Groundwater contamination: risk management and assessment for landfills in Mar del Plata, Argentina

E. Bocanegra · H. Massone · D. Martínez · E. Civit · M. Farenga

Abstract This study presents a program for risk management in the contamination of groundwater resulting from leachate in landfills at Mar del Plata (Argentina). The program includes prediction, prevention, monitory observation and mitigation as actions that must be taken prior to, during, and after contamination occurs. This routine aids in the identification of weak points and failures of the program for those who are in charge of making decisions, and will allow better use of limited financial and technical resources based on planning, thus minimizing disconnected determinations. Results of the prediction stage were confirmed through monitoring. The presence of a plume of contamination more than 100 m from the landfill was shown, thus proving that implementation of a monitoring plan to follow the development of the contaminated plume must be implemented, particularly in populated areas which depend on residential wells for their water supply.

Keywords Risk management · Landfill · Groundwater pollution · Argentina

Introduction

Mar del Plata, the capital of the General Pueyrredón District, is situated on the southeast coast of the Province of Buenos Aires and constitutes the largest seaside resort in Argentina (Fig. 1). The city's permanent population is approximately 600,000 inhabitants but reaches about 2,000,000 during the summer months.

Until the early 1950s, Mar del Plata was a minor summer resort with fishing as its main economic activity. Subsequently, other economic activities such as construction, textiles and tourism were developed. This brought about a very important urban expansion and a considerable influx of people. At present, Mar del Plata is the sixth largest city in Argentina (excluding the Federal District and its suburban centers). The city produces about 700 tons of solid urban wastes per day in the wintertime, and in the summer months the amount of waste goes up to 1200 tons/day; none of which are classified. Mar del Plata takes its drinking water from underground resources, and the municipal network serves 75% of the population.

The final municipal waste disposal is, like in all large cities all over the world, a great problem that very often requires an important portion of public funds. Furthermore, dumping sites for municipal wastes usually produce a leachate that migrates to adjacent areas, resulting in gross pollution of soil, surface water and groundwater. The leachate may contain matter that is resistant to biological or chemical changes, and that thus remains in the soil for many years. The composition of this leachate derives from rainfall, groundwater, initial humidity, and reactions associated with decaying waste. A contamination plume has formed as a result of leaching into the groundwater system, and it has been described in various case studies (Baedecker and Back 1979; Kimmel and Braids 1980; Nicholson and others 1983).

Since the late 1960s, solid waste has been disposed of in several open dumps in the southern sector of the district. Foreseeing the threat of environmental hazards, sanitary landfill techniques have been used since 1993. The company in charge of final waste disposal was required first to build a retaining wall for liquids made of an isolating membrane, and then to collect and treat the leachate within a confined area. However, political and social factors (access to the site by the needy for scavenging, and the fires that this population group often set) have made the site function as an open area.

This project is directed to present a risk management program for leachate contamination of groundwater at two recent sites utilized for the disposal of solid urban wastes in Mar del Plata. The first site, referred to as BV, was in operation from 1979 to 1993, and covered an area of 0.43 km^2. The second site, called BI, has functioned since mid-1993, and occupies an area of 0.20 km^2. Both sites are on a watershed of surface water and recharge water formed by elevated blocks of orthoquarzites (Fig. 1).

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E. Bocanegra (✉) · H. Massone · D. Martínez · E. Civit
M. Farenga
Centro de Geologia de Costas y Cuaternario,
Universidad Nacional de Mar del Plata,
C.C. 722 Correo Central,
7600 Mar del Plata, Argentina
e-mail: ebocaneg@mdp.edu.ar

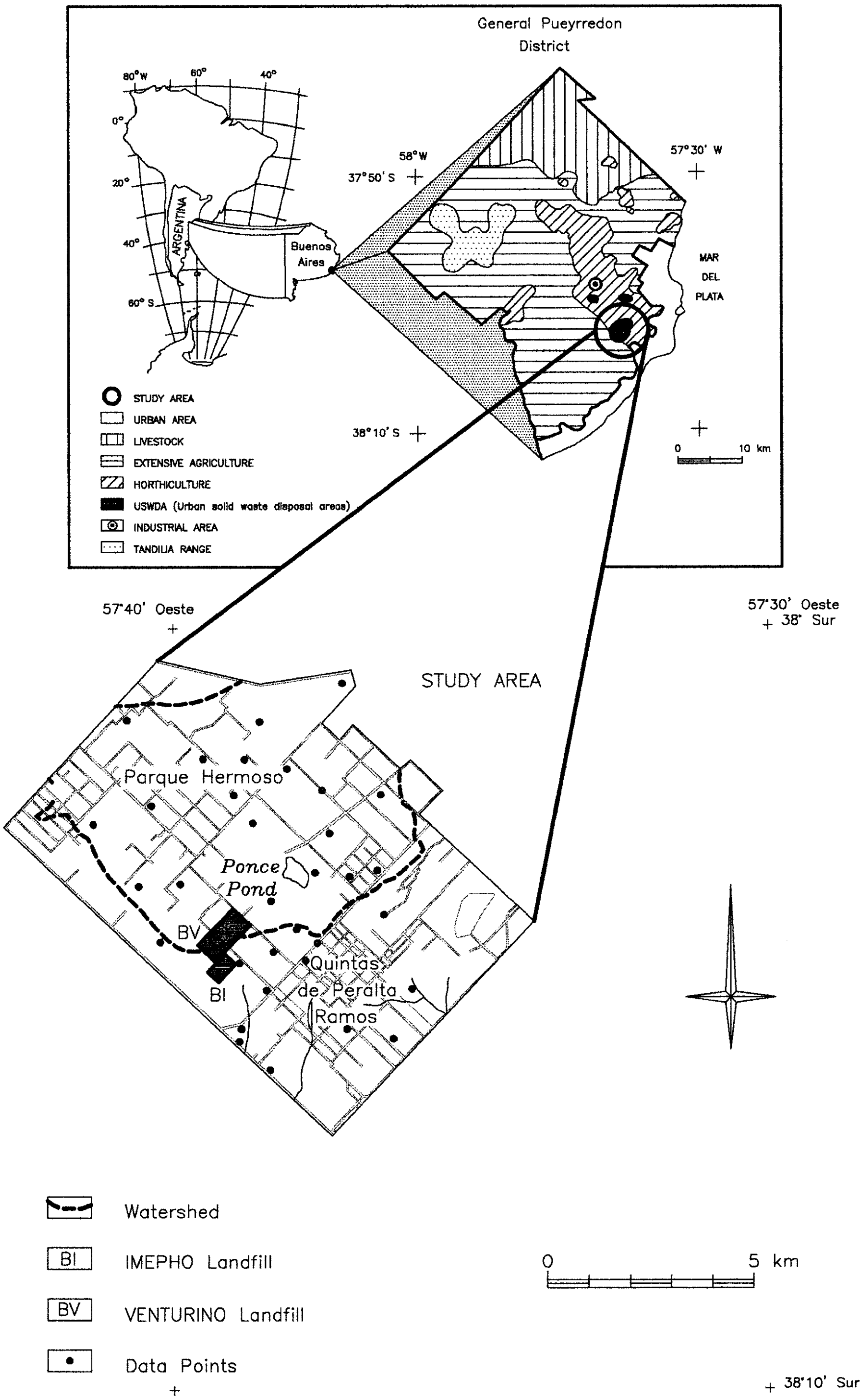

Fig. 1
Map of land use in General Pueyrredon District, and location of the study area

Hydrogeological characterization of the area

The disposal sites under study are at approximately 57°40′W and 38°05′S in a watershed area of the Las Chacras Creek, district of General Pueyrredón (Fig. 1). This district is at the southeast spurs of the Sierras de Tandilia (Teruggi and Kilmurray 1975), a mountain range in blocks lined up according to three Tertiary faulting systems with directions NW–SE, NE–SW and E–W, producing horst and graben structures. Those blocks are formed by Precambrian crystalline basement rocks and Eopaleozoic orthoquartzites. The rocks outcropping in the General Pueyrredón District are Eopaleozoic orthoquartzites from the Balcarce Formation (Dalla Salda and Iñiguez

1989). These rocks present a subhorizontal stratification that generates the morphology of the tablelands. The orthoquartzites are near the surface (4 to 10 m) at the sanitary landfill area, and cause a hilly landscape. Upper Tertiary and Quaternary sediments of silt–sandy and silt–clay textures, the "sedimentos Pampianos", overlie the orthoquartzites.

The orthoquartzites comprise the impermeable hydrogeological basement of the region (Sala 1975). Though generally considered impermeable, the joints of these orthoquarzites produce a secondary porosity which has not yet been studied in detail but which is negligible as an aquifer with regard to the overlying clastic aquifer. The most common sequence presents orthoquartzites topped by loessoides Pleistocene-Holocene sediments and very thin sandy sediments.

Hydrologically, this area represents the most important sequence, as it is the principal aquifer in the region (cf. clastic permeable complex in nomenclature of Sala 1975). The clastic thickness of the analyzed basin was determined by means of drilling and geoelectrical prospecting (Martínez and others 1997). It ranges between 10 and 90 m (Fig. 2a). The orthoquarzitic rock is a few meters below the surface in the southern, western and northwestern borders. At some of these points, the basement outcrops and the quartzite is exploited as commercial rock. These borders are considered as impermeable boundaries in this study. The basement reaches a depth of more than 100 m in the distal zone of the basin.

The aquifer of Mar del Plata is a Pleistocene sedimentary sequence (silt and silty–sand). The permeability varies from 10–15 m/day. Pumping tests show values for transmissivity of 500 m^2/day in the southwestern zone of the district, between 500 and 700 m^2/day in the urban zone, and between 1,000 and 1,200 m^2/day in the northern and northwestern rural zones. The storage coefficient is in the order of 0.001 (Bocanegra and others 1993).

The recharge takes place in the upper part of the aquifer in practically the entire exposed area. The recharge reaches the highest levels of the aquifer and from there drains into deeper levels. Rainfall infiltration produces the recharge to the systems. The annual average rainfall for the period 1901–1987 was 851.6 mm. Although the Penman-Monteith method has been considered the most adequate for plains (Usunoff 1994), the lack of data on meteorological parameters and land use required by this method forced us to calculate evapotranspiration by means of the Thornthwaite method (Custodio and Llamas 1976). The losses produced by evapotranspiration are 719.2 mm/year, leaving a surplus of 132.4 mm/year between July and October.

On the basis of the measured hydraulic heads, a piezometric contour map was prepared (Fig. 2b). The underground watershed coincides with the surface watershed. The underground flow direction is the same as the surface water flow direction on the northern slope, i.e., SSW–NNE, whereas on the southern slope it is NW–SE. The average hydraulic gradient in the foothills is 0.015, whereas on the plains it is 0.008. On the top of the hills

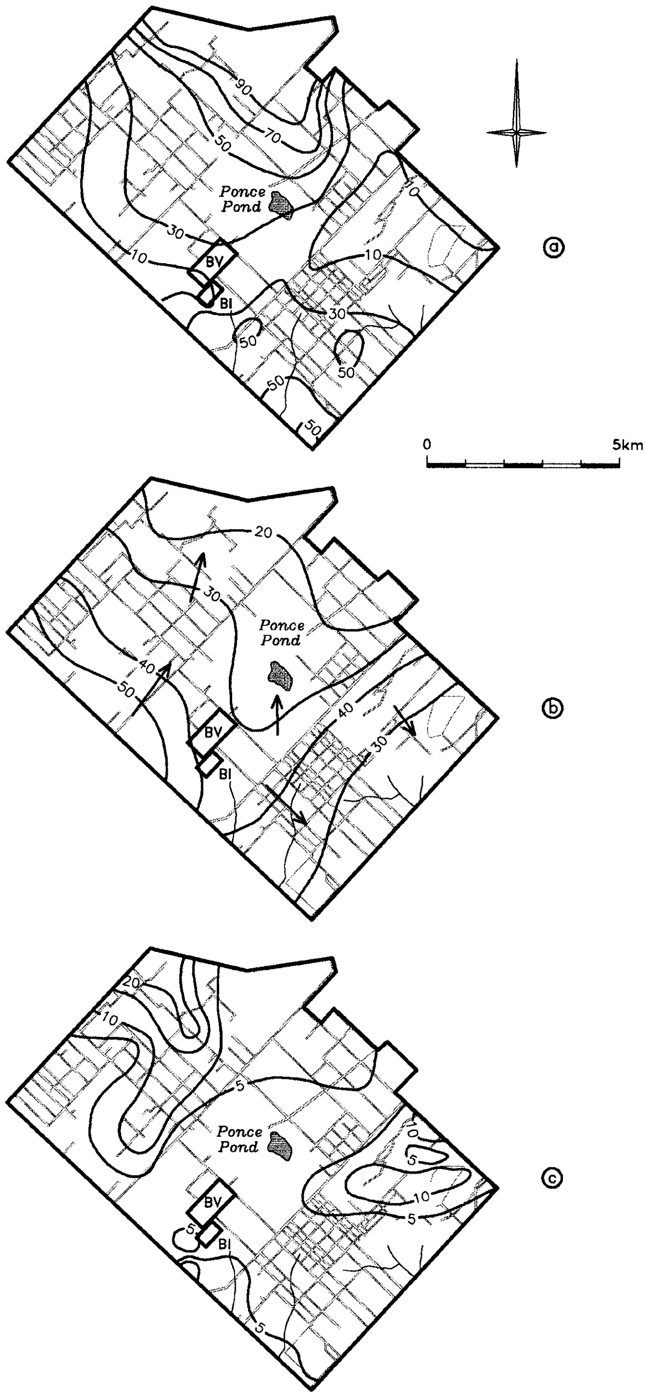

Fig. 2
Hydrogeological features of the study area. **a** Hydrological basement depth (m). **b** Piezometric contour map. **c** Thickness of unsaturated zone (m)

where the landfills are placed, the thickness of the unsaturated zone is about 5 m, increasing to 10 m on the slope and to 15 m on the plains (Fig. 2c).

The groundwater quality in the basin is that of a sodium bicarbonate type, either mixed with calcium compounds or with no dominant cation near the watershed. The anionic composition is fairly constant but the cationic one varies from a calcium type at the recharge zone to a sodium type at the zone of discharge. The salinity ranges from 500–1,500 mg/l of TDS.

Socioeconomic characterization of the area

The main economic activity in the area is horticulture, with small to medium-sized farms in the south and southeast, and large commercial farms in the north and northeastern zone. Urbanized areas surround both zones, i.e., Parque Hermoso, a neighborhood to the north with a population density of 1–5 inhabitants/ha, and Quintas de Peralta Ramos to the southeast which has a density of less than 1 inhabitant/ha. It is relevant to note that, although the population density of both residential areas is rather low, it has increased more that 10% between 1980 and 1990. The families living there are exposed to several risks due to the lack of running water, and have quite limited economic resources.

Risk management and aquifer contamination

Any risk can be understood as a product of the probability of the occurrence of some undesirable or even disastrous event and the damage this event could cause (Rowe 1977). Risk and hazard are frequently used synonymously in the literature. However, hazard is related only to the probability of an undesirable event taking place, without considering its negative effects. Callow (1998) points out that, among specialists, risk assessment is most often discussed as a predominantly scientific process concerned with adverse outcomes that are measurable, whereas risk management is viewed as a primarily legal, political, and administrative process.

The risk evaluation process is embedded in a larger one, that of risk management. This process begins when a potential risk is determined. In this case study, the amount of pollution and the vulnerability of the aquifer determine the potential risk. Both factors must be taken into account when designing a risk management program.

The process of risk assessment for this project is considered to be contained within a larger one, that of risk management, which begins by resolving a potential risk determined by the interaction between the contaminant load and the aquifer vulnerability. Both parameters must be taken into account when designing a program for pollution risk management.

It is well understood that, for situations caused by natural disasters, risk management involves actions taken previously, during, and after the event (Van Molle 1993). These actions may be classified into four, often overlapping stages: prediction, prevention, alert system, and mitigation. Among the many geological risks, the pollution of the aquifer is a clear example of an induced risk. However, by managing the risk of aquifer contamination stemming from final waste, it is possible to propose a similar pattern. The following principles must be taken into consideration:

1. It is a process that takes place slowly over a long period of time.
2. People do not easily understand this problem, especially in a humid zone where the water is considered plentiful and unlimited.
3. It is avoidable.
4. It is predictable and economically adequate.
5. Mitigation is a process that costs much more than prediction.

The two stages preceding the contamination process are the spatial/temporal predictions and the preventive measures (structural and nonstructural), both of which must be carried out to avoid this phenomenon. The third stage, performed after a contamination event occurs, involves an alert system which is based on periodic monitoring. Monitoring is the main tool for following the evolution of the contamination process. When that phenomenon produces damage, the fourth stage, mitigation, is the one of greatest concern to those involved in water supply administration.

Stage 1: prediction

Prediction represents the main element for risk assessment in the administration of sanitary landfills. Petts (1998) proposes four important phases:

1. Calculation of leachate generation and estimation of its quality.
2. Estimation of the amount and rate of leakage.
3. Estimation of the concentration of specific pollutants in the groundwater.
4. Comparison of estimated concentrations with those of acceptable standards.

Calculation of leachate generation and estimation of quality

The amount of leachate depends on different elements such as weather conditions, waste humidity, soil permeability, etc. The volume of the leachate that a sanitary landfill generates can be determined with the following equation:

$$L = I + DH + Ei - Eo$$

where L denotes the leachate, I the infiltration of the remaining rainfall minus evapotranspiration, DH the change in the humidity of the residue, Ei the superficial runoff entering the landfill, and Eo the superficial runoff that exits the landfill.

The HELP code – a model for the hydrological evaluation of sanitary landfill behavior (Schroeder and others 1994) – has been applied in this study to calculate the hydrological balance. The program uses climate, soil, and design data in order to generate the daily estimation of wa-

Table 1
Design for landfill BV (see Fig. 2)

Layer	Type	Thickness (m)	Porosity (v/v)	Wilting point (v/v)	Saturated hydraulic conductivity (cm/s)
1st	Soil	0.15	0.50	0.15	0.19E-03
2nd	Municipal waste	4.00	0.67	0.30	0.10E-02
3rd	Soil	0.15	0.50	0.15	0.19E-03
4th	Municipal waste	4.00	0.67	0.30	0.10E-02
5th	Soil	0.15	0.50	0.15	0.19E-03
6th	Municipal waste	4.00	0.67	0.30	0.10E-02
7th	Soil	0.15	0.50	0.15	0.19E-03
8th	Municipal waste	4.00	0.67	0.30	0.10E-02
9th	Soil	0.15	0.50	0.15	0.19E-03
10th	Municipal waste	4.00	0.67	0.30	0.10E-02
11th	Clay	0.30	0.45	0.45	0.12E-05

ter movement into and out of the landfill. The climate data required by the model are the daily rainfall, temperature, solar radiation, annual average wind speed, and relative average humidity. The maximum depth for the evapotranspiration of silty soil varies between 0.20 and 0.40 m. In this study, a depth of 0.25 m was used. A sparse green cover was also considered. The surface slope was estimated at 2%.

Sanitary landfill BV (1979–1994)

The sanitary landfill design presented in Table 1 includes a sequence of five cells of trash 4 m high. The cells are separated by layers of soil 15 cm thick that lie on a layer of clay. The characteristics of the soil in the different layers correspond to the USDA classification schemes that have been incorporated into the HELP code. The design is a simplification of the natural model, and does not take the effect of daily waste removal into consideration.

The simulation results are shown in Table 2. The loss by evapotranspiration is about 70.1% of the rainfall, and the runoff is about 12.5%. The leachate from the landfill constitutes 16.1% of the rainfall, and it may seep through the clay layer.

Sanitary landfill BI (1994–1999)

The sanitary landfill design presented in Table 3 includes a sequence of six cells of trash 3 m high. The cells are separated by layers of soil 0.30 m thick, and lie on a synthetic geomembrane. Nonpredictable deficiencies due to holes in the membrane and installation defects were taken into consideration.

Simulation results are shown in Table 4. The loss by evapotranspiration is about 68.7% of the rainfall, and the runoff is about 15.4%. The leachate from the landfill is 7.1% of the rainfall, and it could be released through the geomembrane. By improving the performance of the geomembrane, a certain amount of leachate could be contained, removed, and submitted to some type of wastewater treatment.

Table 2
Annual hydraulic balance for landfill BV (see Fig. 2)

	(mm)	(m^3)	(%)
Rainfall	1,050.8	451,844	100.0
Surface drainage	131.5	56,541	12.5
Evapotranspiration	736.4	316,651	70.1
Leachate through 11th layer	169.7	72,985	16.1
Average head on top of 11th layer	5.1		
Change in water storage	13.2	5,666	1.3

Quality of leachate

In July and August of 1998, samples were taken from the leachate to determine in situ temperature, specific conductivity, and pH. The monitoring of liquid chemical and microbiological contamination involved various laboratory tests. Measurements of the concentrations of some heavy metals were performed by means of atomic absorption spectrometry. Samples were collected under USGS standards (Brown and others 1970).

The normal pH value for leachates is about 7.8. These samples were found to be slightly alkaline. Total solids concentrations were 8,220 mg/l for BI, and 3,318 mg/l for BV. Ammonia and nitrites were found in considerable amounts, pointing at the leachate as a reduction medium. Bacteriological tests show very high values for aerobic bacteria, total coliforms, fecal coliforms, *Streptococcus*, and traces of *Pseudomona aeruginosa*. A characterization of leachates taken from active cells at the BV and BI landfills is shown in Table 5.

Estimation of amount and rate of leakage

Based on the data shown above, the annual volume has been estimated as 130,000 m^3 for BV, and 15,000 m^3 for BI. After applying the HELP code, computations show a higher production of leachate from October to Novem-

Table 3
Design for landfill BI (see Fig. 2)

Layer	Type	Thickness (m)	Porosity (v/v)	Wilting point (v/v)	Saturated hydraulic conductivity (cm/s)
1st	Soil	0.30	0.50	0.15	0.19E-03
2nd	Municipal waste	3.00	0.67	0.30	0.10E-02
3rd	Soil	0.30	0.50	0.15	0.19E-03
4th	Municipal waste	3.00	0.67	0.30	0.10E-02
5th	Soil	0.30	0.50	0.15	0.19E-03
6th	Municipal waste	3.00	0.67	0.30	0.10E-02
7th	Soil	0.30	0.50	0.15	0.19E-03
8th	Municipal waste	3.00	0.67	0.30	0.10E-02
9th	Soil	0.30	0.50	0.15	0.19E-03
10th	Municipal waste	3.00	0.67	0.30	0.10E-02
11th	Soil	0.30	0.50	0.15	0.19E-03
12th	Municipal waste	3.00	0.67	0.30	0.10E-02
13th	Membrane	0.0025	0.50	0.15	0.19E-10

Table 4
Annual hydraulic balance for landfill BI (see Fig. 2)

	(mm)	(m^3)	(%)
Rainfall	1,050.8	210,160	100.0
Surface drainage	162.2	32,449	15.4
Evapotranspiration	722.2	144,448	68.7
Leachate through 13th layer	73.9	14,775	7.1
Change in water storage	92.4	18,487	8.8

Table 5
Physicochemical characterization of leachates from landfills BV and BI at two different dates

	BV-07/07/97	BI-07/07/97	BV-08/27/98	BI-08/27/98
Conductivity (μs/cm)	8740	2860	7250	20 000
DBO (mg/l)	390	430	465	1000
Dissolved oxygen (mg/l)	–	1	0.1	0.1
DQO (mg/l)	662	–	–	
Fe (mg/l)	0.54	1.1	5.2	
Zn (mg/l)	0.13	0.07	0.2	0.2
Mn (mg/l)	–	2.1	2.8	
Ni (μg/l)	40	76	108	
Pb (μg/l)	5	5	5	5
Cr (μg/l)	–	–	27	
Cd (μg/l)	1	–	–	
Hg (μg/l)	1	1	–	–

ber, and a lower one from April to May. During most of the year, the quantity of leachate is more or less uniform. The higher daily values are about 122.2 m^3.

With regard to the total leachate in BV, approximately 30% is retained within the landfill, while 70% seeps through and infiltrates a low area located 100 m downcurrent. In the case of BI, there is a small treatment plant that partially treats the leachate and then pours it back into the landfill. The rest of the leachate leaks through the membrane and may affect the groundwater.

Estimation of groundwater specific pollutant concentrations

Flux and transport of the contaminated plume

The three-dimensional modeling of the flux and transport in BV was carried out by the Visual MODFLOW Code (Guiger and Franz 1996). This code solves differential equations by means of finite elements. Visual MODFLOW includes the MODFLOW code for flux simulation, MODPATH for particles tracking, and MT3D for solute transport.

The properties of the domain were distinguished by means of a grid of 2,500 cells (Fig. 3). The aquifer geometry was defined by importing the topographic surfaces and the upper surface of the quartzite formation. The two layers developed were the upper one, corresponding to the clastic aquifer, and the lower one, the hydrogeological basement. The lowest layer presented aquifuge characteristics, and was considered as inactive.

Due to the lack of information concerning the coefficients of retardation for different contaminants, the modeling of solute transporting was carried out by employing the data for the chloride ion. This ion has a conservative character and comes from a simple source. Although the nitrate ion is also conservative, as it comes from multiple sources, it could not be used as a reference. The following parameters were defined:

1. Border conditions: these are based on geological and hydrogeological data, the impermeable border having been fixed in the hills of orthoquartzite towards the south and west of the watershed. The northern border

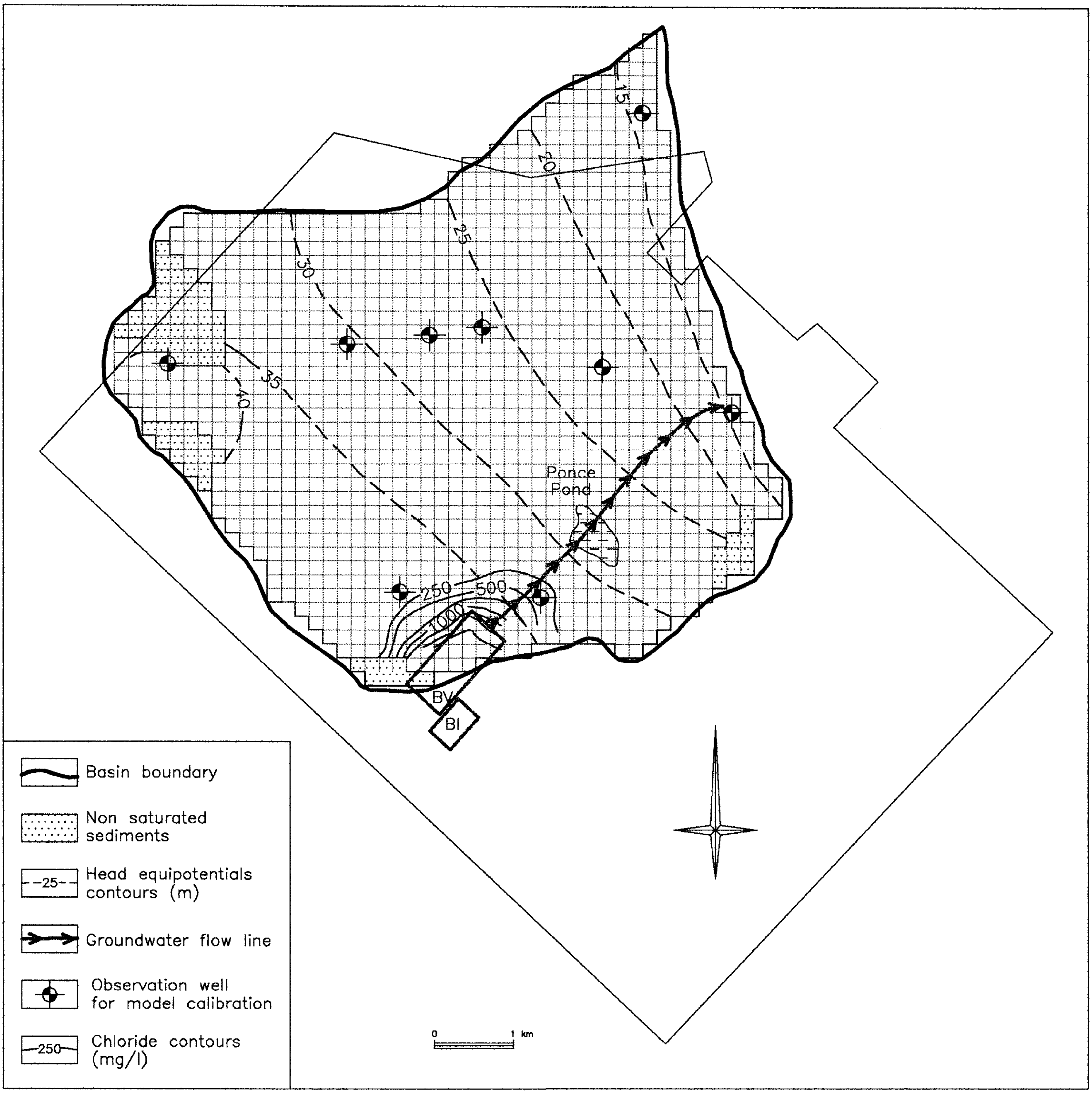

Fig. 3
Results of groundwater flow and solute transport modeling using Visual MODFLOW, showing piezometric heads and chloride concentrations

had the condition of null flow at the watershed. The piezometric line of 15 m constituted the eastern border, establishing a condition of fixed height. The border condition for chloride concentration was 70 mg/l, which is the initial concentration for the entire area.

2. Hydraulic parameters: the parameters considered for modeling correspond to the values of hydraulic conductivity, porosity and specific storage coefficient. These values are 10 m/day, 0.3, and 0.001, respectively (Bocanegra and others 1993). The dispersivity is 200 m (Bocanegra and others 1994).
3. Recharge: a recharge of 130 mm/year was applied for the entire study area. In the landfill, the recharge value was 300 mm. This is the result of adding the leachate through layer 11 plus the surface drainage from the plain adjacent to the landfill. The recharge concentration in this region is 2,000 mm of chloride per liter.

The stationary flow model with a theoretical uniform recharge was set as an initial condition in transitory simulation of flow and chloride transportation. The results, after applying the landfill recharge for 10 years, are

shown in Fig. 3. The dark gray zones show the nonsaturated thickness sector.
The pattern of the flow lines maintains similitude with the observed values. The most important differences are found towards the northeast because of the complexity of the system, and the lack of data concerning the depth of the hydrogeological basement. The adjustment in the major part of the area is acceptable. Figure 4 shows the ratio between calculated and observed levels in eight drilling holes within the area. The average error ($\Sigma[h_{calc}-h_{obs}]$/number of observations) is null. The average absolute error ($\Sigma[h_{calc}-h_{obs}]$/number of observations) is 3 m, and the average quadratic error ($\{\Sigma[h_{calc}-h_{obs}]^2\}^{1/2}$/number of observations) is 4 m.
The values of chloride concentration found in the contaminated plume show that it varied from 900–250 mg/l in the direction of flow. The plume width was around 600 m to the northeast.
In the case of BI, the time of transit through the unsaturated zone was calculated by (Foster and Hirata 1991)

$$t = H \cdot \theta / R$$

where H is the unsaturated depth (2 m), θ is the moisture content (10%), and R is the infiltrated volume (73.9 mm/year).
The time required for the leachate to reach the aquifer is 2.7 years. When applying Darcy's law to the saturated zone, using a hydraulic gradient of 0.0125 and a permeability of 10 m/day, the calculated velocity is 0.125 m/day. This speed indicates that, if there were any contamination, it would not spread out further than 100 m from the point source, Therefore, with our present data, it is impossible to calibrate a model to verify predictions.

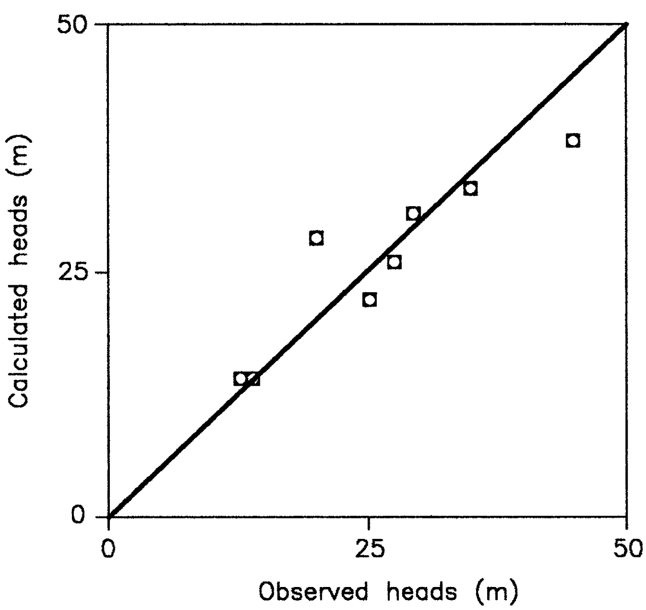

Fig. 4
Correspondence between calculated and observed piezometric heads

Comparison of estimated concentrations with acceptable standards

The concentration of chlorine ion was estimated between 250 and 900 mg/l. The World Health Organization (WHO 1963) stipulates 600 mg/l as the allowable maximum. In the present study, the concentration exceeds this admissible level in only a small portion of the samples. The plume verifies values that reach up to 2.5 times those of the regional background. This represents an alert to the possibility of alteration due to other contaminants.
The rates between area values and standards for other contaminants were evaluated after monitoring-stage data were established.

Stage 2: prevention

Since 1960, Mar del Plata has seen such an increase in population that it has become one of the ten largest cities in Argentina. Final waste disposal was accomplished through rudimentary sanitary landfill techniques. Moreover, the disposal locations belonged to private firms. Until 1979, the only problem was to find a place to dump the waste. In later years, new regulations concerning the presence of superficial drainage, the depth of the quartzitic basement, and the phreatic level appeared. Nevertheless, until the early 1990s, the city still did not have a complete territorial regulation that included waste disposal. In 1993, new and more precise regulations were approved. Prevention incorporates structural and nonstructural measures.

Structural measures

One kind of structural measure is related to the adequate operation of the landfill. Neither the insulation of the bottom (by means of selected soil or use of linings) nor leachate treatment were required until 1993. In the beginning, the BI site complied with these requirements. Later on, however, the poverty-stricken sector of the population began to go to the dump for scavenging, and fires were started. This brought about technical problems that reduced effectiveness to a minimum. At present, there are rigorous technical regulations intended to prevent contamination.

Nonstructural measures

Two important measures of this kind are land-use planning and education. In 1993, the City Hall signed an agreement with the National University of Mar del Plata. The University carried out research in order to select better places for final waste disposal. The Santander's Geoscientific Cartographic methodology was used for this study (Massone and others 1993). As a result, an area located to the south of the city was selected. The site was neither a residential zone nor a place visited by tourists. In addition, it had a phreatic level with a depth of more than 5 m, and a quartzitic basement more than 10 m deep.
Education is essential to become aware of the importance of these regulations designed to prevent contamination. For example, the construction of residential wells with

correct casing, including a cement ring in the first 20–25 m, is the most important requirement.

Stage 3: alert system

Considering that the main purpose of this study is to present a groundwater contamination risk management program, the alert system is principally based on the development of a monitoring program. This program must be carried out during and after site operation. Until now, municipal authorities have not monitored on a regular basis. Nevertheless, the University was asked to conduct monitoring campaigns. The first one took place in 1993, and about 20 residential wells were monitored. In 1997, five more monitoring wells were drilled (Martínez and others 1997). The second monitoring campaign took place that same year, and was based on the flow model pattern. The parameters selected for the measurement were the principal ions, nitrogen dissolved species, and heavy metals.

The hydrochemical characterization cannot be considered to reflect the natural geochemical composition because several groundwater contamination sources had already existed in the zone. Nevertheless, the present information constitutes a very important reference for monitoring the spreading of the present plume. Statistical parameters corresponding to the 1997 monitoring are shown in Table 6.

In most of the sampling sites, nitrate concentration exceeded the admissible 45 mg/l maximum established by WHO (1963). The contamination cannot be attributed exclusively to the urban waste disposal because, if the speed of flow and the age of waste disposal are considered, the flow of a contaminant plume – especially for a conservative contaminant such as nitrate – could not have spread more than 200 or 300 m from that site. Furthermore, there are some other sources of nitrates in the area, such as the localized pollution produced by the residential wells and the diffused pollution resulting from the use of fertilizers in agriculture.

Heavy metals show a long-term lag because the sediment absorbs them. The tests were carried out only in those wells near the dumps, and they were performed to locate only Fe, Mn, Pb, Zn, and Ni. These were considered to be the most dangerous metals because they are present in the leachates.

The potability limits established by the Argentine Food Administration are 5.0 mg/l for Zn, 0.05 mg/l for Pb, Ni, Mn and Cd, and 0.1 mg/l for Fe. At seven of the sites sampled, the levels of iron exceeded these admissible values and, at two others, the levels of manganese were also excessive. The concentrations of zinc were considerably below the limit, and the lead concentration was at the limit at one site. The concentrations of iron and manganese are not good contamination indicators since these metals can appear for natural reasons. The concentrations of zinc, however, did not exceed the admissible limits but they do indicate that the groundwater has been polluted by the leachate. Lead and nickel have not been found as of yet, probably because the soil has absorbed them.

Table 6
Statistical parameters for the chemical composition of groundwater from the sampling wells

	No. of samples	Minimum	Maximum	Mean	Standard deviation
Conductivity	23	523.0	1,644.0	972.74	276.52
pH	23	6.8	7.9	7.48	0.22
Residue	23	466.0	1,242.0	732.0	203.0
Hardness	23	98.0	534.0	274.35	115.61
HCO_3^-	23	292.6	683.9	441.58	114.65
Cl^-	23	39.0	303.0	86,078	54.06
$SO_4^=$	23	11.0	41.0	22.0	6.93
NO_3^-	22	5.0	195.0	77.86	46.44
NO_2^-	15	0.01	0.5	0.083	0.131
F^-	22	0.3	1.7	0.659	0.282
Ca^{2+}	23	21.0	134.0	64.04	30.99
Mg^{2+}	23	1.0	50.0	27.65	12.60
Na^+	23	42.0	244.0	112.39	60.07
K^+	23	5.0	20.0	11.74	6.40

Stage 4: mitigation

Some natural mitigation exists due to the absorption by organic-rich soil in the area, producing a delay in contaminant flow. Nevertheless, it is necessary to think about other mitigating procedures, particularly focusing on the accurate construction of residential wells, since a supply of running water is not economically feasible.

Conclusions

The leachate produced in landfills contaminates groundwater, and actions to resolve the problem have, till now, not been integrated, thus resulting in the misuse of time and money. The implementation of a program for risk management is a new approach to solve this problem. The program proposed emphasizes the process as a whole, but it is organized into four stages. These stages may overlap at times, but they are always well differentiated. Moreover, for the people in charge of making decisions, it may become easier to identify the weak points or failures in the process. At the same time, limited technical and economic resources can be optimized. Finally, decisions can be planned in advance.

In the present case study, the weak points identified for the city of Mar del Plata are the following:

1. Periodical data series are still lacking and, therefore, it is difficult to create a model or obtain precise information regarding initial conditions for any process.
2. Political and social factors hinder the efficacy of the structural prevention steps.
3. Political and social factors hamper the full usage of the results obtained in the prediction stage which, in turn, affects the subsequent stages.

4. Access to information regarding the dynamics of hydrogeological resources is insufficient for those in charge of decision-making and for consumers as well.

The results obtained for the prediction stage were confirmed by monitoring. They showed the presence of a contaminant plume reaching farther than 100 m from the final waste disposal site. The fact that it is an urban zone that utilizes wells for water supply makes it necessary to maintain a monitoring program to closely check the spreading of the contaminated plume.

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References

Baedecker MJ, Back W (1979) Hydrogeological processes and chemical reactions at a landfill. Groundwater 17:417–429

Bocanegra EM, Benavente MA (1994) Simulación numérica del sistema de flujo de las aguas subterráneas en Mar del Plata, Argentina (Período 1920–1969). II Congr Latinoamericano de Hidrología Subterránea, Santiago, Chile, 9–11 Noviembre 1994, Proceedings vol 1, pp 213–223

Bocanegra EM, Martinez DE, Massone H, Cionchi JL (1993) Exploitation effect and salt water intrusion in the Mar del Plata aquifer, Argentina. Study and modeling of salt water intrusion into aquifers. CIMNE-UPC, Barcelona, pp 177–191

Brown E, Skougstad MW, Fishman M (1970) Methods for collection and analysis of water samples for dissolved minerals and gases. USGSTechniques for Water Resources Investigations, Book 5 (A-1), Reston, Virginia

Callow P (1998) Environmental risk assessment and management: the whats, whys and hows? In: Callow P (ed) Handbook of environmental risk assessment and management. Blackwell Science, Oxford, pp 1–8

Custodio EY, Llamas MR (1976) Hidrología subterránea. Omega, Barcelona

Dalla Salda J, Iñiguez M (1989) La Tinta precámbrico y paleozoico de Buenos Aires. VII Congr Geol Argentino, Neuquén, 9–13 Abril 1978, Actas 1, pp 539–550

Foster S, Hirata R (1991) Determinación del riesgo de contaminación de aguas subterráneas. Publ programa regional de prevención y control de la contaminación de aguas subterráneas. CEPIS-WHO, Lima

Guiger N, Franz T (1996) VISUAL MODFLOW. The integrated modeling environment for MODFLOW and MODPATH. Version 2.00. User's manual. Waterloo Hydrogeologic Inc, Ontario, Canada

Kimmel GE, Braids OC (1980) Leachate plumes in groundwater from Babylon and Islip landfills. US Geol Surv Prof Pap 1085, Long Island, New York

Martínez DE, Massone HE, Bocanegra E (1997) Relevamiento geológico en el nuevo sitio de disposición final de residuos y preparación de una red de monitoreo. Conv Univ Nacion Mar del Plata – Municipal General Pueyrredón, Techinical Rep

Massone H, Del Rio JL, Fajardo D, Cionchi J, Martinez D, Bocanegra E (1993) Los residuos sólidos domiciliarios del Partido de General Pueyrredón desde una perspectiva geológico-ambiental. Parte 1: aplicación de la cartografía geocientífica a la selección de sitios de disposición final. XII Congr Geol Argentino, Mendoza, 10–15 Octubre 1993, Actas VI, pp 303–310

Nicholson RV, Cherry JA, Reardon EJ (1983) Migration of contaminants in groundwater at a landfill: a case study. VI. Hydrogeochemistry. J Hydrogeol 63:131–176

Petts J (1998) Risk assessment for waste treatment and disposal. In: Callow P (ed) Handbook of environmental risk assessment and management. Blackwell Science, Oxford, pp 417–452

Rowe W (1977) An anatomy of risk. Wiley, New York

Sala JM (1975) Recursos hídricos (especial mención a las aguas subterráneas). VI Congr Geol Argentino Relatorio, Bahía Blanca, 21–27 Septiembre 1975, pp 169–194

Schroeder PR, Lloyd CM, Zappi PA, Aziz NM (1994) The hydrologic evaluation of landfill performance (HELP) model. Version 3.1. USEPA, Cincinnati, Ohio

Teruggi M, Kilmurray J (1975) Tandilia relatorio geología de la Provincia de Buenos Aires. VI Congr Geol Argentino, Bahía Blanca, 21–27 Septiembre 1975, pp 55–77

Usunoff E (1994) Técnicas de estudio en llanuras. In: Bocanegra EM, Rapaccini (eds) Temas actuales de la hidrología subterránea. Univ Nac Mar del Plata, Mar del Plata, Argentina, pp 103–118

Van Molle M (1993) Natural hazards. In: Nath B, Compton P, Dvuyst D (eds) Environmental management. VUBPRESS, Bruselas, vol 1, pp 305–340

WHO (1963) International Standards for drinking-water, 2nd edn. World Health Organization, Geneva