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Slurry wall containment performance: monitoring and modeling of unsaturated and saturated flow

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Abstract

A specific 2-year program to monitor and test both the vadose zone and the saturated zone, coupled with a numerical analysis, was performed to evaluate the overall performance of slurry wall systems for containment of contaminated areas. Despite local physical confinement (slurry walls keyed into an average 2-m-thick aquitard), for at least two decades, high concentrations of chlorinated solvents (up to 110 mg l−1) have been observed in aquifers that supply drinking water close to the city of Milan (Italy). Results of monitoring and in situ tests have been used to perform an unsaturated-saturated numerical model. These results yielded the necessary quantitative information to be used both for the determination of the hydraulic properties of the different media in the area and for the calibration and validation of the numerical model. Backfill material in the shallower part of the investigated aquifer dramatically affects the natural recharge of the encapsulated area. A transient simulation from wet to drought periods highlights a change in the ratio between leakages from lateral barriers that support a specific scenario of water loss through the containment system. The combination of monitoring and modelling allows a reliable estimate of the overall performance of the physical confinement to be made without using any invasive techniques on slurry wall.

Introduction

Vertical barriers have been used since the late 1970s for containment and environmental pollution control at hazardous sites (Paul et al. 1992; Ryan 1987, 1994; Manassero et al. 1995; Filz and Mitchell 1995) and have been extensively studied in terms of design, performance, permanence, and other relevant issues (Paul et al. 1992; Nash 1974;D'Appolonia 1980; Hajnal et al. 1984; Evans 1993; Xanthakos 1994; Beretta et al. 2003). In Italy, the technique has been widely applied for groundwater remediation strategies in the major contaminated sites (Rolle et al. 2009).

Given the extremely high cost to clean up the sites completely in a short period of time, vertical barriers are useful to confine the source for subsequent removal while protecting downgradient groundwater, or simply to contain exposure to the source. Under certain flow conditions, and when combined with pumpand- treat remediation systems, physical barriers have a dramatic effect on the contaminant plume management strategy (Bayer et al. 2004). In any case, barriers must be designed to be effective for a long-term period, which is a function of many variables, such as the extent and amount of contamination, the hydrogeological condition, and the presence of potential risk in the surroundings. The lifetimes of these systems, therefore, range from about 50 to 100 years (Inyang 2004).

The most widely used technique for containment has been the soil-bentonite slurry wall. It is typically the most economical, utilises low-permeability backfill, and usually allows reuse of all or most ofthe material excavated during trenching. A study (US EPA 1998) on a number of sites in the USA has showed that maintaining the efficiency of containment is strongly related to design, construction quality assurance, construction quality control, and especially monitoring of the systems. Monitoring the water regime of the containments is particularly necessary to assess their proper operation (Brandelik and Huebner 2003).

In fact, one of the most crucial points is the long-term performance, which depends on a number of factors, such as keying between wall sections, keying into an aquitard (Ryan and Day 2003), formation of flaws, lack of continuity of the aquitard, and the contaminants propertie (Candelaria and Matsumoto 2000); all of these can alter the performance of the system. Some authors tried to extrapolate short-time observations to larger scales (Inyang and Tomassoni 1992), but local conditions strongly affect single performances loss; therefore, each case can only be controlled by a reliable monitoring scheme tailored to the local features of the area.

Many techniques can be used, both “non-invasive” (those that do not alter the wall structure) and “invasive” (those that can alter the wall structure). The use of the latter is usually restricted to avoid problems that enhance contamination spreading. Each technique has advantages and disadvantages; for instance, geophysics can give the general state condition of the wall but is not always reliable for detecting small flaws and discontinuities, whereas piezocone is very sensitive (Manassero 1994) but many tests could be required in an extended area and problems of verticality can arise especially in thin and/or deep walls. A modified slug test approach has also been used to determine the hydraulic conductivity of a vertical cut-off wall (Choi and Daniel 2006), even if it has the main limitations of the standard tests, related to the limited extension of the volume of wall investigated during the test. Pumping tests within the containment area in order to measure drawdown outside the area are expensive and are generally extremely sensitive to the position of the largest defects, so such tests are effective only if piezometers are placed close to them (Benson 2002).

The performance of pumping tests by creating a test cell is very time consuming and expensive and cannot be applied if the containment areas are not wide enough to provide an adequate amount of groundwater in order to perform the test on the correct time scale (Ryan 1987). Moreover, it was showed that there is a clear dependence of the cut-off walls' hydraulic conductivity on the sample volume (Britton et al. 2004), so any technique should also considered the actual amount of volume that is investigated by the test. In any case, whatever the approach, the general aim should always be to obtain a reliable assessment of the bulk transmissivity, that is, to quantify the effectiveness of the system considering its overall performance.

In this study, a specific 2-year program of monitoring and testing both the vadose zone and the saturated zone, coupled with a numerical analysis, was performed to evaluate the overall performance of slurry wall systems for contaminated area containment. The activities were focused on finding out whether deficiencies in the integrity of the vertical barrier systems or the local geological conditions could be more likely to determine the release of contaminants into the aquifers. Due to restrictions of the local Environmental Agency, all the activities have to be done using only noninvasive techniques on the slurry wall systems.

The evaluation of containment effectiveness and the dynamics of contaminant release from it are strategic to perform risk assessment of groundwater contamination, that should always be used as an important support for planning remediation system. Given that risk assessment in these contexts should always be evaluated within a probabilistic framework (Tartakovsky and Winter 2008;Winter and Tartakovsky 2008; de Barros et al. 2009; Bolster et al. 2009), a proper understanding of the impact of source behaviour is essential to reduce uncertainty. So, the aim was to:

(a) assess failures in the containment systems, including deficient cut-off walls or inappropriate design of the geological scheme;

(b) estimate the water budget in an equilibrium state and the discharge/recharge behaviour;

(c) define reliable hydraulic parameters to support successive remediation action; and

(d) obtain a quantitative evaluation of groundwater losses through the confined area, in order to support future analysis of the fate of chlorinated compounds released into the aquifers.

All these aspects are necessary to build a reliable conceptual model and to refine analytical estimate of the probability of failure of a remediation effort for future probabilistic risk assessment (Bolster et al. 2009).

The study area

The industrial growth in northern Italy that occurred over the last 50–70 years, together with the lack of specific laws on waste management, has contributed to the gradual depletion of vulnerable groundwater resources. Chemical facilities had a great role in this as well. The former "Bianchi" chemical facility at the Rho district, located few km northwest to the city of Milan (Fig. 1), is an example of how hazardous wastes produced by industrial activities have been stored for decades without proper knowledge of environmental matters and have caused a potential risk situation that currently is a matter of special concern.

In the late 1970s, the first evidence of intense contamination with chlorinated compounds arose and the supposed main source was immediately confined in a specific area (Fig. 2), later indicated as zone 1, through the construction of vertical slurry cutoff walls keyed into a low hydraulic conductivity layer (aquitard). This containment system was never coupled with other remediation techniques, causing a progressive spread of the contamination during the following years into the present. A second confinement solution, that of a hydraulic barrier was set only much later, in 2006, to stop the migration of a well-defined dense non-aqueous phase liquids (DNAPL) plume in the downgradient areas (zone 2). Again, in 2007, however, concentrations of chlorinated solvents were measured up to 110 mg l−1 near the source at the area of the former "Bianchi" chemical facility, in the southern side of the municipality. Pollution has been detected within a depth of 45 m. The city of Milan is located immediately downstream and, consequently, public and private pumping water wells for human direct consumption and farming are directly exposed to the chlorinated plume threat (Fig. 1).

Hydrogeological setting

The geological structure of the area may be classified by adopting the traditional regional subdivisions for the northern Po Plain (e.g. ENI-AGIP 2002). The vulnerable aquifer is the uppermost, multilayered aquifer (A), which is commonly detected within a depth of 60 m in the whole western part of the Milan province. However, several fine layers with low hydraulic conductivity divide this layer into two semi-confined aquifers (A1 and A2), even if, in some zones, the subunits merge because of a thickness reduction or local lack of such low conductivity horizons. On the top of the A1 aquifer, an unconfined (or sometimes perched) aquifer (A0) occasionally appears. This A0 aquifer has a variable depth (7–9 m from the ground surface), and overlays a semi-permeable, discontinuous aquitard (Fig. 2). At the former chemical domain scale, the discontinuous geometry of the aquitard, and the consequent dependence on DNAPL migration to small-scale anomalies, are already shown in Pedretti et al. (2009) (Fig. 3).

The recharge of aquifers is mainly due to precipitation and irrigation water (with a period going from April to mid-October), while the contribution of the small river crossing the area is negligible (Alberti et al. 2007). Within the containment area, recharge is given by precipitation only, even if the nature of backfilling dramatically affects the recharge rate and dynamics, as described later. The characteristics of the containment system are only partially known. The geometry is commonly referred to as a "sarcophagus-shape" polygon (Fig. 4), keyed into the aquitard (Fig. 2).The composition is composed of typical cut-off slurry backfilled walls.

Focusing on the containment system, some aspects become strategic in the conceptual model describing the general water balance. The aquifer connections and the recharge/discharge relationship through the vadose zone affect DNAPLs fate in the groundwater, involving various aspects of the migration of contaminants (acting physical and chemical properties of the compounds). Large uncertainties are thus associated with the vertical barrier system, especially in term of its hydraulic conductivity, that have never been tested in the past and that are not possible to test currently due to the risk of enhancing contaminant migration in the subsoil. Only a few historical monitoring data were available; moreover, these data were extremely discontinuous and basically were not very useful. The only reliable and available historical data were the chlorinated compounds concentrations in wells and in piezometers downgradient from the containment area. At the Rho site, the high and continuous contaminant concentrations in groundwater indicated that the containment system seems to have lost its effectiveness since it was constructed in 1981. Two possible conceptual models are most likely to explain this, as shown in Fig. 2. In the first case (Fig. 2a up), consistent flow occurs outward from the encapsulated area due to lateral spreading and the fact that the aquitard is almost impermeable below source area; the A1 aquifer is polluted by deepening of DNAPLs through discontinuities in the aquitard, and from long-term diffusion processes. In the second case (Fig. 4b below), flow comes from lateral spreading directly into the A0 aquifer, and into the A1 aquifer through the bottom aquitard. The integrity and efficiency of the lateral barrier is contrasted by poor hydrodynamic efficiency of the geological barriers.

Site characterization: in situ monitoring and laboratory tests As a method to cope with problems related to the

containment area, a 2-year monitoring and testing program was developed, due to the need to obtain the necessary information in order to increase the knowledge of the contaminant migration and to support the ultimate remediation design. It should be considered that a thorough hydrogeological characterization can give great benefits in uncertainty reduction in human health risk in cases of small contaminant source, as the one analysed in this study (de Barros et al. 2009).

Because the major uncertainty came from the unknown geometry and permeability of the barrier, there was a need to deeply investigate the geological features of both the vadose zone and the saturated domain in the source area. To avoid the risk of further spreading the contamination by direct surveys in the slurry wall, a monitoring system of both the vadose zone and the aquifers was designed in order to support the definition of a reliable conceptual model (Hudak and Loaiciga 1999). The system was also used for groundwater modelling in the unsaturated and saturated conditions (Chen et al. 1999). A great number of surveys and measurements have been performed to characterise the study area, using both in situ and laboratory tests, in order to collect geologic and hydraulic information of the area and to develop its geologic conceptual model. The location of the surveys is shown in Fig. 4. Standard monitoring of groundwater level and contaminant concentration of the A0 and A1 aquifers has been done both within the encapsulated area (herein, Zone 1) and outside it (herein, Zone2). A specific detailed program of activities was planned in zone 1 and included the following.

- Drilling six small boreholes (S-points in Fig. 4) with the direct push technique, to the maximum depth of 3.6 m, for a detailed reconstruction of the stratigraphy of the vadose zone, including the partial recovery of drilled soils.
- Performing double-ring infiltrometery tests to determine the vertical saturated conductivity of superficial soils, and pumping and slug tests on aquifers and aquitard A0–A1.
- Installing two continuous electrical data loggers for the automatic monitoring of soil moisture content to a maximum depth of 150 cm based on the frequency domain reflectometry (FDR) technique. Each device consists of four probes, located at different depths.
- Installing three tensiometers at different depth for the measure of pore-water pressure in the vadose zone (manual reading) to a maximum depth of 120 cm.
- A recording station for temperature and rainfall.
- Standard laboratory tests on sampled soil have been done, including grain-size analysis, organic content, specific weight, and permeability tests.

Drilling

The analysis of the stratigraphy of the six S-points (Fig. 4) showed that the deposits in zone 1 are very heterogeneous (Fig. 5—SG stands for Soil Gas). This is probably due to the different nature of backfill used and to the presence of the remains of ancient anthropogenic structures in the subsoil. Human impact is remarkable and affects the hydrodynamic properties of the soils. In the past, the pilot area was dug out and then backfilled by waste materials from former structures of the chemical facility. At a depth of approximately 1–2 m, an undistinguished, fine, reddish horizon can be found locally; it is mainly constituted by bricks and other waste material (Fig. 5). This horizon is not continuous and has also been found randomly in the containment area during some surveys performedin the past.

Hydraulic conductivity tests

Saturated hydraulic parameters were obtained from multi-scale sampling tests. Depending on the investigated representative volume, different orders of magnitude should be obtained, due to an increase in soil heterogeneities for different and controversial reasons. However, even if largescale tests (i.e., pumping tests) are assumed normally to deliver better global effective transmissivity values, here the interest was also to characterize small-scale hydraulic properties. In fact, small features (e.g., higher conductivity pathways) are crucial in the solute migration fate into lower conductivity media (Zheng and Bennett 1997).

Falling head permeability laboratory tests were carried out on aquitard A0–A1 samples and an average $1.2 \cdot 10^{-7}$ m s^{-1} was obtained. As shown by double-ring infiltrometery test results, the surficial layer showed different degrees of compaction. This significantly affects infiltration at a local scale, even if the actual influence of this factor on the overall infiltration in zone 1 needs to be investigated through a numerical analysis. According to these field tests, vertical saturated hydraulic conductivity was measured with a total variation of three orders of magnitude (from 10^{-4} to 10^{-7} m s^{-1}), with a consequential high sensitivity to the compaction of the surficial layer. Traditional pumping tests and a slug test were also performed to investigate the hydraulic conductivity of the two aquifers, A0 and A1, and of aquitard A0–A1, respectively. Overall results are presented in Table 1.

Vadose zone monitoring

Monitoring of the vadose zone was a key point in the program, because it allowed the quantification of the recharge rate in the area. Figure 6 represents the observation of soil moisture content using the two continuous electrical data loggers. EVS-1 records are shown on the left side of Fig. 6, while EVS-2 are plotted on the right. The dataset series we considered are limited for both devices to a close time window within the monitoring program. Actually, this monitoring lasted twelve months because of instrument failure during an exceptionally warm summer period. During the considered time, at least two major rainfall events (gray line) occurred, at a rate of more than 5 mm h−1. The texts within the plots indicate the depth of the lecture probes, and thus at which depth the corresponding dataset refers to. The functionality of probes was manually tested with occasional surveys. Some critical aspects concerning the heterogeneous distribution of the soil hydraulic properties may be observed and commented upon. Because of the absence of rainfall events, all probes (excepting the shallower probe of EVS-2) maintain a constant value of moisture content. In specific, probes at 60-, 90- and 120-cm depths for EVS show values above 30% with no remarkable response to rainfall events. This behaviour is typical of fine-grained deposits, which are sparsely distributed at different depths in the study area (see Fig. 5); indeed, EVS-2 was located in a sector with the anthropogenic fine and strongly consolidated deposits. A different question concerns the 30-cm depth probe in EVS-2 which showed an unrealistically fast response to precipitation, probably due to preferential flow (poor adhesion of the sensors to the soil at low depth). On the other hand, EVS-1 show lower mean values of moisture content, and a faster response to rainfall events. We can assume that at this point the deposits can be coarser than the ones at the same depths around EVS-2 locations; especially, at depth 80 cm the moisture content is extremely low and we still observe a minimum response to rainfall events. It is probable than a coarse grained formation is encountered at this depth and that a higher hydraulic connections takes place during recharge events. In general, the results show a high degree of spatial variability of the hydraulic property of the soils, both in the vertical direction and, referring also to Fig. 5, in the horizontal plan.

Laboratory tests

Falling head hydraulic conductivity tests were previously described as performed on fine-grained sediments composing aquitard A0–A1. Other laboratory tests were also carried out. Grain-size analysis and organic content tests were executed on samples collected from drilled core samples at different depths. The purpose was to better characterise the stratigraphy of the upper 3.6 m and to obtain data to indirectly derive the hydraulic function of sediments in the vadose zone. This aspect is extremely important, because these layers influence the infiltration process within the containment area the most. In this way, the determination of reliable physical functions is a key point to obtaining a correct infiltration rate during rainfall events, and to determine the local recharge of the area.

Groundwater heads monitoring

Groundwater levels were monitored twice a year using about twenty piezometers outside the encapsulate area (zone 2), while three points have been chosen to be measured almost daily. These points were located according to the following scheme (Fig. 4): one in zone 1 within the A0 aquifer (A0-int) and the other two in zone 2, just outside the walls, in A0 and A1 aquifers (A0-ext, A1-ext), respectively. The aim was to detect different hydraulic behaviors of the inner and outer zones of the isolated A0 aquifer, in comparison with the A1 aquifer. Measurements in a long, dry period allowed us to observe that the progressive decrease of groundwater head at points A0-int, A0-ext and A1-ext occurred with three different rates (Fig. 9). As they were not influenced by external factors such as precipitation or well pumping, they were considered optimal for model calibration. It is also important to highlight that evapotranspiration processes from the saturated layer were not active either, mainly because piezometric levels were at least 4mbelow the topographic surface, and also because of the presence of low conductivity horizons of backfill at low depth, which inhibits such processes.

Numerical modelling

Model settings

The conceptual model used in the numerical analyses covers the entire containment (about 25 m width, see Fig. 2), up to an average depth of 30 m. The conceptual model included the following assumptions, and according to the hydrostratigraphic scheme in Fig. 2, the model is composed of (Fig. 7): (1) a shallow aquifer (A0), from the ground surface to a depth of 9 m; (2) a fine-grained aquitard (A0–A1), which divides A0 from the lower semi-confined aquifer and has been located from a depth of 9–11 m; (3) the second main aquifer (A1), from a depth of 11–30 m; and (4) the lowest fine-grained aquitard (A1–A2), up to a depth of 33 m. For the top layer, a finer vertical discretisation was chosen to take into account the upper, thin heterogeneities and to increase the ability of the model to effectively evaluate their influence on the recharge dynamics. For the first attempt, the layer was divided according to the position of the device sensors for the water content and tensiometers. The vertical barrier is represented by means of fine-grained materials located from the top surface to a depth of 9.30 m (30 cm inside the aquitard, to represent the keying). The barrier system is 17-m wide, and occupies one third of the total depth of aquifer A. The profile is oriented along the local groundwater flux in the A0 aquifer (Fig. 4). An imposed-valued function (Dirichlet conditions) was adopted as boundary conditions on the lateral sides. SEEP/W allows the use of “infinite elements” to simulate far conditions, limiting border effects on the model (GEO-SLOPE International Ltd. 2002 2006). The bottom side has an imposed-derivative function (Neuman condition), for which no flux was assigned for this problem, since leakages through the second aquitard are considered negligible.

Model calibration

Calibration and verification tasks were performed to identify unique series of proper parameters, adopting current techniques from the groundwater model literature (e.g. Anderson and Woessner 1992). Trial and error procedures were followed, and no automatic method was taken into account. The model was calibrated for both unsaturated and saturated properties. Three different sets of data, derived from the monitoring activities, have been chosen for optimisation: (1) agreement with head values in observation boreholes close to the studied domain; (2) agreement with pore-water pressure obtained by tensiometers monitoring; and (3) agreement with soilmoisture content measured by means of electric probes. As a first attempt, retention curves and hydraulic conductivity functions for materials in the vadose zone were determined using the Gupta and Larrson (1979) technique starting from the grain-size curve, with specific weight and organic content determined through laboratory tests. Hydraulic conductivity functions were calculated from the retention curves with the Green and Corey technique (Green and Corey 1971) using the experimental saturated hydraulic conductivity.

First step: calibration of retention curves

Two pre-rainfall reference temporal moments were chosen to reproduce initial steady-state conditions, with different saturation profiles. Transient analysis was performed to simulate the effect of the registered rainfall events, and applied as a boundary condition to the topographical surface. During the calibration phase, soil moisture content and pore-water pressures have been used, and the model was considered calibrated when there was a good agreement (residual lower than 5%) between experimental and calculated data. Soil retention curves were changed to reach this result, where points with the pair of pore-water pressure and soil moisture content measured at the same depth in the same time were considered as fixed. Examples of results of the calibration phase are shown in Fig. 8. When a randomly distributed heterogeneous layer, at a shallow depth into the soil, was assumed, the simulated soil moisture content fits well with observed values at different depths (on probe sensors) in the pilot area. Saturated properties of this horizon will be discussed later. The backfill layer had a significant impact and was configured with high moisture content and abrupt transitions to upper and lower horizons. This layer, in particular, was calibrated by assigning the first shallow layer of the model a saturated hydraulic conductivity (K_{sat}) = $5 \cdot 10^{-6}$ m s^{-1}; unsaturated hydraulic parameters were then derived from typical low-permeability materials, for example, adopting retention curves that showed a softer decrease of hydraulic conductivity with suction. In this case, however, volumetric soil moisture content holds saturation close to 100%, and the retention curve is not influential.

Second step: calibration of saturated conductivity

Following calibration of the retention curves, transient simulations were set-up for a period of six months, and divided into 600 time steps. A long, dry period was chosen in order to recreate a groundwater head drawdown trend of the system (drainage conditions) and to validate the model. In this way, no recharge flux was applied to the upper model layer, since no rainfall event occurred during the observed time. Two different functions for the imposed head boundary conditions of the models were used in order to detect the actual dynamics through the borders of the encapsulated area. At the upper A0 aquifer, constant head values were applied to infinite elements for the whole transient time in order to leave piezometric levels free to change within the model as timesteps were solved. At the lower (A1) aquifer, timevariable values were applied to the boundaries.

This function was chosen based upon observation at the A1 piezometer, located immediately close to the downstream wall border (Fig. 4). The same function was applied to the upper limit of the A1 aquifer, but modified in absolute values according to the measured hydraulic gradient. The outputs of the transient state simulation were verified by comparing the observed exhaustion curve in aquifer A0 piezometers (Fig. 4) and the computed head.

Initial parameter values were assigned according to laboratory measurements and the in situ test results described in the previous paragraph. Also, for the vertical barriers, since no hydrodynamic information was available, an initial range of lowpermeability values from 5·10−7 m s−1 (i.e., similar to the aquitard vertical conductivity evaluated by laboratory test) to around 5·10−9 m s−1 (typical values of well-consolidated slurry cut-off walls) was considered. Figure 9 shows the good agreement between the observed and computed values. It must be noted that real values tend to oscillate after reaching the bottom of the A0 shallow aquifer while calculated the curve flattened out. This behaviour can be related to the formation of small water pools on the small-scale irregularities at the top of the fine grain aquitard that were not analysed in the model. A randomly distributed, heterogeneous layer at a shallow depth into the soil was maintained, as observed by the "stratified" soil moisture contents (Fig. 6). Table 2 summarises the final hydraulic properties of both the aquifer and the aquitards after the calibration process.

Model results

Water release incidents occurring from the isolated zone were assessed by evaluating flux sections on the lateral and bottom surfaces of the isolated area. An average value (the integral) of the total leakage from one boundary was then calculated. Figure 10 shows a synthetic plot of calculated water leakage. While values at the bottom surface seem to maintain a generally stable amount at 0.5m2 d−1 during the simulation, lateral values seems to decrease from an initial 0.75 to 0.65 m2 d−1 in the simulation time (180 days), delivering a general reduction of about 20% in terms of normalised flux (lateral vs. bottom spreading ratio). In any case, the results showed that outflow is given by both the lateral boundaries through the slurry wall, and the bottom boundary through the aquitard. This result matches the second hypothesis (Case 2) we formulated as a reliable conceptual model for the study area (Fig. 2—down). It is interesting to note that losses are generally higher through the slurry walls, with values reaching about 18 m3 d−1, while the discharge through the bottom is about 13 m3 d−1. This gives a value for total losses of more than 30 m3 d−1, which represent a significant amount of contaminated water released by the containment system.

Evidently, a direct correlation between the saturated volume of the aquifer within the containment system and the lateral spreading exists, while the leakage from the bottom is only slightly dependant on the saturated volume. That means that increasing head within the system has only a small effect on the amount of total leakage; the leakage mainly increases or decreases according to the changes in the filtration surface, which is given only by changes of the lateral filtration surface.

Scenario simulations in recharging conditions

According to the importance of the hydraulic head values within the containment area, as described in the previous paragraph, and in order to describe qualitatively and quantitatively the systems reaction to recharge, a scenario analysis based on two simple cases (A and B) was set-up, based on the discharge model but with increased total simulation time. We focused on the infiltration contribution as a critical point. In scenario A, the hydrogeological background was maintained with conditions identical to the calibration processes. Measured rainfall rates were available at a meteorological station near the Rho site, and were applied as the inward flux (Neuman boundary conditions) at the nodes of the top layer. For the sake of the problem, we assumed complete effective infiltration of water, and also discarded any topographic gradient, and thus no watershed may deliver secondary recharge from the top. In scenario B, we applied the same Neuman boundary conditions, but a homogenous aquifer was considered inside the containment system, as if no buried waste materials existed. In this way, by comparing results of the two scenarios, the actual impact of the backfill on the dynamics of the infiltration process was calculated. The main limitation was the lack of experimental data to directly “verify” scenario outputs, which delivers uncertainty to the analysis even if the numerical models have been thoroughly calibrated. In addition, due to high CPU calculation times (up to 1 week for each simulation of full drainage condition), both scenario simulations have been limited to a short single rainfall event (5 days) after the drought period (in which complete drainage was simulated).

The results revealed the importance of the human impact in the dynamics of aquifer, affecting the vertical recharge profile, and thus the calculated leakages, both through the lateral and bottom sides. Figure 11 shows that leakages from lateral conditions are much higher in the case of incidental heterogeneities in the soil (case A, dark continuous line) than in the case of homogeneous conditions (case B, grey continuous line). On the other hand, the homogeneous aquifer (case B) reacts promptly to instantaneous recharge; bottom leakages react quickly, but then reach an asymptotic value close to 0 m2 d−1. Only lateral flux exists, but lateral leakages become stable at least until the end of the simulation. For the heterogeneous case (A), leakages react later but asymptotic values with non-negligible values are obtained.

This behaviour contrasts with the assumed hypothesis hat in homogeneous conditions (B), infiltration could have been more effective, reaching the bottom of the aquifer, and generating bottom flux. On the other hand, higher values are obtained from lateral spreading in heterogeneous case (A). We can give a physical explanation assuming that the heterogeneous layer, made up by impermeable but saturated media, acts as a “barrier” which makes preferential paths diverge in lateral directions rather than vertically downwards.

Since it is located at shallower depths, water avoids infiltrating into larger unsaturated volumes, which would retain more water and take longer for enough soil moisture content to accumulate in order to permit gravitic seepage. Therefore, since fewer volumes are affected, seepage occurs rapidly and diverges. Figure 12 offers a snapshot of seepage direction (indicated by dark arrows) close to the upstream vertical walls during the initial phase of the infiltration process in the heterogeneous (A) case.

Discussion

The results showed that the poor global performance of the containment system stems from a lack of knowledge about the geologic structure of the area. In fact, a remarkable exchange between the inner and the outer part of the confined source area occurs through the aquitard horizon.

The slurry walls seem to have lost most of their containment properties and presently they have almost the same amount of losses as the aquitard. However, it is probable that in the past, when walls were close to the top of the performance, they may have played a role as a funnel and allowed the plume to head even more directly towards the lower layers. The lack of any other remediation system coupled with the main system contributed to the development of the present groundwater contamination in an extended area.

This indicates, firstly, the need for a revision of the existing remedial action, which can be effectively designed only with the addition of reliable knowledge of processes analysed and quantified, both in the vadose zone and in the aquifer, through a complete and calibrated numerical model. Other, further enhancements may include a better knowledge of the spatial distribution of heterogeneities in the soils. This is remarkably important, especially for anomalous features, to better assess the local influence of infiltration to the general behaviour of the source area. Further solutions by indirect analysis (via numerical method) allowed us to estimate the approximate assets of the subsurface engineering structures. The system seems particularly sensitive to:

– the state of the slurry walls, in terms of hydraulic retention capacity;

– the geometry and the extension of the aquitard on a detailed spatial scale; and

– abrupt changes in the composition of the soil, due to the presence of waste backfilled materials.

Slurry walls are only partly responsible for the lack in performance of the containment system. A non-negligible flux takes place through the aquitard and inevitably contributes to the spread of the plume. Waste backfill could intervene in the alteration of the natural infiltration process of the encapsulated aquifer. The low transmissivity of this fine-grained horizon, even if it is located in heterogeneous and isolated spots, tends to reduce and delay the water infiltration into the shallow confined aquifer.

Conclusions

The present study aimed to achieve quantitative results about the overall effectiveness of a containment system, without using invasive technique, and to demonstrate the feasibility of this approach for future uses. In order to assess global water release from a containment system, an unsaturated–saturated flow model has been calibrated through an extensive monitoring system developed for the specific case study. Both unsaturated and saturated hydraulic parameters were used, whose reliability came from multi-scale tests. Laboratory and in situ-derived unsaturated (soil moisture content and hydraulic conductivity curve) and saturated hydrodynamic parameters were optimised through model calibration in both steady-state and transitory-state conditions. This allowed us to hydraulically characterise all materials, including waste backfilling, even if their actual exact spatial distribution remains unknown due to the lack of information. However, the influence of these heterogeneities on the infiltration process has been generically established and, for a general view, considered as an impediment to normal hydrological cycle and recharge processes.

The most important thing to note is that, presently, losses through the containment system come both from lateral spreading directly into the A0 aquifer, and into the A1 aquifer through the bottom aquitard, according to case 2 proposed in "Hydrogeological setting". Thus, the hot spot in the containment system can be considered responsible for contamination found downgradient in both the aquifers.

The results of this research will be also a reliable base for future development of the study, concerning development of transport, multi-phase modelling. They also furnish the initial input to generate a quantitative analysis of the risk of further spreading of contaminant from the polluted site. Nonetheless, in spite of the fact that stochastic methods are a suitable way to deal with soil heterogeneity, further consideration of the efficiency of the barrier may be obtained only with new direct investigation in the site.

In any case, the combination of monitoring and modelling allowed us to make a reliable estimate of the overall performance of the physical confinement and its possible influence on groundwater contamination, without using any invasive techniques on slurry wall containment systems, as is often required because of the potential risk for an increased release of contaminants.

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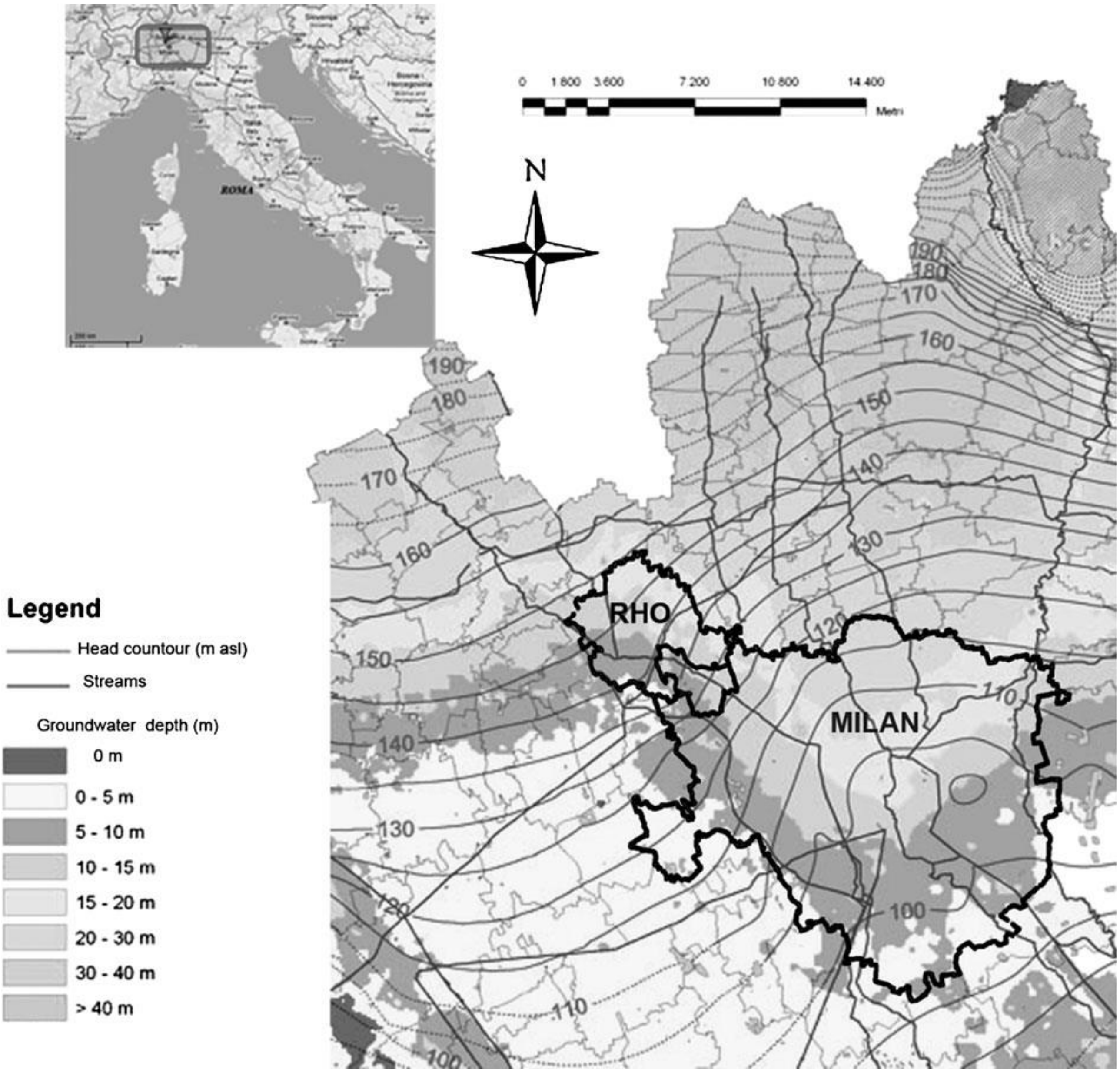

Fig. 1 Location of the area and main hydrogeological features (modified from www.provincia.milano.it). The metropolitan area of Milan is located a few km downstream the polluted site, along the main direction of the regional piezometric trend

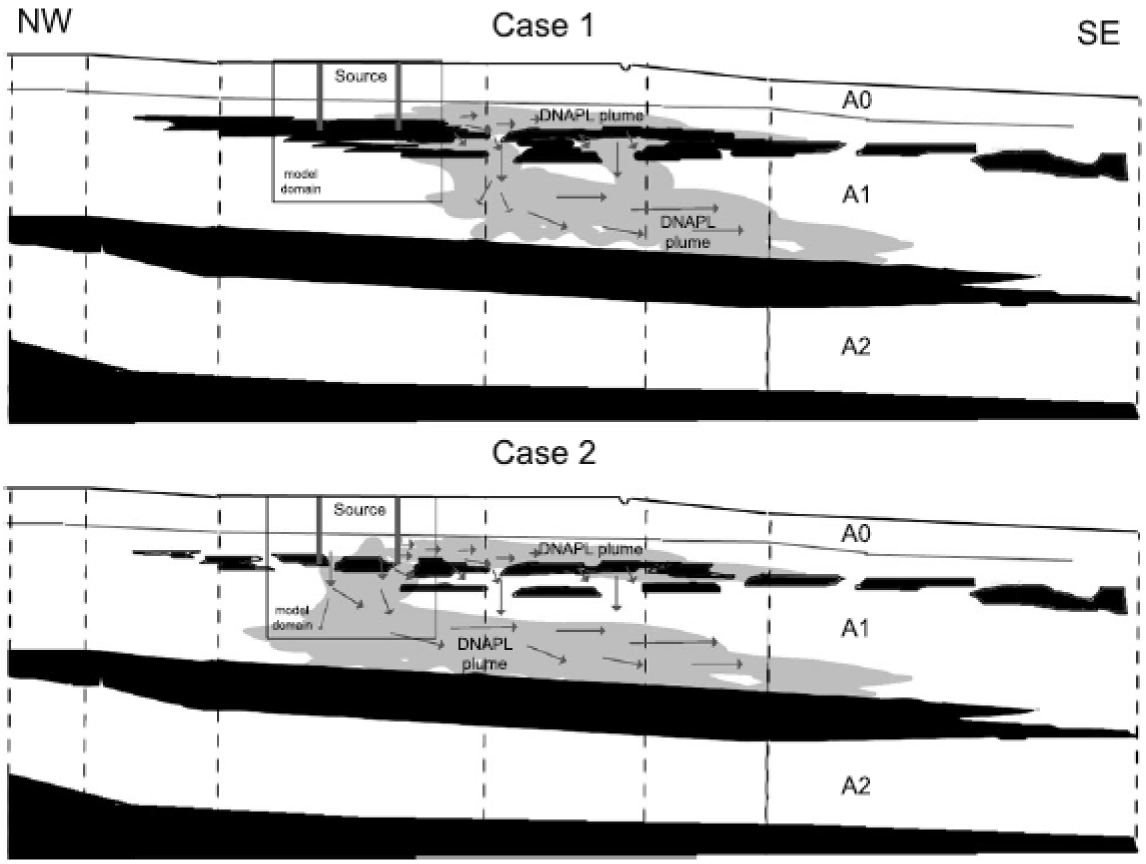

Fig. 2 Alternative conceptual models describing the hydrogeological division of A aquifer in the Rho area. The DNAPLs leak into aquifer A1 depending on the aquitard efficiency below the encapsulated area. Case (1): The aquitard is more efficient to contaminant leakage than lateral barriers. Spreading into A1 mainly occurs through discontinuities along the aquitard. Case (2): Aquitard and lateral barrier efficiencies are equivalent; spreading into A1 occurs also at the bottom of the encapsulated area

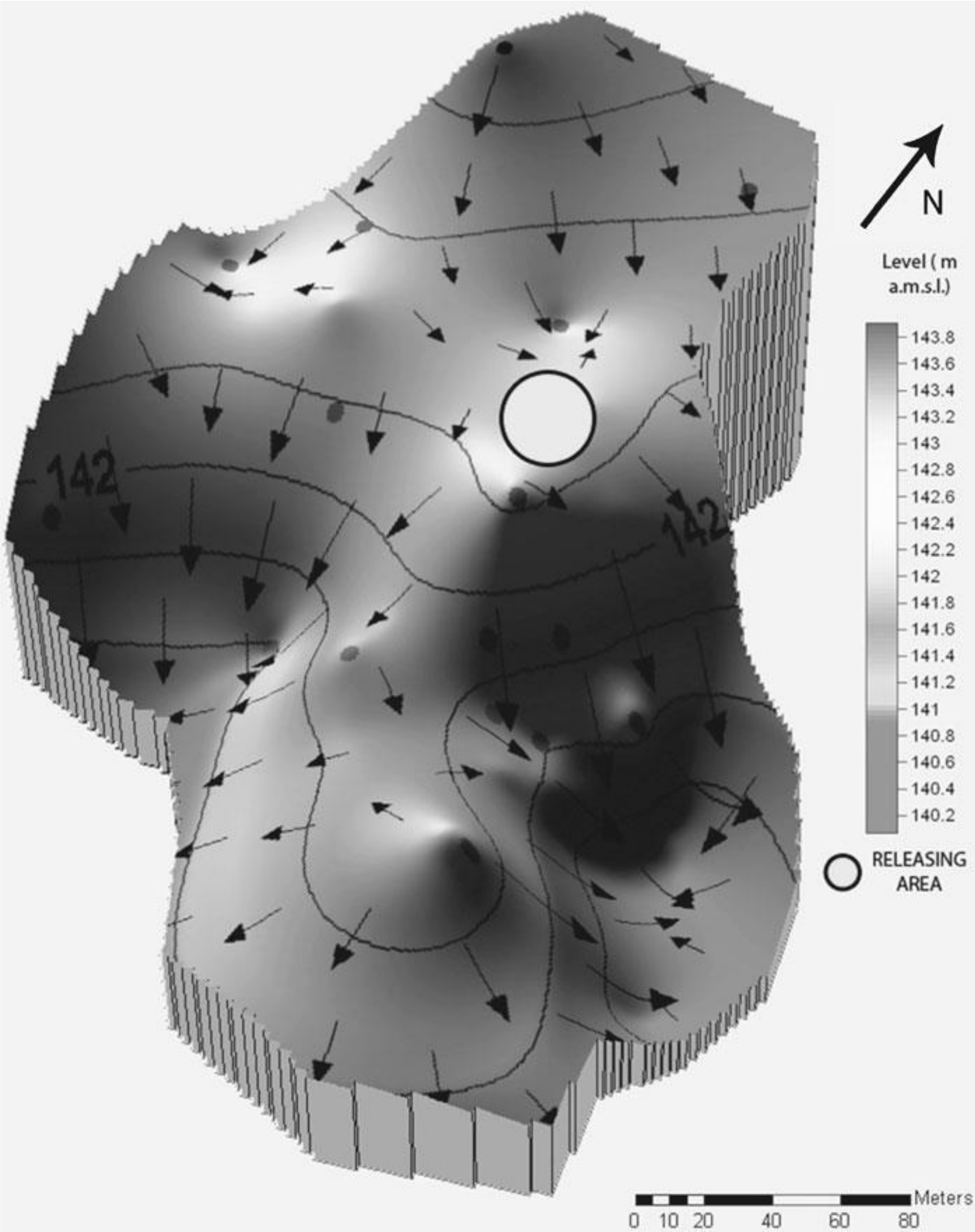

Fig. 3 3D mapping of the A0/A1 aquitard at Rho-Bianchi former chemical facility, after Pedretti et al. (2009). The shadow area is where gradients are maximum towards the southern portion of the domain, thus acting as a potential preferential migration zone of DNAPLs

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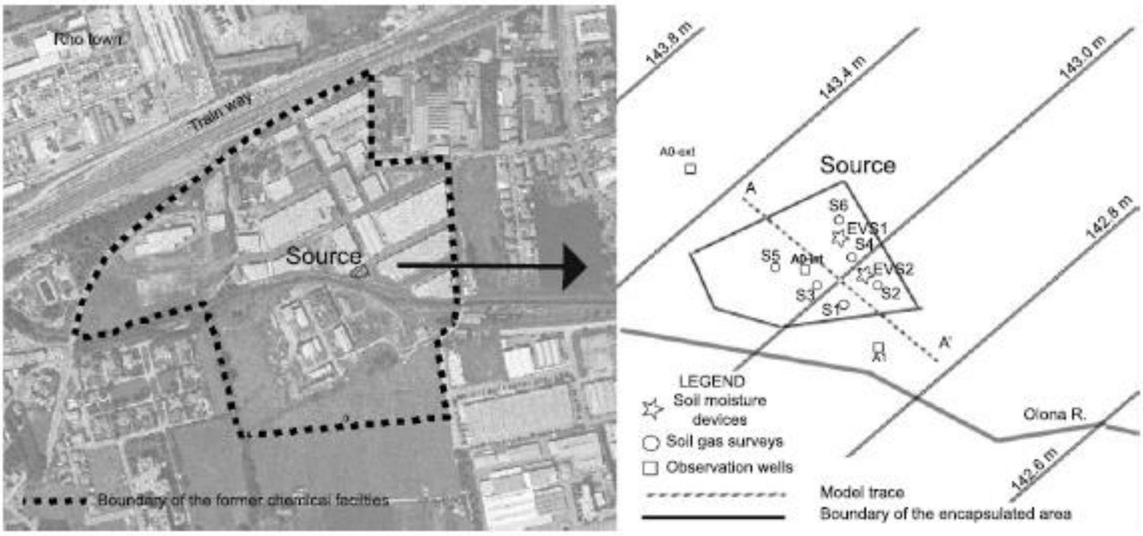

Fig. 4 Synthetic sketch of the local set-up, depicting piezometric isolines (m) of the A0 aquifer (taken from regional piezometry); the SEEP model A’–A trace; the relative position of observation wells, small S-boreholes and EVS devices. The encapsulated zone is Zone 1; the external area is Zone 2

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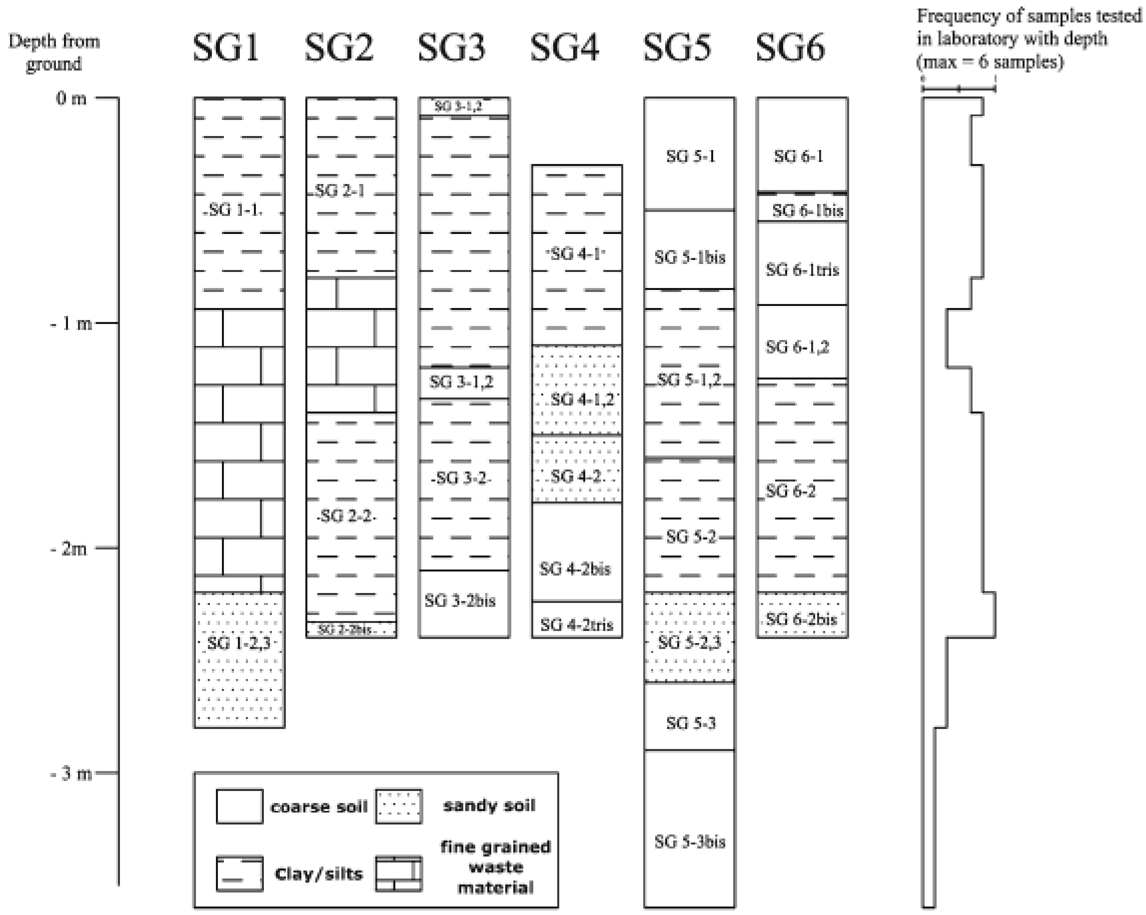

Fig. 5 Stratigraphy and depth of soil samples for laboratory test. The soil is extremely heterogeneous both in the horizontal and in the vertical direction. In some case, anthropogenic materials were observed at shallow depths

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Table 1 Experimental hydraulic conductivities at different horizons of the Rho aquifer using different testing devices

Layer	Saturated hydraulic conductivity (K_s) m/s	Method
A0	3.50E-04	Pumping test
A1	2.00E-03	Pumping test
Aquitard A0–A1	8.30E-08	Slug test
Mean	1.20E-07	Laboratory
St. Dev.	1.35E-07	Laboratory

For laboratory experiments, mean and standard deviations resulting from repeated measurements are also reported

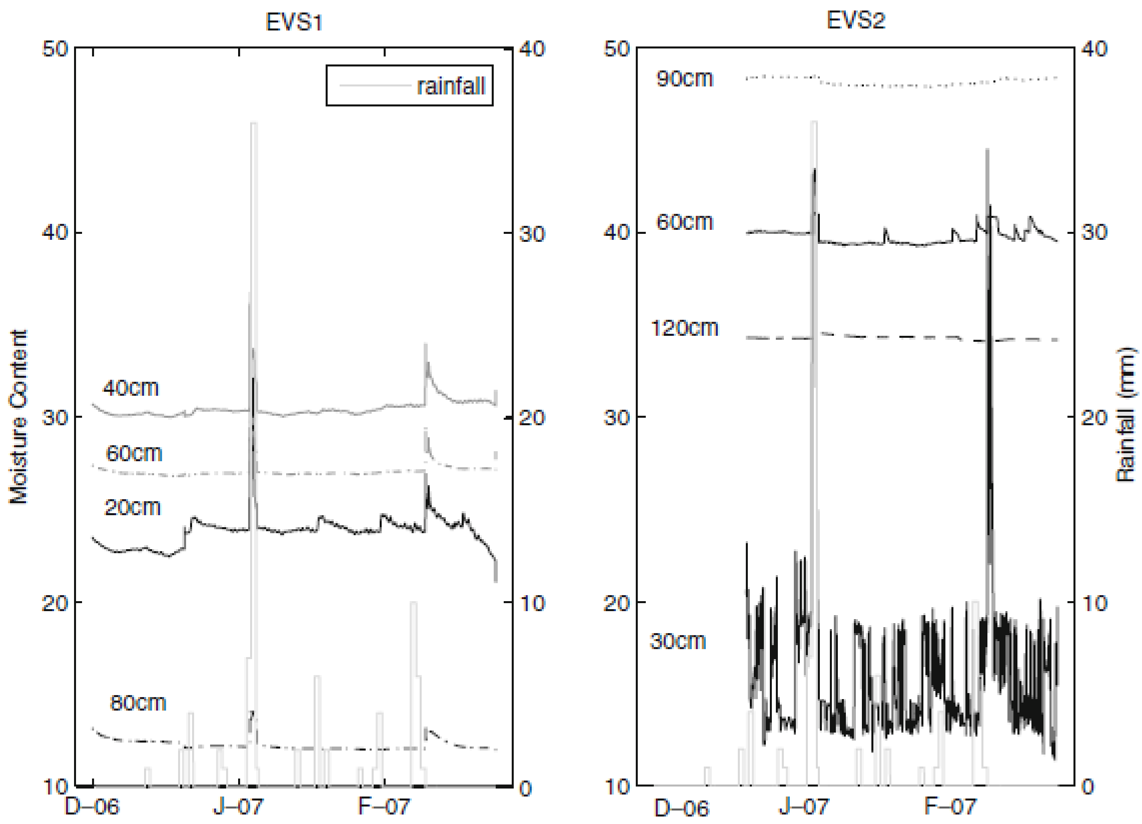

Fig. 6 Experimental soil moisture values at different depths and rainfall heights. It can be seen that the soil is heterogeneously stratified by layer of different moisture content. For the same depths, the device recorded in most cases different values, which reflect the horizontal heterogeneous spatial distribution of hydraulic properties

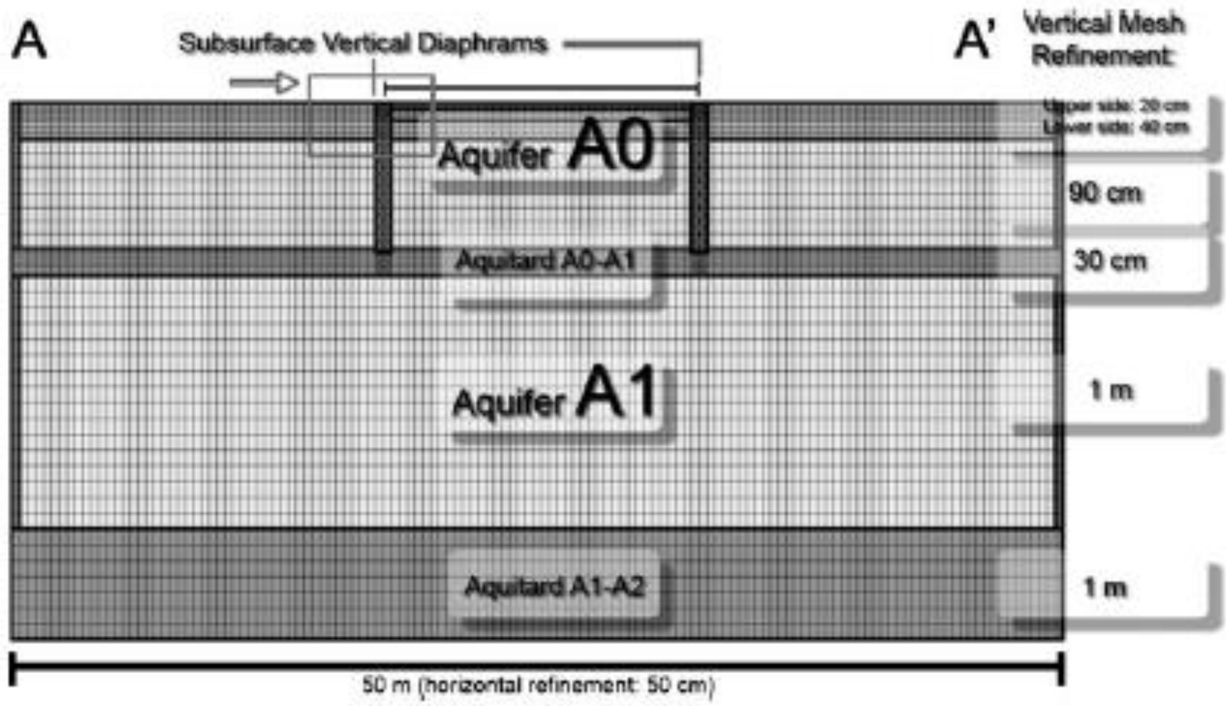

Fig. 7 Hydrogeological conceptual model for numerical analysis. At shallow depth the grid has finer refinement to reproduce the experimental observation in the first centimeters of the encapsulated area

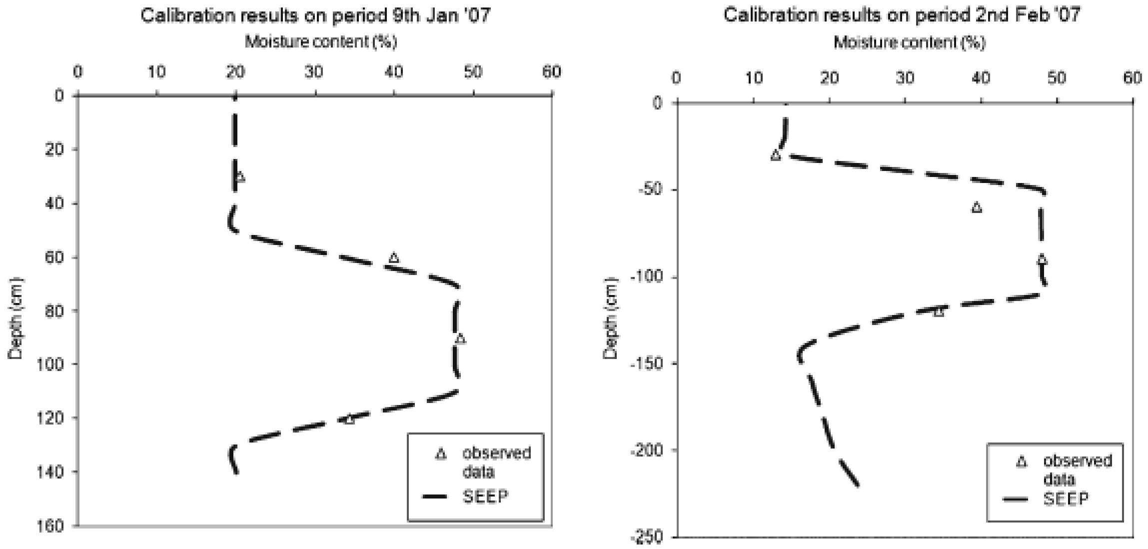

Fig. 8 Results of model calibration for unsaturated flow. The numerical simulation fits the experimental data if a layer with lower permeability and high moisture content is set at a shallow depth

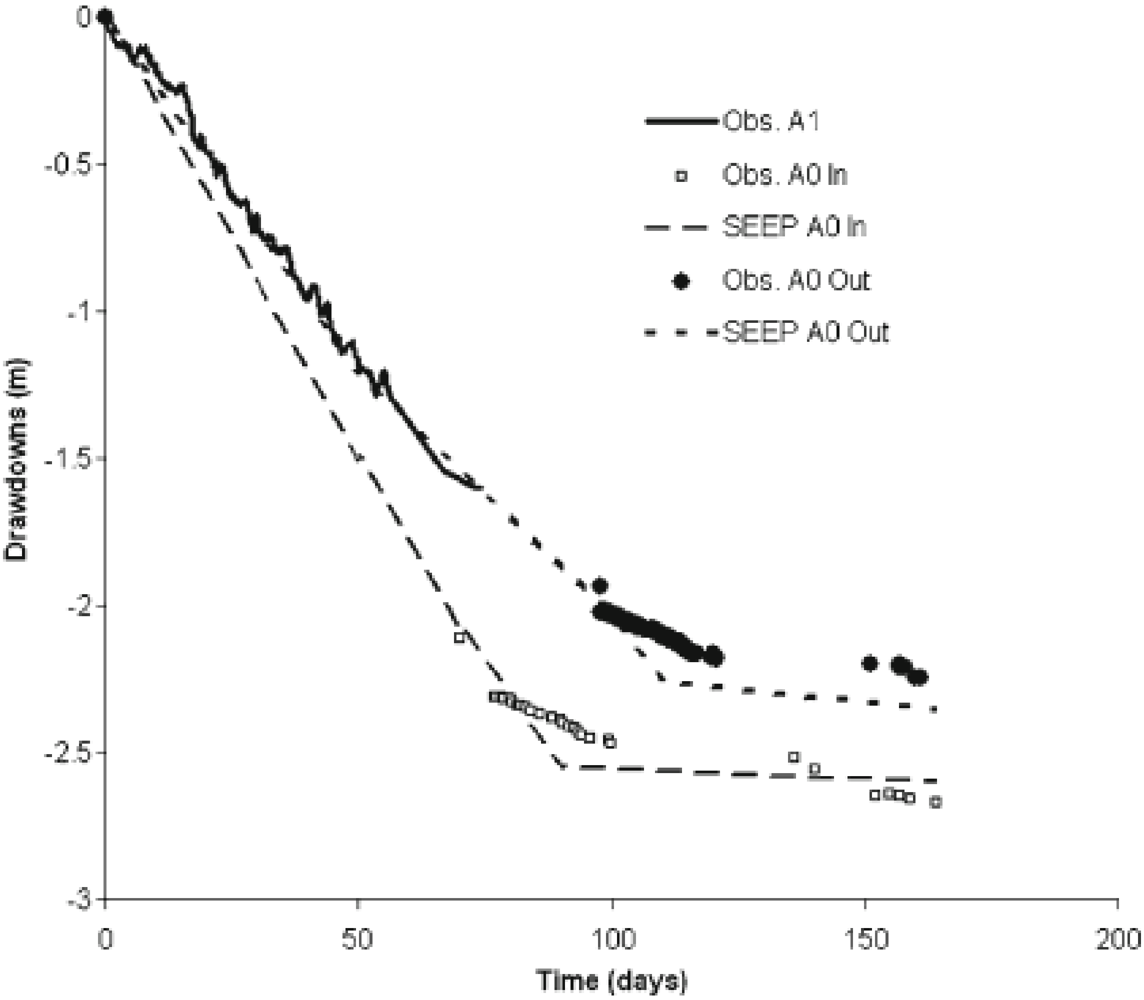

Fig. 9 Observed and simulated drawdown data at two observation points. Value 0 (zero) on drawdown axis represents the reference data measured in the aquifers at the beginning of the drawdown phase

Table 2 Hydraulic conductivities adopted in the SEEP model for different geological units and anthropic features

Layer	Model K_s m/s
A0	3.50E-04
A0 - Brick layer	5.00E-06
Aquitard A0–A1	2.00E-03
A1	1.24E-7
Aquitard A1–A2	1.00E-08
Slurry walls	5.00E-08

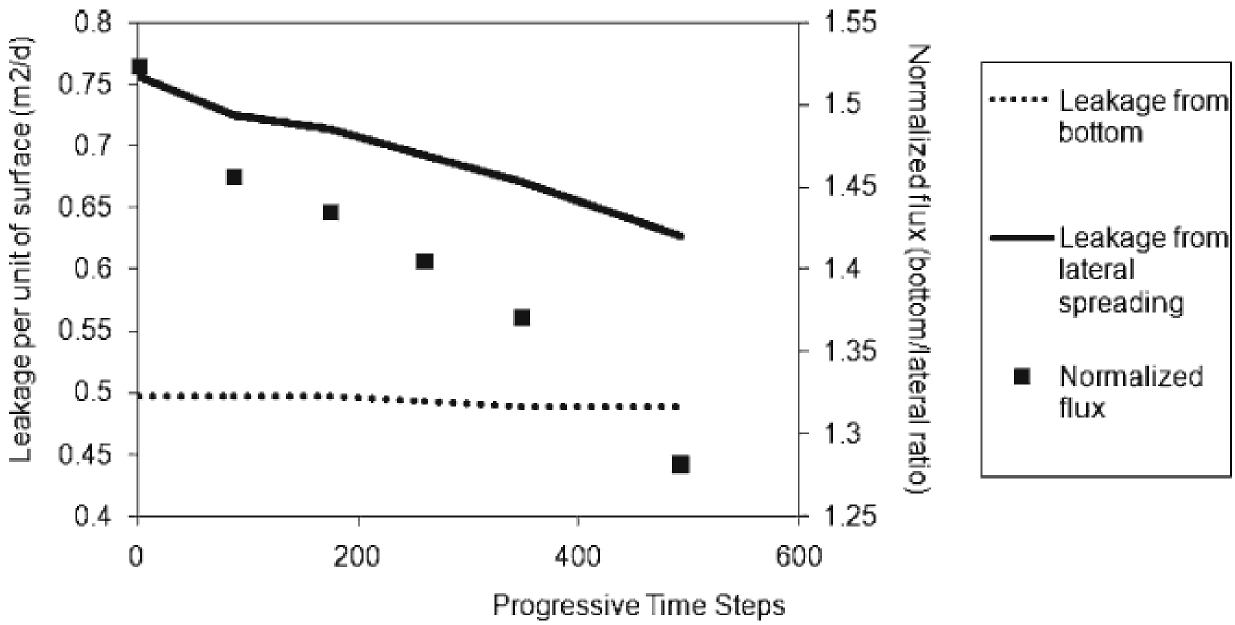

Fig. 10 Change of leakage during the transient simulation through the containment system. The values are expressed in terms of absolute values of the lateral and bottom flux per unit of wall length (left axis) and of the relative proportion of lateral flux with the bottom (right axis)

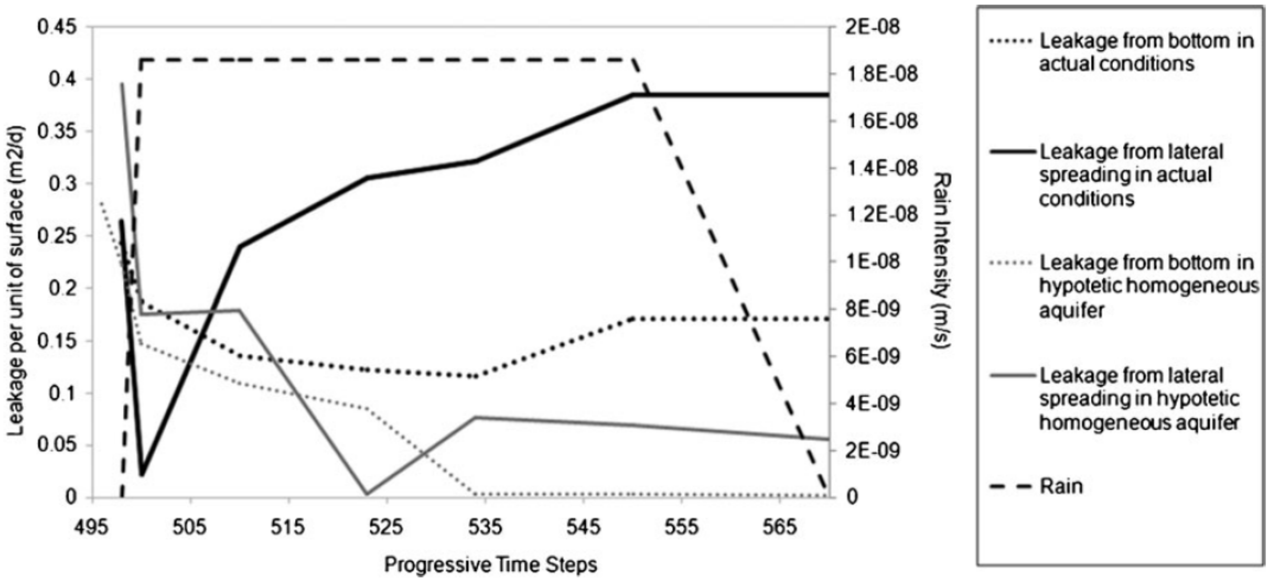

Fig. 11 Evaluation of the change of leakage in time through the containment system under recharge for different scenarios

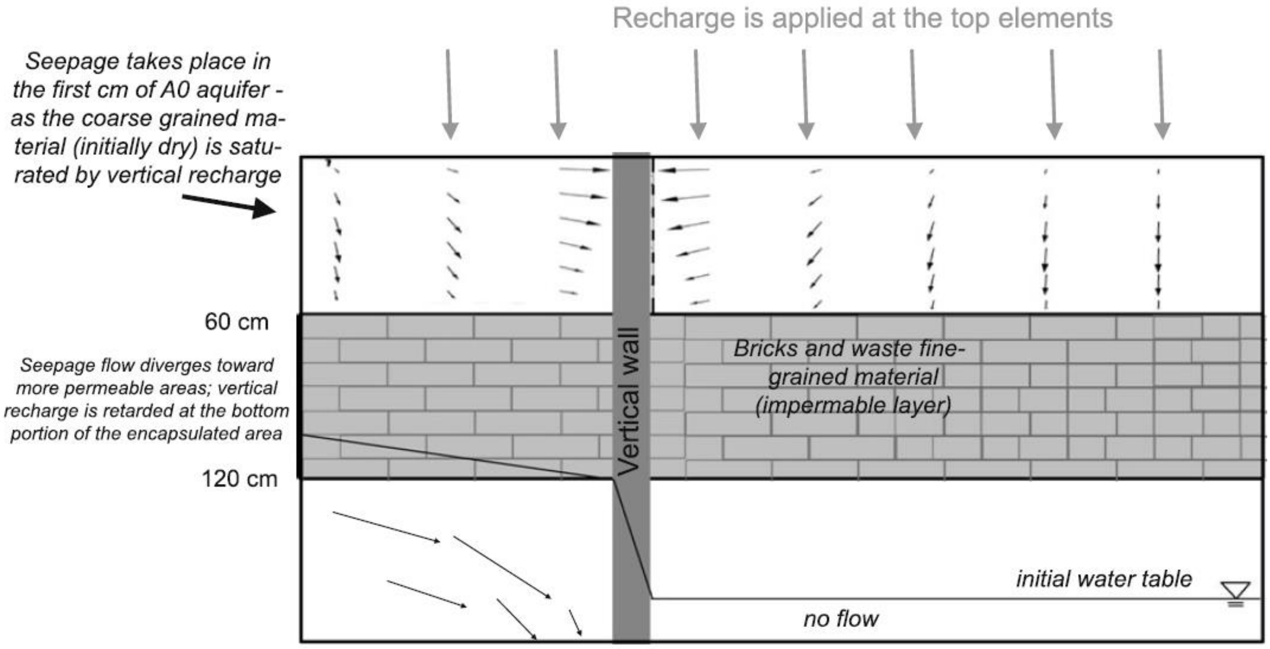

Fig. 12 Qualitative effects of waste backfill and slurry wall on the seepage vectors occurring after a recharge event. The figure refers to the area highlighted in Fig. 7