DC resistivity and induced polarisation investigations at a waste disposal site and its environments

E. Aristodemou *, A. Thomas-Betts

The T.H. Huxley School of Environment, Earth Science and Engineering, Royal School of Mines, Imperial College of Science, Technology and Medicine, Prince Consort Road, London SW7 2BP, UK

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Abstract

We report the results from some geoelectrical surveys carried out to monitor the spread of contamination in underlying aquifers due to a landfill site. The geophysically determined electrical properties of the aquifers have subsequently been used to estimate hydraulic conductivities which are required for modelling contaminant transport. The type of waste deposited and the influence of the geological environment were the crucial factors investigated, employing the DC resistivity and time domain induced polarisation (IP) methods. The landfill was mainly a liquid disposal site with existing borehole information showing that the waste contained high concentrations of not only inorganic material (chlorides, sulphates) but also organic matter (indicated by high values of chemical oxygen demand (COD) and total organic carbon (TOC)). The measured fluid resistivities from the surrounding boreholes showed values as low as 0.25 Ω m, leading us to expect low bulk resistivities \( \rho_b \). The aquifer system in the study area, with an eastwards regional groundwater flow, consists of three sand aquifers with intervening semi-pervious clay aquitards. Clay particles are also present in the sand formations and expected to influence the overall bulk resistivity and chargeability. The presence of organic waste is another factor suggesting that the IP method could be employed as a diagnostic tool. Our resistivity measurements along the survey lines perpendicular to the groundwater flow show systematic reductions of resistivities relative to the control line, the effect decreasing progressively eastwards from the landfill. The resistivities of these contaminated sections were higher than expected and one possible explanation for this could be the presence of the organic waste. However, an alternative explanation could be the low porosities in the sand formations. Low porosity may imply reduced fluid content, and therefore increased bulk resistivities, as well as lower hydraulic conductivity values. Field and core hydraulic conductivity measurements in the area from previous investigations had indicated the relatively low hydraulic conductivity values of \( 9.3 \times 10^{-7} \) and \( 6.2 \times 10^{-6} \) m/s for the top and middle aquifers respectively, thus, favouring the low porosity hypothesis. The IP measurements showed high apparent chargeability values (80–120 ms) on top of the landfill, possibly due to the presence of disseminated solid metallic waste or the high organic load of the liquid waste disposed. The IP line parallel to the groundwater flow direction, and close to the landfill, produced chargeability anomalies which may be associated with a plume of organic waste. No chargeability anomalies are observed on the second IP line, further away from the landfill and in the SE direction. The bulk resistivities \( \rho_b \) obtained from the resistivity inversions and the fluid resistivities \( \rho_f \), from adjacent boreholes, allowed hydraulic conductivities to be estimated. The intrinsic formation factor was first determined from \( \rho_b \) and \( \rho_f \) and was then used in conjunction with a range of Archie’s parameters appropriate for sands to evaluate porosity and its likely bounds of error. The hydraulic conductivities obtained through the Kozeny–Carmen–Bear equation, for the geophysically determined range of

* Corresponding author. E-mail: e.aristodemou@ic.ac.uk

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porosities, give plausible values agreeing within an order of magnitude with each other and with reported values for the formation. © 2000 Elsevier Science B.V. All rights reserved.

**Keywords:** Waste disposal; Aquifers; Landfill

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**1. Introduction**

A vast amount of literature exists showing the application and limitations of the geoelectrical methods in environmental problems associated with groundwater contamination due to leachate movement. The DC resistivity method is favoured for such applications as the inorganic pollutants, present in most leachates, on entering the fluid pathways increase the fluid conductivity due to an increase of the number of ions in the solution. The overall bulk resistivity is therefore expected to be reduced and by monitoring and observing changes to it that are not geologically related, we can draw conclusions as to the spread of contamination. However, other types of pollutants forming a leachate, such as hydrocarbons, may have the opposite effect on the fluid conductivity. According to Benson et al. (1991), hydrocarbon plumes may be delineated as resistivity highs since hydrocarbons typically have high resistivities relative to water. However, although this is true, when inorganic compounds are added to the polluted groundwater for bioremediation purposes or when the hydrocarbons are being biodegraded, the total dissolved solids (TDS) are increased and the hydrocarbon plume may also appear as a resistivity low. This is also observed by Vanhala (1997) in his field experiments, although in this case the author attributes the decrease in resistivity to the detachment of ions from the grain surfaces following their interaction with the oil contaminant.

In most environmental cases, more than one method is applied at the same location as this improves interpretation greatly. Thus, many examples exist where the DC resistivity method is applied in conjunction with other methods such as the electromagnetic or the induced polarisation (IP) methods. Improvements in automatic data acquisition techniques and more importantly in data interpretation, through the development of fast two-dimensional (2D) or three-dimensional (3D) inversion software, increased the popularity of the geoelectrical methods in recent years. Applications dealing with determination of landfill boundaries, thickness of fill and spread of contamination are described by many authors (see, e.g., Barker, 1990; Buselli et al., 1992; Meju, 1993; Bernstone and Dahlin, 1998).

The application of the time domain IP method in environmental problems has been in the past mainly associated with groundwater exploration and with distinguishing between sand aquifers affected by saline intrusion and clay layers (e.g., Seara and Granda, 1987; Draskovits et al., 1990; Roy et al., 1995). However, examples such as those of Cahyna et al. (1990) and Vogeslang (1995) show its potential in contamination problems. Cahyna et al. (1990) applied both the DC resistivity and IP methods in a cyanide groundwater contamination problem where it was found that the IP method identified the contamination of groundwater while this was not possible from the resistivity results. In the Vogeslang (1995) case, IP measurements were carried out over a domestic waste site where high chargeability values (> 80 ms) were detected. These were attributed to the presence of either galvanic sludge or lubricants in the waste. Oldenburg (1996) discusses the joint application of DC resistivity and IP methods in a case study associated with acid mine drainage problems, while more recently, Frangos (1997) described the application of these methods over a simulated hazardous waste dump where high IP responses
Fig. 1. Map of the area showing the landfill, monitoring points of interest and the control line L3.
Fig. 2. Borehole log information associated with the monitoring points of interest.
Fig. 3. Schematic of the landfill — vertical cross-section.
Table 1
Closest survey: lines L1, L2 and L3 — August 1997.
EC: fluid electrical conductivity; COD: chemical oxygen demand; TOC: total organic carbon; NH$_3$–N: ammoniacal nitrogen.
(1a) Important water quality parameters — August 1997

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<th>Borehole monitoring point (MP)</th>
<th>Associated aquifer</th>
<th>EC (µS/cm)</th>
<th>COD (mg/l)</th>
<th>TOC (mg/l)</th>
<th>NH$_3$–N (mg/l)</th>
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<th>Sulphate (mg/l)</th>
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(1b) Important water quality parameters — April 1998

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<th>TDS (mg/l)</th>
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* MP17: no data available since 1997; Data shown from August 1996.
** MP26: no data available for April 1998.
*** MP32: results are reported to be an average over the whole borehole and to have been affected by high dilution due to heavy rainfall in March and April 1998.
**** MP37: results are reported to be an average over the whole borehole.
were detected and attributed to the disseminated metallic content of the waste.

In the past 5 years, much research has been carried out in relation to the spectral IP method and the identification of organic pollutants. This follows the work published by Olhoeft (1985) where the author discusses how different interfacial processes such as the oxidation–reduction process and the clay–organic interactions may be differentiated on the basis of the observed phase spectra. Most of the work published recently relates to spectral IP laboratory studies (Vanhala et al., 1992; Boerner et al., 1993; Vanhala and Soininen, 1995) where the authors use clay or sandstone samples to examine the effect of different contaminants on the complex conductivity. Field applications are reported by Boerner et al. (1996) and Vanhala (1997) where the authors discuss the application of the spectral IP method in (a) detecting organic pollutants in soils with both low and high clay particle content, and (b) in estimating the hydraulic conductivity of the aquifer.

Following the above examples in the literature, our research concentrated on the application of both the DC resistivity and IP methods in areas where groundwater contamination problems existed and were associated with landfill waste. As the geology of the study area consisted of sand and clay layers, it was felt that the IP method could be a suitable method in our investigations. In addition, we were interested in investigating whether the organic contamination detected in the area through high concentrations of chemical oxygen demand (COD) and total organic carbon (TOC) would show any effect on the IP response.

The specific objectives of our study were (a) the characterisation of the landfill waste in terms of both resistivity and chargeability, (b) the determination of the base of the landfill, (c) the geophysical identification of contamination in the underlying geological strata due to leachate movement, (d) the geophysical identification of clay layers in areas where these are shown to exist from other sources (e.g., from boreholes); the identification of these layers is important as clay layers act as natural barriers to the downward movement of leachate. In addition to the above objectives, we attempted to estimate aquifer properties such as porosity and hydraulic conductivity by combining existing measurements of fluid electrical conductivity with the geophysical inversion results.

### 2. Geological setting and site description

Our area of investigation lies in East Anglia, UK, where the geological setting consists of alternating layers of clay and sand of variable thickness. The sediments within the depth of interest are of Quaternary marine and glacial origin, overlying the London clay, which occurs at approximately 70 m depth. At greater depths, the London clay is believed to overlie a Chalk formation. Past geological studies show that the aquifer system consists of a sequence of three sand aquifers (top, middle, and bottom), with intervening semi-pervious clay aquitards (Richardson, 1996). A map of the area together with the boreholes of interest and the up-gradient geophysical control line is shown in Fig. 1. The plan view of the landfill with its lagoons and trenches can also be seen in this figure. The detailed individual borehole logs are shown in Fig. 2. From these, it is observed that the thickness of both the top and middle aquifers is variable throughout our study area. From the log of borehole MP32, the first sand aquifer lies at a depth between 7 and ~16.4 m, while the first semi-pervious silty sand/clay aquitard appears between 16.4 and ~18.0 m. For borehole MP14/15/16/17, the first aquitard appears at ~20 m depth with the top aquifer lying between 5.5 and 20 m. For borehole MP43, there seems to be no aquitard in the first ~29 m and top and middle aquifers (Corton and Cromerian sands) appear as one unit. In all cases, there is a thick top layer (overburden) of boulder clay whose thickness also varies from location to location, ranging between 5.5 (borehole MP 32)
and 11.0 m (borehole MP 37). There are no logs for boreholes MP4/5 and MP 6/7.

A schematic of a vertical cross-section along the central part of the landfill, from south to north, is given in Fig. 3. The site forms a small hill with its highest region approximately 7 m above surrounding ground level (~25 m above ordnance datum, OD) and with a flat region at the top approximately 150 m long. The water level is approximately ~23.5 m below the top flat region of the landfill (equivalent to ~1.5 m above OD). The natural clay base of the landfill in the northern part of the site is ~14 m from the top (~11 m above OD) while the base of the excavated trenches and lagoons in the southern site was ~17 m from the top (~8 m above OD).

Fig. 4. The main geophysical lines in relation to the landfill.
The landfill was set in operation in the 1950s after the closure of a clay pit used for brick making. Initially, the waste deposited was of domestic nature and it was placed in the south-western part of the present site. However, in the early 1970s, the deposition of liquid waste began lasting up to 1989 when the operation of the site was terminated. The site received both oily and non-oily waste with two of its lagoons, B1/2 and B3 in Fig. 1, receiving the liquid waste. These lagoons had a natural clay base, while lagoon A (also Fig. 1) and the existing trenches had no clay base at all, but were excavated straight into the underlying sand layer. From the limited information available on the type of waste deposited, it appears that 65–80% of the waste was agrochemical with solvents, ammonia, and heavy metals being deposited in small but consistent quantities. There was also regular disposal of mercury and copper, as well as of a number of volatile organic carbons and phenolic compounds.

As indicated in the vertical cross-section in Fig. 3, the liquid waste was disposed in the southern part of the site. However, 2 years before the closure of the site, a co-disposal operation was set-up with solid waste being deposited in both the southern and northern parts of the site. An inclined natural clay base was left in the northern part (≈ 11 m above OD) so that any accumulation of leachate would naturally drain into the lagoons in the southern part. The main reason for introducing solid waste to the site was to effect a reduction of a hydraulic mound that developed underneath the landfill, due to the deposition of the liquid waste. This hydraulic mound affected the local groundwater flow movement and resulted in its reversal. Hydrogeological studies carried out since the closure of the site in 1989 indicate the elimination of this hydraulic mound and the local groundwater flow direction following the regional one (Bartlett and Dottridge, 1996).

A substantial amount of water quality information data exists from the boreholes distributed around the site with clear indications of contamination of both the top and middle aquifers in the eastern part of the site. Water samples have been taken on a quarterly basis for a number of years by the landfill operators and have been analysed for both organic contamination parameters (COD, TOC) and inorganic parameters (chlorides, sulphates, ammoniacal nitrogen (NH₃–N)) as well as heavy metals and

<table>
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<th>Survey line</th>
<th>Date</th>
<th>System used</th>
<th>Parameter measured</th>
<th>Survey characteristics</th>
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<td>Dipole–dipole array with a = 5 m; length of line L1 = 140 m; length of line L2 = 140 m; length of line L3 = 110 m</td>
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<td>W1</td>
<td>March 1998</td>
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<td>Apparent resistivity only</td>
<td>Wenner array smallest electrode separation = 4.0 m; total length of line = 320.0 m</td>
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<td>A, B and C</td>
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<td>Soundings S1 and S2</td>
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electrical conductivity. The borehole log information together with the water level and fluid conductivity measurements at these locations are used in the interpretation of the geophysical results. An example of the concentration values of the most important parameters measured at the different monitoring points in August 1997 is given in Table 1a. As can be seen high concentrations of chlorides, sulphates, $\text{NH}_3$–$\text{N}$ and sodium ions as well as of COD and TOC are recorded at boreholes MP32, MP43 and MP44 being indicative of contamination in both the top and middle aquifers. Similar contamination levels are observed in April 1998 as shown

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<th>Survey date</th>
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<th>Water level (m)</th>
<th>Geological unit where monitoring point is placed in.</th>
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<td>March 1998</td>
<td>W1</td>
<td>6</td>
<td>18.3</td>
<td>Corton sands (1st aquifer)</td>
</tr>
<tr>
<td>March 1998</td>
<td>W1</td>
<td>7</td>
<td>18.3</td>
<td>Crag clay (3rd aquifer)</td>
</tr>
<tr>
<td>March 1998,</td>
<td>W1 and A</td>
<td>43</td>
<td>18.4</td>
<td>Corton sands and Cromerian till (1st and 2nd aquifers)</td>
</tr>
<tr>
<td>May 1998</td>
<td>W1 and A</td>
<td>44</td>
<td>18.4</td>
<td>Cromerian sands (2nd aquifer)</td>
</tr>
<tr>
<td>March 1998,</td>
<td>W1 and A</td>
<td>45</td>
<td>18.4</td>
<td>Crag sands (3rd aquifer)</td>
</tr>
<tr>
<td>May 1998</td>
<td>C</td>
<td>37(30)</td>
<td>19.2</td>
<td>Corton sands (1st aquifer)</td>
</tr>
<tr>
<td>May 1998</td>
<td>C</td>
<td>37(50)</td>
<td>19.2</td>
<td>Cromerian sands (2nd aquifer)</td>
</tr>
<tr>
<td>May 1998</td>
<td>Sounding S1</td>
<td>32</td>
<td>18.2</td>
<td>Corton and Cromerian sands (1st and 2nd aquifers)</td>
</tr>
<tr>
<td>May 1998</td>
<td>Sounding S2</td>
<td>43 and 44</td>
<td>18.4</td>
<td>Corton and Cromerian sands (1st and 2nd aquifers)</td>
</tr>
</tbody>
</table>
in Table 1b, with the exception of MP32. This, according to site operators, is most likely due to high dilution effects in an unreliable borehole. Borehole MP44 also shows a considerable contrast between the two surveys dates.

3. Geophysical data acquisition

Geophysical lines were set-up as close to monitoring boreholes as possible within the logistical constraints. Both profile and sounding data were collected at the site over a period of 2 years (1997–1998). As groundwater is known to be moving towards the east our investigation was concentrated east of the landfill with the exception of the control line L3 (see Fig. 1). Fig. 4 shows the landfill in relation to the investigations lines and the sounding centres as well as the boreholes of interest. Hedges, houses, and roads are also shown in order to emphasise the obstacles encountered during the surveys which restricted the lengths of some of our survey lines, thus, limiting the depth of investigation. Line B was right on the top of the landfill where a flat region existed allowing a maximum length of 150 m. Schlumberger sounding measurements were taken at only two centres. All profile data were collected using the dipole–dipole configuration with the exception of the W1 line, where the Wenner array was used. The surveys were carried out with a variety of field equipment which had an effect on both the quantity and quality of the data collected. The length of the lines surveyed, the data acquisition systems used and the parameters measured are summarised in Table 2 while Table 3 shows the geophysical lines in relation to their nearest monitoring points. The water levels measured at these points are also shown.

4. Data analysis

The apparent resistivity and chargeability measurements, with the exception of profile W1, were inverted using the two dimensional commercial software DCIP2D — a resistivity and IP inversion software based on the work by Oldenburg et al. (1993) and Oldenburg and Li (1994). The Wenner resistivity data from profile W1 were inverted using the RES2DINV software (Loke and Barker, 1996). The inversion approaches are explained in detail in the DCIP2D and RES2DINV manuals and in the above mentioned papers and therefore the complexities of inversion algorithms will not be discussed here. It suffices to note that the goal of a resistivity/chargeability inversion algorithm is to recover a physically realistic set of model parameters that adequately reproduces the given set of field observations. To overcome the problem of non-uniqueness, an objective function is defined which when minimised produces a model that follows certain characteristics as specified by the user as well as resulting in a model response that has a good fit to the measured data. For example, in the DCIP2D software, the user can specify how smooth the final model should be in a given direction by choosing appropriate values of the smoothing parameters and can specify how close to a given reference model the final model should be. The fit between the measured data and the model response is assessed through the chi-square misfit function.

5. Interpretation of results

In this section, we first look at the results from the landfill itself in order to characterise the geoelectric response of the different types of fill. We then investigate the spread of contamination by comparing the response of the ground east of the landfill against that of the uncontaminated land to the west, obtained from the control line L3. Profiles W1, A and C and the two Schlumberger soundings penetrate down to the middle aquifer.
5.1. Landfill characterisation

The inversion results along line B (Fig. 4) are shown in Figs. 5 and 6, respectively. In the final resistivity inversion model, three different zones can be distinguished. The low resistivities within the landfill in Zone 1 are believed to be associated with both the liquid and solid waste. These are also consistent with the high concentrations of inorganic material recorded in the boreholes within the site (e.g., borehole MP38). Zone 2 has lower resistivities between 11 and 23 m depth and even lower values at depths below 23 m. One possible explanation for the lower values is the natural clay base that exists below some of the lagoons. Alternatively, they could be attributed to the accumulation of leachate and the contamination of the saturated sand, especially since lagoon A and the trenches were excavated into the underlying sands. The latter explanation is favoured as the water table is recorded at ~ 23.5 m from the top of the landfill. Zone 3 corresponds to the northern part of the landfill, most likely representing the ~ 14 m thick unconsolidated solid waste which was deposited in the last 2 years of operation of the landfill. In this zone, we observe similar resistivities as in Zone 1. From the schematic of the landfill (Fig. 3), a thin sloping clay layer exists on the northern side at ~ 14 m depth from the
top of the landfill, which is intended to drain any formed leachate into the southern part. This would explain the marked resistivity contrast between the north and south sections with the higher resistivities observed in Zone 4 than in Zone 2 in the saturated sections (i.e., below the water table).

The chargeability lateral variations along the profile are small. However, throughout the section the values are high, indicating the presence of the landfill, and ranging between 80 and 120 ms. Zones 1 and 3 show chargeabilities in the lower end of the range while Zones 1 and 4 are in the upper end. The higher chargeabilities in Zones 1 and 4 may be due to higher concentrations of liquid waste and/or an increase in metallic waste content. In particular, the increased chargeabilities in Zone 4 correlate with the saturated zone where, as the resistivity results suggest, contamination due to the accumulation of leachate is believed to be occurring.

Combining resistivity and chargeability contrasts, we may conclude that the following.

(i) It was possible to distinguish between the upper unsaturated and the lower saturated fill and hence the contaminated water table at ∼23 m below the top of the landfill in the southern part of the site. Substantial lowering of resistivities and increase of chargeabilities are observed at this depth which is believed to be due to the contamination of groundwater by the accumulation of leachate.

(ii) Although low resistivities are also observed in the northern part of the profile there does not seem to be a clear contrast at the expected water level; two possible reasons for this, affecting the total amount of leachate available in the northern part of the site, may be (a) the fact that only solid waste is deposited in the northern part and therefore less leachate is produced (b) the presence of the inclined clay base draining any leachate from the north to the south.

(iii) Due to the high concentrations of leachate, it was not possible to identify the inclined clay base in the northern part of the site nor the clay base of some of the lagoons in the southern part.

5.2. The control line L3

A survey (L3) was carried out well away from the site and to the west of it to act as a control. At this location, we expected no contamination as groundwater is moving eastwards (down-gradient). The nearest borehole to the up-gradient survey profile L3 was borehole MP26, shown in Fig. 1. According to the borehole log, the boulder clay is very thin at this location (∼0.5 m thick) compared to the other locations east of the landfill. Thus, our geophysical measurements are representative of the first sand aquifer without the conductive overburden. The water table recorded at borehole MP26 is around 11.3 m, while the measured fluid conductivity has the low value of 721 μS/cm which is considered as the background, uncontaminated value.

The resistivity model obtained for this profile is shown in Fig. 7. From the results, three zones were identified. Zone 1 represents the unsaturated zone above the recorded water table and it is a highly resistive zone with resistivity values exceeding 550 Ω m. For the first ∼70 m of the profile, Zone 2 corresponds to the saturated aquifer at depths below the water table (11.3 m) and its lower resistivities in the range of 150–550 Ω m reflect this. Zone 3, between 70 and 110 m along the profile, corresponds to the resistive zone that extends below the expected water table. This may be due to lower porosities in this part of the profile, which would imply lower fluid content and therefore increased bulk resistivities.

5.3. Profiles L1 and L2 — the unsaturated zone

The first two profiles to be considered east of the landfill are profiles L1 and L2, surveyed in August 1997. The boreholes associated with them are boreholes MP32 and MP14/15/16/17, respectively (Figs. 1 and 4). From the
data shown in Table 3, the average water level measured at these boreholes was \(~18.0\) m. The lengths of L1 and L2 were limited to 150 m due to restrictions in space thus allowing a maximum depth of investigation of \(~20\) m. Considering lines L1 and L2, we could therefore only assess geophysically the unsaturated zone down to \(~18\) m depth. It was not possible to draw any conclusions in relation to the contamination in the saturated zone, as data in this zone were very limited (only two to three meters of information). The final inversion models for L1 and L2 are shown in Figs. 8 and 9, respectively.

For profile L1, the three resistivity zones identified are as shown in Fig. 8. The presence of the conductive overburden identified as Zone 1 seems to be in agreement with the existing borehole data, which indicate a thick layer of boulder clay down to depths of \(~7.0\) m. The resistivities in Zone 2 are much lower than the equivalent unsaturated zone in the control line L3; this may be due to the movement of contaminants or equally likely a residual effect of the subsiding hydraulic mound; another possible cause may be the variations in porosity, although the contrasts in values with the control
5.4. Identification of contamination in the saturated zone

5.4.1. Profile W1

As no decisive conclusions on the contamination in the saturated aquifers could be made from the results of profiles L1 and L2, due to their limited lengths, further measurements were needed. Additional data were taken along a much longer line W1, which lay between L1 and L2. The boreholes surrounding this profile are boreholes MP14/15/16/17 (eastwards),
borehole MP6/7 (in the north) and borehole MP44 (west). The automated ABEM SAS400 instrument was used in this case with the Wenner array. The length of the line was 320 m, thus, allowing a greater lateral coverage for depths up to 50 m. Thus, information on both the top aquifer (down to ~20 m depth) and middle aquifer (down to ~28 m depth) could be obtained.

The measurements as well as the inversion results for this profile are shown in Fig. 10. Looking at the resistivity inversion model along the whole profile, four resistivity zones could be distinguished. Zone 1 represents the conductive overburden (<75 \( \Omega \) m) for the whole length of the profile, in agreement with borehole information which indicates a thick boulder clay top layer. Zone 2 is a more resistive zone with resistivities greater than 200 \( \Omega \) m while Zones 3 and 4 are more conductive (<200 \( \Omega \) m).

Bearing in mind that the regional groundwater flow is eastwards, the lower resistivities found in Zones 3 and 4, for depths greater than ~17 m, may be indicative of an eastward movement of contamination from the landfill. Although the information from line L2 (Fig. 9) was depth limited to ~20 m, the lower resistivities in its Zone 3 (Fig. 9) seems to be consistent with the lower resistivities found in Zone 3 of the profile W1 (Fig. 10).

Two more profiles, profiles A and C, were carried out in order to investigate the southward movement of any contamination as the water samples from boreholes MP43 and MP44 indicated high fluid conductivities (Table 1a and b). Profile A will be discussed first.

5.4.2. Profile A

Both resistivity and chargeability measurements were taken along this profile whose total length was 240 m, thus, allowing a depth of investigation down to ~30 m covering both the top and middle aquifers. According to the borehole information along this profile (boreholes MP43 and MP44), the first aquitard is non-existent and the two aquifers in fact form one unit. The resistivity and chargeability inversion models for line A are shown in Fig. 11a and b, respectively.

5.4.2.1. Resistivity model. Four zones are identified in the resistivity model, with Zone 1 representing the conductive overburden and the unsaturated zone down to the depth of 17 m. Zone 2 is a resistive zone, while Zone 3 is a lower resistivity region which is thought to be associated with the contamination plume. Zone 4 is a highly resistive section between 160 and 200 m with resistivities greater than 550 \( \Omega \) m.

5.4.2.2. Chargeability model. Three chargeability zones are identified in the chargeability model. Zone 1 is a low chargeability zone along the whole length of the profile with chargeabilities <5 ms, while Zone 2 has higher chargeability values in the western part of the profile, between 80 and 120 m, and for depths greater than 17 m. These higher chargeability values correspond to the lower resistivity Zone 3 of the resistivity model and could be an indication of contamination. Zone 3 is a low chargeability region associated with the high resistivity Zone 4 of the resistivity model (Fig. 11a).

Comparing (i) the fluid conductivity values in the boreholes close to line A, (ii) the bulk resistivity values obtained from the resistivity inversion and (iii) the chargeability model results, we observe that although the fluid resistivities are very low in the first and second aquifers (\( \rho_w \sim 0.3 \) and 0.65 \( \Omega \) m, respectively; Table 1b) the absolute bulk resistivities from the inversions do not seem to reflect this. The bulk resistivities in Zone 3 of the resistivity model, are in fact lower than in the adjacent Zones 2 and 4, suggesting contamination. However, with such low fluid resistivities, one would expect that the corresponding bulk resistivities would be much lower than what we obtained. One possible explanation for this apparent ‘inconsistency’ could be the presence of oily/organic waste in this fine/silty/clayey sand aquifers. This may also explain the region in the charge-


Fig. 10. Measured and predicted data together with the resistivity inversion model for profile W1.
Fig. 11. (a) Resistivity inversion model for profile A. (b) Chargeability inversion model for profile A.

ability model (Zone 2) where high chargeabilities exist (up to 20 ms) which cannot be due to the presence of clay particles as the chargeability values due to clay are lower (<10 ms). These higher chargeabilities in the western part of the profile may therefore be related to the presence of the organic waste detected in the boreholes.

A second possible explanation of the low fluid resistivities and high bulk resistivities may be related to the pore geometry itself as discussed by Herrick and Kennedy (1994). According to these authors, in a given pore system, pore throats exhibit high current densities while the nearly stagnant volumes in isolated parts of the pore system exhibit low current densities. This non-uniform current density distribution caused by the pore geometry may affect the overall electrical efficiency of the rock.

5.4.3. Profile C

In order to establish the southern boundary of a contamination ‘plume’, we also looked at both the resistivity and chargeability inversion results along line C, shown in Fig. 12a and b, respectively. The closest boreholes to this profile are MP37 in the NW end of the profile, MP43 in the central region, and MP4/5 in the SE end of the profile. All three boreholes show fluid electrical conductivities above the background, uncontaminated value (Table 1a and b). The resistivity inversion model is shown in Fig. 12a where we can distinguish four zones.

5.4.3.1. Resistivity model. Although there is a lateral variation of resistivities along the profile, it is only in the very early part (for the length of the profile < 50 m) that the resistivities are lower than 300 Ω m, for depths greater than 17
This is represented by Zone 3 in Fig. 12a and the resistivities observed are similar in values to the ones found in Zone 3 of profile A (Fig. 11a). These resistivities could be an indication of the presence of contamination and in particular of inorganic waste. However, as our line is ~200 m away from the borehole MP4/5, it is yet unclear as to whether this is the case. Also, the low resistivities in Zone 3 of profile A were associated with high chargeabilities. There is no such correlation between resistivity and chargeability values for the resistivity Zone 3 of profile C (Fig. 12a and b).

For the rest of the profile and at depths between 17 and 30 m, the resistivities are greater than 300 Ω m. The thick highly resistive zone in the latter part of the profile (> 140 m) for the depths between 17 and 33 m has resistivities greater than 500 Ω m while as we move towards the centre of the profile there is a 'thinning' of this resistive zone with resistivities ranging between 300 and 500 Ω m.

5.4.3.2. Chargeability model. The chargeability model corresponding to this profile (Fig. 12b) shows a layered structure with chargeabilities increasing uniformly with depth. Such a uniformly layered chargeability inversion model would appear to suggest that at least no organic waste has reached this part. The low chargeabilities (< 5 ms) at the top 7 or 8 m seem to correspond to the Boulder clay layer found in
the boreholes east of the landfill, while the high chargeabilities at ~ 30 m depth may be related to the semi-pervious clay layer, also found in the surrounding boreholes. The observed differences in chargeabilities between the two layers of clay may be diagnostic of different compositions.

Comparing the resistivity and chargeability results, it is unclear as to whether the lateral variations in resistivity are associated with any contamination. The chargeability model showed no lateral variations, thus, no link to inorganic or organic contamination could be made as was the case for line A.

Following the above individual description of the results, the output from four ‘parallel’ lines has been displayed in Fig. 13. This shows the variation of resistivities east of the control line L3, and leads us to suggest that the movement of contamination is not only towards the east, but also towards the SE direction.

5.5. Schlumberger soundings

In addition to the 2D measurements using the dipole–dipole and Wenner arrays, we also carried out Schlumberger soundings at two locations, S1 near borehole MP32 and S2 near borehole MP43. Resistivity data were collected to see whether we could resolve the parameters of the different layers, especially their resistivities. Of special interest was the resolution of the clay layer identified in the borehole close to sounding S1 (borehole MP32). The apparent resistivity data were inverted using the 1D inversion program of Sandberg (1990). During the inversion procedure, the borehole log information enabled us to constrain the thickness of the layers.

For sounding S1, the water sampling information from borehole MP32 (Table 1a) showed high levels of contamination within both the top and middle aquifers. The apparent resistivity
curve seen in Fig. 14 seems more suited to a
two-layer case than a five-layer case as indicated
by the borehole log. However, the starting
model for the inversion represented the layered
structure according to the borehole log with
three layers over a half space where the half-
space corresponded to the middle aquifer. Dur-
ing the inversion, all depths were constrained.
The results are shown in Fig. 14 together with
the final models, as well as the log of borehole
MP32. The comparison between model re-
sponse and real measurements show a very
good fit to the data and the parameter values
obtained are physically realistic. However, the
parameter resolution information (see Sandberg,
1990) associated with the final model showed
the resistivity of the third layer, corresponding
to the thin clay layer, was not resolved.

At the second location, the clay layer is
non-existent according to the borehole log for
MP43. The apparent resistivity curve at this
centre also resembles more of a two-layer case.
The starting model used during the inversion
consisted of three layers over a half space and
all depths were constrained again according to
the borehole information. The inversion results
together with the final inversion model are
shown in Fig. 15. Again, although a good fit to
the data was obtained, the resistivities of the
third layer and of the half-space were not re-
solved.

It is important to note that in isolated
Schlumberger soundings only 1D inversion is
possible, whereas the subsurface distribution of
resistivities, contributing to the measured appar-
ent resistivities may be 2D or 3D. It is therefore
not surprising that the model parameters cannot
be resolved.

In these 1D inversions, we also observe the
relatively high resistivities obtained for the lay-
ers representing the aquifers, especially in
sounding S2. Despite the fact that the fluid
conductivity and chloride levels are very high at
these locations, the bulk resistivity is very high
for sounding S2 (862 Ω m). The high resistivity
values obtained in regions where the fluid resis-

\begin{figure}[h]
\centering
\includegraphics[width=\textwidth]{sounding_s1}
\caption{Schlumberger sounding S1 — measurements, predictions and resistivity model.}
\end{figure}
tivity is low (high conductivity) is not what is expected and as explained earlier these high bulk resistivities could be associated with the presence of organic contamination parameters (high concentrations of COD and TOC indicated by borehole MP43) or alternatively with the low porosity and pore geometry. Additionally, it needs to be borne in mind that the distortions introduced by treating a 3D problem with 1D or 2D modelling procedures can be quite significant and may produce a bias towards the high resistivity values.

6. Determination of aquifer properties

In order to obtain quantitative answers in groundwater flow and contaminant transport modelling, it is essential to have estimates for the hydraulic properties of any given aquifer system such as the hydraulic permeability and conductivity. Such values are usually obtained either from pumping tests or laboratory experiments when core samples exist. However, an alternative approach can be applied, which utilises non-invasive geophysical information. Geophysicists have realised for some time that a correlation between the hydraulic and electrical properties of an aquifer should be possible, as both these properties relate to the pore space structure and heterogeneity (Kelly, 1977; Mazac et al., 1985; Huntley, 1986; Mazac et al., 1988; Boerner et al., 1996; Christensen and Sorensen, 1998). The diversity of the relationships reported indicates the complexity of the problem, with all authors emphasising the fact that these relationships are specific to the site under investigation.

Since we had bulk and fluid resistivities at several locations, it was desirable to see if it was possible to obtain hydraulic conductivity values using the Kozeny–Carman–Bear equation (Domenico and Schwartz, 1990). The porosity $\phi$ required in this equation had to be estimated using Archie’s law, with its empirical nature and dependence on Archie’s parameters. We have investigated the dependence of $\phi$ on these parameters as well as calculated the range of hydraulic conductivities produced by the plausible values of these porosities.
6.1. Determination of porosity through the intrinsic formation factor $F_i$

Archie’s law relates the bulk resistivity of a fully saturated granular medium to its porosity and the resistivity of the fluid within the pores according to Eq. (1).

$$\rho_a = a \rho_w \phi^{-m}$$

(1)

where $\rho_a$ is the bulk resistivity, $\rho_w$ is the fluid resistivity, $\phi$ is the porosity of the medium and $m$ is known as the cementation factor, although it is also interpreted as grain-shape or pore-shape factor; the coefficient $a$ is associated with the medium and its value in many cases departs from the commonly assumed value of one. For a clay-free, clean medium, the ratio $\rho_a/\rho_w$ is known as the intrinsic formation factor $F_i$.

The values of the coefficients $a$ and $m$ should, ideally, be determined for each site under investigation. However, due to lack of core samples in our test area, this was not possible and an alternative approach was adopted whereby a wide range of values for $a$ and $m$ reported in the literature was used to obtain porosity estimates. Worthington (1993) reports three different expressions for the intrinsic formation factor in relation to the porosity associated with samples from different locations. We use these three relationships to estimate the porosity, as well as a fourth expression in which the coefficient $a$ has the value of one while $m$ is allowed to vary between 1.3 and 2.5 (Jackson et al., 1978; de Lima and Sharma, 1990).

The first step in our procedure to estimate the hydraulic conductivity is the determination of the intrinsic formation factor $F_i$ from which porosity can be estimated via Eq. (1) for different values of $a$ and $m$. However, for field data a complication arises due to the fact that Archie’s formula (Eq. (1)) is valid only for clay-free, clean, consolidated sediments. Any deviations from these assumptions make the equation invalid as discussed by Worthington (1993). In the case of unclean, clayey sands the ratio of bulk resistivity to fluid resistivity is known as the apparent formation factor $F_a$. As our aquifer system consisted of clayey/silty sand material, a modified representation of the Archie’s equation was required. For this reason, the Waxman-Smits model was considered (Vinegar and Waxman, 1984) which relates the apparent and intrinsic formation factors $F_a$ and $F_i$ after taking into account shale effects. According to Worthington (1993),

$$F_a = F_i (1 + BQ_v \rho_w)^{-1}$$

(2)

where the $BQ_v$ term relates to the effects of surface conduction, mainly due to clay particles. When the surface conduction effects are non-existent, then the apparent formation factor is equal to the intrinsic one, i.e., $F_a = F_i$.

Re-arranging the terms of Eq. (2), we obtain a linear relationship between $1/F_a$ and $\rho_w$,

$$1/F_a = 1/F_i + (BQ_v/F_i)\rho_w$$

(3)

where $1/F_i$ is the intercept of the straight line and $BQ_v/F_i$ represents the gradient (Huntley, 1986; Worthington, 1993). Thus, by plotting $1/F_a$ vs. fluid resistivity $\rho_w$, we should, in principle, obtain a value for the intrinsic formation factor, which will subsequently enable us to estimate porosity.

To follow the above approach, we used bulk resistivities $\rho_a$ from our resistivity inversion results together with the fluid electrical resistivities $\rho_w$ at the nearest boreholes, to obtain values of the apparent formation factor $F_a = \rho_a/\rho_w$ for the saturated top aquifer. From Eq. (3), it is clear that a major source of error would be the wrong estimation of the apparent formation factor which depends on the bulk resistivity as obtained from the inversion models. Thus, assuming the fluid resistivities were measured correctly in the laboratory, we considered by how much the bulk resistivities from the inversions could vary. This uncertainty was mainly due to the fact that some of the boreholes (e.g.,
Table 4
Estimating apparent formation factor from the geophysical data top aquifer

<table>
<thead>
<tr>
<th>MP</th>
<th>Associated line</th>
<th>Depth (m)</th>
<th>Fluid EC (µS/cm)</th>
<th>$\rho_w$ (Ω m)</th>
<th>$\rho_o$ (Ω m)</th>
<th>$F_a$</th>
<th>$1/F_a$</th>
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<tbody>
<tr>
<td>6</td>
<td>W1</td>
<td>19.3</td>
<td>10,370 (April 1998)</td>
<td>0.96</td>
<td>89.1</td>
<td>92.8</td>
<td>0.0108</td>
</tr>
<tr>
<td>6</td>
<td>W1</td>
<td>19.3</td>
<td>10,370 (April 1998)</td>
<td>0.96</td>
<td>129.7</td>
<td>135.1</td>
<td>0.0074</td>
</tr>
<tr>
<td>37</td>
<td>C</td>
<td>20.1</td>
<td>3875 (April 1998)</td>
<td>3.38</td>
<td>198</td>
<td>58.6</td>
<td>0.017</td>
</tr>
<tr>
<td>37</td>
<td>C</td>
<td>20.1</td>
<td>3875 (April 1998)</td>
<td>3.38</td>
<td>296</td>
<td>87.6</td>
<td>0.011</td>
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<tr>
<td>26</td>
<td>L3</td>
<td>12.3</td>
<td>721 (August 1997)</td>
<td>13.8</td>
<td>432</td>
<td>31.30</td>
<td>0.0319</td>
</tr>
<tr>
<td>26</td>
<td>L3</td>
<td>12.3</td>
<td>721 (August 1997)</td>
<td>13.8</td>
<td>530</td>
<td>38.41</td>
<td>0.0260</td>
</tr>
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</table>

borehole MP37, line C, were at some distance away from the region represented in the resistivity model. To obtain a measure of the uncertainty, we considered average bulk resistivities over stretches of the survey lines, thus, enabling estimates of the upper and lower limits of the bulk resistivity value $\rho_w$ for the lines L3, C and W1.

The resistivity values obtained from the inversions, after averaging to allow for lateral variations along each profile, and the estimated apparent formation factors are shown in Table 4. Fig. 16 shows $1/F_a$ plotted against the fluid resistivity $\rho_w$ and the best fit straight line gives the inverse of the intrinsic formation factor $F_i$ which for our data set was 0.0083. The porosities could now be determined through Eq. (1) for reported values of $a$ and $m$ for sands as shown in Table 5.

6.2. Determination of hydraulic conductivity

The estimation of the hydraulic conductivity was achieved through the use of the Kozeny–Carman–Bear equation, given by Domenico and Schwartz (1990) as:

$$K = (\delta_w g / \mu)(d^2 / 180)[\phi^4 / (1 − \phi^2)]$$

where $d$ is the grain size, $\delta_w$ is the fluid density (taken to be 1000 kg/m$^3$), and $\mu$ is the dy-

<table>
<thead>
<tr>
<th>$a$</th>
<th>$m$</th>
<th>Porosity</th>
<th>Hydraulic conductivity (m/s)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1.04</td>
<td>2.3</td>
<td>0.127</td>
<td>1.28E−05</td>
</tr>
<tr>
<td>0.5</td>
<td>2.31</td>
<td>0.093</td>
<td>5.05E−06</td>
</tr>
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Fig. 16. $1/F_a$ vs. fluid resistivity $\rho_w$. 

Table 5
Sensitivity analysis on porosity and hydraulic conductivity estimates

Porosity estimates using different Archie’s coefficients ($1/F_a = 0.0083$), together with the sensitivity analysis investigating the effect of porosity on hydraulic conductivity. Intrinsic formation factor $F_i = 0.0083$. (Grain size = 0.4 mm.)
namic viscosity taken to be 0.0014 kg/ms (Fetter, 1994). The reported values for the grain size range between 0.2 and 0.6 mm (Richardson, 1996). In our estimations, we assumed at this stage a mean particle size of 0.4 mm consistent with exposures on the coastal section. The hydraulic conductivity values using Eq. (4) are shown in Table 5.

The reported geometrical mean hydraulic conductivity value for this aquifer is $9.3 \times 10^2$ m/s while the corrected geometrical mean value becomes $4.4 \times 10^2$ m/s (Richardson, 1996). Looking at Table 5, we can see that the low porosity values within the range of 0.05–0.10 (based on $a = 1$ and $m$ ranging between 1.5 and 2.1) result in hydraulic conductivity values of the same order of magnitude as the ones reported by Richardson (1996).

In comparing these values, it is important to bear in mind that the physical measurements are made in small samples and a limited zone around the borehole, whereas the geophysical methods use properties averaged over large volumes.

7. Summary and conclusions

By applying the geo-electrical methods, we have been able to characterise the landfill in terms of resistivity and chargeability and distinguish between the southern and northern sections, which received different types of waste. A more contaminated saturated zone in the southern part could be distinguished, which is believed to be due to the accumulation of leachate, mainly from the deposited liquid waste. Using the chargeability inversion results we were also able to distinguish the zone below the water table within the landfill.

In addition, we were able to follow the eastwards movement of contamination by comparing the resistivity results from lines L1, L2 and W1 to the control line L3. Resistivity contrasts are observed in lines L1 and W1, suggesting inorganic contamination as detected in the nearby boreholes. Lateral resistivity contrasts along line L2 are less obvious, with only a small central section shown to be affected by contamination.

Lines A and C were investigated in the attempt to establish the southern boundary of the contamination. Both resistivity and chargeability results were available on these lines. Line A shows resistivity contrasts that may be related to contamination, while with profile C the effects of contamination are not obvious. On profile A, there is a marked difference in the chargeability values within the saturated zone, with the increased chargeability values in the western part of the profile suggesting the presence of organic contamination. On the contrary, the results for profile C show no lateral variations, but rather a uniformly layered chargeability model with chargeabilities increasing with depth up to the second clay aquitard at ~28 m depth. From the results of these two profiles, it is clear that IP information helped to distinguish more clearly regions affected by contamination.

An intriguing general observation is that the bulk resistivities obtained from the inversions (whether 2D or 1D) are quite high (200–300 $\Omega\cdot$m). This happens in regions where contamination is known to exist from the low fluid resistivity measurements in the nearby boreholes (~0.3 $\Omega\cdot$m). With such low fluid resistivities one would expect the bulk resistivities also to be low (<50 $\Omega\cdot$m). One possible explanation for the unexpectedly high bulk resistivities may be the presence of organic pollutants since borehole fluids show high concentrations of parameters such as COD and TOC, in addition to the inorganic pollutants. The precise mechanism for this is not clear however and further study is needed; indeed lowering of resistivities by bacterial action on organics has also been reported in the literature (e.g., Benson et al., 1991). An alternative explanation for the high bulk resistivities is low porosity values and the pore geometry itself, as discussed by Herrick and Kennedy (1994), which would affect the overall
electrical efficiency of the system. An additional factor to be taken into account in future studies of this area may be the distortion effects of a 3D problem on the 1D and 2D solutions. To eliminate these effects, a 3D data set would be required, which would inevitably increase the survey time and costs.

Using the bulk resistivities from the inversions, the measured fluid resistivities and the intrinsic formation factor derived from them, an attempt was made to estimate a range of porosity values for the top aquifer, using reported values for the parameters in the empirical Archie’s formula. These porosities were subsequently used to estimate the hydraulic conductivity through the Kozeny–Carman–Bear equation. A range of porosity values was established for which the hydraulic conductivity estimates cluster well and fall within an order of magnitude of values reported from other sources. Given the nature and extent of uncertainties, it is encouraging that this degree of agreement exists between the geophysically based values and the other methods, and it is reasonable to conclude that geophysical estimates, at the very least, provide a useful additional constraint on the hydraulic conductivity values.

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