

Groundwater contamination at Trandum landfill, Southeastern Norway

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The influence of a sanitary landfill on groundwater within the surrounding glaciofluvial deposits has been investigated at Trandum, in the vicinity of Gardermoen international airport, 40 km northeast of Oslo. These glaciofluvial deposits form the largest discrete aquifer in Norway with a volume of approximately $2.75 \times 10^9 \text{ m}^3$ (based on an area of 55 km^2). The average velocity of groundwater flow near the landfill is calculated as 1.5 m/day, whereas earlier investigations have reported 0.04-0.08 m/day in adjacent areas.

The pH of the groundwater ranges between 5.8 and 8.4, the electrical conductivity between 123 and $1480 \mu\text{S/cm}$, and the redox potential between -30 and +257 mV. Contamination of the groundwater is indicated by elevated values of Ba, B, Cl^- , NH_4^+ , H₂S, Fe and Mn beneath the landfill. Chloride and electrical conductivity are useful parameters indicating the shape of the contaminant plume. The contaminated groundwaters are readily classified in a diagram where pH is plotted against electrical conductivity. The quality of the water pumped for domestic use, including drinking water, from wells located c. 1 km northeast of the landfill is acceptable as far as inorganic constituents are concerned. However, it is recommended that both inorganic and organic parameters are monitored in the future until contamination from the landfill is significantly reduced or has ceased.

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Introduction

Trandum is a military base located in the vicinity of Gardermoen international airport, 40 km northeast of Oslo (Fig. 1). The (mainly) glaciofluvial deposits of the area form the largest discrete aquifer in Norway, the Øvre Romerike aquifer, with a volume of approximately $2.75 \times 10^9 \text{ m}^3$, based on the aquifer's central area of 55 km^2 (Fig. 2). The influence of leachate from Trandum landfill (Fig. 3) on groundwater chemistry in the vicinity of Trandum military base has been investigated. The investigation was undertaken as a result of a national survey of hazardous waste within Norwegian landfills (Misund et al. 1991). The primary data on which this paper is based are reported in detail by Misund and Sæther (1991).

The purpose of the investigation was to evaluate the groundwater chemistry at Trandum military base, and its relation to the local hydrogeology, given that there are several potential pollution threats in the area (Fig. 3):

(a) the main landfill, located immediately west of the base.

(b) a washing bay for cleaning large vehicles, located within the base.

(c) spill water from the washing bay which is drained into a pond at a shooting range located northwest of the base.

(d) a smaller, older landfill located within the base, well above the saturated zone of the aquifer, a short distance west of Transjøen Lake.

Drinking water for the base is pumped from two wells northeast of the base at the perimeter of Transjøen lake.

Description of the locality

The Trandum military base covers about 0.5 km^2 (Fig.3) and lies to the west of Transjøen lake. The predominant vegetation in the area is spruce and aspen. The landfill is located in a dead ice hollow, or kettle hole, west of the base. The landfill was begun in 1954 and is

NORTHERN ROMERIKE QUATERNARY MAP

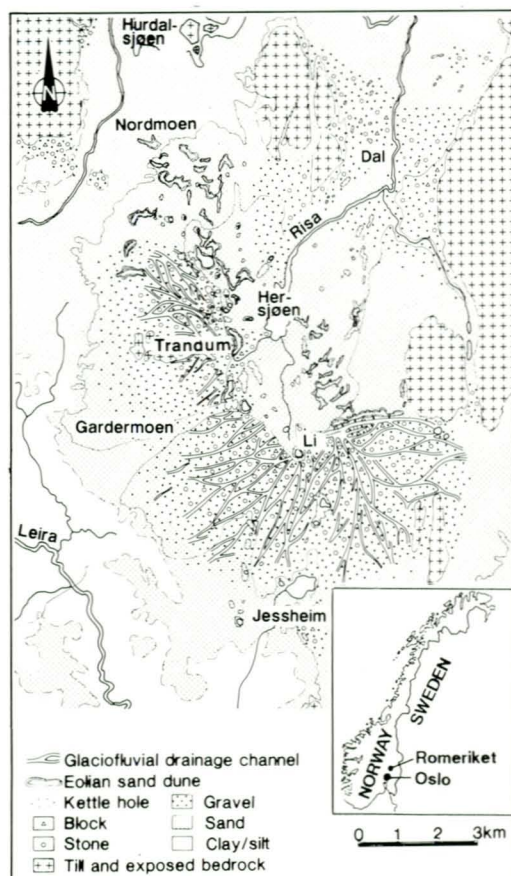


Fig. 1. Map of Pleistocene deposits within the Northern Romerike area (from Jørgensen and Østmo 1990).

now c. 15 m thick. It has been filled to a level of around 195 m above sea level (a.s.l.). The base of the landfill lies just below the water table. Until 1978 it was used as a sanitary landfill for both military and civilian purposes. From 1978 onwards, it has been a site for disposal of solid domestic waste. In addition to domestic waste, there are good reasons to suspect that the landfill contains a wide variety of batteries containing heavy metals (e.g. Pb, Ni, Fe, Hg, Cd, Zn), paints, solvents, used lubricating oil from vehicles etc. Combustible material was incinerated on site.

Monitoring borehole 3 (Fig.3) is located in the centre of the landfill whereas boreholes 4, 21, 22, 23, 24 and 25 are placed in a semi-

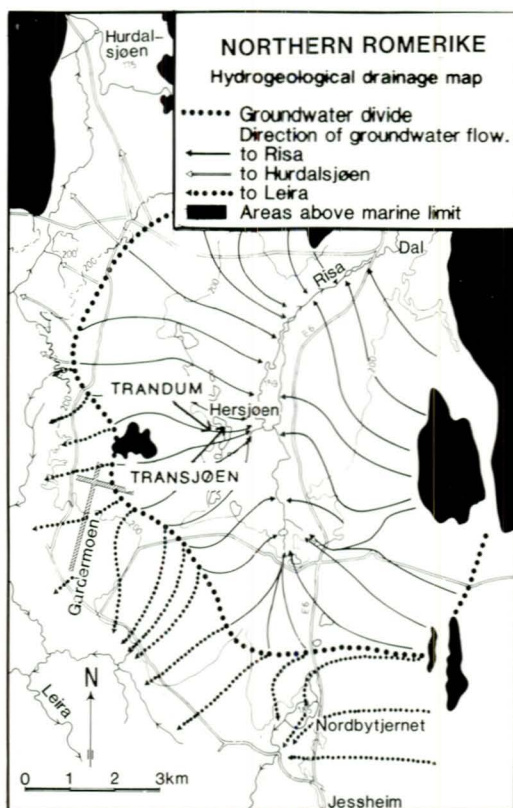


Fig. 2. Simplified groundwater flow map for the northern Romerike area. The location of Trandum is marked (modified from Østmo 1976b and Jørgensen and Østmo 1990).

circle to the east (i.e. downflow) of the landfill. Borehole 26 is located upflow of the investigated area and acts as a background reference.

A shooting range is located in another kettle hole north of the base, and boreholes 32, 34, 35 and 36 are placed here.

The washing bay lies at the northern end of the base, and is used as a site for cleaning (including degreasing) of larger vehicles such as trucks and tanks. Investigatory boreholes 31, 37 and 38 are located here. Water used for cleaning is pumped from two military wells (Fig.3) at the northwestern end of Transjøen lake. Spillwater from the washing bay is drained via a sewer which discharges into a pond in the base of the shooting-range kettle hole adjacent to borehole 32. The water pumped from the military wells is also used for domestic purposes at the Trandum base.

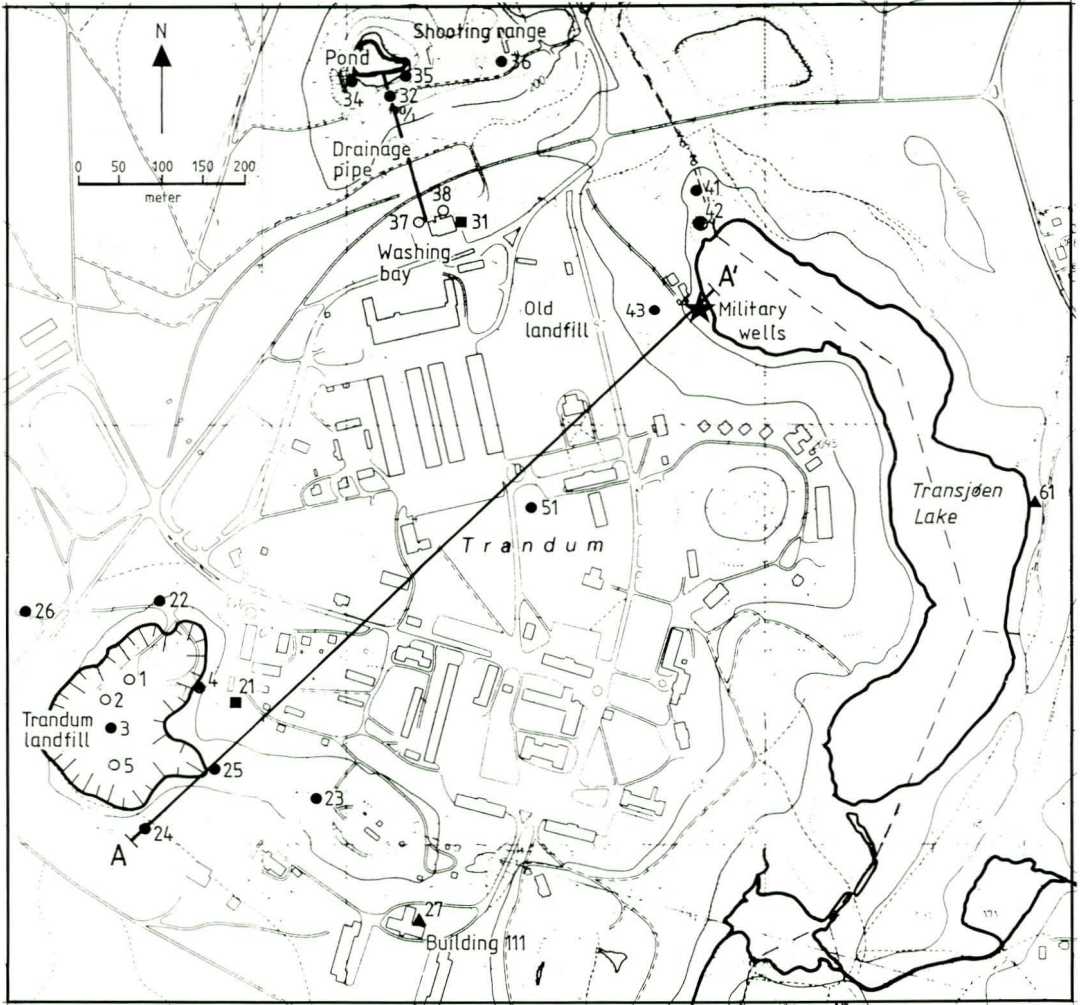


Fig. 3. Map of the investigated area showing the locations of the 18 monitoring boreholes (○ = soil samples only, ● = borehole 2" diam., ■ = borehole 4" diam., ▲ = preexisting boreholes) at Trandum military base as of June 1991.

Geological setting

The genesis of the deposits of the Trandum area is described by Østmo (1975, 1976a) and Longva (1987). They are around 9600–9700 years old (Longva & Thoresen 1989), and are part of an ice-marginal delta with a point-source at Trandum (Hebrand 1990 and Fig. 1). The sediments were deposited as sediment-loaded subglacial streams lost their hydraulic transport capacity on reaching the fjord. In this way, medium to coarse materials were deposited as foreset beds in several discrete depositional bodies up to the marine limit (205 m a.s.l.) closest to the ice-margin. A progradational coarse-grained sandur, with

sub-horizontal bedding, was deposited on top of the delta. The finest material was transported farther away, and is thus dominant at greater distances from the glacier front.

At Trandum, the sandur is built up to 210 m a.s.l. The area's topography is influenced by erosional channels and kettle holes. In the immediate subsurface this sedimentary history has resulted in alternating fine- and coarse-grained beds (Fig. 4). Extensive investigations with georadar (Mauring & Rønning 1991) support the interpretation of the sedimentological structure as described above, but are not discussed further in this paper.



Fig. 4. Sediment texture at the Trandum landfill, in the side-wall of the kettle hole. Note the abrupt changes in grain-size which strongly influence the hydraulic parameters.

Hydrogeology

The hydrogeology of the investigated area is complex due to the heterogeneous sediment texture and structure. A transect A-A' (Figs. 3 and 5), from borehole 24 (at the south rim of the landfill) to borehole 43 (on the northwest side of Transjøen lake) shows a water table with a gradient from 182 m a.s.l. in the SW to 172 m a.s.l. in the NE. The water table is located in strata consisting of fine and coarse sand. West of the landfill site, the coarser, overlying gravel deposits are mostly located above the water table (Fig. 5). In the area of the landfill, the coarser and finer sediment bodies interfinger in both the unsaturated and saturated zone. The contour-lines on the water table strike approximately north-south indicating a groundwater flow to the east (Fig. 6). This coincides with the interpretation of Østmo (1976b).

The annual fluctuation of the water table is about 1 m, while the difference between the lowest and highest levels measured during the period 1967-1991 is 4 m, due to variation in precipitation over a scale of decades (Kirkhusmo & Sønsterud 1988).

Methodology

Sampling design

The locations of 18 monitoring wells and five investigation boreholes from which sediments were sampled are shown in Fig. 3. The well-filters are 3-5 m long, and are placed just below the water table in all wells (i.e. varying from 2 to 35 m below the ground surface). A total of 113 sediment

samples were collected from these and a subset of the samples were analysed for grain-size distribution and sediment chemistry. Groundwater was sampled in 1991 during three periods; weeks no. 22, 26 and 30.

Sampling procedures

The sediment samples were collected during drilling, placed immediately in airtight glass-jars and stored in a refrigerator until analysis.

Prior to groundwater sampling, using a stainless steel submersible pump, a stable redox-potential was achieved by pumping at least five well-volumes from each borehole. Two aliquots of the groundwater sample (100 mL polyethylene bottle and 1000 mL glass bottle) were acidified to $\text{pH} < 2$ with ultrapure nitric acid (14N) and two aliquots (100 mL and 500 mL, both polyethylene bottles) were stored untreated. The 100 mL samples were filtered using a 0.45 μm filter (Kirkegaard 1989).

Field measurements

Temperature, pH, and electrical conductivity were measured in the field. During the second sampling round (week no. 26), the redox-potential was measured using a YSI airtight cell with Orion electrode together with electrodes for pH and temperature.

Laboratory measurements

The sediment samples were extracted with 7N HNO₃ in borosilicate bottles in an autoclave at 120°C for half an hour. This extraction procedure is in accordance with the Norwegian Standard NS4770 (Norsk Standard 1980). The acid extracts were analysed for 29 elements by a Jarell Ash inductively coupled plasma emission spectrometer (Ødegård 1983).

The 100 mL acidified water samples were analysed for 29 elements by inductively coupled plasma emission spectroscopy. Mercury was determined on the one litre acidified samples by cold vapour atomic absorption using a Perkin-Elmer atomic absorption spectrophotometer model 460 and a mercury/hydride system model MHS-20 including a gold trap. Conductivity, pH, and alkalinity were measured both

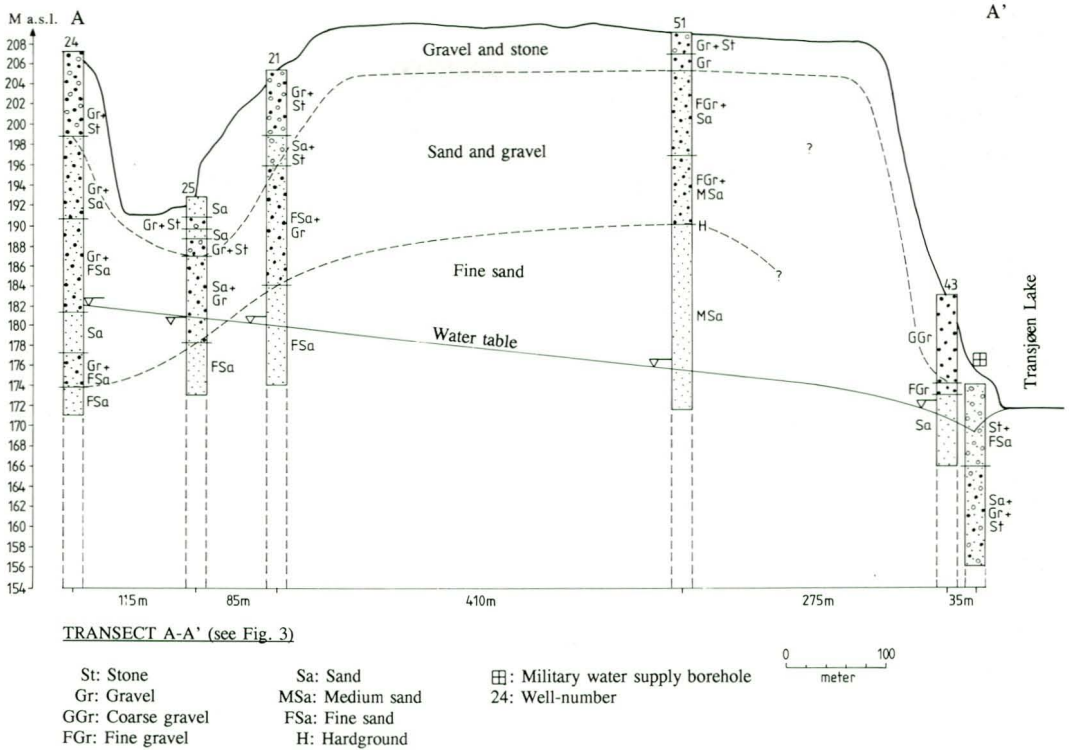


Fig. 5. SW-NE transect showing the sedimentology between Trandum landfill and Transjøen lake.

in the field and in the laboratory on the untreated 500 mL aliquots. Seven anions were analysed on the untreated 100 mL aliquot by HPIC using an ion chromatograph model 2010i from DIONEX. Laboratory measurements were performed mainly according to methods used at the Geological Survey of Norway (Ødegård & Andreassen 1987).

Results

Sediment samples

There is a large variation in the texture of the sediments due to the complex depositional history governed by interfingering deltas and a braided river system. The sediments around borehole 27 were found to be mainly of silt grade below the water table (Storrø 1991, Storrø & Banks 1992). In borehole 23, however, located 180 m northwest of borehole 27, a sediment sample located 4 m below the water table contained 75% gravel. Similarly there are wide variations in grain-size vertically within each borehole, as is illustrated in Figs. 4 and 5. At borehole 61 on the eastern shore of Transjøen Lake the sediments consist of sand and gravel down to at least 45 m below the water table. Many of the sediment samples were found

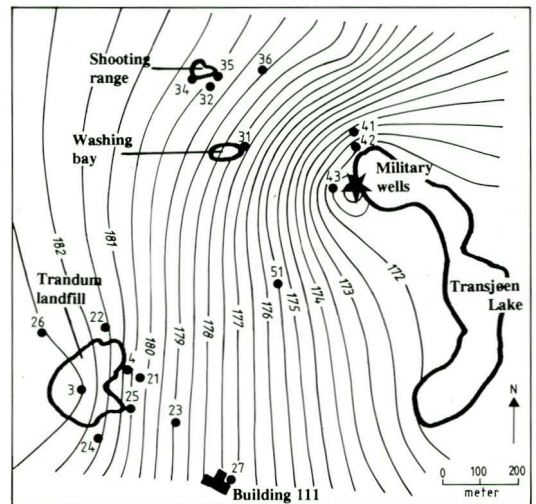


Fig. 6. Map of piezometric head (m a.s.l.) in the investigated area at Trandum military base, for June 1991. Numbers correspond to borehole reference numbers in Fig. 3.

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Many of the sediment samples were found

Table 1. Maximum enrichment factors for metals in sediment samples (relative to background samples from borehole 26). A, B and C refer to the soil criteria used in the Netherlands for contaminated land (Moen et al. 1986). Boreholes are ranked according to decreasing degree of sediment contamination.

Element	Enrichment factor	"ABC"-system	Ranking of borehole
Pb	120 x	>C	Bh 34,5,3,4,35,38,37,32
Zn	15 x	>B	Bh 51,5,34,31,1,4,2,42
Hg	10 x	<A	Bh 5,34,31,43,41,1,4,2
Mn	6 x	-	Bh 4,34,5,23,38,1,32,35
V	3.5 x	-	Bh 5,23,34,2,32,35,38,4
Cu	3 x	>A	Bh 5,34,38,3,4,32,2,1
Fe	2 x	-	Bh 4,2,34,5,23,32,35,38
Ni	1.7 x	<A	Bh 4,5,32,2,51,38,34,43

to contain levels of the heavy metals Pb, Zn, Hg, V, Cu and Ni well above the background values found in borehole 26. Some metals, particularly Pb and Zn, show enrichment factors of 15-120, relative to the background level; e.g. at the shooting range, in the central area of the military base (borehole 51) and below the landfill (borehole 5) (Table 1 and Fig. 3).

Groundwater flow

The hydraulic conductivity, porosity and effective porosity of the aquifer have been estimated from grain-size analyses by a slight modification of the Beyer & Schweiger method as described by Langguth & Voigt (1980). Estimates of the hydraulic parameters, based on sediment samples from below the water table along profile A-A' (Fig. 5) are shown in Table 2. The average hydraulic conductivity is 4×10^{-5} m/s which, with an average effective porosity of 0.29 and a hydraulic gradient of 0.013, implies a groundwater flow-velocity of about 1.5 m/day (1.85×10^{-5} m/s). These results are consis-

tent with values reported by Klemetsrud (1986) and Snekerbakken (1990).

However, the large variation in sediment texture implies a correspondingly large variation in groundwater flow velocity, since a volume of approximately 150 L/s (Klemetsrud 1986) is transmitted through a cross-section of the sub-catchment at any one time. For example, Storrø (1991) deduced a flow velocity of 0.04-0.08 m/day ($5 - 9 \times 10^{-7}$ m/s) at borehole 27. The investigations indicate that a significant part of the contamination in the groundwater at Trandum has its origin at the Trandum landfill, and is transported in the direction of groundwater flow towards the east. There is, however, little evidence of a significant component of southerly spreading of contamination (see Figs.7); for example, there are few traces of contamination found in borehole 27. This may reflect the generally finer grained nature and lower permeability of sediments in the south.

Chemical stratification

To investigate the variation of various chemical parameters with depth, three groundwater samples were taken during drilling at two-meter depth intervals down from the water table in borehole 4 (at the landfill) and borehole 32 (at the shooting-range). The results are shown in Table 3. The fact that the temperature, alkalinity and chloride concentrations decrease with depth at the landfill, and that the Eh and pH increase, indicate that the groundwater is most polluted in the upper zone of the aquifer. Electrical conductivity, and thus total dissolved solids, also shows a slight decrease with depth in borehole 4 (from 1226 to 1142 $\mu\text{S/cm}$; i.e. by 6.9%), but is constant in borehole 32.

Table 2. Estimates of the hydraulic parameters based on grain-size distributions of sediment samples from below the water table (see Langguth & Voigt 1980). d_{20} and d_{60} are grain size parameters, $P(n)$ = porosity, $P(n_e)$ = effective porosity, $P(S_e)$ = correction factor for effective porosity.

Borehole no. & depth	d_{20}	d_{60}	u	C	Hydraulic cond. (m/s)	P (n)	P (S_e)	P (n_e)	Gradient (i)	Real velocity m/s	Real velocity m/d
24-18	0.29	0.64	2	0.01	$8.4 \cdot 10^{-4}$	0.35	0.9	0.32	0.013	$3.4 \cdot 10^{-5}$	2.9
25-14	0.09	1.3	14	0.007	$5.7 \cdot 10^{-4}$	0.28	0.8	0.22	0.013	$3.4 \cdot 10^{-5}$	0.29
21-26	0.09	0.2	2	0.01	$7.4 \cdot 10^{-4}$	0.35	0.8	0.28	0.013	$3.4 \cdot 10^{-5}$	0.29
51-33	0.16	0.47	3	0.009	$2.3 \cdot 10^{-4}$	0.33	0.9	0.29	0.017	$1.3 \cdot 10^{-4}$	1.16
43-18	0.22	0.47	2	0.01	$4.8 \cdot 10^{-4}$	0.35	0.9	0.32	0.025	$3.8 \cdot 10^{-4}$	3.24
44-18	0.28	0.8	3	0.009	$7.4 \cdot 10^{-4}$	0.34	0.9	0.31			

Table 3. Overview of chemical parameters in groundwater as function of depth, in boreholes 4 and 32.

Increasing with depth	Stable	Decreasing with depth
Eh	Al	Temperature
pH	Mg	Conductivity
Fe	Ca	Alkalinity
SO ₄ ²⁻	Na	Mn
	K	Cu
		Zn
		Ni
		Co
		B
		Li
		Cl ⁻
		NO ₃ ⁻

The contaminant plume

The chemical parameters measured in the pumped groundwater samples were very stable during the two months in which sampling took place for most components, except Zn, Fe, and Mn. The variation of these components could be due to different degrees of oxidation of the strongly reduced water during sampling.

The heavy metals Pb, Zn, Cu and Mn appear to be accumulated in the sediments under, or immediately downstream of, the landfill. This is presumably a result of the strongly reducing, sulphide-, bicarbonate- and metal-rich leachate from the landfill mixing with the more alkaline regional groundwaters, and the metals being immobilised in the sediments through adsorption or precipitation. Elevated values for these elements were, in fact, confirmed in the sediment samples below the landfill (see above).

The small vertical variations in groundwater chemistry and the consistency of analytical results during the three sampling rounds justify an evaluation of the lateral variations in hydrochemistry based on comparisons with «background levels» measured in borehole 26.

Chloride and electrical conductivity

The concentration of chloride (Fig. 7a) in the groundwater is highest in the vicinity of the landfill (over 80 mg/L in boreholes 4 & 21), and decreases towards the east in the direction of the groundwater flow in a similar manner to the electrical conductivity. The chloride concentration is only 3-4 mg/L in boreholes

41-43 and in the reference borehole 26. Chloride is a conservative ion which is not significantly retarded by sorption, ion-exchange or precipitation in the subsurface. It can thus be assumed that it is transported at the same rate as the groundwater (Freeze & Cherry 1979), and is regarded as a better tracer of contamination from domestic waste than electrical conductivity.

Electrical conductivity is roughly proportional to the total dissolved solids in aqueous solutions of low ionic strength. The electrical conductivity is highest in groundwaters beneath the landfill (Fig.7b), where it is 1420 µS/cm in borehole 3 and 1013 µS/cm in borehole 4. It decreases to 200 µS/cm in borehole 42. The electrical conductivity of the groundwater at the reference borehole 26 is only 123 µS/cm. The decline in electrical conductivity and chloride away from the landfill illustrate how dilution and dispersion can attenuate a contaminant plume from a point source of pollution.

pH, temperature and alkalinity

As shown in Table 4, the pH of the groundwater in the Romerike aquifer averages 7.9 at thirty metres depth (Jørgensen et al. 1991). In this study a pH of 8.4 has been measured in water from the reference well, borehole 26 (Fig.7c). At the northwest corner of the Transjøen lake where groundwater is abstracted for domestic use, the pH is around 7.5. The lowest pH value of 5.8 was measured in borehole 4 which is located in the eastern margin of the landfill (Fig.3). A slightly alkaline value of pH is common in leachate from old domestic waste landfills (Mather 1992). It is therefore remarkable that such a low pH has been recorded in groundwater (which naturally has a pH around 8) at the Trandum landfill; such low-levels are more characteristic of young landfills, generating abundant organic acids. It is likely therefore that parts of the Trandum landfill are still in their acetogenic phase.

In the area of the shooting range the pH is about 6.5, presumably due to infiltration of surface water drainage via the small pond in the area.

The groundwater temperature is c. 7.9°C beneath the landfill (Fig.7d). This is 3-4°C higher than values determined in the surrounding area, and is presumably a result of exothermal biodegradation of organic material.

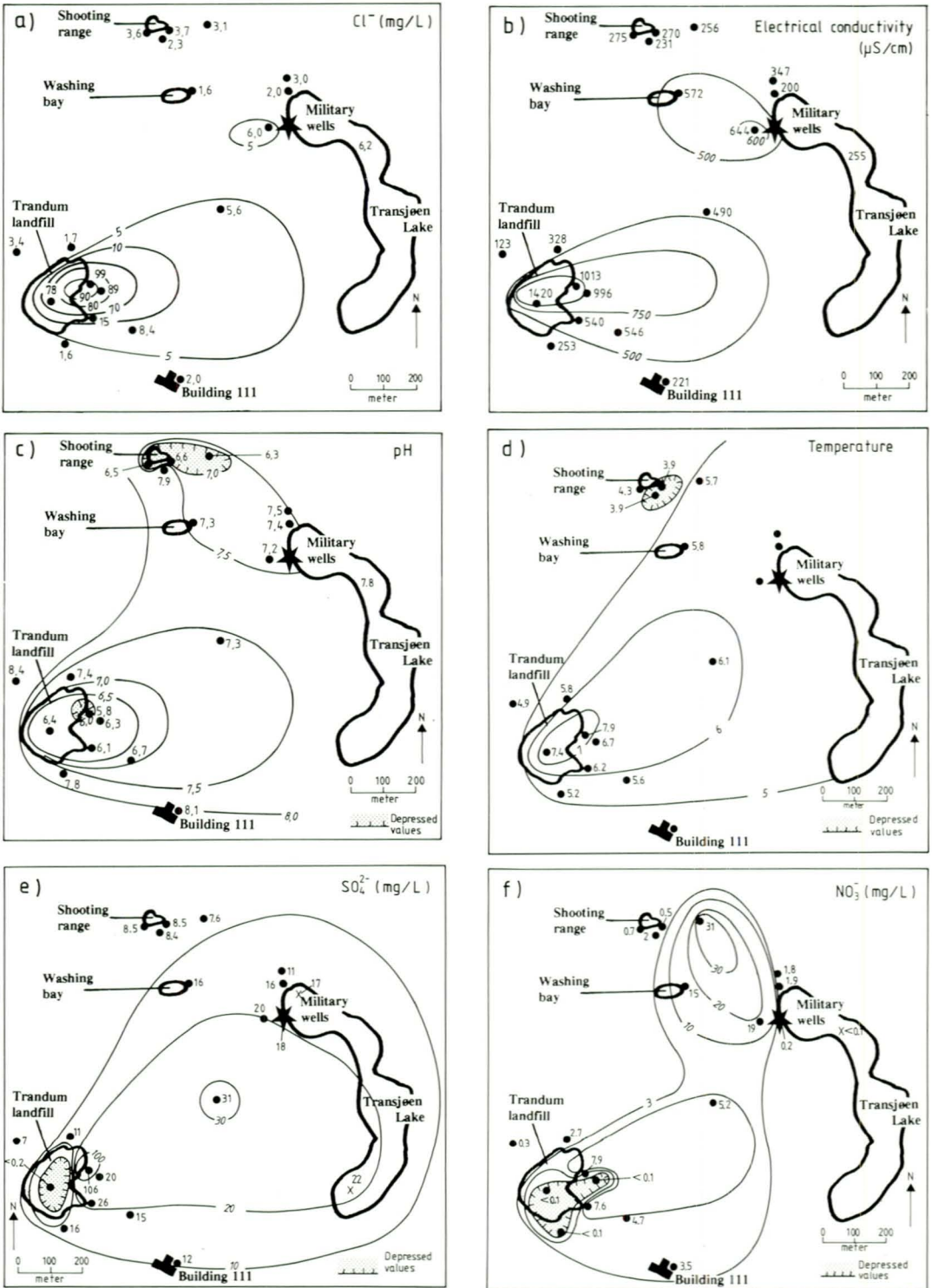


Fig. 7. Contour maps showing the distribution of a) chloride (mg/L), b) electrical conductivity ($\mu\text{S/cm}$), c) pH, d) temperature ($^{\circ}\text{C}$), e) sulphate (mg/L), and f) nitrate (mg NO_3^-/L) in the groundwater of the investigated area.

Table 4. Chemical parameters in groundwater derived from this investigation, results from other investigations and SIFFs quality requirements for inorganic constituents in drinking water. All concentrations are in mg/L. Trsj: Transjøen lake, Msb: Military supply borehole, Dgw: Deep groundwater sample collected at 30 m depth at Furusmo, (G): Good, (A) : Acceptable, (P): Poor quality, SIFF = drinking water requirements (SIFF 1987).

Sample	Na	K	Mg	Ca	Cl ⁻	SO ₄ ²⁻	NO ₃ ⁻ -N	Alk mmol/L	Cond µS/cm	pH
Risa*	3.1	1.2	2.9	31.4	2.9	12.1	0.1	1.6	172	7.7
Aurtjern*	1.4	0.5	0.8	3.7	1.1	3.7	0.1	0.3	30	6.7
Trsj*	5.8	1.5	4.7	50.9	6.2	18.3	<0.1	2.6	255	7.8
Trsj	4.1	<0.2	3.5	51.8	5.5	16.6	<0.1	2.4	259	8.2
Build. 111 ⁺	3.5	1.0	3.0	53.0	2.5	13.0	2.0	2.7	290	8.1
Build. 111	2.5	<0.2	1.7	38.0	2.0	11.7	0.8	2.1	221	8.1
Bh26	2.0	<0.2	1.9	17.8	3.4	7.0	0.1	1.1	123	8.4
Bh3	45	34.6	30	146	78.4	0.2	<0.1	13.8	1420	7.1
Bh43	6.7	<0.2	6.9	126	6.0	19.7	4.4	6.6	644	7.5
Msb	3.4	1.1	3.6	56.2	4.0	17.6	<0.1	2.7	275	8.0
Dgw*	1.9	1.25	2.5	39.1	1.1	14.9	<0.1	1.9	202	7.9
SIFF	<20 (G)	-	>20 (P)	15-25 (G)	>200 (P)	>100 (P)	>10 (P)	<1.0 (G)	-	6.5-9.0 (A)

* Data from Jørgensen et al. (1991)

⁺ Data from Storø (1991).

Alkalinity varies between 1 mmol/L at the reference well to 14 mmol/L in borehole 3. This reflects the high concentration of bicarbonate beneath the landfill due to the disintegration of organic refuse. Johansen & Carlson (1976) report an alkalinity between 6 and 41 mmol/L in leachate from six landfills in Norway.

Sulphur and nitrogen

Concentrations of more than 100 mg/L sulphate were found in borehole 4 (Fig.7e). Elevated values of 10 to 30 mg/L were also found in boreholes 21, 23, 24, 25, 31, 43 and 51. The background value, as measured in borehole 26, was 6-7 mg/L. The concentration of sulphate in borehole 3 was very low (<0.2 mg/L), but a potent smell of H₂S gas was evident, indicating reducing conditions under the landfill. This is reflected in the redox- potential which was found to be -30 mV in borehole 3, in contrast to over +100 mV in pumped samples from the adjacent areas.

The concentration of nitrate is less than 0.3 mg NO₃⁻/L in the reference borehole 26 (Fig. 7f). The nitrate concentrations are below the detection limit (0.05 mg/L) in boreholes 3, 21 and 24 located centrally and just east of the landfill. However, nitrogen in the groundwater in the vicinity of the landfill is present as ammonium ion (20-25 mg/L) due to the strongly reducing conditions. Nitrite was only detected in borehole 25 indicating a transition in redox-conditions in this area. The highest concentrations (31.2 mg/L) of nitrate were found in borehole 36, probably deriving from gunpow-

der from the shooting range. Nitrogen in all three forms is believed to reflect the influence of anthropogenic sources, including the landfill.

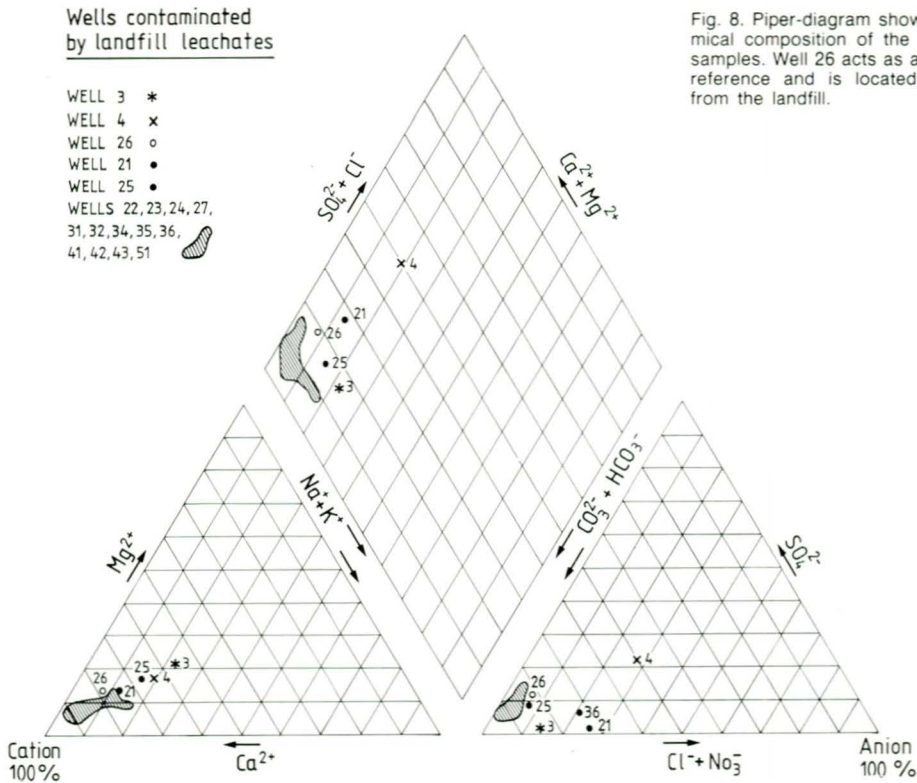
Discussion

Classification of the groundwaters

The groundwater samples are of calcium-bicarbonate type (Fig. 8) with a pH between 5.8 and 8.4. Elevated concentrations of Mg, Na and K are found in the waters beneath the landfill, compared to the less- or uncontaminated waters such as in borehole 26. The percent proportion of Ca is, however, lower in the vicinity of the landfill (Fig.8), relative to the other major cations. Whereas Cl⁻ concentrations are high in the contaminated waters in the vicinity of the landfill, differences in the bicarbonate and sulphate (due to the sulphide to sulphate redox transition) concentrations cause the neighbouring boreholes 3 and 4 to be widely separated on the anion-triangle, and thus on the central diamond, of the Piper diagram (Fig.8).

The contaminated groundwaters are readily classified in a diagram where pH is plotted against electrical conductivity (Fig. 9):

(a) Waters with the lowest pH-values and highest conductivities are found in wells 3, 4, 21, 23 and 25, i.e. those surrounding the landfill. These are the most contaminated groundwaters.



(b) Boreholes 31, 43 and 51 (beneath the military base) contain water with slightly elevated TDS (i.e. higher conductivity) but with «normal» pH values. It is possible that these groundwaters are slightly contaminated by the outer fringes of the contaminant plume.

(c) Groundwater samples from the boreholes at the shooting range (boreholes 34, 35 and 36) have a low pH, but the conductivity is well below 400 $\mu\text{S}/\text{cm}$. They cannot be considered as contaminated. The lower pH is presumably due to infiltration of low-pH, low TDS surface water drainage (including drainage from the washing bay) through the relatively thin unsaturated zone in the base of the kettle hole.

(d) The groundwaters from wells 22, 24, 32, 41, 42 are all fairly similar in quality to that from borehole 26 (normal pH and low TDS).

Saturation indices

The natural groundwaters in the Øvre Romerike aquifer are in equilibrium with the $(\text{Ca}^{++}$,

$\text{Mg}^{++}) \text{CO}_3^-$ -type found in these deposits (Jørgensen et al. 1991). The state of equilibrium between the groundwaters and several minerals has been estimated using a computer code based on thermodynamic calculations of ion activity products (IAP) which in turn are related to equilibrium constants (K_{eq}), ionic strength and ion activity coefficients. A calculation of the saturation indices ($\text{SI} = \log(\text{IAP}/K_{\text{eq}}) = \text{p}K_{\text{eq}} - \text{pIAP}$) for calcite in each well (Fig. 10a) indicates that the groundwaters around the landfill and at the shooting range are significantly undersaturated in spite of the high alkalinity and high concentrations of calcium measured in these areas. This is a result of the low pH which causes the calcite to be consumed at a higher rate than it is produced.

The acid produced by the landfill combined with the anaerobic environment generates reduced species of sulphur (H_2S , HS^-) and nitrogen (NH_4^+ , NO_2^-). The transition between NH_4^+ and NO_2^- is located between boreholes 3 and 25 (Fig. 7f). The redox boundary between H_2S and SO_4^{2-} is indirectly delineated in Fig. 10b where the saturation indices of barite from

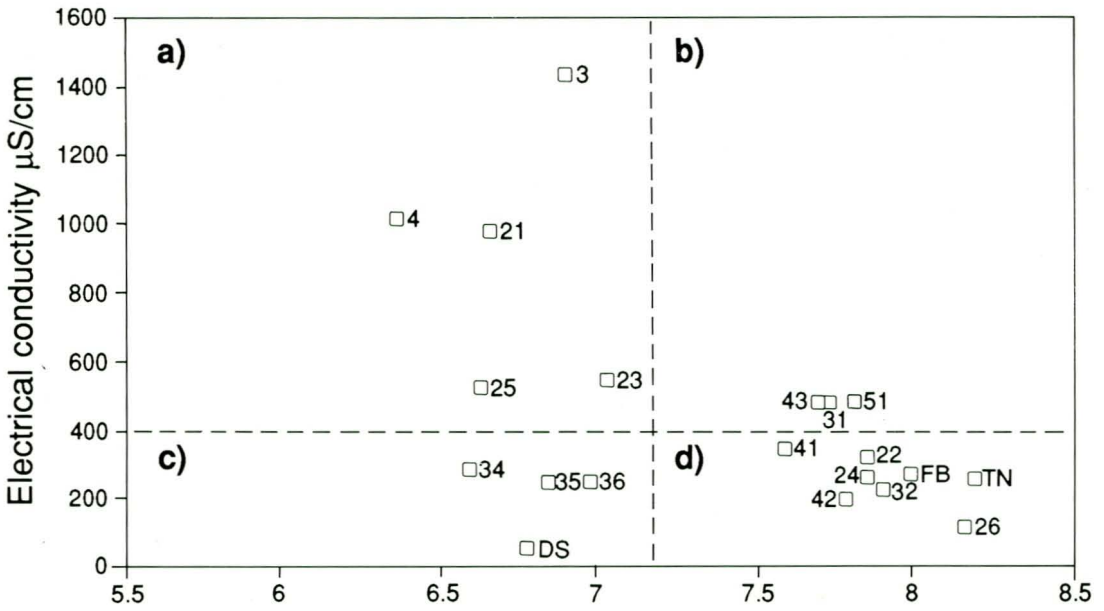


Fig. 9. Average values of electrical conductivity versus pH in groundwater. TN: Transjøen North (lake water), DS: Pond at shooting range, FB: Military supply well for domestic water. a) Contaminated samples, b) slightly contaminated samples, c) and d) uncontaminated samples.

each borehole are contoured. A transition zone from undersaturated to supersaturated waters with respect to barite is located a little to the east of borehole 3. A similar boundary is located south of boreholes 31-36. It is likely that the concentration of barium in the

groundwaters is limited by the solubility of barite.

Both calcite and barite are supersaturated in groundwaters from boreholes 31, 43 and 51. Supersaturation of groundwaters with respect to calcite, dolomite and siderite was

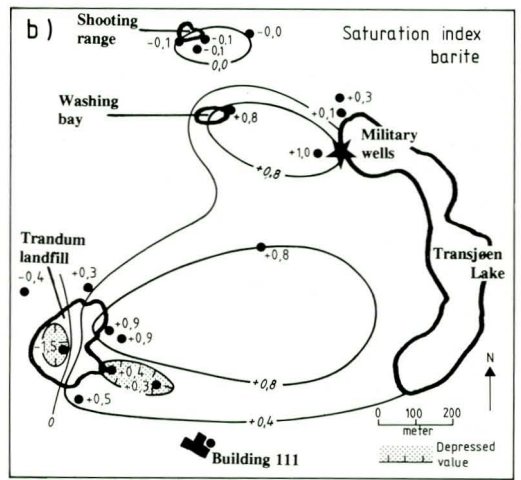
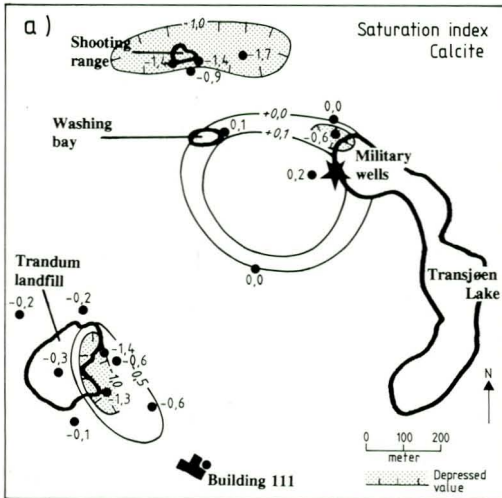


Fig. 10. Contour map showing the distribution of the saturation indices of a) calcite and b) barite in the groundwater of the investigated area.

also detected in a plume from the KL Landfill in Kalamazoo County, Michigan, USA (Kehew & Passero 1990).

Comparison between water types

A comparison of the chemical analyses obtained in this study with results from earlier investigations at Building 111 (Storrø 1991 - Table 4), shows a variation of up to 25% in TDS at borehole 27 (Fig.3). This may be due to different sampling procedures, particularly with respect to pumping before sampling. At Transjøen lake the results vary only by 2% (for TDS) during a six month period.

The concentrations of the major ions Na^+ , K^+ , Mg^{++} , Cl^- , are 10-30 times higher in the contaminated waters at borehole 3 than at the Military supply wells and Transjøen lake (which is almost wholly groundwater-fed), indicating an attenuation of at least one order of magnitude across the investigated area.

There is a two-fold net increase in specific conductivity from the reference borehole 26 to the military domestic water supply wells where drinking water is withdrawn. This latter water is very similar to the waters of Transjøen lake. This implies that the groundwater in the vicinity of the lake has acquired a significant amount of all major ions (except NO_3^-) during its drainage across the investigated area. This water fails to meet the SIFF quality standards required for «good» drinking water because of its high content of calcium (SIFF 1987), but it is regarded as of «acceptable» quality.

Conclusions

The groundwater flow at the site is in an easterly direction with a real velocity of 1.5 m/day.

Significant contamination of soils and groundwater in the immediate surroundings of the landfill at Trandum has been detected. Sediments contain levels of Pb, Zn, Hg, Mn, V, Cu and Ni which are well above background values. However, only Pb and Zn are found to exceed the European Community's requirements for further mapping of the extent of the pollution.

The groundwater is not contaminated below the washing bay nor in the vicinity of its sewer discharge at the shooting range.

The groundwater's electrical conductivity decreases from over 1000 $\mu\text{S}/\text{cm}$ just beneath the landfill to 200 $\mu\text{S}/\text{cm}$ one kilometre down-gradient to the east. Upstream, the background level is 123 $\mu\text{S}/\text{cm}$ at a reference well 100 m NW of the center of the landfill. The concentrations of the major ions Na^+ , K^+ , Mg^{++} , Cl^- , are 10-30 times higher in the contaminated waters at borehole 3 than at the military wells and Transjøen lake, indicating an attenuation of at least one order of magnitude across the investigated area.

The groundwaters, apart from those in the immediate vicinity of the landfill, tend to be supersaturated with respect to calcite and barite suggesting that the concentrations of Ca^{++} , HCO_3^- , and Ba are partially governed by the saturation of these minerals.

It is recommended that both inorganic and organic components of the groundwater are monitored in the future until contamination from the landfill is significantly reduced or has ceased.

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