SARDINIA '89 - 2nd International Landfill Symposium

ASSESSMENT OF ENVIRONMENTAL IMPACTS AND CONTROL MEASURES AT WASTE TIPS

Enrico A.J.M. van den Bogaard, Jan Hoeks

The Winand Staring Centre for Integrated Land, Soil and Water Research, P.O. Box 125, 6700 AC Wageningen, The Netherlands

SUMMARY

Under specific conditions, simple methods can be used in assessing environmental impacts and considered control measures at waste disposal sites. Some models, virtually based on the water balance principle, are demonstrated on estimating migration of pollutants in groundwater, and on calculating leachate effluent and gas production.

1. INTRODUCTION

Since 1986 in the Netherlands new licences for waste disposal sites can only be granted if the environmental impacts have been described in a report as required by the Law on Environmental Impact Assessment (EIA).

Most common impacts in an EIA study for a waste disposal site are:
- risk of soil and groundwater contamination;
- risk of air pollution and injury to vegetation by uncontrolled emission of gases and odours;
- direct health risks, caused by spreading of deseases (by insects or rats) or noise nuisance (increased traffic);
- impacts on landscape.

Measures to control the emission of leachate and gas are: construction of bottom and top liners, completion strategy, leachate management, gas extraction and monitoring systems.

In EIA studies several types of models can be used in order to predict impacts and to design control measures. Complex, detailed models - if applicable in the concerned case - are able to generate more exact results. However, complex models tend to require a lot of different, specific input data c.q. parameters, which usually are not readily available and cannot be collected easily. Under specific conditions, such as in the presented case, the use of simple models can be satisfactory for the intended purpose without unacceptable loss of accuracy. Because less input data are required, simple models can be applied easier and faster.

To demonstrate the applicability of simple models at some aspects of EIA of waste disposal sites, in this paper some results of a case study (VAN DEN BOGAARD and HOEKS, 1988) are presented, for which such models and calculations have been used. After description of the concerned site (Section 2), in the next sections the following aspects are considered:

- the environmental impact of waste disposal compartments on soil and groundwater (Section 3);
- control measures to be taken to reduce or prevent environmental pollution (Section 4) with special reference to leachate management (Section 5) and gas production (Section 6).

2. LOCATION AND DESCRIPTION OF THE SITE

2.1. The waste disposal site

The case study was carried out for the regional waste disposal site Linne/ Montfort. The site is located in the southeastern part of The Netherlands, 10 km south of Roermond, 2300 m east of the river Meuse.

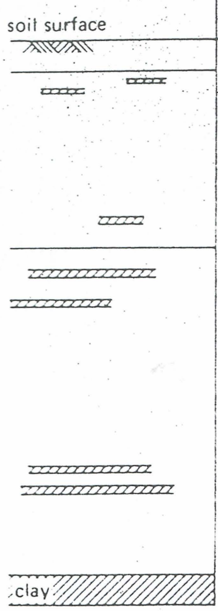
From 1966 until 1986, approximately 175.000 tons of waste have been dumped yearly in a sand mining area of 17 hectares with a depth of 5 m, without any measures taken to control leachate and gas production and discharge.

Since 1986 disposal of waste is carried out according to legislative directives, which means that at new compartments, with an area of in total 19 hectares, lining material will be applied underneath the waste. Until 2000, each year approximately 250,000 tons will have to be disposed off and dumped at the regional site. The waste consists of household and municipal waste (30%), demolition waste (20%) and industrial bulk waste (50%).

Both old and new compartments will ultimately be provided with top surface caps, while measures for leachate management, gas extraction and site monitoring are considered. The intended final use of the complete site is extensive recreation in a more or less natural-grown undulating area.

2.2. Geohydrological situation

An outline of the hydrogeological situation was derived from earlier regional investigations and local drillings. It appeared that underneath the site no protective low-permeable soil layer is present. Leachate from the old, non-protected compartments, is discharged directly into the aquifer. This unconfined



depth below soil surface (m)	height above MSL (m)	thickness of deposits (m)	type of deposits	hydrogeological characterization	hydrogeological parameters (approx.)
	30				
5	25	5	cover sand, fine-course, with clay	of no significance	
		35	course sand and gravel; local clay lenses	well permeable aquifer, isotropic	$k = 35$ à $100\ m.d^{-1}$; $kD = 2800\ m^2.d^{-1}$; $i = 17.5 \times 10^{-4}$; $v_h = 50\ m.yr^{-1}$
40	-10	65	medium fine and course sand with thin layers of clay and coal	medium permeable aquifer, anisotropic	$k_h = 10\ m.d^{-1}$; $kD = 650\ m^2.d^{-1}$; $v_h = 6\ m.yr^{-1}$; $i = 17.5 \times 10^{-4}$
105	-75	7	clay	base of geoh. system	$k \approx 0$, $kD \approx 0$

Fig. 1. Schematic cross-section of the hydrogeological profile at the location of the waste disposal site. MSL = mean sea level

aquifer is about 100 m thick. The upper part (35 m thick) consists of well permeable course sand and gravel with few small clay lenses (fluvial deposits from upper and middle Pleistocene). The lower part (65 m thick) consists of medium permeable sands and thin local clay layers (lower Pleistocene deposits). A 7 m thick layer of clay at a depth of 105 m below soil surface forms the base of the considered hydrogeological system (Fig. 1).

The water table (phreatic surface) is found at 6 m below soil surface. Regional and local the patterns of equipotential lines/isohypse patterns of phreatic and middle deep groundwater have been derived from existing maps and measured hydraulic heads in observation wells near the site.

It appeared that the above mentioned clay lenses in the aquifer do not induce vertical flow resistance or disturbances of the flow pattern. This flow pattern is not very complex here and quite stable throughout the year.

3. GROUNDWATER CONTAMINATION

3.1. Water balance at the site

Combining the collected geological, hydrological and meteorological data, a rough water balance of the yet uncapped area (17 ha) was calculated, in order to get a first indication of the plume of migrating pollutants in the groundwater. As a result of infiltration of precipitation surplus, the vertical volume flux of leachate to the aquifer is 45,000 $m^3.yr^{-1}$. This amount appears to be, nevertheless, only 1% of the horizontal volume flux in the aquifer (apparent velocity v = 50 $m.yr^{-1}$, which is high under Dutch circumstances).

This result indicates that pollutants will not penetrate the aquifer to a large depth.

3.2. Set-up of a simple model

In this study there is a large catchment area of infiltration upstream of the site. Although two layers of different permeability are present in the aquifer, transformation of this situation to an aquifer with homogeneous characteristics is possible. For this transformation, the lowest part of the aquifer with the lower permeability is replaced by a thinner layer with the same permeability as the upper part of the aquifer. For this transformed situation, the use of a relatively uncomplicated model - based on the water balance principle - is allowed to calculate the migration of pollutants in the aquifer (HOEKS, 1981).

The regional groundwater flow was abstracted and considered to be a divergent radial flow. According to HOEKS (1981) in such situations the flow velocity underneath the disposal site is

$$v^* = \frac{x_s N}{2 \epsilon D} \tag{1}$$

where v^* = effective (interstitial) flow velocity of groundwater front ($m.yr^{-1}$)

x_s = distance from watershed (m)

N = infiltrating precipitation surplus ($m.yr^{-1}$)

ϵ = effective waterfilled pore volume (-)

D = thickness of aquifer (m)

This equation can be used for a homogenous aquifer, where the hydraulic conductivity (k) is assumed to be constant with depth. Since there are two different zones in the aquifer (Fig. 1), a replacing (fictive) thickness has to be calculated first. The upper part of the aquifer (with $k = 80\ m.d^{-1}$) is regarded as main flow zone for migrating pollutants (in accordance with the results of Section 3.1 and the data shown in Fig. 1). The total transmissivity, kD, of the aquifer amounts to 2800 + 650 = 3450 $m^2.d^{-1}$, so the replacing thickness D' is 43 m (for $k = 80\ m.d^{-1}$).

Application of eq. (1) for calculation of the flow velocity, v^*, below the waste disposal site yields 131 $m.yr^{-1}$ (other data: $x_s = 15{,}000$ m; $N = 300\ mm.yr^{-1}$ and $\epsilon = 0.40$).

This shows a striking similarity with v^* as derived from the local pattern of equipotential lines (groundwater isohypses):

$$v^* = \frac{k}{\epsilon}\ i = \frac{80}{0.40}\ .\ 17.5 * 10^{-4} * 365 = 128\ m.yr^{-1} \qquad (2)$$

where i = hydraulic head gradient in the aquifer (-)
k = hydraulic conductivity ($m.d^{-1}$)

This similarity justifies the assumption of a radial flow pattern.

3.3. Modelling the migration of pollutants

Next, the transport of pollutants along a streamline in the aquifer can be calculated. For the migration of a solute front the following equation is used:

$$v_i = v^* \left(\frac{1}{1 + R_i}\right) \qquad (3)$$

where: v_i = migration velocity of solute front ($m.yr^{-1}$)
v^* = effective flow velocity of groundwater ($m.yr^{-1}$)
R_i = distribution ratio (-), representing the distribution of solute i over solid phase (adsorption) and soil solution ($1+R_i$ = retardation factor)

Based on eqs. (1) and (3), the following solutions can be derived (HOEKS, 1981) for travelled distance (x_i), depth (d_i) and concentration at the solute front, taking into account adsorption and degradation:

$$x_i = x_s\left(e^{Nt/2\epsilon D(1+R_i)} - 1\right) \qquad (4)$$

$$d_i = \left[1 - \left(\frac{x_s}{x_i + x_s}\right)^2\right]D \qquad (5)$$

$$C_i = C_o\ e^{-kt/(1+R_i)} \qquad (6)$$

where x_i = horizontally travelled distance of solute i, in t years, counted from the point of infiltration (m)
x_s = distance from watershed (m)
t = time since infiltration in aquifer (year)
d_i = penetration depth of contamination front in aquifer (m)
C_i = concentration of solute i at front, after t years ($mg.l^{-1}$)
C_o = concentration of solute i in infiltrating leachate ($mg.l^{-1}$)
k = degradation coefficient ($year^{-1}$)

Table 1. Horizontally travelled distance and relative concentration of contaminants at the solute front near the waste disposal site of Linne/Montfort, as a function of adsorption and degradation processes

R_i = distribution ratio (-). R_i = 0 means a high mobility (e.g. Cl); R_i = 4 means medium (e.g. NH_4, K); R_i = 20 means slow (e.g. Cu, Pb)

K = degradation coefficient ($year^{-1}$). k = 0 means persistent (e.g. heavy metals, persistent organic components); k = 0.2 means medium degradable; k = 0.5 means high degradable (degradable organic pollutants only)

C_o = concentration of solute i in infiltrating leachate ($mg.l^{-1}$)

Time since infiltration (years)	Travelled distance (m)			Concentration C_i (as % of C_o)		
	R_i = 0	R_i = 4	R_i = 20	K = 0	K = 0.2	K = 0.5
0	0	0	0	100	100	100
2	264	52	12	100	67	37
4	532	105	25	100	45	14
6	806	158	37	100	30	5
8	1084	211	50	100	20	2
10	1367	264	62	100	14	1
12	1655	317	75	100	9	0
14	1948	371	87	100	6	0
16	2246	425	100	100	4	0
18	2550	478	113	100	3	0

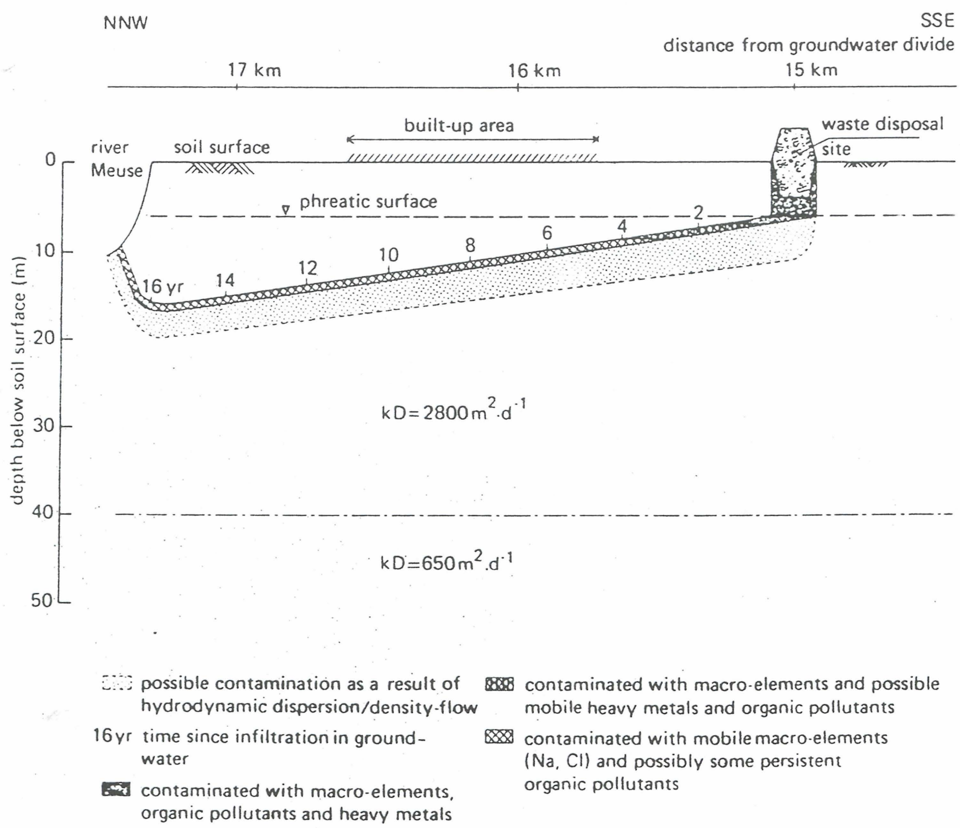


Fig. 2. Contaminated plume downstream of the waste tip, related to residence time; calculated with the model presented in Section 3.

Because the original concentrations of pollutants in the infiltrating leachate (eq. 6) were not available, calculations have been carried out (Table 1) for pollutants with roughly estimated distribution ratios and degradation coefficients (as can be obtained from soil column experiments, e.g. CHRISTENSEN et al., 1987). In this way an indication is obtained concerning the migration of certain groups of conservative and nonconservative pollutants. Theoretically a wide range of combinations of mobility and degradability is conceivable.

Knowing that waste dumping has taken place here since 20 years, the calculations (Table 1) show, that early infiltrated mobile, non-degradable solutes ($k = 0$; $R_i = 0$) in the groundwater will have reached the river Meuse (2300 m NNW) already.

With the analytical solutions presented in eqs. (4), (5) and (6), the course of the contaminated streamline in the aquifer can be calculated and outlined (Fig. 2).

3.4. Testing the model by groundwater sampling

Since the disposal site is stretched out perpendicular to the general streamline pattern over a width of approximately 1000 m, groundwater quality downstream of the site will be endangered in a zone of approximately 250 ha. In this downstream zone (where also the municipality of Linne is situated) several private wells are located, at least eight of which withdraw water from the calculated polluted depth (Fig. 2).

Groundwater samples from private wells were examined on typical tracers, such as chloride, ammonium-nitrogen, electric conductivity and total organic carbon (TOC). These data were compared with the existing wide range of data from monitoring wells surrounding the site.

It appeared that three wells at a distance of 500 m from the site were already severely affected (further analyses proved the presence of chloroform). At larger distances, 8 others showed raised levels of tracer concentrations. Monitoring wells close to the disposal site show a clear increase in concentration of various pollutants over the years, so contamination in private wells located further away is expected to increase in the near future.

The contamination, observed in monitoring wells and private wells, is in good agreement with the predicted situation calculated with the described model.

4. GENERAL ASPECTS WITH RESPECT TO CONTROL MEASURES

4.1. Bottom liners

At new compartments to be constructed at the site, bottom liners are prescribed to prevent infiltration of leachate into the soil and groundwater. For this purpose a locally mined tertiairy clay, consisting of 38.7% clay particles (< 2 μm) and 61.0% silt particles (2-50 μm), is used. In permeability tests as described for natural liners by HOEKS et al. (1987), and HOEKS and RYHINER (1987), the permeability was so low, that at the expected hydraulic head gradients the leakage of leachate through this liner is expected to be negligible.

The liner has to be constructed with a thickness of 50 cm, according to prescribed conditions. Leachate will be collected in a drainage system, pumped out and discharged into a reservoir (Section 5).

4.2. Completion strategy

The advised completion strategy is based on a phased approach, in which compart-

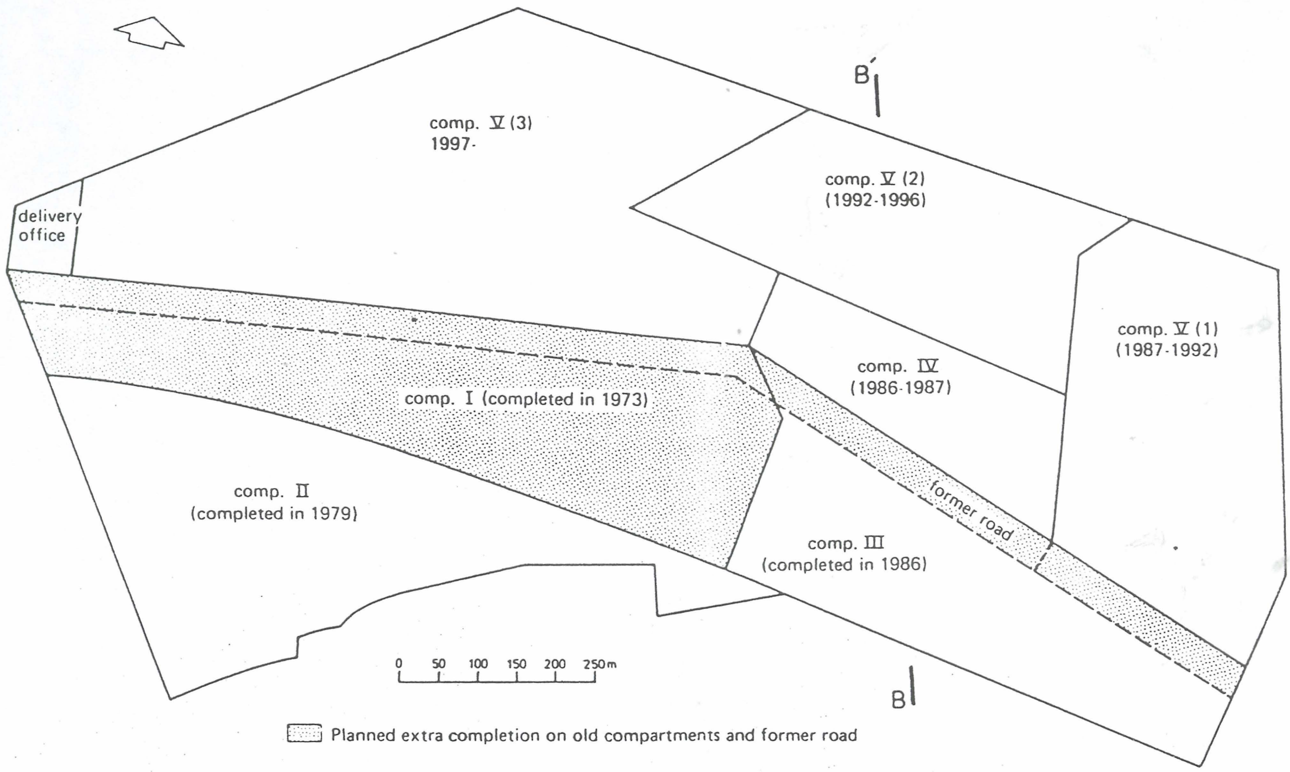


Fig. 3. Phased construction of compartments. The 'old' compartments I. II and III are not provided with bottom liners. Cross-section B-B' is outlined in Fig. 4.

ments are built up fast (up to 25 m), separated from each other, one after another (Fig. 3). Advantages are:
- reduction of collected volume of leachate;
- reduction of consequenses of possible leakages or drainage system failures;
- surface capping and measures for gas extraction and recirculation can be constructed earlier;
- investment profits concerning liner construction.

Waste will be compacted in order to reduce expected consolidation and settlement rates to an estimated 25%, in view of early and safe surface capping.

4.3. Surface capping

Laboratory research and practical experiences with clay liners for surface capping are, among others, described by HOEKS and AGELINK (1982), and HOEKS and RYHINER (1987). Based on these experiences, a technical protocol on surface caps is being prepared now to support new legislative directions.

At the old compartments (which are not underlined) surface caps are necessary to stop infiltration of rainwater, in order to prevent further groundwater contamination. The already mentioned tertiairy clay can be used for this purpose.

Surface capping has also been advised for the new (bottom-lined) compartments, because:
- application of two liners reduces the risk of leakage;
- gas emission is controlled, gas can be extracted and exploited;
- lateral emission of leachate is prevented;
- better chances are provided for final use, e.g. vegetation cover;
- costs of leachate collection and treatment can be reduced substantially (Section 5).

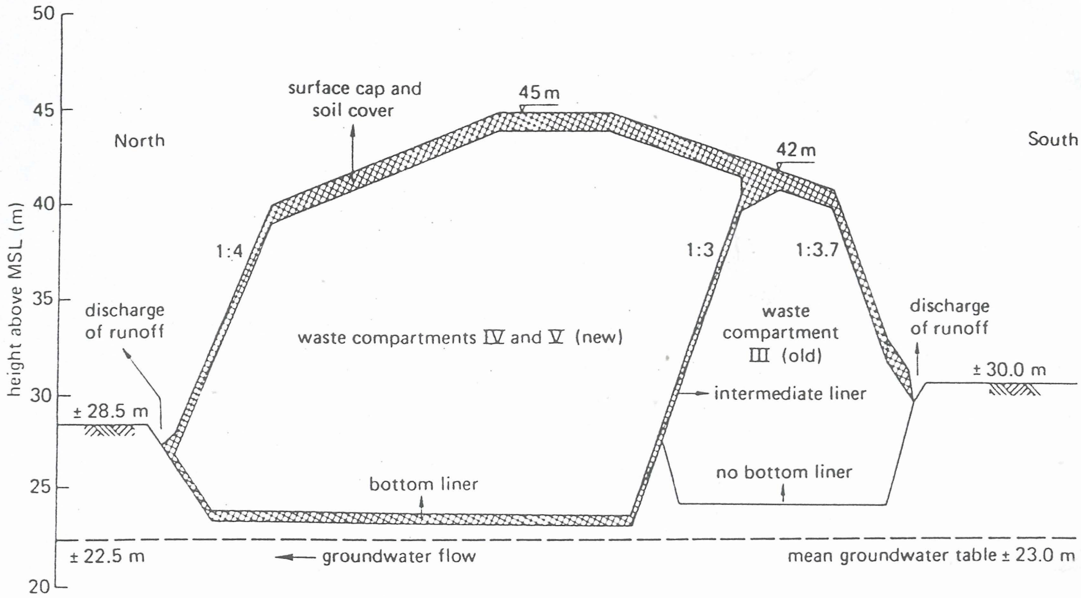


Fig. 4. Outline in cross-section of the final constellation of the waste disposal site, with respect to liners and build-up. MSL = mean sea level

Application of surface caps requires the construction of a drainage system for discharge of rainfall surplus, e.g. with drains in a layer of 25 cm permeable sand. This layer has to be covered with a humic soil layer (75 cm thick) with a water holding capacity of 150-200 mm (depending on the aimed vegetation type).

An outline (in cross-section) of the advised final constellation with respect to liners and build-up of the site, is given in Fig. 4.

In the study a control and monitoring strategy has been set up, which will not be discussed in this paper.

5. ESTIMATION OF LEACHATE PRODUCTION; TREATMENTS AND COSTS

5.1. Treatment strategy

Leachate, collected from the new, bottom-lined compartments, and temporarily stored in a reservoir, will finally have to be discharged. In the present case, the leachate is discharged into a sewage treatment plant of the provincial water board of Limburg. The water board calculates discharge costs by an equation based on quantity and quality of the discharged effluent:

$$\text{n.i.e.} == (Q/136) \times (COD + 4.57\ N_{kj}) + [\text{heavy metals conc.}] \qquad (7)$$

where n.i.e. = number of inhabitant equivalents (-)
Q = discharge of leachate effluent ($m^3.d^{-1}$)
COD = chemical oxygen demand ($mg.l^{-1}\ O_2$)
N_{kj} = concentration of Kjeldahl-nitrogen ($mg.l^{-1}$ N)

The inhabitant equivalent (i.e.) is defined as the discharge of 54 g Biological Oxygen Demand (BOD) per inhabitant per day. The charge per i.e. amounts to Dfl. 55.- (approximately 23 ECU).

Since the concentration of heavy metals appeared to be relatively low, their contribution in eq. (7) is further neglected here.

Sophisticated treatment facilities on the site are not efficient, since a large scale treatment plant of the water board is available nearby. When recirculation at the waste disposal site will be applied, further pretreatment is not interesting because fatty acids are almost completely degraded in the waste tip (VROM, 1984). This leaves the next alternatives for leachate treatment to be considered:

I. Direct discharge of the effluent to the provincial treatment plant; no surface capping applied.

II. Direct discharge of the effluent to the provincial treatment plant; with surface capping. Leachate production will finally decrease to zero, and thus save money for treatment.

III. Discharge, after recirculation on site (surface capped). A further decrease of costs is realized in this case. Recirculation increases biological activity in the waste tip and thus pollutant concentrations are decreasing and mineralization of waste is increasing (EHRIG, 1980; HAM, 1983). This, in fact, is a result of getting into the 'methane fermentation stage' earlier (Section 6). Recirculation takes place with a system to ensure proper infiltration underneath the surface cap.

From a purely environmental point of view, the third alternative is considered best. Nevertheless, external final treatment of leachate cannot be avoided completely by this recirculation of leachate (BLAKEY and MARIS, 1987; DOEDENS and CORD-LANDWEHR, 1987).

5.2. Modelling leachate production

To compare costs of the alternatives for leachate treatment, leachate production and quality have to be calculated.

For quantitative estimations, a calculation model is set up, based on the water balance. Starting points are based on data derived from concerned local administrations, from comparable data of other sites (e.g. HOEKS and RYHINER, 1987; VROM, 1984):
- each compartment covers 3.5 ha, and is completed in 3 years, in 3 layers of 9.5 m each;
- consolidation: 25%; thus final height of compartments: 21.3 m;
- dumped volume of waste: 250,000 $t.yr^{-1}$;
- considered period: 15 years (1986-2000);
- final density of waste after consolidation: 1.0 $t.m^{-3}$;
- water storage capacity - after consolidation: S = 0.04 $m^3.m^{-3}$, of which 60% is used in the first year, and 40% in the second year; in the third year (and on) all waste is assumed to be saturated;
- mean runoff on uncapped compartments: O = 100 $mm.yr^{-1}$;
- mean rainfall: F = 750 $mm.yr^{-1}$;
- mean evapo(transpi)ration on barren compartments: E = 250 $mm.yr^{-1}$;
- water, produced by compaction of saturated paper sludge (which is dumped regularly in large quantities) yields P = 3000 $m^3.yr^{-1}$;
- the vegetation cover is planted one year after completion, inducing an increase of evapotranspiration to E = 450 $mm.yr^{-1}$;
- (alt. II and III only:) surface capping is carried out one year after completion of the compartments. Mean runoff: O = 300 $mm.yr^{-1}$;
- (alt. III only:) during 3 years of completion, all leachate of compartment X+1 is recirculated (R) to compartment X; from the 4th year (after completion) this takes place on compartment X+1 itself.

The yearly expected volume of leachate from one compartment can be estimated now, based on the mentioned starting points, with the following equation (symbols: see starting points):

$$T = F + P - E - S - O \quad (m^3.yr^{-1}) \tag{8}$$

where T = total discharge of leachate ($m^3.yr^{-1}$)

Leachate quality improvement can be estimated, by expressing the sum of COD and $4.57N_{kj}$ from eq. (7) as a Total Oxygen Demand.

A mean development of leachate quality in time is derived from local data and data of several sites with comparable waste composition and management (VROM, 1984).

5.3. Results

The total volume of leachate that has to be discharged, has been calculated with the water balance presented by eq. (8). Resulting volumes in each alternative are shown in Table 2. Qualitative data are shown in Table 3.

Table 2. Estimated yearly volumes of leachate to be discharged ($m^3.yr^{-1}$) from one compartment of 3.5 ha, to be completed in 3 years, calculated with eq.(8), for three alternatives:
alternative I: no surface capping; direct discharge of leachate
alternative II: surface capping; no recirculation (direct discharge)
alternative III: surface capping and recirculation (indirect discharge)

Year of operation	Total discharge of leachate ($m^3.yr^{-1}$)		
	alternative I	alternative II	alternative III
1	11,000	11,000	11,000
2	7,000	7,000	7,000
3	7,000	7,000	7,000
4	10,000	10,000	21,000
5	7,000	0	7,000
6	7,000	0	7,000
7	7,000	0	0
8	etc.	etc.	etc.

Now yearly costs for discharge of leachate treatment to the nearby sewage treatment plant can be calculated: volumes of total discharge (Table 2) and Total Oxygen Demand data (Table 3) are combined for the involved compartments and years. Next, these final figures for each year's total quantity and quality are applied in eq. (7). Calculation results of all alternatives are taken together in Table 4.

Table 3. Estimated development of Total Oxygen Demand TOD ($mg.l^{-1}\ O_2$) of discharged leachate effluent from one considered compartment, expressed as the sum of COD and $4.57N_{kj}$; for alternatives I and II (without recirculation) and alternative III (with recirculation)

Year of operation	Total Oxygen Demand ($mg.l^{-1}\ O_2$)	
	alternative I and II	alternative III
1	25,000	1)
2	25,000	1)
3	20,000	1)
4	15,000	2)
5	10,000	5000
6	5,000	4000
7	4,000	3000
8	3,000	2900
9	2,900	2800
10	2,800	2600
11	2,600	2400
12	2,400	2200
13	2,200	2000
14	2,000	1800
15	1,800	1600

1) recirculation takes place on another, earlier completed compartment
2) recirculation with leachate of a next compartment is started (as long as TOD > 5000 $mg.l^{-1}$, recirculation continues on an earlier completed compartment)

Table 4. Calculated yearly costs (Dfl) for discharge of leachate sewage to the sewage treatment plant, for 3 alternative treatments on site (mean costs per period of 3 years (price level 1987)
alternative I: no surface capping; direct discharge of leachate
alternative II: surface capping; no recirculation (direct discharge)
alternative III: surface capping and recirculation (indirect discharge)

Period of operation	Discharge costs ($Dfl.yr^{-1}$)		
	alternative I	alternative II	alternative III
1986-1988	218,000	218,000	218,000
1989-1991	312,000	273,000	101,000
1992-1994	387,000	273,000	38,000
1995-1997	358,500	273,000	38,000
1998-2000	372,500	273,000	38,000
Total over 15 years	4,944,000	3,930,000	1,299,000

From Table 4 it can be concluded that recirculation of leachate is very attractive, because discharge of leachate after a recirculation treatment (alternative III) is substantially cheaper than direct discharge (alternatives I and II).

For a proper costs/benefits analysis, investment and exploitation costs of the recirculation system and surface capping have to be considered too. However, surface capping is advised for a number of reasons (Section 4), of which the environmental aspects (emission of gas and leachate) are the most important ones. Therefore the savings in treatment costs can be considered as an extra benefit. For this reason the costs of the recirculation system are left out of consideration.

If recirculation is carried out, measures have to be taken to ensure uniform distribution of recirculated leachate under the surface cap into the waste and to prevent formation of preferential channels or 'bypasses'. If possible, a combination should be made with a gas extraction system (Section 6).

6. ESTIMATION OF GAS PRODUCTION AND POSSIBILITIES FOR GAS EXTRACTION

6.1. Point of view with respect to Environmental Impact Assessment

Gas production in waste tips is a well known phenomenon, and has been described and studied by many researchers. Succession of fermentation stages was already described by FARQUHAR and ROVERS (1973). The process and its main controlling parameters have been described and studied by HAM and BARLAZ (1987), among others, and need no further explanation here.

For the Linne/Montfort waste disposal site generation of gas is regarded as an interesting, exploitable feature, but it is considered to be less important than leachate management. Methane fermentation should be promoted in order to improve leachate quality (HOEKS, 1983). Recirculation of leachate and surface capping can increase gas production and extraction efficiency (HAM, 1983 among others; and Section 5), besides the already mentioned positive effects on groundwater protection and saving of treatment costs.

6.2. Modelling gas production

An estimation of gas production is calculated with a model developed by HOEKS (1983). The gas production rate can be calculated as:

$$a_t = \sum_{i=1}^{n} 0.8K_iP_{o,i}e^{-K_it} \qquad (9)$$

where a_t = gas production rate at time t ($m^3.t^{-1}.yr^{-1}$)
$P_{o,i}$ = amount of compound i per ton of waste at t = 0 ($kg.t^{-1}$)
K_i = degradation rate coefficient for compound i (yr^{-1})
t = time (year)
n = number of compounds (-)

Cumulative gas production can be found by integrating eq. (9) over time t, yielding:

$$A_t = \sum_{i=1}^{n} 0.8P_{o,i}(1 - e^{-K_it}) \qquad (10)$$

where A_t = cumulative gas production since t = 0 ($m^3.t^{-1}$)

In this approach a first stage of increasing gas production rate is neglected, because in the present case it is experienced that this period is very short.

In this paper only calculations of gas production will be presented for the new, bottom-lined compartments IV and V (Fig. 4). (In the case study the same model has been applicated for the old compartments too. These compartments yield

Table 5. Organic composition and degradation categories of waste types, dumped at the site of Linne/Montfort. Fractions expressed as percentages of weight. Degradation rate is expressed by factor K, as calculated from 'half-life' data: $K = \ln 2/t_{½}$ (derived from HAM et al., 1979, and empirical Dutch data, collected by HOEKS, 1983). Categories are: fast: K = 0.693; medium: K = 0.139; slow: K = 0.046; inert: K = 0

Type of waste	Fraction of total waste (%)	Fraction of organic material (%) in degradation category				Total
		fast	medium	slow	inert	
Household/ municipal	30	12.0	12.0	16.0	13.2	53.2
Industrial	50	0	8.3	16.7	8.3	33.3
Demolition	20	0	0	7.5	2.5	10.0

significantly smaller amounts of gas, mainly because of the high age of the waste. Extraction barely pays there).

The following set of data has been used for calculations (when no reference is given, data are derived from concerned local administrations):
- yearly weight of the waste to be dumped: 250,000 $t.yr^{-1}$. The intensity of dumping is expected to be constant in time;
- composition of waste in types (Table 5);
- fraction of degradable organic material in each type of waste (Table 5);
- degradation rates of organic material (Table 5).

The mean fraction of organic material in the waste as a whole, amounts to approximately 32%. Data from Table 5 now can be applied in eqs. (9) and (10).

6.3. Calculated gas production and gas extraction

Each compartment (Fig. 3) has been subdivided in 4 subcompartments, in order to make an accurate calculation of the production of gas with eq. (10). For calculation purpooses each subcompartment of 1 ha is assumed to be completed in 1 year, after dumping of 250,000 tons of waste. Results are shown in Table 6.

One has to realize that gas extraction can be carried out only after completion and capping of a compartment. The extraction efficiency is estimated then at 70%, and the biogas is assumed to consist of 50% CH_4 and 50% CO_2.

Calculated gas production rates are comparable with other data on gas production, collected by WILLUMSEN (1987).

Calculations show that in the near future a mean yearly extraction from the whole site of approximately 15 million m^3 is possible. As a result of the very diverse composition of the waste (Table 5) and the phased approach in completion, gas production is extended and relatively stable over a long period. Gas extraction is expected to be paying, up to at least 2020.

The total gas production per ton of waste was calculated here at 270 m^3. This amount fits well with literature data ranging from 150-180 $m^3.t^{-1}$ (STEGMANN, 1981) to 520 $m^3.t^{-1}$ (HAM and BARLAZ, 1987).

Table 6. Gas production and extraction ($m^3.yr^{-1}$) in compartments IV and V, calculated with model of HOEKS (1983) at three dates (1987, 1993 and 2000). Each compartment is divided in 4 subcompartments of 1 ha, on which yearly 250,000 tons of waste is dumped (- = no gas production yet)

Year of operation	Compartment		Gas production (million $m^3.yr^{-1}$)		
	in use	completed	1987	1993	2000
1986	IV		5.7	1.8	1.0
1987	IV		8.5	2.0	1.1
1988	IV		-	2.3	1.2
1989	IV + V(1)		-	2.7	1.3
1990	V(1)	IV	-	3.2	1.4
1991	V(1)		-	4.1	1.5
1992	V(1)		-	5.7	1.7
1993	V(2)	V(1)	-	8.5	1.8
1994	V(2)		-	-	2.0
1995	V(2)		-	-	2.3
1996	V(2)		-	-	2.7
1997	V(3)	V(2)	-	-	3.2
1998	V(3)		-	-	4.1
1999	V(3)		-	-	5.7
2000	V(3)		-	-	8.5
Total gas production			14.2	30.3	39.5
Extractable (after completion and capping; estimated efficiency 70%):			-	15.3	12.6

A proper gas extraction system has to be built up already during filling and completion of each compartment. To ensure a proper pass-through through the surface liner material, possibilities for a combination with the leachate recirculation and infiltration system (like suggested by RUDOLPH, 1987) should be examined. Technical details will not be discussed here.

Meanwhile, in 1989 a small power station is built at the Linne/Montfort site, in cooperation with the Provincial Energy Company. One m^3 of methane (CH_4) is expected to yield approximately 2.5 kWh of electricity. In the future, data of extracted volumes of gas can be compared with the predicted volumes as calculated with the described model.

7. CONCLUSIONS

Under specific conditions, such as in the described case, simple methods can be used in assessing environmental impacts effects of control measures at waste disposal sites.

If the geohydrological structure is transformable to homogeneous characteristics, migration of pollutants in groundwater can be described and predicted by using a simple model. Input data for the model are primary data on site-history, geohydrology, meteorology and behaviour of pollutants (adsorption and degradation).

If the general completion strategy is known, expected volumes and quality of leachate, collected from bottom-lined compartments of waste tips - at different management options - can be estimated by simple model calculations. This model uses the water balance principle and empirical data on leachate quality development.

Gas production at waste tips can be estimated by a model, using data on yearly weight of dumped waste, mean waste composition, and degradation rates and fractions of organic material of the various types of waste.

In the here presented case, surface capping is highly advisable, both at site compartments without and with bottom liners, recirculation of leachate in combination with gas extraction facilities is very attractive because of saving of leachate treatment costs and benefits of gas utilization.

REFERENCES

BLAKEY, N.C. and MARIS, P.J. (1987) 'On-site leachate management - anaerobic processes'. Proc. symp. 'Process Technology and Environmental Impact of Sanitary Landfill', 20-23 October 1987, Cagliari, Italy, Vol. I: XXI, pp. 1-15.

BOGAARD, E.A.J.M. VAN DEN and HOEKS, J. (1988) 'Milieu-effecten en milieutechnische maatregelen betreffende de stortplaats Linne/Montfort'. Rapport 25. Institute for Land and Water Management Research (ICW), Wageningen, The Netherlands. 119 pp.

CHRISTENSEN, T.H.; KJELDSEN, P.; LYNGKILDE, J. and TJELL, J.C. (1987) 'Behaviour of leachate pollutants in groundwater'. Proc. symp. 'Process Technology and Environmental Impact of Sanitary Landfill', 20-23 October 1987, Cagliari, Italy, Vol. II: XXXVIII, pp. 1-17.

DOEDENS, H. and K. CORD-LANDWEHR (1987) 'Leachate recirculation in landfills'. Proc. symp. 'Process Technology and Environmental Impact of Sanitary Landfill', 20-23 October 1987, Cagliari, Italy, Vol. I: XVIII, pp. 1-16.

EHRIG, H.J. (1980) 'Beitrag zum quantitativen und qualitativen Wasserhaushalt von Mülldeponien'. Heft 26. Inst. Stadtbauwesen, Technische Universität Braunschweig, West Germany (2. erweiterte Auflage).

FARQUHAR, G.J. and ROVERS, R.A. (1973) 'Gas production during refuse decomposition". Water, Air and Soil Pollution 2, pp. 483-495.

HAM, R.K. (1983) 'Gas production modelling'. In: R.L. Wenthworth (ed.). Proc. 3th Int. symp. on Anaerobic Digestion, Boston. Cambridge, USA, pp. 203-213.

HAM, R.K. and BARLAZ, M.A. (1987) 'Measurement and prediction of landfill gas quality and quantity'. Proc. symp. 'Process Technology and Environmental Impact of Sanitary Landfill', 20-23 October 1987, Cagliari, Italy, Vol. I: VIII, pp. 1-24.

HAM, R.K.; HEKINIAN, K.K.; KATTEN, S.L.; LOCKMAN, W.J.; LOFY, R.J.; McFADDIN, D.E. and DALEY, E.J. (1979) 'Recovery, processing and utilization of gas from sanitary landfills'. EPA-600/2-79-001. US Environmental Protection Agency (Solid and Hazardous Wastes Res. Div.), Cincinatty, USA.

HOEKS, J. (1981) 'Analytical solutions for transport of conservative and nonconservative contaminants in groundwater systems". Water, Air and Soil Pollution 16, pp. 339-350.

HOEKS, J. (1983) 'Significance of biogas production in waste tips'. Waste Management & Research 1, pp. 323-335.

HOEKS, J. and AGELINK, G.J. (1982) 'Hydrological aspects of sealing waste tips with liners and soil covers. Proc. IAHS Symposium, Exeter, UK, July 1982, pp. 157-167. IAHS Publ. 139. Techn. Bull. 14. Institute for Land and Water Management Research (ICW), Wageningen, The Netherlands.

HOEKS, J.; GLAS, H., HOFKAMP, J. and RYHINER, A.H. (1987) 'Bentonite liners for isolation of waste disposal sites'. Waste Management & Research 5, pp. 93-105.

HOEKS, J. and RYHINER, A.H. (1987) 'Surface capping of waste disposal sites with natural liner materials'. Proc. Symp. 'Proces Technology and Environmental Impact of Sanitary Landfill', 20-23 October 1987, Cagliari, Italy, Vol. II: XXVIII, pp. 1-10. Report 19. Institute for Land and Water Management Research (ICW), Wageningen, The Netherlands.

RUDOLPH, K.U. (1987) 'Deponie-Sickerwasserbehandlung in Gasdomen; eine preisgünstige Möglichkeit der Vorreinigung'. Müll und Abfall 1, pp. 25-29.

STEGMANN, R. (1981) 'Beschreibung eines Verfahrens zur Untersuchung anaerober Umsetzungsprocesse von festen Abfallstoffen im Labormassstab'. Müll und Abfall 2, pp. 35-39.

VROM (1984) 'Opvang en behandeling van percolatiewater van afvalstortterreinen'. Reeks Bodembescherming nr. 35, Staatsuitgeverij, The Hague, The Netherlands.

WILLUMSEN, H.C. (1987) 'Landfill gas utilization, especially optimalization at smalllandfills". Proc. Symp. 'Proces Technology and Environmental Impact of Sanitary Landfill', 20-23 October 1987, Cagliari, Italy, Vol. I: XI, pp. 1-14.