Treatment of landfill leachate – high tech or low tech? A case study

N. Schwarzenbeck, K. Leonhard and P.A. Wilderer
Institute of Water Quality Control and Waste Management, Technical University of Munich,
Am Coulombwall, D – 85748 Garching, Germany (E-mail: *N.Schwarzenbeck@bv.tum.de*)

Abstract At the sanitary landfill of the city of Penzberg (Germany), two diverse approaches to leachate treatment were studied as parts of a three-stage treatment concept. The performance of a simple aerobic pond was compared to that of an advanced multistage treatment unit, the latter comprising a membrane biological reactor and a two-stage activated carbon filter. For 274 days of the year (75%) the pond was able to provide sufficient treatment even under cold weather conditions. For temperatures lower than 5°C, a higher biomass content and temporal storage of the raw leachate (e.g. increasing hydraulic retention time) could close the gap of insufficient treatment. In contrast, the advanced treatment system could only accomplish limited treatment capabilities due to insufficient maintenance, low loading conditions and deficient coordination between the individual treatment steps. As a result, degradation rates were low and operational problems frequent. Limits for N_{tot} were exceeded regularly ($N_{tot,e}$ = 60–70 mg/L), throughput broke down and excessive nitrite production occurred (NO_2-N_e = 10 mg/L) as a result of microbial activity inside the activated carbon filters. This case study clearly suggests aerobic ponds as an appropriate solution for the treatment of landfill leachate in areas where operational independence is essential.
Keywords Activated carbon; aerobic pond; decentralized treatment; landfill leachate; membrane biological reactor

Water Science and Technology **Vol 48 No 11–12 pp 277–284** © IWA Publishing 2003

Introduction

Landfills are typically located in peripheral areas and are enduring sources of emissions of complex composition and often of toxic nature. Whereas landfill gas represents a valuable source of energy, landfill leachate has to be treated before being discharged into surface waters. Consequently, sophisticated and highly independent treatment is demanded in regions where usually space is widely available but sufficient infrastructure is lacking. Hazardous compounds of the leachate (e.g. heavy metals, adsorbable organic halogens (AOX), polyaromatic hydrocarbons (PAH), polychlorinated biphenyls (PCB) or dioxins) usually require physico-chemical treatment. Organic compounds (COD), nutrients (N, P) and even some hazardous compounds (e.g. AOX) can up to a certain level also be removed by means of biological treatment approaches. There is a wide range of different approaches and technologies available for the treatment of special wastewaters. These treatment needs can be satisfied in two distinct ways. High degradation capacities can either be installed by means of complex plants and process automation, which is followed by significant maintenance needs on-site. Or reaction volumes and higher hydraulic retention times are required as a result of lower degradation efficiencies. Therefore these approaches are less demanding concerning maintenance and process control. At the municipal waste landfill of the city of Penzberg (Germany), both approaches could be studied at the same time as part of a decentralized treatment concept.

Local situation

History of the landfill

The municipal waste landfill at Penzberg-Weitfilz (Germany) was in operation from 1959 until 1985. In the early 1990s it was closed and covered with a surface sealing. Over the total period of operation, approximately 480,000 m^3 of municipal and partially industrial waste as well as rubble were dumped at a rate of 20,000 m^3/a. The total surface area covers about 6 ha; the mean filling height is 13 m above level. The base does not have an engineered barrier, but stands on a layer of peat (6 m) and a layer of lacustrine clay (3 m). The surrounding soil is bordered against the landfill by means of a sealed sheet piling. Leachate is drained through a layer of gravel (40 cm) between waste and sheet piling. On parts of the surface area, a composting plant is being operated, whose process water originally was collected together with the leachate, meanwhile it is wasted separately. Leachate could originally be recirculated into the landfill through a re-infiltration shaft at the highest elevation of the landfill. Thus the biological processes inside the waste could be used for the treatment of the leachate and the impact of shock loading on the treatment unit could be reduced. As a result water gauges today indicate an impoundage of 6 m above level.

Leachate yield and flow regime characteristics

Due to the extensive operation in an uncovered state and due to the long-term recirculation of leachate, a water saturation of the landfill body occurred. As a result of the recirculation also the transition into the stable methanogenic phase was accelerated. Together this results in comparably low pollutant concentrations in the leachate and a direct interrelation between precipitation and leachate yield. The mean leachate flow is 58 m^3/d corresponding to a leachate yield of 9.2 m^3/(ha·d). Looking at the dry- and the storm weather flow (Table 1) we can indeed find values that were found to be typical for landfills saturated with water (Ehrig, 1989; Ramke, 1991; Kruse, 1994). The BOD_5/COD-ratio in the leachate is equal to 0.2 und thus typical for landfills of a stable methanogenic phase (Ehrig, 1989; Kruse, 1994). Table 1 gives an overview of the relevant pollutants and parameters compared to data collected in other long term-studies (20–30 years).

Future development of the pollutant concentrations

Only few measured values were available for the current pollutant concentrations in the leachate in the current situation (4 values/pollutant and year). Furthermore these values showed a high fluctuation (Table 1) and thus could not be used as a significant basis for the evaluation of the potential treatment capabilities of the single treatment steps in the future. The future evolution of the pollutant concentrations was hence assessed by means of mathematical modeling. The long-term development of pollutant concentrations (Eq. (1)) in the leachate can be described with a model based on the kinetics of exponential decay (Kruse, 1994).

This approach assumes the development of a pollutant-concentration to be dependent on the properties of the waste, the amount of water available for the transport, and the time passed since operation of the landfill was started.

Table 1 Pollutant concentrations and leachate yield of the landfill in 1999

	COD	BOD_5	TKN	AOX	Yield	
					Dry weather	Storm weather
	mg/L	mg/L	mg/L	µg/L	m^3/(ha·d)	m^3/(ha·d)
(Ehrig, 1989)	500–4500	20–550	50–5000	524–2010	8.10	16.70
(Kruse, 1994)	460–8300	20–700	250–2000	195–3500	10.80	38.00
(Ramke, 1991)	–	–	–	–	9.40	81.90
Penzberg	136–1980*	22–127*	73–130*	150–330*	10.28	39.54

* Considerable fluctuations due to a limited number of data points (4/year) that were available for 1999

$$C_A(t) = m_{0,A} \cdot e^{-kt} \cdot y \cdot h/q \quad (1)$$

$C_A(t)$ evolution of the concentration of pollutant A over the time of landfill operation [mg A/L]
$m_{0,A}$ initial rate of mobilization for pollutant A [mg A/(kgTS·d)]
k constant of decay [–]
y density of the waste [kgTS/m^3]
h filling height of the landfill [m]
q leachate yield [L/m^2·d]

In most cases of closed landfills, parameters describing the properties of the waste are scarce and difficult to be measured ex-post. Transforming Eq. (1) and introducing a single constant for the waste properties allows the modeling of the development of pollutant concentrations based on data describing the concentration changes in the past only (Eq. (2)).

$$C_X(t) = b \cdot e^{-kt} \quad (2)$$

b waste property constant [mg/L]

This procedure allows an estimation of the mean concentration level to be expected currently, the range of variation and the total time elapsing until concentrations will fall below the current discharge limits (maintenance time). In addition, properties of the dumped waste can be calculated and compared to values from other studies in order to confirm the results of the curve fitting. Underlying the leachate flow regime of the last year, an assessment basis for the design of a treatment unit can further be derived. For the evaluation only organic compounds (COD and BOD_5) and nitrogen (N_{tot}, NH_4-N) were considered since all of the other relevant pollutants (i.e. P and AOX) have at present already fallen below discharge limits. The total mass of COD and N, that can potentially be mobilized, is in the present case comparably high where the rate of mobilization is lower than at comparable landfills (Kruse, 1994). Table 2 gives an overview of the results of the mathematical modeling.

Leachate treatment

Plant configuration. Until 1996 an aerobic pond was used as a single stage treatment unit for the leachate and the surface runoff coming from the composting plant. In 1996 the construction of an advanced treatment system was decided on. The new treatment concept includes the aerobic pond as a first stage for uncontrolled pre-treatment and equalization of the leachate flow. The newly built plant consists of a two-stage membrane biological reactor for denitrification and nitrification followed by a two-stage activated carbon filter. The entire plant is insulated, and equipped with heat tracing and fully automated process control.

Methanol is added as an external C-source for the denitrification; dihydrogenphosphate

Table 2 Overview of the future development of pollutant concentrations in the leachate

		COD	BOD_5	N_{tot}	NH_4-N
Average	mg/L	280	90	130	120
Range	mg/L	70–800	80–120	60–250	70–190
Discharge limit	mg/L	200	20	70	18
Maintenance time	a	20	30	35	60

was added for optimum biomass growth, hypochloric acid and sodium hydroxide were added for pH-control.

Performance of the aerobic pond

Within the scope of regular control of the pond operation, detailed data were acquired every fortnight over a period of 9 years (1988–1997) under varying environmental and loading conditions. Thus a realistic picture could be drawn of the possible treatment capabilities. The pond includes a total operating volume of 2,000 m^3 at a surface area of 1,200 m^2 and a water depth of 1–2 m. Surface aerators provide oxygen at a rate of 40 to 60 kgO_2/d and intensive mixing of the pond content.

Influence of temperature. Temperature has a major impact on the degradation processes in the pond, especially on the nitrification. Due to its exposed position and low water depth, variations in ambient temperature are directly reflected in the water temperature (average 13°C). In the present case a direct interrelation between temperature and nitrification efficiency can be observed (Figure 1) and determined (Table 3). At decreasing temperatures nitrification shows sufficient buffer capacities of one month until it breaks down. For rising temperatures nitrification simultaneously recovers without delay (Figure 1). For further calculations the dependence of reaction rates on the temperature was quantified using the Arrhenius-equation (Eq. (3), normalized to 15°C).

$$\mu(T) = \mu \cdot \theta^{(T-15)} \tag{3}$$

$\mu(T)$	growth rate at temperature T [1/d]
T	temperature [°C]
θ	characteristic temperature coefficient [–]

Degradation capacities. Assessing the performance of the aerobic pond as a single stage treatment unit required further evaluation of the data with respect to the degradation capacities. Reaction rates were calculated from a linear curve fitting, using the Lineweaver-Burk plot. In accordance with Tchobanoglous and Schroeder (1987) a continuously stirred tank reactor was chosen as a sufficient model of the pond. Reaction rates and constants were calculated from mass balances (Table 3).

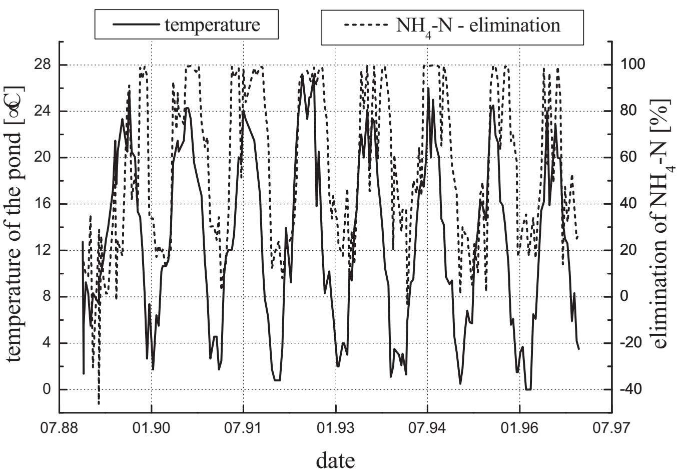

Figure 1 Nitrification efficiency and temperature in the pond

Table 3 Reaction rates and temperature coefficients in the pond

	$r_{SV,max}$ mg/(m³·d)	K_S mg/L	$r_{SX,max}$ mg/(gSS·d)	K_S mg/L	θ –
COD	3260	55	66	410	1.0436
BOD_5	7313	110	–	–	1.0952
Nitrification	3116	33	52	29	1.0572
Denitrification	2246	19	42	21	1.0509

In the next step degradation capacities and temperature coefficients were merged and compared to different scenarios of pollutant loading and hydraulic loading. These loading scenarios result from the different levels of pollutant concentrations as given in Table 2 and the different levels of leachate flow as indicated in Table 1. Superposing the results for the different relevant degradation processes gives clear information under which conditions sufficient treatment capacities can be provided by the pond. The integrated projection is depicted in Figure 2.

The diagram indicates sufficient treatment capacities only for temperatures higher than 5°C. On average, ambient temperature is higher than 5°C on 75% of the days of a year. Mainly nitrification is limiting due to the low discharge limit and the high temperature dependency. These deficiencies could on the other hand be counter-balanced by installing higher biomass content in the pond and increasing hydraulic retention time. In terms of practical application this would mean a temporal storage of the raw leachate, operating the pond in a sequencing batch mode over the winter months, and introducing a carrier material for additional immobilization of biomass in the pond (biofilm growth).

Performance of the advanced treatment concept – biological reactor and activated carbon

Each of the two biological reactors (denitrification and nitrification) has a working volume of 66 m³. The first reactor is equipped with a stirrer; the second reactor is equipped with fine bubble aeration. Biomass is completely retained in the system by means of a microfiltration membrane (pore size 0.1 μm). The activated carbon filter comprises two columns of 11 m³ volume each and was originally included for the removal of AOX and as a safety step for COD-removal. AOX-concentration in the raw leachate and COD in the effluent of the biological reactors are already below the discharge limits.

Influence of basic process conditions. The temperature in the reactors is highly dependent on the ambient temperature although the plant is insulated and equipped with heat tracing

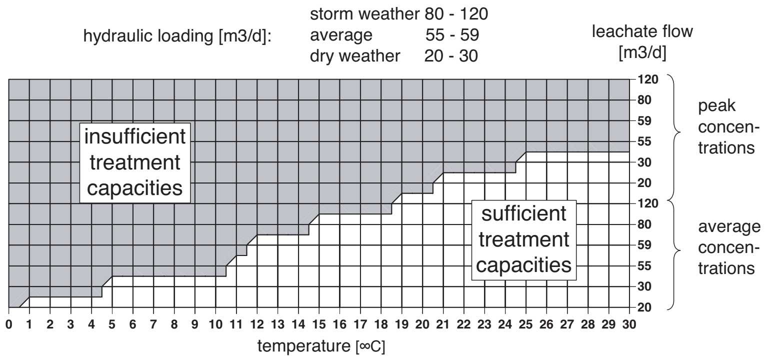

Figure 2 Overall treatment capacities of the pond (superposition of COD- and N-removal)

and heat exchangers. Between summer and winter the temperature in the reactors differs about ±20°C (Figure 3) and thus in the same range as for the pond. Automated pH-control adjusted the pH in the reactors to around pH 7. As a result the microorganisms showed a high level of sensitivity towards deviations from the optimum pH. After pH-control broke down in February 1999, pH increased significantly and nitrification was not recovered until one year had passed (Figures 3 and 4). The sensitivity of the biomass towards pH deterioration was also clearly reflected in the MLSS content. It decreased by 40% after breakdown of the pH-control. The mean biomass content was maintained at 13 gSS/L (8.5 gVSS/L), excess sludge was mainly wasted “accidentally”. The plant is altogether operated at a very high sludge age with simultaneous aerobic sludge stabilization. The latter results in a high demand of energy for aeration.

During operation, the microfiltration unit clearly had a limiting effect on the throughput of the total system (Q_d = 60 m^3/d). Membranes were blocked frequently due to scaling and biofouling until an improved cleaning in place improved the flux. A sufficient level of internal recirculation could be provided, ensuring a theoretical N-removal of 80%.

Removal of COD and NH_4-N. The biological reactors provide sufficient treatment capacities in order to meet the discharge limits all-the-year. COD and BOD_5 are removed at average efficiencies of 65% and 95% respectively. Apart from limited nitrification capacities due to pH-deterioration (January 1999–January 2000), NH_4-N was removed at 97% efficiency. Figure 4 shows the development of pollutant concentrations in the influent and effluent of the biological reactors and the activated carbon filter over the course of operation.

Removal of nitrate and nitrite in the biological reactors. Over the whole period of operation severe problems occurred concerning the removal of nitrate. Nitrate could only be removed at an average efficiency of 22% in the biological reactors (Figure 5 left). The limitations in degradation capacity occurred mainly due to a limitation in biodegradable organic carbon, a very high sludge age and the construction conditioned presence of dissolved oxygen in the denitrifying reactor.

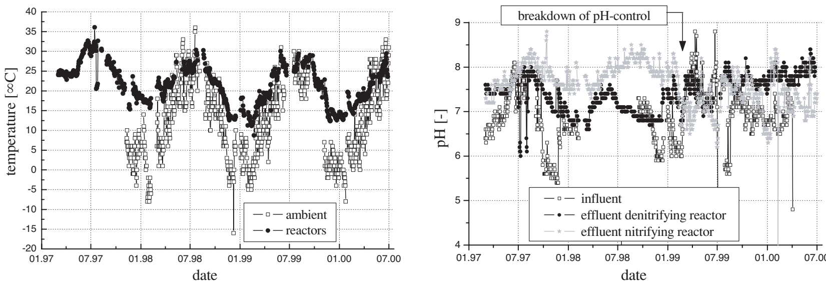

Figure 3 Temperature and pH inside the biological reactors over the course of operation

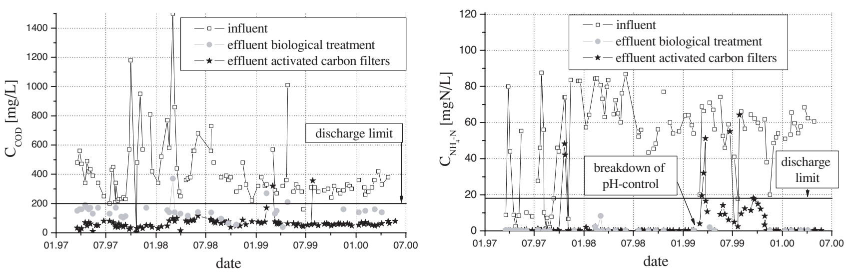

Figure 4 COD and NH_4-N-removal over the course of operation

Denitrification in the activated carbon filters. Denitrification was observed at the activated carbon filters as a result of microbial growth inside the activated carbon filters (see below). Due to a limited availability of biodegradable organic carbon in the influent of the activated carbon filters (= effluent of the biological reactors), only incomplete denitrification occurred, accompanied by the production of nitrite (Figure 5 right).

Activated carbon filters – microbial activity. The activated carbon filters are still running with the first filling. In the effluent of the activated carbon filters the COD does not fall below an equilibrium concentration of 65–65 mg/L. Mass-balancing gives a total of 6,500 kg of COD, that has up to now been removed from the biologically treated waste-water by the activated carbon. At the given equilibrium concentration this would sum up to a carbon consumption of 65 m^3 up to now. This means a volume three times bigger than the existing volume (22 m^3). Together with the nitrite production this clearly indicates the growth of microbial biofilms inside the activated carbon. Furthermore a continuous regen-eration of the activated carbon filters can be assumed and these therefore be regarded as rather efficient continuous flow biofilm reactors giving relevant contribution to the overall removal capacities of the system.

Overall treatment capacities. Reaction rates were calculated for the biological reactors as well as for the activated carbon filters, since significant microbiological activity was found in the activated carbon filters. Table 4 gives an overview and shows the limited treatment capacities of the aerobic pond together with the biological reactors in comparison with the conversion rates of the activated carbon filters. In the current state of operation the activat-ed carbon filters provide higher denitrification capacities than the biological reactors. As a result they contribute significantly to the overall treatment efficiency of the plant.

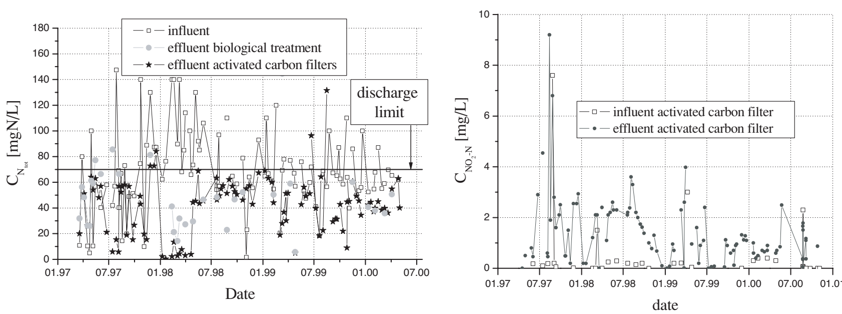

Figure 5 N_{tot}-removal at the biological reactors (left) and NO_2-N-production at the activated carbon filters (right) over the course of operation

Table 4 Reaction rates in the biological reactors and in the activated carbon filters

	Aerobic pond	Biological treatment		Activated carbon filter
	$r_{SV,max}$ g/(m^3·d)	$r_{SV,max}$ g/(m^3·d)	$r_{SX,max}$ g/(gSS·d)	$r_{SV,max}$ g/(m^3·d)
COD	4	452	34	250[A)]
BOD_5	7	78	6	–
Nitrification	6	74	6	–
Denitrification	2	24	2	32

A) Reaction rate estimated from the potential of regeneration

Conclusions

- Aerobic ponds, even under cold climatic conditions, can sufficiently treat leachate from a municipal waste landfill. This applies particularly to cases, where a high level of independence from regular maintenance on site is required.
- Leachate from a municipal waste landfill does not necessarily require highly engineered treatment plants in order to meet the discharge limits.
- Highly advanced treatment approaches show worse treatment efficiencies than low tech approaches if not chosen appropriately and maintained regularly.
- Excessive process control creates a spoiled microbial community, which is rather sensitive to deteriorations from the optimum conditions (e.g. pH-deteriorations).
- Biofilm growth in activated carbon filters can result in significant treatment capacities and a continuous regeneration of the activated carbon filling.

Acknowledgements

We thank Mr. Strauss and Mr. Christ from the Landratsamt Weilheim-Schongau for the excellent cooperation and support during the investigations.

References

Ehrig, H.J. (1989). Sickerwasser aus Hausmülldeponien – Menge und Zusammensetzung. In: Müll- und Abfallhandbuch, Loseblattsammlung, Lfg. 1/89, Kennzahl 4587(1989).

Kruse, K. (1994). *Langfristiges Emissionsgeschehen von Siedlungsabfalldeponien*, PhD thesis, Institute of Sanitary and Environmental Engineering, Technische Universität Braunschweig, Germany.

Ramke, H.-G. (1991). *Hydraulische Beurteilung und Dimensionierung der Basisentwässerung von Deponien fester Siedlungsabfälle – Wasserhaushalt, hydraulische Kennwerte, Berechnungsverfahren*, PhD thesis, Leichtweiss Institute for hydraulic engineering, Technische Universität Braunschweig, Germany.

Tchobanoglous, G. and Schroeder, E. (1987). *Water Quality*. Addison Wesley Publishing Company, Reading-Massachusetts.