

FULL-SCALE PERFORMANCE OF BIOLOGICAL LEACHATE TREATMENT AT LOW TEMPERATURE

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Abstract

This paper deals with biological leachate treatment in an activated sludge system with nitrogen removal (pre-denitrification). The achieved removal efficiencies were 68 % (COD), 95 % (BOD₇), 99 % (NH₄-N) and 55 % (N_{tot}) at temperatures 4.7-15.5 °C. The process was operated for several months at 4.7-6.2 °C and the lowest monthly temperature was 4.7-5.3 °C. A stable nitrification was found even at these extreme conditions, obviously the first time in a leachate treatment plant. The denitrification efficiency was reasonable. COD removal was affected by the decreased influent COD concentration. AOX removal varied between 10-50 % and was affected by the low temperature and by the changes in the biodegradable fraction. A comparison was made to estimate the eutrophication and GWP (global warming potential) impacts of two treatment systems. The study shows that the application of biological processes in leachate treatment at low temperatures is worth considering and the feasible temperature range can be extended down to 4.5-5 °C.

Keywords: landfill leachate, biological treatment, nitrogen removal, AOX removal, low temperature, costs, emissions

1. Introduction

Local biological treatment of leachate has been applied for over 10 years to a great extent, for example, in Germany (Kollbach *et al.* 1993), where the number of this type of plant is the highest in Europe. In general the local treatment of leachate is becoming more popular. Leachate consists of several compounds, such as COD, BOD, nitrogen, heavy metals and some organic compounds, which can be removed partially or to a rather high extent in a biological process. But the removal of salts cannot be managed in a biological way.

The applicability of biological treatment depends on many factors, which may be site specific. This means that such variables must be studied and taken into consideration in designing or upgrading the operation of the treatment process. In Nordic conditions, the water temperature can be rather low and this can be a limiting factor. In 1997-8 a pilot-scale study was made (Pelkonen *et al.* 1997, 1999a, 1999b, Kotro 1998) to determine the design values in the expected operational range and to make cost estimates.

The aim of this paper is to report the results from a full-scale treatment plant and to examine the impact of different factors on the removal of some compounds. The aim is also to evaluate the economic and environmental feasibility of local leachate treatment in Nordic conditions and describe the possible benefits in this type of treatment.

2. Materials and methods

The experiments were conducted at the landfill of the Nurmijärvi municipality, which is situated in the southern part of Finland. During the reported period the age of landfill was seven years, the filling height 7-8 m and area 2.5 hectares. Water from a windrow composting area (1 ha) was collected and treated together with the leachate. For more details see Pelkonen *et al.* (1999b).

In autumn 1999, a full-scale activated sludge process was taken into operation in Nurmijärvi. The design of the process was based on pilot-scale tests and consists of an activated sludge process including denitrification and nitrification stages for nitrogen removal. Phosphorus addition is needed in the process. Special care was taken in design and construction of the plant to maintain a sufficient temperature in winter conditions as well. The average annual design flow was $50 \text{ m}^3\text{d}^{-1}$ and the maximum daily flow $90\text{-}100 \text{ m}^3\text{d}^{-1}$. The process was started in November and the loading was increased slowly step by step. In February the annual average flow was achieved and also exceeded. The sludge retention time (SRT) was over 40 days. The other operation parameters applied and analysis methods used can be found in Pelkonen *et al.* (1999b). The heavy metals were analysed with ICP-MS or ICP-AES (inductively coupled plasma - mass spectroscopy or - atomic emission spectroscopy) and AOX according to ISO 9562 (1989).

3. Results and discussion

3.1 Removal of nitrogen and organic matter

The influent and effluent concentrations of nitrogen compounds, BOD and COD are shown in Table 1 and the corresponding removal efficiencies in Table 2.

Table 1. Influent and effluent concentrations of the treatment plant (winter and spring period in 2000)

	INFLUENT					EFFLUENT				
	Ave	std	min	max	n	ave	std	min	max	n
COD _{tot}	625	109	500	820	10	195	28	150	250	10
COD _{fil}	509	72	380	600	8	167	27	140	220	10
BOD ₇	222	46	130	290	8	10.9	3.8	6	16	8
N _{tot}	93.3	24.3	66	148	8	42.7	13.3	32	71	9
NH ₄ -N	86.3	26.1	62	154	11	0.91	2.2	0.1	7.4	11
NO ₂ +NO ₃ -N						42.1	24.2	19.7	105	11
PO ₄ -P						0.86	0.97	0.09	2.51	12
Process parameters										
Temperature	8.8	4.2	4.7	15.5	11					
HRT	3.1	1.9	1.5	8.3	11					
BOD ₇ /N	2.7	0.8	1.4	3.6	5					
COD _{biod} /N	4.7	1.0	3.0	6.0	7					

Units mg l^{-1} , except temperature ($^{\circ}\text{C}$) and HRT (hydraulic retention time, days). n = number of observations. COD_{biod} denotes biodegradable COD (Pelkonen *et al.* 1999b).

Table 2. Removal efficiencies of the full-scale treatment plant (period as in Table 1).

	Ave	Std	min	max	n
COD _{tot}	68.5	3.6	65.4	75.3	10
COD _{fil}	66.2	3.7	60.9	70.2	8
BOD ₇	95.1	1.3	93.2	96.8	7
NH ₄ -N	98.7	3.2	89.6	99.9	10
N _{tot}	54.8	5.3	50.2	63.8	7

The removal efficiency of organic matter was on average 68 % for COD and 95 % for BOD₇. The effluent BOD was low and it can be assumed that the residual organic matter was non biodegradable, on average 170 mg/l in soluble form. Compared with the results found earlier on a pilot scale, the influent COD concentrations and reductions in the process were lower on full scale. This was due to some dilution of waters (melting), but also due to the observed decrease of organic matter concentrations in water coming directly from the landfill bottom drainage layer. This is an indication of the ageing of the landfill and the change towards an obvious increase in gas production and decrease of organics in the water phase.

The results show a nearly complete nitrification, and the ammonia removal was on average over 98 %. The process temperature was on average 5.3 °C for over three months (4.7-6.2 °C) and the lowest monthly process temperature was 4.7-5.3 °C, which can be characterised as extremely low. The specific nitrification rate was on average 0.014 g NH₄-N (g VSS)⁻¹ d⁻¹ (range 0.009-0.022). This did not considerably differ from the pilot tests, in which the temperature range was approx. 7–11 °C, in spite of the high temperature impact on nitrification (12 % °C⁻¹, determined in pilot studies). These results support extending the feasible temperature range for nitrogen removal down to 4.5-5 °C. An important factor to take into consideration is the possible inhibition of nitrification due to heavy metals or other toxic compounds, which can have a synergetic effect with the decreasing temperature (Randall and Buth 1984). The results indicate that, at least in this case, this phenomenon did not have a significant effect. As a result, it is obviously the first time that a full and stable nitrification has been reported in a leachate treatment process in this temperature range on full scale.

A reasonable denitrification was found-on average the total nitrogen removal was approx. 55 %. Because of the lower concentration of organic matter in the leachate, the BOD₇/N-ratio was rather low, 2.7 on average, and the COD_{biod}/N-ratio was 4.7. This affected, to some extent also the denitrification efficiency, but it was also affected by some operational parameters, which were not optimised (i.e. internal flow rates). With the observed COD/N-ratio, the nitrogen removal efficiency has potential of 65-70 % nitrogen removal, if the process is optimised. However the decreasing COD/N-ratio will affect the process performance in the long term.

3.2 Removal of metals and organohalogen compounds (AOX)

The influent and effluent concentrations of metals and AOX are shown in Table 3 and the removal efficiencies in Table 4.

Table 3. Metal and AOX concentrations in influent and effluent leachate

INFLUENT		EFFLUENT							
		ave	std	min	max	ave	std	min	max
Al	μgl^{-1}	213	209	79	695	62	54	31	193
Cd	μgl^{-1}	0.28	0.17	0.13	0.50	0.09	0.06	0.05	0.22
Cu	μgl^{-1}	20.0	18.0	4.1	45.6	4.0	3.8	1.1	9.2
Pb	μgl^{-1}	5.6	3.2	2.4	12.4	1.0	0.7	0.4	2.7
Ni	mg l^{-1}	0.067	0.039	0.032	0.150	0.033	0.008	0.027	0.050
Zn	mg l^{-1}	0.13	0.06	0.09	0.28	0.05	0.05	0.03	0.16
Fe	mg l^{-1}	14.4	6.6	9.9	30.4	1.1	0.5	0.5	2.0
Mn	mg l^{-1}	1.16	0.21	0.96	1.45	0.91	0.30	0.59	1.34
Mg	mg l^{-1}	76	22	53	120	73	18	50	107
Ca	mg l^{-1}	122	22	98	152	117	15	98	142
Na	mg l^{-1}	245	88	152	417	234	73	149	382
K	mg l^{-1}	131	77	65	289	118	57	62	233
AOX	$\mu\text{g Cl l}^{-1}$	292	129	170	557	215	40	160	283

Table 4. Removal efficiencies (%) of metals and AOX.

	Al	Cd	Cu	Pb	Ni	Zn	Fe	Mg	Mn	Na	Ca	K	AOX
ave	66	64	80	80	42	69	91	4	22	3	3	6	20
std	11	10	8	9	19	11	4	5	19	5	9	9	17
min	54	56	74	67	16	43	86	-4	0	-4	-14	-8	1
Max	81	83	91	96	67	79	96	11	54	13	16	19	49

Number of observations 4 (Cu) and 7-8 (others), results below detection limits omitted, for AOX n=9

The influent metal concentrations here are less than 40 % of the earlier pilot test values, which also indicates the changing phase of the landfill. Also the removal efficiencies found here are 5-30 %-units lower (Fe, Zn, Pb, Al) or slightly higher (Ni). No clear impact of temperature on metal and solids removal could be found. Therefore the lower treatment efficiency in this respect seems to be mainly due to the different influent concentrations. Even these removals for Fe, Zn, Pb, Cu, Cd and Al were reasonable high (60-90 %), but the removal of nickel was lower (42 %). As a result, heavy metals do not seem to be a problem in biological leachate treatment.

The removal of organic halogen compounds in the form of AOX as a function of temperature is shown in Fig. 1. Some additional data has also taken from earlier pilot-scale results.

A clear impact of temperature can be seen. However, the impact of wastewater characteristics cannot be ignored, because during the highest AOX-removals (over 45 %) the influent AOX-concentration was higher (0.49-0.56 mg/l) compared to the other cases (0.17-0.30 mg/l). This was also the case in the pilot results. During the observation period, the influent flow and strength could be controlled to some extent, but not completely, in order to take into account the whole range.

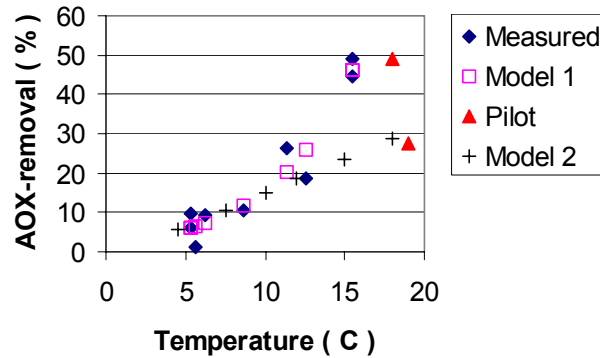


Fig 1. The removal of AOX at different temperatures. Pilot means earlier results from pilot scale; model fits - see text.

Therefore two different models were fitted to the data:

- model 1 in which all full-scale data is included
- model 2 in which results from influent concentrations below 0.3 mg/l are included. (also from pilot scale)

The equations used in the models were:

$$\begin{aligned} \text{AOX}_{r,T} &= R_{1,15} \theta_G^{(T-15)} && \text{(model 1)} \\ \text{AOX}_{r,T} &= R_{1,15} + R_{1,15} \theta_A (T-15) && \text{(model 2), where:} \end{aligned}$$

$\text{AOX}_{r,T}$ = AOX-removal efficiency at temperature T (%)

$R_{1,15}$ = AOX-removal efficiency at 15 °C (%)

θ_G = geometric temperature coefficient (°C⁻¹)

θ_A = arithmetic temperature coefficient (°C⁻¹)

The estimation results are shown in Table 5. The best fits of data are shown. The first model (shown in Fig. 1) fits well with the data ($R^2 = 0.94$), but the temperature coefficient is high ($\theta_G = 1.22$), which does not seem to be realistic because the impact of the change in influent concentrations is included in the coefficient. The second model gives a more realistic description and the temperature coefficient ($\theta_A = 0.073$) is in the expected range.

A model was constructed with multiple regression analysis to evaluate factors affecting AOX removal efficiency in more detail. The key variables were influent AOX concentration, temperature and amount of biomass in the process (MLVSS). Of these, the biomass concentration had the lowest impact. The change of influent AOX concentration was related to the biodegradable fraction of organohalogen compounds. The results indicate that this fraction can vary and the obvious range for that fraction is 25-50 %. This corresponds with the fraction of AOX, which can be removed (degraded) at a temperature of approximately 15 °C or higher. For comparison, in one example from Germany (Ehrig 2000), the biologically removed fraction of AOX was approx. 30-75 %, which is higher than found here. The temperature impacts on AOX removal shown in model 2 and in the multiple regression model were close, and not very far from the results found from AOX removal rates (data not shown).

It is obvious that the evaluation of the biodegradable fraction requires more effort to determine the local influent water and process characteristics that affect the organohalogen degradation and removal. It is interesting to note that the AOX removal has been connected to sol-

ids removal due to adsorption on sludge (Theilen 1995). Our experience from pilot scale does not support this.

Table 5. The estimation results of models 1 and 2

Model	std error of estimate	
Model 1 (influent AOX up to 0.56 mg/l)		
N	9	-
$R_{1,15}$	21.3	2.9
θ_G	1.22	0.034
R^2	0.937	
Model 2 (infl. AOX 0.17-0.3 mg/l)		
N	8	-
$R_{1,15}$	23.6	2.7
θ_A	0.073	0.011
R^2	0.74	

n = number of observations, R^2 = explained variance by the model (models 1, 2 and parameters, see text)

3.3 Other emissions

The effluent phosphate concentration was on average 0.86 mg l^{-1} . This is connected to the control of phosphorus feed, and some observations over 1 mg l^{-1} were found. In most cases, the concentration was below 0.5 mg l^{-1} , which can be seen as an achievable result in the long term.

The sludge production was $0.25\text{-}0.3 \text{ kg SS (kg COD removed)}^{-1}$. This depends to some extent on the sludge removal rate (SRT). The produced sludge is a secondary problem, which has to be considered. The sludge can be composted and converted to utilisable humic material. It can be estimated that leachates during a 15-year period, since the landfilling of waste, produce sludge that is 10 - 30 % of the total amount needed for the growth layer on the landfill surface.

N_2O is a nitrogen compound, which can be formed in the biological treatment system. During this experimental period it was not measured. The approximate emission of $\text{N}_2\text{O-N}$ can be estimated from literature values. In a survey made in German municipal wastewater treatment plants, most including nitrification and rather many denitrification, the average $\text{N}_2\text{O-N}$ emission was 0.6 % of the influent nitrogen load. The emissions include also the effluent water gaseous compounds (Wicht and Beier, 1995). Okayasu *et al.* (1997) observed in a laboratory-scale activated sludge system (denitrification-nitrification), treating wastewater of high $\text{NH}_4\text{-N}$ concentration, that the $\text{N}_2\text{O-N}$ emission in the gas phase was almost always less than 1 % of the removed nitrogen, when the aeration basin oxygen concentration was sufficient ($> 3 \text{ mg/l}$). These results are in good agreement, and as an estimate 0.6 % conversion of influent nitrogen to $\text{N}_2\text{O-N}$ can be used.

The energy consumption in the process has an important role from economical and environmental point of view, and the emissions of energy production must be considered. The total energy consumption in this biological process was 1.8 kWh m^3 treated. This consisted of approximately 50 % of aeration (compressor) and the rest 50 % of pumping, mixing and heating

of the technical room. It can be assumed that the specific energy consumption decreases when the plant size increases. The treatment plant size in this examination is rather small.

3.4 Costs

The costs in a $50 \text{ m}^3 \text{ d}^{-1}$ average flow ($100 \text{ m}^3 \text{ d}^{-1}$ maximum daily flow) were approximately FIM 1 M for investments and FIM 3 - 4 per m^3 operating costs. Total costs per treated leachate m^3 are approximately FIM 9 (1.5 euros). The cost estimate made earlier (Pelkonen *et al.* 1999b) is close to this value.

4. Comparison with a reference system

A comparison was made to evaluate the impact of increased energy consumption on emissions. As a reference, treatment a system was applied in which the energy consumption was 20 kWh m^{-3} (approx. tenfold). The emissions from energy production were CO_2 for GWP (global warming potential) and NO_x for eutrophication. As an energy source, a conventional combined heat and power plant system was assumed (Fogelholm and Heinolainen 1993). A 15-year time frame was used in which the emissions from one ton of waste during the first 15 years was considered (Pelkonen *et al.* 2000). In biological treatment, the average removals of 74 % (COD) and 68 % (N) were assumed during this period. Phosphorus emissions were based on 0.5 mg P l^{-1} effluent concentration. The eutrophication and GWP impacts were calculated based on Lindfors *et al.* (1995) assuming that both nitrogen and phosphorus are limiting factors in the recipient. The amount of leachate was calculated based on a 10 m landfill height and 700 mm a^{-1} precipitation, 45 % of which is formed to leachate.

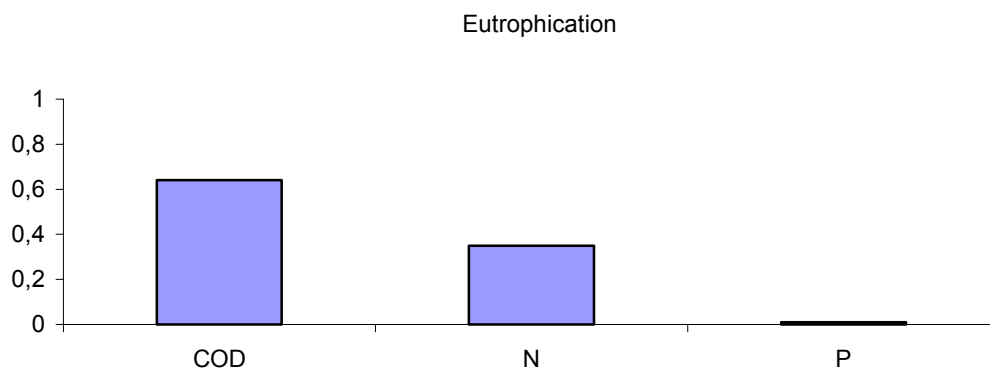


Fig 3. Eutrophication caused by different components in biologically treated water in a 15-year time frame.

The eutrophication impact after biological treatment is caused mainly by COD and nitrogen (Fig 3). Compared to the biological treatment, the tenfold energy consumption in the reference system gives a 2.5 % increase in eutrophication, which is low. On the other hand the GWP of the reference system is 720 % compared to the biological system (Fig. 4). The impact of N_2O emission on GWP in the biological process is approximately 30 %. This proportion can be assumed to increase a little as a function of plant size.

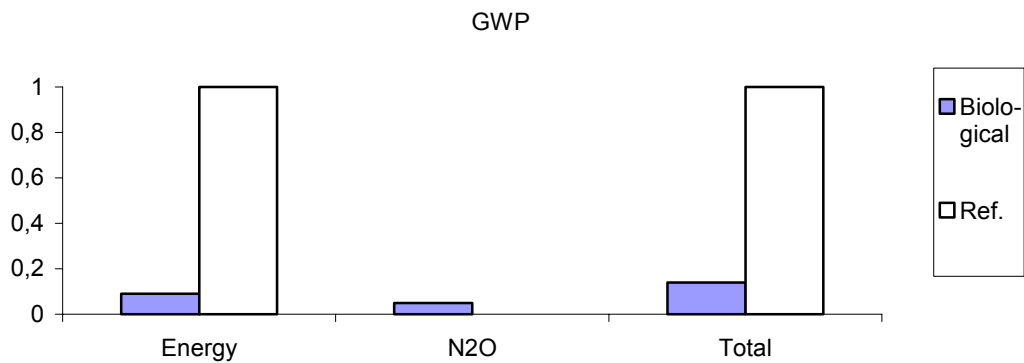


Fig. 4. Global warming potential in biological treatment compared to the reference system

In order to reduce the eutrophication, the COD and nitrogen removals are of importance. It is assumed that all COD in treated water will degrade, which is an obvious upper level estimate. In the long term, the impact of nitrogen is much higher compared to COD. The removal of residual COD after biological treatment can be necessary and also connected to the removal of organohalogen compounds.

5. Conclusions

A biological process was successfully applied to treat cold leachate in winter conditions on full scale. A stable and nearby complete ammonia nitrogen oxidation was maintained, obviously for the first time on full scale at a process temperature of 4.5-5 °C in combination with a reasonable denitrification efficiency. The process was not significantly affected by the inhibition of heavy metals and toxic compounds in these extreme conditions.

The solids and metal removal efficiencies were reasonable and no clear temperature impact was observed. However, the AOX removal efficiency decreased with decreasing temperature and the range was considerable, 10-50 %. This indicates that the removal of organohalogen compounds must be considered as a temperature-dependent and specific case and the biodegradable fraction of AOX (25-50 %) needs more characterisation.

In biologically treated leachate, the eutrophication is caused by COD and nitrogen in the short (15 years) term, but in the long term the role of nitrogen is more important. Therefore, in order to decrease the eutrophication, the C/N-ratio in the leachate, which will limit the nitrogen removal, has to be controlled internally or externally. The residual organic matter can need reducing actions with respect to eutrophication or organohalogen compounds.

The treatment results of COD and nitrogen together with the economical evaluation show that the application of biological processes in leachate treatment at low temperatures is worth considering and the feasible temperature range can be extended down to 4.5-5 °C.

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