FEASIBILITY OF BIOLOGICAL AERATED FILTERS (BAFS) FOR LEACHATE TREATMENT

T. STEPHENSON*, S. J. T. POLLARD** AND E. CARTMELL*

**School of Water Sciences, Cranfield University, Cranfield, Bedfordshire, MK43 0AL, UK*
***Integrated Waste Management Centre, Cranfield University, Cranfield, Bedfordshire, MK43 0AL, UK*

SUMMARY: Ammonia can be removed from landfill leachate through aerobic biological processes. The biological aerated filter (BAF) combines biological treatment and subsequent biomass separation in one reactor providing a small footprint alternative to conventional systems. Leachate from an operational landfill was found to be aerobically treatable using the OECD recommended Modified Zahn-Wellens test. This leachate was used as feed to a pilot-scale BAF at influent COD and ammoniacal-nitrogen concentrations of 765 mg/l and 568 mg/l respectively. During an initial period of stable operation without pH control, 33 % of influent ammonia was removed. The reactor pH was 9.2 with little conversion to total oxidized nitrogen (<45 mg/l). Therefore this removal was accounted for primarily by air stripping. In a second period of stable operation, the reactor pH was reduced to pH 7.2 by addition of hydrochloric acid. Ammonia removal was increased to 97 % with a concomitant increase in effluent nitrite concentration to an average of 524 mg/l. Biological aerated filters can therefore be used to nitrify landfill leachates.

1. INTRODUCTION

Landfill leachates have conventionally been treated with aerobic biological processes when inhibitory substances are absent; the most commonly applied processes are aerated lagoons (Robinson *et al*., 1992) and sequencing batch reactors (Ying *et al.*, 1986). Anaerobic processes are capable of removing biodegradable organic matter, usually quoted as biochemical oxygen demand (BOD) or chemical oxygen demand (COD), from leachates (Kettunen & Rintala, 1998) but aerobic processes are required for ammonia removal. This is achieved by the two step nitrification process involving conversion of ammonia to nitrite, then to nitrate, with the supply of oxygen (Shama & Ahlert, 1977).

Alternative aerobic processes have been proposed for leachate treatment, e.g. rotating biological contactors (Spengel & Dzombak, 1991), suspended carrier reactors (Welander *et al*., 1998); biological aerated filters (BAFs) represent another alternative. Conventionally, BAFs are submerged media wastewater treatment reactors that combine aerobic biological treatment and biomass separation by depth filtration (Mendoza-Espinosa & Stephenson, 1999). They adopt a

Proceedings Sardinia 2003, Ninth International Waste Management and Landfill Symposium
S. Margherita di Pula, Cagliari, Italy; 6 - 10 October 2003
© 2003 by CISA, Environmental Sanitary Engineering Centre, Italy

granular media as the support for microbial biofilms that also provides the depth filtration action (Figure 1). BAFs offer a small footprint alternative to conventional aerobic processes and can operate at biochemical oxygen demand (BOD) loadings much higher than trickling filters and activated sludge plants. Carbonaceous BOD removal, solids filtration and nitrification are achieved in a single unit operation and process modifications allow for denitrification and phosphate removal (Clark *et al*., 1997). BAFs are commonly used in municipal wastewater treatment for tertiary nitrification alone, especially for upgrading existing works, as they can operate at higher space loadings than trickling filters (Table 1); hence their attraction for leachate treatment.

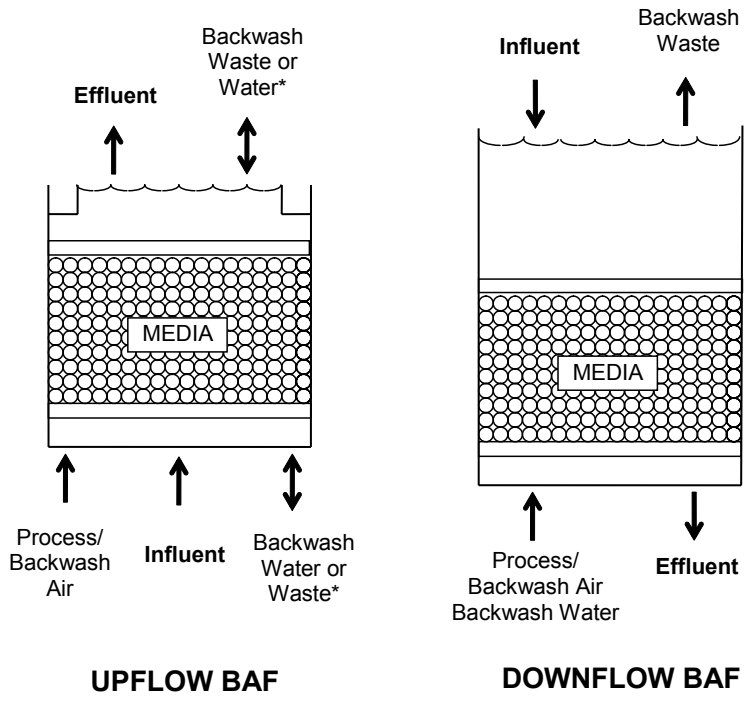


Figure 1. Schematics of upflow and downflow BAFs showing key features

Table 1 - Typical ammoniacal-nitrogen loadings for trickling filters and BAFs to achieve >90% removal when operating in tertiary treatment mode.

Unit process	Ammonia loading (kg NH_4-N/m^3/d)
Tertiary filters	0.1 – 0.6
BAFs	0.6 - 1.0

The current study aimed to investigate the feasibility of applying BAFs to the treatment of methanogenic landfill leachates. This was undertaken through treatment of a real leachate in a pilot-scale BAF.

2. EXPERIMENTAL

2.1 Pilot-scale biological aerated filter plant

The pilot-scale BAF plant used for the investigation was an upflow 5.3 m high column, 0.43 m diameter, based on the proprietary Capital Controls ColOX design. The granular media (height 2.1 m) was supported by 0.90 m of underdrain, giving a media volume of 0.31 m^3. The skid-mounted column was fitted with controls, pumps, blowers, a clearwell and mudwell (Figure 1). The clearwell and mudwell were each approximately 0.5 m^3. The media was a crystal quartz sand with an approximate diameter of 1.5 to 2.2 mm and an effective porosity of 40 %v/v. This media was supported by a layered underdrain of equal heights (0.30 m) of 40 mm, 14 mm and 10 mm gravel. To ensure sufficient wetting of the column, a flow of at least 151 l/h was maintained through a recycle.

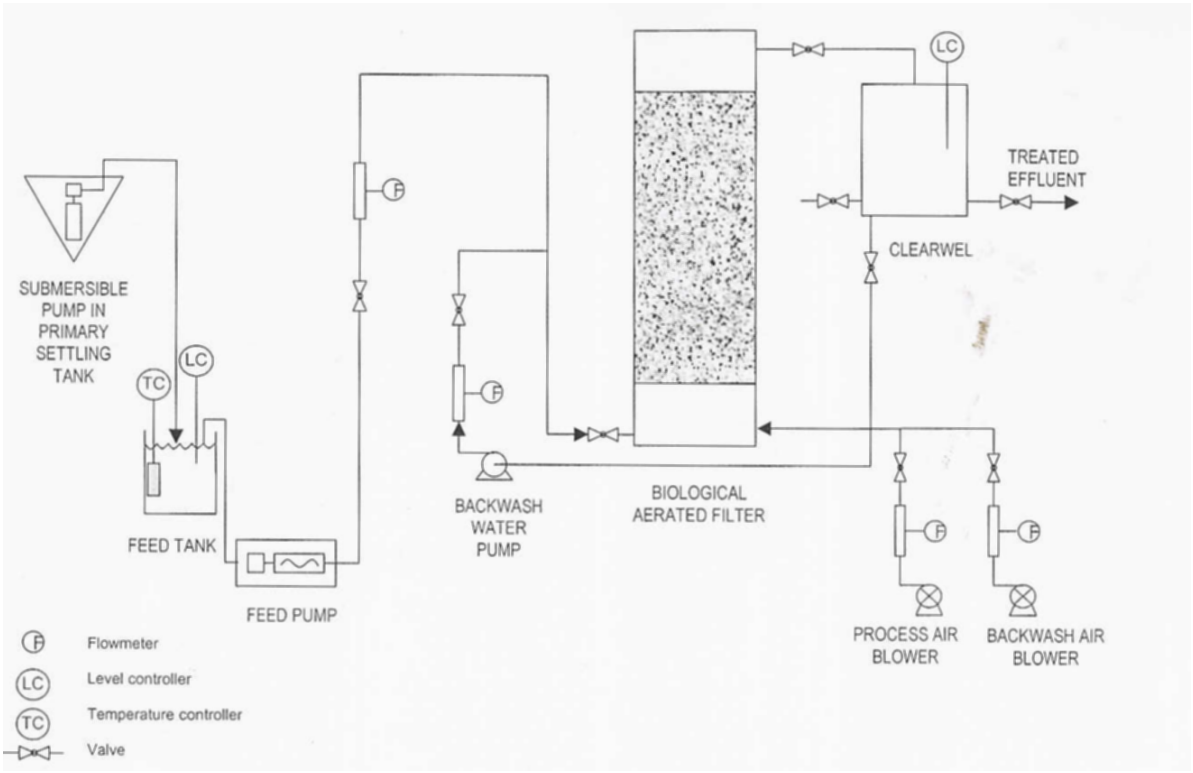


Figure 1. Schematic of pilot-scale BAF and ancillary equipment.

Leachate was fed to the large pilot-scale BAF with the use of a Watson-Marlow 700 series peristaltic pump with a capacity 0-100 $1min^{-1}$ (Watson-Marlow Ltd, Cornwall, UK). Backwashing water was supplied by means of a Grundfos peristaltic pump (model A006422) of 60 m^3/h capacity (Grundfos Pumps Ltd, Leighton Buzzard, UK). Air and liquid flow to the BAF were monitored by means of a Platon plastic tube VA flowmeter with a 40-440 1/min range and a Platon plastic tube VA flowmeter with a 300-3000 1/h range, respectively (CT Platon Ltd, Basingstoke, UK). Bulb vales (Durapipe S&LP, Staffordshire, UK) were used to control both hydraulic and gas flowrates. An automatic motorised valve (Ultravalve Ltd, Halsowen, UK) was used to prevent backwash water from siphoning back through the effluent pipe.

Backwashing was undertaken automatically by means of a timer. The backwash cycle consisted of the following sequence.

1. feed pump off, the outlet valve closes

2. backwash air comes on; runs for five min
3. backwash water on; air and water run simultaneously for 15 min
4. backwash air stops; backwash water still runs for further 5 min (or is tripped off by low level in clearwell)
5. backwash water off
6. two minute delay
7. feed pump comes on; normal operation is resumed

For start-up, seeding with nitrifying activated sludge from a municipal sewage treatment works was used. The pilot-plant was run in batch mode for 3 days until some ammonia removal was established, then influent leachate diluted to approximately 50 mg/l NH_4-N, dosed with alkalinity and phosphate, was started. This corresponded to one tenth of the maximum expected loading to be used, i.e. 0.01 kg NH_4-N/m^3/d, with the loading increasing to a maximum of 0.07 kg NH_4-N/m^3/d, corresponding to a COD loading rate of 0.10 kg COD/m^3/d.. The pilot-plant was operated for two distinct stable operating periods, one with no pH control and one with addition of hydrochloric acid to reduce pH.

2.2 Leachate treatability studies

Seeking to examine the feasibility of BAFs for the treatment of landfill leachate, a methanogenic leachate with a measured COD concentration of 786 mg/l, chloride close to 10000 mg/l and ammoniacal nitrogen of 652 mg/l was sampled from two distinct locations at a landfill site in the United Kingdom and treated in a pilot BAF plant at Cranfield University. A broad analytical suite of priority pollutants, including pesticides and heavy metals, undertaken by a commercial laboratory, confirmed the absence of substances inhibitory to nitrification. Batch biodegradability testing was undertaken prior to the pilot-plant trials. Leachate characteristics were monitored over a 6-week period during feeding to the plant and found to be stable. Landfill leachate is usually deficient in alkalinity and phosphate for biological treatment to be successful. In order to provide 7.14 mg of alkalinity as calcium carbonate per 1 mg of NH_4-N, 4,686 mg/l as calcium carbonate was added to the feed leachate as sodium biocarbonate. To provide a nutrient balance sufficient for biological nitrification, i.e. a COD:P ratio of 100:1 (Ying *et al*., 1986) , the stored leachate was initially dosed with trisodium orthophosphate. After problems were experienced with precipitate formation, a 1:1 mix of sodium dihydrogen orthophosphate and dipotassium hydrogen orthophosphate was used. The pH of the leachate remained altered at pH 7.8 both before and after nutrient dosing.

2.3 Analytical methods

Bench scale biological treatability studies adopting the OECD-recommended Zahn-Wellens test were used to investigate COD removal, with tests being undertaken over 28 days (OECD, 1981). Standard methods were used for routine suspended solids (SS) (UK DOE 1972), BOD_5 (UK DOE, 1988), total and soluble COD (UK DOE, 1986) and ammoniacal nitrogen (UK DOE, 1981a) determinations. Samples were filtered through GF/C grade filter paper (Whatman Ltd, UK) prior to analysis of soluble COD (SCOD) and other soluble species. Total oxidised nitrogen (TON) and nitrite-nitrogen were determined by standard methods (UK DOE, 1981b) and nitrate found by difference. Phosphate, alkalinity and chloride were monitored in the leachate influent to the pilot-scale BAF using recommended techniques (UK DOE, 1980; 1981c;d). Dissolved oxygen and pH in the BAF study were measured by probes (Jenway Models 9070 and 3070 respectively).

3. RESULTS

3.1 Treatability studies

Leachate samples from two different locations on the site had COD concentrations of 1,406 mg/l and 1662 mg/l; they were diluted to initial concentrations of 253 mg/l and 396 mg/l respectively in modified Zahn-Wellens tests. Both samples were found to be biodegradable with >80 % COD removal. A mixture of the two were then tested at three initial COD concentrations of 287, 396 and 479 mg/l. After 28 days, the lowest concentration was found to be most degradable at 97 % COD removal; the highest concentration was the least easily degraded at 49 % COD removal (Figure 2). These tests demonstrated that the leachate was biologically treatable.

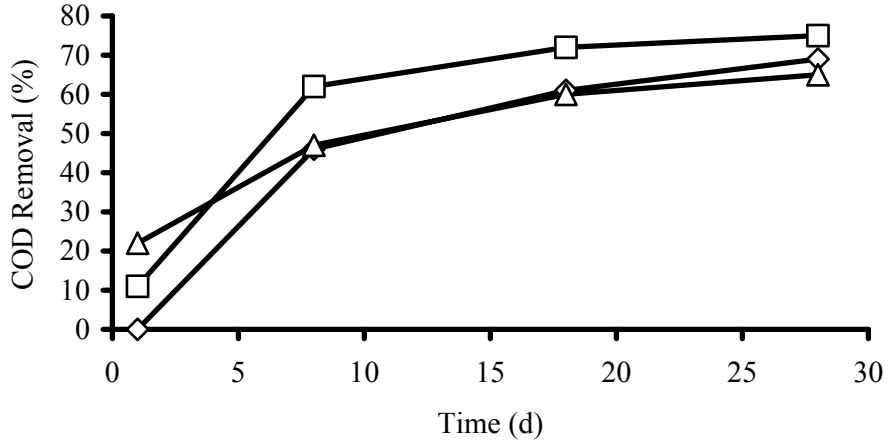


Figure 2. Percentage biodegradation of leachates at starting COD concentrations of 287 mg/l (□), 396 mg/l (△) and 479 mg/l (◇)

3.2 Pilot-plant trials

Influent leachate characteristics were monitored throughout the pilot-plant trial (Table 2). Total and soluble COD, and ammonia concentrations were relatively stable, whereas suspended solids concentrations decreased during operation from approximately 191 mg/l to <100 mg/l. This decrease coincided with delivery of a second batch of leachate and a switch in dosing trisodium orthophosphate to the potassium/sodium hydrogen orthophosphate mixture. A corresponding increase in soluble phosphate from <2 mg/l to >10 mg/l was noted. The pH of the leachate remained constant at pH 7.8; chloride was also constant.

Table 2 – Influent leachate characteristics throughout pilot-plant trial.

Determinand	Average	Maximum	Minimum	No. of tests
Temperature (^oC)	17	19	15	39
pH-value	7.8	7.8	7.8	33
Total alkalinity (mg/l as $CaCO_3$)	4520	4619	4338	12
Soluble orthophosphate	1.7[a]	1.2[a]	1.2[a]	4[a]
(mg/l as P)	10.4[b]	10.5[b]	10.2[b]	3[b]
Chloride (mg/l as Cl)	1,000	1,000	1,000	4
SS (mg/l)	119	191	66	22
TCOD (mg/l)	765	905	625	18
SCOD (mg/l)	722	859	616	10
BOD_5 (mg/l)	76	144	45	4
NH_4-N (mg/l)	568	652	533	19
TON (mg/l)	4	10	0.5	13
NO_2-N (mg/l)	0.3	0.9	0.01	13
NO_3-N (mg/l)	4.1	10	0.3	8

[a]trisodium orthophosphate dosing
[b]sodium/potassium hydrogen phosphate mix

Temperature of the pilot-plant varied from 15 ^oC to 19 ^oC during the study. Dissolved oxygen concentrations in the effluent averaged 6.0 mg/l which indicated that oxygen was not limiting. Effluent alkalinities and phosphate concentrations were always well above the detection limits throughout the trial, again indicating that neither were limiting nitrification. At no point was the BAF backwashed to remove excess solids as there was no significant build of headloss, as indicated by the depth of liquid on the top of the column.

Eight days after start-up, the pilot-plant was observed for a further 17 day stable operating period without pH control. Performance in terms of total BOD removed was excellent, as anticipated from the treatability tests, with >90 % removal (Table 3). Suspended solids removal was also good at >70 %. However, COD removal was poor, indicating that a high proportion of recalcitrant COD was present. During this period, effluent pH rose to an average of pH 9.2 with only 33 % of influent ammonia removed (Table 4). Of this removed ammonia, <8% was accounted for by the average effluent TON.

Table 3 – Average suspended solids, BOD and COD performance for the BAF pilot-plant.

Parameter	Influent (mg/l)	Effluent (mg/l)	Removal (%)
Suspended solids	132	35	73
Total COD	787	506	36
Soluble COD	719	588	18
Total BOD	86	6	93

Table 4 – Average ammonia removal and oxidised nitrogen production during 17 days stable operation without pH control.

Parameter	Influent (mg/l)	Effluent (mg/l)	Removal (%)
Ammonia -N	581	392	33
Total Oxidised N	4.1	45	-
Nitrite -N	0.1	35	-
Nitrate -N	4.3	8.8	-

After the first stable operating period, pH control was initiated. The reactor pH was lowered to pH 7.1 – 7.8 by dosing hydrochloric acid. Over an 8 day period, nitrification improved until steady-state ammonia removal of 99 % was achieved. The BAF was then monitored during stable operation for a further 8 days, during which period ammonia removal remained at 99 % (Table 5). An average mass balance indicated that 95 % of the influent ammonia could be accounted for in the effluent, with the majority present as nitrite. High effluent concentrations of nitrite confirmed that the first stage of nitrification was functioning but the second stage was not (Table 5).

Table 5 – Average ammonia removal and oxidised nitrogen production during 8 days stable operation with pH controlled at pH 7.2.

Parameter	Influent (mg/l)	Effluent (mg/l)	Removal (%)
Ammonia -N	578	6.2	99
Total Oxidised N	0.6	543	-
Nitrite -N	0.3	524	-
Nitrate -N	0.3	4.7	-

4. DISCUSSION

It is well known that ammonia can be removed by stripping from aqueous effluents. Ammoniacal nitrogen is removed by volatilisation of dissolved ammonia; the rate of ammonia transfer is enhanced by increasing the pH thereby converting a greater proportion of the ammonium ion to free ammonia (Metcalf & Eddy, 2002). Benneworth and Morris (1972) found that the optimum pH was in the range pH 10.5 to 11.5 and that air requirements will increase with decreasing pH for a given ammonia removal. The free ammonia (FA) concentration is a function of the total ammonia concentration, pH and temperature of a solution (Anthonisen *et al.*, 1976):

$$\text{FA as } NH_3 \text{ (mg/L)} = (17/14) \times \frac{\text{total ammonia as N (mg/L)} \times 10^{pH}}{(K_b/K_w) + 10^{pH}} \quad (1)$$

where K_b and K_w are the ionisation constants for ammonia and water respectively. Using this equation, during the first stable operating period, the ammoniacal-nitrogen concentration was 581 mg/l; therefore 192 mg/l was present as free ammonia. This was 33 % of the total ammonia concentration – corresponding to the percentage removal of ammonia. However, as there was some conversion of ammonia to nitrite and nitrate accounting for 8 % of the removal, 25 % could be accounted for due to ammonia stripping.

During the second stable operating period, pH adjustment stimulated nitrification, but only through to nitrite as opposed to nitrate. Lowering the pH will decrease the proportion of ammonia present as the dissolved gas compared to the ammonium ion. The former is the substrate for the first stage of nitrification, therefore the conversion to nitrite is stimulated (Shama & Ahlert, 1977). However, as noted by Anthonisen *et al.* (1976) and summarised by Turk and Mavinic (1989), high concentrations of nitrite at the operating pH used in this study will inhibit the conversion to nitrate by *Nitrobacter*.

Discharge of effluent to the environment with high nitrite concentrations would be an unacceptable practice. However, the ability of processes to denitrify nitrite has been demonstrated previously (Abeling & Seyfried, 1992). Therefore nitrification-denitrification can be achieved without the need to supply the additional energy for nitrate production. While it is likely that BAF systems similar to the one tested here can be applied to leachates, optimisation

will need to be undertaken in terms of aeration (Harris *et al.,* 1996), flow direction (Mann *et al*., 1998), media type (Moore *et al*., 1999) and the size of that media (Moore *et al*., 2001).

5. CONCLUSIONS

The study confirms the potential feasibility of the small footprint BAF process for the treatment of domestic leachate, with ammonia-N concentrations reduced to below 10 mg/l and BOD to below 20 mg/l. Onward anoxic denitrification of nitrite-nitrogen and COD polishing is required to reach typical discharge consent standards.

ACKNOWLEDGEMENTS

The authors wish to acknowledge the practical contribution made by Emma Dobson, MSc student in the School of Water Sciences at Cranfield, and Howard Rundle, Capital Controls Ltd, for the supply of the pilot-plant and technical support.

REFERENCES

Abeling, U., & Seyfried, C.F. (1992) Anaerobic-aerobic treatment of high-strength ammonium waste-water – nitrogen removal via nitrite. *Water Sci. Technol*., 26 (5/6), 139-146.

Anthonisen, A.C., Loehr,R.C., Prakasam, T.B.S., & Srinath, E.G. (1976) Inhibition of nitrification by ammonia and nitrous acid. *J. Water Pollut. Control Fed*., 48, 835-851.

Benneworth, N.E., & Morris, N.G. (1972) Removal of ammonia by air stripping. *J. Water Pollut. Control Fed.*, 44, 485–492.

Clark, T., Stephenson, T., & Pearce, P. A. (1997). Phosphorus removal by chemical precipitation in a biological aerated filter. *Water Res*., 31, 2557-2563.

Harris, S. L., Stephenson, T., & Pearce, P. (1996). Aeration investigations of biological aerated filters using off-gas analysis. *Water Sci. Technol.*, 34 (3/4), 307-314.

Kettunen, R.H., & Rintala, J.A. (1998) Performance of an on-site UASB reactor treating leachate at low temperature. *Water Res*., 32, 537-546.

Mann, A., Mendoza-Espinosa, L., & Stephenson, T. (1998). A comparison of floating and sunken media biological aerated filters for nitrification. *J. Chem. Technol. Biotechnol*., 72, 273-279.

Mendoza-Espinosa, L., & Stephenson, T. (1999) A review of biological aerated filters for wastewater treatment. *Environ. Eng. Sci*., 16, 201-216.

Metcalf & Eddy Inc (2002) *Wastewater Engineering* (4th Edition) McGraw-Hill, New York.

Moore, R., Quarmby, J., & Stephenson, T. (1999). BAF media: ideal properties and their measurement. *Trans IChemE (Part B)*, 77, 291-297.

Moore, R., Quarmby, J., & Stephenson, T. (2001) The effects of media size on the performance of biological aerated filters. *Water Res*., 35, 2514-2522.

Organisation for Economic Co-operation and Development (1981) Inherent biodegradability: modified Zahn-Wellens test. *OECD Guidelines for Testing of Chemicals*, OECD, Paris, 1-11.

Robinson, H.D., Barr, M.J., & Last, S.D. (1992) Leachate collection, treatment and disposal. *J. Inst. Water Environ. Manage.*, 6, 321–332.

Shama, B., & Ahlert, R.C. (1977) Nitrification and nitrogen removal. *Water Res*., 11, 897-925.

Spengel, D.B., & Dzombak, D.A. (1991) Treatment of landfill leachate with rotating biological contactors: bench-scale experiments. *J. Water Pollut. Control Fed*., 63, 971-981.

Turk, O., & Mavinic, D.S. (1989) Stability of nitrite build-up in an activated sludge system. *J. Water Pollut. Control Fed*., 61, 1440-1448.

UK Department of the Environment (1972) Suspended solids and total dissolved solids in waters and effluents. *Methods for the Examination of Waters and Associated Materials*. HMSO, London.

UK Department of the Environment (1980) Phosphorus in waters and effluents. *Methods for the Examination of Waters and Associated Materials*. HMSO, London.

UK Department of the Environment (1981a) Ammonia in waters. *Methods for the Examination of Waters and Associated Materials*. HMSO, London.

UK Department of the Environment (1981b) Oxidised nitrogen in waters. *Methods for the Examination of Waters and Associated Materials*. HMSO, London.

UK Department of the Environment (1981c) Determination of alkalinity and acidity of waters. *Methods for the Examination of Waters and Associated Materials*. HMSO, London.

UK Department of the Environment (1981d) Chloride in waters and sewage effluents. *Methods for the Examination of Waters and Associated Materials*. HMSO, London.

UK Department of the Environment (1986) Chemical oxygen demand (dichromate value) of polluted and wastewater. *Methods for the Examination of Waters and Associated Materials*. HMSO, London.

UK Department of the Environment (1988) Five day biochemical oxygen demand (BOD_5) with dissolved oxygen in waters. *Methods for the Examination of Waters and Associated Materials*. HMSO, London.

Welander, U., Henrysson, T., & Welander, T. (1998) Biological nitrogen removal from municipal landfill leachate in a pilot scale suspended carrier biofilm process. *Water Res*., 32, 1564-1570.

Ying, W., Bonk, R.R., Lloyd, V.J., & Sojka, S.A. (1986) Biological treatment of a landfill leachate in a sequencing batch reactor. *Environ. Prog*., 5, 41–50.



https://dspace.org/

Feasibility of biological aerated filters (BAFs) for treating landfill leachate

Stephenson, Tom; Pollard, Simon J. T.; Cartmell, Elise

2004-12-31T00:00:00Z

http://dspace.lib.cranfield.ac.uk/handle/1826/1414

Proceedings Sardinia 2003. 9th International Waste Management and Landfill Symposium. S. Margherita di Pula, Cagliari, Italy; 6-10 October 2003

Downloaded from a DSpace repository. https://dspace.org/