TREATMENT OF HIGH AMMONIA CONCENTRATION LANDFILL LEACHATE WITH AN ANAEROBIC FILTER AND ROTATING BIOLOGICAL CONTACTOR (RBC)

· by

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ABSTRACT

Landfill leachate discharge potentially results in degradation of the receiving environment. One of the primary leachate contaminants of concern is nitrogen, which in the form of ammonia is toxic to aquatic life, and in the form of nitrate may cause eutrophication of the receiving water body. For these reasons, it is often desirable to totally remove nitrogen from leachate prior to discharge to the environment.

Biological nitrogen removal appears to be the most practical method of removing nitrogen from leachate. Rotating biological contactors (RBC's), in particular, have been shown to effectively nitrify high ammonia leachate. If sufficient carbon is available, anaerobic biological treatment processes have been shown to effectively denitrify aerobically treated leachate.

This thesis investigates the use of a predenitrifying anaerobic filter and a rotating biological contactor to remove nitrogen from a high ammonia concentration leachate from a municipal solid waste landfill in Kaohsiung, Republic of China. The primary objective of the research was to determine the effects of high ammonia concentration on RBC ammonia loading and removal. Secondary objectives include removing organics and metals from the leachate.

The research indicated that 97% ammonia removal from high ammonia leachate (mean 2,140 mg/L) can be achieved at RBC loading rates up to 1.5 $g/m^2/day$. At higher loading rates, ammonia removal was inhibited.

Nitrogen removal for the system averaged 66%, including an estimated 54% in the RBC. Nitrogen removal in the RBC was either the result of simultaneous nitrification/denitrification or air stripping of ammonia in combination with nitrification. Both alkalinity consumption and COD removal results support the explanation that simultaneous nitrification/denitrification (potentially aerobic

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denitrification) occurred, but since RBC off-gasses were not monitored, neither theory can be confirmed.

BOD and COD removal in the system averaged 92% and 49%, respectively. COD:BOD removal was 3.7:1.

The system did not effectively remove metals. Overall removal of dissolved metals ranged from -19% for nickel to 59\% for manganese. Organic complexing of the metals most probably resulted in low removals.

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1. INTRODUCTION

Sanitary landfilling is the most common method of disposing of municipal solid waste in North America and much of the rest of the world. Landfilling may detrimentally affect the local environment through the production and discharge of contaminants to the atmosphere and land. A major source of contaminants is the discharge of leachate to ground or surface water.

Landfill leachate is generated by the percolation of water through refuse. The organic and inorganic decomposition products of the waste dissolve in the water producing an odorous, dark liquid known as leachate. Major contaminants of concern include dissolved organic material, ammonia, and metals.

For older landfill leachates, the primary contaminant of concern is ammonia, which is toxic to fish, and if converted to nitrate may result in eutrophication of the receiving environment. For these reasons, complete nitrogen removal from leachate is often desirable.

Several authors have shown that the rotating biological contactor (RBC) is effective at converting leachate ammonia to nitrate (Peddie and Atwater, 1985; Hartmann and Hoffmann, 1990; Hosomi et al., 1991; Masuda et al., 1991). Under certain conditions, RBC's may also simultaneously remove nitrogen from the system (Masuda et al., 1991; Hosomi et al., 1991, Atwater and Bradshaw, 1981). Ammonia has also been shown to be toxic to ammonia oxidizing bacteria at high influent concentrations (Azevedo, 1993), but the RBC process may be more resistant to ammonia toxicity than suspended growth processes (Hartmann and Hoffmann, 1990).

To fully remove nitrogen from wastewater, an anoxic basin is generally used. The most common method is to either use a basin prior to the aerobic reactor with recycle or a basin following the aerobic basin. In each case, supplemental carbon is generally added to provide an energy source for denitrification. If sufficient carbon is available, predenitrification may be possible without

supplemental carbon addition.

The leachate used in this experiment has very high ammonia concentration with relatively high COD. Therefore, it was hypothesized that a process train of a predenitrifying anaerobic filter followed by an RBC may be effective in removing nitrogen from the leachate. The primary objective of the experiment was to use this process train to investigate the effects of high ammonia concentration on loading rates and removal efficiencies. Secondary objectives included removing organics and metals from the leachate.

The experimental work described in this thesis was conducted at the National Sun Yat Sen University, Department of Marine Environment, Kaohsiung, Taiwan, Republic of China, during the winter and spring of 1991/1992.

2. LITERATURE REVIEW

2.1 LANDFILL LEACHATE CHARACTERISTICS, GENERATION AND DETRIMENTAL EFFECTS

2.1.1 Leachate Characteristics

The characteristics of landfill leachate vary from site to site and over time at an individual site. The variation between sites is based on differences in climatic conditions, hydrogeological conditions, and waste composition (Pohland et al., 1985). The variation over time at an individual site is a result of the degree of stabilization of the waste.

Pohland et al. (1985) have divided the stabilization of landfills into a number of discreet stages. A summary of those stages is given in Table 1.

Due to the continuous rather than batch nature of landfilling, and the length of time required to reach Phase V (up to 20 years; Pohland et al., 1985), it is common to describe leachate as either acidic or methanogenic (Ehrig, 1985), or as young, mature, aging or old based on the BOD:COD ratio (young = 0.7, mature = 0.5, aging = 0.3, old = 0.1) (Henry et al., 1987). The length of time associated with the transition from young to old after the start up of a landfill is typically from 2 - 10 years depending on circumstances (Forgie, 1988a).

TABLE 1: LANDFILL STABILIZATION STAGES

Phase I: Initial Adjustment

- Landfill is under aerobic conditions prior to the depletion of all the entrained air in the refuse
- No leachate generation because the refuse has not reached field capacity

Phase II: Transition

- Landfill is in transition from aerobic to anaerobic stage
- Leachate production begins

Phase III: Acid Formation

- Volatile organic fatty acids are the primary component of COD resulting in a high biologically oxidizable organic fraction, and consequently a high BOD:COD ratio.
- Leachate is characterized by low pH
- Metals tend to be mobile

Phase IV: Methane Fermentation

- Volatile organic fatty acids are converted to methane and carbon dioxide
- Leachate is characterized by neutral to slightly alkaline pH
- Reducing environment within the landfill results in the disappearance of nitrates and sulphates.
- Metals are complexed with sulphide and organic ligands and precipitated from solution
- Total organic strength (COD) significantly lower
- Humic and fulvic compounds dominate COD resulting in a low biologically oxidizable fraction, and consequently a low BOD:COD ratio.

Phase V: Final Maturation/Stabilization

- Relative dormancy
- Gas production all but ceases
- Settling ceases
- Recalcitrant organics may be slowly converted to humic substances capable of forming soluble complexes with heavy metals, remobilizing them

SOURCE: Pohland et al., 1985

Expected characteristics of acid phase and methanogenic phase leachate as well as example characteristics are provided in Table 2.

Parameter	Expected Acidic 1	Expected Methanogeni 2	Example c Acidic 3	Example Methanogenic 4	
рН	6.1	8	5.8	8.1	
BOD (mg/L)	13,000	180	9,660	100	
COD (mg/L)	22,000	3000	13,780	1,000	
BOD:COD	0.6	0.06	0.7	0.1	
NH ₃ -N (mg/L)	750	750	42	340	
Total N (mg/L)	1,250	1,250	212	340	
Total P (mg/L)	6	6	0.77	0.2	
Cd $(\mu g/L)$ 5	6	6	0.10	14	
Ni $(\mu g/L)$	200	200	1080	220	
Pb $(\mu g/L)$	90	90	-	<100	
$Cr(\mu g/L)$	300	300	_	60	
Cu $(\mu g/L)$	80	80	190	120	
Mn (mg/L)	25	0.7	-	-	
Fe (mg/L)	780	15	1070	11	
Zn (mg/L)	5	0.6	5.04	0.34	
	· · · ·				

TABLE 2: CHARACTERISTICS OF LANDFILL LEACHATE

SOURCES/NOTES:

Ehrig (1989a)
 Ehrig (1989a)
 Henry et al. (1987)
 Knox (1985)
 Metals are totals for example acidic, but not stated for the rest.

Note that although the expected parameter values for acidic and methanogenic leachate described by Ehrig (1989a) are not exactly the same as the example leachates from Henry et al. (1987) and Knox (1985) major parameter values tend to be similar (e.g. pH, BOD, COD, BOD:COD ratio). Variations from the expected values are a result of site specific conditions.

2.1.2 Leachate Generation

Leachate generation depends on the amount of moisture infiltrating into a landfill.

Various models are used to calculate leachate flow from a landfill. They are all based on creating a water balance for the landfill. A simple water balance, which assumes groundwater flow and surface run-off to be zero, is given in Jasper et al. (1985a) and is shown in Equation 1.

EQUATION 1:	LANDFII	LL WATE	R BALANCE						
Leachate	Product	ion = P	recipitat - Refuse	ion + Refus Uptake - Ev	e In vapot	put transpirati	Lon		
Calculating	the ra	te of	leachate	production	is	necessary	to	size	treatment
facilities.	A comm	nonly u	sed model	for calcula	ating	g leachate	pro	ductio	on rate is

the HELP model, which was developed in the United States and is described in Farguhar (1989).

2.1.3 Detrimental Effects of Leachate

If landfill leachate is discharged directly to a receiving body (ground or surface water), the potential detrimental effects include toxic effects on fish and other aquatic organisms (Cameron and Koch, 1980), and contamination of the groundwater or surface water with organic or inorganic contaminants. Consequently, prior to discharge, treatment to remove both organic and inorganic constituents is required.

2.2 THEORY OF BIOLOGICAL NITROGEN REMOVAL

Nitrogen is a critical component of living organisms. On average, nitrogen makes up 12-14% of cell protein (Barnes and Bliss, 1983). When organisms die or they excrete waste products, the nitrogen is released into the environment.

A degradation product of proteins is ammonia. In solution, ammonia is either in ionic form (ammonium) or dissolved gaseous form (ammonia). At pH 7 and 20 °C,

approximately 100% of ammonia is in the ammonium ion form. At pH 11 and 20 °C, approximately 100% of ammonia is in dissolved gaseous ammonia form (Viessman and Hammer, 1985). Landfill leachate is generally in the pH range of 4.5 to 8 (Ehrig, 1989a). Therefore, unless the pH is artificially adjusted, the majority of ammonia will be in the ammonium ion form. In this thesis, the term ammonia (or the symbol NH_3) is used to refer to the sum of ammonium ion and free ammonia.

Under aerobic conditions, ammonia is converted to nitrate through the reactions shown in Equation 2 collectively termed nitrification.

EQUATION	12:	: NI	[RIF]	CATION								
Step 3 NH4	L ,* +	1.5	0 ₂	->	NO2-	+ H ₂ O +	⊦ 2H⁺	+ (240	- 350	kJ)		
Step 2 NO ₂	? - +	0.5	0 ₂	->	NO3-	+ (65	- 90	kJ)				
SOURCE:	Baı	rnes	and	Bliss,	1983		÷					

The conversion is done by autotrophic organisms (Step 1 <u>Nitrosomas</u>, Step 2 <u>Nitrobacter</u>), which use the energy generated for synthesis and other life processes. These organisms, by definition, use inorganic carbon rather than organic carbon.

Nitrification can be affected by many environmental conditions. These factors are shown in Table 3.

TABLE 3: FACTORS AFFECTING NITRIFICATION

```
Reduced temperatures

Low O_2 values

pH values outside the optimum

7.9 - 8.2 <u>Nitrosomas</u>

7.2 - 7.6 <u>Nitrobacter</u>

Free ammonia (unionized)

Free nitrous acid

Shock nitrogen loading
```

SOURCE: Ehrig, 1985

Note that the pH optimum for <u>Nitrosomas</u> (ammonia to nitrite) is higher than the pH optimum for Nitrobacter (nitrite to nitrate). Alleman (1984) suggests that

<u>Nitrobacter</u> may be selectively inhibited by pH, low temperature, low oxygen partial pressure, low carbon dioxide partial pressure, free ammonia, and process over-loading. This would result in prolonged nitrite build-up.

Gee et al. (1990) found that inhibition of the rate of oxidation of ammonia to nitrite in batch experiments was successfully modelled by the Haldane inhibition model which is presented in Equation 3:

EQUATION	3:	HALDAN	E INHIBITION MODEL
r _s		=	$\frac{\mathbf{k} \mathbf{X} \mathbf{S}}{\mathbf{k}_{s} + \mathbf{S}} + \mathbf{S}^{2}/\mathbf{k}_{i}$
r		=	substrate utilization rate (M/L ³ T)
k		= 1	maximum substrate utilization rate (T^{-1})
k_		= 1	half velocity coefficient (M/L ³)
k		=	inhibition coefficient (M/L ³)
X		= 0	concentration of microbial species carrying out oxidation (M/L ³)
S		= :	substrate concentration (M/L ³)

The conversion of nitrite to nitrate was not effectively modelled by the Haldane inhibition model. It was observed that nitrite oxidation was inhibited by the simultaneous presence of nitrite and ammonia rather than simply nitrite.

In a continuous flow activated sludge experiment, Azevedo (1993) found full nitrification could be achieved up to influent ammonia- N concentration of 1,500 mg/L, but that at 2,000 mg/L ammonia-N effluent ammonia levels increased to 700 mg/L. This occurred at both 10 day and 20 day aerobic solids retention time (SRT). Azevedo (1993) found an accumulation of nitrite began to occur at influent ammonia-N levels of 600 mg/L. Turk and Mavinic (1989) found nitrite build-up began to occur at free ammonia-N levels of 5 mg/L or total ammonia-N concentration of 90 mg/L at pH 8. Turk and Mavinic (1989) could not maintain a nitrite build-up.

As a result of nitrification, hydrogen ions are released (see Equation 2; Step 1). If the leachate is not effectively buffered, a reduction in pH will result leading to suboptimal conditions and reduced nitrification. To maintain pH balance, the theoretical alkalinity demand is 7.14 mg alkalinity as $CaCO_3$ per mg NH₃-N oxidized.

In many cases, nitrified wastewater is released directly to a receiving body, but since nitrogen is an essential nutrient, its release may stimulate aquatic plant growth, which may be undesirable. High nitrate levels in drinking water may cause infant methaemoglobinaemia (blue baby); blockage of haemoglobin with nitrite ions, which prevent oxygen transport and suffocate the infant (Barnes and Bliss, 1983). Therefore, complete nitrogen removal from the leachate is sometimes appropriate.

Under anoxic conditions, nitrate ions can be used as terminal electron acceptors by facultative, heterotrophic, microorganisms (Barnes and Bliss, 1983). The nitrate ions are thus converted to nitrogen gas by the reactions shown in Equation 4 in a process called denitrification.

EQUATION 4: DENITRIFI	CATION		
Step 1 NO ₃ + 1/3 CH ₃ OH	->	$NO_2^- + 1/3 CO_2 + 2/3 H_2O$	
Step 2 NO ₂ + 1/2 CH ₃ OH	->	$N_2 + 1/2 CO_2 + 1/2 H_2O + OH^-$	
SOURCE: Barnes and Bl	iss, 1	983	

In Equation 4, methanol is shown as the electron donor, but other organic substrates may also be used. Since hydroxide ions are produced during denitrification, an increase in alkalinity results. The theoretical increase in alkalinity due to denitrification is 3.57 mg CaCO₃ per mg N.

The first step in denitrification involves converting nitrate back to nitrite. Turk and Mavinic (1989) found that if the second step of nitrification was skipped (conversion of nitrite to nitrate) the overall nitrification/denitrification reaction could be improved resulting in 1) 40% reduction in COD demand, 2) 63% increase in denitrification rate, 3) 300% decrease in sludge production from anaerobic growth. They could not maintain a build-up of nitrite, finding instead, that over time, the oxidation of nitrite to nitrate in the reactor could not be prevented.

Robertson and Kuenen (1984) showed that certain bacteria, for example <u>Thiospaera</u> <u>pantotropha</u>, are capable of simultaneously nitrifying and denitrifying ammonia wastewater under fully aerobic conditions. These bacteria are heterotrophic nitrifiers and aerobic denitrifiers, and are able to simultaneously use nitrite and oxygen as terminal electron acceptors.

Robertson and Kuenen (1984) showed aerobic denitrification occurred by both measuring nitrogen gas production from an aerobic reactor and measuring the effects of adding denitrifying inhibitors. This reaction is still contingent on the presence of a suitable electron donor (e.g. acetate). The aerobic simultaneous nitrification/denitrification reaction is shown in Equation 5.

EQUATION 5: AEROBIC NIT	TRIFICATION/DENITRIFICATION
$\rm NH_4^+ \rightarrow \rm NH_2OH \rightarrow \rm NO_2^- \rightarrow \rm N_2OH$	$D \rightarrow N_2$
SOURCE: Robertson et al	l., 1988

It should be noted that Equation 5 does not include the organic electron donor necessary for the reaction to proceed. Also, since the reaction does not proceed to nitrate, the COD requirement should be 40% less than conventional denitrification.

Robertson and Kuenen (1984) and Robertson et al. (1988) conducted their research at a reactor temperature of 37 °C. The reason for using such a high temperature is not provided in their papers.

2.3 LEACHATE TREATMENT

Due to the complexity of landfill leachate, to completely treat leachate, prior

to discharge to the environment, generally requires a multi-component process train consisting of both biological and physical/chemical processes (Beszedits and Silbert, 1990). The following review explores the benefits and draw-backs of a wide range of processes.

2.3.1 Biological Treatment

Biological treatment methods rely on heterotrophic and autotrophic microorganisms to, by some means, render contaminants in leachate innocuous, either by removing them completely from the leachate or by binding them into sludge so that they can be separated and removed. There are two basic forms of biological treatment used in treating landfill leachate; aerobic and anaerobic treatment systems.

2.3.1.1 Aerobic Treatment

Aerobic leachate treatment is based on the theory that, in the presence of oxygen, aerobic heterotrophic microorganisms will use the organic substrates present as a food source for growth and energy, and convert them to carbon dioxide and water. Consequently, the BOD and COD of the leachate will be reduced. Other leachate contaminants of concern (ammonia, and metals) may also be removed from the wastewater either through assimilation, oxidation or precipitation. Maris et al. (1985) found a BOD:N of 20 was required to effectively remove ammonia from leachate through assimilation. Under favourable conditions, ammonia is oxidized to nitrate by chemo-autotrophic organisms, and metals are removed through the formation and precipitation of insoluble metal oxides and hydroxides (Henry, 1985).

Aerobic treatment is most appropriate in the BOD:COD range of approximately 0.1 - 0.4 (Forgie, 1988c). In this range, oxidation of ammonia to nitrate can be achieved, and biodegradable organics are still present. Below a BOD:COD ratio of approximately 0.1, the majority of biodegradable organics have been removed, and aerobic treatment would only be considered for ammonia removal. For BOD:COD ratios greater than approximately 0.4, less energy intensive anaerobic treatment

is more appropriate (Forgie, 1988c).

The other requirements for aerobic treatment are adequate nutrients (phosphorous addition to achieve a BOD:P ratio of 100:0.5 is often required (Henry, 1985; Ehrig, 1985), and sufficiently low levels of toxic substances. Various authors have discussed toxicity with respect to specific processes, and therefore, toxicity will be dealt with when describing individual processes.

Aerobic biological processes consist of either suspended growth systems (activated sludge, extended aeration and aerated lagoons) or fixed film systems (trickling filter and rotating biological contactors) (Forgie, 1988a). In suspended growth systems, the microorganisms are suspended in the wastewater, in a "floc", either mechanically or with air bubbles. In a fixed film system, the microorganisms are attached to inert media and are placed in contact with the leachate.

Suspended Growth Systems

In the activated sludge system, microorganisms are suspended in a "floc" either mechanically or with bubbles. The unique feature of an activated sludge system is that sludge is recycled and "wasted" to control the sludge concentration and "sludge age" within the reactor. The variables which are controlled in an activated sludge system are aeration rate, sludge recirculation rate, sludge wasting rate, and influent flow. These variables determine loading and operational parameters for the system including BOD loading (kg/m³/day and kg/kg mlvss/day), sludge age, hydraulic retention time, and BOD removal. The typical system parameter values for conventional activated sludge (tapered aeration) wastewater treatment plants are shown in Table 4.

TABLE	4:	ACTIVATED	SLUDGE	SYSTEM	PARAMETERS
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* F/M = Food to microorg	anism ratio or BOD to MLVSS ratio.
Notes:	
BOD Řemoval	80 - 90%
Sludge Recycle	30%
Hydraulic Retention	6 - 7.5 hours
Sludge Age	5 - 15 days
F/M *	0.2 - 0.5 kg/kg
Volume	0.5 - 0.65 kg/m³/day
BOD Loading	

SOURCE: Viessman and Hammer, 1985.

Extended aeration systems are simply a variation of activated sludge systems, providing lower loading rates (0.15 - 0.5 kg $BOD/m^3/day$), and higher hydraulic retention times (20 - 30 hours, Viessman and Hammer, 1985). This results in less sludge production because the microorganisms are in an endogenous state.

Aerated lagoon systems are similar to activated sludge systems except that no recycle is used. Therefore, lower loading rates and less system control are achievable. Municipal wastewater hydraulic retention times for aerated lagoons are typically around 10 days depending on temperature.

Activated sludge has been used extensively to treat landfill leachate.

Albers et al. (1986) used activated sludge to treat landfill leachate effluent from an anoxic reactor with COD up to 15,000 mg/L, and influent TKN up to 2,000 mg/L. They found that, to ensure proper sludge settling, phosphorous addition was necessary. For full nitrification, bicarbonate addition (HCO_3^-) was necessary to achieve an alkalinity as CaCO₃:TKN ratio of at least 6:1.

The plant was run with predenitrification and 500% effluent recycle. Effluent COD was approximately 1500, BOD was less than 25 mg/L and ammonia was less than 20 mg/L. Effluent nitrate-N was reduced from approximately 1000 mg/L to approximately 200 mg/L by mixing 7 parts anaerobic effluent to 3 parts raw leachate feeding the activated sludge plant. This raised the influent COD from an average of approximately 5,000 to an average of approximately 12,500. Albers et al. (1986) attribute the removal of nitrate to denitrification. They found the COD:N ratio required to remove 80% of the nitrate was 4.

Dedhar and Mavinic (1985) used activated sludge following an anoxic fully mixed reactor on an old leachate (BOD:COD = 0.06) to remove nitrogen. The influent ammonia was up to 288 mg/L and influent COD was up to 318 mg/L. The system was run at 15 days solids retention time. Effluent ammonia was less than 1 mg/L, and effluent nitrate was as low as 20 mg/L. Glucose addition into the anoxic reactor up to 1,500 mg/L was required to achieve denitrification. Influent metal concentrations (totals) were zinc 0.019 - 0.155 mg/L, manganese 0.024 - 0.286 mg/L, iron 10.5 - 36.25 mg/L and nickel 0.025 - 0.066 mg/L. Effluent samples were filtered and removal rates were zinc 35-100%, manganese 78-100%, iron 80-100%, nickel 0-20%. As part of the experiment, zinc and manganese spiking was done to investigate the toxicity of these metals. Manganese did not result in inhibition of nitrification at 12.5 mg/L concentration, but inhibition did appear with zinc concentration at 17.6 mg/L.

Jasper et al. (1985b) used a similar system to Dedhar and Mavinic (1985). Their system was run at nominal hydraulic retention time (aerobic basin volume/influent flow) of 24 hours, and sludge age of 5, 10, 15 and 20 days for the aerobic basin. Influent ammonia concentration averaged 161 mg/L. They could not achieve their target of 10 mg/L effluent ammonia except sporadically, and postulated the poor ammonia removal was due to metal toxicity.

Keenan et al. (1984) investigated the full scale treatment of a landfill leachate with influent ammonia of 890 mg/L. Early attempts to treat the leachate were not successful due to phosphorous limitation and ammonia toxicity. To reduce the influent ammonia concentration, an air stripping lagoon at high pH was incorporated into the system. Air stripping reduced the influent ammonia by approximately 50%. The total system ammonia removal was subsequently 72% for two activated sludge vessels in parallel and 99% for two activated sludge vessels in series.

Knox (1985) used a pilot activated sludge system to treat landfill leachate. Knox (1985) found that under certain conditions, extensive foaming occurred, and that good sludge settling characteristics could not be achieved even with phosphorous addition.

Ehrig (1985) investigated the use of full scale aerated lagoons with hydraulic retention times of greater than 10 days and loading rate less than 20 g $BOD/m^3/day$ for leachate treatment. He found that both effluent ammonia and BOD levels were seriously affected at temperatures below 5 °C (effluent BOD > 50 mg/L), and therefore concluded that aerated lagoons were not appropriate for German climatic conditions.

Azevedo (1993) used an continuous flow completely mixed activated sludge reactor with 20 day solids retention time to treat leachate. The experiment involved supplemental ammonia addition to investigate the effects of ammonia toxicity. The system completely nitrified the ammonia up to 1500 mg/L ammonia-N, but complete inhibition of nitrification occurred at ammonia-N concentration of 2000 mg/L. This may have been the result of an insufficient air supply, which failed to provide a mixed liquor dissolved oxygen level of 2-3 mg/L (Dr. Donald Mavinic, University of British Columbia, Personal communication, March 1994).

Robinson and Grantham (1988) used full scale aerobic lagoons to treat leachate with influent COD of 5518 mg/L, BOD of 3670 mg/L, ammonia-N of 130 mg/L. They found that effluent values for COD of 153 mg/L, BOD of 18 mg/L, and ammonia-N of 9.4 mg/L could be achieved down to water temperatures of 2 to 3 °C by maintaining a hydraulic retention time of 10 days. Ammonia removal was by assimilation rather than nitrification. Robinson and Maris (1983) earlier found that BOD:N values greater than 100:3.6 provided complete ammonia removal through

assimilation. Robinson and Grantham's (1988) system provided metal removal as shown in Table 5.

Metal *	Influent (mg/L)	Effluent (mg/L)	Removal (%)	
Fe	242	3.2	98.7	
Zn	4.9	0.2	95.9	
Mn	40	2.4	94.0	
Mq	85	63	25.9	
Cu	0.13	<0.1	>20	
Cr	<0.1	<0.1	· <u> </u>	
Ni	<0.1	<0.1	_	
Cd	<0.1	<0.1	- -	
Pb	<0.1	<0.1	-	
OTES:				
* Authors d	id not state whether	r metals were disso	lved or totals.	

Forgie (1988a) provided the following potential problems with suspended growth systems: leachate foaming, high power consumption, metal inhibition, temperature loss and inorganic precipitates caused by aeration, high sludge production when treating young leachate, and poor sludge settling due to inadequate phosphorous.

Fixed Film Systems

Fixed film systems include a variety of biological treatment systems in which microorganisms grow attached to an inert medium. The microorganisms are "fed" by either passing the wastewater over the medium (as in a trickling filter) or passing the medium through the wastewater (as in a rotating biological contactor). Examples of fixed film systems include trickling filters, rotating biological contactors (RBC), and packed or expanded bed systems (Forgie, 1988a). In this review, only trickling filters and RBC's will be investigated.

Trickling Filters

Trickling filters are packed columns of rocks or other media through which wastewater is percolated or "trickled" at a controlled rate. Design parameters for low-rate trickling filters for municipal wastewater are given in Table 6.

TABLE 6: TRICKLING FILTER DESIGN F	PARAMETERS
BOD Loading (kg BOD/m³/day) Hydraulic Loading (m³/m²/day) Bed Depth (m)	$\begin{array}{rrrrrrrrrrrrrrrrrrrrrrrrrrrrrrrrrrrr$
SOURCE: Viessman and Hammer, 1985	

Knox (1985) investigated the pilot-scale treatment of landfill leachate using a trickling filter over a two year period. Influent concentrations of ammonia were 340 mg/L. The system hydraulic retention time was varied from 4.5 days to 15 hours. Sodium bicarbonate (NaHCO₃) was added to maintain an effluent pH of 7.5 and sodium dihydrogen phosphate (NaH₂PO₄) was added to provide a Total Oxygen Demand:Phosphorous ratio of 100:1. Effluent ammonia ranged from 2-34 mg/L, and ammonia removal rates ranged from 8 - 70 g N/m³/day (unit volume of plastic media). Anticipated problems of scaling and clogging due to inorganic precipitates did not occur. This was potentially because the leachate was lagooned for several months prior to treatment.

A full-scale trickling filter in the United Kingdom failed due to clogging with ferric oxide precipitates and its inability to handle fluctuating flows (Henry, 1985).

Rotating Biological Contactors (RBC's)

Rotating biological contactors (RBC's) have been in use in Europe since 1958. They initially were used for organics removal, but now are also used for ammonia removal (Poon and Chao, 1979). The potential advantages of RBC's over other technologies include excellent response to shock organic and hydraulic loadings (Peddie and Atwater, 1985), tolerance to high influent ammonia-N concentrations (up to 2000 mg/L) (Hartmann and Hoffmann, 1990), low maintenance (Hosomi et al., 1991) and low operating costs (Clark et al., 1978).

The treatment of landfill leachate and other high strength wastewaters using RBC's have been investigated by various authors. Since this thesis examines the treatment of a landfill leachate using an RBC, a significant portion of this literature review will be dedicated to previous work using RBC's.

Rotating biological contactors consist of a series of disks which are immersed approximately 40% in wastewater and rotated at a range of speeds. Weng and Molof (1974) and Wilson and Murphy (1980) showed that disk surface area loading is the key design variable for RBC's rather than hydraulic or volume loading rates.

For municipal wastewater treatment, the critical design parameter is BOD loading. The recommended loading rate is 15 g total $BOD/m^2/day$ at 13 °C. A disk area increase of 15% for each 3 °C decrease in temperature is recommended (Viessman and Hammer, 1985). For high ammonia wastewater such as leachate, ammonia loading is often the determining design factor . According to Forgie (1988c), if the $BOD:NH_3-N$ ratio is less than 1, treatment system design should be based on ammonia loading rather than organic loading. This "rule of thumb" applies to all aerobic systems.

RBC ammonia and organic loading rates from various sources are given in Table 7.

As shown in Table 7, ammonia loading rates for RBC are generally less than 2 $g/m^2/day$. Ehrig (1985) achieved greater than 95% nitrification at ammonia loading rates up to 17 $g/m^2/day$, but he recommended loading rates be restricted to less than 2 $g/m^2/day$ to avoid nitrite build-up and subsequent operating problems. The Pitsea full scale leachate treatment plant was designed based on an ammonia loading rate of 4.8 $g/m^2/day$ at 20-25 °C (Water Quality

International, 1987).

Wastewater	Influent Ammonia	Ammoni Lo	a BOD ading	COD	Speed	Т	Ammonia Removal	Source
	(mg/L)	(g/1	m²/day)	ł	(RPM) (°C)	(%)	
Sewage	40	0.8	NG	5.3	6.75	20	100	1
Leachate	2000	2.0	6	NG				2
Leachate	460	0.77	5	11.7	8-32	NG	59	3
Leachate Sludge	460	0.77	5	11.7	8-32	NG	99	3
Supernata: Sludge	nt 780	2.2	0.5	2.7	13	15	99	4
Supernata	nt 780	4.4	1.0	5.4	13	17	72	4
Synthetic	125	1.0	NG	NG	NG	10-3	0 95	5
AS Effluent	t 20	1.6	1.6	NG	NG	13	95	6
Synthetic	1.65	0.06	NG	6.1	10	NG	96	7
Synthetic	3.85	0.13	NG	13.4	10	NG	63.1	7
Synthetic	39.2	1.4	NG	23.3	10	NG	36.8	7
Leachate	7-46	0.5	6.0	10.9	6	>5	95	8
Leachate	NG	2.0	NG	NG	NG	NG	96	9
Leachate	30	1.0	1.1	16	NG	NG	95	10
Leachate	350	4.8	NG	NG	1.5	20-2	599	11
Sewage	30	1.6	NG	9.9	16	15	87	12
Sewage	22	1.1	NG	13.9	16	15	98	12
Leachate	154 1.2	-7.3 0.2	21-1.3	2.8-28.4	2.3	20	95	13

NG = Not given SOURCES:

- 1. Pretorius, 1971
- 2. Hartmann and Hoffmann, 1990. These values are design recommendations.
- 3. Hosomi et al., 1991. The first values are for a standard RBC. The second values are for a modified RBC containing an anaerobic biofilter in the same tank.
- 4. Lue-Hing et al., 1976 5. Masuda et al., 1991 6. Antonie, 1974

- 7. Ahn and Chang, 1991 8. Peddie and Atwater, 1985
- 9. Ehrig, 1985
- 10. Albers et al., 1986
- 11. Water Quality International, 1987
- 12. Pano and Middlebrooks, 1983
- 13. Spengel and Dzombak, 1991

Factors affecting nitrification rate include RBC speed, temperature, alkalinity availability, BOD, and toxicity.

Hosomi et al. (1991) investigated the effects of varying RBC rotational speed on nitrification rate over a range of 8-32 RPM. Japanese design standards are peripheral velocity of 15-20 m/min (28-38 RPM for the disks used). Hosomi et al.

(1991) found that in a standard RBC, nitrification rates increased with increasing rotational speed, but that at 32 rpm the effluent becomes white and turbid due to detached biofilms. Weng and Molof (1974) found that over the range of 10-42 RPM (peripheral velocity of 5-20 m/min), maximum nitrification was achieved at 42 rpm. Both Hosomi et al. (1991) and Weng and Molof (1974) link increased nitrification with increased tank dissolved oxygen.

Nitrification rate depends on temperature for all aerobic processes. Wilson and Murphy (1980) studied TKN removal from municipal wastewater over a temperature range of 12-18 °C, and used an Arrhenius temperature correlation for RBC's developed by Murphy et al. (1977) to normalize removal rates to 20 °C. The Arrhenius correlation is shown in Equation 6.

EQUATION 6: ARRHENIUS	EQUATION
$K_{\rm T} = K_{20} \Theta^{(\rm T-20)}$	
$K_{T} = TKN removal r$ $K_{20} = TKN removal r$ $\Theta = 1.09$	ate at T °C (g/m²/day) ate at 20 °C (g/m²/day)
SOURCE: Murphy, 1980	

Pano and Middlebrooks (1983) also found nitrification in RBC's of domestic wastewater followed the Arrhenius relationship with Θ = 1.10. They also found no nitrification occurred below 5 °C. Peddie and Atwater (1985) found no nitrification of landfill leachate occurred using an RBC at water temperatures less the 5 °C. Water Quality International (1987) reported that the designers of the full scale Pitsea leachate treatment system used landfill gas to heat leachate to 20-25 °C prior to treatment in an RBC.

During the nitrification process, hydrogen ions are produced, and therefore to maintain a constant pH, alkalinity or buffering is required. The theoretical alkalinity requirement of nitrification is 7.14 mg alkalinity as $CaCO_3$ per mg ammonia-N oxidized (Ehrig, 1985). Chen et al. (1989) found that, in an RBC, nitrification rate was independent of bulk alkalinity at 6.7 mg/L alkalinity as

 $CaCO_3$ per mg/L ammonia-N. Lue-Hing et al. (1976) found a net alkalinity as $CaCO_3$: ammonia-N ratio of 6:1 was required to maintain a stable pH. Chen et al. (1989) found that to maintain nitrification in an RBC, the bulk alkalinity needed to be kept above 50 mg/L as $CaCO_3$.

Various authors have suggested that the autotrophic microorganisms responsible for nitrification (<u>Nitrobacter</u> and <u>Nitrosomas</u>) have slower growth kinetics than the heterotrophic organisms responsible for organic decomposition (Barnes and Bliss, 1983; Antonie, 1974; Harremoes, 1982). In activated sludge systems, this relationship results in a longer sludge age for nitrification than for organic oxidation.

In an RBC plant, nitrification is generally considered to not proceed unless the organics have been removed enough so that the heterotrophs do not "choke out" the slower growing autotrophs. Therefore, organic removal primarily proceeds in the first stages of the RBC and nitrification primarily proceeds in the following stages (Weng and Molof, 1974).

Harremoes (1982) found that at dissolved oxygen concentrations of 3 mg/L a maximum filtered BOD of 20 mg/L was required for nitrification. Weng and Molof (1974) found nitrification did not proceed until the COD and BOD were removed to approximately 50 and 14 mg/L, and a minimum dissolved oxygen concentration of 2 mg/L was present.

Several authors have examined simultaneous nitrification and organic removal, and simultaneous nitrification/denitrification in biofilms.

Pano and Middlebrooks (1983) measured the COD and ammonia concentration in the effluent of each stage of a four stage RBC to develop kinetic relationships for simultaneous organic and ammonia removal. They noted that the disks in the first stage were covered in thick grey biofilm and subsequent disks were covered in a

thinner, smooth, brown biofilm. Weng and Molof (1974) found that grey biofilm indicates a heterotrophic community, and brown biofilm indicates a nitrifying community.

Pano and Middlebrooks (1983) found the majority of COD removal occurred in the first stage, and that COD removal followed Monod kinetics in the first stage and variable order kinetics in subsequent stages. Ammonia removal followed Monod kinetics, but was inhibited by organic loading. In the first stage, the level of inhibition varied linearly with organic loading.

Gonenc and Harremoes (1990) found that pure nitrification (no organic substrate present) is 1/2 order with respect to oxygen concentration (oxygen rate limiting). They found that in the presence of organic substrates, this relationship must be modified to account for the fraction of nitrifiers in the biofilm and the distribution of the nitrifiers through the biofilm. They concluded that to achieve nitrification, the ratio between soluble BOD and dissolved oxygen concentration must be less than five.

Masuda et al. (1991) investigated simultaneous nitrification/denitrification of landfill leachate in a covered RBC with controlled oxygen partial pressure. They found that heterotrophs, nitrifiers and denitrifiers were throughout the thickness of the biofilm. The activity of each of the groups was found to be independent of the biofilm location (surface, middle, bottom), but strongly dependent on oxygen partial pressure in the air over the RBC.

Masuda et al. (1991) found that with a C:N ratio of 3.5:1, ammonia loading rate of 1.0 g/m²/day, and an operating oxygen partial pressure of 0.05 atmospheres, up to 55% of the nitrogen could be removed through denitrification. At atmospheric pressure (oxygen partial pressure of 0.2 atmospheres), approximately 50% of the nitrogen was removed. At C:N of 1.5:1, up to 40% of the nitrogen was removed at oxygen partial pressure of 0.05 atmospheres, but only 5% was removed at 0.2 atmospheres.

Masuda et al. (1991) concluded that heterotrophs, nitrifiers and denitrifiers coexist throughout the biofilm. The activity of each group being strongly dependent on oxygen partial pressure. In an aerobic RBC, if microaerobic environments exist, nitrifiers and denitrifiers can work together to produce simultaneous nitrification/denitrification. The nitrogen removal efficiency was found to depend on the partial pressure of oxygen in the air phase, water temperature, hydraulic detention time, and the ratio of influent organics to ammonia.

Hosomi et al. (1991) compared a standard RBC to a RBC plus submerged anaerobic biofilter. They found approximately 40% nitrogen removal from leachate in a standard RBC at ammonia-N and BOD loading rates of 0.77 $g/m^2/day$ and 5.0 $g/m^2/day$, and up to 90% nitrogen removal in the modified RBC. No additional carbon source was required. They found BOD removals of greater than 95% in each system, but that COD removal was much higher in the modified RBC than standard RBC (85% compared to 65%).

In their paper, Hosomi et al. (1991) do not suggest a mechanism for increased COD and nitrogen removal, but simply state that the combination method of aerobic and anaerobic treatment was effective in reducing refractory organic compounds.

Chen et al. (1989) modelled the simultaneous removal of organics and nitrogen in an aerated, fully submerged, RBC. They found that, for simultaneous removal, a bulk dissolved oxygen concentration, which balanced the aerobic and anoxic parts of the biofilm, was required. For their study, the required bulk dissolved oxygen concentration was 2.5 mg/L. They found approximately 50% of total nitrogen could be removed from synthetic wastewater with influent ammonia-N concentration of 20-30 mg/L and COD of 50-200 mg/L.

Atwater and Bradshaw (1981) found up to 50% nitrogen removal in an RBC treating septic tank effluent with nitrate-N up to 20 mg/L. They attributed the nitrogen removal to denitrification.

Gupta et al. (1994) found up to 87.6% nitrogen removal in an RBC treating high strength synthetic nitrogen fertilizer wastewater. Nitrogen loading rates of 9.4 to 16.1 $g/m^2/day$ were investigated. For influent TKN ranging from 1000 to 2000 mg/L, TKN removal ranged from 96% to 60%. Gupta et al. (1994) attribute the nitrogen removal to simultaneous nitrification/denitrification by <u>Thiospaera</u> pantotropha, a heterotrophic nitrifier and aerobic nitrifier.

The results of the preceding experiments indicate that, under certain circumstances, nitrification and denitrification can occur simultaneously in a primarily aerobic fixed film system. This may be the result of the presence of anoxic regions within the biofilm or, as discussed by Gupta et al. (1994), due to bacteria capable of denitrifying under aerobic conditions.

Leachate constituents, which in excess, could inhibit the RBC nitrification process include metals, ammonia, organics or xenobiotics. Organics have been shown to inhibit nitrification in RBC's as previously discussed. Ammonia is anticipated to inhibit nitrification, but other authors have successfully nitrified high ammonia wastes.

Using an RBC, Lue-Hing et al. (1974) fully nitrified sewage sludge supernatant with ammonia-N concentration of 780 mg/L. Hartmann and Hoffmann (1990) suggest leachate with influent ammonia of up to 2,000 mg/L can be successfully nitrified using an RBC.

Rotating biological contactors are designed to remove ammonia and organics. They have also been shown to remove metals present in the leachate. Peddie and Atwater (1985) found metal removal rates shown in Table 8.

Metal*	Influent (mg/L)	Effluent (mg/L)	Removal (%)	
Mn	1.77	0.21	88.1	
Fe	18.6	3.2	82.8	
Zn	0.11	0.019	82.7	
Pb	0.0036	0.0016	55.6	
Cu	0.0019	0.0009	52.6	
Ni	0.00113	0.00091	19.5	
Cr	0.00068	0.00057	16.2	
Notes:	wag totalg			

The leachate described in Table 8 has lower metals concentration than the leachate described in Table 5: Aerated Lagoon Metal Removal, but the removal efficiencies are in the same range.

Potential improvements to the conventional RBC have been suggested in several papers.

One potential weakness of RBC's is that they tend to produce effluent with higher fine suspended solids than suspended growth systems due to the turbulent shearing action in the RBC tank (Watanabe et al., 1990). Watanabe et al. (1990) developed a two storey laboratory scale RBC in which the bottom one-half was used as a clarifier. They achieved a final suspended solids concentration of approximately 10 mg/L without a final clarifier.

Tanaka et al. (1991) found that the fine suspended solids produced by RBC's could not be effectively removed by gravity settling alone. They found that increasing RBC hydraulic retention time reduced the number of fine particles in the effluent, and that any small particles in the effluent could be most effectively removed using coagulation/flocculation.

Ahn and Chang (1991) compared a two storey RBC with the lower level acting as a
settling tank to a conventional RBC loaded at $6.1-23.3 \text{ g COD/m}^2/\text{day}$ and $0.06-1.4 \text{ g NH}_3-N/m^2/\text{day}$. They found soluble COD removal rates were similar for both units, ammonia removal was slightly higher in the two storey RBC, and suspended solids removal was much better in the two storey RBC producing effluent with 1.8-6.2 mg/L compared to 12.5-19.6 mg/L in the standard unit.

Nyhuis (1990) was able to achieve nitrification rates of 3.3 g $N/m^2/day$ in a full scale sewage treatment plant by filtering the wastewater prior to the RBC to separate organic matter removal and nitrification, and periodically reversing flow in the RBC to produce even growth throughout the stages.

Surampalli and Baumann (1989) improved both organic removal and nitrification in a full scale RBC plant by adding supplemental aeration to the RBC tank. The nitrification rate was increased from 20% to more than 50% for the same effluent COD value (40 mg/L).

Hosomi et al. (1991), treating landfill leachate, compared a standard RBC to a modified RBC, which incorporated an anaerobic biofilter into the tank under the RBC. The modification resulted in nitrogen removal, without organic substrate addition, of more than 70% (up to 88%) compared to 40% in the standard unit, and improved BOD and COD removal.

2.3.1.2 Anaerobic Treatment

Anaerobic leachate treatment methods include both fixed film and suspended growth methods. They potentially provide low maintenance, low energy, methods of organics removal from high strength leachate. Another advantage of anaerobic treatment is that methane is produced as a byproduct, which can be either used to improve reaction rates by heating the system or sold.

Forgie (1988c) suggests that leachate with a BOD:COD ratio greater than approximately 0.4 is amenable to anaerobic treatment. Since high BOD:COD ratios

are typically transient, existing during the acidic phase, conditions conducive to anaerobic treatment will generally be short term.

The effluent from an anaerobic treatment process will be similar to "older" leachate (low BOD:COD ratio), and require further treatment using aerobic or physical chemical treatment methods prior to discharge (Forgie, 1988c).

An alternative method of producing low BOD:COD leachate would be to recirculate leachate through the landfill, since the landfill is essentially a large anaerobic treatment vessel. Leachate recirculation is now accepted as a stabilization method by the United States Environmental Protection Agency (Federal Register, 1991). Leachate recirculation is discussed later in this review.

The results of various authors' experiments treating leachate anaerobically are shown in Table 9.

Treatment	BOD:	HRT I	Loading	Temp	Efficiency	Source
Method	COD	(days)	(kg COD/ m ³ /day)	(°C)	(% COD Rem.)	
Upflow Bed&Fil	lter 0.7	7.7	1.4	35	92.1	1
	0.7	2.7	21.8	35	67.9	1
Upflow Filter	0.7	4.0	1.45	21-25	95	2
-	0.7	2.0	2.89	21-25	68	2
	0.3	1.0	1.26	21-25	95	2
	0.3	0.5	3.14	21-25	60	2
	0.5	1.0	1.35	21-25	90	2
	0.5	0.5	2.66	21-25	88	2
Digester Tank	"old"	8.3	0.1	20&37	35	3
5	"voung"	15	0.29	20	46	3
	"voung"	15	0.29	37	51	3
	"voung"	4	1.1	20	29	3
	"voung"	4	1.1	37	47	3
Upflow Filter	0.7	1.0	1.8-4.0	22	70	4
SOURCES: 1 Chang, 198 2 Henry et a 3 Mendez et 4 Muthukrishr	9 1., 1987 al., 198 nan and <i>P</i>	9 Atwater	, 1985			

Table 9 shows that, at COD loading rates of approximately 1.3 kg COD/m³/day and at BOD:COD ratios of as low as 0.3, high COD removal can be achieved. Table 9 also shows that at high loading rates, temperature affects treatment efficiency. Anaerobic treatment does not effectively remove COD from old leachate.

Mendez et al. (1989) found that phosphorous addition was not necessary for anaerobic treatment, removal was not improved by recycle, and inhibition probably caused by heavy hydrocarbons resulted in a complete halt of the degradation process. They also found that addition of NaCl to 20 g/L produced significant inhibition of degradation activity.

Anaerobic treatment has been shown to be inhibited by unionized hydrogen sulphide concentrations of 110 and 350 mg/L as sulphur, which resulted from influent sulphate concentrations in excess of 625 mg/L as sulphur (Parkin et al., 1991).

A pH outside of 6 - 8 results in a sharp decrease in methane production in an anaerobic reactor (Speece et al., 1986).

Anaerobic organic oxidation has been shown to be strongly inhibited by free ammonia. Heinrichs et al. (1990) found that unionized ammonia concentrations of 200 mg/L totally inhibited methane production.

A second application of anaerobic reactors in leachate treatment is as an anoxic basin for denitrification. Dedhar and Mavinic (1985) achieved up to 100% denitrification by adding glucose to a predenitrifying suspended growth system (anoxic basin precedes aerobic basin). Ehrig (1985) found that a COD:N ratio of 5-6 was required to achieve full denitrification.

2.3.1.3 Biological Metal Removal

Dissolved heavy metals in leachate can be removed during anaerobic or aerobic treatment of the leachate.

In anaerobic treatment, sulphate in the leachate is generally reduced to sulphide by sulphate reducing bacteria. Heavy metal ions form insoluble compounds with the sulphide, which precipitate out as metal sulphides (Chang, 1989). According to Barnes et al. (1991), to achieve sulphate reduction and heavy metal precipitation, a redox potential of -100 mV (compared to -300 mV for methanogenisis), a reactor pH of 5 to 9 (optimum 7.5), and an available carbon substrate are required.

In aerobic processes, metals are oxidized and precipitated out (e.g. as metal hydroxides or oxides). Precipitation may result in clogging of media if using trickling filters (Henry, 1987). To avoid clogging of the aerators from precipitates of iron and calcium, suspended growth systems require coarse bubble aerators (Ehrig, 1985).

2.3.2 Physical/Chemical Treatment

2.3.2.1 Recycle/Recirculation

Leachate recycle involves collecting leachate and either above or below ground irrigating the landfill with the leachate. In the United States EPA Landfill Regulation (Subtitle D) 40 CFR Parts 257 and 258, effective October 9, 1993, if a landfill is equipped with a composite liner and a leachate collection system designed to maintain a maximum hydraulic head of 30 cm on the liner, leachate recirculation is acceptable (Federal Register, 1991). The goals of leachate recirculation are to speed up the stabilization of the landfill through the introduction of additional moisture, improve leachate quality (reduce BOD and COD), and increase the quality and quantity of methane production (Federal Resister, 1991).

Pohland et al. (1985) note that leachate recirculation may reduce the stabilization period from 15-20 years to 2-3 years. The EPA notes (Federal Register, 1991) that under humid conditions, leachate recirculation is not

recommended because sufficient moisture is already available and additional moisture will increase total leachate production and may increase the hydraulic head acting on the liner system.

Birkbeck and Tomlins (1985) note that potential problems associated with leachate recirculation include clogging of irrigation equipment and landfill surface with precipitates of carbonates and iron oxides, odours, and vegetation kill from air spraying leachate.

Pohland et al. (1985) found that even following recirculation, leachate still contains significant organic and inorganic contaminants, and in most cases will require additional treatment prior to disposal. Additionally, since recirculation involves using the landfill as an anaerobic bioreactor and ammonia is not removed through anaerobic decomposition (except through assimilation), recirculation should not significantly reduce leachate ammonia concentration.

Therefore, under certain circumstances (low moisture), recirculation could be used to reduce organic constituents of leachate, but additional treatment will be required prior to discharging the leachate.

2.3.2.2 Land Application

Land application involves discharging leachate to land other than the landfill and relying on evaporation plus the assimilative capacity of the land to renovate the leachate. According to Forgie (1988b), land application of leachate has not been widely investigated. Land application has been successfully used in Great Britain as a treatment system, but in North America its use is restricted to polishing previously treated leachate (Henry, 1985).

A full scale leachate treatment plant in Sarnia, Ontario, discharges treated effluent to wetlands prior to discharge to a creek. The wetlands provide approximately one month of retention time to utilize residual nitrogen and phosphorus (Environmental Science and Engineering, 1991).

2.3.2.3 Air Stripping

The equilibrium between ammonium ion concentration and dissolved ammonia gas in water depends on pH and temperature (lower temperature shifts balance towards ammonium ions) (Viessman and Hammer, 1985). At 25 °C, the equilibrium relationship between ammonium ion concentration and free ammonia concentration is shown in Equation 7.

EQUATION 7:	: AMMONIA/AMMONIUM EQUILIBRIUM EQUATION	
[NH ₃] [H ⁺ [NH ₄ ⁺]	$\frac{1}{1} = 10^{-9.245}$	
T = 25°C		
SOURCE: Vie	essman and Hammer, 1985	_

By raising the pH of wastewater to between 10.8 - 11.5 with NaOH or Ca(OH)₂ and subsequently aerating the wastewater, ammonia gas can be stripped from the water to the air, thus removing the ammonia from the wastewater (Forgie, 1988b).

Keenan et al. (1984) took advantage of the high pH following chemical precipitation of metals using lime to air strip ammonia. The ammonia was stripped in a 1.74 day hydraulic retention time aerated lagoon preceding activated sludge treatment. Keenan et al. (1984) found that influent ammonia concentration of up to 1000 mg/L inhibited the activated sludge microorganisms, but following 50% reduction through air stripping, the ammonia was no longer inhibitory.

The disadvantages of air stripping include large chemical requirements due to the well buffered nature of leachate, chemical costs (lime costs represented 30% of total costs for Keenan et al. (1984)), and sludge disposal requirements.

2.3.2.4 Adsorption

Treatment by adsorption generally involves using either granulated activated

carbon (GAC) or powdered activated carbon (PAC). GAC is used in a filter or column and PAC is added to the leachate as a liquid slurry (Forgie, 1988b). Activated carbon adsorption is most effective at removal of high molecular weight organics, which predominate in either biologically treated leachate or old leachate. Therefore, activated carbon adsorption is most effective as a polishing technique to remove recalcitrant organics (Forgie, 1988b). COD removal on biological treatment plant effluents of 85% are possible (Mendez et al., 1989).

Various authors have shown that activated carbon is also effective at metals removal under certain circumstances (e.g. in the presence of organic complexing agents) (Bhattacharyya and Cheng, 1987; Corapcioglu and Huang, 1987), but coagulation/precipitation techniques are more commonly used for metals removal (Enzminger et al., 1987).

Another potential adsorption method is metals removal by adsorption to peat in a column. Available literature on adsorption of metals with peat is laboratory scale only (McLellan and Rock, 1988).

Corbett (1975) used laboratory scale peat columns to treat landfill leachate. Corbett (1975) achieved 59% metal removal at pH 7.1, and found that a dry weight of approximately 159 kilograms of peat was required per 1,000 litres of leachate. Corbett (1975) also found that "resting" the peat for one month, following treatment, was not sufficient to reuse the peat, and that desorption of metals occurred if water was percolated through the peat.

2.3.2.5 Chemical Coagulation/Precipitation

Chemical coagulation involves creating flocs by adding multivalent metal ions (e.g. Ca^{+2} , Fe^{+3} , Al^{+3}), which settle and trap suspended solids and colloidal matter. Precipitation involves adding agents, which chemically bind with dissolved ions, creating insoluble precipitates. Lime is the most common

precipitation agent (Lema et al, 1988).

Ehrig (1989b) found that flocculation was not a suitable treatment method for high strength leachate, but found iron salts could be used effectively to remove COD (50% removal) from old leachate. Forgie (1988c) states that coagulation can be used to remove residual suspended solids and colour. Lime precipitation can be used as a pretreatment step for metals removal prior to biological treatment processes to avoid potential inhibition of the treatment process (Forgie, 1988c). Keenan et al. (1984) used lime addition to remove metals prior to biological treatment. The resultant high pH was utilized through air stripping to remove ammonia. pH readjustment is required prior to biological treatment (Environmental Science and Engineering, 1991).

In general, physical/chemical processes cannot be considered substitutes for biological treatment methods, but rather should be considered a component of a combined biological/physical chemical treatment train designed to meet increasingly stringent discharge standards. Other methods not discussed in this review include chemical oxidation (e.g. chlorination), membrane techniques such as reverse osmosis and ultra-filtration, and ion exchange.

2.4 LITERATURE REVIEW CONCLUSIONS AND EXPERIMENTAL OBJECTIVES

The leachate used in this experiment was a methanogenic stage leachate, with very high ammonia concentration and relatively high COD concentration. Based on the review of available literature, the system design was based on ammonia removal rather than organics removal.

To remove ammonia, either a suspended growth or a fixed film system should be used. It appears that influent ammonia-N concentrations exceeding 2,000 mg/L completely inhibit the activated sludge process (Azevedo, 1993), although the inhibition observed by Azevedo may have been due to insufficient process aeration

(Dr. Donald Mavinic, University of British Columbia, Personal communication, March 1994). Forgie (1988a) mentions several other potential problems with suspended growth systems.

RBC's appear to nitrify leachate up to ammonia-N concentrations of 2000 mg/L at loading rates up to 2 g/m²/day (Hartmann and Hoffmann, 1990). The additional advantages of RBC include low operating costs (Clark et al., 1978), low maintenance costs (Hosomi et al., 1991), and good response to shock loadings (Peddie and Atwater, 1985). It also appears that RBC's may provide full nitrogen removal under certain conditions (Masuda et al., 1991; Hosomi et al., 1991; Atwater and Bradshaw, 1981; Gupta et al., 1994).

Unless the leachate being treated has a very high COD:ammonia-N ratio, the most effective method to completely remove nitrogen from the leachate appears to be using biological denitrification. This can be achieved in an anoxic basin either preceding or following the aerobic system. Supplemental carbon addition to provide an energy source for denitrification is common, but not necessary if sufficient carbon is available in the leachate.

Based on the review of available literature, the system selected for this experiment was an RBC for nitrification and a predenitrifying anaerobic filter without supplemental carbon addition. It was postulated that nitrification would occur in the RBC and denitrification would occur in the anaerobic filter using the COD in the leachate as an energy source. Organics and metal removal were anticipated to occur simultaneously.

The objectives of this research were to determine the loading rates at which nitrification occurs in an RBC at high influent ammonia concentration, and to determine whether a predenitrifying anaerobic filter could be used on the leachate without supplemental carbon addition. Secondary objectives included removing organics and metals.

3. SITE DESCRIPTION

Leachate for the experiment was collected from Da Liao Landfill near Kaohsiung, located in south-western Taiwan, Republic of China. The landfill serves two small communities on the outskirts of Kaohsiung. A map of Taiwan showing Kaohsiung is provided in Figure 1.

According to the Kaohsiung County Environmental Protection Branch, the Landfill receives refuse from approximately 400,000 people (Mr. Zhang, Personal communication, June 27, 1993). According to Mr. Zhang, the Landfill accepts 450,000 tonne per year of refuse, but the Landfill has no weigh scales. Based on the population estimate provided by Mr. Zhang, and Kaohsiung City's per capita waste generation of 0.4 tonnes/cap/year (Kaohsiung Municipal Government, 1992), a more conservative estimate of the annual refuse disposed of at the landfill is 160,000 tonnes. From observations of truck traffic at the site, 160,000 tonnes per year is probably more accurate.

A schematic drawing of the Landfill is provided in Figure 2, and a site plan is provided in Appendix 1. Based on the plan in Appendix 1, the total area of the Landfill is approximately eight hectares.

The Landfill began operation sometime around 1987.

The landfill is underlaid with a geotextile membrane, and a concrete retaining wall is located at the eastern end of the landfill at which point leachate is collected in a 27 cubic metre sedimentation basin. The leachate flow is measured continuously using a Parshall Flume. The leachate is pumped to an on-site facility treating leachate plus septage. Septage is treated at the facility because, in Taiwan, all buildings have sedimentation/septic tanks.



FIGURE 1: KAOHSIUNG LOCATION PLAN



FIGURE 2: LANDFILL SITE PLAN

The treatment plant includes an anaerobic treatment tank, an oxidation ditch, and a final clarifier. Chemicals including powdered activated carbon, polymer and sodium hydroxide are added to the oxidation ditch. Neither plant performance data nor facility sizing data are available.

Mean monthly precipitation and mean daily temperature data for Kaohsiung for the years 1934-1989 are provided in Table 10. Table 10 also includes precipitation data for the period the experiment was conducted (October 1991 to May 1992). Daily precipitation data for Kaohsiung are provided in Appendix 3.

Month	Precipitation (1934-1989 mea (mm)	n)	Precipitation 1991 & 1992 (mm)	Mean Daily Temp. (°C)
January	15	1992	33.0	18.6
February	18		59.9	19.3
March	38		127.8	21.8
April	55		262.2	24.7
May	171		146.7	27.2
June	412			27.9
July	407			28.4
August	390			28.0
September	166			27.6
October	41	1991	35.1	25.9
November	20		3.4	23.1
December	13		34.6	20.0
SOURCE: Pe	rsonal communication, Ta	iwan C	entral Weather	Agency, 1992

TABLE 10: PRECIPITATION AND TEMPERATURE DATA FOR KAOHSIUNG

An approximate water balance for each month is shown in Table 11. The leachate production results in Table 11 are a summary of the data provided in Appendix 4. Table 11 shows that, on average, approximately 105% of precipitation was collected as leachate. Since some of the precipitation would have evaporated, or been stored in the incoming refuse, it is probable that the actual area from which leachate was collected exceeded eight hectares.

Month	Precipitation Volume (m ³) 1	Leachate Production (m ³) 2	Precipitation/ Leachate Prod.
November	270	4,020	0.07
December	2,770	5,580	0.50
January	2,640	5,210	0.51
February	4,790	4,230	1.1
March	10,220	4,780	2.1
April	20,980	19,580	1.1
Total	41,670	43,400	0.96

TABLE 11: DA LIAO LANDFILL WATER BALANCE

1. Based on 1991 and 1992 precipitation volumes from Table 10 and landfill area of 8 hectares.

2. From weakly leachate production volumes in Appendix 4.

4. EXPERIMENTAL DESIGN

Approximately one year prior to arriving in Taiwan, the author began corresponding with Professor Lin Cheng-Fang of the National Taiwan University in Taipei to develop a research program.

During the summer of 1991, Professor Lin informed the author he and a colleague from National Sun Yat-Sen University in Kaohsiung (Professor Yang Lei) would be investigating the treatment of landfill leachate using the treatment train shown in Figure 3. The treatment train consisted of denitrification in an anaerobic filter, nitrification and organics removal in an activated sludge treatment vessel and clarification in a final clarifier.

Following an initial literature review of potential treatment methods, the author proposed to use a similar treatment train to the one proposed by Professor Lin, but to substitute an RBC for the activated sludge system.



The leachate treatment system investigated in this project is shown schematically in Figure 4. The system consisted of:

- 1) A peristaltic feed pump
- A 20 litre leachate feed container with 0.5 litre increments marked on the side of the container
- 3) A 30 litre anaerobic filter made out of plexiglass and filled with 2.5 cm diameter perforated balls called tri-packs
- 4) A 10 litre RBC with 3 stages, and a total of 75 disks. The disks were 20 cm diameter, 2 mm wide, and spaced on 9.2 mm centres. The total disk area was 4.7 m^2 , and a minimum clearance of 0.5 cm was allowed between the disks and the tank. The RBC was rotated by an electric motor at 10-25 RPM
- 5) A 10 litre final clarifier
- 6) A recirculation peristaltic pump
- 7) System piping was Nalgene 8000 (6.4 mm internal diameter, 1.4 mm wall thickness)



Based on the anticipated high ammonia concentration of the leachate and conversations with Professors Atwater, Lin and Yang, ammonia loading was determined to be the critical design factor for the RBC. Subsequently, based on loading rates shown in Table 7, RBC ammonia-N loading rates of 1, 3 and 5 $g/m^2/day$ were selected for investigation. Based on an estimated ammonia-N concentration of 1,700 mg/L (Chang, 1989), design flow rates of 2.8, 8.3 and 14 litres per day were selected.

The design of the anaerobic filter was based on an estimated COD of 5,000 mg/L, and minimum design loading rate of approximately 0.5 kg/m³/day. These loading rates are based on previous work provided in Table 9. From the design loading rate and design flows from the RBC calculations, a design volume of 30 litres was calculated.

The design of the clarifier was based on achieving a minimum detention time of 2 hours (Viessman and Hammer, 1985) using up to 500% recycle (Jim Atwater, Personal communication, October 1991). Based on maximum design influent and recycle flow, the design clarifier volume was 10 litres.

To achieve denitrification, recycle was from the clarifier to the inlet of the filter. To avoid clogging of the filter, effluent recycle was selected rather than sludge recycle. No facilities were included for sludge removal because the anticipated sludge volume was small, thus allowing manual removal.

5. ANALYTICAL PARAMETERS AND METHODS

Leachate and effluent from the anaerobic filter and the RBC (collected from the final clarifier) were analyzed for a variety of parameters.

The analytical parameters and their monitoring frequency are provided in Table 12.

5.1 BOD (BIOCHEMICAL OXYGEN DEMAND)

BOD analysis was initiated within two hours of sample collection and followed the procedure described in Standard Methods (American Public Health Association et al., 1985). All samples (including blanks) were seeded with effluent from the activated sludge system or RBC system, and samples were not filtered prior to analysis. No nitrification inhibitor was added. Dissolved oxygen was measured using a Syland-Temp-O₂-Mat 4000 L meter. No long-term BOD tests were conducted to determine whether the seed was sufficiently acclimated or whether a nitrification inhibitor was required.

Leachate Weekly BOD (mg/L) COD (mg/L) Ammonia-N (mg/L) Suspended Solids (mg/L) pН Metals (dissolved, mg/L) Cd , Cr, Cu, Fe, Pb, Mn, Ni, Zn $\begin{array}{c} NO_{3}^{-} (mg/L) \\ NO_{2}^{-} (mg/L) \\ Cl^{2} (mg/L) \\ SO_{4}^{-2} (mg/L) \\ SO_{4}^{-2} (mg/L) \end{array}$ Flow (m³/hour) Periodically TKN (mg/L) Total Alkalinity (to pH 4.5) (mg/L as CaCO₃) Anaerobic Filter & RBC Effluent Minimum Twice Per Week Flow (L/day) Temp (RBC and Filter) (°C) RBC Speed (RPM) Weekly BOD (mg/L) COD (mg/L) Ammonia-N (mg/L) pН Metals (dissolved, mg/L) Cd , Cr, Cu, Fe, Pb, Mn, Ni, Zn $\begin{array}{c} NO_{3}^{-} (mg/L) \\ NO_{2}^{-} (mg/L) \\ C1^{-} (mg/L) \end{array}$ Cl^{2} (mg/L) SO_{4}^{-2} (mg/L) Periodically Suspended Solids (mg/L)

TABLE 12: ANALYTICAL PARAMETERS AND MONITORING FREQUENCY

5.2 COD (CHEMICAL OXYGEN DEMAND)

COD analysis was initiated within two hours of sample collection, and followed the $K_2Cr_2O_7$ titration procedure described in Standard Methods (American Public Health Association et al., 1985). Since nitrites are generally assumed to be in small quantities (American Public Health Association et al., 1985) no chemical addition to account for nitrite interference was included. Samples were not filtered prior to analysis.

To calculate RBC effluent COD, measured COD was adjusted to account for nitrite present in the effluent as per Standard Methods (American Public Health

Association et al., 1985).

5.3 AMMONIA

Ammonia was measured within four hours of sample collection. Initially a Schott CG 840 meter with an ammonia membrane probe was used according to Standard Methods (American Public Health Association et al., 1985), but from January 15 to February 15, 1992, replacement membranes were not available. Therefore, an ion chromatograph (IC) (TOA ICA - 5000) with an SIC Chromatocorder 12 strip recorder was used.

Ammonia probe samples were not filtered, but IC samples were filtered with a 0.4 μ m filter to avoid clogging the column.

A sample set of data plus calculation methodology for the ammonia probe is provided in Appendix 2.

5.4 pH

pH was calculated using a Schott CG 840 pH meter. Samples were analyzed within two hours of collection. Samples were not filtered prior to analysis.

5.5 METALS

Dissolved metals (cadmium, chromium, copper, iron, lead, nickel, manganese and zinc) were measured following vacuum filtration with 0.4 μ m filter paper. Samples were directly measured using premixed standards and an 0.05 N HCl blank. A Hitachi Z-8000 Atomic Absorption Mass Spectrometer using flame atomizer, C₂H₂ fuel and air as an oxidant was used. 5.6 FLOW

Leachate was added to the 20 litre feed vessel each one to two days. The mean retention time in the feed vessel was maintained at approximately three days. The effect of storage on leachate quality is unknown since a comparison between fresh and stored leachate was not made.

The leachate feed vessel was marked with 0.5 litre increments. Influent flow was determined by recording the change in leachate feed volume over each one to two day period, and subsequently calculating mean daily influent flow for the period.

Recycle flow was calculated by measuring the system influent and filter effluent flows over a 15 minute period. The difference equalled the recycle flow.

5.7 ANIONS $(NO_3^-, NO_2^-, SO_4^{-2}, Cl^-)$

Anions including nitrate, nitrite, sulphate and chloride were measured in the leachate, filter effluent and RBC effluent using a TOA ICA-5000 ion chromatograph (IC) with a SIC Chromatocorder 12 strip recorder. Samples were filtered, prior to analysis, using a 0.4 μ m filter to avoid clogging the column.

5.8 RBC SPEED

RBC speed was measured by counting the number of disk revolutions in a minute.

5.9 SUSPENDED SOLIDS

Suspended Solids were measured following vacuum filtration with a Whatman 42 filter. Samples were dried at 105 $^{\circ}$ C overnight.

5.10 TKN (Total Kjeldahl Nitrogen)

TKN was analyzed on unfiltered leachate samples using the method described in Standard Methods (American Public Health Association et al., 1985).

5.11 TOTAL ALKALINITY (titration to pH 4.5)

Total alkalinity was measured titrimetrically to pH 4.5 using the procedures described in Standard Methods (American Public Health Association et al., 1985), and a Schott CG 840 pH meter to determine the titration end point.

6. RESULTS

6.1 LEACHATE, FILTER EFFLUENT AND RBC EFFLUENT CHARACTERISTICS

The mean, maximum and minimum parameter values for the leachate are provided in Table 13. Data are provided in Appendix 4. The results in Table 13 summarize the data from the analysis of a total of 21 samples taken weekly over a six month period. Leachate for the experiment was collected on a weekly basis in 20 litre containers, and stored at 4 °C prior to being dispensed into the leachate feed vessel.

Parameter	Mean (1			
pH (units)	8.1	7.8	8.8	
Ammonia-N	2140	1260	2700	
Suspended Solids	220	75	375	
BOD	705	465	1270	
COD	5040	3205	8420	
Total Alkalinity	10820	10510	11100	
TKN	2920	2470	3640	
Cd *	0.01	0.01	0.02	
Cr	3.05	1.91	4.10	
Cu	0.13	0.08	0.19	
Fe	7.30	5.34	14.65	
Mn	0.77	0.41	1.70	
Ni	0.22	0.19	0.27	
Pb	0.26	0.02	0.39	
Zn	1.01	0.49	1.29	
Cl	2450	1960	2860	
SO_4^{-2}	35	0.0	290	
NO3	0.9	0.0	10.0	
NOŽ	0.0	0.0	0.0	

TABLE 13: LEACHATE CHARACTERISTICS

Table 13 shows that the BOD:COD ratio of the leachate was approximately 0.1, which is indicative of a methanogenic phase leachate. The ammonia concentration of the leachate was very high. No higher ammonia concentrations for landfill leachate have been found by the author in the literature. Comparing the metals concentrations to the expected methanogenic leachate composition shown in Table 2, shows that cadmium, copper, iron, manganese and nickel are in the expected range, zinc is slightly higher than expected (a factor of two) and chromium is

particularly high (factor of 10). Given the amount of heavy industry in Taiwan and particularly in Kaohsiung, fairly high metals levels were anticipated.

Mean, maximum and minimum parameter values for the anaerobic filter and RBC effluent are provided in Table 14. Detailed discussion of the results is provided in subsequent sections.

	FILTER	EFFLUENT		RBC EF	FLUENT	
Parameter	Mean	Minimum	Maximum (mg/L unless	Mean stated)	Minimum	Maximum
pH (units)	8.4	8.0	8.6	8.5	7.2	9.3
Ammonia-N	1690	1040	2600	220	0.0	555
Suspended Solids	170	65	290	170	64	280
BOD	415	130	740	63	26	140
COD	4480	2670	7245	2660	1020	3790
Total Alkalinity	8150	6510	10980	3500	2630	4490
Cd *	0.01	0.01	0.02	0.01	0.01	0.01
Cr	3.17	2.42	4.09	3.41	2.52	4.93
Cu	0.09	0.03	0.14	0.13	0.06	0.24
Fe	4.31	2.20	8.15	4.59	2.29	5.94
Mn	0.43	0.24	0.82	0.32	0.12	0.52
Ni	0.25	0.17	0.35	0.27	0.18	0.33
Pb	0.16	0.06	0.28	0.22	0.03	0.92
Zn	0.63	0.40	0.89	0.85	0.52	1.32
C1 ⁻	2510	1940	2920	2740	2100	3050
SO_{A}^{-2}	30	0.0	260	75	0.0	220
NO	0.0	0.0	0.0	30	0.0	150
NO ₂ ⁻	0.0	0.0	0.0	550	20	890

6.2 FLOW

During the experiment, a total of five system flow regimes were investigated over a period of approximately one month per flow regime. The influent and recycle flow during each period are shown in Figure 5.

The maximum mean flow was 6.6 L/day with 90% recycle. The maximum recycle rate was 220% at mean influent flow of 6.1 L/day. The experimental design called for a maximum flow of 14 L/day and 500% recycle, but full nitrification could not be



achieved at the maximum flows used. Therefore, higher flows were not investigated. Daily flow data are provided in Appendix 4.

6.3 MASS BALANCE

As previously discussed, to allow for a mass balance to be calculated across the system, parameters were measured in each of the leachate, anaerobic filter effluent, and RBC effluent. The total mass of any parameter in each system component can be calculated using the relationships shown in Equation 8.

EQUATION 8: MASS BALANCE CALCULATIONS

System In (g/day) Filter In (g/day) Filter Out (g/day) RBC Out (g/day) System Out (g/day)		I*Ci I*Ci + R*Ce (I + R)*Cf (I + R)*Ce I*Ce
Filter Removal (g/day) RBC Removal (g/day) System Removal (g/day)	=	I*Ci + R*Ce - (I+R)*Cf (I+R)*Cf - (I+R)*Ce I*(Ci-Ce)
I = Influent Flo Ci = Influent Co R = Recycle Flo Ce = Effluent Co Cf = Filter Effl	ow (L/d ncentr w (L/d ncentr uent C	day) ation (g/L) ay) ation (g/L) oncentration (g/L)

Parameter removal in the anaerobic filter for COD, nitrogen and iron are shown in Figures 6, 7 and 8. Example mass balance calculations from data in Appendices 5, 6 and 7 are shown in Example 1.

EXAMPLE 1: MASS BALANCE SAMPLE CALCULATIONS

COD					
No Recycle (Day 2) I = 2.7 I R = 0 L/c Ci = 5.06 Cf = 2.67 Ce = 2.13	27) L/day day g/L g/L g/L		·		
Filter Removal	=	I*C:	i + R*Ce	ə —	(I+R)*Cf
	=	6.4	5 g/day	=	478
RBC Removal	=	(I+H	R) *Cf -	(I+	·R)*Ce
	=	1.5	g/day	_	11%
System Removal	=	I*((Ci-Ce)		
	=	7.9	g/day	=	58%
Recycle (Day 34) I = 6.2 I	_/day				
R = 13.4	L/day				
Ci = 3.82	g/L ¯				
Cf = 2.97	g/L				
Ce = 2.36	g/L				
Filter Removal	=	-2.9	g/day	=	-12%
RBC Removal	=	12.0	g/day	=	51%
System Removal	=	9.1	g/day	=	39%

Nitrogen: No Recycle (Day 27) 2.7 L/day I = R = 0 Ci = 1.630 g/L Cf= 1.595 g/L 0.054 g/l Ce = $0.095 \, g/day =$ 28 Filter Removal = = 95% RBC Removal 4.16 g/day 4.26 g/day97% System Removal = Recycle (Day 55) 6.4 L/day Ι = = 13.4 L/day R 2.160 g/L = Ci Cf = 1.040 g/L Ce = 0.452 g/L (Sum of Ammonia-N, Nitrate and Nitrite) -5% Filter Removal -0.71 g/day= = 11.64 g/day= 84% RBC Removal = 10.93 g/day System Removal = 79% = Metals No Recycle (Iron, Day 27) 3.0 L/day Ι = R = 0 Ci = 0.00640 g/L Cf 0.00291 g/L = 0.00229 g/L Ce = Filter Removal = 0.0105 g/day= 55% = 0.0019 g/day 10% RBC Removal = 0.0123 g/day System Removal = = 64% Recycle (Iron, Day 48) 6.0 L/day Ι = = 13.4 L/day R 0.00644 g/L Ci = 0.00477 g/L Cf = = 0.00572 g/L Ce 0.0228 g/day59.0% Filter Removal = -0.0184 g/dayRBC Removal = = -47.6% 0.0044 g/day11.4% System Removal = =

Figures 6, 7 and 8 show that parameter removal across the anaerobic filter varied widely both over time and within individual flow periods. The large variation seems due to insufficient system stabilization time between changes in flow regimes. The total system volume was 50 litres, and the generally accepted requirement to ensure steady state conditions are achieved is three hydraulic retention times. Therefore, for system flows of 2.7 and 6.0 L/day, stabilization times of 55 and 25 days would have been required.

EXAMPLE 1 CONTINUED: MASS BALANCE SAMPLE CALCULATIONS







In order to investigate as many flow loading regimes as possible, flow regimes were changed as soon as system effluent ammonia stabilized. Since ammonia was removed primarily in the RBC (which was only 10 litres) it responded rapidly to changes in influent flow. A more rigorous approach would have been to wait a full three hydraulic detention times prior to implementing flow regime changes.

Since parameter removal in the anaerobic filter and RBC varied widely throughout the experiment, parameter removal in each system component cannot be accurately calculated. Estimates of parameter removal for the anaerobic filter and RBC are provided, based on values from flow period three (See Figure 5), which was the longest flow period (60 days).

Overall system removal results are provided for each parameter for each flow period.

6.4 BOD and COD REMOVAL

According to Forgie (1988c), biological treatment is inappropriate for organics removal for leachate with a BOD:COD ratio of less than approximately 0.1, due to the recalcitrant properties of the existing organic material. The mean BOD:COD ratio of the leachate from Table 13 is 0.14. Therefore, although some degradable carbon was present in the leachate, the COD removal was anticipated to be low. Table 15 shows system BOD and COD removal from data provided in Appendix 5.

Day	0-27	28-55	56-111	112-146	147-163	
Flow (L/day)	2.8	6.1	6.0	6.6	3.6	
Recycle (L/day)	0.0	13.4	0.0	6.0	6.0	
BOD Results						Mean *
Influent BOD (mg/L)	465	670	620	920	1270	705
Effluent BOD (mg/L)	26	47	64	85	57	63
BOD Removal (%)	95	93	90	91	96	92
COD Results						
Influent COD (mg/L)	4100	4590	5160	7220	4898	5040
Effluent COD (mg/L)	2140	2630	2750	2960	2502	2660
COD Removal (%)	48	43	47	59	49	49
COD:BOD Removal	4.5	3.1	4.3	5.1	2.0	3.7

The results in Table 15 show period mean BOD removal ranged from 90-96%, with an overall mean of 92%. Period mean COD removal ranged from 43-59%, with an overall mean of 49%. Period mean COD:BOD removal ranged from 2:1 to 5.1:1, with a mean of 3.7:1.

Given that BOD is generally assumed to represent the biologically degradable portion of COD, the COD:BOD removal values appear high. In other landfill leachate treatment experiments, Hosomi et al. (1991) found a COD:BOD removal of 2.3:1, Albers et al. (1986) found a COD:BOD removal of 7.5:1, Spengel and Dzombak found a COD:BOD removal ratio of 6.6:1. Based on these results, the COD:BOD removal ratios found in this experiment do not seem unreasonable.

These results seem to indicate that either the COD test was overestimating the organic content of the leachate or the BOD test was underestimating the degradable component of the organic matter.

To determine if the BOD test was underestimating the degradable organic content,

long term BOD tests would be required to estimate the ultimate BOD, and determine if the seed was sufficiently acclimated (Dr. David Forgie, Associated Engineering Ltd., Personal communication, April 1994).

According to Standard Methods (American Public Health Association et al., 1985), chloride content exceeding 2000 mg/L may result in interference of the COD measurement. From Table 13, the mean chloride concentration of the leachate was 2450 mg/L. Whether the high chloride concentrations interfered with the measurement of COD is unknown.

Based on the data provided in Appendix 5, both BOD and COD removal were primarily in the RBC.

Figures 9 and 10 show leachate and RBC effluent BOD and COD concentrations over time.

Figure 11 shows system BOD and COD removal with increasing flow. Figure 11 shows that both BOD and COD removal were independent of system flow. This result was anticipated because the system was only lightly organically loaded at all flow rates.






6.5 NITROGEN

Given the low BOD:COD ratio and the high ammonia concentration of the leachate, ammonia and nitrogen removal were the prime goals of the study.

To measure nitrogen transformations within the system, ammonia was measured throughout the duration of the experiment. Nitrite and nitrate were measured commencing in mid-January 1992 due to a delay in receiving an anion column for the ion chromatograph. TKN (total kjeldahl nitrogen) was measured in the leachate mid-December to mid-January, to develop a correlation between ammonia and TKN data. Correlation calculations and leachate, filter effluent and RBC effluent nitrogen data are provided in Appendix 6.

Nitrogen removal across the system is based on influent ammonia-N concentration rather than TKN, because TKN was measured in the leachate only, and was only measured on a few occasions (See Appendix 6). Additionally, ammonia removal rates are more commonly referred to in the literature than TKN removal. The mean TKN:ammonia-N ratio was 1.4:1, which equals the ratio found by Dedhar (1985) in his leachate treatment investigation.

Table 16 shows nitrogen forms and removal efficiencies in the system at varying flows.

Figure 12 shows leachate and RBC effluent ammonia concentration.

TABLE 16: NITROGEN AND AMMC	NIA REMOV	AL				
Day	0-27	28-55	56-111	112-146	147-163	Mean 1
Flow (L/day) Recycle (L/day)	2.8 0.0	6.1 13.4	6.0 0.0	6.6 6.0	3.6 6.0	
Influent NH_3-N (mg/L) RBC NH_3-N Loading (g/m ² /d) 2 Effluent NH_3-N (mg/L) Effluent $NO_2^{-}-N$ (mg/L) Effluent $NO_3^{-}-N$ (mg/L)	1856 2 0.9 59 	2235 2.5 454 71 3	2244 2.0 173 590 3	2254 2.9 251 834 70	1957 1.3 52 495 90	2142 222 552 32
Overall NH ₃ -N Removal (%) Overall Total-N Removal (%)	97	80 76	92 66	89 49	97 67	90 66

NOTES:

1. Mean values are mean of all samples.

2. RBC loading is estimated based on 18% ammonia removal in the anaerobic filter.

Figure 13 shows overall ammonia removal efficiency at varying RBC mass loading rates. These rates are estimated RBC loading rates based on a mean ammonia removal of 18% in the anaerobic filter. This removal rate is based on data from flow period 3. Azevedo (1993) found anoxic ammonia removal of 8%. Carley (1988) and Mavinic and Randall (1990) found anoxic ammonia removals of 6% and 10%.

Figure 13 shows decreasing removal with increasing loading. The linear regressions statistics equal: Slope = $-5.9 %/(g/m^2/day)$, $r^2 = 0.39$, indicating a non-linear relationship between removal and loading.

Ammonia removal throughout the study exceeded 80%, but at loadings greater than 1.5 g ammonia- $N/m^2/day$, mean effluent ammonia exceeded 100 mg/L, which was considered too high.

Figure 14 shows ammonia mass removal with increasing loading. Figure 14 shows a linear relationship, indicating that, on a mass loading basis, the process was not inhibited.







Although Figure 14 shows that, on a mass loading basis, ammonia removal was not inhibited, full ammonia removal could only be achieved up to an RBC loading rate of 1.5 $g/m^2/day$. Given that Spengel and Dzombak achieved 95% ammonia removal at an RBC loading rate of up to 7.3 $g/m^2/day$ at influent ammonia-N of 154 mg/L and 20 °C, the results of this experiment show inhibition of nitrification. Possible reasons for inhibition are discussed subsequently.

6.5.1 Effect of Hydraulic Retention Time on Ammonia Removal

Weng and Molof (1974) and Wilson and Murphy (1980) showed that disk substrate surface area loading is the key design variable for RBC's, rather than hydraulic retention time. However, Peddie (1986) found good correlation between hydraulic retention time and ammonia removal for an RBC treating landfill leachate.

Figure 15 shows ammonia removal compared to system influent flow. Since leachate ammonia concentration did not vary significantly during the study, Figure 15 is similar to Figure 13. The linear regression statistics equal: Slope = -3.0 %/(L/day), $r^2 = 0.52$, which is again indicative of a non-linear relationship.



Figure 16 shows ammonia removal compared to total system flow (influent plus recycle). Figure 16 shows a more linear relationship between ammonia removal and total flow than shown in either Figure 13 or 15. The linear regression parameters equal: Slope = -0.95 %/(L/day) $r^2 = 0.67$, which is a better correlation than either mass loading or influent flow, but is still a relatively poor correlation. No similar results are available in the literature, since RBC's are most often run as a complete treatment system without recycle.

6.5.2 Inhibition of Nitrification

Figure 17 shows RBC effluent nitrite and nitrate concentrations over time. Figure 17 shows a long term (greater than 100 days) nitrite build-up. The buildup seemed to occur with or without recycle and nitrate in the effluent was all but eliminated. The results indicate that the transition from ammonia to nitrite was occurring, but the transition from nitrite to nitrate was being inhibited. Alleman (1984) suggests certain factors (elevated pH, low temperature, low oxygen or carbon dioxide partial pressure, free ammonia and process overloading) may selectively inhibit Nitrobacter, resulting in a prolonged nitrite build-up.

Several factors including pH, metals or ammonia could have caused inhibition of the oxidation of nitrite to nitrate. Given that the influent ammonia concentration to the RBC generally exceeded 1,500 mg/L, inhibition by the ammonia or free ammonia is most likely.

Working with a continuous flow activated sludge reactor, Azevedo (1993) found a nitrite build-up occurred at influent ammonia-N concentrations of 600 mg/L. Gee et al. (1990) found that the rate of conversion of ammonia to nitrite is inhibited by ammonia, and can be modelled by Equation 3: The Haldane Inhibition Model.





FIGURE 17: RBC EFFLUENT NITRITE AND NITRATE

The conversion of nitrite to nitrate was not effectively modelled with the Haldane Inhibition Model, and the oxidation of nitrite to nitrate was found to be strongly inhibited by simultaneous high concentrations of ammonia and nitrite (approximately 1,000 mg/L total N). These results support the conclusion that high influent ammonia concentration caused the inhibition of the conversion of nitrite to nitrate and resulted in the observed nitrite build-up.

Since nitrite/nitrate were not measured until day 45 of the experiment, definitive conclusions cannot be made. But given, ammonia was almost completely removed from the leachate at ammonia-N loading rates up to 1.5 g/m²/day (97%), and initial analysis two weeks after increasing loading rates showed only a slight nitrite build-up (71 mg/L), a plausible conclusion is that nitrite build-up would not occur at low (<1.5 g/m²/day) ammonia loading. This conclusion is supported by Ehrig(1985), who recommends ammonia loading of less than 2 g/m²/day be maintained to avoid nitrite build-up.

As previously mentioned, in addition to an observed nitrite build-up, the removal of ammonia from the system appeared to be inhibited at ammonia loading rates exceeding 1.5 g/m²/day. Spengel and Dzombak (1991) achieved 95% ammonia removal at RBC loading of up to 7.3 g/m²/day. Ehrig (1986) achieved full ammonia removal at ammonia loading rates of 17 g/m²/day. Given the high ammonia and nitrite concentrations in the RBC, it is reasonable to assume that the inhibition of ammonia removal was also a result of the high ammonia and nitrite concentrations. Azevedo (1993), using an completely mixed activated sludge reactor, found complete inhibition of nitrification (effluent ammonia-N equal 700 mg/L) at an influent ammonia concentration of 2000 mg/L.

6.5.3 Nitrogen Removal

As previously discussed, developing an accurate mass balance for major parameters could not be achieved. Although steady-state was not fully achieved, an approximate mass balance for nitrogen for the RBC can be determined by calculating the total nitrogen entering and exiting the RBC. Figure 18 shows RBC nitrogen removal based on RBC influent ammonia and effluent total nitrogen (ammonia plus nitrite and nitrate).

Based on the results shown in Figure 18, the mean total nitrogen removal in the RBC was 54%. The mean system total nitrogen removal was 66%. Nitrogen removal appeared to be independent of loading rate.

The nitrogen removal in the RBC could be the result of assimilation, stripping or denitrification. Given that influent ammonia concentration substantially exceeded influent BOD, only a small portion of the nitrogen removal would have been through assimilation.

The RBC effluent pH ranged from 7.2 to 9.3 and averaged 8.5 (see Table 14). According to Equation 7, 0% to 50% of ammonia would be in dissolved gaseous form. Therefore, air stripping and denitrification would have combined to remove the nitrogen from the system.

Although off-gasses from the RBC were not analyzed, ammonia odours were never present over the RBC. This is not sufficient evidence to show that the ammonia removal was not a result of air stripping. The odour threshold for ammonia is 46.8 ppm (Weiss, 1980). Therefore, given that daily ammonia-N removal ranged up to approximately 10 g/day, sufficient ammonia would have been produced to make approximately 200 litres/day of air exceed the odour threshold. Since the room the experiment was conducted in was not ventilated, this quantity of ammonia may or may not have been detectable.



As shown in Table 14, alkalinity was removed from the system. This is evidence that nitrification occurred in the system. As shown subsequently in Section 6.8, the alkalinity:ammonia-N removal ratio was not sufficient to suggest complete nitrification, unless denitrification is also assumed to have occurred.

As shown in Equation 4, denitrification requires a carbon substrate. Based on the stoichiometric requirements of denitrification using methanol as a carbon substrate, the approximate COD requirements are 3.0 g/g-N for denitrification of nitrate and 1.7 g/g-N for denitrification of nitrite. The overall mean COD/N removal for the experiment was 1.8 g-COD/g-N. Therefore, assuming all of the COD consumed was used for nitrogen removal, and denitrification from nitrite occurred, sufficient COD was consumed to support denitrification.

Various authors have shown nitrogen removal in RBC's, including Masuda et al. (1991), Hosomi et al. (1991), Atwater and Bradshaw (1981), and Gupta et al. (1994). The maximum nitrogen removal observed was 87.6% at influent total nitrogen concentration of 1786 mg/L and total nitrogen loading of $9.36 \text{ g/m}^2/\text{day}$ (Gupta et al., 1994). These authors postulate that nitrogen removal is a result of concurrent nitrification/denitrification in the RBC. Masuda et al. (1991) attribute the results to the simultaneous presence of aerobic and anoxic conditions within the fixed film. Masuda et al. (1991) showed increasing nitrogen removal with decreasing oxygen partial pressure (ie. decreased dissolved oxygen), which supports the hypothesis of denitrification occurring under anoxic conditions within the biofilm. Gupta et al. (1994) attribute the nitrogen removal to simultaneous aerobic nitrification/denitrification as described by Robertson and Kuenen (1984) and shown in Equation 5.

Based on the results of this experiment, two possible explanations of the observed nitrogen removal are possible:

1. Removal is the result of nitrification/denitrification reactions, with

nitrification proceeding only to nitrite as shown in Equation 5.

2. Removal is the result of ammonia stripping occurring simultaneously with nitrification.

In similar work currently being conducted by the author, an RBC is being used to nitrify landfill leachate with influent ammonia-N of up to 300 mg/L, and total BOD of approximately 50 mg/L. The RBC effluent pH ranges from 7.4 to 8.9 with an average of 8.1, and no nitrogen removal is occurring. This observation suggests that little ammonia stripping would occur over the pH range observed in this research (7.2-9.3).

Although neither nitrification/denitrification or ammonia stripping can be proven to have caused the observed RBC nitrogen removal since RBC off-gasses were not analyzed, it seems most likely that the former caused the nitrogen removal. This is because:

1. Robertson and Kuenen (1984) and Kuenen (1988) have shown that aerobic nitrification/denitrification is possible, and other authors have shown nitrification/denitrification can occur in a primarily aerobic biofilm

2. Both alkalinity consumption and COD removal results support the explanation of simultaneous nitrification/denitrification

3. Ongoing research by the author suggests that in the pH range observed, ammonia stripping is unlikely

Although, as previously discussed, system mass balances cannot be used to quantify mass removals in each of the system components, the results of this research show that full denitrification occurred in the anaerobic filter during recycle. This conclusion can be made because nitrite and nitrate were not present in the filter effluent under all operating conditions (see Table 14), including 100% recycle and recycle nitrite-N concentration greater than 800 mg/L.

6.6 METALS

As part of the study, filtered samples were measured for iron, zinc, lead, copper, cadmium, manganese, chromium and nickel on a weekly basis. Data from the analysis are shown in Appendix 7. Overall mean removal ranged from -19% for nickel to 59% for manganese, and are shown in Table 17. Measured values are dissolved only (filtered through a 0.4 μ m filter). Totals were not measured.

TABLE 17:	OVERALL METAL	REMOVAL			
Metal	Leachate	Filter	RBC Effluent	Removal	
	(mg/L)	(mg/L)	(mg/L)	(१)	
Mn	0.77	0.43	0.32	59	
Fe	7.30	4.31	4.59	37	
Pb	0.27	0.16	0.22	17	
Zn	1.01	0.63	0.85	16	
Cd	0.01	0.01	0.01	0	
Cu	0.13	0.09	0.13	0	
Cr	3.05	3.17	3.41	-12	
Ni	0.22	0.25	0.27	-19	

Although system component removals cannot be accurately calculated, based on the concentration of the effluent from the anaerobic filter and RBC shown in Table 17, the majority of metal removal appears to have been in the anaerobic filter. The variation of overall metal removal is shown in Figures 19 to 25 from data in Appendix 7.















Figures 19 to 25 show generally non-linear relationships between flow and metal removal.

Chang (1989) states that the main mechanism for removal of metals in an anaerobic filter would be the formation and precipitation of metal sulphides. Barnes et al. (1991) anaerobically treated acid mine drainage with influent sulphate concentration exceeding 1000 mg/L to remove metals. They achieved 100% removal of zinc, cadmium, cobalt and copper. Chang (1989) removed 95% of soluble iron from a leachate with influent iron of 570 mg/L and influent sulphate of 1830 mg/L.

The leachate was analyzed for sulphate, but sulphate was only present in 2 of 11 samples, and did not exceed 290 mg/L. Sulphate was present in 6 of 11 RBC effluent samples. Hydrogen sulphide odour was not detected in the leachate or anaerobic filter effluent. Given that the odour threshold for detection of hydrogen sulphide is 0.0047 ppm (Weiss, 1980), even trace amounts of hydrogen sulphide should have been noticeable.

These results indicate that sufficient sulphate and sulphide may not have been present in the leachate to fully precipitate the metals present.

The lack of metal removal in the RBC cannot be adequately explained. Results from aerobic systems (aerated lagoon and RBC) in Table 5 and Table 8 show removal efficiency up to 99% for iron in an aerated lagoon system (Robinson and Grantham, 1988) and 88% for total manganese in an RBC system (Peddie and Atwater, 1985). In comparison, based on the information in Table 17 zero or negative removal occurred in the RBC for most metals.

Under aerobic conditions, iron and manganese are removed from wastewater through the oxidation of Fe^{+2} and Mn^{+2} to Fe^{+3} and Mn^{+4} . The oxidized species are much less soluble than the reduced species. For instance, according to Benefield et

al. (1982), at pH 8, Fe^{+2} concentration can be up to 10 mg/L. Under the same conditions, Fe^{+3} concentrations would be in the 10^{-4} mg/L range. Therefore, following oxidation in the RBC, iron and manganese should have been totally removed from the leachate.

pH also affects the metal solubility. An increase in pH occurred across the anaerobic filter for the duration of the experiment, and a further increase in pH occurred across the RBC for the first part of the experiment. Table 18 shows the effect of pH on the solubility of the metals analyzed in compounds with sulphide, hydroxide and oxide.

TABLE	18:	EFFECT	OF	pН	ON	METAL	COMPOUND	SOLUBILITY		
Metal		S ⁺²			OI	H+	O ⁺²			
Mn			A			A		A		
Fe(II)			Α			A		А		
Fe(III	:)		d			Α		A		
Pb			Α			w		w		
Zn			Α			A		W		
Cd			Α			A		A		
Cu			Α			A		A		
Cr			d			Α		a		
Ni			A			w		А		
A:	Inse	oluble i	in v	vate	er,	but so	oluble in	acid		
d:	Deco	omposes	in	wat	er					
w:	Spar	ringly s	solu	ıble	€ ir	n water	c, but sol	luble in acio	d	
a:	Inso	oluble i	in v	vate	er,	sparin	ngly solub	ole in acid		
SOURCE	: CI	RC Handk	000}	s,]	989	9-1990				

For each of the metal compounds shown in Table 18 except chromium sulphide, an increase in pH should result in a decrease in solubility. Therefore, the lack of metal removal cannot be associated with increasing pH.

The metals that increased across the treatment system, chromium and nickel, may simply have not been treated by the system, and the measured increase is the result of experimental error. Peddie (1985) found RBC removals of less than 20% for chromium and nickel compared to removals exceeding 80% for manganese and iron, indicating biological treatment is ineffective at removing these metals.

The high solubility of metals and consequential lack of removal by the system may indicate that the metals were complexing with organic molecules such as humic and fulvic acids. According to Benefield et al. (1982), the presence of these molecules will significantly increase metal solubility. To test this theory, extraction of the organics would be necessary.

Another possible explanation is that metals were released from leachate solids during treatment. This theory cannot be verified because only dissolved metals were measured. In future work, to ensure all metal forms are measured, total metals rather than dissolved should be measured.

Since metal discharge levels are not regulated in Taiwan, RBC effluent metal levels are compared to B.C. Pollution Control Objectives (1976) AA and BB values for discharge to receiving bodies in Table 19.

TABLE 19: RBC EFFLUENT METAL LEVELS COMPARED TO B.C. POLLUTION

C	ONTROL OBJECTIVES			
Metal	Effluent	B.C. Pollution Co AA	ontrol Objec BB	ctives
	(All value	s dissolved mg/L unle	ss stated)	
Cđ	0.01	0.005	0.01	
Cr	3.41	0.1	0.3	*
Cu	0.13	0.2	0.5	
Fe	4.59	0.3	1.0	
Pb	0.22	0.05	0.1	*
Mn	0.32	0.05	0.5	
Ni	0.27	0.3	0.5	
Zn	0.85	0.5	5.0	*

NOTES:

* B.C. Pollution Control Objectives Levels AA and BB for these parameters are totals rather than dissolved.

SOURCE: British Columbia Department of Lands, Forests and Water Resources, 1976

Table 19 shows that the system effluent exceeded both AA and BB levels for lead, iron and chromium, and additional treatment would be required prior to discharge.

North American experience has shown that many metals can be removed incidentally during biological treatment processes. Based on the results of this experiment, it appears that higher strength, more complex, leachates' metals are removed to a lesser extent through biological treatment, and therefore these leachates require tertiary treatment for metals removal. Based on the literature survey of potential physical/chemical treatment methods, precipitation with lime is probably the most appropriate method. The pH of the leachate was measured each week. While the author was in Taiwan, the pH of the filter effluent and RBC effluent were also measured weekly. Subsequently, the pH of the filter and RBC were measured on a less frequent basis.

The pH of the leachate, filter effluent, and RBC effluent are shown over time in Figure 26, and system pH data are provided in Appendix 4.

The leachate pH ranged from 7.8 to 8.8 with a mean of 8.1. This is typical of a methanogenic phase leachate. Filter effluent ranged from 8.0 to 8.6 with a mean of 8.4. RBC effluent ranged from 7.2 to 9.3 with a mean of 8.5. The observed effect of increasing pH across the RBC during the early part of the experiment was unexpected because nitrification results in the release of hydrogen ions and is consequently acidifying. Szwerinski et al. (1986) verified this theoretical result for a nitrifying biofilm.

The phenomenon of increasing pH across a nitrifying biofilm was observed by Spengel and Dzombak (1991). They found a change in pH from 7.4 in the first stage of an RBC to 8.1 in the last stage (influent pH 7.8). They attributed the increase in pH to high carbon dioxide concentration in the leachate prior to aeration. The carbon dioxide was present because of anaerobic decomposition within the landfill and degradation of organic substrates in the leachate. During aeration, the carbon dioxide was stripped from the leachate resulting in an increase in pH.

This phenomenon is explained in detail by Sawyer and McCarty (1978), and is based on the alkalinity equation shown in Equation 9.



EQUATION 9: ALKALINITY RELATIONSHIP

 $CO_2 + H_2O \leftrightarrow H_2CO_3 \leftrightarrow HCO_3^- + H^+$

The equilibrium relationship for carbonic acid, carbonate and hydrogen ion is shown in Equation 10.

EQUATION	10 :	CARBONIC	ACID	EQUILIBRIUM	RELATIONSHIP	
[H ⁺][HCO ₃ ⁻]/[ŀ	$H_2CO_3] = K$	1			

According to Sawyer and McCarty (1978), $[H_2CO_3]$ is the sum of the concentrations of free carbon dioxide and carbonic acid. Free carbon dioxide represents 99% of the total. During aerobic treatment, carbon dioxide present in the leachate due to organic decomposition processes is stripped off to equilibrium with the air. Since the total alkalinity remains constant (Sawyer and McCarty, 1978), the hydrogen ion concentration must drop, and the pH must therefore rise. According to Sawyer and McCarty (1978), at 25 °C, for aerated water with 100 mg/L total alkalinity as CaCO₃, the equilibrium pH will be 8.6. A higher alkalinity would result in a higher equilibrium pH.

This phenomenon may not be observed if pH is not measured at the site. If the pH is measured on a stored sample, a higher carbon dioxide concentration may be present, and a lower pH would be observed. This effect can be seen in work currently being performed by the author in which site measurements of RBC effluent pH are up to a unit higher than laboratory measurements.

6.8 TOTAL ALKALINITY

Total alkalinity (titration to pH 4.5) was measured in the raw leachate, filter effluent, and RBC effluent weekly from day 20 to day 62 of the experiment. The measured data are shown in Table 20.

DAY	LEACHATE ALKALINITY	FILTER EFFLUENT ALKALINITY	RBC EFFLUENT ALKALINITY	REMOVAL
	(mg/L)	(mg/L)	(mg/L)	(mg/L)
20	10,980	10,980	2,630	8,350
27	10,850	9,020	2,630	8,220
34	11,100	8,650	3,510	7,590
41	10,690	6,510	3,325	7,365
48	10,800	6,770	4,490	6,310
55	10,510	6,960	4,395	6,115
Mean	10,820		3,500	7,320

Table 20 shows that the alkalinity of the raw leachate was high. Ehrig (1989a) states that leachate typically has total alkalinity of 6,700 and ranges from 300 to 11,500. Chang (1989), investigating the treatment of leachate from another landfill in southern Taiwan, found a leachate alkalinity of 8,500. The leachate investigated by Chang (1989) was younger than the leachate used in this experiment (pH 7). More volatile fatty acids and consequently lower alkalinity would therefore be expected.

Since the waste characteristics of the landfill are unknown, the source of the high alkalinity is unknown.

Based on the data in Table 20, the mean change in alkalinity as $CaCO_3$ per unit ammonia-N removed is 4.1 g/g. This is below the theoretical requirement of 7.14 g/g for nitrification to nitrite or nitrate.

If one assumes that denitrification occurred in the RBC and anaerobic filter, the net alkalinity requirement would be the requirement for nitrification minus the alkalinity gained from denitrification, which equals 3.57 g/g.

Therefore, as previously discussed, the observed reduction in alkalinity to nitrogen removal ratio, compared to the theoretical requirement, could have been the result of concurrent nitrification/denitrification.

6.9 ANIONS

In addition to nitrate and nitrite, anions including chloride and sulphate were measured in the leachate, filter effluent, and RBC effluent using an ion chromatograph (IC).

Sample output from the IC is shown with each peak labelled in Appendix 8.

In addition to the other anions analyzed, the IC standards also included phosphate, but due to interference from the adjacent chloride peak, phosphate could not be measured in any of the samples. Enough phosphate was added directly to the RBC on a daily basis to ensure a BOD:N:P ratio of 100:5:0.5 was maintained, as suggested by Ehrig (1985). Leachate, filter effluent, and RBC effluent values of chloride, nitrite, nitrate and sulphate are shown in Table 21.

The results in Table 21 show that, although the mean leachate, filter effluent and RBC effluent chloride concentrations were not statistically equal at the 95% confidence level, they were close enough to indicate no removal occurred in the system. This is as expected, as chloride is not removed through aerobic or anaerobic biological treatment.

TZ STORT	OTNE	N DALA										
	Leach	ate			Filte	r Efflu	lent		RBC E	ffluen		
Date	Cl ⁻¹ (mg/L	SO4 ⁻²)	NO ³⁻	NO_2^-	Cl ⁻¹ (mg/L	SO4 ⁻²	NO ³⁻	NO2 ⁻	Cl ⁻¹ (mg/L	SO4 ⁻²)	NO ₃ -	NO_2^-
01/13/92 01/17/92	2400	0	0	0	2540	10	0	20	2680 2690	60 60	00	120
01/27/92					2390	0	0	0	2390	<u> </u>		160
02/13/92	2860	0	0	0	2820	0	0	0	2900	140	00	570
02/21/92	2490	0	0	0	2590	0	0	0	2590	0	0	470
02/27/92	2820	0	0	0	2720	0	0	0	3050	0	10	780
03/02/92	1980	100	0	0	2470	0	0	0	2890	0	0	890
03/16/92	2520	0	10	0	1940	0	0	0	2100	0	10 1	680
03/23/92	2750	0	0	0	2690	0	0	0	2890	70	270	680
03/30/92					2830	260	0	60	3030	220	80	860
04/06/92		a			2920	0	0	0	2920	190	80	870
04/13/92	1960	0	0	0					•) 1)	
04/20/92	2080	290	0	0								
04/27/92	2440	0	0	0	2220	0	0	0	2720	150	300	230
05/04/92	2680	0	0	0	2030	100	0	0	2750	80	80	620
Mean	2453	35	6.0	0	2513	30	0	7	2740	7.5	60	530
STD	310				304				255	1)	
95% +-	183				172				139			

TABLE 21: ANION DATA
The results also show that sulphate was only present in the leachate and filter effluent 20% of the time compared to 60% of the time in the RBC effluent. Nitrite and nitrate were not present in the leachate and only present 13% of the time in the filter effluent, compared to 100% in the RBC effluent. This result indicates that the leachate and filter effluent were generally fully reduced, containing no oxidized species. Oxygen/reduction potential measurements would be required to confirm this hypothesis.

6.10 RBC SPEED

RBC speed could not be controlled precisely because of the type of drive motor used, but was adjusted on an approximately daily basis to maintain a speed between approximately 10 and 25 RPM (6-16 m/min peripheral speed). RBC speed data are provided in Appendix 4. Since nitrification rate has been shown to depend on RBC peripheral speed (because it controls dissolved oxygen concentration and consequently oxygen transfer into the system (Hosomi et al., 1991)), future work should ensure RBC speed is kept constant.

6.11 SUSPENDED SOLIDS

Effluent suspended solids were not measured on a regular basis because RBC is a fixed film process, and suspended solids are not a key design parameter. From the system data provided in Appendix 4, the mean raw leachate suspended solids was 220 mg/L, filter effluent suspended solids was 170 mg/L, and the mean effluent suspended solids was 170 mg/L. Sludge recycle, rather than effluent recycle, may have reduced effluent suspended solids since sludge would not have had time to float, due to denitrification in the sludge. Sludge recycle was not used to avoid clogging the anaerobic filter.

7. SUMMARY OF RESULTS

The leachate used in this experiment had BOD:COD ratio and pH typical of a methanogenic stage leachate. The leachate ammonia-N (mean of 2140 mg/L) and dissolved chromium (mean of 3.05 mg/L) were both particularly high compared to expected values.

1. Parameter removal in each of the RBC and anaerobic filter could not be accurately calculated because removal values varied widely throughout the experiment. This result can be attributed to insufficient stabilization time between changes in system flow. Overall system removal rates for each flow period could be determined.

2. The period mean BOD removal rates ranged from 90-96%, with an overall mean of 92%. The period mean COD removal rates ranged from 43-59%, with an overall mean of 49%. Period mean COD:BOD removal ranged from 2:1 to 5.1:1, with an overall mean of 3.7:1. BOD and COD removal were unaffected by flow.

3. Ammonia removal throughout the experiment exceeded 80%, but at loading rates greater than 1.5 g/m²/day, mean effluent ammonia-N exceeded 100 mg/L, which was considered unacceptable. At RBC loading rates less than 1.5 g/m²/day, 97% ammonia removal was achieved. Other authors have found nitrification at significantly higher loading rates, but lower influent ammonia concentration.

4. Ammonia mass removal increased linearly at all RBC loading rates.

5. Ammonia removal, on a percent removal basis, decreased with increasing mass loading, influent hydraulic loading and total hydraulic loading. The linear regression parameter r^2 was highest for total hydraulic loading at 0.67, indicating the best linear correlation to ammonia removal.

6. A long term (greater than 100 days) build-up of nitrite occurred in the system. RBC effluent nitrite concentrations averaged 530 mg/L.

7. The mean total nitrogen removal in the RBC was approximately 54%, and the total system nitrogen removal was approximately 66%. Nitrogen removal appeared to be independent of flow.

8. Overall mean removals for iron, zinc, lead, copper, cadmium, chromium and manganese and nickel equalled 37%, 16%, 17%, 0%, 0%, -12%, -19% and 59%. The majority of metal removal appeared to be in the anaerobic filter.

9. System effluent metals levels exceeded B.C. Pollution Control Objectives for lead, iron and chromium.

8. CONCLUSIONS

1. The system effectively removed BOD, but COD was not fully removed.

2. RBC ammonia loading, rather than organic loading, was the critical design parameter for the system. Total hydraulic loading appeared to provide the best correlation with ammonia removal.

3. At RBC loading rates exceeding 1.5 $g/m^2/day$, inhibition of nitrification was observed. At higher loading rates, although overall mass removal continued to increase, full nitrification could not be achieved. High influent ammonia concentration was most likely responsible for the observed inhibition of nitrification.

4. Inhibition of <u>Nitrobacter</u>, resulting in a prolonged nitrite build-up, occurred in the system. The high influent ammonia concentration was most likely responsible for the inhibition of Nitrobacter.

5. Nitrogen removal in the RBC can be attributed to either simultaneous nitrification/denitrification or air stripping of ammonia. It is most likely that simultaneous nitrification/denitrification caused the observed nitrogen removal.

6. Low metal removals in the system was most likely the result of complexing of the metals with organics. The North American experience of incidental metal removal during biological treatment may not be the case for stronger, more complex, leachates. Additional physical/chemical treatment processes, potentially lime treatment, would be required to remove system metals to acceptable discharge levels.

9. RECOMMENDATIONS

Future work investigating RBC treatment of the leachate used in this experiment should take the following recommendations into account:

1. Off-gasses from the RBC should be analyzed to determine the fate of nitrogen in the system.

2. Total metals should be analyzed.

3. To determine if metals are complexing with organics, extraction of the organics and subsequent metal analysis would be required.

4. Nitrification inhibitors should be added to BOD analysis samples.

5. Long term BOD tests should be conducted to determine if the seed is sufficiently acclimated, to determine the ultimate BOD of the leachate, and to determine if nitrification affected the BOD results.

6. Nitrite interference inhibitors should be added to COD analysis samples.

7. COD samples should be analyzed by a method suitable for high chloride concentration.

8. Complete steady state conditions should be achieved at each loading rate before adjusting system flow.

9. RBC speed should be kept constant.

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APPENDIX 2: AMMONIA PROBE SAMPLE CALCULATIONS AND OUTPUT

12/24/91 Date: Sample: RBC Effluent Standard LOG Probe Concent. Conc. Reading (mg/L) (m) 10 -33 1 50 1.69897 -76 100 2 -95 200 2.30103 -113 500 2.69897 -136 1000 3 -155 Unknown -77 Linear Regression Regression Output: 27.3 Constant Std Err of Y Est 0.6 1.00 R Squared No. of Observations 6.0 Degrees of Freedom 4.0 X Coefficient(s) Std Err of Coef. -60.8 0.4 LOG Unknown Concentration 1.72 Unknown Concentration 52 mg/L



APPENDIX 3: KAOHSIUNG DAILY PRECIPITATION DATA

STN.:467440 KIND:A69 YEAR:1992 REMARKS: 高雄 KAOHSIUNG 降水量 PRECIPITATION

UNIT:0.1 mm

D/M	1	2	3	4	5	6	7	8	9	10	11	12	M/D
1		24° 100	95		More		100	1	And Call	-	-044	BALLAN	1
2	-	-		62	ac-801	5	1		25	8 14034			2
3		-	88	426	73		224			Calasian			3
4				38	-	******	897	68	2005			-	4
5	165		420	9			2214		1433				5
6	6		190	285			11		316				6
7	***	50				11	137		25			Т	7
8						608	230						8
9				56	605	74		-	7				9
10				-	6	137		-	31				10
11		43		406		Т			134	*****			11
12		171		120			·						12
13		83		13		41	ADDA		-				13
14		T		22		1		5	-				14
15		53		7								-	15
16	55	15						49					16
17		28				-	5	138	Т				17
18	35		·		436		- 75	322	Ť	5		-	18
19	52	156	·	521	35			128	*****	ź			19
20	15						30	205		-			20
21			-		37	1814		105	58	-			21
22		-		510	147			10	166				22
23				147					625				23
24	- T-					-	305		470				24
25			-										25
26					50		-		270				26
27			terror		14			140			Place		27
28						-	т	77					28
29			470			51				-			29
รก			15		6	26	308	976					30
31	2				58			92					31
T/M	330	599	1278	2622	1467	954	4544	2316	5615	12	0	0	T/M
D/M	1	2	3	4	5	6	7	8	Э	10	11	12	D/M
REFER	RENCE:	TOTAL	= 19	737, M	EAN=	1645,	MAX=	5615	i/ 2214*.	, MIN=		0/	0*
T : TF	RACE,	-:N0	RAIN/2	ERO,	X: TRO	UBLE,	∕:UNK	NOWN,	*:REF	ERENCE	VALU	E	

Appendix 4: Leachate, Filter Effluent and RBC Effluent Data

Leachate Data

(I/6	0.21	0.25	0.19	0.20 0.21 0.27	0.22		0.22
ΞÔĔ	3.35 3.25	0.20 2.59 2.79	2.97	3.64 4.10 2.09 2.09	3.49	2.70 3.24 1.91 2.75	3.05
C (D) (mg/)	42	20 25	84	57 55 30 30	43 42	51 79 47	77
Mn (D) (mg/l)		9.44	0	0004	ÖÖ	00-0	ю
(ng/l) (mg/l)	0.0	0.00	0.11	0.00			0.13
) (l/bu	6.75 6.79 6.40	9.49 6.44	6.33	6.12 6.31 6.34 14.65	6.00 5.34	5.89 5.99 11.30 6.70	7.30
E E E E	0.01	0.0	0.01				0.01
ЪŪ	0.33 0.29 0.29	0.33	0.31	0.24 0.23 0.31 0.02	0.16 0.28	0.39 0.26 0.28 0.20	0.26
Pb (D) (mg/l)	24 16 26	89 89 89	96	99 03 49	62 90	18 29 96	10
Zinc (D) (mg/l)	+- +- +	- 00	Ö	00-0	ÖÖ		-
KN (mg/l)	0760	3253 3640 2755 2755					2918.60
alinity g/i Ca)	10980 10860	11100 10690 10690 10510					0821.67
() Alk	5002 3204 4155 3454 3707	3820 4337 5620	5191	4519 4392 5154 7015	4706 5247	8417 7999 5176 4621	139.80
(mg/	165	330 900	320	980 520 540	920	270	00 50
BOD (mg/	196 224 76 343	330 6 330 6 153 9	145	281 274 137 260	147 155	218 376 228 11 205 11	.74 705
(INg/I)	00000			∞ <i>−</i> 0 4	40	0 0 0	0 219
Ammonia (mg/l)	1826 2111 126(2318 2318 2318 2318 2318	253	247 257 190 198	198	270 160 231	2142.0
H units)	7.77 8.8 8.8 8.5 8.5	7.87 7.87 7.98 7.98 8.02	80	8 7 7 8 8 7 9 8	8.2 8.2	8 8 8 9 7 9 8 8 8 8 9 9 9 9 9 9 9 9 9 9	8.14
ean p ow ^^3/hr) (0 7 7 0 7 0 0 0 7 0 0 0	7.6 7.9 7.0 7.0	8.0	6.3 6.7	6.4	29.2	8.39
ч м Ш Ц Ц Ц Ц Ц Ц Ц	231630 232720 233439 234417 235787 235787	237190 238471 239806 240982 240982	243188	247415 248536	251773	279816	
- E G L O C L	ນ. ນີ້ນີ້ ອີນ ນີ້ນີ້ ອີນ	0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0 0	5.1	5.5 6.5	6.6	7.4	6.22
Inst. Flow (m^3/I	M M M M M M M M M M M M M M M M M M M	A A A A A A A A A A A A A A A A A A A	AM		AM AM AM	AM	·
Time	1 10:30	2 10:30 2 10:30 2 10:30 2 10:30 2 10:30	2 10:30		2 10:30 10:30	2 2 2 10:30	
Collect. Date	18/11/9 25/11/9 02/12/9 09/12/9 16/12/9	23/12/9 30/12/9 06/01/9 13/01/9	27/01/9	02/02/9 17/02/9 02/03/9	16/03/9 23/03/9	20/05/9 06/04/9 20/04/9 27/04/9 02/05/9	Mean

Appendix 4: Leachate, Filter Effluent and RBC Effluent Data

RBC Influent/Anaerobic Filter Effluent Data

ć	(I/Bu					0.205					0.21										30.0	C7.0						0.29										0.21	
Z	-) (l/b					2.47					2.42										30 0	C.A.7						2.82										2.81	
Ϋ́) E)					0.28					0.24											0.4Z						0.59										0.67	
UM V	<u>)</u> (015					0.01										5	10.0						0.01										0.01	
ЪĘ) Ű					25 0.					75 (60										90	
Sé	(mg/l)					5 0.0					0.0											4						6										2	
đ	(I/6m)					0.05					0.0										ć	5						0.1										0	
Zu	(l/gm)					0.395					0.475										22.0	0.0						0.72										0.6	
еĘ	(mg/l)					2.2					2.91											0.10						4.77										3.46	
Sulfate	(I/Gm)																-														12.6	i							
loride S	(i/ɓu																														2540	2							2390
o z) (l/ɓu												1924																										
at TK	// Ca) (n						0980		9020		8650								1000	0090			6100	0120							6770	-		0969					
Tot	ů Ű	490 662	600		775		820 1		670				963											500	021										330				
COD	l/ɓɯ)	~ ~ ~	N	0	0		0		2				0		ļ	2					ç	2		C	N		0								ч				
BOD	(I/gm)			26					47							24					, r	77			-		33												
FS	(I/gm)																								000														
TS	(i/ɓɯ)																								12400														
SS	(I/gm)										134												33	3							94	,	290						
Ammonia S	(mg/l)	2015	C061	1720			1475				1715		1585			1290					0007	0701	GL11					1030		1120	-		1040			2070			
I	units)						8.05		8.04		8.01					8.52						00 0	8.62						8 48	5				8.5					
ecycl p)) (%	00	00	0	0	0	0	0	0	0	0	0	12.3												13.0						144	ŕ	C		c	• O			
PO4 R	(I/day) (°		0.0 0.0	0.3	0.4	0.5	0.4	0.2	0.6	0.8	0.6	1.1	1.5	1.5	0.7	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	00											
-ilter																																		19	2 C	19.8	20	22	23
ach. F	(day)	Ċ	3 G 3 K	23	4.2	5.0	2.3	1.7	2.9	0.3	2.5	4.2	5.8	5.7	6.1	7.0	5.6	6.0	<u>6</u> .0	6 [.] O	0.0	0.0	- C - C		4 6		2.2	64		5.8	7 20		2 C 2 G	6.9	67	5.8	6.3	8.0	6.0
يار Jay		13	91	20	21	22	23	24	25	27	28	29	30	31	32	34	90 90	37	38	39	4	5	44	3:	44 4	25	48	64	205	56	5	1 U 1 U	99	22	. 6 <u>5</u>	62	63	64	65
Collect. [Date	09/12/91	13/12/91	16/12/91	17/12/91	18/12/91	19/12/91	20/12/91	21/12/91	23/12/91	24/12/91	25/12/91	26/12/91	27/12/91	28/12/91	30/12/91	01/01/92	02/01/92	03/01/92	04/01/92	05/01/92	06/U1/92	07/01/92	76/1.0/90	26/10/60	26/10/01	13/01/92	14/01/92	15/01/92	16/01/92	17/01/92		20/01/92	22/01/92	24/01/92	27/01/92	28/01/92	29/01/92	30/01/92

Appendix 4: Leachate, Filter Effluent and RBC Effluent Data

ē	(l/gu						0.35					0.23					0.29																	0.17						•		
ΞŸ	-)-) (2)					0	3.19					3.78					4.09						3.94																	3.25		
ς	D E					ļ	ŝ					35					66						ğ											32						53		
٩	(ing/)					č	0					ö					ö						ö											õ						ö		•
ßé	(ng/l)																																									
Ъć	(I/Gm)					000	0.09					0.13					0.14																									
و ز	(mg/l)						0.12					0.27					0.28						0.08											0.1						0.15		
- 6	(I) (I)					u c	0.5					0.78					0.84						0.89											0.43						0.65		
7	1) (//6						3.1					3.82					4.75						5.16											4.52						3.72		
tte Fe	<u>)</u> E																																								ţ	159
sulfa	(mg						¢	9					93					9						33											4						8	2
Cloride	(I/ɓɯ)						00	281					255					271						247											192						265	704
TKN	(µgm) (
Total Mod	mg/l Ca																																									
Q	(I/bu			0007	4000					4157						7692										5852						4941						4639				
8	ע) (ע			000	320					600						600					550																					
BOL	0m) (I																																									
FS	/ɓɯ)																																									
TS	(I/ɓɯ)																					_																	10			
SS	(µgm)																					26(15			
mmonia	(I/ɓu			0001	7761					2479						1405					1768					1837						1301						2142		i	1702	/
4	lits) (i																									8.2						8.2						8.4				
ycl pH	(r																																									
Rec	(%) (/																																									
PO4	r Iow (I/da)	23 20	LC.	20	5.2		50	20	9.8	9.6	<u>.</u> 5	3.2	3.9	3.1	19	9,1		21	5.	24	5.5	25		26	51	22	23	24	1.5	22	5.5	י. כי	<u>,</u> 5	1.5		5.5		25		28	28	<u>ה</u>
Filter	(C)		3				_	~	۳ ۳	÷	3	2	4 18	1	2	¢₽ 0	•	0	2	m	6	0		<i>(</i> 0	~	~	6	0	4	2	36	5	3 21	5	-	3	0			. 2	-	5
Leach.	(I/day)	ũũ					4	ŝ	<u>.</u>	7.	2.2	9	-'-	ŵ	Ö	ы. С		<u>.</u>	80	ŝ	ŝ	e e		ö	~	4	е П	e	ġ,		0.0		Ö	5.	O	Ö	.9			9 0	~	
Day		99 69	2	21	99	2 8	18	6/	80	83	84	85	86	87	88	6	92	33	94	32	67	86	66	100	5	104	105	106	107	108	109	11	112	113	114	115	116	118	119	120	121	126
Collect.	Date	31/01/92 03/02/92	05/02/92	07/02/92	10/02/92	76/ZN/11	12/02/92	13/02/92	14/02/92	17/02/92	18/02/92	19/02/92	20/02/92	21/02/92	22/02/92	24/02/92	26/02/92	27/02/92	28/02/92	29/02/92	02/03/92	03/03/92	04/03/92	05/03/92	06/03/92	09/03/92	10/03/92	11/03/92	12/03/92	13/03/92	14/03/92	16/03/92	17/03/92	18/03/92	19/03/92	20/03/92	21/03/92	23/03/92	24/03/92	25/03/92	26/03/92	31/03/92

RBC Effluent Data
Filter Effluent and I
Appendix 4: Leachate,

240 240 2830 255 7.92 129 0.02 0.22 130 3781 2918 7.92 129 0.22 0.22 738 4314 7.92 1.29 0.22 0.22 738 4314 7.92 1.29 0.22 0.34 738 4314 7.15 7.92 1.29 0.32 738 4314 7.245 7.15 0.34 0.34 738 2316 2.215 0.5 0.34 0.34 3009 2168 2.168 3.30 3.05 1.58 3000 2020 158 1.58 1.58	Day Leach. Filter PO4 Recycl pH Ammonia SS TS Flow Temp Flow ////////////////////////////////////	Leach. Filter PO4 Recycl pH Ammonia SS TS Flow Temp Flow (1997) (2011) (2011)	Filter PO4 Recycl pH Ammonia SS TS Temp Flow (2011) (2011) (2011) (2011) (2011) (2011) (2011) (2011)	PO4 Recycl pH Ammonia SS TS Flow (1142-1) (2010) (2010)	Recycl pH Ammonia SS TS	pH Ammonia SS TS	Ammonia SS TS	SS TS	TS (moll)	FS (moll)	BOD	COD (mod)	Total TKN Akal. (mod/Ca) (mod)	Cloride (mo/l)	Sulfate	Fe (D)	Zn (D)	4 (D)	n (D)	Cq (D)	Mn (D)	ъ Ĵ
240 2830 255 130 3781 7.32 129 0.22 738 4314 7.32 129 0.22 0.22 736 4314 7.32 129 0.22 0.23 736 4314 7.45 7.45 0.23 0.34 736 7.245 4.54 0.5 0.21 0.34 7155 7.155 2.216 0.5 0.21 0.34 3999 2168 2.216 3.5 1.53 1.54 0.34 5909 2.020 1.56 1.58 1.56 1.54 1.54 3000 2.020 1.58 1.56 1.56 1.56	(I/day) (C) (I/day) (%) (units) (mg/l) (mg/l) (mg/l)	(l/day) (C) (l/day) (%) (units) (mg/l) (mg/l) (mg/l)	(C) (I/day) (%) (units) (mg/i) (mg/i) (mg/i)	(I/day) (%) (units) (mg/l) (mg/l)	(1/6m) (mg/i) (mg/i) (mg/i) (%)	(Ingm) (Ingm) (Ingm) (Ingm)	(Ivgm) (Ivgm) (Ivgm)	(I/GW) (I/GW)	(Ingm)	(ı/ɓɯ)	(Ingm)	(mgm)	(mg/i ca) (mg/i)	(ingm)	(iv@m)	(1 <i>1</i> 6ш)	(Iv6m)	(Ingm)	(ingm)	(IvBm)	(Mgm)	(vbu)
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$\begin{array}{cccccccccccccccccccccccccccccccccccc$	132 6.8 24.5 8.6 2600	9.6 24.5 8.6 2600	: 24.5 8.6 2600	8.6 2600	8.6 2600	8.6 2600	2600				130	3781										
738 4314 7245 7245 7245 1245 7155 4.54 0.5 3999 2168 5909 2168 3000 2020 158	134 6.6 25	1 6.6 25	25													7.92	1.29	0.22			0.22	2.79
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5909 3000 2020 158	158 3.6 28.5	3 3.6 28.5	1 28.5																			
3000 2020 158	160 3.6 28.5 8.6 1517	3.6 28.5 8.6 1517	3 28.5 8.6 1517	8.6 1517	8.6 1517	8.6 1517	1517					5906	•									
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Appenix 4: Leachate, Filter Effluent and RBC Effluent Data

RBC Effluent Measured from the Final Clarifier

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ర్	(D) (ng/l)			2.52	2.895			3.2			2.9							2 1 0	0.0						4.3
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loride h	(l/bu									7676	C 107			0697					2390						
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0) T	units) (9.32 9.32	9.4	9.01		8.89				8.93			90.6										
a 	رت ن	ά	<u>5 6 6</u>	5 t t	50	20 15		18.5	16.5 15.5	17	: ;	: £	13.5	<u>t</u> 4	16	17 16 5	16	17	18	50	ן א ה ל	2.2	18	8 u	10.0
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Collect. Date		09/12/ 12/12/ 13/12/	17/12/	19/12/	23/12/	26/12/5 30/12/5	01/01/	06/01/ 07/01/5	08/01/ 09/01/5	10/01/5	13/01/5	15/01/5	16/01/5	21/01/5	22/01/5	24/01/5 27/01/5	28/01/5	29/01/5	30/01/5	31/01/5	05/02/5	07/02/5	10/02/5	11/02/5	シアハアト

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۳ M)Ë e																																									
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Б.	(l/gm)				0.91						1.27						0.82												0.67					0.78								1.32
ΦĈ	(l/bu				4.36						5.18						5.04												5.05					4.46								5.94
Ifate F) (I/6L	136																																	83			223				
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Total Alkalinit	(mg/l C																																									
COD	(l/gm)		3137						4231						4105						3815						3451					3536			3660					3741		
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Speed	(RPM)	17.	22	15	7	14.	14.	10	<u>~</u> ;		[;	- '	N .	- 1		<u>6</u>		2	0	7	0	19.	17.	13.	-	-	-	7	2		2	~~	•	~ ~		- •	-	•	-	~	Ŧ	-
Hd	(units)																				80						80					Ø								7.2		
RBC Temp.	<u>(</u>)	16.5 16.5	2.01 7	17	16.5	15	15	15	13.5	16	0.01		1/.5	0,12 0,12	23.5	23		24	18.5	18	20	21	22	22	23	23	24.5	25	22		23	24		26	0.02	25	0.07	ſ	q	20	300	0.UA
Collect. F Date	I	13/02/92	17/02/92	18/02/92	19/02/92	20/02/92	21/02/92	22/02/92	24/02/92	25/02/92	76/70/97	76/70/77	28/0.2/82	26/20/62		03/03/92	04/03/92	05/03/92	06/03/92	07/03/92	09/03/92	10/03/92	11/03/92	12/03/92	13/03/92	14/03/92	16/03/92	17/03/92	18/03/92	19/03/92	20/03/92	23/03/92	24/03/92	25/03/92	26/03/92	31/03/92	78/10/10	02/04/92	03/04/92	06/04/92	76/10/10	70/00/00

Appenix 4: Leachate, Filter Effluent and RBC Effluent Data

Appenix 4: Leachate, Filter Effluent and RBC Effluent Data

ź	() (n)												-	_						
ΰ	(n) (ng/l)												20 %	5						
ЧW Ч	(n) (//gm)												0.16	5						
Ъć	(n) (mg/l)																			
ъć	(n))																			
ą Ę	(n) (1/6m)												0 11							
u Ž	(l/gm)												0.86							
e Ç	(l/gm)												3.49							
Sulfate	(l/gm)	194												152					11	
Nitrate	(J/gm)	83												298	9				25	
Nitrite	(µgm)	866												232	517				615	
Cloride	(l/gm)	2918												2715	2168				2545	
Total C Atkalinity	(mg/l Ca (
COD	(I/gm)			4706			3736				3008				3000		3030			
BOD	(I/gm)			88							57									
FS	(l/gm)		•																	
TS	(I/gm)		682					317				636			436			235		
SS	(l/gm)																			
Ammoni	(I/gm)										62				35		46			
Speed	(RPM)		19	8	19.5	18		12	19	14.5		21.5		15.5		19.5	19			20.5
Hď	(units)										8.2				8.13		8			
RBC Temn	0		21	20	21	21		24	24	26		24		27		26	26			26.5
Collect.		09/04/92	10/04/92	13/04/92	16/04/92	17/04/92	20/04/92	21/04/92	24/04/92	25/04/92	27/04/92	28/04/92	29/04/92	30/04/92	01/05/92	02/05/92	04/05/92	05/05/92	07/05/92	08/05/92

Appendix 5: BOD and COD Data

Biochemical Oxygen Demand

				Leachate		Filter Efflu	ent		RBC Eff	luent				
Date	Day	Flow	Recycle	BOD	Mean BOD	BOD	Mean BOD	Treat. Eff.	BOD	Mean BOD	Treat. Eff.	Overall Treat. Fff	Overall Mean Treat	
		(L/day)	(L/day)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	1 (%)	(mg/L)	(mg/L)	2 (%)		Eff. (%)	
18/11/91 26/11/91 02/12/91	ΨΟΦ	2.7	0		465		368				26			95
09/12/91 16/12/91 23/12/91	₽22	3.0	000	465		260 475			(1	Q	τŋ	ý ý	у5 У5	
30/12/91 06/01/92 13/01/92 20/01/92	8 4 4 S	4 – 2 :0 0.0	2 13.4 13.4 13.4 13.4	630 490 900	673	245 490 330	355	-12 -210 -1	001	4 8 6	47 10 30 5	Q Q Q Q	888	63
27/01/92	62 60	69 6	00	320	616		518			3	34	0,	8	8
10/02/92 17/02/92 24/02/92 02/03/92 09/03/92	200 200 200 200 200 200 200 200 200 200			980 620 6240		320 600 550		-2 a a 67	<u>7</u> (1(1) w w		- 00 00 to	α 4 4 Φ α ο, ο, α	98 72 75	
23/03/92 30/03/92 06/04/92 13/04/92 20/04/92	115 132 132 132 132 146		000000	920	920	240 130 740	370	108			55 -1	~	2 2 0 0	91
27/04/92 04/05/92 07/05/92 Mean	153 160 163	3.5 3.7 3.6	000 10 N (0	1270	1270	415	415	108	υψ	3 4		0.00	8 Q	90
Notes: 1. Fitter Tra 2. RBC Tra 2. RBC Tra	atment atment Ci=Le Ci=Le Ce=E Cf=Fil	Efficiency Efficiency uent flow (I achate con ecycle flow ffluent conc Iter effluent	= ((Ce*R + (= (Cf - Ce) _/day) icentration (m (∐day) centration (m)	Ci*I) - (R + I)*(R + I)/(Ci g/L) ⊅/L) n (mg/L))*Cf)/(Ci*l)*-	00								

Appendix 5: BOD and COD Data

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Chemical Oxygen Demand

	_		48		43		47					59		49	
	Overal Mean Treat.	Eff. (%)		8 4 19	8	4		~	ر م		0	0	(0 m	~ ~	, A
	Overall Treat. Eff.	(%)		<u>ю</u> ,4 Ю	ň	ů		7	¥ ¢	οùο	4	ŭ	ងស	Ω4 0.14	46
	ਦ	-		10 11 10	51	217	83	62	88 8	64-	45	76	22 13	247 143	
	Tre2 Eff.	e (%)	, 143		327		750					958		502	
	Mean Adjust COD	(mg/L)	3		й 2		1					З		56	
	Mean COD	(mg/L)	214		270		340					3875		3067	
	lo2 coD coD	2 mg/L)		2340 1960 2130	2357	2577 2947		1020	2485 3580	3455	3165 2800	2618	2823 2823 3788 2818	2444 2449	2658
	2 < 0	- -	0		71		591					834		514	
	Mean NO2	(mg/l				0.01	0	(0)	~ ~	01				10.10	
ant	NO2	(mg/L)				22 12	162	200	778	892	677	777 856	866	375 615	552
RBC Efflue	0	(T/gm		2340 1960 2130	2435	2655 3025		1670	4230	4105	3815 3450	3535 3660	3740 4705 3735	3010 3015	3176
-	ند	Ŭ		23 25 47	-12		17	Ņ	u 4		'n	-26	4 4 0	-195 -96	
	Trea Eff.	- (%)	e E		8		52					5		ß	
lent	Mean COD	(mg/L)	271		292		526					496		485	
Filter Efflu	cop	(mg/L)		2675 2795 2670	2965 2860	2	4330	4605	7690		2020 4940	4640	3780 4315 7245	7155 4955 3000	4481
	lean COD	mg/L)	4097		4592		5163					7220		4898	
eachate	a 00	mg/L) (I	5000 3205 4155	3454 3710 5060	3820 4337	5620	5190	4520	4330 5155	7015	4705	5245	8415 8000	5175 4621	5040
-	ean ecycle	/day) (00	0000	13.4 13.4	13.4	00	00	00	00	00	ပပ	မမမ	999	>
	ŽŽ	ך (ך	2.7	3.0 3.0 3.0	6.2 5.9	6.0 4.9	6.3 6.5	ר, ה ה	0.0	6.7 8.2	5.5	6.2 6.9	6.5 9.8	3.5 3.7 3.6	2
	Mean Flow	(L/da)	ထုဝဖ	2030	<u>7</u> =	: 8 <u>3</u> 12	ស្ត ស្ត	0 c	20	6 5	t	ωŅ	ក្ត្លូល	n c u	,
	Jay			+- (N (N	~ 4	נע א	99	iνα	<i>ν</i> υ	υç	11	1 1	004	10 16 16	
	Date [18/11/91 26/11/91 02/12/91	09/12/91 16/12/91 23/12/91	30/12/91 06/01/92	13/01/92 20/01/92	27/01/92 03/02/92	10/02/92 17/02/92	24/02/92	02/03/92	16/03/92	23/03/92 30/03/92	06/04/92 13/04/92 20/04/92	27/04/92 04/05/92 07/05/92	Mean

Notes:

Hiter Treatment Efficiency = ((Ce*R + Ci*l) - (R + I)*Cf)/(Ci*l)*100
 According to Standard Methods (1985), nitrite produces a COD of 1.1 g/g NO2.
 According to Standard Methods (1985), nitrite produces a COD of 1.1 g/g NO2.
 RBC Treatment Efficiency = (Cf - Ce)*(R + I)/(Ci*l)*100
 RBC Treatment Efficiency = (Cf - Ce)*(R + I)/(Ci*l)*100
 RBC Treatment flow (L/day)
 Ci = Leachate concentration (mg/L)
 R = Recycle flow (L/day)
 Co = Effluent concentration (mg/L)
 Cf = Filter effluent concentration (mg/L)

Appendix 6: Nitrogen Data

Nitrogen Values

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			_	eachate	-	ilter Efflue	ent		œ	BC Efflue	ŧ													
Date	Jay F	a N N	ecycl.	Ammon. A	Aean / Ammon.	Ammon. N A	tean Ar mmon. Tr Ef	mmo N eat. f.	itr. A eat. f.	mmon. M Ar	ean R mmon A	BC Me mmon. RE bad. Lo	ean NO 3C ad.	о N N N	5 Zi		san An tal Tre r. Eff	at. Trea	tt Amn Trea	Over. no Ammo t. Treat.	over. Dn. Nitr. Treat.	Over. Mean Ammo	Over. Mean Nitr	
							**		0		Ę	3 /m^2 (ci	CV				4	5	Ë		ЕЩ. 9	Freat. Eff.	Ireat. Eff.	
	U)	/day) (L	/day) (, (1/gm	, (J)gn	ug/L) (r	6) (1/6u	, (9	r) (r	u) (T/gr	5) (1/6u	9 9 9 9	Ű ")رr) (سَ	a/r) (m	g/L) (m	6/L) (%	8) (1	(%) ((g/day	(%)	(%)	(%)	
26/11/91 02/12/91 09/12/91 16/12/91 23/12/91	0 6 20 27	2.7 2.7 3.0 2.8 2.7	0.0 0.0 0.0	1826 2115 2450 1260 1630	1856	2000 1720 1595	1772	18	18	54	26	0.9 1.0 0.6 0.8	6.0					79 95		76 76	4 1	6		
30/12/91 06/01/92 13/01/92 20/01/92	34 41 55	6.2 6.0 6.4	13.4 13.4 13.4 13.4	2285 2185 2310 2160	2235	1438 1067 1075 1040	1155	-47 -20 -7 -7	က က	555 380 450 430	454	527 529 52	2.5	00	120 22	576 452	528	123 87 87	57 57	8858		92 80	76	
27/01/92 03/02/92	62 69	6.3 6.5	000	2535	2244	2070	1826	18	18	180	173	2.9	2.0	0	162	342	768	75	83	83	15 8	7 92	66	
10/02/92 17/02/92 24/02/92 02/03/92 09/03/92 16/03/92	76 97 97 104 111	55740 57740 57740 57740		2478 2570 1910 1985 1985		1922 2479 1405 1768 1837 1301		22 4 11 34 34	34 - 1 26 4 22	0 80 250 327 184		0047000		၀၀၀၀ စ	566 473 778 892 677	566 553 979 1142 870		78 93 64 76 56	71 78 33 35 33 33 33 33 33 33	868886	440040	0 0.00		
23/03/92 30/03/92 06/04/92 13/04/92 20/04/92	118 125 132 139	6.2 6.5 8 8	6.0 0.0 0.0 0.0 0.0	2403 2700	2254	1922 2600	1928	-44 -82	-11 -48	327 251 176	251	2.7 3.1	2.9	47 84 83	777 859 866	1151 1194 1125	157	131 175	40 4 57 \$	8 8	9 2	8 8	49	
27/04/92 04/05/92 07/05/92 Mean	153 160 163	3.5 3.7 3.6	6.0 6.0 6.0	1600 2313 2142	1957	1611 1577 1690	1594	-167 -77 -18	-110 -32 -4	62 41 222	52	1.5	1.3	154 25 32	375 615 552	591 681	636	263 175 108	27 22	968 06 7	5 8 10 6 7	3 97	67	
A REC ATT 1. Filter ATT 2. Filter ATT 2. Filter Nitr 4. REC ATT 6. Since REC Nitr 6. Since REC Nitr 7. A A A A A A A A A A A A A A A A A A A	monia per Tr monia la monia la mitroç f = Me e = Me e = Me	freatment eatment E sading = <i>p</i> ading = <i>p</i> treatment eatment E eatment gen remov gen remov an influer an influer san effluer san effluer san effluer	Efficienc Efficiency Vit 82/4, Efficiency Efficiency Efficiency Ifficency Ifficent am Uday Iffuent am Iffuent am Iffuent am Iffuent am Iffuent am	 /= ((Ae*R) = ((Ne*R) = ((Ne*R) = (1+R)* = (1+R)* = (1+Ne/A)* = (1-Ne/A)* = (ng/L)* = nonia-N (rogin (mg/n)* rogen (mg/n)* 	- + Ai*I) - (I + Ai*I) - (R seed on 18 (Af - Ae)/(I 7*100)*100)*100); mg/L) ;/L)	 x + i)*Af)/(x x + i)*Af)/(x x = i)*Af)/(x x = i)*100 x = i)*100 i = i = i 	Ai-1)*100 In filter, I a to the F en to the F	mean of tBC rath	flow peri	od three) ie system	. filter plu	s RBC rer	noval does	i not eque	al overall	emoval.								
TKN Data																								
Date A	L mm	ÅI V	KN/Amm.																					

1.12 1.27 1.27 1.2 2470 3253 3640 2755 2475 23/12/91 1630 30/12/91 2285 06/01/92 2185 13/01/92 2310 20/01/92 2160

Appendix 7: Metal Data

			Ļ	sachate								Anaerobi	ic Filter	Effluent					-	RBC Em	uent							liter
Date Da	y Flow	/ Reci	yc. Fí	е Д	ط د	ں م	n N	- Dd	VIN (<u>ح</u>		Ц	zu 1) dq	0	v p	4n C	<u>د</u>	7	² e Z	ц ц	ت م	Ŭ	Ā	Ū	ž	- 12	ron Remov.
	(L/dĉ	3D/) (VE	ay) (n	u) (T/Gu	ng/L) (r	ug/L) (i	") (') (') (T/Bm) (T/ɓw)	, (1/gm) (J/gm) (T/ɓɯ;	, (J/gm	(mg/L) ((1) (1) (1	ug/L) (I) (1/gm	") (') ('	mg/L)	(1) (i	1) (1/Bu	л) (Л) (п	л) (л	ш) (¬/бі	л) (л/bi	ig/L) (m	g/L)	(%)
26/11/91 02/12/91 09/12/91 16/12/91 23/12/91	0 13 20 27	2.7 2.7 3.0 3.0	00000	6.75 6.79 6.40	1.24 1.16 1.26	0.33 0.29 0.38	0.08 0.12 0.17	0.01 0.02 0.01	0.42 0.41	3.35 3.20	0.21 0.24	2.20 2.91	0.40 0.48	0.06	0.03 0.08	0.02	0.28 0.24	2.47 2.42	0.21	2.49 2.29	0.52 0.58	0.03	0.06	0.01	0.12	2.52	0.25 0.25	68 55
30/12/91 06/01/92 13/01/92 20/01/92	34 41 55	0000 000 000	13.4 13.4 13.4	9.49 6.44	0.95 0.89	0.33	0.12	0.01	1.70	2.59 2.79	0.25	8.15 4.77	0.76 0.72	0.14 0.16	0.10	0.01	0.42 0.59	2.95	0.25 0.29	6.83 5.72	0.88 0.84	0.14 0.26	0.13 0.13	0.01	0.28	3.20	0.27 0.26	-17 59
27/01/92 03/02/92	62 69	6.3 6.5	00	6.33	0.96	0.31	0.11	0.01	0.84	2.97	0.19	3.46	0.60	0.20	0.06	0.01	0.67	2.81	0.21	4.65 4.18	0.73 0.94	0.20 0.22	0.10 0.16	0.01	0.46 0.36	3.18 4.30	0.33	45
10/02/92 17/02/92 24/02/92	76 90 20	1.5 5.5 6.0	0000	6.12 6.31 6.34	0.99 0.97 1.03	0.24	0.19 0.13 0.16		0.57 0.55 0.49	3.64 3.82 3.82	0.20 0.21 0.27	3.10 3.82 4.75	0.50 0.78 0.84	0.12 0.27 0.28	0.09 0.13 0.14		0.35 0.35 0.39	3.19 3.78 4.09	0.35 0.23 0.29	4.36 5.18	0.91 1.27	0.23 0.24	0.16 0.24		0.33	3.57	0.27	39 39 25
09/03/92 1	50	0.7 8.2	00	14.00	0.43	n.uz			1.30	60.7		01.0 4.52	0.89 0.43	0.10 0.10			0.32 0.82	3.94	0.17	5.04	0.82	0.07			0.24	3.84		65
16/03/92 1	11	5.5	0	6.00	0.79	0.16			0.43		0.22									5.05	0.67	0,14			0.37	-	0.18	
23/03/92 1 30/03/92 1	18 25	6.2 6.9	ю u	5.34	0.90	0.28			0.42	3.49										4.46	0.78	0.21				4.00		
06/04/92 1 13/04/92 1 20/04/92 1	46 33 32	9 9 9 9 9 9 9 9	9999	5.89 5.99 11.30	1.18 1.29 1.13	0.39 0.26 0.28			0.51 0.79 1.04	2.70 3.24 1.91										5.94	1.32	0.92			0.28	2.54		
27/04/92 1 04/05/92 1	53 60	3.5 3.7	မမ	6.70	0.96	0.20			0.47	2.75		4.54	0.50	0.21			0.34	3.21		3.49	0.86	0.11			0.16	3.04		ς,
Mean Values(m	()/			7.30	1.01	0.27	0.13	0.01	0.77	3.05	0.22	4.31	0.63	0.16	0.0	0.01	0.43	3.17	0.25	4.59	0.85	0.22	0.13	0.01	0.32	3.41 (0.27	
Treatment (%) Overall Treatmer	ıt (%)											41	38	42	33	9	43	4	တု	-4	-23 16	-25 17	-35 -2	14 0	15 59	8- <u>1</u> -	-10	

Appendix 7: Metal Data

(L/day) (L/day) (mg/L) (mg/L) (mg/L) (mg/L) (mg/L) (mg/L) ī 1025 4 4 6 - 63 43 -15 -7 ø ÷ ັບ 100 72 4 8 84 66 45 8226 45 99 Mn 0 20 00 0 ß φ. 6-354 б -53 -23 С Overall Treatment Efficiency (%) Fe Zn Pb Cu 0 23 -250 -136 90 74 35 13 25 58 -13 45 67 -23 -12 56 54 ~ 9 24 15 9 13 63 11 31 18 66 16 16 27 7 48 0.0 0.0 0.0 13.4 13.4 13.4 13.4 6.0 6.0 6.0 6.0 6.0 Recyc. 2.7 3.0 3.0 6.2 6.0 6.4 6.3 6.5 6.7 6.7 5.5 5.5 6.2 6.3 9.8 3.5 3.7 Flow 0 13 20 27 55 4 4 34 55 4 4 5 62 69 76 90 97 111 1125 125 132 139 153 160 Day 23/03/92 30/03/92 06/04/92 13/04/92 20/04/92 30/12/91 06/01/92 13/01/92 20/01/92 27/01/92 03/02/92 10/02/92 17/02/92 24/02/92 02/03/92 09/03/92 27/04/92 04/05/92 26/11/91 02/12/91 09/12/91 16/12/91 23/12/91 Date

-29 -74 9 Q -13 ₆ 18





TOTAL

APPENDIX ... ION **CHROMATOGRAPH** SAMPLE OUTPUT

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