INFORMATION TO USERS

This manuscript has been reproduced from the microfilm master. UMI films the text directly from the original or copy submitted. Thus, some thesis and dissertation copies are in typewriter face, while others may be from any type of computer printer.

The quality of this reproduction is dependent upon the quality of the copy submitted. Broken or indistinct print, colored or poor quality illustrations and photographs, print bleedthrough, substandard margins, and improper alignment can adversely affect reproduction.

In the unlikely event that the author did not send UMI a complete manuscript and there are missing pages, these will be noted. Also, if unauthorized copyright material had to be removed, a note will indicate the deletion.

Oversize materials (e.g., maps, drawings, charts) are reproduced by sectioning the original, beginning at the upper left-hand corner and continuing from left to right in equal sections with small overlaps.

Photographs included in the original manuscript have been reproduced xerographically in this copy. Higher quality 6" x 9" black and white photographic prints are available for any photographs or illustrations appearing in this copy for an additional charge. Contact UMI directly to order.

ProQuest Information and Learning
300 North Zeeb Road, Ann Arbor, MI 48106-1346 USA
800-521-0600

Université d'Ottawa · University of Ottawa

ANAEROBIC TREATMENT OF LANDFILL LEACHATE USING A PEAT MOSS FILTER FOR PRE-TREATMENT

Presented by

CATHERINE SAVARD

B. Sc. spécialisé en chimie, Université de Montréal, 1997

A thesis under the supervision of

Dr. RONALD L. DROSTE

Dr. KEVIN J. KENNEDY

A THESIS SUBMITTED IN PARTIAL FULFILLEMENT OF THE REQUIREMENTS FOR THE DEGREE OF MASTER OF APPLIED SCIENCES IN CHEMICAL (ENVIRONMENTAL) ENGINEERING AT THE FACULTY OF GRADUATE AND POSTDOCTORAL STUDIES OF THE UNIVERSITY OF OTTAWA

Department of Chemical Engineering

Faculty of Engineering

University of Ottawa

Ottawa, ON

January 2001

ACKNOWLEDGEMENTS

The author would like to express all her gratitude to her two supervisors, Dr. Ronald Droste, and Dr. Kevin Kennedy, for their academic support, the sharing of their valuable experience and extensive knowledge. Their doors were always opened for a constructive discussion; it was sincerely enjoyable to work with them.

The author would like to express deep appreciation for the financial support from the Natural Sciences and Engineering Research Council (NSERC), Canadian Waste Services Inc., West Carleton Landfill (Carp Road Landfill) and the University of Ottawa. The author wishes to express special gratitude to Mr. Michael Walters of Canadian Waste Services Inc. who gave her his entire trust on carrying the project in its entirety.

Special thanks to Francisco Aposaga for his inexhaustible patience, his resourcefulness, and his unremitting support for the laboratory work. Thanks to Veronique and Anna, resourceful and joyful high school students participating in an academic term project in the Civil Engineering Department and conscientiously performed routine tests for the project. Thanks to Dr. Nimal De Silva, Adjunct Research Professor in the Department of Chemistry at the Carleton University, for his expertise in ICP analysis.

The author would also like to thank all her colleagues for their friendship and mutual aid: Kevin, Awni, Louise, Tham, Majid, Nicole, Paul, Tarek, Lucie, Nancy, and all the others.

Finally, special appreciation goes to the author's family for demonstrating a continuous support for her academic goals, especially her husband Pascal for his patience, his love, and his assistance.

Abstract

The feasibility of anaerobically treating a leachate from a landfill site in the Ottawa-Carleton Region, ON, was studied. The levels of specific pollutants present in the leachate, COD, Fe, B and Ba, were required by the landfill owners to be treated in order to meet the local sewer user bylaw (SUB) discharge limits. The leachate had a high COD concentration (from 14 to 40 g COD L^{-1}); however most of the organic content was readily biodegradable (up to 98%).

In preliminary work, the landfill leachate was treated without peat pre-treatment in a bench-scale UASB reactor. Reactors were filled with different amounts of sludge (33, 50 and 66% of the total volume of the reactor) and OLRs varied from 3.0 to 6.2 g COD L^{-1} d^{-1}. In these initial runs, inorganic suspended solids present in or precipitating from leachate accumulated in the reactors, displacing biomass and increasing biomass inert solids content. Biomass in the bottom part of reactor was the most affected by this accumulation. To overcome the inert solid accumulation, peat moss filtration used as a pre-treatment was considered because peat is a cheap and easily available medium, which is re-usable in a landfill site as daily cover material.

The studied leachate was concluded to stimulate anaerobic bacterial microbial activity; the maximum SLR to be fed to reactors was found to be 0.42 g COD g^{-1} VSS d^{-1}. Therefore the minimal biomass amount required for a given removal efficiency on CRL could be calculated from OLRs and reactors volumes. The reactors were able to achieve high COD removal efficiencies (82-97%), however, the continuing accumulation of FSS indicated that the reactors would not be able to sustain their performance in the long term.

The filtration of the leachate by peat moss was found to be an excellent means of removing suspended solids. Peat moss filtration had another important advantage; it had the capacity to adsorb specific soluble metals and inorganic ions. The removal of Al, B, Ba, Ca, Co, Fe, Mn, and Zn was studied; peat was found highly efficient for removal of Al, B, and Zn (at studied concentrations). However, peat was inefficient in achieving sufficient removal of Mn for a permitted discharge value of 5.00 mg L^{-1}. Ba desorption from the peat was observed, which was concerning since Ba has a low discharge limit of 1.20 mg L^{-1}. Fe, Co and Ca removal was variable but deemed unimportant because influent concentrations were below the discharge limit.

The core experiments of this work were the treatment of the leachate by the combined peat-filter-UASB reactor process. Three series of reactor runs were conducted; feed leachate COD concentration and reactors flow rates were varied in order to mimic seasonal landfill leachate conditions (spring, summer and fall conditions). Within the range of operational conditions studied, the combined peat-UASB treatment could remove high COD concentrations

and meet the regional SUB discharge limits. The adequate operational settings were concluded to be: an HRT as low as 0.95 d and an OLR of 6.5 g COD L_r^{-1} d^{-1} for spring conditions; an HRT of 1.4 d and an OLR of 4.7 g COD L_r^{-1} d^{-1} for fall conditions; an HRT of 1.6 d and a maximum OLR of 10.0 g COD L_r^{-1} d^{-1} for summer conditions. These operating conditions assume peat pre-filtration, leachate concentrations tested, an adequate COD:P ratio of 500:1, an operation temperature of 32°C and a minimal recycle to feed ratio of 6.25:1.

From solids profiles, the studied leachate treatment stimulated anaerobic biomass growth. However, at moderate and high feeding rates, while biomass growth increased, significant inert solids accumulation occurred, causing biomass calcification and biomass displacement. Peat moss pre-treatment had an irrefutable positive effect on solids accumulations, but long-term operation inevitably resulted in inert solids accumulation in sludge beds. Therefore, other pre-treatment methods or improvement of peat filtration should still be investigated.

Finally, the removal of soluble inorganic ions by the combined treatment was studied. High Ca removal efficiencies were observed because Ca was the main precipitating element in UASB reactors. In order to prevent Ca precipitation in reactors, it was suggested to add lime or caustic prior to peat filtration; Ca would precipitate as calcium carbonate ($CaCO_3$) and could be physically removed by filtration. B and Ba were removed at levels below their SUB discharge limits; Fe initial concentration was low but its removal was good. If higher Fe concentrations were experienced in the studied leachate, peat filtration would efficiently remove suspended Fe whereas soluble Fe would precipitate within UASB reactors under anaerobic conditions. Al, Mn, and Zn removals in the peat filter-UASB reactor process were high whereas Co results were not conclusive.

Abrégé

L'objet de la présente étude est la faisabilité du traitement d'eaux de lixiviat d'un site d'enfouissement de déchets situé dans la région d'Ottawa Carleton, ON, par un traitement biologique anaérobie. À la demande des propriétaires du site d'enfouissement, les concentrations de polluants spécifiques, tels que la matière organique, le fer, le bore et le baryum, devaient être abaissées pour rencontrer les niveaux fixés par le règlement municipal de décharge dans les égouts. Les eaux de lixiviat étudiées avaient une forte concentration en demande chimique en oxygène (DCO) (de 14 à 40 g DCO L^{-1}); malgré cela, la majeure partie de la matière organique était facilement biodégradable (jusqu'à 98%).

Dans un premier temps, les eaux de lixiviat furent traitées directement, sans pré-traitement, par un réacteur anaérobie à écoulement ascendant et à lit de boues (AÉALB) de petite dimension (environ 6 L). Les réacteurs furent remplis à différents niveaux avec de la boue activée (33, 50 et 66% du volume total des réacteurs) et les débits volumiques d'alimentation en matière organique (DVAO) variaient de 3.0 à 6.2 g DCO L^{-1} j^{-1}. Durant ces essais initiaux, la matière inorganique en suspension ou en solution présente dans le lixiviat a provoqué une accumulation de matière inerte dans les granules de boues activées, déplaçant ainsi les micro-organismes du lit de boues et provoquant la calcification de celui-ci. La partie inférieure du lit de boues a été affectée de façon plus importante que le reste du lit. Pour surmonter le problème d'accumulation de matière inorganique, un filtre de mousse de tourbe a été utilisé en guise de pré-traitement; le choix de la mousse de tourbe comme substance filtrante a été pris en raison du faible prix et de l'abondance de cette matière, de même que du fait que la tourbe peut-être réutilisée dans les sites d'enfouissement en guise de couche d'épandage quotidien sur les déchets.

Une des conclusions de l'étude sur les eaux de lixiviat est que celles-ci stimulent l'activité biologique anaérobie; un débit massique d'alimentation en matière organique de 0.42 g DCO g^{-1} MVS j^{-1} (matière en suspension volatile, MSV) maximum a été mesuré. Ainsi, la quantité minimale de biomasse requise pour le traitement adéquat des eaux de lixiviat peut-être calculé à partir de ce débit massique, de la concentration en DCO des eaux et du volume des réacteurs. Les réacteurs AÉALB ont traité efficacement le lixiviat; de 82 à 97% de la DCO a été enlevée des eaux au terme du traitement; par contre, l'accumulation de matière inorganique a nuit considérablement au traitement si bien qu'un traitement à long terme pourrait être difficilement réalisable.

Une autre conclusion de l'étude est que la filtration du lixiviat par la mousse de tourbe est un excellent moyen pour enlever la matière en suspension dans celui-ci. La mousse de tourbe possède un deuxième avantage important; elle a la capacité d'adsorber certains métaux et ions inorganiques présents dans le lixiviat. L'enlèvement des éléments suivant a été étudié: Al, B, Ba, Ca, Co, Fe, Mn, et Zn. La mousse de tourbe a été très efficace pour enlever les éléments Al, B, et Zn aux concentrations étudiées. Par contre, la tourbe a été inefficace pour l'enlèvement du Mn en deçà de la limite de déversement de 5.00 mg L^{-1}. Une désorption de l'élément Ba a été observé, ce qui est préoccupant puisque la limite de déversement du Ba est basse (1.20 mg L^{-1}). L'enlèvement des éléments Fe, Co et Ca a été variable quoique non cruciale puisque tous ces éléments se retrouvaient initialement déjà en deçà de la limite permise.

Les expériences centrales de cette étude furent le traitement des eaux de lixiviat par la combinaison d'un filtre de tourbe et d'un réacteur AÉALB. Trois séries d'expériences furent menées; trois réacteurs furent opérés simultanément, les concentrations en DCO et les débits des réacteurs ont été variés de façon à simuler les changements saisonniers (printemps, été, automne) des eaux de lixiviat dans un site d'enfouissement réel. Le traitement du lixiviat par le filtre et le réacteur combinés a été hautement efficace pour abaisser le niveau de la DCO en deçà de la limite de déversement permise. Les conditions optimales d'opération pour les réacteurs furent fixées comme étant un temps de rétention hydraulique (TRH) de 0.95 j et un DVAO de 6.5 g DCO L_r^{-1} j^{-1} pour les conditions printanières, un TRH de 1.4 j et un DVAO de 4.7 g DCO L_r^{-1} j^{-1} pour les conditions automnales et un TRH de 1.6 j et un DVAO maximum de 10.0 g DCO L_r^{-1} j^{-1} pour les conditions estivales. Ces conditions sous-entendent que le lixiviat est pré-traité par un filtre de tourbe, que la DCO des eaux de lixiviat demeurent à l'intérieur des limites des concentrations testées, qu'un ratio DCO:P adéquat de 500:1 soit disponible pour les micro-organismes, que la température des réacteurs soit de 32°C et qu'un ratio minimum de 6.25:1 soit utilisé pour le débit de recyclage versus celui de l'alimentation.

À partir du profile de concentration des bio-solides, il a été établi que le traitement combiné stimule l'activité biologique anaérobie mais qu'en alimentant les réacteurs à forts débits ou à débits modérés, le problème d'accumulation de matière inorganique demeure et entraîne toujours une calcification du lit de boues. La filtration du lixiviat par la tourbe a définitivement eu un impact positif sur le traitement, mais le traitement à long terme reste problématique et résulte inévitablement en une accumulation de matière inorganique dans le lit du réacteur. Une amélioration de l'étape de filtration ou la recherche de méthodes plus efficaces de pré-traitement du lixiviat devront être envisagées.

Finalement, l'enlèvement des ions inorganiques par le traitement combinés a été étudié. Grâce à sa précipitation dans le lit des réacteurs, le Ca a été éliminé du lixiviat avec succès. Pour prévenir cette précipitation dans les réacteurs, il est fortement suggéré que le Ca soit enlevé du lixiviat par précipitation chimique (addition de chaux ou de caustique) avant d'être filtré par la tourbe: le Ca précipiterait ainsi sous forme de carbonate ($CaCO_3$) et pourrait être physiquement éliminé du lixiviat avant que celui-ci soit alimenté aux réacteurs. Les concentrations de B et de Ba du lixiviat ont été abaissées en deçà de la limite de déversement permise; l'enlèvement du Fe était satisfaisant étant donné que sa concentration était déjà en deçà de la limite permise de déversement. Les concentrations des éléments Al, Mn, et Zn ont été abaissées avec succès en deçà de la limite de déversement, par contre, les études sur le Co furent non-concluantes.

TABLE OF CONTENT

LIST OF TABLES

LIST OF FIGURES

Abbreviations used in this work:

AAT	acetoclastic activity test
BOD	biochemical oxygen demand:
BOD_x	biochemical oxygen demand: x corresponds to the test length in days
COD	chemical oxygen demand
COD_{rem}	chemical oxygen demand removed
CRL	Carp Road Landfill
EPA	Environment Protection Agency
FAS	ferrous ammonium sulphate
FS	fixed solids
FSS	fixed suspended solids
GC	gas chromatograph
HRT	hydraulic retention time
K_s	half-velocity constant (mg COD L^{-1})
L	litre
L_g	litre of gas
L_r	litre of reactor
M	molar (mole L^{-1})
Mt	metric ton
NA	natural attenuation
N/D	non detectable
OLL	old landfill leachate
OLR	organic loading rate
POTW	publicly owned treatment works
RMOC	Regional Municipality of Ottawa-Carleton
SLR	specific organic loading rate
SRR	specific organic removal rate
SRT	solids retention time
SS	suspended solids
SUB	sewer use bylaw
TKN	total Kjeldahl nitrogen
TOC	total organic carbon
TP	total phosphorus
TSS	total suspended solids
UASB	upflow anaerobic sludge blanket
VFA	volatile fatty acid
VRR	volumetric removal rate g COD L_r^{-1} d^{-1}
VS	volatile solids
VSS	volatile suspended solids
YLL	young landfill leachate

CHAPTER 1

INTRODUCTION

Landfilling is the cheapest and the most widely used means of solid waste disposal. Up until about thirty years ago, solid wastes were disposed directly on the ground without any aftercare or they were buried with little concern for their potential environmental hazard. In Canada, this type of landfill is referred to as a natural attenuation landfill and is now illegal, unless very small, because of the potential contamination of surrounding groundwaters and land. Today, engineered landfill cells, or waste containment systems, are generally adopted as the prime landfill type and are used for solid waste disposal. Landfill cells are lined with synthetic liners and clay and covered with impermeable liners, in order to prevent pollutant emissions from escaping from the landfill site.

These new landfill cell systems are not an environmental panacea. The two main sources of pollution emanating from these landfill cells are gaseous emission, called biogas, and liquid emission, called landfill leachate. Those two emissions have to be controlled or treated in order to minimize their noxious effects on environment. Landfill leachate is the subject of interest in this research. Landfill leachate production is the result of rainwater trickling through the solid waste, and becoming contaminated by the organic and inorganic matter that it is contacting. Since landfill liners are impervious to water, landfill leachate accumulates on the liner leading to liquid mounding in the landfill. Landfill leachate is a highly polluted wastewater and its treatment is particular and complex, as are its physico-chemical characteristics. Since leachate cannot accumulate indefinitely on the liner, it eventually has to be treated.

The objective of this research is to find an efficient means of treating an industrial landfill leachate, by concomitantly lowering the concentration of its organic and inorganic contaminants. The landfill owner has proposed that the leachate be roughly treated prior to being discharged in the regional publicly owned treatment works (POTW) as a final treatment. The treated leachate is required to meet the regional sewer use discharge limits, which are higher than discharge limits in to natural waterways.

1.1 LANDFILLS IN ONTARIO

Ontario is the most populated province of Canada with a population of approximately 11 million in 1997. Of that number, 750 000 are residents of the Regional Municipality of Ottawa-Carleton (RMOC). In Canada, the average per capita solid waste production rate is approximately 1.7 kg d^{-1}. Extrapolation of this production rate to RMOC's population results in over a million kilograms per day of municipal solid waste. Despite the sustained efforts of the RMOC to encourage its citizens to recycle, the amount of waste directed to landfills remains enormous. The size of this problem demands that solid waste disposal should be properly managed to minimize its impact on the environment.

According to the Ontario Ministry of Environment web site (http://www.ene.gov.on.ca/envision/faq/index.htm#Doesusing, 1999), there were 38 public landfills, nine private ones and several smaller specific landfills registered in Ontario. Adding to that, 10% of municipal waste is trucked to American landfills, and only 5% are burnt in three energy-from-waste incinerators (Government of Ontario, 2000).

Ontario municipal regulation is clear about landfill design and the protection of the surroundings of a landfill site, as can be seen from this excerpt:

> "Ontario Regulation 232/98
>
> *Landfill sites must be well designed for groundwater and surface water protection to prevent effects from site operation and to facilitate site closure and post-closure care. The site design report required by the standards must describe all site features in sufficient detail to confirm the design will be acceptable. The design report must address:*
>
> *1- site boundaries, buffer area, waste fill area and contours, surface water control works, on-site roads and structures and final cover design,*
>
> *2- the design of any liner and leachate collection system,*
>
> *3- any air emission control works,*

4- monitoring facilities for groundwater, leachate and surface water,

5- a contingency plan for leachate control and

6- site closure and post-closure care requirements. "

It is the fifth point, a need for a contingency plan that has launched this research, which is supported by Canadian Waste Services (CWS). CWS operates the Carp Road Landfill (CRL, Carp, ON) site, which is currently receiving domestic and industrial solid wastes. Presently, CWS discharges its landfill leachate to the RMOC wastewater treatment plant in Gloucester, ON. However, long-term predictions demonstrate a probability of exceeding legal sewer discharge limits. Consequently, CWS has begun to evaluate the treatability of their landfill leachate using peat pre-treatment followed by anaerobic digesters as part of their contingency plan.

Various types of treatment are available for leachate; anaerobic bioreactors are known to efficiently treat high strength wastewater such as leachate. The upflow anaerobic sludge bed (UASB) reactor has two important advantages: high biomass solid retention times (SRTs) and ability to treat high volumetric organic loads. The UASB reactor was chosen in this study as the best option for removal of organic matter from the Carp Road leachate.

However, one deficiency of the UASB system is the potential negative affects of inert solids accumulation in the lower part of the UASB reactor resulting in biomass displacement. In order to minimize this accumulation of inert solids in the reactor, leachate pre-treatment was required. Peat filters were used as a pre-treatment to remove suspended solids as well as some other soluble inorganics. Peat was chosen because of its filtration properties, low price, and its abundance in Canada. Moreover, peat can potentially remove some soluble metals present in landfill leachate; this potential for metal removal was studied, as a side bonus of suspended solids removal.

CWS required that their leachate be pre-treated to decrease its chemical oxygen demand (COD), iron (Fe), barium (Ba), and boron (B) concentrations in order to meet RMOC sewer discharge limits at all times.

1.2 RESEARCH OBJECTIVES

The four objectives of this study were

i. decrease COD, Fe, B, and Ba concentrations of CRL leachate to meet sewer discharge regulations;

ii. demonstrate that the UASB reactor is an efficient means of achieving (i);

iii. remove suspended inorganic solids from the CRL leachate by pre-treating it with a peat filter;

iv. study the soluble metals removal capacity of a peat filter treating raw CRL leachate.

1.3 THESIS LAYOUT

In Chapter 2, the literature review describes the main background theory of this research: landfill design, leachate production and quality, the Carp Road Landfill site, anaerobic digestion, UASB reactors, and peat as a filtering media.

In Chapter 3, apparatus, material, and methods are described. In Chapters 4, 5, and 6, results of laboratory experiments are detailed. The structure of those three chapters is the following: a short introduction outlines the research to be presented, followed by specific materials and methods used only for the described experiment, followed by results, discussion and conclusions for each aspect of the experiment. In Chapter 4, preliminary study results on UASB treatment are disclosed. In Chapter 5, peat filtration, as leachate pre-treatment, results are described and Chapter 6, regarding combined peat filter and UASB reactor treatment, discloses the main core of thesis experiments. Finally, in Chapter 7, the summary conclusions correlate the experimental results to the research objectives.

CHAPTER 2

LITERATURE REVIEW

Prior to choosing a treatment process for a landfill leachate, characterization of the leachate is required. The landfill design, type of solid waste, landfill age, landfill operations/maintenance and climatic conditions are important parameters influencing the quality and quantity characteristics of landfill leachate. These points are addressed in Sections 2.1 and 2.2. A description of the CRL Site is found in Section 2.3. The general principles of anaerobic digestion and the application of UASB reactors for treatment of leachate are overviewed in Sections 2.4 and 2.5. Finally peat moss and its use as a filter pre-treatment are described in Section 2.6.

2.1 LANDFILLS

The design and construction of an engineered landfill are complex and expensive, however, new waste containment systems are efficient in controlling landfill pollution sources (leachate) and minimizing their impact on the environment. The general design of a landfill, its leachate collection system, and the landfill closure are overviewed in this section.

2.1.1 LANDFILL DESIGN

Since the early 1970s, scientists and engineers have been concerned with liquid pollutant discharges from landfill sites. These aqueous liquids, called leachates, are generated from liquid resulting from precipitation percolating through the waste pile, contacting and dissolving or transporting organic and inorganic pollutants emanating from waste degradation. Leachate is highly polluted and difficult to characterize because its composition varies greatly with landfill age, season, climate, landfill operational management, sampling point, and solid waste type.

In the past, natural unlined attenuation landfills were used as a means of land disposal of solid wastes; however, their failure to contain leachate within the boundary of the landfill site led to the design of more elaborate means: liquid and solid waste containment cells ($LSWC^2$). $LSWC^2$ systems are lined with layers of natural (clay) or synthetic material (PVC liners)

restricting leachate leakage out of the landfill and comprise a leachate collection/recycling system. Once a landfill is filled to its maximum capacity, the waste pile is covered with a synthetic and/or clay liner to minimize further rainfall infiltration and concomitantly decrease leachate production.

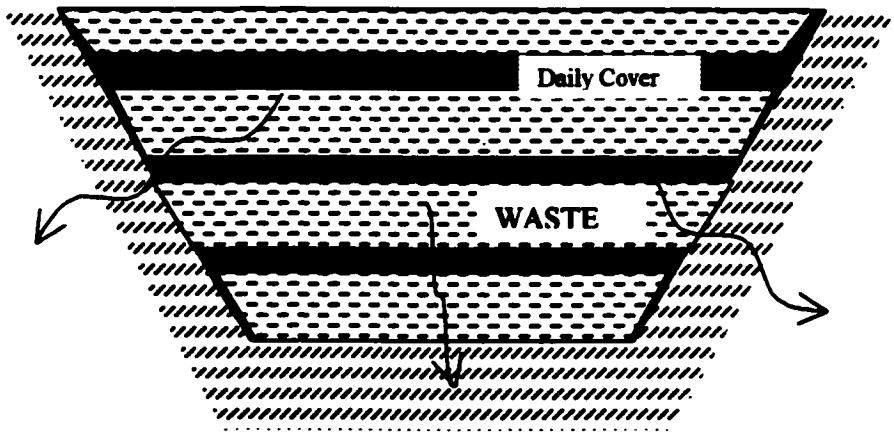


Figure 2.1: Natural attenuation landfill

2.1.1.a Natural attenuation landfills

Natural attenuation (NA) landfills were typically used as the main landfill technique in the past; no barriers were put in place to prevent liquid emissions from escaping from the landfill base and migrating freely in the soil (Figure 2.1). Leachate purification was expected to occur naturally by unsaturated soil filtration and adsorption. However, soil saturation was generally reached and contamination of surrounding groundwater and soils was unavoidable. Now, NA landfills are rarely permitted in Canada except for small landfills containing only domestic waste.

The oldest part of the CRL consists of closed and covered NA cells. Leachate has migrated in soil beneath those cells and has contaminated the groundwater aquifer. These contaminated groundwaters are less concentrated than leachate produced from open operating cells and they were not used in the present study. However these groundwaters could be used to decrease the organic strength of the active cell leachate by dilution prior to treatment in a reactor and therefore facilitate operating and loading conditions of a possible on-site treatment. Presently dilute leachate is pumped from wells and trucked to the RMOC's R.J Pickard secondary

wastewater treatment plant. The newer section of the CRL (Figure 2.2) is designed as a waste containment system: it includes a leak-proof synthetic liner, clay liner, and a leachate collection system.

2.1.1.b Waste containment systems

The waste containment landfill concept is to severely restrict leachate percolation into and through the soil beneath the landfill base by lining the landfill with impermeable layers (Figure 2.2). These impermeable layers consist of clay liners or synthetic geomembranes liners (polyvinylchloride, PVC or high density polyethylene, HDPE), or most often a combination of both, and a leachate collection system. The CRL cell from which the studied leachate originates from is a waste containment system, designed with impermeable clay and synthetic membrane layers. Different types and layers of liners are installed in relation to the environmental hazards associated with the waste contained in the landfill: domestic waste requires fewer layers of liners than highly toxic refuse.

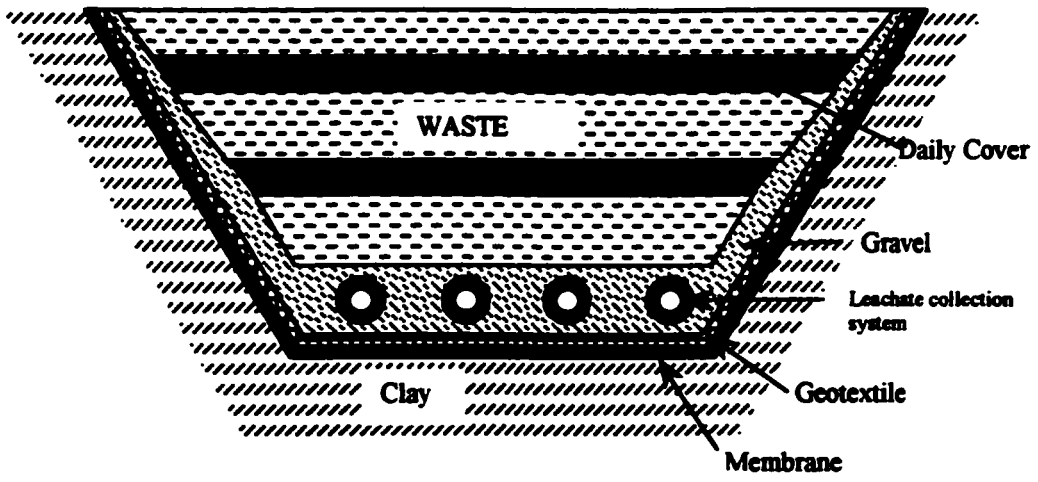


Figure 2.2: Waste containment system

2.1.1.c Leachate collection system (LCS)

The role of the LCS is firstly to minimize the leachate head on the landfill liner by pumping the leachate out of the landfill. The LCS is made of pipes placed horizontally on top of the liners; their role is to convey the leachate to wells or sumps. Leachate collected in the wells

can be pumped and hauled to a municipal sewer system, can be piped directly into a sewage collector, re-injected on top of the refuse to increase degradation, or directed to an on-site treatment system. The CRL leachate used in the present study was sampled directly from a LCS well.

2.1.1.d Landfill closure

The waste containment system is covered with an impermeable barrier (Figure 2.3) at the end of its life span to minimize new leachate accumulation and to lower maintenance costs, including leachate treatment. Following closure of the landfill, the leachate composition and volume change and the method of treatment of leachate used while the landfill was open tend not to be appropriate or as efficient as when the landfill was operational. As the landfill ages, the main changes in leachate composition are a decrease in the biodegradability of the organic matter and an increased concentration of soluble metals. Therefore, in order to suggest a valuable long-term leachate treatment to the client, the life span of the landfill should be taken into consideration.

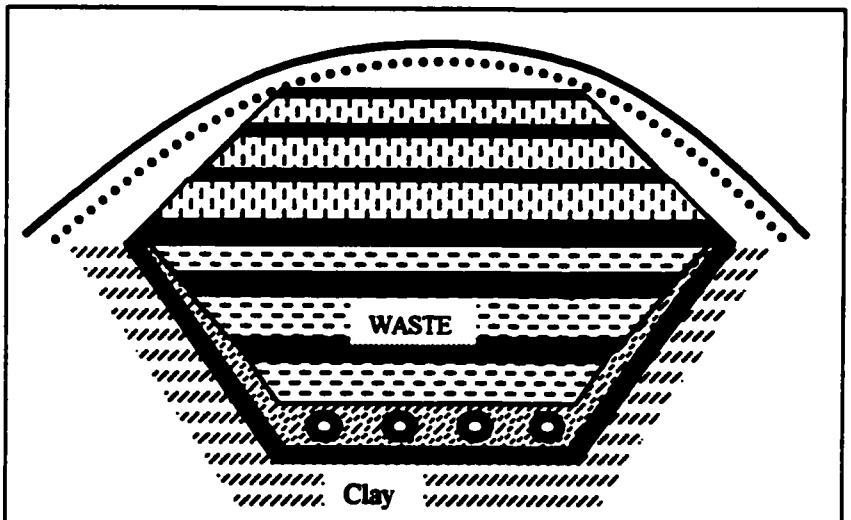


Figure 2.3: Closed landfill

2.2 LANDFILL LEACHATE PRODUCTION AND QUALITY

The quantity of leachate produced is directly related to the climatic and geographic conditions of the landfill site and the size of the landfill. On the other side, the quality of the leachate is related to landfill age and landfill operation/management, leachate-sampling point, refuse composition, and seasons. These factors have a great impact on the leachate composition and are briefly described in this section.

2.2.1 LEACHATE PRODUCTION

The volume of leachate produced varies widely due to climatic and geological conditions and landfill size. Based on United States Environment Protection Agency (EPA) studies an average landfill is estimated to have a leachate generation rate of approximately 114m^3 d^{-1} (30 000 gallons per day (gpd), EPA, 2000). Seasonal variations in rainfall cause fluctuations in leachate volume. In a rainy season such as fall, the volume of leachate produced is greater than during a dryer season such as summer. Moreover, the leachate pollutant concentrations change due to dilution effects and volume of leachate generated.

Leachate generation rates vary with annual local precipitation, which relates to geographic location. In Canada, there will be low leachate production during winter because precipitation is in the form of snow (solid). In springtime however, the production will suddenly increase with the melting of snow cover and leachate produced will be more diluted. In general the leachate flow rate is directly correlated to the surface area of the active portion of the landfill.

2.2.2 LEACHATE QUALITY

Concentrations of organic and inorganic pollutants in the leachate are related to external factors such as landfill age, season, and internal factors such as landfill operation/management and refuse composition. It is the interrelationship of all these factors that determine the quality of the leachate.

2.2.2.a Landfill age

Leachate quality varies with landfill age. Méndez *et al.* (1989) published a detailed study on the treatment of a young landfill leachate (YLL) from a recently opened landfill, versus a 20-

year-old landfill leachate (OLL) from a landfill that was about to be closed (Table 2.1). Both landfills are located in Spain. According to their study, significant differences were discovered between the two leachates: larger concentrations of proteins, ammonia-nitrogen, suspended solids (SS), volatile suspended solids (VSS) and chemical oxygen demand (COD: soluble and total) and a higher alkalinity were found in the YLL compared to the OLL values. On the other hand, phosphorus (as phosphate) was not detected in the YLL, whereas it was problematic in the OLL; where it exceeded the sewer discharge limit of 10 mg L^{-1}. Therefore, while the OLL needed phosphorus removal to meet regulations the YLL required addition of phosphorus for adequate biological treatment of the leachate (see anaerobic digestion nutrients requirements, section 2.4.2.d).

Table 2.1: Characteristics of leachate versus their age (Méndez *et al.*, 1989)

	Leachate from older landfill (20 years)	Leachate from younger landfill (recently opened)
PH	7.6	7.3
COD total	0.83	4.10
COD soluble	0.65	3.50
SS	0.20	1.20
VSS	0.12	0.60
$N\text{-}NH^{4+}$	0.28	0.80
$P\text{-}PO_4^{3-}$	0.12	0.00
Proteins	0.00	0.50
Alkalinity	1.9	3.16

All units are g L^{-1} except for pH; alkalinity is as $CaCO_3$

It has been observed that the leachate from OLL is generally less biodegradable than YLL ones; the BOD:COD ratio of leachates from younger landfills typically exhibit a ratio of approximately 0.8, while older landfills exhibit a ratio as low as 0.1 (EPA, 2000). Méndez *et al.* (1989) also found that the COD from the OLL contained a high non-biodegradable organic fraction (40%) whereas the YLL COD was mainly highly degradable volatile fatty acids (VFAs). The refractory fraction of the OLL was reported to be made up of mainly high-molecular weight organic polymers. Britz *et al.* (1990) reported that YLL could be treated anaerobically in a UASB reactor with over 90% COD removal, indicating a BOD:COD ratio of close to 0.9. Spengel and Dzombak (1991) treated an OLL (close to 50 years-old) with a rotating biological contactor

(RBC) to remove COD and nitrogen; they could only remove a maximum of 38% of COD and 80% of BOD_5: the BOD:COD ratio was reported to be less than 0.5.

Iza and co-workers (1992) studied on-site the treatability of a YLL from a landfill in Massachusetts, USA. Characteristics of the leachate are shown in Table 2.2; COD concentrations varied between 0.2 to 6.4 g L^{-1} while the BOD_5 concentration was between 0.03 to 4.5 g L^{-1}, the average BOD:COD ratio was therefore 0.6. Iza *et al.* reported the leachate to be high in VFAs and highly biodegradable; biological COD removal efficiencies averaged 90%.

The EPA (2000) reported that no specific time frame or relation could be used to describe changes in leachate characteristic. As mentioned previously, many factors influence the leachate quality (refuse composition, landfill operations, climatic data) and the ageing of landfill leachate. EPA (2000) noted that, depending on landfill construction, leachate could originate from different cells of different ages, opened or closed; and that the characteristics of the resulting leachate mixture will not reflect any particular landfill cell age.

2.2.2.b Leachate quality variation: batch experiment

Pohland *et al.* (1983) studied batch decomposition of domestic refuse and monitored changes with time of various parameters (Figure 2.4). Parameters of interest providing information on landfill ageing and leachate quality are COD and total volatile acids (TVAs) curves: both parameters reached a maximum within 200 days after the initiation of the experiment. In a batch experiment, soluble COD and TVA concentrations are simultaneously increasing due to hydrolysis and decreasing as a result of biodegradation of the organic fraction; the rate of hydrolysis is greater than the rate of degradation until day 200, thereafter it is the opposite situation that occurs. In a full-scale landfill, a continuous supply of new waste would maintain the biodegradable fraction; COD and TVA would achieve a steady state and the respective curves would plateau rather than decreasing after 200 days.

Table 2.2: Characteristics of raw leachates found in literature

Reference / Parameter	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(8)	(9)
COD (tot)	—	3.47-8.22	58.4	55-70 [c]	37-66.7 [c]	—	0.2-6.4 [d]	0.2-30.7 [d]	27.3-36.7 [d]	1.7-52
COD (sol)	0.63-3.2	2.91-6.37	—	16.4	—	5.1-8.3	—	—	—	—
BOD_x	1.2-1.9 [a]	—	40.0 [b]	—	15.7 [b]	2.2-4.2 [b]	0.03-4.5	—	—	—
pH	6.5-7.0	7.2-8.0	5.5	5.9	6.3-6.4	7.6-9.3	6.1-7.8	7.8-8.8	7.6-8.3	6.3-8.6
Alkalinity	1.6-2.67	3.16	8.5	4.6	2.33	1.3-2.3	—	0.75-12.1	7.0-11.7	0.73-14.3
Type	YLL, D	YLL, D	YLL, D	OLL, I	YLL, D	I	YLL, D	—	—	YLL

All values are in g / L except pH

a: BOD_x; x = 7

b: BOD_x; x = 5

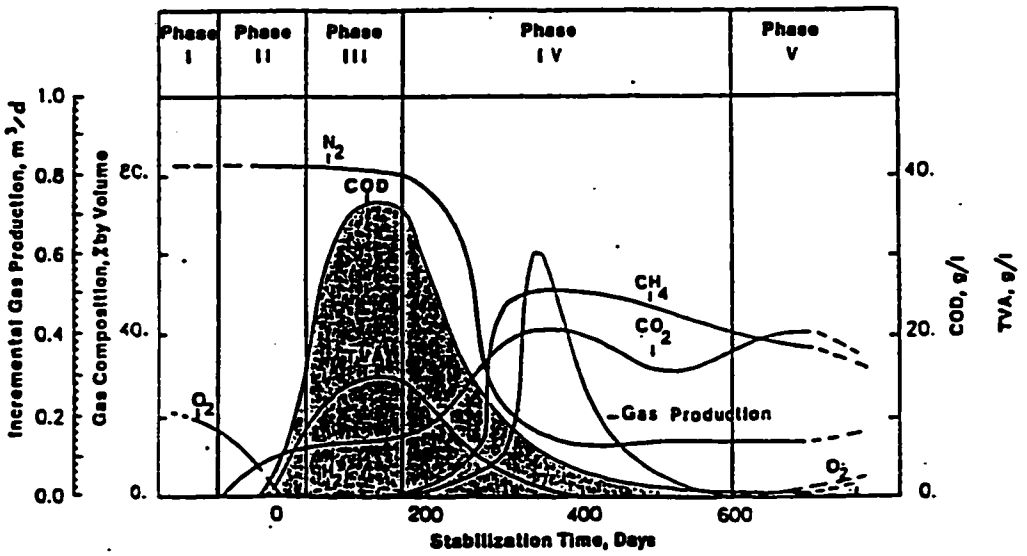
c: tot COD on a different leachate than other parameters

d: not specified if tot. or sol. COD

Type: YLL = young landfill leachate
OLL = old landfill leachate
D = domestic waste
I = industrial waste

(1): Kettunen and Rintala (1998)
(2): Méndez *et al. (1989)*
(3): Chang (1989)
(4): Rumpf and Ferguson (1990)
(5): Lin (1991)
(6): Sung (1997)
(7): Iza *et al. (1992)*
(8):Myburg and Britz (1993)
(9): Britz *et al. (1990)*

Figure 2.4 : Phases of landfill stabilization (after Pohland *et al.* 1983)



2.2.2.c Refuse composition

The composition of the received refuse is the most representative characteristic of a landfill and, therefore, of the leachate generated since its main contaminants are derived from the landfilled material. Waste types are divided in two categories and leachates emanating from each type of landfill will be categorized: non-hazardous materials (domestic solid waste) to hazardous wastes (industrial and toxic solid wastes). Differences between hazardous and non-hazardous landfill reside not only in the nature of the pollutants themselves but also their concentrations. According to the EPA (2000), 32 pollutants monitored are of interest for non-hazardous landfills whereas 63 pollutants are of interest and must be monitored for hazardous landfills.

The CRL receives domestic and industrial solid wastes and is classified as a hazardous industrial landfill site; therefore, the leachate from that landfill is classified as hazardous.

2.2.2.d Landfill Operations and sampling point

Landfill operations, especially the material used as the daily cover can influence the leachate quality. Daily cover is essential to properly operate a landfill, particularly when a large refuse fraction is putrescent material, and the site is close to housing. Cover controls odour, reduces windblown debris, facilitates access for trucks and provides a media for partial attenuation of the leachate (Bagchi, 1990). In specific cases, daily cover can be contaminated

material and the degree of contamination can influence the leachate quality. Hydrocarbon impacted soils or chemical contaminated soils can be used in a hazardous or co-disposal landfill as daily covers (practised at CRL); in this case, leachate contamination could increase.

The leachate sampling point within the landfill site is also a parameter affecting the reporting of leachate quality. Méndez *et al.* (1989) reported on the influence of the sampling location on analytical characteristics of the leachate. All samples must preferably be taken from the same location in a landfill site to eliminate the possibility of variation caused by location. In the current study, the samples were all taken from the same sampling location and on the same date, decreasing variation related to time and location.

Table 2.3: Six-year-old landfill leachate concentration seasonal variations (Méndez *et al.*, 1989)

	Winter	Spring	Summer	Autumn
COD total	3.72	8.22	5.84	3.47
COD soluble	3.57	6.37	4.89	2.91
PH	8.0	8.0	7.2	7.9
SS	0.71	1.29	1.22	1.25
VSS	0.47	0.71	0.65	0.64
$N\text{-}NH_4^+$	1.05	1.04	0.80	0.70

All units are g L^{-1} except for pH.

2.2.2.e Seasonal changes

Iza *et al.* (1992) evaluated the seasonal biotreatability and anaerobic toxicity of a YLL and concluded that seasonal changes did not influence leachate toxicity effects that inhibited anaerobic micro-flora. Only the volume of leachate and its concentration change with season affected biodegradation efficiency. According to Méndez *et al.* (1989), and shown in Table 2.3, leachate in Spain is the most concentrated in spring, and less concentrated in the fall season. Sampling of the leachate to characterize it or to store it for subsequent study should therefore take into consideration the time of the year in order to avoid unrepresentative results.

2.2.3 LEACHATE COMPOSITION

As discussed above, leachate composition is highly variable from one landfill to another according to factors described above. General characterization of landfill leachate is given as the range of concentrations within which most leachate components are usually found.

2.2.3.a Organic content

Several organic compounds are solubilized during the waste stabilization process; some are readily degradable organics and others are complex and harder to characterize. According to EPA (2000), because of the complex nature of landfill leachate, it is common to find the sum of individual organic contaminants not always matching the measured total organic carbon (TOC) and/or COD value. This difference is attributed to high molecular weight organics, from which exact carbon content is hard to estimate. Most of the soluble organics in leachate are VFAs; more complex organic fractions include carbohydrates, proteins, and humic and fulvic-like substances (EPA, 2000).

Table 2.2 lists characteristics of leachates reported in the literature; COD concentrations range from 0.2 to 70 g COD L^{-1} for total COD and from 0.63 to 16.4 g COD L^{-1} for soluble COD. BOD varies from as low as 0.03 to 40 g L^{-1} for both domestic waste and young landfill leachate.

2.2.3.b Metals, pH and sulphide content

Landfill leachate usually has a high metal and salt content, in both soluble and suspended forms. Metals such as iron, zinc, aluminium, lead, and copper are present in landfill leachate in ranges from 0 to 5500 mg L^{-1} (Table 2.4). Certain metals (e.g., Fe or Cu) stimulate anaerobic bacteria at low concentration but are inhibitory to anaerobic microbial activity at higher concentrations. In general, soluble heavy metals are toxic to anaerobic micro-flora, especially methanogenic bacteria (see Section 2.4.1) at concentrations as low as 0.1-10 mg L^{-1} (Parkin and Owen, 1986).

The pH of leachate, reported on in the literature (Table 2.2), varies from 5.5 to 9.3 whereas the alkalinity ranged from 0.75 to 14.3 g L^{-1} as $CaCO_3$. Because carbonates and other buffering ions are typically present in leachate, leachates rarely have extremely high or low pHs.

EPA (2000) published typical pH ranges from 3.7-8.9 for domestic waste leachate and from 5.8-11 for industrial, hazardous leachates (Table 2.4). The alkalinity range, from the same table, is typically 0 to 20 850 mg L^{-1} as $CaCO_3$.

Soluble sulphur in leachate is most often present as sulphate and will be reduced, under anaerobic conditions, to hydrogen sulphide, which will precipitate with iron and other soluble metals present in landfill leachate. This precipitation decreases the toxic effect of sulphide and metals on anaerobic flora. Carbonates, which also form insoluble metal precipitates, will also precipitate with cations if they are present in sufficient concentrations (Iza *et al.*, 1992).

Table 2.4: Literature leachate characteristics (EPA, 2000)

Leachate constituent	Units	Municipal Concentration Range	Hazardous Concentration Range
BOD	mg BOD L^{-1}	7 – 720 000	22-2963
COD	mg COD L^{-1}	0 – 750 000	270-6873
TOC	mg TOC L^{-1}	0 – 30 500	2-3824
TKN	mg TKN L^{-1}	0 – 3320	
Nitrate nitrogen (NO_3^- N)	mg NO_3^- L^{-1}	0 – 1300	0.38 – 193
Total phosphorus (PO_4^- P)	mg PO_4^- L^{-1}	0 – 234	0.01 – 15.9
Total alkalinity (as $CaCO_3$)	mg $CaCO_3^-$ L^{-1}	0 – 20 850	
Solids (TSS)	mg L^{-1}	0 – 59 200	32 – 568
PH		3.7 – 8.9	5.8 – 11
Al	mg L^{-1}	0 – 122	---
Ba	mg L^{-1}	0 – 12.5	---
B	mg L^{-1}	0.3 – 73	0.5 – 8.2
Ca	mg L^{-1}	5 – 7200	---
Cu	mg L^{-1}	0 – 10	0.009 – 0.61
Fe	mg L^{-1}	0 – 5500	3.6 – 36.8
Pb	mg L^{-1}	0 – 14.2	---
Mg	mg L^{-1}	0 – 15 600	8.3 – 440.8
Mn	mg L^{-1}	0.03 –1400	0.081 – 9.045
Ni	mg L^{-1}	0.01 – 7.5	0.06 – 2.87
K	mg L^{-1}	2.8 – 2800	---
Na	mg L^{-1}	0 – 7700	---
Zn	mg L^{-1}	0 – 1000	0.046 – 0.846

2.3 CARP ROAD LANDFILL SITE

A geographical description of the CRL site, its evolution in time, the waste composition and the origin of the leachate produced are discussed in this section.

2.3.1 LANDFILL SITE

The leachate used in this study was collected in September 15, 1998 from the CRL. The site is located in the eastern part of the Regional Municipality of Ottawa-Carleton (RMOC), ON, Canada. The total site area, including buffer lands, is 1.385 km^2; the approved permitted landfill footprint is 0.346 km^2 of which 0.243 km^2 are already used and covered, while 0.103 km^2 are still in operation. The site is generally flat; no important elevation, other than the pile of waste itself, can be seen.

The average climate data for the Region of Ottawa Carleton (45°19-N 75°40-W) according to Environment Canada (1998), from 1961 to 1990, are summarized in Table 2.5.

Table 2.5: Ottawa climate normals 1961-1990 (Environment Canada, 1998)

	January	April	July	October	Year
Daily mean temperature (°C)	-10.8	5.6	20.8	7.9	5.8
Precipitation: rainfall (mm)	15.3	58.0	88.1	70.3	701.8
Precipitation: snowfall (cm)	49.6	9.1	0.0	4.1	221.5
Precipitation: overall (mm)	58.0	69.0	88.1	74.8	910.5
Days with measurable precipitation	16	12	11	13	159

2.3.2 LANDFILL CELLS

The landfill site has evolved over time. Initially CRL started with natural attenuation (NA) cells, which were opened in 1970. These cells are now closed and covered. The (NA) cells are not equipped with a leachate collection system or leak-proof liners. In 1992, the present active 25.5 acres cell was opened and lined with a polymeric liner and clay material. This cell is classified as a waste containment system and is projected to be operational until 2012.

2.3.3 WASTE COMPOSITION

Waste composition in this landfill is 40% domestic waste and 60% industrial and commercial waste. The composition as a whole is, by weight, petroleum-impacted soil (45.3%), commercial (30.5%) and residential (16.4%) wastes, wastewater sewage solids (5.8%) and special wastes (1.9%). Approximately 240 000 metric tons (mt) of waste and up to 140 000 mt of hydrocarbon impacted soils are accepted each year. Of these, 210 000 mt of waste are landfilled, the hydrocarbon impacted soils are used as daily cover, and the rest of the waste is recycled. Annual recycling breakdown is wood 32% (13 000 mt), biosolids 62% (25 000 mt) and metals 6% (2500 mt).

2.3.4 LEACHATE

2.3.4.a Leachate production

The leachate produced originates from two different sources: groundwater purge wells and the active cell leachate collection system. The original 60 acres of natural attenuation cells has contaminated surrounding groundwater by pollutant migration. This contaminated groundwater is collected by 11 purge wells and fed into the to the municipal sewage system collection network at a flow rate of 44 300 L h^{-1}. The active waste containment cell produces leachate at 2840 L h^{-1} over a surface area of 25.5 acres. Total leachate production is roughly 47 000 L h^{-1} or one million L d^{-1}.

2.3.4.b Leachate sampling

Leachate was sampled from the leachate collection system well from the active waste containment system cell. No contaminated groundwater from the natural attenuation cells was sampled or studied. The leachate was collected in 200 L plastic barrels, and transported to the University of Ottawa where it has been kept frozen at –4°C until needed. When fresh leachate was needed, a barrel was thawed and kept at 4°C until emptied. Barrels were usually half to two-thirds full, and the last 20-35 L in the barrel bottom was not used because of its high content of suspended solids and grit.

CRL leachate characteristics are shown in Section 3.2.1.

2.4 ANAEROBIC DIGESTION

Anaerobic digestion is the metabolism of carbon-based substrates by microorganisms in absence of oxygen with the production of methane and carbon dioxide biogas as waste stabilization end products. An important difference between anaerobic and aerobic microbial treatment resides in the composition of their flora: aerobic flora is composed of a multitude of different microorganisms whereas anaerobic flora is composed mainly by a defined consortium of bacteria. The key to successful anaerobic digestion is the interactions among its bacterial populations. If these interaction mechanisms are neglected or ignored, instability or failure of the process can occur.

The basis of anaerobic digestion microbiology and biochemistry and the factors influencing anaerobic digestion will be presented in this section.

2.4.1 ANAEROBIC MICROBIOLOGY AND BIOCHEMISTRY

Anaerobic digestion is a succession of closely related biochemical and enzymatic activities composed essentially of four main phases accomplished by five different bacterial communities. This consortium is organized in a multi-step food chain degrading particulate organics to soluble organics, and finally to methane, hydrogen, and carbon dioxide as end products.

The primary step of anaerobic digestion is the liquefaction of insoluble particulate organics; this biochemical process is called hydrolysis. Insoluble organic matter is hydrolysed by extracellular bacterial enzymes and reduced to smaller, soluble organic molecules. This step does not contribute to organic stabilisation; organics are only converted into readily assimilated forms for microorganisms. For liquid wastes comprising an important fraction of suspended particles, such as municipal solid waste, hydrolysis is the rate-limiting step of the food chain considering the substrate availability only (Parkin, and Owen, 1986).

Soluble molecules are transported across the bacterial cell membrane and are transformed further into short-chain volatile acids, alcohols, lactic acid, and mineral compounds as end products: this second step is called acidogenesis. The bacteria involved in that step are mainly producing VFAs by fermentation and are therefore called acid-forming bacteria. The main VFAs produced are acetic, propionic, butyric, valeric, and caproic acids. Hydrogen is also produced by

acidogenesis and is inhibitory to acid-forming bacteria if not removed from the liquor. Fortunately hydrogen is removed by the last microbial biochemical reaction.

The third biochemical process, called acetogenesis, produces acetate and hydrogen as main end products. A group of bacteria forms directly acetate from hydrolysed substrate. Another group, called acetogens, forms acetate from hydrogen (H_2) and carbon dioxide (CO_2). The fourth and final step referred to is methanogenesis. Methane-producing bacteria, called methanogens, are subdivided into two groups: hydrogenophilic and acetophilic methanogens. Hydrogenophilic methanogens produce methane autotrophically from H_2 and CO_2:

$$4H_2 + CO_2 \rightarrow CH_4 + 2H_2O$$

Acetophilic methanogens produce methane from acetate:

$$CH_3CO_2H \rightarrow CH_4 + CO_2$$

Each represents 28% and 72% of methane production in anaerobic reactors respectively (Droste, 1997).

If the methanogenesis process did not proceed, no substantial COD removal would be noticed from the treated wastewater. This final step is the major stabilisation step; as the COD is removed from wastewater in the form of methane gas (insoluble in water). For soluble wastes, methanogenesis is usually the rate-limiting step; the slow growing rate of the methanogens and their sensitivity to environmental factors make this step the most crucial step of the sequential anaerobic digestion process.

The oxidation reaction of methane is

$$CH_4 + 2\,O_2 \rightarrow 2H_2O + CO_2$$

Therefore, two moles (64 g) of COD are removed for each mole of methane (16 g) produced by the anaerobic reactor. At standard temperature and pressure, this ratio can be express as 0.35 m^3 of methane produced per kg of COD removed (0.39 $m^3\ kg^{-1}\ COD_{rem}$ at 32°C; van Haandel and Lettinga, 1994). This relation is useful to approximate the volume of methane produced by waste treated (see section 2.4.2.e).

2.4.2 FACTORS AFFECTING ANAEROBIC DIGESTION PERFORMANCE

2.4.2.a Rapid loading variation

Methanogens are sensitive to rapid increases of the organic loading; increase in the loading is one of the primary causes of process imbalance and eventual failure. Sudden increases in the rate of organic matter loading stimulates microbial acid production, which can potentially inhibit the methanogenic activity by inducing a brutal drop in pH and a sudden VFA accumulation.

Britz *et al.* (1990) observed that phenomenon for a UASB reactor subjected to abrupt organic load changes; the recovery of the reactor's performance took a few days. However, Britz and co-workers reported that as the degree of the abrupt increases of ORL was enlarging, the UASB reactor took longer to recover a stable operating state, until total failure (no gas production) was observed. The failure was attributed by Britz *et al.* to an imbalance between specific acid production (fast) and the slower conversion of VFA and hydrogen and carbon dioxide to methane.

2.4.2.b Temperature

Anaerobic microorganisms are classified according to the temperature range: psychrophilic (15–20°C), mesophilic (30–40°C), or thermophilic (50–60°C). Mesophilic range is the most commonly used in anaerobic digesters; for reasonable methane production, reactor temperature should be maintained above 20°C. Experiments conducted at lower temperature (from 13°C to 23°C) by Kettunen and Rintala (1998) proved that COD removal efficiencies similar to mesophilic digestion can be maintained only if the OLR are decreased due to slower metabolic reaction rates at lower temperatures.

2.4.2.c pH

The optimum pH range is set in respect to the needs of the more sensitive microorganisms, methanogenic bacteria, in a pH range of 6.5 to 7.7. Neutral pH reduces inhibitory effect of VFA on the methanogenic population; at lower pH, VFAs are protonated and inhibit acid-forming bacteria.

Maintaining a stable neutral pH is essential; the use of a carbonate-buffered solution helps keeping pH at a constant value. Careful monitoring of the pH, buffering capacities and alkalinity of the bulk liquid is important to avoid reactor failure.

2.4.2.d Nutrients requirements

Nutrients are essential to maintain a healthy anaerobic flora; the two most important nutrients are nitrogen (N) and phosphorus (P). The microbial yield of anaerobic bacteria is low, and therefore the ratio BOD to N to P to be used in anaerobic digesters is also low: 500:5: 1 (BOD:N:P) (van Haandel and Lettinga, 1994; Droste, 1997)

2.4.2.e Gas production

Anaerobic digestion produces two main gases: methane (CH_4) and carbon dioxide (CO_2). CH_4 production usually ranges from 60-70% of total gaseous emissions, CO_2 represents the complementary portion. Small amount of hydrogen, hydrogen sulphide (H_2S), ammonia, water vapour, and other gases are also present. Methane is high in potential energy and can be used as a combustible energy source. Therefore, it is desirable to obtain the highest possible methane production to gain high-energy return.

2.5 UASB REACTOR

Lettinga and co-workers developed the UASB reactor in the 1970s. This anaerobic reactor system has been successfully treating high strength wastewater, such as landfill leachate, and has resulted in high treatment efficiencies (Kettunen and Rintala, 1998, Méndez *et al.*, 1989, and Myburg and Britz, 1993 in Table 2.2). Presently, pilot- and full-scale UASB reactors are operated throughout the world to treat many industrial wastewaters, as well as landfill leachate and domestic sewage. The UASB schematic set-up is shown in Figure 2.5: the main features of the UASB are high biomass retention capacity and high-rate treatment at short hydraulic retention times (HRT). Relevant characteristics and features of the UASB reactor, useful for experimental bench-scale treatment of landfill leachate, are reviewed in this section.

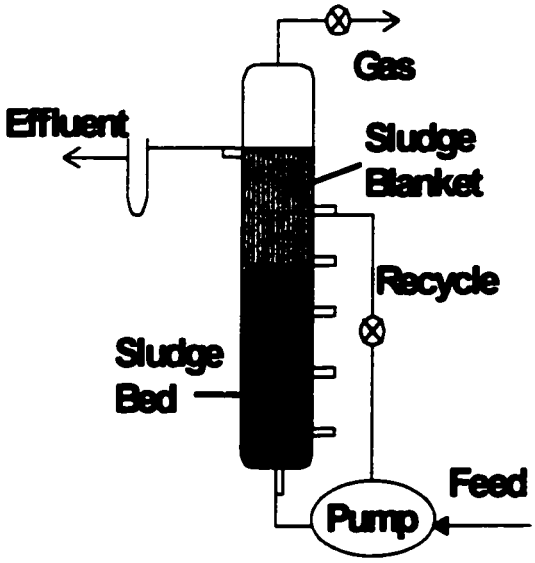


Figure 2.5: UASB reactor schematic configuration

2.5.1 UASB CONFIGURATION

The UASB reactor system is configured as a biological reaction zone and a sedimentation zone. The major part of the anaerobic digestion occurs in the biological reaction zone, called the sludge bed. This set-up has the advantage to be operated at high SRT without sludge recycle. The feed flows in the upward direction through the bed, which is composed of highly active (usually granular) anaerobic microbial sludge.

Some improvements have been made to the original UASB reactor, such as the addition of an anaerobic filter (AF) in the upper sedimentation zone of the reactor. The combination of those two configurations is called the UASB-AF or hybrid reactor. The original UASB configuration was preferred in the present experiment because of lower capital cost and shorter start-up time (for film growth on the AF).

2.5.1.a UASB to treat leachate

Some UASB, UASB-AF and other types of reactors have been or are currently treating landfill leachate (Table 2.6). In Sweden, a pilot-scale UASB reactor has successfully treated landfill leachate at room and lower temperatures (Kettunen and Rintala, 1998). Rumpf and Ferguson (1990) achieved a COD removal over 90% by feeding an UASB reactor at loading rate of 9.4 kg COD m^{-3} d^{-1}. Britz *et al.* (1989) successfully treated a YLL with an UASB-AF reactor,

Table 2.6: Literature anaerobic treatment of landfill leachate

Reactor type	Temperature (°C)	OLR kg COD m^{-3} d^{-1}	HRT d	average COD removal (%)	Methane yield L CH_4 g^{-1} COD_{rem}	Reference and notes
UASB	18-23	3.3	0.96	57-60		(1)
UASB	19-23	3.2	0.54	71-77		(1)
UASB	12-14	1.5	1.3	49-55		(1)
UASB	37	0.44 to 1.47	10 to 3	44-58 a)		(2) digesters 2,3 and 4
UAF	37	0.31 to 1.92	10.67 to 1.85	55.8-64.5 b)		(2)
UASB-AF	35	1.43 to 21.79	7.67 to 2.68	67.9-92.8 a)		(3)
UASB-AF	35	21.65 to 29.0	1.0	84.5-95	0.19-0.24	(4)
UASB-AF	37	9.15 to 12.99	2.0 to 1.4	91-93	0.203-0.276	(5): YLL
UASB-AF	37	14.53 to 20.54	1.2 to 0.9	82-89	0.281-0.299	(5): YLL
UASB	35	9.4	2.3	87 a)		(6) recycle velocity = 2 L min^{-1}
CM	35	1.14 to 2.84	20 to 8	92-95		(7)

(1): Kettunen and Rintala (1998)
(2): Méndez et al (1989)
(3): Chang (1989)
(4): Myburg and Britz (1993)
(5): Britz et al (1990)
(6): Rumpf and Ferguson (1990)
(7): Lin (1991)

a) total COD
b) soluble COD

achieving COD removals from 82 to 93% for OLRs of 18.6 to 10.3 kg COD m^{-3} d^{-1}, respectively. Iza *et al.* (1992) mentioned that YLL characteristics are favourable for high rate anaerobic treatment and many research groups have successfully treated this type of leachate by high-rate anaerobic digestion.

2.5.1.b Problems with UASB configuration

Among the problems encountered with the UASB system is the possibility of inert material accumulation in the sludge bed, leading to a decrease of specific metabolic activity. This problem is discussed in details in Section 2.5.5 of this chapter. Secondly, UASB reactors are an unprofitable option for inorganic particulate removal from raw wastewater because there is no frequent sludge wastage. Furthermore, UASB reactor has a history of granule flotation or sludge bed washout after a transient shock loads or sudden temperature changes (Kennedy, 1999). Sludge flotation has also occurred when reactors were fed after a long starvation period. Biomass settling properties are discussed further in Section 2.5.2b. Finally, while UASB reactors remove large amounts of COD from concentrated wastes, the effluent quality does not generally meet surface water discharge criteria.

2.5.2 GRANULAR SLUDGE

UASB reactor's sludge is usually granular. The UASB configuration takes advantage of the two major benefits of this type of sludge: excellent settling properties and high microbial activity. Good sludge settling properties enable the SRT to be longer than the liquid HRT. Additionally high specific microbial activity is coupled with high microbial concentrations in reactors, enabling reactors to be operated at high organic loads and short HRT while achieving substantial COD removal efficiencies. The sludge acclimation, settling properties, and storage are reviewed in this section.

2.5.2.a Sludge acclimation

The process of sludge granulation is relatively long and delicate and is not fully understood. To avoid such a lag time, active granulated sludge from another waste treatment stream may be used to fill UASB reactors to accelerate start-up. In such a case, an acclimation period is required since the inoculum sludge is often accustomed to a different type of wastewater. A low loading rate is initially favoured followed by an incremental increase of

loading rates while maintaining constant removal efficiencies; this procedure results in acclimation.

Most of experiments reported in the literature have used already granulated sludge from a running reactor and the length of the acclimation period is varied. Examples of acclimation are:

- Méndez *et al.* (1989) studied the digestibility of a YLL with a sugar refinery granular sludge. A long HRT was used at first (15 days), followed by decreasing HRTs (10, 8, 6, 5, 4, and 3 days) over a period of 250 days allowing acclimation of sludge to new substrate type.
- Iza *et al.* (1992) utilised an inoculum previously fed on lactose and batch fed it with leachate over an acclimation period of 45 days prior to starting their experiments.
- Britz *et al.* (1989) utilised a mixture of active laboratory sludge and municipal sewage-sludge. Their reactor was fed with a synthetic acetate-propionate solution over a period of 10 days to stimulate anaerobic bacteria growth. The studied leachate replaced synthetic feed and the flow rate was slowly increased until the desired HRT was reached.

2.5.2.b Sludge settling properties

Sludge settling properties are important to provide a SRT longer than the critical bacterial doubling time, and avoid sludge washout. Washout of microorganisms occurs when a small diameter granule is carried out of the reactor with effluent liquid, or when a normal-size granule has a gas bubble attached to it. In the latter case, granules tend to float and may escape from reactor with the effluent, unless a gas-solid-liquid separation device is installed.

Generally larger granules have poorer settling properties than smaller ones. When granules grow to a point that substrate can no longer diffuse to the inside of the granule, the microbial cells in this inner region start endogenous decay. Cells decay and biogas resulting from it cause a gas-filled core to appear in granules centre; their densities become lower than the bulk liquid density and granules will tend to float.

2.5.2.c Sludge storage

It has been demonstrated that long-term storage has little effect on mesophilic granule activity when kept under favourable conditions. Wu *et al.* (from Kettunen and Rintala article, 1998) observed negligible effects on biomass activity when stored anaerobically at 5°C for 2-8 weeks. Jian *et al.* (1989) studied long-term storage effect on microbial activity and reported after

a latent period of 3 months an entire return of activity within 2 to 7 days. Granules stored at room temperature (20°C) for 9 months recovered their initial activities within two weeks in a UASB reactor (Kennedy, 1999). From the same reference, it is noted that periodic feeding of stored granules will maintain their metabolic activity.

2.5.3 UASB OPERATION

2.5.3.a Hydraulic retention time and loading

The HRT of a system is a function of the treatment capacity of UASB reactors and the influent concentration whereas the treatment capacity is related to sludge inventory and biomass activity. If a sludge bed has a high biomass concentration, higher loading rates (*i.e.* lower HRTs) may be achieved. HRTs and reactor loading rates are also a function of operating temperatures: a lower temperature (less than the optimum mesophilic temperature) is usually related to a higher HRT or a low volumetric organic loading rate.

HRTs reported in the literature:

- Kettunen and Rintala (1998) reported that a UASB reactor could successfully treat landfill leachate at a HRT of 0.96± 0.02 d at a temperature of 23 ± 1°C. To maintain the same COD removal efficiency at lower temperature (13!1°C) they had to increase the HRT to 1.30± 0.07 d.
- Britz *et al.* (1989) concluded that a HRT of 0.9 d, for a leachate feed COD concentration of 18 000 mg L^{-1} (ORL of 20.54 kg COD m^{-3} d^{-1}) was the lowest HRT possible for a hybrid UASB-AF, otherwise, reactor failure occurred within 24 hours.

2.5.3.b Temperature

Kettunen and Rintala (1998) studied UASB reactor performances at low temperatures. Their results showed a wide variation of biogas production with temperature changes: from 210 to 410 mL CH_4 g^{-1} $COD_{removed}$ for a temperature variation from 13 to 23°C respectively. When the organic loading rate was maintained constant, COD and BOD_7 removals went down: from an average of 74% and 95% respectively at 18-23°C, removals dropped to 52% and 72% for COD and BOD, respectively, at 13-14°C (Table 2.2). They concluded that leachate treatment with a

UASB reactor was feasible at temperature as low as 13°C; however, at the expense of a lower removal rates and efficiencies.

2.5.3.c Mixing and recycling

One important factor to obtain high COD removals is optimizing contact between substrate and substrate-utilising bacteria. In UASB reactors, effluent recycling and gas bubbles are responsible for mixing. Effluent recycle also dilutes the feed concentration, decreasing toxic component concentrations or shock load impact while increasing reactor upflow velocities that appear to help the granulation process. In general the recycle-to-feed ratio employed in UASB reactors is about 4-6 to 1.

2.5.3.d Pseudo-steady state

Iza *et al.* (1992) reported that a period of one month was necessary to establish a pseudo-steady state for a UASB reactor running at a HRT of 1 d. Their leachate was highly variable and was the cause of this long acclimation time (on-site pilot-scale UASB-AF). Myburg and Britz (1993) defined the pseudo-steady state as a stable state reached after five volume turnovers when there was less than 10 % variation of parameters. However, when running a UASB-AF reactor on leachate at a HRT of 3.5 d, Britz *et al.* (1990) defined the steady state as less than 10 % variation but assumed that four volume turnovers were sufficient.

2.5.3.e Metabolic activity

In order to quantify UASB reactor performance, specific methanogenic and acetoclastic activities, and specific COD removal rates are commonly measured. Generally, higher volumetric activities are found in granular sludge compared to flocculent sludge because of higher VSS content per volume. Mesophilic activities range from 7-8 kg COD kg^{-1} VSS d^{-1} for pure culture consuming readily degradable volatile acids, to 0.5-1.0 kg COD kg^{-1} VSS d^{-1} for heterogeneous anaerobic consortia treating a complex wastewater. For VFA mixtures, methanogenic activities of 1.3-1.8 kg CH_4–COD kg^{-1} VSS d^{-1} and 2.1 kg CH_4–COD kg^{-1} VSS d^{-1} for acetate only feed have been reported (Kennedy, 1999). Kettunen and Rintala (1998) reported activities from 0.05 to 0.16 kg CH_4–COD kg^{-1} VSS d^{-1} for an UASB reactor treating a landfill leachate at 22-24°C.

2.5.3.f Methane and biogas theoretical production

Van Haandel and Lettinga (1994) have detailed theoretical methane and biogas productions. As mentioned in Section 2.4 of this thesis (on anaerobic digestion), for each 16 g of methane produced, 64 g of COD is removed; or equivalently, for each kg of COD removed, 250 g of methane is produced. Assuming the ideal gas law to be valid, the volume of one mole (16 g) of methane produced at a temperature T (in °K) is expressed as 22.4 T / 273. Thus, to calculate the theoretical methane volumetric production per kg of COD removed for a temperature T, equation 2.5.1 may be used (van Haandel and Lettinga, 1994):

$$(250/16) \times 22.4\ T / 273 = 1.28\ T \quad L\ CH_4\ kg^{-1}\ COD_{removed} \tag{2.5.1}$$

To calculate biogas production, partial pressure of methane, p_m, and methane collected fraction, f_m are used. Methane partial pressure in digesters is usually 0.8 atm (Van Haandel and Lettinga, 1994). Methane collected fraction, f_m, is a correction factor used to incorporate methane lost by dissolution in effluent or by leakage from gas tubing. Methane solubility in water at atmospheric pressure is about 20 mg L^{-1}; if methane represents 70% of gaseous phase, 14 mg L^{-1} could be lost in the effluent. Biogas production, v_b, is theoretically calculated with equation 2.5.2 (van Haandel and Lettinga, 1994):

$$v_b = f_m\ 1.28\ T / p_m \quad L\ kg^{-1}\ COD_{removed} \tag{2.5.2}$$

Myburg and Britz (1993) obtained methane production of approximately 0.19 to 0.24 m^3 kg^{-1} $COD_{removed}$ for leachate and reported a theoretical optimum on a glucose substrate of 0.395 m^3 kg^{-1} $COD_{removed}$.

2.5.3.g Operational problems

Clogging and precipitation in pipes are frequent problems in UASB or UASB-AF as reported by Iza and co-workers (1992). Clogging of the influent piping, inorganic precipitation and denser granular sludge were also observed to lead to operational problems (Kennedy *et al.*, 1988).

2.5.4 UASB REACTOR FEED

2.5.4.a Nutrients

Méndez *et al.* (1989) reported the need for phosphate addition during their experiment because of low phosphorus concentration in their leachate. Rumpf and Ferguson (1990) did not require any nutrients addition at an OLR of 9.4 kg m^{-3} d^{-1}, total P initial concentration was 3.0 mg L^{-1} and was not limiting. They reported 79% P removed.

2.5.4.b Influent pre-treatment

In order to avoid inert solids or particulate accumulation in the lower part of the UASB reactor, influent pre-treatment is to be considered if feed has a high-suspended solids content. Iza *et al.* (1992) concluded, after running an UASB-AF reactor treating a YLL for several months, that a pre-treatment was required if long-term operations were planned. Pre-treatment would remove inert solids as well as metals and inorganic ions in order to prevent the calcification of the sludge bed.

Rumpf and Ferguson (1990) suggested pre-treating landfill leachate to remove iron and calcium prior to feeding it to an UASB reactor. They suggested recycling a portion of the reactor's effluent to a mixing tank with raw leachate so that the effluent's higher pH and alkalinity would precipitate Fe and Ca out of the leachate.

2.5.5 FIXED SOLID ACCUMULATION AND METALS REMOVAL

One problem with the UASB reactor's upflow treatment through a sludge bed is the accumulation of fixed solids in reactor granules. When an accumulation of inert material occurs in UASB reactors, the biomass activity decreases and granule calcification occurs. On the other hand, this accumulation does result in a beneficial metal removal from the wastewater, though at the expense of biomass displacement from the sludge bed.

Kettunen and Rintala (1998) observed inert solids accumulation in their UASB reactor. They reported that sludge VS content decreased from 51% in the inoculum to 21 to 30% in the reactor's bottom and middle zones respectively. Main precipitants were reported to be calcium (24% of TSS) and iron (7% of TSS).

Iza *et al.* (1992) also observed an accumulation of heavy metals in the reactor's sludge bed after several months of operation on leachate. Mineralization of the sludge occurred and reactor failures were noted due to bed compaction and channelling. They published a hypothesis on the envelopment of biomass by solid precipitation causing biomass flotation: entrapped bacteria would be deprived of fresh substrate and would undergo endogenous decay. This would create hollow granules and eventually lead to sludge bed washout because of the granules increased buoyancy.

Rumpf and Ferguson (1990) reported soluble metal removal in an UASB reactor treating landfill leachate; full removal of soluble Cd, Cr, Co, Cu, and Pd, high removals (≥ 94%) of Ca, Fe, Mn, and Zn, and significant removals of Mg (36%), P (79%), and Ni (68%) were measured. Precipitates on their reactor's wall were analysed and identified as $CaCO_3$, $Ca_5(PO_4)_3(OH)$, $FeCO_3$, and FeS; because of low sulphide and phosphate concentrations, carbonate precipitates were predominant. However, heavy metals precipitated predominantly as metal sulphides. At the end of the running period, a change in sludge physical character was observed: it became denser, coarser, and highly inorganic. Kennedy (1999) reported granule ash fractions of 30% and above for a UASB reactor treating landfill leachate; major cations were Na, K, Ca, Mg, and Fe, whereas P and S were the main non-metal precipitates.

Kettunen and Rintala (1998) observed a decrease of soluble metal out of UASB reactors treating leachate. The major metal removal mechanism was concluded to be precipitation: in Table 2.7, removals of metals of interest from Kettunen and Rintala's experiments are shown. Boron, aluminium, cobalt, and zinc were present but no significant removals were observed.

Table 2.7: Kettunen and Rintala (1998) results for leachate metal removal by a UASB reactor

Metal	Initial concentration range ($mg\ L^{-1}$)	Days 69-71 (approx. T = 24°C)	Days 132-135 (approx. T = 22°C)	Days 223-226 (approx. T = 15°C)
Al	< 0.14	*no significant removal*	*no significant removal*	*no significant removal*
B	1.13 – 1.99	*no significant removal*	*no significant removal*	*no significant removal*
Ba	0.206 – 0.303	15	25	12
Ca	233 – 524	31	7	*no significant removal*
Co	<0.02	*no significant removal*	*no significant removal*	*no significant removal*
Fe	15.0 – 27.1	73	78	50
Mn	1.14 – 2.37	72	34	18
Zn	0.057 – 0.076	*no significant removal*	*no significant removal*	*no significant removal*

Chang (1989) also observed metals removal from an hybrid UASB-AF reactor treating leachate. Total Fe removal ranged from 14.6% to 69.8% whereas initial concentrations (total Fe) were from 179 to 515 mg L^{-1} respectively. Soluble Fe removal was constantly above 90%, ranging from 90 to 95.9% for initial concentrations of 82 and 232 mg L^{-1}, respectively. Chang reported that sulphate influent concentration increased with increasing Fe concentration, therefore, the iron removal mechanism was assumed to be iron sulphide precipitation.

2.6 PEAT MOSS

Peat moss has been used since the 1900s as a filtering media because it is inexpensive, plentiful, and easily accessible. Peat moss is also known to be an excellent sorbent for organics as well as dissolved metals. In landfilling operation, peat moss is used as a filter or sorbent because it is tolerant to variations in hydraulic and contaminant loading; moreover, after being used as a filtration media, it may be used as daily cover.

Canada is the world's largest peat moss producer (600 000 dry tonnes y^{-1}) and has the largest peat moss reserves (150 million hectares of peat mosslands) (*Use of adsorbents for the removal of pollutants from wastewater*, 1996). Important information to acquire prior to using peat moss are; its chemical structure, mechanisms of sorption, and factors influencing sorption and compaction. Peat moss composition, characteristics and applications in water pollution control are reviewed in this section.

2.6.1 PEAT MOSS COMPOSITION

Peat moss is essentially fossilized plants and trees; its composition varies greatly from one source to another. Its basic constituents are a mixture of lignin, cellulose and humic acid. In a macroscopic visual examination, peat moss is a brownish fibrous and spongy material. It might contain small pieces of wood, roots, or bough. Depending on the type of trees or plant that peat moss originates from, the morphology of peat moss will vary: for instance non-woody fine fibrous versus amorphous-granular peat moss (MacFarlane, 1969). On the chemical (molecular) scale, peat moss is described as natural wood polymers with surfacial polar functional groups. Polar functional groups include carboxylic acid, alcohol, aldehyde, ketone, and phenolic hydroxide acids (Viraraghavan and Rana, 1987). These functional groups can be involved in

hydrogen or ionic bonding. The chemisorption capacity of peat moss is important; its polar functional groups can potentially interact with ions such as metals or polar organic molecules.

2.6.2 PEAT MOSS FILTERING CAPACITIES

Peat moss may be used as a filter to remove pollutants from a wastewater by three means: physical removal, chemical sorption and biological removal. Physical removal is entrapment of suspended solids from wastewater by the peat moss fibres. Chemical adsorption is the natural capacity of peat moss to act as a resin; dissolved ions are chemically bonded to peat moss. The third removal mechanism is biological: a microflora is present in the first layers of peat moss top and is responsible for degrading BOD, as well as killing pathogenic and faecal coliforms. This microflora is mainly composed of antibacterial-producing fungi; growth of bacteria is inhibited by these fungi (Chevalier, 1996).

2.6.3 PEAT MOSS ADSORPTION CAPACITIES

Adsorption capacity of peat moss is assumed to rely on different mechanisms: ion exchange, chelation and electrostatic attraction. Although many research groups have investigated metal adsorption mechanisms, a shared perspective on the precise mechanism of adsorption could not be adopted (*Use of adsorbents for the removal of pollutants from wastewater*, 1996).

Peat moss ion exchange, like natural zeolites, is a cation removal mechanism. Cations, such as Ca and Mg, and heavy metals (Hg, Cd, Zn, Pb, Cu, Fe, Ni, Cr, and Ag) are assumed to be removed by this mechanism. Two main conditions have to be respected for high removal rates: maintenance of low flow rates and treatment of small volumes.

Chelation is the ionic or hydrogen bonding of a metal ion with two adjacent carboxylate ($-COO^-$) or hydroxylic (–OH) groups on aromatic structures. Some researchers debate this mechanism, arguing that no evidence by NMR (nuclear magnetic resonance) proves the occurrence of this type of binding with peat moss. Another group of researchers references demonstrated that metal ions binding to peat moss were in their hydrated forms. Thus, chelation cannot be the only mechanism of metal removal.

The role of humic acids for metal binding is inconclusive. Conflicting reports indicate in some cases humic acids, a natural component of peat moss, are contributing to metal removal; but

another study, using peat moss from which all humic acids were removed, concluded that the studied peat was more efficient for removal of Cu than the original peat moss (*Use of adsorbents for the removal of pollutants from wastewater*, 1996).

Electrostatical adsorption is defined as "attraction between the negatively charged surface and a positively charged metal ion" *(Use of adsorbents for the removal of pollutants from wastewater*, 1996). It is the third assumed method of metal removal by peat moss, nevertheless the exact mechanism is yet to be settled.

Kadlec and Hammer (1980) reported that ortho-phosphates are also adsorbed on peat moss. Therefore, special attention must be given to phosphate removal, especially when peat moss filtrate is fed to a biological reactor for further treatment, as is the case in the present experiment.

2.6.4 PEAT MOSS pH

The pH of peat moss itself will vary depending on the variety of used. Peat moss waters will generally be acidic, *i.e.* pH ranging from 4 to 7, because of the presence of carbon dioxide and humic acids (MarFarlane, 1969) in peat moss.

Despite an incomplete understanding of peat moss removal mechanisms, it is generally accepted that pH variation and peat moss adsorption capacity are directly related. At low pH, the adsorption process is irreversible and it is assumed that a strong type of adsorption is prevalent. At moderate to high pHs, other mechanisms than electrostatical attraction are involved. The pH of a solution is critical to the adsorption process: the percent of metal removed varies from almost negligible to almost complete removal within 4 to 5 pH units. The pH of the wastewater should be maintained above 3 to avoid metal leaching out of peat moss. The optimum adsorption range is from pH 3.5 to 6.5. At lower pH, there is lower humic or fulvic acid leaching observed; at higher pH, a brownish colour is observed leaching out of the filter. Above pH of 8.5, the peat moss structure itself is not stable.

Cameron (1978) used peat moss filtration as main method for metal removal from a landfill leachate. He found that increasing the pH of a landfill leachate from 4.8 to 8.4 resulted in a higher metal removal for Fe, Mn, Zn, K, Na, Ca, and Mg. He concluded that at pH 8.4, greater

removal was induced by higher solids precipitation of metal complexes. At pH above 8.4, precipitates caused serious filter clogging problems and experiments had to be terminated.

Cameron (1978) observed a preferential initial adsorption of zinc, followed by a desorption leading to a sharp rise of Zn effluent concentration. He reported a decrease of COD, P, and TKN at lower pH and a rapid excess of Fe and Mn beyond maximum acceptable concentrations for direct discharge of the effluent. Table 2.8 discloses Cameron's results for specific metals; filters contained 793 g of dry peat moss and had a flux of 70 L $h^{-1}m^{-2}$.

Table 2.8: Metals removal from a landfill leachate from Cameron (1978)

Parameter	Influent (mg L^{-1})	Maximum influent concentration (mg L^{-1})	Volume throughput to reach maximum concentration
Fe	26.8	0.30	< 2 L
Mn	0.52	0.05	< 2 L
Zn	0.43	0.50	2.5 L

Chaney and Hurderman (1979) found a reduction in the adsorption of specific ions with pH variation or presence of other ions. Strong chelating agents or salts, such as cyanide and chloride, reduce peat moss removal efficiency. Lee and Low (1996) established the following order of adsorption: Pb, Cu > Cd > Ni, Zn.

2.6.5 REGENERATION, ASH AND HUMIDITY CONTENT

Because peat moss is inexpensive and plentiful, it is generally accepted that regeneration of peat moss is not economically attractive, unless specific conditions apply. Peat moss is typically made up mainly of carbonaceous matter and of only 2% fixed solids (FS). When the FS fraction is higher than 2 percent, peat moss may be classified as organic clay or silt (MacFarlane, 1969).

The affinity of peat moss for water is one of its particular characteristics; peat moss has a great capacity of taking up and holding water like a sponge. Peat moss water content may vary form 100% to over 3000% (MacFarlane, 1969). In order to obtain a maximum surface contact between bulk liquid and pores, it is important to thoroughly wet the peat moss.

2.6.6 COMPACTION

Viraranghan and Rana (1987) reported that peat moss should not be highly compacted when used for filtration: at densities such greater than 150 kg m^{-3}, dry weight, major clogging problems could occur. The more compaction there is, either in the natural absorbent or in the filter set-up, the less surface area is available for adsorption, the lower will be the removal. On the other hand, good compaction is necessary to avoid channelling and to obtain a uniform flow. Therefore, in order to allow adequate percolation, a lightly compacted peat moss, of a density of 100 kg m^{-3}, is suggested to use in a filter. Field sphagnum peat moss has a bulk density of 80 to 100 kg m^{-3}, based on its dry weight.

2.6.7 CLOGGING PROBLEMS

Cameron (1978) has reported clogging problems and flow rate reduction with peat moss filters. Surface clogging by metal precipitation was assumed to be causing this reduction. To overcome the problem, Cameron applied a slight negative pressure on the filter to maintain a constant flux. Chevalier (1996) reported flux decreasing with time that could reach a point of total clogging. Maintaining a satisfactory hydraulic capacity requires periodical removal of fines and bacteria accumulated on the top of the filter by excavating the whole top part of the peat moss filter and replacing it with fresh peat moss.

2.6.8 APPLICATIONS OF PEAT MOSS IN POLLUTION CONTROL

Peat moss is used for its filtering and adsorption capacities in various applications in environmental engineering. Examples of such use of peat moss are (Cameron, 1978; Viraraghavan and Ayyaswami, 1987; Warith, 1999):

- Treatment of slaughterhouse wastewater, dairy and animal wastes effluent, septic tank effluent, landfill leachate and municipal wastewater,
- Oil and heavy metal removal from wastewaters,
- Pit latrines bedding,
- Daily-cover media in landfilling operation.

CHAPTER 3

MATERIALS AND METHODS

The apparatus, experimental setup, characteristics of materials used such as leachate, biomass and peat, as well as the test procedures and analyses methods performed are detailed in this chapter.

3.1 SETUP AND APPARATUS

3.1.1 SETUP

The setup of the UASB reactors used in this study is schematically shown in Figure 3.1. Three UASB reactors were run in parallel; a centrifugal pump recycled the feed leachate from a low temperature storage unit (4°C) to a main line to which all influent feed pumps were connected. The unused feed was recycled to the cold storage unit maintaining it at low temperature, therefore preventing biomass growth in the feed storage and keeping feed COD concentration constant over a running period.

Peristaltic feed pumps transferred leachate from the main line to each UASB reactor at a rate set to achieve the desired HRTs. Figure 3.1 shows the four sampling ports used in the setup; X_1 was the sampling port for the effluent (COD, SS, and pH), X_2 was the recycling line sampling port for VFAs, and X_3 was the gas sampling port for biogas composition; all three reactors had these sampling ports. Finally, X_4 was the feed leachate sampling port; one measurement was assumed to be valid to describe feed characteristic for all three reactors.

The reactors were maintained in a temperature-controlled room kept at the desired operation temperature (32°C). The room temperature was monitored by a mercury thermometer (0 to 100°C) in a 125 mL erlenmeyer flask filled with tap water; no temperature monitoring within reactors was made.

Biogas volume was measure using a pre-calibrated Wet Tip Meter, (Wet Tip Inc.); calibrated using a Fisher Scientific Lab-Crest flow meter (cat. no. 448-118) with a stainless steel float. Wet Tip Meters were recalibrated before each set of experiments. Gas flow rate was

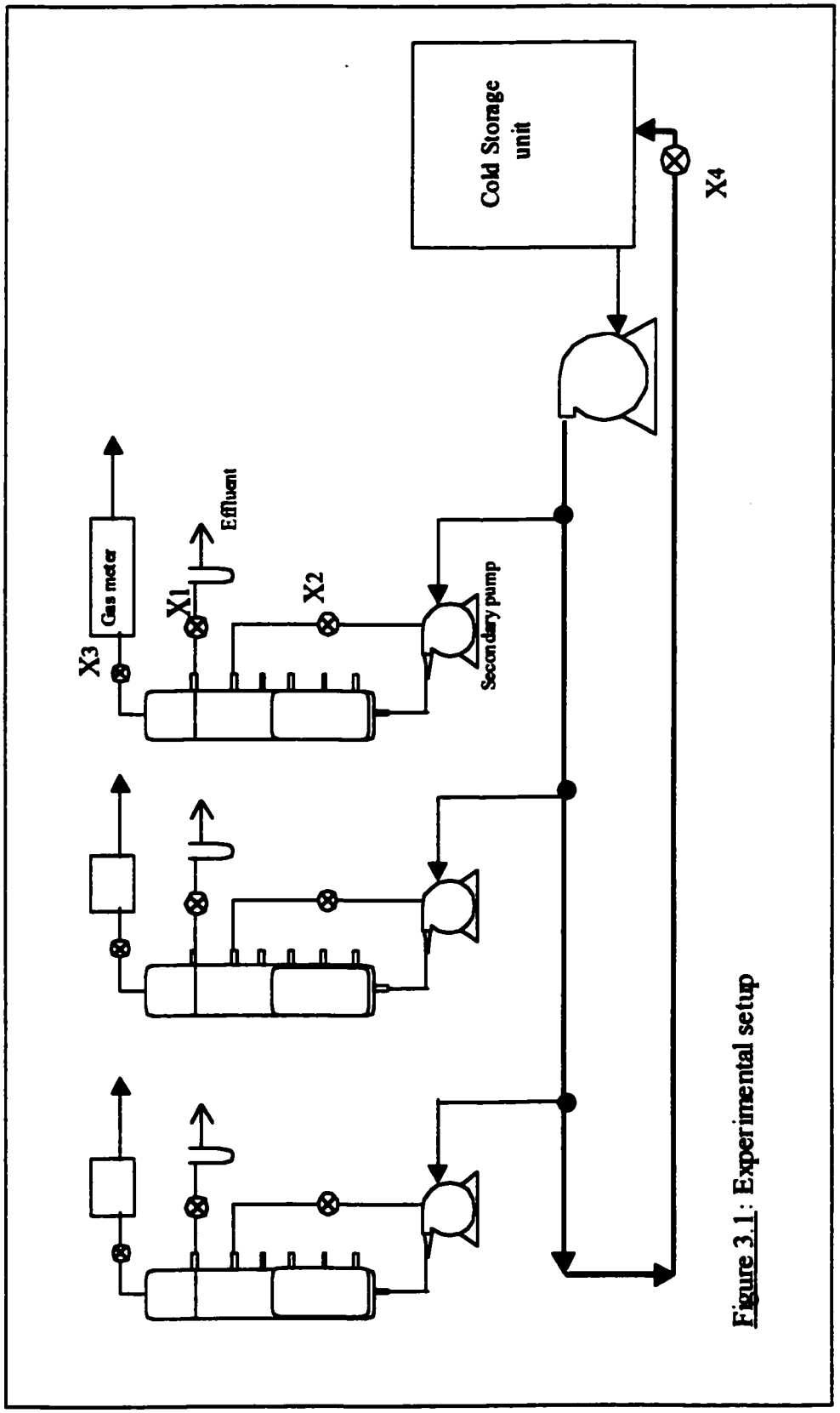


Figure 3.1: Experimental setup

calculated by dividing the measured gas volume by the measured time period of biogas production.

3.1.2 UASB REACTOR

All three UASB reactors were made of glass and had similar sizes and shapes (Figure 3.2). Each reactor had an approximate total volume of 7 L and 5.75 ± 0.5L of liquid capacity. Reactors were 13 cm in diameter and 60 cm in height. The influent inlet was a vertical connection leading to a T-shape at the bottom of the reactor whereas the outlet was a horizontal port connection at the top on the reactor, approximately at the 5.75 L volume mark. The vertical orientation of the inlet led to some clogging problems; calcified high-density biomass granules were falling into the inlet tubing and were forming clogs, despite the T-shape of the reactor entrance. It is strongly suggested, for subsequent experiments using the same reactor type, to fix the inlet clogging problems in order to improve reactor operations.

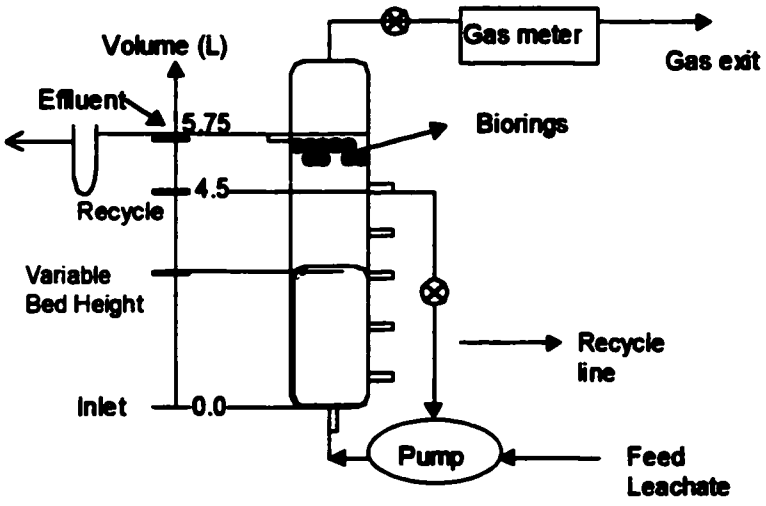


Figure 3.2: UASB reactors setup

On the effluent line, a U-shaped glass tube filled with effluent was used to seal the reactor from the surrounding atmosphere and to visually monitor the inner reactor pressure. This aqueous bridge prevented oxygen from entering and displacing the anaerobic atmosphere in the reactor. The water level difference between both sides of the u-tube provided a quick and easy way to verify if the pressure inside the reactors was always higher than the surrounding

atmospheric pressure. A positive pressure confirmed two important facts; 1- gas production was most probably taking place and 2- no gas leaks affected reactor junctions.

The reactor top was sealed with a glass lid comprising two or more openings to access inside reactors and a gas exit to allow biogas to progress to Wet Tip Meters. On top of the sludge blanket, 4 cm of plastic Koch™ bio-rings were placed to act as deflectors, preventing excessive biomass washout by detaching gas bubbles from granules when floating granules hit them.

3.1.3 PUMPING SYSTEM

Harvard Apparatus® double-head peristaltic pumps were used for feed and recycling lines. This type of pump allowed effluent and recycling lines to be pumped simultaneously and at the same rate. The recycle to feed ratio is calculated from the ratio of the square of the inner diameter of tubing used for each line, respectively. Recycling circuits withdrew liquid from the sludge blanket, below the effluent level. The tubing used for the recycling line had 0.794 cm diameter (5/16 in.) whereas the feed tubing diameter was 0.318 cm (1/8 inch); the recycling to feed ratio was therefore 6.25:1 or 25:4. The main centrifugal recycle pump was operated at a constant rate of approximately 200 L h^{-1} whereas secondary influent feed pumps were set to obtain the desired HRTs (0.95 to 3.8 d).

Tubing used for all lines in the setup, were made of clear Nalgene® or Tygon®. Clear tubing was useful in quickly identifying potential clogs and scaling problems.

3.1.4 PEAT FILTER

Peat filters were constructed from a one-meter high Plexiglas® container, having a square cross-section of 18 by 18 cm^2 (Figure 3.3). As disclosed by Figure 3.3, a board (made of Plexiglas®) angled at 45° was sitting in the container bottom to direct the flow toward the exit channel.

The bottom of the peat filter was first filled with 1 L of 1.2 cm glass marbles, followed by 2 L of 10% 20 mesh or coarser sand with an effective filtering diameter of 0.25 mm. The filter was then filled with peat; peat was weighed and pre-wetted prior to filling the filter in order to form a slurry and decrease dry peat flotation and dust problems. The peat slurry was poured inside the container and allowed to settle for 24 hours. Peat settling and compaction occurred naturally; no compression other than brief hand packing was made on the peat column. Successive additions and settling of slurry were made until 45 cm of packed peat had

accumulated. Finally, small stones were added on top of the peat to prevent cavity formation when pouring leachate onto the filter.

Raw leachate was poured directly onto the top of the peat column and peat filtered leachate exited the bottom of the container. Leachate filtration was driven by gravity; no pressure or vacuum was applied. The filter was operated in a batch mode; a liquid head varying from 30 to 2 cm of leachate (from beginning to end of a batch process) forced the liquid through the filter. Peat filters were not allowed to run dry because when they did, serious channelling problems occurred.

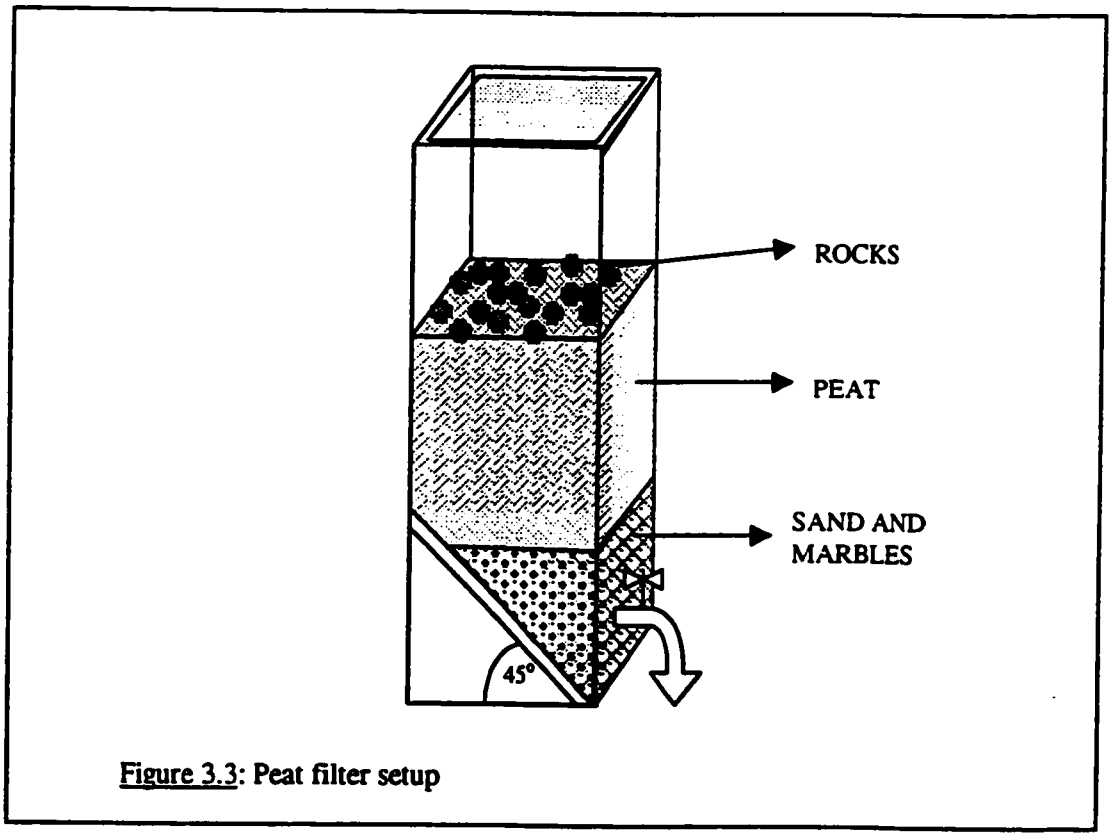


Figure 3.3: Peat filter setup

3.2 MATERIAL; LEACHATE, BIOMASS AND PEAT

3.2.1 CARP ROAD LANDFILL LEACHATE

Qualitatively, CRL leachate had a high strength; it had an orange to amber colour and a considerable amount of suspended fines and colloids, which made the liquid opaque. If a sample was left settling over five days, an orange powdery precipitate was observed whereas the bulk liquid became clear. CRL also had a strong malodorous smell and produced considerable foaming when stirred.

Table 3.1 details the raw leachate characteristics. It had a high total COD concentration (from 18 000 to 40 000 mg COD L^{-1}) and an average BOD_5 of 15 267 mg BOD L^{-1}. The pH was acidic (5.6 to 6.2) and the TSS concentration averaged 515 mg L^{-1}. The fixed suspended solids (FSS) fraction of total suspended solids (TSS) represents on average 67%; the main inorganic salts of the suspended solids (SS) were Ca, Mg, K, and Na, whereas main metals were Al, Fe, and Sr.

Nitrogen as TKN was present in sufficient concentrations (470 mg L^{-1}) according to the COD:N ratio required to sustain anaerobic microbial growth (500:5). However, phosphorus (P) as total P (TP) was low (2.2 mg L^{-1}); P addition had to be considered to improve bioreactor COD removal efficiency (Chapter 6). No pH or buffering capacity adjustment was necessary; the raw leachate has a high alkalinity of 5.16 g as $CaCO_3$ L^{-1}. Chlorides were high (2.82 g L^{-1}) and could be problematic for sewer discharge and peat adsorption of metals. Other potentially problematic elements were boron, iron, manganese, and zinc. Unexpectedly, barium had a low concentration in studied samples and did not exceed the municipal sewer discharge limit. Ba adsorption and removal were studied despite its low concentration.

Table 3.1: Carp Road Landfill leachate characteristics

Element	Detection limit	Average dissolved (mg L^{-1})	Average total (mg L^{-1})
Al	0.01	2.90	6.7
As	0.1	<0.1	<0.1
Ba	0.005	0.495	0.60
Be	0.005	<0.005	<0.005
Bi	0.05	<0.05	<0.05
B	0.01	2.9075	—
Cd	0.02	<0.02	<0.02
Ca	0.03	2260	2588
Cr	0.01	0.035	0.18
Co	0.01	0.1025	0.14
Cu	0.01	<0.01	0.03
Ga	0.05	<0.05	<0.05
Fe	0.02	7.26	307
Pb	0.1	<0.1	<0.1
Li	0.005	0.43625	0.52
Mg	0.01	331	428
Mn	0.01	10.745	16
Mo	0.02	<0.02	0.08
Ni	0.02	0.57	0.84
P	0.1	1.825	2.5
K	0.4	711.75	821
Si	0.05	3.475	—
Ag	0.02	<0.02	<0.02
Na	0.2	1358.75	1638
Sr	0.005	73.725	113
Sn	0.2	<0.2	—
Ti	0.01	<0.01	<0.01
V	0.005	<0.005	0.02
Y	0.005	<0.005	<0.005
Zn	0.01	4.45	6.0
Zr	0.01	0.0175	—
Cl	0.1	2820	—
Nitrate-N	0.1	<0.1	—
Nitrite-N	1	5	—
Sulphate	1	489	—
Alkalinity as $CaCO_3$	1	5160	—
TKN	0.05	470	—
TP	0.01	2.2	—
BOD_5	1	15267	—
COD	50	24000-44000	24000-45000
TOC	10	3980	—
TSS	1	—	515
FSS	1	—	345

3.2.2 GRANULAR SLUDGE

A granular sludge previously treating a pulp and paper wastewater from Lake Utopia Pulp and Paper in New Brunswick, Canada, was use to inoculate reactors. Granules were dark grey to black and had approximate diameters from 0.2 to 2.0 mm. No granulation process was used in the present experiments; only a short period of acclimation (less than two weeks) for microorganisms was necessary. It was deduced that the acclimation period was short because CRL leachate was essentially composed of volatile acids, which are readily degradable substrate. Granules used for inoculation had an approximate specific acetoclastic activity of 0.3 g COD g^{-1} VSS d^{-1}.

3.2.3 PEAT MOSS

Peat moss used in peat filters was a commercially available (in gardening stores) research quality control grade Canadian sphagnum peat moss, sold under trademark Premier®, available in 5 kg dry peat bags. The dry peat (moisture content was less than 5%) was medium brown and highly dusty, contained small wood pieces and tiny branches that were mostly removed by hand.

3.3 TESTS AND ANALYSES

3.3.1 TESTS AND ANALYSES ON INFLUENT AND EFFLUENT

3.3.1.a COD

COD tests were conducted as described in *Standard Methods* (1992): 5220 C. Closed Reflux, Titrimetric Method. Samples taken for soluble COD analysis were centrifuged for 10 minutes at 10 000 RFG (relative centrifugal force) prior to analysis instead of being filtered as described in *Standard Methods*. Digestion vessels used were 20 x 150 mm, therefore according to Table 5220:I of *Standard Methods*, 5.0 mL samples were analysed and corresponding reagents volumes were used. Ferrous ammonium sulphate (FAS) molarity was measured weekly and the FAS solution bottle was kept in the dark to prevent its degradation. All digestion solutions used were prepared as directed in *Standard Methods*.

3.3.1.b pH

Sample pHs were measured with a Fisher Accumet pH meter (model 320, expanded scale research pH meter). A standard VWR Scientific pH 7.00 buffer solution (yellow) was used to calibrate pH meter before each use. Samples were analysed directly without filtration or centrifugation.

3.3.1.c BOD, alkalinity, nutrients and mineral content

BOD, alkalinity, nutrients and mineral content of the raw leachate were analysed by a private laboratory: Seprotech, Laboratories, located at 2378 Hollylane, Ottawa, ON. Results are disclosed in Appendix C.

3.3.1.d Metals

Metals concentrations were measured by Dr. Nimal de Silva from Carleton University, by inductively coupled plasma (ICP). Calibration concentration and wavelengths used are given in Table 3.2.

Table 3.2: Calibrations concentration and wavelengths of ICP

Metal	Wavelengths (nanometres)	Calibration concentration (ppm)
Al	308.2	10
B	249.7	10
Ba	493.4	10
Ca	315.8	100
Co	228.6	10
Fe	259.9	10
Mn	257.6	10
Zn	213.8	10

3.3.1.e Suspended solids

Total and volatile SS were measured by filtering influent or effluent with glass-fibre filters according to *Standard Methods* (1992), Methods 2540D and 2540E. Residual solids were

dried in a 105°C oven, cooled in a dessicator for 24 hours, weighed for TSS, reduced to ashes in a 550°C furnace and weighed a second time for FSS.

Whatman glass microfibre filters (cat. no. 1822 070), 7.0 cm in diameter, were used. Volumes were measured with a 250 mL graduated cylinder. Samples were kept for 24 to 72 hours at 4°C until filtered; they were allowed to reach room temperature before being filtered. Filters were prepared as described in *Standard Methods*. They were pre-dried, put in an aluminium tare tray and kept in a dessicator until weighted for tare. Filtration apparatus was set on a vacuum pump, and solids attached to the filter support walls were washed out with distilled water and incorporated with the filter in the aluminium tare tray prior to drying of samples.

3.3.1.f Volatile fatty acids

VFAs were analysed according to the method of Ackman (1972). Procedures used to calibrate and analyse VFAs were described by C.M. Elliott (1989). VFAs concentrations were measured by a Hewlett Packard (HP) 5840A packed column gas chromatograph (GC), equipped with a flame ionisation detector. The GC column temperature was set at 180°C, and the injector temperature, at 400°C. GC apparatus was fitted with an auto-sampler (HP 7672A Automatic Sampler), and an integrator (5840A GC HP Terminal). The carrier gas was 99.9% pure helium and formic acid was use to suppress peak tailing.

A VFA analysis on samples from the sludge blanket, the effluent, and the recycling line sampling port revealed no significant differences between VFA concentrations measured in those three sampling locations. Therefore, it was assumed that sampling the VFA from the recycling line-sampling port (X2) was representative of the bulk liquid in the sludge blanket, and a trustworthy indication of the reactor operation. Consequently, all samples were taken from the sampling port of the recycling line.

Samples of 1.0 mL were centrifuged prior to analysis with an Eppendorf 5415 Centrifuge, for a minimum of 4 minutes at 7000 rpm. Samples, VFA mixture and internal standard volumes were measured with an Eppendorf 4700 micropipette (500 µL).

GC was calibrated with a standard solution of a VFA mixture and internal standard. VFA standard mixture was composed of three organic acids of exactly 0.2000% concentration (2000 mg L^{-1} acetic, 2000 mg L^{-1} propionic and 2000 mg L^{-1} butyric acids). The internal standard was a solution of exactly 0.2000% isobutyric acid (2000 mg L^{-1}). The detection limit of the VFA analysis is 3 mg L^{-1}.

3.3.2 TESTS AND ANALYSES ON BIOMASS

3.3.2.a Acetoclastic activity (AAT)

Acetoclastic activity tests were conducted as described in Kennedy (1999). 10 mL of anaerobic sludge was cored with a hollow Plexiglas® rod at the desired height in the sludge bed. Samples were transferred into a bottle with 40 mL of defined reduced medium, which was prepared as described in the reference. All bottles and pipettes were purged with nitrogen prior to use and sealed with a rubber septum and an aluminium cap. Bottles were place on a rotary shaker at 100 rpm and at the same ambient temperature as reactors for 24 hours. Samples were always tested in duplicate.

Tests were initiated by injecting samples with 100 mg of acetate from prepared acetate stock solution; 0.8 mL of defined medium was thereafter sampled at 90 or 120 minutes intervals. These samples were centrifuged and analysed for acetate concentration as described for VFA test in Section 3.3.1f.

3.3.2.b Suspended solids

Biomass solids content were sampled simultaneously with AAT samples and therefore by the same technique (Section 3.3.2a). An important source of error on biomass measurements was derived from the difficulty to accurately measure sludge with a pipette, even with a wide-mouth pipette. Volume measurements were therefore not as precise as desired. Solids content was measured as described in Section 3.3.1e.

3.3.3 TESTS AND ANALYSES ON BIOGAS

Biogas composition was measured by a Hewlett Packard 5710A gas chromatograph fitted with a thermal conductivity detector, on which the column temperature was set at 150°C and the injector temperature at 100°C. The carrier gas used was helium. Biogas samples were taken by a gas-tight syringe in the sampling port X4 as described in section 3.1.1 above. The samples were injected in the GC within seconds after sampling to avoid any possible alteration.

3.3.4 TESTS AND ANALYSES ON PEAT AND PEAT FILTER

3.3.4.a Total organic carbon

TOC was analysed by a Rosemount Dohrmann TOC Analyser DC-190, fitted with a Panasonic KX-P1150 Multi-mode printer. Soluble TOC was measured on samples that were filtered through glass micro-fibre filters (Whatman glass microfibre filters (cat. no. 1822 070)). A standard solution of KHP 100 mg C L^{-1} was used to calibrate the TOC analyser.

3.3.4.b Solids, metals, pH, nutrients, and COD

All these elements were tested and measured as described in Section 3.3.1 on influent and effluent measurements.

3.3.4.c Flux

Raw leachate was poured on top of the filter's until a 20 cm head was obtained. The experiment was initiated by opening the effluent tubing valve and the time required for the head to be reduced to 15 cm was measured. Volumes of leachate filtered and the peat column surface were measured and flux could be calculated from these two parameters.

When peat filters were not to be used for over 24 hours, raw leachate on top of the filter was removed to prevent accumulation of settled solids from raw leachate. If filters were left unused for less than 24 hours, then raw leachate on top of the filters was stirred to re-suspend solids that could have settled and to prevent cake formation on the filter's top.

CHAPTER 4

PRELIMINARY EXPERIMENTS: LEACHATE TREATABILITY WITH A UASB REACTOR

4.1 INTRODUCTION

Two main objectives were set for these preliminary experiments: evaluate the treatability of the CRL leachate with a UASB reactor and determine the minimal biomass concentration required for efficient treatment. To assess CRL treatability, the following operating parameters were optimized: COD removal, biogas production and biomass activity. To assess the minimal biomass concentration, specific loading rates versus fixed solids accumulation rates were analysed and the amount of biomass required was established. Finally, preliminary experiments allowed an overall observation of the treatment's nutrient requirements and possible toxic effects.

4.2 SPECIFIC MATERIAL AND METHODS

Three reactors, as described in Chapter 3 Section 3.1, were run in parallel at an ambient temperature of 25 ± 1°C. This moderate temperature was chosen in order to verify the anaerobic digestion potential efficiency at 25°C and to hopefully decrease the operating costs of a speculative full-scale application, if these operation conditions were found to be efficient at 25°C.

Constant flow rates were technically difficult to maintain because of pumping failure problems. When a reactor VFAs concentration was high, (i.e. when acetate concentration was greater than 800 mg L^{-1}, propionate, greater than 400 mg L^{-1} and butyrate, greater than 50 mg L^{-1}), the feed rate was decreased or stopped and the reactor was left on recycle until VFAs were low or undetectable. Data from days when such failures occur were not used for the efficiency averages.

Each reactor's gas volume, liquid flow rate, and VFA concentrations were monitored on a daily basis during weekdays. Biogas production was monitored on a daily basis by a Wet Tip Meter and its methane fraction was measured by GC. The pH and COD values were monitored

approximately every two days. Only soluble COD was measured during the course of preliminary experiments.

4.3 RESULTS AND DISCUSSION

UASB reactors were found to be an efficient means of treatment for CRL leachate. Standard operating conditions were defined and used to lay the foundations for Chapter 6 experiments. Excellent COD removal efficiencies, gas production and activity rates were observed. However, accumulation of inert solids was observed in reactor biomass and a decrease of activity ensued from biomass calcification. Based on the above parameters, the required biomass amount to adequately treat CRL leachate was determined.

4.3.1 OPERATING CONDITIONS

Table 4.1 describes reactors R1, R2, and R3 operating conditions: biomass fractions, sludge bed volumes, biomass concentrations, HRTs, OLRs, and pH. The average HRT and feed leachate COD concentration were 2.3 ± 0.1 d and 11.3 ± 1.7 g L^{-1} respectively. The average organic loading rate was 5.1 ± 0.2 g COD L^{-1} d^{-1}. Figure 4.1 shows the variation in COD concentration of experimental influent and effluents.

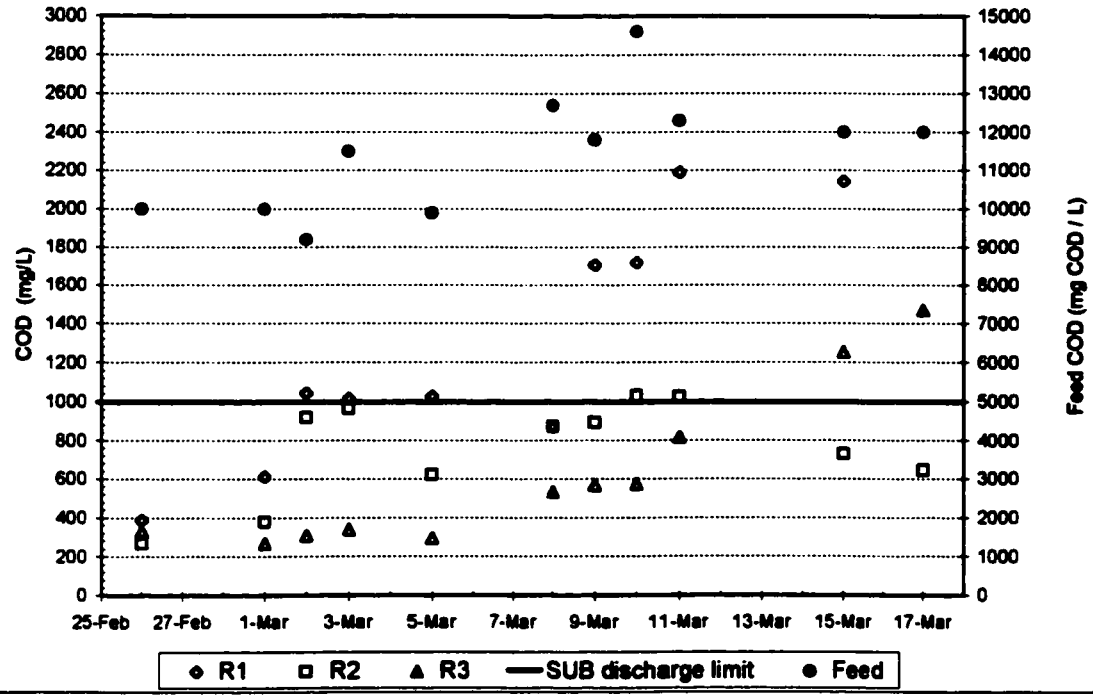
The three reactors were initially filled with different amounts of biomass; R1 was filled up to one-third of the reactor sludge blanket volume, R2 to one half and R3, two thirds. Measured bed volumes averaged 1.9 L, 3.0 L and 3.7 L for R1, R2, and R3 respectively. Sludge bed volumes remained fairly constant throughout the experiment despite a significant variation in the solids concentration. Solids concentration variations are discussed in detail in Section 4.3.5. The pH remained constant at about 7.3 in all reactors, indicating the good stability of the reactors. The high stability of the pH was a positive side effect of the high alkalinity of the CRL leachate; it was providing a high buffering capacity preventing drastic pH variations.

To initially prevent a shock load on reactors, feed leachate was diluted to approximately 3000 mg COD L^{-1} and increased slowly to 6000 mg COD L^{-1} over a period of 10 days (Feb. 14 to 23). Since VFAs remained low and reactors' functioning was stable, feed concentration was increased to the target value of 10 000 mg COD L^{-1} over 60 hours (Feb. 23 to 25). HRTs were kept constant and no significant VFA increase was noted during this acclimation period. The

Table 4.1: Reactor operational conditions for R1, R2 and R3

	Targeted sludge bed fraction	Observed sludge bed average volume	Reactor final biomass concentration (average)	HRT (d)	average OLR ($g\ COD\ L_r^{-1}\ d^{-1}$)	average pH
R1	1 / 3	1.9 L	10 g VSS / L	2.4	4.9	7.4 ! 0.2
R2	1 / 2	3.0 L	18 g VSS / L	2.2	5.3	7.3 ! 0.2
R3	2 / 3	3.7 L	23 g VSS / L	2.3	5.1	7.4 ! 0.2

Figure 4.1: Feed and effluents COD variation for R1, R2 and R3



experiment was then started on February 26th for a period of 20 days and a pseudo-steady state was maintained for a period of eight times the HRT.

Values of the feed and effluents COD concentration are shown on Figure 4.1; the average feed COD was 11 300 mg COD L^{-1} from February 26th to March 17th. There was an unexpected, important variation in the leachate COD concentration on March 10th: the feed COD concentration unexpectedly jumped to 14 600 mg COD L^{-1} (30% variation). It was concluded that such a sudden variation was uncontrolled and caused by inappropriate storage and stirring methods of raw CRL leachate.

The leachate was stored in 200 L barrels, stirred prior to use and collected in small volumes of 20L. However, the stirring was ineffective in re-suspending the settled colloidal and suspended matter from the bottom of the barrel; a concentration gradient was most probably created from top to bottom. From March 10th, a change in the mixing technique was made to prevent this situation from re-occurring. New feed barrels were sampled and characterized at least two or more times for COD content in order to obtain a representative measurement of organics concentration.

4.3.2 COD REMOVAL

Two types of information were extracted from reactors' COD removal efficiencies data: the degree of biodegradability of CRL leachate and the capacity of each reactor to decrease effluent COD below the SUB discharge limit of 1000 mg L^{-1} (Figure 4.1).

The degree of biodegradability was calculated from the maximum COD removal achieved by any of the reactors. R3 removed up to 97% of COD in the first two weeks of the run; it was therefore concluded that CRL leachate was highly degradable by anaerobic digestion, up to 97% at a HRT of 2.3 d corresponding to a volumetric organic loading of 5.2 g COD L^{-1} d^{-1}.

The capacity of UASB reactors to treat CRL leachate to COD levels meeting the SUB discharge limit was based on the overall individual performance. All runs achieved relatively high COD removal: 89% for R1, 94% for R2, and 95% for R3 (average values). However, no reactor achieved full compliance with the SUB discharge limit: they all at one point or another exceeded the limit (Figure 4.1).

R1 exceeded the limit within four days after the run started. R2 and R3 met the SUB limit most of the time; R2 exceeded the limit when the March 10th sudden feed COD increases

occurred but recovered quickly afterward, and R3 exceeded it when experiencing severe pumping and recycling problems from March 14th until the end of the experiment. Therefore, without operational problems, R2 and R3 would most probably have achieved compliance with SUB COD limit.

Results for COD removal, OLR and VRR are shown in Table 4.2: R2 and R3 COD removal performance were similar (94% and 95%, respectively) even though R3 had a higher biomass concentration. When compared to R2, it was concluded that R3 excess biomass volume compared to R2 was not essential. R1 had the lowest COD removal efficiency (an average of 89% removal), was not showing recovery signs and it was concluded that it had an insufficient biomass inventory to achieve the desired removal.

Table 4.2: Average values of OLR and removal for R1 to R3.

	Targeted sludge bed fraction	COD removal % COD	OLR g COD L_r^{-1} d^{-1}	VRR g COD_{rem} L_r^{-1} d^{-1}
R1	1/3	89	5.03 ± 0.50	4.5
R2	1/2	94	5.53 ± 0.41	5.2
R3	2/3	95	5.28 ± 0.65	5.0

Resulting OLRs ranged from 4.9 to 5.3 g COD L_r^{-1} d^{-1}. These OLRs were higher than OLRs reported by Kettunen and Rintala (1998) on 19-23°C UASB reactors (3.3 g COD L_r^{-1} d^{-1}), but lower than OLRs reported by Myburg and Britz (1993) fed on 35°C UASB-AF reactors (21.65 to 29.0 g COD L_r^{-1} d^{-1}) (Table 2.2). The former had a lower COD removal (57-60%) whereas the latter had a similar COD removal range (84.5-95%) to the present study.

It was therefore concluded that an OLR of 5.5 g COD L_r^{-1} d^{-1} was a suitable volumetric loading rate upper limit for feeding raw CRL leachate to an UASB reactor having R2 biomass concentration. However in this case, specific loading rates are more useful and practical to set standard parameters especially in experiments involving reactors of similar volumes but different biomass amounts; COD fed per g VSS per day results are discussed in Section 4.3.4b.

4.3.3 BIOGAS PRODUCTION

4.3.3.a Gas production and methane fraction

High biogas production was observed; all runs yielded a production close to the theoretical methane production at 25°C, 0.38 L CH_4 g^{-1} COD_{rem}, within a divergence of ± 0.01.

The efficiency of each reactor to produce biogas from the feed leachate and methane fractions are shown in Table 4.3.

Methane fractions in biogas were satisfactory in all runs: ranging from 73-77% in these experiments whereas methane fractions reported in literature (Kennedy, 1999) were ranging from 60 to 80%. The resulting methane production per COD removed was ranging from 0.37 to 0.38 L $_{CH4}$ g^{-1} COD_{rem} which was excellent compared to methane productions of 0.20 to 0.30 L $_{CH4}$ g^{-1} COD_{rem} published by Britz and co-workers (1990). This production was indicative of good operational procedures, healthy methanogenic activity and also a low toxicity of the CRL leachate on methanogens.

R2 yielded the highest biogas production with 2.4 L_g L_r^{-1} d^{-1}. However, this reactor had also the highest OLR and therefore, the largest amount of COD to remove. R1 and R3 had satisfactory biogas production of 2.0 and 2.1 L_g L_r^{-1} d^{-1}. It was concluded that R3 was possibly not at its maximum biogas production rate (higher organic loading was possible) whereas R1 was overfed.

Table 4.3: Average values of biogas production for R1 to R3

	Biogas production (L_g L_r^{-1} d^{-1})	Methane fraction of biogas %	CH_4 production (L_m L_r^{-1} d^{-1})	CH_4 produced per COD removed L $_{CH4}$ g^{-1} COD_{rem}
R1	2.0	73	1.60	0.37
R2	2.4	74	2.01	0.38
R3 (until March 5th)	2.1	77	1.57	0.37

From March 8th, R3 methane production had doubled compared to the March 5th value; it was producing twice the theoretical amount calculated with equation 2.3.1. The gas meter was found defective; it was doubling the gas volume reading. Similar problems have occurred in the past (unpublished results). Because of this defect, R3 gas production results were used for averages only over the days on which the meter was functioning correctly (until March 5th). Prior to all subsequent experiments, Wet Tip Meters were cleaned and re-calibrated and worn parts replaced to prevent this situation from occurring again.

4.3.3.b Collected methane fraction

The collected methane fraction was calculated with equation 2.5.2 and the biogas production per kg COD removed was calculated from the biogas volumetric production (L L_r^{-1} d^{-1}) and the VRR (g COD L_r^{-1} d^{-1}). The methane partial pressure was calculated with the

measured methane fraction and by assuming a total pressure inside the reactor of 1 atm and using a value of 298°K for the temperature T. Calculations results are shown in Table 4.4.

Collected methane fractions (f_m) of 0.89 to 0.97 were found and concluded plausible considering effluent dissolved methane, possible gas lost through tubing and joints and reading errors on gas meters. The average value of calculated f_m, 0.92, was used as a constant for the subsequent experiments.

Table 4.4: Calculated collected methane fraction

	Biogas volume	Measured methane fraction	VRR	Calculated f_m
	$L\ L_r^{-1}\ d^{-1}$	%	$g\ COD_{rem}\ L_r^{-1}\ d^{-1}$	
R1	2.13 ± 0.16	73.0 ± 0.9	4.5	0.91
R2	2.62 ±0.31	73.5 ± 0.6	5.2	0.97
R3*	1.87 ± 0.25	74.7 ± 0.9	4.1	0.89
Averages	2.2	74	4.6	0.92

* R3 gas production and VRR from February 26th to March 5th

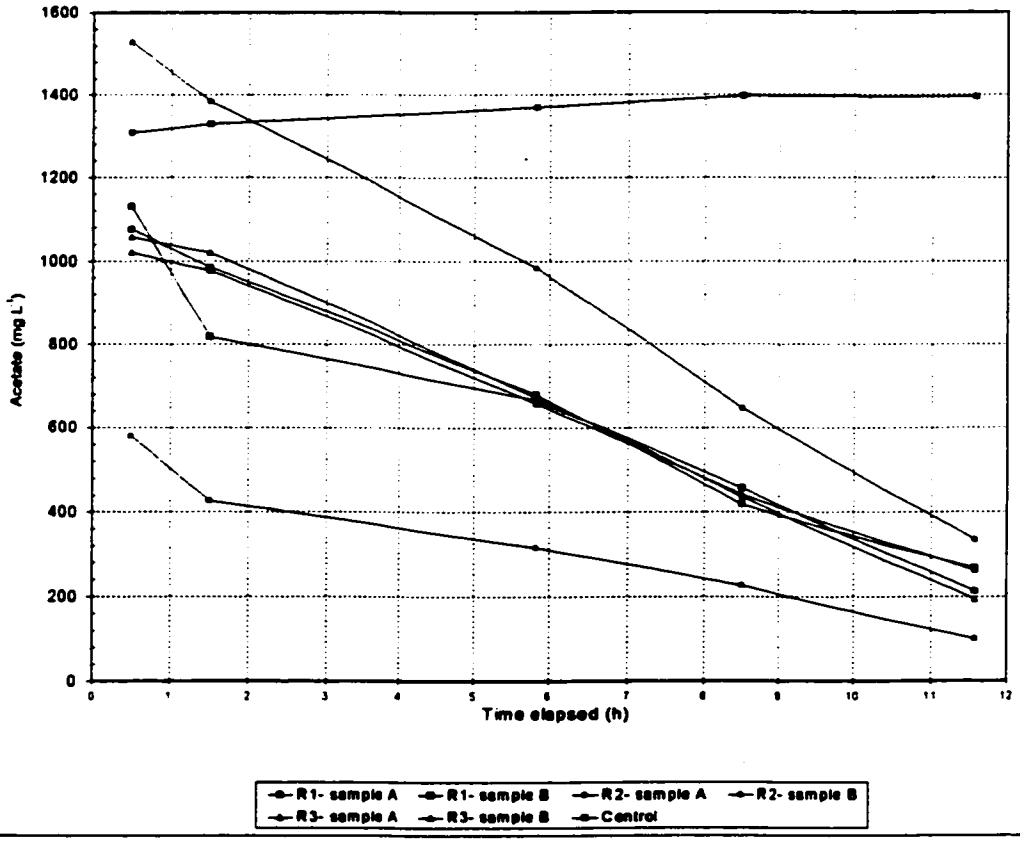
4.3.4 BIOMASS ACTIVITY

The biomass activity was measured by two means: an acetoclastic activity test on reactor's biomass samples was performed and specific removal rates (SRR) of reactors were calculated. Reactor specific removal rates provided excellent indications of reactors performance whereas AAT failed to reveal any pertinent information. Despite the poor quality of results obtained, a discussion of AAT results was deemed to be important to understand what led to this outcome.

4.3.4.a Acetoclastic activity test

AAT of preliminary experiment were conducted on duplicate samples originating solely from biomass harvested from the bottom of the reactor sludge bed. Higher activities were expected from bottom samples; because of their proximity to the influent entrance, thus their exposure to higher substrate concentrations. However, these samples were found to be unrepresentative of the whole reactor biomass. In order to obtain an accurate representation of the sludge bed activity, samples from different parts of the bed were tested in subsequent AAT experiments.

Figure 4.2: Acetoclastic activity test



In these AATs, initial acetate concentrations were observed to be not sufficiently high to induce the maximum consumption rate. R2 sample A (Figure 4.2) was accidentally fed a higher initial acetate concentration and yielded a greater activity rate compared to other samples with lower initial acetate concentrations. Thus, it was concluded that higher activities could have been achieved at higher initial acetate concentration for all samples. R1 and R3 initial acetate concentrations were roughly ranging from 1000 to 1120 mg L^{-1} whereas the mesophilic half-velocity constant (K_s) for acetate was reported to range from 11 to 421 mg COD L^{-1} (Pavlostathis and Giraldo-Gomez, 1991). Therefore, a reasonable initial acetate concentration for AATs should yield a curve with all data points above 800 mg L^{-1}; so as to obtain activity measurements independent of substrate concentration.

Moreover, samples were transferred to test bottles and directly tested without an acclimation period; therefore propionate was present in samples in significant concentration (95 ± 10 mg L^{-1} for R1 and 63 ! 2 mg L^{-1} for R3) to interfere with the acetoclastic activity rate.

Because propionate was competitively consumed by microorganisms, results were concluded to not reflect actual operating reactor maximum specific activities.

Table 4.5: ATT and reactor operation specific activity results

	AAT results g acetate g^{-1} VSS d^{-1}	AAT results g COD g^{-1} VSS d^{-1}	SLR* g COD g^{-1} VSS d^{-1}	SRR* g COD g^{-1} VSS d^{-1}
R1	0.26	0.27	0.51	0.42
R2-A	0.28	0.30	0.32	0.30
R3	0.20	0.21	0.23	0.22

* based on reactor operational data

Table 4.5 discloses AAT results: all AAT specific consumption rates were lower than SRR of the corresponding operational reactor. This observation reinforces the above assumptions on AAT activities being neither representative of the actual activity nor at their maximum values. R2 sample A had the highest acetoclastic activity and the closest acetate (as COD) consumption rate from the specific removal rates: it had the highest initial acetate concentration and was therefore closer to the maximum and actual reactor activity.

Objectives of subsequent AATs were to obtain greater acetate consumption rates (as COD) compared to respective reactor SRR for all biomass samples and to obtain representative results of corresponding reactors biomass activity. A greater consumption rate in AAT versus in reactors was expected because acetate used as sole substrate should be consumed faster than a mixture of soluble or less-soluble organic substrates such as found in leachate.

4.3.4.b Specific removal rate

Specific removal rates in Table 4.5 were calculated with average sludge bed VSS concentrations (average initial and final VSS concentrations from solid tests multiplied by average bed heights). Specific removal efficiencies were low (0.22 to 0.42 g COD g^{-1} VSS L^{-1}) compared to the literature. For mesophilic conditions activities the range for complex wastewaters is reported in the range of 0.5-1.0 g^{-1} COD g^{-1} VSS d^{-1} (Kennedy, 1999). SRR results would have been expected to be higher because of the large COD fraction of the raw CRL leachate, essentially composed of VFAs (close to 50%). It is possible that other components in the leachate have some inhibitory effect on the rate at which organics are removed.

R1, which had the least amount of biomass (9 g L^{-1}), and the highest HRT (2.4 d) had a peak SLR up to 0.68 g COD g^{-1} VSS d^{-1}, while an average SRR of 0.42 g COD g^{-1} VSS d^{-1} was obtained. R1 effluent COD was over the SUB discharge limit of 1000 mg COD L^{-1} and its

propionic acid concentration sporadically exceeded 500 mg L^{-1}. It was again concluded that R1 biomass was significantly overloaded (about 25% higher than its capacity) and no efficient removal could be expected at SLR as high as 0.5 g COD g^{-1} VSS d^{-1} when SRR capacity was around 0.4 g COD g^{-1} VSS d^{-1}. Secondly, it was concluded that the probable maximum SRR for UASB reactor digesting CRL leachate was 0.42 g COD g^{-1} VSS d^{-1} based on the average SRR achieved by R1.

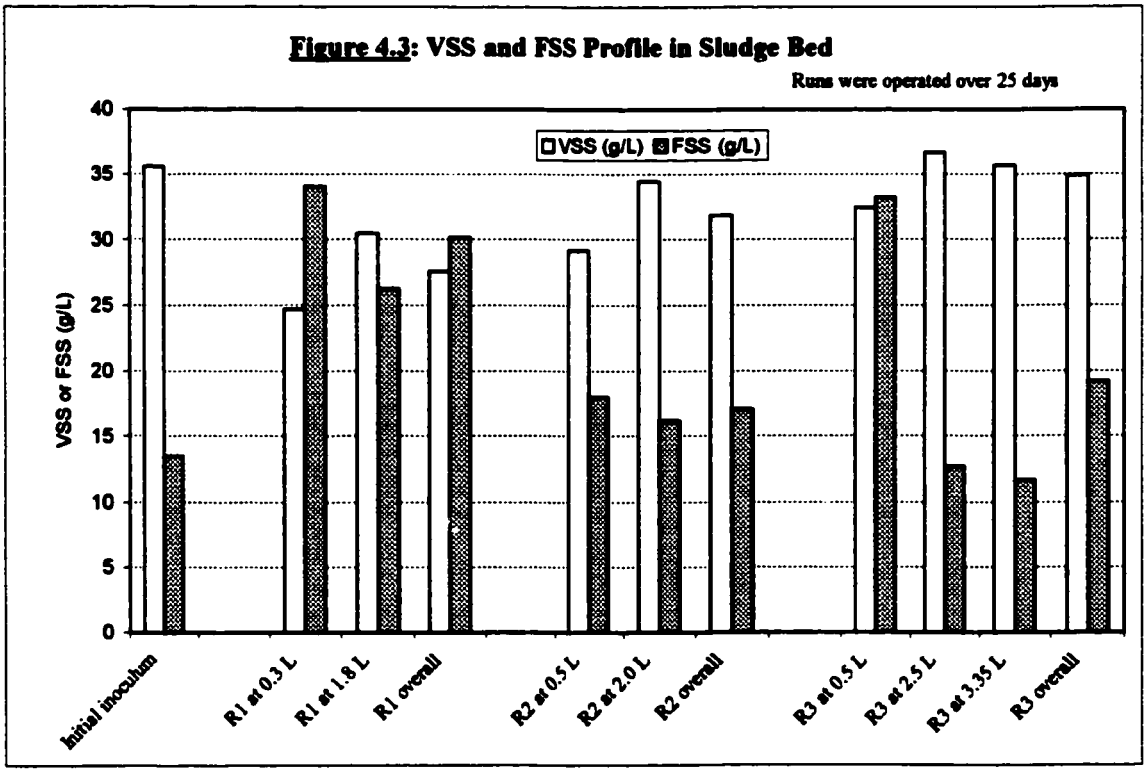
R2, had almost twice R1 biomass inventory (17 g L^{-1}), an average HRT of 2.1 d and an average effluent COD of 653 mg COD L^{-1}. The examination of the SRR (0.30 g COD g^{-1} VSS d^{-1}) versus the SLR (0.32 g COD g^{-1} VSS d^{-1}) led to the conclusions that R2 was slightly overfed; however running under a slight stress was an excellent means of optimizing the biomass activity. R2 effluent experienced COD concentration peaks corresponding to influent COD concentration increases but had steadily recovered and produced an effluent compliant with the SUB discharge limit. Individual VFAs concentrations remained below 400 mg L^{-1}, even while experiencing a shock load.

R3 had the largest biomass inventory (23 g L^{-1}), an average HRT of 2.2 d and achieved the best COD removal performance. R3 effluent COD concentration was high only during a short period (March 15th to 19th) because of pump failure problems; otherwise VFAs concentrations were stable at low levels prior to March 15th indicating normal functioning of the anaerobic digestion. The maximum SLR just slightly exceeded 0.3 g COD g^{-1} VSS d^{-1}, but the average SLR was 0.23 g COD g^{-1} VSS d^{-1} while the average SRR was 0.22 g COD g^{-1} VSS d^{-1}. R3 reactor was concluded not to be stressed and was being fed below its COD removal capacity. R3, that had a SRR that was even closer to its SLR (0.22 and 0.23 g COD g^{-1} VSS d^{-1}, respectively) than R2, achieved a moderate SRR and it was concluded that less activity stimulation was occurring compared to R2.

It was concluded that CRL leachate had the capacity of stimulating anaerobic bacterial microbial activity. It was also concluded that further acclimation and increase in removal efficiency were possible. A preferable operating SLR range was set to between 0.2 to 0.42 g COD g^{-1} VSS d^{-1}, most preferably around 0.32 g COD g^{-1} VSS d^{-1}. The maximum SLR to be fed to reactors was set at 0.42 g COD g^{-1} VSS d^{-1}; therefore, the minimum biomass amount required for a given removal efficiency on CRL could be calculated from OLRs and reactors volumes.

Table 4.6: Solids concentration variation in sludge beds for a running period of 25 days

	Average total biomass	g TSS L^{-1}	g VSS L^{-1}	% VSS
Initial inoculum		49	36	73
R1 (March 20)	59 g VSS	58	28	48
R2 (March 20)	101 g VSS	49	31	65
R3 (March 20)	131 g VSS	54	35	65



4.3.5 SOLIDS ACCUMULATION

There was a substantial accumulation of inert solids in all reactors after running preliminary experiments. Table 4.6 discloses the solid content of biomass samples from two or three different locations in reactor sludge beds. There was an increase of TSS concentration in all reactors, excepted for R2 where it remained stable. Unfortunately, the TSS increase was not attributed to new biomass growth but to an increase of FSS content in granules, especially in bottom granules where an overall biomass displacement by inert solids occurred.

Solid profiles on Figure 4.3 show a clear accumulation of FSS in all reactor bottom samples. R1 bottom sample (at 0.3 L) FSS final fraction was even more significant than VSS final fraction (58% FSS); R2 and R3 had FSS final fractions of 38% and 51% respectively versus an initial FSS fraction of 27%. Overall, there was a decrease of VSS fractions from 73% in the inoculum to 48% in R1 and to 65% in both R2 and R3. However, R2 and R3 top samples VSS fractions remained high; 68% for R2, and 75% for R3.

Inert solid accumulation rates were approximately 1.4, 0.5, and 0.96 g FSS d^{-1} for R1, R2, and R3 respectively. At a rate of 1.4 g FSS d^{-1} and starting from an initial inoculum of 49 g TSS L^{-1} (73% VSS), hypothetically assuming the worst case scenario of no biomass growth and no FSS washout, R1 sludge bed would have been completely turned into FSS within 27 days. Because there was FSS washout and VSS growth, the calcification process was actually slower but a complete calcification of the sludge bed was expected in a short-term. Approximated rates of g VSS and g FSS accumulation versus kg COD fed are given in Table 4.7. All VSS accumulation rates were negative whereas the FSS accumulation was dramatically high.

Table 4.7: Accumulation rates of biomass and inert solids in R1-R3

Run	Biomass accumulation g VSS kg^{-1} COD_{fed} d^{-1}	Inert solids accumulation g FSS kg^{-1} COD_{fed} d^{-1}
R1	-0.13	0.27
R2	-0.09	0.09
R3	-0.02	0.17

It was concluded that CRL leachate was significantly inducing inert solids accumulation and that CRL pre-treatment was mandatory to remove suspended and precipitating inert solids if a UASB reactor was kept as the preferred treatment. Because high COD removals were obtained and confidence in UASB reactor to achieve compliance with SUB COD discharge limit was

maintained, it was decided to keep UASB as the favoured treatment and use a pre-treatment to overcome the inert solid accumulation problem.

4.3.6 REQUIRED BIOMASS AMOUNT

The amounts of biomass in each reactor are tabulated in Table 4.1: R1, R2, and R3 were respectively filled to the 1.8, 3.0, and 3.7 L volumetric lines; their representative biomass fractions were $^1/_3$, $^1/_2$, and $^2/_3$ respectively.

From an operational point of view, R3 was found inefficient to adequately treat CRL leachate; because the sludge bed top was too close from the recycling line inlet and significant amounts of biomass granules were dragged in the recycling line resulting in major clogging of recycling line and pumps failure. Thus, the maximum suggested sludge level for operational reasons is 3.5 L, or sufficiently far from the recycle inlet to avoid excessive biomass transport into the recycle line.

On the treatment efficiency point-of-view, R1 failed to comply with the SUB discharge limit whereas R2 yielded satisfactory treatment results and achieved compliance with the limit. The required biomass concentration could therefore be calculated from the feed concentration (g COD L^{-1}), the desired flow rate (L d^{-1}) and the optimum SLR found in Section 4.3.4b: calculations with 0.21 g COD g^{-1} VSS d^{-1} SLR leaves a safety factor of 2 according to a maximum measured SRR of 0.42 g COD g^{-1} VSS d^{-1}.

4.3.7 NUTRIENTS REQUIREMENTS

No evidence of nutrients deficiency was noticed; removal rates and biogas productions were high. N and P were already present in raw CRL leachate; N was present in sufficient quantities in the raw leachate (470 mg TKN L^{-1}) however P was low (2.2 mg L^{-1} as total P). It was decided not to proceed to P addition in order to test the actual treatability of the raw leachate with as few additives as possible. Reactors should be pushed to their maximum efficiency before P addition would be considered.

4.3.8 TOXIC EFFECTS

The treatment of the CRL leachate did not reveal signs of strong toxic effects on the anaerobic flora. SRRs were lower than expected from literature results but anaerobic digestion

was working well, since gas production and high removal of COD were observed. And because COD removal rates were high, it was concluded that if a toxic substance had been present in the feed, removal efficiencies would have been lower. CRL leachate was concluded non-toxic to anaerobic digestion in the range of studied loading rates and concentrations.

4.4 CONCLUSIONS AND DIRECTIONS FOR SUBSEQUENT EXPERIMENTS

CRL leachate was concluded to be highly degradable, up to possibly 97% by UASB treatment, over a HRT 2.3 d. UASB reactor was an adequate means of COD removal, no signs of toxicity or nutrients deficiency to the anaerobic flora were observed. However, the serious weakness of UASB treatment was its susceptibility to inert solids accumulation causing biomass displacement. Therefore, there was a significant need for a pre-treatment of CRL leachate in order to remove suspended inert solids from the influent and hopefully prevent inert solids from accumulating in reactors. The suggested pre-treatment was a filtration by a peat moss bed. Peat moss is a low-cost, well-known filtration media and possesses the important advantage to potentially adsorb soluble inorganic pollutants from landfill leachates.

Satisfactory biogas production was measured: 2.0-2.4 $L_g\ L_r^{-1}\ d^{-1}$, with a methane fraction averaging 73-77%. In the subsequent experiments, it was suggested to use higher temperatures in order to hopefully increase biomass activity. Calculations of required biomass amounts were concluded to be appropriate from feed leachate COD concentration, reactor flow rate, and volume and the specific loading rate of R2, 0.21 g COD g VSS d^{-1}.

Finally, in subsequent AATs, initial acetate concentration should be sufficiently high to allow measured rates to be independent of substrate concentration; thus higher than 800 mg L^{-1} throughout the testing period.

CHAPTER 5

PEAT FILTRATION

5.1 INTRODUCTION

The objective of the peat filtration pre-treatment was the removal of TSS in CRL leachate prior to feeding it to UASB reactors. The TSS concentration in CRL leachate ranged from 500-1100 mg L^{-1} and TSS content had an average FSS fraction of 67%. CRL leachate TSS were highly non-biodegradable, accumulating in UASB reactors, and most probably inducing biomass calcification and decrease of specific microbial activity. Table 5.1 indicates the main inorganic salts and metals in the CRL leachate and average concentrations. Specific material and methods relating to peat filtering experiments are detailed Section 5.2. Results of preliminary peat filter studies are present Section 5.3. TSS removal and flux variation versus filtered volumes are discussed in Section 5.4. In parallel with TSS filtration, the adsorption of inorganic salts and metals by peat filter was studied. Removal results of selected elements are detailed in the Section 5.5.

Table 5.1: Concentration of selected solids in the CRL leachate

Element	Total solids (mg L^{-1})	Soluble solids (mg L^{-1})	Suspended solids (mg L^{-1})
Al	6.7	2.9	3.8
Ca	2588	2260	328
Fe	307	7.3	300
K	821	712	109
Na	1638	1359	279
Mg	428	331	97
Sr	113	74	39

5.2 SPECIFIC MATERIAL AND METHODS

5.2.1 OPERATIONAL CONDITIONS

Table 5.2 details the main characteristics of the peat filters used in this experiment. Peat filters were bench-scale size (see Section 3.1.4 for filter dimensions and setup) and were run at

ambient temperature (20–23°C). In preliminary experiments only, the filters were filled with dry peat and thereafter saturated with tap water. Tap water was slowly introduced in to the filters by pumping it in through the outlet tubing, at the filter's bottom. By introducing water through the bottom, peat was slowly wetted and air pockets between peat fragments was reduced. This method diminished peat flotation problems and accelerated peat wetting, however, preparing peat slurry as detailed in section 3.1.4 was found to be a more efficient filling technique.

Table 5.2: Peat filter characteristics

Parameter	Experimental values	
Amount of water in filter	5.65 L	5.64 kg
Amount of peat in filter		1.245 kg
Total weight of wet peat		6.895 kg
water/ kg peat	4.54 L_{water} kg_{peat}^{-1}	4.53 kg_{water} kg_{peat}^{-1}
% water over total filter	40% (v/v)	82 % (w/w)
Total volume of peat	14.3 L	0.0143 m^3
Compaction: kg peat per m^3	87.1 kg_{peat} m^{-3} (dry weight)	482 kg_{peat} m^{-3} (wet weight)

5.2.2 COMPACTION

Filters were not mechanically compacted; natural gravitational settling of wet peat yielded a compaction of 482 kg_{peat} m^{-3} (wet weight) or 87 kg_{peat} m^{-3} (dry weight), which was within the range of natural sphagnum peat compaction (80 to 100 kg m^{-3}, dry weight, from Viraranghan and Rana, 1987). Filters were over-saturated with water (water content represented 82% by weight of the whole filtration media) and water content was within the natural water absorption range of natural peat (over 450% of peat dry weight). It was concluded from preliminary experiments that if excess water was removed (passing from an over-saturation to a saturation condition), risks of channelling and concomitant problems increased. Consequently, peat filters were maintained over-saturated, even when not operated, to prevent air pockets and channelling.

5.2.3 REPLACEMENT VOLUME

The replacement volume was a concept introduced to facilitate measurements of peat capacity to filter or adsorb studied elements detected in CRL wastewater. The measured saturation volume of water in filters, 5.65 L, was set as the calculation base for replacement volume, V_r. A V_r was defined as 5.65 L of filtered liquid entering the filter, pushing out the liquid

contained in the filter, in a plug flow manner. This assumption infers that dilution and back mixing in the filter are negligible.

5.3 PRELIMINARY RESULTS

The objective of preliminary peat filter studies was to qualitatively determine the filter capacity for removing TSS from CRL leachate. No other measurements, other than COD variation from influent to effluent, were made. Raw leachate was filtered up to seven V_r, and the test was terminated.

A rapid and qualitative visual observation of filtered leachate was conclusive: that peat filtration was an excellent method of TSS removal from CRL leachate. Raw CRL leachate was opaque and had a rusty colour whereas filtered leachate was clear with a slight yellow tint. The filtered leachate remained clear from the first filtered fraction up to the end of the preliminary experiment.

COD tests on raw and filtered leachates were made and results are presented in Appendix B. There was an apparent high removal of COD by the peat for the first two V_r; however, the value of the adsorption result is inconclusive. The first V_r being most probably water initially contained in the filter and pushed out by leachate and the second one, a mixture of water remaining and some adsorption, no pertinent information could be deduced from those samples. After three V_r, the effluent COD concentration was about 50% of the influent COD concentration and after four V_r, no COD removal was observed.

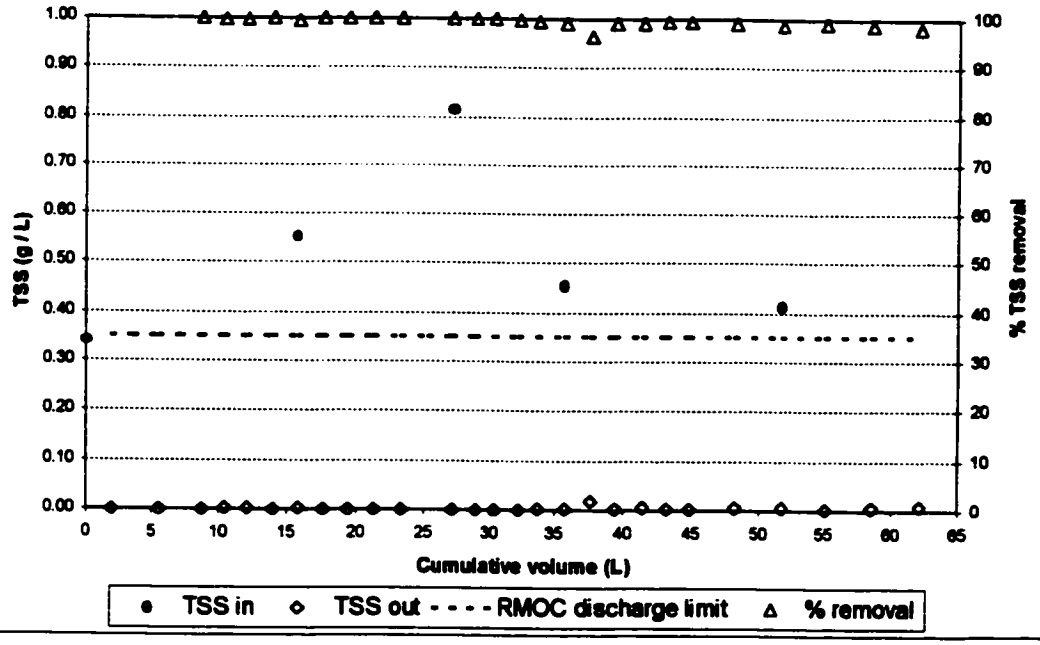
In subsequent experiments, fractionation volumes of effluent should be decreased to differentiate initial saturation water leaching out of peat versus a decrease of COD concentration resulting from peat adsorption. Channelling problems were encountered because the peat was accidentally allowed to run dry, *i.e.*, the filter was not over-saturated with liquid. It was concluded that special attention should be given to not allowing peat filters to run out of excess liquid, otherwise filtration performance could be seriously affected.

5.4 RESULTS AND DISCUSSION: TSS REMOVAL AND FLUX VARIATION

5.4.1 TSS REMOVAL

The removal of TSS was quantitatively measured and was concluded to be successfully achieved by peat filtration. TSS removal of 100% was obtained after up to five times V_r and for an average removal of 99.5% was measured after 62.1 L of leachate filtered (approximately eleven times V_r). Figure 5.1 shows TSS removal versus filtered leachate volume. The peat filtration pre-treatment was concluded to be an excellent technique of TSS removal from CRL leachate.

Figure 5.1: TSS in and out of peat filter

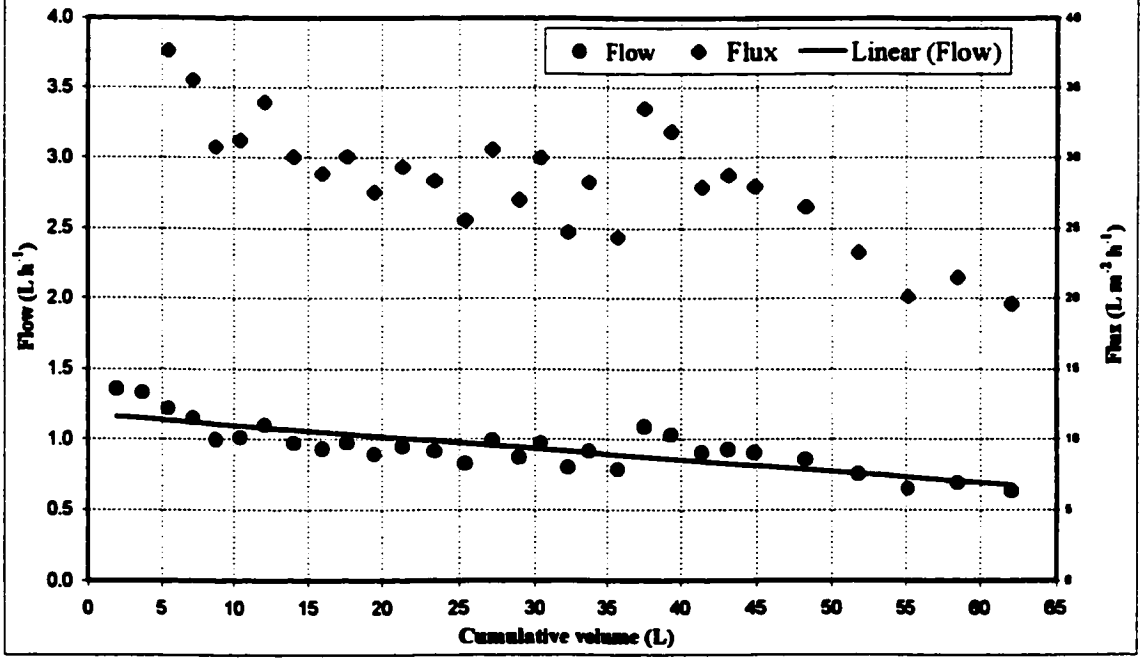


The calculated TSS accumulation on top of the peat filter was 27.4 g of TSS (from solids test results and assuming 100% removal). This accumulation was found to be equivalent to 846 g TSS per square meter of peat surface until flux was reduce by 50%, this corresponded to the filtration of 1630 L of raw CRL leachate per square meter of peat surface, (raw CRL leachate contained an average of 0.52 g TSS L^{-1}).

5.4.2 FLUX VARIATION

The leachate flow through peat filters decreased with time (Figure 5.2). At first, a flow rate of 1.3 L h^{-1} was measured, then after five V_r, it was at 0.87 L h^{-1} (a decrease of 33%) whereas at the end (after eleven V_r), the flow rate had decreased by 50%, to 0.64 L h^{-1}. The surface area of the peat filter being 0.0324 m^2, fluxes were calculated to vary from 40 L m^{-2} h^{-1} initially to 20 L m^{-2} h^{-1} at the end of the experiment. Solids precipitates on the filter top, which were powdery and dense, were forming a cake and were most likely responsible for the flux decrease, in combination with peat column compaction.

Figure 5.2: Peat filter flux variation versus cumulative volume



Assuming the flux decrease being linear and remaining linear until total obstruction, the zero flux would be reached after filtration of 147 L of raw leachate. The clogging problem may be overcome by removing the top layer of peat, discarding it and replacing it with fresh peat. However, if specific metal adsorption by peat was examined simultaneously with SS removal, then replacement of the whole filter might be necessary. The replacement of peat inside filters should be set according to a design criterion, either TSS or metal removal or any element in the effluent which concentration could fail meeting the POTW discharge limit.

5.5 RESULTS AND DISCUSSION: SPECIFIC METALS AND IONS REMOVAL

The removal of soluble metals and inorganic ions was studied as a parallel benefit of using peat as a filtering agent. Interestingly, the removal efficiencies of some metals were high, whereas some other elements were barely removed or not removed at all.

The selection of metals and inorganic ions was based on different criteria;

- Boron, barium, and iron were studied as requested by CWS,
- Manganese and zinc were studied because they were found in an initial ICP analysis to be exceeding the RMOC discharge limits in the raw leachate,
- Calcium was studied because it was found to be one of the main inorganic elements accumulating in UASB reactors,
- Aluminium and cobalt were initially measured in high concentrations in the raw leachate, however both of them were found not to be problematic for POTW discharge limit. Their results are discussed regardless of this situation.

Since filtered CRL leachate was to be fed to biological reactors, essential biological nutrients and TOC removal by peat pre-filters were studied. In the following sections, metals and inorganic ions (in alphabetical order) removal are discussed first, followed by the nutrients and TOC effluent content.

5.5.1 ALUMINIUM

Interestingly, aluminium was leaching out of the peat filter in random concentrations until two V_r were filtered, excluding the first V_r, which was the filter's initial water content (Figure 5.3). Thereafter, the removal was total (100% removal). It was assumed that some elements in the leachate reacted, with or without peat involvement in the reaction, to complex aluminium and precipitated it out of solution. No maximum adsorption value was observed and no indication of saturation was measured. CRL leachate average aluminium concentration was 0.166 mg L^{-1}; therefore, excluding the first 14 L of cumulative volume, a total of 9.19 mg of aluminium was adsorbed or precipitated in the peat filter, yielding an adsorption of 7.35 mg L^{-1} Al kg^{-1} peat.

Figure 5.3: Aluminium adsorption on peat

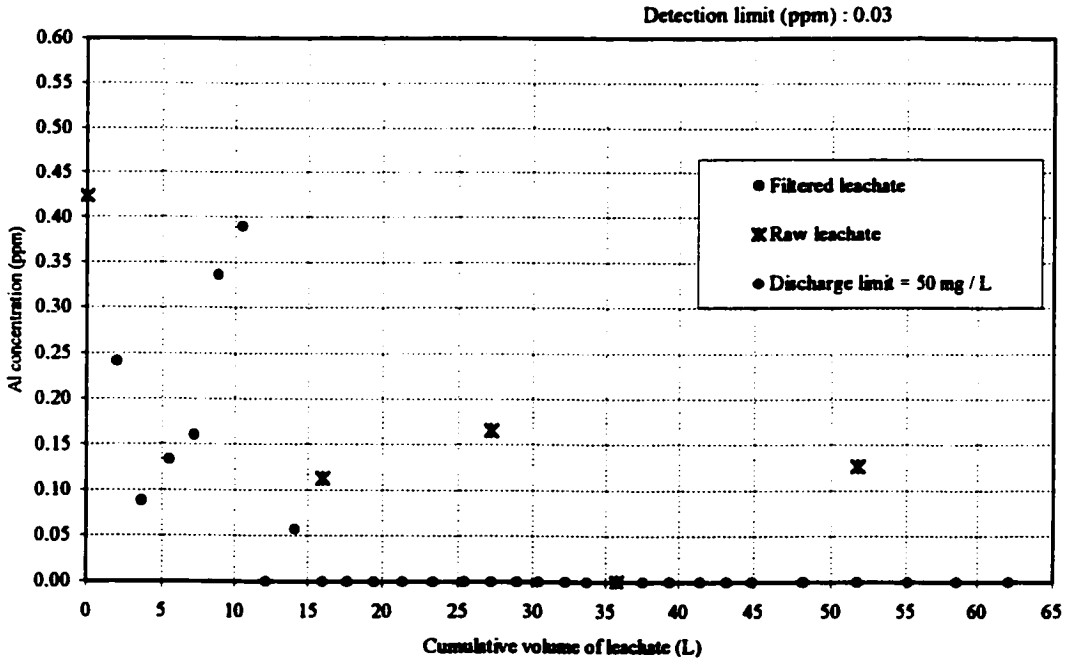
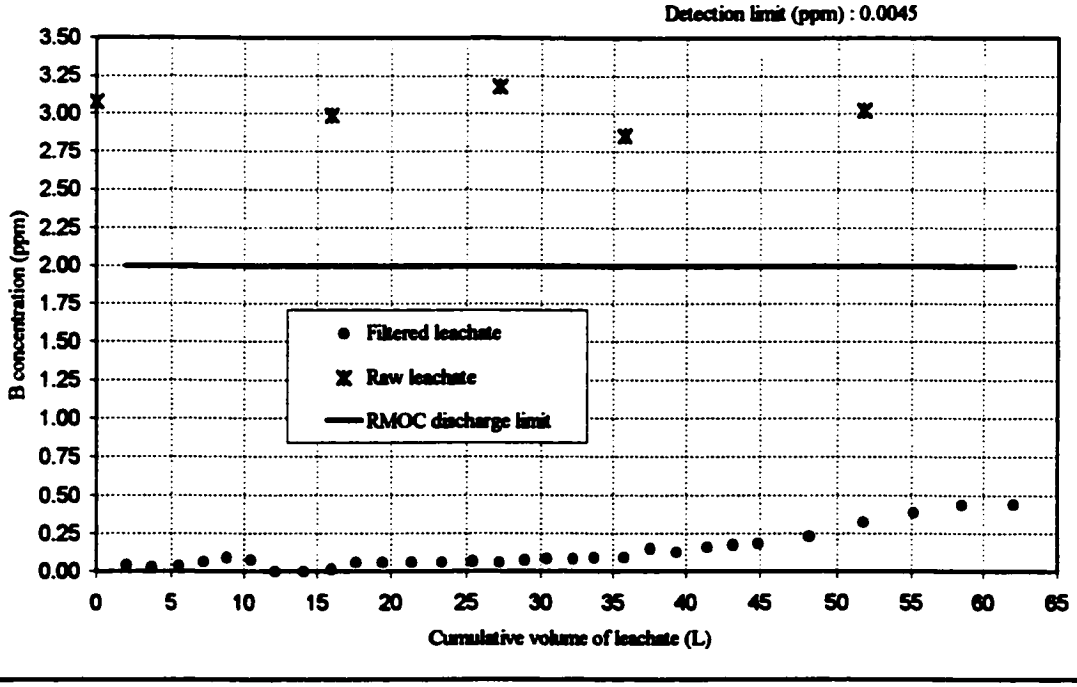


Figure 5.4: Boron adsorption on peat



5.5.2 BORON

There was excellent boron removal by peat (Figure 5.4); 97% removal and above was maintained until six V_r were passed through the filter. Boron was initially leaching out of the peat filter; and was detected within the same concentration range in the first V_r than after six V_r had been filtered.

After a cumulative volume of 44.9 L (8 V_r), the effluent boron concentration started to increase. The concentration curve was assumed to be linearly increasing from 44.9 L until the average raw leachate concentration (3.03 mg B L^{-1}) would be reached (outside data range on Figure 5.4). A linear regression was calculated with Microsoft® Excel 97 from the cumulative volume 44.9 L point to the last data point (62.1 L). The regression predicted a leach out of boron at the average raw leachate concentration at cumulative volume of 217 L (38 V_r). Since the RMOC discharge limit must be lower than 2.00 mg B L^{-1}, according to the same linear regression, after a cumulative volume of 155 L (27 V_r), the effluent boron concentration from the peat pre-filter should reach this limit.

The overall removal efficiency of boron, until the flux was 50% of its initial value, was 95%. Therefore to this point, it was estimated that 177 mg B was adsorbed on the filter, yielding an unsaturated adsorption of 142 mg B kg^{-1} dry peat.

5.5.3 BARIUM

Barium removal was poor; in fact results indicate that barium even leached out of the peat filter material, resulting in a higher concentration in the filter effluent compared to influent (Figure 5.5). The saturation point, where the effluent concentration was equal to the inlet concentration, was at a cumulative volume of about 14 L, which correspond to the volume of water initially contained in the filter, plus 1.5 V_r of leachate. There was a net desorption of barium of 5.87 mg out of the filter or 4.69 mg Ba kg^{-1} dry peat. Before the saturation point, there was a barium adsorption of 2.31 mg kg^{-1} dry peat (or 2.88 mg Ba for the whole filter) but of 33.9 mg barium input in the filter, 39.8 mg barium leached out; the net desorption was 24%.

This situation is unfortunate because, as observed in Figure 5.5, the discharge limit of 1.2 mg Ba L^{-1} is close to being reached as barium is leached out of the filter. Therefore, a cautious monitoring of barium in and out of peat filters should be made with respect to the municipal discharge limit.

Figure 5.5: Barium adsorption on peat

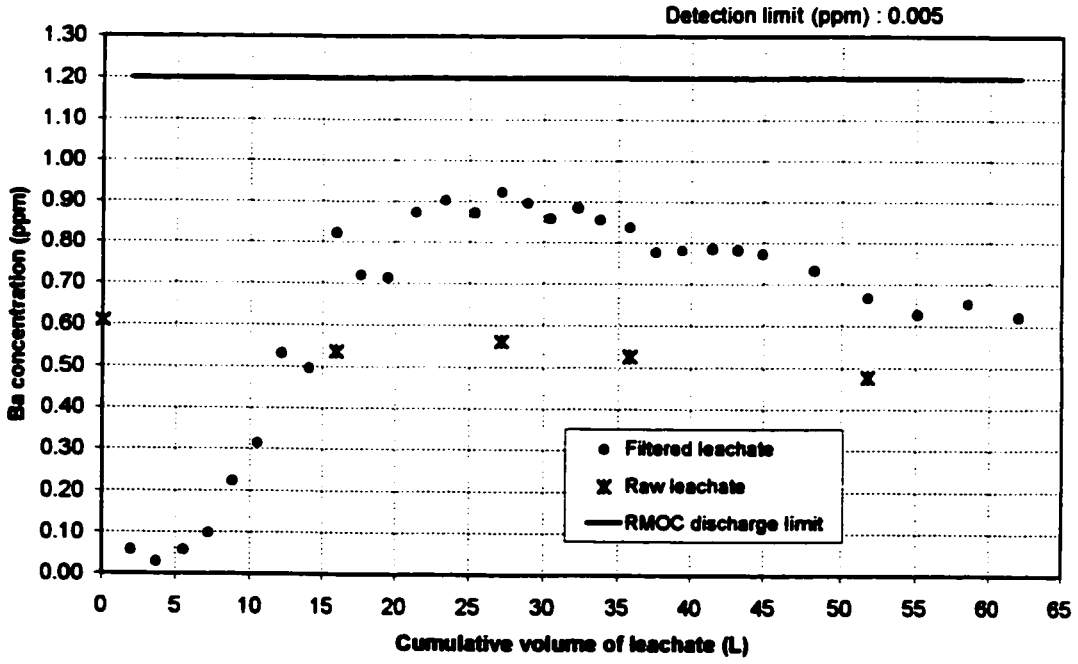
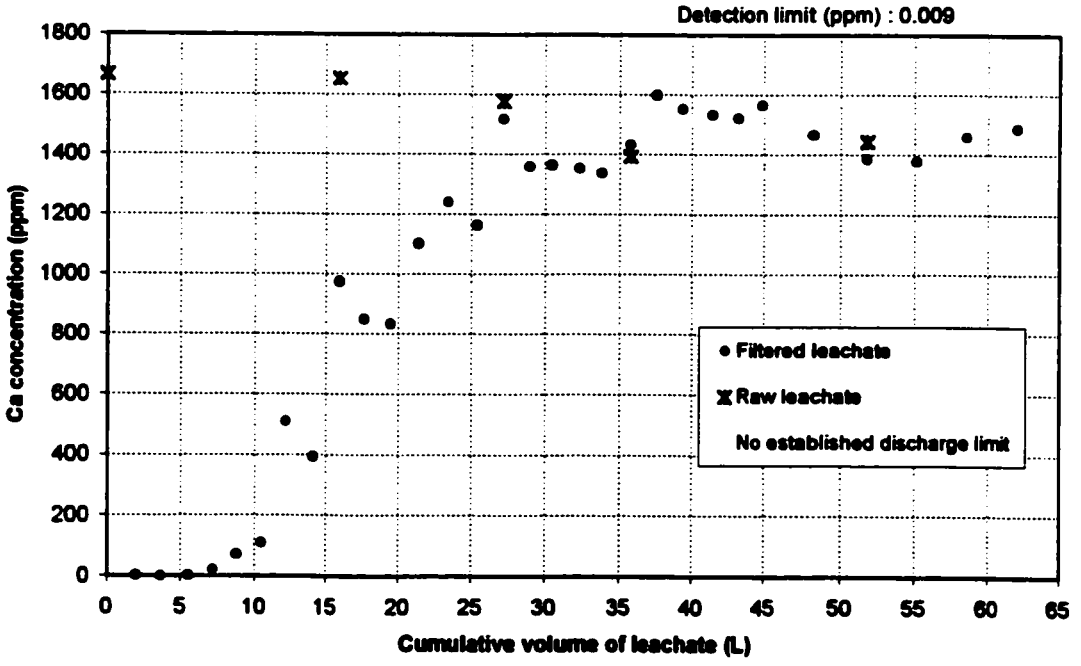


Figure 5.6: Calcium adsorption on peat



5.5.4 CALCIUM

There was an initial calcium adsorption on peat (Figure 5.6) but its concentration increased rapidly and effluent concentration reached influent concentration after five V_r (27.2 L). High concentration of calcium in CRL leachate (1547 mg Ca L^{-1}) explained the rapid saturation of the filter. It was concluded that it would be difficult to remove such high concentrations of calcium by peat adsorption. If calcium concentrations were to be lowered, a chemical precipitation prior to peat filtration could be more efficient. Precipitated calcium would most probably accumulate on the peat filter top and could be removed periodically along with TSS.

Despite the rapid saturation, there was an overall calcium removal by peat filtration; 27% of calcium was precipitated or adsorbed by peat. Before the saturation point was reached at 27.2 L, an adsorption of 16.4 g of calcium was measured for the whole filter, or 13.1 g Ca kg^{-1} dry peat.

5.5.5 COBALT

Cobalt adsorption was good; after three V_r, an overall removal of 74% was achieved (Figure 5.7). Despite a small initial concentration, the effluent cobalt concentration consistently increased; the assumption of a linear increase to a saturation point from a cumulative volume of 7.2 L was made. A linear regression calculated with Microsoft® Excel 97 from the 7.2 L data point to 62.1 L predicted that the average cobalt raw leachate concentration would be reached at a cumulative treatment volume of 163 L (28.8 V_r). However, because the RMOC discharge limit was higher than CRL leachate concentration, high removal of cobalt by the peat filter was not an objective.

The overall removal of cobalt in this experiment was 74% (calculation excludes results for cumulative volume up to 5.5 L). There was only 7.0 mg cobalt adsorbed on the filter, yielding an unsaturated adsorption of 5.6 mg Co kg^{-1} dry peat.

5.5.6 IRON

Iron removal results by peat filtration were variable; no general trendline could be established (Figure 5.8). Influent iron concentration was also variable, ranging from 51 to 14 mg Fe L^{-1}. The overall average removal was 61% and an unsaturated adsorption of 922 mg Fe kg^{-1}dry peat was observed. Iron concentration was not problematic for POTW discharge

Figure 5.7: Cobalt adsorption on peat

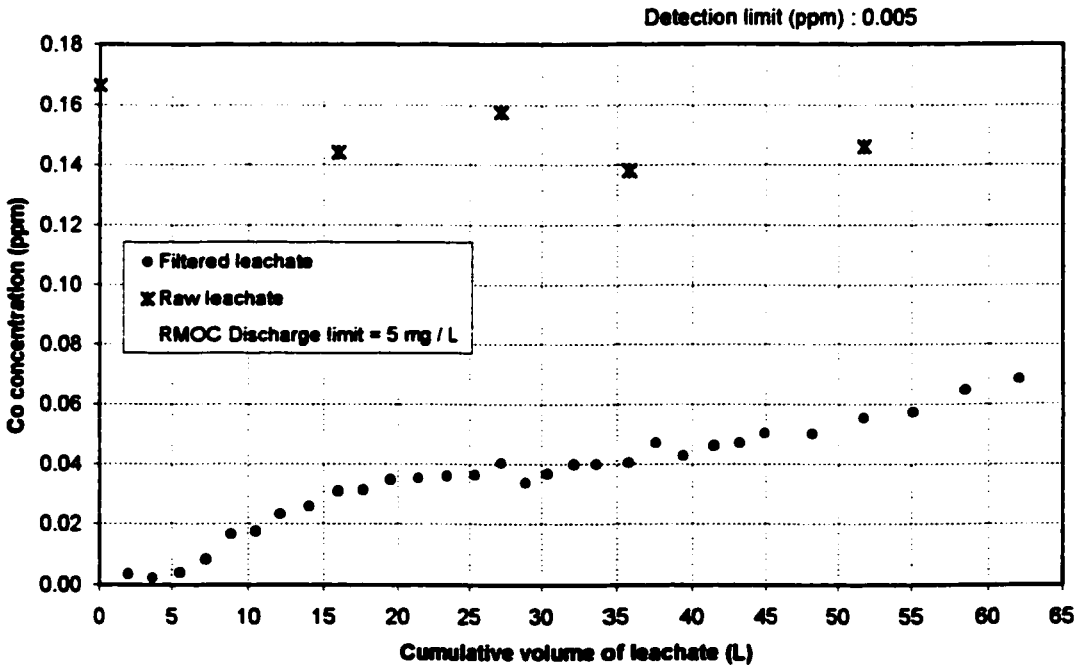
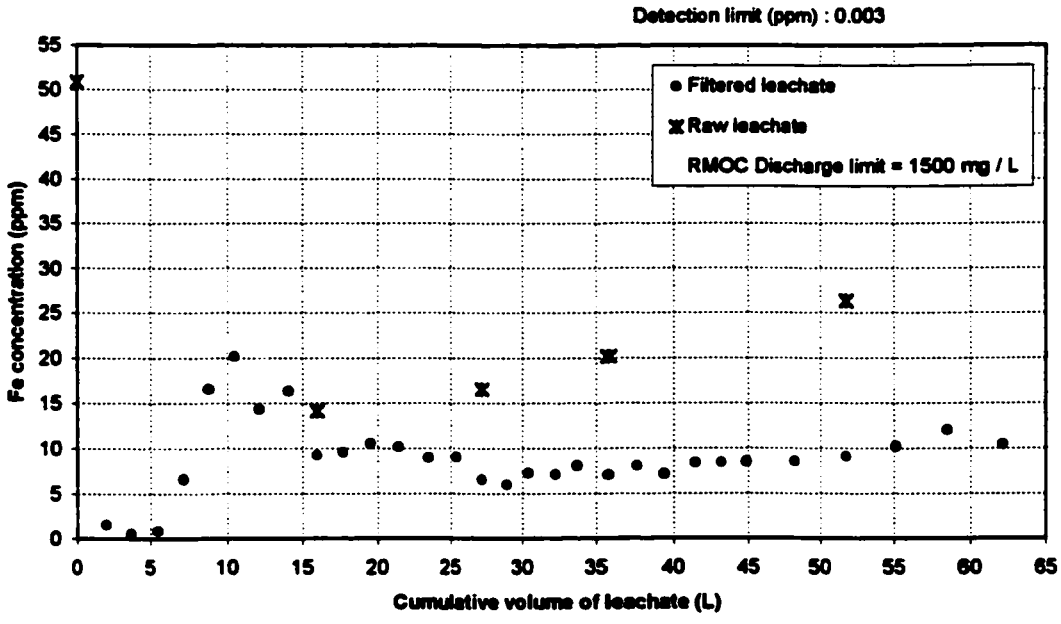


Figure 5.8: Iron adsorption on peat



requirements (1500 mg L^{-1}); however, it was problematic for UASB reactors; as it accumulated in the sludge bed and caused biomass displacement and decrease in specific activity. The highest concentration of iron in raw CRL leachate was found in the TSS (Table 5.1); over 97% of iron was contained in TSS. Therefore, in the specific case of iron, it was more important to remove TSS than soluble iron.

5.5.7 MANGANESE

Manganese removal was moderate; the overall average removal was 40%. Manganese effluent concentration continuously increased until it reached the influent value after pre-treatment of 37.6 L (6.6 V_r) (Figure 5.9). Manganese concentrations exceeded the POWT discharge limit of 5.00 mg L^{-1} after three V_r, including the initial water content (cumulative volume of 16.0 L). Therefore, manganese could be problematic if the peat treated effluent was to be discharged to a municipally managed wastewater treatment facility. However, removal by UASB reactors, discussed in Section 6.3.6g, solved the high effluent manganese concentration problem.

There was 292 mg of manganese adsorbed per kg of dry peat before saturation was observed at 35.8 L (6.3 V_r).

5.5.8 ZINC

High zinc removals were achieved; overall average removal was 95%. Initially, there was an increase above the influent zinc concentration (Figure 5.10), up to a cumulative volume of 12.1 L, including the initial water volume. However, this increase was followed by a sudden decrease, down to an average effluent concentration of 0.4 mg Zn L^{-1} which was maintained until the end of the experiment. This observation was also observed in the peat studies conducted by Cameron (1978) who also observed initial zinc desorption followed by a zinc adsorption.

No saturation prediction could be established because effluent zinc did not increase within the experimental limits of the test. Peat was therefore concluded to be efficient in removing zinc, bringing its concentration to below the POTW discharge limit of 3.00 mg Zn L^{-1}. An unsaturated adsorption of 227 mg Zn kg^{-1} dry peat was observed in this study.

Figure 5.9: Manganse adsorption on peat

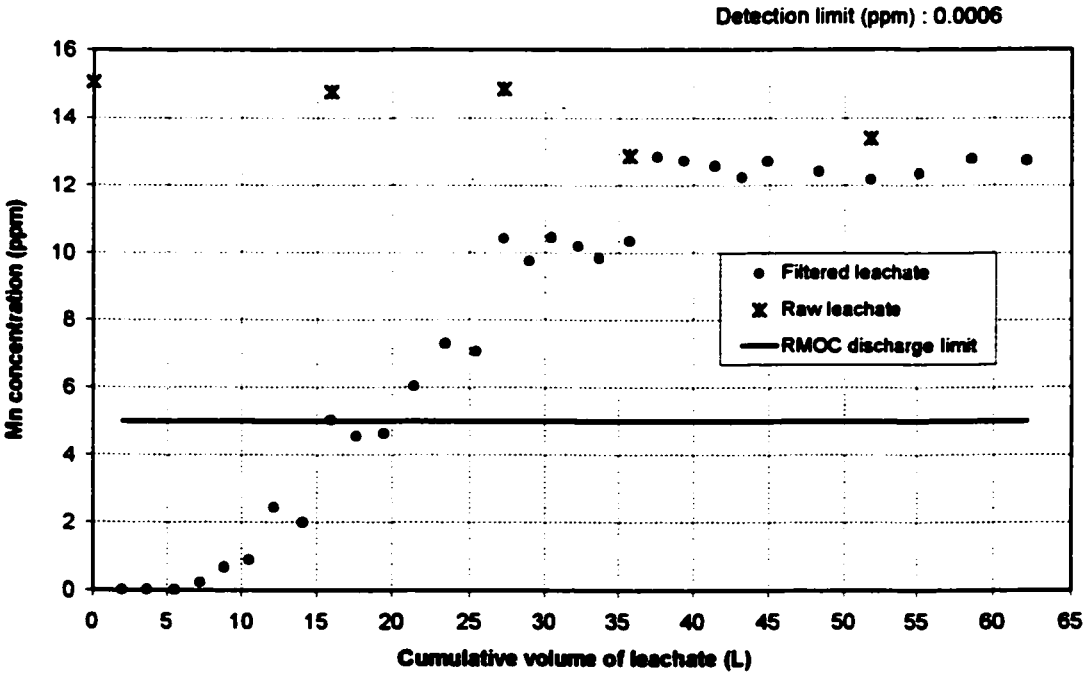
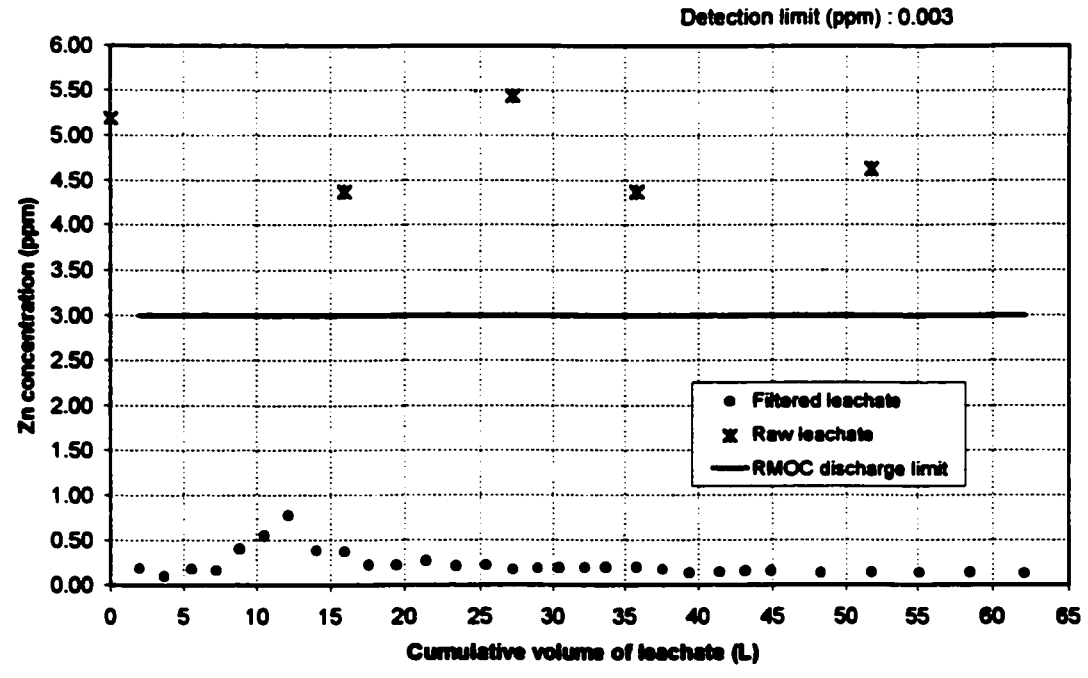


Figure 5.10: Zinc adsorption on peat



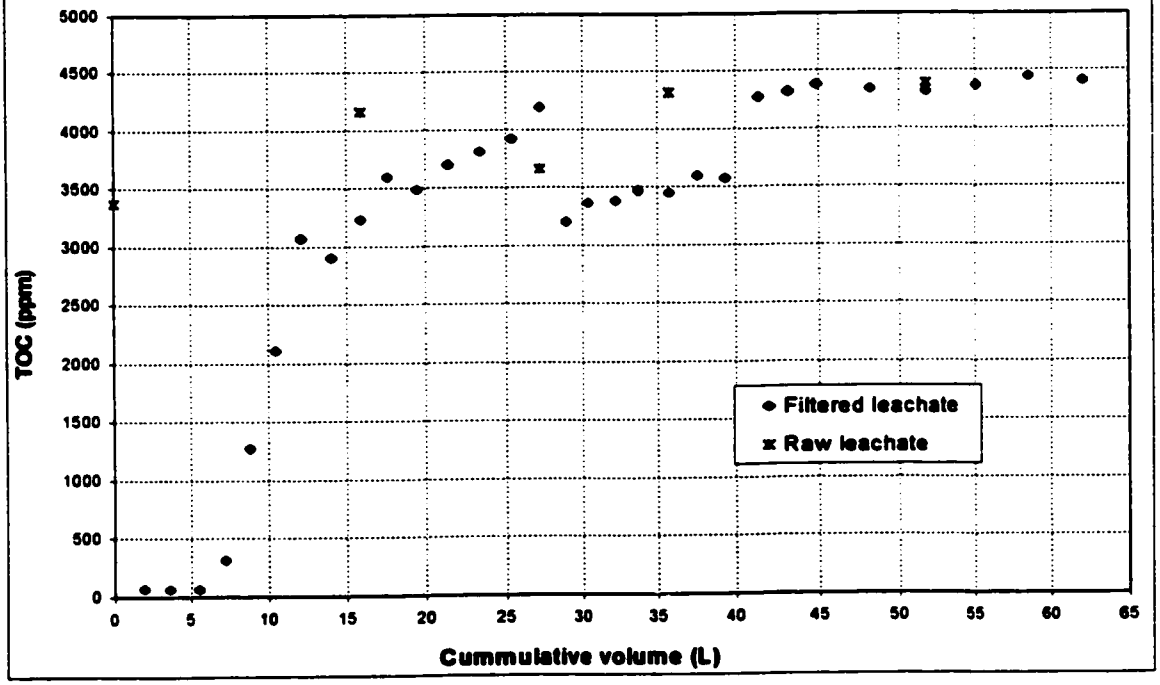
5.5.9 PHOSPHORUS AND TKN

Both phosphorus (P, as total phosphorus) and nitrogen (N, as total Kjeldahl nitrogen) were measured on three filtered leachate samples and compared to corresponding raw leachate concentrations. The removal of both P and N was approximately 50% (Table 5.3). It was therefore concluded to be essential to consider nutrient monitoring or addition after peat filtration when the filtered leachate is fed to a biological reactor. P and N concentrations should be at a ratio of 500:5:1 COD:N:P for anaerobic treatment. In the present case, P ratio was low, considering that the COD concentration was averaging 20 000 mg COD L^{-1}, it resulted in a COD:P ratio of 20 000:1. N was not deficient; filtered leachate had, at 200 mg L^{-1}, a COD:N ratio of 100:1 which was an adequate N requirement for anaerobic treatment.

Table 5.3: Removal of nutrients P and N by peat filtration

	Raw leachate			Filtered leachate			Average removal (%)
Sample #	1	2	3	1	2	3	
TKN	451	528	432	226	230	204	
TP	2.04	2.26	2.35	1.18	1.11	1.08	
Ave TKN		470			220		53
Ave TP		2.22			1.12		49

Figure 5.11: TOC variation versus cumulative volume



5.5.10 TOTAL ORGANIC CARBON

There was a modest removal of TOC during peat pre-treatment: the overall average TOC removal was 15% (Figure 5.11). TOC concentration in raw leachate was high (3980 mg TOC L^{-1}) and peat was expected to rapidly reach its saturation point from previous preliminary experiments. The saturation adsorption was estimated to be 32 g TOC kg^{-1} dry peat. The low TOC removal by peat filtration resulted in an effluent ready to be fed to biological reactors: rich in soluble organic matter but free of suspended solids.

The balance between TSS and metals removal expense versus enhanced biological activity induced by high organic but low TSS content feed should result in better functioning of UASB reactors.

5.6 CONCLUSION

Peat filtration is an excellent means of TSS removal for CRL leachate; moreover, peat is cheap, easily available and re-usable as a daily cover in a landfill site. However, if TSS removal exclusively was required, a back-washable sand filter could be an efficient alternative. Nevertheless, peat filtration has an important advantage over sand filtration: it has the capacity to adsorb specific soluble metals and inorganic ions.

Peat was highly efficient for removal of aluminium, boron, and zinc at inlet concentrations found in the CRL leachate. Peat was not efficient in achieving sufficient removal of manganese (influent concentration of 15 mg L^{-1}) for a permitted discharge value of 5.00 mg L^{-1}. Peat pre-treatment of barium was also poor, additionally, barium desorption from the peat occurred and barium has a low discharge limit of 1.20 mg L^{-1}. Iron removal was variable but deemed unimportant because influent concentrations were below the discharge limit, as was the case for cobalt. Chemical precipitation of calcium prior to peat filtration should be studied to prevent Ca accumulation/precipitation in UASB treatment.

In order to prevent the flux from decreasing until filter clogging occurs, the filter's top layer should be periodically scraped/discarded and replaced with fresh peat. However, depending on the objectives of peat filtration, all the peat in the filter could be replaced if the adsorption concentration of a metal was exceeded or the effluent concentration exceeded the maximum municipal discharge limit.

CHAPTER 6

UASB TREATMENT OF FILTERED LANDFILL LEACHATE

6.1 INTRODUCTION

The same set up as described in Chapter 4, consisting of three UASB reactors was run at various organic loading rates and HRTs but with similar anaerobic granular biomass concentrations. Three series of reactor runs were conducted: feed leachate COD concentration and reactor's flow rates varied in order to mimic seasonal landfill leachate conditions (dilute in spring, concentrated in summer and fall). Raw CRL leachate was first pre-treated with sphagnum peat moss filters. The COD concentration of CRL leachate was then adjusted with tap water to achieve the targeted seasonal feed COD concentration. COD removal efficiency, biogas production, biomass activity, solids accumulation, and metals removal were studied.

The objective of these experiments was to demonstrate the efficiency of the full treatment process, which combined peat filter pre-treatment followed by anaerobic treatment in a UASB reactor, for various practical operating conditions.

6.2 SPECIFIC MATERIALS AND METHODS

In these series of experiments, the reactor temperature was 32 ±1°C. VFAs, flow rate, CODs were monitored daily, or every two days for runs R10-R12. For runs R4, R5, and R6, soluble COD was measured whereas for runs R7 to R12, both soluble and total COD were measured. The pH (for all runs) and effluent TSS for runs R7-R12 were measured twice weekly and the bed height, once a week. Raw leachate was pre-treated by peat filtration then diluted to the desired concentration with tap water to produce reactor feed. Similar types of pump failures as mentioned in Chapter 4 also occurred during these sets of experiments. When such failures occurred, no measurements were taken, or data were not included in calculations.

Averages, standard deviations, and linear regressions were calculated using Microsoft® Excel 97 functions. Reactor data were analysed at pseudo-steady state: the pseudo-steady state was defined as a running period of at least six HRTs and a variation of less than 15% of parameters.

The choice of 15% was not ideal but was a consequence of numerous pumping failures. During these series of tests, the methane gas fraction of the biogas could not be measured because the GC was out of order. Therefore, methane fractions were calculated with equation 2.5.2, based on volumetric COD removal rates, biogas productions, and the methane collected fraction, f_m, based on runs R1-R3: 0.92.

Fresh anaerobic granular biomass from Lake Utopia Paper Ltd. was mixed with leachate acclimated biomass from R1-R3 and this mixture was used to equally fill all reactors for runs R4-R6. For these runs, it was assumed that the initial solids profile and microbial activity was the same in the biomass sludge bed of all three reactors since their biomass originated from the same source. Reactors were left 48 hours on recycle then an AAT and a solid profile measurement were performed on one of the reactors to determine initial solids and specific acetoclastic methanogenic activity conditions.

A period of acclimation was given prior to starting each run; reactors were fed with diluted landfill leachate at low rate. The feed rate was thereafter incrementally increased and the reactors condition, by means of VFAs monitoring, was carefully monitored until the desired operating conditions were reached. For runs R4-R6, an acclimation period of 48 hours with a leachate concentration of 6000 mg L^{-1}, was sufficient to fully recover the original activity and COD removal efficiency after 8 weeks of storage at room temperature. The same biomass from runs R4-R6 was reused for runs R7 to R9; an acclimation period of 15 days was required because biomass had been inactive for a period of five months. For the entire period of inactivity, biomass was stored in reactors at room temperature.

6.3 RESULTS AND DISCUSSION

6.3.1 OPERATION CONDITIONS

It is known that leachate concentration and flow vary with seasonal changes (Méndez *et al.*, 1989); concomitantly feed CODs were adjusted to simulate Canadian seasonal variation. Winter was not simulated because microbial activity and leachate production in landfills is low during the wintertime; average temperatures are below 0°C and in most cases meteorological precipitation is in the form of snow. The three other seasonal conditions were simulated; a dilute but voluminous leachate simulated springtime conditions, a concentrated and moderate flow of leachate simulated summertime, and a moderate concentration and moderate flow rate simulated

fall conditions, corresponding to HRTs of 1.9, 1.3, and 0.95 d respectively. Runs R7-R9 (fall conditions) had a moderated leachate COD concentration of 7500 mg COD L^{-1}, HRTs of 3.8, 2.2, and 1.5 d, corresponding to OLRs of 2, 3.5 and 5 g COD L_r^{-1} d^{-1} for R7, R8, and R9, respectively. Finally, runs R10-R12 (summer conditions) had a leachate concentration of 15,000 mg COD L^{-1}, HRTs of 3.0, 2.0, and 1.5 d, resulting in OLR of 5, 7.5, and 10 g COD L_r^{-1} d^{-1} for R10, R11, and R12, respectively. Spring condition runs were conducted from May 26th to June 16th (22 days), fall condition runs from November 14th to December 4th (21 days) and summer condition runs from January 4th to 26th (20 days).

The targeted final effluent COD quality (COD % removal) was set according to the RMOC SUB discharge limit of 1000 mg COD L^{-1}. The measured values from experiments R4-R12 of percent removal efficiency are compared to corresponding targeted values of feed concentration, HRT and OLR are summarised in Table 6.1.

In most cases the SUB target values were reached during season simulation experiments.

Table 6.1: Targeted and measured values for R4 to R12

	TARGETED VALUES				MEASURED VALUES (± standard deviation)		
	Flow rate (L/d)	HRT (d)	OLR (g COD L_r^{-1} d^{-1})	Removal (%)	Flow rate[1] (L/d)	HRT[1] (d)	OLR[1,2] (g COD L_r^{-1} d^{-1})
Spring, influent COD: 6000 mg L^{-1}					Influent COD[1,2]: 5839±580 mg L^{-1}		
R4	3.0	1.9	3.1	> 83	2.8 ± 0.1	2.1 ± 0.1	2.9 ± 0.3
R5	4.5	1.3	4.7	> 83	3.9 ± 0.3	1.5 ± 0.1	3.9 ± 0.6
R6	6.0	0.95	6.3	> 83	6.7 ± 0.8	0.9 ± 0.2	6.5 ± 1.2
Fall: effluent COD 7500 mg L^{-1}					Influent COD[1,2]: 6521± 845 mg L^{-1}		
R7	1.5	3.8	2.0	> 87	1.7 ± 0.2	3.4 ± 0.2	2.1 ± 0.4
R8	2.7	2.2	3.5	> 87	2.9 ± 0.1	2.0 ± 0.1	3.0 ± 0.4
R9	3.8	1.5	5.0	> 87	4.2 ± 0.1	1.4 ± 0.04	4.7 ± 0.6
Summer: effluent COD 15,000 mg L^{-1}					Influent COD[1,2]: 16 261± 594 mg L^{-1}		
R10	1.9	3.0	5.0	> 93	1.9 ± 0.1	3.1 ± 0.2	5.2 ± 0.5
R11	2.9	2.0	7.5	> 93	2.8 ± 0.2	2.2 ± 0.2	7.4 ± 0.5
R12	3.8	1.5	10	> 93	3.5 ± 0.2	1.6 ± 0.1	10.0 ± 0.7

1- average values 2- soluble COD

6.3.2 COD REMOVAL

In order to achieve COD compliance to the SUB, COD removal efficiencies had to range from 83 to 93% based on targeted influent conditions (Table 6.1). In all experiments R4-R12, targeted removal efficiencies were exceeded (Table 6.2). The substantial COD removals achieved in runs R4-R12 have re-demonstrated the high biodegradability of the CRL leachate; COD removal efficiencies exceeded 90% most of the time, up to average soluble COD removals of 98% for runs R7, R8 and R10 which simulated summer and fall conditions (Tables 6.2 and 6.3).

Removal efficiencies of R4-R6, which simulated springtime conditions, will be discussed separately from R7-R12: total and soluble COD were measured for the latter, whereas only soluble COD was determined for the former. In general total COD and soluble COD results for the same sample were generally insignificantly different; the difference was concluded to be within experimental data error. As an example, R7-R9 filtered feed leachate had an undetectable TSS fraction; however experimental values for influent soluble and total COD were, on average, 6815 and 6521 mg COD L^{-1} respectively. Total COD would have been expected to be at least equal to or greater than soluble COD; however dilutions and experimental manipulations were deemed as the major source of variance. On the other hand, TSS in the effluent contributed measurably to total and soluble COD concentration difference. For instance, R9 average total and soluble COD effluent concentrations were 248 and 164 mg COD L^{-1} respectively. The contribution of suspended COD overall was small and in most cases it was assumed not significant enough to be discussed in detail. Therefore, only soluble COD results will be discussed in subsequent sections, unless explicitly mentioned otherwise.

Table 6.2: Removal efficiencies and loading rates for R2, R4-R6

	HRT d	OLR g COD L_r^{-1} d^{-1}	VRR g COD L_r^{-1} d^{-1}	COD Removal %
R2	2.2	5.3	5.1	94
R4	2.1 ± 0.08	2.9 ± 0.35	2.6	96 ± 1.3
R5	1.5 ± 0.14	3.9 ± 0.61	4.2	94 ± 1.5
R6	0.9 ± 0.19	6.5 ± 1.2	6.2	91 ± 0.8

6.3.2.a COD efficiency results for R4-R6

R4-R6 had the lowest HRT range of all runs as springtime conditions implied high volumetric production of leachate, *i.e.* high flow rates. COD removals of 91% to 96% were achieved at average OLRs of 2.9 to 6.5 g COD L_r^{-1} d^{-1} (Table 6.2). Runs R4 and R5 were stable and experienced no major problem; VFAs concentration remained low excepted for a short period for R5 when propionic acid concentration started to increase to above 100 mg L^{-1} around June 11th.

R6 experienced some operational problems; propionic acid averaged 150 mg L^{-1} and butyric acid was sporadically found in the effluent. R6 had the highest OLR of this set of runs (6.5 g COD L_r^{-1} d^{-1}), and the lowest COD removal efficiency; yet no effluent COD concentration higher than 1000 mg COD L^{-1} was detected. Considering the possible occurrence of a shock load, R6 would most probably have experienced possible SUB excess and difficulties returning to pre-overload conditions because of its high VFA concentrations, in particular propionic acid. Increased stability and higher removals would have been expected if P would have been added and was not limiting, as was the case for R10-R12 (discussed in section 6.3.2 b). This hypothesis should be verified for extended treatment of CRL leachate under R6 operational conditions.

While treating CRL leachate under simulated spring conditions, R5 conditions, a moderate HRT of 1.5 d and an OLR of 3.9 g COD L_r^{-1} d^{-1}, seemed to provide more stable treatment over R4 or R6 conditions. However, if P limitations did occur, then it is possible that R6 conditions may have produced equal or better results. At this time, nutrients limitations can only be speculated and conditions should be checked if this limitation does exist. P addition is strongly recommended in order to increase stability and resistance to shock loads of reactors. Compliance to SUB discharge limit and reactor stability should be tested more thoroughly prior to moving to a larger scale.

6.3.2.b COD efficiency results for R7-R12

R7 to R11 COD removal efficiencies were the highest of all experiments (Table 6.3 and Figure 6.1); 96% to 98% total and soluble COD removal efficiencies were achieved with a low variance at OLRs ranging from 2.1 to 7.4 g $COD_{(soluble)}$ L_r^{-1} d^{-1}. R12 had the highest of all OLRs (10.0 g $COD_{(soluble)}$ L_r^{-1} d^{-1}) and achieved a satisfactory average COD removal of 94%. R12 high removal efficiency was possible in part because P was added before a potential reactor failure (please see details below) could occur.

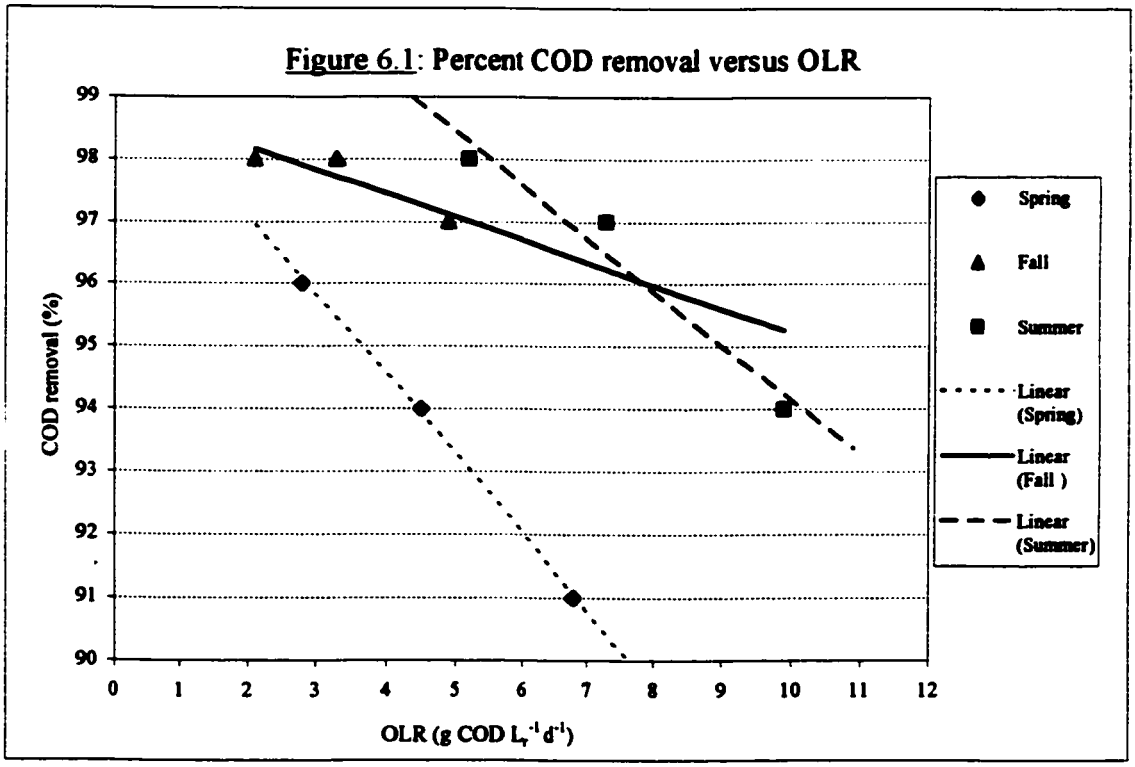


Figure 6.1: Percent COD removal versus OLR
COD removal (%)
99
98
97
96
95
94
93
92
91
90
0
1
2
3
4
5
6
7
8
9
10
11
12
OLR (g COD L-1 d-1)
Spring
Fall
Summer
Linear (Spring)
Linear (Fall)
Linear (Summer)

R7-R9 reactors (fall condition simulation) were stable: VFAs concentrations were undetectable (< 5 mg L^{-1}) or at low concentrations in all runs. As expected, because R7-R9 OLR and flow rate ranges were lower, R7-R9 achieved higher removals (96-98%) compared to spring conditions (R4-R6: 91-96%). Running UASB reactors under fall conditions resulted in high removal efficiencies; no significant problems were encountered except achieving the targeted influent COD concentration because of the concentration variability of filtered leachate. Therefore, if run on a large scale and during fall conditions, reactors could be operated using R9 parameters, *i.e.* an OLR of 4.6 g COD L_r^{-1} d^{-1}, or even higher as long as no sudden variations in influent COD concentration are anticipated.

Table 6.3: Removal efficiencies and loading rates for R7 to R12

	OLR (soluble COD) (g COD L_r^{-1} d^{-1})	OLR (total COD) (g COD L_r^{-1} d^{-1})	COD removal, total (%)	COD removal, soluble (%)
R7	2.1 ± 0.4	2.0 ± 0.3	97 ± 0.2	98 ± 0.1
R8	3.0 ± 0.4	3.1 ± 0.4	97 ± 0.2	98 ± 0.1
R9	4.7 ± 0.6	4.6 ± 0.5	96 ± 0.3	97 ± 0.2
R10	5.2 ± 0.5	5.4 ± 0.5	97 ± 0.2	98 ± 0.1
R11	7.4 ± 0.5	7.4 ± 0.5	96 ± 0.5	97 ± 0.4
R12	10.0 ± 0.6	10.0 ± 0.7	94 ± 1.4	94 ± 1.5

Summer simulating runs (R10-R11) also resulted in high COD removals (96-98%) at high OLRs (5.2-7.4 g COD L_r^{-1} d^{-1}) but overall removal efficiencies were strongly dependent on P addition (January 20th, please see Appendix A). Prior to that date, R12 experienced severe VFA accumulation; it had the highest OLR (10.0 g COD L_r^{-1} d^{-1}) of all runs and the effluent rapidly exceeded the SUB discharge limit. VFAs were high (averages of 256 mg acetic, 171 mg propionic and 11 mg butyric acids per litre of effluent) indicative of microbial stress and no signs of recovery or stabilisation was apparent despite a high COD removal efficiency.

Phosphate addition was initiated on January 25th because the effluent quality, especially of R12, was systematically decreasing. A positive and rapid response of all reactors was observed and all VFAs concentrations promptly went down to non-detectable limits. R12 met the discharge limit within 24 hours. Additionally, R11 effluent COD decreased from above 600 to below 200 mg COD L^{-1}. As discussed in Chapter 5, peat filtration was unexpectedly removing approximately half the P content from CRL leachate; reactors fed on filtered leachate were therefore more susceptible to be P-limited compared to those fed on raw leachate.

From R10-R12 experiments, it was suggested that P (as phosphate) would be added in order to provide a nutrient safety factor while ensuring that at all times the bylaw discharge limit for P was not exceeded. If no P additions were to be performed, lower OLRs are suggested, but the likely possibility of poor or unstable results caused by phosphorus deficiency will remain. For summer conditions, R12 conditions were determined to be the optimal conditions for UASB operation (with mandatory P addition).

It was concluded that if R4-R6 runs (spring conditions) were restarted after runs R10-R12 but with P addition, higher COD removals would most probably be obtained. Despite R4-R6 lower OLR ranges (2.8-6.8 g $COD_{(soluble)}$ L_r^{-1} d^{-1} versus 5.2-10.0 g $COD_{(soluble)}$ L_r^{-1} d^{-1} for R10-R12), spring runs (R4-R6) achieved lower removal efficiencies (91-96%) compared to summer runs (R10-R12) (94-98%) (Figure 6.1). It is believed that this difference was partly related the shorter HRTs but most likely also due to surplus P and better acclimation of the biomass for runs R10-R12 than for runs R4-R6.

6.3.2.c COD results analysis and predictions

From Figure 6.1 linear regression curves describing COD removal efficiency as a function of OLR for summer, fall and spring conditions (leachate strength) can be developed. Linear regression provides a simple solution for establishing the relation between organic loading versus removal. The objective is to establish a general relationship between OLR and COD removal within the experimental range studied, and is not an attempt to produce a rigorous model of the reactors performance. Therefore these equations are specific for the range of OLR, the biomass inventory, waste studied and reactor conditions.

Table 6.4: COD removal versus OLR

Season	Waste concentration mg L^{-1}	Biomass range g VSS L^{-1}	OLR range g COD L_r^{-1} d^{-1}	COD removal equation*	R^2
Spring	5 839 ± 580	35-42	2.9-6.5	y = –1.3 x + 100	0.9992
Summer	16 261 ± 594	30-48	2.1-4.7	y = –0.9 x + 103	0.9525
Fall	6 521 ± 845	30-48	5.2-10.0	y = –0.4 x + 99	0.8126
Winter	—	—	—	—	—

*Linear regression: y is % COD removal; x is OLR (g COD L_r^{-1} d^{-1})

Table 6.4 shows the waste concentration, biomass inventory, and the calculated linear equations for COD removal efficiency for each season as a function of OLR. Figure 6.1 and Table

6.4 indicate that higher removal efficiency is to be expected at similar volumetric loading rates if the effluent is more concentrated.

At low waste concentration (spring), it is hypothesised that the high hydraulic factor (low HRT) caused a more pronounced decline of COD removal efficiency than for other conditions; insufficient contact time between substrate and microorganisms might be causing the steepness of the slope (−1.3). Best results seems to occur for medium concentration and medium hydraulic factor conditions (fall: slope of −0.4); sufficient contact time is allowed and the waste concentration is less inhibitory to the microorganisms than at higher concentrations such as summer. It should be noted however, that the fitting of the curve is imprecise and comparing a curve with the others should be made with prudence. For the most concentrated waste (summer), the steeper slope (-0.9) of the curve of COD removal versus OLR might be related to the higher concentration of toxic substances that have inhibitory effect on microorganisms.

From these results, it was concluded that, despite good COD removals, reactors running under more extreme conditions, such as high flow rates or high feed concentration, seem to be more fragile to increases in loading rates than reactors running at moderated conditions. However, this hypothesis should be verified by increasing the number of runs for each condition, therefore increasing the number of data points.

6.3.2.d Risk zone for exceeding SUB limit

The combination of a UASB reactor and peat moss filter was a successful means of treatment for CRL leachate under each simulated seasonal condition and for almost all conditions produced effluent within the SUB COD discharge limit. It is possible that higher OLR could potentially be achieved: Myburg and Britz (1993) achieved removals of 84.5-95% with loading rates of 21.65 to 29.0 g COD L_r^{-1} d^{-1} using a UASB-AF reactor but none of their average effluent COD values would have complied with RMOC SUB limits. Therefore, while higher organic loading rates were possible in the present study, it was concluded that higher OLR might result in an unacceptable risk zone resulting in a process that could not meet discharge limits.

It is suggested that OLR be kept below 10 g COD L_r^{-1} d^{-1} for full scale operating conditions in order to maintain a reasonable OLR below the risk zone (Figure 6.2). If pre-treatment of leachate with peat is carried out, it is also recommended that P be monitored to ensure that P limited conditions do not occur. And if P limitation occurs, P addition should be instituted to ensure a healthy anaerobic flora, long term reactor stability and achievement of compliance with the sewer discharge limit are anticipated.

Figure 6.2: VRR versus OLR

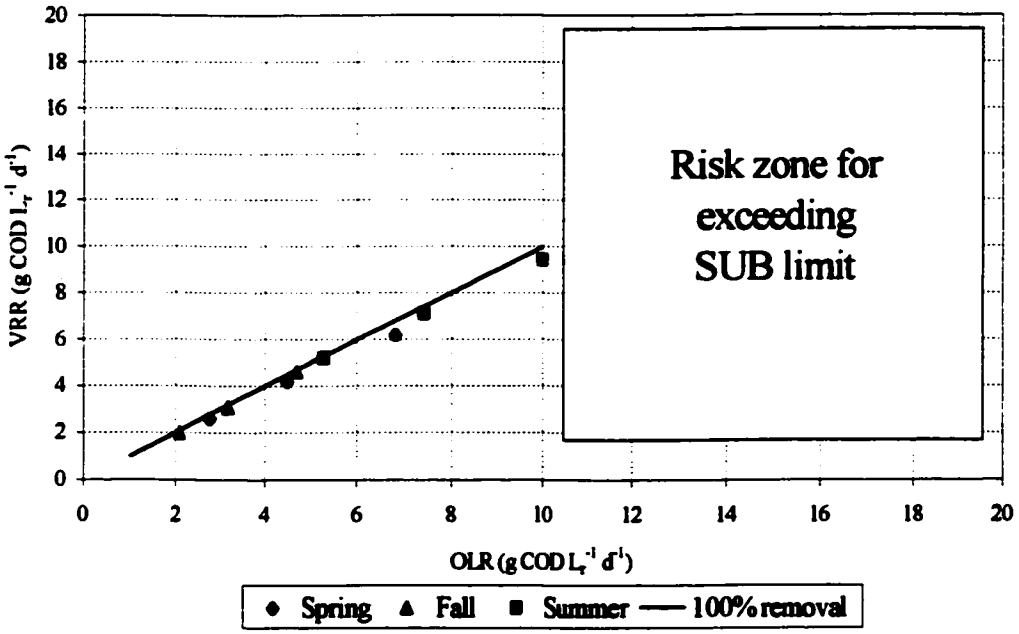
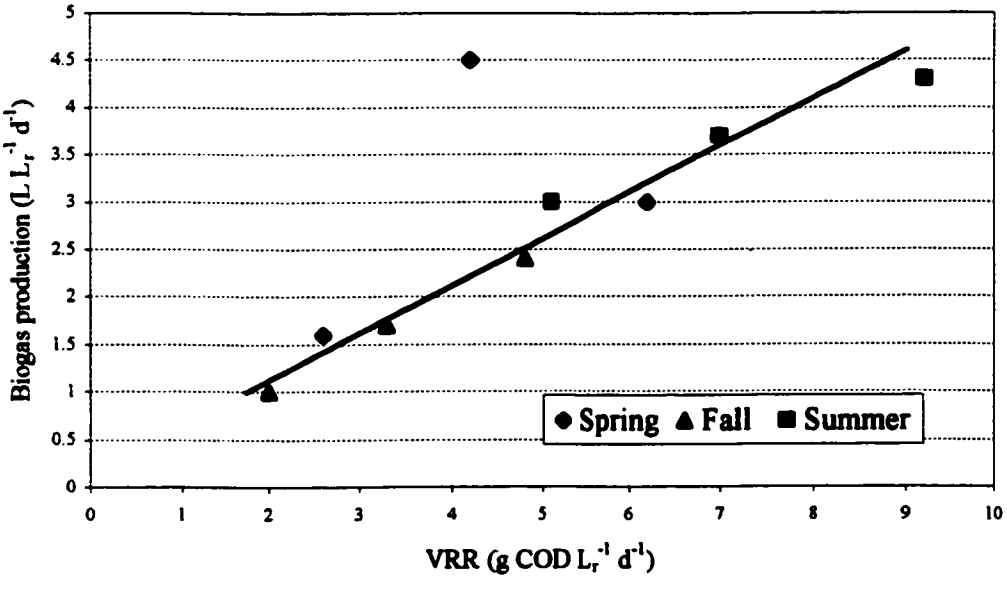


Figure 6.3: Biogas production versus VRR



6.3.3 BIOGAS PRODUCTION

Table 6.5 discloses biogas productions and calculated methane fractions for runs R4 to R12. Biogas production was, as expected, increasing with increasing VRR (Figure 6.3). One data point was apart from the general trendline: R5, data point 4.2 g COD L_r^{-1} d^{-1}: 4.5 L_g L_r^{-1} d^{-1}. R5 experienced a Wet Tip Meter failure and was not representative of the actual gas production; it was therefore discarded from biogas production calculations.

Table 6.5: Biogas production and methane fraction runs R4-R12

	Average biogas Production L_g L_r^{-1} d^{-1}	Average VRR g COD L_r^{-1} d^{-1}	Calculated methane fraction[1] %
R2	2.4	5.1	76
R4	1.5 ±0.2	2.6	62
R5	3.9 ± 0.6	4.2	39
R6	2.9 ± 0.5	6.2	77
R7	1.0 ± 0.2	2.0	72
R8	1.6 ± 0.2	3.3	74
R9	2.3 ± 0.3	4.8	75
R10	2.9 ± 0.4	5.1	63
R11	3.7 ± 0.3	7.0	68
R12	4.1 ± 0.3	9.2	81
Average			72

1- R2 is measured; assumed f_m = 0.92

A linear regression calculation was made with Microsoft® Excel 97 on Figure 6.3 (biogas production versus VRR) data points; a slope of 0.44± 0.04 L biogas produced per g COD removed was found. Hence, with the collected fraction from R1-R3 (f_m = 0.92), the calculated corrected biogas production was approximately 0.48 L biogas per g COD removed. Consequently, the methane production, calculated with the average methane fraction of 72% from Table 6.5, was calculated to be 0.35 L methane per g COD removed. Calculated methane production was low compared to the theoretical methane production at 32°C, 0.39 L methane per g COD removed. Various sources of error could have influenced this result; calibration of gas meter, feeding line maintenance interruptions, pumps failures, inaccuracy of f_m (0.92) and average methane fraction of 72%.

Since biogas composition could not be measured directly, biogas methane fractions were approximated with equation 2.5.2 (Table 6.5), assuming therefore the ideal gas law being valid and the total pressure inside the reactors to be 1 atmosphere. The methane fractions were calculated

using average biogas volumetric productions ($L_g\ L_r^{-1}\ d^{-1}$), average COD volumetric removal rate (g COD $L_r^{-1}\ d^{-1}$), a methane collected fraction of 92% (from R1-R3), a temperature of 32°C (305K) and a total pressure of 1 atmosphere. R6-R9 methane calculated fractions were similar to observed fractions in preliminary results, R4, R10 and R11 were however lower whereas R12 was higher. In order to be conclusive, further studies should be made on biogas to assess accurate production and composition of biogas from anaerobic treatment of CRL leachate.

6.3.4 BIOMASS ACTIVITY

6.3.4.a Acetoclastic activity test

The final biomass activities between the fall and the summer runs were not measured; the overall change of acetoclastic activity, after running combined fall and summer conditions, was analysed as R7', R8', and R9'. For R4 to R9' batch acetoclastic activity tests (AAT), all samples were conducted with excess acetate and the rates measured were the maximum acetate consumption rates. Maximum rates were not reached for runs R1-R3, because of low initial acetate concentrations (see section 4.3.4). For runs R4-R9', initial acetate concentrations were higher than 800 mg L^{-1} and therefore AATs were assumed not to be substrate-limited. Acetoclastic activity test for runs R4-R9' tests were all conducted in duplicate samples from the top and bottom parts of the anaerobic granular sludge bed. Measured specific acetoclastic activity rates fell within the range reported for various complex wastewaters for mixed anaerobic cultures, 0.19 to 0.80 g COD g^{-1} VSS d^{-1} (Table 6.6).

As expected, activities differed according to the sampling location. Samples for the bottom part of the granular sludge bed, closer to the feed inlet and therefore exposed to higher substrate concentrations, had in most cases (initial and final AATs) a higher specific acetoclastic consumption rate than sludge samples for the top of the sludge bed (Table 6.6). For instance, for runs R4, R7' and R8' the bottom biomass samples tested at the end of the run had a consumption rate at least 0.1 g COD g^{-1} VSS d^{-1} greater than sludge sampled at the top of the bed.

Run R4 had the largest disparity between top sample activity (0.19 g COD g^{-1} VSS d^{-1}) and bottom sample one (0.39 g COD g^{-1} VSS d^{-1}). Run R4 also had the lowest SLR (0.14 g COD g^{-1} VSS d^{-1}) and it was assumed that the top portion of the sludge bed was not exposed to substrate or was exposed to low concentrations compared to the biomass in the lower portion of the sludge bed.

Table 6.6: Initial and final AAT and SLR for runs R4-R9', per g VSS

	Initial Top	Initial Bottom	Final Top	Final Bottom	SLR (soluble)
	g COD g^{-1} VSS d^{-1}	g COD g^{-1} VSS d^{-1}	g COD g^{-1} VSS d^{-1}	g COD g^{-1} VSS d^{-1}	g COD g^{-1} VSS d^{-1}
	Initial acetate: 2 003 ± 36 mg L^{-1}		Initial acetate: 2 217 ± 32 mg L^{-1}		22 d
R4	0.41	0.52	0.19	0.39	0.14
R5	0.41	0.52	0.61	0.62	0.23
R6	0.41	0.52	0.66	0.65	0.29
	Initial acetate: 1 761 ± 15 mg L^{-1}		Initial acetate: 2 095 ± 47 mg L^{-1}		64 d
R7'	0.99	1.12	0.50	0.61	0.12 – 0.32
R8'	1.25	1.30	0.69	0.80	0.21 – 0.43
R9'	1.00	1.12	0.67	0.55	0.23 – 0.40

R5 and R6 had moderate SLRs (0.23 and 0.29 g COD g^{-1} VSS d^{-1} respectively) and yielded a final uniform activity profile throughout their sludge bed; combined top and bottom final activities were averaging 0.62 and 0.65 g COD g^{-1} VSS d^{-1} respectively. This uniformity might also be related to low HRTs (1.5 and 0.9 d R5 and R6 respectively) that produce higher flow rates and upflow velocities, resulting in an increased mixing and a more homogeneous substrate distribution. Therefore, lower HRT should promote improved substrate distribution and uniformity of biomass activity in the sludge bed.

R7' and R8' bottom biomass samples had higher acetoclastic activities relative to top samples (Table 6.6). Compared to R5 or R6, R8' was under a higher stress; R8' SLRs ranged from 0.21 to 0.43 g COD g^{-1} VSS d^{-1} whereas R5 and R6 SLRs were averaging 0.23 and 0.29 g COD g^{-1} VSS d^{-1}, respectively. R8' acetoclastic final activity profile was higher (0.69 and 0.80 g COD g^{-1} VSS d^{-1} top and bottom respectively) relative to R5 and R6 final activity profiles. Therefore, bacteria in runs R5 and R6 were neither at their maximum acclimation capacity nor their maximum potential substrate consumption capacities compared to R8'.

Table 6.7: Initial and final AAT and SLR for runs R4-R9', per g TSS

	Initial Top	Initial Bottom	Final Top	Final Bottom
	g COD g^{-1} TSS d^{-1}	g COD g^{-1} TSS d^{-1}	g COD g^{-1} TSS d^{-1}	g COD g^{-1} TSS d^{-1}
	Initial acetate: 2 003 ± 36 mg L^{-1}		Initial acetate: 2 217 ± 32 mg L^{-1}	
R4	0.31	0.26	0.14	0.28
R5	0.31	0.26	0.48	0.44
R6	0.31	0.26	0.48	0.41
	Initial acetate: 1 761 ± 15 mg L^{-1}		Initial acetate: 2 095 ± 47 mg L^{-1}	
R7'	0.78	0.49	0.36	0.35
R8'	0.94	0.80	0.48	0.30
R9'	0.75	0.53	0.38	0.10

R9' was the only run where higher specific acetoclastic activities were observed in samples from the top of the sludge bed; the final acetoclastic activity profile was 0.55 g COD g^{-1} VSS d^{-1} for bottom sludge and 0.67 g COD g^{-1} VSS d^{-1} for top sludge. This observation was related to inert solids accumulation in the bottom sludge (see section 6.3.5) resulting in a decrease of biomass concentration and activity. The bottom biomass had a final TSS activity of 0.10 g COD g^{-1} TSS d^{-1} (Table 6.7), dropping from an initial activity of 0.53 g COD g^{-1} TSS. Therefore biomass displacement had occurred in R9' after long-term operation, resulting in a decrease of the overall reactor activity.

Higher HRTs (3.1-3.4 d for R7' and 2.0-2.2 d for R8') with higher SLRs were concluded to result in uniformly higher activities distributed throughout the sludge bed; both factors must therefore be weighted against each other before setting standard operating conditions.

6.3.4.b Activity of total solids

Significant differences were noted between top and bottom VSS activities versus TSS activities (Table 6.6 versus Table 6.7). R5, R6, R8, and R9 top samples had higher consumption rates per g TSS compared to their respective bottom samples, despite the fact that R5 and R6 had homogeneous VSS activities and R8 had a higher bottom VSS activity. This observation was concluded to be a consequence of higher fixed suspended solids (FSS) concentrations in bottom samples relative to top samples. Hence, bottom samples had greater VSS activities but VSS contributed to a small weight fraction of total granules.

VSS removal rate efficiency should be weighted against TSS efficiency before commenting on overall reactor removal capacity. In R9 case, the biomass volatile activity appeared to be satisfactory (0.61 g COD g^{-1} VSS d^{-1}) whereas the actual total activity was in fact low (0.24 g COD g^{-1} TSS d^{-1}). As a general rule, after the acclimation period a reactor should have a total activity above 0.20 g COD g^{-1} TSS d^{-1}, otherwise it is suggested that the biomass be replaced with fresh, less calcified biomass. A periodical removal of high-inert material containing granules would be essential in order to maintain a satisfactory total activity. An alternative option to periodical removal would be the improvement of pre-treatment removal efficiency of potentially accumulating inert elements in the leachate.

6.3.4.c Specific removal rate

Reactors specific removal rates (SRR) (Table 6.8) were compared with specific loading rates (SLR) and AAT results to quantify the UASB reactor capacity to treat CRL leachate. Biomass from R4 to R10 runs were concluded not to be at maximum treatment capacity because the applied SLRs were below the maximum SRR found in R1 of 0.42 g COD g^{-1} VSS d^{-1}. R11 and R12 were at or near the maximum assumed capacity with SRRs of 0.42 and 0.38 g COD g^{-1} VSS d^{-1}, respectively.

Table 6.8: Removed loading rates, specific loading and removal rates runs R4-R12

	OLR (soluble COD) (g COD L_r^{-1} d^{-1})	VRR (sol) (soluble COD) (g COD L_r^{-1} d^{-1})	SLR (sol) (soluble COD) (g COD g^{-1} VSS d^{-1})	SRR (sol) (soluble COD) (g COD g^{-1} VSS d^{-1})
R4	2.9 ± 0.35	2.6	0.14	0.13
R5	3.9 ± 0.61	4.2	0.23	0.21
R6	6.5 ± 1.2	6.2	0.29	0.26
R7	2.1 ± 0.4	2.0	0.12	0.12
R8	3.0 ± 0.4	3.3	0.21	0.21
R9	4.7 ± 0.6	4.8	0.23	0.22
R10	5.2 ± 0.5	5.1	0.32	0.31
R11	7.4 ± 0.5	7.0	0.43	0.42
R12	10.0 ± 0.6	9.2	0.40	0.38

R4 to R10 were submitted to SLR lower than 0.42 g COD g^{-1} VSS d^{-1} (from R1), ranging from 0.14 to 0.32 g COD g^{-1} VSS d^{-1} and achieved SRR ranging from 0.13 to 0.31 g COD g^{-1} VSS d^{-1} (Table 6.8). The SRR had invariably a difference of 0.01 to 0.03 g COD g^{-1} VSS d^{-1} with the SLR, this difference being related to the approximate 2% non-biodegradable fraction of CRL leachate.

SLR for run R11 was approximately (0.43 g COD g^{-1} VSS d^{-1}) at the maximum removal rate set by R1 whereas R12 was slightly below it with a SLR of 0.40 g COD g^{-1} VSS d^{-1}. They both achieved high SRRs of 0.42 and 0.38 g COD g^{-1} VSS d^{-1}, respectively. Because P may have been limiting in run R1 and because acclimation was probably greater for R10-R12 biomass compared to R1, the actual maximum capacity of R11 and R12 might be even higher than 0.42 g COD g^{-1} VSS d^{-1}. This conclusion in part is supported by Figure 6.4, which shows that the SRR data points were close to the 100% removal line and no maximum plateau was observed. Moreover, the treatment efficiency was high (R11 had an efficiency of 97% and R12 of 94%) and VFAs low, confirming that maximum removal capacity on CRL leachate was not reached or

Figure 6.4: SRR versus SLR

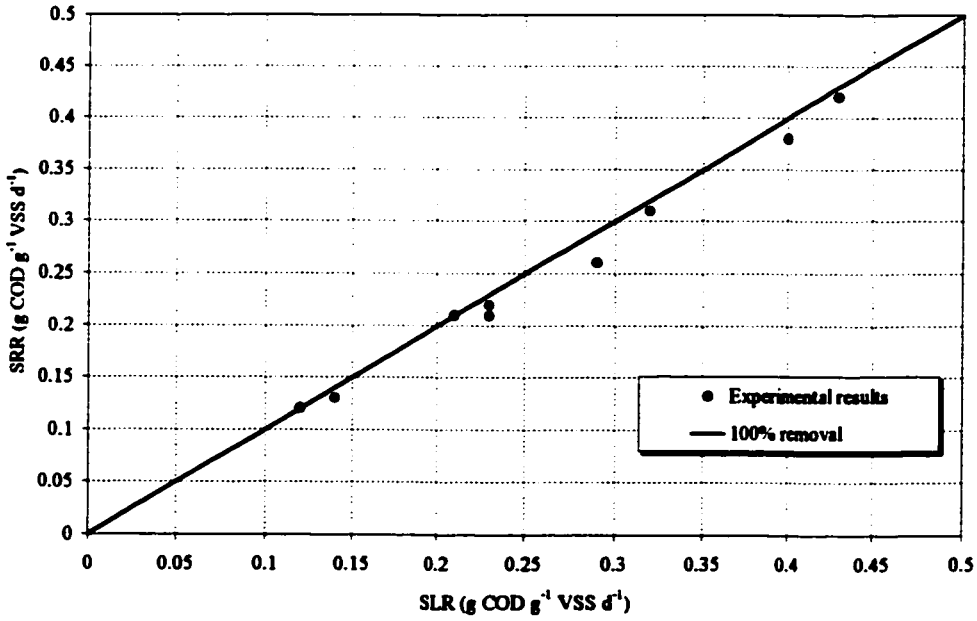
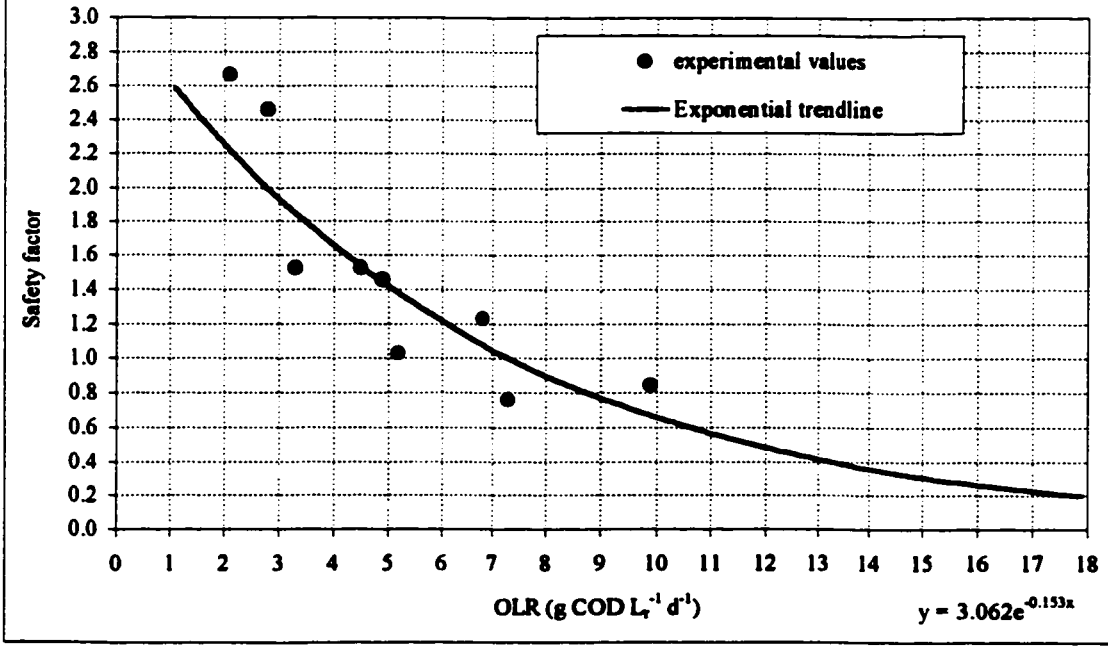


Figure 6.5: Specific loading rate based safety factor versus OLR
for CRL leachate treatment



exceeded. A higher SLR could have been achieved and possibly reaching a maximum possible activity but the SRR would maybe not have been sufficient to produce an effluent complying with SUB discharge limit; the bylaw limit was the real restrictive factor.

A maximum activity of 0.80 g COD g^{-1} VSS d^{-1} was measured (R8' AAT) but such a high activity was not expected to be obtained while digesting CRL leachate; the complexity of CRL leachate organic substrates compared to acetate was expected to yield a lower specific digestion rate. When comparing AAT results versus SRRs, a relation could be established; SRR:AAT ratio (averaging top and bottom results) was approximately 1:2. The rate of acetate consumption was therefore two folds faster relative to the rate of CRL leachate substrate consumption in the UASB reactors.

6.3.4.d Biomass activity summary conclusions

In summary, spring conditions favoured homogeneous biomass activity throughout the UASB reactors at low HRTs (1.5 and 0.9 d). Moderate SLR (higher than 0.2 g COD g^{-1} VSS d^{-1}) should be selected instead of low SLR for diluted leachate. Fall and summer conditions are susceptible to FSS accumulation because of higher feed concentrations, which may potentially lead to biomass displacement. Therefore, SLR less or equal to 0.40 g COD g^{-1} VSS d^{-1} are advised when no complementary removal of soluble inert elements of filtered leachate would be used. Increasing the recycle-to-feed ratio could be used to increase upflow velocities, thus mixing in reactors, and hopefully a uniform substrate distribution and a more uniform acetoclastic activity profile could be obtained.

Interestingly, higher activities were measured after the longest storage period (R7'-R9', initial AAT) than in any other test. This situation was unexpected and this phenomenon may be interesting to study in subsequent research.

Final AAT bottom samples consumption rates (excluding R4) ranged from 0.55 g COD g^{-1} VSS d^{-1} for R9', to 0.80 g COD g^{-1} VSS d^{-1} for R8. The hypothetical maximum attainable activity for acetate digestion was therefore set to 0.80 g COD g^{-1} VSS d^{-1} and the maximum SLR for CRL leachate using the ratio 1:2 was 0.40 g COD g^{-1} VSS d^{-1}. This value was used to calculate SLR safety factors for the treatment of CRL leachate by a UASB reactor (Figure 6.5).

The SLR safety factor was determined from the maximum attainable activity of anaerobic flora fed on CRL leachate and the actual achieved SRR for R4 to R12 inclusively versus SLR (Figure 6.5). A general trendline, which roughly connected experimental data points, was plotted

using Excel® data series format options. The exponential trendline yielded a good fitting curve; nevertheless it is again not an attempt to give a precise model but to establish a general relationship between experimental data points. This trendline predicts a safety factor higher than 1.0 for OLR values below 0.40 g COD g^{-1} VSS d^{-1}. At a SLR of 0.5 g COD g^{-1} VSS d^{-1}, the calculated safety factor would be 0.67, and at a SLR above 0.58 g COD g^{-1} VSS d^{-1}, the safety factor would fall below 0.5. Therefore, in order to keep a conservative safety margin it is suggested that SLRs should be less than 0.4 g COD g^{-1} VSS d^{-1}.

6.3.5 SOLIDS ACCUMULATION

6.3.5.a Reactor solids profiles

Volatile and fixed suspended solids were measured in each reactor from two sampling locations to establish initial and final solids profiles. Prior to runs R4-R6, only one reactor was initially fully characterized and the others were assumed to have similar solids profiles. For R7-R9 and R10-R12, solid profiles were measured, as for the activity test, before fall conditions and after summer conditions; solids content was therefore studied over the two periods, identified as R7', R8', and R9'. Reactor solids content varied significantly from the beginning to end of a run and despite peat pre-treatment, there was a significant fixed solids accumulation.

In R4-R6, there was an overall VSS fraction increase from 62% to 68-74% (Table 6.9 and Figure 6.6), indicating inert solids accumulation was less than microbial growth, result of influent SS removal by peat filtration. VSS percent fraction increase was more important in bottom samples than top samples (Figure 6.6), showing healthier conditions and washout of high-inert content granules. On the other hand, there was an overall TSS concentration decrease in sludge beds, from 67 g TSS L^{-1} to 48-62 g TSS L^{-1}. Sludge bed VSS concentrations had also decreased (Table 6.8) from 40 g VSS L^{-1} to 36 and 35 g VSS L^{-1} for R4 and R5 respectively but increased slightly for R6 to 42 g VSS L^{-1}. VSS concentration remained stable in all bottom fractions but was lower in R4 and R5 top sludge bed samples (Figure 6.7) whereas VSS was generally stable for R6.

TSS concentrations decrease was attributed to washout of small diameter and high FSS content granules for the beneficial development of lower density but larger granules composed of structured anaerobic consortia. R6 had the highest SLR of spring condition simulation run and had also the highest final VSS concentration. Higher substrate loading (6.5 g COD L_r^{-1} d^{-1}) was concluded to stimulate biomass growth and promoted higher VSS concentrations in reactors.

Figure 6.6: SS fraction in runs R4, R5 and R6

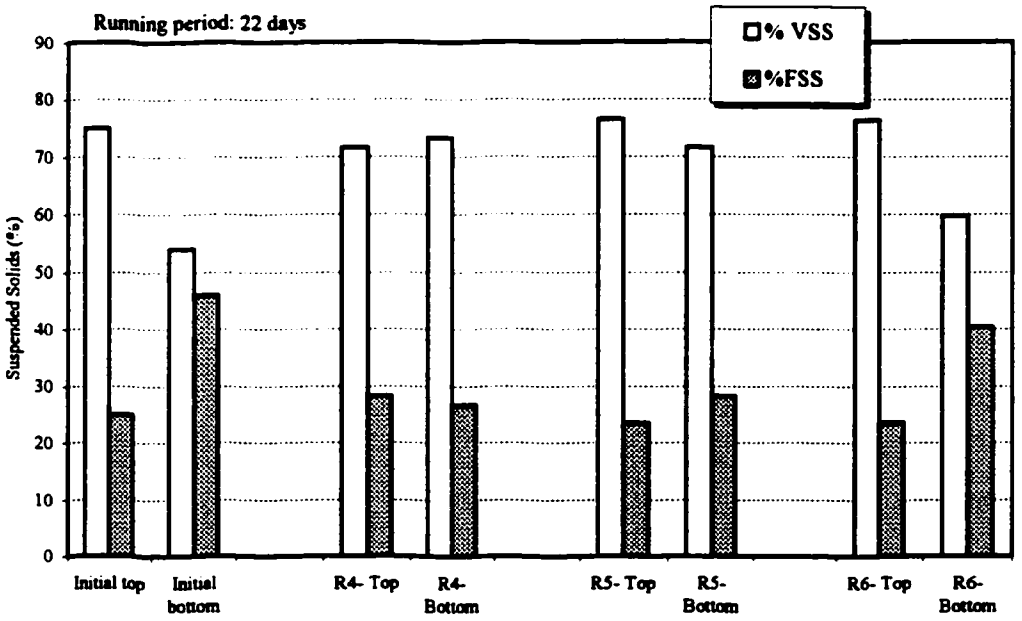
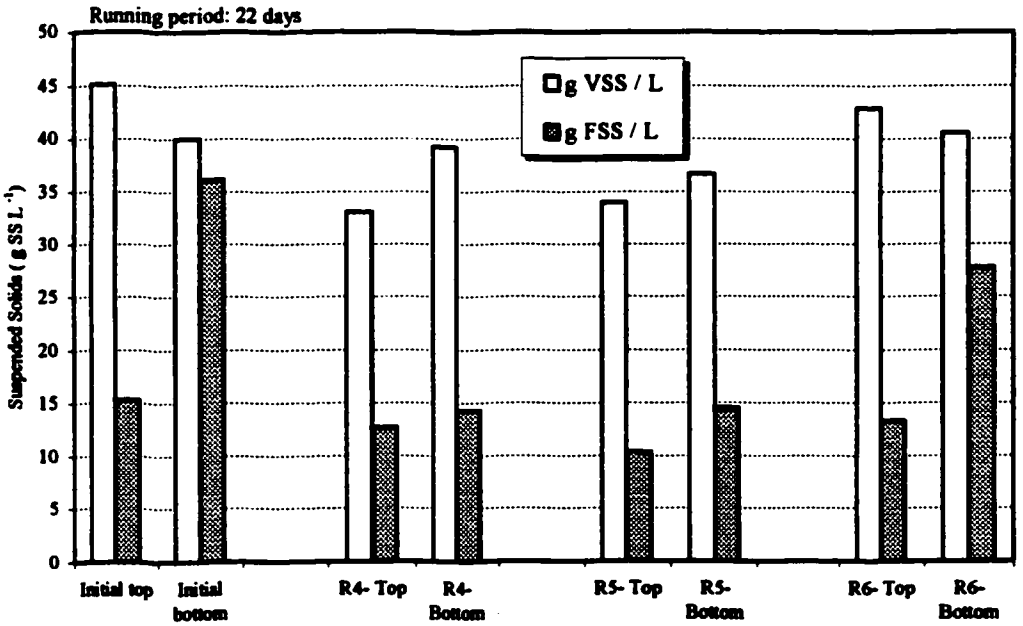


Figure 6.7: Sludge bed SS concentration in R4, R5 and R6



However, R6 also had the highest final FSS concentration and could therefore still be susceptible to bed calcification despite peat filtration. The pre-treatment of CRL leachate with a peat filter was concluded to promote biomass growth and FSS washout but at high SLRs, the possibility of biomass calcification problems still existed

In runs R7' to R9' solids profile changes were significantly (Table 6.9, Figures 6.8 and 6.9). R7' changes were inverted compare to R8' and R9'; R7' VSS and TSS concentrations had decreased over 64 operating days whereas R8' and R9' concentrations increased, and R7' % VSS fraction increased whereas it had decreased for R8' and R9'. Moreover, there was an important fixed solids accumulation in R8' and R9'.

R7' SS concentrations decreased from 35 to 30 g VSS L^{-1} and 64 to 47 g TSS L^{-1} but the VSS fraction increased from 55% to 63%. R7' was the least stressed reactor during fall and summer simulations (SLR of 0.12 and 0.32 g COD g^{-1} VSS L^{-1}). It was concluded that milder operating conditions prevented inert solids accumulation but induced solids concentration decreases.

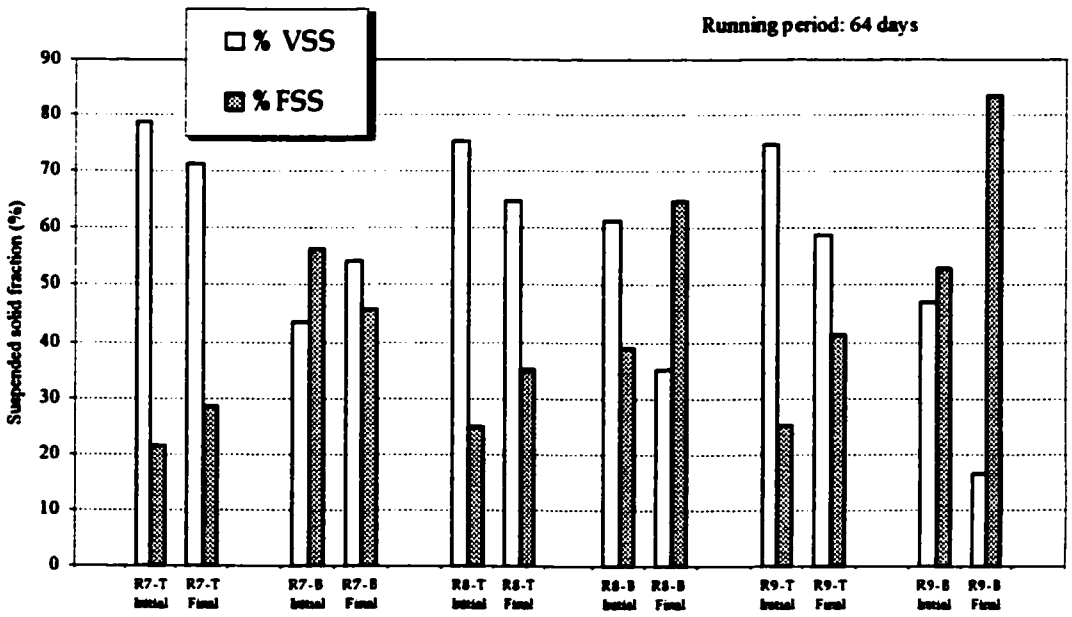
Table 6.9: Solids content variation runs R4-R9'

	Initial			Final (22 days)		
	g VSS L^{-1}	g TSS L^{-1}	% VSS	g VSS L^{-1}	g TSS L^{-1}	% VSS
R4	40	67	62	36	50	73
R5	40	67	62	35	48	74
R6	40	67	62	42	62	68
	Initial			Final* (after 64 days)		
R7'	35	64	55	30	47	63
R8'	30	45	67	35	69	50
R9'	35	60	58	48	129	38

R8' SS concentrations increased from 30 to 35 g VSS L^{-1} and from 45 to 69 g TSS L^{-1}; however, VSS overall fraction decreased from 67 to 50% (Table 6.9). R8' was operated at medium level SLRs in both fall and summer conditions (0.12 to 0.43 g COD g^{-1} VSS d^{-1}) that resulted in significant inert solids accumulation. Run R8' bottom sample FSS concentration was higher than 100 g FSS L^{-1} after 64 operation days. Therefore, inert solids were accumulating despite peat pre-treatment and increased biomass growth.

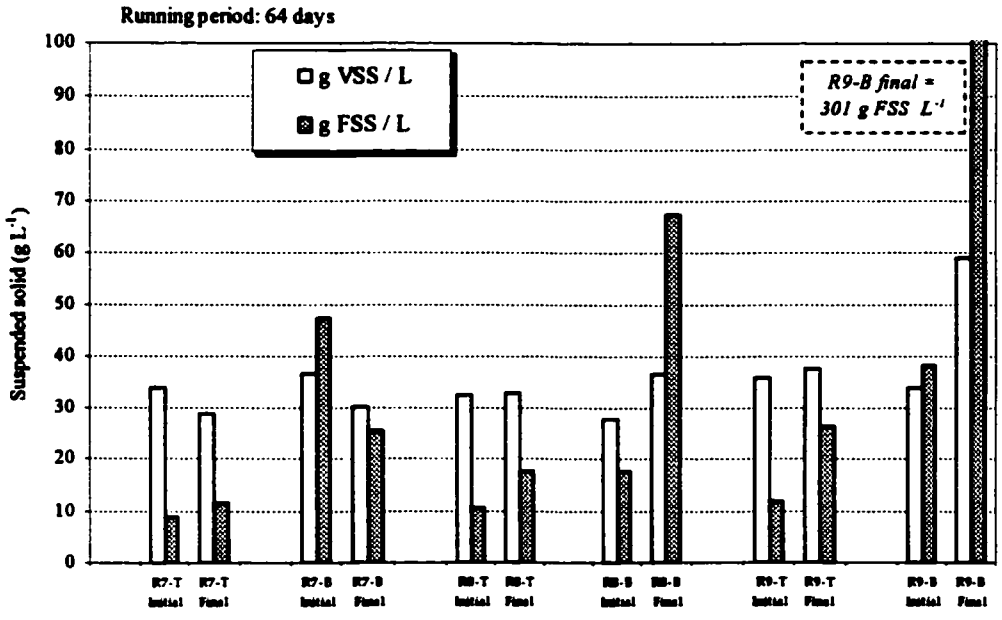
There was a significant biomass displacement in the bottom of R9' sludge bed. After 64 days of operations, the R9' VSS fraction dropped from an initial 58% to a final fraction of 38% (Table 6.9). Sludge from the bottom of R9' was essentially composed of inert solids (83% FSS) at the end of combined fall and summer runs (Figure 6.8). R9' SS concentrations increased from 35

Figure 6.8: SS profile in R7', R8' and R9'



T = top B = bottom

Figure 6.9: Sludge bed SS profile in R7', R8' and R9'



T = top B = bottom

to 48 g VSS L^{-1} and from 60 to 129 g TSS L^{-1}. Despite inert solids accumulation, R9' bottom sludge samples had the greatest VSS concentration increase: from 34 to 59 g VSS L^{-1} (Figure 6.9). It was therefore concluded that high loading rates (R9' SLRs were 0.23 to 0.40 g COD g^{-1} VSS d^{-1}) promoted biomass growth though although severe inert solids accumulation also occurred.

6.3.5.b Rate of solids accumulation

Yields of solids accumulation and rates for combined fall-summer conditions were approximated for each reactor based on the COD fed over the 64-day test period and compared to preliminary study results (Table 6.10).

In preliminary studies (Section 4.3.5) there was an overall negative accumulation (*i.e.* washout) of VSS whereas there was significant FSS accumulation: from –0.33 to –2.58 g VSS kg^{-1} COD_{fed} and from 1.74 to 5.39 g FSS kg^{-1} COD_{fed} respectively. Accumulated solids were assumed to accumulate evenly over the operating period. An approximated accumulation rate (corrected accumulation, Table 6.10) over the running period was calculated. It was understood that solid variation over a running period might not be linear, but for establishing a rough estimation of reactor behaviour, it is assumed to be a reasonable method of approximation.

Table 6.10: Solids accumulation R1-R3 and R7-R9

	Yield of solid accumulation (measured) over the running period		Yield, for non-biomass FSS*	Corrected accumulation over the running period	
	g VSS kg^{-1} COD fed	g FSS kg^{-1} COD fed	g FSS' kg^{-1} COD fed	g VSS kg^{-1} COD fed d^{-1}	g FSS' kg^{-1} COD fed d^{-1}
Running time: 20 d					
R1	–2.58	5.39	4.3	–0.13	0.27
R2	–1.87	1.74	1.4	–0.09	0.09
R3	–0.33	3.49	2.8	–0.02	0.17
Running time: 64 d					
R7'	–1.19	–2.10	–0.13	–0.02	–0.03
R8'	0.61	3.91	–0.09	0.01	0.05
R9'	1.59	16.2	–0.08	0.03	0.20

* FSS' are FSS without FSS fraction (assumed to be 20%) associated with biomass

The rate of solids accumulation (corrected) for R1-R3 varied from –0.02 to –0.13 g VSS kg^{-1} COD_{fed} for volatile solids and from 0.09 to 0.27 g FSS kg^{-1} COD_{fed} for inert solids (Table 6.10). The rate of accumulation for combined fall-summer test conditions varied from -0.02 (R7) to 0.03 (R9) g VSS kg^{-1} COD_{fed} for volatile solids and from -0.03 (R7) to 0.20 (R9) g FSS kg^{-1} COD_{fed} for inert solids. From these rates, it was concluded that keeping high rates such as R9'

promoted biomass growth but to the expense of severe inert solids accumulation, as if no peat filtration was performed. However, moderate conditions such as R8' promoted a positive but low biomass accumulation rate and positive and also low inert solids accumulation.

Keeping moderate R8' conditions would inevitably imply slower treatment, which would result in the requirement for a large reactor volume to treat the same leachate production compared to R9' conditions. On the other hand, R9' would inevitably require more frequent periodical biomass replacement compared to R8'. The merits of higher capital costs versus higher operation costs would have to be carefully considered to which option is suitable for commercial viability.

A third but unexplored option would be to decrease the raw leachate calcium content prior to feeding to reactors. As discussed in the next section, calcium precipitated in significant quantities in reactors, and perhaps is playing an important role in inert solids accumulation, even more than SS present in CRL leachate. Softening raw leachate with lime prior to filtering it with peat would be an interesting and possibly viable option to readily overcome rapid calcification of the anaerobic granular sludge.

6.3.6 METALS REMOVAL BY UASB REACTOR

During fall and summer simulation conditions, the removal of selected inorganic elements by UASB reactors was sporadically monitored. As discussed in the previous section 6.3.5, inorganic elements were precipitating in reactors, giving rise to undesirable inert solids accumulation. However precipitation of inorganic solids in UASB reactors could be regarded as an efficient removal mechanism of soluble metals from treated wastewater.

The removal of metals from CRL leachate was one of the objectives of this present research: specifically the removal of boron, barium, and iron. Therefore, any stage in the whole treatment contributing to metal removal including during biological treatment should be regarded as a means of accomplishing this objective. Adding to the mandatory elements above, removal efficiencies of metals discussed during peat filtration were also analysed: aluminium, calcium, cobalt, manganese, and zinc. Most of the studied elements were removed with high efficiencies in the UASB reactor.

One of the two peat filters used for leachate pre-treatment might have experienced channelling problems; elements such as aluminium or boron were found in high concentrations in the influent fed to the UASB although they were importantly removed by peat filtration. Therefore, UASB reactor metal removal results were unexpectedly measured under problematic

experimental conditions: a leak in peat filters or channelling problems. The impact on the study is however positive; the efficiency of the UASB reactor for metal removal could be measured on higher metal concentrations.

6.3.6.a Removal of selected inorganic element: aluminium

Aluminium concentration was expected to be undetectable in feed leachate because it was completely removed from raw leachate by peat filtration (Chapter 5). However, as explained above, the feed leachate used was pre-treated by a peat filter afflicted with serious channelling problems. In the three measurements wherein aluminium concentration was detected, aluminium was completely removed by reactors from the leachate (Figure 6.10). Aluminium was detected in influent in concentrations varying from 0.02 to 0.12 mg L^{-1} but had never been detected in effluent samples.

Aluminium was, as expected, below the POTW discharge limit in the final UASB effluent; it was removed to undetectable levels by combination of the peat filter and UASB reactors. Moreover, aluminium discharge limit (50 mg L^{-1}) was higher than all aluminium measurements made on the raw leachate soluble effluent. The only possible way of exceeding the SUB limit would be to directly discharge raw CRL leachate in to the RMOC sewer system; aluminium concentration was found to be the highest in SS, which were removed mainly by peat filtration.

6.3.6.b Removal of selected inorganic element: boron

Despite excellent boron removal with peat (98%), a high concentration of boron was fed to reactors, most probably because of peat filter channelling problems. As shown in Figure 6.11, there was little or no boron removal by UASB reactors: a maximum of 14% was removed whereas a maximum increase of 33% based on the initial concentration was measured on December 15th.

UASB reactors were concluded to be inefficient for boron removal and therefore boron should be removed below SUB discharge limits by peat filtration.

6.3.6.c Removal of selected inorganic element: barium

Removal of barium by the peat filter was poor (Section 5.5.3) but fortunately barium removal by UASB reactors was quite efficient; a maximum removal of 87% was measured in the UASB on January 19th (Figure 6.12). Increases in effluent concentration were observed at the two last data points, which were the least reliable because these results occurred when the feed

Figure 6.10: Aluminium in and out of reactors

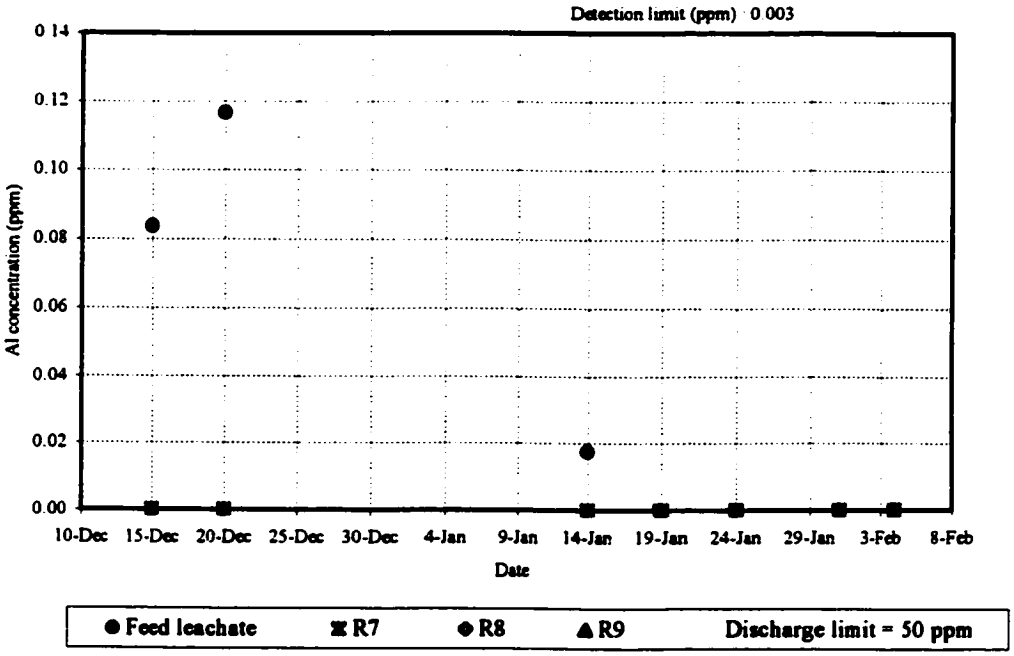
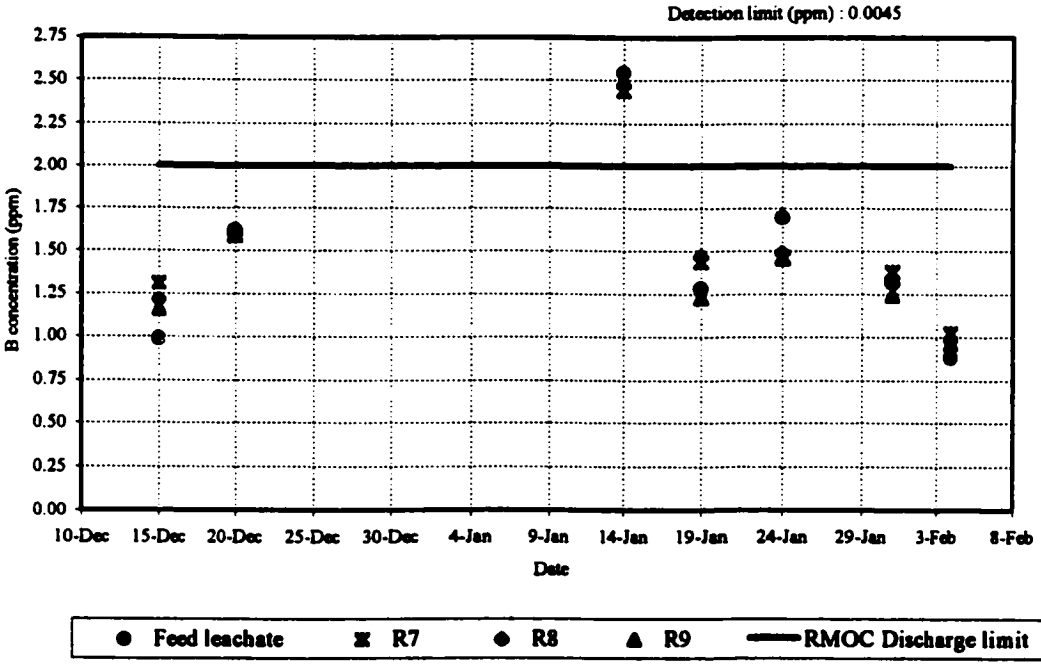


Figure 6.11: Boron in and out of reactors



concentration was at its lowest (0.05 mg Ba L^{-1}) and measurement error was highest. When the influent feed concentration exceeded 0.30 mg Ba L^{-1}, the minimum barium removal achieved was 68% (occurred on December 20th).

The overall barium removal by UASB reactors was high and final effluent concentrations did not exceeded 0.20 mg L^{-1}, whereas the influent concentrations ranged from 0.44 to 0.67 mg L^{-1}. Despite a low discharge limit (1.2 mg L^{-1}), the effluent quality was always at least 1.0 mg L^{-1} below the limit. Barium removal was variable for peat, but good for UASB reactors.

6.3.6.d Removal of selected inorganic element: calcium

Calcium was not problematic for discharge into the local POTW since there was no established RMOC discharge limit for calcium. However, calcium was the main precipitating element in UASB reactors; therefore, it was responsible for inert solids accumulation. High calcium removal efficiencies were observed in UASB reactors: a minimum of 60% removal and a maximum of 97% (Figure 6.13). The feed calcium was as high as 1300 mg L^{-1} whereas the effluent was as low as 40 mg L^{-1}.

Calcium is, with magnesium, a hardness ion. The solubility of calcium carbonate ($CaCO_3$) is low (18 mg L^{-1} at 25°C) compared to calcium chloride ($CaCl_2$: 750 000 mg L^{-1} at 25°C) or calcium sulphate ($CaSO_4$: 1620 mg L^{-1} at 25°C) (Droste, 1997). Sulphate and chloride concentrations were high in raw leachate (Cl^-: 2 820 mg L^{-1} and SO_4^{2-}: 489 mg L^{-1}). Because CRL leachate was at low pH, there was likely a low concentration of carbonates; therefore, a possible solution to prevent calcium precipitation in reactors would be to increase pH with lime or caustic, therefore precipitating $CaCO_3$ prior to feeding to reactors.

Precipitated $CaCO_3$ could be removed by peat filtration. The overall cost of lime addition and $CaCO_3$ removal should be weighted against biomass periodical replacement. A benefit of lime addition would be the parallel removal of Mg, another hardness ion. Mg was also accumulating in reactors and was an important cause of tubing obstructions.

6.3.6.e Removal of selected inorganic element: cobalt

Cobalt removal efficiencies in UASB reactor were variable: a maximum removal of 78% was obtained on January 31st and a minimum of 32% on January 19th (Figure 6.14). The RMOC discharge limit for cobalt was 5 mg L^{-1} and this limit was far from being exceeded in raw CRL leachate itself (average cobalt was 0.1 mg L^{-1}); therefore, the importance of cobalt removal was

Figure 6.12: Barium in and out of reactors

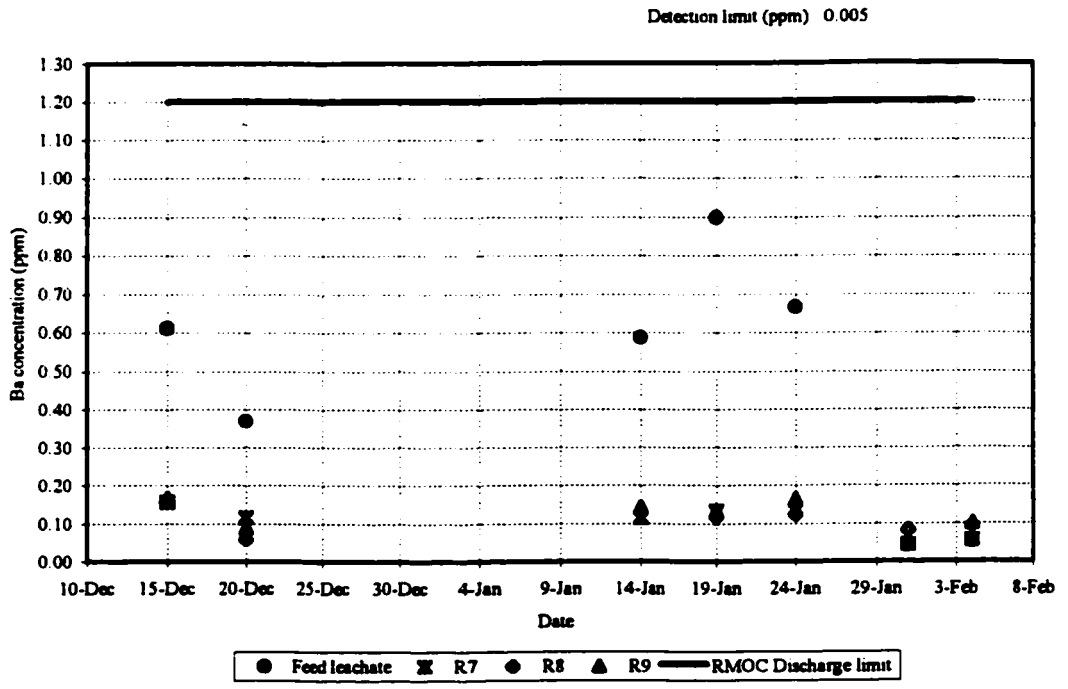
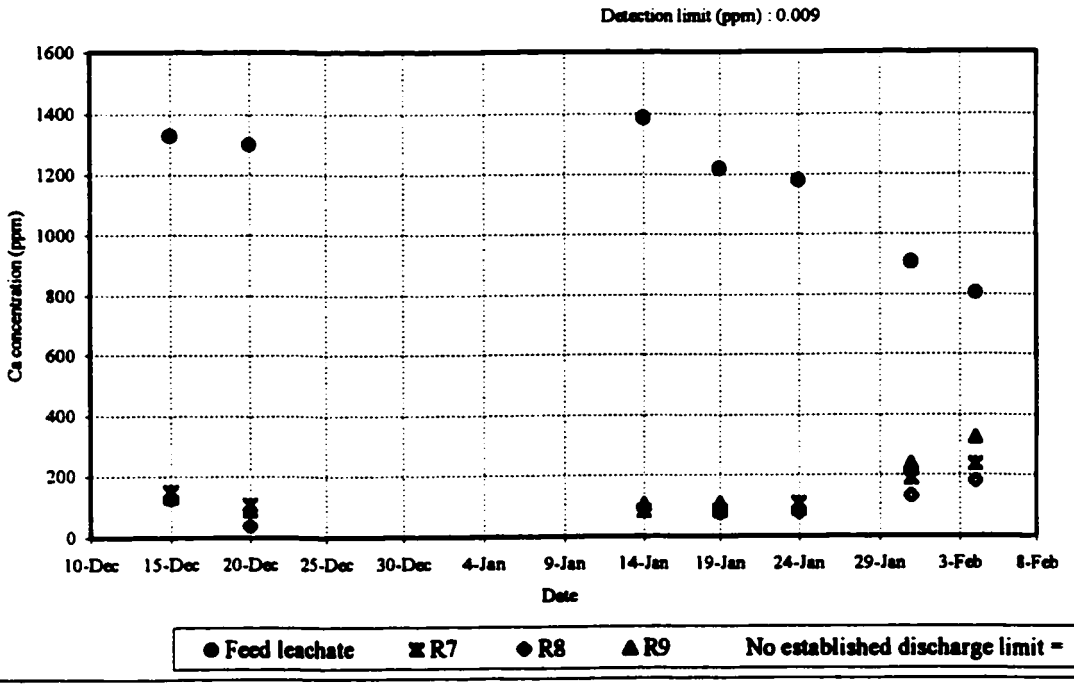


Figure 6.13: Calcium in and out of reactors



minimal. Nevertheless, the overall cobalt removal capacity of the whole treatment remained uncertain: if high cobalt concentration occurred in CRL leachate, there would be no information available on the performance of combined peat-UASB reactor treatment to meet the discharge limit concentration. It is suggested therefore that, in subsequent studies, cobalt-spiked samples would be studied to measure cobalt removal efficiencies of the combined treatment process at higher cobalt concentrations.

6.3.6.f Removal of selected inorganic element: iron

Iron removal by UASB reactor was high (Figure 6.15): ranging from a minimum of 73% (December 15th) to a maximum of 98% removal (January 14th). Iron was most probably precipitating with sulphides produced by the anaerobic environment in reactors. The removal efficiency was high and far below the discharge limit 1 500 mg L^{-1}. Iron was mainly found in raw leachate SS at concentrations usually below the RMOC discharge limit. Therefore iron was not problematic regarding the observance of the SUB, but caused inert solids accumulation problems in reactors. It was concluded that a benefit of peat filtration was to increase the iron removal capacity prior to UASB treatment. This slowed the accumulation of inert FSS in the reactor.

6.3.6.g Removal of selected inorganic element: manganese

Manganese removal by UASB reactors was the highest of all metals, except for aluminium. Removal efficiencies ranged from 93% (February 4th) to almost full removals (>99%) in five out of seven data samples (Figure 6.16). This was fortunate considering that the manganese feed concentration, which ranged from 5.5 to 12.1 mg L^{-1}, exceeded the SUB discharge limit of 5 mg L^{-1}. Additionally, the peat filter's ability to remove manganese was modest (40%). The combination peat-UASB reactor was concluded to be an efficient mean of manganese removal from the CRL leachate.

6.3.6.h Removal of selected inorganic element: zinc

High removals of zinc were attained by peat filtration. For reasons previously mentioned for aluminium and boron, filtered leachate had high zinc concentrations. Zinc removal efficiencies by UASB reactors were also among the highest (Figure 6.17) with a minimum removal of 72% (January 24th) and a maximum of 99% (December 20th). The minimal removal occurred when zinc feed concentration was at its lowest (0.31 mg L^{-1}). Feed zinc concentrations did not exceed the discharge limit (3 mg L^{-1}); the high removal efficiencies only increased the safety factor. Both

Figure 6.14: Cobalt in and out of reactors

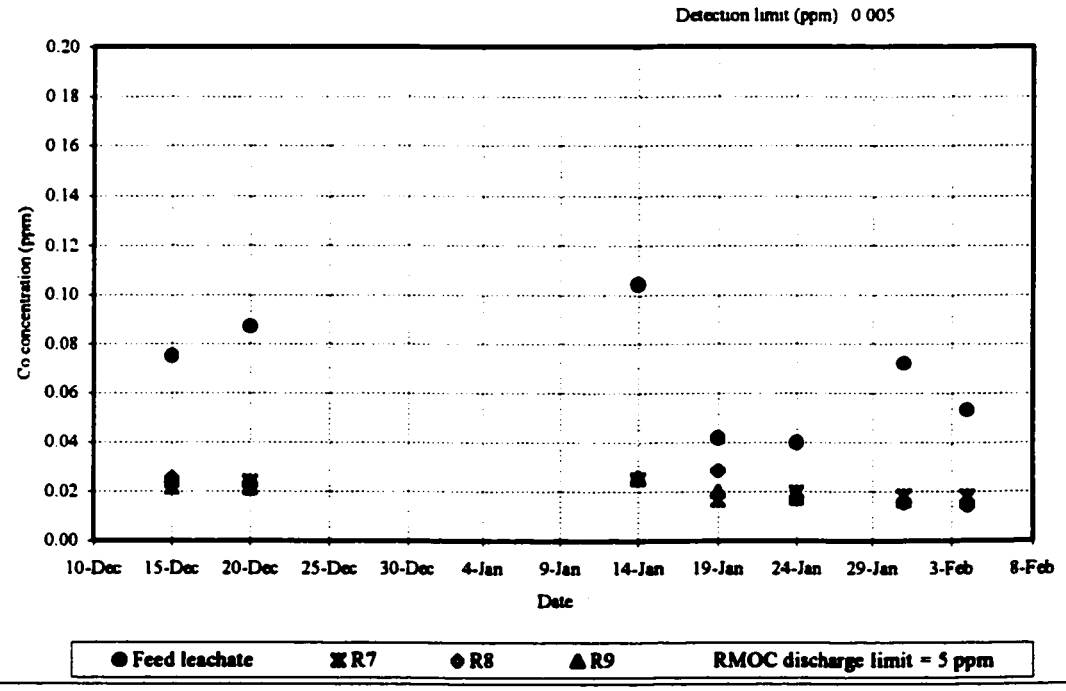


Figure 6.15: Iron in and out of reactors

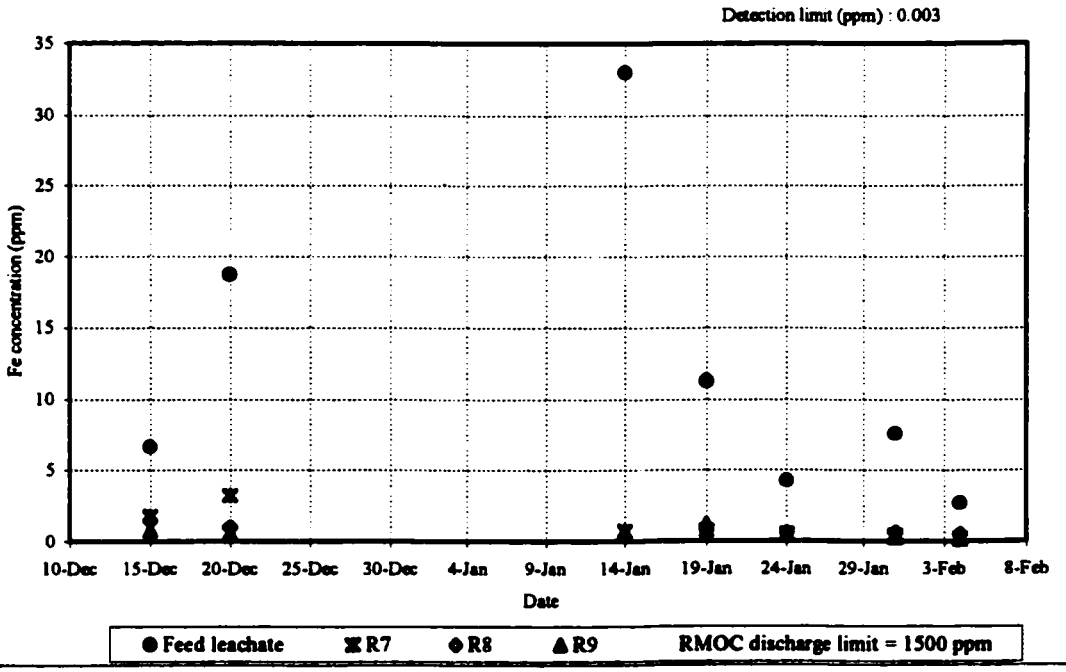


Figure 6.16: Manganese in and out of reactors

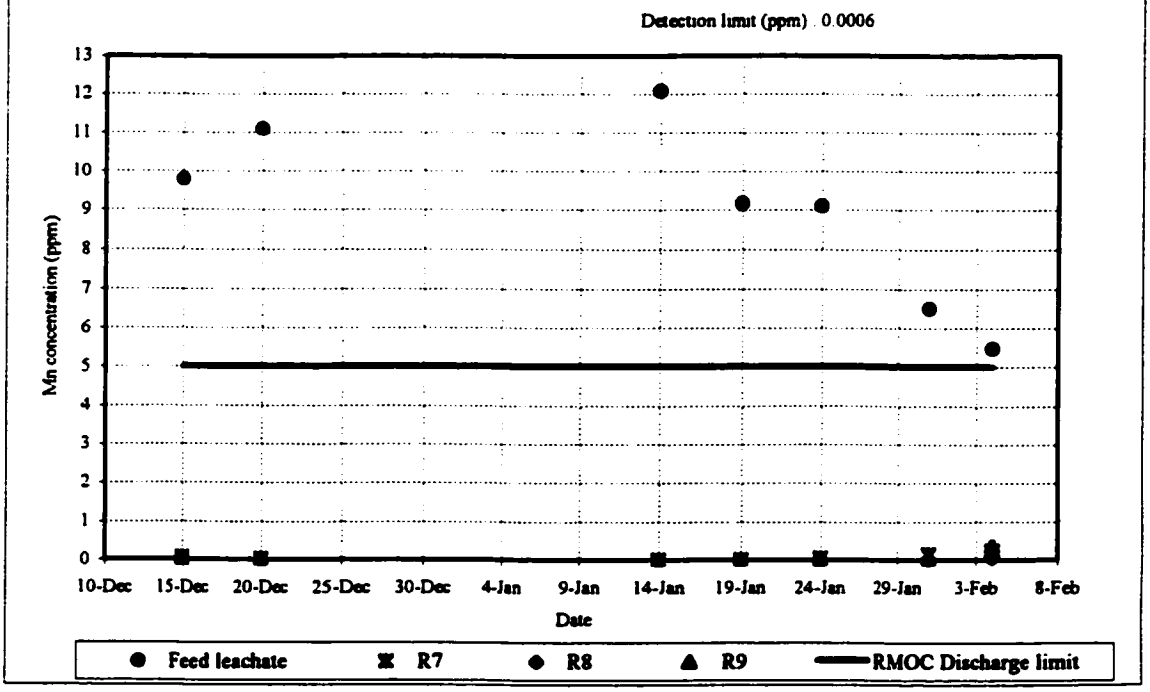
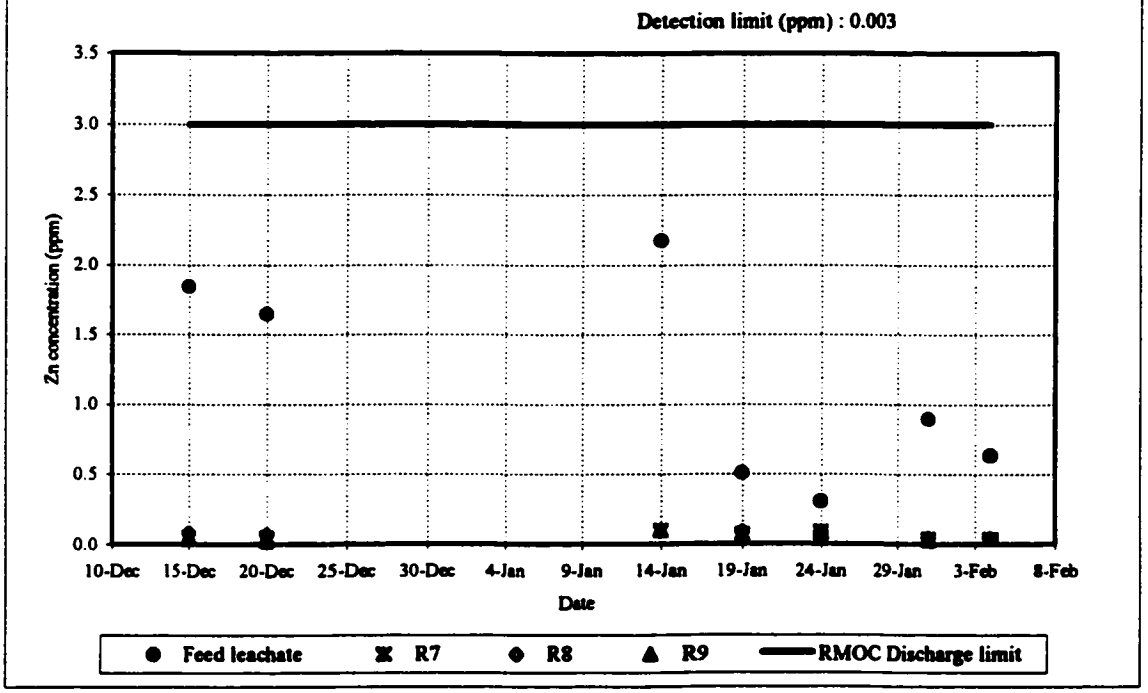


Figure 6.17: Zinc in and out of reactors



peat filters and UASB reactors had high zinc removal efficiencies and despite the low SUB discharge limit, treated-effluent zinc levels would be below the SUB POTW requirements. Zinc removal in the UASB, similarly to iron removal, was most probably the result of sulphide precipitation; therefore, sulphate concentration in the CRL leachate should be monitored to ensure high removals of iron and zinc.

6.4 CONCLUSIONS

CRL leachate was effectively treated by a combined peat-filter-UASB reactor process. Improvements in treatment are still to be made, however, the main operating conditions and potential problem areas are now clearer.

Within the range of operational conditions studied in this research combined peat-UASB treatment could remove high COD concentrations and meet RMOC SUB discharge limits. For simulated spring conditions, an HRT as low as 0.95 d and an OLR of 6.5 g COD L_r^{-1} d^{-1} were concluded to be adequate operational settings. For simulated fall conditions, an HRT of 1.4 d and an OLR of 4.7 g COD L_r^{-1} d^{-1} would meet regulatory discharge requirements. Finally for simulated summer conditions, an HRT of 1.6 d and a maximum OLR of 10.0 g COD L_r^{-1} d^{-1} were concluded to be adequate settings. These operating conditions assume peat pre-filtration, leachate concentrations tested, an adequate COD:P ratio of 500:1, an operation temperature of 32°C and a minimal recycle to feed ratio of 6.25:1.

Methane production results provided little insight into the process because of equipment problems. A more thorough study of methane production should be made if the methane production was of specific interest.

Based on bacterial activity in UASB reactors, it was concluded that the maximum SLR applied to UASB reactors should be 0.40 g COD g^{-1} VSS d^{-1} in order to ensure a SLR safety factor for reactor stability.

From solids profiles, CRL leachate treatment stimulated biomass growth. However, at moderate and high feeding rates, while biomass growth increased, more significant inert solids accumulation occurred, causing biomass calcification and biomass displacement. Peat pre-treatment had an irrefutable positive effect on solids accumulations, but long-term operation after peat treatment of the raw CRL inevitably resulted in FSS accumulation in sludge beds. Therefore, other pre-treatment methods, such as lime addition, should still be investigated.

Both elements, boron and barium, were removed from CRL leachate at levels below their SUB discharge limits by the combined peat filter-UASB treatment.

Iron, the third mandatory element to remove from CRL leachate, was already present in CRL leachate in lower concentrations than SUB discharge limit. However, iron removal by combined treatment was good and if higher iron concentrations were experienced in CRL leachate, the peat filter would efficiently remove suspended iron whereas soluble iron would precipitate within UASB reactors under anaerobic conditions. If soluble iron was to be problematic, a careful monitoring of sulphate concentration in the UASB reactor influent should assure a stoichiometrical precipitation of iron with sulphide.

Aluminium, manganese, and zinc removals in the peat filter-UASB reactors process were high whereas cobalt removal was inconclusive cobalt-spiked leachate should be tested to ascertain the efficiency of the combined treatment process for cobalt removal.

CHAPTER 7

SUMMARY CONCLUSIONS

The combined peat moss filter and UASB reactor setup provided an excellent treatment in terms of COD and soluble metals removal for the Carp Road Landfill leachate. In addition, the objectives of this study were all met:

- UASB reactor was demonstrated to be an efficient mean of decreasing COD, Fe, B and Ba concentrations and the effluents were found to meet sewer discharge regulations at the studied conditions;
- Pre-treatment of the leachate with a peat moss filter successfully removed suspended inorganic solids and selected soluble metals of the raw leachate.

7.1 SUMMARY OF EXPERIMENTAL RESULTS

7.1.1 PRELIMARY STUDIES

Preliminary treatment studies of the CRL leachate by UASB reactors led us to the conclusion that:

- The leachate was highly biodegradable, up to 97-98%;
- Leachate pre-treatment to remove inorganic suspended solids was required to prevent rapid sludge calcification;
- Required biomass inventory can be calculated from the leachate COD concentration and flow rate, and half (safety factor of 2) the maximum achieved specific removal rate, 0.21 g COD g VSS d^{-1} (the max SRR observed being 0.42 g COD g^{-1} VSS d^{-1}).

7.1.2 PEAT MOSS FILTRATION

Peat moss filtration was concluded:

- To be an excellent means of TSS removal for CRL leachate;

- To be highly efficient for removal of Al, B and Zn at inlet concentrations found in the raw leachate;
- To be inefficient for the removal of Mn ($[Mn]_{effluent}$ > SUB limit of 5.00 mg L^{-1}) and of Ba, which experienced desorption from peat moss ($[Ba]_{effluent} > [Ba]_{influent}$);
- To be variable for Fe removal at high concentration such as found in the raw leachate;
- To be inefficient for Ca and COD removal at high concentration such as found in the raw leachate (> 1500 mg Ca L^{-1} and > 30 000 mg COD L^{-1});
- To require periodical removal of the peat moss to prevent decreasing of the flux and/or of metal adsorption capacity.

7.1.3 UASB REACTORS

UASB reactor performance results are:

- COD removals between 91% and 98% were obtained;
- An average biogas of 0.44! 0.04 L biogas produced per g COD removed was found;
- The maximum assumed capacity with specific removal rates are of 0.42 g COD g^{-1} VSS d^{-1}, therefore, specific loading rates less or equal to 0.40 g COD g^{-1} VSS d^{-1} are advised to be applied on reactor digesting Carp Road Landfill leachate;
- OLRs over 10 g COD L_r^{-1} d^{-1} were concluded to be a risk zone, meaning that it is a range where the risk to exceed the SUB discharge limit is uncertain;
- Operation settings:
  - For spring, an HRT as low as 0.95 d and an OLR of 6.5 g COD L_r^{-1} d^{-1};
  - For fall, an HRT of 1.4 d and an OLR of 4.7 g COD L_r^{-1} d^{-1};
  - For summer, an HRT of 1.6 d and a maximum OLR of 10.0g COD L_r^{-1} d^{-1};
- These operating conditions assume peat moss pre-filtration, leachate concentrations tested, an adequate phosphorus:COD ratio of 500:1, an operation temperature of 32°C and a minimal recycle to feed ratio of 6.25:1.

Combined UASB reactor - peat moss filter treatment performance results (solids):

- Peat moss pre-treatment decreased biomass displacement, accumulations of -0.02 (R7) to 0.03 (R9) g VSS kg^{-1} COD_{fed} were measured;
- However, at moderate and high feeding rates, inert solids accumulation in biomass occurred despite the pre-treatment, the rate of inert solids accumulation was found to be ranging from -0.03 (R7) to 0.20 (R9) g FSS kg^{-1} COD_{fed};
- Therefore, because long-term operations inevitably resulted in FSS accumulation, other pre-treatment methods should still be investigated.

Combined UASB reactor - peat moss filter treatment performance results (metals):

- B and Ba were efficiently removed from CRL leachate at levels below their SUB discharge limits;
- Indifferently from the fact that Fe initial concentration was already lower than the SUB discharge limit, Fe removal was good; suspended Fe was mostly removed by the peat moss filter whereas soluble Fe would precipitate within UASB reactors under anaerobic conditions.
- Al, Mn and Zn removals were high (up to 99%) whereas Co removal was inconclusive.

7.2 RECOMMENDATIONS TO THE CLIENT

Recommendations to the clients are made based on the characteristics of the sampled leachate and on the information provided by the client at the time of the study; the results should be modify if these two factors varied. The recommendations are applicable for the Carp Road Landfill, in Ottawa, Ontario.

The design of the reactor set-up should be flexible because of the high variability of landfill leachate; variations in flow rates and in concentration are to be foreseen. General recommendations for peat moss filter and UASB reactor design are presented below.

7.2.1 PEAT FILTER DESIGN

The peat moss filter design was based on the following elements:

- Assuming 40 L m^{-2} h^{-1} as being acceptable as an initial flux;

- Assuming a raw leachate production from the lined (opened) portion of the Carp Road Landfill to be 2,840 L h^{-1};
- Assuming the physico-chemical characteristics of the leachate remains the same as the studied leachate
- Assuming all metals identified as being potentially problematic adequately removed by the combined treatment (below SUB limits) and only the flux variation is a time factor.

A surface of 71 m^2 of peat moss filter would be required initially; but because precipitants accumulate on peat top causing a flux decrease and because the head of leachate on peat moss must roughly be constant, the required surface area will increase as the flux decreases. Therefore, calculations must include flux variations; hypothesis a safety factor of 0.4 for hourly leachate production and a minimal flux of 20 L m^{-2} h^{-1} was made, a peat surface area of 199 m^2 was found to be required. The client is therefore recommended to construct two peat filters of 200 m^2; two filters are preferred to only one for keeping one in service while the other is replaced, or is on maintenance, or is experiencing problems.

7.2.2 REACTOR DESIGN

Iza *et al.* (1992) recommended that when implanting a pilot scale reactor on-site, the set-up should be very flexible to respond to the variability of the landfill leachate and to insure the long-term efficiency of the treatment. They also advised that a full-scale reactor would have the same flexibility to avoid additional costs due to redesign of another treatment adapted to older landfill leachate or closed landfill leachate. It is suggested to the client to revise the efficiency of the combined treatment, as the landfill age will increase, to verify if the treatment is still efficient and applicable. Méndez *et al.* (1989) underlined that anaerobic treatment is inefficient for older landfill leachate because of their high refractory fraction. A complementary study on removal of high polymeric molecules, such as those present in older landfill leachate, by peat moss filter would perhaps conclude that the filtration is efficient in low pH ranges, such as the results reported by Cameron (1978).

The reactor could be in operation the whole year, however, if required, a shut down of operations during wintertime should not be problematic. The current experiments demonstrated a

rapid biomass activity recovery after a latent period: biomass stored in reactors at room temperature for a period of five months required a relatively short acclimation period of 15 days.

The UASB reactor design was based on the following elements:

- Assuming a raw leachate production from the lined (opened) portion of the Carp Road Landfill to be 2,840 L h^{-1};
- Assuming the physico-chemical characteristics of the leachate remains the same as the studied leachate;
- Assuming the summer conditions valid as design criterion because landfill leachate has the highest concentration during that the season (15 000 mg COD L^{-1});
- Assuming the risk zone and the maximum OLR (10 g COD L_r^{-1} d^{-1}) to be valid;
- Assuming the maximum specific loading rates equal to 0.40 g COD g^{-1} VSS d^{-1} to be valid;
- Assuming a volatile solid composition of 60% to be valid (60% of TSS are VSS);
- Assuming peat moss pre-filtration, an adequate phosphorus:COD ratio of 500:1, an operation temperature of 32°C and a minimal recycle to feed ratio of 6.25:1.

Two reactors of 100 m^3 would be required according to the above elements; it is recommended to build two reactors for the same reasons as previously mentioned for the peat moss filter. A minimum of 2430 kg VSS would be required in each reactor, which implies at least 4000 kg of biomass (as TSS) in each reactor. In order to obtain a flexible design, it is suggested that the reactors could be expanded up to at least 150 m^3 (safety factor of 1.5) and that the recycle inlet and the outlet lines could be transferred to higher volume marks in the sludge blanket, allowing an expansion of the sludge bed.

7.3 FUTURE EXPERIMENTS

Suggestions of research topics, ensuing from unexplored elements related to this research, are:

- The study of the efficiency of the combined treatment on leachate samples taken directly from collection wells of the closed old parts of Carp Road Landfill; this leachate might be found, after physico-chemical analysis, to be significantly different from the studied leachate and maybe close to what the leachate might be constituted

of in a long term perspective. This experiment could be a guideline for leachate composition variation with time and the performance of the combined treatment could be roughly established.

- The improvement of peat moss filtration, firstly by softening the leachate with lime addition prior to filtration to precipitate out some soluble inorganics, such as Ca, that cause calcification of the sludge, secondly by spiking with Co similar leachate as the one studied to measure the filter and the combined treatment removal capacity for soluble Co, and finally perhaps by extending the analyzed metals to other metals that can potentially be present in Carp Road landfill leachate or in landfill leachate in general, such as copper (Cu), lead (Pb), nickel (Ni) and strontium (Sr), to verify the peat moss filter efficiency in removing other soluble metals.
- The study of phosphorus concentration variation during the combined treatment by closely monitoring P concentrations and dilutions and study thoroughly the impact of P addition, to improve the treatment and provide a greater safety merge in case of sudden feed COD increase.

7.4 CONCLUSION

The Carp Road landfill leachate contingency plan is now well-establish; it is known that the combined peat moss filter UASB reactor is efficient to treat this leachate and lower its constituents below the SUB discharge limits. The next procedures, if required, would be the construction of a pilot-scale treatment apparatus, thereafter a full-scale plant.

References:

Ackman, R.G., *Porous Polymer Bead Packing and Formic Acid in the GLC of Volatile Free Fatty Acids*. J. of Chromatographic Science, 1972, vol. 10, pp. 560-565.

Bagchi, Amalendu, *Design, Construction, and Monitoring of Sanitary Landfill*, John Wiley & Sons Inc., USA, 1990.

Britz, T.J., Venter, C.A. and Tracey, R.P., *Anaerobic Treatment of Municipal Landfill Leachate using an Anaerobic Hybrid Digester*. Biological Wastes, 1990, vol. 32, pp. 181-191.

Britz, Trevor J., *Microbiology of Landfill Sites*, Edited by Eric Senior, 2nd ed., Lewis Publishers, CRC Press Inc., Boca Raton, USA, 1995.

Cameron, Robert D., *Treatment of a Complex Landfill Leachate with Peat*, Canadian Journal of Civil Engineering, 1978, vol. 5, pp. 83-97.

Chang, Juu-En, *Treatment of Landfill Leachate with an Upflow Anaerobic Reactor Combining a Sludge Bed and a Filter*, Water Science Technologies, 1989, vol. 21, Brighton, pp. 133-143.

Chevalier, Pierre et collaborateurs, *Technologies d'assainissement et prévention de la pollution*. Télé-université, Presses de l'Université du Québec, Sainte-Foy, Québec, 1996.

Droste, Ronald L., *Theory and Practice of Water and Wastewater Treatment*, John Wiley and Sons, Inc., USA, 1997.

Elliott, C.M., *Gas Chromatography of Volatile Fatty Acids*, unpublished booklet, University of Ottawa, May 1989.

Environment Canada, 1998, THE CANADIAN CLIMATE AND WATER INFORMATION AND DATA SITE, http://www.cmc.ec.gc.ca/climate/normals/ONTO007.HTM .

Environment Protection Agency (EPA), *Development Document for Final Effluent Limitations Guidelines and Standards for the Landfills Point Source Category*, URL: http://www.epa.gov/OST/guide/landfills/final/index.html , Revised January 14, 2000.

Gurbuz Beker, Ulker, Cergel, Akif and Recepoglu, Oguz, *Removal of Boron from Geothermal Power Plant Wastewater in Kizildere, Turkey*, Energy Sources, 1996, vol. 18, 6, p. 645-654.

Government of Ontario, Ministry of the Environment web site, March 2000, http://www.ene.gov.on.ca/envision/faq/index.htm#Doesusing .

Iza, J., Keenan, P.J. and Switzenbaum, M.S., *Anaerobic Treatment of Municipal Solid Waste Landfill Leachate: Operation of a Pilot Scale Hybrid UASB/AF Reactor*, Water Science Technologies, 1992, vol. 25, 7, pp. 255-264.

Jian, M.K., Wu, W.M., Thiele, J.H. and Zeikus, J.G., *Performance of Ecoengineered Biomethanation Granules after Long-term Storage at Reduced Temperature*, Poster presented at

the 62nd Annual Conference of Water Pollution Control Federation, October 15-19 1989, San Francisco, CA.

Kadlec, R.H. and Hammer, D.E., *Wetland Utilization for Management of Community Wastewater*, 1979 Operation Summary, Report to NSF-ASRA, February 1980.

Kettunen, Riitta H. and Rintala, Jukka A., *Performance of an On-site UASB Reactor Treating Leachate at Low Temperature*, Water Research, 1998, vol. 32, 3, pp. 537-546.

Kennedy, K. J., Hamoda, M. F. and Guiot, S. G., *Anaerobic Treatment of Leachate using Fixed Film and Sludge Bed Systems*, Journal WPCF, 1988, vol. 60, 9, pp. 1675-1683.

Kennedy, K.J., *Anaerobic Digestion*, CVG 5179 course notes, University of Ottawa, 1999.

Lin, C-Y, *Anaerobic digestion of landfill leachate*, Water S.A., vol. 17, 4, Oct. 1991, pp. 301-306.

MacFarlane, Ivan C., *Engineering Characteristics of Peat*, Muskeg Engineering handbook, by the Muskeg Subcommittee of the NRC Associate Committee on Geotechnical Research, ed. by Ivan C. MacFarlane, University of Toronto Press, Toronto, 1969.

Méndez, R., Lema, J.M., Blàzquez, Pan, M. and Forjan, C., *Characterization, Digestibility and Anaerobic Treatment of Leachates from Old and Young Landfills*, Water Science Technologies, Brighton, 1989, vol. 21, pp. 145-155.

Myburg, C. and Britz, T.J., *Influence of Higher Organic Loading Rates on the Efficiency of an Anaerobic Hybrid Digester while Treating Landfill Leachate*, Water S.A., Oct. 1993, vol. 19, 4, pp. 319-324.

Ontario Regulation 232/98, *Landfill Standards – A Guideline on the Regulatory and Approval Requirements for New or Expanding Landfilling Sites*, Ministry of the Environment, Government of Ontario, June 1998.

Parkin, Gene F. and Owen, William F., *Fundamentals of Anaerobic Digestion of Wastewater Sludges*, Journal of Environmental Engineering Society, Am. Soc. of Civ. Eng., 1986, vol. 112, 5, pp. 867-920.

Pavlostathis, S.G. and Giraldo-Gomez, E., *Kinetic of Anaerobic Treatment:: A Critical Review*, Critical Review in Environmental Control, 1991, vol. 21 (5, 6), pp. 411-490.

Pohland, F.G., Dertien J.T. and Ghosh, S.B., *Leachate and Gas Quality Changes during Landfill Stabilisation of Municipal Refuse*, Proceedings of the Third International Symposium on Anaerobic Digestion, Boston, Massachusetts, August, 1983.

Regional Municipality of Ottawa-Carleton, Environmental Service Department, Water Environment Protection Division, *Regional Sewer Regulations, Regional Regulatory Code*, Chapter 5 – *Sewer and Waste Disposal*, part 5.2 – Sewers, Sewage Works and Control of Discharges, 800 Green Creek Drive, Gloucester, ON, K1J 1A6.

Reinhart, Debra R. and Al-Yousfi, A. Basel, *The Impact of Leachate Recirculation on Municipal Solid Waste Landfill Operating Characteristics*, Waste Management & Research, 1996, vol. 14, pp. 337-346.

Rumpf, Mary Idzorek and Ferguson, John F., *Anaerobic Pretreatment of a Landfill Leachate for Metals and Organics Removal*, Environmental Engineering: Proceedings of the 1990 Speciality Conference, edited by Charles R. O'Melia, American Society of Civil Engineering, New York, USA, 1990.

Spengel, Douglas B. and Dzombak, David A., *Treatment of Landfill Leachate with Rotating Biological Contactors Bench Scale Experiments*, Research Journal of Water Pollution Control Federation, 1991, vol. 63, pp. 971-981.

Standard Methods for the Examination of Water and Wastewater, Edited by Greenberrg, A. E., Clesceri, L.S. and Eaton, A.D., published by APHA, AWWA and WEF, 18th Edition, 1992.

Sung, Moon Sung, Chang, Duk and Lee, Hwa Young, *Performance Improvement of an Unstable Anaerobic Leachate Treatment System in an Industrial Waste Landfill*, Water Science Technologies, 1997, vol. 36, 12, pp. 333-340.

Sung, Moon Sung, Chang, Duk and Lee, Hwa Young, *Performance Improvement of an Unstable Anaerobic System in an Industrial Waste Landfill*, Water Science Technologies, 1997, vol. 36, 12, pp. 333-340.

Use of Adsorbents for the Removal of Pollutants from Wastewater, edited by Gordon McKay, Boca Raton, CRC Press, 1996.

van Haandel, Andrianus C. and Lettinga, Gatze, *Anaerobic Sewage Treatment, A Practical Guide for Regions with Hot Climate*, John Wiley & Sons Ltd, England, 1994.

Viraraghavan, T. and Rana, Shahid, *Use of Peat in Septic Tank Effluent Treatment – Column Studies*, Water Poll. Res. J. Canada, vol. 22, no 3, 1987

Warith, Mostafa A., *Design of Waste Containment Systems, course notes*, University of Ottawa, 1999.

Yong, Raymond N., The Montreal Geotechnical Group *Workshop on Soil Barriers to Control Groundwater Contamination at Landwaste Disposal Sites*, Montreal, Quebec, March 31, 1987.

Appendix A

Complied data for each runs:

- Reactor 4, 5, and 6: spring run
- Reactor 7, 8, and 9: fall run
- Reactor 10, 11, and 12: summer run

Biomass occupies approximately 1/2 of total liquid volume and temperature is constant at 31 +/- 1°C.
Mass of biomass in reactor 4 sludge bed=> initial = 40 g VSS / L ; final = 33.5 g VSS/L
Bed height => initial = 3.50 L ; final = 3.50 L, thus total mass of VSS = 140g VSS initially, 117 g VSS at the end
AAT rate of COD (as acetate) consumption (bottom sample) = 0.29 g COD / g VSS / d
Biomass concentration in reactor=> initial: 24 g VSS / L: final = 20 g VSS / L
Reactor total volume = 5.75L

Assuming a CH_4 concentration of 74%
(based on 0.39 L CH4 / g COD removed at 35°C)

Reactor 4: [HRT = 2.08 d]

Day	Feed (30%)	Effluent	Flow rate	% COD	OLR	Removed Loading	Approximated Biomass	Spec. Loading	Spec. Removal	HRT	Gas Prod°.	App. CH_4	Calculated	L CH_4 per	L CH_4 per
	mgCOD/L	mgCOD/L	L/d	removal	gCOD/L/d	gCOD/L/d	in reactor	gCOD/gVSS/d	gCOD/gVSS/d	d	(g) / L (r) /d	production	CH_4 Prod	g COD	g COD
	soluble	soluble		(weekly)		(weekly)	g VSS		(weekly)			L (g) / L (r) /d	L (CH_4) / L (r) /d	added	removed
26-May	5550	500	3.1		2.99			0.123		1.85	1.45	1.07		0.36	
27-May	6800	495	2.9		3.43			0.141		1.98	1.78	1.32		0.39	
28-May			2.9							1.98	1.83	1.36			
29-May	6175	497.5	3.0	91.9	3.21	2.95	140	0.131	0.120	1.94	1.69	1.25	1.15	0.39	0.42
30-May															
31-May															
1-Jun		243													
2-Jun	4050	158	2.7		1.90			0.083		2.13	1.09	0.81		0.43	
3-Jun		136	2.7							2.13	1.10	0.82			
4-Jun	5150		2.6		2.33			0.101		2.21	1.18	0.88		0.38	
5-Jun	4600	179	2.7	96.1	2.12	2.03	132	0.089	0.089	2.16	1.13	0.83	0.79	0.39	0.41
6-Jun	6260	112	2.4		2.61			0.121		2.40	1.20	0.89		0.34	
7-Jun		176	2.6							2.21	1.63	1.21			
8-Jun	5500	141	3.2		3.01			0.139		1.83	1.78	1.32		0.44	
9-Jun			2.6							2.05	1.68	1.24			
10-Jun	8140	186	2.9		4.11			0.189		1.98	2.18	1.61		0.39	
11-Jun			3.2							1.80	2.37	1.75			
12-Jun	6633	153.75	2.8	97.7	3.24	3.17	125	0.151	0.148	2.02	1.90	1.34	1.24	0.41	0.42
13-Jun															
14-Jun	5950	182													
15-Jun	5150	188	2.7		2.42			0.119		2.13	1.57	1.16		0.48	
16-Jun		124	4.4							1.31	2.22	1.64			
	5550	165	3.6	97.0	2.42	2.36	117	0.168	0.163	1.62	1.89	1.40	0.92	0.58	0.60
Averages	5749	249	2.78	95.7	2.75	2.63	129	0.136	0.130	2.08	1.63	1.20	1.02	0.44	0.46

(average HRT from June 2 to 11)

shaded areas are weekends
numbers in bold are weekly averages

Biomass occupies approximately 1/2 of total liquid volume and temperature is constant at 31 +/- 1°C.
Mass of biomass in reactor 5 sludge bed=> initial = 40 g VSS / L ; final = 32 g VSS / L
Bed height => initial = 3.3 L ; final = 3.2 L, thus total mass of VSS = 132g VSS initially, 102 g VSS at the end
AAT rate of COD (as acetate) consumption (bottom sample) = 0.62 g COD/gVSS/d
Biomass concentration in reactor: 18 g / L
Reactor total volume = 5.70L

* Assuming a CH_4 concentration of 74%
(based on 0.39 L CH4 / g COD removed at 35°C)

Reactor 5: [HRT = 1.35 d]

Day	Feed (30%	Effluent	Flow rate	% COD	OLR	Removed Loading	Biomass	Spec. Loading	Spec. Removal	HRT	Gas Prod*.	Approx. CH_4	Calculated	L CH_4 per	L CH_4 per
	mgCOD/L	mgCOD/L	L/d	removal	gCOD/L/d	gCOD/L/d	in reactor	gCOD/gVSS/d	gCOD/gVSS/d	d	L (g) / L (r) /d	production	CH_4 Prod	g COD	g COD
	soluble	soluble		(weekly)		(weekly)	g VSS		(weekly)			L (CH_4) / L (r) /d	L (CH_4) / L (r) /d	added	removed
26-May	5550	540	2.6		2.53			0.109		2.19	2.44	1.81		0.71	
27-May	6800	690	3.5		4.18			0.180		1.63	4.07	3.01		0.72	
28-May			4.0							1.43	3.99	2.95			
29-May	**6175**	**615**	**3.4**	**90.0**	**3.65**	**3.28**	**132**	**0.157**	**0.142**	**1.69**	**3.50**	**2.59**	**1.28**	**0.71**	**0.79**
30-May															
31-May															
1-Jun		208													
2-Jun	4050	140	3.7		2.63			0.123		1.54	2.83	2.10		0.80	
3-Jun		158	3.8							1.50	3.03	2.24			
4-Jun	5150		3.7		3.34			0.156		1.54	3.02	2.24		0.67	
5-Jun	**4900**	**168**	**3.7**	**96.3**	**3.01**	**2.90**	**122**	**0.141**	**0.136**	**1.53**	**2.96**	**2.19**	**1.13**	**0.73**	**0.76**
6-Jun	6260	117	3.6		3.95			0.201		1.58	3.55	2.62		0.66	
7-Jun		226	4.6							1.24	4.57	3.38			
8-Jun	5500	222	4.5		4.34			0.221		1.27	4.68	3.61		0.83	
9-Jun			4.4							1.30	4.68	3.46			
10-Jun	8140	374	4.5		6.43			0.327		1.27	6.32	4.68		0.73	
11-Jun			6.5							0.88	7.39	5.47			
12-Jun	**8833**	**235**	**4.7**	**98.5**	**5.45**	**5.26**	**112**	**0.277**	**0.268**	**1.22**	**5.23**	**3.87**	**2.06**	**0.71**	**0.74**
13-Jun															
14-Jun	5950	762													
15-Jun	5150	199	4.4		3.98			0.222		1.30				0.00	
16-Jun		196	7.4							0.77	6.14	4.54			
	5550	**366**	**5.9**	**93.1**	**5.74**	**5.35**	**102**	**0.321**	**0.299**	**0.97**	**6.14**	**4.54**	**2.06**	**0.79**	**0.85**
Averages	**5740**	**351**	**4.37**	**94.0**	**4.46**	**4.20**	**117**	**0.224**	**0.211**	**1.35**	**4.46**	**3.30**	**1.64**	**0.73**	**0.78**

shaded areas are weekends
numbers in bold are weekly averages

(average HRT from June 2 to 11)

Biomass occupies approximately 1/2 of total liquid volume and temperature is constant at 31 +/- 1°C.
Mass of biomass in reactor sludge bed=> initial = 40 g VSS / L ; final = 42.4 g VSS / L
Bed height => initial = 3.25 L ; final = 3.2 L, thus total mass of VSS = 130g VSS initially, 136 g VSS at the end
AAT rate of COD (as acetate) consumption (bottom sample) = 0.65 g COD/gVSS/d
Biomass concentration in reactor => initial: 23 g VSS / L: final = 24 g VSS / L
Reactor total volume = 5.70L

Assuming a CH_4 concentration of 74%
(based on 0.39 L CH4 / g COD removed at 35°C)

Reactor 6: [HRT = 0.76 d]

Day	Feed (30%)	Effluent	Flow rate	% COD	OLR	Removed Loading	Biomass	Spec. Loading	Spec. Removal	HRT	Gas Prod°.	App. CH_4	Calculated	L CH_4 per	L CH_4 per
	mgCOD/L	mgCOD/L	L/d	removal	mgCOD/L/d	gCOD/L/d	in reactor	gCOD/gVSS/d	gCOD/gVSS/d	d	L (g) / L (r) /d	production	CH_4 Prod	g COD	g COD
	soluble	soluble		(weekly)		(weekly)	g VSS		(weekly)			L (g) / L (r) /d	L(CH_4) / L (r) /d	added	removed
26-May	5550	620	2.9		2.82			0.124		1.97	1.06	0.78		0.28	
27-May	6800	755	4.5		5.37			0.235		1.27	2.08	1.54		0.29	
28-May			5.0							1.14	2.57	1.90			
29-May	6175	688	4.1	88.9	4.46	3.96	130	0.186	0.174	1.38	1.90	1.41	1.55	0.31	0.35
30-May															
31-May															
1-Jun															
2-Jun	4050	385	8.1		5.76			0.249		0.70	2.43	1.80		0.31	
3-Jun		274	7.3							0.78	2.30	1.70			
4-Jun	5150		7.4		6.69			0.289		0.77	2.65	1.96		0.29	
5-Jun	4900	330	7.6	92.8	6.13	5.69	132	0.266	0.246	0.75	2.46	1.82	2.22	0.30	0.32
6-Jun	6260	378	6.9		7.56			0.322		0.83	2.84	2.10		0.26	
7-Jun		552	7.6							0.75	3.63	2.69			
8-Jun	5600	660	7.4		7.14			0.304		0.77	3.81	2.82		0.39	
9-Jun			7.2							0.79	3.61	2.67			
10-Jun	8140	980	7.6		11.14			0.474		0.73	4.43	3.28		0.29	
11-Jun			7.9							0.72	4.84	3.58			
12-Jun	6833	643	7.5	90.9	8.69	7.86	134	0.370	0.334	0.76	3.88	2.86	3.06	0.33	0.36
13-Jun															
14-Jun	5950	600													
15-Jun	5150	282	8.4		5.78			0.242		0.89	3.21	2.37		0.41	
16-Jun		724	9.8							0.58	4.05	3.00			
	5689	535	8.1	90.4	7.89	7.13	136	0.331	0.299	0.70	3.63	2.69	2.78	0.34	0.38
Averages	5749	549	7.51	90.6	6.80	6.16	133	0.290	0.263	0.76	2.96	2.19	2.40	0.32	0.36

Approximated

shaded areas are weekends
numbers in bold are weekly averages

(average HRT from June 2 to 11)

Initial biomass concentration in reactor bed (Nov. 1st) => 35 g VSS/L (●2.75 L: 34 g VSS L^{-1}, ● 0.5 L: 36 g VSS L^{-1})
Final biomass concentration in reactor bed (Feb 12th) => 30 g VSS/L (●2.75 L: 29 g VSS L^{-1}, ● 0.5 L: 30 g VSS L^{-1})
Ave. bed height = 2.9L, thus total mass of VSS = 102 g VSS initially, 87 g VSS finally
Initial AAT rate of COD (as acetate) consumption (bottom sample) = 1.05 g COD g^{-1} VSS d^{-1}
Final AAT rate of COD (as acetate) consumption (bottom sample) = 0.56 g COD g^{-1} VSS d^{-1}
Biomass concentration in reactor: 18 g VSS L^{-1} initially, 15 g VSS L^{-1} finally

Organic Loading rate (reactor total volume = 5.70L)

R 7

Day	Feed	Feed	Effluent	Effluent	Flow rate	% COD	% COD	OLR	OLR	Approximated Biomass	Removed Loading	Removed Loading	Spec Loading	Spec Loading	Spec Removal
	mgCOD/L	mgCOD/L	mgCOD/L	mgCOD/L	L/d	removal	removal	gCOD/L/d	gCOD/L/d	in reactor	gCOD/L/d	gCOD/L/d	gCOD/gVSS/d	gCOD/gVSS/d	gCOD/gVSS/d
	total	soluble	total	soluble		total	soluble	total	soluble	g VSS	total	soluble	total	soluble	total
11-Nov	4620	4702	141	98	2.23	97	98	1.61	1.64	102	1.75	1.78	0.101	0.103	
12-Nov	5120	4430	167	106	2.09	97	98	1.82	1.58		1.77	1.53	0.102	0.088	
13-Nov						97	98						0.102	0.088	0.088
14-Nov	4960	4702	195	111	1.68	96	96	1.46	1.39		1.41	1.33	0.083	0.079	
15-Nov	4888	4889	142	111	1.66	97	98	1.42	1.42		1.38	1.38	0.081	0.081	
16-Nov	5140	5319	163	106	1.71	97	98	1.54	1.60		1.49	1.54	0.088	0.091	
17-Nov	4337	5118	163	108	1.65	96	96	1.25	1.48		1.21	1.42	0.071	0.084	
18-Nov	4879	4779	148	94	1.63	97	98	1.39	1.37		1.35	1.32	0.079	0.078	
19-Nov	4936		170	104	1.62	97		1.40			1.36		0.080		
20-Nov	4857	4661	164	106	1.7	97	98	1.41	1.45	100	1.37	1.40	0.080	0.083	0.078
21-Nov															
22-Nov			157	98	1.68										
23-Nov	5723	5555	129	117	1.55	98	98	1.55	1.51		1.52	1.47	0.090	0.087	
24-Nov	6545	6385	181	105	1.45	97	98	1.67	1.62		1.62	1.58	0.097	0.094	
25-Nov	6671	6363	203	120	1.44	97	98	1.69	1.61		1.64	1.56	0.098	0.094	
26-Nov	7283	7260	304	195	2.79	96	97	3.56	3.56		3.41	3.41	0.207	0.206	
27-Nov	7436	7520	280	173	1.62	97	98	2.11	2.14		2.04	2.06	0.123	0.124	
28-Nov	6729	6617	218	142	1.77	97	98	2.12	2.09		2.06	2.02	0.123	0.121	0.119
29-Nov	7786	7540	421	274	3.23	95	96	4.41	4.27		4.17	4.04	0.256	0.248	
30-Nov	8620	8780	304	194	1.68	97	98	2.60	2.58	98	2.51	2.49	0.151	0.150	
1-Dec	7975	9000	235	174	1.59	97	98	2.23	2.52		2.17	2.44	0.129	0.146	
2-Dec	9120	9600	279	198	1.75	97	98	2.80	2.95		2.71	2.86	0.162	0.171	
3-Dec	9120	10160	261	194	1.71	97	98	2.73	3.04		2.65	2.96	0.158	0.177	
4-Dec	8759	9360	270	190	1.7	97	98	2.59	2.77		2.51	2.69	0.150	0.161	0.146
5-Dec	10645				1.70			3.18					0.188		
6-Dec															
7-Dec	9240	9584	307	174	1.66	97	98	2.69	2.79		2.60	2.70	0.159	0.165	
8-Dec	9000	9624	267	182	1.66	97	98	2.66	2.84		2.58	2.76	0.157	0.168	
9-Dec	11100		684			94									
10-Dec		11823			1.35				2.80	96				0.166	
11-Dec	13212	11747	306	224	1.76	98	98	4.08	3.63		3.99	3.55	0.242	0.215	
12-Dec					1.63	98	98	3.15	3.02		3.08	3.00	0.186	0.178	0.180
13-Dec	13094	12867	343	218	1.66	97	98	3.82	3.76		3.72	3.66	0.231	0.227	
14-Dec	14384	12944	555	359	2.26	96	97	5.70	5.13		5.48	4.93	0.344	0.310	
18-Dec						97	98	4.76	4.45		4.60	4.30	0.287	0.268	0.278
19-Dec															
20-Dec	17900		695		2.06	96		6.46		95	6.21	0.00	0.389		
	6781	6666	216	146	1.7	97	98	2.0	2.1		2.0	2.0	0.118	0.121	0.114
Average	6521	6815	209	140	1.70	97	98	1.96	2.06		1.9	2.0	0.113	0.119	0.110
Std dev.	1689	1914	80	41	0.315	0.496	0.253	0.663	0.738		0.66	0.71	0.04	0.04	
+/-	845	957	30	21	0.16	0.25	0.13	0.342	0.369		0.328	0.355	0.020	0.022	

R 7

(based on a f_m ratio of 0.92 at 32°C)

Day	Spec Removal gCOD/gVSS/d soluble	HRT d	Gas Flow (L/d)	Gas Prod'n. L (g) / L (r) /d	CH_4 gas fraction total	CH_4 gas fraction soluble	
11-Nov		2.55	5.27	0.82	0.68	0.69	
12-Nov		2.81	4.89	0.82	0.77	0.67	
13-Nov	0.093	2.7	5.0	0.9	0.73	0.68	
14-Nov		3.39	3.79	0.67	0.76	0.72	
15-Nov		3.44	3.92	0.69	0.72	0.72	
16-Nov		3.33	4.37	0.77	0.70	0.72	
17-Nov		3.46	4.12	0.72	0.60	0.71	
18-Nov		3.50	3.93	0.69	0.70	0.69	
19-Nov		3.52	4.04	0.71	0.69		
20-Nov	0.081	3.44	4.0	0.7	0.69	0.71	
21-Nov							
22-Nov		3.39	3.45	0.61			
23-Nov		3.69	4.01	0.70	0.77	0.75	
24-Nov		3.93	4.41	0.77	0.75	0.73	
25-Nov		3.95	4.92	0.86	0.68	0.65	
26-Nov		2.04	10.03	1.76	0.70	0.70	
27-Nov		3.52	5.87	1.03	0.71	0.72	
28-Nov	0.119	3.42	5.4	1.0	0.72 e	0.71	
29-Nov		1.77	12.84	2.16	0.69	0.67	Pump failure
30-Nov		3.39	7.93	1.39	0.65	0.64	
1-Dec		3.57	7.51	1.32	0.59	0.67	
2-Dec		3.26	8.80	1.54	0.63	0.66	
3-Dec		3.34	9.07	1.59	0.60	0.67	
4-Dec	0.159	3.39	8.53	1.5	0.62	0.66	
5-Dec		3.35	9.28	1.63			
6-Dec							
7-Dec		3.43	8.99	1.58	0.59	0.61	
8-Dec		3.39	8.78	1.54	0.60	0.64	
9-Dec			9.90	1.74			
10-Dec		4.22					
11-Dec		3.24	11.10				
12-Dec	0.175	3.5	9.6	1.6	0.60	0.63	
13-Dec		3.42	12.32	2.16	0.62	0.61	
14-Dec		2.52	16.95	2.96	0.67	0.60	
18-Dec	0.202	3.0	14.8	2.56	0.64	0.60	
19-Dec							
20-Dec		2.77	19.20	3.37	0.66		
	0.119	3.42	5.9	1.04	0.68	0.66	Average November 14th to December 3rd
Average	0.116	3.4	5.8	1.01	0.68	0.70	
Std dev.		0.43	2.24	0.39	0.06	0.03	
+/-		0.22	1.12	0.20	0.03	0.02	

Initial biomass concentration in reactor bed (Nov. 1st) => 30 g VSS/L (@2.75 L: 33 g VSS L^{-1}, @ 0.5 L: 28 g VSS L^{-1})
Final biomass concentration in reactor bed (Feb 12th) => 35 g VSS/L (@2.75 L: 33 g VSS L^{-1}, @ 0.5 L: 37 g VSS L^{-1})
bed height = 3.0 L, thus total mass of VSS = 90 g VSS initially, 105 g VSS finally
Initial AAT rate of COD (as acetate) consumption (bottom sample) = 1.27 g COD g^{-1} VSS d^{-1}
Final AAT rate of COD (as acetate) consumption (bottom sample) = 0.75 g COD g^{-1} VSS d^{-1}
Biomass concentration in reactor: 15 g VSS L^{-1} initially, 18 g VSS L^{-1} finally
Organic Loading rate (reactor total volume = 5.85L)

R8

(based on a f_m ratio of 0.92 at 32°C)

Day	Feed (30%)	Feed (30%)	Effluent	Effluent	Flow rate	% COD	% COD	OLR	OLR	Approximated							HRT	Gas	Gas Prod°	CH_4 gas	CH_4 gas
	mgCOD/L	mgCOD/L	mgCOD/L	mgCOD/L	L/d	removal	removal	COD/L	COD/L								d	Flow (L/d)	L (g) / L (r) /d	fraction	fraction
	total	soluble	total	soluble		total	soluble	total	soluble											total	soluble
11-Nov	4830	4702	186	98	2.77	96	98	2.19	2.29	98	2.19	2.14	0.142	0.148			2.11	5.86	1.68	0.45	0.46
12-Nov	5182	4230	211	139	3.23	96	97	2.53	2.44		2.71	2.91	0.164	0.189			1.81	6.46	1.46	0.67	0.68
13-Nov						96	96						0.163	0.153	0.156	0.146	2.0	9.2	1.6	0.56	0.52
14-Nov	4980	4702	198	123	3.31	96	97	2.60	2.68		2.69	2.55	0.178	0.168			1.77	9.47	1.45	0.67	0.63
15-Nov	4998	4998	186	125	3.23	97	97	2.70	2.70		2.61	2.61	0.172	0.172			1.81	8.21	1.40	0.67	0.67
16-Nov	5140	5319	189	121	3.26	96	96	2.68	2.68		2.78	2.67	0.184	0.190			1.78	9.25	1.41	0.71	0.73
17-Nov	4397	5118	180	139	2.71	96	97	2.01	2.37		1.93	2.27	0.128	0.151			2.16	7.81	1.33	0.52	0.61
18-Nov	4979	4779	172	115	2.89	96	96	2.41	2.38		2.33	2.28	0.154	0.150			2.02	7.60	1.30	0.64	0.63
19-Nov	4936		142	118	2.89	97		2.43			2.36		0.155				2.03	7.53	1.29	0.66	
20-Nov	4857	4861	175	123	3.1	96	96	2.54	2.51	92	2.46	2.52	0.162	0.166	0.156	0.162	1.93	8.9	1.4	0.64	0.65
21-Nov																					
22-Nov			156	93	2.91												2.01	6.53	1.12		
23-Nov	5723	5655	155	84	2.73	97	98	2.67	2.59		2.60	2.52	0.167	0.162			2.14	7.98	1.35	0.69	0.67
24-Nov	6545	6385	190	127	2.87	97	98	3.21	3.13		3.12	3.04	0.200	0.195			2.04	9.74	1.48	0.75	0.73
25-Nov	6671	6383	212	154	2.68	97	98	3.06	2.92		2.96	2.83	0.191	0.182			2.18	9.66	1.66	0.64	0.61
26-Nov	7263	7290	220	169	2.77	97	98	3.44	3.44		3.33	3.33	0.215	0.214			2.11	10.49	1.79	0.67	0.67
27-Nov	7439	7520	247	182	2.90	97	98	2.82	2.99		2.83	2.98	0.182	0.194			2.54	8.98	1.53	0.66	0.67
28-Nov	6798	6617	287	146	2.87	97	98	3.00	3.01		2.97	2.92	0.191	0.188	0.186	0.184	2.17	8.7	1.5	0.66	0.67
29-Nov	7768	7540	274	209	2.77	96	97	3.66	3.56		3.56	3.44	0.230	0.222			2.12	11.25	1.62	0.66	0.64
30-Nov	6820	6780	244	188	3.10	97	98	4.66	4.64	94	4.56	4.51	0.292	0.290			1.66	13.75	2.35	0.69	0.69
1-Dec	7975	8000	284	230	3.14	96	97	4.29	4.93		4.12	4.66	0.267	0.301			1.66	14.44	2.47	0.60	0.66
2-Dec	8188	8836	310	286	2.94	97	97	4.99	4.93		4.48	4.69	0.288	0.301			1.66	14.66	2.63	0.63	0.66
3-Dec	8188	10100	318	340	2.66	97	96	4.87	5.66		4.61	4.89	0.294	0.318			1.66	14.66	2.66	0.62	0.69
4-Dec	6184	6433	271	211	3.0	97	97	4.36	4.55		4.21	4.44	0.273	0.286	0.283	0.276	2.0	13.0	2.4	0.66	0.67
5-Dec	10845				3.01			6.47					0.335				1.95	16.03	2.74		
6-Dec																					
7-Dec	8940	9684	341	256	3.12	96	97	4.89	5.11		4.75	4.92	0.301	0.313			1.66	16.19	2.77	0.62	0.64
8-Dec	9000	9624	296	226	2.92	97	98	4.46	4.80		4.36	4.66	0.275	0.294			2.00	14.66	2.56	0.61	0.66
9-Dec	11100		431			96												15.70	2.66		
10-Dec		11823		282	2.96				5.99	99			0.366				1.97	18.13	3.10		
11-Dec	13212	11747	405	357	3.07	97	97	6.94	6.17		6.73	5.98	0.424	0.377			1.90	20.15	3.44	0.70	0.62
12-Dec						97	97	5.46	5.53		5.36	5.19	0.334	0.336	0.332	0.336	1.9	18.9	2.9	0.64	0.64
13-Dec	13084	12987	446	372	3.19	97	97	7.15	7.03		6.91	6.79	0.429	0.422			1.83	21.75	3.72	0.67	0.66
14-Dec	14384	12944	583	427	3.20	96	97	7.67	7.06		7.55	6.79	0.472	0.425			1.83	24.15	4.13	0.66	0.59
15-Dec						96	97	7.51	7.65		7.23	6.79	0.461	0.423	0.434	0.419	1.9	23.9	3.92	0.66	0.62
16-Dec																		11.5	2.8		
20-Dec	17800		1175		3.23	93		9.87		96	9.22	0.00	0.562				1.81	26.39	4.51	0.73	
	8563	8678	217	158	2.9	97	98	3.3	3.4		3.2	3.3	0.208	0.214	0.261	0.268	2.02	10.2	1.7	0.66	0.66
Average	6240	6389	207	150	2.9	97	98	3.1	3.2		3.0	3.1	0.194	0.199			2.03	9.5	1.6	0.66	0.66
Std dev.	1425	1524	43	38	0.29	0.46	0.28	####	####		0.720	0.787	0.045	0.050			0.21	2.26	0.38	0.05	0.04
+/-	712	762	21	20	0.14	0.23	0.14	####	####		0.360	0.394	0.023	0.025			0.11	1.13	0.19	0.03	0.02

Average November 14th to December 3rd

Initial biomass concentration in reactor bed (Nov. 1st) => 35 g VSS/L (●2.75 L: 36 g VSS L^{-1}, ● 0.5 L: 34 g VSS L^{-1})
Final biomass concentration in reactor bed (Feb 12th) => 48 g VSS/L (●2.75 L: 38 g VSS L^{-1}, ● 0.5 L: 59 g VSS L^{-1})
bed height = 3.3 L, thus total mass of VSS = 116 g VSS initially, 158 g VSS finally
Initial AAT rate of COD (as acetate) consumption (bottom sample) = 1.06 g COD g^{-1} VSS d^{-1}
Final AAT rate of COD (as acetate) consumption (bottom sample) = 0.61 g COD g^{-1} VSS d^{-1}
Biomass concentration in reactor: 20 g VSS L-1 initially, 27 g VSS L-1 finally

Organic Loading rate (reactor total volume = 5.75L)

R9

Day	Feed (30%)	Feed (30%)	Effluent	Effluent	Flow rate	% COD	% COD	OLR	OLR	Biomass	Removed Loading	Removed Loading	Spec. Loading	Spec. Loading	Spec. Removal
	mgCOD/L	mgCOD/L	mgCOD/L	mgCOD/L	L/d	removal	removal	gCOD/L/d	gCOD/L/d	in reactor	gCOD/L/d	gCOD/L/d	gCOD/gVSS/d	gCOD/gVSS/d	gCOD/gVSS/d
	total	soluble	total	soluble		total	soluble	total	soluble	g VSS	total	soluble	total	soluble	total
11-Nov	4620	4702	189	103	3.34	96	96	2.68	2.73	116	2.57	2.67	0.133	0.135	
12-Nov	5120	4430	256	157	6.79	95	96	6.04	5.23		5.74	5.04	0.300	0.259	
13-Nov						96	97						0.216	0.197	0.206
14-Nov	4960	4702	253	159	4.62	95	97	3.98	3.78		3.78	3.65	0.189	0.179	
15-Nov	4988	4589	217	124	4.63	96	97	3.93	3.93		3.76	3.63	0.187	0.187	
16-Nov	5140	5319	211	128	4.22	96	98	3.77	3.90		3.61	3.81	0.179	0.185	
17-Nov	4337	5118	223	137	4.19	95	97	3.16	3.73		3.00	3.63	0.150	0.177	
18-Nov	4879	4779	195	117	4.06	96	98	3.44	3.37		3.30	3.29	0.163	0.160	
19-Nov	4936		189	109	4.11	97		3.52			3.40		0.167		
20-Nov	4957	4981	211	128	4.3	96	97	3.64	3.74	121	3.48	3.64	0.172	0.177	0.185
21-Nov															
22-Nov			155	102	4.04										
23-Nov	5723	5555	197	112	3.92	97	98	3.90	3.78		3.76	3.71	0.177	0.172	
24-Nov	6545	6365	204	132	3.75	97	98	4.27	4.17		4.14	4.08	0.194	0.189	
25-Nov	6671	6363	251	152	3.93	96	98	4.56	4.35		4.39	4.25	0.207	0.198	
26-Nov	7263	7260	284	192	3.99	96	97	5.04	5.04		4.84	4.90	0.229	0.229	
27-Nov	7438	7520	302	218	4.30	96	97	5.57	5.63		5.34	5.46	0.253	0.256	
28-Nov						96	98	4.87	4.98		4.49	4.48	0.212	0.208	0.204
29-Nov	7788	7540	323	230	4.24	96	97	5.75	5.56		5.51	5.39	0.261	0.253	
30-Nov	8820	8780	313	238	4.39	96	97	6.73	6.68	127	6.49	6.50	0.306	0.304	
1-Dec	7975	9000	334	250	4.39	96	97	6.09	6.86		5.84	6.69	0.277	0.313	
2-Dec	9140	9600	407	346	4.76	96	96	7.43	7.88		7.17	7.64	0.343	0.360	
3-Dec	9140	10100	407	366	4.66	96	96	7.27	8.04		6.98	7.76	0.329	0.366	
4-Dec	7278	7298	278	181	4.1	96	97	6.18	6.37		5.96	6.19	0.281	0.290	0.270
5-Dec	10645				4.55			8.43					0.386		
6-Dec															
7-Dec	9240	9584	384	295	4.48	96	97	7.20	7.47		6.90	7.24	0.314	0.326	
8-Dec	9000	9624	360	301	4.52	96	97	7.06	7.57		6.80	7.33	0.309	0.330	
9-Dec	11100														
10-Dec		11823		4380	4.51				9.28	192				0.405	
11-Dec	13212	11747	1386	1078	2.14	89	91	4.91	4.37		4.39	3.97	0.214	0.191	
12-Dec						96	97	7.57	7.53		6.86	7.285	0.311	0.328	0.298
13-Dec	13094	12667	541	389	3.97	96	97	9.03	8.88		8.66	8.61	0.379	0.373	
14-Dec	14384	12944	613	436	3.67	96	97	9.17	8.25		8.78	7.98	0.385	0.346	
15-Dec						96	97	9.10	8.57		8.72	8.28	0.382	0.360	0.366
16-Dec															
17-Dec															
18-Dec															
19-Dec															
20-Dec	17600		1386		3.72	92		11.68		197	10.68		0.486		
	6667	6130	244	180	4.2	96	97	4.8	4.9		4.6	4.8	0.222	0.225	0.213
Average	6240	6399	248	184	4.2	96	97	4.6	4.7		4.4	4.6	0.210	0.215	
Std dev.	1425	1524	52	49	0.27	0.61	0.37	1.092	1.189		1.056	1.153	0.047	0.052	
+/-	712	762	26	25	0.13	0.30	0.18	0.546	0.595		0.528	0.576	0.024	0.026	

R9

(based on a f_m ratio of 0.92 at 32°C)

Day	Spec. Removal gCOD/gVSS/d soluble	HRT d	Gas Flow (L/d)	Gas Prod°. L (g) / L (r) /d	CH_4 gas fraction total	CH_4 gas fraction soluble	
11-Nov		1.72	8.28	1.44	0.64	0.67	
12-Nov		0.85	14.67	2.69	0.80	0.70	
13-Nov	0.192	1.3	11.8	2.9	0.72	0.68	
14-Nov		1.25	11.63	2.02	0.67	0.65	
15-Nov		1.24	11.56	2.01	0.67	0.66	
16-Nov		1.36	11.37	1.98	0.66	0.69	
17-Nov		1.37	10.53	1.63	0.59	0.71	
18-Nov		1.42	10.00	1.74	0.68	0.68	
19-Nov		1.40	10.11	1.76	0.70		
20-Nov	0.173	1.3	10.9	1.9	0.66	0.68	
21-Nov							
22-Nov		1.42	7.64	1.33			
23-Nov		1.47	9.39	1.63	0.63	0.62	
24-Nov		1.53	11.11	1.93	0.77	0.76	
25-Nov		1.46	13.76	2.39	0.66	0.64	
26-Nov		1.44	14.95	2.60	0.67	0.68	
27-Nov		1.34	15.71	2.73	0.70	0.72	
28-Nov	0.204	1.4	12.1	2.1	0.73	0.72	
29-Nov		1.36	16.88	2.93	0.67	0.66	
30-Nov		1.31	19.45	3.38	0.69	0.69	
1-Dec		1.31	20.48	3.56	0.59	0.67	
2-Dec		1.31	24.15	4.04	0.64	0.66	Not use for the average
3-Dec		1.26	24.06	4.18	0.69	0.67	Not use for the average
4-Dec	0.291	1.32	19.93	3.29	0.66	0.67	
5-Dec		1.26	23.21	4.04			
6-Dec							
7-Dec		1.28	22.53	3.92	0.63	0.66	
8-Dec		1.27	23.62	4.11	0.59	0.64	
9-Dec			26.03	4.53			
10-Dec		1.27					recyling line / pump failure
11-Dec		2.66					recyling line / pump failure
12-Dec	0.318	1.3	23.9	4.1	0.61	0.65	
13-Dec		1.45	18.60	3.23	0.96	0.96	
14-Dec		1.57	19.33	3.36	0.94	0.85	
15-Dec	0.348	1.5	19.0	3.30	0.66	0.90	
19-Dec							
20-Dec		1.55	30.25	5.26	0.73		
	0.219	1.37	14	2.4	0.68	0.69	Average November 14th to December 1st
Average		1.36	13.3	2.3	0.68	0.70	
Std dev.		0.09	3.60	0.63	0.06	0.05	
+/-		0.04	1.80	0.31	0.03	0.02	

Initial biomass concentration in reactor bed (Nov. 1st) => 35 g VSS/L (●2.75 L: 34 g VSS L^{-1}, ● 0.5 L: 36 g VSS L^{-1})
Final biomass concentration in reactor bed (Feb 12th) => 30 g VSS/L (●2.75 L: 29 g VSS L^{-1}, ● 0.5 L: 30 g VSS L^{-1})
Ave. bed height = 2.9L, thus total mass of VSS = 102 g VSS initially, 87 g VSS finally
Initial AAT rate of COD (as acetate) consumption (bottom sample) = 1.05 g COD g^{-1} VSS d^{-1}
Final AAT rate of COD (as acetate) consumption (bottom sample) = 0.56 g COD g^{-1} VSS d^{-1}
Biomass concentration in reactor:18 g VSS L^{-1} initially, 15 g VSS L^{-1} finally **R10**
Organic Loading rate (reactor total volume = 5.70L)

	Feed	Feed	Effluent	Effluent	Flow rate	% COD	% COD	OLR	OLR	Approximated Biomass	Removed Loading	Removed Loading	Spec. Loading	Spec. Loading	Spec Removal
	mgCOD/L	mgCOD/L	mgCOD/L	mgCOD/L	L/d	removal	removal	gCOD/L/d	gCOD/L/d	in reactor	gCOD/L/d	gCOD/L/d	gCOD/gVSS/d	gCOD/gVSS/d	gCOD/gVSS/d
	total	soluble	total	soluble		total	soluble	total	soluble	gVSS	total	soluble	total	soluble	total
4-Jan	18180	18020			1.41			4.49	4.45	95			0.271	0.268	
5-Jan															
6-Jan	14550	14380	358	323	1.71	98	98	4.36	4.31		4.25	4.20	0.263	0.260	
7-Jan	15330	15350	399	333	1.74	97	98	4.67	4.67		4.55	4.55	0.282	0.282	
8-Jan	15615	16160		595	0.53		96	1.45	1.50			1.44		0.091	
9-Jan	14940	14865	378	328	1.7	97	98	4.51	4.49		4.40	4.39	0.272	0.271	0.265
10-Jan	16400	16080	374	305	1.40	98	98	4.02	3.95		3.93	3.86	0.248	0.243	
11-Jan															
12-Jan	17800	17800	566	443	2.16	97	97	6.67	6.67		6.46	6.46	0.411	0.411	
13-Jan	18580	17680	553	454	2.07	97	97	6.74	6.41		6.54	6.22	0.415	0.395	
14-Jan					1.96					93					
15-Jan	16000	15920	496	405	1.72	97	97	4.62	4.79		4.67	4.65	0.297	0.295	
16-Jan	17145	16820	498	402	1.6	97	98	5.56	5.46		5.40	5.30	0.342	0.336	0.332
17-Jan	15840	16080	470	408	1.98	97	97	5.51	5.59		5.35	5.43	0.339	0.344	
18-Jan															
19-Jan	16982	16900	416	328	2.00	98	98	5.91	5.92		5.76	5.77	0.371	0.372	
20-Jan					1.87										
21-Jan	16235	15800	400	313	2.18	98	98	6.20	6.04		6.05	5.89	0.390	0.379	
22-Jan															
23-Jan															
24-Jan	15215	15800	412	306	1.80	97	98	4.80	4.99	91	4.67	4.85	0.302	0.313	
25-Jan					2.03										
26-Jan	16036	16145	425	339	2.0	97	98	5.61	5.63		5.46	5.48	0.350	0.352	0.341
27-Jan	14800														
28-Jan	13900	12250	397	244	2.18	97	98	5.21	4.69		5.06	4.56	0.334	0.301	
29-Jan					2.24										
30-Jan	10980	9715	313	206	2.53	97	98	4.87	4.31		4.73	4.18	0.312	0.276	
31-Jan					1.58										
1-Feb	9447	8141	320	215	3.05	97	97	5.06	4.36		4.88	4.21	0.324	0.279	
2-Feb					3.48										
3-Feb	10153	8802	332	196	3.52	97	98	6.27	5.31	89	6.07	5.14	0.402	0.341	
4-Feb					3.78										
5-Feb					2.80	97	98	5.35	4.67		5.18	4.52	0.343	0.299	0.333
12-Feb					1.94					87					
	16041	15843	433	356	1.7	97	98	5.23	5.19		5.09	5.05	0.322	0.320	0.313
Average	16261	16159	444	362	1.88	97	98	5.37	5.33		5.22	5.19	0.332	0.329	
Std dev.	1187	1005	74	59	0.22	0.3	0.3	0.97	0.92		0.94	0.89	0.062	0.059	
+/-	594	502	37	29	0.11	0.16	0.14	0.487	0.462		0.469	0.445	0.031	0.030	

R10

	Spec. Removal gCOD/gVSS/d soluble	HRT d	Gas Flow (L/d)	Gas Prod'n. L (g) / L (r) /d	CH_4 gas fraction total	CH_4 gas fraction soluble	
4-Jan		4.05	12.80	2.25			
5-Jan			13.93	2.44			
6-Jan		3.34	14.88	2.61	0.58	0.58	
7-Jan		3.28	14.56	2.55	0.64	0.64	
8-Jan		10.75	7.30	1.28		0.40	pump failure
9-Jan	0.295	3.51	14.7	2.6	0.61	0.61	
10-Jan		4.07	12.50	2.19	0.64	0.63	
11-Jan			2.09	0.37			
12-Jan		2.84	20.91	3.67	0.63	0.63	
13-Jan		2.76	20.04	3.52	0.67	0.64	
14-Jan		2.91	19.64	3.45			
15-Jan		3.32	17.86	3.13	0.53	0.53	
16-Jan	0.338	3.14	18.3	3.2	0.62	0.61	
17-Jan		2.87	19.33	3.39	0.57	0.57	
18-Jan							
19-Jan		2.85	19.52	3.25	0.64	0.64	
20-Jan		3.05	19.99	3.40			START OF PHOSPHORUS ADDITION
21-Jan		2.82	19.84	3.48	0.62	0.61	
22-Jan			20.00	3.51			
23-Jan							
24-Jan		3.17					Gas tubing was strangled
25-Jan		2.81	17.76	3.12			
26-Jan	0.345	2.90	19.1	3.4	0.61	0.61	
27-Jan			19.1	3.36			
28-Jan		2.61	22.3	3.91	0.46	0.42	
29-Jan		2.55	22.9	4.02			
30-Jan		2.26	23.5	4.13	0.41	0.38	
31-Jan		3.60	24.1	4.23			
1-Feb		1.87	24.8	4.35	0.40	0.35	
2-Feb		1.64	29.0	5.08			
3-Feb		1.62	33.8	5.93	0.37	0.31	
4-Feb		1.51	36.4	6.39			
5-Feb	0.293	2.2	30.3	4.6	0.41	0.36	
12-Feb							
	0.312	3.1	17.4	3.0	0.61	0.61	Average January 8th to January 25th
Average		3.1	16.7	2.9	0.61	0.61	
Std dev.		0.40	5.0	0.9	0.04	0.04	
+/-		0.20	2.50	0.44	0.02	0.02	

0.20 g COD g^{-1} TSS d^{-1}

Initial biomass concentration in reactor bed (Nov. 1st) => 30 g VSS/L (@2.75 L: 33 g VSS L^{-1}, @ 0.5 L: 28 g VSS L^{-1})

Final biomass concentration in reactor bed (Feb 12th) => 35 g VSS/L (@2.75 L: 33 g VSS L^{-1}, @ 0.5 L: 37 g VSS L^{-1})

bed height = 3.0 L, thus total mass of VSS = 90 g VSS initially, 105 g VSS finally

Initial AAT rate of COD (as acetate) consumption (bottom sample) = 1.27 g COD g^{-1} VSS d^{-1}

Final AAT rate of COD (as acetate) consumption (bottom sample) = 0.75 g COD g^{-1} VSS d^{-1}

Biomass concentration in reactor: 15 g VSS L^{-1} initially, 18 g VSS L^{-1} finally

Organic Loading rate (reactor total volume = 5.85L)

R11

	Feed (30%)	Feed (30%)	Effluent	Effluent	Flow rate	% COD	% COD	OLR	OLR	Approximated							HRT	Gas	Gas Prod°.	CH_4 gas	CH_4 gas	
	mgCOD/L	mgCOD/L	mgCOD/L	mgCOD/L	L/d																	
	total	soluble	total	soluble													d	Flow (L/d)	L (g) / L (r) /d	fraction	fraction	
						removal	removal													total	soluble	
4-Jan	15190	15020			2.66			6.40	6.34	99			0.381	0.380			2.84	18.16	3.11			
5-Jan																		20.72	3.54			
6-Jan	14650	14380	602	424	2.71	96	97	6.74	6.66		6.46	6.39	0.404	0.400			2.16	19.43	3.32	0.70	0.99	
7-Jan	15330	15350	881	771	2.74	94	95	7.18	7.20		6.73	6.74	0.431	0.432			2.13	18.84	2.88	0.84	0.84	
8-Jan	15615	16160	995	466	2.62		97	7.00	7.25				0.420	0.435			2.23	21.2	3.62		0.00	
9-Jan	15165	15397	756	554	2.7	95	96	6.98	7.04		6.56	6.56	0.419	0.422	0.397	0.407	2.2	19.2	3.3	0.77	0.77	
10-Jan	16400	16080	845	426	2.39	96	97	6.69	6.56		6.43	6.31	0.384	0.388			2.45	19.77	3.38	0.69	0.67	
11-Jan					2.53												2.31	20.91	3.57			
12-Jan	17900	17900	726	620	2.66	96	96	7.98	7.98		7.65	7.65	0.469	0.469			2.21	23.37	4.00	0.69	0.69	
13-Jan	18580	17980	743	666	2.54	96	96	8.06	7.67		7.74	7.39	0.474	0.451			2.31	23.97	4.10	0.69	0.65	
14-Jan					3.71					99							1.57	19.95	3.24			very high VFA, left on recycle
15-Jan	18880	18830																12.8	2.20			
16-Jan	17537	17139	795	599	2.53	96	97	7.59	7.49		7.27	7.11	0.446	0.436	0.466	0.481	2.53	22.9	3.76	0.69	0.67	
17-Jan	18940	18080	569	466	2.00	96	97	5.41	5.46		5.22	5.30	0.319	0.323			2.93	16.62	2.84	0.69	0.67	
18-Jan																						
19-Jan	18982	18900	648	566	3.12	96	97	8.99	8.00		8.64	8.66	0.519	0.520			1.69	24.36	4.16	0.74	0.75	
20-Jan					2.62												2.00	26.46	4.53			START OF PHOSPHORUS ADDITION
21-Jan	18236	18900	463	451	2.98	97	97	8.20	8.08		8.04	7.83	0.478	0.466			1.96	28.39	4.91	0.84	0.82	
22-Jan																		26.10	4.46			
23-Jan																						
24-Jan	18218	18800	441	322	3.68	97	98	7.88	8.18	101	7.66	7.84	0.466	0.479			1.63	23.86	4.10	0.67	0.70	
25-Jan					2.66												1.86	23.69	4.06			
26-Jan	18858	18146	539	446	2.8	97	97	7.54	7.58		7.26	7.43	0.443	0.446	0.466	0.488	2.11	23.9	4.1	0.69	0.68	
27-Jan	14809																	24.27	4.15			
28-Jan	13900	12250	417	232	3.76	97	98	8.73	7.67		8.47	7.63	0.466	0.448			1.56	24.25	4.15	0.73	0.66	
29-Jan					4.62												1.26	22.47	3.84			
30-Jan	10889	9715	351	208	4.84	97	98	8.70	7.70		8.42	7.46	0.484	0.497			1.20	24.36	4.16	0.73	0.64	
31-Jan					4.86												1.16	28.84	4.55			
1-Feb	9447	8141	340	168	5.19	96	98	8.38	7.23		8.09	6.97	0.478	0.410			1.13	24.31	4.15	0.70	0.60	
2-Feb					6.11												0.96	27.33	4.67			
3-Feb	16183	8602	338	187	5.90	97	98	####	8.59	103	9.74	8.25	0.571	0.484			1.01	27.26	4.66	0.75	0.64	
4-Feb					5.99												0.99	24.80	4.24			
5-Feb						97	98	9.55	7.89		8.88	7.57	0.560	0.444	0.466	0.486	1.11	24.9	4.2	0.73	0.64	
12-Feb										106												
	16643	16167	663	523	3.7	96	97	7.4	7.4		7.1	7.0	0.436	0.434	0.418	0.431	2.2	21.7	3.7	0.71	0.71	Average January 6th to January 25th
Average	16223	16183	649	515	2.9	96	97	7.4	7.4		7.2	7.1	0.437	0.439			2.2	21.9	3.7	0.70	0.70	
Std dev.	1203	1001	154	136	0.41	1.0	0.8	1.0	1.0		1.05	1.03	0.056	0.055			0.32	3.29	0.56	0.06	0.06	
+/-	601	501	77	68	0.21	0.52	0.40	####	####		0.523	0.517	0.029	0.027			0.16	1.64	0.29	0.03	0.03	

Initial biomass concentration in reactor bed (Nov. 1st) => 35 g VSS/L (●2.75 L: 36 g VSS L^{-1}, ● 0.5 L: 34 g VSS L^{-1})
Final biomass concentration in reactor bed (Feb 12th) => 48 g VSS/L (●2.75 L: 38 g VSS L^{-1}, ● 0.5 L: 59 g VSS L^{-1})
bed height = 3.3 L, thus total mass of VSS = 116 g VSS initially, 158 g VSS finally
Initial AAT rate of COD (as acetate) consumption (bottom sample) = 1.06 g COD g^{-1} VSS d^{-1}
Final AAT rate of COD (as acetate) consumption (bottom sample) = 0.61 g COD g^{-1} VSS d^{-1}
Biomass concentration in reactor: 20 g VSS L-1 initially, 27 g VSS L-1 finally
Organic Loading rate (reactor total volume = 5.75L)

R12

	Feed (30%)	Feed (30%)	Effluent	Effluent	Flow rate	% COD	% COD	OLR	OLR	Approximated Biomass	Removed Loading	Removed Loading	Spec Loading	Spec Loading
	mgCOD/L	mgCOD/L	mgCOD/L	mgCOD/L	L/d	removal	removal	gCOD/L/d	gCOD/L/d	in reactor	gCOD/L/d	gCOD/L/d	gCOD/gVSS/d	gCOD/gVSS/d
	total	soluble	total	soluble		total	soluble	total	soluble	g VSS	total	soluble	total	soluble
4-Jan	18180	18020			2.40			7.60	7.53	137			0.319	0.316
5-Jan														
6-Jan	14550	14380	602	424	3.22	96	97	8.14	8.04		7.80	7.81	0.342	0.338
7-Jan	15330	15350	619	449	3.27	96	97	8.72	8.73		8.36	8.47	0.366	0.366
8-Jan	15815	16160	722	595	3.33		96	9.04	9.35				0.379	0.393
9-Jan	15165	15297	648	499	3.3	96	97	8.63	8.71		8.08	8.14	0.362	0.366
10-Jan	16400	16080	784	690	3.28	95	96	9.36	9.18		8.91	8.78	0.378	0.371
11-Jan					2.97									
12-Jan	17600	17600	1290	1299	3.89	93	93	11.90	11.90		11.03	11.03	0.481	0.481
13-Jan	18580	17680	1706	1699	3.91	91	90	12.64	12.03		11.48	10.87	0.511	0.486
14-Jan					3.68					142				
15-Jan	16000	15920	1827	1821	3.82	89	89	10.63	10.58		9.42	9.37	0.430	0.428
16-Jan	17145	16820	1402	1377	3.6	92	92	11.13	10.92		10.21	10.01	0.450	0.442
17-Jan	15840	16080	1427	1308	3.69	91	92	10.15	10.31		9.24	9.47	0.410	0.417
18-Jan														
19-Jan	16862	16900	1255	1180	3.36	93	93	9.86	9.88		9.12	9.19	0.384	0.385
20-Jan					3.70									
21-Jan	16235	15800	924	845	3.79	94	95	10.71	10.42		10.10	9.86	0.417	0.406
22-Jan														
23-Jan														
24-Jan	15215	15800	455	341	3.55	97	98	9.39	9.75	148	9.11	9.54	0.366	0.380
25-Jan					3.78									
26-Jan	16038	16145	1015	918	3.6	94	94	10.03	10.09		9.39	9.51	0.394	0.397
27-Jan	14800													
28-Jan	13600	12250	355	250	4.07	97	98	9.62	8.66		9.36	8.48	0.362	0.326
29-Jan					4.92									
30-Jan	10980	9715	347	204	5.76	97	98	11.00	9.73		10.65	9.52	0.414	0.366
31-Jan					6.29									
1-Feb	9447	8141	340	208	7.41	96	97	12.18	10.49		11.74	10.22	0.458	0.395
2-Feb					7.74									
3-Feb	10153	8802	378	207	7.98	96	98	14.09	11.94	153	13.57	11.65	0.531	0.450
4-Feb					8.07									
5-Feb					7.5	96	98	13.14	11.22		12.65	10.94	0.494	0.422
12-Feb										158				
	16116	16087	1022	928	3.5	94	94	9.9	9.9		9.2	9.2	0.402	0.401
Average	16202	16159	1056	968	3.5	93	94	10.0	10.0		9.5	9.4	0.406	0.405
Std dev.	1143	953	470	522	0.30	2.72	3.03	1.355	1.224		1.131	0.992	0.052	0.046
+/-	572	477	235	261	0.15	1.36	1.52	0.677	0.612		0.565	0.496	0.026	0.023

R12

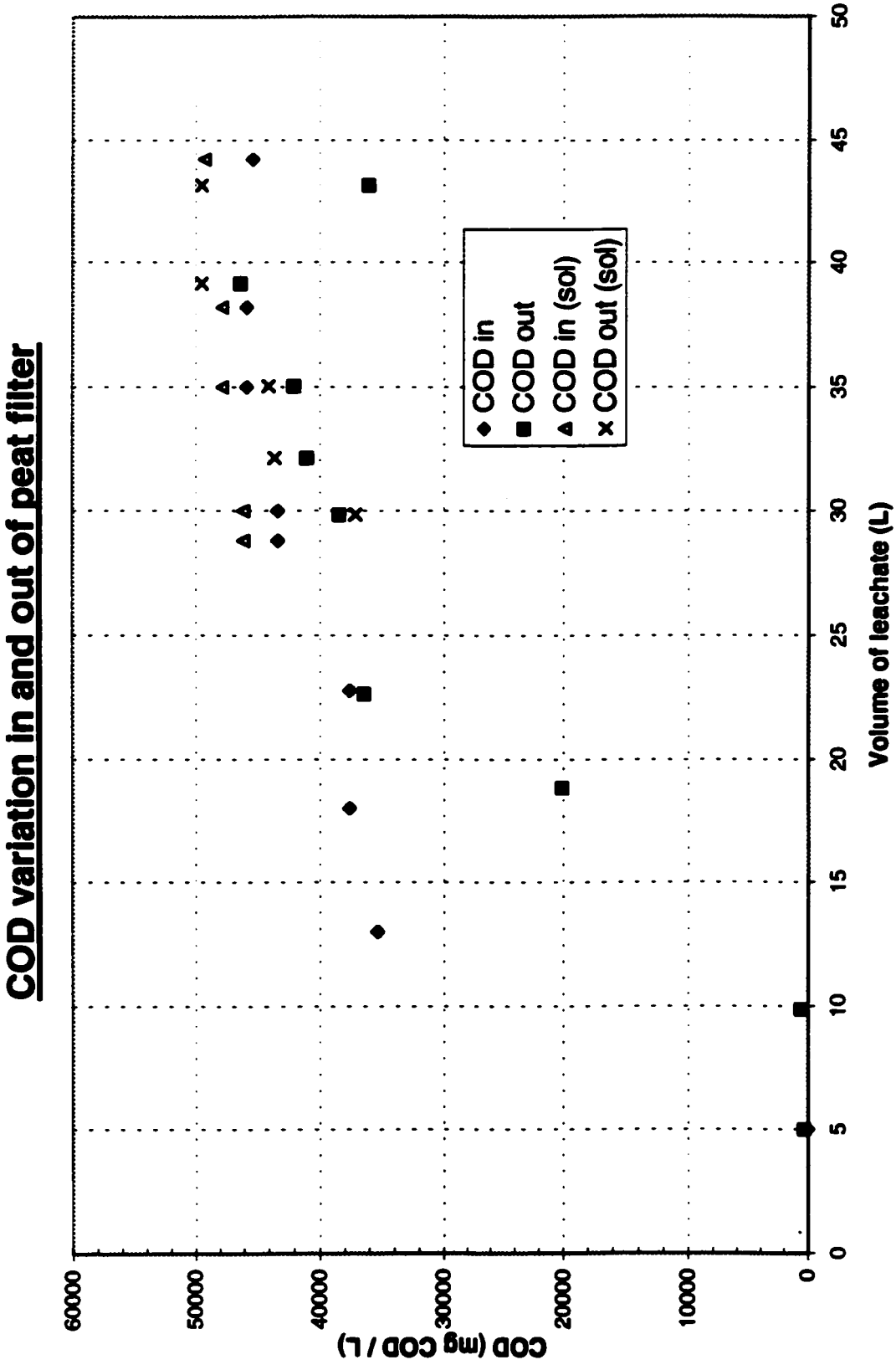
	Spec. Removal gCOD/gVSS/d total	Spec. Removal gCOD/gVSS/d soluble	HRT d	Gas Flow (L/d)	Gas Prod^{n}. L (g) / L (r) /d	CH_4 gas fraction total	CH_4 gas fraction soluble	
4-Jan			2.39	12.89	2.24			
5-Jan				13.37	2.33			
6-Jan			1.79	24.47	4.26	0.66	0.66	
7-Jan			1.76					
8-Jan			1.73	25.74	4.48			
9-Jan	0.347	0.354	1.8	26.1	4.4	0.66	0.66	
10-Jan			1.75	19.16	3.33	0.96	0.95	
11-Jan			1.93	16.14	2.81			
12-Jan			1.48	25.76	4.48	0.88	0.88	
13-Jan			1.47	27.24	4.74	0.87	0.82	
14-Jan			1.48	26.56	4.62			
15-Jan			1.50	26.68	4.64	0.73	0.72	
16-Jan	0.413	0.466	1.8	23.6	4.1	0.86	0.84	
17-Jan			1.56	25.36	4.41	0.75	0.77	
18-Jan								
19-Jan			1.71	23.09	4.02	0.82	0.82	
20-Jan			1.55	24.19	4.21			START OF PHOSPHORUS ADDITION
21-Jan			1.52	31.36	5.45	0.68	0.65	
22-Jan								
23-Jan								
24-Jan			1.62	24.44	4.25	0.77	0.81	
25-Jan			1.52	25.88	4.50			
26-Jan	0.370	0.375	1.8	26.7	4.5	0.75	0.76	
27-Jan				23.93	4.16			
28-Jan			1.41	24.33	4.23	0.79	0.72	Transition to new HRT
29-Jan			1.17	23.02	4.00			
30-Jan			1.00	27.04	4.70	0.81	0.78	
31-Jan			0.91	32.27	5.61			
1-Feb			0.78	31.34	5.45	0.77	0.67	
2-Feb			0.74	32.93	5.73			
3-Feb			0.72	34.24	5.96	0.82	0.70	
4-Feb			0.71	31.03	5.40			
5-Feb	0.476	0.412	0.77	32.36	5.63	0.80	0.71	
12-Feb								
	0.377	0.378	1.6	24.8	4.3	0.76	0.76	Average January 6th to January 25th
Average			1.6	23.8	4.1	0.79	0.79	
Std dev.			0.15	3.74	0.65	0.10	0.10	
+/-			0.07	1.67	0.33	0.05	0.05	

Appendix B

Results of COD test on peat moss filtration samples (in and out of filter)

COD variation in and out the peat filter; COD is in mg COD L^{-1}, volumes are in L

Volume IN	OUT + sample	Volume Cumm. In	COD in (tot)	COD in (sol)	Volume Cumm. Out	COD out (tot)	COD out (sol)
5.0	5.0	5	0		5	314	
8.0	4.9	13	35300		9.85	583	
5.0	9.0	18	37600		18.85	20100	
4.8	3.8	22.8	37600		22.65	36400	
6.0	7.2	28.8	43250	46000	29.85	38400	37100
1.2	2.3	30	43250	46000	32.15	41000	43500
5.0	2.9	35	45750	47750	35.05	42000	44000
3.2	4.1	38.2	45750	47750	39.15	46250	49500
6.0	4.0	44.2	45250	49250	43.15	36000	49500
	total =	235		total =	235.7		


COD variation in and out of peat filter
COD (mg COD / L)
Volume of leachate (L)
COD in
COD out
COD in (sol)
COD out (sol)
0
10000
20000
30000
40000
50000
60000
0
5
10
15
20
25
30
35
40
45
50

Appendix C

Official communications:

- Certificate of analysis of raw CRL leachate
- Memo from CWS on CRL specifications
- RMOC SUB for discharge of landfill leachate

Seprotech Laboratories

Certificate of Analysis

Client:
University of Ottawa
161 Louis Pasteur St.
Ottawa, Ontario
K1N 6N5

Report: 200002182
Project:
Submitted by: C.S.
Date submitted: April 14, 2000
Date printed: April 26, 2000

Attention: Catherine Savard
page 1 of 3

Matrix: solution

Parameter (dissolved)	Units	Det. Limit	Raw Jan 13	Raw Feb 7	Raw Mar 30	Filter 1	Filter 2	Filter 3
aluminum	mg/L	0.01	3.00	3.34	3.22			
antimony	mg/L	0.05	<0.05	<0.05	<0.05			
arsenic	mg/L	0.1	<0.1	0.1	<0.1			
barium	mg/L	0.005	0.485	0.480	0.485			
beryllium	mg/L	0.005	<0.005	<0.005	<0.005			
bismuth	mg/L	0.05	<0.05	<0.05	<0.05			
boron	mg/L	0.01	2.31	3.34	2.75			
cadmium	mg/L	0.01	<0.01	<0.01	<0.01			
calcium	mg/L	0.03	2200	2640	2060			
chromium	mg/L	0.01	0.04	0.05	0.01			
cobalt	mg/L	0.01	0.07	0.12	0.07			
copper	mg/L	0.01	<0.01	<0.01	<0.01			
gallium	mg/L	0.05	<0.05	<0.05	<0.05			
iron	mg/L	0.02	6.78	3.68	7.08			
lead	mg/L	0.1	<0.1	<0.1	<0.01			
lithium	mg/L	0.005	0.350	0.530	0.435			
magnesium	mg/L	0.01	349	421	329			
manganese	mg/L	0.01	12.0	16.2	13.1			
molybdenum	mg/L	0.02	<0.02	<0.02	<0.02			
nickel	mg/L	0.02	0.69	0.85	0.63			
niobium	mg/L	0.02	<0.02	<0.02	<0.02			
phosphorus	mg/L	0.1	1.8	2.4	1.8			
potassium	mg/L	0.4	745	899	725			
silicon	mg/L	0.05	3.06	3.67	3.06			
silver	mg/L	0.01	<0.01	<0.01	<0.01			
sodium	mg/L	0.2	1450	1710	1490			
strontium	mg/L	0.005	92.5	111	75.1			
tin	mg/L	0.2	<0.2	<0.2	<0.2			
titanium	mg/L	0.01	<0.01	<0.01	<0.01			

Seprotech Laboratories
2378 Holly Lane, Ottawa, Ontario, K1V 7P1, Canada
Tel: (613)523-1641, Fax: (613)731-0851

Dave Peeler, Lab Supervisor

Seprotech Laboratories

Certificate of Analysis

Client:
University of Ottawa
161 Louis Pasteur St.
Ottawa, Ontario
K1N 6N5

Report: 200002182
Project:
Submitted by: C.S.
Date submitted: April 14, 2000
Date printed: April 26, 2000

Attention: Catherine Savard
page 2 of 3

Matrix: solution

Parameter (dissolved)	Units	Det. Limit	Raw Jan 13	Raw Feb 7	Raw Mar 30	Filter 1	Filter 2	Filter 3
tungsten	mg/L	0.05	<0.05	<0.05	<0.05			
vanadium	mg/L	0.005	<0.005	<0.005	<0.005			
yttrium	mg/L	0.005	<0.005	<0.005	<0.005			
zinc	mg/L	0.01	4.73	4.73	4.21			
zirconium	mg/L	0.01	<0.01	<0.01	<0.01			
sulphate	mg/L	1	430	550	495			
alkalinity as $CaCO_3$	mg/L	1	4420	5660	5400			
total kjeldahl nitrogen	mg/L	0.05	451	528	432	226	230	204
total phosphorus	mg/L	0.01	2.04	2.26	2.35	1.18	1.11	1.08
B.O.D.$_5$	mg/L	1	13500	18100	14200			

Seprotech Laboratories
2378 Holly Lane, Ottawa, Ontario, K1V 7P1, Canada
Tel: (613)523-1641, Fax: (613)731-0851

Dave Peeler, Lab Supervisor

Seprotech Laboratories

Certificate of Analysis

Client:
University of Ottawa
161 Louis Pasteur St.
Ottawa, Ontario
K1N 6N5

Report: 200002182
Project:
Submitted by: C.S.
Date submitted: April 14, 2000
Date printed: April 26, 2000

Attention: Catherine Savard
page 3 of 3

Matrix: solution

Parameter (total)	Units	Det. Limit	Raw Jan 13	Raw Feb 7	Raw Mar 30	Filter 1	Filter 2	Filter 3
aluminum	mg/L	0.01	6.76	7.64	7.03			
arsenic	mg/L	0.1	<0.1	<0.1	<0.1			
barium	mg/L	0.005	0.625	0.580	0.565			
beryllium	mg/L	0.005	<0.005	<0.005	<0.005			
bismuth	mg/L	0.05	<0.05	<0.05	<0.05			
cadmium	mg/L	0.02	<0.02	<0.02	<0.02			
calcium	mg/L	0.03	2450	3060	2630			
chromium	mg/L	0.01	0.14	0.22	0.19			
cobalt	mg/L	0.01	0.08	0.26	0.08			
copper	mg/L	0.01	<0.01	0.04	<0.01			
gallium	mg/L	0.05	<0.05	<0.05	<0.05			
iron	mg/L	0.02	278	377	323			
lead	mg/L	0.1	<0.1	<0.1	<0.1			
lithium	mg/L	0.005	0.380	0.560	0.510			
magnesium	mg/L	0.01	392	495	425			
manganese	mg/L	0.01	15.1	18.9	16.0			
molybdenum	mg/L	0.02	0.16	0.10	0.06			
nickel	mg/L	0.02	0.87	1.04	0.88			
phosphorus	mg/L	0.1	2.2	3.3	2.5			
potassium	mg/L	0.4	761	949	809			
silver	mg/L	0.02	<0.02	<0.02	<0.02			
sodium	mg/L	0.2	1600	1990	1730			
strontium	mg/L	0.005	103	130	111			
titanium	mg/L	0.01	<0.01	<0.01	<0.01			
vanadium	mg/L	0.005	0.010	0.035	0.030			
yttrium	mg/L	0.005	<0.005	<0.005	<0.005			
zinc	mg/L	0.01	6.07	7.08	5.95			

Seprotech Laboratories
2378 Holly Lane, Ottawa, Ontario, K1V 7P1, Canada
Tel: (613)523-1641, Fax: (613)731-0851

Dave Peeler, Lab Supervisor

CANADIAN WASTE
A WASTE MANAGEMENT COMPANY

MEMO

TO: Catherine Savard

From: Michael Walters

SUBJECT: **Canadian Waste Services Inc.**
West Carleton Landfill Information

DATE: September 22, 1999

Listed below is the information you required in your memo of September 21st, 1999.

Age of Landfill: landfill began operation in 1970.

Leachate Sample Location: the cell where you received the samples became operational in 1992. This cell will continue to be used for the remaining life of the site (12 years).

Total Landfill Area: we own 342.2 acres of land that included the approved permitted footprint. The approved permitted is 85.5 acres. The cell where the leachate was obtained is approximately 25 acres.

Quantity of Waste Received: since 1989 the site received approximately 240,000 mt of MSW and up to 140,000 mt of hydrocarbon impacted soil per year.

Percentage Recycled: we have diverted, as high as 50%; however, our normal diversion total is 40%. The majority of our waste diversion comes from our reuse of the hydrocarbon impacted soils as cover. On average 210,000 mt is actually landfilled per year. We recycle wood 13,000 per year. Biosolids – 25,000 mt per year, and metal 2,500 mt per year.

Waste Composted: Residential waste 40%.
Industrial Commercial 60%

Note: no recycled tonnage is included in this breakdown.

80 to 85% of the landfill permitted volume is used for waste and 15 to 20% of the permitted volume is daily cover.

Leachate Production: There are two (2) separate streams for leachate production.

Purge Wells
This is for the 60 acres of unlined landfill. There are 11 purge wells that collect and pump the leachate off site through an off-site forcemain to the Regions Sewage Treatment Plant. The flow of leachate impacted ground water is 195 gallons per minute.

Liner System
This is for 25 acres. The leachate-generated rate is 12.5 gallons per minute.

Currently we haul 1000m3 of leachate to the Pickard Sewage Treatment Plant per week. No leachate is currently being recirculated. We have just received approval to construct our off-site forcemain that will allow all leachate from both systems to be pumped off-site for treatment. We are planning to recirculate our liner leachate back into the lined portion in 2000.

We have designed for 317mm per year of infiltration through uncapped section and 242 mm per year through the capped section.

Region of Ottawa-Carleton
Robert O. Pickard Environmental Centre
800 Green Creek Drive
Gloucester, Ontario K1J 1A6
Environment and Transportation Department
Water Environment Protection Division
Tel. (613) 560-1335
Fax. (613) 745-9197

Région d'Ottawa-Carleton
Centre environnemental Robert O. Pickard
800, promenade Green Creek
Gloucester (Ontario) K1J 1A6
Services de l'environment et des transports
Division de la protection du milieu aquatique
Tél. (613) 560-1335
Télécopieur (613) 745-9197

2 May 2000

File: 50 40-93-0052-IW

Catherine Savard, Graduate Student
Environmental Engineering
University of Ottawa
161, rue Louis-Pasteur
C.P. 450, SUCC. A
Ottawa, ON K1N 6N5

Dear Ms. Savard

Re: Landfill Leachate Limits

The Region of Ottawa-Carleton has a sewer use bylaw that prohibits the discharge of waste disposal site leachate to the sanitary sewer system. Under specific conditions listed in Part 5.2.2(9) of the Regional Regulatory Code and subject to Regional Council approval, leachate may be discharged to the sewer system under the authority of a leachate agreement negotiated with the landfill site. The agreement clearly delineates discharge limits for the parameters found in the leachate, volume limits and fees, as well as when and where the leachate is to be discharged including any special conditions for the discharge.

The leachate agreement specifies parameter limits established in Part 5.2.2 of the Regional Regulatory Code as well as limits for parameters normally contained in leachate. In response to your request for specific parameter discharge limits for landfill site leachate, these limits are as follows:

Parameter	Discharge Limit (mg/L)
Aluminium	50
Boron	2
Barium	1.2
Calcium	no limit established
Cobalt	5
Iron	1500
Manganese	5
Zinc	3
COD	1000
BOD	300

If you have questions or concerns, please do not hesitate to contact me at 560-6086, extension 2837.

Sincerely,

George Reimer
Industrial Waste Inspector
Technical Support and Wastewater Collection Branch

GR/gr/ng

G:\Industrial Waste\REIMER\MISC\Leachate limits.doc