UNIVERSITY OF NEWCASTLE UPON TYNE
DEPARTMENT OF CIVIL ENGINEERING
DIVISION OF ENVIRONMENTAL ENGINEERING

COMPARATIVE STUDIES OF LANDFILL LEACHATE TREATMENT USING AEROBIC, ANAEROBIC AND ADSORPTION SYSTEMS

BY

MOHD RAZMAN SALIM, B. Sc., M. Sc.

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ABSTRACT

Landfill leachate with its variable and complex characteristics poses a well established threat to the environment. Enhancement of the environmental quality through the minimization of the leachate problem should therefore be the major objective of good landfill management. The need to control and manage landfill leachate has resulted in various treatment alternatives which include both biological and physical-chemical processes.

The research described in this thesis discusses the feasibility of biological and physical-chemical treatment of leachate based on laboratory-scale reactors. After a short introduction, a review of the relevant literature on solid waste disposal including landfilling, leachate generation and the treatment alternatives was presented. Comparative experimental studies were then carried out using an aerobic rotating biological contactor (RBC), an upflow anaerobic filter (UAF) and an activated carbon (AC) adsorption column for treating landfill leachate. The effect of a range of parameters on the performance and operation of the RBC, the UAF and the AC column has been evaluated in the study

From the experimental results, an RBC was found to achieve a better performance when treating a low strength (LS) leachate, whereas a high strength (HS) leachate would be much better treated by a UAF. For the LS leachate treatment, a COD removal of 80% at a loading rate of 6 kg COD/m^3.d was achieved by the RBC as compared to only 60% by the UAF. Whereas for the HS leachate the RBC achieved a COD removal of only 50% at the loading rate of 14 kg COD/m^3.d as compared to 60% by the UAF.

Direct physical-chemical treatment process in treating leachate using an AC adsorption was also investigated. The results obtained showed that the adsorption process was not capable of achieving the desired effluent requirement, with 20% residual organic fractions still remaining in the effluent. The need to remove this biodegradable organic matter by biological processes was found to be necessary.

It is suggested that to achieve satisfactory treatment, anaerobic UAF treatment of leachate followed by aerobic RBC and a final polishing with AC column should be used.

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LIST OF ABBREVIATIONS AND NOTATIONS

AC	Activated Carbon
AOLR	Areal Organic Loading Rate ($g/m^2.day$)
AORR	Areal Organic Removal Rate ($g/m^2.day$)
BDST	Bed Depth-Service Time
BOD_5	Five-Days Biochemical Oxygen Demand (mg/l)
BOD_{5e}	Effluent Five-Days Biochemical Oxygen Demand (mg/l)
COD	Chemical Oxygen Demand (mg/l)
DO	Dissolved Oxygen (mg/l)
EBCT	Empty Bed Contact Time (min)
EMP	Efficiency of Methane Production (m^3 CH_4/kg COD removed)
$FCOD_e$	Effluent Filtered Chemical Oxygen Demand (mg/l)
GAC	Granular Activated Carbon
HRT	Hydraulic Retention Time (days)
HS	High Strength
HS S-S	High Strength Steady State
LS	Low Strength
LS S-S	Low Strength Steady State
MLSS	Mixed Liquor Suspended Solids (mg/l)
MLVSS	Mixed Liquor Volatile Suspended Solids (mg/l)
MSW	Municipal Solid Waste
NH_3-N	Ammonia Nitrogen (mg/l)
OLR	Organic Loading Rate ($kg/m^3.day$)
PAC	Powdered Activated Carbon
RBC	Rotating Biological Contactor
SRT	Solids Retention Time (days)
SS	Suspended Solids (mg/l)
TKN	Total Kjeldahl Nitrogen (mg/l)

TOC	Total Organic Carbon (mg/l)
TOC_e	Effluent Total Organic Carbon (mg/l)
TSS	Total Suspended Solids (mg/l)
UAF	Upflow Anaerobic Filter
VOLR	Volumetric Organic Loading Rate (kg/m^3.day)
VORR	Volumetric Organic Removal Rate (kg/m^3.day)
VFA	Volatile Fatty Acid (mg/l)
VSS	Volatile Suspended Solids (mg/l)

CHAPTER ONE

INTRODUCTION

Solid domestic, municipal and industrial wastes are commonly disposed of by placing them in landfill sites where they are left to decompose. In nature the organic fraction of these waste materials decompose biologically with the process being either aerobic or anaerobic. The slow decomposition process takes place spontaneously under the appropriate conditions. The process develops through the combined work of microorganisms in the presence of water in the form of precipitation which infiltrates the landfill site.

The liquid that is released from the landfill site is known as leachate. This leachate often contains a high concentration of organic matter and inorganic ions, including heavy metals. The combined effects of all of the characteristics of leachate give an awesome polluting potential that threatens receiving water bodies and the environmental quality in general (Brown et al, 1983; Ehrig, 1984; Robinson and Maris, 1985).

The leachate from landfills can seriously degrade the quality of both surface and groundwater and hence can be a potential hazard for human health. The art of landfill management is fast becoming the art of leachate control since this is the key to many of the problems associated with landfill disposal of wastes. The basic design and resulting management and site practices should be aimed at controlling the production of leachate to a minimum and preventing its movement to environmentally unacceptable areas.

Recently, concerned awareness of the deleterious nature of leachate has highlighted the need to treat these highly complex and hazardous discharges. This can be achieved by designing all landfills with provisions for collection and treatment of leachate, thereby making the selection of the landfill site and operation of the landfill more manageable. (Johansen and Carlson, 1976)

A number of options are available for the treatment of leachate including the biological and physical-chemical processes suitable for treating wastewaters. However, the degree of treatment required is dependent upon the nature and strength of the leachate. Also the design of treatment plant needs to be robust and flexible because the type of treatment may change from biological to a combination of biological and chemical processes as the emplaced wastes age.

From the early fifties, aerobic processes have been firmly established for the biological treatment of wastewater. Many small-scale experimental studies demonstrated that leachate from landfills containing domestic solid waste has been effectively stabilized using aerobic biological processes. Both small-scale batch aeration experiments (DeWalle and Chian, 1974; Chian and Dewalle, 1975; Robinson, 1980) and investigations using laboratory-scale continuous flow aeration units (Boyle and Ham, 1972, 1974; Cook and Foree, 1974; Palit and Qasim, 1977; Uloth and Mavinic, 1977; Stegmann and Ehrig, 1980; Zafpe-Gilje and Mavinic, 1981) have been reported.

Activated sludge treatment process and aerated lagoon were employed in most of the studies reported. Besides activated sludge treatment process and aerated lagoon, biological filter (Maris et al, 1984) and rotating biological contactor (Albers and Mennerich, 1986) have also been used especially for reducing the ammonia-nitrogen content of leachate from aged wastes.

However, aerobic treatment processes use the energy contained in the organic matter to produce large volumes of new organisms (sludge). Most recently, as a result of the environmental debate, reuse and energy conservation have become the current topics of research interest. Much attention has, therefore, been paid to the possible use of anaerobic processes for treating organic wastewaters, especially the stronger effluent from many manufacturing processes (Vigneswaran et al, 1986). Also the possibility of treating landfill leachate to recover resources such as biogas has made anaerobic treatment a viable technique.

The anaerobic process is, in many ways, ideal for waste treatment, having several significant advantages over other available methods. It has been used for many years for the stabilization of municipal wastewater treatment plant sludge and has considerable potential for the treatment of many industrial wastewaters (Anderson and Donnelly, 1977). Interest in anaerobic biotechnology has grown considerably, both in the harnessing of the process for industrial wastewater treatment and in the bioconversion of crop-grown biomass to methane (Chynoweth et al., 1979; Moo-Young et al, 1979; Sheridan, 1982; Stenroos et al, 1986).

When first introduced several decades ago, these units were unmixed and unheated; hence the process was very slow and inefficient. More recently, significant advances in both the basic understanding of the anaerobic process and the engineering application of this process have taken place. Significant research contributions have been made by the United States and European microbiologists. This new interest, supported by advances in process engineering, has been translated into numerous treatability studies in the field of wastewater treatment (Speece, 1983).

The research and development of heated, completely mixed, high rate suspended growth systems then took place. These innovations have lowered the retention time to 15 days or less, and greatly increased the allowable organic loading rates. The novel application of fixed film processes in the anaerobic treatment of wastewater was also developed. Several full-scale suspended growth and fixed film systems are now available on a commercial scale, such as the anaerobic contact process, the upflow anaerobic sludge blanket process, the upflow anaerobic filter process, the anaerobic expanded/fluidized bed process and the downflow anaerobic stationary fixed-film process.

The requirement for achieving higher quality effluent from municipal and industrial waste treatment processes has led to the use of physical-chemical treatment processes (Weber et al, 1970; Annesini et al, 1987; Bencheikh-Lehocine, 1989). Although physical-chemical treatment is usually carried out as tertiary treatment process, treatment of strong raw wastewater directly using a physical-chemical treatment process with complete elimination of the biological processes has been proposed and results of pilot studies reported (Hager and Reilly, 1970; Weber et al, 1970; Rebhun and Streit, 1974).

A study of the treatability of leachate by physical-chemical methods such as precipitation, coagulation, oxidation and carbon adsorption has also been investigated (Thornton and Blanc, 1973; Ho et al, 1974; Bull et al, 1983). Bull et al (1983) investigated the usage of biological processes in treating leachate, and results indicated that for surface water discharge of treated leachate, a post-treatment by lime addition to increase the pH and ammonia stripping was required.

The purpose of this study is to compare the performances of the aerobic, anaerobic and adsorption processes for the treatment of landfill leachate.

CHAPTER TWO

LITERATURE REVIEW

2.1 Solid Waste Disposal

Solid waste disposal is an urgent and critical problem perpetuated by the rapid urbanization and industrial development. Tchobanoglous et al (1977) defined solid wastes as all those wastes arising from human and animal activities that are normally solid in nature and are discarded as useless or unwanted. Waste materials can be both putrescible and nonputrescible. Solid wastes are produced from mining operations, commercial, agricultural, domestic, and industrial activities.

The quality and quantity of solid wastes produced are greatly influenced by the climate, habits, standard of living, and the nature of the waste. Some of the wastes from industrial activities are hazardous to health and present a serious pollution problem. Unfortunately, very little information is available on the quantities of hazardous wastes generated in various industries (Peavy et al, 1986). Therefore it is pertinent to handle and dispose all solid wastes with care.

2.2 Waste Disposal by Landfilling

Although the recycling of raw materials and energy recovery from wastes is being given considerable attention, landfilling is still the most common and economical method for the ultimate disposal of urban and industrial solid wastes despite the several disadvantages (Schmidell et al, 1986; Gourdon et al, 1989).

The disadvantages of landfilling are mainly:

i) the possibility of the leachate contaminating the ground water table,

ii) the low potentiality of biogas utilization, and

iii) the problem of finding a suitable area in the vicinity due to health risk.

Sanitary landfill, as it is known in the USA, is defined by the American Society of Civil Engineers as:

> "A method of disposing of refuse on land without creating nuisance or hazards to public health or safety, by utilizing the principles of engineering to confine the refuse to the smallest practical area, to reduce it to the smallest practical volume, and to cover it with a layer of earth at the conclusion of each day's operation, or at such more frequent intervals as may be necessary"
>
> (Baum and Parker, 1974)

Therefore a landfill site should not be seen simply as a euphemism for the old open dump or uncontrolled tip. A modern landfill should conform to strict codes of practice, based on sound engineering principles. In the United Kingdom, the landfilling of wastes is governed by guidelines which are set out by the Department of the Environment (DoE, 1986). Unfortunately, leachate which is generated in landfills often leads to the pollution of groundwater and municipal water supply (Chian and DeWalle, 1976).

In order to minimize the potential hazardous effects of gas and leachate generated from solid waste disposal, the best solution as proposed by Cook and Foree (1974) and Gourdon et al (1989) is to give special attention in the selection of the landfill site. A proper selection of landfill site not only could reduce the health risk but also could safeguard the public from the resulting hazard.

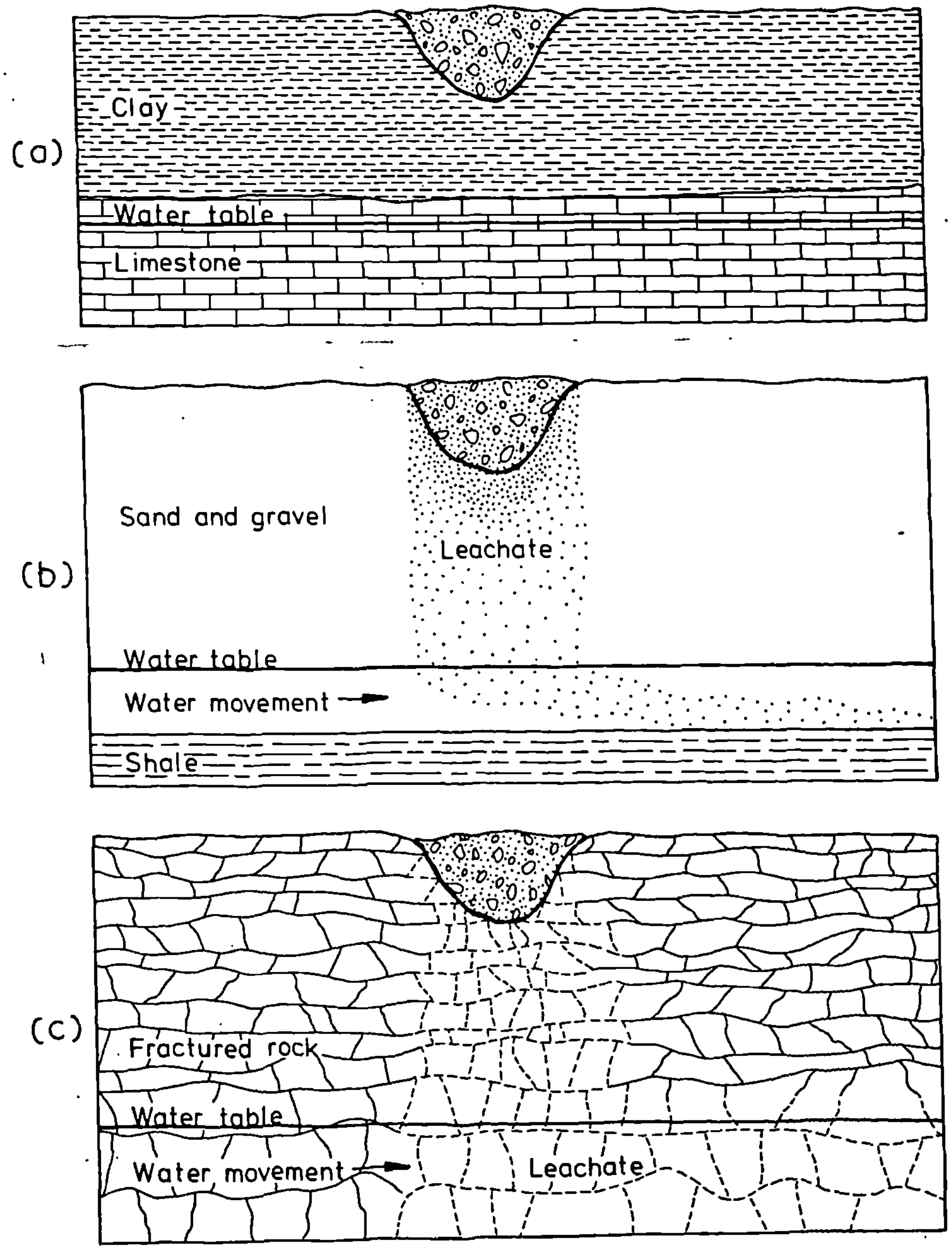
Effective management starts with containment of the gas and leachate followed by treatment and disposal in an environmentally and economically sound manner (Kang et al, 1989). Therefore, the design, construction and operation of the landfill site must be properly planned. The landfill site should also be designed with provisions for collection and treatment of the leachate produced.

2.3 Types of Landfill Sites

Basically the landfill sites can be categorized into three types, depending on the nature of the solid wastes deposited, and environmental and climatic conditions (Wilson, 1981). The types of landfill sites, as illustrated in Fig. 2.1, are:

i) attenuate and disperse (non-containment site),
ii) concentrate and contain (containment site), and
iii) rapid migration site.

Evison (1978) reported that for the disposal of toxic chemical wastes, only "attenuate and disperse" and "concentrate and contain" sites are suitable. A rapid migration site is only suitable for the disposal of relatively inert wastes.



(a) *Class 1 or containment site*. The leachate is contained within the landfill by the impermeable clay.

(b) *Class 2 or attentuation site*. The sand and gravel allows slow leachate migration with significant attenuation. The underlying, impermeable, shale stratum protects deeper aquifers.

(c) *Class 3 or rapid migration site*. The fissured strata allow rapid leachate migration into the aquifer, with insignificant attenuation.

Figure 2.1: Types of Landfill Sites (Wilson, 1981)

2.3.1 Attenuate and Disperse

In non-containment sites the seepage of leachate to the environment occurs slowly. The risk of the waste polluting the environment in such a site is prevented by the attenuation mechanisms operating within the waste and in the strata beneath, and adjacent to, the landfill site. The mechanisms include those of dilution and dispersion which assist by reducing the effect of leachate on water resources.

The rate of leachate migration within a landfill site, as pointed out by the Department of the Environment (DoE, 1986), must not be too fast in order to allow for attenuation to take place through physical-chemical and biological processes. The effective operation of these processes in abetting attenuation is influenced by the geochemistry of the strata and the prevailing hydrogeological conditions. The uncertainty about the mechanics of attenuation precludes accurate prediction of the pollution risk at such sites. The United States Environmental Protection Agency (USEPA) and indeed several other environmental agencies restrict the use of non-containment sites.

The use of non-containment sites in the UK is still accepted, perhaps due to good past and present landfill management practice. However some researchers still regard this type of landfilling practice as unacceptable (Cope et al, 1983), and it should never be used for disposal of hazardous wastes. The disapproval of using non-containment sites for the disposal of solid wastes is further substantiated by the accident at Love Canal landfill site in Niagara, where a school and housing estate had to be evacuated (McDougall et al, 1980).

2.3.2 Concentrate and Contain

Containment sites are designed to prevent the wastes deposited, and subsequently the leachate generated, from escaping to the environment. This is undertaken by making use of the geological strata of naturally low permeability to prevent leachate percolation, or by lining sites artificially either by the use of clay or man-made materials. Attenuation mechanisms take place almost completely within the body of the waste and through the passage of time these processes result in the reduction of the organic strength of leachate produced.

On completion, the landfill must be capped using an impermeable or low permeability material, to safeguard against vermin and emission of foul odours. Capping layers, which are usually contoured to encourage runoff, also assist in preventing rain or surface water from saturating the wastes. (Haxo, 1979).

Generally, containment sites are not suitable for the disposal of large volumes of liquid wastes. Disposing of large volumes of liquid wastes will inevitably saturate the landfill site unless water entry is minimized by capping. However, the prevention of water ingress will tend to reduce the degradation rate of organic wastes which will delay stabilization of the site. In this case, it is advisable to design and construct the site with provision for collecting the leachate for either treatment or subsequent discharge to a receiving watercourse or sewer. When properly managed, this type of landfill site is a safe method of waste disposal when compared to non-containment sites.

2.4 Potential Problems Associated with Landfill

Landfill is not without potential problems but, through proper landfill management, these can be overcome, or at least controlled, to acceptable levels. This practice involves planning and application of sound engineering principles and construction techniques (General Electric, 1975). The problems usually associated with landfill are:

i) pollution of ground and surface waters through indiscriminate site selection and landfill management,

ii) risk of explosion from landfill gas in nearby properties, and death of vegetation due to landfill gas, and

iii) settlement.

2.4.1 Pollution of Ground and Surface Waters

Injudicious site selection and landfill management results in the leaching of pollutants from the solid wastes into ground and surface waters. Rainfall precipitating through the emplaced waste will further aggravate the situation causing groundwater contamination (Kennedy et al, 1988). Landfill leachate will be discussed in detail later as the study involves the treatment of leachate.

2.4.2 Landfill Gas

Gases are produced to a greater or lesser extent in all sites containing biodegradable wastes. Hillman (1988) stated that awareness of the presence of gas whilst not new, has become more pronounced with changes in domestic waste compositions, landfill management and site engineering practices particularly with regard to leachate control.

Landfill gas produced in landfilling of solid wastes is predominantly composed of methane and carbon dioxide, although traces of hydrogen, carbon monoxide and hydrogen sulphide may also be present (Hill, 1985). The landfill gas originates through methanogenesis of organic wastes in sanitary landfills. Gendebien and Nyns (1991) in their study of sanitary landfilling biotechnology reported that the landfill gas generated in landfills could be controlled and exploited through proper site management.

If not properly monitored and controlled landfill gas could give rise to flammability, toxicity, asphyxiation and explosive hazards (Richards, 1988; DoE, 1989). The presence of methane between 5 to 15 percent in the air may result in explosion. Several cases have been reported on the destruction of buildings erected on former landfill sites, or even at a distance of several hundred metres from such a site, and some of the accidents reported were also fatal (Hill, 1985). The first experience of a landfill gas explosion in the United Kingdom was at Loscoe landfill site in Derbyshire injuring 3 people. In the United States of America, 9 people were killed in explosions caused by landfill gas and a further 57 have been injured (Hoather and Wright, 1988).

Carbon dioxide and other gases are toxic to some vegetable root systems. Stonell (1985) stated that the harm to vegetation is primarily due to the displacement of oxygen around the roots by landfill gas, thus causing extensive die-back.

2.4.3 Settlement

Settlement of putrescible waste is inevitable. This is due to the aerobic and anaerobic breakdown of the waste as well as the incomplete compaction of the waste during the landfill operation. Studies have shown that 90 percent of the ultimate settlement occurred in the first 5 years after a landfill is completed (General Electric, 1975).

Uneven settlement of the landfill can have significant effects on the restoration of a landfill site, such as soft spots, ponding, soil loss and damage to agricultural machinery. Incomplete settlement can also damage site capping, resulting in surface water ingress. This will increase the landfill problems.

2.5 Problems of Leachate

The design of any treatment processes is largely dependent on the waste to be treated. The characteristics of the waste and the processes by which it is generated must also be studied. The analysis made on the waste will assist determining which treatment processes should be considered.

The waste related to this study is leachate which is obtained from a landfill site. Leachate is an unpleasant, odorous liquid which contains a high concentration of organic matters, inorganic ions and heavy metals in an electrochemically reduced environment. The pollutants of concern in leachate are copper, lead, zinc, ammonium, potassium, sodium, magnesium, iron, BOD_5, COD, nitrate and sulphate (Amalendu, 1982).

The composition of leachate is affected by the nature of the solid wastes deposited in the landfill, the earth surrounding the landfill and whether the composition is aerobic or anaerobic. Secondary factors influencing the composition are: the quantity of solid wastes disposed, time of storage or landfill age, degree of compaction, amount of water in contact with the solid wastes and the temperature.

The volume of leachate generated from a landfill is difficult to estimate. Using data collected from 15 landfill sites of less than 12 years, Ehrig (1983) has shown that an average of 4.7 m^3/ha.day (range of 0.4 to 10.6 m^3/ha.day) of leachate was produced.

Amalendu (1982), based on field observations and experimental works carried out in the laboratory, gave a generalized concentration variation plot, as illustrated in Fig. 2.2.

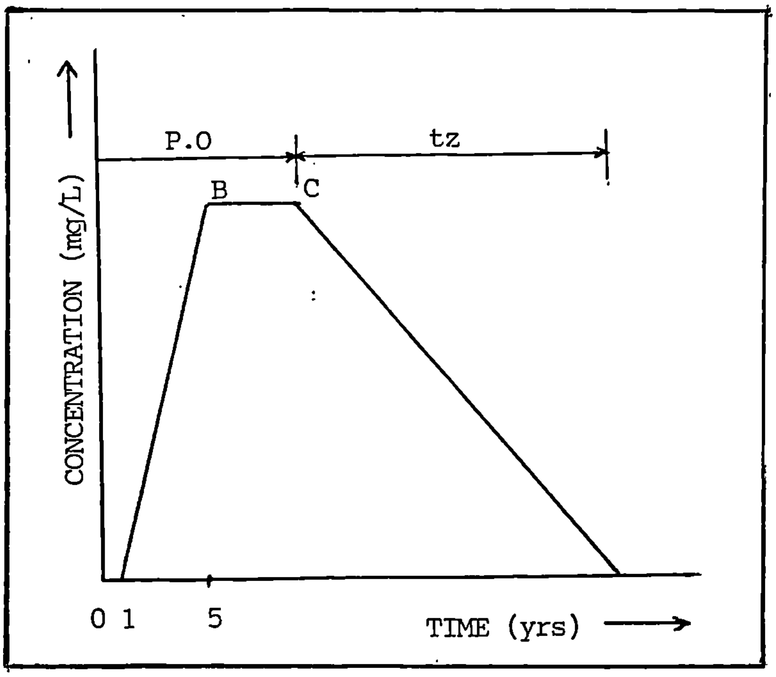


Figure 2.2: A Generalized Concentration Variation Plot
(Amalendu, 1982)

The figure could be interpreted as follows:

i) the period from the start of landfill up to its closure is represented by P.O. For P.O equals five years, point C coincides with point B, and

ii) the time (since closure of landfill) by which time the leachate concentration will be low enough so as not to cause any environmental problems is denoted by tz.

2.5.1 Control of Leachate Production In Landfill Site

The management and site practices for a landfill can affect the way in which the landfill matures and stabilizes, consequently the nature of leachate produced may be controlled to a certain extent. Landfill stabilization is affected by moisture, pH, temperature, particle size and solid wastes density. The stabilization of landfill described by Pohland et al (1983) includes: initial lag or adjustment, transition from aerobic to anaerobic stabilization, acid formation, methane fermentation and final maturation. This was later updated by Barlaz et al (1989) who divided the decomposition processes of the solid wastes into four phases:

i) an aerobic phase,
ii) an anaerobic acid phase,
iii) an accelerated methane production phase, and
iv) a decelerated methane production phase.

The composition of municipal solid wastes (MSW) varies considerably from one municipality to another. However, the ranges of values that are typically found in municipal solid wastes and their relative distribution are reported in Table 2.1.

Table 2.1 Typical Composition of MSW

Component	Percent by Weight			
	Range	Typical	Davis California*	Merida Venezuela*
Food Wastes	6 - 26	14	8.3	27.4
Paper	15 - 45	34	35.8	15.5
Cardboard	3 - 15	7	10.9	13.0
Plastics	2 - 8	5	6.9	4.6
Textiles	0 - 4	2	2.5	2.3
Rubber	0 - 2	0.5	2.5	0.4
Leather	0 - 2	0.5	0.7	1.3
Garden trimmings	0 - 20	12	10.8	5.8
Wood	1 - 4	2	1.9	3.6
Misc. organic	0 - 5	2	2.0	0.6
Glass	4 - 16	8	7.5	10.3
Tin cans	2 - 8	6	5.1	8.3
Nonferrous metals	0 - 1	1	1.6	0.1
Ferrous metals	1 - 4	2	2.2	1.2
Dirt,ashes,brick	0 - 10	4	1.3	5.6

* Based on measurements made during the month of October over a 5-year period (1978 through 1982)

* Based on measurements made during the month of July over a 3-year period (1978 through 1980)

Source: Peavy et al, 1986

Table 2.2 presents a typical approximate and ultimate analysis data for the components in MSW (Peavy et al, 1986).

2.5.2 Leachate Production

Most water which enters landfill refuse cells will not appear as leachate until all the refuse layers have reached the field capacity (i.e all voids are filled with water). This water is needed to sustain the microbial processes or organic decomposition. As stated above, landfill leachate is produced by a combination of:

i) the activity of microorganism within the landfill, and

ii) the action of water as it percolates through the landfill.

Although this is a simple means of explaining the origins of a landfill leachate, it is far from accurate and gives no indication of the mechanisms involved. In fact, there are several mechanisms and stages involved before a particular compound or element appears as a contaminant in the leachate. The processes that are responsible for the appearance of the ions of heavy metals and cations in the leachate are primary leaching and chemical reduction. These inorganic matters act as inhibitors in biological processes and are toxic to aquatic life (Klerks and Weis, 1987).

Other processes involved, which may either be chemical or biological, are too numerous and complex to be considered individually in this study. However, for convenience and as an aid to understanding as to what is happening within the landfill, the mechanisms by which contaminants are leached out can be depicted as in Fig. 2.3.

Table 2.2 Approximate and Ultimate Chemical Analysis of MSW

	Value, percent*	
	Range	Typical
Approximate analysis		
Moisture	15.0 - 40.0	20.0
Volatile matter	40.0 - 60.0	53.0
Fixed carbon	5.0 - 12.0	7.0
Noncombustibles	15.0 - 30.0	20.0
Ultimate analysis (combustibles components)		
Carbon	40.0 - 60.0	47.0
Hydrogen	4.0 - 8.0	6.0
Oxygen	30.0 - 50.0	40.0
Nitrogen	0.2 - 1.0	0.8
Sulphur	0.05 - 0.3	0.2
Ash	1.0 - 10.0	6.0
Heating Value#		
Organic fraction, kJ/kg	12,000 - 16,000	14,000
Total, kJ/kg	8,000 - 12,000	10,500

* By weight

\# As-discarded basis

Source: Peavy et al, 1986

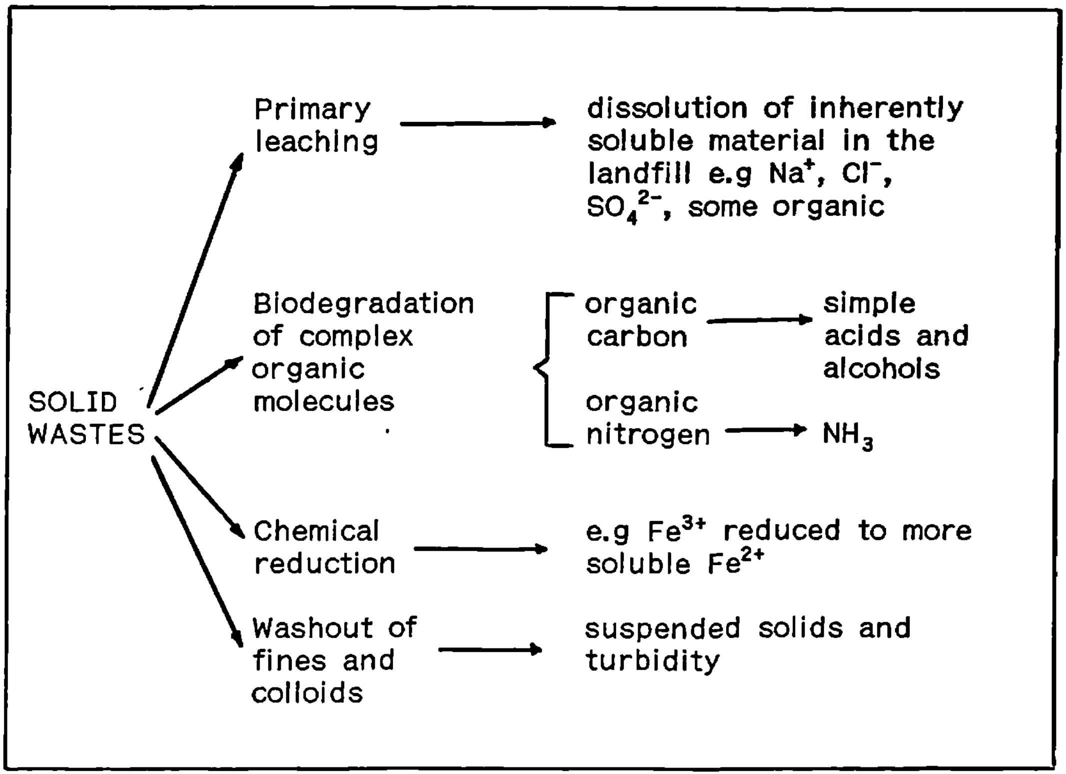


Figure 2.3: Schematic Representation of the Main Mechanism by which Material is Leached from a Landfill (Crawford and Smith, 1985)

The biodegradation of organic matters within a landfill affects the environment and consequently the chemical reactions that takes place. This microbial activity starts as soon as the solid wastes are deposited. The initial decomposition is aerobic where, within the first two years of the landfill operation, the action of aerobic bacteria on organic matter depletes the level of oxygen as the fill settles. The aerobic biodegradation of organic matter is given in Fig. 2.4a. During this "maturing period" of the landfill the changes in the microbial population within the landfill are reflected in the leachate and the gas which is produced.

The maturing period will be followed by a stabilization period in which the activity of the microorganisms within the landfill continues in an anaerobic steady state condition. The number of bacteria present and their activity will continue until they deplete the organic matter on which they feed. The organic matter conversion by anaerobic bacteria is shown in Fig. 2.4b.

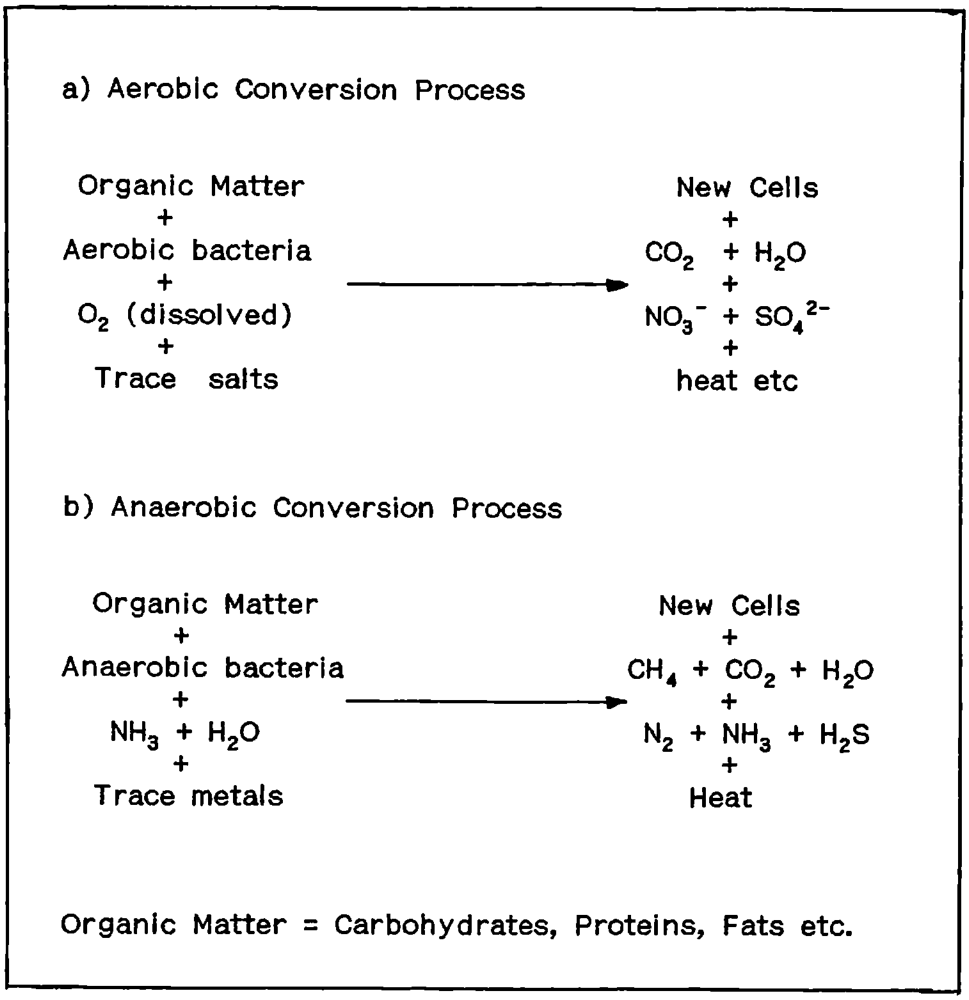


Figure 2.4: Aerobic and Anaerobic Biodegradation of Organic Matter

Anaerobic decomposition is the dominant reaction in the formation of leachate. The anaerobic organic reactions shown above occur in two distinct stages. The first is liquification and acid formation, where complex organic compounds are broken down principally to volatile fatty acids. The simplest of these is acetic acid. In the second stage, the volatile fatty acids are further broken down into gaseous end-products.

The nitrogen system also undergoes a biochemical process reducing nitrite and nitrate to nitrogen gas (denitrification). Ammonia gas is formed from organic compounds that contains nitrogen. It is present throughout the period of high BOD_5 of the leachate and always in concentrations of the order of 100 - 200 mg/l, and this level will begin to rise as the landfill stabilizes (Weiss, 1974). Ammonia may rise to a peak of greater than 700 mg/l which may occur when the BOD_5 has already decreased substantially from its peak value. Furthermore, levels in excess of 100 mg/l may continue to exist for several years after the BOD_5 has finally reduced to its low, stabilized level.

2.5.3 Leachate Generation (Water Balance Analysis)

A method which has been developed to reasonably estimate the amount of leachate produced at a landfill, is the Water Balance Analysis (Crawford and Smith, 1985). The various components of the moisture which need to be considered to evaluate leachate at a typical sanitary landfill site are precipitation, surface runoff, infiltration of surface and groundwaters, evapotranspiration and soil moisture storage. The principal source generally being precipitation over the landfill site. Whenever the moisture exceeds the field capacity of the soil it percolates down to the solid waste.

The addition of moisture to solid waste over a period of time saturates the solid waste to its field capacity moisture content. At that stage moisture from the solid waste percolates to the virgin ground below in the form of leachate. The rate of moisture percolation to the solid waste which in turn, after an initial delay, is equal to the rate of leachate generation. Calculations on the water balance in landfill sites are based on the principle that any water which enters the waste and does not evaporate or remain stored within it, must leave by percolation or as surface seepage. A schematic diagram showing the various parameters affecting the water balance at a landfill is given in Fig. 2.5. The volume of leachate produced is affected by the absorptive capacity of the waste, which is, in turn, a function of the degree of compaction and of the nature of the waste. In practice, the leachate volume produced is difficult to estimate, and for this reason many leachate treatment plants are often designed after waste emplacement.

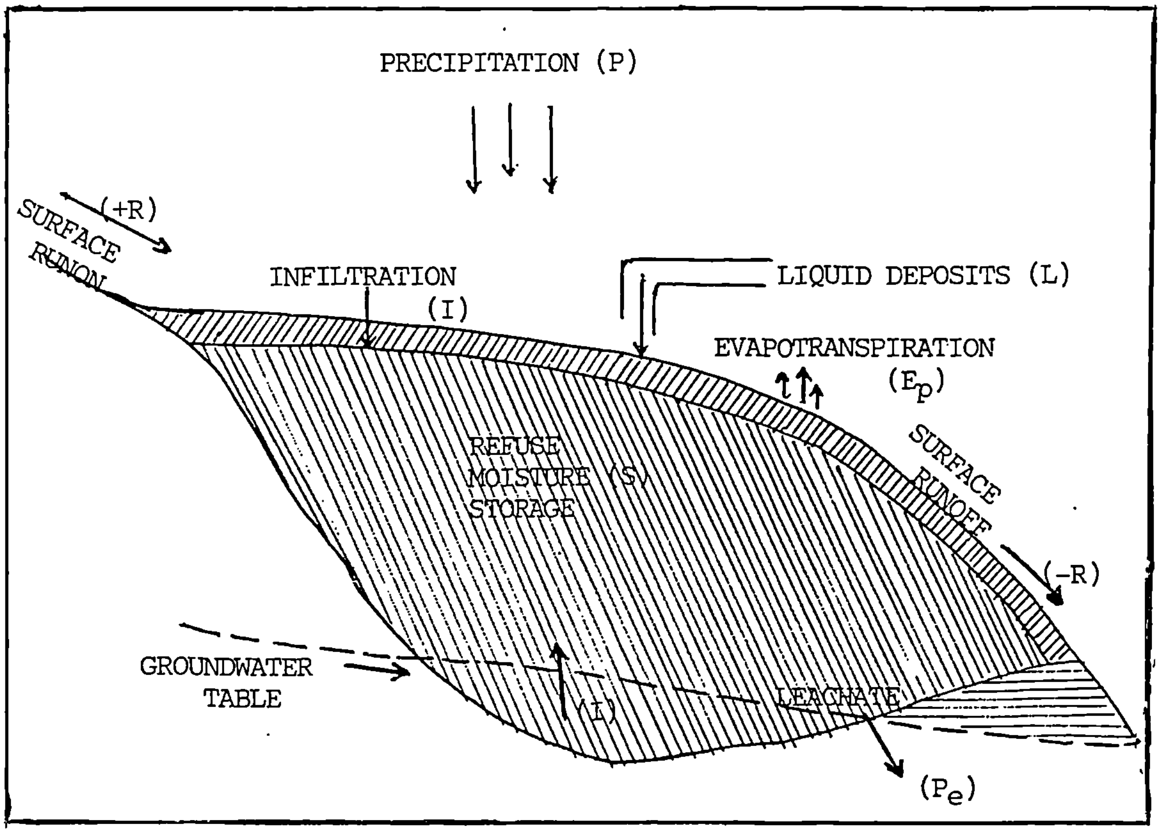


Figure 2.5: Water Balance Calculation of a Landfill Site (Crawford and Smith, 1985)

The form of balance applicable to landfills may be stated as:

$$P_e = P + I + L \pm R - S - E_p \qquad (2.1)$$

where

P_e = leachate production

P = precipitation

I = infiltration of surface and groundwater

L = liquid deposits

R = run-off (-) run-on (+)

S = liquid absorbed by the emplaced wastes

E_p = actual evaporation

2.5.4 Leachate Characteristics and Pollutants

Chemical constituents and their concentration in leachate vary over a wide range. This variability is due to the management and site practices such as solid waste composition, landfill age and climate (Boyle and Ham, 1974). Mennerich and Albers (1986) reported that leachate from recently emplaced wastes contained high concentrations of dissolved organic matter with up to 80% of the COD may be accounted for as volatile fatty acids, indicating hydrolysis and acidification processes are taking place within the landfill.

Robinson and Maris (1979) reported the range of values of the composition of 23 samples of leachate collected from sites of various ages in the UK obtained by the Water Research Centre (WRC) and are summarized in Table 2.3. High variation in the composition of leachate was also reported by Pohland and Kang (1975), Johansen and Carlson (1976), Ghassemi et al (1984) and Millot et al (1987).

Table 2.3 Leachate Composition from Various Aged Landfill Sites in the United Kingdom

Constituent		Range All values except pH in mg/l Minimum	 Maximum
pH		6.2	7.45
BOD_5		2	8000
COD		66	11600
TOC		21	4400
Ammoniacal-Nitrogen		5	730
Organic-Nitrogen		not detectable	155
Nitrate-Nitrogen		0.5	4.9
Nitrite-Nitrogen		0.02	1.84
Ortho-Phosphorus		0.02	4.43
Chloride		70	2777
Sulphate		55	456
Sodium	Na	43	2500
Magnesium	Mg	12	480
Potassium	K	20	650
Calcium	Ca	130	1150
Chromium	Cr	0.005	0.14
Manganese	Mn	0.19	26.5
Iron	Fe	0.09	380
Nickel	Ni	0.02	0.16
Copper	Cu	0.004	0.15
Zinc	Zn	0.05	0.95
Cadmium	Cd	0.005	0.013
Lead	Pb	0.05	0.22

Source: Robinson and Maris (1979)

Carter et al (1984) have established some data on leachate characteristics from the studies made using household and commercial solid wastes. Table 2.4 gives a comparative composition of leachate from fresh and aged wastes, where aged and stabilized wastes are considered to be greater than 5 years. It should be noted that the strength of leachate from fresh wastes is some 35 times stronger than domestic sewage.

The complex interaction between physical-chemical and biological processes occurring within the landfill, makes it difficult to predict leachate quality at any given site. Variability in the composition of leachate was studied at the Water Research Centre and is indicated in Fig. 2.6, where ammoniacal-nitrogen and BOD_5 concentrations in leachate samples from various sites are compared.

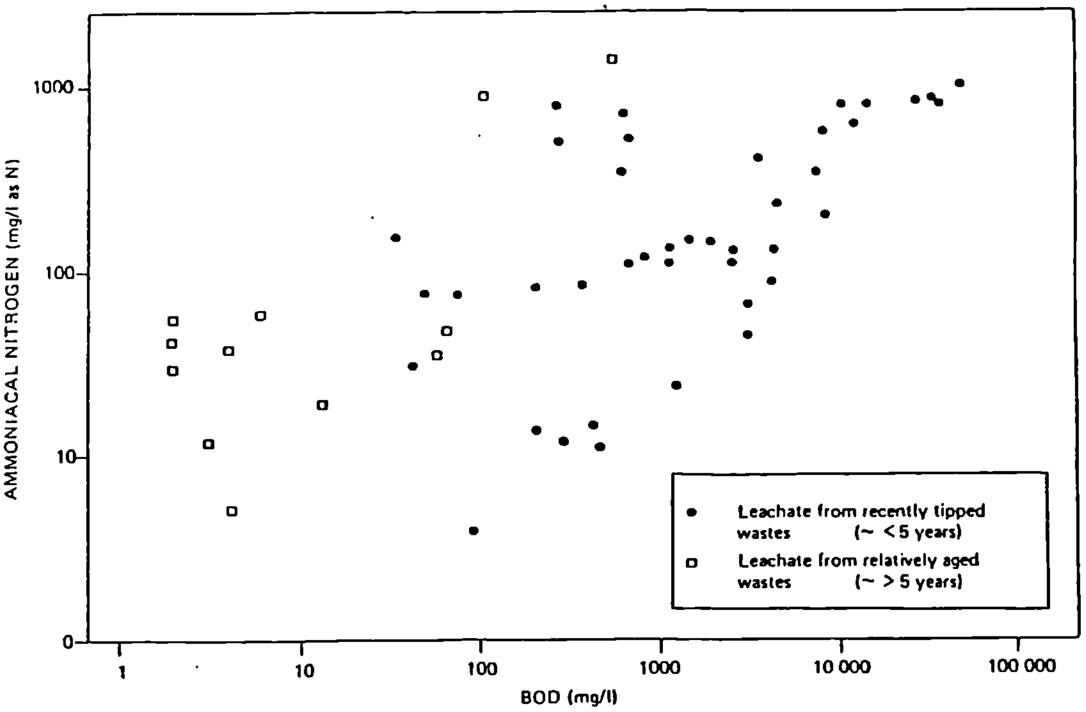


Figure 2.6: Concentration of Ammoniacal-Nitrogen Plotted Against BOD_5 for Leachate (Robinson and Maris, 1979)

Table 2.4: Comparison of Leachate Compositions between Fresh and Aged Wastes with Stevenage Domestic Sewage

All results in mg/l except pH

Constituent	Fresh wastes	Aged wastes	Domestic sewage
pH	6.2	7.5	7.5
COD	24000	700	700
BOD_5	13600	70	400
TOC	8000	400	240
Volatile fatty acids (as C)	6000	5.0	40.0
Ammoniacal N	600	260	46
Oxidized N	0.5	7.5	0.5
Orthophosphate	0.7	0.5	14.0
Chloride	1300	1400	120
Sodium	960	880	100
Magnesium	250	130	4.5
Potassium	780	340	20
Calcium	1820	200	110
Chromium	0.56	0.07	0.005
Manganese	26.5	1.7	0.07
Iron	540	10	0.03
Zinc	21.5	0.2	0.16

Source: Robinson and Maris, 1979

Studies made by Levine et al (1985) on leachate characteristics illustrated that over a thousand compounds have been identified in various leachates. These compounds can be classified into a size range based on molecular size. An overview of the types and size ranges of contaminants, typically found in leachates is presented in Fig. 2.7. In general, the inorganic constituents in leachate can be divided into four major size categories designated as group I - IV. The composition of each group varies widely depending on particular site characteristics.

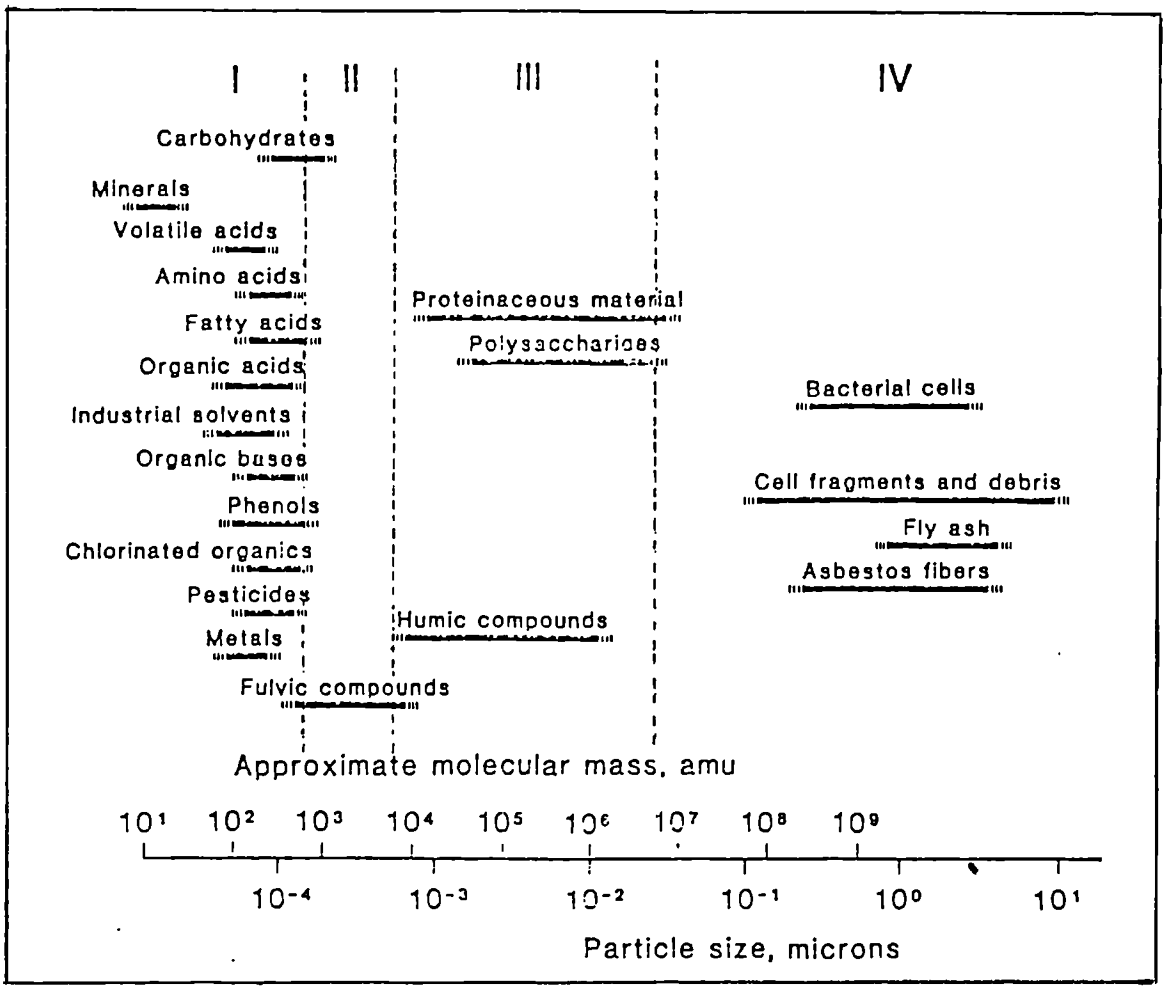


Figure 2.7: Size Range of Contaminants in Leachate (Levine, 1985)

The main physical aspects of leachate affecting water quality are suspended solids, colour, turbidity and temperature. The first three characteristics will reduce the light intensity in the receiving water and therefore, reduce the oxygen level by reducing photosynthetic activity. Suspended solids may also smother plant life and benthic organisms, affecting food chains and further reducing the level of photosynthesis. A change in average stream temperature of only a few degrees will effect the flora and fauna of a stream, by disrupting the life cycles of certain organisms and interfering with the delicate respiratory systems of others.

- Organic Pollutants

A general trend of leachate composition can be identified as a landfill ages. Untreated leachate mainly from young landfill is highly polluting due to the readily biodegradable organic matters, such as volatile fatty acids, and gives a high ratio of BOD_5:COD. The deleterious effect of a high BOD_5 waste discharged into a surface water source is well-known. However, in the case of leachate pollution, the oxygen deficit downstream of the discharge point may not be the only factor to be considered. Leachate may contain toxic organic chemicals (such as phenol) and other refractory organic compounds which may have an adverse effect on the flora and fauna of a receiving water, and under extreme conditions may render the river sterile (Maris et al, 1984).

The first indication of the contamination of surface water sources by toxic chemicals, is the presence of dead or dying fish. Typical examples of the organic toxins which may be present in a leachate and their approximate lethal concentrations are given in Table 2.5. These values are only an approximate guide as there are many experimental difficulties in determining the toxicity of substances to fish.

Table 2.5: Toxicity of Some Organic Compounds

Chemical	Species	Lethal dose (mg/l)	Exposure time (hr)
Acetic acid	goldfish	423	20
Aniline	brown trout	100	48
DDT	goldfish	0.027	96
	brown trout	0.32	36
Napthalene	perch	20	1
Parathion	fathead minnow	1.4 - 2.7	96
	minnow	20	4
	rainbow trout	6	3
	perch	9	1
Tannic acid	goldfish	100	200

Source: Klein, 1972

As industry is producing new organic compounds, resulting in unknown environmental effects, the presence of these compounds needs to be studied. The biodegradation of existing toxic organic chemicals cannot be assured, and the possible presence of an "unknown quantity", which may have disastrous consequences if it appears in the landfill leachate, means that careful consideration is advisable before allowing such wastes to be disposed together with domestic solid waste.

- Inorganic Pollutants

The main inorganic pollutants that can cause problems with landfill leachates are ammonia, iron, heavy metals and to a lesser extent chlorides, sulphates, phosphates and calcium. Leachates from aged wastes often contains a high level of ammonia even though the ratio of BOD_5:COD may have fallen to low values. Ammonia-nitrogen in surface waters can present a problem at concentrations as low as 0.4 mg/l.

Leachates containing ferrous iron are particularly objectionable as they can produce "ochre" deposits. Chemical oxidation occurs due to the reaction with dissolved oxygen to produce ferric compounds thus exerting an oxygen demand on the river. Biochemical oxidation also occurs, resulting in a reddish-brown bacterial slime containing ferric hydroxide. Apart from the oxygen demand exerted by these bacteria, the turbidity they produce will cut out light changing the flora and fauna of the river.

Odour may be a major source of irritation and is often associated with reduced sulphur compounds, which appear to be an environmental nuisance rather than a toxicity hazard. Heavy metals can be toxic to fish at relatively low concentrations. Examples of these lethal limits are given in Table 2.6.

Again these values are only a guide to the possible lethal doses as the conditions in the river may have a significant effect. Some work on the adaptation of aquatic organisms to long term exposure has been carried out, but only on simple life forms (Klerks and Weis, 1986).

Table 2.6: Toxicity of Some Inorganic Compounds

Chemical	Species	Lethal dose (mg/l)	Exposure time (hr)
Cadmium	goldfish	0.017	43 - 48
	stickleback	0.3	190
Copper	goldfish	0.019	3 - 7
	stickleback	0.02 - 0.03	160 - 190
	salmon	0.18	-
Lead	goldfish	25	96
	stickleback	0.1	336
	rainbow trout	1.0	100
Nickel	goldfish	1.0	200
	stickleback	1.0	156
Tin	goldfish	100	180
	salmon	4.8	-
Zinc	goldfish	100	120
	stickleback	0.3	204
	rainbow trout	0.5	64

Source: Klein, 1972

2.6 Treatment and Disposal Options

The minimization of the leachate problem is the major objective of good landfill management. This can be achieved by controlling the production of leachate through surface sealing or by containment techniques, revegetation to promote evapotranspiration and contour grading. These control techniques, which are comparatively expensive, are rarely used. As a result leachate treatment techniques have been implemented.

Several studies have been reported in the literature on the treatment of leachate from sanitary landfills (Ho et al., 1974; Boyle and Ham, 1974; Pohland, 1980; Bull et al, 1983; Wu et al, 1988). The variable and unpredictable nature of landfill leachate, such as its volume and quality which are subject to seasonal variation, presents operational difficulties for treatment processes as compared to the treatment of wastewater with consistent strength and volume.

These changes, together with the progressive changes of leachate characteristics as the landfill ages, means that an appropriate treatment technique for "young" landfill may not be effective for "aged" landfill. Hence, general recommendations for special treatment processes for leachate are not possible. In order to adapt to these changes, the treatment system should be flexible. Young and Maris (1986) proposed a multiple treatment system, the first phase is for the removal of degradable organic matters, whilst the second phase removes inorganic constituents, particularly ammoniacal-nitrogen.

Different forms of treatment techniques were tried, though as yet no one method has been put forward as an environmentally acceptable and cost effective means of dealing with the problems posed by landfill leachate. The overall objective for the treatment of leachate must reflect the selected disposal outlet (Otieno, 1989). Where landfills are located in the vicinity of a wastewater treatment plant, leachate can be treated along with the wastewater at the treatment plant. For a UK treatment plant, it would require a leachate BOD_5 of less than 300 mg/l and a total heavy metal content of less than 1 mg/l.

It should be noted that the problem of leachate treatment has only recently aroused the interest of wastewater and water resources engineers, mostly in the area of groundwater pollution, and as a consequence, data on treatment alternatives have been difficult to obtain. The dominant way of disposal is by discharging the leachate to sewer, land water courses or tidal waters under consent conditions issued by the regulatory authority concerned (Dass, 1977). Treatment of leachate discharged to sewer takes place at the treatment plant which is a form of off-site treatment. At coastal landfill sites, disposal via sea outfall in admixture with wastewater may be the least expensive process (Robinson and Maris, 1979).

The on-site leachate treatment generally produced a quality of effluent which is not suitable for discharge to inland surface waters. The on-site treatment of industrial wastewaters is, however, carried out in a number of industries in order to meet consent conditions set by regulatory agencies. In the UK, sewer discharges are controlled by the Regional Water Companies whereas river and groundwaters discharges are controlled by the National Rivers Authority (NRA). The treatment and disposal options will be briefly discussed in the following sections.

2.6.1 Combined Treatment with Municipal Wastewater

Treatment of leachate in combination with domestic wastewater at a municipal wastewater treatment plant is a potential disposal technique where access to the sewer system is available near to a landfill site. Experimental studies to determine the proportion of leachate which can be tolerated in a treatment plant influent, without causing deterioration in effluent quality had been conducted by several researchers. However, it is rather complicated to compare the results obtained by them due to differences in compositions of both leachate and domestic wastewater, as well as the differing experimental procedures.

Biological oxidation of leachate is inhibited by the high organic and ammonia content, leachate load variability, the low phosphate content and the possible presence of trace metals. Boyle and Ham (1974) investigated the treatment of various proportions of leachate with domestic wastewater (BOD_5 140 mg/l) in a laboratory-scale activated sludge plant. They reported that the addition of 2 per cent leachate, having a BOD_5 of 8800 mg/l, to the wastewater had no noticeable effect on the process. When this was increased by 5 per cent (a total daily organic loading of approximately 0.15 kg BOD_5/kg MLSS), the effluent quality was impaired and its BOD_5 increased by 50 per cent. Additions of leachate beyond 5 per cent resulted in substantial solids production, increased oxygen uptake rates, generating solid separation problems and high BOD_5 levels in the effluent.

Chian and Dewalle (1977) carried out similar studies using a stronger leachate (BOD_5 24700 mg/l) and confirmed that at 2 per cent leachate, the system operates well although the COD of the effluent increased with an increasing proportion of the leachate in the influent. They reported system failure at 4 per cent leachate, as shown by high effluent BOD_5 and deteriorating sludge separation. This was attributed to the increases in BOD_5:phosphorus ratio to values above 130:1. To this end, these authors deduced that a BOD_5:phosphorus ratio of about 100:1 represented a limiting constraint.

Winkler (1986) reported on the treatment of leachate by introducing it into the intake of the Soers wastewater treatment plant (West Germany). He indicated that no difficulties arose as long as the additional BOD_5 loading was 5 per cent or less of the total. He further pointed out that with a high strength leachate, problems due to excessive COD were encountered which resulted in excess sludge production and an increase in the organic content of the primary and secondary sludges prior to stabilization, with the consequence that the required degree of stabilization was no longer achieved in the usual 28 days retention period.

Jank (1981) has reviewed published reports concerning the effects of landfill leachate on the performance of activated sludge systems for municipal domestic wastewater treatment. He reported that data regarding full-scale treatment plants is scanty but there exists several reports on laboratory- and pilot-scale facilities for treating domestic wastewater with very small amounts of leachate or diluted leachate. He further reported that evidence of sustained nitrification at influent ammonia levels of 1000 mg/l has been obtained, showing that biological treatment is feasible in the absence of toxic levels of other pollutants.

These factors coupled with the usual long distance of the landfill sites to many wastewater treatment plants probably explain why very few treatment plants accept landfill leachate and those that do, generally limit the quantity of the leachate accepted to below 2 per cent of the total influent.

2.6.2 Leachate Recirculation onto the Landfill

Pilot-scale studies (Pohland, 1980; Tittlebaum, 1982) have shown that a major benefit of leachate recirculation onto the landfill, is the production of a leachate with a low organic strength (with a particular reduction in volatile fatty acids) in a relatively short period of time. Pohland (1980) and Barber and Maris (1984) also reported that the solid wastes were degraded and stabilized more rapidly due to the increased moisture content in the landfill obtained by leachate recycle.

Robinson et al (1982) showed that the volume of leachate could also be reduced by evaporation, if leachate was recycled by spraying onto the surface of an experimental landfill. Although recirculation of leachate greatly reduces the volume and organic strength of leachate, it is not a complete answer to leachate problems. The overall quantity of the leachate available for recirculation increased with time and that this excess leachate would inevitably require a disposal route other than recirculation. Moreover, other constituents of leachate such as ammonia, chlorides and heavy metals are not significantly removed. Hence, alternative treatment methods at some stage may be required.

2.6.3 Spray Irrigation

Spray irrigation of leachate to adjacent land or to a completed landfill has proved to be an effective process. The problems that occurred are freezing during winter season and overspraying leading to water-logged conditions. It would appear to be an appropriate solution for weaker leachate. Imhoff et al (1956) reported that domestic wastewater having a 230 mg/l BOD_5 is suitable for treatment by spray irrigation at a loading rate of 112 m^3/ha.d (12000 gallons/acre.day). The results from a laboratory investigation carried out by Bull et al (1983) to evaluate leachate treatment and disposal technique indicated that anaerobically treated leachate would be suitable for disposal by spray irrigation.

It seems that not all of the works using spray irrigation were successful. Newton (1979) experimented by spraying leachate with a BOD_5 concentration of 100 mg/l onto grasslands in Gloucestershire, UK. It was found that even with this very low BOD_5 leachate, both discolouration of the grass and ponding resulted. The experiment was thus discontinued.

Several studies on spray irrigation of domestic solid wastes carried out by Rowe (1979) at four landfill sites in Cornwall were not successful mainly because he employed developed techniques used for sewage treatment and these could not cope with such high BOD_5 and/or widely fluctuating flows of the leachate. The organic content of leachate was found to be much higher than that of sewage, and unlike sewage Rowe

(1979) argued that much of the organic content of the leachate cannot be settled out without initial treatment.

2.6.4 Biological Treatment

Leachates from recently deposited landfills which contain mainly volatile fatty acids can be readily degraded by biological means. Usually, leachates have very high concentrations of dissolved organic matters, therefore biological treatment processes probably will be the most appropriate treatment methods. Biological treatment may be in the form of aerobic or anaerobic treatment processes and since this study involved both aerobic and anaerobic treatment processes, the processes will be discussed in detail in Chapter 3.

2.6.5 Physical-Chemical Treatment

Physical-Chemical treatment methods have frequently been employed to purify industrial wastewater. The apparent difficulty in operating biological processes has led many workers to investigate physical and chemical processes. (Bull et al., 1983; Keenan et al., 1983). As a landfill stabilizes with the passage of time, the biodegradable organic content of the leachate decreases, and consequently the effectiveness of a biological treatment process decreases and physical-chemical processes may become more appropriate. In Chapter 3 a detailed literature review of some physical-chemical processes will be undertaken.

CHAPTER THREE

TREATMENT FUNDAMENTALS AND PROCESSES

3.1 Biological Wastewater Treatment

Biodegradation of waste is the dominant feature in the removal of organic pollutants both in natural stream self-purification and in biological wastewater treatment. It has been in existence since time immemorial. Sterritt and Lester (1988) reported that the evolution of wastewater treatment processes was prompted by the spread of waterborne diseases in overpopulated areas. The treatment processes developed could also help in reducing odour problems due to putrefaction of solid wastes.

A well operated and controlled treatment system will accelerate the time taken for removing organic matter. Through the advancement of technology treatment processes are becoming more sophisticated, but the fundamental biological reactions occurring in these processes remain practically the same (Farquhar, 1979). In order to optimize the microbial metabolism involved in biological processes, a proper understanding of the fundamentals of the microbiology and the process development of the treatment systems is essential (Loehr, 1977; Anderson, 1981). Achieving this will assist in the rational predictions of design and performance of treatment processes.

Biological treatment methods may be either aerobic or anaerobic, depending on the environmental conditions and process development. In aerobic processes, the oxidation of organic matter which utilize dissolved oxygen is accompanied by the liberation of heat energy (Morgan and Gunson, 1988). The energy liberated is utilized for cell synthesis and reproduction. In anaerobic processes the organic matter is degraded in the absence of oxygen. Organic fermentation yields new cells, carbon dioxide, water as well as other carbonaceous residuals such as methane.

In both cases, in order to make treatment more complete, it is generally necessary to provide for the removal of the microbial cells from the wastewater subsequent to treatment. The steps involved in biological wastewater treatment are summarized in Fig. 3.1.

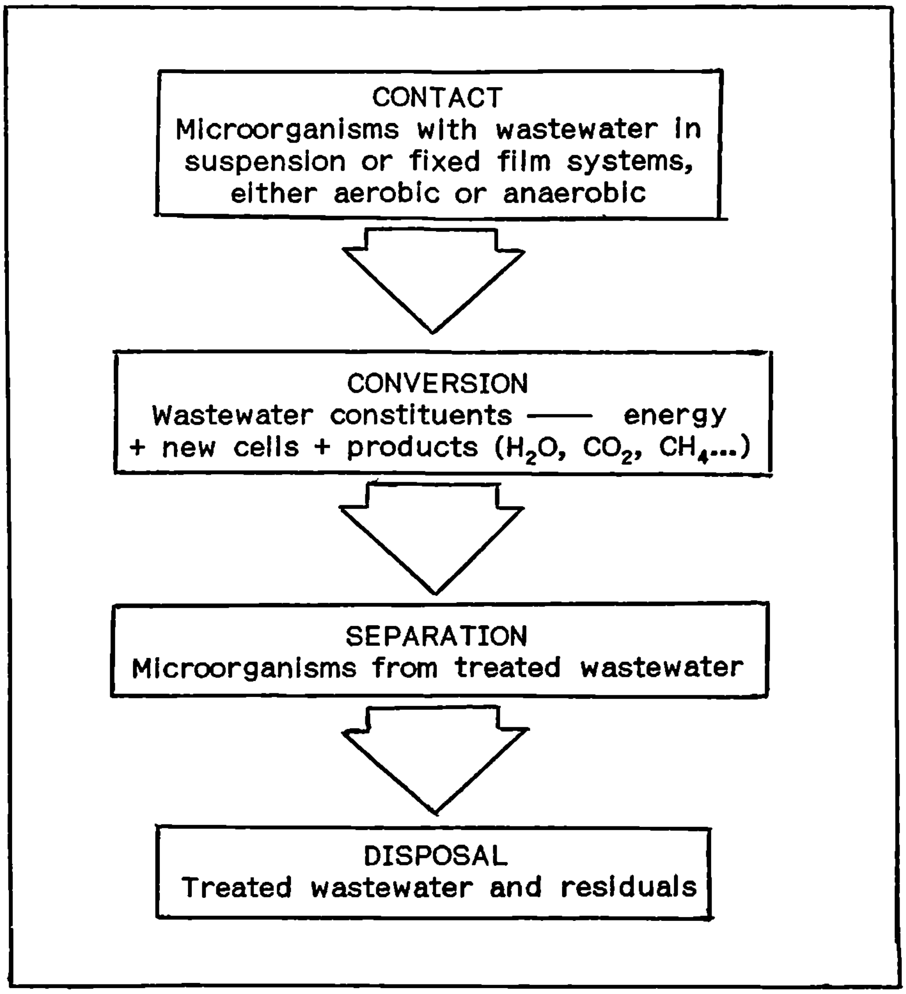


Figure 3.1: Steps in Biological Wastewater Treatment (Farquhar, 1979)

The design of the processes is based on an assessment of microbial growth and substrate conversion kinetics. These mostly involve the use of Monod-type equations together with cell yield and decay expressions. These are then combined with the hydraulic properties of the reactor to yield efficiency expressions.

3.2 Aerobic Treatment Process

The aerobic process of wastewater treatment has been well studied and the equations involved are also well known (Chiu et al, 1972; Chen and Hashimoto, 1980; Bovendeur and Klapwijk, 1986; Vochten et al, 1988). The process basically involves the biological oxidation and biosynthesis of organic matter (both dissolved and suspended) that is present in the wastewater by microorganisms established in the treatment plant. Both of these biological processes result in the removal of organic matter. The desired products of organic matter conversion are new cells, carbon dioxide and water.

Biological oxidation (respiration) results in mineralized end-products which are discharged in solution in the effluent. Biosynthesis converts organic matter, which is either mostly in a soluble or a suspended form, into particulate biomass which can subsequently be removed by settlement as excess activated sludge or as "humus" solids from biological filters (Metcalf and Eddy, 1979).

The interaction between organic matter and microorganisms can be maximized by feeding the influent wastewater over a film of biomass fixed to media surfaces or by a suspension of biomass in the wastewater (Peavy et al, 1986).

3.2.1 Process Development

The microorganisms involved in the aerobic biological treatment of wastewater are essentially similar to those that degrade organic matter in the self-purification of natural water systems. The types of microorganism that can be found include bacteria, fungi, algae, protozoa, rotifers, crustacea, worms and insect larvae depending upon the prevailing environmental conditions.

The predominance of the different types of microorganism in biological systems may be used as indicators of the performance and environmental conditions in the system. One useful tool that can be utilized for assessing the quality of the effluent, the degree of treatment accomplished, and changes occurring in the system is the periodic microscopic examination of the microorganisms existing in the system (Gaudy and Gaudy, 1980).

The biochemical reactions involved in the microbial metabolism in the aerobic degradation of waste are rather complex and are not fully understood (Peavy et al, 1986). However, two distinct types of metabolic processes that occur simultaneously, are known which are:

i) the process involving the degradation of the substrate and furnishing of energy for the synthesis of new cells. This process is collectively termed catabolism (Steritt and Lester, 1988), and

ii) anabolism, the other process, that provides the material necessary for cell growth. In the absence of external food sources the organisms will use previously stored endogenous food supplies for their respiration.

3.2.2 Aerobic Reactions

As can be seen previously, although different types of microorganism have different metabolic pathways, the principles of energy, synthesis, and endogenous catabolism consistently remained the same. The rates at which these reactions occur are a function of the environmental conditions imposed by and/or on a given biological treatment process. The generalized metabolic processes are shown schematically in Fig. 3.2.

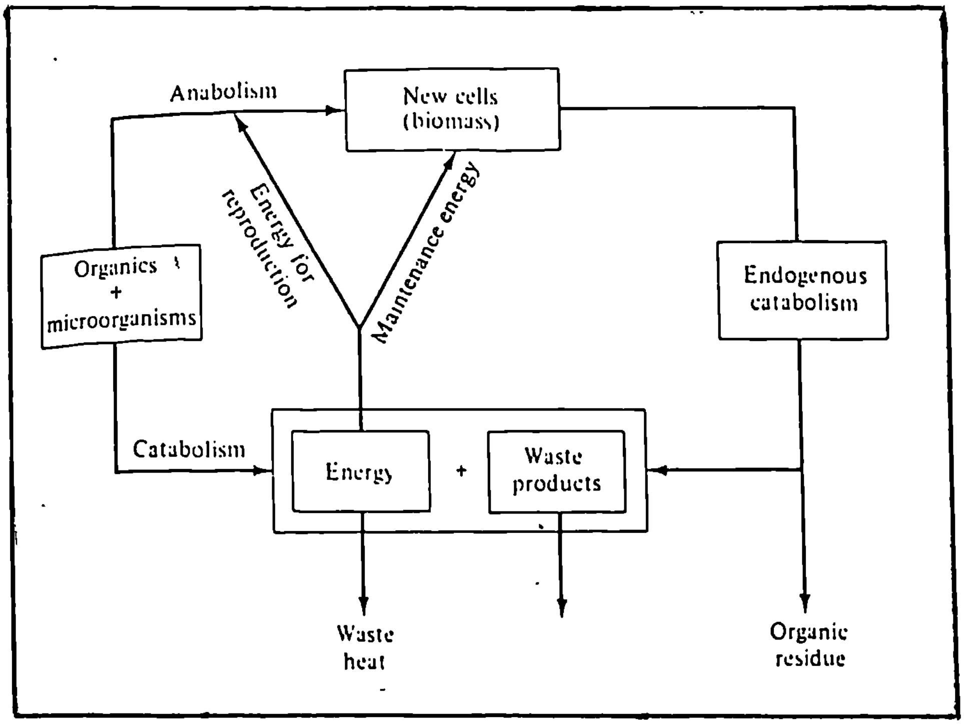


Figure 3.2: Generalized Metabolic Pathway (Peavy et al, 1986)

Biodegradation of organic matter under aerobic conditions involves the oxidation of organic compounds to carbon dioxide, water, nitrate, phosphate and sulphate. The process of decomposition during biological treatment can be determined by these changes in the effluent. These transformations are discussed in subsequent sections.

- Carbon Dioxide

In the presence of sufficient nutrients, vital for the aerobic growth of microorganisms, a fraction of the biodegradable matter is converted into carbon dioxide. The utilization of oxygen as well as the conversion to carbon dioxide represents the effects of respiration. The energy that is generated during this process is then used to fix the substrate carbon to form additional microbial biomass.

Hamer (1989) stated that biodegradable and/or non-biodegradable products can also be generated together with a corresponding reduction of energy. The reactions during this process will depend both on the nature of the original substrate and an oxygen availability.

- Nitrogen

One of the essential nutrients in the biological treatment systems is nitrogen. In wastewater, nitrogen is present in the form of organic nitrogen and ammonia nitrogen. The oxidation of nitrogen compounds takes place in two stages. In the first stage, in the presence of oxygen, living organisms will convert organic nitrogen to ammonia nitrogen and in the second stage ammonia is converted to nitrite and nitrate nitrogen.

In order for nitrification to be significant, the solids retention time must be long enough for the conversion of ammonia nitrogen to nitrite and nitrate nitrogen. A further requirement is that for optimum nitrification, a residual dissolved oxygen concentration of about 2 mg/l must be present in the wastewater (Loehr, 1977).

Nitrification is accomplished by the presence of nitrifying autotrophic bacteria, Nitrosomonas and Nitrobacter which exist in rivers, lakes and wastewater treatment plants. Nitrosomonas catalysed the oxidation of ammonia to nitrite using molecular oxygen, while Nitrobacter further oxidized nitrite to nitrate using oxygen derived from the water molecule (Horan, 1990). They use carbon dioxide as their source of carbon while the oxidation of ammonia gives them the required energy:

$$2NH_3 + 3O_2 \text{ ----> (Nitrosomonas) ---> } 2NO_2^- + 2H^+ + 2H_2O \qquad (3.1)$$

$$2NO_2^- + O_2 \text{ ----> (Nitrobacter) ---> } 2NO_3^{2-} \qquad (3.2)$$

Nitrifying bacteria are relatively slow growing, and function best at temperatures above 25°C. At temperature below 5°C, their metabolism may slow down to the extent that the bacteria become almost dormant. Temperature plays significant role in the design and operation of a treatment plant. Although nitrification may be achieved during summer months, in winter the growth of nitrifying bacteria is inhibited, and action must be taken to limit the ammonia existing in the wastewater.

The performance of a conventional treatment plant can usually be predicted after the quantities of oxygen consumed during the first stage of oxidation are obtained. However, eutrophication of receiving waterbodies may result with a highly nitrified effluent (Barnes and Bliss, 1983). In order to control this, many wastewater treatment plants are being modified to incorporate both nitrification and denitrification as a means of removing nitrogen from the effluent. Denitrification is adopted to reduce the nitrate form in the highly nitrified effluent, in which it can be converted to nitrogen gas by facultative heterotrophic bacteria in an anoxic environment.

- Phosphorus

The sources of phosphorus in wastewaters include organically bound phosphorus originating from body and food wastes, polyphosphates coming from synthetic detergents, and the urine of man and animals. The organic phosphorus is transformed to inorganic phosphorus in the form of orthophosphates during biological decomposition. Phosphorus is required by microorganisms in treatment processes, but excess phosphorus may reduce the treatment efficiencies by interfering with the chemical coagulation of turbidity and may also result in eutrophication of waterbodies. Hence when introducing phosphorus into low phosphate wastewater, a proper knowledge is required of the amount of phosphorus needed in order to ensure sufficient nutrient is available and to prevent excess nutrient in the resultant effluent.

- Sulphur

Microbial transformations of sulphur are similar to those of nitrogen. The decomposition of organic sulphur-bearing waste yields sulphide, which in turned is oxidized to sulphate under aerobic conditions.

3.2.3 Factors Affecting Aerobic Process

The most common factors affecting the rates of biological activity include temperature, dissolved oxygen (DO) concentration, pH and nutrient levels.

- Temperature

Microorganisms do not have any means of controlling the internal temperature, thus the temperature within the cell is determined by the external temperature. Somerville (1985) reported that the operating temperature in aerobic treatment processes is not usually a controllable operating variable. This is due to the fact that the design of most aerobic bioreactors frequently incorporates rotating, surface aerators which tends to equilibrate between the ambient temperature and the process operating temperature. Most investigations have concluded that the effect of temperature on reaction rates can be expressed using the modified Arrhenius equation of the form (Farquhar, 1979; Peavy, 1986):

$$k_T = k_{20}\,\theta^{T-20} \qquad \text{..... (3.3)}$$

where k_T, k_{20} = reaction rate constants at temperature T and 20°C

θ = temperature correction coefficient

T = temperature (°C)

The values of the reaction rates tend to increase with temperature up to a maximum value, after which time cell deterioration leads to a rapid reduction of reaction rates.

- Dissolved Oxygen Concentration

The rates of aerobic biological reactions are affected by DO concentrations up to 2 to 3 mg/l range, beyond which they become independent of DO (Farquhar, 1979). The critical DO concentration for a flocculated heterogeneous microbial population has usually been found to be more than 0.5 mg/l. Hence, to provide for a factor of safety, maintenance of a 2.0 mg/l concentration is generally recommended.

- pH

Biological activity can alter the pH of a treatment unit. Photosynthesis, denitrification and organic nitrogen breakdown cause an increase in pH, while pH decreases during sulphate oxidation, nitrification and organic carbon oxidation. The relative changes in pH will be affected by the buffer capacity of the liquid and amount of substrate utilized by the microorganisms.

- Nutrient Requirement

The maximization of substrate conversion rates can be achieved by ensuring an adequate nutrient level within the wastewater, with the required nutrients being in the form of nitrogen, phosphorus and trace minerals. Loehr (1977) stated that nutrient deficiencies result in a decrease in microbial growth rate, lowering the treatment efficiencies as well as impairing the settling characteristics of a sludge. In certain wastes, specific nutrients such as nitrogen, may be in excess. As stated before, an excess of nutrients can be a cause of eutrophication in receiving waters when the treated effluent is discharged. Hence, methods of controlling and removing these excess nutrients are required.

The usual BOD_5:N:P ratio for aerobic systems is 100:5:1 (Farquhar, 1979), although the actual nutritional requirements will be related to the operational behaviour of the treatment processes. A high rate process resulting in a high rate of microbial growth will need a greater quantity of nutrient. However most treatment systems, with a long solids retention time, will require less nutrients. In this case, the endogenous respiration of the microbial cells will release the additional nutrients required for the synthesis of new cells (Gaudy and Gaudy, 1980).

3.2.4 Effect of Inhibitory Substances

The rate at which biological oxidation takes place is dependent on the composition of organic matter and the presence of inhibitory substances. In general the aerobic microorganisms responsible for biodegradation are less sensitive to the presence of dissolved inorganic ions, such as chlorides and sulphides, than their anaerobic counterparts. However, they show a greater sensitivity to pesticides and heavy metals such as lead and arsenic. Weiss (1974) has given a list of approximate values for the concentrations at which the most common inhibitory substances take effect.

3.2.5 Types of Treatment Processes

Examples of aerobic treatment processes in use are the activated sludge process, oxidation ditches, waste stabilization ponds and lagoons, biological filters and aerobic rotating biological contactors (RBCs). A brief description of the processes will be given below and a detailed overview of the historical and process developments of an aerobic RBC process will be discussed in another section since this study involves the use of RBC units.

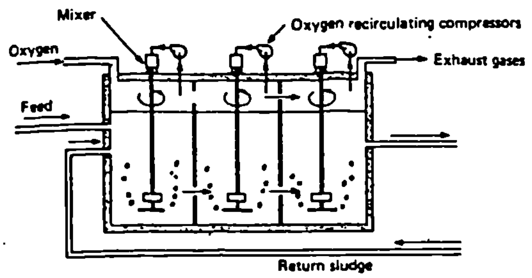
- Activated Sludge Process

The activated sludge process is a suspended growth system that has been in use since the early 1900s (Peavy et al, 1986). The activated sludge process is aerobic, with oxygen being supplied by dissolution from entrained air or pure oxygen. The entrained air helps in maintaining the microorganism flocs in suspension and the existence of the microbial cells in the form of flocs permits them to be consolidated by sedimentation under quiescent conditions in a separate secondary sedimentation tank.

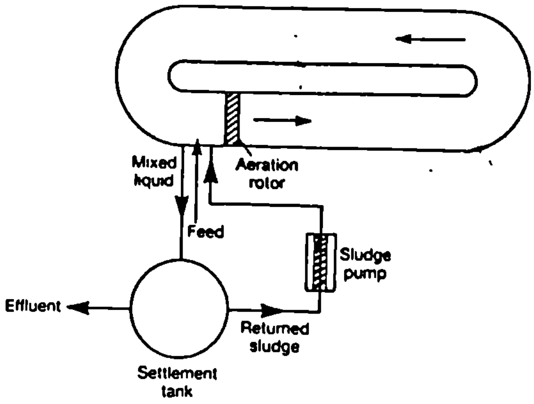
The process derives its name from the fact that settled sludge containing living, or active, microorganisms is returned to the reactor to increase the available biomass and speed up the reactions. The mixture of biomass and wastewater is termed mixed liquor. The activated sludge process can either be a completely mixed or a plug flow process. A schematic diagram of the activated sludge process using a mechanical aerator is shown in Fig. 3.3a.

- Oxidation Ditches

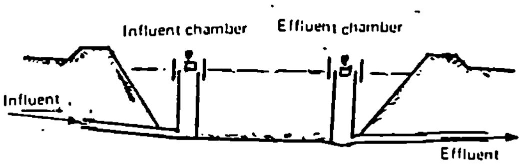
An oxidation ditch was developed by the Institute of Public Health Engineering in the Netherlands (Loehr, 1977). The ditch is an earthen or concrete tank of specific shape with arrangements for a sufficient supply



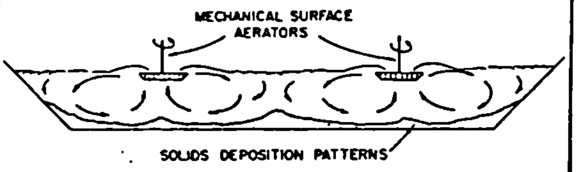
a. Activated Sludge with Mechanical Aerator



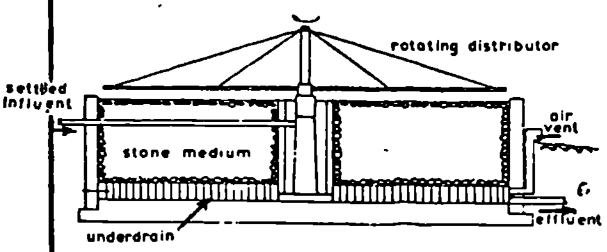
b. Oxidation Ditch



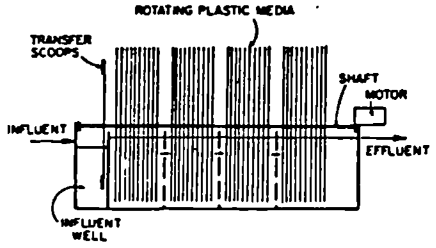
c. Waste Stabilization Pond



d. Aerated Lagoon



e. Biological Filter



f. Rotating Biological Contactor

Figure 3.3: Schematic Diagrams of Principal Aerobic Treatment Processes (Barnes et al, 1981; Tebbutt, 1983; Peavy, 1986)

of oxygen. Raw wastewater is brought into the tank and aerated for an extended period of time. An oxidation ditch as illustrated in Fig. 3.3b can achieve a high degree of purification.

The activated sludge in the oxidation ditch removes the organic matter and converts it to cell protoplasm which will then degrade if aeration is continued further (30 to 60 days mean cell residence time). The sludge formed in the oxidation ditch is mineralized to such an extent that it can be dried on sand beds without odour problems (Metcalf and Eddy, 1979).

- Waste Stabilization Ponds and Lagoons

A wastewater pond, alternatively known as a stabilization pond (Fig. 33c) and sewage lagoon (Peavy et al, 1986), consists of a large, shallow earthen basin in which wastewater is retained long enough for natural purification processes to provide the necessary degree of treatment. The phenomenon of lagoon treatment is dependent on factors such as influent quality, type of microorganism and aquatic growth (algae), which in turn depends on light, temperature, dissolved oxygen, evaporation, percolation and sedimentation (Metcalf and Eddy, 1979). The pH value and toxicity of the substances in the wastewater also affect the treatment process.

At least part of the system must be aerobic to produce an acceptable effluent. Although some oxygen is provided by diffusion from the air, the bulk of the oxygen in ponds is provided by photosynthesis. Lagoons are distinguished from ponds in that oxygen for lagoons is provided by artificial aeration (Peavy et al, 1986). For aerobic system, a shallow aerated lagoon as in Fig 3.3d is employed. The organic matter in wastewater is converted to carbon dioxide and ammonia and then to algae in the presence of sunlight. However, the large land area required causes this process to be impractical in a metropolitan area or in places where land purchase price is very high.

- Biological Filters

A biological filter is an example of an attached growth system. It is a reactor in which randomly packed solid forms provide a surface area for biofilm growth. The term "filter" is a misnomer, since the reactor contains media which are too large to serve as a filter. Instead, sorption and subsequent biological oxidation are the primary means of substrate removal (Peavy et al, 1986). An example of a biological filter is shown in Fig. 3.3e.

The important characteristics of the media include specific surface and porosity. The application of wastewater onto the media is usually accomplished by a rotating distribution system which spray the wastewater uniformly over the surface of the media.

Many factors affect the operation of a biological filter, the most important being the organic loading rate, hydraulic flow rate and the temperature of the wastewater and ambient air. A high organic loading rate results in a rapid growth of biomass although excessive growth may result in the plugging of pore space and subsequent flooding of portions of the media. Increasing the hydraulic loading rate increases sloughing and helps to keep the bed open. These loading rates limit the depth of a conventional biological filter to 2 m because of head loss through the randomly packed media.

3.2.6 Rotating Biological Contactor (RBC) Process

The use of RBC's for wastewater treatment highlighting process description, advantages and disadvantages, history, applications and process parameter had been reviewed by several authors (Antonie, 1976; Lumbers, 1983; Strom and Chung, 1985).

- Historical Background

The rotating biological contactor was described as early as 1900 by Weigand in Germany. However, the Weigand idea remained dormant until Doman (1929) carried out an experimental work with laboratory-scale equipment consisting of metal discs situated in an Imhoff tank type of reactor. Later in 1931, Maltby patented a process that was based on the biological wheel principles. The revival of the RBC process occurred in about 1960 when Hartman (1964) developed an "immersion drip-filter" (trauchtropfkorpern). The research of Hartman was continued by Pöpel (1964) using a large-scale unit from which an empirical correlation for the efficiency of substrate removal in terms of the operating variables was derived. The first commercial RBC was installed in West Germany in 1960 and soon after it was widely applied throughout Europe (Antonie, 1976)

RBCs using rotating discs were introduced into both the United States and the United Kingdom to be used particularly for treatment of wastewater from small isolated communities (Welch, 1968). A paper published by Bruce et al (1973) gave a comprehensive detail of a Bio-Disc plant tested at the Water Pollution Research Laboratory in Stevenage. The plant ran entirely on domestic sewage from a population equivalent of about 25 persons. The reported BOD_5 removal was about 87% to 97%.

By the middle of the 1970s, in both Europe and the United States, came an important breakthrough in improving the efficiency of RBCs. The development of a more compact disc with much more surface area for a given volume of RBC greatly enhanced the performance of the RBC. This has been achieved in several ways, probably the most usual being the use of the packed cage, with plastic media of ever increasing specific surface.

The European Plastic Machinery Co. (Sorensen, 1974) have developed a rotating biological packed-drum (RBPD) unit called the Biodrum in which hollow plastic balls were packed into the drum. The results of a pilot-scale study using a single-stage Biodrum process to treat dairy waste, indicated that it was possible to remove 95 percent of the influent BOD_5 at an average organic loading of 3 kg $BOD_5/m^3.d$. This loading is about 15 times greater than that in a low loaded conventional biological filter with a very high recirculation ratio.

Another method of increasing the specific surface was patented by the Autotrol Corporation in 1972, using corrugated sheets of polythene. This increased the specific surface to 120 m^2/m^3, compared to the usual 50 m^2/m^3 for discs.

Several experimental studies using RBPDs were carried out at the Department of Civil Engineering, the University of Newcastle upon Tyne.

Vitoonpunyakij (1976) studied the performance of a RBPD unit and reported that 90 percent BOD_5 removal was achieved at an organic loading rate of 3 kg BOD_5/m^3.day. Norton (1984), using RBPDs to treat strong organic wastewaters, found out that under aerobic conditions the RBPD was probably most cost-effective when used at a loading rate in the region of 50 g BOD_5/m^2.day. Ibrahim (1986) investigated the applicability of using RBPD to treat industrial wastewater containing high lipid content and concluded that non-hydrocarbon lipid did not interfere with the RBC performance as seen from the high COD reduction (over 80%) at an optimum loading rate of 60 g COD/m^2.day. Using the RBPD unit to treat a high strength industrial wastewater from petrochemical and synthetic fibre manufacturing plant, George (1988) achieved a COD removal of up to 96%.

- Process Description

The RBC process is similar in function to the biological filter in that both operate as fixed-film biological reactors (Chittenden and Wells, 1971). The wastewater is allowed to percolate through the contact media in a biological filter while in an RBC the contact media is moved against the wastewater. Both the processes also exhibit similar principles of substrate, nitrogen and oxygen conversions. However, as the RBC consists of a series of closely spaced discs (Fig 3.4), the influent-effluent characteristics of a wastewater vary greatly from stage to stage (Ouano and Pescod, 1976). Watanabe et al, (1978) applied a fully submerged biofilm model to his nitrification experiment on an RBC which is partly submerged. The kinetics of substrate utilization by the biofilm in RBCs have also been studied using a general fixed-film model (Rittman and McCarty, 1978; Atasi and Borchardt, 1983).

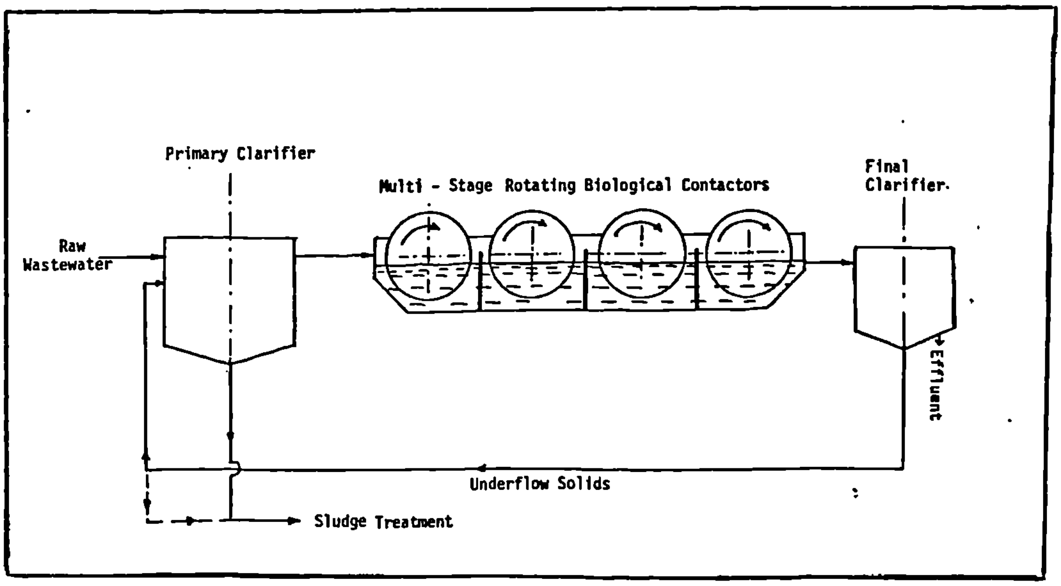


Figure 3.4: Typical Flow Diagram of RBC (Task Committee, 1983)

The series of closely spaced discs are mounted on a horizontal shaft, with 40 percent being immersed in the wastewater, and is set up in a basin, through which the wastewater flows, and may be baffled either parallel to the shafts if there is more than one, or perpendicular to the shafts, to prevent channelling.

Gilbert et al (1986) reported that for optimization of the process in the most energy-efficient manner, the process must be flexible. It is recommended in the design, to include for removable baffles between stages and the ability to bypass part of influent flow to subsequent stages (stepfeed). The process control and operational flexibility can also be increased by installing a shaft with variable rotational speeds (Evans, 1985). This flexibility will allow for adequate media surface area for treating a high strength organic wastewater.

The shaft is rotated slowly, causing an alternating exposure of the discs to the atmosphere and the wastewater. As the discs rotate they carry a film of wastewater into the atmosphere. The slow rotation develops a biological slime which oxidizes the wastewater as it passes through it and colonies of bacteria continue to propagate (in the presence of adequate oxygen and food material) on both sides of the discs. The oxygen is transferred from the atmosphere to this liquid film. The organisms attached to the discs then remove both dissolved oxygen and nutrients from this film of wastewater.

Fujie et al (1983) reported that the use of RBC, because of its low sludge production, easy operation and maintenance and more importantly because of its comparatively low power requirement, has become more attractive at small scale wastewater treatment plants. The fluctuation in inflow rate of wastewater and influent BOD_5 concentration increased the power consumption considerably thus, in order to achieve a higher power economy, the installation of a flow balancing tank was recommended.

Trulear and Characklis (1982) found out that the biofilm developed on a surface exposed to wastewater is the net result of physical transport and biological growth rate processes. The overall biofilm accumulation is initiated by the adsorption of an organic monolayer and occurs within a few minutes after exposure of the solid surface to the wastewater containing the dispersed microorganisms, nutrients and organic matters. The organic adsorption is a prerequisite for further biological development because it conditions the surface.

Diffusion of substrate plays an important role in the fixed-film model. La Motta (1976) has divided all the steps occurring in the overall process into three major types:

i) external diffusion which consists of diffusion of the substrate from the bulk of the liquid to the interface between the liquid and the biological film,

ii) internal diffusion in which diffusion of the substrate takes place within the porous biological slime, and

iii) biochemical reactions within the biofilm.

When the slime layer becomes too thick it sloughs off and is settled, either in a sludge storage zone which forms part of the basin or in a separate tank. Sludge production is not large for normal wastewaters, therefore, in the first instance sludge removal may only take place two to four times a year, due to the degree of anaerobic digestion of the sludge.

An RBC is generally designed as a "once-through" flow process with no sludge or effluent recycle. The operation without sludge or effluent recycle has made the RBC simple to operate and increases process efficiency (Antonie, 1978; Task Committee, 1983). However, some researchers believe recycling capabilities should be designed into any RBC plant for increased flexibility (Poon et al, 1979).

The RBC process, like any other treatment technology, has its own inherent advantages and disadvantages which potential users should be cognisant with. The advantages claimed for RBC systems are flexibility, a high degree of efficiency, stability, low maintenance and power consumption, short process retention time, excellent shock and toxic load capabilities, provisions for nitrification and improved sludge settling characteristics (Wu and Smith, 1982; Hamoda and Wilson, 1989). In addition, as referred to earlier, it requires no sludge or effluent recycle stream. The system consumes relatively little energy since the medium is evenly balanced about the rotating driving shaft, and unlike most other "contact-filter" systems, the head loss in the RBC is low, thus, adding an RBC to an existing plant improves the performance of the plant without the need to add pumping facilities (EPA, 1976).

The disadvantages of the systems are as follows (Dallaire, 1979):

i) for a good size plant (>210 m^3/day), the number of RBC units required may be substantial, which means that a large number of electro-mechanical drives will need maintenance and upkeep,

ii) effluent quality may not be as predictable as a suspended growth process.

iii) RBC shaft media and drive failures have been experienced, requiring major repairs,

iv) oil leaks from drive units may cause problems, and

v) larger plants require more space than equally sized suspended growth systems, depending upon site layout and the depth of tanks.

- Process Development

A factor which contributes to the efficiency of a biological ecosystem is ambient temperature. In 1976, Presner et al., using a very small unit and an exceptionally high hydraulic loading rate found that biomass developed more quickly and grew thicker on heated discs than on unheated ones, but after 7 days there was a greater accumulation of biomass on the unheated discs. The COD removals were found to reach a maximum at the optimum metabolic rate of mesophilic bacteria (30°C), but fell off with further increase in temperature until at 43°C the rate was less than for the unheated discs.

Antonie (1976) reported that the reaction velocity changes by a factor of between two to three for each 10°C change in temperature. Lower temperatures gave a lower removal rate at a high BOD_5 concentration but gave higher a removal rate at low BOD_5 concentration. Huang et al (1985) used pilot-scale RBCs in their study to treat phenol-formaldehyde resin wastewater. From the study, 99.6% phenol, 93% formaldehyde and 60 - 90% COD were removed at an HRT of 2.8 hours. The COD removal efficiency increased by 10 - 15%, when the wastewater temperature was increased from 2 - 7°C to 23 - 24°C.

A study carried out by Lehman (1983) indicated that the maximum biomass growth developed at an optimum peripheral velocity of 0.27 to 1.5 m/s with a hydraulic loading rate of 0.091 $m^3/m^2.d$ with a final effluent quality of 25 mg/l BOD_5 and 30 mg/l suspended solids giving a total removal efficiency of 88% and 99% respectively. Pescod and Norton (1983) reported studies on RBPD, using small random packing media, rotating at 10 rpm (peripheral velocity of 9.4 cm/sec) and ambient temperature of 20 to 22°C to achieve a removal efficiency of 90% COD removal at a neutral pH value for both aerobic and anaerobic treatment processes.

The variation of suspended solids concentration in a RBC system is likely to be stage-dependent. A higher suspended solids concentration is usually found at that stage where the weight of the biomass is also higher, and that the cell mass tends to build up to a certain point and then detach from the disc surface. These sloughed off biological slimes may suspend or partially accumulate in each compartment (Torpey, 1971; Pretorius, 1971; Labella et al, 1972). However, an evaluation of the performance of an RBC installation in Gladstone, Michigan revealed that mixed liquor suspended solids concentration variations between stages was quite small (Autotrol Corp., 1975).

Pescod and Nair (1972) investigated the effect of retention time on process development. Using a substrate obtained by diluting a nightsoil supernatant liquid to a COD of 400 - 450 mg/l in conjunction with a small pilot plant, they found that the retention time had little effect on COD removal, more than 80% being removed in a 2 hour retention time, which they felt could have been reduced even further, however, the areal organic loading rate was quite low at 8 g COD/m^2.d. Suwanarat (1968) using an HRT of 4.5 hours reported a very high BOD_5 removal efficiency of 98% for assimilated milk waste at an AOLR of 11 g BOD_5/m^2.day.

Labella et al (1972) also had a fairly high areal organic loading rate with a reasonable removal rate, using brewery waste. Using waste from pulp and paper mills at various loading rates, Gillespie et al (1974) found that as areal loading rate increased, BOD_5 and COD removals decreased. Pajak and Loehr (1975) using RBC to treat a wide range of poultry manure waste, reported a similar finding.

The studies made by Antonie et al, (1974) and Torpey et al, (1974) indicated that if the RBC surface area is fixed and the hydraulic loading rate decreases, thus producing a longer residence time, the BOD_5 removal achieved by the RBC increases. This is also implying that, as the total organic loading rate to the RBC is decreased, the removal efficiency increases.

It is apparent that both the influent wastewater substrate concentration and the hydraulic loading rates affect the removal efficiencies achieved by the RBC process. Del Borghi et al (1985) in their study, showed that the hydraulic loading rate, the stage number and the wastewater temperature are the most significant variables affecting the RBC system performance. Therefore, the variability of both of these parameters must be considered when assessing the performance of an RBC.

Antonie (1976) stated that peripheral speeds above 0.3 m/s had no further beneficial effect and that for a BOD_5 removal of 85% with a 2-stage system the maximum hydraulic load rate would be 0.074 $m^3/m^2.d$, but with a 4-stage system of the same dimensions the loading could be up to 0.122 $m^3/m^2.d$, for BOD_5 concentrations up to 300 mg/l which is rather higher than established practice at 37 g $BOD_5/m^2.d$.

Pescod and Ware (1988) in their investigation of a full-scale study with an anaerobic/aerobic RBC unit on the treatment of brewery wastewater stated that excessive turbulence along with the variable operating conditions such as loading rate, pH, etc. affected the performance of the system although an overall COD removal of 85% could be achieved.

Table 3.1 summarizes removal rates from domestic wastewaters where sufficient data was given in the paper for areal organic loading rate in terms of BOD_5 and COD.

Table 3.1 BOD$_5$/COD Removal Rates from Domestic Wastewaters

Reference	BOD$_5$ Applied (g/m^2.d)	COD Applied (g/m^2.d)	Removal (%)	Type of Unit
W.R. Newsheet	6.8		93	Disc
W.R. Newsheet	27.8		92	Disc
Antonie	6.5		92	Disc
Antonie	18.4		83	Disc
Antonie	18.4		88	Bio-Surf^R
Antonie et al	3.2		94	Disc
Bruce	6.0		90	Disc
Pescod & Nair		8.0	80	Disc
Poon & Mikucki	38.0		92	Bio-Surf^R
Pretorius		29.0	73	Disc

The response of the RBC to hydraulic surges was investigated by Antonie (1970). It was observed that the RBC was not upset by hydraulic surges due to the large captive biomass that was retained on the disc which absorbed the increased organic overload brought about by the hydraulic surge. The RBC performance rapidly returned to steady state condition after the surge ended.

Stratta and Long (1980) concentrated their work on the effect of pH on fixed film RBC nitrification. The research was carried out under two phases of an RBC system that is:

i) nitrification as a function of pH in which the system was capable of nitrifying the natural wastewater containing approximately 290 mg CaCO$_3$/l of alkalinity, and

ii) pH adjustment for optimization of the process of treatment using various chemical, such as lime, soda ash, sodium bicarbonate and sodium hydroxide.

Shammas (1986), investigating the interactions of temperature, pH and biomass on the nitrification process, concluded that the design and operation of the nitrification RBC process must be based on the combined environmental and operational conditions which are mainly a function of pH, temperature and MLVSS. RBCs may also be useful in achieving nitrification of landfill leachate (Harrington and Maris, 1986)

The effect of metals and toxic organic pollutants on the operation of RBCs was also studied by several researchers (Nakamura et al, 1986; Pisano et al, 1989). Nakamura et al (1986) using a laboratory-scale RBC to treat wastewater containing ferrous iron observed that low pH (1.5 to 2.6) and temperature (10 to 40°C) did not affect the Fe^{2+} oxidation rate. At peripheral disc velocities of 4.7 to 28.2 m/min, Fe^{2+} oxidation increased rapidly.

Janczukowicz et al (1990), from investigations carried out to determine the influence of waste system feeding on technological parameters of an RBC, concluded that the change of wastes feeding system by step feeding wastes to the first and second stage of a four-stages RBC caused a uniform biomass growth in all the stages. Higher COD removal was achieved due to the improved conditions for organic substrate utilization

The predominance of the various forms of microorganisms in biological systems may at time be indicative of the performance and environmental conditions in the systems. Microscopic examination of the biological system can be utilized as a tentative guide to the quality of the effluent, the degree of treatment that has been accomplished, and changes occurring in the systems.

Torpey et al (1974), Pescod and Nair, (1972) and Sack et al, (1973) conducted examinations to determine biological solids characteristic on the media under various operational conditions in treating domestic and industrial wastes. The general findings based upon these studies reported by the above investigators are summarised by the Task Committee (1983) as followed:

"The predominant organisms including Sphaerotilus and zoogleal bacteria are present on all discs. Besides these two important kinds, the diversity and abundance of free-swimming protozoa (Paramaecium, Cyclidium, Ocomanas, Oxytrichia, and Euglena) are present in the first few stages. The growth of rotifers (Epiphanes and Proales), and a loop-forming fungus (Anthrobotyrs) together with algae (Coelastrum, Chlorella, Fragilaria, and Pinnularia) occur in the last few stages only when the organic loading rate is low but sufficiently high to support microbial growth. The quickly developed biofilm during the earlier stages on the RBC system is much thicker than bacterial slime produced on the later discs.

The mechanisms of attached growth in a RBC treatment system are described as the filamentous organisms (Sphaerotilus, Geotrichum, Bacillus) actually serving as a sort of skeletal system on which other microorganisms are able to attach. The thickness of the biofilm is substantially reduced in each stage as a result of significant reduction in filamentous populations, and that is caused by the marked change in carbon-energy level in wastewater after passing it through each stage. Both Pseudomonas denitrificans and Beggiatoa alba are also present in the RBC system indicating that there are involvements of both nitrogen and sulphur transfers inherent in the systems."

3.3 Anaerobic Treatment Process

Despite the widespread use of anaerobic digestion in wastewater treatment, the basic microbiology and biochemistry of the process are still poorly understood. Several papers have been presented comprehensively reviewing the application of anaerobic treatment processes (McCarty, 1981; Henze and Harremoes, 1983; Brown and Tata, 1985; Parkin and Owen, 1986; Harper and Pohland, 1986; Vigneswaran et al, 1986). Tilche and Vieira (1991) and Weiland and Rozzi (1991) in their discussion reports reviewed some general aspects of start-up, operation behaviour and hydrodynamic of some anaerobic reactors and their relationship with process efficiency. The anaerobic process is usually described as a three-step process (Karmano et al, 1986). The mechanism of the process indicating the three distinct phases in the operation is illustrated in Fig. 3.5.

3.3.1 Microbial Metabolism

The formation of biogas from waste organic matter in an anaerobic fermentation is a complex process. The conversion of complex organic matter to methane and carbon dioxide, in the absence of molecular oxygen, is generally accomplished by two major groups of bacteria, namely acid producing and methane producing bacteria (Anderson and Donnelly, 1977). Metabolic activity links these two groups in anaerobic digestion (Zeikus, 1980; Mah et al, 1981). Initially, the complex organic compounds are converted to less complex, soluble organic compounds by enzymatic hydrolysis. In the acid-forming step, these hydrolysis products are further fermented to simpler organic compounds, mainly volatile fatty acids.

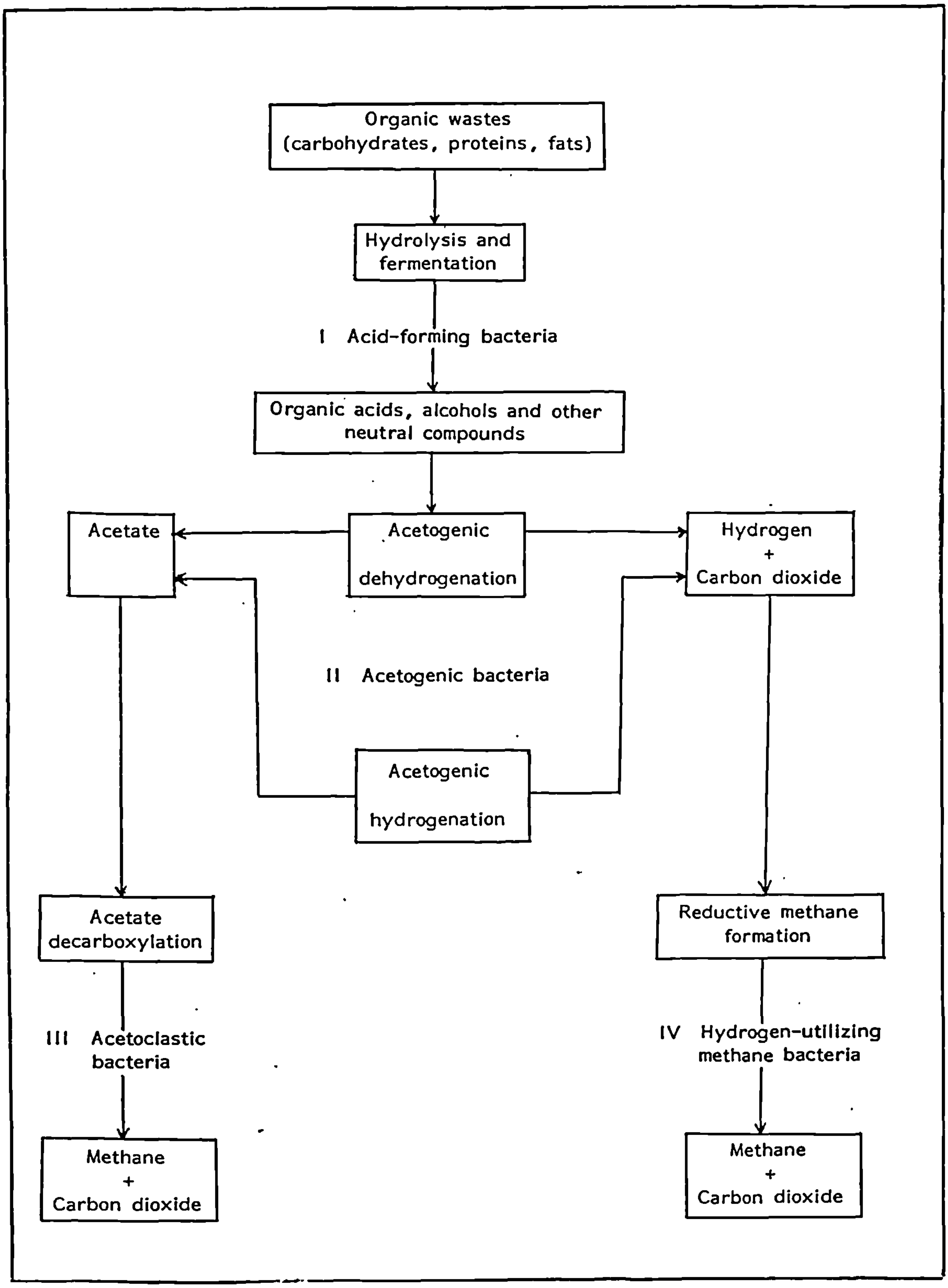


Figure 3.5: The Mechanism of the Anaerobic Digestion Process (Brown and Tata, 1985)

In the third and final step, the simple organic compounds are fermented to methane and carbon dioxide by a group of substrate specific, strictly anaerobic bacteria called "methane-formers".

Acid producing bacteria are subdivided into hydrolytic (acid forming) bacteria and acetogenic bacteria. Methane producing bacteria, which are obligate anaerobes, are subdivided into acetoclastic methanogens and hydrogen-utilising (hydrogenophilic) methanogens (Brown and Tata, 1985). In addition a small number of protozoa, fungi and yeast (Toerien and Hattingh, 1969) have frequently been observed.

Mosey (1982) and Novaes (1986) briefly mentioned the main feature of the four groups of bacteria as follows:

- Acid forming Bacteria

Acid forming (hydrolytic) bacteria are fast-growing bacteria, which are tolerant to pH and have a doubling time of about 14 hours. These bacteria can roughly be grouped into: aminolitic, cellulolytic, proteolytic and lipolytic (Whiteman, 1985). By hydrolysing many of the organic compounds in the substrate, these bacteria render many of the materials water soluble. The volatile fatty acids produced at this stage are a mixture of acetic, propionic and butyric acids according to the reactions:

$$C_6H_{12}O_6 + 2H_2O \text{ -------> } 2CH_3COOH \text{ (acetic) } + 2CO_2 + 4H_2 \qquad \text{...(3.4)}$$

$$C_6H_{12}O_6 + 2H_2 \text{ -------> } 2CH_3CH_2COOH \text{ (propionic) } + 2H_2O \qquad \text{...(3.5)}$$

$C_6H_{12}O_6$ -------> $CH_3CH_2CH_2COOH$ (butyric) + $2CO_2$ + $2H_2$ ....(3.6)

Pohland and Bloodgood (1963) reported the presence of acetic and propionic acids during high organic loadings in laboratory studies on anaerobic digestion of sewage sludge, with smaller quantities of butyric, valeric and formic acids although the preferred product observed in most studies was acetic acid. Acetic acid provides the acid forming bacteria with the biggest energy yield for growth and it provides the acetoclastic methanogens with their prime substrate for methane production.

During conditions of imbalance, when the hydrogen concentration is high or the pH is low, the total volatile fatty acids increase with propionic acid probably becoming the most prevalent. The conversion of glucose to propionic acid is used by anaerobic bacteria as a method of removing surplus hydrogen from the system (McInerney et al, 1979; Mosey and Fernandes, 1984).

- Acetogenic Bacteria

These bacteria are the bacteria that convert propionic and butyric acids into acetic acid according to the equations:

CH_3CH_2COOH (propionic) + $2H_2O$ ----> CH_3COOH + CO_2 + $3H_2$ ....(3.7)

$CH_3CH_2CH_2COOH$ (butyric) + $2H_2O$ ----> $2CH_3COOH$ + $2H_2$ ....(3.8)

The existence of these bacteria has not yet been demonstrated but Henze and Harremoes (1983) have quoted that it has only been deduced by

McInerney et al (1979) from the inability of any known methanogens to metabolise propionate and butyrate directly.

- Acetoclastic Methanogens

The acetic acid formed in the metabolism of organic matter is decarboxylated to methane by a group of bacteria called acetoclastic bacteria. These bacteria are responsible for most of the methane produced by the anaerobic process. The bacteria normally control the pH value of fermentation by converting acetic acid into a mixture of carbon dioxide and methane according to the reaction:

$$CH_3COOH \text{ ----------> } CH_4 + CO_2 \quad \text{...(3.9)}$$

They are generally considered to be most oxygen-sensitive and hence the most strictly anaerobic bacteria, highly sensitive to low pH which, due to their extremely long doubling time (3 - 5 days) are adversely affected by any abrupt nutritional or environmental change (Anderson and Donnelly, 1977; Kirsop, 1984; Stronach et al, 1986).

- Hydrogen-Utilising Methanogen

These bacteria are hydrogen-scavengers. They obtain energy for growth from the reaction:

$$4H_2 + CO_2 \text{ --------> } CH_4 + 2H_2O \quad (3.10)$$

and in doing so, they remove almost all the hydrogen from the system. They grow quite relatively quickly with a minimum doubling time of around 6 hours.

The traces of hydrogen remaining, regulate both the total rate of acid production and the mixture of acids produced by the acid-forming bacteria. As the hydrogen concentration is lowered by the bacteria, the propionic and butyric acids will be converted to acetic acid. These bacteria regulate the formation of volatile fatty acids.

3.3.2 Factors Affecting Anaerobic Process

A fundamental concern in process design is the identification of the overall rate-controlling step. The rate and extent of methane production is affected mainly by three things (Speece, 1983; Brown and Tata, 1985):

i) the nature of the substrate,
ii) the environmental and nutritional requirements, and
iii) the process configurations of the anaerobic reactor system.

Substrate that cannot be degraded biologically is obviously not appropriate for biomethanation. Biomass that is readily biodegradable on the other hand, can be digested under anaerobic conditions to produce biogas. Raw cellulosic materials such as lignin, chitin, barks and feathers are some examples of recalcitrant materials that are not easily degraded by microorganisms, owing to the slow solubilization of cellulose fibres (Brown and Tata, 1985; Temmes and Mettälä, 1986). Hobson et al (1981) reported that the recalcitrance of lignin to anaerobic biodegradation severely limits the hydrolysis rate of the raw cellulose.

Food processing industrial wastewaters are often high in starch and sugar content because of cooking operations and these simple organic matters are rapidly fermented to volatile fatty acids but, the rate-controlling step is the conversion of the volatile fatty acids to methane. Since complex wastewaters containing organic matters have a wide range of degradation rates, at low loading rates, the rate-controlling step may be acid formation, as evidence by low volatile fatty acids concentrations. But as the loading rate increases, the methanogenesis stage may gradually become the rate-controlling step, as evidenced by the accumulation of volatile fatty acids (Speece, 1983).

Methanogenesis can be initiated quickly in an anaerobic system if it is initially seeded with a suitable inoculum, such as digesting sludge. The key to starting and maintaining successful methane production lies in providing a balanced distribution of the acid formers and methane formers. As long as there is an adequate population of methane bacteria in the digester to utilize the volatile fatty acids produced by the acid formers, the process can be maintained when the proper environmental conditions are provided for the organisms.

Methanogens, unlike acid forming bacteria, are fastidious anaerobes, even low concentrations of dissolved oxygen proving detrimental whilst acid forming bacteria may be obligative or facultative, and as such more tolerant of low concentrations of oxygen (Anderson and Donnelly, 1977).

The successful production of biogas depends on providing a favourable environment for all groups of microorganism responsible for the production of methane. Methanogenic bacteria have unique environmental and nutritional requirements which should be met in order to initiate and sustain methanogenesis in a digester. Aside from maintaining anaerobic conditions, the major environmental factors that influence the production of biogas are pH, temperature, toxic inhibitors, mixing, and the nutrient levels.

- pH

A very important environmental factor is the effect of hydrogen-ion concentration in the digester. Although low pH inhibits methanogenesis, Anderson et al (1982) reported that the effect is not bactericidal. Methanogenic bacteria have been reported recovering after surviving fairly severe inhibitory conditions in laboratory reactors. Kirsop (1984) stated that anaerobic digestion can be operated successfully at any pH in the range 6.8 - 7.4. The optimum range for a well balanced biochemical reaction being 7.0 to 7.2 (McCarty, 1964). Generally, Anderson et al (1982) reported that anaerobic fermentation process of methanogenesis is impaired at pH values below 6.5 and above 8.2. The fermentative bacteria will continue to produce acids until the pH decreases to around 4.5 when the digester became "pickled" or "stuck" (Pfeffer, 1980).

Excessive accumulation of volatile fatty acids, which may occur when the organic loading rates are very high and/or when inhibitory factors are introduced into the digester, may lead to a decrease in the pH value to 6.0 and below. The situation can be corrected in two ways when this has occurred:

i) the feeding of the digester can be stopped to permit the methanogens to utilize the accumulated volatile fatty acids at their own pace with normal loading of the digester being resumed soon after the optimal gas production rates have been reestablished, and ii) the pH may be adjusted to neutrality by the addition of lime. Anderson and Donnelly (1977) and Brown and Tata (1985) stated that a good buffering capacity is generally ensured in the digester if the alkalinity of the digesting medium is maintained within the range of 2,500 to 5,000 mg/l.

- Temperature

The three temperature ranges for optimum growth in anaerobic digestion are:

i)	psychrophilic	5 - 15°C
ii)	mesophilic	20 - 45°C
iii)	thermophilic	50 - 65°C

In the U.K. sludge digesters are usually operated in the mesophilic range of 33 - 37°C, but abrupt, small temperature changes are known to be detrimental. In order to maintain the temperature within the mesophilic range, auxiliary heating may be required. The optimum temperature of growth of anaerobic microorganisms is 35°C or greater. Although anaerobic digesters have been reported to operate at lower temperature, such as 20°C (Switzenbaum and Jewell, 1980), the anaerobic growth under this operating condition was found to be protracted. Difficulties in the start-up of some reactors have also been reported (Salkinoja-Salonen, 1982).

Schraa and Jewell (1984) reported that for a stabilized thermophilic fixed-film reactor operation, careful management of the microorganisms and stringent control of operational parameters should be applied. The advantages of thermophilic processes are increased metabolic rate and increased pathogen destruction, but the major problem with these processes is the low net yield which results in very slow start-up and very slow adaptation to variation in loading rates, substrate changes or toxic inhibitors. Most of the research in thermophilic anaerobic digestion has been carried out at 55°C (Wiegant et al, 1986; Hajipakkos, 1987).

- Toxic Inhibitors

Although exceptions have been noted, methanogenic bacteria are commonly considered to be most sensitive to toxicity of all the microorganisms in the overall consortium for anaerobic conversion of organic matters to methane. Toxicity in anaerobic processes may originate from a number of sources. Accumulation of volatile fatty acids and undissociated ammonia are commonly associated with digester failure. Molecular oxygen also inhibits methane generation. However, anaerobic bacteria, like most microorganisms, can tolerate a wide varieties of toxicant (Parkin et al, 1983; Speece and Parkin, 1983) and even biodegrade some of them (Stuckey et al, 1980).

Of singular significance is the fact that acclimatization to toxicity and reversibility of toxicity are commonly noted (Parkin and Speece, 1982). Of the cations studied by McCarty and McKinney (1961), the ammonium ion was the only one which did not permit acclimatization by the microorganisms. From their study, they concluded that free ammonia concentrations above 150 mg/l are toxic to methane bacteria. A list of the more commonly encountered inhibitors to the anaerobic treatment process is given in Table 3.2 (Brown and Tata, 1985).

Table 3.2 Inhibitors to the Anaerobic Treatment Process

Parameter	Inhibiting concentration (mg/l)
Volatile fatty acids	> 2000 (as acetic acid) (a)
Ammonia nitrogen	1500 - 3000 (at pH > 7.6)
Sulphide (soluble) (b)	> 200; > 3000 toxic
Calcium	2500 - 4500; 8000 strongly inhibitory
Magnesium	1000 - 1500; 3000 strongly inhibitory
Potassium	2500 - 4500; 12000 strongly inhibitory
Sodium	3500 - 5500; 8000 strongly inhibitory
Copper	0.5 (soluble metal)
Cadmium	150 (c)
Iron	1710 (c)
Chromium^{+6}	3
Chromium^{+3}	500
Nickel	2

Notes:

a) Within the pH range of 6.6 to 7.4 and with adequate buffering capacity, volatile fatty acids concentrations of 6000 to 8000 mg/l may be tolerated.
b) Off-gas concentration of 6% is toxic (Speece, 1984).
c) Millimoles of metal per kg of dry solids.
d) Nickel promotes methane formation at low concentrations. It is required by methanogens.

Source: EPA (1979)

The toxic action of salts as seen from Table 3.2, is found to be predominantly determined by the cations, many of which are present in significant concentrations in wastes which can be treated anaerobically. Though inhibition is caused by an excessive amount of any one of the ions,it has been further demonstrated that this could be counteracted by some other ions (antagonistic ions), while it can also be exacerbated by others (synergistic ions) (McCarty, 1964). Table 3.3 listed the ions that are known to exhibit such interactions.

Table 3.3 Synergistic and Antagonistic Cations in Anaerobic Digestion Systems

toxic cations	synergistic cations	antagonistic cations
Ammonium-N (NH_4)	Ca, Mg, K	Na
Sodium (Na)	NH_4, Ca, Mg	K
Magnesium (Mg)	NH_4, Ca	K, Na
Potassium (K)	-	NH_4, Ca, Mg, Na
Calcium (Ca)	NH_4, Mg	K, Na

Source: EPA (1979)

The studies carried out by Kugelmann and McCarty (1966) on the interactions of these ions can be seen as in Fig. 3.6.

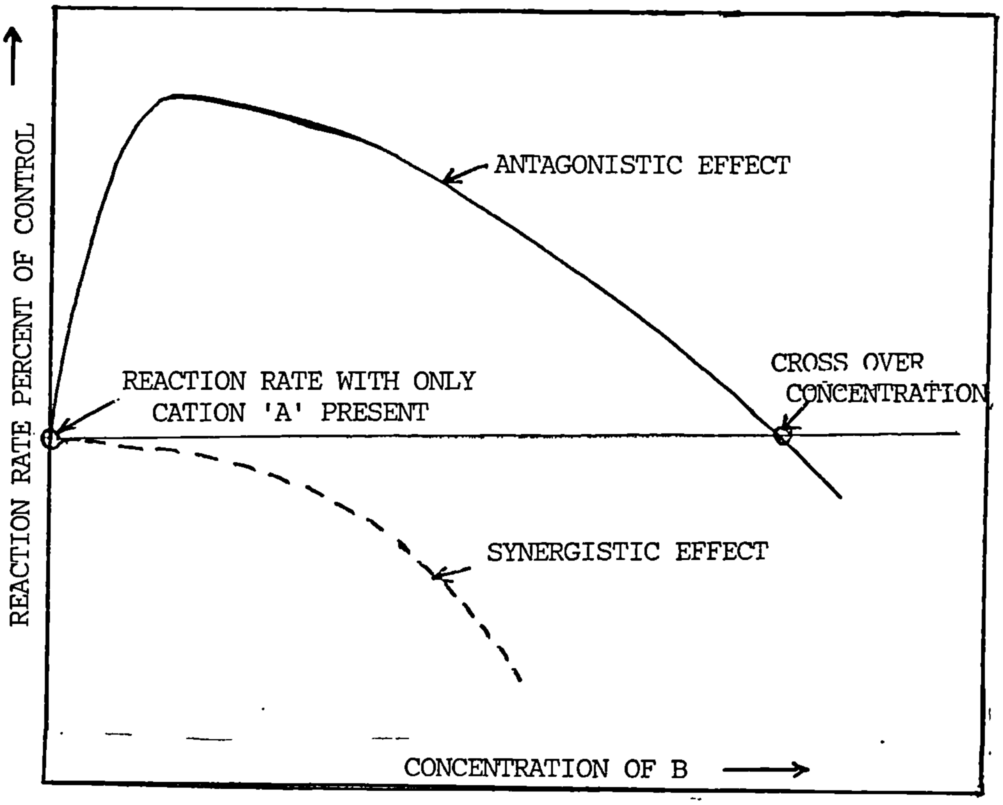


Figure 3.6: Antagonistic and Synergistic Effects of Ions on Biological Reactions (Kugelman and McCarty, 1966)

Heavy metal toxicity, with the exception of chromium, can be attenuated or relieved if sulphides are present in the wastewaters, which combine with the heavy metals to form very insoluble salts (Lawrence and McCarty, 1965). Carbon dioxide in solution as fermentation proceeds could also precipitate some metals such as zinc, cadmium, copper and lead as carbonates (Hobson et al, 1981), thus a high alkalinity is preferred when digesting wastes with high metal contents in order to reduce heavy metal toxicity by precipitation or by preventing sulphide being given of as hydrogen sulphide.

- Mixing

Mixing reduces the settlement of solids and the separation of supernatant. It provides an intimate contact between the bacterial populations and their substrate. Hence, higher rates of gas production can be realized with mixing than without it. The simplest types of reactor are unstirred, of course, but they are insufficient in performance, as evidence by their low gas production rates, low loading rates (less than 1.6 kg VS/m^3.day) and high detention time (more than 30 days) (Barnett, 1978). Mixing also reduces the potential of scum formation.

- Nutrient Mix

Nutrients are of primary importance in all forms of biological treatment. A deficiency or imbalance in the nutrients required by microorganisms may result in low bacterial metabolism, which adversely affects methane production.

Nitrogen and phosphorus are by far the major nutrients that are required for methanation. Systems operated at higher solids retention times have lower nitrogen requirements than those operating at lower solids retention times (Speece, 1984). Other nutrients such as Fe, Mg, Co, K, Na and Ca have been found also to be necessary nutrients in the anaerobic stabilization of wastes. Attention only to traditional nitrogen and phosphorus nutrient requirements appears to be grossly inadequate for methanogens. Trace metal deficiency may be the reason why even food processing wastewaters, which are among the most readily biodegradable candidates, could not support proper methane fermentation when anaerobic treatability studies on fruit cannery wastewaters were conducted at San Jose, California in the 1960s (Speece, 1983).

Four elements - iron, cobalt, nickel and sulphide - have been shown to be obligatory nutrient requirements for methanogens to convert acetate to methane (Speece et al, 1986; Hoban and van den Berg, 1979). One of the distinct features of methanogens is the nickel requirement, since nickel is generally not essential for the growth of bacteria. It was observed that the high contamination level in defined media is the reason why the nickel requirement for methanogens has long been overlooked.

Stephenson and Lester (1986) reported that high-rate anaerobic treatment systems were resistant to shock load of orthophosphoric acid. However, the presence of high concentrations of nutrients in the digester are detrimental to anaerobic digestion. Thus for those industrial wastes which have a low concentration of one or more nutrients, it becomes of engineering importance to supplement the deficiency, and of economic importance not to add too much (Anderson and Donnelly, 1977).

3.3.3 Process Configurations

The principal objective of any advanced biological reactor configuration should be to bring the substrate and enzymes into intimate contact for a sufficient time to allow the reactions to occur. For anaerobic methane fermentation processes, long microbial residence times are necessary due to the slow growth rate of the methane producing bacteria. Until now, attempts to increase the growth rate of these bacteria have essentially been unsuccessful except by changing the digestion temperature from the mesophilic (35°C optimum) to the thermophilic range (50 - 60°C) (Schraa and Jewell, 1984). By preventing bacteria from escaping in the effluent, the digestion process eventually becomes independent of growth rate. In this way it is possible to achieve a high concentration of bacteria and hence high rates of reaction in spite of very slow growth rates. This is the principle upon which advanced anaerobic technologies are based.

Anaerobic digesters can be broadly characterised as suspended-growth and attached-growth reactors. In suspended-growth reactors, the biological solids are suspended in the contents of the digester, whereas in fixed-film reactors they are made to attach themselves to surfaces such as rock, plastic, or ceramic media.

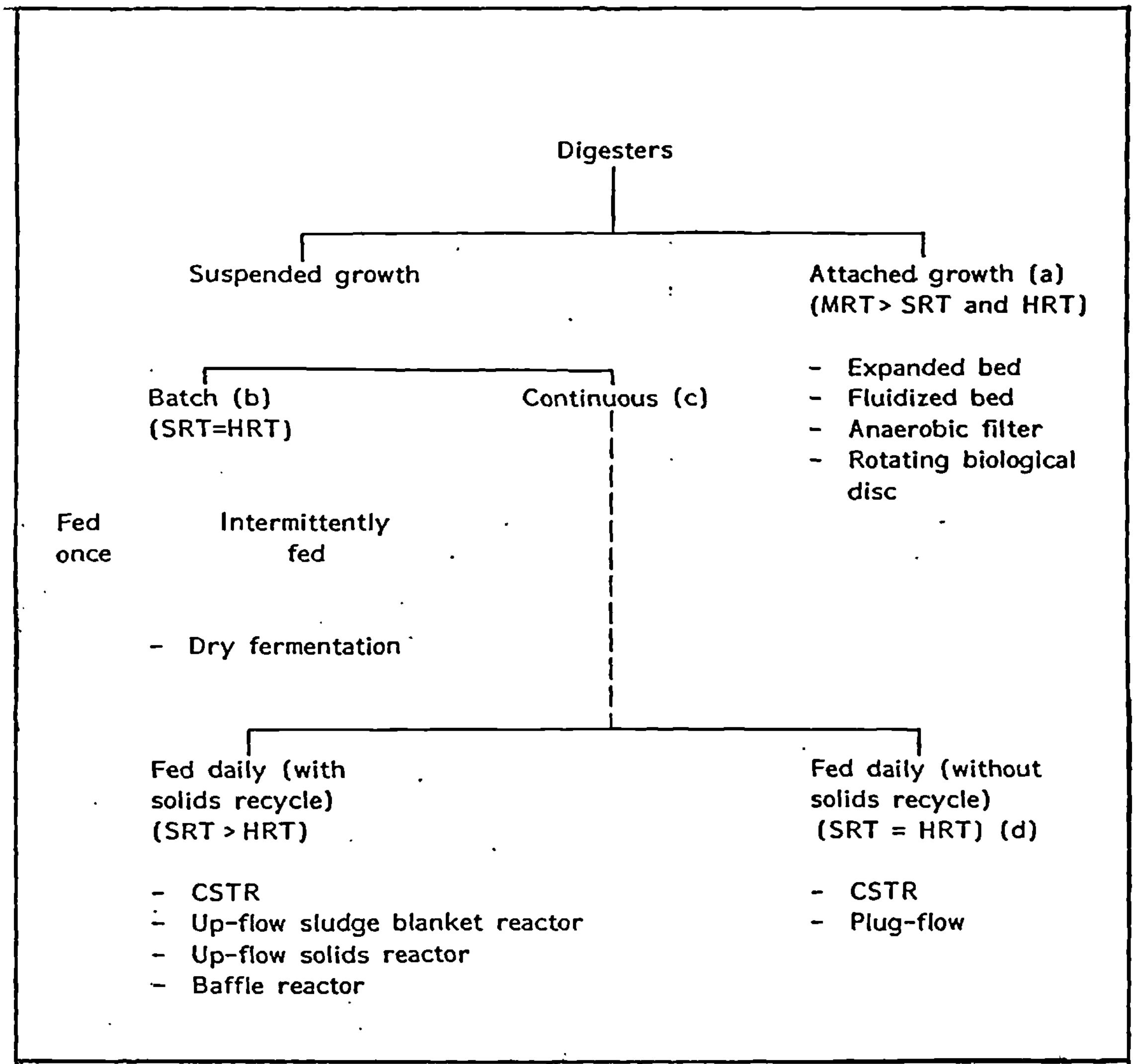
Fixed-film or attached-growth reactors are well suited to the treatment of very highly biodegradable wastes that contain a very low concentration of suspended solids. In these heterogeneous systems, the microorganisms grow in a film by attaching to the media while organic matter is removed from the liquid flowing past them. In contrast to the suspended-growth digesters, which have gained acceptance in most countries, attached-growth reactors are of relatively recent origin. The new reactors are "retained biomass reactors" and their mode of operation relies on the propensity of bacteria, especially the methanogens, for attachment to solid surface (Colleran et al., 1982)

Fig. 3.7 shows a classification of digester configurations. The classification was based on hydraulic characteristics, relationship between solids retention time (SRT) and hydraulic retention time (HRT). A brief summary of the development of various types of anaerobic digester are given below:

- Conventional Anaerobic Digester

The conventional anaerobic digester is a straight-through completely mixed reactor and is illustrated as in Fig. 3.8a. It consists of a heated digestion tank containing the waste and those bacteria responsible for anaerobic treatment. Raw waste is introduced either periodically or continuously and is preferably mixed with the digester contents. Usually the wastes are maintained in the reactor for 30 to 60 days (Vigneswaran et al, 1986). The mixed, treated waste and microorganisms are usually removed together for final disposal. Long retention times are required to produce sufficient stabilisation of sludges and to avoid washout of the slowest growing bacteria.

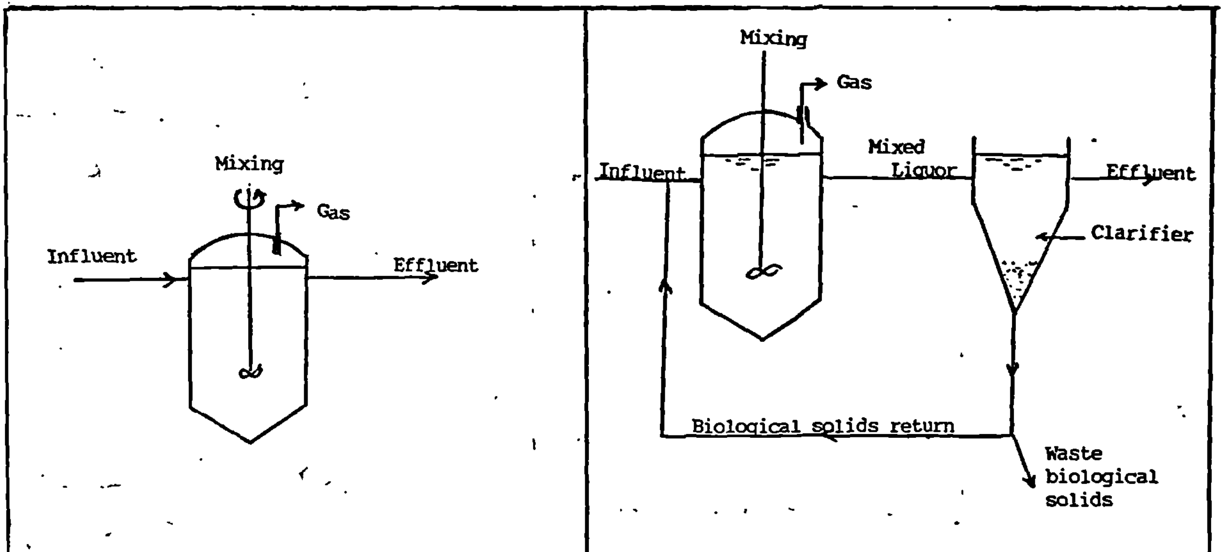
This process is most suitable for the treatment of wastes with a high organic solids content, and is generally used at a larger wastewater treatment plant for conditioning the sludge for disposal on to land. The methane produced may be used for heat or power (Isaac, 1982)



Notes: SRT = solids retention time
HRT = hydraulic retention time
MRT = microorganism retention time
CSTR = completely strirred tank reactor

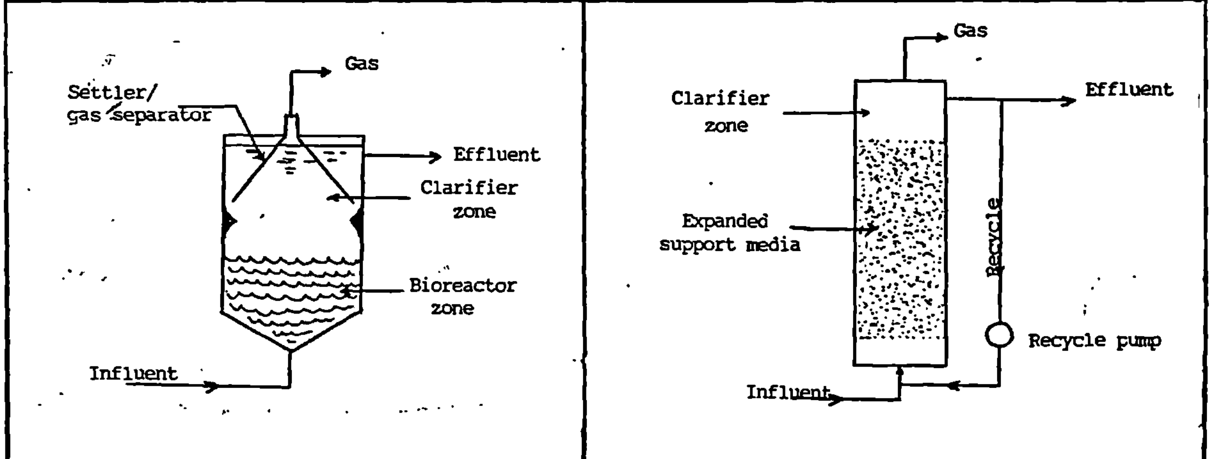
(a) usually continuously fed
(b) can be fed daily, or fed seasonally with irregular frequency
(c) with or without partial mixing
(d) SRT = HRT only when completely mixed

Figure 3.7: Classification of Digester Configurations (Fannin and Biljetina, 1984)



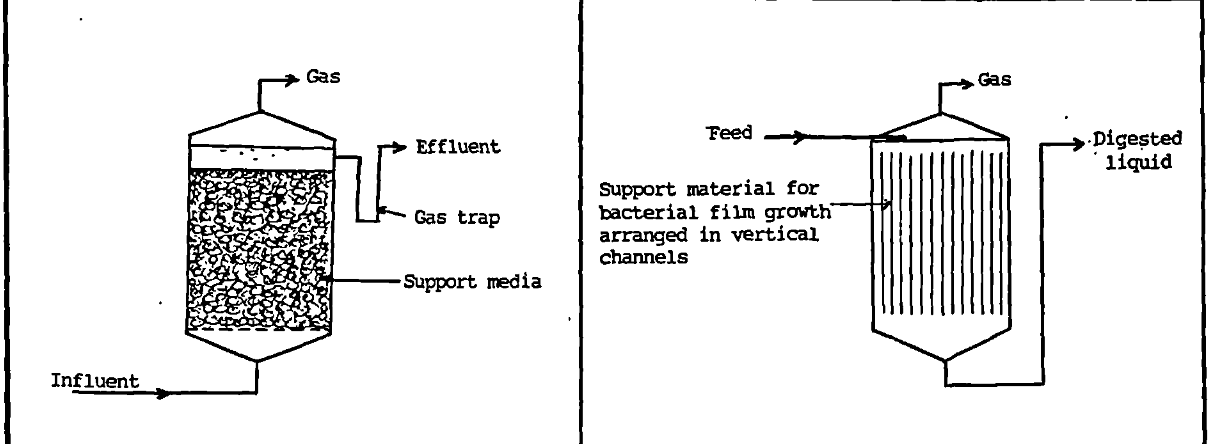
a. Conventional Anaerobic Digestor

b. Anaerobic Contact Process



c. Upflow Anaerobic Sludge Blanket

d. Anaerobic Expanded/ Fluidized Bed Reactor



e. Upflow Anaerobic Filter

f. Anaerobic Downflow Stationary Fixed Film Reactor

Figure 3.8: Diagrams of Typical Anaerobic Digesters

- Anaerobic Contact Process

Although a high SRT is necessary for efficient methane fermentation, a low HRT is preferable for system economy. The limitation of the conventional process has resulted in the development of the anaerobic contact process (Fig. 3.8b).

This process was devised to retain solids, and hence bacteria, while allowing the wastewater to flow through the system. This is accomplished by discharging the liquid/solid mixture into a clarifier and recycling the solids to the system after separation. Thus, the HRT becomes independent of the SRT, and can be reduced to a fraction of the time required in a conventional digester.

- Upflow Anaerobic Sludge Blanket (UASB) Reactor

This reactor as illustrated in Fig. 3.8c, was developed in the Netherlands in the 1970s by Lettinga et al (1980). The digester has three distinct zones:

i) a densely packed sludge blanket at the bottom,
ii) a sludge blanket at the middle, and
iii) a supernatant layer at the top.

In the UASB reactor, the microbes attach themselves to each other or to small particles of suspended matter to form conglomerates or granules. Wastewater enters at the bottom and passes through the sludge blanket. As the wastewater continues to pass upwards, the solids are separated via an internal gas-solid-liquid separator. The solids fall toward the digesting zone, thus creating a long solids retention time and a high concentration of sludge solids in the system. It has been reported that 80 to 90 percent of the decomposition of the organic matter takes place in the sludge blanket zone, which occupies approximately 30 percent of the total volume of the reactor (Lettinga et al, 1980).

- Anaerobic Expanded/ Fluidised Bed Reactor

The first application of fluidized bed technology to anaerobic treatment was developed in Jewell's laboratory at Cornell University (Jewell, 1981) for the treatment of domestic wastewater. The reactor, as shown in Fig. 3.8d, consists of inert sand-sized particles which expand (or remain in the fluidized state) by the upward flow of waste through the reactor (Biljetina, 1984).

The exact difference between an expanded and fluidized bed is somewhat ambiguous. In the expanded bed reactor, the flow has a velocity sufficient to expand the bed without necessarily causing vigorous agitation. The upward velocities are greater in the fluidized bed reactor and the bed is agitated, which results in a reactor whose contents are completely mixed (Brown and Tata, 1985).

- Upflow Anaerobic Filter

The anaerobic treatment process of special interest in this study for the treatment of leachate is the upflow anaerobic filter (Fig. 3.8e). An overview of the upflow anaerobic filter will be described later in another section.

- Anaerobic Downflow Stationary Fixed-Film Reactor

Downflow stationary fixed film (DSFF) reactors are a relatively recent addition to the family of advanced high rate-anaerobic reactors. The reactor was developed by the National Research Council in Ottawa (van den Berg et al., 1980). The DSFF reactor distinguishes itself from other type of reactors by the downflow mode of operation, the architecture of its packing (fixed biofilm support), and the absence or near-absence of suspended growth. (Vigneswaran et al., 1986). A schematic diagram is shown in Fig. 3.8f.

3.3.4 Upflow Anaerobic Filter (UAF) Treatment Process

The use of UAFs for industrial wastewater treatment is well documented. Mueller and Mancini (1975) listed information on anaerobic filters treating various industrial wastes. Henze and Harremoes (1983) gave an excellent review on the application of UAF while Witt et al (1979) and Anderson et al (1984) have also summarised the full scale treatment experiences with this system. A survey of laboratory-, pilot- and full-scale UAF installations emphasising the most important features of each of these reactors has been reported by Bonastre and Paris (1989) while the summary of design and operating factors for UAFs has been listed by Young (1991).

The UAF offers several advantages in comparison to other biological treatment systems such as (Young, 1983; Siino et al, 1985; Backman et al, 1985):

i) a high substrate removal efficiency even when operated at high loading rates,

ii) a very low biological solids production which yields less residual sludge. The high biological solids retention allows the systems to accept severe organic shock loads, and

iii) the ability to operate intermittently.

- Historical Background

According to the report made by Bonastre and Paris (1989), the first application of anaerobic filters was by Makkonen in 1953. The reactor consisted of an upflow filter of gravel and sand to treat septic tank effluent. Another study around the same period was performed by Coulter et al (1957) using a rock-filled column as part of an anaerobic treatment plant for raw sewage. The biophysical filter used was the first analogue of the anaerobic filter, though the aims of their study were to remove suspended solids from the effluent without necessarily utilizing the resulting concentration of microorganisms. Pretorius (1971) described a similar digester system consisting of a contact chamber combined with a settling zone and a biophysical filtration system. The system was used for the treatment of raw sewage at 20°C.

The concept of using an anaerobic filter as an anaerobic digestion system was first developed by Young and McCarty (1967) and it was further extended by Plummer et al (1968) to treat a high carbohydrate food processing waste. The filter media used was a mixture of Raschig rings and Berl saddles. Four filters of varying HRTs were operated, by altering the depth of filter media to give a required reaction volume for a desired retention time. The filters were operated at a temperature of 35°C and the raw waste had a COD of 8475 mg/l. It was found that the rate of substrate removal increased with an increase in retention time or a decrease in the hydraulic loading rate. Also, suspended solids in shallow anaerobic filters have been demonstrated to decrease the quality of effluent produced to a significant extent

Shucksmith (1971) analysed the performance of three UAFs, each one containing different contact media. From the experimental works carried out, it has been found that most of the bacterial activity took place in the early part of the UAF. As a result, very high concentration of volatile fatty acids have been produced in this zone which tend to lower the pH, and to overcome this large quantities of buffer such as sodium bicarbonate was added to the feed.

After Young and McCarty (1967), the UAF has since being used in numerous applications in both high and low strength industrial wastewaters such as brewery press liquor (Lovan and Foree, 1971), pharmaceutical waste (Jennet and Dennis, 1975; Seeler and Jennet, 1978; Sachs et al, 1982), dairy waste (Peterson, 1975; Caine et al, 1990), leachate from solid waste lysis (DeWalle and Chian, 1976; Chian and DeWalle, 1977, Wu et al, 1982), shellfish and food processing wastes (Hudson et al, 1978; Carrondo, 1982; Barry and Colleran, 1984; Weiland and Wulfert, 1986, 1988), liquor from activated sludge dewatering (Haug and Raksit, 1977; Donovan, 1980) and liquor from mining seepage (Gordon et al, 1988).

In general, the organic composition and concentration fluctuate from one industrial wastewater to another, and thus for better understanding of the treatment process, some researchers have tried to establish design factors affecting the UAF performance (Dahab and Young, 1982; Bonastre et al, 1986; Young and Yang, 1989) and the process kinetic (Rittmann et al, 1982; Stover et al, 1984; Gourdon et al, 1989).

- Process Description and Development

The anaerobic filter (or packed bed or submerged filter) is essentially a filter column packed with stationary media such as rocks, plastic or even glass. Typically anaerobic filters are operated in an upflow mode thus ensuring that the media are fully submerged, hence maintaining anaerobic conditions. The filter can also be operated in the downflow (van der Berg et al, 1980) or in the horizontal mode (Landine et al, 1982).

The biological reactions that take place in a UAF are identical to those in any other anaerobic system (Anderson et al, 1984), however, the advantages of an attached growth system may be summarized as follows:

i) a high mass biomass concentration may be retained within the reactor, with up to 100000 mg/l being reported by Mitchell (1981),

ii) the prevention of washout under adverse conditions and increased resistance to potentially inhibitory conditions (Duarte and Anderson, 1983) make the process more stable, and

iii) the natural stratification of the various microorganism types will allow the optimal species conditions to prevail.

The maximum organic loading rate to achieve a given treatment efficiency depends upon the wastewater characteristics and the capacity of the reactor to retain active microorganisms. High concentrations of biomass, having long solid retention periods, are achieved by promoting microbial growth on the support media (Anderson and Saw, 1986). Unlike suspended growth systems, the UAF is not limited by the settling rate of flocculated microorganisms but rather by the capacity of the media to entrap and retain the solids (Young and Dahab, 1982), and to distribute the flow (Song and Young, 1986). This allows the filter to be operated at a wide range of loading rates under stable conditions since biomass washout does not take place.

From the study conducted by Young and McCarty (1969), the UAF was found to be better suited for treatment of soluble wastes than the anaerobic contact process. This is further supported by Carter et al (1984) when they achieved superior BOD_5 and COD removals using UAFs as compared to contact processes. By using anaerobic contact processes for soluble wastes the biological solids often remain dispersed or lightly flocculated. A significant fraction may be lost with the effluent.

Tilche and Vieira (1991) in their discussion report stated that, in a UAF many authors have indicated that higher activity is associated with the biological solids retained in the void spaces in the filter media. This has also been studied by Young and McCarty (1967) who summarized that trapped suspended solids between the void spaces are mainly responsible for enhancing the performance of UAFs which was later substantiated by van den Berg and Lentz (1980).

Investigations were performed on UAF treatment efficiency using different types of media of both porous and non-porous structure. It is essential that the support materials used have a high area/volume ratio, are biologically inert, mechanically strong and resistant and economical. Baker (1991) reported that the UAF filled with porous media gave an average COD removal efficiency of 85% and was able to achieve much higher organic loading rate of up to 13 kg COD/m^3.day as compared with the one filled with non-porous media. The non-porous media could only achieved an average COD removal of 50% and an increased in organic loading rate beyond 1 kg COD/m^3.day reduced the reactor performance.

Other factors associated with support media that might affect treatment efficiency such as shape, size, porosity, pore size and presence of nutrient or inhibitory substances were also studied (Dahab and Young, 1982; Wilkie and Colleran, 1984). A high COD removal was achieved with plastic modular media of 98 m^2/m^3 surface area as compared to similar smaller media of unit surface area 138 m^2/m^3. From this finding it seems probable that shape and void size of the media are more important than actual surface area for the establishment of a satisfactory treatment performance.

Most anaerobic treatment processes, including UAF performed satisfactorily when operated in the mesophilic temperature range. Young (1991) stated that to eliminate hydrolysis from becoming the rate limiting step, complex wastes are generally treated at temperatures above 25°C. Viraraghavan and Kikkeri (1989) observed that the COD removal during the start-up was 5 - 6% more at 30°C than that at 21°C, but the variation did not affect the subsequent steady state performance. From the study conducted by Kobayashi et al (1983) the effluent quality, from a UAF with media of high surface area treating domestic waste, was found to be poor as the temperature decreased from 25°C to 20°C. However, the performance at 25°C and 35°C did not give any significant difference.

The start up of anaerobic treatment processes play an important role in the success of the processes. Anaerobic filters are most sensitive to changes in pH during start up but once steady state is achieved, a moderate change in pH is tolerable. Response to adverse pH levels was investigated by Clark and Speece (1971) who observed a rapid recovery with systems exposed to a pH of 5.4 for 12 hours.

Howerton and Young (1987) carried out a laboratory study to evaluate the performance of UAFs when operating in the two-stage cyclic mode. The two-stage cyclic mode operation was reported to significantly improve the UAF performance as compared to single-stage unit at the same OLR and HRT. The improvement in COD removal was between 8% at a loading rate of 4 kg COD/m^3.day and 11% at 8 kg COD/m^3.day.

3.4 Kinetic Models for Fixed Film Processes

The prediction of reactor performances in both aerobic and anaerobic fixed-film reactors have been studied by several researchers (Friedman et al, 1976; Chen and Hashimoto, 1980; Rittman, 1984; Braha and Hafner, 1987; Stover and Gonzalez, 1988). The outcome of which several empirical and semi-empirical steady state and mathematical models have been developed to explain the mechanism of substrate removal by biological treatment processes. All of these models have indicated that at high substrate concentrations the rate of substrate removal per unit cells remain constant to a limiting substrate concentration below which the rate will become concentration dependent and decrease, but the rate of cell growth in a treatment process may continue at a maximum longer than the rate of substrate removal due to assimilation of stored substrate.

The relationships describing the substrate conversion and microbial growth are usually developed along the line of Monod's work using equations in the Michaelis-Menten format. Such kinetic expressions are incorporated into diffusive mass transport material balances and various hydraulic properties of the reactors in order to arrive at overall rates of substrate transformation within the biofilms (Farquhar, 1979; Meunier and Williamson, 1981).

Hence, understanding the mechanisms that affect microbial growth and substrate utilization, and those that affect the transport within the microbial film, is essential for modelling biofilm reactors.

3.4.1 Microbial Growth Rate

Three fundamental relationships underlying the theory of biological growth are (Donnelly, 1984):

i. growth rate,
ii. growth yield, and
iii. relationship between an essential nutrient concentration and growth rate.

The log-growth phase can be used to describe the bacterial growth in a batch culture. The rate of microbial growth during this phase is proportional to an increase in their weight, and is described by the autocatalytic equation below:

$$\frac{dX}{dt} = \mu X \qquad \text{.....(3.11)}$$

where:

dX/dt = rate of microorganisms generation (mg/l/day)

X = microorganism concentration (mg/l)

μ = specific growth rate (day^{-1})

Endogenous respiration that is the self-destruction of biomass, cell maintenance, predation, cell death and lysis are processes leading to a decrease in microbial biomass (Pavlostathis and Giraldo-Gomez, 1991). All these factors represent the microbial decay. The microorganism decay rate is usually employed for the modification of the growth rate (Lawrence, 1971):

$$\frac{dx}{dt} = (\mu - K_d)\, X \qquad \text{.....(3.12)}$$

where:

dX/dt = net rate of microorganism generation (mg/l/day)

K_d = specific microorganism decay rate (day^{-1})

Monod (1949) adequately described the effect of a limiting substrate (i.e., the essential nutrient) concentration on the rate of microbial growth:

$$\mu = \frac{\mu_{max}\, S}{K_s + S} \qquad \text{....(3.13)}$$

where:

μ_{max} = maximum specific growth rate (day^{-1})

S = growth limiting substrate concentration (mg/l)

K_s = half-velocity coefficient (i.e., substrate concentration at one-half maximum specific growth rate) (mg/l)

This is the basis for all continuous-flow treatment processes in biological wastewater treatment in which microorganisms are continuously cultivated but the overall rate of metabolism is controlled by the substrate concentration. This equation has the same form as the Michaelis-Menton equation which describes the rate of reaction of an enzyme with the substrate concentration (Benefield and Randall, 1980).

The specific growth rate of the equation of microbial growth as in Equ. 3.11 can be replaced by the Monod function in Equ. 3.13 so that

$$\frac{dX}{dt} = \frac{\mu_{max}\, S.X}{K_s + S} \qquad \text{....(3.14)}$$

Substituting Equ. 3.13 into Equ. 3.12 gives:

$$\frac{dX}{dt} = \frac{\mu_{max}\, X\, S}{K_s + S} - K_d\, X \qquad \text{....(3.15)}$$

Microbial growth rate can be related to the substrate utilization rate as follows (Monod, 1949):

$$Y = -\frac{dX/dt}{dS/dt} \qquad \text{....(3.16)}$$

where:

Y = growth yield coefficient (mg cells generated/mg substrate utilized)

dS/dt = substrate utilization rate (mg/l/day)

Combining Equ. 3.14 with Equ. 3.16 gives:

$$\frac{dS}{dt} = -\frac{\mu_{max} X S}{Y (K_s + S)} \qquad \text{....(3.17)}$$

3.4.2 Model Prediction of Rotating Biological Contactor

Several models have been developed to predict the rate of substrate removal in a rotating biological contactor operating under aerobic conditions (Steels, 1974; Kornegay, 1975; Friedman et al, 1976; Ouano and Pescod, 1976; Hansford et al, 1978; Famularo et al, 1978).

Most of these models have been applied to the treatment of weak organic wastewaters (up to 500 mg/l BOD_5) and at fairly low organic loading rates (20 g BOD_5/m^2.d or less). However, the design chart developed by Steels can be used to predict removal rates for loading rates of up to 80 g BOD_5/m^2.d.

Most of the models predicted were based on specific operating parameters and/or empirical constants, which differ from one model to the other. Hence comparison between the models will not be reliable more so with the unavailability of published material.

Norton (1984) using regression analysis of published data on aerobic RBC's has presented an empirical model which gave a reasonable assessment of the likely performance of any type of aerobic RBC system based on applied loading rate and also allowed the selection of a suitable design loading rate to suit any particular influent condition and effluent quality constraint. The regression equation is as follows:

$$g/m^2.d\ BOD_5\ Removed = 0.986\ (g/m^2.d\ BOD_5\ Loading)^{0.965} \qquad ....(3.18)$$

Norton (1984) showed that, from tracer study of a once-through flow, the RBC to be approximately a completely mixed reactor rather than a plug flow. It is thus apparent that staging the reactor did not influence the flow and the performance of the reactor.

Several models that have been used to predict process operation assumed a steady state relationship of the form (Kornegay and Andrews, 1967; Kincannon and Stover, 1982, Hamoda and Wilson, 1989):

$$\frac{Q}{A}(S_i - S_e) = \frac{dS}{dt_A} \qquad ....(3.19)$$

where dS/dt_A = substrate utilization rate is defined as:

$$\frac{dS}{dt_A} = \frac{U_{max}\,\frac{QS_i}{A}}{K_B + \frac{QS_i}{A}} \qquad \text{....(3.20)}$$

where

Q = flow rate (l/day)

A = total media surface area (m^2)

S_i = influent substrate concentration (mg/l)

S_e = effluent substrate concentration (mg/l)

U_{max} = maximum substrate removal rate ($g/m^2.day$)

K_B = proportional constant ($g/m^2.day$)

The above kinetic expressions have also been applied to predict the performance of the UAF (Stover et al, 1984; Stover and Gonzalez (1988).

3.4.3 Model Prediction of Anaerobic Filter

Kinetic analysis of anaerobic filter performance frequently employs a first order assumption: (Mueller and Mancini, 1975; Rittman, 1982; Rittman et al, 1982; Lindgren, 1982). Rittman (1982) stated that the first order relations were used since they described the results better than other simple alternatives such as zero order and one-half order.

From most of the studies performed (Chavadej, 1978), anaerobic filter with no recycle was found to approximate a plug flow relation. The first order relation used is:

$$\frac{dS}{dt} = -kS \qquad \text{....(3.21)}$$

where

S = substrate concentration (mg/l)

t = time (days)

k = first-order coefficient (day^{-1})

Integrating Equ. 3.21 gives

$$\ln\frac{S^{o}}{S} = k\theta \qquad \text{....(3.22)}$$

where

S^{o} = influent concentration (mg/l)

θ = reactor detention time (day)

Young (1980) developed another kinetic expression that is:

$$\eta = 100(1-\frac{\varepsilon}{HRT}) \qquad \text{....(3.23)}$$

where

η = ultimate substrate removal efficiency (%)

ϵ = proportional coefficient (day)

HRT = hydraulic retention time (day)

3.5 Physical and Chemical Treatment Processes

The advancement in science and technology has created various treatment processes which are already used in large-scale operations. The requirement for achieving higher quality effluent from municipal and industrial waste treatment processes has led to the use of more and more physical and chemical treatment processes (Annesini et al, 1987). Besides activated carbon adsorption, other physical and chemical treatment or advanced treatment processes generally employed are chemical precipitation, coagulation, ultrafiltration, microfiltration, ion exchange, reverse osmosis and electrodialysis.

The character of compounds presence in the wastewater, such as molecular weight and chemical structure, determined the effectiveness of the physical-chemical process in removing the organic matters (Chian, 1977). Generally, chemical precipitation and coagulation are used in removing colour and turbidity. Chemical precipitation is also used to remove manganese and iron from wastewater by the addition of lime or soda ash (Metcalf and Eddy, 1979). Ho et al (1974) treated leachate with lime and found out that it did little to remove anything but iron and colour. Hence it can be seen that the physical-chemical treatment processes have greater capability for removing heavy metals as compared to the biological treatment processes. But the physical-treatment processes do not remove all inorganic and organic pollutants and thus the problem of build up of resistant compounds is not completely solved (Bishop et al, 1972).

Rebhun and Streit (1974) reported that using a chemical flocculation-clarification process, high colour as well as all the suspended and most of the colloidal fractions were removed from a strong municipal wastewater but the organic matters removal were low. Introducing the effluent from the flocculation-clarification process to activated carbon adsorption columns, further removal of organic matters was achieved, but the residual organic fractions were biochemically unstable indicating the needs for a biological treatment.

Among tertiary treatment system, activated carbon adsorption is the most commonly used process. Although activated carbon adsorption in fixed beds is widely used in wastewater treatment, little fundamental knowledge is available for the design of carbon beds for this purpose. This knowledge is essential in order to ascertain the suitability of activated carbon for treating wastewater, as well as for design purposes.

3.5.1 Activated Carbon Process

The first occurrence of adsorption was noted by Scheele in 1777, when he observed the selective removal of gases from air by charcoal (Weber, 1968) and today it is recognized that adsorptive reactions are prevalent in most natural physical-chemical and biological processes. Adsorption onto activated carbon has been a useful and effective process for treating industrial wastewaters and for advanced treatment of effluent from biological treatment plants. In the mid-nineteenth century, carbon was used to remove odours and tastes in drinking waters and since then, water and wastewater treatment with carbon has become widespread in municipal and industrial processes (Cheremisinoff and Morresi, 1978).

However its use in the removal of inorganic metal ions from wastewater is rather rare (Corapcioglu and Huang, 1987).

The performance of an activated carbon column is often measured by the reduction in concentration of collective parameters such as TOC, COD or dissolved organic carbon (DOC) (Summers and Roberts, 1984). However, the treatment process is contingent on many factors and extensive experimental pilot plant studies are required to determine the appropriate design methods (Yen and Singer, 1984). These factors include the amenability of the dissolved constituents to sorption, the presence of other substances which promote or inhibit the sorption process, the soundness of engineering, and proper operation and maintenance of the system (Ford, 1981).

The activated carbon process, regardless of the applied mode, has process limitations and should be carefully investigated prior to making process commitments. It should be recognized that many classes of organic compounds are not amenable to carbon adsorption - particularly oxygenated organic substances - and show up as residual BOD_5, COD, or TOC in carbon column effluent. This limits the overall process efficiency of activated carbon when treating many industrial wastewaters.

3.5.2 Factors Affecting the Adsorption Capabilities

The adsorption capabilities of activated carbon are influenced by several factors. Some of the factors affecting adsorption include (Cheremisinoff and Morresi, 1978; Ford, 1981):

i) the physical and chemical characteristics of the adsorbent such as surface area, pore size, chemical composition, etc. Generally the larger the surface area, the more adsorption that can take place,

ii) the physical and chemical characteristics of the adsorbate, such as molecular size, molecular polarity, chemical composition, etc. Ford (1981) stated that as a rule, branched-chain compounds are more adsorbable than straight-chain compounds. Also molecules with low polarity are more sorbable than highly polar ones. Unless the screening action of the carbon pores actually impedes, large molecules are more sorbable than small molecules of similar chemical nature. This is attributed to more solute carbon chemical bonds being formed, making desorption more difficult,

iii) the concentration of the adsorbate in the liquid phase (solution). Generally , strong ionised solutions are not as adsorbable as weakly ionised ones (that is undissociated molecules which have low solubility are in general preferentially adsorbed),

iv) the characteristics of the liquid phase such as pH, temperature. A low pH promotes the adsorption of organic acids whereas a high pH would favour the adsorption of organic bases. Adsorption reactions are generally exothermic and hence, high temperature impedes adsorption process, and

v) the residence time of the system.

3.5.3 Activated Carbon Systems

Basically, there are two forms of activated carbon, powdered and granular. The former are particles that are less than U.S. Sieve Series No. 50, while the latter are larger (EPA, 1973). The adsorption rate is influenced by carbon particle size, but not the adsorptive capacity which is related to the total surface area. By reducing the particle size, the surface area of a given weight is not affected. Particle size contributes mainly to a system's hydraulics, filterability and handling characteristics (Cheremisinoff and Morresi, 1978).

The criteria employed for selection of the particular carbon to be used are (Weber and Morris, 1963):

i) the reasonable adsorptive capacity,
ii) the freedom from substances that might interfere with analytical methods,
iii) the ease of preparation in desired particle sizes, and
iv) the resistance to attrition.

The applicability of granular or powdered carbon systems must first be proven using bench- or pilot-scale analyses. In many applications, the preliminary evaluation programme may take the form of batch isotherm studies. The use of carbon isotherm tests, while not providing a basis for design, does provide a "screening" analysis for assessing the effectiveness of a given carbon in removing defined organic constituents (Ford, 1981).

3.5.4 Adsorption Isotherm Study

The adsorption capacities of carbon are usually determined by batch experiments. In such studies, a liquid-phase isotherm shows the distribution of adsorbate between the adsorbed phase and the solution phase at equilibrium. This distribution changes with adsorbate concentration due to changes in the driving force for adsorption. At a low concentration the driving force is reduced resulting in lower adsorption capacities (Wagner and Jula, 1981).

The adsorption isotherm for organic substances in an aqueous solution can be expressed by using the empirical Freundlich equation or Langmuir equation. Pirbazari and Weber (1984) investigated several different models to mathematically describe the equilibrium data for dieldrin removal from water, and the Freundlich equation was found to provide the best statistical fit.

The Freundlich adsorption equation is perhaps the most widely used mathematical description of adsorption in aqueous systems (Faust and Aly, 1987). The equation, which relates the amount of adsorbate in the solution phase to that in the adsorbed phase, can be expressed as:

$$\frac{x}{m} = kC^{\frac{1}{n}} \qquad \text{.... (3.24)}$$

where

x = amount of adsorbate adsorbed (mg)
m = weight of carbon (g)
c = equilibrium concentration of adsorbate in solution (mg/l)
k & n = constants

for linearization of data, the equation can be expressed in logarithms to give:

$$\log \frac{x}{m} = \log k + \frac{1}{n} \log C \qquad \text{.... (3.25)}$$

3.5.5 Dynamic Adsorption

The next step in the adsorption studies is to evaluate the liquid in a dynamic test. It is well-known that adsorption on granular activated carbon is a diffusion process consisting of the following steps (Schuliger, 1978):

i) bulk diffusion of the adsorbate from the liquid to the film around the carbon particle,
ii) diffusion through the film, and
iii) internal pore diffusion to the adsorption sites.

Usually, the film diffusion or the pore diffusion acts as the rate-limiting step. Utilising this basic understanding of the adsorption process, one can frequently make changes in the operation of dynamic systems to improve the overall efficiency (Wagner and Jula, 1981).

Schuliger (1978) stated that ideally the performance of carbon can be predicted from the equilibrium data using mathematical techniques. However, due to limitations, it will be necessary to test each solution in a dynamic system. Before conducting the column test, several factors should be taken into consideration as they determined the shape of the exhaustion curve and the height of the adsorption zone. These factors are listed below:

i) location of test such as in the plant or in the laboratory,

ii) size and type of system, that is the column diameter and quantity of carbon,

iii) carbon type and particle size,

iv) linear velocity of liquid in the carbon bed, and

v) temperature and pH.

3.5.6 Process Description

There are several types of carbon system used in the treatment of organic substances in wastewater, each with their own advantages and disadvantages. The most common type of contactor is the stationary or fixed adsorbent bed (Hutchins, 1981). The influent flows through the bed to the breakthrough point or until the adsorptive capacity of the bed is exhausted. The entire bed is then removed from the contactor and replaced by virgin carbon. The selection of a particular system can be made by assessing the flow rate, wastewater characteristics, effluent requirements, application, treatment process and economics (Cheremisinoff and Morresi, 1978). Although different approaches in applying activated carbon to the successful treatment of wastewaters are constantly evolving, the most commonly modes utilized are:

i) Downflow Fixed-Bed Columns,

ii) Countercurrent Pulse-Bed Columns, and

iii) Upflow-Downflow Columns.

- Downflow Fixed-Bed Columns

These contactors can be either of the pressure or gravity type, with pressurized systems being the more prevalent. A typical pressurized downflow carbon contactor is shown in Fig. 3.9a. The columns are used to removed organic substances by adsorption and suspended solids by filtration. However, backwashing should be provided due to excessive head losses, and this could incur higher operating costs which offset the economic gain of using this type of adsorption system.

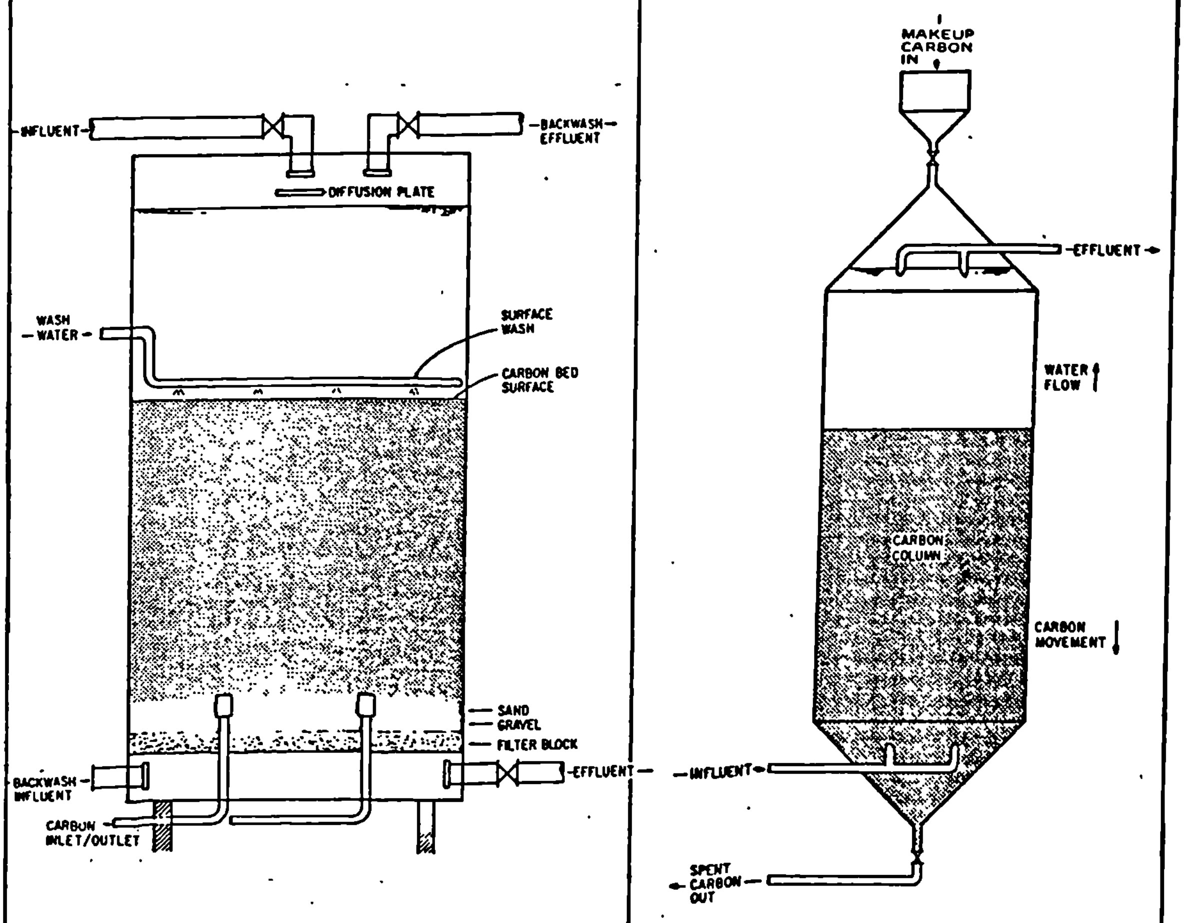
- Countercurrent Pulse-Bed Columns

The pressurized pulse-bed column or moving bed contactor operates on the countercurrent principle by continuously withdrawing and replacing small quantities of carbon. The influent feed enters the bottom of the column and flows upward through the carbon bed while spent carbon is removed from the bottom and an equal volume of fresh carbon is added to the top.

The system effects a highly efficient use of carbon, reducing regeneration and carbon makeup costs. Because of the efficient utilization of the carbon adsorptive capacity, this system is used when the carbon usage rate is high (Faust and Aly, 1987). These system, shown in Fig. 3.9b when properly executed, means that no carbon is withdrawn from use until it is completely exhausted.

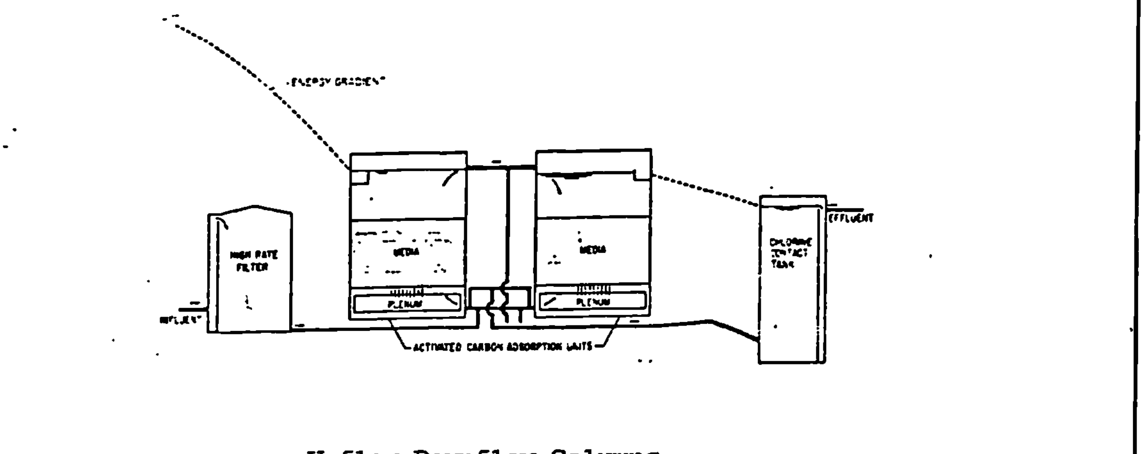
- Upflow-Downflow Columns

The tandem upflow-downflow concept of using activated carbon columns in treating wastewater, which is developed by Zurn Industries (Ford, 1978), provides a countercurrent, two-bed series system as illustrated in Fig. 3.9c. The two beds are arranged so that the gravity,



a. Downflow Fixed-Bed Column

b. Countercurrent Pulse-Bed Column



c. Upflow-Downflow Columns

Figure 3.9: Typical Activated Carbon Column Systems (Ford, 1978)

open top structures are operated as a series upflow "roughing" contactor. Once breakthrough occurs, the pair of columns are taken off line, the spent upflow column regenerated, and the unused capacity of the downflow column is used by reversing the flow and employing it as the upflow reactor using the former upflow column containing regenerated carbon as the downflow polishing unit.

3.6 Biological and Physical-Chemical Treatments of Leachate

3.6.1 Aerobic Treatment Methods

Activated sludge plants have been used to treat landfill leachate (especially in Germany where the pollution problems of leachate first became a public issue due to their active environmental groups), but data on full-scale plants are rare and most treatment studies carried out in this country have been on a laboratory- or pilot-scale plants (Stegmann and Ehrig, 1980). Palit and Qasim (1977) showed that leachate could be treated using a conventional activated-sludge process, although occasional problems with sludge bulking and poor solid/liquid separation were encountered.

The activated sludge plant required to treat the leachate from a large landfill site may be extensive and will therefore incur high capital costs. The comparison of costs between on-site treatment method and direct discharge to sewer indicate that little economic benefit may be obtained by using a direct discharge unless the strength of leachate is greater than 2000 mg/l COD. In order to optimized and extend the operational life of the activated sludge plant, Cameron and Koch (1980) suggested recirculation of leachate onto landfill site, which also helped in reducing the toxicity of leachate. Bull et al (1983) reported that when possible, the addition of leachate into a municipal sewer is the preferred option.

Boyle and Ham (1974) investigated the treatment of different proportions of leachate with domestic sewage (BOD_5 140 mg/l) in a laboratory-scale activated-sludge plant. The leachate used in this investigation had a BOD_5 of 8800 mg/l and was added in proportions up to a maximum level of 20% of the influent by volume. Sewage containing 5% leachate in the influent (a total daily organic loading rate of approximately 0.15 kg BOD/kg MLSS) could be treated without seriously impairing effluent quality. Additions of leachate greater than 5% resulted in substantial solids production, increased oxygen-uptake rates, poorer mixed liquor separation and unsatisfactory levels of effluent BOD.

Chian and DeWalle (1977) performed similar experiments using a stronger leachate with an influent BOD_5 of 24700 mg/l. A constant daily organic loading rate of about 0.3 kg BOD_5/kg MLSS was maintained by varying the retention time and increasing the leachate being added. However, effluent BOD_5 increased with increasing proportions of leachate, showing that large quantities of refractory substances were present. Units receiving a 4% leachate by volume failed, as indicated by a high effluent BOD_5 and deterioration of sludge characteristics.

As a general rule a leachate volume greater than 5% is not normally acceptable to a receiving works. This is because of the very high organic content and ammonia nitrogen concentration (5,000 - 30,000 mg/l BOD_5 or COD and 100 - 300 mg/l NH_3-N).

Although high COD and BOD_5 were found to be substantially reduced in a combined treatment of leachate with domestic wastewater, Kelly (1987) stated that many uncertainties still remained about the feasibility of combined treatment concerning ammonia conversions, temperature effects, sludge production, foaming problems, settleability difficulties, and heavy metal accumulations, as well as effects of precipitation on treatment plant operation.

The amenability of leachate to biological treatment varies due to the variation of leachate characteristics from one landfill site to another (Gaudy et al, 1986). The combination of the various types of leachate with the different types of wastewater (that is domestic, agricultural and industrial) would also effect the performance of the combined treatment differently. Hence, evaluation on the combined treatment for each specific leachate and wastewater is required.

Kang et al (1989) in their study, successfully removed BOD_5 and COD in excess of 90% from leachate originating from a hazardous waste landfill by using a conventional activated sludge and a powdered carbon activated sludge treatment systems. The powdered activated carbon treatment (PACT) mode is a modified form of activated sludge treatment process. Another adaptation of the activated sludge process is the oxidation ditch, or Pasveer ditch, which can be used as a low-cost alternative to the conventional process. However, retention times must be increased to achieve the same performance and therefore, a greater land area is required to accommodate the plant.

Another process to be considered is an aerated lagoon. A study carried out by the Water Research Centre at Ugley landfill (Maris et al, 1984) using a pilot-scale aerated lagoon gave a high BOD_5 and COD removal of over 99% and 96% respectively although residual suspended solids (SS) are a problem. At an organic loading rate of 0.2 kg BOD_5/kg MLVSS an average influent BOD_5 of 10000 mg/l was reduced to 30 mg/l. However, the SS could be effectively reduced by further settlement in a maturation pond.

Maris et al (1984) also reported a full-scale 10-day aerated lagoon in operation at the Bryn Posteg disposal site in Montgomery District Council in Wales treating 115 m^3/day of leachate. The results showed that a reductions in COD from 4000 mg/l to 85 mg/l and BOD_5 from 3000 mg/l to less than 5 mg/l were being achieved. Ammonia-nitrogen was also reduced from 100 mg/l to less than 1 mg/l.

Until very recently, operation of aerobic processes has not been wholly successful and many full-scale plants have failed. Comparing the aerobic processes, the aerated lagoon is to be preferred to the activated sludge process because it is felt that it provides the degree of flexibility required to accommodate the transitory nature of the leachate treatability and strength.

Other options that could be used for treating leachate are biological filters and rotating biological contactors (RBCs). The treatment processes, especially for nitrification of leachate from "aged wastes", may be an option for the future. A review of the rotating biological contactor has been discussed earlier in this chapter.

3.6.2 Anaerobic Treatment Methods

Anaerobic treatment processes have been described fully in section 3.3. These processes not only take place within an anaerobic digester, but also within the landfill producing the organic fractions which appear in the leachate, giving the leachate its high BOD. When used for the treatment of wastewater, anaerobic digestion takes place in a reactor vessel where the bacteria may be suspended within a mixed liquor (conventional digester), or attached to some form of medium (RBC or filter). The pH in the reactor is kept above 7.0 to obtain methane, which is then used to maintain the temperature in the reaction vessel at around 35°C (mesophilic digestion). The production of a potentially saleable end-product, and the low volumes of waste sludge produced make the anaerobic process an attractive alternative to aerobic processes.

A major disadvantages of these processes are the long retention time required, the large capital costs and the difficulties of operating the plant to maintain an adequate population of methanogenic bacteria to provide treatment. The bacteria are inhibited by acidic pH values and are also sensitive to the presence of some heavy metals. These inhibitions can cause reduced growth rates and lead to a net washing-out of microbial cells from a completely-mixed reactor system. Another disadvantage is that the environment within an anaerobic digester does not provide suitable conditions for the removal of nitrogen, in the form of ammonia, and ferrous iron. These will pass through the reactor unchanged and may in fact inhibit the methanogenic bacteria. Data on the performance of anaerobic processes for leachate treatment is again only limited to laboratory- and pilot-scale plants.

Wu and Kennedy (1984) reported that more than 96% of the COD could be removed from a high strength acidic leachate when the organic loading rate was less than 1.2 kg COD/m^3.day. Mennerich and Albers (1986) achieved a COD reduction above 70% at organic loading rates up to 30 kg COD/m^3.day using an anaerobic filter. They also stated that the upflow anaerobic sludge bed process might be a suitable method for leachate treatment.

Boyle and Ham (1974) showed that greater than 90% removal of organic matter from leachate, as measured by COD and BOD$_5$, was possible by storage under anaerobic conditions for 10 - 12 days at a-temperature of between 23°C and 30°C. The organic loading rate was 1.05 kg COD/m^3.d. Further experiments showed that temperature was an important factor affecting the efficiency of anaerobic units in the range of 11°C to 23°C; with an organic loading rate of 0.67 kg COD/m^{3}d, removal efficiency dropped from 87% at 23°C to 22% at 11°C, with a retention time of 12.5 days.

In another experiment, Foree and Reid (1973) operated five completely-mixed, fill-and-draw anaerobic digester units of 1.5 l capacity under various conditions of organic loading and temperature, with and without additions of lime and nutrients. They concluded that the addition of nutrient and lime did not contribute significantly to the removal of organic matter (the leachate initially had a COD of 12900 mg/l and total soluble phosphorus concentration of 12.5 mg/l). 95% COD removal was achieved at 35°C with an organic loading rate of 0.64 kg COD/m^3.day, but only 77% removal was obtained when the organic loading rate was increased to 1.28 kg COD/m^3.day at 35°C. Results for this digester could be compared with data from a UAF with an organic loading of 1.28 kg COD/m^3.day at 35°C. This filter consisted of a column 1.8 m high, 150 mm diameter, containing limestone fragments through which leachate was pumped upwards. A COD removal of 96.1% was achieved when the unit had reached a steady state after 79 days. The greater efficiency of UAF when compared with corresponding completely-mixed digester is explained by the fact that microorganisms are largely retained within a filter, whereas they may be lost in the effluent from a digester.

Carter et al (1985) reported that alkalinity in the leachate was predominantly due to salts of the volatile fatty acids and not the normal bicarbonate alkalinity found in industrial wastes. These neutralized volatile fatty acids could readily be treated anaerobically. They observed that the volatile fatty acids at concentrations of 8000 mg/l or less were not toxic to the anaerobic bacteria during start-up of an UAF treating leachate, as long as the pH is maintained in a suitable range. The BOD$_5$ removal achieved was over 90% at organic loading rate of 14.2 kg BOD$_5$/m^3.day.

Henry et al (1987) conducted a laboratory study using a UAF to remove organic substances in raw leachate from older (COD of 3750 mg/l) and relatively new (COD of 14000 mg/l) landfill sites. The results obtained were compared with results from study a previously carried out using leachate from a "mature" landfill (COD of 1900 mg/l). The UAF was found to reduce the COD from different landfills by 90 percent at loading rates between 1 to 2 kg COD/m^3.day with hydraulic retention times of 24 to 96 hours without any phosphorus supplement even though the phosphorus content in the raw leachate was low.

Carter et al (1984) used pilot plants consisting of two anaerobic contact digesters and two UAFs to treat leachate which consisted primarily of volatile fatty acids. One UAF was designed using media that enhances a cross flow pattern in the contact digester while the other filter contains a media that allowed only vertical flow through the contact digester. The two UAFs achieved the highest soluble COD removals of 85% as compared to 65% and 50% for the two contact digesters. The soluble COD results indicated excellent biological activity in the two UAFs. Although the effluent suspended solids from UAFS were much higher than those from contact digesters, the poor COD removal efficiencies of the two contact digesters were attributed to high volatile fatty acid concentrations. The volatile fatty acid concentration of the two UAFs were about 2300 mg/l while those for contact digesters averaged 7400 mg/l.

A study carried out by Chian and DeWalle (1977) indicated that more than 95% of the organic matter from a leachate with a COD of 54000 mg/l and pH of 5.4 could be removed, when the influent leachate was diluted with recirculated effluent in a completely-mixed anaerobic filter. They observed that the effect of a large concentration of toxic metals present in the leachate could possibly be eliminated by the addition of sulphide.

Further work by Chian and DeWalle (1977) looked in more detail at the removal of toxic metals from leachate in an anaerobic filter. They concluded that the percentage removal of iron, zinc, nickel, cadmium, lead and chromium increased with increasing concentrations of metals in the leachate, and also with increasing hydraulic retention time. Metals were precipitated as sulphides, carbonates and hydroxides, and most removal took place in the lower part of the filter.

Using a two-stage anaerobic filter to treat leachate from a solid waste landfill, Wu et al (1988) achieved the COD removal of 91% at an organic loading rate of up to 4 g COD/m^2.day. Almost all of the organic matter reduction occurred in the first stage reactor, which agrees with the observation of other researchers (Young and McCarty, 1969; Mosey, 1978; Witt et al, 1984) who found that the first quarter of the filter depth are responsible for most of the biological activity. This was reflected by the high biogas production in the first stage reactor.

3.6.3 Physical-Chemical Treatment Methods

Apart from aerobic and anaerobic treatment processes, investigations on physical and chemical methods have also been made. Precipitation and coagulation using either lime, ferric chloride or alum have been shown to have little effect on the removal of organic matter. Thornton and Blanc (1973) reported that the methods of leachate treatment with lime or alum was only effective if used as complements to other chemical or biological treatment processes. This is further substantiated by Ho et al (1974) who observed that precipitation has been proved effective in the removal of colour, turbidity, and heavy metals, and is therefore particularly useful in complementing processes capable of removing organic matter effectively, such as biological treatment processes.

Keenan et al (1983) in their study on raw leachate treatment using high calcium hydrated lime as oxidant concluded that temperature and pH have an effect on the concentration of heavy metals in the lime treated effluent, though the response was not identical for all heavy metals. Chromium, copper and mercury produced a U-shaped response to pH, with minimum concentrations occurring at pH 10.2 to 11.2, 10 to 11 and 9.8 to 10.8 respectively. Effluent zinc and iron concentrations increased at lower temperature. Approximately 50% of the organic matter and 75% of suspended solids in the leachate were removed by lime treatment.

Chemical oxidation using hydrogen peroxide, ozone, calcium hypochlorite and potassium permanganate were also tried. Although they were effective in removing colour, turbidity and iron, extremely high dosages are required which proved expensive. Ho et al (1974) and Bjorkman and Mavinic (1977) experimented with ozone as a means of reducing COD concentration in untreated leachate. It was concluded that in order to achieve a reasonable improvement in leachate quality, large amount of ozone (up to 7700 mg/l), with a long period of contact (3 - 4 hr), would be necessary.

For the removal of ammoniacal nitrogen, Maris et al (1984) reported that air stripping has been carried out together with a full-scale experimental activated sludge treatment plant in Pennsylvania, United States of America. From the experimental studies, it was proposed that in order to reduce ammonia-nitrogen below 35 mg/l, a system involving activated sludge treatment preceded by lime addition, clarification and ammonia stripping and followed by chlorination would be required.

Bull et al (1983) concluded from their laboratory investigations that anaerobic treatment process can be effectively employed for simultaneous removal of iron and BOD_5 from sanitary landfill leachate. However, to meet the required discharge consent, further reduction in the organic and nitrogen content of leachate would be necessary. The post-treatment suggested is ammonia-stripping and then followed by aerobic polishing. However, the pH of the leachate must be raised above 10.5 to convert it to gaseous phase, requiring the addition of large quantities of alkali.

Activated carbon has been investigated by Ho et al (1974) to treat leachate, and from batch studies the optimum carbon dosage of 4000 mg/l was suggested for best removal of COD possible. Adsorption equilibrium of COD was achieved in less than 30 minutes. In the laboratory carbon column study, COD and iron removals of 55% and 60% respectively were achieved at 20 minutes detention time.

The advantages of applying physical and chemical methods are that start-up periods are short, simple equipment can be used, processes are generally insensitive to temperature and many of the methods lend themselves to automation. The disadvantages being the operating costs can be high because of excessive chemical dose rates and increased sludge production. For complete treatment, these processes are inadequate as only low organic matter removal has been reported.

When the organic analysis was related to the leachate treatability, it was noted that leachate collected from recently leaching landfills is best treated by aerobic or anaerobic biological treatment processes as they are most effective in removing the free volatile fatty acids that are present in large quantities. Physical-chemical processes are most effective in treating leachate from stabilized landfills or in further removing organic matter in the effluent from biological units treating leachate (Amalendu, 1982).

CHAPTER FOUR

AIMS OF THE STUDY

The importance of landfill management especially with respect to the control of leachate production had been briefly dealt with in the literature review. Landfill leachate when not properly controlled may become an environmental problem, mainly in the area of groundwater pollution and has aroused concern over the need to treat landfill leachate.

Many publications have been produced reviewing a number of treatment methods employed to treat industrial, agricultural and domestic wastewaters. However, little relates to the treatment of leachate. Those limited studies carried out in the treatment of leachate, as described in the literature review, have employed a wide range of treatment techniques. As yet no single technique has been proposed as the most cost effective, and environmentally acceptable means of dealing with the problem posed by landfill leachate. Consequently, data on the treatment alternatives, for comparison purposes, have been difficult to obtain.

A number of options are available including biological and physical-chemical processes. However, the degree of treatment required is often dependent upon the nature and strength of the leachate concerned and the overall objective for the treatment of leachate must reflect the selected disposal outlet (Otieno, 1989). It was therefore felt pertinent to investigate the feasibility of treating leachate using aerobic, anaerobic and physical-chemical processes.

The basic objectives that should be considered in order to achieve optimum design are the maximizing of substrate removal efficiency and the minimizing of operating cost and problems. Generally, these requirements are interrelated and a compromise is necessary as long as the performance of the treatment process is not impaired. During the experimental study, the design factors such as organic loading rate, hydraulic retention time, substrate concentration and other external conditions affecting the process performance should be taken into account.

The main objective of the research was to carry out a feasibility study on the treatment of landfill leachate using aerobic rotating biological contactors (RBCs), upflow anaerobic filters (UAFs) and activated carbon (AC) adsorption columns. An initial investigation was carried out to identify the various parameters which affect the performance of the various reactors. The specific aims were as follows:

i) an investigation into the effects of pH on the performance of the RBC and UAF reactors which was intended to obtain the optimum pH value during steady state conditions,

ii) an investigation into the effects of organic loading rate on the behaviour of the RBC and the UAF reactors which was aimed at providing additional information concerning the stability of the processes under varying loading conditions.

iii) an evaluation of the capabilities of the RBC and UAF reactors as a means of ammonia removal from leachate,

iv) an investigation into the effects of shock loadings on the behaviour of the RBC and the UAF reactors, subjecting the reactors to shock loads, by varying the hydraulic retention times or substrate concentration. Intermittent operation of the reactors are also investigated,

v) an investigation into the toxicity effect of heavy metals on the performance of the RBC and the UAF reactors. This is carried out by studying the effect of step addition of zinc on the performance of the reactors, and

vi) an investigation into the performances of an activated carbon adsorption at varying pH and organic loading conditions.

CHAPTER FIVE

EXPERIMENTAL EQUIPMENTS AND PROCEDURES

The characteristic of leachate varies from one landfill site to another which depended on the design and operation of the landfills, type of solid waste deposited and climatic conditions. The age of the landfill also contributed to the variation of leachate composition. From the literature review, it was reported that leachate from younger landfill had a very high organic concentrations as compared to those of the older landfill, of which 60 to 80% of the total COD in young landfill leachate are made up of volatile fatty acids. The composition of leachate from older landfill have a much higher molecular weight compounds.

In this study the raw leachate samples used were collected from two landfill sites with different compositions and of different ages. A comparative study on the performance of the biological and physical-chemical processes used to treat the leachate was made. The wider spectrum of leachate strength and characteristic obtained, could be used during the treatability studies which allow assessment of the best possible treatment option for the leachate treated (Gaudy et al, 1986).

The leachate used during the initial phase of experimental study was obtained from a landfill site at Old Fisher Lane in Blyth Valley District. The site was established in 1974 and was in operation for nearly 16 years before it was closed down nearly 2 years ago. Therefore, the site could be considered as an "ageing" landfill site (Henry et al, 1987) characterized by the partially stabilized leachate, as listed in Table 5.1. Apart from dead

Table 5.1: Average Compositions of Leachate from Old Fisher Lane and Birtley Landfill Sites

(Results, except for pH, are expressed in mg/l)

Constituent	Old Fisher Lane	Birtley
pH value	6.9	5.9
COD	2610	32400
BOD_5	834	19200
TOC	621	9430
Total Solids	8030	31620
Volatile Total Solids	940	20980
Suspended Solids	372	585
Volatile Suspended Solids	163	341
Volatile Fatty Acids	942	11830
Ammonia-Nitrogen	65	555
Organic-Nitrogen	28	462
Alkalinity	1525	3250
Sodium	774	2485
Calcium	512	1720
Potassium	438	574
Magnesium	127	280
Iron	3.8	356
Zinc	0.7	83
Chromium	0.3	0.2
Cadmium	0.1	0.6
Nickel	0.4	4.0
Manganese	2.2	46.5
Orthophosphate	0.6	2.5

branches, household furniture, items of clothing and footwear, the landfill consisted mainly of the residual incinerator of domestic solid waste. The leachate samples were collected from a trench downstream from the site.

During the later stages of the study the leachate used was collected from a landfill site at Birtley, Gateshead District. Historically, the site accepted almost every type of solid wastes (i.e solid industrial wastes and MSW) including pharmaceutical wastewater, but with restriction imposed on the disposal of hazardous waste, the site currently accepts interceptor wastes (i.e oily waste from garages), commercial wastes, old batteries and MSW. Although the landfill has been in operation for almost twenty years, the leachate produced are still quite high. This is probably due to the leachate being contained in the landfill through recirculation as well as the continuous disposal of wastes on to the site. Hence, the landfill site could be considered in its "maturing" stage with partly degraded leachate as listed in Table 5.1. The leachate samples were extracted from boreholes situated at the base of the landfill.

The experimental study involved the use of several bench-scale laboratory systems with an influent of low strength (LS) and diluted high strength (HS) leachate. The bench-scale reactors were installed at the laboratories of the Environmental Engineering Division of the Department of Civil Engineering, University of Newcastle upon Tyne. The materials and procedure used will be discussed in detail later. The laboratory analyses were carried out in accordance with Standard Methods for the Examination of Water and Wastewater, Sixteenth Edition (APHA, 1985). The analytical procedures used in monitoring the performance of the reactors are described later in this chapter.

5.1 Description of the Laboratory-Scale Experiments

The laboratory-scale reactors used consisted of two RBCs, two UAFs and two AC adsorption columns. The reactors arrangement are illustrated in Plate 5.1.

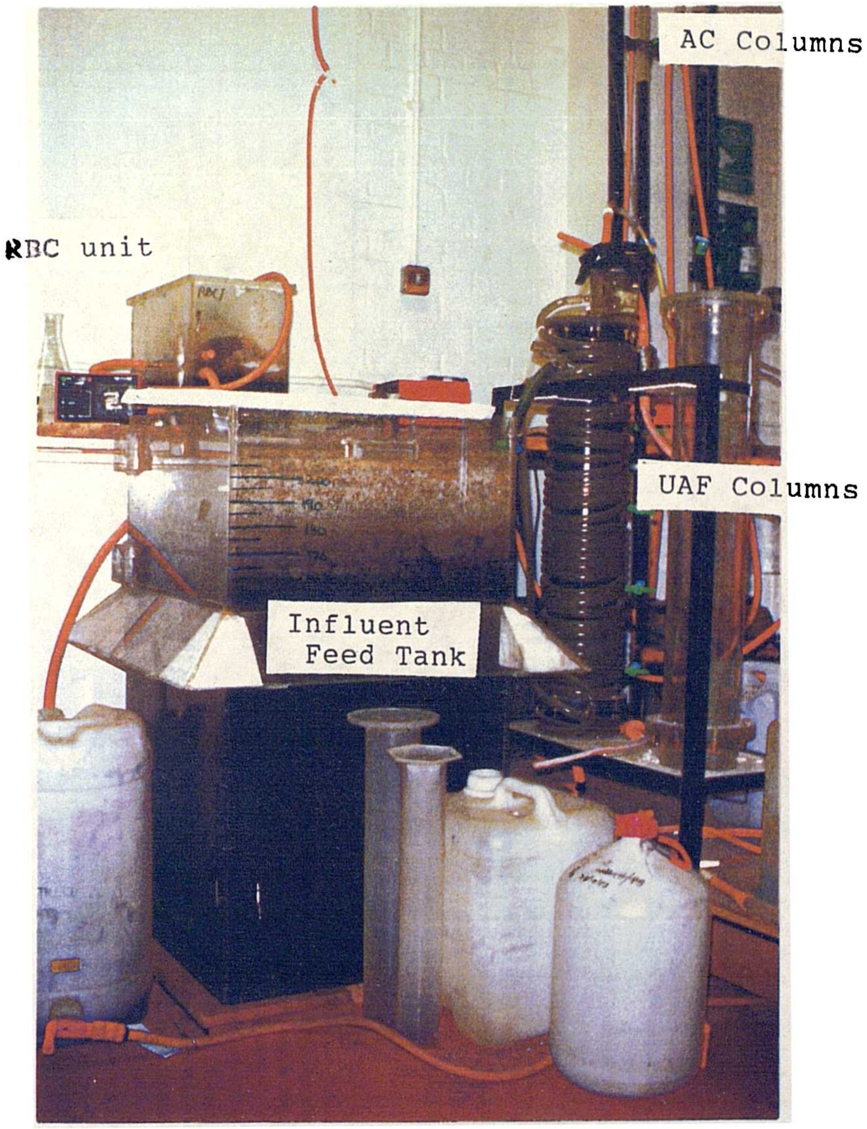


Plate 5.1: The Overall Systems Arrangement

5.1.1 Rotating Biological Contactors

Two existing aerobic RBCs from a previous project in the form of a packed drum were used in this study, without any modifications. The wall of each RBC unit was constructed from perspex and of a rectangular shape. The RBC consisted of four packed wire mesh drums mounted on a horizontal shaft which ran along the entire length of the tank. The shaft revolved at a constant speed of 2 rpm with the drums being immersed in the liquor to approximately 40 percent of their surface area.

Each drum was 150 mm in diameter and 60 mm long. The drums were filled with "Plastic Bioring 25" media supplied by Norton G.B and Company, of Akron, Ohio, the United States of America. The media have a specific area of 200 m^2/m^3 having a fully packed drum volume of 1.06×10^{-3} m^3 and a total surface area for each 4-stage unit of 0.212 m^2. Each RBC unit, which ran at ambient temperature, was divided into 4 equal compartments or stages as shown in Plate 5.2 and Fig. 5.1.

The influent was fed from a storage container into the RBCs using peristaltic pumps.

5.1.2 Upflow Anaerobic Filters

The UAFs used were constructed of perspex columns with an internal diameter of 100 mm and a height of 0.7 m. Each filter column was filled with random packed plastic media similar to the media used in the RBC unit. The column was kept at a constant temperature of approximately 36°C by an external hot water coil which was controlled by a Grant water

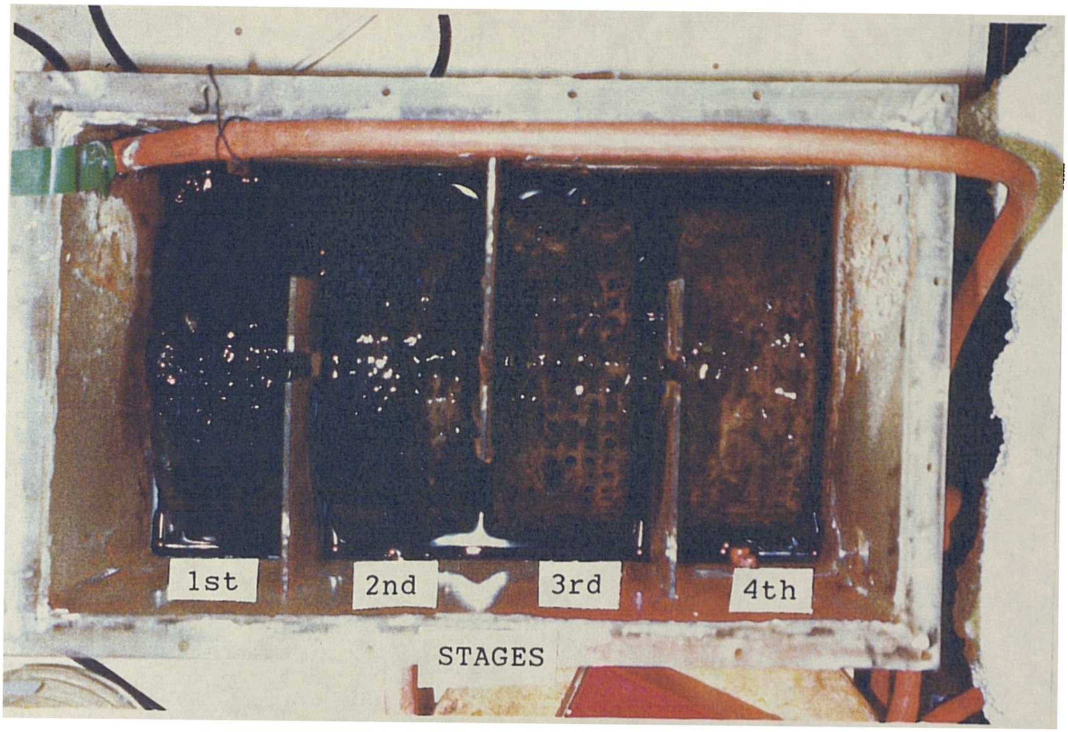


Plate 5.2: The RBC Showing Four Packed Drums with Attached Biomass

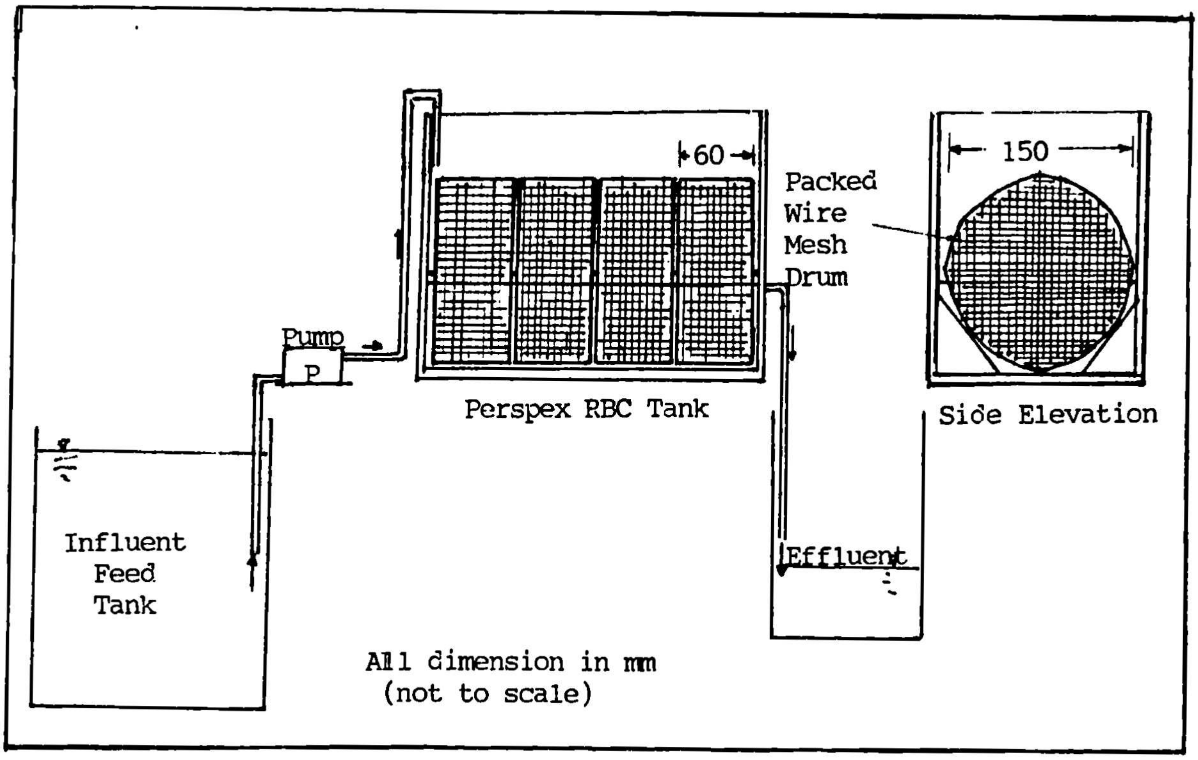


Figure 5.1: The Four-Stage Rotating Biological Contactor

heater. A thermometer was inserted into the top of the column to monitor the temperature. The UAFs with all the necessary equipment are shown in Plate 5.3 and Fig. 5.2.

The total empty bed volume of each filter column was 6.7 litres with a void volume of 6.3 litres. An influent feed port was located at the bottom of each filter and the influent continuously fed from a storage container using variable speed peristaltic pumps. The influent flow was distributed uniformly by a distribution plate placed above the influent port.

Initially a water displacement method (using a 20 litre aspirator) incorporating a water trap was used for biogas measurement. It was later replaced by a wet-test gas meter positioned after a Dreshell bottle. Gas samples were taken from the self-sealing serum cap at the top of the columns.

5.1.3 Carbon Adsorption Columns

The granular activated carbon (GAC) for the adsorption reactor was supported by fibreglass wool and glass marbles placed in a perspex column. Two perspex columns of 1.6 m in height and 25 mm internal diameter were used, as illustrated in Plate 5.4 and Fig. 5.3. The physical properties of the carbon are given in Table 5.2. Before each adsorption run, distilled water was passed through the carbon beds to rinse out any extraneous carbonaceous materials from the carbon so that they would not interfere with subsequent measurements of the carbon content in the effluent.

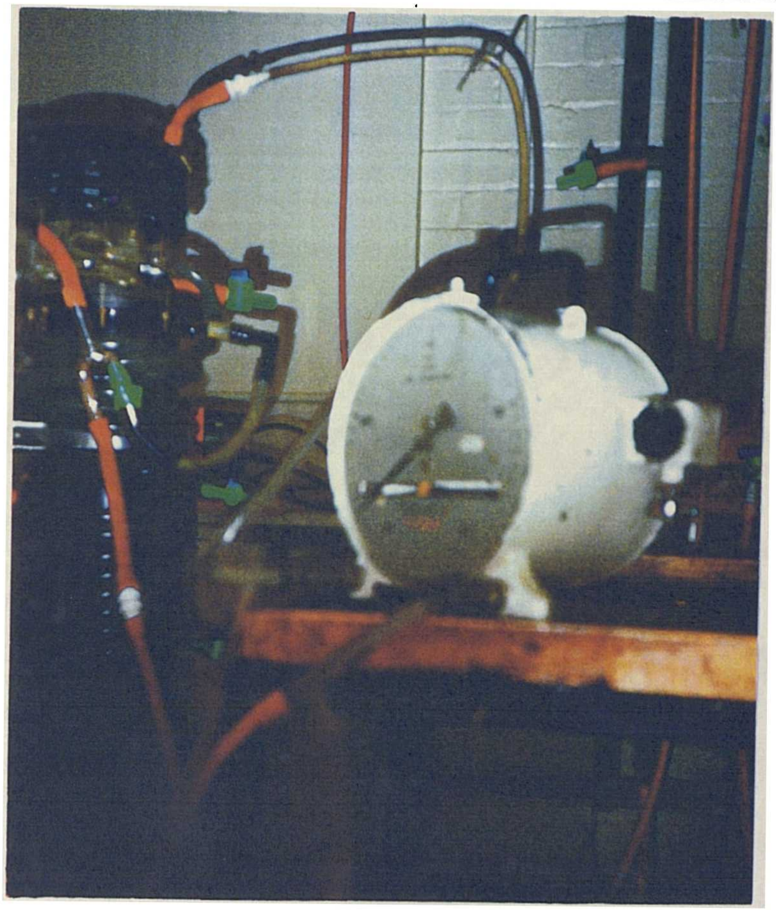


Plate 5.3: The Upflow Anaerobic Filter with Wet-Test Gas Meter

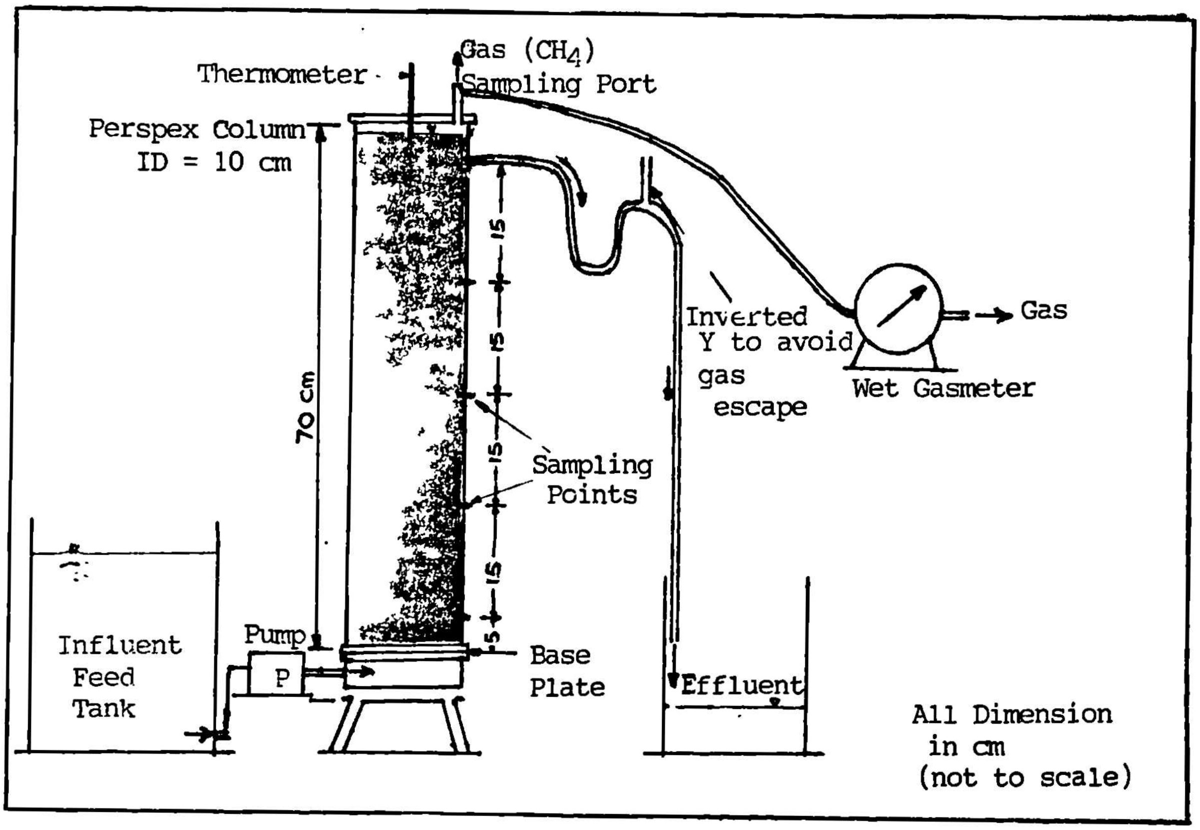


Figure 5.2: Laboratory-Scale UAF Set Up

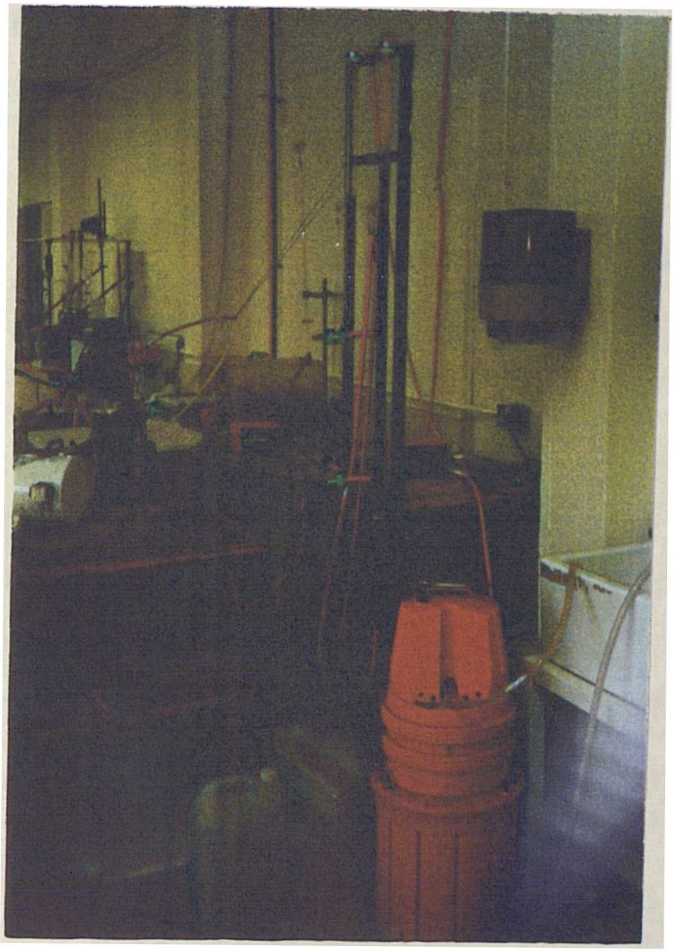


Plate 5.4: The AC Columns and the Effluent Auto Sampler

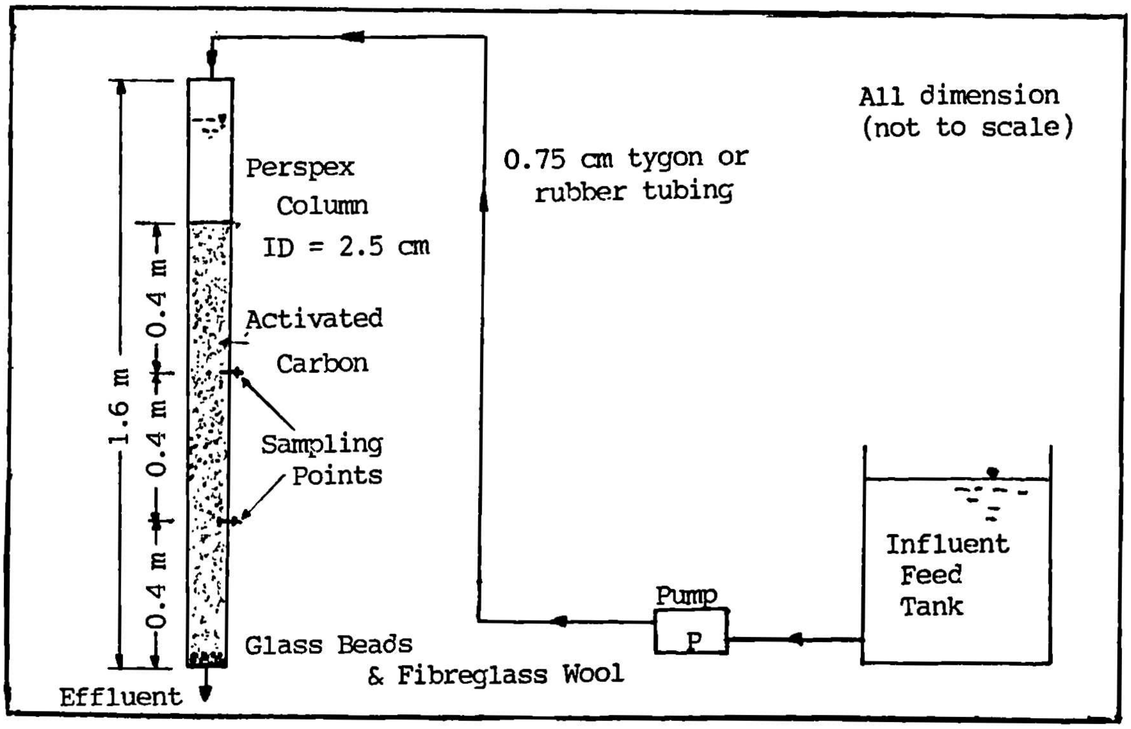


Figure 5.3: Laboratory-Scale Activated Carbon Column Set Up

Table 5.2: The Physical Properties of GAC Used

Type of Carbon	Charcoal Granular
Particle Size (mm)	0.85 - 1.70
Mesh Size	10 - 18
Loss on Drying at 130°C	Not more than 12%

Two sampling ports were spaced equally over the height of the column at 0.4 m intervals. The influent was continuously fed from the top using a variable speed peristaltic pump.

5.2 Laboratory Analyses

During the earlier stages, the influent and effluent from the RBC units and the UAF columns were sampled daily. The samples were analyzed for pH, alkalinity, COD, BOD_5, TOC and various other relevant parameters. The analytical procedures used in this study are summarized in Appendix A. All analyses were carried out in accordance with Standard Methods (APHA, 1985), with the procedures summarized as follow:

5.2.1 Chemical Oxygen Demand (COD)

Chemical Oxygen Demand (COD) is a measure of the oxygen equivalent of the organic matter content of a sample that is susceptible to oxidation by a strong chemical oxidant. The COD test gives a meaningful method for the determination of the organic contents in the influent, effluent and in other samples taken from the sampling ports, because of the rapid nature of the chemical oxidation process. The analyses were carried out at least three times a week. The closed reflux method for COD analysis was adopted as described in the Laboratory Manual.

For influent samples, the total COD was determined, while the effluent samples were analyzed for both filtered and total COD with GF/A filter papers being used to filter the samples.

5.2.2 Five Day Biochemical Oxygen Demand (BOD_5)

The five day Biochemical Oxygen Demand (BOD_5) is a measure of the amount of biochemically degradable organic matter in a sample of waste. BOD_5 was determined using the dilution technique described in Standard Methods (APHA, 1985). The BOD_5 determination was carried out at least three times a week.

5.2.3 Total Organic Carbon (TOC)

The Total Organic Carbon (TOC) test is a rapid test procedure. The concentrations of dissolved organic carbon and inorganic carbon were measured using a Beckman 915B TOC Analyzer. For total carbon, 30 micro-litres of sample was injected into a tube containing a catalyst and maintained at 950°C. The carbon dioxide produced by the oxidation of any carbonaceous material present in the sample was detected by an infra-red analyzer.

Inorganic carbon, such as carbonate was measured separately using an acid catalyst at 150°C. The value of TOC was obtained by subtracting the value for inorganic carbon from the total carbon.

5.2.4 Volatile Fatty Acid (VFA) Determination

The method used was that adopted in a technical report (TR 76, February 1976) presented by the Water Research Centre. Full information is given in the manual on automated-gas-chromatographic procedures for the determination of VFAs. Usually 10 mls of sample were treated with 1 ml of low concentration formic acid and then transferred to small tubes. The VFAs present in the sample were then determined using the gas liquid chromatography.

5.2.5 pH values and Alkalinity

These are very important parameters of the wastewater. The pH values of the influent and effluent samples were measured daily using an external pH meter while the alkalinity of the influent and effluent sample were measured daily using a volumetric method.

5.2.6 Suspended and Volatile Suspended Solids (SS and VSS)

The procedures used for analyzing both SS and VSS were adopted from Standard Methods (APHA, 1985). Using a pre-dried and weighed GF/A filter paper, 50 ml from each influent and effluent sample were filtered. The filtered samples were dried in a 105°C oven for the SS determination and for the VSS determination, the dried filtered samples were placed in a 550°C furnace. The SS and VSS analyses were carried out at least twice a week.

The SS and VSS of the sludge samples periodically taken from all stages and sampling ports of the RBC and the UAF reactors respectively were also determined for profile studies.

5.2.7 Gas Analysis

The gas composition from the UAFs was determined using a Becker Model 403 thermal conductivity detector chromatography using Poropak Q packing and Helium as the carrier gas. Gas samples of about 0.9 ml were obtained from the serum cap of each filter and injected through the injection port. The gas compositions were reported as a percentage of methane, carbon dioxide and nitrogen.

5.2.8 Nitrogen

Ammoniacal-nitrogen (NH_3-N) and Total Kjeldahl Nitrogen (TKN) of the influent and effluent samples of the RBC and the UAF were determined at least twice a week using the distillation-titration technique as described by Standard Methods (APHA, 1985).

Nitrite-nitrogen (NO_2-N) and nitrate-nitrogen (NO_3-N) in the influent and effluent samples were also periodically determined. The NO_2-N determination was carried out using a portion of the filtered samples diluted to a final volume of 40 ml. 2 ml Griess-Ilosvay's Reagent I and 5 ml of Griess-Ilosvay's Reagent II were then added to the samples. The absorbances of the samples were measured and compared with the reagent blank at 525 nm using a spectrophotometer.

The NO_3-N was determined by the Modified Brucine Method developed by Jenkins and Medsker (1964). Absorbances of the samples against reagent blank at 410 nm were determined using a spectrophotometer.

5.2.9 Phosphate

To ensure that the phosphorus requirement in the influent was sufficient, phosphorus was periodically determined using the Ascorbic Acid Method in accordance with Standard Methods (APHA, 1985)

5.2.10 Metals

Heavy metals in the influent and effluent samples were determined using an atomic absorption spectrophotometer (AAS). Soluble metal analysis were carried out with samples pretreated according to the methods described in Standard Methods (APHA, 1985).

CHAPTER SIX

START-UP AND STEADY STATE OPERATIONS OF ROTATING BIOLOGICAL CONTACTOR AND UPFLOW ANAEROBIC FILTER

6.1 Introduction

The operations of aerobic and anaerobic reactors can be divided into two stages namely start-up and steady state. Both stages are affected in a different way by variations in loading and operating conditions.

The success or failure of any treatment process lies within the initial period of operation, that is during the start-up. Start-up provides a stabilization phase in which acclimatization and adaptation of the microorganisms in the seed sludge to the wastewater can be achieved, thus providing a suitable biofilm development and hence ensuring an acceptable effluent quality. Careful control of start-up operation is necessary.

Start-up operation varies from one treatment process to another, especially more so between aerobic and anaerobic systems. Generally the start-up of anaerobic reactors is more time-consuming than for aerobic reactors. The generation of sufficient appropriate microbial culture for particular wastes is often posed as a vital obstacle. The main reasons are the slow growth rates of the methanogenic bacteria and the long acclimatization of the microorganisms to new types of wastes. Start-up of anaerobic reactors is also affected by external and internal disturbances. In order to achieve stable operating conditions a good biomass growth must be maintained in the reactors.

Anaerobic biomass, as referred to above, grows much more slowly than does aerobic biomass due to the lower net energy yield of anaerobic degradation, which is about one-tenth of that in aerobic degradation (Salkinoja-Salonen et al, 1983). Hence seeding is very much more important in anaerobic reactor start-up when compared to an aerobic process. Moreover, initial loading during start-up must be kept low, examples of as low as 0.05 to 0.1 kg COD/kg VSS.day are quoted (de Zeeuw and Lettinga, 1980; Hulshoff Pol et al, 1983). An increase in loading must be conducted gradually and carefully. Henze and Harremoes (1983) reported that the start-up period was usually greater than one month.

The aerobic reactor of concern in this study is an RBC. Information on start-up of RBC units is very limited with most references encountered not giving a comprehensive review on RBC start-up. Fry et al (1984) stated that the literature available on start-up characteristics of RBC units are brief, incomplete and incidental in nature because the research was not focused on start-up. The scattered information mostly centred on the establishment of observable biofilm attachment onto the disc or drum.

During start-up of an RBC, biomass growth on the disc or drum is established. According to Antonie (1976), the thickness of the attached biofilm generally ranges from 2 to 4 mm, one week after start-up. In another study by Bracewell et al (1980), an observable biomass in an RBC treating phenol-formaldehyde resin wastewater was established approximately two weeks from the beginning of operation. Hence based on the preceding studies it is apparent that a measurable or observable biofilm will result 1 or 2 weeks after start-up begins. The characteristics of the biomass in turn changed from one stage to another in the RBC.

Further necessary information relating to the start-up operation in an RBC is the time required to attain steady-state conditions. Trinh (1981) reported the acclimatization of an RBC unit within two weeks (in terms of BOD_5 removal) with a loading rate of 7.3 kg $BOD_5/100\ m^2$.day. Based on this information it appears that approximately two to three weeks are required to reach steady state conditions in terms of BOD_5 or COD removal.

The start-up in an anaerobic process has always been considered the most unstable and difficult phase and the rate of start-up in anaerobic reactors is depended on the type of inoculum, the type and strength of wastewater, level of volatile fatty acids maintained and the characteristics of the support material used (Vigneswaran et al, 1986). Reducing the time of start-up is one of the keys to greater competitiveness of anaerobic digestion (Camilleri, 1988). In general, the start-up of anaerobic reactors proceeds easily and quickly when the seed sludge is more adapted to the composition of the waste.

Using continuous stirred tank reactors and anaerobic fluidized bed reactors and arranged as a single- and two-stage systems to treat synthetic meat waste, Stephenson and Lester (1986) achieved a rapid start-up within 50 days in all the systems by following a procedure involving stepped increases in organic loading rates in addition to a methanol substitution and a trace element supplementation for the encouragement of the methanogenic bacterial growth.

After seeding five bench-scale UAFs with screened anaerobic sludge and acclimatizing the reactors with the wastewater at 2 to 3 days hydraulic retention time for two weeks on a full recycling mode, Capobianco and

Blanç (1989) achieved a rapid start-up of about 25 days in all the reactors, using a soft drink syrup and bottling wastewater with an influent strength of 1200 mg/l and at HRTs ranging from 8 to 52 hours.

Young and McCarty (1967) noted that microorganisms in an unseeded or lightly seeded filter remained dispersed and a significant fraction was washed out with the filter effluent, whereas in a highly seeded filter, rapid flocculation was observed at the filter base, causing the biomass to remain in the filter. Suspended growth of biomass leads to wash out of the microorganisms which consequently disrupted the reactor operation (Salkinoja-Salonen et al, 1983). Raman and Khan (1977) in their study of sewage treatment using a UAF reported that, without seeding, four to six weeks of continuous operation at a temperature of between 25°C and 32°C were required for start-up, and three months were required before the filter became fully mature.

The start-up of an upflow reactor using an unacclimatized seed sludge, such as digested sewage sludge, can take from two months to more than half a year, depending on the quality of the seed sludge and the wastewater characteristics (de Zeeuw and Lettinga, 1983). Clearly, it seems apparent that the duration of acclimatization fluctuates and that even stabilization of the measured chemical parameters of the sludge may not be a true indication of the stabilization of the microorganism's activity.

Donnelly (1984) stated that a good acclimatized sludge can be successfully achieved by seeding with a high concentration of suitable bacteria followed by the gradual introduction to the new substrate at low loadings accompanied by good mixing. Minimizing biomass washout as well as controlling the pH through buffer addition and/or feeding interruption are also vital during start-up operation.

Kennedy and Droste (1991) stated that the start-up of anaerobic reactors is affected by four factors:

i) the quality of the seed sludge,

ii) the rate of acclimatization of methanogenic bacteria to the waste,

iii) the rate of growth of anaerobic microorganisms, and

iv) the rate of loss of anaerobic microorganisms.

For a fixed-film reactor, the attachment and growth of microorganisms onto the media is the vital element (van den Berg and Kennedy, 1982). Start-up will improve if the higher rate of attachment is achieved. The start-up time of a UAF is directly proportional to the concentration of the microbial population. The start-up time for UAFs has ranged from 10 to 180 days with the shorter times corresponding to the use of large amounts of an active seed while longer times were associated with the use of light seeding (Young and McCarty, 1967). Thus it can be seen that heavy seeding of a UAF using digested sludge solids would be preferable to a light seed for a rapid filter start-up.

The effect of pH on RBC and UAF performance has also been studied, since it is one of the major factors influencing the performance of the reactors. The role of pH on the performance of an RBC and a UAF has been briefly discussed in Chapter 3. In this study, preliminary runs were undertaken to establish the range of influent pH for optimum performance of both the reactors.

Jennett and Rand (1980) stated that, as a rule of thumb, steady state condition in a UAF implies that for a constant influent strength and loading rate, the effluent substrate concentration and all other operational parameters remain constant for an indefinite period of time. However, the nature of the UAF with its dynamic biological solids concentration and the continual state of fluctuation within the system, means that true steady state conditions probably never existed (Young, 1968). For this study, steady state conditions were assumed to exist when stable substrate effluent concentration was achieved, accompanied by a stable gas production rate.

Organic loading rate (OLR) has been recognized as the major factor affecting treatment efficiency of both the aerobic and the anaerobic processes. The organic loading applied to a reactor is a function of both the influent waste strength and the hydraulic retention time (Young, 1991). However, conflicting ideas are afforded by numerous researchers on the relative importance of these two parameters on reactor performance (Gillespie et al, 1974; Clark et al, 1978; Dewalle et al, 1980; Kobayashi et al, 1983; Surampalli and Baumann, 1986; Young and Yang, 1989).

Gillespie et al (1974), treating pulp and paper mill waste using an RBC, suggested that the hydraulic loading rate had the most profound effect on performance. Norton (1984) as well as Surampalli and Baumann (1986) observed that removal rates and process efficiency of the RBC were indeed dependent on the areal organic loading rate (AOLR) rather than wastewater concentration or flow rate individually.

Dewalle et al (1980) reported the influent BOD_5 concentration to be the most important predictor variable in the percentage BOD_5 removals of a UAF treating domestic sewage. On the contrary, Young and Yang (1989) found that the HRT was the most significant operating parameter. Thus from the above, it is apparent that both the influent wastewater substrate concentration and the hydraulic loading affect the removal efficiencies achieved. Therefore, the variability of both these parameters must be considered when assessing the reactor performance, especially during steady state operation.

Treatment efficiency can be expressed in various forms, such as areal organic removal rate (AORR), volumetric organic removal rate (VORR) and substrate removal (% as COD or BOD_5). Each is suitable for different purposes, for example both AORR and VORR are preferred as design criteria (Henze and Harremoes, 1983) while substrate removal is the simplest parameter for the comparison of the efficiency of various reactors within the same range of OLR. In most literature, VORR (kg $COD/m^3.d$ or kg $BOD_5/m^3.d$) is generally used for all types of reactors including UAF reactors, whereas AORR (g $COD/m^2.d$ or g $BOD_5/m^2.d$) is only relevant to fixed film reactors.

This is especially true for the study using RBC reactors, where most researchers tend to use AORR (g $BOD_5/m^2.d$) over VORR (kg $COD/m^3.d$) in their performance analysis. In this study, with the exception of substrate utilization kinetic analysis using AOLR and AORR in the determination of the kinetic constants (for comparison between the aerobic RBC and the anaerobic UAF), treatment efficiency will be presented using the parameter VORR ($kg/m^3.d$) and substrate removal (%), with AORR ($g/m^2.d$) being inserted in parentheses.

6.2 Experimental Programme

The experimental work was divided into several phases. In this chapter three phases were investigated that is:

i) the start-up,

ii) the determination of optimum pH range, and

iii) the quasi steady state operations.

6.2.1 Phase I: Start-Up Operation

Based on the information gathered from previous literature reviews, start-up of the RBC units was delayed for approximately 40 days from the start-up of UAFs. This step was taken in order to allow for acclimatization and adaptation of anaerobic bacteria in the filter to the substrate introduced. Also, by following this procedure ample time could be allocated before both the RBC and the UAF received the same wastewater (leachate) coming from the same feed tank.

- Start-up of RBC

During start-up, the RBC units were seeded using return activated sludge from Cramlington Wastewater Treatment Plant. The reactors were initially batch fed with 1 to 2 kg whey powder/m^3 reactor, to assist the biofilm formation on the media. After 4 days of batch feeding, the reactors were fed continuously using whey solution with an average COD concentration of 1210 mg/l at a mean VOLR of 3.7 kg COD/m^3.day (mean AOLR of 19.2 g COD/m^2.day) and an average HRT of 8 hours.

Initial influent and effluent characteristics during the start-up of the RBC units were determined three times a week. After about two weeks the whey solution was replaced by a landfill leachate obtained from Old Fisher Lane. The reactors were then monitored until a quasi steady state condition was achieved.

In order to maximize substrate conversion rates, it is pertinent to have a sufficient quantity of nutrients in the wastewater. The nitrogen content in the leachate was low, while phosphate was very much deficient. In order to rectify the situation, sufficient nutrients were added to the leachate sample to ensure that the carbonaceous content of the feed was the only growth limiting substrate. The nutrients added were in the form of potassium diphosphate (KH_2PO_4) and ammonium bicarbonate (NH_4HCO_3) which yielded a COD:N:P ratio within the range of 200:4.5:0.8 to 200:8:1.5. Trace metal solution was also added to the leachate sample. The composition of the trace metal solution is listed in Appendix B.

- Start-Up of UAF

The seed sludge used for the start-up of the UAFs was collected from the anaerobic digesters of the Durham Wastewater Treatment Plant treating basically domestic wastewaters. Shapiro and Switzenbaum (1984) reported that to achieve a good initial biofilm attachment onto supporting media a high density of biological solids should be used. The sludge obtained from the anaerobic digesters was very thin. In order to produce seed sludge with higher density, the sludge was first screened and thickened before it was used as inoculum for the start-up of the process.

The seed sludge was first sieved three times through 3 mm mesh and then thickened. The solids characteristics of the thickened seed sludge were:

- total solids: 28.2 g/l
- volatile solids: 16.5 g/l
- percentage of volatile solids: 58.5%

Initially, approximately 4 litres of seed sludge was placed in each UAF reactor, thus providing 66 grams of volatile solids. The reactors were then purged with oxygen-free nitrogen in order to remove the air entrapped in the seed sludge. Nitrogen was left to flow at a pressure of 35 kPa for 20 minutes. The heating system was then switched on in order to warm up the sludge from room temperature to the operating temperature of 36°C. The reactors were operated by introducing synthetic feed through the inlet systems. The average COD concentration of the diluted synthetic feed used was 995 mg/l. The compositions of the synthetic feed are listed in Table 6.1.

Table 6.1 Compositions of Synthetic Feed

Ingredients	Concentration
Glucose	8.0 g/l
Bacteria Peptone	4.8 g/l
Lab Lemco	3.2 g/l
$NaHCO_3$	0.8 g/l
KH_2PO_4	0.8 g/l
NH_4HCO_3	0.8 g/l
Trace Metal Solution	1 ml/l

The VOLR of the synthetic feed was maintained at an average of 1.25 kg COD/m^3.day with an average HRT of 20 hours. The low loading rate was selected in order to avoid overloading during the acclimatization period. This assists in the build up of the microbial population required for complete breakdown of the applied substrate, whilst at the same time purging the inert solid matter from the system. Acclimatization was considered to be completed when the effluent COD had levelled off and decreasing volatile fatty acid concentration was achieved.

After about four weeks of feeding with synthetic wastewater, the substrate was gradually replaced by a portion of low strength (LS) leachate from Old Fisher Lane. The gradual introduction of the leachate was necessary to allow for an adaptation period of the bacteria. The initial ratio of synthetic wastewater to leachate was 5:1. The proportion of leachate in the substrate was increased step by step, and by the end of day 46, the UAF units received an influent feed comprising totally of leachate. Around this period of time, the leachate used was also introduced as feed for the RBC units.

6.2.2 Phase II: Determination of Optimum pH Ranges

After a steady state condition had been achieved, one of the RBC units and one of the UAF units were subjected to the first volumetric shock loading, and the results will be discussed in another chapter. In order to determine the pH range for optimum performance of the RBC and the UAF units, the pH value of the influent feed was initially decreased gradually by adding sulphuric acid (H_2SO_4). Then the pH value was gradually increased by adding sodium bicarbonate ($NaHCO_3$).

6.2.3 Phase III - Quasi Steady State Operation

Following the experiments of reactor start-up and pH investigation, quasi steady-state operations of the RBC and the UAF were carried out over a range of various loading rates. Two different leachate strengths were used in this Phase III study. The low strength leachate (LS leachate) used was obtained from Old Fisher Lane landfill site while the high strength leachate (HS leachate) came from Birtley landfill site.

The reactors were initially operated with the LS leachate. The performances of the reactors at four different loading rates were studied by changing the flow rates. After quasi steady state operation for each loading rate had been achieved, the reactors were allowed to run for a further few weeks to determine the substrate removal efficiencies of both reactors for each loading rate. A simple regression study on each reactor was determined to describe the reactor performance.

After the completion of LS steady state (LS S-S) performances study, one each of the RBC and UAF units were subjected to a first organic shock loading. A few days after the shock analysis, all reactors were shut down for about three weeks. The units were soon restarted and analysis were carried out on the performances after shut down. The results for all these investigations will be discussed as Phase IV study in Chapter 7.

Soon after the new start-up and a second volumetric shock loading, investigations on the quasi steady state performances of the RBC and UAF units using HS leachate were undertaken. The leachate, which had an average COD of 32400 mg/l was diluted to the required strength (an average COD of 6250 mg/l) before being fed to the reactors. The nutrients and trace metal requirements in the leachate sample were also corrected.

6.3 Results and Discussion

The aim of the Phase I study was to determine the start-up characteristics of the RBC and the UAF while in Phase II, investigations were conducted to examine the response of the reactors to pH variations. During these phases of study the operational parameters including the influent COD, flow rate and nutrient addition were monitored to maintain the required organic loading rate. The data analyzed from both samples were plotted to illustrate the trends revealed.

The operational conditions and influent characteristics during start-up of the RBC and the UAF are summarized in Tables 6.2 and 6.3 respectively.

6.3.1 Response to Start-Up Operation

For the first three days of operation, biological growth was not detected on the rotating cages of the RBC units. However from the results obtained, a BOD_5 removal of 48.5% (COD removal of 51.0%) was achieved after two days operation. This removals were achieved with the help of the aeration, flocculation and sedimentation processes brought about by the rotating cages. After about 5 days operation a slight growth of biomass was noticeable, which formed a transparent layer surrounding the rotating surfaces. Gradually, approximately ten days of operation, the biomass in the first two stages become thickèr and a greyish-brown in colour. The

Table 6.2 Operational Conditions in RBC during Start-Up

Parameters	Whey Powder		LS Leachate	
	Range	Mean	Range	Mean
Time of Operations	1 - 13	(14 days)	14 - 44	(31 days)
Infl. Total COD (mg/l)	910 - 1560	1210	950 - 1600	1240
Infl. Total BOD_5 (mg/l)	400 - 775	635	304 - 735	485
Infl. TOC (mg/l)	245 - 390	325	185 - 520	330
Infl. pH	5.4 - 5.5	5.45	6.4 - 7.9	7.10
Infl. TKN (mg/l)		80	65 - 100	80
Infl. NH_3-N (mg/l)		35	25 - 60	45
Infl. TSS (mg/l)			270 - 640	390
Infl. VSS (mg/l)			215 - 510	325
Org.L.R. (kg COD/m^3.d)	2.3 - 5.0	3.7	3.8 - 6.3	5.0
Org.L.R. (kg BOD_5/m^3.d)	1.3 - 2.5	1.9	1.2 - 2.8	2.0
Org. L.R. (kg TOC/m^3.d)	0.8 - 1.3	1.0	0.9 - 1.9	1.4
Areal L.R. (g COD/m^2.d)	12.0 - 25.8	19.2	14.7 - 28.1	21.0
Areal L.R. (g BOD_5/m^2.d)	6.6 - 12.8	10.0	4.5 - 12.4	8.2
Areal L.R. (g TOC/m^2.d)	4.0 - 6.5	5.1	3.4 - 8.5	5.6
HRT (hrs)	7.5 - 8.5	8.0	4.5 - 7.5	6.0

Table 6.3: Operational Conditions in UAF during Start-Up

Parameters	Synth. Feed (SF)		SF + LS Leachate		LS Leachate	
	Range	Mean	Range	Mean	Range	Mean
Duration	1 - 26 (27 days)		27 - 45 (19 days)		47 - 86 (40 days)	
Infl. Total COD (mg/l)	825 - 1150	995	810 - 1400	1040	950 - 1600	1255
Infl. Total BOD_5 (mg/l)	300 - 445	385	400 - 675	545	305 - 785	520
Infl. TOC (mg/l)	190 - 275	225	175 - 315	220	185 - 520	330
Infl. pH	5.4 - 6.0	5.7	5.6 - 6.3	6.0	5.6 - 7.9	6.9
Infl. TKN (mg/l)			80 - 95	90	65 - 100	75
Infl. NH_3-N (mg/l)			35 - 50	40	25 - 60	45
Infl. TSS (mg/l)	370 - 570	470	415 - 630	495	270 - 655	420
Infl. VSS (mg/l)	300 - 490	395	330 - 520	400	220 - 510	340
Org.L.R. (kg COD/m^3.d)	0.9 - 1.5	1.25	1.1 - 1.5	1.30	1.0 - 1.6	1.20
Org. L.R. (kg BOD_5/m^3.d)	0.3 - 0.6	0.50	0.6 - 0.8	0.70	0.3 - 0.7	0.50
Org. L.R. (kg TOC/m^3.d)	0.2 - 0.4	0.30	0.2 - 0.3	0.25	0.2 - 0.5	0.35
Areal L.R. (g COD/m^3.d)	5.5 - 8.8	7.10	6.5 - 8.5	7.40	5.6 - 9.2	7.00
Areal L.R. (g BOD_5/m^3.d)	1.9 - 3.6	2.75	3.2 - 4.8	3.90	1.7 - 3.9	2.90
Areal L.R. (g TOC/m^3.d)	1.1 - 1.9	1.60	1.3 - 1.9	1.55	1.2 - 2.7	1.80
HRT (hrs)	16.6 - 24.4	20.0	16.8 - 27.5	19.5	20.0 - 30.2	25.0

surfaces of the four cages were almost covered with the biofilm after two weeks operation with the last two stages acquiring a reddish brown appearance, while the first two stages became filamentous.

Some sloughing of the biomass was apparent but the treatment efficiency was not significantly affected since the BOD_5 removal at this time was found to be 88.5% (COD removal of 77.0%). From the results, it can be said that the anticipated level of BOD_5 removal has been achieved. The average effluent BOD_5 concentration after two weeks operation was 78 mg/l. In order for nitrification to occur, Antonie (1976) stated that effluent BOD concentration should be below 20 mg/l. The beginnings of nitrification can be determined by the presence of nitrite ion (NO_2^-), in the effluent. When nitrite disappears and nitrate ion (NO_3^-) appears complete, nitrification is then underway. From the results obtained, there was no evidence of nitrification occurring at this stage of operation as no noticeable nitrite- and nitrate-nitrogen in the effluent, while the ammonia-nitrogen (NH_3-N) concentration was seen to increase. The reason being that at this early stage, most of the organic matters (and organic nitrogen) were utilized in the substrate removal.

Upon the introduction of LS leachate a sudden decrease in COD and TOC removal efficiencies was noted in the RBC units (COD removal fell from 77% to 69% and TOC removal from 86% to 76%). The BOD_5 removal was not affected and remained fairly constant at 86%. The substrate removal efficiencies are illustrated in Figs. 6.3 and 6.4. However after a day or two, the removal rate increased until quasi steady-state conditions were achieved. Consistent effluent substrate (COD, BOD_5 and TOC) concentrations were observed after 20 days.

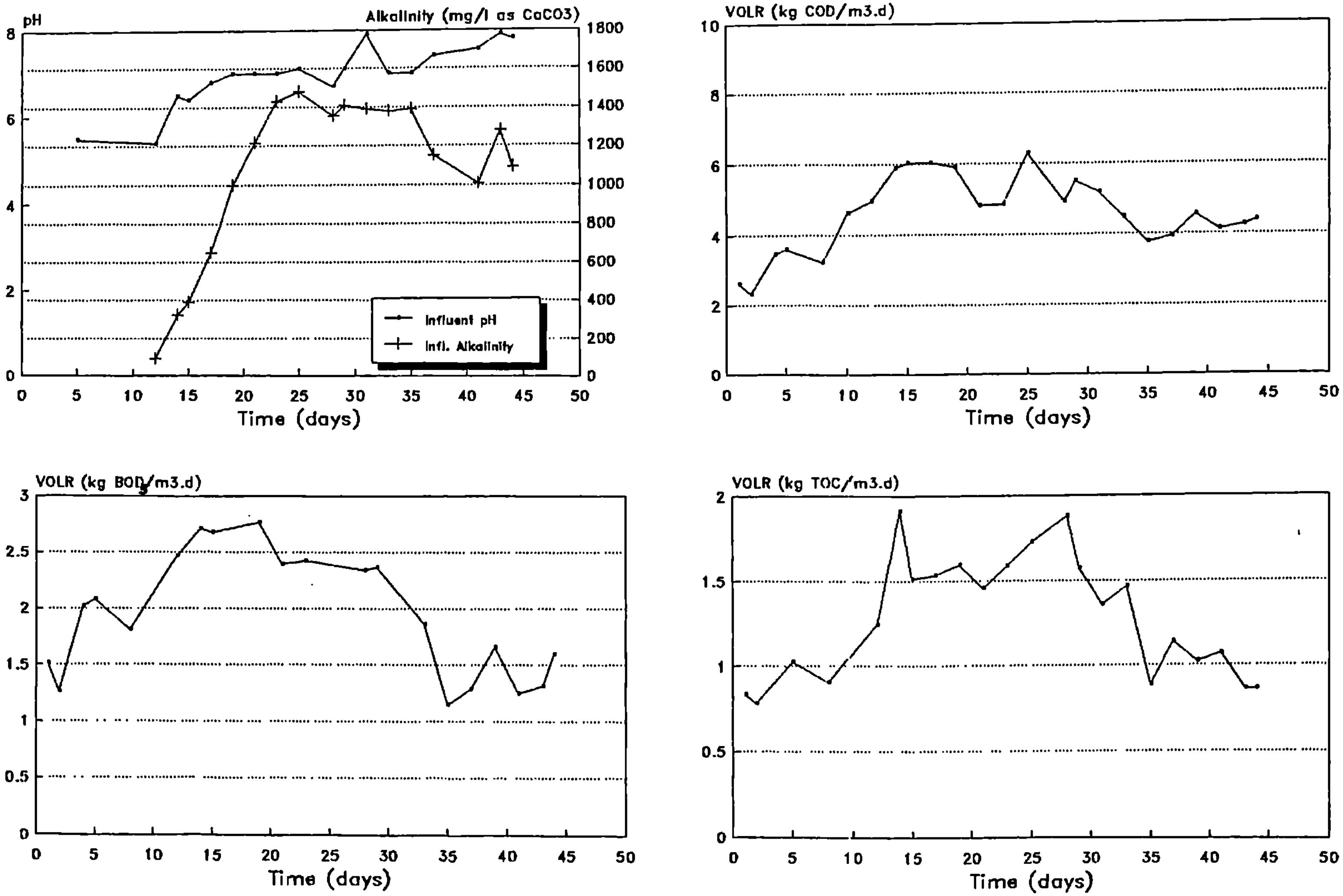


Figure 6.1: Influent pH & Alkalinity and Operational Conditions for RBC Start-Up (Phase I Study)

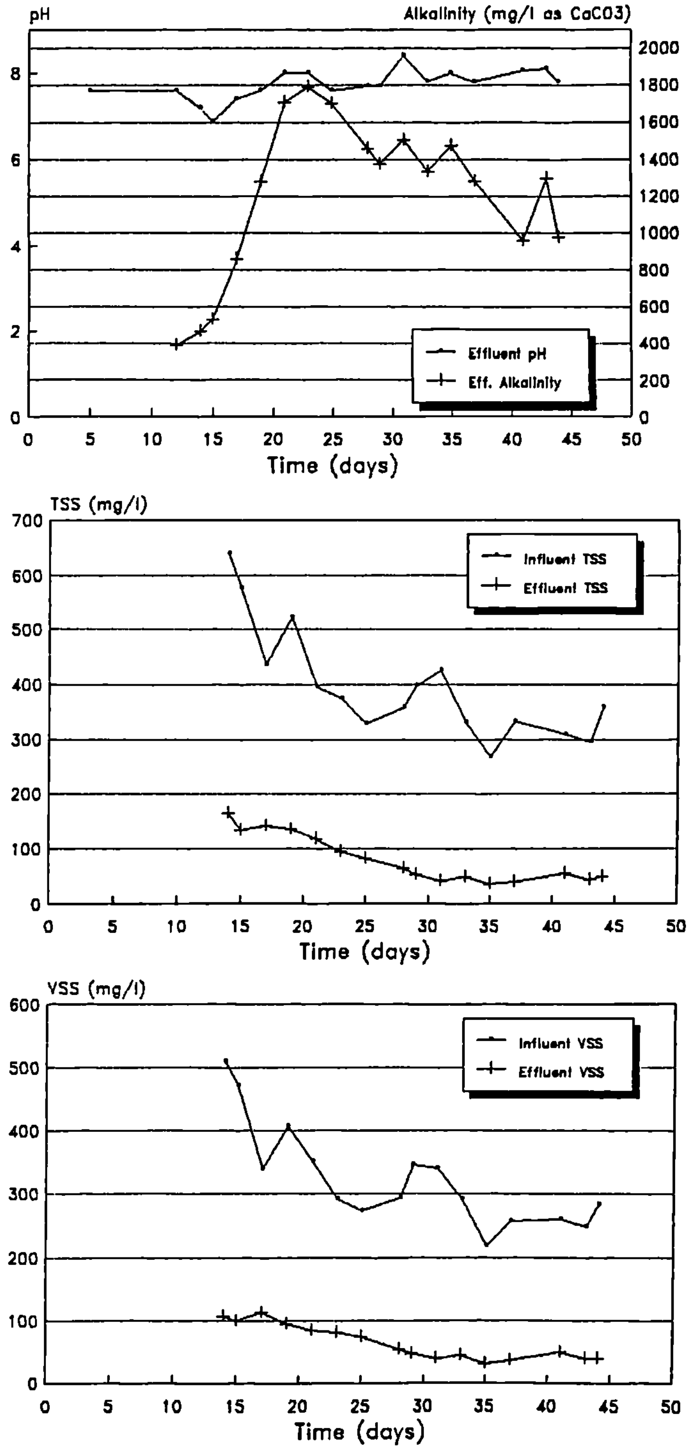


Figure 6.2: Effluent pH & Alkalinity, Influent and Effluent Total And Volatile SS for RBC during Start-Up (Phase I)

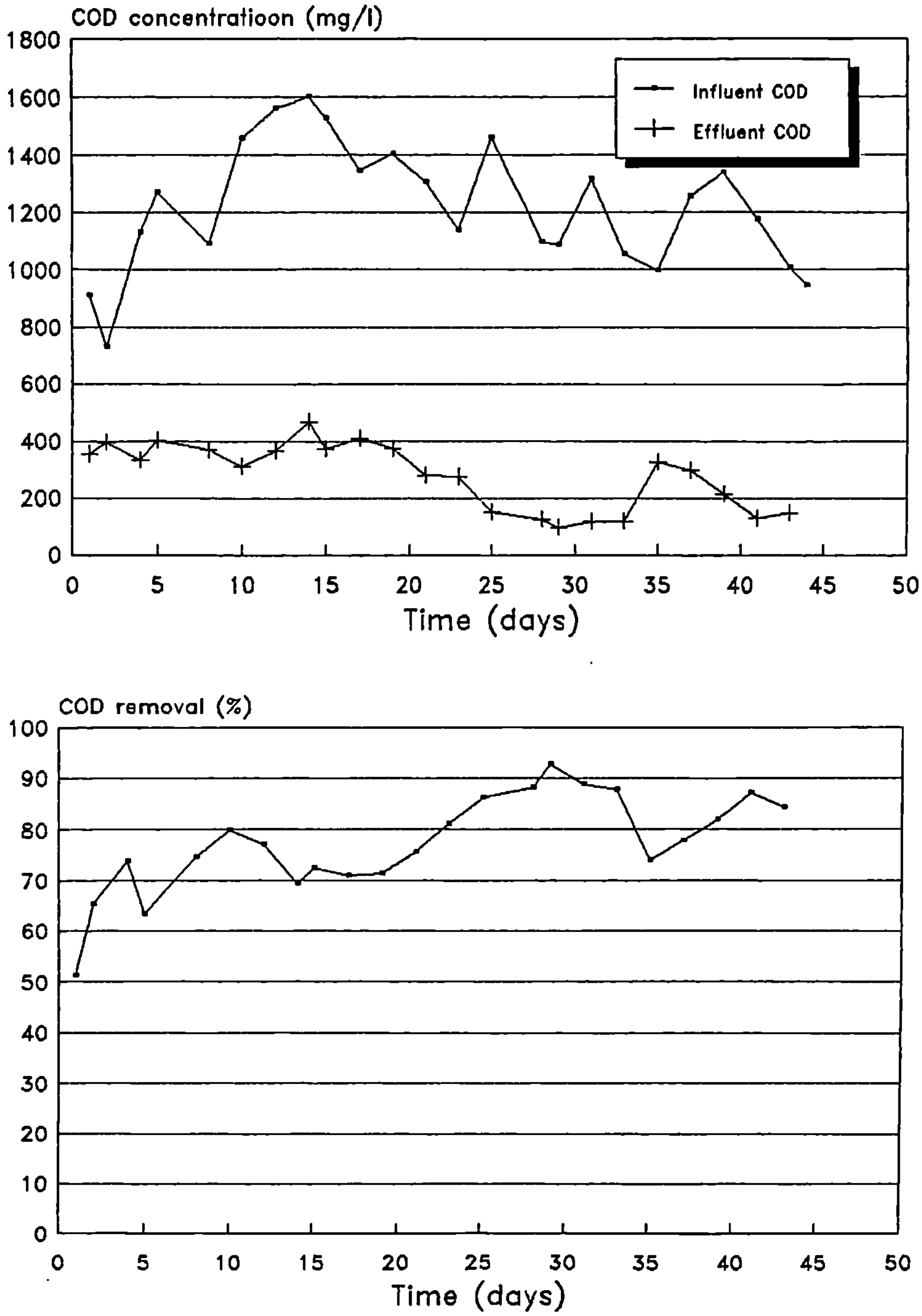


Figure 6.3: COD Concentrations and Removal for RBC during Start–Up (Phase I Study)

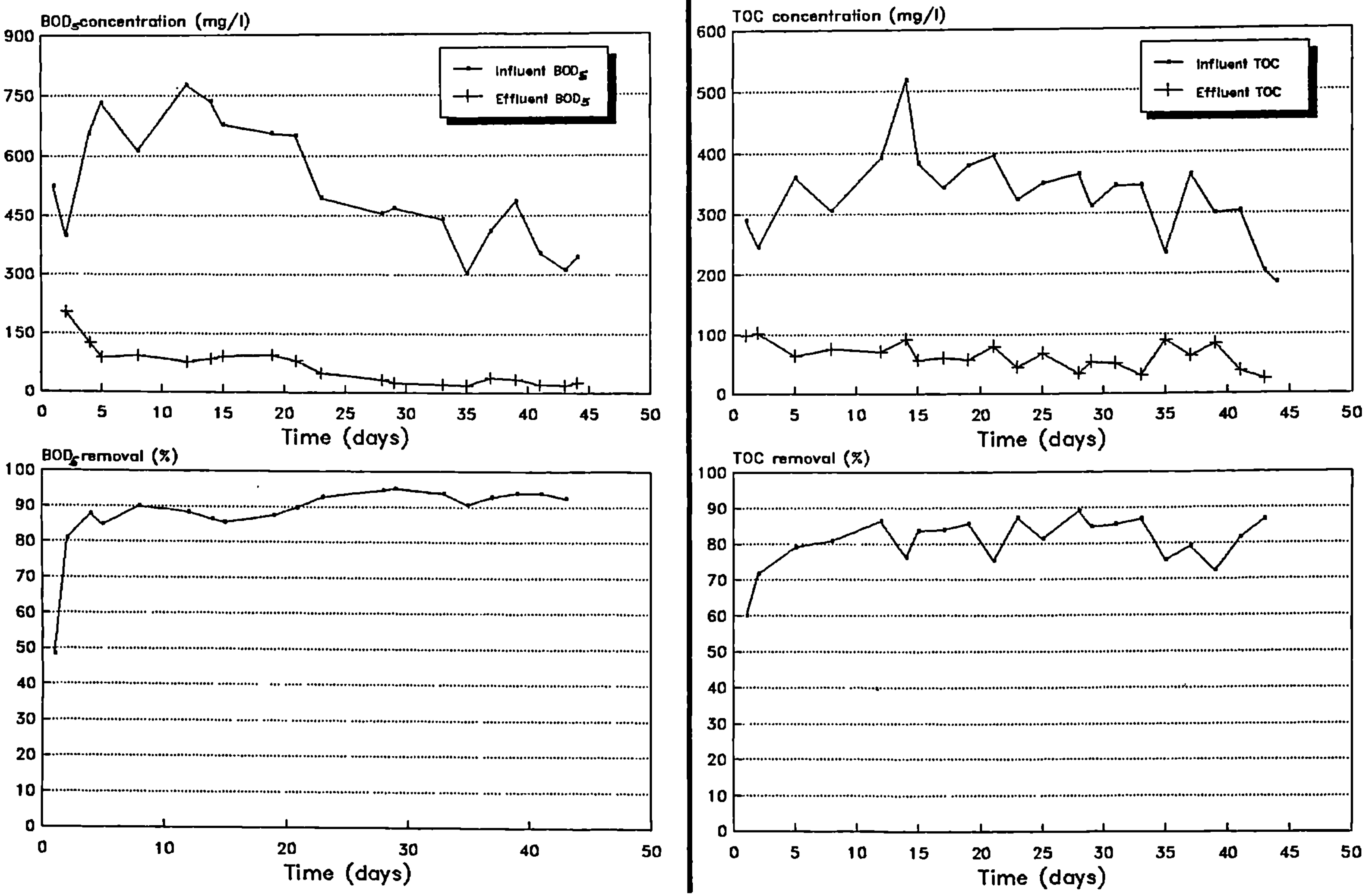


Figure 6.4: BOD_5 and TOC Concentrations & Removal for RBC during Start-Up (Phase I Study)

As for the UAF treatment process, at the beginning of feeding, the percentage methane content in the biogas was found to be around 40% and the percentage methane increased rapidly during the first 10 days of feeding and was maintained at between 85 to 90% during the remainder of the Phase I study. The biogas production at the start of the UAF was found to be 0.3 l/day (0.15 m^3 biogas/kg COD removed). The biogas gradually increased (see Fig. 6.6) until it stabilized at an average of 2.2 l/day (0.35 m^3/kg COD removed) 20 days after the system start-up. The quantity of methane produced per kilogram of COD removed was calculated. The results are listed in Table 6.5. Since the rates of biogas production responded rapidly to substrate loading changes, acclimatization could not be gauged using biogas production. The acclimatization period in this case was determined by monitoring the effluent substrate and the concentration of suspended solids.

An average volumetric loading rate of 1.25 kg COD/m^3.day (based on the active liquid volume) in the UAF was maintained throughout the start-up phase. The performances of the RBC and UAF units during start-up are summarized in Tables 6.4 and 6.5.

In Figs. 6.8 and 6.9, it can be seen that the UAF units were capable of removing around 33% of the COD (44% BOD_5 removal) from the synthetic feed as soon as the units began to function. Most of the COD came from glucose which was the main ingredient in the synthetic wastewater. After about 20 days continuous operation, the COD reduction had increased to over 70% (over 80% BOD_5 reduction).

Upon the introduction of 20% LS leachate to the synthetic feed, the

Table 6.4 Performance of the RBC during Start-Up

Parameter	whey powder		LS leachate	
	Range	Mean	Range	Mean
Effl. Settled COD (mg/l)	350 - 455	405	110 - 500	285
SCOD reduction (%)	45.5 - 77.5	64	67.0 - 91.5	78
Effl. Filtered COD (mg/l)	315 - 400	360	100 - 470	250
FCOD reduction (%)	51.0 - 80.0	68	69.5 - 92.5	80
Effl. Filtered BOD$_5$ (mg/l)	80 - 205	120	20 - 95	45
BOD$_5$ reduction (%)	48.5 - 90.0	78	85.5 - 95.0	91
Effl. TOC (mg/l)	65 - 100	85	25 - 90	60
TOC reduction (%)	60.0 - 81.0	73	72.5 - 89.0	82
Effluent pH		7.6	6.9 - 8.4	7.8
Effluent TSS (mg/l)			35 - 165	80
TSS reductiom (%)			67.5 - 90.0	80
Effluent VSS (mg/l)			30 - 115	65
VSS reduction (%)			67.0 - 88.5	80

Table 6.5: Performance of the UAF during Start-Up

Parameter	Synth. Feed (SF)		SF + LS Leachate		LS Leachate	
	Range	Mean	Range	Mean	Range	Mean
Effl. Settled COD (mg/l)	240 - 690	415	190 - 390	250	50 - 305	160
COD reduction (settled) (%)	30.0 - 74.0	58	63.5 - 84.5	75	74.5 - 95.0	87
Effl. Filtered COD (mg/l)	210 - 655	385	165 - 370	225	40 - 275	135
COD reduction (filtered) (%)	33.0 - 77.5	61	77.0 - 87.0	78	77.0 - 96.0	89
Effl. Filtered BOD_5 (mg/l)	65 - 185	120	55 - 105	75	10 - 65	30
BOD_5 reduction (%)	44.0 - 85.5	67	80.5 - 91.5	86	89.0 - 97.5	94
Effl. TOC (mg/l)	45 - 130	80	30 - 75	45	20 - 60	40
TOC reduction (%)	37.5 - 79.5	64	67.5 - 85.5	80	70.5 - 92.0	86
Effluent pH	6.3 - 6.9	6.6	6.6 - 7.4	7.0	6.8 - 8.6	7.7
Effluent TSS (mg/l)	435 - 770	625	65 - 355	225	25 - 95	50
TSS reductiom (%)	-72.5 - -12.0	-34	16.0 - 88.5	51	74.0 - 93.0	87
Effluent VSS (mg/l)	315 - 540	420	55 - 290	165	25 - 75	40
VSS reduction (%)	-43.0 - 20.0	-6	18.5 - 87.5	56	75.5 - 92.5	87
Effluent Total VFA (mg/l)			32.5 - 64.5	44	24.5 - 78.0	47
Effl. Acetic Acid (mg/l)			6.0 - 35.0	16	1.0 - 48.0	17
Effl.Propionic Acid (mg/l)			26.5 - 30.5	28	19.5 - 35.0	29
Gas Production (l/day)	0.30 - 2.45	1.45	2.00 - 2.50	2.25	1.70 - 3.30	2.40
Methane Yield (m^3 CH_4/kg COD)	0.06 - 0.33	0.21	0.27 - 0.33	0.30	0.23 - 0.37	0.30

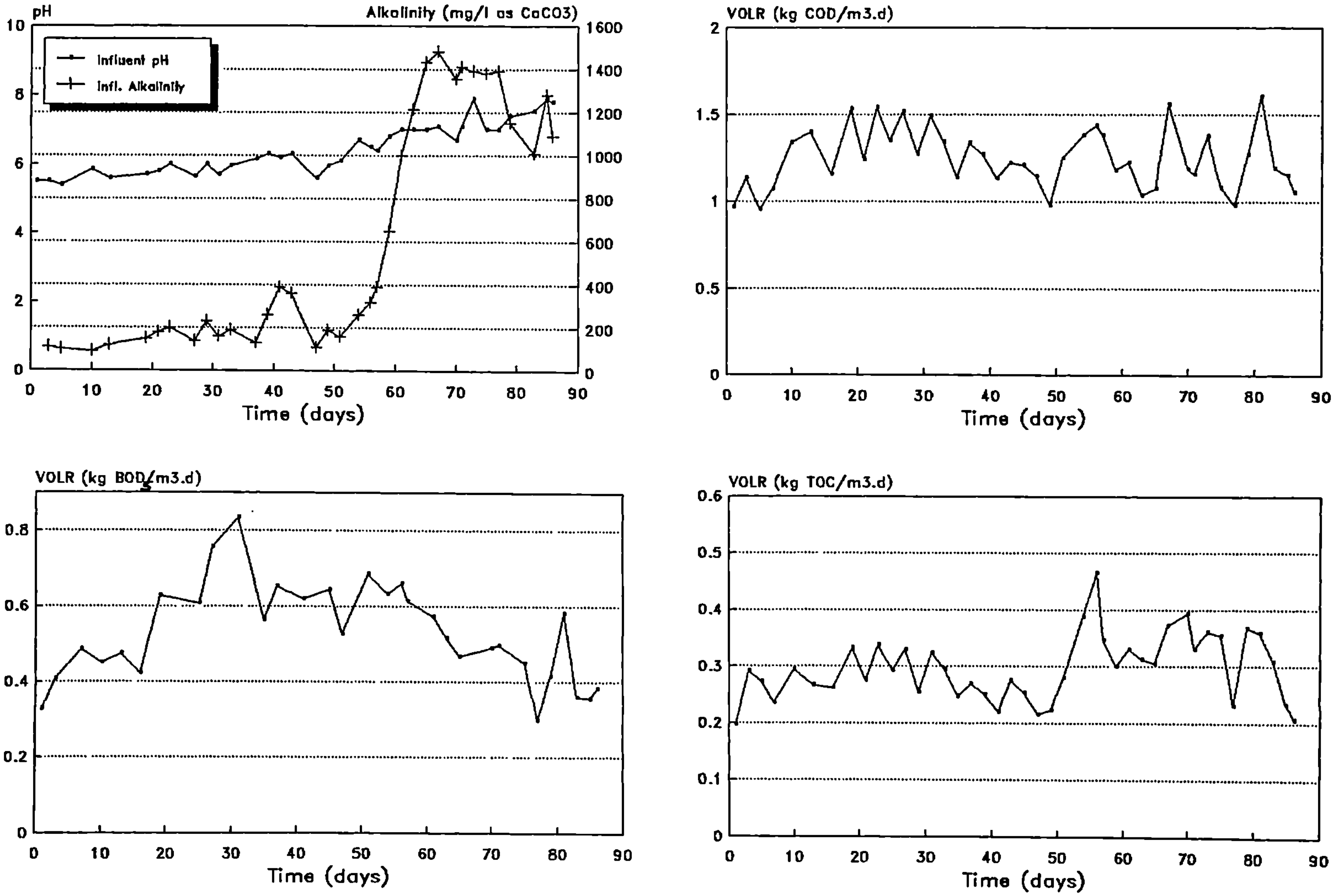


Figure 6.5: Influent pH & Alkalinity and Operational Conditions for UAF during Start-Up (Phase I Study)

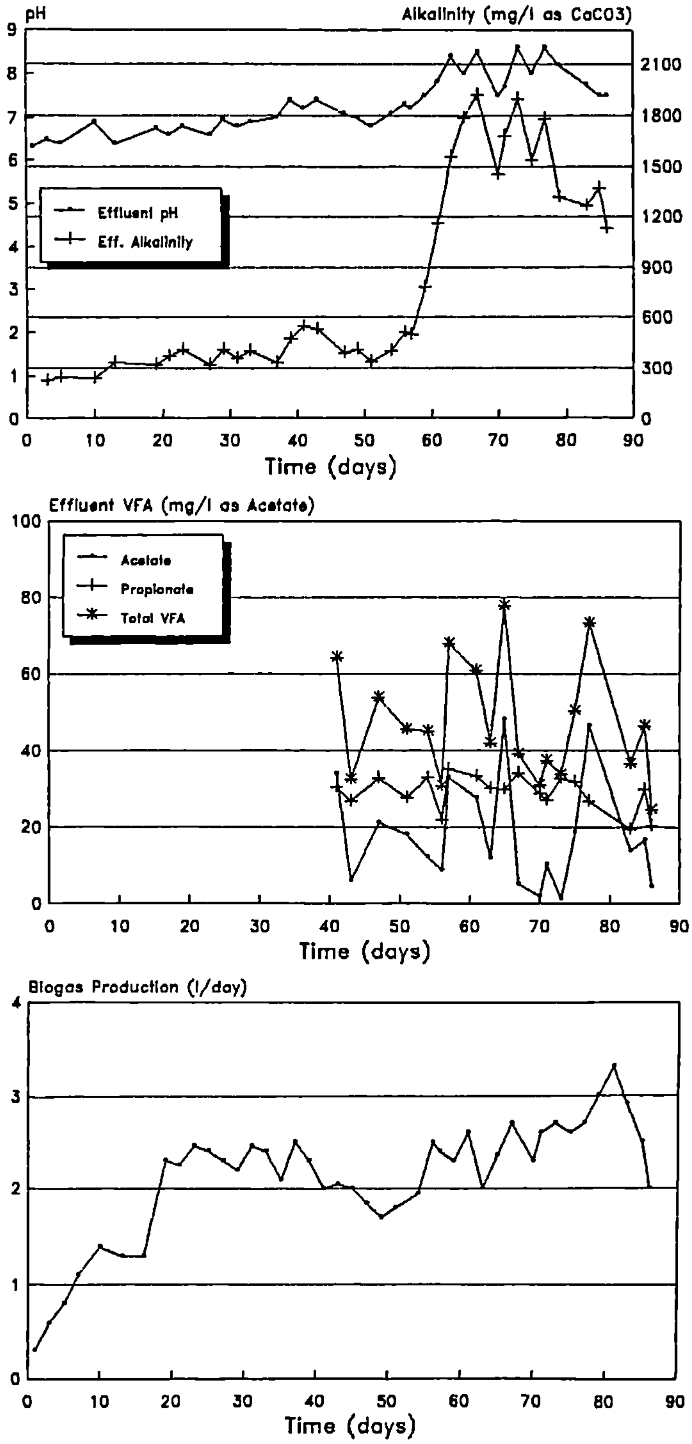


Figure 6.6: Effluent pH, Alkalinity & VFAs and Biogas Production for UAF during Start-Up (Phase I Study)

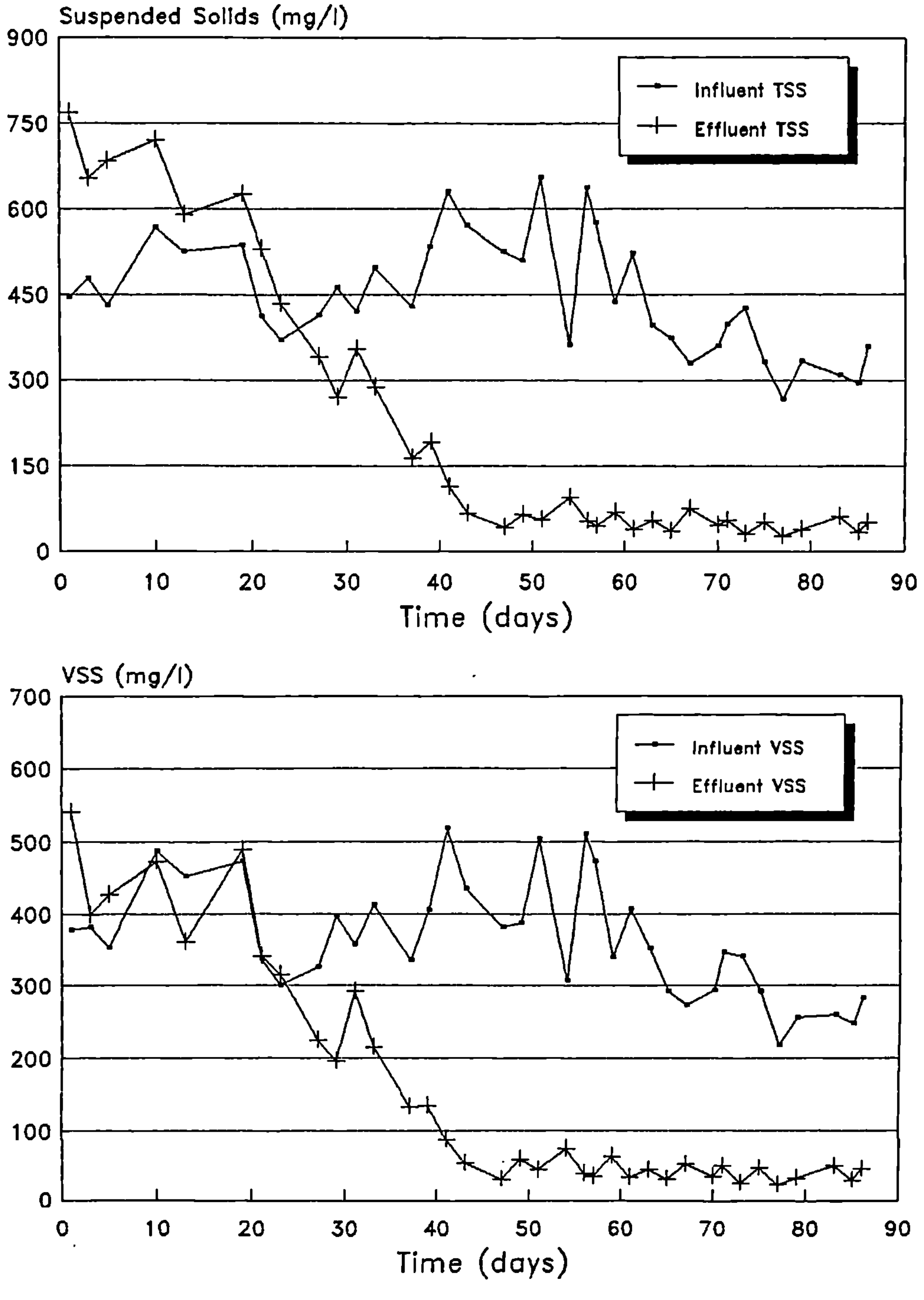


Figure 6.7: Influent & Effluent TSS and VSS for UAF during Start-Up (Phase I Study)

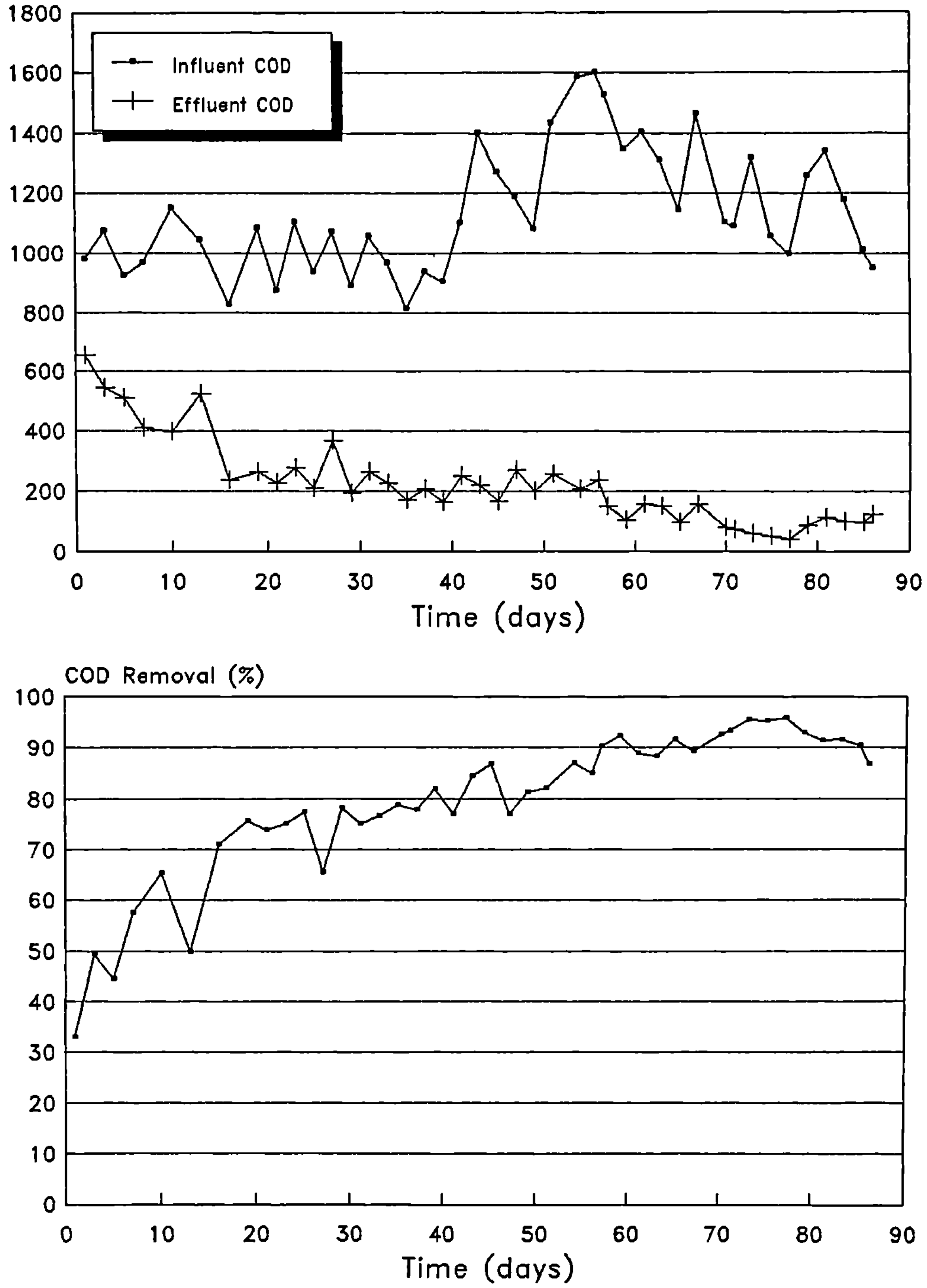


Figure 6.8: COD Concentrations and Removal for UAF during Start-Up (Phase I Study)

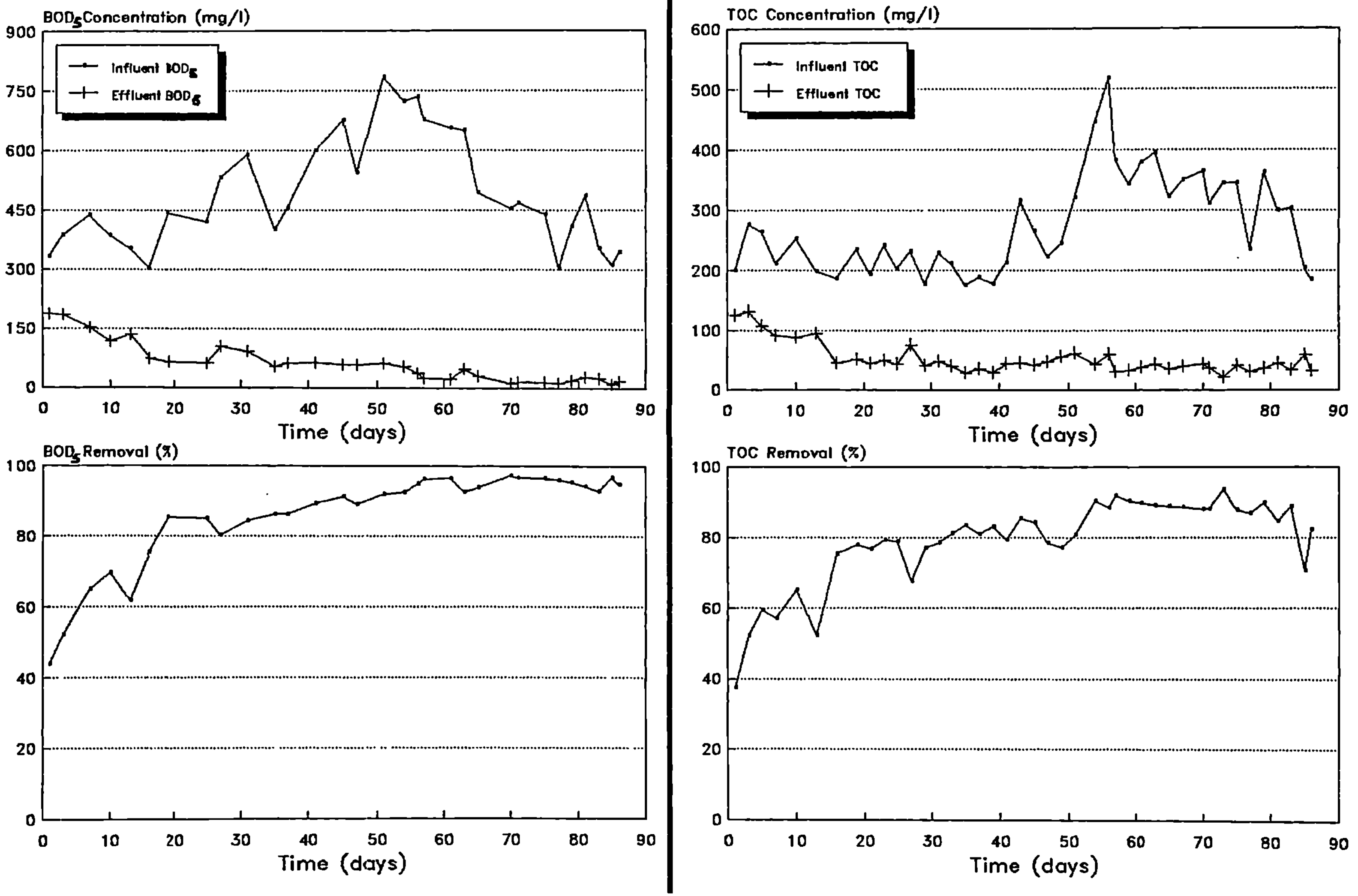


Figure 6.9: BOD_5 & TOC Concentrations and Removal for UAF during Start–Up (Phase I Study)

removal efficiencies for the three parameters (COD,BOD_5, and TOC) were found to decrease (a decrease in COD removal from 77% to 65%, BOD_5 removal from 85% to 80%, and TOC removal from 79% to 68%). This sudden reduction of removal efficiencies is attributed to the microorganisms adjusting to the new substrate. Quasi steady state conditions in the UAF units were clearly established by day 60 - 70.

The effluent total volatile fatty acids (VFAs) during the Phase I study of the UAF and measured 40 days after starting, were within the range of 25 to 80 mg/l (an average of 45 mg/l). Although the influent VFAs in the UAF units fluctuated, even up to a maximum of 590 mg/l as $CaCO_3$, the effluent VFAs remained low indicating that microbial activity was continuing within the reactors.

It was difficult to see when attachment of the biomass occurred in the UAF reactors due to their being completely enclosed and their walls covered with a black sludge. Hence, during the early stages an indication that biomass retention in the UAF existed was acknowledged by considering the amount of TSS and VSS removed with the effluent. The average effluent TSS (625 mg/l) in UAF in the beginning of start-up operation was found to be greater than the average influent TSS (470 mg/l). The probable cause of this observation might be that biological solids, which remained dispersed during the early stages of operation, had been washed out with the filter effluent. Effluent TSS and VSS decreased, as the biomass became attached to the media. In both the RBC and UAF units, the suspended solids, built up during the first few days, began to decrease as the reactors approached quasi steady-state conditions. After day 27 onward during the Phase I study for the UAF reactors, the TSS and VSS removal began to increase (see Fig. 6.7) until they stabilized at around 87%.

6.3.2 Response to pH Variations

The results for the Phase II study on pH variations are shown in Table 6.6 and Figs 6.10 and 6.11. The study was conducted using LS leachate. In the RBC units, a pH of between 5.5 and 8.0 produced a COD removal of greater than 70%. For a BOD removal of greater than 80%, the pH was found to lie between 5.3 and 7.5. Outside this range the removal efficiency decreased, though the decreases in substrate reduction was small. Norton (1984) in his study of the RBC treatment process reported an optimum pH range of between 6.0 and 8.0 with only a small reduction in the removal rate at a pH of 9.0.

Table 6.6 pH versus Removal Efficiencies

	% removal in RBC			% removal in UAF		
pH	COD	BOD_5	TOC	COD	BOD_5	TOC
4.2	63.0	69.4	55.8	55.0	66.8	43.0
5.2	65.5	78.9	59.1	62.4	77.8	48.0
5.5	69.6	76.7	63.8	62.8	72.9	54.8
5.9	70.1	84.9	57.4	67.6	75.9	51.6
6.4	76.8	85.1	60.6	74.8	82.7	59.3
6.6	70.9	82.4	58.1	68.4	80.8	53.8
7.0	76.0	91.9	64.5	74.9	89.7	63.6
7.4	77.2	75.3	58.2	77.7	79.1	57.8
7.5	75.4	74.0	57.6	74.1	74.6	52.8
7.6	71.5	69.0	50.1	71.4	73.0	48.9
8.0	67.8	60.8	47.3	67.5	68.0	46.1
8.3	64.5	60.3	50.5	63.4	60.4	48.4
8.6	62.9	56.1	46.8	60.1	60.9	43.4

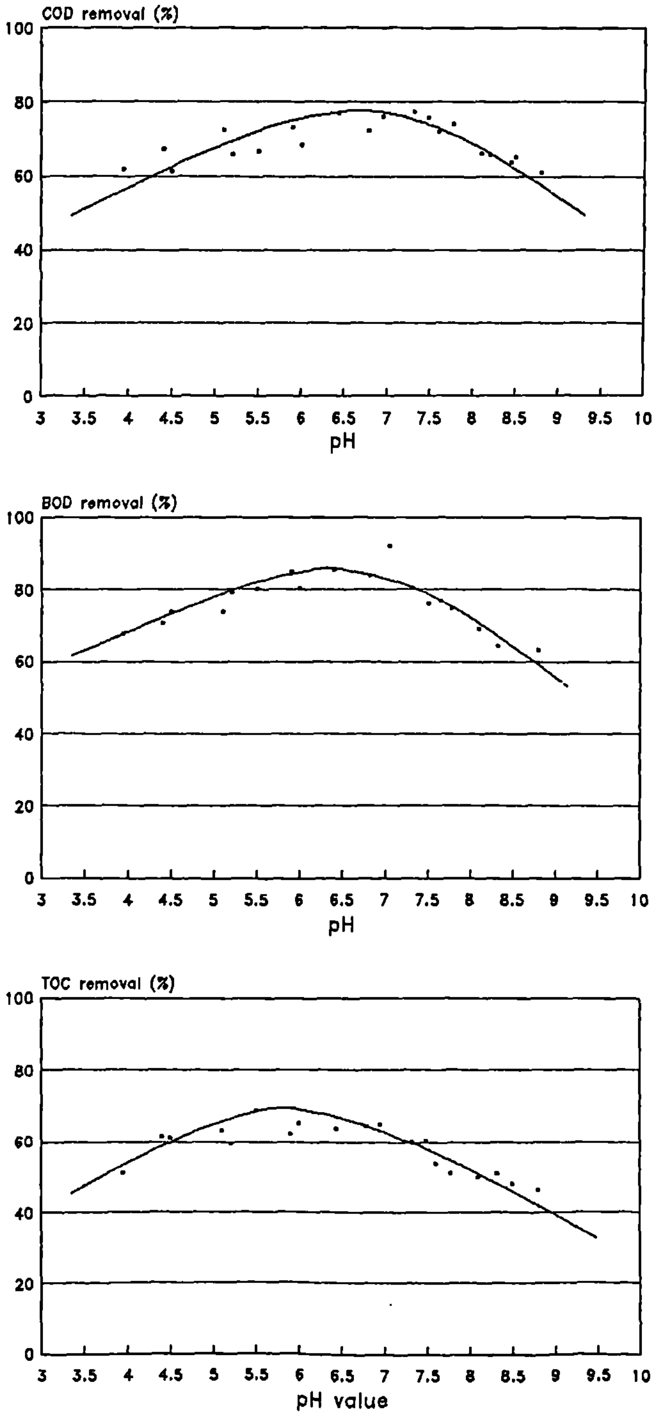


Figure 6.10: Relationship between pH and COD, BOD_5 and TOC Removals of RBC (Phase II Study)

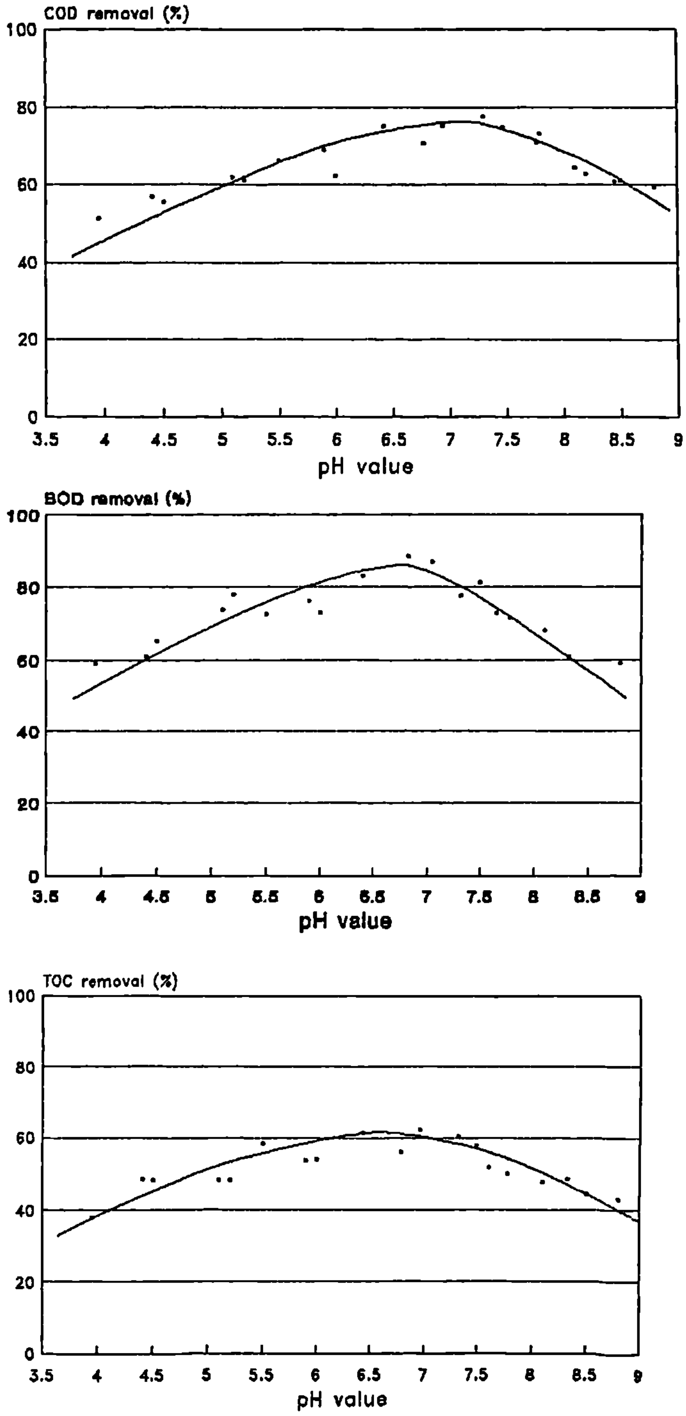


Figure 6.11: Relationship between pH and COD, BOD_5 and TOC Removals of UAF (Phase II Study)

For the UAF, a pH of between 6.0 and 7.5 gave a COD removal of over 70% (and a BOD_5 removal of over 80%). Anaerobic digestion requires a stable pH for optimum organic removal. This can be seen from the removal efficiencies in the UAF which decreased sharply when the pH values were outside the optimum range. Thus, adjustment of the buffer is required to maintain the required pH range. Without adequate pH control, a prolonged imbalance in the digester could lead to total inhibition in the anaerobic digestion process (McCarty, 1964). However in the case of the RBC, since the latitude in pH is large, the need for automatic correction of feed pH is not so vital.

Automatic pH correction which is widely used includes a pH probe installed at a control point in the digester system, an alkali and/or acid pumping system and an electronic control circuit. However for this system to work effectively and economically, the liquid phase of the digester should be constantly and completely mixed. However, Young (1991) reported that the mixing conditions in a full-scale UAF heavily depend on the biogas flux that is being produced, which in turn depends on the loading and the environmental conditions, including the pH. But in practice, in a full-scale treatment plant, recirculation is most probably the dominant criterion. Hence, this cyclic dependency makes the reliability of this type of control system questionable (Yang and Anderson, 1990).

In this study, pH control was carried out using the method developed by Yang and Anderson (1990). It was shown from laboratory studies that bicarbonate is a more sensitive parameter than both pH and total alkalinity to represent the dynamic changes in anaerobic reactors. Hence the pH in the digester during the steady state performance study was maintained by controlling the capacity of the bicarbonate buffer system.

6.3.3 LS and HS Quasi Steady State Performances

Experimental runs during Phase III were made using both LS and HS leachates. The influent LS and HS leachate characteristics, and the operational conditions for both the RBC and the UAF units during the Phase III study are listed in Tables 6.7, 6.8 and 6.9 respectively.

- TOC, BOD_5 and COD Relationships

In this experimental phase, attempts were made to examine the validity of using total organic carbon (TOC), besides the conventional method which used COD or BOD_5, in determining the organic substrate changes. Although TOC analysis is generally faster, more accurate and more reproducible than the conventional BOD_5 test, Hamoda and Wilson (1989) reported that the use of TOC was very limited, probably due to the high variability of the relationships quoted for different wastes between BOD_5 and TOC values and to the high cost of the analytical system. Table 6.10 listed the relationships obtained between the influent and effluent TOC, COD and BOD_5 for both the RBC and the UAF.

The results from Table 6.10, indicated that BOD_5 relates consistently with TOC and COD for the influent LS leachate. The regression coefficients established were significant at levels considerably better than the 0.1% level. As for the effluent LS leachate, with the exception of the RBC BOD_5:COD relationship, the others showed a strong linear relationship. Although the influent BOD_5:COD relationship for the HS leachate was found

Table 6.7: Influent Leachate Characteristics for RBC and UAF during Phase III Study*

Parameters	LS Leachate				HS Leachate			
No. of Runs	1st	2nd	3rd	4th	1st	2nd	3rd	4th
Period (days)	51	56	53	51	31	31	27	20
Infl. Total COD (mg/l)	1280	1785	1800	1800	5605	6070	6090	7205
Infl. Total BOD_5 (mg/l)	385	565	720	745	3190	3615	3335	3765
Infl. TOC (mg/l)	290	400	420	420	1370	1495	1815	2015
Infl. pH	7.15	7.15	7.10	7.15	7.15	7.25	7.45	7.45
Alkalinity infl. (mg/l)	665	805	860	940	2355	2705	2120	1860
Infl. TKN (mg/l)	80	110	95	115	405	465	520	
Infl. NH_3-N (mg/l)	50	60	60	75	195	355	420	
Infl. TSS (mg/l)	460	555	495	510	385	470	435	440
Infl. VSS (mg/l)	355	400	375	385	200	265	255	220

Note: * Average values in each experimental run

Table 6.8: Operational Conditions for RBC Phase III Study

Parameters	LS Leachate				HS Leachate			
No. of Runs	1st	2nd	3rd	4th	1st	2nd	3rd	4th
Period (days)	51	56	53	51	31	31	27	20
Org.L.R. (kg COD/m^3.d)	4.65	7.00	9.00	12.00	3.45	6.20	8.95	17.65
Org.L.R. (kg BOD_5/m^3.d)	1.40	2.25	3.60	5.00	1.90	3.65	4.90	9.25
Org. L.R. (kg TOC/m^3.d)	1.05	1.60	2.15	2.80	0.85	1.50	2.70	4.90
Areal L.R. (g COD/m^2.d)	18.15	27.65	35.10	46.65	13.40	24.20	34.90	68.65
Areal L.R. (g BOD_5/m^2.d)	5.45	8.70	14.05	19.35	7.70	12.15	13.95	21.00
Areal L.R. (g TOC/m^2.d)	4.10	6.20	8.35	10.90	3.30	6.00	9.75	16.75
HRT (hrs)	6.5	6.0	5.0	3.5	39.5	23.5	16.5	10.0

Table 6.9: Operational Conditions for UAF Phase III Study

Parameters	LS Leachate				HS Leachate			
No. of Runs	1st	2nd	3rd	4th	1st	2nd	3rd	4th
Period (days)	51	56	53	51	31	31	27	20
Org.L.R. (Kg COD/m^3.d)	1.10	2.35	3.85	6.10	2.05	4.65	7.55	12.90
Org.L.R. (Kg BOD_5/m^3.d)	0.35	0.75	1.55	2.55	1.15	2.80	4.05	6.75
Org. L.R. (Kg TOC/m^3.d)	0.25	0.55	0.90	1.40	0.50	1.15	2.25	3.60
Areal L.R (g COD/m^2.d)	6.15	13.40	22.00	35.05	11.80	26.70	43.20	73.70
Areal L.R. (g BOD_5/m^2.d)	1.85	4.20	8.85	14.55	6.60	16.00	23.30	38.55
Areal L.R. (g TOC/m^2.d)	1.40	3.05	5.30	8.15	2.90	6.60	12.85	20.65
HRT (hrs)	29.0	18.5	11.5	7.0	67.5	31.5	19.5	13.5

Table 6.10: Relationship between TOC,COD and BOD_5 for LS and HS Leachates

Relationship for LS Leachate	R	R^2	No. of Observations
TOC = 0.213COD + 24.95 (influent)	0.888	0.789	103
TOC = 0.172COD + 27.54 (AF effluent)	0.921	0.849	103
TOC = 0.207COD + 24.24 (RBC effluent)	0.848	0.720	103
BOD_5 = 1.892TOC - 119.50 (influent)	0.788	0.621	103
BOD_5 = 1.133TOC - 33.23 (AF effluent)	0.881	0.776	103
BOD_5 = 1.699TOC - 72.97 (RBC effluent)	0.854	0.730	103
BOD_5 = 0.464COD - 175.70 (influent)	0.807	0.652	103
BOD_5 = 0.228COD - 15.80 (AF effluent)	0.948	0.898	103
BOD_5 = 0.089COD -58.79 (RBC effluent)	0.441	0.195	103
Relationship for HS Leachate	**R**	**R^2**	**No. of Observations**
TOC = 0.354COD - 535.22 (influent)	0.802	0.644	40
TOC = 0.392COD - 186.54 (AF effluent)	0.958	0.918	40
TOC = 0.347COD - 149.63 (RBC effluent)	0.954	0.911	40
BOD_5 = 0.587TOC + 2486.86 (influent)	0.559	0.313	40
BOD_5 = 0.489TOC + 209.75 (AF effluent)	0.860	0.740	40
BOD_5 = 0.982TOC + 265.66 (RBC effluent)	0.874	0.764	40
BOD_5 = 0.353COD + 1283.41 (influent)	0.763	0.582	40
BOD_5 = 0.199COD + 106.83 (AF effluent)	0.858	0.737	40
BOD_5 = 0.379COD + 46.47 (RBC effluent)	0.928	0.861	40

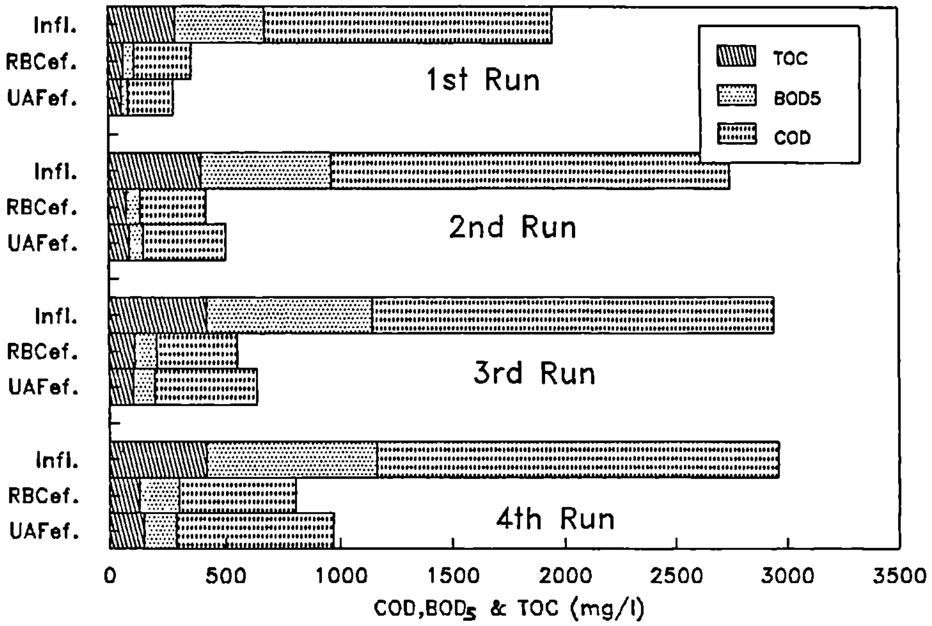
to be significant, it is rather low. The regression coefficient was disappointing in the influent BOD_5:TOC relationship. A good linear relationship was exhibited between BOD_5 with TOC and COD for the effluent samples. Similarly, both the influent and effluent LS and HS leachates gave significant TOC:COD relationships. From these results, it is quite safe to say that TOC could be used in lieu of the other two parameters to study changes in the organic substrate.

From Table 6.7, the ratio of BOD_5/COD for LS leachate was found to lie within the range of 0.30 to 0.41, indicating that the leachate was partially stabilized (a characteristic of an "ageing" landfill) and only a small amount of biodegradable materials remained in the leachate composition. The BOD_5/COD ratio for HS leachate fell within 0.52 to 0.60, which showed that the leachate came from a "maturing" landfill and was in the process of degradation. The relationships between COD, BOD_5 and TOC for LS and HS leachates are illustrated in Figs. 6.12a and 6.12b respectively.

- LS Steady State (LS S-S) Performance

The first quasi steady state condition for the RBC was attained at an average VOLR of 4.65 kg COD/m^3.day (AOLR of 18.15 g COD/m^2.day). As for the UAF reactor, quasi steady state was achieved at an average OLR of 1.1 kg COD/m^3.day (AOLR of 6.15 g COD/m^2.day). The first experimental run was operated continuously for 51 days. After a lapse of two days, the second experimental run was operated for 56 days. In order to increase the loading rate for the second run, the HRT was reduced. Similarly, the operations were repeated for the third and fourth experimental runs.

a: LS Leachate



b: HS Leachate

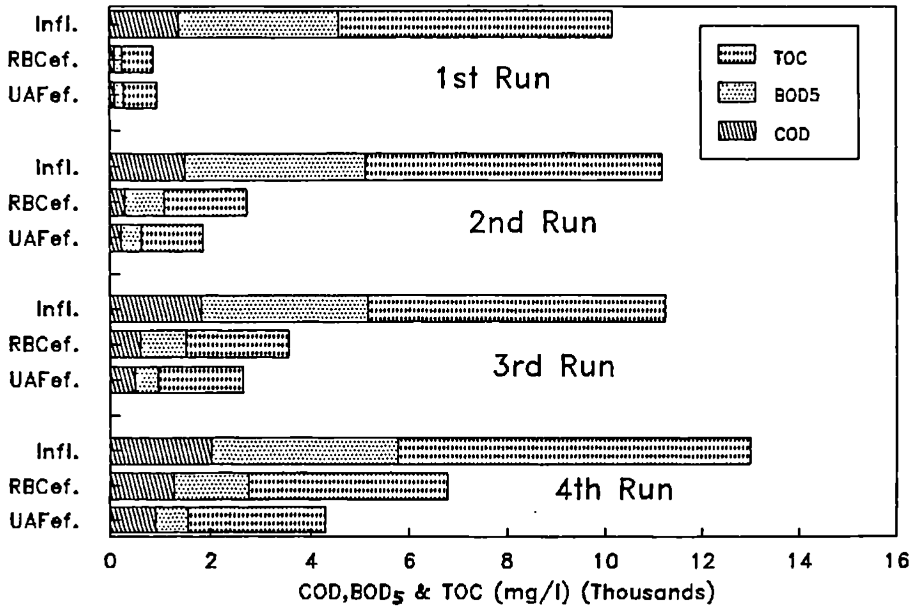


Figure 6.12: COD, BOD$_5$ and TOC Relationships

The performances of the RBC and the UAF reactors during the four experimental runs are summarized in Tables 6.11 and 6.12 respectively. The average influent leachate in the first experimental run was much lower than the other three runs (see Table 6.7). Figs. 6.13 to 6.16 illustrate the RBC performance, while the UAF performance is presented in Figs. 6.17 to 6.21.

Although, the influent leachate varies between the experimental runs, from Figs. 6.15 and 6.16, it can be observed that the RBC reactors produced a consistent substrate removal during the first to the third runs. The highest substrate removal based on filtered effluent was achieved during the second run with an average VOLR of 7 kg COD/m^3.day (AOLR of 27.65 g COD/m^2.day) or 2.25 kg BOD$_5$/m^3.day (AOLR of 8.7 g BOD$_5$/m^2.day) or 1.6 kg TOC/m^3.day (AOLR of 6.2 g COD/m^2.day) at 84% as COD, 90% as BOD$_5$ and 80% as TOC. At the highest loading of 12.0 kg COD/m^3.day (AOLR of 46.65 g COD/m^2.day), as in the fourth run, the substrate removal decreased to 72% as COD, 77% as BOD$_5$ and 69% as TOC.

On the other hand, from Figs. 6.20 and 6.21, the substrate removal for the UAF reactor showed downward trends from the first run to the fourth run. An increase in OLR (i.e a decrease in HRT) was found to reduce the performance of the UAF. The first run, with an average VOLR of 1.1 kg COD/m^3.day (AOLR of 6.15 g COD/m^2.day) or 0.35 kg BOD$_5$/m^3.day (AOLR of 1.85 g BOD$_5$/m^2.day) or 0.25 kg TOC/m^3.day (AOLR of 1.4 g TOC/m^2.day) was observed to give the highest substrate removal based on filtered effluent at 85% as COD, 93% as BOD$_5$ and 80% as TOC. While the fourth run with an average VOLR of 6.1 kg COD/m^3.day (AOLR of 35.0 g COD/m^2.day) produced substrate removal of 62% as COD, 81% as BOD$_5$ and 64% as TOC.

Table 6.11: Performance of the RBC during LS and HS Steady State Conditions*

Parameter	LS Leachate				HS Leachate			
No. of Runs	1st	2nd	3rd	4th	1st	2nd	3rd	4th
Period (days)	51	56	53	51	31	31	27	20
SCOD$_e$ (mg/l)	285	330	415	565				
Eff. SCOD reduction (%)	77	81	77	69				
FCOD$_e$ (mg/l)	250	285	345	500	625	1620	2040	4025
Eff. FCOD reduction (%)	81	84	81	72	89	73	66	44
BOD$_{5e}$ (mg/l)	45	60	95	170	145	790	915	1485
Eff. BOD$_5$ reduction (%)	89	90	87	77	95	78	73	61
TOC$_e$ (mg/l)	65	80	110	130	100	310	600	1275
Eff. TOC reduction (%)	77	80	74	69	93	80	67	37
Effluent pH	7.8	7.9	8.0	8.1	8.1	8.2	8.2	8.1
Alkalinity effl. (mg/l)	705	880	935	1000	1870	2920	2590	2090
Effluent TKN (mg/l)	75	75	75	50	170	395	495	
Effluent NH$_3$N (mg/l)	55	55	55	35	125	285	415	
Effluent TSS (mg/l)	45	60	90	160	145	470	385	435
TSS reduction (%)	90	88	80	68	56	0.25	5	-3.5
Effluent VSS (mg/l)	35	50	70	120	95	315	255	265
VSS reduction (%)	89	85	79	68	37	-17	-10	-25

*Average values for each experimental runs

Table 6.12 Performance of UAF during LS and HS Steady State Conditions*

Parameter	LS Leachate				HS Leachate			
No. of Runs	1st	2nd	3rd	4th	1st	2nd	3rd	4th
Period (days)	51	56	53	51	31	31	27	20
$SCOD_e$ (mg/l)	235	390	470	720				
Eff. SCOD reduction (%)	82	78	73	60				
$FCOD_e$ (mg/l)	195	350	440	680	660	1215	1680	2755
Eff. FCOD reduction (%)	85	80	75	62	88	80	72	62
BOD_{5e} (mg/l)	25	60	90	140	190	400	460	645
Eff. BOD_5 reduction (%)	93	89	87	81	94	89	86	83
TOC_e (mg/l)	60	90	105	150	95	230	500	905
Eff.TOC reduction (%)	80	77	75	64	93	84	72	55
Effluent pH	7.1	7.0	7.0	7.1	7.2	7.4	7.4	7.4
Alkalinity effl. (mg/l)	765	990	1050	1125	2160	3510	2975	2575
TKN_e (mg/l)	80	95	100	100	175	460	495	
NH_3N_e (mg/l)	60	75	70	75	135	370	415	
TSS_e (mg/l)	40	60	95	100	120	310	280	290
TSSred (%)	91	87	78	79	68	36	35	32
VSS_e (mg/l)	30	45	75	80	70	160	150	160
VSSred (%)	90	86	77	77	62	40	39	24
Effl. Total VFA (mg/l)	55	75	80	130	165	300	385	895
Effl. Acetic Acid (mg/l)	25	30	35	60	80	190	215	530
Ef. Propionic Acid (mg/l)	30	35	35	45	70	90	100	220
Biogas (l/d)	1.9	3.9	6.1	9.1	4.4	8.8	13.6	20.6
Methane (m^3 CH_4/kg COD)	0.29	0.29	0.30	0.33	0.33	0.31	0.33	0.34

*Average values for the experimental runs

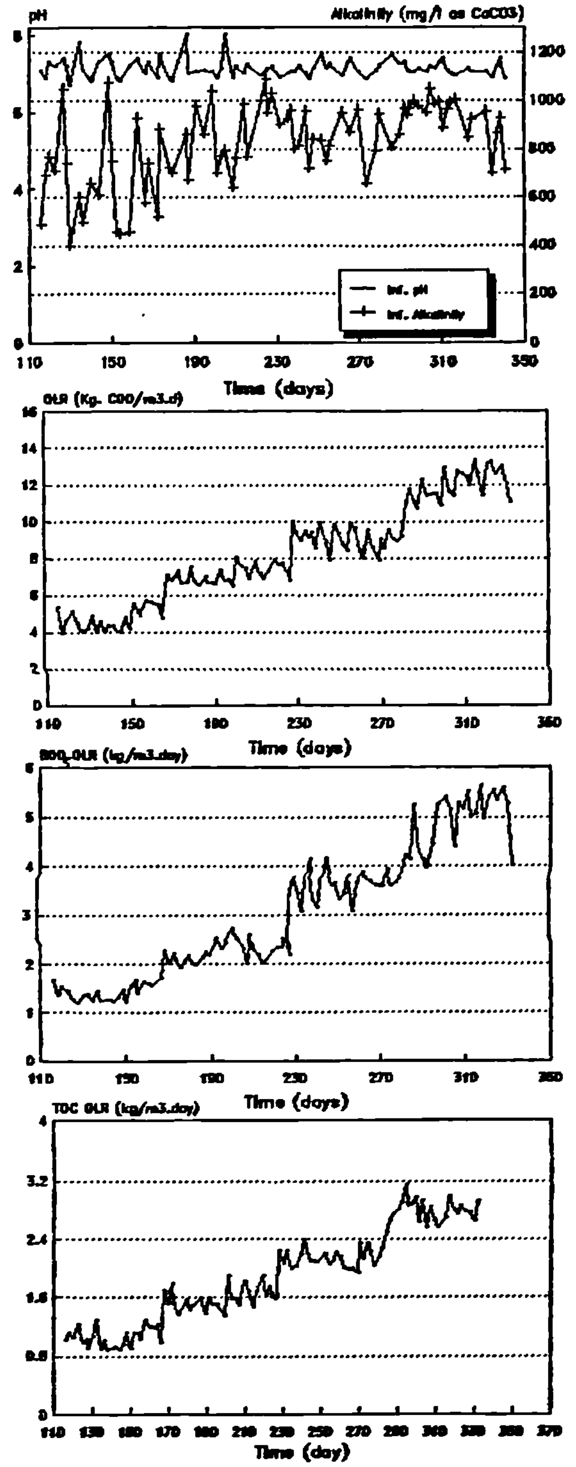


Figure 6.13: Influent pH & Alkalinity and Operational Conditions for RBC LS Steady State Performance

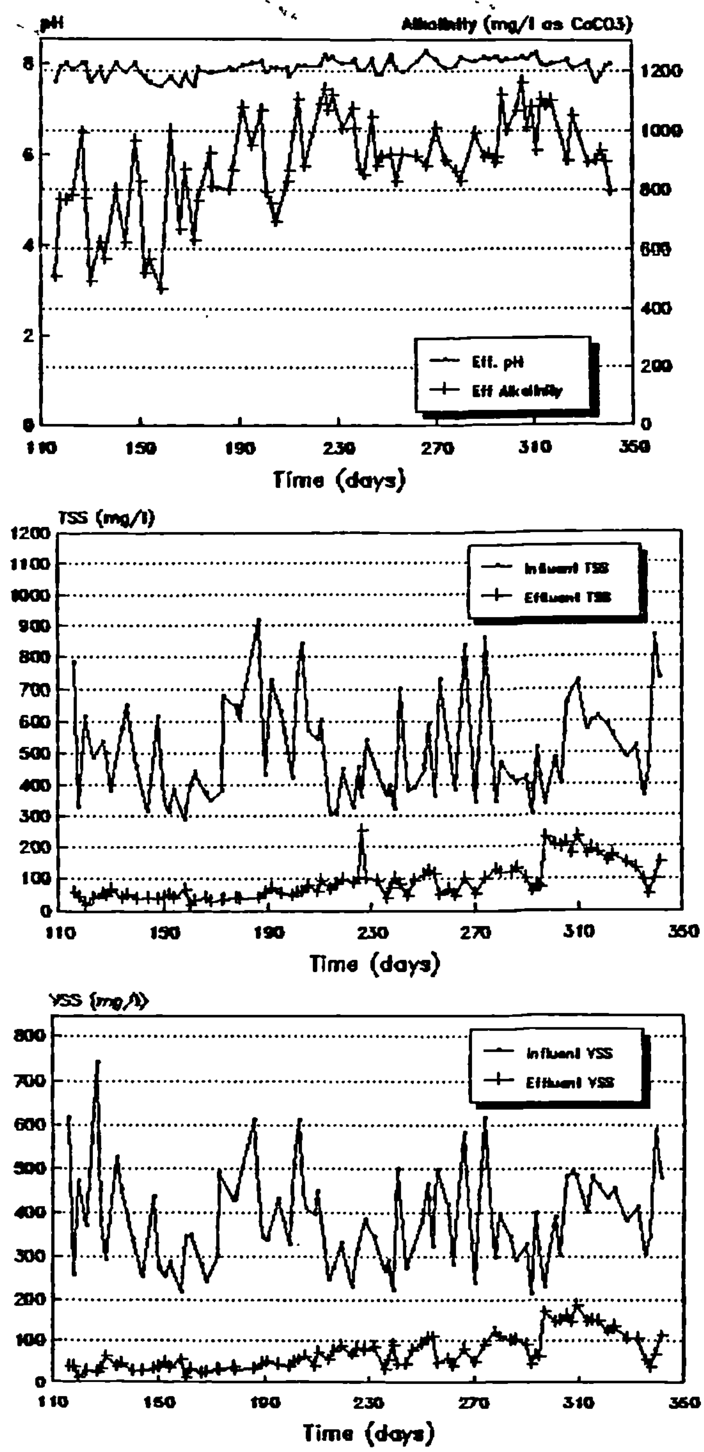


Figure 6.14: Effluent pH & Alkalinity, Influent and Effluent TSS & VSS for RBC LS Steady State Performance

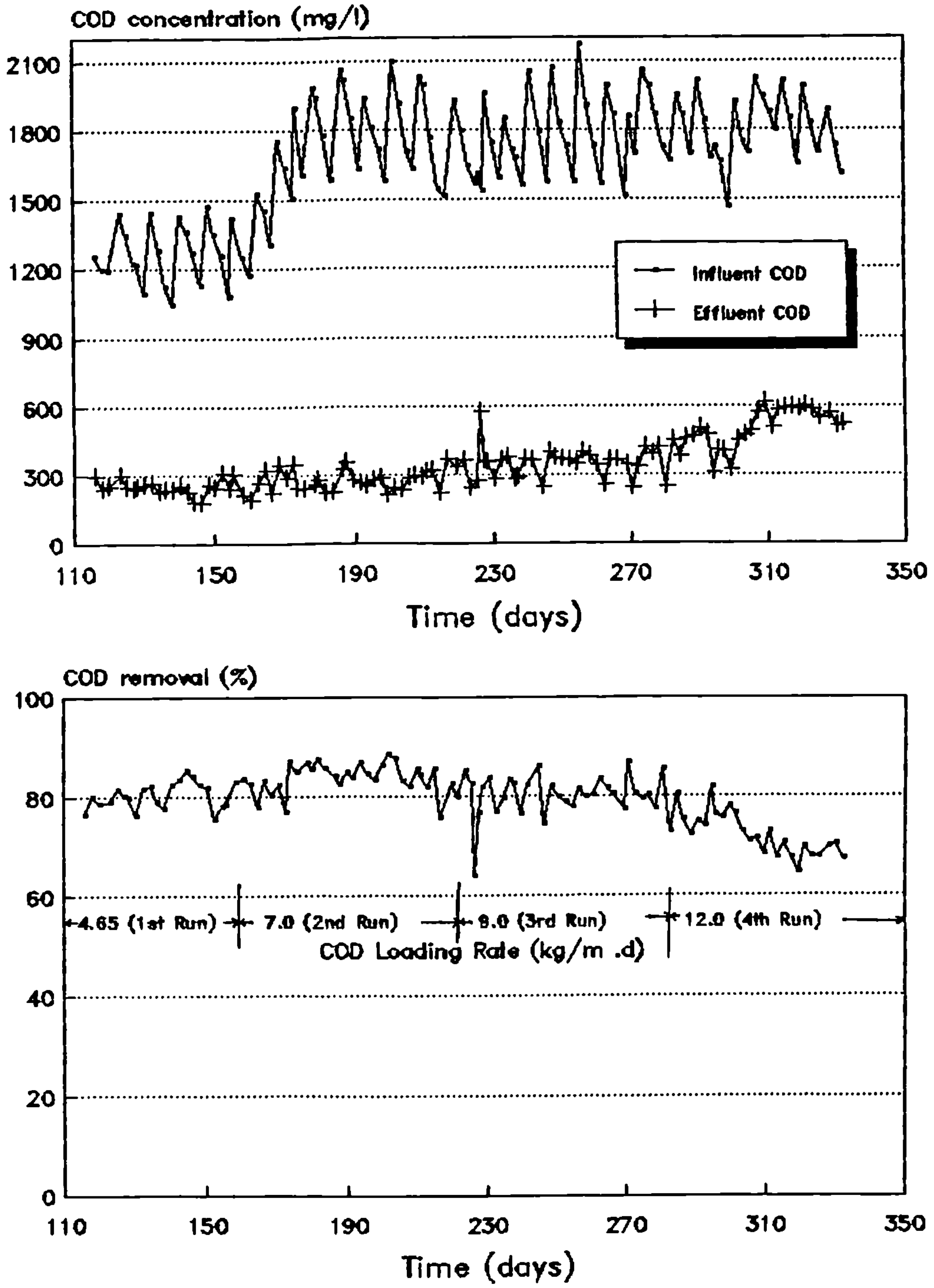


Figure 6.15: COD Concentrations and Removal for RBC LS Steady State Performance

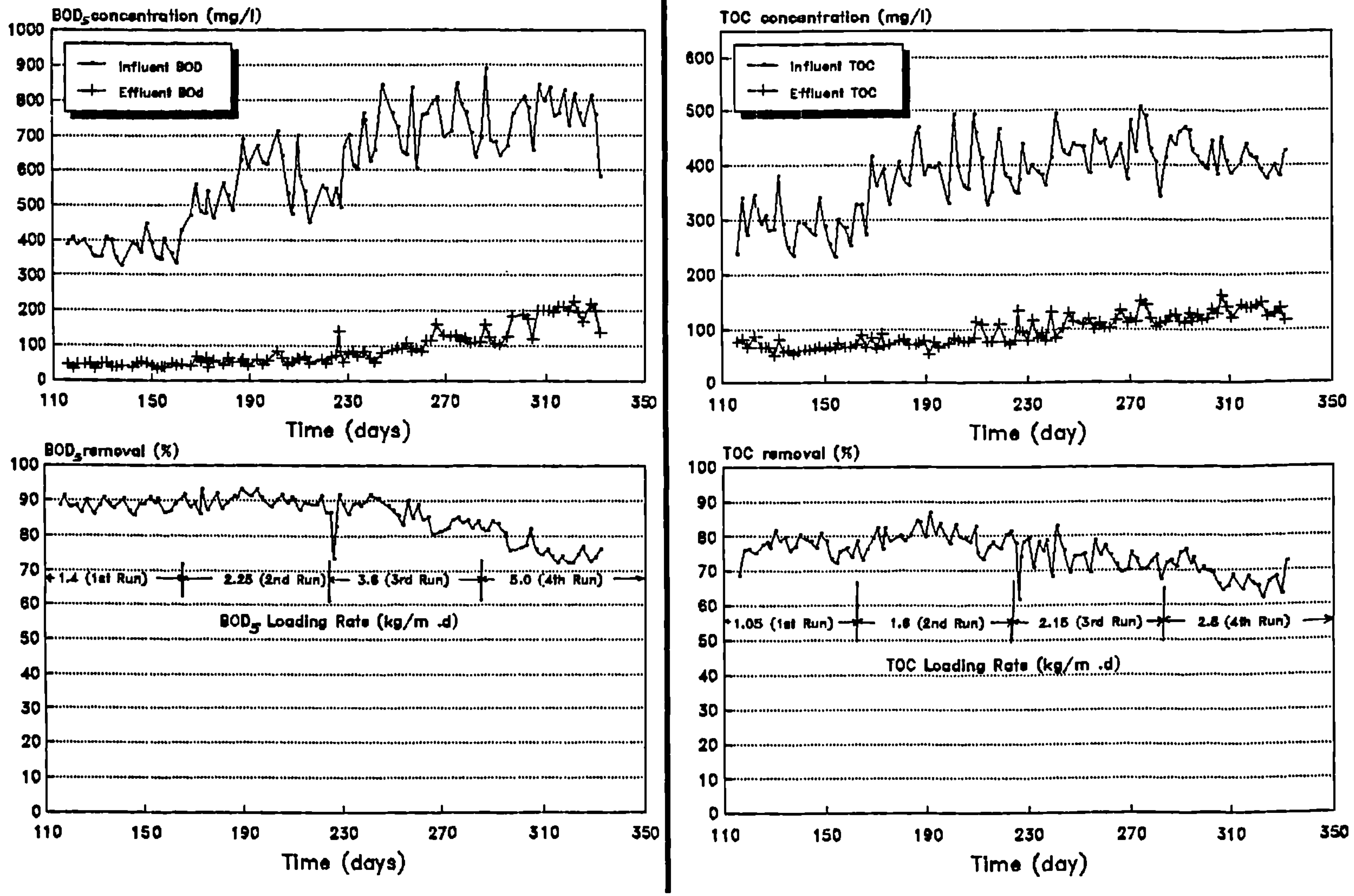


Figure 6.16: BOD_5 & TOC Concentrations and Removal for RBC LS Steady State Performance

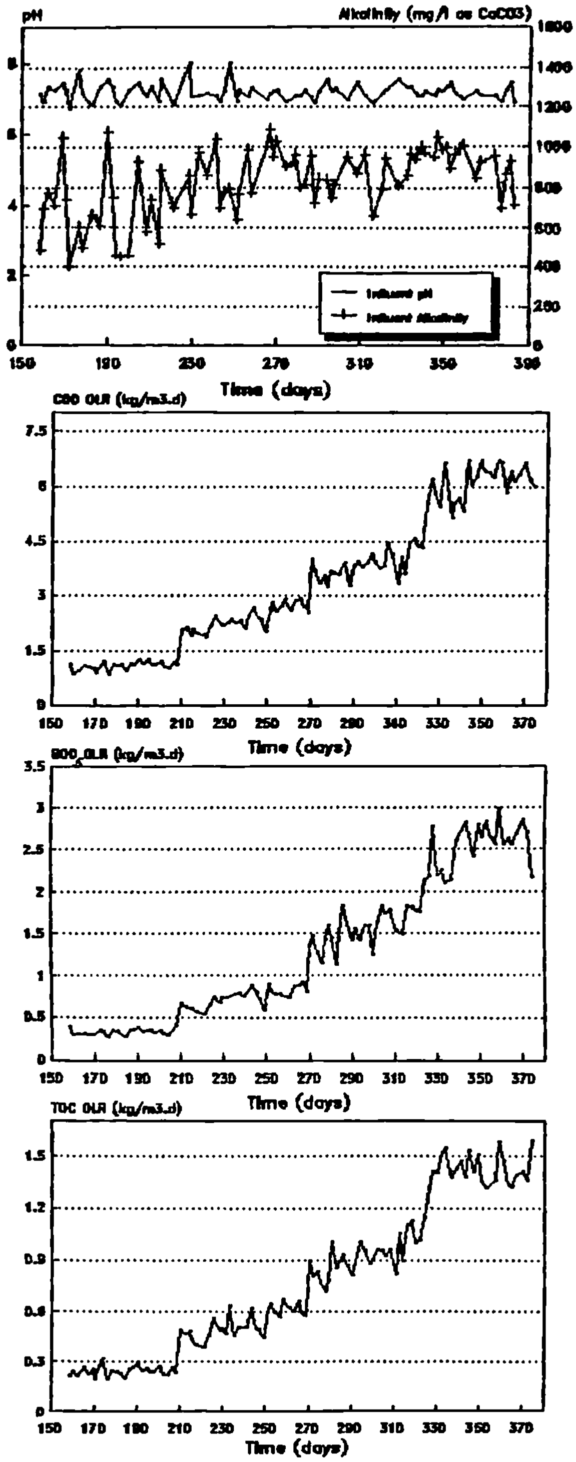


Figure 6.17: Influent pH & Alkalinity and Operational Conditions for UAF LS Steady State Performance

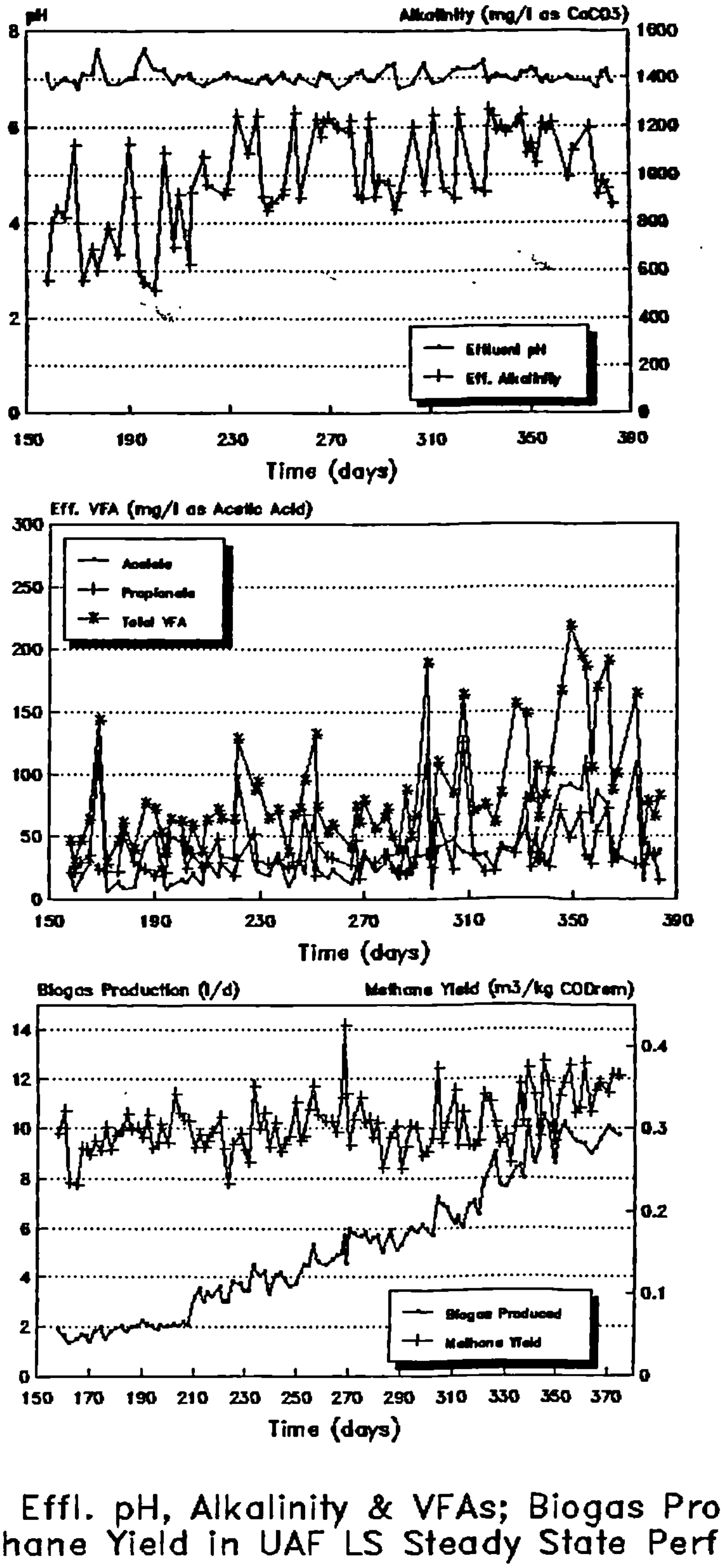


Figure 6.18: Effl. pH, Alkalinity & VFAs; Biogas Production and Methane Yield in UAF LS Steady State Performance

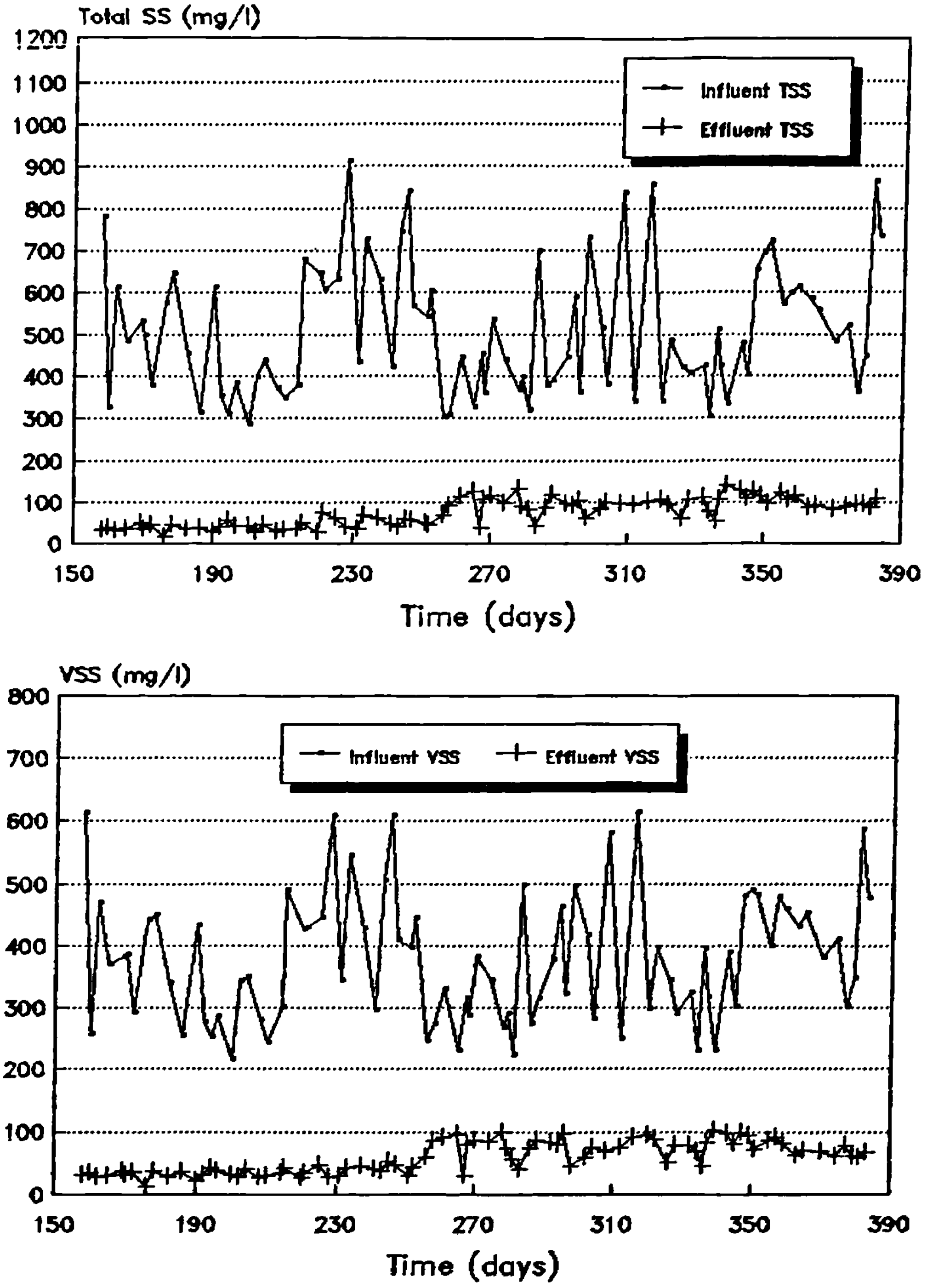


Figure 6.19: Influent and Effluent TSS & VSS for UAF LS Steady State Performance

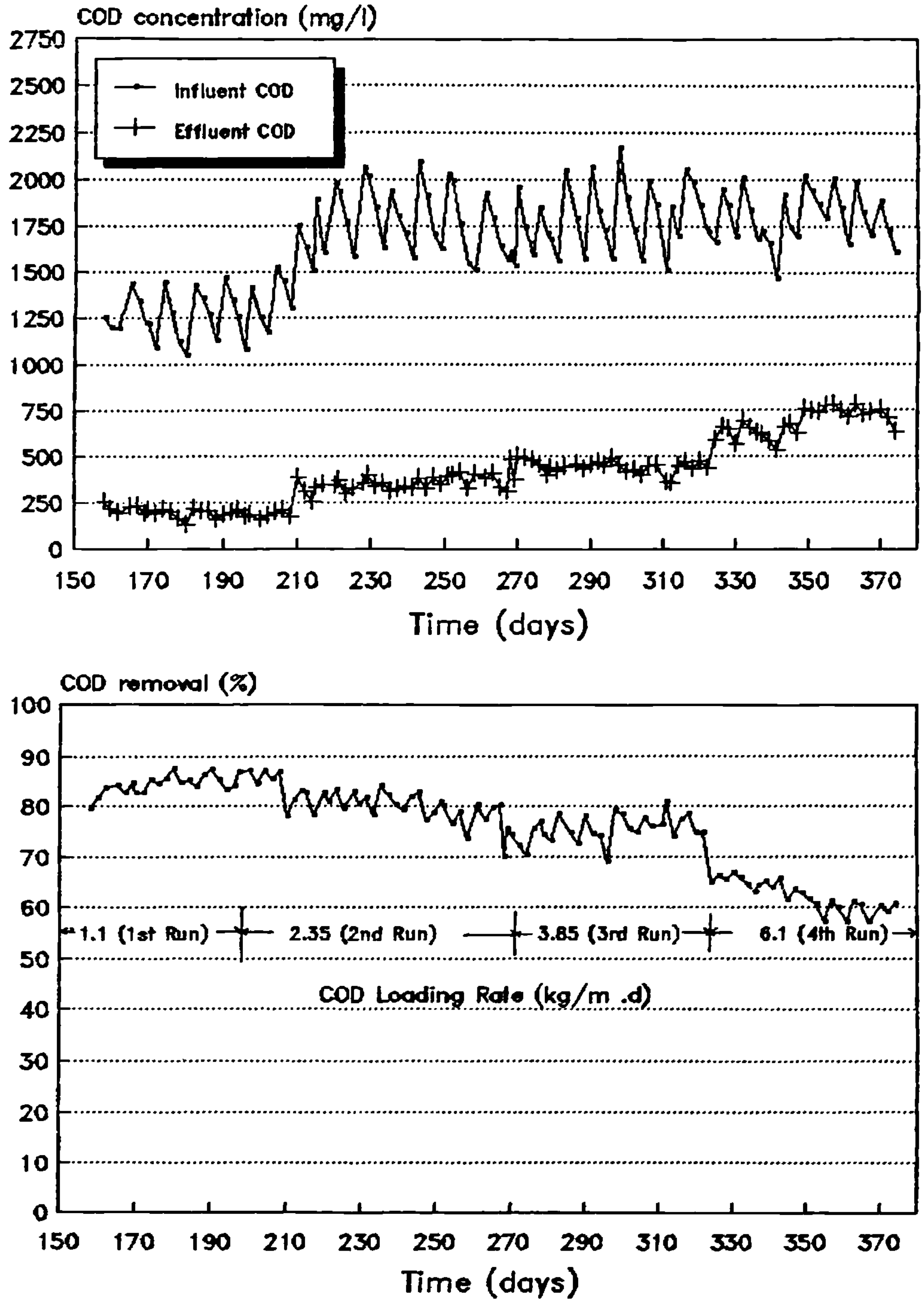


Figure 6.20: COD Concentrations and Removal for UAF LS Steady State Performance

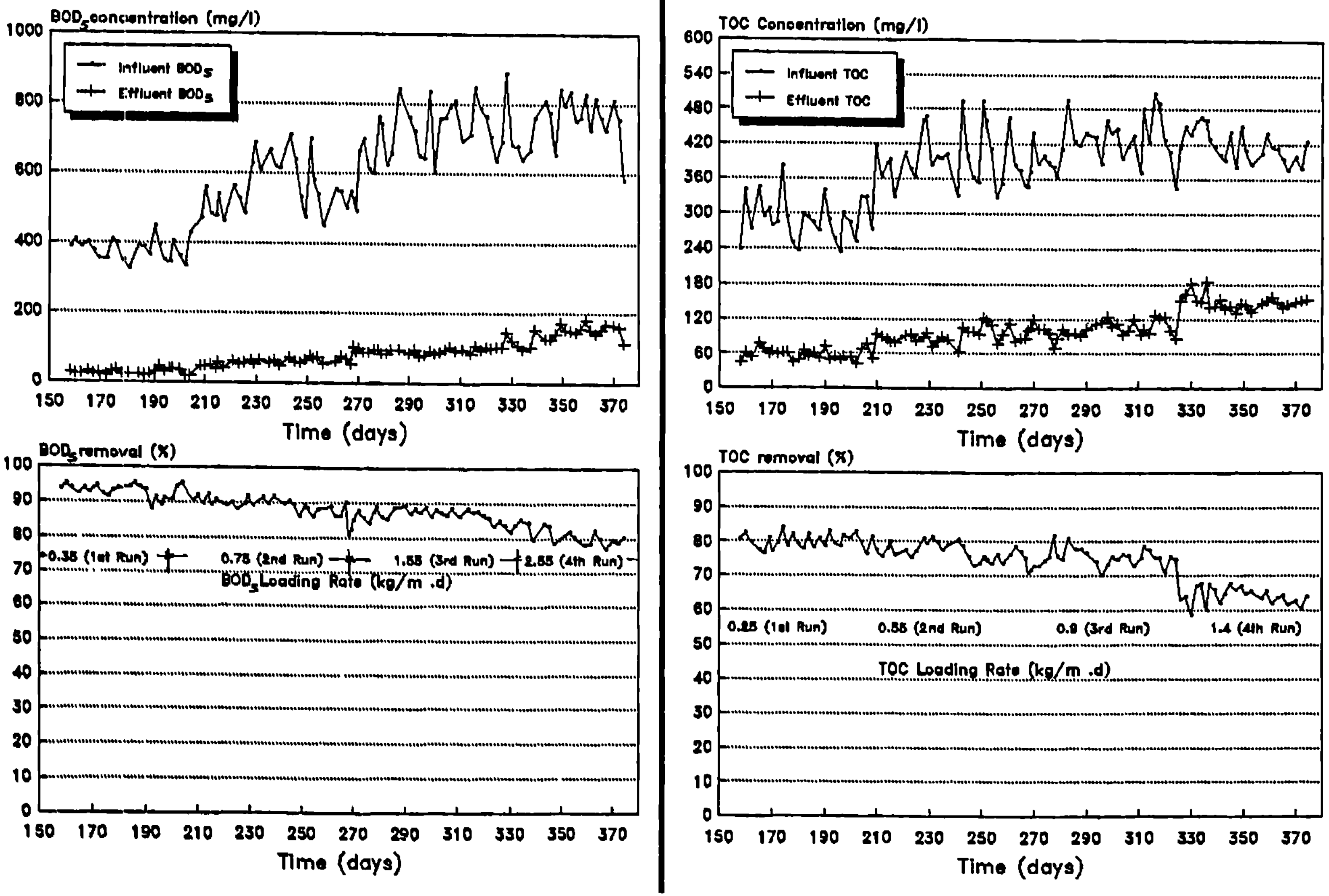


Figure 6.21: BOD_5 & TOC Concentration and Removal for UAF_{LS} Steady State Performance

When the OLRs of the RBC and the UAF reactors during the LS S-S analysis were taken into account, the overall performance of the RBC is comparatively higher than the UAF. Despite the seemingly low BOD_5 reduction in the RBC during the fourth run as compared to the UAF reactor, RBC still performed better than UAF. At 81% BOD_5 removal, the VOLR for the UAF was 2.5 kg BOD_5/m^3.day whereas BOD_5 removal of 86% to 90% were achieved by RBC for VOLR ranging between 2.2 to 3.6 kg BOD_5/m^3.day.

As far as biomass retention is concerned, both the RBC and the UAF seemed to be able to retain a significant amount of biological solids. Throughout the LS S-S experimental runs, the VSS which were removed together with the effluent were found to be between 32 mg/l to 121 mg/l (67% to 90% retention). Although the contents of TSS and VSS fluctuated up to around 900 mg/l, the effluent TSS and VSS were found to be low, as seen in Fig. 6.14 for the RBC and Fig. 6.19 for the UAF. The importance of biomass in fixed film bioreactors cannot be ignored. In this study, biomass retention in terms of TVS (attached and suspended) was determined in each reactor. Profiles of attached and suspended VSs were plotted to see the distribution. The biological solids accumulation will be discussed in detailed in Chapter 7.

- HS Steady State (HS S-S) Performance

In the HS S-S investigations, both the RBC and the UAF units exhibited a downward trend. The RBC and the UAF performances are illustrated in Figs. 6.22 to 6.25 and Figs. 6.26 to 6.30 respectively. Maximum substrate removals for both the RBC and the UAF units were

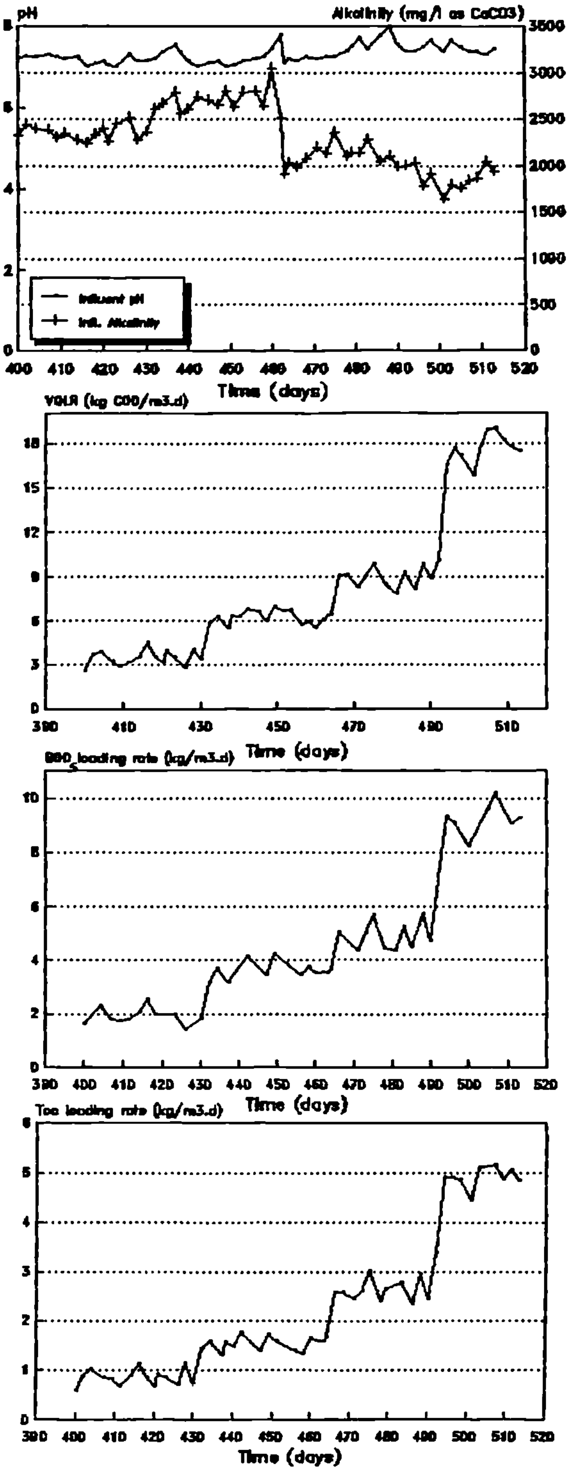


Figure 6.22: Influent pH & Alkalinity and Operational Conditions for RBC HS Steady State Performance

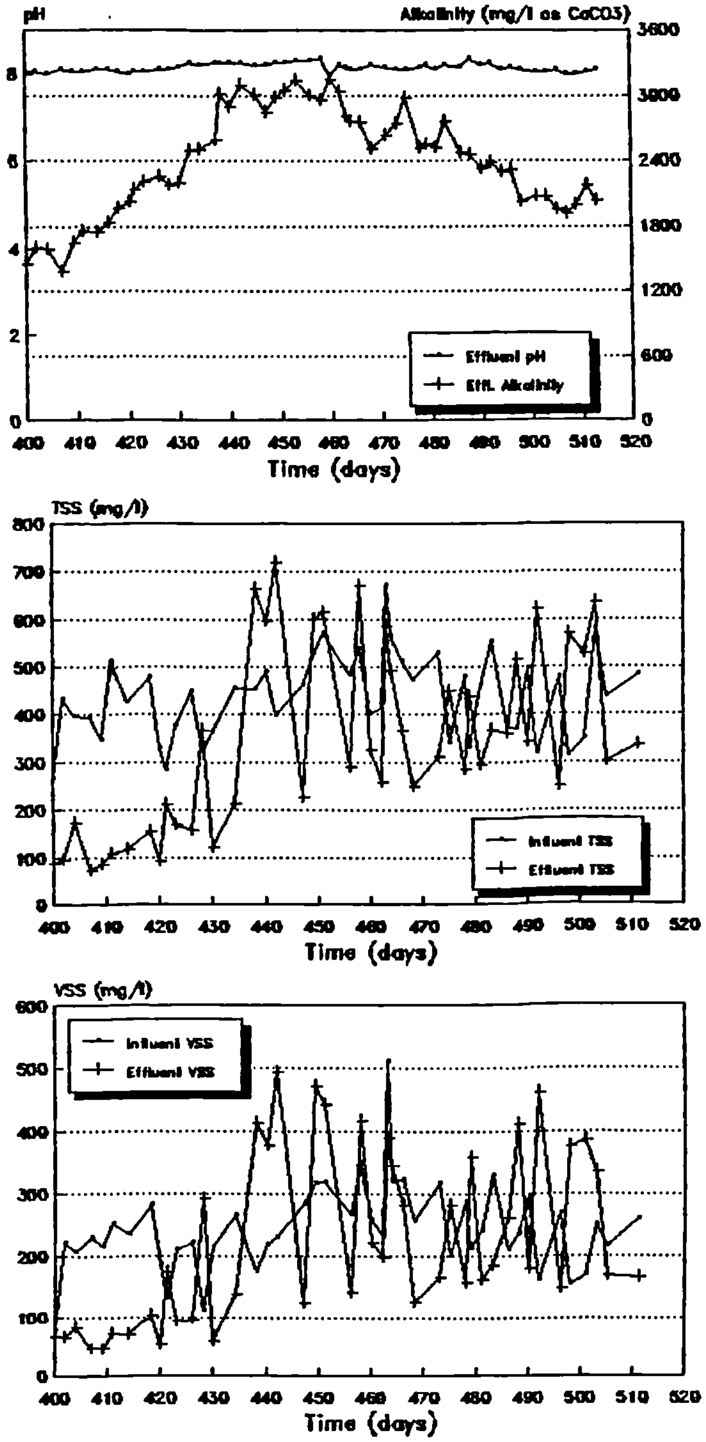


Figure 6.23: Eff. pH & Alkalinity, Inf. and Eff. TSS and VSS for RBC HS Steady State Performance

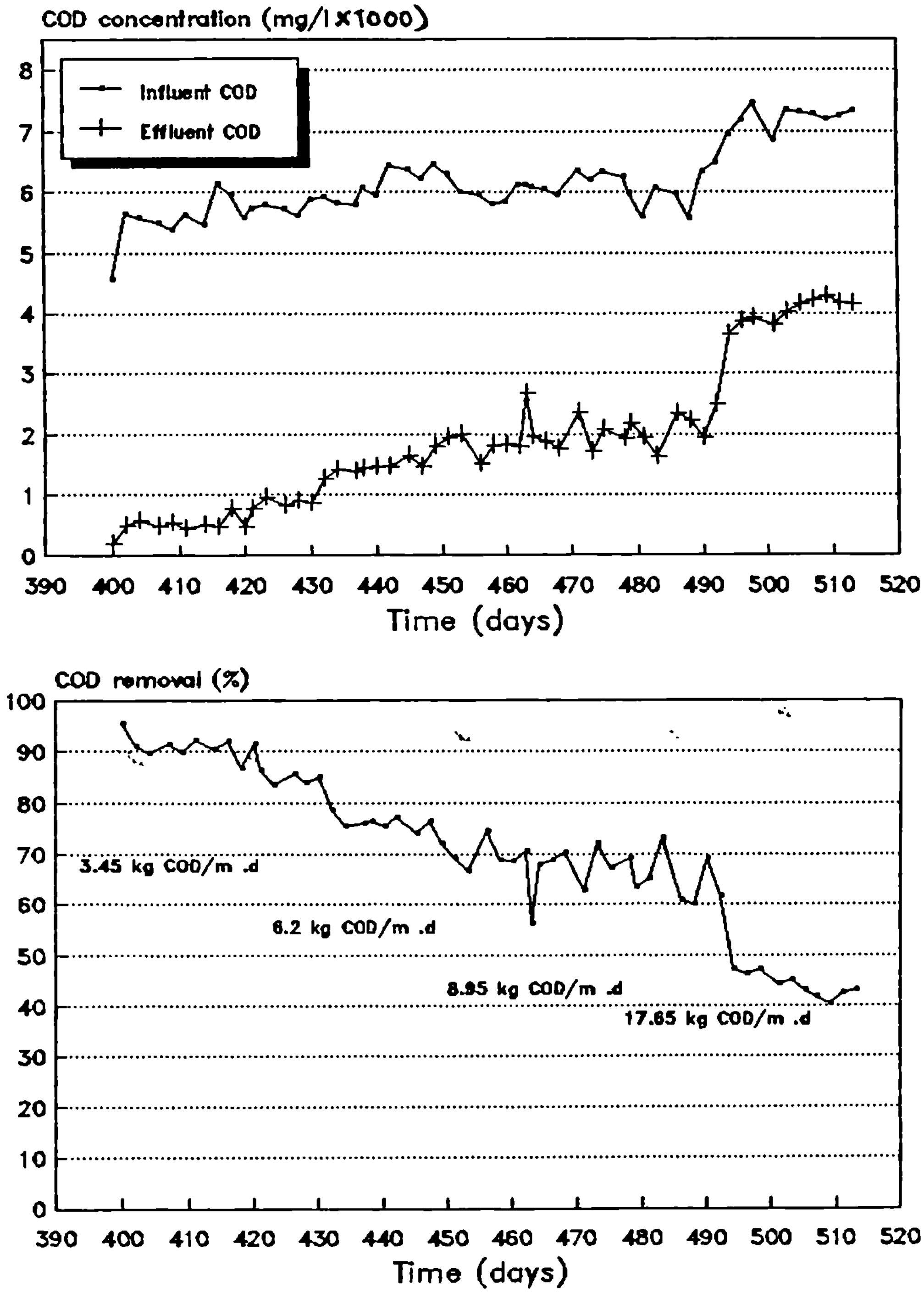


Figure 6.24: COD Concentrations and Removal for RBC HS Steady State Performance

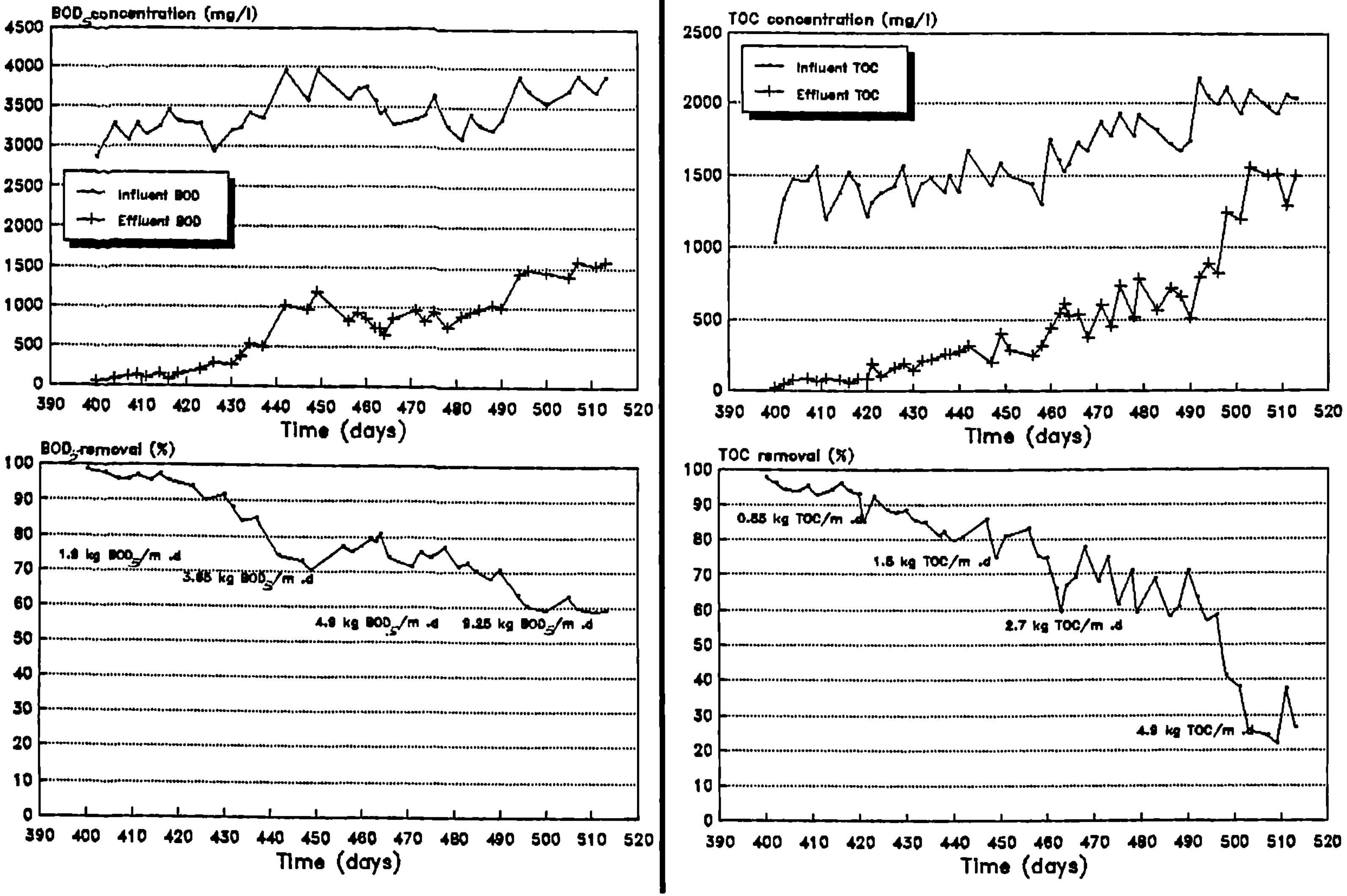


Figure 6.25: BOD$_5$ & TOC Concentrations and Removal for RBC HS Steady State Performance

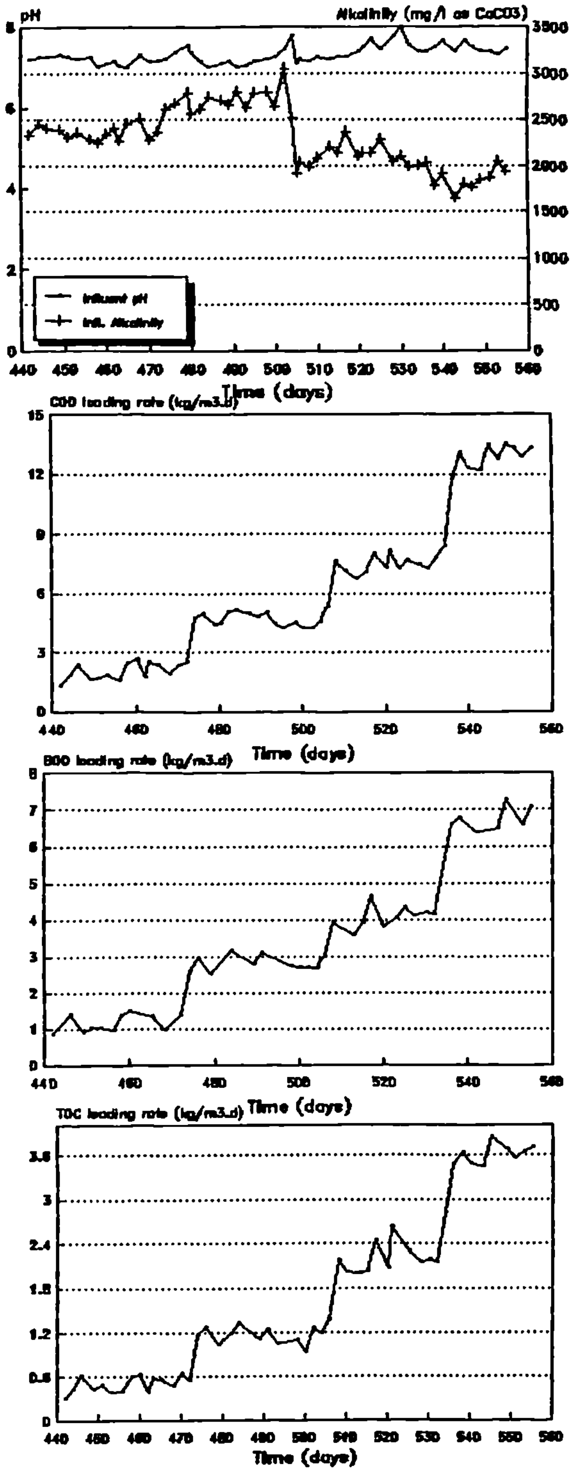


Figure 6.26: Influent pH & Alkalinity and Operational Conditions for UAF HS Steady State Performance

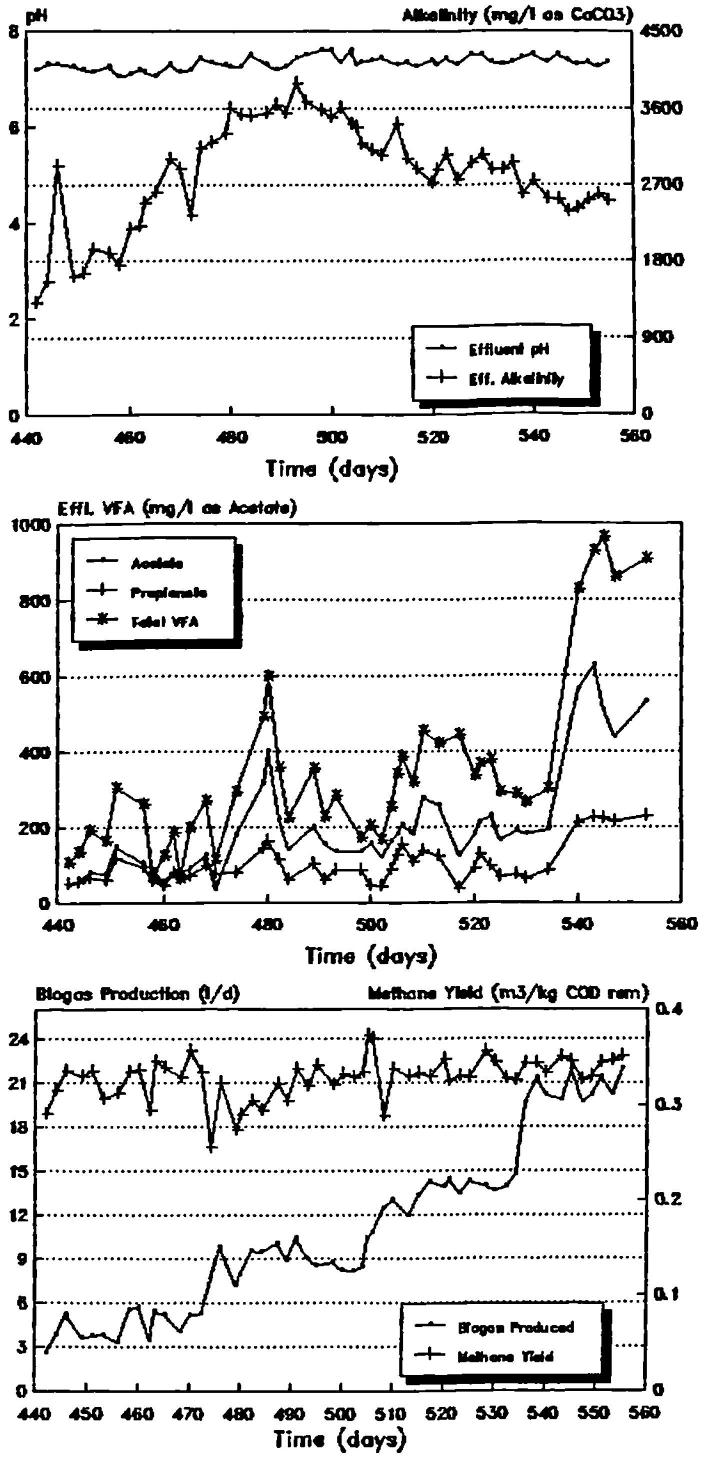


Figure 6.27: Eff. pH, ALkalinity & VFAs; Biogas Production and Methane Yield for UAF HS Steady State Performance

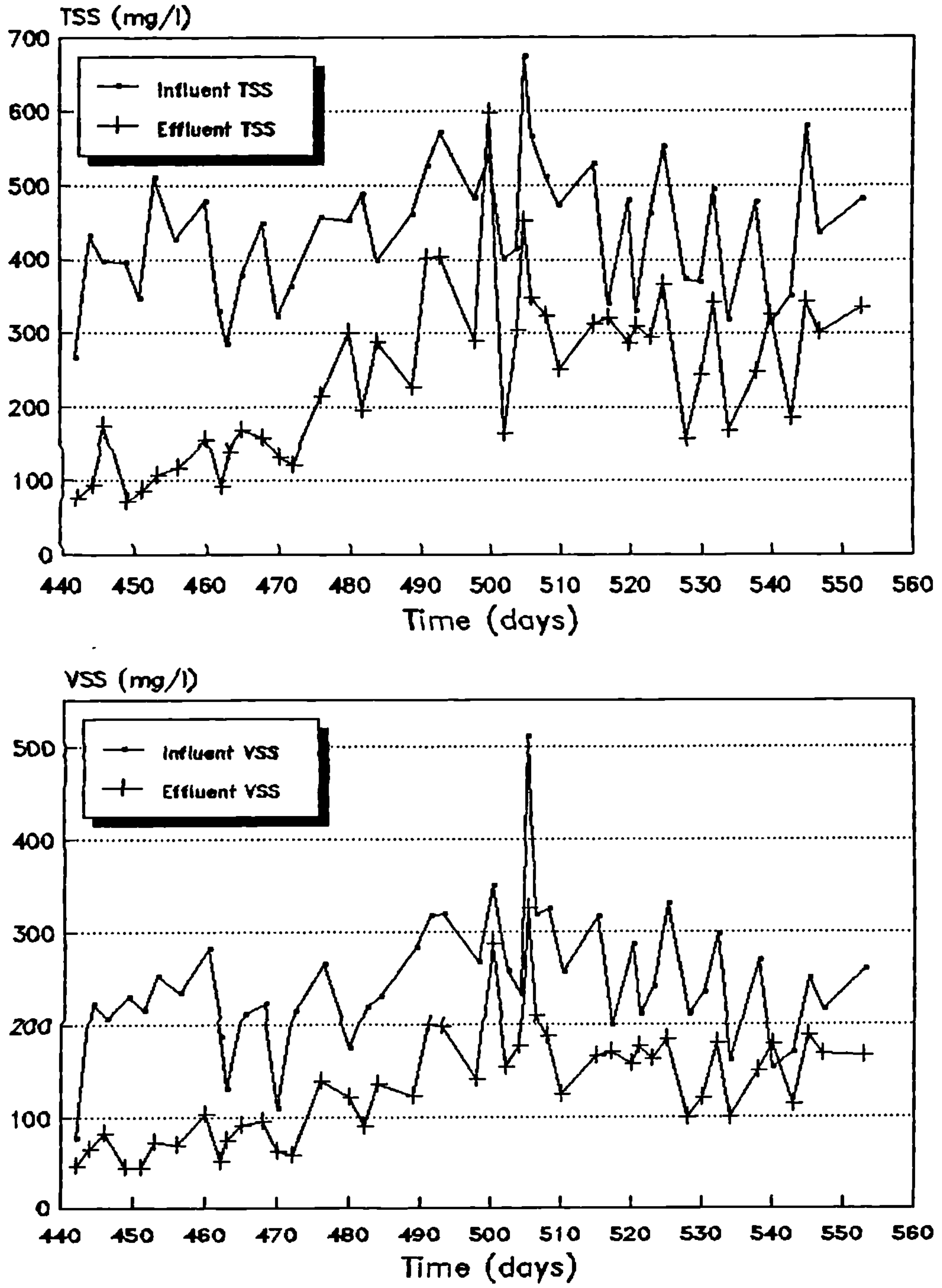


Figure 6.28: Influent and Effluent TSS & VSS for UAF HS Steady State Performance

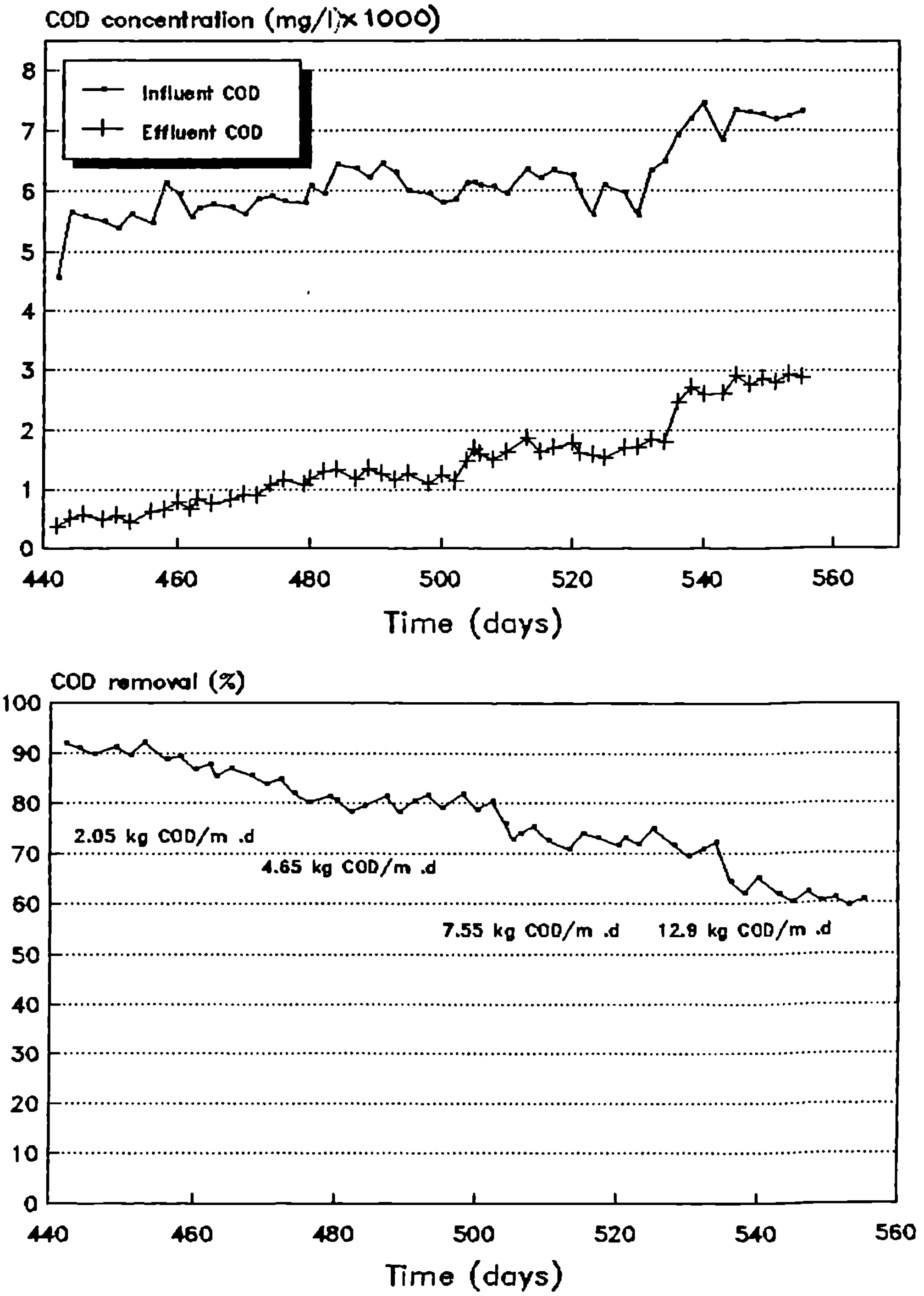


Figure 6.29: COD Concentrations and Removal for UAF HS Steady State Performance

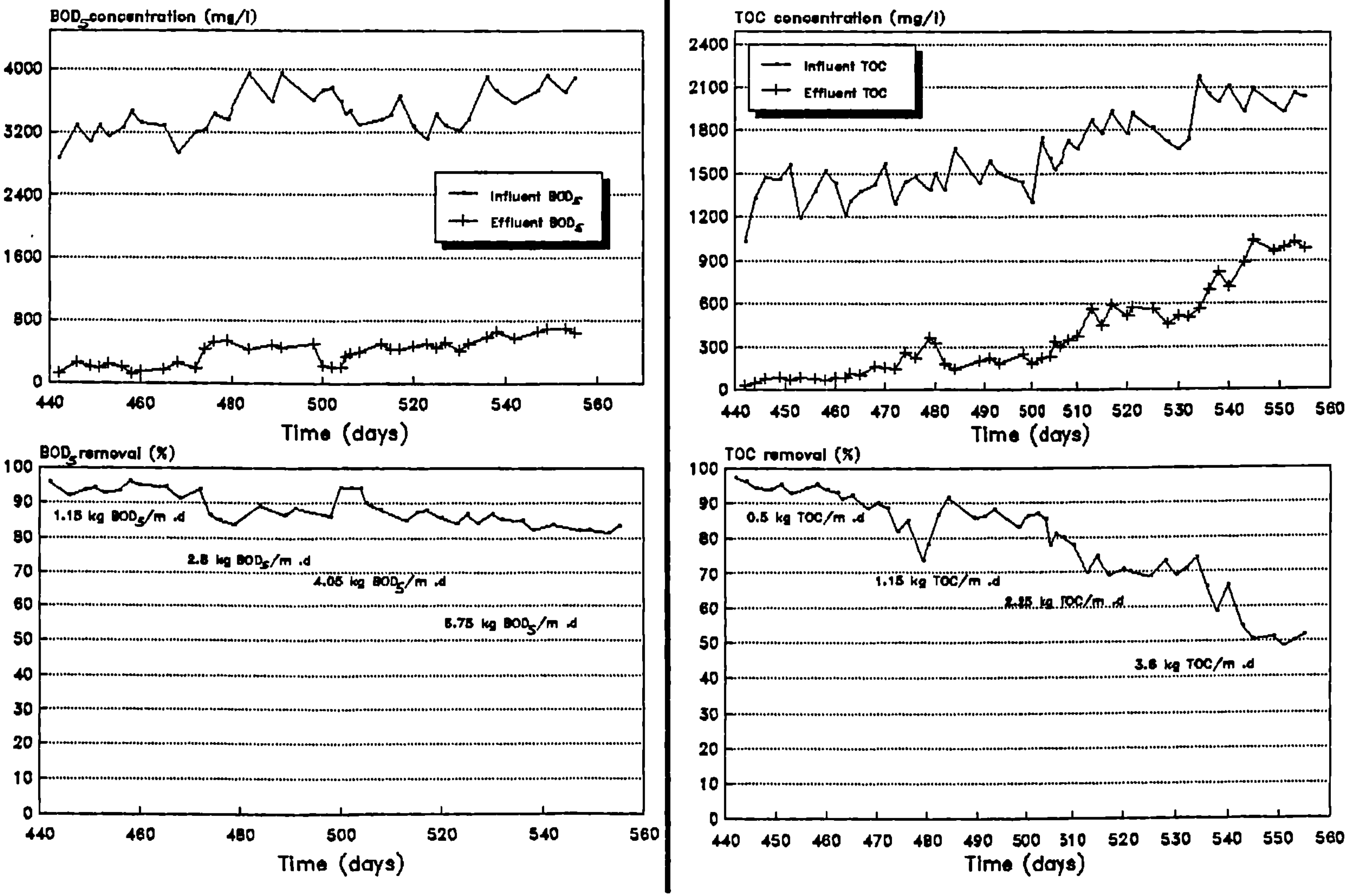


Figure 6.30: BOD_5 & TOC Concentrations and Removal for UAF HS Steady State Performance

achieved in the first run (refer to Tables 6.11 and 6.12). In this first run, the RBC reactor yielded an average COD reduction of 89% (or 95% as BOD_5 and 93% as TOC removals); whereas the UAF reactor yielded an average COD reduction of 88% (or 94% as BOD_5 and 93% as TOC). Despite these initial high removal rates for both the RBC and the UAF reactors in the first run, the substrate removals in the RBC reactor decreased more rapidly from those in the UAF reactor in the subsequent runs (see Figs. 6.24 and 6.25 for the RBC; and Figs. 6.29 and 6.30 for the UAF).

The performances of the RBC and the UAF during the HS S-S conditions gave opposite results from those during LS S-S conditions. The UAF performed much better than the RBC, where at the same VOLR of 10 kg COD/m^3.day the average COD removal in RBC was 61% while that of the UAF was 65% (refer to Figs 6.31b and 6.34b).

The biological solids wash-out increased in both the RBC and the UAF with the increase in OLR. The increase in VSS removal during the HS S-S condition was more drastic than those during the LS S-S since with high OLR (or the reduction in HRT), the flowrate increases. Although biological solids removal increased in both the RBC and the UAF, both behaved differently. The RBC performance deteriorated more rapidly due to more attached biomass being sloughed off from the drum, whereas the increase in biological solids removal did not affect the UAF to any great extent judging from the satisfactory performance based on substrate removal.

- Effect of OLR and HRT on RBC and UAF Steady State Performances

Organic loading rate has been recognized as the major factor

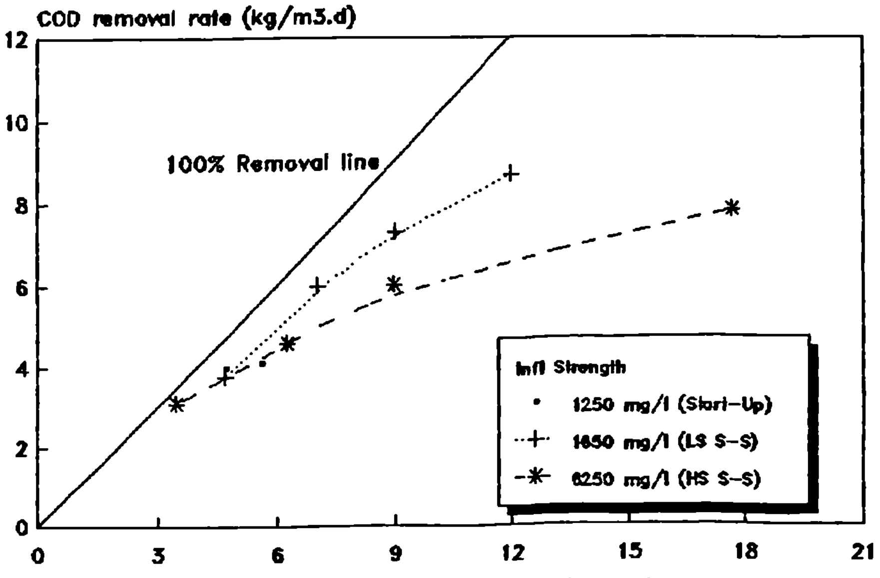


Figure 6.31a: COD VORR versus VOLR for RBC Steady State (S-S) Performance

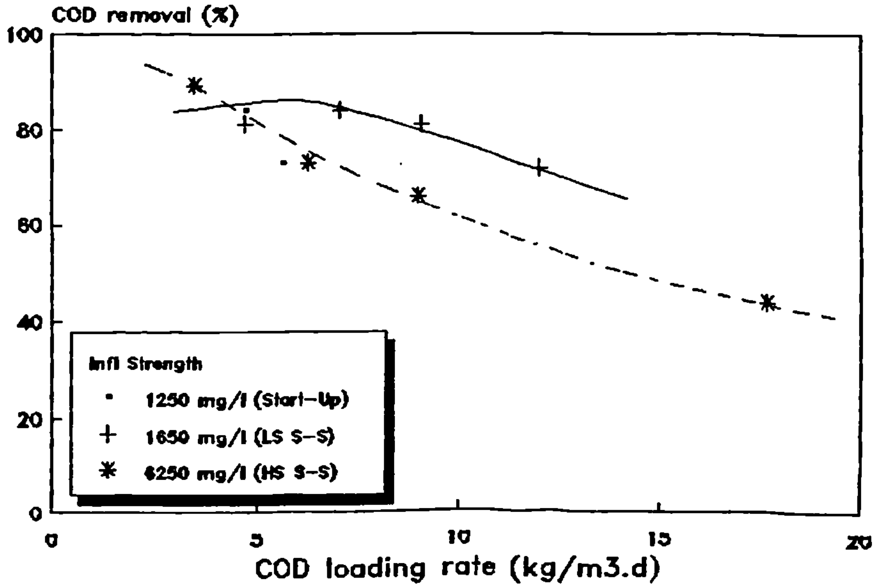


Figure 6.31b: COD Removal versus VOLR for RBC Steady State (S-S) Performance

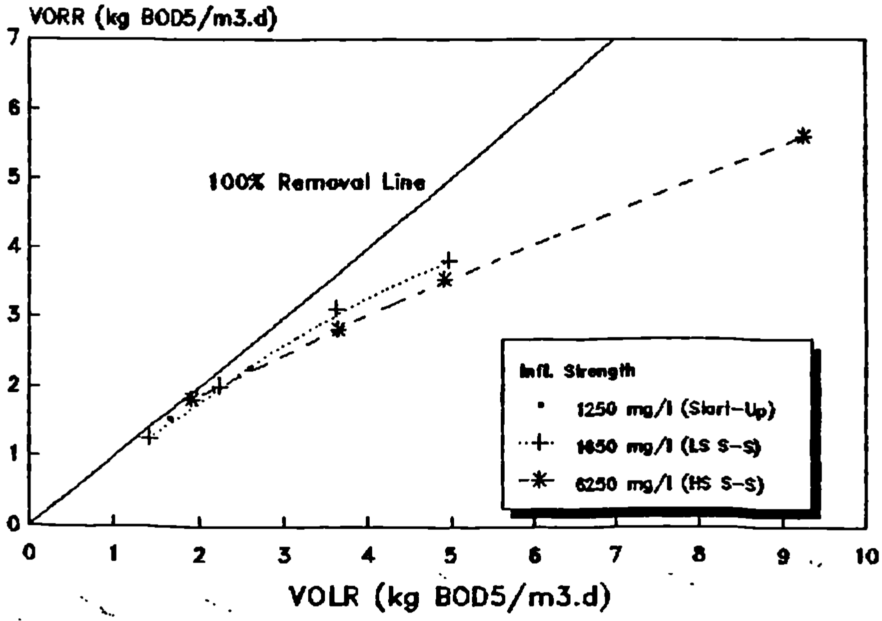


Figure 6.32a: BOD_5 VORR versus VOLR for RBC Steady State (S-S) Performance

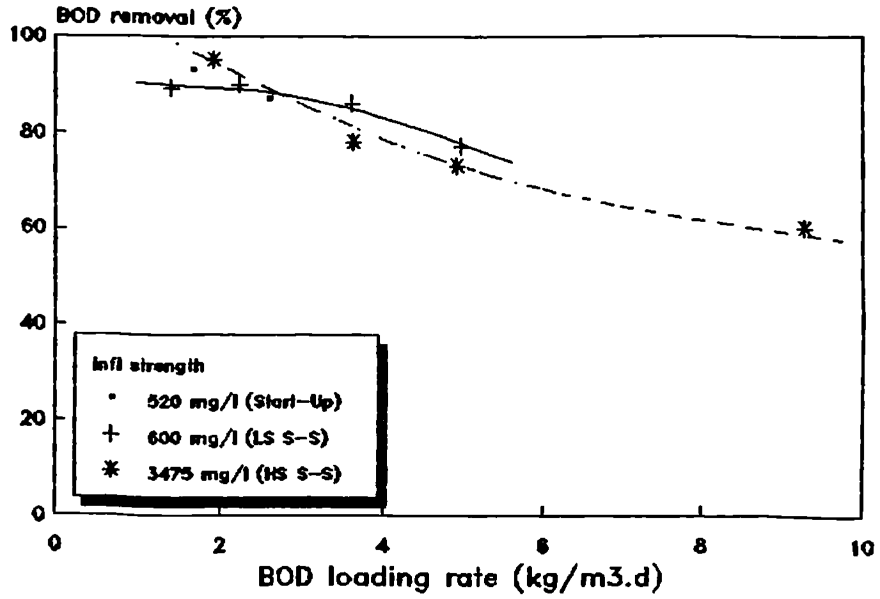


Figure 6.32b: BOD_5 Removal versus VOLR for RBC Steady State (S-S) Performance

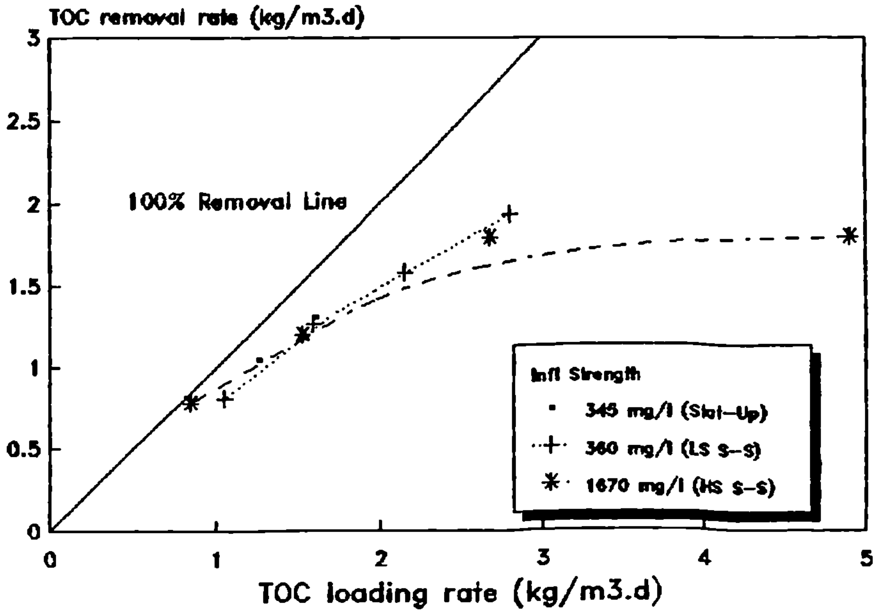


Figure 6.33a: TOC VORR versus VOLR for RBC Steady State (S-S) Performance

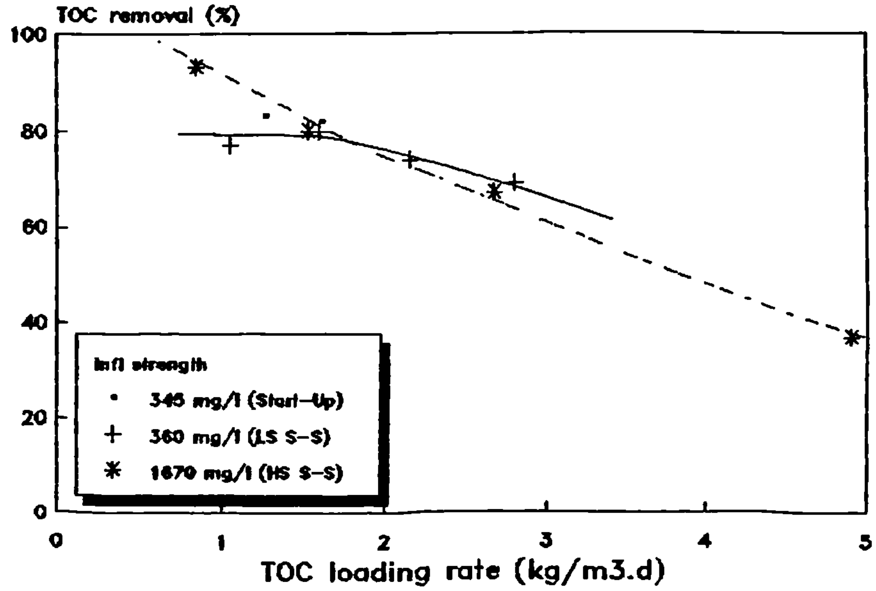


Figure 6.33b: TOC Removal versus VOLR for RBC Steady State (S-S) Performance

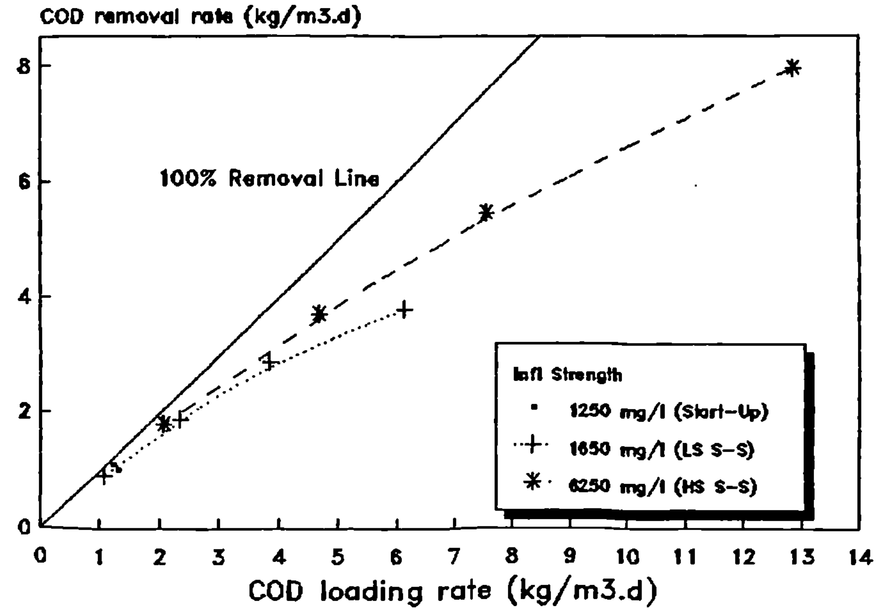


Figure 6.34a: COD VORR versus VOLR for UAF Steady State (S-S) Performance

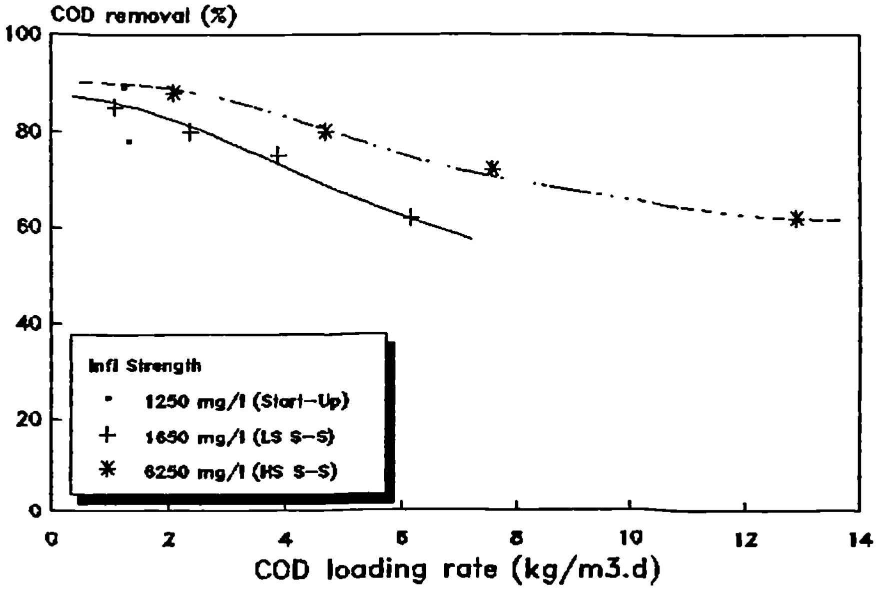


Figure 6.34b: COD Removal versus VOLR for UAF Steady State (S-S) Performance

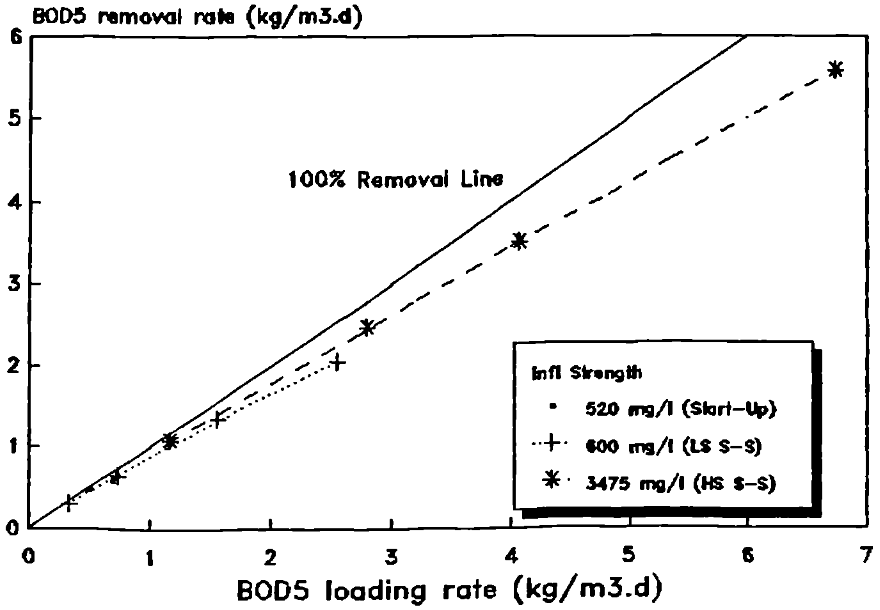


Figure 6.35a: BOD_5 VORR versus VOLR for UAF Steady State (S–S) Performance

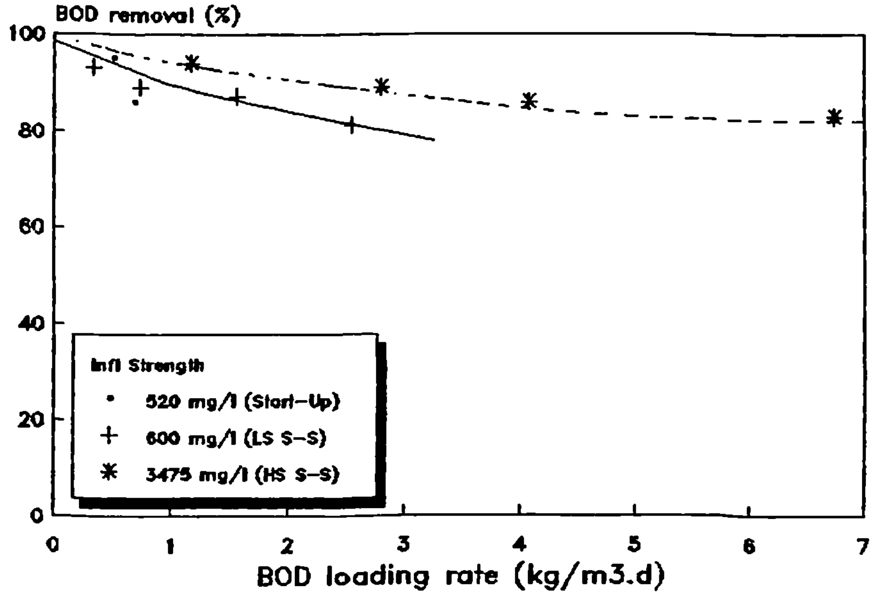


Figure 6.35b: BOD_5 Removal versus VOLR for UAF Steady State (S–S) Performance

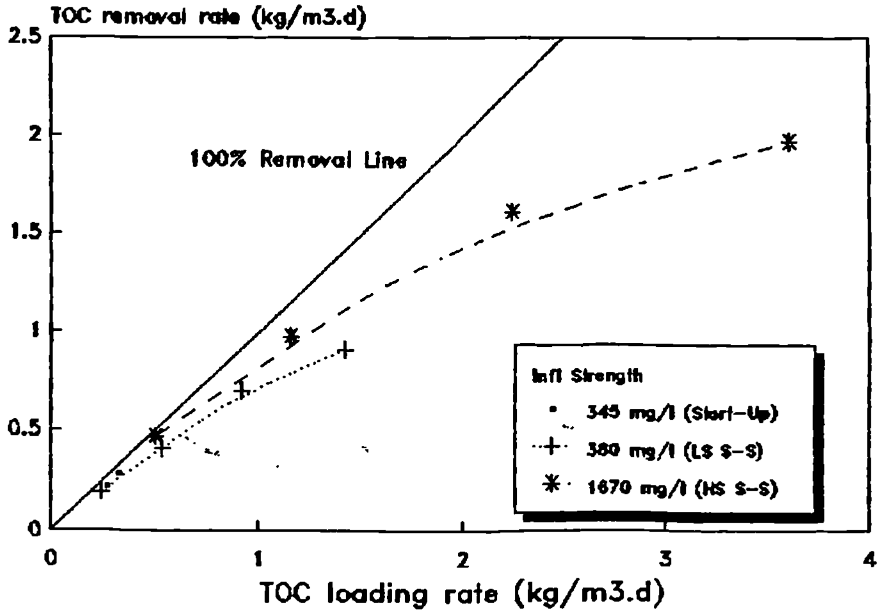


Figure 6.36a: TOC VORR versus VOLR for UAF Steady State (S-S) Performance

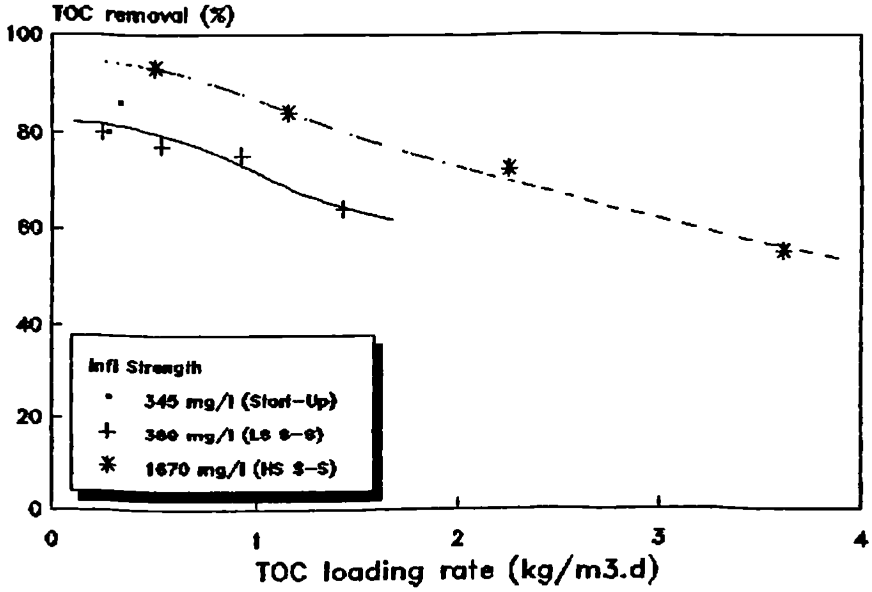


Figure 6.36b: TOC Removal versus VOLR for UAF Steady State (S-S) Performance

affecting treatment efficiency. Both the RBC and the UAF units seem to possess their own response to OLR. For this study, the effects of VOLR on VORR and substrate removals for the RBC and the UAF reactors are shown in Figs. 6.31 to 6.33 and Figs. 6.34 to 6.36 respectively.

Apparently, from the plotted graphs, the VOLR was found to affect treatment efficiency in three distinct stages. This finding was also observed by Echaroj (1986) in his study of an anaerobic RBC. At low VOLRs, constantly high substrate reduction for both the LS and the HS steady state conditions were achieved, thus giving VORR as a linear (first order) function of VOLR (refer to Figs 6.31a, 6.32a and 6.33a for the RBC and 6.34a, 6.35a and 6.36a for the UAF).

From observations based on the HS S-S performance study, the RBC reactor yielded a more or less constant maximum COD reduction of 90% (BOD_5 and TOC removals of 93%) for a VOLR of up to 5 kg COD/m^3.day (Fig. 6.31b) or 2 kg BOD_5/m^3.day (Fig. 6.32b) or 0.8 kg TOC/m^3.day (Fig. 6.33b).

In the case of the UAF, the highest VOLR that can be applied to yield a constant maximum COD removal of about 95% (BOD_5 removal of 98% or TOC removal of 93%) was 1 kg COD/m^3.day (Fig. 6.34b) or 0.3 kg BOD_5/m^3.day (Fig. 6.35b) or 0.1 kg TOC/m^3.day (Fig. 6.36b). This results in this study using HS leachate are comparable to the results achieved by Choi and Burkhead (1985) who reported a COD removal of 95% at 37°C with a VOLR of 0.96 kg COD/m^3.day (at an HRT of 5 days) having an influent concentration of 4800 mg/l.

Secondly, a further increase in the VOLR beyond 5 kg COD/m^3.day for the RBC and beyond 1 kg COD/m^3.day for the UAF resulted in a decrease in treatment efficiency as seen by the departure of the VORR line from the theoretical 100% removal line. At this stage, the VORR was no longer linearly related to the VOLR.

Finally, as the VOLR was further increased, a constant VORR was achieved which indicated that the organic removal rate at this point was no longer dependent on the VOLR and the rate limiting phase became apparent (i.e a zero order function). It can also be seen that substrate removal decreased with increasing OLR. Although BOD_5 reduction at the end of the experimental runs did not appear to reach this rate limiting phase, the phase can be seen developing from analysis of COD removal (Fig. 6.31a for the RBC and Fig. 6.34a for the UAF) and more distinctly from the TOC removal investigation (Fig. 6.33a for the RBC and Fig. 6.36a for the UAF).

The effect of HRT on substrate removals in the RBC and the UAF are illustrated in Figs 6.37 and 6.38 respectively. The relationships obtained show that substrate removal efficiencies increased with the increase in HRT irrespective of substrate concentrations but it can be seen that the RBC and the UAF behaved differently with respect to the rate of substrate removals.

During a period of varying HRT at the same VOLR but with different leachate strength, a greater percentage removal was noted in the UAF reactor treating HS leachate which was associated with a longer HRT and higher biomass retention. The UAF treating the LS leachate was loaded at

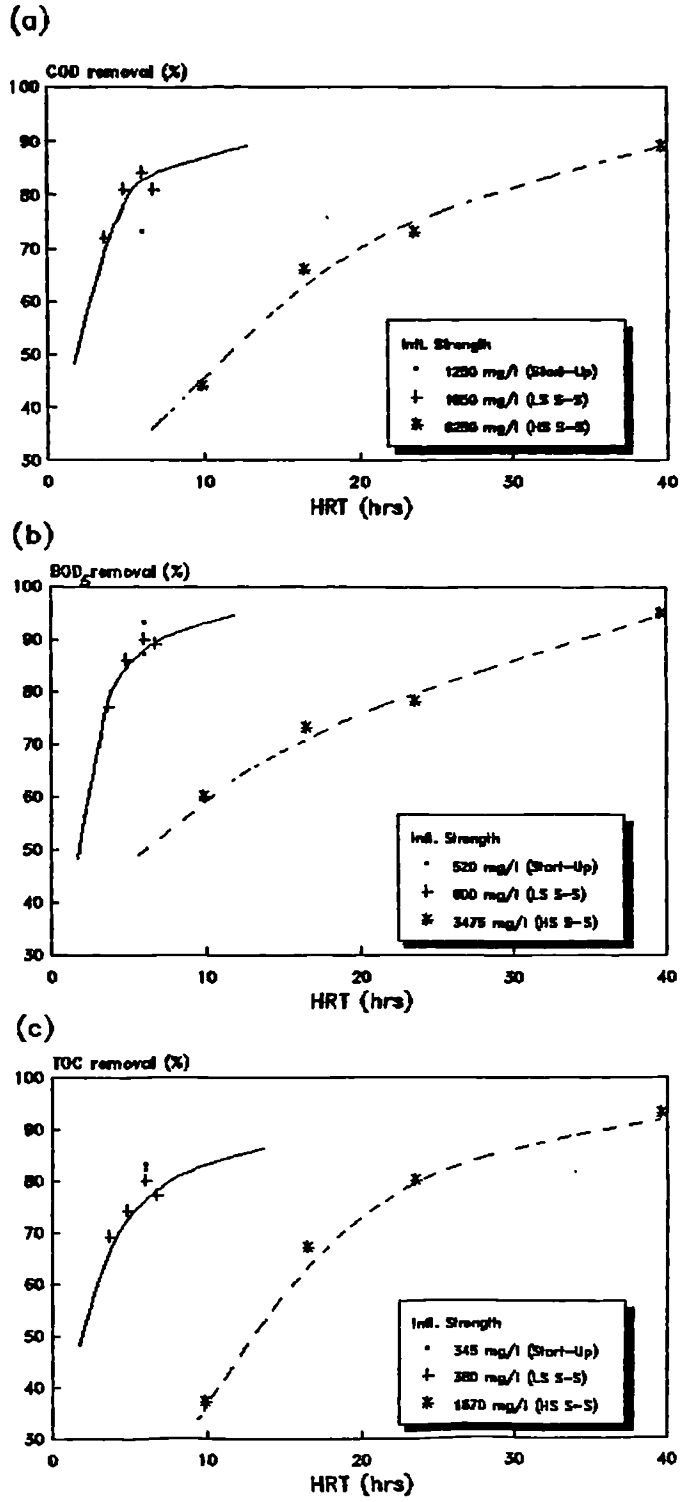


Figure 6.37: Effect of HRT on (a) COD (b) BOD_5 and (c) TOC in RBC during Steady State (S–S) Performance

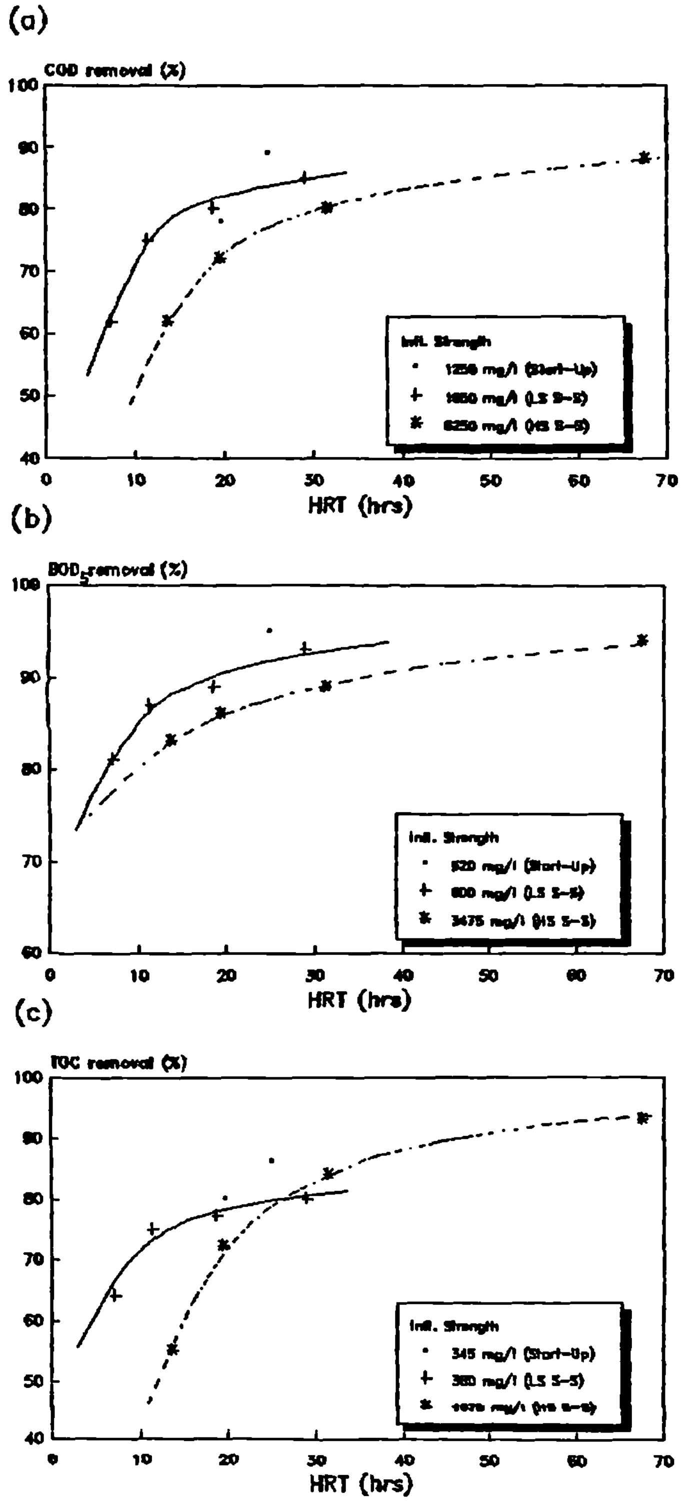


Figure 6.38: Effect of HRT on (a) COD (b) BOD_5 and (c) TOC in UAF during Steady State (S–S) Performance

up to 6 kg COD/m^3.day before any appreciable loss in efficiency was recorded, whereas when the HS leachate was used a higher treatment efficiency was maintained up to a loading rate of 13 kg COD/m^3.day. Evidently, the UAF treating the LS leachate resulted in higher substrate removals at identical HRTs (or lower substrate removals at similar VOLRs). It was further noted that the LS leachate exhibited a more pronounced response to changes in HRT or OLR than the HS leachate, as seen from the rapid decreased in the substrate removal efficiencies especially at HRTs shorter than 8 hours. The UAF treating the LS leachate at HRTs shorter than 8 hours was probably overloaded as seen from the high concentration of volatile fatty acids in the effluent.

In contrast, comparing the steady state performance of HS leachate with the performance using LS leachate in the RBC, and at the same OLR, the LS leachate exhibited a higher treatment efficiency even though the HRT is much shorter. This is due to the inability of the RBC to cope with the high substrate concentration introduced to the reactor. Norton (1984) in his study found that at higher organic loadings the food-to-microorganisms ratio (F/M) had an increasing influence on performance. In order to yield a BOD$_5$ removal in excess of 90% irrespective of wastewater concentrations, the F/M ratio should not exceed 0.6 g BOD$_5$/g attached VS.day. Hence a much longer HRT (or lower OLR) is needed for HS leachate in order to reduce the F/M ratio to an acceptable value.

Despite the decrease in treatment efficiency in both the RBC and the UAF however, it is evident from the experimental runs that the reactors were able to absorb a doubling of the OLR and a corresponding reduction in HRT with a minimum effect on the effluent substrate concentration.

Clearly then, both organic loading and hydraulic loading rates are important when considering waste treatment by fixed film reactors.

6.3.4 Substrate Utilization Kinetics for RBC and UAF Treatment Processes

Although not stated as a specific objective of these studies, a kinetic description of the removal of the soluble organic fraction during leachate treatment was attempted. Such an effort may be worthwhile in that it can provide a further insight into those factors governing the operational processes as well as allowing some rational indication of system scale-up requirements and/or limitations specific to the wastewater of concern.

Primarily the mathematical description of the substrate removal rate in a treatment process is developed for modelling and predicting substrate removal and treatment efficiency. Various documented mathematical models involving a steady state mass balance with basic first order or Monod-type hyperbolic rate equations have been used to describe the kinetic of substrate removal. Although a general model could be mathematically developed, a comparison of the various models is made difficult by the fact that the curve fitting constants obtained are usually apply to a particular reactor and/or a particular substrate.

Hudson et al (1976) used modified Monod-type hyperbolic rate equations in their study to describe a kinetic expression of the RBC reactor treating shellfish processing wastewater. On the other hand, from literature reviews, some researchers have concluded that substrate removal for an RBC system appears to follow a first order kinetic pattern (Stover and Kincannon, 1976; Friedman et al, 1976). The removal of substrate by microorganisms in the RBC process was based on monomolecular kinetics with substrate utilization expressed as a function of the mass substrate loading rate as in Equation 3.20.

Similar empirical relationships can also be applied to anaerobic systems. Stover and Gonzalez (1988) reported that accurate prediction and modelling of both treatment performance and methane production have been accomplished when substrate utilization and methane production were expressed as functions of the mass substrate loading rate for both suspended and fixed film systems. Extensive studies by Stover et al (1984) using anaerobic reactors, have shown the reliability of the kinetic expression.

The effect of the mass substrate loading on the substrate utilization of the RBC is illustrated in Figs. 6.39a, 6.40a and 6.41a, while the effect on the UAF performance is shown in Figs. 6.42a, 6.43a and 6.44a. The plotted results were of the same form as those obtained by several other researchers (Kincannon and Stover, 1982; Stover and Gonzalez, 1988; Hamoda and Wilson, 1989).

The main feature in the graphs was the gradual loss in efficiency with increasing loading rate with a distinct point of departure illustrating the existence of two phases. The point of departure from a linear graph, with the exception of that for BOD_5 removal in the UAF (Fig. 6.43a), was noticeable in all the other graphs especially for COD and TOC removals. It is therefore apparent that a further increase in loading could have been applied to the UAF and still maintain a satisfactory BOD_5 removal.

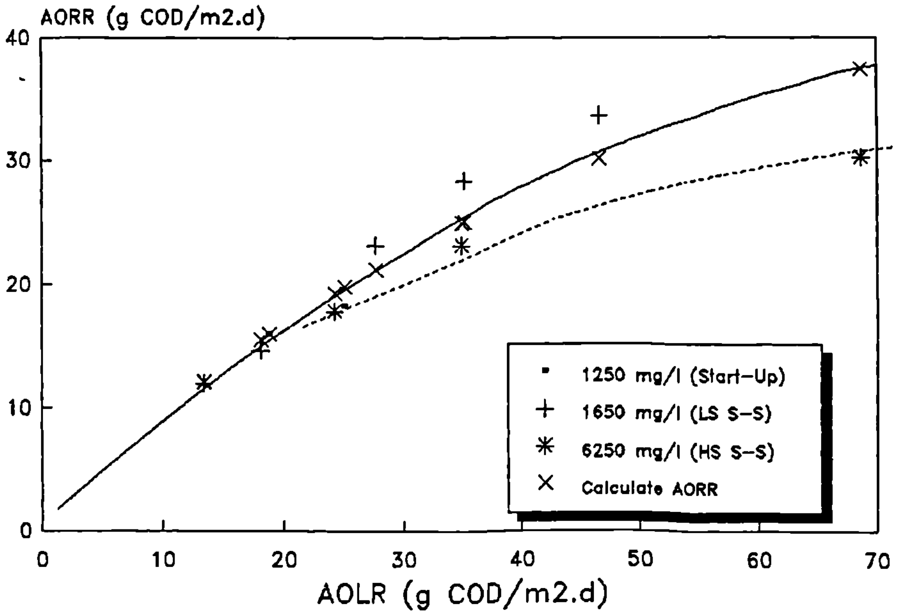


Figure 6.39a: COD AORR versus AOLR for RBC during Steady State (S-S) Performance

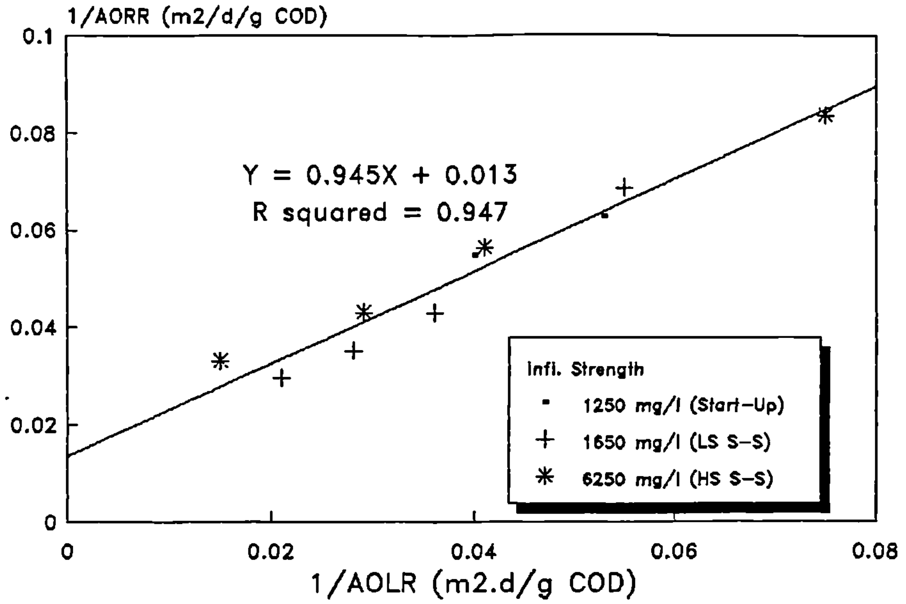


Figure 6.39b: 1/AORR versus 1/AOLR for RBC during Steady State (S-S) Performance

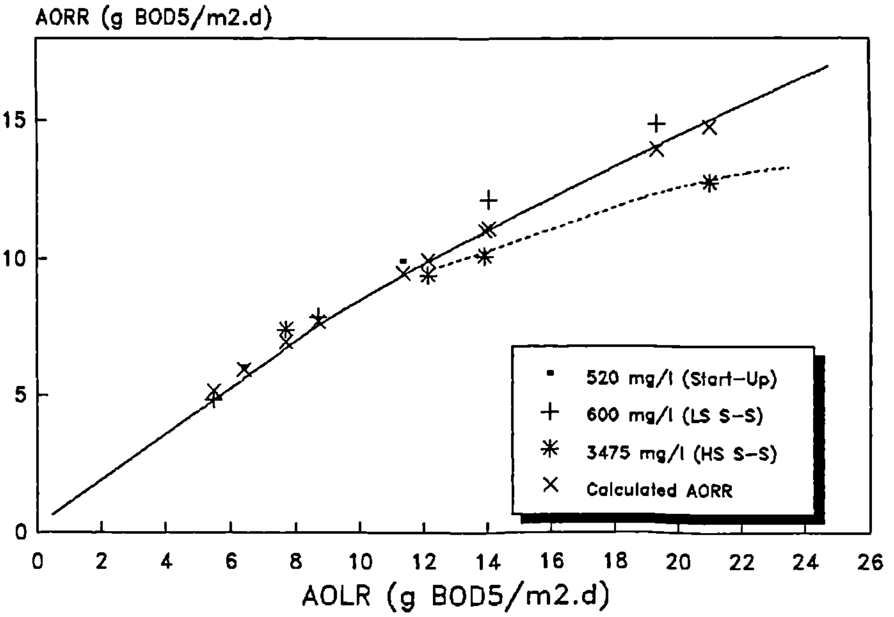


Figure 6.40a: BOD_5 AORR versus AOLR for RBC during Steady State (S–S) Performance

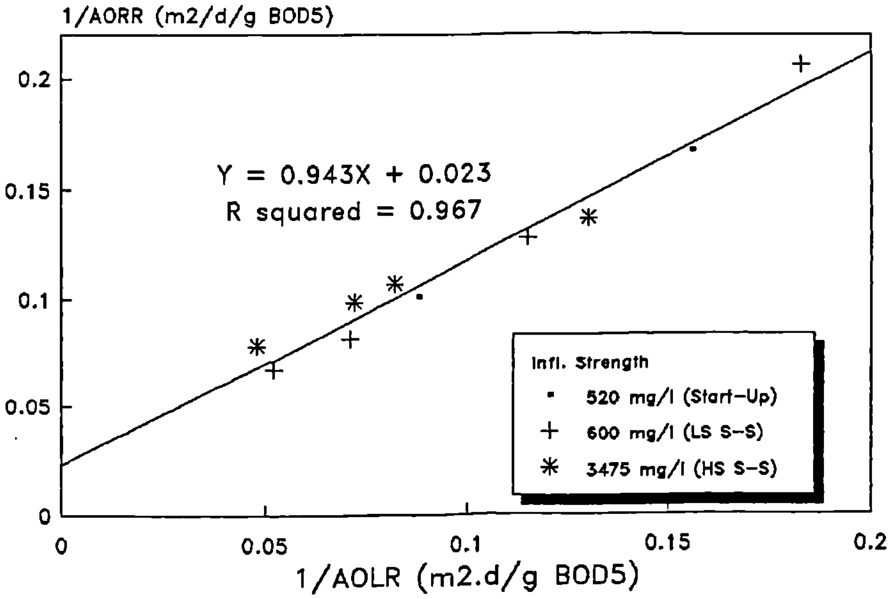


Figure 6.40b: 1/AORR versus 1/AOLR for RBC during Steady State (S–S) Performance

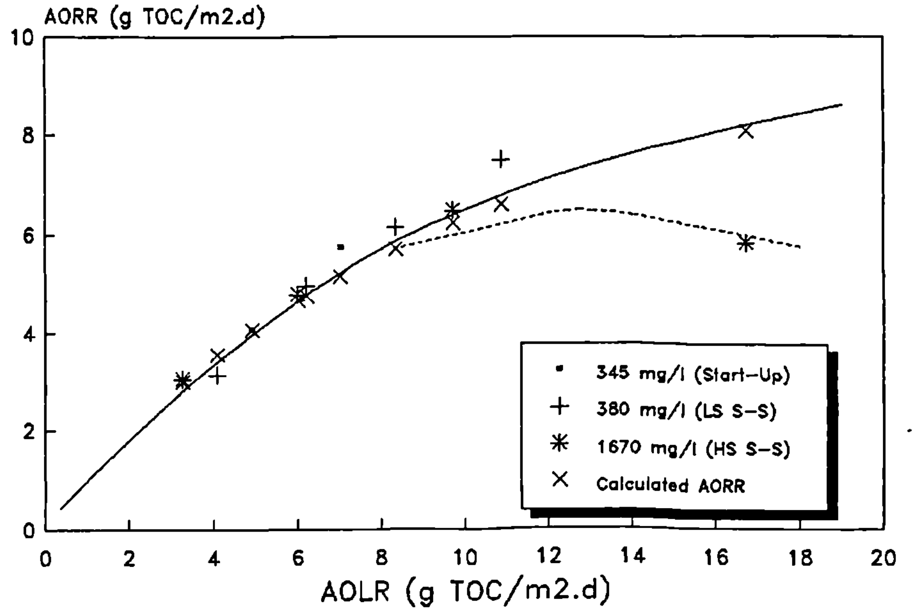


Figure 6.41a: TOC AORR versus AOLR for RBC during Steady State (S-S) Performance

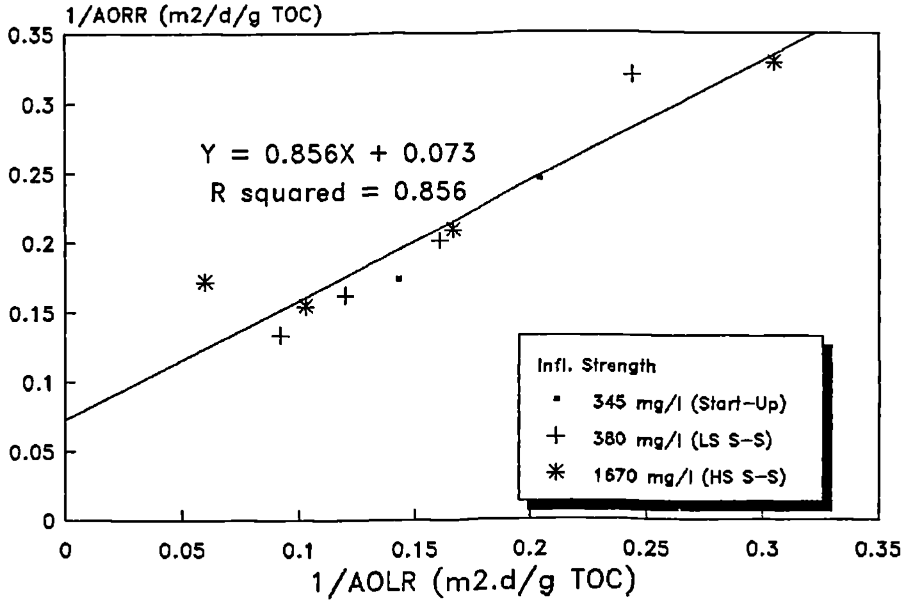


Figure 6.41b: 1/AORR versus 1/AOLR for RBC during Steady State (S-S) Performance

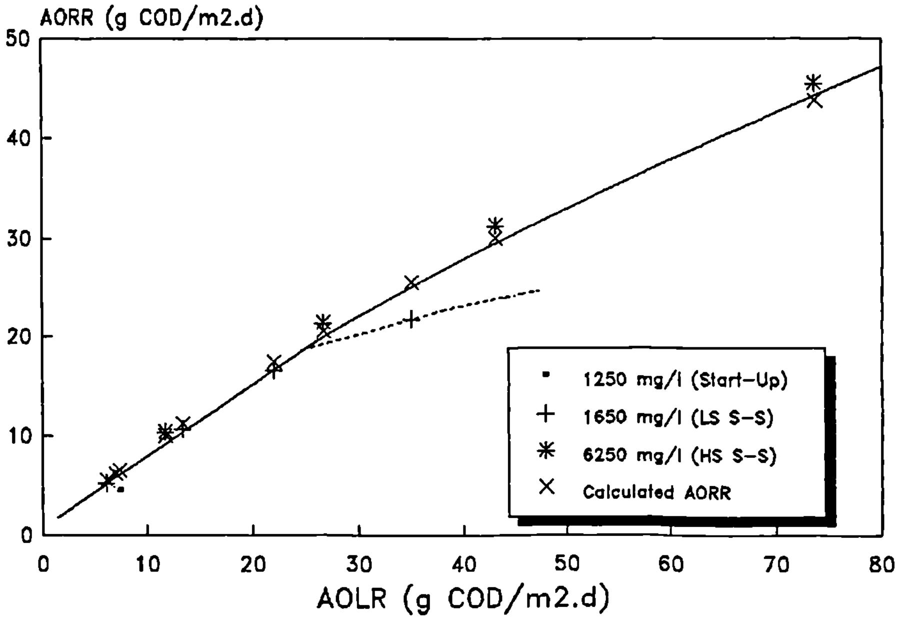


Figure 6.42a: COD AORR versus AOLR for UAF during Steady State (S-S) Performance

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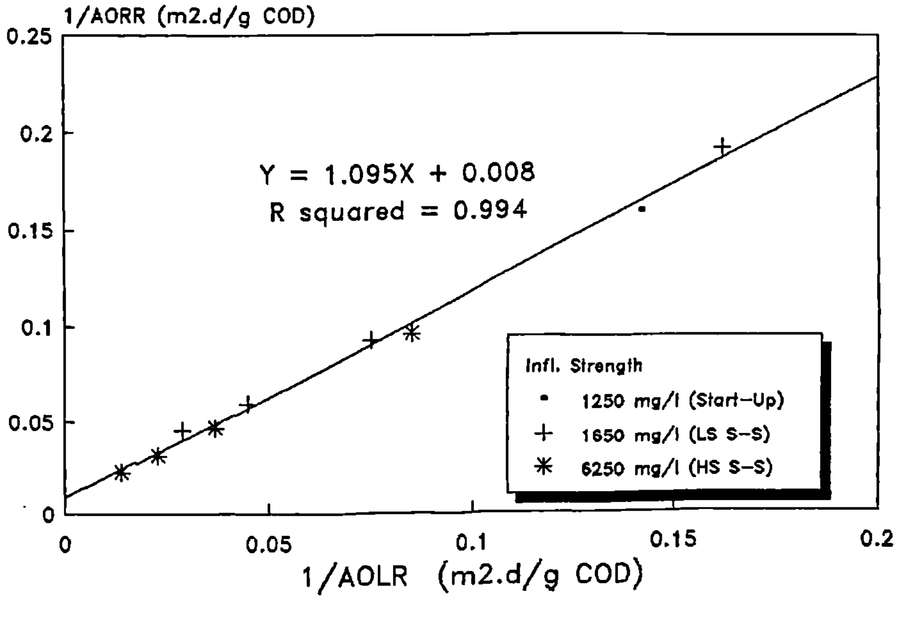

Figure 6.42b: 1/AORR versus 1/AOLR for UAF during Steady State (S-S) Performance

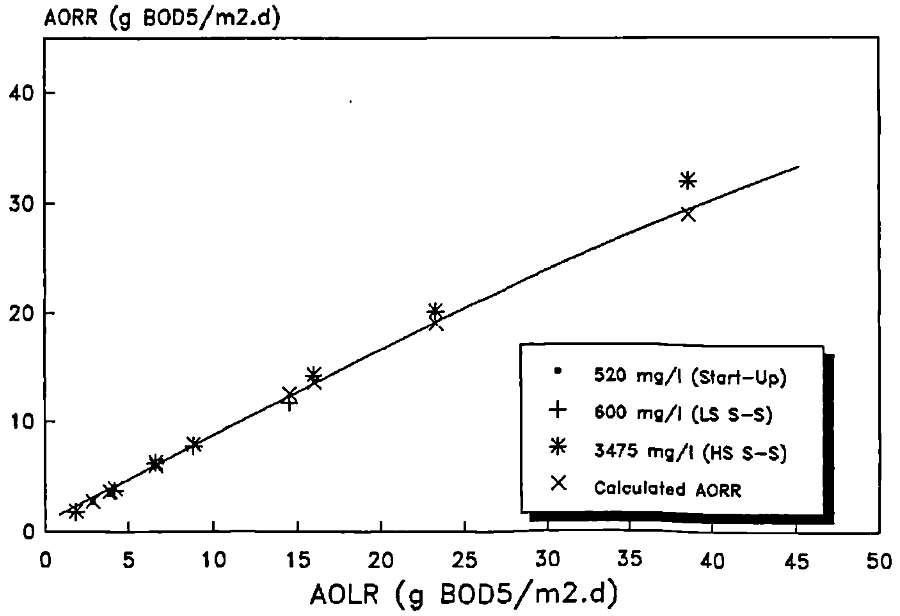

Figure 6.43a: BOD$_5$ AORR versus AOLR for UAF during Steady State (S-S) Performance

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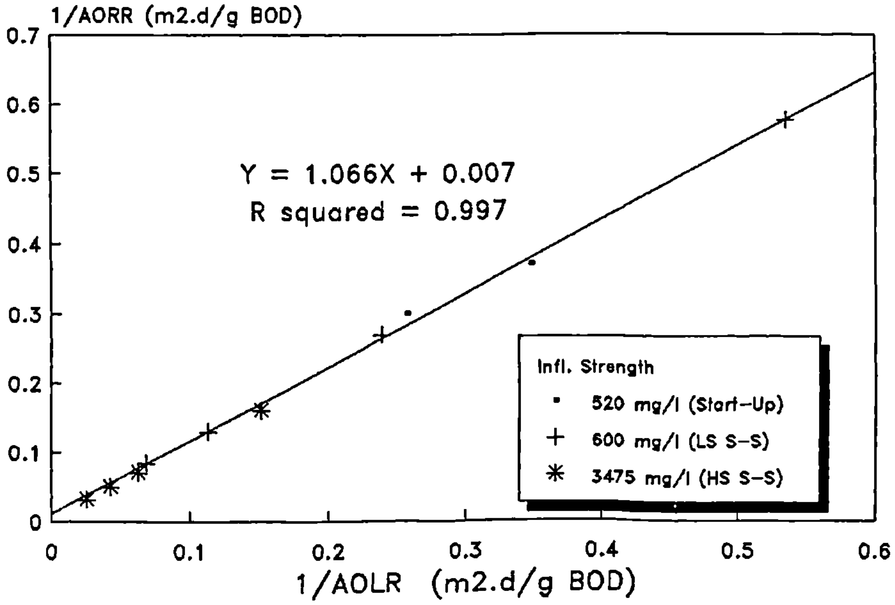


Figure 6.43b: 1/AORR versus 1/AOLR for UAF during Steady State (S-S) Performance

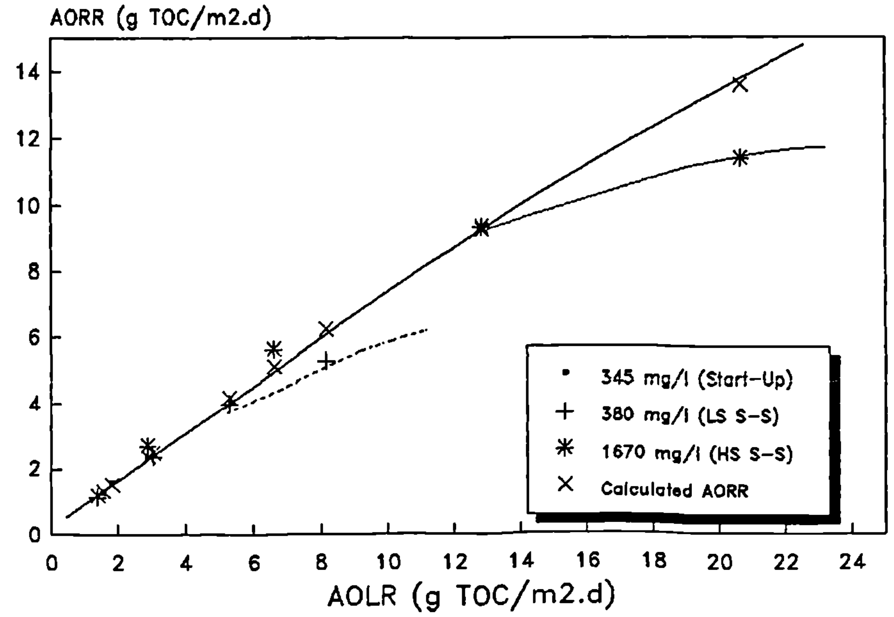


Figure 6.44a: TOC AORR versus AOLR for UAF during Steady State (S-S) Performance

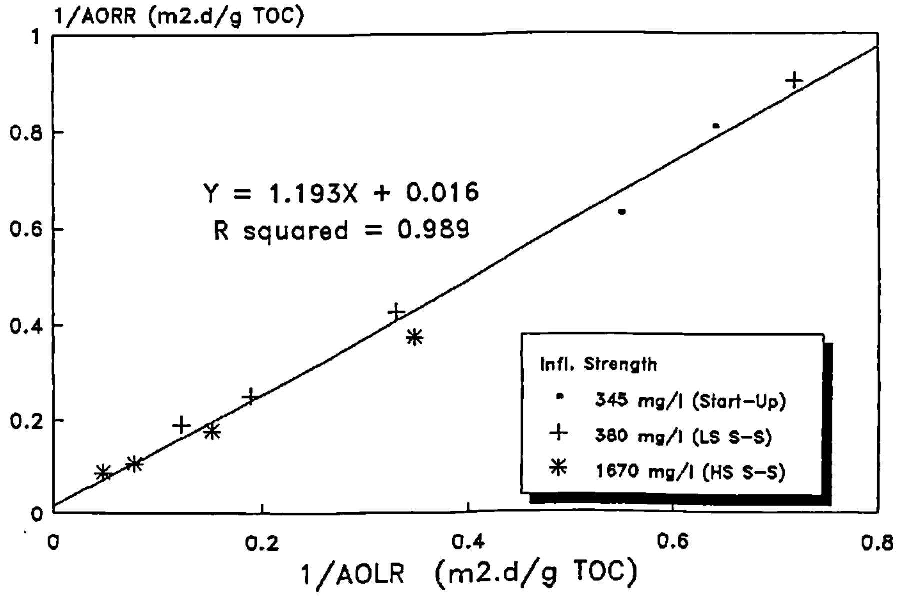


Figure 6.44b: 1/AORR versus 1/AOLR for UAF during Steady State (S-S) Performance

In order to evaluate the validity of the total organic loading concept adopted in the semi-empirical Equation 3.20, a linear representation of the equation was attempted giving:

$$\frac{dt_A}{dS} = \frac{A}{Q(S_i - S_e)} = \frac{K_B}{U_{max}} . \frac{A}{QS_i} + \frac{1}{U_{max}} \qquad (6.1)$$

where

A = total surface area of media (m^2)

Q = flow rate (l/day)

S_i = influent substrate concentration (mg/l)

S_e = effluent substrate concentration (mg/l)

K_B = proportional constant ($kg/m^3.day$)

U_{max} = maximum substrate removal rate ($kg/m^3.day$)

Expressing dt_A/dS as $A/Q(S_i-S_e)$ and plotting this against A/QS_i according to Equ. 6.1 a straight line is obtained. The value of K_B/U_{max} was determined from the slope while the intercept gave the value of $1/U_{max}$. The kinetic plots expressing this relationship are shown in Figs. 6.39b, 6.40b and 6.41b for the RBC whereas Figs. 6.42b, 6.43b and 6.44b show similar relationship for the UAF.

In order to confirm the applicability of the model, a regression line was obtained for the loading rates of each experimental run. From these graphical presentations, the biological kinetic constants, K_B and U_{max} as expressed in term of COD, BOD_5 and TOC, were determined and listed in Table 6.13.

Table 6.13: Substrate Utilization Kinetic Constants

Kinetic Constants	RBC			UAF		
	COD	BOD_5	TOC	COD	BOD_5	TOC
U_{max} (g/m^2.d)	76.9	43.5	13.7	125.0	142.9	62.5
K_B (g/m^2.d)	72.7	41.0	11.7	136.9	152.3	74.6
R^2	0.95	0.97	0.86	0.99	1.00	0.99

The solid lines drawn in the graphs of AORR versus AOLR (i.e in Figs. 6.39a, 6.40a and 6.41a for the RBC and Figs. 6.42a, 6.43a and 6.44a for the UAF) were determined using kinetic constants established from the graph of 1/AORR versus 1/AOLR. The calculated maximum substrate utilization rate was much greater than the actual observed rates for both reactors. The maximum predicted substrate utilization rate for the RBC was approximately 77.0 g COD/m^2.day.

However, in the RBC treating the HS leachate gross sloughing of biological solids from the drums resulted in an AOLR much above 35 g COD/m^2.day (or an AOLR of 14 g BOD_5/m^2.day or 10 g TOC/m^2.day), which corresponded to a AORR of around 23 g COD/m^2.day (or AORR of 10 g BOD_5/m^2.day or 6 g TOC/m^2.day) When the last experimental run for the RBC treating LS leachate was terminated at about 50 g COD/m^2.day (or 20 g BOD_5/m^2.day or 11 g TOC/m^2.day), it was observed that the AORR in terms of all the three parameters had still not reached the maximum, thus the reactor could still have been subjected to a further loading increment. although this would have reduced the treatment efficiency.

In the case of the UAF, the problem was mainly due to limitations of methanogenic bacteria and the build up of VFAs at higher loading rates. The higher applied loading rate was found to affect the UAF treating LS leachate more than when treating the HS leachate. The actual substrate utilization rate for the UAF treating the LS leachate peaked at around 22 g COD/m^2.day (20 g BOD$_5$/m^2.day or 8 g TOC/m^2.day) as compared to the predicted maximum of 125.0 g COD.m^2.day (143 g BOD$_5$/m^2.day or 62.5 g TOC/m^2.day). For the UAF treating the HS leachate the substrate utilization rate went as high as 50 g COD/m^2.day.

For a UAF, Young (1980) developed an empirical kinetic expression as follows:

$$\%removal,E = 100\left(1 - \frac{\epsilon}{HRT}\right) \qquad (6.2)$$

where,

E = ultimate substrate removal efficiency

ϵ = proportional coefficient (hrs)

In order to compare the performance of the RBC and the UAF, an attempt will be made using this expression. Graphical presentations of substrate removals against the reciprocal HRT are shown in Figs. 6.45, 6.46a and 6.46b for the RBC, while Figs. 6.47, 6.48a and 6.48b show similar relationship for the UAF. The values of ϵ for the reactors were evaluated and listed in Table 6.14.

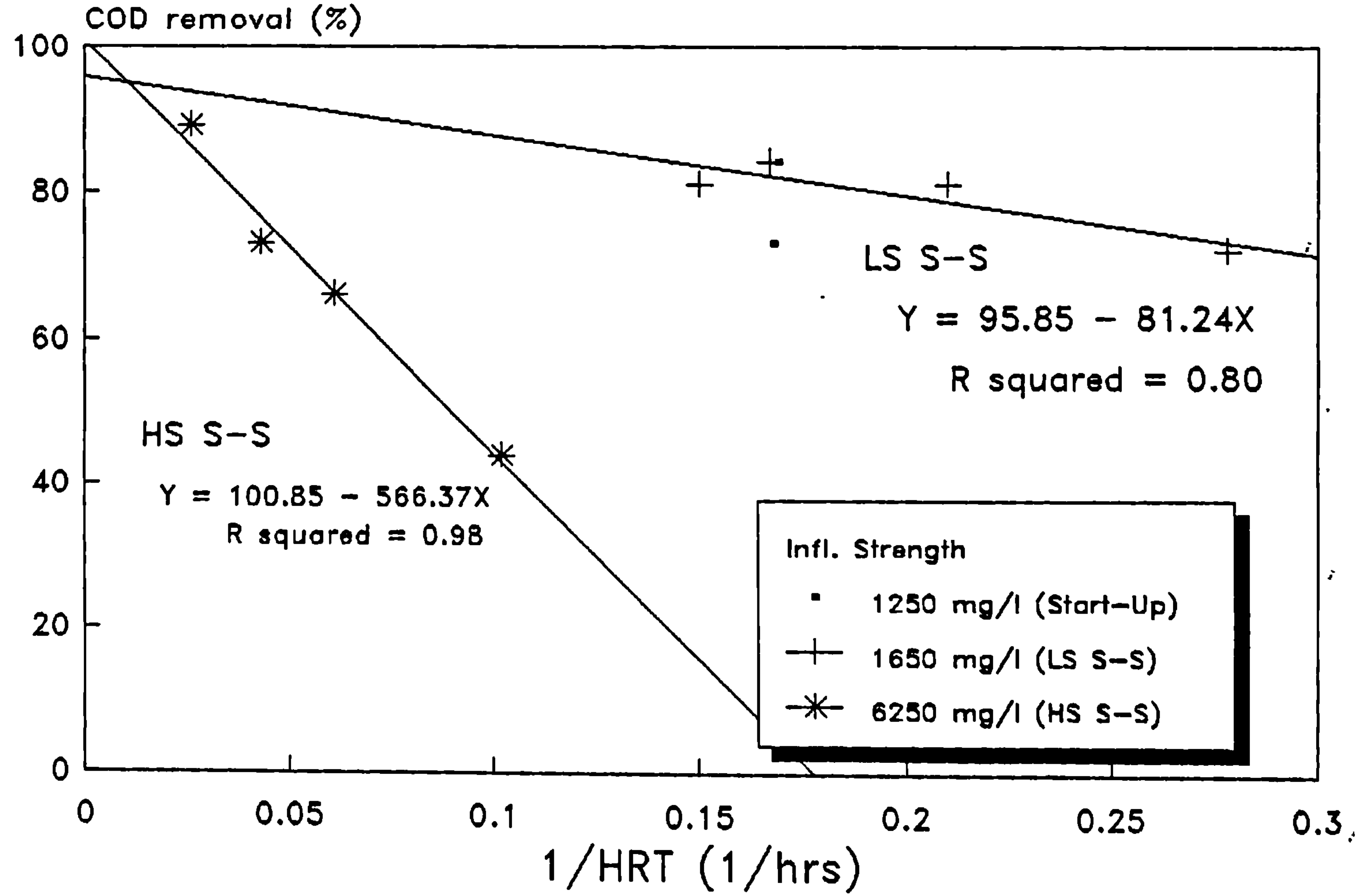


Figure 6.45: COD Removal versus 1/HRT for RBC during Steady State (S-S) Performance

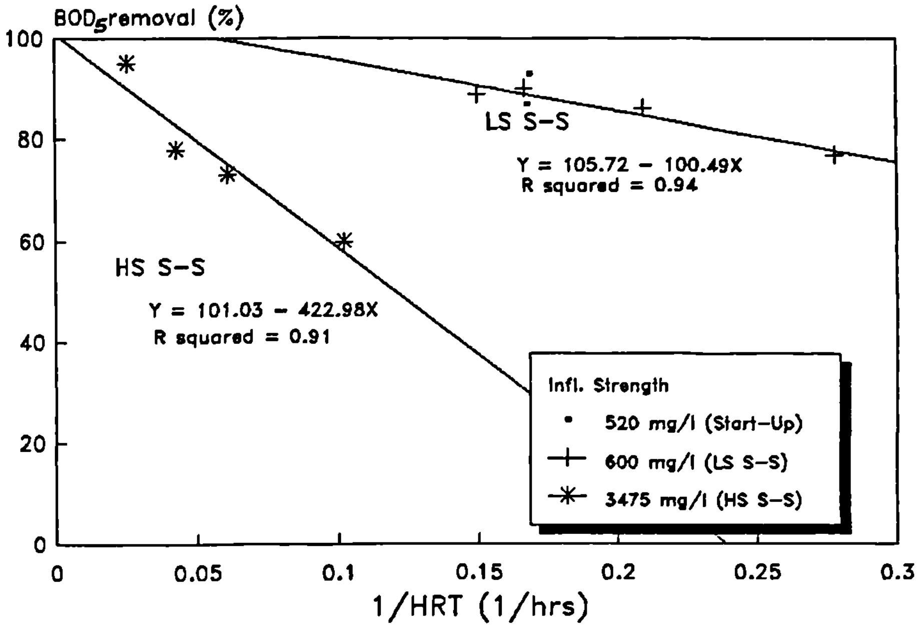


Figure 6.46a: BOD$_5$ Removal versus 1/HRT for RBC during Steady State (S–S) Performance

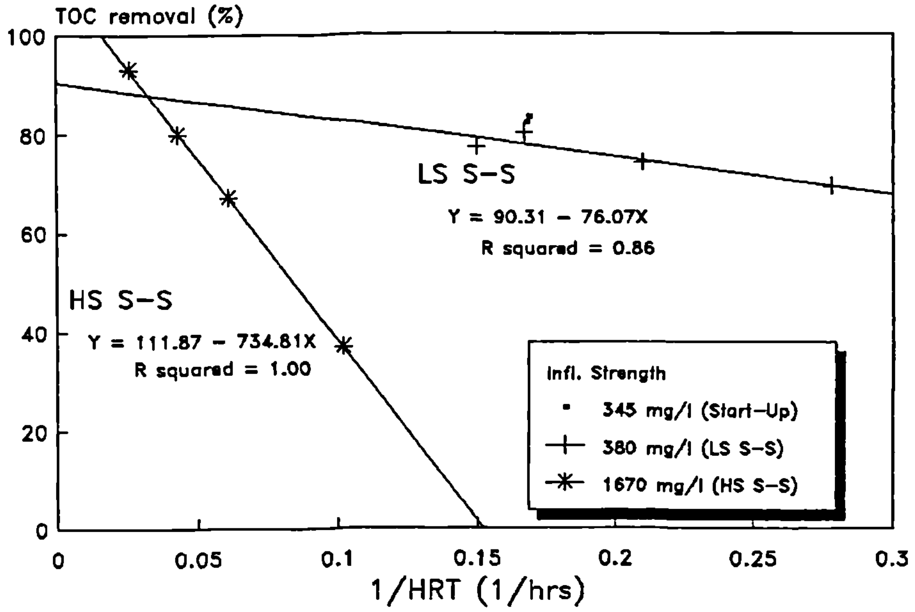


Figure 6.46b: TOC Removal versus 1/HRT for RBC during Steady State (S–S) Performance

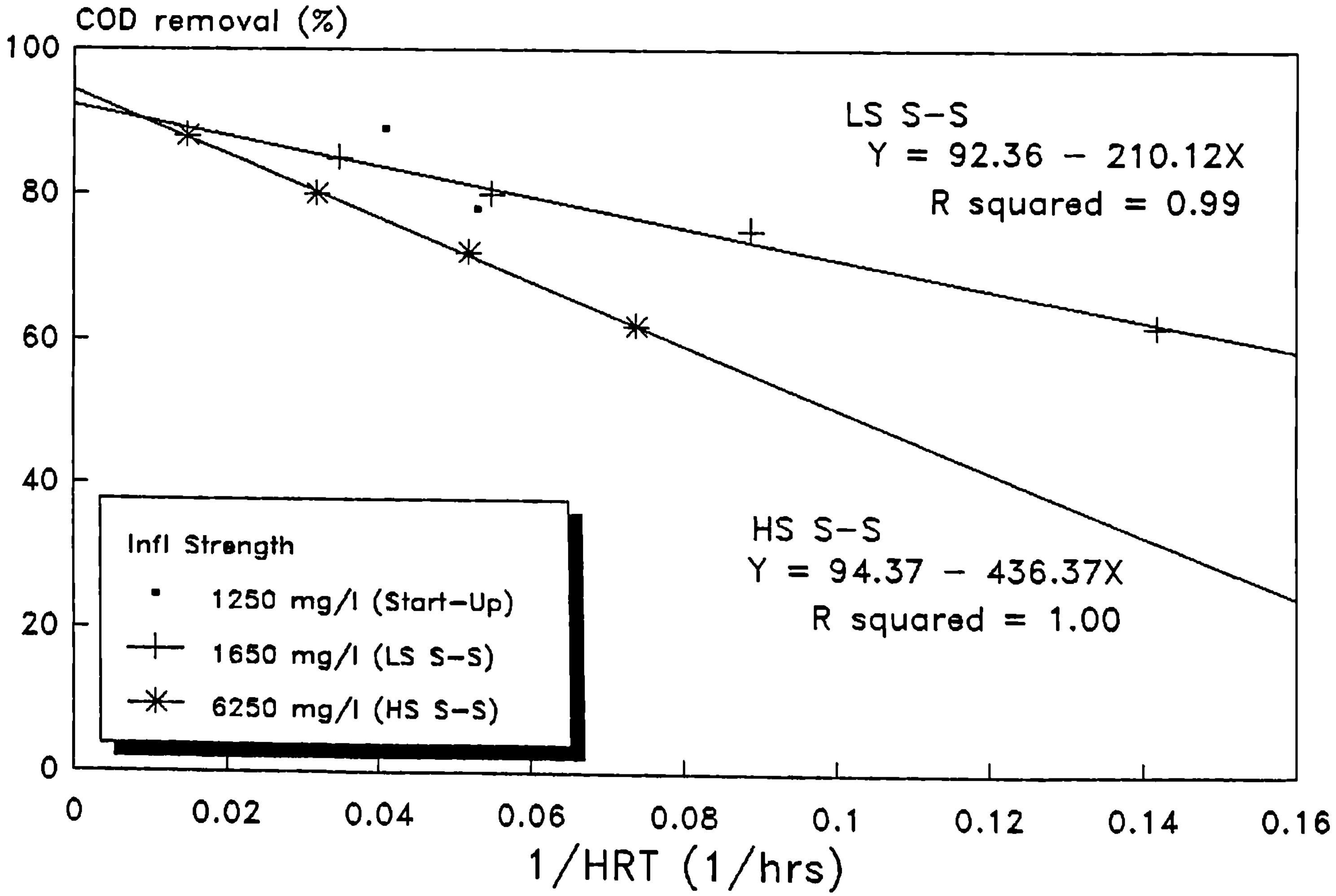


Figure 6.47: COD Removal versus 1/HRT for UAF during Steady State (S-S) Performance

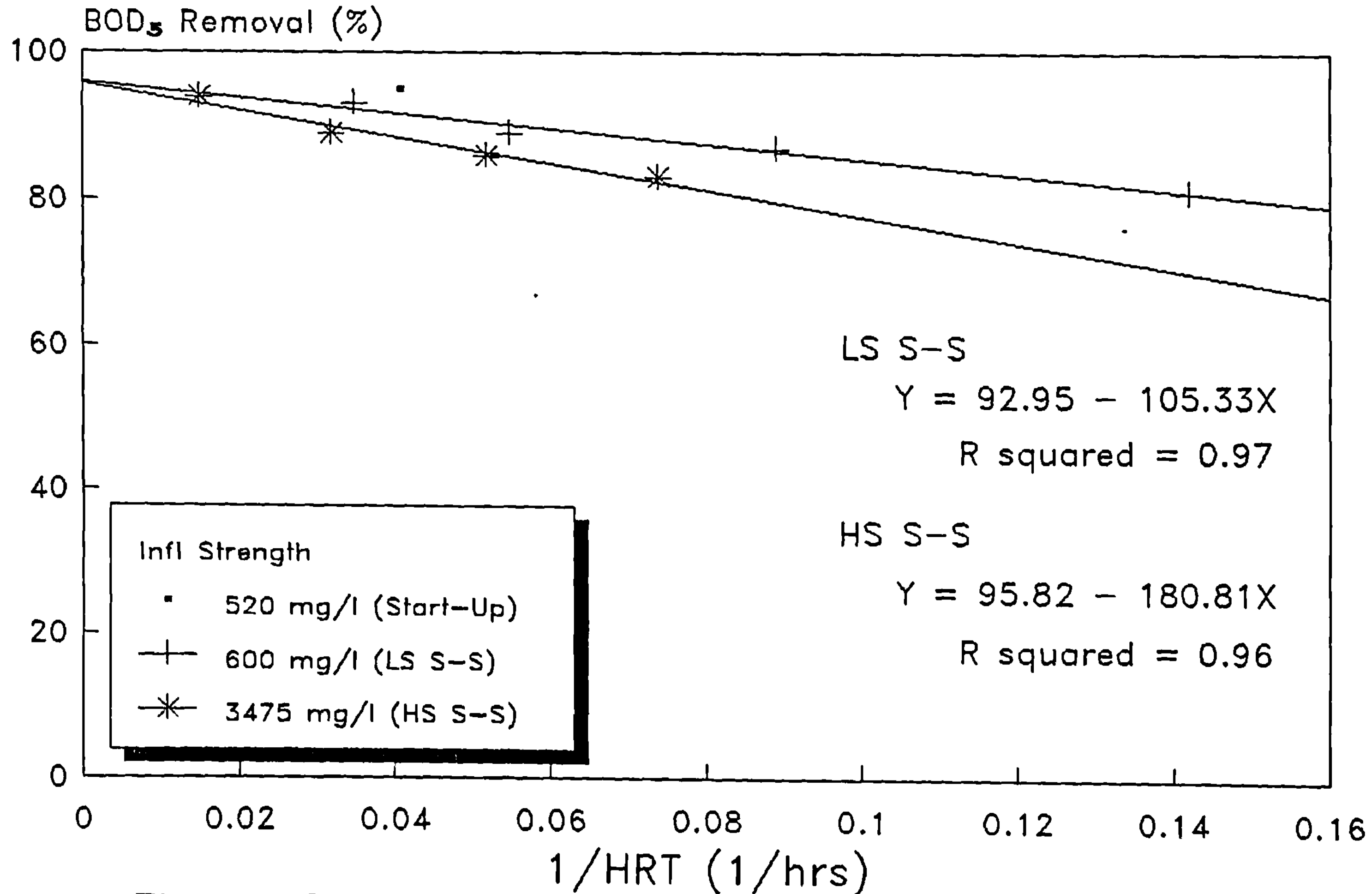


Figure 6.48a: BOD$_5$ Removal versus 1/HRT for UAF during Steady State (S-S) Performance

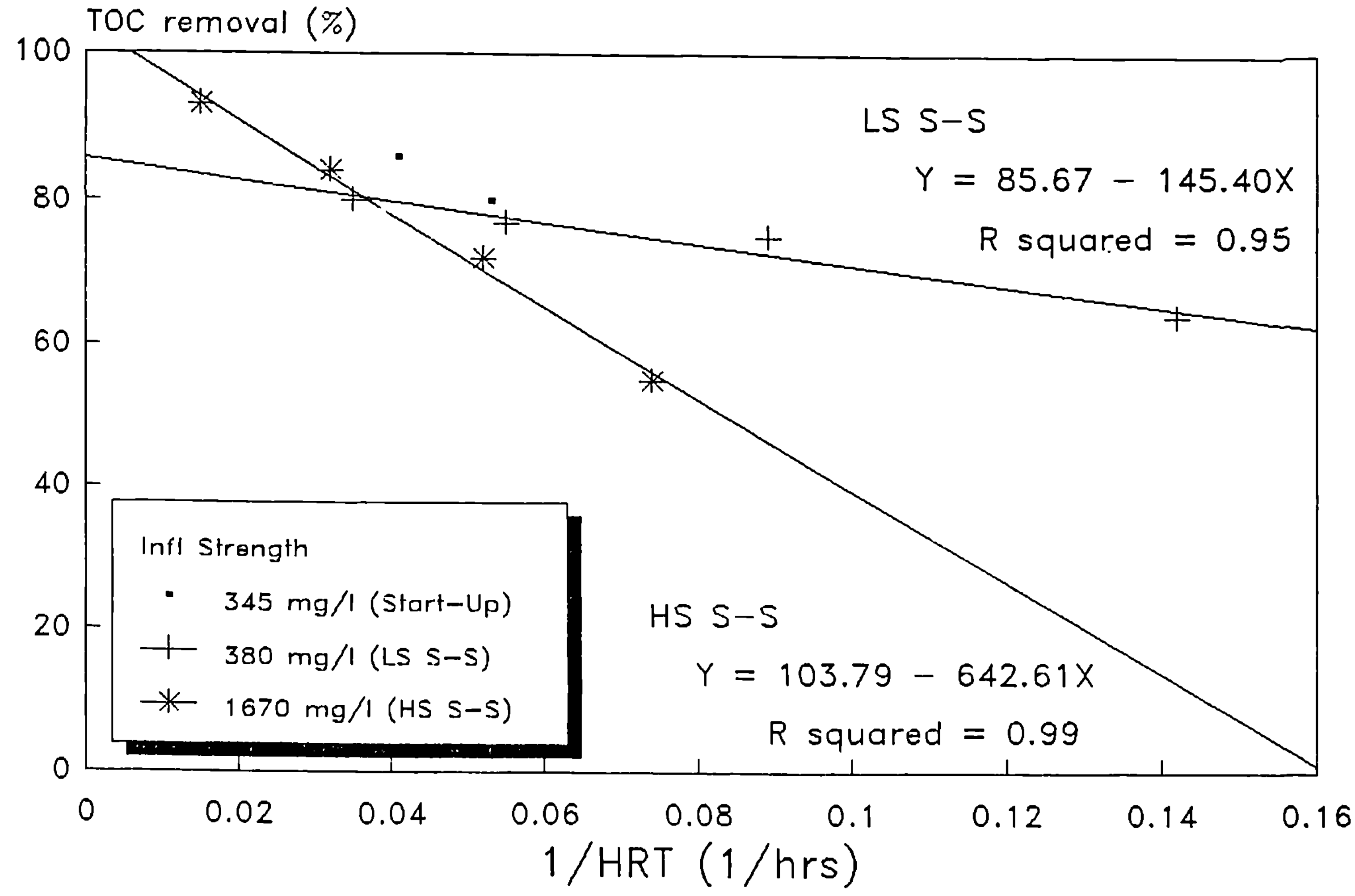


Figure 6.48b: TOC Removal versus 1/HRT for UAF during Steady State (S–S) Performance

Table 6.14: Proportional Coefficient for RBC and UAF

a. LS Steady State (LS S-S) Performance

Proportional Coefficient	RBC			UAF		
	COD	BOD_5	TOC	COD	BOD_5	TOC
ϵ (hrs)	0.81	1.00	0.76	2.10	1.05	1.45
R^2	0.80	0.94	0.86	0.99	0.97	0.95

b. HS Steady State (HS S-S) Performance

Proportional Coefficient	RBC			UAF		
	COD	BOD_5	TOC	COD	BOD_5	TOC
ϵ (hrs)	5.66	4.23	7.35	4.36	1.81	6.43
R^2	0.98	0.91	1.00	1.00	0.96	0.99

The R^2 values in Table 6.14 indicate that the experimental data fit the kinetic expression given in Equ. 6.2. However the expression is of limited value for prediction purposes because the values obtained for the proportional coefficient ϵ, are specific to particular leachate characteristics. As previously discussed these characteristics will change over time for any given landfill leachate.

6.3.5 Biogas Production in the UAF.

The ability of an UAF to generate a useable methane gas beside substrate removals is a bonus in anaerobic processes. Methane production in the UAF reactor as a function of OLR and HRT are demonstrated in Figs. 6.49 and 6.50a. From the graphs, it is clearly seen that the methane production was affected by the substrate concentration. Treatment using the HS leachate yielded a higher methane production compared to that of the LS leachate , especially at low HRT and high OLR. At the same OLR, HS S-S conditions appeared to result in a better performance as well as producing higher methane when compared to LS S-S condition. Conversely, a higher substrate concentration generated a lower percentage methane at the same OLR.

From Fig. 6.49, the percentage methane was observed to decrease with an increase in OLR for both LS and HS leachates. This reduction in percentage methane is related to the lower COD removal. The reason being that the rate of substrate removal is greater than the rate of VFA removal, as seen in the greater build up of VFA concentration at higher loading rates (refer to Figs. 6.18 and 6.27). Hence the methanogenic bacteria when subjected to the higher concentration of VFA, were further actively stimulated, thus producing higher rate of methane production.

The efficiency of methane production (EMP) can generally be used to indicate process stability. In this present study, a good correlation was achieved between methane production and the amount of waste removed as illustrated in Fig. 6.50b. The results for both LS and HS leachates fitted the linear regression with a correlation coefficient of 0.998 indicating a

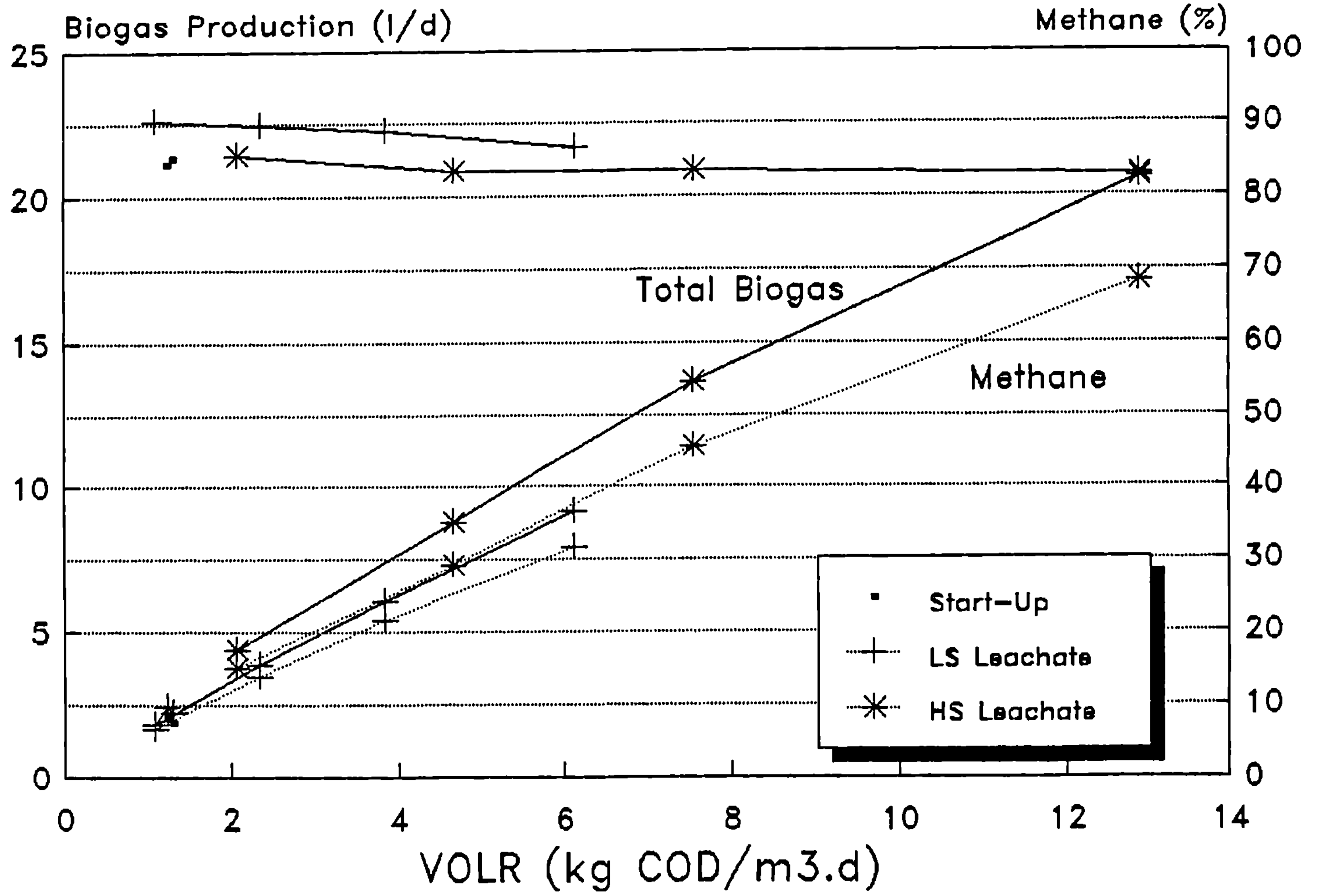


Figure 6.49: Effect of VOLR on Biogas Production and Methane Percentage for UAF Steady State (S-S) Performance

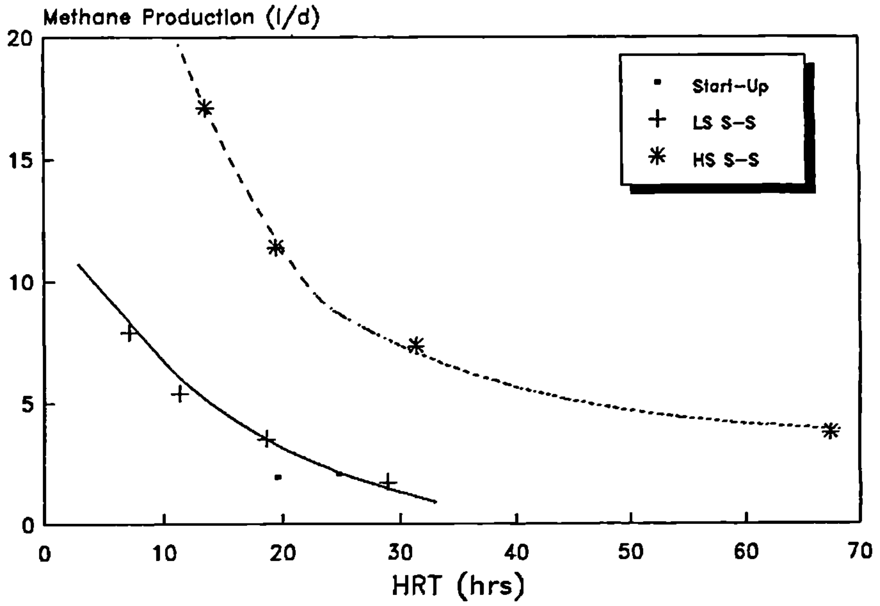


Figure 6.50a: Effect of HRT on Methane Production for UAF during Steady State (S-S) Performance

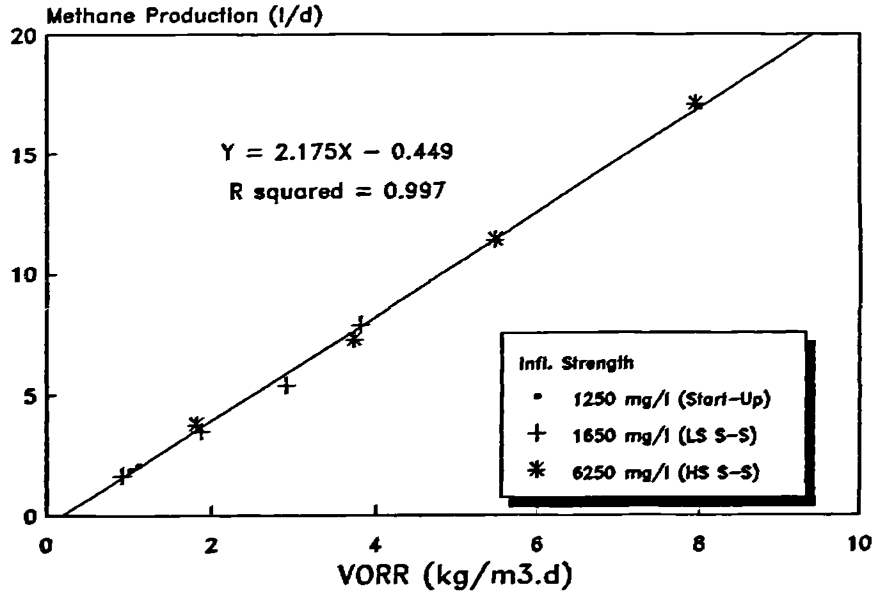


Figure 6.50b: Methane Production versus VORR for UAF during Steady State (S-S) Performance

significance at better than the 0.1% level. The regression suggests that the relationship between methane production and substrate removal is independent of substrate concentration.

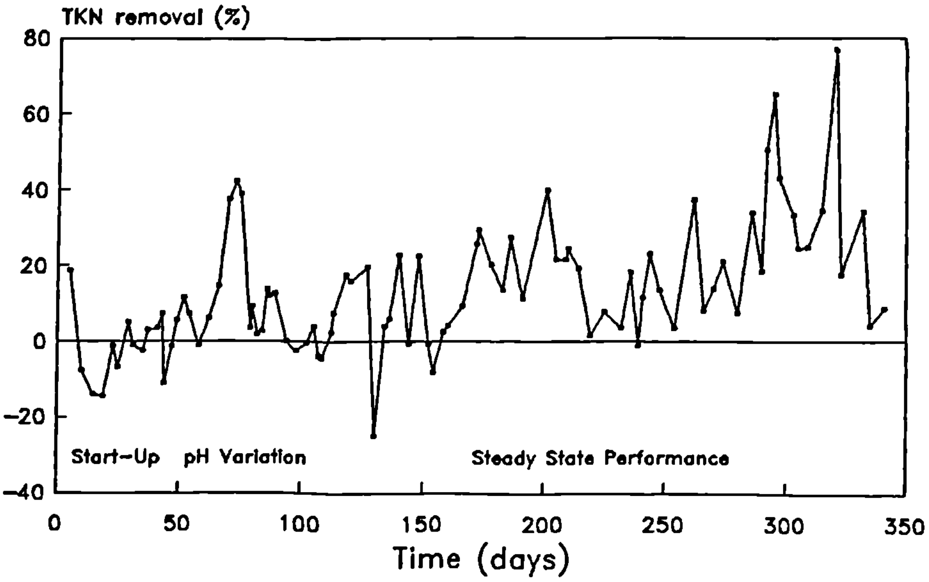
Considering all the OLRs throughout the study, the average daily methane yield at 37°C in the UAF was 0.304 m^3/kg COD removed for LS S-S condition, and 0.327 m^3/kg COD removed for HS S-S condition. The methane yields were slightly lower than the theoretical value of 0.35 m^3/kg COD removed (at STP), but the yields were comparable with literature values (Boyle and Ham, 1974; Wu et al, 1982; Henry, 1987). The reason may be due to the existence of non-biodegradable fraction in the COD removed and COD fraction allocated for bacterial metabolism. The latter implies that a higher bacterial growth rate can reduce methane production since more COD has to be used for cell synthesis.

6.3.6: Nitrogen Removal Results

The results for nitrogen removal are shown in Figs. 6.51 and 6.52 for the RBC and the UAF respectively. Although, the scope of the study was not extended to include the investigation of nitrification and denitrification in the RBC and the UAF, evaluation of nitrogen removal was made along with the other investigations. Apart from assessing the requirement for nutrient and occasional determination of nitrite- and nitrate-nitrogen, monitoring of ammonia-nitrogen (NH_3-N) loadings were not determined.

From Fig. 6.51, it was observed that after the Phase I study, nitrogen removal in the form the Total Kjeldahl Nitrogen (TKN) was achieved in the RBC treating LS leachate, especially in the Phase III study.

TKN Removal by RBC



Ammonia−Nitrogen Removal by RBC

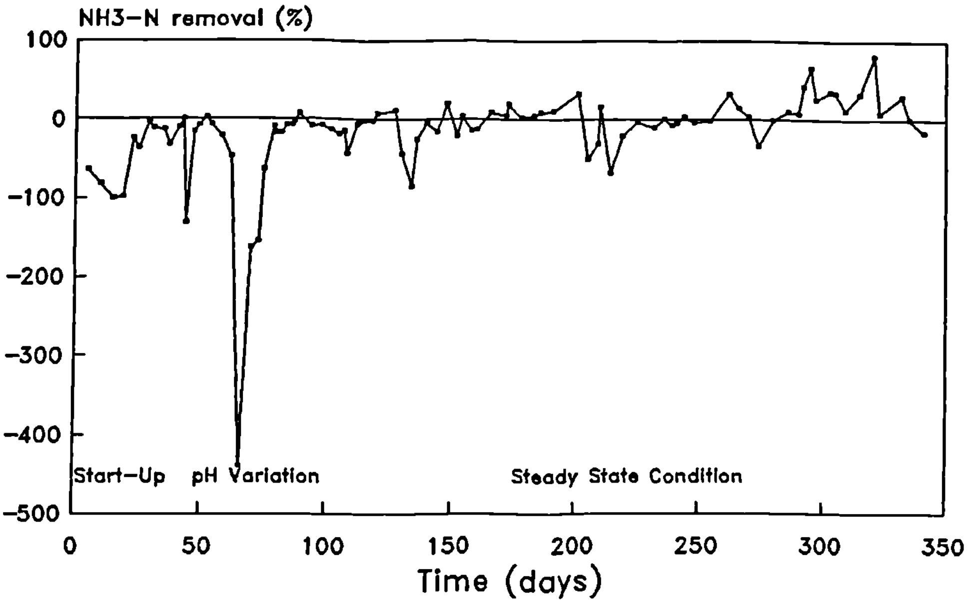


Figure 6.51: Nitrogen Removal by RBC for LS Leachate

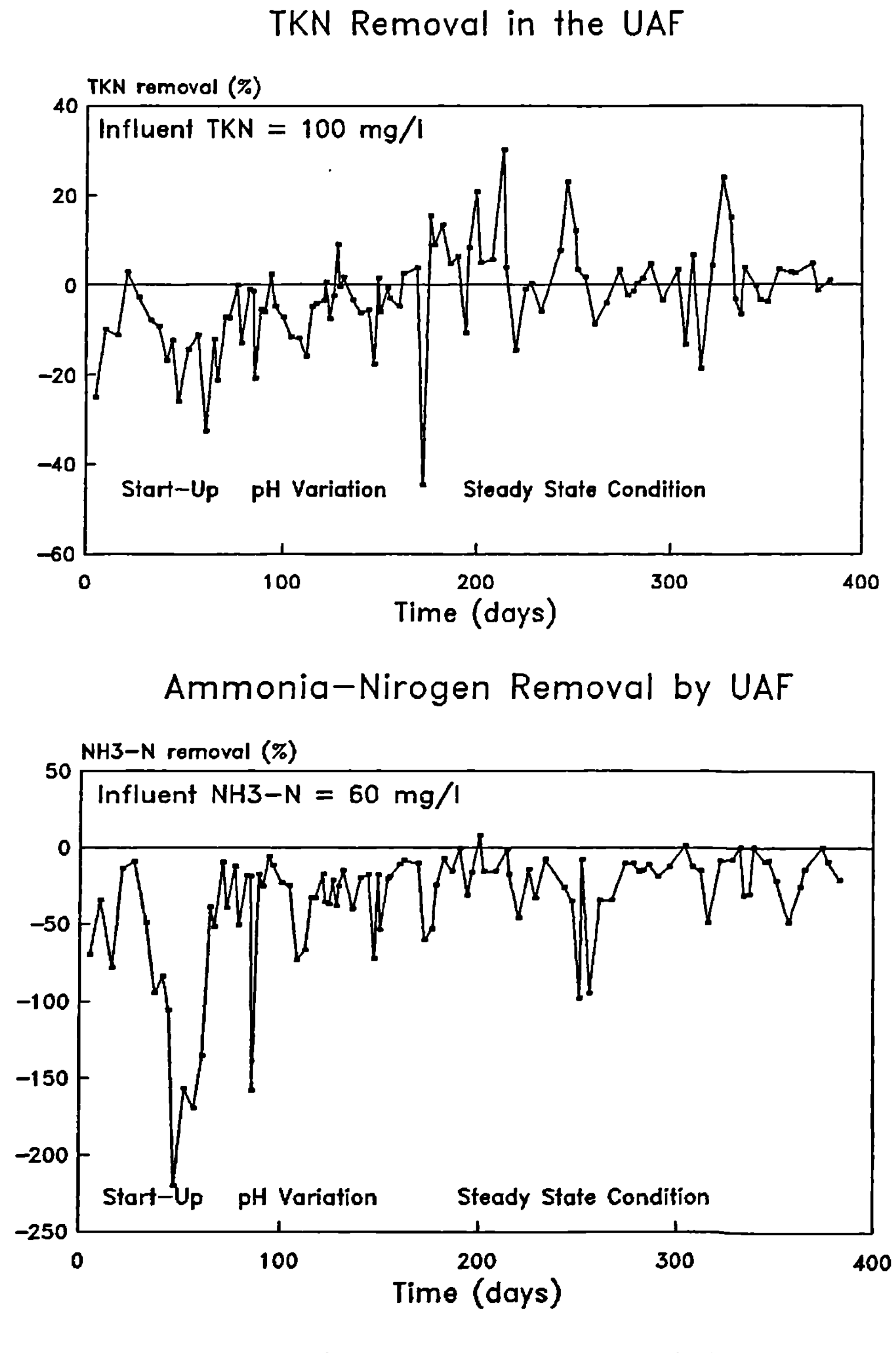


Figure 6.52: Nitrogen Removal by UAF for LS Leachate

NH_3-N reduction was also found, although the reduction was not consistent and high enough. The highest TKN removal efficiency was 77% while the highest for NH_3-N removal was 81%. From the results, an increase in the nitrate-nitrogen in the LS leachate effluent was observed, which could possibly be due to the occurrence of nitrification in the RBC.

As for the UAF treating LS leachate, the nitrogen removal was only observed in term of TKN' removal. Even the TKN removal was not very high as seen in Fig. 6.52. The increase in the NH_3-N in the effluent was probably due to the biological assimilation which took place that removed the organic nitrogen. The highest TKN removal efficiency was 30%.

6.4 Conclusion

Comparing the start-up of the RBC and the UAF units, the RBC start-up is faster. The aerobic biomass growth on the drum can be seen within two weeks from start-up, whereas the biomass in the UAF units appeared to grow at a slower rate as seen from the longer time required before stable conditions were achieved. In the early stages of start-up operation, a considerable fraction of the biological solids were washed out with the filter effluent. This also affected the UAF performance. The rate of biomass attachment and development onto the supporting media in the UAF reactor depended on the media configuration. However, after a quasi steady state condition had been established, both the RBC and the UAF units gave good substrate removals.

Salkinoja-Salonen et al (1983) pointed out that for fast start-up of anaerobic fixed film reactor, the OLR should be less than 0.1 kg COD/kg

VSS.day with a high HRT (more than 30 days) to prevent washout. An OLR of approximately 1.0 kg COD/m^3.day was applied to the UAF reactor during the start-up operation. This OLR was capable of reducing the acclimatization period for the methanogenic bacteria to the waste introduced.

Provided effective contact is maintained between the active biomass and the influent feed materials, the loading capacity of an anaerobic treatment plant can be retained in the reactor. The UAF reactor designs can maintain a larger amount of anaerobic biomass per unit volume of reactor in which biomass retention is achieved solely by attachment to the support surface media and is limited by the surface area to volume ratio of the packing material (Colleran et al, 1986). Young and Dahab (1982), stated that long term operation may result in excessive biomass entrapment in the interstitial cavities in the matrix bed, with resultant problems of plugging and channelling. But from these investigations the UAF is capable of accepting a higher loading rate without plugging or channelling, as seen from the satisfactory substrate removal rates.

From the pH variation studies, the performance of the UAF units as compared to the RBC reactors was found to be greatly affected by extremes of pH. A sharp reduction in substrate removal efficiency resulted when the pH fell below 6 and rose to over 8. In order to rectify this, sufficient buffer must be added to the influent feed to give a pH value near to neutrality. This is very important during start-up of an UAF reactor.

Substrate utilization in both the reactors was found to be a function of the mass loading rate with the reaction described by monomolecular kinetics. As the substrate loading rate increased, the substrate removal rates decreased. The RBC and the UAF exhibited opposite effects when treating LS and HS leachates. High substrate removal rates were obtained in the RBC treating LS leachate whereas for the UAF high substrate removal rates were found when the HS leachate was used.

An increase on OLR (a decrease in HRT) was found to reduce the performance of both the RBC and the UAF, regardless of the origin or type of leachate used. This general trend was similar to those observed with other fixed film reactors treating different types of wastewaters (Del Borghi et al, 1985; Bonastre and Paris, 1989). The decline in system efficiency noted is due to the increased dilution rates where, in such instances, the substrate utilization capacity of the system's biomass is exceeded by the hydraulic application rate of organic substrate with a resultant diminishment of overall removal.

Although various differences in the performances existed between the RBC and the UAF reactors, what is seen is that both the reactors achieved satisfactory substrate removals at low OLR. The study also indicates the validity of using TOC measurements for kinetic analysis. The TOC concentrations in the influent and effluent of both LS and HS leachates were found to relate consistently with BOD_5 and COD concentrations.

Although, nitrification was not actually analyzed, it seemed that the RBC was capable of removing nitrogen from the leachate. From various investigations (Ito and Matsuo, 1980; Pano and Middlebrooks, 1983; Lin et al, 1984) nitrification was reported to occur by the appearance of nitrate-nitrogen in the effluent. From the study, it was found that nitrate-nitrogen was measured in the effluent. Therefore, it could be inferred that nitrification did take place in the RBC. Therefore, it is recommended that further study should be made to investigated the nitrification process, in order to confirm this finding.

Another salient observation highlighted by the performance of both the reactors was noted during the transitional period between one quasi steady state and another. Biological reactors are generally considered to be auto-catalytic, which implies that for a given steady state, the active biomass population in the reactor would be proportional to the flux of growth limiting substrate, hence an acclimatization phase would result. But from investigations made during the Phase III study indicated that the biomass in both reactors adapted well during the transitional period, with only a small decreased in removal efficiency soon after the OLR was increased. Stabilization occurred two days after the loading changes.

The overall performance of the RBC and the UAF under various operating conditions, basically revealed that the RBC was more better suited for lower strength wastes, while the UAF could be operated for both lower and higher strength wastes. Better performance in the UAF treatment study is demonstrated with HS leachate when operated at lower OLR (i.e higher HRT). Therefore in terms of organic loading, lower OLR (higher HRT) may be advantageous for better effluent quality although optimal utilization of reactor volume in organic waste reduction calls for the application of higher OLR (shorter HRT).

CHAPTER SEVEN

PERFORMANCES OF THE RBC AND THE UAF DURING RESTART-UP, PROFILE STUDIES, SHOCK LOADINGS AND ZINC TOXICITY

7.1 Introduction

The fast growing technological and industrial advancements have encouraged organic materials, nutrients and toxins to increase in most water resources. Water quality regulations cannot completely solve these ever increasing problems. Therefore, in order to address these problems, environmental engineers must examine and analyze the effect of these pollutants on wastewater treatment processes. Biological treatment systems, because of their sensitivity to loading variation, external conditions, and toxins, are of particular concern.

The provision of proper environmental conditions in order to enhance the growth of biomass in aerobic and anaerobic systems is the key to maintaining process control and stable operations in biological treatment systems (Stover and Gonzalez, 1988). Any changes to the environmental conditions, especially fluctuations in wastewater characteristics, tend to disrupt those steady state conditions which such biological treatment facilities were designed to approach. The two most critical parameters of concern when ensuring stable operating conditions are the hydraulic flow rate and the organic loading rate (OLR). Apart from these two parameters other parameters such as pH, temperature, nutrients, and the absence of toxic or inhibitory substances are also critical to successful operations of the systems.

These environmental changes, which may be in the form of shock or intermittent loadings, tend to disrupt steady state conditions. Unless these changes are taken care of by preventive engineering measures, they must be accommodated by the systems solely through successful biological response or by combined biological and operational remedial responses. Four types of transient and/or shock loadings that can affect reactor performances are (Young, 1980):

i) variation in loading as a result of changes in flow rate or waste strength,

ii) intermittent operation,

iii) changes in pH, temperature and waste composition, and

iv) influx of organic toxins or heavy metals.

Anaerobic filters are much more resistant to variations in waste load and environmental factors such as pH and temperature than was originally thought (Wu et al, 1982). Young (1980) reported that UAFs have been shown to accept considerable adverse operating conditions without permanent loss of treatment efficiency even when operated at organic loading rates well in excess of the loading capabilities of conventional aerobic and anaerobic systems. Fourfold instantaneous increases in loading have caused no permanent adverse effects on filter performance. The ability of the UAF to recover rapidly to its former steady state conditions was also found by Young (1980) in his study; although the recovery time increased as the magnitude and duration of the change in flow and load increased (Young, 1991).

Short term loading increases having a duration of one or two HRT can be expected to produce a slight, short term change in effluent quality or gas production. Long term changes, however, will cause the COD and volatile acid profiles, and no doubt the population dynamics and solids concentrations, to shift until a new quasi steady state level of performance is reached.

As for aerobic processes, the ability of RBC units in handling shock organic and hydraulic loadings as well as toxic load capabilities are well documented (Bracewell et al, 1980; Dupont and McKinney, 1980; Fry et al, 1984). Bracewell et al (1980) in their studies of an RBC process treating phenol-formaldehyde resin wastewater found that the RBC exhibited excellent stability in withstanding periodic shock loadings. The RBC was found to rapidly recover within 24 hours after the termination of the shock loadings. One important characteristic of RBC units is the ability to retain the attached biomass when exposed to large hydraulic shocks (Fry et al, 1984).

In contrast, Dupont and McKinney (1980) after studying the performance of a municipal RBC installation in Kirksville, Missouri, found treatment efficiency was reduced as a result of variable hydraulic loadings. This reduction in treatment efficiency was attributed to reduced contact time within the RBC units and hydraulic surges on the final clarifiers.

The results of a study by Poon et al (1980) agree with the Kirksville study. The effluent soluble BOD_5 increased rapidly as the hydraulic shocks increased. Even though the soluble BOD_5 removal actually improved, the effluent quality deteriorated significantly.

Intermittent operation might be used in practice for weekend operation. Young (1968) and Jennet and Rand (1980) in their studies to evaluate the performances of UAF units after several days of no feeding and flow found that the COD removal efficiency initially decreased upon restarting, but full COD removal capacity and gas production were achieved after only three to four days of operation.

It has been generally assumed that anaerobic processes are unable to cope with waste streams containing toxicant and therefore are unsuitable for the treatment of many wastewaters. Toxicant do alter the kinetic parameters of methanogens and thus increase their generation time and decrease pollutant removal efficiency. However, these adverse effects can be offset by proper attention to solids retention time (SRT) (Wu et al, 1982). Proper acclimatization procedures can also increase the threshold concentration of toxicant which cause inhibition. The magnitude of the toxic effect generated by a substance can be reduced significantly if the concentration is increased slowly. In evaluating data from toxicity studies for design purposes, the engineer should consider the test conditions used and whether toxic materials may be introduced into the waste stream to be treated as a slug dose of high concentration or as a constant component to which a population may become acclimatized. Speece et al (1980) showed that methanogenic bacteria could acclimatize to toxicant concentrations that were 100 times greater than the concentrations which caused inhibition of unacclimatized cultures. The early warning of possible metal toxicity is given by a gradual decrease in gas production and an increase of the effluent COD.

The loading capacity of a biological wastewater treatment system is essentially dictated by the amount of active biomass retained in the system, provided sufficient contact between active biomass and waste organics can be assured. Generally, the active biomass in a fixed film reactor consists of attached biomass (biofilm) with a small amount of suspended biomass (mixed liquor). In order to determine the actual concentration of active biomass in the reactor, profile studies have been carried out by a number of researchers. Young and McCarty (1969) cut three UAFs into sections and the quantity of biological solids present at various height were determined. In studies carried out by Donovan (1980), the filter media in a UAF were removed at the end of the experimental runs and total biological solids, both attached and suspended were determined.

Speece (1983) stated that, due to the comparatively high synthesis ratio of aerobic organisms, an effluent suspended solids concentration of 500 mg/l may border on solids wash-out failure for a waste strength of 1000 mg/l COD, whereas, for anaerobic systems a solids loss of 30 mg/l would apply for the same waste.

In the study of biomass retention, the main parameter of concern is SRT. To achieve maximum removal efficiency and process stability, the SRT should be at least 10 times the minimum bacterial doubling time (Jewell, 1987). SRT depends on both the daily loss of solids in the effluent and the total sludge contained in the reactor. The daily loss should be controlled in order to keep a net sludge increase in the reactor and it is especially important in the treatment of dilute waste with a sludge yield close to the daily loss. In a high rate system, the sludge washed-out is independent of HRT, but depends on the OLR. The effluent suspended solids from a laboratory scale UAF (Frostell, 1981) clearly showed this. At a constant OLR, the effluent suspended solids reduced when the HRT was reduced, so that daily loss of sludge from the reactor was constant. The daily sludge loss in the reactor was increased at higher OLR. Sooner or later a reactor becomes saturated with sludge, so that loss of solids in the effluent becomes equal to the net sludge production. Therefore, at steady state, SRT is influenced by the capability of a reactor to retain a high concentration of biomass.

Lettinga et al (1983) stated that the maximum amount of sludge that can be retained within a reactor for a given sludge etc. is mainly dictated by the applied OLR, i.e the maximum OLR and the maximum achievable sludge retention are interrelated. Increasing the OLR will increase the gas production and probably the expansion of the sludge bed in a UASB. As a result the sludge bed will completely fill-up the reactor causing the increase in solid wash-out.

Longer SRTs provide more concentrated biomass in the reactor which consequently cause lower applied sludge loading rates, less nutrient requirement, less surplus sludge production and higher stability in the case of shock loading and/or fluctuation in environmental factors (Henze and Harremoes, 1983)

7.2 Experimental Programme

The scope of this Phase IV study was to evaluate the response capabilities under intermittent loading (total feed shutdown periods), controlled organic and hydraulic shock loadings as well as toxic shock load i.e gradual step addition of zinc (Zn) concentrations in both the RBC and UAF units. Periodic profile studies were also made.

7.2.1 Restart-Up Operation

On completion of Phase III, the RBC and the UAF units were shutdown for twenty six days. Wastewater feeding was then resumed at a loading rate of 3 kg COD/m^3.day (AOLR of 9 g COD/m^2.day) for the RBC, while the UAF was started with a loading of 1.5 kg COD/m^3.day. Effluent samples from the RBC and the UAF were then analyzed to determine their ability to withstand intermittent operation.

7.2.2 Profile Study

Periodically, mixed liquor samples were collected throughout the period of operation from each stage of the RBC and from the sampling points in the UAF. The samples were analyzed for soluble COD, BOD$_5$, TOC, VFAs and suspended solids. Periodical wastage or scraping of attached solids from the RBC drums were carried out in order to control the SRT more effectively, whereas at no time during the operation were solids wasted from the UAF except for the small amounts removed with sample analysis.

After terminating the Phase III study, one each of the RBC and the UAF units were dismantled to determine the biological solids distribution throughout each unit and to examine the manner in which solids were held by the media. The other RBC and UAF units were dismantled at the end of the entire study period. The unattached solids or mixed liquor suspended solids (MLSS) in each reactor were drained out and were measured to determine the mass of MLSS and the mixed liquor volatile suspended solids (MLVSS).

The plastic media in each drum of the RBC were removed, and a random representative sample measured to determine the mass of the attached biological solids. Similarly, the plastic media in the UAF were removed carefully, in sections, and random sampling taken to determine the mass of attached biological solids.

7.2.3 Shock Loading Operations

Both the RBC and UAF were subjected to a series of volumetric and organic shock loadings. Two separate volumetric shock were conducted to evaluate the response of the RBC and the UAF units. The initial shock loading was performed for 12 hours while the second was a 24 hours shock loading. From the base loading at 6 hours HRT of 5.5 kg COD/m^3.day (AOLR of 22 g COD/m^2.day) for the RBC during the initial shock load, the loading was increased to 17.5 kg COD/m^3.day (AOLR of 70 g COD/m^2.day) by decreasing the HRT to 2 hours. For the second shock loading of the RBC, the loading was raised from the base loading of 3.3 kg COD/m^3.day (AOLR of 13 g COD/m^2.day) to 14.2 kg COD/m^3.day (AOLR of 55 g COD/m^2.day) by decreasing the HRT from 11 hours to 2.5 hours.

For the UAF, the first shock loading was carried out from the base loading at 18 hours HRT of 1.75 kg COD/m^3.day to a loading of 7 kg COD/m^3.day by decreasing the HRT to 4.7 hours. In the second shock loading, the loading was increased to 9.5 kg COD/m^3.day from the base loading of 2.5 kg COD/m^3.day by reducing the HRT from 14 hours to 3.8 hours.

Similarly, the effects of an organic shock loading without an increase in hydraulic loading was carried on two separate occasion. The initial 12 hours organic shock loading was performed using LS leachate, while the second organic shock loading was carried out for 10 hours using HS leachate. A step feed increase in the wastewater strength was produced by thoroughly mixing concentrated HS leachate with the wastewater in another feed tank, in order to raise the wastewater COD to approximately 4600 mg/l for the initial shock loading and 13500 mg/l for the second.

Analysis were carried out throughout the entire shock period and for a further period of time until steady state condition was achieved.

7.2.4 Zinc Addition for Toxicity Effect

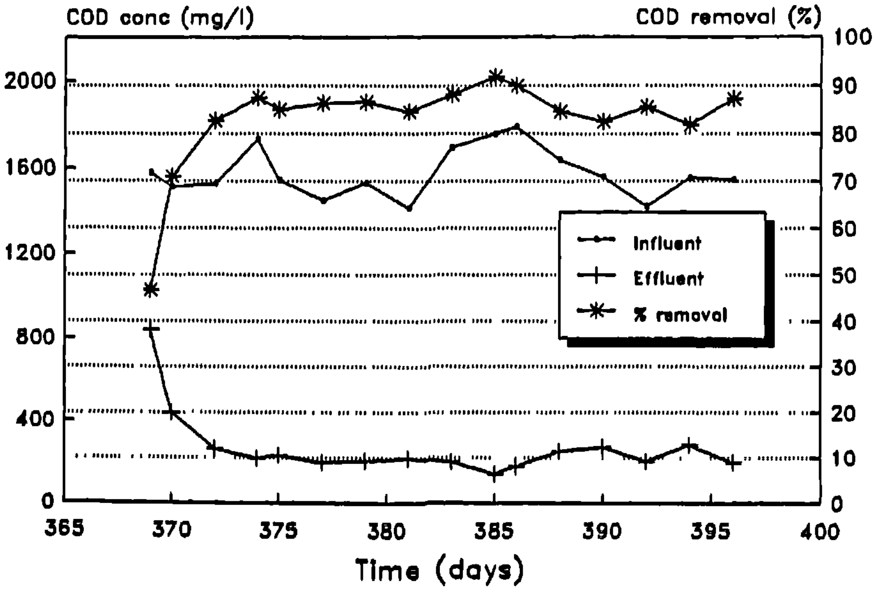
Heavy metal removal in term of Zn and iron (Fe) were continuously analyzed throughout the operational conditions. The tolerance of the RBC and the UAF units at different heavy metal concentrations and furthermore the comparison of the influence of heavy metal (in this case zinc) on both the reactors were of interest. In order to monitor the effect of zinc concentrations on the RBC and UAF treatment processes, Zn in the form of zinc nitrate was added in step to the wastewater. Performance was determined by monitoring the COD removal in both the RBC and the UAF.

7.3 Results and Discussion

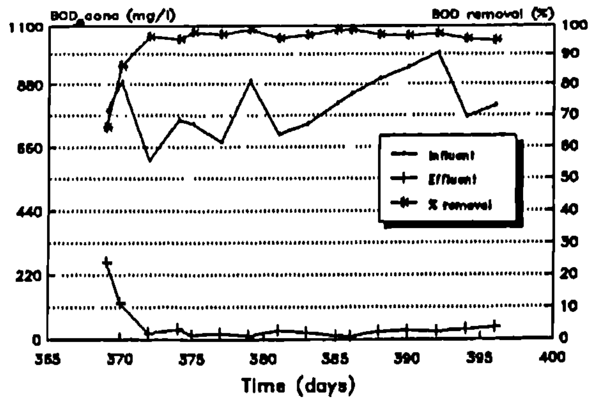
7.3.1 Response to Restart-Up Operation

The ability of the RBC and UAF to adapt to a period of shutdown can be observed from Figs. 7.1 to 7.3. The UAF performance was illustrated by

a. COD



b. BOD$_5$



c. TOC

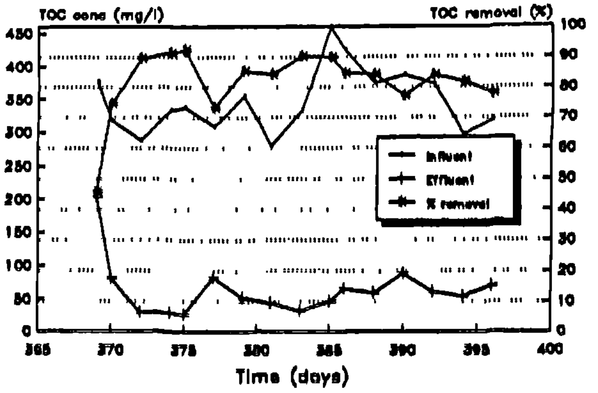


Figure 7.1: Performance of the RBC during Restart-Up (Phase IV Study)

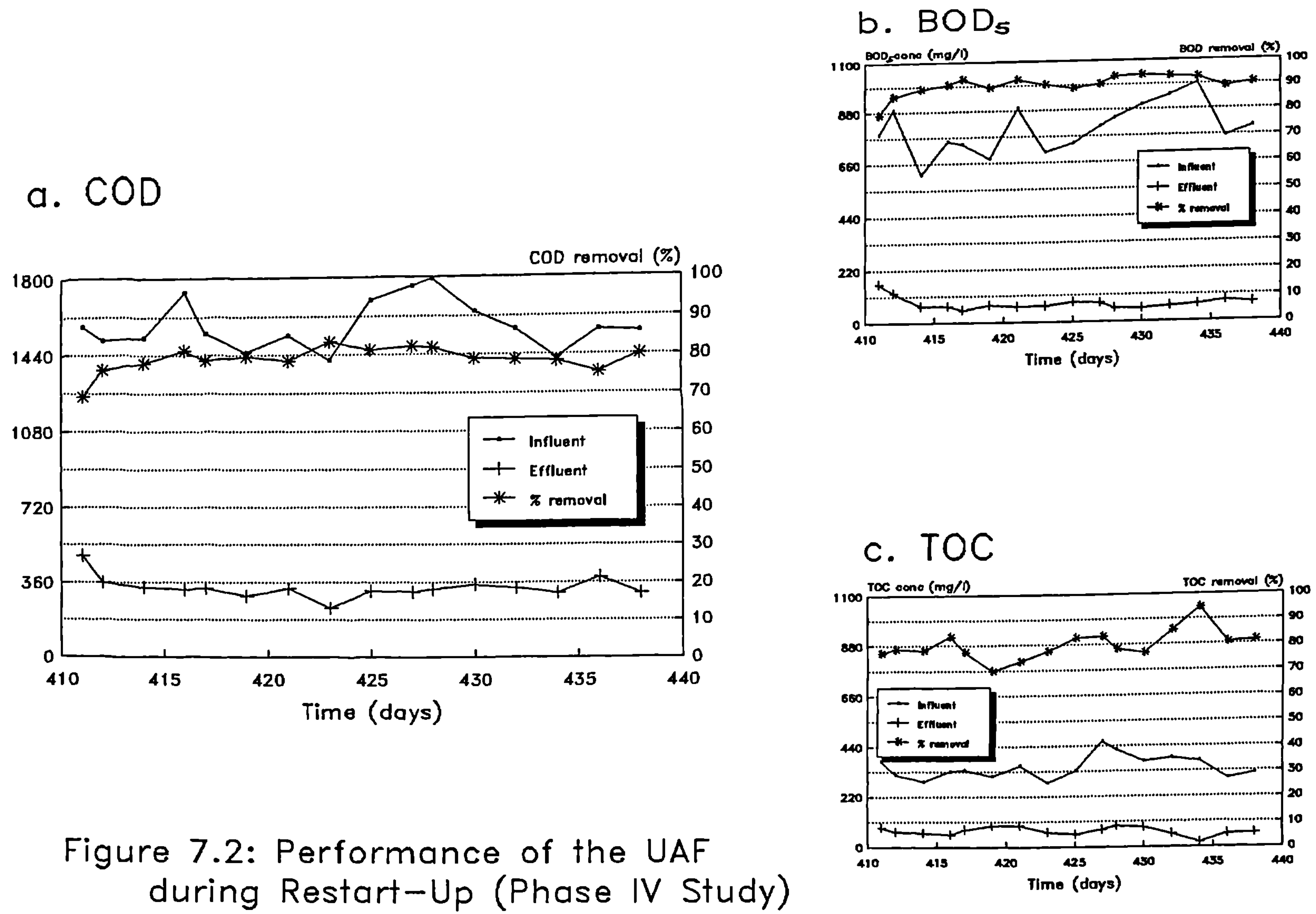


Figure 7.2: Performance of the UAF during Restart-Up (Phase IV Study)

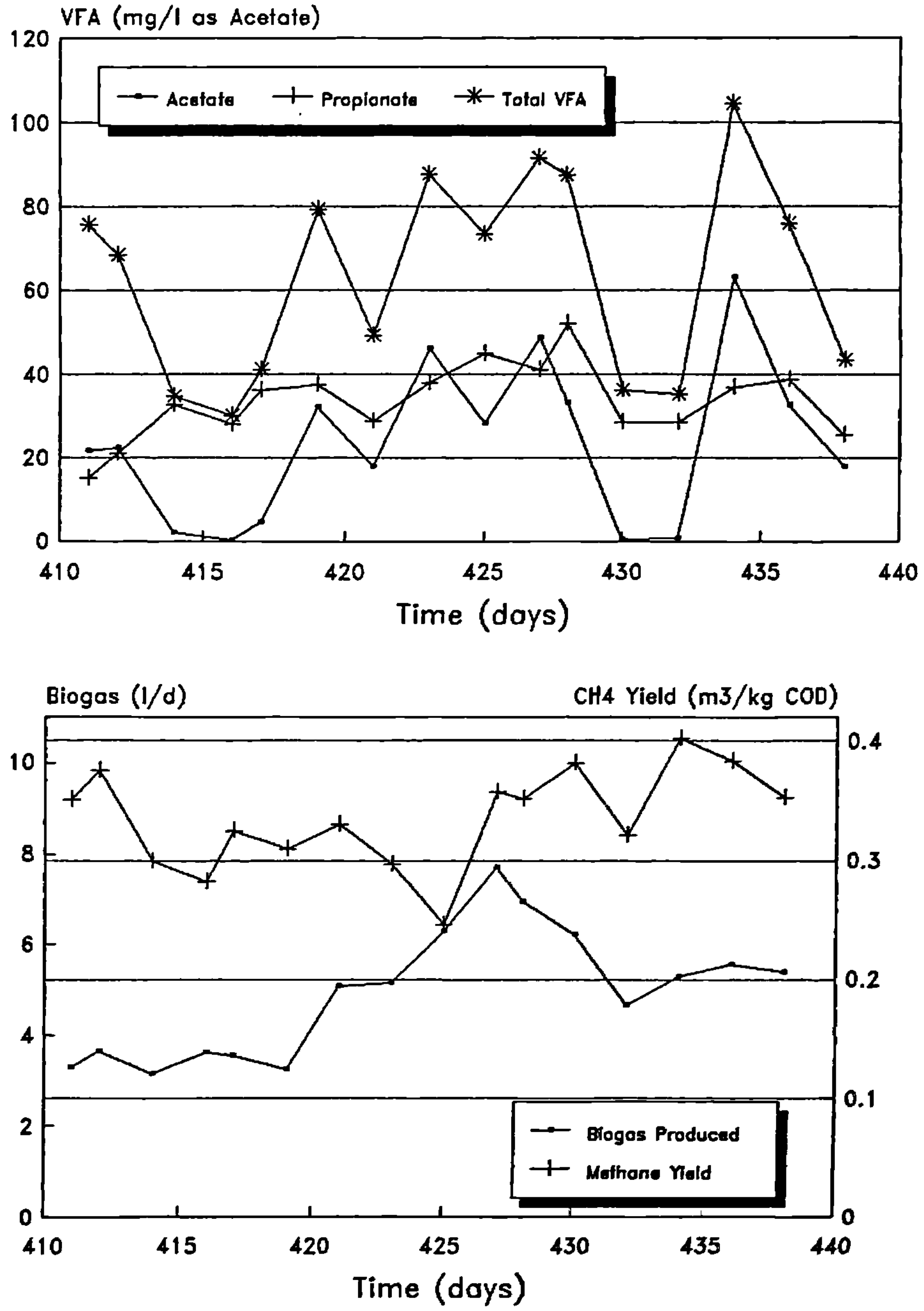


Figure 7.3: VFAs, Biogas Production and Methane Yield in the UAF during Restart–Up (Phase IV Study)

the total gas production data (Fig. 7.3b), which increased gradually from zero until it reached quasi steady state conditions in a matter of four days. From the analysis of effluent quality, it was clearly seen that both the RBC and UAF could withstand intermittent operation.

From Fig 7.1a, it can be seen that the RBC performed better than the UAF with a COD removal of 85% for the RBC while the UAF average 77% (Fig. 7.2a). This is probably attributed to the removal of sludge from the bottom of the RBC before restarting. As for the UAF, the fast recovery of the reactor without the need of a further reseeding indicated that intermittent shutdown did not seem to affect the anaerobic microorganisms in the reactor. The idle stage was found to enhance the UAF performance as seen by the increased COD removal compared to that before shutdown.

7.3.2 Results of Profile Studies

The profile studies carried out on the RBC and UAF are presented in Tables 7.1 and 7.2. The substrate distribution in the RBC is depicted in Fig. 7.4 while that for the UAF is shown in Figs. 7.5 and 7.6. From Fig 7.4, it can be seen that most of the substrate removal in the RBC took place in the first stage. After that only a small amount of removal was observed in the other stages. The performance increased with the days of operation, but a higher OLR tends to lower the substrate removal efficiency, as seen at day 270 (AOLR of 35.2 g COD/m^2.day) which exhibited higher effluent COD concentrations. The MLVSS distribution (Fig. 7.4d) in the RBC stages was found to be consistent in the stages, with the drum in the first stage accumulating the highest VSS concentration. This agreed with the finding of high substrate removal in the first stage.

Table 7.1: Results for Profile Studies of the RBC

Profile for LS Leachate Date 8/3/90 (Day 209)

VOLR = 7.5 kg COD/m^3.d (AOLR = 29.1 g COD/m^2.d)

Stage	Infl.	1st	2nd	3rd	4th	Effl.
pH	6.9	7.5	7.4	7.6	7.7	7.9
COD	1815	405	294	310	303	291
BOD	800	118	75	82	74	64
TOC	450	152	136	121	119	84
Sus. S	375	9360	7540	7750	3260	60
VSS	270	7814	5730	5093	1940	40

Date 8/5/90 (Day 270)

VOLR = 9.1 kg COD/m^3.d (AOLR = 35.2 g COD/m^2.d)

pH	7.1	7.6	7.4	7.8	7.9	8.1
COD	1815	380	326	295	308	242
BOD	800	134	118	92	103	63
TOC	450	126	104	120	115	109
Sus. S	375	17450	14720	13990	11470	55
VSS	270	12400	10300	9750	8415	44

Date 5/9/90 (Day 352

VOLR = 3.5 kg COD/m^3.d (AOLR = 10.6 g COD/m^2.d)

pH	7.1	7.2	7.1	7.1	7.1	6.9
COD	1815	370	307	286	290	271
BOD	800	106	72	54	50	33
TOC	450	141	124	125	123	88
Sus. S	375	18540	15840	16080	16680	152
VSS	270	11900	10320	10600	10800	118

Table 7.2: Results for Profile Studies of the UAF

Profile for LS Leachate Date 8/3/90 (Day 251)

VOLR = 2.6 kg COD/m^3.d (AOLR = 14.8 g COD/m^2.d)

	AVG					
Height	0.0	0.5	2.0	3.5	5.0	6.5
pH	6.9	6.8	6.9	6.9	6.9	7.1
acetate	315	17	18	23	62	73
propionate	105	27	42	46	49	19
total VFA	550	72	60	70	111	132
COD	1815	780	465	530	520	391
BOD	800	261	137	132	122	76
TOC	450	145	118	128	126	121
Sus. S	375	9938	1384	270	236	50
VSS	270	7631	923	184	175	30

Date 8/5/90 (Day 312)

VOLR = 4.0 kg COD/m^3.d (AOLR = 23.2 g COD/m^2.d)

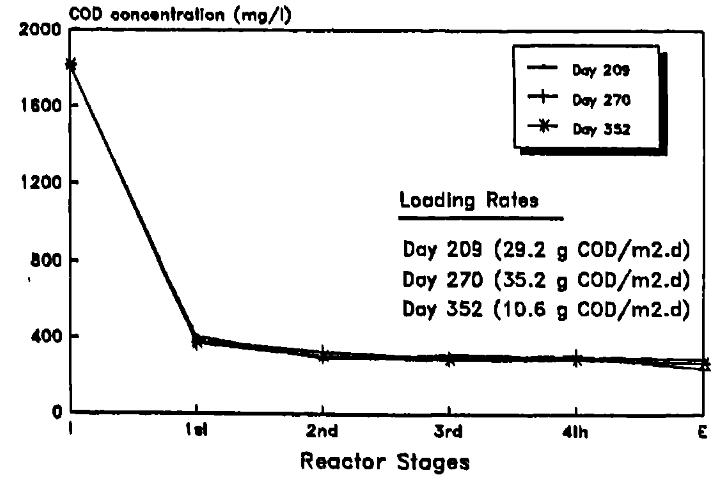
pH	7.0	6.9	6.9	6.9	6.8	6.9
acetate	315	17	21	27	30	36
propionate	105	33	30	33	33	34
total VFA	550	50	52	61	66	70
COD	1815	490	400	376	372	355
BOD	800	168	122	97	91	73
TOC	450	90	74	38	49	103
Sus. S	375	13367	4567	326	424	95
VSS	270	10680	3585	230	304	75

Date 5/9/90 (Day 394)

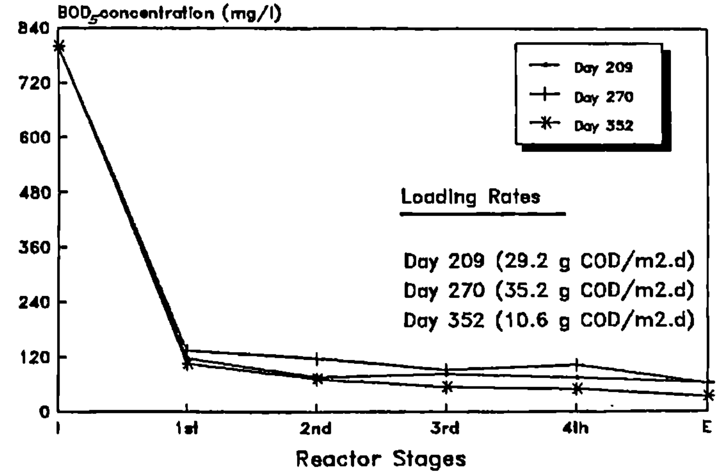
VOLR = 2.7 kg COD/m^3.d (AOLR = 15.3 g COD/m^2.d)

pH	7.0					6.9
acetate	315	4	4	5	4	0
propionate	105	23	23	21	22	29
total VFA	550	27	27	26	26	35
COD	1815	462	311	327	342	333
BOD	800	235	126	118	96	62
TOC	450	159	137	93	125	56
Sus. S	375	15700	6480	252	272	152
VSS	270	12420	4940	156	148	118

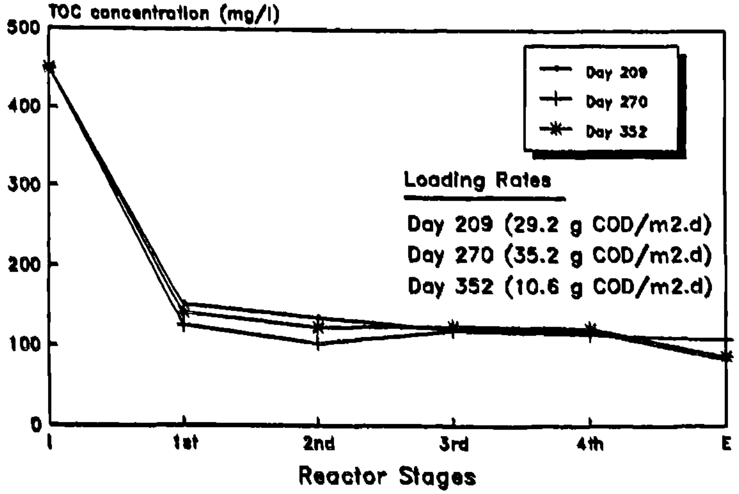
a. COD



b. BOD$_5$



c. TOC



d. VSS

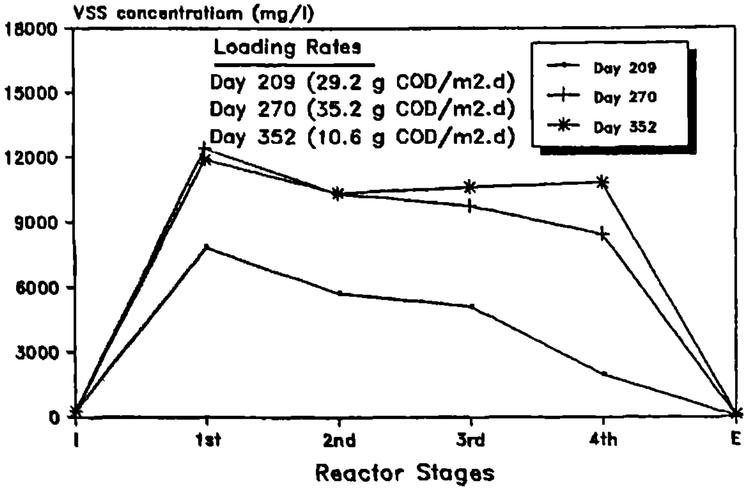


Figure 7.4: Profile Study in RBC Phase IV

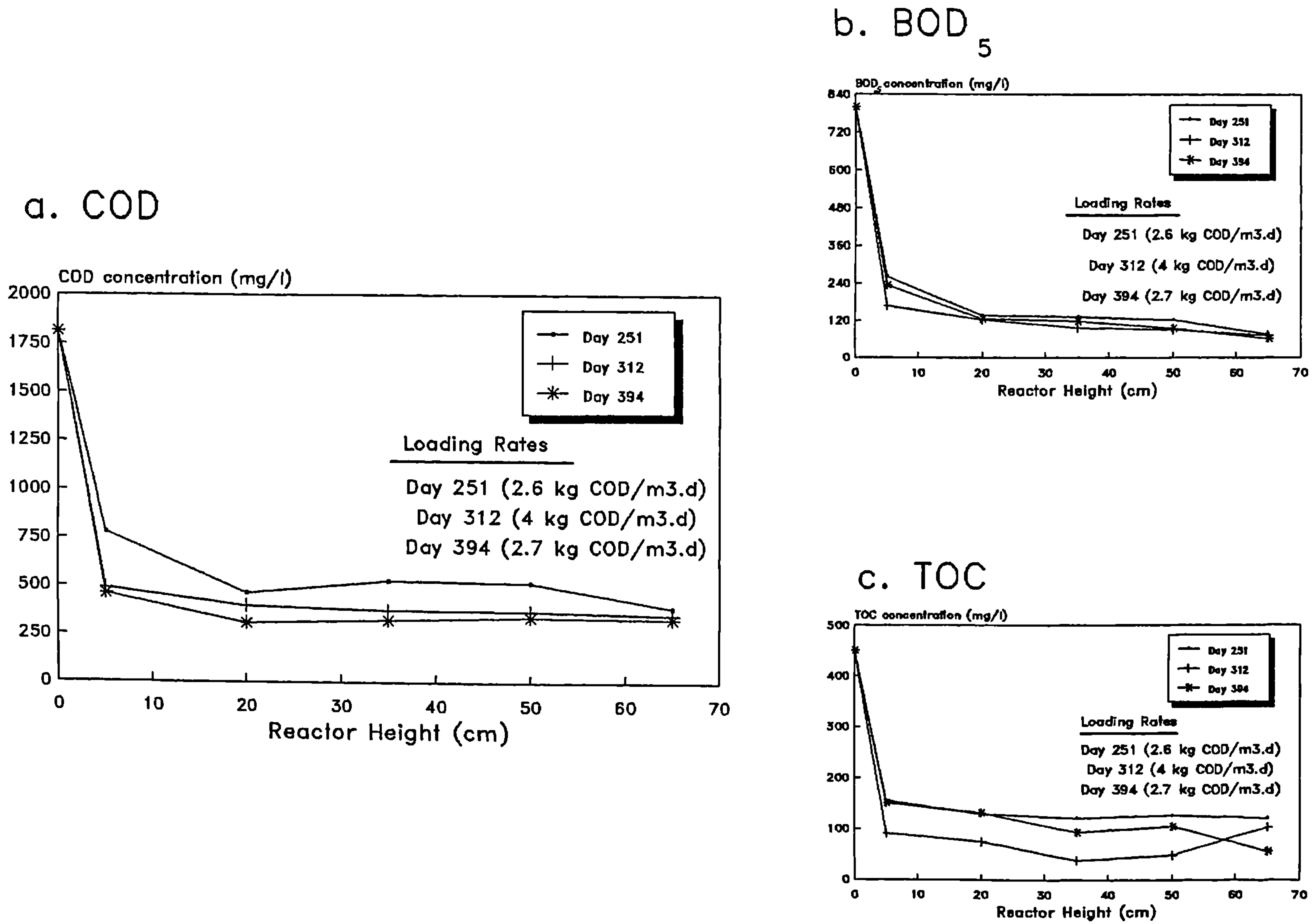
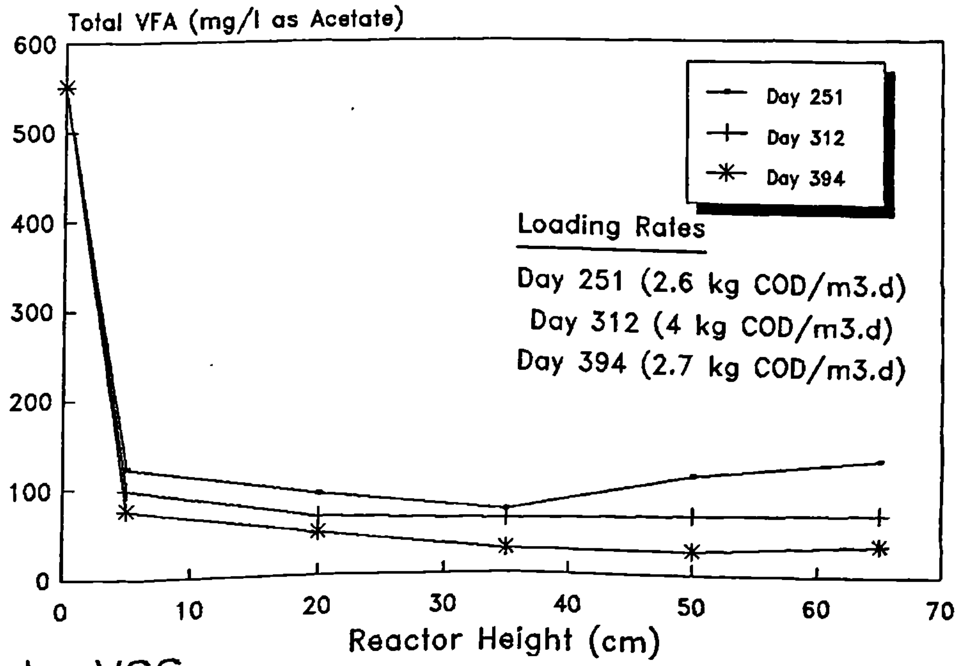


Figure 7.5: Profile Study in UAF Phase IV

a. VFAs



b. VSS

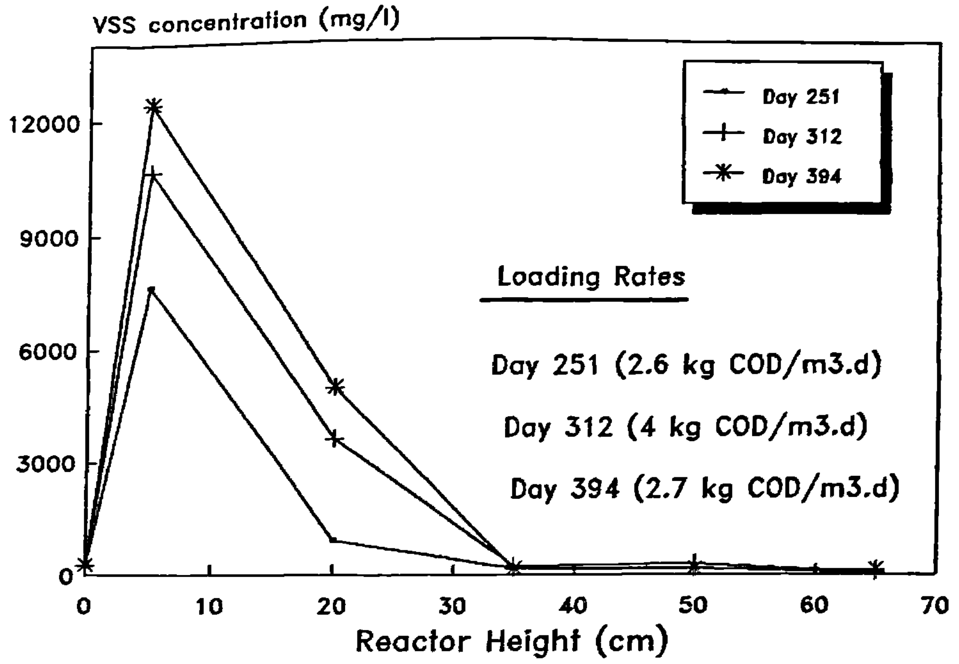


Figure 7.6: Profile Study in UAF Phase IV

In all the work which has been carried out on the UAF it has been found that most of the bacterial activity took place in the lower part of the filter i.e the bottom 50 cm. This is indicated by the high reduction in substrate removal (Fig. 7.5) in the lower one quarter of the UAF section. The microbial activity resulted in an overall higher concentration of VFAs (Fig. 7.6a) being produced in this zone when compared to the other reactor heights. This contributed to the lowering of the pH. However the initial high VFAs at the bottom was not quite sufficient to cause any problem in the UAF performance, since addition of external buffering in the form of sodium bicarbonate increased the bicarbonate alkalinity in the UAF. The bicarbonate alkalinity produced in the later stages of UAF together with the high biomass present in the packing material help to alleviate the need of excessive buffer addition. This was observed from the low level of VFAs production in the reactor heights, since most of the VFAs were converted to methane gas by the biomass entrapped in the high voidage support media.

Dahab and Young (1982) made a comparison between the COD profiles and biological solids distribution profiles, to determine the association between the two parameters. It was observed that the high concentrations of biological solids (Fig 7.6b) in the bottom section coincided with the rapid COD removal (Fig. 7.5a) at the lower one quarter of the reactor height.

7.3.3 Biological Solids Accumulation

At the end of the Phase III, one each of the RBC and UAF units were dismantled; while the other RBC and UAF units were dismantled at the very end of the entire study period. The biological solids concentration in the

RBC and UAF were determined for both attached and suspended solids. The results for the biological solids accumulation in the Phase III and at the end of entire study are listed in Tables 7.3 and 7.4 respectively.

The attached solids were seen to be equally distributed in each stage of the RBC, with a slightly higher concentration in the first stage. The MLVSS distribution compared well with the results obtained for the profile study, with the highest MLVSS concentration in the first stage and the lowest concentration in the fourth stage. Results of the analysis for the biological solids distribution at the end of Phase III showed that about 150 g (equivalent to 175 g TS/m^2 drum area) of dry total solids were attached to the drums (of which 65% total solids were volatile) and 25 g were in the MLSS (7000 mg/l).

As for the results after the completion of study, the dry total attached solids were found to be 330 g (equivalent to 380 g TS/m^2 drum area). Only 35% of the attached TS were volatile. The average MLSS was found to be 50 g/l, that is approximately 165 g. The MLVSS contributed to 25% of the MLSS. The probable reason was due to the inability of the RBC to accommodate the high loading rate of the HS leachate applied onto the reactor during this stage of experimental study. A substantial decreased in the substrate removal efficiency was observed which might be due to the decrease of the active biomass in the MLSS.

Using the measured biological solids data for the period ending after Phase III, the SRT of the RBC was determined. The SRT was calculated following the procedure employed by Saunders et al (1980). The calculation was carried out using values for total attached and suspended volatile

Table 7.3: Biological Solids Accumulation in Phase III

a) For RBC at the end of Day 223

Stage No.	Suspended TS		Suspended VTS	
	(mg/l)	(gm)	(mg/l)	(gm)
1	9568	7.9	6185	5.1
2	8103	6.7	5834	4.8
3	7250	6.0	4638	3.8
4	3762	3.1	2257	1.9
Avg MLSS	7171	23.7	4729	15.6

Stage No.	Attached TS		Attached VTS	
	(g/m2)	(gm)	(g/m2)	(gm)
1	216.9	46.0	126.2	26.8
2	172.6	36.6	114.3	24.2
3	147.0	31.2	105.8	22.4
4	155.3	32.9	102.6	21.8
Total		146.7		95.2
Total TS		170.3		110.8

b) For UAF at the end of Day 265

Port Ht. (mm)	Suspended TS		Suspended VTS	
	(mg/l)	(gm)	(mg/l)	(gm)
50	10564		7923	
200	2196		1480	
350	379		282	
500	382		275	
Avg MLSS	3380	21.3	2490	15.7

Port Ht. (mm)	Attached TS		Attached VTS	
	(g/m2)	(gm)	(g/m2)	(gm)
50	11.4	34.1	9.0	27.1
200	3.4	12.0	2.2	7.7
350	1.1	3.9	0.8	2.9
500	0.9	3.0	0.6	2.1
Total	4.2	52.9	3.2	39.8
Total TS		74.2		55.5

Table 7.4: Biological Solids Accumulation in Phase IV

1. For RBC at the end of Phase IV

Stage No.	Suspended TS		Suspended VTS	
	(mg/l)	(gm)	(mg/l)	(gm)
1	59612	49.2	15091	12.5
2	47685	39.3	13006	10.7
3	41418	34.2	9964	8.2
4	49661	41.0	11576	9.6
Avg MLSS	49594	163.7	12409	41.0

Stage No.	Attached TS		Attached VTS	
	(g/m2)	(gm)	(g/m2)	(gm)
1	410.9	87.1	138.4	29.3
2	389.7	82.6	128.6	27.3
3	400.0	84.8	148.1	31.4
4	363.4	77.0	117.2	24.9
Total		331.6		112.9
Total TS		495.2		153.8

2. For UAF at the end of Phase IV

Port Ht. (mm)	Suspended TS		Suspended VTS	
	(mg/l)	(gm)	(mg/l)	(gm)
50	34670		24268	
200	31455		18590	
350	14686		8573	
500	1898		1015	
Avg MLSS	20677	130.3	13112	82.6

Port Ht. (mm)	Attached TS		Attached VTS	
	g/20unit	(g)	g/20unit	(g)
50	22.0	66.1	14.0	42.0
200	9.3	32.4	6.4	22.4
350	5.1	17.8	3.5	12.3
500	3.8	13.3	2.3	8.0
Total	10.0	129.7	6.6	84.7
		259.9		167.3

solids in the RBC , M_T, and the average rate of volatile solids wasted, r_w. SRT was then calculated using Equation 7.1.

$$SRT = \frac{M_T}{r_w} \qquad \text{.............}7.1$$

The rate of wastage of volatile solids was equal to the summation of the effluent VSS plus VS periodically scraped and the MLVSS removed for the profile study. The value of SRT was found to be 14 days.

As for the UAF, a high percentage of solids was found at the bottom section in both the results after Phase III and the results at the end of the study period. These results confirmed the finding of the profile study. On dismantling the reactor, it was observed that the attachment was very thin, but the suspended growth was settled in a compacted form inside the media's void spaces. The media which were located at the bottom of the reactor appeared to be blocked. The dry weight of total attached and suspended TS in the UAF for results after Phase III was 75 g (of which 75% were volatile); whereas the total attached and suspended TS after the completion of study was 260 g (of which 65% were volatile). The average SRT in the UAF for the results after Phase III was calculated according to Young and McCarty (1969) and was found to be 66 days.

The high percentage of volatile solids in the UAF at the end of the study when compared to the volatile solids in the RBC indicated the stability of the UAF during Phase IV study. Although the study was discontinued with the highest OLR of 13 kg COD/m^3.day for the UAF, the active biomass in the UAF could still probably be able to remove the substrate in the leachate with further increases in organic loading rate; whereas the RBC would probably fail as seen from the drastic reduction in COD removal efficiency for the RBC. The capability for retaining an active biomass (attached and suspended) and proper mixing are known to be crucial factors affecting the performance of a biological reactor. Mixing in the RBC is accomplished by the rotating drums, while the distribution of flow in the UAF assists in proper mixing. This helps to maintain the required contact between the biological solids and wastewater. The gas and liquid up-flow velocity, foaming and sludge settleability are factors which affect the retention of suspended growth. In attached growth systems, the specific surface area and roughness of the surface media are very important.

7.3.4 Performance under Shock Loading

Effluent samples taken from the RBC and UAF during the shock loading operations were analyzed and the results are tabulated in Tables D.1 to D.3 in Appendix D. The operating conditions and performance during shock loading are given in Tables 7.5 to 7.7 The variation in effluent substrate and substrate removal are illustrated in Figs. 7.7 to 7.10 for the RBC and Figs 7.11 to 7.18 for the UAF.

- Volumetric Shock Loading

The COD removals for both the RBC and the UAF decreased slight y. The BOD removals in the RBC were found to decreased more when compared to the UAF, whereas TOC removals in both the RBC and the UAF were

Table 7.5: Influent Leachate during Shock Loading Studies

Expt. No.	Influent Leachate	Average Influent Concentration COD	BOD (mg/l)	TOC	Duration (hours)
Volumetric Shock Loading					
1	LS leachate	1400	500	330	12
2	diluted HS leachate	1490	745	370	24
Organic Shock Loading					
1	LS leachate	1565	545	350	12
		4650	2065	1075	
2	diluted HS leachate from	6120	3575	1600	
	increase				10
	to	13550	7820	3290	

Table 7.6: Performance of the RBC during Shock Loading Studies

Volumetric Shock Loading	1st Run LS Leachate	2nd Run HS Leachate
Effluent		
COD	550	630
BOD	230	295
TOC	215	145
% removal		
COD	60	58
BOD	54	60
TOC	38	61

Organic Shock Loading	1st Run LS Leachate	2nd Run HS Leachate
Effluent		
COD	1080	4400
BOD	650	1570
TOC	430	960
% removal		
COD	77	67
BOD	69	80
TOC	60	71

Table 7.7: Performance of the UAF during Shock Loading Studies

Volumetric Shock Loading

	1st Run LS Leachate	2nd Run HS Leachate
Effluent		
COD	490	675
BOD	110	330
TOC	180	180
% removal		
COD	65	55
BOD	78	56
TOC	45	51

Organic Shock Loading

	1st Run LS Leachate	2nd Run HS Leachate
Effluent		
COD	890	3440
BOD	280	725
TOC	280	565
% removal		
COD	77	75
BOD	69	91
TOC	60	83

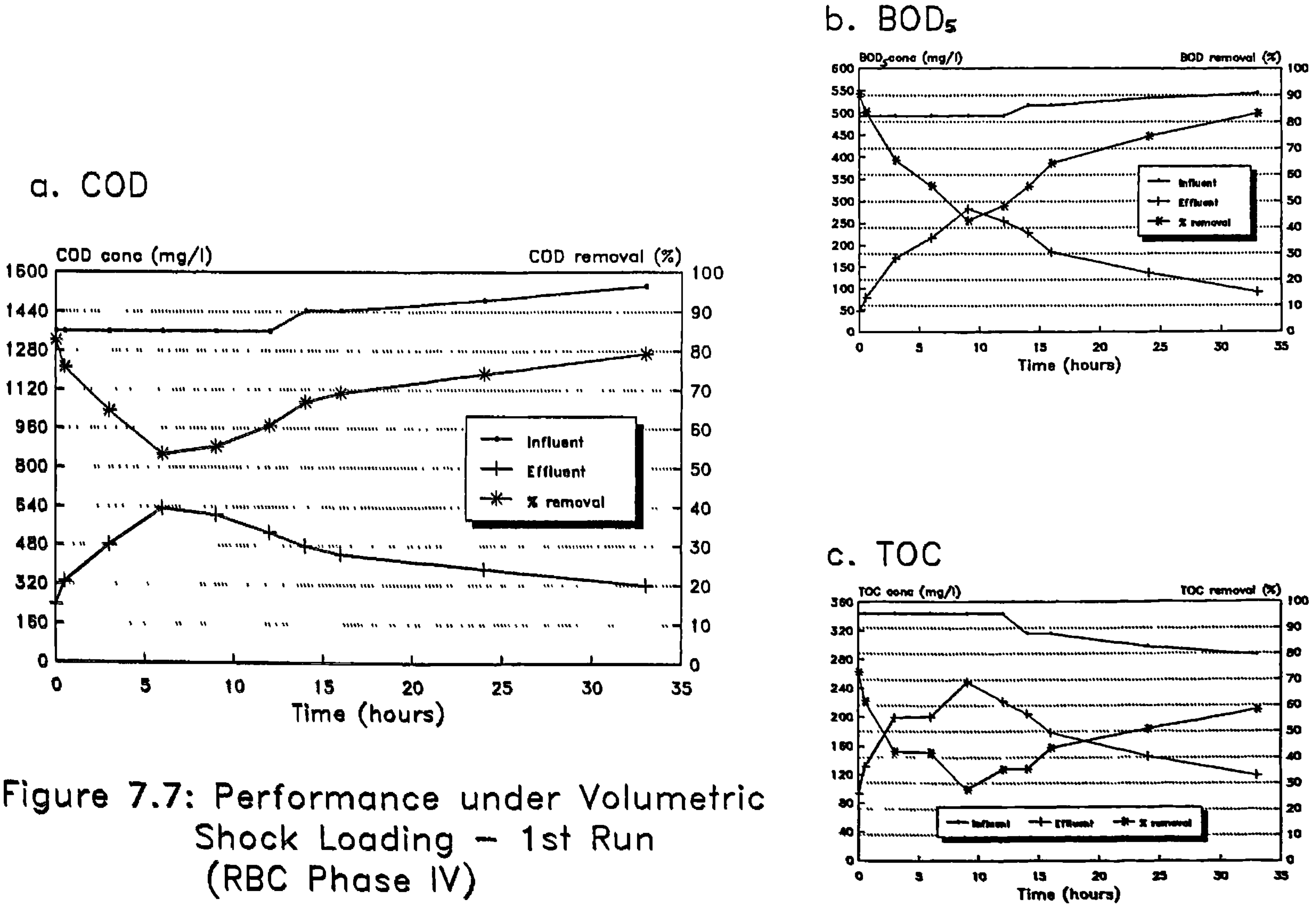


Figure 7.7: Performance under Volumetric Shock Loading – 1st Run (RBC Phase IV)

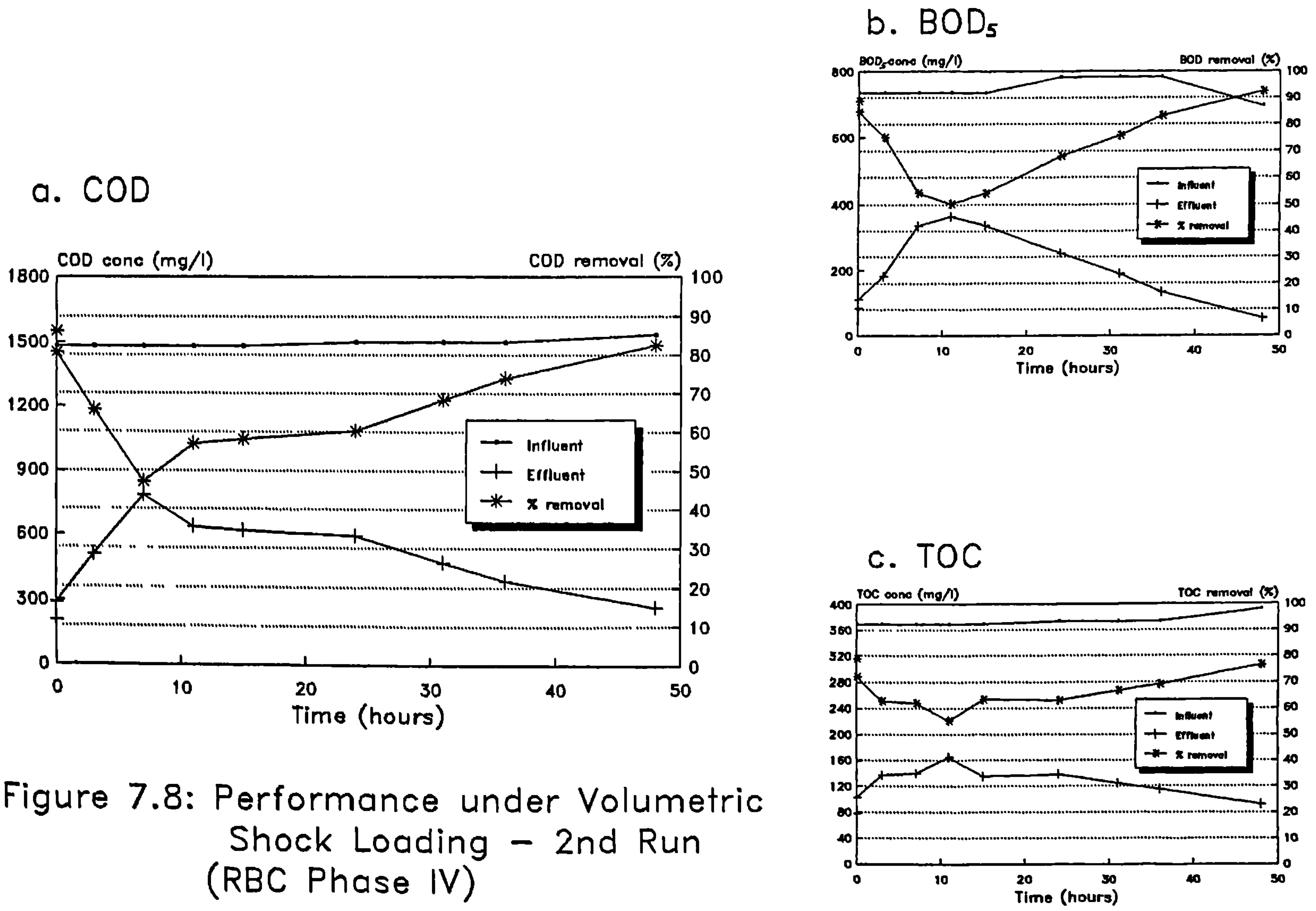


Figure 7.8: Performance under Volumetric Shock Loading – 2nd Run (RBC Phase IV)

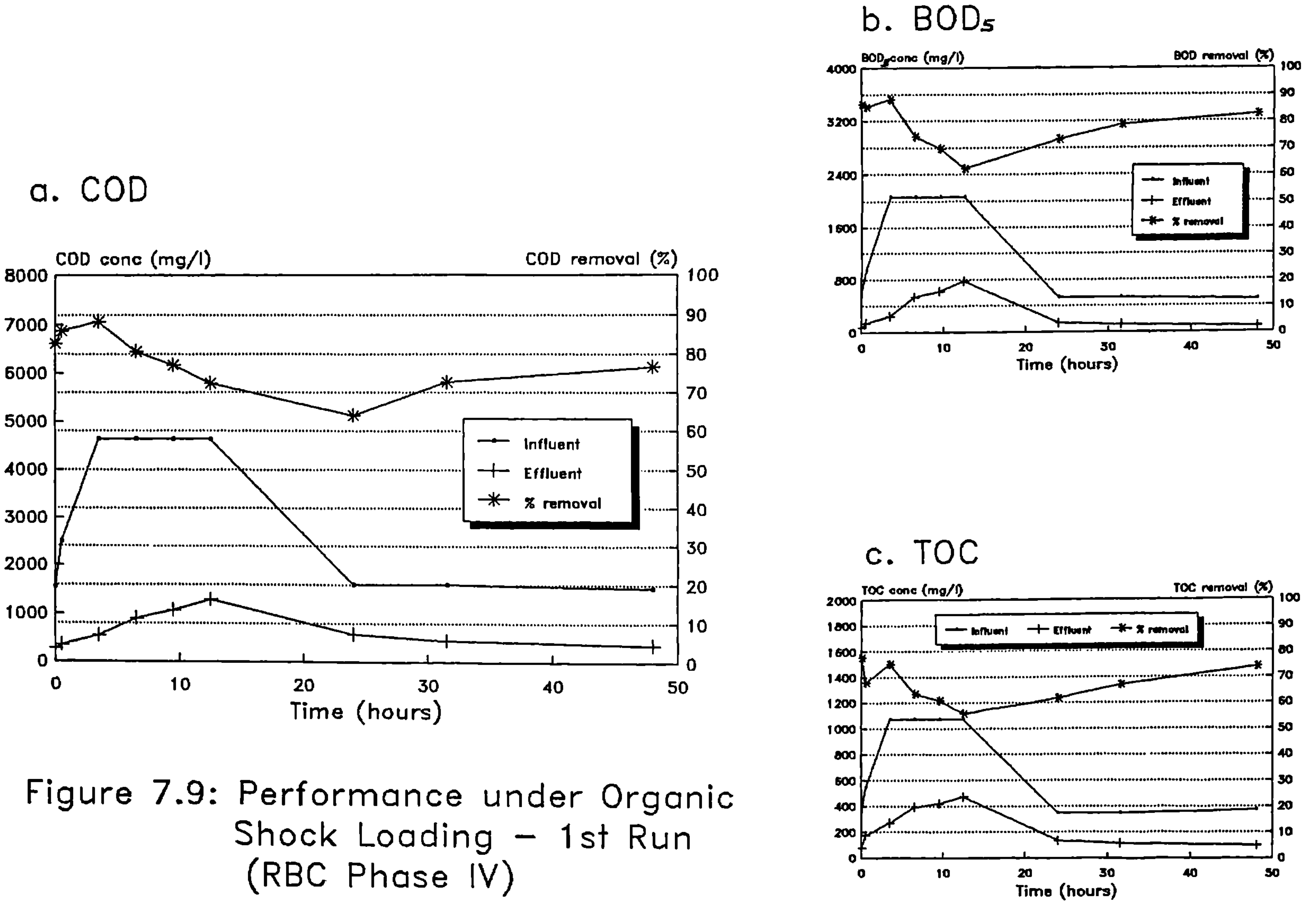


Figure 7.9: Performance under Organic Shock Loading – 1st Run (RBC Phase IV)

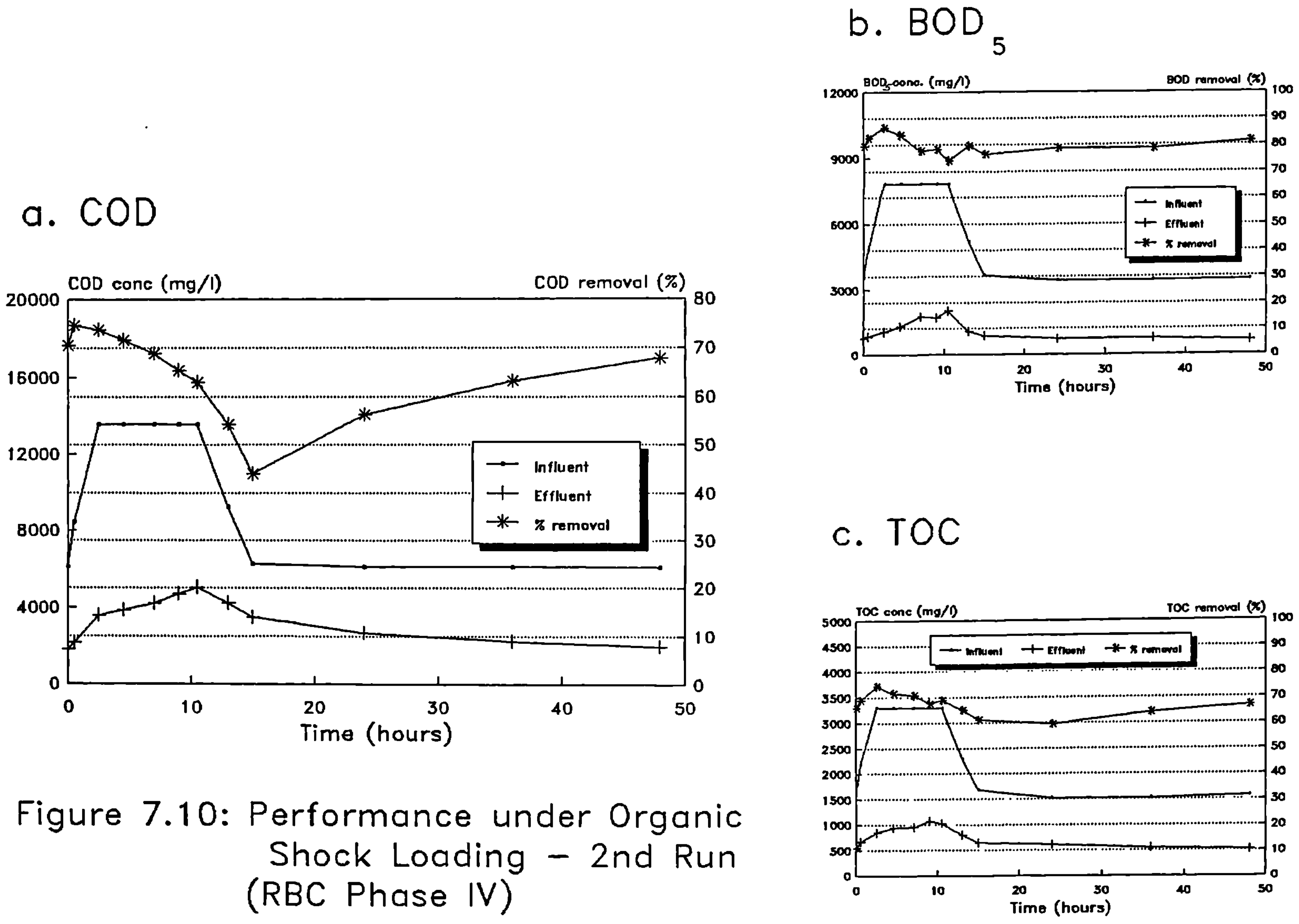


Figure 7.10: Performance under Organic Shock Loading - 2nd Run (RBC Phase IV)

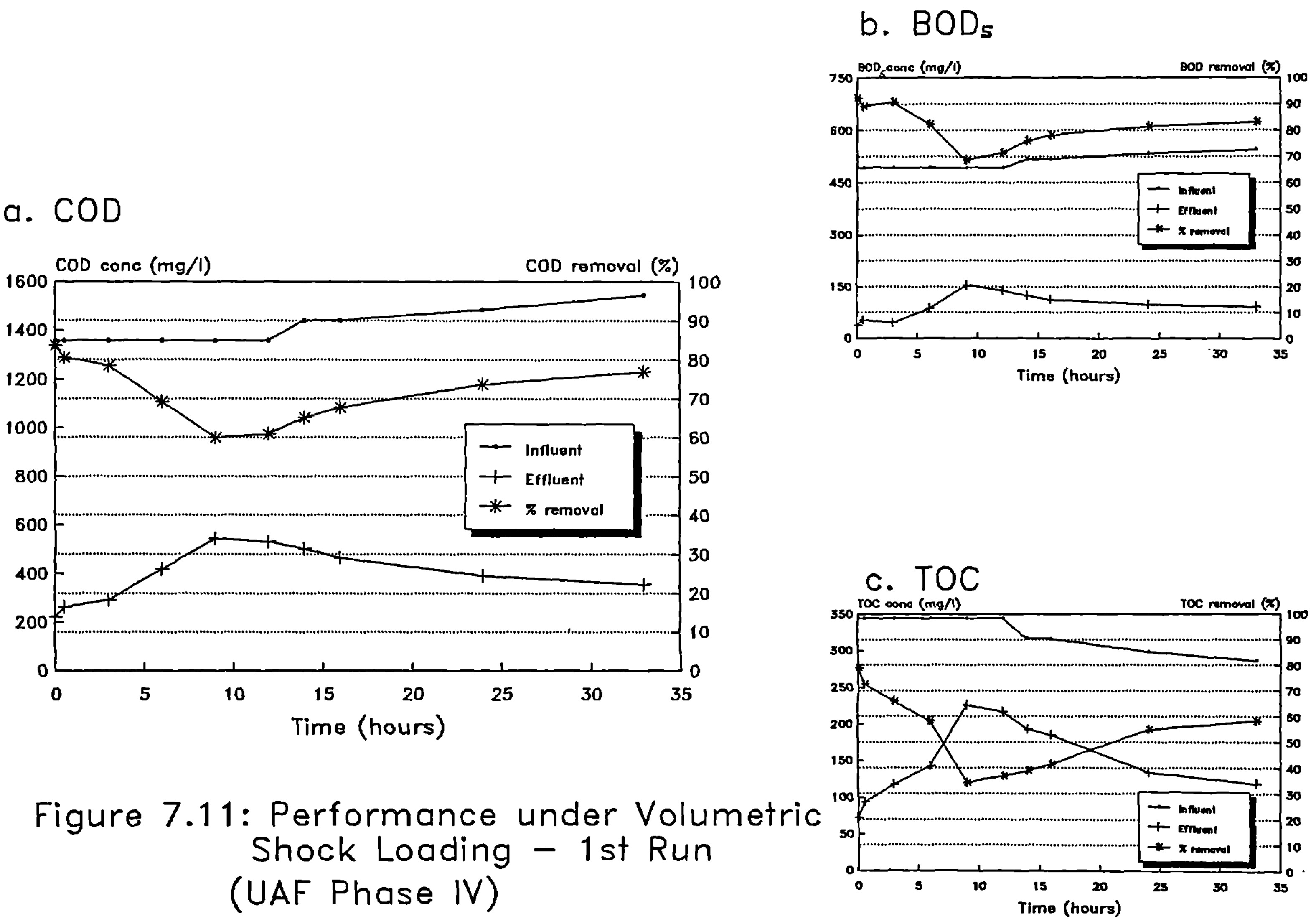


Figure 7.11: Performance under Volumetric Shock Loading – 1st Run (UAF Phase IV)

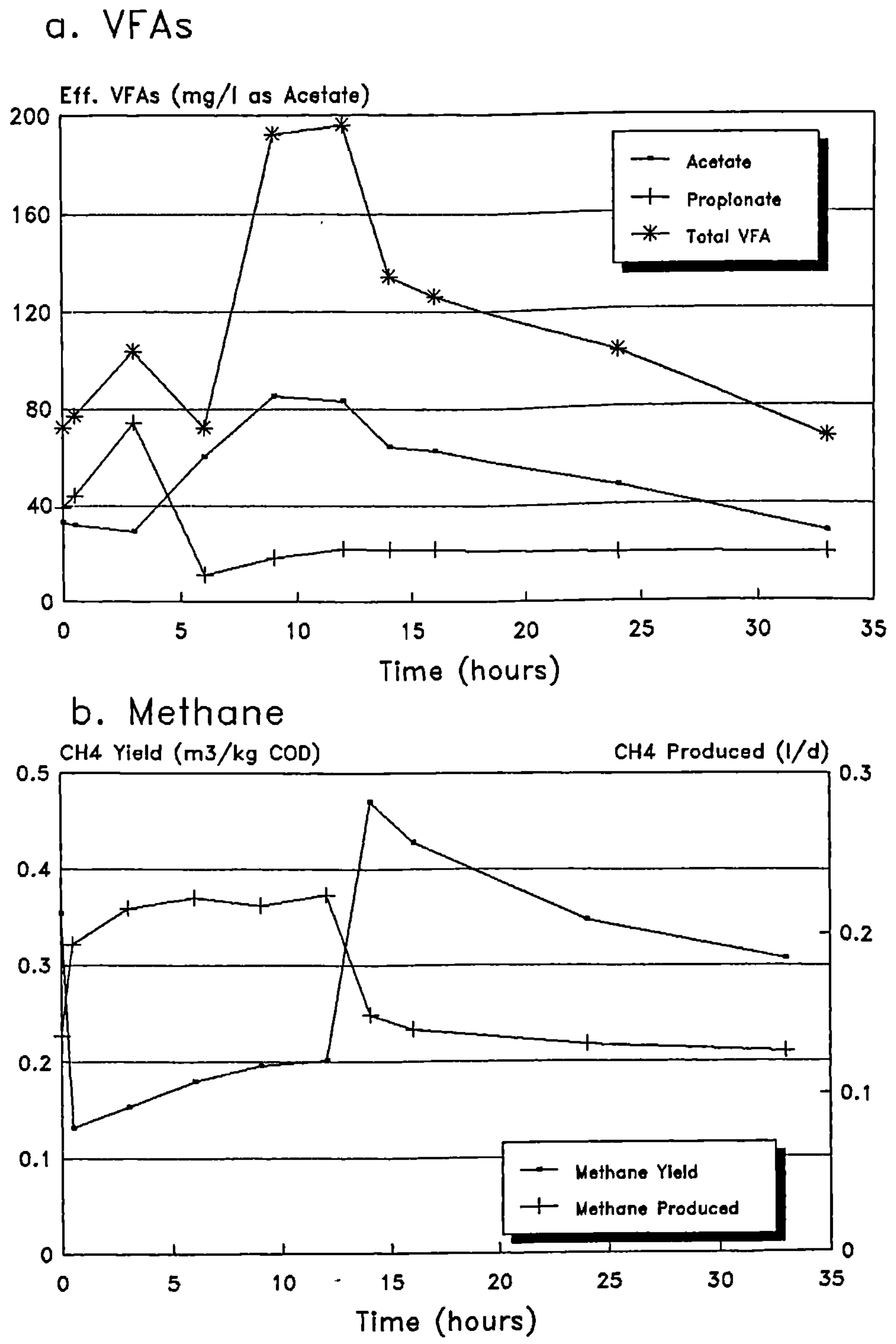


Figure 7.12: Performance under Volumetric Shock Loading – 1st Run (UAF Phasae IV)

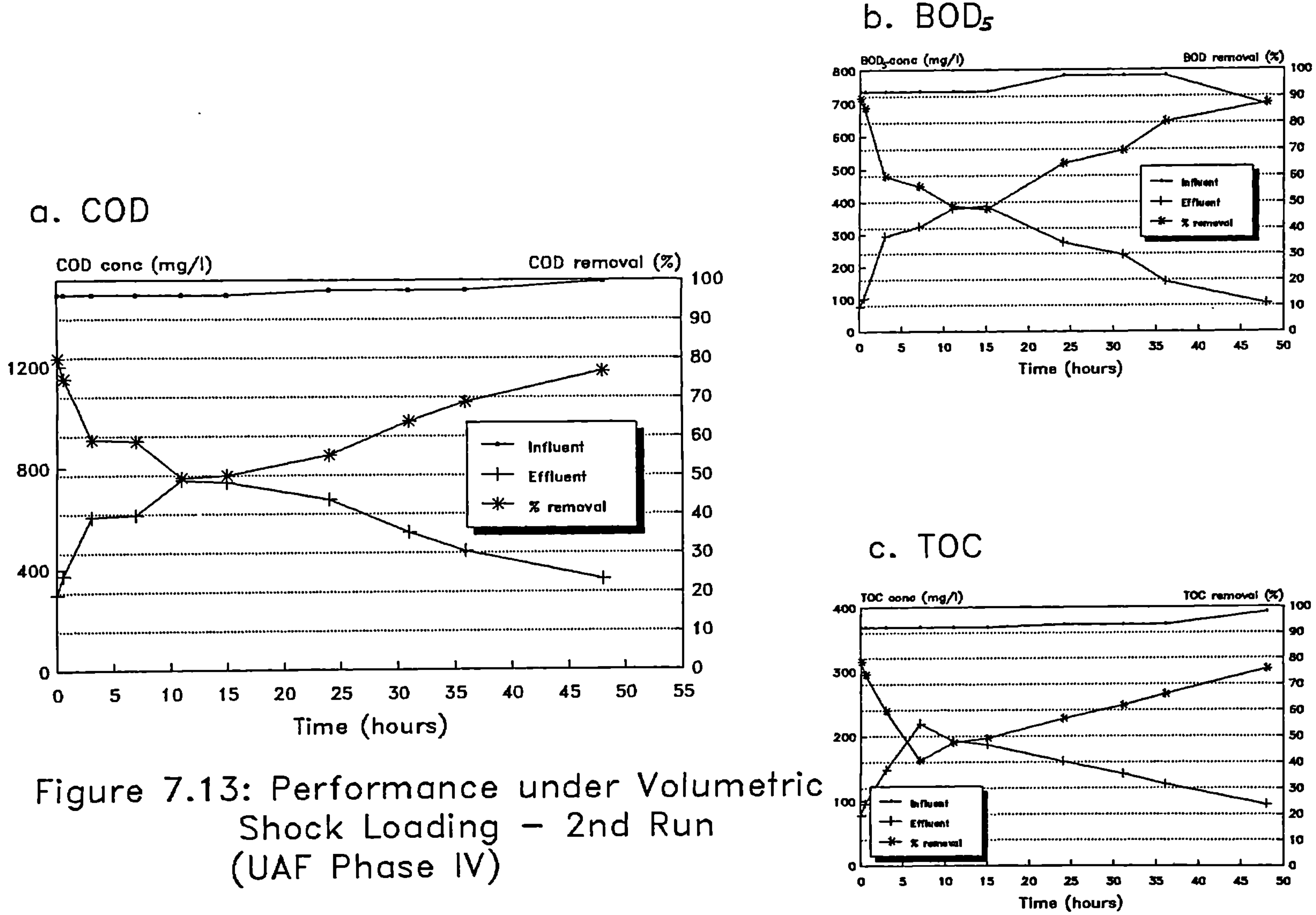
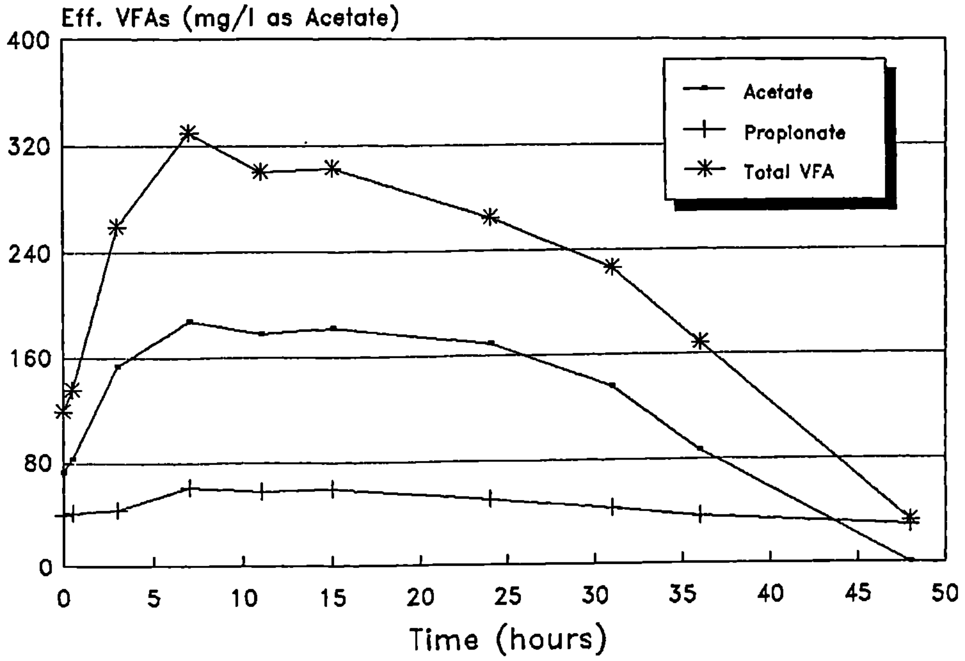


Figure 7.13: Performance under Volumetric Shock Loading – 2nd Run (UAF Phase IV)

a. VFAs



b. Methane

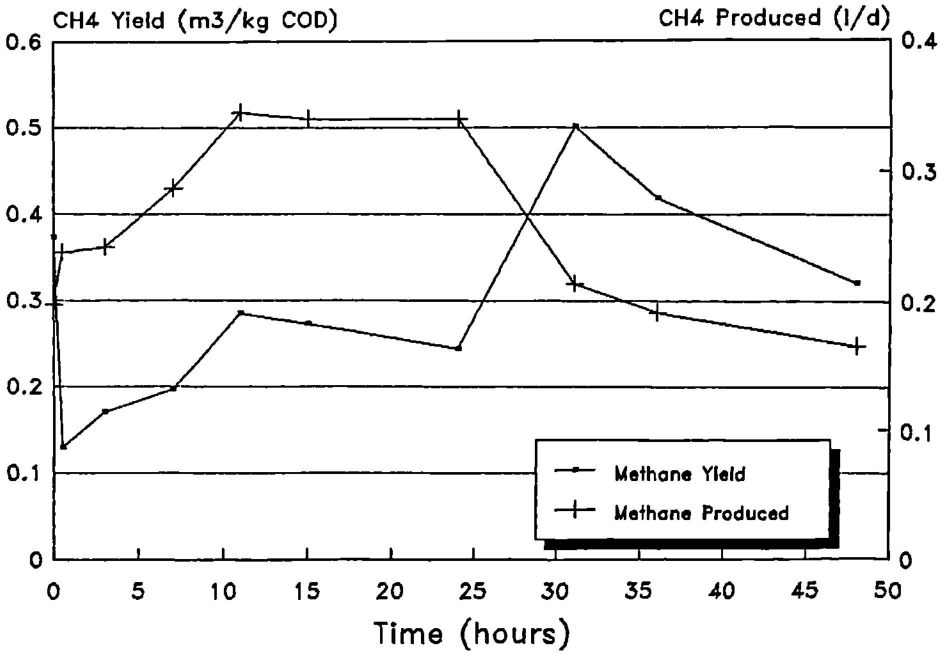


Figure 7.14: Performance under Volumetric Shock Loading - 2nd Run (UAF Phase IV)

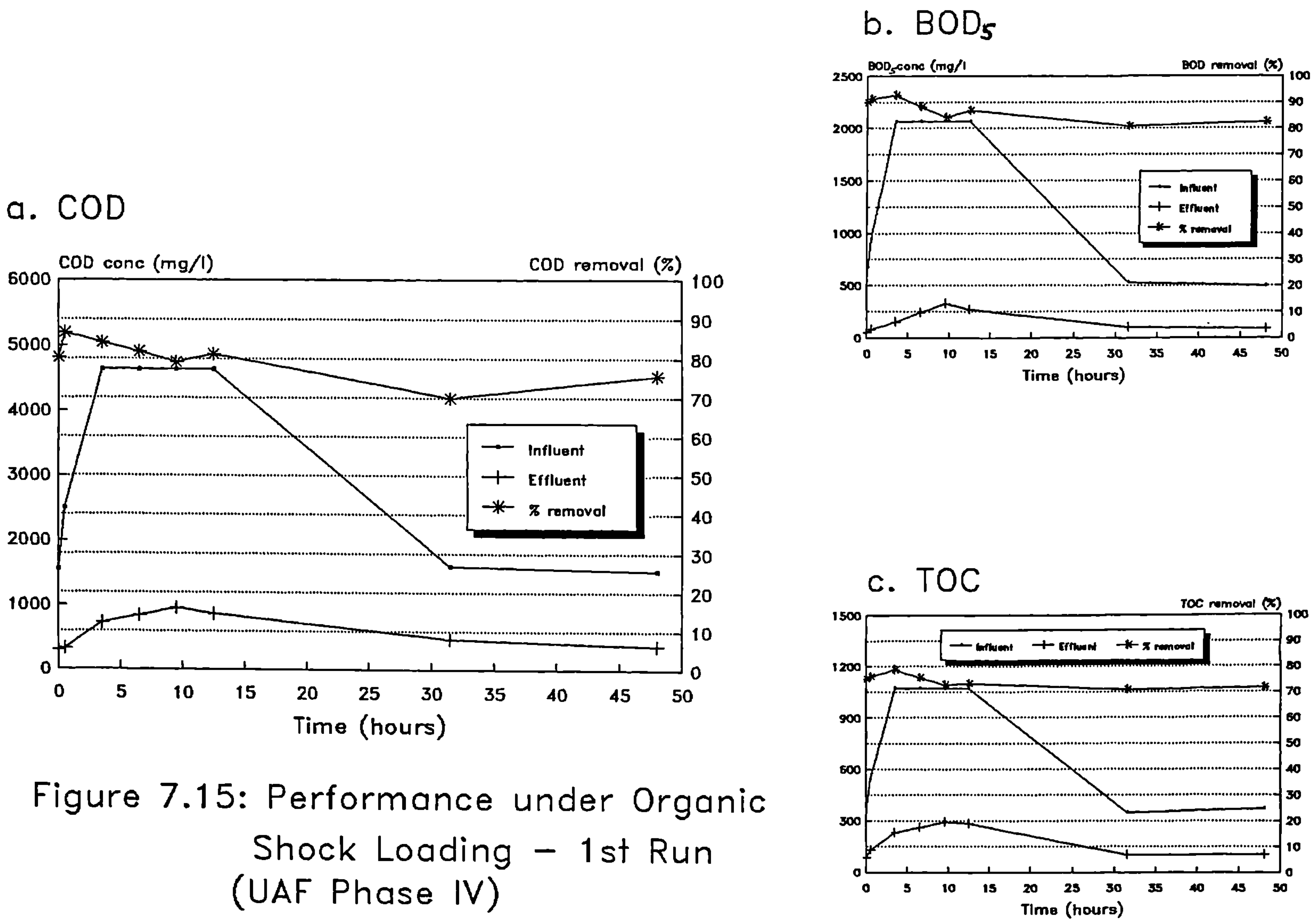
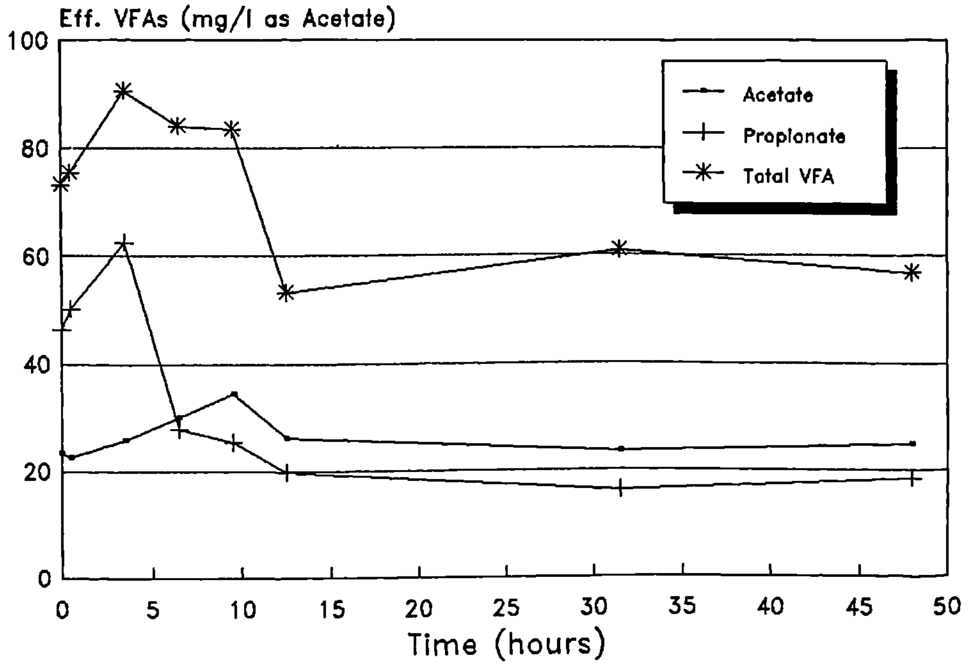
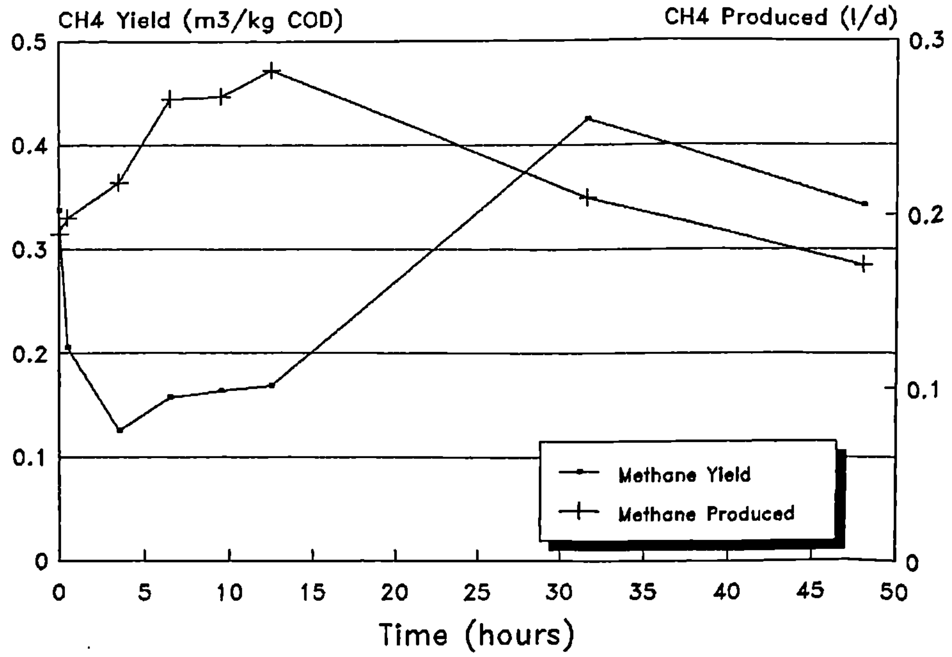


Figure 7.15: Performance under Organic Shock Loading – 1st Run (UAF Phase IV)



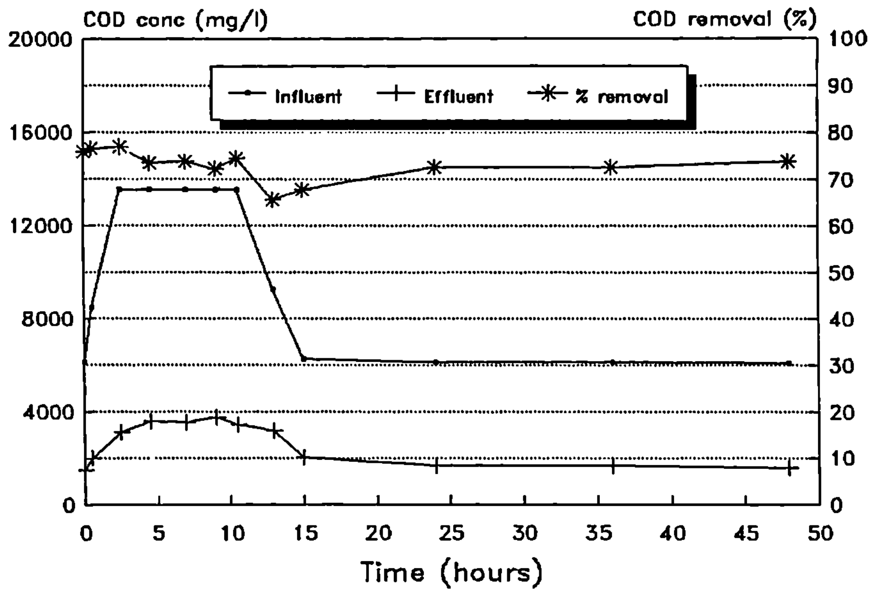
a. VFAs



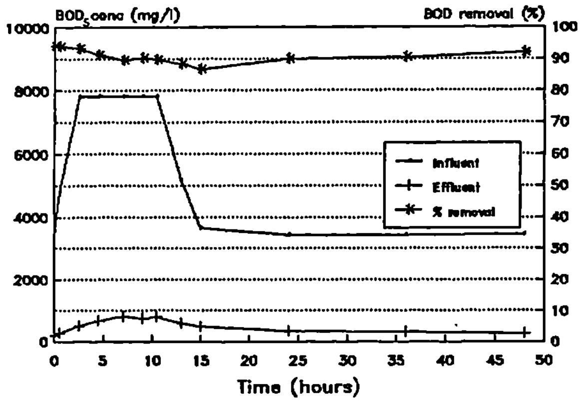
b. Methane

Figure 7.16: Performance under Organic Shock Loading – 1st Run (UAF Phase IV)

a. COD



b. BOD_5



c. TOC

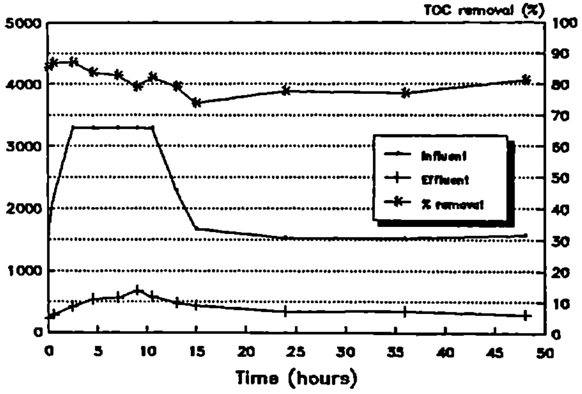
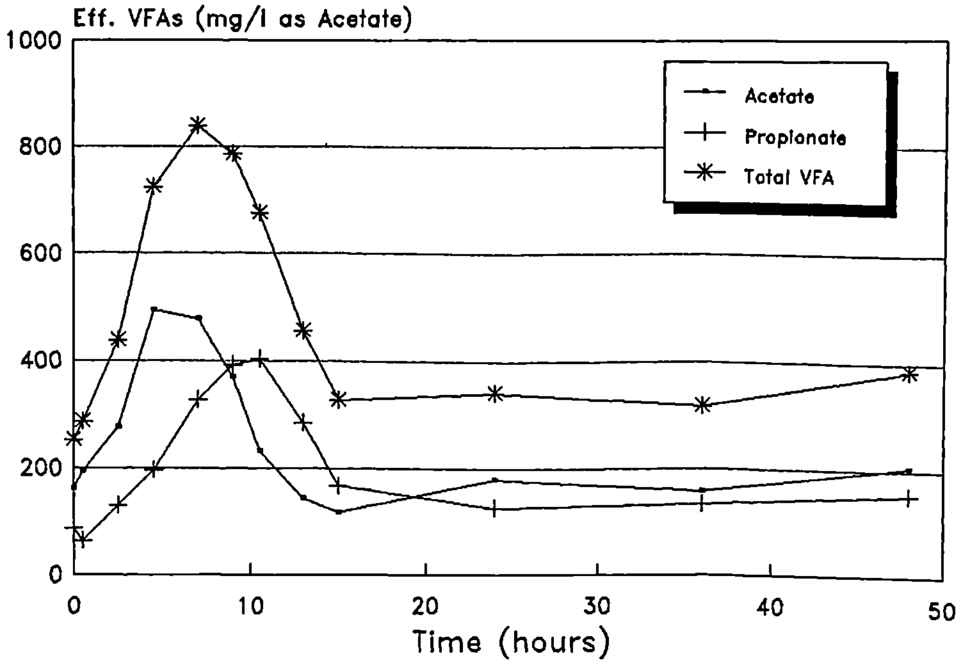


Figure 7.17: Performance under Organic Shock Loading – 2nd Run (UAF Phase IV)

a. VFAs



b. Methane

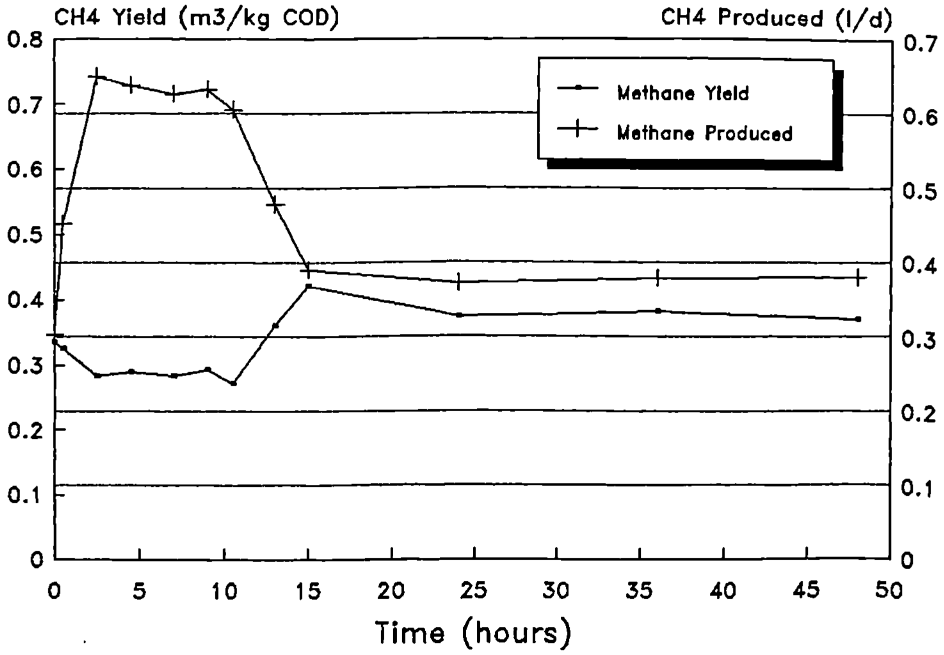


Figure 7.18: Performance under Organic Shock Loading – 2nd Run (UAF Phase IV)

significantly affected, falling to around 30% (Figs 7.7c and 7.11c for the RBC and the UAF respectively). Although the RBC performance appeared to be poorer than that of the UAF, where during the first run the lowest COD reduction in the RBC was 53% (Fig 7.7a) while that in the UAF was 60% (Fig 7.11a), the recovery was much more rapid in the RBC. Within 24 hours, both the RBC and the UAF were observed to have recovered from the shock loading, as indicated by the stabilized COD removal efficiency.

This lower COD reduction in the RBC was apparently due to the fact that the VOLR of the RBC was higher than that in the UAF, that is 18 kg COD/m^3.day, whereas the VOLR of the UAF was only 7 kg COD/m^3.day. Taking this into account, the RBC appeared to perform better than the UAF. Similarly, the results from the second run confirmed the above observation.

Comparing the results from the two runs, the performance for both the RBC and UAF fair slightly better in the second run. The second run was carried out after restarting of the reactors in the Phase IV study. A possible explanation for the RBC is perhaps due to the removal of sludge lying at the bottom of the tank prior to restarting, which enhanced the mixing potential of the biological solids. attached to the rotating drums. The period of shutdown allowed most of the VSS in the UAF to settle and thus during the shock loading, the VSS which had accumulated at the bottom of the UAF assisted in the build up of further biological solids and enhanced the substrates removal efficiency.

Effluent VFAs in the UAF during volumetric shock loading were found to increase from 72 mg/l to 200 mg/l in the first run and from 120 mg/l to

330 mg/l in the second run. Biogas and methane production rapidly increased which reduced the methane yield, reflecting the increase in VOLR and revealing the incumbent process instability. Methane yield decreased from 0.35 m^3 CH_4/kg COD removed to 0.15 m^3 CH_4/kg COD removed in the first run and from 0.37 m^3 CH_4/kg COD removed to 0.17 m^3 CH_4/kg COD removed in the second run. However, the overall process stability was maintained and the recovery period was short. Once a steady state condition was achieved, the VFAs decreased.

- Organic Shock Loading

The RBC was found to suffer more than the UAF during organic shock loading. The stress in the RBC was possibly brought about by the increased wastewater substrate concentration entering the RBC. The instability resulted in sloughing of the attached biological solids. Comparing the first and the second run, the performance during the second run was lower (44% COD removal) than the first run (64% COD removal). This is due to the very high leachate concentration (increased to 13500 mg/l COD concentration) in the second run while the first run was subjected to an increase of 4600 mg/l COD concentration. Despite the stress, the RBC recovered within 24 hours after the termination of shock loading.

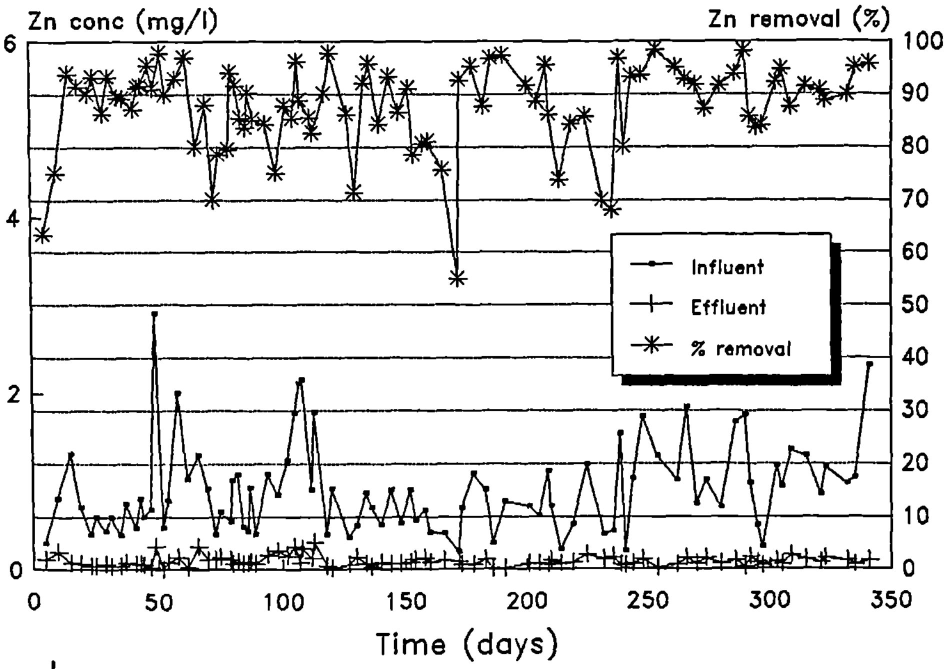
As for the UAF the increase at low VOLR brought about by increase in leachate concentration, i.e from 2.5 kg COD/m^3.day to 8 kg COD/m^3.day in the first run and from 5 kg COD/m^3.day to 11.5 kg COD/m^3.day in the second run, did not seem to affect the UAF performance. Although the biogas and methane production increased, indicating the increased VOLR, stability was maintained throughout the shock period. The UAF was capable of coping with the high build up of total VFAs in the second run (Fig 7.18a) of 820 mg/l. The two- to threefold OLR increase did not reduce the UAF performance when compared to the performance of the RBC.

7.3.5 Response to Zinc Toxicity

It is known that anaerobic bacteria, especially methanogens, are the most sensitive to any changes when compared to aerobic bacteria. Apart from zinc and iron, the other heavy metals in the leachate were present in quite low enough concentrations not to affect the performance of the RBC and the UAF. Even the zinc and iron concentrations were not capable of causing any problems since the concentrations are much lower than the toxic limit i.e an average soluble Zn concentration of 2 mg/l and a soluble Fe concentration of 8 mg/l. The results for Zn and Fe removals throughout the entire phase of the studies are seen in Figs. 7.19 and 7.20 for the RBC and the UAF respectively. As indicated, both the RBC and the UAF were found to yield high degree of Zn and Fe removals (90% Zn removal and 80% Fe removal). This observation compared well with the finding of Wu et al (1982) who reported Zn and Fe removal of over 92%.

Table 7.8 gives the performance of the RBC and UAF during step addition of Zn. Figures 7.21 and 7.22 show the performances of the RBC and the UAF during the step addition of Zn. The maximum amount of dosage added (45 mg/l of soluble Zn) showed a lowering in COD removal efficiency, especially in the UAF. The RBC was found to better withstand the influx of Zn in term of COD and Zn removal. The Fe removal in the RBC (Fig 7.21b) was greatly affected by Zn increment compared to that in

a. Zinc



b. Iron

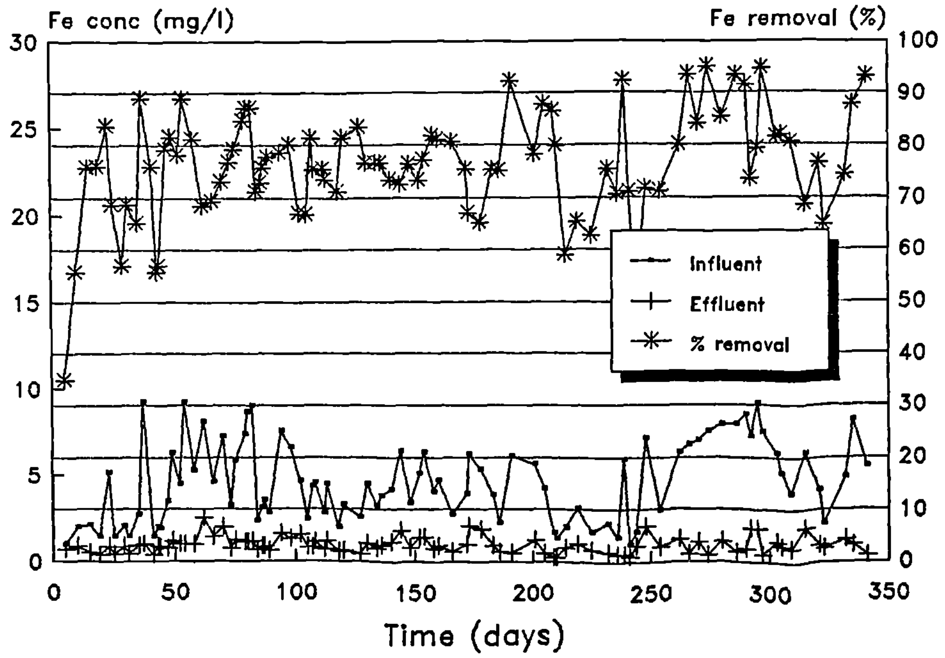
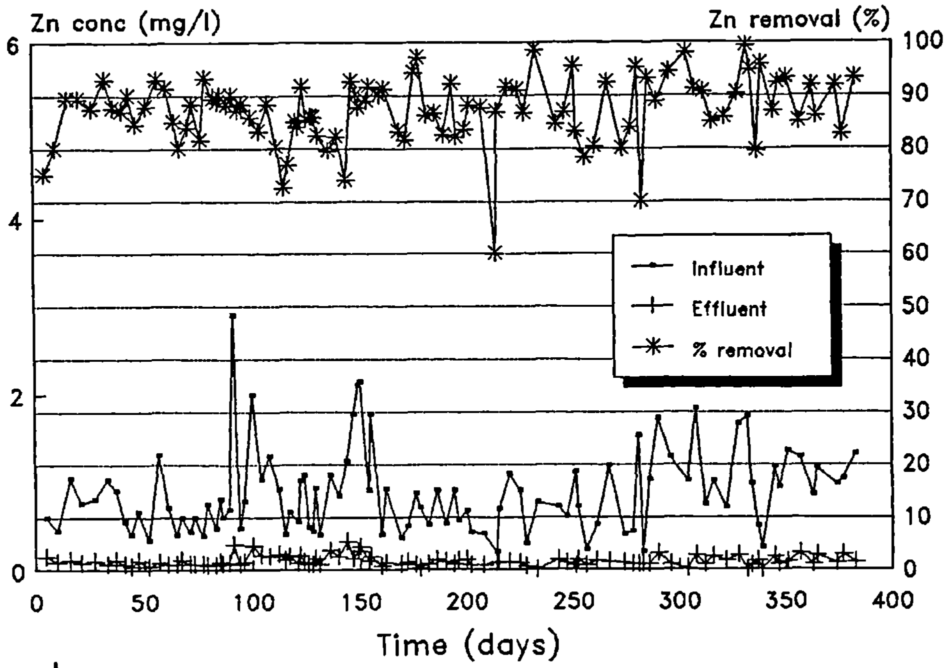


Figure 7.19: Zinc and Iron Removal in RBC Performance (Phase IV)

a. Zinc



b. Iron

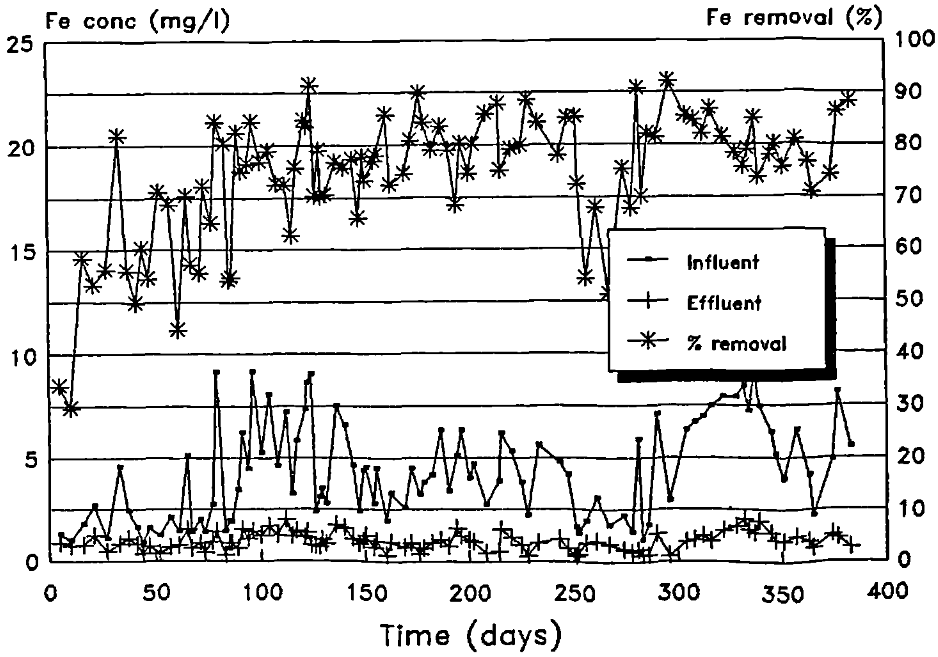


Figure 7.20: Zinc and Iron Removal in UAF Performance (Phase IV)

Table 7.8: Performance of RBC and UAF during Step Addition of Zinc

	Zinc Addition (mg/l)				
	1	10	20	40	25
R B C (VOLR = 1.9 kg COD/m .d)					
% removal					
COD	85	80	72	70	77
Zn	85	91	92	93	87
Fe	79	81	65	72	80
U A F (VOLR = 2.8 kg COD/m .d)					
% removal					
COD	78	71	69	63	71
Zn	84	90	88	94	85
Fe	77	84	36	44	59
Biogas (l/d)	4.3	2.8	2.1	2.2	3.6
Methane Yield (m /kg CODrem)	0.28	0.16	0.12	0.1	0.19

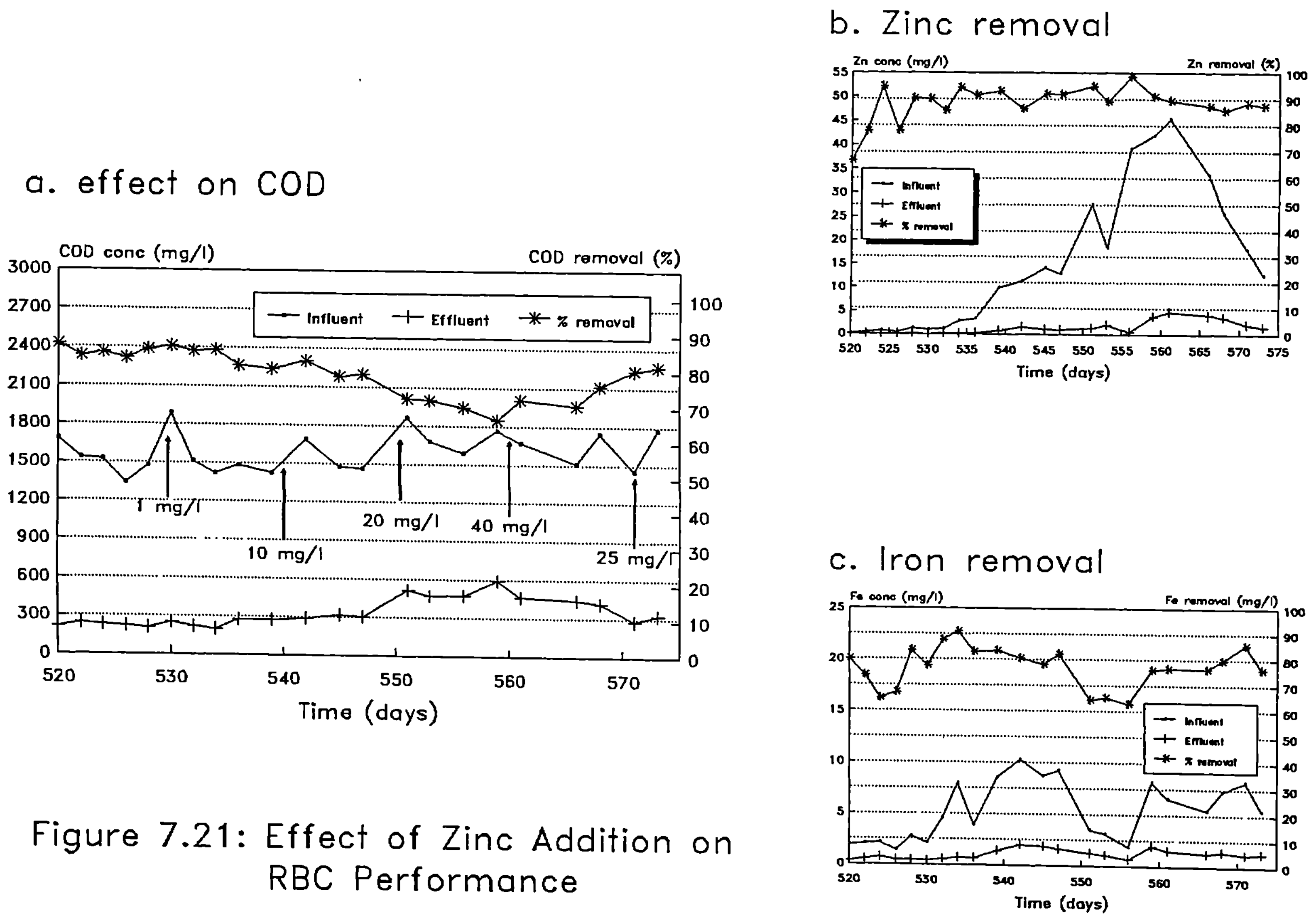


Figure 7.21: Effect of Zinc Addition on RBC Performance

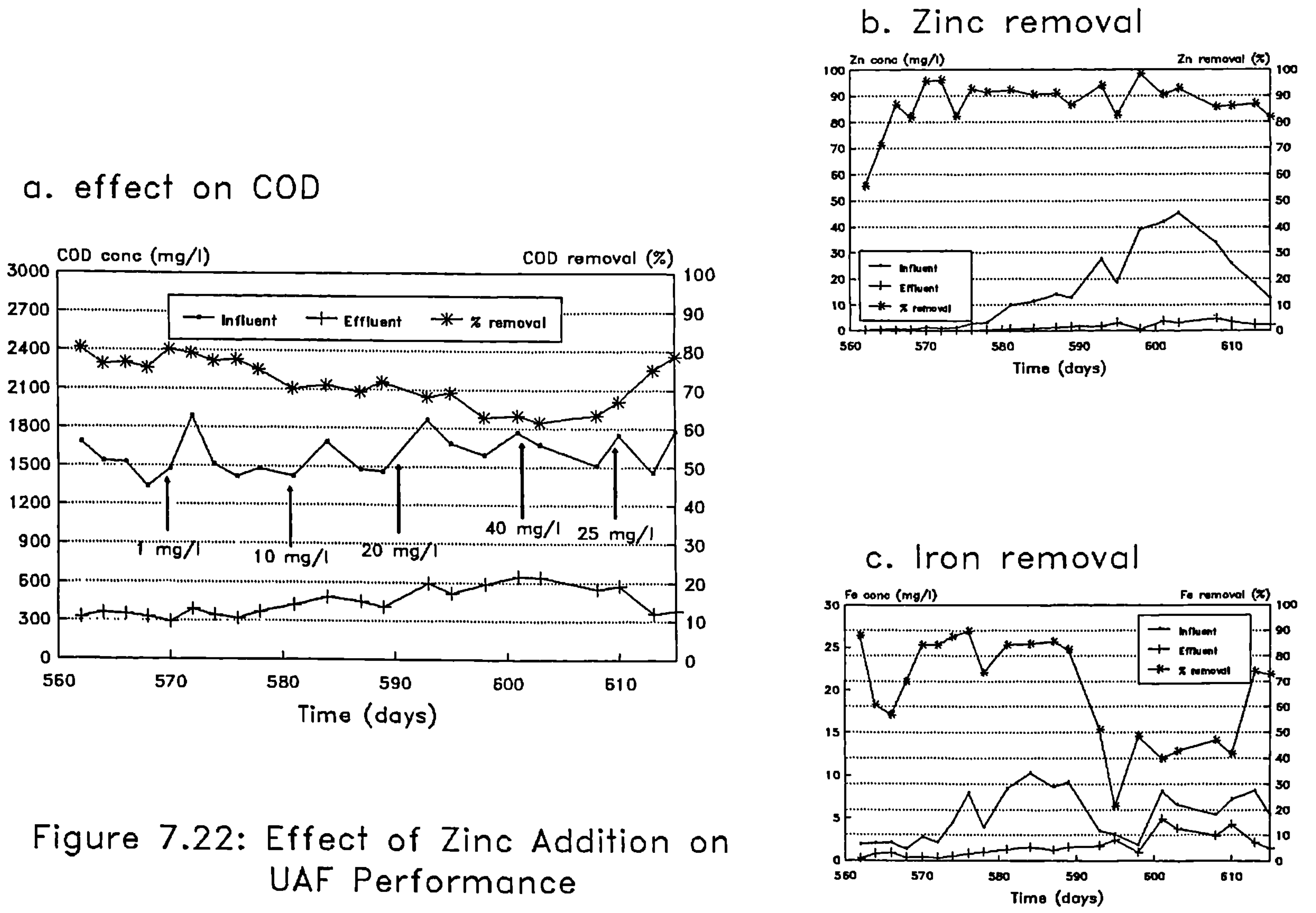


Figure 7.22: Effect of Zinc Addition on UAF Performance

the UAF (Fig 7.22b). The lowest Fe removal being 20% in the RBC. The Zn toxicity was rather low since by gradual addition of Zn, the microorganism in the RBC and the UAF were allowed to adapt to the new environmental condition. Analytically, inhibition in the UAF is shown by a significant decrease in total gas produced, an accumulation of VFAs, a drop in pH and alkalinity, a decrease of the biogas and methane produced, and a decrease in the substrate removal efficiency.

In any case, resistance to a toxic substance often involves an increase in the concentration of the substance which can be tolerated rather than acquisition of total resistance to the substance at any level. When the concentration of the toxic substance is increased slowly, the microbial population can acquire increased resistance through all the mechanisms available to it such as mutation of one or more species in the population; or the alteration of the metabolism of one or more species to overcome the metabolic block produced by the toxic material. However, if a large concentration of toxic material is introduced suddenly, the effects are quite different than when the same concentration is reached after an adequate series of acclimatization because no time is allowed for any of the available mechanisms to operate, and most of the population will be destroyed.

7.4 Conclusion

The RBC and the UAF responded remarkably to the restart-up operation after a period of shutdown. It was seen that the RBC gave a better performance than the UAF. Once the reactors were restarted, the microorganism which had been idle immediately began to activate, shown by the spontaneous reduction in substrate in the wastewater.

From the profile studies and biological solids accumulation, the behaviour of the RBC and the UAF can be identified. The biological solids distribution in the RBC indicated that the RBC was approximately operating as a completely mixed system, whereas the UAF approached a plug flow configuration. These observations were based on the finding of Chavadej (1978) and Norton (1984). The COD removal in the UAF was found to correlate with the biological solids build up (either attached or entrapped within the media's void spaces) in the lower one quarter of the reactor height.

Recovery from volumetric and organic shock loadings was very rapid. The RBC was seen to be affected by both the volumetric and the organic shock loadings, whereas for the UAF, only the volumetric shock loading was found to lower the performance. The overall performance indicated that both the RBC and UAF were able to resist a series of two- to fourfold shock loadings.

Both the RBC and the UAF are capable of removing high percentages of Zn and Fe. The effect of Zn addition resulted in the lowering of the RBC and UAF performances, although the reactors are able to accommodate further Zn addition.

CHAPTER 8

LEACHATE TREATMENT BY ACTIVATED CARBON ADSORPTION

8.1 Introduction

The treatment of wastewaters using activated carbon (AC) adsorption has recently received wide attention, especially as an advanced treatment of effluent from biological treatment plants. Although conventional secondary biological processes are generally employed for municipal and industrial wastewater treatment, the processes are not totally effective in the removal of many organic pollutants, particularly those which are synthetic in nature (Huang and Steffens, 1976). Therefore, more efficient methods, usually physical-chemical treatment processes, especially AC, were used either as a supplement or as a substitute for conventional biological methods. Effluent quality from carbon adsorption process is influenced by the previous treatment processes through which the wastewaters have passed

Adsorption of effluent from biological treatment processes which contain the biologically resistant portions of a wastewater have been investigated and of the adsorbent materials thus far evaluated, AC has shown to be the most promising (Burleson et al, 1968). In this case, the activated carbon adsorption system should be considered as a tertiary treatment process.

However, the addition of any tertiary treatment process incurs significant additional treatment expense, and the effectiveness of the tertiary treatment depends greatly on the consistent and efficient operation of the biological secondary process which may have been subjected to problems from toxic materials as well as the variation in waste composition and hydraulic conditions (Weber et al, 1970).

Peoples et al (1972) reported the applicability of using direct physical chemical treatment namely a filtration system followed by AC adsorption for the removal of oil, SS and dissolved organics from a refinery wastewater as an alternative to conventional biological treatment. The study carried out by Hager and Reilly (1970) indicated that the AC adsorption phase of a clarification-carbon adsorption process was the most expensive unit process, but it still remained the vital phase in the study as substantial removal of soluble organic matters only took place during the adsorption stage. As a result, most research efforts aimed at optimizing physical-chemical treatment processes have generally been concentrated on the AC process.

Srivastava et al (1987) stated that the process of adsorption has an edge over other methods, such as precipitation and coagulation, due to its sludge free, clean operation. Despite the search for other, low cost adsorbents such as coal fly ash (Sen and De, 1987; Letten, 1984), peat moss (Chaney and Hundermann, 1979; Bencheikh-Lehocine, 1989), crushed coconut shells and straw (Larsen and Schierup, 1981), AC is still universally used and is generally acknowledged to be the most feasible process for removing a wide range of trace toxic and carcinogenic pollutants from wastewater.

Evaluating the use of powdered activated carbon (PAC) addition to an activated-sludge aeration basin to enhance COD removal from a pharmaceutical wastewater, Osantowski et al (1985) observed that, although the process could increase the soluble COD removal, it could not be recommended as viable due to an occurrence of viscous floating MLSS resulting in a significant loss of both volatile MLSS (MLVSS) and PAC.

Granular activated carbon (GAC) is preferred over powdered activated carbon (PAC) in a continuous fixed-bed column. Van Gils et al (1984) in their studies used GAC in a fixed-bed column to provide final polishing and removal of soluble organics from industrial laundry wastewaters.

It was noted that from the literature review of the AC process in Chapter 3, the process has limitations and should be carefully investigated prior to making process commitments. The applicability of AC, may it be in powdered or granular form, must first be tested in the laboratory in order to determine the appropriate design methods (Yen and Singer, 1984). The degree of organic removal achievable by adsorption, even in ideal equilibrium batch adsorption tests, varies widely between different wastewaters and can also differ significantly between virgin and regenerated carbons treating the same wastewater (Lawson et al, 1978). Adsorption may be an efficient and appropriate treatment for removing some specific chemical from one particular waste, while totally unacceptable for removing the same compound from another waste.

The studies that are usually conducted in determining the suitability of the activated carbon are batch isotherm and continuous carbon column breakthrough studies. The initial study using batch isotherms is performed to evaluate the adsorbability of the AC while the continuous study is used to evaluate the dynamic adsorption test. Although the data obtained from isotherm study cannot be used to directly predict the performance of a continuous carbon column study, the isotherm technique is useful for qualitatively comparing the amenability of various wastewaters to carbon treatment.

The leachate of concern in this study came from both "ageing" and "maturing" landfills, hence the leachate has basically passed through a certain level of degradation. Also the study involved a comparison of treatment processes, namely the RBC, the UAF and the AC adsorption processes. In order to compare the performances of all three reactors, the leachate was directly treated by the AC adsorption process without prior primary and secondary treatments. The effect of the concentrations of the leachate used on adsorption was also evaluated.

8.2 Experimental Programme

In this particular study, the feasibility of using GAC to treat leachate was carried out. The effectiveness of the GAC as an adsorbent for leachate was first determined through laboratory investigation. Generally, the experimental programme involved two parts. The first were the batch adsorption tests - preliminary isotherm tests, which were conducted to illustrate the feasibility of the GAC treatment. During this stage the contact time appropriate for the carbon used was also determined. The batch adsorption tests consisted of:

- tests for the determination of contact time and
- adsorption isotherm tests

Secondly, dynamic, laboratory-scale carbon column tests were carried out to simulate the actual treatment process used in a treatment plant. Unlike other treatment processes, which are mainly biological, an adsorption column can be investigated in less than a month. Although adsorption is a physical-chemical process, a certain amount of biological activity will usually take place in a carbon bed. The biological activity may increase and become significant with a longer operating time of the column bed. In this case the effective removal capacity of the carbon will be not only due to the physical-chemical process but also because of the biological degradation of organic matter in the wastewater (Lyman, 1978).

It should be noted that adsorption characteristics are actually the net result of a large, unknown number of interacting adsorption, thus for empirical curve-fitting parameters, a well established parameter such as COD or TOC will be used instead (Sweeney et al, 1982), although the COD and TOC will not represent the adsorption capacity of the other parameters.

8.2.1 Determination of Contact Time

Contact time is very critical to the adsorption process (Wagner and Jula, 1981). In order for an adsorption equilibrium to be reached, ample contact time must be allowed between the GAC and wastewater of concern. The experimental test to determine the contact time was carried out by adding 0.5 g of pulverised GAC, into several 100 ml portions of leachate which were placed in 250 ml conical flasks. The flasks were then clamped to a shaker and agitated for various time periods at ambient temperature. At the end of each required time, the flask was unclamped and the carbon removed from the leachate sample by vacuum filtration through a Whatman GC/C filter paper. The filtered solutions were then used in the determination of the required contact time.

The effect of pH on the adsorption rate was also studied. Besides the actual pH of the leachate of 6.6, two other pH readings of 7.8 and 5.4 were investigated.

8.2.2 Adsorption Isotherm Tests

The adsorption isotherm for both the LS and HS leachate were carried out according to the procedure given in Appendix E. Fixed quantities of leachate were tested with a series of increasing measured quantities of pulverised GAC. The carbon-leachate slurry was agitated for a minimum of the contact time determined prior to conducting the test. The carbon was then removed from the leachate by filtering through a Whatman GC/C filter paper and the residual adsorbate in solution was determined. The data obtained were then used for plotting isotherms using Freundlich Isotherm procedures.

8.2.3 Dynamic Carbon Column Tests

Fixed-bed continuous column studies were conducted as described in the Experimental Methods in Chapter 5. Three different, empty bed contact times (EBCT) were used in these studies. The intent of the column tests was to obtain breakthrough curves showing how the concentration of the effluent varies with time or volume of leachate treated. The carbon usage at a specific breakpoint time was determined for the three different EBCTs. Heavy metal removals (zinc and iron) were also investigated in the AC column study.

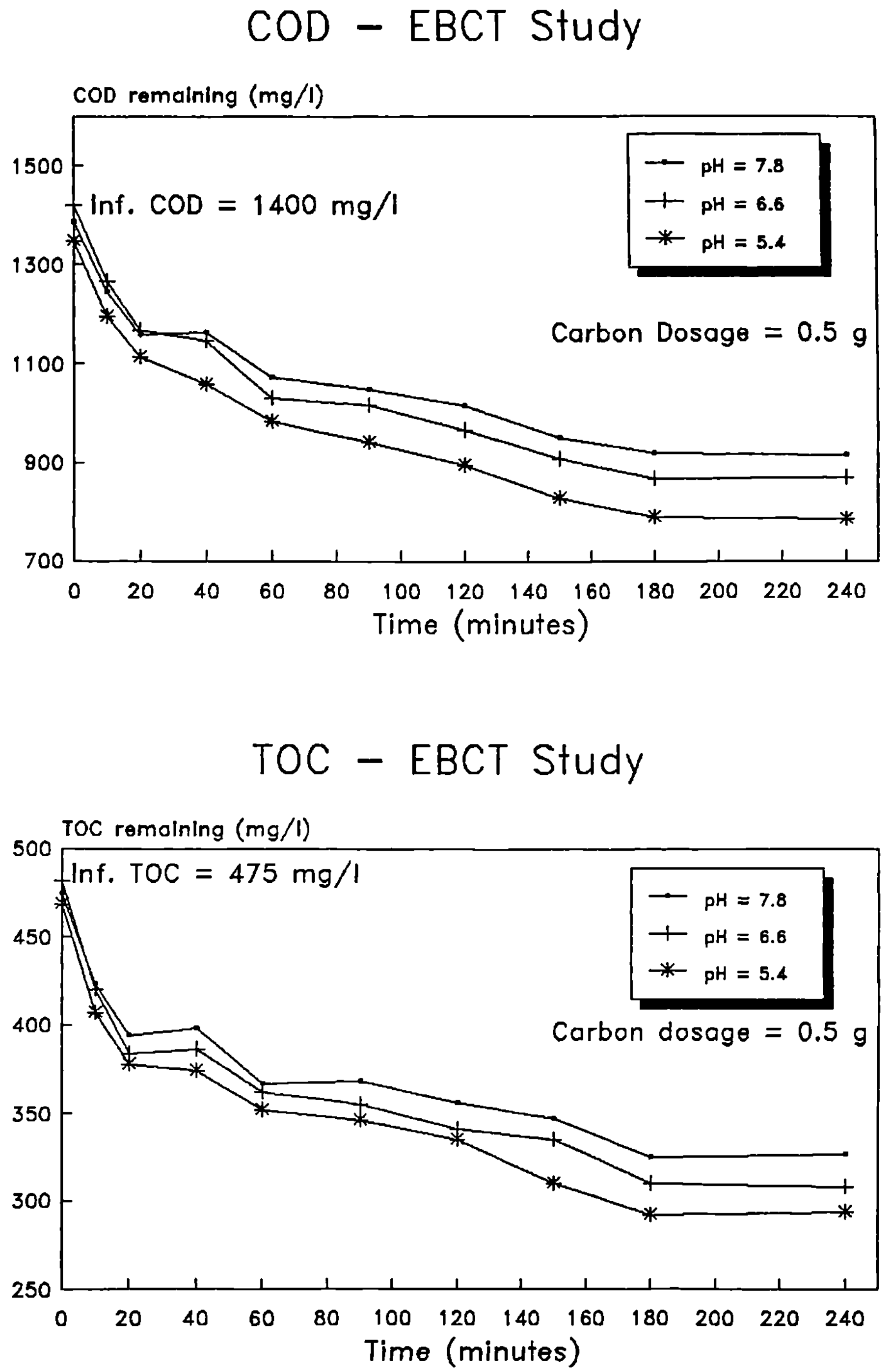
8.3 Results and Discussion

8.3.1 Rate of Adsorption Results

Figure 8.1 illustrates the variations of adsorption rates for the different pH readings as determined by COD and TOC. From both graphs, the adsorptions of COD and TOC were essentially completed within about 180 min. It should be noted that the contact time to achieve equilibrium is only applicable for the specific type of adsorbent and wastewater used, and in this case the leachate. The results for this study is tabulated in Table E.1 in Appendix E.

From the pH evaluation, the lower pH was observed to give a higher adsorption rate than the higher pH, although equilibrium was reached at the same contact time. Wang et al (1972) stated that almost all organic wastes are multi-component, and since the influence of pH on AC adsorption of organic compounds from wastewater depends on the physical-chemical properties of individual organic species, the adjustment of pH may increase the removal of one organic species while at the same time suppressing the removal of another.

Although the results obtained in this study (based on COD and TOC) were consistent with the finding of Weber and Morris (1963), Zuckerman and Molof (1970) and Wang et al (1972), they are not conclusive since other organic species present in a leachate might produce opposite results as observed by Wang et al (1972). Further study is recommended to investigate the effect of pH on the rate of adsorption for different organic species.



EBCT = Empty Bed Contact Time

Figure 8.1: AC Adsorption Rate (Contact Time) Study

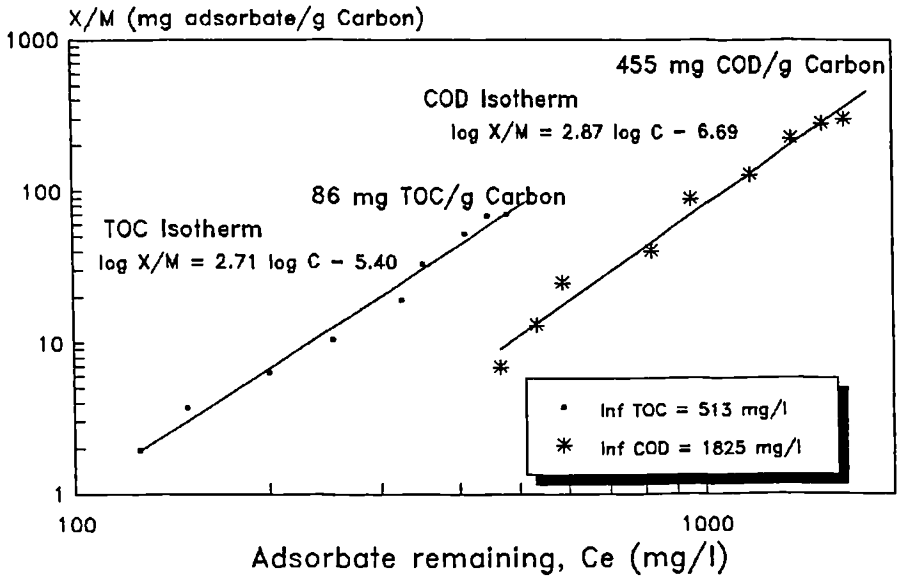
8.3.2 Adsorption Isotherm Results

From the above adsorption rate study, the pH variations were found to affect the degree of adsorption. The influent pH value of the leachate was between the range of 6.5 to 7.2, and at this pH range AC adsorption process was observed to be capable of removing high percentage of the COD and TOC. Therefore, to avoid the effect of pH on the isotherm study, the influent pH value of the leachate was kept within the observed range during the rest of the adsorption isotherm study.

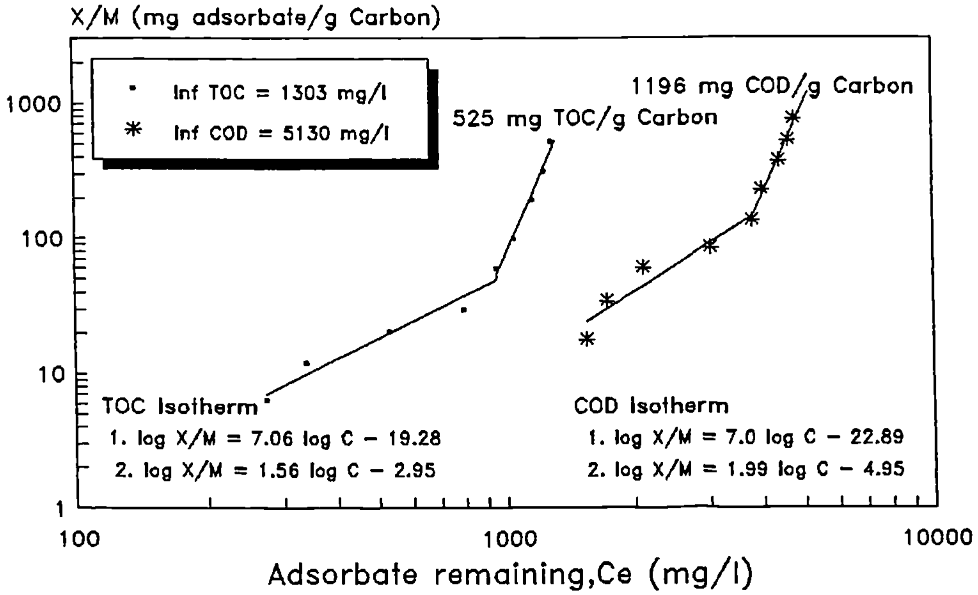
The equilibrium studies carried out on the LS and HS leachate indicated less favourable adsorption of both the leachate concentrations by the carbon as demonstrated by the linear plot (Fig. E.1 in Appendix E) of the adsorption parameters. The isotherms indicated a decreasing adsorption capacity with increasing carbon dosages. Thus in order to reduce high quantities of removable adsorbable organic compounds, massive carbon dosages would be necessary.

The results of the adsorption tests on the logarithmic plots are shown in Fig. 8.2. From Fig. 8.2 a fraction of organic matters which cannot be removed by carbon adsorption was observed. These residual organic concentrations were determined by subjecting the leachate to a maximum carbon dosage of 30 g per 100 ml leachate sample. The values obtained, together with the influent leachate values are summarized in Table 8.1.

The results indicated that a high fraction of the organic species cannot be removed by activated carbon, especially in terms of COD removal.



Isotherm for LS Leachate



Isotherm for HS Leachate

Figure 8.2: Freundlich Adsorption Isotherm

Table 8.1: COD and TOC values before and after Carbon Adsorption

Types of Leachate	Organic species before carbon adsorption (mg/l)		Residual organic species after carbon adsorption (mg/l)		% non-adsorbable organics (mg/l)	
	COD	TOC	COD	TOC	COD	TOC
LS Leachate	1825	513	440	120	24	23
HS leachate	5130	1303	1400	250	27	19

These non-adsorbable fractions could be due to the presence of adsorption resistant organic compounds with low molecular weight and which may be biodegradable such as dissolved carbohydrates and organic acids and hence they are much more amenable to biological treatment processes (Westermark, 1975; Ford and Manning, 1978).

From the HS isotherm study, two distinct phases were observed (Fig. 8.2). The first phase was dominated by the less or weakly adsorbable solute. As a general rule, the more adsorbable solute should always be adsorbed before the less adsorbable solute (Petura, 1981). The probable reason is due to the presence of a higher concentration of the less adsorbable solute when compared to the more adsorbable solute. Adsorption is a function of both adsorbability and concentration, since the less adsorbable solute has much greater concentration it dominates the first phase of the isotherm.

Another reason is that the weight of the less adsorbable solute may be much greater than that of the more adsorbable solute. Therefore, although fewer moles are adsorbed, they account for the bulk of the measurement and the adsorption characteristic of the less adsorbable solute dominates the first phase. When most of the less adsorbable component has been adsorbed, the isotherm changes slope to show the higher adsorbability of the remaining solute.

The major factors determining the shape of an isotherm are the number of compounds in solution and their relative adsorbabilities, the initial concentrations in solution, the relative contributions to the total COD and TOC, the degree of competition among solutes for adsorption sites, and the characteristics of the specific carbon. Therefore the number of different isotherms is therefore clearly immense.

In this study, the Freundlich isotherm could only be applied to the adsorbable fractions due to the limitation resulting from the complex nature of leachate with a high concentration of the weakly adsorbable solute. An understanding of the complexity of leachate components is recommended for further study. The constants K and n along with the corresponding correlation coefficients are listed in Table 8.2.

Table 8.2: Freundlich Parameters for Adsorption of Leachate

	K (mg/g)	n	R^2
LS leachate			
COD	1.2×10^{-3}	0.348	0.98
TOC	4.5×10^{-3}	0.369	0.99
HS leachate			
COD (phase 1)	1.1×10^{-10}	0.143	0.98
TOC (phase 1)	4.2×10^{-9}	0.142	0.98
COD (phase 2)	7.1×10^{-3}	0.502	0.92
TOC (phase 2)	0.052	0.641	0.95

The low K and n values from the Freundlich isotherm data indicated the increase in leachate complexity (Reimers et al, 1976). The LS leachate exhibited an overall higher adsorption capacity than the HS leachate, although the phase 1 adsorption in HS leachate indicated a higher adsorption capacity but once the less adsorbable component had been completely adsorbed, the phase 2 adsorption took place at a lower adsorption capacity.

The adsorption capacity and carbon usage rate based on adsorbable fractions of COD and TOC for the LS and HS leachate are listed in Table 8.3.

Table 8.3: Adsorption Capacity and Carbon Usage Rate for LS and HS Leachate

	Ultimate Capacity (mg COD/g Carbon)	Carbon Usage Rate (g Carbon/l leachate)
LS leachate		
COD	455	3.0
TOC	86	4.6
HS leachate		
COD (phase 1)	1196	1.1
TOC (phase 1)	525	0.7
COD (phase 2)	142	16.8
TOC (phase 2)	52	13.4

The existence of the less adsorbable solute in phase 1 of the HS leachate even though showing a higher adsorption capacity of 1196 mg COD adsorbed/g Carbon and 525 mg TOC adsorbed/g Carbon only resulted in a maximum of 26% COD and 27% TOC removal. The removal of another 47% COD and 54% TOC in phase 2 of the HS leachate required carbon usage rates of

16.8 g/l leachate and 13.4 g/l leachate respectively which were rather high. Thus HS leachate was not used in the dynamic studies, since greater quantity of carbon would be required which is not cost-effective.

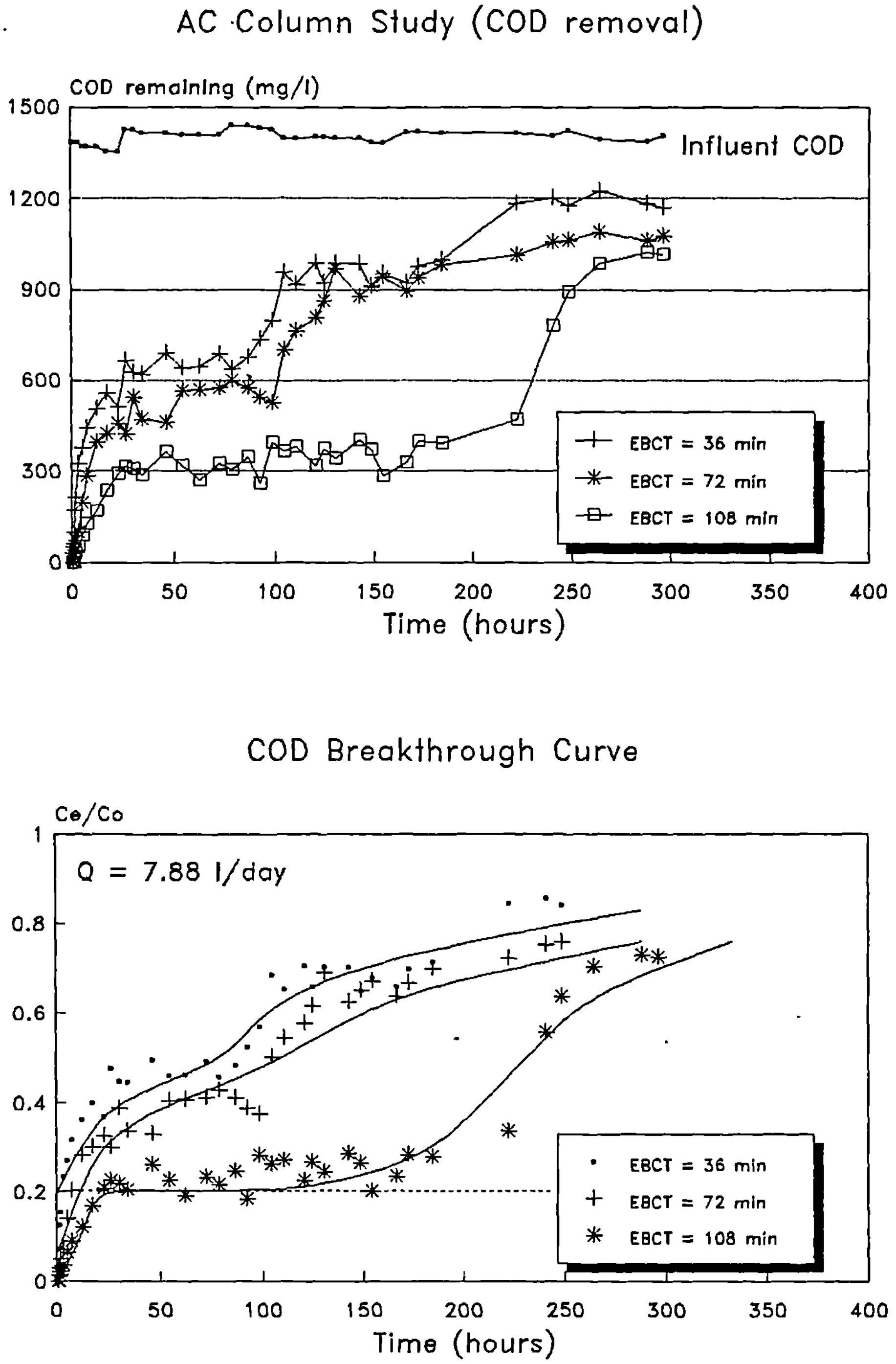
8.3.3 Dynamic Column Results

Operating conditions for the AC adsorption column study are described in Table 8.4. The carbon column effluent breakthrough curves are plotted in Figs. 8.3 and 8.4 for a flowrate of 7.9 l/day and in Figs. E.2 and E.3 (Appendix E) for a flowrate of 17.9 l/day. From the graphs, the data showed some scattering which varies from one set to another, but in all cases smoothed breakthrough curves could be drawn.

Table 8.4: Operating Conditions and Description of AC Adsorption Column (LS Leachate)

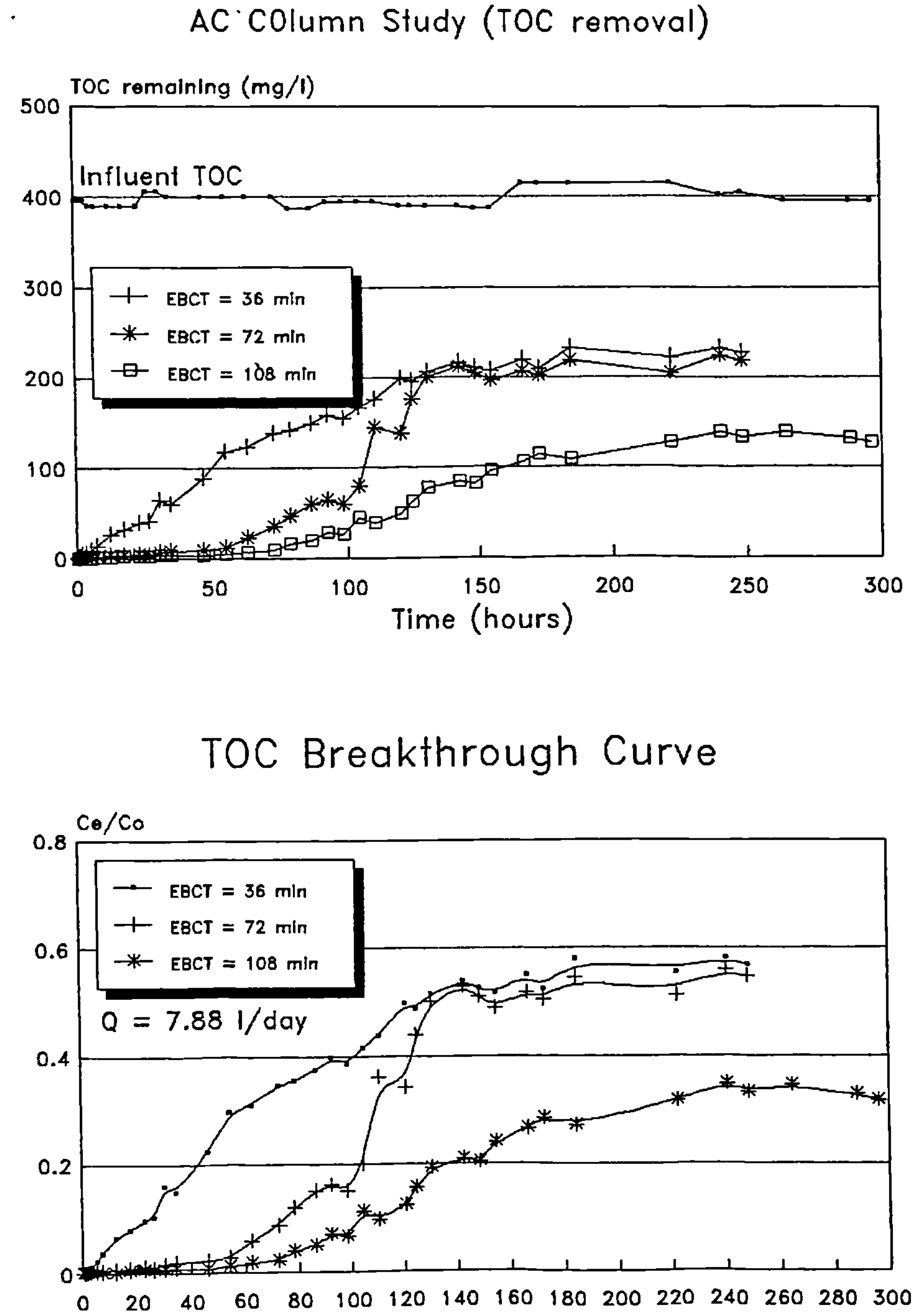
Test Column Sampling Points	1	2	3	3
Inf. COD concentration (mg/l)	1400	1400	1400	1600
Inf. TOC Concentration (mg/l)	400	400	400	454
Flowrate (l/day)	7.9	7.9	7.9	17.9
Dry Weight (g)	98	196	294	294
Bed depth (m)	0.4	0.8	1.2	1.2
EBCT (min)	36	72	108	45
Test Duration (hours)	250	250	300	150

In order to evaluate the adsorption capacity of AC in a continuous column study, the breakthrough level $C_e/C_o = 0.2$ representing 80% removal efficiency was selected based on the percentage non-adsorbable organic fractions remaining in the leachate during adsorption isotherm study. This breakthrough level passes through the breakpoint (that is the point when the effluent level began to increase consistently and at a significant rate)



EBCT = Empty Bed Contact Time

Figure 8.3: AC Breakthrough Curve Study for COD Removal (LS Leachate)



EBCT = Empty Bed Contact Time

Figure 8.4: AC Breakthrough Curve Study for TOC Removal (LS Leachate)

for a COD removal at EBCT of 108 min, as shown in Fig. 8.3. In pilot-scale studies of municipal wastewater treatment by Joyce et al (1966), the effluent quality was considered satisfactory with C_e/C_0 ranging from 0.25 up to 0.5. The results together with the maximum concentration reached at the "exhaustion point" for each carbon test sampling point are tabulated in Table 8.5.

Table 8.5: Dynamic Column Performance Test Results at C_e/C_0 = 0.2 and Maximum Concentrations at Exhaustion Point (LS Leachate)

Test Column Sampling Points	1	2	3	3
COD Removal				
Breakpoint Time (hours)	3.0	12.0	34.0	4.0
Volume Treated at Breakpoint (l)	0.9	3.95	11.2	3.05
Carbon Usage Prior to Breakpoint (g Carbon/l leachate)	117	50	26	96
Carbon Loading Prior to Breakpoint (mg COD adsorbed/g Carbon)	9.6	22.6	42.7	13.3
Max. Concentration at exhaustion (mg/l)	1200	1065	1025	1200
TOC Removal				
Breakpoint Time (hours)	44.0	95.0	140.0	3.0
Volume Treated at Breakpoint (l)	12.7	31.2	46.4	2.3
Carbon Usage Prior to Breakpoint (g Carbon/l leachate)	7.7	6.3	6.3	127.8
Carbon Loading Prior to Breakpoint (mg COD adsorbed/g Carbon)	41.1	52.7	50.1	2.8
Max. Concentration at exhaustion (mg/l)	230*	225*	140*	410

* Exhaustion points had still not been reached when the column operation was stopped.

Carbon usage rates against EBCTs for COD and TOC are illustrated in Fig. 8.5. The curves are usually used to evaluate the economic balance between EBCT for a single fixed bed, which translates into capital cost, and carbon exhaustion rate, which translates into direct operating expense (Petura, 1981). The carbon usage rate decreased rapidly with the increase

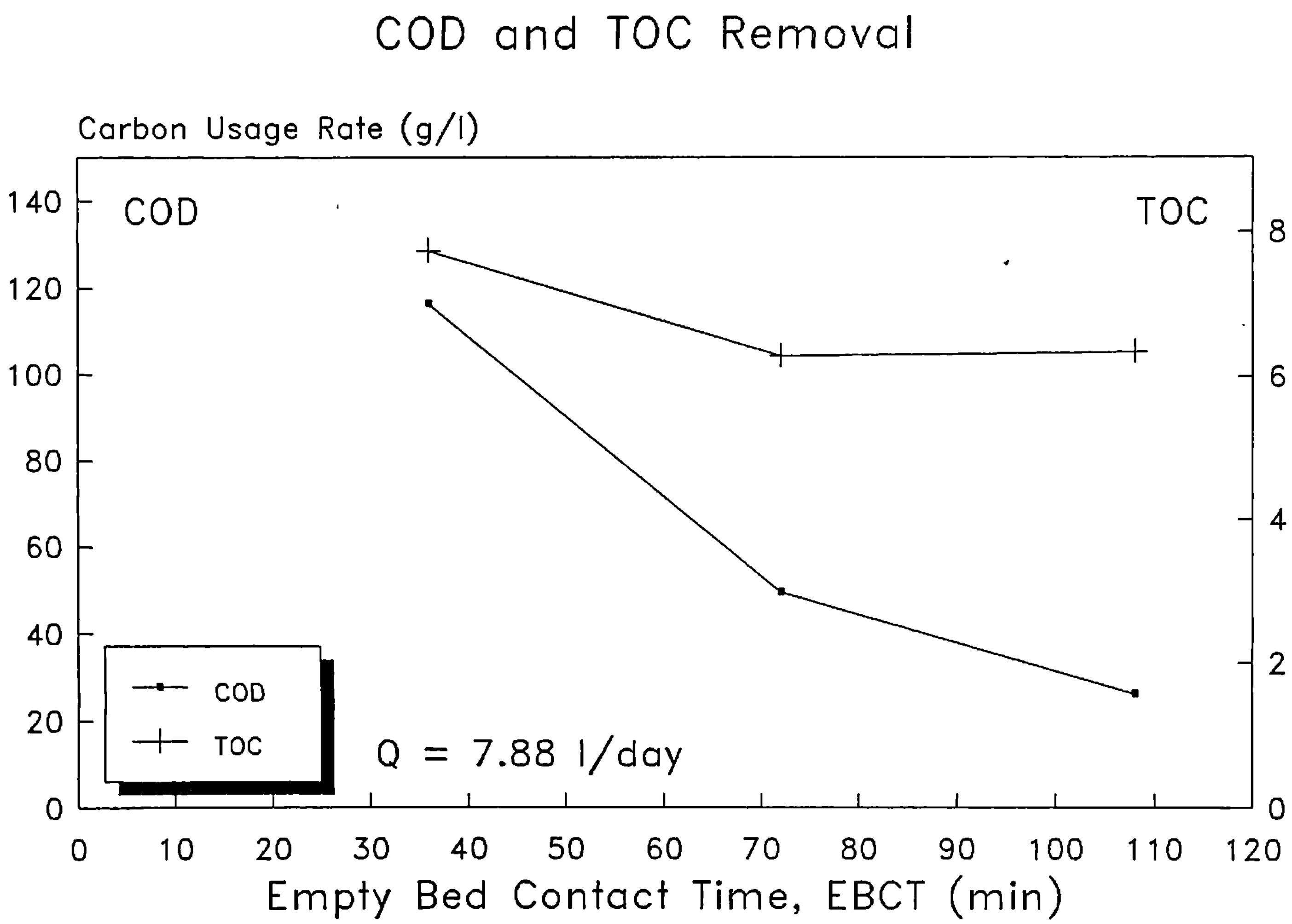


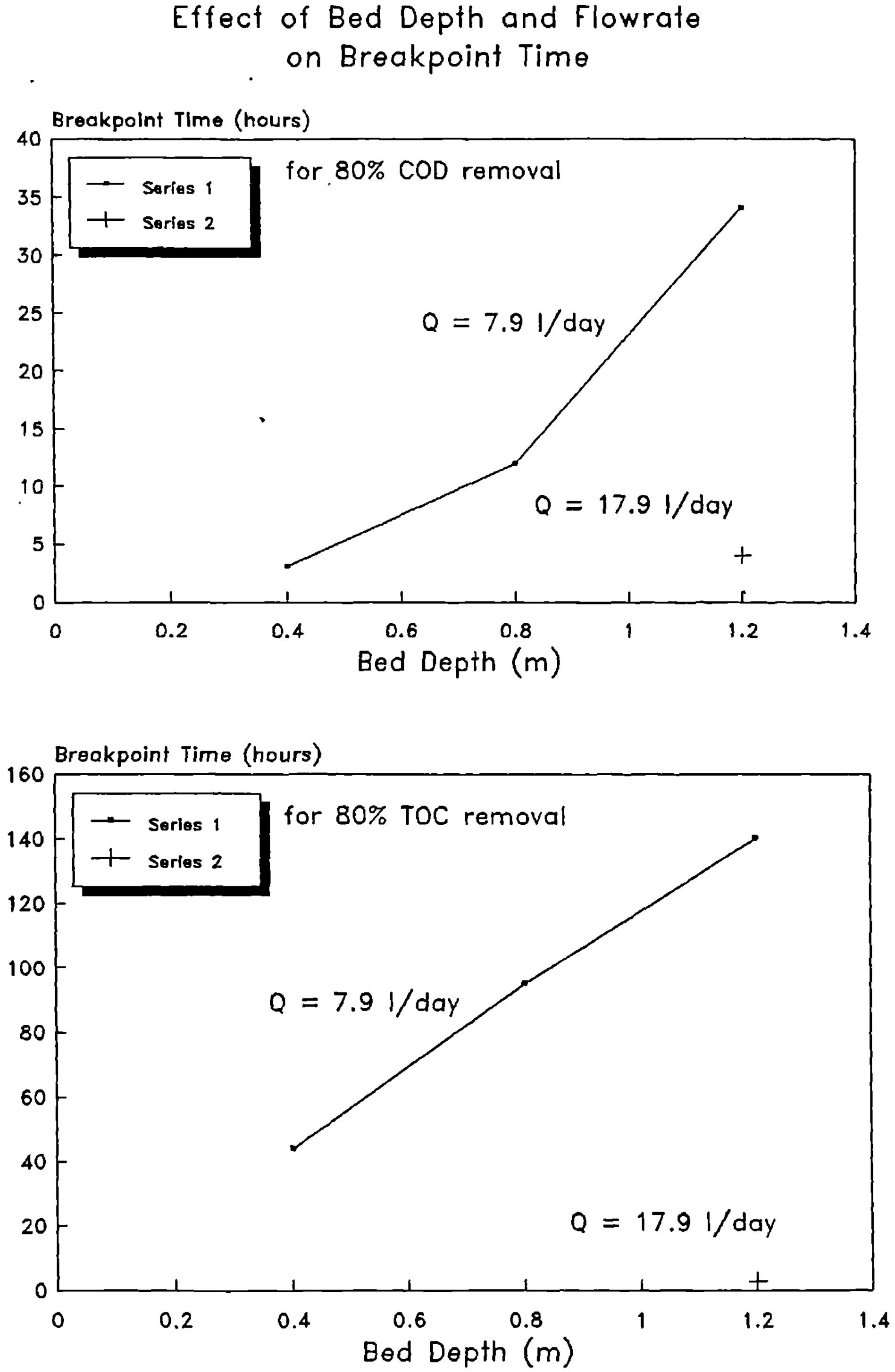
Figure 8.5: Carbon Usage Rate versus EBCT for LS Leachate

in EBCT. The low carbon usage rate is obtained at about 90 min EBCT, where increasing the EBCT beyond this value results in a minor reduction in usage rate.

The effect of flowrates and column bed depths on the breakpoint time in the experimental runs are illustrated in Fig 8.6. The graph represent a relationship between bed depth and breakpoint time or bed depth-service time (BDST) in actual treatment plant as proposed by Bohart and Adams (1920). In this case the adsorption rate was assumed to be proportional to both the remaining adsorbate concentration and the residual carbon capacity. As expected, regardless of the breakthrough levels chosen, the higher flowrate or the lower bed depth were responsible for the earlier breakpoints. These results confirmed the observation made by Faust and Aly (1987) who stated that the immediate breakthrough occurring at C_e/C_o range of 0.1 to 0.5 after start-up was contributed by EBCT or bed depth which presents as limits to the operation of an AC column. High flowrate would exhaust the bed more rapidly as well as reducing the contact time of the adsorbate with the adsorbent, consequently the amount of adsorbates being adsorbed became smaller. A shorter contact time tends to give steeper breakthrough curves, therefore with respect to the breakpoint time, steepness increases with decreasing breakpoint time.

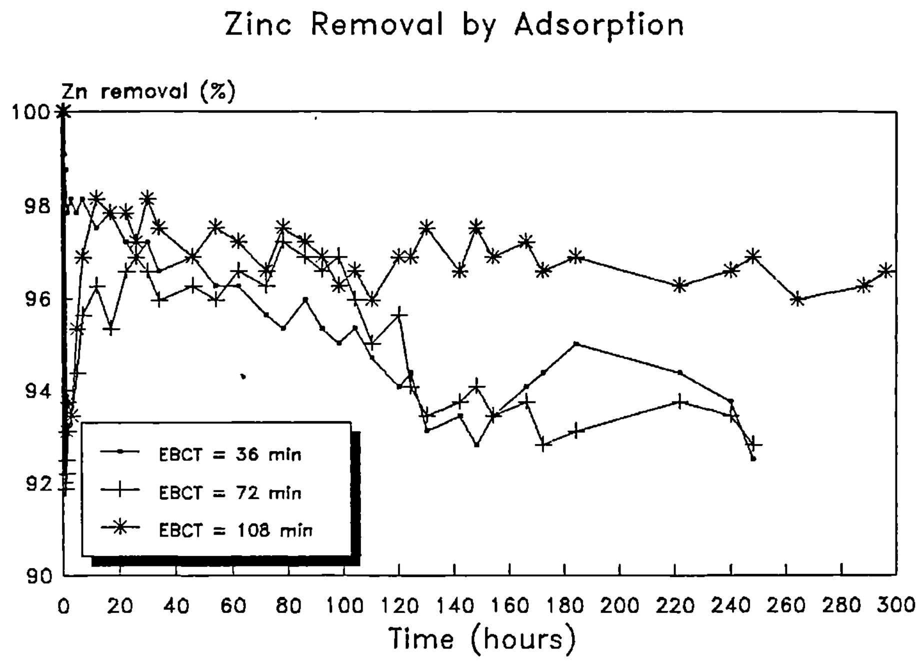
8.3.4 Zinc and Iron Removals by AC Adsorption Column

The results for zinc and iron removal by adsorption are shown in Fig. 8.7. Even though the exhaustion points for COD and TOC removals were reached, a very high removal of zinc (93%) and iron (96%) could still be achieved. This indicated that AC adsorption could be used as a direct

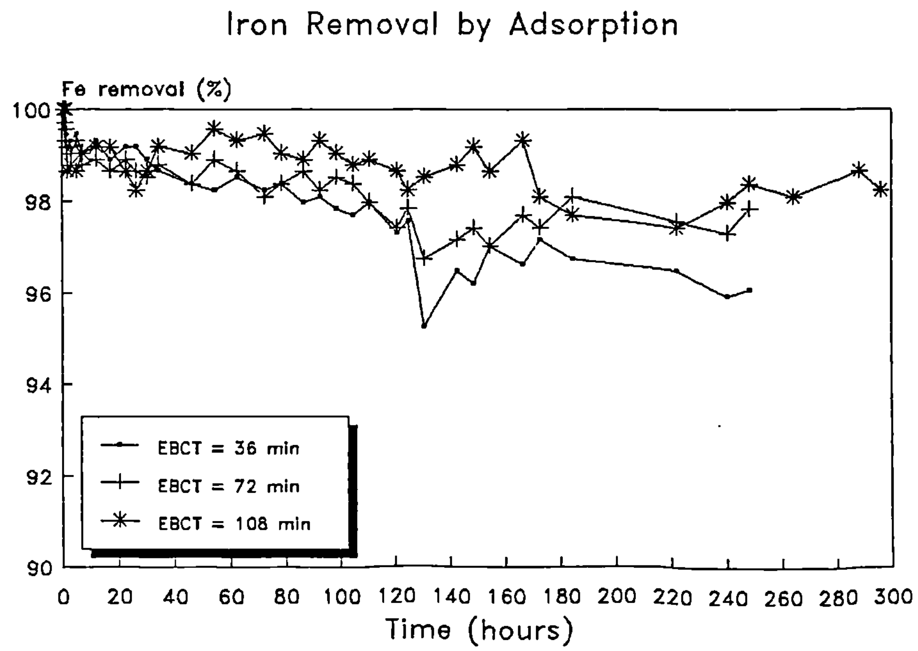


Effect of Bed Depth and Flowrate on Breakpoint Time

Figure 8.6: Bed Depth–Service Time (BDST) Curves at Ce/Co = 0.2 for LS Leachate



Zinc Removal by Adsorption



Iron Removal by Adsorption

EBCT = Empty Bed Contact Time

Figure 8.7: Zinc and Iron Removal by AC Adsorption

treatment process for heavy metal removal.

8.4 Batch Isotherm versus Continuous Column Results

The very large percentage COD (TOC) removals seen in the batch (powdered carbon) isotherm tests at very high carbon dosages cannot be related to removals achievable in GAC beds at economically reasonable contact time and service life. From the column studies, the low contact time resulted in rapid breakthrough. This is an indication that more than one solute was present resulting in the adsorbates competing for sites of adsorption on the carbon surface. Although from the column studies an optimum EBCT of 90 minutes could minimize the carbon usage rate resulting in a COD removal efficiency of 80% and TOC removal of 90%, the non-adsorbable resistant organic fractions remaining were still high. Therefore, direct treatment with AC without any primary or secondary treatment is not sufficient to remove organic matter from the leachate.

Westermark (1975) and Rebhun and Streit (1974) have shown that a high fraction of the organics not adsorbed by activated carbon treatment consist of dissolved carbohydrates and organic acids. Therefore for leachate with high concentration of organic acids (as observed in the HS leachate used) low organic removal efficiencies may be expected in a direct physical-chemical treatment process.

The difference in nature between a batch and a continuous system could be the reason for the capacity differences. Because leachate contains a mixture of compounds of different adsorbabilities, their individual compositions in equilibrium with the carbon are different in batch and continuous systems. In the batch system, a larger number of weakly adsorbable compounds is present in proportion to the equilibrium concentration than in the continuous system. Therefore, the batch system may be expected to provide a somewhat lower capacity than the continuous system. In addition, a higher capacity in a continuous system might be expected because the adsorption process could be enhanced by the occurrence of biological activities (Lyman, 1978; Ford and Manning, 1978).

8.5 Conclusion

The removal by adsorption in a GAC column of specific organics from wastewater are not necessarily parallel to the removal profile of COD and cannot be predicted from batch equilibrium adsorption tests. Furthermore the monitoring for a generalized pollutant parameter, such as COD or TOC, will not provide sufficient information regarding breakthrough of specific solutes. Competitive adsorption by components in the background matrix dramatically affects the carbon removal performance for individual solutes, in this case decreasing individual solute removal capacities by as much as 80 to 97%. Therefore extensive pilot testing is required to develop design information for a specific wastewater matrix (McManus et al,1984).

Multiple column tests with regenerated carbon, operated through several staged countercurrent cycles, are required to provide data for a complete cost optimization. The undefined nature of leachate used has made the interpretation or generalization of results difficult. In most studies, the choice of adsorbates has been limited to the commonly known organics, which are measured in terms of TOC and COD, although treatment usually involves a multitude of adsorbates competing (Jain and Snoeyink,

1973) for adsorption sites on the carbon surface. The removal of organic contaminants in term of COD or TOC was not necessarily found to be a good prediction of specific organic removal. Monitoring for a generalized pollutant parameter, such as COD or TOC, will not provide sufficient information regarding breakthrough of specific solutes. Thus, a thorough understanding of the competitive effect of various organic compounds would appear to be important in any future study.

The removal by direct carbon adsorption, resulted in a prohibitively high dosage of carbon. Hence a GAC column is not effective for treating leachate without prior secondary treatment. Although, the study does not include the treatment of effluent from secondary biological treatment processes, it must not be precluded since previous investigations by several researchers have shown that GAC may effectively be used as a tertiary treatment.

It can be seen that comparing maximum removals from isotherms with typical removals in a continuous column is not valid since the very high batch dosage correspond to the period of low cumulative throughput in continuous adsorbers. A column-type operation would appear to have a distinct advantage over batch treatment because rates of uptake depend on the concentration of solute in the solution that is in contact with carbon. For column operation the carbon is continuously in contact with a fresh solution, consequently the concentration in the solution in contact with a given layer of carbon in a column is relatively constant, For batch treatment, the concentration of solute in contact with a specific quantity of carbon steadily decreases as adsorption proceeds, thereby decreasing the effectiveness of the adsorbent for removing the solute from solution.

Although the primary function of the activated carbon adsorption process is to remove soluble organics, not all soluble organic substances are removed by activated carbon which confirms the report by Bishop et al (1972). The adsorption isotherm study on the LS leachate exhibited a linear trend as seen in Fig. 8.3, but the percentage of non-adsorbable fractions was quite high. Thus, it may be seen that pretreatment by biological processes is essential for reducing the biodegradable organics and allowing for better usage of carbon adsorption. Many classes of organic compounds are not amenable to carbon adsorption - particularly oxygenated organics (Ford and Manning, 1978) - and show up as residual COD and TOC in carbon column effluent. This limits the overall process efficiency of pure physical-chemical treatment systems.

As many of these residual compounds are biodegradable, activated carbon as a polishing process is generally capable of producing a better quality of effluent than is a strictly physical-chemical application. Thus, provision of a secondary biological treatment as well as other physical-chemical treatment processes is required such as filtration and lime or alum precipitation. Becker and Wilson (1978) stated, in their review of pesticide waste treatment experiences, that "the best technology available appears to be a process including pretreatment, filtration and adsorption on AC and/or resin".

In this study only one type of carbon was used and thus it will not give a true picture of the adsorptive capability of activated carbon. Different types of carbon would produce different results, therefore the effect of various types and particle sizes of carbon should also be investigated to find the best carbon type for the specific wastewater.

CHAPTER NINE

CONCLUSIONS AND RECOMMENDATIONS

A broad overview of those criteria directly applicable to wastewater treatment is given in Table 9.1. Obviously, such a listing is only qualitative and the choice of parameters listed is subjective, but if emphasis is placed upon the quality of the end product it is clear that aerobic treatment has nothing to fear as yet from its anaerobic counterpart. High strength (HS) and low strength (LS) leachate were used in this study to investigate the biological treatabilities as well as physical-chemical treatment using activated carbon (AC) adsorption. Both aerobic and anaerobic treatment systems i.e aerobic RBC and anaerobic UAF were used to study the treatment, start-up, operation and performance under various loading rates and pH as well as to evaluate the effect of zinc. For comparative purposes, the experimental works were carried out by using all the reactors individually to assess their capability in treating leachate.

The biological treatment processes exhibited variable performance for the different landfill leachates. The evaluation of criteria for comparison is therefore difficult since leachate compositions from one landfill to another vary widely in concentration depending on landfill age. Hence, for the basis of comparison, removal efficiency and process stability were taken into consideration using the gross all embracing parameters such as BOD_5, COD and TOC. The difficulty is also exemplified when the biological treatment processes are to be compared with physical-chemical treatment process. Once a comparative study has been made, recommendation for future study may be developed. This recommendation should take into account all the contributing factors which could assist in producing an effective leachate treatment.

Table 9.1: Comparison of Direct Treatment of Leachate by the RBC, the UAF and the AC Column

Criterium	RBC	UAF	AC Column
Range of concentration	Good for LS	Good for both LS and HS	Poor
Degree of Treatment	High for LS Moderate for HS	High for HS Good for LS	Low for both LS and HS
Sludge Production	High	Low	Low
Process Stability:			
Volumetric Shock loading	Good	Moderate	-
Organic Shock Load	Moderate	Good	-
Intermittent operation	Good	Good	-
Start-Up Time	14 to 20 days	60 - 70 days	-
Zinc and Iron Removals	Good	Good	High
Biogas Production	None	Methane Production	-
pH	Good at 5.5 to 8.0	Good at 6.0 to 7.5	Good at low pH
Alkalinity Requirement	Low	Buffer needed to increase the pH	-
NH_3-N removal	Good	Poor	-

9.1 Conclusions

Based on the experimental results, the following conclusions can be made:

1. The start-up of the RBC was more rapid i.e within 14 days when compared to that of the UAF (60 days). Acclimatization of the methanogenic bacteria in the UAF took a long time before stable conditions were achieved. The methanogenic bacteria activities were also affected by the fluctuation in pH, hence the optimum pH range of 6.5 - 7.5 should be controlled.

2. During steady state conditions, the RBC operated most effectively over a range of pH 5.5 to 8.0 whereas the UAF performed well only within the pH range of 6.0 to 7.5, beyond which the substrate removal efficiencies decreased. In order to maintain pH near neutrality, buffer is needed for the UAF. This is more crucial with the HS leachate were the bicarbonate alkalinity should be kept above 2500 mg/l in order to provide sufficient buffer capacity to handle increases in volatile fatty acid.

3. The performance of an AC column is generally based on a parameter such as TOC. In order to compare the performance of the AC column with the RBC and the UAF, the validity of using TOC when determining the organic substrate changes in the RBC and the UAF were investigated. The investigation revealed that TOC could be used in lieu of the other common parameters (BOD_5 and COD) where high correlations were found between the TOC and BOD_5 as well as between TOC and COD.

4. The performances of both the RBC and the UAF were found to decrease with an increase in OLR (or a decrease in HRT) regardless of the strength of the leachate. Both the RBC and the UAF performed well for LS leachate, although the UAF could only be subjected to a maximum loading rate of 6 kg COD/m^3.day for a removal efficiency above 65% as compared to the RBC which could go beyond 12 kg COD/m^3.day when the experimental studies were discontinued.

5. With respect to the HS leachate, the UAF produced a much better performance than the RBC. Although at a low loading rate of up to 5 kg COD/m^3.day the RBC achieved higher removal efficiency of 90%, the COD removal decreased more rapidly when compared to that of the UAF as the

OLR increased. The maximum OLR for the RBC to give a COD removal of 60% was 9 kg COD/m^3.day whereas the UAF can go up to 13 kg COD/m^3.day to achieve the same removal efficiency.

6. The biological solids wash-out increased in both the RBC and the UAF with the increase in OLR, and more dramatically when HS leachate was used. The rapid biomass sloughing from the RBC during the treatment of HS leachate was one of the contributing factors for the deteriorating performance of the RBC as the OLR increased.

7. The removal of nitrogen through nitrification and denitrification has been recognized as a process step of major importance (Vochten et al, 1988). The RBC was found capable of nitrification once a steady-state condition had been achieved where up to 55% NH_3-N removal was observed, whereas generally the ammonia concentration in the UAF was found to increase.

8. Although both the RBC and the UAF could withstand intermittent operation such as during restart operations, the RBC responded better when compared to the UAF. This was attributed to the removal of sludge from the base of the RBC which assisted in a better mixing of influent leachate with the microorganisms attached to the drum.

9. The RBC performed better than the UAF during volumetric shock loading. Both the reactors recovered rapidly within 24 hours after the shock loading had stopped. In the case of organic shock loading the performance of the RBC decreased due to the sudden high influx of leachate concentration which caused a stress, as seen from the build up

and sloughing of biological solids (expressed as VSS). As for the case of the UAF, it could cope quite well during organic shock loading although signs of stress could be seen from the increase in VFAs in the reactor.

10. Biological treatment processes removed organic matter more efficiently although the effluent still contained a high concentration of refractory organics, whereas refractory organics are readily adsorbed by activated carbon. Therefore, activated carbon adsorption could provide a major complimentary role to secondary biological treatment methods.

11. The presence of considerable quantities of non-adsorbable material both in leachate would indicate that direct physical-chemical treatment such as AC adsorption is probably not a particularly cost-effective treatment method for removal of organic although a very high removal of zinc and iron could be achieved. The AC adsorption in this study indicated an adsorption capacity of 455 mg COD adsorbed/g carbon for LS leachate, but the highly adsorbable fractions in the HS leachate could only reach an adsorption capacity of 142 mg COD adsorbed /g carbon.

12. The results from the AC adsorption process showed that, for an effective AC adsorption, the AC column should be incorporated as a complimentary treatment process together with secondary biological treatments. The high level of non-adsorbable organic matter of both the LS and HS leachate, but which is probably biodegradable, is better removed first by aerobic or anaerobic treatment followed by AC adsorption which could act as a final polishing process.

13. Based on the above conclusion, the best possible option for the

treatment of leachate could be through the combination of all three treatment processes. The first phase would be by an anaerobic UAF which could remove most of the organic substrates followed by an aerobic RBC which would further removed some organic fractions as well as achieving nitrification. For final polishing, an AC column could be incorporated for removing the refractory organics not removed by the biological processes and also for heavy metal removals.

9.2 Recommendations for Future Study

1. In order to better understand the treatment performance of all the reactors, recommendations for future study should include the study of the effect of temperature on the performance of all the reactors. Bacterial activities are affected by temperature as well as pH and loading rate variations. Microscopic examinations to monitor the bacterial activities could also be carried out.

2. Optimization of the aerobic RBC process through the study of the effect of rotational speeds and percentage media submergence on leachate treatment performance would seem a viable investigation which could provide a cost-effective evaluation. Different media configuration should also be used in a future study.

3. Since the existence of non-adsorbable and very low adsorbable organics in the raw leachate, especially in the HS leachate was noted, a knowledge of the relative adsorbabilities of these organics is essential. It is also important to carry out a study on the post-treatment of leachate before AC adsorption. One such study would be to make a comparison

between the use of AC adsorption process to treat settled leachate and biologically degraded leachate.

4. To investigate the feasibility of the option proposed in the conclusion, any future study carried out should include the evaluation of leachate treatment using a combination of the three treatment processes. A complete process study together with a cost analysis should also be attempted to find the optimum degree of treatment from one phase to another.

5. Several types of activated carbon should be used to determine the best possible type to be used for treating leachate. The removal of specific organics in the leachate cannot be simply determined from the evaluation of COD or TOC removal, therefore in order to investigate the competitive adsorption in leachate, an evaluation of such organic constituents should be carried out.

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APPENDIX A

ANALYTICAL METHODS

The parameters monitored during the different stages of the study and the methods used for the analyses are summarized in Table A.1. A detailed description of the analytical instruments and other facilities used is as follows:

GAS CHROMATOGRAPH FOR THE DETRMINATION OF CH_4 AND CO_2

Model: Becker 403 with Thermal Conductivity Detector
Carrier Gas: Helium 50 ml/min
Packing: Poropack Q with metal column 1.5 m x 4 mm bore
Column Temperature: 55°C
Sample Size: 1 ml

GAS-LIQUID CHROMATOGRAPH FOR THE DETERMINATION OF VFA

Model: PYE Unicam 304 with Flame Ionization Detector, incorporated with PU 4700 Autojector and CDP4 Philips Computing Integrater
Column: 2 m x 2 mm bore packed with 10% AT-1000 on 80/100 Mesh Chromosorb W-AW
Oven Temperature: 145°C
Injector Temperature: 165°C
Detector Temperature: 165°C
Sample Size: 1 μl
Carrier Gas: Nitrogen at 25 ml/min

ATOMIC ABSORPTION SPECTROPHOTOMETER FOR METAL DETERMINATION

Model: PYE Unicam SP9 incorporated with SP9 Computer
Liquid Sampling Rate: 6 ml/min at 0.5 sec interval
Flame: Air/Acetylene, fuel lean
Burner Type: Nebulizer - Spray Chamber

TOTAL ORGANIC CARBON ANALYZER

Model: Beckman Model 915B TOC Computational System
Carrier Gas: Oxygen
Temperature: Total Carbon Furnace (953°C)
Inorganic Carbon Furnace (162°C)
Sample Size: 30 µl

INSTRUMENTATION/MANUFACTURERS/DISTRIBUTORS

INSTRUMENTATION	MANUFACTURER
SHAKER (Orbital Mixer)	Denley Ins. Ltd. Natts Lane Billinghurst Sussex RI14 9EY
SPECTROPHOTOMETER (SP 500 Series)	Unicam UK
PORTABLE WASTEWATER SAMPLER (Epic 1011 Auto-Sampler)	Epic Products Ltd.
CENTRIFUGE (MSE Multex)	Fison Plc. Crawley Sussex RH10 2UL

Table A.1: Parameters Monitored and Methods Used

Parameter	Method	Reference/Instrument
COD	Dichromate Reflux	Standard Methods (1985)
TOC	Combustion Infrared	Beckman 915A Analyzer
BOD_5	Dilution Technique	Standard Methods (1985)
pH	pH Meter	Corning EEL Model 7
Alkalinity	Volumetric Method	Yang & Anderson (1990)
VFA	Gas-Liquid Chromatography	PYE Unicam 304
Solids (SS & VSS)	Gravimetric	Standard Methods (1985)
Gas Production	Wet Gas Meter	M75-IN5 Alexander Wright
Gas Composition	Gas Chromatography	Becker 403 (CH_4 & CO_2)
TKN & NH_3-N	Distillation/ Titration	Standard Methods (1985)
Phosphate	Ascorbic Acid Method	Standard Methods (1985)
Metal Analysis	Atomic Absorption Spectrophotometer	PYE Unicam SP9

APPENDIX B

TRACE METAL SOLUTION

	g/l
Ferric Chloride	5.0
Calcium Chloride	5.0
Potassium Chloride	5.0
Cobalt Chloride	5.0
Magnesium Sulphate	5.0
Distilled Water	

APPENDIX C

Experimental Data for the RBC and the UAF during Phases I, II and III studies.

RESULTS FOR ROTATING BIOLOGICAL CONTACTOR

Date 1990	Day	C O D mg/l infl.	set.eff	% rem	fil.eff	% rem	Q (l/d)	HRT (hrs)	VOLR Kg COD /m3.d	CODres Kg COD /m3.d	CODrem Kg COD /m3.d	AOLR g COD/ m2.d	resCOD g COD/ m2.d	remCOD g COD/ m2.d
Aug 12	1	910.00	whey powder				12.70	8.315	2.63	0.00	2.63	13.63	0.00	13.63
13	2	730.00	400.00	45.21	356.00	51.23	14.00	7.543	2.32	1.13	1.19	12.05	5.88	6.17
15	4	1132.00	456.00	59.72	394.00	65.19	13.50	7.822	3.47	1.21	2.26	18.02	6.27	11.75
16	5	1272.00	378.00	70.28	333.00	73.82	12.50	8.448	3.61	0.95	2.67	18.75	4.91	13.84
19	8	1093.00	442.00	59.56	402.00	63.22	13.00	8.123	3.23	1.19	2.04	16.76	6.16	10.59
21	10	1458.00	409.00	71.95	371.00	74.55	14.00	7.543	4.64	1.18	3.46	24.07	6.13	17.95
23	12	1562.00	350.00	77.59	313.00	79.96	14.00	7.543	4.97	1.00	3.97	25.79	5.17	20.62
25	14	1600.00	439.00	72.56	368.00	77.00	14.00	6.514	5.89	1.36	4.54	26.42	6.08	20.34
26	15	1527.00	500.00	67.26	467.00	69.42	15.00	6.080	6.03	1.84	4.18	27.01	8.26	18.75
28	17	1347.00	400.00	70.30	373.00	72.31	17.00	5.365	6.03	1.67	4.36	27.00	7.48	19.53
30	19	1404.00	462.00	67.09	409.00	70.87	16.00	5.700	5.91	1.72	4.19	26.49	7.72	18.77
Sept 01	21	1308.00	432.00	66.97	374.00	71.41	14.00	6.514	4.82	1.38	3.44	21.59	6.17	15.42
03	23	1141.00	306.00	73.18	278.00	75.64	16.20	5.630	4.86	1.19	3.68	21.80	5.31	16.49
05	25	1464.00	302.00	79.37	277.00	81.08	16.30	5.595	6.28	1.19	5.09	28.14	5.32	22.82
08	28	1101.00	188.00	82.92	153.00	86.10	17.00	5.365	4.93	0.68	4.24	22.07	3.07	19.00
09	29	1090.00	160.00	85.32	128.00	88.26	16.70	4.743	5.52	0.65	4.87	21.47	2.52	18.95
11	31	1317.00	112.00	91.50	98.00	92.56	13.00	6.092	5.19	0.39	4.80	20.19	1.50	18.69
13	33	1057.00	140.00	86.75	119.00	88.74	14.00	5.657	4.48	0.50	3.98	17.45	1.96	15.49
15	35	997.00	140.00	85.96	121.00	87.86	12.50	6.336	3.78	0.46	3.32	14.70	1.78	12.91
17	37	1256.00	384.00	69.43	327.00	73.96	10.37	7.637	3.95	1.03	2.92	15.36	4.00	11.36
19	39	1340.00	323.00	75.90	298.00	77.76	11.24	7.046	4.56	1.02	3.55	17.76	3.95	13.81
21	41	1176.00	248.00	78.91	213.00	81.89	11.61	6.822	4.14	0.75	3.39	16.10	2.92	13.18
23	43	1010.00	152.00	84.95	129.00	87.23	13.90	5.698	4.25	0.54	3.71	16.56	2.11	14.44
24	44	949.00	159.00	83.25	148.00	84.40	15.30	5.176	4.40	0.69	3.71	17.12	2.67	14.45

	Average C O D												
whey powder	1207.83	405.83	64.05	361.50	68.00	13.50	7.837	3.71	1.11	2.60	19.24	5.75	13.49
leachate (C)	1387.83	423.17	69.56	378.17	72.77	15.37	5.967	5.59	1.53	4.07	25.05	6.84	18.22
leachate (FL)	1159.73	209.82	82.21	182.82	84.53	13.81	6.015	4.68	0.72	3.96	18.81	2.89	15.92

RESULTS FOR ROTATING BIOLOGICAL CONTACTOR START-UP

		B O D mg/l			Q	HRT	VOLR	BODres	BODrem	AOLR	resBOD	remBOD
Date 1989	Day	infl.	fil.eff	% rem	(l/d)	(hrs)	Kg BOD /m3.d	Kg BOD /m3.d	Kg BOD /m3.d	g BOD/ m2.d	g BOD/ m2.d	g BOD/ m2.d
Aug 12	1	525.00			12.70	8.315	1.52	0.00	1.52	7.86	0.00	7.86
13	2	398.00	205.00	48.49	14.00	7.543	1.27	0.65	0.61	6.57	3.38	3.19
15	4	657.00	126.00	80.82	13.50	7.822	2.02	0.39	1.63	10.46	2.01	8.45
16	5	732.00	89.00	87.84	12.50	8.448	2.08	0.25	1.83	10.79	1.31	9.48
19	8	614.00	94.00	84.69	13.00	8.123	1.81	0.28	1.54	9.41	1.44	7.97
23	12	776.00	77.50	90.01	14.00	7.543	2.47	0.25	2.22	12.81	1.28	11.53
25	14	735.00	85.00	88.44	14.00	6.514	2.71	0.31	2.39	12.13	1.40	10.73
26	15	678.00	92.00	86.43	15.00	6.080	2.68	0.36	2.31	11.99	1.63	10.37
30	19	656.00	95.00	85.52	16.00	5.700	2.76	0.40	2.36	12.38	1.79	10.58
Sept 01	21	650.00	81.00	87.54	14.00	6.514	2.39	0.30	2.10	10.73	1.34	9.39
03	23	495.00	50.00	89.90	16.20	4.889	2.43	0.25	2.18	9.46	0.96	8.50
08	28	454.00	33.00	92.73	17.00	4.659	2.34	0.17	2.17	9.10	0.66	8.44
09	29	468.00	25.00	94.66	16.70	4.743	2.37	0.13	2.24	9.22	0.49	8.72
13	33	440.00	21.00	95.23	14.00	5.657	1.87	0.09	1.78	7.26	0.35	6.92
15	35	304.00	19.00	93.75	12.50	6.336	1.15	0.07	1.08	4.48	0.28	4.20
17	37	410.00	38.00	90.73	10.37	7.637	1.29	0.12	1.17	5.01	0.46	4.55
19	39	487.00	35.00	92.81	11.24	7.046	1.66	0.12	1.54	6.46	0.46	5.99
21	41	355.00	21.50	93.94	11.61	6.822	1.25	0.08	1.17	4.86	0.29	4.57
23	43	312.00	19.00	93.91	13.90	5.698	1.31	0.08	1.23	5.11	0.31	4.80
24	44	345.00	26.00	92.46	15.30	5.176	1.60	0.12	1.48	6.22	0.47	5.76
		Average BOD										
whey powder		635.40	118.30	78.37	13.40	7.966	1.93	0.36	1.57	10.01	1.88	8.12
leachate (C)		642.80	80.60	87.56	15.04	5.939	2.59	0.32	2.27	11.34	1.42	9.92
leachate (FL)		397.22	26.39	93.36	13.62	5.975	1.65	0.11	1.54	6.41	0.42	5.99

RESULTS FOR ROTATING BIOLOGICAL CONTACTOR DURING START-UP

		T O·C mg/l			Q	HRT	VOLR	TOCres	TOCrem	AOLR	resTOC	remTOC
Date 1990	Day	infl.	fil.eff	% rem	(l/d)	(hrs)	Kg TOC /m3.d	Kg TOC /m3.d	Kg TOC /m3.d	g TOC/ m2.d	g TOC/ m2.d	g TOC/ m2.d
Aug 12	1	289.00			12.70	8.315	0.83	0.00	0.83	4.33	0.00	4.33
13	2	245.00	98.00	60.00	14.00	7.543	0.78	0.31	0.47	4.04	1.62	2.43
16	5	361.00	102.00	71.75	12.50	8.448	1.03	0.29	0.74	5.32	1.50	3.82
19	8	306.00	64.00	79.08	13.00	8.123	0.90	0.19	0.71	4.69	0.98	3.71
23	12	392.00	75.00	80.87	14.00	7.543	1.25	0.24	1.01	6.47	1.24	5.23
25	14	518.00	71.00	86.29	14.00	6.514	1.91	0.26	1.65	8.55	1.17	7.38
26	15	383.00	91.00	76.24	15.00	6.080	1.51	0.36	1.15	6.77	1.61	5.17
28	17	343.00	56.00	83.67	17.00	5.365	1.53	0.25	1.28	6.88	1.12	5.75
30	19	379.00	61.00	83.91	16.00	5.700	1.60	0.26	1.34	7.15	1.15	6.00
Sept 01	21	395.00	57.00	85.57	14.00	6.514	1.46	0.21	1.25	6.52	0.94	5.58
03	23	323.00	80.00	75.23	16.20	4.889	1.59	0.39	1.19	6.17	1.53	4.64
05	25	350.00	45.00	87.14	16.30	4.859	1.73	0.22	1.51	6.73	0.86	5.86
08	28	365.00	68.00	81.37	17.00	4.659	1.88	0.35	1.53	7.32	1.36	5.95
09	29	311.00	34.00	89.07	16.70	4.743	1.57	0.17	1.40	6.12	0.67	5.46
11	31	345.00	53.00	84.64	13.00	6.092	1.36	0.21	1.15	5.29	0.81	4.48
13	33	346.00	51.00	85.26	14.00	5.657	1.47	0.22	1.25	5.71	0.84	4.87
15	35	235.00	31.00	86.81	12.50	6.336	0.89	0.12	0.77	3.46	0.46	3.01
17	37	363.00	90.00	75.21	10.37	7.637	1.14	0.28	0.86	4.44	1.10	3.34
19	39	300.00	63.00	79.00	11.24	7.046	1.02	0.21	0.81	3.98	0.84	3.14
21	41	304.00	84.00	72.37	11.61	6.822	1.07	0.30	0.77	4.16	1.15	3.01
23	43	205.00	38.00	81.46	13.90	5.698	0.86	0.16	0.70	3.36	0.62	2.74
24	44	186.00	25.00	86.56	15.30	5.176	0.86	0.12	0.75	3.36	0.45	2.90

	Average T O C										
whey powder	326.00	84.75	72.92	13.38	7.914	0.99	0.26	0.73	5.13	1.34	3.80
leachate (C)	390.17	69.33	81.82	15.37	5.844	1.60	0.29	1.31	7.01	1.25	5.75
leachate (FL)	300.91	52.91	82.63	13.81	5.884	1.26	0.21	1.05	4.90	0.83	4.07

RESULTS FOR ANAEROBIC FILTER START-UP
Vol = 6.3 litres

		C O D mg/l					biogas	CH4	CO2	Q	HRT	VOLR	AOLR	EMP
Date 1989	Day	infl.	set.eff	% rem	fil.eff	% rem	(l/d)	(%)	(%)	(l/d)	(hrs)	Kg COD /m3.d	g COD /m2.d	m3 CH4/ Kg CODred
July 01	1	980.00	689.00	29.69	656.00	33.06	0.30	38.00	52.00	6.20	24.39	0.964	5.524	0.057
03	3	1073.00	574.00	46.51	543.00	49.39	0.60	43.70	56.30	6.66	22.70	1.134	6.497	0.074
05	5	922.00	535.00	41.97	512.00	44.47	0.80	49.10	50.90	6.52	23.19	0.954	5.465	0.147
07	7	965.00	446.00	53.78	410.00	57.51	1.10	62.60	37.40	7.00	21.60	1.072	6.141	0.177
10	10	1149.00	423.00	63.19	398.00	65.36	1.40	77.20	22.80	7.34	20.60	1.339	7.667	0.196
13	13	1042.00	565.00	45.78	524.00	49.71	1.30	78.00	22.00	8.45	17.89	1.398	8.004	0.232
16	16	826.00	253.00	69.37	239.00	71.07	1.30	82.60	17.40	8.83	17.12	1.158	6.631	0.207
19	19	1084.00	291.00	73.15	264.00	75.65	2.30	87.20	12.80	8.90	16.99	1.531	8.771	0.275
21	21	873.00	256.00	70.68	228.00	73.88	2.25	83.90	16.10	8.95	16.89	1.240	7.103	0.327
23	23	1104.00	303.00	72.55	276.00	75.00	2.45	84.30	15.70	8.80	17.18	1.542	8.832	0.283
25	25	934.00	242.00	74.09	211.00	77.41	2.40	85.00	15.00	9.10	16.62	1.349	7.727	0.310
27	27	1068.00	390.00	63.48	368.00	65.54	2.30	82.40	17.60	8.95	16.89	1.517	8.690	0.303
29	29	889.00	222.00	75.03	194.00	78.18	2.20	86.70	13.30	9.03	16.74	1.274	7.298	0.304
31	31	1056.00	288.00	72.73	263.00	75.09	2.45	84.00	16.00	8.90	16.99	1.492	8.544	0.292
Aug 02	33	966.00	257.00	73.40	226.00	76.60	2.40	81.90	18.10	8.75	17.28	1.342	7.684	0.304
04	35	812.00	195.00	75.99	172.00	78.82	2.10	83.20	16.80	8.85	17.08	1.141	6.533	0.308
06	37	936.00	240.00	74.36	208.00	77.78	2.50	85.50	14.50	9.00	16.80	1.337	7.658	0.326
08	39	902.00	189.00	79.05	164.00	81.82	2.30	86.10	13.90	8.88	17.03	1.271	7.282	0.302
10	41	1098.00	275.00	74.95	252.00	77.05	2.00	86.80	13.20	6.50	23.26	1.133	6.488	0.316
12	43	1401.00	229.00	83.65	219.00	84.37	2.05	88.00	12.00	5.50	27.49	1.223	7.005	0.277
14	45	1268.00	195.00	84.62	167.00	86.83	2.00	88.40	11.60	6.01	25.16	1.210	6.928	0.267
16	47	1186.00	300.00	74.70	272.00	77.07	1.85	84.10	15.90	6.09	24.83	1.146	6.566	0.280
18	49	1078.00	234.00	78.29	202.00	81.26	1.70	85.00	15.00	5.74	26.34	0.982	5.625	0.287
20	51	1433.00	297.00	79.27	256.00	82.14	1.80	88.70	11.30	5.50	27.49	1.251	7.165	0.247
23	54	1584.00	280.00	82.32	206.00	86.99	1.95	87.90	12.10	5.50	27.49	1.383	7.920	0.226
25	56	1600.00	307.00	80.81	239.00	85.06	2.50	85.00	15.00	5.66	26.71	1.437	8.233	0.276
26	57	1527.00	192.00	87.43	150.00	90.18	2.40	82.40	17.60	5.70	26.53	1.382	7.913	0.252
28	59	1347.00	129.00	90.42	103.00	92.35	2.30	83.10	16.90	5.52	27.39	1.180	6.759	0.278
30	61	1404.00	178.00	87.32	156.00	88.89	2.60	84.00	16.00	5.50	27.49	1.226	7.020	0.318
Sept 01	63	1308.00	170.00	87.00	152.00	88.38	2.00	78.90	21.10	5.00	30.24	1.038	5.945	0.273
03	65	1141.00	108.00	90.53	96.00	91.59	2.35	81.00	19.00	5.95	25.41	1.078	6.172	0.306
05	67	1464.00	209.00	85.72	158.00	89.21	2.70	85.70	14.30	6.70	22.57	1.557	8.917	0.264
08	70	1101.00	92.00	91.64	82.00	92.55	2.30	86.70	13.30	6.80	22.24	1.188	6.806	0.288
09	71	1090.00	80.00	92.66	73.00	93.30	2.60	82.00	18.00	6.70	22.57	1.159	6.639	0.313
11	73	1317.00	65.00	95.06	60.00	95.44	2.70	86.00	14.00	6.60	22.91	1.380	7.902	0.280
13	75	1057.00	60.00	94.32	50.00	95.27	2.60	83.70	16.30	6.45	23.44	1.082	6.198	0.335
15	77	997.00	48.00	95.19	41.00	95.89	2.70	80.40	19.60	6.20	24.39	0.981	5.619	0.366
17	79	1256.00	95.00	92.44	87.00	93.07	3.00	82.50	17.50	6.39	23.66	1.274	7.296	0.331
19	81	1340.00	136.00	89.85	114.00	91.49	3.30	87.80	12.20	7.55	20.03	1.606	9.197	0.313
21	83	1176.00	112.00	90.48	99.00	91.58	2.90	84.00	16.00	6.40	23.62	1.195	6.842	0.353
23	85	1010.00	101.00	90.00	96.00	90.50	2.50	87.00	13.00	7.20	21.00	1.154	6.611	0.331
24	86	949.00	134.00	85.88	125.00	86.83	2.00	89.10	10.90	7.00	21.60	1.054	6.039	0.309

Average C O D

	(Infl.)	(S.Eff.)		(F.Eff.)						HRT	OLR		
Synt. feed	995.64	416.09	58.25	387.36	61.14	1.47	70.15	28.95	7.89	19.56	1.244	7.124	0.208
Leachate (C)	1039.60	248.00	75.73	223.30	78.21	2.23	85.30	14.70	8.04	19.47	1.294	7.411	0.300
Leachate (FL)	1360.80	219.50	83.81	183.20	86.39	2.15	84.01	15.99	5.62	26.99	1.210	6.932	0.274
Leachate (FL)	1159.73	102.91	91.20	89.55	92.28	2.66	84.99	15.01	6.73	22.55	1.239	7.097	0.317
(FL57+FL58)	1255.48	158.43	87.68	134.14	89.48	2.42	84.52	15.48	6.20	24.66	1.225	7.018	0.297

RESULTS FOR ANAEROBIC FILTER START-UP
Vol = 6.3 litres

Date 1989	Day	B O D mg/l infl.	fil.eff	% rem	Q (l/d)	HRT (hrs)	VOLR Kg BOD /m3.d	AOLR g BOD /m2.d
July 01	1	333.00	187.00	43.84	6.20	24.39	0.328	1.877
03	3	387.00	185.00	52.20	6.66	22.70	0.409	2.343
07	7	438.00	153.00	65.07	7.00	21.60	0.487	2.787
10	10	387.00	117.00	69.77	7.34	20.60	0.451	2.582
13	13	354.00	135.00	61.86	8.45	17.89	0.475	2.719
16	16	302.00	74.00	75.50	8.83	17.12	0.423	2.424
19	19	444.00	65.00	85.36	8.90	16.99	0.627	3.592
25	25	421.00	62.50	85.15	9.10	16.62	0.608	3.483
27	27	532.00	105.00	80.26	8.95	16.89	0.756	4.329
31	31	590.00	92.50	84.32	8.90	16.99	0.833	4.774
Aug 04	35	402.00	55.00	86.32	8.85	17.08	0.565	3.234
06	37	457.00	62.50	86.32	9.00	16.80	0.653	3.739
10	41	600.00	64.00	89.33	6.50	23.26	0.619	3.545
14	45	676.00	59.00	91.27	6.01	25.16	0.645	3.693
16	47	545.00	59.50	89.08	6.09	24.83	0.527	3.017
20	51	785.00	62.50	92.04	5.50	27.49	0.685	3.925
23	54	724.00	54.00	92.54	5.50	27.49	0.632	3.620
25	56	735.00	37.50	94.90	5.66	26.71	0.660	3.782
26	57	678.00	25.00	96.31	5.70	26.53	0.613	3.513
30	61	656.00	22.50	96.57	5.50	27.49	0.573	3.280
Sept 01	63	650.00	47.50	92.69	5.00	30.24	0.516	2.955
03	65	495.00	30.00	93.94	5.95	25.41	0.468	2.677
08	70	454.00	12.00	97.36	6.80	22.24	0.490	2.807
09	71	468.00	15.50	96.69	6.70	22.57	0.498	2.851
13	75	440.00	15.00	96.59	6.45	23.44	0.450	2.580
15	77	304.00	12.00	96.05	6.20	24.39	0.299	1.713
17	79	410.00	18.50	95.49	6.39	23.66	0.416	2.382
19	81	487.00	28.00	94.25	7.55	20.03	0.584	3.343
21	83	355.00	25.00	92.96	6.40	23.62	0.361	2.065
23	85	312.00	10.00	96.79	7.20	21.00	0.357	2.042
24	86	345.00	18.00	94.78	7.00	21.60	0.383	2.195

	Avg. B O D				HRT	OLR	
synthetic	383.25	122.31	67.34	7.81	19.74	0.476	2.726
leachate (C)	542.83	73.00	86.31	8.04	19.36	0.678	3.886
leachate (FL)	658.50	42.31	93.51	5.61	27.02	0.584	3.346
leachate (FL)	397.22	17.11	95.66	6.74	22.50	0.426	2.442
(FL45+FL46)	520.18	28.97	94.65	6.21	24.63	0.501	2.867

RESULTS FOR ANAEROBIC FILTER START-UP

		T O C mg/l			Q	HRT	VOLR	AOLR
Date 1989	Day	infl.	fil.eff	% rem	(l/d)	(hrs)	Kg TOC /m3.d	g TOC /m2.d
July 01	1	200.00	125.00	37.50	6.20	24.39	0.197	1.127
03	3	276.00	132.00	52.17	6.66	22.70	0.292	1.671
05	5	264.00	107.00	59.47	6.52	23.19	0.273	1.565
07	7	212.00	91.00	57.08	7.00	21.60	0.236	1.349
10	10	253.00	88.00	65.22	7.34	20.60	0.295	1.688
13	13	199.00	95.00	52.26	8.45	17.89	0.267	1.529
16	16	187.00	46.00	75.40	8.83	17.12	0.262	1.501
19	19	235.00	52.00	77.87	8.90	16.99	0.332	1.901
21	21	194.00	45.00	76.80	8.95	16.89	0.276	1.578
23	23	242.00	50.00	79.34	8.80	17.18	0.338	1.936
25	25	203.00	43.00	78.82	9.10	16.62	0.293	1.679
27	27	232.00	75.00	67.67	8.95	16.89	0.330	1.888
29	29	178.00	41.00	76.97	9.03	16.74	0.255	1.461
31	31	229.00	49.00	78.60	8.90	16.99	0.324	1.853
Aug 02	33	212.00	40.00	81.13	8.75	17.28	0.294	1.686
04	35	176.00	29.00	83.52	8.85	17.08	0.247	1.416
06	37	189.00	36.00	80.95	9.00	16.80	0.270	1.546
08	39	178.00	30.00	83.15	8.88	17.03	0.251	1.437
10	41	213.00	44.00	79.34	6.50	23.26	0.220	1.259
12	43	316.00	46.00	85.44	5.50	27.49	0.276	1.580
14	45	266.00	42.00	84.21	6.01	25.16	0.254	1.453
16	47	223.00	48.00	78.48	6.09	24.83	0.216	1.235
18	49	245.00	56.00	77.14	5.74	26.34	0.223	1.278
20	51	321.00	62.00	80.69	5.50	27.49	0.280	1.605
23	54	445.00	43.00	90.34	5.50	27.49	0.388	2.225
25	56	518.00	60.00	88.42	5.66	26.71	0.465	2.665
26	57	383.00	31.00	91.91	5.70	26.53	0.347	1.985
28	59	343.00	33.00	90.38	5.52	27.39	0.301	1.721
30	61	379.00	39.00	89.71	5.50	27.49	0.331	1.895
Sept 01	63	395.00	43.00	89.11	5.00	30.24	0.313	1.795
03	65	323.00	36.00	88.85	5.95	25.41	0.305	1.747
05	67	350.00	40.00	88.57	6.70	22.57	0.372	2.132
08	70	365.00	44.00	87.95	6.80	22.24	0.394	2.256
09	71	311.00	37.00	88.10	6.70	22.57	0.331	1.894
11	73	345.00	22.00	93.62	6.60	22.91	0.361	2.070
13	75	346.00	42.00	87.86	6.45	23.44	0.354	2.029
15	77	235.00	31.00	86.81	6.20	24.39	0.231	1.325
17	79	363.00	37.00	89.81	6.39	23.66	0.368	2.109
19	81	300.00	46.00	84.67	7.55	20.03	0.360	2.059
21	83	304.00	34.00	88.82	6.40	23.62	0.309	1.769
23	85	205.00	60.00	70.73	7.20	21.00	0.234	1.342
24	86	186.00	33.00	82.26	7.00	21.60	0.207	1.184

	Avg. T O C						
	(Infl.)	(F.Eff.)	(%)		HRT	OLR	
Synt. feed	224.09	79.45	64.72	7.89	19.56	0.278	1.593
Leachate (C)	218.90	43.20	80.10	8.04	19.47	0.272	1.558
Leachate (FL)	357.50	45.10	86.50	5.62	26.99	0.317	1.815
Leachate (FL)	300.91	38.73	86.29	6.73	22.55	0.320	1.833
(FL56+FL57)	327.86	41.76	86.39	6.20	24.66	0.319	1.825

RESULTS FOR ROTATING BIOLOGICAL CONTACTOR (pH VARIATION)

.		C O D mg/l			Q	HRT	VOLR	AOLR	pH value		alkalinity	
Date 1989	Day	infl.	fil.eff	% rem	(l/d)	(day)	Kg COD /m3.d	g COD /m2.d	infl.	eff.	infl.	eff.
Sept 27	47	1640.00	364.00	77.80	13.68	5.789	6.80	26.46	7.05	8.05	550	820
29	49	1468.00	360.00	75.48	12.82	6.178	5.70	22.19	6.75	7.65	600	700
Oct 01	51	1236.00	314.00	74.60	13.62	5.815	5.10	19.85	7.05	7.75	650	690
02	52	1640.00	320.00	80.49	11.59	6.833	5.76	22.41	6.30	7.80	360	640
04	54	1340.00	323.00	75.90	11.24	7.046	4.56	17.76	6.50	7.75	480	620
05	55	1344.00	348.00	74.11	10.39	7.623	4.23	16.47	6.50	7.60	410	510
07	57	1251.00	351.00	71.94	10.08	7.857	3.82	14.87	5.95	7.70	225	365
08	58	1279.00	334.00	73.89	11.70	6.769	4.53	17.65	5.85	7.70	200	500
10	60	1450.00	512.00	64.69	11.24	7.046	4.94	19.22	5.20	7.60	115	490
12	62	1384.00	458.00	66.91	9.97	7.944	4.18	16.27	5.35	7.70	120	400
14	64	1322.00	462.00	65.05	11.04	7.174	4.42	17.21	5.05	7.80	50	380
16	66	1504.00	585.00	61.10	12.20	6.492	5.56	21.64	4.50	7.70	-ve	410
18	68	1311.00	508.00	61.25	13.98	5.665	5.55	21.61	3.90	7.80	-ve	390
20	70	1095.00	412.00	62.37	12.05	6.573	4.00	15.56	4.00	7.70	-ve	360
21	71	1426.00	469.00	67.11	12.50	6.336	5.40	21.02	4.40	7.85	-ve	380
23	73	1270.00	353.00	72.20	14.06	5.633	5.41	21.06	5.10	7.60	-ve	390
25	75	1465.00	489.00	66.62	10.23	7.742	4.54	17.67	5.50	7.60	110	420
27	77	1333.00	400.00	69.99	10.98	7.213	4.44	17.26	5.95	7.55	230	395
29	79	1152.00	388.00	66.32	11.94	6.633	4.17	16.22	6.05	7.65	235	425
30	80	1266.00	374.00	70.46	11.98	6.611	4.60	17.89	6.70	7.30	210	255
Nov 01	82	1680.00	433.00	74.23	11.71	6.763	5.96	23.20	6.75	7.75	650	880
03	84	1424.00	399.00	71.98	11.52	6.875	4.97	19.34	6.90	7.80	1005	1065
05	86	1616.00	412.00	74.50	12.33	6.423	6.04	23.50	7.45	8.05	1490	1720
06	87	1592.00	352.00	77.89	12.19	6.497	5.88	22.88	7.55	8.30	1585	1640
08	89	1440.00	363.00	74.79	11.33	6.990	4.94	19.24	7.45	8.15	1585	1610
10	91	1608.00	361.00	77.55	11.76	6.735	5.73	22.30	7.70	8.25	1600	1480
13	94	1320.00	397.00	69.92	11.90	6.655	4.76	18.52	7.85	8.20	2060	1950
15	96	1250.00	428.00	65.76	10.94	7.239	4.14	16.13	8.10	8.35	2095	1900
17	98	1585.00	555.00	64.98	11.64	6.804	5.59	21.76	8.50	8.80	2135	2040
19	100	1472.00	578.00	60.73	11.86	6.678	5.29	20.59	8.80	9.00	2220	2065
21	102	1396.00	510.00	63.47	11.64	6.804	4.92	19.16	8.45	8.85	2210	2070
22	103	1422.00	489.00	65.61	12.28	6.450	5.29	20.59	8.20	8.50	2100	1950
24	105	1284.00	362.00	71.81	11.75	6.740	4.57	17.79	7.75	8.05	1290	1300
26	107	1136.00	323.00	71.57	11.96	6.622	4.12	16.02	7.55	8.10	975	1005
27	108	1212.00	334.00	72.44	12.48	6.346	4.58	17.84	7.50	8.00	1055	1105
28	109	1800.00	381.00	78.83	11.84	6.689	6.46	25.13	7.30	8.15	1410	1620
Dec 01	112	1336.00	316.00	76.35	12.31	6.434	4.98	19.39	7.35	8.05	975	1115
02	113	1360.00	329.00	75.81	12.45	6.361	5.13	19.97	7.30	8.00	825	1025
		1397.61			11.87	6.71	5.03	19.57				

RESULTS FOR ROTATING BIOLOGICAL CONTACTOR (pH VARIATION)

		B O D mg/l			Q	HRT	VOLR	AOLR	pH value		alkalinity	
Date	Day	infl.	fil.eff	% rem	(l/d)	(hrs)	Kg BOD	g BOD	infl.	eff.	infl.	eff.
1989							/m3.d	/m2.d				
Sept 27	47	584.00	43.00	92.64	13.68	5.789	2.42	9.42	7.05	8.05	550	820
Oct 01	51	354.00	31.50	91.10	13.62	5.815	1.46	5.69	7.05	7.75	650	690
02	52	463.00	72.00	84.45	11.59	6.833	1.63	6.33	6.30	7.80	360	640
05	55	385.00	55.00	85.71	10.39	7.623	1.21	4.72	6.50	7.60	410	510
07	57	276.00	34.00	87.68	10.08	7.857	0.84	3.28	5.95	7.70	225	365
08	58	424.00	76.00	82.08	11.70	6.769	1.50	5.85	5.85	7.70	200	500
10	60	395.00	78.30	80.18	11.24	7.046	1.35	5.24	5.20	7.60	115	490
12	62	339.00	72.00	78.76	9.97	7.944	1.02	3.99	5.35	7.70	120	400
14	64	398.00	89.00	77.64	11.04	7.174	1.33	5.18	5.05	7.80	50	380
16	66	471.00	124.00	73.67	12.20	6.492	1.74	6.78	4.50	7.70	-ve	410
18	68	356.00	114.00	67.98	13.98	5.665	1.51	5.87	3.90	7.80	-ve	390
20	70	264.00	88.00	66.67	12.05	6.573	0.96	3.75	4.00	7.70	-ve	360
21	71	352.00	105.00	70.17	10.08	7.857	1.08	4.18	4.40	7.85	-ve	380
23	73	374.00	99.00	73.53	14.06	5.633	1.59	6.20	5.10	7.60	-ve	390
25	75	411.00	82.80	79.85	10.23	7.742	1.27	4.96	5.50	7.60	110	420
27	77	300.00	59.00	80.33	10.98	7.213	1.00	3.88	5.95	7.55	230	395
29	79	310.00	62.50	79.84	11.94	6.633	1.12	4.36	6.05	7.65	235	425
Nov 01	82	610.00	106.30	82.57	11.71	6.763	2.16	8.42	6.75	7.75	650	880
03	84	497.00	77.00	84.51	11.52	6.875	1.73	6.75	6.90	7.80	1005	1065
06	87	538.00	115.00	78.62	12.19	6.497	1.99	7.73	7.55	8.30	1585	1640
08	89	680.00	182.50	73.16	11.33	6.990	2.33	9.09	7.45	8.15	1585	1610
10	91	425.00	111.30	73.81	11.76	6.735	1.51	5.89	7.70	8.25	1600	1480
13	94	645.00	161.30	74.99	11.90	6.655	2.33	9.05	7.85	8.20	2060	1950
15	96	375.00	117.00	68.80	10.94	7.239	1.24	4.84	8.10	8.35	2095	1900
19	100	453.00	168.00	62.91	11.86	6.678	1.63	6.34	8.80	9.00	2220	2065
21	102	472.00	163.00	65.47	11.64	6.804	1.66	6.48	8.45	8.85	2210	2070
22	103	406.00	152.00	62.56	12.28	6.450	1.51	5.88	8.20	8.50	2100	1950
24	105	333.00	88.50	73.42	11.75	6.740	1.19	4.61	7.75	8.05	1290	1300
26	107	356.00	72.00	79.78	11.96	6.622	1.29	5.02	7.55	8.10	975	1005
28	109	503.00	88.00	82.50	11.84	6.689	1.80	7.02	7.30	8.15	1410	1620
Dec 01	112	395.00	94.00	76.20	12.31	6.434	1.47	5.73	7.35	8.05	975	1115
02	113	463.00	85.00	81.64	12.45	6.361	1.75	6.80	7.30	8.00	825	1025

RESULTS FOR ROTATING BIOLOGICAL CONTACTOR (pH VARIATION)

Date 1990	Day	pH value infl.	pH value eff.	T O C mg/l infl.	T O C mg/l fil.eff	T O C mg/l % rem	Q (l/d)	HRT (hrs)	VOLR Kg TOC /m3.d	AOLR g TOC /m2.d	alkalinity infl.	alkalinity eff.
Sept 27	47	7.05	8.05	379.00	120.00	68.34	13.68	5.789	1.57	6.11	550	820
29	49	6.75	7.65	264.00	113.00	57.20	12.82	6.178	1.03	3.99	600	700
Oct 01	51	7.05	7.75	213.00	68.00	68.08	13.62	5.815	0.88	3.42	650	690
02	52	6.30	7.80	296.00	98.00	66.89	11.59	6.833	1.04	4.05	360	640
04	54	6.50	7.75	408.00	147.00	63.97	11.24	7.046	1.39	5.41	480	620
05	55	6.50	7.60	239.00	97.00	59.41	10.39	7.623	0.75	2.93	410	510
07	57	5.95	7.70	248.00	85.00	65.73	10.08	7.857	0.76	2.95	225	365
08	58	5.85	7.70	238.00	99.00	58.40	11.70	6.769	0.84	3.28	200	500
10	60	5.20	7.60	261.00	116.00	55.56	11.24	7.046	0.89	3.46	115	490
12	62	5.35	7.70	288.00	109.00	62.15	9.97	7.944	0.87	3.39	120	400
14	64	5.05	7.80	246.00	99.00	59.76	11.04	7.174	0.82	3.20	50	380
16	66	4.50	7.70	301.00	118.00	60.80	12.20	6.492	1.11	4.33	-ve	410
18	68	3.90	7.80	256.00	122.00	52.34	13.98	5.665	1.08	4.22	-ve	390
20	70	4.00	7.70	228.00	116.00	49.12	12.05	6.573	0.83	3.24	-ve	360
21	71	4.40	7.85	267.00	104.00	61.05	12.50	6.336	1.01	3.94	-ve	380
23	73	5.10	7.60	240.00	89.00	62.92	14.06	5.633	1.02	3.98	-ve	390
25	75	5.50	7.60	309.00	97.00	68.61	10.23	7.742	0.96	3.73	110	420
27	77	5.95	7.55	284.00	94.00	66.90	10.98	7.213	0.94	3.68	230	395
29	79	6.05	7.65	247.00	91.00	63.16	11.94	6.633	0.89	3.48	235	425
30	80	6.70	7.30	222.00	87.00	60.81	11.98	6.611	0.81	3.14	210	255
Nov 01	82	6.75	7.75	437.00	153.00	64.99	11.71	6.763	1.55	6.03	650	880
03	84	6.90	7.80	387.00	130.00	66.41	11.52	6.875	1.35	5.26	1005	1065
05	86	7.45	8.05	441.00	177.00	59.86	12.33	6.423	1.65	6.41	1490	1720
06	87	7.55	8.30	452.00	185.00	59.07	12.19	6.497	1.67	6.50	1585	1640
08	89	7.45	8.15	334.00	127.00	61.98	11.33	6.990	1.15	4.46	1585	1610
10	91	7.70	8.25	282.00	144.00	48.94	11.76	6.735	1.00	3.91	1600	1480
13	94	7.85	8.20	308.00	146.00	52.60	11.90	6.655	1.11	4.32	2060	1950
15	96	8.10	8.35	228.00	115.00	49.56	10.94	7.239	0.76	2.94	2095	1900
17	98	8.50	8.80	295.00	154.00	47.80	11.64	6.804	1.04	4.05	2135	2040
19	100	8.80	9.00	325.00	176.00	45.85	11.86	6.678	1.17	4.55	2220	2065
21	102	8.45	8.85	298.00	146.00	51.01	11.64	6.804	1.05	4.09	2210	2070
22	103	8.20	8.50	268.00	134.00	50.00	12.28	6.450	1.00	3.88	2100	1950
24	105	7.75	8.05	227.00	119.00	47.58	11.75	6.740	0.81	3.15	1290	1300
26	107	7.55	8.10	226.00	108.00	52.21	11.96	6.622	0.82	3.19	975	1005
27	108	7.50	8.00	270.00	106.00	60.74	12.48	6.346	1.02	3.97	1055	1105
28	109	7.30	8.15	335.00	117.00	65.07	11.84	6.689	1.20	4.68	1410	1620
Dec 01	112	7.35	8.05	286.00	127.00	55.59	12.31	6.434	1.07	4.15	975	1115
02	113	7.30	8.00	344.00	141.00	59.01	12.45	6.361	1.30	5.05	825	1025

RESULTS FOR ANAEROBIC FILTER (pH VARIATION)
Vol = 6.3 litres

Date 1990	Day	pH value infl	pH value eff	alkalinity infl.	alkalinity eff.	C O D mg/l infl.	C O D mg/l fil.eff	C O D mg/l % rem	biogas (l/d)	CH4 (%)	CO2 (%)	Q (l/d)	HRT (hrs)	VOLR Kg COD /m3.d	EMP m3 CH4/ Kg CODred
Sept 27	89	7.05	6.75	550	860	1640.00	383.00	76.65	4.00	86.40	13.60	8.41	17.98	2.19	0.327
29	91	6.75	6.65	600	760	1468.00	365.00	75.14	3.90	87.20	12.80	8.34	18.13	1.94	0.370
Oct 01	93	7.05	6.80	650	780	1236.00	336.00	72.82	3.00	88.00	12.00	7.86	19.24	1.54	0.373
02	94	6.30	6.70	360	760	1640.00	355.00	78.35	3.50	86.90	13.10	8.06	18.76	2.10	0.294
04	96	6.50	6.90	480	670	1340.00	345.00	74.25	3.15	87.80	12.20	7.75	19.51	1.65	0.359
05	97	6.50	6.70	410	640	1344.00	380.00	71.73	3.10	86.60	13.40	8.11	18.64	1.73	0.343
07	99	5.95	6.90	225	590	1251.00	387.00	69.06	2.70	84.00	16.00	7.04	21.48	1.40	0.373
08	100	5.85	6.95	200	520	1279.00	404.00	68.41	3.20	87.80	12.20	8.10	18.67	1.64	0.396
10	102	5.20	6.55	115	410	1450.00	562.00	61.24	3.10	85.50	14.50	8.22	18.39	1.89	0.363
12	104	5.35	6.70	120	420	1384.00	548.00	60.40	2.50	84.90	15.10	8.00	18.90	1.76	0.317
14	106	5.05	6.60	50	330	1322.00	510.00	61.42	2.40	86.00	14.00	7.84	19.29	1.65	0.324
16	108	4.50	6.50	0	270	1504.00	672.00	55.32	2.95	86.40	13.60	7.70	19.64	1.84	0.398
18	110	3.90	6.45	0	300	1311.00	624.00	52.40	2.50	87.20	12.80	7.54	20.05	1.57	0.421
20	112	4.00	6.60	0	255	1095.00	552.00	49.59	2.00	87.90	12.10	7.86	19.24	1.37	0.412
21	113	4.40	6.55	0	210	1426.00	616.00	56.80	2.80	86.70	13.30	7.92	19.09	1.79	0.378
23	115	5.10	6.80	0	240	1270.00	486.00	61.73	2.60	85.80	14.20	7.58	19.95	1.53	0.375
25	117	5.50	6.75	110	490	1465.00	499.00	65.94	2.80	85.60	14.60	7.62	19.84	1.77	0.326
27	119	5.95	6.80	230	435	1333.00	523.00	60.77	2.60	87.00	13.00	7.49	20.19	1.58	0.373
29	121	6.05	6.60	235	470	1152.00	424.00	63.19	2.30	86.90	13.10	7.78	19.43	1.42	0.353
30	122	6.70	6.50	210	330	1266.00	413.00	67.38	2.20	89.40	10.60	7.85	19.26	1.58	0.294
Nov 01	124	6.75	7.00	650	860	1680.00	444.00	73.57	4.20	87.00	13.00	8.61	17.56	2.30	0.343
03	126	6.90	6.90	1005	1090	1424.00	426.00	70.08	3.00	88.10	11.90	8.11	18.64	1.83	0.327
05	128	7.45	7.30	1490	1700	1616.00	414.00	74.38	4.30	84.40	15.60	9.40	16.09	2.41	0.321
06	129	7.55	7.25	1585	1740	1592.00	372.00	76.63	4.15	87.30	12.70	8.00	18.90	2.02	0.371
08	131	7.45	7.05	1585	1740	1440.00	388.00	73.06	3.00	89.60	10.40	8.22	18.39	1.88	0.311
10	133	7.70	7.20	1600	1680	1608.00	416.00	74.13	2.90	88.70	11.30	7.97	18.97	2.03	0.271
13	136	7.85	7.60	2060	1960	1320.00	435.00	67.05	2.20	89.60	10.40	7.85	19.26	1.64	0.284
15	138	8.10	7.45	2095	2170	1250.00	448.00	64.16	3.20	91.30	8.70	9.59	15.77	1.90	0.380
17	140	8.50	7.65	2135	1985	1585.00	618.00	61.01	4.00	90.10	9.90	9.14	16.54	2.30	0.408
19	142	8.80	7.80	2220	2020	1472.00	602.00	59.10	3.90	89.50	10.50	8.86	17.07	2.07	0.453
21	144	8.45	7.70	2210	2095	1396.00	551.00	60.53	4.05	89.80	10.20	9.32	16.22	2.07	0.462
22	145	8.20	7.50	2100	1920	1422.00	533.00	62.52	3.90	88.90	11.10	9.00	16.80	2.03	0.433
24	147	7.75	7.20	1290	1405	1284.00	336.00	73.83	2.00	90.40	9.60	6.50	23.26	1.32	0.293
26	149	7.55	7.10	975	1085	1136.00	327.00	71.21	2.10	91.80	8.20	8.43	17.94	1.52	0.283
27	150	7.50	7.05	1055	1210	1212.00	316.00	73.93	2.90	90.00	10.00	8.76	17.26	1.69	0.333
28	151	7.30	7.45	1410	1530	1800.00	377.00	79.06	4.60	89.10	10.90	9.70	15.59	2.77	0.297
Dec 01	154	7.35	7.05	975	1180	1336.00	312.00	76.65	3.30	90.60	9.40	7.92	19.09	1.68	0.369
02	155	7.30	7.20	825	1160	1360.00	314.00	76.91	3.20	89.20	10.80	7.92	19.09	1.71	0.345
						1397.61						8.17	18.63	1.82	

RESULTS FOR ANAEROBIC FILTER (pH VARIATION)

		pH value		alkalinity		B O D mg/l			Q	HRT	VOLR
Date	Day	infl	eff	infl.	eff.	infl.	fil.eff	% rem	(l/d)	(hrs)	Kg BOD /m3.d
Phase 2											
Sept 27	89	7.05	6.75	550	860	548.00	72.00	86.86	8.41	17.98	2.19
Oct 01	93	7.05	6.80	650	780	354.00	47.50	86.58	7.86	19.24	1.54
02	94	6.30	6.70	360	760	463.00	73.00	84.23	8.06	18.76	2.10
05	97	6.50	6.70	410	640	385.00	72.50	81.17	8.11	18.64	1.73
07	99	5.95	6.90	225	590	276.00	73.00	73.55	7.04	21.48	1.40
08	100	5.85	6.95	200	520	424.00	92.00	78.30	8.10	18.67	1.64
10	102	5.20	6.55	115	410	395.00	88.00	77.72	8.22	18.39	1.89
12	104	5.35	6.70	120	420	339.00	68.00	79.94	8.00	18.90	1.76
14	106	5.05	6.60	50	330	398.00	96.00	75.88	7.84	19.29	1.65
16	108	4.50	6.50	-ve	270	471.00	165.00	64.97	7.70	19.64	1.84
18	110	3.90	6.45	-ve	300	356.00	133.00	62.64	7.54	20.05	1.57
20	112	4.00	6.60	-ve	255	264.00	119.00	54.92	7.86	19.24	1.37
21	113	4.40	6.55	-ve	210	352.00	138.00	60.80	7.92	19.09	1.79
23	115	5.10	6.80	-ve	240	374.00	99.00	73.53	7.58	19.95	1.53
25	117	5.50	6.75	110	490	411.00	114.00	72.26	7.62	19.84	1.77
27	119	5.95	6.80	320	570	300.00	81.00	73.00	7.49	20.19	1.58
29	121	6.05	6.60	235	470	310.00	86.30	72.16	7.78	19.43	1.42
Nov 01	124	6.75	7.00	650	860	610.00	62.50	89.75	8.61	17.56	2.30
03	126	6.90	6.90	1005	1090	497.00	66.00	86.72	8.11	18.64	1.83
06	129	7.55	7.25	1585	1740	538.00	83.00	84.57	8.00	18.90	2.02
08	131	7.45	7.05	1585	1740	680.00	152.50	77.57	8.22	18.39	1.88
10	133	7.70	7.20	1600	1680	425.00	136.50	67.88	7.97	18.97	2.03
13	136	7.85	7.60	2060	1960	645.00	158.80	75.38	7.85	19.26	1.64
15	138	8.10	7.45	2095	2170	375.00	120.00	68.00	9.59	15.77	1.90
19	142	8.80	7.80	2220	2020	453.00	187.00	58.72	8.86	17.07	2.07
21	144	8.45	7.70	2210	2095	472.00	184.00	61.02	9.32	16.22	2.07
22	145	8.20	7.50	2100	1920	406.00	163.00	59.85	9.00	16.80	2.03
24	147	7.75	7.20	1290	1405	333.00	96.00	71.17	6.50	23.26	1.32
26	149	7.55	7.10	975	1085	356.00	91.00	74.44	8.43	17.94	1.52
28	151	7.30	7.45	1410	1530	503.00	115.00	77.14	9.70	15.59	2.77
Dec 01	154	7.35	7.05	975	1180	395.00	90.00	77.22	7.92	19.09	1.68
02	155	7.30	7.20	825	1160	463.00	102.00	77.97	7.92	19.09	1.71

RESULTS FOR ANAEROBIC FILTER (pH VARIATION)

Date 1989	Day	pH value infl	pH value eff	alkalinity infl.	alkalinity eff.	T O C mg/l infl.	T O C mg/l fil.eff	% rem	Q (l/d)	HRT (hrs)	VOLR Kg COD /m3.d
Sept 27	89	7.05	6.75	550	860	379.00	147.00	61.21	8.41	17.98	2.19
29	91	6.75	6.65	600	760	264.00	97.00	63.26	8.34	18.13	1.94
Oct 01	93	7.05	6.80	650	780	213.00	82.00	61.50	7.86	19.24	1.54
02	94	6.30	6.70	360	760	296.00	121.00	59.12	8.06	18.76	2.10
04	96	6.50	6.90	480	670	408.00	150.00	63.24	7.75	19.51	1.65
05	97	6.50	6.70	410	640	239.00	93.00	61.09	8.11	18.64	1.73
07	99	5.95	6.90	225	590	248.00	113.00	54.44	7.04	21.48	1.40
08	100	5.85	6.95	200	520	238.00	113.00	52.52	8.10	18.67	1.64
10	102	5.20	6.55	115	410	261.00	136.00	47.89	8.22	18.39	1.89
12	104	5.35	6.70	120	420	288.00	141.00	51.04	8.00	18.90	1.76
14	106	5.05	6.60	50	330	246.00	135.00	45.12	7.84	19.29	1.65
16	108	4.50	6.50	0	270	301.00	156.00	48.17	7.70	19.64	1.84
18	110	3.90	6.45	0	300	256.00	158.00	38.28	7.54	20.05	1.57
20	112	4.00	6.60	0	255	228.00	143.00	37.28	7.86	19.24	1.37
21	113	4.40	6.55	0	210	267.00	138.00	48.31	7.92	19.09	1.79
23	115	5.10	6.80	0	240	240.00	125.00	47.92	7.58	19.95	1.53
25	117	5.50	6.75	110	490	309.00	129.00	58.25	7.62	19.84	1.77
27	119	5.95	6.80	230	435	284.00	119.00	58.10	7.49	20.19	1.58
29	121	6.05	6.60	235	470	247.00	125.50	49.19	7.78	19.43	1.42
30	122	6.70	6.50	210	330	222.00	105.00	52.70	7.85	19.26	1.58
Nov 01	124	6.75	7.00	650	860	437.00	188.00	56.98	8.61	17.56	2.30
03	126	6.90	6.90	1005	1090	387.00	163.00	57.88	8.11	18.64	1.83
05	128	7.45	7.30	1490	1700	441.00	165.00	62.59	9.40	16.09	2.41
06	129	7.55	7.25	1585	1740	452.00	201.00	55.53	8.00	18.90	2.02
08	131	7.45	7.05	1585	1740	334.00	153.00	54.19	8.22	18.39	1.88
10	133	7.70	7.20	1600	1680	282.00	137.00	51.42	7.97	18.97	2.03
13	136	7.85	7.60	2060	1960	308.00	160.00	48.05	7.85	19.26	1.64
15	138	8.10	7.45	2095	2170	228.00	120.00	47.37	9.59	15.77	1.90
17	140	8.50	7.65	2135	1985	295.00	165.00	44.07	9.14	16.54	2.30
19	142	8.80	7.80	2220	2020	325.00	186.00	42.77	8.86	17.07	2.07
21	144	8.45	7.70	2210	2095	298.00	155.00	47.99	9.32	16.22	2.07
22	145	8.20	7.50	2100	1920	268.00	137.00	48.88	9.00	16.80	2.03
24	147	7.75	7.20	1290	1405	227.00	108.00	52.42	4.50	33.60	0.92
26	149	7.55	7.10	975	1085	226.00	112.00	50.44	8.43	17.94	1.52
27	150	7.50	7.05	1055	1210	270.00	130.00	51.85	8.76	17.26	1.69
28	151	7.30	7.45	1410	1530	335.00	123.00	63.28	9.70	15.59	2.77
Dec 01	154	7.35	7.05	975	1180	286.00	127.00	55.59	7.92	19.09	1.68
02	155	7.30	7.20	825	1160	344.00	132.00	61.63	7.92	19.09	1.71

RESULTS FOR RBC STEADY STATE PERFORMANCE (LS LEACHATE)

Date	Day			C O D mg/l			Q	HRT	VOLR	AOLR
1990		infl.	set.eff	% rem	fil.eff	% rem	(l/d)	(hrs)	Kg COD/m3.d	g COD/m2.d
Dec 05	116	1255.00	340.00	72.91	294.00	76.57	14.00	5.66	5.32	20.72
07	118	1196.00	257.00	78.51	238.00	80.10	10.80	7.33	3.91	15.23
09	120	1191.00	271.00	77.25	254.00	78.67	12.80	6.19	4.62	17.98
12	123	1438.00	336.00	76.63	301.00	79.07	11.75	6.74	5.12	19.93
14	125	1345.00	298.00	77.84	250.00	81.41	11.00	7.20	4.48	17.45
16	127	1224.00	282.00	76.96	241.00	80.31	11.08	7.15	4.11	15.99
17	128	1220.00	293.00	75.98	247.00	79.75	10.80	7.33	3.99	15.54
19	130	1092.00	324.00	70.33	258.00	76.37	12.50	6.34	4.14	16.10
21	132	1444.00	290.00	79.92	264.00	81.72	11.12	7.12	4.87	18.94
23	134	1281.00	243.00	81.03	228.00	82.20	10.19	7.77	3.96	15.39
25	136	1122.00	287.00	74.42	236.00	78.97	13.50	5.87	4.59	17.86
27	138	1047.00	275.00	73.73	234.00	77.65	12.46	6.36	3.95	15.38
29	140	1428.00	256.00	82.07	251.00	82.42	10.00	7.92	4.33	16.84
31	142	1361.00	271.00	80.09	227.00	83.32	10.46	7.57	4.31	16.79
Jan2/90	144	1271.00	237.00	81.35	185.00	85.44	10.30	7.69	3.97	15.44
04	146	1129.00	221.00	80.43	179.00	84.15	11.70	6.77	4.00	15.58
06	148	1470.00	296.00	79.86	258.00	82.45	10.81	7.33	4.82	18.74
08	150	1350.00	260.00	80.74	245.00	81.85	10.23	7.74	4.19	16.29
10	152	1260.00	372.00	70.48	309.00	75.48	14.50	5.46	5.54	21.54
12	154	1080.00	286.00	73.52	238.00	77.96	15.77	5.02	5.16	20.08
13	155	1417.00	345.00	75.65	306.00	78.41	11.28	7.02	4.84	18.85
16	158	1251.00	268.00	78.58	212.00	83.05	14.95	5.30	5.67	22.05
18	160	1172.00	225.00	80.80	192.00	83.62	15.80	5.01	5.61	21.84
20	162	1525.00	286.00	81.25	264.00	82.69	11.93	6.64	5.51	21.45
22	164	1452.00	359.00	75.28	322.00	77.82	12.35	6.41	5.43	21.15
24	166	1306.00	256.00	80.40	220.00	83.15	11.96	6.62	4.73	18.42
26	168	1750.00	379.00	78.34	345.00	80.29	13.42	5.90	7.12	27.69
28	170	1638.00	303.00	81.50	287.00	82.48	13.75	5.76	6.83	26.56
30	172	1506.00	380.00	74.77	348.00	76.89	15.35	5.16	7.01	27.26
31	173	1894.00	258.00	86.38	244.00	87.12	12.80	6.19	7.35	28.59
Feb 02	175	1604.00	252.00	84.29	240.00	85.04	13.62	5.81	6.62	25.76
05	178	1980.00	280.00	85.86	258.00	86.97	11.10	7.14	6.66	25.92
06	179	1936.00	297.00	84.66	280.00	85.54	12.75	6.21	7.48	29.11
08	181	1774.00	253.00	85.74	220.00	87.60	12.50	6.34	6.72	26.15
10	183	1584.00	249.00	84.28	226.00	85.73	13.54	5.85	6.50	25.29
13	186	2060.00	369.00	82.09	325.00	84.22	11.20	7.07	6.99	27.21
14	187	2016.00	414.00	79.46	356.00	82.34	10.80	7.33	6.60	25.68
16	189	1849.00	329.00	82.21	277.00	85.02	11.86	6.68	6.65	25.86
18	191	1632.00	323.00	80.21	264.00	83.82	13.25	5.98	6.55	25.50
20	193	1933.00	286.00	85.20	252.00	86.96	12.54	6.32	7.35	28.58
22	195	1807.00	307.00	83.01	279.00	84.56	12.30	6.44	6.74	26.21
24	197	1712.00	310.00	81.89	285.00	83.35	13.06	6.06	6.78	26.37
26	199	1578.00	253.00	83.97	214.00	86.44	13.45	5.89	6.43	25.03
28	201	2095.00	286.00	86.35	242.00	88.45	12.70	6.24	8.06	31.38
Mac 02	203	1914.00	292.00	84.74	234.00	87.77	13.00	6.09	7.54	29.34
04	205	1705.00	331.00	80.59	288.00	83.11	14.42	5.49	7.45	28.99
06	207	1629.00	328.00	79.86	294.00	81.95	13.85	5.72	6.84	26.61
08	209	2030.00	329.00	83.79	291.00	85.67	12.20	6.49	7.50	29.21
09	210	1995.00	372.00	81.35	311.00	84.41	12.94	6.12	7.82	30.44
11	212	1763.00	379.00	78.50	322.00	81.74	13.40	5.91	7.16	27.86

13	214	1544.00	340.00	77.98	223.00	85.56	14.70	5.39	6.88	26.77
15	216	1512.00	475.00	68.58	368.00	75.66	15.82	5.01	7.25	28.21
18	219	1924.00	396.00	79.42	335.00	82.59	13.47	5.88	7.85	30.56
20	221	1795.00	435.00	75.77	363.00	79.78	14.00	5.66	7.62	29.63
22	223	1644.00	300.00	81.75	244.00	85.16	15.40	5.14	7.67	29.86
24	225	1565.00	338.00	78.40	272.00	82.62	15.20	5.21	7.21	28.05
25	226	1608.00	476.00	70.40	578.00	64.05	14.93	5.30	7.27	28.31
26	227	1536.00	392.00	74.48	359.00	76.63	14.50	5.46	6.75	26.26
27	228	1957.00	409.00	79.10	355.00	81.86	16.80	4.71	9.96	38.77
29	230	1745.00	351.00	79.89	282.00	83.84	17.74	4.46	9.38	36.51
31	232	1593.00	420.00	73.63	367.00	76.96	18.50	4.28	8.93	34.75
Apr 02	234	1847.00	444.00	75.96	377.00	79.59	16.90	4.69	9.46	36.81
04	236	1714.00	327.00	80.92	284.00	83.43	17.48	4.53	9.08	35.33
05	237	1680.00	340.00	79.76	291.00	82.68	18.45	4.29	9.39	36.55
07	239	1563.00	478.00	69.42	369.00	76.39	18.00	4.40	8.53	33.18
09	241	2050.00	376.00	81.66	363.00	82.29	15.92	4.97	9.89	38.49
12	244	1792.00	281.00	84.32	248.00	86.16	16.30	4.86	8.85	34.45
14	246	1572.00	496.00	68.45	400.00	74.55	16.54	4.79	7.88	30.66
16	248	2067.00	402.00	80.55	372.00	82.00	15.64	5.06	9.80	38.12
18	250	1832.00	428.00	76.64	370.00	79.80	16.75	4.73	9.30	36.19
20	252	1730.00	440.00	74.57	366.00	78.84	16.84	4.70	8.83	34.36
22	254	1575.00	512.00	67.49	349.00	77.84	17.52	4.52	8.36	32.54
24	256	2170.00	489.00	77.47	401.00	81.52	15.00	5.28	9.86	38.38
26	258	1904.00	476.00	75.00	384.00	79.83	16.68	4.75	9.62	37.45
28	260	1731.00	367.00	78.80	342.00	80.24	16.00	4.95	8.39	32.66
30	262	1564.00	328.00	79.03	255.00	83.70	16.80	4.71	7.96	30.98
May 02	264	1990.00	447.00	77.54	364.00	81.71	15.78	5.02	9.52	37.03
04	266	1865.00	456.00	75.55	366.00	80.38	15.12	5.24	8.55	33.25
07	269	1512.00	422.00	72.09	345.00	77.18	17.20	4.60	7.88	30.67
08	270	1856.00	300.00	83.84	242.00	86.96	16.10	4.92	9.06	35.24
10	272	1696.00	429.00	74.71	333.00	80.37	16.60	4.77	8.53	33.20
12	274	2055.00	478.00	76.74	422.00	79.46	15.28	5.18	9.52	37.03
14	276	1991.00	480.00	75.89	393.00	80.26	15.00	5.28	9.05	35.22
16	278	1865.00	504.00	72.98	422.00	77.37	15.75	5.03	8.90	34.64
18	280	1724.00	319.00	81.50	248.00	85.61	17.51	4.52	9.15	35.60
20	282	1665.00	495.00	70.27	450.00	72.97	21.90	3.62	11.05	43.00
22	284	1948.00	439.00	77.46	383.00	80.34	19.85	3.99	11.72	45.60
24	286	1865.00	565.00	69.71	462.00	75.23	19.54	4.05	11.04	42.97
26	288	1695.00	572.00	66.25	470.00	72.27	20.80	3.81	10.68	41.58
28	290	2011.00	576.00	71.36	503.00	74.99	20.10	3.94	12.25	47.67
30	292	1840.00	531.00	71.14	478.00	74.02	20.45	3.87	11.40	44.37
Jun 01	294	1680.00	379.00	77.44	307.00	81.73	22.50	3.52	11.45	44.58
02	295	1724.00	503.00	70.82	409.00	76.28	22.00	3.60	11.49	44.73
04	297	1660.00	463.00	72.11	402.00	75.78	22.88	3.46	11.51	44.79
06	299	1468.00	389.00	73.50	321.00	78.13	24.40	3.25	10.85	42.24
08	301	1919.00	498.00	74.05	452.00	76.45	22.20	3.57	12.91	50.24
10	303	1748.00	558.00	68.08	473.00	72.94	21.90	3.62	11.60	45.14
12	305	1698.00	549.00	67.67	493.00	70.97	22.15	3.58	11.40	44.35
14	307	2020.00	654.00	67.62	575.00	71.53	20.84	3.80	12.76	49.64
16	309	1942.00	690.00	64.47	616.00	68.28	21.43	3.70	12.61	49.08
18	311	1872.00	582.00	68.91	506.00	72.97	21.90	3.62	12.42	48.35
20	313	1795.00	628.00	65.01	581.00	67.63	22.00	3.60	11.97	46.57
22	315	2007.00	656.00	67.31	590.00	70.60	21.92	3.61	13.33	51.88
24	317	1846.00	635.00	65.60	597.00	67.66	22.55	3.51	12.61	49.09
26	319	1652.00	632.00	61.74	586.00	64.53	22.70	3.49	11.36	44.22
28	321	1985.00	674.00	66.05	600.00	69.77	21.90	3.62	13.17	51.26
30	323	1827.00	650.00	64.42	587.00	67.87	24.00	3.30	13.29	51.71

July 02	325	1702.00	594.00	65.10	549.00	67.74	24.40	3.25	12.58	48.97
05	328	1884.00	627.00	66.72	570.00	69.75	22.76	3.48	12.99	50.57
07	330	1730.00	584.00	66.24	512.00	70.40	23.00	3.44	12.06	46.92
09	332	1608.00	582.00	63.81	523.00	67.48	22.68	3.49	11.05	43.01
12	335	1672.00	430.00	74.28	379.00	77.33	13.30	5.95	6.74	26.22
14	337	1650.00	433.00	73.76	398.00	75.88	13.20	6.00	6.60	25.68
16	339	1620.00	382.00	76.42	343.00	78.83	13.44	5.89	6.60	25.68
18	341	1665.00	424.00	74.53	380.00	77.18	14.40	5.50	7.27	28.27
		1281.81	285.92	77.54	248.19	80.56	12.08	6.68	4.66	18.14
		1786.31	327.76	81.47	283.28	84.01	13.21	6.04	7.10	27.64
		1801.48	414.78	76.80	344.81	80.77	16.60	4.79	9.02	35.11
		1799.65	565.58	68.57	499.81	72.24	22.03	3.61	11.98	46.64
		1651.75	417.25	74.75	375.00	77.30	13.59	5.84	6.80	26.46

RESULTS FOR RBC STEADY STATE PERFORMANCE (LS LEACHATE)

		B O D mg/l			Q	HRT	VOLR	AOLR
Date 1989		infl.	fil.eff	% rem	(l/d)	(hrs)	Kg BOD /m3.d	g BOD/ m2.d
Dec 05	116	387.00	44.50	88.50	14.00	5.66	1.64	6.39
07	118	408.00	35.30	91.35	10.80	7.33	1.34	5.20
09	120	388.00	47.00	87.89	12.80	6.19	1.50	5.86
12	123	399.00	46.00	88.47	11.75	6.74	1.42	5.53
14	125	376.50	51.00	86.45	11.00	7.20	1.26	4.88
16	127	352.50	35.00	90.07	11.08	7.15	1.18	4.61
19	130	351.00	49.00	86.04	12.50	6.34	1.33	5.17
21	132	405.00	47.50	88.27	11.12	7.12	1.36	5.31
23	134	400.00	37.00	90.75	10.19	7.77	1.24	4.81
25	136	349.00	39.00	88.83	13.50	5.87	1.43	5.56
27	138	324.00	40.00	87.65	12.46	6.36	1.22	4.76
31	142	392.00	38.30	90.23	10.46	7.57	1.24	4.84
Jan2/90	144	388.00	51.00	86.86	10.30	7.69	1.21	4.71
04	146	363.00	52.30	85.59	11.70	6.77	1.29	5.01
06	148	445.00	49.00	88.99	10.81	7.33	1.46	5.67
08	150	392.00	43.00	89.03	10.23	7.74	1.22	4.73
10	152	350.00	32.50	90.71	14.50	5.46	1.54	5.98
12	154	343.30	37.00	89.22	15.77	5.02	1.64	6.38
13	155	404.00	39.50	90.22	11.28	7.02	1.38	5.37
16	158	359.00	48.00	86.63	14.95	5.30	1.63	6.33
18	160	331.00	43.00	87.01	15.80	5.01	1.58	6.17
20	162	426.00	47.00	88.97	11.93	6.64	1.54	5.99
24	166	467.00	40.00	91.43	11.96	6.62	1.69	6.59
26	168	558.00	67.30	87.94	13.42	5.90	2.27	8.83
28	170	481.00	52.00	89.19	13.75	5.76	2.00	7.80
30	172	475.00	66.00	86.11	15.35	5.16	2.21	8.60
31	173	540.00	36.50	93.24	12.80	6.19	2.09	8.15
Feb 02	175	462.00	58.00	87.45	13.62	5.81	1.91	7.42
06	179	561.00	45.00	91.98	12.75	6.21	2.17	8.43
08	181	526.00	65.00	87.64	12.50	6.34	1.99	7.75
10	183	484.00	53.00	89.05	13.54	5.85	1.99	7.73
13	186	630.00	55.00	91.27	11.20	7.07	2.14	8.32
14	187	687.50	62.00	90.98	10.80	7.33	2.25	8.76
16	189	608.00	42.50	93.01	11.86	6.68	2.19	8.50
20	193	666.00	59.50	91.07	12.54	6.32	2.53	9.85
22	195	622.00	44.00	92.93	12.30	6.44	2.32	9.02
24	197	615.00	57.00	90.73	13.06	6.06	2.43	9.47
28	201	710.00	83.80	88.20	12.70	6.24	2.73	10.63
Mac 02	203	640.30	64.00	90.00	13.00	6.09	2.52	9.82
04	205	530.00	45.00	91.51	14.42	5.49	2.32	9.01
06	207	474.00	51.30	89.18	13.85	5.72	1.99	7.74
08	209	699.00	63.50	90.92	12.20	6.49	2.58	10.06
09	210	583.00	59.00	89.88	12.94	6.12	2.29	8.90
11	212	539.00	69.00	87.20	13.40	5.91	2.19	8.52
13	214	451.00	48.00	89.36	14.70	5.39	2.01	7.82
18	219	555.00	62.50	88.74	13.47	5.88	2.27	8.82
20	221	548.00	48.50	91.15	14.00	5.66	2.32	9.05
22	223	499.00	67.00	86.57	15.40	5.14	2.33	9.06
24	225	546.00	74.00	86.45	15.20	5.21	2.51	9.79
25	226	529.00	142.00	73.16	14.93	5.30	2.39	9.31
26	227	494.00	86.00	82.59	14.50	5.46	2.17	8.45
27	228	664.00	55.00	91.72	16.80	4.71	3.38	13.15
29	230	700.00	77.00	89.00	17.74	4.46	3.76	14.64

31	232	614.00	85.00	86.16	18.50	4.28	3.44	13.40
Apr 02	234	599.00	67.00	88.81	16.90	4.69	3.07	11.94
04	236	760.00	83.00	89.08	17.48	4.53	4.03	15.67
05	237	740.00	85.00	88.51	18.45	4.29	4.14	16.10
07	239	625.00	66.00	89.44	17.50	4.53	3.31	12.90
09	241	654.00	54.00	91.74	15.92	4.97	3.16	12.28
12	244	840.00	82.00	90.24	16.30	4.86	4.15	16.15
16	248	760.00	89.00	88.29	15.64	5.06	3.60	14.02
18	250	721.00	93.00	87.10	16.75	4.73	3.66	14.24
20	252	650.00	92.00	85.85	16.84	4.70	3.32	12.91
22	254	645.00	109.00	83.10	17.52	4.52	3.42	13.33
24	256	835.00	82.50	90.12	15.00	5.28	3.80	14.77
26	258	605.00	91.00	84.96	16.68	4.75	3.06	11.90
28	260	755.00	85.30	88.70	16.00	4.95	3.66	14.25
30	262	760.00	117.00	84.61	16.80	4.71	3.87	15.06
May 02	264	787.00	114.00	85.51	15.78	5.02	3.76	14.64
04	266	806.30	160.00	80.16	15.12	5.24	3.69	14.38
07	269	693.00	131.00	81.10	17.20	4.60	3.61	14.06
10	272	711.00	126.00	82.28	16.60	4.77	3.58	13.92
12	274	845.00	132.00	84.38	15.28	5.18	3.91	15.23
14	276	789.00	116.00	85.30	15.00	5.28	3.59	13.96
16	278	765.00	123.00	83.92	15.75	5.03	3.65	14.21
18	280	704.00	108.00	84.66	17.51	4.52	3.74	14.54
20	282	637.00	113.00	82.26	21.90	3.62	4.23	16.45
22	284	689.00	110.00	84.03	19.85	3.99	4.14	16.13
24	286	887.50	162.00	81.75	19.54	4.05	5.26	20.45
26	288	684.00	126.50	81.51	20.80	3.81	4.31	16.78
28	290	679.00	108.00	84.09	20.10	3.94	4.14	16.09
30	292	640.00	105.00	83.59	20.45	3.87	3.97	15.43
Jun 02	295	666.00	128.00	80.78	22.00	3.60	4.44	17.28
04	297	860.00	207.00	75.93	22.88	3.46	5.96	23.20
08	301	805.00	190.00	76.40	22.20	3.57	5.42	21.07
10	303	776.00	177.00	77.19	21.90	3.62	5.15	20.04
12	305	656.00	119.00	81.86	22.15	3.58	4.40	17.13
14	307	840.00	200.00	76.19	20.84	3.80	5.30	20.64
16	309	797.00	202.00	74.65	21.43	3.70	5.18	20.14
18	311	833.00	199.00	76.11	21.90	3.62	5.53	21.51
20	313	754.00	196.00	74.01	22.00	3.60	5.03	19.56
22	315	762.00	211.00	72.31	21.92	3.61	5.06	19.70
24	317	825.00	213.00	74.18	22.55	3.51	5.64	21.94
26	319	724.00	202.00	72.10	22.70	3.49	4.98	19.38
28	321	815.00	228.00	72.02	21.90	3.62	5.41	21.05
30	323	762.00	195.00	74.41	24.00	3.30	5.54	21.57
July 02	325	724.00	168.30	76.75	24.40	3.25	5.35	20.83
05	328	810.00	222.00	72.59	22.76	3.48	5.59	21.74
07	330	757.00	200.00	73.58	23.00	3.44	5.28	20.53
09	332	582.00	139.00	76.12	22.68	3.49	4.00	15.57
12	335	538.00	68.00	87.36	13.30	5.95	2.17	8 44
14	337	595.00	42.00	92.94	13.20	6.00	2.38	9.26
18	341	572.00	51.00	91.08	14.40	5.50	2.50	9.71

382.62	43.13	88.66	12.21	6.60	1.41	5.47
565.75	56.98	89.81	13.18	6.05	2.24	8.72
721.09	96.91	86.59	16.60	4.79	3.61	14.06
748.52	171.70	77.27	21.91	3.63	4.97	19.34
568.33	53.67	90.46	13.63	5.82	2.35	9.14

RESULTS FOR RBC STEADY STATE PERFORMANCE (LS LEACHATE)

Date 1989		T O C mg/l infl.		% rem	Q (l/d)	HRT (hrs)	VOLR Kg TOC /m3.d	AOLR g TOC/ m2.d
		infl.	fil.eff	% rem				
Dec 05	116	238.00	75.00	68.49	14.00	5.66	1.01	3.93
07	118	339.00	82.00	75.81	10.80	7.33	1.11	4.32
09	120	272.00	65.00	76.10	12.80	6.19	1.06	4.11
12	123	345.00	86.00	75.07	11.75	6.74	1.23	4.78
14	125	293.00	66.00	77.47	11.00	7.20	0.98	3.80
16	127	307.00	67.00	78.18	11.08	7.15	1.03	4.01
17	128	280.00	65.00	76.79	10.80	7.33	0.92	3.57
19	130	283.00	52.00	81.63	12.50	6.34	1.07	4.17
21	132	381.00	82.00	78.48	11.12	7.12	1.28	5.00
23	134	292.00	59.00	79.79	10.19	7.77	0.90	3.51
25	136	249.00	60.00	75.90	13.50	5.87	1.02	3.96
27	138	235.00	54.00	77.02	12.46	6.36	0.89	3.45
29	140	298.00	58.00	80.54	10.00	7.92	0.90	3.51
31	142	293.00	62.00	78.84	10.46	7.57	0.93	3.61
Jan2/90	144	283.00	61.00	78.45	10.30	7.69	0.88	3.44
04	146	271.00	63.00	76.75	11.70	6.77	0.96	3.74
06	148	339.00	65.00	80.83	10.81	7.33	1.11	4.32
08	150	288.00	62.00	78.47	10.23	7.74	0.89	3.47
10	152	255.00	67.00	73.73	14.50	5.46	1.12	4.36
12	154	233.00	64.00	72.53	15.77	5.02	1.11	4.33
13	155	300.00	73.00	75.67	11.28	7.02	1.03	3.99
16	158	284.00	66.00	76.76	14.95	5.30	1.29	5.01
18	160	251.00	65.00	74.10	15.80	5.01	1.20	4.68
20	162	328.00	70.00	78.66	11.93	6.64	1.19	4.61
22	164	328.00	88.00	73.17	12.35	6.41	1.23	4.78
24	166	273.00	65.00	76.19	11.96	6.62	0.99	3.85
26	168	415.00	85.00	79.52	13.42	5.90	1.69	6.57
28	170	363.00	64.00	82.37	13.75	5.76	1.51	5.89
30	172	385.00	91.00	76.36	15.35	5.16	1.79	6.97
31	173	392.00	69.00	82.40	12.80	6.19	1.52	5.92
Feb 02	175	328.00	70.00	78.66	13.62	5.81	1.35	5.27
06	179	405.00	80.00	80.25	12.75	6.21	1.56	6.09
08	181	376.00	81.00	78.46	12.50	6.34	1.42	5.54
10	183	362.00	72.00	80.11	13.54	5.85	1.49	5.78
13	186	459.00	71.00	84.53	11.20	7.07	1.56	6.06
14	187	467.00	74.00	84.15	11.20	7.07	1.58	6.17
16	189	384.00	78.00	79.69	11.86	6.68	1.38	5.37
18	191	397.00	53.00	86.65	13.25	5.98	1.59	6.20
20	193	395.00	77.00	80.51	12.54	6.32	1.50	5.84
22	195	402.00	66.00	83.58	12.30	6.44	1.50	5.83
26	199	331.00	74.00	77.64	13.45	5.89	1.35	5.25
28	201	493.00	83.00	83.16	12.70	6.24	1.90	7.38
Mac 02	203	399.00	80.00	79.95	13.00	6.09	1.57	6.12
04	205	362.00	75.00	79.28	14.42	5.49	1.58	6.16
06	207	354.00	77.00	78.25	13.85	5.72	1.49	5.78
08	209	493.00	84.00	82.96	12.20	6.49	1.82	7.09
09	210	460.00	115.00	75.00	12.94	6.12	1.80	7.02
11	212	412.00	110.00	73.30	13.40	5.91	1.67	6.51
13	214	328.00	77.00	76.52	14.70	5.39	1.46	5.69

15	216	351.00	77.00	78.06	15.82	5.01	1.68	6.55
18	219	465.00	110.00	76.34	13.47	5.88	1.90	7.39
20	221	383.00	76.00	80.16	14.00	5.66	1.62	6.32
22	223	375.00	70.00	81.33	15.40	5.14	1.75	6.81
24	225	349.00	78.00	77.65	15.20	5.21	1.61	6.26
25	226	347.00	133.00	61.67	14.93	5.30	1.57	6.11
26	227	372.00	97.00	73.92	14.50	5.46	1.63	6.36
27	228	439.00	95.00	78.36	16.80	4.71	2.23	8.70
29	230	385.00	80.00	79.22	17.74	4.46	2.07	8.05
31	232	400.00	116.00	71.00	18.50	4.28	2.24	8.73
Apr 02	234	389.00	85.00	78.15	16.90	4.69	1.99	7.75
04	236	382.00	94.00	75.39	17.48	4.53	2.02	7.87
05	237	363.00	78.00	78.51	18.45	4.29	2.03	7.90
07	239	412.00	131.00	68.20	17.50	4.53	2.18	8.50
09	241	494.00	85.00	82.79	15.92	4.97	2.38	9.27
12	244	425.00	102.00	76.00	16.30	4.86	2.10	8.17
14	246	418.00	128.00	69.38	16.54	4.79	2.10	8.15
16	248	439.00	115.00	73.80	15.64	5.06	2.08	8.10
20	252	432.00	109.00	74.77	16.84	4.70	2.20	8.58
22	254	386.00	119.00	69.17	17.52	4.52	2.05	7.97
24	256	460.00	98.00	78.70	15.00	5.28	2.09	8.14
26	258	439.00	112.00	74.49	16.68	4.75	2.22	8.64
28	260	445.00	102.00	77.08	16.00	4.95	2.16	8.40
30	262	396.00	101.00	74.49	16.80	4.71	2.02	7.85
May 02	264	415.00	117.00	71.81	15.78	5.02	1.98	7.72
04	266	435.00	133.00	69.43	15.12	5.24	1.99	7.76
07	269	373.00	111.00	70.24	17.20	4.60	1.94	7.57
08	270	480.00	119.00	75.21	16.10	4.92	2.34	9.11
10	272	424.00	114.00	73.11	16.60	4.77	2.13	8.30
12	274	507.00	151.00	70.22	15.28	5.18	2.35	9.14
14	276	489.00	144.00	70.55	15.00	5.28	2.22	8.65
16	278	427.00	118.00	72.37	15.75	5.03	2.04	7.93
18	280	406.00	104.00	74.38	17.51	4.52	2.15	8.38
20	282	343.00	112.00	67.35	21.90	3.62	2.28	8.86
22	284	414.00	115.00	72.22	19.85	3.99	2.49	9.69
24	286	451.00	123.00	72.73	19.54	4.05	2.67	10.39
26	288	436.00	127.00	70.87	20.80	3.81	2.75	10.69
28	290	460.00	114.00	75.22	20.10	3.94	2.80	10.90
30	292	468.00	112.00	76.07	20.45	3.87	2.90	11.29
Jun 01	294	462.00	129.00	72.08	22.50	3.52	3.15	12.26
02	295	429.00	113.00	73.66	22.00	3.60	2.86	11.13
04	297	416.00	127.00	69.47	22.88	3.46	2.88	11.22
06	299	403.00	117.00	70.97	24.40	3.25	2.98	11.60
08	301	392.00	118.00	69.90	22.20	3.57	2.64	10.26
10	303	442.00	136.00	69.23	21.90	3.62	2.93	11.41
12	305	382.00	130.00	65.97	22.15	3.58	2.56	9.98
14	307	451.00	161.00	64.30	20.84	3.80	2.85	11.08
16	309	409.00	142.00	65.28	21.43	3.70	2.66	10.34
18	311	386.00	121.00	68.65	21.90	3.62	2.56	9.97
22	315	405.00	145.00	64.20	21.92	3.61	2.69	10.47
24	317	438.00	139.00	68.26	22.55	3.51	2.99	11.65
26	319	417.00	140.00	66.43	22.20	3.57	2.81	10.92
28	321	413.00	143.00	65.38	21.90	3.62	2.74	10.67
30	323	394.00	150.00	61.93	24.00	3.30	2.87	11.15
July 02	325	376.00	125.00	66.76	24.40	3.25	2.78	10.82
05	328	400.00	128.00	68.00	22.76	3.48	2.76	10.74

07	330	380.00	140.00	63.16	23.00	3.44	2.65	10.31
09	332	426.00	117.00	72.54	22.68	3.49	2.93	11.39
12	335	342.00	99.00	71.05	13.30	5.95	1.38	5.36
14	337	365.00	100.00	72.60	13.20	6.00	1.46	5.68
16	339	329.00	67.00	79.64	13.44	5.89	1.34	5.21
18	341	418.00	86.00	79.43	14.40	5.50	1.82	7.10

289.92	67.00	76.75	12.08	6.68	1.05	4.09
397.63	79.22	79.97	13.31	5.99	1.59	6.21
422.04	110.29	73.80	16.88	4.72	2.15	8.35
418.96	130.30	68.74	22.11	3.59	2.80	10.90
363.50	88.00	75.68	13.59	5.84	1.50	5.84

RESULTS FOR UAF STEADY STATE PERFORMANCE (LS LEACHATE)

		C O Dmg/l					biogas	CH4	CO2	Q	HRT	VOLR	AOLR	EMP m3
Date	Day	infl.	set.eff	% rem	fil.eff	% rem	(l/d)	(%)	(%)	(l/d)	(hrs)	Kg COD	g COD	CH4/Kg
1989												/m3.d	/m2.d	CODred
Dec 05	158	1255.00	289.00	76.97	258.00	79.44	1.85	89.10	9.90	5.61	26.95	1.12	6.40	0.295
07	160	1196.00	257.00	78.51	219.00	81.69	1.60	88.40	11.60	4.50	33.60	0.85	4.89	0.322
09	162	1191.00	231.00	80.60	195.00	83.63	1.30	90.20	9.80	5.00	30.24	0.95	5.41	0.235
12	165	1438.00	262.00	81.78	228.00	84.14	1.50	89.50	10.50	4.77	31.70	1.09	6.24	0.233
14	167	1345.00	273.00	79.70	235.00	82.53	1.65	90.40	9.60	4.86	31.11	1.04	5.94	0.276
16	169	1224.00	231.00	81.13	189.00	84.56	1.60	91.30	8.70	5.14	29.42	1.00	5.72	0.275
17	170	1220.00	252.00	79.34	213.00	82.54	1.35	89.70	10.30	4.47	33.83	0.87	4.96	0.269
19	172	1092.00	239.00	78.11	190.00	82.60	1.80	88.30	11.70	6.17	24.51	1.07	6.13	0.286
21	174	1444.00	247.00	82.89	214.00	85.18	1.95	90.40	9.60	5.25	28.80	1.20	6.89	0.273
23	176	1281.00	243.00	81.03	201.00	84.31	1.50	91.80	8.20	4.22	35.83	0.86	4.91	0.302
25	178	1122.00	211.00	81.19	164.00	85.38	1.80	91.40	8.60	6.24	24.23	1.11	6.36	0.275
27	180	1047.00	165.00	84.24	131.00	87.49	1.90	90.40	9.60	6.43	23.51	1.07	6.12	0.292
29	182	1428.00	254.00	82.21	219.00	84.66	2.00	87.70	12.30	4.87	31.05	1.10	6.32	0.298
31	184	1361.00	249.00	81.70	202.00	85.16	1.80	88.40	11.60	4.32	35.00	0.93	5.35	0.318
Jan2/90	186	1271.00	241.00	81.04	205.00	83.87	1.95	91.30	8.70	5.60	27.00	1.13	6.47	0.298
04	188	1129.00	183.00	83.79	156.00	86.18	2.00	90.70	9.30	6.20	24.39	1.11	6.36	0.301
06	190	1470.00	228.00	84.49	185.00	87.41	2.20	91.60	8.40	5.42	27.90	1.26	7.24	0.289
08	192	1350.00	248.00	81.63	199.00	85.26	2.10	92.70	7.30	5.35	28.26	1.15	6.57	0.316
10	194	1260.00	257.00	79.60	212.00	83.17	2.00	91.00	9.00	6.30	24.00	1.26	7.22	0.276
12	196	1080.00	204.00	81.11	172.00	84.07	1.85	90.60	9.40	6.50	23.26	1.11	6.38	0.284
13	197	1417.00	229.00	83.84	188.00	86.73	2.05	91.30	8.70	4.99	30.30	1.12	6.43	0.305
16	200	1251.00	197.00	84.25	160.00	87.21	2.00	91.80	8.20	5.96	25.37	1.18	6.78	0.282
18	202	1172.00	233.00	80.12	182.00	84.47	2.10	92.00	8.00	5.72	26.43	1.06	6.09	0.341
20	204	1525.00	222.00	85.44	196.00	87.15	2.00	89.80	10.20	4.25	35.58	1.03	5.89	0.318
22	206	1452.00	247.00	82.99	211.00	85.47	2.15	90.50	9.50	5.05	29.94	1.16	6.67	0.310
24	208	1306.00	205.00	84.30	174.00	86.68	2.05	91.10	8.90	5.34	28.31	1.11	6.34	0.309
26	210	1750.00	426.00	75.66	385.00	78.00	3.15	89.90	10.10	7.47	20.24	2.08	11.88	0.278
28	212	1638.00	343.00	79.06	309.00	81.14	3.50	91.00	9.00	8.20	18.44	2.13	12.21	0.292
30	214	1506.00	307.00	79.61	255.00	83.07	3.00	92.20	7.80	7.96	18.99	1.90	10.90	0.278
31	215	1894.00	342.00	81.94	330.00	82.58	3.35	92.30	7.70	6.94	21.79	2.09	11.95	0.285
Feb 02	217	1604.00	375.00	76.62	348.00	78.30	3.20	89.10	10.90	7.70	19.64	1.96	11.23	0.295
05	220	1980.00	368.00	81.41	345.00	82.58	3.60	87.40	12.60	6.15	24.59	1.93	11.07	0.313
06	221	1936.00	390.00	79.86	370.00	80.89	3.00	87.60	12.40	6.07	24.91	1.87	10.68	0.276
08	223	1774.00	342.00	80.72	299.00	83.15	3.00	88.70	11.30	7.71	19.61	2.17	12.43	0.234
10	225	1584.00	355.00	77.59	324.00	79.55	3.80	90.10	9.90	9.68	15.62	2.43	13.94	0.281
13	228	2060.00	374.00	81.84	356.00	82.72	3.70	91.50	8.50	6.75	22.40	2.21	12.64	0.294
14	229	2016.00	412.00	79.56	396.00	80.36	3.40	89.40	10.60	6.83	22.14	2.19	12.52	0.275
16	231	1849.00	386.00	79.12	338.00	81.72	3.40	89.10	10.90	7.72	19.59	2.27	12.98	0.260
18	233	1632.00	387.00	76.29	355.00	78.25	4.50	89.70	10.30	9.00	16.80	2.33	13.35	0.351
20	235	1933.00	359.00	81.43	310.00	83.96	4.00	88.90	11.10	7.33	20.63	2.25	12.88	0.299
22	237	1807.00	374.00	79.30	322.00	82.18	4.22	90.00	10.00	8.04	18.81	2.31	13.21	0.318
24	239	1712.00	371.00	78.33	337.00	80.32	3.30	90.20	9.80	7.80	19.38	2.12	12.14	0.278
26	241	1578.00	365.00	76.87	329.00	79.15	4.07	91.80	8.20	9.75	15.51	2.44	13.99	0.307
28	243	2095.00	388.00	81.48	381.00	81.81	4.20	87.80	12.20	7.90	19.14	2.63	15.05	0.272
Mac 02	245	1914.00	386.00	79.83	328.00	82.86	3.90	91.00	9.00	7.94	19.04	2.41	13.82	0.282
04	247	1705.00	438.00	74.31	388.00	77.24	3.60	91.70	8.30	8.64	17.50	2.34	13.39	0.290
06	249	1629.00	385.00	76.37	347.00	78.70	3.70	90.20	9.80	7.85	19.26	2.03	11.63	0.332
08	251	2030.00	426.00	79.01	391.00	80.74	4.20	89.50	10.50	8.04	18.81	2.59	14.84	0.285
09	252	1995.00	436.00	78.15	399.00	80.00	4.50	90.60	9.40	8.82	17.14	2.79	16.00	0.290

11	254	1763.00	463.00	73.74	414.00	76.52	4.45	88.80	11.20	9.10	16.62	2.55	14.58	0.322	
13	256	1544.00	360.00	76.68	325.00	78.95	5.28	89.00	11.00	10.99	13.76	2.69	15.43	0.351	
15	258	1512.00	447.00	70.44	400.00	73.54	4.60	91.50	9.50	11.97	12.63	2.87	16.45	0.316	
18	261	1924.00	435.00	77.39	382.00	80.15	4.50	89.30	10.70	8.46	17.87	2.58	14.80	0.308	
20	263	1795.00	462.00	74.26	407.00	77.33	4.72	90.00	10.00	9.94	15.21	2.83	16.22	0.308	
22	265	1644.00	378.00	77.01	333.00	79.74	4.87	88.70	11.30	11.16	13.55	2.91	16.68	0.295	
24	267	1565.00	342.00	78.15	309.00	80.26	4.94	91.70	8.30	10.70	14.13	2.66	15.22	0.337	
25	268	1608.00	524.00	67.41	482.00	70.02	5.69	88.50	11.50	10.52	14.37	2.69	15.38	0.425	
26	269	1536.00	420.00	72.66	375.00	75.59	4.51	90.70	9.30	10.31	14.67	2.51	14.40	0.342	
27	270	1957.00	533.00	72.76	501.00	74.40	5.93	88.20	11.80	12.82	11.79	3.98	22.81	0.280	
29	272	1745.00	516.00	70.43	488.00	72.03	5.78	89.10	10.90	13.18	11.47	3.65	20.91	0.311	
31	274	1593.00	509.00	68.05	477.00	70.06	5.64	87.00	13.00	13.05	11.59	3.30	18.90	0.337	
Apr 02	276	1847.00	483.00	73.85	453.00	75.47	5.80	88.70	11.30	12.00	12.60	3.52	20.15	0.308	
04	278	1714.00	425.00	75.20	398.00	76.78	5.44	89.00	11.00	11.85	12.76	3.22	18.46	0.310	
05	279	1680.00	467.00	72.20	433.00	74.23	5.58	87.60	12.40	13.59	11.13	3.62	20.76	0.288	
07	281	1563.00	452.00	71.08	420.00	73.13	5.70	89.20	10.80	14.52	10.41	3.60	20.63	0.306	
09	283	2050.00	469.00	77.12	441.00	78.49	5.00	88.50	11.50	10.90	13.87	3.55	20.31	0.252	
12	286	1792.00	482.00	73.10	454.00	74.67	5.89	89.90	10.10	13.72	11.02	3.90	22.35	0.288	
14	288	1572.00	465.00	70.42	431.00	72.58	5.10	88.40	11.60	13.06	11.58	3.26	18.66	0.303	
16	290	2067.00	483.00	76.63	452.00	78.13	5.30	89.40	10.60	11.65	12.98	3.82	21.89	0.252	
18	292	1832.00	500.00	72.71	467.00	74.51	5.72	90.00	10.00	13.56	11.15	3.94	22.58	0.278	
20	294	1730.00	481.00	72.20	448.00	74.10	6.00	88.40	11.40	13.64	11.09	3.75	21.45	0.303	
22	296	1575.00	522.00	66.86	489.00	68.95	5.80	87.50	12.50	15.55	9.72	3.89	22.26	0.301	
24	298	2170.00	489.00	77.47	452.00	79.17	6.10	90.30	9.70	12.00	12.60	4.13	23.67	0.267	
26	300	1904.00	443.00	76.73	411.00	78.41	5.95	88.80	11.20	13.00	11.63	3.93	22.50	0.272	
28	302	1731.00	458.00	73.54	425.00	75.45	5.70	89.60	10.40	13.60	11.12	3.74	21.40	0.288	
30	304	1564.00	430.00	72.51	397.00	74.62	7.25	91.20	8.80	15.18	9.96	3.77	21.58	0.373	
May 02	306	1990.00	481.00	75.83	446.00	77.59	6.91	88.50	11.50	14.00	10.80	4.42	25.33	0.283	
04	308	1865.00	489.00	73.78	451.00	75.82	6.80	88.90	11.10	13.92	10.86	4.12	23.60	0.307	
07	311	1512.00	393.00	74.01	358.00	76.32	6.14	89.80	10.20	13.84	10.92	3.32	19.02	0.345	
08	312	1856.00	386.00	79.20	355.00	80.87	6.46	89.40	10.60	13.76	10.99	4.05	23.22	0.280	
10	314	1696.00	478.00	71.82	444.00	73.82	5.98	88.70	11.30	13.24	11.42	3.56	20.41	0.320	
12	316	2055.00	490.00	76.16	468.00	77.23	6.95	87.40	12.60	13.65	11.08	4.45	25.50	0.280	
14	318	1991.00	462.00	76.80	428.00	78.50	7.05	89.40	10.60	14.50	10.43	4.58	26.24	0.278	
16	320	1865.00	511.00	72.60	470.00	74.80	6.50	90.20	9.80	14.72	10.27	4.36	24.96	0.286	
18	322	1724.00	468.00	72.85	436.00	74.71	7.83	88.20	11.80	15.78	9.58	4.32	24.73	0.340	
20	324	1665.00	623.00	62.58	587.00	64.74	8.40	89.50	10.50	21.03	7.19	5.56	31.83	0.332	
22	326	1948.00	689.00	64.63	658.00	66.22	9.00	88.10	11.90	20.00	7.56	6.18	35.42	0.307	
24	328	1865.00	692.00	62.90	645.00	65.42	7.70	87.70	12.30	19.70	7.68	5.83	33.40	0.281	
26	330	1695.00	602.00	64.48	564.00	66.73	7.60	88.00	12.00	20.25	7.47	5.45	31.20	0.292	
28	332	2011.00	725.00	63.95	689.00	65.74	7.96	89.00	11.00	20.80	7.27	6.64	38.03	0.258	
30	334	1840.00	700.00	61.96	653.00	64.51	8.36	88.70	11.30	20.76	7.28	6.06	34.73	0.301	
Jun 01	336	1680.00	678.00	59.64	622.00	62.98	8.46	85.00	15.00	19.21	7.87	5.12	29.34	0.354	
02	337	1724.00	653.00	62.12	616.00	64.27	7.97	86.90	13.10	20.26	7.46	5.54	31.75	0.309	
04	339	1660.00	634.00	61.81	581.00	65.00	10.23	84.80	15.20	21.60	7.00	5.69	32.60	0.372	
06	341	1468.00	589.00	59.88	531.00	63.83	8.57	85.20	14.80	22.93	6.59	5.34	30.60	0.340	
08	343	1919.00	690.00	64.04	659.00	65.66	9.10	88.70	11.30	22.10	6.84	6.73	38.55	0.290	
10	345	1748.00	721.00	58.75	673.00	61.50	10.50	84.90	15.10	21.80	6.94	6.05	34.64	0.380	
12	347	1698.00	675.00	60.25	622.00	63.37	10.12	87.80	12.80	23.20	6.52	6.25	35.81	0.356	
14	349	2020.00	789.00	60.94	752.00	62.77	8.55	88.90	11.10	21.00	7.20	6.73	38.56	0.285	
16	351	1942.00	777.00	59.99	743.00	61.74	9.80	86.50	13.50	20.86	7.25	6.43	36.83	0.339	
18	353	1872.00	773.00	58.71	738.00	60.58	10.25	84.00	16.00	21.45	7.05	6.37	36.50	0.354	
20	355	1795.00	806.00	55.10	771.00	57.05	9.77	86.80	13.20	21.97	6.88	6.26	35.85	0.377	
22	357	2007.00	815.00	59.39	778.00	61.24	9.50	87.00	13.00	21.10	7.17	6.72	38.50	0.319	
24	359	1846.00	776.00	57.96	742.00	59.80	9.41	86.90	13.10	22.75	6.65	6.67	38.18	0.326	
26	361	1652.00	743.00	55.02	710.00	57.02	9.38	84.70	15.30	22.22	6.80	5.83	33.37	0.380	

28	363	1985.00	827.00	58.34	774.00	61.01	8.98	87.70	12.30	20.31	7.44	6.40	36.65	0.320
30	365	1827.00	779.00	57.36	723.00	60.43	9.19	87.80	12.20	21.16	7.15	6.14	35.14	0.345
July 02	367	1702.00	765.00	55.05	731.00	57.05	9.53	84.20	15.80	23.32	6.48	6.30	36.08	0.354
05	370	1884.00	801.00	57.48	748.00	60.30	10.05	86.10	13.90	22.18	6.82	6.63	37.99	0.343
07	372	1730.00	753.00	56.47	708.00	59.08	9.90	84.70	15.30	22.50	6.72	6.18	35.39	0.365
09	374	1608.00	670.00	58.33	632.00	60.70	9.71	85.60	14.40	23.43	6.45	5.98	34.25	0.363
12	377	1672.00	523.00	68.72	490.00	70.69	5.55	89.50	10.50	12.44	12.15	3.30	18.91	0.338
14	379	1650.00	459.00	72.18	421.00	74.48	5.71	90.30	9.70	13.26	11.40	3.47	19.89	0.316
16	381	1620.00	433.00	73.27	398.00	75.43	5.84	91.00	11.00	13.92	10.86	3.58	20.50	0.312
18	383	1665.00	447.00	73.15	423.00	74.59	5.62	87.60	12.40	13.20	11.45	3.49	19.98	0.300

1281.81	234.50	81.62	196.08	84.65	1.85	90.44	9.52	5.33	28.87	1.08	6.16	0.291
1786.31	388.97	78.07	351.83	80.19	3.89	89.90	10.14	8.34	18.61	2.34	13.41	0.295
1801.48	472.78	73.55	440.48	75.36	6.09	88.94	11.06	13.49	11.29	3.84	22.01	0.298
1799.65	720.96	59.89	678.85	62.26	9.15	86.74	13.28	21.46	7.07	6.12	35.05	0.332
1651.75	465.50	71.83	433.00	73.80	5.68	89.60	10.90	13.21	11.47	3.46	19.82	0.317

RESULTS FOR UAF STEADY STATE PERFORMANCE (LS LEACHATE)

		B O D mg/l			Q	HRT	VOLR	AOLR
Date 1989	Day	infl.	fil.eff	% rem	(l/d)	(hrs)	Kg BOD /m3.d	g BOD /m2.d
Dec 05	158	387.00	25.00	93.54	6.15	24.59	0.38	2.16
07	160	408.00	20.50	94.98	4.50	33.60	0.29	1.67
09	162	388.00	24.00	93.81	5.00	30.24	0.31	1.76
12	165	399.00	31.00	92.23	4.77	31.70	0.30	1.73
14	167	376.50	23.00	93.89	4.86	31.11	0.29	1.66
16	169	352.50	25.00	92.91	5.14	29.42	0.29	1.65
19	172	351.00	18.50	94.73	6.17	24.51	0.34	1.97
21	174	405.00	30.50	92.47	5.25	28.80	0.34	1.93
23	176	400.00	33.00	91.75	4.22	35.83	0.27	1.53
25	178	349.00	23.50	93.27	6.24	24.23	0.35	1.98
27	180	324.00	20.30	93.73	6.43	23.51	0.33	1.89
31	184	392.00	23.00	94.13	4.32	35.00	0.27	1.54
Jan2/90	186	388.00	18.00	95.36	5.60	27.00	0.34	1.98
04	188	363.00	21.00	94.21	6.20	24.39	0.36	2.05
06	190	445.00	29.00	93.48	5.42	27.90	0.38	2.19
08	192	392.00	46.30	88.19	5.35	28.26	0.33	1.91
10	194	350.00	30.00	91.43	6.30	24.00	0.35	2.00
12	196	343.30	37.00	89.22	6.50	23.26	0.35	2.03
13	197	404.00	36.00	91.09	4.99	30.30	0.32	1.83
16	200	359.00	33.00	90.81	5.96	25.37	0.34	1.95
18	202	331.00	18.50	94.41	5.72	26.43	0.30	1.72
20	204	426.00	20.00	95.31	4.25	35.58	0.29	1.65
24	208	467.00	45.00	90.36	5.34	28.31	0.40	2.27
26	210	558.00	44.00	92.11	7.47	20.24	0.66	3.79
28	212	481.00	50.00	89.60	8.20	18.44	0.63	3.59
30	214	475.00	36.00	92.42	7.96	18.99	0.60	3.44
31	215	540.00	57.50	89.35	6.94	21.79	0.59	3.41
Feb 02	217	462.00	42.00	90.91	7.70	19.64	0.56	3.23
06	221	561.00	59.50	89.39	6.07	24.91	0.54	3.10
08	223	526.00	52.00	90.11	7.71	19.61	0.64	3.69
10	225	484.00	58.00	88.02	9.68	15.62	0.74	4.26
13	228	630.00	65.00	89.68	6.75	22.40	0.68	3.87
14	229	687.50	55.00	92.00	6.83	22.14	0.75	4.27
16	231	608.00	66.00	89.14	7.72	19.59	0.75	4.27
20	235	666.00	56.50	91.52	7.33	20.63	0.77	4.44
22	237	622.00	64.00	89.71	8.04	18.81	0.79	4.55
24	239	615.00	49.00	92.03	7.80	19.38	0.76	4.36
28	243	710.00	73.30	89.68	7.90	19.14	0.89	5.10
Mac 02	245	640.30	62.00	90.32	7.94	19.04	0.81	4.62
04	247	530.00	55.30	89.57	8.64	17.50	0.73	4.16
06	249	474.00	65.50	86.18	7.85	19.26	0.59	3.38
08	251	699.00	76.00	89.13	8.04	18.81	0.89	5.11
09	252	583.00	68.00	88.34	8.82	17.14	0.82	4.67
11	254	539.00	74.00	86.27	9.10	16.62	0.78	4.46
13	256	451.00	54.30	87.96	10.99	13.76	0.79	4.51
18	261	555.00	62.00	88.83	8.46	17.87	0.75	4.27
20	263	548.00	77.00	85.95	9.94	15.21	0.86	4.95
22	265	499.00	69.00	86.17	11.16	13.55	0.88	5.06
24	267	546.00	55.00	89.93	10.70	14.13	0.93	5.31

25	268	529.00	102.00	80.72	10.52	14.37	0.88	5.06
26	269	494.00	87.00	82.39	10.31	14.67	0.81	4.63
27	270	664.00	99.00	85.09	12.82	11.79	1.35	7.74
29	272	700.00	86.30	87.67	13.18	11.47	1.46	8.39
31	274	614.00	87.00	85.83	13.05	11.59	1.27	7.28
Apr 02	276	599.00	95.50	84.06	12.00	12.60	1.14	6.53
04	278	760.00	97.00	87.24	11.85	12.76	1.43	8.19
05	279	740.00	82.50	88.85	13.59	11.13	1.60	9.14
07	281	625.00	86.00	86.24	14.52	10.41	1.44	8.25
09	283	654.00	97.00	85.17	10.90	13.87	1.13	6.48
12	286	840.00	96.80	88.48	13.72	11.02	1.83	10.48
16	290	760.00	82.50	89.14	11.65	12.98	1.41	8.05
18	292	721.00	94.80	86.85	13.57	11.14	1.55	8.89
20	294	650.00	76.00	88.31	13.64	11.09	1.41	8.06
22	296	645.00	82.00	87.29	15.55	9.72	1.59	9.12
24	298	835.00	92.00	88.98	12.00	12.60	1.59	9.11
26	300	605.00	85.00	85.95	13.00	11.63	1.25	7.15
28	302	755.00	90.00	88.08	13.60	11.12	1.63	9.33
30	304	760.00	97.50	87.17	15.18	9.96	1.83	10.49
May 02	306	787.00	105.00	86.66	14.00	10.80	1.75	10.02
04	308	806.30	92.00	88.59	13.92	10.86	1.78	10.20
07	311	693.00	96.30	86.10	13.87	10.90	1.53	8.74
10	314	711.00	82.50	88.40	13.24	11.42	1.49	8.56
12	316	845.00	108.00	87.22	13.65	11.08	1.83	10.49
14	318	789.00	96.00	87.83	14.50	10.43	1.82	10.40
16	320	765.00	103.30	86.50	14.72	10.27	1.79	10.24
18	322	704.00	101.50	85.58	15.78	9.58	1.76	10.10
20	324	637.00	105.00	83.52	21.03	7.19	2.13	12.18
22	326	689.00	104.50	84.83	20.00	7.56	2.19	12.53
24	328	887.50	145.00	83.66	19.70	7.68	2.78	15.89
26	330	684.00	124.50	81.80	20.25	7.47	2.20	12.59
28	332	679.00	109.00	83.95	20.80	7.27	2.24	12.84
30	334	640.00	95.00	85.16	20.76	7.28	2.11	12.08
Jun 02	337	666.00	104.00	84.38	20.26	7.46	2.14	12.27
04	339	760.00	156.00	79.47	21.60	7.00	2.61	14.92
08	343	805.00	128.00	84.10	22.10	6.84	2.82	16.17
10	345	776.00	128.00	83.51	21.80	6.94	2.69	15.38
12	347	656.00	142.00	78.35	23.20	6.52	2.42	13.84
14	349	840.00	172.00	79.52	21.00	7.20	2.80	16.04
16	351	797.00	155.00	80.55	20.86	7.25	2.64	15.11
18	353	833.00	150.00	81.99	21.45	7.05	2.84	16.24
20	355	754.00	148.00	80.37	21.97	6.88	2.63	15.06
22	357	762.00	159.00	79.13	21.10	7.17	2.55	14.62
24	359	825.00	183.00	77.82	22.75	6.65	2.98	17.06
26	361	724.00	156.00	78.45	22.22	6.80	2.55	14.62
28	363	815.00	144.00	82.33	20.31	7.44	2.63	15.05
30	365	762.00	154.00	79.79	21.16	7.15	2.56	14.66
July 02	367	724.00	169.00	76.66	23.32	6.48	2.68	15.35
05	370	810.00	165.00	79.63	22.18	6.82	2.85	16.33
07	372	757.00	160.00	78.86	22.50	6.72	2.70	15.48
09	374	582.00	114.50	80.33	23.43	6.45	2.16	12.40
12	377	538.00	84.50	84.29	12.44	12.15	1.06	6.08
14	379	595.00	70.00	88.24	13.26	11.40	1.25	7.17
18	383	572.00	59.30	89.63	13.20	11.45	1.20	6.86

RESULTS FOR UAF STEADY STATE PERFORMANCE (LS LEACHATE)

		T O C mg/l			Q	HRT	VOLR	AOLR
Date	Day	infl.	fil.eff	% rem	(l/d)	(hrs)	Kg TOC	g TOC
1989							/m3.d	/m2.d
Dec 05	158	238.00	46.00	80.67	5.61	26.95	0.21	1.21
07	160	339.00	60.00	82.30	4.50	33.60	0.24	1.39
09	162	272.00	55.00	79.78	5.00	30.24	0.22	1.24
12	165	345.00	78.00	77.39	4.77	31.70	0.26	1.50
14	167	293.00	69.00	76.45	4.86	31.11	0.23	1.29
16	169	307.00	59.00	80.78	5.14	29.42	0.25	1.43
17	170	280.00	64.00	77.14	4.47	33.83	0.20	1.14
19	172	283.00	58.00	79.51	6.17	24.51	0.28	1.59
21	174	381.00	61.00	83.99	5.25	28.80	0.32	1.82
23	176	292.00	62.00	78.77	4.22	35.83	0.20	1.12
25	178	249.00	45.00	81.93	6.24	24.23	0.25	1.41
27	180	235.00	49.00	79.15	6.43	23.51	0.24	1.37
29	182	298.00	66.00	77.85	4.87	31.05	0.23	1.32
31	184	293.00	53.00	81.91	4.32	35.00	0.20	1.15
Jan2/90	186	283.00	62.00	78.09	5.60	27.00	0.25	1.44
04	188	271.00	52.00	80.81	6.20	24.39	0.27	1.53
06	190	339.00	73.00	78.47	5.42	27.90	0.29	1.67
08	192	288.00	49.00	82.99	5.35	28.26	0.24	1.40
10	194	255.00	53.00	79.22	6.30	24.00	0.26	1.46
12	196	233.00	50.00	78.54	6.50	23.26	0.24	1.38
13	197	300.00	55.00	81.67	4.99	30.30	0.24	1.36
16	200	284.00	54.00	80.99	5.96	25.37	0.27	1.54
18	202	251.00	43.00	82.87	5.72	26.43	0.23	1.31
20	204	328.00	67.00	79.57	4.25	35.58	0.22	1.27
22	206	328.00	78.00	76.22	5.05	29.94	0.26	1.51
24	208	273.00	51.00	81.32	5.34	28.31	0.23	1.33
26	210	415.00	94.00	77.35	7.47	20.24	0.49	2.82
28	212	363.00	89.00	75.48	8.20	18.44	0.47	2.71
30	214	385.00	85.00	77.92	7.96	18.99	0.49	2.79
31	215	392.00	79.00	79.85	6.94	21.79	0.43	2.47
Feb 02	217	328.00	79.00	75.91	7.70	19.64	0.40	2.30
06	221	405.00	91.00	77.53	6.07	24.91	0.39	2.23
08	223	376.00	94.00	75.00	7.71	19.61	0.46	2.64
10	225	362.00	83.00	77.07	9.68	15.62	0.56	3.19
13	228	459.00	88.00	80.83	6.75	22.40	0.49	2.82
14	229	467.00	96.00	79.44	6.83	22.14	0.51	2.90
16	231	384.00	72.00	81.25	7.72	19.59	0.47	2.69
18	233	397.00	81.00	79.60	10.00	15.12	0.63	3.61
20	235	395.00	89.00	77.47	7.33	20.63	0.46	2.63
22	237	402.00	85.00	78.86	8.04	18.81	0.51	2.94
26	241	331.00	64.00	80.66	9.75	15.51	0.51	2.93
28	243	493.00	105.00	78.70	7.90	19.14	0.62	3.54
Mac 02	245	399.00	98.00	75.44	7.94	19.04	0.50	2.88
04	247	362.00	99.00	72.65	8.64	17.50	0.50	2.84
06	249	354.00	93.00	73.73	7.85	19.26	0.44	2.53
08	251	493.00	121.00	75.46	8.04	18.81	0.63	3.60
09	252	460.00	117.00	74.57	8.82	17.14	0.64	3.69
11	254	412.00	109.00	73.54	9.10	16.62	0.60	3.41
13	256	328.00	78.00	76.22	10.99	13.76	0.57	3.28

15	258	351.00	93.00	73.50	11.97	12.63	0.67	3.82
18	261	465.00	111.00	76.13	8.46	17.87	0.62	3.58
20	263	383.00	82.00	78.59	9.94	15.21	0.60	3.46
22	265	375.00	86.00	77.07	11.16	13.55	0.66	3.80
24	267	349.00	87.00	75.07	10.70	14.13	0.59	3.39
25	268	347.00	101.00	70.89	10.52	14.37	0.58	3.32
26	269	372.00	105.00	71.77	10.31	14.67	0.61	3.49
27	270	439.00	120.00	72.67	12.82	11.79	0.89	5.12
29	272	385.00	104.00	72.99	13.18	11.47	0.81	4.61
31	274	400.00	102.00	74.50	13.05	11.59	0.83	4.75
Apr 02	276	389.00	95.00	75.58	12.00	12.60	0.74	4.24
04	278	382.00	70.00	81.68	11.85	12.76	0.72	4.12
05	279	363.00	87.00	76.03	13.59	11.13	0.78	4.48
07	281	412.00	104.00	74.76	15.32	9.87	1.00	5.74
09	283	494.00	94.00	80.97	10.90	13.87	0.85	4.90
12	286	425.00	95.00	77.65	13.72	11.02	0.93	5.30
14	288	418.00	92.00	77.99	13.06	11.58	0.87	4.96
16	290	439.00	102.00	76.77	11.65	12.98	0.81	4.65
20	294	432.00	112.00	74.07	14.64	10.33	1.00	5.75
22	296	386.00	115.00	70.21	15.55	9.72	0.95	5.46
24	298	460.00	124.00	73.04	12.00	12.60	0.88	5.02
26	300	439.00	108.00	75.40	13.00	11.63	0.91	5.19
28	302	445.00	112.00	74.83	13.60	11.12	0.96	5.50
30	304	396.00	94.00	76.26	15.18	9.96	0.95	5.46
May 02	306	415.00	100.00	75.90	14.00	10.80	0.92	5.28
04	308	435.00	119.00	72.64	13.92	10.86	0.96	5.50
07	311	373.00	93.00	75.07	13.84	10.92	0.82	4.69
08	312	480.00	103.00	78.54	13.76	10.99	1.05	6.00
10	314	424.00	95.00	77.59	13.24	11.42	0.89	5.10
12	316	507.00	126.00	75.15	13.65	11.08	1.10	6.29
14	318	489.00	122.00	75.05	14.50	10.43	1.13	6.45
16	320	427.00	124.00	70.96	14.72	10.27	1.00	5.71
18	322	406.00	100.00	75.37	15.78	9.58	1.02	5.82
20	324	343.00	86.00	74.93	21.03	7.19	1.14	6.56
22	326	414.00	152.00	63.29	20.00	7.56	1.31	7.53
24	328	451.00	162.00	64.08	19.70	7.68	1.41	8.08
26	330	436.00	181.00	58.49	20.25	7.47	1.40	8.03
28	332	460.00	152.00	66.96	20.80	7.27	1.52	8.70
30	334	468.00	150.00	67.95	20.76	7.28	1.54	8.83
Jun 01	336	462.00	184.00	60.17	19.21	7.87	1.41	8.07
02	337	429.00	140.00	67.37	20.26	7.46	1.38	7.90
04	339	416.00	142.00	65.87	21.60	7.00	1.43	8.17
06	341	403.00	153.00	62.03	22.93	6.59	1.47	8.40
08	343	392.00	138.00	64.80	22.10	6.84	1.38	7.88
10	345	442.00	142.00	67.87	21.80	6.94	1.53	8.76
12	347	382.00	131.00	65.71	23.20	6.52	1.41	8.06
14	349	451.00	148.00	67.18	21.00	7.20	1.50	8.61
16	351	409.00	145.00	64.55	20.86	7.25	1.35	7.76
18	353	386.00	133.00	65.54	21.45	7.05	1.31	7.53
22	357	405.00	148.00	63.46	21.10	7.17	1.36	7.77
24	359	438.00	151.00	65.53	22.75	6.65	1.58	9.06
26	361	417.00	159.00	61.87	22.22	6.80	1.47	8.42
28	363	413.00	150.00	63.68	20.31	7.44	1.33	7.63
30	365	394.00	141.00	64.21	21.16	7.15	1.32	7.58
July 02	367	376.00	145.00	61.44	23.32	6.48	1.39	7.97
05	370	400.00	149.00	62.75	22.18	6.82	1.41	8.07

07	372	380.00	151.00	60.26	22.50	6.72	1.36	7.77
09	374	426.00	153.00	64.08	23.43	6.45	1.58	9.07
12	377	342.00	111.00	67.54	12.44	12.15	0.68	3.87
14	379	365.00	108.00	70.41	13.26	11.40	0.77	4.40
16	381	329.00	92.00	72.04	13.92	10.86	0.73	4.16
18	383	418.00	101.00	75.84	13.20	11.45	0.88	5.02

289.92	58.15	79.94	5.33	28.87	0.24	1.39
397.63	91.15	77.03	8.48	18.29	0.53	3.04
422.33	103.63	75.43	13.84	11.09	0.92	5.28
418.75	150.00	64.13	21.45	7.07	1.42	8.15
363.50	103.00	71.46	13.21	11.47	0.76	4.36

RESULTS FOR RBC STEADY STATE PERFORMANCE (HS lEACHATE)

Date 1990	Day	COD mg/l infl.	fil.eff	% rem	Q (l/d)	HRT (hrs)	VOLR Kg COD /m3.d	CODres Kg COD /m3.d	CODrem Kg COD /m3.d	AOLR g COD /m2.d
Sept 15	400	4565.00	204.00	95.53	1.87	42.35	2.59	0.12	2.47	10.07
17	402	5640.00	504.00	91.06	2.15	36.84	3.67	0.33	3.35	14.30
19	404	5580.00	574.00	89.71	2.30	34.43	3.89	0.40	3.49	15.13
22	407	5495.00	473.00	91.39	1.92	41.25	3.20	0.28	2.92	12.44
24	409	5385.00	541.00	89.95	1.74	45.52	2.84	0.29	2.55	11.05
26	411	5620.00	439.00	92.19	1.86	42.58	3.17	0.25	2.92	12.33
29	414	5455.00	518.00	90.50	2.11	37.54	3.49	0.33	3.16	13.57
Oct 01	416	6140.00	484.00	92.12	2.44	32.46	4.54	0.36	4.18	17.67
03	418	5955.00	783.00	86.85	1.96	40.41	3.54	0.47	3.07	13.76
05	420	5570.00	477.00	91.44	1.85	42.81	3.12	0.27	2.86	12.15
06	421	5715.00	778.00	86.39	2.28	34.74	3.95	0.54	3.41	15.37
08	423	5781.00	955.00	83.48	2.01	39.40	3.52	0.58	2.94	13.70
11	426	5725.00	823.00	85.62	1.62	48.89	2.81	0.40	2.41	10.94
13	428	5610.00	908.00	83.81	2.37	33.42	4.03	0.65	3.38	15.68
15	430	5870.00	879.00	85.03	1.90	41.68	3.38	0.51	2.87	13.15
17	432	5920.00	1269.00	78.56	3.27	24.22	5.87	1.26	4.61	22.83
19	434	5830.00	1425.00	75.56	3.55	22.31	6.27	1.53	4.74	24.41
22	437	5795.00	1385.00	76.10	3.13	25.30	5.50	1.31	4.18	21.39
23	438	6075.00	1431.00	76.44	3.45	22.96	6.35	1.50	4.86	24.72
25	440	5940.00	1461.00	75.40	3.50	22.63	6.30	1.55	4.75	24.52
27	442	6430.00	1467.00	77.19	3.48	22.76	6.78	1.55	5.23	26.39
30	445	6360.00	1650.00	74.06	3.47	22.82	6.69	1.74	4.95	26.03
Nov 01	447	6220.00	1470.00	76.37	3.20	24.75	6.03	1.43	4.61	23.47
03	449	6450.00	1800.00	72.09	3.57	22.18	6.98	1.95	5.03	27.15
05	451	6300.00	1940.00	69.21	3.48	22.76	6.64	2.05	4.60	25.85
07	453	6000.00	1996.00	66.73	3.71	21.35	6.75	2.24	4.50	26.25
10	456	5955.00	1518.00	74.51	3.16	25.06	5.70	1.45	4.25	22.19
12	458	5805.00	1812.00	68.79	3.36	23.57	5.91	1.84	4.07	23.00
14	460	5845.00	1836.00	68.59	3.12	25.38	5.53	1.74	3.79	21.51
16	462	6120.00	1804.00	70.52	3.26	24.29	6.05	1.78	4.26	23.53
17	463	6125.00	2680.00	56.24	3.42	23.16	6.35	2.78	3.57	24.70
18	464	6090.00	1957.00	67.87	3.51	22.56	6.48	2.08	4.40	25.21
20	466	6060.00	1888.00	68.84	4.94	16.03	9.07	2.83	6.25	35.30
22	468	5950.00	1759.00	70.44	5.10	15.53	9.20	2.72	6.48	35.78
25	471	6345.00	2360.00	62.81	4.32	18.33	8.31	3.09	5.22	32.32
27	473	6195.00	1721.00	72.22	4.86	16.30	9.12	2.53	6.59	35.50
29	475	6330.00	2074.00	67.24	5.16	15.35	9.90	3.24	6.65	38.52
Dec 02	478	6255.00	1927.00	69.19	4.50	17.60	8.53	2.63	5.90	33.19
03	479	5980.00	2184.00	63.48	4.56	17.37	8.26	3.02	5.25	32.16
05	481	5595.00	1950.00	65.15	4.62	17.14	7.83	2.73	5.10	30.48
07	483	6075.00	1628.00	73.20	5.04	15.71	9.28	2.49	6.79	36.11
10	486	5970.00	2340.00	60.80	4.50	17.60	8.14	3.19	4.95	31.68
12	488	5580.00	2230.00	60.04	5.85	13.54	9.89	3.95	5.94	38.49
14	490	6325.00	1946.00	69.23	4.64	17.07	8.89	2.74	6.16	34.61
16	492	6480.00	2490.00	61.57	5.14	15.41	10.09	3.88	6.21	39.28
18	494	6920.00	3650.00	47.25	7.91	10.01	16.59	8.75	7.84	64.55
20	496	7180.00	3862.00	46.21	8.11	9.77	17.65	9.49	8.15	68.67

22	498	7455.00	3930.00	47.28	7.61	10.41	17.19	9.06	8.13	66.90
25	501	6840.00	3810.00	44.30	7.63	10.38	15.81	8.81	7.01	61.54
27	503	7335.00	4028.00	45.09	8.06	9.83	17.92	9.84	8.08	69.72
29	505	7310.00	4150.00	43.23	8.50	9.32	18.83	10.69	8.14	73.27
31	507	7265.00	4230.00	41.78	8.62	9.19	18.98	11.05	7.93	73.85
2/1/91	509	7190.00	4290.00	40.33	8.34	9.50	18.17	10.84	7.33	70.71
04	511	7240.00	4165.00	42.47	8.10	9.78	17.77	10.22	7.55	69.16
06	513	7320.00	4160.00	43.17	7.87	10.06	17.46	9.92	7.54	67.93
		5607.07	622.67	89.01	2.03	39.62	3.45	0.38	3.07	13.42
		6069.67	1617.60	73.34	3.38	23.49	6.22	1.66	4.56	24.21
		6087.69	2038.23	66.48	4.86	16.38	8.96	3.00	5.96	34.88
		7205.50	4027.50	44.11	8.08	9.82	17.64	9.87	7.77	68.63

RESULTS FOR RBC STEADY STATE PERFORMANCE (HS LEACHATE)

Date	Day	B O D mg/l			VOLR	BODres	BODrem	AOLR	BODres	BODrem
1990		infl.	fil.eff	% rem	Kg BOD /m3.d	Kg BOD /m3.d	Kg BOD /m3.d	g BOD /m2.d	g BOD /m2.d	g BOD /m2.d
Sept 15	400	2865.00	42.00	98.53	1.62	0.02	1.60	6.32	0.09	6.23
19	404	3290.00	73.00	97.78	2.29	0.05	2.24	8.34	0.19	8.16
22	407	3070.00	121.00	96.06	1.79	0.07	1.72	8.33	0.33	8.00
24	409	3280.00	138.00	95.79	1.73	0.07	1.66	7.43	0.31	7.11
26	411	3145.00	94.00	97.01	1.77	0.05	1.72	6.45	0.19	6.26
29	414	3240.00	147.00	95.46	2.07	0.09	1.98	7.11	0.32	6.78
Oct 01	416	3450.00	87.50	97.46	2.55	0.06	2.49	8.58	0.22	8.37
03	418	3320.00	146.00	95.60	1.97	0.09	1.89	9.55	0.42	9.13
08	423	3290.00	195.00	94.07	2.00	0.12	1.89	7.60	0.45	7.15
11	426	2930.00	283.00	90.34	1.44	0.14	1.30	6.39	0.62	5.77
15	430	3190.00	266.00	91.66	1.84	0.15	1.68	8.58	0.72	7.86
17	432	3230.00	381.00	88.20	3.20	0.38	2.82	7.66	0.90	6.75
19	434	3425.00	528.00	84.58	3.68	0.57	3.12	6.54	1.01	5.53
22	437	3350.00	495.00	85.22	3.18	0.47	2.71	9.36	1.38	7.98
27	442	3950.00	1010.00	74.43	4.17	1.07	3.10	8.85	2.26	6.59
Nov 01	447	3575.00	964.00	73.03	3.47	0.93	2.53	13.79	3.72	10.07
03	449	3950.00	1180.00	70.13	4.27	1.28	3.00	16.54	4.94	11.60
10	456	3600.00	820.00	77.22	3.45	0.79	2.66	13.29	3.03	10.26
12	458	3725.00	915.00	75.44	3.79	0.93	2.86	15.15	3.72	11.43
14	460	3750.00	853.00	77.25	3.55	0.81	2.74	15.48	3.52	11.96
16	462	3575.00	735.00	79.44	3.53	0.73	2.81	14.67	3.02	11.65
17	463	3425.00	728.00	78.74	3.55	0.75	2.80	14.02	2.98	11.04
18	464	3450.00	651.00	81.13	3.67	0.69	2.98	13.02	2.46	10.56
20	466	3280.00	849.00	74.12	5.07	1.31	3.76	13.81	3.57	10.23
25	471	3350.00	958.00	71.40	4.39	1.25	3.13	13.75	3.93	9.82
27	473	3400.00	826.00	75.71	5.01	1.22	3.79	14.88	3.61	11.26
29	475	3645.00	943.00	74.13	5.70	1.47	4.22	13.58	3.51	10.07
Dec 02	478	3260.00	746.00	77.12	4.45	1.02	3.43	12.92	2.96	9.96
05	481	3115.00	885.00	71.59	4.36	1.24	3.12	11.46	3.26	8.20
07	483	3425.00	940.00	72.55	5.23	1.44	3.80	13.17	3.61	9.55
09	485	3290.00	973.00	70.43	4.49	1.33	3.16	13.27	3.92	9.34
12	488	3220.00	1029.00	68.04	5.71	1.82	3.88	13.33	4.26	9.07
14	490	3355.00	984.00	70.67	4.72	1.38	3.33	19.54	5.73	13.81
18	494	3890.00	1425.00	63.37	9.32	3.42	5.91	23.40	8.57	14.82
20	496	3710.00	1470.00	60.38	9.12	3.61	5.50	18.90	7.49	11.41
24	500	3560.00	1448.00	59.33	8.23	3.35	4.88	20.40	8.30	12.10
29	505	3720.00	1380.00	62.90	9.58	3.55	6.03	22.64	8.40	14.24
31	507	3905.00	1574.00	59.69	10.20	4.11	6.09	20.72	8.35	12.37
4/1/91	511	3700.00	1520.00	58.92	9.08	3.73	5.35	19.90	8.17	11.72
06	513	3880.00	1577.00	59.36	9.25	3.76	5.49	21.14	8.59	12.55
		3188.18	144.77	95.43	1.92	0.08	1.83	7.70	0.35	7.35
		3613.00	788.10	78.50	3.63	0.79	2.83	12.13	2.75	9.38
		3334.00	913.30	72.58	4.91	1.35	3.56	13.97	3.84	10.13
		3766.43	1484.86	60.56	9.26	3.65	5.61	21.01	8.27	12.75

RESULTS FOR RBC STEADY STATE PERFORMANCE (HS LEACHATE)

		T O C mg/l			VOLR	TOCres	TOCrem	AOLR	TOCres	TOCrem
Date 1990	Day	infl.	fil.eff	% rem	Kg TOC /m3.d	Kg TOC /m3.d	Kg TOC /m3.d	g TOC /m2.d	g TOC /m2.d	g TOC /m2.d
Sept 15	400	1034.00	25.00	97.58	0.59	0.01	0.57	2.28	0.06	2.23
17	402	1324.00	53.00	96.00	0.86	0.03	0.83	3.36	0.13	3.22
19	404	1473.00	84.00	94.30	1.03	0.06	0.97	4.00	0.23	3.77
22	407	1463.00	90.00	93.85	0.85	0.05	0.80	3.31	0.20	3.11
24	409	1558.00	72.00	95.38	0.82	0.04	0.78	3.20	0.15	3.05
26	411	1192.00	88.00	92.62	0.67	0.05	0.62	2.61	0.19	2.42
29	414	1375.00	79.00	94.25	0.88	0.05	0.83	3.42	0.20	3.22
Oct 01	416	1517.00	58.00	96.18	1.12	0.04	1.08	4.36	0.17	4.20
03	418	1426.00	87.00	93.90	0.85	0.05	0.80	3.30	0.20	3.09
05	420	1209.00	86.00	92.89	0.68	0.05	0.63	2.64	0.19	2.45
06	421	1311.00	198.00	84.90	0.91	0.14	0.77	3.52	0.53	2.99
08	423	1375.00	104.00	92.44	0.84	0.06	0.77	3.26	0.25	3.01
11	426	1420.00	164.00	88.45	0.70	0.08	0.62	2.71	0.31	2.40
13	428	1561.00	192.00	87.70	1.12	0.14	0.98	4.36	0.54	3.83
15	430	1289.00	148.00	88.52	0.74	0.09	0.66	2.89	0.33	2.56
17	432	1438.00	212.00	85.26	1.42	0.21	1.21	5.55	0.82	4.73
19	434	1475.00	222.00	84.95	1.59	0.24	1.35	6.17	0.93	5.25
22	437	1381.00	259.00	81.25	1.31	0.25	1.06	5.10	0.96	4.14
23	438	1501.00	263.00	82.48	1.57	0.27	1.29	6.11	1.07	5.04
25	440	1387.00	282.00	79.67	1.47	0.30	1.17	5.72	1.16	4.56
27	442	1668.00	318.00	80.94	1.76	0.34	1.42	6.85	1.31	5.54
Nov 01	447	1426.00	204.00	85.69	1.38	0.20	1.18	5.84	0.83	5.00
03	449	1582.00	402.00	74.59	1.71	0.43	1.28	5.97	1.52	4.45
05	451	1494.00	284.00	80.99	1.58	0.30	1.28	6.29	1.20	5.09
10	456	1442.00	246.00	82.94	1.38	0.24	1.15	5.92	1.01	4.91
12	458	1295.00	320.00	75.29	1.32	0.33	0.99	5.67	1.40	4.27
14	460	1743.00	443.00	74.58	1.65	0.42	1.23	6.50	1.65	4.84
16	462	1598.00	545.00	65.89	1.58	0.54	1.04	6.33	2.16	4.17
17	463	1524.00	616.00	59.58	1.58	0.64	0.94	5.61	2.27	3.34
18	464	1572.00	523.00	66.73	1.67	0.56	1.12	6.04	2.01	4.03
20	466	1724.00	534.00	69.03	2.58	0.80	1.78	6.95	2.15	4.80
22	468	1671.00	374.00	77.62	2.58	0.58	2.00	6.92	1.55	5.37
25	471	1868.00	603.00	67.72	2.45	0.79	1.66	10.88	3.51	7.37
27	473	1769.00	446.00	74.79	2.61	0.66	1.95	10.64	2.68	7.96
29	475	1925.00	740.00	61.56	3.01	1.16	1.85	9.81	3.77	6.04
Dec 02	478	1773.00	513.00	71.07	2.42	0.70	1.72	10.16	2.94	7.22
03	479	1916.00	780.00	59.29	2.65	1.08	1.57	11.66	4.75	6.91
07	483	1814.00	566.00	68.80	2.77	0.86	1.91	9.63	3.00	6.62
10	486	1712.00	716.00	58.18	2.33	0.98	1.36	9.21	3.85	5.36
12	488	1670.00	656.00	60.72	2.96	1.16	1.80	9.10	3.57	5.52
14	490	1739.00	505.00	70.96	2.45	0.71	1.74	10.34	3.00	7.33
16	492	2174.00	795.00	63.43	3.39	1.24	2.15	11.54	4.22	7.32
18	494	2045.00	884.00	56.77	4.90	2.12	2.78	14.11	6.10	8.01
20	496	1991.00	826.00	58.51	4.89	2.03	2.86	10.89	4.52	6.37
22	498	2108.00	1238.00	41.27	4.86	2.85	2.01	12.78	7.50	5.27
25	501	1926.00	1195.00	37.95	4.45	2.76	1.69	17.97	11.15	6.82
27	503	2088.00	1550.00	25.77	5.10	3.79	1.31	19.97	14.82	5.15
31	507	1975.00	1496.00	24.25	5.16	3.91	1.25	17.72	13.43	4.30
2/1/91	509	1924.00	1503.00	21.88	4.86	3.80	1.06	17.31	13.52	3.79

RESULTS FOR UAF STEADY STATE PERFORMANCE (HS LEACHATE)

Date 1990	Day	COD mg/l			biogas	CH4	CO2	Q	HRT	VOLR	EMP	AOLR
		infl.	fil.eff	% rem	(l/d)	(%)	(%)	(l/d)	(hrs)	Kg COD /m3.d	m3 CH4/ Kg CODred	g COD /m2.d
Sept 15	442	4565.00	366.00	91.98	2.58	87.00	13.00	1.84	82.17	1.33	0.291	7.64
17	444	5640.00	514.00	90.89	3.80	87.30	12.70	2.05	73.76	1.84	0.316	10.51
19	446	5580.00	574.00	89.71	5.20	86.20	13.80	2.67	56.63	2.36	0.335	13.54
22	449	5495.00	483.00	91.21	3.55	86.10	13.90	1.85	81.73	1.61	0.330	9.24
24	451	5385.00	561.00	89.58	3.73	86.50	13.50	1.99	75.98	1.70	0.336	9.74
26	453	5620.00	439.00	92.19	3.80	86.00	14.00	2.06	73.40	1.84	0.306	10.52
29	456	5455.00	618.00	88.67	3.30	85.00	15.00	1.86	81.29	1.61	0.312	9.22
Oct 01	458	6140.00	660.00	89.25	5.49	84.10	15.90	2.52	60.00	2.46	0.334	14.07
03	460	5955.00	783.00	86.85	5.70	86.80	13.20	2.84	53.24	2.68	0.337	15.37
05	462	5570.00	677.00	87.85	3.40	84.70	15.30	2.00	75.60	1.77	0.294	10.13
06	463	5715.00	841.00	85.28	5.40	87.30	12.70	2.80	54.00	2.54	0.345	14.55
08	465	5781.00	755.00	86.94	5.20	85.00	15.00	2.59	58.38	2.38	0.340	13.61
11	468	5725.00	823.00	85.62	4.00	84.70	15.30	2.10	72.00	1.91	0.329	10.93
13	470	5610.00	904.00	83.89	5.15	86.00	14.00	2.64	57.27	2.35	0.356	13.46
15	472	5870.00	897.00	84.72	5.29	84.60	15.40	2.69	56.21	2.51	0.335	14.35
17	474	5920.00	1068.00	81.96	7.64	82.80	17.20	5.10	29.65	4.79	0.256	27.45
19	476	5830.00	1154.00	80.21	9.81	83.50	16.50	5.42	27.90	5.02	0.323	28.73
22	479	5795.00	1075.00	81.45	7.16	85.80	14.20	4.75	31.83	4.37	0.274	25.02
23	480	6075.00	1173.00	80.69	7.90	82.90	17.10	4.62	32.73	4.46	0.289	25.51
25	482	5940.00	1290.00	78.28	9.50	80.10	19.90	5.38	28.10	5.07	0.304	29.05
27	484	6430.00	1319.00	79.49	9.44	80.10	19.90	5.04	30.00	5.14	0.294	29.46
30	487	6360.00	1177.00	81.49	10.00	82.50	17.50	4.95	30.55	5.00	0.322	28.62
Nov 01	489	6220.00	1350.00	78.30	8.85	82.00	18.00	4.90	30.86	4.84	0.304	27.71
03	491	6450.00	1265.00	80.39	10.37	83.80	16.20	4.96	30.48	5.08	0.338	29.08
05	493	6300.00	1154.00	81.68	9.00	81.80	18.20	4.47	33.83	4.47	0.320	25.60
07	495	6000.00	1260.00	79.00	8.50	84.80	15.20	4.45	33.98	4.24	0.342	24.27
10	498	5955.00	1085.00	81.78	8.70	86.00	14.00	4.79	31.57	4.53	0.321	25.93
12	500	5805.00	1244.00	78.57	8.20	84.20	15.80	4.56	33.16	4.20	0.332	24.06
14	502	5845.00	1145.00	80.41	8.10	86.50	13.50	4.53	33.38	4.20	0.329	24.07
16	504	6120.00	1476.00	75.88	8.42	86.40	13.60	4.68	32.31	4.55	0.335	26.04
17	505	6125.00	1676.00	72.64	10.39	85.60	14.40	5.36	28.21	5.21	0.373	29.85
18	506	6090.00	1595.00	73.81	10.75	84.90	15.10	5.50	27.49	5.32	0.369	30.45
20	508	6060.00	1496.00	75.31	12.42	83.80	16.20	7.92	19.09	7.62	0.288	43.63
22	510	5950.00	1629.00	72.62	13.00	84.80	15.20	7.56	20.00	7.14	0.337	40.89
25	513	6345.00	1858.00	70.72	11.92	83.60	16.40	6.72	22.50	6.77	0.330	38.76
27	515	6195.00	1621.00	73.83	13.30	82.50	17.50	7.20	21.00	7.08	0.333	40.55
29	517	6330.00	1702.00	73.11	14.30	85.40	14.60	8.00	18.90	8.04	0.330	46.04
Dec 02	520	6255.00	1782.00	71.51	13.90	82.50	17.50	7.37	20.52	7.32	0.348	41.91
03	521	5980.00	1604.00	73.18	14.41	85.30	14.70	8.64	17.50	8.20	0.325	46.97
05	523	5595.00	1580.00	71.76	13.50	80.10	19.90	8.16	18.53	7.25	0.330	41.50
07	525	6075.00	1528.00	74.85	14.24	83.90	16.90	7.99	18.92	7.70	0.329	44.13
10	528	5970.00	1695.00	71.61	14.00	85.50	14.50	7.87	19.21	7.46	0.356	42.71
12	530	5580.00	1705.00	69.44	13.60	80.80	19.20	8.22	18.39	7.28	0.345	41.70
14	532	6325.00	1846.00	70.81	13.90	82.30	17.70	7.78	19.43	7.81	0.328	44.74
16	534	6480.00	1800.00	72.22	14.70	84.70	15.30	8.16	18.53	8.39	0.326	48.07
18	536	6920.00	2460.00	64.45	19.60	83.70	16.30	10.70	14.13	11.75	0.344	67.31
20	538	7180.00	2725.00	62.05	21.30	82.50	17.50	11.50	13.15	13.11	0.343	75.06
22	540	7455.00	2595.00	65.19	20.10	84.00	16.00	10.42	14.51	12.33	0.333	70.62

25	543	6840.00	2610.00	61.84	19.84	83.90	16.10	11.25	13.44	12.21	0.350	69.95
27	545	7335.00	2910.00	60.33	21.90	81.00	19.00	11.60	13.03	13.51	0.346	77.35
29	547	7310.00	2760.00	62.24	19.70	82.90	17.10	11.00	13.75	12.76	0.326	73.10
31	549	7265.00	2870.00	60.50	20.14	84.60	15.40	11.73	12.89	13.53	0.331	77.47
2/1/91	551	7190.00	2805.00	60.99	21.41	82.20	17.80	11.66	12.97	13.31	0.344	76.21
04	553	7240.00	2940.00	59.39	20.18	82.60	17.40	11.18	13.52	12.85	0.347	73.58
06	555	7320.00	2885.00	60.59	21.96	81.30	18.70	11.47	13.18	13.33	0.351	76.33
		5607.07	659.67	88.31	4.37	85.82	14.18	2.30	67.44	2.06	0.326	11.79
		6069.67	1215.67	79.97	8.77	83.55	16.45	4.84	31.35	4.66	0.312	26.71
		6087.69	1680.46	72.38	13.63	83.48	16.58	7.81	19.43	7.54	0.331	43.20
		7205.50	2756.00	61.76	20.61	82.87	17.13	11.25	13.46	12.87	0.341	73.70

RESULTS FOR UAF STEADY STATE PERFORMANCE (HS LEACHATE)

		B O D mg/l			VOLR	BODres	BODrem	AOLR	BODres	BODrem
Date 1990	Day	infl.	fil.eff	% rem	Kg BOD /m3.d	Kg BOD /m3.d	Kg BOD /m3.d	g BOD /m2.d	g BOD /m2.d	g BOD /m2.d
Sept 15	442	2865.00	114.00	96.02	0.84	0.03	0.80	4.79	0.19	4.60
19	446	3290.00	264.00	91.98	1.39	0.11	1.28	7.99	0.64	7.34
22	449	3070.00	198.00	93.55	0.90	0.06	0.84	5.16	0.33	4.83
24	451	3280.00	189.00	94.24	1.04	0.06	0.98	5.93	0.34	5.59
26	453	3145.00	232.00	92.62	1.03	0.08	0.95	5.89	0.43	5.46
29	456	3240.00	211.00	93.49	0.96	0.06	0.89	5.48	0.36	5.12
Oct 01	458	3450.00	125.00	96.38	1.38	0.05	1.33	7.90	0.29	7.62
03	460	3320.00	158.00	95.24	1.50	0.07	1.43	8.57	0.41	8.16
08	465	3290.00	173.00	94.74	1.35	0.07	1.28	7.75	0.41	7.34
11	468	2930.00	261.00	91.09	0.98	0.09	0.89	5.59	0.50	5.10
15	472	3190.00	193.00	93.95	1.36	0.08	1.28	7.80	0.47	7.33
17	474	3230.00	439.00	86.41	2.61	0.36	2.26	14.98	2.04	12.94
19	476	3425.00	521.00	84.79	2.95	0.45	2.50	16.88	2.57	14.31
22	479	3350.00	537.00	83.97	2.53	0.40	2.12	14.47	2.32	12.15
27	484	3950.00	430.00	89.11	3.16	0.34	2.82	18.10	1.97	16.13
Nov 01	489	3575.00	478.00	86.63	2.78	0.37	2.41	15.93	2.13	13.80
03	491	3950.00	450.00	88.61	3.11	0.35	2.76	17.81	2.03	15.78
10	498	3600.00	502.00	86.06	2.74	0.38	2.36	15.68	2.19	13.49
12	500	3725.00	220.00	94.09	2.70	0.16	2.54	15.44	0.91	14.53
14	502	3750.00	212.00	94.35	2.70	0.15	2.54	15.44	0.87	14.57
16	504	3575.00	205.00	94.27	2.66	0.15	2.50	15.21	0.87	14.34
17	505	3425.00	346.00	89.90	2.91	0.29	2.62	16.69	1.69	15.00
18	506	3450.00	378.00	89.04	3.01	0.33	2.68	17.25	1.89	15.36
20	508	3280.00	394.00	87.99	3.94	0.47	3.46	22.54	2.71	19.83
25	513	3350.00	502.00	85.01	3.57	0.54	3.04	20.47	3.07	17.40
27	515	3400.00	428.00	87.41	3.89	0.49	3.40	22.25	2.80	19.45
29	517	3645.00	435.00	88.07	4.63	0.55	4.08	26.51	3.16	23.35
Dec 02	520	3260.00	467.00	85.67	3.81	0.55	3.27	21.84	3.13	18.71
05	523	3115.00	494.00	84.14	4.03	0.64	3.39	23.11	3.66	19.44
07	525	3425.00	453.00	86.77	4.34	0.57	3.77	24.88	3.29	21.59
09	527	3290.00	516.00	84.32	4.11	0.64	3.47	23.54	3.69	19.85
12	530	3220.00	420.00	86.96	4.20	0.55	3.65	24.06	3.14	20.92
14	532	3355.00	498.00	85.16	4.14	0.61	3.53	23.73	3.52	20.21
18	536	3890.00	585.00	84.96	6.61	0.99	5.61	37.84	5.69	32.15
20	538	3710.00	663.00	82.13	6.77	1.21	5.56	38.79	6.93	31.85
24	542	3560.00	572.00	83.93	6.36	1.02	5.34	36.41	5.85	30.56
29	547	3720.00	654.00	82.42	6.50	1.14	5.35	37.20	6.54	30.66
31	549	3905.00	688.00	82.38	7.27	1.28	5.99	41.64	7.34	34.30
4/1/91	553	3700.00	692.00	81.30	6.57	1.23	5.34	37.61	7.03	30.57
06	555	3880.00	645.00	83.38	7.06	1.17	5.89	40.46	6.73	33.73
		3188.18	192.55	93.94	1.16	0.07	1.09	6.62	0.40	6.23
		3613.00	399.40	88.83	2.79	0.31	2.48	15.99	1.79	14.20
		3334.00	460.70	86.15	4.07	0.56	3.51	23.29	3.22	20.08
		3766.43	642.71	82.93	6.73	1.15	5.58	38.56	6.59	31.98

RESULTS FOR UAF STEADY STATE PERFORMANCE (HS LEACHATE)

Date 1990	Day	T O C mg/l infl.	 fil.eff	 % rem	VOLR Kg TOC /m3.d	TOCres Kg TOC /m3.d	TOCrem Kg TOC /m3.d	AOLR g TOC /m2.d	TOCres g TOC /m2.d	TOCrem g TOC /m2.d
Sept 15	442	1034.00	27.00	97.39	0.30	0.01	0.29	1.73	0.05	1.68
17	444	1324.00	53.00	96.00	0.43	0.02	0.41	2.47	0.10	2.37
19	446	1473.00	84.00	94.30	0.62	0.04	0.59	3.58	0.20	3.37
22	449	1463.00	90.00	93.85	0.43	0.03	0.40	2.46	0.15	2.31
24	451	1558.00	72.00	95.38	0.49	0.02	0.47	2.82	0.13	2.69
26	453	1192.00	88.00	92.62	0.39	0.03	0.36	2.23	0.16	2.07
29	456	1375.00	79.00	94.25	0.41	0.02	0.38	2.32	0.13	2.19
Oct 01	458	1517.00	70.00	95.39	0.61	0.03	0.58	3.48	0.16	3.31
03	460	1426.00	87.00	93.90	0.64	0.04	0.60	3.68	0.22	3.46
05	462	1209.00	86.00	92.89	0.38	0.03	0.36	2.20	0.16	2.04
06	463	1311.00	117.00	91.08	0.58	0.05	0.53	3.34	0.30	3.04
08	465	1375.00	104.00	92.44	0.57	0.04	0.52	3.24	0.24	2.99
11	468	1420.00	164.00	88.45	0.47	0.05	0.42	2.71	0.31	2.40
13	470	1561.00	155.00	90.07	0.65	0.06	0.59	3.75	0.37	3.37
15	472	1289.00	148.00	88.52	0.55	0.06	0.49	3.15	0.36	2.79
17	474	1438.00	257.00	82.13	1.16	0.21	0.96	6.67	1.19	5.48
19	476	1475.00	222.00	84.95	1.27	0.19	1.08	7.27	1.09	6.17
22	479	1381.00	363.00	73.71	1.04	0.27	0.77	5.96	1.57	4.40
23	480	1501.00	325.00	78.35	1.10	0.24	0.86	6.30	1.37	4.94
25	482	1387.00	185.00	86.66	1.18	0.16	1.03	6.78	0.90	5.88
27	484	1668.00	145.00	91.31	1.33	0.12	1.22	7.64	0.66	6.98
Nov 01	489	1426.00	204.00	85.69	1.11	0.16	0.95	6.35	0.91	5.44
03	491	1582.00	221.00	86.03	1.25	0.17	1.07	7.13	1.00	6.14
05	493	1494.00	180.00	87.95	1.06	0.13	0.93	6.07	0.73	5.34
10	498	1442.00	246.00	82.94	1.10	0.19	0.91	6.28	1.07	5.21
12	500	1295.00	180.00	86.10	0.94	0.13	0.81	5.37	0.75	4.62
14	502	1743.00	225.00	87.09	1.25	0.16	1.09	7.18	0.93	6.25
16	504	1598.00	232.00	85.48	1.19	0.17	1.01	6.80	0.99	5.81
17	505	1524.00	337.00	77.89	1.30	0.29	1.01	7.43	1.64	5.78
18	506	1572.00	295.00	81.23	1.37	0.26	1.11	7.86	1.47	6.39
20	508	1724.00	345.00	79.99	2.17	0.43	1.73	12.41	2.48	9.93
22	510	1671.00	374.00	77.62	2.01	0.45	1.56	11.48	2.57	8.91
25	513	1868.00	562.00	69.91	1.99	0.60	1.39	11.41	3.43	7.98
27	515	1769.00	446.00	74.79	2.02	0.51	1.51	11.58	2.92	8.66
29	517	1925.00	595.00	69.09	2.44	0.76	1.69	14.00	4.33	9.67
Dec 02	520	1773.00	513.00	71.07	2.07	0.60	1.47	11.88	3.44	8.44
03	521	1916.00	570.00	70.25	2.63	0.78	1.85	15.05	4.48	10.57
07	525	1814.00	566.00	68.80	2.30	0.72	1.58	13.18	4.11	9.07
10	528	1712.00	457.00	73.31	2.14	0.57	1.57	12.25	3.27	8.98
12	530	1670.00	515.00	69.16	2.18	0.67	1.51	12.48	3.85	8.63
14	532	1739.00	505.00	70.96	2.15	0.62	1.52	12.30	3.57	8.73
16	534	2174.00	563.00	74.10	2.82	0.73	2.09	16.13	4.18	11.95
18	536	2045.00	700.00	65.77	3.47	1.19	2.28	19.89	6.81	13.08
20	538	1991.00	826.00	58.51	3.63	1.51	2.13	20.81	8.64	12.18
22	540	2108.00	715.00	66.08	3.49	1.18	2.30	19.97	6.77	13.20
25	543	1926.00	884.00	54.10	3.44	1.58	1.86	19.70	9.04	10.66
27	545	2088.00	1039.00	50.24	3.84	1.91	1.93	22.02	10.96	11.06
31	549	1975.00	966.00	51.09	3.68	1.80	1.88	21.06	10.30	10.76
2/1/91	551	1924.00	990.00	48.54	3.56	1.83	1.73	20.39	10.49	9.90

APPENDIX D

Shock Loading Experimental Data

Table D.1: Results for RBC Volumetric and Organic Shock Loadings

Time	pH value		alkalinity		C O D mg/l			B O D mg/l			T O C mg/l			HRT	VOLR
(hrs)	infl	eff	infl.	eff.	infl.	fil.eff	% rem	infl.	fil.eff	% rem	infl.	fil.eff	% rem	(hrs)	Kg COD /m3.d
Volumetric Shock Loading (Low Strength Leachate)											25th - 26th Sept. 1989				
0.0	7.30	8.00	825	1025	1360.00	236.00	82.65	493.00	47.00	90.47	344.00	93.00	72.97	5.89	5.54
0.5		8.00		1025	1360.00	329.00	75.81	493.00	79.00	83.98	344.00	131.00	61.92	1.79	18.20
3.0		7.95		1005	1360.00	482.00	64.56	493.00	170.00	65.52	344.00	199.00	42.15	1.96	16.62
6.0		7.80		990	1360.00	634.00	53.38	493.00	217.50	55.88	344.00	200.00	41.86	2.05	15.95
9.0		7.85		960	1360.00	608.00	55.29	493.00	282.50	42.70	344.00	249.00	27.62	1.72	18.93
12.0		7.80		995	1360.00	534.00	60.74	493.00	255.00	48.28	344.00	222.00	35.47	1.89	17.23
14.0	7.25	7.90	850	1000	1442.00	479.00	66.78	517.00	229.00	55.71	316.00	204.00	35.44	6.02	5.75
16.0	7.25	7.85	850	990	1442.00	445.00	69.14	517.00	184.00	64.41	316.00	178.00	43.67	6.09	5.68
24.0	7.20	7.95	870	975	1485.00	385.00	74.07	534.00	136.00	74.53	298.00	146.00	51.01	6.00	5.94
33.0	7.20	8.00	915	960	1544.00	320.00	79.27	543.80	91.30	83.21	286.00	119.00	58.39	5.98	6.19
Volumetric Shock Loading (High Strength Leachate)											12th - 14th Sept 1990				
0.0	7.00	8.05	995	1000	1482.00	205.00	86.17	735.00	84.00	88.57	369.00	78.00	78.86	10.70	3.32
0.0	7.00	8.05	995	1025	1482.00	287.00	80.63	735.00	112.00	84.76	369.00	103.00	72.09	2.51	14.19
3.0		8.00		1080	1482.00	508.00	65.72	735.00	182.50	75.17	369.00	137.00	62.87	2.51	14.18
7.0		7.85		1080	1482.00	784.00	47.10	735.00	336.00	54.29	369.00	140.00	62.06	2.50	14.23
11.0		7.90		1050	1482.00	638.00	56.95	735.00	365.00	50.34	369.00	165.00	55.28	2.51	14.16
15.0		7.90		1050	1482.00	620.00	58.16	735.00	337.50	54.08	369.00	135.00	63.41	2.52	14.13
24.0	7.05	7.85	990	1075	1500.00	596.00	60.27	783.00	251.00	67.94	373.00	138.00	63.00	2.52	14.27
31.0	7.05	7.90	990	1070	1500.00	474.00	68.40	783.00	189.00	75.86	373.00	124.00	66.76	10.97	3.28
36.0		7.90		1075	1500.00	392.00	73.87	783.00	133.00	83.01	373.00	115.00	69.17	10.97	3.28
48.0	7.15	7.95	985	1080	1536.00	269.00	82.49	694.00	54.00	92.22	392.00	92.00	76.53	11.43	3.23
Organic Shock Loading (Low Strength Leachate)											24th - 26th March 1990				
0.0	7.10	8.20	1090	1090	1565.00	272.00	82.62	546.00	74.00	86.45	349.00	78.00	77.65	5.21	7.21
0.5		8.10		1090	2500.00	351.00	85.96	950.00	138.00	85.47	548.00	176.00	67.88	5.21	11.52
3.5		7.95		1210	4638.00	544.00	88.27	2065.00	247.50	88.01	1074.00	267.00	75.14	5.21	21.36
6.5		7.90		1330	4638.00	896.00	80.68	2065.00	535.00	74.09	1074.00	393.00	63.41	5.21	21.36
9.5		8.05		1460	4638.00	1060.00	77.15	2065.00	627.50	69.61	1074.00	421.00	60.80	5.21	21.36
12.5		8.05		1215	4638.00	1285.00	72.29	2065.00	780.00	62.23	1074.00	474.00	55.87	5.21	21.36
24.0	7.15	8.10	950	1220	1608.00	578.00	64.05	529.00	142.00	73.16	347.00	133.00	61.67	5.30	7.27
31.5		8.10		1220	1608.00	440.00	72.64	529.00	113.00	78.64	347.00	115.00	66.86	5.30	7.27
48.0		8.00		1210	1536.00	359.00	76.63	494.00	86.00	82.59	372.00	97.00	73.92	5.46	6.75
Organic Shock Loading (High Strength Leachate)											16th - 18th Nov. 1990				
0.0	7.80	8.20	2520	3040	6120.00	1804.00	70.52	3575.00	735.00	79.44	1598.00	545.00	65.89	24.29	6.05
0.5					8460.00	2148.00	74.61	4810.00	842.00	82.49	2165.00	672.00	68.96	23.29	8.72
2.5					13550.00	3561.00	73.72	7820.00	1056.00	86.50	3291.00	846.00	74.29	23.50	13.84
4.5					13550.00	3850.00	71.59	7820.00	1284.00	83.58	3291.00	940.00	71.44	23.50	13.84
7.0					13550.00	4215.00	68.89	7820.00	1748.00	77.65	3291.00	960.00	70.83	23.50	13.84
9.0					13550.00	4693.00	65.37	7820.00	1710.00	78.13	3291.00	1064.00	67.67	23.50	13.84
10.5					13550.00	5028.00	62.89	7820.00	2036.00	73.96	3291.00	1025.00	68.85	23.50	13.84
13.0					9264.00	4236.00	54.27	5180.00	1065.00	79.44	2284.00	797.00	65.11	22.96	9.69
15.0					6270.00	3512.00	43.99	3650.00	864.00	76.33	1673.00	648.00	61.27	22.96	6.56
24.0	7.10	8.15	1920	2820	6125.00	2680.00	56.24	3425.00	728.00	78.74	1524.00	616.00	59.58	23.16	6.35
36.0					6125.00	2249.00	63.28	3425.00	736.00	78.51	1524.00	548.00	64.04	23.16	6.35
48.0	7.20	8.10	2030	2770	6090.00	1957.00	67.87	3450.00	651.00	81.13	1572.00	523.00	66.73	22.56	6.48

Table D.2: Results for UAF Volumetric and Organic Shock Loadings

Time	pH value		alkalinity		Acetate	Propionate	T. VFA	B O D mg/l			T O C mg/l		
(hrs)	infl	eff	infl.	eff.	eff.	eff.	eff.	infl.	fil.eff	% rem	infl.	fil.eff	% rem
				Volumetric Shock Loading (LS Leachate)				25th - 26th Sept. 1989					
0.0	7.30	7.20	825	1160	33.1	38.9	72.3	493.00	38.00	92.29	344.00	72.00	79.07
0.5		7.15		1130	31.5	44.0	77.1	493.00	54.00	89.05	344.00	94.00	72.67
3.0		7.15		1055	29.4	74.5	103.9	493.00	45.00	90.87	344.00	117.00	65.99
6.0		7.05		1045	60.2	11.4	72.2	493.00	87.50	82.25	344.00	143.00	58.43
9.0		7.05		1030	85.3	18.3	192.5	493.00	155.00	68.56	344.00	226.00	34.30
12.0		7.05		1015	83.4	21.9	196.0	493.00	140.00	71.60	344.00	217.00	36.92
14.0	7.25	7.05	850	1040	64.5	21.3	134.7	517.00	124.00	76.02	316.00	193.00	38.92
16.0	7.25	7.10	850	1035	62.8	21.5	126.4	517.00	113.00	78.14	316.00	185.00	41.46
24.0	7.20	7.00	870	1055	47.6	20.4	102.8	534.00	99.00	81.46	298.00	134.00	55.03
33.0	7.20	7.00	915	1070	28.3	19.8	67.3	543.80	91.50	83.17	286.00	119.00	58.39
				Volumetric Shock Loading (HS leachate)				12th - 14th Sept. 1990					
0.0	7.00	6.95	995	1090	72.3	39.5	118.7	735.00	78.00	89.39	369.00	78.00	78.86
0.5		7.00		1100	83.0	41.2	135.6	735.00	104.00	85.85	369.00	96.00	73.98
3.0		6.95		1125	153.8	43.6	259.1	735.00	295.00	59.86	369.00	148.00	59.89
7.0		6.90		1120	187.6	61.3	329.6	735.00	323.00	56.05	369.00	219.00	40.65
11.0		6.95		1095	178.3	57.3	299.9	735.00	379.00	48.44	369.00	193.00	47.70
15.0		6.90		1080	182.4	59.1	303.0	735.00	387.50	47.28	369.00	187.00	49.32
24.0	7.00	6.95	990	1100	169.0	50.6	264.6	783.00	276.00	64.75	373.00	161.00	56.84
31.0	7.05	7.00	990	1140	135.9	42.5	226.4	783.00	238.00	69.60	373.00	142.00	61.93
36.0		7.00		1165	86.5	36.4	168.3	783.00	154.00	80.33	373.00	126.00	66.22
48.0	7.15	7.05	985	1235	0.0	29.2	32.2	694.00	87.00	87.46	392.00	95.00	75.77
				Organic Shock Loading (LS Leachate)				24th - 26th March 1990					
0.0	7.10	7.10	1090	1160	23.3	46.4	73.3	546.00	55.00	89.93	349.00	87.00	75.07
0.5		7.10		1160	22.5	50.2	75.4	950.00	83.00	91.26	548.00	131.00	76.09
3.5		7.25		1200	25.7	62.5	90.6	2065.00	152.00	92.64	1074.00	229.00	78.68
6.5		7.20		1240	30.0	27.9	84.1	2065.00	242.00	88.28	1074.00	261.00	75.70
9.5		7.15		1440	34.5	25.4	83.5	2065.00	327.50	84.14	1074.00	291.00	72.91
12.5		7.45		1470	26.2	19.7	53.2	2065.00	272.00	86.83	1074.00	285.00	73.46
31.5	7.15	7.10	950	1220	23.3	16.1	60.9	529.00	102.00	80.72	347.00	101.00	70.89
48.0		7.15		1205	24.8	18.3	56.8	494.00	87.00	82.39	372.00	105.00	71.77
				Organic Shock Loading (HS Leachate)				16th - 18th Nov. 1990					
0.0	7.80	7.60	2520	3420	161.1	86.6	251.2	3575.00	205.00	94.27	1598.00	232.00	85.48
0.5					193.6	62.5	285.4	4810.00	286.00	94.05	2165.00	285.00	86.84
2.5					275.9	128.3	437.7	7820.00	517.00	93.39	3291.00	422.00	87.18
4.5					493.8	194.8	724.3	7820.00	674.00	91.38	3291.00	534.00	83.77
7.0					477.2	326.5	838.0	7820.00	815.00	89.58	3291.00	560.00	82.98
9.0		7.10			368.5	391.2	786.5	7820.00	752.00	90.38	3291.00	684.00	79.22
10.5					231.6	403.1	675.9	7820.00	794.00	89.85	3291.00	583.00	82.29
13.0					142.0	284.2	455.8	5180.00	601.00	88.40	2284.00	475.00	79.20
15.0					116.7	167.9	327.4	3650.00	487.00	86.66	1673.00	436.00	73.94
24.0	7.10	7.30	1920	3380	179.3	124.5	340.2	3425.00	346.00	89.90	1524.00	337.00	77.89
36.0					156.9	133.0	316.5	3425.00	328.00	90.42	1524.00	349.00	77.10
48.0	7.20	7.35	2030	3190	204.6	152.3	384.7	3450.00	278.00	91.94	1572.00	295.00	81.23

Table D.3: Results for UAF Volumetric and Organic Shock Loadings

C O D mg/l infl.	fil.eff	% rem	biogas (l/h)	CH4 (%)	CO2 (%)	CH4 (l/h)	Q (l/d)	HRT (hrs)	VOLR Kg COD /m3.d	CODres Kg COD /m3.d	CODrem Kg COD /m3.d	EMP m3 CH4/ Kg CODred
Volumetric Shock Loading (LS Leachate) 25th - 26th Sept. 1989												
1360.00	223.00	83.60	0.151	90.20	9.80	0.136	8.12	18.62	1.75	0.29	1.47	0.354
1360.00	264.00	80.59	0.215	89.90	10.10	0.193	32.26	4.69	6.96	1.35	5.61	0.131
1360.00	292.00	78.53	0.238	90.40	10.60	0.215	31.54	4.79	6.81	1.46	5.35	0.153
1360.00	418.00	69.26	0.250	88.70	11.30	0.222	31.50	4.80	6.80	2.09	4.71	0.179
1360.00	544.00	60.00	0.244	89.00	11.00	0.217	32.51	4.65	7.02	2.81	4.21	0.196
1360.00	532.00	60.88	0.255	87.80	12.60	0.224	32.23	4.69	6.96	2.72	4.24	0.201
1442.00	503.00	65.12	0.174	85.60	14.40	0.149	8.10	18.67	1.85	0.65	1.21	0.470
1442.00	465.00	67.75	0.162	86.30	13.70	0.140	8.04	18.81	1.84	0.59	1.25	0.427
1485.00	391.00	73.67	0.150	87.00	13.00	0.131	8.24	18.35	1.94	0.51	1.43	0.347
1544.00	356.00	76.94	0.146	86.50	13.50	0.126	8.29	18.24	2.03	0.47	1.56	0.307
Volumetric Shock Loading (HS leachate) 12th - 14th Sept. 1990												
1482.00	298.00	79.89	0.219	89.60	10.40	0.196	10.69	14.14	2.51	0.51	2.01	0.372
1482.00	375.00	74.70	0.287	82.50	10.30	0.237	39.84	3.80	9.37	2.37	7.00	0.129
1482.00	606.00	59.11	0.324	74.30	8.50	0.241	38.77	3.90	9.12	3.73	5.39	0.170
1482.00	612.00	58.70	0.360	79.40	6.40	0.286	40.32	3.75	9.48	3.92	5.57	0.196
1482.00	750.00	49.39	0.400	86.10	5.80	0.344	39.60	3.82	9.32	4.71	4.60	0.285
1482.00	742.00	49.93	0.400	84.90	7.40	0.340	40.50	3.73	9.53	4.77	4.76	0.272
1500.00	672.00	55.20	0.405	83.90	6.60	0.340	40.32	3.75	9.60	4.30	5.30	0.244
1500.00	543.00	63.80	0.245	87.00	13.00	0.213	10.67	14.17	2.54	0.92	1.62	0.501
1500.00	468.00	68.80	0.216	88.50	11.50	0.191	10.62	14.24	2.53	0.79	1.74	0.419
1536.00	357.00	76.76	0.188	88.10	11.90	0.166	10.50	14.40	2.56	0.60	1.97	0.321
Organic Shock Loading (LS Leachate) 24th - 26th March 1990												
1565.00	309.00	80.26	0.206	91.70	8.30	0.189	10.70	14.13	2.66	0.52	2.13	0.337
2500.00	335.00	86.60	0.216	91.70	8.30	0.198	10.70	14.13	4.25	0.57	3.68	0.205
4638.00	733.00	84.20	0.240	91.00	9.00	0.218	10.70	14.13	7.88	1.24	6.63	0.125
4638.00	840.00	81.89	0.297	89.80	10.20	0.267	10.70	14.13	7.88	1.43	6.45	0.158
4638.00	964.00	79.22	0.312	85.90	14.10	0.268	10.70	14.13	7.88	1.64	6.24	0.164
4638.00	868.00	81.29	0.334	84.70	13.30	0.283	10.70	14.13	7.88	1.47	6.40	0.168
1608.00	482.00	70.02	0.237	88.50	11.50	0.210	10.52	14.37	2.69	0.80	1.88	0.425
1536.00	375,00	75.59	0.188	90.70	9.30	0.171	10.31	14.67	2.51	0.61	1.90	0.342
Organic Shock Loading (HS Leachate) 16th - 18th Nov. 1990												
6120.00	1476.00	75.88	0.351	86.40	13.60	0.303	4.68	32.31	4.55	1.10	3.45	0.335
8460.00	1985.00	76.54	0.524	86.20	13.80	0.452	5.15	29.36	6.92	1.62	5.29	0.325
13550.00	3125.00	76.94	0.772	84.10	15.90	0.649	5.30	28.53	11.40	2.63	8.77	0.282
13550.00	3592.00	73.49	0.765	83.30	16.70	0.637	5.30	28.53	11.40	3.02	8.38	0.290
13550.00	3540.00	73.87	0.758	82.60	17.40	0.626	5.30	28.53	11.40	2.98	8.42	0.283
13550.00	3764.00	72.22	0.786	80.40	19.60	0.632	5.30	28.53	11.40	3.17	8.23	0.292
13550.00	3451.00	74.53	0.770	78.50	21.50	0.604	5.30	28.53	11.40	2.90	8.50	0.271
9264.00	3183.00	65.64	0.583	82.00	18.00	0.478	5.24	28.85	7.71	2.65	5.06	0.360
6270.00	2025.00	67.70	0.459	85.10	14.90	0.391	5.24	28.85	5.22	1.68	3.53	0.421
6125.00	1676.00	72.64	0.433	85.60	14.40	0.371	5.36	28.21	5.21	1.43	3.79	0.373
6125.00	1682.00	72.54	0.440	86.00	14.00	0.378	5.36	28.21	5.21	1.43	3.78	0.381
6090.00	1595.00	73.81	0.448	84.90	15.10	0.380	5.50	27.49	5.32	1.39	3.92	0.369

APPENDIX E

Procedure for Adsorption Isotherm Study - Carbon Dosage

For meaningful results, Wagner and Jula (1981) recommended at least nine dosages of carbon i.e 0.05, 0.1, 0.2, 0.5, 1, 2, 5, 10 and 20 g.

The granular activated carbon (GAC) was first pulverised and then oven dry. The required dosages were weighed out and transferred into suitable copntainers. Appropriate volume of sample (100 ml) was then added into each container. The containers were agitated for a required contact time, after which the samples were filtered and analyzed for COD and TOC remaining in the solution.

The results obtained are tabulated in Tables E.2 and E.3.

Table E.1: Results for Contact Time - Adsorption rate Study 6/2/91

pH	7.8		6.6		5.0	
Time	COD	Ct/Co	COD	Ct/Co	COD	Ct/Co
0	1386	1.000	1422	1.000	1348	1.000
15	1245	0.898	1266	0.890	1196	0.887
30	1159	0.836	1168	0.821	1114	0.826
60	1164	0.840	1145	0.805	1058	0.785
90	1074	0.775	1031	0.725	983	0.729
120	1047	0.755	1014	0.713	940	0.697
180	1014	0.732	966	0.679	895	0.664
240	951	0.686	908	0.639	828	0.614
360	923	0.666	872	0.613	794	0.589
480	915	0.660	860	0.605	782	0.580

pH	7.8		6.6		5.0	
Time	TOC	Ct/Co	TOC	Ct/Co	TOC	Ct/Co
0	475	1.000	482	1.000	469	1.000
15	423	0.891	420	0.871	407	0.868
30	394	0.829	384	0.797	378	0.806
60	398	0.838	386	0.801	374	0.797
90	367	0.773	362	0.751	352	0.751
120	368	0.775	355	0.737	346	0.738
180	356	0.749	341	0.707	335	0.714
240	347	0.731	335	0.695	310	0.661
360	325	0.684	310	0.643	295	0.629
480	327	0.688	303	0.629	288	0.614

Table E.2:

RESULTS FOR ADSORPTION ISOTHERM (LS LEACHATE)

Isotherm Data - COD

M wt of carbon g/100 ml sample	Co concentration of adsorbate in solution (mg/l)	C constituent remaining in solution (mg)	X adsorbate adsorbed (mg)	X/M adsorbate adsorbed/ g carbon (mg/g)
0	1825	182.5		
0.05	1676	167.6	14.9	298
0.1	1545	154.5	28	280
0.2	1380	138	44.5	222.5
0.5	1185	118.5	64	128
1	950	95	87.5	87.5
2.5	820	82	100.5	40.2
5	588	58.8	123.7	24.74
10	533	53.3	129.2	12.92
20	465	46.5	136	6.8

Isotherm Data - TOC

M wt of carbon g/100 ml sample	Co concentration of adsorbate in solution (mg/l)	C constituent remaining in solution (mg)	X adsorbate adsorbed (mg)	X/M adsorbate adsorbed/ g carbon (mg/g)
0	513	51.3		
0.05	478	47.8	3.5	70
0.1	445	44.5	6.8	68
0.2	409	40.9	10.4	52
0.5	350	35	16.3	32.6
1	324	32.4	18.9	18.9
2.5	252	25.2	26.1	10.44
5	200	20	31.3	6.26
10	149	14.9	36.4	3.64
20	126	12.6	38.7	1.935

Table E.3:

RESULTS FOR ADSORPTION ISOTHERM (HS LEACHATE)

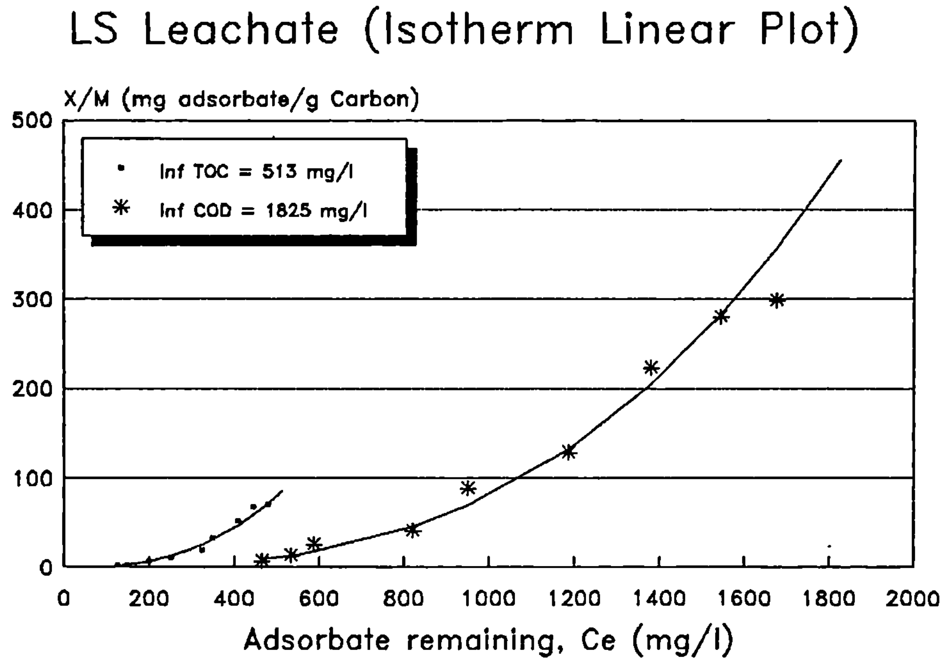
Isotherm Data - COD

Date 8/2/91

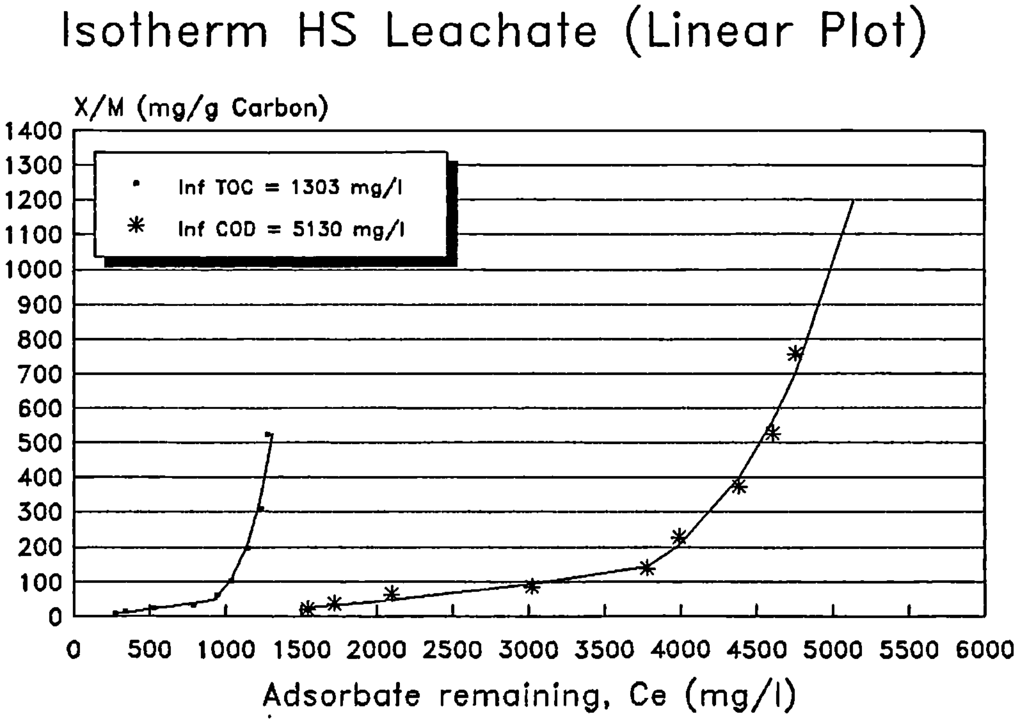
M wt. of carbon g/100 ml sample	Co concentration of adsorbate in solution (mg/l)	C constituent remaining in solution (mg)	X adsorbate adsorbed (mg)	X/M adsorbate adsorbed/ g carbon (mg/g)
0	5130	513		
0.05	4752	475.2	37.8	756
0.1	4605	460.5	52.5	525
0.2	4382	438.2	74.8	374
0.5	3990	399	114	228
1	3780	378	135	135
2.5	3020	302	211	84.4
5	2100	210	303	60.6
10	1720	172	341	34.1
20	1540	154	359	17.95

Isotherm Data - TOC

0	1303	130.3		
0.05	1273	127.3	26.1	522
0.1	1225	122.5	30.9	309
0.2	1146	114.6	38.8	194
0.5	1037	103.7	49.7	99.4
1	945	94.5	58.9	58.9
2.5	789	78.9	74.5	29.8
5	527	52.7	100.7	20.14
10	337	33.7	119.7	11.97
20	272	27.2	126.2	6.31

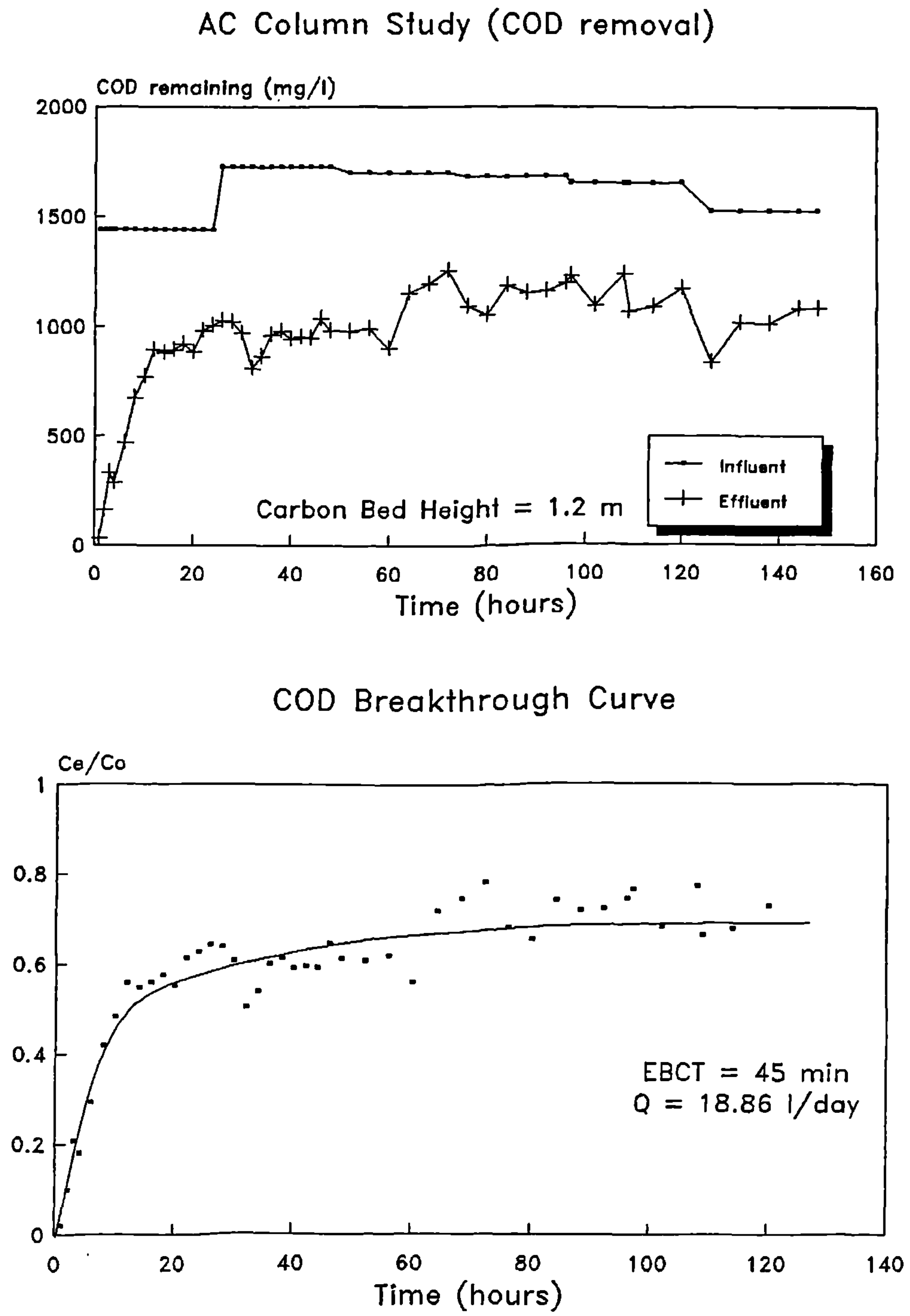


LS Leachate (Isotherm Linear Plot)



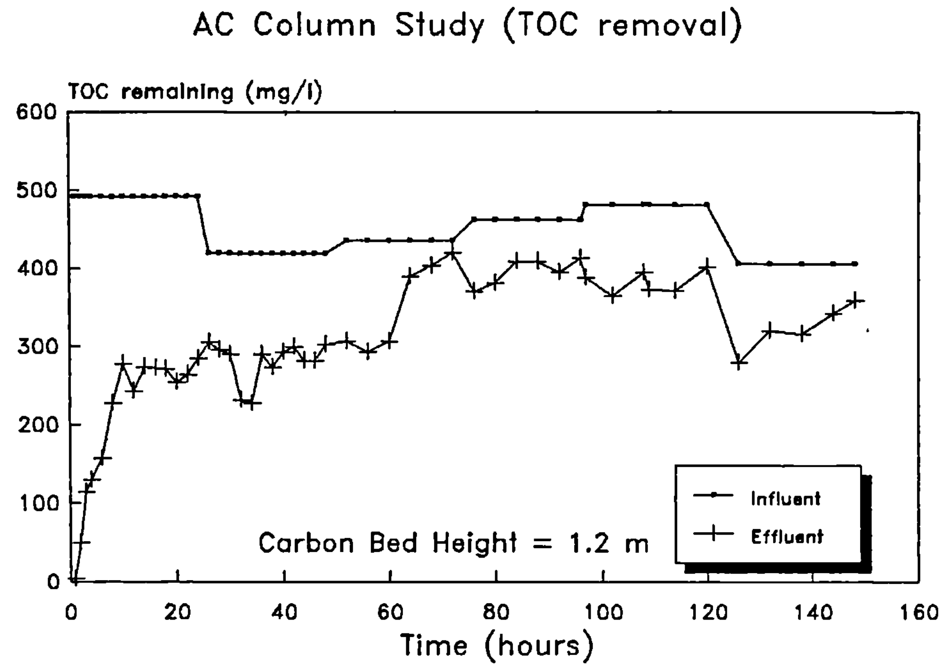
Isotherm HS Leachate (Linear Plot)

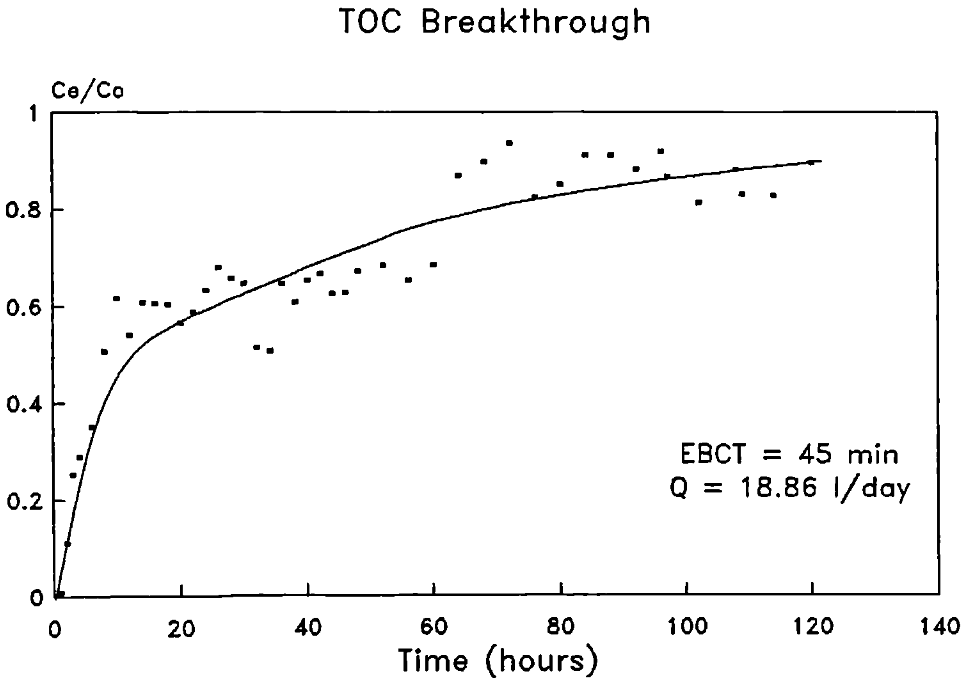
Figure E.1: Adsorption Isotherm (Linear Plot)



EBCT = Empty Bed Contact Time

Figure E.2: AC Breakthrough Study for COD Removal (Bed Height = 1.2 m)





EBCT = Empty Bed Contact Time

Figure E.3: AC Breakthrough Study for TOC Removal (Bed Height = 1.2 m)