EFFECT OF POWDERED ACTIVATED CARBON ADDITION

ON SUBSTRATE REMOVAL AND SLUDGE CHARACTERISTICS IN

LEACHATE TREATMENT

by

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ABSTRACT

The performance of activated sludge process depends on substrate removal efficiency and a good separation of biomass from the supernatant. Powdered Activated Carbon (PAC) addition to activated sludge is a promising method in the treatment of wastewater containing non-biodegradable and inhibitory substances. Substrate removal efficiency can be enhanced via addition of PAC to activated sludge systems in the treatment of leachate combined with domestic wastewater. The increase in substrate removal is due to adsorption of non-biodegradable and inhibitory and/or toxic compounds in the leachate. Moreover, addition of PAC has an effect on settling and dewatering characteristics of activated sludge. The purpose of this study was to investigate the effect of powdered activated carbon (PAC) addition on substrate removal and sludge characteristics in the co-treatment of leachate and domestic wastewater.

In the first part of the experimental path of the study, leachate and domestic wastewater were mixed at different ratios and PAC was added at various concentrations to batch reactors. Also, a control reactor was used which was fed under the same conditions as the PAC added reactor. Results showed that leachate could be treated successfully even if the leachate ratios was high (20-25 %) in the activated sludge system. The inert COD in PAC added activated sludge systems was lower than in activated sludge reactors. Results of the study also showed that the Specific Resistance to Filtration (SRF) increased during removal of readily biodegradable COD and then tended to decline. Moreover, the SRFs of the sludges were higher if the leachate ratio increased. This showed that the dewaterability of the sludge decreased when the leachate ratio in the feed increased. As a general trend, in most of the runs, PAC addition suppressed these SRF increases.

In the next step, leachate (diluted) and domestic wastewater were treated alone. The aim was to have an idea about the effect of leachate and domestic wastewater in the case of combined treatment. In the case of diluted leachate, although COD removal was considerable, sludge characteristics were negatively affected. PAC addition to leachate had a positive effect. The effect of PAC was most obvious in the case of diluted leachate

showing that leachate constituents were highly adsorbable onto activated carbon. The presence of domestic wastewater always resulted in a decrease in SRF or an increase in dewaterability.

In the treatment of domestic wastewater alone, very high COD removal was observed and PAC addition did not result in additional COD removal. The SRF values of this sludge were much lower compared to leachate and co-treatment.

ÖZET

Aktif çamur metodunun başarısı organik madde giderimine ve çamurun arıtılmış atıksudan iyi bir şekilde ayrılmasına bağlıdır. Aktif çamura Toz Aktif Karbon eklenmesi özellikle biyolojik olarak ayrıştırılması zor olan organik maddeler içeren atıksuların arıtılmasında etkili bir uygulamadır. Çöp sızıntı suyu ve evsel atıksuların beraber arıtıldığı bir sistemde, Toz Aktif Karbon eklenmesiyle birlikte organik madde giderimi artar. Bu artış, biyolojik olarak ayrıştırılması zor maddelerin ve biyolojik arıtmada inhibisyona neden olan maddelerin aktif karbon üzerine adsorbe olmaları nedeniyledir. Bunlara ek olarak Toz Aktif Karbon ilavesi, çamurun çökmesine ve susuzlaştırılmasına da etki etmektedir. Bu çalışmanın amacı, Toz Aktif Karbonun organik madde giderimine ve çamur özelliklerine olan etkisini evsel ve çöp sızıntı suyunun birlikte arıtıldığı bir sistemde incelemektir.

Deneylerde kesikli reaktörler kullanılmıştır. Tüm deneylerde Toz Aktif Karbon ilavesi yapılan reaktörlerin yanı sıra aynı şartlarda bir aktif çamur reaktörü kontrol reaktörü olarak bulunmaktadır. Deneysel çalışmanın ilk kademesinde çöp sızıntı suyu ve evsel atıksu çeşitli oranlarda karıştırılarak, değişik Toz Aktif Karbon dozları eklenmiştir. Çalışmanın sonuçları, çöp sızıntı suyu oranının yüksek olması halinde bile arıtılabildiğini göstermiştir. Toz Aktif Karbon eklenen reaktörlerde arıtılamayan organik madde miktarı, Toz Aktif Karbon eklenmeyen reaktörlere göre daha azdır. Ayrıca sonuçlar, çamur özelliğinin çabuk ve kolay tüketilen maddelerin giderilmesi sırasında kötüleştiğini (Çamurun Özgül Direncinin arttığını) ve daha sonra tekrar iyileştiğini göstermiştir. Çamur özellikleri sızıntı suyu oranları ile bağlantılıdır ve yüksek sızıntı suyu oranlarında çamur çökmesi ve susuzlaştırılması bakımından daha kötü bir çamur elde edilmiştir.

Deneylerin ikinci ve üçüncü bölümlerinde sadece seyreltilmiş çöp sızıntı suyu ve sentetik evsel atıksu kullanılmıştır. Bu deneyler, çöp sızıntı suyu ve evsel atıksuyun birlikte arıtılmaları halinde her ikisinin rolünü ayrı ayrı görmek için yapılmıştır. Seyreltilmiş çöp sızıntı suyunun arıtılmasında, organik madde giderimi iyi olmasına rağmen çamur özellikleri kötü bir şekilde etkilenmiştir. Toz Aktif Karbonun organik madde giderimine etkisi seyreltilmiş çöp sızıntı suyunun arıtımında iyi bir şekilde gözlenmiştir. Çünkü sızıntı

suyu aktif karbon üzerine adsorbe olabilen maddeler içermektedir. Ortamda evsel atıksuyun bulunması her zaman çamur özelliklerinin iyileşmesine yol açmıştır.

Evsel atıksuyun arıtılmasında çok yüksek organik madde gideriminin sağlanmış ve Toz Aktif Karbon ek bir organik madde giderimine yol açmamıştır. Çamur özellikleri diğer alternatiflerle karşılaştırıldığında çok daha iyidir.

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BOD_5 : 5 days Biochemical Oxygen Demand, mg/l

CFSTR   : Continuous-Flow Stirred Tank Reactor

COD   : Chemical Oxygen Demand, mg/l

F/M   : Food to microorganism ratio, mg COD/mg MLVSS.h

GAC   : Granular Activated Carbon

k   : Zero-order substrate removal rate, mg COD/l.h

MLSS   : Mixed Liquor Suspended Solids, mg/l

MLVSS   : Mixed Liquor Volatile Suspended Solids, mg/l

NO_x-N   : NO_2-N + NO_3-N, mgN/l

OUR   : Oxygen Uptake Rate, mg O_2/l.h

q   : Zero-order specific substrate removal rate, mg COD/mg MLVSS.h

PAC   : Powdered Activated Carbon

r_{su} : Substrate removal rate, mg COD/l.h

S   : Substrate concentration, mg/l

S_i : Inert substrate concentration, mg/l

S_o : Initial substrate concentration, mg/l

SCFB   : Semi-Continuously Fed Batch

SCOD   : Soluble COD (COD of filtrated sample), mg/l

TCOD   : Total COD (COD of unfiltrated sample), mg/l

TKN   : Total Kjeldahl Nitrogen, mgN/l

VSS   : Volatile Suspended Solids, mg/l

TSS   : Total Suspended Solids, mg/l

X   : Average biomass concentration, mg MLVSS/l

k'   : First-order substrate removal rate, h^{-1}

q'   : First-order specific substrate removal rate, l/mg MLVSS.h

θ : Hydraulic retention time, h

θ_c : Solids retention time, days

1. INTRODUCTION

Sanitary landfilling is one of the most applied alternatives in the disposal of solid wastes. However, the production of leachate, is a disadvantage of this alternative due to the fact that it may cause pollution of both groundwater, and soil. Moreover, treatment of landfill leachates is problematic due to high organics strength, variability in both quality and quantity.

Treatment of leachate can be achieved by a number of methods. Among the treatment alternatives, activated sludge may be appropriate for a young leachate which is mainly composed of biodegradable organics. The most important drawback of activated sludge is the failure due to toxic and inhibitory substances coming from leachate and inefficient removal of refractory organics. Moreover, severe changes in sludge characteristics are also encountered in activated sludges treating landfill leachates.

Some of problems stemming from leachate can be overcome by the co-treatment of leachate and domestic wastewaters. The presence of domestic wastewater in the influent suppresses the negative effects of leachate and also supplies enough nutrients for biological treatment. The extent of the negative effect is strongly related to the leachate ratio, biodegradability of leachate and presence of toxic and inhibitory substances. With a proper leachate ratio, leachate can be treated in activated sludge with a considerable efficiency.

PAC addition to an activated sludge reactor is especially used for wastewaters containing inhibitory and toxic matters. With the adsorption of toxic and inhibitory substances on activated carbon, the negative effects of leachate can be overcome. In addition to this, in PAC added activated sludge systems, non-biodegradable organics can also be removed, which are not removed in a conventional activated sludge. Moreover, increased nitrification is also observed in the case of PAC addition to activated sludge. Besides substrate removal, sludge characteristics are also influenced by the presence of PAC. In the case of PAC a decrease in SVI (Sludge Volume Index) and an improved dewaterability are observed. The aim of this study was to investigate the substrate removal and sludge characteristics in the co-treatment of leachate and domestic wastewater.

Experiments were conducted in lab-scale batch reactors. Different leachate ratios were employed to determine the optimum leachate ratio for biological treatment. PAC was added to the reactors at dosages ranging from 1000 to 3000 mg/L. Both substrate removal, sludge dewaterability and settling were thoroughly studied.

2. LITERATURE SURVEY

2.1 Leachate

Leachate may be termed as the water passed through waste materials in a landfill. Many chemical and biological reactions as well as physical processes such as sorption and diffusion occur while movement of percolating water and as a result leachate is highly polluted by extracting dissolved and suspended materials. Major environmental problems arise since leachate contaminates surface and groundwater.

2.1.1 Factors Affecting Leachate Composition

Chemical composition and quantity of the leachate depends on numerous factors. These interacting factors can be summarised as type and depth of solid waste, age of fill, climate, hydrogeologic, chemical and biological activities, moisture content, temperature, pH, and degree of the stabilisation.

Hydrogeologic conditions in the landfill affect leachate generation by influencing water entering the landfill. Important factors that govern the rate of percolation are precipitation, surface run-off and soil moisture storage [1]. Depending upon surface slope, the degree of the permeability of the soil cover and the extent and type of the vegetation, a portion of precipitation leaves the site before reaching solid waste deposited by surface run-off while a portion infiltrates to the site. Moreover, a portion of infiltrated water is lost either by evaporation directly to atmosphere or by transpiration to atmosphere through vegetative cover. Actually, once percolation takes place, it affects leachate generation, but a delay is encountered. At the beginning of the filling, although water infiltrates into the landfill, moisture is stored by the soil and solid waste and no leachate is produced. After field capacity of the landfill is exceeded, leachate generation initiates.

Climate is a very important parameter in leachate generation due to the fact that the rate of precipitation falling onto landfill strongly depends on the climate of the landfill site. However, depending upon site topography and final cover material, leachate generation may be enhanced or delayed.

In addition to climatic effects, waste conditions such as availability of moisture, temperature, age and composition of the refuse, availability of oxygen also contribute to leachate generation. Leachate composition is greatly affected by the solid waste composition and extent of the refuse processing, such as shredding and balling. Researches indicate that shredded solid waste resulted in higher concentrations of pollutants than unshredded solid waste. The rate of pollutant removal, solid waste decomposition rate and the cumulative mass of pollutants released per unit volume of leachate is significantly increased when compared with unshredded fills. Conversely, balled solid waste produces a large volume of dilute leachate with a longer period of stabilization than required for balled refuse.

The depth of the refuse has also significant effects on leachate composition. In the case of deep fills, water has a greater contact time with solid waste, hence, concentrated leachates are obtained from deeper fills operating under similar conditions. However, an elevated stabilisation time of the refuse is also encountered.

The temperature of the refuse influences leachate by increasing the rate of bacterial growth and the refuse decomposition. Increased temperatures also favour the solubility of most salts and affect the rate of biochemical reactions.

The rate of moisture addition plays an important role in leachate generation by affecting solute dissolution and microbial decay. At high infiltration rates, soluble organics and even microbial cells may be flushed from the refuse, however, at low flow rates, microbial activity has a significant effect on leachate generation. Co-disposal of municipal solid waste and sludge from municipal wastewater treatment plant also enhanced leachate generation due to increased moisture, microbial seeding and nutrient enrichment. Enhanced methanogenic activity is also encountered. Studies indicate that co-disposal posed no more a pollution threat than the leachate from the municipal landfill [1].

Contrary to the addition of sludge, co-disposal of sorptive wastes, such as incinerator ash, fly ash, kiln dust, limestone lead to a reduction in hazardous chemicals in the leachate. This may be due to adsorption and sorption of metallic ions, formation of less soluble calcium and carbonate compounds and raising of pH may cause precipitation of metals [1].

One of the most important factors influencing leachate composition is the age of the fill, in other words, the degree of the stabilisation. Although the concentration of various leachate constituents varies considerably from leachate to leachate, it is found that landfill age is the most relevant factor affecting leachate composition.

2.1.2 Decomposition of Solid Waste and Leachate Generation

Decomposition of solid waste placed in the landfill takes place in three stages. Firstly, aerobic decomposition predominates because of the oxygen entrained at the time of refuse placement. This aerobic decomposition lasts only few days due to the fact that oxygen supply is used up after refuse placement. In this phase, temperature of the landfill is raised. Temperature increase can be much higher when moisture content of solid waste is also high.

Consumption of oxygen results in the initiation of anaerobic processes. During this phase, carbon dioxide and large amounts of volatile acids (mainly acetic acid) are produced. Large amounts of organic materials dissolved in the percolating water contribute to the high organics concentration in the leachate as measured by high BOD and COD. As a consequence of dissolved CO_2 and volatile fatty acids, the pH of the leachate is reduced to between 4 and 5, in turn this situation helps to dissolve inorganic materials (Cl^-, SO_4^{2-}, Ca^{2+}, Mg^{2+}, and Na^+). Therefore, leachate has a high specific conductance and a negative redox potential. Ammonia concentration is also high at this stage due to hydrolysis and fermentation of protein containing compounds.

The third phase of decomposition takes place when methanogens build up, which convert acetic acid and hydrogen to methane and carbon dioxide. Since methanogens utilise volatile acids, volatile fatty acids are diminished and therefore a pH rise up to 7-8 occurs. This pH increase stimulates methane production and also because of neutral pH fewer inorganic constituents are solubilized and conductivity falls. With the production of landfill

gases, COD and BOD values of leachate are reduced because much of the COD and BOD materials are being converted to gas [2]. Leachate characteristics in acidic and methanogenic phases obtained from various landfills are shown in Table 2.1.

2.1.3 Characteristics of Landfill Leachate

Landfill leachates from municipal solid waste dumping sites have generally higher concentrations of both organic and inorganic chemicals than municipal and industrial wastewater. The amount, types and production rates of contaminants present in the leachate depend on numerous factors, including solid waste type and composition, depth, moisture content of refuse, and temperature. Organic constituents (represented as COD, BOD, and TOC) show a trend of peak in early stages (within 2-3 years) and decline over time [2]. Other constituents (iron, zinc, phosphate, chloride, sodium, copper, organic nitrogen, total solids and suspended solids) increase to a plateau and tend to remain at a relatively constant level for a lengthy period of time, and exhibit steady decreases in concentration. The mechanisms of reduction of these organic and inorganic constituents are also different. Inorganic compounds in leachate decrease due to flushing of refuse bed, in other words dilution by infiltrated water. However, organic constituents decrease more rapidly due to decomposition and washout. On the other hand, fluctuations in heavy metal concentrations occur because of dissolution, adsorption, complexation mechanisms that may retain or mobilize the metals within the landfill microenvironment.

Leachate quality shows a large variation. However, it is found that the age of the fill has a significant effect on the composition of leachate. Young and old leachates have different characteristics, young leachates consist of readily biodegradable organic matter, mainly volatile fatty acids. COD and BOD concentrations and the BOD_5/COD ratio (in the range of 0,4 - 0,8) are high. With time, the volatile fatty acid fraction decreases by production of landfill gasses and poorly biodegradable organics (humic and fulvic acid) remain in leachate. The decrease of volatile fatty acids results in a decrease in the BOD_5/COD ratio and BOD and COD concentration. However, NH_3-N concentration increases as landfill ages. Such a leachate cannot be treated by biological processes but by physical and

TABLE 2.1 Leachate characteristics (parameters with differences between acetic and methanogenic phase) [3].

Acetic phase	average	range
pH	6.1	4.5-7.5
BOD_5 (mg/L)	13000	4000-40000
COD (mg/L)	22000	6000-60000
BOD_5/COD	0.58	
SO_4 (mg/L)	500	70-1750
Ca (mg/L)	1200	10-2500
Mg (mg/L)	470	50-1150
Fe (mg/L)	780	20-21000
Mn (mg/L)	25	0.3-65
Zn (mg/L)	5	0.1-120

Methanogenic Phase

	average	range
pH	8	7.5-9.0
BOD_5 (mg/L)	180	20-550
COD (mg/L)	3000	500-4500
BOD_5/COD	0.06	
SO_4 (mg/L)	80	10-420
Ca (mg/L)	60	20-600
Mg (mg/L)	180	40-350
Fe (mg/L)	15	3-280
Mn (mg/L)	0.7	0.03-45
Zn (mg/L)	0.6	0.03-4

(No differences between phases could be observed)

	average	range
Cl (mg/L)	2100	100-5000
Na (mg/L)	1350	50-400
K (mg/L)	1100	10-2500
Alkalinity (mg $CaCO_3$/L)	6700	300-11500
NH_4-N (mg/L)	750	30-3000
Org N (mg/L)	600	10-4250
Total N (mg/L)	1250	50-5000
NO_3-N (mg/L)	3	0.1-50
NO_2-N (mg/L)	0.5	0-25
Total P (mg/L)	6	0.1-30
AOX * (μg/L)	2000	320-3500
As (μg/L)	160	5-1600
Cd (μg/L)	6	0.5-140
Co (μg/L)	55	4-950
Ni (μg/L)	200	20-250
Pb (μg/L)	90	8-1020
Cr (μg/L)	300	30-160
Cu (μg/L)	80	4-1400
Hg (μg/L)	10	0.2-50

*adsorbable organic halogen

chemical processes [2]. With aging of landfill, pathogenic and viral population is also deactivated. This deactivation process occurs due to adverse conditions such as initial temperature rise, low pH levels, and presence of chemicals that are generally associated with solids decomposition processes.

One of the important features of leachate is the phosphorous deficiency. Since an optimum ratio of BOD: P of 100:1 is widely recommended for biological treatment processes [4] , biological treatment of leachate is inhibited due to phosphorous deficiency unless phosphorous is added. The pH of leachate is low during the acidic phase of decomposition. This affects participation and dissolution of metals, redox and sorption reactions in the waste matrix. Therefore, neutralisation is required for the treatment of young leachates.

In addition to temporal variations in the concentration, the flow variations of leachate are also significant. Leachate production rates change in response to changes in precipitation rates. The highest leachate generation usually occurs during winter and spring months, when temperatures are lowest. In this case, the low temperature inhibits the biological treatment of leachate [2].

2.2. Adsorption

Adsorption is a significant phenomenon in most natural physical, biological and chemical processes. The process has gained importance in both chemical industry, energy utilisation and in the fields of environmental pollution.

Adsorption involves the interphase accumulation or concentration of substances at a surface or interface. The process can occur between liquid-liquid, gas-solid and liquid-solid interfaces. The material being concentrated or adsorbed is termed adsorbate and the adsorbing phase is termed the adsorbent.

2.2.1. The Relationship between Surface Tension and Adsorption

Adsorption is a surface phenomenon; as such the adsorbate is transferred from solution and concentrated at a surface of the adsorbent. Hence, the forces active within the phase boundaries or surface boundaries influence and alter such surface reactions.

When a pure liquid is placed on a flat plate, the drop will resist spreading and tend to retain a nearly spherical shape by minimising surface area. The reason of the development of surface tension, which tends to keep the drop from spreading, is the attractive forces between molecules of liquid within the drop. Enlarging a surface of the drop requires an input of work in the excess of the necessary to compensate the tension at the surface.

A large number of soluble materials are known as having ability to alter the surface tension of liquid. For example, detergents can lower surface tension dramatically and can thus act to cause spreading of the water drop on the flat plate. These materials are termed "wetting" agents or "surface-active" agents. When a surface-active materials are present in the liquid system, a decrease in tension at the surface will occur upon movement of the solute to the surface.

2.2.2. Causes and Types of Adsorption

Adsorption in a solid-liquid system occurs a consequence of lyophobic (solvent-disliking) character of solute relative to the particular solvent, or of high affinity of solute for the solute for the solid. One or the combination of these forces may play important role in the adsorption process.

A hydrophilic substance will be less likely to be adsorbed on a solid because such substances like the solvent system. Conversely, a hydrophobic substance (water-disliking) will be more likely be adsorbed from aqueous solution.

The second primary driving force for adsorption results from a specific affinity of solute for solid. According to this, adsorption is caused by electrical attraction of a solute to the adsorbent, by van der Walls attraction or a by a chemical nature.

Adsorption resulting from van der Walls forces and electrical force between adsorbate and the atoms that compose the adsorbent surface is termed as physical adsorption. In this type of adsorption, to some extent, adsorbed molecules are free to undergo translational movement within the interface because molecule is not affixed to a specific surface site at the surface. For conditions in which adsorption is caused by chemical interaction between adsorbate and adsorbent, this phenomenon is termed as " chemisorption" or "chemical adsorption".

Physical adsorption occurs usually at low temperatures, in other words, the energy of physical adsorption is low and the adsorbate is not held as strongly to the adsorbent as for chemical adsorption. Conversely, chemical adsorption processes exhibit high energies of adsorption, because adsorbate forms strong localized bonds at the active sites on the adsorbent. And chemical adsorption is favoured by higher temperature, because chemical reactions proceed more rapidly at elevated temperatures than lower temperatures.

In the adsorption phenomena several forces influence the different types of adsorption. Most of adsorption processes are combination of physical and chemical adsorption. And it is not easy to distinguish between adsorption types.

2.2.3. Factors Affecting Adsorption

2.2.3.1 Surface Area

Because adsorption is a surface phenomenon the extent of adsorption is proportional to specific surface area. Specific surface area can be termed as that portion of the total surface area that is available for adsorption. Hence, the breaking up of large particles to form smaller ones results in open, some tiny channels which might then become available for adsorption.

2.2.3.2. Nature of the Adsorbent

The extent of the solubility of the solute is one of the most significant factors in the adsorption equilibrium. It is postulated that before adsorption can occur some form of the solute-solvent bonds must be broken [5]. The greater the solubility, the stronger are the solute-solvent bonds. Hence, an inverse relationship exists between the degree of the solubility in the solvent and the extent of the adsorption.

Ionization is one of the factors effecting adsorption. For simple compounds, adsorption is at a minimum for the charged species and a maximum for neutral species. On the other hand, as compounds become more complex the effect of ionization losses its importance.

The effect of solute polarity on adsorption is another important factor. The polar solute will prefer the phase which is more polar. This means a polar solute will be more strongly adsorbed from a nonpolar solvent by a polar adsorbent, but will much prefer a polar solvent to a nonpolar adsorbent.

High molecular weight, low polarity, low ionic character, low pH for organic acids and high pH for inorganic bases, and aromatic structures are correlate with increased adsorption. As a rule of thumb, molecules of higher molecular weights are attached more strongly than are molecules of lower molecular weights [6]

2.2.3.3. pH

The pH of the solution affects the adsorption process due to the fact that the adsorption of the ions other than hydrogen and hydroxide is influenced by pH because hydrogen and hydroxide ions are adsorbed quite strongly. In general, adsorption of typical organic pollutants from water is increased with decreasing pH. In many cases, this may be a result of neutralization of negative charges at the surface of the carbon with increasing hydrogen-ion concentration. Therefore, hindrance to diffusion is reduced and more of the active surface of the carbon becomes available.

2.2.3.4. Temperature

Adsorption reactions are normally exothermic. Therefore, the extent of adsorption generally increases with decreasing temperature.

2.2.4. Activated Carbon

Activated carbon adsorption is the best developed and one of the most efficient processes available for the removal of most organic and some inorganic materials from water and wastewater. Removal of some organic materials also results in removal of taste, odour and colour. Beside organic materials, some inorganic materials including some potentially toxic trace materials are also effectively removed from aqueous solutions.

The most important characteristics of the activated carbon is the large surface area. The surface area of several activated carbons manufactured for wastewater treatment is about 1000 m^2/g [7].

Activated carbons are prepared from carbonaceous material such as coal (anthracite or brown coal), wood, nut shell, peat, and other carbon containing material. Activated carbon production consists of two distinct phases. Firstly, carbon sources are dehydrated and carbonised by slowly heating in the absence of air at several hundred degrees centigrade As a consequence excess water including structural water, the volatile fraction and low molecular products are driven off. Sufficient porosity and thereby a large surface area are achieved by activation of this carbonized material activation can be made by oxidizing

gases, such as steam at above 800°C or by carbon dioxide at higher temperature. When suitable substances are added to the starting material before carbonization process, reaction proceeds at lower temperatures and carbonized product of high adsorption capacity are obtained. Moreover, by this method larger micropores are formed than those formed by gas phase activation [8]. Larger micropores may be preferred for the adsorption of larger molecules

2.2.4.1 Powdered Activated Carbon

Commercially used activated carbons are classified mainly in two forms, powder form and granular or pelletilized forms. Powdered activated carbons in most cases are produced from wood in the form of saw dust. The average size of PAC is in the range of 15 to 25 μm. PAC is used for adsorption of liquids. In the liquid phase, surface charge of PAC becomes an important factor because it acts as a coagulant agent thus affects coagulation, sedimentation and filtration. Decoloration in the food processing, such as sugar refinery, oil production and sodium glutamate productions as well as wine production are some of the major industrial PAC applications in the industry.

2.2.4.2. Granular Activated Carbon

Granular activated carbons (GAC) are either in the form of crushed granules (coal or shell) or in the pelletized form of prepared by granulation of pulverized powders using binders such as coal tar pitch. Size of granules differs depending on the application. For gas adsorption cylindrically extruded pellets of between 4 to 6 mm or crushed or sieved granules of 4/8 mesh to 10/20 mesh are often used. Solvent recovery, air purification, gas purification are the main applications of gas phase adsorption. On the other hand, in the liquid phase adsorption, smaller particle size is advantageous because intraparticle diffusion becomes the rate-limiting step of the adsorption. However, from the viewpoint of the operational requirement such as ease of handling, low pressure drop in the adsorption bed, little elutration or abrasion during back washing lower limit of the particle size is determined. Granular activated carbon adsorption and regeneration are used as a tertiary treatment method in wastewater treatment. Beside, decoloration in sugar refinery, removal of organic substances, odour and trace pollutants in drinking water treatment are the major applications of liquid phase adsorption.

2.2.4.3 Activated Carbon Features

Micropores

Micropores of activated carbons, where most adsorption occurs, are in the form of two-dimensional spaces between two graphite–like walls, two-dimensional crystal planes composed of carbon atoms. The distance between the two neighbouring planes of graphite is 3.76 Å (0.376 nm), but in the case of the activated carbons the planar layers are arranged at different angles to a perpedicular axis passing trough them and they overlap in an irregular manner (so called turbostratic structure).(Figure 2.1)

Micropores include pores with effective radii of less than about 18-20 Å. The volume of micropores varies with different types of activated carbon between 0.15-0.50 cm^3/g and their area is usually a minimum of 95 % of the total surface area.

Surface Oxide Groups

Most activated carbons contain some oxygen complexes that arise from either source materials or from chemical adsorption or air (oxidation) during the activation stage or during storage after activation. Surface oxides add a polar nature to activated carbons. Oxygen complexes on the surface exist in the form of four different acetic surface oxides, namely; strong carboxylic groups, weak carboxylic groups, phenolic groups, carbonyl groups. Besides oxygen complexes, there are several other forms of oxides including basic groups such as cyclic ether groups. Surface oxide groups can be removed by heat treatment of carbons in an inert atmosphere or under vacuum [8]
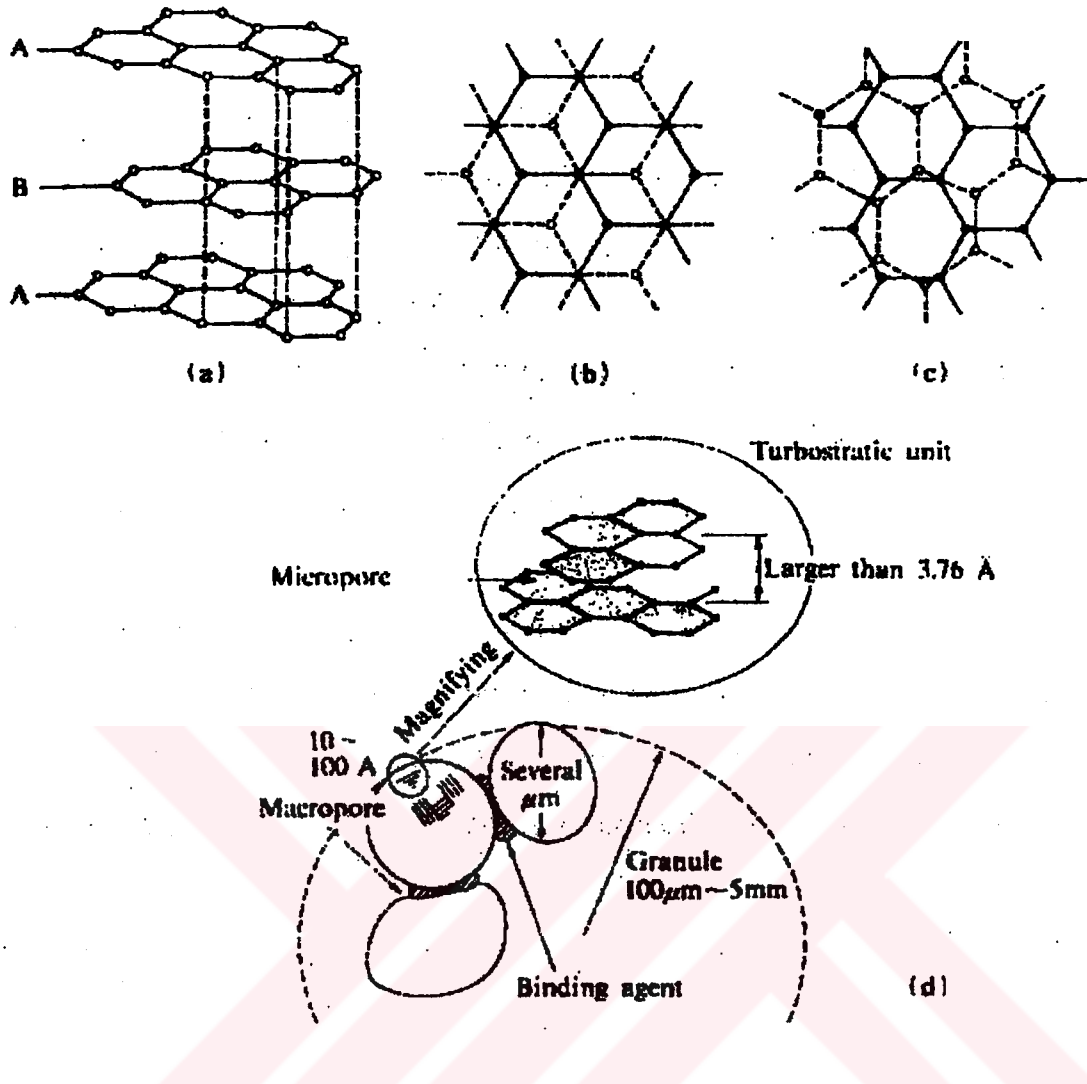
FIGURE 2.1. Graphite structure (a,b) and turbostratic structure (c) of activated carbon. Conceptual illustration of granular activated carbon (d) [8]

Activated carbons activated at low temperatures (< 500 °) are "acidic" and those activated at higher temperatures are "basic" carbons. Acidic carbons are defined as carbons that are capable of lowering the pH of neutral or alkaline distilled water, and that are relatively hydrophilic. Basic carbons are not really basic in the acid-base sense, as they interact with acidic solutions in a specific anion adsorption manner, but they are characterised by the ability to raise the pH of a neutral or acidic solution, and by a relative hydrophobicity.

Ashes

Activated carbon also contains to some extent ashes derived from starting materials. The amount of ash ranges from 1 % to 12 %. Ashes consist mainly of silica, alumina, iron, alkaline and alkaline earth metals. Presence of ashes in PAC increases hydrophobicity, which is advantageous in the water treatment. Another function of ashes is the catalytic effects of alkaline-earth and some other metals during activation or regeneration processes modifies pore size distribution to a larger range.

2.2.5. Adsorption Equilibrium and Adsorption Isotherm

Adsorption in a solid-liquid system results in the removal of solutes from solution and their concentration at the surface of the solid, to some extent that the concentration of the solute remaining in solution is in the equilibrium with at the surface. At this point of the equilibrium, there is a defined distribution of solute between the liquid and solute phases. The distribution ratio is a measure of the position of equilibrium in the adsorption process; it may be a function of concentration of the solute, the concentration and the nature of competing solutes, the nature of solution, and so on.

Generally, the amount of material adsorbed is determined as a function of the concentration at a constant temperature, and the resulting function is called an adsorption isotherm. The theoretical adsorption capacity of the carbon for a particular contaminant can be determined by calculating its adsorption isotherm. Equations that are often used to describe the experimental isotherm data are as follows: Freundlich , Langmuir, and Brunauer, Emmet and Teller (BET isotherm) . Of the three, the Freundlich isotherm is used most commonly to describe the adsorption characteristics of the activated carbon used in water and wastewater treatment [4].

2.3. Treatment of Landfill Leachate in Activated Sludge Systems

2.3.1 Activated Sludge Processes

Activated sludge is a continuous or semicontinuous (fill-and–draw) aerobic biological wastewater treatment process that involves the production of an activated mass of microorganisms capable of stabilising a waste aerobically. Aeration of wastewater with a mixed culture of microorganisms results in consumption of organic materials in the waste stream and biological growth of microorganisms. Afterwards, treated wastewater and biomass are separated from each other.

Activated sludge treatment method removes dissolved and colloidal biodegradable organics as well as unsettleable suspended solids and other constituents, which can be sorbed or entrapped by the activated sludge floc. The mineral nutrients (phosphorus and nitrogen compounds) can be also partially removed. However, the colour of many industrial influents can not be removed by this method and even there may be an increase in colour due to the formation of highly coloured intermediates resulting from biological oxidation of certain compounds [9]. Activated sludge consists of a complicated mixture of viruses, bacteria, protozoa and other organisms, found in single/ or clumped together, often enmeshed in a fabric of organic debris, dead cells and other waste products.

Degradation of organic matter by activated sludge is affected by environmental conditions, such as pH, temperature, nutrient conditions, and turbulence and waste composition. The mixture of activated sludge and wastewater is named mixed liquor suspended solids (MLSS) and the volatile organic fraction is called mixed liquor volatile suspended solids (MLVSS) [1]. In a conventional activated sludge system treating municipal wastewater, the active biomass usually represents only about 30 percent or less of mixed liquor suspended solids.

Microorganisms are separated from wastewater and then a portion is returned to aeration basin so that desired concentration of microorganisms and F/M ratio (Food to microorganisms) can be maintained.

The organic matter in a wastewater can be divided into three categories [10] as readily biodegradable, slowly biodegradable and non-biodegradable (inert) organic compounds. Readily biodegradable compounds, such as alcohols, volatile fatty acids and glucose can be immediately metabolized by the cells and converted to storage products or biomass. On the other hand, non-biodegradable compounds pass through all biological treatment systems unchanged in form. Inert compounds can be present in the wastewater in both soluble and particulate forms and are also generated during activated sludge process. Biodegradable organic compounds are termed as compounds which have to be hydrolysed by means of extracellular enzymes before they can be transported into cells for intracellular metabolism slowly biodegradable organics are subdivided as soluble and suspended organics.

In the activated sludge system, when wastewater and microorganisms are contact in the presence of dissolved oxygen, soluble organic substances in the wastewater are adsorbed on the surface of activated sludge flocs. At the same time, a fraction of wastewater organics is converted into new bacterial cell. A certain fraction of organics is decomposed to carbon dioxide and water.

The simplified stoichiometry of organic matter removal is shown in Equation 2.1, in which CHONS represents the organic substances and $C_5H_7NO_2$ represents the new bacteria cells produced.

$$CHONS + O_2 + nutrients \xrightarrow{\text{enzymes}} CO_2 + NH_3 + C_5H_7NO_2 \tag{2.1}$$
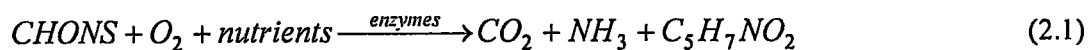
After depletion of available organic substrate, the microorganisms are forced to metabolize their own cells. The stoichiometry of endogenous respiration is shown in Equation 2.2.

$$C_5H_7NO_2 + 5O_2 \xrightarrow{\text{enzymes}} 5CO_2 + 2H_2O + NH_3 + energy \tag{2.2}$$
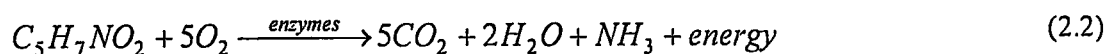
Bacterial populations in aerobic conditions are capable also of converting ammonia nitrogen into nitrite and nitrate nitrogen. This process is called nitrification. Nitrification bacteria are autotrophic microorganisms, which use carbon dioxide rather than organic

carbon as carbon source. In general, nitrification is achieved in two steps. Firstly, ammonia is oxidized to nitrite by *Nitrosomonas spp* and subsequently, nitrite is oxidized to nitrate by *Nitrobacter spp.*

The first phase of nitrification process is presented as follows:

$$NH_4^+ + \frac{3}{2}O_2 \xrightarrow{\quad Nitrosomonas \quad} NO_2^- + 2H^+ + H_2O \tag{2.3}$$
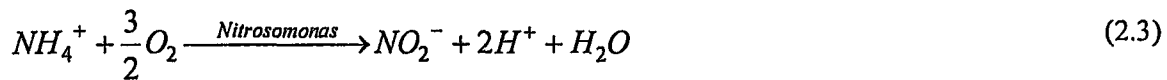
While the second step is as follows:

$$NO_2^- + \frac{1}{2}O_2 \xrightarrow{\quad Nitrobacter \quad} NO_3^- \tag{2.4}$$

2.3.2 Treatment of Leachate by Activated Sludge

Leachate can be treated by aerobic or anaerobic biological processes and also by physical and chemical treatment alternatives depending upon characteristics of leachate. Treatment of leachate by the activated sludge method is one of the most favourable alternatives. From the results of many researches, up to 80-99 % BOD and COD removal and up to 80-99 % metal removal can be achieved. However, due to the large amount of organic matter in leachate rather extensive operational conditions are required such as MLVSS concentrations of 5000-10000 mg/L, F/M ratio of 0.02-0.06 BOD_5/MLVSS.day , hydraulic retention time of 1-10 days and solid residence time of 15-60 days [1]

The problems arising in the activated sludge treatment of leachate can be stated as follows [1,2]:

(a) Foaming due to high metal concentration
(b) Requirement of an anti-foaming agent and mechanical mixing to reduce foaming
(c) Inhibition of biological processes due to nutrient deficiency (especially the phosphorous concentration is not sufficient for biological treatment)
(d) Inhibition of the process due metals and other constituents in the leachate
(e) Higher sludge production due to increased biomass and metal participates

Similar to other biological treatment options, applicability of activated sludge to leachate depends on the BOD_5/COD ratio. The high BOD_5 /COD ratio in a young leachate results from high amounts of volatile fatty acids and this favors biological treatment. However, in the case of an old leachate containing large amounts of refractory organic matter, this method is not suitable. Chemical- physical treatment processes are more convenient for leachates from old landfills.

In the treatment of leachate by activated sludge, nitrogen removal is essential, especially in the case of old leachates with high ammonia concentrations [2]. In addition to organic removal, metals are also removed by precipitation and adsorption on biomass. High per cent removal of some metals in the settled biological floc, especially Iron (>98), Zinc (>99), Calcium (>93), Manganese (>95), Cadmium (>96), Lead (> 79) and Magnesium (>54) is reported [2]. The MLVSS/MLSS ratio decreases in an activated sludge reactor because of heavy metal precipitation [1]. Pretreatment of metals prior to biological treatment is required due to inhibition of activated sludge at high heavy metal concentrations [11].

Due to contamination with chemicals, even leachates from municipal solid waste landfill sites contain trace amounts of pharmaceuticals, solvents, cleaners, herbicides and pesticides. There are concerns about their potential and carcinogenic properties.

Similar to organic pollutants, the presence of heavy metals in leachate may cause an inhibitory or toxic effect on biological treatment [1]. Heavy metals effect biological treatment by reacting with microbiological enzymes. Inhibition is especially important at low pHs due to solubilization of precipitated metals. A list of pollutants that have an effect on the activated sludge process is presented in Table 2.2.

TABLE 2.2. Threshold concentrations of pollutants inhibitory to activated sludge process [4].

Pollutant	Concentration (mg/L)	
	Carbonaceous removal	Nitrification
Aluminium	15-26	-
Ammonia	480	-
Arsenic	0.1	-
Borate	0.05-100	-
Cadmium	10-100	-
Calcium	2500	-
Chromium(Hexavalent)	1-10	0.25
Chromium(Trivalent)	50	-
Copper	1.0	0.005-0.5
Cyanide	0.1-5	0.34
Iron	1000	-
Manganese	10	-
Magnesium	-	50
Mercury	0.1-5.0	-
Nickel	1.0-2.5	0.25
Silver	5	-
Sulfate	-	500
Zinc	0.8-10	0.08-0.5
Phenol	200	4-10
Cresol	-	4-16
2-4 Dinitrophenol	-	150

2.3.3. Co-Treatment of Leachate and Domestic Wastewater

The leachate treatment at an existing municipal wastewater treatment plant is a convenient method. When the landfill is placed near a publicly owned sewer, it is a cost-effective option when leachate is combined with domestic wastewater. The requirements are as follows: availability of sewer system, plant capacity to assimilate the waste, process

compatibility with leachate characteristics, and facility to handle increased sludge production.

On the other hand, the addition of leachate to municipal sewage may decrease the possibility of obtaining an effluent suitable for water recycling, recharging groundwater resources. Furthermore, occasional problems with sludge bulking and poor solid/liquid separation and requirement of nutrient addition are encountered [1].

Combined treatment of landfill leachate and domestic sewage has received significant consideration [1-3,12-19]. Municipal sewage may be a satisfactory source of nutrients and microorganism supply. Moreover, sewage may also dilute some inhibitory components of the leachate, making the combined treatment much easier and more effective than would be possible for leachate only.

Researches have been conducted to determine the leachate ratio that can be tolerated in the municipal wastewater without causing deterioration in effluent quality [3, 16]. Comparison of the results is difficult because both leachate and sewage have different composition and different experimental procedures are used. However the impact of leachate on biological treatment is evident. Therefore, the maximum leachate ratio at which no reduction in removal is observed must be determined in each case.

It was shown that high strength leachates (24000 mg/l COD) when combined at less than 2% by volume with normal municipal wastewater have no effect upon the performance of municipal wastewater treatment. However, if the percentage was increased to 5%, plant performance could deteriorate [1]. In another study, leachate was successfully treated when combined at 20 and 40% by volume with municipal wastewater in aerobic batch (fill and draw) reactors with sludge residence time (SRT) of 5,10 and 20 days [1]. In the combined treatment of leachate, even over 90 percent COD and BOD removal efficiency and complete nitrification (over 80%) can be obtained with a 10 day sludge residence time The effect of leachate to biological system varies from case to case [3]. The possible reason is that leachate quality has some impact on the efficiency of treatment.

In addition to reductions in removal efficiencies, modifications of biomass and foaming are reported. Addition of leachate may promote filamentous growth such as *Microthrix Parvicella* and *Triothrix spp.* [12]. *Microthrix Parvicella* is known to feed on slowly biodegradable substrates and *Triothrix spp* grows in nutrient deficient systems [20].

In co-treatment of landfill leachate and domestic wastewater sludge production increases due to the precipitations of heavy metals such as Fe, Cu, and Ni [11].

2.4. Powdered Activated Carbon Addition to Activated Sludge

2.4.1. Basic Features of PAC Added Activated Sludge Systems

Powdered Activated Carbon application to activated sludge system (under the name of PACT) has been widely used for wide variety of the pollutants. Studies showed that the PACT process was applicable for wastewaters ranging from municipal to highly contaminated wastewaters with biodegradable and non-biodegradable organics.

Studies about PAC in conjunction with activated sludge showed that the PACT system had several advantages over conventional activated sludge systems. The benefits can be stated as follows: improved COD, BOD, and residual organics removal; stabilization of effluent quality; removal of colour, heavy metals and biodegradable and non-biodegradable organics, which cannot be removed in a conventional activated sludge system; less tendency to foam in the aeration basin; more stability against shock loadings, toxic upsets and variable organics loadings [6, 16-34]. Addition of PAC to activated sludge enhances nitrification due to adsorption of nitrification inhibitors. Moreover, in PACT systems, an improvement was observed in sludge settling, dewatering characteristics and SVI. Also substantial improvements in the filter yield in vacuum filtration was observed at lower doses of cationic polymers [21].

In a PACT system, PAC can be directly added to activated sludge aeration basin or mixed with the effluent of the primary clarifier. After providing a sufficient contact time for the association of mixed liquor and PAC, the mixture of PAC and mixed liquor solids are settled from the treated effluent. Spent carbon in the sludge can be regenerated or dewatered and disposed with sludge directly. Wet air oxidation is used for the regeneration of PAC [21]. The simple schematic diagram of the PAC addition to activated sludge systems is shown in Figure 2.2.
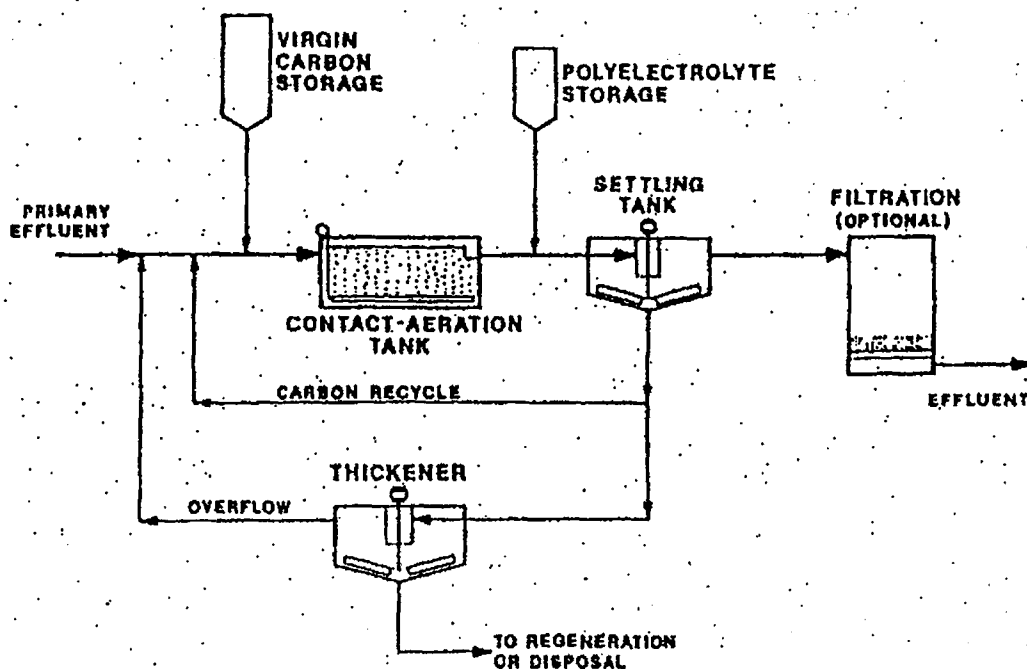
FIGURE 2.2. Schematic representation of a typical powdered activated carbon- activated sludge system [21].

In PAC enhanced systems, an increase in residual suspended solids in the effluent is observed. Therefore, a small dose of liquid cationic polyelectrolyte may be added just ahead of the final clarifier to ensure maximum overflow clarity. Doses typically range from 0.5 to 1.5 mg/L [21]. Following clarification, effluent filtration is also employed if low suspended solids requirements must be met. Similar to the conventional activated sludge system, a portion of the biological solids and spent activated carbon are wasted daily while a portion is returned to aeration basin to reach the desired level of microorganisms. In order to maintain the desired activated carbon concentration, virgin activated carbon addition may also be added.

2.4.2. Enhancement of Organic Matter Removal and Nitrification with PAC Addition

In a PACT system, organic matter removal is due to simultaneous physical adsorption and biodegradation. However, results of many studies showed that organic matter could be removed more effectively than either biodegradation or carbon adsorption alone. Therefore, a synergism is speculated by many authors [25, 26, 31]. The mechanisms of improvement are said to be due to stimulation of biological activity by the action of PAC and bioregeneration. Bioregeneration can be termed as the renewal of adsorption capacity by microorganisms for further adsorption.

Enhancement of activated sludge with PAC addition is primarily due to adsorption of inhibitory and toxic substances. Presence of such substances in the influent may cause total failure of conventional activated sludge system. On the other hand, due to adsorption of these substances, microorganisms are exposed to lower concentrations of toxins and therefore a stimulation of bioactivity results. Moreover, PAC provides an improved stability to shock loadings. Thus, PACT system may be applicable in the treatment of leachate and industrial wastewaters which vary in quality and quantity.

Besides the enhanced removal in COD, improvements in BOD removal and biological activity are also reported [18, 26, 27, 30]. BOD represents, however, biodegradable substances which are rather weakly adsorbable onto activated carbon. Therefore, their enhanced removal due to PAC addition could be due to a stimulation of bioactivity. Oxygen Uptake Rate (OUR) is a measure of bioactivity of microorganisms. OUR measurements revealed that addition of PAC to AS resulted in an increase in OUR even in the presence of inhibitory substances [26, 27]. Moreover, it was found that the presence of PAC affected the available oxygen for the microorganisms by adsorbing oxygen. In a PACT system, PAC acted as O_2 reservoir [31].

The MLSS (Mixed Liquor Suspended Solids) and MLVSS (Mixed Liquor Volatile Suspended Solids) levels (corrected with PAC) in PAC added activated sludge systems are lower than expected [24, 26, 31]. This decrease may be a consequence of enhanced endogenous respiration in the PAC reactors resulting in lower yields. Higher organic removal at a lower MLVSS level might be an evidence of stimulated bioactivity.

Generally, activated carbon is suitable for organics with low water solubility and high molecular weight. However, a decrease in the extent of adsorption was observed as the molecular weight of the material increased [35]. On the other hand, it is suggested that activated carbon preferentially adsorbs the refractory organics with low molecular weight (LMW) than high molecular weight (HMW) [28]. Moreover, LMW organics are less refractory or more amenable to biodegradation than high molecular weight organics. Therefore, microorganisms prefer LMW compounds for biodegradation during the bioregeneration of activated carbon.. A non-adsorbable substrate could be removed by adsorption when it is in certain mixtures although it showed little affinity for activated carbon when alone in the solution [25].

In a PACT system, enhancement of nitrification may be observed [16, 24, 29, 33]. Increased removal of ammonia is primarily due to adsorption of inhibitory and toxic compounds, which otherwise cause an upset of the nitrification process. Nitrifiers are known to be more sensitive than heterotrophic bacteria. Therefore adsorption of inhibitors is an important factor in the enhancement of nitrification.

The presence of PAC increases also the solid-liquid surface. Hence, the PACT system is considered to be favorable for nitrifiers growing on the surface of PAC particles because nitrifying bacteria grow mainly in attached form. The immobilized microorganisms form a biofilm on the PAC surface. Thus, bacteria are prevented from the wash-out from the bioreactor at high hydraulic loadings. Moreover, nitrification efficiency may be increased due to acclimation of organisms to nitrification inhibitors [16].

2.4.3.Bioregeneration

The synergisms between PAC and AS microorganisms might be explained by bioregeneration. According to the bioregeneration hypothesis, in a PACT system microorganisms can regenerate or free the adsorption capacity for further adsorption. Desorption of adsorbed organics by exoenzymatic reactions results in bioregeneration [25, 26, 31]. Some enzymes excreted by the bacteria could easily diffuse into the micropores and react with the adsorbed substrates and further desorption may take place due to the weak absorbability of enzyme reacted organic matter. Due to bioregeneration the broken-down products desorb and new molecules can be adsorbed on vacant sites. However, in the literature, there is an ambiguity about bioregeneration hypothesis. Some researches observed a bioregeneration [25, 26, 31], whereas according to some others the synergism was due to a simple combination of adsorption and biodegradation [16, 17, 18, 31, 34, 36]. It is stated that neither bacteria nor exoenzymes are small enough to migrate into the micropores of PAC [32].

In a PACT system, the removal of non-biodegradable organics can only be achieved by simple adsorption. However, the removal of slowly biodegradable and adsorbable compounds is a time dependend process. Therefore, if sufficient time is given, these compounds can be degraded. In a conventional activated sludge system these substances are in contact with biomass for a length of time equal to the hydraulic retention time. On the other hand, in a PACT system, the retention time of adsorbable and biodegradable substances is equal to the sludge residence time once adsorbed onto activated carbon. Then the removal of these substances must depend on the sludge age and increase with increasing sludge age [25]. At this point, biologically activated GAC systems are more advantageous than PACT systems due to the fact that the sludge age in these systems are higher than PACT. In a biologically activated GAC system, higher sludge ages provide additional removal of refractory organics.

The desorption of adsorbed organics may also take place due to the concentration gradient between the PAC surface and the bulk liquid. The energy of adsorption, however, may be an indicator of this potential because significant amounts of adsorbed substrates can be strongly adsorbed and may not be available for biodegradation [26].

On the other hand, some recent studies indicated that the synergism in the PACT system was due to a combination of simple biodegradation and adsorption rather than bioregeneration [32, 34, 36]. Results indicated that after exhaustion of adsorption capacity, the removal rate in a PAC added activated sludge reactor was almost the same as in a biological reactor [32].

2.5. Sludge Characteristics

The performance of an activated sludge system is strongly related to the performance of both the organics removal efficiency and the separation of biological solids from treated wastewater in the clarifier. In some cases, even if the organics removal efficiency is satisfactory, activated sludge system can not meet the discharge parameters due to inefficient separation of treated wastewater and biosolids. Moreover, this situation results in additional operational problems. In such a system, in the aeration basin the MLSS (Mixed Liquor Suspended Solids) concentration cannot be maintained at a desired level due to the low suspended solids concentration in the return sludge. The treatment efficiency is also affected negatively due to the low MLSS concentration in the aeration basin.

Besides the solid-liquid separation problems, dewatering of the produced sludge is also of concern. Therefore, for efficient and economical treatment of wastewaters, the dewatering characteristics of the sludge deserve special attention.

2.5.1. Factors Affecting Sludge Settling and Dewaterability

The settleability depends mainly on the structure, size and density of activated sludge flocs. These properties can change fast with varying conditions in the wastewater treatment system.

The effect of filamentous and floc-forming bacteria on settling and dewatering of activated sludge was investigated by several authors [20, 37-41]. For a well-developed floc, which can be settled and compacted easily leaving a clear supernatant, filamentous and zoogleal (floc-forming) bacteria must be in balance because different types of microorganisms play an important role in the formation of flocs. Filamentous bacteria serve as a backbone so that floc-forming bacteria cling onto them [20, 37]. Total disappearance of one of the two types of bacteria results in disturbances in settling and dewatering properties of the sludge. If the filamentous bacteria are present in large numbers, the produced sludge has a poor settling character with a corresponding increase in SVI (Sludge Volume Index). On the other hand, in the absence of filamentous microorganisms, a sludge with poor compaction

is produced (zoogleal or non-filamentous). These types of sludges have a voluminous character. The flocs are weaker and by turbulence they break into smaller particles and a turbid supernatant is obtained.

The effect of dissolved oxygen (DO) on sludge settling characteristics is through effecting the microorganisms and consequently particle size distribution in sludge. Studies showed that at low DO concentrations, flocs with poor settling and dewatering characteristics were produced due to the excessive growth of filamentous bacteria [42]. The main reason of overgrowth of filamentous bacteria at low DO (as well as at low nutrient concentrations) was that many types of filamentous microorganisms were also capable of utilizing these compounds more effectively at low concentrations due to the kinetic selection [43]. On the other hand, flocs grown at low DO concentrations are known as having low exopolymers and therefore smaller in size than that of grown at high DO concentrations [42]. Low DO concentrations also correlate with high turbidity. This can be primarily due to inefficient flocculation or deflocculation of flocculated bacteria at low dissolved oxygen concentrations Another reason of turbidity may be the disappearance of eucaryotes, which scavenge on the dead bacteria and other colloids in the bulk liquid at low DO concentrations [44].

Since reactor configuration affects the available substrate concentration and the biomass type, there is a strong relationship between the reactor type and the settling of activated sludge. In practice, activated sludges from plug-flow reactors usually settle better than that from completely mixed systems and tend to have fewer filaments. It was seen that the relative abundance of filamentous and floc-forming bacteria was related to their relative growth rates when exposed to varying concentrations of substrate [4, 43, 45]. Low concentrations of substrates and nutrients in the complete mixed system give filamentous bacteria a competitive advantage due to the fact that filamentous bacteria have a low μ_{max} but a high affinity for the substrates. On the other hand, in plug-flow reactors (or the start of the aeration in the case of the batch treatment system), the floc-forming microorganisms, which are capable of utilising the substrate at a faster rate at high concentrations, can outcomplete the filamentous microorganisms [43]. Therefore, overgrowth of filamentous bacteria can be eliminated in plug-flow reactors.

The relationship between particle size distribution and dewatering is important. In literature, there is a general agreement that dewaterability decreases with decreasing particle size distribution [44, 46]. This is due to the fact that as particle size gets smaller, smaller particles tend to blind the passage ways of water through the cake and the filter medium during filtration. The resistance to the flow of water is increased and as a result the specific resistance to filtration (SRF) increases.

2.5.2. Effect of Powdered Activated Carbon (PAC) on Activated Sludge Characteristics

Powdered Activated Carbon (PAC) addition to activated sludge system is a widely applied procedure. In addition to improvement in COD removal and nitrification, PAC improves sludge settleability [4, 16, 25, 42]. In PAC added activated sludge systems, the mixed liquor containing PAC settles more rapidly and produces a lower Sludge Volume Index (SVI). Settling of activated sludge was good even in the presence of a significant number of filamentous organisms [20]. The main reason of this enhanced settling was the weighting effect of activated carbon particles. The density of activated carbon and mixed liquor was higher than either PAC or microorganisms alone. Production of heavier and well-structured flocs resulted in an increased sludge settling velocity and an efficient solid-liquid separation.

Dewaterability of activated sludge was affected by addition of PAC to activated sludge reactors. Studies about PAC addition revealed that in PAC added AS systems, floc size distribution was lower than conventional activated sludge systems [23, 33, 47]. Kim et al. (1998) observed that the median size of particles in AS and PAC-added AS systems were as 100 and 82 μm, respectively. In another study, although particle size distribution was in a lower range, the mean floc diameter in the PAC added activated sludge was 29.2 μm smaller than the diameter of activated sludge (44 μm) [33]. In such a system, attached growth is predominant over suspended growth. Therefore, simple mixture of PAC and activated sludge is quite different due to the fact that in this case no biofilm is formed during such a short time period. When powdered activated carbon was mixed with activated sludge in a beaker, the diameter of the sludge flocs was increased and measured as 134 μm.

Paradoxically, even in the cases when the particle size in the PAC added activated sludge is lower than the conventional activated sludge, an increased dewaterability (measured as lower SRF) is observed. Therefore, the mechanism of this increase in sludge dewaterability due PAC addition must be explained in another way. It is postulated that increased dewaterability by PAC addition may be due to the change of porosity of sludge [33]. The porosity of PAC containing activated sludge was measured higher than activated sludge. [33].

One of the characteristics of the PAC added AS reactors is a shift in ECP (Extracellular Polymers) levels. Settling of activated sludge is very related to the ECP content of the flocs. ECP originate either from the microorganisms in the activated sludge or from the wastewater itself. Constituents of ECP (protein, carbohydrate and lipid) influence the settling characteristics of the activated sludge via effecting the surface charges [48]. Moreover, it was observed that a decrease of ECP content was associated with an increase of the dewatering property in the case of PAC addition [33]. The reduction of ECP might have been caused either by adsorption and/or attachment of extractable ECPs to the PAC surface or by the reduction in the excretion of ECP from the attached growth microorganisms than suspended growth microorganisms[33].

2.5.3.Effect of Leachate on Activated Sludge Characteristics

In most of the studies about the co-treatment of leachate and domestic wastewater, biomass alterations, foaming and dominance of filamentous growth were reported [1, 2, 12]. The presence of leachate in the feed affected sludge characteristics by influencing the bacterial composition. Slowly biodegradable substrates in leachate promote filamentous growth [20]. *Microthrix Parcivella* and *Thiothrix ssp.* were some of the filamentous microorganisms observed in the co-treatment of leachate and domestic wastewater [12]. *Microthrix Parcivella* is known as having an affinity to slowly biodegradable (even particulate) substrates. The presence of long chain fatty acids (as in the case of young leachates) can have a positive effect on *M. Parcivella* due to the fact that the hydrophobic property of *M. Parcivella* offers an additional advantage for uptake of these substances. Overgrowth of filaments leads to larger flocs with poorer settling and compaction, corresponding with an increase of SVI (Sludge Volume Index). The general characteristic

of filamentous bacteria is their relatively large surface area, which also give an additional advantage for the uptake of substrates and nutrients, but makes settling more difficult [40, 45]. On the other hand, filamentous bacteria increase the porosity of flocs if they proliferate inside the flocs.

One of the most important factors affecting the sludge characteristics is the nutrient deficiency in leachate. Sludge characteristics were affected by favouring filamentous growth due to nutrient deficiency of nitrogen, phosphorous or trace elements or combination of them [43]. Moreover, presence of toxic materials and heavy metals can result in dispersed growth due to deflocculation.

Another important factor influencing sludge characteristics is the presence of high dissolved solids concentrations including calcium, potassium, sodium etc. Divalent cations are important in the formation of a well-structured floc. Activated sludge flocs are held together to form a 3-dimentional matrix by means of exopolymers (biopolymers or extracellular polymers) and divalent cations. The role of cations is to form complexes or ion pairs between the functional groups of the exopolymers and the bacteria. Among the divalent cations, Ca^{2+} seems to be most important cation for the ability of the bridging between themselves and bacteria in the activated sludge. Additions of cations resulted in turbidity accompanied by an increase in SRF [49]. The decrease in dewaterability (increased SRF) may be due to the weakening of sludge structure through the displacement of Ca^{2+} with other cations. The displacement capacity of these cations varies (Mg $^{2+}$ > Na $^{+}$>K^{+}). When Ca^{2+} is replaced or removed from activated sludge flocs, a weakening of sludge structure and appearance of smaller particles may be expected [46]. Beside divalent cations, monovalent cations such as K^{+} and Na^{+} have an effect on sludge characteristics. At low concentrations of potassium, addition of small concentrations of potassium improves activated sludge settling and dewatering characteristics. However, excess monovalent cations such as sodium and potassium can cause deterioration in settling and dewatering properties of activated sludge [46].

3. EXPERIMENTAL WORK

3.1 Experimental Procedure

The effect of PAC on substrate removal and settling characteristics was investigated in the case of co-treatment of leachate and domestic wastewater in activated sludge reactors. Experiments were performed mainly in three phases:

- Co-treatment of landfill leachate and domestic wastewater
- Treatment of leachate alone
- Treatment of domestic wastewater alone

In the co-treatment phase, leachate and domestic wastewater were mixed at different volumetric ratios and COD removal and sludge characteristics were examined. Based on previous works [3,16], it was decided that the leachate ratio should vary from 5-25 %. In the treatment of leachate alone, leachate was diluted because of the high concentrations. The dilution was made according to the former co-treatment runs. Similarly, stock domestic wastewater was diluted to approximately the same strength and treated alone in activated sludge reactors. The characteristics of these batch runs are presented in Table 3.1.

2 litre plexiglass batch reactors were used in the experimental path of the study. Throughout the runs, a control reactor (AS: Activated Sludge) was fed under the same conditions as the AS+PAC (PAC added Activated Sludge) reactor. The stock synthetic domestic wastewater was diluted to 500 mg/L (to the strength of a typical domestic wastewater) and added to both AS and AS+PAC reactors. The characteristics of the domestic wastewater are presented in Section 3.3. Additionally, the characteristics of the PAC used in the study are given in Section 3.4.

Both AS and AS+PAC reactors were inoculated with a sludge taken from a five-litre stock activated sludge reactor. In each run, a new sludge sample was taken from this reactor. This stock activated sludge reactor had been fed with synthetic domestic wastewater for three years and sludge age of activated sludge was kept at 20 days by wasting daily 1/20 of

the mixed liquor[3, 16]. The reactor was fed with the synthetic wastewater daily at a loading rate of 500-mg COD/L.day.

TABLE 3.1. The Characteristics of Batch Runs Performed throughout the Study (in chronological order)

RUN	Leachate	Volumetric Leachate Ratio (v/v)	AS Reactors	AS+PAC Reactors	
			PAC Dose (mg/L)		
1	Leachate A	5	0	1000	-
2	Leachate A	20	0	1000	-
3	-	Domestic w.w Alone	0	1000	-
4	-	Domestic w.w Alone	0	1000	-
5	Leachate B	5	0	1000	-
6	Leachate B	5	0	1500	-
7	Leachate B	10	0	1500	-
8	Leachate B	15	0	1500	2000
9	Leachate B	20	0	1500	2000
10	Leachate B	25	0	1500	2000
11	Leachate B	25	0	2500	3000
12	Leachate B	Diluted Leachate	0	1000	2000
13	Leachate B	20	0	2500	3000
14	Leachate B	15	0	2500	3000
15	Leachate B	10	0	2500	3000
16	Leachate B	5	0	2000	2500
17	Leachate B	10	0	2500	3000
18	Leachate B	Diluted Leachate	0	2500	-
19	-	Domestic w.w Alone	0	2500	-

In this study, two different leachate samples were used. First, the leachate from a previous study was used (Leachate A). The runs performed with this leachate served as a preliminary work for further studies. The major part of runs were conducted with a new leachate (Leachate B) taken from the same location. The characteristic of the leachates are presented in Section 3.2.

In the experimental path of the study, separate adsorption studies were not performed since results of a previous study indicated that in AS+PAC reactors organics removal was higher than that of calculated from adsorption isotherms [16]. The results of all batch runs were evaluated with respect to substrate removal and sludge characteristics. Soluble COD (SCOD) and Total COD (TCOD), NO_2-N, MLSS (Mixed Liquor Suspended Solid), TS, (Total Solids), pH, were regularly measured in all runs. NO_2-N was measured in order to correct COD measurements. The sludge properties were characterised by SRF (Specific Resistance to Filtration), which is a measure of the resistance of sludge to filtration or dewatering. Moreover, since specific resistance is a true nature of the filterability of a sample, it is possible to compare sludges numerically. Total Solids (T.S.) measurements were needed for calculation of SRF. Additionally, for comparison purposes, the SRFs belonging to leachate and PAC were also determined. In addition to these, in batch runs Sludge Volume Index (SVI) values and Settling Curves were also examined to have an idea about the settling of activated sludge.

3.2. Landfill Leachate

Throughout the experimental work of the study, leachate generated from the Kemerburgaz Sanitary Landfill Site was used. This site is placed on the European side of Istanbul. Two leachate samples were studied. Leachate A was taken from the leachate storage lagoon on July 1998 and Leachate B was taken on November 1999 from the same location. Both leachates were stored at 4° C in closed plastic barrels. The characteristics of the leachate samples are presented in Table 3.2.

Kemerburgaz Landfill Site is one of the two landfill sites in Istanbul receiving solid wastes from the three transfer stations on the European side and three stations on the Asian side. Only municipal solid wastes are received.

The landfill is placed in an area in which there is no groundwater and the soil is rich in clay acting as an impermeable layer. High-density membrane liners are used to prevent leachate leakage. The solid wastes are landfilled by the cell method and covered daily by soil. Gas collection chimneys are used to collect landfill gas and the leachate generated is collected by special pipes.

TABLE 3.2 Characteristics of Landfill Leachates A and B Used throughout the Study

PARAMETER	Leachate A (used in Runs 1,2)	Leachate B (used in Runs 5-19)
pH	8.49	7.70
TCOD (mg/L)	8662	18422
SCOD (mg/L)	7772	17763
BOD_5 (mg/L)	5940	9660
TKN (mg/L)	2159	2350
Ammonia (mg/L)	1755	1946
NO_x-N (mg/L)	167	167
Alkalinity ($mgCaCO_3$/L)	9500	10700
TSS (mg/L)	710	1540
Total Solids (mg/L)	16015	17655

3.3. Synthetic Domestic Wastewater

Domestic wastewater was prepared synthetically in the laboratory. It was prepared as a stock solution having a COD 10000 mg/L, TKN 1060 mg/L, phosphorous 812 mg/L, and pH 7.1. It was diluted 20-folds before being used in the reactors. This composition represented the composition of domestic wastewater. Phosphorous was supplied in excess amounts in order to provide enough phosphorous in co-treatment studies and to buffer severe pH decreases.

TABLE 3.3. Composition of Synthetic Domestic Wastewater Stock Solution

Component	Concentration (mg/L)
CH_3COONa	6000
Glucose	5600
Peptone	2000
$(NH_4)_2SO_4$	5000
KH_2PO_4	2000
K_2HPO_4	2000
$MgSO_4.7H_2O$	2000
$CaCl_2.2H_2O$	600
$FeCl_3. 6 H_2O$	200

3.4. Powdered Activated Carbon (PAC)

The activated carbon used throughout the study was Norit SA 4 in powdered form. PAC was dried at 103° C before using in the experiments. The properties of activated carbon are shown in Table 3.4.

TABLE 3.4: Properties of Activated Carbon Used in the Experiments

Apparent density (g/L)	190
Moisture(%)	5
Ash Content (%)	6
Phenol Adsorption (%)	4
Molasses Number	525
Methylene Blue Adsorption (g/100 g)	10
Iodine Number (mg/g)	750
Internal Surface Area (m^2/g)	650
Particle Size	
>10 micron (%)	80
>44 micron (%)	37
>79 micron (%)	20
150 micron (%)	5

3.5. Analytical Methods and Other Measurements

The analytical methods used in the experiments followed the Standard Methods [50]:

1) COD Analysis: COD analysis was performed by the dichromate closed reflux method. Samples were refluxed with $K_2Cr_2O_7$ and H_2SO_4 for 2 hours at 150 °C in a Hach COD digester, in the presence of Ag_2SO_4 acting as catalyst. H_2SO_4 was used to prevent chloride interference. COD was measured colorimetrically by measuring the absorbance of samples at 600 nm with the Hach 2100 spectrophotometer. Standard KHP (Potassium Hydrogen Phthalate) solutions were used to prepare calibration curves. In all runs, COD values were corrected for nitrite interference and corrected data was reported in all experiments. In previous studies [3, 15], it was found that 1 mg/L NO_2-N led to an increase of 1.1 mg/L COD. Therefore, correction of nitrite interference was made by subtracting 1.1 mg/L COD from measured COD for each determined NO_2-N concentration.

Total COD (TCOD) is the COD of the supernatant and soluble COD (SCOD) is the COD of the supernatant filtered through 0.45 μm Millipore filter. SCOD was used as the main organic parameter throughout the study.

2) NO_2-N Analysis: Nitrite concentration was determined by the spectrophotometric method. Nitriver 2 test kits were added to samples and the absorbance at 560 nm was measured by the Hach DR/3 spectrophotometer. Samples containing high concentrations of nitrite were diluted with distilled water. Calibration curves were prepared using a $NaNO_2$ solution.

3) pH Analysis : Orion SAS20 pH meter was used.

4) MLSS Analysis: Mixed Liquor Suspended Solids measurements were done by drying the residue on 1.2 μm glassfiber filterpaper for one hour at 103 °C.

5) Total Solids Analysis: Total solids measurements were done by evaporation and drying the residue for one hour at 103 °C.

6) Total Suspended Solids Analysis: Total Suspended Solids measurements were done by drying the residue on 1.2 μm glassfiber filterpaper for one hour at 103 °C.

Sludge characteristics were examined by the following measurements. For this purpose, samples were taken from the mixed liquor of all reactors.

1) SVI (Sludge Volume Index) Measurements: SVI was performed in a 1 litre graduated cylinder without stirring. Sample was allowed to settle down in 30 minutes. SVI was calculated as follows:

$$SVI \ (mL/g) = \frac{ml \ settled \ volume \ of \ sludge \ /1 \ L \ sludge \ x \ 1000}{MLSS \ (mg/l)}$$

2) Sludge Settling Curves: The sample was placed in a 1 L graduated cylinder and well mixed. It was then allowed to settle quiescently. Settling curve were obtained from the decrease of the sludge height with respect to time.

3) SRF Measurement: Specific Resistance to Filtration Test (SRF) on biological sludge was performed as described by Christensen and Dick (1985). Vacuum filtered cake solid measurements were performed using a Buchner funnel with a vacuum pressure of 18 psi. 1.2 μm glassfiber filterpaper was used in the experiments. Specific Resistance, r, values were determined by the equation given below [51].

$$r = \frac{2pA^2b}{\mu W}$$

where ,

r = specific resistance (m/kg)

p = pressure differential (N/m^2)

A = filtering area (m^2)

b = slope of t/V vs V plot in specific resistance test (h/m^6)

μ = filtrate viscosity (N.s/m^2)

w = mass of dry cake deposited per volume of filtrate collected (kg/m^3)

An example to SRF calculations is given as follows:

P = 18 psi = 1.242 x 10^5 N/ m^2

A = 8.55 cm^2 = 8.55x 10^{-4} m^2

μ = 1.002 x 10^{-3} N.S./m^2 (for 20 °C water)

W is calculated from the following equation

$$W = \frac{C_0 C_k}{100(C_k - C_0)}$$

C_0 = Feed solids concentration (per cent)

C_k = Cake solids concentrations (per cent)

C_k was calculated from

$$C_k = \frac{Dry\ Solids}{Wet\ Solids} \times 100$$

b was obtained from the slope of t/V versus V plot.
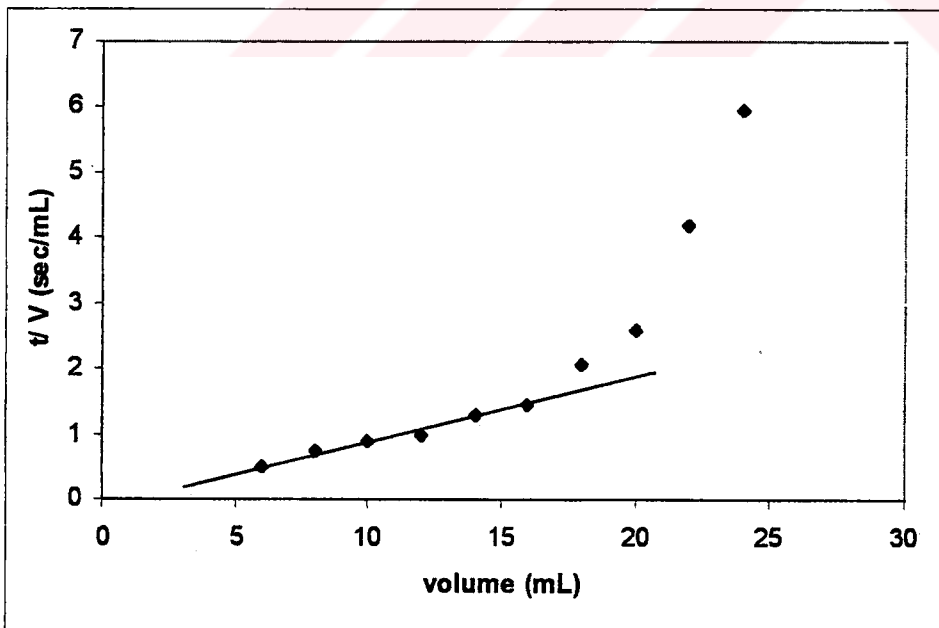
FİGURE 3.1. Typical Plot of t/V versus V in which b was obtained

$C_k = 16.91\ \%$

$C_0 = 0.444\ \%$

$W = 4.559\ kg/m^3$

$b = 0.1377\ h/m^6$

$$r = \frac{2 \times 1.242 \times 10^5 \times (8.55 \times 10^{-4})^2 \times 0.1377}{1.002 \times 10^{-3} \times 4.559} \times 10^{12}$$

$r = 5.482 \times 10^{12}\ m/kg$

4. RESULTS AND DISCUSSIONS

4.1. Co-Treatment of Landfill Leachate and Domestic Wastewater

4.1.1. Batch Runs Carried out with Leachate A and Domestic Wastewater

Leachate A was used in RUNS 1 and 2. These runs served as a preliminary work for further experiments. In RUN 1 the leachate ratio was kept at 5 %. In RUN 2 it was kept at 20 % in order to investigate this extreme case. The PAC dosage was as 1000 mg/L in both runs. The PAC addition at 1000 mg/L was more effective at the low leachate ratio of (5 %) than 20 % (Figure 4.1 and 4.2). The trend in RUN 1 was similar as in the treatment of domestic wastewater alone since the leachate ratio was low. 61% of the COD stemmed from domestic wastewater. After the removal of readily biodegradable substrates, the effect of PAC became more obvious. Then, the slowly biodegradable part of COD was removed and PAC was more effective in the removal of this fraction. Moreover, the effect of PAC depended on the contact the time of the wastewater and PAC. The adsorption equilibrium time for the same leachate had been determined about 3 days in a previous study [16].

In RUN 2, the volumetric leachate ratio was increased to 20 % in order to investigate the effect of extreme leachate addition. This high leachate ratio resulted in higher inert COD concentrations and therefore lower COD removal efficiencies. With the increase of leachate ratio, the non-biodegradable or the slowly biodegradable COD fraction of the feed wastewater increased dramatically. As a consequence, substrate removal was completed in a larger time span. The PAC dose of 1000 mg/L was not enough to stimulate COD removal at such a high leachate ratio (Figure 4.2).
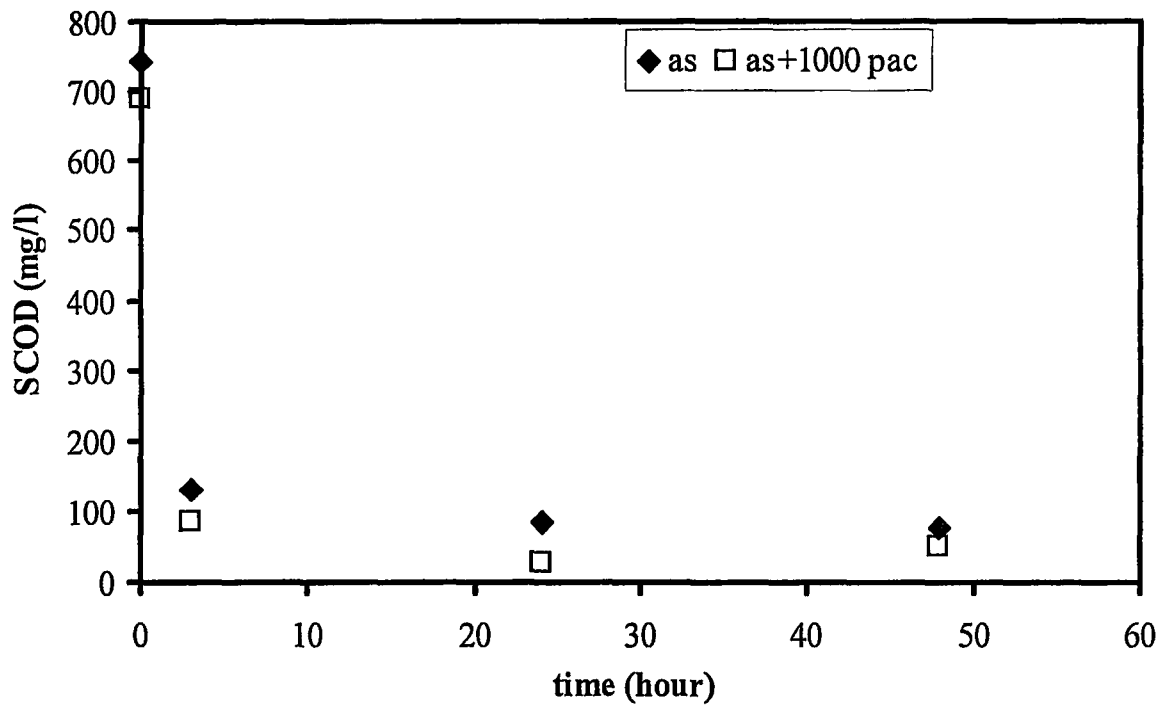
FIGURE 4.1. COD Profiles of RUN 1 (Leachate A, 5 % leachate ratio)
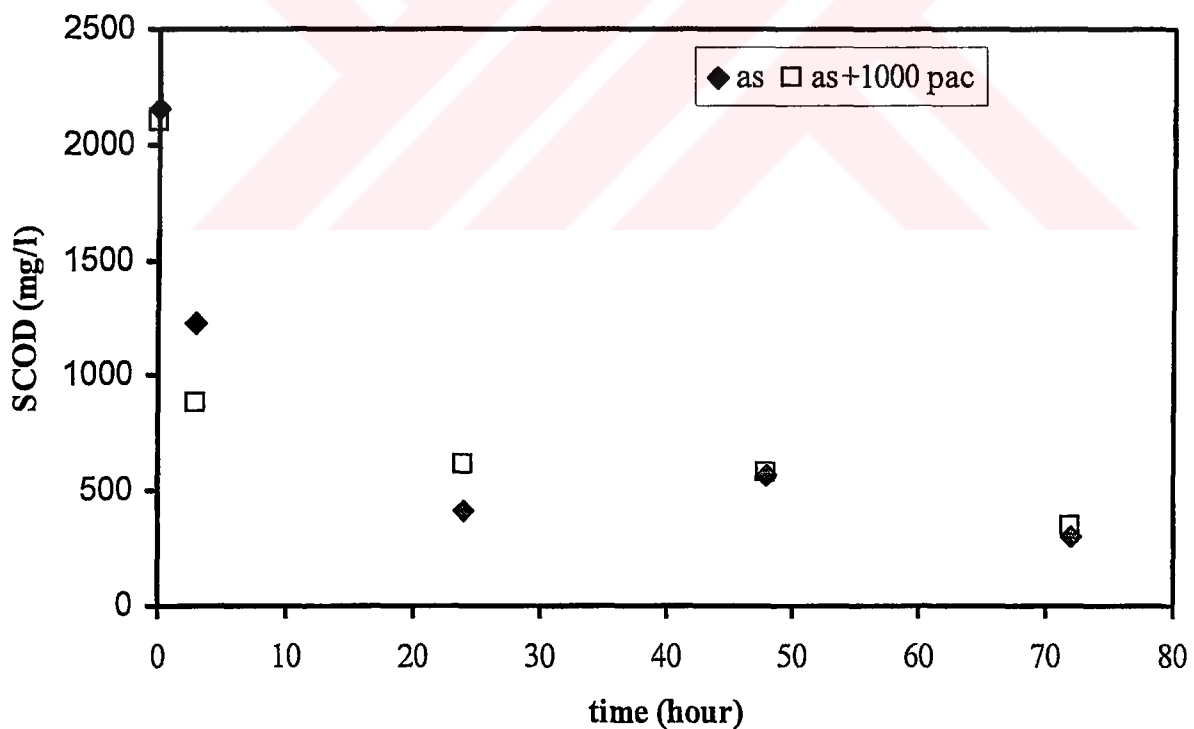
FIGURE 4.2. COD Profiles of RUN 2 (Leachate A, 20 % leachate ratio)

4.1.2. Batch Runs Carried out with Leachate B and Domestic Wastewater

After RUN 5, experiments were carried out with a new leachate (Table 3.1). In this part of the study the leachate ratio in the feed was kept in the range of 5-25 % (on volumetric basis) and the PAC dosage were varied from 1000 to 3000 mg/L (Figure 4.3-4.14).

As seen in Table 3.1, in RUNS 6 and 7 the leachate ratio was 5 % and 10 %, respectively. The dose of PAC was 1500 mg/L. The increase in leachate ratio resulted in higher inert COD and therefore the SCOD removal efficiency decreased from 97 % to 89 % in AS+PAC reactors. This indicated that the effect of PAC was dependent on the leachate percentage in the feed. The COD profiles (Figure 4.4 and 4. 5) show that the differences in AS and AS+PAC reactors were more obvious after 24 hours of aeration.

Throughout RUNS 8-17, three reactors were operated (see Table 3.1). One of them was a control reactor (AS reactor) which contained no PAC. The others (AS+PAC reactors) received two different PAC dosages. The effect of PAC addition was most obvious at increased leachate ratios.

To some extent, PAC was more effective at high dosages. For example, in RUNS 10 and 11, the leachate ratio was about 25 % and the final COD concentrations were reduced to lower values at increased PAC dosages. The residual SCOD fell to 475 mg/L at a PAC dosage of 1500 mg/L, whereas that was as low as 92 mg/L in the case of 2500 mg/L PAC addition (Figure 4.8 and 4.9). However in some cases (RUN 11), there were no differences in residual SCOD between 2500 and 3000 mg/L PAC addition (Figure 4.9). This showed that additional PAC doses did not always result in additional COD removal because leachate may also contain non-adsorbable compounds that cannot be removed by PAC. Hence, an optimum PAC concentration should be determined.
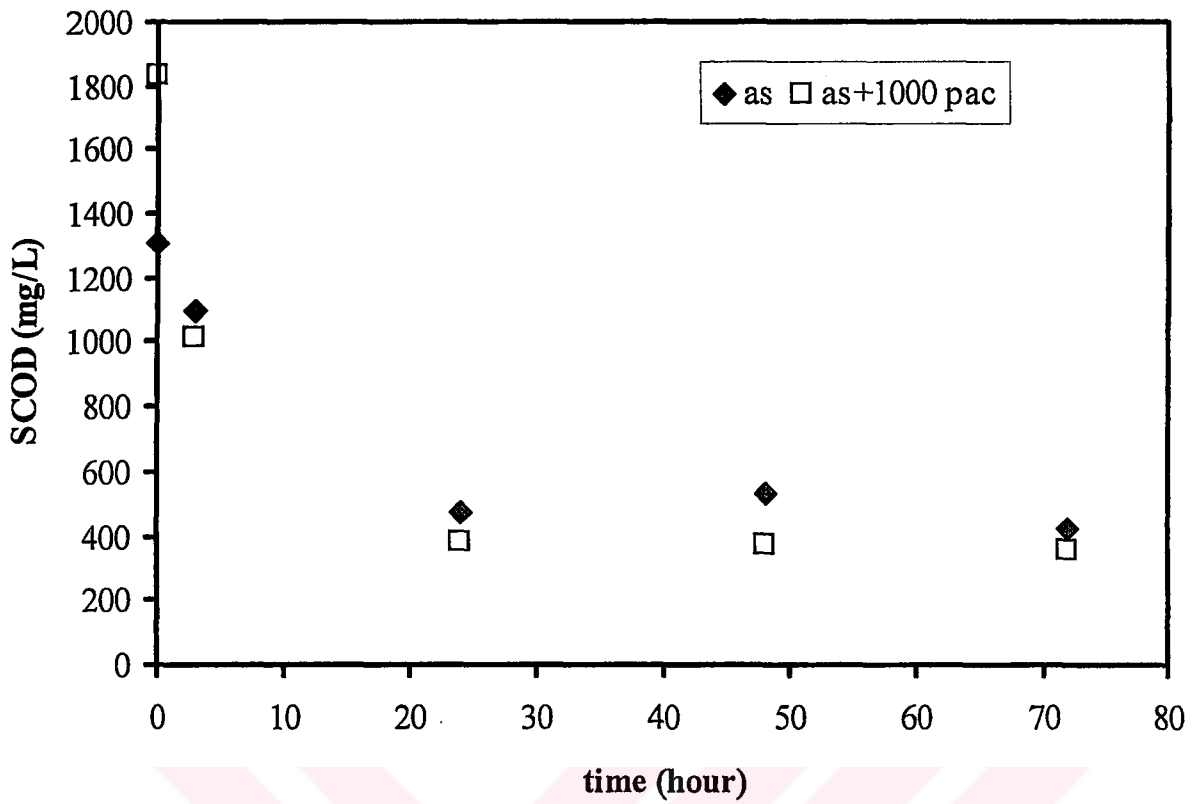
FIGURE 4.3. COD Profiles of RUN 5 (Leachate B, 5 % Leachate ratio)
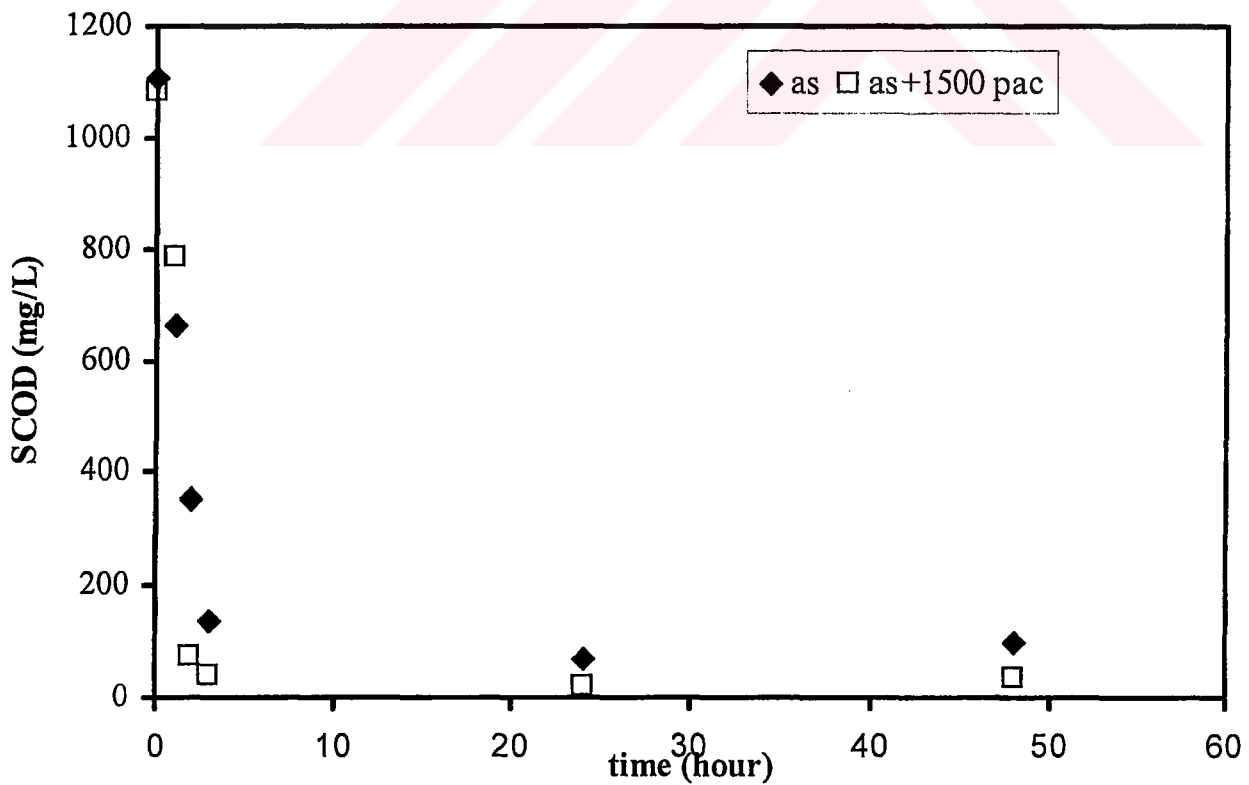
FIGURE 4.4. COD Profiles of RUN 6 (Leachate B, 5 % Leachate ratio)
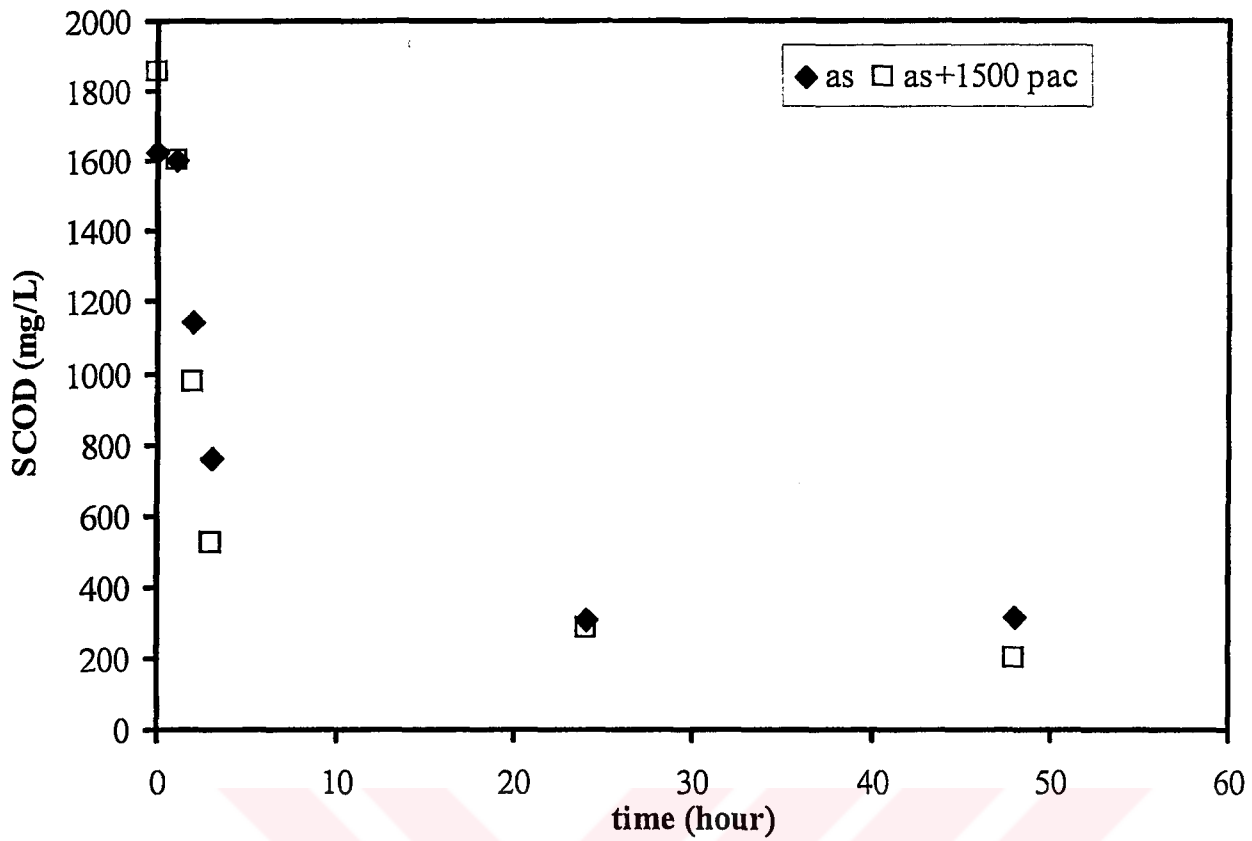
FIGURE 4.5. COD Profiles of RUN 7 (Leachate B, 10 % Leachate ratio)
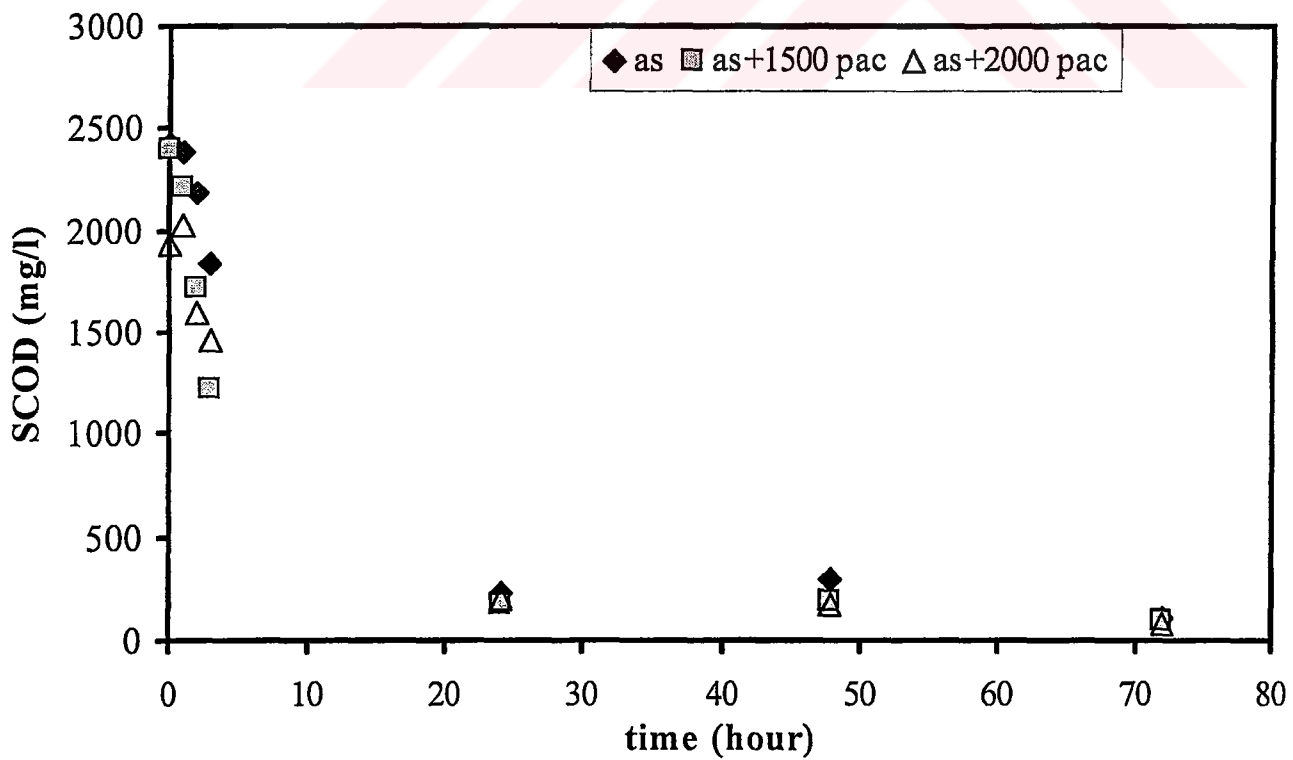
FIGURE 4.6. COD Profiles of RUN 8 (Leachate B, 15 % Leachate ratio)
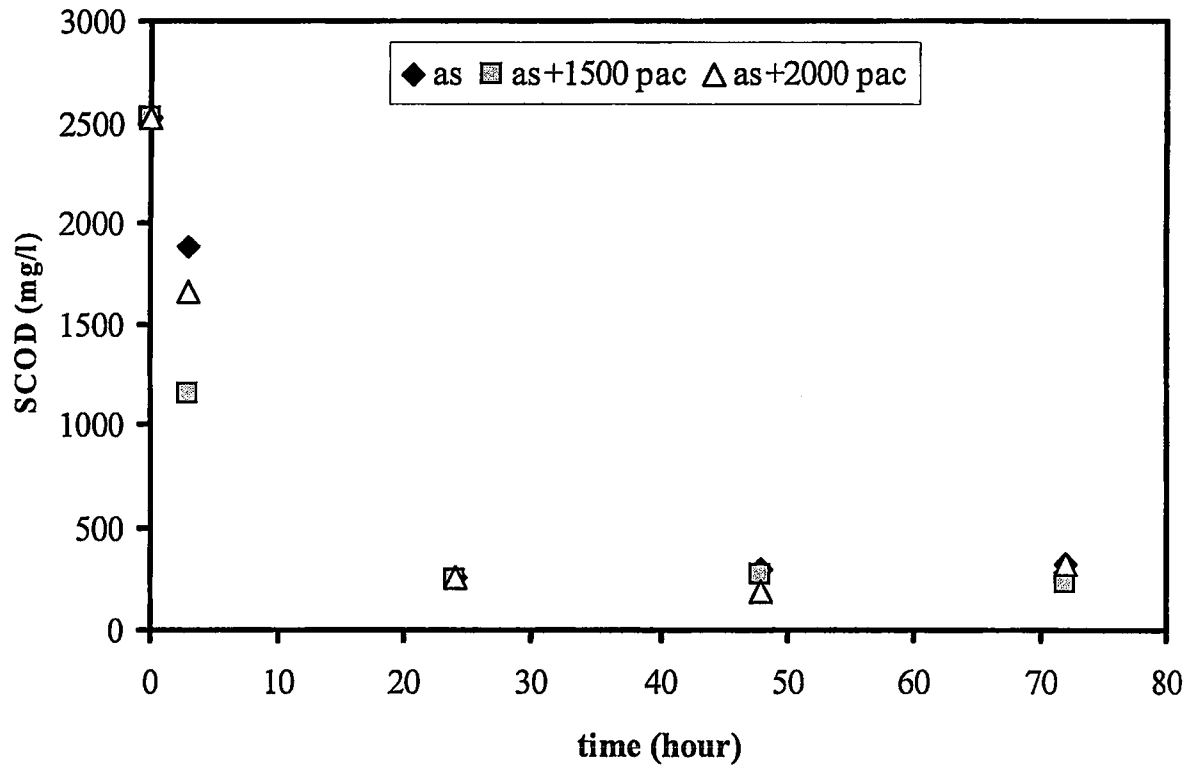
FIGURE 4.7. COD Profiles of RUN 9 (Leachate B, 20 % Leachate ratio)
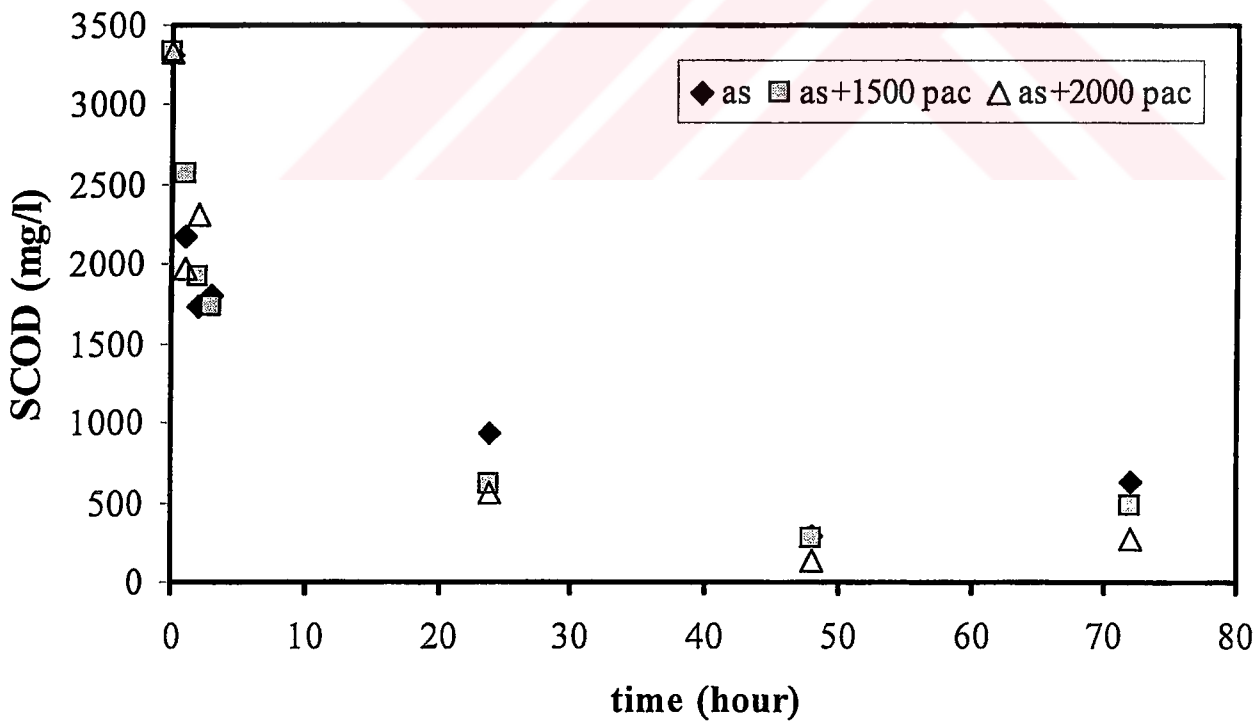
FIGURE 4.8. COD Profiles of RUN 10 (Leachate B, 25 % Leachate ratio)
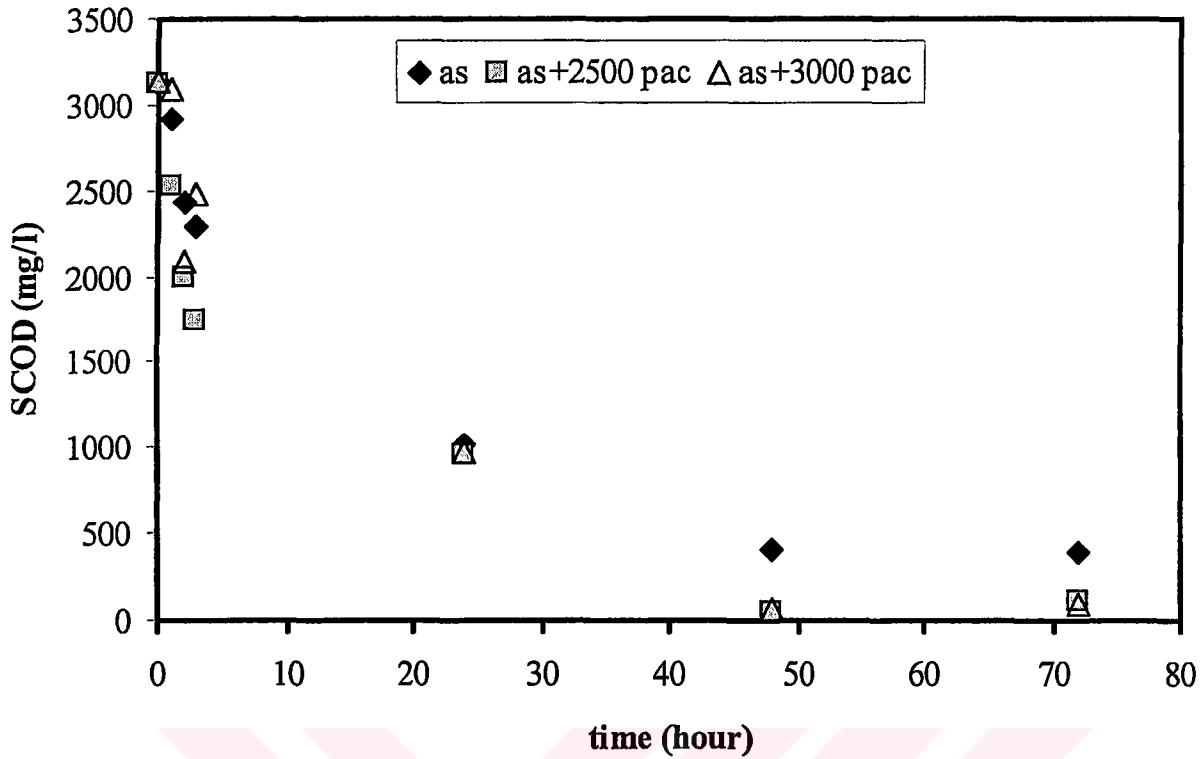
FIGURE 4.9. COD Profiles of RUN 11 (Leachate B, 25 % leachate ratio)
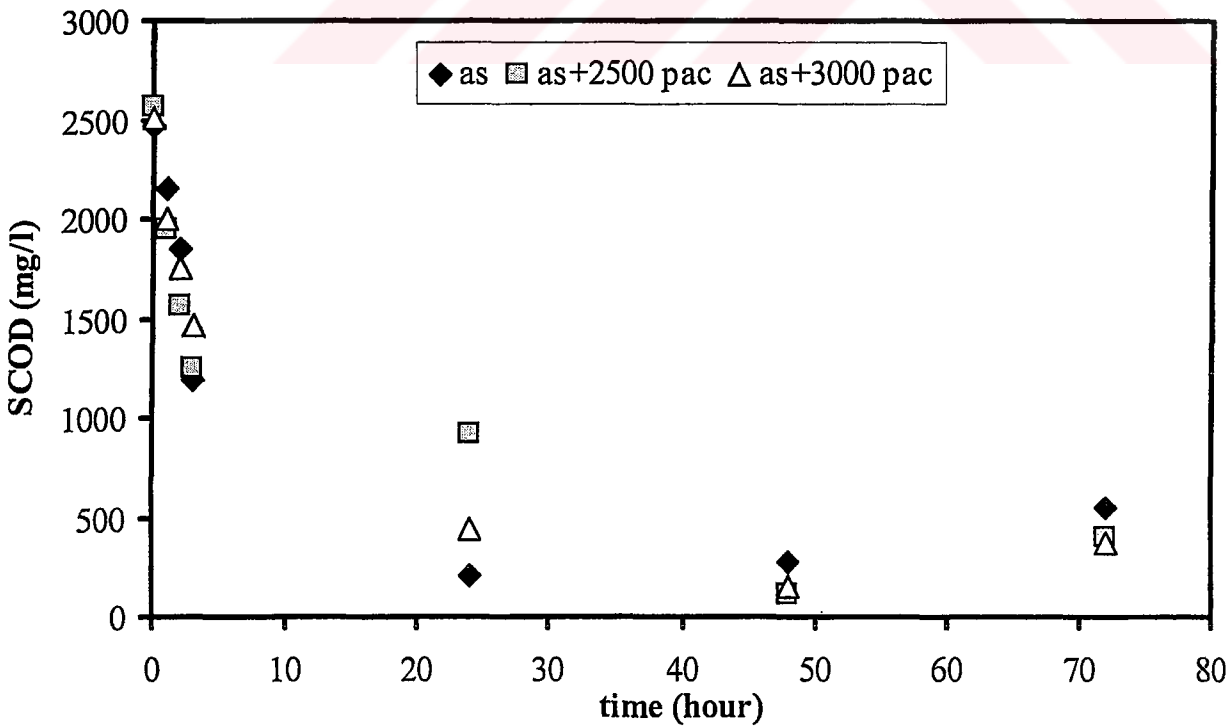
FIGURE 4.10. COD Profiles of RUN 13 (Leachate B, 20 % leachate ratio)
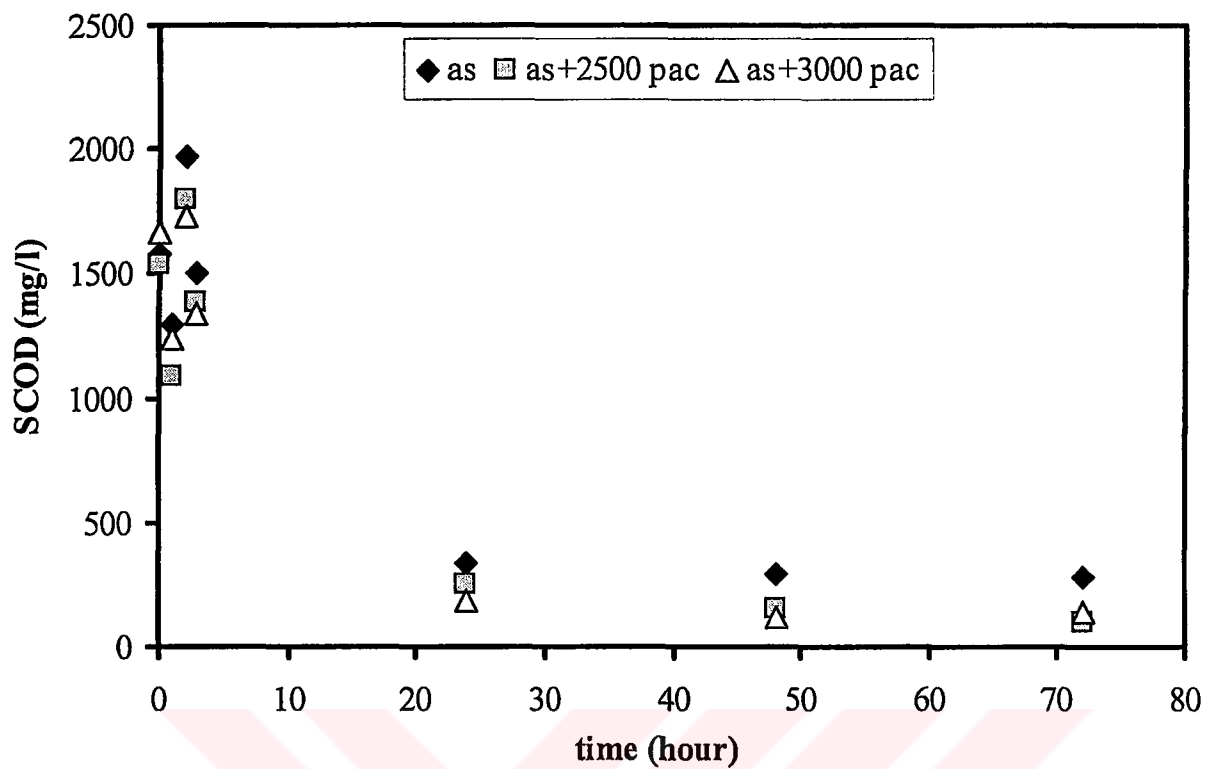
FIGURE 4.11. COD profiles of RUN 14 (Leachate B, 15 % Leachate ratio)
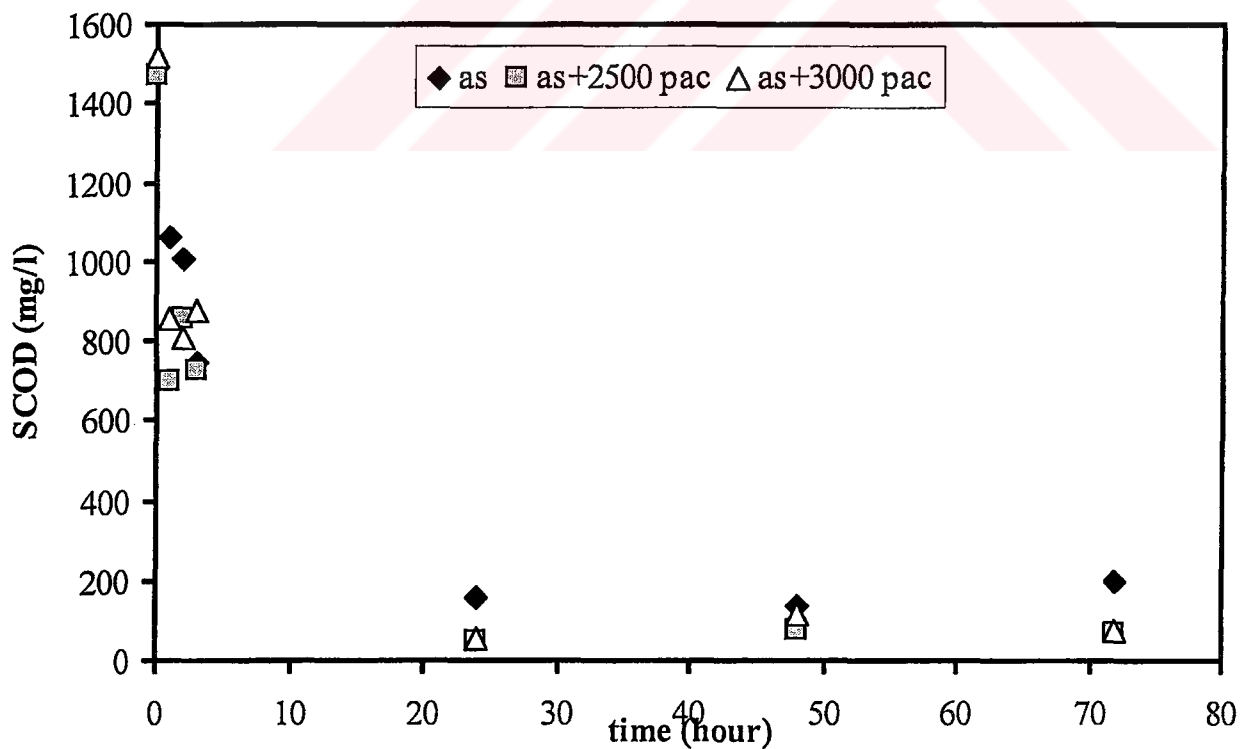
FIGURE 4.12. COD profiles of RUN 15 (Leachate B, 10 % Leachate ratio)
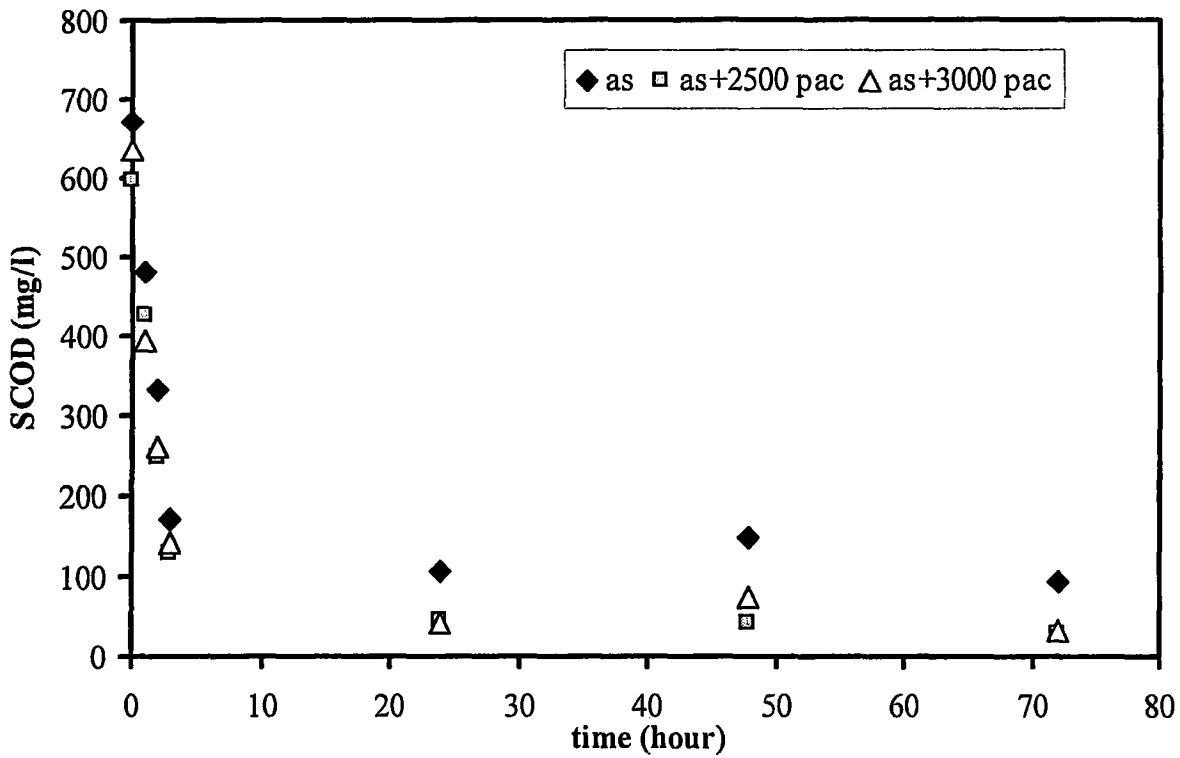
FIGURE 4.13. COD Profiles of RUN 16 (Leachate B, 5 % Leachate Ratio)
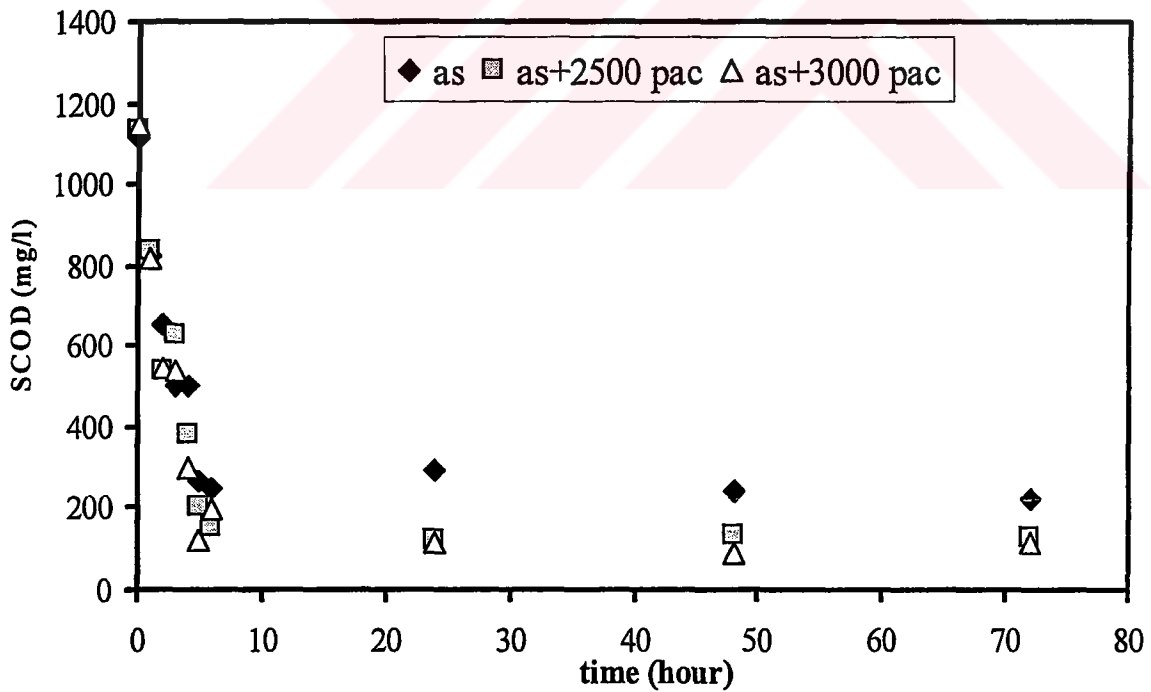
FIGURE 4.14. COD Profiles of RUN 17 (Leachate B, 10 % Leachate Ratio)

4.1.2.1. Temporary Increases in SCOD within the Aeration Period

As a general trend, in most of the runs (RUNS 1,2,3,5,6,8,9,10,11,12,13,14,15,16,17,18) an increase of SCOD concentration took place at the 48th or 72nd hours. This increase in SCOD may be due to several reasons. First, hydrolysis of slowly biodegradable materials in the leachate may lead to an increase in soluble COD. Secondly, the increase in SCOD may be a result of desorption from activated carbon or biomass. Desorption may take place due to a concentration gradient. This is especially the case when the COD concentration in the reactor becomes low enough [31]. The increase in COD concentration may also be due to the formation of soluble microbial products. These products may arise from growth-associated or decay-associated mechanisms [10].

Soluble microbial products are associated with biomass growth and substrate metabolism or with biomass decay [10,48]. In this study, the COD increase took place when substrate concentration in the reactor became low. Therefore, it was more likely that Soluble Microbial Products (SMP) formation took place due to decay associated mechanisms. In AS+PAC reactors, the SCOD increases at 48th and 72nd hour were usually lower than in AS reactors. Soluble Microbial Products (SMP) might be adsorbed onto PAC as stated in literature [48].

4.1.2.2. Comparison of Results at the same Initial PAC Doses

For comparison purposes, the results of the runs at PAC dosages of 2500 and 3000 mg/L were presented in Table 4.1 and Table 4.2. In these tables, the effect of PAC dose at different leachate ratios can be seen. At all leachate ratios, the final COD values in PAC added reactors were always lower than the control AS reactors. The difference between AS and AS+PAC reactors reached about 300-400 mg/L COD with an increase in leachate ratio. As the leachate ratio increased, the COD removal efficiency in the AS+ PAC reactors remained at a high level, but in the AS reactors a decrease was seen. This indicated that the presence of leachate had an inhibitory effect and with PAC addition this inhibitory effect could be prevented. Moreover, the reduction in residual COD by PAC was higher at higher

TABLE 4.1. Comparison of the SCOD results at PAC dose of 2500 mg/L

Run No	Leachate Ratio %	Reactor	PAC Dosage (mg/L)	Initial SCOD mg/L	Final SCOD mg/L	SCOD removal %
16	5	AS	0	670	95	86
16	5	AS+PAC	2500	598	29	95
15	10	AS	0	1492	200	87
15	10	AS+PAC	2500	1467	68	95
14	15	AS	0	1575	280	82
14	15	AS+PAC	2500	1538	99	94
13	20	AS	0	2466	548	78
13	20	AS+PAC	2500	2556	112	95
11	25	AS	0	3135	399	87
11	25	AS+PAC	2500	3128	109	97

TABLE 4.2. Comparison of the SCOD results at PAC dose of 3000 mg/L

Run No	Leachate Ratio %	Reactor	PAC Dosage (mg/L)	Initial SCOD mg/L	Final SCOD mg/L	SCOD removal %
16	5	AS	0	670	95	86
16	5	AS+PAC	3000	637	31	95
15	10	AS	0	1492	200	87
15	10	AS+PAC	3000	1516	75	95
17	10	AS	0	1114	221	80
17	10	AS+PAC	3000	1144	113	90
14	15	AS	0	1575	280	82
14	15	AS+PAC	3000	1660	118	93
13	20	AS	0	2466	548	78
13	20	AS+PAC	3000	2506	156	94
11	25	AS	0	3135	399	87
11	25	AS+PAC	3000	3133	92	97

leachate ratios. This showed that PAC was more effective at high leachate ratios. In these cases, PAC increased the removal by adsorbing either toxic substances that caused an inhibition or slowly biodegradable adsorbable materials in leachate.

4.1.2.3. Comparison of SCOD Decreases in AS and AS+PAC Reactors

In the Figures 4.15 and 4.16, the general trend of COD removal in AS and AS+PAC reactors is illustrated. There remained an inert SCOD and the amount was related to the leachate ratio. The addition of PAC caused a decrease in this inert COD (Figure 4.16). Similar results were observed in a previous study [16, 17, 18, 19] and the positive effect of PAC was observed in the later of the aeration where inert SCOD become adsorbed onto PAC [19].
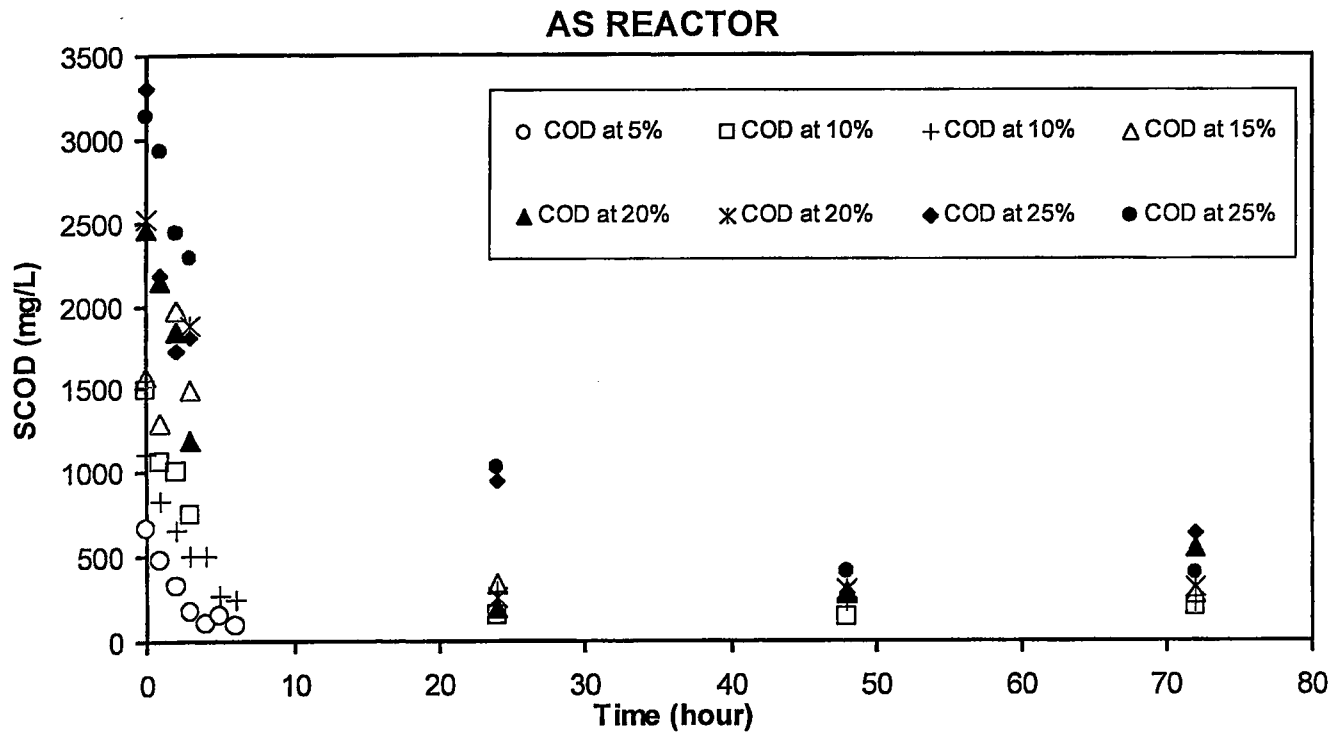
AS REACTOR

FIGURE 4.15. The SCOD Profiles in AS (Activated Sludge) Reactors in which the Leachate ranged from 5-25 %.
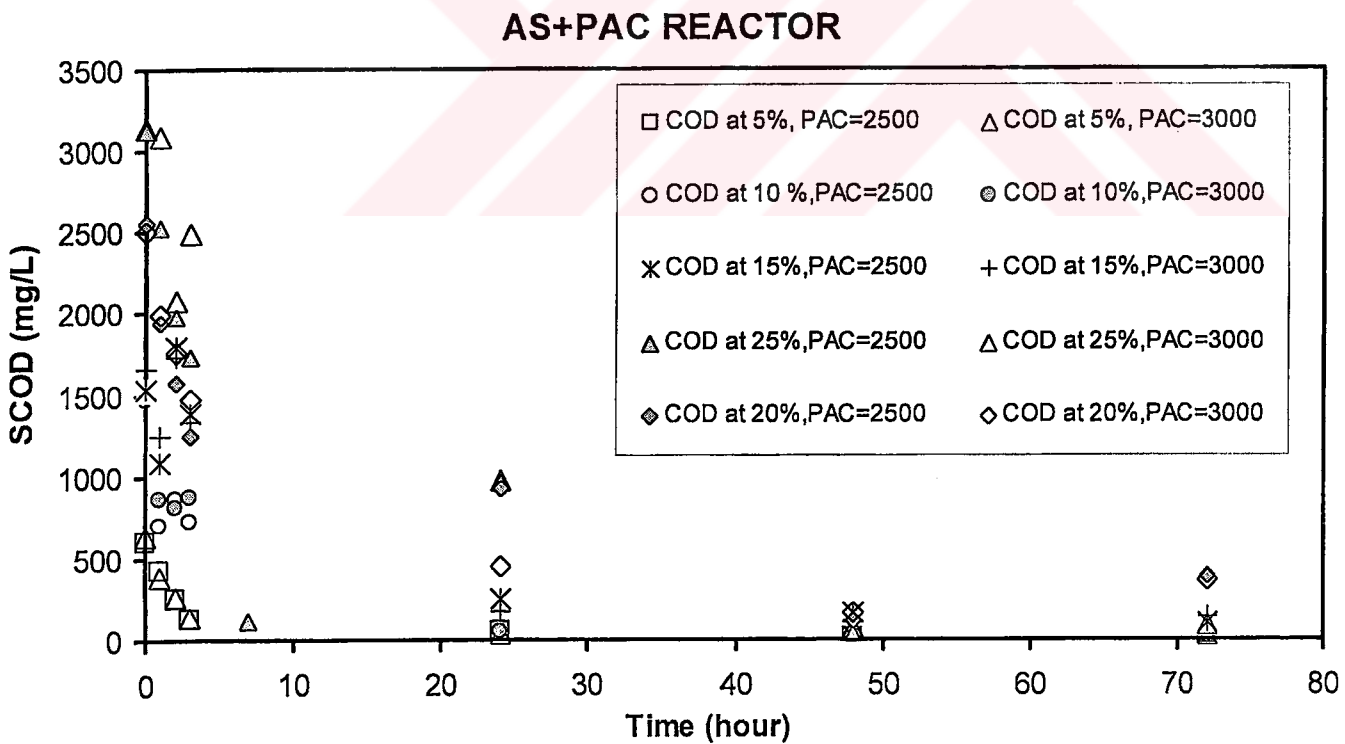
AS+PAC REACTOR

FIGURE 4.16. The SCOD Profiles in AS+PAC (PAC Added Activated Sludge) Reactors in which the Leachate ranged from 5-25 %.

4.2. Treatment of Synthetic Domestic Wastewater in Batch Reactors

In RUNS 3, 4 and 19 only domestic wastewater was used. These runs served as control runs for co-treatment studies. Stock synthetic domestic wastewater was diluted to about 3700, 1500 and 2400 mg/L SCOD in RUNS 3,4 and 19, respectively. The results were compared to leachate treatment.

The major part of COD was removed in the first hours (especially in 24 hours). After 24 hours, SCOD fell to negligible levels and the concentrations remained at a constant level. (Figure 4.17, 4.18 and 4.19).

In contrast to co-treatment and treatment of leachate alone, PAC addition did not enhance COD removal. The reason was that synthetic domestic wastewater was mainly composed of readily biodegradable and non-adsorbable organic matter. It is known that the effect of PAC is more obvious in the removal of non-biodegradable, adsorbable, and slowly biodegradable compounds [25]. Therefore, PAC addition was not required in domestic wastewater treatment.

In RUN 19, COD was monitored hourly within the first six hours to detect the variations because the major part of COD removal took place in the first hours. The removal of readily biodegradable substrate was achieved in the first 5-6 hours. In this run COD concentrations in the 6th and 24th hours are almost the same (Figure 4.19)
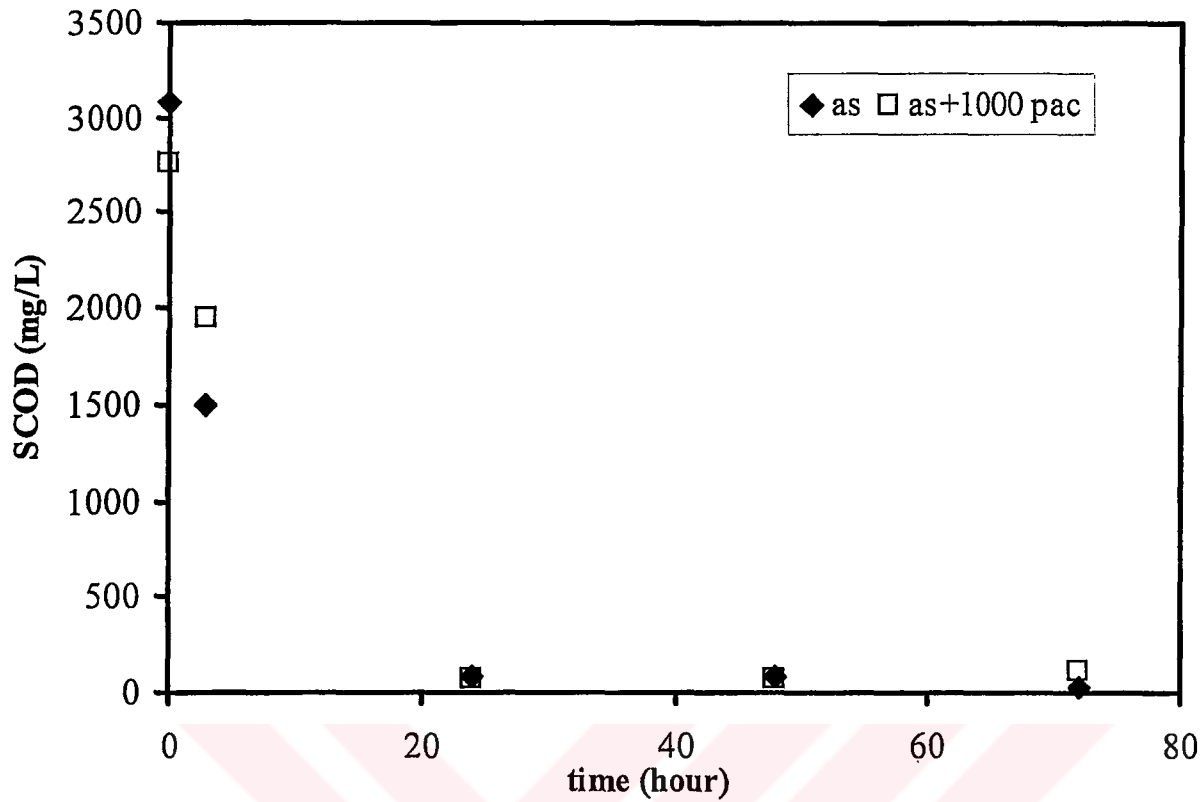
FIGURE 4.17. COD Profiles of RUN 3 (Domestic Wastewater Alone)
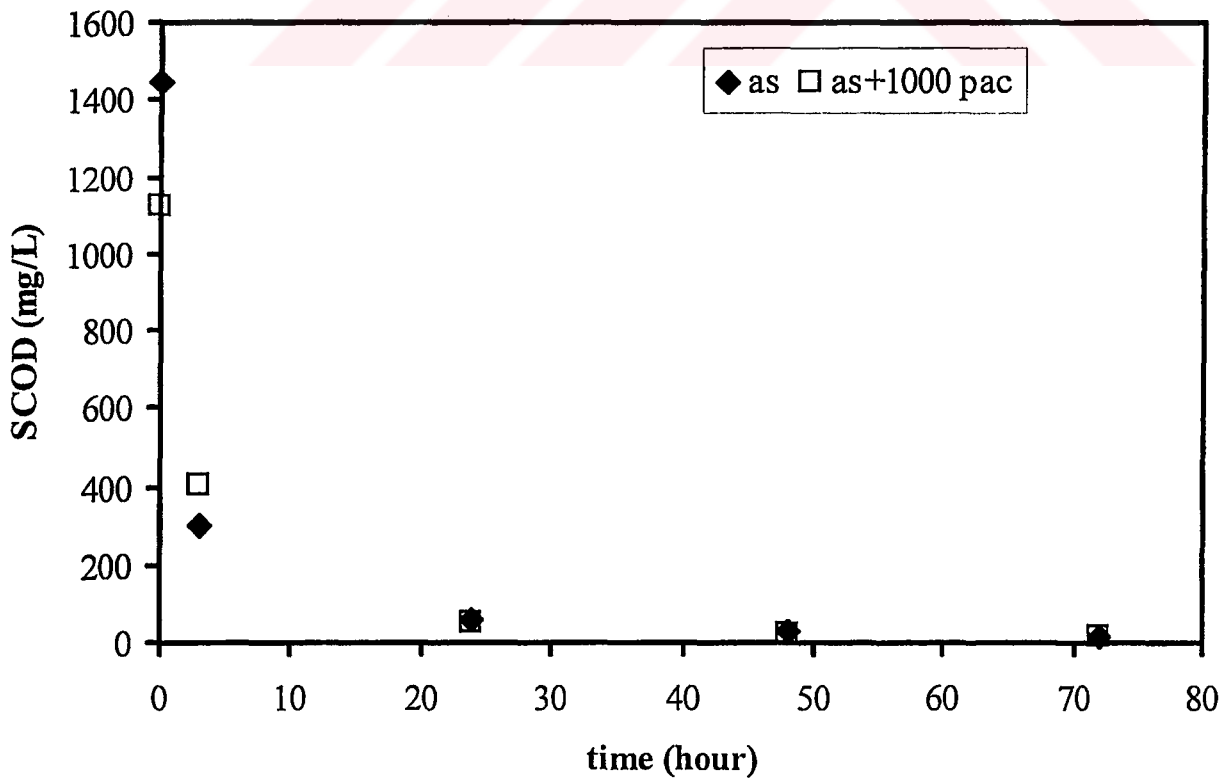
FIGURE 4.18. COD Profiles of RUN 4 (Domestic Wastewater Alone)
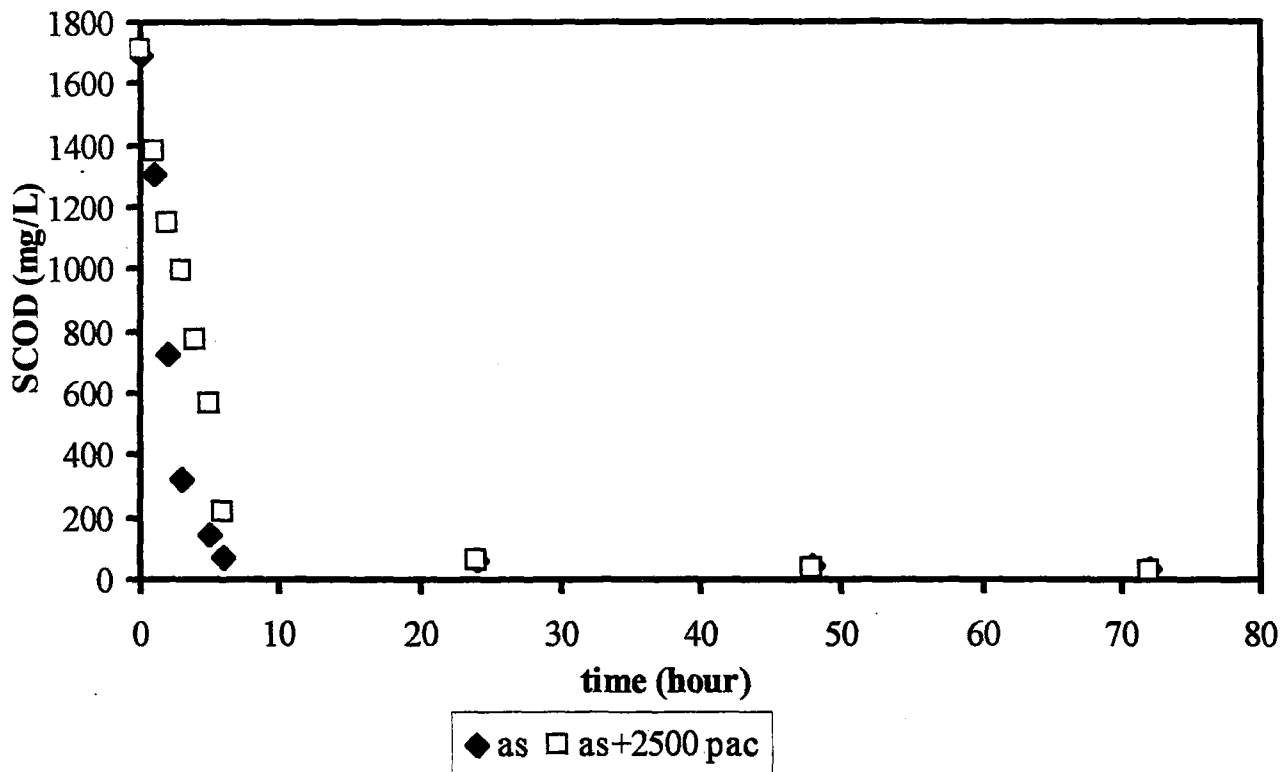
FIGURE 4.19. COD profiles of RUN 19 (Domestic Wastewater Alone)

4.3. Treatment of Leachate alone (Leachate B) in Batch Activated Sludge Reactors

RUNS 12 and 18 were performed with diluted leachate without the addition of domestic wastewater. The aim was to investigate the effect of leachate wastewater in the co-treatment of leachate and domestic wastewater. The results showed that diluted leachate could be treated without severe inhibition. Moreover, high COD removal was observed even at high leachate ratios (20-25 %).

In the treatment of leachate alone, significant COD removal was achieved both with and without PAC addition (Figure 4.20 and 4.21). On the other hand, the sludge characteristics were negatively affected when the feed was composed of leachate only. The possible reason was that the presence of domestic wastewater supplied enough nutrients, diluted the effects of toxic materials and heavy metals [9].
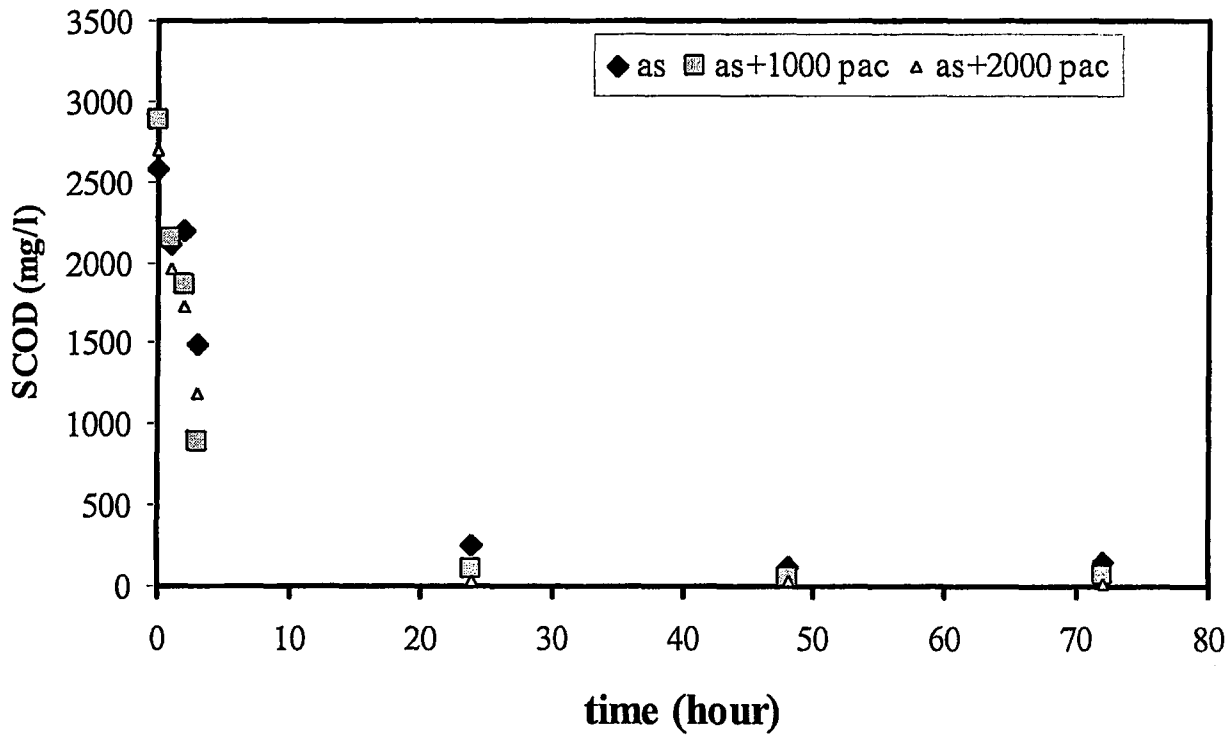
FIGURE 4.20. COD Profiles of RUN 12 (Leachate Alone)
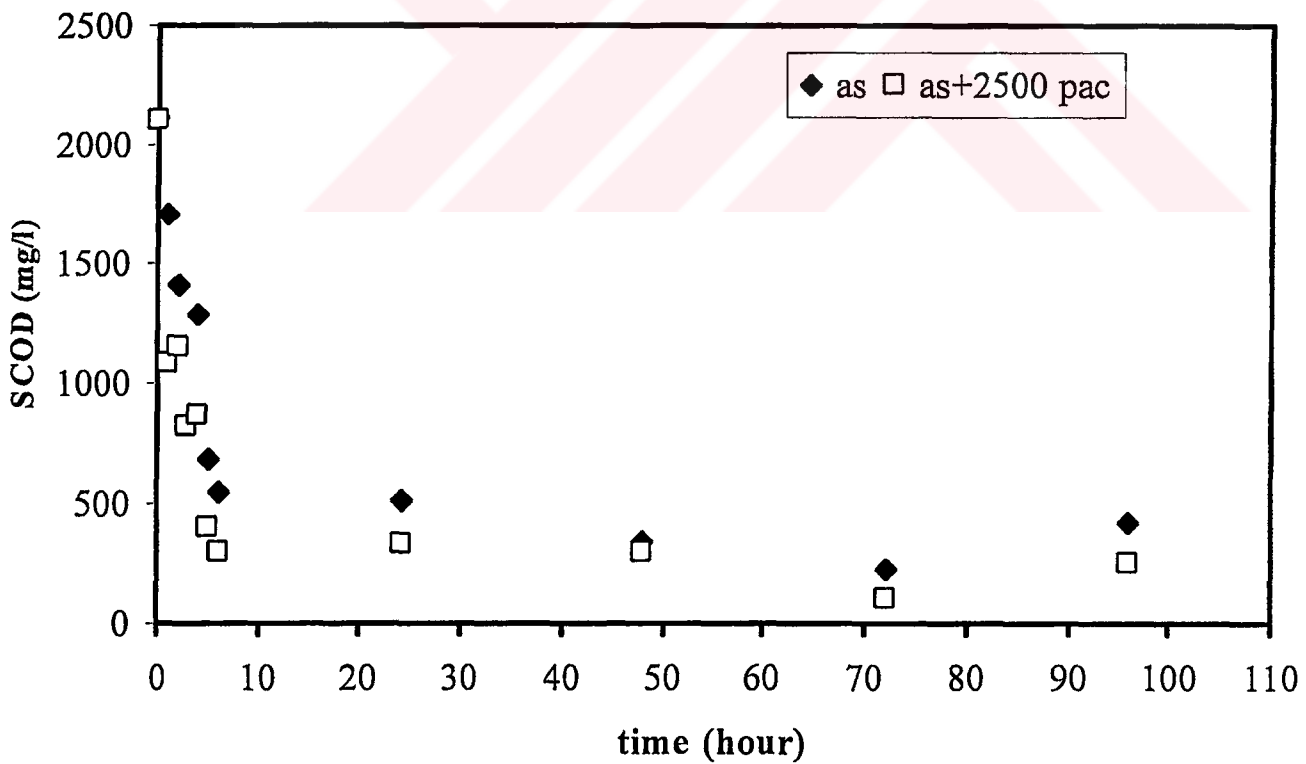
FIGURE 4.21. COD Profiles of RUN 18 (Leachate Alone)

4.4. Sludge Characteristics with and without Powdered Activated Carbon (PAC) Addition

Throughout the runs, the settling properties of activated sludge were determined by Sludge Volume Index (SVI) and Batch Settling Curve. The dewaterability of the sludge was determined by Specific Resistance to Filtration (SRF) measurements.

4.4.1. SVI (Sludge Volume Index) Measurements:

Results of all runs showed that the SVIs in the AS+PAC reactors were always lower that those in AS reactors (Data are shown in Appendices B1, B2, B3, and B4). In SVI measurements, the volume of sludge settled in 30 minutes in the AS and AS+PAC reactors were almost the same (Data are not given) In some cases this was even lower in AS+PAC reactors than in the AS reactors. However, in AS+PAC reactors more solids occupied this volume since this sludge also contained PAC particles. Therefore, lower SVI values were obtained in AS+PAC reactors. The presence of more solids in a smaller volume indicated that PAC addition led to a compaction of sludge.

4.4.2. Settling Curves:

In addition to SVI measurements, batch settling tests were conducted to have an idea about the settling characteristics of activated sludge. However, settling tests are not very reliable since they are based on visual examinations rather than measurements. In AS+PAC reactors, no reliable data could be obtained because of the dark black colour caused by the PAC particles. Therefore batch settling tests were only interpreted for the case of AS. Based on visual examinations, addition of leachate resulted in smaller flocs compared to the control reactor which was fed with domestic wastewater alone. In Figures 4.22 and 4.23, the settling curves belonging to RUN 15 and 16 were presented, respectively. As seen in the figures, as the run proceeded, the time required for reaching the same sludge-water interface height changed, indicating a change in the settling properties. Usually after 24 hours, settling was improved. However, at the 24th hour, particularly in the runs carried out at high leachate ratios settling velocity (the slope of the settling curve) slightly decreased indicating that settling properties worsened. Settling curves for other runs are presented in Appendix C.
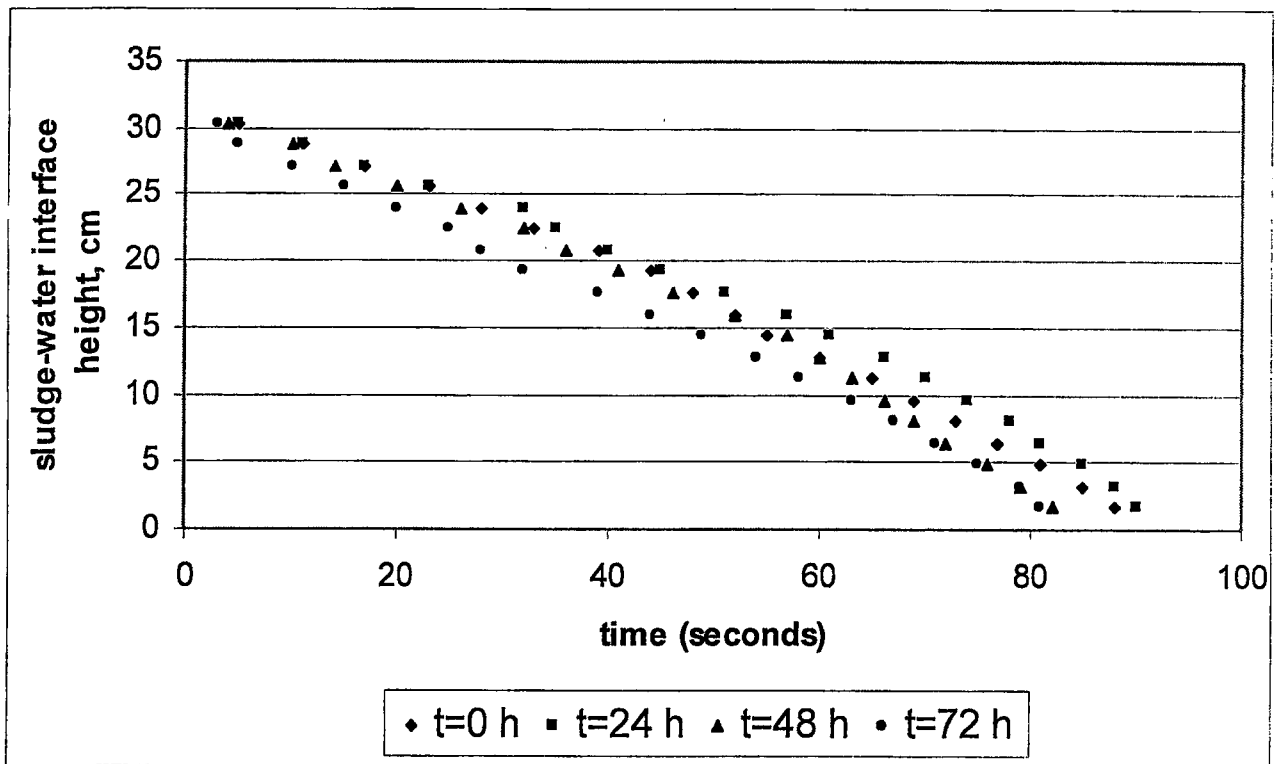
FIGURE 4.22. Settling Curves belonging to the AS Reactor in RUN 15 (10 % Leachate Ratio)
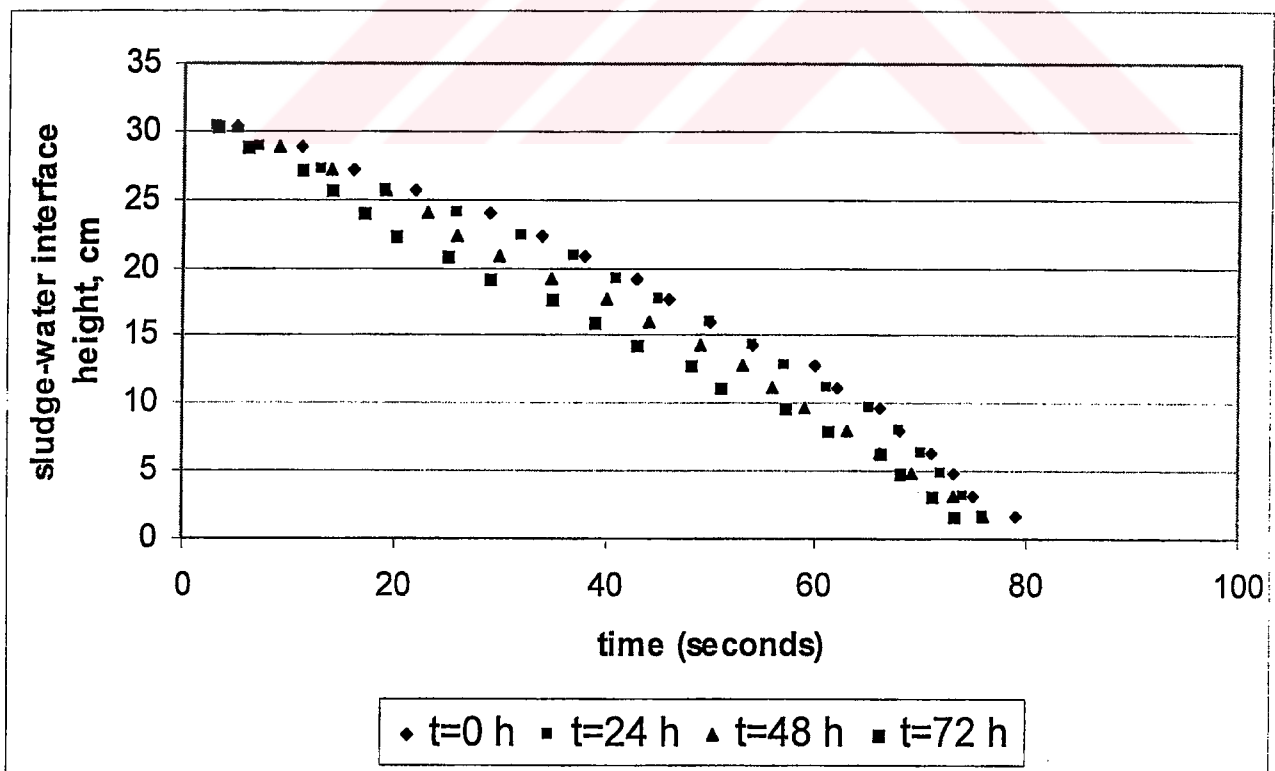
FIGURE 4.23. Settling Curves belonging to the AS Reactor in RUN 16 (5 % Leachate Ratio)

4.4.3. Evaluation of Sludge Dewaterability based on SRF Measurements

Specific Resistance to Filtration (SRF) measurement is a more reliable way of determining the effect of PAC addition. For control purposes, also the SRFs of the leachate (diluted with tap water) and the PAC solutions alone (prepared with tap water) were measured. The concentrations of PAC and the leachate were kept as in the other runs. The results are presented in Table 4.3 and Table 4.4. The SRFs of the PAC solutions were very small with respect to leachate and tended to decrease with increasing PAC concentrations. On the other hand, the SRF of the leachate was very high if the leachate concentration was high.

TABLE 4.3 The SRF of Various PAC solutions

PAC Concentration (mg/L)	SRF x10^{12} (m/kg)
1000	1.61
1500	0.78
2000	1.40
2500	0.46
3000	0.34

TABLE 4.4 The SRF of Leachate diluted to various strength

% of Leachate	SRF x 10^{12} (m/kg)
5	2.73
10	10.64
15	3.51
20	15.11
25	63.89

In RUN 8 and succeeding runs, in both AS and AS+PAC reactors SRF was measured at the 24th hour and at the 72nd hour. But, in RUN 8, the filter paper used was different (having 0.45 µm pore size). Due to smaller pore size, filter paper was clogged in a short time and the filtration time for the sample was longer. Due to that reason, very high SRF values were obtained during the first hours of RUN 8. Therefore, the results of RUN 8 could not be compared to others. All SRF values were presented in Appendices B.2, B.3, and B.4.

The results showed that the dewaterability of activated sludge was closely related to the leachate ratio in the feed. The higher leachate ratio in the feed resulted in higher SRF. In other words, dewaterability of sludge decreased with an increase in leachate ratio. Also, the maximum SRF values were higher if the leachate ratio was high. The effect of leachate ratio on sludge dewaterability can be clearly seen in Figure 4.24 and 4.25 in AS and AS+PAC reactors, respectively. All runs except RUNS 10 and 11 (with 25 % leachate ratio) showed the same trend: an increase in SRF at the 24th hour and a decrease in SRF at the 72nd hour. During the removal of readily biodegradable substrates, SRF tended to increase. After depletion of readily biodegradable substrate, SRF decreased to the initial level.

AS Reactor

—△— SRF at 5%   - -○- - SRF at 10%   — ■ — SRF at 15%   — -◇- ·· SRF at 25%   - -※- - SRF at 20%
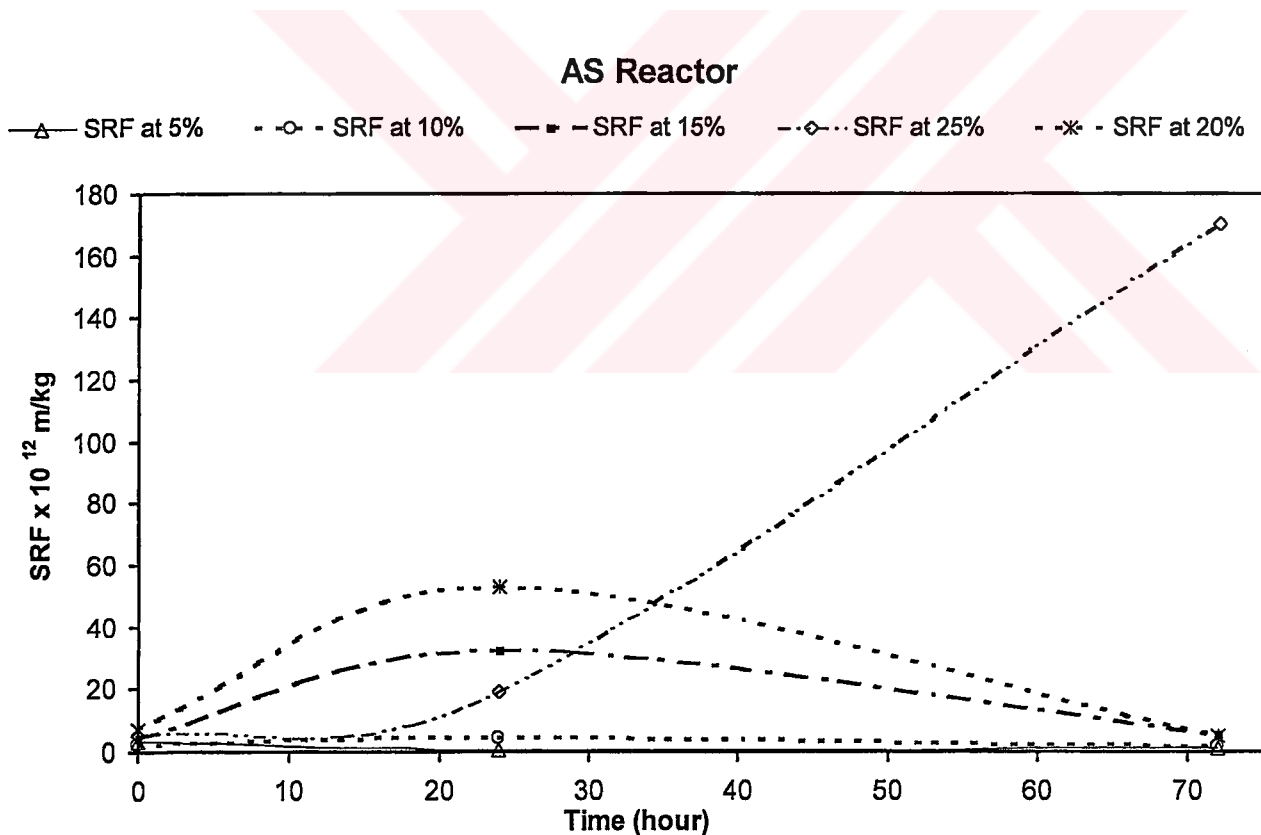
FIGURE 4.24. Change of Specific Resistance to Filtration (SRF) with respect to Time in AS Reactors

AS+PAC Reactor

——■—— SRF at 5%,PAC= 2500	···o··· SRF at 5%,PAC= 3000
——✳—— SRF at 15%,PAC= 2500	——◆—— SRF at 15%,PAC=3000
——●—— SRF at 20%, PAC= 2500	——o—— SRF at 20%,PAC= 3000

——■—— SRF at 10%,PAC= 2500	——▲—— SRF at 10%,PAC= 3000
- -+- - SRF at 25%,PAC= 2500	——▲—— SRF at 25%, PAC= 3000
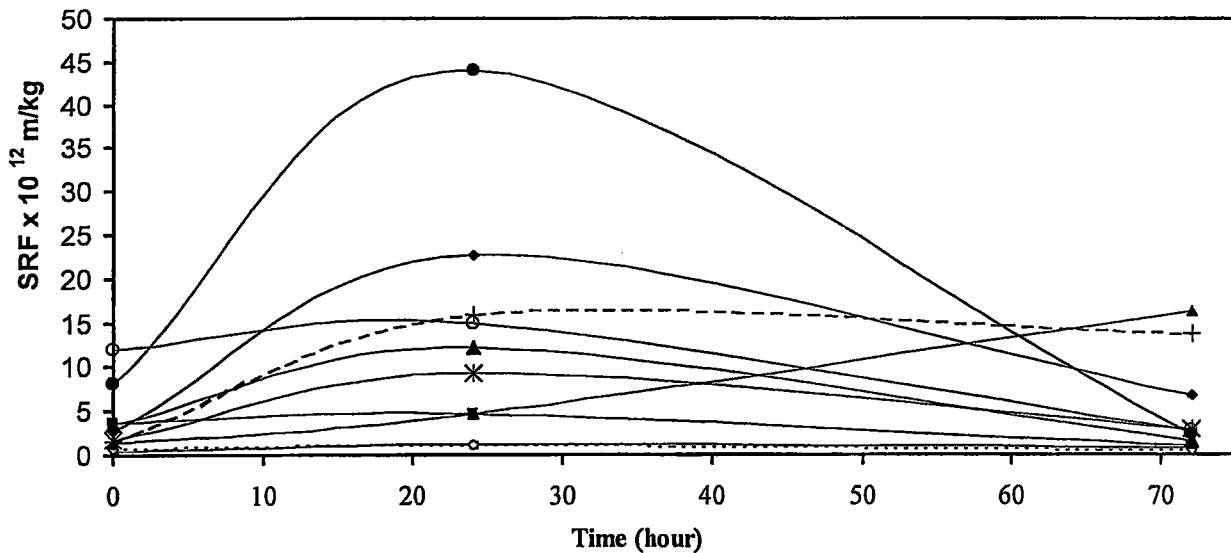
FIGURE 4.25. Change of Specific Resistance to Filtration (SRF) with respect to Time in AS+PAC (PAC Added Activated Sludge) Reactors

RUNS 10 and 11 were performed to determine the effect of high leachate contribution (25 %). In these runs, the increase in SRF lasted till 72nd hour. This increase in SRF at the 72nd hour can also be explained by a delay of substrate consumption. Therefore it can be concluded that there is a clear relationship between COD removal and sludge characteristics.

4.4.4. Relationship between Substrate Removal and SRF

In the Figure 4.26 and 4.27, this relationship can be clearly seen. These figures indicate that in both AS and AS+PAC reactors the removal of readily biodegradable substrates was achieved in the first hours. The residual COD concentrations were strongly related to the leachate amount in the feed. The increase of SRF in the first hours can be attributed to the increase in growth of filamentous organisms as a result of high amount of readily biodegradable substrates. High amounts of readily biodegradable substrates changed the initial F/M ratio and so the bacterial composition in the system. After depletion of readily biodegradable COD, the remaining part of COD consists of slowly biodegradable and non-biodegradable compounds. In this case, filamentous bacteria may disappear and as a consequence a decrease in SRF was observed. In AS+PAC reactors, the peak SRF values were usually lower than in AS reactors (Figure 4.26). This indicated an improvement of dewaterability with PAC addition.

AS Reactors

▲ COD at 5%　　● COD at10%　　■ COD at 15% ·　　◆ COD at 25%　　✕ COD at 20%

—△— SRF at 5%　　···⊙··· SRF at 10%　　—·■·— SRF at 15%　　—◇— SRF at 25%　　···✕·· SRF at 20%
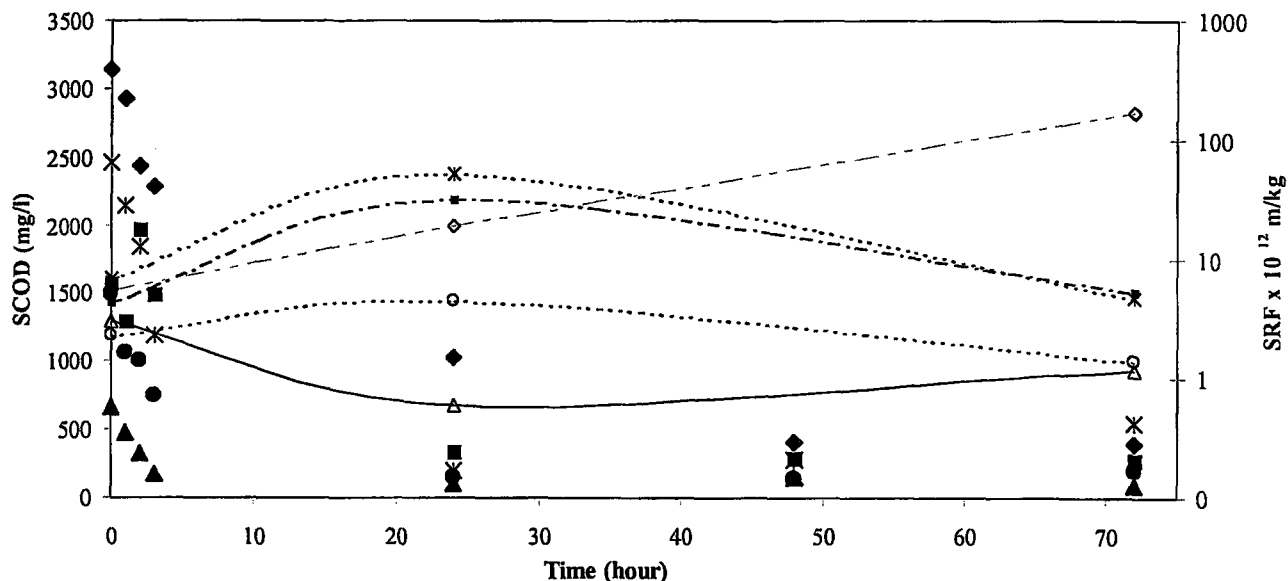
FIGURE 4.26. The Relationship between SCOD Decrease and the SRF of the Sludge in AS Reactors

AS+PAC Reactors

□ COD at 5%, PAC=2500	✕ COD at 5%, PAC=3000	✕ COD at 10 %,PAC=2500	✕ COD at 10%,PAC=3000
○ COD at 15%,PAC=2500	+ COD at 15%,PAC=3000	△ COD at 25%,PAC=2500	○ COD at 25%,PAC=3000
◇ COD at 20%,PAC=2500	✕ COD at 20%,PAC=3000	■— SRF at 5%,PAC= 2500	···◆··· SRF at 5%,PAC= 3000
—◆— SRF at 10%,PAC= 2500	···▲·· SRF at 10%,PAC= 3000	◇— SRF at 15%,PAC= 2500	···■·· SRF at 15%,PAC=3000
—◆— SRF at 25%,PAC= 2500	···■··· SRF at 25%, PAC= 3000	■— SRF at 20%, PAC= 2500	·· —■— SRF at 20%,PAC= 3000
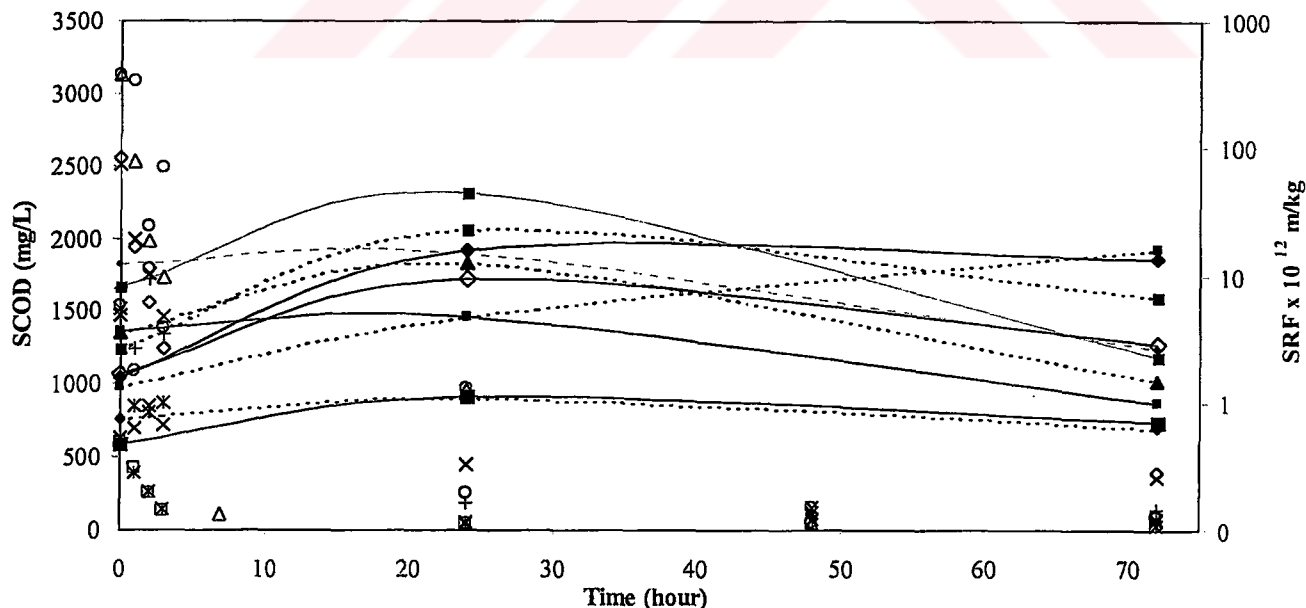
FIGURE 4.27. The Relationship between SCOD Decrease and the SRF of the Sludge in AS+PAC Reactors

4.4.5. Comparison of the Co-treatment Alternative with Leachate and Domestic Wastewater Treatment

Combined treatment of leachate with domestic wastewater was investigated in RUN 17. In RUN 18 and 19 diluted leachate and domestic wastewater were used, respectively. In these runs, the initial COD concentrations were about the same. Therefore, the results could be compared. The changes in SCOD and SRF are shown in Figure 4.28.
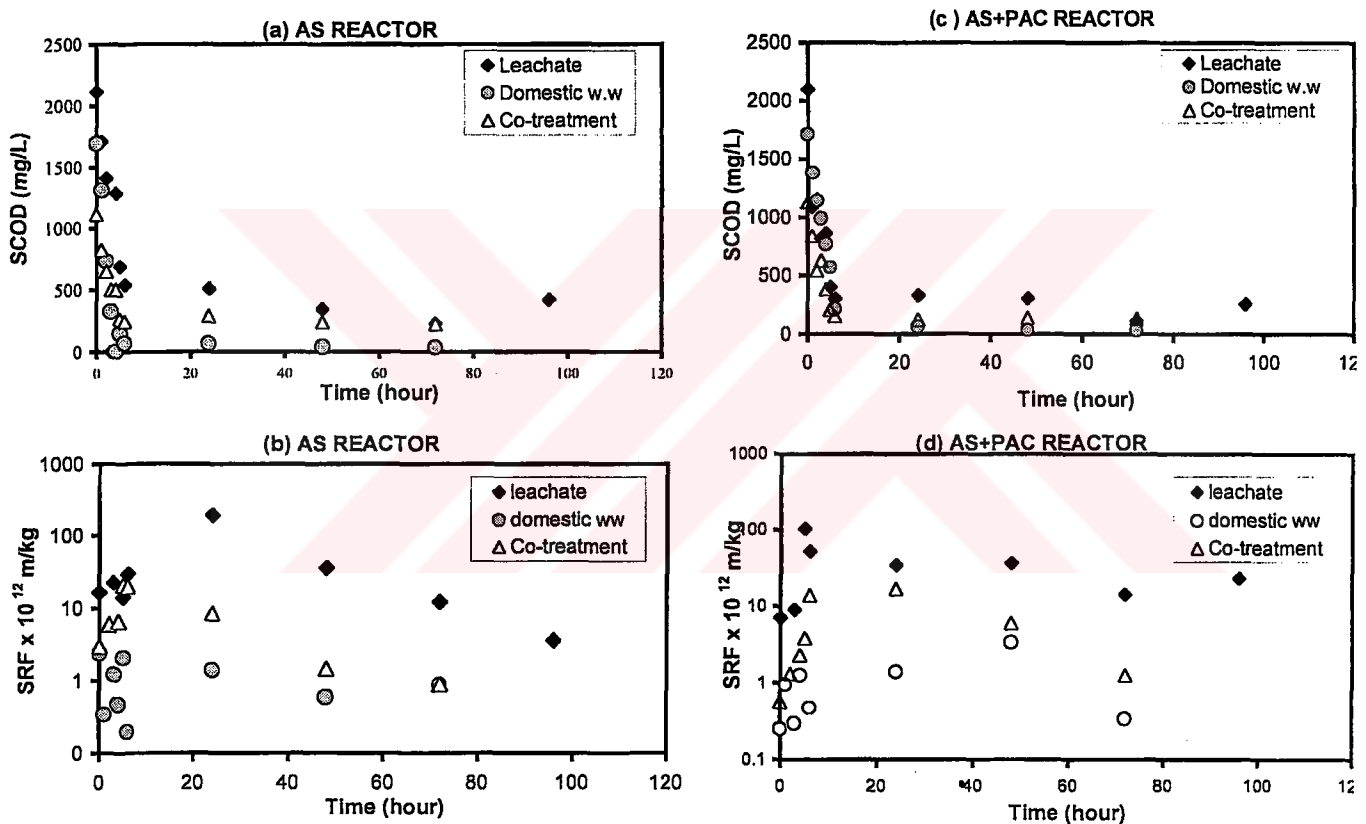
FIGURE 4.28. (a) SCOD in Domestic Wastewater Treatment, Leachate Treatment and the Co-treatment of Leachate and Domestic Wastewater in AS (Activated Sludge) Reactors (b) the corresponding SRFs (c) SCOD in Domestic Wastewater Treatment, Leachate Treatment and the Co-treatment of Leachate and Domestic Wastewater in AS+PAC (PAC Added Activated Sludge) Reactors (d) the corresponding SRFs

In the case of domestic wastewater treatment, SRF values were lower than leachate treatment or co-treatment of leachate and domestic wastewater. In domestic wastewater treatment, the SRF remained almost constant compared to other runs with leachate addition. The SRF values belonging to diluted leachate were higher than in of domestic wastewater treatment probably due to the complexity of leachate. On the other hand, in the co-treatment of domestic wastewater and leachate, moderate SRF values were observed. The addition of leachate to the feed decreased sludge dewaterability whereas the presence of domestic wastewater helped to overcome the negative effects of leachate. A reason for this may be that the domestic wastewater provides enough nutrients for biological treatment. An influent containing insufficient amounts of certain nutrients may cause a deterioration of activated sludge settling [38, 41]. In our study, synthetic domestic wastewater contained excess amounts of phosphorous. This is a very important factor since phosphorous deficiency is usually encountered in leachate treatment. Additionally, the high dissolved solids including monovalent and divalent ion concentration in the leachate may be one of the reasons of the increased SRF during treatment of leachate [46].

Dewaterability of sludge may be affected by the extent of the flocculation, in other words, by the ratio of the filamentous and non-filamentous (zoogloeal) microorganisms in the biomass population. On the other hand, some of filamentous microorganisms are believed to mainly utilise the readily biodegradable substrate, especially in completely mixed reactors and under nutrient deficient conditions [43]. Moreover, in our study, during the removal of readily biodegradable substrate dewaterability of activated sludge decreased (SRF value increased). The increase of SRF may arise through a rapid filamentous growth at the beginning of batch runs. Additionally, after removal of readily biodegradable substrate, the SRF values tended to decrease, probably due to disappearance of filamentous microorganisms. Therefore, there may be a correlation between the SRF and the removal of readily biodegradable substrate by filamentous bacteria. Composition of wastewater also affects settling and dewatering characteristics by influencing biomass population. For example, presence of long chain fatty acids in the leachate might positively affect the growth of *M. Parvicella* [39]. Moreover, the hydrophobic nature of *M. Parvicella* causes a poor dewaterability [40].

4.5. Kinetic Evaluation of Organic Substrate Removal in Batch Activated Sludge Reactors with and without Powdered Activated Carbon (PAC) Addition

Application of a Zero-Order Model in the Early Stages of Aeration

The depletion of substrates was expressed by several kinetic models. The Monod model (Eq. 4.1) reduces to a zero-order equation (Eq. 4.2) if the substrate concentration is much higher than half-velocity constant.

$$r_S = -\frac{dS}{dT} = qX = \frac{\mu_m}{Y} \frac{SX}{K_S + S} \tag{4-1}$$

if $S >> K_s$

$$r_s = \frac{\mu_m X}{Y} = k \tag{4.2}$$

S = Substrate concentration, mass COD/volume

X = Average biomass concentration in the reactor, mass MLVSS/volume

q = Zero order specific substrate removal rate, mass COD/ mass MLVSS/time

k = Zero order substrate removal rate coefficient, mass COD/volume/time

Y = Yield coefficient, mass/mass

K_s = Half-velocity constant, mass COD/volume

μ_m = Maximum specific growth rate , 1/time

Due to the high initial concentrations, substrate concentration was not expected to be rate limiting in the first hours. Hence, during the first hours (3-4 hours) of aeration, the zero-order model was more applicable. In the calculation of zero-order rate constants the initial slope of COD curves was used. The regression coefficients were high, except in runs in which COD increase occurred due to solubilization of TCOD. Also, the zero order rate constants were based on 24-hour period as seen in Table 4.6. The zero order approximation has both advantages and disadvantages. Since some particulate organics are solubilized, the time period of 24 hours may better reflect the changes in substrate concentration since TCOD is converted to SCOD. On the other hand, the zero order approximation may not be

valid for the whole period since most of the readily biodegradable compounds removed. The kinetic modelling in RUNS 6-7 and 9-19 is more reliable since more data were available during this period. Zero order reaction constants increased with an increase in leachate ratio since the initial substrate concentrations increased.

Application of a First-Order Model That Incorporates the Non-biodegradable COD Fraction

With removal of readily biodegradable organics, substrate concentration becomes rate limiting and the substrate removal kinetics approaches a first-order reaction. However, the non-biodegradable fraction of the leachate makes the first-order model inapplicable in these batch runs. A model proposed by Braha [52] takes into account the non-biodegradable COD fraction. According to this model, substrate utilization can be expressed as follows;

$$r_s = -\frac{dS}{dt} = q'X(S - S_i) = k'(S - S_i)$$

(4-3)

Where,

S_i = Non-biodegradable substrate concentration, mass COD/volume

q' = First order specific substrate removal rate, volume/mass MLVSS/time

k' = First order substrate removal rate constant, time $^{-1}$

The residual COD concentration obtained at the end of aeration in experiments was accepted as the inert COD (S_i). However, in some experiments in which COD concentration increased in 72 hours, the data at the 48th hour were accepted as the inert COD. The rate constant, k', was estimated by a trial and error procedure until reaching a k' that best fitted the actual profile. The k' values found at low leachate ratios were relatively higher than those at high leachate ratios. (Table 4.5). This showed that the leachate in the feed had an inhibitory effect on substrate removal. Moreover, experimental data did not fit Braha's model very well at high leachate ratios. The results showed that the validity of

Braha's model depended on the leachate ratio. In Figure 4.29 and 4.30 two examples of model fitting are shown.
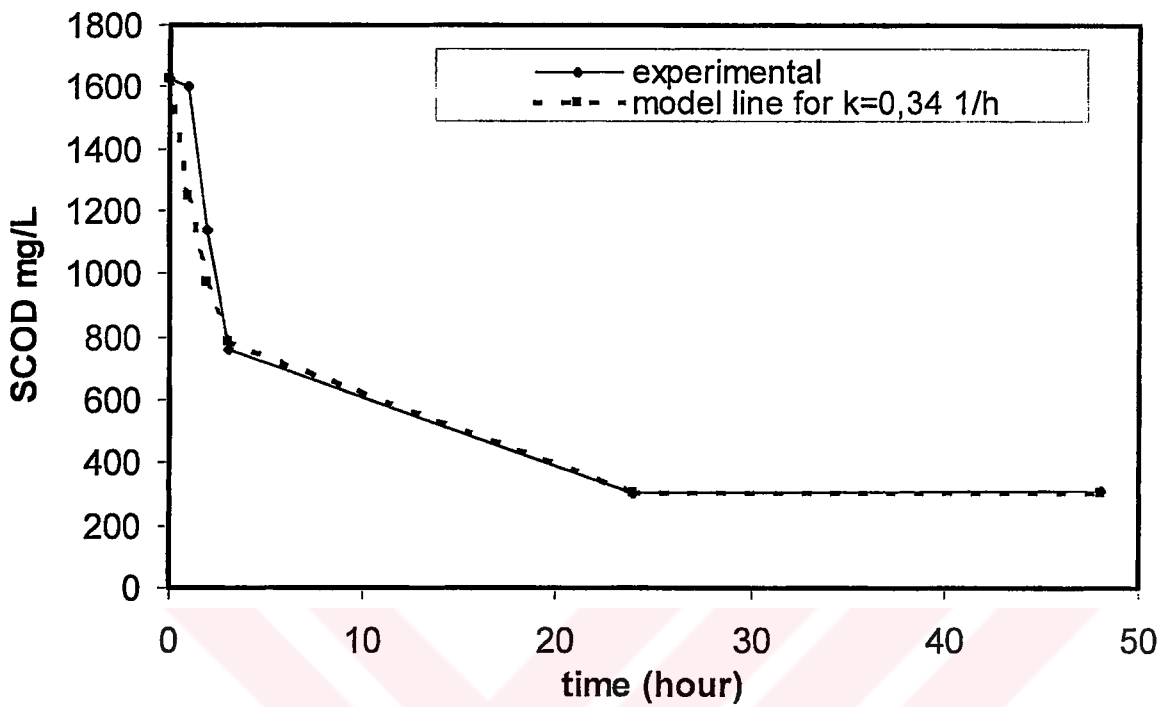
FIGURE 4.29. Application of the First-order Model (Eq. 4.3) to SCOD Data in RUN 7 (5 % Leachate Ratio)
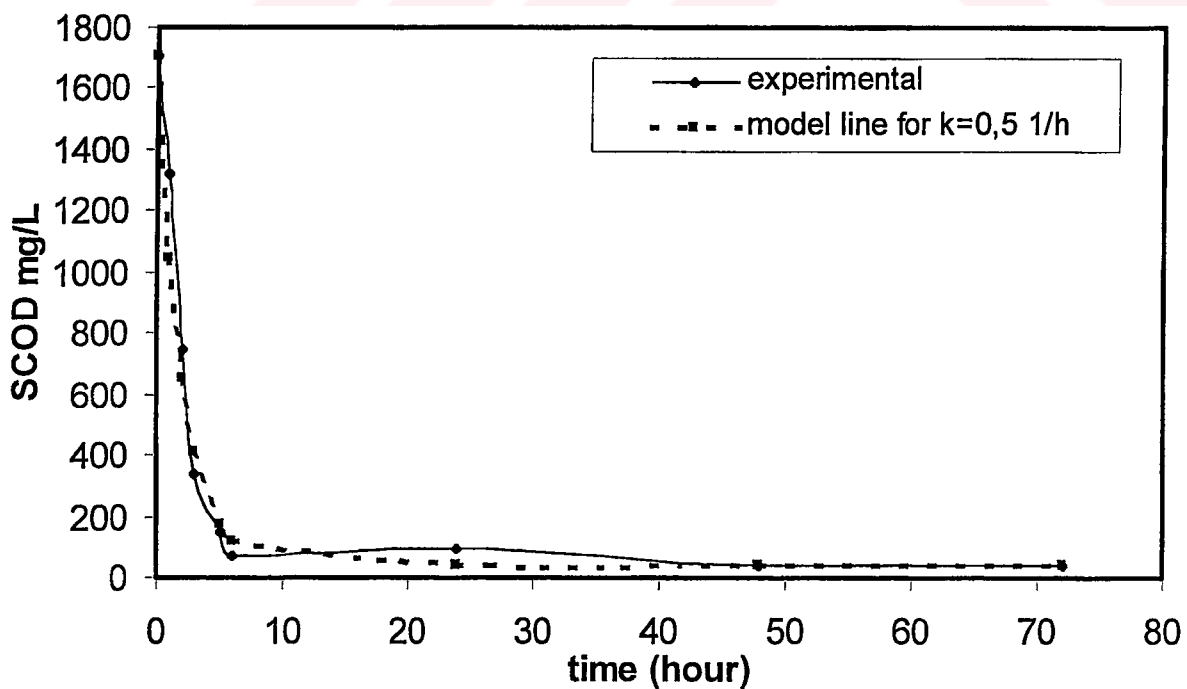
FIGURE 4.30. Application of the First-order Model (Eq. 4.3) to SCOD Data in RUN 19 (Domestic Wastewater Alone)

On the other hand, the kinetic rate constants in the case of diluted leachate were higher than expected. Therefore, in the treatment of leachate in conjunction with domestic wastewater, the volumetric ratio causing no deterioration in the biological treatment must be determined. The rate constants in the case of domestic wastewater were higher than diluted leachate as expected.

Also, from the results obtained in AS+PAC reactors, the rate constants were determined for comparison purposes (Table 4.6). In AS+PAC reactors, in addition to biodegradation also adsorption took place. Therefore, these values represent not the real but "apparent" kinetic constants. These "apparent" rate constants in AS+PAC reactors were usually equal or higher than those in AS Reactors.

TABLE 4.5 Evaluation of Kinetic Rate Constants in AS Reactors

Run	Leachate ratio (Volume %)	Basis	Braha's model (According to Eq. 4-3) k' 1/h	Zero-order Approximation (in the first 3 hours) (According to Eq 4-2) k mgCOD/Lh	Zero-order Approximation (in the first 24 hours) (According to Eq 4-2) k mgCOD/Lh
1	5	TCOD	0.67	201[*]	29[*]
		SCOD	0.76	204[*]	27[*]
2	20	TCOD	0.15	272[*]	55[*]
		SCOD	0.2	305[*]	72[*]
3	(Domestic ww)	TCOD	0.31	719[*]	141
		SCOD	0.255	529[*]	125[*]
4	(Domestic ww)	TCOD	0.4	352[*]	60[*]
		SCOD	0.5	382[*]	58[*]
5	5	TCOD	0.09	71[*]	35[*]
		SCOD	0.11	170[*]	47[*]
6	5	TCOD	0.8	319	43
		SCOD	0.88	321	42
7	10	TCOD	0.34	396	55
		SCOD	0.34	306	69
8	15	TCOD	0.097	238	92
		SCOD	0.11	197	85
9	20	TCOD	0.09	263[*]	94[*]
		SCOD	0.13	457[*]	110[*]
10	25	TCOD	0.46	760	98
		SCOD	0.25	495	104
11	25	TCOD	0.065	198	88
		SCOD	0.105	304	106
12	(Diluted Leachate)	TCOD	0.1	342	97
		SCOD	0.22	316	90
13	20	TCOD	0.12	284	94
		SCOD	0.28	414	72
14	15	TCOD	0.12	149	51
		SCOD	0.1	44	67
15	10	TCOD	0.11		56
		SCOD	0.26	229	65
16	5	TCOD	0.6	174	23
		SCOD	0.45	165	25
17	10	TCOD	0.25	149[a]	34
		SCOD	0.31	139[a]	40
18	(Diluted Leachate)	TCOD	0.22	236[a]	72
		SCOD	0.3	246[a]	67
19	(Domestic ww)	TCOD	0.56	351[a]	90
		SCOD	0.5	269[a]	67

a :calculations based on 6th hour COD value

* : uncertainty in results due to limited data

TABLE 4.6 "Apparent" Rate Constants for AS+PAC Reactors

Run	Leachate ratio(Volume %)	PAC conc (mg/L)	Basis	Braha's model (According to Eq. 4-3) k' 1/h	Zero-order Approximation (in the first 3 hours) (According to Eq 4-2) k mgCOD/Lh	Zero-order Approximation (in the first 24 hours) (According to Eq 4-2) k mgCOD/Lh
1	5	1000	TCOD	0.35	178*	28*
			SCOD	0.9	202*	28*
2	20	1000	TCOD	0.19	227*	49*
			SCOD	0.33	219*	61*
3	(domestic ww)	1000	TCOD	0.21	553*	153*
			SCOD	0.145	268*	60*
4	(domestic ww)	1000	TCOD	0.37	320*	44*
			SCOD	0.35	240*	
5	5	1000	TCOD	0.13	196*	46*
			SCOD	0.23	272*	60*
6	5	1500	TCOD	0.35	266	38
			SCOD	0.98	383	32
7	10	1500	TCOD	0.26	290	62
			SCOD	0.65	463	65
8	15	1500	TCOD	0.09	344	96
			SCOD	0.11	406	93
8		2000	TCOD	0.2	394	100
			SCOD	0.22	184	72
9	20	1500	TCOD	0.085	263*	101*
			SCOD	0.3	457*	95*
9		2000	TCOD	0.08	185*	103*
			SCOD	0.15	291*	94*
10	25	1500	TCOD	0.05	334	79
			SCOD	0.025	344	113
10		2000	TCOD	0.075	377	96
			SCOD	0.13	-	115
11	25	2500	TCOD	0.08	146	104
			SCOD	0.18	472	90
11		3000	TCOD	0.07	353	107
			SCOD	0.055	294	90
12	(Diluted Leachate)	1000	TCOD	0.1	432	70
			SCOD	0.39	626	116
12		2000	TCOD	0.15	403	77
			SCOD	0.2	473	111
13	20	2500	TCOD	0.15	306	76
			SCOD	0.3	432	68
13		3000	TCOD	0.18	330	74

TABLE 4.6 Continued

Run	Leachate ratio(Volume %)	PAC conc (mg/L)	Basis	Braha's model (According to Eq. 4-3) k' 1/h	Zero-order Approximation (in the first 3 hours) (According to Eq 4-2) k mgCOD/Lh	Zero-order Approximation (in the first 24 hours) (According to Eq 4-2) k mgCOD/Lh
			SCOD	0.26	336	86
14	15	2500	TCOD	0.1	183	68
			SCOD	0.09	-	54
14		3000	TCOD	0.11	205	73
			SCOD	0.11	48	62
15	10	2500	TCOD	0.09	-	64
			SCOD	0.1	208	59
15		3000	TCOD	0.3	-	56
			SCOD	0.3	198	61
16	5	2500	TCOD	0.3	132	24
			SCOD	0.57	158	23
16		3000	TCOD	0.5	180	30
			SCOD	0.6	162	25
17	10	2500	TCOD	0.3	193	42
			SCOD	0.5	182	42
17		3000	TCOD	0.5	203	42
			SCOD	0.5	208	43
18	(diluted Leachate)	2500	TCOD	-	159	48
			SCOD	0.42	300	73.83
19	(domestic ww)	2500	TCOD	0.28	328	89.04
			SCOD	0.22	250	68.67

a :calculations based on 6th hour COD value

* : uncertainty in results due to limited data

5. CONCLUSIONS AND RECOMMENDATIONS

The purpose of this study was to examine the effect of PAC addition on the substrate removal and activated sludge characteristics in the co-treatment of landfill leachate and domestic wastewater. In addition to co-treatment, leachate and domestic wastewater were also treated alone in order to compare the results.

In the co-treatment of leachate and domestic wastewater, the leachate ratio in the total wastewater was kept at 5,10, 15, 20, 25 % (on volumetric basis). Even at low leachate ratios, the COD of the influent was mainly composed of leachate COD due to the high strength of leachate. Considerable COD removal was achieved in the runs performed at low leachate ratios. This indicated that the inhibitory and toxic substances in the leachate were diluted when the leachate ratio was low. At extremely high leachate ratios, 90 % of COD stemmed from leachate and COD removal was slightly reduced.

PAC addition led to an increased COD removal in most of the co-treatment runs. The mechanism of the improvement with addition of PAC was mostly due to the adsorption of inhibitory and toxic substances and/or slowly biodegradable and non-biodegradable matter, which could not be removed in a conventional activated sludge system. Therefore, by PAC addition many problems in the activated sludge systems treating leachate and domestic wastewater together may be overcome.

If the leachate ratio in the feed increased, the residual COD also increased indicating that residual organics stemmed from leachate. The inert COD concentrations in PAC added reactors were lower than in normal activated sludge reactors. This showed that the residual organics were adsorbed onto the activated carbon. Moreover, the effect of PAC was more obvious when the leachate ratio was high.

In the case of domestic wastewater, COD was almost completely removed. PAC addition did not lead to an improvement in COD removal because of the fact that domestic wastewater was mostly composed of non-adsorbable organics. In the case of leachate alone, PAC addition usually led to an additional COD removal. However, when the PAC

dose increased from 2500 mg/L to 3000 mg/L, this did not considerably affect the residual differences. Therefore, the optimum PAC dose must be determined taking the operational cost and sludge production into account.

Dewaterability of sludge was represented by Specific Resistance to Filtration (SRF) and settling characteristics were represented by Sludge Volume Index (SVI) and Settling Curves. Leachate affected activated sludge characteristics. As a general trend, in most of the runs SRF values first increased (sludge dewaterability was decreased) and then tended to decline to initial values. This may be attributed to the enhanced growth of filamentous bacteria due to the high concentrations of readily biodegradable substrates (Volatile Fatty Acids) in the initial periods of aeration. When the leachate ratio in the feed increased, the SRF and settling properties of activated sludge worsened.

The addition of PAC to activated sludge increased the dewaterability of the activated sludge. Results showed that presence of PAC suppressed the increase of SRF during the removal of readily biodegradable substrate. The SVI values in PAC added activated sludge reactors were always lower than normal activated sludge reactors.

The experiments may also be performed in a Continuous Flow Stirred Tank Reactor (CFSTR). In that case substrate removal and sludge characteristics can be examined together. Additionally, microscopic studies should also be performed to determine the effect of filamentous bacteria on settling and dewatering of activated sludge. This makes it possible to identify the real causes rather than the possible reasons.

In conclusion, PAC led to an improvement in substrate removal in the co-treatment of leachate and domestic wastewater. Inert COD concentrations were reduced to lower levels. Moreover, addition of PAC had an effect on sludge characteristics. The dewaterability of sludge was improved with PAC addition.

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APPENDIX A

Table A1. Efficiencies and Final COD Concentrations in AS and AS+PAC Reactors

	RUN No	Leachate Ratio in the Feed	Reactor Type	Initial COD (mg/L)	Initial Leachate COD (mg/L)	Final COD (mg/L)	Decrease in Leachate COD (mg/L)	Decrease in COD (mg/L)	Leachate COD Removal (%)	COD Removal (%)
TCOD	1	5	AS	820	320	141	179	679	56	83
SCOD	1	5	AS	742	242	77	165	665	68	90
TCOD	1	5	AS+1000 PAC	897	397	164	233	733	59	82
SCOD	1	5	AS+1000 PAC	690	190	48	142	642	75	93
TCOD	2	20	AS	2184	1684	551	1133	1633	67	75
SCOD	2	20	AS	2150	1650	297	1353	1853	82	86
TCOD	2	20	AS+1000 PAC	2220	1720	818	902	1402	52	63
SCOD	2	20	AS+1000 PAC	2099	1599	343	1256	1756	79	84
TCOD	5	5	AS	2001	1501	611	890	1390	59	69
SCOD	5	5	AS	1306	806	422	384	884	48	68
TCOD	5	5	AS+PAC	1963	1463	608	855	1355	58	69
SCOD	5	5	AS+1000 PAC	1823	1029	351	678	1472	66	81
TCOD	6	5	AS	1203	703	179	524	1024	75	85
SCOD	6	5	AS	1105	605	97	508	1008	84	91
TCOD	6	5	AS+1000 PAC	1173	673	177	496	996	74	85
SCOD	6	5	AS+1000 PAC	1081	581	37	544	1044	94	97
TCOD	7	10	AS	2135	1635	395	1240	1740	76	81
SCOD	7	10	AS	1624	1124	312	812	1312	72	81
TCOD	7	10	AS+1500 PAC	2047	1547	520	1027	1527	66	75
SCOD	7	10	AS+1500 PAC	1855	1355	197	1158	1658	85	89
TCOD	8	15	AS	2915	2415	530	1885	2385	78	82
SCOD	8	15	AS	2429	1929	108	1821	2321	94	96
TCOD	8	15	AS+1500 PAC	2903	2403	661	1742	2242	72	77
SCOD	8	15	AS+1500 PAC	2398	1898	100	1798	2298	95	96
TCOD	8	15	AS+2000 PAC	3005	2505	417	2088	2588	83	86
SCOD	8	15	AS+2000 PAC	1924	1424	88	1336	1836	94	95

Table A1. Continued

	RUN No	Leachate Ratio in the Feed	Reactor Type	Initial COD (mg/L)	Initial Leachate COD (mg/L)	Final COD (mg/L)	Decrease in Leachate COD (mg/L)	Decrease in COD (mg/L)	Leachate COD Removal (%)	COD Removal (%)
TCOD	9	20	AS	3560	3060	717	2343	2843	77	80
SCOD	9	20	AS	2520	2020	315	1705	2205	84	88
TCOD	9	20	AS+1500 PAC	3650	3060	755	2305	2895	75	79
SCOD	9	20	AS+1500 PAC	2520	2020	227	1793	2293	89	91
TCOD	9	20	AS+2000 PAC	3650	3060	733	2327	2917	76	80
SCOD	9	20	AS+2000 PAC	2520	2020	315	1705	2205	84	88
TCOD	10	25	AS	4367	3867	1226	2641	3141	68	72
SCOD	10	25	AS	3306	2806	629	2177	2677	78	81
TCOD	10	25	AS+1500 PAC	4350	3850	1572	2278	2778	59	64
SCOD	10	25	AS+1500 PAC	3326	2826	474	2352	2852	83	86
TCOD	10	25	AS+2000 PAC	4348	3848	1353	2495	2995	65	69
SCOD	10	25	AS+2000 PAC	3326	2826	273	2553	3053	90	92
TCOD	11	25	AS	4416	3916	1194	2722	3222	70	73
SCOD	11	25	AS	3135	2635	399	2236	2736	85	87
TCOD	11	25	AS+2500 PAC	4289	3789	1563	2226	2726	59	64
SCOD	11	25	AS+2500 PAC	3128	2628	109	2519	3019	96	97
TCOD	11	25	AS+3000 PAC	4915	4415	1431	2984	3484	68	71
SCOD	11	25	AS+3000 PAC	3133	2633	92	2541	3041	97	97
TCOD	13	20	AS	2706	2206	608	1598	2098	72	78
SCOD	13	20	AS	2466	1966	548	1418	1918	72	78
TCOD	13	20	AS+2500 PAC	2856	2356	415	1941	2441	82	85
SCOD	13	20	AS+2500 PAC	2556	2056	391	1665	2165	81	85
TCOD	13	20	AS+3000 PAC	2816	2316	408	1908	2408	82	86
SCOD	13	20	AS+3000 PAC	2506	2006	364	1642	2142	82	85

Table A1. Continued

	RUN No	Leachate Ratio in the Feed	Reactor Type	Initial COD (mg/L)	Initial Leachate COD (mg/L)	Final COD (mg/L)	Decrease in Leachate COD (mg/L)	Decrease in COD (mg/L)	Leachate COD Removal (%)	COD Removal (%)
TCOD	14	15	AS	2309	1809	597	1212	1712	67	74
SCOD	14	15	AS	1575	1075	280	795	1295	74	82
TCOD	14	15	AS+2500 PAC	2369	1869	524	1345	1845	72	78
SCOD	14	15	AS+2500 PAC	1538	1038	99	939	1439	90	94
TCOD	14	15	AS+3000 PAC	2428	1928	459	1469	1969	76	81
SCOD	14	15	AS+3000 PAC	1660	1160	137	1023	1523	88	92
TCOD	15	10	AS	1825	1325	302	1023	1523	77	83
SCOD	15	10	AS	1492	992	200	792	1292	80	87
TCOD	15	10	AS+2500 PAC	1744	1244	200	1044	1544	84	89
SCOD	15	10	AS+2500 PAC	1467	967	68	899	1399	93	95
TCOD	15	10	AS+3000 PAC	1660	1160	198	962	1462	83	88
SCOD	15	10	AS+3000 PAC	1516	1016	75	941	1441	93	95
TCOD	16	5	AS	810	310	284	26	526	8	65
SCOD	16	5	AS	670	170	95	75	575	44	86
TCOD	16	5	AS+2500 PAC	735	235	388	-153	347	-65	47
SCOD	16	5	AS+2500 PAC	598	98	29	69	569	70	95
TCOD	16	5	AS+3000 PAC	900	400	280	120	620	30	69
SCOD	16	5	AS+3000 PAC	637	137	31	106	606	77	95
TCOD	17	10	AS	1463	963	322	641	1141	67	78
SCOD	17	10	AS	1114	614	221	393	893	64	80
TCOD	17	10	AS+1500 PAC	1433	933	320	613	1113	66	78
SCOD	17	10	AS+1500 PAC	1132	632	127	505	1005	80	89
TCOD	17	10	AS+3000 PAC	1140	632	414	218	726	34	64
SCOD	17	10	AS+3000 PAC	1144	644	113	531	1031	82	90

APPENDIX B

TABLE B.1. Results of the Co-treatment of Leachate A and Domestic Wastewater in Batch Activated Sludge Reactors

Run	Leachate Ratio Volume Per cent	Leachate Ratio COD Contribution (%)	Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO_2-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)
1	5	39	AS	0	820	742	7.88	-	650	-	63
				3	217	130	8.54	-	-	-	-
				24	116	83	8.67	3.5	1560	-	36
				48	141	77	8.54	7.7	1425	-	35
				72	-	-	8.07	8.7	1800	-	28
1	5	39	AS+1000 PAC	0	897	690	8.01	-	1440	-	-
				3	362	84	8.71	-	-	-	23
				24	217	27	8.68	5.7	1805	-	-
				48	164	48	8.44	9.8	1910	-	24
				72	-	-	7.65	5.8	750	-	23
2	20	77	AS	0	2184	2150	8.13	-	1455	6625	17
				3	1368	1233	-	-	-	5430	28
				24	863	411	8.95	5	-	5765	-
				48	837	574	9.04	-	-	-	-
				72	551	297	-	12	-	-	-
2	20	77	AS+1000 PAC	0	2220	2099	8.53	-	2568	6555	16
				3	1539	880	-	-	-	-	-
				24	1021	619	9.12	5	-	5990	-
				48	829	579	9.02	7	-	7885	-
				72	818	343	-	20	-	-	-

TABLE B.2. Results of the Co-treatment of Leachate B and Domestic Wastewater in Batch Activated Sludge Reactors

Run	Leachate Ratio Volume Fraction (%)	COD Contribution (%)	Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO_2-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)	SRF x 10^{12} m/kg
5	5	75	AS	0	2001	1306	8.52	-	1700	6420	30	-
				3	1490	1094	-	-	-	-	-	-
				24	873	470	8.99	6	-	4955	-	-
				48	773	529	9.03	9.5	1710	-	29	-
				72	611	422	9.05	13	1580	6245	25	-
5	5	75	AS+1000 PAC	0	1963	1829	8.62	-	2100	6585	17	-
				3	1374	1014	-	-	-	-	-	-
				24	854	378	9.09	13	-	6275	-	-
				48	726	374	9.13	15	2215	-	18	-
				72	608	351	9.12	26	2305	5545	13	-
6	5	58	AS	0	1203	1105	8.61	-	1480	4340	-	-
				1	758	660	-	-	-	-	-	-
				2	425	355	-	-	-	-	-	-
				3	252	136	-	-	-	-	-	-
				24	195	67	8.78	13	2495	4050	20	-
				48	179	97	8.42	15	2770	4185	-	-
6	5	58	AS+1500 PAC	0	1173	1081	8.53	-	-	5335	-	-
				1	1136	782	-	-	-	-	-	-
				2	526	75	-	-	-	-	-	-
				3	490	39	-	-	-	-	-	-
				24	233	20	8.58	12	3845	5795	11	-
				48	177	37	8.22	14	3575	4700	-	-

TABLE B.2 Continued

| Run | Leachate Ratio | | Reactor Type | Time (Hour) | TCOD (mg/L) | SCOD (mg/L) | pH | NO$_2$-N (mg/L) | MLSS (mg/L) | T. Solids (mg/L) | SVI (mL/g) | SRF x 10^{12} m/kg |
	Volume Fraction (%)	COD Contribution (%)										
7	10	75	AS	0	2135	1624	8.41	-	3575	6460	20	-
				1	1690	1605	-	-	-	-	-	-
				2	1240	1140	-	-	-	-	-	-
				3	965	758	-	-	-	-	-	-
				24	467	306	8.93	8.3	3800	5615	-	-
				48	395	312	8.9	5	3045	5715	23	-
7	10	75	AS+1500 PAC	0	2047	1855	8.39	-	4645	6780	12	-
				1	1965	1605	-	-	-	-	-	-
				2	1435	977	-	-	-	-	-	-
				3	1258	520	-	-	-	-	-	-
				24	562	287	8.8	8.3	-	7045	-	-
				48	520	197	8.75	10	3100	5675	21	-
8	15	83	AS	0	2915	2429	8.21	-	2360	6070	22	1209.849
				1	2681	2386	-	-	-	-	-	-
				2	2464	2182	-	-	-	-	-	-
				3	2194	1840	-	-	-	-	-	-
				24	884	227	8.83	12	3400	6615	21	-
				48	739	291	8.88	12	3155	5885	21	8.656
				72	530	108	9.06	11	3640	5630	15	4.718
8	15	83	AS+1500 PAC	0	2903	2398	8.48	-	3840	7050	11	-
				1	2783	2215	-	-	-	-	-	-
				2	2246	1708	-	-	-	-	-	-
				3	1936	1215	-	-	-	-	-	-
				24	607	168	8.92	18	3910	7035	15	10.129
				48	652	191	8.93	17	4330	6600	13	5.638
				72	661	100	8.88	24	3760	6050	13	4.912

TABLE B.2 Continued

Run	Leachate Ratio Volume Fraction (%)	COD Contribution (%)	Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO$_2$-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)	SRF x 10^{12} m/kg
8	15	83	AS+2000 PAC	0	3005	1924	8.38	-	-	7350	-	-
				1	2380	2026	-	-	-	-	-	-
				2	2296	1594	-	-	-	-	-	-
				3	1720	1455	-	-	-	-	-	-
				24	595	202	8.8	23	4660	6410	14	13.399
				48	500	174	8.81	26	4545	6760	11	73.682
				72	417	82	8.78	40	3805	6120	13	3.482
9	20	86	AS	0	3650	2520	7.23	-	1940	6060	23	34.804
				3	2741	1876	-	-	-	-	-	-
				24	1007	259	9.03	18	3020	7005	16	108.295
				48	851	299	9.1	14	-	5960	-	-
				72	717	315	7.63	9	2775	5985	21	6.837
9	20	86	AS+1500 PAC	0	3650	2520	7.21	-	4000	8580	10	67.563
				3	2861	1149	-	-	-	-	-	-
				24	1231	245	9.11	19	4445	7965	9	4.862
				48	868	267	9.1	21	3950	7760	-	-
				72	755	227	9	24	4185	7315	11	6.640
9	20	86	AS+2000 PAC	0	3650	2520	7.23	-	4120	8325	7	65.229
				3	3095	1648	-	-	-	-	-	-
				24	1166	256	8.95	33	4420	7695	9	8.549
				48	853	192	9.01	24	4175	7385	-	-
				72	733	315	8.9	31	4305	7470	12	7.417

TABLE B.2 Continued

Run	Leachate Ratio Volume Fraction (%)	COD Contribution (%)	Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO₂-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)	SRF x 10¹² m/kg
10	25	88	AS	0	4367	3306	8.63	-	1930	6705	21	0.370
				1	3938	2176	-	-	-	-	-	-
				2	3667	1725	-	-	-	-	-	-
				3	1925	1805	-	-	-	-	-	-
				24	1866	941	8.87	31	2750	6220	15	-
				48	1361	281	9.14	31	2530	6410	-	-
				72	1226	629	9.21	31	2610	6040	-	54.491
10	25	88	AS+1500 PAC	0	4350	3326	8.72	-	3022	7270	10	15.793
				1	3817	2566	-	-	-	-	-	-
				2	3607	1916	-	-	-	-	-	-
				3	3307	1737	-	-	-	-	-	-
				24	2443	616	9	32	4195	8135	8	2.482
				48	1485	265	9.16	28	2265	6285	15	-
				72	1572	474	9.22	19	3190	-	11	10.015
10	25	88	AS+2000 PAC	0	4348	3326	8.67	-	3715	7530	-	8.711
				1	3487	1976	-	-	-	-	-	-
				2	3317	2306	-	-	-	-	-	-
				3	3147	-	-	-	-	-	-	-
				24	2043	560	8.92	34	3925	8305	-	-
				48	1478	133	9.1	28	4110	8345	-	-
				72	1353	273	9.14	26	3745	-	-	26.122

TABLE B.2 Continued

Run	Leachate Ratio Volume Fraction (%)	COD Contribution (%)	Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO$_2$-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)	SRF x 10^{12} m/kg
11	25	89	AS	0	4416	3135	7.5	11	1950	8110	21	5.331
				1	3936	2925	-	-	-	-	-	-
				2	3875	2434	-	-	-	-	-	-
				3	3775	2284	-	-	-	-	-	-
				24	1872	1021	9.06	12	3120	7585	11	19.209
				48	1351	409	8.97	24	3215	6685	-	-
				72	1194	399	9.19	26	3308	-	10	170.234
11	25	89	AS+2500 PAC	0	4289	3128	7.57	17	4170	12470	9	1.583
				1	3868	2527	-	-	-	-	-	-
				2	3828	1986	-	-	-	-	-	-
				3	3816	1736	-	-	-	-	-	-
				24	1788	956	9.18	43	4610	8900	7	15.814
				48	1289	40	9.18	52	-	7690	-	-
				72	1563	109	9.16	38	4796	7325	6	13.709
11	25	89	AS+3000 PAC	0	4915	3133	7.56	13	5015	10480	7	1.340
				1	3917	3087	-	-	-	-	-	-
				2	3707	2085	-	-	-	-	-	-
				3	3807	2486	-	-	-	-	-	-
				24	2346	975	9	9	4280	7760	7	4.660
				48	1271	58	9.02	25	-	7365	-	-
				72	1431	92	9.05	59	3408	6120	9	16.269

TABLE B.2 Continued

Run	Leachate Ratio		Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO₂-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)	SRF x 10^{12} m/kg
	Volume Fraction (%)	COD Contribution (%)										
13	20	82	AS	0	2706	2466	8.27	-	2312	7365	30	7.011
				1	2436	2146	-	-	-	-	-	-
				2	2356	1845	-	-	-	-	-	-
				3	1785	1185	-	-	-	-	-	-
				24	978	204	9.1	22	2940	6275	24	53.071
				48	819	276	9.11	13	-	5845	-	-
				72	608	548	9.1	12	2496	5595	28	4.842
13	20	82	AS+2500 PAC	0	2856	2556	8.44	-	4456	8635	14	8.166
				1	2549	1946	-	-	-	-	-	-
				2	2399	1564	-	-	-	-	-	-
				3	1885	1245	-	-	-	-	-	-
				24	1037	917	9.02	7	5660	7530	14	43.994
				48	710	112	9.04	3	-	8148	-	-
				72	415	391	9.02	27	3672	6715	19	2.253
13	20	82	AS+3000 PAC	0	2816	2506	8.36	-	4296	8540	14	11.939
				1	2536	1995	-	-	-	-	-	-
				2	2336	1755	-	-	-	-	-	-
				3	1785	1465	-	-	-	-	-	-
				24	1044	443	8.89	1	5465	8245	15	14.842
				48	618	156	8.93	18	-	8915	-	-
				72	408	364	8.76	77	5468	9040	15	2.624

TABLE B.2 Continued

Run	Leachate Ratio Volume Fraction (%)	COD Contribution (%)	Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO_2-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)	SRF x 10^{12} m/kg
14	15	78	AS	0	2309	1575	8.1	-	1384	5435	5	4.338
				1	2260	1293	-	-	-	-	-	-
				2	2230	1966	-	-	-	-	-	-
				3	1822	1498	-	-	-	-	-	-
				24	690	342	8.95	5	2428	5305	28	32.378
				48	543	295	9.08	-	-	4675	-	-
				72	597	280	9.13	3	2180	4440	32	5.482
14	15	78	AS+2500 PAC	0	2369	1538	8.66	-	3812	8040	17	1.664
				1	2321	1089	-	-	-	-	-	-
				2	2182	1792	-	-	-	-	-	-
				3	1804	1385	-	-	-	-	-	-
				24	727	250	8.98	12	3664	7925	19	9.340
				48	530	158	8.85	5	-	6695	-	-
				72	524	99	8.61	72	3984	6575	20	2.859
14	15	78	AS+3000 PAC	0	2428	1660	8.72	-	4812	7630	12	2.642
				1	2333	1240	-	-	-	-	-	-
				2	2080	1726	-	-	-	-	-	-
				3	1828	1338	-	-	-	-	-	-
				24	665	181	8.94	14	4536	7045	14	22.771
				48	422	118	9.01	14	-	4795	-	-
				72	459	137	9.03	6	4408	7035	18	6.835

TABLE B.2 Continued

Run	Volume Fraction (%)	COD Contribution (%)	Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO₂-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)	SRF x 10¹² m/kg
15	10	73	AS	0	1825	1492	8.46	-	1472	3700	27	2.284
				1	1635	1059	-	-	-	-	-	-
				2	-	1005	-	-	-	-	-	-
				3	-	747	-	-	-	-	-	-
				24	275	155	8.96	13	1740	3905	23	4.472
				48	469	137	9.03	11	-	3650	-	-
				72	302	200	9.01	11	1543	3430	26	1.415
15	10	73	AS+2500 PAC	0	1744	1467	8.62	-	3356	5595	20	3.605
				1	1792	699	-	-	-	-	-	-
				2	-	855	-	-	-	-	-	-
				3	1984	723	-		-	-	-	4.715
				24	211	48	8.95	17	4296	6525	12	-
				48	568	75	8.93	50	-	5045	-	-
				72	200	68	8.71	39	3013	5055	13	0.991
15	10	73	AS+3000 PAC	0	1660	1516	8.48	-	3384	5525	-	3.472
				1	-	855	-	-	-	-	-	-
				2	2068	807	-	-	-	-	-	-
				3	1936	873	-	-	-	-	-	-
				24	322	53	8.87	14	4028	6395	12	12.228
				48	632	116	8.9	24	-	5080	-	-
				72	198	75	8.75	35	3555	5845	11	1.500

TABLE B.2 Continued

Run	Leachate Ratio Volume Fraction (%)	COD Contribution (%)	Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO$_2$-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)	SRF x 10^{12} m/kg
16	5	77	AS	0	810	670	8.46	-	1906	3190	25	3.062
				1	634	481	-	-	-	-	-	-
				2	435	331	-	-	-	-	-	-
				3	295	171	-	-	-	-	-	-
				24	215	108	8.65	3	2123	3275	18	0.589
				48	201	147	8.76	6	-	3275	-	-
				72	284	95	8.64	9	2023	3210	25	1.158
16	5	77	AS+2500 PAC	0	735	598	8.48	-	4234	5560	12	0.464
				1	542	427	-	-	-	-	-	-
				2	408	250	-	-	-	-	-	-
				3	338	129	-	-	-	-	-	-
				24	153	45	8.64	9	4140	5680	12	1.116
				48	284	42	8.69	10	-	5335	-	-
				72	388	29	8.44	17	3011	4315	12	0.701
16	5	77	AS+3000 PAC	0	900	637	8.46	-	-	5520	-	0.752
				1	553	392	-	-	-	-	-	-
				2	406	262	-	-	-	-	-	-
				3	349	141	-	-	-	-	-	-
				24	190	43	8.64	5	4291	5810	14	1.091
				48	392	76	8.65	5	-	5090	-	-
				72	280	31	8,52	13	4405	5455	11	0.636

TABLE B.2 Continued

Run	Leachate Ratio Volume Fraction (%)	COD Contribution (%)	Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO$_2$-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)	SRF x 10^{12} m/kg
17	10	66	AS	0	1463	1114	-	-	1567	4120	-	2.994
				1	1253	825	-	-	1853	3920	-	-
				2	1066	650	-	-	2020	4360	-	5.953
				3	957	501	-	-	-	4315	-	-
				4	758	501	-	-	1847	4350	-	6.529
				5	783	264	-	-	2100	4095	-	20.512
				6	489	244	-	-	2220	4260	-	19.697
				24	490	289	-	7	2220	4165	-	8.458
				48	422	239	-	1	1900	3545	-	1.454
				72	322	221	-	5	1870	3705	-	0.899
17	10	66	AS+2500 PAC	0	1433	1132	-	-	2893	5060	-	2.071
				1	1319	837	-	-	2454	4890	-	5.672
				2	963	541	-	-	2986	5220	-	9.456
				3	909	625	-	-	-	5530	-	-
				4	553	380	-	-	3320	5550	-	2.318
				5	893	204	-	-	-	5200	-	26.952
				6	405	155	-	-	3143	5500	-	1.710
				24	424	122	-	9	3523	5410	-	8.525
				48	433	134	-	10	3123	4895	-	5.053
				72	320	127	-	5	3260	4890	-	

TABLE B.2 Continued

Run	Leachate Ratio		Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO₂-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)	SRF x 10¹² m/kg
	Volume Fraction (%)	COD Contribution (%)										
17	10	66	AS+3000 PAC	0	1440	1144	-	-	4603	7445	-	0.567
				1	1216	819	-	-	4447	4505	-	15.573
				2	921	547	-	-	4667	7190	-	1.313
				3	861	541	-	-	-	7245	-	-
				4	565	300	-	-	4210	6170	-	2.284
				5	429	119	-	-	-	5610	-	3.862
				6	417	196	-	-	3616	5620	-	13.818
				24	420	117	-	9	4883	6425	-	16.707
				48	392	86	-	43	3826	5530	-	6.027
				72	414	113	-	12	4313	6670	-	1.258

TABLE B.3. Results of the Batch Runs Performed with Synthetic Domestic Wastewater Alone

Run	Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO$_2$-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)	SRF x 10^{12} m/kg
3	AS	0	3672	3087	6.59	-	3465	10365	-	-
		3	1514	1501	-	-	-	-	-	-
		24	287	81	8.2	12	3635	8660	20	-
		48	185	87	7.79	17	3825	8585	22	-
		72	115	34	7.23	50	3485	8315	25	-
3	AS+1000 PAC	0	3783	2758	6.96	-	2920	10035	-	-
		3	2124	1953	-	-	-	-	-	-
		24	113	77	8.48	10	4090	9540	17	-
		48	192	65	8.55	9	4295	8460	17	-
		72	177	109	8.23	4	4020	8510	18	-
4	AS	0	1526	1446	7.52	-	1430	4375	28	-
		3	471	300	-	-	-	-	-	-
		24	89	60	8.21	4	2260	4450	25	-
		48	61	26	7.85	4.4	1795	3795	30	-
		72	27	15	6.62	5	1880	3880	37	-
4	AS+1000 PAC	0	1500	1123	7.62	-	2150	5105	14	-
		3	540	404	-	-	-	-	-	-
		24	63	55	8.25	10	2940	4970	19	-
		48	38	19	8.2	4.5	3120	5105	16	-
		72	38	13	7.75	3.5	2070	3835	31	-

TABLE B.3 Continued

Run	Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO$_2$-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)	SRF x 10^{12} m/kg
19	AS	0	2302	1693	-	9.4	2556	6685	-	2.350
		1	1816	1310	-	13	2840	7190	-	0.340
		2	899	730	-	14	-	-	-	-
		3	-	327	-	9.4	3736	8200	-	1.219
		4	-	-	-	-	3730	8380	-	0.470
		5	214	147	-	-	4066	7690	-	2.069
		6	170	71	-	-	5500	8905	-	0.197
		24	117	66	-	25	3773	7690	-	1.390
		48	200	41	-	2	3520	8180	-	0.590
		72	158	36	-	1	3426	8215	-	0.866
19	AS+2500 PAC	0	2310	1713	-	2	3260	7245	-	0.253
		1	1898	1379	-	5	4253	8090	-	0.953
		2	1282	1143	-	5	-	-	-	-
		3	1251	992	-	0	3890	8195	-	0.299
		4	950	769	-	-	3846	8440	-	1.263
		5	721	564	-	-	3756	7290	-	-
		6	344	212	-	-	4083	8150	-	0.473
		24	171	63	-	3.5	6360	9860	-	1.391
		48	244	33	-	21	5970	9885	-	3.430
		72	89	29	-	4	5663	9460	-	0.340

TABLE B.4. Results of the Batch Runs Performed with Diluted Leachate (Leachate B)

Run	Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO₂-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)	SRF x 10¹² m/kg
12	AS	0	2993	2572	8.65	-	1465	3265	20	0.603
		1	2590	2116	-	-	-	-	-	-
		2	2338	2200	-	-	-	-	-	-
		3	1936	1491	-	-	-	-	-	-
		24	835	246	8.9	7	2748	3390	11	136.430
		48	404	112	8.99	8	-	3445	-	-
		72	326	158	8.91	13	1432	3060	21	1.600
12	AS+1000 PAC	0	3053	2887	8.63	-	2620	4485	15	0.615
		1	2771	2140	-	-	-	-	-	-
		2	2128	1864	-	-	-	-	-	-
		3	1828	891	-	-	-	-	-	-
		24	1382	96	8.84	7.68	2172	4805	17	69.040
		48	581	47	8.9	12	-	4410	-	-
		72	387	67	8.79	18	3052	4925	15	2.341
12	AS+2000 PAC	0	3185	2693	8.51	-	2940	5150	13	3.626
		1	2590	1960	-	-	-	-	-	-
		2	2104	1732	-	-	-	-	-	-
		3	2004	1191	-	-	-	-	-	-
		24	1327	27	8.79	10.11	2992	5185	13	28.402
		48	617	37	8.87	11	-	4540	-	-
		72	320	13	8.81	29	2732	4400	15	5.861

TABLE B.4 Continued

| Run	Reactor Type	Time (Hour)	TCOD (mg/L)	SCOD (mg/L)	pH	NO2-N (mg/L)	MLSS (mg/L)	T. Solids (mg/L)	SVI (mL/g)	SRF x 10^{12} m/kg
18	AS	0	2820	2110	-	52.29		7660	-	16.160
		1	2238	1708	-	35			-	
		2	2058	1407	-	23			-	
		3	-	-	-	9.23		8315	-	22.330
		4	1538	1284	-	14			-	
		5	1456	684	-			7815	-	14.020
		6	1311	540	-			8100	-	29.20
		24	1074	507	-	4		6520	-	190.110
		48	1020	345	-	3.4		6650	-	35.590
		72	540	224	-	6		6600	-	12.070
		96	787	416	-	1		6255	-	3.540
18	AS+2500 PAC	0	2315	2098	-	30		9415	-	7.010
		1	2304	1085	-	62			-	-
		2	2152	1152	-	14			-	-
		3	2370	815	-	14		8950	-	8.970
		4	2018	860	-	5			-	-
		5	1422	397	-	9.23		8960	-	102.260
		6	1360	299	-			9960	-	51.480
		24	1152	326	-	9		8755	-	34.240
		48	1245	300	-	7		8975	-	36.570
		72	583	106	-	2		8190	-	14.110
		96	823	253	-	0		7620	-	23.320

APPENDIX C
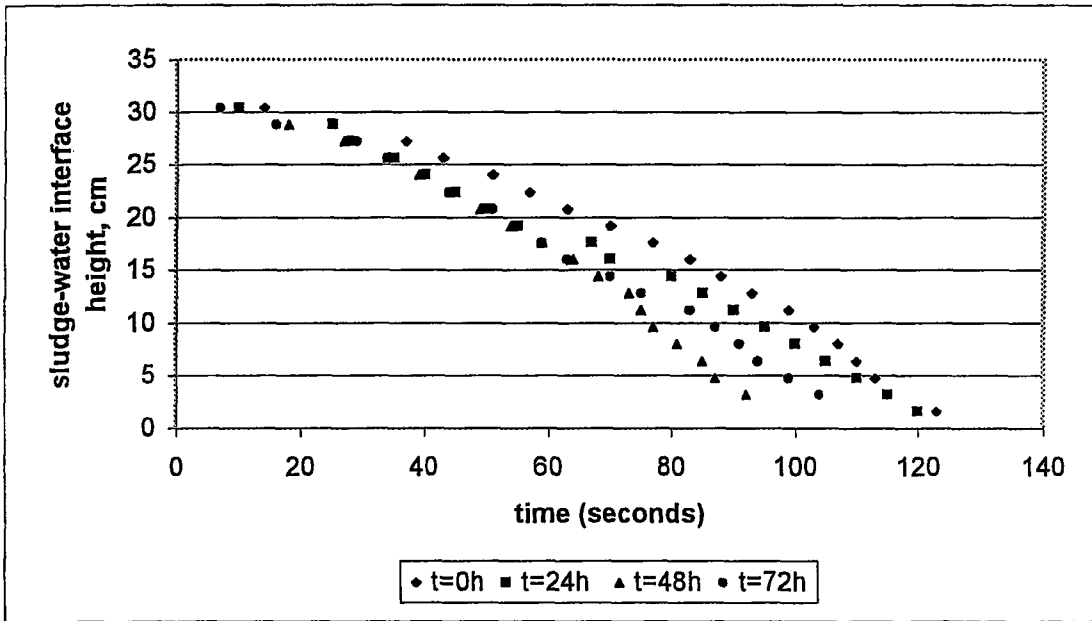
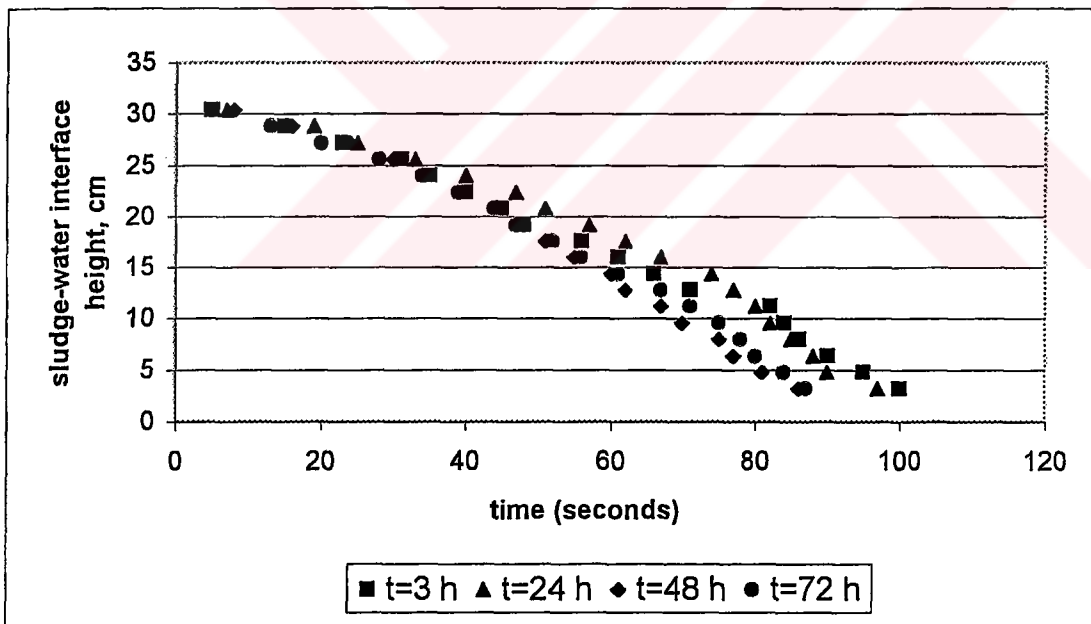FIGURE C.1. Settling Curve of RUN 8 (15 % Leachate Ratio)

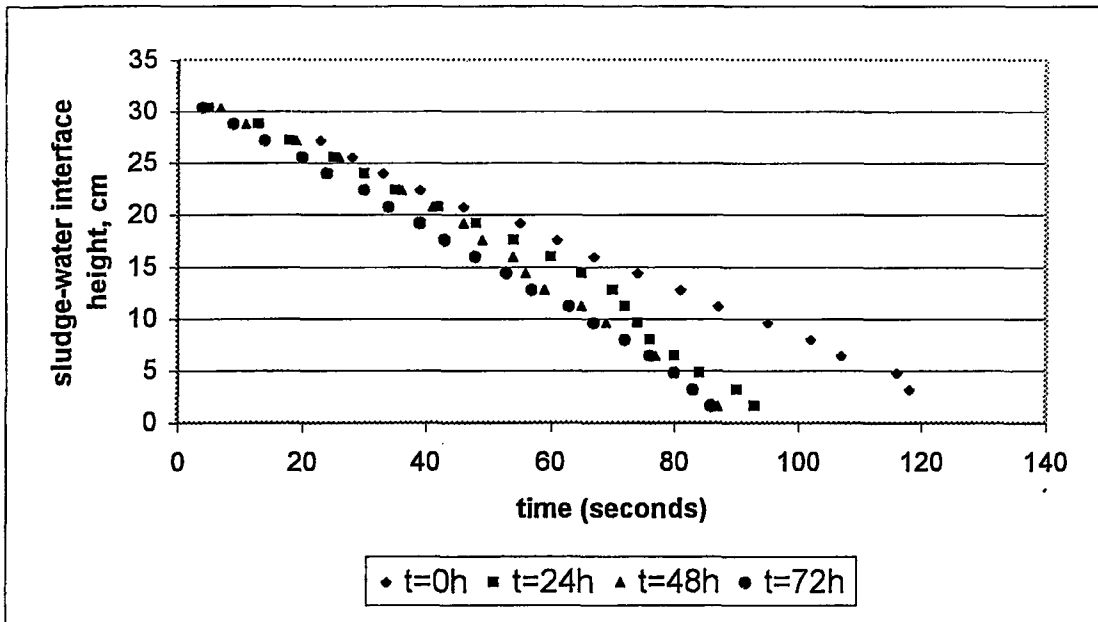FIGURE C.2. Settling Curve of RUN 9 (20 % Leachate Ratio)

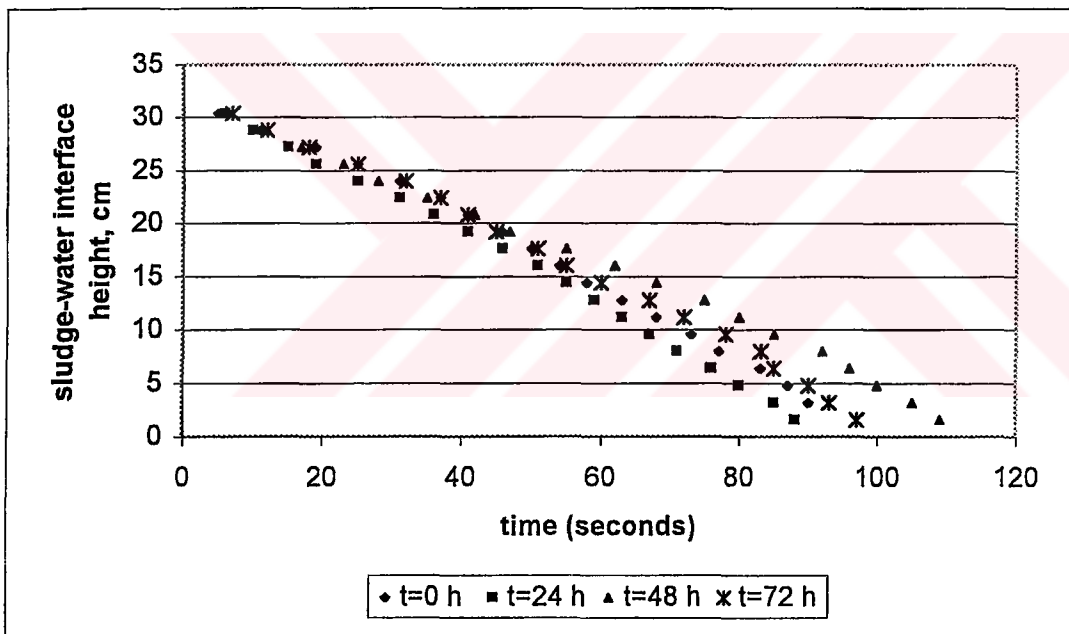FIGURE C.3. Settling Curve of RUN 10 (25 % Leachate Ratio)

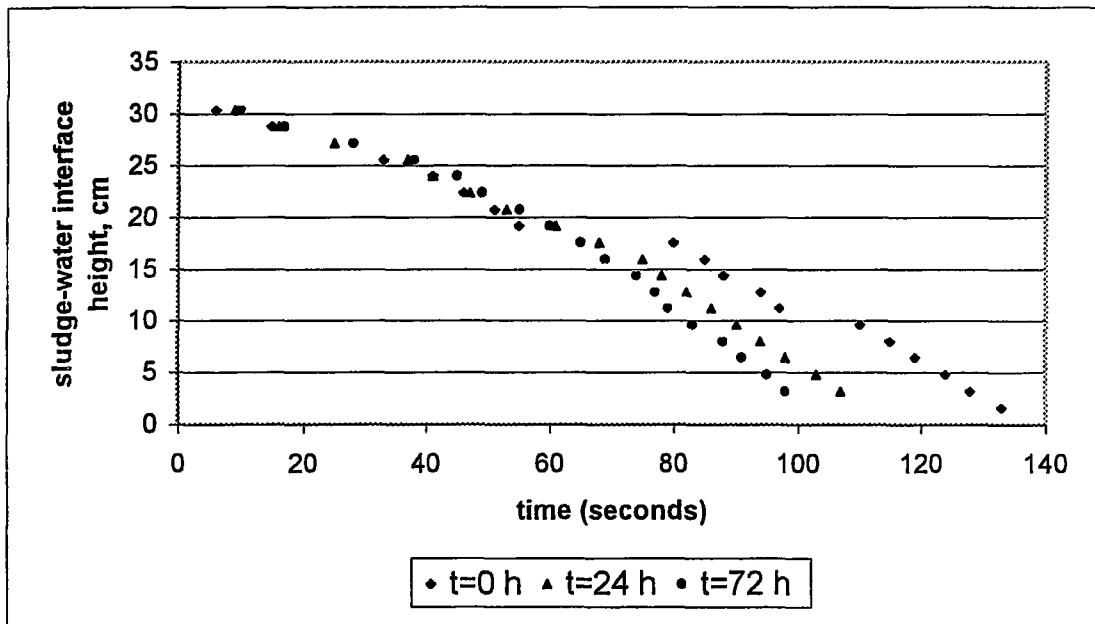FIGURE C.4. Settling Curve of RUN 12 (Diluted Leachate)

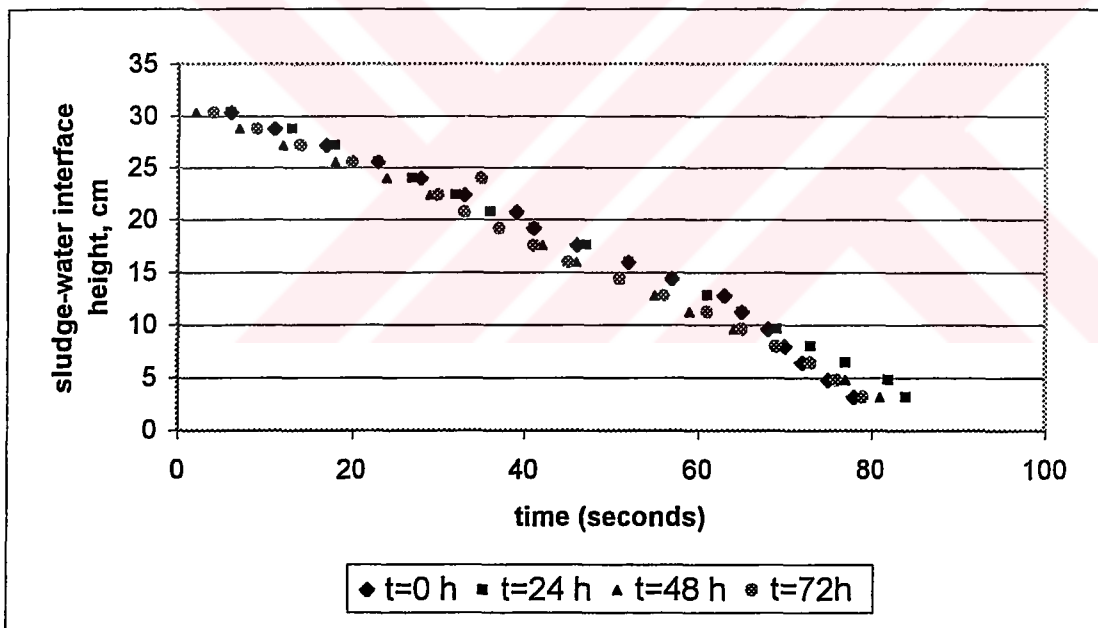FIGURE C.5. Settling Curve of RUN 13 (20 % Leachate Ratio)
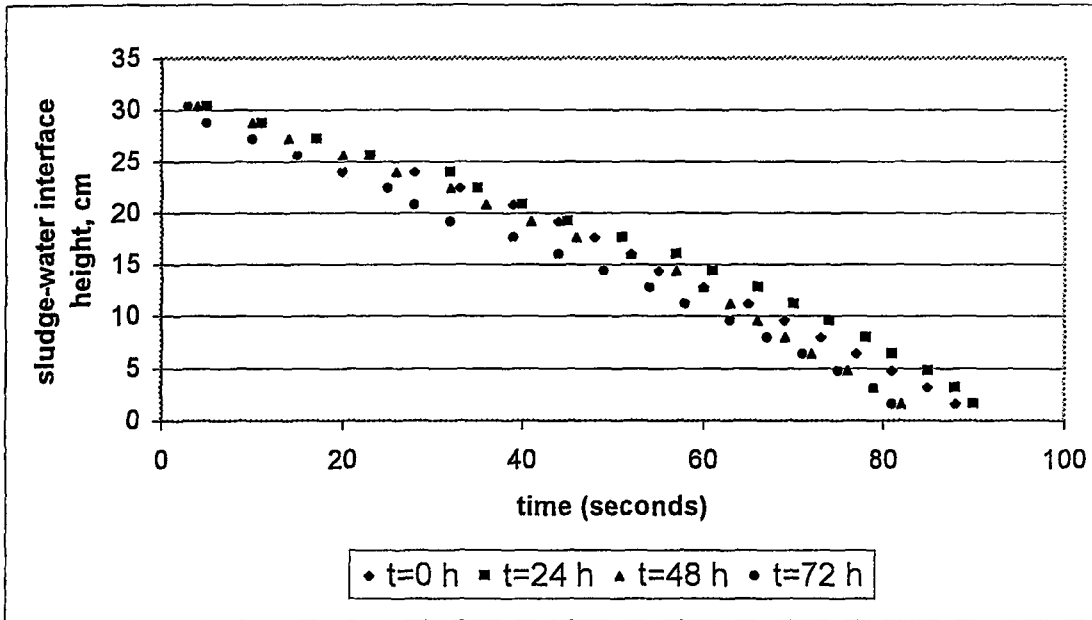

FIGURE C.6. Settling Curve of RUN 14 (15 % Leachate Ratio)

FIGURE C.7. Settling Curve of RUN 17 (10 % Leachate Ratio)