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Organic Matter and Size Distribution Transformation During a Simulated Bulk Phase of Sewer Transport

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Organic Matter and Size Distribution Transformations During a Simulated Bulk Phase of Sewer Transport

by

Paul Gardner

A Dissertation Submitted to the School of Graduate Studies in Partial Fulfillment of the Requirement for the Degree of

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Abstract

As wastewater is transported in a sewer system, in-sewer transformations occur that allow the sewer system to be considered as a pipeline reactor. Domestic wastewater from the local wastewater treatment facility was monitored for a 3 day period to simulate the bulk phase of sewer flow. Aerobic and anoxic reactive environments were studied to determine the change in organic matter and size distribution. Aerobic and anoxic conditions showed similar treatment of organic matter with an average of 55% of the total chemical oxygen demand (COD_t) removed in 72 hours. The COD_t consumption rate increased to a mean maximum of 8.4 mg/L/h a mean experiment time of 24 hours. The majority of the organic removal occurred in the first 42 hours. Aerobic conditions led to the formation of larger microbial flocs than anoxic conditions demonstrated by an increasing trend of mean area particle diameter (D_a) with the mean increase 8 microns compared to a 3 micron decrease for anoxic conditions. Initial substrate and biomass quality strongly influenced the observed transformations.

Keywords: wastewater, sewer system, pipeline reactor, particle size distribution, aerobic, anoxic, bulk phase, in-sewer transformations, downstream effects

Résumé

Quand une eau usée s'écoule à l'intérieur d'un système d'égout domestique, des transformations internes ont lieu et permettent de conceptualiser le système d'égout comme un réacteur pipeline. De l'eau usée domestique échantillonnée à partir de l'usine locale de traitement des eaux usées fut observée pendant une période de trois jours pour simuler l'écoulement de la phase massive d'un égout domestique. Des milieux réactifs, aérobique et anoxique, furent étudiés pour déterminer les changements de la matière organique et de la distribution de la taille des particules. Ces deux conditions ont démontré un taux semblable d'efficacité de traitement de la matière organique, avec une réduction moyenne de 55% de la demande chimique en oxygène totale (DCO_t), en 72 heures. Le taux de réduction de la DCO_t a augmenté jususqu'à une moyenne maximale de 8.4 mg DCO_t/L/h pour une durée expérimentale moyenne de 24 heures. La majorité du contenu organique fut réduit au cours des premières 48 heures.

La condition aérobique à mener à la formation de particules microbiologiques de tailles supérieures à celles du milieu anoxique. Le rapport moyen surface/diamètre d'une particule à augmenter de 8 μm, en moyenne, pour la condition aérobique et à diminuer de 3 μm, en moyenne, pour la condition anoxique. La qualité initiale du substrat et de la biomasse ont exercé le plus grand effet sur les transformations internes observées.

Mots clés: eaux usées, égouts, réacteur pipeline, distribution de la taille des particules, aérobique, anoxique, phase massive, transformations internes, effets en aval.

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Nomenclature

%F	Percent of particle volume from 11 to 22 μm, % particle volume
%FP	Percent of particle volume smaller than 11 μm, % particle volume
%GP	Percent of particle volume larger than 52 μm, % particle volume
%MF	Percent of particle volume from 22 to 52 μm, % particle volume
%TVC	Percent of total particle volume in channel
AR	Aeration rate, mg/L/h
ASM#1	Activated sludge model N° 1
ASVR	Automated small volume recirculator
BOD	Biochemical oxygen demand, mg/L
C	Dissolved oxygen concentration, mg/L
C*	Saturation dissolved oxygen concentration, mg/L
cfs	Cubic feet per second
CM	Completely mixed
COD	Chemical oxygen demand, mg/L
COD_p	Particulate chemical oxygen demand, mg/L
CODR	COD_t removed, mg/L
COD_s	Soluble chemical oxygen demand, mg/L
COD_t	Total chemical oxygen demand, mg/L
D_a	Area characteristic diameter, μm
DBAR	Diffused bubble aeration reactor
D_n	Number characteristic diameter, μm
DO	Dissolved oxygen, mg/L
D_p	Diameter of particle, μm
D_v	Volume characteristic diameter, μm
gpm	Gallons per minute
HRT	Hydraulic residence time, h
K_La	Volumetric mass transfer coefficient, min^{-1}

MLD	Mega-litres per day (10^6 L/d or 10^3 m^3/d)
MnP	Mean net rate of COD_p consumption, mg/L/h
MPN	Most probable number
MxS	Maximum nominal rate of COD_s consumption, mg/L/h
MxT	Maximum nominal rate of COD_t consumption, mg/L/h
NH_3	Ammonia, mg/L as N
NO_3^-	Nitrate, mg/L as N
O	Gaseous oxygen concentration, mg/L
ORP	Oxidation reduction potential, mV
OUR	Oxygen utilization rate, mg/L/h
PF	Plug flow
pH	negative log proton concentration
PO_4^{3-}	Ortho-phosphate, mg/L as P
PR	Pipeline reactor
Q	Volumetric flow rate, MLD, m^3/s
r	Rate constant for Koch and Zandi, 1973, g O_2/h/g cells
RES	Respiration rate
RMOC	Regional Municipality of Ottawa-Carleton
ROPEC	Robert O. Picard Environmental Centre
RSS	Resuspended sewer sediments
S	Substrate concentration, mg/L
S_m	Monod substrate constant for Koch and Zandi, 1973, mg/L
s.d.	Standard deviation
TDS	Total dissolved solids, mg/L
TE	Treatment efficiency, % COD_t(Influent)
T_l	Initial lag time, h
T_m	Maximum rate time, h
T_r	Readily degraded substrate consumed time, h
TOC	Total organic carbon, mg/L
TFS	Total film solids, mg/L
TS	Total solids, mg/L

TSS	Total suspended solids, mg/L
T_s	Stabilization time, h
TVS	Volatile solids, mg/L
VDS	Volatile dissolved solids, mg/L
VFS	Volatile film solids, mg/L
X	Biomass concentration, mg/L
Y	Yield coefficient (mg biomass generated/mg substrate consumed)
μ	COD_t rate constant, h^{-1}
μ_{max}	Maximum specific growth rate for Koch and Zandi, 1973, mg/L/h

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CHAPTER 1 Introduction

Wastewater, water that has been contaminated through use, is usually conveyed to a centralized treatment facility using a sewerage system or network of sewers. The time the wastewater takes to reach the treatment facility is the hydraulic residence time (HRT) of the sewer system. The sewerage system is not just a mechanical conveyance device but also an active environment. Hydrodynamic, chemical, physical, and biological processes occurring during the transport of wastewater in sewers result in substantial changes in wastewater characteristics. One observed process is the generation of hydrogen sulphide (H_2S). Hydrogen sulphide generation is an operational problem causing odours, maintenance costs and safety risks. Attempts to solve the sulphide generation problem have led to a greater understanding of the many mechanisms and reactive environments present in the sewerage system. This knowledge of in-sewer mechanistic behaviour allows the consideration of sewer pipelines as reactors [upstream of the treatment facility or discharge/reuse point].

Understanding the reactive nature of sewer systems allows them to be considered part of an integrated wastewater system with the wastewater treatment plant and the receiving water body. An integrated wastewater system will allow minimization of wastewater treatment costs. Failure to meet water quality standards is a cost. Wastewater treatment is a cost. Excess treatment is a cost. Wastewater treatment costs should be minimized while ensuring that all quality goals are met.

The transformation of wastewater in sewers is complex. Wastewater is highly variable in flow volume and quality. Many processes occur within sewer systems. Transformations in sewers may be evaluated by several methods. Wastewater transformations may be monitored in-situ either in a sewer or an analogous body (e.g., studying riverbeds to estimate sewer benthos) or wastewater may be sampled from a sewer system and transformations evaluated in a pilot or laboratory scale simulation.

Transformations may be evaluated as a change in overall organic content, changes in specific groups of organic compounds, by the respiration rate of the microorganisms, or as the consumption of electron acceptors. Understanding the transformations is necessary for appropriate application of technology. Inappropriate application of technology leads to unforeseen problems and remediation costs.

Wastewater flow within sewers may be considered a near ideal plug flow environment. The in-sewer HRT may be relatively large (up to 72 h). The use of sewer HRT for wastewater treatment offers potential financial savings. Control of sewage septicity is one application of pipeline reactor technology. Operational hazards and maintenance costs have been reduced by the addition of oxidizing agents to convert compounds to a more stable state, chemicals that raise the oxidation potential, and bactericides which destroy or lower microbial activity (Sercombe, 1995).

Pipeline reactor technology may reduce organic loading by oxidizing organic constituents to CO_2. Wastewater treatment facilities that are operating over their capacity may be remedied by pre-treating individual high strength sewer reaches. This may be an attractive solution where no space exists for wastewater treatment plant expansion or where land costs are prohibitively large. If agricultural reuse of wastewater is employed and involves significant transport, pipeline reactors may reduce required treatment facilities. Financial savings may also be realized by using pipeline reactors to minimize the cost of disposal of waste sludge. Pipeline reactors will provide a greater integration of control allowing dynamic operation of a sewer system to minimize treatment plant operating costs by maximizing pipeline treatment. Sludge disposal and treatment costs may be reduced using in-sewer treatment.

Uncertainty surrounds the choice of a relatively new technology. Operational difficulties with pipeline reactors may arise both in-sewer and downstream. In-sewer problems may occur due to pump clogging, excess biological growth, or the variable nature of wastewater composition and flow. Appropriate dosing strategies need to be evaluated for specific sewer reaches. Downstream at the wastewater treatment facility, problems may arise due to the change in influent wastewater quality (nature and size of particulate fraction).

The questions are financial and practical. Is this method of treatment cost effective? What physical factors affect the use of this technology? How much treatment is possible and at what cost?

1.1 DESCRIPTION OF DEMONSTRATION LOCATION

The Regional Municipality of Ottawa-Carleton (RMOC), located in Canada, covers approximately 2,700 km^2 with a population of 725,000. Over 10,000 km of small-diameter sanitary and combined sewers feed over 200 km of large-diameter trunk sewers in the RMOC sewage collection system. An average of 450 million L/d (MLD) of RMOC wastewater is treated at the Robert O. Picard Environmental Centre (ROPEC). ROPEC is a secondary wastewater treatment facility that treats domestic, commercial and industrial wastewater prior to discharge into the Ottawa River. ROPEC has capacity to meet the regions wastewater treatment needs until the year 2011 when the population is expected to be 900,000. As ROPEC reaches capacity, pipeline reactors may prove economical in the RMOC instead of further treatment plant expansion.

Treating distant wastewater in trunk sewers during transport to the wastewater treatment plant (reaches with sewer HRT greater than 30 h) may reduce treatment costs by reducing sludge production. Additionally, future suburban wastewater treatment needs may be partially met using pipeline reactor technology. Population expansion and new development in the RMOC is often located more than 30 km from ROPEC.

1.2 SCOPE OF INVESTIGATION

ROPEC influent wastewater was studied in a laboratory scale batch experiment to simulate a discrete element of the plug flow environment of sewer transport. Transformations of total organic content and particle size distribution of solids were measured in aerobic and anoxic reactive environments over a 72 h period. The kinetics and treatability of organic matter were evaluated.

This experiment simulated the bulk phase of a pipeline reactor to study the transformations of organic matter with specific focus on the changes in the size distribution of particulate matter. Organic matter was studied with respect to amount and

rate of treatment. Particle size distribution was studied to relate change in size distribution to kinetic changes (easily degraded substrate or hydrolysis limited regions) and with respect to downstream effects on existing or planned treatment units. Three variables were manipulated: reactive environment, initial substrate and initial microorganisms.

Aerobic and anoxic reactive environments were studied. Anaerobic conditions were not studied as they are septic (a condition to be avoided) and the transformations are generally too slow to significantly change the organic matter concentration. Dissolved oxygen (DO) and nitrate concentrations were adjusted to maintain a desired reactive environment (aerobic or anoxic conditions).

Influent substrate was manipulated to assess the response of the system to the wide range of composition that may be observed in a sewer. Initial substrate concentration and composition were adjusted by supplementing the wastewater with soluble or particulate substrate additions. Commercial dog and baby foods were used to augment the initial particulate and soluble substrate.

Influent microorganism concentration and quality were manipulated to assess the seeding requirements and determine the conditions in which pipeline reactors would be effective for augmenting treatment. Initial microorganism concentration and composition were adjusted by supplementing the initial wastewater with seed microorganisms, either activated sludge or sewer biofilm biomass. Waste activated sludge (from ROPEC secondary clarifier) and sewer biofilm (sampled locally from the ROPEC collection system) were used to augment initial biomass concentration.

The effects of aerobic and anoxic conditions on the transformation of the particle size distribution of solids were evaluated to understand downstream effects and to determine if a change in particle size distribution correlated with the conversion of slowly degraded compounds. The following sections of this thesis examine the present understanding of the sewerage environment, recent research into various aspects of pipeline reactors and evaluates pipeline reactor treatment for ROPEC wastewater.

The objectives of this project were to:

- determine the applicability of pipeline reactors to the RMOC wastewater
- determine the size distribution transformations that occur during sewer transport

- evaluate different reactive environments (aerobic and anoxic) as treatment strategies
- estimate microorganism seeding requirements
- determine if the pipeline reactor system is robust to the wide range of conditions encountered during in-sewer transport.

CHAPTER 2 Background and Literature Review

The primary objective of this work was to assess the feasibility of treating ROPEC wastewater as it is transported through the RMOC sewer system by considering the sewer system as a pipeline reactor. This chapter outlines some basic knowledge of wastewater, sewer systems and the transformations that occur during the transport of wastewater within a sewer system.

2.1 WASTEWATER CHARACTERISTICS

Different users consume water and contaminate it with different organic and inorganic compounds. The nature of the contaminants and the volume of wastewater generated are determined by the users. The users can be domestic, commercial, industrial or agricultural in nature. RMOC water users are primarily domestic with some light industrial and commercial users. Substances subject to regulatory discharge controls are considered contaminants and may be organic or inorganic in nature.

Contaminants may be suspended or dissolved within the wastewater. Substances are considered dissolved if they pass through a 2 μm filter paper. Dissolved contaminants cannot normally be seen as discrete particles but may be observed as cloudy or odiferous water. Suspended contaminants, including wood, paper, sand, fecal matter, sanitary solids, and bicycles, may be classified by composition, degradability, density and size. Suspended content is measured using either the contents retained on the separation filter paper or by subtraction of the dissolved (filtered) content from the total measured content.

The volume of wastewater generated depends on the amount and type of water users. Wastewater flow volume varies with life cycles. There is variation of flow daily, weekly and seasonally.

2.1.1 ORGANIC CONTAMINANTS

Many organic compounds are found in wastewater. Wastewater is subject to discharge regulation that considers both the overall amount of organic compounds and the amount of specific organic compounds that pose known risks. Carbohydrates, lipids, and proteins, the principal structural materials of living cells, normally form the majority of organic compounds found in wastewater.

Carbohydrates are a group of organic compounds that include sugars, starches, and cellulose. Carbohydrates may be monosaccharides, such as fructose and glucose, disaccharides, such as lactose, maltose and sucrose, or polysaccharides made of chains of many monosaccharides including cellulose, glycogen and starch.

Lipids, including fats, oils, waxes, sterols and triglycerides are a group of organic compounds that are insoluble in water but soluble in common organic solvents. Proteins are macromolecules composed of one or more chains of amino acids. Microorganisms commonly found in wastewater and sewers are mainly aerobic and consume organic compounds causing them to degrade primarily into CO_2. Typically carbohydrates are most easily consumed, followed by proteins and lipids. Substances that cannot be consumed by microorganisms may be considered biologically inert.

The overall organic content of wastewater is often defined and commonly measured as the oxygen required to fully oxidize the organic matter. Oxidation can be biological (biochemical oxygen demand; BOD) or chemical (chemical oxygen demand; COD) and is measured in units of mg O_2/L. As the BOD is time dependant, a five day period (BOD_5) is commonly used. As more substances can be oxidized chemically than biologically the COD is normally larger than the BOD. The ratio of BOD to COD varies from 0.4 to 0.8 depending on the composition, age and treatment of the wastewater. The Ontario discharge regulation is <25 mg BOD_5/L. This experiment used COD to estimate the wastewater organic content as the time required for COD tests was 4 hours compared to 5 days for BOD_5 tests.

2.1.2 INORGANIC CONTAMINANTS

Many inorganic compounds including nitrogen, phosphorus, sulphur, ammonia, calcium, sodium, mercury and heavy metals may be found in wastewater. Some inorganics, such as nitrogen and phosphorus, are essential nutrients for life while others,

such as mercury, lead or arsenic, are toxic. Inorganic compounds monitored in this project were ammonia (NH_3), nitrate (NO_3^-), and orthophosphate (PO_4^{3-}).

2.2 WASTEWATER COLLECTION SYSTEMS

Wastewater from various sources (domestic, commercial, agriculture, industry and runoff) is collected in small diameter systems. The smaller diameter pipes flow to junctions with larger pipes. Mixing occurs at junctions. The wastewater is transported to the treatment facility by large diameter trunk sewers. A schematic representation of this may be seen in Figure 2.1. Trunk sewer transport time or HRT greatly affects the wastewater quality transformations. The consideration of trunk sewers as pipeline reactors may reduce wastewater treatment costs.

Sewer flow may be driven by gravity or pumping. Gravity sewers are normally designed to flow approximately half full with a design flow velocity ranging from 0.5 to 1.0 m/s with a gradient of 0.5 to 1.5% (Nielsen et al., 1992). Gravity sewers often self-clean at high design flow rates. The possible presence of air in the head space may allow for some re-aeration of the sewer stream. The DO within a gravity sewer may range from 0 to greater than 5 mg O_2/L. The reactive environment of a sewer changes from aerobic to anoxic to anaerobic as oxygen is depleted.

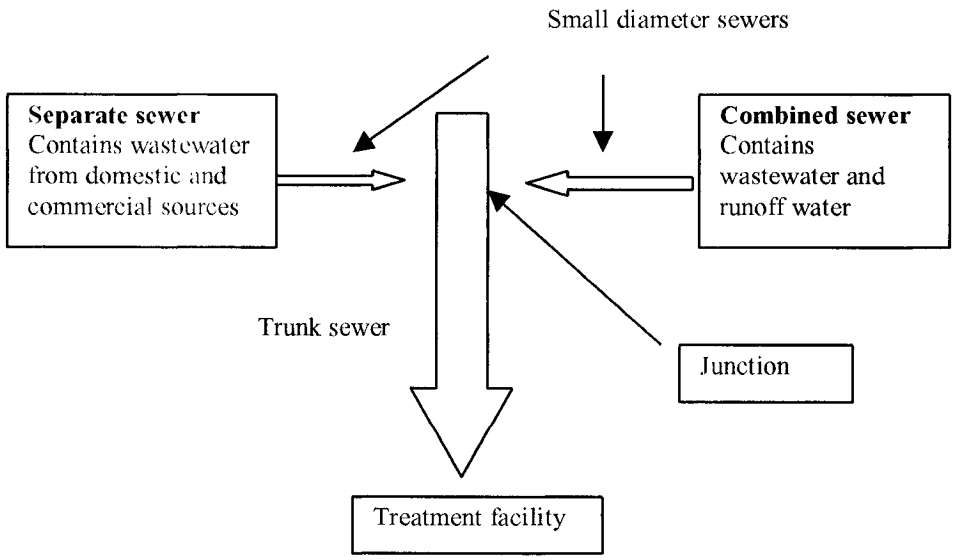


Figure 2.1 A schematic wastewater collection system

Pressure sewers are designed to be pumped periodically with a full flowing sewer. Flow velocities vary from 0.8 to 1.5 m/s (Nielsen et al., 1992) and are dependent on the

wastewater flow and pump characteristics. Flow velocities in pressure sewers are normally sufficient for self-cleansing of sewer sediments. Average flow velocities, which consider non-pumping time (up to 90% of total time (Nielsen et al., 1992)), are lower than actual flow velocities and wastewater may have a longer HRT than gravity sewers of similar length. Pressure sewers may be aerobic in the upstream (initial) portion but the lack of re-aeration ensures that anaerobic conditions are normally present downstream (after the oxygen has been depleted). Sulphide generation in the anaerobic regions is a serious operational problem causing increased maintenance costs and health risks.

Wastewater flow is made up of sanitary flow, water discharged to the sewers by users, drainage flow, originating from drainage systems including stormwater runoff and building foundation drainage, and extraneous flow, that leaks into the sewer system. If a sewer transports only sanitary flow it is a separate sewer or if it transports a combination of sanitary and drainage flow it is a combined sewer. The sewerage system that exists defines many of the characteristics of the possible transformations. Separate sewers may contain wastewater from domestic and industrial users. The flow volume and composition varies with local water use and is normally periodic with the time of day/week/month/year.

Combined sewers contain both wastewater from domestic and industrial users and storm runoff. This affects the wastewater by causing some periodic and some event caused variations. For combined sewers there may be greater than two orders of magnitude difference between extreme flow events (dry weather/low flow to storm events/high flow). Combined sewers are subject to overflow due to storm events, which leads to discharge of untreated wastewater.

The quality of runoff water is dependent on the land use of the area in question. Different land use patterns will contribute different quality wastewater (roads contribute lead and oils, agriculture contributes fertilisers and pesticides, and industry contributes heavy metals and hydrocarbons, etc.). Uncontrolled runoff may lead to health risks.

The qualitative variability of wastewater streams is dependant on many factors which include source of the wastewater (domestic or from different trades), the time of day/week/season, the sewer characteristics (flow rate, velocity, pipe diameter, gravity or

pressure flow regimes, residence time within sewer), and the level of biological activity within the sewer. The variation of these and other variables make it very difficult to model the sewer environment.

Natural selection influences the biological growth on the sewer wall (biofilm). Local conditions (water quality, geometry, hydrodynamic, atmospheric pressure) will set conditions to which the biofilm will naturally adapt.

2.3 ROPEC/RMOC

The RMOC has a population greater than 700,000 and covers 2,700 km^2 (see Figure 2.2). Land use in RMOC is approximately 44% residential, 17% commercial or institutional, 7% industrial, 13% roads, and the remainder being open space (www.city.ottawa.on.ca/inside_govt/about/about_2_en).

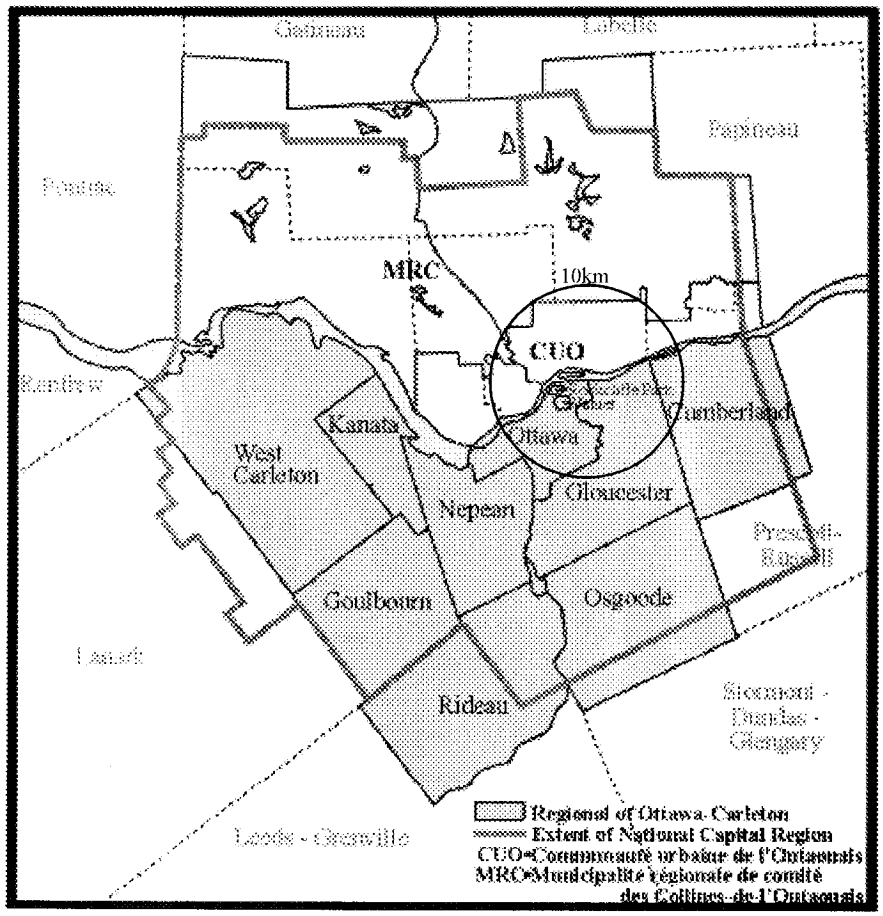


Figure 2.2 Map of RMOC

Four main trunk collector sewers serve Ottawa-Carleton. The Ottawa interceptor serves a population greater than 300,000, the Orleans/Cumberland collector 70,000, the Greens Creek collector 30,000, and the South Ottawa collector over 200,000 (RMOC, 1995). Collector sewers have a combined length of 220 km. A mixture of both combined (30%) and separate (70%) sewers transports ROPEC influent wastewater with diameters ranging from 60 cm to 3 m (RMOC, 1995). The combined collector sewers are located primarily in the old core region of Ottawa and feed into the Ottawa interceptor. The annual daily influent flow to ROPEC is greater than 450 MLD (RMOC, 1995). In-sewer HRT varies across the RMOC with distance and weather conditions. A typical dry weather HRT from Kanata to ROPEC is 4 days (Manery, K., private correspondence). It is estimated that 60% of the population of RMOC lives greater than 10 km from ROPEC and may benefit from the use of pipeline reactor technology.

2.3.1 ROPEC WASTEWATER

ROPEC influent is subject to variation daily in quality and quantity. Weather and water use patterns cause a large variation of wastewater flow. The daily influent flow to ROPEC for the month of June 1998 is shown in Figure 2.3. Wastewater production varies with water use and climatic conditions. For the month of June 1998, ROPEC mean daily influent flow was 387 MLD, however the daily influent wastewater flow ranged from 313 to 483 MLD (Hall, S., private correspondence, 1998).

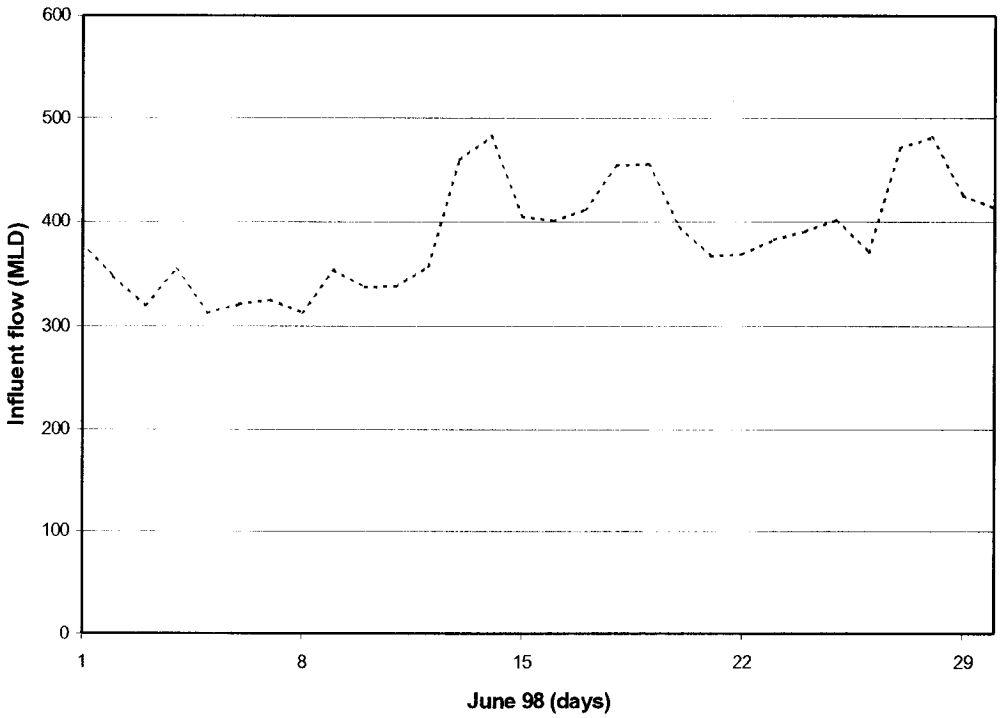


Figure 2.3 Quantitative flow variation of ROPEC daily influent, June 98 (Hall, S., private correspondence, 1998)

ROPEC wastewater quality also varies seasonally due to precipitation and runoff entering from the combined portion of the sewer network. Autumn will contribute fallen leaves and spring will contribute thawed animal feces. The influent total COD (COD_t) variation for the months of February (spring), June (summer), and October (fall) 1998 is shown in Figure 2.4.

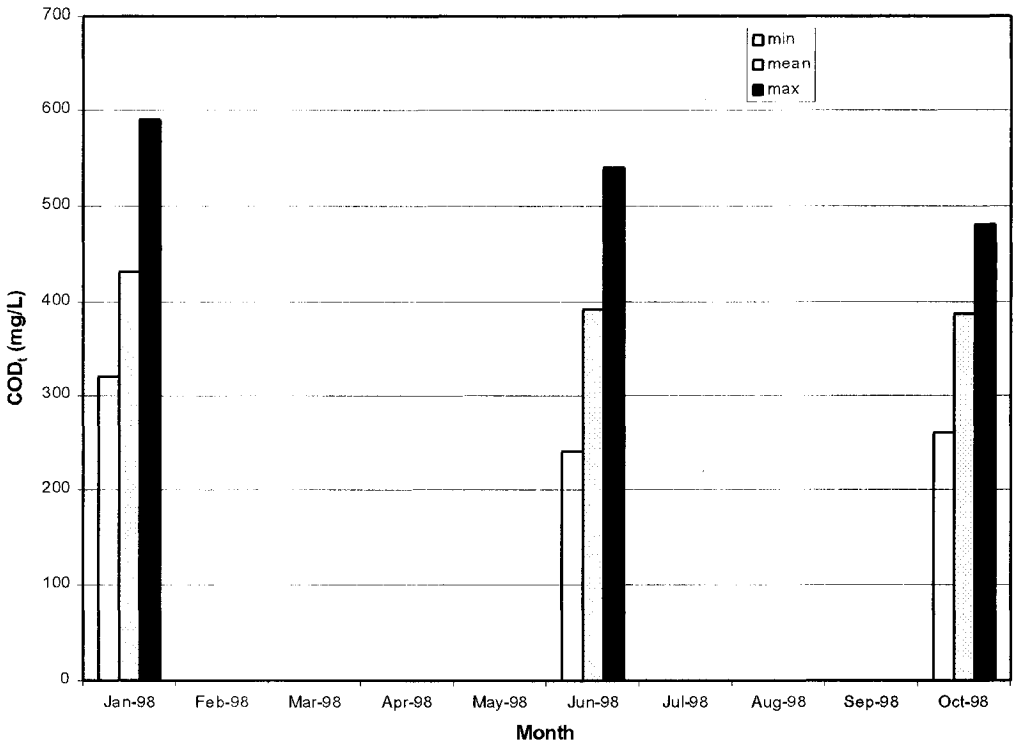


Figure 2.4 Monthly variation of ROPEC influent COD_t in 1998
(Hall, S., private correspondence, 1998)

Although the mean COD_t is fairly constant, extreme changes in wastewater composition are sometimes observed. Pipeline reactor design must consider variation of wastewater quality. Wastewater composition for the months of February, June and October 1998 are shown in Table 2.1. ROPEC mean influent composition is approximately COD_t 400 mg/L, BOD 100 mg/L, soluble COD (COD_s) 100 mg/L, total suspended solids (TSS) 220 mg/L, volatile suspended solids (VSS) 190 mg/L and ammonia 20 mg/L (Hall, S., private correspondence, 1998). October wastewater contained more SO_4^{2} and nitrogen, possibly due to fallen leaves and other autumnal debris. February wastewater contained a lower COD_s, possibly due to low sewer temperatures reducing contaminant solubility.

Table 2.1 ROPEC influent composition for February, June, and October 1998.
(S. Hall, personal correspondence, 1998)

	February			June			October		
	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
BOD (mg/L)	102	48	300	94	56	166	92	77	104
COD_t (mg/L)	441	240	1150	392	240	540	387	260	480
COD_s (mg/L)	98	85	105	118	90	145	113	84	133
TSS (mg/L)	233	124	640	212	70	270	221	169	310
VSS (% of TSS)	81	73	84	84	78	90	80	69	84
TP (mg/L)	6	3	19	4	3	6	5	4	7
TKN (mg/L)	28	22	37	28	21	34	32	27	35
NH_3 (mg/L)	20	15	23	21	13	25	22	18	24
NO_x (mg/L)	0.2	0.2	1	0	0	0	0	0	0
SO_4^{2-} (mg/L)	52	48	55	54	45	60	59	54	64

2.4 PIPELINE REACTORS

To consider a sewer as a pipeline reactor entails understanding the physical and biological properties of the sewer environment. A sewer is a three phase dynamic system (solid/liquid/gas) that has four distinct regions, bulk liquid, biofilm, benthos and head space (see Figure 2.5). Most of the wastewater is transported in the bulk liquid region. If the sewer is not full a head space exists above the bulk liquid. Biofilm grows on the sides. The bottom sediment is sometimes referred to as the benthos.

The biological oxidation of organic matter to carbon dioxide is the most important process studied in this project. Bacteria consume organic matter to sustain life. The overall process is summarized below.

Organic matter + electron acceptor → Bacteria cells + decay products Equation 2.1

(CHONPS) (O_2, NO_3, S) (CHONPS)

Decay products may be (CO_2+H_2O+NH_3+PO_4^{3-}+SO_4^{2-}).

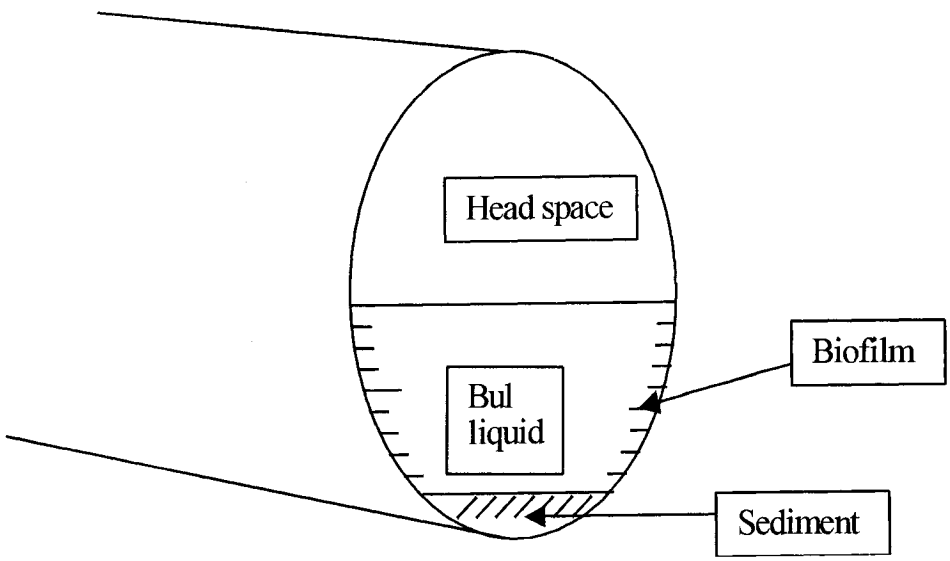


Figure 2.5 Sections of a sewer system

2.4.1 THE BULK LIQUID

The bulk liquid phase consists of the free flowing wastewater (not in contact) beyond the boundary conditions of the walls (biofilm) and the bottom (sediment). Many processes interact to affect the bulk phase (see Figure 2.6).

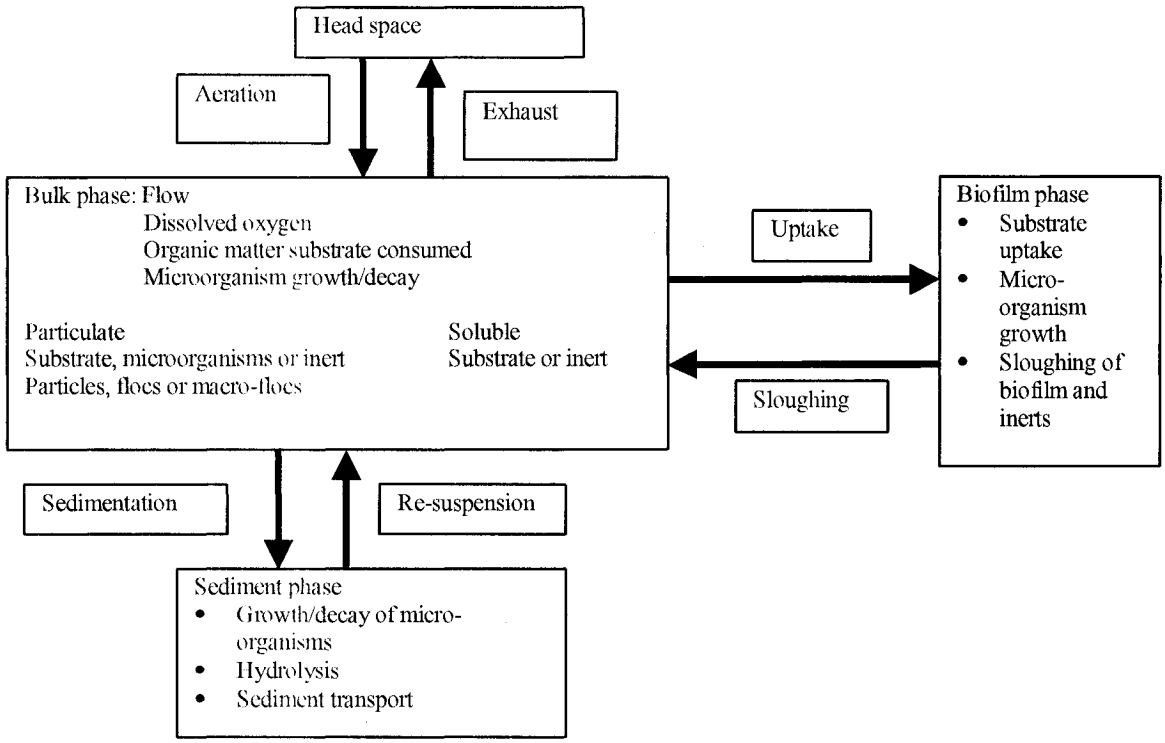


Figure 2.6 Mechanisms within a sewer system

The bulk liquid phase transports the majority of material reaching the receiving plant or discharge points. The material within the bulk phase is subject to mechanisms involving transport and water quality processes. Bjerre et al. (1998) found that the bulk liquid phase is responsible for 80% of the transformations within a sewer system. The wastewater undergoes complex physical/chemical/biological reactions in the bulk phase, including the mechanisms of:

- Flocculation/deflocculation (dispersion) – particles become larger/smaller by mechanical forces, physical-chemical reactions, and microbial action/activity
- Sedimentation/resuspension – particles settle to bed/from bed
- Hydrolysis of particulate substrate by micro-organisms into readily consumable (soluble) substrate (+inert products?)
- Microorganisms consume substrate for maintenance and growth
- Microorganisms decay/die producing inert products and consumable substrate
- Soluble substrate is taken up by biofilm, biofilm breaks off walls and enters the bulk phase

This phase is the primary area of interest of this project, bulk phase transformations of organic matter and solids.

2.4.2 THE BIOFILM

The biofilm phase consists of the attached biological growth on the side of the sewer wall. The biofilm is typically 1-3 mm thick, 14% dry solid matter containing 37-85% volatile solids (Lemmer et al., 1994). The biofilm receives substrate from the bulk phase and discharges biomass due to sloughing (see Figure 2.7). Sloughing may be caused by mechanical shear stress or insufficient nutrient penetration to the inner layer of the biofilm (Schurmann, 1987). The biofilm has a diffusion limit of oxygen permeation. This leads to anaerobic or anoxic zones close to the wall. Tanaka and Hvitved-Jacobsen (1998) observed sulphide generation and denitrification activity within a biofilm even when the bulk water was aerobic.

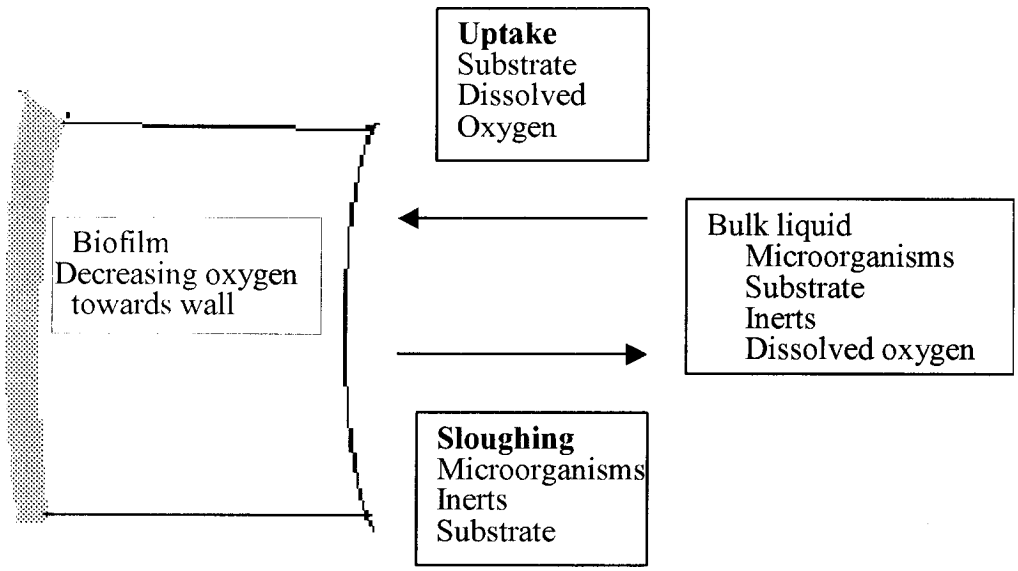


Figure 2.7 Biofilm mechanisms

A varied heterogeneous microbial consortium is present in the sewer system, microorganisms are present from both the influent waste and from the growth of sewer biofilm. A large variation in biofilm composition depending on waste conditions has also been observed (Roth and Lemmer, 1994, Nielsen and Hvitved-Jacobsen, 1998). Servais et al. (1999) concluded that most of the heterotrophic microorganisms present in

wastewater grew in the sewers. It is assumed that natural selection will cause biofilm conditioning. Microorganisms found in the sewer biofilm have been shown to be similar to those found in high load activated sludge plants (Table 2.2).

Table 2.2 Population densities of various groups of heterotrophic bacteria originating from sewer biofilms and activated sludge (Roth and Lemmer, 1994)

Bacteria	Units	Biofilm from wastewater	High load activated sludge
Heterotrophic	MPN*/g dry wt	1.3×10^{10}	2.7×10^{10}
Proteolytic	MPN/g dry wt	3.7×10^{9}	1.6×10^{9}
Polysaccharide degrading	MPN/g dry wt	8.1×10^{9}	8.1×10^{10}
Lipolytic	MPN/g dry wt	2.7×10^{9}	1.4×10^{10}
Ammonifying	MPN/g dry wt	1.1×10^{10}	1.3×10^{10}
Nitrate reducing	MPN/g dry wt	2.8×10^{11}	3.0×10^{10}
Denitrifying	MPN/g dry wt	3.0×10^{10}	3.1×10^{9}

*MPN is most probable number, an index of the number of bacteria that, more probably than any other number, would give the result shown by the laboratory examination.

The activated sludge has more polysaccharide and lipolytic bacteria than the biofilm. The long sludge age of activated sludge (typically 3 to 7 d) allows bacteria to consume a larger fraction of the available substrates. The biofilm has more denitrifying and nitrate reducing bacteria than the activated sludge. The larger concentrations of nitrate reducing and denitrifying bacteria in the biofilm indicate that this biofilm consortia may have been conditioned by an oxygen deficient environment.
In this project reactors were seeded with activated sludge and sewer biofilm to increase the initial biomass and assess the effect of different consortia. The biofilm phase is of secondary interest to this project.

2.4.3 SEDIMENT (BENTHOS)

The benthos is the sediment at the bottom of the sewer. The benthic phase is subject to the mechanisms of sedimentation, resuspension, anaerobic decay, hydrolysis and bed transport. The denser particles settle out of the liquid phase into the benthos. Some particles are resuspended, particularly during high flow events, and may contribute substantially to the organic load of the waste stream. Vollertson and Hvitved-Jacobsen (1998) studied organic matter transformations of resuspended sewer sediments (RSS). It was observed that particles with the smallest size or lowest specific gravity are preferentially resuspended. These particles have differing levels of treatibility or reactivity and consequentially affect downstream oxygen uptake. Verbanck et al. (1990) concluded that resuspended sewer sediments contributed mainly small particles (10-100 μm) with 20-40% organic content during high flow events. The sewer sediment is relatively coarse with little material smaller than fine sand (60 μm) and about 18% organic content (Nielsen et al., 1992). In this project reactors were mixed sufficiently to ensure all material remained suspended (i.e., no benthos present).

2.4.4 HEAD SPACE

Head space is the space above the bulk liquid. The head space is a gaseous mixture of air and vapours from the wastewater. If the head space contains oxygen then some aeration may occur due to the oxygen gradient. Head space can also contain carbon dioxide, methane and sulfurous fumes that can kill or be odiferous (causing complaints and health risks).

Aeration is often only from the head space atmosphere. The change in concentration of the DO in the bulk liquid phase of a sewer system fluctuates as a result of the microbial consumption processes and re-aeration. Surface aeration, turbulent effects and drop stations usually provide natural oxygenation of wastewater. Potential energy between the head and the tail of the pipe is consumed during the aeration. The available energy is constant for a pipe and additional oxygen intrusion may be required. For a system subject to aeration only from the head space atmosphere the DO balance may be given as (Nielsen et al., 1992):

$$\Delta S_O/\Delta t = K_L a(S_{O,m} - S_O) - (r_w + r_b + r_s) \qquad \text{Equation 2.2}$$

where K_La is the aeration transfer coefficient (h^{-1}), $S_{O,m}$ and S_O are the saturation and actual DO concentrations in the wastewater (mg/L), and r_w, r_b and r_s are the oxygen consumption rates in the bulk water, biofilm and sediment respectively (mg/L/h).

The aeration transfer coefficient has been found to be dependent on wastewater velocity, hydraulic mean depth, slope of the sewer and temperature (Pomeroy and Parkhurst, 1973; Nielsen et al., 1992).

2.5 REACTIONS IN SEWERS

Three different reactive environments (primary electron acceptors) are known to exist in sewers: aerobic, anoxic, and anaerobic. Each environment is characterised by the physical state and the reactions that take place. The physical differences between these environments can be observed by the DO and oxidation/reduction potential (ORP). In each environment, specific groups of microorganisms utilise specific electron acceptors to consume substrate. Some microorganisms can survive in differing environments while others can only survive in a specific environment. These categories, in order of declining ORP are aerobic (oxygen), anoxic (nitrogen) and anaerobic (sulfur) conditions.

2.5.1 AEROBIC

Aerobic conditions exist where oxygen is available in sufficient quantity to not limit heterotrophic microbial growth. This is the most active condition with the fastest rates of microbial growth and substrate consumption. This occurs when the DO is greater than 0.5 mg O_2/L and results in a large ORP in the oxygen rich water. Microorganisms in an aerobic environment use oxygen to oxidise the substrate organic material into water and carbon dioxide, capturing some of the energy released for life functions. Aerobically the reduction half reaction is:

$$e^- + \tfrac{1}{4} O_2 + H^+ \rightarrow \tfrac{1}{2} H_2O \qquad \text{Equation 2.3}$$

The DO in water varies with temperature and normally has a range from 0 to 10 mg/L. Oxygen consumption is due to direct microbial oxidation of organic matter and oxidation of reduced substances (nitrogen or sulfur).

Bulk water oxygen consumption rates in gravity sewers do not differ from those in pressure mains. As oxygen addition is often used as a sulphide prevention strategy in pressure mains, research has been done into the oxygen consumption rates (Boon et al., 1977; Nielsen et al., 1992). Bulk water oxygen consumption rates greater than 20 mg/L/h have been observed.

Biofilm oxygen consumption rates have been observed up to 1.4 g $O_2/m^2/h$ (Pomeroy and Parkhurst, 1973). Estimates of sediment oxygen consumption based on rivers have ranged from 0.4 to 2.9 g $O_2/m^2/h$ (Hickey, 1998; Boyle and Scott, 1994). In this study aerobic conditions were studied with DO maintained above 5 mg/L to ensure the presence of excess O_2.

2.5.2 ANOXIC

Anoxic conditions are characterised by denitrification of nitrate to nitrogen gas. This occurs when the DO is low (DO<0.5 mg/L) and ORP is negative or slightly positive. Denitrifying microorganisms selectively consume O_2, NO_3^-, NO_2^- or NO (Copp and Dold, 1998). The net reduction of nitrate to nitrogen half reaction is:

$$e^- + 1/5\ NO_3^- + 6/5\ H^+ \rightarrow 1/10\ N_2 + 3/5\ H_2O \qquad \text{Equation 2.4}$$

Organic and inorganic nitrogen is commonly present in wastewater. The processes of microbial growth, nitrification and denitrification affect nitrogen concentrations. Ammonia is produced from the decomposition of organic matter:

$$\text{Organic matter} + O_2 \rightarrow NH_3 + \text{microorganisms} + CO_2 \qquad \text{Equation 2.5}$$

Nitrifying bacteria convert ammonia to nitrates:

$$NH_3 + 2O_2 \rightarrow NO_3^- + H_2O + H^+ \qquad \text{Equation 2.6}$$

Denitrifying bacteria use nitrate as an electron acceptor to oxidise organic matter:

$$\text{Organic matter} + NO_3^- \rightarrow N_2 + \text{microorganisms} \qquad \text{Equation 2.7}$$

Denitrification is the dominant mechanism of anoxic systems. Microorganisms preferentially consume oxygen, followed by nitrate, and nitrite with a theoretically equivalent 2.86 g O_2/ g NO_3^- (Copp and Dold, 1998; Tseng et al., 1998). The nitrate reduction is least sensitive to the presence of DO whereas NO_2 and NO reduction is almost completely inhibited by the presence of oxygen (Kornaros and Lyberatos, 1998). The denitrifyers are able to switch more rapidly from NO_2^-/NO_3^- to O_2 than from O_2 to NO_2^-/NO_3^- as electron acceptors (Copp and Dold, 1998). Anoxic conditions were studied with excess NO_3^- and DO maintained as low as possible. Nitrogen in the form of ammonia and nitrate was measured in this project.

2.5.3 ANAEROBIC

Anaerobic conditions exist when the DO is zero and the ORP is negative. Anaerobic growth is slower than aerobic or anoxic. Anaerobic conditions are characterised by the reduction of sulfate to sulphide.

$$1/8\ SO_4^{2-} + 5/4\ H^+ + e^- \rightarrow 1/8\ H_2S + \tfrac{1}{2}\ H_2O \qquad \text{Equation 2.8}$$

The prevention of sulphide formation and septic conditions is a useful application of pipeline reactor technology. Kitagawa et al. (1998) and Delgado et al. (1999) showed that although sulfides may be formed in the bulk liquid phase, formation in the benthic and biofilm phases is the dominant formation mechanism under aerobic or anoxic conditions. Sulphide generation in the bulk phase may not occur until DO<0.5 mg/L and ORP< -150 mV. Sulfate reduction depends on the organic matter concentration (Nielsen et. al., 1998; Delgado et. al., 1999). Sulphide generation depends on temperature when little variation of organic matter is present (Kitagawa et al., 1998). Transformations of sulphurous compounds were not measured in this experiment.

Anaerobic conditions are generally too slow to have a significant effect on the concentration of organic components over a typical sewer residence time but may be used for pre-acidification. Anaerobic conditions are not dealt with in this project. Avoidance of sewage septicity is one of the goals of using pipeline reactors.

Attempts to understand, model and predict sewer behaviour are as old as sewer systems themselves. This section chronicles some attempts to model treatment in collection

systems. Studies of in-sewers transformations can be performed in-situ using sewers or analogous bodies, in a laboratory, bench or pilot scale simulation or theoretically by hypothesizing behaviour and solving the resulting equations mathematically.

An experimental solution of an engineering problem involves translation into a bench, pilot, or full scale experiment. The formulation of the experiment, the analytical method and the interpretation of the results must be correct for the solution to be valid. In-situ experiments must take into account sampling difficulties.

Models attempt to simulate wastewater that is highly variable in both quantity and quality. Extreme events may cause a failure of the system (model or real treatment).

2.6 MODELS OF TREATMENT IN COLLECTION SYSTEMS

Koch and Zandi (1973) developed a theoretical model of aerobic biological transformations. They assumed that a 3 phase equilibrium system exists between the oxygen in the gaseous phase and DO in the water available to microorganisms (solid). Microorganisms utilize DO to oxidize organic matter. Carbon dioxide and other gaseous byproducts are released by the decomposition of organic matter. The reduction in DO causes an oxygen gradient between the gaseous and liquid phases. At some point the oxygen utilization rate (OUR) will exceed the aeration rate (AR) and the DO will decrease. A series of completely mixed reactors was used to approximate the pipeline (plug flow) reactor using the following mass balance equations between the reactors where (i) is the number of the reactor in the sequence.

Cell balance

$$X(i-1)*Q(i-1) + [\text{Cell growth}] - [\text{Endogenous decay and maintenance}] = X(i)*Q(i) \qquad \text{Equation. 2.9}$$

Substrate balance

$$S(i-1)*Q(i-1) - [\text{Substrate oxidised}] = S(i)*Q(i) \qquad \text{Equation 2.10}$$

DO balance

$$C(i-1)*Q'(i-1) + [\text{Aeration rate}] - [\text{Oxygen utilisation rate}] = C(i)*Q'(i) \quad \text{Equation 2.11}$$

Gaseous oxygen balance

$$O(i-1)*Q''(i-1) - [\text{Aeration rate}] = O(i)*Q''(i) \quad \text{Equation 2.12}$$

where Q(i) is volumetric flow rate(m^3/s; cfs; gpm), X(i) is microbial cell concentration (mg/L), S(i) is concentration of biologically oxidizable fraction of solid wastes or BOD (mg/L), C(i) is DO (mg/L), O(i) is gaseous oxygen concentration (mg/L), Q'(i) is volumetric liquid flow rate (m^3/s; cfs), and Q'' is volumetric gas flow rate (m^3/s; cfs). Rate of cell growth (r) was estimated assuming a Monod (mixed second-order) type growth expression.

$$\text{Rate of cell growth (r)} = k_m \times \frac{S}{(S_m + S)} \times X \quad \text{Equation 2.13}$$

where k_m is maximum specific growth rate (h^{-1}) and S_m is Monod substrate constant (mg/L). Substrate consumption was estimated assuming a constant yield factor, where Y is the yield factor (g cells /g BOD).

$$Y = \frac{\Delta X}{\Delta S} \quad \text{Equation 2.14}$$

OUR was calculated assuming a direct relationship to biomass.

$$OUR = r * X \quad \text{Equation 2.15}$$

where the rate constant (r) was assumed to be constant (0.375 g O_2/h/g cells). The aeration rate was estimated using the following equation.

$$dC/dt = K_La*(C*-C) \quad \text{Equation 2.16}$$

where t is time, C* is saturation oxygen concentration (mg/L) C is oxygen concentration (mg/L) and K_La is volumetric mass transfer coefficient (min^{-1}).

Endogenous and maintenance respiration was assumed to be negligible (sufficient substrate is always maintained so that endogenous/maintenance rates are much smaller than growth rates).

A sensitivity analysis of this model was conducted for K_La, respiration rate and initial biomass concentration (X(0)). While respiration rate and initial biomass had a large effect on DO, K_La did not.

The assumption that endogenous decay and maintenance respiration rates are negligible is not valid if the model is applied beyond the region of substrate limited growth. As readily degraded substrate becomes scarce, decay and maintenance rates make up an increasing fraction of the overall respiration rate. Different mechanisms (particularly the hydrolysis of particulate matter) limit the overall respiration (organic consumption) rate. These different mechanisms also reduce the validity of using a single Monod growth equation with constant yield and respiration coefficients. Different microbial groups will need to be represented with different growth, yield and respiration coefficients.

These assumptions simplify the model to allow solution but reduce the model's ability to accurately predict in-sewer behaviour over longer periods.
Koch and Zandi (1973) solved this system [with punch cards and patience] by iteration for a case study of municipal wastewater flowing at a rate of 8.5 m^3/min (5 cfs) in a 30.5 cm (12-in.) diameter pipeline. A 30% BOD reduction (initial BOD 150 mg/L) required a 42.7 km (26.5 mile) pipeline with one oxygen or three air injection points. This distance corresponds to a 6 h HRT (flow velocity about 7 km/h) for a 45 mg BOD/L decrease (7.5 mg/L/h zero order rate). A 30% BOD reduction is a reasonable assumption for readily degraded substrate not to be limiting.

Green et al. (1985) evaluated the feasibility of using pipeline reactors to solve the problem of a wastewater treatment plant operating over capacity. They performed a bench scale (15 L reactor vessel) simulation of the Tel Aviv sewerage system as a step-fed, plug flow reactor. They assumed that microbial seeding would be achieved using an

8 km return biological sludge pipeline and a 10 h retention time for the loop. The continuous feed was simulated using 4 feed pulses (see Table 2.3).

Table 2.3 Reactor influent for Green et al. (1985)

Time (h)	Volume added to reactor (% of reactor volume)	Contents added to reactor
0	0.3	Return sludge
2	11.3	500 mg/L COD
3.7	13.3	500 mg/L COD
5	43	900 mg/L COD
8	32.1	560 mg/L COD
10	Termination of experiment	

The system was studied for two initial microorganism concentrations and under differing pressures (atmospheric in all stages or atmospheric in all stages except the third stage, which was at two atmosphere pressure). Initial microorganism concentration and flow regime (gravity/pressure) were found to influence the overall treatment. High initial biomass atmospheric pressure and low initial biomass 2-atmosphere pressure systems provided better treatment than low initial biomass atmospheric pressure system.

Green et al. observed a 73 to 81% COD removal that corresponded to an 85 to 93% BOD removal. The BOD/COD ratio changed from about 0.5 to 0.2. Microscopic inspection of the flocs showed an average floc size of 16 (s.d.=7.6) μm for the atmospheric system and 11 (s.d.=5) μm for the pressurized system. Final effluent had an average BOD of 31.6 mg/L (COD of 77.7 mg/L). This is very close to the discharge standard of 25 mg BOD/L.

A 50% construction cost savings was estimated by using the pipeline reactor system over doubling the size of the existing treatment plant. No reduction in operating or maintenance costs was estimated between the two systems (plant expansion or pipeline reactor treatment).

Kaijun et al. (1995) evaluated natural degradation of wastewater at various temperatures under aerobic, anaerobic and micro-aerophilic (anoxic) conditions. Six litre reactor vessels were used and experimental periods lasted up to thirty days. It was observed that the anaerobic system was much less reactive than the aerobic or micro-aerophilic systems. The use of seeding by a hydrolysing upflow sludge blanket provided hydrolysis bacteria that increased the rate of treatment. The majority of treatment occurred within the initial 48 hours for the aerobic system and within the initial 72 hours for the micro-aerophilic system.

Cao and Alaerts (1995) used a synthetic wastewater and a recirculating channel to simulate aerobic transformations and microbial population in the bulk and biofilm phases. They measured the specific activity of the biofilm and bulk phase microorganisms using a biological oxygen monitor. The biofilm had a lower specific activity than the suspended biomass (0.5 to 0.75 of suspended activity). The suspended biomass contributes greatly to the respiration process. The biofilm was observed microscopically and filaments were the predominant microbial morphology in both the bulk liquid and biofilm phases. The dominant filament was a rod shaped cell, 1.2-2 μm in diameter, 2-5 μm in length. Most of the suspended biomass was in the form of individual cells or small aggregates. They concluded that the microbial community was highly influenced by source wastewater quality but the validity of this conclusion is limited as glucose was used as the sole soluble carbon source. The rate of growth was found to increase by a factor of 1.7 as the temperature increased from 20 to 28°C.

Ozer and Kasirga (1995) studied substrate removal in the bulk and biofilm using 10 L batch reactors. Biofilm mechanisms were found to be dominant for small diameter pipelines, bulk liquid mechanisms were dominant for large diameter pipelines. A simple relationship between respiration rate and available substrate was developed to minimize the number of kinetic parameters that need to be estimated/calculated independently.

$$RES=0.0077 + 0.00064\ S \qquad \text{Equation 2.17}$$

where RES is the respiration rate in (mg/L/min) and S is the biodegradable fraction of the filtered COD in (mg/L). Respiration rate can be used to estimate COD transformations as

respiration rates converge with substrate utilisation rate. This model does not consider biomass or DO concentrations. This simple model is very useful as it only has two kinetic constants and does not require a large degree of curve fitting.

Reitsma (1997) used the diffused bubble aeration reactor (DBAR) module of Toxchem, a computer model developed by Enviromega Limited to model in-sewer transformations. The DBAR module of Toxchem uses a mixed second-order rate expression to simulate a diffused bubble aeration activated sludge system. He found that the results observed by Ozer and Kasirga (1995) could be accurately simulated if biofilm was considered as equivalent biomass showing that activated sludge type models may be used to simulate in-sewer processes. Sensitivity analysis indicated that increasing the availability of electron acceptors (oxygen, nitrate, etc.) above some limiting concentration had no further effect on transformations. The initial biomass concentration (food/microorganism ration) also had a significant effect on the overall modelled treatment. Recirculating mixed liquor suspended solids (MLSS) or increasing biofilm contact were postulated to increase biomass concentration. Reducing pipe diameter reduces the flow/surface area ratio, increasing biofilm contact. Reitsma recommended using multiple small diameter pipelines and increasing sewer wall roughness to facilitate biofilm contact.

For over 10 years, Thorkild Hvitved-Jacobsen and his colleagues have studied in-sewer transformations occurring during transport. An early study on sulphide generation (Nielsen and Hvitved-Jacobsen, 1988) performed in-situ and in a laboratory noted that various sources of wastewater resulted in a selection of bacteria that consumed the most suitable organic compounds present.

Raunkjaer et al. (1995) measured the transformations of various groups of organic components, COD and total organic carbon (TOC) during 3 h of transport in a gravity sewer. The wastewater varied in strength (with no particular trend) between 200 to 370 mg/L COD_t and 75 to 225 mg/L COD_p. Their other results may be seen in Table 2.4.

Table 2.4 Transformations observed by Raunkjaer et al. (1995)

Variable	Mean (or range) of Consumption Rate	Comments
COD_s	8 mg/L/h	Up to 30 mg/L/h observed
DOC	4 mg/L/h	3 to 6 mg/L/h
Carbohydrates	0-10 mg/L/h	No removal at concentrations below 5 mg/L
Acetate	5-16 mg/L/h	
Protein	4-9 mg/L/h	
Lipid	2.4-8.7 mg/L/h	
Ammonia		Constant (and varying) over experiment

Bjerre et al. (1998) measured the OUR of readily and slowly biodegradable organic matter and viable biomass in the Emscher river in Germany. They found that the observed data could be simulated by extending an activated sludge model (ASM#1) (Henze et al., 1987) to include 3 hydrolysis processes. Sensitivity analysis of the extended model showed the specific growth rate of heterotrophs (μ_h) and heterotrophic yield (Y_h) to have the greatest effect on the modelled results. This model did not account for biofilm processes, estimated to be responsible for less than 20% of the overall transformations in this case. This was still a simplified model of the actual transformation processes. Of the 10 kinetic constants required for this model only 2 were measured from the experimental data, 2 were selected from activated sludge modelling and the other 6 (including Y_h) were estimated by curve fitting. They concluded that the transformations depended on the nature of the organic matter present and the availability of electron acceptors. They concluded that biomass naturally selected to consume the available substrate. Although the model was extended to include up to 5 hydrolysis fractions, it was found that the use of 3 hydrolysis fractions would optimise the fit of the model.

Vollertson and Hvitved-Jacobsen (1998) measured the OUR of resuspended sewer sediments (RSS) in a lab scale experiment. They observed that the OUR could be used to estimate the effect on DO and biotransformations of RSS on receiving water bodies. Only a small portion of the RSS was found to be readily degradable or easily hydrolysable, the oxygen uptake occurred over 30 days in some cases. This corresponds to the observed phenomenon of delayed DO depletion in streams receiving combined sewer overflows. Vollertson and Hvitved-Jacobsen (1998) divided the sediment by settling velocity. Most of the material was found to settle quickly but the slowly settling material was more biodegradable.

Tanaka and Hvitved-Jacobsen (1998) evaluated changing aerobic/anaerobic conditions in a real pressure sewer and a lab scale study. The anaerobic conditions were observed to generate readily degraded substrate although 20 kinetic constants were required to simulate the results.

A process and model concept for microbial wastewater transformations in gravity sewers was published by Hvitved-Jacobsen et al. in 1998. This model employed 2 hydrolysis processes, a maintenance energy requirement, re-aeration and suspended and biofilm growth processes and can be extended to include resuspended sewer sediments, anoxic and anaerobic zones. This model requires that the wastewater be considered as unquantifyable fractions and uses 12 kinetic parameters. Curve fitting is required to use this complex model although it is still a simplification of actual in-sewer transformations. This Peterson model matrix may be seen in Appendix C.

Almieda et al. (1999) presented a model for aerobic in-sewer transformations considering the processes of re-aeration, growth and decay of heterotrophic bacteria, hydrolysis, ammonification, and biofilm consumption of substrate and oxygen. The heterotrophic growth function is a Monod type based on the ASM#1 (Henze et al., 1987). They chose to use one hydrolysis fraction as increasing the number of hydrolysis fractions leads to model identification problems that are difficult to solve using experimental evidence. They assumed that as less substrate is readily available per unit biomass, more microorganisms would use endogenous material for maintenance. Hence the microorganism decay was controlled by a switching function based on specific substrate loading (easily degraded substrate to microorganism ratio). Almieda et al.

(1999) also considered the presence of inert substances both initially in the feed and as generated as hydrolysis by-products. Sensitivity analysis confirmed that heterotrophic yield (Y_h) and maximum specific growth (μ_{max}) were the most significant parameters and should be determined experimentally. This Peterson model matrix may be seen in Appendix C.

2.7 MODEL COMPARISON

The models described above range from simple empirical models to complex theoretical heterogeneous models based on the ASM#1 (Henze et al., 1987). The models proposed by Koch and Zandi (1973), Ozer and Kasirga (1995), Hvitved-Jacobsen et al. (1998), and Almieda et al. (1999) were solved for a series of completely mixed (CM) reactors using the coefficients reported. As the original studies were conducted on different strength wastewaters these results were scaled to an identical influent COD_t of 100 mg/L. Sufficient electron acceptors were assumed present to not limit microbial growth. The COD_t transformations for the different selected model forms may be seen in Figure 2.8.

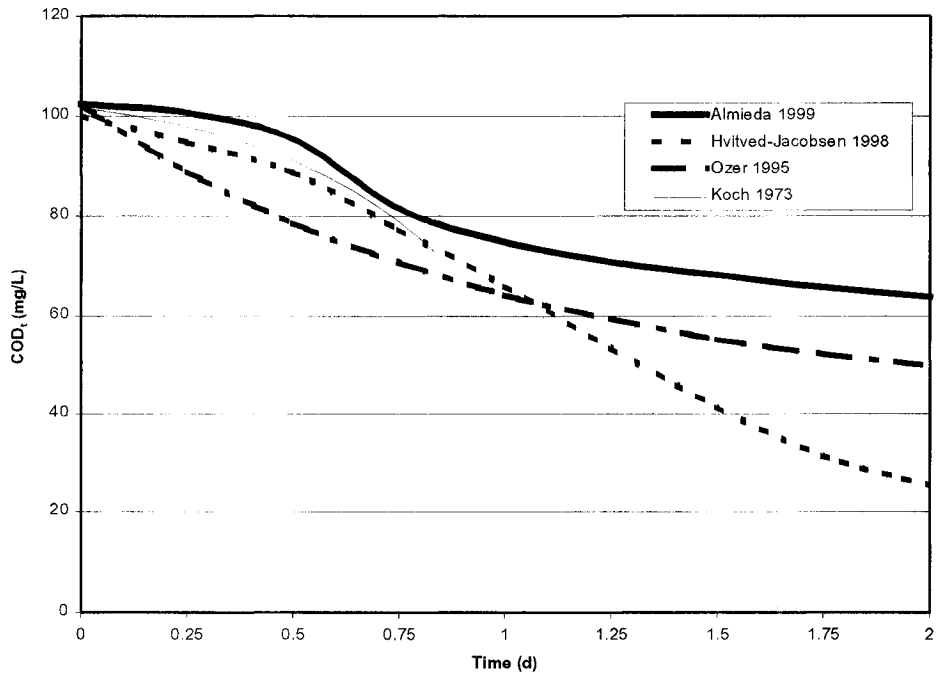


Figure 2.8 COD_t transformations for different proposed models

The models proposed by Ozer and Kasirga (1995), Hvitved-Jacobsen et al. (1998), and Almieda et al. (1999) are presented for 2 d of transport as Kaijun et al. (1995) indicated that the majority of the transformations occurred in the initial 2 d. The model proposed by Koch and Zandi (1973) was solved for only 20 h of transport as it assumed that endogenous decay was negligible. Beyond 20 h this assumption may cause significant errors.

All four models seem to describe the same observed phenomena. The three mixed second-order (Monod) models are similar until the hydrolysis/endogenous decay dominated region (following time of 0.75 days). The models of Hvitved-Jacobsen et al. (1998) and Almieda et al. (1999) differ in the hydrolysis/endogenous decay region due to differing model assumptions and the different wastewater quality from which they are calibrated.

Although the three simulations are for different influents, with different parameters for yield and growth, they all show similar qualitative behaviour. The choice

of model depends on the reason for modelling and the data available. The use of a simple model increases uncertainty due to assumptions about mechanisms. The use of a complex model increases uncertainty due to the large number of parameters that require experimental investigation or estimation by curve fitting. Sensitivity analyses by Almieda et al. (1999) and Hvitved-Jacobsen et al. (1998) have shown that the maximum specific growth rate (μ_h) and yield coefficient (Y_h) have the largest effect on model behaviour. It is recommended that these should be experimentally determined independent of any model fitting exercise. Complex models using Monod type kinetics and decay functions better represent the theoretical transformations although simple models using first-order differential equations may be sufficiently accurate for design purposes.

The COD_t consumption rates for the scaled models are presented in Figure 2.9. The exponential decay equation (Ozer and Kasirga, 1995) based model is initially at the maximum rate while the Monod (Almieda et al., 1999 and Hvitved-Jacobsen et al., 1998) based models increase to a maximum rate then decrease. The maximum rate occurs at 12 h for Almieda et al. (1999) and at 24 h for Hvitved-Jacobsen et al. (1998). Following the maximum observed COD_t consumption rate, the rate predicted by Almieda et al. declines faster than the rate predicted by Hvitved-Jacobsen et al. The availability of readily consumed substrate limits the overall COD_t consumption rate. The availability of readily consumed substrate is dependent upon the hydrolysis and endogenous decay rates.

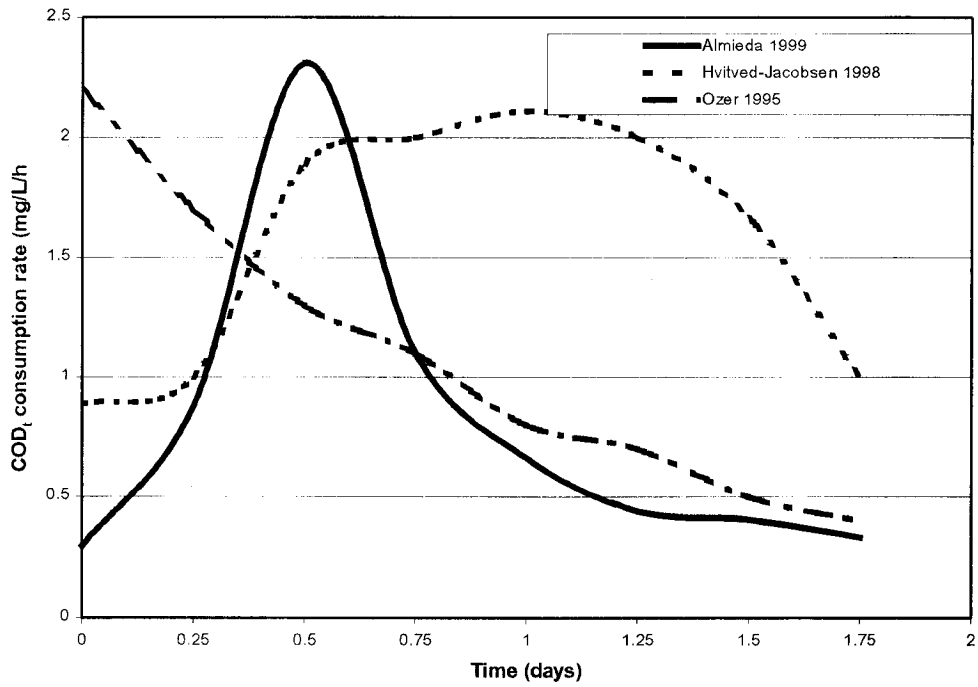


Figure 2.9 COD_t consumption rates for scaled models proposed by Ozer and Kasirga (1995), Hvitved-Jacobsen et al. (1998), and Almieda et al. (1999)

The complex activated sludge (Monod) based models are useful to predict microorganism behaviour and their effect on overall organic matter concentration. Figure 2.10 shows the microorganism concentration for the models of Almieda et al. (1999) and Hvitved-Jacobsen et al. (1998).

After approximately one day, microorganisms reach a maximum concentration from which they slowly decay endogenously. Over a larger time frame (up to 72 hours), the effect of microorganism or biomass concentration appears as only an initial lag and a small decay.

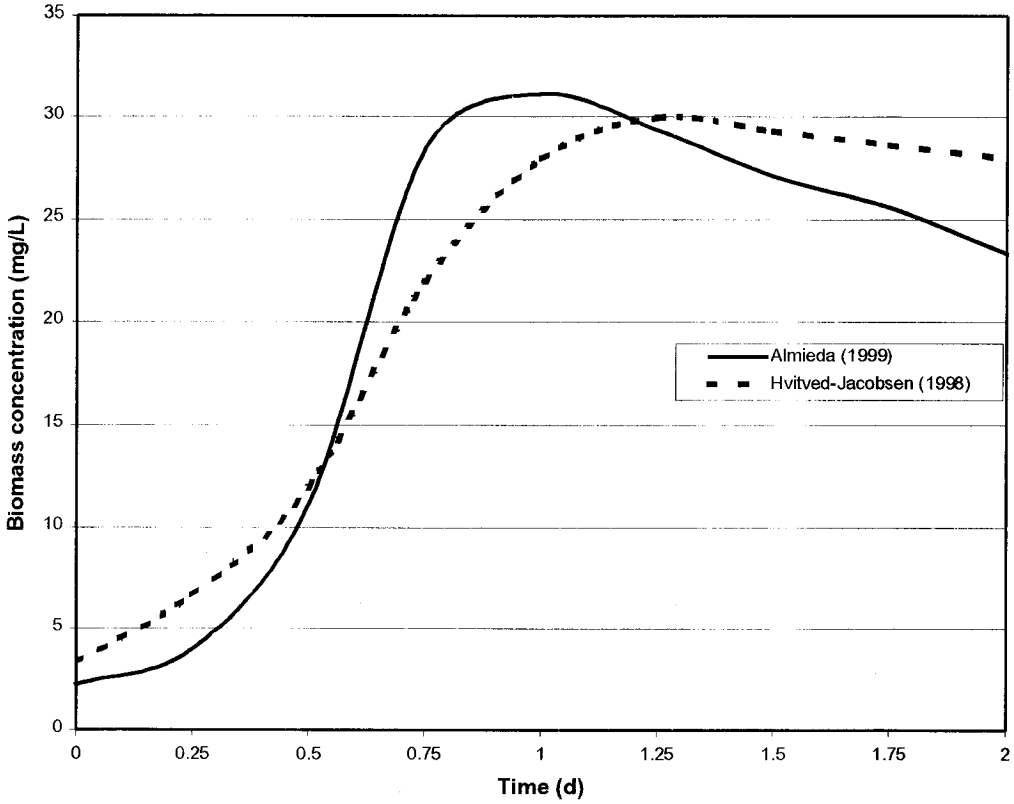


Figure 2.10 Biomass transformations for models of Almieda et al., (1999) and Hvitved-Jacobsen et al., (1998)

The model chosen should be problem oriented (Ashley et al., 1999). Full theoretical modelling of in-sewer transformations is 'naïve' (Ashley et al., 1999); the situation to be modelled is too complex as there are too many variables and interacting processes. The more complex models probably estimate the internal behaviour better but have too many coefficients that cannot be independently measured. This makes their application to real world situations difficult (Harremoes and Madsen, 1999). Specific solutions require that a large number of coefficients be independently evaluated. These models may not be robust over the wide range of influent conditions that occur in sewer transport.

The less complex models have simplifications that make their solutions limited in scope. They cannot be applied to a wide range of situations or for a long HRT as the organic consumption rate varies with quality of substrate and microbial consortia.

2.8 PARTICLE SIZE DISTRIBUTION

The secondary objective of this thesis/project was to monitor the particle size distribution of wastewater solids both for a greater understanding of transformation mechanisms and to estimate potential downstream effects on wastewater treatment facilities considering employing pipeline reactor technology.

The availability of economical high accuracy particle size analysers has led to a greater understanding of the role of particle size distribution in many processes including activated sludge. This author does not know of any studies of particle size distribution transformations for in-sewer transformations; however, Andral et al. (1999) studied the size distribution of particles in roadway runoff. Particles smaller than 50 μm were found to make up 90% of the particulate mass of roadway runoff. The particles were observed to form flocs easily. Sonification was employed to separate these flocs prior to analysis.

The organic matter transformations of pipeline reactors are analogous to activated sludge systems. Snidaro et al. (1997) and Wilen and Balmer (1999) studied the particle size of activated sludge flocs. Snidaro et al. (1997) found that particles existed on three levels. Particles were found to average 2.5 μm nominal diameter, flocs were 13 μm mean nominal diameter and macroflocs were 125 μm mean nominal diameter. Wilen and Balmer (1999) found that organic matter content and DO were the most important factors affecting floc size distribution. Smaller, more irregular and porous flocs were formed at low DO (<0.5 mg/L).

Influent particle size distribution has also been shown to affect processes involving filtration, flotation, and sedimentation.

Adin and Alon (1993), studying particle size distribution to improve the design for filtering secondary effluents, found that particle size distribution has a significant effect on particle removal and head loss in granular filtration units. The filters showed a lower efficiency removing smaller particles (D_p<10 μm). Smaller particles exhibited lower removal efficiencies at high flow rates. Larger particles (D_p>10 μm) had good removal efficiency over the range of media and flow rates studied. Filter media had a larger effect on the removal of small particles. It was concluded that particle size should not be represented by one characteristic diameter for modelling purposes.

Mackie and Bai (1993) studied the effect of influent particle size distribution on deep bed filters. Different mechanisms influenced removal of different sized particles. They observed that small particles were more difficult to remove than large particles. They also found that the presence of coarse particles increased the removal of fine particles. The deposition of small particles in the filter bed caused a greater head loss. They recommended that head loss and removal models must take particle size distribution into account.

Gregory (1997) studied the fractal nature of particle aggregates and their effect on sedimentation, flotation and filtration. The two most important properties of flocs are size and density. In many cases it has been observed that as the floc size increases, the floc density decreases. The density of flocs affects the floc strength and resistance to shear. It was concluded that large flocs are preferential for deep bed filtration and flotation, as the increased floc size enhances the probability of their capture by other flocs, filter grains or air bubbles, while smaller, denser flocs are preferential for sedimentation and de-watering processes.

Greb and Bannerman (1997) found that influent particle size distribution affects removal efficiency of wet detention ponds. Particles smaller than 30 μm were not observed to be removed in significant quantities by the wet detention pond. They suggest that greater suspended solids and pollutant removal may be achieved by controlling the quantity of particles smaller than 2 μm.

2.9 DOWNSTREAM EFFECTS

The transformations that occur in sewers have been observed to have a significant effect on wastewater quality. The effect of in-sewer transformations on the performance of existing and planned downstream process units must be considered before the application of pipeline reactors. Pipeline reactors may be considered as part of an integrated wastewater system as in-sewer changes in the organic concentration, the chemical composition, the microorganism concentration, the particle size distribution and the nutrient concentration affect downstream process units.

Primary treatment will be affected by changes of settling and flotation characteristics and by pH if chemical addition is employed. Secondary treatment is

affected by organic loading and nutrient concentrations. Filtration and wet ponds are affected by influent particle size distribution. As this is a relatively new technology, the effects on downstream treatment facilities are not well understood.

Gall et al. (1995) modified the GPS-X model, by Hydromantis Inc., to simulate the effects of reactions in a wastewater collection system on treatment plant performance. The GPS-X computer model was used to simulate both the reactive sewer collection system and the treatment facility. A 50% decrease of COD_s and a 10% increase in COD_p made up an overall 15% decrease in COD_t. This change in wastewater quality led to a 17% increase in raw sludge produced in the primary clarifier due to the increased particulate fraction. A decrease in organic loading to the aeration basins led to lower activated sludge production. This model did not account for changes of raw sludge settleability due to the increased particulate fraction. They concluded that small changes in COD_t can mask a significant variation of wastewater quality that will have some effect on treatment plant performance.

This investigation examines the possible downstream effects on the primary and secondary ROPEC treatment units and other treatment units that are strongly influenced by particle size distribution. Specifically this project assesses the applicability of ROPEC wastewater to pipeline treatment and generally assesses the transformations of particle size within aerobic and anoxic environments.

CHAPTER 3 Experimental Method

3.1 EXPERIMENTAL DESIGN

This experimental investigation was designed to study the transformations of organic matter with a specific focus on changes in the size distribution of particulate matter by simulating the bulk phase of a pipeline reactor to. Organic matter was monitored with respect to amount of COD removal and rate of treatment calculated. The particle size distribution was studied to relate change in size distribution to removal of easily degraded substrate or particle hydrolysis and with respect to downstream effects on existing or planned treatment units.

Three variables were manipulated: reactive environment, initial substrate and initial microorganisms. A differential element of bulk phase wastewater in transport was simulated using batch reactors.

Aerobic and anoxic reactive environments were studied using two identical reactor vessels. DO and nitrate concentrations were adjusted to obtain the desired reactive environment. Experiments were conducted as run pairs made up of one aerobic reactor, one anoxic reactor, using the same initial reactor conditions.

Influent substrate was manipulated to assess the robustness of the system to respond to the wide range in composition that may be observed in a particular sewer. Initial substrate concentration and composition were adjusted by supplementing the wastewater with soluble or particulate substrate additions.

Influent microorganism concentration and composition were manipulated to assess the seeding requirements and determine the conditions for which pipeline reactors can be conducive for sludge disposal. Initial microorganism concentration and composition were adjusted by supplementing the initial wastewater with seed microorganisms from activated sludge or sewer biofilm. Table 3.1 shows the initial conditions of the experimental aerobic/anoxic run pairs in which the only difference was an aerobic or anoxic reactive environment.

Table 3.1 Aerobic/anoxic run pair supplements

Run pair ID	Substrate addition	Microorganism addition
1	—	—
2	—	—
3	—	Activated sludge
4	—	Activated sludge
5	Soluble	Activated sludge
6	Soluble	—
7	Soluble and particulate	—
8	Soluble and particulate	—
9	—	Biofilm
10	Particulate	Biofilm

Twenty experimental runs were conducted as ten aerobic/anoxic run pairs. Ten runs were aerobic, ten runs were anoxic. Ten runs (5 pairs) were conducted without microorganism seed, six runs (3 pairs) were conducted with activated sludge seed and four runs (2 pairs) were conducted with biofilm seed. Ten runs (5 pairs) were conducted without substrate addition, four runs (2 pairs) with only soluble substrate addition, two runs(1 pair) with only particulate substrate addition and four runs (2 pairs) with both soluble and particulate substrate additions. Conditions were evaluated using measured and calculated responses. Variables measured were COD_t, COD_s, total solids (TS), total dissolved solids (TDS), total volatile solids (TVS), volatile dissolved solids (VDS), pH, particle size distribution, ammonia, nitrate, phosphorus and when applicable biofilm growth.

This design allows for evaluation of the following objectives:

Application of ROPEC waste to pipeline treatment (n=20)

Reactive environment

Aerobic (n=10) vs anoxic (n=10)

Initial microorganism

No seed (n=10) vs seed (n=10)

Initial microorganism source

Biofilm seed (n=4) vs activated sludge seed (n=6)

Initial substrate

No substrate addition (n=10) vs substrate addition (n=10)

Initial substrate addition type

Soluble (n=4) vs particulate (n=2) vs soluble and particulate (n=4)

Variables chosen to characterize the change in organic matter and particle size distribution over the course of the experiment are shown in Table 3.2.

Table 3.2 Responses variables selected to characterize observed behaviour

Response	Units	Description
TE	%influent COD_t removed	Maximum percent of initial COD_t removed
CODR	mg/L	Maximum amount of COD_t removed
MxT	mg/L/h	Maximum nominal COD_t consumption rate
MxS	mg/L/h	Maximum nominal COD_s consumption rate
MnP	mg/L/h	Mean net COD_p consumption rate
T_l	h	Initial lag time
T_m	h	Time of maximum COD_t consumption rate
T_r	h	Time to consume readily degraded substrate
T_s	h	Stabilization time
D_a	μm	Mean area diameter
%FP	% total particle volume	Volume of particles <11μm
%F	% total particle volume	Volume of particles from 11 to 22 μm
%MF	% total particle volume	Volume of particles from 22 to 52 μm
%GP	% total particle volume	Volume of particles >52 μm

3.2 WASTEWATER SAMPLES

Municipal wastewater was selected as reactor contents since it most accurately represents the range of compounds found in domestic sewers. Wastewater was collected from the ROPEC facility in Ottawa, Canada on June 1, 1999; the weather was dry with no rain occurring in the preceding 5 days. Although the wastewater had already undergone sewer transport with some resultant degradation, this degradation was considered insignificant as sewer transport is considered anaerobic with low reaction rates. The wastewater was collected post-grit chamber to minimise the presence of inert solids such as sand that do not undergo reactions. Wastewater was sampled by immersing a sump pump into the open waste channel and pumping directly into barrels. Approximately 600 L of wastewater was collected. Sample collection took approximately 2 h and transportation to the refrigeration unit approximately 1 h. To avoid the problem of different wastewater composition due to multiple sampling over time only one wastewater sample was collected and the wastewater was stored at 4°C until needed. The use of aged wastewater may have an effect on the composition of the wastewater even with refrigeration at 4°C (Kaijun et al., 1995). It is believed that the wastewater underwent some degradation of easily degraded substrate during transport and possibly pre-acidification during storage. Wastewater ageing is dealt with in Section 4.2. The mean reactor influent wastewater composition prior to any supplement additions for all runs and the typical ROPEC influent are presented in Table 3.3.

Wastewater analysis parameters are explained in detail in Section 3.10. The source wastewater used for this investigation would be classified as weak wastewater (Metcalf and Eddy, 1991) and it was weaker than the average ROPEC influent.

Table 3.3 Influent wastewater quality for ROPEC and experimental project prior to supplement additions

Component	Mean influent value		Standard deviation (of mean project influent)
	ROPEC	Project	
COD_t (mg/L)	400	226	73
BOD_5 (mg/L)	100		
COD_s (mg/L)	100	90	34
TS (mg/L)	220	556	241
TDS (mg/L)		448	202
TVS (mg/L)	175	211	158
VDS (mg/L)		145	115
Ammonia (mg/L as N)	20	10	3
Nitrate (mg/L as N)	0	5	2
Phosphorus (mg/L)	5	4	2

3.3 SUBSTRATE ADDITION

The effect of additional soluble and particulate substrate was studied to assess the durability of the system to a wide range of influent conditions. Baby food and dog food were chosen as substrate additions as they contain a wide variety of biodegradable components. Major ingredients of the baby food were lactose, skim milk powder, palm oil, whey protein concentrate, soy oil, coconut oil, sunflower oil, and corn syrup solids. Enfalac brand commercial baby food was added as soluble substrate to a dose of 100 mg COD_s/L. One can of baby food supplied all soluble supplements. Major ingredients of the dog food were lamb meal, rice, sunflower oil and dried egg. Nutros brand commercial dog food was added for particulate substrate addition at a dose of either 100 or 200 mg COD/L. The dog food was ground finely in a food processor and screened to 354 μm prior to being added. The compositions of major organic groups in these additives may be seen in Table 3.4.

Table 3.4 Wastewater and substrate additive composition

Component	Wastewater (Raunkjaer et al., 1994) (percent COD_t)	Commercial baby food (percent by mass)	Commercial brand dog food (percent by mass)
Protein	8 to 28	11.2	21
Carbohydrates	6 to 18	54	31
Lipids	12 to 31	29	12

Wastewater has a wide range of possible compositions. Both supplements had protein and lipid fractions within the range reported by Raunkjaer et al. (1994) although carbohydrate fractions were higher. The higher fraction of readily degradable carbohydrates will provide readily degraded substrate.

3.4 SLUDGE SAMPLES

Waste activated sludge, collected post secondary clarifier from ROPEC, was used as seed sludge. ROPEC waste activated sludge consumes ROPEC influent wastewater and is expected to have a naturally optimized microbial consortium that is acclimatized to ROPEC wastewater. No unusual operating circumstances occurred prior to collection of seed sludge. ROPEC was designed to operate with a sludge age of 4 to 7 d and actually operates with a sludge age of 3 to 5 d. Seed sludge had a VSS of 4200 mg/L. Seed sludge was added at a dose of 10 mg VSS/L wastewater.

3.5 BIOFILM SAMPLES

A locally sampled sewer biofilm was used to ensure that the biofilm could adequately consume (was conditioned to) local municipal wastewater. Biofilm seed was collected from the combined sewer located next to the Colonel By engineering building on campus at the University of Ottawa. This sewer flows into the main Ottawa collector before being transported to ROPEC. A confined access technician collected the biofilm sample from the sewer wall just below the water line. The biofilm sample appeared dark

grey, with slime and lumps, and with a total film solids (TFS) of 900 mg/L and a volatile film solids (VFS) of 720 mg/L. This is within the range for biofilm as reported by Lemmer et al. (1994).

The sample was divided into 4 equal portions for two experimental runs. The first experimental run was started immediately following biofilm sampling. The second experimental run was started three days later. The biofilm samples for the second experimental run were kept in a tap water solution at 4°C to prevent drying or decay until needed. The size of the biofilm sample did not allow excessive testing of composition prior to use. Biofilm seed dose was approximately 3 mg VSS/L wastewater.

3.6 REACTOR DESIGN

Two identical 35 by 70 cm cylindrical glass vessels with working volumes of 20 L and a headspace of 20 L were used as reactors. Reactors were maintained at 20°C for the duration of each run. A schematic representation of a reactor is shown in Figure 3.1.

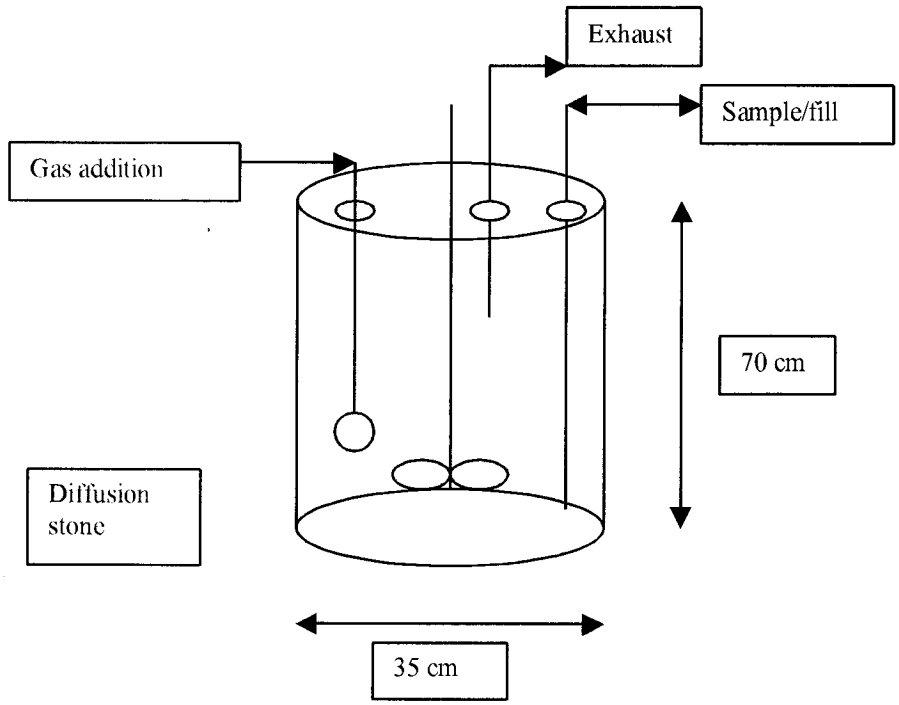


Figure 3.1 A schematic drawing of the experimental reactors

Reactor vessels contained 3-bladed internal impellers driven by an external motor at a speed of 72 rpm. This provided sufficient stirring (Reynolds number greater than

10^5) to keep all solids suspended within the reactor. Reactors were sealed using a greased rubber phalange. Each reactor was subject to constant gas addition through diffusion stones.

Air was added at a rate of 2 mL/s to the aerobic reactor to ensure surplus available oxygen. The anoxic reactor was continuously sparged at a rate of 13 mL/s using nitrogen gas from a cylinder at a rate to maintain the DO below 0.5 mg/L. Reactors were at atmospheric pressure, vented to a fumehood via tubing. Reactors were filled through ports located in the top of the reactor. Samples were withdrawn via a centrifugal pump connected to glass tubes ending near the bottom of the reactor. The reactors were covered with black cloth to exclude sunlight and resultant phototrophic growth. A photograph of the operating system may be seen in Figure 3.2.

3.7 REACTOR FILL PROCEDURE

The reactor fill procedure involved transferring 20 L of source wastewater from the storage facility to each reactor. The storage drum was first mixed to ensure a representative sample. The 40 L sample was then pumped using a sump pump into an 80 L vessel for transport to the reactors. The activated sludge seed, and the soluble and the particulate substrate additives were mixed into the 80 L vessel prior to filling the reactors. Biofilm seed was added following reactor fill-up through the sampling ports located at the top of each reactor. The reactors were filled in sequence through ports in the top of the reactor using the sump pump. The impellers and the gas additions (air to aerobic, nitrogen to anoxic) were started and 200 mg/L calcium dinitrate [$Ca(NO_3)_2$] was added to the anoxic reactor. An initial sample was then withdrawn from each reactor.

3.8 SAMPLING PROCEDURE

Samples were withdrawn every twelve hours using a centrifugal pump connected to the sampling line. The sampling procedure was devised to minimise oxygen intrusion into the anoxic reactor. The sampling procedure started with opening the clamps to the aerobic reactor. The sampling line was then primed with tap water to initiate a siphon effect. The centrifugal pump was started, the initial 150 mL of fluid was discarded and a

sample of 400 mL was taken. The aerobic line was then re-clamped and the anoxic line unclamped. Again, the initial 150 mL was discarded before the sample was taken. The initial 150 mL of sample corresponds to the volume of the tubing measured using tracer tests. Reactor temperature and DO were measured using an Orion 860 DO meter sealed within the reactor. Sample time was recorded. Testing of the samples began immediately after they were withdrawn from the reactor. A total of 7 samples was withdrawn from each reactor (16.5% of influent sample volume) for each experimental run.

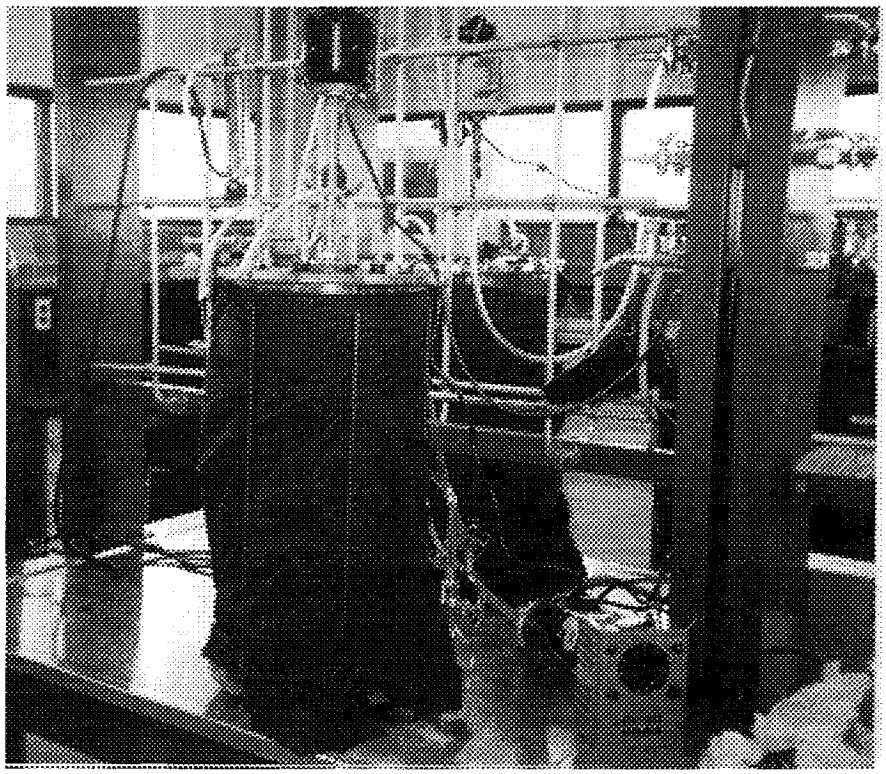


Figure 3.2 An operating reactor setup

3.9 REACTOR CLEANING AND ESTIMATE OF BIOFILM GROWTH

At the completion of each run the reactors were cleaned and tested for biofilm growth. Two methods were used to sample biofilm growth. A measured portion of the reactor wall growth was scraped off and the biofilm tested for total and volatile solids. This method did not account for differing thickness of the biofilm. The reactor was scrubbed as much as possible using a long metal spatula and rinsed using tap water. No

detergents were used to avoid their effect on microorganism growth. The water from the first rinse was also tested for total and volatile solids. Any suspended matter adhering to the reactor wall as the reactor was emptied caused this method to be biased. Reactors were rinsed repeatedly until no more biofilm could be removed (normally 5 times). Some biofilm remained in the cracks and crevices of the baffles after each cleaning. All runs, after the first, had a small amount of biofilm present at the beginning of each run.

3.10 TESTING PROCEDURE (ANALYTICAL METHOD)

Analysis of the samples began immediately after they were withdrawn from the reactor. The 400 mL sample was divided and roughly 150 mL was filtered through Whatman#1 filter papers. All tests were conducted according to Standard Methods (APHA, 1998). All reagents were prepared using reagent quality chemicals supplied by VWR. The sample was sub-sampled for testing (as shown in Table 3.5) as required. The remainder of the sample was retained for re-testing failed tests (i.e., COD above range, contamination of solids, difference between duplicates too large, etc.). Results reported are mean results of 2 or 4 repeated tests. As subsampling was considered to be the largest source of deviation for the COD and solids tests, the error for these tests were estimated as the mean difference between replicate subsamples. The error for the nutrient tests was estimated from the calibration curves. These error estimates may be seen in Appendix B. The size distribution test is described in the following section.

Table 3.5 Division of sample for testing and test procedure

Test	Unfiltered (mL)	Filtered (mL)	Total (mL)	Test procedure Standard Methods (1998)	Apparatus
COD	20	20	40	COD 5220c	
Solids	20	20	40	Solids 2540	
Ammonia		25	25	4500-NH_3 D	Orion 95-12 Fischer Accumet 750
Nitrate		25	25	4500-NO_3 D	Orion 90-07, Orion 90-02, Fischer Accumet 750
pH					Radiometer model 26
Phosphorus		25	25	4500-P D	Bausch and Lomb Spectronic 70
Size distribution	200 (screened)		200		LECOTRAC LT-100
Total	240	115	355		

3.10.1 PARTICLE SIZE DISTRIBUTION

Particle size distribution was measured with a LECOTRAC LT-100 connected to an automated small volume recirculator (ASVR). This instrument uses the principle of laser diffraction with a three-laser system, one primary on-axis and two secondary off-axis laser diodes. Particles scatter light at specific angles related to their size. A collector lens transforms the angular distribution of scattered light into a spatial distribution in the focal plane of the lens. This pattern is measured by two segmented photodetector arrays (one forward, one high angle) and is converted to particle size data. Particle size distribution was measured as the percent total particle volume in each size channel assuming particle characteristics (a requirement of the apparatus) of an absorbent, irregular particle carried in water. These particle characteristics were selected as they correspond to theoretical estimates of wastewater particles. Size ranges of 0.02 to 700 μm were measured using 120 channels.

All samples were pre-screened (at either 0.354 or 0.105 mm nominal size wire mesh screens) prior to particle size analysis to prevent damage to the testing apparatus. All samples were subject to sonification for 10 seconds at power of 25 W. The ASVR was used to ensure that all samples were handled consistently, improving repeatability and minimizing operator induced error. The use of the ASVR should prevent any sample settling or flocculation within the testing apparatus.

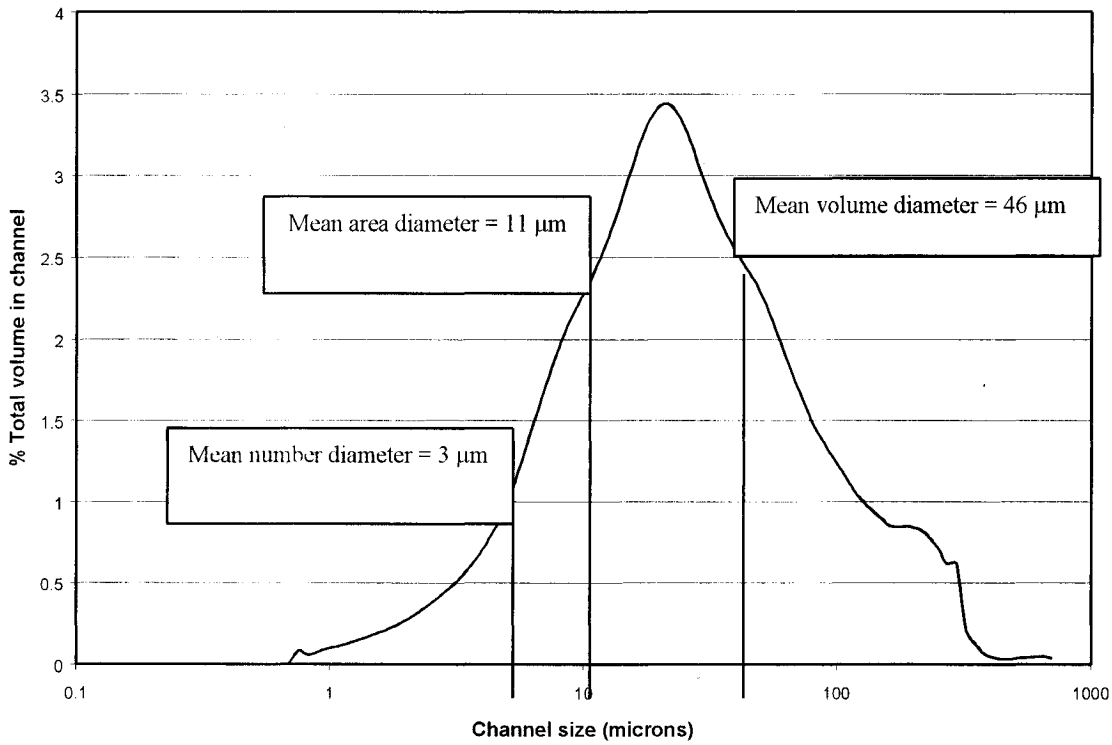


Figure 3.3 Mean influent particle size distribution

Size distribution data were recorded as the percent total particle volume in a channel. The mean influent particle size distribution for all runs is presented in Figure 3.3. It was evident that particles were not uniform in size. The LECOTRAC calculates three measures of the mean particle diameter (volume, area and number). These characteristic mean diameters may also be calculated from the percent total volume in channel data as follows (Allen, 1990):

Mean volume diameter (D_v) = Σ (%TVC) * (channel root mean size) Equation 3.1

Mean area diameter (D_a) = Σ (%TVC) * (channel root mean size)$^{2/3}$ Equation 3.2

Mean number diameter (D_n) = Σ (%TVC) * (channel root mean size)$^{1/3}$ Equation 3.3

where channel root mean size is $(D_{Lower}*D_{Upper})^{1/2}$ and %TVC is the percent of total particle volume in a channel.

These mean diameters are shown on Figure 3.3.

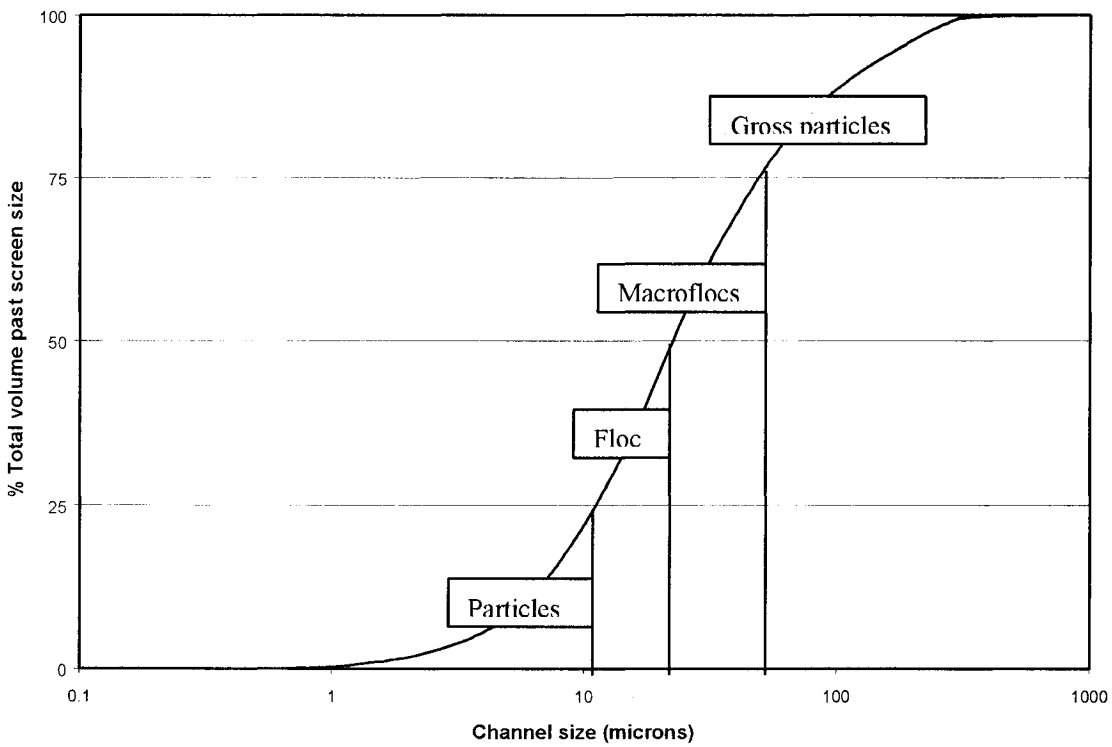


Figure 3.4 Mean influent cumulative percent total volume past screen size

The percent total volume in channel data are summed to calculate the cumulative percent total volume past screen size. The cumulative percent total volume past screen size of the mean influent particle size distribution for all runs is shown in Figure 3.4. Size ranges were chosen considering the probable size of flocs. Green et al. (1985) reported the mean floc size to be 16 μm (±7 μm) hence the floc size range was chosen as 11 to 22 μm. Particles were defined as smaller than 11 μm. Andral et al. (1999) reported that 90% of total mass of particles was smaller than 50 μm for roadway runoff. Particles larger than 52 μm were considered to be gross particles. The size range from 22 to 52

μm was considered to be an agglomeration of 2 or 3 flocs and this range is referred to as macroflocs.

The percent total volume in each size range may be calculated from Figure 3.4. Conveniently these size ranges corresponded roughly to the mean experimental influent 25, 50 and 75 percent total volume pass sizes.

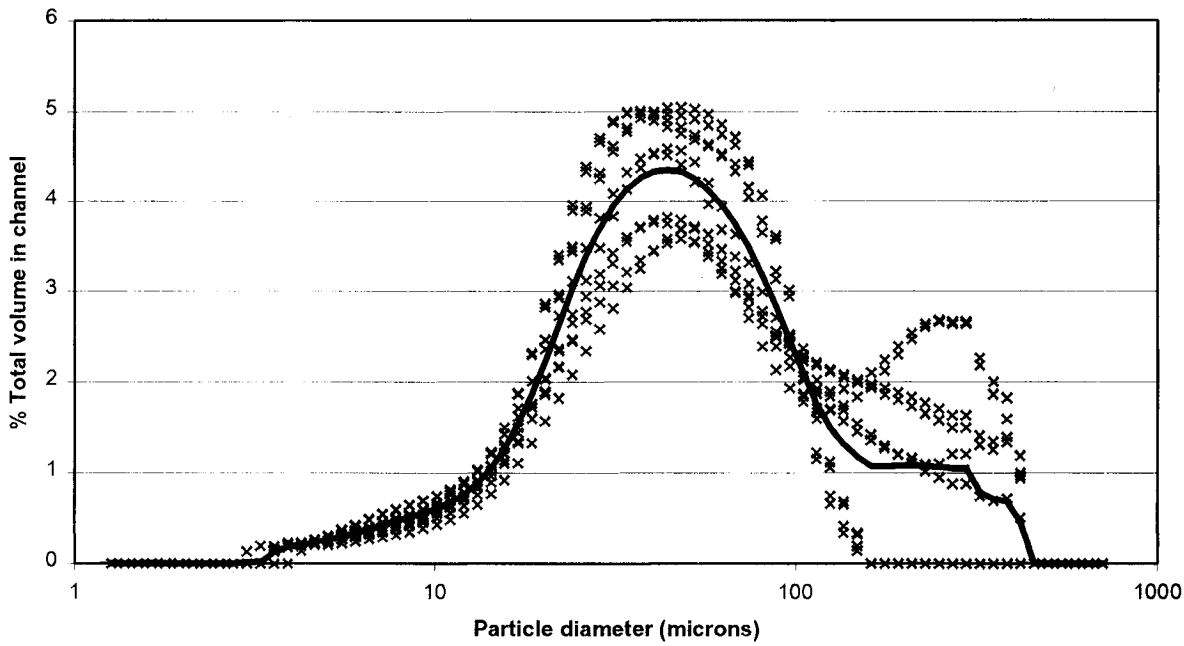


Figure 3.5 Size distribution of the error estimate subsamples

The error of various particle size distribution parameters was determined as the standard deviation of five subsamples prepared from the same sample. The size distribution of the error estimate subsamples are given in Figure 3.5. It can be seen that the volume of particles larger than 100 μm has a greater variability due to the effect of individual large diameter particles on total particle volume.

The mean, standard deviation and range of the 11, 22 and 52 μm percent total volume pass size are given in Table 3.6. The standard deviation of the percent total volume past screen was found to be approximately a sixth of the mean percent total volume past screen. The range was approximately a half of the mean percent total

volume past screen. The error of the percent total volume in a size range was estimated by summing the standard deviation of the percent total volume of a size range. Estimated errors for size distribution and other measured parameters are shown in Table 3.7.

Table 3.6 Mean, standard deviation and range of 11, 22, and 52 μm percent total volume pass size for the error estimate subsamples

Pass size (μm)	Mean percent volume past size (%TV*)	Standard deviation (%TV)	Range of samples (highest-lowest) (%TV)
11	5.3	0.9	3
22	17.5	2.8	9.3
52	57.3	9.0	26.0

*Where %TV is the percent total particle volume

3.10.2 OTHER ANALYSES PERFORMED

A settling test and a BOD_5 test were conducted to further characterize the influent and effluent quality. These tests were not conducted on all run pairs.

A limited settling test was performed to determine if the pipeline reactor significantly affected the settling characteristics of the wastewater. A reduced version of the settling test was conducted using four 100 mL graduated cylinders as the influent sample size was approximately 400 mL and direct comparison of influent to effluent quality was desired. The graduated cylinder was filled with the sample and the height of the accumulated sludge was measured initially and again after 1h, 2 h, and 6h. This test was conducted on the influent and effluent of both the aerobic and anoxic reactors for experimental run pair 7.

The BOD_5 was measured according to test 5210 B Standard Methods (APHA, 1998) to estimate the influent and effluent BOD/COD ratio for the aerobic and anoxic systems for experimental run pair 8.

Table 3.7 Estimated error for measured parameters

Parameter (units)	Method of estimation	Error estimate	Typical value or range of values
COD (mg/L)	Mean difference of replicate subsamples	40	280
Total solids (mg/L)		165 or 25%	500-800
Dissolved solids (mg/L)		40 or 20%	150-300
pH	Instrument calibration	0.1	7.0-8.0
NO_3^- (mg/L as N)		2.5	10-100
NH_3 (mg/L as N)		4	10
PO_4^{3-} (mg/L as P)		3	2-10
D_v (μm)	Replicate subsample variation, (see Figure 3.5 and Table 3.6 above)	20	20-50
D_a (μm)		15	10-30
D_n (μm)		0.45	1-10
Fine particle volume (% total volume)		1	25
Floc volume (% total volume)		4	25
Macrofloc volume (% total volume)		12	25
Gross particle volume (% total volume)		9	25

CHAPTER 4 Results and Discussion

The results and discussion chapter is organised as follows. A run pair, consisting of one aerobic and one anoxic run(O indicates aerobic, X indicates anoxic), is defined and evaluated detailing the observed transformations and mechanisms. Responses variables are identified for conditional analysis. Possible systematic errors that may bias the conditional analysis are evaluated. The overall results are presented comparing the organic matter and size distribution transformations for the different conditions including aerobic versus anoxic reactive environment, seeding, and available substrate. The downstream effects of the observed transformations are considered and a design study was performed to assess the technical feasibility of treatment assistance in sewer systems.

4.1 TYPICAL EXPERIMENTAL RUN PAIR

Similar trends were observed over the course of the experimental investigation. Run pair 5 (runs 5O and 5X) was chosen to demonstrate the typical case. The source wastewater was augmented with activated sludge seed (COD_t of 15 mg/L) and baby food feed (COD_s of 100 mg/L) for run pair 5. Furthermore 200 mg/L NO_3^- was added to the anoxic reactor. This gave initial conditions with a large biomass consortia and readily degraded substrate. Since many variables were monitored, a typical experimental run pair was evaluated for all measured parameters and response variables selected for overall analysis were identified. These responses were chosen for conditional analysis to characterize the amount and kinetics of the organic matter and the particle size distribution transformations.

4.1.1 INFLUENT

The initial samples were withdrawn immediately after the experimental reactors were operating. The initial reactor contents must be similar for a valid comparison of results to be conducted. Table 4.1 shows the initial conditions for the anoxic and aerobic reactors as well as overall mean influent and error estimates. The influent wastewater quality was similar for the aerobic and anoxic reactors. The difference between aerobic

and anoxic influent water quality parameters was less than the error estimate for all water quality parameters except COD_s. The sequential reactor filling procedure may have caused the higher concentration of CODs in the anoxic reactor. Systematic error resulting from reactor filling procedure is discussed later in Section 4.2.

Table 4.1 Influent quality for run pair 5

Variable	Mean project influent (n=40)	Estimated error	Aerobic (n=2)	Anoxic (n=2)
pH	7.2	0.1	7.3	7.2
COD_t (mg/L)	290	40	304	328
COD_s (mg/L)	180	40	184	248
TS (mg/L)	820	25%	660	735
TDS (mg/L)	630	20%	610	665
TVS (mg/L)	310	25%	105	80
NH_3 (mg/L as N)	10	4	9	8
NO_3^- (mg/L as N)	60	2.5	200	4
PO_4^{3-} (mg/L)	4	3	4	4
D_v (μm)	46	20	20	20
D_a (μm)	15	4.7	9.5	8.7
D_n (μm)	2.4	0.45	1.5	1.45

4.1.2 ORGANIC MATTER TRANSFORMATIONS

COD was used to estimate the overall organic concentration. The change in COD_t over the course of the experimental run pair may be seen in Figure 4.1. A mean overall decrease of 210 mg/L (from 310 to 100 mg/L) of COD_t was observed for run pair 5. This is a 67% removal of initial COD_t.

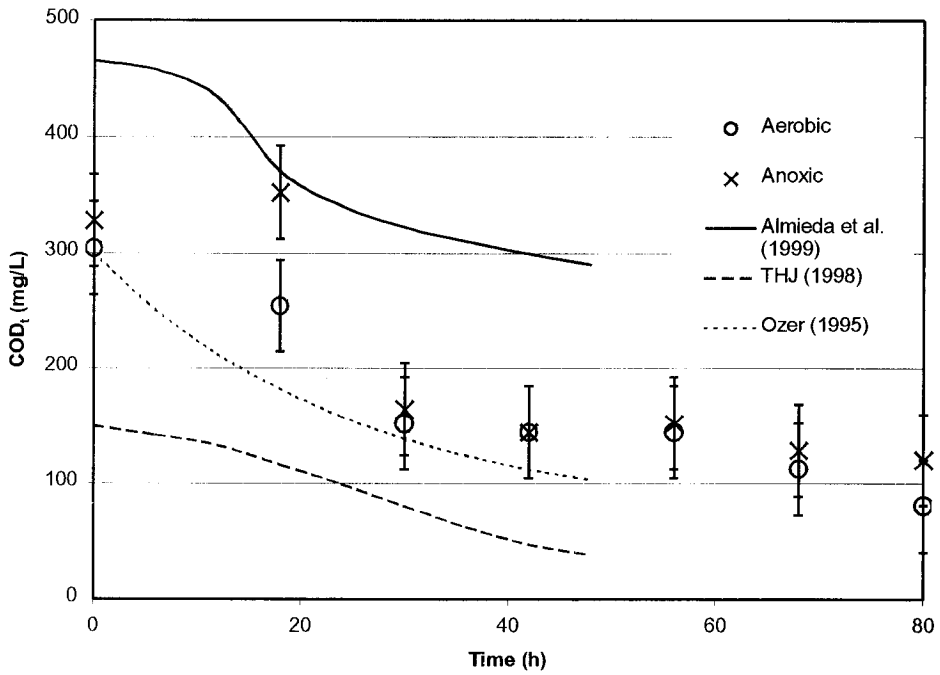


Figure 4.1 COD_t transformations for run pair 5 with error bars and modelled results of Almieda et al. (1999), Hvitved-Jacobsen et al. (1998), and Ozer and Kasirga (1995)

The transformation of total chemical oxygen demand (COD_t) with error bars for the aerobic and anoxic experimental results are compared with the modelled solutions of Almieda et al. (1999), Hvitved-Jacobsen et al. (1998), and Ozer and Kasirga (1995) in Figure 4.1. The COD_t decreased rapidly initially for the aerobic case, as the readily degraded substrate was consumed fuelling microbial growth, then slowly as the less readily degraded substrate was subject to hydrolysis (for particulate substrate) or endogenous decay (for microorganisms). The wastewater was considered stabilized at the stabilization time (T_s) when the COD_t no longer decreased. Any COD_t remaining at T_s was considered non-biodegradable. The COD_t transformations appeared similar for the aerobic and anoxic cases (except for the data points at time 20 hours), as shown by the juxtaposition of the error bars. The three published models, Ozer and Kasirga (1995), Hvitved-Jacobsen et al. (1998), and Almieda et al. (1999) are presented with different initial COD_t to clearly show the different model attributes. The experimental COD_t

transformations are qualitatively similar to all three models presented. The COD_t seemed to increase over the initial 18 hours for the anoxic case, possibly due to sampling variation. Although the increase in COD_t was less than the error estimate, the difference between the aerobic and anoxic reactors indicated that the aerobic reactor was initially more active. This may be because the activated sludge came from an aerobic environment and did not require as much acclimatization as the anoxic. Any of the three proposed model forms appears to be able to describe the data. The stabilization time (T_s), amount of COD_t removed (CODR) and the treatment efficiency (TE, the percent of the influent COD_t removed) were the responses calculated directly from COD_t measurements. The net COD_t consumption rates, calculated as forward differences, for run pair 5 are given with respect to time in Figure 4.2.

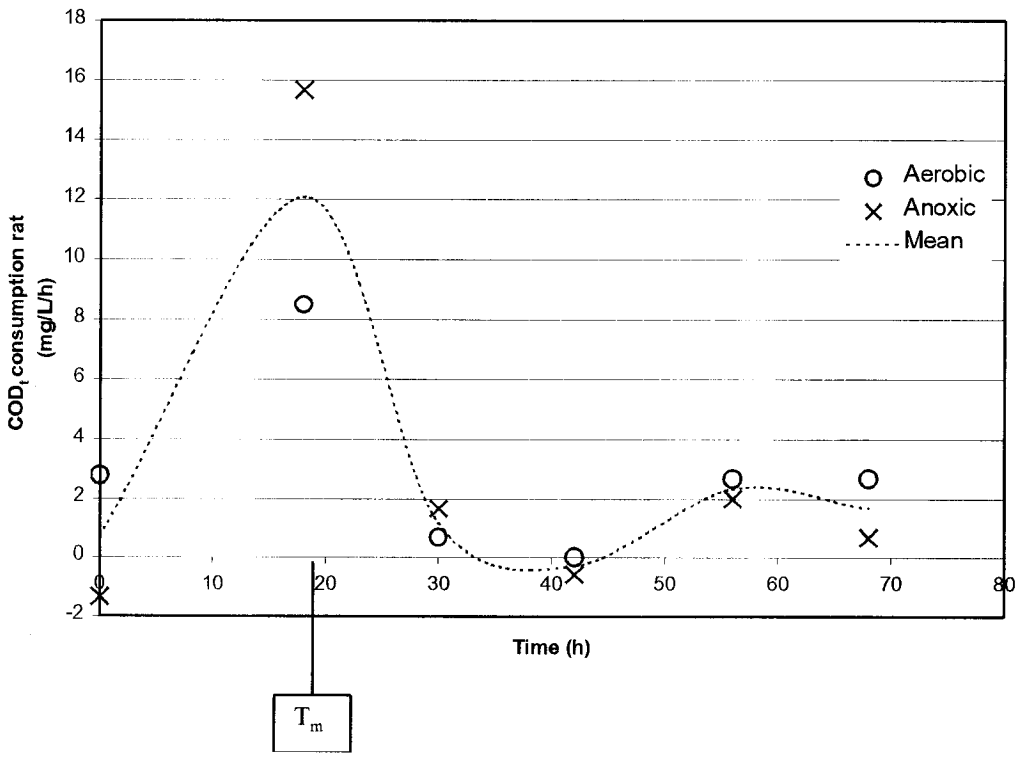


Figure 4.2 COD_t consumption rate for run pair 5

The net COD_t consumption rate increased to a mean maximum of 12 mg/L/h then decreased. COD_t consumption is initially microorganism limited with ample readily consumable substrate available, then substrate limited as the rate decreases towards zero mg/L/h as hydrolysis and endogenous decay become the dominant decay mechanisms.

This behaviour is similar to that forecast by Almieda et al. (1999) where the specific substrate to microorganism ratio determines endogenous decay.

Figure 4.2 clearly demonstrates that although a single simple rate expression cannot explain all the organic matter transformations throughout the complete cycle, the models (Figure 2.9) proposed by Almieda et al. (1999) and Hvitved-Jacobsen et al. (1998) could simulate these data. The rate was assumed to be initially limited by the availability of appropriate microorganisms, then readily degraded substrate availability and finally by the hydrolysis and endogenous decay rates. The maximum nominal COD_t (MxT) consumption rate and time (T_m) of the maximum nominal COD_t consumption rate were selected as responses. All further COD consumption rates are nominal rates.

The fraction of observed COD_t reduction with time and the pH variation are given in Figure 4.3.

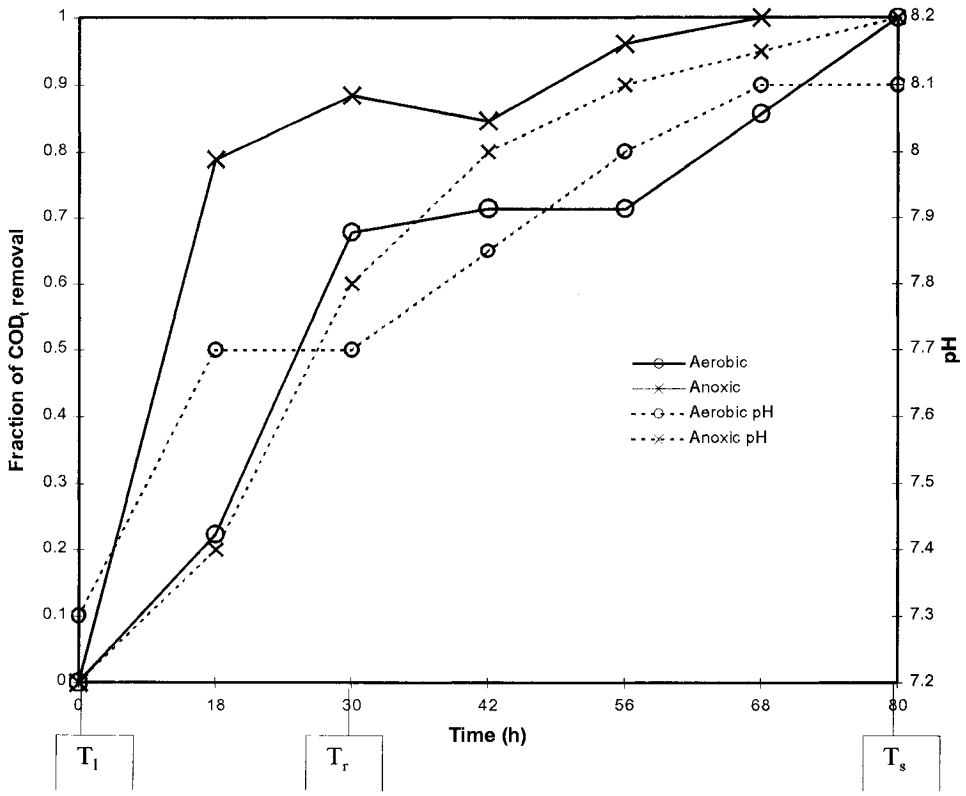


Figure 4.3 Fraction of observed COD_t removal and pH for aerobic/anoxic run pair 5

The pH showed similar behaviour to the fractional COD_t removal (Figure 4.3). The wastewater became more basic as COD_t was consumed. This agrees with theoretical H^+ consumption for both aerobic and anoxic reactions, although the net reaction of

aerobic conditions may produce H^+ (Droste, 1997). The majority (75%) of the observed COD_t reduction occurred in the initial 30 hours. This COD_t reduction in the observed first phase was the readily degraded fraction, which is a function of the composition of the wastewater and the environmental conditions of the pipeline. Initial lag time (T_l) as indicated by a change in pH was 0 h for this run pair. The time to consume the readily degraded fraction (T_r) was 30 h. The stabilization time (T_s), selected as the time after which no further COD_t reduction was observed or as the time of the final measurement, was 80 h. T_l, T_r and T_s were selected as response variables.

Organic matter is composed of soluble and particulate portions. Figure 4.4 shows the transformations of COD_s components for run pair 5. Mean net COD_s removal for both reactors was 160 mg/L, 110 mg/L COD_s of which was removed in the initial 30 hours. The mean net COD_s decreased by approximately 50 mg/L from 30 hours to the end of the experiment. Both aerobic and anoxic reactors consumed the largest fraction of the COD_s between 18 and 30 hours.

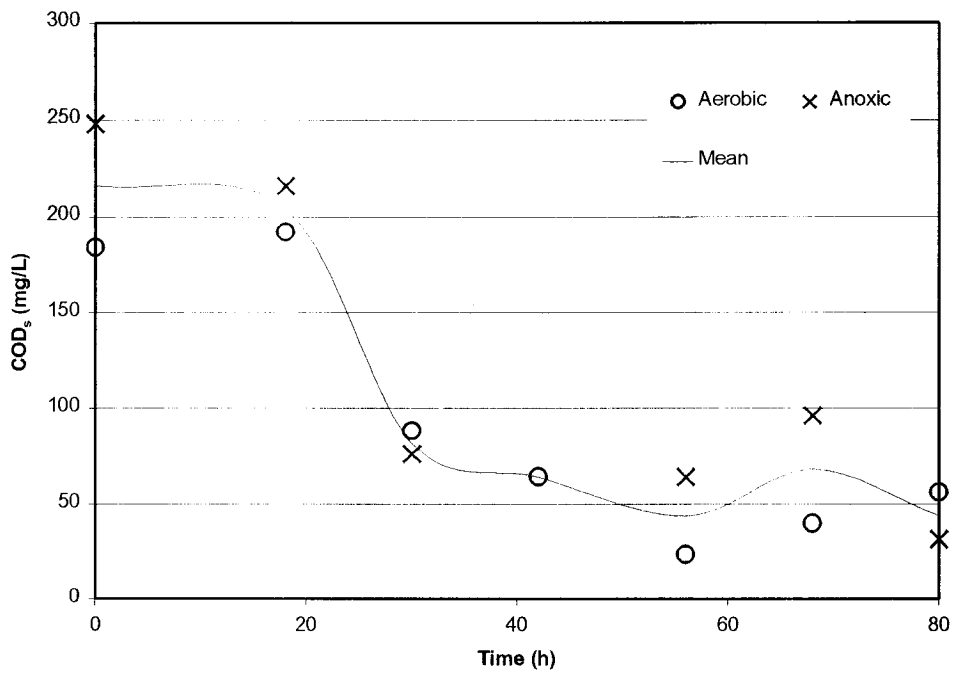


Figure 4.4 COD_s transformations for run pair 5

The mean COD_p decreased from 95 to 50 mg/L as shown in Figure 4.5. Overall COD_p showed a weak decreasing trend subject to much scatter. This was due to the fact that COD_p was calculated from two measured parameters (COD_t and COD_s), increasing error due to calculation and that the particulate matter is heterogeneous, causing a larger sampling variation. The COD_p decreased at a mean net rate of 0.6 mg/L/h.

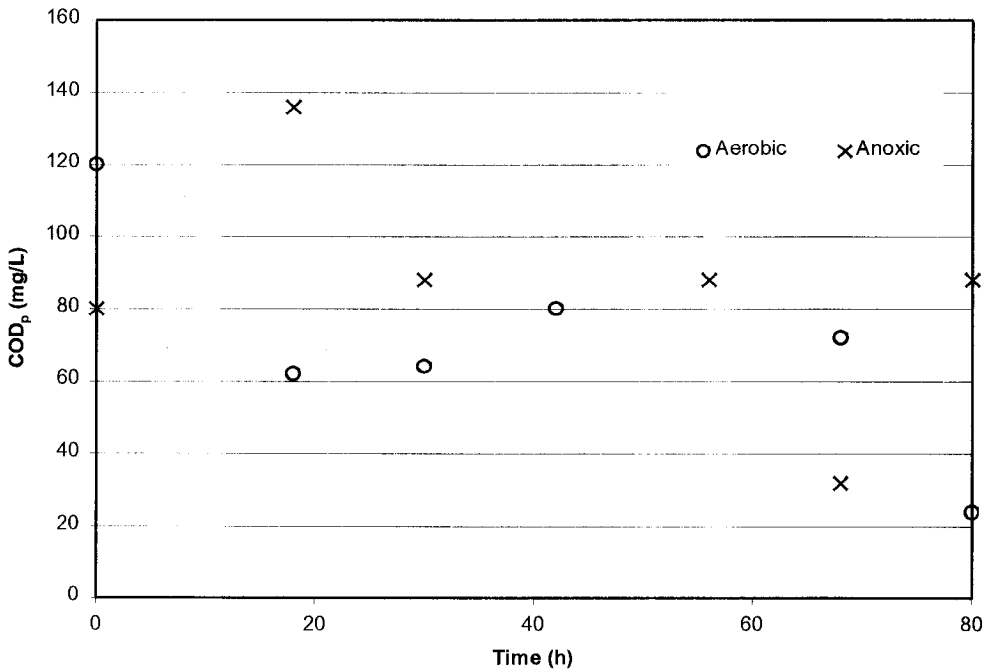


Figure 4.5 COD_p transformations for run pair 5

The soluble and particulate net COD consumption rates, calculated as forward differences, for the aerobic case are given in Figure 4.6. It can be observed that initially the soluble organics were removed as the soluble substrate is more readily consumed and finally the particulate organics were removed due to the endogenous decay of biomass and the hydrolysis of complex substrate. Negative particulate consumption rates indicate microbial growth. The negative soluble consumption rates indicate a generation of COD_s, perhaps inert hydrolysis byproducts. It is interesting to note that at t=42 h, the negligible overall net COD_t (Figure 4.2) rate actually conceals an increase of particulate matter due to the microbial growth process and a decrease of soluble substrate due to the consumption process. After the soluble substrate was fully consumed, only the COD_p

decreased. A positive nominal COD_p consumption rate was comprised of both particulate hydrolysis and endogenous decay of microbial biomass rates. Maximum nominal COD_s (MxS) and mean nominal COD_p (MnP) consumption rates were selected as responses.

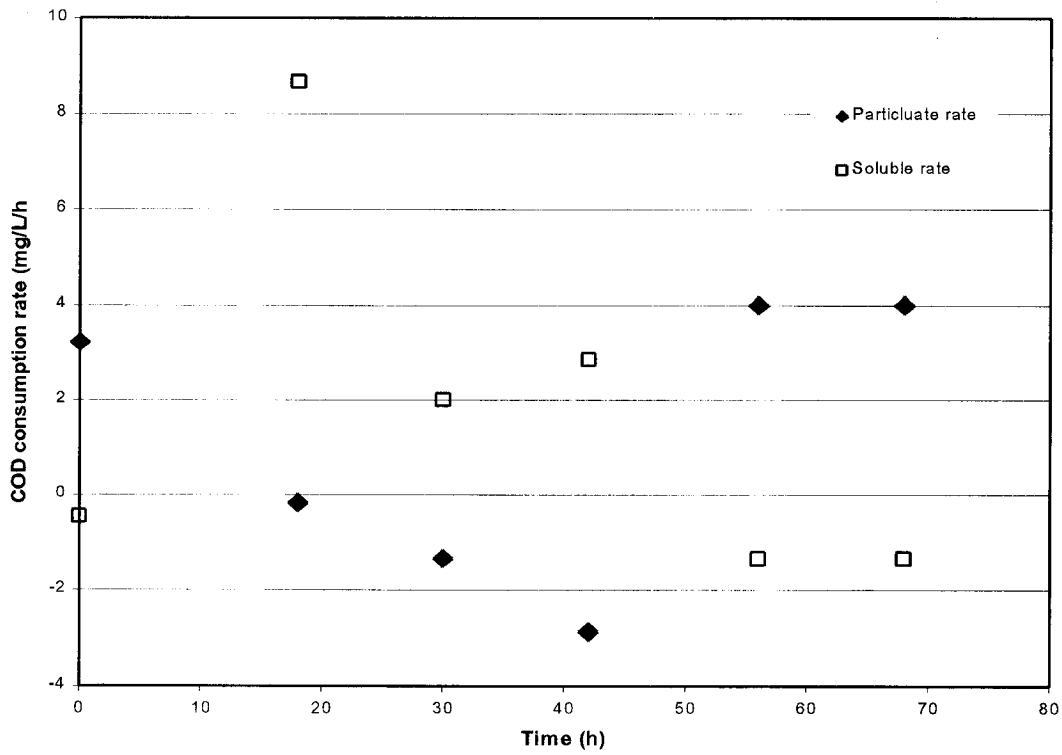


Figure 4.6 Particulate and soluble nominal COD consumption rates for aerobic case 5O

Cross reference and examination of Figures 4.1 through 4.6 allows for a comprehensive estimation of the organic matter transformations for the aerobic and anoxic cases. The transformations of COD appeared similar for both the aerobic and anoxic cases. The transformations appeared to fall into three regions: microbial growth and readily degraded substrate consumption (0 to 30 h), a transition region (30 to 58 h) and hydrolysis/endogenous decay region (58 to 80 h).

After 18 h, both reactors showed evidence of microbial activity (no initial lag). The aerobic case showed a reduction of COD_t, and no change in COD_s, possibly due to the hydrolysis of easily hydrolyzed particulate matter into soluble substrate at a similar rate to the consumption of soluble substrate. The anoxic case showed an increase of

COD_t (that may not be significant) and a decrease of COD_s. This region appeared to be limited by the overall acclimated microorganism concentration.

From 18 to 30 h the microorganisms were conditioned to the wastewater quality and present in sufficient quantity to rapidly consume the readily degraded substrate. After 30 h, both reactors appeared to have consumed all readily degradable substrate. The overall COD_t reduction (of 150 mg/L) was mostly COD_s (100 mg/L). The availability of readily consumable substrate limited the rapid reduction of organic substances.

From 30 to 58 h the COD_t did not change appreciably for either the aerobic or anoxic reactors. This appeared to be a transition zone between the region of rapid substrate consumption and hydrolysis. Further COD reduction was COD_p degradation. The hydrolysis and decay rates limit any further COD_t reduction. Hydrolysis, or endogenous decay, transforms the particulate matter into soluble substrate that is rapidly consumed by the microorganisms. The mean particulate nominal COD consumption rate was approximately 0.6 mg COD_p/L/h for this experimental run pair. The aerobic reactor appeared to show a faster hydrolysis degradation rate. For this run pair, both the aerobic and anoxic trials showed similar behaviour but this was not always the case.

4.1.2.1 Kinetic evaluation

The kinetic relationship between substrate and consumption rate was evaluated to determine if any simple rate equation could model the observed transformations. Substrate concentration, estimated by COD_t and COD_s, is shown with COD_t consumption rate in Figure 4.7.

Poor linear regression coefficients indicate that no simple relationship exists over all the regions of graph. The different regions cannot be described using a simple first-order rate constant. A first-order rate with respect to substrate may be valid after the microorganism limiting phase prior to the hydrolysis limiting phase (i.e., from 18 to 40 h for run pair 5).

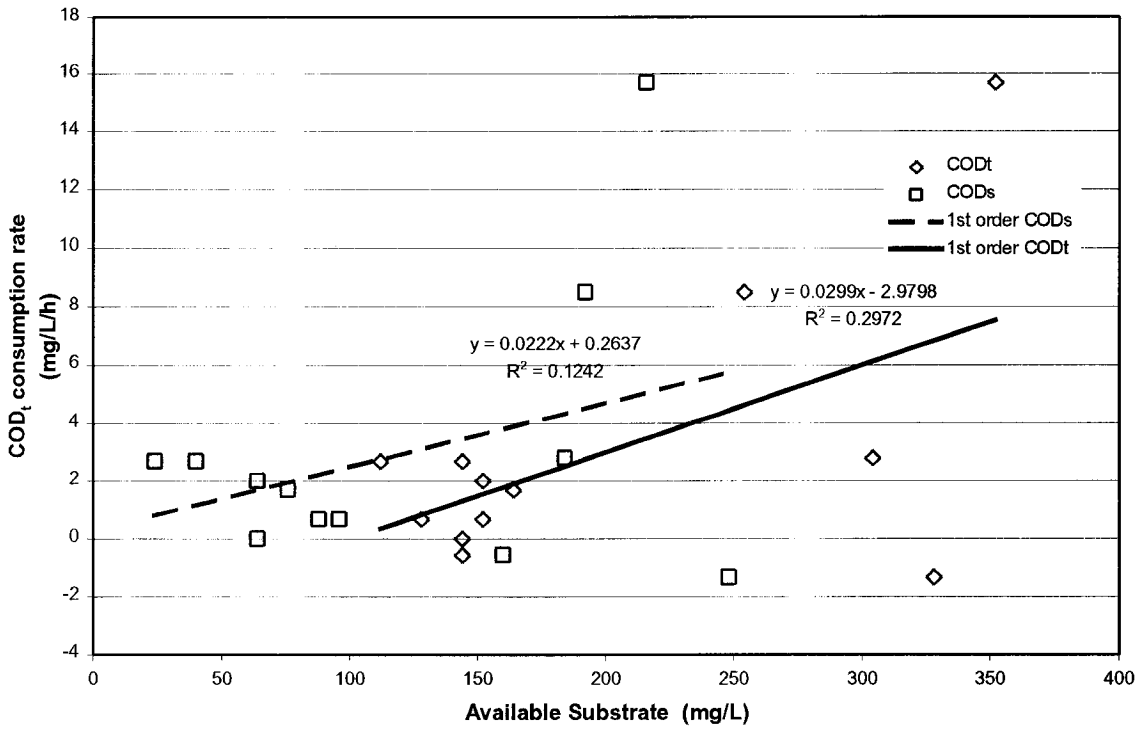


Figure 4.7 Available substrate (as COD_t or COD_s) versus COD_t rate for run pair 5

To determine if any simple biomass kinetic relationship was valid over the entire run, COD_t consumption rate was plotted against biomass (as COD_p and VSS) in Figure 4.8. No simple relationship can describe the behaviour over all the regions of Figure 4.8. Initially the microorganisms may limit the reaction but then reach a concentration where they are no longer limiting.

The wastewater substrate and microbial quality both appear to influence the kinetics. As no simple expression may be used over the entire region, it is proposed that a COD_t consumption rate expression be derived by a method similar to Ozer and Kasirga (1995). This method involves plotting the maximum COD_t consumption rate versus the biodegradable portion of the COD_t to determine the exponential decay of the COD_t. This method should allow reasonable modelling of transformations following the microbial (biomass) limiting period.

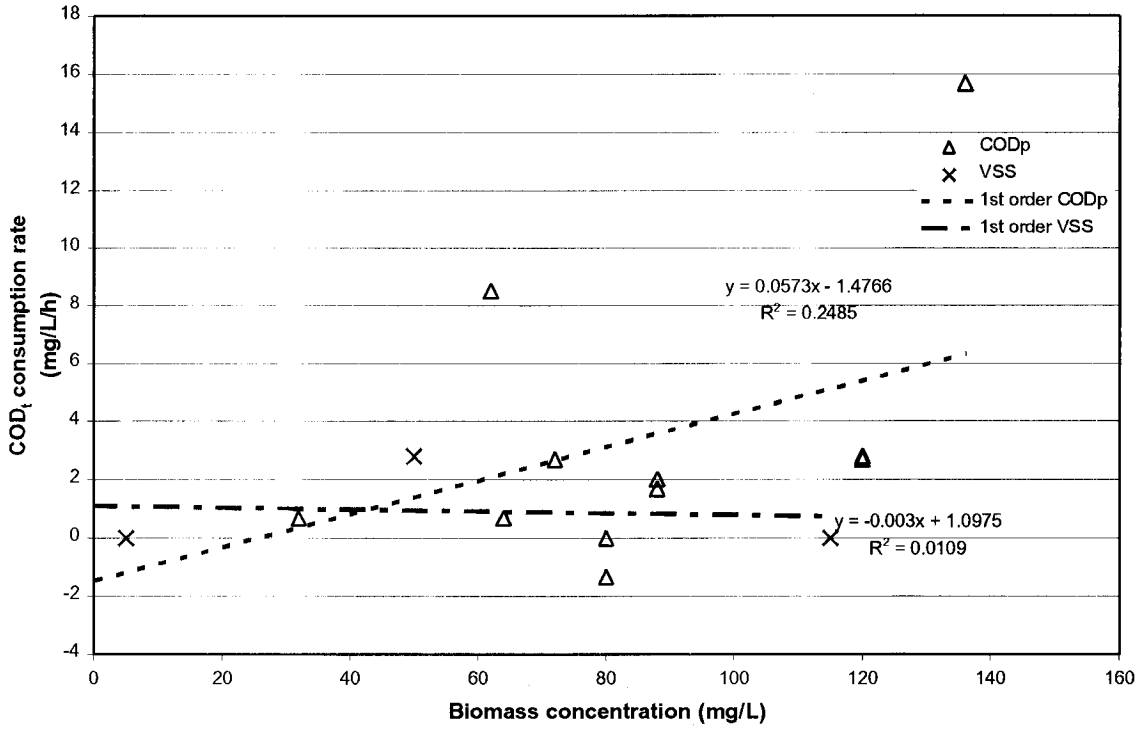


Figure 4.8 Observed COD_t consumption rate versus biomass concentration for run pair 5

4.1.2 NUTRIENTS

Nitrogen (as NO_3^--N and NH_3-N) and orthophosphate were the nutrients monitored in this study. Figure 4.9 shows nitrogen transformations for aerobic and anoxic run pair 5. Phosphorus transformations were not presented since no activity was observed although theoretically some phosphorus is consumed by microbial growth.

Nitrate was consumed in the anoxic reactor in the first 40 hours while ammonia and phosphorus remained unchanged for the duration of the experiment. This conversion of nitrate is an indication that denitrification occurred. No significant change of nutrients was observed in the aerobic reactor although some initial nitrification may have occurred as indicated by the increase in nitrate concentration. The nutrients monitored were always present and never appeared to limit microbial growth. Other aerobic runs indicated that ammonia was converted to nitrate.

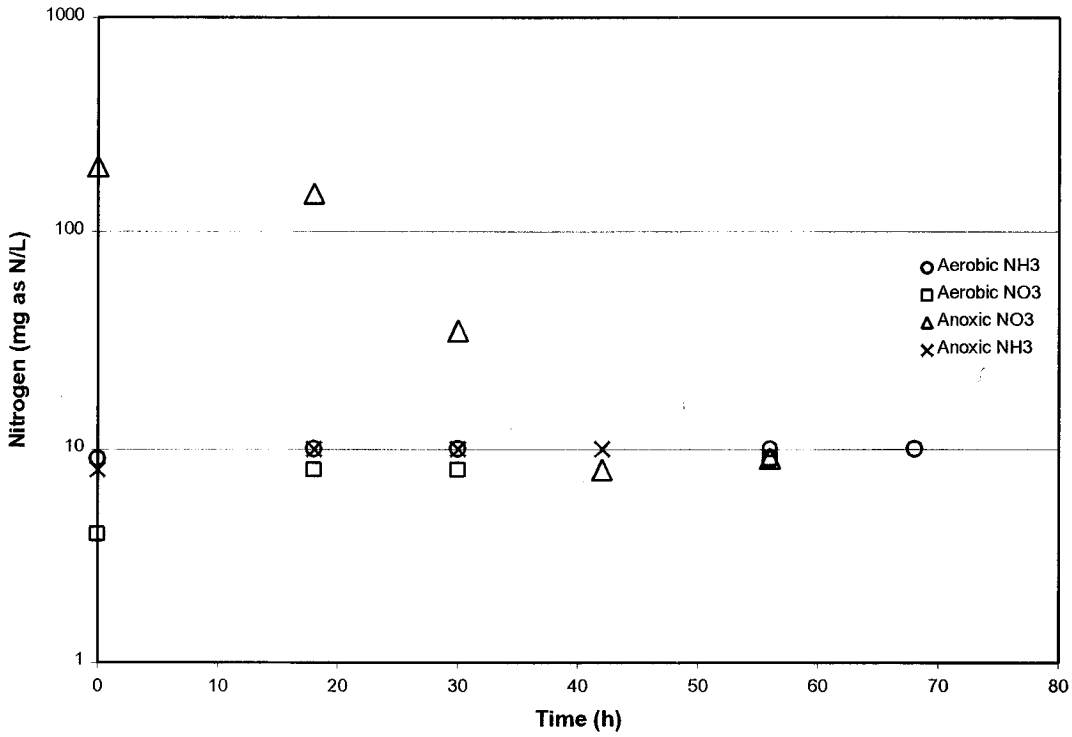


Figure 4.9 Nutrient transformations for run pair 5

4.1.3 SOLIDS

The variation of total solids (with error bars) and total volatile solids (without error bars) are shown in Figure 4.10. The solids tests were conducted in a different laboratory to the reactors as the building was under renovation. It is possible that airborne contamination in the form of construction dust changed the results of the solids tests. The error of the total solids tests was estimated to be 25%(see Appendix B). A small decrease (<2 mg/L/h) in total solids was observed for both the anoxic and aerobic reactors over the run pair and an increasing trend (1.8 mg/L/h) of volatile solids was also present. The magnitude of observed transformations was less than the error estimate. Although trends were observed, their magnitude was not significant.

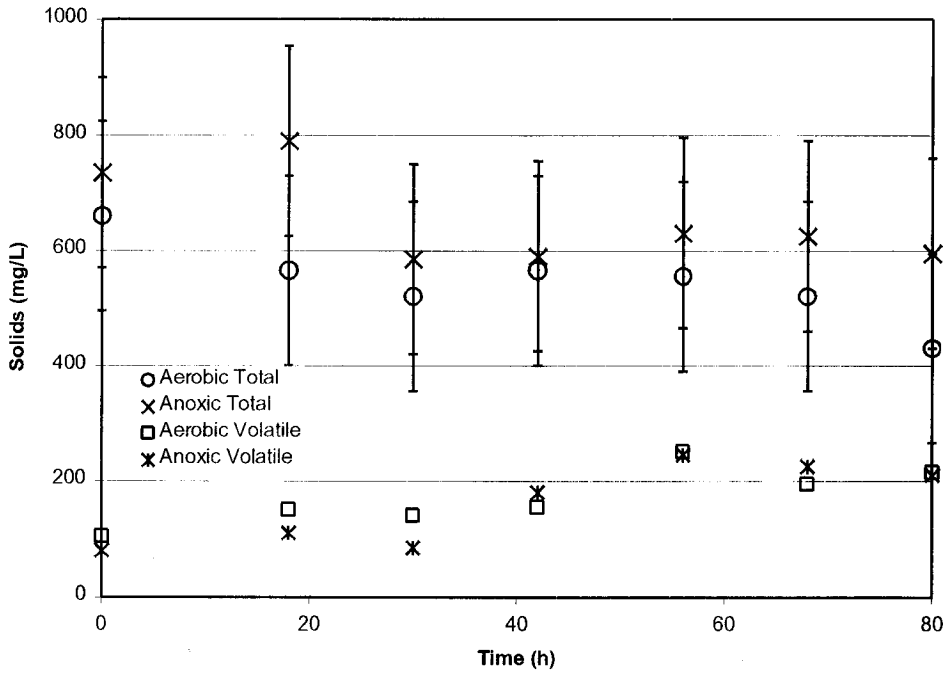


Figure 4.10 Variations of total and volatile solids for run pair 5

The dissolved solids variation over the course of the run pair is presented in Figure 4.11. Total solids are presented with error bars and volatile solids are presented without error bars. A small decrease (<2 mg/L/h) in total dissolved solids and a small increase (1.8 mg/L/h) in volatile dissolved solids were observed. Volatile dissolved solids increased approximately 100 mg/L for both the aerobic and anoxic cases over the course of the experiment.

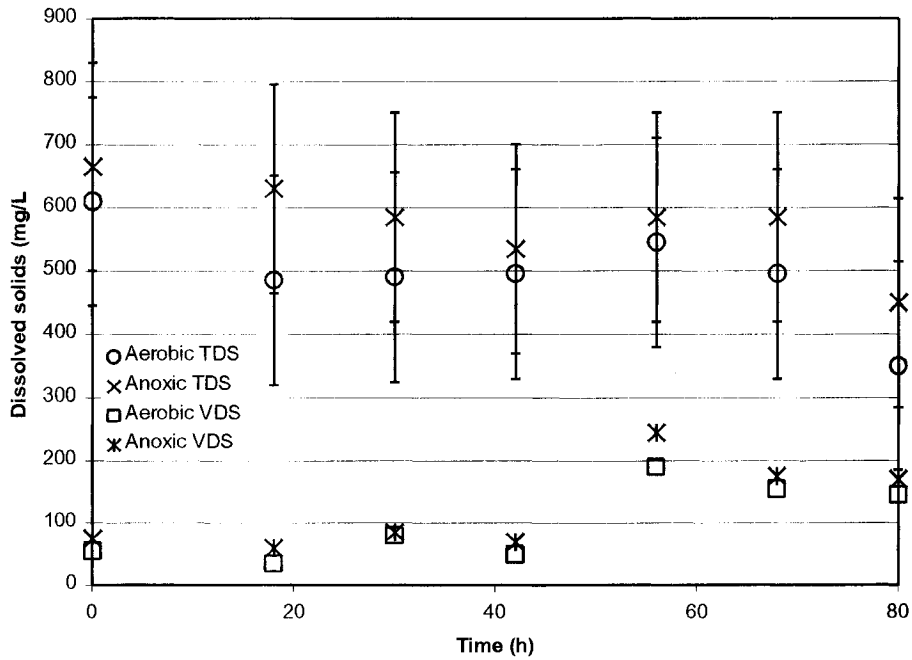


Figure 4.11 Variations of dissolved solids for aerobic/anoxic run pair 5

The large error bars (representing the mean difference between replicated subsamples) indicate that the magnitude of observed dissolved solids transformations were not significant. Both total and dissolved portions showed an insignificant decrease in total solids and an insignificant increase in volatile solids.

The total (TS) biofilm growth was 110 mg/L for the anoxic case and 275 mg/L for the aerobic case. This difference is the same size as the error estimate for the solids tests. The VFS increased 70 mg/L for the anoxic case and 105 mg/L for the aerobic case. The TVS increase of the bulk phase was estimated as 100 mg TVS/L. The overall (bulk TVS and biofilm VFS) microbial growth was 170 mg TVS/L for the anoxic case and 205 mg TVS/L for the aerobic case. Given that approximately 200 mg/L of COD_t was consumed, the mean yield coefficient was estimated at 0.9 mg TVS generated/mg COD_t consumed.

It is believed that solids test integrity was disrupted by the building renovation. Dust permeated the building causing an increased fixed solids content and variation. For this reason no responses were calculated from the solids data.

4.1.6 PARTICLE SIZE DISTRIBUTION

Particle size distribution data may be represented as volume in size range (percent of total particle volume) or volume to pass a size (percent of total particle volume). The influent to effluent change in size distribution from for the aerobic and anoxic run pair 5 may be seen as percent total particle volume in size range in Figure 4.12.

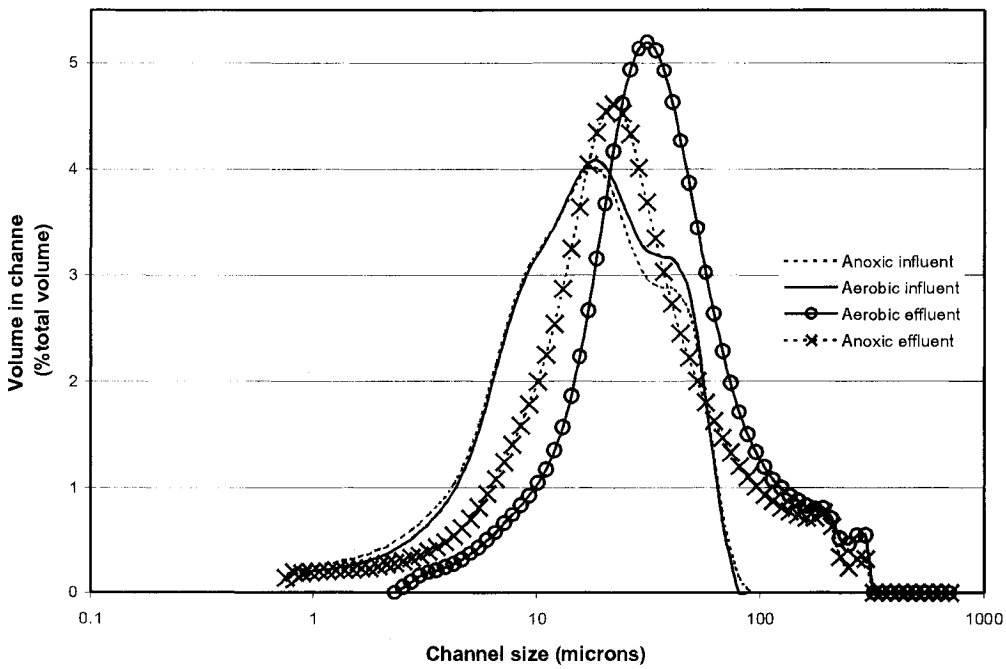


Figure 4.12 Volume in channel for influent and effluent of run pair 5

Influent particle size distribution was similar for the aerobic and anoxic reactors. The size distribution curve shifted to the right indicating an uptake of small particles for both reactors. The aerobic reactor size distribution curve shifted farther to the right indicating that a greater net consumption of small particles occurred in the aerobic reactor. The aerobic effluent distribution curve appears more regular than the anoxic effluent distribution curve. Larger particles seem to have formed in the aerobic environment than the anoxic environment.

Each size distribution curve may be represented by characteristic mean diameters related to volume (D_v), area (D_a), or number (D_n). Characteristic mean particle diameters for aerobic and anoxic run pair 5 are shown in Figure 4.13.

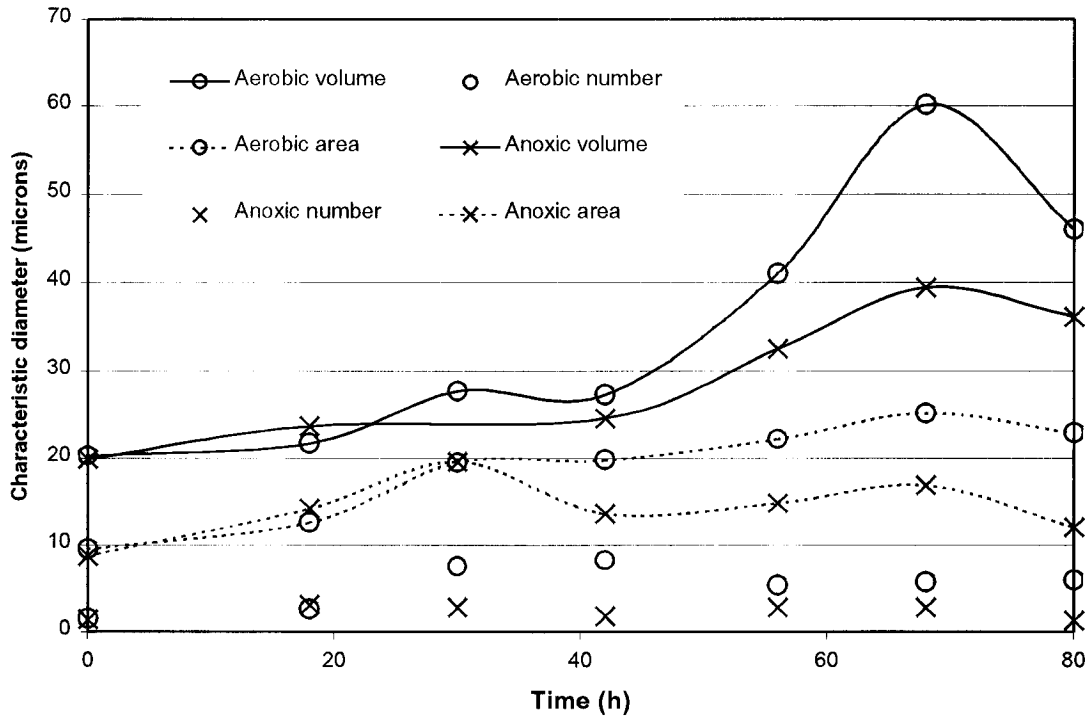
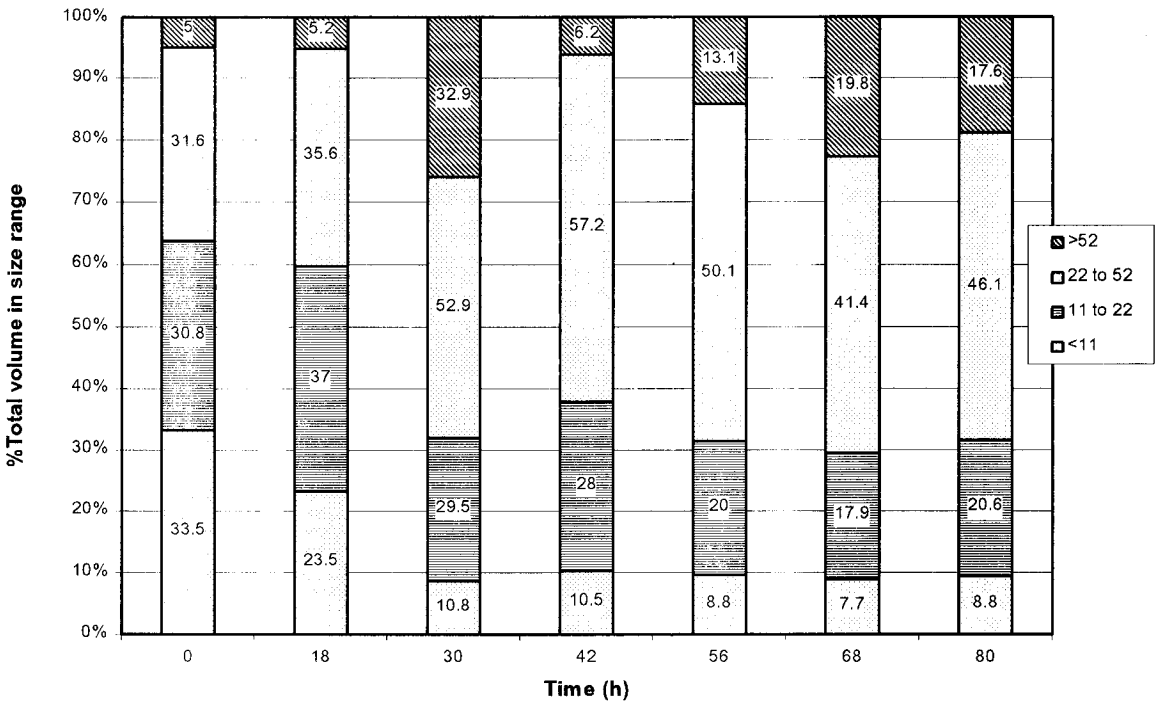


Figure 4.13 Characteristic mean diameters versus time for run pair 5

Figure 4.13 shows the number, area and volume characteristic mean diameters over the course of run pair 5. The D_v was fairly constant in the initial 40 h for both cases then increased, a sharper increase was observed for the aerobic reactor. The D_n was unchanged for the anoxic case but for the aerobic case reached a maximum at 40 h then declined as larger particles are formed. The D_a increased initially for both cases but showed a distinct difference after 30 h between the aerobic and anoxic cases (the aerobic continues increasing while the anoxic decreased to a stable value). Although similar organic matter consumption occurred in both reactors, D_a behaved differently following the consumption of readily consumed substrate. As particle size is dependant upon microbial consortia quality, this is an indication that the reactive environment influences the quality of the microbial consortia, the anoxic environment appears less able to form large stable flocs. The change in D_a from influent to effluent was selected as a representative response as particle breakdown is considered as a surface area dominated mechanism related to the activity of exopolymeric enzymes (Nielsen et al., 1992).

To study the mechanisms involved, particle size distribution was evaluated by the percent of total volume within four size ranges. Size ranges were selected to represent fine particles (<11 μm), flocs (11 to 22 μm), macroflocs (22 to 52 μm), and gross particles (>52 μm). The percent volume in size range for the aerobic case is shown in Figure 4.14.

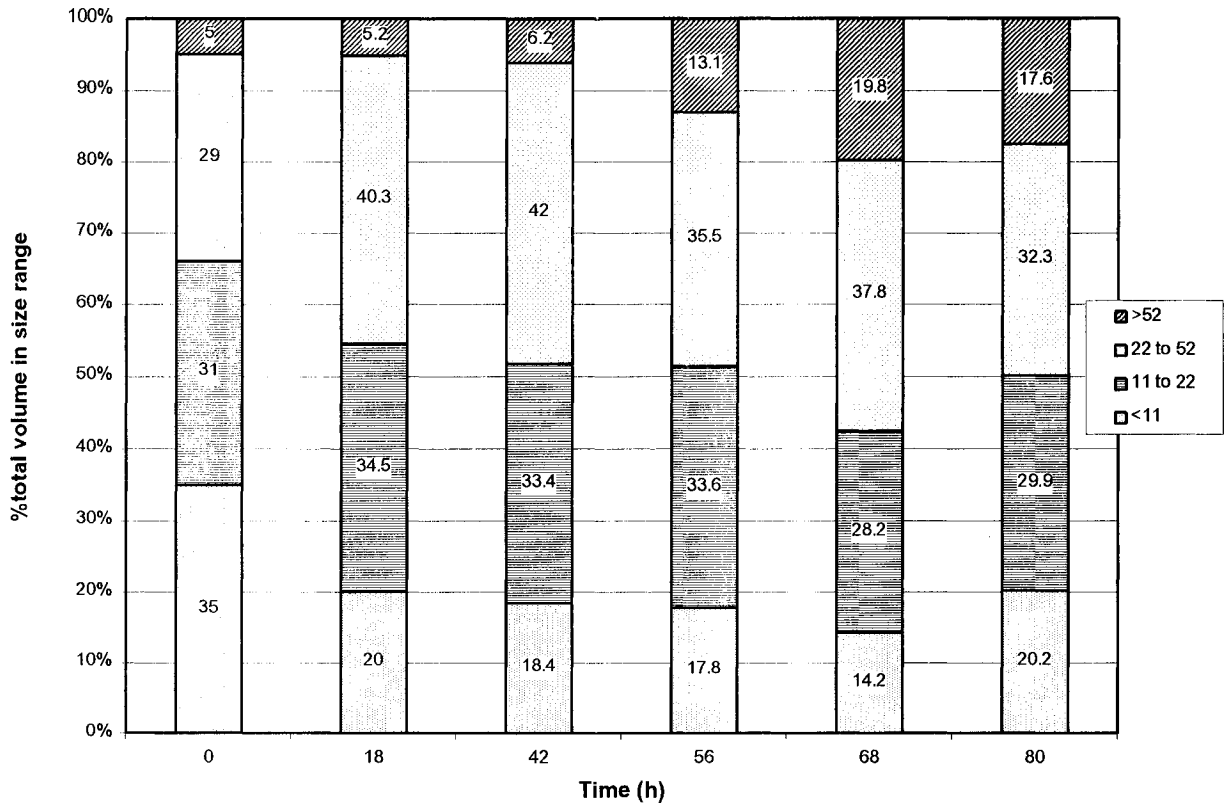


Legend entries refer to size in microns.

Figure 4.14 The percent total volume in size range for aerobic case 5O

Particles smaller than 11 μm showed a rapid initial net uptake for 30 h. These fine particles likely formed flocs as the 11 to 22 μm size range reaches a maximum volume at 18 h. These flocs then appeared to form into macroflocs as the 22 to 52 μm size range reached a maximum at 42 h. Following 42 h these macroflocs appeared to form into gross particles. Overall, a steady flocculating mechanism was dominant for the aerobic case.

The percent of particle volume in size ranges for the anoxic case is shown in Figure 4.15.



Legend entries refer to size in microns.

Figure 4.15 The percent of total particle volume in size range for anoxic case 5X

The percent volume of fine particles (smaller than 11 μm) decreased initially for 18 h, similar to the aerobic case, then did not change significantly for the remainder of the experiment. The percent volume of flocs (11 to 22 μm) showed a small net decrease over the experiment. The macroflocs (22 to 52 μm) increased to a maximum from 18 to 42 h and then decreased. The percent total volume of gross particles (larger than 52 μm) remained steady until 42 h then increased. Initially, until T_r, the anoxic case shows a net uptake of fine particles similar to the aerobic case. Following T_r, the anoxic case showed less change in particle size distribution than the aerobic case although a similar amount of organic matter was consumed. This could indicate that the anoxic microbial consortia formed flocs less resistant to shear stress than the aerobic microbial consortia.

The size distribution change as percent volume in each size range, from influent to T_r and from T_r to effluent were, selected as responses.

4.1.7 RESPONSE VARIABLES ANALYSED

Responses were selected to characterize the change in organic matter and particle size distribution over the course of the experiment. The responses selected relating to amount of organic matter transformations were the COD_t removal (as mg/L and percent of initial COD_t), the maximum COD_t consumption rate (mg/L/h), the maximum COD_s consumption rate (mg/L/h), and the mean COD_p consumption rate (mg/L/h). The responses selected relating to the time, in hours, of organic matter transformations were the initial lag time (T_l), the time of maximum COD_t consumption rate (T_m), the time to consume readily degraded substrate (T_r), and the stabilization time (T_s). The time from T_l to T_m was considered biomass limited, the time from T_m to T_r, substrate limited, and the time from T_r to T_s hydrolysis or endogenous decay limited. The responses selected concerning particle size distribution transformations were mean area diameter (D_a) and percent volume in size range. D_a was evaluated only from influent to effluent while the percent total particle volume in size range was evaluated from influent to T_r, and from T_r to effluent.

4.2 SYSTEMATIC ERRORS

Error sources may be random or systematic. Random errors should be normally dispersed about the mean response but systematic errors may bias the observed results and lead to incorrect observations or conclusions. Two possible sources of systematic error have been identified. The source wastewater changed over the duration of the investigation due to aging and the anoxic and aerobic reactors were filled sequentially possibly leading to differing initial contents. As complex biological systems do not lend

themselves to rigorous statistical analysis, conditions were considered significant if the difference between the responses was greater than the standard deviation of all measured responses.

4.2.1 AGING

Some change of source wastewater quality occurred between the beginning and end of the experiment as wastewater will go septic even when stored at 4° C. Run pair 10 was conducted with septic source wastewater. The experiment was concluded after run pair 10 as the source wastewater had turned septic. In this experiment systematic error due to aging is evaluated by monitoring the change of influent organic content (as COD_t and particle size distribution) and as the change in responses observed between the 1st half and the 2nd half of the investigation.

The influent COD_t varied over the course of the experiment due to aging and supplement addition. The mean influent COD_t for all 20 runs (with substrate and biomass supplements) was 290 mg/L with a standard deviation of 115 mg/L. The run pair mean influent organic content with supplement additions (as COD_t) over the duration of the experiment is shown in Figure 4.16.

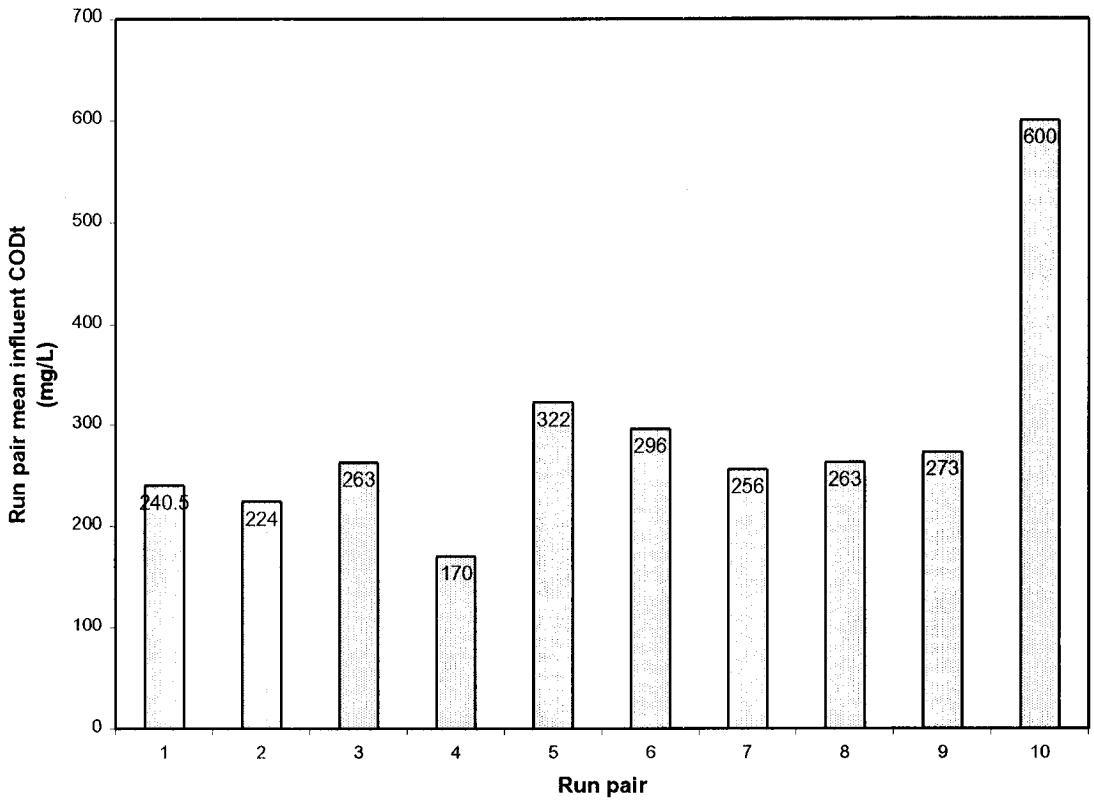


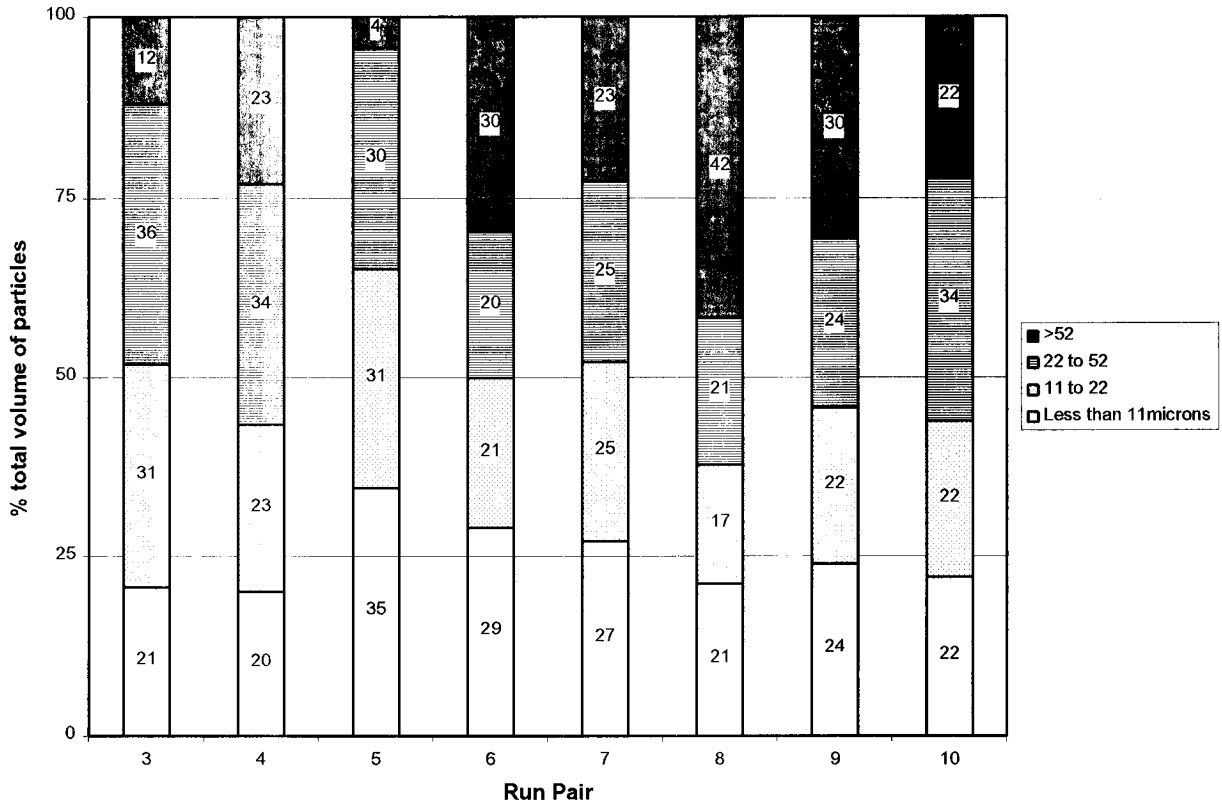
Figure 4.16 Run pair mean influent organic content (as COD_t)

Other than run pair 10, which was septic and contained a larger COD_t, the influent organic content did not appear to systematically change over the duration of the investigation as demonstrated by Figure 4.16.

Even though run pair 10 was conducted with septic wastewater that appeared black and odiferous, the measured responses showed similar behaviour to runs conducted with non-septic wastewater. The aerobic reactor changed the colour of the wastewater to brown within 4 h while the anoxic reactor took 12 h to change the colour of the wastewater. As septic wastewater is a possible influent and as the responses were within the range of observed responses, the results for run pair 10 were included in the conditional analysis.

The influent particle size distribution varied over the course of the experiment due to settling of the source wastewater in the storage containers. The mean influent particle size distribution for all runs was 25% of total volume smaller than 11 μm (s.d.=8 percent total volume), 24% of total volume between 11 and 22 μm (s.d.=5 percent total volume),

28% of total volume between 22 and 52 μm (s.d.=7 percent total volume), and 23% of total volume greater than 52 μm (s.d.=11 percent total volume). The run pair mean particle size distribution (as percent total volume in size range) over the duration of the investigation is shown in Figure 4.17.



Legend refer to size in microns.

Figure 4.17 Run pair mean influent particle size distribution (as percent of total volume in size range)

Although there was considerable variation over the course of the experimental investigation, no systematic change in particle size distribution can be observed in Figure 4.17. The largest variation was in the largest size range (particles greater than 52 μm), which ranged from 4 to 42% of the total volume.

The change in observed results over the course of the experiment was evaluated by comparing the results from the initial 5 run pairs to the results from the final 5 run pairs (see Table 4.2).

Table 4.2 Comparison of results between initial 5 run pairs and final 5 run pairs

Response	Units	Experimental mean	Standard deviation	Initial 5 run pairs	Final 5 run pairs
Organic content					
TE	%COD_t(inf)	55.6	14.9	60.7	50.4
CODR	mg/L	166	75	153	180
MxT	mg/L/h	8.4	5.4	7.5	9.0
MxS	mg/L/h	4.9	3.1	4.7	5.0
MnP	mg/L/h	1.1	1.4	1.2	1.1
Time					
T_l	h	4	11	7	0
T_m	h	25	16.2	25.5	24.5
T_r	h	41.8	13.3	40.2	43.4
T_s	h	74	15.3	70.2	77.8
Particle size distribution					
Inf to eff					
D_a	μm	2.7	9.0	2.4	2.8
Inf to T_r					
<11 μm	% total vol.	6.6	9.3	7.4	6.3
11 to 22 μm	% total vol.	1.2	7.1	-0.8	0.6
22 to 52 μm	% total vol.	-9.9	9.9	-10.6	-9.6
T_r to eff					
<11 μm	% total vol.	1.4	6.8	-0.2	2.1
11 to 22 μm	% total vol.	0.1	5.6	3.0	-1.2
22 to 52 μm	% total vol.	4.5	6.5	4.8	4.4

The observed responses did not differ significantly between the initial 5 run pairs and the final 5 run pairs. The COD_t reduction was 60% for the initial 5 run pairs and 50% for the final 5 run pairs but this difference (10%) was still less than the standard deviation of all results (15%). Aging of the influent wastewater may have reduced the treatable fraction of the wastewater but this difference is not considered significant. The difference in time responses (T_l to T_m, T_m to T_r, and T_r to T_s) all indicate that the influent became less readily degradable over the course of the experimental investigation.

Although the influent conditions and observed responses varied over the duration of the experimental investigation, systematic error due to aging is believed to not bias this investigation.

4.2.2 INITIAL CONTENTS

A systematic error may have occurred as the aerobic and anoxic reactors were filled sequentially from a mixed vessel. Differing initial reactor composition would bias the comparison between the aerobic and anoxic reactive environments. The difference between the aerobic and anoxic initial reactor organic content (as measured by COD_t and COD_s) is shown in Figure 4.18. No bar indicates no difference in initial reactor contents.

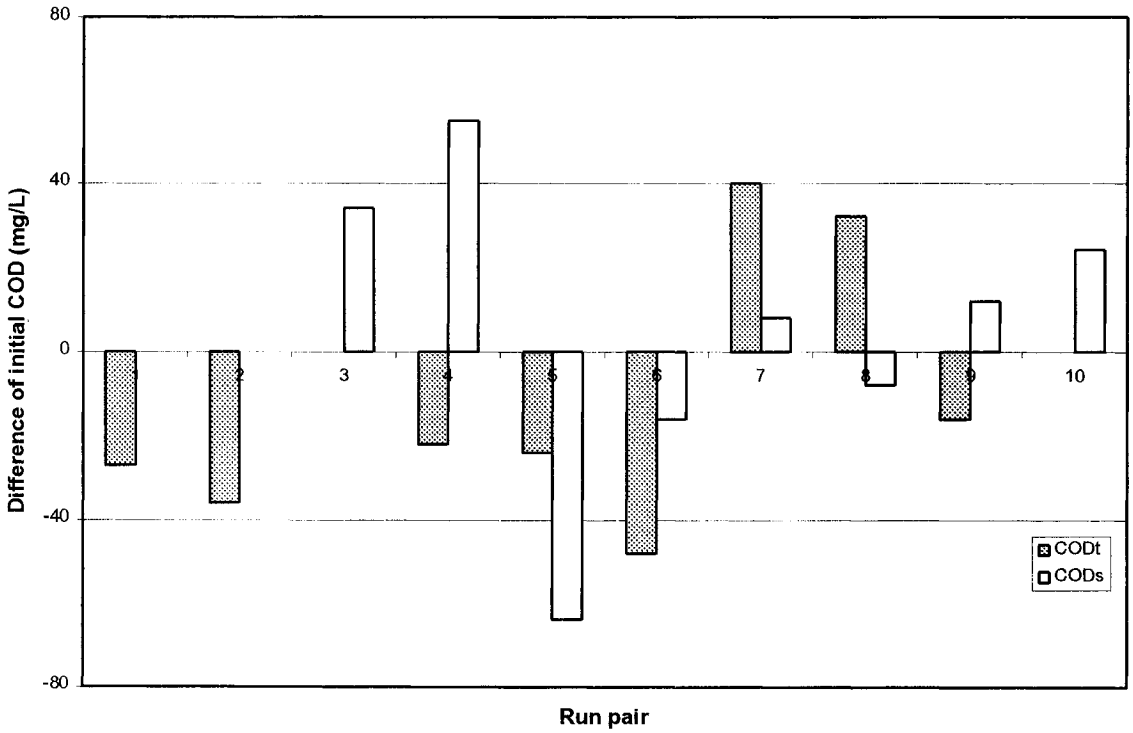


Figure 4.18 Difference (aerobic-anoxic) of initial reactor organic content

The anoxic reactor was normally filled before the aerobic reactor causing the anoxic reactor to be on average 10 mg/L stronger in COD_t and 5 mg/L weaker in COD_s. This mean reactor difference was small and is not considered significant. Run pairs 4 and 5 had differing COD_s concentrations and run pair 6 had a differing COD_t concentration.

The small mean difference between initial reactor contents and their distribution on either side of the 'no difference' line indicates that no systematic error in reactor organic content arose from the reactor filling procedure.

The difference between initial (post supplement addition) aerobic and anoxic reactor particle size distribution (as percent of total volume in size range) is shown in Figure 4.19.

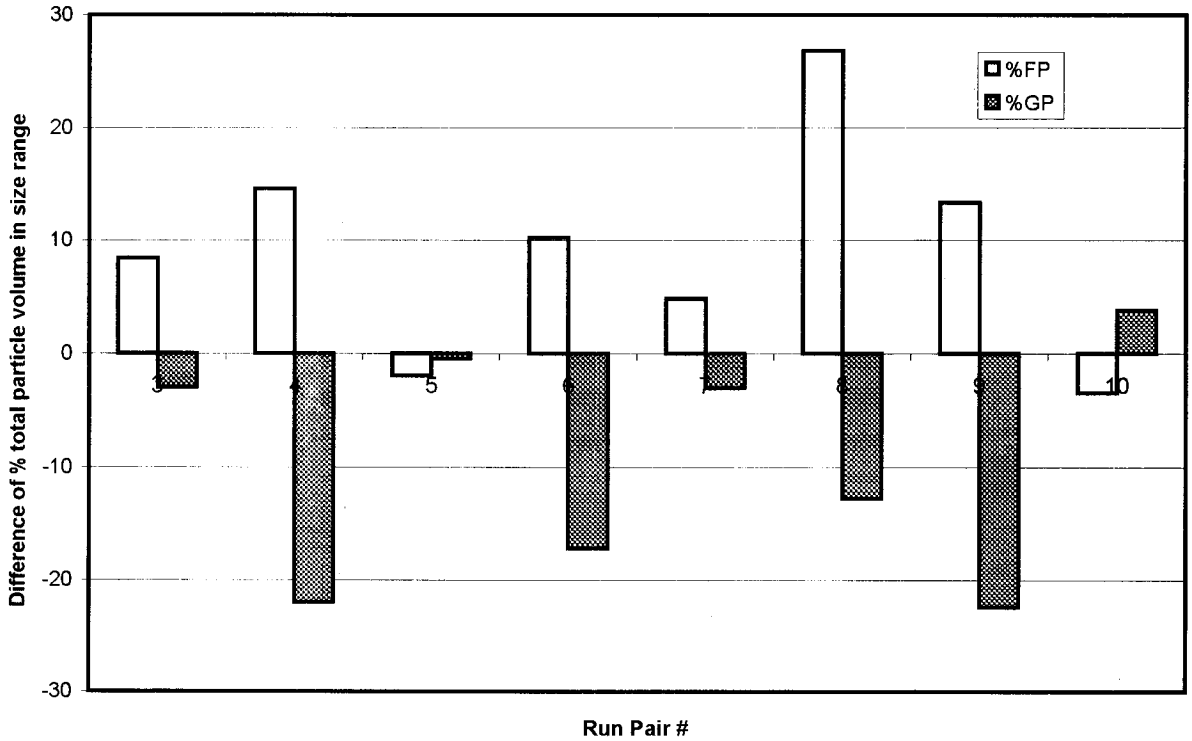


Figure 4.19 Difference between aerobic and anoxic initial reactor particle size distribution

It appears that the initial particle size distribution varied between the reactors in four of the eight run pairs of the experiment. The aerobic reactor had a larger fraction of fine particles (<11 μm) in five of the eight experimental run pairs and the anoxic reactor had a larger fraction of gross particles (>52 μm) in four of the eight experimental run

pairs. The aerobic reactor had a mean 10% total volume more %FP (< 11 μm) and the anoxic reactor had a mean of 9% more %GP (> 52 μm). This systematic error must be considered when evaluating overall particle size distribution transformations.

4.3 SUMMARY OF OVERALL RESULTS

The overall behaviour of ROPEC influent in pipeline reactors was estimated using the mean response of all 20 experimental runs. The responses for each experimental run and for each condition analyzed may be described by organic content responses (Table 4.3), time responses (Table 4.4), and particle size distribution responses (Table 4.5). Overall organic content (measured as COD_t) decreased by an average of 167 mg/L (or 55% (s.d.=15%) of the initial COD_t) over 72 h. The decrease occurred in the following manner.

Initially a lag (mean T_l = 4 h, s.d=11 h) was observed in which the micro organisms acclimatized themselves to their environment. The duration of this lag depended upon the initial quality of the wastewater as appropriate microorganisms must be present in order to consume the available substrate. The initial quality of the microbial consortia and available substrate limit the growth of the microorganisms. The COD_t consumption rate increased until some limiting microbial concentration was reached (T_m = mean 24 h, s.d=16 h). The availability of readily consumable substrate then limits the consumption of COD_t (mean maximum COD_t consumption rate = 8.4 mg/L/h, s.d.=5.5 mg/L/h). The substrate being readily consumed is primarily soluble COD (mean maximum COD_s consumption rate = 5.0 mg/L/h, s.d.=3.1 mg/L/h). As the readily degraded substrate was consumed, endogenous decay and particle hydrolysis made up an increasing part of the COD_t decrease. The readily consumable substrate was consumed (T_r = mean 41 h, s.d.=13 h) after a mean of 17 h following the maximum observed COD_t consumption rate. Endogenous decay and particulate hydrolysis continued at a mean COD_p consumption rate of 1.1 mg/L/h. No further decrease in COD_t was observed after a mean experiment time of (T_s) 75 hours (s.d.=15 h).

Table 4.3 Organic content responses

Run Id	Treatment efficiency %COD_t(inf)	COD_t removed (mg/L)	Max COD_t rate (mg/L/h)	Max COD_s rate (mg/L/h)	Mean COD_p rate (mg/L/h)
1O	66	149	5.3	2.3	1.9
1X	72	184	6.1	1.8	3.0
2O	19	35	2.2	2.3	0.4
2X	64	142	12.2		
3O	69	180	7.8	5.4	1.3
3X	79	211	7.4	5.5	1.1
4O	55	87	3.6	3.8	-0.1
4X	47	85	6.2	1.1	1.2
5O	74	224	8.5	8.7	1.1
5X	63	232	15.7	11.7	0.4
6O	57	156	4.8	5.2	0.3
6X	56	180	7.7	4.4	0.4
7O	49	148	5.7	1.7	1.4
7X	28	64	2.6	3.6	-1.2
8O	64	176	4.4	2.7	1.2
8X	59	156	6.2	4.0	0.9
9O	42	136	8.7	7.7	-1.3
9X	46	128	8.0	8.3	1.0
10O	50	332	18.2	2.4	4.4
10X	53	320	23.6	10.2	3.6
mean	55	167	8.4	5.0	1.1
s.d.	15	77	5.5	3.1	1.4
aerobic	54	162	7	4	1
anoxic	57	170	10	6	1
seed	58	194	11	6	1
no seed	53	139	6	3	1
sludge	64	170	8	6	1
biofilm	48	229	15	7	2
no substrate	56	134	7	4	1
substrate	55	134	10	5	1

Table 4.4 Time responses

Run Id	Lag time T_l (h)	Max rate time T_m (h)	Readily consume time T_r (h)	Stabilization time T_s (h)
1O	0	46.5	58	58
1X	0	11	34.5	46
2O	47	47	59	71
2X	11.5	24	33	59
3O	0	0	21	89
3X	0	8	32	89
4O	5.5	41	52	65
4X	5.5	41	52	65
5O	0	18	30	80
5X	0	18	30	80
6O	0	29	53	89
6X	0	53	65	89
7O	0	17	52	64.5
7X	0	0	29	52.5
8O	0	41	52	89
8X	0	41	52	89
9O	0	15.5	40	90
9X	0	15.5	40	90
10O	0	16.5	25.5	74.5
10X	0	16.5	25.5	50.5
mean	4	24	41	75
s.d.	11	16	13	15
aerobic	5	27	44	77
anoxic	2	23	39	71
seed	1	19	35	77
no seed	6	31	49	71
sludge	2	21	36	78
biofilm	0	16	33	76
substrate	0	25	41	76
no substrate	7	25	42	72

Table 4.5 Particle size distribution responses

	Influent to effluent	Influent to T_r (decrease of percent particle volume in range)				T_r to effluent (decrease of percent particle volume in range)			
Run Id	Area diameter Da (μm)	<11 μm	11 to 22 μm	22 to 52 μm	>52 μm	<11 μm	11 to 22 μm	22 to 52 μm	>52 μm
3O	3.7	11.3	6.4	-0.8	-16.9	-5.2	-4.6	-12.4	22.2
3X	-0.5	-10.3	5.9	17.0	-12.6	3.7	-2.9	-12.3	11.5
4O	2.8	7.4	-2.3	-10.1	-5.0	3.0	6.0	6.7	-15.7
4X	-7.5	-13.6	-10.0	-9.1	32.7	1.2	-3.4	-0.4	2.6
5O	13.0	18.0	6.0	-23.6	-0.4	2.3	9.0	13.5	-24.8
5X	3.0	9.4	-0.9	-7.2	-1.3	1.2	4.4	6.9	-12.5
6O	15.5	16.9	6.1	-21.5	-1.5	3.1	5.2	3.9	-12.2
6X	-21.9	8.3	2.6	-18.0	7.1	0.4	-0.1	1.9	-2.2
7O	6.2	11.1	2.4	-16.9	3.4	-0.7	3.7	4.5	-7.5
7X	7.2	8.3	5.0	-15.3	2.0	0.3	0.3	0.5	-1.1
8O	13.9	18.6	10.4	-11.7	-17.3	-0.5	0.9	8.0	-8.4
8X	0.0	-0.4	-2.1	-10.6	13.1	-1.5	-4.0	-4.0	9.5
9O	6.3	10.0	-1.1	-2.3	-6.6	7.0	-7.8	17.2	-16.4
9X	-1.9	-0.4	-4.6	-3.6	8.6	-1.7	-1.3	0.6	2.4
10O	4.1	7.1	-14.4	7.4	-0.1	7.0	-7.8	17.2	-16.4
10X	-1.2	-1.2	-3.6	-2.7	7.5	-5.8	3.3	7.1	-4.6
mean	2.7	6.3	0.4	-8.1	1.4	0.9	0.1	3.7	-4.7
s.d.	9.0	9.3	6.6	10.6	11.6	3.6	5.0	8.7	12.1
aerobic	8	13	2	-10	-5	2	1	7	-10
anoxic	-3	0	-1	-6	7	0	0	0	0
seed	2	4	-2	-4	2	1	-1	4	-4
no seed	3	10	4	-16	2	0	1	2	-3
sludge	2	4	1	-6	1	1	1	0	-2
biofilm	2	4	-6	0	2	2	-3	11	-10
substrate	4	10	1	-12	1	1	2	6	-9
no substrate	0	1	-1	-1	1	1	-2	0	1

Table 4.6 Supplement Additions

Run Pair	Microorganism supplement	Substrate supplement
1	None	None
2	None	None
3	10 VSS mg/L AS	None
4	10 VSS mg/L AS	None
5	10 VSS mg/L AS	100 mg/L soluble
6	None	100 mg/L soluble
7	None	100 mg/L soluble 100 mg/L particulate
8	None	100 mg/L soluble 100 mg/L particulate
9	3 mg/L VFS BF	None
10	3 mg/L VFS BF	200 mg/L particulate

4.3.1 CONDITIONAL ANALYSIS OF OVERALL RESULTS

Evaluating these results by condition can provide a greater understanding of pipeline reactor behaviour. Three operating conditions, reactive environment, initial microorganism composition, and initial substrate composition were manipulated. The conditional responses may be seen in Table 4.3, Table 4.4, and Table 4.5.

4.3.2 REACTIVE ENVIRONMENT

The choice of an aerobic or anoxic reactive environment did not influence the overall organic removal (Table 4.3) or time of organic removal (Table 4.4) provided sufficient electron acceptors were present to not limit the microbial growth. The choice of aerobic or anoxic conditions did influence the observed particle size distribution transformations (Table 4.5). The mean area diameter increased by an average of 8 μm for the aerobic reactor and decreased by an average of 3 μm for the anoxic reactor. This indicates that larger flocs formed under aerobic conditions than anoxic conditions. Overall the aerobic reactive environment consumed fine particles and flocs smaller than 22 μm to form macroflocs and gross particles larger than 22 μm while the anoxic reactive environment consumed gross particles larger than 52 μm to form flocs and macroflocs between 11 and 52 μm in size. The aerobic environment formed larger flocs than the anoxic environment.

The aerobic reactive environment initially (from the beginning of the experiment until T_r) consumed fine particles smaller than 11 μm to form flocs primarily into the 22 to 52 μm size range (see Table 4.5). After T_r, the smaller flocs continued to grow as particles smaller than 52 μm were transformed into gross particles larger than 52 μm. These results suggest that steady floc growth occurred in the aerobic environment.

The anoxic reactive environment initially (from the beginning of the experiment to T_r) generated particles in the 11 to 52 μm size range from gross particles larger than 52 μm (see Table 4.5). No change in particle size distribution was observed after the readily degraded substrate was consumed. These results suggest that the initial transformation was large particles breaking up into smaller discrete particles due to endogenous decay, hydrolysis, or shear stress. Following T_r, the anoxic microorganisms appeared to reach

some optimum spatial arrangement that maximized the floc surface area to mass ratio. Anoxic conditions produced smaller flocs than aerobic conditions that appeared less resistant to shear stress.

Particle size analysis must consider the systematic error induced by the sequential reactor filling. This was evaluated by comparison of the influent to effluent change in D_a for two replicated sets of run pairs (3,4 and 7,8, see Table 4.6). Comparison of the aerobic and anoxic runs of run pairs 4 and 8 (skewed influent) to the aerobic and anoxic runs of run pairs 3 and 7 (non-skewed influent) resulted in a 't' number of 0.39. Comparison of all aerobic runs to all anoxic runs of run pairs 3,4,7 and 8 resulted in a 't' number of 1.75. This larger 't' number indicates that the reactive environment more probably caused the difference in D_a response than the skewed influent particle size distribution. A 't' number of 1.75 indicates a confidence interval of approximately 80% for 6 degrees of freedom (Miller and Miller, 1993).

Table 4.7 Experimental design considering skewed influent size distribution

Run pair ID	Substrate addition	Microorganism addition	Skewed influent
1	—	—	
2	—	—	
3	—	Activated sludge	0
4	—	Activated sludge	1
5	Soluble	Activated sludge	0
6	Soluble	—	1
7	Soluble and particulate	—	0
8	Soluble and particulate	—	1
9	—	Biofilm	1
10	Particulate	Biofilm	0

4.3.3 BIOMASS ADDITION

Enhanced initial biomass due to reactor seeding increased the mean maximum rate of COD_t consumption from 6 to 11 mg/L/h and the mean maximum rate of COD_s consumption from 3 to 6 mg/L/h (see Table 4.3). Reactor seeding reduced the initial lag from 6 to 1 h and the time of maximum COD_t consumption from 31 to 19 h (see Table 4.4). The biomass limited growth period, from the initial lag time to the time of maximum COD_t consumption rate, was shorter for seeded runs (18 vs 25 h) due to the higher quality of the initial biomass consortia. Seeding did not affect the length of time to consume readily degraded substrate after the maximum COD_t rate was reached, indicating that substrate availability was limiting microbial growth and resultant COD_t consumption. Seeding did affect the size distribution transformations. Initially (influent to T_r) particles were generated into the 22 to 52 μm size range from small particles (<11 μm) for unseeded runs (see Table 4.5). No significant size distribution change occurred following T_r for the unseeded runs and for all seeded runs.

The source of the microbial seed also influenced the observed transformations. Activated sludge seed had the largest mean COD_t reduction of 64% while biofilm seed or no seed reduced the COD_t by a mean of 51% (see Table 4.3). This increased COD_t reduction indicates that the activated sludge contained microorganisms that were not present in the sewer biofilm. Runs seeded with sewer biofilm had the fastest mean maximum COD_t consumption rate of 15 mg/L/h compared to 8 mg/L/h for runs seeded with activated sludge (see Table 4.3). The sole difference in size distribution transformations due to seed source was a tendency for runs seeded with biofilm to form macroflocs and gross particles larger than 22 μm following T_r (see Table 4.5). This increased activity (observed as maximum COD_t consumption rate) indicates that the sewer biofilm originated from a more synergistic microbial consortium than the activated sludge or that the sewer biofilm had a larger fraction of viable microorganisms. But fixed film biomass is normally considered less active than free floating biomass and a smaller amount of film biomass was added compared to activated sludge seed.

4.3.4 SUBSTRATE ADDITION

Initial substrate was manipulated to determine pipeline reactor system response to a wide range of influent wastewater conditions. Substrate addition did not affect overall organic removal or rate of organic removal (see Tables 4.3 and 4.4) although it did affect the particle size distribution transformations (see Table 4.5). Runs without substrate addition had no significant change in size distribution. Flocs formed initially (influent to T_r) in the 22 to 52 μm size range and finally larger than 52 μm for runs subject to substrate addition. This suggests that the availability of readily degraded substrate influenced microbial activity. The pipeline reactor system performance was consistent over a wide range of influent wastewater conditions.

4.4 DOWNSTREAM EFFECTS

In-sewer transformations can have a significant effect on wastewater quality that will affect downstream treatment units. Due diligence requires that these downstream effects be estimated prior to the application of pipeline reactor technology both to avoid disrupting existing treatment facilities and to properly estimate the actual cost of pipeline reactors. The primary goal of pipeline reactors is to reduce the organic loading to the treatment facility although other changes in wastewater quality may occur. The transformations observed in this experimental investigation were a reduction in overall organic content, a conversion of the organic content to particulate matter, and a differing particle size distribution dependant on the quality of electron acceptors available.

A reduction in overall organic content is the goal of pipeline reactor treatment. Reduced organic loading should reduce treatment facility costs and increase capacity by requiring less downstream treatment to achieve discharge or reuse regulations. The aeration cost of secondary treatment will be spread into the wastewater collection system and possibly reduced, as there would be no aeration mixing requirement for pipeline reactors. The necessity for biological treatment of wastewater at the treatment facility will be reduced. Gall et al. (1995) forecast an increased particulate fraction of wastewater organics due to microbial growth and assumed no change in settling

characteristics due to pipeline reactor treatment to estimate downstream effects to a secondary treatment plant.

It is believed that the wastewater had stabilized as BOD_5 tests on the final reactor contents of run pair 8 showed a mean BOD_5 of 29 mg/L, close to the discharge limit of 25 mg/L. The BOD_5/COD ratio changed from 0.35 to 0.23 over an experimental run. This result is similar to the phenomena observed by Green et al. (1985). Influent COD had a mean soluble fraction of 0.62 and effluent COD had a mean soluble fraction of 0.42 (see Figure 4.20). This result is similar to the increase of COD_p predicted by Gall et al. (1995).

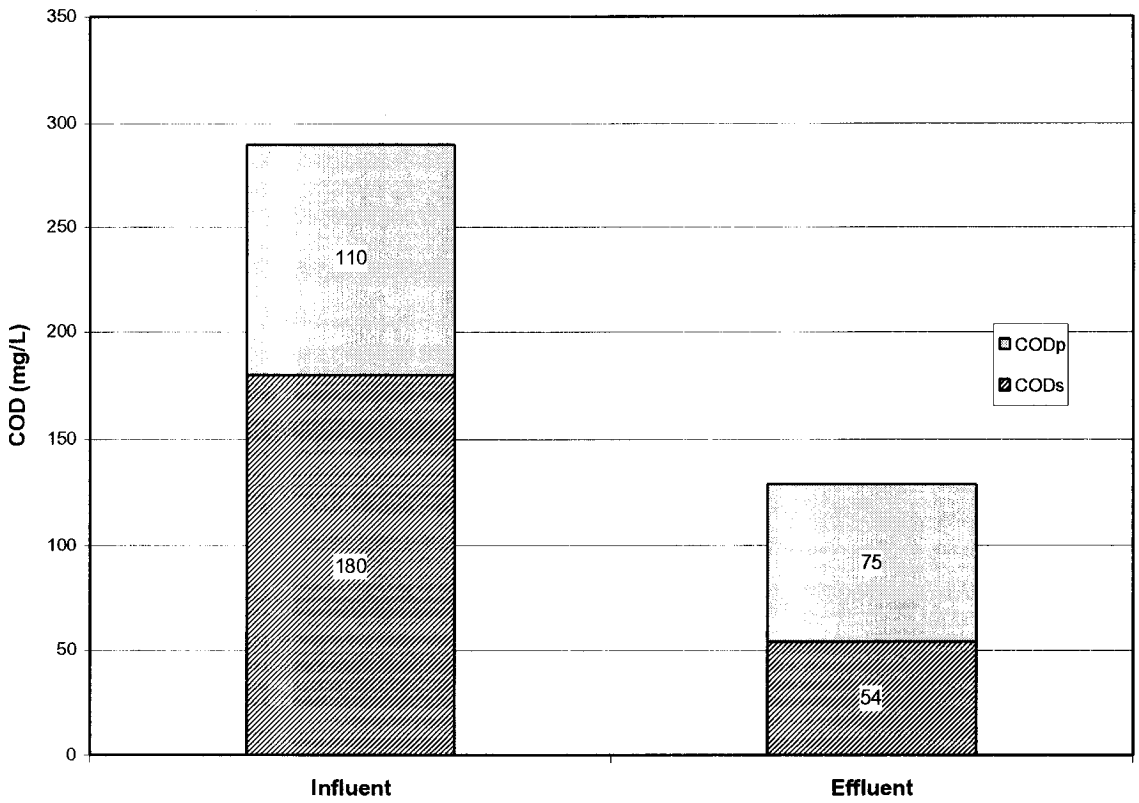


Figure 4.20 Mean organic strength and content for experimental influent and effluent

Gall et al. (1995) assumed no change in settling characteristics would occur due to pipeline reactor treatment. The settling characteristics did not appear to change significantly due to pipeline reactor treatment for either the aerobic or the anoxic reactive environments (see Figure 4.21). The effluent from the anoxic reactor generated 12 mL

sludge, slightly more sludge than the effluent from the aerobic reactor (10 mL sludge) but this difference was less than the error estimate for this test (±4 mL).

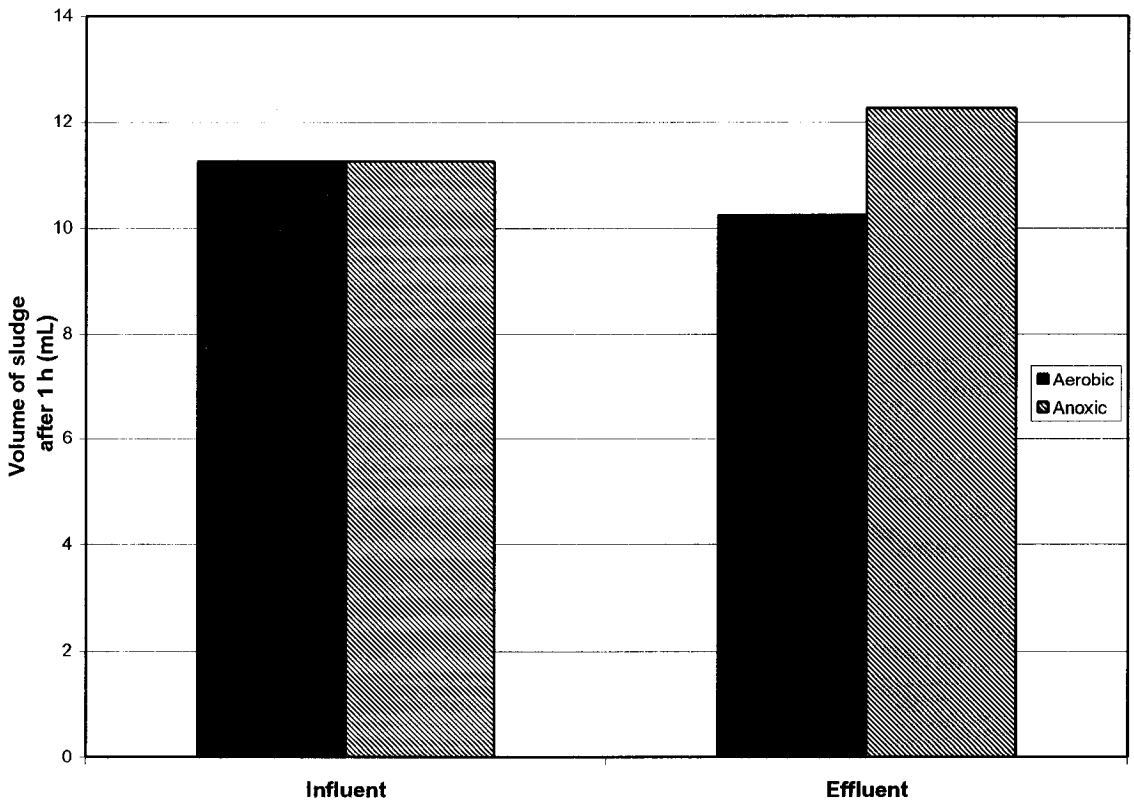


Figure 4.21 Settling test for influent and effluent of an aerobic/anoxic run pair 7

The increased particulate fraction of treatment plant influent should result in an increase of primary sludge (Gall et al., 1995). Enhanced primary treatment (removal of particulate matter) and a lower soluble organic load should reduce the secondary treatment requirements.

The size distribution of the treatment plant influent wastewater particulate fraction will also be effected by pipeline reactor treatment. An aerobic reactive environment led to the generation of larger flocs than an anoxic reactive environment. Larger floc size is believed to preferential for units involving filtration and floatation while smaller floc size is believed to be preferential for sedimentation and de-watering processes (Gregory, 1997). The settling test data were inconclusive to confirm or refute this statement. Theoretically smaller flocs would be preferred as ROPEC employs primary settling and sludge dewatering. This implies the use of anoxic pipeline treatment to increase primary

sludge production. The reduced BOD and increased particulate fraction of the organics indicate that primary treatment may accomplish all required treatment if sufficient pipeline reactor HRT is available.

4.5 ESTIMATION OF BIOMASS GROWTH AND HYDROLYSIS RATES

Biomass growth and particulate hydrolysis rates for this experimental investigation were estimated for 3 assumed yield factors, using data from the influent to T_r as endogenous decay can be considered negligible as long as readily degraded substrate is present. The data and method used to calculate these rates may be seen in Appendix C. The results may be seen in Table 4.7. These rates are in the range predicted by Almieda et al. (1999) and Hvitved-Jacobsen et al. (1998). The aerobic biomass growth and particulate hydrolysis rates are approximately 1.3 times greater than the anoxic rates.

Table 4.8 Biomass growth and Hydrolysis rates

	Aerobic		Anoxic	
Yield (mg biomass/ mg substrate)	Biomass growth rate (mg/L/h)	Particulate hydrolysis rate (mg/L/h)	Biomass growth rate (mg/L/h)	Particulate hydrolysis rate (mg/L/h)
0.6	3.5	3.0	2.6	2.4
0.55	2.8	2.4	2.1	1.9
0.5	2.3	1.9	1.7	1.5

4.6 DESIGN STUDY

The design of a pipeline reactor treatment system must first and foremost consider local wastewater and sewer conditions. As the wastewater quality always varies, the ASM#1 (Henze et al., 1987) based (Monod kinetics) models are too complex for practical use. These models require extensive calibration or they may only be used to forecast general estimates of transformations. A practical method is to consider pipeline reactor treatment as three reactive regions; microorganism limited initially, then substrate limited, and finally hydrolysis and endogenous decay limited.

In-sewer transformations are initially limited by the availability of an appropriate microbial consortium to consume the available substrate. Sewer biofilm should naturally optimize to local wastewater providing an appropriate microbial consortium to consume local wastewater. The duration of the biomass limited period was 16 h for runs seeded with biofilm for this experimental investigation. Although a conservative estimate of the duration of the biomass limited period should be used, theoretically the biofilm performance should improve with natural selection. No COD_t reduction should be estimated during the biomass limited period due to the variation of the amount of treatment received. This assumption is not true, some substrate will always be consumed as the biomass grows, but the estimate is conservative. A 10% mean treatment was observed at T_m. This initial biomass limited period will only occur once for each sewer reach, other fresh wastewater entering the selected sewer reach may be considered as substrate limited provided the fresh wastewater volumetric flow is not significantly larger than the pipeline reactor sewer volumetric flow.

The substrate limited period (from T_m to T_r) may be characterized by an initial maximum substrate consumption rate that declines for a period of time. The duration of the substrate limited period was approximately 18 h for this experimental investigation. A method similar to that used by Ozer and Kasirga (1995) is proposed to estimate the COD_t consumption rate for this period. This method involves plotting the maximum observed COD_t consumption rate versus the biodegradable portion of the COD_t to determine a rate expression in the form:

$$COD_t \text{ consumption rate} = \mu * [COD_t - COD_t(\text{non-biodegradable})] \quad \text{Equation 4.1}$$

where μ is the COD_t rate constant (h^{-1}). This equation may be solved to yield an expression for $COD_t(t)$ that is valid from T_m to T_r:

$$COD_t(t) = \text{non biodegradable } COD_t + \text{biodegradable } COD_t * e^{-\mu * t} \quad \text{Equation 4.2}$$

Figure 4.22 shows the maximum observed COD_t consumption rate versus the degradable COD_t for this experimental investigation. A best fit line with an intercept at

the origin estimated the COD_t rate constant as $\mu = 0.063\ h^{-1}$. Data used to estimate biodegradable and non biodegradable fractions of the COD_t can be seen in Appendix A.

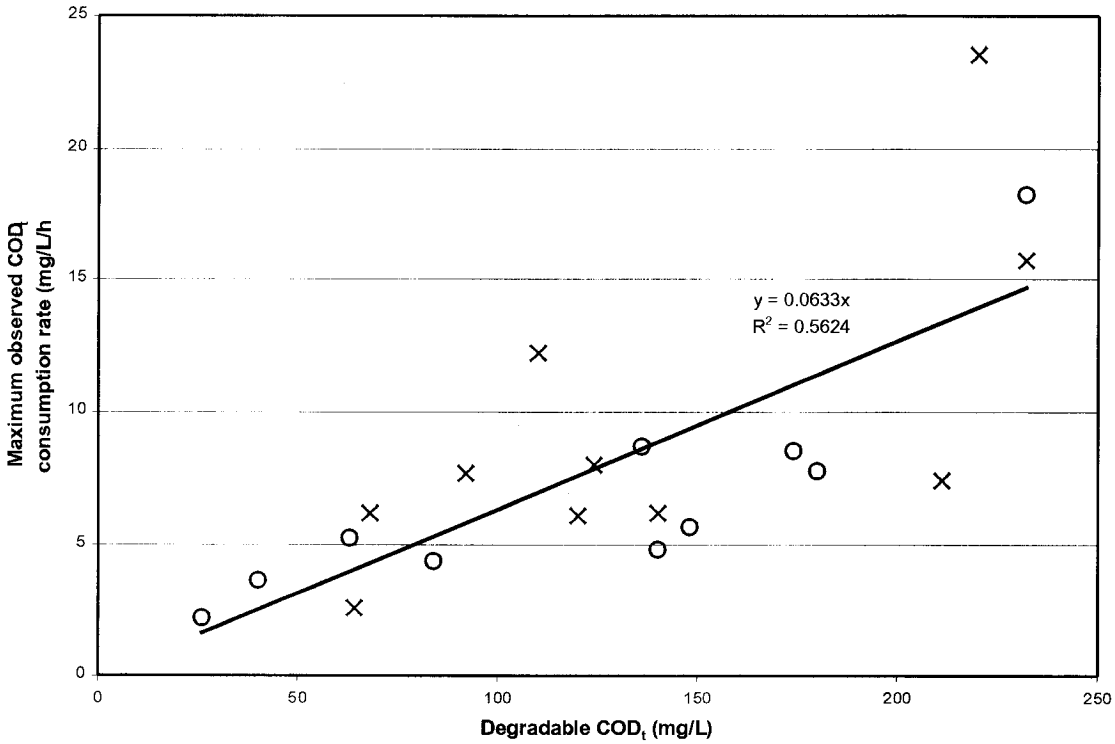


Figure 4.22 The maximum observed COD_t consumption rate versus the degradable COD_t for this experimental investigation

This is equivalent to 70% of the degradable COD_t being consumed over the 18 hours between T_m and T_r. The low regression coefficient, due to varying wastewater quality, indicates that conservative estimates of treatment received should be employed. This rate equation may be considered valid provided readily degraded substrate is present.

If the in-sewer HRT is longer than the duration of the substrate limited period without additional wastewater inflow then further COD_t consumption may be considered as particulate hydrolysis and endogenous decay. The first-order rate expression derived above is no longer valid in the hydrolysis/decay limited period. A zero-order rate expression is proposed to estimate the transformations from T_r to T_s. This experimental investigation had a mean hydrolysis limited period of 34 h that may be used as an

estimated time to consume the remaining 30% of the degradable COD_t. This is equivalent to a hydrolysis/endogenous decay rate of approximately 1% biodegradable COD_t/h.

In winter, sewer temperatures will drop below 10°C in the RMOC. Microbial growth rates are affected by reduced temperatures. Cao and Alaerts (1995) observed that the microbial growth rates decreased by a factor of 1.7 as the temperature decreased from 28° to 20°C. Hvitved-Jacobsen et al. (1998) used temperature coefficients of 1.03 for biofilm growth or 1.07 for bulk phase growth and maintenance. Winter treatment rates can be estimated as ½ of summer treatment rates requiring twice the HRT (60 h) for treatment of the readily consumed fraction of the wastewater.

The proposed models are presented in Figure 4.23 with normalized experimental data. Fourteen of the eighty experimental data points presented are above the summer model (a 1/7 model failure rate).

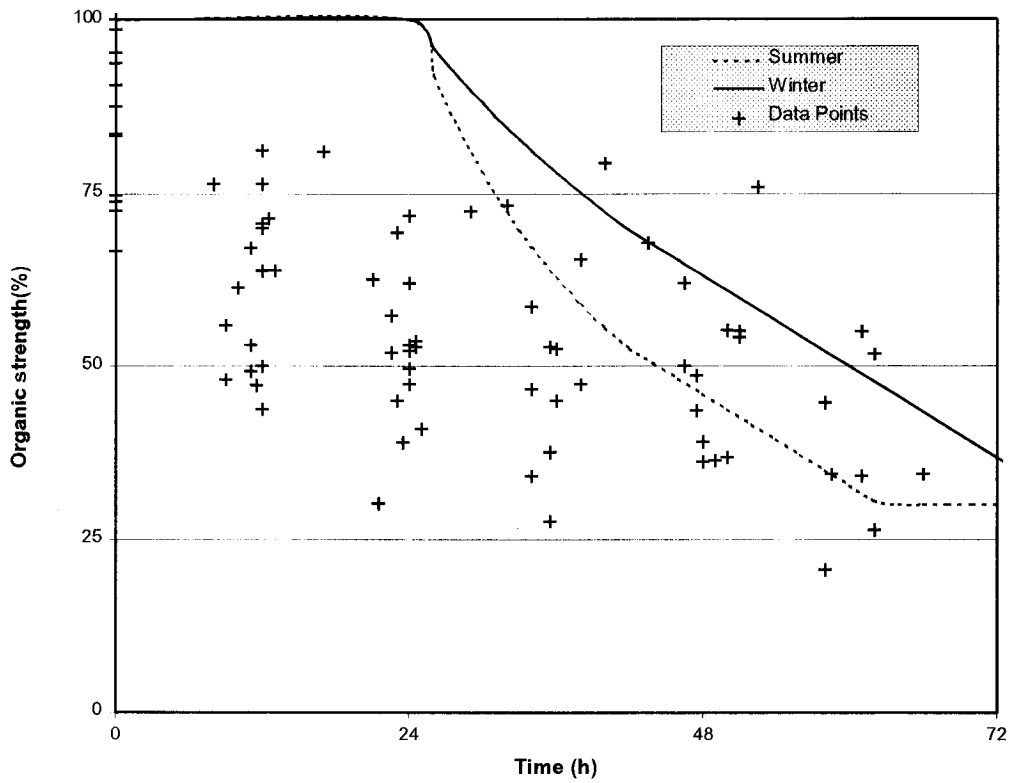


Figure 4.23 Proposed model with experimental data

The reduction in total organic load to ROPEC was estimated for the use of pipeline reactor technology from Stittsville to Kanata to Nepean to ROPEC assuming that 5% of the total ROPEC organic load entered the system at Stittsville and was conveyed 72 hours, 10% of the total organic load entered the system at Kanata and was conveyed 48 hours and 20% of the total organic load entered the system at Nepean and was conveyed 24 hours. Organic load was estimated on the basis of population and conveyance time was estimated conservatively. The lag due to microbial acclimatization was assumed to affect only the initial Stittsville to Kanata leg. This may be seen graphically in Figure 4.24. It was estimated that the total organic load to ROPEC would be reduced by 21% in summer or 18% in winter if pipeline reactor technology were applied.

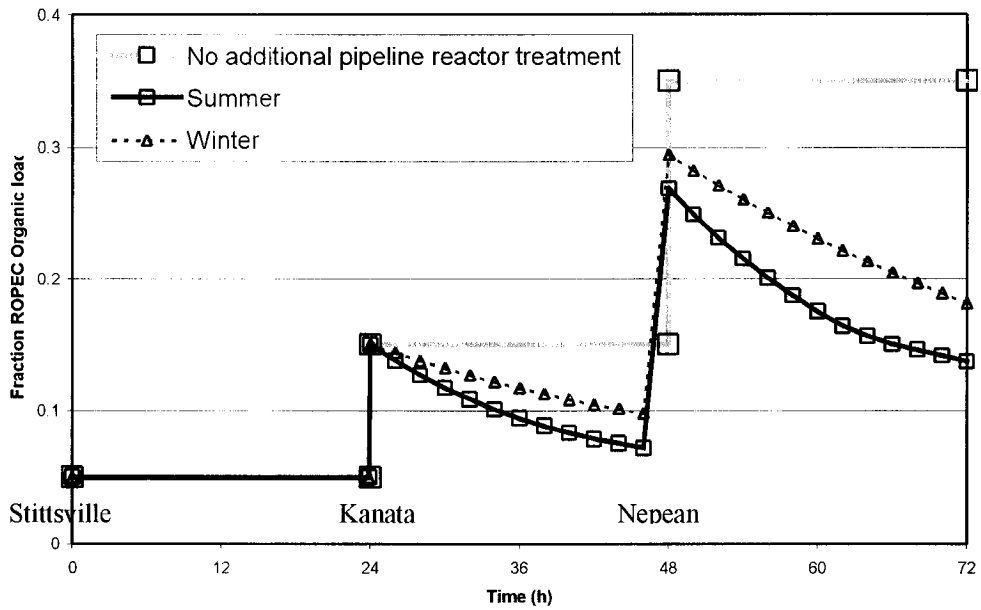


Figure 4.24 Design study applied to RMOC region

A mass balance based on the fraction of total treatment plant influent receiving pipeline reactor treatment must be used to estimate the overall effect of pipeline reactor treatment on treatment plant influent. Until more detailed wastewater characterization is performed the mass balance may be divided into soluble and particulate fractions.

Wastewater transformed from 0.62 soluble to 0.42 soluble during this experimental investigation.

The choice of electron acceptor will be based on cost analysis. No settling change between the aerobic and anoxic reactive environments was observed but theoretically anoxic conditions should improve settling (Lawler, 1997). Additional mixing at electron acceptor dosing points may improve the performance of pipeline reactors. Nitrate addition may be preferred as only one dosing point would be required as the entire dose could be added at once (Sercombe, 1995). However excess nitrate could lead to problems as nitrite may cause health risks. Oxygen addition would require multiple dosing points dispersed within the sewer system. The use of an oxygen addition strategy could reduce treatment costs as the wastewater collection system became integrated into the wastewater treatment system (Shultz et al., 1999).

Additional upstream activated sludge recycle for sludge disposal or as pipeline reactor seeding may increase pipeline reactor treatment efficiency by introducing microorganisms that may consume a larger fraction of the wastewater. Activated sludge may improve substrate consumption, particulate hydrolysis or endogenous decay rates. Optimum sludge and electron addition strategies should be investigated for a given sewer reach.

CHAPTER 5 Conclusions and Recommendations

5.1 CONCLUSIONS

This investigation was designed to determine the applicability of pipeline treatment to ROPEC wastewater and to determine if changes in particle size distribution could be related to readily degraded substrate or hydrolysis. Batch reactors were used to replicate the residence time available in a pipeline reactor. The results demonstrated that ROPEC wastewater is suitable for pipeline reactor treatment and that the quality of electron acceptor affects the microbial consortia floc strength. The experiments were carried out in a laboratory at temperatures between 21 and 23°C. A significant fraction of RMOC wastewater has in-sewer HRT greater than 48 h and consequently may be practical for pipeline reactor treatment. The extreme local weather conditions indicate that in-situ studies be used to evaluate winter pipeline reactor behaviour.

The following conclusions can be drawn from the results of this investigation.

The performance was consistent over the wide range of influent wastewater compositions and strength.

Pipeline reactor treatment requires a sewer reach with sufficient HRT to stabilize wastewater, so that minimum settling or filtration of biomass would be required at the treatment plant to achieve discharge standards.

Type of electron acceptor does not affect rate or amount of treatment but may lead to differing downstream effects. Aerobic or anoxic conditions have no significant effect during the initial 36 h (microorganism or readily degraded substrate limited) on size distribution transformations. Aerobic conditions lead to larger flocs than anoxic conditions during hydrolysis limited region.

Additional seeding would not be required as biofilm should account for seeding, providing a naturally optimized consortia of microorganisms to treat wastewater. Although not necessary for seeding purposes, upstream activated sludge disposal may increase treatment by providing microorganisms to consume different organic groups.

Advanced models (ASM#1 (Henze et al., 1987) based) show good theoretical agreement with observed phenomena but are not applicable for use without detailed in-sewer calibration.

5.2 RECOMMENDATIONS FOR FURTHER RESEARCH

These results indicate that pipeline reactor treatment is technically feasible although further research is required to verify the transformations at low (winter) temperatures is necessary. Continued laboratory studies could also be performed to determine the applicability of pipeline reactor technology for the remediation of hazardous wastes, such as landfill leachate, and for disposal of waste activated sludge.

Laboratory studies could be improved by testing a withdrawn sample for microbial activity, to determine a 'point COD consumption rate' to accompany the gross rates calculated from the COD removal.

Continuation of this study must be performed 'in-situ'. A sewer reach with sufficient HRT should be selected for process evaluation purposes. This particular sewer reach wastewater should be characterized for flow volume and quality for modelling purposes. This will allow for an economic analysis of practicality. In-situ testing should be performed to calibrate the selected process model and determine/monitor the actual cost of treatment. Electron acceptor dosing strategies must be evaluated either as nitrate dosage or as aeration requirements.

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Appendix A

Raw data

Run Pair 1 Supplements No supplements

Aerobic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm 22 µm 52µm (% volume in size range)
0		227	70											
46.5		141	56											
58.5		78	28											
69.5		78	42											

Anoxic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm 22 µm 52µm (% volume in size range)
0		254	70						34					
11		190	50											
22.5		120	85											
34.5		99	63											
46.5		70	78											
58.5		70	71											
69.5		70	91											

Run Pair 2 Supplements No supplements

Aerobic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm 22 µm 52µm (% volume in size range)
0		186	71	480	307									
11.5		177	71	563	313									
24		186	62	517	350									
33		177	72	563	357									
47		177	40	500	343									
59		151	50	480	360									
71			71	457	393									

Anoxic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm 22 µm 52µm (% volume in size range)
0		222												
11.5		262												
24		190												
33		80												
59		116												

Run Pair 3 Supplements Activated sludge seed

Aerobic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
0	7.7	262	78	730	500	365	245	15			2.3	12.0	26.1	24.8	56.4	89.4
8	7.9	200	35	630	530	180	125									
21	7.8	164	87	675	500	165	140				4.1	19.0	60.9	13.5	39.3	72.6
32	8	192	72	590	475	495	390				3.1	14.6	36.5	18.6	49.7	86.4
43.5	8	178	40	740	600	415	595				4.5	18.1	32.6	12.9	40.8	85.7
66	8	90	35	700	580	115	35				7.3	18.6	26.7	12.6	42.4	95.3
89	7.5	82	41	710	520	130	95				3.2	12.8	22.5	24.6	58.2	95.2

Anoxic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
0	7.4	262	144	790	630	325	240	15	40		3.4	15.7	29.9	16.4	46.6	86.4
8	7.7	266	140	760	505	240	210				2.1	11.0	22.9	28.0	58.7	92.4
21	8.2	170	85	650	505	160	90				2.5	12.9	33.4	23.7	53.1	82.9
32	8.4	132	24	700	530	715	355				2.8	13.4	26.0	20.9	54.2	88.9
43.5	8.5	100	44	690	620	480	465				1.4	10.7	51.7	26.7	51.1	74.0
66		55	41	725	680	135	560				3.2	15.2	31.2	18.3	44.4	80.9
89	8.7	69	55	705	595	120	140				1.7	10.4	24.4	27.8	56.1	89.9

Run Pair 4 Supplements Activated sludge seed

Aerobic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
5.5	7.8	159	136	525	505	40	75	10			1.2	10.3	27.5	27.3	52.8	87.8
17.5	7.9	149	91	525	575	120	105	10			3.9	12.3	19.5	27.5	58.2	100
29	8	140	48	665	485	220	180	10								
41	8	112	68	520	595	140	180				5.6	15.5	22.6	19.2	53.8	99.8
52	8.1	72	56	740	685	190	45				5.1	15.1	22.9	20.0	50.4	98.9
65	8.05	72	48	530	500	45					2.1	14.8	35.6	16.9	41.7	80.8
											1.4	13.4	30.7	17.1	41.1	85.3

Anoxic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
5.5	7.3	181	81	180	160	115	20	10			3.3	19.8	78.3	12.7	34.0	66.1
17.5	8.2	176	86	575	540	155	90	10								
29	8.5	164	88	640	560	205	220				3.8	13.2	21.4	23.2	57.4	98.9
41	8.5	164	84	570	540	130	90	10			1.8	11.4	27.2	25.3	54.5	90.9
52	8.5	96	72	810	750	165	430	10			1.7	11.3	27.5	25.1	55.6	88.4
65	8.4	96	60	690	520	140	90	10			4.3	13.6	21.3	23.2	57.9	98.8

Run Pair 5 Supplements Activated sludge seed and soluble substrate

Aerobic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
0	7.3	304	184	660	610	105	55	9	4	4	1.5	9.5	20.3	33.5	64.4	95.9
18	7.7	254	192	565	485	150	35	10	8		2.6	12.6	21.7	23.4	60.5	96.1
30	7.7	152	88	520	490	140	80	10	8	6	7.4	19.4	27.7	10.8	40.3	93.2
42	7.85	144	64	565	495	155	50				8.1	19.8	27.3	10.4	38.4	95.7
56	8	144	24	555	545	250	190	10	9	4	5.3	22.2	41.0	8.8	28.8	78.9
68	8.1	112	40	520	495	195	155	10			5.7	25.2	60.1	7.7	25.6	67.0
80	8.1	80	56	430	350	215	145				5.9	22.9	46.0	8.8	29.4	75.4

Anoxic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
0	7.2	328	248	735	665	80	75	8	200	4	1.4	8.7	19.9	35.5	66.0	95.4
18	7.4	352	216	790	630	110	60	10	180	5	3.1	14.2	23.7	20.1	54.5	94.8
30	7.8	164	76	585	585	85	85	10	70		2.8	19.6	64.1	12.5	36.8	67.1
42	8	144	160	590	535	180	70		8		1.8	13.6	24.6	18.4	51.8	93.8
56	8.1	152	64	630	585	245	245	10	9	5	2.8	14.8	32.5	17.6	51.4	86.9
68	8.15	128	96	625	585	225	175				2.8	16.8	39.4	14.2	42.4	80.2
80	8.2	120	32	595	450	210	170				1.3	12.0	36.1	20.2	50.1	82.4

Run Pair 6 Supplements Soluble substrate

Aerobic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
0	7.3	272	168	640	565	245	160	8		4	1.1	8.9	38.1	33.9	57.9	79.2
19	7.9	260	176	735	640	275	185				1.4	12.1	35.4	21.0	52.4	83.9
29	7.3	216	124	665	625	260	155	12								
44	8.1	144	72	565	565	180	125		0.7		4.3	20.3	40.1	10.6	32.0	79.8
53	7.8	120	56	645	750	35	95				5.8	24.4	45.3	7.6	24.1	75.2
65	7.9	168	56	600	790	120	190				6.7	23.8	43.7	7.8	27.0	77.1
89	7.9	116	88	750	605	175	570	9	0.5	4	5.4	28.2	74.8	6.7	23.7	73.8

Anoxic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
0	7.3	320	184	935	970	455	310	8	180	4	2.3	14.2	55.2	23.9	42.3	61.9
19	7.3	268	136	910	785	350	255				6.6	23.4	41.8	7.3	27.6	80.0
29	7.1	248	116	799	730	345	260	12			7.4	21.7	31.4	8.8	31.4	88.6
44	7.5	240	80	755	640	235	155				8.0	23.8	35.3	7.8	26.1	83.2
53	8.15	232	40	930	820	325	260		5.5		6.4	25.1	44.0	7.4	22.6	76.3
65	8.5	140	60	940	805	330	235	9	8	4	6.6	24.6	43.0	7.4	24.0	78.1
89	8.6	168	48	850	725	300	210				7.0	25.8	48.8	6.7	23.7	73.8

Run Pair 7 Supplements Soluble and particulate substrate

Aerobic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
0	7.1	272	144	805	600	285	80	10	2.8	5	2.0	11.4	32.6	29.4	54.1	78.9
17	7.7	288	120	620	600	105	115	11	9.8		5.1	17.5	28.8	13.4	44.1	87.6
29	7.5	220	100	725	660	195	175	10	5.6		6.6	20.2	31.0	10.9	35.7	86.8
40	7.9	200	96	790	710	280	205	10	3.9		6.0	20.6	32.5	10.3	33.7	84.0
52.5	8	152	104	660	550	160	70	10	3.9	4	6.9	21.2	32.8	10.0	32.9	83.7
64.5	8	140	88	595	595	70	60	10	3.5		4.7	20.0	36.4	11.6	32.7	79.3

Anoxic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
0	7.1	232	136	1915	1925	875	990	10	210	2	1.8	12.2	43.6	24.7	50.3	76.1
17	7.35	188	112	805	595	245	155	11	9.8		5.7	28.7	79.2	6.7	21.9	56.8
29	7.2	168	88	775	540	295	80				5.7	25.4	49.9	7.2	23.1	70.2
40	7.2	184	48	690	605	185	165	10	12.6	4	6.2	25.9	51.3	7.3	22.8	68.3
52.5	7.1	176	53	900	690	355	210	10	9.8		6.6	28.8	66.3	6.5	20.1	60.8
64.5	7.2	208	52	855	730	310	215				6.3	26.5	52.6	7.1	22.2	67.3

Run Pair 8 Supplements Soluble and particulate substrate

Aerobic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
0	7.2	276	120	635	525	215	135				1.1	7.7	20.4	38.2	66.0	92.7
17	7.5	232	124	620	615	195	170				4.3	18.4	40.3	13.2	38.2	80.5
26.5	7.4	236	120	590	550	350	380	7	4.4		5.2	22.6	45.7	9.4	27.4	74.3
41	7.6	184	86	585	530	395	370	8			6.9	26.4	50.4	7.1	22.8	68.8
52	7.7	136	56	485	470						6.9	27.6	54.5	6.5	21.3	65.7
65	7.6	144	56	470	515	130	155				6.7	29.1	61.1	6.2	19.8	60.4
89	7.55	100	86	545	420	130	130	0.8	9.8		4.5	24.8	63.0	8.9	24.7	60.8

Anoxic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
0	7.2	244	128	915	875	410	390		180		6.4	48.9	117.4	4.1	9.4	23.9
17	7.3	216	60	875	725	395	300				4.4	19.5	41.0	11.4	36.0	79.5
26.5	7.4	256	86	705	858	390	425		35		7.9	25.1	38.3	6.8	24.5	77.4
41	8	240	108	625	550	420	420	7			5.8	22.3	37.8	9.2	28.2	77.7
52	8.25	172	64	735	545	225	65	10			5.2	21.6	44.2	9.7	60.8	77.1
65	8.45	184	84	640	595	160	70				4.8	24.8	68.1	8.2	25.4	65.3
89	7.55	100	86	545	420	130	130	10			5.5	19.7	32.5	11.0	36.2	84.1

Run Pair 9 Supplements Biofilm seed

Aerobic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
0	7.1	260	116	530	485	175	105	8	0.7	4	1.7	10.4	34.1	30.8	55.8	80.9
15.5	7.2	288	120	670	460	245	85	10	0.7		1.8	10.0	24.5	32.1	60.9	87.6
27.5	7.6	184	28	430	290	210	100	10	0.7	3	2.1	10.8	24.9	29.8	59.7	87.8
40	7.9	152	16	650	445	170	40	10	0.7		2.8	14.2	46.1	21.4	49.4	77.7
66.5	7.8	156	8	520	395	145	45	10	0.7	3	3.1	14.0	27.9	21.1	53.0	86.3
74.5	7.8	172	72	515	345	140	20	9	0.7		3.7	16.5	42.4	16.7	45.8	78.3
90	7.75	160	16	580	385	125	20	9	0.5		3.7	16.5	38.3	16.2	45.1	80.4

Anoxic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
0	7.1	276	104	840	600	295	185	10	70	4	3.3	18.1	72.7	17.7	36.1	58.8
15.5	7.9	272	124	685	565	195	160	10	56		2.8	13.6	45.6	23.9	49.8	75.9
27.5	8.2	176	24	510	430	200	160	10	40	6	2.3	12.8	48.1	25.2	50.0	75.4
40	8.25	148	16	630	515	165	135	9	21		3.1	14.0	46.2	22.9	49.8	76.8
66.5	8.4	152	48	625	470	200	100	9	15	4	3.1	14.8	46.8	21.0	47.8	75.8
74.5		168	72	585	500	140	80				2.8	13.9	43.0	22.7	50.7	78.3
90	8.55	172	60	605	490	125	35	12	4		3.1	14.1	45.2	22.8	50.8	78.1

Run Pair 10 Supplements Particulate substrate and biofilm seed

Aerobic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
0	7.3	600	128	1885	415	680	130	10	3.9		3.3	12.4	52.7	20.3	46.0	75.0
16.5	7.55	500	104	4095	465	3120	85	10	8.4		3.4	13.2	58.4	16.1	40.1	74.1
25.5	7.5	336	92	2250	500	1685	70	9	8.4		3.5	13.4	57.7	11.1	34.1	74.9
39	7.5	344	92	960	530	470	100	8	5.6		3.6	14.3	62.9	8.8	27.8	70.2
50.5	7.5	352	64	860	570	365	85	7	8.4		3.7	15.1	67.1	5.9	22.1	66.1
63	7.5	300	64	800	545	385	115	4.7	11.9		3.8	16.0	67.1	4.4	17.4	61.4
74.5	7.3	268	36	880	585	370	130	0.85	5.6		4.0	17.7	84.4	3.8	14.5	55.3

Anoxic

Time h	pH	CODt mg/L	CODs mg/L	TS mg/L	TVS mg/L	TDS mg/L	VDS mg/L	NH3 mg/L	NO3 mg/L	PO4 mg/L	Dn µm	Da µm	Dv µm	11 µm (% volume in size range)	22 µm	52µm
0	7.3	600	104	765	115	305	200	10.5	70		3.1	11.2	45.6	23.7	52.4	79.6
16.5	8	500	172	1640	565	1140	150	11	63	4	3.4	12.9	54.9	14.3	40.3	75.6
25.5	8	288	80	885	550	410	180				3.1	9.9	34.1	16.6	46.7	85.1
39	8.2	312	96	1045	605	500	130	9.5	11.2	4	3.0	9.7	33.1	20.2	51.5	77.1
50.5	8.1	280	64	840	595	270	135	8	9.1		3.3	11.7	43.4	10.6	33.3	77.2
63	8.4	396	116	835	600	315	160	10	6.3	5	2.9	9.2	31.6	26.3	56.6	85.0
74.5	8.4	392	144	835	600	300	120	10	5.6		3.3	12.5	55.9	26.1	50.0	75.5

Table of total, non biodegradible (NBD) and biodegradible (BD)CODt

Run ID	Influent	NBD	BD
1o	227	78	149
1x	254	70	184
2o	186	151	35
2x	242	116	126
3o	262	82	180
3x	264	55	209
4o	159	72	87
4x	181	96	85
5o	304	80	224
5x	340	120	220
6o	272	116	156
6x	320	168	152
7o	280	140	140
7x	232	176	56
8o	276	100	176
8x	244	100	144
9o	274	152	122
9x	274	148	126
10o	600	268	332
10x	600	280	320

Appendix B

Error estimates

COD TESTS

Error of COD test was estimated as the mean difference of replicate subsamples.

Mean Value	Difference	% Difference
260	0	0
144	0	0.0
262	0	0.0
200	0	0.0
216	0	0.0
168	12	7.1
116	12	10.3
276	12	4.3
232	12	5.2
262	12	4.6
266	12	4.5
140	12	8.6
320	12	3.8
304	24	7.9
164	24	14.6
181	24	13.3
222	24	10.8
328	24	7.3
262	24	9.2
186	24	12.9
272	24	8.8
254	24	9.4
159	35	22.0
177	35	19.8
141	35	24.8
232	47	20.3
190	47	24.7
244	47	19.3
99	47	47.5
272	47	17.3
227	47	20.7
151	47	31.1
600	59	9.8
276	59	21.4
208	71	34.1
168	71	42.3
312	71	22.8
336	94	28.0
500	141	28.2

	600	259	43.2
Mean	248	39	15
S.D.	110	46	12

SOLIDS

Error of solids tests was estimated as the mean difference of replicate subsamples. Total solids and total dissolved solids cannot be compared on a line basis as they are ordered in ascending difference.

Total Solids (TS)			Total Dissolved Solids (TDS)		
Mean Value	Difference	% Difference	Mean Value	Difference	% Difference
440	10	2.3	40	0	0
520	20	3.8	120	0	0.0
690	25	3.6	220	10	4.5
575	30	5.2	210	10	4.8
590	40	6.8	245	10	4.1
480	40	8.3	190	15	7.9
470	40	8.5	70	15	21.4
790	45	5.7	350	15	4.3
705	60	8.5	45	20	44.4
730	60	8.2	115	20	17.4
640	80	12.5	155	20	12.9
520	80	15.4	295	20	6.8
625	100	16.0	205	25	12.2
515	100	19.4	130	25	19.2
670	100	14.9	355	25	7.0
563	100	17.8	140	25	17.9
545	100	18.3	160	25	15.6
810	120	14.8	165	30	18.2
525	123	23.4	245	30	12.2
485	125	25.8	195	30	15.4
635	128	20.2	395	30	7.6
640	130	20.3	195	30	15.4
665	130	19.5	225	30	13.3
530	160	30.2	215	35	16.3
625	165	26.4	125	35	28.0
790	180	22.8	325	40	12.3
640	180	28.1	115	40	34.8
430	200	46.5	130	40	30.8
735	200	27.2	365	45	12.3
790	200	25.3	130	45	34.6
810	200	24.7	420	45	10.7
665	220	33.1	185	50	27.0
180	220	122.2	175	50	28.6
520	225	43.3	130	50	38.5

	765	270	35.3	105	50	47.6
	660	300	45.5	160	60	37.5
	530	300	56.6	390	65	16.7
	805	300	37.3	280	90	32.1
	1885	305	16.2	285	90	31.6
	840	370	44.0	240	100	41.7
	915	400	43.7	480	110	22.9
	935	500	53.5	630	110	17.5
	1915	500	26.1	715	120	16.8
Mean	693	167	25	234	40	19
S.D.	306	123	21	143	30	12

NUTRIENTS

Sample calibration curves are presented.

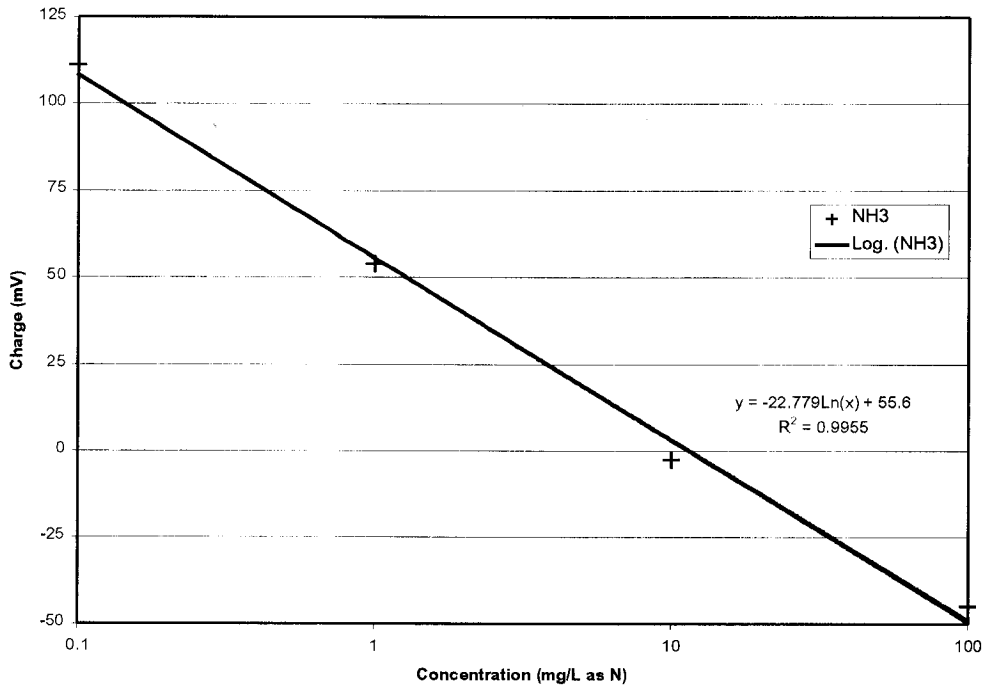

Figure B1 Sample NH_3 calibration curve

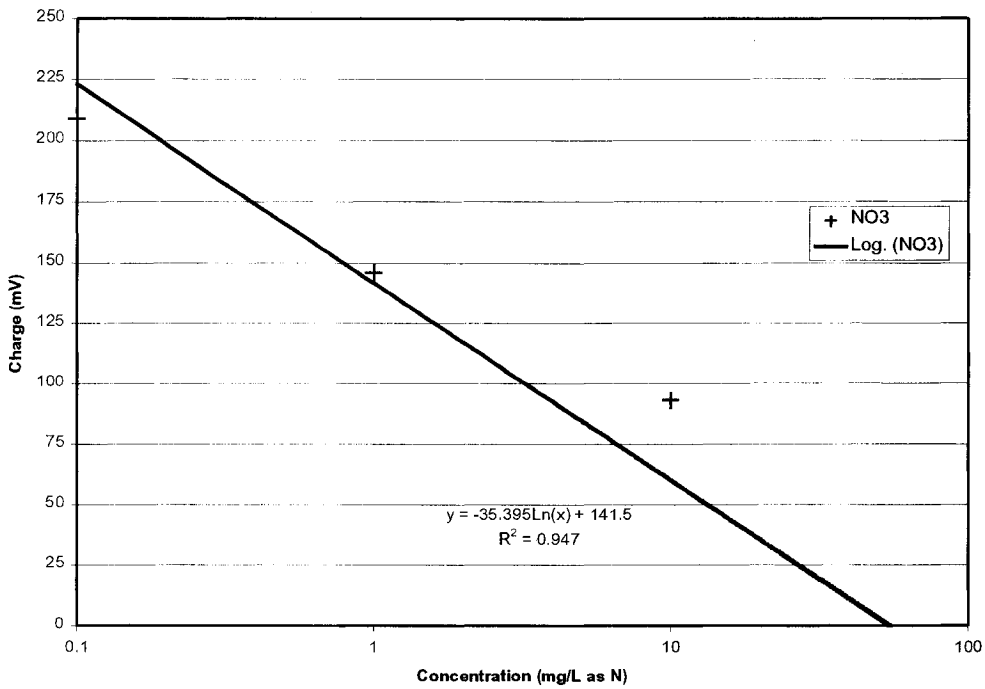

Figure B2 Sample NO_3 calibration curve

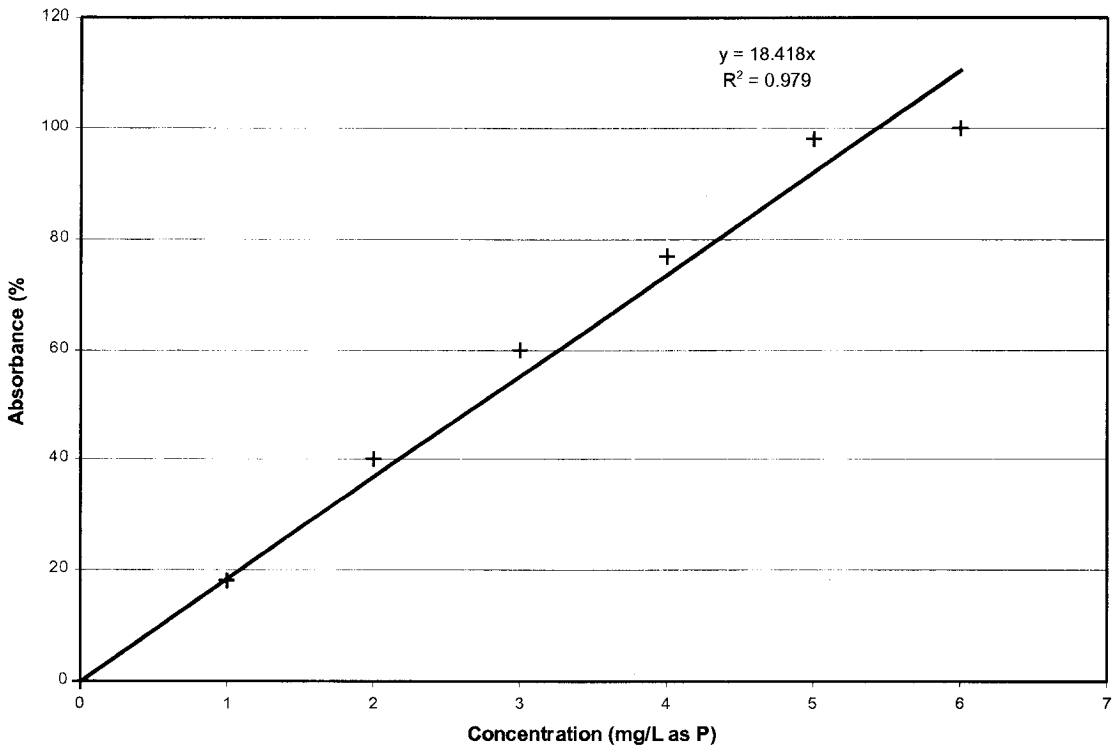

Figure B3 Sample PO_4 calibration curve

Appendix C

Sample calculations

Sample calculations are shown for run 5O

Time (h)	0	18	30	42	56	68	80
CODt (mg/L)	304	254	152	144	144	112	80
	Tl	Tm	Tr			Ts	

Organic content

COD_t removed (CODR); calculated from influent to T_s

$= COD_t(influent) - COD_t(stabilized)$

$= 304 - 80$ mg/L

$= 224$ mg/L

Treatment efficiency (TE), calculated from influent to effluent

$= CODR/COD_t(influent)$

$= 224/304$

$= 74$ %

COD removal rate, calculated from time =0 h to time = 18 h

$= [COD(n) - COD(n+1)] / [time(n+1) - time(n)]$

$= [304 - 254] / [18 - 0]$ mg/L/h

$= 2.8$ mg/L/h

Size distribution; influent to effluent change of 11 to 22 μm size range shown

%particle volume in size range = (%particle volume past upper limit) – (%particle volume past lower limit)

= 64.4 – 33.5 %particle volume (time = 0 h, 11 to 22 μm size range)

= 30.9 %particle volume

Influent = mean of 1st and 2nd measurements (11 to 22 μm pass volume)

= (30.9 + 37.1) / 2 %total particle volume

= 34.0 %total particle volume

Effluent = mean of final and 2nd last measurements

= (17.9 + 20.6) / 2 %total particle volume

= 19.25 %total particle volume

Influent to effluent decrease of %particle volume in size range

= %particle volume in size range (influent) - %particle volume in size range (effluent)

= 34.0 – 19.25 %particle volume in size range

= 14.75 decrease of %particle volume in size range

Calculation of particulate hydolysis, and biomass growth rates

Data table for microorganism growth and hydrolysis rates

	Aerobic				Anoxic			
Run ID	time		CODt	CODs	time		CODt	CODs
	3	0.0	262.0	78.0	0.0	262.0	44.0	
		43.5	178.0	40.0	43.5	100.0	44.0	
Rate	(mg/L/h)		1.9	0.9		3.7	0.0	
	4	5.5	159.0	136.0	5.5	181.0	81.0	
		41.0	112.0	68.0	41.0	164.0	84.0	
Rate	(mg/L/h)		1.3	1.9		0.5	-0.1	
	5	0.0	304.0	230.0	0.0	360.0	280.0	
		42.0	144.0	64.0	42.0	150.0	80.0	
Rate	(mg/L/h)		3.8	4.0		5.0	4.8	
	6	0.0	256.0	152.0	0.0	304.0	168.0	
		44.0	128.0	56.0	44.0	224.0	64.0	
Rate	(mg/L/h)		2.9	2.2		1.8	2.4	
	7	0.0	272.0	144.0	0.0	232.0	136.0	
		40.0	160.0	56.0	40.0	208.0	48.0	
Rate	(mg/L/h)		2.8	2.2		0.6	2.2	
	8	0.0	276.0	120.0	0.0	244.0	128.0	
		41.0	136.0	56.0	41.0	167.0	60.0	
Rate	(mg/L/h)		3.4	1.6		1.9	1.7	
	9	0.0	228.0	84.0	0.0	290.0	72.0	
		40.0	152.0	16.0	40.0	300.0	40.0	
Rate	(mg/L/h)		1.9	1.7		-0.3	0.8	
	10	0.0	500.0	128.0	0.0	344.0	112.0	
		39.0	500.0	92.0	39.0	320.0	104.0	
Rate	(mg/L/h)		0.0	0.9		0.6	0.2	
Mean			2.3	1.9		1.7	1.5	

Summary	
Mean Tr (h)	41
Aerobic Mean CODt rate (mg/L/h)	2.3
Mean CODs rate (mg/L/h)	1.9
Anoxic Mean CODt rate (mg/L/h)	1.7
Mean CODs rate (mg/L/h)	1.5

COD_t consumed = Substrate consumed – Biomass growth
Biomass growth = Substrate consumed * Yield
Substrate consumed = COD_t consumed / (1-Yield)
Particulate hydrolysis = Yield * Substrate consumed - COD_P consumed
Particulate hydrolysis = Yield *Substrate consumed – COD_t consumed + COD_s consumed

Calculated for aerobic case,
Y=0.5, Rate of COD_t consumed =2.3 mg/L/h, Rate of COD_s consumed = 1.9 mg/L/h

Rate of total substrate uptake

=Rate of COD_t removal/(1-Y)
=2.3/(1-0.5)
=4.6

Rate of biomass growth

=Y*(rate of total substrate uptake)
=0.5*4.6
=2.3 mg/L

Rate of particulate hydolysis

= Yield * rate of Substrate consumed – rate of COD_t consumed + rate of COD_s consumed
=2.3-2.3 +1.9
=1.9 mg/L

The model matrix for Hvitved-Jacobsen et al., (1998) is presented as a process matrix, governing equations, nomenclature and sample calculations.

Process	1 Ss	2 Xs1	3 Xs2	4 Xbw	5 -So
Re-aeration					-1
Suspended biomass	-1/Yhw			1	(1-Yhw)/Yhw
Biofilm	-1/Yhf			1	(1-Yhw)/Yhw
Maintenence	-1			-1*	1
Hydrolysis, fraction 1	1		-1		
Hydrolysis, fraction 2	1			-1	

Heterotrophic growth

$$\mu_h \times \frac{S_s}{(K_s + S_s)} \times \frac{S_o}{(K_{o2} + S_s)} \times X_H \times 1.07^{T-20}$$

Maintenance energy

$$q_m \times \frac{S_O}{K_O + S_O} \times X_H \times 1.07^{T-20}$$

Biofilm

$$k_{1/2} \times S_O^{0.5} \times \left(\frac{Y_{Hf}}{1 - Y_{Hf}}\right) \times \left(A/V\right) \times \left(\frac{S_s}{K_{Sf} + S_s}\right) \times 1.03^{T-20}$$

Hydrolysis (for hydrolysis fraction n)

$$k_{hn} \times \frac{X_{Sn}/X_{Bw}}{\left(K_{Xn} + X_{Sn}/X_{Bw}\right)} \times \left(X_{Bw} + X_{Bf}(A/V)\right) \times 1.07^{T-20}$$

Re-aeration

$$Kla \times \left(S_{O,S} - S_O\right)$$

K_la

$$Kla = 0.86\left(1 + 0.2F^2\right)(su)^{3.8} d_m^{-1} \times 1.024^{T-20}$$

Nomenclature

Y_{hw}, heterotrophic yield constant	0.55 g COD biomass/g COD substrate
Y_{hf}, biofilm yield constant	0.55 g COD biomass/g COD substrate
μ_h, maximum specific growth rate	3.25 d^{-1}
Ks, saturation constant for readily biodegradable substrate	1.0 g COD/m3
Ko, saturation constant for DO	0.5 g O2/m3
K_La, oxygen transfer coefficient	d^{-1}
T, temperature	20 ° C
Sos, dissolved oxygen saturation concentration at T (°C)	9 g/m^3
F, Froude number $(Y/(g*d_m)^{0.5}$	dimensionless
u, mean flow velocity	m/s
g, gravitational acceleration	m/s^2
d_m, hydraulic mean depth	m
s, slope	0.003 m/m
q_m, maintenance energy requirement rate constant	1.0 d^{-1}
$k_{1/2}$, half order rate constant	4.8 g $O_2^{0.5}m^{-0.5}d^{-1}$
A/V, sewer pipe surface area – volume ration	0.5 m^{-1}
Ksf, saturation constant for readily biodegradable substrate in biofilm	1.0 g COD/m^3
k_{hn}, hydrolysis rate constant, fraction n	n=1 4.0 n=2 1.0 d^{-1}
K_{Xn}, saturation constant for hydrolysis, fraction n	n=1 0.5 n=2 0.2 g COD/g COD
ε, efficiency constant for the biofilm biomass	0.5 dimensionless
Xbw heterotrophic active biomass in the water phase	40 g COD/m3
Xs1 hydrolyzable substrate, fast biodegradable	75 g COD/m3
Xs2 hydrolyzable substrate, slowly biodegradable (includes inert organic matter)	325 g COD/m3
Ss readily biodegradable substrate	25 g COD/m3
So dissolved oxygen	5 gO2/m3

Rates

Heterotrophic growth

$= 3.25\ d^{-1} * 25/(25+1) * 5/(5+0.5) * 40\ mg/L * 1.07^{20-20}$

$= 113.6\ mg/L/d$

Maintenance energy requirement

$= 1\ d^{-1} * 5/(5+0.5) * 40\ mg/L * 1.07^{20-20}$

$= 36.4\ mg/L/d$

Biofilm

$= 4.8\ g\ O_2^{0.5}m^{0.5}d^{-1} * 5^{0.5} * (0.55/0.45) * 0.5 * 25/(25+1) * 1.03^{20-20}$

$= 12.6\ mg/L/d$

Hydrolysis (fraction 1)

$= 4.0\ d^{-1} * [(75/40) / (0.5+(75/40))] * (40 + 0.1 * 40 * 0.5)\ mg/L$

$= 132.6\ mg/L/d$

Hydrolysis (fraction 2)

$= 1.0\ d^{-1} * [(325/40) / (0.2+(325/40))] * (40 + 0.1 * 40 * 0.5)\ mg/L$

$= 41.0\ mg/L/d$

Mass balance on readily degraded substrate for a HRT(dt) of 0.01 d

Ss(1) = Ss(0) +[–(1/Yhw)*(heterotrophic growth) –(1/Yhf)*(biofilm growth) –(maintenance energy) + (hydrolysis generation)]*dt

= 25 mg/L +[–(1/0.55)*(113.6 mg/L/d) –(1/0.55)*(12.6 mg/L/d) –36.4 mg/L/d +132.6 mg/L/d +41.0 mg/L/d] * 0.01 d
= 25 mg/L –(91.4 mg/L/d) * 0.01 d
= 24.1 mg/L

The model matrix for Almieda et al., (1999) is presented as a process matrix, governing equations, nomenclature and sample calculations.

Process	1 Si	2 Ss	3 Xi	4 Xs	5 Xh	6 So	7 Snh
Heterotrophic growth		-1/Yh			1	-(1-Yh)/Yh	$-I_{NBM}$
Decay			fxi	1-fxi	-1		
Hydrolysis	fsi	1-fsi		-1			v3,nh
Biofilm		-1/(1-Yh)				-1	
Re-aeration						1	

Heterotrophic growth

$$\mu_h \times \frac{S_s}{(K_s + S_s)} \times \frac{S_o}{(K_{o2} + S_s)} \times X_H$$

Endogenous decay

$$b_h \times \frac{K_b}{K_b + \left(S_s / X_H\right)} \times X_H$$

Hydrolysis

$$k_h \times \frac{\left(X_s / X_H\right)}{K_x + \left(X_s / X_H\right)} \times X_H$$

Biofilm

$$k_{su} \times (sU)^{0.5} \times \left(A_b / V\right)$$

Re-aeration

$$Kla \times \left(S_{O,S} - S_O\right)$$

Nomenclature

μ_h maximum specific growth rate for biomass 5 d^{-1}

Ks	saturation coefficient for biomass		5 g/m^3
b_H	decay coefficient for heterotrophic biomass	1	
k_H	specific hydrolysis rate		7 d^{-1}
K_b	saturation coefficient for decay		1
K_x	saturation constant for hydrolysis		0.3
K_O	saturation coefficient for oxygen		1
k_{su}	coefficient for biofilm		7
Y_h	heterotrophic yield		0.55 g COD biomass/g
COD substrate			
fxi	biomass yielding inert particulate products		0.1 fraction
fsi	inert soluble products resulting from hydrolysis		0.1 fraction
v3,nh	ammonia produced on hydrolysis		
I_{NBM}	nitrogen used in cell synthesis		
Si	inert soluble matter		50 g/m^3
Ss	readily biodegradable matter		110 g/m^3
Xi	inert particulate matter		75 g/m^3
Xh	biomass		40 g/m^3
Xs	slowly biodegradable matter		225 g/m^3
So	dissolved oxygen		5 g/m^3
S_{NH}	ammonia plus ammonium nitrogen		g/m^3

Rates
Heterotrophic growth
= 5 d^{-1} * 110/(110+5) * 5/(5+0.5) *40 mg/L
= 173.9 mg/L/d
Endogenous decay
= 1 d^{-1} *1/(1+(110/40)) * 40 mg/L
= 10.7 mg/L/d

Hydrolysis
= 7 d^{-1} * (225/40)/(0.3+(225/40)) * 40 mg/L
= 265 mg/L/d

Biofilm
= 7 * (0.002*1)$^{0.5}$ * (0.5)
= 0.15 mg/L/d

Mass balance on Heterotrophic biomass for a HRT(dt) of 0.01 d
$X_h(1)$ = $X_h(0)$ +[Heterotrophic growht – decay]*dt
= 40mg/L + [173.9 –10.7] mg/L/d *0.01 d
= 41.6 mg/L