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# BIOLOGICAL PHOSPHORUS REMOVAL FROM MUNICIPAL WASTEWATER

by

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## ABSTRACT

A laboratory scale study was undertaken to investigate the possibility of removing "enhanced" amounts of phosphorus from municipal wastewater using solely biological means. The system consisted of an anaerobic-aerobic treatment sequence. Several modifications were studied to observe their effect on the capability of the system to remove phosphorus. These included temperature changes and the addition of a sludge conditioning reactor to minimize nitrate input to the anaerobic reactor and thus eliminating the possible interference with the release of phosphorus in this reactor and the subsequent uptake in the aerobic reactor. Parameters such as COD, MLSS, DO, ORP, nitrates, nitrites, TKN, pH, alkalinity, and heavy metals were monitored throughout the study in addition to the determination of orthophosphates and total phosphates.

The results show that nitrification activity, once established in the aerobic reactor, was extremely difficult to curtail due to the small scale effects. The denitrification of the return sludge in the sludge conditioning reactor did not in itself result in the manifestation of "enhanced" phosphorus removal by the system. It is hypothesized that excellent phosphorus removal (up to 98%) can be obtained using essentially an anaerobic-aerobic treatment scheme only when the system is operated at exceptionally long sludge ages (approximately 86 days in this study).

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## SUMMARY

This thesis examines the conditions required for the enhanced biological removal of phosphorus in a modified activated sludge treatment scheme. The effect of phosphorus on the aquatic environment and reasons for the control of this nutrient, by judicious management of point source discharges, are presented. Methods of control presently available to the wastewater engineering discipline are briefly outlined. An overview of current literature dealing with the enhanced phosphorus removal capabilities of several biological wastewater treatment schemes and the mechanisms involved in phosphorus uptake are discussed.

The physical characteristics of the system investigated in this study, together with the operating conditions and regimes, are outlined. Sampling, analysis, and monitoring procedures are also described. Equations used to calculate hydraulic retention times, solids retention times, $\Delta P/\Delta COD$ ratios, and reaction rates are given.

The study involves two phases. Phase A deals with the preliminary investigations undertaken to determine the response of the system to various modifications introduced to encourage the removal of phosphates. Up to 55% phosphorus removal was obtained without chemical addition. Phase A is characterized by constant physical and operational alterations aiming at curtailing possible effects of nitrates (particularly in the anaerobic reactor) on the phosphorus release and uptake

phenomena. The denitrification of oxidized nitrogen species in the sludge return was finally achieved in the sludge conditioning reactor, thus precluding denitrification from taking place in the anaerobic reactor. No improvement in either the release or uptake of phosphorus resulted however.

Phase B represents the period of study when enhanced phosphorus removal was realized. Percentage removals were significantly higher than exhibited during Phase A, reaching a high of 98%. There were no physical modifications effected during Phase B. The only significant operating change was the maintenance of exceptionally long solids retention times.

# CHAPTER 1

## INTRODUCTION

As the end of the twentieth century approaches, the populace today is becoming increasingly aware of the limits of our environment to assimilate man's wastes. Municipal wastewater, if improperly treated, may have numerous detrimental effects on the water body into which it is discharged. Early objectives of municipal treatment schemes aimed at removing biodegradable organics and suspended and floatable matter, as well as the elimination of pathogens for obvious health reasons. As a better understanding of the effects on the environment of wastewater discharges develops, an increase in the degree of treatment has become the trend. Limits on nutrients (nitrogen and phosphorus), trace metals, refractory organics, and dissolved inorganic solids, have been incorporated into the standards of numerous communities all over the world.

This chapter will introduce the concept of water quality degradation due to the addition of nutrients from point sources. Since phosphorus is often the limiting nutrient, methods by which its discharge can be curtailed will be presented.

## 1.1 The Effects of Nutrients on the Aquatic Environment

The impact of nutrients on natural water bodies is of greater significance to lakes than rivers because of the accumulating potential of the former. Lakes fall into three major limnological classes: oligotrophic, mesotrophic, and eutrophic. Oligotrophic lakes are relatively young, having a low input of nutrients and a sparse biological population. Lakes with higher levels of biological productivity (characterized by the presence of fish), due to an adequate nutrient supply, are known as mesotrophic. As additional nutrients are added, productivity increases until the habitat is no longer optimal for the species present in the system. At this point the lake becomes eutrophic. This stage in lake life is characterized by excessive biological growth, in particular algal blooms. A multitude of problems tend to result. These include turbidity and dissolved oxygen depletion due to organic matter decay. Algal blooms are a particular nuisance and yield numerous physical and aesthetic problems including foul taste and odour of the water, the disruption and possible curtailment of recreational activities, and the proliferation of undesirable biological species which tolerate such polluted conditions. Numerous other problems occur when the lake is used as a source of water supply; clogging of intakes, the necessity for an increase in filter backwashing frequency, as well as dictating a need for coagulation and clarification facilities, (Black and Khettry (1980)). Eutrophication affects the life-cycles of all levels of the biotic population present in lakes including

benthos, phytoplankton, zooplankton, macrophytes, and fish. As nutrient concentrations in a given lake increase, the relative concentrations of the biological populations may shift and possibly alter the numbers of species present.

## 1.2 Phosphorus as the Limiting Nutrient

The three major nutrients required for biological growth are carbon, nitrogen, and phosphorus. To control the growth of certain undesirable fresh water species, the limiting nutrient must first be identified. Subsequent control of that nutrient input will curtail the rapid growth of such species.

Control of carbon is a well established practice via biological wastewater treatment. It is impossible however to curtail the aquatic input of carbon to the extent that it will become the limiting nutrient. Even if this could be achieved, several species of aquatic plants can utilize the bicarbonate ion to sustain their photosynthetic function, which would make this practice futile (Black and Khettry (1980)).

Algae require nitrogen in a combined inorganic form for growth. In many instances, controlling the input of this nutrient to maintain low algal populations to a given lake or water body is irrational. For example, blue-green algae are capable of fixing dissolved nitrogen gas. The supply of this gas in such an open system is virtually infinite because of its abundant availability from the atmosphere. Blue-green algae therefore are able to thrive under "nitrogen limiting"

conditions due to the presence of phosphorus. These algae are a particular nuisance in terms of producing unpleasant taste and odors in drinking water. Unsightly suspended algal growth may also be manifested (Black and Khettry (1980)).

This leaves the control of phosphorus as the only other means of curtailing eutrophication of natural water bodies. By limiting the phosphorus content in the aquatic environment, the phosphorus:nitrogen ratio would be reduced. Consequently, phosphorus would be rapidly assimilated in the growth process of the green algae, to the extent that eventually the existence of the nitrogen-fixing blue-green algae would be threatened, and hopefully brought under control (Black and Khettry (1980)).

Phosphorus is gradually being recognized as the limiting nutrient in the majority of eurtophication situations around the world. Porter (1975) reports that in most temperate lakes of the United States, phosphorus is limiting. Black and Khettry (1980) cite numerous examples of basins in Canada where the control of phosphorus is rapidly being considered as the only solution to the escalating rates of entrophication. Examples of these basins include: Lake Erie, Ontario; Qu'Appelle River Basin, Saskatchewan; Saint John River Basin, New Brunswick; and the Okanagan Basin of British Columbia. The central regions of South Africa, where fresh water is in critically short supply, are also potential candidates for eutrophication control by proper phosphorus management techniques.

## 1.3 Control of Phosphorus

This section will discuss naturally occurring phosphorus in the aquatic environment and typical raw sewage phosphorus constituents. Reasons for employing point source control of phosphorus and methods whereby this control can be realized will also be given.

### 1.3.1 Phosphorus Naturally Occurring in the Aquatic Environment

A practically infinite reserve of phosphorus-containing minerals is located in the earth's crust. Most of these however, are only to a very low degree soluble in water. This has resulted in relatively minor concentrations of phosphorus derived from such sources in "natural" water.

### 1.3.2 Sources of Phosphorus Input to the Aquatic Environment

The input of phosphorus to a water body may be roughly divided into two forms: the soluble and the particulate. Particulate P may be biologically incorporated into the cell protoplasm of aquatic organisms, or may be adsorbed onto particulate material. This form of phosphorus may enhance the growth of the rooted aquatic vegetation in a lake environment. Soluble P is the most readily available form of phosphorus for uptake by the aquatic population. In this capacity, soluble P acts as a fertilizer, eventually contributing to the problems associated with eutrophication. Black and Khettry (1980) listed a summary of phosphorus sources, which upon addition to a

natural water body may cause eutrophication. These sources are presented in Table 1.1.

### 1.3.3 Phosphorus Forms in Wastewater

The P concentrations in wastewater may be classed as total P, particulate P, and soluble P. Orthophosphates are the major constituent of the soluble P fraction and are the most readily available form. Inorganic condensed phosphates are largely derived from the detergent input into municipal wastewater. A further subdivision may also be made, based on chemical reactivity and biological availability. Soluble reactive phosphorus, nonapatite inorganic phosphorus, and certain forms of organic phosphorus are of note since they have the greatest potential for being utilized by the biological community in the wastewater (Williams et al.(1976)).

### 1.3.4 Point Source Control of Phosphorus

From Table 1.1, although it is evident that many non-point sources contribute to the P input to a natural water body, these are diffuse, difficult to distinguish or classify, and for the most part their control would require a monumental effort. To reduce the input of phosphorus from these sources, a concious effort must be directed at proper management of fertilizer application to agricultural land, and in general controlling the extent of man's activities particularly in terms of extensive urbanization. Point sources, on the other hand, are much more

TABLE 1.1

POSSIBLE SOURCES OF PHOSPHORUS INPUT TO THE AQUATIC ENVIRONMENT

(according to Black and Khettry (1980))

| | |
|---|---|
| i. | Municipal waste treatment plant effluents and private waste disposal |
| ii. | Untreated industrial wastes |
| iii. | Chemicals derived from water treatment |
| iv. | Surface runoff |
| v. | Groundwater |
| vi. | Derived from physical, chemical, and biological reactions inherent in the aquatic environment |
| vii. | Atmosphere |

easily identified and control can be based on site-specific needs. A comprehensive study by the U.S. Environmental Protection Agency (EPA) indicated that point source nutrient loadings are in a relatively concentrated form and are the most readily controlable (Gakstatter et al. (1978)). The management of point source waste, in terms of controlling the phosphorus input to the environment derived from man's actvities, is at present the considered practical method.

### 1.3.5 Methods of Point Source Phosphorus Control

Depending on the effluent regulations in effect for a given point source discharge, one of three major treatment methods may be selected: physical, chemical, or biological. Combinations of these may also be used to suit the requirements of different projects.

#### 1.3.5.1 Physical Methods of Phosphorus Control

Primary sedimentation and clarification usually results in up to 10% removal of the total P content from raw sewage. This amount essentially represents the insoluble P fraction in wastewater. Because of these low percentage removals, solely physical sedimentation methods are not successful and must be accompanied by either chemical or biological treatment to yield significant P removal efficiencies (in the 80% removal range).

1.3.5.2 Chemical Methods of Phosphorus Control

Active removal of P from wastewater in Canada was encouraged in 1969 by the International Joint Commission (IJC) which recommended that P discharges into the Great Lakes be reduced to the lowest practical level. In 1972 a further specific effluent objective of 1 mg/L total P was set, leading to extensive research in the field of physical-chemical P removal schemes in Canada, and particularly in Ontario. Precipitating phosphates with certain chemicals, followed by coagulation and subsequent sedimentation is a proven method of obtaining extremely low effluent P levels. Three chemicals are commonly used for this purpose: lime, alum, and ferric chloride. The aim of these compounds is to provide cations for the precipitation with phosphates. Waste materials from various industial operations, upon initial investigation, appear to be economically beneficial as alternatives to purchased chemicals. Problems with such materials include their fluctuating chemical composition and the presence of impurities such as trace metals (affecting sludge disposal methods). Their availability may be inconsistent in terms of quantities and their transportability from the industry to the wastewater treatment site.

In a primary treatment plant such chemicals are usually added to a preaeration unit at the head end. In the case of secondary treatment, enhanced P removal is typically realized either by lime addition to a preaeration unit prior to primary clarification, or by alum or ferric chloride addition directly

to the biological reactor, with subsequent sedimentation occurring in the secondary clarifier. Jar tests have proven excellent indicators of full scale precipitation techniques.

1.3.5.3 Biological Methods of Phosphorus Control

In standard rate activated sludge treatment receiving primary effluent, between 1.0 and 1.5 mg/L of phosphorus is removed per 200 mg/L of COD removed (Environmental Protection Agency (1976)). This translates to a reduction in P of 20% to 30% across the plant for a typical raw sewage P content of 8 mg/L (Metcalf and Eddy (1979)). Unfortunately such removals are generally inadequate since influent P concentrations are usually high; thus effluent levels below 1 mg/L of phosphorus are seldom realized.

For this reason new approaches have recently been initiated in the area of biological treatment; these approaches aim at enhanced phosphorus removal. Basically, the majority of new research is concerned with the inclusion of stressful conditions which lead to subsequent uptake of P in amounts greater than are required for ordinary microbial growth. These methods of enhanced biological removal of P are discussed in detail in Chapter 2.

# CHAPTER 2

## LITERATURE BACKGROUND

The realization that natural water eutrophication can often be effectively controlled or at least curtailed by maintaining low P concentrations in sewage treatment plant effluents, has lead to extensive research in this field. This chapter will discuss the results of various studies conducted on bench, pilot, and full-scale schemes designed for the enhanced biological removal of P from municipal wastewater. The mechanisms of removal, as proposed by various researchers, will be presented. Case studies, together with their findings, will be given as examples of the research programs being conducted. Because of the variation in the local characteristics and constituents of raw sewage, as well as the seemingly inexhaustible combination of treatment systems available to the researcher, comparison of results is difficult. Also, conclusions derived from these studies are case specific and do not easily lend themselves to universal application and acceptance. It is "generally" agreed that certain conditions must be met in a biological treatment scheme to yield significant levels of P removal. These will be dealt with in Section 2.3 of this chapter. Finally, the rationale for choosing the treatment mode employed in this study will be presented, with reference to the work of other researchers.

## 2.1 Mechanisms of Phosphate Uptake

There are three schools of thought regarding the method whereby P is removed in a biological treatment unit where no chemical addition is practised; these include normal assimilation of P by the biological solids, chemical precipitation, and luxury uptake.

### 2.1.1 Normal Assimilation Mechanism

Proponents of this form of P uptake maintain that P removed via a given biological treatment scheme is due to the assimilation of P (in the various forms in which it is present in the wastewater substrate - See Section 1.3.3) by the microorganisms in the system. The amount assimilated is dependent on the growth requirements of the biomass (Hall and Engelbrecht (1967), Sherrard and Shroeder (1972)).

Conventional activated sludge treatment is capable of removing 20 to 30 % of the influent P, typically yielding approximately 3% phosphorus in the sludge on a dry weight basis (Menar and Jenkins (1970), Levin and Shapiro (1965)). Morgan and Fruh (1972) reported 2.5% P content in sludge using activated sludge batch experiments with pH adjusted to preclude the possible precipitation of calcium phosphate.

Several researchers have found that the P content of sludges can be significantly greater than the above reported levels. Vacker et al. (1967) found up to 7% P concentrated in

the biomass (on a dry weight basis) from a full scale activated sludge plant at San Antonio, Texas. Studies of modified activated sludge schemes, incorporating nitrification and/or denitrification processes, have yielded up to 10% P in the sludge (on a day weight basis) (Simpkins and McLaren (1978), Timmerman (1979)). Among others, these results have prompted further research of the P removal mechanism.

### 2.1.2 Chemical Precipitation Mechanism

The chemical precipitation hypothesis proposes the formation of an insoluble calcium phosphate precipitate on the activated sludge floc during the aeration process and subsequent entrapment of the precipitate in the zoogleal mass. Calcium in sewage may be in the form of calcium ions, particulate salts complexed with fatty acids, and particulate or dissolved complexes with phosphorus. Menar and Jenkins (1970) proposed that activated sludge treatment encourages the increase of both orthophosphate and calcium ions. During activated sludge treatment, phosphates are hydrolyzed to orthophosphates, thus rendering the bound calcium free for the subsequent precipitation process. Additional calcium is obtained from the microbial degradation of fatty acid salts of calcium. Release of P in the clarifier is proposed to be the result of the dissolution of the precipitate in the clarifier unit due to carbon dioxide accumulation and a subsequently lower pH (Menar and Jenkins (1970)). Supporters of this mechanism propose the removal of P is via the raising of pH due to carbon dioxide

stripping in the aeration basin which, in schemes designed for the enhanced removal of P, is usually at an above average dissolved oxygen level. Addition of an alkaline solution to the contents of the aeration basin has also lead to excellent P removals (Menar and Jenkins (1970)). Several researchers disagree with this claim however. For example, Barnard (1975), did not find any correlation of pH with the phosphates in the effluent. The fluctuations in pH due to $CO_2$ stripping in the aerobic unit were incidental in Barnard's study. Thus the pH may not have reached a high enough level for the manifestation of an increase in the precipitation of phosphates by the aerobic biomass.

### 2.1.3 Luxury Uptake Mechanism

The luxury uptake of P is defined as occurring when microbial growth is arrested due to the lack of a nutrient other than P and/or the lack of an energy source. In activated sludge systems all nutrients are in abundant supply and organism growth is controlled by the input of energy sources (Fuhs and Chen (1975)). Researchers in this area have modified this definition to include stress conditions (such as anaerobiosis) as also being capable of producing growth limiting conditions.

From experiments on activated sludge samples, Fuhs and Chen (1975) concluded that P stored in the sludge is in the form of an acid-soluble granular polyphosphate (sometimes also referred to as an acid soluble fraction). Inhibition of biological

phosphate accumulation with 2,4-dinitrophenol yielded results which supported the biological luxury uptake mechanism. A single organism or several closely related forms of the genus Acinetobacter were isolated and found capable of storing highly packed mineral polyphosphates. The authors further determined that these microorganisms grow in abundance under anaerobic conditions. Their results indicate that the inclusion of an anaerobic basin at the head end of an activated sludge treatment scheme would aid the growth of facultative anaerobic bacteria. The function of these organisms would be to produce an energy source in the form of low-molecular weight intermediates for the P accumulating microorganisms (for example, Acinetobacter ). The conventional activated sludge process typically consists of obligate aerobes which are unable to produce the intermediate compounds necessary for Acinetobacter growth. Also, Acinetobacter organisms were found incapable of utilizing sugars or polysaccharides and would be under heavy competition for amino acids in the conventional activated sludge process. Thus their development would, in general, be suppressed. Fuhs and Chen (1975) attributed the release of P in the anaerobic stage to the lysing of some Acinetobacter cells.

Timmerman (1979) used staining techniques which indicated the presence of polymetaphosphate compounds (known as volutin granules) in certain bacterial cells. These organisms were large gram-negative cocci containing poly B-hydroxybutyrate (PHB). Phosphate storage was in the acid soluble fraction; chromatography techniques recorded numerous chain lengths of

polyphosphates in the extract. She suggested that for enhanced P uptake to occur, the presence of two organisms is required. The microorganisms in the anaerobic zone would be responsible for the breakdown of organics for the use of a second type of organism (such as Acinetobacter ) active in the aerobic region. The latter organism was postulated to be involved in enhanced P and carbonaceous material removal. Timmerman realized however, that the uptake of organics in the anaerobic zone was not answered by this approach alone. It was further proposed that bacteria capable of containing high concentrations of P are able to transport "food" across their membranes in an anaerobic environment to satisfy energy and cell synthesis reactions. The organics which constitute this "food" are then assimilated in the aerobic environment at which point P uptake also occurs.

The presence of PHB was also identified in the anaerobic zone by Osborn et al. (1979). Staining methods showed a decreased presence of polyphosphates stored in volutin granules in the anaerobic phase. PHB storage was not exhibited in the aerobic zone, while poly P storage was enhanced by aerobic conditions. Research by Osborn and Nicholls (1977), also considered the mechanism of P uptake and the mode of storage of the P within the bacterial cell. It was noted that Acinetobacter was found in all treatment plants in the Johannesburg, S.A. area which incorporated an anaerobic zone at the head end of the process. This observation was supported by Osborn et al. (1979) who reported large concentrations of Acinetobacter in sludges from Johannesburg S.A treatment plants. Acinetobacter was noted

by Venter et al. (1978) to be present in greatest concentrations with schemes exhibiting enhanced P uptake. The exclusive attribution of P uptake to the Acinetobacter genus, was questioned by Osborn and Nicholls (1977) in that anaerobiosis per se, and not fermentation products such as acetic acid, was deemed the important factor in enhanced P removal schemes by these researchers. Also, extended anaerobiosis did not affect the P removal capacity of the biomass once aeration was resumed. Obligate aerobes such as Acinetobacter would find such long anaerobic periods unfavorable, whereas facultative microorganisms would eventually adapt to this adverse environment. Osborn and Nicholls (1977), in their study of a treatment scheme in the anaerobic - anoxic - aerobic configuration, also proposed that facultative denitrifiers were involved in P uptake in the anoxic zone. This zone is generally defined as contaning nitrates but being free of dissolved oxygen. However, this form of uptake was slower than that typical of the aerobic zone, and appeared to cease altogether upon completion of the denitrification process.

Osborn and Nicholls (1977) maintain that several species of bacteria are involved in P uptake via the "polyphosphate overplus" mechanism. This phenomenon, often confused with the luxury uptake mechanism, occurs when microorganisms lacking the supply of P are suddenly placed in a P rich environment (Fuhs and Chen (1975)). The "phosphate overplus" phenomenon is inapplicable to municipal wastewater treatment schemes since the supply of P is generally over-abundant. Osborn and

Nicholls (1977), in conflict with the above statement, defined the role of the aerobic zone as allowing the oxidation of both carbonaceous matter and ammonia to occur, with luxury uptake of P taking place towards the outflow end of this zone, due to nutrient limitation. Upon entering the aerobic region, obligate aerobes (for example, Acinetobacter ) and facultative anaerobes have a disturbed internal P balance, due to the release which theoretically occurred in the anaerobic and/or anoxic zones. This condition stimulates the rapid assimilation of P. Further aeration induces nutrient limiting conditions conducive to luxury uptake. Osborn and Nicholls (1977) also observed that long aerobic HRT's appeared to be beneficial to the enhanced uptake of P. The luxury P uptake mechanism was also supported by Levin and Shapiro (1965), Borchardt and Azad (1967), Connell and Vacker (1967), Barnard (1976), and Yall et al. (1972).

## 2.2 Results and Observations of Selected Phosphorus Removal Studies

The research conducted to date can be divided into two categories: a) that concerned with the observation of enhanced P removal efficiencies in conventional activated sludge schemes (Section 2.2.1) and b) modified activated sludge processes incorporating anaerobic and/or anoxic treatment to purposefully enhance the removal of phosphates from wastewater (Section 2.2.2).

### 2.2.1 Phosphorus Removal in Conventional Activated Sludge Treatment

Vacker et al. (1967) reported up to 96% P removal in full scale conventional activated sludge treatment. These researchers recommended that digester liquors not be recycled through the plant due to high P content. The DO in the aeration basin was maintained at 5 mg/L, which was adequate to preclude P release in the clarifiers (although the liquid phase of the return sludge was reported to contain up to 13 mg-P/L of phosphates). High nitrification rates generally corelated with reduced P uptake in the aerobic unit.

Milbury et al. (1971) investigated the removal of P in a conventional activated sludge scheme. Primary effluent orthophosphate concentrations fluctuated between 7.7 and 9.7 mg-P/L while the secondary effluent P concentrations were below 1 mg-P/L, with removal efficiencies of 89% realized. Orthophosphate release of up to 46 mg-P/L at the head end of the aeration tank was reported. Several operating parameters recommended by these researchers will be discussed in Section 2.3.

Several preliminary batch type studies by Shapiro et al. (1967), on the P release phenomenon, indicated that the oxidation reduction potential (ORP) must be low (<+150 mV) for release to take place. The rate of release was independent of the mixed liquor concentration and appeared to be a reversible process. Further studies showed that P release was also

triggered by the lowering of pH. Phosphorus release, in the form of orthophosphates, was postulated to have been from the acid soluble phosphorus fraction of the sludge mass. To prevent P release in the clarifier, high DO levels in the aerobic unit and rapid sludge removal from the clarifier was recommended.

A review of the literature shows that enhanced P removal in conventional activated sludge is an unreliable occurrence. Many more investigations are needed to catalogue the effects of the variation of the multitude of parameters which play a role in wastewater treatment on the P removal capabilities of such systems.

### 2.2.2 Phosphorus Removal via Modified Activated Sludge Treatment

The latest research in this field has been, to a large degree, a South African endeavour. The critical shortage of water in that country has spurred the need for economical tertiary treatment schemes. The Bardenpho (BARnard-DENitrification-PHosphorous removal) process, developed by J.L. Barnard in the early 1970's, is perhaps the most famous. This process is capable of removing up to 95% of nitrogen, 97% of phosphorus and 90% of carbonaceous material from wastewater via a biological sequence of plug-flow activated sludge reactors; these consist of two alternating anoxic-aerobic units followed by a clarifier. The system is operated in an extended aeration mode.

The original Bardenpho scheme was investigated by Barnard

(1974) on a 100 $m^3/d$ pilot plant, at the Daspoort Sewage Treatment Works in Pretoria, S.A.. The influent COD was 300 to 600 mg/L. Influent orthophosphates were between 5 and 8 mg-P/L. The system yielded an effluent total P content of <1 mg-P/L. There was no problem with the settling of solids in the clarifier. The system SRT was 16 to 18 days. The SRT was then increased to greater than 25 days and a MLSS content of approximately 6000 mg/L was maintained. At these operating conditions, the system yielded poor P uptake, and supersaturation of the sludge with P was blamed. An interconnection between the final aeration basin and the waste sludge tank was in effect during this period however; this may have had deleterious effects as the P rich sludge may have entered the aeration basin, causing excessively high P readings.

Venter et al. (1978) reported enhanced P removal in a full scale Bardenpho plant only when the effluent nitrate levels were below 2 mg-N/L. No significant drop in calcium concentration was found through the plant, thus supporting the purely biological uptake mechanism as opposed to chemical precipitation. Further observations were made on the treatment performance of a modified Bardenpho scheme with acid sludge addition (a source of short-chain organics) at the head end of the process. The results supported the view of Osborn and Nicholls (1977) in that both of these groups found acid sludge addition (designed to stimulate anaerobic conditions at the head end of the scheme) of no benefit. Orthophosphate release of up to 100 mg-P/L was reported by Venter et al. (1978) at the head (anaerobic) end of

the process. The sludge filtrate of the scheme, which did not employ acid sludge addition, contained up to 700 mg/L of acetic acid. Prolonged periods of anaerobiosis were found not to be optimal for the removal of P through the plant. Enhanced P removal was accompanied by low nitrification rates in the aerobic zone. Effluent nitrate levels below 5 mg-N/L were found to yield a system capable of removing the greatest amount of P. These researchers deemed necessary the inclusion of an anaerobic zone in a P removing treatment scheme.

Subsequent to his initial research, Barnard proposed the addition of an anaerobic reactor either at the head end of the Bardenpho scheme, to condition the biomass for enhanced P removal (referred to as the Modified Bardenpho Process), or simply prior to an aerobic basin in which nitrification is suppressed. Both these schemes were termed Phoredox by Barnard (1976). This principle involved a high sludge recycle rate, since the nitrate content in the sludge of such systems would be typically low. The sludge return was to be mixed with the feed at the head end of the treatment scheme, and then flow into the anaerobic reactor where phosphorus release would theoretically occur. The mixed liquor would then pass on to either a standard Bardenpho plant for denitrification or a simple aeration basin if nitrification is not a problem. Nicholls (1975) confirmed the effectiveness of the Phoredox principle during full scale operation at the Alexandra Plant (S.A.). Waste sludge was allowed to go anaerobic and was recirculated to the head end of the aeration basin. The mixed liquor dissolved P content dropped

to 0.3 mg-P/L. Some re-solution of P occurred in the clarifier as a result of accumulating sludge. The system hydraulic retention time was 27 hours.

Pilot scale studies by Simpkins (1979) on the modified Bardenpho scheme, indicated that raw sewage COD levels, although higher than that of settled sewage, did not yield appreciably higher P removals. During the period of study the $\Delta P/\Delta COD$ ratios were 0.011 and 0.017 for the raw and settled influent phases respectively. These ratios are lower than those typically corresponding to enhanced P removal. Ratios of $\Delta P/\Delta COD$ of approximately 0.020 yielded greater than 86% P removal in the study of Simpkins and McLaren (1978).

Osborn et al. (1979) reported that the enhanced removal of P corresponded to high influent COD levels in their investigation of the Goudkoppies Works, Johannesburg (S.A.), operated in the modified Bardenpho mode. These researchers claimed that high nitrate concentrations did not always yield poor process P removal efficiencies. High DO in the return sludge appeared to lower the degree of anaerobiosis in the anaerobic zone. Up to 24 mg-P/L release was reported and it was noted that high release rates corresponded to a low sludge return flow. Phosphorus release was observed when effluent nitrate concentrations were below 1 mg-N/L, and was found to be a prerequisite for P removal. The extent of anaerobiosis in the scheme studied by Osborn et al. (1979) was questionable since typical anaerobic reactor ORP values were reported as +35 mV.

The primary anoxic zone was found to be involved both in denitrification and P uptake by these researchers. It was proposed that denitrifying bacteria also function in a P removal capacity. Filamentous organisms proliferated when the aerated fraction of the system was reduced, yielding poor sludge settling characteristics. Subjecting the entire treatment plant to anaerobiosis failed to eliminate these types of organisms. Poor clarifier sludge settling was blamed on overaeration in the secondary aeration basin, as well as on high nitrates (causing denitrification with subsequent sludge rising problems).

Dew (1979) studied the modified Bardenpho scheme under temperature variability. His results indicate P removal is not a function of temperature and obtained an overall average P removal of 89% (approximately 6.7 mg/L total P removal based on the average influent total P concentration of 7.5 mg/L). The SRT was typically long, as required by nitrogen removing systems, ranging from 15 days to 98 days.

Simpkins and McLaren (1978) also investigated a modified Bardenpho pilot plant. With an SRT of 15 days, P removal was greater than 86%, except when a high mixed liquor recycle rate was in effect. The luxury uptake mechanism was supported and it was calculated that the high P removals could not have been explained solely by the physical-chemical precipitation theory of Menar and Jenkins (1970). The anaerobic release of P was found to vary with the influent COD concentration. Ratios of $\Delta P/\Delta COD$ from 0.014 to 0.021 were reported, based on the total P

content of unfiltered feed and orthophosphate concentrations in filtered effluent samples. The rate of P uptake in the aerobic zone was directly proportional to the rate of release in the anaerobic stage. Uptake occurred in the initial anoxic stage and tended to compete with the aerobic stage for P. Little fluctuation of pH was observed. The MLVSS:MLSS ratio was 0.70 during periods of enhanced P removal by the pilot plant of that study. A ratio of 0.85 was obtained at the VCSD Water Reclamation Plant, San Ramon, California where phosphorus uptake was typical of activated sludge treatment (Menar and Jenkins (1970)). The average COD of the feed varied from 250 to 315 mg/L with COD percentage removals exceeding 90%. Effluent nitrate levels were typically less than 5 mg-N/L during periods of significant P removal. Although the testing schedule used in this research was extensive, (24 hour composite samples were analysed daily), and a large data base was compiled, fluctuations were a problem.

Pitman (1980) reports effluent orthophosphate results from three South African sewage treatment plants. Data from the Alexandra and Olifantsvlei plants in the Johannesburg (S.A.) area, indicate that at comparable influent COD levels (240 to 820 mg/L), and a similar operating scheme (a lack of aeration at the head end of both treatment plants), enhanced P removal occurred only at the Olifantsvlei plant where the SRT was 15 to 20 days. The Alexandra plant SRT was 11 to 15 days. Limited data is also presented for the Goudkoppies Bardenpho scheme operated at a SRT of between 20 and 22 days. Effluent nitrate levels were

below 4 mg/L and influent COD ranged from 320 to 350 mg/L. Phosphorus removals in the range of 60% were reported during the short sampling period. The data given does not distinguish filtered from unfiltered samples. Pitman (1980), also studied the settling characteristics of sludges, when >20% of a plant consists of anaerobic and/or anoxic zones. The reaserch results yielded the conclusion that bulking in the clarifier is a possible occurrence due to the active growth of filamentous organisms.

Jank et al. (1978) achieved high P removal efficiencies in a pilot system operated in a denitrification-nitrification mode, including mixed liquor recycle and methanol addition. The SRT's studied were: 5, 10, and 23 days at 7.5, 15, and 25$^{o}$C respectively. A system was also investigated without methanol addition and operated at slightly lower SRT's of 4, 8, and 18 days. These authors determined that enhanced P removal occurred in schemes which employed denitrification at the head end of the system, coupled with a high influent COD and periods of high COD percentage removals. Although enhanced P removal was reported at high $\Delta P/\Delta COD$ ratios, the systems investigated did not vary to an appreciable degree, in terms of their removal capabilities, to warrant such an observation. In fact, at similar operating conditions, a greater $\Delta P/\Delta COD$ ratio was realized with the scheme which did not fare as well in terms of effluent P concentrations than did the set-up which included the addition of methanol. These researchers also found that the variation in the P content of the sludge was negligible at SRT's ranging from approximately

3 to 24 days. An anaerobic stage at the head end of a treatment scheme was found to have conflicting effects on the capability of the subsequent aerobic stage to remove high percentages of P.

Marais (1979a) maintained that the Bardenpho scheme is not optimal for the removal of P because of its dependency on low TKN:COD ratios. He proposed an anaerobic-anoxic-aerobic sequence incorporating both anoxic mixed liquor recycle to the anaerobic reactor and the "conventional" aerobic to anoxic recycle. Sludge return was to the anoxic reactor. This scheme was cited as having a greater flexibility than the Bardenpho in terms of P removal in that the TKN/COD fluctuations in the influent did not disrupt the efficiency of P removed by the process. Maximum anaerobic sludge mass fractions were also presented as guidelines to prevent inefficient nitrification in such schemes. Efficiencies of P removal by the process were not presented however.

MacLaren and Wood (1976) studied the P release phenomenon using a laboratory treatment model in an anaerobic-anoxic-aerobic configuration. A SRT of 15 days was maintained at an operating temperature of 20$^{o}$C. An average of 11.4 mg-P/L of orthophosphates was recorded in the anaerobic unit, which was only slightly higher than the feed concentration. P uptake in the aerobic reactor began immediately, yielding orthophosphate concentrations of 0.1 mg-P/L in this unit. The effluent orthophosphates were reported as 0.4 mg-P/L, indicating slight release in the clarifier. The

$\Delta P/\Delta COD$ ratio during laboratory testing was 0.019. Further pilot plant studies by these researchers, on a modified Bardenpho scheme yielded no P release when the HRT of the anaerobic reactor was maintained at 2 hours. Increasing the anaerobic basin HRT to 4 hours caused release of up to 4 mg-P/L and subsequent uptake to levels typically below 1 mg/L total P. The operating SRT was 25 days. A drop of SRT to 15 days did not have any detrimental effects in terms of P removal. The removal of the first anoxic basin from the system, as well as returning the anaerobic reactor to a 1 hour HRT, did not cause a reduction in the P removal efficiency. Further batch experiments by these researchers supported the biological or luxury P uptake mode. Phosphorus was removed in the presence of nitrates and aeration. Population selection was cited as the reason for the lack of P removal in the initial pilot plant studies. This conclusion conflicts with the results reported for the bench scale studies conducted by McLaren and Wood (1976), where immediate P removal was observed. Fuhs and Chen (1975) also supported the requirement for an acclimation period to encourage the accumulation of P removing microorganisms.

Osborn and Nicholls (1977) investigated the operation of a pilot plant in an anaerobic-anoxic-aerobic configuration. The system SRT was 25 days. Pilot scale studies yielded no improvement in P removal when acid sludge was added to the treatment process at the anaerobic stage. Addition of acetic acid to the anoxic zone was also found unadvantageous. An anaerobic zone was deemed indispenable for the development of

volutin granules in bacteria. A comprehensive list of volutin forming bacteria was presented in that report. High P removal was associated with high MLSS levels, since this condition led to a greater concentration of volutin containing bacteria. Osborn and Nicholls (1977) agreed that nitrates must be absent from the anaerobic zone for enhanced P uptake across the plant to occur. It was proposed that this condition would minimize oxygen availability to the denitrifying species causing the release of P through catalysis by the enzyme polyphosphatase, thus allowing the activation of the enzyme polyphosphate kinase in P-storing bacteria for subsequent uptake of P in the aerobic reactor. A maximum nitrate level of 2 mg-N/L in the effluent from the aeration basin was suggested, which could be achieved with sufficient MLSS recycle to the anoxic reactor. Increasing the anaerobic HRT lead to increased P levels in this zone. Periods greater than 2 hours were suggested as having possible detrimental effects on the growth of volutin forming bacteria, which the general consensus is, are either strict or facultative aerobes. Release of 5 mg-P/L was found to be adequate for subsequent enhanced P uptake in the aerobic reactor.

Fuhs and Chen (1975) could not induce P uptake without previous P release under anaerobic conditions. The anaerobic phase per se may not be responsible for the triggering of the P release mechanism; high $CO_2$ concentrations and a low pH were suggested as important factors. They reported that P release could be induced by bubbling of carbon dioxide through the mixed liquor of the anaerobic reactor. However, Osborn and Nicholls

(1977) reported that once this "forced" release occurred, subsequent uptake under aerobic conditions was not realized. High DO in the influent to the clarifier was recommended by Osborn and Nicholls (1977) to avoid anaerobiosis, and hence P release, from developing in the sludge blanket. However, aerobic HRT's less than 4 hours were recommended to discourage the continuation of the respiration processes in the clarifier, which were reported to cause the release of P.

Levin (1972), proposed a plug-flow unit (known as the Pho-strip process) which incorporated principles similar to those proposed by Barnard in the Phoredox process, except that the clarifier underflow was thickened in Levin's operation. Thickening induced anaerobiosis, causing P to be released into the supernatant which was decanted and underwent lime precipitation. The thickened anaerobic sludge was then returned to the head end of an aeration basin. Barnard (1976) claimed that the essential part of this stripping operation is the creation of anaerobic conditions in the sludge and proposed that appropriate sludge wasting would remove the P from the system. Vacker et al. (1967), Milbury et al. (1971), and Garber (1972), also agreed that stripping is an unnecessary operation. Levin's results indicate that of the P removed, only 25 % is removed from the system via stripping; the remaining 75% being removed in the waste sludge.

Timmerman (1979) reported on the studies undertaken by Air Products and Chemicals, Inc., using laboratory scale units

operated in the anaerobic-aerobic configuration. Phosphorus release greater than 13 mg-P/L was noted in the anaerobic reactor when the nominal HRT in this unit was maintained at 1.0 hour. Release was found to be a prerequisite for P uptake. Nitrates in the oxic unit were not found to be detrimental to the P uptake mechanism. Only slight release of P occurred in the clarifier. Fluctuations in influent P concentrations did not alter the high efficiency of removal, once this had been established (approximately 14 weeks after start-up). Due to this lengthy period at the beginning of the experiments, where P removal was not in effect, the author suggested that ample time must be allowed for the development of the necessary P removing organisms. Biological uptake was supported by a low MLVSS:MLSS ratio (in the 0.70 range), indicating that a large portion of the sludge was inorganic. The system SRT was not reported.

## 2.3 Operating Conditions Required for P Removal

This section will list the conditions which have become widely accepted as required for the occurance of enhanced P removal in modified activated sludge treatment schemes:

(a) Plug flow conditions are one prerequisite. Barnard (1976) recommends a length to width ratio of 20:1. Milbury et al. (1971) obtained P removals in the 50% range with a bench scale activated sludge unit employing the plug flow mode. The effectiveness of this mode is also supported with observations by Vacker et al. (1967), Levin et al. (1972), and Garber (1972).

(b) All basins must be completely mixed (Barnard (1976)).

(c) The dissolved oxygen in the final aeration basin, prior to clarification must be greater than is typical for conventional activated sludge treatment. Generally 3 to 4 mg/L are recommended to avoid P release in the clarifier unit due to the onset of anaerobiosis in the sludge blanket (Vacker et al. (1967), Milbury et al. (1971), Levin et al. (1972), Garber (1972), and Barnard (1976)). High dissolved oxygen in the final aeration basin also causes carbon dioxide stripping from the mixed liquor, thus increasing the pH and encouraging the possible precipitation of P out of solution (Menar and Jenkins (1970)). Barnard (1976) however, using a plug flow compartamentalized aerated unit, reports no improvement in the P removal efficiency when a lime slurry was added to increase the pH of the mixed liquor to 8.5.

The settling charactristics in the clarifier are also enhanced by high dissolved oxygen levels influent to this unit in that denitrification is less likely to occur, thus minimizing the problem of rising sludge (Barnard (1976)).

Dissolved oxygen control is in general more difficult with bench scale units than in full scale operations. Small fluctuations in the compressed air supply tend to have significant effects on the former (Barnard (1976)).

(d) Nitirification should be minimized as it contributes to high nitrate concentrations being returned to the anaerobic

reactor, thus reducing the degree of P removal. Excellent P removal during periods of low nitrate concentrations in the effluent was confirmed by Barnard (1976). It must be noted that while poor phosphorus removal efficiencies were manifested simultaneously with a high nitrification rate, feed flow fluctuations and excessive dissolved oxygen levels in the aerobic basin were also a norm. A high nitrate content in this basin was coupled with a low P removal efficiency. It appears that other forms of nitrogen (for example, ammonia) have no effect on P removal (Barnard (1976)). In full scale conventional activated sludge treatment, nitrification activity is, in general, insignificant since sludge recycle ratios are in the order of 0.25 (Vacker et al. (1967), Milbury et al. (1971), Levin et al. (1972), and Garber (1972)). Also, the feed to full scale operations often arrives in a septic state, encouraging the reaction of nitrates in the return sludge with reduced substances in the feed. Of course, fluctuations in the degree of anaerobiosis of the feed would trigger erratic P removal efficiencies in a given system.

(e) Reducing conditions are required at the head end of the treatment scheme. Anaerobiosis has been found to yield the release of P which is a prerequisite of P removal (Milbury et al. (1971)). These researchers also claimed that P uptake was a relatively slow process. Barnard (1976) reported that P release is not always a consequence of anaerobic conditions. He suggested that the oxidation reduction potential (ORP) may play an important role in the P release phenomenon and that at some

minimum ORP level P release occurs. Care must be exercised not to reach such a low ORP that sulphates are reduced, thus yielding odor problems (Barnard (1976)). Barnard's research indicates that the release of P leads to subsequent P removal in the aerobic zone (Barnard (1974)). Complete anaerobiosis appears to be essential such that the appropriate stress conditions are manifested. For this reason the inflow of nitrates to the anaerobic zone must be curtailed in order that the ORP is maintained at a sufficiently low level.

Milbury et al. (1971) obtained P concentrations of 30 mg-P/L in filtered samples from the anaerobic end of the treatment scheme studied. Fuhs and Chen (1975) stated that the anaerobic phase per se did not necessarily stimulate P release; the accumulation of carbon dioxide and the resulting low pH were cited as possibly having an influence on the release phenomenon. It was suggested, by these researchers, that the anaerobic stage may be required for the production of an appropriate carbon source (by facultative anaerobes) which is required for the growth of P-accumulating organisms. P release in the anaerobic stage is a possible result of lysing of the P rich microorganisms under these stressful conditions (Fuhs and Chen (1975)).

(f) The return sludge should be mixed with the feed (raw or settled sewage) at the head end of the plant to encourage a high oxygen demand, thus improved anaerobic conditions (Vacker et al. (1967), Milbury et al. (1971), Garber (1972), Levin et al.

(1972), and Barnard (1976).

(g) The buildup of solids in the clarifier must be avoided to prevent P release, resulting from anaerobic conditions of the sludge (Barnard (1974)).

(h) The supernatant from the sludge digesters should not be returned to the treatment plant because of the high phosphorus concentration usually present therein (Vacker et al. (1967), Milbury et al. (1971), Garber (1972), Yall et al. (1972), and Levin et al. (1972)).

(i) Garber (1972) proposed that a relatively high C:P ratio is required for the establishement of enhanced P removal conditions. When the influent carbon content is high, a high oxygen demand is exerted at the head end of the treatment scheme, thus allowing anaerobiosis to be established rapidly and to a higher degree than with a low influent carbon content.

## 2.4 Research Rationale - This Work

Based on the principles discussed earlier in this chapter, a system "designed" for the enhanced removal of P was built and studied at the Environmental Engineering Laboratory at the University of British Columbia. Initially the model consisted of two reactors (anaerobic followed by aerobic). The theory regarding the P removal capabilities of this system was based to a great extent on the Phoredox principle described by Barnard (1976). Essentially, anaerobic conditions were to be created at

the influent end of the system, followed by a period of aerobic uptake. Both the sewage feed, and the sludge return were to enter the system via the anaerobic zone. Nitrification was to be curtailed by maintaining a low sludge age. The system was then to be monitored while operational changes were implemented.

The aim of this study was to make certain operational changes to the system, observe the results in terms of P removal, and make subsequent modifications to improve the efficiency of this removal. Chapter 3 will deal with the details of the physical and operational characteristics of the system. The results obtained will be discussed in Chapters 4 and 5. Conclusions and recommendations derived from this study are presented in Chapter 6.

# CHAPTER 3

# EXPERIMENTAL SYSTEM AND PROCEDURES

This chapter describes the physical and operational characteristics of the system used in this study, and presents the sampling procedures, and sample analysis techniques used. System start-up, monitoring procedures, and various modifications are discussed.

## 3.1 Physical System Characteristics

The system was initially designed for the efficient removal of phosphates from municipal wastewater, based on information obtained from other research (See Chapter 2). Two environmentally controlled rooms were employed; one was maintained at $4^{o}$ C to facilitate lengthy periods of raw sewage storage (Section 3.2.1), and another to house the treatment system at temperatures which could be independently controlled.

Initially the treatment scheme consisted of two reactors (an anaerobic followed by an aerobic) and a clarifier. A third sludge conditioning reactor was added at a later date for reasons discussed in Section 4.2.2. The original clarifier proved to be inefficient in terms of suspended solids removal and was substituted for by another model half way through the study (Section 4.2.3).

All components were built of clear 0.6 cm perspex; their dimensions are given in Table 3.1. The rectangular reactors were fitted with a vertical perspex plate, 2 cm from the outflow wall with a 12 cm clearance from the reactor bottom to the lowest edge of the baffle. The initial clarifier consisted of a cylinder with an inverted cone forming the base. Sludge was withdrawn throught a 0.5 cm opening in the apex of this cone. Inflow to the clarifier was through a 2.5 cm vertical cylinder which extended into the clarifier, to enhance solids settling characteristics. The second "improved" clarifier was an inclined cylinder with a flat bottom. Sludge withdrawal was via a 0.5 cm opening at the lowest point.

## 3.2 System Operation

### 3.2.1 Raw Sewage Supply

Approximately 450 litres of fresh, unsettled raw sewage was obtained bimonthly during the course of the study. The initial source was the Lulu Island Treatment Plant of the Greater Vancouver Regional District, for reasons recommended by Dew (1979). An analysis of raw sewage from the Maple Ridge Treatment Plant (Maple Ridge, B.C.) however, indicated much lower trace metals concentration than Lulu Island samples. On this basis the Maple Ridge source was opted for after the initial 450 litres had been used in the acclimation process.

The raw sewage feed was stored in two 225 litre, stainless

TABLE 3.1

SYSTEM DIMENSIONS

| | LENGTH (cm) | WIDTH (cm) | HEIGHT (cm) | EXPERIMENTAL VOLUME (L) |
|---|---|---|---|---|
| Anaerobic | 32.1 | 9.8 | 31.8 | 5.9 |
| Aerobic | 55.9 | 9.8 | 31.8 | 12.4 |
| Sludge Conditioning | 24.1 | 9.8 | 31.8 | 4.6 |

| | DIAMETER (cm) | VOLUME (L) |
|---|---|---|
| Initial Clarifier | 12.7 | 4.5 |
| Substituted Clarifier | 6.4 | 2.6 |

steel drums at 4°C (Figure 3.1). Synthetic sewage was used to raise the feed COD as required. (Synthetic sewage components are listed in Appendix I).

### 3.2.2 System Hydraulics

#### 3.2.2.1 Flow Regime

The feed was pumped to the system, located in the second environmental room, out of one drum at a time. The feed line passed through a water bath to bring the feed temperature up to that of the system (Figure 3.2). The feed then entered the anaerobic reactor (Figure 3.3); flowed by gravity to the aerobic reactor (Figure 3.4), and on to the clarifier (Figures 3.5 and 3.6). Initially the clarifier underflow was pumped to the anaerobic reactor. Upon the installation of the sludge conditioning reactor however (Figure 3.7), the sludge was returned to this reactor and the mixed contents flowed by gravity to the anaerobic reactor. At the beginning of the study sludge wasting was from the sludge return line. However; due to inconsistent sludge suspended solids concentrations, the wasting procedure was modified (after one and half months) such that waste mixed liquor was pumped out of the aerobic reactor. Figures 3.8 and 3.9 show schematics of the flow pattern through the system without and with the sludge conditioning reactor respectively. The change in clarifiers is also presented in these figures.

FIGURE 3.1: FEED STORAGE DRUMS

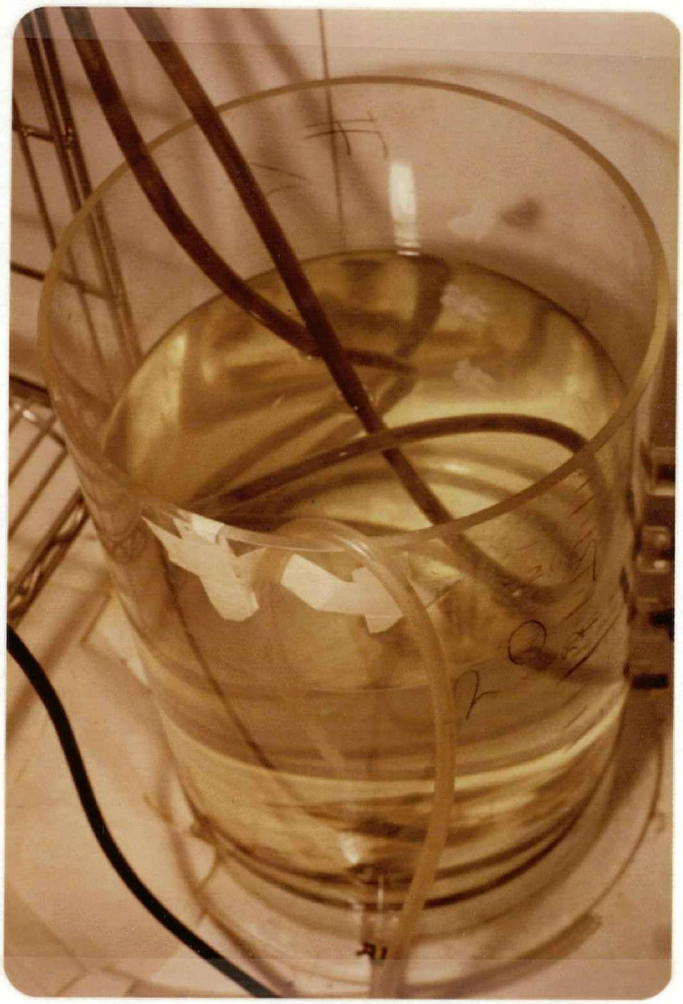


FIGURE 3.2: WATER BATH

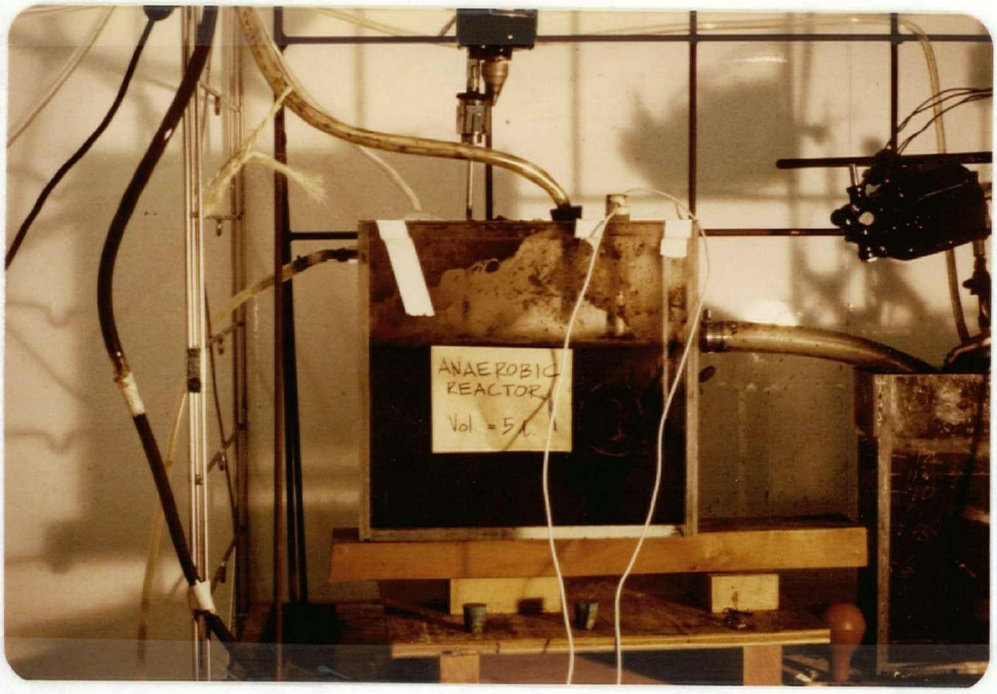



FIGURE 3.3: ANAEROBIC REACTOR

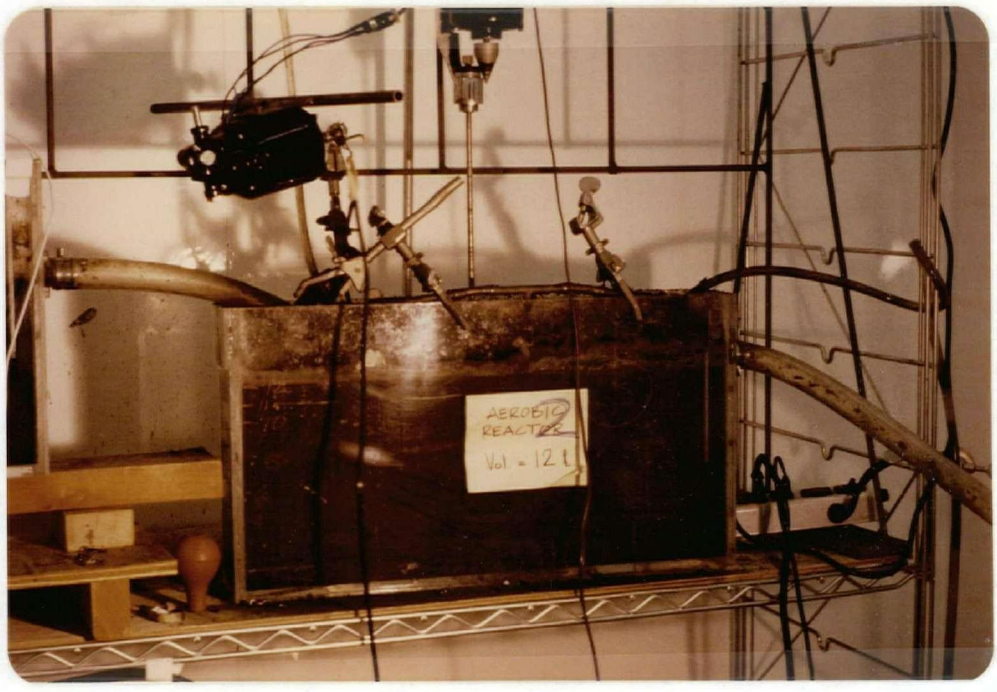



FIGURE 3.4: AEROBIC REACTOR

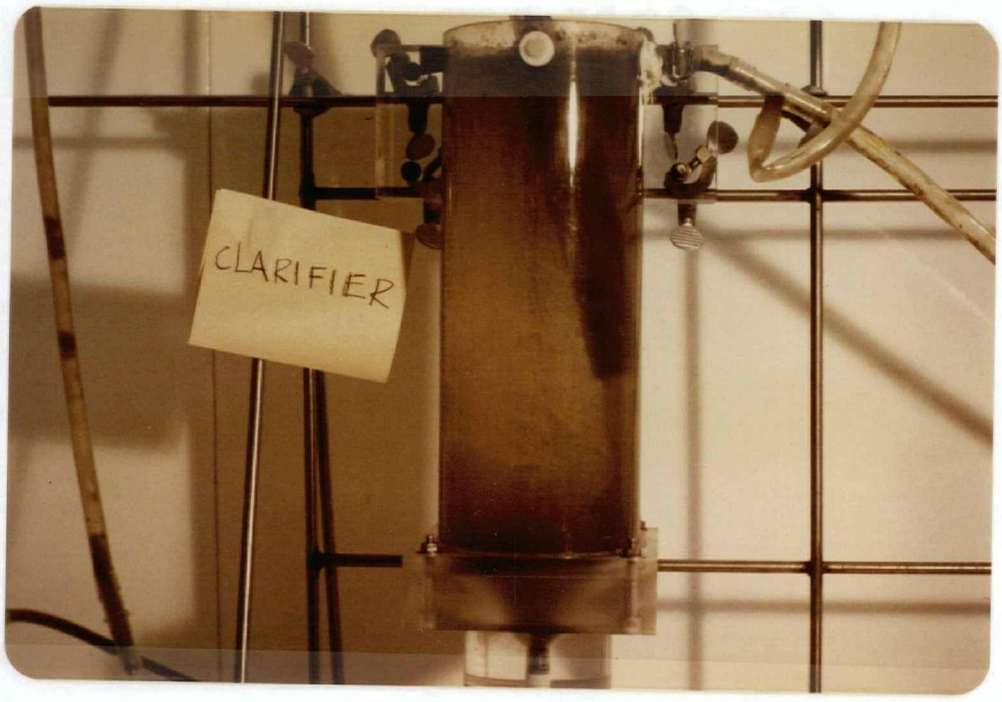



FIGURE 3.5: CLARIFIER (INITIAL MODEL)

FIGURE 3.6: CLARIFIER (SUBSTITUTED MODEL)

FIGURE 3.7: SLUDGE CONDITIONING REACTOR

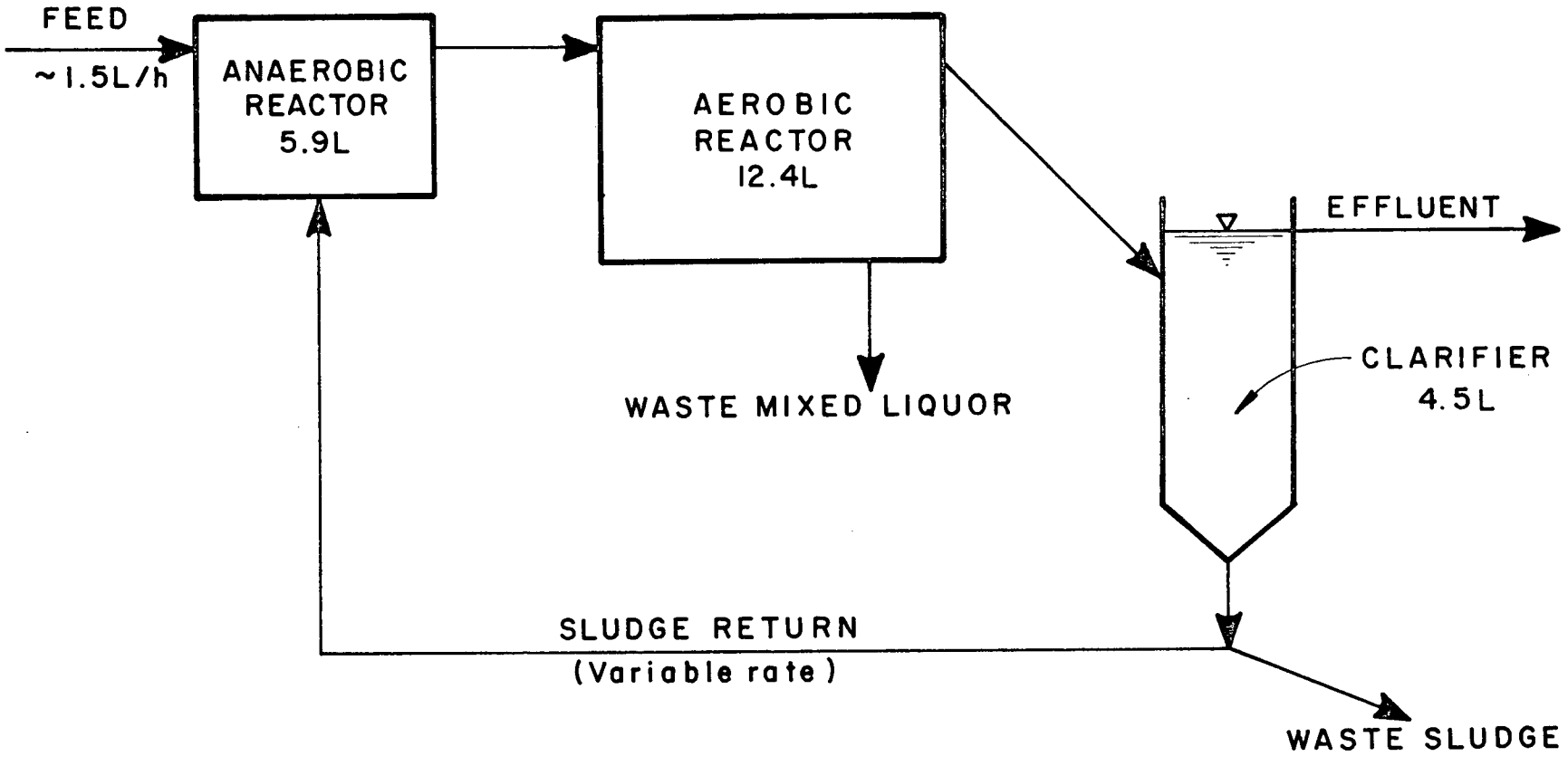



FIG. 3.8: INITIAL PHOSPHORUS REMOVAL SYSTEM.

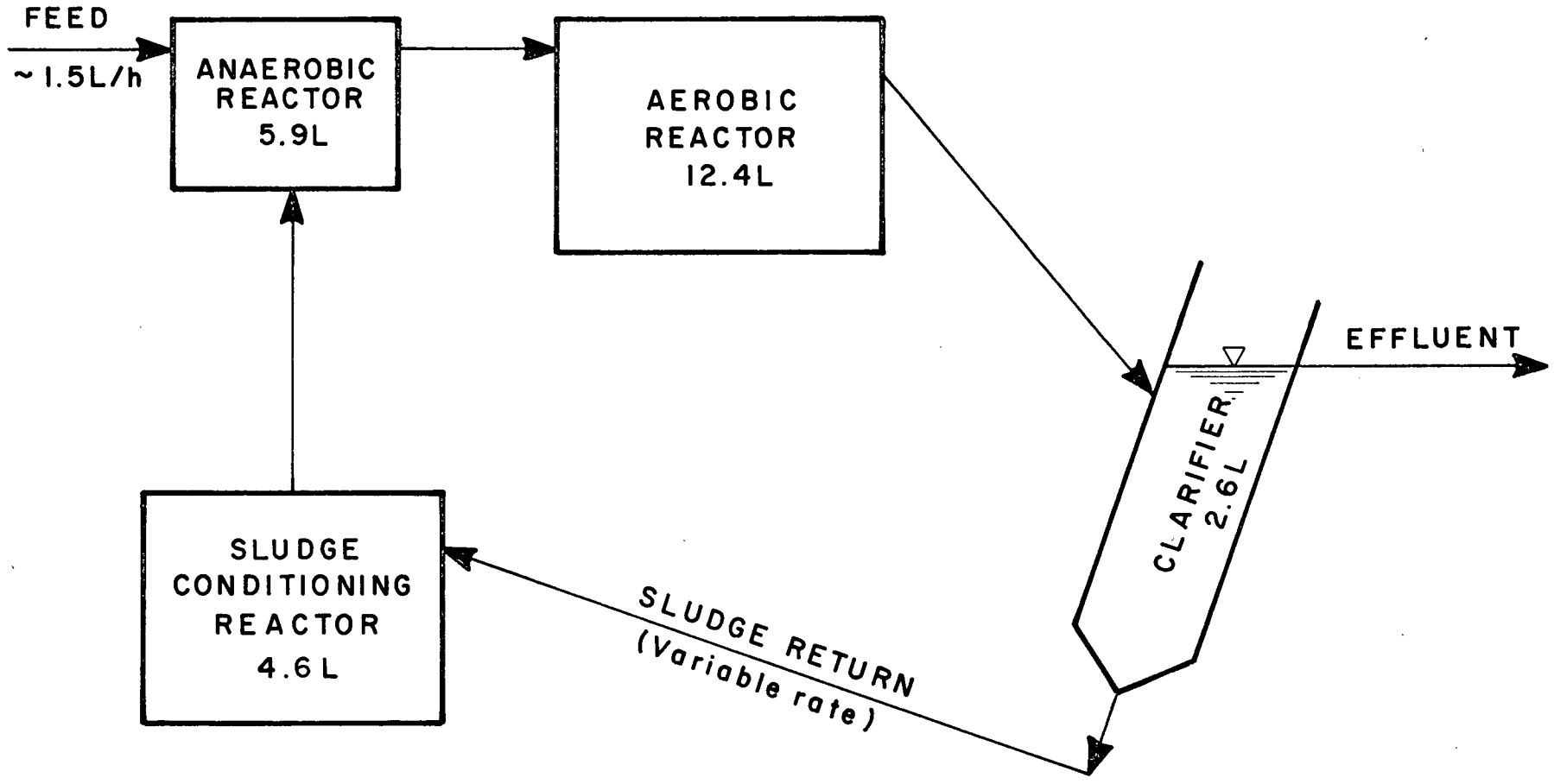



FIG. 3.9: MODIFIED PHOSPHORUS REMOVAL SYSTEM.

### 3.2.2.2 Pumping Aspects

Cole-Parmer variable speed peristaltic pumps were employed, together with Cole-Parmer silicone tubing, type 6411. About 50 cm of this tubing was used within the "pump head" and extended to the outside of the head where a transition to larger diameter (up to 2.5 cm) Tygon tubing was made to minimize blockage of the lines by large particles.

Flow control was via Eagle Signal Flexopause Timers, each of which was associated with one pump (the feed, recycle and wasting pumps). Many, but short, ON - OFF cycles were used to yield a "continuous" flow system, while at the same time minimizing the deposition of solids in the tubing by maintaining a higher flow rate. ON-OFF cycles were in the order of 10 to 20 cycles per hour, with typical ON times ranging from 20 to 40 seconds.

### 3.2.3 Mixing

Each reactor was continuously mixed with a stainless steel propellor powered by a Sargent-Welch cone drive stirrer. Mixing was vigorous, in order to prevent the settling of solids. The storage drums were also mixed continuously when in use. Some solids settling still occurred in these drums however, since restricted access precluded the use of a larger propellor.

### 3.2.4 Nitrogen Supply

The anaerobic reactor was covered by a fitted perspex cover to ensure low dissolved oxygen levels in the mixed liquor. Nitrogen gas was continuously bubbled through the mixed liquor of the anaerobic reactor. This gas is inert and maintained a positive pressure in the reactor, preventing atmospheric air from entering, thus aiding in keeping the reactor contents anaerobic. In full scale treatment this practice would not be required since anaerobiosis would be easily acheived (and maintained) due to the higher volume to surface ratio than is inherent in laboratory scale units.

### 3.2.5 Aeration

Three Fisher Gas Dispersion Tubes with fritted glass cylinders connected, via Tygon tubing, to the building compressed air supply, provided fine bubble aeration to the aerobic reactor mixed liquor. The air flow rate was adjusted to maintain the required dissolved oxygen level (See Section 4.3.4 and 5.2.4) using a pressure regulator and a needle valve. A water trap was provided on line to avoid air pressure fluctuations.

### 3.2.6 Light

The system was operated in darkness except during sampling, routine daily maintenance, and inspection periods.

## 3.3 Sampling Procedures

In general samples were all "grab", except the effluent samples; in this case 6 hour composites were taken, to account for suspended solids fluctuations that occurred.

Sample pre-treatment and preparation conformed to the individual analysis requirements of each of the tests performed as discussed in Section 3.4. Analysis was done immediately upon sampling in all cases.

The raw sewage was sampled upon arrival at the laboratory. The feed to the system (raw plus synthetic sewage) was sampled at the inlet to the anaerobic reactor. Samples of the mixed liquor of each of the reactors were taken at the outlet ends of the reactors. The effluent was collected in a 22.5 litre container via an outlet line very near the effluent exit point from the clarifier (to avoid any problems of settling occurring in the effluent line and subsequent possible erroneous results). Except for suspended solids, effluent concentrations were assumed to equal the return sludge levels for dissolved impurities. Table 3.2 presents the frequency and type of analysis performed during the course of this study, as well as indicating sampling points.

TABLE 3.2

SAMPLING FREQUENCY

| PARAMETER | NO. OF SAMPLES PER WEEK | SAMPLING POINTS |
|---|---|---|
| Alkalinity | 1 | ALL* |
| pH | 1 | ALL* |
| Phosphorus | | |
| Orthophosphates | 3 | ALL* |
| Total Phosphates | 1 | Feed, effluent |
| Nitrogen | | |
| Nitrates | 1 | ALL* |
| Nitrites | 1 | ALL* |
| Suspended Solids | | |
| MLSS | 7 | All reactors, effluent |
| Oxygen | | |
| BOD | 2 | Feed, effluent |
| COD | 2 | Feed, effluent |
| DO | Continuous | Aerobic reactor |
| DO | Irregular | Anaerobic reactor, sludge conditioning reactor |
| Oxidation Reduction Potential | Continuous | Anaerobic reactor |
| Temperature | Continuous | Air temperature |
| Flow Rates | 2 | Feed, sludge return |

*ALL = Feed, anaerobic reactor, aerobic reactor, sludge conditioning reactor, and effluent.

## 3.4 Analytical Procedures

### 3.4.1 Chemical Oxygen Demand ( COD )

The chemical oxygen demand was measured regularly during this study, since its removal in a given process can be directly related to energy and cell growth. The analysis of this parameter was opted for, rather than BOD or TOC, because of its relative reliability and simplicity. In addition, a quick visual indication of the levels of carbonaceous material present is afforded by this test. The amount of "available" substrate was thus obtained in this study. The determination of COD followed the requirements outlined in Standard Methods (1976). Unfiltered samples of raw sewage were analyzed immediately upon arrival at the laboratory, to aid in the proportioning of the synthetic sewage to the large storage drums. A feed COD of between 500 and 700 mg/L was to have been maintained ideally, but large fluctuations occurred due to solids settling in the storage drums (as a result of inadequate mixing (Section 3.2.3)). Feed samples were not filtered prior to analysis. The effluent was filtered through a No. 4 Whatman paper filter.

### 3.4.2 Total Suspended Solids ( TSS )

Feed, mixed liquor (from each reactor), and effluent TSS determinations were in accordance with Standard Methods (1976) for "Total Nonfilterable Residue".

### 3.4.3 Mixed Liquor Volatile Suspended Solids ( MLVSS )

The MLVSS results were obtained by ashing the mixed liquor samples on which TSS values had been previously obtained. The procedure is outlined in Standard Methods (1976) under the heading "Total Volatile Residue".

### 3.4.4 Alkalinity

All samples were unfiltered. Titration was performed with 0.02N $H_2SO_4$ to a endpoint pH of 4.8, as suggested in Standard Methods (1976), for expected alkalinities in the range of 150 mg/L as $CaCO_3$.

### 3.4.5 pH

The pH values were obtained at the time of alkalinity measurements prior to titration. Glass electrodes (VanLab 34106-002), in conjunction with a saturated calomel electrode (Fisher Calomel Reference Electrode 13-639-51), were used. Electrode calibration was with pH 7.0 and 4.0 buffer solutions. Readings were taken with a Fisher Accument Model 310 pH meter.

### 3.4.6 Trace Metals

The determination of trace metals followed the requirements of the Methods for Chemical Analysis of Water and Wastes (EPA (1979)) using a Jarrell Ash 810 Atomic Absorption Spectrophotometer.

### 3.4.7 Nitrogen Forms

#### 3.4.7.1 Nitrates and Nitrites

Both of these nitrogen forms were determined in duplicate on the supernatant of centrifuged samples, filtered through a 0.45 μm membrane filter and appropriately diluted. Nitrate plus nitrite concentrations were obtained in the presence of a cadmium reduction column, followed by nitrite only determination without the column. Nitrate levels were then calculated by subtraction. The Technicon Auto Analyzer II was used and the procedure followed was recommended by the Technicon Industrial Method 100-70W. All values were determined in mg-N/L.

A comparison of nitrate results was attempted with duplicates being tested both by the Auto Analyzer method and by ultraviolet spectrophotometry using a Pye Unicam SP8-100 Ultraviolet Spectrophotometer as per Standard Methods (1976). No correlation of results could be derived between these two methods, possibly due to phosphate interference in the Auto Analyzer (Olson (1980)), and dissolved organic matter interference in the ultraviolet determination. It was apparent that the former type of interference would not be as significant as that of the ultraviolet (since the cadmium column was well maintained). For this reason the latter method was not pursued further.

3.4.7.2 Total Kjeldahl Nitrogen ( TKN )

The analysis of samples for TKN concentrations was performed as outlined in Standard Methods (1976). This included preliminary distillation, followed by acidimetric titration with 0.02N $H_2SO_4$. Results are given in mg-N/L and are the sum of the organic and ammonia nitrogen forms.

3.4.8 Phosphate Forms

3.4.8.1 Orthophosphates

The procedure followed for orthophosphate determinations is outlined in Standard Methods (1976) "Stannous Chloride Method ". Standards were prepared on each day of sampling. A Bausch and Lomb Spectronic 88 was used to obtain absorption readings. Feed samples were unfiltered yielding total orthophosphate values. All other samples (mixed liquor of each reactor and effluent) were centrifuged and the supernatant was subsequently filtered through a 0.45 µm membrane filter, to give filterable orthophosphates. All values are given as mg-P/L and are based on the analysis of duplicate samples.

3.4.8.2 Total Phosphates

Feed samples for total phosphate analysis were unfiltered, while the effluent was first centrifuged and the supernatant was filtered through a 0.45 µm membrane filter. Mixed liquor total phosphates of the three reactors were not determined. The

procedure is given in Standard Methods (1976). Persulfate digestion was followed by total phosphate determination via the Stannous Chloride Method, to give results in mg-P/L. As was the case with orthophosphate determinations, fresh standards were used and absorption readings were obtained via a Bausch and Lomb Spectronic 88.

## 3.5 Monitoring Procedures

This section will discuss several parameters which were monitored during this study.

### 3.5.1 Temperature

Mixed liquor temperature was monitored with a mercury thermometer. The temperature of the environmentally controlled room was then adjusted to maintain the required liquid temperature.

### 3.5.2 Dissolved Oxygen ( DO )

The aerobic reactor DO was monitored continuously for the duration of this research work. A Yellow Springs Instrument Company submersible DO probe (Model No. 5740) and Model 54 meter were used. Readings were recorded on a chart recorder. To prevent membrane fouling, as well as aiding "fresh" liquid to come in contact with the membrane, the probe was shaken continuously via a cam mechanisn (Figure 3.10).

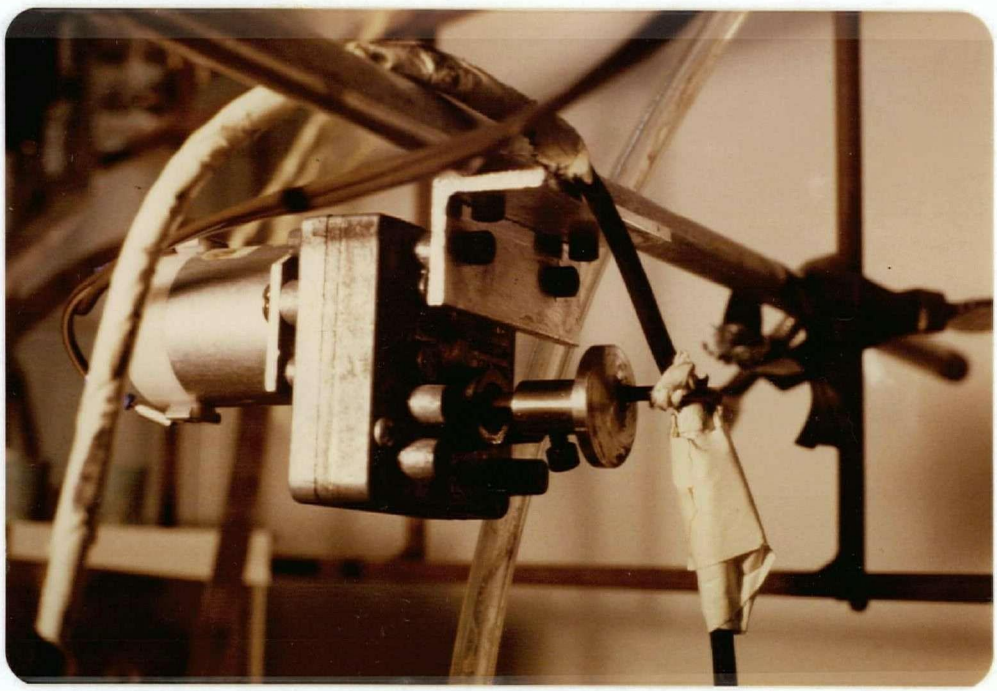


FIGURE 3.10: DISSOLVED OXYGEN PROBE AGITATION MECHANISM

The probe was calibrated (typically once weekly) against a tap water sample whose DO concentration had been determined following the Azide Modification of the Iodometric Method, as described in Standard Methods (1976). The anaerobic and sludge conditioning reactor DO concentrations were usually measured subsequent to the probe standardization procedure. The DO of the sludge in the clarifier was also determined sporadically.

### 3.5.3 Oxidation Reduction Potential ( ORP )

The ORP of the anaerobic reactor mixed liquor was monitored continuously. ORP measurements were via a Fisher platinum wire probe (No. 13-639-115), referenced to a saturated calomel electrode (as was used in pH determinations). Both electrodes were washed thoroughly and the reference electrode was refilled with saturated potassium chloride solution once weekly. Standardization procedures were also followed on these occasions as outlined by the probe manufacturers. Results were recorded using a chart recorder calibrated in negative millivolt (-mV) units.

### 3.5.4 Flow Rates

Both feed and sludge return flow rates were generally monitored twice weekly or when flows were purposely altered. The amount of fluid (feed or return sludge) delivered into a graduated cylinder per pumping cycle, timed with a stopwatch, was determined and reported as L/h.

## 3.6 Hydraulic Retention Time (HRT)

Reactor volumes were constant throughout this study. The feed flow rate was ideally maintained at approximately 1.5 L/h; a rate which provided an appropriate F/M ratio (Section 4.1), as well as being reasonable in terms of the bimonthly volume of raw sewage transported to the laboratory. The solids fluctuations in the feed flow dictated that the HRT be adjusted during certain periods however. For example, high concentrations of solids in the feed yielded a high influent COD, thus necessitating a reduction in the feed flow to maintain a reasonable F/M ratio. This, in turn, resulted in longer liquid retention in the system. The sludge return flow rate was altered according to the clarifier performance and mixed liquor suspended solids requirements of the reactors.

Two forms of HRT are presented in Table 3.3. The nominal values are based on the feed rate only (Equation 3.1), while actual HRT's include the effect of the sludge return flow (Equation 3.2).

$$\text{Nominal HRT (h)} = \frac{\text{Reactor Volume (L)}}{\text{Feed Flow Rate (L/h)}} \qquad (3.1)$$

$$\text{Actual HRT (h)} = \frac{\text{Reactor Volume (L)}}{\text{(Feed + Sludge Return) Flow Rate (L/h)}} \qquad (3.2)$$

TABLE 3.3

HYDRAULIC RETENTION TIME

| DAY | ANAEROBIC REACTOR (hours) | | AEROBIC REACTOR (hours) | | SLUDGE CONDITIONING REACTOR (hours)* | | SYSTEM (hours) excluding clarifier | |
|---|---|---|---|---|---|---|---|---|
| | Nominal | Actual | Nominal | Actual | Nominal | Actual | Nominal | Actual |
| 1A-183A | | | | | | | | |
| Range | 3-11 | 1-3 | 6-14 | 3-7 | 4 | 1-3 | 13-29 | 5-13 |
| Typical Value | 4 | 2 | 8 | 4 | 4 | 2 | 16 | 8 |
| 184A-68B | | | | | | | | |
| Range | 4-6 | 1-4 | 8-12 | 3-9 | 4-5 | 1-4 | 16-23 | 5-17 |
| Typical Value | 4 | 3 | 9 | 6 | 4 | 3 | 17 | 12 |

*Sludge Conditioning Reactor was only in operation from Day 127A.

## 3.7 Solids Retention Time ( SRT )

Since a comprehensive data base of solids concentrations in each reactor and the effluent was developed, a predetermined design SRT could be maintained by daily wasting of an appropriate amount of microorganisms (initially from the sludge return line and later from the aerobic reactor). Sample volumes withdrawn for analysis, formed a significant portion of solids removed from the system, and were accounted for in the calculation of SRT. Poor clarifier performance necessitated the collection of large composite samples of the effluent for suspended solids analysis. It was felt that this practice yielded a better estimate of the suspended solids content than provided by grab type samples.

Two forms of SRT will be discussed in this work: the aerobic SRT and the system SRT. Sample calculations for these are given in Equations 3.3 and 3.4 respectively.

$$\text{Aerobic SRT (d)} = \frac{\text{Aerobic Reactor Suspended Solids (mg)}}{\text{Suspended Solids Wasted from System (mg/d)}} \quad (3.3)$$

$$\text{System SRT (d)} = \frac{\text{Mass of Suspended Solids in All Reactors (mg)}}{\text{Suspended Solids Wasted from System (mg/d)}} \quad (3.4)$$

The effluent suspended solids values were averaged over each individual period for which SRT's are given. All significantly large samples removed for analysis were included in the SRT calculations.

Both aerobic and system SRT's are presented chronologically in all the figures in Chapters 4 and 5. The SRT's quoted are only estimates however, since only at the beginning of the study, when SRT's were short, was enough time allowed for complete system equilibrium to be achieved. At least one sludge age, after a change in operation (for example, a temperature change) should be maintained as a rule of thumb for reaching steady state conditions in a given treatment system. Gaps in the SRT data indicate periods during which system operation was being altered.

## 3.8 Nitrate Reaction Rates

Nitrate reaction rates were calculated as per Equation 3.5. These rates were based on actual flow rates and considered the mass of nitrates removed daily via intentional wasting and sampling. When nitrate reaction rates were positive, denitrification occurred in a given reactor, while negative rates indicated nitrification (See Section 4.3.8.1(b), and 5.2.8.1(b)).

$$\text{Nitrate Reaction Rate (mg-N/g MLSS/h)} = \frac{\text{Nitrate (In-Out) (mg-N/h)}}{\text{Reactor Volume} \times \text{Reactor MLSS (L.g.L}^{-1}\text{)}} \qquad (3.5)$$

## 3.9 Orthophosphate Reaction Rates

Equation 3.6 shows the method by which the orthophosphate reaction rates, (as reported and discussed in Section 4.3.9.1(c) and 5.2.9.1(c)) were calculated. The sludge return flow was included in the determination of these rates (in that actual flows were considered), as were the daily wastage and sample volumes removed from the system. Orthophosphate uptake was indicated by postive reaction rates while release was denoted by negative values.

$$\text{Orthophosphate Reaction Rate (mg-P/g MLSS/h)} = \frac{\text{Orthophosphate (In-Out) (mg-P/h)}}{\text{Reactor Volume} \times \text{Reactor MLSS } (L.g.L^{-1})} \quad (3.6)$$

## 3.10 $\Delta P/\Delta COD$ Ratio

Several researchers have referred to the $\Delta P/\Delta COD$ ratio as indicative of the expected P removal capabilities of a given treatment system. This ratio was determined in this study by considering the change between the unfiltered feed and filtered aerobic reactor mixed liquor total phosphate concentrations. Since only the orthophosphates were analytically determined for the aerobic reactor, the conversion to total phosphates was based on the assumption that the orthophosphate: total phosphate ratio calculated for the effluent was also the same in the aerobic reactor. The change in COD was obtained by calculating the difference between the influent to and the effluent from the system. Note that due to laboratory analysis scheduling, only

those $\Delta P/\Delta COD$ ratios were calculated for which phosphates and COD determinations were within a day or so of each other.

$$\Delta P/\Delta COD = \frac{\text{Total P (Influent - Aerobic Reactor) (mg/L)}}{\text{COD (Influent - Effluent) (mg/L)}} \qquad (3.7)$$

## 3.11 Research Program

The system was operated over a period of 10 months. The operational scheme was divided into two phases. Phase A is discussed in detail in Chapter 4 and is concerned with the start-up of the system and the subsequent 8 months of system modifications intended to yield satisfactory phosphorus removal. Such modifications included the improvement of clarifier design, the incorporation of a sludge conditioning reactor and liquid temperature changes. Numerous operational and design problems will be discussed and results of this period presented. A further two months of study, Phase B, involved the achievement of enhanced phosphorus removal and the necessary prerequisites for this to occur. This second phase is discussed in Chapter 5.

# CHAPTER 4

# PHASE A: A PRELIMINARY INVESTIGATION OF PARAMETERS AFFECTING THE BIOLOGICAL REMOVAL OF PHOSPHATES FROM WASTEWATER

## 4.1 Description of System Operating Conditions

This chapter describes the operating conditions of the system at startup, the subsequent modifications, and results obtained during Phase A of the study. Phase A spans the initial 238 days of system operation. The physical and operational characteristics of the system were discussed in Chapter 3. For reasons derived from other research (Chapter 2), enhanced P removal was expected and intensive testing was to be undertaken once this phenomenon was observed. Several major modifications of the sysytem were necessary however, as experience dictated, causing numerous disruptions in the testing schedule depicted in Table 3.2.

The initial raw sewage source was the Lulu Island Treatment Plant; but because of high metal concentrations found therein, subsequent feed sewage was obtained from the Maple Ridge Treatment Plant. Both the anaerobic and aerobic reactors were seeded with secondary sludge from the Squamish Treatment Plant to encourage the rapid development of the biomass. Approximately one month was allowed for system stabilization, with only the mixed liquor solids being actively monitored during this period. To gain experience with the testing procedures and to overcome

any sampling and analysis problems, preliminary tests were also performed on several parameters such as COD, nitrates, nitrites, total Kjeldahl nitrogen, and ortho and total phosphates. Sampling points were identified and methods were developed such that the sampling and analysis techniques described in Chapter 3 could be closely adhered to during subsequent months of system operation.

The initial design parameters were developed on the assumption that the anaerobic reactor biomass did not participate in the assimilation of organics. To avoid the recirculation of formed nitrates from the clarifier to the anaerobic reactor, an aerobic sludge age of < 5 days (excluding the clarifier) was maintained initially. At the influent flow rate of 1.5 L/h and an anticipated $BOD_5$ of approximately 200 mg/L, the design F/M ratio was 0.6 kg $BOD_5$ applied/kg MLVSS/d. The initial Volumetric Loading was 0.7 kg $BOD_5$ applied/$m^3$/d.

## 4.2 System Modifications

The effluent P levels did not approach the anticipated low concentration of < 1 mg/L P during the initial days of the study. Several modifications of the physical system and operating conditions were undertaken during Phase A in an effort to fulfill the objective of the successful removal of P. The following modifications were implemented, the effects of which will be discussed in Section 4.3.

### 4.2.1 Temperature

Initially the liquid temperature in the reactors was maintained at 18°C. Although the sludge age was low, a thriving nitrifier population was established in the aerobic reactor during the first two months of the study. Research by Barnard (1976) indicated phosphates could not be removed simultaneously with high nitrate levels in the effluent. Therefore the curtailment of nitrification became a major objective. Since the growth of nitrifying autotrophic bacteria (namely Nitrosomonas and Nitrobacter ) is susceptible to temperature changes, the temperature was lowered on several occasions. This measure was effective in reducing nitrification rates over a short time period only, especially since the aerobic SRT was increased as Phase A proceeded (thus improving the conditions for a thriving nitrifier population to develop).

### 4.2.2 Sludge Conditioning Reactor

Temperature changes alone were ineffective in lowering the nitrification rates in the aerobic reactor for any appreciable length of time. A sludge conditioning reactor was therefore added to the system on Day 127A, with the physical characteristics described in Section 3.1. The aim of this modification was the provision of a basin where denitrification of the return sludge would occur prior to the sludge entering the anaerobic reactor. It thereby ensured that no significant levels of nitrates entered the anaerobic reactor. This measure

precluded any possible interference of returned nitrates with the release of P in this reactor and the subsequent uptake in the aerobic reactor.

### 4.2.3 Clarifier

Numerous problems were encountered with the operation of the system due to poor clarifier performance. Excessive denitrification caused sludge rising to the surface of the clarifier, causing suspended solids to escape with the effluent. The result was large fluctuations in mixed liquor suspended solids in all three reactors due to the return sludge having inconsistent solids levels. On Day 175A, a new clarifier was installed with the characteristics described in Section 3.1. A similar clarifier showed promising settling properties at the University of Cape Town (Marais (1979b)). Unfortunately the problems of poor effluent suspended solids quality prevailed. These will be discussed in more detail in Section 4.3.2.

### 4.2.4 System Shut-down

On Day 160A, the compressed air supply to the aerobic reactor was shut off, stirring was discontinued and all pumping ceased for 12 hours. These measures were undertaken in order to induce complete anaerobiosis of the whole system followed by a reversal to pre-shut-down conditions. Tests were performed on the settled supernatant in each of the units at the end of the shut-down period, as well as shortly after the system was

restarted. Pertinent results will be discussed in the following section.

## 4.3 Results and Discussion

The maximum Phase A percentage removal of phosphates was 55% and that of COD was 96%.

Throughout the presentation and discussion of the results obtained during Phase A, one should bear in mind the three temperature regimes studied (as indicated on each of the following figures). Temperature effects are important from the point of view of the reaction rates for both nitrates and phosphates. Unfortunately, the effect of temperature on these rates could not be developed, due to the substantial frequent variation of other parameters (especially SRT) during the course of this phase. All parameters will be presented based on their chronological variation in the study.

### 4.3.1 Chemical Oxygen Demand ( COD )

Throughout Phase A, influent COD levels varied between 140 and 1650 mg/L. Although synthetic feed spiking of the raw sewage was practised to maintain a feed COD of approximately 600 mg/L, difficulties with stirring the large feed storage drums still caused large COD fluctuations (See Section 3.1.3). Influent and effluent COD values are presented in Appendix II, Figure II-1. Percentage removals, as indicated in Figure 4.1, varied between 55 and 96%. Periods of low removal corresponded to low feed COD.

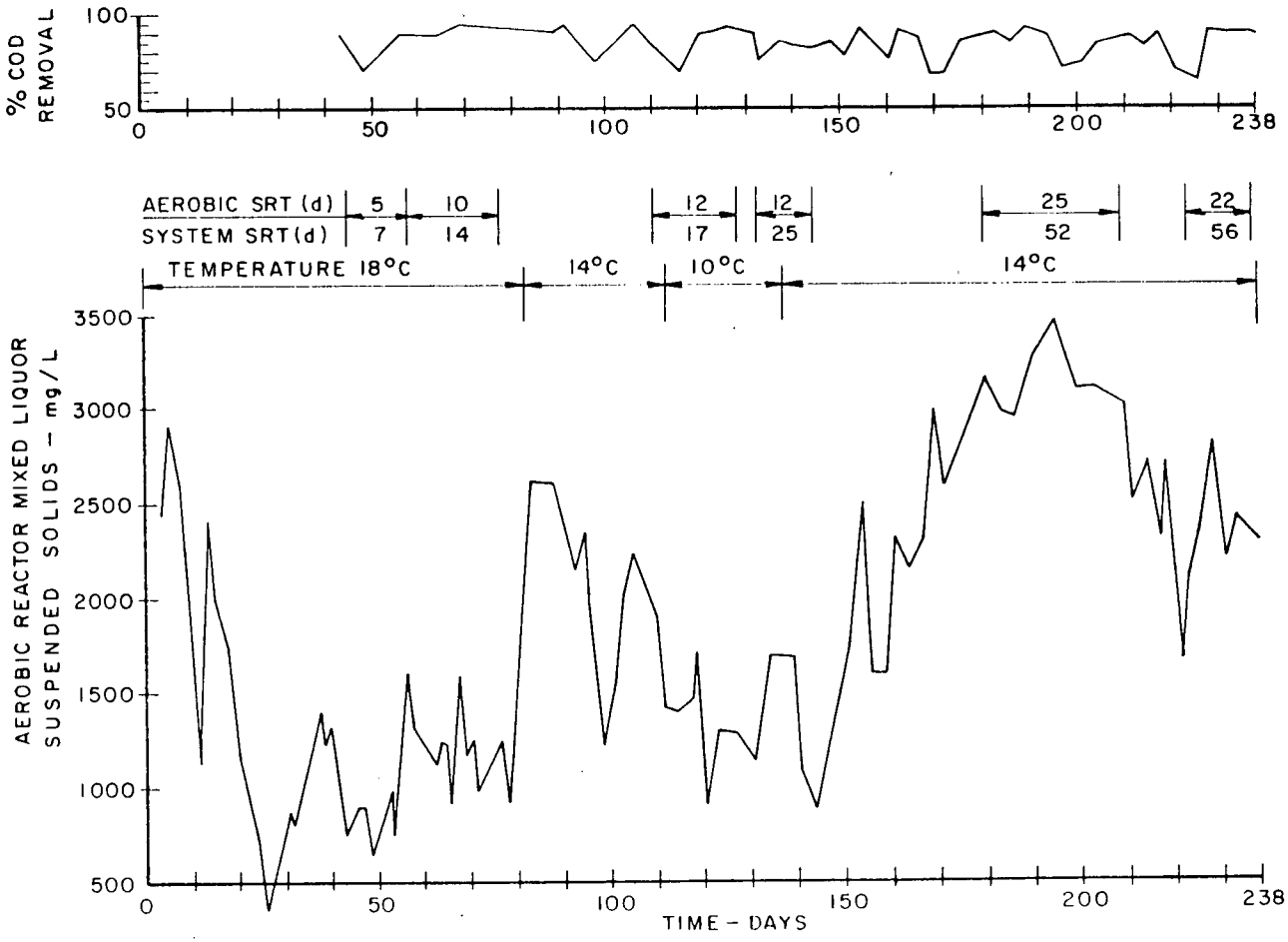



FIG. 4.1: SYSTEM PERCENTAGE COD REMOVAL AND AEROBIC MLSS LEVELS (PHASE A).

### 4.3.2 Total Suspended Solids ( TSS )

Fluctuations in the feed suspended solids content were primarily a result of insufficient stirring of storage drum contents, typically yielding high TSS as the raw sewage level approached the drum bottom (where solids concentrations were higher due to settling).

The clarifier was unable to produce an effluent with acceptably low SS, thus percentage solids removals are not reported. The effluent SS content generally ranged between 6 and 89 mg/L; sometimes exceeding feed SS levels. Due to these fluctuations, effluent suspended solids were monitored only for the purpose of SRT calculations and are reported in Appendix II, Figure II-2. Probable reasons for the poor performance of the clarifier include: scale effects, dispersion of the bioflocs in the aeration tank, and denitrification resulting in rising sludge. The latter is believed to have had the greatest impact on the high effluent SS levels encountered throughout this study. Sludge bridging occurred across the bottom cone of the clarifier, resulting in clear liquid forming part of the sludge return. The accumulating sludge was thus afforded ample time to denitrify and rise. Increasing the sludge return caused clear effluent to be pumped out of the clarifier, while sludge accumulation around the clarifier walls persisted.

Similar problems were encountered upon the installation of the new clarifier. Although wall effects were not as pronounced in the new clarifier itself, denitrification effects were

visible in the flooded influent line, especially at the junction of this line and the clarifier. It is not expected that this problem would prevail in pilot or full scale operations. It is unlikely that the poor sludge settling observed during Phase A of this study may have resulted from the activity of filamentous organisms as described by Pitman (1980). The anaerobic zone alone constituted 31% of the system volume, and the inclusion of the sludge conditioning reactor increased this to 46%. Unlike the observations made by Pitman, a distinct sludge matrix was evident in this study, thus precluding the activity of filamentous organisms. Unfortunately, microscopic examinations of the sludge were beyond the scope of this research work.

The MLSS of the anaerobic reactor closely paralleled the concentrations in the aerobic reactor, therefore only the latter values are presented in Figure 4.1. Upon the installation of the sludge conditioning reactor on Day 127A, the monitoring of the MLSS in this reactor was also initiated. MLSS ranged between 1870 mg/L upon installation, and 7170 mg/L near the end of Phase A. Since the only inflow to this reactor originated from the underflow of the clarifier, the solids concentration in this reactor was highly dependent on the quality of sludge settled in the clarifier. When sludge rising occurred, the sludge return included a large amount of clear effluent. This diluted the solids in the sludge conditioning reactor, with subsequent lowering of solids levels in the anaerobic and aerobic reactors. This unreliable solids concentration in the sludge return was the major reason for the solids fluctuations in all the

reactors, causing additional problems with estimating daily wasting volumes.

### 4.3.3 Mixed Liquor Volatile Suspended Solids ( MLVSS )

The MLVSS typically constituted 87% of MLSS in all three reactors throughout Phase A.

### 4.3.4 Dissolved Oxygen ( DO )

Dissolved oxygen levels were maintained typically higher than required by conventional (1-2 mg/L) activated sludge treatment; in general between 2.0 and 4.0 mg/L. This was a prerequisite for the prevention of clarifier sludge becoming anaerobic, causing the release of P as well as other factors discussed in Section 2.3. At startup, DO control was difficult due to solids fluctuations and problems with diffuser clogging, probe membrane fouling and fluctuations in the laboratory compressed air supply. Subsequent minor operational modifications (such as the provision of a mechanism to agitate the probe) resulted in a reliable DO monitoring program (See Section 3.5.2).

The DO in the anaerobic reactor was always below 0.5 mg/L and typically fluctuated about the 0.1 mg/L level. The anoxic reactor DO was not actively monitored.

Infrequent DO measurements on the settled sludge on the bottom of the clarifier indicated levels in the neighbourhood of

0.3 mg/L, even though the aerobic reactor DO was relatively high.

### 4.3.5 Oxidation Reduction Potential ( ORP )

In general the anaerobic reactor was maintained "truly" anaerobic during Phase A, since the ORP values were for the most part below -130 mV, typically fluctuating between -130 and -520 mV (levels above -130mV were only observed on four occasions during this phase of study). This indicated that negligible dissolved oxygen and nitrates were present at all times. Fluctuations of approximately 30 mV were observed during the ON time of the sludge return pump, when the sludge was recycled to the anaerobic reactor. These fluctuations were eliminated upon the installation of the sludge conditioning reactor.

### 4.3.6 Alkalinity and pH

Alkalinity and pH results are reported in Appendix II Table II-1. The anaerobic reactor pH was up to 0.9 units higher than the pH of the aerobic reactor. This is atypical in that anaerobiosis is, in general, accompanied by a low pH in conventional activated sludge treatment schemes. Beer and Wang (1978) indicate however that endogenous nitrate respiration, although yielding no carbon dioxide, produces bicarbonates which would tend to buffer the pH. This explains the high pH observed in the anaerobic and the sludge conditioning reactors in this study. The anaerobic reactor

exhibited higher than usual pH because the probable high bicarbonate content in the flow from the sludge conditioning reactor afforded a high degree of buffering capacity.

There was no discernable trend in the raising or lowering of pH upon treatment. In general, alkalinity was removed via the treatment scheme with the bulk of the removal occurring in the aerobic reactor (due in part to the nitrification reactions). As expected, low pH paralleled low alkalinity in the aerobic reactor and effluent samples.

### 4.3.7 Trace Metals

The concentrations of trace metals in the feed (not spiked with synthetic sewage) are presented in Appendix II Table II-2. Sampling and digestion were performed immediately upon arrival at the laboratory.

There was no appreciable change in the feed trace metal concentrations throughout Phase A. It is probable that accumulations of these metals in the sludge increased with increasing sludge age. There is also the possibility that a toxicity threshold was reached, but this aspect was not investigated in this study (See Section 5.2.7).

### 4.3.8 Nitrogen Forms

Because results in the literature suggest that nitrates are important in relation to the P release and uptake phenomena,

this form of nitrogen was monitored extensively. Nitrites and total Kjeldahl nitrogen were also monitored as sources of further information regarding the operating performance of the system.

4.3.8.1 Nitrates

(a) Concentrations

Feed nitrate levels were typically <0.02 mg-N/L. Anaerobic reactor concentrations (prior to installing the sludge conditioning reactor) were dependent on the degree of nitrification in the aerobic reactor, reaching a high of 5.65 mg-N/L. During the latter stages of Phase A, the nitrates in the anaerobic reactor were in general <0.1 mg-N/L as a result of the excellent denitrification capabilities of the sludge conditioning reactor.

Due to the purposeful temperature curtailment (that was practised during the first 126 days of the study) of nitrification, the aerobic reactor nitrate levels fluctuated appreciably. Subsequent to installing the sludge conditioning reactor, such fluctuations were no longer an issue. Throughout Phase A, the aerobic reactor nitrates paralleled those in the effluent but were, in general, slightly higher. Effluent nitrate levels are shown in Figure 4.2.

The attempt at limiting nitrification via maintaining a low sludge age, and thus recirculating very low nitrate levels into

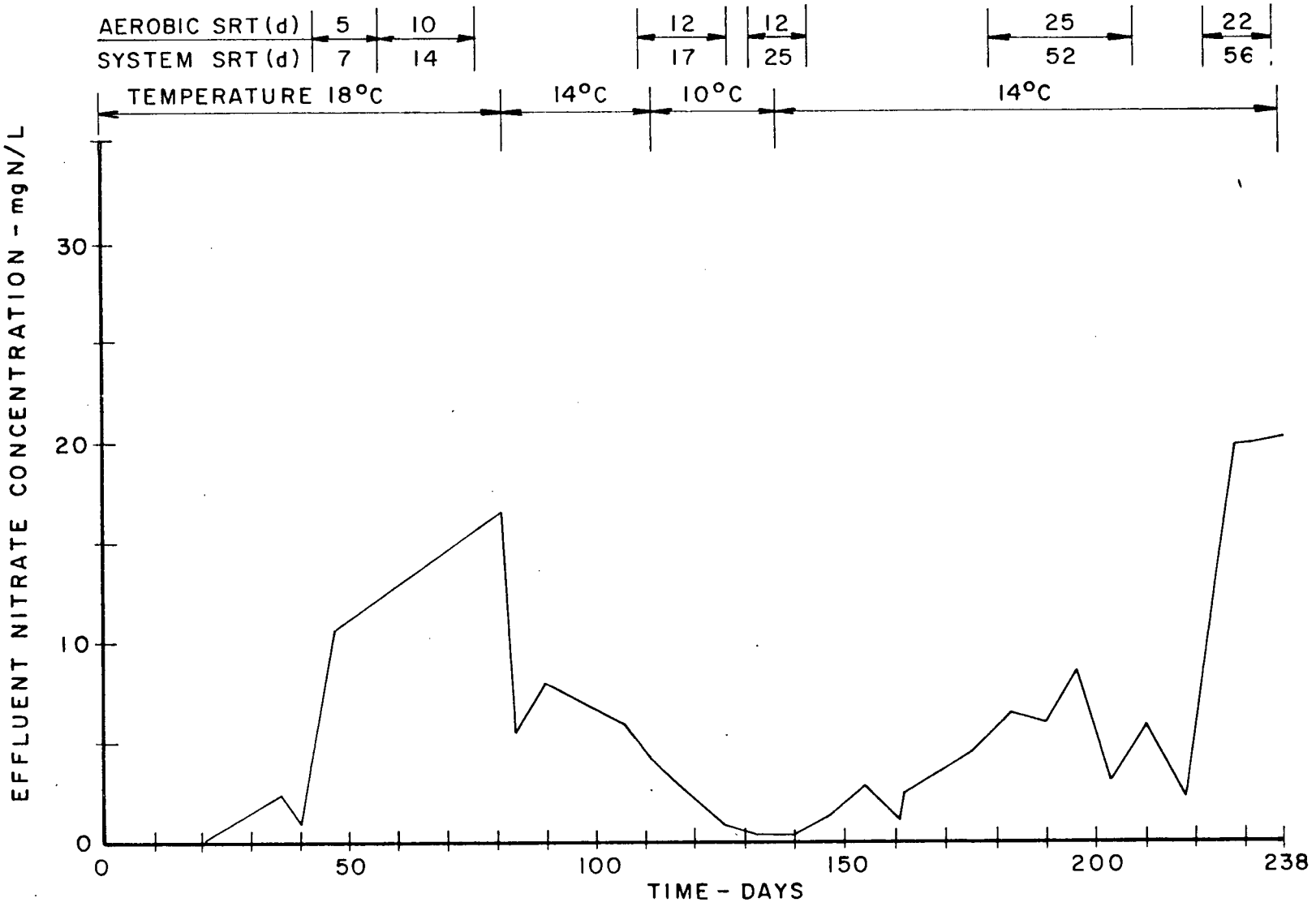



FIG. 4.2: EFFLUENT NITRATES ( PHASE A ).

the anaerobic reactor during the initial 127 days of Phase A, was unsuccessful. Liquid temperature was lowered twice, to a minimum of 10°C on Day 113A, but the lowest nitrate in the effluent (and thus assumed in the sludge return) was 3.0 mg-N/L (excluding the initial 40 days of study during which the nitrifier population was establishing itself and little nitrification occurred). Upon the installation of the sludge conditioning reactor, the highest nitrate concentration entering the anaerobic reactor via the sludge conditioning reactor, was 0.3 mg-N/L, and typically was below 0.1 mg-N/L.

(b) Nitrification and Denitrification Rates

The nitrification and denitrification activities of the biomass in each of the three reactors are presented chronologically in Figure 4.3.

The anaerobic reactor was intensively involved in denitrification until Day 127A, when the sludge conditioning reactor was installed. Negligible denitrification activity was observed in the anaerobic reactor once the sludge conditioning reactor was operational. A maximum denitrification rate of 7.06 mg-N/g MLSS/h occurred in the anaerobic reactor on Day 81A at 18°C. This high value suggests that a healthy denitrifier population had established itself, since the outflow from this reactor contained very low nitrate levels (0.08 mg-N/L). The anaerobic reactor denitrification rates typically fluctuated about the 0.5 mg-N/g MLSS/h level. Dew (1979) reported mean anaerobic denitrification rates of 0.53 mg

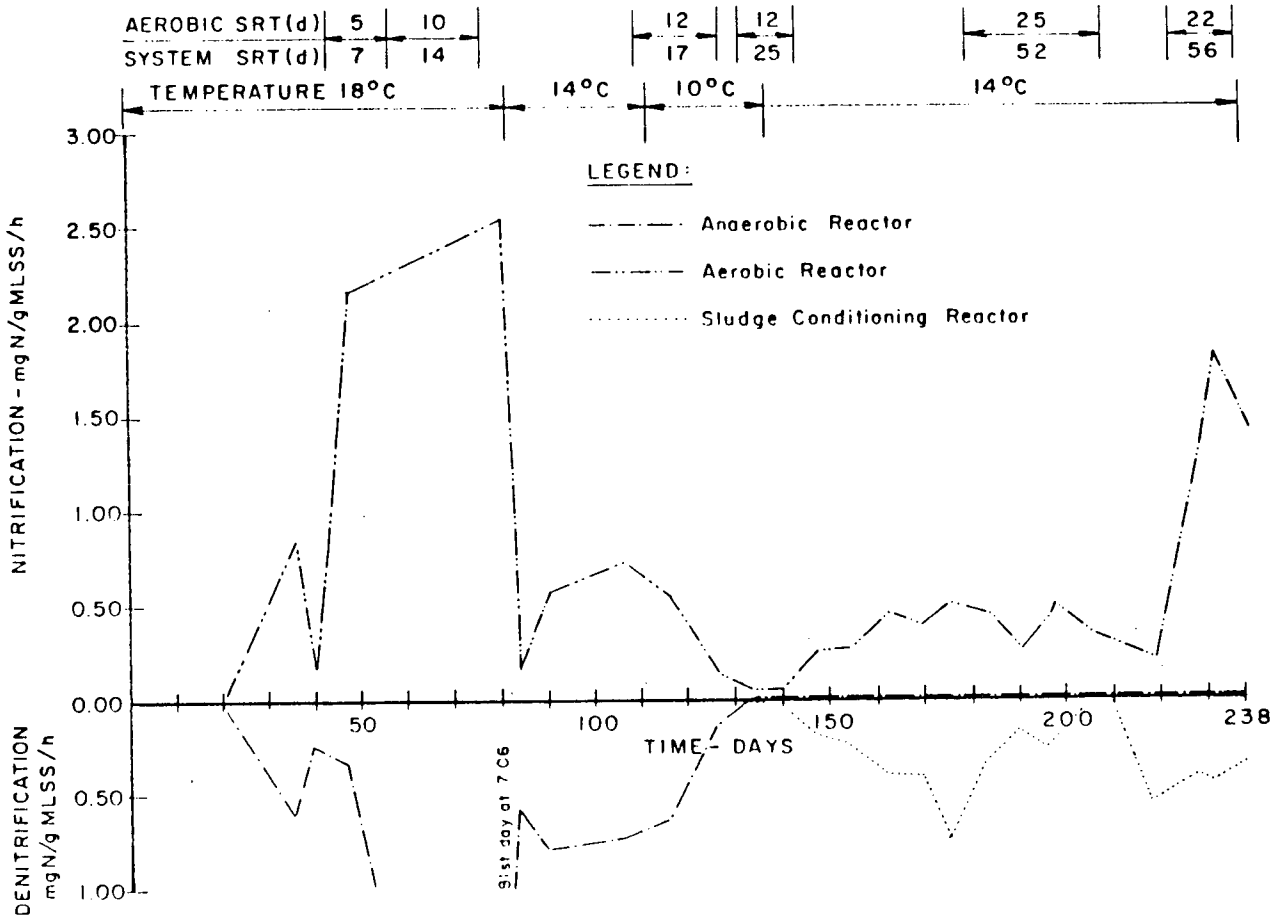



FIG.4.3: NITRIFICATION AND DENITRIFICATION RATES (PHASE A).

(nitrate+nitrite)-N/g MLSS/h at 18° C, with a high of 7.8 mg (nitrate+nitrite)-N/g MLSS/h recorded at this temperature.

Nitrification in the aerobic reactor fluctuated between 0.03 and 2.53 mg-N/g MLSS/h during Phase A. The maximum rate was observed on Day 81A, which coincided with the high denitrification rate observed in the anaerobic reactor on this day. Research by Dew (1979), indicated a range of 0 to 1.76 mg (nitrate+nitrite)-N/g MLSS/h at 18° C in the aerobic basin of the modified Bardenpho scheme studied.

Effluent nitrate concentrations closely paralleled but were generally lower than those in the aerobic reactor, supporting the visual observation of denitrification occurring in the clarifier system.

The denitrification rates in the sludge conditioning reactor fluctuated according to the amount of nitrification activity in the aerobic reactor. Because of the low nitrate content in the outflow from the sludge conditioning reactor, it appeared that this reactor had excess denitrifying capacity and the temperature could be increased back up to 14°C, as indeed it was on Day 138A.

4.3.8.2 Nitrites

Nitrite concentrations were determined simultaneously with nitrates. Feed nitrites were at all times <0.07 mg-N/L during

Phase A. The anaerobic reactor nitrites were <0.08 mg-N/L. Aerobic reactor nitrite concentrations were generally higher than those of the anaerobic and sludge conditioning reactors due to a greater concentration of oxidized nitrogen species, reaching a maximum of 6.70 mg-N/L on Day 203A at 14°C. In general however, values fluctuated around the 1.0 mg-N/L level. Nitrite values approached nitrate levels during periods of incomplete nitrification in the aerobic reactor. Effluent nitrite concentrations paralleled those of the aerobic reactor, but were slightly lower, due to denitrification in the clarifier unit. The sludge conditioning reactor nitrites never exceeded 0.2 mg-N/L.

4.3.8.3 Total Kjeldahl Nitrogen ( TKN )

Feed TKN concentrations varied between 37 and 70 mg-N/L. The effluent values ranged from 3 to 35 mg-N/L. The high effluent TKN concentration on Day 148A occurred at a time when low nitrification rates were recorded in the aerobic reactor. Appendix II, Table II-3 summarizes the TKN results of this study.

4.3.9 Phosphates

Both orthophosphates and total phosphates were analyzed during this study. The methods of phosphate analysis were presented in Section 3.4.8. The efficiency of the treatment scheme, in terms of P removal, was based on orthophosphate

determinations only. Total phosphate determinations were performed so that comparisons with data from other research could be made.

4.3.9.1 Orthophosphates

(a) Concentrations

As discussed in Section 3.4.8.1, orthophosphate determinations were performed on samples filtered through a 0.45 μm membrane filter, except in the case of feed samples which were unfiltered.

Concentrations of orthophosphates in the feed, all reactors, and the effluent are given in Figure 4.4. The lowest orthophosphate level in the aerobic reactor of 3.3 mg-P/L occurred on Day 28A. This relatively low result corresponded to an unusually high orthophosphate concentration of 13.2 mg-P/L in the anaerobic reactor. Although not as marked, this phenomenon was also observed on Day 218A. It appears that high orthophosphate levels in the anaerobic reactor, are a prerequisite for obtaining low P concentrations in the aerobic reactor. In general, effluent orthophosphate levels slightly exceeded those of the aerobic reactor ( by up to 2.9 mg-P/L).

(b) Percentage Removals

Orthophosphate percentage removals are based on a difference in feed and aerobic reactor samples because of

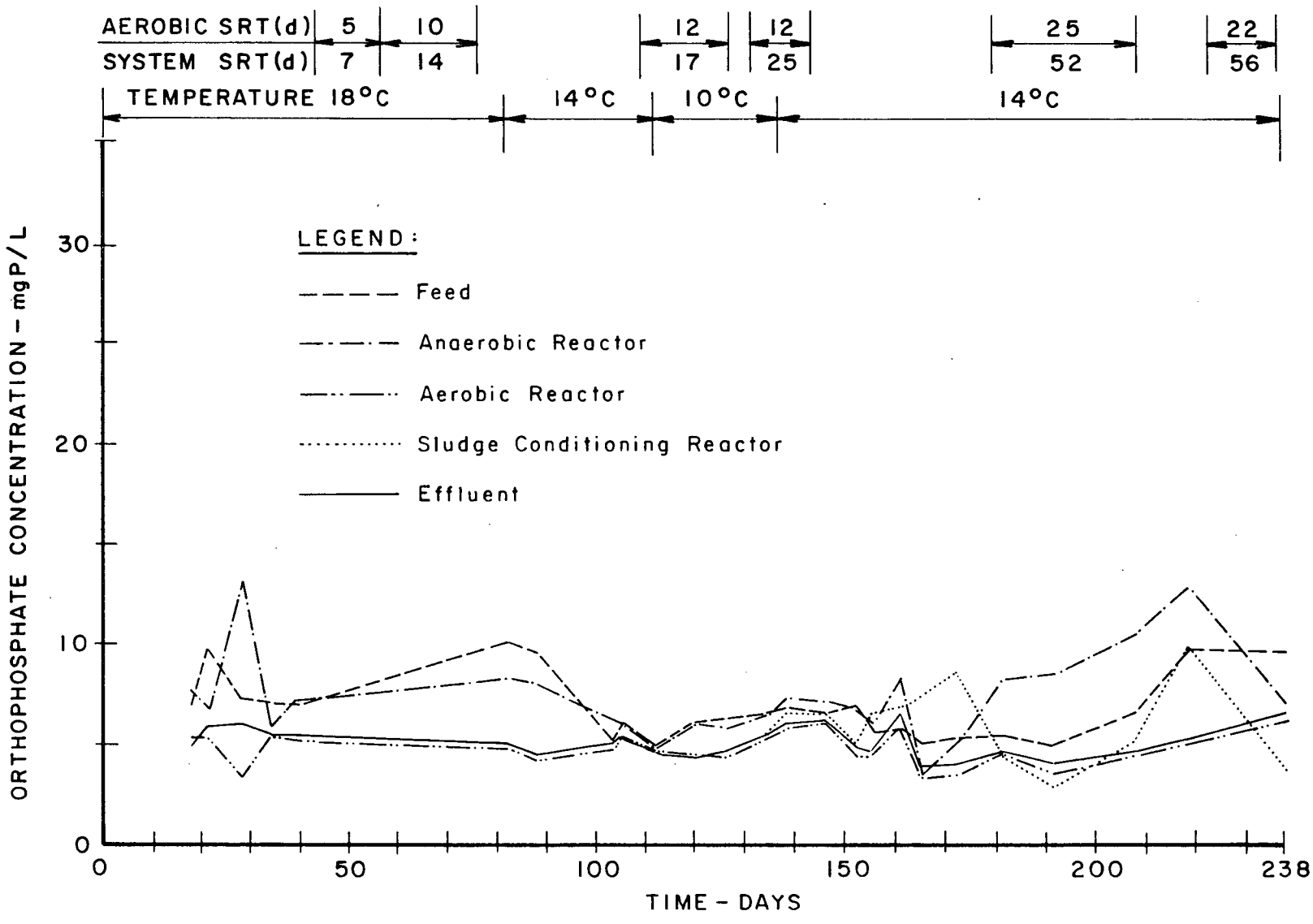



FIG. 4.4: ORTHOPHOSPHATES (PHASE A).

potential P release in the clarifier (due to anaerobic conditions developing in the sludge blanket). Clarifier release of P was substantiated on numerous occasions in this study, since typical effluent P concentrations were greater than those in the aerobic reactor. Careful design and operation of clarifiers in full scale treatment facilities would avoid such discrepancies between the aerobic reactor and effluent P concentrations. Percentage removals are shown in Figure 4.5. The maximum removals obtained occurred on Days 28A and 88A when 55% of the influent orthophosphates were removed by the system.

(c) Orthophosphate Reaction Rates

The anaerobic, aerobic and sludge conditioning reactor orthophosphate reaction rates are presented in Figure 4.6. In general, the anaerobic reactor biomass was involved in the release of orthophosphates. Especially high release rates occurred simultaneously with the higher orthophosphate removal percentages discussed in Section 4.3.8.1. (b). Release in the anaerobic reactor was not affected by the installation of the sludge conditioning reactor and results continued to fluctuate appreciably (between -0.43 mg-P/g MLSS/h (nitrification) and 4.72 mg-P/g MLSS/h). Simpkins (1979) reported anaerobic release rates of up to 10 mg-P/g MLSS/h.

The orthophosphate reaction rates in the aerobic reactor were at all times in the form of uptake; varying between 0.02 and 4.37 mg-P/g MLSS/h (compared to approximately

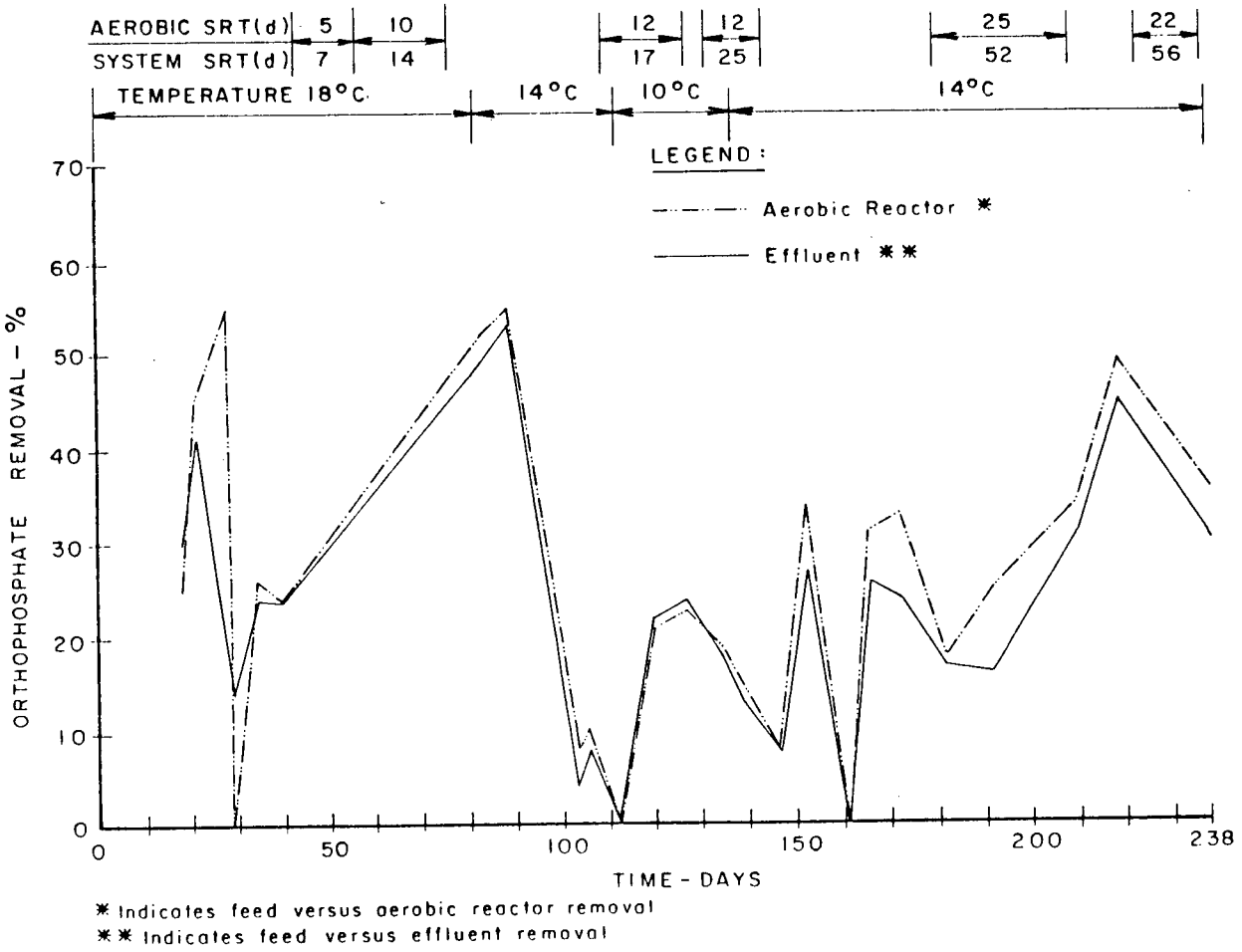



FIG.4.5: PERCENTAGE ORTHOPHOSPHATE REMOVAL ( PHASE A ).

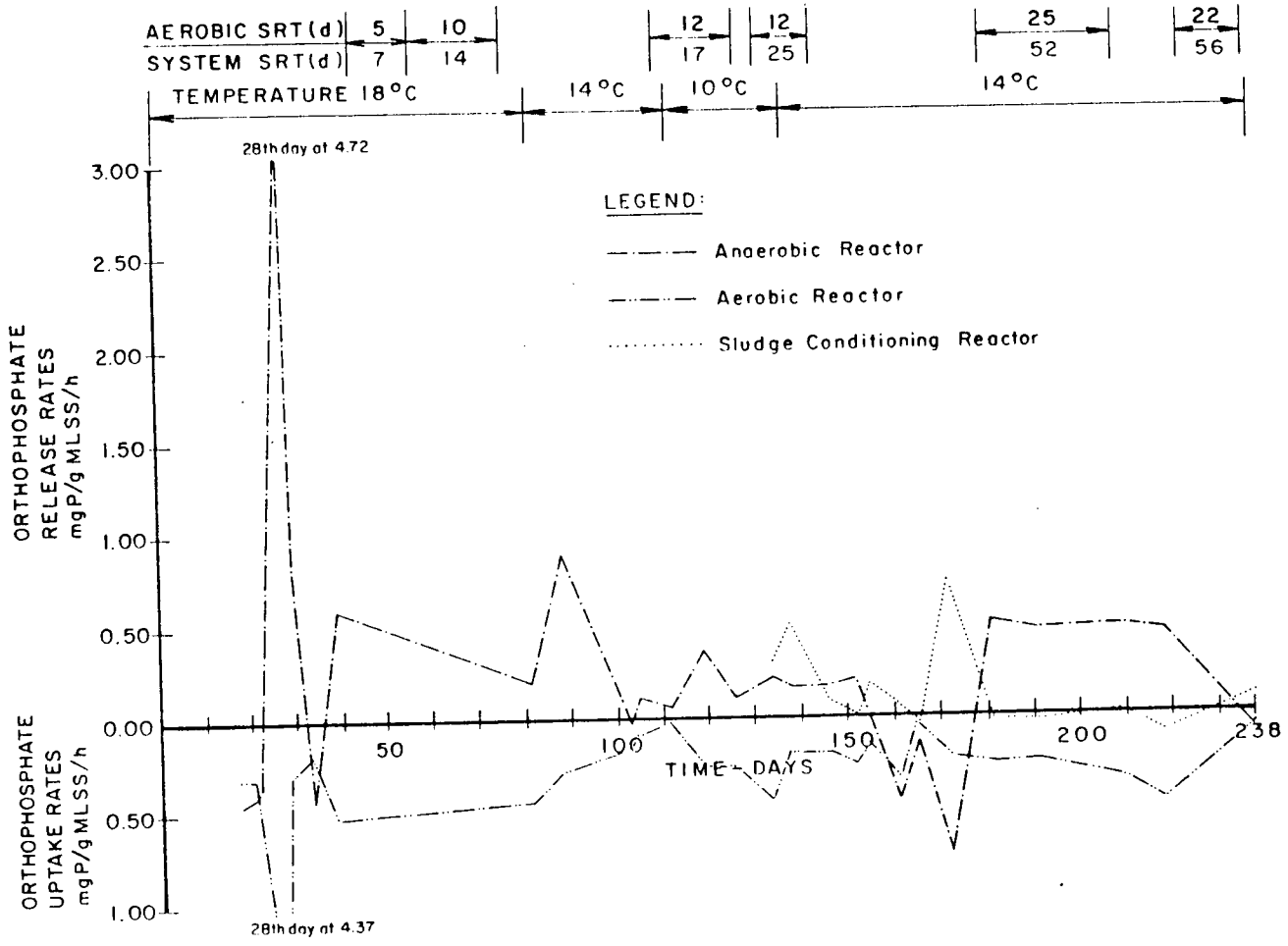



FIG. 4.6: ORTHOPHOSPHATE REACTION RATES ( PHASE A ).

7 mg-P/g MLSS/h observed by Simpkins (1979)). When release rates were relatively high in the anaerobic reactor, the aerobic uptake rates were also comparatively high.

4.3.9.2 Total Phosphates

Total phosphates were monitored only occasionally and are reported in Appendix II, Table II-4. Results indicate that the ratio of orthophosphates to total phosphates in unfiltered feed samples ranged between 0.58 and 0.99.

Virtually all of the filtered effluent total phosphates consisted of orthophosphates.

4.3.10 $\Delta P/\Delta COD$

Appendix II, Figure II-3 summarizes the $\Delta P/\Delta COD$ ratios for this phase of study. The calculation methods used were presented in Section 3.10. Orthophosphates in the aerobic reactor were assumed to constitute 100% of total phosphates. Phase A $\Delta P/\Delta COD$ ratios varied from 0.004 to 0.027, yielding P removals of 10% to 52% respectively. For the most part, $\Delta P/\Delta COD$ ratios were at the lower end of the range determined by other reaserchers which were discussed in Section 2.2.2.

4.3.11 Hydraulic Retention Time ( HRT )

Both nominal and actual hydraulic retention times maintained in each of the three reactors were presented in Table

3.3. The HRT's were relatively constant in each reactor until Day 186A, when flows were generally reduced; this caused sightly longer liquid retention.

4.3.12 Solids Retention Time ( SRT )

As noted both the aerobic and system SRT's are presented with each figure in this chapter. The days for which SRT's are not given represent periods of system instability (See Section 3.7). The solids retention times were increased as Phase A proceeded to achieve higher suspended solids in the reactors, thus increasing the potential for the growth of volutin forming bacteria (Section 2.1.3).

4.4 Conditions Affecting Phosphate Removal

Due to daily (and perhaps hourly) fluctuations of the influent COD, no effect of this parameter on the phosphate uptake rate in any of the three reactors could be accurately discerned.

There was no apparent role played by the reactor MLSS concentrations, per se, on the phosphate reaction rates. MLVSS was used as an indicator of the viable biomass in the system in terms of phosphate release or uptake. No knowledge is currently available regarding the actual organism involved in the uptake and release of phosphates (see Section 2.1.3); thus the choice of MLVSS is arbitrary at this point. Unlike results reported by Menar and Jenkins (1970), enhanced P removal did not parallel a

reduction in the ratio of MLVSS to MLSS in the aeration tank during Phase A.

Since the DO in the reactors was maintained relatively constant throughout this phase (<0.3 mg/L) in the anaerobic and sludge conditioning reactors, and typically 2 to 4 mg/L in the aerobic reactor, no effect of this parameter on phosphate reaction rates could be determined.

Complete anaerobiosis (ORP typically <<-130 mV) was maintained in the anaerobic reactor; thus the release of phosphates should have occurred throughout Phase A (according to Jank et al. (1978)). Anaerobic release was acheived only sporadically however, with subsequent uptake in the aerobic reactor also fluctuating appreciably. At the beginning of Phase A, phosphate release and uptake rates were high (Day 28A). This phenomenon can probably be attributed to the instability of the system at that time. On Day 160A the system was shut down (see Section 4.2.4). Phosphate analysis of the anaerobic reactor contents (after the system had been restarted) indicated the uptake of phosphates in this unit. Uptake continued during the subsequent two weeks. This may have been due to the anaerobic biomass attempting to overcome the endogenous phase which resulted from the shut-down. Upon restarting the process, anaerobic respiration by the biomass in this unit would have resulted in rapid uptake of carbonaceous material and simultaneously also of phosphates. It may be noted that phosphorus would of course be abundant since the endogenous

conditions during the shut-down period would have been conducive to the triggering of the release phenomenon. No discernable effects on the behavior of the aerobic biomass were observed as a result of this system modification. Phosphate uptake rates in the aerobic reactor were relatively low however.

The effect of pH and alkalinity on phosphate reaction rates was not substantiated during this phase of study. There is no evidence that a reduced alkalinity in the effluent occured simultaneously with enhanced P uptake, which the proponents of the chemical precipitation mechanism maintain (Menar and Jenkins (1970)).

Trace metal effects may play an important role but much more information is required before these can be isolated. The feed used in this study contained relatively low metal concentrations. Enhanced removal of P (by chemical precipitation) did not materialize, even though relatively high calcium and magnesium ion concentrations were manifested in the feed during Phase A. The content of metals in the feed was in general constant; thus their effects may be considered as unchanged for the duration of this phase of study. It is necessary to have a better microbiological understanding of the effects on phosphate reaction rates in wastewater treatment schemes and "threshold" metals concentration, before this aspect can be addressed. Trace metals may be harmful to the phosphate accumulating organisms at higher sludge ages since their accumulation in the sludge may be substantial. A more detailed

discussion on the role of trace metals is presented in Section 5.2.7.

Numerous researchers, to date, have emphasized the importance of maintaining a truly anaerobic reactor at the head end of biological P removal treatment schemes (Section 2.2). It was evident however, from this work, that significant levels of nitrates were recycled to the anaerobic reactor via the sludge return (up to 16 mg-N/L). These nitrates were rapidly denitrified with the aid of a carbon source derived from the raw sewage feed. Stemming nitrification in the aerobic reactor by temperature reduction had insignificant effects in the long term. The sludge conditioning reactor was installed to provide a basin where denitrification of the return sludge would take place prior to the anaerobic reactor. In this way, the possible interference of denitrification with the phosphate release mechanism in the anaerobic reactor was avoided. Although the sludge conditioning reactor was involved in endogenous denitrification, which is typically slower than denitrification rates with raw wastewater as a substrate, it was capable of almost completely denitrifying the sludge return. The denitrification rates in the anaerobic reactor, subsequent to the installation of the sludge conditioning reactor, were negligible. The release of phosphates in the anaerobic zone did not change as a result of this modification however.

The reaction rates given are, in the case of both nitrates and phosphates, a function of MLSS levels maintained in each of

the reactors - thus high MLSS concentrations yield low reaction rates and vice versa. This assumption could be misleading, especially if it is determined at a later date that that MLSS is not the most appropriate parameter in reaction rates calculations. Temperature effects must also be considered when dealing with the magnitudes of the reaction rates.

# CHAPTER 5

# PHASE B: CONDITIONS REQUIRED FOR THE SUCCESSFUL BIOLOGICAL REMOVAL OF PHOSPHATES FROM WASTEWATER

## 5.1 Description of System Operating Conditions

This chapter describes in detail the conditions necessary for the successful removal of phosphates from municipal wastewater, as determined for the configuration of the system used in this study. Although there were no major physical modifications during this phase, the operational characteristics were significantly altered. From the results obtained during Phase A, it was apparent that P removal was not occurring with a low sludge age system, even when the input of nitrates to the anaerobic reactor was negligible. Thus Phase B was characterized by a high mean cell residence period. Wasting of mixed liquor from the aerobic reactor was halted on Day 1B following the completion of Phase A. No wasting was practised until Day 27B. The liquid temperature was raised to $18^{o}$ C on Day 1B and maintained at this level throughout Phase B. These measures were taken to promote the growth of microorganisms and thus achieve a high MLSS content in the reactors. A high sludge age would intuitively promote the establishment of a large population of organisms which are capable of P uptake.

The sludge recycle rate was maintained typically at half the feed rate. This was necessary since net solids growth is

slow at high sludge ages, yielding little settled sludge in the clarifier. Care also had to be taken to avoid recirculation of clear effluent to the sludge conditioning reactor with a possibility of subsequent dilution of all three reactors.

## 5.2 Results and Discussion

The system removed up to 98% of phosphates and 96% of COD during Phase B. Markedly improved P removal was initiated on Day 17B when an unprecedented 92% P removal (feed versus aerobic reactor) was acheived. Consistently high percentage removal rates were obtained for 30 days following this observation, while a high solids retention time was maintained. The sludge age was reduced beginning on Day 40B, by increasing the wasting rate, in an effort to determine the optimum sludge age below which P removal is not possible. The system failed in terms of its capability to remove P on Day 52B, at an approximate aerobic sludge age of 24 days. A further two weeks of testing showed that the system did not regain its ability for enhanced P removal. In fact, on three occasions, P concentrations of filtered samples from the aerobic reactor were higher than those in the unfiltered influent samples.

### 5.2.1 Chemical Oxygen Demand ( COD )

Synthetic feed addition was practised as dictated by the raw sewage COD concentrations throughout Phase B. Consistent influent COD concentrations were not achieved however, and

values ranged between 180 and 1410 mg/L (Appendix II, Figure II-1).

COD percentage removals for the system during Phase B varied from 83 to 96%, with an exceptionally low removal of 55% on Day 18B. A chronological presentation of percentage COD removals during Phase B is given in Figure 5.1. Once again, as previously pointed out in Section 4.3.1, low COD removals occurred concurrently with low influent COD. One reason for this occurrence may have been that the influent COD was atypical on the given days of sampling. This would have had a particularly noticible effect, since the effluent was sampled simultaneously with the feed, and would not have yet adjusted to any increase in influent COD. Also, there may be a baseline level of non-biodegradable organics in the feed, which would be conducive to the above results, in that a high COD would not necessarily indicate a high amount of available carbonaceous material.

### 5.2.2 Total Suspended Solids ( TSS )

To achieve a high sludge age, microbial growth was encouraged in all three reactors by controlling the daily wasting rate. The solids concentrations in the anaerobic and aerobic reactors were 2500 mg/L at the beginning of Phase B, reaching a peak of 5600 mg/L after approximately 30 days, and gradually decreasing to 2500 mg/L at the end of this phase (See Figure 5.1).

The TSS in the sludge conditioning reactor ranged from 3570

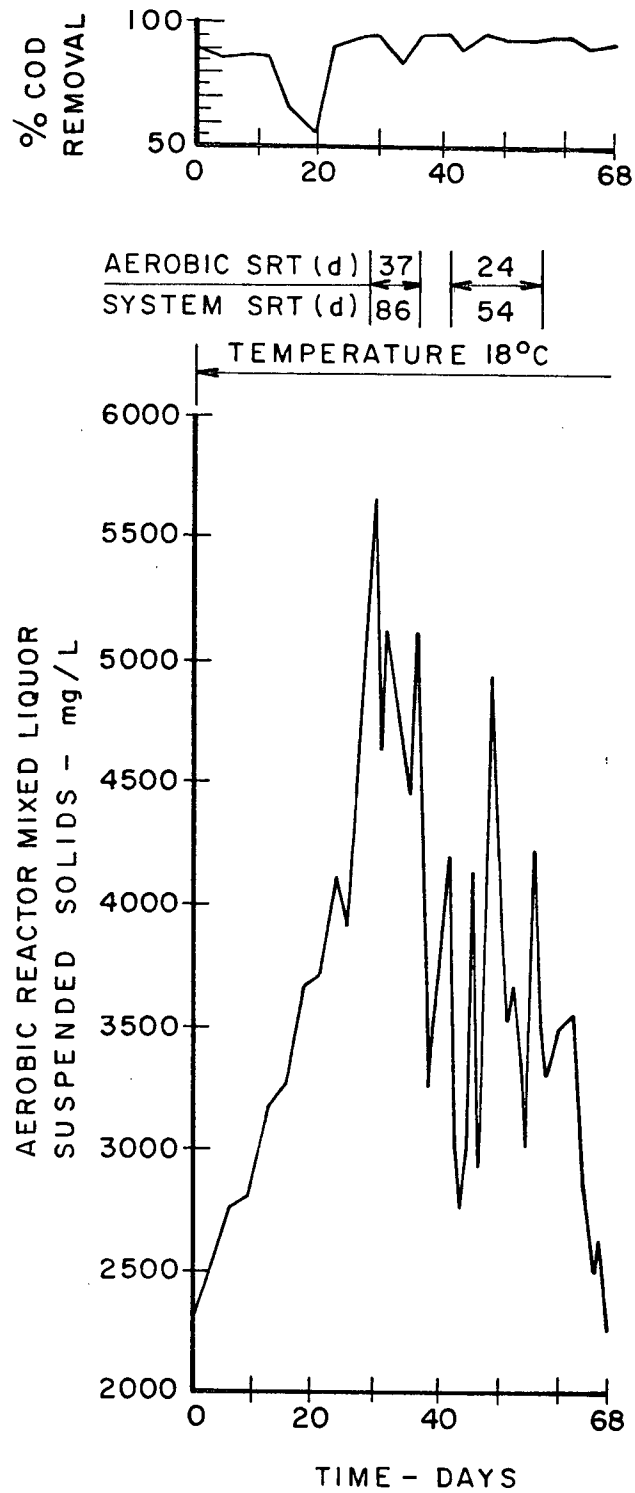



FIG. 5.1 : SYSTEM PERCENTAGE COD REMOVAL AND AEROBIC MLSS LEVELS ( PHASE B ).

to 12,940 mg/L. This large variation was primarily due to poor clarifier performance (Section 4.3.2.). Manual, gentle stirring of the clarifier was practised (typically twice daily) to dislodge the nitrogen gas bubbles trapped in the sludge. This alleviated the problem somewhat but conditions reverted to the adverse at night. Because of the small size, little buffering capacity was inherent in the system (to overcome the feed strength fluctuations and inconsistency in the sludge return solids content) adversely affecting the system performance. The small scale pumping system was the major cause of the large fluctuations in feed composition. Often, partial blockages of the feed line occurred causing a filtration effect on the feed. Also such factors as cleaning of the reactor walls to dislodge any biomass adhering to the surfaces caused shock loading on the system. This was a result of the relatively high surface area to volume ratios inherent in a small scale system, compared to full scale operations.

Effluent TSS concentrations are presented in Appendix II, Figure II-2. Daily fluctuations were common for reasons discussed in Section 4.3.2.

Because of the poor results obtained with this small scale system, no percentage removals of suspended solids through the system are presented. Large scale clarification would preclude the problems encountered with bench scale units (Barnard (1974)). The excessive TSS lost in the effluent caused problems with achieving and/or maintaining high solids concentrations in

the reactors. Rigorous maintenance of the system was required to prevent washout. Average effluent suspended solids concentrations were used in SRT calculations (Section 3.7).

### 5.2.3. Mixed Liquor Volatile Suspended Solids ( MLVSS )

On an average basis the volatile suspended solids content typically constituted 84% of the total suspended solids for all three reactors during this phase. This was slightly lower than determined during Phase A.

### 5.2.4 Dissolved Oxygen ( DO )

Dissolved oxygen levels in the aerobic reactor were typically maintained between 2 and 4 mg/L during this phase. Nightime values were generally slightly higher than day time, due to a less dense sludge return (causing dilution of reactor solids). In effect, nighttime conditions caused the increase of the F/M ratio, thus lowering the oxygen demand.

The DO of both the anaerobic and the sludge conditioning reactors did not exceed 0.2 mg/L at any time, and were typically negligible.

### 5.2.5. Oxidation Reduction Potential ( ORP )

Highly anaerobic conditions existed in the anaerobic reactor throughout Phase B. The ORP fluctuated between -300 and -500 mV (typically approaching the latter higher value); well

below the anoxic cut-off point of -130 mV.

### 5.2.6 Alkalinity and pH

Appendix II, Table II-1 summarizes alkalinity and pH data obtained during Phase B. Although the alkalinity of the raw sewage, as analyzed upon arrival at the laboratory, was in general similar to that typical of Phase A, the feed alkalinity was higher throughout Phase B . This can be accounted for by the relatively high additions of the synthetic sewage (which had a high bicarbonate content) during Phase B. Alkalinity was lost in the aerobic reactor, while it was gained in the anaerobic and sludge conditioning reactors. This observation supports the concept that alkalinity is formed during the reduction of nitrates, at a rate which is twice that of the alkalinity used up during the nitrification of ammonia in aerobic reactions (Beer and Wang (1978)).

Trends in pH could not be substantiated during this phase, due to insufficient data and results which were extremely close in magnitude; thus it was difficult to discount experimental error. It can be noted however, that low aerobic reactor and effluent pH values occurred simultaneously with low alkalinities.

### 5.2.7 Trace Metals

As described in Section 4.3.7, trace metal analysis was performed on each fresh delivery of raw sewage to the

laboratory. Feed trace metals concentrations did not change appreciably from those determined in Phase A (Appendix II, Table II-2).

On Day 61B a scan was performed of the trace metal content of unfiltered samples of the influent, the contents of each reactor, and the effluent (Appendix II, Table II-5). The results indicate high metals concentrations in the mixed liquor of all three reactors, particularly in the sludge conditioning reactor. Research by Neufeld and Hermann (1975) indicates that the ratio of metal in the biomass, to that in the surrounding liquid on a weight basis, may vary from 4000 to 10000 for cadmium and zinc. A U.S. Public Health Service study (1965) indicated that if the influent levels of chromium, copper, nickel, and zinc exceeded 10 mg/L in any combination, a 5% decrease in treatment efficiency (in terms of BOD and SS removal) can be expected in activated sludge schemes; nitrification was practically eliminated at these concentrations. Barth et al. (1967) showed that up to 60% removal of metals is possible with secondary sewage treatment, causing the inflow to the sludge digesters to contain up to 37 times the metals concentration of the plant influent. Day 61B results indicate over 70% removal of iron from influent to effluent, while other metal percentage removals were typically lower. Results from this study show that the sludge return metal concentration (based on the sludge conditioning reactor metal levels) were as much as 120 times (as was the case for chromium) that of the feed. In fact, only calcium and magnesium concentrations in the sludge were less than 37 times

that in the feed. In the above discussion, the sludge conditioning reactor metal concentrations have been compared with the digestor inflow levels reported in other research work.

The feed calcium and magnesium ion concentrations were approximately 8.4 and 2.2 mg/L respectively. Day 61B results show that 8.3 mg/L calcium and 1.9 mg/L magnesium were present in the effluent. Thus an insignificant reduction of these ions occurred via the system used in this study, even though excellent P removals were realized. Barnard (1974) obtained enhanced P uptake while the $Ca^{++}$ and $Mg^{++}$ effluent concentrations were 38 and 18 mg/L respectively (influent levels were not reported). On this basis, P precipitation seems an unlikely mechanism.

### 5.2.8 Nitrogen Forms

Because of the widely variable results obtained during Phase A, the effects of the presence of the various nitrogen forms was not substantiated in terms of either phosphate release or uptake. The monitoring of nitrates, nitrites, and total Kjeldahl nitrogen was thus continued during Phase B to substantiate these findings. The effects of nitrification in the aerobic reactor on phosphate uptake rates also required further clarification.

5.2.8.1 Nitrates

(a) Concentrations

Typical feed nitrate levels did not exceed 0.04 mg-N/L. Relatively high values (for example, 15 mg-N/L on Day 13B) most probably resulted when feed-storage, drum liquid levels were low, encouraging air to be entrained via agitation by stirring.

The anaerobic reactor nitrate concentrations varied from <0.01 to 0.11 mg-N/L. The latter high value cited is atypical of this phase. It is possible that the "sudden" reduction of aerobic sludge age, from 37 days to 24 days, may have temporarily altered the characteristics and/or functions of the biomass.

Nitrate levels in the aerobic reactor during Phase B were characterized by erratic fluctuations. Values between 0.25 and 34.2 mg-N/L were recorded.

Effluent nitrate concentrations paralleled the aerobic reactor values. In general, as was also the case in Phase A, the effluent was typically lower in nitrates (as much as 14.3 mg-N/L on Day 48B). This difference accounts for the frequently observed denitrification in the clarifier unit. Effluent nitrate levels are given in Figure 5.2. Relatively low nitrate concentrations occurred at the high system sludge age of 86 days. This phenomenon was probably caused by the destruction of the nitrifying portion of the biomass due to an excessively long anaerobic SRT (Marais (1979a)). The reduction in system SRT to

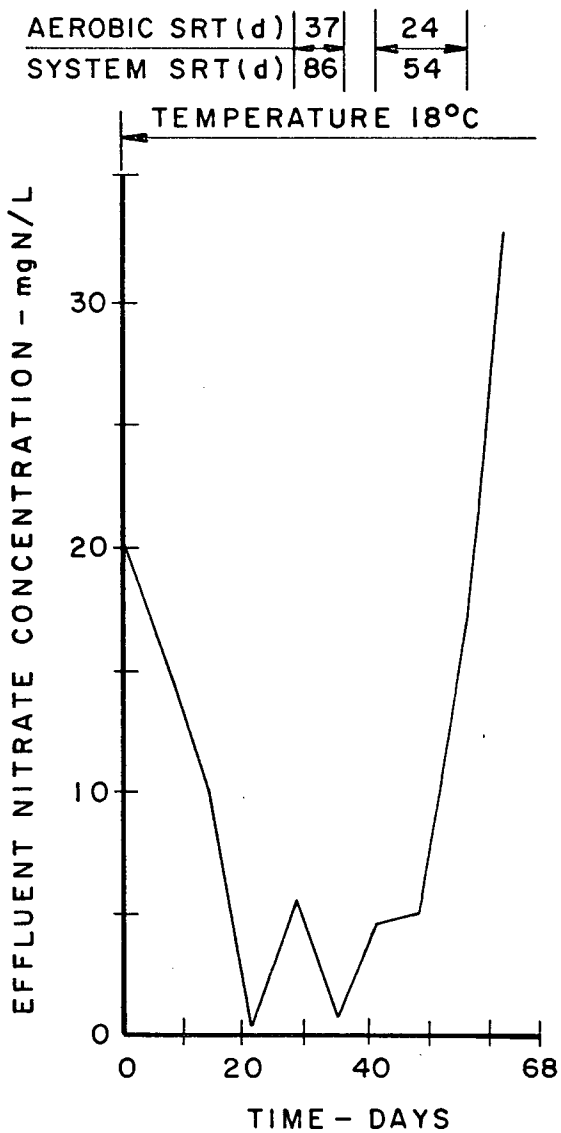



FIG. 5.2: EFFLUENT NITRATES ( PHASE B ).

54 days returned the nitrates in the effluent (due to nitrification activity in the aerobic reactor) to previously elevated levels.

The sludge conditioning reactor, whose primary function was the denitrification of the clarifier underflow, contained consistently low nitrate levels, generally <0.1 mg-N/L.

(b) Nitrification and Denitrification Rates

The nitification and denitrification rates were calculated as described in Section 3.8 and are presented in Figure 5.3 for each of the reactors.

Denitrification was the norm in the anaerobic reactor, although two instances of apparent nitrification occurred on Days 41B and 48B. These rates were negligible in magnitude (<0.007 mg-N/g MLSS/h). At these low rates, factors such as sampling techniques and sample preparation and analysis may play significant roles in determining whether the anaerobic biomass apparently plays a nitrifying or a denitrifying role.

The aerobic reactor microorganisms were involved in nitrification throughout Phase B. The degree of nitrification was highly variable, ranging from <0.01 to 1.42 mg-N/g MLSS/h. As expected, the low rates corresponded to a low nitrate concentration in the aerobic reactor and vice versa.

As was previously mentioned, the sludge conditioning reactor was installed to denitrify the return sludge. This

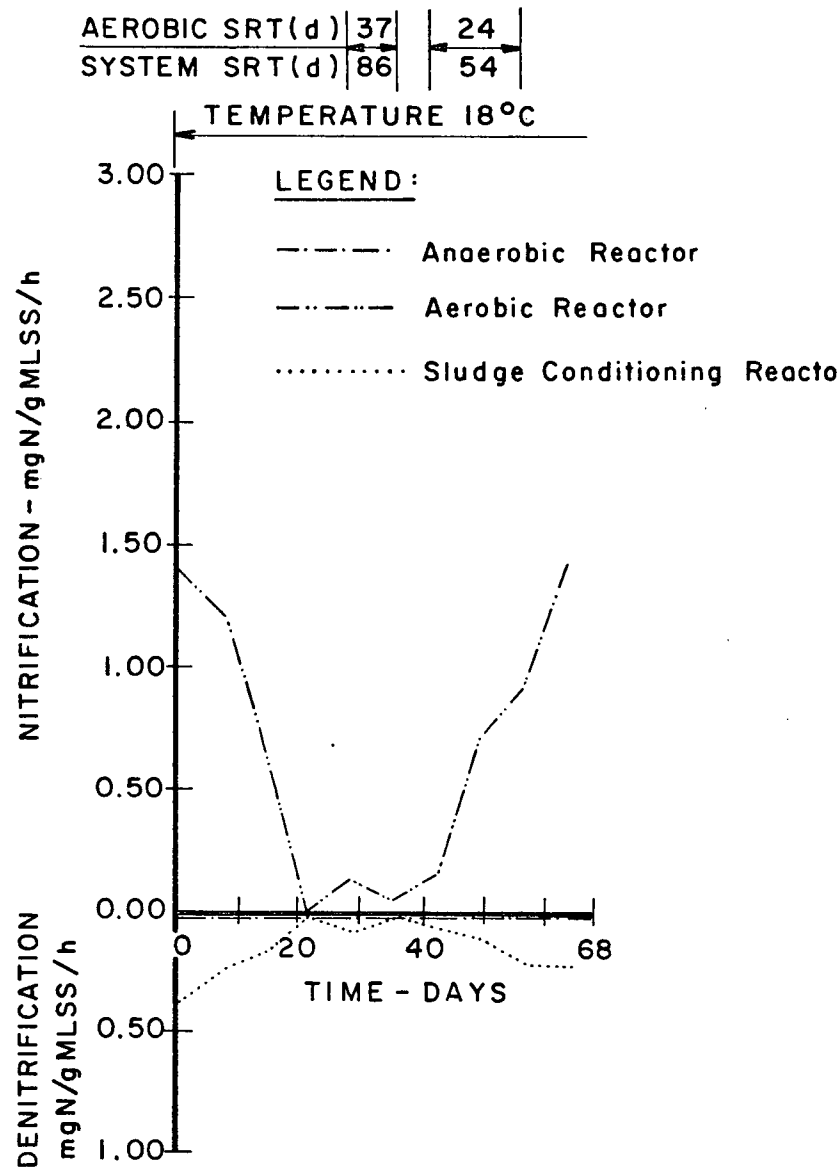



FIG.5.3 : NITRIFICATION AND DENITRIFICATION RATES (PHASE B).

measure (as described in Section 4.3.8.1 (b)) proved successful. Denitrification rates in this reactor were less than 0.36 mg-N/g MLSS/h and although of lesser magnitude, were a mirror image of the nitrification rates in the aerobic reactor.

The function of the clarifier contents, in terms of nitrification or denitrification activity, could not be determined, due to variations in the sludge blanket height and concentration. The visual observation of bubbles (assumed to be nitrogen gas) in the sludge, and the lower nitrate concentrations in the effluent, as compared to the aerobic reactor, strongly point to denitrification activity in the clarifier unit.

5.2.8.2 Nitrites

Feed nitrite concentrations were typically <0.02 mg-N/L throughout Phase B. The anaerobic reactor nitrites during this phase were <0.04 mg-N/L. Nitrite levels in the aerobic reactor ranged between 0.02 and 4.75 mg-N/L with similar concentrations found in the effluent during this phase. The high nitrite concentrations recorded for the aerobic reactor corresponded to periods of incomplete nitrification. The nitrite values in the sludge conditioning reactor were between 0.01 and 0.12 mg-N/L.

5.2.8.3 Total Kjeldahl Nitrogen ( TKN )

Feed TKN concentrations ranged from 42.7 to 56.6 mg-N/L. The effluent values varied between 4.1 and 31.5 mg-N/L (Appendix

II, Figure II-3). As observed during Phase A, periods of low effluent TKN occurred simultaneously with relatively high aerobic reactor nitrate contents. This was a result of a large fraction of the TKN having been converted to oxidized nitrogen forms of which nitrates were the major portion, throughout this study.

### 5.2.9 Phosphates

Both orthophosphate and total phosphate concentrations (in particular the former) were determined during this phase of study. Since successful P removal was realized, sampling during this phase was more frequent than that of Phase A.

#### 5.2.9.1 Orthophosphates

##### (a) Concentrations

The orthophosphate content of the filterable fraction of each reactor and the effluent, as well as the nonfiltered feed, will be discussed in this section. These results are shown chronologically in Figure 5.4.

The unfiltered feed orthophosphate concentrations were similar to levels encountered during Phase A, and ranged between 5.04 and 9.60 mg-P/L. Filterable orthophosphate concentrations in the anaerobic reactor varied from 12.9 to 49.5 mg-P/L (whereas Phase A levels did not exceed 13.2 mg-P/L). The aerobic reactor filterable orthophosphates during Phase B, ranged from

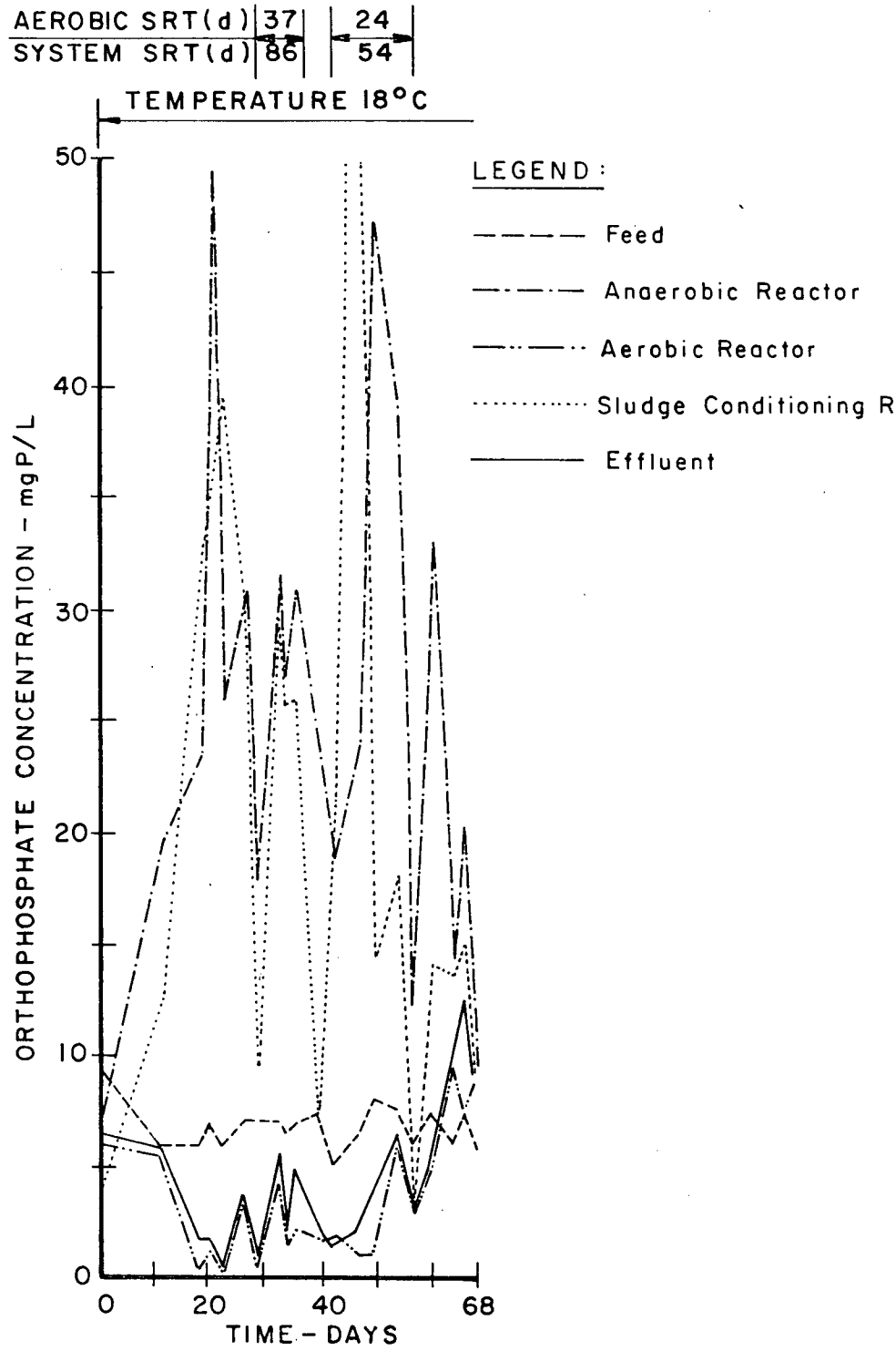


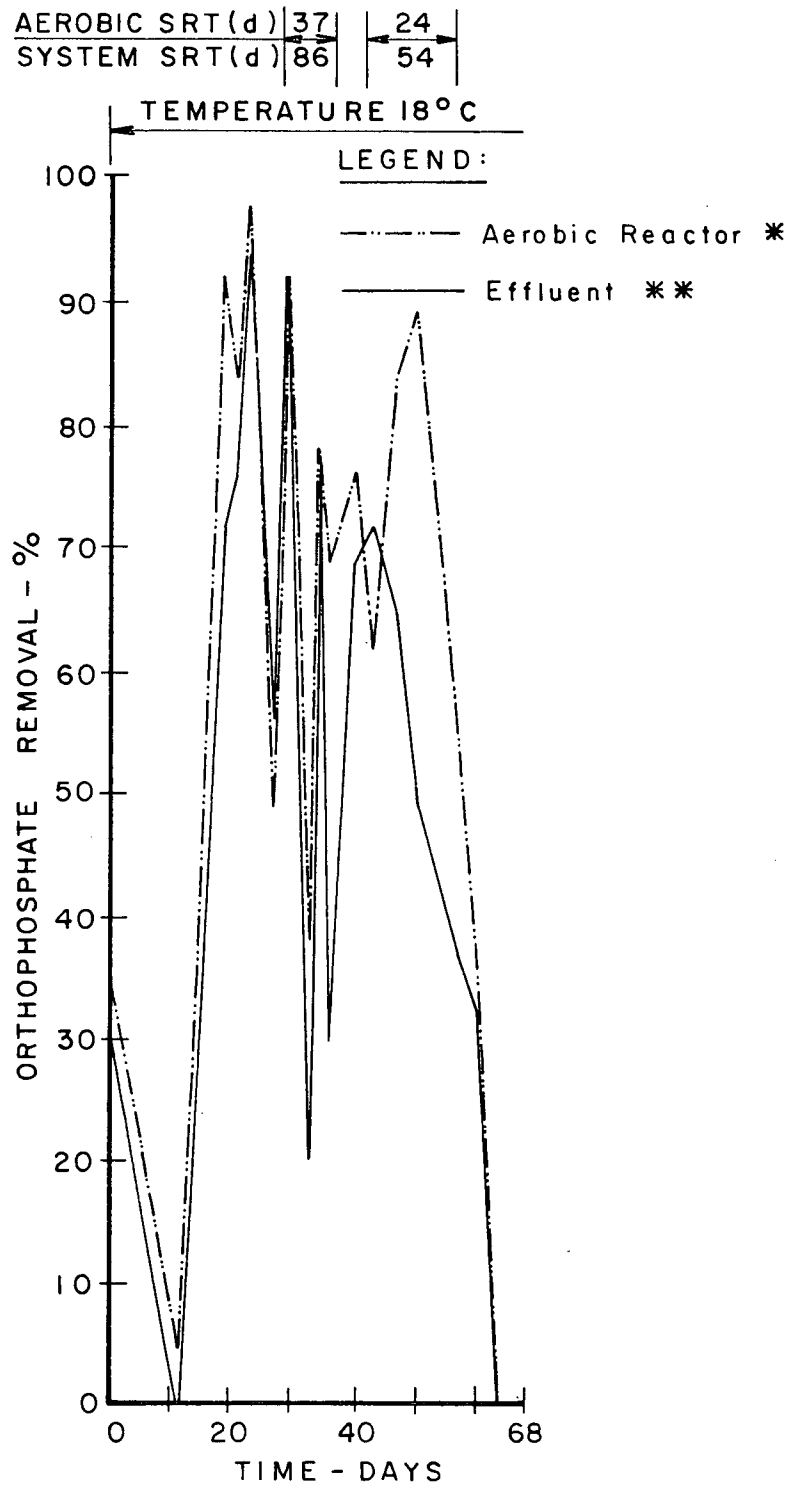

FIG.5.4: ORTHOPHOSPHATES (PHASE B).

0.14 to 15.12 mg-P/L. The latter value corresponded to the failure of the system in terms of P removal (Section 5.2.9.1 (b)). Filtered effluent orthophosphate concentrations were, in general, slightly higher than aerobic reactor levels, probably as a result of P release in the clarifier under anoxic conditions. The orthophosphates of the filterable portion of the sludge conditioning reactor contents varied from 3 to 76 mg-P/L, compared to a maximum of 10 mg-P/L during Phase A when enhanced P uptake was not a norm. It appears that high rates of P release in the sludge conditioning reactor, are conducive to the uptake of P.

In general, the filterable orthophosphate concentrations in the anaerobic reactor were slightly higher than those in the sludge conditioning reactor during Phase B. Periods of high orthophosphate concentrations in these two reactors were concurrent with low values in the aerobic reactor.

(b) Percentage Removals

The percentage of orthophosphates removed during Phase A ranged from less than zero (when release occurred in the aerobic reactor) to 98%. These removal values were determined by considering the difference in unfilterable orthophosphate concentrations in the feed and filterable orthophosphate levels in the aerobic reactor (See Figure 5.5). Release of P in the clarifier was common throughout this phase, causing slightly lower system P removals. During this phase, the system maximum




* Indicates feed versus aerobic reactor removal

** Indicates feed versus effluent removal

FIG.5.5: PERCENTAGE ORTHOPHOSPHATE REMOVAL (PHASE B).

orthophosphate removal was 95% (feed versus effluent). It is assumed that full scale operating conditions would include an efficient clarifier, in which anoxic conditions, and thus P release would be avoided.

(c) Orthophosphate Reaction Rates

The orthophosphate reaction rates were calculated on the basis of the method described in Section 3.9. A chronological plot of these rates, in each of the reactors, during Phase B is given in Figure 5.6. Throughout this phase, the anaerobic reactor biomass released phosphates. The rate of release fluctuated from 0.31 to 3.34 mg-P/g MLSS/h. Orthophosphate uptake occurred in the aerobic reactor, except on Day 68B. This day however, corresponds to complete system failure with regard to orthophosphate removal (induced by reducing the sludge age of the biomass). Aerobic reactor uptake rates varied between 0.04 and 2.05 mg-P/g MLSS/h during Phase B. High release rates in the anaerobic reactor occurred simultaneously with enhanced uptake rates in the aerobic reactor. Although the rates in the sludge conditioning reactor were generally much lower than in the anaerobic reactor, the release of orthophosphates was the norm except for Day 68B. Release rate values in this reactor ranged between -0.04 (orthophosphate uptake on Day 68B) to 0.68 mg-P/g MLSS/h.

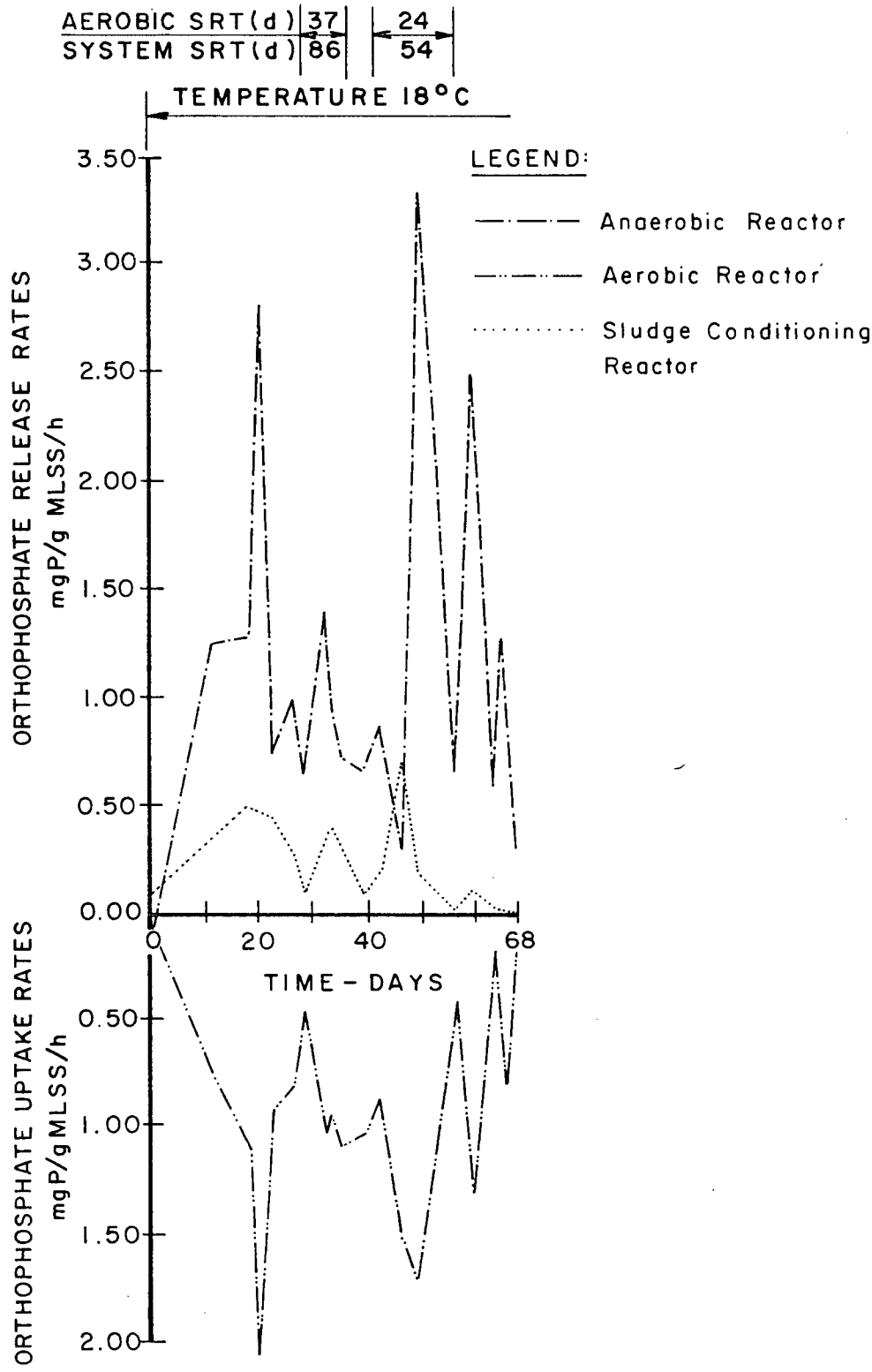



FIG. 5.6: ORTHOPHOSPHATE REACTION RATES (PHASE B).

5.2.9.2 Total Phosphates

Total phosphate determinations on unfiltered influent samples varied from 6.5 to 13.0 mg-P/L during this phase of the study. This indicated that, of the total phosphates in the influent, between 62 and 79% were in the form of orthophosphates.

The filterable total phosphate levels in the effluent ranged from 1.6 to 10.1 mg-P/L. Between 86 and 139% of the total filterable phosphates were in the form of filterable orthophosphates. Results greater than 100% were mainly due to inaccuracy in sampling and analysis. There is a possibility that water entered the flasks at the time of sample cooling. Only two such occasions were recorded.

The purpose of the analysis of total phosphates was to compare the sewage characteristics used in this study to typical values determined by other researchers. It is generally expected that orthophosphates constitute roughly 70% of the total phosphates in sewage. Thus orthophosphate to total phosphate ratios found in this study were typical, particularly in the case of the influent sample ratios. The effluent ratios were at the upper limit of expected results.

5.2.10 $\Delta P/\Delta COD$ Ratio

Total P was used to calculate the change in P through the system. An estimate of the total P in the aerobic reactor was

based on typical effluent total P: orthophosphate ratios, since these were not available for the aerobic reactor contents. The ΔP/ΔCOD ratios are presented in Appendix II, Figure II-3. The ΔP/ΔCOD ratios determined during Phase B vary from 0.005 to 0.081, yielding respective P removals of 62% to 84%. Note that a relatively "low" ΔP/ΔCOD ratio of 0.018 yielded 89% removal of P. This ratio occurred at the time in the study when enhanced removal had already been established. In general, higher ΔP/ΔCOD ratios were obtained during the phase of enhanced P removal by this system, than during Phase A, when the removal mechanism was not well established. This finding agrees well with results of other researchers (for example, Jank et al. (1978)) who claim that high ΔP/ΔCOD ratios occur simultaneously with high P removal.

### 5.2.11 Hydraulic Retention Time ( HRT )

Table 3.3, shows HRT's in all three reactors during Phase B. The HRT in each reactor during the initial six months of Phase A was slightly lower than those maintained in Phase B. There was no difference in HRT's between the last two months of Phase A and that of Phase B.

### 5.2.12 Solids Retention Time ( SRT )

A discussion of SRT calculations (both aerobic and system) is given in Section 3.7. Phase B was characterized by long SRT's. All wasting was terminated on Day 1B. After a build-up of

solids had occurred in all three reactors, wasting was resumed and a 37 day aerobic SRT was maintained. A reduction of SRT on Day 40B occurred as solids wasting was gradually increased.

## 5.3 Conditions Affecting Phosphate Reaction Rates

As was the case with Phase A, large fluctuations of feed COD were also observed during Phase B. Thus the effects of COD on phosphate reaction rates could not be substantiated during the course of this study.

It appears that MLSS concentrations, per se, in the reactors are not responsible for enhanced P removal. However, the MLSS concentrations are important in terms of allowing high sludge ages to be maintained without altering the hydraulic retention times and/or reactor volumes. MLVSS were important only in terms of indicating the concentration of viable microorganisms in the system. The determination of the actual fraction of these microorganisms involved in P uptake and release reactions was beyond the scope of this study.

The dissolved oxygen in the aerobic reactor was maintained at levels comparable to those of Phase A. No conclusion can thus be drawn regarding the possible effect of DO on the activity of the biomass. Anaerobiosis in the anaerobic reactor was maintained (in terms of the dissolved oxygen level and nitrate content) as shown by the oxidation reduction potential results. In general, the ORP levels were lower (more negative) than those exhibited during Phase A. Since the P removal mechanism was not

manifested during Phase A, it is not possible to discern the role played by ORP in the P removal capability of a given biological system. It is clear however that ORP values of approximately -300 to -500 mV are not detrimental to the P uptake phenomenon, and may infact be beneficial.

The lack of comprehensive pH and alkalinity data during Phase B has made it impossible for any relationship to be drawn regarding the correlation of these parameters to the release and/or uptake of P.

There were no discernable differences in influent trace metal concentrations during Phase A and Phase B. The sludge accumulated large quantities of trace metals but these accumulations apparently were not detrimental to the P release and uptake rates which occurred during Phase B.

By observing reaction rates as opposed to removal percentages, it becomes clear that nitrates had no effect on the ability of the aerobic biomass to remove phosphates. At first glance, it appears that orthophosphate removal was high only at times of low nitrate concentrations in the aerobic reactor (with consequent low nitrate levels in the sludge returned to the anaerobic reactor via the sludge conditioning reactor). On examining individual reaction rates however, in particular P uptake rates in the aerobic reactor, it appears that nitrates did not play a role in reducing the P removal capacity of the system. This is clearly shown by considering the results of Day 58B and 17B. On Day 58B, high nitrate levels (between 20 and 35

mg-N/L) occurred in the aerobic reactor and the effluent, while at the same time orthophosphate removal was only 38% (based on the influent and the aerobic reactor). The orthophosphate reaction rate on this day in the aerobic reactor was 1.30 mg-P/g MLSS/h. On Day 17B, at at slightly lower aerobic orthophosphate uptake rate of (1.09 mg-P/g MLSS/h), 92% P removal occurred, while at the same time nitrate concentrations in the aerobic reactor were comparable to those of Day 58B (between 15 and 20 mg-N/L). Thus, the high nitrate content in the aerobic reactor, at comparable orthophosphate reaction rates, caused two very different removal percentages to occur. The two days were operated at similar conditions except the system SRT on Day 17B was between 56 and 86 days, whereas on Day 58B the system SRT was maintained at 54 days. Therefore, there is strong evidence provided by the results obtained during this phase that when the SRT is sufficiently long, nitrate levels in the aerobic reactor are not of consequence, as long as the anaerobic reactor influent nitrates are sufficiently curtailed. The mass of P removed by the system was 0.102 g on Day 58B and 0.178 g on Day 17B (based on nominal flow rates). Thus removal amounts and percentages appear related.

Comparison of orthophosphate reaction rates in the aerobic reactor, and removal percentages determined for individual days, does not show a correlation due to fluctuation of the data. Thus process optimization was not acheived during this phase of the study. In general however, high removals are only possible when reaction rates are generally high. This can also be said of the

anaerobic and the sludge conditioning reactors, except in this case, the reaction rates constitute the release of orthophosphates.

The significantly increased P release and uptake rates during Phase B occurred simultaneously with a significant increase of system solids retention time. It appears that the microorganisms responsible for P removal, prior to Phase B, were washed out. During Phase B, ample time was given the bacteria to become involved in the removal mechanism. Phase B results indicate that, once the sludge age reaches a point where enhanced P removal is evident, a reduction in the SRT (from 86 to 54 days in this study) is possible without any loss of P removing biomass activity. Further lowering of the sludge age is detrimental however, as indicated by results subsequent to Day 48B. A reduction in sludge age on Day 40B was followed by eight days where the P removal was maintained at enhanced levels. Although fluctuations were evident, the orthophosphate release and uptake rates were also relatively high (similar to rates exhibited during the first five weeks of Phase A). Subsequent to Day 48B however, both the P release and the P uptake rates returned to typical Phase A levels in all three reactors. The percentage removals also gradually diminished, until Day 62B when the aerobic reactor biomass was involved in the release of orthophosphates. Nitrates returned to the anaerobic reactor remained at the same, relatively low, level encountered during the period of enhanced P removal, in that the denitrification by the sludge conditioning reactor biomass did not suddenly stop.

# CHAPTER 6

# CONCLUSIONS AND RECOMMENDATIONS

## 6.1 Conclusions

The anaerobic-aerobic system studied is capable of enhanced phosphorus removal, provided a sufficiently long solids retention time is maintained. At a system SRT of approximately 86 days, up to 98% of phosphates was removed. Once enhanced removal is achieved, subsequent reduction to a lower system SRT (54 days in this study) appears to have no deleterious effects on the phosphorus removal capabilities of the system.

In general, high orthophosphate concentrations were a prerequisite for obtaining low P concentrations in the aerobic reactor. Orthophosphate reaction rates followed this pattern also, in that high release rates in the anaerobic reactor yielded high uptake rates in the aerobic reactor. These phenomena were only observed at high SRT's however. Orthophosphate release rates of up to 3.34 mg-P/g MLSS/h occurred in the anaerobic reactor, while the aerobic reactor phosphate uptake rates reached a maximum of 2.05 mg-P/g MLSS/h during the period of enhanced phosphorus removal by the system. Although the release and uptake rates in the anaerobic and aerobic reactors reached higher levels during Phase A of this study (4.72 and 4.37 mg-P/g MLSS/h respectively), enhanced system phosphorus removal was not a norm.

Typical anaerobic reactor nitrate concentrations were in the neighbourhood of 0.1 mg-N/L, thus precluding the possible interference of nitrates with the phosphate release phenomenon throughout the study. Denitrification rates in this reactor reached a maximum of 7.06 mg-N/g MLSS/h during Phase A. The installation of the sludge conditioning reactor virtually eliminated the denitrification activity in the anaerobic reactor; the release of phosphates did not occur, as a consequence, however.

The presence of nitrates or high nitrification rates in the aerobic reactor had no direct bearing on the phosphate uptake rates in this unit. At comparable nitrate levels and nitrification rates (15-35 mg-N/L and 0.7-0.9 mg-N/g MLSS/h respectively), the phosphate removals were 38% and 92%. The phosphate uptake rates corresponding to these removal percentages were 1.30 and 1.09 mg-P/g MLSS/h respectively.

The $\Delta P/\Delta COD$ ratios were, in general, high during periods of enhanced phosphorus removal by the system. No conclusions could be made regarding the effects of temperature changes on P uptake, since these changes were implemented during Phase A of the study, when enhanced P removal was not a norm. The role played by factors such as COD, MLSS, alkalinity, pH, and trace metals on the P release/uptake mechanism were not substantiated from results of this study. Extensive data fluctuations was caused by the difficulty in achieving system equilibrium.

## 6.2 Recommendations

The major problems encountered during this study involved the effects of size on the operation of the system. Plugging of tubing and outlets was a frequent occurrence, usually severely disrupting the performance of the system. A larger system would preclude these problems. Further, the time lag involved in observing the effects on the performance of the system, resulting from a physical or operating change, usually was so long that inevitably other mishaps (such as spills) occurred in this time span. It was thus difficult to separate the effects of the actual induced change and effects of accidental occurrences resulting from equipment malfunction. Because of the presence of relatively large particles in raw sewage, it is recommended that a continuous flow treatment model should be operated on the pilot plant scale, rather than on a bench scale.

Many additional studies can be pursued to better define the operation of this phosphorus removal treatment scheme. Preferably, a "dual" system should be employed where one stream of raw sewage enters two systems, one of which is maintained as the "baseline" unit, while the other undergoes the required operational changes.

To substantiate the findings of this study, the SRT in future studies should be carefully maintained at various levels. Gradual changes in SRT should be made and the resulting phosphorus removals should be analyzed more frequently. The minimum SRT necessary for enhanced phosphorus uptake should be

determined. Once enhanced phosphorus uptake is realized, the SRT should be further increased to determine if there is a critical SRT beyond which removal will not take place. Also, a gradual decrease in SRT should then be investigated to substantiate the finding of this study that, once enhanced removal of phosphorus has been initiated by the system, a reduction of SRT does not reduce the removal capabilities.

Additional studies should be directed at the optimization of phosphorus removal using a two-reactor system only (an anaerobic-aerobic scheme). Results from this study indicate that the sludge conditioning reactor does not appear necessary for enhanced phosphorus removal to occur.

The role of the hydraulic retention time should also be investigated. In addition, future reaserch should be directed at discerning the effects of such parameters as influent COD, anaerobic reactor ORP, DO of the aerobic reactor, and MLSS concentrations.

Improving laboratory scale equipment and operation is a major concern. Also, it should be determined how well such units compare to similar pilot and full-scale operations.

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# APPENDICES

# APPENDIX I

## SYNTHETIC SEWAGE

| | |
|---|---|
| NaCl | 12,000 mg |
| KCl | 2,800 mg |
| $NaHCO_3$ | 67,200 mg |
| $CaCl_2$ | 2,800 mg |
| $MgSO_4$ | 2,000 mg |
| Nutrient Broth | 10 g |
| Soap | 20,000 mg |
| Urea | 12,000 mg |
| Starch | 40,000 mg |
| $Na_2HPO_4$ | 10,000 mg |
| $Al_2(SO_4)$ $18H_20$ | 10,000 mg |

The above amounts were dissolved in 4L of water to yield a theoretical $BOD_5$ of 20,000 mg/L.

# APPENDIX II

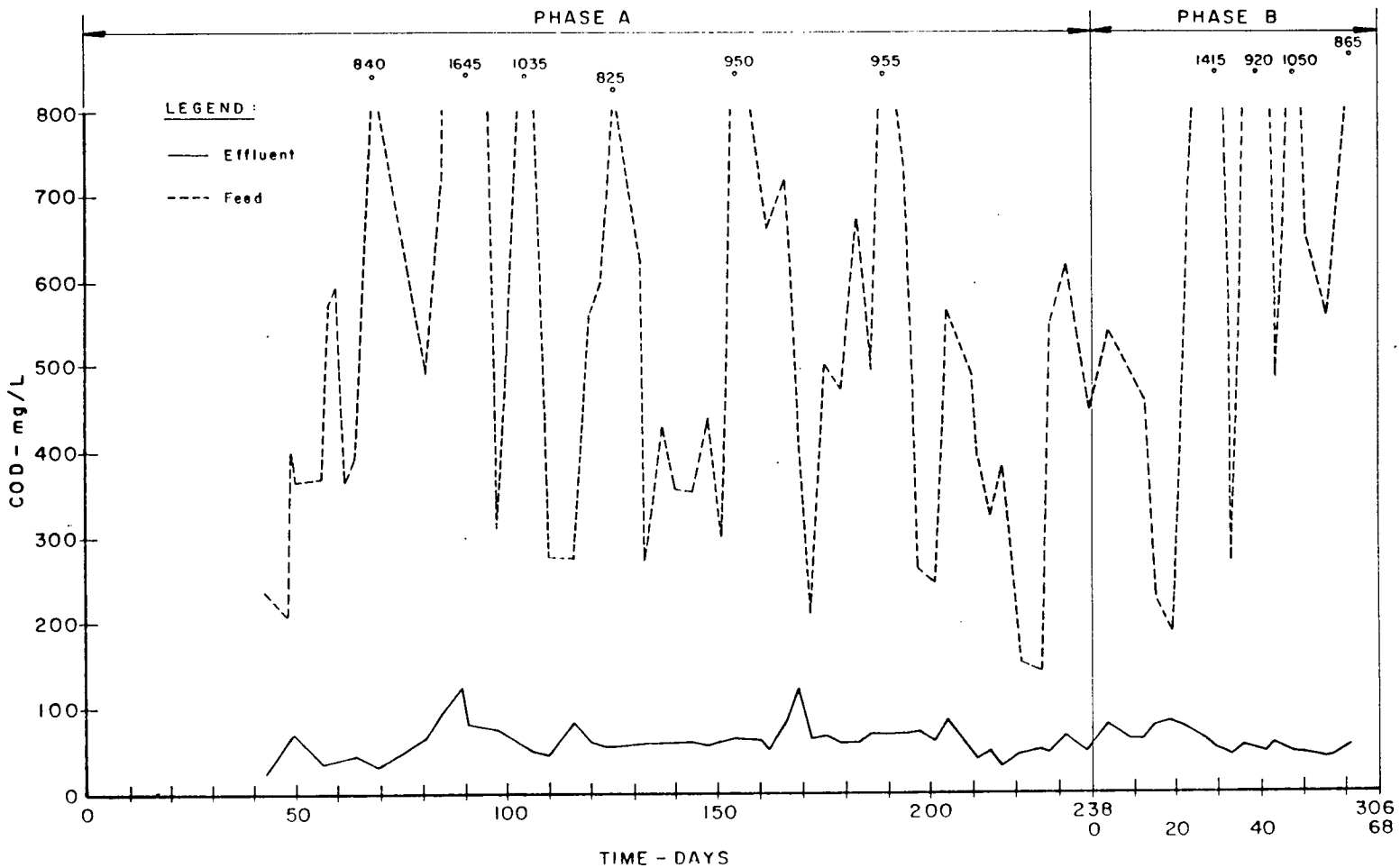


FIG. II-1: FEED AND EFFLUENT COD CONCENTRATION.

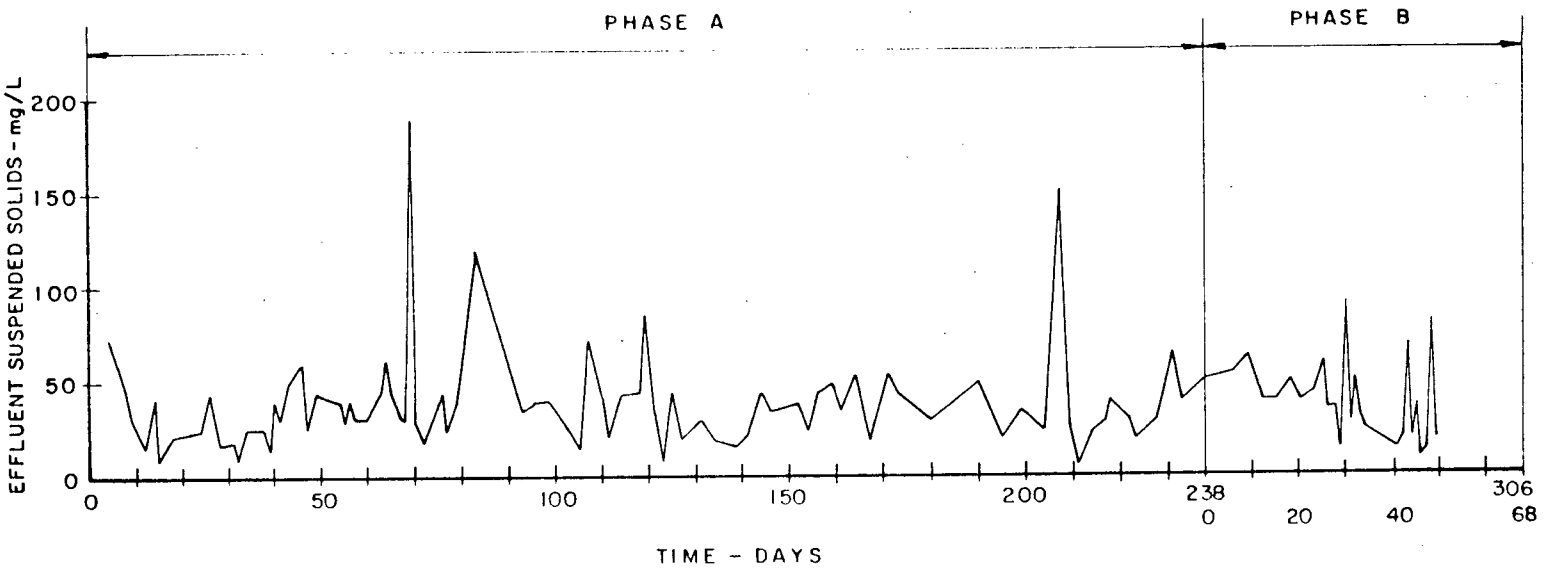


FIG. II - 2: EFFLUENT SUSPENDED SOLIDS.

TABLE II-1

pH

| DAY | FEED | ANAEROBIC REACTOR | AEROBIC REACTOR | SLUDGE CONDITIONING REACTOR | EFFLUENT |
|---|---|---|---|---|---|
| 23A | 7.3 | 7.5 | 7.5 | | 7.6 |
| 43A | 7.15 | 7.40 | 7.05 | | 7.12 |
| 57A | 7.19 | 7.34 | 7.15 | | 7.22 |
| 64A | 7.15 | 7.40 | 6.90 | | 7.17 |
| 69A | 7.10 | 7.47 | 6.60 | | 6.80 |
| 88A | 7.05 | 7.37 | 7.0 | | 7.18 |
| 99A | 7.35 | 7.52 | 7.28 | | 7.47 |
| 124A | 7.30 | 7.40 | 7.20 | | 7.20 |
| 186A | 7.56 | 7.30 | 6.65 | 6.87 | 6.83 |
| 8B | 7.75 | 7.32 | 6.51 | 6.80 | 6.70 |
| 23B | 7.58 | 7.30 | 7.54 | 6.94 | 7.68 |
| 47B | 7.47 | 7.37 | 7.08 | 7.22 | 7.29 |

ALKALINITY (mg/L as $CaCO_3$)

| | | | | | |
|---|---|---|---|---|---|
| 23A | 150 | 175 | 160 | | 160 |
| 32A | 180 | 180 | 173 | | 165 |
| 43A | 168 | 133 | 78 | | 68 |
| 57A | 208 | 131 | 83 | | 57 |
| 64A | 213 | 156 | 65 | | 54 |
| 69A | 203 | 159 | 37 | | 31 |
| 88A | 207 | 158 | 88 | | 75 |
| 99A | 161 | 158 | 120 | | 100 |
| 124A | 142 | 157 | 132 | | 118 |
| 186A | 150 | 131 | 52 | 100 | 37 |
| 8A | 237 | 221 | 38 | 168 | 22 |
| 23A | 185 | 225 | 224 | 239 | 186 |
| 47A | 245 | 298 | 105 | 232 | 99 |

TABLE II-2

RAW SEWAGE HEAVY METALS

| | LULU ISLAND SOURCE (mg/L) | MAPLE RIDGE SOURCE-PHASE A (mg/L) | | MAPLE RIDGE SOURCE-PHASE B (mg/L) | |
|---|---|---|---|---|---|
| | | Low | High | Low | High |
| Iron | 5.5 | 0.47 | 1.91 | 0.59 | 2.59 |
| Nickel | 0.176 | 0.002 | 0.009 | 0.002 | 0.005 |
| Chromium | 0.208 | 0.004 | <0.01 | 0.002 | 0.003 |
| Lead | 0.370 | 0.012 | 0.05 | 0.012 | 0.037 |
| Copper | 0.26 | 0.13 | 0.34 | 0.12 | 0.34 |
| Zinc | 0.69 | 0.06 | 0.26 | 0.08 | 0.19 |
| Manganese | 0.17 | 0.04 | 0.09 | — | — |
| Cadmium | 0.019 | 0.001 | <0.003 | — | — |
| Calcium | 17.0 | 6.3 | 12.1 | 7.96 | 9.47 |
| Manganesium | 4.74 | 1.47 | 3.10 | 1.98 | 2.52 |

TABLE II-3

TOTAL KJELDAHL NITROGEN (mg-N/L)

| DAY | FEED | EFFLUENT |
|---|---|---|
| 57A | 49.6 | 9.7 |
| 74A | 51.4 | 3.2 |
| 98A | 49.6 | 19.5 |
| 110A | 38.4 | 20.4 |
| 120A | 37.0 | 16.0 |
| 132A | 54.3 | 24.2 |
| 148A | 50.4 | 35.0 |
| 160A | 40.6 | 17.2 |
| 176A | 49.4 | 18.7 |
| 183A | 46.1 | 9.5 |
| 204A | 58.6 | 19.4 |
| 211A | 51.2 | 15.0 |
| 232A | 69.9 | 12.2 |
| 7B | 55.7 | 4.1 |
| 21B | 50.2 | 30.4 |
| 34B | 50.3 | 31.5 |
| 53B | 42.7 | 8.1 |

TABLE II-4

TOTAL PHOSPHATES (mg-P/L)

| DAY | FEED (Unfiltered) | EFFLUENT (Filtered) |
|---|---|---|
| 29A | 6.7 | |
| 34A | 12.5 | 5.90 |
| 39A | 8.36 | 5.28 |
| 82A | 16.6 | 5.6 |
| 103A | 6.64 | 4.99 |
| 105A | 9.18 | 5.80 |
| 126A | 9.68 | 4.79 |
| 181A | 8.06 | 4.64 |
| 238A | 13.0 | 6.9 |
| 19B | 9.50 | 1.85 |
| 25B | 11.40 | 3.27 |
| 31B | 10.2 | 6.25 |
| 34B | 9.74 | 4.6 |
| 41B | 6.50 | 1.59 |
| 48B | 10.2 | 2.9 |
| 55B | 8.54 | 3.94 |
| 66B | 8.8 | 10.1 |

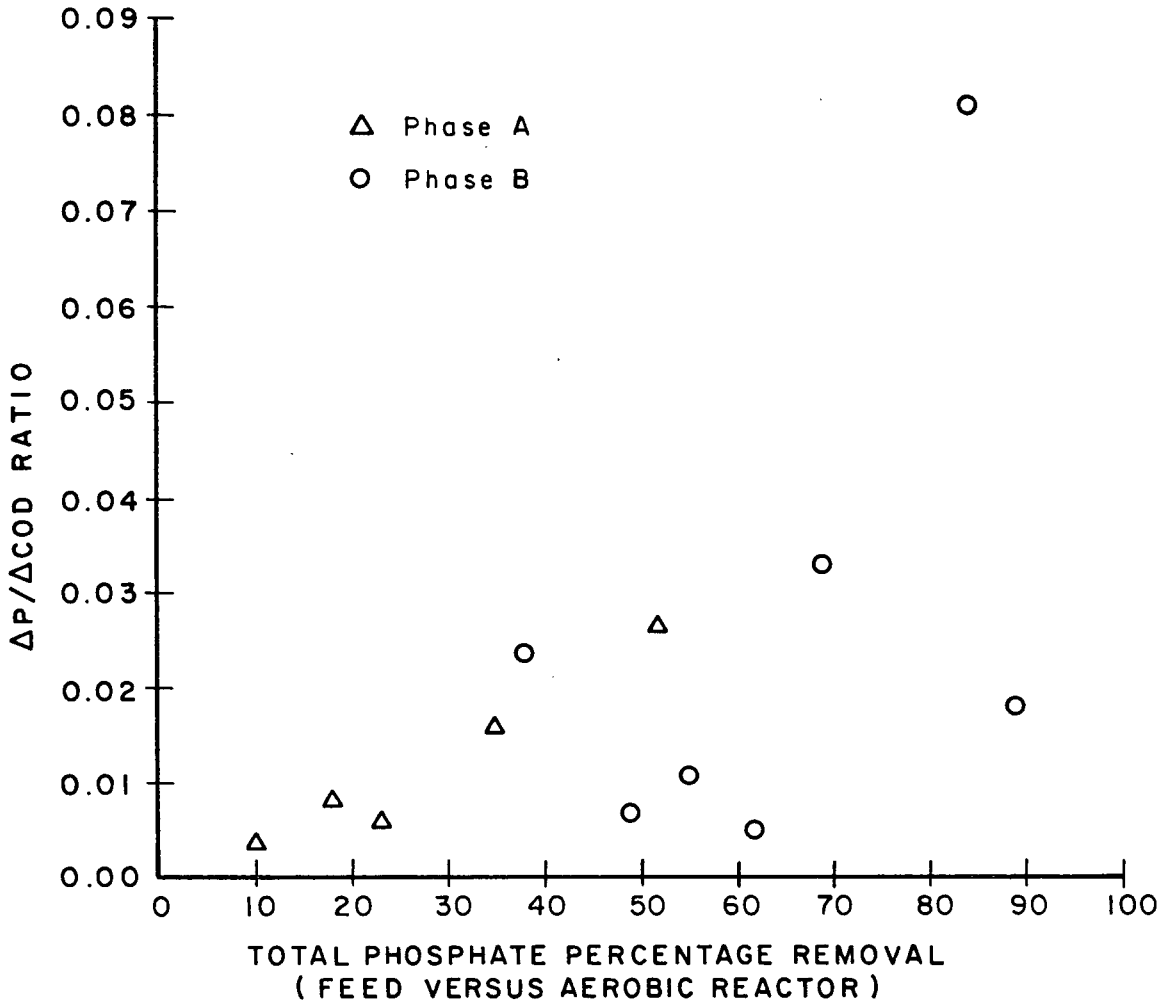


FIG. II - 3 : ΔP/ΔCOD RATIO VS. %P REMOVED BY SYSTEM.

TABLE II-5

HEAVY METALS SCAN THROUGH SYSTEM

(DAY 61B) (mg/L)

| ELEMENT | FEED | ANAEROBIC REACTOR | AEROBIC REACTOR | SLUDGE CONDITIONING REACTOR | EFFLUENT |
|---|---|---|---|---|---|
| Calcium | 10.5 | 38.1 | 37.5 | 100. | 8.3 |
| Copper | 0.27 | 4.96 | 6.75 | 9.5 | 0.13 |
| Iron | 1.34 | 21.8 | 23.0 | 82.0 | 0.32 |
| Zinc | 0.13 | 2.78 | 2.67 | 10.75 | 0.07 |
| Magnesium | 2.95 | 21.6 | 22.6 | 75.5 | 1.85 |
| Lead | 0.026 | 0.45 | 0.47 | 1.75 | 0.012 |
| Nickel | 0.025 | 0.538 | 0.567 | 1.90 | 0.020 |
| Chromium | 0.010 | 0.463 | 0.383 | 1.28 | 0.007 |
| Cadmium | <0.001 | 0.010 | 0.010 | 0.036 | <0.001 |