A LABORATORY STUDY

ON THE

ANAEROBIC DIGESTION OF

YEAST WASTE

BY

HAROLD IAN HENRY RICHTER

SUBMITTED IN PARTIAL FULFILMENT OF
THE REQUIREMENTS FOR THE DEGREE OF
MASTER OF SCIENCE IN ENGINEERING

UNIVERSITY OF CAPE TOWN
JANUARY 1973

ABSTRACT

This thesis reports the establishment of a new type of internally clarified digester as a means of treating a strong industrial waste. Factors pertaining to both the system and the waste itself were investigated.

A batch system was used to determine the susceptibility of the waste chosen, viz. that from the first separator of a local yeast factory, to anaerobic digestion.

The reactor-clarifiers were constructed and operated at a wide range of loadings. A constant hydraulic retention time was maintained for each of the digesters and sufficient sludge withdrawn daily to maintain a constant sludge concentration over the range of loadings used.

The ability of each of the parameters used to monitor digestion, to predict periods of digester imbalance was investigated.

A mathematical model was derived to describe the process and the kinetic parameters related to the anaerobic digestion of yeast waste evaluated.

Possible inhibitory substances present in the waste and/or generated during digestion and their influence on the system were investigated.

A C K N O W L E D G E M E N T S

The author wishes to express his gratitude to Dr. G.S. Hansford for his assistance and guidance throughout this investigation. The advice obtained from Dr. H.O. Buhr, Professor G. v. R. Marais during the study is also sincerely appreciated.

Valuable assistance was given by Mr. K.E. Wheeler and Mr. G.P.E. Corewijn during the construction of the experimental apparatus.

The financial assistance of the Council for Scientific and Industrial Research made this investigation possible and is gratefully acknowledged.

C O N T E N T S

L I S T O F F I G U R E S

operation and the development of various anaerobic processes, is presented in Chapter 2.

The development of a batch system to evaluate the feasibility of treating yeast waste by conventional anaerobic means is discussed in Chapter 3. The development of the reactor-clarifier system which was used to establish the kinetic parameters is also presented. A description of the operating methods for both the batch and continuous processes concludes the chapter.

In Chapter 4, the general results relating to the typical anaerobic system parameters are presented and where possible compared to values obtained for the treatment of other industrial wastes. The relative significance of the various parameters used in the daily monitoring of digester condition, is evaluated in this chapter.

A brief review of the literature on the application of kinetics to anaerobic digestion can be found in Chapter 5. The development of a model to describe the anaerobic process is described and its practical application to full scale digesters discussed. The model is adapted to describe the 'reactor-clarifier' system used in this investigation. Finally, the kinetic parameters obtained from this model are evaluated and compared to other values in the literature.

Possible sources of inhibition relating to the present study are discussed in Chapter 6. An attempt to evaluate the inhibitory effects of these materials is also described in this chapter.

TABLE 1.1. Characteristics of First Separation Waste.

PARAMETER	RANGE
Biological Oxygen Demand	25 000 - 38 000
Chemical Oxygen Demand	45 000 - 70 000
Oxygen Absorbed	17 000 - 25 000
Total Dissolved Solids	77 000 -100 000
Total Volatile Dissolved Solids	56 000 - 75 000
Total Suspended Solids	-
Total N	800 - 1 000
NH_3-N	300 - 400
Sulphates	2 800 - 4 500

The following conclusions were arrived at after the completion of the investigation :

(1) approximately 75 per cent. of the yeast waste was readily susceptible to anaerobic treatment. The remaining 25 per cent. did not appear to be degradable;

(2) the reactor-clarifier proved to be suitable for the anaerobic treatment of strong industrial wastes. The system was operated satisfactorily over a wide range of loadings;

(3) total gas production was found to be the most reliable indicator of digester performance;

(4) the kinetic parameters pertaining to the anaerobic treatment of yeast waste were found to be of the same order of magnitude as other published constants;

(5) attempts to relate a limiting loading concentration to potassium, chlorides and sulphides were inconclusive.

CHAPTER 2

LITERATURE SURVEY AND BACKGROUND THEORY

2.1 INTRODUCTION TO LITERATURE SURVEY.

A survey of the literature concerning the anaerobic process, relating in particular to the objectives listed in Chapter 1, was carried out.

Consequently the literature survey as presented in this chapter is divided into the following sections :

(a) A brief description of the anaerobic process, and an indication of the areas where its application is favoured.

(b) An elementary discussion of the microbiology and biochemistry applying to the anaerobic system.

(c) A survey of the necessary environmental and operating conditions required to maintain satisfactory process performance.

(d) A summary of the parameters used to monitor digestion and their respective ability to indicate the start of process failure.

(e) The survey is concluded with a review of the development of anaerobic processes from the septic tank until the various contact systems used at the present time.

The portion of the literature survey relating to the application of kinetics to anaerobic digestion is presented with the kinetics pertaining to the present study in Chapter 6.

2.2 BRIEF DESCRIPTION OF PROCESS.

2.2.1 Introductory Definition.

Malina has defined anaerobic waste treatment as a process consisting of the use of various micro-organisms, under anaerobic conditions, to stabilize organic wastes by conversion to gaseous end products (M3).

2.2.2 Advantages of Anaerobic Digestion.

The advantages to be derived from the use of anaerobic digestion, as opposed to conventional aerobic processes, have been summarized by McCarty (M8) :

(a) a low cellular yield. The sludge disposal problem is significantly reduced because only a small portion of the waste is converted to new cells;

(b) as a result of the low level of microbial growth, there are correspondingly less nutrient requirements in the anaerobic system;

(c) the conversion of a major portion of the degradable waste to methane, a useful end product;

(d) no oxygen requirements and consequent reduction in capital expenditure;

(e) the ability to handle more concentrated wastes than practically possible for aerobic processes.

2.2.3 Disadvantages of Anaerobic Digestion.

The anaerobic process does have disadvantages which have limited its use in certain cases (M8) :

(a) the slow growth rate of the anaerobic organism has necessitated long retention times in the digesters and consequently resulted in large volume requirements. The slow growth rate further limits the rate at which the process can adjust to changes in the environment, particularly to step changes in loading. This is a serious limitation of the anaerobic process;

(b) a susceptibility to toxic materials in the waste. This is commonly a major reason for the rejection of the anaerobic process as a method of treating industrial wastes;

(c) the relatively high temperatures required for digestion, often in the region of 30 - 35°C. Dilute wastes may not produce sufficient methane during digestion to maintain these temperatures.

The development of the anaerobic contact process has offset some of these disadvantages to a considerable extent, as shown in section 2.6.7.

2.2.4 Summary.

Initially the advantages of anaerobic digestion appear to be substantial. However, the process has suffered from a poor record of performance stability in the past.

The lower the concentration of a particular waste, the more prominent become the disadvantages of relatively high temperature requirements and long retention times. The choice of treating such wastes anaerobically becomes uneconomical and/or unfeasible.

In the past, anaerobic digestion was used primarily to stabilize sewage sludge. Recently, however, increased application of the anaerobic process has been reported in the treatment of organic industrial wastes, Cillie et al. (C2). The same writers have suggested that anaerobic digestion would appear to be uneconomical, for treating wastes with a COD below 4 000 mg/l.

Such wastes are frequently treated by aerobic means. Highly seasonal wastes are often treated by various lagooning systems, provided sufficient land is available. Because of the large volumes characteristic of the latter systems, the effects of sudden shock loads or presence of inhibitory substances is often minimized. This is a significant advantage of such systems compared to conventional anaerobic processes.

Should economics or sludge disposal problems preclude the use of aerobic systems, and siting prove inadequate for the establishment of a lagoon system, the use of the anaerobic contact process, section 2.6.7, could be investigated. Schroepfer et al. (S4) have reported successful use of such a system for the treatment of meat packing wastes.

The volatile solids content and well balanced supply of nutrients in sewage sludge, together with the uniformity of loading commonly experienced in most sewage systems, enhances the selection of conventional anaerobic processes for such treatment.

Generally the application of anaerobic digestion is more suited to the treatment of concentrated industrial wastes.

However, the presence of toxic materials in the waste, nutrient deficiency and erratic loading rates, have frequently prevented the treatment of concentrated wastes by anaerobic digestion. In such cases the use of lagooning as a form of pretreatment is popular. The effluent from such a preliminary lagoon can be treated by further systems and may, as such, act as a buffer to shock loads. If economics permit, the lagoon effluent can be transfered directly to a sewage system.

2.3 MICROBIOLOGY AND BIOCHEMISTRY.

2.3.1 General Introduction.

The occurence of anaerobic digestion in nature is comparatively widespread. In a review of anaerobic digestion, Toerien and Hattingh (T1) reported that methanogenic bacteria have been found in ordinary garden soil, black mud, in the rumen of ruminants, in marsh and lake areas, as well as in sewage and sewage treatment processes.

The microbial decomposition of complex organic substrates provides a source of energy to the bacterial cells, which convert a large portion of the organic material to methane and carbon dioxide.

Malina (M3) has reported that the microbial population consists of various facultative and anaerobic bacteria which use chemically-bound oxygen in the form of organic molecules, carbon dioxide, nitrates, sulphates or organic materials, in order to reduce them to methane, ammonia, hydrogen sulphide and reduced organic compounds respectively.

The facultative micro-organisms are able to use molecular oxygen during metabolism and this group of organisms can therefore protect the strictly anaerobic bacteria from small amounts of free, dissolved oxygen which enter the system with the feed (M3).

2.3.2 Reaction Steps.

Willimon and Andrews (W3), amongst others, have shown anaerobic digestion to be a sequence of steps as shown in Figure 2.1.

In the first step, liquefaction takes place and the insoluble organic material is converted to soluble material by extracellular enzymes. Liquefaction is confined to sludge digestion and those wastes with insoluble organic material present.

In the treatment of soluble organic wastes digestion proceeds from the second stage as described below.

In the second stage a heterogeneous group of facultative and anaerobic bacteria, commonly termed the 'acid-formers', convert the soluble proteins, carbohydrates and fats into various intermediate compounds, primarily fatty acids.

Toerien et al. (T1) have reported similar digestion paths in both anaerobic digesters and the rumen of herbivorous animals. They suggest that as the main bacterial groups in the rumen are obligate anaerobes, the role of the latter in the acid-forming phase of anaerobic digestion has been under-estimated in the past.

The strictly anaerobic bacteria then utilize the organic acids in the final stage, converting them into methane and carbon dioxide (M3).

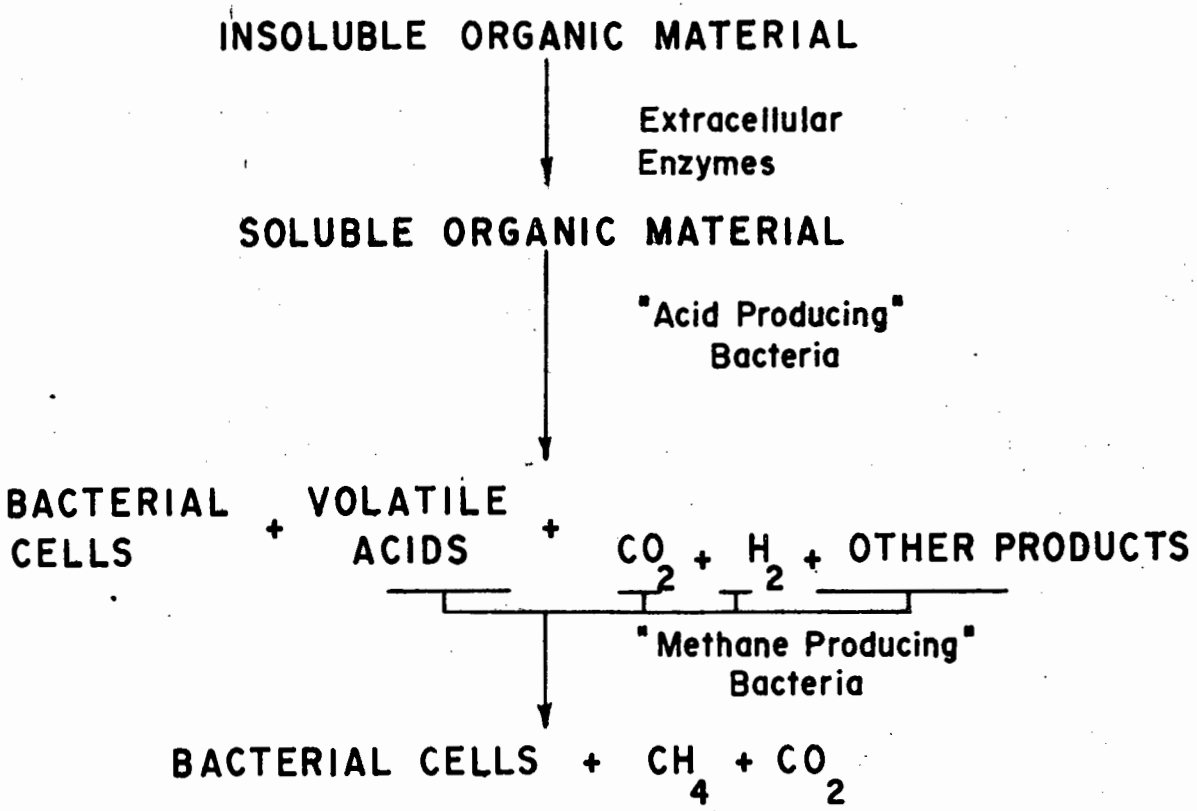

Figure 2.1 Simplified concept of the anaerobic decomposition of organic wastes.

2.3.3 Acid-Forming Stage.

McCarty (M8) described the 'acid-forming' stage of anaerobic treatment as a waste conversion step. Virtually no stabilization, of the Biological Oxygen Demand (BOD) of the organic matter in solution, takes place in this 'acid-forming' stage. A simple change of form takes place, a portion being converted to various end-products, such as organic acids, and the other portion being converted to new bacterial cells.

Short-chain, carboxylic acids are the major products of acid fermentation. These volatile acids in turn provide a substrate for the methane forming bacteria.

The primary acids produced are acetic, propionic and butyric acids. However, smaller quantities of formic, valeric, isovaleric and caproic acids are frequently found during anaerobic treatment.

The 'acid-forming' bacteria are relatively tolerant to changes in pH and temperature and have a faster growth rate than the methane bacteria (M8), (M3).

2.3.4 Methane-Forming Stage.

It is in the final methane-forming stage that true stabilization of the oxidisable organics takes place. McCarty (M8) has shown this stabilization to be directly proportional to the amount of methane produced from theoretical considerations, supported by experimental evidence.

The methane-forming bacteria are all strict anaerobes. Each species of methane bacteria can only ferment a relatively

restricted group of simple compounds to methane or lower chain acids. Consequently a number of different species of methane-formers are necessary to completely stabilize a waste complex.

Little is known about the pathways followed by these bacteria, Jeris and McCarty (J2) concluded that about 70 per cent. of the methane produced in the anaerobic digestion process, results from the degradation of acetic acid. Further studies by McCarty et al. (M7) indicated that approximately 85 per cent. of the total methane formed during the complete treatment of a complex waste was due to the fermentation of propionic and acetic acids. The significance of the fermentation of these two acids to the overall process is therefore confirmed. It should be emphasized that the above figures were based on the fermentation of sewage sludge and could therefore differ for other wastes.

Smith (S7), (S8), did further work indicating that approximately 73 per cent. of the methane produced by sludge fermentation could be attributed to acetic acid. He further suggests that a large portion of the remaining methane is formed from carbon dioxide and hydrogen. This postulation is backed by extensive experimental studies.

Unlike the 'acid-forming' bacteria, the methane bacteria are extremely sensitive to changes in temperature and pH.

The most important methane bacteria have low growth rates. Minimum sludge retention times of four days or longer are required for their growth (M8).

Malina (M3) emphasizes that the low rate of acid utilization constitutes the rate limiting step of the process. This step governs the design of anaerobic digesters.

2.3.5 Process Imbalance.

McCarty (M8) has defined the anerobic system to be stable, and functioning satisfactorily, when the acids are converted as fast as they are produced. Unstable conditions result when the acids are produced at a faster rate than they are converted into gases (M8).

The biological reactions can be inhibited and may cease altogether if unstable conditions are permitted to persist.

These unstable conditions can result from :

(a) too low a relative concentration of methane bacteria in the system. This situation often arises during periods of digester start-up and is further apparent when sudden increases of loading are made on the system. The methane bacteria are unable to cope with the sudden increase in volatile acids resulting from such situations;

(b) unfavourable environmental conditions causing a decrease in the activity of the methane bacteria. This can occur particularly in the treatment of industrial wastes, due to the presence of certain materials in inhibitory concentrations.

To maintain balanced conditions in a digester it is therefore essential to have a thorough understanding of the optimum environmental conditions necessary for these bacteria.

2.4 ENVIRONMENTAL CONDITIONS.

2.4.1 pH.

One of the most important environmental requirements for the methane bacteria is that a satisfactory pH be maintained.

As mentioned earlier digester imbalance is accompanied by an increase in the concentration of volatile acids in the system. If these acids are allowed to build up to sufficiently large concentrations they may depress the pH to a level unsuitable for the methane bacteria.

In a summary of recent research, McCarty (M9) found that a pH range of approximately 6,6 to 7,6 is generally considered satisfactory for anaerobic digestion to proceed. Digesters have operated successfully outside of this range, but generally the efficiency of digestion tends to decrease rapidly as the pH moves beyond these limits.

The range of pH suitable for the methane bacteria is further dependent on the components of the waste, particularly the quantity of nitrates present. The significance of ammonia in digester systems is discussed in section 2.4.7.

Albertson (A3) found the optimum pH for methane fermentation to be 7,1, while McCarty (M9) suggests an optimum range of 7,0 to 7,2.

2.4.2 Temperature.

Temperature greatly affects microbial activity. It has been found that the rate of microbial metabolism will approximately double for every 10°C rise, up to a certain limiting temperature (L4).

Fair and Moore (F1) established that there were two temperature ranges favourable for practical anaerobic digestion :

(a) the mesophilic range 30 - 37°C;

(b) the thermophilic range 55 - 60°C.

There is some disagreement concerning the value of thermophilic digestion. Some writers feel that gas yield and volatile matter destruction are greater under thermophilic conditions (H3), (K1), while others indicate that there is no significant difference in the end products derived from mesophilic and thermophilic digestion (G1), (G4), (M1).

Thermophilic digestion has not been demonstrated to be either practical or economical as yet. A disadvantage of thermophilic conditions would be the possibility of thermal degradation of the bacteria if temperatures rose above the suggested range for thermophilic digestion. A tremendous decline in metabolism would be the result of such an occurence.

It does not follow that anaerobic treatment should occur at optimum mesophilic temperatures. Satisfactory anaerobic digestion can, and has taken place at lower temperatures, provided an adequate active biological concentration is maintained in the digester.

As shown in section 2.6.7, the anaerobic contact and similar processes can operate at lower temperatures than the conventional process. This permits the treatment of more dilute wastes where insufficient methane is evolved to maintain a conventional system at a temperature suitable

for digestion to proceed. Further for countries experiencing a high seasonal variation in ambient temperature, a significant advantage can be derived from the use of such a system.

The ability of the anaerobic contact process to operate at lower temperatures reduces the heat requirements, which might make waste treatment uneconomical during cold winter periods.

2.4.3 Alkalinity.

As mentioned in section 2.3.5, a characteristic of digester instability is an increase in the concentration of the intermediate volatile acids. The control of pH depends upon the maintenance of an adequate buffer system in the digester, in order to counter the tendency of these organic acids to depress the pH.

Alkalinity may either be generated in the digester itself, or be added to the system in the form of a suitable chemical. Pohland et al. (P3), (P4), (P5), in a series of articles, has described the function of alkalinity in the digester system and established an equilibrium model to predict the influence of various concentrations of acids and bases and their associated equilibria on the anaerobic environment. The systems discussed by Pohland (P3), include carbonic acid, ammonia, acetic acid and hydrogen sulphide. Their respective titration curves are shown in Figure 2.2. The maximum buffer capacity for a system is commonly known to occur when pH is equal to the pK of the species concerned. From Figure 2.2 it can be seen that within the pH range normally encountered during anaerobic digestion, viz. 6,8 - 7,4, only the first dissociation steps of the hydrogen sulphide and carbonic acid systems would be expected to contribute toward the buffering system present in the digester. As will be shown in

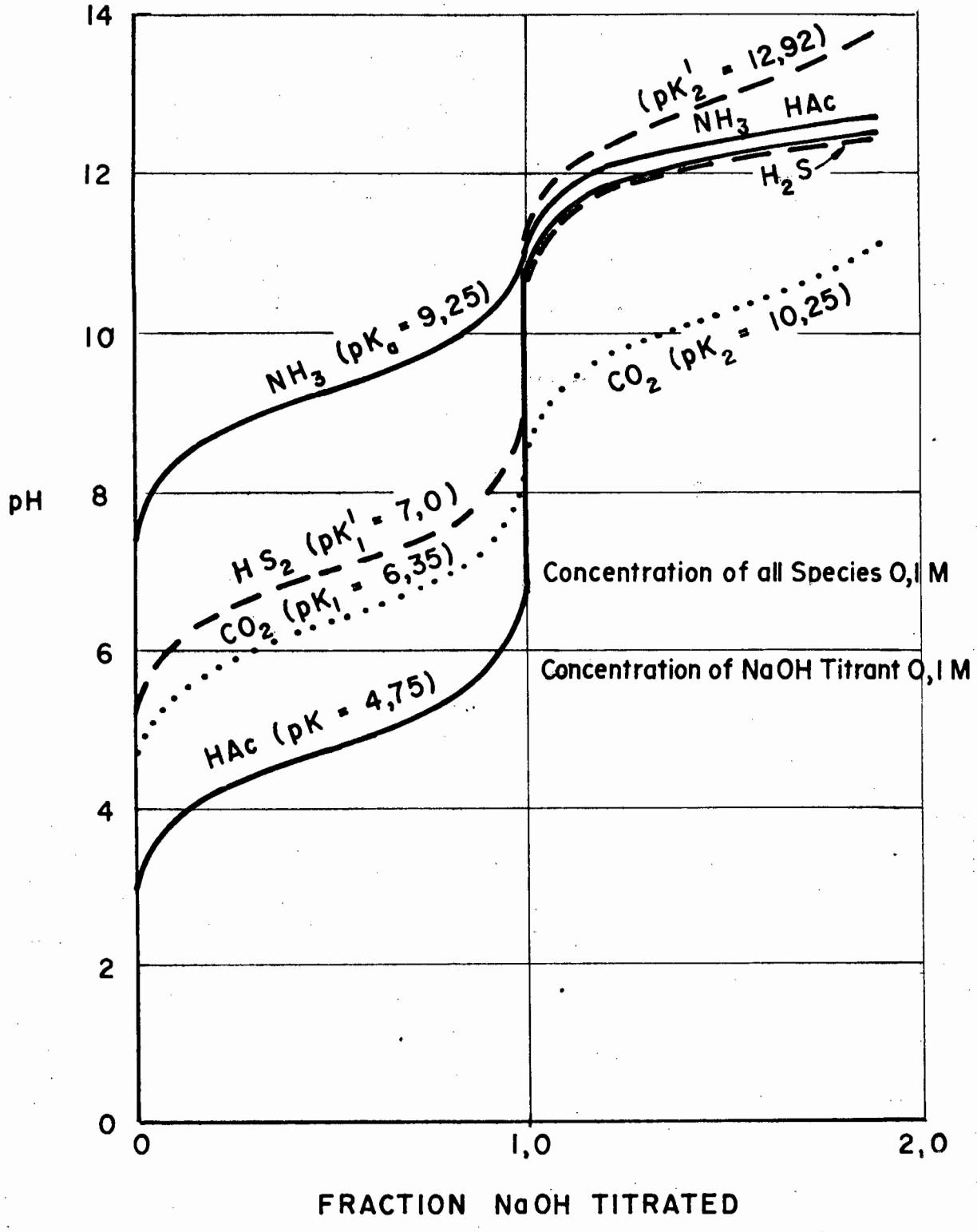

Figure 2.2 Titration of buffer systems prevalent in anaerobic digestion.

section 4.2, in the present study, sulphides contributed ten per cent. to the total buffer capacity when present in a concentration of 100 mg/l in the digesters. In cases where the concentration of sulphides in solution was low, the maintenance of a neutral pH can be attributed solely to the bicarbonate system.

Slight deviations from this condition can arise from the periodical generation of increased concentrations of volatile acids or ammonia in the system. If these systems are present in sufficient quantities, the pH tends to be displaced in their favour. An unusually high generation of ammonia results in a higher pH being established, while an increase in the presence of volatile acids, depresses the pH. An increase in the pH was experienced at higher loadings of the yeast waste under consideration, thus indicating the presence of ammonia in the system.

As described above, it has been shown that in most digestion systems, the major buffer system controlling the pH is the bicarbonate system. McCarty (M10) and Banta and Pomeroy (B1) have shown the bicarbonate alkalinity to be a function of pH and the carbon dioxide content of the digester gas. McCarty (M10) has shown that during normal periods of digestion, the bicarbonate alkalinity can be in the range of 1 000 to 5 000 mg/l as $CaCO_3$ depending on the pH prevailing and the carbon dioxide content of the gas in the digester environment. Rather than operate at extreme values in this range, Malina (M3) suggests an optimum value of 3 000 mg/l as $CaCO_3$ for the bicarbonate alkalinity.

Choice of Chemical Additives.

The use of lime for the purpose of controlling pH in digestion systems has been subject to wide controversy over the years.

McCarty (M9), has indicated that the major advantage to be derived from the use of lime for the purpose of pH control is the relative availability and cheapness of the material. Further the calcium cation has been shown by McCarty and McKinney (M6) to be less toxic than sodium, potassium, calcium and magnesium.

While conceding these advantages, McCarty (M9) points out that various difficulties have arisen with the use of lime in the past. These difficulties are related to the relative insolubility of the calcium salts which form in the digester.

When lime is added the biocarbonate alkalinity is increased initially in the following manner :

$$Ca(OH)_2 + 2CO_2 \rightarrow Ca(HCO_3)_2 \quad \ldots\ldots\ldots\ldots\ldots (2.1)$$

However, as soon as the solubility product for $CaCO_3$ is exceeded, further additions of lime merely result in the precipitation of the comparatively insoluble calcium carbonate as follows :

$$CA(OH)_2 + CO_2 \rightarrow CaCO_3 + H_2O \quad \ldots\ldots\ldots\ldots\ldots (2.2)$$

Consequently, lime additions beyond this point are of little use as the insoluble calcium carbonate is quite ineffective in the neutralization of excessive volatile acids or in the control of pH (M9).

From the above discussion it is clear that for the use of lime to be economical and effective, it is necessary to monitor closely the amount of lime added and the effect of these additions on the pH prevailing in the digester.

Sodium Bicarbonate.

The use of sodium bicarbonate, which has seldom been used in the past, has been advocated by McCarty (M9).

The major advantage of sodium bicarbonate over lime is its greater solubility. The relatively high solubility enables it to be dissolved before addition to the digester, thereby allowing more efficient mixing of the buffer system with the digester contents.

Although McCarty and McKinney (M6) have shown sodium to be more toxic than calcium, it is unlikely to be present in sufficient concentrations as to adversely effect digestion. The use of sodium bicarbonate to attain reasonably high alkalinities of 5 000 to 6 000 mg/l has been reported without any toxic effects being shown (M9).

Sodium bicarbonate would therefore appear to be extremely useful in laboratory scale digestion systems. There has, however, been very little mention in the literature concerning the application of sodium bicarbonate to full scale systems. The relatively high cost of using the salt on such a scale appears to present a formidable barrier to its application in the field.

2.4.4 Nutrient Requirements.

Sludge growth in the anaerobic process is significantly less than that in aerobic systems and nutrient requirements are correspondingly less for anaerobic digesters.

Nevertheless, the anaerobic bacteria do require various inorganic nutrients for bacterial growth and cell synthesis. Nitrogen and phosphorus constitute the primary requirements but various other materials in trace quantities are necessary for optimum growth (M3).

Nutrients are nearly always present in sufficient quantities in municipal sewage sludge. Industrial wastes are usually very specific in composition and it is sometimes necessary to add nutrients (usually phosphorus or nitrogen).

The nutritional requirements are related to the quantity of micro-organisms synthesized in the system. Loehr (L4) has reported the nitrogen and phosphorus requirements as being approximately 12 per cent. and 2 per cent. respectively of the volatile fraction of the bacterial cells synthesized.

Oxygen and oxidised material, such as nitrates, should not be added to anaerobic systems since they tend to be used as hydrogen acceptors in preference to the oxidised organic matter. Hence, when nutrient additives are required in an anaerobic system, they should be added in the reduced forms.

Nitrates should therefore not be used as nutrient additives. It has been shown that volatile acids have increased and gas production significantly reduced when nitrates have been used as a source of nitrogen in highly loaded digesters (C4), (S16).

Ammonium chloride, carbonate, hydroxide and phosphate have been used as suitable nitrogen sources for methane fermentation of nutrient deficient wastes.
Indiscriminate nitrogen additions should be avoided since McCarty has shown ammonia nitrogen concentrations above 1 500 mg/l to be inhibitory (M10).

McCarty (M11) feels that more research should be undertaken in this sphere of anaerobic digestion, in order to obtain a better understanding of the nutrient requirements of the methane bacteria. Speece and McCarty (S9) have found additions of iron, in concentrations between 20 and

60 mg/l, to be beneficial for digestion, but it is felt that various other inorganic and organic stimulants are needed to obtain maximum rates of digestion (M11).

As mentioned earlier in this section, industrial wastes are frequently nutrient deficient. The necessity of continually adding comparatively large amounts of nitrogen or phosphorus can make the use of conventional anaerobic treatment unattractive in such cases. Aerobic micro-organisms, with their higher growth rates, are more susceptible to nutrient requirements. Consequently, physical methods may have to be resorted to for the treatment of such wastes.

As shown in Chapter 4, the yeast waste being treated in this study was not nutrient limiting.

2.4.5 Mixing.

The major advantage to be derived from providing adequate mixing in a digester is the establishment of adequate contact between the organic material (food) and the micro-organisms.

Loehr (L5) and Malina (M2) have indicated other useful functions performed by mixing to include :

(a) the maintenance of uniform temperatures through the digester;

(b) the prevention of the formation of any scum layers;

(c) dispersion of potential metabolic inhibitors throughout the tank, avoiding the build-up of more concentrated regions.

Three methods have been employed to mix digesters :

(a) recirculation of sludge;

(b) mechanical agitation;

(c) gas recirculation.

The relatively large size of most digestion tanks tends to make it difficult for sludge recirculation alone to provide adequate contact between the organic material and the micro-organisms (L4).

However, sludge recirculation, assisted by gas recirculation, has been found by Garrison et al. (G2) to keep high-rate digesters adequately mixed.

Gas recirculation was used as a mixing system in this study, and was found to be ideally suited to laboratory scale digesters, as shown in Chapter 3.

2.4.6 Loading.

(a) Introduction.

Loading rates used in anaerobic processes are frequently higher than those applied to aerobic units.

However, this advantage is considerably offset by the poor record of process efficiency common to many anaerobic treatment plants. As mentioned in part (c), high-rate digestion is only feasible provided a relatively constant loading rate is maintained. Fluctuations in loading rates result in sudden increases in volatile acid concentrations, with which the slower growing methane bacteria are unable to cope. The deleterious effect of fluctuations in loading

has a more pronounced effect on digestion as the loading rates are increased.

McMahon (M14) has shown a sudden loading increase of 2,2 times to result in process failure, even at comparatively low operating rates.

(b) Parameters Used.

The loading rates on digestion tanks are usually expressed in terms of the amount of volatile solids added per digester capacity per day ($kgVS/m^3/day$).

For the treatment of soluble wastes the loading rates are often expressed in terms of the biological or chemical oxygen demand, BOD or COD respectively, i.e. kg $BOD/m^3/day$.

The above terms relate the amount of digestible organic matter available to the active microbial population, over a definite period of time. Since the number of active organisms are not readily determined, a unit volume of the digester contents is assumed to be related to the microbial population (M2). The significance of thorough mixing is self evident.

The use of different methods to express loading rates on a system can lead to confusion and makes comparison difficult. Although the purpose of digestion may vary from process to process, loadings and reductions are often meaningless for partially degradable wastes unless referred to some ultimate capacity of the digester. This is particularly relevant to the yeast waste under consideration where approximately one quarter of the waste was found to be not degradable.

(c) Comparative Loading Rates.

Up until comparatively recent times, loading rates recommended for anaerobic units treating sewage and industrial wastes of a similar nature, have been in the range of 0,48 to 1,12 kg VS/m^3/day (M2), (L4), (W1). This range of loadings has commonly been referred to as 'low-rate' digestion (W1).

Recently, both laboratory and pilot plant anaerobic systems have demonstrated that higher loadings are possible, provided that optimum environmental conditions are maintained in the digesters. Such conditions are suggested by Loehr (L4) to include a uniform feed, adequate SRT, thorough mixing and a beneficial temperature.

When these conditions have been incorporated into anaerobic units, loading rates between 1,6 and 6,1 kg VS/m^3/day have been reported (T2), (S1), (M21). This range is now commonly refered to as 'high-rate' digestion. Full scale digesters are rarely loaded at such high rates. However, Estrada (E1) reviewed the loading rates at a number of municipal sewage treatment plants and found loading rates varying between 1,6 and 3,52 kg VS/m^3/day. Other reports have been made of digester loadings in the field ranging from 2,4 to 6,1 kg VS/m^3/day (G2), (N1).

(d) Limiting Factors.

It should be emphasized that the trend towards higher loading rates has only become possible by maintaining optimum environmental conditions in the digesters.

The development of the anaerobic contact process, section 2.7.7, has provided a means of maintaining an adequate solids concentration (and therefore adequate sludge age) to withstand these higher loadings.

A curtailing factor in the use of higher loadings, particularly prevalent in the treatment of industrial wastes, is the build-up of toxic materials to inhibitory concentrations. The concept of toxicity is discussed in section 2.4.7.

A further limit to the loadings used in high-rate digestion is the difficulty experienced in solids/liquid separation in the contact process (M2). The amount of sludge recycled to the digester must be sufficient to maintain a large biological concentration in the process, to cope with the high loading rates.

2.4.7 Toxicity.

(a) Introduction.

There are a number of materials which may be inhibitory or toxic to the anaerobic waste treatment process. The loading rates on the system, treating the yeast waste under consideration, appeared to be limited by the presence of one or more inhibitory substances.

The effects of salt concentration upon the rate of biological reaction have been shown by Malina (M3) and McCarty (M10) and are presented in a general manner in Figure 2.6. At low concentrations the salts may have some stimulatory effect on the system. However, as the concentration reaches a level at which the salt begins to reduce the rate of reaction below the observed maximum, inhibition takes place. Eventually by continuously increasing the concentration of salt, the biological activity decreases rapidly and finally ceases completely. Micro-organisms can adapt to inhibitory substances. The extent of adaptation varies, and depends on the relative concentrations of other salts present as well as on the digester environment itself.

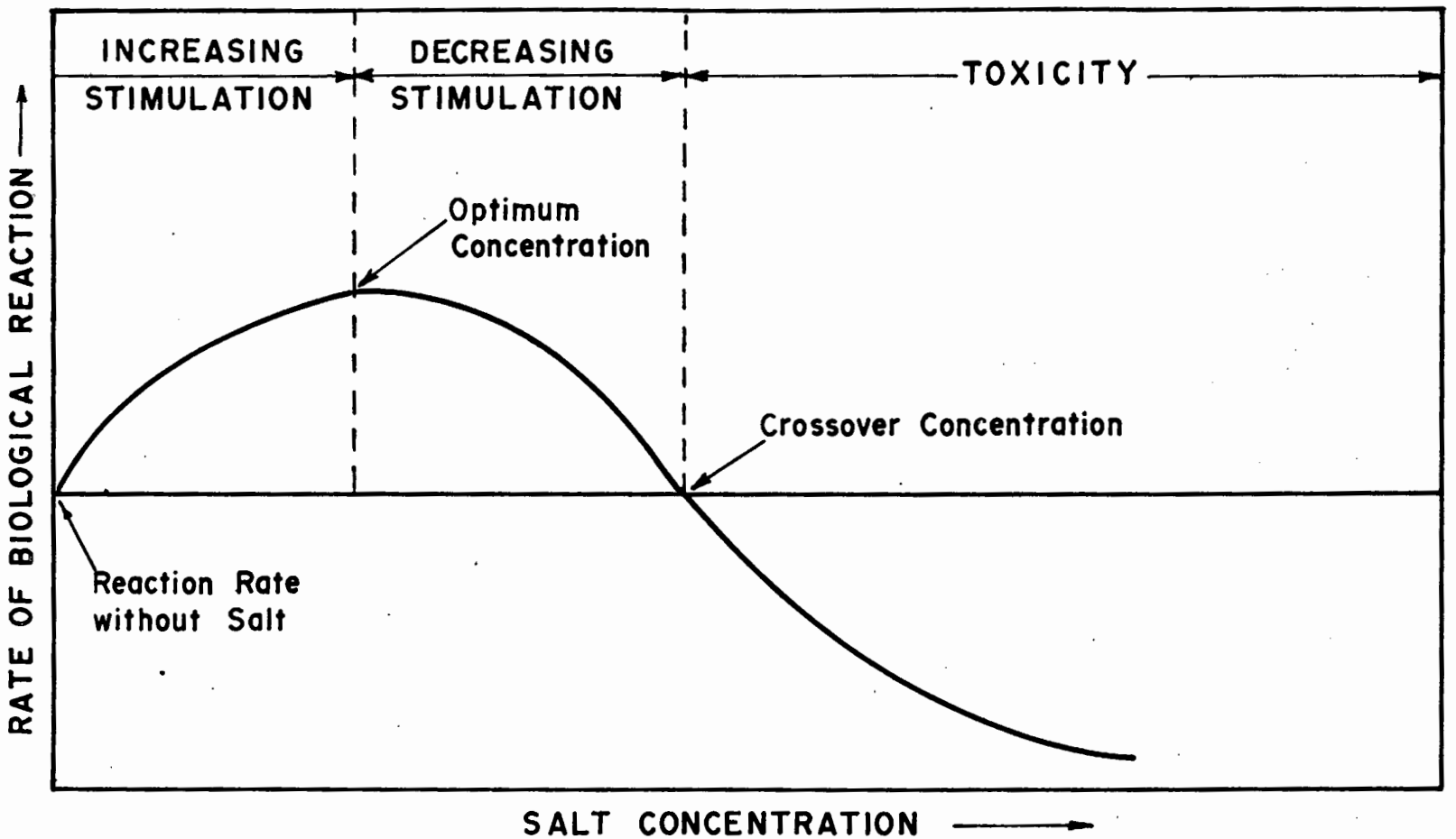

Figure 2.3 General effect of salts or other materials on biological reactions.

(b) Volatile Acids.

As shown in section 2.3.2, volatile acids serve as the major source of food for the methane bacteria.

Although it is commonly agreed that a build-up of volatile acids is the result of unstable digester conditions, the actual effect of a high volatile acid concentration on the digestion process has been strongly debated.

One group of investigators believe that volatile acids are toxic to the methane bacteria only in an indirect manner, through a reduction in pH (C1), (S1), (K1). They feel that a decrease in the activity of the methane bacteria is the result and not the cause of digester imbalance. The same group feel that such imbalanced conditions can be corrected by addition of a buffer material to restore the pH to a value more suitable for digestion to proceed.

The other group, headed by Buswell et al. (B4), suggested that a high concentration of volatile acids was inhibitory to the digestion process even if a neutral pH was maintained. Buswell (B4) further recommended that the volatile acid concentration be kept below 2 000 mg/l (as acetic acid) for digestion to proceed satisfactorily. He suggested that the maximum tolerable volatile acid concentration was approximately 3 000 mg/l. This view has been supported by Schulze and Raju (S5) and Mueller et al. (M23).

In an extensive study McCarty and McKinney (M5) showed that the decreased activity in digestion, during periods characterized by high concentrations of volatile acids, was due to salt toxicity rather than volatile acid toxicity. They showed that a rapid build-up of volatile acids and subsequent neutralization with alkaline materials, is similar to a slug loading of volatile acid salts. The

deleterious effect of these volatile acid salts was similar to that obtained when adding equivalent amounts of other common salts.

These two writers showed that volatile acid concentrations up to 10 000 mg/l have been tolerated, provided they were neutralized with alkalies containing non-toxic cations.

In a more recent publication, Andrews (A6) has suggested that the un-ionized portion of the volatile acids present in the digester could inhibit the methane bacteria.

The practical significance of the above work is doubtful, as volatile acids concentrations above 2 000 mg/l are extremely rare in full scale plants, during normal periods of digestion. However, there should be an increased awareness of the possibility of the various alkali and alkaline earth metals being present in inhibitory concentrations, particularly in the case of industrial wastes.

(c) Alkali and Alkaline Earth Salt Toxicty.

In municipal waste sludge, the concentrations of alkali and alkaline-earth metal salts, viz. sodium, potassium, calcium and magnesium, are usually sufficiently low so as not to inhibit digestion.

In industrial wastes the concentration of these salts may be quite high, especially if additions are made in order to control the pH of the system.

McCarty and McKinney (M5), (M6), have shown that toxicity is normally associated with the cation, rather than the anion portion of these salts.

The nature of the inhibitory effect of these salts is quite complex. The situation is further complicated by

the influence various combinations of these cations have on the inhibitory effect of others. McCarty (M10) has shown some ions to act antagonistically, i.e. reducing the toxicity of other cations, while others act synergistically, i.e. increasing the toxicity of other cations.

If an inhibitory concentration is present in a waste, its inhibitory effect may be considerably reduced by the presence of an antagonistic ion, either present in or added to the waste. Kugelman and McCarty (K3) have reported various antagonistic and synergistic combinations of cations. McCarty has shown sodium and potassium to be the most effective antagonists (M10).

It is apparent that there is difficulty in laying down definite levels of concentration, above which salts become inhibitory. However, the general guidelines, shown in Table 2.4, have been suggested by McCarty (M10).

TABLE 2.4. Stimulatory and Inhibitory Concentrations of Alkali and Alkaline-Earth Cations.

Cation	Stimulatory	Concentrations in mg/l	
		Moderately Inhibitory	Strongly Inhibitory
Sodium	100 - 200	3 500 - 5 500	8 000
Potassium	200 - 400	2 500 - 4 500	12 000
Calcium	100 - 200	2 500 - 4 500	8 000
Magnesium	75 - 150	1 000 - 1 500	3 000

At high feed concentrations of the yeast waste under consideration, potassium was found to be present at concentrations in the range likely to cause moderate inhibition, Table 2.4. This subject is discussed in Chapter 6.

(d) Heavy Metals.

Of the various heavy metals likely to be encountered in anaerobic digestion, copper, zinc, nickel and hexavalent chromium are likely to be most toxic (M20).

Hexavalent chromium may exert a toxic effect on the system, but this metal is normally reduced under anaerobic conditions to the trivalent state, which is relatively insoluble at the pH levels encountered in anaerobic systems, and therefore not very toxic (M3).

Iron and aluminium are virtually insoluble at the pH levels prevalent during digestion and consequently are not toxic.

The concentrations of the relatively toxic heavy metals which can be tolerated in an anaerobic system, are dependent on the concentration of soluble sulphides in the environment. Although soluble sulphides in sufficient contrations are by themselves toxic to the anaerobic system, this anion will react with the soluble heavy metal ions to form a relatively insoluble metal precipitate (L1), (M4).

Malina (M3) has suggested that in evaluating the tolerance of anaerobic treatment systems to heavy metals, the concentration of anaerobically reduced sulphur compounds must be considered (M3).

The addition of a sulphide, or sulphate salt which will be reduced to sulphide under anaerobic conditions, is therefore an effective method of controlling this type of toxicity. Otherwise, precipitation of the metals in chemically controlled systems, frequently as hydroxides, has been suggested by Loehr (L5).

The concentrations, above which heavy metals become toxic, suggested by Barth et al. (B2), are shown in Table 2.5.

TABLE 2.5. Toxic Concentration of Heavy Metals.

Metal	Concentration
Cr^{6+}	50 mg/l
Cu	10 mg/l
Ni	40 mg/l
Zn	10 mg/l

None of the heavy metals mentioned in Table 2.5 were found to be present in the yeast waste under consideration, at a concentration level greater than 2 mg/l. Consequently no form of heavy metal toxicity was suspected.

(e) Ammonia Toxicity.

Inhibitory concentrations of ammonia may be approached in industrial wastes containing high nitrogen concentrations, or in highly concentrated municipal waste sludges being used in high rate digestion, McCarty (M10).

In anaerobic systems ammonia may be present either in the form of the ammonium ion (NH_4+) or as dissolved ammonia gas (NH_3). These two forms are in equilibrium with each other as indicated by the following expression (L5) :

$$NH_4^+ \rightleftarrows NH_3 + H^+ \quad \ldots\ldots\ldots\ldots\ldots \quad (2.3)$$

As the pH of the system increases, equilibrium shifts to the right and the concentration of free ammonia may become inhibitory. Conversely, if the pH drops below a value of about 7,2, the equilibrium is shifted to the left and inhibition becomes related to the ammonium ion concentration (M10).

Ammonia nitrogen concentrations which have an adverse effect on anaerobic treatment are indicated by McCarty (M6) and Albertson (A3) and shown in Table 2.6.

TABLE 2.6. Effect of Ammonia Nitrogen on Anaerobic Treatment.

Ammonia Nitrogen Concentration mg/l	Effect on Anaerobic Treatment
50 - 200	Beneficial
200 - 1 000	No adverse effect
1 500 - 3 000	Inhibitory at higher pH values
Above 3 000	Toxic

If the ammonia concentration is between 1 500 and 3 000 mg/l, and if the pH rises to a value of 7,3 or higher, the more toxic free ammonia predominates and becomes inhibitory. Depression of the pH to a value below 7,2 by addition of hydrochloric acid can relieve this inhibition (M10).

When the ammonia-nitrogen concentration exceeds 3 000 mg/l, then the ammonium ion itself becomes toxic. Reduction of pH will not prevent the process from failing and either dilution or removal of the nitrogen source itself would have to be considered.

The possibilities of high rate digesters acclimatising to ammonia nitrogen toxicity have recently been reported by Melbinger and Donnellon (M15) and Zablatzky (Z1).

At extremely high loading rates, during the preliminary studies ammonia was thought to have an inhibitory effect on the digestion process. However, at the loading rates subsequently used, ammonia was not present in sufficient quantities to cause inhibition.

(f) Sulphide Toxicity.

Sulphides present in a digester may originate in either of the following ways :

(a) introduction of sulphides with the raw waste;

(b) biological production in the digester, from the reduction of sulphates and other sulphur containing compounds.

Part of the sulphides leave the system as hydrogen sulphide gas, and a further portion of the sulphides will be precipitated as heavy metal salts if these metals are present.

The remaining soluble sulphides form a weak acid which ionizes in aqueous solution, the extent depending upon the pH. At the neutral pH required for anaerobic treatment, only the first dissociation of hydrogen sulphide is of importance.

Sulphides, therefore, may be distributed between a soluble form, insoluble form and the gaseous hydrogen sulphide.

The actual distribution of these sulphides is dependent on the pH of the digester and on the amount of gas produced from the waste as is shown in Figure 2.4, (L1). As can be seen from Figure 2.4, the higher the gas production per volume of waste, the greater the amount of sulphides driven off from solution. Similarly, the lower the pH, the lower the amount of sulphides remaining in solution, provided the decrease in gas production is not too significant with a consequent reduction in the amount of sulphides driven off from solution.

Concentrations of soluble sulphides up to 100 mg/l can be tolerated in anaerobic processes with little or no

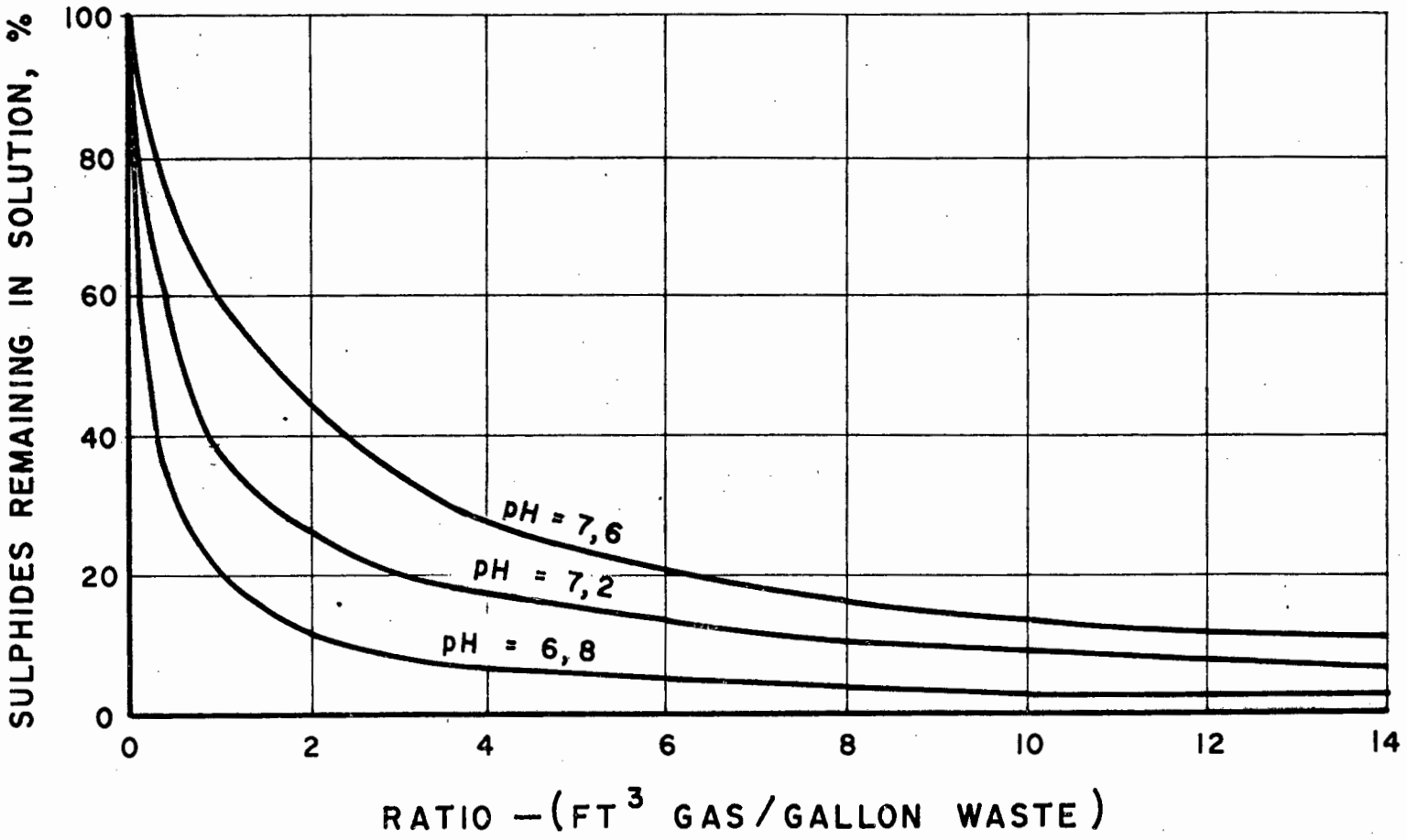

Figure 2.4 Effect of gas production and pH on the fraction of soluble sulphides in solution.

acclimatization required. With continuous operation and some acclimatization, concentrations up to 200 mg/l have been tolerated. Concentrations above 200 mg/l are reported to be toxic (L1).

Excessive concentrations of sulphides can be precipitated to reduce the sulphide toxicity in an anaerobic digester. Iron salts are very suitable for this purpose, due to their low toxicity.

Since the toxic effects of heavy metals can be decreased by precipitation and since toxic concentrations of sulphides can in turn be decreased by precipitation of their metal salts, the toxicity of both materials can be decreased at the same time (L5).

Gas recirculation, with corresponding stripping of the soluble sulphides, has also been reported as a means of reducing sulphide toxicity (L1).

As shown in Chapter 6, precipitation of some of the sulphides from the digester environment did not appear to have any influence on the efficiency of the process, even at the high loading rates where inhibition was suspected. Consequently, the sulphides generated in the treatment of the yeast waste were not inhibitory. Stander et al. (S12) found the addition of sulphates to yeast waste, in a similar study, not to influence the system.

(g) Chloride Toxicity.

Very erratic behaviour followed by partial recovery or total failure, depending on the load on a digester, has been described as typical of chloride inhibition (L6).

Up to concentrations of 2 000 mg/l as NaCl no inhibitory effect is likely.

Chloride additions of between 2 000 and 5 000 mg/l as NaCl have resulted in a serious decrease in gas production with partial recovery when the load on the digester was reduced.

Chloride additions of 10 000 mg/l, or larger, promptly resulted in negligible gas production and little recovery after six weeks operation (L6).

Delayed effects, biota adaptation and effects of digester past history made it difficult to suggest a definite limit of tolerance for chlorides during anaerobic digestion. A given chloride increase frequently exerted delayed effects, with performance decreases occuring long after steady state would normally have been established (L6).

Niles and Frook (N2), while studying the digestion of Monosodium Glutamate, found a concentration of 5 000 mg/l as NaCl as the limiting amount of chlorides their digesters could tolerate.

Buswell et al. (B5), using pure acetic acid, found that with acclimatization, chloride concentrations of 10 000 mg/l as NaCl could be tolerated.

Very little work has been reported on the acclimatization of digesters to high chloride concentrations. It is possible that the halophilic organisms present in salt pans might contribute to such acclimatization, but no evidence of such treatment appears in the literature.

Chlorides were present in relatively high concentrations in the yeast waste under consideration and could have exerted an inhibitory influence on the system at high loading rates, as shown in Chapter 6.

(h) General Factors.

Various organic materials, ranging from organic solvents to common materials such as long chain fatty acids, may at high concentrations, inhibit the digestion process. These have been summarized in the literature and rarely affect continuously fed digesters (M10).

(i) Control Methods.

The type of method used to control toxic materials will depend on both the degree and type of inhibition.

The four main methods used can be summarized as follows (M10) :

(a) removal of the toxic material from the waste;

(b) dilution of the waste until the concentration of the toxic material falls below its reputed threshold value;

(c) the formation of insoluble complexes or precipitates;

(d) addition of a suitable antagonistic material.

2.4.8 Summary of Optimum Environmental Parameters.

A summary of the environmental parameters which are considered optimum for anaerobic digestion is shown in Table 2.7.

TABLE 2.7. Optimum Environmental Parameters for Anaerobic Digestion.

Variable	Optimum	Extreme
Temperature ^{o}C	30 - 35	25 - 40
pH	6,8 - 7,4	6,2 - 7,8
Volatile acids (mg/l CH_3COOH) [1]	50 - 500	2 000
Alkalinity (mg/l $CaCO_3$) [2]	2 000 - 3 000	1 000 - 5 000
Redox Potl (mv)	-520 to -530	-490 to -550
	Sufficient nutrient requirements	
	Absence of toxic materials	

1 Depends on system - could be 10 000 provided sufficient alkalinity is present.

2 Depends on volatile acid concentration.

Having established what sort of an environment is deemed favourable for digestion to proceed, it is now necessary to ascertain what indicators are used to assess digester performance.

2.5 INDICATORS OF DIGESTER PERFORMANCE.

2.5.1 Introduction.

The various parameters used to evaluate digester performance are discussed in this section.

Besides indicating the relative condition of a digester, these parameters should enable an operator to detect any sign of impending problems, so that corrective action can be taken before the digester fails.

Once digestion has failed, and especially if a seed sludge is not readily accessible, severe difficulties are

experienced in restarting digesters. The significance of reliable indicators of digester performance is self evident. This is particularly pertinent for industrial waste treatment systems, where economics frequently do not allow the construction of alternative systems for diverting the feed during unstable periods of digestion.

2.5.2 pH.

A decrease in pH is usually the result of a high volatile acid concentration, and therefore can be considered as an indicator of retarded digestion.

A significant drop in pH, however, is unlikely to occur, until the digester is seriously affected, and corrective action in such cases may be too late to prevent process failure.

It is therefore exceedingly dangerous to rely on pH to predict periods of digester imbalance.

2.5.3 Volatile Acids.

The monitoring of volatile acids concentration has often been mentioned as one of the best parameters for following digester progress (M9).

If the methane bacteria become affected by adverse environmental conditions, their rate of utilization of volatile acids will decrease, and the concentration of the latter will begin to increase. A sudden increase in volatile acids is therefore one of the first indicators of digester imbalance (M9).

As shown earlier a high value of acid concentration on its own does not necessarily signify retarded digestion.

It is the relative change in volatile acids concentration, rather than the absolute concentration, which signifies a decrease in digester performance. Used in this manner, volatile acids concentration can give a valuable insight into the condition of a digester.

In cases where high volatile acids concentrations are characteristic of the system, sudden increases in volatile acids concentration may be masked by the high absolute concentration in the digester, until inhibition is fairly established.

In such cases, some investigators feel that by monitoring the ratio of volatile acids to total alkalinity (W1) or 'volatile acid salts' alkalinity to total alkalinity (P2), a more sensitive indication of digester performance can be obtained.

Pohland and Bloodgood (P2) showed that individual volatile acids were affected differently by adverse changes in their environment. They found that the degradation of propionic acid to either gas or lower acids was affected to the greatest extent during times of retarded digestion. The butyric-valeric group was found to be less affected and acetic acid most resistant to these changes.

Monitoring the changes in individual acid concentrations would therefore seem to be an excellent method of analysing digester performance. The striking difference in the changes in concentration of the individual volatile acid components during periods of normal and retarded digestion is shown in Figure 2.5.

2.5.4 Gas Composition.

Increases in the volatile acids concentration and their subsequent neutralization have been shown by Pohland et al. (P3) to result in the release of carbon dioxide to

Waste treatment plants, unlike other types of chemical plants, are frequently limited by the variation in concentration and/or the volumetric flow rate of the waste that is to be treated. The vulnerability of anaerobic digestion to sudden loading changes has been pointed out earlier in this section.

Clough (C3) has indicated that such variations in volumetric flow and load concentration can be partially offset or even eliminated by the use of holding and/or balancing tanks.

A holding tank is one in which the volume is kept constant, but where mixing takes place, thereby reducing fluctuations in the waste concentration. Holding tanks consequently have no influence on volumetric flow rate.

A balancing tank is one in which the volume is allowed to fluctuate, therefore enabling a constant flow rate to be maintained to the digesters.

A compromise between holding and balancing tanks is often desirable and is possible to obtain by allowing the volume in a holding tank to fluctuate over a limited range.

The use of these tanks is dependent on the extent and variation of the expected load on the system. The capital costs of such a tank should be weighed against the potential saving to be derived from the improved treatment efficiency resulting from a steadier load (C3).

Up to this stage, the mechanism of anaerobic digestion, the environment necessary for digestion to proceed and the indicators of digester performance have been discussed.

Attention will now be focussed on the development of digester design and the type of digester designs in common use nowadays.

2.6 TYPE OF DIGESTERS USED AND THEIR DEVELOPMENT.

2.6.1 Preliminary Treatment.

The application of anaerobic digestion to waste treatment appears to date from the Austin tank, first constructed in 1857 (M16). A French engineer, L.H. Mouras, designed the 'Mouras automatic scavenger' a few years later.

The principal action of these preliminary tanks was thought to be the liquification of organic solids after sedimentation had taken place. The bacteriologists of the time were unaware that a considerable amount of the organic matter was in fact removed as a gas, an attractive characteristic of the process.

These preliminary tanks, however, served principally for sedimentation purposes.

It was not until 1895, when Donald Cameron designed and constructed a septic tank for the City of Exeter, that anaerobic digestion came into more frequent use.

2.6.2 Septic Tank.

Although septic tanks (Figure 2.6) are basically similar to plain sedimentation basins, the retention times used for the former are normally much longer so that the deposited sludge can undergo partial or complete digestion.

The septic tank consisted therefore of one stage, with no attempt to separate the phases of the process. No control was exercised other than the specification of the tank volume, which for a given flow rate fixed the retention time.

Two major problems were encountered with the septic tank :

(a) the lifting of solids into the clarification

into the sedimentation compartment.

Travis considered it essential to pass a certain proportion of the sewage through the digestion, or 'liquifying' compartment in order to seed the liquid there with fresh bacteria. Consequently the lower digestion compartment was made reasonably small and the retention times in these compartments were neither long enough to produce a fresh effluent nor a properly digested sludge.

The Imhoff tank was a two-storey tank in which all sewage passed through the upper, sedimentation compartment. The suspended matter fell through slots into the lower digestion compartment where there was no sewage flow. The digestion compartments of Imhoff tanks were considerably larger in relation to the sedimentation sections than was the case for the Travis tank.

The bottom of the settling compartment was overlapped to prevent gas or sludge rising into this section, while vents were provided at the side of the digestion section to allow for the release of gas.

Although the separation of functions in the Imhoff tank had increased the quality of the effluent over that obtained in the septic tank, there was still no control over the process other than the hydraulic retention time.

2.6.4 Separate Digestion Tanks.

By the early 1920's, the separate digestion tank began to gain favour. The solids were removed from the wastes before entering the digester, and only biological degradation of the solids and thickening occurred in this process.

In conducting sludge-digestion in separate tanks, rather than combining sludge-digestion and sedimentation

in the same tank, certain advantages became apparent.

The most significant of these was the ability to exert greater control on both the individual processes. This increase in operational control tended to justify the cost arising from the construction of the separate structures.

2.6.5 Developments in Control.

By this time two significant improvements had emerged which further aided the control of the digestion process :

(a) the heating of the digester contents to temperatures more favourable for microbial action;

(b) the use of artificial mixing to maintain a continuous supply of food material to the anaerobic micro-organisms and to prevent scum formation.

Since artificial mixing is incompatible with the thickening of digested sludge, the next development in anaerobic digestion was the separation of the biological action of digestion from the thickening process.

This resulted in a two-stage process, Figure 2.8, which allowed the heating and mixing of the digester contents in a first stage. Thickening and storage took place in a second stage where heating is not necessary.

Two further advantages were derived from this development :

(a) the ability to make use of the entire volume of the first stage for biological stabilization;

(b) the possibility of designing such a unit for optimum biological action.

2.6.6 Two-stage Process.

The imprdvements in operational control and modifications incorporated into the two-stage digester, i.e. the separation of the physical phases of the process, reduced retention times significantly.

It is of interest to note that the changes in digester design and operation have not significantly affected the biological process. In other words the digester is still a one stage system with respect to the biological reactions even though the physical stages of digestion and sludge thickening have been separated.

To summarize, the various control parameters available to the operator up to this point in the development of anaerobic digestion were :

(a) the hydraulic residence time;

(b) an ability to heat the digesters to an optimum temperature for biological degradation; and

(c) various methods of mixing the digester contents so as to ensure adequate contact between the organisms and organic material.

However, in spite of the improvements in operational control, the size of the digesters were still discouraging due to the relatively long retention times necessary for digestion to proceed satisfactorily.

The next stage in the development of anaerobic digestion, the anaerobic contact process, was linked with the solids

retention time (defined in Chapter 5).

2.6.7 Anaerobic Contact Process.

The anaerobic contact process, Figure 2.9, is basically similar to the two stage system, except that some of the thickened sludge is recycled to the digester.

This system permits the maintenance of a high biological population while operating at a relatively low hydraulic retention time. In other words, it is possible to vary the solids retention time independent of the hydraulic retention time. As can be seen from the theory, 6.1.4, the effluent quality is a function of solids retention time and not of the hydraulic retention time.

The contact process can readily be applied to dilute wastes where hydraulic residence times should be short to make the process economical. In systems without recycle, the dilute wastes may not support an adequate, microbial population. Digester volumes for treating dilute wastes become prohibitively large in order to have an adequate hydraulic residence time, which in turn have to be greater than the minimum solids retention time for stabilization to take place. Solids recycle permits the use of smaller digester volumes for maintaining a sufficiently large sludge concentration to obtain satisfactory waste stabilization.

The major problem arising from the use of the anaerobic contact process to date is related to the inability to efficiently separate the bacterial solids from the effluent stream for recycle back to the digester.

When operating at short hydraulic retention times, high separation efficiency is required to provide adequate sludge concentrations in the digester and thereby to maintain

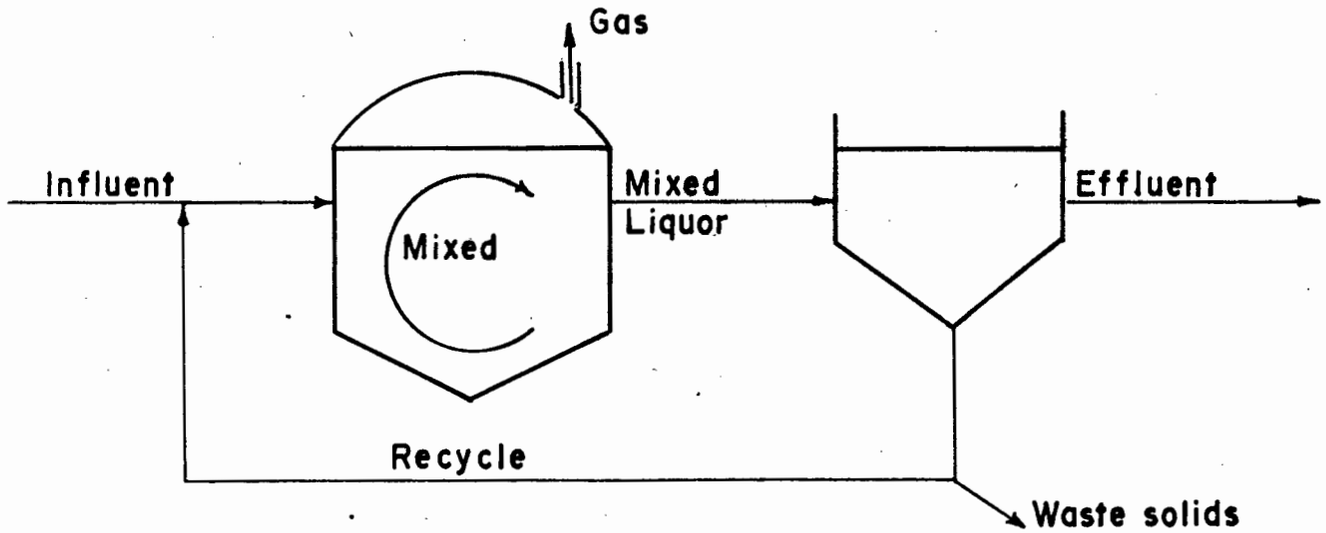

Figure 2.9. Anaerobic contact process.

the required solids retention time for such a system.

In the successful full-scale treatment of meat-packing wastes reported by Steffen and Bedker (S14), a vacuum degasifier has been used between the digester and final settling tank to remove gases which tend to hinder settling. The effluent from the digesters are pulled into the degasifier tanks under a 20" vacuum produced by vacuum pumps installed in the control building. The mixed liquor splashes down into a series of slats which aid in the release of the gases, which in turn are exhausted by means of a vacuum pump.

Other systems for solid/liquid separation, such as centrifuges, filters and flotation units, have been used experimentally, but none has so far been used on a significant commercial scale (C3).

Generally the development of the anaerobic contact process has meant that smaller volumes and lower temperatures can be used to treat a given waste. Further, the ability of these systems to handle changes in the digester environment has been greatly improved. However, loadings used in the anaerobic contact process are limited by the efficiency of sludge separation.

2.6.8 Internal Maintenance of Cell Concentration.

Similar maintenance of active cell material and biota in a digester can be achieved by internally returning the sludge to the mixing zone.

This type of internally clarified digester is often referred to as a 'reactor-clarifier', the conventional flow pattern of which is given in Figure 2.10.

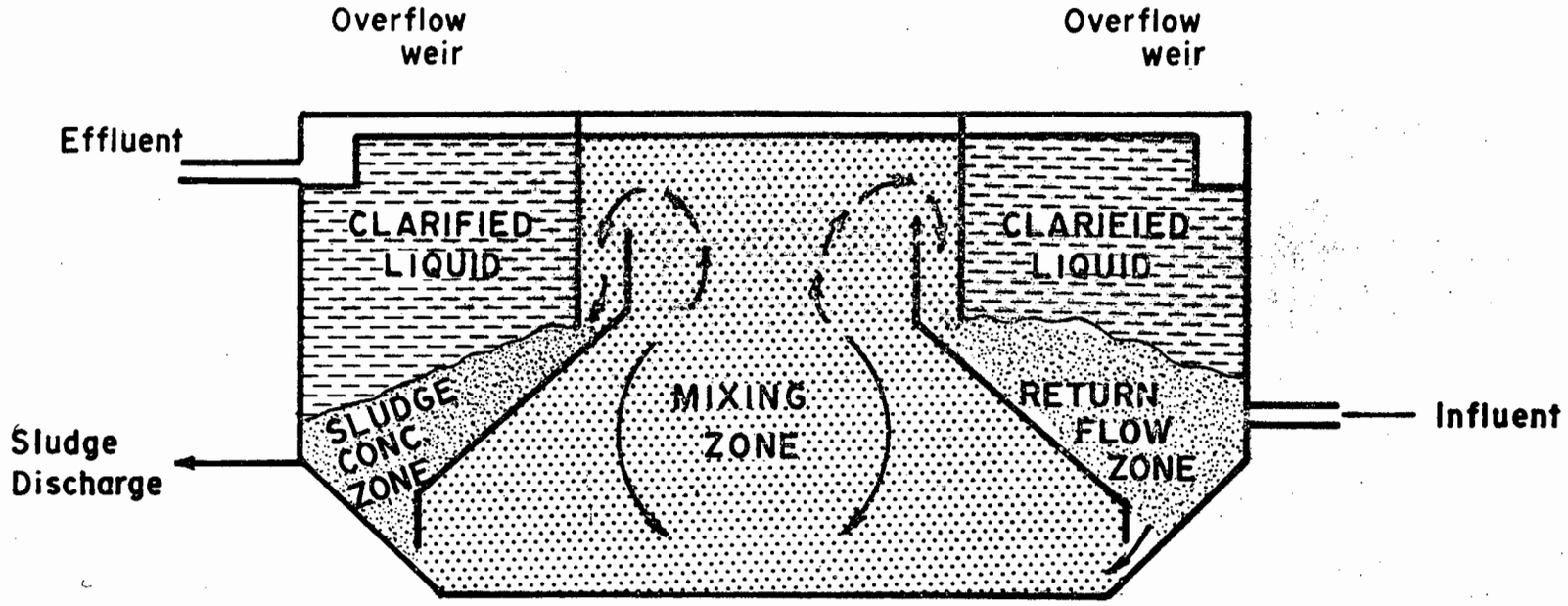

Figure 2.10. Flow patterns in typical reactor-clarifier system.

Various reports on this type of digester have appeared in the literature (P7), (H2), (S11). Stander (S11) reported reductions consistently higher than 96% (COD) in a reverse flow clarifier, treating a wine waste, with organic loadings up to 0,2 lbs COD/ft^3/day.

The major advantage of a clarifier unit is its ability to maintain sludge in an integrated cycle rather than by use of external recycling.

However, even when facilitated, by the provision of fixed scrapers, sludge return can be inefficient.

A further disquieting limitation, inherent to this type of anaerobic treatment, concerns the behaviour of sludge density with changes in feed rate (S11). As mentioned earlier, a sufficiently large sludge concentration must be maintained in the digester, to cope with increases in loading. Sludge accumulation in the clarifier can only be reduced if the concentration on the floor of the clarifying section (from where it is scraped into the digester section) is higher than that in the digester compartment. The rapid evolution of gas, characteristic of high loading rates, takes place in the clarifying section as well, and hinders sludge thickening. Consequently, sludge builds up in the clarifier and eventually sludge appears in the effluent.

Various design methods, including the construction of an annular compartment in the clarifier section, were used in an attempt to solve this problem (S11), and met with moderate success. As will be shown later, a similar disadvantage became apparent in the digesters used in this thesis. The maximum sludge concentrations possible in the digesters seemed to decrease with decreases in hydraulic retention time, which is unfortunately contrary to the biological requirements for maximum loading. However, the simplicity of the process, as opposed to the anaerobic contact process, is an advantage. The extent of the loading

will determine to what degree this offsets the above disadvantages.

This type of digester is particularly applicable to laboratory scale digesters in that adequate sludge concentration can be maintained with the minimum equipment.

2.6.9 Anaerobic Filter.

Insufficient methane is produced from anaerobic treatment of dilute soluble wastes to maintain digesters at suitable temperatures. Consequently, in some cases, treatment at temperatures near ambient becomes a practical necessity. Efficient waste treatment at these low temperatures becomes dependent on the maintenance of long solids retention times, McCarty (M12).

The anaerobic contact process has been shown to treat such wastes quite successfully, particularly meat-packing wastes and others containing a significant concentration of suspended solids (M11). The micro-organisms tend to become attached to the solids and are readily removed in the settling tank for recycle back to the digester. However, with soluble wastes, many of the bacteria remain dispersed and a significant number may leave with the effluent, thereby making it difficult to maintain the required solids retention times (M12).

The anaerobic filter, (M12), (P1), (Y1), might prove to be the solution to the problem of treating these soluble wastes.

In this device, Figure 2.11, the bacteria grow attached to the walls of the confining structure as well as to the packing and remain in the filter, making extremely long solids retention times possible. Because of the long biological solids retention times possible, Young and

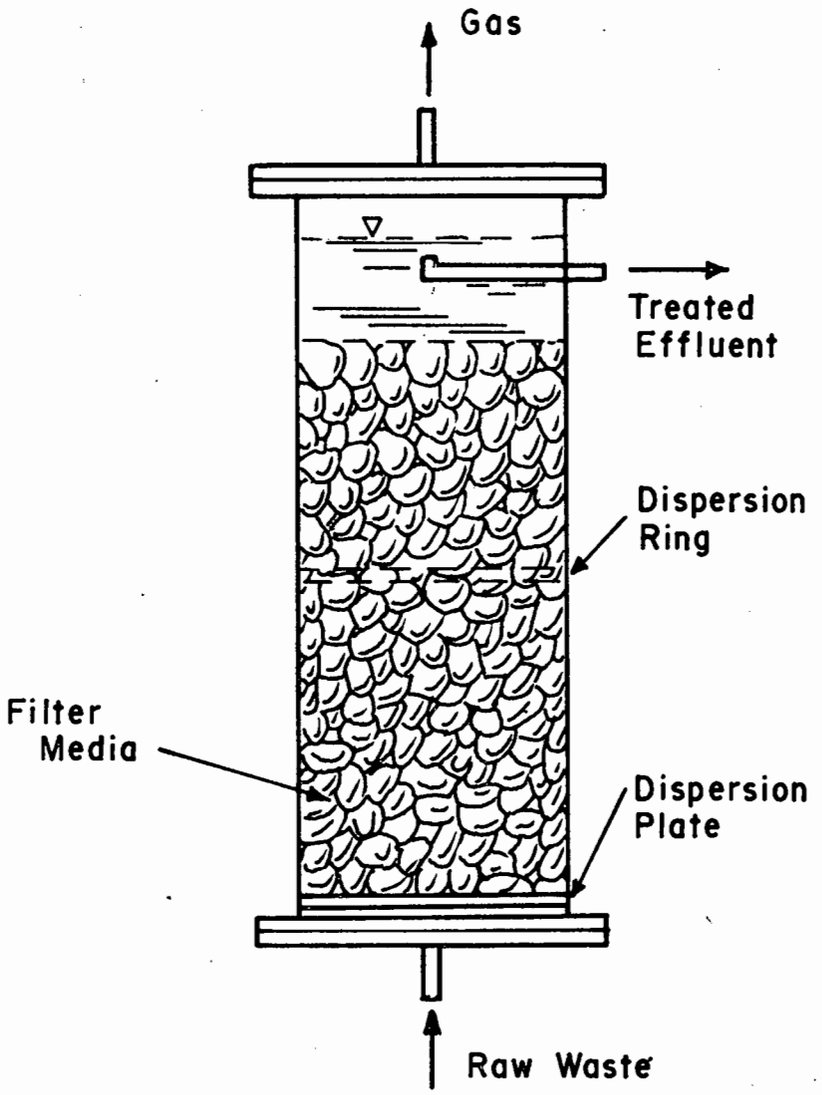

Figure 2.11. Anaerobic filter.

McCarty have shown that the anaerobic filter can be efficiently operated at ambient temperatures (Y1). Therefore the filter is extremely suitable for the treatment of dilute wastes, from which insufficient methane is produced to heat the incoming waste.

A further advantage of the filter over the anaerobic contact process is the retention of the biological solids in the filter, eliminating the need for recycle with its inherent separation difficulties.

A disadvantage of the filter, is the short-circuiting of the raw waste along the paths caused by gas bubbles, mentioned by Plummer et al. (P1). They suggest staging of the filter to overcome the difficulty.

However, it should be born in mind that as the recycle ratio increases, the filter approaches a continuously mixed system and the advantages characteristic of the plug flow system become less apparent.

Up to the present, the filter has largely been confined to laboratory scale studies, treating a synthetic volatile acid waste (M12) and a carbohydrate-protein waste (Y1).

2.6.10 Multi-stage Systems.

The use of multi-stage systems for waste treatment has been reported by Rich (R1) and Wuhrmann (W4).

In the former, Figure 2.12, the waste stream flows into a contact basin (retention time of 25 to 90 minutes) where the organics are primarily absorbed by the sludge flow. The activated sludge is then separated from the mixed liquor in a sedimentation tank and passed into a

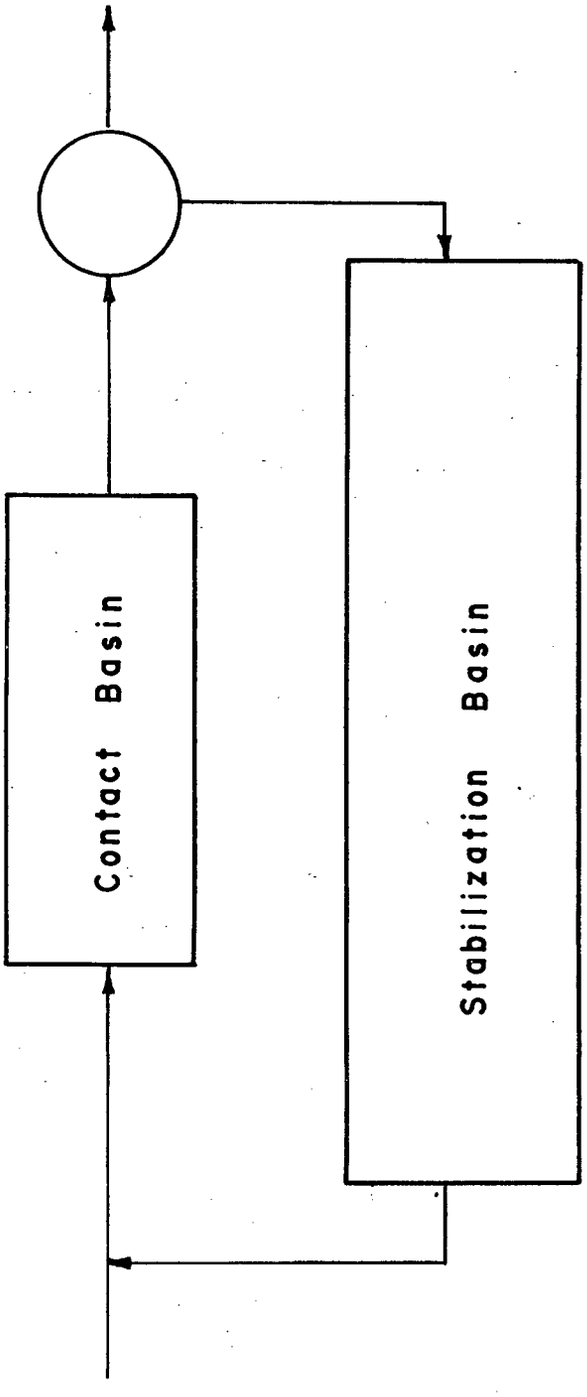

Figure 2.12. Contact stabilization.

stabilization basin in which the absorbed organics are metabolized by the organisms. The retention time in the stabilization basin is 1,5 to 5 hours depending on the strength of the waste.

In the process suggested by Wuhrmann, Figure 2.13, the nitrogen compounds are initially oxidized to NO_2^- and NO_3^- under aerobic conditions. In the second stage anaerobic denitrifying bacteria are used to reduce the nitrates and nitrites to gaseous N_2 or N_2O.

Willimon and Andrews (W3) have suggested that such a multi-stage process might be applicable to anaerobic digestion. They depict the anaerobic process as three biological reactions occuring in series, and maintain that the optimum environmental conditions for the different organisms responsible for each reaction, should vary from group to group. They further speculate that the isolation of the biological reactions into separate vessels would enable the environmental conditions to be varied to maximize the rate of each reaction.

Possible advantages that might apply to such a system are suggested as :

(a) improved removal;

(b) increased stability;

(c) better quality control.

Disadvantages of such a system would include :

(a) increased costs;

(b) more control requirements;

(c) increased process complexity.

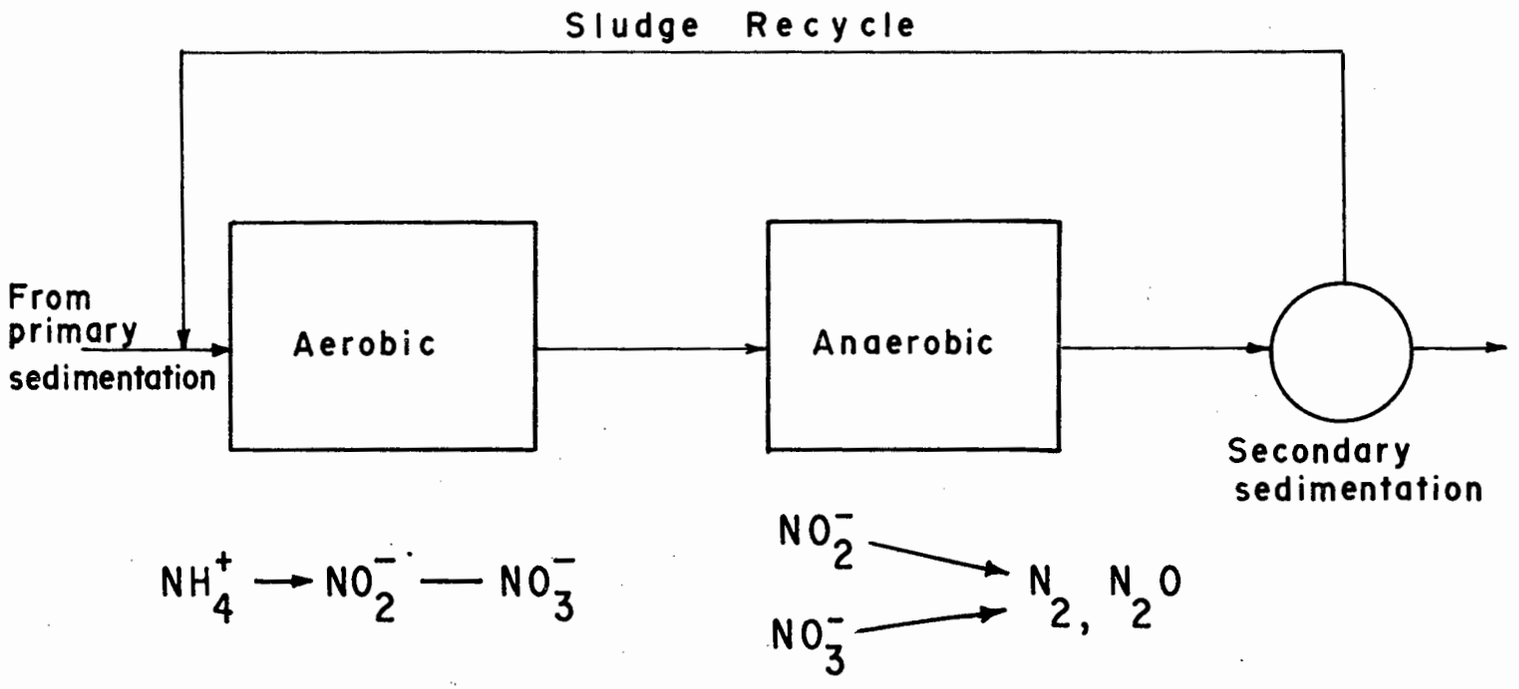

Figure 2.13. Nitrification and denitrification.

The system is purely at the exploratory stage as far as anaerobic digestion is concerned and it is conceded that the environmental conditions required for the optimization of each stage are still largely unknown (W3).

2.6.11 Summary of Types of Digesters in Common Use.

The two most common processes used for anaerobic treatment at present, are the conventional process and the anaerobic contact process, Figure 2.14, (M11).

The conventional process is the simpler type, involving a single tank in which the bacteria and waste are mixed together. The hydraulic retention time equals the solids retention time for this process as the bacteria and treated wastes are removed together for disposal. The stabilization may be increased by making some provision for the separation of the effluent solids from the effluent stream. The conventional process is usually used for the treatment of more concentrated wastes.

The anaerobic contact process, on the other hand, can be used to treat most waste concentrations, but is especially useful for the treatment of dilute wastes. The ability to use economically short hydraulic retention times, while maintaining comparatively long solids retention times, allows the use of lower operating temperatures and results in more stable operation.

The major problem arising from the use of anaerobic contact process to date concerns the efficiency of the solids separation from the effluent stream for recycle back to the digester.

The anaerobic filter has been developed as a possible solution to the above problem, but itself suffers other

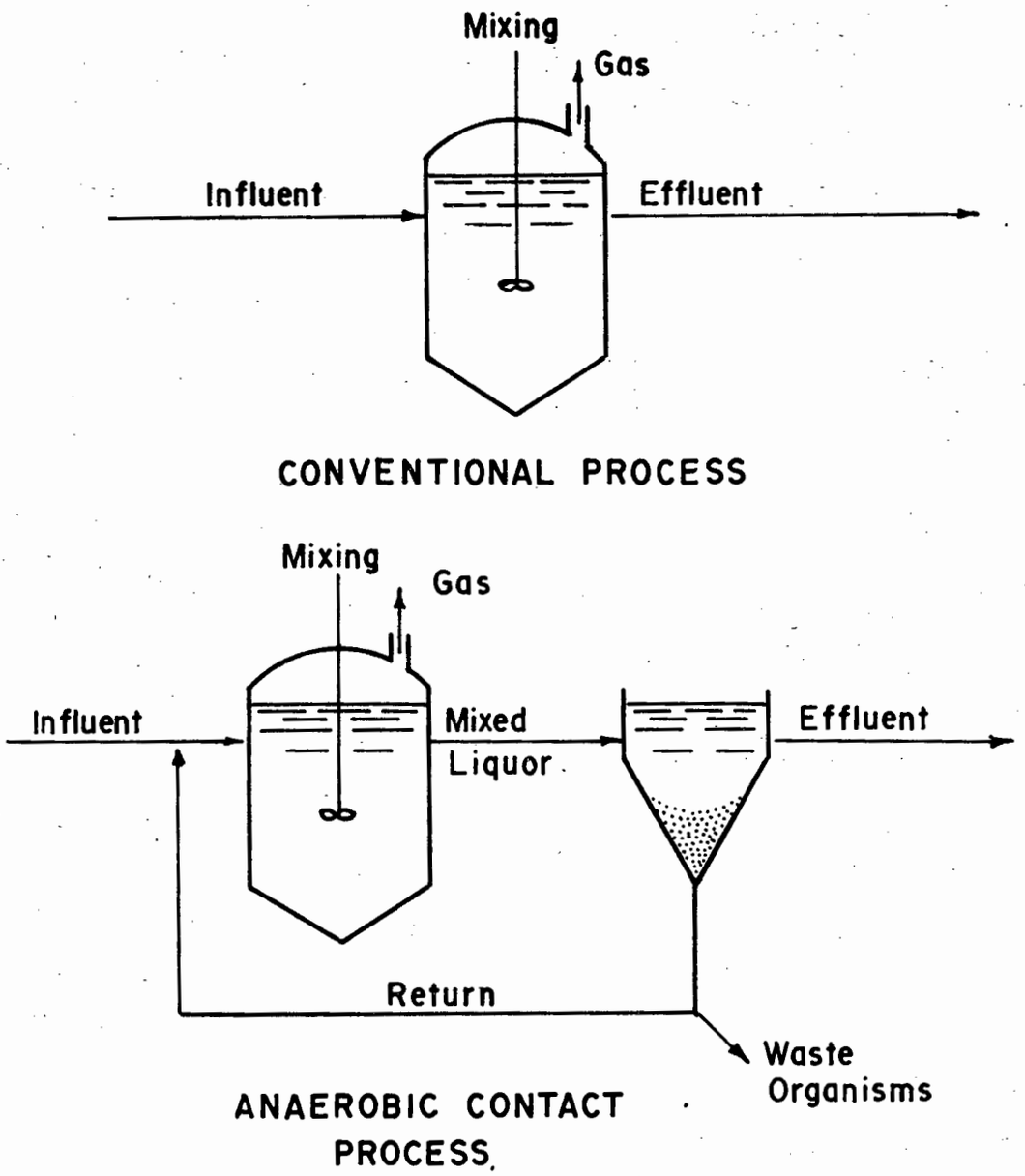

Figure 2.14. Conventional and contact processes.

disadvantages.

A brief survey of the development and application of kinetics to anaerobic digestion has been placed in Chapter 6.

CHAPTER 3

APPARATUS AND EXPERIMENTAL TECHNIQUE

3.1 INTRODUCTION.

The experimental work was carried out in two different types of apparatus, viz. a batch digester and a continously fed digester of the 'reactor-clarifier' type.

The batch system was used for preliminary work in determining the feasibility of treating yeast waste by a conventional form of anaerobic digestion. Further use was derived from the apparatus in determining ultimate biodegradability and for investigating the potential toxicity of various ions present in the yeast waste.

The continuous flow system was designed to permit the independent removal of sludge and clarified effluent, and consequently the independent variation of sludge and hydraulic retention times.

The operating parameters used for each digester are presented and the experimental technique of arriving at steady state conditions discussed.

3.2 THE BATCH SYSTEM.

The batch system used during this investigation is shown in Figure 3.1. The digesters consisted of two-litre Buchner flasks, each containing 1,5 litres of digesting liquor.

Each digester was connected to a 5-litre aspirator flask, which was used as a gas collection bottle. Each gas collection bottle contained a sulphuric acid solution (5 ml concentrated H_2SO_4/litre) with a pH value below 1,5. The displaced liquid from the gas collection bottles was allowed to run into a similar 5-litre aspirator flask which was left open to the atmosphere. This system permitted the measurement of gas production at atmospheric pressure by maintaining the liquid levels of both aspirator bottles at the same height, while

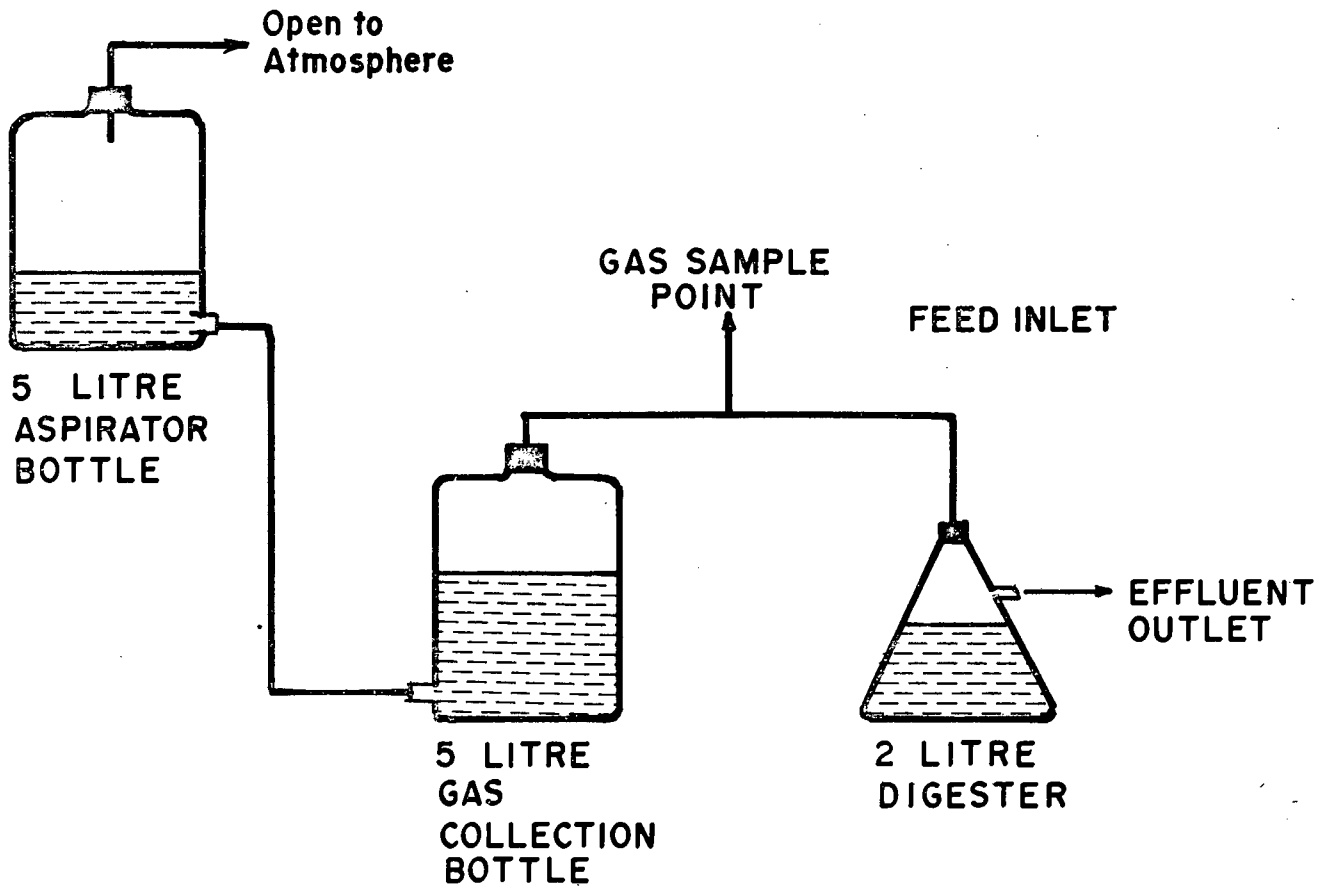

Figure 3.1. Batch process used in this investigation.

readings were being taken.

Gas samples for the analysis of the digester gas were taken from the gas sampling cock, shown in Figure 3.1.

The daily addition of feed and withdrawal of effluent was made under pressure to prevent air from entering the system. Sufficient pressure was obtained during these processes by raising the aspirator bottle to a suitable level above the level of liquid in the gas collection bottle.

The digesters were operated in a constant temperature room, maintained at 30°C. Fluctuations were kept to a minimum by monitoring the temperature daily, on a maximum-and-minimum thermometer.

The digesters were initially seeded with digested sludge from the Athlone sewage works and the following start-up procedure was used for all the batch digesters:

One litre of the seed sludge was put into the digester together with 500 ml of the diluted yeast waste. The system was purged with nitrogen gas, carefully sealed, and digestion allowed to proceed. As shown in section 4.2.1, virtually no acclimatisation period was required. Consequently, no further additions of seed sludge were required.

As a low hydraulic residence time was thought to be essential for treatment of the yeast waste to be economical, the following method of operation was used for the batch digesters.

The digester contents were shaken at frequent intervals throughout the day and allowed to settle overnight. This permitted the daily withdrawal of effluent without significant sludge wastage. Consequently, a low hydraulic residence time, viz. 3 days, was made possible while maintaining a sludge age of between 40 and 60 days.

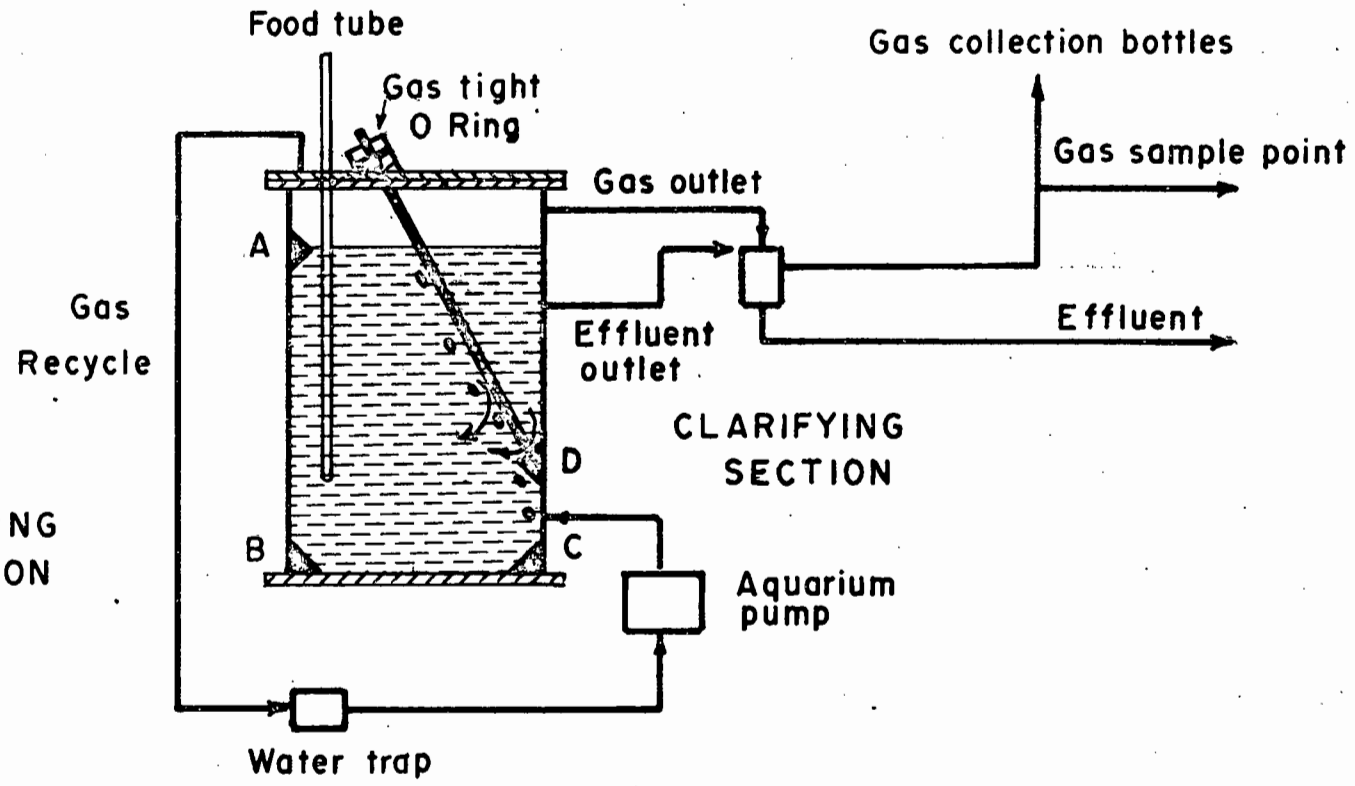

Figure 3.2. Reactor-Clarifier used in this investigation.

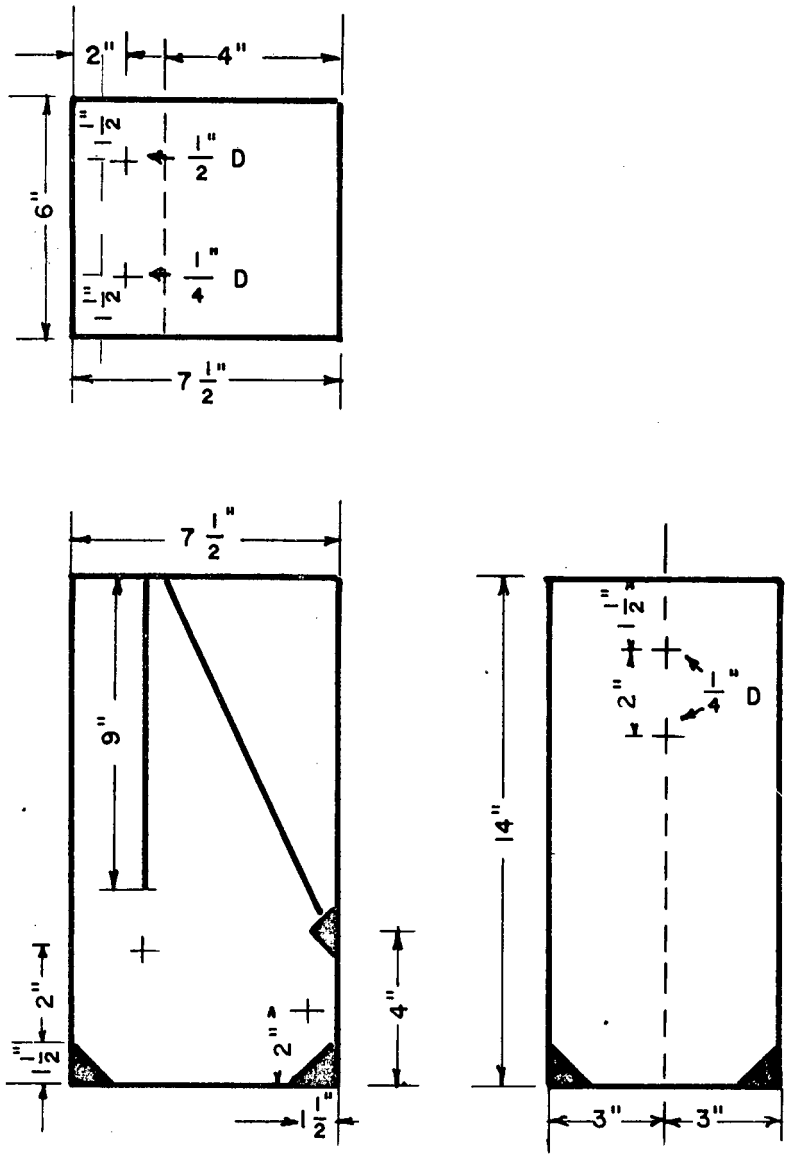

Figure 3.3. Dimensions of reactor-clarifier.

ensured that the digester was sealed effectively without restricting versatility of operation.

The baffles A, B and C shown in Figure 3.2, aided in maintaining adequate flow patterns in the digesting section of the system and thereby eliminating any dead spots which lead to inefficient utilization of the digester volume, or where settling could take place.

The baffle D was an important component of the clarifying mechanism in the digesters. Recirculated gas entered the digesters at the position F indicated in Figure 3.2. The bubbles, in rising, struck the baffle D, placed just beneath the settling section and continued upwards to the liquid surface. The flow pattern of the gas bubbles around this baffle resulted in the creation of eddies as shown in Figure 3.2, which drew the sludge out of the settling section. This system worked efficiently in practice and the settling section was kept clear throughout the investigation. An opening of 4 - 8 mm between the sliding baffle and the digester wall was found to give the most satisfactory results.

The degree of mixing achieved by the gas recirculation, aided by the positioning of baffles A, B and C in the digesting section, proved adequate throughout the investigation.

An aquarium pump was used to recirculate the gas and a water trap included in the circuit, as shown in Figure 3.2, to prevent any liquid from entering the pump.

The level of the liquid in the digester and consequently the volume of the digester contents could be varied by raising or lowering the outlet weir, F, in Figure 3.2. The volume of each of the digesting sections of the reactor-clarifier was maintained at 7,5 litres throughout the study.

A constant daily flow rate was applied to each of the digesters, viz. 5,155; 4,42 and 2,125 litres/day.

Consequently, each digester had a fixed HRT throughout the study, viz. 1,46 - 1,70 and 3,53 days respectively.

A 'drip-feed' system was used to feed the digesters. The system was based on the restriction of air entering the feed bottle. This resistance was created by placing a certain length of wire, C, into a capillary tube, providing the only access for air to enter the feed bottle, as shown in Figure 3.4. Minor adjustments to the flow rate could be made by varying the length of wire in the capillary tube, A. Major adjustments to the flow rate could be achieved by varying the height of the outlet weir, B, in Figure 3.4.

In order to simulate a continuous process as closely as possible, fluctuations in feed rate, commonly due to sludge growth in the feed times, had to be avoided. This was achieved by preparing new feed daily and by regular cleansing of the feed lines.

The feed tube entering the digester extended some 6" below the liquid surface in the digester, thus constituting an effective seal to air entering the system.

Digester effluent and gas lines are shown in Figure 3.2, the liquid seal in the effluent tube preventing gas from escaping through this outlet.

Gas was collected in precisely the same manner as that used in the batch system, described in section 3.2. The gas collection bottles were considerably larger than those used for the batch process, varying from 13 - 16 litres in volume.

A gas sampling point was included along the gas collection line as shown in Figure 3.2.

Sludge was wasted daily from the system due to the inherent difficultues in establishing a continuous removal system for such small quantities of sludge. Simulation studies

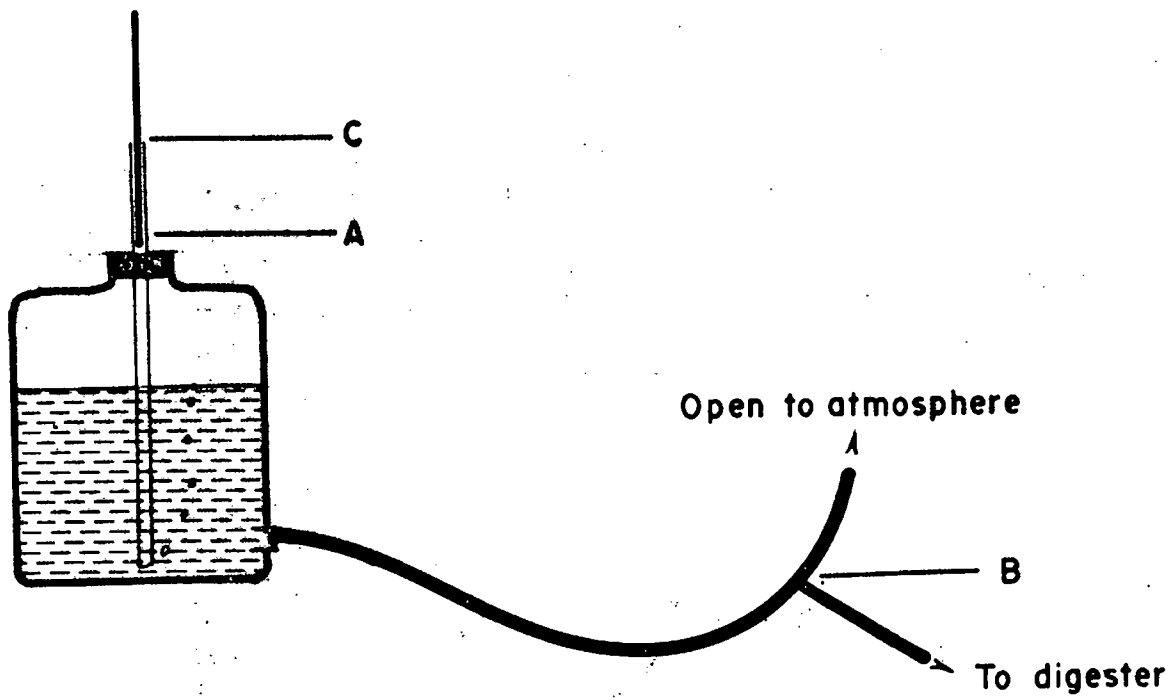

Figure 3.4. Drip feed system.

being carried out at present in the Department of Chemical Engineering, University of Cape Town, by Hansford and Ledermann (H1) have indicated that the range of errors, applicable to this intermittent method of sludge withdrawal would be minimal for anaerobic systems.

As mentioned in section 3.2, the temperature in the room containing the digester was maintained at 30 $\pm$ 1°C. This was achieved by monitoring the temperature daily on a maximum-and-minimum thermometer.

Batches of yeast waste were stored in a refrigerator at 4 $\pm$ 2°C. Daily feed volumes were prepared by diluting these batches of yeast waste with tap water. Each batch was seldom used for longer than two weeks before a fresh batch was obtained. The variation in chemical oxygen demand of the yeast waste did not exceed 2 per cent. during this period.

3.4 OPERATION OF REACTOR-CLARIFIER DIGESTERS.

As mentioned in section 3.3, the hydraulic retention time of each of the digesters was kept constant, viz. at 1,46 - 1,70 and 3,53 days respectively. Similarly to the batch system, increases in the loading rates on the digesters were achieved by increasing the feed concentration.

The tanks were sensitive to step increases in loading, of greater than ten per cent. at moderate loadings and five per cent. at higher loadings. This indicates the difficulty of applying the anaerobic digestion process to industrial wastes with widely varying waste streams. The use of a balancing or equalising tank would be essential if conventional anaerobic systems were to be used in such cases.

Maximum tolerable loadings varied from 14,5 kg/COD/m^3/day to 6,5 kg/COD/m^3/day as the hydraulic retention time increased from 1,46 to 3,53 days respectively. These loadings, however, corresponded to a similar concentration of feed entering the

system, viz. 0,021 - 0,023 kg COD/l. This indicated that the digesters were failing due to the presence of one or more materials, in inhibitory concentrations, at these loadings. The possible sources of this inhibition are treated more comprehensively in Chapter 6.

The digester with the shortest hydraulic retention time, viz. 1,46 days, was found to be more susceptible to sudden changes in the environment, than the other digesters. It would consequently appear that the digesters used in this study would probably fail at hydraulic retention times much below 1,6 days, due to an insufficient contact period between the microorganisms and the yeast waste. In the internally clarified type of system, this limiting hydraulic retention time is probably longer than that possible in the anaerobic contact process where an external recycle system is used. This is due to the limiting sludge concentration, imposed on an internally clarified system, by the hydraulics of the system. A further discussion of this particular disadvantage inherent to the reactor-clarifier type of system is presented in section 4.2.7.

In order to investigate the kinetics of the system it was necessary to evaluate the performance of the digesters over a range of solids retention times.

In conventional processes where no solids/liquid separation takes place, i.e. solids retention time is fixed by the hydraulic residence time, the retention time is varied by changing the flow rate through the system.

The extension of this method to the laboratory scale reactor-clarifier, used in this investigation, involves operating at various sludge ages controlled by fixing the amount of sludge removed per day. Under these conditions the cell contration adjusts itself to the particular steady state value required.

This method of operation suffers from one major disadvantage which is particularly pronounced in anaerobic systems, viz. the long period of time required by the anaerobic microorganisms to adjust from one steady state to another.

Process dynamics (C5) show that a completely mixed system should be left for at least five residence times before steady state conditions can be assumed.

In the reactor-clarifier system used in the present investigation comparatively long sludge retention times, viz. 40 - 60 days, were used while operating at a hydraulic residence time of two days.

Consequently to apply the period recommended by process dynamics to the contact process before reaching steady state, approximately 5 x 50 days would be required between steady state readings.

As this period of time was impractical, a novel method of operation was investigated in this study. Rather than maintain a fixed sludge age, the quantity of sludge wasted per day, i.e. the sludge age, was adjusted to maintain a constant sludge concentration in the digester at all loadings.

Under these circumstances, no changes in cell concentration of the slow growing anaerobic organisms was required. Therefore by maintaining a constant sludge concentration in the system a considerably shorter period should be required for a steady state condition to be established. Further it was felt that attainment of steady state could then be related more closely to hydraulic residence time rather than solids retention time.

This method of operation could be limited by the following problems :

(1) a difficulty in selecting the correct amount of sludge to waste in order to maintain a

constant sludge concentration in the reactors;

(2) a period related to a number of hydraulic residence times might prove insufficient for steady state to be attained.

However the results in moving from one steady state value to another as shown in Figure 3.5 indicate the following :

(a) it was relatively simple to select the required amount of sludge to be wasted per day in order to maintain a constant sludge concentration in the system. The level of sludge in the settling section was a useful indicator for this method of operation;

(b) steady state values as shown in Figure 3.5 and Table C.2, Appendix C, appeared to be reached within a period of two weeks which is considerably less than any sludge age used during the study and very close to a period of five times the hydraulic retention time. This indicates that the hydraulic retention time may be the characteristic time constant for such a system.

Pretorius (P7) has commented that researchers in the anaerobic fermentation field have been vague in their respective definitions of steady state. A period of three hydraulic retention times was used by Andrews and Pearson (A5) in a study of the kinetics applying to anaerobic systems. In the present study, each digester was loaded at the same rate and operated at the same environmental conditions, for a period equal to the solids retention time. Readings for the appropriate steady state were then based on the average of one week's readings.

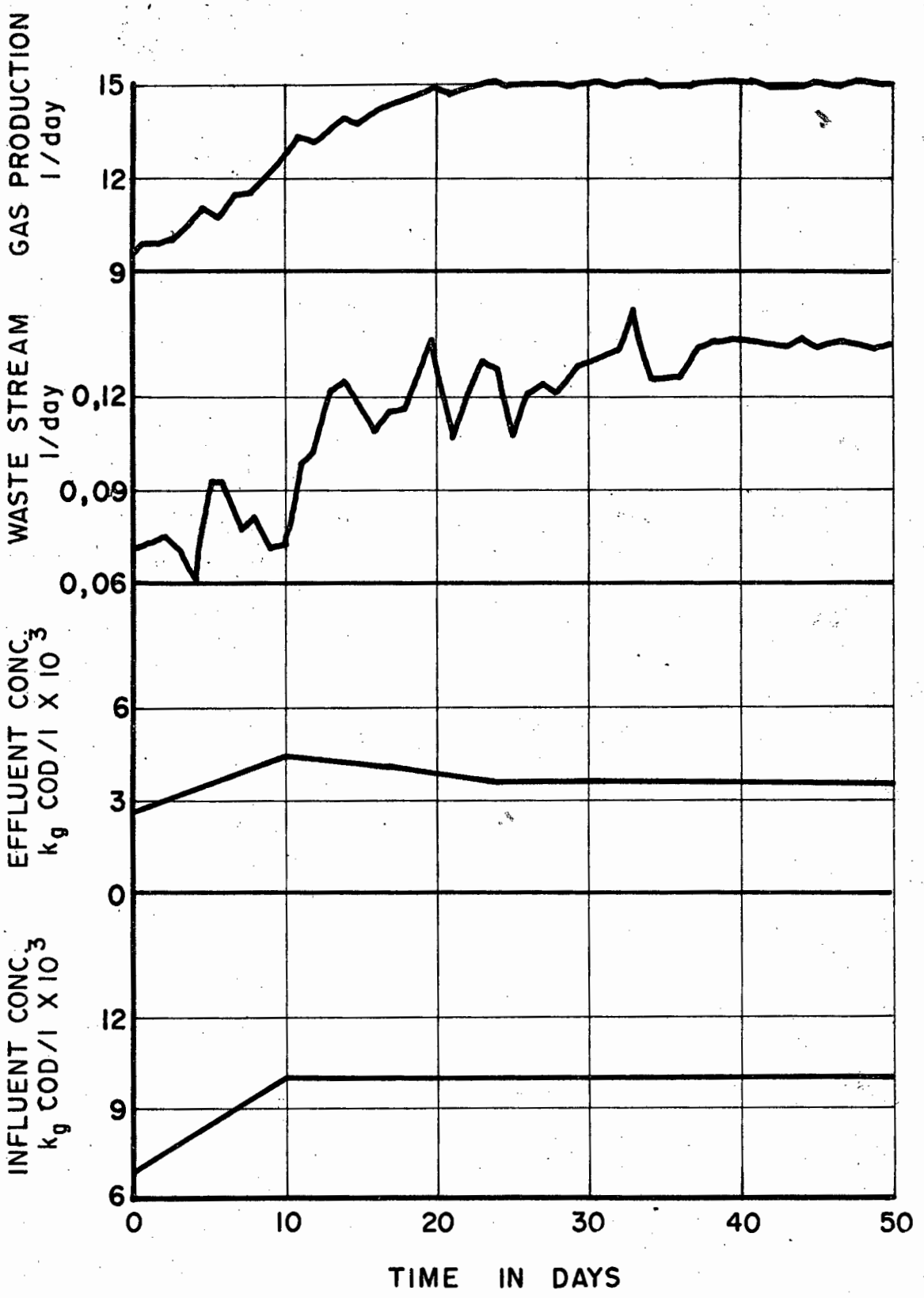

Figure 3.5. Changes in values of parameters from one steady state to another.

A change in loading rate on the digesters was accompanied by an increase in sludge generation and consequently more sludge was wasted to maintain a constant sludge concentration in the reactor section. The new steady state conditions were obtained by a trial and error method. Daily monitoring of sludge in the settling section was performed until the correct amount of sludge was wasted from the system, corresponding to the new environmental conditions imposed on the digesters.

A histogram of the various parameters relating to the digester as the system was moved from one steady state set of conditions to another is shown in Figure 3.5. As can be seen from Figure 3.5, the parameters describing the condition of the digesters remained constant over the period readings were taken.

Pretorius (P7), reported that although typical system parameters might indicate a steady state condition, recent research has shown that certain enzyme reactions had been found to show wide fluctuations which only tended to settle down at a later period.

Further research is necessary in this field in order to obtain a more specific definition of steady state conditions.

A summary of the various 'steady state' values for each digester is presented in Table 5.1.

All the parameters used to monitor digestion and to determine relative digester efficiency are discussed in section 3.5.

3.5 ANALYTICAL PROCEDURE.

The following analyses were performed to monitor digester performance and to evaluate the relative efficiency of the process.

Type of Analysis.	Reference.	
Biological Oxygen Demand	Standard Methods	(S15)
Chemical Oxygen Demand	Standard Methods	(S15)
Oxygen Absorption	Standard Methods	(S15)
Dissolved Solids: (Total, Volatile and Fixed)	Standard Methods	(S15)
Suspended Solids: (Total, Volatile and Fixed)	Standard Methods	(S15)
Total Nitrogen	Standard Methods	(S15)
Ammonia Nitrogen	Standard Methods	(S15)
[1]Volitile Acids Concentration	DiLallo and Albertson	(D1)
[1]Total Alkalinity	DiLallo and Albertson	(D1)
[2]Gas Composition	-	See below
Chlorides	Standard Methods	(S15)
Total Sulphides	Standard Methods	(S15)
[3]Potassium, Sodium	-	See below
[3]Copper, Iron, Zinc and Nickel	-	See below
[4]pH	-	See below

[1] Volatile acids analysis was performed by the method advocated by DiLallo and Albertson (D1). For comparative purposes a series of volatile acids analyses were performed using both the above method and a calorimetric method reported by Montgomery, Dymock and Thom (M22). The results of both methods appeared to be consistent with each other as shown in Table B.1, in Appendix B.

[2] Gas analysis was performed by the use of gas liquid chromatography. A Hewlett-Packard model 5750 chromatograph was used for the analyses. The various operating parameters are summarized in Table B.2, Appendix B.

3 Atomic absorption spectroscopy was used to determine the concentrations of potassium, sodium, copper, iron, zinc and nickel in the digesters. A Varian-Techtron model 1000 instrument was used for the analysis. The conditions applying during the analysis of each element are listed in Table B.3, Appendix B.

4 pH was monitored daily on a Beckman pH meter, (model 55-3).

3.6 SUMMARY.

The batch and 'reactor-clarifier' systems used in this investigation and the parameters pertaining to their operation, both operational and analytical, have been described.

The relevant results describing the general performance of these digesters are discussed in Chapter 4.

CHAPTER 4

GENERAL PERFORMANCE: RESULTS AND DISCUSSION

4.1 INTRODUCTION.

The susceptibility of yeast waste to anaerobic digestion is discussed. The relative behaviour of the parameters relating to anaerobic digestion is also presented.

A large portion of the detailed daily analyses performed during the investigation has been omitted, as it was felt that it would be largely repetitive and of little information, particularly during much of the preliminary periods of the study. Where relevant, data has been abstracted from the daily readings. This data can be found in Appendix C. For ease of interpretation, relevant portions of this data have been plotted chronologically as well as being presented in a tabular form.

The location of the data in this thesis together with its relevant application is summarized below :

(a) The behaviour of the parameters used to monitor digestion during a typical start-up period is presented in Figure 4.1 and in Table C.1, Appendix C.

(b) A comparison of digester performance after the addition of ammonium phosphate, as a possible nutrient source, compared to a normally operated digester, is shown in Figure 4.2.

(c) The values of the parameters used to monitor digestion in the reactor-clarifier system, as an increase in loading concentration caused a shift from one steady state to another, is

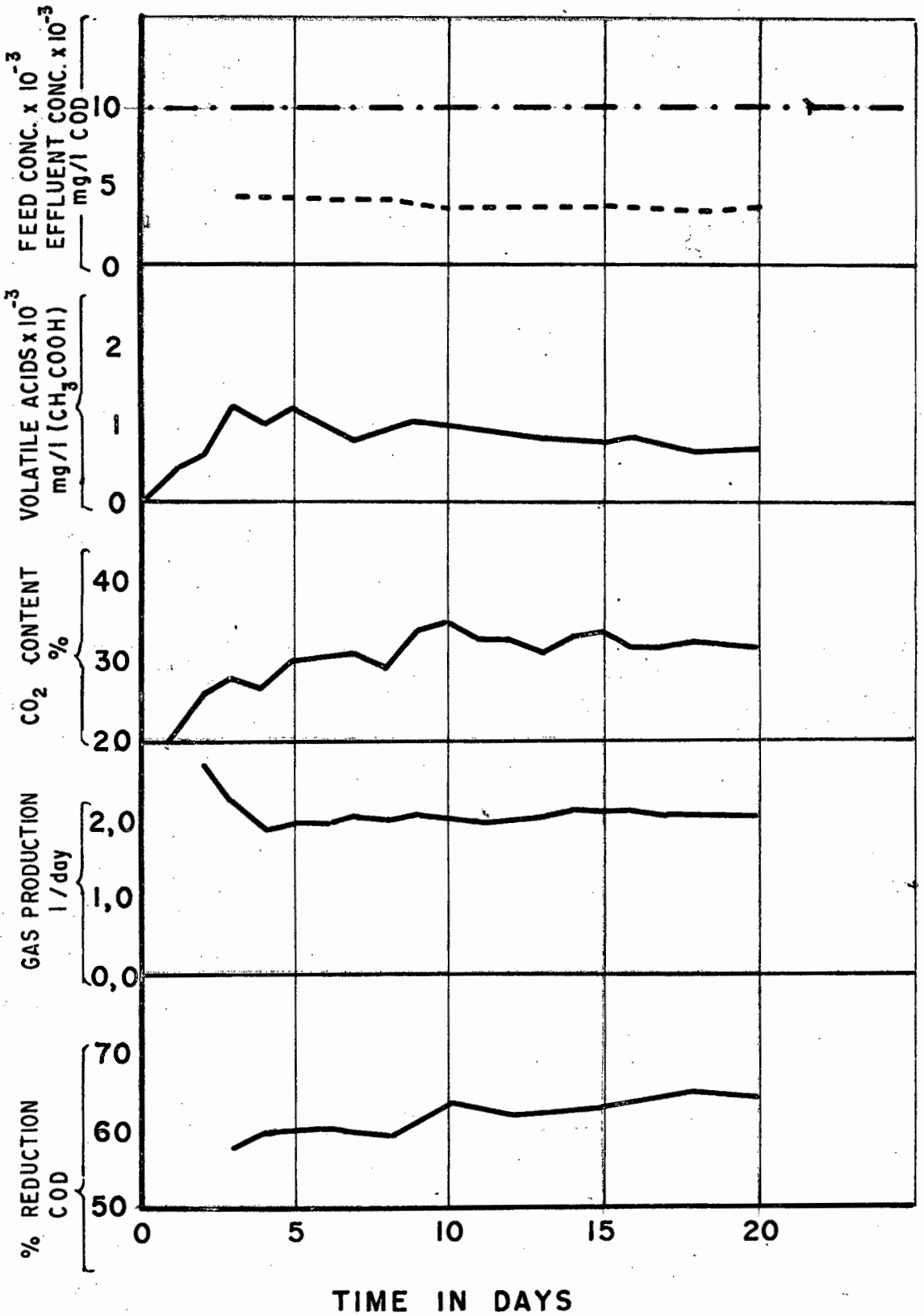

Figure 4.1. Behaviour of parameters used to monitor digestion during typical start-up period.

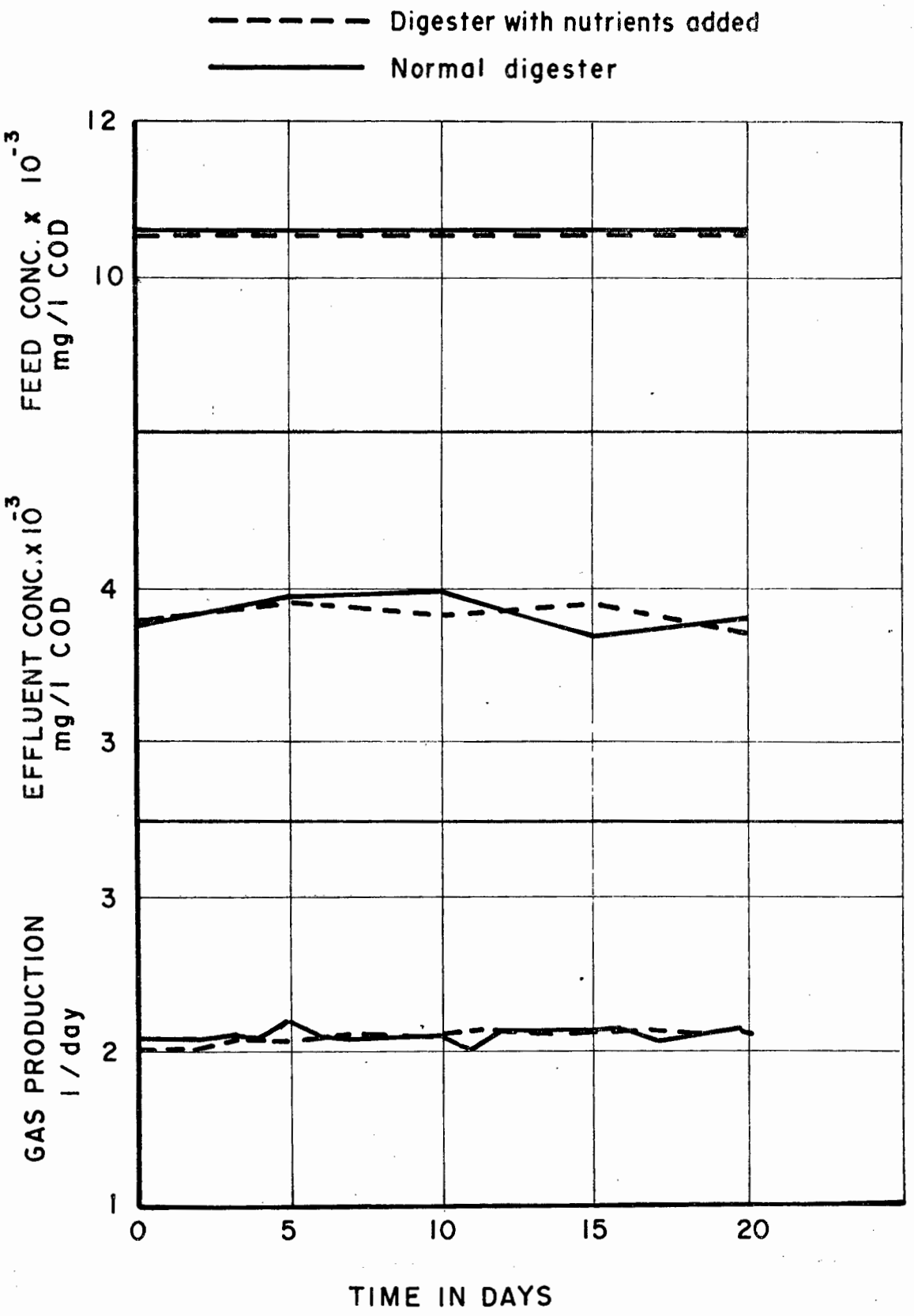

Figure 4.2. Effect of the addition of nutrients to the digester.

shown in Figure 3.5, and can be found in Table C.2, Appendix C.

(d) A summary of the steady state values obtained for the reactor-clarifier systems is listed in Table C.3 in Appendix C. Figures 4.4, 4.5, 4.6 and 4.7 contain data abstracted from Table C.3, and summarize the variation of the common parameters, viz. total gas production, gas composition and volatile acids concentration, with changes in loading rate. Where possible, these values have been compared to values arising from the anaerobic treatment of other industrial wastes.

(e) The significance of changes in the values of the above parameters in predicting process imbalance, is shown in Figure 4.8.

(f) Finally, the relevant data pertaining to the investigation on inhibition is listed in Table C.4. The subject of inhibition is discussed in more detail in Chapter 6.

4.2 GENERAL OPERATION.

4.2.1 Start Up.

Seed from a primary digester at the Athlone sewage works was used to start up the various digesters.

Virtually no acclimatization period was found necessary as digestion of the yeast waste proceeded almost immediately. Maximum efficiency was rapidly attained as shown in Figure 4.1.

The rapidity with which maximum efficiency was reached, contrasted with the acclimatization period required for sewage sludge seed to adapt to spent wine as a substrate, Stander (S11).

In an aerobic study carried out in the Department of Chemical Engineering, University of Cape Town, (S13), acclimatization periods of approximately one month were required before maximum utilization of the same yeast waste was attained.

4.2.2 Nutrients.

Satisfactory performance over a wide range of loadings, indicated that no extra nutrients were required by the various micro-organisms in utilizing yeast waste as a substrate.

In preliminary studies of the system, both nitrogen and phosphorus were added in various amounts to the digesters without any effect on process efficiency, as shown in Figure 4.2.

Similar reductions obtained from both the anaerobic and aerobic methods of treating the yeast waste under consideration, indicated that the waste was not nutrient deficient (S13). Had this actually been the case, reductions in the aerobic process would be suspected to have been lower than the anaerobic process due to the greater sludge sunthesis and consequently increased nutrient requirements of the aerobic process.

4.2.3 Buffer Requirements for Anaerobic Digestion.

The following buffer systems were generally present in the digesters:

(a) the bicarbonate system either generated during

digestion as shown in sections 2.4.3 (a) and 2.4.3 (f), or added to the system in the form of sodium bicarbonate;

(b) the sulphide system, resulting from the reduction of sulphates, present in the yeast waste;

(c) the ammonia system, resulting from the ammonia originally present in the yeast waste, as well as that generated in the system from the reduction of organic nitrogen;

(d) a volatile acids system, resulting from the acids generated in the intermediate stages of digestion and their consequent neutralization by the various forms of alkalinity present in the digester.

In the normal range of pH experienced during this study, viz. 6,8 - 7,2 , only the first two buffer systems had a significant effect on the digestion process. The relative contributions of both systems to the total buffer capacity for a typical digester environment is shown in Figure 4.3. At a pH of 7,0 , the sulphide system contributes approximately 10 per cent. of the total buffer capacity for a sulphide concentration of 75 mg/l.

Increases in pH values above 7,2 were noted during the preliminary periods of the study when extremely high loading rates were attempted. These increases were due to an increase in the ammonia concentration in the digester and consequent displacement of the pH as shown in section 2,4,3 (e).

Similarly, periods of unstable digestion, characterized by high concentrations of volatile acids, resulted in a depression of the pH.

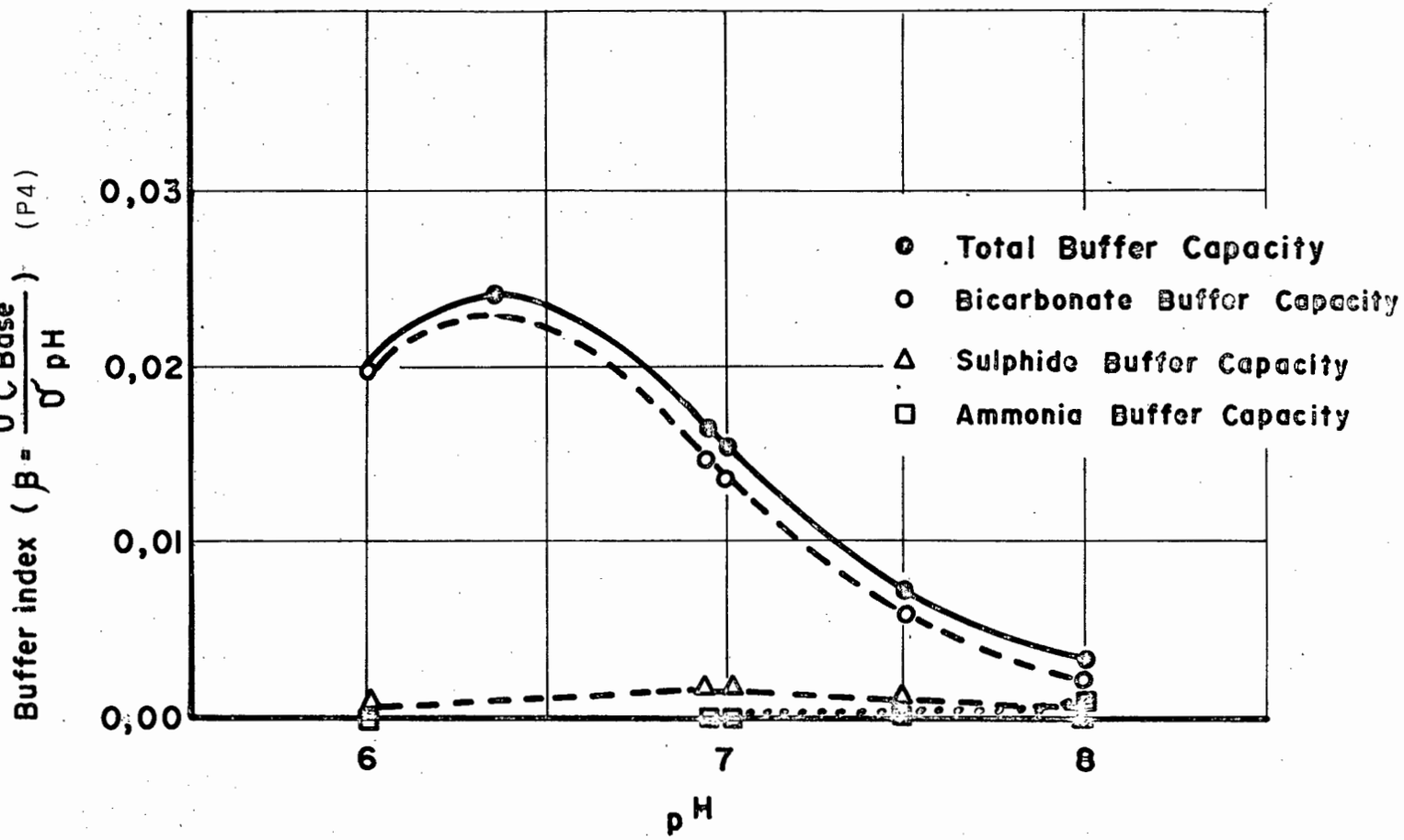

Figure 4.3. Typical components of the buffer capacity present in the reactor-clarifier.

As mentioned in section 2.4.3, an alkalinity level of approximately 3 000 mg/l as $CaCO_3$ is recommended for digestion, to act as a buffer against any build up of volatile acids during periods of digester imbalance.

At low feed concentrations, insufficient alkalinity was present in the waste, or generated during treatment, to maintain satisfactory digestion. In such cases, sodium bicarbonate was added to the waste to maintain a satisfactory level of alkalinity in the system.

However, at loading rates greater than 5 kg COD/day/m^3 sufficient alkalinity was present, in the yeast waste and generated during digestion, to sustain the process.

4.2.4 General Performance.

A typical performance of the digesters for a given loading rate is presented in Table 4.1 while approximate values of the range of reductions obtained over the entire investigation is shown in Table 4.2.

TABLE 4.1. Digester Performance Data.

Parameter	Influent mg/l	Effluent mg/l	Reduction %
Total Chemical Oxygen Demand (COD)	16 100	5 300	67,1
Degradable COD	12 300	1 500	87,9
Total Dissolved Solids	16 700	9 600	42,5
Total Dissolved Volatile Solids	11 000	4 100	62,6
Total Fixed Volatile Solids	5 700	5 500	3,5
Oxygen Absorbed	6 100	1 800	70,5
Biological Oxygen Demand	8 800	1 010	88,5
Total Nitrogen	280	142	49,3
NH_3 Nitrogen	77	132	-
Sulphates	1 000	450	55,0

TABLE 4.2. Range of reductions covering all loading rates. (All values rounded off.)

Parameter	Range of Reductions %
Total Chemical Oxygen Demand (COD)	60 - 70
Degradable COD	80 - 90
Degradable COD as a function of Total COD	70 - 80
Total Dissolved Solids	40 - 55
Total Dissolved Volatile Solids	60 - 70
Total Dissolved Fixed Solids	0 - 10
Oxygen Absorbed	65 - 80
Total Nitrogen	40 - 60
Sulphates	40 - 60

Overall COD reductions of 60 to 70 per cent. were consistently obtained over a comparatively wide range of loadings. Reductions of the biodegradable portion of the waste ranged from 80 to just over 90 per cent.

The biodegradable portion of the waste varied between 70 and 80 per cent. of the total COD. Consequently some 20 to 30 per cent. of the yeast waste was not susceptible to biological treatment.

A study, carried out in the Department of Chemical Engineering, University of Cape Town, on the aerobic treatment of the same yeast waste yielded similar results in evaluating the degradability of the yeast waste (S13).

Reductions in total dissolved solids ranged from 40 to 55 per cent. This comparatively low reduction was due to the large amount of inorganic salts present in the yeast waste. Part of the very small reduction in fixed or non-volatile dissolved solids can be attributed to the reduction of sulphates in the waste to hydrogen sulphide. Reductions

in the volatile content of the dissolved solids appeared to be similar to the reductions obtained in the total COD.

Representative values of the above reductions, over the range of loadings used during this study, are summarized in Table 4.2.

Gas production, as a function of kg COD destroyed, is compared to the theoretical amount, on a percentage basis in Figure 4.2. An average value of 86 per cent. was reported by Cillie et al. (C2) for the treatment of wine waste. Part of the deficit is due to the synthesis of approximately 10 per cent. of the applied COD into cellular material.

The parameters used to monitor digestion are discussed in section 4.3.

4.2.5 Sludge Concentration.

As seen from Equation (5.13) in section 5.3 the following expression can be derived relating the sludge concentration to the hydraulic retention time for the reactor-clarifier :

$$X = \frac{(So - S)Y}{1 + k_d\theta_c} \cdot \frac{\theta_c}{\theta}$$

From the above expression it can be seen that a decrease in hydraulic residence time (θ), should be accompanied by an increase in the solids concentration in the digester (X). In other words, to compensate for a decrease in contact time between the micro-organisms and waste, a consequence of the decrease in hydraulic residence time, an increase in sludge concentration is required for satisfactory digester performance to be maintained.

Unfortunately, contrary to the biological requirements indicated by Equation (5.13), the sludge concentration in the

reactor-clarifier used during this study was lowest for the digester being operated at the lowest hydraulic retention time. This would appear to be related to the increased amount of micro-organisms carried over in the effluent stream at increased flow rates through the system. Consequently, high loadings on the process at low hydraulic retention times could be limited by insufficient sludge concentrations. Stander (S11) has reported a similar limitation on the loading of spent wine on a full-scale clarigester.

This limitation constitutes the major disadvantage of the system used in this study and of all similar forms of clarigesters. The significance of the above limitation will depend on the strength of the waste to be treated. Nevertheless, loadings used on the digesters treating the yeast waste under consideration, were comparable to many of the higher loading rates reported in the literature.

Sludge concentrations varied from approximately 14 500 mg/l to 11 500 mg/l as the hydraulic retention time decreased from 3,53 to 1,46 days. The volatile fraction appeared to remain at a constant value between 80 and 87 per cent. of the total sludge concentration.

4.2.6 Summary.

It has been shown that approximately 70 to 78 per cent. of the yeast waste is biodegradable.

The degradable portion of the yeast waste is readily amenable to conventional anaerobic treatment.

Virtually no acclimatization period was found to be necessary and no additional nutrients required to sustain digestion.

Chemical additions were required at low feed concentrations to maintain a satisfactory alkalinity level in the digester. At higher loadings the alkalinity present in the waste and generated during digestion, proved adequate for digestion to proceed satisfactorily.

Loadings appeared to be limited to a certain feed concentration, indicating the presence of one or more inhibitory materials in the waste. A hydraulic residence time of just below 1,6 days was thought to be the lowest value possible for sufficient contact to be established between the waste and the micro-organisms.

4.3 PARAMETERS USED TO MONITOR DIGESTION.

4.3.1 Introduction.

Values of the various parameters used to monitor digestion, over the range of loading rates used during this study, are presented in Figures 4.4, 4.5, 4.6 and 4.7. Where possible, these values are compared to other published values recorded during the anaerobic treatment of other industrial wastes.

4.3.2 Comparison of Parameter Values.

Unfortunately, comparison of the various parameters pertaining to anaerobic digestion is complicated by the variety of expressions used to describe both the loading rates and performance of these digesters. Although more than one expression is frequently used to describe the characteristics of a process, these expressions are often limited to the influent conditions or merely to the waste itself. Any comparison concerned with the overall reduction of the process is often meaningless unless some ultimate reduction is quoted, particularly in the case of wastes which are only partially

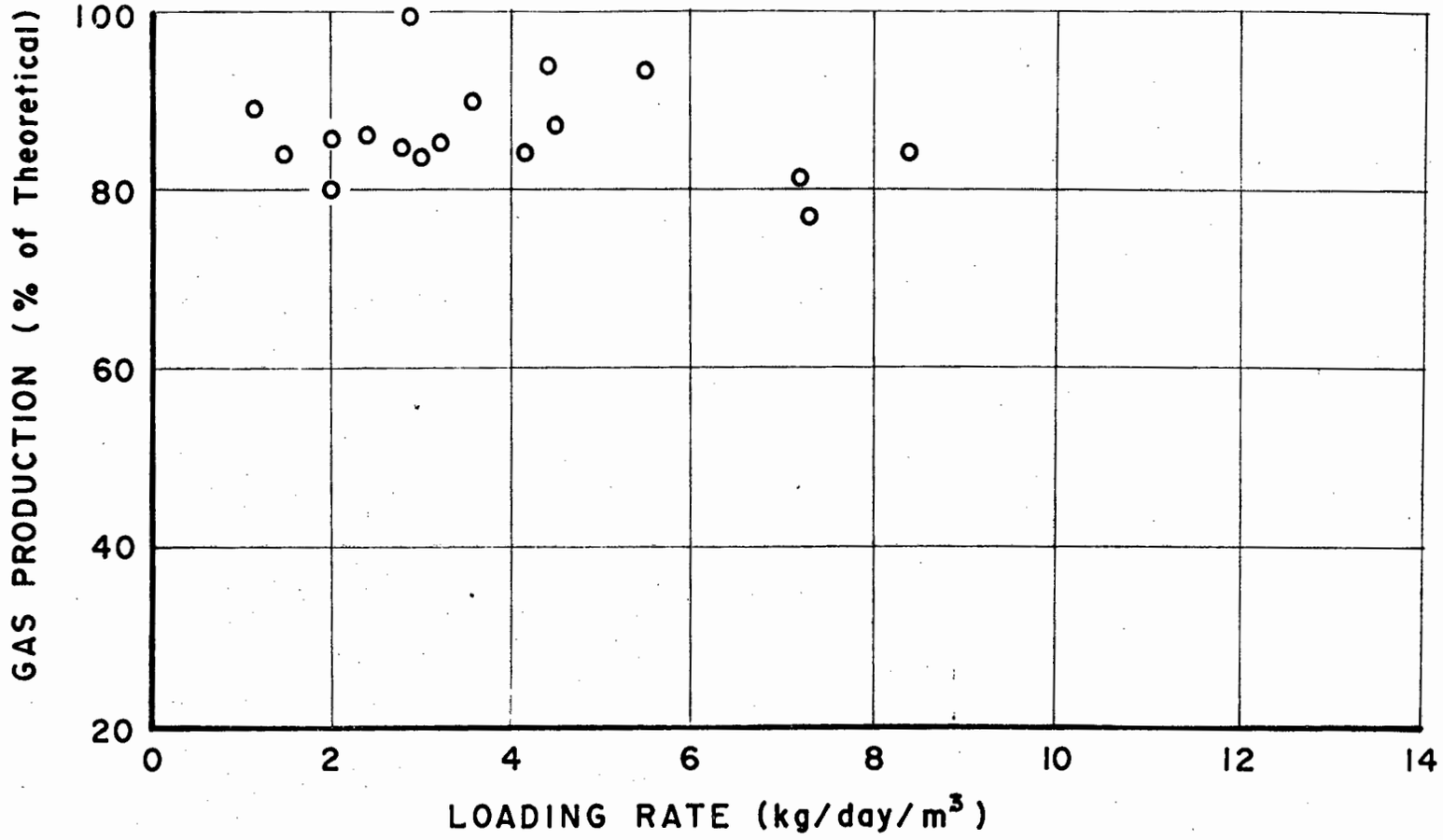

Figure 4.4. Gas production as a percentage of the theoretical amount based on substrate utilization.

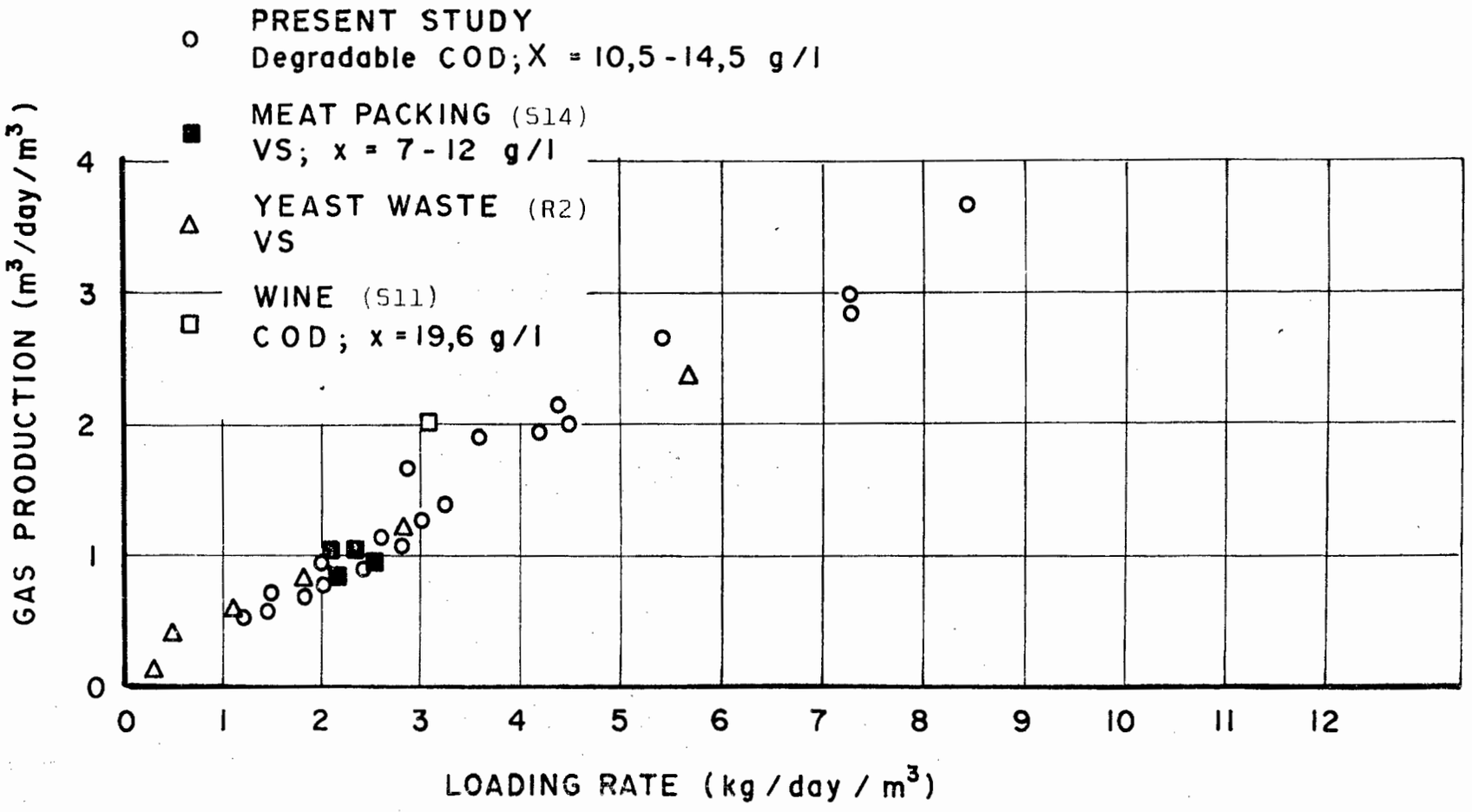

Figure 4.5. Variation of gas production with loading rate.

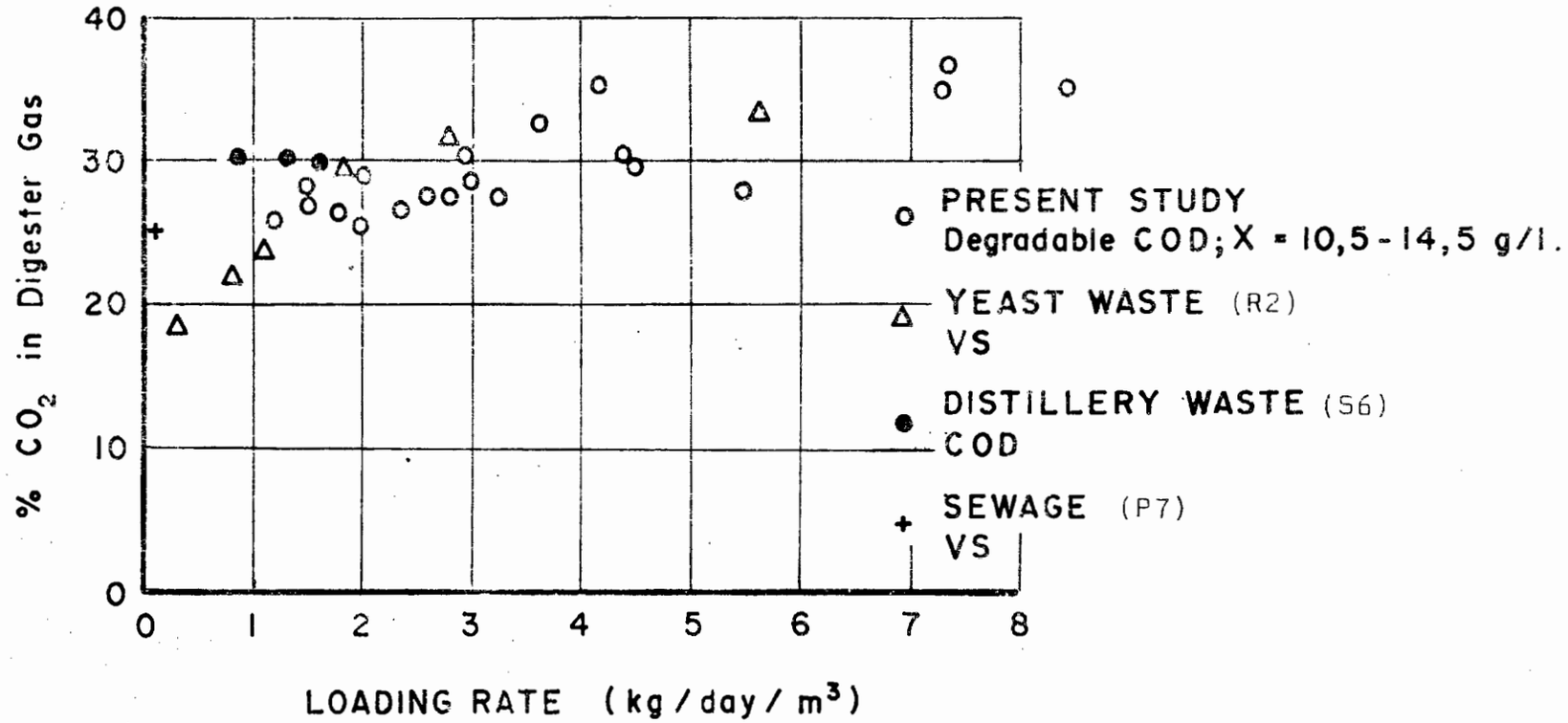

Figure 4.6. Variation in carbon dioxide content of the digester gas with changes in loading rate.

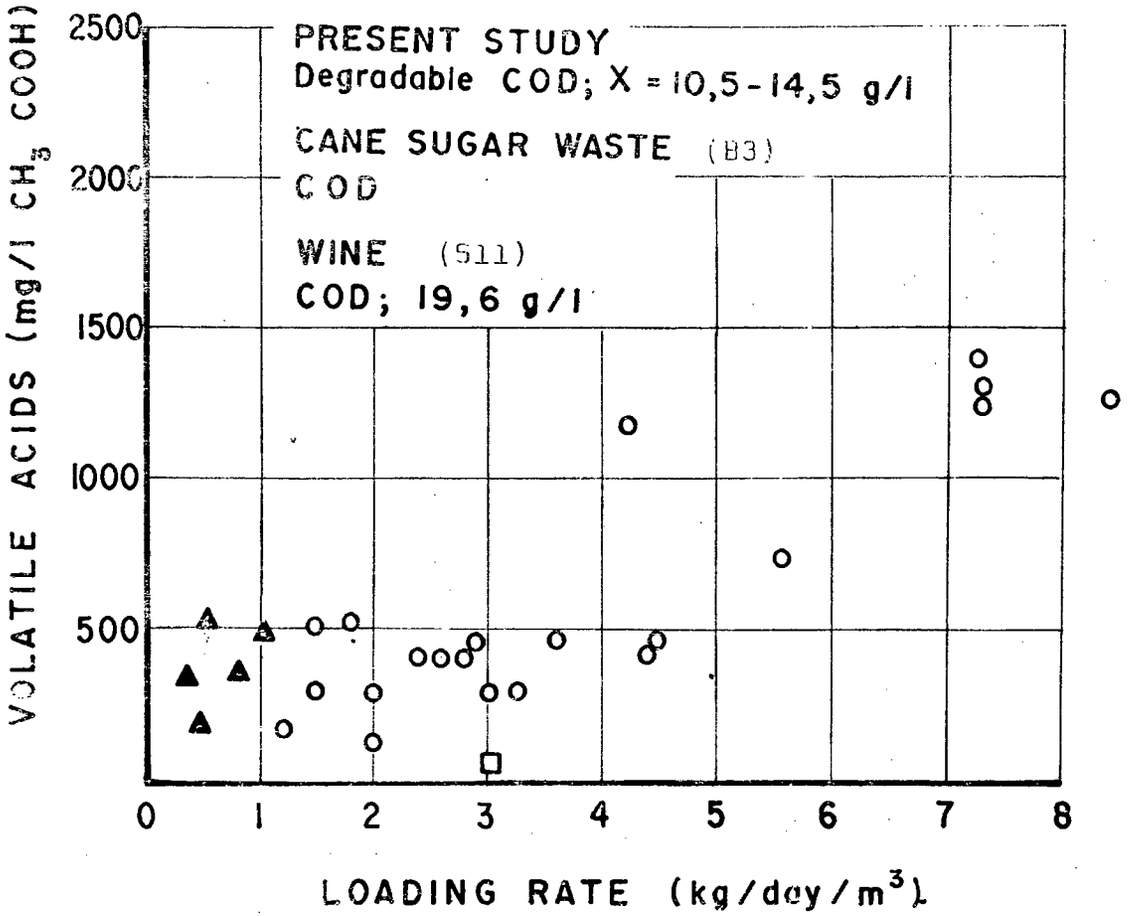

Figure 4.7. Variation in the concentration of volatile acids present in the system at different loading rates.

degradable. Further, it would appear that loading rates should be related to the concentration of solids in the digester in order to make meangingful comparisons between various processes. This has become more apparent with the development of various contact processes, permitting a wide range of sludge concentrations during anaerobic treatment.

Where possible in this thesis, comparisons have been made of the parameters used to monitor digester performance, for a range of loading rates related to degradable Chemical Oxygen Demand or Total Dissolved Volatile Solids. These comparisons are shown in Figures 4.5, 4.6 and 4.7.

It should be emphasized that the comparisons concerning the carbon dioxide content of the gas and the volatile acids concentrations are seriously limited by the lack of information on sludge concentrations. Where possible they are included on the graphs. However, it was decided to include these comparisons merely to indicate the general similarity of both the above parameters over a wide range of loadings for different wastes, viz. the carbon dioxide content of the gas varies between 25 and 35 per cent. while the volatile acid concentrations lie below 500 mg/l, as acetic acid, for the major portion of the loading rates presented.

The comparison between the various gas productions of the processes present is not subject to the above limitation. Both gas production rates and loading rates are plotted as a function of digester volume.

Figures 4.2, 4.3 and 4.4 do indicate that the values of all the parameters used to monitor digestion are within the range of values common to most digestion systems.

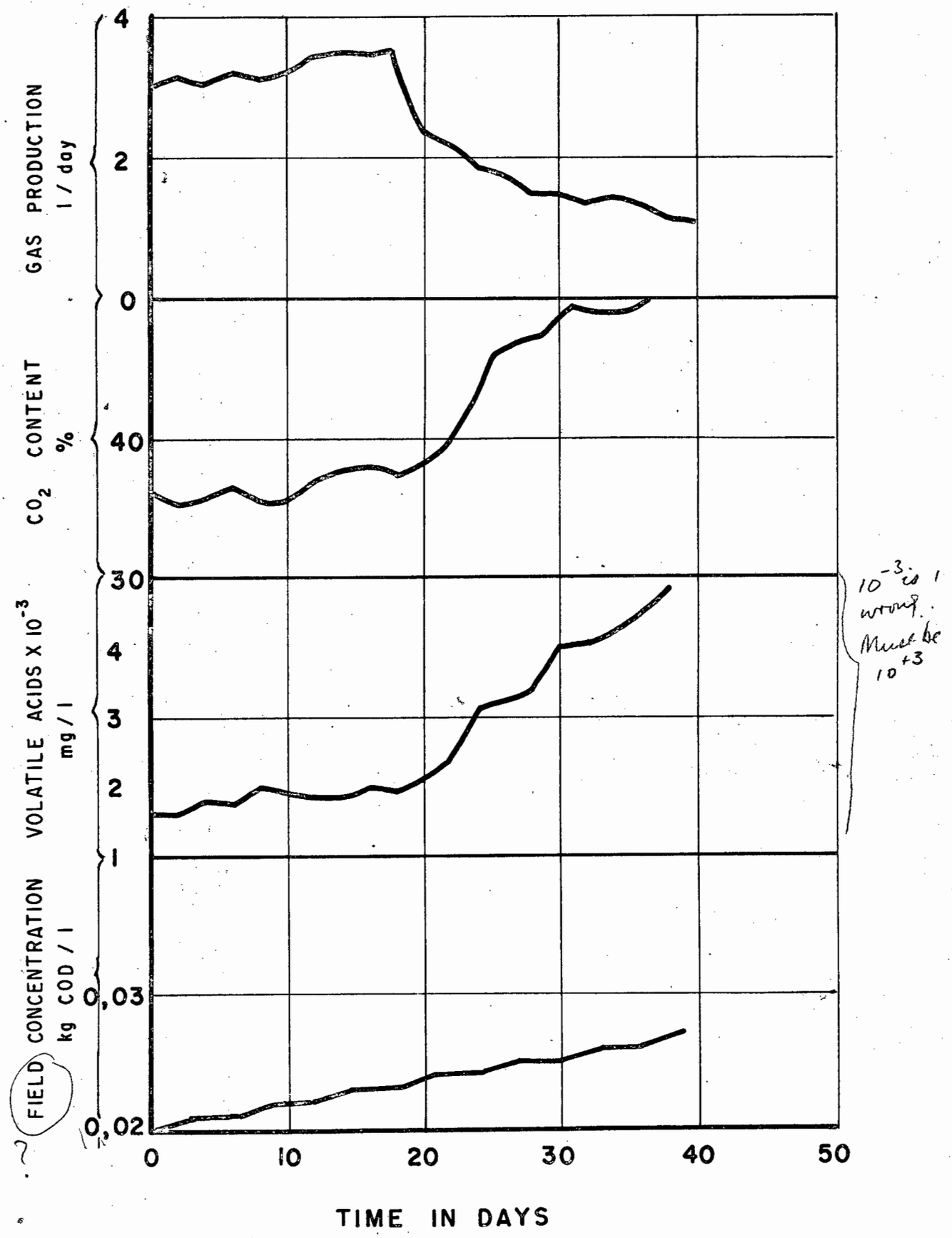

Figure 4.8. Behaviour of parameters used to monitor digestion during unstable periods.

4.3.3 Summary.

Difficulties have been indicated in the comparison of the various parameters used to monitor digestion, for a variety of different wastes.

Where possible comparisons of these parameters have been made for the anaerobic treatment of various industrial wastes and found to be similar to those values obtained during this study.

4.4 PARAMETERS RELATED TO DIGESTER FAILURE.

4.4.1 Introduction.

In order to gain an insight into the relative ability of each of the parameters, to predict impending process imbalance, one of the digesters was loaded to failure.

A histogram of the relative changes in the values of total gas production rate, gas analysis and volatile acids concentration covering the period of digester failure, is shown in Figure 4.8. Fig 4.8. is not a "Histogram" but a frequency polygon!

4.4.2 Gas Production.

The total gas production of the digester was found to decrease markedly at the first sign of process instability. The rapidity with which total gas production responded to changes in digester environment made it the most significant parameter for determining the relative condition of the digesters. The immediate indication total gas production gives of impending process failure, is shown in Figure 4.8.

It should be emphasized that fluctuations in loading rates were kept to a minimum throughout the study. In

larger plants, severe fluctuations in loading rates could limit the use of total gas production as a method of monitoring digester performance. However, even in such cases the extreme sensitivity of gas production to digester environment should overcome the above limitation to its use in anaerobic digestion.

4.4.3 Gas Composition.

As seen from Figure 4.8, the carbon dioxide content of the digester gas was found to be a reasonably sensitive indicator of digester performance. However it can clearly be seen that the drop in overall gas production on day 20 in Figure 4.8 is more marked than the rise in carbon dioxide content of the gas.

As shown in section 2.5.3, Miller and Barron (M17) have reported the carbon dioxide content of the gas as being a reliable indicator of digester performance. In the present study, however, gas production was found to be the more sensitive indicator.

4.4.4 Volatile Acids.

Figure 4.8 indicates the steep climb in volatile acids concentration once digester instability becomes marked. However, at the relatively high operating level of volatile acids in the digester under consideration, the onset of unfavourable digestion may be marked, as shown in Figure 4.8. It is quite clear that a severe drop in gas production has taken place before a significant increase in the volatile acids concentration is noted. Consequently in this study the volatile acids concentration was not found to be as sensitive to digester performance as the total gas production or gas composition.

Steffen and Bedker (S14) have also reported that volatile acids concentration responded at a slower rate than gas production to changes in the digester environment.

4.4.5 Alkalinity.

An increase in alkalinity was noted as the digester began to fail. This increase was due to an increased amount of volatile acids alkalinity present in the system, and eventually the influence of this type of alkalinity, on the process, resulted in a significant drop in the pH prevailing in the digesters.

The use of total alkalinity minus the volatile acids alkalinity, as a means of detecting the onset of digester imbalance has been found informative by Pohland and Bloodgood (P2).

In the present study, the overall gas production rate was found to be more sensitive than the above parameters in predicting digester imbalance.

4.5 SUMMARY.

The degradable portion of the yeast waste under consideration was found to be readily susceptible to anaerobic digestion. Digestion was self sustaining at higher loading rates.

The values of the parameters used to monitor digestion were similar to those reported during the anaerobic treatment of other wastes.

Total gas production was found to be the most reliable indicator of digester performance.

CHAPTER 5

K I N E T I C S

5.1 INTRODUCTION.

The sanitary engineering approach to the kinetics of waste treatment has been based largely on substrate utilization, McCarty (M13), Weston and Eckenfelder (W2), as opposed to the traditional culture theory which uses bacterial growth rate as a basis, Aiba et al. (A2).

Unfortunately there has been little uniformity in the systems of nomenclature used by workers doing research in the field of digester kinetics, e.g. McCarty (M13), Andrews and Pearson (A5) and Andrews (A6).

The nomenclature used in this thesis, as detailed in Appendix A, is the same as that recommended by a committee organized at the Symposium on Continuous Fermentation held in Czechoslavakia in 1962 (F2). This system has also been used by Aiba et al (A2) in a bio-chemical engineering text and further by Andrews (A6) and Pretorius (P6) in the field of waste treatment.

A mathematical model is developed to describe the reactor-clarifier system used in this investigation, and the application of similar models to full scale digesters is discussed.

A method of obtaining the kinetic parameters relating to the anaerobic treatment of yeast waste, is outlined. These parameters are evaluated and compared to other values found in the literature.

5.2 BACKGROUND THEORY.

Continuous culture theory has been shown by Andrews and Pearson (A5) to be applied to such diverse areas as the study of bacterial mutation, the study of rumen metabolism and the production of yeast, ethanol and lactic acid, as well as waste treatment.

However, classical continuous culture theory applies to the growth of a single species of micro-organisms in the presence of a single growth limiting substrate.

Anaerobic treatment, as mentioned in section 2.2.3, is a complex multi-step process involving a mixture of degradable compounds and a mixed microbial population.

Consequently, several assumptions have had to be made in order to apply continuous culture theory to anaerobic digestion, as shown below.

Anaerobic digestion can be represented by the following steps (W3) :

$$\underset{\text{liquefying bacteria}}{\text{solids}} \rightarrow \text{soluble organics} \xrightarrow{k_1} \underset{\text{acid forming bacteria}}{\text{acids}} \xrightarrow{k_2} \underset{\text{methane bacteria}}{CH_4+CO_2}$$

The following assumptions can be made relating to the above reaction sequence :

(a) the liquifaction step can be omitted in the case of soluble wastes;

(b) the reaction constant k_2 is assumed to be much lower than k_1. The methane forming step becomes rate limiting for the overall process. This assumption was suggested

by Sawyer and Roy (S3), and verified by McCarty (M13) and O'Rourke (O1);

(c) the rate limiting methane forming step is assumed to be described by a Monod relationship, Equation (5.1). In practice, this process-controlling step involves the utilization of a number of intermediate volatile acids by a mixed bacterial population. Consequently, a further assumption is made that a form of the Monod relationship can be applied to the fermentation of the volatile acids, where the volatile portion of the total dissolved solids concentration is used in the place of the concentration of growth-limiting substrate, Agardy et al. (A1) and Andrews et al. (A4).

Consequently, in the following development of the anaerobic model, the mixed population is considered to be represented by a single species and the complex organics, as measured by the degradable COD, to constitute the growth limiting substrate.

In continuous culture theory the growth rate of cells is given by the following expression :

$$\mu = \frac{1}{X^*} \frac{dX^*}{dt} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (5.1)$$

where X^* = cell concentration

The relationship between growth rate and the growth limiting substrate is given by the Monod expression :

$$\mu = \frac{\hat{\mu}\, S^*}{K + S^*} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (5.2)$$

where S^* = concentration of the growth limiting substrate.

Several other expressions describing growth rate as a function of substrate have been suggested, e.g. Moser (M22). However, in the following development, the Monod function will be adhered to, due to its wider acceptance.

The same expression has been widely used in the waste treatment field, with the following modifications :

(a) X* is replaced by X, which refers to the organism concentration in the particular waste system;

(b) S* is replaced by S which represents the rate limiting substrate in the complex waste.

The volatile portion of the sludge concentration in the reactor has frequently been used as a measure of the organism concentration, Andrews et al. (A4).

Biological Oxygen Demand and degradable Chemical Oxygen Demand have been commonly used as the growth limiting substrate in studies on the kinetics of anaerobic systems (A1), (A4). Not all of the studies reported in the literature have used an energy source as a limiting substrate. James (J1) has indicated that a wide variety of substances can act as growth limiting substrates.

For efficient waste treatment, both aerobic and anaerobic processes usually operate at low substrate values, and consequently, as can be seen from the Monod expression, Equation (5.2), at low specific growth rates. Under such circumstances the micro-organisms are retained in the system long enough for organism decay to exert an influence on the overall process.

To provide for all the factors, other than growth rate, influencing the mass of micro-organisms in a biological system, a specific decay rate k_d is included

Andrews (A7) suggested that the effects of cell maintenance, organism decay and subsequent lysis, on the overall growth rate, be included in this manner.

A number of writers (M13), (A7), (P6), have shown that the net growth rate can be expressed as follows :

$$\frac{dX}{dt} = (\mu - k_d)X \quad \ldots\ldots\ldots\ldots\ldots\ldots \quad (5.3)$$

Andrews (A7) has recognised that the specific organism decay rate is not a true constant, and reports that it has been found to decrease with a decrease in sludge age. However, the same writer feels that the organism decay term can be satisfactorily used over a relatively limited range of solids retention times.

Yield Coefficient.

Monod (M18) established that for a given organism, and given limiting nutrient, the mass of organisms produced per mass of nutrient utilized, under similar conditions, is constant. Further research by other microbiologists, using different organisms and substrates, have confirmed Monod's contention (A5).

Growth yield has commonly been defined (P6), (A5), as follows :

$$Y = \frac{\text{weight of organisms formed}}{\text{weight of essential limiting substrate utilized}}$$

Moser (M22) expressed the yield as a differential as shown in Equation (5.4) :

$$\frac{dX}{dt} = - Y \frac{dS}{dt} \quad \ldots\ldots\ldots\ldots\ldots\ldots \quad (5.4)$$

The above concept (Equation (5.4)) will be adhered to during the development of the model in this chapter.

Andrews (A7) reports the yield coefficient, as used in Equation (5.4), to be a function of the type of substrate, species of micro-organisms and environmental conditions. He concedes, however, that the yield coefficient has been found to be constant, for a specific biological process treating a specific waste.

5.3 DEVELOPMENT OF MATHEMATICAL MODEL.

Using the basic concepts of Equations (5.2), (5.3) and (5.4), and applying relevant material balances, it is possible to derive expressions for sludge concentration and effluent strength from waste digesters with and without recycle. The digester with recycle, Figure 5.1, is considered to consist of a completely mixed section of volume, V^1, and a settling section of negligible volume.

The same final equations, obtained from the above model, can be derived for a single unit, Figure 5.2, having different rates of removal of clear settled liquid and fully mixed liquor. This model has been adopted in preference to those described in the above paragraph, as it approximates more closely the reactor-clarifier units employed in the present study and does not involve assumptions concerning the unclarified and recycle streams.

As shown in Figure 5.2, a single unit, with one influent stream and two exit streams, is considered. The feed stream, F, enters the digester as shown in Figure 5.2. This stream is assumed to contain no micro-organisms. The stream, E, leaving the digester, represents the clear overflow from the settling stream and is assumed to contain no sludge. The underflow stream, W, is assumed to contain sludge at the same concentration as the digester. Because of the assumption of complete mixing, both E and W have the same waste strength, S, as the digester itself.

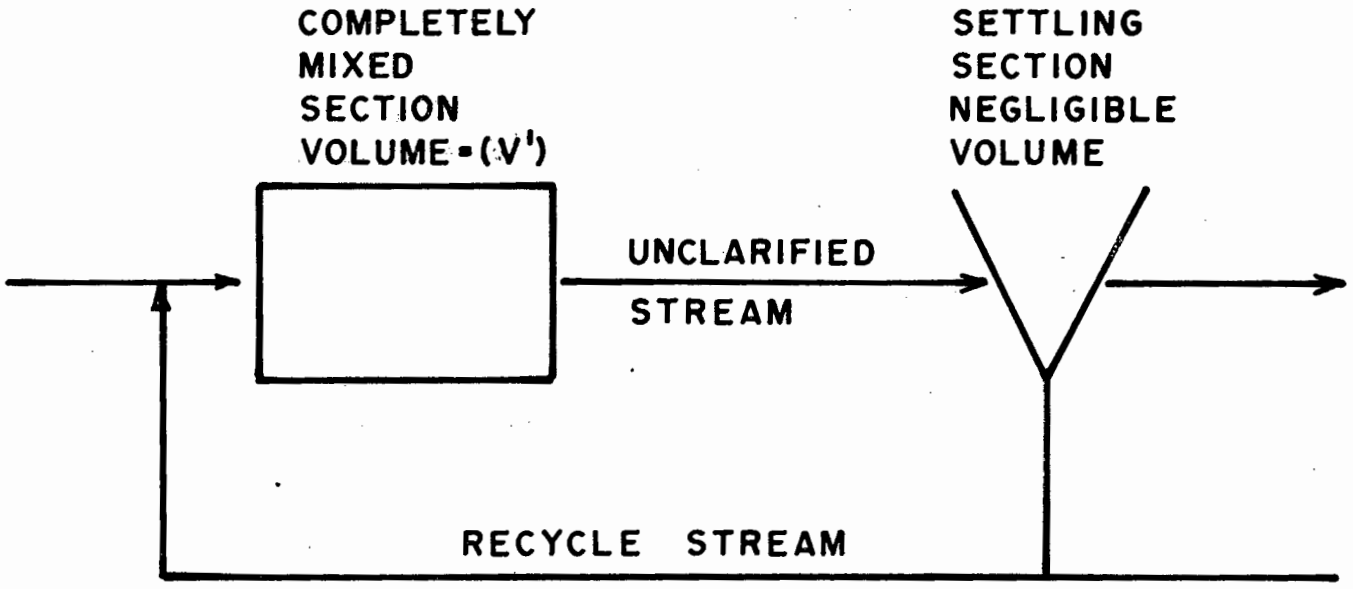

Figure 5.1. Digester with recycle.

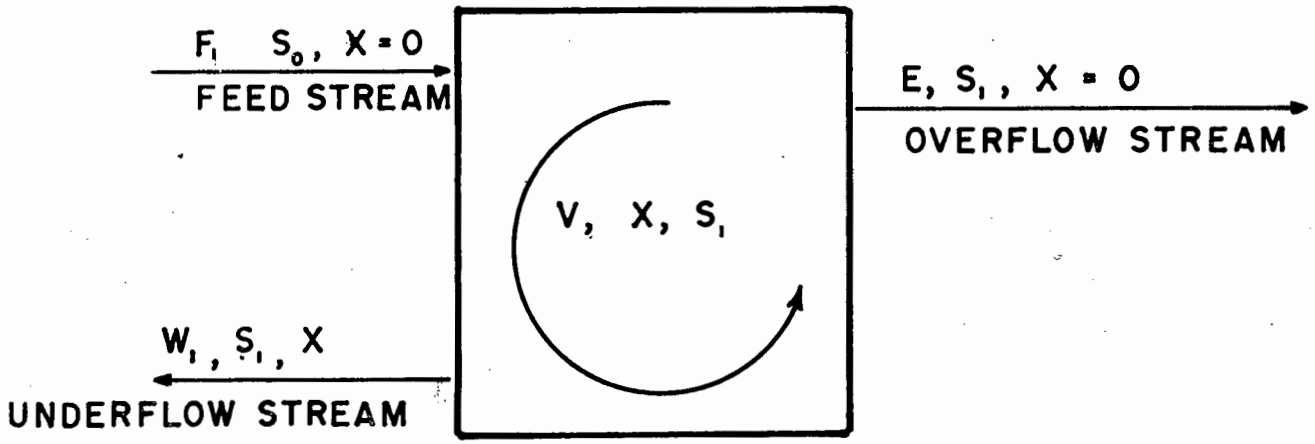

Figure 5.2. Single Unit.

Apart from the general assumptions, described in section 5.2, the following conditions were assumed to apply :

(1) a constant portion of the organisms were active under all operating conditions;

(2) the mass of organisms produced per mass of substrate utilized was constant, i.e. the yield was assumed constant;

(3) only the degradable portion of the yeast waste was assumed as the growth limiting substrate. This assumption was necessary, due to the high non-biodegradable fraction of the yeast waste.

A sludge age, (solids retention time), θ_c, is defined as follows (M13) :

$$\theta_c = \frac{\text{mass of cells in the tank}}{\text{mass of cells wasted per day}}$$

This can be expressed in terms of V, W and X as :

$$\theta_c = \frac{VX}{WX} = \frac{V}{W} \quad \ldots\ldots\ldots\ldots\ldots\ldots \quad (5.5)$$

The sludge age is controlled by the amount of sludge wasted per day.

The hydraulic retention time, θ, is defined as the volumetric feed rate divided by the volume of the digester, i.e.

$$\theta = \frac{V}{F} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (5.6)$$

From equations (5.5) and (5.6) it can be seen that the hydraulic and solids retention times are related by the following expression :

$$\frac{\theta_c}{\theta} = \frac{F}{W} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (5.7)$$

A material balance on the sludge concentration over the digester can be written as follows :

Overall rate of sludge concentration in digester	=	Rate of input of sludge in feed	-	Rate of removal of sludge in overflow	-	Rate of removal of sludge in underflow	+	Rate of increase of sludge due to growth

- Rate of decrease of sludge due to endogenous respiration etc.

$$V\left(\frac{dX}{dt}\right)_{overall} = 0 - 0 - WX + \mu VX - k_d VX \quad \ldots\ldots \quad (5.8)$$

Applying steady state conditions:

$$\mu V = W + k_d V$$

Substituting Equation (5.5)

$$\mu = \frac{1}{\theta_c} + k_d \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (5.9)$$

This shows that at steady state the sludge growth rate is equal to the sum of sludge removal and decay rate.

From Equation (5.9)

$$\frac{1}{\theta_c} = \mu - k_d = \frac{\hat{\mu} S_1}{K_s + S_1} - k_d \quad \ldots\ldots\ldots\ldots \quad (5.10)$$

From the definition of the yield constant

$$\left(\frac{dS}{dt}\right)_{utilized} = -\frac{1}{Y}\left(\frac{dX}{dt}\right)_{growth} = -\frac{1}{Y}\mu X \quad \ldots \quad (5.4)$$

At steady state:

$$V\left(\frac{dS}{dt}\right)_{utilized} = F\,So - FS_1$$

$$\text{Therefore } \left(\frac{dS}{dt}\right)_{utilized} = \frac{So - S_1}{\theta} \quad \ldots\ldots\ldots \quad (5.11)$$

Substituting into Equation (5.4) and re-arranging :

$$X = \frac{(So-S_1)Y}{\mu\ \theta} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (5.12)$$

and then substituting for μ from Equation (5.9)

$$X = \frac{(So-S_1)Y}{1+k_d\ \theta_c} \cdot \frac{\theta_c}{\theta} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (5.13)$$

This shows that the sludge concentration is a function of the sludge age, the hydraulic retention time and the change in waste strength.

An expression for the effluent strength, S_1, is obtained by re-arranging the Monod expression, Equation (5.2)

$$S_1 = \frac{K_s\ \mu}{\hat{\mu} - \mu} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (5.14)$$

Substituting for μ from Equation (5.9)

$$S_1 = \frac{K_s(1+k_d\ \theta_c)}{\theta_c(\hat{\mu} - k_d)-1} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (5.15)$$

This shows that the waste strength leaving the digester is a function of the sludge age only.

5.4 SUMMARY OF EQUATIONS.

The equations derived in this section, and summarized below, can be seen to be the same as those derived by Lawrence and McCarty (L3), for the aerobic activated sludge and the anaerobic contact processes.

$$\mu = \frac{1}{\theta_c} + k_d \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (5.9)$$

$$\frac{1}{\theta_c} = \frac{\hat{\mu}\ S_1}{K_s + S_1} - k_d \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (5.10)$$

$$X = \frac{Y(So-S_1)}{1+k_d\ \theta_c} \cdot \frac{\theta_c}{\theta} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (5.13)$$

$$S_1 = \frac{K_s(1+k_d\,\theta_c)}{\theta_c(\hat{\mu} - k_d)-1} \quad \ldots\ldots\ldots\ldots\ldots\ldots \quad (5.15)$$

5.5 LIMITATIONS.

The major limitations related to the above model, as used in this investigation, can be summarized as follows :

(1) A possible variation in the microbial character present at different steady state periods;

(2) Consequently, the total volatile suspended solids may not be a reliable measure of the active portion of the sludge present in the reactor;

(3) The presence of toxic materials in the waste which would effect the choice of degradable Chemical Oxygen Demand, as the rate-limiting substrate present in the system;

(4) A similar effect as (3) above on the system due to changes in the digester environment, viz. pH, alkalinity, volatile acid concentration.

5.6 APPLICATION OF KINETICS TO ANAEROBIC DIGESTION.

Equation (5.10) predicts that under ideal conditions, the effluent concentration is independent of influent substrate concentration. However, the presence of inhibitory substances in certain wastes can set an upper limit to the concentration of these wastes applied to the system.

Equation (5.10) further indicates that effluent substrate concentration is solely dependent on the sludge age. Consequently, a reduction in the hydraulic retention time will not effect the effluent quality provided sludge age is maintained at the same value. This has made possible the treatment of more dilute wastes previously uneconomical to treat by conventional anaerobic processes. Steffan and Bedker (S14), for example, have been able to operate a field digester, treating meat packing wastes, at hydraulic residence times of 0,5 to 1,0 days as opposed to the usual 20 - 30 day period frequently used in conventional digesters.

5.7 EVALUATION OF KINETIC PARAMETERS.

Because of the difficulties encountered in the measurement of active microbial populations and obtaining a reliable identification and determination of the growth limiting substrate, the evaluation of kinetic parameters has been confined largely to laboratory processes using pure substrates, and only the methane producing stage considered. Pretorius (P6) has summarized most of the data available in the literature.

However, studies on the kinetics of the overall process using mixed cultures and more complex substrates have been undertaken. The data from these studies has been summarized by Pretorius (P6). All the above values have been tabulated in Tables 5.3 and 5.4.

A number of readings were obtained during the present study in an attempt to evaluate the kinetic parameters, viz. Y, k_d, Ks and $\hat{\mu}$, for the anaerobic treatment of yeast waste.

The investigation was based on degradable Chemical Oxygen Demand as the growth limiting substrate and volatile suspended solids as a measure of the concentration of

micro-organisms in the system. The results of all the steady state values obtained during the investigation are shown in Table 5.1.

(a) Evaluation of Y and kd.

Equation (5.13) can be re-arranged to give the following expression :

$$\frac{So-S_1}{X_1\,\theta} = \frac{1}{\theta_c}\cdot\frac{1}{Y} + \frac{k_d}{Y} \qquad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (5.16)$$

A plot of $\frac{So-S_1}{X_1\,\theta}$ against $\frac{1}{\theta_c}$

yields a straight line, as shown in Figure 5.3, with slope equal to the reciprocal of the yield and intercept equal to the organism decay constant divided by the yield.

(b) Evaluation of $\hat{\mu}$ and Ks.

Equation (5.2) can be re-arranged to the following form enabling the evaluation of the Monod constants :

$$\frac{1}{\mu} = \frac{1}{S_1}\left(\frac{K_s}{\hat{\mu}}\right) + \frac{1}{\hat{\mu}} \qquad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (5.17)$$

where $\mu = \frac{1}{\theta_c} + k_d$

Consequently, a plot of the reciprocal of $(\frac{1}{\theta_c} + k_d)$ against the reciprocal of the substrate concentration, yields a line of slope = $\frac{K_s}{\hat{\mu}}$ and intercept = $\frac{1}{\hat{\mu}}$, as shown in Figure 5.4.

5.8 RESULTS.

Reported values of the kinetic parameters related to anaerobic digestion are presented in Tables 5.3 and 5.4. Table 5.3 summarizes the values of the parameters obtained

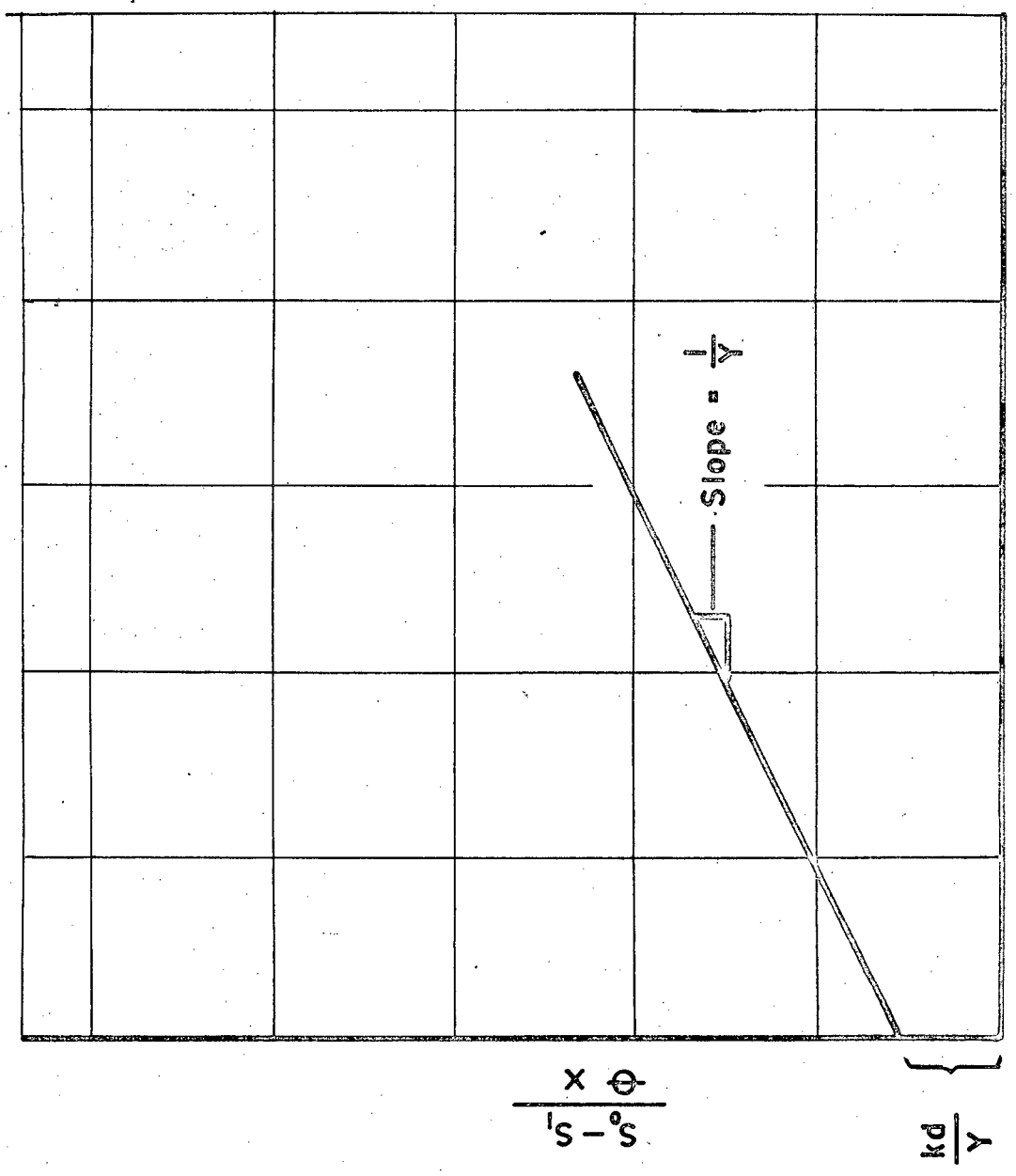

Figure 5.3. Evaluation of yield coefficient and decay constant.

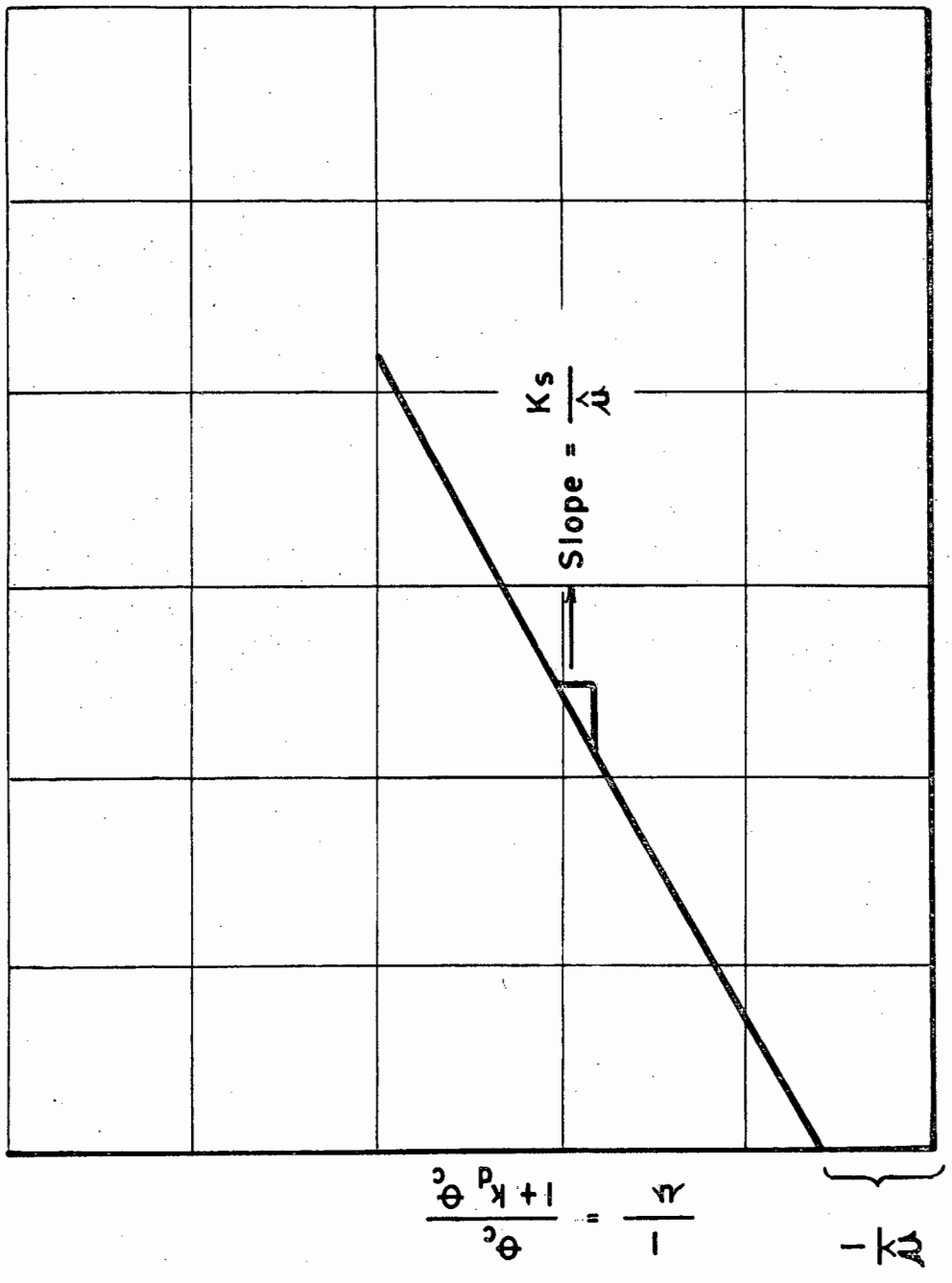

Figure 5.4. Evaluation of saturation constant and maximum specific growth rate.

Table 5.1. Summary of steady state values obtained during this investigation

θ (days)	Gas Production (ℓ/day/ℓ)	Volatile Acids (mg/ℓ CH_3COOH)	CO_2 Content (%)	S_o (mg/ℓCOD)	S_1 (mg/ℓCOD)	S_o Degradable (mg/ℓ COD)	S_1 (mg/ℓCOD)	X (Total) (mg/ℓ)	X (Volatile) (mg/ℓ)	W (ℓ/day)
1,70	0,985	402	26,3	5 574	2 210	4 035	671	13 500	11 500	0,059
	1,063	405	27,6	6 207	2 174	4 767	734	12 800	10 800	0,063
	1,250	277	28,3	6 943	2 598	5 132	787	13 000	11 200	0 072
	2 006	450	29,1	10 050	3 540	7 688	1 178	13 000	11 100	0,133
	2,830	1 236	31,2	16 100	6 092	12 300	2 292	13 500	11 500	0,212
	3,680	1 250	32,0	19 302	6 302	14 300	2 300	13 400	11 300	0,293
1,45	0,765	117	25,3	4 252	1 814	2 922	484	12 000	10 200	0,041
	1,140	408	27,1	6 177	2 152	4 725	700	13 000	11 100	0,105
	1,361	296	27,5	8 330	2 758	6 372	800	13 100	11 400	0,138
	2,132	416	29,9	5 230	1 960	3 786	516	12 000	10 000	0,079
	2,645	731	32,9	10 287	3 754	7 940	1 407	11 500	9 500	0,180
	2 960	1 400	32,6	13 600	5 150	10 550	2 100	11 500	9 600	0,265
3,53	0,549	162	25,6	6 491	2 758	4 136	403	14 100	11 800	0,028
	0,685	284	27,5	7 563	2 663	5 360	460	14 500	12 200	0,034
	0,936	280	28,6	10 390	3 935	7 067	612	14 000	12 100	0,053
	1,640	428	30,4	13 319	3 956	10 189	1 125	14 200	11 620	0,116
	1,890	461	32,5	16 420	4 735	12 810	1 125	13 600	11 200	0,138
	1,938	1 160	35,7	19 320	6 943	14 780	2 403	13 000	11 000	0,163

TABLE 5.2. Summary of computed values required for the evaluation of kinetic constants.

θ	$\frac{So - S_1}{X\ \theta}$	$\frac{1}{\theta_c}$	$\frac{\theta_c}{1 + k_d\ \theta_c} = \frac{1}{\mu}$	$\frac{1}{S_1}$
1,70	0,173	0,0079	79	0,00149
	0,221	0,0084	75	0,00136
	0,228	0,0096	69	0,00127
	0,347	0,0177	44	0,00085
	0,511	0,0283	30	0,00042
	0,625	0,0391	23	0,00435
1,46	0,164	0,0055	97	0,00207
	0,250	0,0140	53	0,00143
	0,338	0,0184	43	0,00125
	0,225	0,0105	65	0,00194
	0,474	0,0240	35	0,00071
	0,604	0,0354	25	0,00048
3.53	0,090	0,0037	117	0,00248
	0,114	0,0045	107	0,00217
	0,151	0,0071	84	0,00163
	0,228	0,0155	49	0,00121
	0,295	0,0184	43	0,00089
	0,318	0,0217	38	0,00042

Table 5.3 Kinetic Parameters Representative of Various Substrates Comprising the Methane-Fermentation Step.

Substrate	Yield Coefficient (mg/mg) Y	Decay Rate (day^{-1}) k_d	Saturation Constant (mg/ℓ) K_s	Maximum Specific Growth Rate (day^{-1}) μ	Temperature ^oC	Reference
Acetic Acid	0,04	0,015	2130	0,144		O'Rourke (O2)
Propionic Acid	0,04	0,015	2850	—		O'Rourke (O2)
Stearic and Palmitic Acids	0,04	0,015	4620	0,154		O'Rourke (O2)
Mixed Acids	0,04	0,015	10600	0,154		O'Rourke (O2)
Acetic Acid	0,054	0,011	869	0,25	20	Lawrence et.al. (L2)
Propionic Acid	0,041	0,040	613	0,40		Lawrence et.al. (L2)
Stearic and Palmitic Acids	0,04	0,015	3720	0,186		O'Rourke (O2)
Mixed Acids	0,04	0,015	5790	0,186		O'Rourke (O2)
Acetic Acid	0,058	0,037	333	0,278	25	Lawrence et.al. (L2)
Acetic Acid	0,044	0,015	154	0,356	35	Lawrence et.al. (L2)
Propionic Acid	0,054	0,010	32			Lawrence et.al. (L2)
Butyric Acid	0,025	0,027	5	0,39		Lawrence et.al. (L2)
Stearic and Palmitic Acids	0,040	0,015	2000	0,267		O'Rourke (O2)
Mixed Acids	0,040	0,015	2235	0,267		O'Rourke (O2)

Table 5.4 Anaerobic Biological Process Parameters

bstrate	Yield Coefficient (mg/mg) Y	Decay Rate (day^{-1}) k_d	Saturation Constant (mg/ℓ) K_s	Maximum Specific Growth Rate (day^{-1}) μ	Temperature °C	Reference
st Waste	0,073	0,005	3 100	0,084	30	Present study
trose, tryptone, f extract	0,18	0,025	5 700	0,067	38	Stewart (S15)
trose, tryptone, f extract	0,114	0,03	6 700	0,09	38	Agardy et.al (A1)
trose, tryptone, f extract	0,104	0,02	13 000	0,11	38	Agardy et.al (A1)
cose and Starch no and fatty ds rient broth	0,46 0,054 0,076	0,088 0,038 0,014	— — —	— — —	—	Speece et.al. (S9)
trose, tryptone f extract d Production hane Production	0,54 0,14	0,87 0,02	— —	— —	38	Andrews and Pearson (A5)
thetic Milk te	0,37	0,07	24,3	0,14	20-25	Gates et.al (G3)
trose, peptone, f extract	0,14	0,10	4200	0,186	—	Andrews, Cole & Pearson (A4)
trose, tryptone, f extract	—	—	2760	—	—	Almgren et.al. as reported by Pretorius (P6)

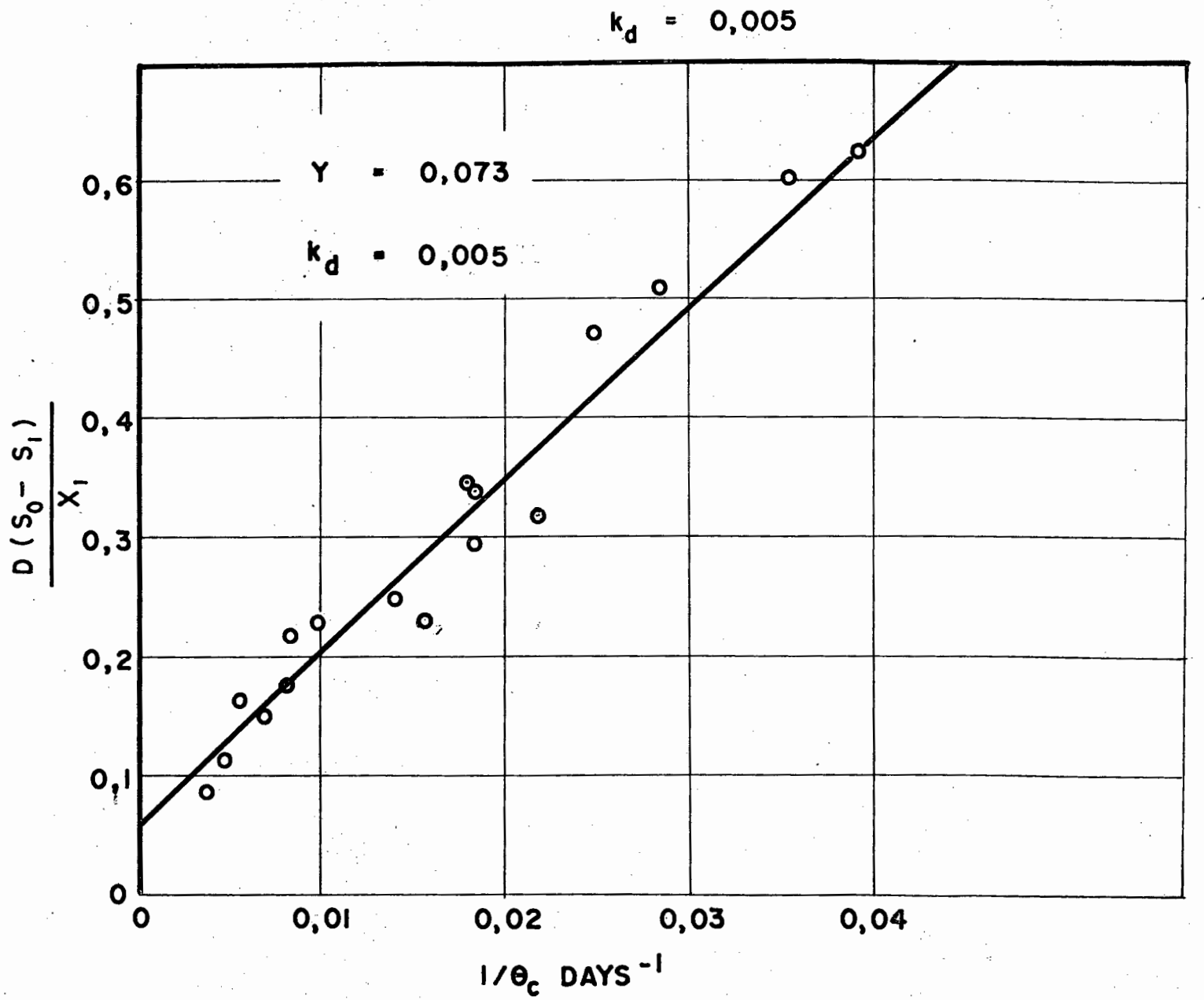

Figure 5.5. Evaluation of yield coefficient and decay constant from the

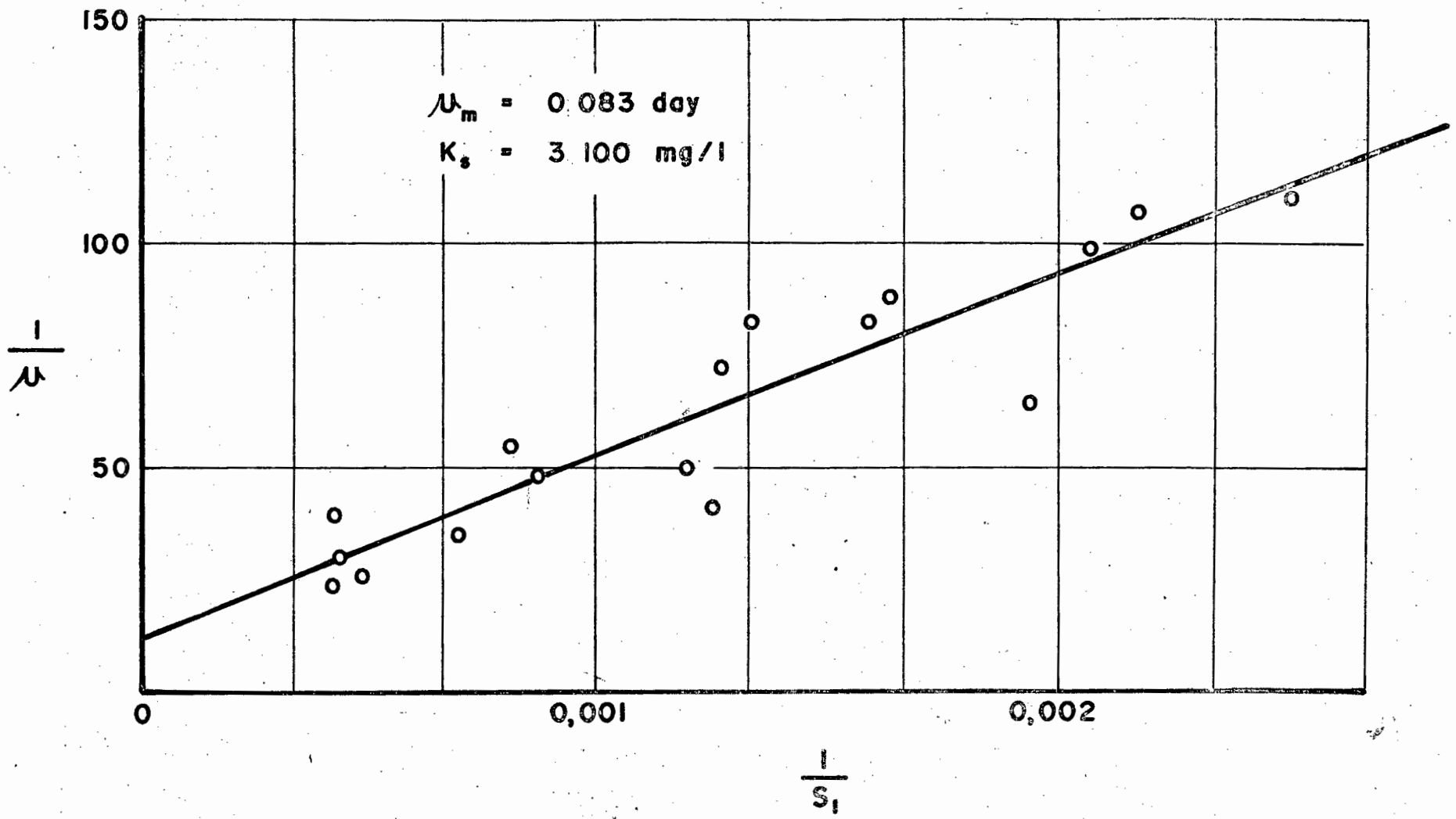

Figure 5.6. Determination of K_S and $\hat{\mu}$ from the results obtained in this

from studies representing the methane fermentation step, using both pure and mixed substrates. The parameters obtained in various mixed culture studies, representing the overall processes, are shown in Table 5.4. The steady state values obtained during this investigation are shown in Table 5.1. The above data was used to compute the required values of $\frac{So-S_1}{X_1\,\theta}$, $\frac{1}{\theta_c}$, $\frac{1}{\theta_c} + k_d$, and $\frac{1}{S_1}$, respectively.

These values are presented in Table 5.2 and were used in the construction of Figures 5.5 and 5.6 from which the parameters Y, kd, $\hat{\mu}$ and K_s, were obtained by the method shown in section 5.5.

The values of the parameters obtained from these plots are depicted on Figures 5.5 and 5.6, and are summarized below :

Parameter.	Value.	Units.
Y	0,073	mg VS/mg COD
k_d	0,005	day^{-1}
$\hat{\mu}$	0,083	day^{-1}
K_s	3 100	mg/l

By comparing the experimental results obtained in this study with those shown in Tables 5.3 and 5.4, it can be seen that :

(a) the value of the maximum specific growth rate ($\hat{\mu}$), viz. 0,083, obtained from the present study, lies within the range reported in the literature;

(b) the saturation constant (K_s) similarly agrees very closely with those reported in other studies with different substrates;

(c) the yield constant (Y) of 0,073 is within the range of experimental values reported, but is lower than many of these values;

(d) the value of the decay rate constant (k_d) shows considerable discrepancy with those reported in the literature. The value of 0,005 day^{-1} is between two or three times lower than any other reported in Tables 5.3 and 5.4.

To facilitate comparison of the kinetic parameters obtained in this research, with typical values reported in the literature, Table 5.5 has been drawn up as a representative range of the data contained in Tables 5.3 and 5.4.

TABLE 5.5. Comparison of published kinetic values related to anaerobic digestion.

Parameter	Units	Present Study	Table 5.3.	Table 5.4
Y	$\frac{mg\ VS}{mg\ COD}$	0,073	0,025 - 0,058	0,054 - 0,18
k_d	days $^{-1}$	0,005	0,010 - 0,040	0,014 - 0,088
$\hat{\mu}$	days $^{-1}$	0,083	0,144 - 0,40	0,067 - 0,186
K_s	mg/l	3 100	2 000 - 5 790	2 760 - 6 700

The kinetic parameters obtained from the complex yeast waste by the experimental techniques employed in this study, confirm the validity of the assumptions made in section 5.3 and indicate that the experimental technique employed, yielded satisfactory results.

It should be noted that the values of the parameters obtained in this study agree more closely with those obtained

by workers with systems that involve the overall process. This is understandable as the elimination of the acid forming step from studies on the kinetics of the anaerobic process must effect at least the yield and decay rate constants. The effect on K_s and $\hat{\mu}$ is expected to be minimal.

CHAPTER 6

T O X I C I T Y

6.1 INTRODUCTION.

As suggested in section 3.4, the failure of all the reactor-clarifier systems at a similar loading concentration indicated that one or more substances were present in concentrations likely to be inhibitory to the system.

An analysis of the various anions and cations present in the yeast waste, was made to determine whether any of these ions were present in inhibitory or even toxic concentrations. The results are shown in Table 6.1.

TABLE 6.1. Analysis of the anions and cations present in the digester influent and effluent streams.

Ion	Digester Feed ppm	Digester Effluent ppm	Inhibitory Conc. ppm	Toxic Conc. ppm	Reference
Cu	1	0,5	-	10	Barth *et al.* (B2)
Zn	1	0,5	-	10	Barth *et al.* (B2)
Ni	1	0,5	-	40	Barth *et al.* (B2)
Fe	10-15	1-2	-	-	-
Na^+	100-200	250	3 500-5 500	8 000	McCarty (M10)
K^+	2 000-3 000	2 000-3 000	2 500-4 500	12 000	McCarty (M10)
Ca^{++}	600-1 000	-	2 500-4 500	8 000	McCarty (M11)
Mg^{++}	250-500	-	1 000-1 500	3 000	McCarty (M11)
$S^=$	-	50-150	100-200	200	Lawrence *et al.* (L1)
NH_3-N	-	200	1 500-3 000	3 000	McCarty (M10)
Cl^-		6 500	2 000-8 000	10 000	Ludzack & Noran (L6)
Cl^-				5 000	Niles & Frook (N2)

An attempt was made to determine the effect of various cations and anions on the system. The experimental methods used are described in section 6.3 and the conclusions discussed in section 6.4.

It should be emphasized that this part of the study had the following limitations :

(a) a variation in the concentration of the respective ions for different batches of the yeast waste;

(b) the presence of a variety of synergistic and antagonistic combinations, defined in section 2.4.7(c), in the yeast waste. The potential inhibition of a particular ion could therefore vary with different concentrations of other ions present in the system.

Consequently the results obtained in this portion of the study are by no means conclusive, and are merely indications as to the possible cause of the toxicity experienced in the digesters.

In order to obtain conclusive evidence of the inhibitory effect of various anions and cations, it is necessary to conduct the investigation with a synthetic feed in which concentrations of the ions concerned can be maintained at known levels or eliminated if required.

6.2 POSSIBLE SOURCES OF INHIBITION.

The concentrations of copper, zinc and nickel in the digester are minimal as shown in Table 6.1. The low concentrations of these heavy metals entering with the feed were precipitated out as insoluble metal sulphide salts. Clearly no heavy metal toxicity was suspected during treatment of the yeast waste.

The following alkali and alkaline earth metals were analyzed: sodium, potassium, calcium and magnesium. By comparing the concentrations of these metals with those reported to be inhibitory by McCarty (M10), it can be seen from

Table 6.1 that only potassium was found to be present in concentrations likely to cause inhibition.

Ammonia nitrogen was not found to be present in inhibitory concentrations at the loading rates used in this study. The concentration of NH_3-N did not exceed 200 mg/l at any stage of this study, which is considerably less than the inhibitory range of 1 500 - 3 000 mg/l reported by McCarty (M10), and discussed in section 2.4.7(e).

Sulphates were present in comparatively large concentrations viz. 2 800 - 4 500 mg/l, in the yeast waste, as shown in Table 1.1, Chapter 1. Consequently a typical concentration of sulphates in the diluted feed to the digesters was 1 000 mg/l as shown in Table 4.1, section 4. From Table 4.2, section 4, it can be seen that under the operating conditions used in this study, 40-60 per cent. of the sulphates entering the digesters were reduced to sulphides.

If all the sulphides produced during digestion remained in solution, the concentration of sulphides would be 2 - 3 times the toxic threshold concentration of 200 mg/l, as reported by Lawrence et al. (L1). However as can be seen from Table 6.1, a major portion of the sulphide content left the system as hydrogen sulphide gas or was precipitated out of the liquid as insoluble metal salts.

As seen from Table 6.1, the concentrations of copper, zinc, nickel and iron in the influent declined during digestion. Iron in particular appeared to be the major metal utilized in the precipitation of sulphides from the system.

Studies on the same waste, undertaken by Stander et al. (S12) indicated that sulphides had not inhibited the process. Nevertheless, the possibility of sulphide inhibition was not discounted in the present study. Sulphide concentrations in the digesters used in the present study were found to vary from 50 to 150 mg/l, as shown in Table 6.1. This is below the

toxicity threshold value of 200 mg/l for sulphides, as quoted by Lawrence et al. (L1). However the same writers (L1) have shown that sulphide concentrations above 100 mg/l can inhibit the methane bacteria. Consequently, as seen from Table 6.1, sulphides were suspected of exerting an inhibitory effect on the methane bacteria, when present in higher concentrations.

Chlorides were found to be present in relatively high concentrations, viz. 6 500 mg/l as NaCl, at higher loading rates as shown in Table 6.1. These values are within the range of 2 000 - 8 000 mg/l reported as being inhibitory by Ludzack and Noran (L6) and greater than the toxic threshold value reported by Niles and Frook (N2).

Consequently, from the data summarized in Table 6.1, the ions suspected of exerting an inhibitory effect on the process, at higher loadings, were potassium, sulphide and chloride.

6.3 EXPERIMENTAL PROCEDURE AND RESULTS.

The following methods were used in an attempt to isolate the effect on the system of potassium, sulphide and chloride, respectively, at different concentration levels.

Sulphides.

Two batch digesters were used in an attempt to evaluate whether sulphides were limiting the load rates tolerated in the present study.

The first digester was operated normally, with no additions to the daily feed. In the second digester sufficient $FeCl_3$ was added to the feed to maintain an overall Fe concentration of 30-35 ppm more than the feed entering the first digester. The concentrations of Fe in the influent and effluent are clearly shown in Figure 6.1.

As shown in Figure 6.1, the effluents of both digesters contained approximately the same concentration of iron.

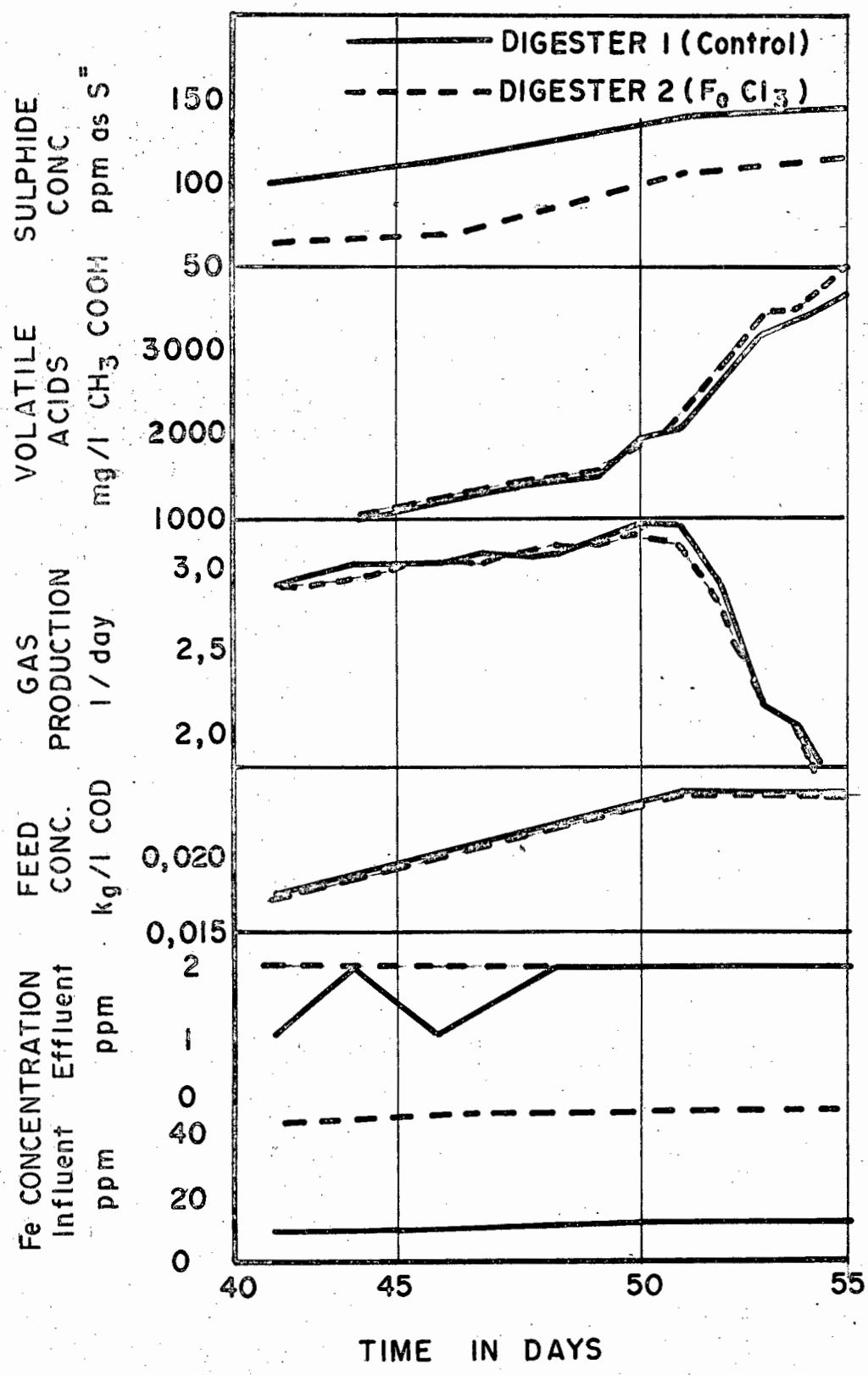

Figure 6.1. Influence of sulphides on digester performance.

Consequently as the feed applied to the second digester contained 30 ppm more Fe than in the first digester, it can be seen that 30 ppm more iron was precipitating out, as iron sulphide, in the second digester.

Both digesters were loaded to failure with constant monitoring of the relevant parameters, viz. gas production, volatile acids concentration, Fe concentrations (in the influent and effluent) and total sulphide concentrations. The results are shown for both digesters in Figure 6.1.

If sulphides were the cause of digester failure, digester 1 would be expected to fail at a lower loading concentration than digester 2, due to its higher sulphide concentration.

As shown in Figure 6.1, both digesters failed at the same feed concentration level, viz. 0,023 kg/l. This indicates that sulphides were not the cause of total digester failure.

Potassium.

As seen from Table 6.1, the maximum concentration of potassium obtained during digestion was 3 000 ppm.

In order to evaluate the effect of increasing levels of potassium concentration on the system, a further two digesters were operated. To avoid confusion they will be referred to as digester 3 and digester 4.

Digester 3 was used as a control and was loaded at a constant rate of 5,33 kgCOD/day/m^3. This corresponded to a feed concentration of 0,016 kgCOD/l.

As seen from Table 6.1, the influent and effluent streams have the same potassium concentration levels, i.e. there is no change in the potassium content during digestion.

Digester 4 was loaded at the same feed concentration as digester 3, viz. 0,016 kgCOD/l. However the potassium

concentration in the feed (and consequently the effluent as well), was increased in increments of 250 ppm/day. These increments were continued until a potassium level of 4 500 mg/l had been reached. K_2CO_3 was used to increase the potassium concentration during this part of the study. The relevant parameters for both digesters are plotted in Figure 6.2.

what about pH?

As can be seen in Figure 6.2, the performance of digester 4 paralleled digester 3 until a potassium concentration of approximately 3 000 ppm was reached. Increases in the potassium concentration above this value resulted in a decrease in the gas production rate at digester 4.

The decrease in gas production rate was not severe, as shown in Figure 6.2. This indicated that although exerting some inhibitory effect on the anaerobic system at concentrations above 3 000 ppm, potassium could not be held responsible for total digester failure.

Chlorides.

A similar method as that used to determine the inhibitory effect of potassium, was used to investigate whether chlorides had any detrimental effect on the digesters at concentrations experienced during this study, viz. a maximum of 6 500 mg/l.

Digester 3 was used as a control and a further digester 5 operated by incrementing the chloride concentration in the feed by 250 mg/l per day. NaCl was used for this purpose.

As seen from Figure 6.2, chloride concentrations of nearly 10 000 mg/l as NaCl were tolerated by digester 5 with no reduction in efficiency. The performance of digester 5 was found to be in no way inferior to the control, digester 3, for a range of chloride concentrations considerably above the maximum experienced by the digesters throughout the investigation.

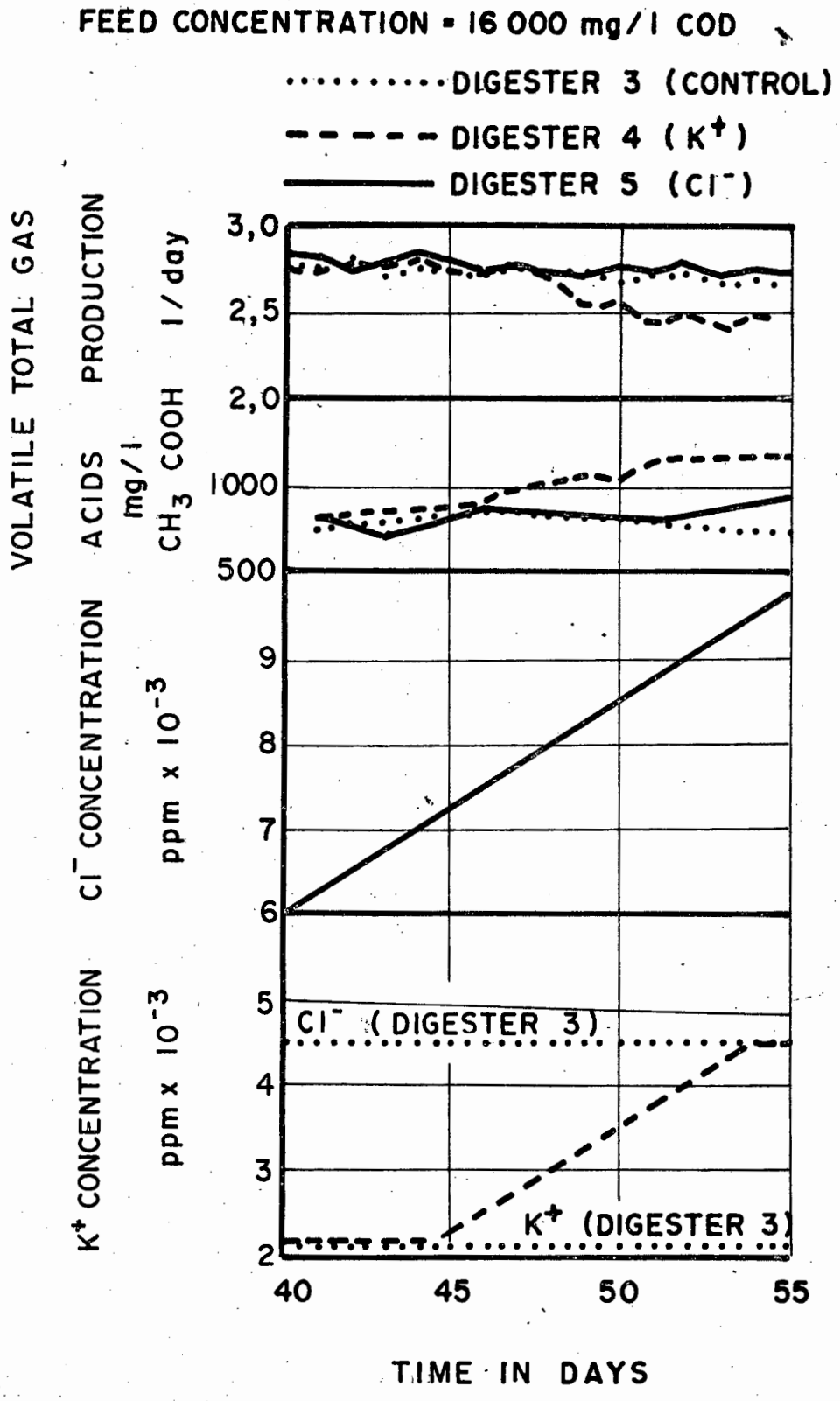

Figure 6.2. Influence of potassium and chloride on digester performance.

6.4 CONCLUSIONS.

From the relevant data, shown in Figures 6.1 and 6.2, and discussed in section 6.3, it was concluded that neither potassium, chlorides nor sulphides were solely responsible for total digester failure.

Potassium was indicated to exert an inhibitory influence on the system at high loadings.

The results of this part of the investigation were felt to be inconclusive. The reasons for this finding are suspected to be twofold :

(a) a significant influence on the inhibitory effect of a particular ion by the other ions present in the digesters;

(b) a change in the concentrations of various ions for different batches of the yeast waste.

Consequently it is recommended that further research on inhibition should be undertaken, particularly in evaluating the antagonistic and synergistic effects of various ions upon one another.

CHAPTER 7

CONCLUSIONS AND RECOMMENDATIONS

As mentioned in Chapter 1, this investigation was carried out with the following objectives in mind :

(1) to establish the feasibility of treating yeast waste by a conventional anaerobic method;

(2) the construction of a laboratory scale digester, of the reactor-clarifier type, and the establishment of a reliable operating technique;

(3) to evaluate the significance of the various parameters used to monitor digester performance, particularly in predicting periods of process imbalance;

(4) the estimation of the kinetic parameters relating to the anaerobic treatment of yeast waste and the comparison of these values to others reported in the literature;

(5) the identification of any inhibitory substances present in the process.

From the results described in Chapters 4, 5 and 6, the following conclusions were made :

(1) Approximately 75 per cent. of the yeast waste was readily susceptible to anaerobic digestion. The remaining 25 per cent. of the yeast waste was not degradable.

(2) The reactor-clarifier type of digester used in this investigation yielded a clear effluent of satisfactory quality. A novel experimental technique permitted a more rapid approach to steady state conditions. Further research on a pure synthetic waste is recommended to confirm the validity of this method.

This is not a good method, since the population composition changes

(3) Total gas production was found to be the most sensitive indicator to changes in the process environment.

(4) The values of the kinetic parameters evaluated in this study were within the range reported by other workers in the literature, with the exception of the cell decay rate constant which was considerably less than any other reported value.

(5) Attempts to relate the presence of inhibitory substances to a limiting feed concentration were unsuccessful. Further research is recommended on synthetic substrates to evaluate the antagonistic and synergistic effects of the anions and cations commonly present in industrial waste systems.

B I B L I O G R A P H Y

A1 Agardy, F.J., Cole, R.D. and Pearson, E.A., "Kinetic and Activity Parameters of Anaerobic Fermentation Systems", Sanitary Engineering Research Laboratory Report No. 63-2, University of California, Berkeley, (1963).

A2 Aiba, S, Humphrey, A.E. and Millis, N.F., "Biochemical Engineering", Academic Press, New York, (1965).

A3 Albertson, O.E., "Ammonia Nitrogen and the Anaerobic Environment", Journal Water Pollution Control Federation, 33, No. 9, pp. 978 - 995, (Sept. 1961).

A4 Andrews, J.F., Cole, R.D. and Pearson, E.A., "Kinetics and Characteristics of Multi-stage Methane Fermentation", Sanitary Engineering Research Laboratory Report No. 64-11, University of California, Berkely, (1964).

A5 Andrews, J.F. and Pearson, E.A., "Kinetics and Characteristics of Volatile Acid Production in Anaerobic Fermentation Processes", International Journal Air Water Pollution, 9, pp. 439 - 461, (1965).

A6 Andrews, J.F., "Dynamic Model of the Anaerobic Digestion Process", Proceedings of the American Society of Civil Engineers, Journal of the Sanitary Engineers Division, SA 1, pp. 95 - 116, (Feb. 1969).

A7 Andrews, J.F., "Kinetic Models of Biological Waste Treatment Processes", Biotechnology and Bioengineering, Symposium No. 2, pp. 5 - 53, (1971).

B1 Banta, A.P. and Pomeroy, R., "Bicarbonate Equilibrium in Digesting Sludge", Sewage Works Journal, 6, No. 2, pp. 234 - 245, (1934).

B2 Barth, E.F., Ettinger, M.B., Salotto, B.V. and McDermott, G.N., Summary Report on the Effects of Heavy Metals on the Biological Treatment Process, Journal Water Pollution Control Federation, 37, No. 1, pp. 86 - 96, (1965).

B3 Bhaskaran, T.R. and Chakarbarty, R.N., "Pilot Plant for Treatment of Cane-Sugar Waste", Journal Water Pollution Control Federation, 38, No. 7, pp. 1160 - 1169, (1966).

B4 Buswell, A.M., "Anaerobic Fermentations", Bulletin No. 32, Illinois State Water Survey, Urbana, (1939).

B5 Buswell, A.M., Pagano and Sollo, "Effect of Chloride Ions on Sludge Digestion", Industrial and Engineering Chem., 41, p. 596, (1949).

C1 Casswell, E.A. and Sawyer, C.N., "A Method of Starting High-Rate Digesters", Sewage and Industrial Wastes, 31, No. 2, p. 123, (1939).

C2 Cillie, G.G., Henzen, M.R., Stander, G.J. and Baillie, R.D., "Anaerobic Digestion - IV. The Application of the Process in Waste Purification", Water Research, 3, pp. 623 - 643, (1969).

C3 Clough, G.F.G., "Chemical Engineering Aspects of Waste Water Treatment", The Chemical Engineer, pp. CE14 - 18, (Jan./Feb. 1970).

C4 Cramer, F.J., "Curing a Sick Digester", Journal Water Pollution Control Federation, 37, pp. 1317 - 1318, (1965).

C5 Coughanower, D.R. and Koppel, L.B., "Process Systems Analysis and Control", McGraw-Hill Book Company, New York, (1965).

D1 DiLallo, R. and Albertson, O.E., "Volatile Acids by Direct Titration", Journal Water Pollution Control Federation, 33, No. 4, pp. 356 - 365, (1961).

E1 Estrada, A.A., "Design and Cost Considerations in High-Rate Sludge Digestion", Journal of the Sanitary Engineering Division, American Society of Civil Engineers, 86, No. SA 3, pp. 111 - 127, (1960).

F1 Fair, G.M. and Moore, E.W., "Time and Rates of Sludge Digestion and their Variation with Temperature", Sewage Works Journal, 6, pp. 3 - 13, (1934).

F2 Fencl, Z., "A Uniform System of Basic Symbols for Continuous Cultivation of Microorganisms", Folio Microbial 8, 192.

G1 Garber, W.F., "Plant Scale Studies of Thermophilic Digestion at Los Angeles", Sewage and Industrial Wastes, 26, p. 1202 - 1216, (1954).

G2 Garrison, N.E., Parkhurst, J.D. and Nagel, C.A., "Gas Recirculation - Natural, Artificial", Water Works and Wastes Engineering, 1, pp. 58 - 63, (Feb. 1964).

G3 Gates, W.E., Smith, J.H., Lin, S-D. and Ris III, C.H., "A Rational Model for the Anaerobic Contact Process", Journal Water Pollution Control Federation, 39, No. 12, pp. 1951 - 1970, (1967).

G4 Gollieke, C.G., "Temperature Effects on Anaerobic Digestion of Raw Sewage Sludge", Sewage and Industrial Wastes, 30, pp. 1225 - 1232, (1958).

H1 Hansford, G.S. and Ledermann, W., Private Communication.

H2 Hemens, J., et al., "Full-Scale Anaerobic Digestion of Effluents from the Production of Maize Starch", Water and Waste Treatment, (1962).

H3 Heukelekian, H., "Thermophilic Digestion of Daily Charges of Fresh Solids and Activated Sludge", Sewage Works Journal, 3, pp. 3 - 19, (1931).

J1 James, T.W., "Continuous Culture of Micro-organisms", Annual Review of Microbiology, 15, pp. 27 - 46, (1961).

J2 Jeris, J.S. and McCarty, P.L., "The Biochemistry of Methane Fermentation Using C^{14} Tracers", Journal Water Pollution Control Federation, 37, No. 2, pp. 178 - 192, (1965).

K1 Kaplovsky, A.J., "Volatile Acid Production During the Digestion of Seeded, Unseeded and Limed Fresh Solids", Sewage and Industrial Wastes, 23, No. 6, pp. 713 - 721, (1951).

K2 Keefer, C.E. and Kratz, H., "The Effect of Gases on Sewage Sludge Digestion", Sewage Works Journal, 4, pp. 247 - 251, (1932).

K3 Kugelman, I.J. and McCarty, P.L., "Cation Toxicity and Stimulation in Anaerobic Waste Treatment. 2. Daily Feed Studies", Proceedings 19th Industrial Waste Conference, Purdue University, pp. 667 - 686, (1964).

L1 Lawrence, A.W., McCarty, P.L. and Guerin, F.J.A., "The Effect of Sulphides on Anaerobic Treatment", Proceedings 19th Industrial Waste Conference, Purdue University, pp. 343 - 357, (1964).

L2 Lawrence, A.W. and McCarty, P.L., "Kinetics of Methane Fermentation in Anaerobic Treatment", Journal Water Pollution Control Federation, 41, No. 2, Part 2, pp. R1 - R17, (Feb. 1969).

L3 Lawrence, A.W. and McCarty, P.L., "Unified Basis for Biological Treatment Design and Operation", Proceedings of the American Society of Civil Engineers, Journal of the Sanitary Engineering Division, SA 3, pp. 757 - 778, (June 1970).

L4 Loehr, R.C., "Design of Anaerobic Digestion Systems", Proceedings of the American Society of Civil Engineers, Journal of the Sanitary Engineering Division, SA 1, pp. 19 - 29, (Feb. 1966).

L5 Loehr, R.C., "Developments in Industrial Microbiology", 9, Chapter 12, pp. 160 - 174, (1968).

L6 Ludzack, F.J. and Noran, D.K., "Tolerance of High Salinities by Conventional Waste Water Treatment Processes", Journal Water Pollution Control Federation, 37, No. 10, pp. 1404 - 1416, (1965).

M1 Malina, J.F., Jr., "The Effect of Temperature on High-Rate Digestion of Activated Sludge", Proceedings of 16th Industrial Waste Conference, Purdue University, pp. 232 - 250, (1961).

M2 Malina, J.F., Jr., "Variables Affecting Anaerobic Digestion", Public Works, pp. 113 - 116, (Sept. 1962).

M3 Malina, J.F., Jr., "Anaerobic Waste Treatment", Water Resource Management Series, 1, Manual of Treatment Processes, Environmental Science Services Corporation, (1969).

M4 Maselli, J.W., Maselli, N.W. and Burford, M.G., "The Occurence of Copper in Water, Sewage and Sludge and its Effects on Sludge Digestion", New England Interstate Water Pollution Control Commission Report, (June 1961).

M5 McCarty, P.L. and McKinney, R.E., "Volatile Acid Toxicity in Anaerobic Digestion", Journal Water Pollution Control Federation, 33, No. 3, pp. 223 - 232, (1961).

M6 McCarty, P.L. and McKinney, R.E., "Salt Toxicity in Anaerobic Digestion", Journal Water Pollution Control Federation, 33, No. 4, pp. 399 - 415, (1961).

M7 McCarty, P.L., Jeris, J.S. and Murdoch, W., "Individual Volatile Acids in Anaerobic Treatment", Journal Water Pollution Control Federation, 35, No. 12, pp. 1501 - 1516, (1963).

M8 McCarty, P.L., "Anaerobic Waste Treatment Fundamentals. Part One: Chemistry and Microbiology", Public Works, pp. 107 - 112, (Sept. 1964).

M9 McCarty, P.L., "Anaerobic Waste Treatment Fundamentals. Part Two: Environmental Requirements and Control", Public Works, pp. 123 - 126, (Oct. 1964).

M10 McCarty, P.L., "Anaerobic Waste Treatment Fundamentals. Part Three: Toxic Materials and their Control", Public Works, pp. 91 - 94, (November 1964).

M11 McCarty, P.L., "Anaerobic Waste Treatment Fundamentals. Part Four: Process Design", Public Works, pp. 95 - 99, (December 1964).

M12 McCarty, P.L., "Anaerobic Treatment of Soluble Wastes", Special Lecture Series on Advance in Water Quality Improvement, The University of Texas, April 4 - 7, (1966).

M13 McCarty, P.L., "Kinetics of Waste Assimilation in Anaerobic Treatment", in "Developments in Industrial Microbiology", 7, p. 144, American Institute Biological Science, Washington D.C., (1966).

M14 McMahon, T.E., "Anaerobic Sludge Digestion" at Topeka, Kansas, M.S. Thesis, University of Kansas, (1967).

M15 Melbinger, N.R. and Donnellon, J., "Toxic Effects of Ammonia Nitrogen in High-Rate Digestion", Journal Water Pollution Control Federation, 43, No. 8, pp. 1658 - 1668, (1971).

M16 Metcalf, L, Transactions of the American Society of Civil Engineers, 46, 456, (1901).

M17 Miller, F.H. and Barron, W.T., "The CO_2 Alarm in Digester Operation", Water and Sewage Works, 104, pp. 362 - 365, (1957).

M18 Monod, J., "The Growth of Bacterial Cultures", A. Rev. Microbiol., 3, p. 371, (1949).

M19 Montgomery, H.A.C., Dymock, J.F. and Thom, N.S., "The Rapid Determination of Organic Acids and their Salts in Sewage Sludge Liquor", The Analyst, 87, p. 949, (Dec. 1962).

M20 Moore, W.A., McDermott, G.N., Post, M.A., Mandia, J.W. and Ettiner, M.B., "Effects of Chromium on the Activated Sludge Process", Journal Water Pollution Control Federation, 33, pp. 54 - 72, (1961).

M21 Morgan, P.F., "Accelerated Process, Gas Diffusion, Pilot Studies", Sewage and Industrial Wastes, 26, No. 4, pp. 462 - 476, (1954).

M22 Moser, H., "The Dynamics of Bacterial Population Maintained in the Chemostat", Carnegie Institute Washington, Publication No. 614, (1958).

M23 Mueller, L.E., Hindin, E., Lunsford, J.W. and Dunstan, G., "Some Characteristics of Anaerobic Sludge Digestion 1. Effect of Loading", Sewage and Industrial Wastes, 31, No. 1, (Jan. 1959).

N1 Nash, N. and Chasick, A.H., "High-Rate Digester Performance at Jamaica", Journal of the Water Pollution Control Federation, 32, pp. 526 - 536, (1960).

N2 Niles, A.H. and Frook, J.E., "Studies on Digestion of End Liquor from Production of Monosodium Glutomate", Proceedings from the 7th Industrial Waste Conference, Purdue University, pp. 269 - 281, (1952).

O1 O'Rourke, J.T., "Kinetics of Anaerobic Treatment at Reduced Temperatures", Thesis presented to Stanford University, Stanford, California, in 1968, in partial fulfilment of the requirements for the Degree of Doctor of Philosophy.

P1 Plummer, A.G. Jr., Malina, J.F. Jr. and Eckenfelder, W.W. Jr., "Stabilization of a Low Solids Carbohydrate Waste by an Anaerobic Submerged Filter", Proceedings of the 23rd Industrial Waste Conference, Part One, pp. 462 - 473, (1968).

P2 Pohland, F.G. and Bloodgood, D.E., "Laboratory Studies on Mesophilic and Thermophilic Anaerobic Sludge Digestion", Journal Water Pollution Control Federation, 35, No. 1, pp. 11 - 42, (1963).

P3 Pohland, F.G. and Engstrom, R.J., "High-Rate Digestion Control I. Fundamental Concepts of Acid-Base Equilibrium", Proceedings 19th Industrial Waste Conference, Purdue University, pp. 80 - 90, (1965).

P4 Pohland, F.G., "High-Rate Digestion Control II. Techniques for Evaluating Acid-Base Equilibrium", Proceedings 22nd Industrial Waste Conference, Purdue University, pp. 353 - 365, (1967).

P5 Pohland, F.G., "High-Rate Digestion Control III. Acid-Base Equilibrium and Buffer Capacity", Proceedings 23rd Industrial Waste Conference, Purdue University, pp. 275 - 284, (1968).

P6 Pretorius, W.A., "Anaerobic Digestion 3. Kinetics of Anaerobic Fermentation", Water Research, 3, pp. 545 - 558, (1969).

P7 Pretorius, W.A., "Anaerobic Digestion of Raw Sewage", Water Research, 5, No. 9, pp. 681 - 687, (1971).

R1 Rich, L.G., "Unit Processes of Sanitary Engineering", John Wiley and Sons, Inc., New York, (1963).

R2 Rudolfs, W. and Trubnick, E.H., "Compressed Yeast Wastes Treatment". 3. Pilot Plant Digestion Studies", Sewage Works Journal, 21, No. 2, pp. 294 - 305, (March 1949).

S1 Sawyer, C.N., Howard, F.S. and Pershe, E.R., "Scientific Basis for Liming of Digesters", Sewage and Industrial Wastes, 26, No. 9, p. 935, (1954).

S2 Sawyer, C.N. and Schmidt, H.E., "High-Rate Sludge Digestion", Journal of the Boston Society of Civil Engineers, 42, pp. 1 - 17, (1955).

S3 Sawyer, C.N. and Roy, H.K., "A Laboratory Evaluation of High-Rate Sludge Digestion", Sewage and Industrial Wastes, 27, No. 12, pp. 1356 - 1363, (1955).

S4 Schroepfer, G.J., Fullen, W.F., Johnson, A.S., Ziemke, N.R. and Anderson, J.J., "The Anaerobic Contact Process as Applied to Packinghouse Wastes", Sewage and Industrial Wastes, 27, pp. 460 - 486, (1955).

S5 Schulze, K.L. and Raju, B.N., "Studies on Sludge Digestion and Methane Fermentation II. Methane Fermentation of Organic Acids", Sewage and Industrial Wastes, 27, No. 12, p. 1356, (Dec. 1955).

S6 Sen, B.P. and Bhaskaran, J.R., "Anaerobic Digestion of Liquid Molasses Distillery Wastes", Journal Water Pollution Control Federation, 34, No. 10, pp. 1015 - 1025, (1962).

S7 Smith, P.H., "The Microbial Ecology of Sludge Methanogenesis", Developments in Industrial Microbiology, 14, No. 3, pp. 368 - 371, (1966).

S8 Smith, P.H. and Mah, R.A., "Kinetics of Acetate Metabolism During Sludge Digestion", Applied Microbiology, 14, No. 3, pp. 368 - 371, (1966).

S9 Speece, R.E. and McCarty, P.L., "Nutrient Requirements and Biological Solids Accumulation in Anaerobic Digestion", Proceedings of the 1st International Conference on Water Pollution Research, 1962, Pergamon Press, London, (1964).

S10 Standard Methods for the Examination of Water and Waste Water (12th Edn.), American Public Health Association, New York, N.Y., (1965).

S11 Stander, G.J., "Treatment of Wine Distillery Wastes by Anaerobic Digestion", Proceedings 22nd Industrial Conference, Part 2, pp. 892 - 907, (1967).

S12 Stander, G.J., Cillie, G.G., Ross, W.R., Bailie, R.D., Gouws, E.P. and Flinke, J.W., "Treatment and Disposal of Yeast Wastes", National Institute for Water Research, South African Council for Scientific and Industrial Research, Research Report No. 305, Pretoria, (June 1971).

S13 Steenveld, G.N., Private Communication.

S14 Steffen, A.J. and Bedker, M., "Operation of Full-Scale Anaerobic Contact Treatment Plant for Meat Packing Wastes", Proceedings 16th Industrial Waste Conference, Purdue University, pp. 423 - 437, (1961).

S15 Stewart, M.J., "Reaction Kinetics and Operational Parameters of Continuous Flow Anaerobic Fermentation Processes", Sanitary Engineering Research Laboratory Report No. 2, University of California, Berkeley, (1958).

S16 Straub, C.P., "Effect of Addition of Nitrogen on Digestion of Paper Pulp", Sewage Works Journal, 16, pp. 30 - 41, (1944).

T1 Toerien, D.F. and Hattingh, W.H.J., "Anaerobic Digestion - 1. The Microbiology of Anaerobic Digestion", Water Research, 3, pp. 385 - 416, (1969).

T2 Torpey, W.N., "High-Rate Digestion and Concentrated Primary and Activated Sludge", Sewage and Industrial Wastes, 26, pp. 479 - 494, (1954).

W1 Water Pollution Control Federation Manual of Practice No. 16: "Anaerobic Sludge Digestion", Water Pollution Control Federation, Washington, p. 41, (1968).

W2 Weston, R.F. and Eckenfelder, W.N., "Application of Biological Treatment of Industrial Wastes: 1. Kinetics and Equilibria of Oxidative Treatment", Sewage and Industrial Wastes, 27, No. 7, p. 802, (1955).

W3 Willimon, E.P. Jr. and Andrews, J.F., "Multi-Stage Biological Processes for Waste Treatment", Proceedings 22nd Industrial Waste Conference, Part 2, pp. 645 - 660, (1967).

W4 Wuhrmann, K., "Microbial Aspects of Water Pollution Control", Advances in Applied Microbiology", 6, Academic Press, New York, (1964).

Y1 Young, J.C. and McCarty, P.L., "The Anaerobic Filter for Waste Treatment", Journal Water Pollution Control Federation, 41, No. 5, Part 2, pp. 160 - 173, (1969).

Z1 Zablatzsky, H.R.,"Toxic Effects of Ammonia Nitrogen in High-Rate Digestion", A Discussion. Journal Water Pollution Control Federation, 43, No. 8, pp. 1668 - 1670, (1971).

APPENDIX A

N O M E N C L A T U R E

As mentioned in Chapter 5, the system of nomenclature used in this thesis, is based as closely as possible on that recommended by Fencl (F2), following the Symposium on Continuous Fermentation held in Czechoslovakia in 1967.

Symbol	Description	Units
F	Feed rate	mass/volume
k_d	Specific decay rate	time $^{-1}$
K_s	Limiting substrate concentration at $\frac{1}{2}$ maximum specific growth rate; (saturation constant)	mass/volume
S	Substrate concentration	mass/volume
t	Time	
V	Volume of reactor	
W	Waste underflow rate	volume/time
X	Sludge concentration	mass/volume
Y	Yield coefficient	
μ	Specific growth rate	time^{-1}
$\hat{\mu}$	Maximum specific growth rate	time $^{-1}$
θ	Hydraulic residence time	
θ_c	Sludge age (solids retention time)	
1	Subscript denoting reactor effluence	
0	Subscript denoting reactor influent	

APPENDIX B

M E T H O D S O F A N A L Y S I S

O P E R A T I N G D A T A

Table B.1 compares the results obtained for the analysis of volatile acids by the methods of DiLallo and Albertson (D1) and Montgomery, Dymock and Thom (M22) respectively.

TABLE B.1. Comparison of concentrations of volatile acids determined by titrometric and calorimetric methods.

TITROMETRIC METHOD (D1)	CALORIMETRIC METHOD (M22)
580	550
610	620
1 110	1 050
1 420	1 350
1 680	1 810
2 270	2 800

Table B.2 contains a summary of the operating conditions used during the analysis of digester gas by gas/liquid chromatography.

TABLE B.2. Data pertaining to analysis of digester gas by g/l chromatography.

Column:	9' x 3/16" O.D. stainless steel packed with Porapak S, 80 - 100 mesh
Temperature:	Oven -100°C Injection Part - 25°C Detector - 25°C
Carrier Gas:	Hydrogen at 30 psig
Bridge Current:	150 mA
Recorder:	Beckman, model 1005
Chart Speed:	0,2" per minute
Sample Size:	0,5 ml
Syringe:	Hamilton gas-tight syringe
Attenuation:	32

Table B.3 summarizes the operating parameters used during the atomic absorption analysis of potassium, sodium, copper, iron, zinc and nickel.

TABLE B.3. Data pertaining to analysis of various cations by atomic absorption spectroscopy.

Element	Wave Length (λ) (NM)	Slit Width (NM)	Lamp Current (ma)
Na	589	0,20	5
K	766,5	0,20	5
Cu	324,8	0,20	3
Fe	248,3	0,20	5
Ni	232,0	0,20	5
Zn	213,9	0,50	5

A P P E N D I X C

TABLE C.1. Values of the parameters used to monitor digestion during a typical start-up period of a batch digester.

Day	Feed Conc. (mg/l COD)	Effluent Conc. (mg/l COD)	Volatile Acids (mg/l CH_3 COOH)	Gas Prod. (l/day)	CO_2 Content %
1	10 000		439		20
2	10 000		590	2,73	26
3	10 000	4 270	1 220	2,19	28
4	10 000		1 010	1,84	27
5	10 000	4 100	1 220	1,97	30
6	10 000			1,91	
7	10 000	4 160	811	2,01	31
8	10 000			1,99	29
9	10 000		1 010	2,03	34
10	10 000	3 610		1,99	35
11	10 000		893	1,94	33
12	10 000	3 730		1,98	33
13	10 000		802	2,01	31
14	10 000			2,10	33
15	10 000	3 780	782	2,09	34
16	10 000		807	2,08	32
17	10 000			2,02	32
18	10 000	3 350	619	2,04	33
19	10 000			2,02	
20	10 000	3 520	646	2,02	31

TABLE C.2. Values of the parameters used to monitor digestion during a change in the steady state conditions applying to the system.

Day	Influent (mg/l COD)	Effluent (mg/l COD)	Gas Prod. (l/day)	CO_2 Content (%)	Volatile Acids mg/l (CH_3COOH)	Waste Stream (l/day)	Sludge Conc.	
							Total (mg/l)	Volatile (mg/l)
0	6 943	2 580	9,39	28,1	310	0,070	13 300	11 300
1	7 243		9,84	29,4		0,072		
2	7 543		9,86	32,2		0,075		
3	7 843		10,01	31,9		0,071		
4	8 143		10,48	30,9		0,060		
5	8 443		10,91			0,093		
6	8 743		10,57			0,092		
7	9 043		11,46			0,076		
8	9 343		11,52			0,081		
9	9 643		12,01			0,070		
10	9 943	4 400	12,63	32,7	760	0,072		
11	10 050		13,31			0,096		
12	10 050		13,17			0,102		
13	10 050		13,87			0,120		
14	10 050		13,87			0,120	11 400	9 700
15	10 050		13,76			0,124		
16	10 050		13,95			0,108		
17	10 050	4 080	14,30	30,3	620	0,115		
18	10 050		14,51			0,115		
19	10 050		14,68			0,128		
20	10 050		14,85			0,140		
21	10 050		14,69			0,105		
22	10 050		14,86			0,120		
23	10 050		14,91			0,132		
24	10 050	3 620	15,04	29,7	490	0,130	12 400	10 300
25	10 050		14,89			0,100		
26	10 050		14,87			0,120		
27	10 050		14,98			0,124		

TABLE C.2. Continued.

Day	Influent (mg/l COD)	Effluent (mg/l COD)	Gas Prod. (l/day)	CO_2 Content (%)	Volatile Acids (mg/l CH_3COOH)	Waste Stream (l/day)	Sludge Conc. Total (mg/l)	Sludge Conc. Volatile (mg/l)
28	10 050		15,01			0,122		
29	10 050		14,98			0,128		
30	10 050		15,03			0,130	12 600	10 600
31	10 050	3 610	15,09	30,2	530	0,132		
32	10 050		14,99			0,134		
33	10 050		15,01			0,148		
34	10 050		15,12			0,126		
35	10 050		14,92			0,120		
36	10 050		14,87			0,120	13 900	11 400
37	10 050		15,13			0,130		
38	10 050	3 570	15,03	29,5	490	0,134		
39	10 050		15,11			0,134		
40	10 050		15,13			0,136		
41	10 050		14,97			0,134		
42	10 050		14,86			0,131		
43	10 050		14,93			0,130		
44	10 050	3 530	14,90	28,8	410	0,137	12 900	11 000
45	10 050		15,01			0,130		
46	10 050		15,00	30,1	490	0,132		
47	10 050	3 590	14,94	28,9	470	0,134	13 500	11 200
48	10 050		15,04	28,4		0,132		
49	10 050	3 550	15,03	29,6		0,131		
50	10 050	3 500	15,01	28,9	420	0,133	12 700	11 000

Table C.3. Summary of steady state values obtained during this investigation

θ (days)	Gas Production (ℓ/day/ℓ)	Volatile Acids (mg/ℓ CH_3COOH)	CO_2 Content (%)	S_o (mg/ℓCOD)	S_1 (mg/ℓCOD)	S_o Degradable (mg/ℓ COD)	S_1 (mg/ℓCOD)	X (Total) (mg/ℓ)	X (Volatile) (mg/ℓ)	W (ℓ/day)
1,70	0,985	402	26,3	5 574	2 210	4 035	671	13 500	11 500	0,059
	1,063	405	27,6	6 207	2 174	4 767	734	12 800	10 800	0,063
	1,250	277	28,3	6 943	2 598	5 132	787	13 000	11 200	0 072
	2 006	450	29,1	10 050	3 540	7 688	1 178	13 000	11 100	0,133
	2,830	1 236	31,2	16 100	6 092	12 300	2 292	13 500	11 500	0,212
	3,680	1 250	32,0	19 302	6 302	14 300	2 300	13 400	11 300	0,293
1,45	0,765	117	25,3	4 252	1 814	2 922	484	12 000	10 200	0,041
	1,140	408	27,1	6 177	2 152	4 725	700	13 000	11 100	0,105
	1,361	296	27,5	8 330	2 758	6 372	800	13 100	11 400	0,138
	2,132	416	29,9	5 230	1 960	3 786	516	12 000	10 000	0,079
	2,645	731	32,9	10 287	3 754	7 940	1 407	11 500	9 500	0,180
	2 960	1 400	32,6	13 600	5 150	10 550	2 100	11 500	9 600	0,265
3,53	0,549	162	25,6	6 491	2 758	4 136	403	14 100	11 800	0,028
	0,685	284	27,5	7 563	2 663	5 360	460	14 500	12 200	0,034
	0,936	280	28,6	10 390	3 935	7 067	612	14 000	12 100	0,053
	1,640	428	30,4	13 319	3 956	10 189	1 125	14 200	11 620	0,116
	1,890	461	32,5	16 420	4 735	12 810	1 125	13 600	11 200	0,138
	1,938	1 160	35,7	19 320	6 943	14 780	2 403	13 000	11 000	0,163

TABLE C.4. Data pertaining to the possible toxic effects of sulphide, chloride and potassium on digestion.

Day	Digester	Feed Conc. (mg/l COD)	Gas Prod. (l/day)	Volatile Acids (mg/l CH_3COOH)	Fe Concentration		Concentration		
					Influent (ppm)	Effluent (ppm)	$S^=$ (ppm)	K^+ (ppm)	Cl^- (ppm)
41	1	17 500	2,88	975	10	1	98		
	2	17 500	2,91	950	43	2	65		
	3	16 000	2,75	730				2 100	4 500
	4	16 000	2,71	810				2 100	
	5	16 000	2,81	820					6 250
42	1	18 100	2,95	960					
	2	18 100	2,93	1 010					
	3	16 000	2,80					2 100	4 500
	4	16 000	2,76					2 100	
	5	16 000	2,73						6 500
43	1	18 700	3,01	990	10	2			
	2	18 700	2,94	990	44	2			
	3	16 000	2,69					2 100	4 500
	4	16 000	2,76					2 100	
	5	16 000	2,79	710					6 750
44	1	19 300	3,02						
	2	19 300	2,98						
	3	16 000	2,73					2 100	4 500
	4	16 000	2,78					2 100	
	5	16 000	2,83						7 000
45	1	19 900	3,04		11	1	114		
	2	19 900	3,03		45	2	70		
	3	16 000	2,70					2 100	4 500
	4	16 000	2,71					2 250	
	5	16 000	2,79						7 250
46	1	20 500	3,09	1 320					
	2	20 500	3,01	1 350					
	3	16 000	2,69	840				2 100	4 500
	4	16 000	2,72	890				2 500	
	5	16 000	2,76	910					7 500
47	1	21 100	3,07	1 390					
	2	21 100	3,07	1 380					
	3	16 000	2,77					2 100	4 500
	4	16 000	2,78					2 750	
	5	16 000	2,79	850					7 750
48	1	21 700	3,10		12	2			
	2	21 700	3,11		46	2			
	3	16 000	2,71					2 100	4 500
	4	16 000	2,69					3 000	
	5	16 000	2,71						8 000
49	1	22 300	3,20	1 510					
	2	22 300	3,13	1 570					
	3	16 000	2,73					2 100	4 500
	4	16 000	2,51	1 090				3 250	
	5	16 000	2,69						8 250

TABLE C.4. Continued.

Day	Digester	Feed Conc. (mg/l COD)	Gas Prod. (l/day)	Volatile Acids mg/l (CH_3COOH)	Fe Concentration		Concentration		
					Influent (ppm)	Effluent (ppm)	$S^{=}$ (ppm)	K^{+} (ppm)	Cl^{-} (ppm)
50	1	22 900	3,28	1 940					
	2	22 900	3,21	1 880					
	3	16 000	2,67					2 100	4 500
	4	16 000	2,55	1 040				3 500	
	5	16 000	2,75						8 500
51	1	23 500	3,25	2 070	13	2	138	2 950	
	2	23 500	3,13	2 130	47	2	107	2 780	
	3	16 000	2,69	790				2 100	4 500
	4	16 000	2,41	1 130				3.750	
	5	16 000	2,73	820					8 750
52	1	23 500	2,85	2 720					
	2	23 500	2,76	2 900					
	3	16 000	2,71					2 100	4 500
	4	16 000	2,49					4 000	
	5	16 000	2,79						9 000
53	1	23 500	2,16	3 190					
	2	23 500	2,19	3 350					
	3	16 000	2,65					2 100	4 500
	4	16 000	2,42					4 250	
	5	16 000	2,71						9 250
54	1	23 500	2,03	3 370					
	2	23 500	1,99	3 510					
	3	16 000	2,69					2 100	4 500
	4	16 000	2,49					4 500	
	5	16 000	2,74						9 500
55	1	23 500	1,59	3 690	13	2	146	2 900	
	2	23 500	1,44	3 850	47	2	116	2 900	
	3	16 000	2,63	710				2 100	4 500
	4	16 000	2,45	1 190				4 500	
	5	16 000	2,76	950					9 750