ANAEROBIC FERMENTATION OF SOLIDS-SEPARATED SWINE SLURRY

A Thesis Presented to
The Faculty of Graduate Studies
The University of Manitoba

In Partial Fulfillment
of the Requirements for the Degree
Master of Science
in Civil Engineering

by
JOE WONG

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ABSTRACT

The application of the anaerobic fermentation process as a method of reducing the pollutional characteristics of solids-separated swine slurry and of producing usable energy from such wastewaters was evaluated with laboratory-scale flow-through anaerobic reactors without sludge recycle. At 35°C, with once-a-day feeding and with manual mixing, the diluted swine slurry was satisfactorily fermented anaerobically at SRT's of 15, 20, and 25 days. Total VS removal varied from 38.32% at a 15-day SRT to 44.07% at a 25-day SRT, while total COD removal ranged from 45.41% at a SRT of 15 days to 54.16% at a SRT of 25 days. Gas production varied from 0.44 - 0.54 m^3/kg VS added. Analyses of gas produced showed a methane content of about 69% for each reactor. The biodegradable fraction was determined to be 56% of the influent VS concentration. The substrate removal rate was determined to be 0.48 day^{-1} using the model developed by Morris *et al.* (1977). The results indicated that the flow-through anaerobic reactor without sludge recycle can effectively treat diluted swine slurries. Flow-through digestion in full scale would, however, be unfeasible economically due to the large reactor volume required to maintain adequate detention time for organics removal and gas production.

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

GLOSSARY

TS	total solids
VS	volatile solids
BOD	biochemical oxygen demand, mass/volume
COD	chemical oxygen demand, mass/volume
HRT	hydraulic retention time, time
SRT	solids retention time, time
STP	standard temperature (0°C) and pressure (1 atmosphere)
V_{STP}	volume at standard temperature and pressure
V_{Ta}	volume at temperature "a"
T_{STP}	temperature at STP = 273.16°K
T_a	temperature "a" = $C_a°$ + 273.16°
R	refractory fraction or the ratio of the refractory VS concentration to the total influent VS concentration, expressed as a decimal
S_o	influent total substrate concentration, mass/volume
S_1	effluent total substrate concentration, mass/volume
$(S_b)_1$	effluent biodegradable substrate concentration, mass/volume
k	rate of substrate removal, $time^{-1}$

CHAPTER 1

INTRODUCTION

1.1 WASTE HANDLING ON INTENSIVE SWINE FARMS

In recent years, the swine industry has shifted toward total confinement production. Mechanization, improved agricultural practices and advances in nutrition and disease control have made it possible for swine producers to manage thousands of animals on a fraction of the land area previously required. The change-over from traditional animal husbandry to the system of confined, intensive raising of animals has greatly increased the quality and quantity of animals produced, but not without increased environmental problems.

Waste handling and disposal is perhaps the largest single problem arising from confinement production of swine. For the purpose of reducing labour requirements, intensive swine units are commonly equipped with hydraulic flushing systems for waste removal. Diluted wastewaters containing swine feces, urine and spillage water are therefore produced rather than the traditional farmyard manure. Flush systems are popular among swine producers because of their relatively low construction costs, low odour in the confinement units due to frequent waste removal, and low labour requirements due to a more automated waste handling process (Jones and Nye, 1981). Flushing is also a very effective method for removing waste collected in gutters or in canals under slotted floors (Jelínek, 1977). Solids settling out of the resulting slurry can, however, create major pumping or handling problems, tending to negate the advantages of a hydraulic transport system (Hegg *et al.*, 1981). Thus, mechanical separation

of the coarse solids is often utilized to convert the swine waste slurry into solid and liquid fractions.

Mechanical separation is being recognized as a suitable processing method for converting farm animal slurries into solid and liquid fractions (Pain *et al.*, 1978; Rorick *et al.*, 1980). Effective separation produces a stackable, fibrous product which could be readily handled in the same way as traditional farmyard manure and a free-flowing liquid, rich in soluble nutrients.

1.2 WASTE TREATMENT AND DISPOSAL

The separated liquid fraction of the swine waste slurry poses a major problem for the swine producers. Disposal of the liquid fraction by land application can overburden the land with the liquid slurry since the area of land available around such intensive systems is often not commensurate with the volumes of slurries produced. In addition, land application cannot be undertaken continuously because of climatic conditions, especially in the colder climates, or crop requirements. Storage of the liquid slurry must therefore be provided. The transport, storage, and land application of swine slurry can lead to problems of air pollution from malodours, and to problems of land and water pollution by run-off or by seepage (Summers and Bousfield, 1980). The surplus liquid fraction of swine slurry becomes a major pollutant on many intensive swine units utilizing flush systems and mechanical separators. It is necessary, therefore, to treat this waste stream to remove its pollutants.

A process which is capable of reducing the environmental pollutant, while at the same time producing sufficient energy to offset the costs of operating the process or, better still, generating a surplus of usable energy, is of great advantage to an intensive swine producer. Anaerobic fermentation is such a

process. In anaerobic treatment, organic material is microbiologically converted to "biogas", a mixture of methane and carbon dioxide which is a very useful source of energy. Energy production is of particular importance to intensive swine producers because of the increasing prices of conventional fuels, the possible shortages of these fuels, and the necessity of thermal and electrical energy to raise hogs.

Intensive swine units typically require thermal energy for the comfort of baby pigs in the farrowing and nursery houses and electrical energy throughout the farm for lighting, ventilation, and waste removal (Fischer *et al.*, 1979b). Since both thermal and electrical energy are required every day of the year, there exists a continuous demand for the energy that can be obtained from the biogas produced from anaerobic treatment systems.

Anaerobic treatment, because of the production of a useful fuel, provides an attractive alternative to energy-consuming treatment systems. Aerobic treatment systems, for instance, require large energy inputs, whether designed as part of a complete treatment system or only for odour control. Aeration requires a minimum energy input of 36 kWh per hog place per year to prevent malodorous nuisance during storage and land spreading of swine manure (van der Hoek, 1977). Mills (1977), in a comparison of aerobic and anaerobic treatment of swine waste from the same source, has shown aerobic systems to have a lower capital cost but higher operating costs, due to the need to provide constant aeration.

The use of the other product of anaerobic fermentation, the digested residue, as fertilizer is an additional benefit of the process. The digested residue,

a blackish, free-flowing, stabilized liquid with no offensive odour, retains elements such as nitrogen, phosphorus, and potassium from the untreated waste, which enables the reuse of these compounds for fertilizer purposes.

Anaerobic fermentation therefore offers certain benefits to swine producers - pollution abatement, energy production, and the production of a stabilized residue which retains the fertilizer value of the untreated waste.

1.3 ANAEROBIC TREATMENT OF SWINE WASTES

Research conducted by Jeffrey *et al.* (1965), Schmid and Lipper (1969), Cross and Duran (1970), Gramms *et al.* (1971), and Fischer *et al.*(1975) have determined that swine wastes can be anaerobically fermented for methane production. The majority of the laboratory swine waste digestion research was conducted on swine feces alone or on swine slurry (feces and urine) of 6-10% total solids. Swine wastes of 3-5% total solids have been treated anaerobically (Hobson and Shaw, 1973; Summers and Bousfield, 1976), but studies on swine slurries with less than 3% total solids are lacking.

The waste slurry produced from intensive swine units equipped with hydraulic flushing systems is usually 3% or less total solids (Hill and Tollner, 1980; Fischer and Iannotti, 1981). The use of a mechanical separator to remove the coarse solids from the slurry would result in a waste stream containing even less total solids. According to Hill and Tollner (1980), this dilute waste slurry must be concentrated before utilizing anaerobic treatment. Fischer and Iannotti (1981) expressed the same opinion. According to these two researchers, a total solids concentration of approximately 8% is required for optimal loading of an anaerobic flow-through reactor. They recommended sedimentation as an economical method of obtaining this solids concentration.

The sedimentation of swine manure slurries with subsequent wastage of clarified liquid results in losses of dissolved and suspended solids and thus, potential methane production from an anaerobic treatment system (Sievers *et al.*, 1981). According to Sievers *et al.* (1981), methane losses up to 45% can result from the sedimentation of swine manure slurries with subsequent removal of all clarified liquid. Fischer and Iannotti (1981) determined that as much as 30% of the biodegradable volatile solids can be removed in the excess liquid as the solids content is increased from 1.5% - 8% by sedimentation. In consequence, it would not be advantageous to concentrate the solids in the liquid fraction of swine slurry since the excess liquid would still possess a high pollution potential. In light of energy production, the excess liquid represents a loss in potential methane production.

1.4 OBJECTIVES

Intensive swine units with hydraulic transport of wastes and mechanical separation of coarse solids produce significant quantities of wastewaters that are rather dilute and of high pollution potential. The characteristics of this wastewater are such that it warrants investigations of its treatability for stream discharge.

The purpose of this study was to assess the potential of anaerobic fermentation to produce usable energy from swine slurries with low initial solids content and to reduce the pollutional characteristics of such wastewaters. The specific objectives of this study were to prove the feasibility of treating dilute swine slurries anaerobically, and to establish limits of biodegradability, energy production, substrate utilization rate, and effluent qualities that can be expected.

CHAPTER 2

BACKGROUND

2.1 GENERAL CHARACTERISTICS OF SWINE WASTES

The characteristics of swine wastes are a function of the digestibility of the feed ration, the protein and fiber content and the nature of the other feed elements (Taiganides and Hazen, 1966; Loehr, 1974). Fecal wastes from swine contain undigested feed, mostly cellulose fiber, residue from the digestive fluids, worn-out cells from the intestinal linings, waste mineral matter, mucus, bacteria, and foreign matter such as dirt consumed along with the feed ration (Loehr, 1974). Calcium, magnesium, and iron are also excreted in the feces.

Most of the nitrogen of the feed is contained in the proteins. Proteins vary in digestibility. Nitrogen in the digested proteins is absorbed and used to build flesh in the animal. Excess nitrogen from the digested proteins is excreted in the urine. Undigested proteins and the nitrogen they contain are voided in the feces. The potassium content of the feed is absorbed during digestion. Almost all of the absorbed potassium is, however, eventually excreted. Likewise, part of the phosphorus in the feed is absorbed, but the majority is voided in the feces.

Swine wastes can contain feed spilled on the pen floors. Therefore, all of the ingredients of the swine feed ration, whether in their original form or in chemically simpler forms, will eventually end up in the wastes.

The daily waste production from a hog is dependent on the breed and size of the animal, the feed ration, and the temperature and humidity within the building in which the animal grows (Loehr, 1974). The quantity of waste produced increases with the weight of the animal and the amount of feed it consumes. Table 2.1 shows the mass of wet manure produced per g of animal

per day and the solids concentration of wet manures for swine (Middlebrooks, 1974).

Accórding to Loehr (1974), waste production can range from 6-8% of the body weight of a hog per day. Taiganides *et al.* (1964) noted that the daily production of swine waste varies considerably with the time of year and suggested an average manure production value of 5.0 kg per day per 100 kg live weight be used for purposes of estimating waste quantities.

Jelínek (1977) found that young pigs of 5 to 15 kg live weight generated almost 50% more excrement per unit of live weight than fattened pigs weighing 66 to 100 kg. He also measured the waste (feces and urine) production from seven large swine feedlots and determined that the waste produced per mean weight of the pigs in the feedlots ranged from 5.5 to 10.5 kg of waste per day per pig fattened.

Table 2.2 shows the BOD, COD, and nutrient contents of swine manures in weight per weight of animal per day x 10^{-3} as reported in the literature (Middlebrooks, 1974).

2.1.1 Malodours in Anaerobically Stored Swine Wastes

Odours from anaerobically stored swine wastes could be considered a nuisance when exhausted from a building or when the swine waste slurry is spread on land (Loehr, 1974). Over sixty compounds have been identified in both the air of swine confinement buildings and anaerobically stored swine wastes (Spoelstra, 1978). The main compounds responsible for the malodours are volatile fatty acids (particularly butyric acid), diacethyl, phenol, p-cresol, indole, and skatole (Schaefer *et al.*, 1974).

TABLE 2.1 PHYSICAL CHARACTERISTICS OF SWINE DEFECATION
(Middlebrooks, 1974)

Animal	Wet Manure (gg^{-1} of animal-day)	Total Solids (gg^{-1} of animal-day)	Total Solids (% of W.W.)	Volatile Solids (gg^{-1} of animal-day)	Volatile Solids (% of T.S.)	References
Swine	0.084	0.011	13.1	-	-	Moore, 1969
	0.028-0.095	0.008-0.016	12-28	0.0068-0.0136	83-87	Taiganides and Hazen, 1966
	0.087	0.016	18	-	-	Hart, 1960
	-	0.0080	-	0.0063	78.5	Hart and Turner, 1965
	-	0.0097	-	0.0080	82.5	Taiganides, *et al.*, 1964
	-	0.0050	-	0.0035	70	Clark, 1965
	-	0.0071	-	-	-	Dept. Sc. & Indust. Res., 1964
	-	0.0048	-	0.0033	68.8	Humenik, 1972
	0.074	0.0059	8	0.0047	79.7	Schmid and Lipper, 1969
	-	0.0099	-	0.0070	71	Townshend, *et al.*, 1969
Average	0.074	0.0089	17.4	0.0054	76.5	

TABLE 2.2 Nutrient and Sanitary Characteristics of Swine Manures (Middlebrooks, 1974)

(gg^{-1} of Animal-day x 10^{-3})

Animal	BOD	COD	Ammonia Nitrogen	Total Nitrogen	Phosphorus (P_2O_5)	Potassium (K_2O)	References
Swine	-	-	-	0.51	0.32	0.62	Moore, 1969
	-	-	-	0.42 - 0.60	0.29 - 0.32	0.34 - 0.62	Taiganides and Hazen, 1966
	-	-	-	0.53	-	-	Hart, 1960
	2.0	7.6	0.24	0.32	0.25	0.11	Hart and Turner, 1965
	4.3	5.4	-	0.64	-	-	Taiganides, *et al.*, 1964
	-	4.7	-	-	-	-	Clark, 1965
	2.5	-	-	-	-	-	Little, 1966
	2.2	-	-	0.70	-	-	Poelma, 1966
	5.6	-	-	-	-	-	Dept. Sc. & Indust. Research, 1964
	-	-	-	0.41	0.55	-	Vollenweider, 1968
	3.1	6.4	-	-	-	-	Humenik, 1972
	2.0	5.2	-	-	-	-	Schmid and Lipper, 1969
	3.2	9.3	-	0.44	0.67	-	Townshend, *et al.*, 1969
Average	3.1	6.4	0.24	0.51	0.42	0.40	

The odorous compounds are gaseous products of microbial fermentations under anaerobic conditions. The anaerobic degradation of the organic materials in the swine wastes results in the formation of intermediate fermentation products such as those previously mentioned which are responsible for the offensive odours. Under the controlled conditions of an anaerobic fermentation process, however, these intermediates are further degraded into methane and carbon dioxide which are odorless compounds (Welsh *et al.*, 1977). Anaerobic treatment is thus effective in reducing odours from swine wastes.

2.1.2 Feed Additives in Swine Wastes

Swine producers commonly supplement the feed ration of their animals with feed additives to promote weight gains, improve feed efficiencies, and reduce animal diseases. Certain feed additives, when excreted, affect the decomposition of animal waste. Copper salts are an example of such additives. Copper, in the form copper sulfate, has been used as a growth promoting feed additive in swine diets since 1955 (Barber *et al.*, 1955). The beneficial effects of copper on animal performance have been well documented (Lucas *et al.*, 1962; Castell and Bowland, 1968; Braude and Ryder, 1973; Braude and Hosking, 1975), but others have found that copper in swine diets inhibits the anaerobic decomposition of the stored waste (Taiganides, 1963; Ariail *et al.*, 1971) Arsanilic acid (paraaminophenylarsonic acid) has been used as a feed additive in swine diets since the early 1950's to prevent swine dysentery, and according to Brumm *et al.* (1977), the presence of this feed additive in swine diets and the resultant presence of its metabolites in the waste enhanced the dry matter decomposition in anaerobic pits.

Morrison *et al.* (1969) studied the role of excreted antibiotic in

modifying microbial decomposition of feedlot waste. These authors suggested that excreted chlortetracycline (Aureomycin), a feed antibiotic, may have decreased the conversion efficiency of the microbial flora participating in the decomposition of the waste. Other antibiotics have also been suspected of having an inhibitory effect on the microbial population responsible for the decomposition of animal waste. Fischer *et al.* (1975) speculated that tylosin and lyncomycin, two intramuscular antibiotics which were injected into one of their hogs, caused an upset in their digester.

2.2 ANAEROBIC FERMENTATION

2.2.1 General Background on the Anaerobic Process - Historical Perspective

Anaerobic microbial metabolism may occur at any place where organic materials accumulate and the supply of free oxygen is stopped completely or limited to such an extent that the oxygen present is rapidly removed by aerobic microbial metabolism (Hobson *et al.*, 1974). In nature, the anaerobic fermentation process is a part of the carbon cycle since this process plays a role in the mineralization of organic material (van Velsen, 1981).

The fact that organic material, under conditions where free oxygen is lacking, will decompose and produce an inflammable gas (which is primarily methane, the simplest organic compound of carbon and hydrogen) has been known for centuries, particularly in the phenomenon of marsh gas. The occasional dancing flames of this marsh gas (ignited, perhaps, by sparks from a nearby fire), seen at night, have spawned the legends of the "will-o'-the-wisp", or fool's fire (Anon., 1977). As early as 1630, van Helmont mentioned, among 15 distinct types of gas, a combustible gas that is produced during putrefaction and also is contained in intestinal gas (Partington, 1960). In 1790, Priestley(1790) reported, in his Observations on Inflammable Air, of "air produced by substances putrefying in

water". Priestley confirmed the work of Volta, who in 1776 was the first to recognize the close relationship between the decaying vegetation in the sediment of lakes and streams and the appearance of inflammable gas (Tietjen, 1975).

In 1808, Humphrey Davy (1814) collected methane from strawy cattle manure kept in a retort under vacuum. This might be considered the beginning of anaerobic fermentation research, but Davy's experiments were not directed towards solving energy problems with natural fuel gas; he was more interested in evaluating the fertilizer value of digested and undigested manure for crop production (Tietjen, 1975).

Gayon, a pupil of Pasteur, collected such large quantities of gas during his experiments with digesting animal manure in 1883-1884 that Pasteur concluded anaerobic fermentation of animal manure might supply gas for heating and illumination under special circumstances (Tietjen, 1975). In 1895, Donald Cameron utilized gas from a "carefully designed" septic tank for street lighting in Exeter, England (McCabe and Eckenfelder, Jr., 1957). Since then, the use of the anaerobic fermentation process for the stabilization of domestic sewage sludge has been increasingly applied.

The anaerobic fermentation process has also been utilized for producing energy from agricultural residues whenever fossil energy was in short supply, as in France, Algeria, and Germany during and shortly after World War II, when methane thus produced was utilized to run vehicles (Anon., 1977). In the post war period, however, the interest in anaerobic fermentation diminished due to the availability of inexpensive fossil fuels. On the other hand, in countries hampered by low natural abundance or inadequate distribution of energy supplies, the development of anaerobic manure fermentation processes has been continued.

In some Asian countries, the use of the anaerobic fermentation process

to provide fuel for small-scale applications has been extensive (Pyle, 1980). In the People's Republic of China, the practice has been promoted vigorously and available reports indicate that tens of thousands of anaerobic reactors are in operation throughout the country, based on the use of night soil and animal manures as raw materials (Anon., 1977).

In India, experiments to develop a system to provide fuel from dried cow dung without destroying its fertilizer value were initiated in 1939 (Anon., 1977). These initial experiments have led to the construction of thousands of methane generators - the majority of them in rural areas and serving from one to two families.

Korea, too, has wide experience with small-scale methane generators, with more than 29,000 units installed.

In developed countries, the energy crisis of 1973 and the consequent prospect of ever-increasing prices of finite natural resources has revived the interest and research into the anaerobic fermentation process as an energy-producing method and an energy-saving waste treatment technology (van Velsen, 1981).

2.2.2 The Anaerobic Process for Waste Treatment

The anaerobic waste treatment process converts organic matter to methane gas and carbon dioxide in the absence of molecular oxygen (Switzenbaum, 1978). The anaerobic fermentation process has been traditionally relegated to a supporting role in the field of biological waste treatment, in which organic solids emanating from various points in the liquid processing train of an aerobic waste treatment system are converted to gas (Kirsch and Sykes, 1971; Grady and Lim, 1980). Recently, it has been recognized that the anaerobic fermentation

process can provide an effective means of treating high strength wastewaters. In fact, the anaerobic process may offer certain advantages over the traditional aerobic waste treatment processes. McCarty (1964) pointed out that the principal merits associated with the anaerobic fermentation process are directly attributable to the unique characteristics of the anaerobic microorganisms. Table 2.3 lists five important advantages of anaerobic fermentation processes.

TABLE 2.3 ADVANTAGES OF ANAEROBIC TREATMENT (McCarty, 1964)

1. A high degree of waste stabilization is possible.
2. Low production of waste biological sludge.
3. Low nutrient requirements.
4. No oxygen requirements.
5. Methane is a useful fuel.

Anaerobic processes can convert 80-90% of the biodegradable organic portion of wastes to gas, even in highly loaded systems (McCarty, 1964). Conventional aerobic waste treatment systems, in contrast, actually stabilize only about 50% of the degradable organic fraction of wastes.

Conventional aerobic waste treatment utilizes aerobic microorganisms to convert degradable organics to carbon dioxide and water in the presence of a sufficient supply of oxygen. Since aerobic respiration gives a high energy yield, a large portion of the substrate carbon and nitrogen is converted to new microbial cells (Hobson *et al.*, 1974). The fraction converted to microbial cells is not stabilized as the cells themselves are subject to microbial attack, and so are

potentially pollutional. Removal of these cells from the treated liquid is necessary. The disposal of the resulting biological sludge, however, presents a significant problem. In aerobic treatment, at least 40-60% of the input chemical oxygen demand (COD) ends up in the unstabilized sludge (Olthof and Oleszkiewicz, 1982).

Anaerobic metabolism, on the other hand, yields little energy for growth. The rate of growth of an anaerobic microorganism is thus slow and only a small portion of the waste is converted to new microbial cells (McCarty, 1964; Hobson *et al.*, 1974). As a result, the volume of residual microorganisms to be further disposed of is much less than in an aerobic system. The residual sludge production from an anaerobic process is equivalent to only 3-10% of the incoming COD load (Oleszkiewicz and Olthof, 1982). The residual sludge is also stable and odorless and does not present problems with disposal (Hobson *et al.*, 1974).

In anaerobic treatment, the majority of the degradable organic waste is converted to methane gas. Such conversion to methane gas represents waste stabilization as methane is insoluble and spontaneously escapes from the waste stream. The methane gas produced is, of course, a valuable resource. Approximately 90% of the substrate energy is retained in the methane gas (Bryant, 1979). The energy value of the methane gas produced from well-designed systems is more than the energy required to operate the facility (Oleszkiewicz and Olthof, 1982). Figure 2.1 illustrates the comparison of energy use (or yields) in aerobic and anaerobic systems.

Anaerobic microorganisms, because of the low cellular yields, require relatively few inorganic nutrients. The anaerobic process is thus able to treat wastes that have levels of nutrients that would be far too small for aerobic treatment. Since the anaerobic treatment process does not require oxygen, the rate of supply of oxygen does not limit the process as it does in aerobic systems

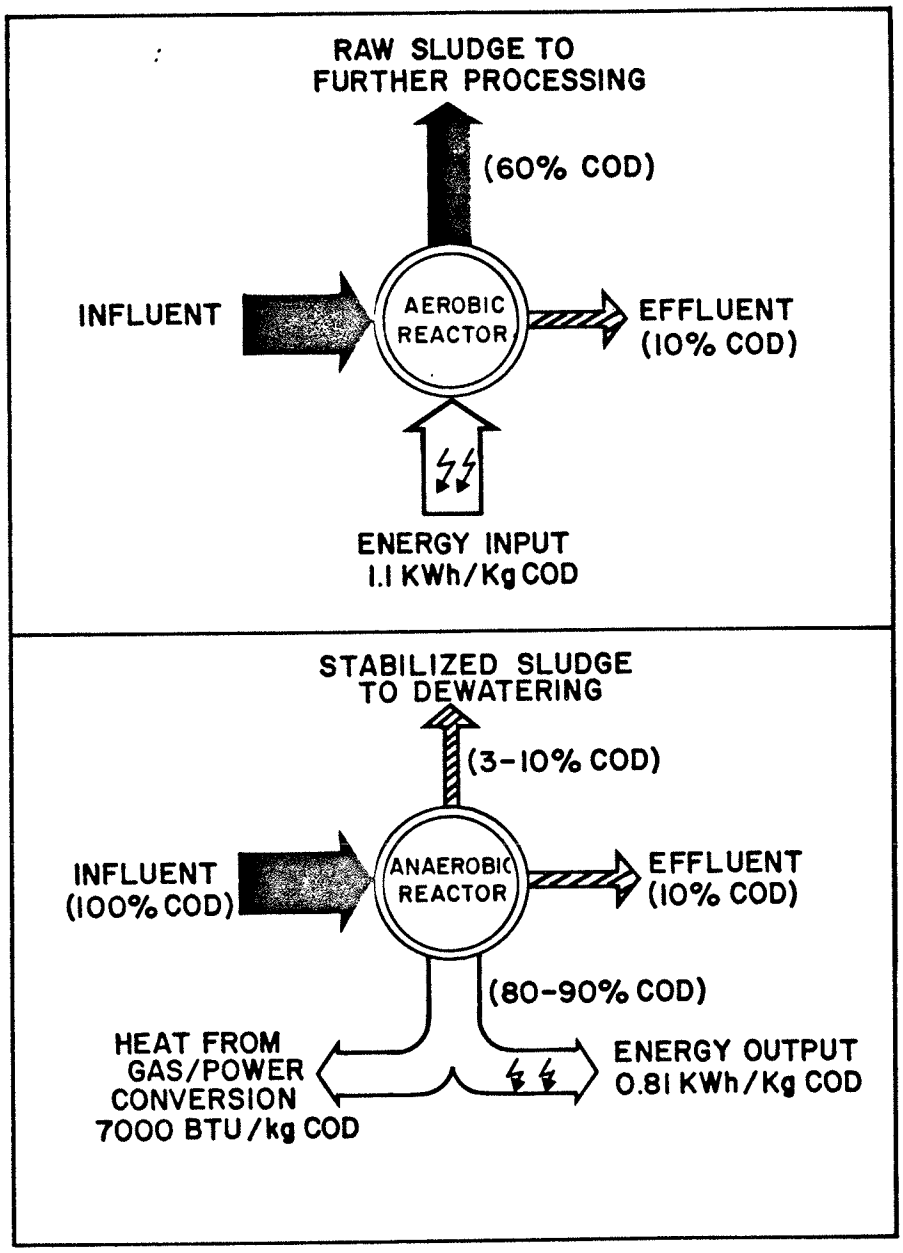

Figure 2.1 Comparison of energy use in aerobic and anaerobic processes based on 90% removal of chemical oxygen demand (COD) (Oleszkiewicz and Olthof, 1982)

(Kugelman and Jeris, 1981). Decisions regarding the reactor volume are consequently not influenced by the limitations of oxygen transfer. Loadings per unit volume of reactor for an anaerobic system can be higher than for aerobic systems. The required power inputs are also greatly reduced due to the fact that no oxygen is required.

Other benefits of anaerobic treatment include: ability to degrade complex compounds at high concentrations; absence of offensive odours due to the fact that anaerobic reactors are sealed; and effective reduction of pathogenic microorganisms.

Poor past experience with municipal sludge digestion has hindered the professional acceptance of anaerobic processes, as municipal sludge digesters have often been plagued with toxicity and instability problems (Oleszkiewicz and Olthof, 1982).

Table 2.4 presents some of the common misconceptions about anaerobic treatment contrasted with factual information about the process (Olthof and Oleszkiewicz, 1982).

Anaerobic processes, when designed properly and fully acclimated, can: withstand slugs of inhibitory or toxic organic compounds and eventually degrade them; operate at low temperatures; provide steady performance under unstable load conditions; and offer treatment efficiency as high as or better than aerobic systems using comparable or smaller reactor volumes and with lower operating cost (Oleszkiewicz and Olthof, 1982; Olthof and Oleszkiewicz, 1982).

It has been possible, with recent advancements in anaerobic technology, to anaerobically treat wastes with COD ranging from 300 to 100,000 mg/l at hydraulic retention times (HRT's) from 20 days down to 3 hours and at temperatures from 5° to 55°C (Oleszkiewicz and Olthof, 1982).

TABLE 2.4 Common Misconceptions About Anaerobic Treatment (Olthof and Oleszkiewicz, 1982)

Misconceptions	Facts
Anaerobic treatment processes are ...	In fact, anaerobic treatment ...
- applicable only to concentrated wastes, slurries and sludges.	- has been applied to streams with COD as low as 1,000 mg/L.
- not applicable to streams containing difficult-to-degrade organics.	- may be acclimated to degrade organic compounds, even some that aerobic treatment cannot degrade.
- not applicable to streams having no suspended solids.	- processes soluble wastes more quickly.
- slow, requiring 8-10 day retention times and, therefore, high reactor volume.	- requires hydraulic retention times comparable to those in aerobic treatment.
- energy-inefficient, as the reactors must be heated.	- generates surplus energy when treating streams with more than 3,000 mg/L COD.
- requires costly chemicals for process control.	- requires only 10-20% of the nutrients that aerobic treatment does, and controls alkalinity where needed by recycle.

The anaerobic process, while presenting several advantages over aerobic systems, also has some drawbacks which, coincidentally, are based on the same properties as some of the advantages. For example, oxidizing agents are toxic to some of the strict anaerobes and must be excluded from the anaerobic process. Even nitrate, a mild oxidizing agent, has been reported to be toxic (Kugelman and Jeris, 1981).

It was noted previously in this section that the cellular yield in the anaerobic process is low, which is a disadvantage because it interferes with the ability of the process to recover quickly after a toxic shock. However, when exposed to toxics, anaerobic bacteria attached to a substrate can become dormant and thereby protect themselves (Olthof and Oleszkiewicz, 1982). In some anaerobic systems, the dormant bacteria become active again once the toxic stress is removed and favourable conditions are reestablished.

The low growth rates of the anaerobic organisms also require the systems to be designed with relatively longer solids rentention time (SRT). This does not, however, necessarily mean a larger reactor volume, for the hydraulic residence time (HRT) need not be greater. Longer SRT's can be attained through recycle of sludge. Recycling solids back into a system will increase the SRT of the system without increasing its HRT.

The retention of sludge on fixed media will also enable longer SRT's to be maintained without increasing the HRT. Since the sludge is retained on the media and not washed off in the effluent, SRT's on the order of 100 days or more can be attained with short HRT's.

Another disadvantage of the anaerobic system relates to the inherent inability of the anaerobic bacteria to degrade various species of organic compounds (Kirsch and Sykes, 1971). Many synthetic organic chemicals are degraded slowly,

if at all by anaerobic processes, thereby limiting the applicability of these systems.

The anaerobic process is never entirely complete and in most cases the effluent from an anaerobic system may not be suitable for direct discharge. The effluent generally contains enough soluble biodegradable organic matter to warrant further treatment before its final discharge.

Finally, the capital costs of anaerobic reactors can be high because of the requirement for heat exchange. In order to maintain the reactors in the optimal temperature range, especially in the colder climates, heat must be added to the system and the need to heat the process requires the initial outlay of funds for the appropriate equipment, such as a boiler, heat exchanger, associated piping and a control system. Nevertheless, the merits of the anerobic process generally outweigh the drawbacks, especially when high strength wastes are being treated.

2.2.3 Microbiology and Biochemistry of Anaerobic Fermentation

The combined and coordinated metabolic activity of diverse genera of obligate and facultative anaerobic bacteria is responsible for the anaerobic degradation of organic matter. The coordinated activity of these different bacterial groups as a whole ensures process stability during anaerobic fermentation (Zeikus, 1980).

In accordance with the present knowledge of the microbiology and biochemistry of anaerobic fermentation, the effective conversion of complex organic matter to methane proceeds according to the scheme depicted in Figure 2.2 (McInerney *et al.*, 1980). The responsible microbial population is comprised of several major trophic groups of bacteria, each with a different carbon catabolizing function. At present, at least four distinct trophic groups are recognizable (Figure 2.2). The microbial population includes: hydrolytic bacteria

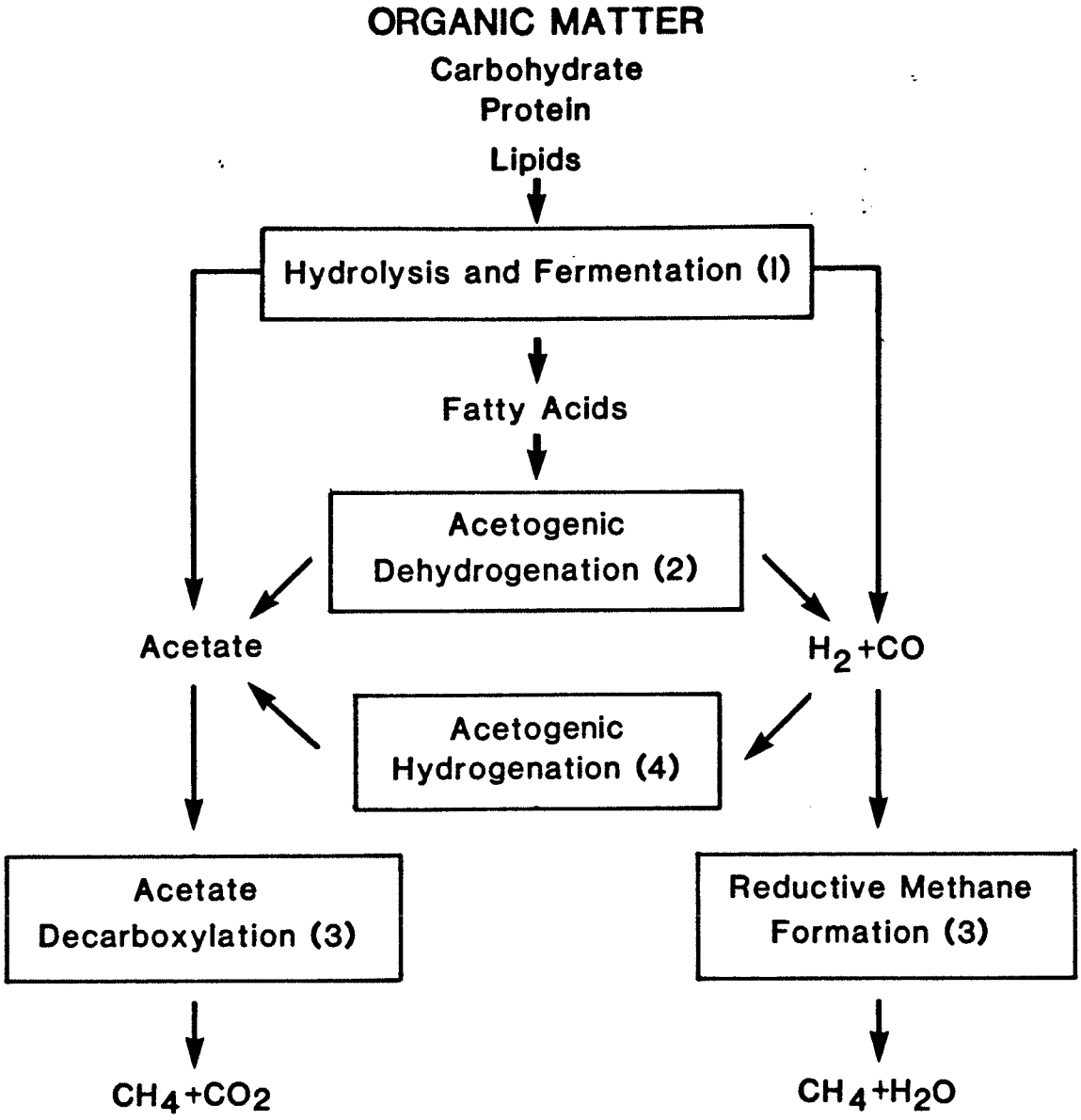

Figure 2.2 Idealized scheme for the complete anaerobic degradation of organics (McInerney *et al.*, 1980). (1. The hydrolytic bacteria; 2. The obligate hydrogen-producing acetogenic bacteria; 3. The methanogenic bacteria; 4. The homoacetogenic bacteria.)

(group 1); hydrogen-producing acetogenic bacteria (group 2); methanogenic bacteria (group 3); and homoacetogenic bacteria (group 4) (Zeikus, 1980; McInerney *et al.*, 1980).

Hydrolytic bacteria (group 1) hydrolyze polymers such as polysaccharides, proteins, and lipid materials into lower molecular weight materials such as sugars. These lower molecular weight materials are then transported into the bacterial cells and fermented to a variety of end products such as acetate, longer-chain fatty acids, organic acids, alcohols, hydrogen, carbon dioxide, ammonia nitrogen, and sulphide (McInerney *et al.*, 1980; McInerney and Bryant, 1981). The hydrolytic bacteria possess the ability to produce and excrete hydrolyzing enzymes (exo-enzymes) (van Velsen, 1981). They are thus able to utilize high molecular weight compounds in both dissolved or undissolved form.

Hydrolytic bacteria are comprised of a very complex mixture of many bacterial species, consisting mostly of obligate anaerobes. Some facultative anaerobes such as streptococci and enterics may, however, be numerous in some ecosystems (McInerney and Bryant, 1981). Hydrolytic bacteria isolated from anaerobic digester populations are classified according to the type cf exo-enzymes produced. *Anaerovibrio lipolytica* (Hobson *et al.*, 1974) is a lypolytic bacteria present in digesters. Proteolytic bacteria identified in anaerobic processes include *Micrococcus* spp., *Clostridia* spp. (Hobson *et al.*, 1974), and *Eubacterium* (Zeikus, 1980). Cellulolytic bacteria include *Ruminococcus* spp., *Butyrivibrio fibrisolvens* (Hobson *et al.*, 1974), and *Clostridium thermocellum* (Ng *et al.*, 1977). Streptococci, according to Hobson and Shaw (1974), are the predominant hydrolytic bacteria in swine waste digesters.

The total population of hydrolytic bacteria in mesophilic sewage sludge, as documented by several investigators, amounts to 10^8 - 10^9 hydrolytic

bacteria per ml of mesophilic sewage sludge (Toerien and Siebert, 1967; Mah and Sussman, 1968; Kirsch, 1969) or 10^{10} - 10^{11} hydrolytic bacteria per gram of volatile solids (Toerien and Siebert, 1967; Kotzé *et al.*, 1968; Kirsch, 1969).

The obligate hydrogen-producing acetogenic bacteria (group 2) are essential to anaerobic degradation because they, as a group, degrade propionate and longer-chained fatty acids, alcohols, some aromatic and aliphatic organic acids produced by the hydrolytic bacteria (McInerney *et al.*, 1980; McInerney and Bryant, 1981). This group produces acetate, H_2 and, in the case of odd-numbered-carbon sources, CO_2 is also produced (McInerney and Bryant, 1981).

The H_2-producing acetogenic bacteria, because of their metabolic activity, form an intermediate group linking the fermentative and methanogenic stages of the anaerobic fermentation process. The acetogenic bacteria can grow only in the presence of hydrogen-utilizing bacteria (e.g., methanogens and sulphate-reducing bacteria) since the catabolic reactions are thermodynamically unfavourable unless the hydrogen partial pressure is kept extremely low (Zeikus, 1980; McInerney *et al.*, 1980). The degradation of ethanol becomes energetically favourable when the partial pressure of H_2 is below 0.15 atm while the degradation of butyrate and propionate is not energetically favourable unless the partial pressure of H_2 is below 2×10^{-3} atm or 9×10^{-5} atm, respectively (McInerney *et al.*, 1980). The partial pressure of H_2 has only to increase slightly to prevent the degradation of these compounds. Propionate catabolism would be the first reaction to be inhibited followed by butyrate then ethanol catabolism. In cocultured growth, the methanogens, because of their great affinity for H_2, provide thermodynamically favourable conditions for H_2-producing acetogenic bacteria by keeping the concentration of H_2 very low.

The classic example of a H_2-producing acetogenic bacterium is the 'S organism' isolated from "*Methanobacillus omelianskii*". M. *omelianskii* had been regarded as a methanogen and was thought to oxidize ethanol to acetate and reduce CO_2 to CH_4. This fermentation was, however, demonstrated by Bryant *et al.*, (1967) to be a syntrophic association of two distinct bacterial species, viz., the 'S organism' and the *Methanobacterium* strain MOH. The nonmethanogenic "S organism' catabolizes ethanol to acetate and H_2 while the methanogen (*Methanobacterium* strain MOH) utilized the H_2 to reduce CO_2 to CH_4.

Although not taxonomically identified, the 'S organism' is presently the only physiologically well characterized H_2-producing acetogenic bacterium (Zeikus, 1980). Populations of 4.2×10^6 H_2-producing acetogenic bacteria per ml of sewage sludge have been reported (Zeikus, 1980).

The methanogenic bacteria (group 3) are a philogenetically unique group of bacteria that are composed of many species with very different cell morphology. All species studied so far have a similar and peculiar energy metabolism which is as yet not fully understood (Zeikus, 1977; Bryant, 1979; McInerney and Bryant, 1981). The methanogens produce methane from acetate and unicarbon compounds such as hydrogen/carbon dioxide (carbon monoxide) mixtures, methanol, formate, and methylamine (Balch *et al.*, 1979). The methanogens, owing to their ability to produce a reduced gaseous end product, methane, are the "key" organisms in the anaerobic fermentation process. These organisms provide thermodynamically favourable conditions for the nonmethanogenic organisms. Without this unique group of microorganisms, effective degradation of the total organic materials would cease due to the accumulation of nongaseous, reduced fatty acid and alcohol products of the

fermentative bacteria and other H_2-consuming species (McInerney *et al.*, 1980; McInerney and Bryant, 1981).

The methanogenic microorganisms are characterized by their relatively slow growth rate, their requirement of strictly anaerobic environments for growth (redox potential below -300 mV), and their utilization of a narrow range of substrate as energy sources. Yet, the methanogens appear to show more microbiological diversity than the other three trophic groups associated with the anaerobic fermentation process. Balch *et al.* (1979) proposed a new taxonomic scheme for methanogenic bacteria, based on comparative cataloging of the 16S ribosomal RNA molecule and substrates used for growth and methanogenesis (Table 2.5). The new taxonomic scheme classifies the methanogenic bacteria in three orders, viz., *Methanobacteriales*, *Methanococcales*, and *Methanomicrobiales*. The three orders are phylogenetically as distinct from one another as gram-positive sporing bacilli are from gram-negative enteric bacteria (McInerney and Bryant, 1981).

The metabolic interactions between the methanogenic and nonmethanogenic species are of great importance to the anaerobic fermentation process. The methanogens serve as bioregulators of process stability and activity. As shown in Table 2.6, the three most significant functions performed by methanogens are: proton regulation, electron regulation, and nutrient regulation (Zeikus, 1980).

Proton regulation is the most important ecological function of methanogens in anaerobic ecosystems. The methanogens, via acetic acid catabolism, remove toxic protons from the anaerobic ecosystem and maintain the optimal pH range for growth of representative species in all four trophic groups associated with anaerobic fermentation. Several facts support the notion

TABLE 2.5 Proposed taxonomic scheme of Balch *et al.* (1979) for methanogenic bacteria based on comparative cataloging of the 16S ribosomal RNA and substrates used for growth and methanogenesis

	Type strain	Former designation	Substrates for growth and CH_4 production
Order I. *Methanobacteriales* (type order)			
Family I. *Methanobacteriaceae*			
Genus I. *Methanobacterium* (type genus)			
1. *Methanobacterium formicicum* (neotype species)	MF	*Methanobacterium formicicum*	H_2, formate
2. *Methanobacterium bryantii*	M.o.H.	*Methanobacterium* sp. strain M.o.H.	H_2
Methanobacterium bryantii strain M.o.H.G.		*Methanobacterium* sp. strain M.o.H.G.	H_2
3. *Methanobacterium thermoautotrophicum*	H	*Methanobacterium thermoautotrophicum*	H_2
Genus II. *Methanobrevibacter*			
1. *Methanobrevibacter ruminantium* (type species)	MI	*Methanobacterium ruminantium* strain MI	H_2, formate
2. *Methanobrevibacter arboriphilus*	DHI	*Methanobacterium arbophilicum*	H_2
Methanobrevibacter arboriphilus strain AZ		*Methanobacterium* sp. strain AZ	H_2
Methanobrevibacter arboriphilus strain DC		*Methanobacterium* strain DC	H_2
3. *Methanobrevibacter smithii*	PS	*Methanobacterium ruminantium* strain PS	H_2 , formate
Order II. *Methanococcales*			
Family I. *Methanococcaceae*			
Genus I. *Methanococcus*			
1. *Methanococcus vannielii* (neotype species)	SB	*Methanococcus vannielii*	H_2, formate
2. *Methanococcus voltae*	PS	*Methanococcus* sp. strain PS	H_2, formate
Order III. *Methanomicrobiales*			
Family I. *Methanomicrobiaceae* (type family)			
Genus I. *Methanomicrobium* (type genus)			
1. *Methanomicrobium mobile* (type species)	BP	*Methanobacterium mobile*	H_2, formate
Genus II. *Methanogenium*			
1. *Methanogenium cariaci* (type species)	JR1	Cariaco isolate JR1	H_2, formate
2. *Methanogenium marisnigri*	JR1	Black Sea isolate JR1	H_2, formate
Genus III. *Methanospirillum*			
1. *Methanospirillum hungatii*	JF1	*Methanospirillum hungatii*	H_2, formate
Family II. *Methanosarcinaceae*			
Genus II. *Methanosarcine* (type genus)			
1. *Methanosarcina barkeri* (type species)	MS	*Methanosarcina barkeri*	H_2, CH_3OH, CH_3NH_2, acetate
Methanosarcina barkeri strain 227		*Methanosarcina barkeri* strain 227	H_2, CH_3OH, CH_3NH_2, acetate
Methanosarcina barkeri strain W		*Methanosarcina barkeri* strain W	H_2, CH_3OH, CH_3NH_2, acetate

TABLE 2.6 Role of Methanogen as Bioregulators of the Anaerobic Fermentation Process (Zeikus, 1980)

Function performed		Metabolic reaction	Process significance
I.	Proton regulation	$CH_3 COO^- + H^+ \quad CH_4 + CO_2$	1. Removes a toxic metabolite 2. Maintains pH
II.	Electron regulation	$4H_2 + CO_2 \quad CH_4 + 2H_2O$	1. Creates favourable conditions for metabolism of certain metabolites 2. Prevents accumulation of some toxic metabolites 3. Increases metabolic rates
III.	Nutrient regulation	Excretion of growth factors	1. Stimulates growth of heterotrophs

that proton regulation is the ecologically most important function of methanogenic bacteria in anaerobic systems. Acetate is an important intermediate product of the anaerobic fermentation process since the majority of the methane produced during anaerobic fermentation is derived from this volatile acid intermediate. Also, H_2 catabolism by both methanogenic and acetogenic bacteria is inhibited by high proton concentrations. Hydrogen catabolism by methanogenic bacteria leads to electron regulation. This process creates thermodynamically favourable conditions for the catabolism of multicarbon compounds (e.g., alcohols, fatty acids, aromatics) by hydrogen-producing acetogenic bacteria, and increases substrate utilization by hydrolytic bacteria. Nutrient regulation via synthesis and excretion of organic growth factors by certain methanogens is not well documented. This process appears, however, to stimulate the growth of heterotrophs (Zeikus, 1980).

In mesophilic sewage sludge, populations of 10^8 methanogenic bacteria per ml have been reported (Zeikus, 1980). Four genera of hydrogen-oxidizing methanogens, viz., *Methanobacterium* , *Methanospirillum* , *Methanosarcina*, and *Methanococcus* , have been reported to be present at populations of 10^6 - 10^8 per ml of sludge (Smith, 1966).

The hydrogen-consuming acetogenic or homoacetogenic bacteria (group 4) display a mixotrophic metabolism and are capable of catabolizing unicarbon compounds (e.g., hydrogen/carbon dioxide mixtures) or hydrolyzing multicarbon compounds (e.g., sugars). The homoacetogenic bacteria reduces CO_2 to acetate and sometimes to acids as butyrate using H_2 as the electron donor (McInerney and Bryant, 1981). Although the homoacetogenic bacteria can produce acetate and longer chain volatile fatty acids from hydrogen/carbon dioxide mixtures, the methanogenic bacteria appear to successfully out-compete

them for hydrogen in the gastrointestinal environment (Prins and Lankhorst, 1977).

The exact role of hydrogen-consuming acetogenic bacteria in the anaerobic fermentation process is not yet clear. Mah *et al.* (1976) have suggested that the homoacetogenic bacteria, because of their catabolism of hydrogen/carbon dioxide mixtures, increase the significance of acetate as an immediate methane precursor. The net result of homoacetogen metabolism is, according to Zeikus (1980), the maintenance of low hydrogen partial pressures, thus contributing to process stability (van Velsen, 1981).

The only recognized genera of hydrogen-oxidizing homoacetogenic bacteria are *Clostridium* and *Acetobacterium* (Zeikus, 1980). Populations of 10^5 - 10^6 homoacetogenic bacteria per ml have been reported in sewage sludge (Ohwaki and Hungate, 1977; Braun *et al.*, 1979).

2.2.4 Simplified Scheme for the Anaerobic Fermentation Process

The above mentioned scheme for the anaerobic fermentation process describes the process on the basis of four diverse trophic groups of bacteria found in anaerobic fermentation populations. As the metabolism of each of these four groups of bacteria is highly dependent upon the others, the above scheme is very effective for understanding the anaerobic fermentation process. This scheme, because of its complexity, is not, however, satisfactory for describing the course of the anaerobic process from routine analyses.

A simplified scheme, shown in Figure 2.3, for describing the anaerobic degradation of complex, undissolved substrates has been proposed by van Velsen (1981). According to the proposed scheme, the anaerobic fermentation process is divided into three stages, viz., hydrolysis of undissolved organic compounds,

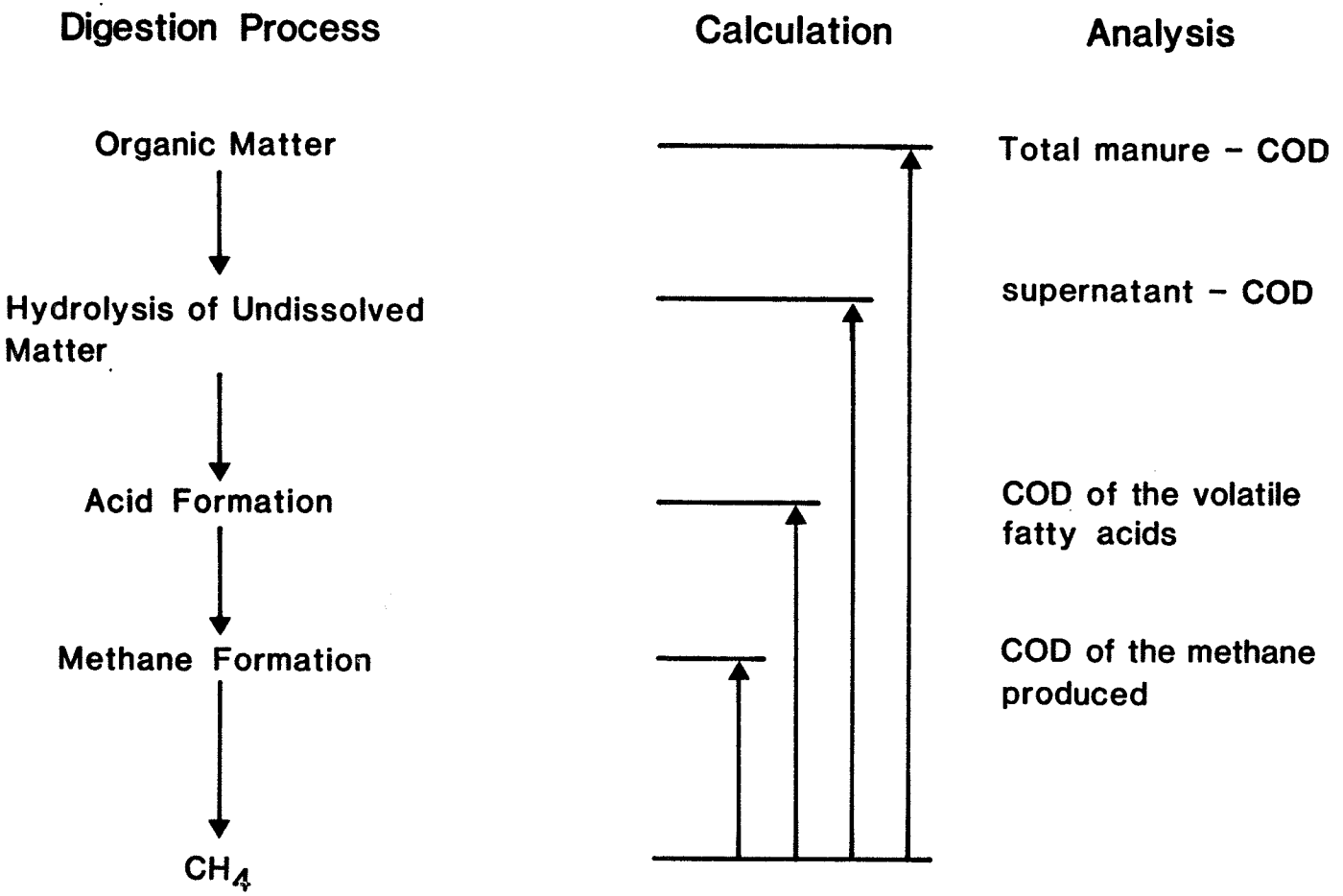

Figure 2.3 Simplified scheme for describing the anaerobic degradation of organic material on the basis of chemical routine analysis (van Velsen, 1981)

acid formation, and methane formation. The degree of conversion in each distinct stage can be roughly estimated from routine chemical analyses, by expressing the analytical data in terms of chemical oxygen demand (COD).

In van Velsen's scheme, hydrolysis is limited to the liquefaction of undissolved organic compounds and solely indicates the net activity of the hydrolyzing exo-enzymes produced by the hydrolytic bacteria (van Velsen, 1981). The extent to which the input COD is converted to volatile fatty acids is indicated by the acid formation stage. For this purpose, van Velsen (1981) assumed (a) that methane is only formed from end products of hydrolytic and acetogenic bacteria, and (b) that the volatile fatty acids are the predominant intermediates of the anaerobic fermentation process. The portion of the input COD which is ultimately converted to methane is indicated by the methane formation stage. Methane formation thus expressed is a measure of the overall process efficiency under the conditions studied (van Velsen, 1981).

The above simplified scheme provides a means to gain insight in the conversion degree of the distinct stages of the anaerobic process under the circumstances studied. By no means, however, do the three distinct stages coincide with the previously discussed trophic groups of bacteria. In consequence, conclusions drawn from the simplified scheme do not necessarily demonstrate the activity of the four trophic groups of bacteria associated with the anaerobic fermentation process (van Velsen, 1981).

CHAPTER 3

EXPERIMENTAL APPROACH

3.1 SCOPE OF STUDY

In accomplishing the objectives outlined in Chapter 1, a study was conducted in the laboratory over a 107 day period. Table 3.1 outlines the parameters measured under the conditions investigated. This study was conducted to assess the potentials of anaerobic treatment to reduce the pollutional characteristics of dilute swine slurry and to produce usable energy from such wastewaters. This information would be of value to many intensive swine producers. The methods and materials utilized to achieve the previously mentioned objectives are outlined in this chapter.

3.2 MATERIALS AND METHODS

3.2.1 Apparatus

Laboratory experiments were conducted in four identical anaerobic reactors. The laboratory reactors were glass aspirator bottles, having approximately 4.5 litre capacity. Each of the four aspirator bottles was sealed with a number 10 rubber stopper. A 250-ml separatory funnel was fitted into each stopper. The separatory funnel served as the feed inlet. Tygon tubing was attached to the funnel to extend the feed tube below the liquid surface. The second hole in the stopper served as the port for gas outlet. To ensure an airtight seal, a silicon rubber compound was applied to all glass-rubber interfaces.

The volume of gas produced daily from each reactor was determined by liquid displacement in graduated plexiglass collection cylinders. The liquid displacement system consisted of four calibrated gas collection cylinders connected

TABLE 3.1 Scope of Study - Environmental Conditions Investigated and Parameters Monitored

Environmental Conditions

Hydraulic retention time	15, 20, 25 days
Influent substrate concentration	13.2 - 14.2 g VS/l 25.7 - 27.3 g COD/l
Organic loading rate	0.53 - 0.95 kg VS/m^3-day 1.03 - 1.82 kg COD/m^3-day
Temperature	35°C

Parameters Monitored

Total VS removal
Total COD removal
Effluent total solids
Effluent total VS
Effluent total COD
pH
Alkalinity
Gas production
Composition of gas (% methane)
Total Kjeldahl Nitrogen
Ammonia nitrogen

in series to two reservoirs. The larger reservoir was used to collect the displaced liquid while the smaller reservoir was used to obtain gas measurements at atmospheric pressure and to level the volume of the displacement liquid in the gas collection cylinders. A piezometric tube was connected to the system. Tygon tubing was used to connect the reservoirs, gas collection cylinders, and reactors. All glass-rubber interfaces throughout the gas collection system were sealed with a silicon rubber compound. The displacement liquid was a saturated sodium chloride solution containing 5% sulphuric acid and methyl orange for color. Figure 3.1 presents a schematic diagram of the anaerobic fermentation system utilized in the study.

A mesophilic temperature of 35°C was maintained in each reactor. This was accomplished by placing the reactors in a water bath which was maintained at the desired temperature by a thermostatically controlled water heater.

3.2.2 Materials

The solids-separated swine waste slurry used in the experiments was obtained from Blue Bell Farms Ltd. which operates a 2,400 sow farrow-to-finish hog operation producing approximately 40,000 market hogs annually (Anon., 1982). Blue Bell Farms utilizes an open gutter flush system to handle the waste generated in the barns. The combined feces and urine slurry from the barns is pumped to a central manure handling facility. Here the slurry is screened using a Sweco vibrating separator. The screened solids are hauled away while the liquid fraction is recycled to flush feces and urine from the barns. Excess liquid is pumped

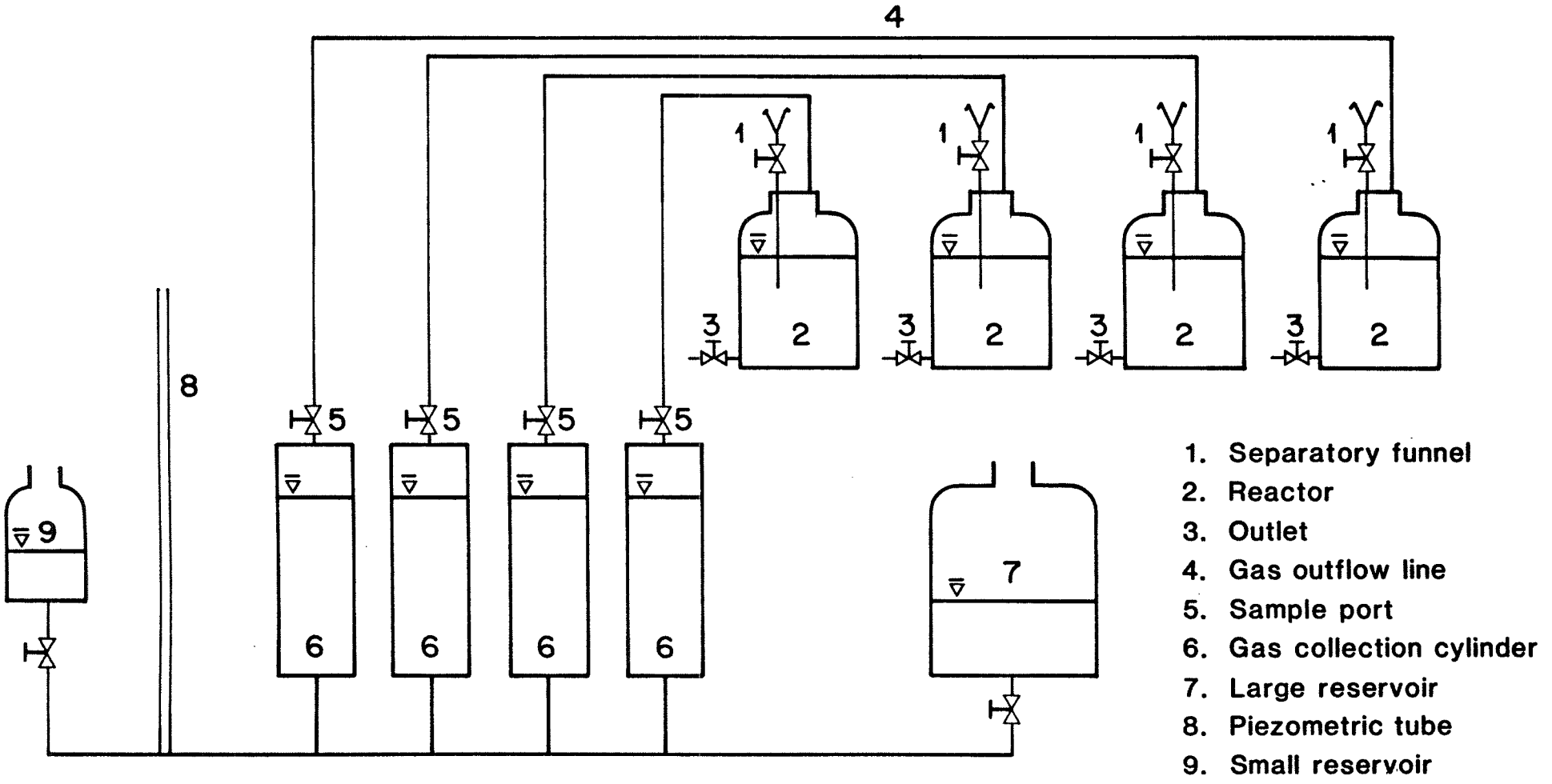

Figure 3.1 Schematic diagram of the anaerobic fermentation system utilized in this study.

to a storage lagoon.

The liquid fraction of the swine slurry was collected directly from the vibrating screen separator. The experiments were conducted with two different batches of the solids-separated swine slurry, which were stored at 4°C to minimize degradation.

3.2.3 Program of Experimentation

The laboratory reactors contained 4 litres of fermenting material and were operated on a semi-continuously fed basis. They were initially seeded with screened anaerobically fermented swine waste from previously operating reactors. The seed material was screened with cheesecloth to remove undigested grains and large solids. Each of the reactors initially contained two litres of fresh solids-separated swine slurry and two litres of the seed material.

All reactors were fed daily during the course of the study. The feeding procedure was as follows. After determination of the gas production, each reactor was vigorously mixed by hand to provide a homogeneous effluent, and the prescribed volume (determined by the retention time) of the mixed reactor contents was removed from the outlet at the bottom of the reactor. An equal quantity of the separated liquid fraction of swine slurry was then added to the reactor through the separatory funnel. The reactor was then manually mixed again to ensure a uniform distribution of the feed. Because of this procedure, the reactors can be considered to have been completely mixed anaerobic fermenters. Prior to feeding, the required volume of solids-separated swine slurry was preheated to the exact reactor temperature to prevent thermal shock to the microorganisms. Since the contents of each reactor were completely mixed during wasting and feeding, and since no recycle of solids was employed,

the solids retention time (SRT) was equal to the hydraulic retention time (HRT).

The determination of the daily gas production from each reactor was carried out as follows. To measure the volume of gas collected in the cylinders, the valve on the large reservoir was closed and the clamp on the tubing leading to the small reservoir was opened. The liquid level in the small reservoir was then brought to the same elevation or atmospheric pressure as in the piezometric tube, at which point the gas pressure inside the collection cylinders was equal to atmospheric. This procedure enabled gas measurements to be made at atmospheric pressure. When the system was collecting gas from the reactors, the tubing to the small reservoir was clamped and raised to a level above the reactors.

The four reactors were initially fed at low rates to allow the requisite bacterial flora to develop. The loading rate was increased in increments until the desired loading rate for each reactor (based on retention time) was achieved. A period equal to one retention time was allowed to pass for each reactor before collection of data proceeded. Reactor A was operated at a 15-day SRT during the course of the study. Reactors B_1 and B_2 were maintained at a 20-day SRT and reactor C at a 25-day SRT.

The total biodegradability of the liquid fraction of swine slurry was determined in batch experiments. The biodegradability determinations were conducted on the fermenting contents of the four semi-continuous fermenters. The four reactors were maintained at a temperature of 35°C and manually mixed on a daily basis. Samples were collected intermittently over a period of 37 days and analyzed for total volatile solids. This data was then analyzed and the total biodegradability of the liquid fraction of swine slurry was determined.

3.3 SAMPLING AND ANALYSIS PROGRAM

The four semi-continuous reactors were set as described in the previous section and seeded with screened anaerobically fermented swine waste. After 35 days of operation to allow the requisite bacterial flora to develop, data was collected for the three solids retention times investigated.

The prescribed volume of fermenting material wasted from each reactor was analyzed to determine thereactor's operating characteristics. The analyses were carried out according to the time schedule shown in Table 3.2. Gas composition samples were taken through the sample ports of each gas collection cylinder.

All analyses, except for the Total Kjeldahl Nitrogen (TKN), were conducted immediately after withdrawal from the reactors so that no sample storage technique was utilized. Samples for TKN were acidified to pH 2.0 and refrigerated at 4°C. They were analyzed the following day.

All glassware used was washed with hot soapy water and rinsed with tap water. The glassware was then washed with concentrated chromic-sulphuric acid cleaning solution, rinsed three times with tap water, three times with distilled water, and dried at 103°C in a drying oven.

3.4 ANALYTICAL TECHNIQUE

3.4.1 pH

The pH of the contents of each reactor was determined by the glass electrode method given in *Standard Methods* (Anon., 1980). A Radiometer Type PHM 29 b pH meter was used. The reference electrode was a Fisher Scientific calomel reverse-sleeve reference electrode (cat. no. 13-639-61) while the pH electrode was a Fisher Scientific universal glass pH electrode (cat. no.

TABLE 3.2 Time Schedule of the Analytical Measurements

Analysis Parameter	Influent Feed	Daily	Effluent Every 5 days	Every 10 days	Gas Daily
Total Solids	X		X		
Total Volatile Solids	X		X		
Total COD	X		X		
pH	X	X			
Alkalinity	X	X			
Total Gas Measurement					X
Gas Composition					X
Total Kjeldahl Nitrogen	X			X	
Ammonia Nitrogen	X			X	
Temperature		X			

13-639-3). Before each set of pH analyses was begun, the pH meter was calibrated with a set of standard buffer solutions at the temperature of the samples to be analyzed. The relative accuracy of the pH meter was ±0.03 pH units.

3.4.2 Alkalinity

Alkalinity was determined using method 403.4c, "Potentiometric Titration to Preselected pH", as described in Standard Methods (Anon., 1980). The samples were titrated to a pH 4.5 end point. Duplicates were analyzed for each sample.

3.4.3 Total and Volatile Solids

The total and volatile solids were determined using method 209 G, "Volatile and Fixed Matter in Nonfiltrable Residue and in Solid and Semisolid Samples", as described in Standard Methods (Anon., 1980). Triplicates of each sample were analyzed immediately after collection.

3.4.4 Chemical Oxygen Demand

Chemical oxygen demand (COD) was determined by the colorimetric method described by Knechtel (1978). Knechtel's method involved placing samples in Kimax 25 x 150 mm culture tubes. After the reagents were added, the culture tubes were tightly capped with teflon-lined bakelite caps, inverted three times to thoroughly mix the contents, and placed in a 150°C oven for two hours to digest the oxygen demanding material. After cooling, the culture tubes were placed in a Bausch and Lomb Spectronic 20 spectrophotometer. Absorbance readings were taken at a 600 nm wavelength. Triplicates of an appropriate set of potassium acid phthalate standards were measured to obtain a calibration curve. The COD of the unknown samples were read against this calibration curve.

Each set of COD analyses consisted of three replicates of each sample and three of the standards.

3.4.5 Gas Analysis

Total gas production for each reactor was measured daily by liquid displacement as described in the previous part of this chapter. Determinations of methane were made using a Fisher-Hamilton model 29 gas partitioner. The model 29 incorporated a dual-column/dual-detector chromatographic system to separate and quantitatively measure carbon dioxide, oxygen, nitrogen, methane, and carbon monoxide.

The two chromatographic columns used in the instrument were made of aluminum. Column 1 was 6 feet long by $\frac{1}{4}$ inch in diameter, and filled with a packing of 30% Di-2-ethylhexylsebacate (DEHS) on 60-80 mesh Chromosorb P. Column 2 was $6\frac{1}{2}$ feet long by 3/4 inch in diameter and packed with 40-60 mesh activated Molecular Sieve 13X. The columns were operated at ambient temperatures, the detectors at 70°C. Helium was used as the carrier gas at 40 ml/min. Gas samples were introduced into the instrument by a 1 ml gas syringe.

3.4.6 Total Kjeldahl Nitrogen (TKN) and Ammonia Nitrogen (NH_3-N)

Total Kjeldahl Nitrogen (TKN) and ammonia nitrogen (NH_3-N) were determined using the KJELTEC System 1 developed by Tecator Inc. The KJELTEC System 1 employed the same analytical principles as outlined in Standard Methods (Anon., 1980). The TKN and NH_3-N measurements were determined using method 417 B, "Nesslerization Method (Direct and Following Distillation)", as described in Standard Methods (Anon., 1980).

Samples taken for TKN determination were acidified and stored at

4°C for no more than 36 hours prior to analysis. Samples for NH_3-N determination were analyzed within hours after being collected. Duplicates were analyzed for each TKN and NH_3-N sample.

CHAPTER 4

RESULTS

4.1 SUMMARY OF DATA

The results of the laboratory experiments with the liquid fraction of swine slurry together with the environmental conditions applied are summarized in Tables 4.1 to 4.3. The data presented in these tables represent the average values of the analytical results during the thirty-fifth through seventieth day of operation when the reactors were operating at steady state conditions. The overall stability exhibited by each of the reactors indicates the ability of a flow-through anaerobic reactor to treat dilute swine slurry under conditions that most likely would be encountered on a typical farm.

Figures 4.1 and 4.2 relate waste removal efficiency, effluent concentration and gas production to SRT on both a volatile solids (VS) and COD basis. A clear trend is shown in both figures for the three parameters. The percent removal and gas production increases while effluent concentration decreases with respect to SRT. This means that the longer the SRT, the greater the gas production relative to the available substrate.

Figure A.1 in Appendix A shows the daily gas production from each of the four reactors during the period of steady state operation. Reactor A produced more gas on a daily basis that the other reactors. This is to be expected as reactor A was operated on a 15-day SRT and more substrate was available to this reactor than to the other three reactors which were maintained at longer SRT's. pH and alkalinity were also monitored to serve as supportive indicators of fermenter malfunction. These data are presented in Figure A.2 of Appendix A.

TABLE 4.1 Physical Parameters

Parameter	Reactor			
	A	B_1	B_2	C
Detention Time, Days	15	20	20	25
Temperature, °C	35	35	35	35
Influent Concentration				
g TS/l	21.6	21.6	21.6	21.6
g VS/l	13.6	13.6	13.6	13.6
g COD/l	26.3	26.3	26.3	26.3
Effluent Concentration (mixed liquor consideration)				
g TS/l	15.7	15.1	15.2	14.9
g VS/l	8.4	8.0	8.0	7.6
g COD/l	14.3	13.5	13.1	12.1
Organic Loading Rate				
kg VS/m^3 - day	0.91	0.68	0.68	0.55
kg COD/m^3 - day	1.75	1.32	1.32	1.05

TABLE 4.2 Chemical Parameters

Parameter	Reactor			
	A	B_1	B_2	C
pH				
Influent	7.15	7.15	7.15	7.15
Effluent	7.55	7.55	7.58	7.61
Alkalinity, mg/l $CaCO_3$				
Influent	7600	7600	7600	7600
Effluent	9000	8900	9000	9100
Total Kjeldahl -N				
Influent mg/l -N	1980	1980	1980	1980
Effluent mg/l - N	1880	1790	1700	2015
Ammonia - N mg/l				
Influent	1395	1395	1395	1395
Effluent	1655	1620	1490	1580

TABLE 4.3 Summary of Results

Parameter	Reactor			
	A	B_1	B_2	C
SRT, Days	15	20	20	25
Total Organic Matter Removal				
Percent VS	38.32	40.91	41.06	44.07
Percent COD	45.41	48.65	49.99	54.16
Gas Production Rate				
m^3/kg VS added	0.44	0.45	0.47	0.54
m^3/kg VS removed	1.14	1.11	1.15	1.24
m^3/kg COD added	0.23	0.23	0.24	0.28
m^3/kg COD removed	0.50	0.48	0.49	0.52
Gas Composition (% CH_4)	69.35	68.94	68.78	68.86
Methane Production				
g COD/day (methane equivalent)	2.77	2.13	2.23	2.05
m^3/kg COD removed @ 35°C	0.34	0.33	0.34	0.36
m^3/kg COD removed @ STP	0.30	0.29	0.30	0.32

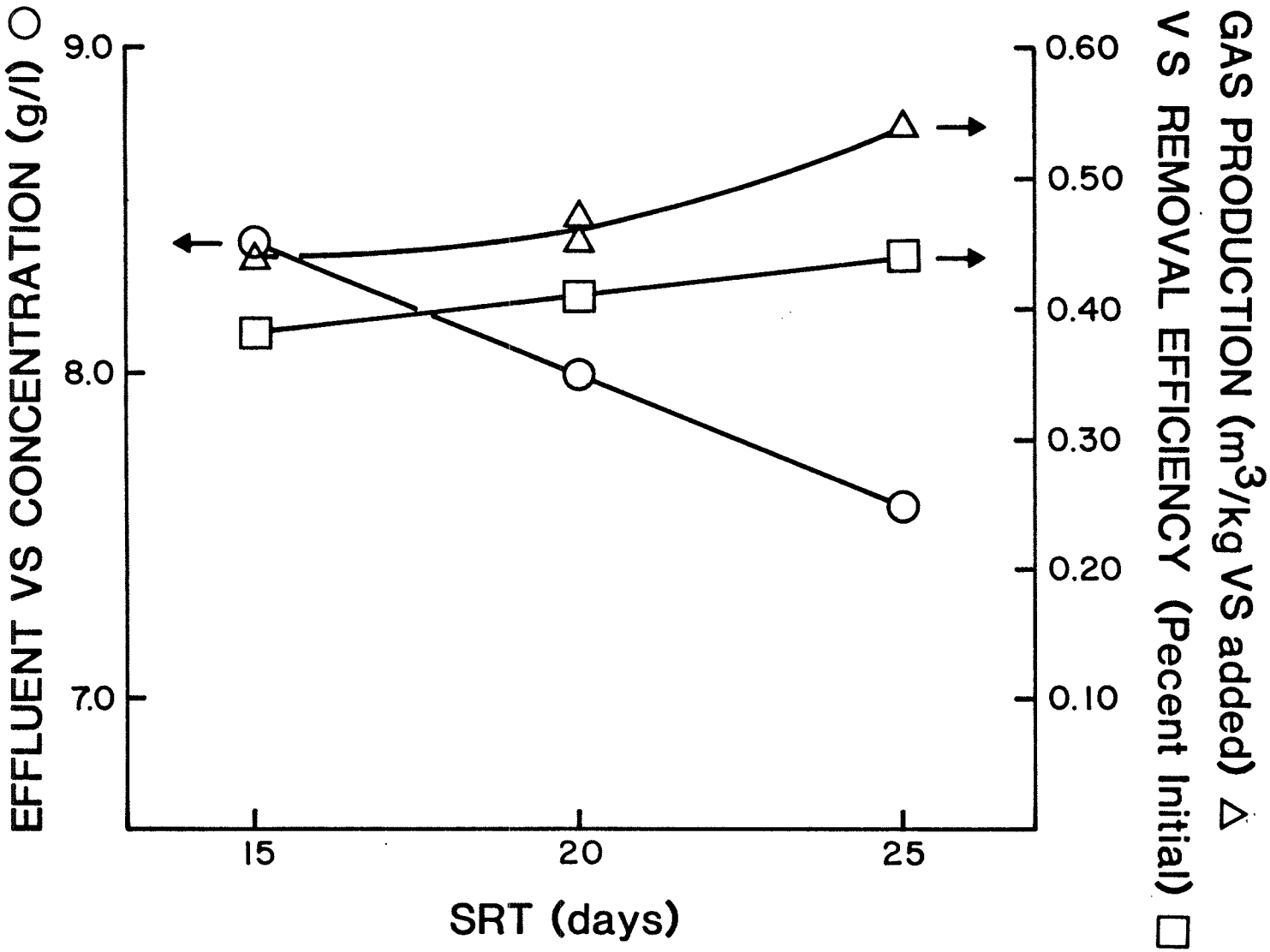

Figure 4.1 Effect of SRT on effluent VS concentration, VS removal efficiency and gas production rates.

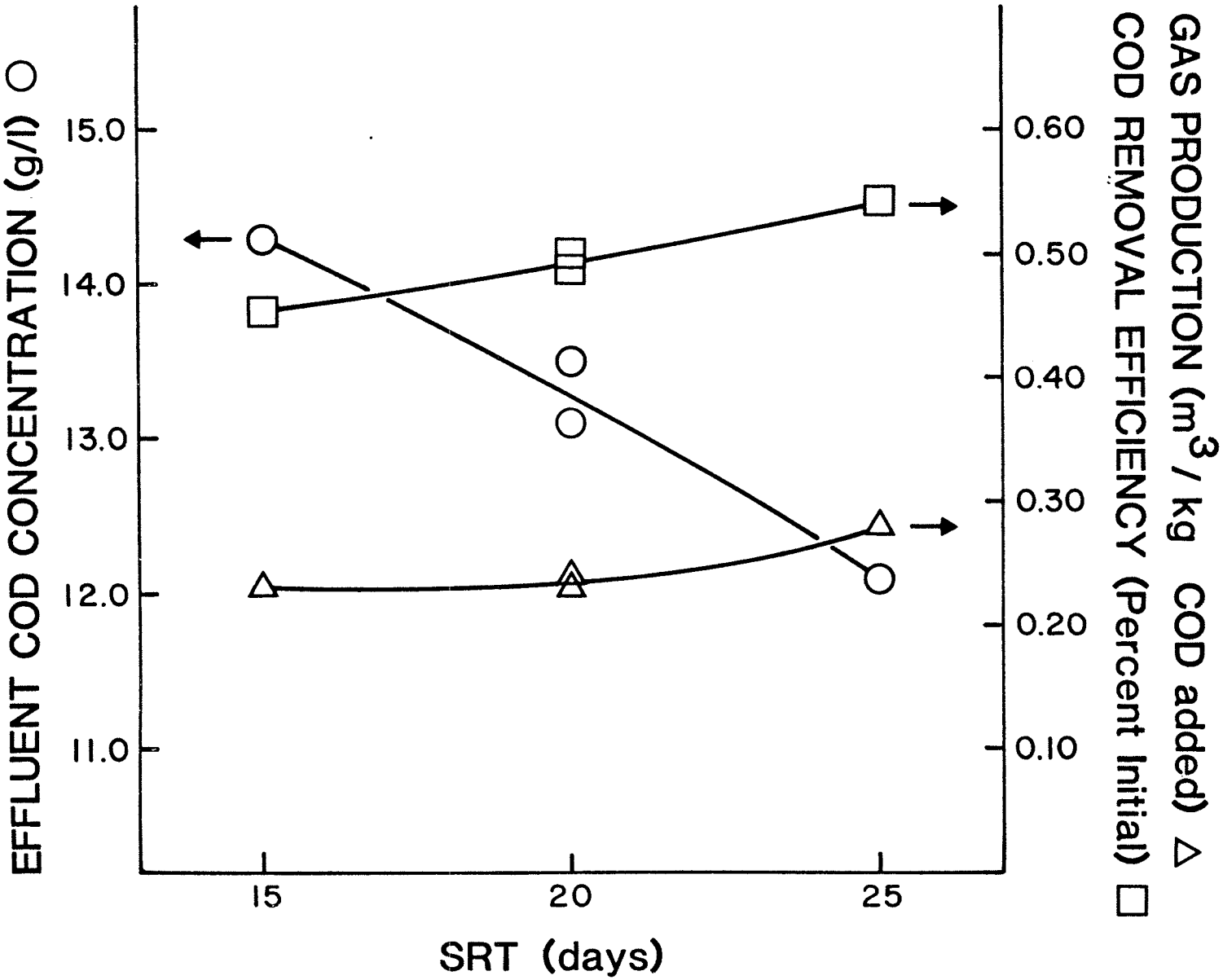

Figure 4.2 Effect of SRT on the effluent COD concentration, COD removal efficiency and gas production rates.

Figures 4.3 and 4.4 show daily methane production, in terms of its COD equivalent, as a function of the organic loading rate on both a VS and COD basis. Evidently, and not surprisingly, more methane was produced at the higher organic loading rates.

In order to convert the volume of methane produced to its COD equivalent, the ultimate oxygen demand of methane gas must first be calculated. Methane combines with oxygen to produce carbon dioxide and water as shown in the equation below:

$$CH_4 + 2O_2 \longrightarrow CO_2 + 2H_2O$$

According to the above equation, one mole of methane requires two moles of oxygen molecules for complete oxidation. Theoretically, on a molecular weight basis, the complete oxidation of one mole of methane requires 4 (15.9994) = 63.9976 grams of oxygen or COD. Since one mole of any gas at standard temperature and pressure (STP- 0°C, 1atmosphere) occupies a volume of 22.414 litres (Sawyer and McCarty, 1978), it follows then that (1 mole CH_4 / 63.9976 g COD) (22.414 l / 1 mole CH_4 @ STP) =

350.23 ml CH_4 @ STP / g COD

This relationship can be further modified for temperature effects by applying Charles' Law. According to Charles' Law, the volume of a given gas at constant pressure varies in direct proportion to the absolute temperature (Sawyer and McCarty, 1978). This relationship can be expressed as follows:

$$\frac{V_{STP}}{T_{STP}} = \frac{V_{Ta}}{T_a}$$

where $V_{S\,TP}$ = volume at standard temperature and pressure (0°C, 1 atmosphere)

V_{Ta} = volume at temperature "a"

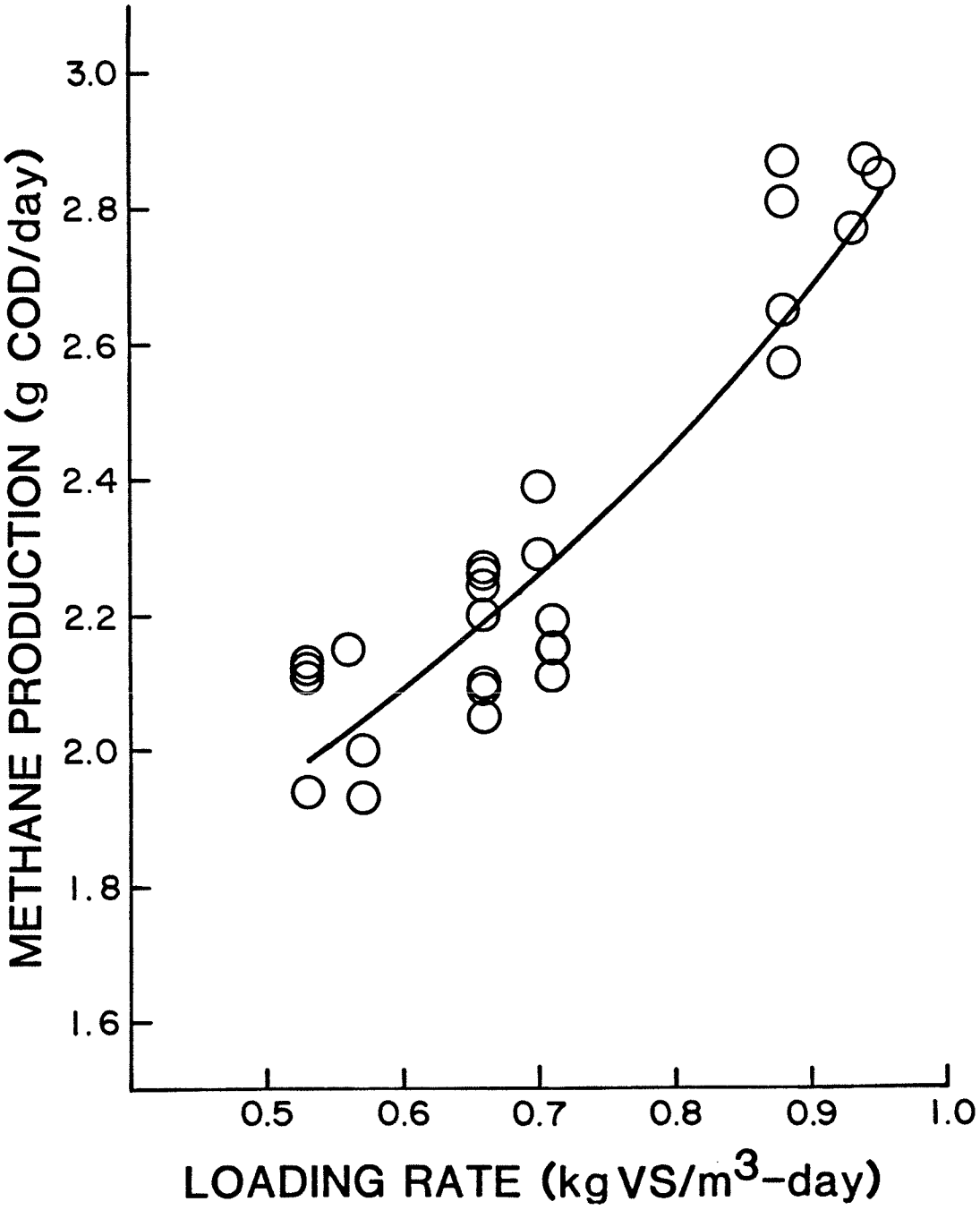

Figure 4.3 Daily methane production as a function of VS loading rate

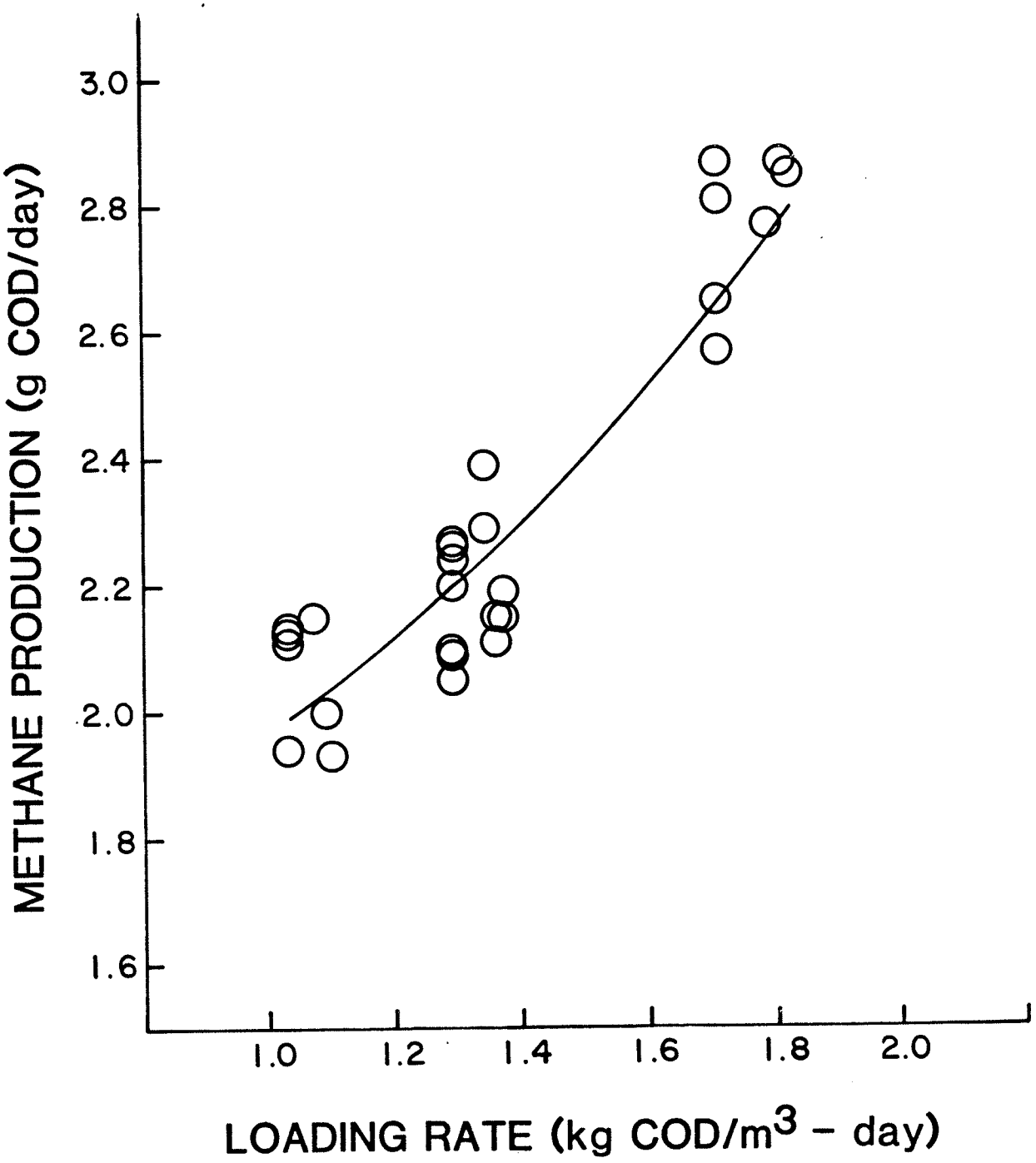

Figure 4.4 Daily methane production as a function of COD loading rate

T_{STP} = temperature at STP = 273.16°K

T_a = temperature "a" = $C_a°$ + 273.16°

Using the above relationship it was calculated that 1 gram of COD at 35°C is equal to 395.11 ml of CH_4. This quantity was used to convert the volume of methane produced to COD mass units.

4.2 CHEMICAL OXYGEN DEMAND BALANCES

Organic balances were evaluated for the four reactors during the period of steady state operation. These balances can be obtained by converting all the influent and effluent streams of each reactor to a COD equivalent basis and comparing the amount placed into each system with those quantities leaving. These balances are useful since they act as a cross checking mechanism and give additional support to the experimental results obtained.

Since anaerobic fermentation involves no external source of oxygen, a mass balance on the COD may be evaluated in the following manner: influent COD = efluent COD + gaseous methane production as COD.

Tables 4.4 to 4.7 show the COD balances for each of the four reactors during steady state operation. The columns in these tables are explained as follows. The "influent" column, expressed as g COD/day, is the product of the influent COD and the flow rate. Likewise, the "effluent" column was calculated by multiplying the effluent COD by the flow rate. The "CH_4 production" column was calculated by multiplying the daily gas production by the percentage of methane as determined by gas chromatography and then converting this quantity to its COD equivalent by using the following relationship which was derived previously in Section 4.1:

1 g COD = 350.23 ml CH_4 @ 35°C

TABLE 4.4 COD Balance for Reactor A

Influent (g COD/day)	Effluent (g COD/day)	CH_4 Production (g COD/day)	Total Out (g COD/day)	% Recovered*
7.28	3.81	2.85	6.66	91.5
7.23	3.95	2.87	6.82	94.3
7.15	3.84	2.77	6.62	92.6
6.85	3.81	2.57	6.38	93.1
6.85	3.73	2.81	6.54	95.5
6.85	3.76	2.87	6.63	96.8
6.85	3.87	2.65	6.52	95.2

* Mean = 94.1

TABLE 4.5 COD Balance for Reactor B_1

Influent (g COD/day)	Effluent (g COD/day)	CH_4 Production (g COD/day)	Total Out (g COD/day)	% Recovered*
5.46	2.64	2.19	4.83	88.5
5.42	2.68	2.15	4.83	89.1
5.36	2.72	2.29	5.01	93.5
5.14	2.66	2.05	4.71	91.6
5.14	2.68	2.09	4.77	92.8
5.14	2.72	2.05	4.77	92.8
5.14	2.78	2.10	4.88	94.9

* Mean = 91.9

TABLE 4.6 COD Balance for Reactor B_2

Influent (g COD/day)	Effluent (g COD/day)	CH_4 Production (g COD/day)	Total Out (g COD/day)	% Recovered*
5.46	2.70	2.15	4.85	88.8
5.42	2.66	2.11	4.77	88.8
5.36	2.72	2.39	5.11	95.3
5.14	2.50	2.20	4.70	91.4
5.14	2.60	2.24	4.84	94.2
5.14	2.64	2.24	4.88	94.9
5.14	2.58	2.26	4.84	94.2

* Mean = 92.4

TABLE 4.7 COD Balance for Reactor C

Influent (g COD/day)	Effluent (g COD/day)	CH_4 Production (g COD/day)	Total Out (g COD/day)	% Recovered*
4.42	1.87	1.93	3.8	86.0
4.37	1.94	2.00	3.94	90.2
4.29	1.98	2.15	4.13	96.3
4.11	2.00	2.12	4.12	100.2
4.11	1.89	2.13	4.02	97.8
4.11	1.89	2.11	4.00	97.3
4.11	1.95	1.94	3.89	94.6

* Mean = 94.6

The "total out" column is the sum of the effluent column and the CH_4 production column and represents the cumulative COD exiting the system. The last column, "% recovered", is the ratio of the COD recovered leaving the system to the COD entering the system multiplied by 100% (i.e., the ratio of the total out quantities to their corresponding influent quantities times 100%).

The mean values of COD recovered for reactors A, B_1, B_2 and C were 94.1%, 91.9%, 92.4% and 94.6% respectively. The high values obtained for the COD recovered indicate a good account of the organics entering and leaving the laboratory reactors. These high values also verify the reliability of the experimental results obtained in this study. A sample COD balance is shown in Appendix B.

4.3 BIODEGRADABILITY STUDY

A biodegradability study was initiated to determine the refractory fraction (R) of dilute swine slurry volatile solids. The refractory fraction of the organic volatile solids is defined as that portion of the initial quantity of volatile solids which is resistant to biological degradation over long periods of time and remains undegraded after the rate of degradation of the initial volatile solids has decreased to a very low level (Jewell *et al.*, 1978; Morris *et al.*, 1977).

The refractory fraction of the influent VS concentration of the dilute swine slurry was determined graphically, employing a method used by Anthonisen *et al.*, (1968), and Wood *et al.*, (1974) and modified by Morris (1976). The assumption in this method is that the biodegradable portion of the volatile solids will be destroyed as the SRT approaches infinity, leaving only the refractory fraction remaining. In this method the substrate is anaerobically fermented

in a batch reactor and samples are withdrawn at various intervals for VS analyses. The "R" of the substrate can then be determined by plotting S_1/S_0 versus $1/(S_0 \cdot SRT)$ where S_1 is the VS concentration of effluent samples taken at various SRT's and S_0 is the initial VS concentration of the substrate. This plot yields a linear relationship with the ordinate intercept being the refractory fraction, R.

Figure 4.5 presents the results of the biodegradability batch tests conducted on the effluents from the four laboratory reactors. Each batch reactor initially contained four litres of fermenting material. A sample volume of 150 ml was removed, after mixing, from each reactor periodically and analyzed for VS. As can be seen from the data presented in Figure 4.5, the refractory fraction, R, of the VS concentration of the solids-separated swine slurry ranged from 0.42 to 0.46, or 42 to 46% of the influent total volatile solids. The total biodegradable fraction of the swine slurry is thus 54 to 58% of the influent VS concentration at 35°C. It is felt that the differences in the respective values of "R" is due to sampling and experimental error and thus, a representative value of 0.44 was chosen for "R". The average biodegradable fraction of the solids-separated swine slurry anaerobically fermented at 35°C was 56% of the influent volatile solids concentration.

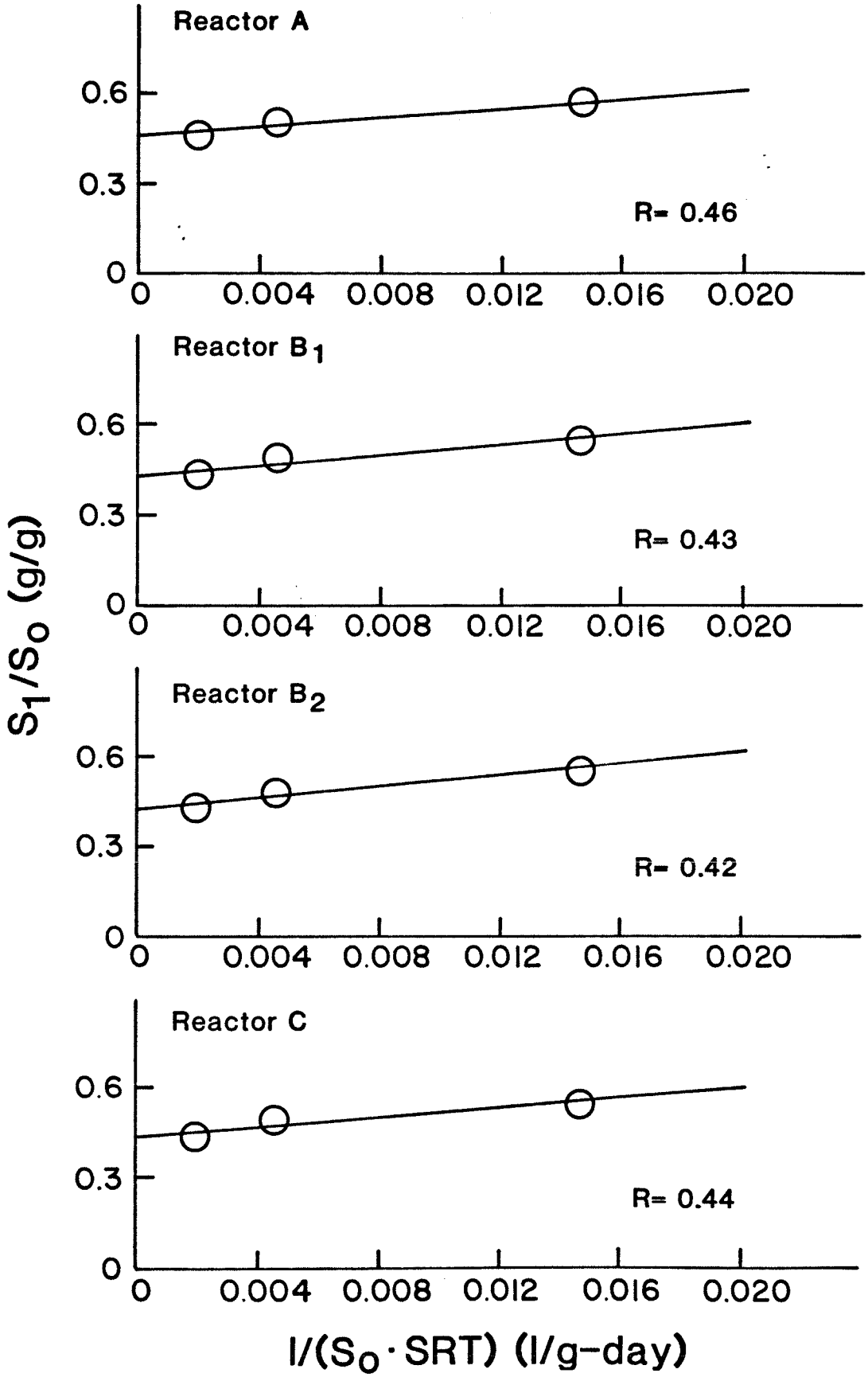

Figure 4.5 Graphical analysis of VS data to determine R

CHAPTER 5

DISCUSSION

5.1 PROCESS EFFICIENCY

Anaerobic fermentation of swine waste has received considerable attention, but there is a lack of evidence as to the minimum total solids (TS) that can be effectively digested. Summers and Bousfield (1976) reported that 2% TS was the minimum concentration that could be effectively digested without washout of digester bacteria. Oleszkiewicz and Koziarski (1982) proved that dilute, sieved, swine wastes with VS concentrations of 0.5% can be effectively digested in conventional flow-through anaerobic reactors without recycle. The results presented in Chapter 4 tend to support the claim by Oleszkiewicz and Koziarski (1982) that the flow-through anaerobic reactor without sludge recycle can effectively treat dilute swine slurries. The results shown in Chapter 4 indicate that the liquid fraction of swine slurry can be successfully fermented in a flow-through anaerobic reactor at SRT's of 15, 20, and 25 days at 35°C. Washout of digester bacteria was not a problem at these SRT's.

The total VS removal obtained in the laboratory experiments varied from 38.32% at a 15-day SRT to 44.07% at a 25-day SRT, which is lower than the VS removal of 60.9% at a 15-day detention time reported by Gramms *et al.* (1971) and of 53-62% at a 20-day detention time reported by Jeffrey *et al.* (1964). The total VS removals obtained in this study were, however, similar to the removals obtained by Kroeker *et al.* (1975). Kroeker *et al.* (1975) reported VS removals of 36% at a 15-day SRT and 44% at a 30-day SRT. The total COD removal obtained in this study ranged from 45.41% at a SRT of 15 days to 54.16% at a SRT of 25 days. Hobson *et al.* (1979) reported

COD removals of 49-53% while Gramms *et al.* (1971) reported a COD removal of 54.6% at a 15-day detention time. The differences in the removal of organics obtained in this study and with the values reported by other researchers should be attributed to the different process conditions, such as temperature and pH, as well as to differences in the composition of the swine wastes and in the analyses used. Furthermore, the results shown in Table 4.3 indicate that VS and COD removals tend to increase at increasing SRT's.

The gas production from the liquid fraction of swine slurry in well-established anaerobic reactors varied from 0.44 - 0.54 m^3/kg VS added. The gas yields obtained in this study are summarized in Table 5.1 together with the gas production rates attained by other researchers from swine wastes. Table 5.1 shows that the gas production rates obtained in this study are of the same order of magnitude as those reported in the literature.

Each of the four reactors produced gas with approximately the same percentage of methane. The results presented in Table 4.3 show that reactor C, which was operated at a SRT of 25 days, produced more methane relative to the available substrate than the other three reactors. On a daily basis, however, more methane was produced by reactor A, which was maintained at a 15-day SRT. Trade-offs between methane production and removal of organic matter appear to exist as indicated by the results shown in Table 4.3. Although total methane production was higher in reactor A than in C, the fraction of VS and COD removed was greater at the lower organic loading rates and the higher SRT. It is not clear from the data gathered in this study how much SRT could be decreased and/or the organic loading rate increased without adversely affecting process stability and methane production rates.

The theoretical carbon to methane conversion has been calculated

TABLE 5.1 Gas Production from Swine Waste

Temp. (°C)	Detention time (days)	Organic Load (kg VS/m^3-day)	Gas Production (m^3/kg VS added)	Reference
35	10 - 50	0.32 - 3.20	0.49 - 0.64	Taiganides *et al.* (1963)
35	20	2.41 - 3.05	0.37 - 0.54	Jeffrey *et al.* (1964)
32.5	10 - 15	1.92 - 3.85	0.26 - 0.45	Gramms *et al.* (1971)
35	15 - 30	1.05 - 2.10	0.62 - 0.82	Kroeker *et al.* (1975)
35	15	4.0	0.56	Fischer *et al.* (1979)
35	26.5	2.4	0.36 - 0.42	Haga *et al.* (1979)
35	10	1.4 - 4.5	0.43	Hobson *et al.* (1980)
30	15 - 20	3.4 - 4.5	0.32 - 0.33	van Velsen (1981)
35	15	0.91	0.44	this study
35	20	0.68	0.45 - 0.47	this study
35	25	0.55	0.54	this study

to be 0.35 m^3 of methane for each kilogram of COD removed at STP. The conversions obtained in this study, as shown in Table 4.3, are lower than the theoretical value. The highest conversion ratio was 0.32 m^3 CH_4/kg COD removed at STP, which was attained at a SRT of 25 days. A discrepancy between theory and practice appears to exist as researchers have been reporting higher methane yields per kilogram of COD removed than the theoretical value of 0.35 m^3 CH_4/kg COD removed at STP. Fischer *et al.* (1979) reported a methane production of 0.43 m^3/kg COD removed at STP for swine waste. Oleszkiewicz and Koziarski (1982) attained 0.48 m^3 CH_4/kg COD removed at STP for dilute, sieved, swine wastes. Pipyn and Verstraete (1980) reported methane production values as high as 0.58 m^3/kg COD removed at STP for swine wastes. The apparent discrepancy between theory and practice is an interesting problem which warrants further investigations.

It is obvious from the results shown in Table 4.1 that the flow-through anaerobic reactor without sludge recycle is not capable of producing an effluent of a standard suitable for direct discharge to most surface waters when treating very high-strength wastes such as the liquid fraction of swine slurry. The average COD of the swine slurry used in this study was 26,300 mg/l. In order to bring the COD of this slurry down to a standard suitable for direct discharge to surface waters, a reduction in COD of greater than 99.9% is required. This efficiency is impossible to obtain in any single-stage process.

Anaerobic treatment as a first step provides a means to reduce the carbonaceous material. The digested effluent can then be further treated by a variety of post-treatment methods ranging from simple lagoons and land application systems to complex treatment systems designed to remove nutrients such as nitrogen and phosphorus and produce effluents that can be directly

discharged to surface waters.

Gólcz *et al.* (1982), in their study of various pretreatments of dilute swine slurries, demonstrated the superiority of anaerobic fermentation over plain sedimentation and chemical precipitation as a method of pretreatment before the activated-sludge process. Their results indicated that anaerobic fermentation produced the most appropriate effluent for further treatment by the activated-sludge process.

5.2 ANAEROBIC TREATMENT OF DILUTE SWINE SLURRIES

The results of this study indicate that dilute swine slurry can be effectively treated in a conventional flow-through anaerobic reactor without recycle. The large amount of dilution water, however, makes conventional flow-through digestion in full scale questionable because of the large reactor volume that would be required to maintain an adequate detention time for organics removal and gas production. A significant portion of the methane produced would be needed to meet the heating requirements of this large reactor. This would lead to a decrease in the amount of methane available for external use. Also, a larger reactor volume leads to increased capital costs for construction.

An alternative to the conventional flow-through reactor is the anaerobic contact process. This process is designed to attain high SRT's at low HRT, resulting in high efficiency despite small reactor volume. In this process, the effluent from the anaerobic reactor is pumped to a settling unit where a portion of the settled sludge is returned to the reactor, enabling a high concentration of active biomass to be maintained.

Oleszkiewicz *et al.* (1981) applied the anaerobic contact process

to the treatment of dilute swine wastes. The HRT of their reactor varied from 1.25 to 7 days at SRT's up to 38 days. Total COD removals of 80-91% were attained and gas production increased with increasing retention time, from 0.161 to 0.242 m^3/kg COD removed. The authors found the process to be stable at a wide range of loading.

Another alternative to the conventional flow-through reactor is the anaerobic filter. The anaerobic filter is a column filled with solid media which act as a stationary surface for microbial attachment. As anaerobic bacteria are retained on the filter medium and not washed off in the effluent, large values of SRT's can be achieved with short HRT's. The short HRT's decrease considerably the size of reactor needed. The long SRT's and the resultant accumulation of high concentration of active biomass allow treatment at temperatures 10-15°C below the optimum mesophilic temperature of 35°C (Brumm and Nye, 1982). Operation at these lower temperatures would increase the amount of methane available for external use.

Oleszkiewicz *et al.* (1981) used anaerobic filters to treat dilute swine wastes. Total COD removals achieved varied from 91 to 61% for COD loadings ranging from 1 to 7.3 kg/m^3-day, at flow rates of 6 to 12 m^3/m^3-day and HRT's varying from 10 to 2 days.

Brumm and Nye (1982) found the anaerobic filter to be an effective system for treating dilute swine wastes at ambient temperatures. They found that the optimal operating condition for their system was between 1 and 2 days detention time (4 kg VS/m^3-day and 2 kg VS/m^3-day) at 24°C. At these operating conditions, VS removals of about 50% and methane efficiencies of 0.45 and 0.62 m^3/kg VS removed were achieved.

The anaerobic filter is less sensitive to changes in environment and

operating conditions than the conventional flow-through reactor. This characteristic of the anaerobic filter is suited for an on-farm situation where the farmer is often limited in the amount of time he can spend operating and maintaining his anaerobic reactor.

5.3 BIODEGRADABILITY OF SOLIDS-SEPARATED SWINE SLURRY

The refractory fraction (R) of the solids-separated swine slurry was determined graphically as shown in Figure 4.5. The R value averaged 0.44. In other words, the biodegradable fraction of the solids-separated swine slurry was 56% of the influent VS concentration. Oleszkiewicz and Koziarski (1982) reported a R value of 0.51 for dilute, sieved, swine waste.

The relatively low biodegradable fraction indicates the resistance of the organic fraction of the solids-separated swine slurry to biological degradation. The main reason for this resistance can be attributed to the presence of lignocellulosic material. This material originates from plant cell walls and mainly consists of cellulose and hemicellulose incorporated in a lignic complex. The incorporated polysaccharides are hardly available for degradation since lignin is regarded as virtually undegradable by anaerobic processes (Hobson *et al.*, 1974) and the cellulolytic enzymes cannot penetrate the lignin matrix because of steric hindrance (de Wit, 1980). As a result, the biodegradation of these materials can only be improved by applying a proper pretreatment method, directed towards a breakdown of the lignin matrix. Such a method may involve a physical, chemical, mechanical or combined treatment of the waste.

Although the biodegradability of refractory materials can be improved by applying a proper pretreatment method, it should be recognized that such methods will be accompanied by high investment and running costs. The

application of these methods on an individual farm is not attractive.

5.4 SUBSTRATE REMOVAL RATE

Morris *et al.* (1977) examined the influence of retention time, volumetric organic loading rate, and influent volatile solids concentration on the anaerobic fermentation process. An important observation made in their investigation was that the rate of VS degradation could be expressed as a function of these process design variables. Morris *et al.*(1977) showed that the effluent biodegradable VS concentration, $(S_b)_1$, is dependent upon both the influent VS concentration and the HRT of the system. $(S_b)_1$ is therefore observed as a linear function of S_o/HRT or the volumetric organic loading rate. This relationship can be expressed mathematically as (Morris *et al.* 1977):

$$(S_b)_1 = S_o/(k \cdot HRT)$$

or

$$S_1 = [S_o/(k \cdot HRT)] + RS_o$$

where

$(S_b)_1$ = effluent biodegradable VS concentration

S_o = influent total VS concentration

S_1 = effluent total VS concentration

k = rate of substrate removal

R = refractory fraction of the influent VS concentration

HRT = hydraulic retention time

This model is a hyperbolic relationship between time and fraction remaining of influent VS. The coefficient k, defined as the rate of substrate removal and expressed as days^{-1}, can be graphically determined by plotting $(S_b)_1$ versus S_o/HRT as shown in Figure 5.1. The coefficient k represents the inverse of the slope of the line.

The plot shown in Figure 5.1 yields k = 0.48 day^{-1} for this study. This

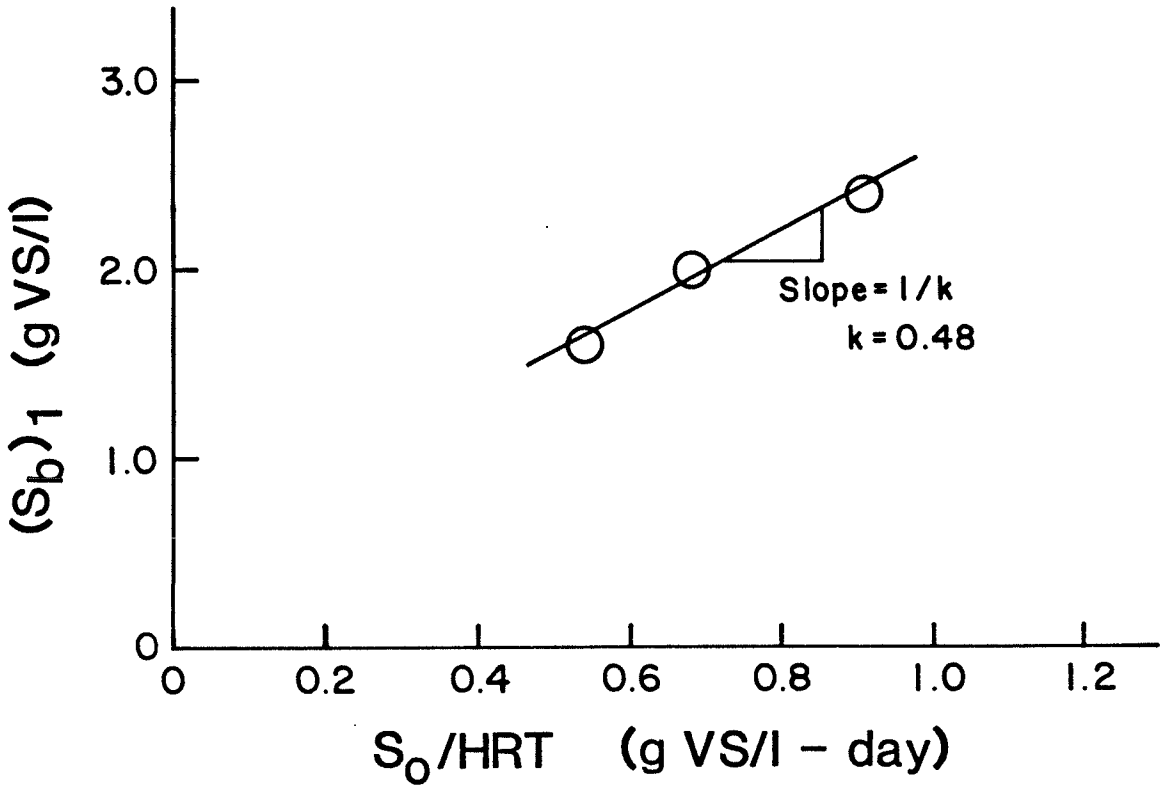

Figure 5.1 Effect of S_o and HRT upon the biodegradable effluent VS concentration

k value applies for a fermentation temperature of 35°C and for a range of HRT's from 15 to 25 days. In theory, the model assumes that as HRT approaches infinity, the influent biodegradable VS should all be removed. In other words, at S_o/HRT = 0, $(S_b)_1$ should theoretically also equal zero. However, HRT's greater than 25 days were not investigated in this study and it would not be proper to extend the model beyond the limits of the data gathered in this study.

Oleszkiewicz and Koziarski (1982) obtained a k value of 0.89 day^{-1} for dilute, sieved, swine waste. Their k value is greater than the value obtained in this study. This difference may be attributed to the different process conditions, such as pH, temperature and ammonia nitrogen concentration, as well as to differences in the composition and origin of the respective swine slurries. The swine slurry used in this present study may contain some residual feed additives or antibiotics which may have an effect on the substrate removal rate.

Table 5.2 compares the effluent VS concentrations obtained from the experimental reactors with the effluent concentrations predicted by the model.

Table 5.2 Experimental and Predicted Effluent VS Concentrations

SRT days	Experimental VS Concentration (g/l)	Predicted* VS Concentration (g/l)
15	8.4	7.9
20	8.0	7.4
25	7.6	7.1

* R = 0.44

The model appeared to predict the effluent characteristics of the experimental reactors reasonably well. The predicted effluent concentrations were generally within 8% of the actual effluent concentration. This variation is very good when considering the sampling and analyses errors that may have occurred during the experimental program.

CHAPTER 6

CONCLUSIONS

The results of this study lead to the following conclusions:

1) The flow-through anaerobic reactor can effectively treat solids-separated swine slurries at 35°C at SRT's of 15, 20, and 25 days.
2) Treatment of dilute swine slurries in a full scale conventional flow-through anaerobic digester without sludge recycle would be questionable economically because of the large reactor volume that would be required to maintain an adequate detention time for organics removal and gas production.
3) The removal of organic matter increased at increasing SRT's. Total VS removal varied from 45.41% at a SRT of 15 days to 54.16% at a SRT of 25 days. Total COD removal varied from 45.41% at a 15-day SRT to 54.16% at a 25-day SRT.
4) Total gas production varied from 0.44 - 0.54 m^3/kg VS added at SRT's of 15 to 25 days.
5) Methane content did not vary with detention time. Each reactor produced gas with a methane content of about 69%.
6) Maximum methane production occurred at the higher organic loading rate and shorter SRT while removal of organic matter was greater at the lower organic loading rate and longer SRT.
7) The refractory fraction, R, of the solids-separated swine slurry was determined to be 0.44. In other words, the biodegradable fraction of the solids-separated swine slurry was 56% of the influent VS concentration.

8) A substrate removal rate of 0.48 day^{-1} was obtained for the solids-separated swine slurry.

9) The digested effluent would have to be further treated by proper post-treatment methods before it can be discharged to surface waters.

CHAPTER 7

SUGGESTIONS FOR FURTHER STUDY

Based on the findings of this study, the following topics are suggested as possible subjects for further study:

1) examine the efficiency of anaerobic processes with SRT's greater than their HRT for treating dilute swine slurries;
2) develop a complete treatment system for dilute swine slurries which incorporates anaerobic fermentation as a method of pretreatment and is capable of producing an effluent that can be discharged directly to surface waters;
3) examine various pretreatment methods which would improve the biodegradability of the volatile solids in dilute swine slurries;
4) conduct a pilot scale study of an anaerobic process treating dilute swine slurry over a prolonged period of time.

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APPENDIX A

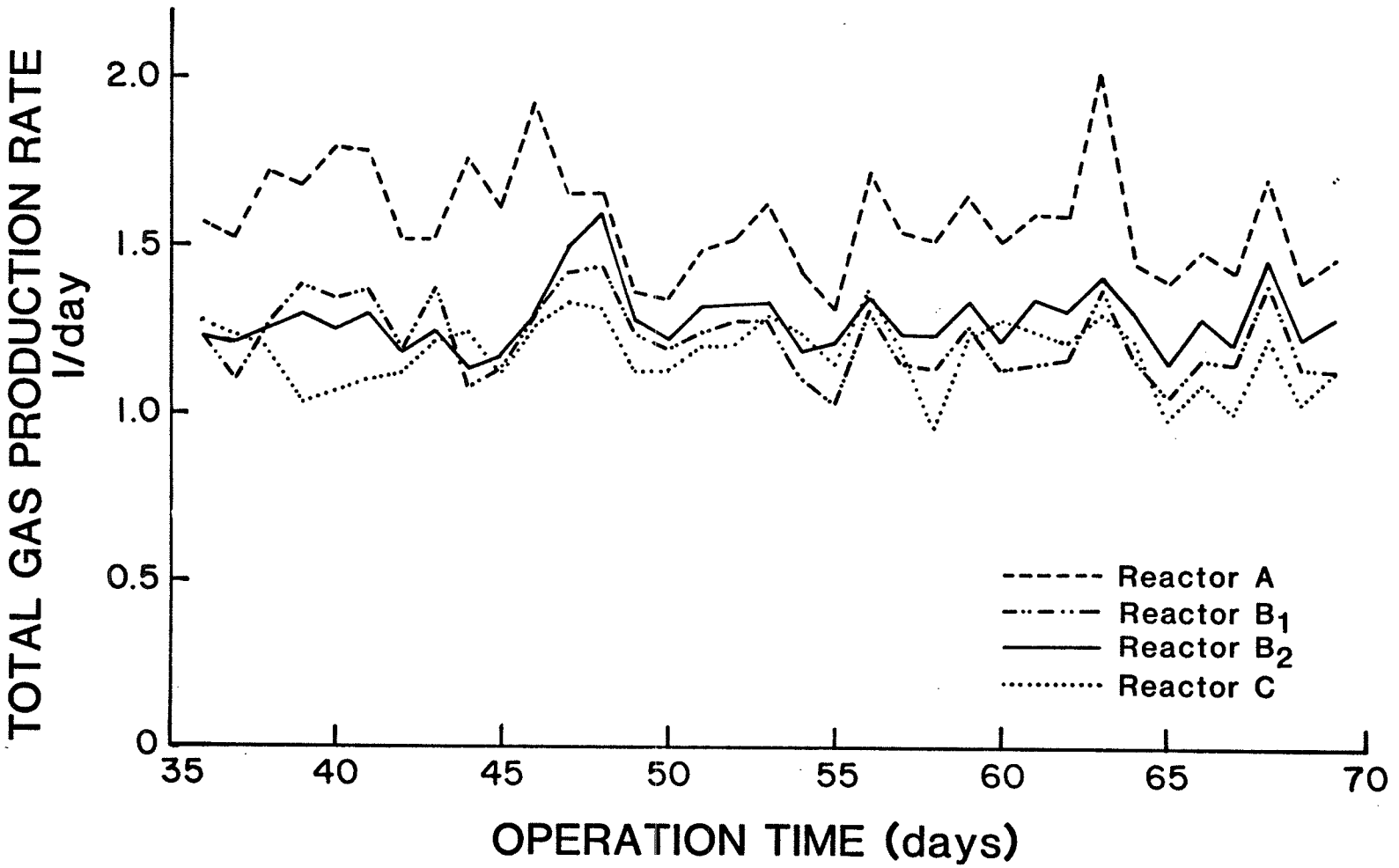

Figure A.1 Daily gas production

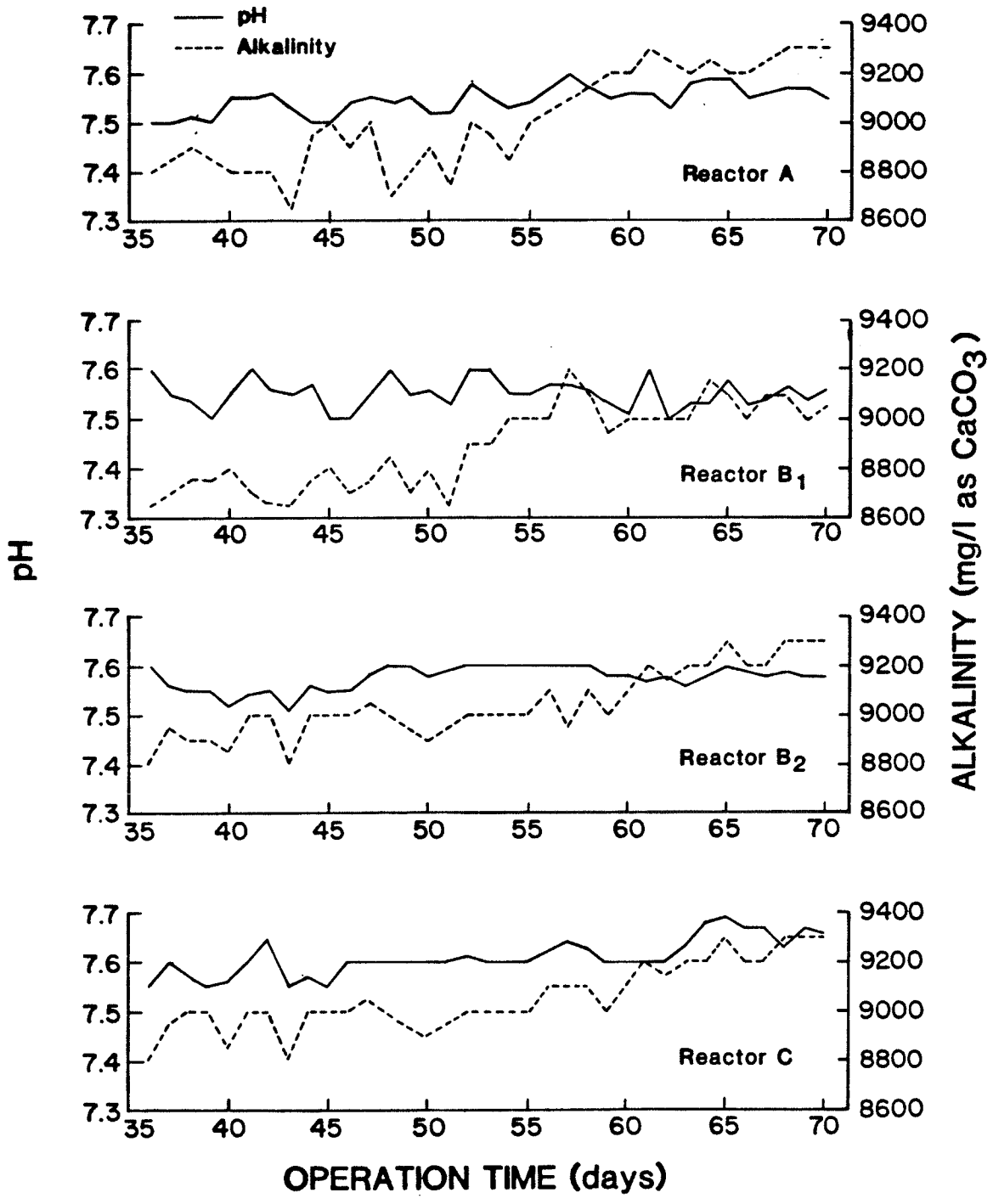

Figure A.2 Daily pH and alkalinity measurements

APPENDIX B

Sample Calculation for COD Balance

As described in section 4.2, a mass balance on the COD may be evaluated in the following manner:

influent COD = effluent COD + gaseous methane production as COD.

Example: Reactor A

1. Influent COD: multiply influent substrate concentration by flow rate.

   27.3 g COD/l x 4/15 l/day = 7.28 g COD/day

2. Effluent COD: multiply effluent substrate concentration by flow rate.

   14.3 g COD /l x 4/15 l/day = 3.81 g COD/day

3. Gaseous CH_4 Production: multiply the amount of gas produced by CH_4 composition; then convert to COD equivalent;

   1.66 l gas/day x 67.8% CH_4 = 1.125 l CH_4/day

   1.125 l CH_4/day ÷ 0.39511 l CH_4/g CH_4 - COD @ 35° = 2.85 g COD/day

4. Total Out: sum of # 2 + # 3.

   Total out = 3.81 + 2.85 = 6.66 g COD/day.

5. Percent Recovered: Total out divided by influent:

   % recovered = (6.66/7.28) x 100% = 91.5%