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This is the first of a series of four articles on a process of waste treatment that has not been too well understood and consequently has not been as widely used as it might deserve. Part One discusses the advantages and disadvantages of anaerobic waste treatment, conventional practices and the present concepts of the microbiology and chemistry involved. Parts Two and Three will cover the environmental requirements for achieving control of the anaerobic process and preventing or correcting toxicity in the system. Part Four will outline the application of these various concepts in treatment plant design.

Anaerobic Waste Treatment Fundamentals

PART ONE | Chemistry and Microbiology

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THE anaerobic process is in many ways ideal for waste treatment. It has several significant advantages over other available methods and is almost certainly assured of increased usage in the future. Anaerobic treatment is presently employed at most municipal treatment plants, and is responsible for the major portion of waste stabilization that occurs there. However, in spite of the present significance and large future potential of this process, it has not generally enjoyed the favorable reputation it truly deserves. The primary obstacle has been a lack of fundamental understanding of the process, required both to explain and control the occasional upsets which may occur, and to extend successfully this process to the treatment of a wide variety of industrial wastes.

An increasing realization of the potentials of anaerobic treatment is evident from the reporting each year of larger numbers of research investigations on this process. Already, significant advances have been made extending the process so it can be used successfully on many more organic wastes. This series of articles is intended to summarize our present knowledge of anaerobic treatment and to point out the important parameters for design, operation, and control. This first article is concerned with a general description, together with the chemistry and microbiology of the process. The subsequent three articles will deal with treatment control and design.

Advantages

The advantages of anaerobic treatment can best be indicated by comparing this process with aerobic treatment. In aerobic treatment, as represented by the activated sludge and trickling filter processes, the waste is mixed with large quantities of microorganisms and air. Microorganisms use the organic waste for food, and use the oxygen in the air to burn a portion of this food to carbon dioxide and water for energy. Since these organisms obtain much energy from this oxidation, their growth is rapid and a large portion of the organic waste is converted into new cells. The portion converted to cells is not actually stabilized, but is simply changed in form. Although these cells can be removed from the waste stream, the biological sludge they produce still presents a significant disposal problem.

In anaerobic treatment, the waste is also mixed with large quantities of microorganisms, but here, air is excluded. Under these conditions, bacteria grow which are capable of converting the organic waste to carbon dioxide and methane gas. Unlike aerobic oxidation, the anaerobic conversion to methane gas yields relatively little energy to the microorganisms. Thus, their rate of growth is slow and only a small portion of the waste is converted to new cells, the major portion of the degradable waste being converted to methane gas. Such conversion to methane gas represents waste stabilization since this gas is insoluble and escapes from the

Table 1—Advantages of Anaerobic Treatment

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 end product.

stages.[3] Each stage represents the culmination of growth of a population of methane formers capable of fermenting one particular group of compounds. The process is not completely operational until all the groups of methane formers are finally established. This may take several weeks if the process is started without the benefit of "seed" sludge containing the methane formers required for the specific acids present.

While there are many different methane forming bacteria, there are also many different acid forming bacteria. Waste stabilization requires a balance among all these organisms. The establishment and maintenance of this balance is normally indicated by one of the most important control tests, that for the concentration of volatile acids. The volatile acids are the short chain organic acids indicated in Table 2. The acids shown are the major intermediates produced by the first stage conversion. They represent the intermediate compounds of most importance in anaerobic treatment, and most of the methane formed from this process results from fermentation of these acids by the methane bacteria.

When the system is in balance, the methane bacteria use the acid intermediates as rapidly as they appear. However, if the methane bacteria are not present in suitable numbers, or are being slowed down by unfavorable environmental conditions, they will not use the acids as rapidly as they are produced by the acid formers, and the volatile acids will increase in concentration. Thus, an increase in acid concentration indicates the methane formers are not in balance with the acid formers. An analysis for the individual acids present will indicate the particular methane bacteria not carrying out their portion of the treatment. Unfortunately, the volatile acids analysis does not indicate an unbalance in the acid forming organisms. At present, no satisfactory method is available to determine the relative populations of the bacteria specifically responsible for production of certain acids.

Methane Formation

The methane producing bacteria have proven to be very difficult to isolate and study. Consequently, relatively little is known of their basic biochemistry. The conversion of organic matter into methane no doubt proceeds through a long series of complex biochemical steps. Although almost nothing is known of the individual steps involved, tracer studies have indicated the major sources of methane as shown in Table 3.[4,5] One source of methane is the direct cleavage of acetic acid into methane and carbon dioxide. This acid is one of the most important volatile acids formed from the decomposition of complex organics and is the source of most methane in anaerobic treatment. The methyl carbon of acetic acid, marked with an asterisk in Table 3, together with its three hydrogen atoms, are converted intact into methane gas. The carbonyl carbon, shown without an asterisk, is converted to carbon dioxide.

Most of the remaining methane in anaerobic treatment is formed from the reduction of carbon dioxide. Here, hydrogen, which is removed from organic compounds by enzymes, reduces carbon dioxide to methane gas. The carbon dioxide here functions as a hydrogen or electron acceptor, just as oxygen in aerobic treatment. There is always a large excess of carbon dioxide available in anaerobic treatment, and thus the availability of carbon dioxide for this reduction is never a limiting factor in treatment of complex materials.

Volatile Acid Intermediates

The two major volatile acid intermediates formed in anaerobic treatment are acetic acid and propionic acid.[6,7] The importance of these two acids as precursors of methane is indicated in Fig. 3, which shows the pathways by which mixed complex organic materials are converted to methane gas. The percentages shown are based on COD conversion and are for methane fermentation of complex materials such as municipal waste sludge or other wastes of similar composition. The percentages would be different for other wastes.

The complete methane fermentation of complex wastes has been compared to a factory assembly line operation[8] in that the processing of raw waste material to the final methane product requires the help of several different workers. The raw material must be worked on by each group of organisms to prepare it for handling by the next. Although each group's contribution to the overall processing may be small, it is still necessary to the formation of the final product. Thus, if just one group of workers fails to do its job, the final product cannot be formed. For example, 30 percent of the complex waste shown in Fig. 3 becomes propionic acid through the action of the methane bacteria, and if these organisms are not functioning, this portion cannot be converted to methane gas. This is true even though the propionic acid bacteria themselves directly produce only 13 percent of the methane. They convert the remainder of the propionic acid, or 17 percent, to acetic acid.

The acetic acid fermenting methane bacteria are also very important, since if they fail, 72 percent of the waste cannot be converted to methane gas. It is interesting to note that acetic acid is formed by several routes and through the action of many different bacteria. Only about 20 percent of the waste is converted directly to acetic acid

Table 2—Common Volatile Acid Intermediates

Acid	Chemical Formula
Formic Acid	$HCOOH$
Acetic Acid	CH_3COOH
Propionic Acid	CH_3CH_2COOH
Butyric Acid	$CH_3CH_2CH_2COOH$
Valeric Acid	$CH_3CH_2CH_2CH_2COOH$
Isovaleric Acid	$(CH_3)_2CHCH_2COOH$
Caproic Acid	$CH_3CH_2CH_2CH_2CH_2COOH$

Table 3—Major Mechanisms of Methane Formation

I. Acetic Acid Cleavage:

$$C^*H_3COOH \rightarrow C^*H_4 + CO_2$$

II. Carbon Dioxide Reduction:

$$CO_2 + 8H \rightarrow CH_4 + 2H_2O$$

waste stream where it can be collected and burned to carbon dioxide and water for heat.

As much as 80 to 90 percent of the degradable organic portion of a waste can be stabilized in anaerobic treatment by conversion to methane gas, even in highly loaded systems. This is in contrast to aerobic systems, where only about 50 percent of the waste is actually stabilized even at conventional loadings.

Other advantages of anaerobic treatment are shown in Table 1. Since only a small portion of the waste is converted to cells, the problem of disposal of excess sludge is greatly minimized. Also, the requirements for the nutrients, nitrogen and phosphorus, are proportionately reduced. This is especially important in the treatment of industrial wastes which lack these materials. The sludge produced is quite stable and will not present a nuisance problem.

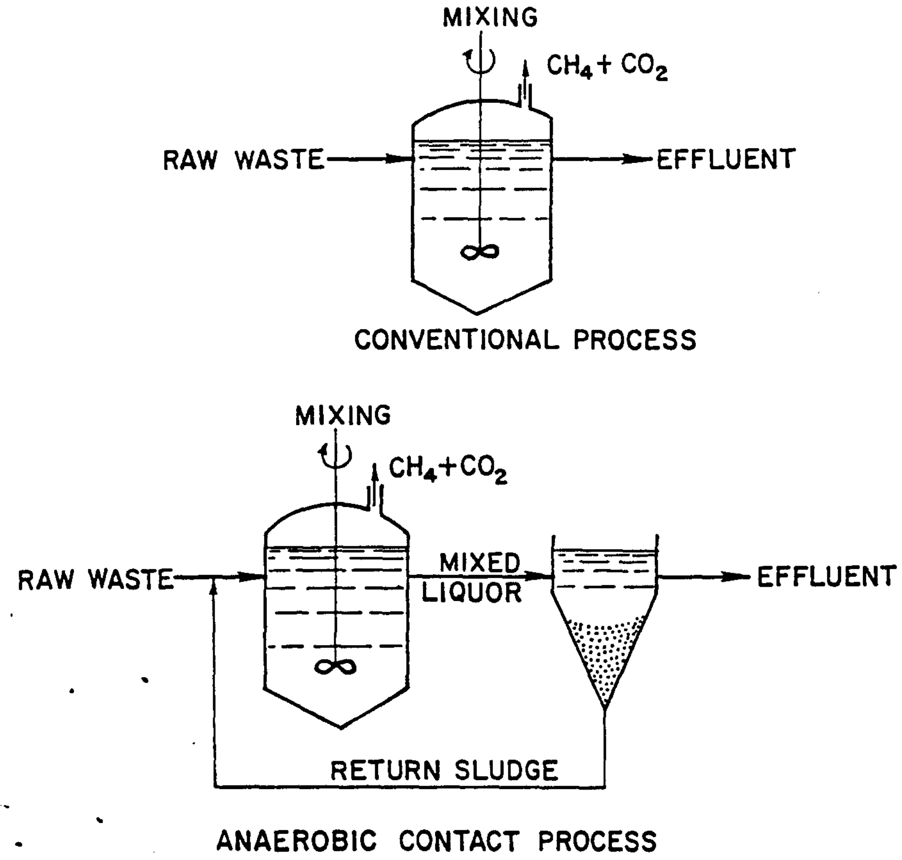
Since anaerobic treatment does not require oxygen, treatment rates are not limited by oxygen transfer. The absence of a need for oxygen also reduces power requirements for treatment. In contrast, the methane gas produced by anaerobic treatment is a good source of fuel energy and is frequently used for heating buildings, running engines, or producing electricity.

The anaerobic treatment process does have some disadvantages which may limit the use of this process for certain industrial wastes. The major disadvantage is that relatively high temperatures are required for optimum operation; temperatures in the range from 85° to 95° F are preferred. Dilute wastes may not produce sufficient methane for waste heating and this may represent a major limitation. This limitation suggests a need for more research on low temperature anaerobic treatment, as there are indications that much lower temperatures can be used if the systems are adequately designed.

Another disadvantage of anaerobic treatment is related to the slow rate of growth of the methane producing bacteria. Because of it, longer periods of time are required for starting the process. This slow rate of growth also limits the rate at which the process can adjust to changing waste loads, temperatures, or other environmental conditions.

The advantages of anaerobic treatment are quite significant, while the disadvantages are relatively few. The advantages normally far outweigh the disadvantages for more concentrated wastes, with BOD values greater than 10,000 mg/L. For less concentrated wastes, the disadvantages become more important, and may limit the use of this process. A noted exception is the successful anaerobic treatment of meat packing wastes with BOD concentrations as low as 1,000 mg/L.[1] These wastes are fairly warm and the temperature requirement does not present a limitation.

FIGURE 1. The two basic anaerobic process designs are diagrammed below.



Process Description

In anaerobic treatment, there are two basically different process designs. One is the "conventional process" most widely used for the treatment of concentrated wastes such as primary and secondary sludges at municipal treatment plants. The other process is one designed to handle more dilute waste and has been termed the "anaerobic contact process."[1,2] Schematic diagrams of each process are shown in Fig. 1.

The conventional anaerobic treatment process consists of a heated digestion tank containing waste and bacteria responsible for anaerobic treatment. Raw waste is introduced either periodically or continuously and is perferably mixed with the digester contents. The mixed treated waste and microorganisms are usually removed together for final disposal. Sometimes this mixture is introduced into a second tank where the suspended material is allowed to settle and concentrate for more efficient disposal.

As the detention time in the conventional process is reduced, an increased percentage of bacteria are removed from the tank each day with the effluent. The limiting detention time is reached when the bacteria are being removed from the system faster than they can reproduce themselves, occurring after about three to five days at temperatures of operation of 95°F. For practical control and reliable treatment, a detention time much above this, or about ten to thirty days, is normally used.

With dilute wastes, hydraulic detention times should be very short if the process is to be economical. These are possible in the anaerobic contact process. Here, the bacteria are not lost with the effluent, but are maintained in the system. In this case, a digester is used. However, it is followed by a settling tank which removes the active biological suspended solids from the effluent stream for recycle back to the digester. This system is similar in operation to the activated sludge process and permits the maintenance of a high biological population for rapid decomposition, while operating at a relatively low hydraulic detention time. Such a system has been found economical with wastes having BOD concentrations of about 1,000 mg/L and detention times of less than 6 to 12 hours.

The gas produced in anaerobic treatment makes the suspended particles buoyant and difficult to settle. Therefore, a degasifier is frequently required between the digester and the settling tank in the anaerobic contact process to permit proper settling of the suspended solids. A flotation process making use of the large quantities of dissolved gases to float and concentrate the solids for return to the digester also appears feasible.

The important parameter governing the efficiency and operation of both the conventional process and the anaerobic contact process is the biological solids retention time. This is similar to the sludge age concept used in aerobic treatment and is defined as follows:

$$SRT = \frac{M_t}{M_e} \ldots\ldots\ldots\ldots\ldots (1)$$

where,

SRT = solids retention time,

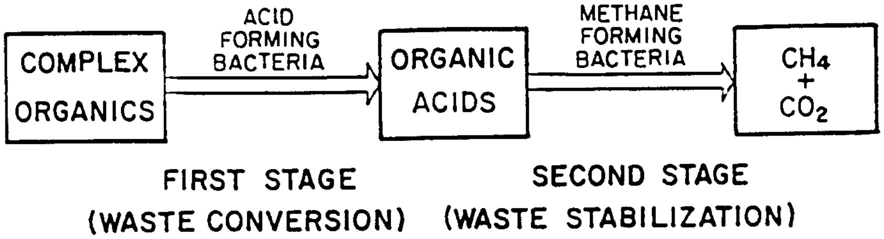
M_t = total weight of suspended solids in treatment system

M_e = total weight of suspended solids leaving the system per day, including both that deliberately wasted and that passing out with the plant effluent.

The weight of suspended solids leaving the system per day refers to the sum total of the suspended solids lost in the effluent plus the suspended solids deliberately removed as "waste sludge." The SRT relates treatment operation to the age and quantity of microorganisms in the system, and is a sound parameter for design. The major requirement of both the conventional process and the anaerobic contact process is the SRT be at least ten days for temperatures of operation of 95°F. The required SRT is about doubled for each 20°F lower temperature.

Microbiology and Biochemistry

It can generally be said that any waste susceptible to aerobic treatment can also be treated anaerobically. There are few exceptions to this statement. In addition, there are certain wastes, such as those containing cellulose, which are more readily treated by the anaerobic process than by aerobic treatment.



■ FIGURE 2. The two stages of anaerobic treatment consist of waste conversion by acid forming bacteria followed by stabilization with methane forming bacteria.

It is commonly considered that anaerobic treatment is only useful for the destruction of suspended solids. This feeling has probably resulted from the extensive use of anaerobic treatment for sludge digestion. However, the process is also well suited to the treatment of soluble wastes.

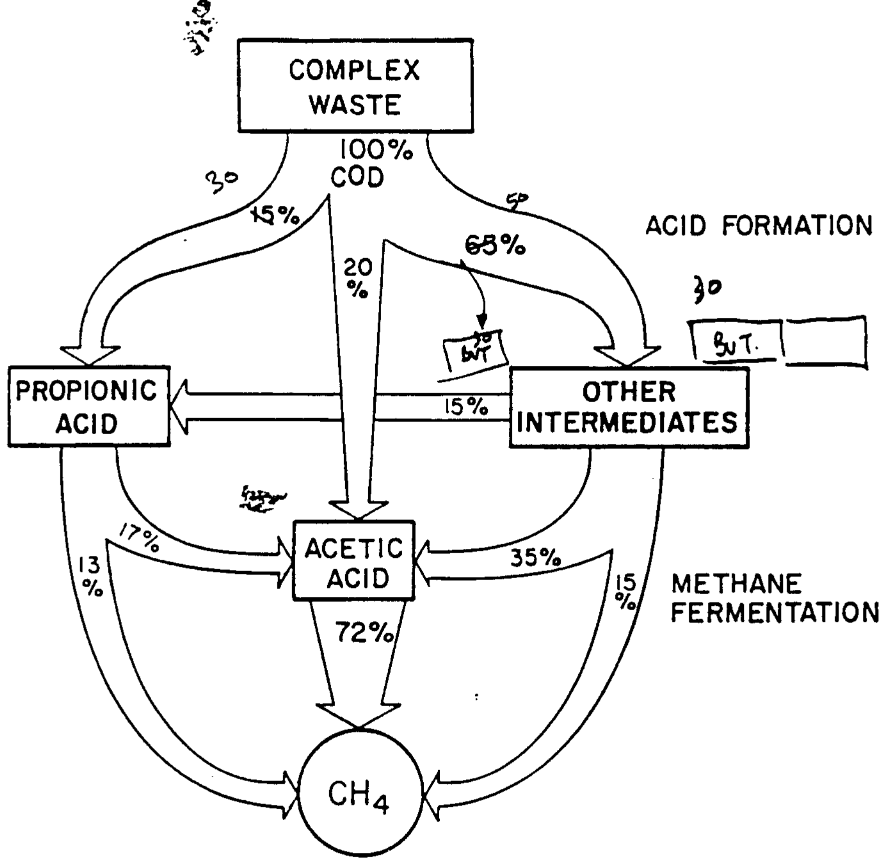
Another common fallacy is that anaerobic treatment is an inefficient process. This belief is also related to experience with sludge digestion, where most of the organic material being treated is not readily susceptible to biological degradation, and only about 50 percent reduction in solids is possible. However, such wastes cannot be treated any better by aerobic processes. Parameters of waste strength such as BOD, which indicate the biological degradability of the waste, should be used to compare the two processes on an equal basis. By using such a comparison, it can be shown the two processes are quite comparable in efficiency of treatment at similar volumetric loadings.

Two-Stage Process

Anaerobic treatment of complex organic materials is normally considered to be a two-stage process as indicated in Fig. 2. In the first stage, there is no methane production and hence no waste stabilization. In this stage, the complex organics are changed in form by a group of facultative and anaerobic bacteria commonly termed the "acid formers." Complex materials such as fats, proteins, and carbohydrates are hydrolized, fermented, and biologically converted to simple organic materials. For the most part, the end products of this first-stage conversion are organic fatty acids. Acid forming bacteria bring about these initial conversions to obtain the small amounts of energy released for growth, and a small portion of the organic waste is converted to cells. Although no waste stabilization occurs during the first stage of treatment, it is required to place the organic matter in a form suitable for the second stage of treatment.

It is in the second stage of methane fermentation that real waste stabilization occurs. During this stage, the organic acids are converted by a special group of bacteria termed the "methane formers" into the gaseous end products, carbon dioxide and methane. The methane forming bacteria are strictly anaerobic and even small quantities of oxygen are harmful to them. There are several different groups of methane formers, and each group is characterized by its ability to ferment a relatively limited number of organic compounds. Thus, in the complete methane fermentation of complex materials, several different methane bacteria are required. The methane formers which use materials such as formic acid and methanol grow very rapidly and can thrive at sludge retention times of less than two days. However, the most important methane formers, which live on acetic and propionic acids, grow quite slowly, and sludge retention times of four days or longer are required for their growth. These bacteria carry out the major portion of waste stabilization. Their slow growth and low rate of acid utilization normally represents the limiting step around which the anaerobic treatment process must be designed.

The many different methane forming organisms responsible for anaerobic treatment, their different sources of food, and their different rates of growth are responsible for some confusion as to when good waste treatment is well under way. For example, during the start-up of the anaerobic treatment process, some methane formation is often noted during the early stages. However, this is produced only from certain materials that are fermented to methane readily. Significant methane production does not occur for several days or weeks, and when it does, it comes in



■ **FIGURE 3. Pathways in methane fermentation of complex wastes such as municipal waste sludges. Percentages represent conversion of waste COD by various routes.**

during the acid formation stage. A much larger portion (52 percent) is formed from the action of various methane producing bacteria which ferment propionic acid and other intermediates to acetic acid and methane.

For different industrial wastes, the percentages shown in Fig. 3 may be different. However, the largest percentage of methane will still result from acetic acid fermentation, which is the most prevalent volatile acid produced by fermentation of carbohydrates, proteins, and fats. Propionic acid, on the other hand, is formed mainly during fermentation of carbohydrates and proteins. The other volatile acids, although significant, are of minor importance.

Thus, although many different organisms are required in anaerobic treatment, the two groups of methane bacteria which handle acetic and propionic acids, are the most important in the methane fermentation. Unfortunately, they also appear to be among the slowest growing methane bacteria and the most sensitive to environmental changes.

Waste Stabilization

Waste stabilization in anaerobic treatment is directly related to methane production. Buswell and coworkers[9] gave the formula shown in Table 4 to predict the quantity of methane from a knowledge of the waste chemical composition. From this formula, it can be shown that the ultimate oxygen demand of the waste being degraded is eequal to the ultimate oxygen demand of the methane gas produced. This fact allows prediction of methane production in another way, that is, from an estimate of COD or BOD_L (ultimate BOD) stabilization. The ultimate oxygen demand of methane gas is as follows:

$$CH_4 + 2O_2 \rightarrow CO_2 + 2H_2O \ldots (2)$$

This formula shows one mol of methane is equivalent to two mols of oxygen. Converting to cubic feet of methane per pound of oxygen, the value shown in Table 4 for relation between waste stabilization and methane production is obtained. Measured values for methane production per pound of COD or BOD_L stabilization for a wide variety of wastes varying from pure laboratory substrates to complex waste sludge have shown the validity of this relationship and the close accuracy with which it can be used to predict methane production.

The relationship between methane production and waste stabilization can also be used in another way in anaerobic waste treatment operation. Here, the methane production can readily be determined. Such a determination gives a direct and rapid measurement of actual waste stabilization and permits closely following the efficiency of waste treatment. For example, if 1,500 pounds of waste COD are added to an anaerobic waste treatment system per day, and the methane production is 5620 cubic feet STP (standard conditions of temperature and pressure), 1000 pounds of COD are being stabilized by conversion to methane gas. Thus, the efficiency of waste stabilization is 67 percent.

Anaerobic Biological Growth

The most important advantages of the anaerobic waste treatment processes are the high percentage of stabilization obtained and the low percentage of conversion of organic matter to biological cells. The small quantities of sludge growth minimizes the problems of biological sludge disposal, as well as the requirements for the inorganic nutrients, nitrogen and phosphorus.

The biological growth resulting from anaerobic treatment of different types of wastes are shown in Fig. 4.[10] Resulting biological suspended solids under anaerobic conditions vary considerably from one type of waste to the next. Thus, the growth cannot be predicted from a knowledge of the waste strength alone, as it is also related to waste composition. The two extremes in growth are represented by fatty acid wastes, which produce the lowest growth, to carbohydrates, which

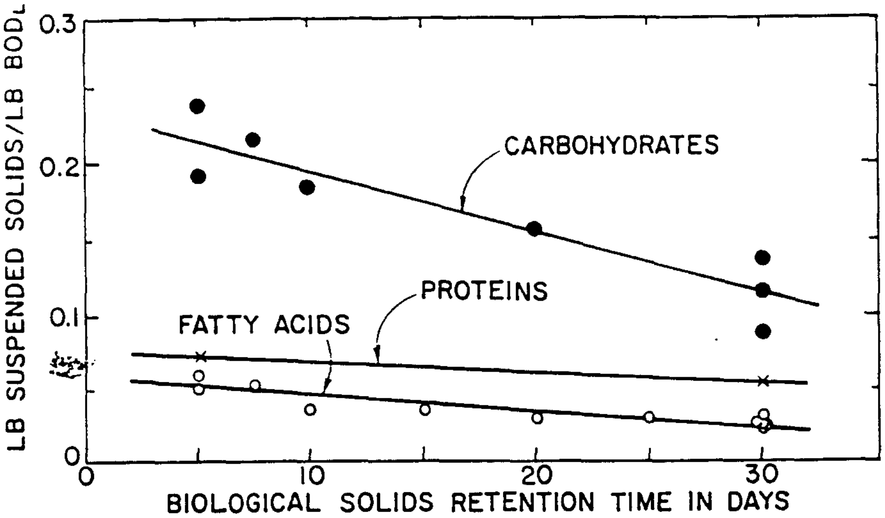
Table 4—Methods of Predicting Methane Production

I. Prediction from Waste Chemical Composition:[9]

$$C_nH_aO_b + \left(n - \frac{a}{4} - \frac{b}{2}\right)H_2O \rightarrow \left(\frac{n}{2} - \frac{a}{8} + \frac{b}{4}\right)CO_2 + \left(\frac{n}{2} + \frac{a}{8} - \frac{b}{4}\right)CH_4$$

II. Prediction from Waste Stabilization:

One pound BOD_L or COD stabilized = 5.62 cubic feet CH_4 (STP)



■ **FIGURE 4. Biological solids production resulting from methane fermentation.**

produce the highest. Other types of waste can be expected to vary between these two extremes.

Fig. 4 shows that the quantity of waste converted to biological suspended solids decreases with increase in sludge retention time. When cells are maintained for long periods of time, they consume themselves for energy, with the result that the net growths are less. Thus, greater waste stabilization and lower biological cell production is obtained at long sludge retention times. Such retention times also result in higher efficiencies of treatment.

In order for any biological process to operate, inorganic nutrients required by the bacteria for their growth must be supplied. The inorganic materials required in highest concentration for this growth are nitrogen and phosphorus. Since these materials may be absent in many industrial wastes, it is important to know the quantities which may have to be added. The requirements for nitrogen may be determined from the cell growth and the fraction of nitrogen in the cells. Based on an average chemical formulation of biological cells of $C_5H_9O_3N$, the nitrogen requirement is about 11 percent of the cell volatile solids weight. The requirement for phosphorus has been found to be about one-fifth that for nitrogen, or about 2 percent of the biological solids weight. Thus, if the solids production were 0.1 lb./lb. of BOD_L, the nitrogen requirement would be 11 percent of this or 0.011 lb/lb. of BOD_L, and the phosphorous requirement would be 2 percent or 0.002 lb./lb. of BOD_L.

Theoretically, the biological sludge production and nitrogen and phosphorus requirements should be based on the fraction of waste removed during treatment, rather than on waste added. However, it is better in anaerobic treatment, to base such requirements on waste additions. The reason for this is that in highly loaded systems, the first stage of acid formation may take place to a larger extent than the second stage of methane formation or stabilization. The first stage bacteria would grow and require nitrogen and phosphorus, even though the waste at this point is not being stabilized. Thus, estimates of growth and nutrient requirements based on stabilization alone, may be much too low.

It should be noted that the suspended solids formed in anaerobic treatment as indicated by Fig. 4 only represents the growth of new cells. Many wastes, notably municipal sludges, contain large quantities of suspended solids which also contribute to the suspended solids in the digester. In this case, the suspended solids for final disposal would be much higher than indicated by Fig. 4. Wastes similar to municipal sludge are quite complex and the increase in biological solids which occurs during treatment may be far overshadowed by the large changes in waste suspended solids occurring during anaerobic treatment. Fig. 4 is of most value for predicting requirements for nutrient deficient wastes, as well as predicting suspended solids production for relatively soluble wastes.

Summary

The anaerobic process has several advantages over aerobic processes for waste treatment. Use of the anaerobic contact process, or a similar modification, permits the use of this process for the treatment of relatively dilute waste. Although the microbiology and biochemistry of the process is complex, it normally operates quite well with a minimum of control. The bacteria responsible for this treatment are widespread in nature and grow well by themselves when provided with the proper environment.

This first in a series of three articles was intended to give an understanding of the bacteriology involved in anaerobic waste treatment and the biochemical steps resulting in the formation of acetic and propionic acids as intermediate products before a waste is finally converted to methane gas.

The next article in this series will be concerned with the control and operation of anaerobic treatment systems and will indicate the environmental requirements for proper digestion, indicators of treatment unbalance and methods for pH control. □□□

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Anaerobic Waste Treatment Fundamentals

PART TWO | Environmental Requirements and Control

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THE ANAEROBIC PROCESS has many advantages over other methods of organic waste treatment. This process has been widely used for the stabilization of municipal waste sludges and has good potential for the treatment of many industrial wastes. In this series of articles, a summary of the current information on the biochemistry and chemistry as related to process design and control is being presented. The first article in this series[1] considered the basic microbiology and biochemistry. This article summarizes the environmental requirements for anaerobic treatment and describes methods of process and pH control.

Environmental Requirements

The methane bacteria, which are responsible for the majority of waste stabilization in anaerobic treatment, grow quite slowly compared to aerobic organisms and so a longer time is required for them to adjust to changes in organic loading, temperature or other environmental conditions. For this reason, it is usually desirable in design and operation to strive for optimum environmental conditions so that more efficient and rapid treatment might be obtained. A summary of optimum environmental conditions for anaerobic treatment are listed in Table 1.

Table 1—Optimum Conditions for Anaerobic Treatment

Optimum Temperatures
  Mesophilic Range
  85° to 100°F
  Thermophilic Range
  120° to 135°F
Anaerobic Conditions
Sufficient Biological Nutrients
  Nitrogen
  Phosphorous
  Others
Optimum pH—6.6 to 7.6
Absence of Toxic Materials

At higher temperatures, rates of reaction proceed much faster, resulting in more efficient operation and smaller tank sizes. Two optimum temperature levels for anaerobic treatment have been reported,[2,3] one in the mesophilic range from 85° to 100°F, and the other in the thermophilic range from 120° to 135°F. Although treatment proceeds much more rapidly at thermophilic temperatures, the additional heat required to maintain such temperatures may offset the advantage obtained. Therefore, most treatment systems are designed to operate in the mesophilic range or lower.

Another environmental requirement for anaerobic treatment is that anaerobic conditions be maintained. Small quantities of oxygen can be quite detrimental to the methane-formers and other anaerobic organisms involved. This requirement usually necessitates a closed digestion tank, which is also desirable so the methane gas can be collected for heating.

The anaerobic process is dependent upon bacteria, which require nitrogen, phosphorus and other materials in trace quantities for optimum growth. Municipal waste sludge normally contains a variety of these materials, and thus usually provides an ideal environment for growth. However, industrial wastes are frequently more specific in composition and biological nutrients must be added for optimum operation. For such wastes, it has been found that materials in addition to nitrogen and phosphorus are frequently required.[6] In some cases, it has been found beneficial to add from 30 to 60 mg/L of iron in the form of ferric chloride.[7] In addition, the inclusion of domestic wastes along with industrial wastes for treatment can be of benefit by supplying inorganic and organic materials which stimulate growth, resulting in more efficient and rapid treatment.

One of the most important environmental requirements is that for a proper pH.[8] Anaerobic treatment can proceed quite well with a pH varying from about 6.6 to 7.6, with an optimum range of about 7.0 to 7.2. Beyond these limits, digestion can proceed, but with less efficiency. At pH values below 6.2, the efficiency drops off rapidly, and the acidic conditions produced can become quite toxic to the methane bacteria. For this reason, it is important that the pH not be allowed to drop below this value for a significant period of time. Because this parameter is so important, the control of pH will be discussed in more detail in a following section.

A last requirement for successful anaerobic treatment is that the waste be free from toxic materials. Normally, concentrated wastes are more susceptible to anaerobic treatment. However, such wastes are also more likely to have high or inhibitory concentrations of various materials ranging from inorganic salts to toxic organic compounds. With municipal wastes, the major problem usually results from heavy metals. Industrial wastes, on the other hand, may have inhibitory concentrations of various common salts such as those containing sodium, potassium, magnesium, calcium, ammonium, or sulfide. Heavy metals may also be a problem. An understanding of the nature of the toxicity caused by these materials and their control is quite important in evaluating the potential of the anaerobic process for treatment for industrial wastes, and will be considered in more detail in the following article in this issue.

Indicators of Treatment Unbalance

Under normal conditions, anaerobic waste treatment proceeds with a minimum of control. However, if environmental conditions are suddenly changed, or if toxic materials are introduced to the digester, the process may become unbalanced. An "unbalanced digester" is defined as one which is operating at less than normal efficiency. In extreme cases, the efficiency may decrease to almost zero, in which case a "stuck" digester results. It is important to determine when a digester first becomes "unbalanced" so that control measures can be applied before control is lost. A stuck digester is difficult to restart, and, if a supply of seed sludge containing high concentrations of methane bacteria is not available, this may take several weeks.

There is no single parameter which will always tell of the onset of unbalanced conditions, and several parameters must be watched for good control. Several of the parameters of importance are listed in Table 2.

Of the many parameters, the best individual one is that for the concentration of volatile acids. As indicated in the previous article,[1] the volatile acids are formed as intermediate compounds during the complete anaerobic treatment of complex organic materials. The methane bacteria are responsible for destruction of the volatile acids, and if they become affected by adverse conditions, their rate of utilization will slow down, and the volatile acid concentration will increase. A sudden increase in volatile acid concentration is frequently one of the first indicators of digester unbalance and often will indicate the onset of adverse conditions long before any of the other parameters are affected. It should be noted that a high volatile acid concentration is the result of unbalanced treatment and not the cause as is sometimes believed.[5] Thus, a high volatile acid concentration in itself is not harmful, but indicates that some other factor is affecting the methane bacteria.

Another indicator of digester unbalance is a decreasing pH, which usually results from a high volatile acid concentration. A significant drop in pH, however, does not usually occur until the digester is seriously affected, and conditions resulting in a "stuck" digester are near.

Table 2—Indicators of Unbalanced Treatment

Parameters Increasing

Volatile Acids Concentration
CO_2 Percentage in Gas

Parameters Decreasing

pH
Total Gas Production
Waste Stabilization

With some types of toxicity, the first indication is a decrease in total gas production. However, this parameter is useful as an indicator only when the daily feed is quite uniform and the daily gas production does not vary too widely from day to day under normal conditions.

Changes in the percentage of carbon dioxide in the digester gas may sometimes indicate the onset of unbalanced condition,[4] as unbalanced treatment often results in decreased methane production which is accompanied by an increase in carbon dioxide percentage. Another indication of unbalanced conditions is a decrease in efficiency of operation. Such a decrease in efficiency may be evidenced from a drop in methane production per pound of volatile solids added, as frequently determined for municipal sludge, or may be indicated by an increase in effluent COD in the treatment of industrial waste.

Although none of the above parameters may be a sure sign of digester unbalance when used individually, together they give a good picture of digester operation. The best and most significant individual parameter, however, is the volatile acids concentration, and this should always be closely followed.

Cause and Control of Treatment Unbalance

Digester unbalance must be controlled to prevent the serious conditions resulting from a stuck digester. Once the start of an unbalance is detected, the steps listed in Table 3 should be observed.

The first thing to do is control pH near neutrality. Unbalance is usually accompanied by an increase in volatile acids, which, if allowed to go unchecked, may depress the pH below 6. This, in itself, can rapidly result in an inoperable digester, a difficult situation to correct. By maintaining pH, this condition can be prevented. The proper pH can be maintained either by decreasing the waste feed to the digester, if this is possible; or by addition of neutralizing materials such as lime; or both.

Table 3—Steps to Follow in Controlling Unbalance

1. Maintain pH near neutrality.
2. Determine cause of unbalance.
3. Correct cause of unbalance.
4. Provide pH control until treatment returns to normal.

Once the pH is under control, the next item is to determine the cause of the unbalance. The unbalance may be temporary in nature or it may be prolonged, as indicated in Table 4. Temporary unbalance can be caused by sudden changes in temperature, organic loading or the nature of the waste. Such unbalances take place while the bacteria are adjusting to the new conditions. What is needed here is time for the adjustment. By providing optimum environmental conditions and controlling pH, a temporary unbalanced condition will soon correct itself.

A prolonged unbalance may be caused by the introduction of toxic materials to the digester. It may also result from an extreme drop in pH when adequate pH control is not maintained, or may result during initial digester start-up when a sufficient population of methane formers is not present. In all cases the control is much more difficult than if the unbalance is only temporary in nature. If toxic materials have been introduced, pH control alone will not correct the situation. The toxic materials themselves must be removed or controlled. However, pH control will prevent a disastrous drop in pH, and may give additional time to correct the undesirable condition.

If the prolonged unbalance is caused by an extreme drop in pH, and no toxic materials are involved, then pH control alone can correct the situation. However, time for adjustment will be similar to that required during initial process start-up. This may vary from a few weeks to months, as required to allow a new population of methane formers to grow up.

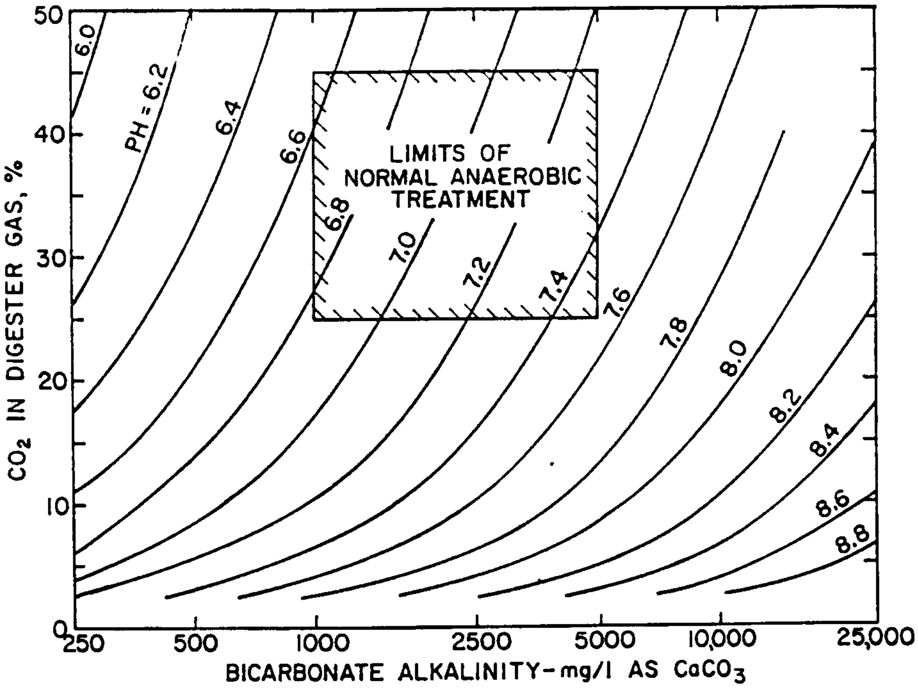
Once the cause of the unbalance is determined and corrected, then the proper pH should be maintained until the system can adjust itself and return to a balanced condition. Because of the various chemical equilibria existing in a digester, pH control can be somewhat difficult unless the factors affecting pH are understood. This is discussed in the following section.

pH Control

The pH of liquor undergoing anaerobic treatment is related to several different acid - base chemical equilibria. However, at the near neutral pH of interest for anaerobic treatment (between 6 and 8) the major chemical system controlling pH is the carbon dioxide-bicarbonate system, which is related to pH or hydrogen ion concentration through the following equilibrium equation:

$$[H^+] = K_1 \frac{[H_2CO_3]}{[HCO_3^-]} \quad \ldots\ldots\ldots\ldots \quad (1)$$

The carbonic acid concentration (H_2CO_3) is related to the percentage of carbon dioxide in the digester gas, K_1 is the ionization constant for carbonic acid, and the bicarbonate ion concentration (HCO_3^-) forms a part of the total alkalinity in the system. Fig. 1 shows the relationship between these factors for anaerobic treatment near 95°F.



■ **FIGURE 1. Relationship between pH and bicarbonate concentration near 95°F.**

The bicarbonate ion concentration or bicarbonate alkalinity is approximately equivalent to the total alkalinity for most wastes when the volatile acid concentration is very low. When the volatile acids begin to increase in concentration, they are neutralized by the bicarbonate alkalinity, and in its place form volatile acid alkalinity.[9] Under these conditions, the total alkalinity is composed of both bicarbonate alkalinity and volatile acid alkalinity. Under these conditions, the bicarbonate alkalinity can be approximated by the following formula:

$$BA = TA - (0.85)(0.833)\,TVA \quad \ldots\ldots\ldots\ldots \quad (2)$$

where:

BA = bicarbonate alkalinity, mg/L as $CaCO_3$,

TA = total alkalinity, mg/L as $CaCO_3$,

TVA = total volatile acid contration, mg/L as acetic acid.

This formula is similar to that used by Pohland and Bloodgood,[9] but includes a factor (0.85) to account for the fact that only 85 percent of the volatile acid alkalinity is measured by titration of total alkalinity to pH 4. The equation also assumes there is no significant concentration of other materials such as phosphates, silicates, or other acid salts which will also produce a significant alkalinity.

Table 4—Factors Causing Unbalanced Treatment

Temporary Unbalance

- Sudden change in temperature.
- Sudden change in organic loading.
- Sudden change in nature of waste.

Prolonged Unbalance

- Presence of toxic materials.
- Extreme drop in pH.
- Slow bacterial growth during start-up.

Fig. 1 indicates that when the bicarbonate alkalinity is about 1,000 mg/L and the percentage of carbon dioxide is between 30 and 40 percent, the pH will be about 6.7. If the bicarbonate alkalinity drops below this value, the pH will drop to undesirable levels. Such a low alkalinity does not give much safety factor for anaerobic treatment, for a small increase in volatile acids will result in a significant decrease in bicarbonate alkalinity and digester pH.

On the other hand, a bicarbonate alkalinity in the more desirable range of 2,500 to 5,000 mg/L provides much "buffer capacity" so that a much larger increase in volatile acids can be handled with a minimum drop in pH.[10] This gives a good factor of safety and allows time for control if an upset results.

If an increase in volatile acid concentration drops the bicarbonate concentration too low as calculated by equation 2, and a serious drop in pH threatens, then the bicarbonate alkalinity should be controlled. This may be done by reducing the feed rate to allow the volatile acids to be utilized and decrease in concentration, or it may be done by the addition of alkaline materials such as lime or sodium bicarbonate.

Liming a Digester

Lime is the most widely used material for controlling pH in anaerobic treatment, mainly because it is readily available and fairly inexpensive. However, occasionally some problems have arisen from its use which are related to the relative insolubility of some of the calcium salts which form in the digester. Because of this problem, close control over lime additions is required, and a knowledge of the solubility problem with lime is helpful.

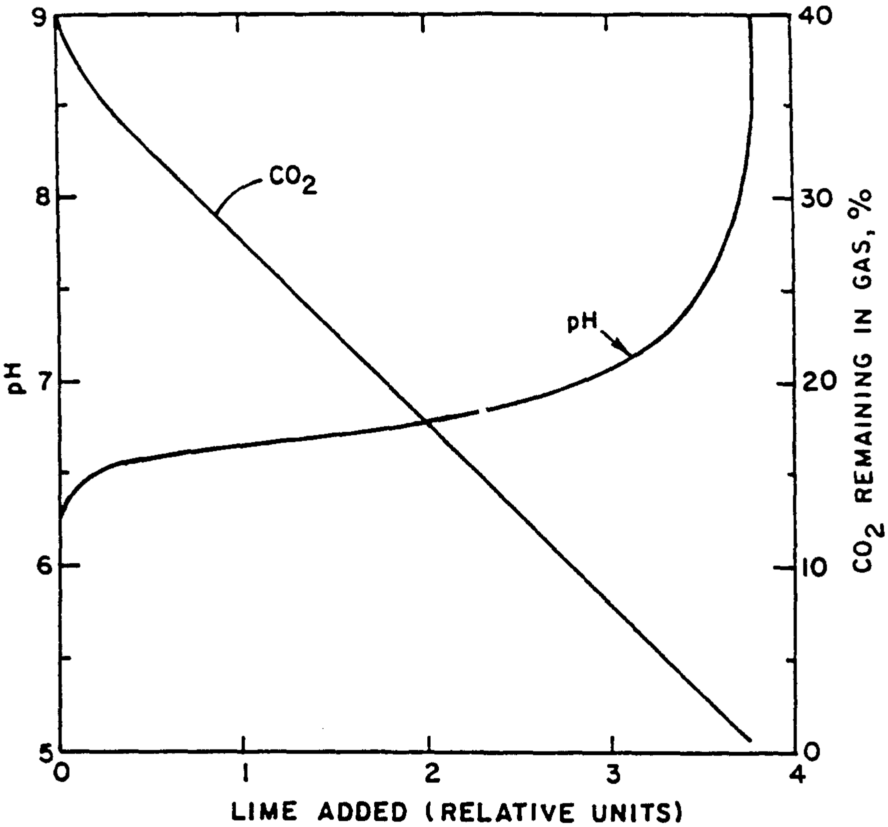
Control of pH is usually considered when it appears likely to drop below 6.5 to 6.6. If lime is then added, it initially increases the bicarbonate alkalinity by combination with the carbon dioxide present as follows:

$$Ca(OH)_2 + 2CO_2 \rightarrow Ca(HCO_3)_2 \quad (3)$$

However, the calcium bicarbonate formed is not very soluble, and when the bicarbonate alkalinity reaches some point between 500 and 1,000 mg/L, additional lime additions result in the formation of the insoluble calcium carbonate as follows:

$$Ca(OH)_2 + CO_2 \rightarrow \underline{CaCO_3} + H_2O \quad (4)$$

Lime additions beyond this point do not increase the soluble bicarbonate alkalinity, and so have little direct effect on digester pH. Fig. 2



■ **FIGURE 2. The effect of lime additions on pH and carbon dioxide percentage.**

is an illustration of what happens to the pH and carbon dioxide percentage in the gas when lime is added after this point is reached. The pH remains between 6.5 and 7, until the CO_2 concentration has decreased to less than about 10 percent by reaction with the lime as indicated in equations 3 and 4. The pH then suddenly increases above 7, and approaches 8 largely as a result of decrease in CO_2 percentage as indicated in Fig. 1. After a short period of time when biological action occurs, the percentage of CO_2 in the gas will begin to increase again. As soon as it exceeds 10 percent, the pH will again drop below 7. This may occur even without the formation of any additional volatile acids. If lime is then added again, the cycle repeats itself.

Thus, nothing beneficial is obtained if additional lime is added to raise the pH above 6.7 to 6.8. After this point, the lime simply combines with the carbon dioxide in the gas to form insoluble calcium carbonate, which precipitates in the digester. This insoluble calcium carbonate is quite ineffective for the neutralization of excessive volatile acids or for raising the pH.

Thus, for effective use of lime, it should not be added until the pH drops below 6.5. A quantity should be added then sufficient only to raise the pH to about 6.7 to 6.8. Once the lime is added, the pH in the digester must be closely watched. As soon as it drops below a value of 6.4 to 6.5, additional lime additions must be made. If this procedure is followed, and pH is closely watched, then lime can serve as a cheap and effective method for controlling pH. Good mixing of the lime is required in the digester and caution must be excercised to prevent the creation of a vacuum from the removal of the carbon dioxide from the gas by combination with the lime.

Sodium Bicarbonate for pH Control

Sodium bicarbonate, although seldom used, is one of the most effective materials for pH control in anaerobic treatment. This material has significant advantages over other materials. It is relatively inexpensive when purchased in large quantities. It does not react with carbon dioxide to create a vacuum in the digester, and there is little danger that it will raise the pH to undesirable levels. It is quite soluble and can be dissolved prior to addition to the digester for more effective mixing. This material can be added to give alkalinity in the digester of 5,000 to 6,000 mg/L without producing any adverse or toxic effects. Although it is more expensive than lime, less quantities are required because it does not precipitate from solution. The ease of control, addition, and handling, make it a very desirable material for pH control in digesters. It is expected this material will be used more in the future.

Conclusion

The successful control of the anaerobic treatment process depends upon a knowledge of the various environmental factors which affect the microorganisms responsible for waste degradation. Of the various factors, pH is one of the most important to control. This control depends upon the maintenance of an adequate bicarbonate buffer system both to counteract the acidity of the carbon dioxide and that of organic acids produced during anaerobic treatment. It is also important to control materials which may produce an adverse environment for the anaerobic microorganisms. The toxicity which may be caused by common materials as well as their control will be discussed in the next article in this series.

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Anaerobic Waste Treatment Fundamentals

PART THREE | Toxic Materials and their Control

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THE ANAEROBIC process is widely used for treatment of municipal waste sludges and has excellent potential for treatment of many industrial wastes. Recent research has helped to explain the complex chemistry and microbiology of anaerobic treatment, and this should stimulate further application of the process to waste treatment. This series of articles is intended to summarize our current knowledge of anaerobic treatment fundamentals, design and control. The part that follows is concerned with toxic materials and control.

There are many materials, both organic and inorganic, which may to toxic or inhibitory to the anaerobic waste treatment process. The term "toxic" is relative and the concentration at which a material becomes toxic or inhibitory may vary from a fraction of an mg/L to several thousand mg/L. Fig. 1 indicates the general effect which results from the addition of most substances to biological systems. At some very low concentration, stimulation of activity is usually achieved. This stimulatory concentration may range from only a fraction of an mg/L for heavy metal salts to over one hundred mg/L for sodium or calcium salts. As the concentration is increased above the stimulatory concentration, the rate of biological activity begins to decrease. A point is then reached where inhibition is apparent and the rate of biological activity is less than that achieved in the absence of the material. Finally, at some high concentration, the biological activity approaches zero.

Microorganisms usually have the ability to adapt to some extent to inhibitory concentrations of most materials. The extent of adaptation is relative, and in some cases the activity after acclimation may approach that obtained in the absence of the inhibitory material, and in other cases the acclimation may be much less than this.

Control of Toxicity or Inhibition

It is desirable to control inhibitory or toxic materials to achieve higher efficiencies or more economical operation of the waste treatment systems. Table 1 lists some methods which may be used in this control.

Removal of toxic materials from the waste stream or dilution of the waste below the "toxic threshold" of the material are the most obvious solutions, although not always the easiest to perform.

The removal of the toxic material from solution by precipitation or complex formation will control toxicity resulting from some materials. This makes use of the principle that only materials in solution can be toxic to biological life. In some cases, addition of an antagonistic material may be beneficial. An "antagonist" is a material which, when added, will decrease or antagonize the toxicity of another material. Little is known about how an antagonist works, but in some cases their use can be very effective.

Not all of the above methods are applicable in all cases. However,

■ **FIGURE 1. General effect of salts or other materials on biological reactions.**

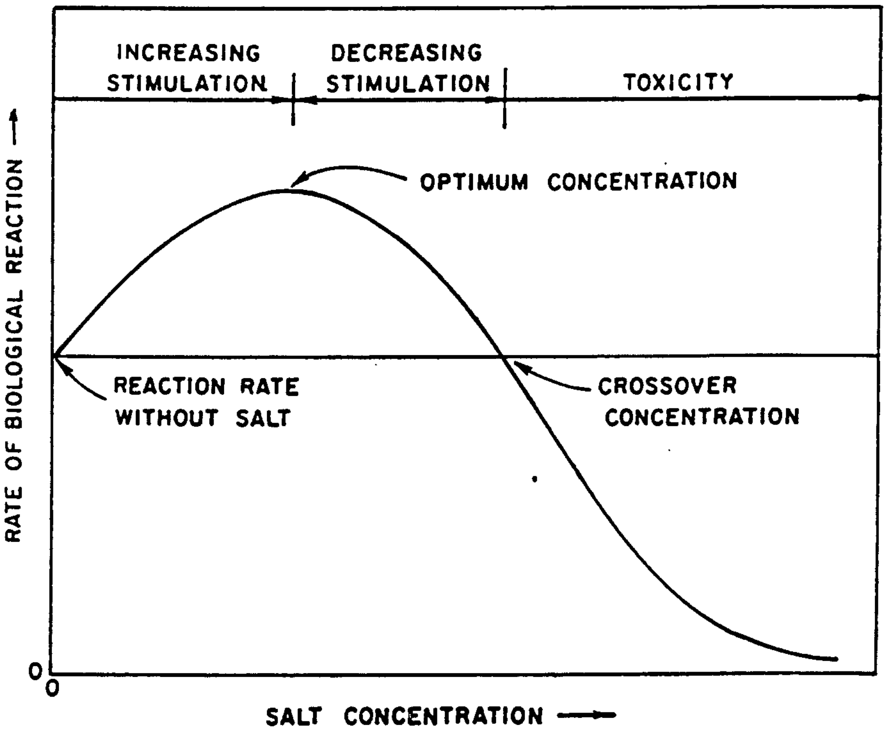


Table 1—Possible Methods to Control Toxic Materials

1. Remove toxic material from waste.
2. Dilute below toxic threshold.
3. Form insoluble complex or precipitate.
4. Antagonize toxicity with another material.

Table 2—Stimulatory and Inhibitory Concentrations of Alkali and Alkaline-Earth Cations

Concentrations in mg/L

Cation	Stimulatory	Moderately Inhibitory	Strongly Inhibitory
Sodium	100—200	3500—5500	8,000
Potassium	200—400	2500—4500	12,000
Calcium	100—200	2500—4500	8,000
Magnesium	75—150	1000—1500	3,000

most inhibition can be controlled by either one or a combination of these procedures.

Alkali and Alkaline-Earth Salt Toxicity

The concentrations of alkali and alkaline earth-metal salts such as those of sodium, potassium, calcium or magnesium, may be quite high in industrial wastes, and are frequently the cause of inefficiency in, or failure of, anaerobic treatment. In municipal waste sludge, however, the concentration of these salts is normally sufficiently low so they will not cause a problem, unless introduced at high concentration for pH control. It has been found that toxicity is normally associated with the cation, rather than the anion portion of the salt. The nature of the inhibitory effect of these salts is quite complex, but general guidelines can be given to indicate when inhibition may be suspected, and how it may be controlled.

Listed in Table 2 are concentrations of the cations of these salts which may be stimulatory and those which may be inhibitory to anaerobic treatment.[1, 2] The concentrations listed as stimulatory are those which are desirable and will permit maximum efficiency of the process.

The concentrations listed as moderately inhibitory are those which normally can be tolerated but require some acclimation by the microorganisms. When introduced suddenly, these concentrations can be expected to retard the process significantly for periods ranging from a few days to over a week.

Concentrations listed as strongly inhibitory are those which will normally retard the process to such an extent that the efficiency will be quite low, and time required for effective treatment may be excessively long. Such concentrations are normally quite undesirable for successful anaerobic treatment.

When combinations of these cations are present, the nature of the effect becomes more complex as some of the cations act antagonistically, reducing the toxicity of other cations, while others act synergistically, increasing the toxicity of the other cations.

If an inhibitory concentration of one cation is present in a waste, this inhibition can be significantly reduced if an antagonistic ion is present or is added to the waste. Sodium and potassium are the best antagonists for this purpose and are most effective if present at the stimulatory concentrations listed in Table 2. Higher concentrations are not so effective, and if too high, will actually increase the toxicity.

Calcium and magnesium are normally poor primary antagonists and when added will normally increase rather than decrease the toxicity caused by other cations. However, they may become stimulatory if another antagonist is already present. For example, it has been found that 7,000 mg/L of sodium may significantly retard anaerobic treatment. If 300 mg/L of potassium is added, this retardation may be reduced by 80 percent. If 150 mg/L of calcium is then added, the inhibition may be completely eliminated. However, if calcium were added in the absence of potassium, no beneficial effect at all would be achieved.

Antagonists are best added as the chloride salts. If such additions are not sufficiently beneficial or economical then the best solution to a toxic salt concentration may be dilution of the waste.

Ammonia Toxicity

Ammonia is usually formed in anaerobic treatment from the degradation of wastes containing proteins or urea. Inhibitory concentrations may be approached in industrial wastes containing high concentrations of these materials or in highly concentrated municipal waste sludges.

Ammonia may be present during treatment either in the form of the ammonium ion (NH_4^+) or as dissolved ammonia gas (NH_3). These two forms are in equilibrium with each other, the relative concentration of each depending upon the pH or hydrogen ion concentration as indicated by the following equilibrium equation:

$$NH_4^+ \rightleftarrows NH_3 + H^+ \ldots (1)$$

When the hydrogen ion concentration is sufficiently high (pH of 7.2 or lower), the equilibrium is shifted to the left so that inhibition is related to the ammonium ion concentration. At higher pH levels, the equilibrium shifts to the right and the ammonia gas concentration may become inhibitory. The ammonia gas is inhibitory at a much lower concentration than the ammonium ion.

The ammonia nitrogen analysis gives the sum total of the am-

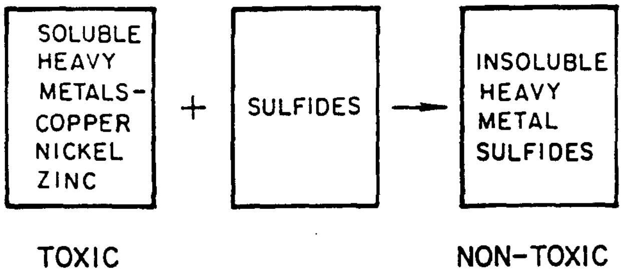
Table 3—Effect of Ammonia Nitrogen on Anaerobic Treatment

Ammonia Nitrogen Concentration mg/L	Effect on Anaerobic Treatment
50— 200	Beneficial
200—1000	No adverse effect
1500—3000	Inhibitory at higher pH values
Above 3000	Toxic

monium ion plus ammonia gas concentrations.

In Table 3 are listed the ammonia nitrogen concentrations which may have an adverse effect on anaerobic treatment.[3, 4]. If the concentration is between 1,500 and 3,000 mg/L, and the pH is greater than 7.4 to 7.6, the ammonia gas concentration can become inhibitory. This condition is characterized by an increase in volatile acid concentration which tends to decrease the pH, temporarily relieving the inhibitory condition. The volatile acid concentration here will then remain quite high unless the pH is depressed by some other means, such as by adding hydrochloric acid to maintain the pH between 7.0 and 7.2

When the ammonia-nitrogen concentration exceeds 3,000 mg/L, then the ammonium ion itself becomes quite toxic regardless of pH and the process can be expected to fail. The best solution then is either dilution or removal of the source of ammonia-nitrogen from the waste itself.



QUANTITY OF SULFIDE SALTS REQUIRED FOR PRECIPITATION

SULFIDE SALT ADDED	CONCENTRATION OF HEAVY METALS PRECIPITATED
1 mg/l SULFIDES ($S^=$)	1.8–2.0 mg/l
1 mg/l SODIUM SULFIDE (Na_2S)	0.75–0.84 mg/l
1 mg/l SODIUM SULFIDE ($Na_2S \cdot 9H_2O$)	0.24–0.27 mg/l

■ **FIGURE 3. The control of heavy metal toxicity by precipitation with sulfides.**

Sulfide Toxicity

Sulfides in anaerobic treatment can result from 1) introduction of sulfides with the raw waste and/or 2) biological production in the digester from reduction of sulfates and other sulfur-containing inorganic compounds, as well as from anaerobic protein degradation. Sulfate salts usually represent the major precursors of sulfides in industrial wastes.

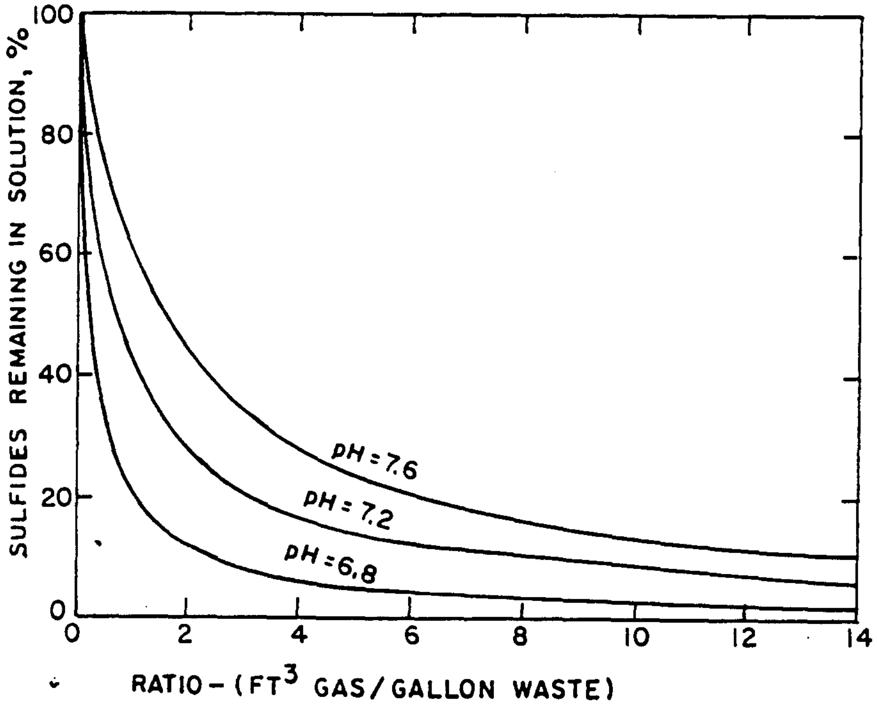
Sulfides produced in anaerobic treatment may exist in a soluble or insoluble form, depending upon the cations with which they become associated. Heavy metal sulfides are insoluble and precipitate from solution so they are not harmful to the microorganism. The remaining soluble sulfide forms a weak acid which ionizes in solution, the extent depending upon the pH. Also, because of limited solubility of hydrogen sulfide, a certain portion of that formed will escape with the digester gas produced. Thus, sulfides may be distributed between an insoluble form, a soluble form, and gaseous hydrogen sulfide.

The actual distribution of sulfides depends upon digester pH and the quantity of gas produced from the waste as shown in Fig. 2.[5] The higher the gas production per gallon of waste, the higher will be the amount of sulfides driven from solution as a gas, and the lower the concentration remaining in solution.

For example, if the concentration of soluble sulfide precursors in a waste entering a digester were 800 mg/L as sulfur, the pH were 7.0, and three cubic feet of gas were produced per gallon of waste added, only about 20 percent, or 160 mg/L of sulfides would remain in solution in the digester. The remainder, or 640 mg/L would escape with the other gases produced during treatment.

Concentrations of soluble sulfide varying from 50 to 100 mg/L, can be tolerated in anaerobic treatment with little or no acclimation required. With continuous operation and some acclimation, concentrations up to 200 mg/L of soluble sulfides can be tolerated with no significant inhibitory effect on anaerobic treatment. Thus, the 160 mg/L, remaining in the example above could be tolerated. Concentrations above 200 mg/L, however, are quite toxic.

■ **FIGURE 2. Graph showing the effect of gas production and pH on the fraction of soluble sulfides formed which remain in solution in the waste during treatment.**



Toxic concentrations of sulfide may be reduced by gas scrubbing, use of iron salts to precipitate sulfides, dilution of the waste, or separation of sulfate or other sulfur containing streams from the waste to be treated.

Heavy Metal Toxicity

The heavy metals have been blamed for many digester failures. Low, but soluble, concentrations of copper, zinc and nickel salts are quite toxic and these salts are associated with most of the problems of heavy metal toxicity in anaerobic treatment. Hexavalent chromium can also be toxic to anaerobic treatment. However, this metal ion is normally reduced to the trivalent form which is relatively insoluble at normal digester pH levels and consequently is not very toxic.[6] Iron and aluminum salts are also not toxic because of their low solubility.

Concentrations of the more toxic heavy metals (copper, zinc and nickel) which can be tolerated are related to the concentration of sulfides available to combine with the heavy metals to form the very insoluble sulfide salts, as indicated in Fig. 3.[7,8] Such salts are quite inert and do not adversely affect the microorganisms. When the sulfide concentration available for this precipitation is low, only small quantities of heavy metals can be tolerated. However, when the concentration of sulfides is very high, then relatively high concentrations of heavy metals can be tolerated with no detrimental effects.

It is interesting to note that sulfides, by themselves, are quite toxic to anaerobic treatment, as are the heavy metals. However, when combined together, they form insoluble salts which have no detrimental effect.

One mole of sulfide is required per mole of heavy metals for precipitation. The heavy metals, copper, zinc, and nickel, have molecular weights ranging from 58 to 65, while that for sulfur is 32. Thus, about one-half milligram per liter of sulfide is required to precipitate one milligram per liter of these heavy metals.

Sufficient sulfide must be available to precipitate all the heavy metals. If sufficient sulfide is not formed during waste treatment, then sodium sulfide, or a sulfate salt, which will be reduced to sulfide under anaerobic conditions, may be added. This is one of the most effective procedures for control of this type of toxicity. Sodium sulfide can be easily added and from this the possibility of upset by heavy metals can be readily ascertained.

Toxic Organic Materials

The preceding discussion includes most materials which may be suspected of causing digester upsets, or of preventing satisfactory treatment of a waste. There are also many organic materials which may inhibit the digestion process. These range from organic solvents to many common materials such as the alcohols and long-chain fatty acids. Organic materials which are toxic at high concentration, but which can be anaerobically treated at low concentration, can be adequately handled by continuous feed to the treatment unit. By continuous feed, these materials are degraded as rapidly as they are added, and the concentrations actually in the digester can be maintained very low, well below that of the feed itself. For example, methanol may be detrimental to anaerobic treatment in concentrations of about 1,000 to 2,000 mg/L. However, concentrations as high as 10,000 mg/L have been treated successfully by continuous feed.

Other toxic organic materials can be treated successfully if they can be precipitated from solution. For example, sodium oleate, a common fatty acid which forms a base for ordinary soap, was found to inhibit anaerobic treatment in concentrations over 500 mg/L. However, by adding calcium chloride, the insoluble calcium oleate salt was formed, which could be treated successfully even when the concentration in the digester exceeded 2,000 to 3,000 mg/L. Fatty acids normally are present in municipal waste sludges as the insoluble calcium salt and thus do not adversely affect the anaerobic treatment process.

Summary

There are many materials which may produce an adverse environment for the anaerobic microorganisms. Usually, these materials are not present in significant concentrations in municipal waste sludges. However, they frequently occur in industrial waste and may reach municipal plants from this source. They also may present a problem in the direct anaerobic treatment of many industrial wastes. A knowledge of these materials, their inhibitory concentrations, and their chemistry, should help quickly to evaluate the potential effect of these materials and lead to effective measures for their control. The next and last article in this series will discuss the various factors related to anaerobic waste treatment design. □□□

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A Brighter Pittsburgh

Pittsburgh, Pennsylvania, will spend more than a quarter of a million dollars in 1964 to place new mercury vapor street lights on 39 miles of arteries and main business thoroughfares. This amount is nearly three times the usual annual expenditure for street lighting improvements. In reporting on the program, Fred S. Poorman, Director of Public Works, said: "The new lighting system which we started here in 1961 is one of the most popular of public improvements. Both mayor and council have been deluged with requests for new fixtures. There is no question that neighborhood morale rises and the image of Pittsburgh to outsiders is considerably brightened by this kind of program. Improved lighting is a stamp of a progressive community." Westinghouse luminaires with Lifeguard electrodes will be used in the modernization.

Anaerobic Waste Treatment Fundamentals

PART FOUR | Process Design

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THE ANAEROBIC waste treatment process has been widely used for the stabilization of concentrated sludges at municipal waste treatment plants, and has also been used to a limited extent for the treatment of industrial wastes. During the last decade, a better understanding of the process has been obtained and the significant advantages offered by this process have become more evident. Because of this, the anaerobic process is expected to receive wider usage for the treatment of industrial wastes in the future.

The first three articles in this series[1,2,3] were concerned with the microbiology, chemistry and operating parameters for anaerobic treatment. This last article will summarize the fundamental considerations in anaerobic waste treatment plant design. This will be directed mainly toward process design for the treatment of industrial wastes, although the principles apply equally well to design of municipal waste treatment plants.

Waste Characteristics

Practical experience in the anaerobic treatment of industrial wastes is still fairly limited and so caution needs to be exercised in the design of full-scale treatment facilities until preliminary pilot plant studies have been conducted. There is, however, a sufficient understanding of the principles involved so that the potential feasibility of the process may be evaluated from a knowledge of a few basic chemical characteristics of a waste under consideration. This preliminary evaluation will indicate the best type of treatment system to use, and will allow estimation of biological solids production, nutrient requirements, methane gas production, and heat requirements. A summary of the important waste characteristics is shown in Table 1.

One important characteristic is the waste strength in terms of the concentration of biologically degradable organics it contains. This is best measured as the ultimate BOD (BOD_L), which may be roughly estimated from the waste COD or the waste 5-day BOD (BOD_5). The COD normally gives a high measure of BOD_L, as it measures organic materials that are not biodegradable as well as those that are. The BOD_5, when multiplied by an appropriate constant (1.5 is commonly used), may also give a fair indication of BOD_L. However, the value obtained may be low as some materials, such as cellulose, are not degraded readily under the aerobic conditions of the BOD test, but are quite susceptible to anaerobic treatment. The best indication of organic waste strength is that given by laboratory anaerobic waste treatment studies. An indication of the relative concentration of carbohydrates, proteins, and fats in the waste is also helpful in anaerobic treatment evaluation.

The alkalinity or buffering capacity of the waste is another important parameter, as this affects the pH. The pH must be near neutral for satisfactory treatment, and this requires a bicarbonate alkalinity of at least 1500 mg/L for waste treatment in the presence of an atmosphere containing about 30 percent carbon dioxide.[2] A higher alkalinity of 3,000 to 4,000 mg/L is more desirable, as it gives better cushion against a drop in pH resulting from excessive volatile acid increase. The alkalinity of importance is that of the waste during treatment, which is not necessarily the same as that of the raw waste. Certain materials, such as proteins, release ammonia nitrogen during biodegradation, and this combines with carbon dioxide and water to form ammonium bicarbonate alkalinity. Alkalinity of such a waste will thus increase during treatment. This is the case with municipal waste sludges. An analysis for organic nitrogen will indicate the potential for formation of this type of alkalinity. Wastes with insufficient alkalinity will require supplementation. Sodium bicarbonate is the best supplement, but lime or ammonium bicarbonate may also be used if added with caution.[2].

Another characteristic of the waste of importance is the concentration of inorganic nutrients, nitrogen and phosphorus, present. These materials are required for the growth of the microorganisms responsible for treatment. Nitrate or nitrite nitrogen is unavailable for growth under anaerobic conditions, as it is reduced to nitrogen gas, and lost from the waste. Ammonia nitrogen and the portion of the organic nitrogen released during waste degradation are the forms used under anaerobic conditions for biological growth. All forms of inorganic phosphorus and the portion of organic phosphorus released during waste degradation are all normally suitable for biological use.

Another important waste characteristic is its temperature. This is

Table 1—Important Waste Characteristics for Anaerobic Treatment Evaluation

1. Organic strength and composition.
2. Alkalinity.
3. Inorganic nutrient content.
4. Temperature.
5. Content of potentially toxic materials.

Table 2—Growth Constants and Endogenous Respiration Rates

(*after Speece*[1])

Waste	Growth Constant a	Endogenous Respiration Rate b
Fatty Acid	0.054	0.038
Carbohydrate	0.240	0.033
Protein	0.076	0.014

Table 3—Design for Solids Retention Times

Solids Retention Times, Days

Operating Temperature °F	Minimum	Suggested for Design
65	11	28
75	8	20
85	6	14
95	4	10
105	4	10

especially true for dilute waste, for which the methane production may be insufficient to heat the waste to a temperature high enough for optimum rates of treatment. It is highly desirable to have a warm waste and any design features which would insure this should be given due consideration.

The last important characteristic for evaluation of a waste ~~is its~~ content of potentially toxic materials, such as the inorganic ions sodium, ~~potassium, magnesium,~~ calcium, or the heavy metals; such as copper, zinc, nickel or lead. Toxic concentrations of these materials and their control were discussed in the third article in this series.[3] Dilution of the waste may be required if the concentrations of these materials are too high, and if other control procedures are not feasible. Such a solution is not desirable from an economic standpoint and should be avoided if possible. Once the above waste characteristics are estimated, the feasibility of the anaerobic process for treatment of the waste can be ascertained. The considerations of importance are discussed in the following.

Methane Production and Heat Requirements

The rate of anaerobic treatment increases with temperature up to about 95 to 100°F. Beyond that, the rate does not increase significantly, and in fact may decrease until a temperature in the thermophilic range near 130°F is reached. Although higher rates of treatment are possible at thermophilic temperatures, practical considerations indicate that more reliable operation can be expected at mesophilic temperatures of about 95°F.

In anaerobic treatment the methane gas produced is an important source of fuel for raising the temperature to a more desirable operating level. Unfortunately, dilute wastes do not usually produce sufficient methane to increase their temperatures significantly. Thus, these wastes must usually be treated at their incoming temperature, as it is usually uneconomical to heat them by use of an external heat supply.

Methane production may be estimated from waste strength by use of the following formula:

$$C = 5.62(eF - 1.42A) \quad \ldots (1)$$

where: C = cubic feet of CH_4 produced per day (STP),
e = efficiency of waste utilization,
F = pounds of BOD_L added per day,
A = pounds volatile biological solids produced per day.

The value 5.62 is the theoretical methane production from stabilization of one pound of BOD_L,[1] and the constant 1.42 is the factor for conversion of pounds of volatile biological solids to BOD_L. The efficiency of waste utilization (*e*) normally ranges from 0.80 to 0.95 under satisfactory operating conditions.

Figure 1 indicates the increase in waste temperature which might be achieved if the methane gas produced from waste treatment were used for waste heating. One cubic foot of methane (STP) has a net heating value of 960 Btu. The values shown were calculated using $e = 0.90$, and $A = 0.1F$. An efficiency of heat transfer from the burning of methane of 80 percent was also used. Heat losses from the conversion of pounds of volatile biin these calculations. The curve in this figure indicates that organic waste concentrations of 5,000 mg/L or above are required before methane production could be sufficient to raise the waste temperature significantly. Thus, wastes with organic concentrations less than 2,000 to 5,000 mg/L must be warm to begin with for optimum treatment, or else must be treated at less than the optimum temperatures.

Nutrient Requirements

In anaerobic treatment, a portion of the organic waste is converted to biological cells, while the remainder is stablized by conversion to methane and carbon dioxide. It is necessary to determine the fraction converted to cells so the methane production can be estimated, and the quantity of nitrogen and phosphorus required for biological growth can be determined. A figure showing the growth of microorganisms as a function of biological solids retention time was given previously.[1] Such a growth can also be approximated by the following formula:

$$A = \frac{aF}{1 + b(SRT)} \quad \ldots\ldots (2)$$

where: A = pounds volatile biological solids produced per day,
F = pounds BOD_L added per day,
SRT = solids retention time in days,[1]
a = growth constant,
b = endogenous respiration rate.

Values for *a* and *b* as found for various wastes are shown in Table 2. The growths obtained from carbohydrate are much higher than those obtained with protein or fatty acid type waste. Waste containing a combination of these materials will have biological growth intermediate between these two extremes. Growth is also less at long sludge retention times.

The quantity of the biological ~~nutrients, nitrogen~~ and phosphorus, ~~required by~~ the microorganisms is ~~directly proportional~~ to their ~~growth.~~ ~~The daily~~ nitrogen requirement is equal to about 0.11A, while the phosphorus requirement is equal to about 0.02A. If these quantities of nutrients are not present in the waste, then they must be added for satisfactory treatment.

BOD Stabilization

BOD may be removed in anaerobic treatment by conversion of organic matter to methane gas, or by separation of BOD producing bacterial cells and suspended solids from the treated effluent. Only that portion converted to methane gas is actually stabilized, and the suspended solids portion removed must undergo further processing for final disposal. One significant advantage of anaerobic treatment is that a relatively high percentage of the organic matter is stabilized by conversion to methane gas, even at high loadings. The percentage of added BOD_L which is stabilized (S) is given by the following formula:

$$S = \frac{100C}{5.62F}$$

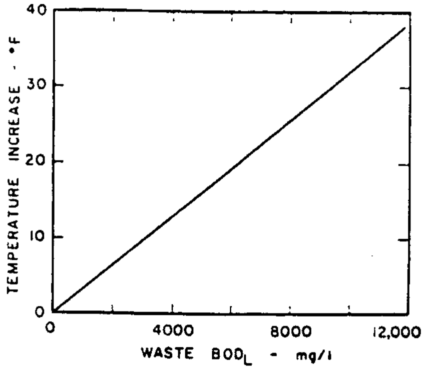
$$= \frac{100(eF - 1.42A)}{F} \quad \ldots (3)$$

Figure 2 shows the relationship between methane production and BOD_L stabilization per 1,000 cubic feet of digester tank volume per day.

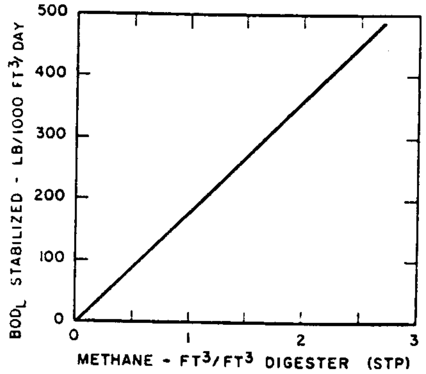
The efficiency of anaerobic treatment is related to the solids retention time (SRT).[1] As the retention time is decreased, the percentage of microorganisms wasted from the digester each day is increased. At some minimum SRT, the microorganisms are wasted from the system faster than they can reproduce themselves and failure of the process results. This minimum SRT is dependent upon temperature as shown in Table 3. Although operation near the minimum SRT is possible, the efficiencies are low and process dependability is poor. It is recommended that design SRT be at least 2½ times the minimum, as indicated in Table 3. More reliability, but little increase in efficiency, is obtained at longer SRT. Ninety to ninety-five percent of maximum efficiency should be obtained at the design SRT shown.

Process Design

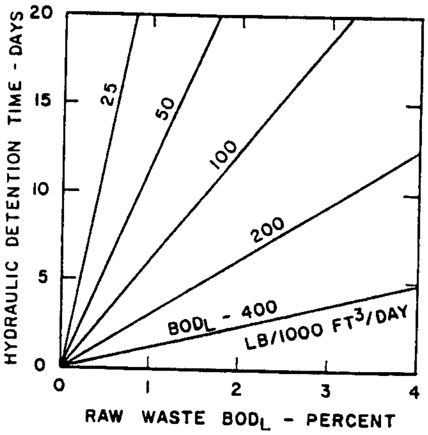
Two major processes are available for anaerobic treatment, the conventional process and the anaerobic contact process.[1] The conventional process is simpler, as it involves one tank in which the bacteria and waste are mixed together for treatment. The bacteria and the treated waste stream are removed together for disposal. For this process the hydraulic detention time and solids retention time are essentially the same. Here, BOD removal is equal to the BOD stabilized by conversion to methane gas, unless further provision is made to separate the effluent solids from the effluent stream. This process or a similar modification is presently used for treatment of concentrated wastes.



■ FIGURE 1. Maximum increase in waste temperature obtained by using methane produced for waste heating.



■ FIGURE 2. Relationship between methane production and stabilization.



■ FIGURE 3. Relationship between loading and hydraulic detention time.

The anaerobic contact process is designed to treat economically dilute organic wastes. In this system, a settling tank follows the digester so that the bacteria can be removed from the effluent stream, and recycled back into the digester. In this case, a short hydraulic detention time can be used, while maintaining the long SRT required for adequate treatment as given in Table 3.

Figure 3 indicates the relationship between raw waste organic concentration, organic loading, and hydraulic detention time. This figure shows that for a given waste concentration, a higher organic loading can only be obtained by decreasing the hydraulic detention time. The conventional process is applicable as long as the hydraulic detention time is greater than the minimum SRT listed in Table 3. The anaerobic contact process should be used whenever the desired organic loading requires a hydraulic detention time less than the recommended SRT.

BOD_5 loadings normally used vary from about 50 to 250 lb/1,000 cu ft./day. In general, the anaerobic contact process becomes the one of choice for wastes with organic concentrations less than one percent.

The major problem arising from use of the anaerobic contact process to date is related to an inability to separate efficiently the bacterial solids from the effluent stream for recycle back to the digester. High efficiency is necessary to maintain the required long sludge retention times while operating at short hydraulic detention times. In the successful full-scale treatment of meat-packing wastes,[5] a vacuum degasifier has been used between the digester and final settling tank to remove gases which tend to float the solids rather than allowing them to settle in the settling tank. Either this scheme, or hopefully even better ones, are needed for high efficiency of effluent solids separation, which is required for the successful treatment of cool and dilute wastes by the anaerobic treatment process. The recycle rates used for return of biological solids in the anaerobic contact process has to date been quite high, usually in the range of 2:1 to 4:1 based on recycle flow rate to raw waste flow rate. Such high rates are required also because of the solids separation problem.

Operational Data

A summary of data reported from treatment of wastes by the anaerobic contact process are shown in Table 4 and by the conventional process are shown in Table 5. The data are from laboratory and pilot plant studies as well as from full-scale plant operation. For the anaerobic contact process, successful operation has been reported with BOD_5 loadings varying from 74 to 730 lb/1,000 cu. ft./day. In two cases, successful treatment was reported with temperatures of only about 75° F and BOD_5 loadings of about 100 lb/1,000 cu. ft./day. Successful treatment by this process has been reported to date only for wastes with BOD_5 concentrations greater than 1,000 mg/L.

The values for BOD_5 stabilized listed both in Tables 4 and 5 were computed from reported or estimated values of methane production. The BOD_5 stabilized values are shown for comparative purposes and were estimated by multiplying the BOD_L values obtained from Fig. 2 by 0.67. From Table 4 for the anaerobic contact process, the BOD_5 stabilized varies from about 60 to 90 percent of the BOD_5 removed, with an average of about 75 percent. This is quite high for such highly loaded systems and indicates one of the advantages of the anaerobic treatment process.

In Table 5 results are listed from operation of the conventional process. Results have usually been expressed in terms of volatile solids loadings, rather than BOD_5 loadings. However, the computed values for BOD_5 stabilized indicates the BOD_5 loadings must have been very high in many cases, much higher than normally considered desirable or possible with aerobic treatment. These data indicate the potential of the anaerobic treatment process for the stabilization of industrial wastes.

Future Research Needs

The anaerobic treatment process has been successfully used for the treatment of both municipal and industrial wastes. However, in order to obtain its full potential, certain technological developments are yet required. One of these has already been mentioned, the need for better methods of solids separation to efficiently remove the bacteria from the effluent streams and return them to the treatment system. This will allow successful treatment of very dilute wastes and at low temperatures.

The other development which is required for successful treatment of many industrial wastes is a better understanding of the complete nutritional requirements of the methane bacteria, which are the limiting organisms around which the process must be designed. Meat packing wastes and municipal sewage sludge are well balanced nutritionally for maximum bacterial growth. However, many industrial wastes are not. The exceptionally high rates for treatment of winery waste listed in Table 4 and acetic acid and butyric acid waste listed in Table 5 were obtained by addition to the digester of some of the nutrient materials other than nitrogen and phosphorus contained in digested municipal sludge. Without this addition, these high rates were not possible. However, there has been limited success to date in determining just which materials in this digested sludge were responsible for stimulation of the methane bacterial growth. Iron in concentrations from 20 to 60 mg/L has been found beneficial,[4] however, other inorganic or organic stimulants are also needed to obtain the exceptionally high rates shown. Several laboratories are now working on this phase of anaerobic treatment because of its importance to the future of the process. Hopefully an answer is near.

Summary

Because of the present limited practical experience with the anaerobic process for the treatment of industrial wastes, pilot plant studies should be conducted before full scale design is undertaken. However, a preliminary evaluation of the type of system to design, additional nutrient requirements, and expected degree of waste treatment and stabilization can be made based on a few basic waste characteristics. The anaerobic waste treatment process is now sufficiently well understood so that many of the common treatment problems which may arise can

Table 4—Anaerobic Treatment Performance for the Contact Stabilization Process

Waste	Hydraulic Detention Time Days	Digestion Temperature °F	Raw Waste mg/L	BOD_5 lb./1,000 cu. ft./Day Added	Removed	Stabilized	Percent Removed	Reference
Maize Starch	3.3	73	6,280	110	97	85	88	6
Whisky Distillery	6.2	92	25,000	250	237	164	95	7
Cotton Kiering	1.3	86	1,600	74	50	42	67	8
Citrus	1.3	92	4,600	214	186	141	87	9
Brewery	2.3	. . .	3,900	127	122	. . .	96	10
Starch-Gluten	3.8	95	14,000*	100*	80*	. . .	80*	11
Wine	2.0	92	23,400*	730*	620*	735	85*	12
Yeast	2.0	92	11,900*	372*	242*	146	65*	12
Molasses	3.8	92	32,800*	546*	376*	222	69*	12
Meat-Packing	1.3	92	2,000	110	104	77	95	8
Meat-Packing	0.5	92	1,380	156	142	66	91	5
Meat-Packing	0.5	95	1,430	164	156	. . .	95	13
Meat-Packing	0.5	85	1,310	152	143	. . .	94	13
Meat-Packing	0.5	75	1,110	131	119	. . .	91	13

*Volatile suspended solids, rather than BOD_5.

be anticipated before they occur and can be controlled when they do develop. The process has several advantages over anaerobic treatment for wastes with BOD_5 concentrations greater than 1,000 mg/L. When effective methods for solids separation are developed and the nutritional requirements for maximum growth of the microorganisms are understood, then the full potential of the process for treatment of dilute wastes and at low temperatures can be realized. □□□

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Table 5—Anaerobic Treatment Performance for the Conventional Process

Waste	Hydraulic Detention Times Days	Digestion Temperature °F	lb./1,000 cu. ft./Day				Reference
			Volatile Solids Added	Volatile Solids Stabilized	BOD_5 Added	BOD_5 Stabilized	
Pea Blancher	3.5	131	700	582	. . .	510	14
Pea Blancher	6.0	99	400	340	. . .	288	14
Winery	. . .	97	200	174	. . .	212	15
Butanol	10.0	. . .	190	. . .	110	75	16
Rye Fermentation	2.0	130	930*	. . .	. . .	500	16
Corn Fermentation	4.0	130	330*	. . .	. . .	250	16
Whey Waste	29.0	130	150*	. . .	. . .	107	16
Sewage Sludge	7.0	95	440	158	. . .	207	17
Sewage Sludge	3.2	97	870	357	. . .	394	18
Sewage Sludge	12.0	90	300	139	. . .	159	19
Sewage Sludge	12.0	108	300	141	. . .	107	19
Sewage Sludge	12.0	126	300	146	. . .	138	19
Acetic Acid	30.0	95	1,370	. . .	975	876	20
Butyric Acid	30.0	95	830	. . .	1,000	910	20

*Total Solids.

ANAEROBIC TREATMENT OF LOW STRENGTH WASTEWATER

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Abstract- Anaerobic filters (AFs) and upflow anaerobic sludge blanket (UASB) reactors are finding wide-scale acceptance for treating various types of wastewater. They are frequently used for medium to high strength wastewater (2,000 to 20,000 mg/L COD), but have fewer applications to low strength wastewater (< 1,000 mg/L COD). In order to understand the applicability of anaerobic treatment for low strength wastewater, such as domestic sewage, a literature review was performed and a dynamic mathematical model was developed. The review showed two main variations of anaerobic wastewater treatment techniques (AF and UASB) and a number of modifications of these two themes. A total of 136 references were found that documented anaerobic wastewater treatment, ranging in strength from 58 mg/L to 62,000 mg/L COD in 34 different countries. A Monod-type kinetic model, which predicts treatment efficiency and gas production, was developed to describe some of the literature observations. The results of the extensive literature review and model predictions suggests that anaerobic treatment is very promising and economical for treating low strength wastewater. This is contrary to experience in the United States where anaerobic wastewater treatment is seldom performed.

Key words---anaerobic filter, fixed film reactor, UASB reactor, hybrid filter, modified process, EGSB reactor, low strength wastewater, domestic wastewater

INTRODUCTION

Anaerobic treatment has traditionally been used for treatment of sludges and especially those derived from wastewater treatment plants. Treatment is provided to reduce sludge mass, increase dewaterability, and reduce pathogen content while producing a useful energy by product - methane gas. The restriction to sludges or high strength wastewater existed because elevated temperatures were required for the slow growing methanogens, and the methane produced from the concentrated sludges was required for heating. Figure 1 shows the heating value of digester gas using the stoichiometric methane yield from chemical oxygen demand (COD) destruction. The heating value is plotted as a function of wastewater strength, and specific points for mesophilic (37^{o}C) and thermophilic (55^{o}C) conditions are shown, assuming an ambient temperature of 20^{o}C. The graph shows two lines: 100% heat conversion and 50% heat conversion efficiency, which is typical of modern boiler and heat exchanger efficiency (the graph neglects any heat recovery that might be obtained from the digested sludge). For example increasing the temperature of a wastewater with an ambient temperature of 20^{o}C to 37^{o}C requires over 11,000 mg/L COD destruction at 50% heat conversion efficiency.

Young and McCarty (1969) and others (Witt *et al.*, 1979; Lettinga & Vinken, 1980; Braun & Huss, 1982) extended anaerobic treatment to high and medium strength wastewater by developing methods to retain cells in the reactors. The new anaerobic systems such as the anaerobic filter (AF), upflow anaerobic sludge blanket (UASB) and hybrid reactors (a combination of UASB and AF) allow treatment of low strength wastes such as domestic wastewater by maintaining long solids retention time (SRT) independent of the hydraulic retention time (HRT). This reduces or eliminates the need for elevated temperatures.

The new anaerobic systems may provide economical and efficient solutions for domestic wastewater when compared to conventional aerobic systems. They have several advantages over the conventional systems: they are simple, energy efficient, produce less sludge, do not require complex equipment and are easy to operate. These systems have had worldwide practical applications. Anaerobic treatment for domestic wastewater is especially suitable for tropical and sub tropical regions, for rural areas such as villages or small communities with a need for compact, simple systems without highly qualified staff and sophisticated equipment and for coastal and tourist cities.

The objective of this paper is to review the previous anaerobic treatment processes, evaluate their potential use for low strength wastewater, describe a model that can be used for AFs, and demonstrate treatment efficiency of domestic wastewater.

ANAEROBIC TREATMENT PROCESS

Anaerobic treatment of waste is a complex biological process involving several groups of microorganisms (Cha & Noike, 1997; Harper & Pohland, 1997; Jianrong *et al*., 1997). In general complex wastes are stabilized in three basic steps: hydrolysis, acid fermentation and methanogenesis. In the acid fermentation step the organic waste is decomposed into lower fatty acids such as acetic and propionic by acid forming bacteria. In methanogenesis these fatty acids are broken down into CO_2 and CH_4 by methanogens (Speece, 1996). The growth rate of the methanogens is low and is usually the rate-limiting step. Long SRT is required to retain the slow growing methanogens.

The conventional anaerobic digestion uses a completely mixed reactor and is mainly used to digest municipal sludge. This process is limited because the HRT is equal to the SRT, which results in large reactor volumes and low volumetric loading rates. The minimum SRT required is approximately 10-15 days at $35^{o}C$.

The first improvement over complete mixing was the anaerobic contact process (Schreopfer *et al*., 1955, 1959). This process used a completely mixed reactor followed by a settling tank, analogous to the activated sludge process, to separate and recycle cells to maintain high SRT with low HRT. The mixing of the reactor was done either with mechanical stirrers or by recirculating the biogas. A major disadvantage of the process was the need for a degasifier between the digester and the settling tank to prevent gas lifting of sludge particles. This process has been used for treating sugar, distillery, yeast, dairy and meat processing wastewater. The removal efficiencies ranged between 65-98% depending on different substrates and operational conditions (Nahle, 1991).

Coulter (1957) was the first to develop AF process. Wastewater flows through rock or synthetic media, which retains biomass on the surfaces and/or in the voids. This process was not investigated again until 1969 when Young and McCarty studied the treatment of a protein-carbohydrate wastewater (1500-6000 mg/L COD) at 25^{o}C, at organic loading rates (OLR) of 0.96-3.40 kg COD/m^3d. Pretorius (1971) used a modified digester (similar to a UASB) followed by a biophysical filter to treat 500 mg/L of raw sewage at 24 hr retention time at 20^{o}C. The digester concentrated the suspended solids and hydrolyzed the complex molecules, which were broken down to methane and carbon dioxide in the filter. He achieved COD removal efficiencies as high as 90%, and concluded that hydraulic loading was a better design parameter than waste concentration for low strength wastewater.

The UASB process was later developed, which employs a dense granular sludge bed at the bottom. A gas solids-separator is used at the top to capture digester gas while preventing solids from leaving the reactor (Lettinga & Pol, 1986, 1991; Souza, 1986). Lettinga (1980) treated raw domestic sewage (140-1100 mg/L COD) at ambient temperatures of 8-20^{o}C using a UASB. Removal efficiency of 65-90% was achieved for influent COD greater than 400 mg/L and an efficiency of 50-65% was obtained for COD’s less than 300 mg/L. Temperature had limited effect on removal efficiency.

More recently the UASB and AF processes have been modified to use the best features of each. The expanded granular sludge bed (EGSB) reactors use recycle to improve wastewater/sludge contact. EGSB reactors are designed with a higher height/diameter ratio as compared to UASB reactors, to accommodate an upward recycle flow (liquid superficial velocity) of 4 to 10 m/h (Seghezzo *et al*., 1998). The hybrid reactor is a combination of UASB and AF reactor concepts. Packing media is placed in the top of a UASB (Guiot & Van den Berg, 1985; Di Berardino, 1997).

The following sections describe the early development of each process with a detailed list of the published demonstrations or applications of each technology. The tables are divided by classifying the studies into laboratory, pilot, demonstration or full-scale application.

ANAEROBIC FILTERS

Table 1 shows 24 previously published studies of laboratory scale (< 10 L) AFs. Wastewater strengths ranged from 54,000 mg/L COD highest (Veiga *et al*., 1994) to 207 mg/L COD lowest (Viraraghavan & Varadarajan, 1996). Pilot and large pilot scale (10 to 100 L, and 100 to 1000 L, respectively) investigations are shown in Table 2 and there are 24 citations. They range in concentrations from 26 mg/L TOC (~ 65 mg/L COD) to 62,000 mg/L COD. Table 3 shows the demonstration and full-scale installations (13 citations), influent wastewater strengths ranged from 60 mg/L BOD to 68,400 mg/L soluble TOD. Only 20 citations were found for low strength wastewater (< 1000 mg/L COD), and none were full-scale installations. Three (Chung, 1982; Kobayashi *et al*., 1983; Abramson, 1987) were from our laboratory and the partial results will be used later in the model calibration. The lack of full-scale installations suggests that the technology is not yet accepted. This may be in part due to lack of experience or preference for UASBs.

Hudson (1978) used an AF to treat low strength shellfish processing wastewater with COD removal efficiencies ranging from 33 to 81% with 8 to 75 hr HRT with two different packing media. Koon *et al*. (1979) used an AF to treat domestic wastewater, and found BOD removal efficiency from 43 to 60 % at 12-48 hr HRT. His cost analysis showed that for a design flow of 189 m^3/d about 20% reduction in total annual costs could be achieved over the activated sludge process. Genung *et al*. (1979) reported 55% BOD removal from domestic wastewater in a demonstration facility. Kobayashi *et al*. (1983) evaluated a 16 L AF treating domestic wastewater at three temperatures (20, 25 and 35^oC), and found an average COD removal of 73%. Abramson (1987) showed 40 to 90% TOC removal in large pilot scale reactors. Iyo *et al*. (1996), Kim *et al*. (1997), Bodik *et al*. (2000), Elmitwalli *et al*. (2000), Kondo and Kondo (2000), Camargo and Nour (2001) also had varied success in treating low strength waste in anaerobic filters.

In contrast to low strength wastewater, AF treatment of medium and higher strength wastewater has been more extensively investigated. Chian and DeWalle (1977), Frostell (1981), Guerrero *et al*. (1997), Leal *et al*. (1998), Wilson *et al*. (1998), Ince *et al*. (2000), Alves *et al*. (2001), Garrido *et al*. (2001) are some notable examples.

UPFLOW ANAEROBIC SLUDGE BLANKET REACTORS

Tables 4 and 5 show the laboratory, pilot, demonstration and full-scale investigations of UASBs for wastewater treatment. There are 56 citations and 44 of them address low strength wastewater. More than 20 are full-scale investigations. The UASB has had much greater acceptance but not in the United States. The cited full-scale installations are in Europe, South America and Southeast Asia.

Lab scale studies using UASBs to treat low strength wastewater began as early as 1976, with Lettinga *et al*. (1983) performing many of the early studies. De Man *et al*. (1986) and Campos *et al*. (1986) were among the first to demonstrate low strength wastewater treatment in UASBs in large scale reactors. Table 5 shows many recent investigations using low strength wastewater. All are outside the United States. Draaijer *et al*. (1992) used a 1200 m^3 UASB reactor to treat municipal wastewater in Kanpur, India. The highest removal efficiency obtained was 74%. Vieira *et al*. (1994) performed a full-scale study on sewage discharged from low-income community in Sumare, Brazil, obtaining 74% removal efficiency. In another Brazilian study, Chernicharo and Cardoso (1999) treated domestic sewage from small villages using a partitioned UASB reactor. The partitioned reactor included three digestion chambers working in parallel to accommodate influent flow rate fluctuations. Removal efficiency reached 79% at HRT of 7.5 hr. The cost evaluation showed that partitioned UASB reactor was much less expensive than the conventional UASB reactor. Karnchanawong *et al*. (1999) investigated UASB domestic wastewater treatment in Thailand obtaining 53-69% BOD removal efficiency. Karnchanawong *et al*. (1999) also studied domestic wastewater treatment from apartment complexes in Bangkok. The removal efficiency ranged from 60 to 76%. He suggested an HRT of 10-12 hr as a design criterion for full-scale UASB reactors to achieve 75% BOD removal.

MODIFIED UPFLOW ANAEROBIC SLUDGE BLANKET REACTORS AND ANAEROBIC FILTERS

Tables 6 and 7 show the modified reactor studies. Kennedy and Van den Berg (1982) among others, investigated downflow AFs with varying success. Guiot and Van den Berg (1985) were the first to use packing above a UASB to improve efficiency. After 1989 there are 16 reported investigations using a hybrid AF, and 4 used low strength wastewater. Elmitwalli *et al*. (1999, 2001) used the hybrid concepts to treat domestic wastewater. Again, the experience is all outside of the United States, and there are currently no full-scale installations treating low strength wastewater.

Table 7 lists the modified UASBs for 9 investigations for domestic or low strength wastewater and several more treating septic tank effluents. Only one study was at full-scale for low strength wastewater, and all were outside the United States. De Man *et al*. (1988) was the first to use an EGSB to treat low strength wastewater, obtaining 20 to 60% soluble COD removal. Van der Last and Lettinga (1992) investigated an EGSB reactor treating domestic sewage, obtaining about 30% COD removal efficiency. EGSB reactors have also been used for industrial wastewater (Kato *et al*., 1997).

SUMMARY OF PREVIOUS WORK

UASBs, AFs and modified reactors have demonstrated excellent performance for high and medium strength wastewater. There are fewer but significant examples for low strength wastes, in different parts of the world but mostly in developing countries with tropical and moderate climates.

The efficiencies ranged from 5% COD removal to as high as 99% COD removal. Temperatures were as low as 2^{o}C. Hydraulic retention times ranged from 1.5 hrs to 10 days for UASBs and 1.5 hrs to 74 days for AFs. The anaerobic systems alone were usually insufficient to meet secondary discharge definitions (less than 30 mg/L BOD_5 and 30 mg/L TSS), and to achieve nutrient removal.

In order to overcome these shortcomings, aerobic reactors (such as sequencing batch reactors (SBRs), tricking filters, activated sludge, stabilization ponds, packed columns, biofilters, rotating biological contactors (RBCs), hanging sponge cubes, etc.) were used for polishing. Also, partitioned or staged anaerobic reactors were suggested for wastewater with high suspended solids or with high influent fluctuations, and for better colloidal suspended solids removal.

Gas composition and production have been less frequently reported, but are a function of different factors such as temperature, waste type and strength. Methane content when reported ranged from 45-95%. The reactors for low strength wastewater could usually be operated at low HRTs ranging from 3 to 24 hrs. Waste type, OLR, HRT, start up conditions, temperature, porosity, media configuration, feeding policy, flow pattern, and gas separation devices are some of the factors that need special attention in order to obtain good solids retention and prevent operational problems. Generally, the daily fluctuations in influent wastewater did not have an adverse effect on removal efficiency.

The previously cited studies show good success with anaerobic wastewater treatment at ambient temperatures, but there are few full-scale implementations, especially in the United States and especially for anaerobic filters. This review and the following research were performed in order to better understand anaerobic treatment and in the hopes that it can be more frequently adopted. In order to better understand the application for low strength wastewater, we developed a model that can predict reactor efficiency, gas production and gas composition as a function of key process variables.

MODEL DEVELOPMENT

The model developed is a dynamic model describing anaerobic treatment using anaerobic filters. The model predicts treatment efficiency as well as gas production and composition. The model assumes methane formation from acetate is the rate-limiting step. Therefore the model was simplified to methanogenesis, and hydrolysis and fermentation steps were not considered. This is a valid assumption for low strength wastewater. The model was based in part on earlier models developed by Andrews (1969, 1971). The model is restricted to low strength influents, and does not require the more advanced concepts that separate substrates and biomasses into different pools (Mosey, 1983; Moletta *et al*., 1986; Suidan *et al*., 1994; Jeyaseelan, 1997; Batstone *et al*., 2000; Karama *et al*., 2000). The model includes the physical, chemical and biological interaction between gas, liquid and biological phases, which are shown in Figure 2. The model is composed of 10 ordinary differential equations. The general material balance equation (Accumulation = Input – Output + Production – Utilization) was used for the corresponding 10 state variables: substrate, and biomass in the biological phase; CO_2, N_2 and CH_4 partial pressures in the gas phase; alkalinity, dissolved CO_2, N_2, CH_4 and NH_3 in the liquid phase.

STOICHIOMETRY

A generalized stoichiometric relationship showing the conversion of acetic acid to methane and carbon dioxide with the synthesis of biomass and the decay of biomass is given respectively in equations (1) and (2). For acetic acid the carbon dioxide and methane yield will be equal to each other as shown in equation (5).

$$CH_3COOH + Y_{XS}.Y_{NH_3X^1}NH_3 \rightarrow Y_{XS}C_5H_7NO_2 + Y_{XS}.Y_{CO_2X^1}CO_2 + Y_{XS}.Y_{CH_4X^1}CH_4 + aH_2O \qquad (1)$$

$$C_5H_7NO_2 + bH_2O \rightarrow Y_{CO_2X^2}CO_2 + Y_{CH_4X^2}CH_4 + Y_{NH_3X^2}NH_3 \qquad (2)$$

$$a = 2 - 2Y_{XS}(1 + Y_{CO_2X^1}) \qquad (3)$$

$$b = 3 \qquad (4)$$

$$Y_{CO_2X^1} = Y_{CH_4X^1} = 0.5(\frac{2}{Y_{XS}} - 5) \qquad (5)$$

from oxidation-reduction balance

BIOLOGICAL PHASE

The rate of change of substrate concentration in the reactor at any time depends on the influent and the utilization of substrate for biomass growth (eq. 7). Monod-type kinetics in equation (6) was used to describe the utilization of substrate.

$$\mu = \frac{\mu_{\max} S}{\left(K_S + S\right)} \qquad (6)$$

$$\mu_{\max} = f\left(Temp\right)$$

$$\frac{dS}{dt} = \frac{Q}{V}\left(S_o - S\right) - \frac{\mu X}{Y_{XS}} \qquad (7)$$

The rate of change of biomass concentration in the reactor is a function of the influent and effluent biomass concentration and the biomass growth and decay in the reactor (eq. 8). In AF the biomass concentration in the reactor is much higher than the effluent biomass concentration as the biomass is retained in the packing media.

$$\frac{dX}{dt} = \frac{Q}{V}\left(X_o - X_E\right) + \left(\mu - k_d\right)X \tag{8}$$

The production and utilization of dissolved CO_2, CH_4 gases and NH_3 during the biological reactions are given in equation (9). The production and utilization rates of CO_2, CH_4 and NH_3 during biomass growth are shown by r_1, r_3 and r_5 respectively. Similarly the production rates of CO_2, CH_4 and NH_3 during decay are represented by r_2, r_4 and r_6.

$$\begin{aligned}
r_1 &= \mu X Y_{CO_2X^1} \\
r_2 &= k_d X Y_{CO_2X^2} \\
r_3 &= \mu X Y_{CH_4X^1} \\
r_4 &= k_d X Y_{CH_4X^2} \\
r_5 &= -\mu X Y_{NH_3X^1} \\
r_6 &= k_d X Y_{NH_3X^2}
\end{aligned} \tag{9}$$

LIQUID PHASE

The net rate of CO_2, CH_4 and N_2 transfer between the liquid and gas phases can be expressed by two-film theory in equation (10). Henry's Law was used to determine the concentration of the gases in the liquid phase at equilibrium with the partial pressure of the gases in the gas phase. Henry's Law constants are a function of the temperature.

$$T_{Gi} = K_L a_i \left(C_i^* - C_i\right) \tag{10}$$

$$C_i^* = K_{Hi} P_i$$

$$K_{Hi} = f(Temp)$$

The charge balance in the reactor gives the alkalinity equation (11).

$$Z = \left[HCO_3^-\right] + 2\left[CO_3^{-2}\right] + \left[NH_3\right] + \left[OH^-\right] - \left[H^+\right] \tag{11}$$

The mass balance for the total carbonic acid system is shown in equation (12)

$$\left[CO_2\right]_T = \left[CO_2\right]_D + \left[HCO_3^-\right] + \left[CO_3^{-2}\right] \tag{12}$$

$$f_{HCO_3^-} = \frac{1}{\left(1 + \frac{K_2}{\left[H^+\right]} + \frac{\left[H^+\right]}{K_1}\right)} \tag{13}$$

$$f_{CO_3^{-2}} = \frac{1}{\left(1 + \frac{\left[H^+\right]}{K_2} + \frac{\left[H^+\right]^2}{K_1 K_2}\right)} \tag{14}$$

$$f_{CO_2} = \frac{1}{\left(1 + \frac{K_1}{\left[H^+\right]} + \frac{K_1 K_2}{\left[H^+\right]^2}\right)} \tag{15}$$

$$f_{NH_3} = \frac{1}{\left(1 + \frac{\left[H^+\right]}{K_{NH_3}}\right)} \tag{16}$$

$$\frac{dHCO_3^-}{dt} = f_{HCO_3^-} \frac{d(CO_2)_T}{dt} \tag{17}$$

$$\frac{dCO_3^{-2}}{dt} = f_{CO_3^{-2}} \frac{d(CO_2)_T}{dt} \tag{18}$$

$$\frac{dNH_3}{dt} = f_{NH_3} \frac{d(NH_3)_T}{dt} \tag{19}$$

The rate of change of alkalinity in the reactor (eq. 20) depends on the influent alkalinity and the change of bicarbonate (eq. 17), carbonate (eq. 18) and ammonia concentrations (eq. 19) in the liquid phase.

$$\frac{dZ}{dt} = \frac{Q}{V}(Zo - Z) + \frac{dHCO_3^-}{dt} + 2\frac{dCO_3^{-2}}{dt} + \frac{dNH_3}{dt} \tag{20}$$

The rate of change of total carbonic acid concentration in the reactor (eq. 12) is a function of the influent carbonic acid concentration and gas transfer rate of dissolved carbon dioxide (eq. 21) and the rate of dissolved carbon dioxide production during biological growth and decay as shown in equation (22).

$$T_{GCO_2} = K_L a_{CO_2} \left((CO_2)_D^* - f_{CO_2} (CO_2)_T \right) \tag{21}$$

$$\frac{d(CO_2)_T}{dt} = \frac{Q}{V}\left[(CO_2)_{TO} - (CO_2)_T \right] + T_{GCO_2} + r_1 + r_2 \tag{22}$$

The rate of change of dissolved N_2 in the reactor depends on the influent N_2 and the gas transfer rate of N_2 (eq. 23). The N_2 gas does not undergo any biological or chemical reaction in the reactor.

$$\frac{d(N_2)_D}{dt} = \frac{Q}{V}\left[(N_2)_{DO} - (N_2)_D \right] + T_{GN_2} \tag{23}$$

The rate of change of dissolved methane gas in the reactor is a function of the influent methane concentration and the gas transfer rate of methane and the rate of dissolved methane production during biological growth and decay (eq. 24).

$$\frac{d(CH_4)_D}{dt} = \frac{Q}{V}\left[(CH_4)_{DO} - (CH_4)_D \right] + T_{GCH_4} + r_3 + r_4 \tag{24}$$

The rate of change of total ammonia in the reactor depends on the influent ammonia concentration and the reaction rates during biological growth and decay (eq. 25).

$$\frac{d(NH_3)_T}{dt} = \frac{Q}{V}\left[(NH_3)_{TO} - (NH_3)_T \right] + r_5 + r_6 \tag{25}$$

GAS PHASE

The partial pressures of CO_2, CH_4 and N_2 gases in the gas phase are a function of the gas transfer rate and the outflow from the gas phase (eq. 26).

$$\frac{dP_i}{dt} = -P_T D T_{Gi} \left(\frac{V}{V_g} \right) - P_i \left(\frac{Q_g}{V_g} \right) \tag{26}$$

$$D = R(273.15 + Temp)$$

$$P_{H_2O} = f\left(Temp\right)$$

$$Q_i = -DVT_{Gi} \tag{27}$$

$$Q_g = \sum_{i=1}^{i=3} Q_i + Q_{H_2O}$$

MODEL RESULTS

Kobayashi *et al.* (1983) and Abramson's (1987) AF data were used to calibrate the model. Figures 3 and 4 show the calibration graph for removal rate and effluent substrate concentration as a function of solids retention time. Pairs of points are shown, with one pair representing the observed data, and the second pair representing the simulation for those conditions. The simulations are not on a smooth line, as shown in later figures, since each observed data point was collected at different temperatures, hydraulic retention times and influent substrate concentrations. Model predictions of the gas composition of the effluent as a function of solids retention time and influent concentration are given in Figures 5 and 6. The model accurately predicts the high nitrogen partial pressure for low strength wastewater. This is due to the dissolved nitrogen in the influent wastewater.

CONCLUSIONS

The literature review showed that anaerobic treatment using AFs, UASBs and modified reactors is an efficient and economical method for treating various types of wastewater, and there are some examples of low strength wastewater treatment, such as domestic wastewater. World wide, there is an increase in the number of pilot scale investigations and full-scale applications. For example many UASB reactors were built in the last 20 years to treat domestic sewage in tropical and sub tropical countries (Monroy *et al.*, 2000). There are fewer large scale AFs and modified reactors treating low strength wastewater. Research has mainly been limited to laboratory or pilot scale. Therefore more investigations are necessary to understand the applicability of these reactors on treating low strength wastewater.

The developed dynamic model was able to predict treatment efficiency from previous pilot scale AF studies. Furthermore the model simulates the gas composition of the effluent from influent characteristics. The previous data and the model suggest that 24 hr HRT is required to achieve greater than 60% COD removal. Methane composition will be less than 50% below influent substrate concentrations of 130 mg/L COD at ambient temperature of 20^{o}C.

Hopefully, the reported advantages of anaerobic reactors such as low energy consumption, easy operation, and less sludge production can be utilized more frequently in the United States and other areas where anaerobic wastewater treatment is less frequently used. Anaerobic treatment may be useful for pretreatment at secondary wastewater treatment plants that are at capacity or overloaded. The anaerobic process may be useful in reducing the load on the secondary treatment system.

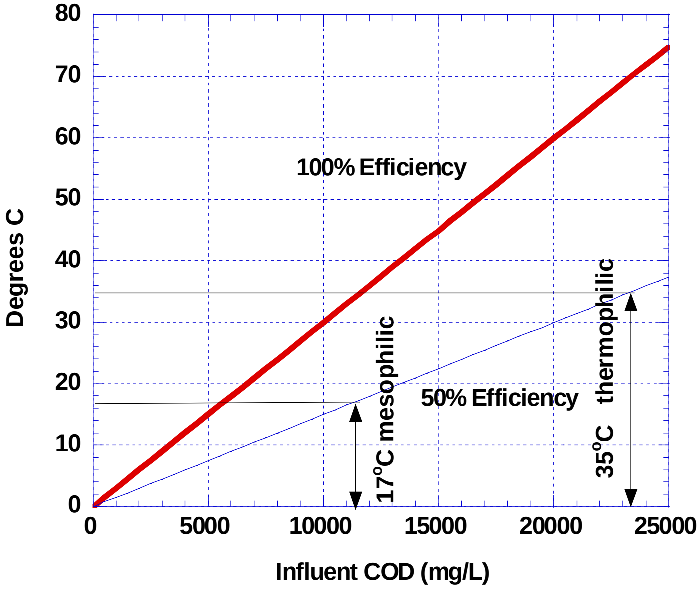

Fig. 1. Heat value of influent wastewater as a function of influent COD

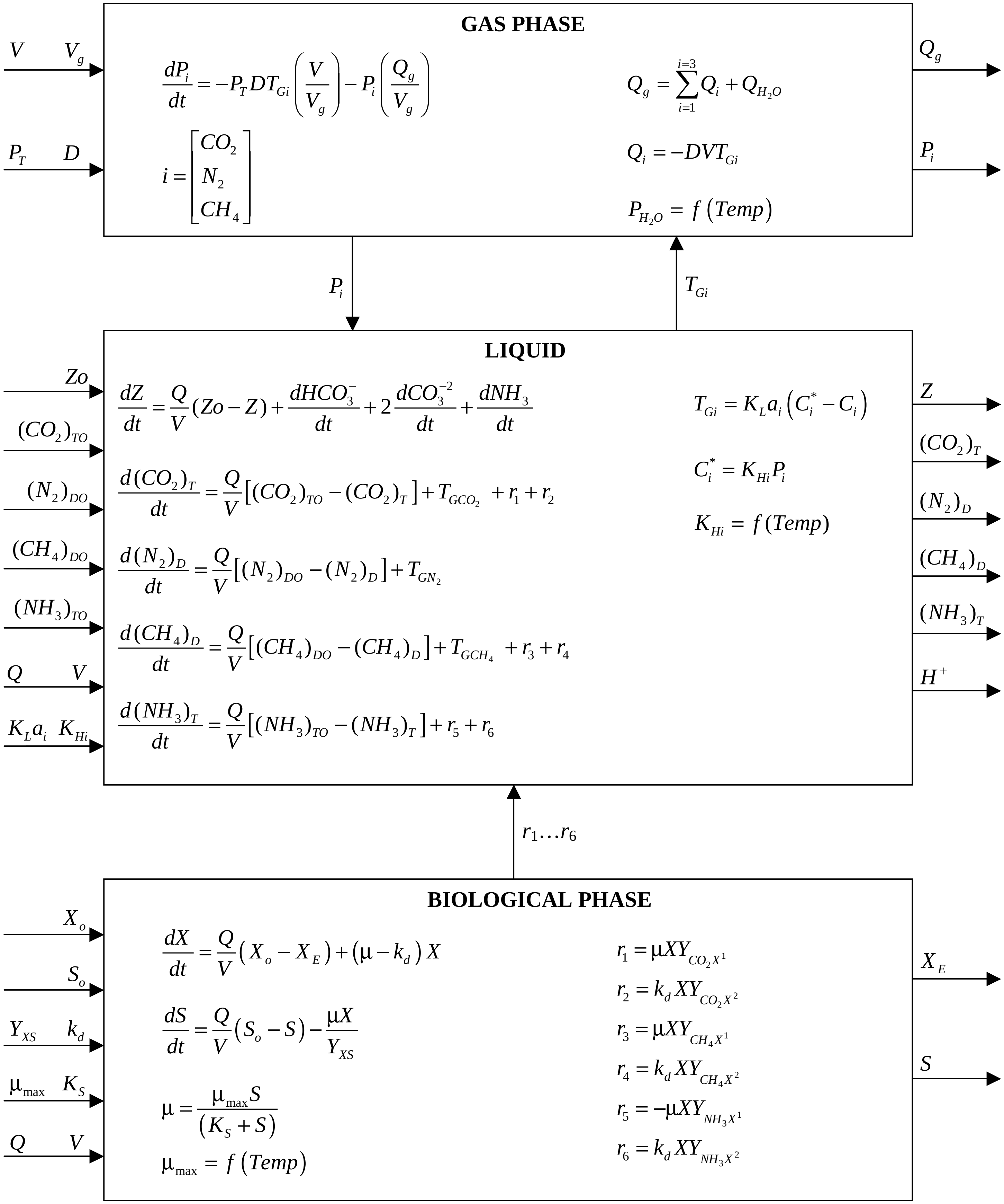

Fig. 2. Model Flow Diagram

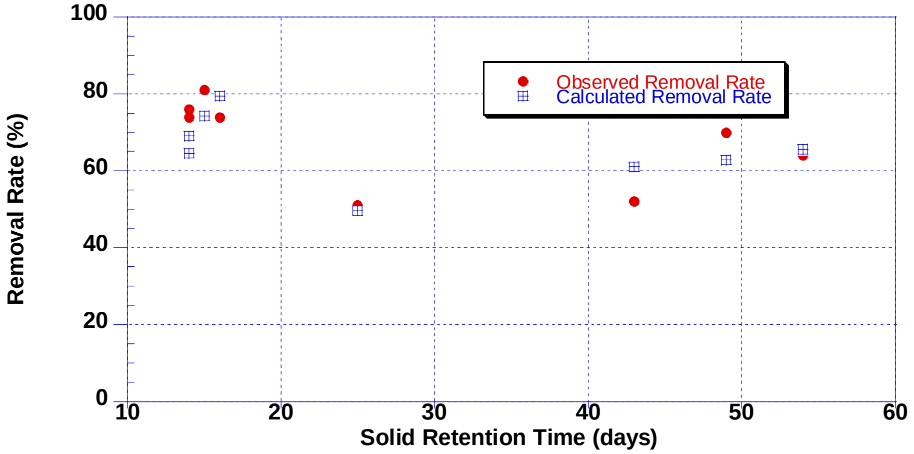

Fig. 3. Removal rate as a function of SRT

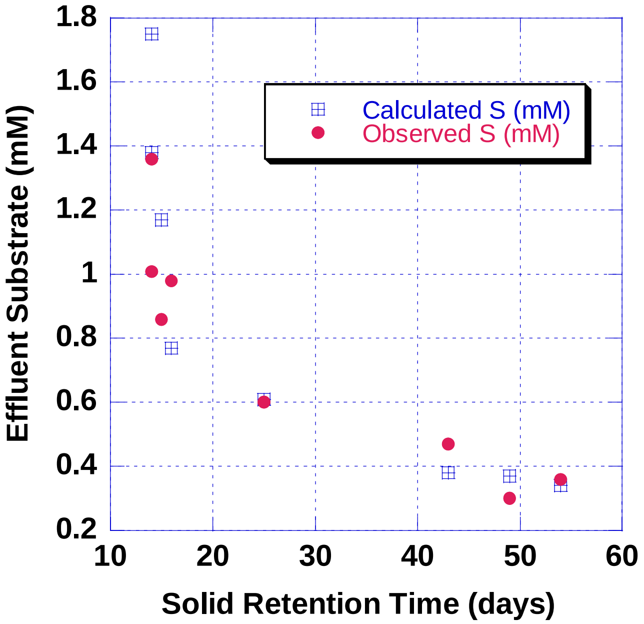

Fig. 4. Effluent substrate concentration as a function of SRT

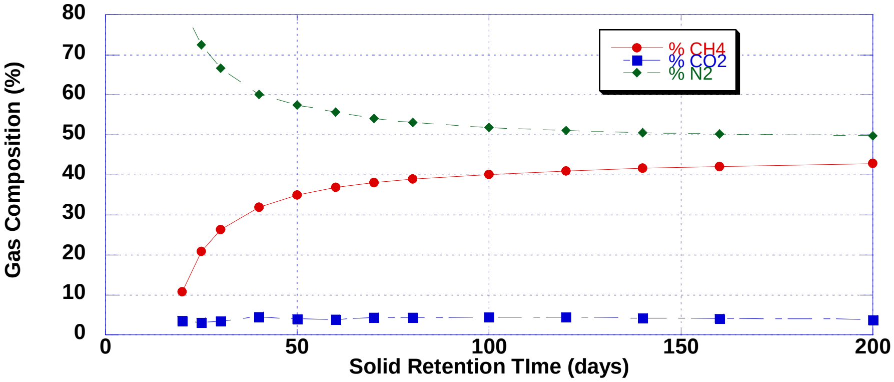

Fig. 5. Simulated gas composition as a function of SRT
(symbols represent calculated values).

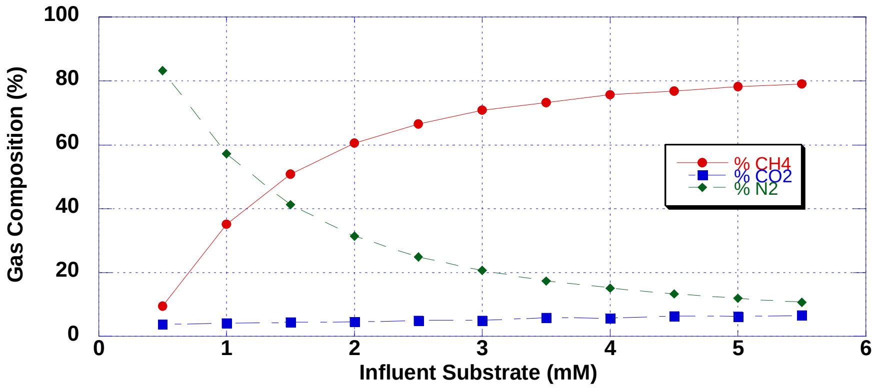

Fig. 6. Simulated gas composition as a function of influent substrate concentration (symbols represent calculated values).

Table 1. Laboratory Scale Studies of Anaerobic Filter on Wastewater Treatment

Reference and Region	Waste[1]	Organic loading rate[2] ($kg/m^3.d$)	Retention time (h)	Efficiency (%)	Packing material	Temp. (C)
Plummer *et al.* (1968) USA	Synthetic waste (1500-3000 mg/L)	0.424-3.392	4.5-72	36.7-92.1	Raschig rings and berl saddles mixture n=0.65-0.70	35
Pretorius (1971) South Africa	Raw sewage (500mg/L)	0.48	24-45	90	Stone n =0.6	20
El-Shafie & Bloodgood (1973) USA	Metrecal (10 g/L)	40.96	3-18	70.5	Hand-graded gravel	30
Frostell (1981) Sweden	Synthetic (8700 mg/L)	0.757-0.992	7.2-29	79-93	Polyurethane plastic material 200 m^2/m^3 * n=0.96	30
Landine *et al.* (1982) Canada	Potato processing wastewater	0.47-1.28	4-10 days	45-68	Rock media	22
Hanaki *et al.* (1990) Japan	Cafeteria (1300-2500 mg/L) 30% lipids		>1.3 days (SPS)[a] 3.3-10.1 days (TPS)[b]	80	Ring type plastic media 206 m^2/m^3 n=0.89	20
Viraraghavan *et al.* (1990) Canada	Dairy wastewater (4000 mg/L)	0.63-4.03	1-6 days	45-78 (12.5 C) 55-85 (21 C) 76-92 (30 C)	Plastic ballast rings 114 m^2/m^3 n=0.965	12.5-30
Hamdi & Garcia (1991) France	Olive mill wastewater (30 g/L)	2	15 days	60	PVC rings n=0.83	35
Hamdi & Ellouz (1993) France	Olive mill wastewater (9.22 g/L)	1.31	7 days	67	PVC rings n=0.83	35
Van der Merwe & Britz (1993) South Africa	Baker's yeast wastewater (5-30 g/L)	1.8-10	3 days	43-74	Synthetic rings 230 m^2/m^3 n=0.95	35
Borja & Gonzalez (1994) Spain	Olive mill wastewater (30 g/L)	2	15 days	70	Sepiolite rings n=0.69	35
Hanaki *et al.* (1994) Japan	Synthetic wastewater (2000-2500 mg/L)	0.27-0.82	3-9 days	81-90	Plastic tubes n=0.83	20
Veiga *et al.* (1994) Spain	Tuna processing wastewater (20-54 g/L) 80% protein 20% fatty acids + fats	3-13	24-96	75	PVC Raschig rings 300 m^2/m^3	37
Smith (1995) USA	Hazardous landfill leachate (3628 mg/L)	2.8	31.2	66-82	Plastic pack 331 m^2/m^3 n=0.88	36
Viraraghavan & Varadarajan (1996) Canada	Septic-tank effluent (207-286 mg/L)	0.09-0.17	1.20-3.17 days	5-52 (5 C) 25-62 (10 C) 49-65 (20 C)	Plastic ballast rings 114 m^2/m^3 n=0.965	5, 10, 20
Viraraghavan & Varadarajan (1996) Canada	Whey wastewater (3400-5200 mg/L)	2-10.1	0.52-1.7 days	69-93	Ceramic saddles n=0.57	16- 30
Guerrero *et al.* (1997) Spain	Fish meal processing wastewater (10.4-34 g/L)	1.62-5.26	4.41-12.22 days	80-90	PVC rings 450 m^2/m^3 n=0.94	37
Punal *et al.* (1999) Spain	Cheese whey wastewater (9000 mg/L)	0-35	8.4	60-95 (SFR)[c] 85-95 (MFR)[d]	PVC Raschig rings 228 m^2/m^3 n=0.94	
Reyes *et al.* (1999) Spain	Piggery wastewater (941 mg/L) (five upflow and downflow mode)		1,2,4 days 8,12	70 (BOD) 60	Waste tyre rubber 5 m^2/m^3 n=0.66	30-35

Reference and Region	Waste[1]	Organic loading rate[2] (kg/m^3.d)	Retention time (h)	Efficiency (%)	Packing material	Temp. (C)
Yilmazer & Yenigun (1999) Turkey *CSTR+ AF*	Cheese whey powder (11 g/L)	(1) 3.67 (2) 2.75 (3) 1.83	24h+3 day 24h+4 day 24h+6 day	(1) 63 (2) 95 (3) 67	Plastic pall rings 322 m^2/m^3 n=0.90	35
Bodik *et al*. (2000) Slovak Republic	Municipal wastewater (490-780 mg/L)		10, 20, 46	46-90	Plastic filling	9, 15, 23
Di Berardino *et al.* (2000) Italy	Food industry wastewater (0.53-2.62 g/L)	0.41-1.23	31-133	81.7-92.5	PVC tubes	35
Elmitwalli *et al.* (2000) Netherlands	(1) Raw sewage (772 mg/L) (2) Synthetic sewage (595 mg/L) (3) Skimmed milk		0.5-8	(2) 53-68	Reticulated polyurethane foam sheets 500 m^2/m^3	18-22
Ince *et al*. (2000) Turkey	Dairy wastewater (2000-6000 mg/L)	5-21	12	80	Raschig rings of glass media	35
Punal *et al*. (2000) Italy	Synthetic wastewater (1) 7200 mg/L (nitogen limited) (2) 6900 mg/L (nitrogen balanced)	1.5-4.5	1.5-4.6 days	(1) 76-86 (2) 80-90	PVC Raschig rings 228 m^2/m^3 n=0.85	35

[1] mg/L COD if not otherwise indicated

[2] COD unless otherwise indicated

[*] Specific surface area

[a] Single-phase system, [b] Two-phase system, [c] SFR: Single fed reactor, [d] MFR: Multiple-fed reactor

Scale: 0-10 liter Laboratory

Table 2. Pilot Scale Studies of Anaerobic Filter on Wastewater Treatment

Reference and Region	Waste[1]	Organic loading rate[2] ($kg/m^3.d$)	Retention time (h)	Efficiency (%)	Packing material	Temp. (C)
Young & McCarty (1969) USA	Synthetic waste (1500-6000 mg/L)	0.96-3.392	4.5-72	36.7-98	Smooth quartzite stone	25
Lovan & Foree (1971) USA	Brewery press liquor (6000-24000 mg/L)	0.8	15-330	90	Crushed limestone	34
Jennett & Dennis (1975) USA	Pharmaceutical wastes 95% methanol (1250-16000 mg/L)	0.221-3.52	12-48	94-98	Hand-graded quartzitic gravel n =0.47	37
Chian & DeWalle (1977) USA	Leachate (19.5-62 g/L)		7.5-74 days	94-98		
Hudson *et al*. (1978) USA	Shellfish processing wastewater (121-466 mg/L)	a. 0.18-0.34 b. 0.15-0.36	7.92-74.4	a. 33-55 b. 45-81	a. Granitic stone packing 130 m^2/m^{3*} n=0.53 b. Oyster shells n=0.82	9.8-26
DeWalle *et al*. (1979) USA	Landfill leachate (0.027-430 mg/L ions)		4.2-34	75 metal ion	Plastic medium 206 m^2/m^3 n=0.94	Room temp
Braun & Huss (1982) Austria	Molasses distillery slops (45-50 g/L)	30-50 VS	26.4-38.4	34-50	Plastic-ball packing material	42
Kobayashi *et al*. (1983) USA	Domestic wastewater (288 mg/L)	0.32	24	73	PVC pack 44 ft^2/ft^3 n=0.97	20-35
Lindgren (1983) Sweden	Synthetic (150-600 mg/L)				Polyurethane plastic material n=0.95	20-35
Noyola *et al*. (1988) France	Domestic sewage (407 mg/L)	0.5-12	4-72	45-80	PVC packing 170 m^2/m^3 n=0.85	16, 29
Abe *et al*. (1991) Japan *Aerobic soil column + AF (denitrifying reactor)*	Livestock wastewater (200 TOC mg/L)		1.8-2.6 days		a. Carbonized rice husks b. Carbonized rice husks with 20% straw c. Volcanic ash soil d. Charcoal chips	25
Akunna *et al*. (1994) France	Synthetic wastewater (5318 mg/L)	0.53-5.55	23h-10 days	60-77 99 (overall)	PVC rings	37
Viraraghavan & Varadarajan (1996) Canada	Potato-processing wastewater (220-840 mg/L)	0.14-0.35	1.5 days	17-56	Stone n=0.42	2-20
Wilson *et al*. (1998) Singapore	a. Domestic (0.26-0.54 g/L) b. Soy-bean processing (7.52-11.45g/L)	a. 0.96-2.04 b. 4.41-22.25	a. 0.42-0.21 day b. 1.04-0.42 days	a. 75-52 b. 92-75	a. Cylindirical plastic rings b. Soft fibrous media 1560 m^2/m^3	a. 17-28 b. 35
Show & Tay (1999) Singapore	Synthetic waste (2500-10000 mg/L)	2-16	15-30	a. 78-97 b. 77-95 c. 57-95	a. Glass Raschig ring 187 m^2/m^3 n=0.75 b. PVC Raschig ring 132 m^2/m^3 n=0.90 c. PVC Raschig ring 187 m^2/m^3 n=0.75	35
Jawed & Tare (2000) South Africa	Synthetic feed (2.30-8.74 g/L)	2-12	0.8-1.1 days	40-80	PVC module 102 m^2/m^3 n>0.97	34-36
Alves *et al*. (2001) Portugal	Synthetic dairy wastewater (3-12 g/L)	3.33-8.6	0.9-1.4 days	>90	PVC Raschig ring 230 m^2/m^3 n=0.925	35

Reference and Region	Waste[1]	Organic loading rate[2] (kg/m^3.d)	Retention time (h)	Efficiency (%)	Packing material	Temp. (C)
Picanco *et al*. (2001) Brazil	Synthetic wastewater (1267 mg/L)	1.27	24	68	a. Polyurethane foam n=0.92 b. PVC n=0.015 c. Special ceramic n=0.64 d. Refractory brick n=0.35	30
---------------------- Large Pilot Scale Studies ----------------------						
Donovan *et al*. (1979) USA	Heat treatment liquor (10-11 g/L)	1.56-9.39	16.56-152.64	17-68	Plastic media n=0.95	35
Chung (1982) USA	Domestic wastewater (25.6 TOC mg/L)	0.16	24	60	PVC pack 44 ft^2/ft^3 n=0.97	22.4
Abramson (1987) USA	Domestic wastewater (30-500 TOC mg/L)		6-60	40-90 TOC	PVC packing material	27.2
Sarner (1990) Sweden	Sodium based sulphite pulp mill wastewater (10-26 g/L)	20-40		85 inorganic sulphur removal	Plastic medium 140 m^2/m^3	
Kim *et al*. (1997) Japan	Sewage a. (222 BOD mg/L) b. (200.9 BOD mg/L)	a. 0.73 BOD b. 0.85 BOD	a. 7.3 b. 5.7	a. 96.1BOD b. 97 BOD	Polypropylene foam tube	
Camargo & Nour (2001) Brazil	Sewage (996 mg/L)	2.66-11.95	2-9	60-80	Whole and cut bamboo rings	

[1] mg/L COD if not otherwise indicated

[2] COD unless otherwise indicated

[*] Specific surface area

Scale: 10-100 liter Pilot, 100-1000 Large Pilot

Table 3. Demonstration and Full Scale Studies of Anaerobic Filter on Wastewater Treatment

Reference and Region	Waste[1]	Organic loading rate[2] (kg/m^3.d)	Retention time (h)	Efficiency (%)	Packing material	Temp. (C)
Genung *et al*. (1979) USA	Sewage (60-220 BOD mg/L)	0.048-0.608 BOD	2.5-10.5	55 BOD	Raschig unglazed ceramic rings	15-20
Koon *et al*. (1979) USA	Domestic sewage (92-209 BOD mg/L)	0.24-0.608 BOD	12-48	43-59.8 BOD	Raschig unglazed ceramic rings	13-25
Harper *et al*. (1990) USA	Poultry processing wastewater (2478 mg/L)	2.8	21	70 92 FOG (fat, oil and grease)	Polyethylene random pack	35
Hogetsu *et al*. (1992) Japan	Wool scouring wastewater (68.4 g/L soluble TOD)	3-45 TOD	Several days	60	Polypropylene media 65 m^2/m^{3*} n=0.95	37-53
Watanabe *et al.* (1993) Japan	Sewage (13 g BOD/c.d blackwater)** (27 g BOD/c.d graywater)			90 BOD		
Iyo *et al*. (1996) Japan	Domestic sewage a. 141.6 BOD mg/L b. 180.4 BOD mg/L c. 166.7 BOD mg/L	a. 0.06 b. 0.08 c. 0.075 (BOD)	a. 57 b. 54 c. 53 (overall)	a. 94.4 BOD b. 91.8 BOD c. 95.1 BOD (overall)	Polypropylene 82 m^2/m^3 n=0.39	a. 22-27 b. 16-22 c. 16-20
Viraraghavan & Varadarajan (1996) Canada	Slaughterhouse wastewater (1194-5900 mg/L)	0.47-2.98	0.8-4.9 days	37-77	Plastic ballast rings 105 m^2/m^3 n=0.90	23.6-27.1
Leal *et al*. (1998) Venezuela	Brewery wastewater (1400-3900 mg/L)	8	10	96	PVC Raschig rings	34-39
Kondo & Kondo (2000) USA	Domestic wastewater (130-550 BOD mg/L)	a. 0.68 b. 0.136 (BOD)	a. 9.6 hr b. 2 days (overall)	a. 97 BOD b. 98 BOD (overall)	Plastic media	14-21
---------- Full Scale Studies ----------						
Witt *et al*. (1979) USA	Guar (9140 mg/L)	7.52	24	60		36.6
Campos *et al*. (1986) Brazil	Meat processing wastewater (1878 mg/L)	1.4	13	76	Broken stones n=0.40	24-25
Defour *et al*. (1994) Ireland	Citric acid wastewater (16.6 g/L)	11.3	1.46 days	65		
Garrido *et al*. (2001) Spain	Dairy wastewater (6-15 g/L)	0.5-8	1.5	50-85	PVC packing	37

[1] mg/L COD if not otherwise indicated

[2] COD unless otherwise indicated

* Specific surface area, ** g/c.d refers to gram per capita per day

Scale: 1000-10000 liter Demonstration, >10000 liter Full

Table 4. Laboratory and Pilot Scale Studies of UASB on Wastewater Treatment

Reference and Region	Waste[1]	Organic loading rate[2] ($kg/m^3.d$)	Retention time (h)	Efficiency (%)	Temp (C) and Scale
Pretorius (1971) South Africa	Raw sewage (500 mg/L)	0.5	24	90	20, L
Frostell (1981) Sweden	Synthetic	2.5-10	20.6-53.3	68-87	30, L
Kato *et al*. (1997) Brazil	Synthetic (whey and ethanol) (113-722 mg/L) (127-675 mg/L)	0.2-6.8	2.6-29	30-99	30, L
Ruiz *et al*. (1997) Spain	Slaughterhouse wastewater (5200-11400 mg/L)	1.03-6.58	28.8-156	93-59	37, L
Kalyuzhnyi *et al*. (1998) Mexico	Potato-maize (raw) (5500-18100 mg/L)	0.63-13.89	15.6-144	63.4-81.3	35, L
Kalyuzhnyi *et al*. (1998) Mexico	Potato-maize (preclarified) (3600-9000 mg/L)	5.02-15	14.4-43.2	71.1-93.6	35, L
Elmitwalli *et al*. (1999) Netherlands	1. Raw sewage (456 mg/L) 2. Pre-settled sewage (344 mg/L)	1.37 1.03	8	65 59	13, L
Bodik *et al*. (2000) Slovak Republic	Municipal wastewater (310 mg/L)	0.62	12	37-48	9,15, L
Syutsubo *et al*. (2000) Japan	1. Alcohol distillery wastewater 2. Synthetic acetate wastewater 3. Sucrose wastewater (3000 mg/L)	9	8	94-99	55, L
Kalogo *et al*. (2001) Belgium	Raw domestic sewage (320 mg/L)	1.99	4.0	65	29, L
Kalyuzhnyi *et al*. (2001) Russia	Winery wastewater (2000-4200 mg/L)	1.7-4.7	0.86-1.15 days	57-68	4.8-10.3, L
Kalyuzhnyi *et al*. (2001) Russia *Two-stage UASB+UASB*	Winery wastewater (1500-4300 mg/L)	1.3-2.2	1.8-2.0 days (overall)	71-78 (overall)	3.9-10.2, L
Lacalle *et al*. (2001) Spain *UASB+ Upflow Aerated Filter*	Food industry wastewater (10.4 g/L)	1.27-2.76	4.51-13.0 days (overall)	96-99 (overall)	33, L
Nadais *et al*. (2001) Portugal	Dairy wastewater 1. 5.9, 11.9 g/L 2. 5.9, 5.8 g/L 3. 5.9, 5.6 g/L	 1. 11.8, 23..8 2. 11.8, 11.6 3. 11.8, 22.4	 1. 12 2. 12 3. 12, 6	 1. 93, 85 2. 93, 93 3. 93, 74	35, L
Nunez & Martinez (2001) Spain *UASB+Activated Sludge Process*	Slaughterhouse wastewater (1533-1744 mg/L)	2.62-6.73	6-16	85 (overall)	35, L
Lettinga *et al*. (1983) Netherlands	Raw domestic sewage (520-590 mg/L)	1.39-1.57	9	57-79	21, P
Gnanadipathy & Polprasert (1993) Thailand	Domestic wastewater (450-750 mg/L)	0.9-6.0	3-12	90	30, P
Sayed & Fergala (1995) Egypt *Two-stage UASB reactor system*	Domestic sewage (200-700 mg/L)	1.22-2.75[a] 1.70-6.20[b]	10 (8+2) 8 (6+2) 6 (4+2)	61-66[a] 32-46[b] 74-82 (overall)	18-20, P and L
Tang *et al*. (1995) Puerto Rico	Domestic wastewater (782 mg/L)	0.782-3.128	6-72	70.9	~20, P
Agrawal *et al*. (1997) Japan *UASB+ Hanging Sponge Cubes*	Raw sewage (300 mg/L)	1.03	7	(70 mg/L)	7-30, P

Reference and Region	Waste[1]	Organic loading rate[2] (kg/m^3.d)	Retention time (h)	Efficiency (%)	Temp (C) and Scale
Cheng *et al*. (1997) Taiwan	PTA manufacturing wastewater (4.66 g/L)	0.39-3.25	1.5-4.6 days	21-73	35, P
Gonzalez *et al*. (1998) Cuba	Sugar cane molasses (3640-3820 mg/L)	2.3-7.15	0.52-1.65 days	59.9-91	24-32, P
Goncalves *et al*. (1999) Brazil *UASB+ Aerated Biofilter*	Domestic wastewater (297-463 mg/L)	1.39-1.84	4-8 4.11-8.23 (overall)	68-73 82-92 (overall)	P
Lettinga *et al*. (1983) Netherlands	Raw domestic sewage (420-920 mg/L)		32-40	48-70	12-18, LP
Lettinga *et al*. (1983) Netherlands	Raw domestic sewage (248-581 mg/L)		12	72	18-20, LP
De Man *et al*. (1986) Netherlands	Municipal wastewater (100-900 mg/L)		4-14	45-72	7-18, LP
Vieira & Souza (1986) Brazil	1. Settled sewage (341 mg/L) 2. Raw sewage (424, 406 mg/L)	1. 2.05 2. 2.54, 2.44	4	1. 65 2. 60, 65	1. 35, LP 2. 20, 23, LP
De Man *et al*. (1988) Netherlands	Low strength wastewater (190-1180 mg/L)		7-8	30-75	12-20, LP
Monroy *et al*. (1988) Mexico	Sewage (465 mg/L)		12-18	65	12-18, LP
Barbosa & Sant'Anna (1989) Brazil	Raw domestic sewage (627 mg/L)	3.76	4	74	19-28, LP
Singh *et al*. (1996) Thailand	Synthetic wastewater (500 mg/L)	4 3 2 1.2	3 4 6 6	90-92	20-35, LP
Chernicharo & Machado (1998) Brazil *UASB/AF system*[c]	Domestic sewage (640 mg/L)		4-6 1.5-24 (AF)	80 85-90 (overall)	LP
Castillo *et al*. (1999) Spain *UASB+ two RBC reactors*	Domestic sewage 1. 363-625 mg/L 2. 613-666 mg/L	1. 1.45-10 2. 2.13-9.81	1.5-7.5 3-10 (overall)	1. 27-70 2. 22-55 82-99 (overall)	1. 18-20, LP 2. 12-13, LP
Chernicharo & Nascimento (2001) Brazil *UASB+Trickling Filter*	Domestic sewage (420-666 mg/L)	0.44-2.52	4	65-77 74-88 (overall)	LP
Torres & Foresti (2001) Brazil *UASB + SBR*	Domestic sewage (103-250 mg/L)	0.412-1	6	65 92 (overall)	14-25, LP
Von Sperling *et al*. (2001) Brazil *UASB+ Activated Sludge Process*	Municipal wastewater (386-734 mg/L)	2.32-4.4	4 7.9-11.2 (overall)	68-84 85-93(overall)	LP

[1] mg/L COD if not otherwise indicated

[2] COD unless otherwise indicated

[a]-This corresponds to the first stage which consists of two flocculent sludge UASB reactors working alternately (one at a time)

[b]-This corresponds to the second stage which consists of one granular sludge UASB reactor

[c]- The system consists of a UASB reactor followed by downflow and upflow anaerobic filters in parallel with blast furnace slag media

Scale: 0-10 liter Laboratory (L), 10-100 liter Pilot (P), 100-1000 liter Large Pilot (LP)

Table 5. Demonstration and Full Scale Studies of UASB on Wastewater Treatment

Reference and Region	Waste[1]	Organic loading rate[2] (kg/m^3.d)	Retention time (h)	Efficiency (%)	Temp (C)
Demonstration Scale Studies					
Craverio *et al*. (1986) Brazil *Two-stage (CSTR+ UASB)*	Brewery/soft drink wastewater (1.3-8 g/L)	2-13	6-8	80.9 84.4 (overall)	35
De Man *et al*. (1986) Netherlands	Municipal wastewater (100-900 mg/L)		9-16	46-60	10-18
Karnchanawong *et al*. (1999) Thailand	Domestic wastewater (64.6-94.7 BOD mg/l)	0.13-0.51	4.5-12	52.6-69.4 BOD	
Martinez *et a*l. (2001) Uruguay	Malting wastewater	0.25-6		85	15, 28, 30
Full Scale Studies					
Campos *et al*. (1986) Brazil	Vegetable/fruit processing wastewater (394-872 mg/L)	0.78-1.36	7.5-24	66-76	29-30
De Man *et al*. (1986) Netherlands	Municipal wastewater (100-900 mg/L) (150-5500 mg/L)		6.2-18	31-49	11-19
Pol & Lettinga (1986) Netherlands	a. Brewery wastewater (1-1.5 g/L) b. Alcohol distillery wastewater (4-5 g/L) c. Maize starch wastewater (10 g/L) d. Paper industry wastewater (3 g/L) e. Paper mill wastewater (~1 g/L)	a. 4.5-7.0 b. 11.5-14.5 c. 15 d. 10.5 e. 4.4-5	a. 5.6 b. 8.2 c. 18.3 d. 8-10 e. 5.5	a. 75-80 b. 92 c. 90-95 d. 75 e. 70-72	a. 20-24 b. 32-35 c. 40 d. 30-40 e. 26-30
Louwe Kooijmans & van Velsen (1986) Lettinga *et al*. (1987) Colombia	Domestic sewage (267 mg/L)	2	6-8	75-82	25
Collivignarelli *et al*. (1991) Maaskant *et al*. (1991) Italy	Municipal wastewater (205-326 mg/L)		12-42	31-56	7-27
Draaijer *et al*. (1992) India	Municipal wastewater (563 mg/L)	2.25	6	74	20-30
Kiriyama *et al*. (1992) Japan	Municipal sewage a. (297 mg/L) b. (286 mg/L) c. (394 mg/L)	a. 0.65 b. 0.73 c. 0.97	1.8	a. 58 b. 69 c. 73	a. 12 b. 24 c. 28
Van der Last & Lettinga (1992) Netherlands	Pre-settled domestic sewage (391 mg/L)	1.34-4.69	2-7	16-34	>13
Schellinkhout & Collazos (1992) Colombia *UASB+ facultative pond/lagoon*	Raw sewage	b. 2.0	a. 5-19 b. 5.2	a. 66-72 b. 18-44	
Vieira & Garcia (1992) Brazil	Domestic wastewater (188-459 mg/L)	0.62-1.88	5-15	60	18-28
Defour *et al*. (1994) Belgium	Potato wastewater (2600 mg/L)	8	7	90	
Defour *et al*. (1994) Belgium	Potato wastewater (12,500 mg/L)	12	18	78	
Defour *et al*. (1994) France	Brewery wastewater (4200 mg/L)	5	17	89	
Defour *et al*. (1994) Netherlands	Starch wastewater (5500 mg/L)	18	7.5	82	
Schellinkhout & Osorio (1994) Colombia	Sewage (380 mg/L)	1.82	5	45-60	24
Vieira *et al*. (1994) Brazil	Sewage (402 mg/L)	1.38	7	74	16-23
Tare *et al*. (1997) India	Domestic wastewater (1183 mg/L)	3.55	8	51-63	18-32

Reference and Region	Waste[1]	Organic loading rate[2] (kg/m^3.d)	Retention time (h)	Efficiency (%)	Temp (C)
Tare *et al*. (1997) India	Domestic wastewater (404 mg/L)	1.21	8	62-72	18-32
Chernicharo & Borges (1997) Brazil	Domestic sewage (600 mg/L)	1.11	13	68	
Vinod *et al*. (1997) India	Domestic sewage (133-254 mg/L)	1.1	8	49-65	
Vinod *et al*. (1997) India	Domestic sewage (551-730 mg/L)	5.63	8	24-50	
Yu *et al*. (1997) India	Municipal wastewater	0.7	12	49-78	15-25
Chernicharo & Cardoso (1999) Brazil *Partitioned Reactor*	Domestic sewage (712 mg/L)	2.28	7.5	79	
Karnchanawong *et al*. (1999) Thailand	Domestic wastewater (409.5-517.7 mg/L)	0.41-2.16	4.5-24	59.9-76.4	30.9
Del Nery *et al*. (2001) Brazil *DAF+UASB reactors R1, R2*	Poultry slaughterhouse wastewater (2631 mg/L)	0.51-2.11	1.47-5.29 days	47.8-84.4 (R1) 54.5-83.4 (R2)	
Florencio *et al*. (2001) Brazil *UASB+polishing pond*	Domestic sewage (290-563 mg/L)	0.79-1.40	8.8-9.7	71-83 79-84 (overall)	30.2-31
Rodriguez *et al*. (2001) Colombia	Domestic sewage (463-538 mg/L)	0.037-1.81	6.7-24.9	73-84	24-27

[1] mg/L COD if not otherwise indicated

[2] COD unless otherwise indicated

Scale: 1000-10000 liter Demonstration, >10000 liter Full

Table 6. Studies Using Modified Anaerobic Filter Process on Wastewater Treatment

Reference and Region	Waste[1]	Organic loading rate[2] (kg/m^3.d)	Retention time (h)	Efficiency (%)	Packing material	Temp(C) and Scale
Kennedy & Van den Berg (1982) *Downflow Fixed Film Reactors* Canada	Bean blanching	0.5-7.5	1-25 days	79-83	Clay packing 120 m^2/m^3[*] n=0.52-0.55	35, P
Kennedy & Droste (1983) *Downflow Fixed Film Reactors* Canada	Sucrose substrate (5-20 g/L)	4-4.5	0.5-15 days	56-85	NPP (needle punched polyester) packing n=0.92	L
Guiot & Van den Berg (1985) *Upflow Blanket Filter* Canada	Sugar wastewater (2500 mg/l)	5-51	2-18	96 (5-25 kg/m^3.d) 63 (36 kg/m^3.d) 64 (51 kg/m^3.d)	Plastic rings 2/3 sludge blanket 235 m^2/m^3	27, L
Kennedy & Guiot (1986) *Upflow Blanket Filter* Canada	Synthetic sucrose wastewater a. 2.5-10.6 g/L b. 300 mg/L c. 5000 mg/L	a. 10 b. 2.85-4.9 c. 5-16	a. 7.2-24 b. 1.6-3 c. 7.2-24	a. 96 b. 73-93 c. 77-97 (8%) 79-97 (16%) 72-97 (32%)	Plastic rings n=0.80 2/3 sludge blanket (a, b) 8, 16, 32% packing depth (c)	27, L
Kennedy & Guiot (1986) *Upflow Blanket Filter* Canada	Landfill leachate (15-25 g/L)	4.8-14.7	1.5-4.2 days	97-98	Plastic rings n=0.80 2/3 sludge blanket	35, P
Chang (1989) *Hybrid Upflow Anaerobic Filter* Taiwan	Leachate from solid waste landfill (11-58.4 g/L)	1.43-21.97	1.25-7.67 days	92 (OLR< 13 kg/m3.d) 70 (OLR=21.97 kg/m3.d)	Ceramic raschig rings 312 m^2/m^3 n=0.59	35, P
Chung & Choi (1993) *Hybrid Upflow Anaerobic Filter* Korea	Naked barley distillery wastewater (3-6 g/L)	1-3	72-144	89-94 (AUBF-1/7)[**] 91-94 (AUBF-1/2) 93-95 (AF)	Polyethylene rings 280 m^2/m^3 n=0.88	35, L
van der Merwe & Britz (1993) *Hybrid Upflow Anaerobic Filter* South Africa	Baker's yeast wastewater (5-30 g/L)	1.8-10	3 days	42-84	Polyethylene foam 0.77 kg/m^3	35, L
Austermann-Haun & Seyfried (1994) *Hybrid Upflow Anaerobic Filter* Germany	Industrial wastewater (11.4 g/L)	1.7	6.8 days	81	BIONET 100 m^2/m^3 34% packed	36.1, F
Miyahara & Noike (1994) *Hybrid Upflow Anaerobic Filter* Japan	Synthetic wastewater (550 mg/L)	0.55	24	75	Vinylidene chloride looped fibre (Ring Lace)	20, L
Tilche *et al.* (1994) *Hybrid Upflow Anaerobic Filter* Italy	Piggery wastewater	8.5-9.7	72	55	BIO-ECO polypropylene random pack	31-36, F
Borja *et al.* (1995) *Hybrid Upflow Anaerobic Filter* UK	Slaughterhouse wastewater (2450 mg/L)	5-45	2-12	69 (45 g/ l.d) 75 (32 g/ l.d) 98 (5-22 g/ l.d)	1/3 clay-ring support medium (bentonite) 250 m^2/g n=0.63 2/3 sludge blanket	35, L

Reference and Region	Waste[1]	Organic loading rate[2] (kg/m^3.d)	Retention time (h)	Efficiency (%)	Packing material	Temp(C) and Scale
Cordoba *et al.* (1995) *Hybrid Upflow Anaerobic Filter* Argentina	Dairy wastewater (1.82-8.39 g/L)	1.8-8.4	24	89.9-95.8	Polyurethane foam n=0.91 8/75 sludge blanket	30, L
Fang &Kwong (1995) *Hybrid Upflow Anaerobic Filter* Hong Kong	Corn starch Wastewater (3-25 g/L)	3-50	9.6-24	40-90	Plastic rings 235 m^2/m^3 21/31 sludge blanket	37, L
Di Berardino *et al.* (1997) *Hybrid Upflow Anaerobic Filter* Portugal	Food processing wastewater (300-2200 mg/L)	0.17-0.42	2.5 days	a. 60 b. 83	Plastic rings	a. 25, P b. 30, P
Timur *et al.* (1997) *Hybrid Upflow Anaerobic Filter* Turkey	Landfill leachate (14.9-19.98 g/L)	0.77-16.53	0.9-5.1 days	81.4 TOC	Plastic pall rings 322 m^2/m^3 n=0.90	35, L
Bello-Mendoza & Castillo-Rivera (1998) *Anaerobic Hybrid Reactor* Mexico	Coffee processing wastewater (2030 mg/L)	0.21-2.59	10-59	22.4-88.6	Volcanic rocks 2/3 sludge blanket	18-23, D
Borja *et al.* (1998) *Hybrid Upflow Anaerobic Filter* Spain	Slaughterhouse wastewater (3.74-10.41g/L)	2.49-20.82	0.5-1.5 days	90.2-93.4	Polyurethane foam n=0.5 2/3 sludge blanket	35, L
Elmitwalli *et al.* (1999) *Anaerobic Hybrid Reactor* Netherlands	a. Raw sewage (456 mg/L) b. Pre-settled sewage (344 mg/L)		8	a. 66 b. 61	Reticulated polyurethane foam sheets 500 m^2/m^3	13, L
Hutnan *et al.*(1999) *Anaerobic Hybrid Reactor* Slovakia	Synthetic wastewater (6000 mg/L)	0.5-15	0.4-12 days	80-90	Tubular plastic carrier 544 m^2/m^3 n=0.93	37, L
Wu *et al.* (2000) *Anaerobic Hybrid Reactor* Singapore	Synthetic wastewater (5000 mg/L)	1-24	5-60	71-98	Raschig rings 20%, 40%, 60% and 75% packing height	35, L
Elmitwalli *et al.* (2001) *AF + Anaerobic Hybrid Reactor* Egypt	Raw domestic sewage		a. 4+8 b. 2+4 c. 3+6	a. 70.9 b. 58.6 c. 63 (overall)	Vertical sheets of RPF 2400 m^2/m^3 n=0.97	13, P

[1] mg/L COD if not otherwise indicated

[2] COD unless otherwise indicated

[*] Specific surface area, [**]AUBF-1/7 refers to 1/7 packed anaerobic upflow sludge bed filter

Scale: 0-10 liter Laboratory (L), 10-100 liter Pilot (P), 100-1000 liter Large Pilot (LP), 1000-10000 liter Demonstration (D), >10000 liter Full (F)

Table 7. Studies Using Modified UASB Process on Wastewater Treatment

Reference and Region	Waste[1]	Organic loading rate[2] (kg/m^3.d)	Retention time (h)	Efficiency (%)	Temp(C) and Scale
De Man *et al*. (1988) *EGSB reactor* Netherlands	Low strength wastewater (150-600 mg/L)		2-3	20-60 CODs	12-20, LP
Van der Last & Lettinga (1992) *EGSB reactor* Netherlands	Domestic sewage (391 mg/L)	2.7-9.4	1.5-5.8	~30	16-19, F
Bogte *et al*. (1993) *UASB-septic-tank* Netherlands	Domestic wastewater (976 mg/L)	0.53	44.3	33	13.8, D
Bogte *et al*. (1993) *UASB-septic-tank* Netherlands	Domestic wastewater (821 mg/L)	0.34	57.2	3.8	12.9, D
Bogte *et al*. (1993) *UASB-septic-tank* Netherlands	Domestic wastewater (1716 mg/L)	0.20	202.5	60	11.7, D
Lettinga *et al*. (1993) *UASB-septic-tank* Indonesia	Domestic sewage Black water		360	90-93	LP
Lettinga *et al*. (1993 *UASB-septic-tank* Indonesia	Domestic sewage Grey + black water		34	67-77	LP
Wang (1994) *HUSB reactor* Netherlands	Sewage (650 mg/L)	5.2	3	37-38	15.8, LP
Wang (1994) *EGSB reactor* Netherlands	Sewage (397 mg/L)	4.76	2	27-48	15.8, LP
Kato *et al*. (1997) *EGSB reactor* Brazil	Synthetic wastewater with ethanol (127-675 mg/L)	3.9-32.4	0.2-2.1	56-97	30, P
Kato *et al*. (1997) *EGSB reactor* Brazil	Brewery wastewater (666-886 mg/L)	9-14.4	1.3-2.4	70-91	15-30, LP
Van Lier *et al*. (1997) *EGSB reactor* Netherlands	Synthetic wastewater (550-1100 mg/L)	5.1-6.7	4	97	8, L
Yu *et al*. (1997) *ABR reactor* [a] Britain	Municipal wastewater (338-516 mg/L)		2-16	67.8-83.5 (overall)	18-28, L
Driessen & Yspeert (1999) *IC reactor* Netherlands	Dairy industry wastewater (820-2950 mg/L)	8.5-24	2.6-4	51	37, F
Driessen & Yspeert (1999) *IC reactor* Netherlands	Food industry wastewater (1000-7500 mg/L)	5-42	3.6-9.1	80	27, F
Driessen & Yspeert (1999) *IC reactor* Netherlands	Brewery wastewater (3000-23000 mg/L)	4-36	8-24	70-90	35, F

[1] mg/L COD if not otherwise indicated

[2] COD unless otherwise indicated

[a]- ABR is the shortcut for Anaerobic Baffled Reactor. The system consists of three chambers. The first is a UASB reactor without a gas-solid-liquid separator, the second is a down flow fixed film reactor with plastic packing and the third one is a hybrid UASB-AF with plastic media at the top 3/5

Scale: 0-10 liter Laboratory (L), 10-100 liter Pilot (P), 100-1000 liter Large Pilot (LP), 1000-10000 liter Demonstration (D), >10000 liter Full (F)

NOMENCLATURE

a	stoichiometric coefficient from oxidation-reduction balance
b	stoichiometric coefficient from oxidation-reduction balance
CH_3COOH	molecular formula for acetic acid
$C_5H_7NO_2$	empirical molecular formula for biomass
C_i^*	saturation concentration of gases in liquid phase at equilibrium (mM)
C_i	concentration of gases in liquid phase (mM)
$\left[CO_2\right]_T$	total concentration of all forms of carbonic acid (mM)
$\left[CO_2\right]_D$	concentration of carbonic acid and dissolved carbon dioxide (mM)
$\left[CO_3^{-2}\right]$	carbonate ion concentration (mM)
D	conversion factor (L gas/mole gas)
$f_{CO_3^{-2}}$	fraction of carbonate ion in the carbonic acid system
f_{CO_2}	fraction of dissolved carbon dioxide in the carbonic acid system
$f_{HCO_3^-}$	fraction of bicarbonate ion in the carbonic acid system
f_{NH_3}	fraction of ammonia in the total ammonia system
$\left[H^+\right]$	hydrogen ion concentration (mM)
$\left[HCO_3^-\right]$	bicarbonate ion concentration (mM)
k_d	decay rate (d^{-1})
K_La_i	overall gas transfer film coefficient (d^{-1})
K_{Hi}	Henry's Law constant (mM/mmHg)
K_S	saturation constant (mM)
μ	specific growth rate (d^{-1})
μ_{max}	maximum specific growth rate (d^{-1})
$\left[NH_3\right]$	ammonia concentration (mM)
$\left[OH^-\right]$	hydroxyl ion concentration (mM)
P_i	partial pressure of gases in the gas phase (mmHg)
P_T	total gas pressure of CO_2, CH_4, N_2 gases and water vapor (760 mmHg)
Q	liquid flowrate (L/d)
Q_g	gas outflow from the reactor (L/d)
Q_i	gas outflow of CO_2, CH_4 and N_2 gases (L/d)
Q_{H_2O}	gas outflow of water vapor (L/d)
R	universal gas constant (0.082057 L-atm/mole. ^oK)
S	limiting substrate concentration in the reactor (mM)
S_o	influent substrate concentration (mM)

$Temp$	temperature (° C)
T_{Gi}	gas transfer rate (mM/d)
V	liquid volume in the reactor (L)
V_g	gas volume in the reactor (L)
X	biomass concentration in the reactor (mM)
X_o	influent biomass concentration (mM)
X_E	effluent biomass concentration (mM)
Y_{XS}	biomass yield per substrate utilized (mole/mole)
$Y_{CO_2X^1}$	carbon dioxide yield per substrate utilized (mole/mole)
$Y_{CH_4X^1}$	methane yield per substrate utilized (mole/mole)
$Y_{NH_3X^1}$	ammonia yield per substrate utilized (mole/mole)
$Y_{CO_2X^2}$	carbon dioxide yield per biomass decay (2.5 mole/mole)
$Y_{CH_4X^2}$	methane yield per biomass decay (2.5 mole/mole)
$Y_{NH_3X^2}$	ammonia yield per biomass decay (mole/mole)
Z	alkalinity in the reactor (meq/L)
Zo	influent alkalinity (meq/L)

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