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Aerobic Digestion of Wastewater Sludge Under Low Temperature Conditions

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By

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TABLE OF CONTENTS

Table of Contents - Continued

LIST OF FIGURES

LIST OF TABLES

SECTION I

INTRODUCTION

In recent years interest in wastewater sludge treatment and disposal has increased significantly. This increased interest is due primarily to economic considerations since in most wastewater treatment plants the costs for sludge handling, treatment and disposal amount to approximately 25 to 50 percent of the overall capital and operating investment for the facility. Present predictions are that the volumes of waste sludge will increase 60 to 70 percent within the next 15 years (1). Therefore, any reduction in the quantity of sludge to be processed is of the utmost importance in the design of efficient and economically feasible treatment facilities.

The main objective of sludge stabilization is to convert raw or untreated sludge into a nonoffensive form, with respect to odor and pathogenic microorganism, that is suitable for land disposal. Volatile solids and total solids reduction are also essential since this will reduce the amount of sludge which must be disposed of.

At the present time, the most widely accepted methods of sludge treatment are anaerobic digestion, oxidation ponds, aerobic digestion and vacuum filtration. These processes are usually proceeded by some form of sludge thickening, either gravity settling, centrifugation or flotation. Any reduction in the volumes of sludge to be handled is reflected in reduced equipment size and, therefore, costs. The ultimate disposal of the treated sludge is currently the most serious problem, since considerable land is required for landfilling and drying beds. This is usually expensive and at times environmentally offensive. In the case of incineration, initial investments are high and with the present fuel situation operating costs have become nearly prohibitive for small communities to bear.

Anaerobic digestion is presently being used in many cases for sludge volume reduction and stabilization. However, due to the many variables which effect the efficient operation of the anaerobic system and render it at times very erratic, this system is being looked at with disfavor by many municipalities. Constant monitoring of pH, organic acids production, alkalinity, and gas production are required and in many cases qualified personnel are not provided or are not available to perform the necessary adjustments to insure proper functioning. Large initial investments are also required for the anaerobic digester. Since the digester supernatant usually contains large amounts of nitrogen, additional treatment is often necessary. Only about 4 to 9 percent of the nitrogen in raw sewage which enters the digester is removed by anaerobic digestion, with the remaining 91 to 96 percent returned to the treatment plant in the supernatant liquor. This can pass through, often unchanged, to the receiving stream (1).

One process which has gained considerable recognition in the field of sludge volume reduction and stabilization is aerobic digestion. At the present time, limited use is being made of this process due to insufficient knowledge of the kinetics, design principles and operating costs of the system. However, in the installations where it is being utilized, results have been good and advantages of aerobic digestion over anaerobic digestion have been claimed to include:

1. Volatile Solids reduction approximately equal to that obtained anaerobically.
2. Lower BOD concentrations in supernatant liquor.
3. Production of an odorless, humus-like, biologically stable end product that can be disposed of easily.
4. Production of a sludge with excellent dewatering characteristics.

5. Recovery of more of the basic fertilizer values in the sludge.

6. Fewer operational problems.

7. Lower capital cost.

The portion of the cell growth phase in which aerobic digestion occurs is termed the endogenous phase. During the early stages of cell development there is sufficient food available for the microorganisms to grow at an accelerated rate since they are not subjected to any nutritional limitations. However, as cell growth continues and the population requirements dictate additional food, a point is reached where either declining growth will occur or additional food must be supplied. This initial growth phase can be visualized as that which occurs in the conventional activated sludge process. The point at which sludge wastage occurs is when the organisms have reached the declining growth phase due to complete assimilation of the organic nutrients in the raw sludge. The biological cells have now reached the endogenous growth phase and without further substrate, auto-oxidation of their own cellular materials and surrounding dead cells occurs (1,2,3,4,5,6).

The aerobic digestion process capitalizes upon this biological phenomenon to reduce the volume of waste activated sludge to be disposed of and to provide a biologically stable end product.

Experimental results have shown the aerobic digestion process to be a worthwhile and feasible unit operation in the treatment of both municipal and industrial biological sludges. However, limited information is presently available on the effects of lower temperatures on the efficiency of the process and, therefore, acceptance has often been limited to climatic regions where temperature is not a contributing variable in the process. In addition, conflicting results have been presented concerning the fate of nutrients during aerobic digestion. Consequently, the main purposes of this research

were to investigate the effects of temperature on the aerobic digestion of a wastewater sludge and to determine the fate of nitrogen and phosphorus during digestion. This was done by use of both batch and continuous flow reactors and both primary and secondary waste sludges.

SECTION II

LITERATURE REVIEW

Sludge Stabilization During Aerobic Digestion

Aerobic digestion can be used to stabilize primary or biological sludges or mixtures of the two. Reynolds (37) has presented a review of the basic theories of aerobic sludge stabilization which is based on the principle that biological cells will use their own cell material and dead cells as food if there is no external source of nutrients in their environment.

Eckenfelder and Ford (60) describe aerobic digestion as a commonly used process where waste activated and digestible primary sludge is aerated for long periods of time resulting in cellular destruction with a decrease in volatile suspended solids (VSS). If endogenous bacterial cells are represented by the chemical formula $C_5H_7NO_2$, then cellular destruction through aerobic digestion can be represented by $C_5H_7NO_2 + 5O_2 \longrightarrow 5CO_2 + 2H_2O + NH_3$. This occurs when substrate in an aerobic system is insufficient for both synthesis and energy, and the rate of cellular destruction exceeds that of growth.

Aerobic digestion of biological sludges from various operations in the treatment of municipal and industrial wastewaters is not a new process. Studies as far back as 1933 were conducted by Heukelekian (2) on the effect of seeding both anaerobic and aerobic sludges prior to treatment. This early research revealed that the aerobic process was superior to the anaerobic process since similar detention times for the same wastes treated aerobically and anaerobically were shown to have a higher degree of stabilization after having undergone aerobic digestion. Heukelekian reported, as many other researchers have since that time, that the deciding factor in the choice between aerobic or anaerobic digestion is the cost involved in supplying air to the aerobic system. The findings of Heukelekian opened up the field to

further research. Since that time many aspects of the aerobic digestion process have been examined by other investigators whose primary objectives were with the drainability, settleability, loading rate, volatile matter removal rates, etc., of the treated sludge. However, little information is available concerning the specific effect of temperature on the efficiency of the aerobic digestion process.

Early studies in aerobic stabilization were concerned mainly with the effect of aerobic treatment on the dewatering of sewage sludge. Murphy (7) studied aerobic digestion as a method of sludge conditioning prior to filtration and settling. His research, performed on a mixture of primary and waste activated sludge at 15°C, revealed that after digestion for six-day periods the reduction in volatile solids was not appreciable. In addition, vigorous aeration for short periods of time reduced the filterability of the sludge.

Coackley (8, 9), whose main interests were also in the dewatering characteristics of digested sludge, also reported observations concerning reduction of volatile solids and organic nitrogen. His early studies were performed on a sludge previously subjected to anaerobic digestion. The work was divided into three phases, the first being the aerobic digestion of anaerobic sludge without aerobic inoculation. The second phase consisted of inoculating the sludge with aerobic organisms from a compost heap, and the last phase was continuation of the anaerobic digestion to act as a basis of comparison. He found that at a temperature of 18°C the reduction of volatile solids after 48 days was not appreciable even in the aerobically inoculated phase. At a temperature of 37°C, the volatile solids were reduced from 2.98 percent to 1.08 percent after 47 days in the non-inoculated sample. In the inoculated series at the same temperature, the reduction was from 2.66 percent to 1.52 percent. A decrease in organic nitrogen was also observed in the aerobic samples while this did

not occur in the anaerobic sample. The sludge produced from the aerated phases was stable and odor free, which was confirmed by trying to initiate anaerobic digestion in them with an inoculation of anaerobic organisms. Anaerobic digestion could not be initiated and it was concluded that the anaerobic process could not further stabilize the aerobically treated sludge.

Akers (10), in studies conducted in 1950, investigated the factors which affected the auto-oxidation (endogenous respiration) rate of biological sludges. He considered two factors:

1. The effect of auto-oxidation rate on the mean sludge age (pounds of volatile suspended solids in the system divided by the pounds of volatile suspended solids added to the system daily).
2. The effect of separate aeration (aerobic digestion) of the mixed liquor without the addition of substrate during the aeration period.

In this study the average reduction in volatile solids in 8 days at 23°C was greater than 20 percent. Reductions in volatile suspended solids, BOD and COD were 30, 60, and 35 percent, respectively.

Work performed by Jaworski, et al (11), was undertaken in 1961 to determine the effects of temperature, time, and loading rates on the aerobic digestion of blends of primary and waste activated sludge. Both batch-type and incremental or continuous feeding systems were used. Temperature ranges from 15°C to 35°C and detention times of from 5 to 60 days were investigated. The data obtained from this study indicated that the aerobic digestion of mixtures of raw and activated sludge was feasible. They concluded that the reduction of volatile solids is a function of time, but that beyond digestion times of 15 days only small increases in solids reduction were obtained. A reduction in volatile solids of 21 percent was obtained at a detention time of 15 days at 15°C. At 15 days at 20°C the reduction was 43 percent, while for 60 days at 20°C it was

46 percent. In general, it was observed that greater reductions in volatile matter were obtained at higher temperatures and at lower loading rates, settleability of sludges digested for 30 days or less was generally poorer than that of the undigested sludge, drainability of sludges digested for periods greater than 5 days was satisfactory, and drying of aerobically digested sludge produced little odor. It was also noted that supernatant liquors from the aerobic digester exhibited relatively low BOD values when compared to anaerobic digester liquors.

Lawton and Norman (12) conducted a three-year aerobic digestion study at the University of Wisconsin to investigate the effects of detention time, temperature, pH, aeration rate, and loading rate on the reduction of volatile solids and the characteristics of the digested sludge. Their studies indicated that while temperature has a singificant effect at short detention times, the effect decreases appreciably as the time of aeration is lengthened or the loading rate is decreased. It was noted that pH values increased for detention times up to about 10 days to a maximum of approximately 8.0 and then decreased slowly to values near 5.0. These low values, which were reached at equilibrium after long detention times, were not considered to significantly affect digestion efficiency. A statistical analysis of the data obtained in this test indicated that higher temperatures resulted in highly significant increases in volatile solids reductions. Increasing the air rates produced no significant improvement in volatile solids reduction. On the other hand, longer detention times produced a significant increase in volatile solids reduction as loading was increased. Artificial control of pH resulted in no significant decrease in volatile solids reduction over an uncontrolled sample digested under the same conditions. Other noteworthy conclusions were that volatile solids reduction showed a high correlation to sludge age, and that for any given

degree of volatile solids reduction, the sludge age observed is expected to vary with the composition of the particular sludge and the frequency of feeding employed.

Eckenfelder (13), in his studies of the oxidation kinetics of biological sludges, used thickened waste activated sludge from a conventional activated sludge treatment plant. The sludge was placed in four liter digesters and aerated for a period of seven days at 25°C. After this period of digestion, reductions of 48.5 percent for COD, 38.2 percent for volatile suspended solids and a decrease in volatile solids from 76.5 to 63.5 percent were noted. Auto-oxidation rates of 10 percent per day were recorded, and it was observed that the soluble nitrogen increased from 46 to 94 mg/l.

Viraraghaven (14), performed tests on raw sewage sludge in Madras, India. The average temperature maintained during his experiments was 31°C and the results he obtained were similar to those previously reported. Detention times of 5, 10, 15 and 20 days were investigated and it was concluded that beyond 15 days the reduction in volatile solids was not significant.

Dreier (15), observed that for long detention times (60 days) the effect of temperature on the aerobic digestion of a mixture of raw and waste activated sludge was minor, and that digestion was essentially complete at all temperatures at this long detention time. For short detention times of 5 days, he noted that the effect of temperature was again significant due to the high organic loading imposed on the system. However, at intermediate detention times temperature did have a decided effect. The higher the temperature the more efficient the reduction in volatile solids became.

Aerobic stabilization of a primary wastewater sludge was performed by Malina and Burton (3) in 10 liter laboratory scale units at two organic loading rates: 0.10 lb VS/day/cu ft and 0.14 lb VS/day/cu ft. The theoretical detention

time of the system was 15 days and the temperature was maintained at 35°C. The results obtained from this series of tests showed that about 43 and 33 percent of the volatile solids were degraded at the high and low loadings, respectively. These results differed from those achieved by other investigators (11, 12) who recommended a loading of 0.10 lb VS/day/cu ft for the same type of sludge. The treated sludge showed a redox potential of slightly greater than +250 mv with respect to hydrogen, indicating that stabilization through oxidation had occurred at both loadings.

The results of an investigation by Reyes and Kruse (4) into the possibilities of aerobic digestion of night soil (urine, feces, and personal cleaning materials) at solids concentrations of 3, 6 and 8 percent indicated that stabilization could be accomplished within 20 days. They found that temperature had a significant effect on the rate and degree of stabilization and that the effect compared favorably with results obtained for the aerobic digestion of sewage solids. At a digestion temperature of 45°C a reduction in volatile matter of from 40 to 50 percent was reported after 15 to 20 days. At temperatures of 8°C and 60°C the reductions in volatile matter were 35 and 67 percent, respectively. It was also noted that, in general, more dilute night soils are more readily stabilized in terms of volatile matter reduction, drainability and color than those with high total solids.

Barnhard (16), while studying the application of aerobic digestion to the treatment of industrial wastes sludges, concluded that the solids reductions obtained through aerobic digestion were comparable to that obtained from anaerobic digestion. He also noted that the effects of temperatures below 20°C were significant enough to cause retardation of the aerobic digestion process. A detention time of 15 days produced acceptable stabilization in all cases.

Woodley (17), in a study of biochemical oxidation at thermophilic and mesophilic temperatures, discovered that primary sewage sludge can be degraded under mesophilic (35°C) and thermophilic (52°C) conditions. The reduction in volatile solids was higher in the mesophilic than in the thermophilic range. Mesophilic-oxidized sludge settled readily and gave a clear supernatant, whereas thermophilic oxidation was noticeably more efficient in removing soluble and nitrogeneous material.

Moriarty (18), confirmed the earlier studies of Heukelekian and Coackley (2,8,9) comparing the results obtained through aerobic and anaerobic digestion of similar sludge samples. He compared the two processes under the same conditions of temperature, loading rate and time, and measured CO_2 production as an indication of the degree of stabilization of the digested sludges. His results revealed, as did earlier researchers with the exception of Eckenfelder (13), that decomposition of volatile solids by aerobic digestion was more complete than by anaerobic digestion.

The dewatering characteristics of aerobically digested sludge are of great importance. This can be expressed by its filterability and settleability.

The filterability of a sludge is indicated by its specific resistance value. For digested sludge the range of 1-6 x 10^{10} sec^2/gram is listed by Metcalf and Eddy (19). Benedek, et al. (20) and Randall, et al. (21) reported in their batch studies that at temperatures of 10° and 20°C the specific resistance fluctuated with no major increase or decrease. At 10°C the specific resistance centered around 0.19 x 10^{10} and at 20°C around 0.15 x 10^{10} sec^2/gram.

The settling characteristics of digested sludge is usually poor. Jaworski, et al (22) reported very poor settling at low temperatures. Sludge that has been digested for more than 30 days formed gas bubbles which caused the solids to float. A probable cause for this phenomena is denitrification. Sludges

with extended digestion times at 15 and 20°C of at least 60 days showed improved settling. At 35°C, settleability improved after a 10 day detention time.

Dreier (15) reported that for detention times of 10 days or more the effect of digestion temperature on sludge drainability appeared to be of minor importance; approximately 70 to 80 percent of the original sludge volume appeared as filtrate in one hour of filtration. This compared to a value of 40 percent obtained after one hour from an untreated sludge sample. For sludges digested less than 5 days, the drainability was less than for untreated sludges. Settleability test results indicated that only after 30 days of digestion did the settling characteristics of the treated sludge improve over the untreated sludge.

Reyes and Kruse (4) investigated aerobic digestion of night soil. Seattleability and filterability observations of the digested night soils indicated that at 8°C hardly any settling occurred and that settling markedly improved at temperatures of 27°C and higher. The improvement in settleability at the higher temperatures was attributed to the observed clumping of solids. Filterability tests, with the addition of 1 percent alum or 0.5 percent ferric sulfate, yielded results which indicated that the digested night soil would be ammenable to vacuum filtration.

Lawton, et al. (12) conducted studies on the effect of pH on aerobic digestion. Two digesters, one operated at a pH of 7.1 and the other at 5.1, showed almost identical reductions. They concluded that there was a complete lack of correlation.

Jaworski, et al. (22) reported that during the digestion period the pH values initially increased to 8 and then fluctuated down toward a pH of 5, which could be considered unsuitable for aerobic metabolism. However, in only one case did there appear to be an appreciable build-up of volatile solids in the digester after long aeration following equilibrium conditions.

Variations in pH in a two and one half liter continuous flow reactor were reported by Singh, et al. (23). It was operated with a four day retention time and used as a control. The pH of this reactor fluctuated between 5.8 and 7.0.

Eikum, et al. (24), in their batch studies of primary and chemical sludge, found that the oxygen uptake rate would decrease with increasing detention times and would increase with increasing temperatures. They reported uptake rates ranging from 0.024 to 0.126 mg O_2/mgVS/day.

Singh, et al. (23) reported O_2 uptake values in their continuous flow digester of 0.055 to 0.137 mg O_2/mgVS/day. The digester had a four day detention time.

A survey of existing aerobic digestion field installations by Dreier (15) in 1963 showed considerable promise for the process. This was based on data received from the following plants:

Paramus, N.J., This facility, serving the Garden State Shopping Plaza, received sludge from a contact stabilization plant. The influent sludge to the digester had the following characteristics: pH of 7, total solids of 1 percent, alkalinity of 374 mg/l, 0.25 mg/l nitrate and 48.8 percent volatile solids. A 32 percent reduction in volatile solids was obtained, even though the feed sludge was already quite low in volatile solids. The digested sludge at Paramus exhibited no obnoxious odors in either the wet or dry state. The population served by the plant was equivalent to 1900 persons and the amount of sludge disposed of amounted to 14,000 gals. per month or approximately 0.25 gal/day/capita.

Batavia, Illinois, This was a municipal plant with a connected population of 8500, using primary settling followed by contact stabilization activated

sludge. A previously used holding tank (10,000 cu ft) for the rotary vacuum filter had been equipped with a decant swing pipe and was being employed as an aerobic digester. Decant thickening was practiced in this installation with good success. The feed concentration averaged about 4.75 percent total solids, and the digester contents averaged above 2.77 percent total solids. The digester contents ranged from about 1.4 to 5.0 percent total solids at various stages from after decanting to time of filtering. No mention of volatile solids reduction was made for this case since the system was operated mainly to cut down on the amount of solids going to the vacuum filter and thereby reduce operation and maintenance costs.

Rockford, Illinois. This municipality operated a high rate filter plant with anaerobic digestion and lagoons for the digested sludge. Filter sludge was returned to the primary tanks and settled with the primary sludge. In an effort to improve the plant operation, a 70 foot diameter sludge storage tank was converted to an aerobic digester. Sludge was drawn from the anaerobic digesters and aerobically digested prior to discharge to the lagoons in order to eliminate an odor problem. No specific data was reported in this case, as the system had only been in operation for a short time. However, early results indicated that the system was functioning properly and more permanent facilities were to be installed based on these results.

Several other new facilities were reported by Dreier (15), but no detailed removal results or operational data were presented since the facilities had only been in operation for a short period at the time. All the results from preliminary investigations, however seemed to indicate that the facilities were performing their designed functions well.

An aerobic digestion plant at the Metropolitan Denver Treatment facility proved quite satisfactory (25). Aeration basins from the activated sludge

treatment plant were converted to aerobic digesters with an eight-day detention time. The primary aim of their study was to reduce the volume of sludge processed by the vacuum filters and incinerators. By this method it was possible to nearly double the solids concentration in the aerobically digested sludge discharged to the flotation concentrator, with no increase in polymer dosage. This amounted to a savings of almost 25 percent in filter conditioning chemical costs for vacuum filtration. Costs for the operation of the aerobic digestion portion of the treatment facility were estimated at $0.35/mil gal of waste treated. This amounted to $1.33/ton for air and $1.15/ton for auxiliary fuel and a savings of $40/ton for chemical and sludge disposal.

Loehr (6) investigated the factors affecting the design of aerobic digestion systems. His results were based on theoretical assumptions of synthesis and oxidation rates. His general conclusions were that aerobic digestion did offer distinct advantages when compared to anaerobic digestion, but that the design engineer should make careful adjustments to each particular installation, especially with respect to temperature during winter months. He reported that a $10^{o}C$ change in temperature would change the rate of metabolism by a factor of two.

Riter (26), reported the results of his experiences using diffused aeration for sludge digestion. His results were based on observations made at the following three plants: Palmertown, Adamstown and Sinking Spring, Pennsylvania. All three plants digested waste activated sludge from contact stabilization processes. His primary conclusions were that the digesters at each plant produced a stable sludge which dried readily and was a useful soil conditioner. He also noted that anaerobic digestion accomplished greater volatile solids reductions than did the aerobic digestion process. Manual decanting of the digester supernatant proved to be adequate in producing an effluent with a low

BOD, but automatic decanting was not successful. Diffused-air aerobic digestion prevented freezing of the digester contents as opposed to mechanically aerated systems in which freezing was a common problem. The use of air-sock type diffusers prevented diffuser clogging. It was reported that construction costs for anaerobic digesters were more than for aerobic digesters. Operating costs depended on the population served. For 3000 people or less, aerobic was cheaper than anaerobic; as the population approached 8000, operating costs for anaerobic were less than those for aerobic. In general, the power costs for aerobic digesters increased as the population served increased.

Studies on the design and operation of turbine aeration for aerobic sludge digestion were performed by Levis, et al (27), in Millersville, Pa. The source of feed sludge in this case was from an activated sludge contact stabilization plant. The plant's outmoded anaerobic digester had been replaced with an aerobic digestion system. In this case, the results obtained after digestion indicated that the supernatant was more stable than that from the previous anaerobic system. The sludge produced from the system dewatered easily on sludge drying beds and had no offensive odor or appearance. Occasional icing of the turbine blades occurred during the winter months; however, it was not serious enough to affect the plant operation. No noticable difference in sludge stabilizaton rate was observed with respect to temperature at the Millersville plant, even during the winter months.

Seven treatment plants in the Province of Ontario using aerobic digestion were evaluated by Ahlberg and Boyko (28). All the aerobic digesters in this study utilized diffused-air aeration systems. The principle observations made by the two investigators were:

1. The design of aerobic digesters should provide a sufficient solids retention time rather than a specific hydaulic retention time. This detention time will determine the air requirements of the system.

2. Specific oxygen uptake is one of the most reliable indicators of the conditions and stability of aerobically digested sludge. The rate is temperature dependent.
3. Aerobic digestion produces a low strength organic supernatant that represents an insignificant load when returned to the treatment process.
4. Settling characteristics of the treated sludge deteriorate with increasing solids concentrations and low residual DO levels.
5. Operational problems occur with extremes of temperature. Foaming was a problem at higher temperatures and icing of the turbine blades occurred at the lower temperatures. This could be minimized by proper selection of the physical plant layout.
6. Sludge drying beds proved to be impractical for use with aerobic digesters in the particular cases evaluated.

One item of specific interest in this evaluation was the effect of temperature on the operation of the system. The authors stated that the location of the digester can be an important factor if it is an isolated tank. The worst possible situation for heat loss in winter is a steel tank completely above grade. The use of earth embankments to place the tank below grade are an important design consideration. The maximum observed temperature ranges in this study were 29°C (0° to 29°C). In the plants where proper positioning of the tanks with respect to temperature considerations was made, the temperature range of the digester contents was only 11.5°C (13.5° to 25°C). The authors go on to state that tank positioning is especially critical if mechanical aeration is used since there is more of a heat loss than with diffused air aeration, where the reactor contents are not being continuously exposed to the surrounding atmosphere.

Nutrient Transformations During Aerobic Digestion

Most research work to date on aerobic digestion has been concerned with the rate and extent of volatile solids and total solids reduction, and the effect of digestion on the settleability and drainability of sludges. A few researchers have, in addition, reported data concerning various forms of nitrogen and phosphorus during aerobic digestion. It is generally believed that aerobic digestion produces a supernatant substantially lower in nitrogen and phosphorus than supernatant produced from anaerobic digestion. While most research has supported this conclusion, significant differences in the extent of nitrification and denitrification during aerobic digestion have been reported. In most cases complete nitrogen mass balances were not kept, so uncertainties exist concerning the fate of nitrogen during aerobic digestion. Similarly, most researchers have not kept mass balances of phosphorus.

Eckenfelder (13), in studies on the oxidation of biological sludges, aerated secondary sludge from a conventional activated sludge plant treating domestic sewage for seven days. He reported that 10 to 12 percent per day of the volatile solids were oxidized at 25^{o}C. After 5 days of aeration first order kinetics were no longer approximated, and the oxidation rate decreased rapidly. Soluble nitrogen increased in an approximately stoichiometric ratio to the sludge oxidized.

Jaworski et al. (11) experimented with a mixture of primary and waste activated sludge. Detention time and temperature were varied to study their effect on aerobic digestion. There data reveals a correlation between nitrification, consumption of alkalinity, and decreasing pH. It was concluded that beyond 15 days detention time only small increases in volatile solids reduction are obtained. However, the extent of nitrification increased with increasing detention time beyond 15 days. The data also showed that nitrites were of little importance in the overall nitrogen balance.

Malina and Burton (3) conducted studies on the aerobic digestion of primary sewage sludge in continuously fed digesters. Experiments were performed at 35°C, 15 days detention time, and loading rates of 0.1 and 0.14 pounds VS/day/ft^3. Results indicated that most of the nitrogen in the feed sludge remained associated with the digested sludge solids. At both loading rates effluent supernatant total nitrogen was less than 15 mg/l; less than 4 mg/l of this appeared as nitrate nitrogen. Effluent pH at both loading rates was about 8.0. At the lower loading rate alkalinity decreased from about 1050 to 500 mg/l; at the higher loading rate the alkalinity decreased from about 800 to 450 mg/l.

Irgens and Halvorson (5) analyzed the effect of different temperatures and detention times on nitrogen and phosphorus during aerobic digestion. Digesters were fed on a continuous basis with mixed primary and activated sludge. At 30°C detention times ranged from 7 to 40 days, and at 23°C detention times of 13 and 20 days were used. It was reported that TKN concentrations of the digested sludge supernatant were between 4 and 8 mg/l, and nitrites, nitrates, and ammonium were less than 1 mg/l (as nitrogen) for all detention times and temperatures. At 30°C the sludge TKN decreased about 14 percent regardless of detention time. Loss of nitrogen was attributed to biological denitrification in the anaerobic zones of the sludge particles. Phosphate levels for all supernatants were less than 1 percent of the sludge phosphate for all digesters.

Viraraghaven (14) investigated the aerobic digestion of raw sewage sludge (4 percent solids) in Madras, India. His digesters were operated on a batch basis at ambient temperature, which averaged 31°C during the study. Viraraghaven also reported that nitrification occurred during digestion, however the total of nitrate and nitrite nitrogen never exceeded 4 mg/l. This indicates that a

significant amount of nitrogen was lost during digestion, although this was not noted by the author.

Bruemmer (29) compared aerobic digestion of primary sewage sludge using pure oxygen and air. His digesters were operated on a continuous feed basis, with detention times ranging from 6 to 22 days. Filtrates from stabilized sludge, using both oxygen and air, contained about 2 mg/l of phosphate and 8 mg/l of total nitrogen. Analyses of nitrate and ammonia nitrogen indicated that less than 1 mg/l of these forms was present. In view of the high partial pressure of oxygen in the oxygen digester it was thought that significant denitrification did not occur. A nitrogen balance was not kept, so data to substantiate this was not available.

Eikum et al. (30) investigated the feasibility of using aerobic digestion to stabilize mixed primary-chemical (alum) sludge. Digestion was carried out in continuous flow laboratory reactors at detention times from 5 to 35 days and 7, 12, 18, and 25°C. Tests on primary sludge were also carried out to make sure factors other than alum were not inhibiting the digestion process. From experimental results it was concluded that:

1. The aluminum content of sludge from a primary-chemical treatment plant using alum will not inhibit aerobic digestion.
2. Total nitrogen content of the sludge was reduced during aerobic digestion; higher reductions were found at higher temperatures.
3. Nitrification occurred in reactors treating both primary and mixed primary-chemical sludges. Alum did not inhibit nitrifying bacteria.
4. Nitrogen content as a percentage of the volatile suspended solids would increase during aerobic digestion in spite of the nitrogen lost from the system.

Eikum et al. (31) examined the release of phosphorus during storage of aerobically digested sludge from a mixed primary-chemical treatment plant. Experimental conditions were the same as described in the previous work. No significant release of solid phase orthophosphate occurred during aerobic digestion of both primary and mixed sludges. With increasing detention time volatile suspended soils concentration was reduced, but the amount of solid phase phosphorus remained nearly constant. In primary sludge digesters the percentage of phosphorus in the sludge increased from about 0.6 to 1.6 percent on a dry weight basis. Concerning anoxic storage, it was concluded that phosphorus is always released from the solid to liquid phase; the rate of release is high during the first 3 to 4 days. No release of phosphorus occurs during storage of mixed primary-alum sludge.

Cohen and Fullerton (32) reported the results of a one year study of aerobic digestion of waste activated sludge. Digestion using oxygen and air was compared. The air digester was a conventional activated sludge tank, consisting of three channels, operated in the usual plug flow mode. On the third and final pass the air was shut off to allow sludge settling and decanting of the supernatant. The oxygen digester was pilot-plant scale and well mixed, using a fine bubble, open tank oxygenation system.

In the air digester, loading rate and temperature varied during the study. Consequently, the individual effect of temperature or loading rate could not be distinguished. Volatile suspended solids reduction correlated best with a lumped parameter, SRT x Temperature, termed the temperature time factor. For the range of SRT (solids retention time) from 0 to 12 days and temperature from 12 to 22^{o}C the correlation coefficient was 0.93. A nitrogen balance kept on the air digester revealed that 9.6 percent of the nitrogen was lost during the year. This loss was attributed to biological denitrification.

Nitrification in the air digester correlated well with the temperature-time factor; r = 0.96. Nitrification also showed a correlation with changes in pH and alkalinity. When high nitrate levels appeared in the effluent, pH was reduced by about 0.5 units and alkalinity was reduced by about 200 mg/l during digestion. When nitrification ceased, pH and alkalinity remained virtually unchanged.

The extent of nitrification and denitrification in the oxygen aerobic digester was significantly different. Minimal nitrification occurred regardless of the loading rate or temperature. The pH and alkalinity in the digester effluent were always higher than in the feed sludge. It was assumed that a high concentration of biomass, high mixing energy, and polymer created an unfavorable environment for nitrifying bacertia.

Theory of Aerobic Digestion

Aerobic digestion of waste activated sludge is similar in concept to extended aeration. In the absence of an external carbon source to supply energy, activated sludge bacteria enter the endogenous phase of their life cycle and utilize internal sources of cellular carbon. Some bacteria undergo cell lysis and release their protoplasm into solution, which then becomes food to other organisms. The net result is a decrease in the volatile portion of the sludge mass (33).

Total Oxidation Theory

Under normal conditions aerobic digestion of biological sludges usually results in a maximum volatile solids reduction of 40 to 60 per cent (28, 32, 34). The residual volatile solids are thought to be polysaccharide in nature, consisting of the remains of the external slime layer and bacterial cell wall

(35, 36, 37). Washington and Symons (36) concluded that the inert volatile solids which would accumulate in activated sludge systems would contain 47 to 56 percent polysaccharide, 39 to 47 percent protein, and 3 to 8 percent fat. Many researchers have stated that the accumulation of this inert material is evidence that the total oxidation of all organic matter in activated sludge systems is not possible (35, 38, 39).

Obayashi and Gaudy (40) have demonstrated, however, that extracellular polysaccharides are readily degradable. They isolated extracellular polysaccharides from five different species of bacteria and used them as the sole carbon source for growth of wastewater organisms. They reported COD removals of 80 to 90 percent and values for bacterial yield and growth rate similar to values reported for simple sugar substrates. It was concluded that the buildup of extracellular heteropolysaccharides "cannot be validly cited as evidence against total oxidation."

Total oxidation of the organic influent substrate in a waste-water treatment system was first proposed by Porges, et al (41) in 1953. They developed a Two Phase Theory in which the substrate was removed and assimilated into new cells in Phase I, followed by self-oxidation (endogenous respiration) of the entire cell mass in Phase II. It was claimed that in Phase I, 52 percent of the substrate reappeared as new cells. In Phase II they measured an endogenous respiration rate of one percent per hour, or a total of 460 hours for 99 percent endogenous oxidation of a unit-mass of newly created cells. These data indicated that there should exist an equilibrium point between the weight of new sludge created and the weight of sludge lost by endogenous oxidation. At this point there should exist a system in which all of the influent organic substrate was eventually oxidized, and thus there would be no sludge removal required.

The above theory was further supported by the initial work of Forney and Kountz (42), who also concluded that total-oxidation was possible. Later, however, Kountz and Forney (38) reported that the earlier concept of total oxidation was theoretically unsound. Their second experiment was performed using a solution of skim milk as a source of organic matter. A metabolic energy balance of their multiple-unit total oxidation system was performed and the following conclusions were drawn:

1. Total endogenous oxidation is not possible within reasonable times and sizes of treatment systems, as there is a residual material remaining equivalent to 20 to 25 percent by weight of the new activated sludge produced.
2. The actual endogenous loss is 2 percent per day of the total weight of activated sludge in the system.
3. The accumulation of non-oxidizable sludge is 0.6 percent per day of the total weight of activated sludge in the system.

Additional research into the total oxidation theory by Symons and McKinney (35), using a batch-feed system in which nitrogen was limiting, revealed that total oxidation was not possible due to the buildup of extracellular polysaccharides. They pointed out that extracellular polysaccharides often accumulated as a result of internal metabolism, and that being a waste product, it would not be subject to further biological action. They concluded that the polysaccharides were an inert portion of the sludge that was formed. Washington and Symons (36) arrived at similar conclusions by reasoning that the inert material that would accumulate wuld be the remains of the cells capsular and external slime, "for this is the material which is least degradable by the organisms themselves". On the other hand, Washington, et al. (43) in a later study reported that

through a long term acclimation period of the microorganisms, an activated sludge treatment plant could be operated without sludge wastage. These results were obtained from the operation of a batch reactor for one year.

Busch and Myrick (39) initiated a study in early 1958 to determine the limitations of the total oxidation process by means of bench-scale units operated on a continuous-flow basis. After operation of the system for as long as 103 days without any intentional sludge wastage they reported that no food-population equilibrium was attained in either the batch or continuous system. Both batch and continuous-flow systems consistently produced median effluent BOD values of less than 10 mg/l even though the continuous unit operated at loadings up to 30 times higher than the batch unit. They concluded that total oxidation is neither theoretically nor practicably attainable due to the buildup of biological solids.

Recent long-term studies into the aerobic digestion-total oxidation concept by Gaudy, et al. (44, 45, 46, 47), have led to the most thorough and widely accepted theories on the subject in question. In the two earlier studies (45,46), return of all the biological solids was positively monitored and controlled in a continuously fed, completely-mixed extended aeration laboratory reactor. All the biological solids which did not settle out in the clarifier chamber were returned daily to the unit through the use of a centrifuge. This three-year study demonstrated conclusively that the total oxidation theory was not unsound and that biological solids would not continually build up in a total oxidation unit. Additional evidence was later obtained from a closed (batch) system undergoing endogenous metabolism. This study showed that under prolonged endogenous metabolism, total oxidation of an amount of sludge equal to that synthesized in the previous growth phase was in fact possible. Results from a hydrolytic pretreatment study (47) indicated that a

means of gaining engineering control of the biological solids concentrations in the system might be possible. Sludge drawn from the settling tank was acidified and subjected to hydrolysis, after which it was returned to the reactor. In this way, the most difficult step for the cells to perform was done chemically. The cells were then able to assimilate the hydrolyzed sludge.

Gaudy's most recent research endeavor (44) concerning the total oxidation theory was performed to determine if the extracellular microbial polysaccharides generated during the endogenous phase could be utilized as a substrate. As mentioned earlier, Symons and McKinney (35) concluded that total oxidation was not possible due to the buildup of these materials. However, Gaudy theorized that since the system studied by Symons and McKinney was nitrogen deficient, breakdown of these heteropolysaccharides would be severely hampered since a nitrogen source was required. Therefore, using extracellular microbial polysaccharides as a source of organic carbon for the growth of other microorganisms, Gaudy proved that these waste products were readily amenable to biological treatment. He concluded that the buildup of extracellular polysaccharides could not be considered as evidence against the total oxidation theory.

Based on the results obtained by the aforementioned researchers, it would seem that the theory of total oxidation is, in fact, sound. However, the results obtained were based on laboratory or ideal situations and through long term acclimation of the systems studied. Under these circumstances it was possible to demonstrate the total oxidation of the sludge produced. However, in actual aerobic digestion systems where detention times and other factors would be far different from those experienced in the laboratory, a definite accumulation of extracellualar waste products could be expected. Detention times could be lengthened to decrease the quantity of this material buildup, but by doing this the advantages of the aerobic digestion system's short detention time as compared to other methods of treatment would be lost.

Kinetics of Volatile Solids Degradation

Studies concerned with the kinetics of aerobic digestion are usually based on the endogenous respiration taking place during digestion. Endogenous respiration is that portion of the cell growth phase in which microbial organisms use their own cell material and surrounding dead cells as a food source. This occurs due to the lack of available substrate from other organic sources which is the case in the aerobic digestion of wastewater sludge. However, not all of this surrounding organic matter will be consumed by the organisms in the normal detention times of aerobic digestion, since a portion of the cell material is composed of hemicellulose and cellulose which are not readily degradable. Therefore a certain buildup of these residual materials does occur in the process.

The destruction of volatile suspended soils (or volatile solids) in batch fed or continuously fed, plug flow aerobic digesters is frequently predicted using first order reaction kinetics (33, 37, 48, 49). To apply a mathematical model the volatile suspended solids in the feed sludge are divided into a degradable fraction and a nondegradable fraction, the nondegradable portion being the inert residue discussed in the previous section. The mathematical model based on first order reaction kinetics predicts that the rate of volatile suspended solids reduction, or endogenous respiration, is proportional to the amount of degradable organic matter remaining. Mathematically,

$$\frac{dS}{dt} = -k_d S \qquad [1]$$

S = concentration of biodegradable cell material at time t.

dS = change in biodegradable cell material

dt = time interval

k_d = first order reaction rate constant.

Integration of equation [1] between definite limits gives

$$\frac{S_t}{S_o} = e^{-k_d t} \qquad [2]$$

S_t = concentration of biodegradable cell material at time t

S_o = concentration of biodegradable cell material at time zero, or in feed sludge.

Equation [2] can be used to predict performance of batch or plug flow aerobic digestion. It is not, however, directly applicable to continuous flow completely mixed systems. A cellular mass balance around the completely mixed digester shown in Figure 1 under steady state conditions can be used to apply the first order reaction rate constant to these systems:

$$\text{Net change} = \text{Input} - \text{Output} - \text{Amount degraded} \qquad [3]$$

$$VdS_{net} = QS_o dt - QSdt - VdS \qquad [4]$$

V = volume of digester

Q = continuous flow rate

S = concentration of degradable VSS in digester, and digester effluent.

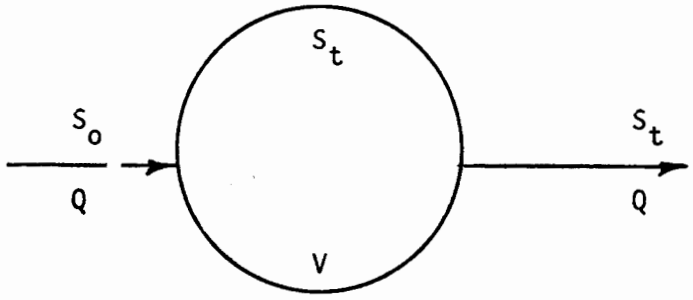

Figure 1

SCHEMATIC OF COMPLETELY MIXED AEROBIC DIGESTER

By noting that $dS = k_d S dt$, equation [4] becomes

$$V dS_{net} = QS_o dt - QS dt - Vk_d S dt \qquad [5]$$

$$\frac{dS_{net}}{dt_{net}} = \frac{Q}{V} S_o - \frac{Q}{V} S - k_d S \qquad [6]$$

Since $Q/V = 1/\theta$, where is the ideal digester residence time, and $(dS/dt)_{net} = 0$,

$$0 = \frac{S_o}{\theta} - \frac{S}{\theta} - k_d S \qquad [7]$$

Rearranging equation [7] gives

$$\theta = \frac{S_o - S}{k_d S} \qquad [8]$$

from which the required residence times can be calculated for varying degrees of VSS reduction. The development of this model is taken from work by Reynolds (37), and is similarly presented by others (33, 48, 49).

Stein, et al. (50) have developed a mathematical model for describing and analyzing the aerobic sludge digestion process for design purposes. Their design approach is based on first determining the biodegradable fraction of the sludge to be digested and then analyzing the biodegradable solids reduction data to find the reaction rate constant, K_b, for the following equation:

$$\frac{X_e - X_n}{X_o - X_n} = e^{-k_b t} \qquad [9]$$

where X_e = effluent total VSS remaining at time, t(mg/l)

X_o = influent total VSS at time t = 0 (mg/l)

X_n = non-degradable portion of VSS, assumed constant throughout aeration basin (mg/l)

Using a solids mass balance for a completely mixed system and integrating, it is then possible to solve for the required detention time for the desired degree of stabilization from the equation:

$$t_o = \frac{X_o - X_e}{K_b (X_e - X_n)} \qquad [10]$$

This method assumes a completely endogenous system with no active growth stage. In a batch digestion study with long detention times the growth stage is negligible.

This approach differs from others in that only the biodegradable portion of the volatile suspended solids are considered. The more common approaches are based on total volatile suspended solids or, occasionally, on total suspended solids with the use of K_d, the decay coefficient.

In batch-fed reactors or continuously-fed plug flow systems, the destruction of volatile degradable organics may be approximated by first order kinetics as follows:

$$\frac{(X_d)_e}{(X_d)_o} = e^{-k_b t} \qquad [11]$$

where:

$(X_d)_e$ = degradable VSS remaining after batch aeration time t (mg/l)

$(X_d)_o$ = initial degradable VSS at time 0 (mg/l)

K_b = batch reaction rate for degradable VSS

t = time of aeration (days)

The degradable VSS, X_d, may be expressed as a function of total VSS X_o, by incorporating the non-degradable residue X_n:

$$(X_d)_e = (X_e - X_n) \qquad [12]$$

$$(X_d)_o = (X_o - X_n) \qquad [13]$$

where:

X_e = effluent total VSS remaining at time t (mg/l)

X_o = influent total VSS at time 0 (mg/l)

X_n = non-degradable portion of VSS assumed constant throughout aeration period (mg/l)

Equation [11] may therefore be modified to consider total VSS rather than degradable VSS by incorporating equations (h and i):

$$\frac{(X_e - Xn)}{(X_o - X_n)} = c^{-K_{bt}} \qquad [14]$$

Unfortunately, digesters are not operated on a batch or plug flow basis and this approach often results in under-designed systems. As a rule, most aerobic digesters are operated as completely-mixed, flow-through basins. Adams et al (33) felt that first order kinetics could not be used to accurately predict the performance of the digester and found that a mass balance around

the aeration basin yielded a representative pseudo-second order relationship. It was shown experimentally that a rate coefficient determined in the laboratory under batch conditions could be incorporated into the pseudo-second order model to predict volatile solid destruction efficiencies under completely mixed conditions.

In completely mixed flow-through systems the mass balance around the digester is represented by:

Degradables in - Degradables out = Degradables destroyed

$$Q(X_d)_o - Q(X_d)_e = \frac{d(X_d)_e}{dt} V \qquad [15]$$

$$(X_d)_o - (X_d)_e = K(X_d)_e t$$

Substitution equations [12] and [13]:

$$(X_o - X_n) - (X_e - X_n) = k(K_e - X_n)t \qquad [16]$$

Solving for detention time:

$$t = \frac{X_o - X_e}{K_b(X_e - X_n)} \qquad [17]$$

Also equation [16] may be arranged to predict the effluent VSS at a given time:

$$X_e = \frac{X_o + K_b t X_n}{1 + K_b t} \qquad [18]$$

Based on the above sequence of equations, it is speculated that a batch experimental procedure may be utilized to establish the reaction rate coefficient, K_b, but the first order equations [11] and [14] cannot be used to predict a completely-mixed, flow-through system. Instead, the batch coefficient can be utilized in the pseudo-second order equation [17] or [18] to predict the performance of the completely -mixed basin.

Kinetics of Oxygen Uptake

To determine the oxygen requirements in both plug flow and completely mixed systesm, Reynolds (37) uses equations based on the empirical chemical equation for bacterial cells of $C_5H_7NO_2$ with a molecular weight of 113. In the case of oxidation where the nitrogen is not completely nitrified but remains in the ammonia form, the biochemical equation is,

$$C_5H_7NO_2 + 5O_2 \longrightarrow 5CO_2 + 2H_2O + NH_3$$

The amount of oxygen required per pound of cells is 5 x 32/113 or 1.42 pounds of oxygen to oxidize each pound of cells. During long periods of aeration the ammonia will eventually be converted to nitrate ion and the equation for the biochemical reaction in this case is,

$$C_5H_7NO_2 + 7O_2 \longrightarrow 5CO_2 + 3H_2O + H^+ + NO_3^-$$

The amount of oxygen required under these conditions will be 7 x 32/113 or 1.98 pounds per pound of cells.

Assuming detention times less than that required for complete conversion of the ammonia nitrogen to nitrate and substituting dO_2 for dS in Equation [1] yields,

$$\frac{dO_2}{dt} = 1.42\ K_d S \qquad [19]$$

where dO_2 = oxygen uptake and $1.42K_d$ = rate constant.

This is the instantaneous oxygen uptake rate under plug flow conditions. The oxygen required will vary along the length of the reactor due to differences in cell mass concentrations. The biodegradable cell concentrations at various points along the reactor can be calculated by dividing the theoretical detention time by the distances to the various points from the upper to the lower end of the reactor and using this value in Equation [5]. The value obtained for the cell concentration can now be substituted in Equation [19]. The oxygen utilization rates at the various points along the tank can be computed and an average value determined.

For completely mixed systems the oxygen utilization rate may be obtained by using Equation [8] and remembering that the equation,

$$\frac{dO_2}{dt} = 1.42 \left(\frac{dS_t}{dt}\right)_{Decay} \qquad [20]$$

which is obtained by substitution of Equation [1] into Equation [19], holds true as in the case of plug flow. Rearranging Equation [8] by dividing by V and dt yields,

$$\left(\frac{dS_t}{dt}\right)_{Net} = \frac{FS_o}{V} - \frac{FS_t}{V} - \left(\frac{dS_t}{dt}\right)_{Decay} \qquad [21]$$

where $\frac{F}{V} = \frac{1}{\theta}$. For steady state, $\left(\frac{dS_t}{dt}\right)_{Net} = 0.$

Therefore Equation [21] will rearrange to,

$$\left(\frac{dS_t}{dt}\right)_{Decay} = \frac{S_o - S_t}{\theta} \qquad [22]$$

which after substitution into Equation [20] yields the expression for the oxygen utilization under completely mixed conditions at steady state,

$$\frac{dO_2}{dt} = 1.42 \quad \frac{S_o - S_t}{\theta} \qquad [23]$$

Kinetics of Nitrogen Transformation

Most of the nitrogen in raw primary or secondary sludge is organically bound in the solid phase as protein. A significant amount of nitrogen may also be present as ammonium, while smaller amounts of nitrate or nitrite nitrogen may also be detectable. During aerobic digestion several different biologically mediated reactions may take place, and several different nitrogen forms may occur. The important reactions are ammonification, assimilation, nitrification, and denitrification.

Ammonification is the change from organic nitrogen to ammonium. Assimilation is the use of mineralized forms of nitrogen, either ammonium or nitrate, to form animal or plant protein. Nitrification describes the biochemical oxidation of ammonium to nitrate, and denitrification describes the reduction of nitrate or nitrite to nitrogen gases. As is evidenced by the apparent variety of

results discussed in the literature review, all of these reactions may or may not occur in a particular aerobic digester.

Biological nitrification of ammonium is considered a two step reaction. The first step involves oxidation of ammonium to nitrite by the nitrosomas group of bacteria. The second step is oxidation of nitrite to nitrate by nitrobacter. Both of these groups of bacteria are chemoautotrophic, in that they derive energy for metabolism from inorganic nitrogen. The net stoichiometric relationship for the oxidation of ammonium to nitrate by nitrifying bacteria can be represented by

$$NH_4^+ + 2O_2 \longrightarrow NO_3^- + 2H^+ + H_2O \qquad [24]$$

In the aquatic environment of an aerobic digester the production of free hydrogen ions generally results in reaction with bicarbonate ions. Equation [24] can be modified to show this:

$$NH_4^+ + 2O_2 + 2HCO_3^- \longrightarrow NO_3^- + 2H_2CO_3 + H_2O \qquad [25]$$

From equation [25] it can be calculated that the oxidation of 1 gram of ammonium nitrogen will consume 7.14 grams of alkalinity (as $CaCO_3$). Experimentally measured values for alkalinity consumption range from 6.0 to 7.4 grams of alkalinity per gram of ammonium nitrogen (34).

Because of the consumption of alkalinity and production of carbonic acid during nitrification, there is a strong tendency for the pH to drop. This is an important consideration because of the relationship between biological reaction rates and environmental pH. Wild, et al. (51) investigated the effect

of pH, in the range of 6.0 to 10.0, on nitrification rates in activated sludge. Maximum nitrification rate occurred at pH 8.4. At pH 7.0 the nitrification rate was about 47 percent of the maximum; at pH 6.0 the nitrification rate was only 14 percent of the maximum.

Downing, et al. (52) suggested an empirical linear equation to estimate the nitrification rate in the pH range of 6.0 to 7.2; the rate is taken to be constant and maximum from pH 7.2 to 8.0. It dorps linearly as a function of pH from the maximum at 7.2 to zero at 6.0. Downing's equation is

$$u = u_m(1 - 0.833(7.2 - pH)) \qquad [26]$$

u = nitrification rate at any pH

u_m = maximum nitrification rate.

No information was found concerning nitrification rates below pH 6.0, except the above equation which is taken to indicate a zero nitrification rate below pH 6.0.

In addition to pH, dissolved oxygen concentration and ammonium concentration are considered to have major effects on nitrification kinetics. The Monod function has been used to describe the relationship between both of these factors and nitrifier growth rates. Temperature affects both the maximum growth rate and half saturation constant for nitrogen (34). (A summary of kinetic constants taken from the literature is presented in reference 34.)

The process of biological denitrification involves the biochemical conversion of nitrate or nitrite to a gaseous nitrogen compound, primarily nitrogen gas. Nitrate and nitrite can be used by a wide variety of bacteria under anoxic conditions to replace oxygen in metabolic reactions. Although denitrification occurs in oxygen depleted environments the biochemical pathways involved are

not anaerobic but are similar to pathways which utilize oxygen as the terminal electron acceptor (34).

Equation [27] can be used to describe biological denitrification with glucose serving as the electron donor.

$$5C_6H_{12}O_6 + 24NO_3^- \longrightarrow 30CO_2 + 18H_2O + 24OH^- + 12N_2 \qquad [27]$$

Production of free hydroxide ions will result in a reaction with components of the carbonic acid system. It can be calculated that 3.57 grams of alkalinity are generated per gram of nitrate nitrogen reduced. Theoretically, denitrification can restore half of the alkalinity that is consumed during nitrification. Alkalinity production, measured in the laboratory, has been reported to be slightly less than the theoretical amount. To reflect laboratory observations the use of 3.0 grams of alkalinity produced per gram of nitrate nitrogen reduced has been suggested for alkalinity calculations in nitrification-denitrification systems (34).

Mineralization of organic nitrogen will also affect digester alkalinity, as described by equation [28].

$$NH_3 + HCO_3^- \longrightarrow NH_4^+ + CO_3^= \qquad [28]$$

For each gram of organic nitrogen mineralized to ammonia, 3.57 grams of alkalinity are theoretically produced. Both mineralization and denitrification help to restore alkalinity destroyed during nitrification.

The presence of dissolved oxygen will inhibit biological denitrification. In as much as aerobic digesters supply oxygen to microbes in the digesting

sludge, it may seen surprising that denitrification has frequently been reported during aerobic digestion (3, 5, 30). This has generally been attributed to a dissolved oxygen gradient in the system where some organisms are at a level of zero dissolved oxygen.

Kinetics of Phosphorus Transformation

Phosphorus is an essential nutrient for bacterial growth. Activated sludge contains approximately 2 to 3 percent phosphorus on a dry weight basis. The principal cellular components that contain phosphorus are nucleic acids, coenzymes, polyphosphates, and phospholipids. Usually one third to one half of bacterial phosphorus is contained in RNA, and from 5 to 10 percent is contained in DNA (53).

Coenzymes are organic compounds that act with enzymes to catalyze specific biochemical reactions. Perhaps the best known phosphorus containing coenzyme is adenosine triphosphate (ATP). ATP is an important intermediate involved in energy transfer in bacterial cells.

Polyphosphates are linear chain anions formed by the condensation of orthophosphate. The polyphosphate content of bacteria can fluctuate widely. They usually cannot be detected in phosphate starved cells, and are low or undetectable during logarithmic growth. In a nutrient deficient (other than phosphates) environment, polyphosphates tend to accumulate in bacteria. These observations have led to the concept that polyphosphates regulate cellular phosphorus and serve as a reservoir from which rapid synthesis of nucleic acids and phospholipids may take place (54).

Most of the lipid material in a bacterial cell is associated with the cell membrane, and most of this lipid contains phosphours. The lipid content of the cell wall is a characteristic of different bacterial groups. The cell

wall of gram positive bacteria contains only traces of lipids, while the cell wall of gram negative bacteria may contain up to 20 percent lipids (55).

Domestic sewage activated sludge bacteria grow in a carbon limited environment, and a sludge containing 2 to 3 percent phosphorus is considered normal. This usually accounts for a removal of about 20 percent of the total phosphorus in the plant influent (56). A few activated sludge wastewater treatment plants, particularly the Rillings Road plant in San Antonio, Texas, have consistently experienced much higher phosphorus removals and higher sludge phosphorus levels. This enhanced phosphorus removal has been attributed to "luxury biological uptake" by some investigators (57, 58, 59). Proponents of the luxury biological uptake theory consider that a high dissolved oxygen concentration enhances phosphate uptake and is responsible for the high phosphorus removal at the Rillings Road plant. In disagreement, Menar and Jenkins (56) proposed that the performance at the Rillings plant is explained by chemical precipitation and entrapment of the precipitate in the sludge flocs. Conditions of increasing pH and high calcium concentration would favor the formation of a calcium phosphate percipitate. It was noted by Menar and Jenkins that high hardness levels in the San Antonio public water supply provide calcium and that high dissolved oxygen levels in the aeration basins coincide with increasing pH.

The actual mode of enhanced phosphorus removal by activated sludge has some implications concerning the aerobic digestion of activated sludge. If biological uptake is responsible for excessive phosphorus uptake, it would suggest that a high dissolved oxygen concentration in an aerobic digester could minimize the mineralization of sludge phosphorus.

Very little work has been reported concerning the release of phosphorus during aerobic digestion of waste activated sludge. With primary sludge it is generally concluded that the digested sludge will retain essentially all the phosphorus in the solid phase.

SECTION III

MATERIALS AND METHODS

This research consisted of investigation of the effect of cold temperatures on the aerobic digestion process and of the fate of nutrients in the sludge during digestion. Studies were performed using primary sludge from two locations and waste activated sludge from two others. Both batch and continuous flow reactors were used.

The effect of temperature on digestion operation and digestion kinetics was determined by operating digesters at varying temperatures, detention times and loading rates, and analyzing for both solids and organic destruction. Changes in settleability and filterability of the sludge were also measured.

Nutrient transformations during digestion were determined by maintaining a nutrient mass balance during the aerobic digestion process.

Physical Apparatus

Several different reactor designs were used during the course of this research, depending on the type of sludge used and objectives of that phase of the research. Description of the apparatus for each phase follows.

Batch Treatment Apparatus

Two types of reactors were used for the batch treatment studies. Experiments performed using waste activated sludge from Somersworth, N.H., and primary sludge from Portsmouth, N.H., utilized one-liter reactors, while nutrient transformation studies performed using waste activated sludge from Hooksett, N.H., utilized 4-liter reactors.

The one-liter reactors were cylindrical glass containers with a conical bottom fitted with a porous glass diffuser. They had a 1-liter capacity with a 5 inch freeboard to allow for possible foaming of the sludge. The reactors were fitted with plastic covers to reduce losses from splashing and evaporation. Air flow rate was measured using rotameters and was adjusted to 1.5 standard cubic feet per hour (SCFH) at a pressure of 3 pounds per square inch (psig). This rate was necessary to provide satisfactory mixing of the reactor contents and was sufficient to maintain approximately 8 mg/l dissolved oxygen in the reactors. The reactors were placed in a 25-gallon water bath to maintain the reactor contents at the desired temperature. The temperature of the water bath was controlled through the combination of a Blue M cooling unit and a Braun Thermomix heating unit. The apparatus used in this part of the research is shown in Figures 2 and 3.

Nutrient transformation studies were performed using inverted 4-liter erlenmeyer flasks which were partially submerged in a controlled temperature water bath. All work in this study was conducted at $20^{o}C$. The water bath was equipped with a Blue M constant flow cooling unit, a Sargent - Welch heater and circulator, and a thermometer. This equipment maintained the water bath temperature at $20 \pm 0.5^{o}C$ throughout the experiment.

Compressed air was bubbled through water to saturate the air and prevent evaporation of digester contents, then passed through a Dwyer visi-float flowmeter, and finally piped into each digester through a fritted glass air sparger. Air flow to each digester was measured and regulated by the flowmeters. Dissolved oxygen concentration in the digesters was measured with a YSI dissolved oxygen meter.

A detailed drawing of the laboratory scale aerobic digesters is shown in Figure 4. The digesters were calibrated at the 3 liter level; this was the

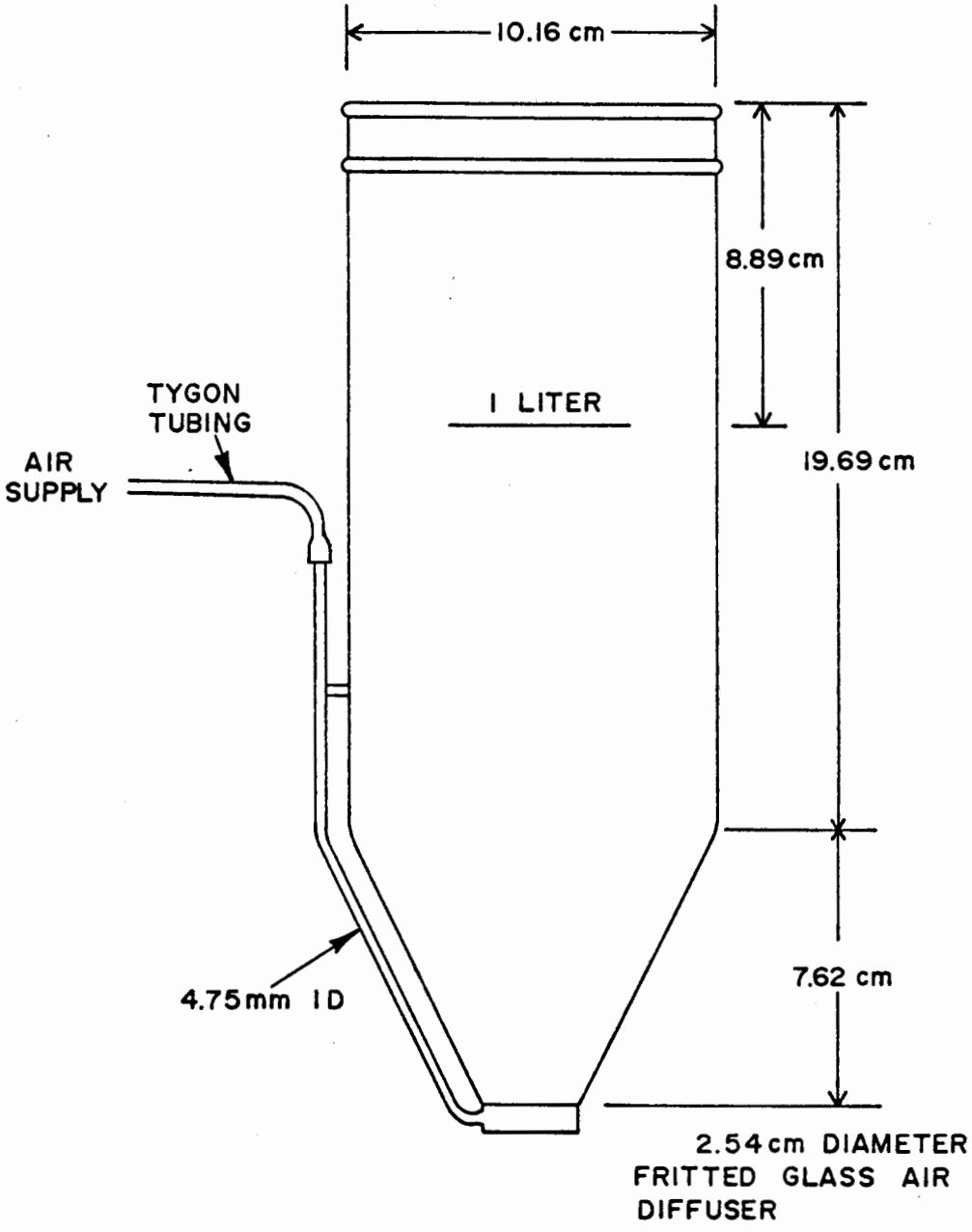

Figure 2

DETAIL OF 1-LITER REACTOR

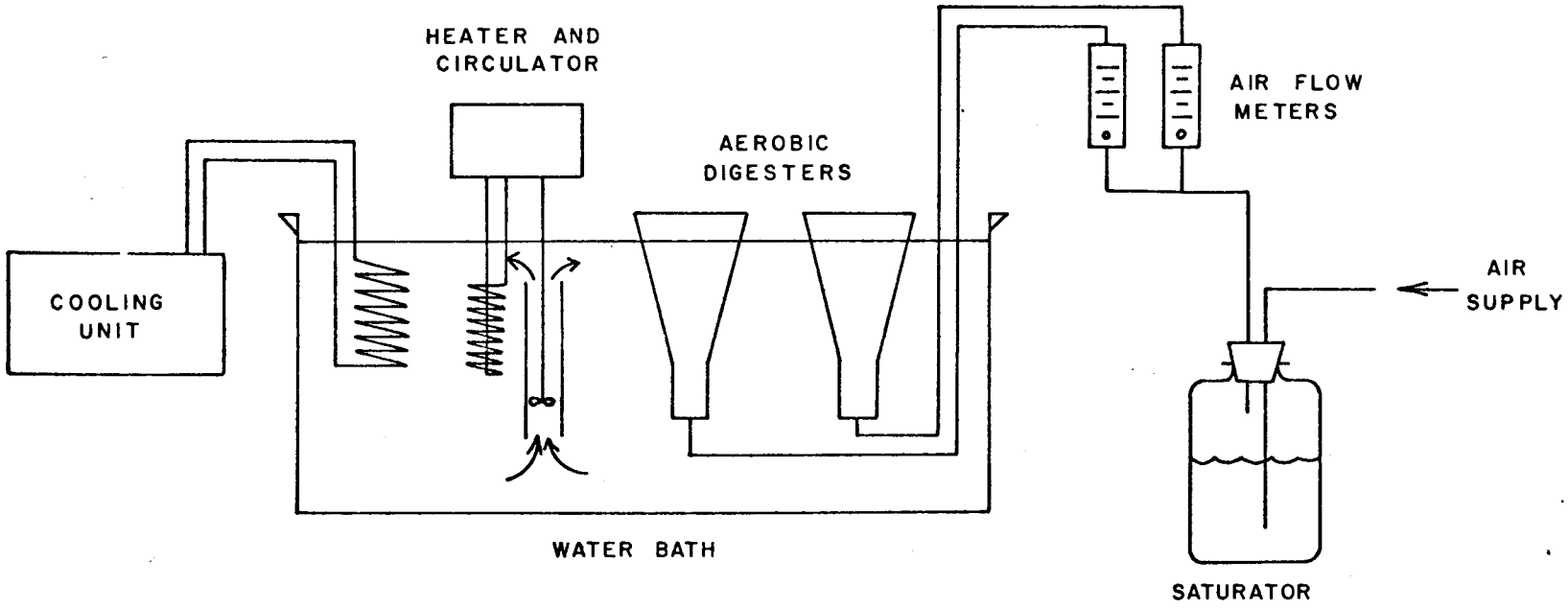

Figure 3

SCHEMATIC OF BATCH TREATMENT APPARATUS - 1-LITER REACTORS

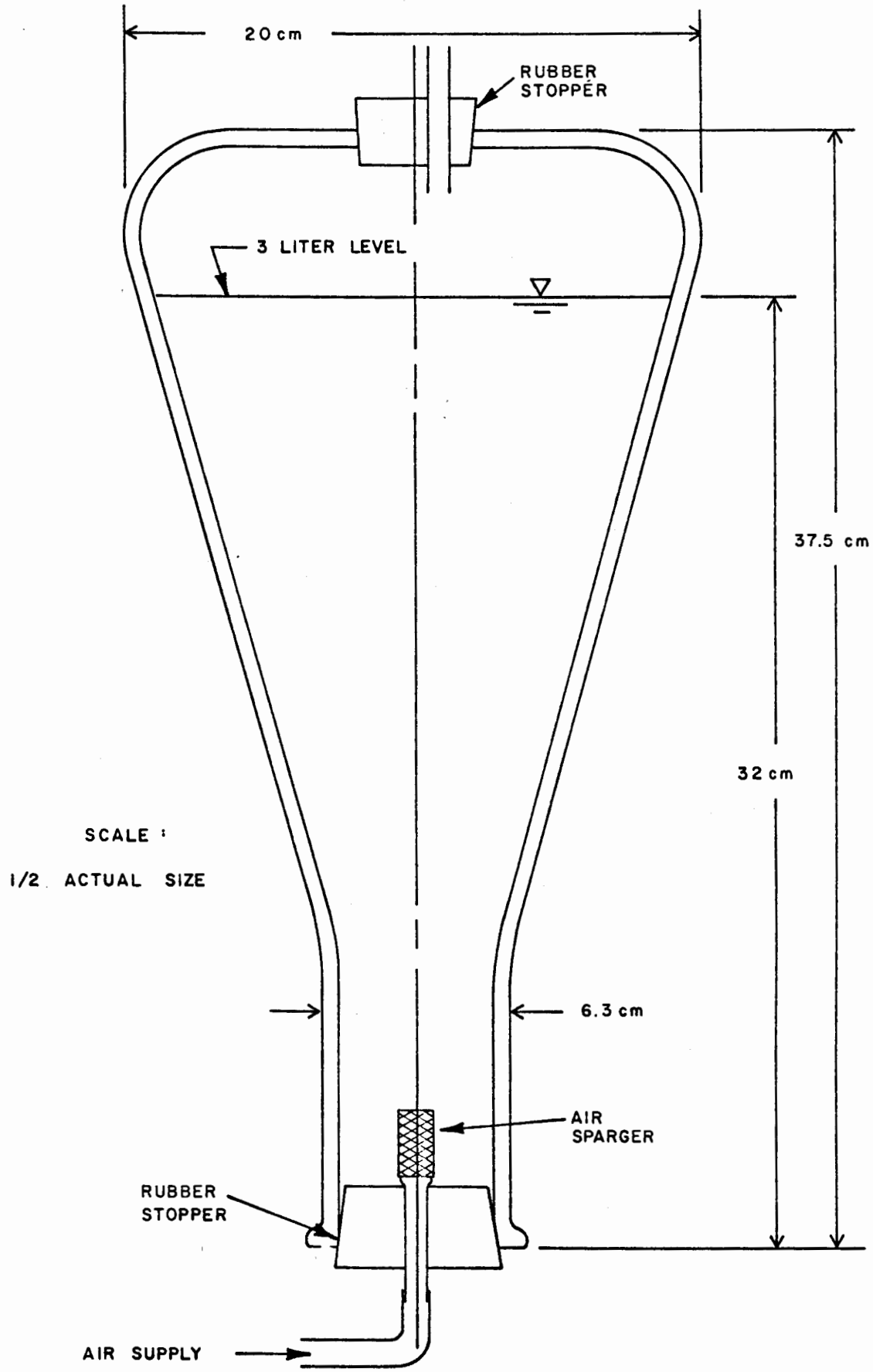

Figure 4

DETAIL OF 4-LITER REACTOR

normal sludge level during digester operation. An air sparger was centered in the neck of each flask and positioned by a rubber stopper which plugged the bottom of each digester (top of the erlenmeyer flask). A hole was drilled in the top of each digester (bottom of flask) and a second rubber stopper with a glass tube passing through it was used to seal this hole. The purpose of this stopper and glass tube was to provide an air vent that would prevent foam from overflowing and escaping the digesters. This stopper also provided access to the digester contents for sampling and feeding.

Continuous Flow Apparatus

The apparatus used for continuous flow studies in all but the nutrient transformation experiments utilized 3-liter reactors. A schematic of this apparatus can be seen in Figure 5 and details of the reactor design can be seen in Figure 6.

The temperature of the individual reactors was maintained within $\pm 0.5^{o}C$ of the desired temperature throughout each trial. A Blue M Constant Flow Portable Cooling Unit was utilized to maintain the operating temperature in the below ambient temperature digester and also served as a source of cooling water for the above ambient digester, which used a Braun Thermomix Variable constant temperature heating unit. The water baths, in both cases, were constantly agitated with mechanical mixers to insure uniform temperature surrounding the digesters.

Compressed air was delivered to the digesters through a pressure regulator, pressure gauge, cotton filter, saturator, rotameter, and a fritted glass air sparger mounted in the bottom center of the digester, as shown in Figure 5. Dissolved oxygen levels were monitored daily with a YSI Oxygen Meter to insure proper DO levels.

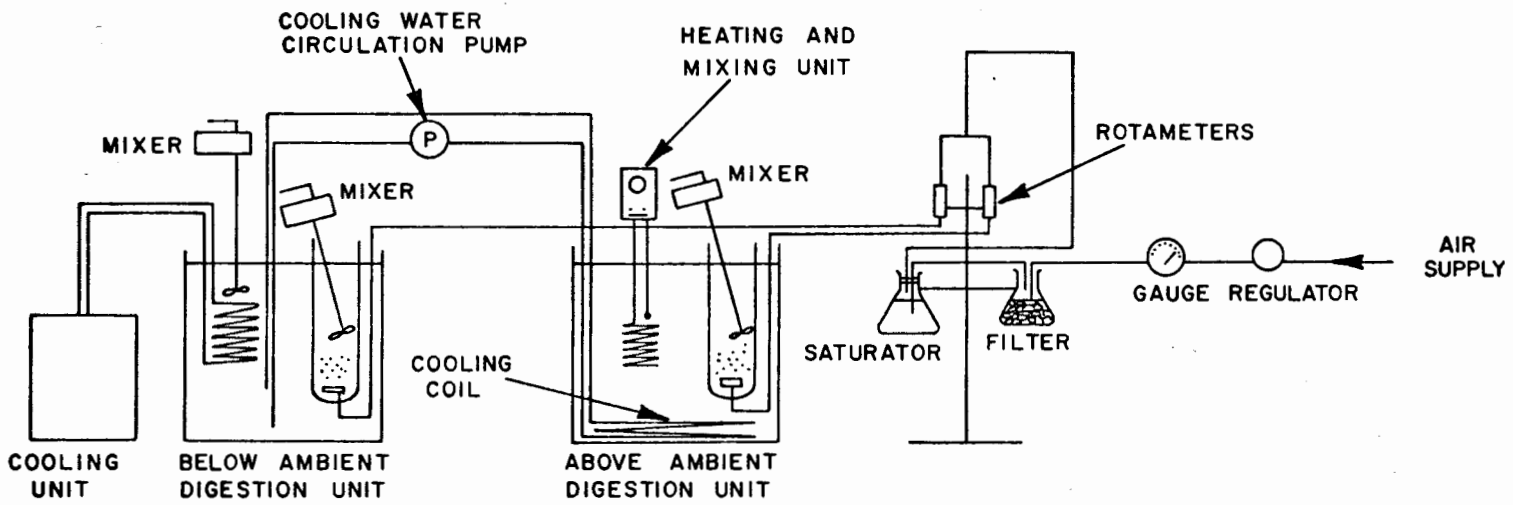

Figure 5

SCHEMATIC OF CONTINUOUS FLOW APPARATUS - 3-liter reactors

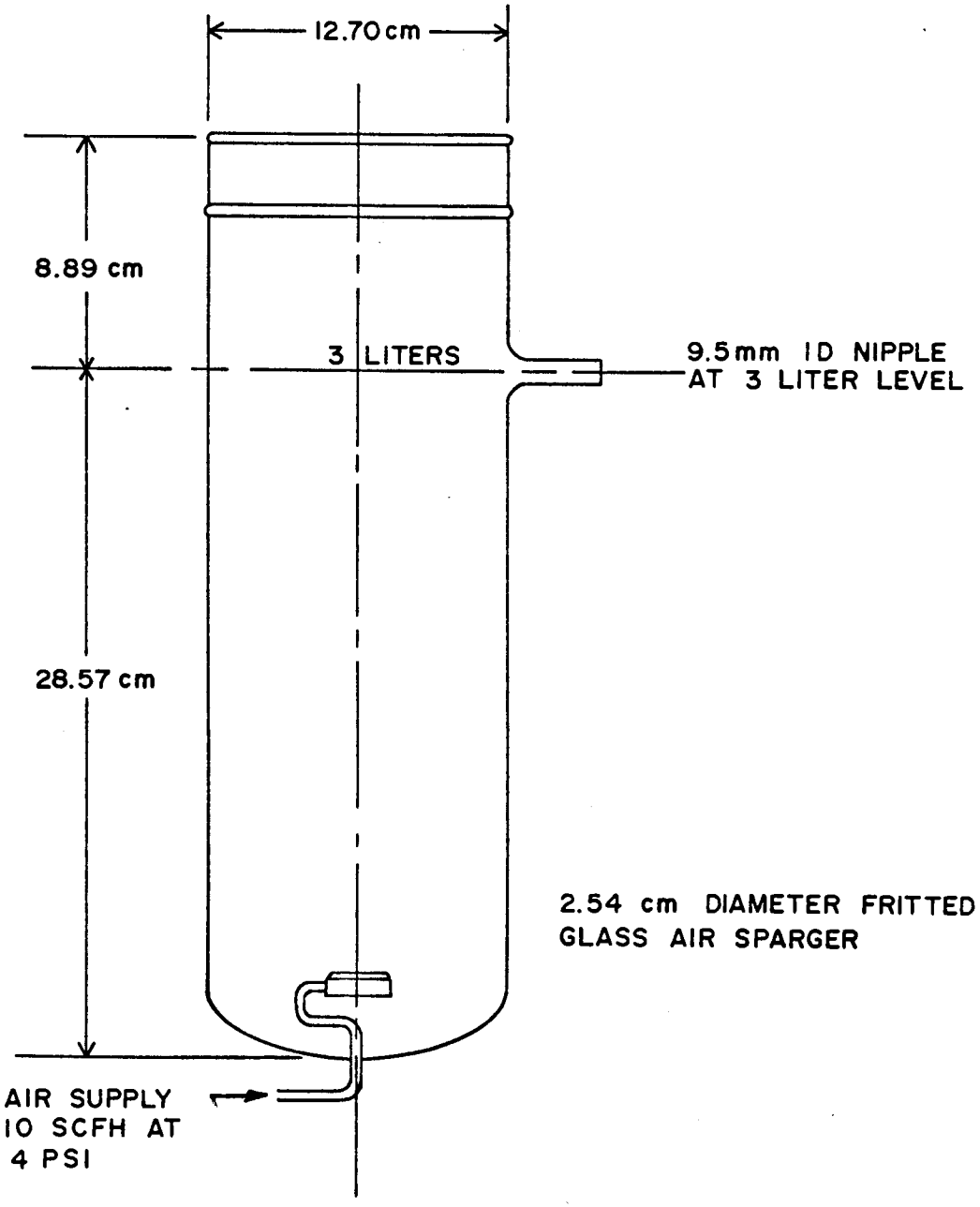

Figure 6

DETAIL OF 3-LITER REACTOR

Adequate mixing was maintained by adjusting the air flow to 10 SCFH and by using a variable speed propeller type stirrer when settling of the sludge was noted to occur.

The continuous flow phase of the nutrient transformation study was performed using the equipment previously described under Batch Treatment apparatus. Sludge was fed to and mixed liquor removed from the reactors daily through the glass tube placed in the top of the reactors.

Mixing Studies

Prior to startup of the aerobic digestion systems, mixing studies were performed on the reactors to determine if completely mixed systems existed. A tracer study, employing Uranine as the tracer, was performed on the reactors under two different modes of mixing. A description of the procedure used for the 3-liter reactor follows.

The concentrated tracer solution was introduced into the reactor by a variable flow peristaltic pump, adjusted to produce a theoretical detention time of 90 minutes in the 3 liter reactor. In the first study, diffused air aeration was the only method of mixing, while in the second study both diffused aeration and mechanical mixing were investigated. Mechanical mixing was accomplished by a variable speed propeller type laboratory stirrer. Measurements of the reactor tracer concentration were made at 2 to 5 minute intervals with a Beckman DB Spectrophotometer. Readings were made for a period of approximately 100 minutes.

Graphical representations of the results obtained from both of these tracer studies are shown in Figures 7 and 8. Evaluation of these graphs reveals that the system, under both modes of mixing, exhibits the characteristics of a completely mixed reactor.

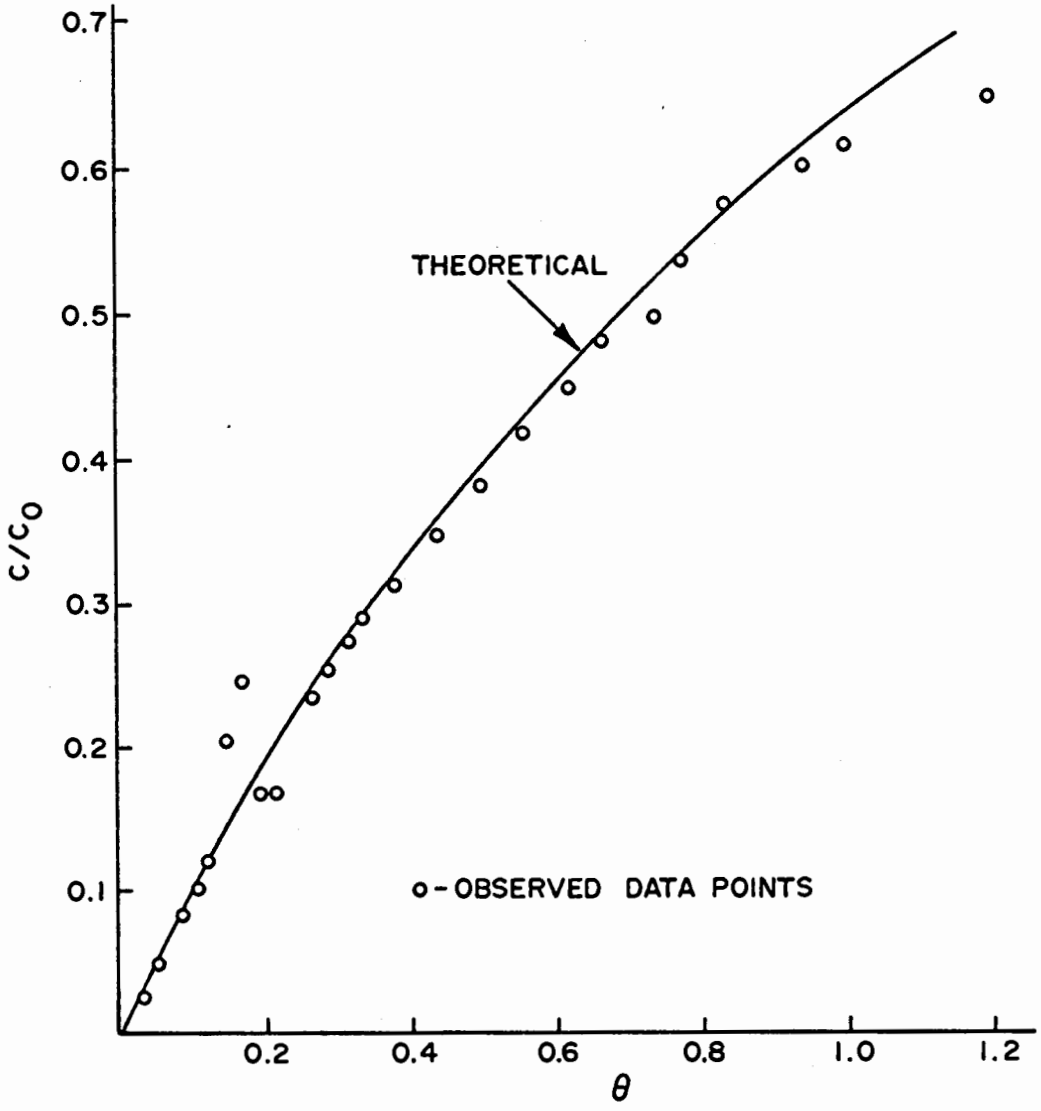

Figure 7

Mixing Study-Aeration and Mechanical Mixing

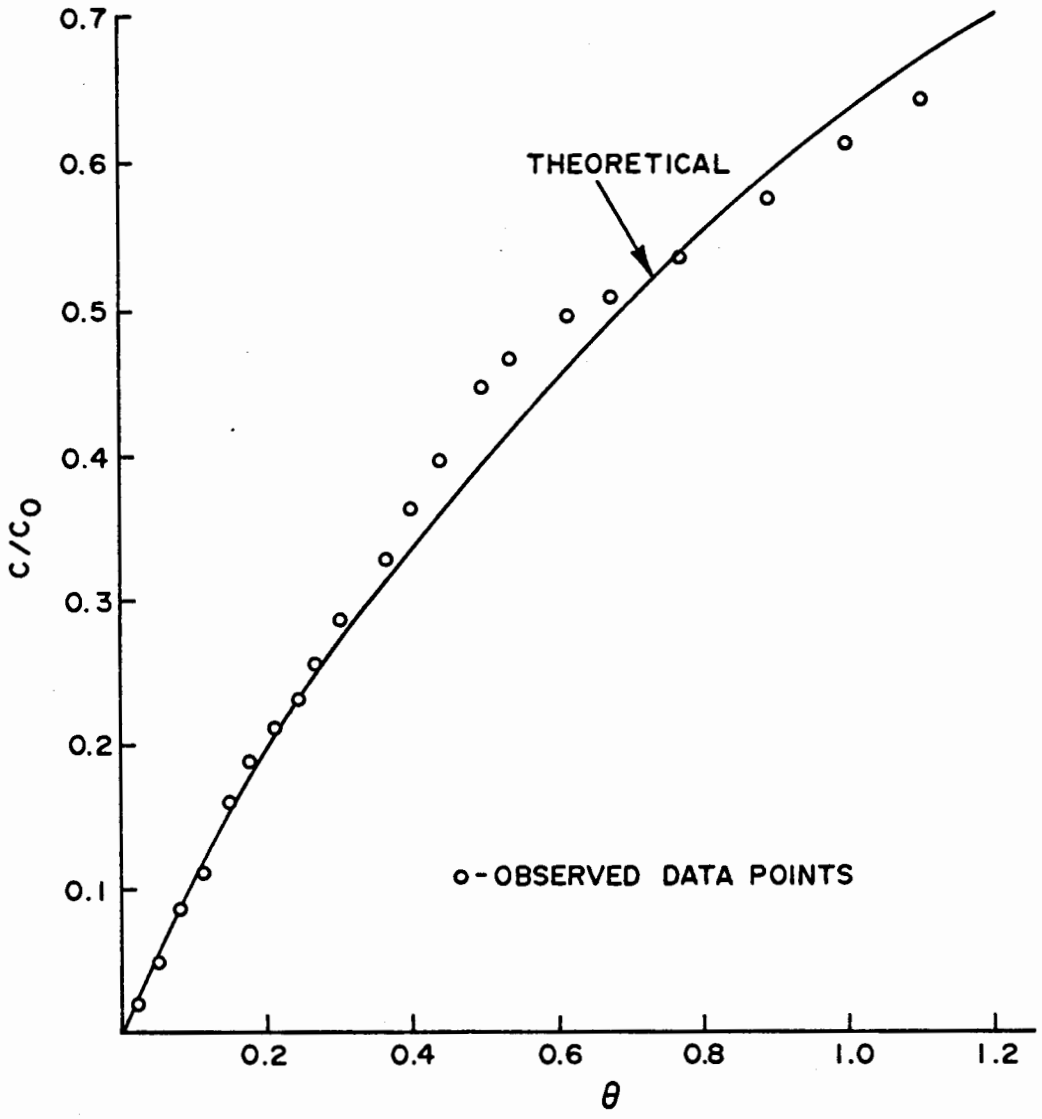

Figure 8
Mixing Study-Aeration Only

Further mixing trials were performed using the actual sludge which was to later be treated in the digester. It was obviously impossible to perform the same type of tracer study as noted above; however, through the use of a YSI Oxygen Meter it was possible to make dissolved oxygen readings at various points in the reactor. No significant difference was noted in the readings and no accumulation of solids occurred in the bottom of the reactor. Therefore it was concluded that the reactor was suitable for the research to be performed, in that complete mixing and proper oxygen levels could be maintained in the digester.

Sludge Characteristics

Sludge used in these studies was obtained from four wastewater treatment plants. These were located at Pease Air Force Base, Newington, N.H.; Somersworth, N.H.; Portsmouth, N.H.; and Hooksett, N.H.

The treatment plant at Pease Air Force Base is a secondary treatment facility utilizing primary classification, high rate trickling filtration, secondary clarification, anaerobic sludge digestion, and sludge drying beds. Sludge samples were obtained from the primary clarifier, homogenized using a Waring blender, separated into 500 ml portions and frozen. These samples served as the initial reactor contents and the daily feed sludge. Characteristics of this primary sludge were as shown in Table 1.

The Somersworth, N.H., wastewater treatment plant is an extended aeration activated sludge facility with air flotation and vacuum filtration of the resulting sludge. Sludge samples were obtained from the return activated sludge lines. Fresh sludge samples were obtained for each batch run. Samples were not frozen since studies performed on the Pease samples indicated that freezing had an effect on some of the physical characteristics of the sludge.

Table 1

RAW SLUDGE CHARACTERISTICS

Parameter	Pease Air Force Base[1]	Somersworth, N.H.[2]	Portsmouth, N.H.[1]	Hooksett, N.H.[2]
BOD_5, mg/l	7,775	-	-	-
COD, mg/l	39,500	12,400	34,200	9,400
Total Solids, mg/l	30,000	13,200	31,000	7,400
Total Volatile Solids, mg/l	23,800	5,700	26,200	5,300
Filterability, %	48.7	-	-	-
Setteability, ml	86	-	-	-

Notes:

1. Primary sludge
2. Secondary sludge

Sludge settleability was increased and sludge filterability was decreased by freezing. The effect on digestion kinetics was negligible.

Portsmouth, N.H., utilizes a primary treatment system followed by chlorination. Samples of sludge were obtained from the sludge wastage line on the day of start-up of the batch laboratory reactors.

The treatment plant in Hooksett, N.H., is a modified extended aeration activated sludge facility treating primarily domestic wastewater. The sludge is digested aerobically and then dewatered on a sludge press. Prior to each run, sludge was collected from the return activated sludge pipe and stored at $4^{o}C$ so that a homogeneous sludge sample could be used for each series of tests.

Sludge solids concentrations were varied for some of the experiments. A Damon IEC B-204 centrifuge was used to concentrate samples to the desired concentrations, with centrate being used for dilution.

Experimental Procedures

Batch Treatment Studies

Fresh sludge samples were collected just prior to start-up of the batch reactors and were blended in a Waring blender. The reactors, suspended in a temperature controlled water bath, were then filled with sludge and aeration was begun. Periodically any accumulated matter on the sides of the reactor was washed down. The reactor contents volume was maintained at the proper level by the addition of deionized water. The pH tended to drop in all reactors except in the control reactor. In some cases the pH was maintained between 6.0 and 7.3 by the addition of 1N sodium hydroxide. Sludge samples were removed at specified time intervals for analysis. Throughout the test runs dissolved oxygen concentration in the reactors was monitored.

Continuous Flow Studies

The continuous flow reactors were designed to operate at theoretical hydraulic retention times of 10, 15 or 42 days. The reactors were initially filled with fresh sludge with the desired volatile solids concentration. The reactors were fed a volume of new sludge daily after removing the same volume of reactor mixed liquor. The feed volume was based upon the desired hydraulic retention time. Feeding was done the same time each day $\pm$ 0.5 hours. The reactors were operated at temperatures ranging from 5 to 30^{o}C. Withdrawal of sludge was accomplished by use of a vacuum pump, with the suction tube placed in the center of the reactor. After the waste sludge was removed, it was measured in a graduated cylinder for accuracy.

Foaming during digestion was a problem in a few of the early trials, particularly if the sludge was septic. To eliminate this problem, Dow Corning Antifoam B was used in dosages of approximately 1 drop per liter. This proved to be very effective. However, later analysis of this material indicated a COD of about 285,000 mg/l and a BOD_5 of 17,500 mg/l. Consequently, use of this anti-foaming agent was discontinued. When fresh sludge was used, foaming was not a problem.

Analytical Procedures

The methods used to analyze the reactor contents and feed sludge for BOD_5, COD, total solids, total volatile solids, nitrogen forms and phosphorus were in accordance with the 14th Edition of Standard Methods for the Examination of Water and Wastewater (61). The procedure used for filterability was obtained from Eckenfelder and Ford (60), and the settleability determinations were based on an experimental procedure used by Graves, et al (62).

Settleability. The settleability of the treated and untreated sludge was obtained by following the procedure of Graves, et al (62). Due to the high solids concentration it was impossible to perform this test by the standard Imhoff Cone procedure. The apparatus used for this determination was a 100 ml graduated cylinder. Treated sludge was removed from the digester and placed in the graduated cylinder up to the 100 ml mark. Readings were then taken every hour for four hours on the amount of sludge settled. Results were reported as ml of sludge settled after four hours. The results obtained from this procedure were rather erratic. This could have been due to several reasons, one of which may have been the effects of sidewall interference between the glass cylinder and the sludge particles. This subject will be discussed at greater length in a later section.

Filterability. As mentioned earlier, the Eckenfelder and Ford (60) procedure for determining the filterability of the treated and untreated sludge was used. A schematic of the apparatus used in this determination is shown in Figure 9.

At the beginning of the test, 200 ml of sludge was poured into the Buchner funnel and the vacuum was set at 30 inches Hg with the stopcock closed. The stopcock was then opened and the stopwatch started. Readings of the volume of filtrate collected in ml were taken at the following intervals: 10, 20, 30, 40, 50 and 60 seconds and 5, 10, 15, 30 and 60 minutes.

Nitrogen

Total kjeldahl nitrogen, ammonium nitrogen and organic nitrogen were determined by acid digestion, distillation, and colorimetric titration. Nitrate ion concentration was measured by use of an Orion Research specific ion nitrate electrode and a Model 701 digital pH/millivolt meter.

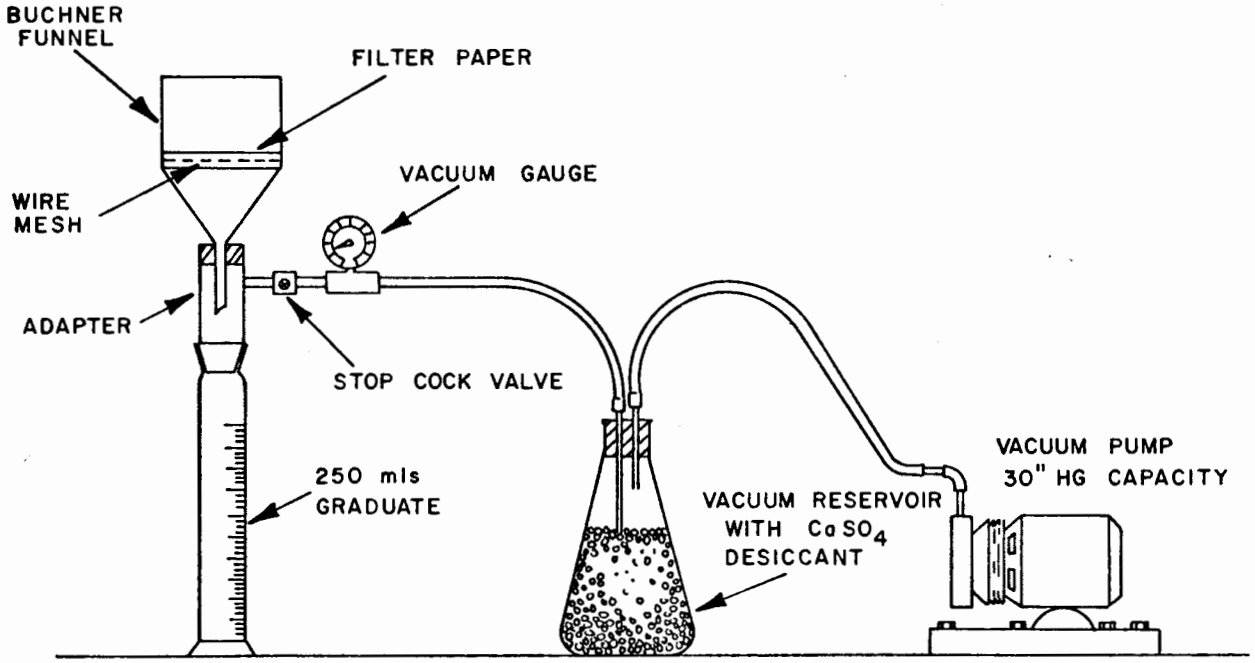

Figure 9

SCHEMATIC OF FILTERABILITY APPARATUS

Phosphorus

Total phosphorus was measured using the persulfate acid digestion procedure. The phosphorus concentration was determined colorimetrically using a Bausch & Lomb Model 710 spectrophotometer.

Specific Oxygen Uptake Rate

A YSI dissolved oxygen meter was used to perform the specific oxygen uptake test. A 150-ml flask was filled with well aerated sludge. The DO probe was immediately placed tightly in the neck of the flask, allowing the displaced liquid to overflow. The contents of the flask were then mixed using a magnetic stirrer. Dissolved oxygen readings were recorded at various time intervals and the uptake rate expressed in units of mg/l/unit time. The specific uptake was then determined by dividing this value by the total volatile solids concentration in the sample.

SECTION IV

DISCUSSION OF RESULTS

Investigations concerning the effect of cold temperatures on the aerobic digestion of waste water sludges and on the fate of nutrients during digestion were carried out using both primary and secondary sludges. The results of these studies are presented below.

Primary Sludge

Aerobic digestion kinetics were evaluated using primary sludge from Pease Air Force Base and from Portsmouth, N.H. Reductions in BOD_5, COD and total volatile solids were used to measure digestion efficiency.

BOD_5 Removal

The results obtained from the BOD_5 determinations on sludge from Pease AFB are shown numerically in Table 2 and graphically in Figure 10. Obvious correlations between detention time, temperature, and BOD removal can be drawn from this figure. Equilibrium was reached after approximately 15 days in all cases except at 5°C where stabilization continued even after 23 days of aerobic digestion. No significant increase in BOD removal occurred after 15 days at the higher temperatures. This observation was confirmed by extending the digestion time for the 10° and 20°C trials to 32 days. No additional reduction occurred between days 15 and 32 at these two temperatures.

The percent BOD reduction at equilibrium with respect to temperature is shown in Figure 11. An increase in BOD reduction from 37 percent at 5°C to 87 percent at 30°C was obtained in 15 days. The 15 day value was selected since no significant increase in removal occurred beyond that point. These results illustrate that higher temperatures are definitely more effective in promoting

Table 2

EXPERIMENTAL DATA - PEASE AIR FORCE BASE

Day	BOD_5 mg/l	COD mg/l	Total Solids %	Total Solids % Volatile	Total Volatile Solids %	Filterability % Filtrate	Settle-ability
				5° C			
0	7775	34,500	3.00	78.0	2.34	48.7	86
2	7275	31,716	2.90	75.1	2.17	14	96
4	6525	31,922	2.85	73.9	2.10	11	96.5
7	5500	28,618	2.66	75.1	1.99	15	96
12	5125	28,148	2.64	75.0	1.98	16.5	96
17	4875	27,622	2.60	74.6	1.93	17	97
23	4500	27,195	2.57	74.2	1.90	18.5	98
				10°C			
0	7775	34,500	3.00	81.00	2.43	48.7	86
2	6000	23,600	2.64	80.85	2.13	9	98
7	3000	16,550	2.03	80.8	1.64	12.5	95.5
9	2800	14,600	1.71	80.5	1.37	13.5	96
11	3100	18,580	1.85	80.0	1.48	12.5	98
14	2550	14,100	1.64	79.7	1.30	16	95
16	1450	17.360	1.53	79.5	1.21	18	94
18	1500	14,250	1.45	78.7	1.14	19	97
21	2100	12,600	1.61	77.4	1.24	25	91
23	1800	14,530	1.62	75.9	1.22	21.5	86
28	1815	15,851	1.60	76.6	1.22	----	----
32	1800	19,730	2.01	73.6	1.47	----	----
				15°C			
0	7775	34,500	3.00	78.0	2.34	48.7	86
2	5430	25,940	2.80	77.8	2.17	11	99
4	2175	26,200	2.75	77.6	2.13	9	98
7	2100	23,600	2.50	79.9	1.99	10	99
9	1950	23,500	2.37	79.6	1.88	11	100
11	1875	21,900	2.29	2.29	1.80	16.25	99
14	1720	18,450	2.9	78.7	2.28	17.5	100
16	1575	17,727	2.13	76.2	1.62	20.7	99
				20°C			
0	7775	34,500	3.00	81.0	2.43	48.7	86
2	3830	21,790	2.45	79.6	1.95	13.5	97
7	1950	18,280	1.73	78.8	1.36	18	96.5
9	2550	17,600	1.60	78.7	1.25	19	97.5
11	2000	17,780	1.66	76.7	1.27	28.5	96
14	2200	15,960	1.59	73.4	1.16	39	92

Table 2 (Continued)

Day	BOD_5 mg/l	COD mg/l	Total Solids %	Total Solids % Volatile	Total Volatile Solids %	Filterability % Filtrate	Settle-ability
16	1900	15,450	1.51	73.6	1.11	37	92.5
18	1850	16,100	1.47	73.3	1.07	41	75
21	2033	16,200	1.48	72.2	1.06	33.5	94
23	1583	13,976	1.45	70.6	1.02	29	88
28	1598	13,634	1.44	69.80	1.00	----	----
32	1700	14,374	1.36	70.7	0.96	----	----
				25°C			
0	7775	34,500	3.00	78.0	2.34	48.7	86
2	4575	31,024	2.83	75.9	2.14	17	94
4	1900	24,595	2.94	74.4	2.18	8.5	96
7	1875	25,055	2.32	74.1	1.71	15.5	97.5
12	1500	20,231	2.14	72.1	1.54	21.2	95
17	1425	18,784	1.98	70.2	1.38	23	97
23	1305	15,846	1.85	69.2	1.28	22	97.5
				30°F			
0	7775	34,500	3.00	81.00	2.43	48.7	86
2	3525	16,050	2.45	79.45	1.94	11	96
4	1380	19,100	2.21	78.6	1.73	15	97.5
7	1250	15,120	2.27	78.7	1.78	17	98
9	1125	18,100	2.20	78.6	1.72	21	100
11	1085	18,700	2.15	76.2	1.63	24.5	98
14	975	17,443	2.04	72.2	1.47	29	98
16	1350	19,639	1.92	70.4	1.35	29	99
18	1650	19,345	1.89	69.0	1.30	24.5	99

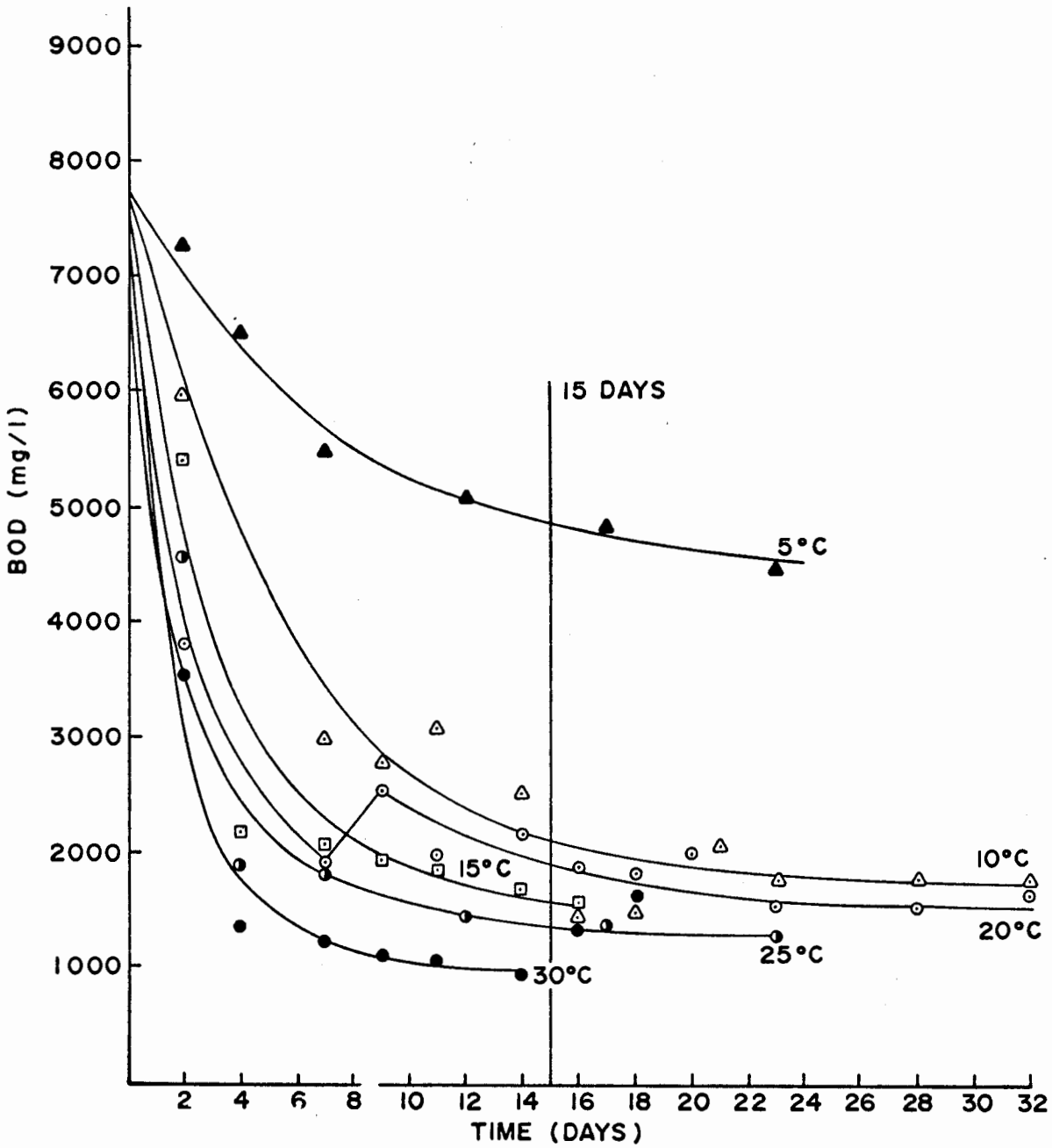

Figure 10

BOD VS TIME - PEASE AIR FORCE BASE

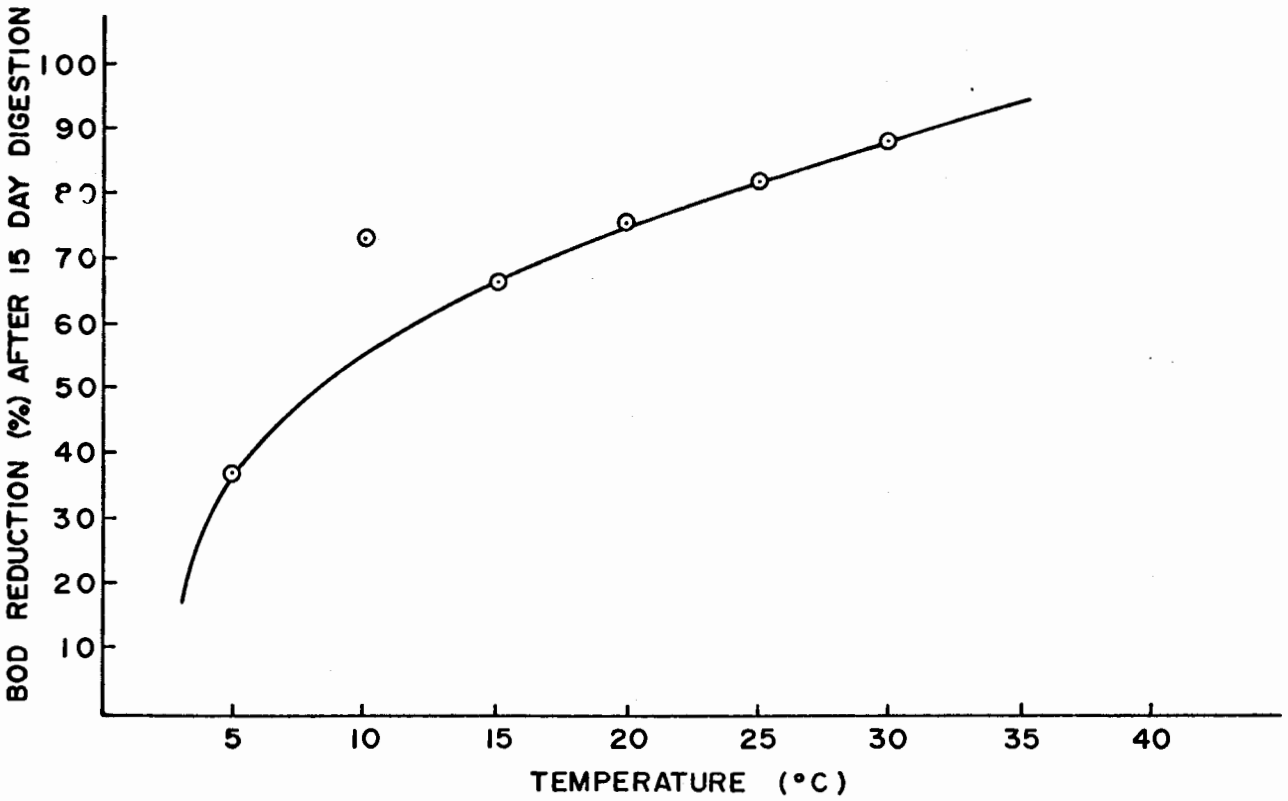

Figure 11

BOD REDUCTION VS TEMPERATURE - PEASE AIR FORCE BASE

BOD reduction, especially where short detention times are involved. However, since the curves in Figure 10 tend to converge after longer periods of digestion, it would seem that the effect of temperature is not as significant at longer detention times.

The organic loading to the digester in all trials was equivalent to approximately 0.01 lbs. BOD_5/cu.ft./day. This loading was low as compared to the values reported by other researchers. However, the primary objective of this investigation was to study the effects of temperature on the aerobic digestion process and a higher organic loading might have exerted more influence on the rate of removal than the temperature.

The effect of a change in operating temperature on the digestion efficiency was demonstrated during the 20°C trial. After approximately 7 days of operation, the heating unit for this reactor failed and the temperature dropped rapidly to 7°C. Since the heating unit could not be repaired immediately, the cooling water flow to the water bath was shut off and the remainder of the trial was carried out at room temperature (average 23°C). Approximately 48 hours elapsed between the time the temperature dropped to 7°C until it stabilized at 23°C. The initial path of the BOD curve at 20°C fell between the 15° and 25°C plots. However, when the temperature dropped the degree of removal also dropped to a rate less than that for the 15°C digester. Once the temperature began to rise and finally level off again, stabilization of the sludge then followed the path shown in Figure 10. This incident proved beneficial in that it served to demonstrate the ability of the system to recover from a severe drop in temperature without causing complete destruction of the acclimated biological mass.

COD Removal

Raw sludge samples from Pease AFB contained 23,800 mg/l total volatile solids and 39,500 mg/l COD. Reactors were operated at temperatures of 5, 10, 15, 20, 25 and 30°C and were fed on a daily basis. Results of COD analyses performed on reactor contents are shown in Figures 12 and 13.

As can be seen the COD reduction trend appears to be the same as for BOD_5 removals except at the higher temperatures where the percent reduction in COD levels off. Unfortunately, there was a significant degree of scatter in this data making analysis difficult.

COD reductions during aerobic digestion were also determined at 20°C using primary sludge from Portsmouth, N.H. The results of this study are shown in Table 3 and Figure 14. As was the case at Pease AFB, the COD of the sludge dropped rapidly at the start of aeration. Essentially all of the COD degradation took place within the first 15 days. Approximately 65 percent of the COD was destroyed during this period.

Solids Removal

The results of the total volatile solids determinations using Pease AFB sludge are shown in Table 2. This data is shown graphically in Figure 15. As expected, the results show that solids reduction increased with increasing time of digestion. Total volatile solids reductions were 13 percent at 5°C and 37 percent at 30°C after 15 days detention. Therefore, a 25°C increase in temperature resulted in nearly a three-fold improvement in volatile solids reduction. This is consistent with the van't Hoff rule which states that biological reaction rates double with each ten degree increase in temperature.

The 10°C and 20°C curves clearly show that no significant decrease in solids occurred between 15 and 30 days detention. These results are in agreement

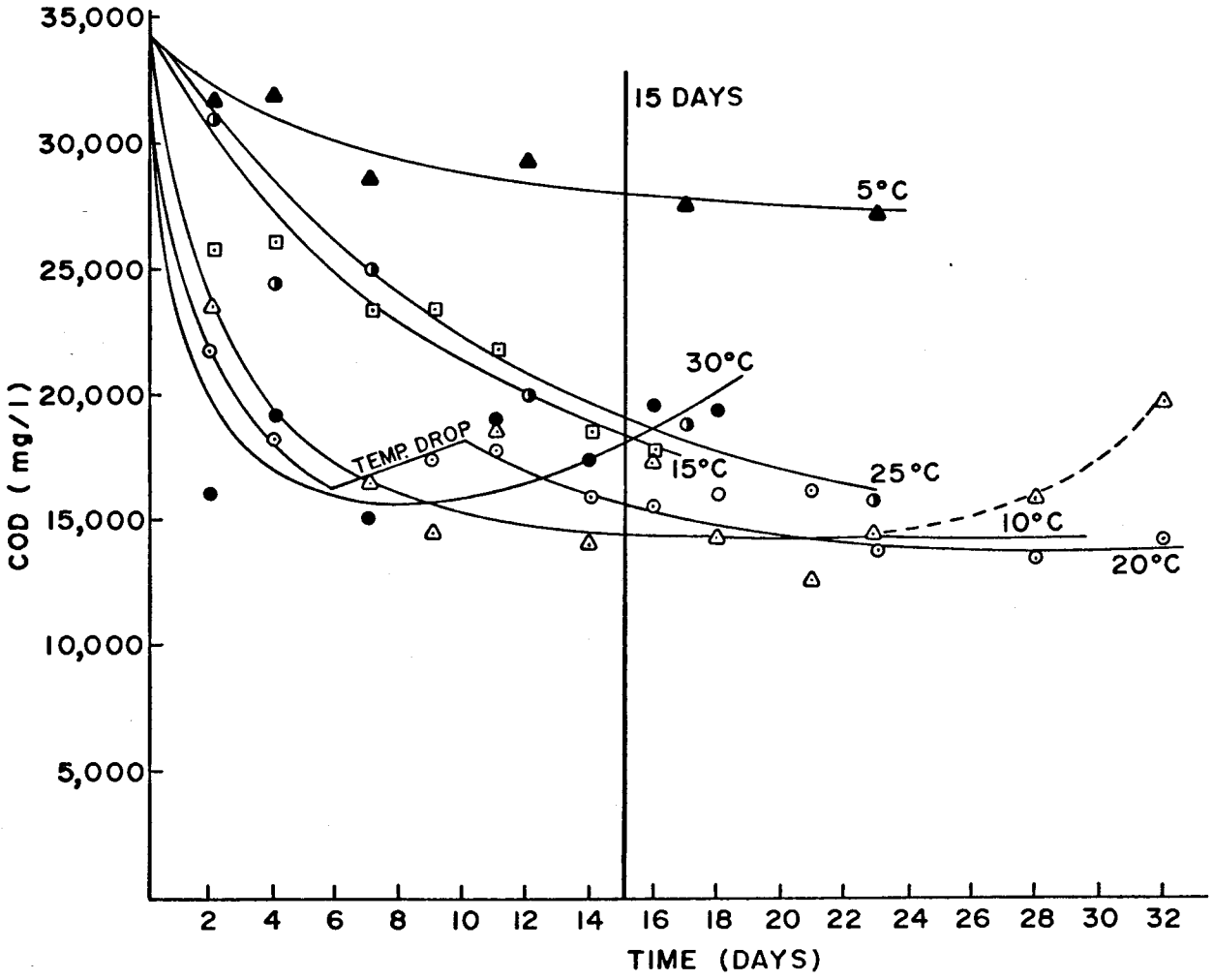

Figure 12

COD VS TIME - PEASE AIR FORCE BASE

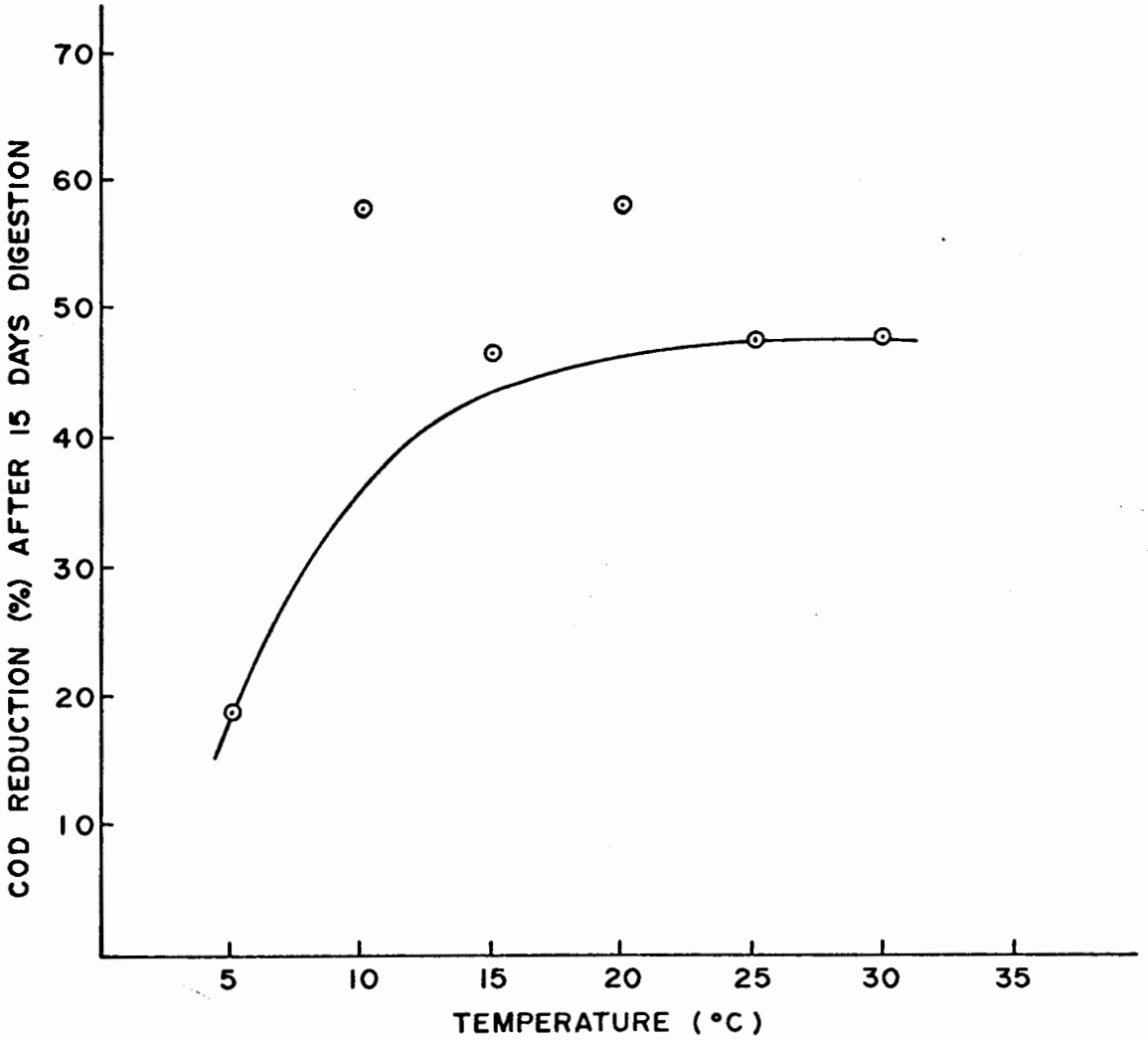

Figure 13

COD REDUCTION VS TEMPERATURE - PEASE AIR FORCE BASE

Table 3

EXPERIMENTAL DATA - PORTSMOUTH, N.H.

Day	Temperature	COD mg/l	Total Volatile Solids %
0	20	34,200	2.62
2	20	31,100	1.98
4	20	19,700	1.33
6	20	16,100	1.21
8	20	14,800	1.14
10	20	13,600	1.07
12	20	13,100	1.04
15	20	12,700	1.00
18	20	12,400	1.02
20	20	12,300	0.97
25	20	12,300	0.96
30	20	12,100	0.98

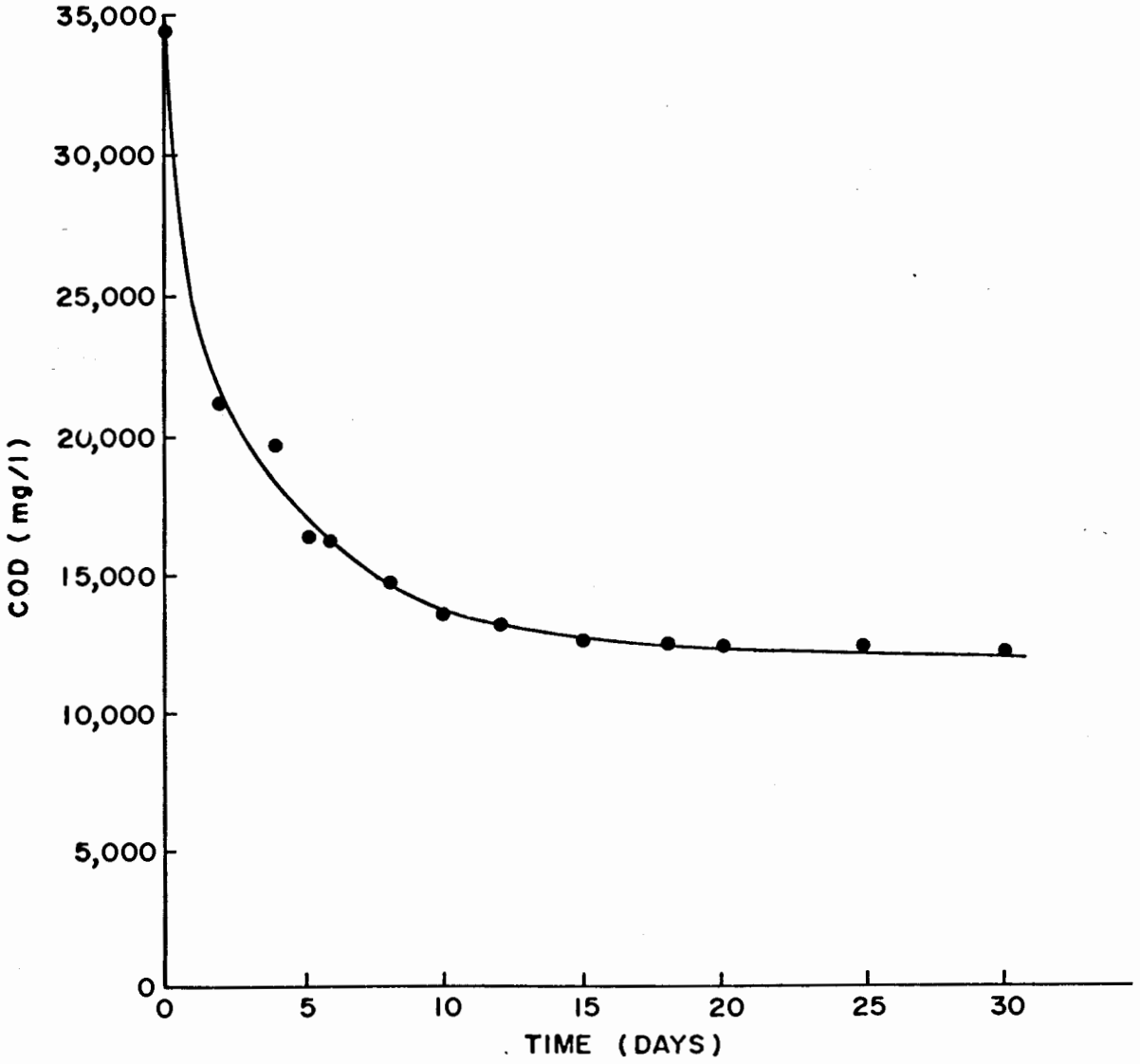

Figure 14

COD VS TIME - PORTSMOUTH, N.H.

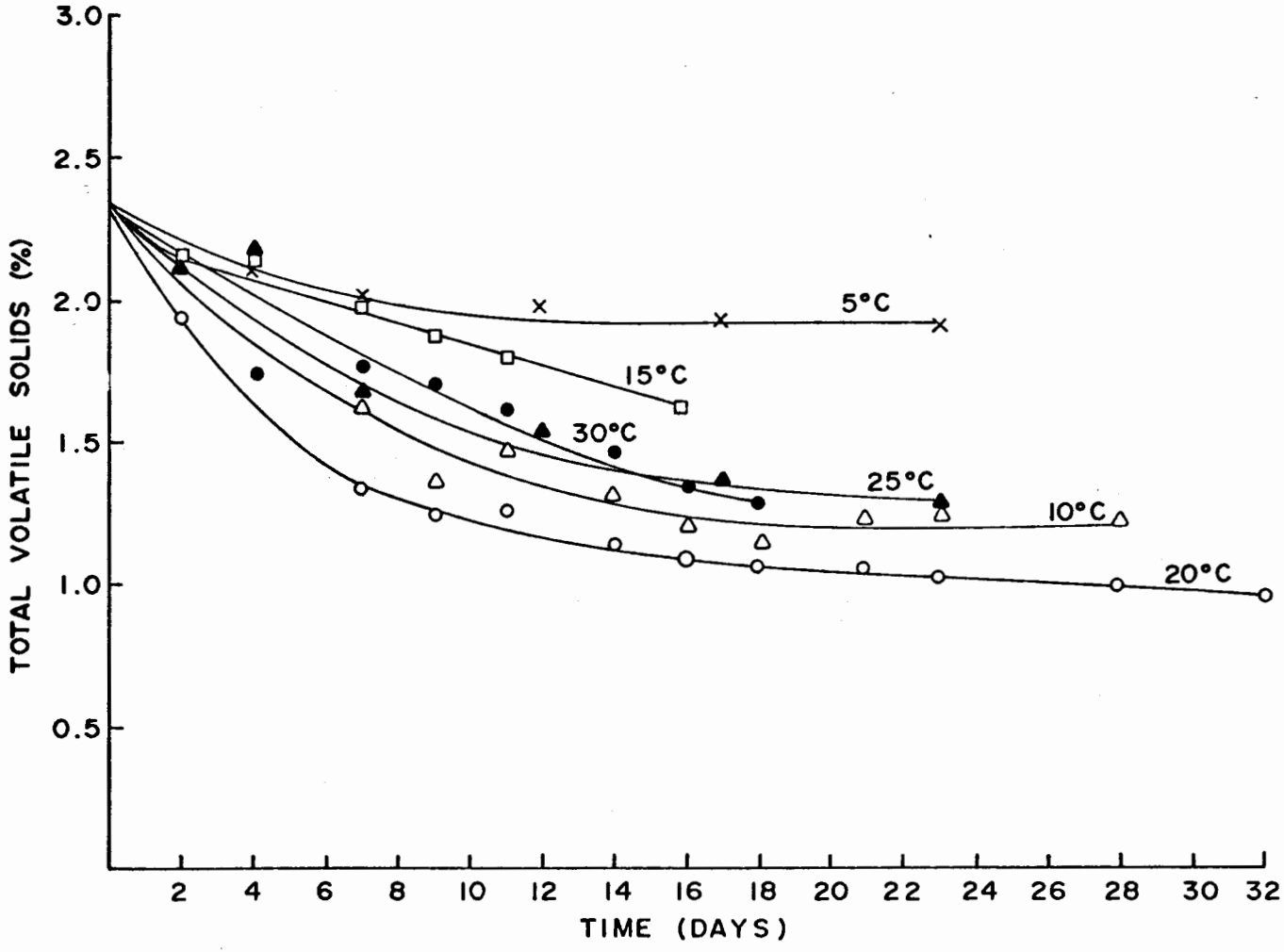

Figure 15

TOTAL VOLATILE SOLIDS REDUCTION - PEASE AIR FORCE BASE

with those of other researchers. At temperatures below 10°C, however, an increase in detention time beyond 15 days would probably result in some additional solids reduction. At higher temperatures, a detention time of 15 days is probably sufficient.

Total volatile solids reductions using the Portsmouth, N.H. sludge followed essentially the same pattern as at Pease AFB. This can be seen in Table 3 and Figure 16. The volatile solids were reduced 63 percent by day 15 when operating at 20°C.

Dewatering Characteristics

The dewatering characteristics of the aerobically digested sludge from Pease AFB, as presented in Figures 17 and 18, show a definite correlation with respect to detention time and temperature. It can be seen that the filterability of the digested sludge does not start to exhibit an increase until after 4 or 5 days of digestion. The filterability steadily increased from that point on until 15 days of digestion had been completed, at which time the filterability started to drop off rather rapidly, especially at the higher temperatures. These results compare well with those obtained by Graves, et al (62). Their results on the filterability of a primary sludge which had under gone aerobic digestion were essentially the same as those shown in Figure 17.

The decrease in the filterability of the digested sludge after 15 days detention can most likely be attributed to the fact that equilibrium had been reached by that time and only a minor degree of organic reduction occurred beyond that point. At this point only a small amount of degradable organic and non-degradable inorganic matter was present. This material was of a much finer consistency than the original digester contents and therefore would have a greater tendency to clog or blind the filter surface and reduce the amount

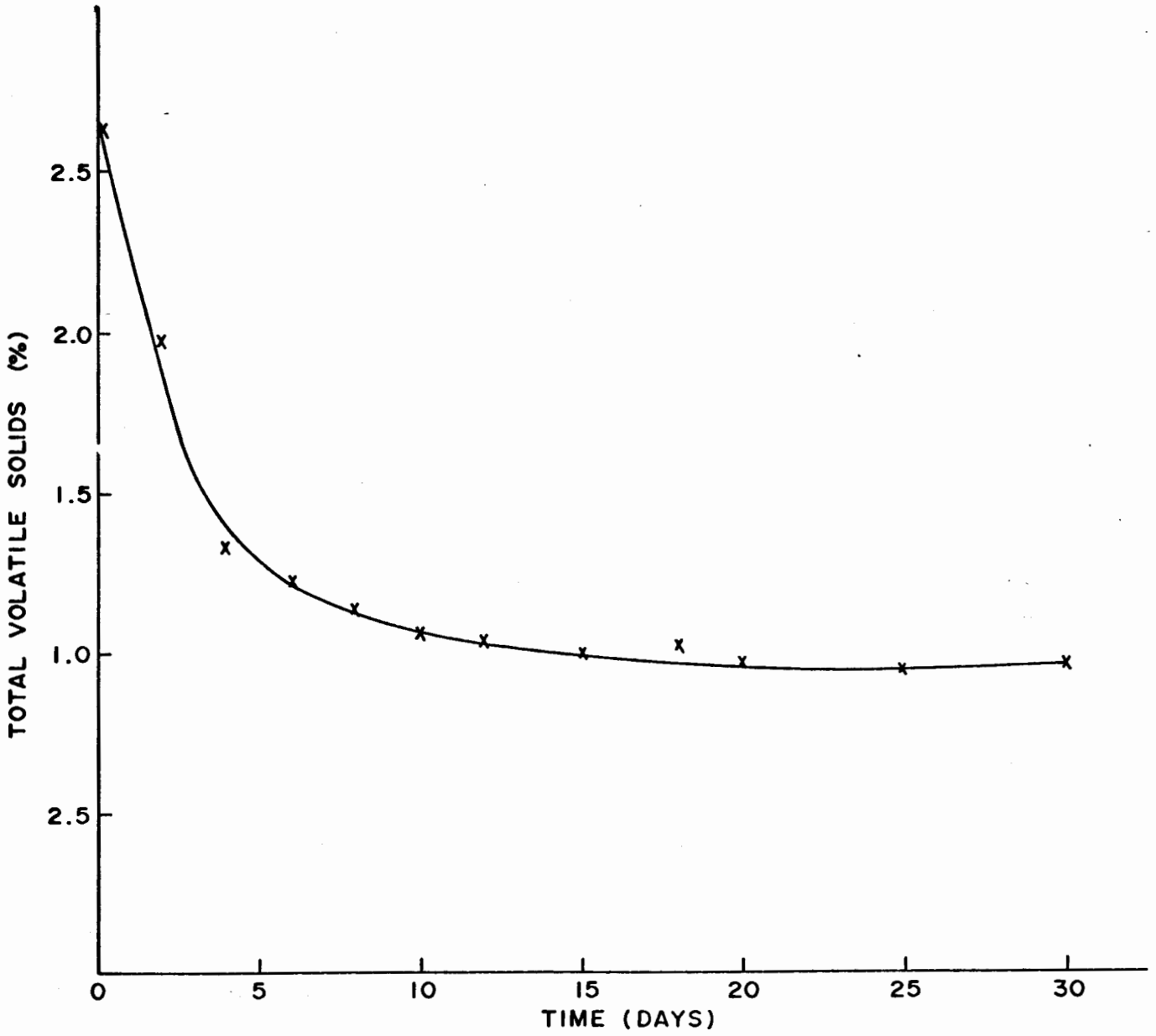

Figure 16

TOTAL VOLATILE SOLIDS REDUCTION - PORTSMOUTH, N.H.

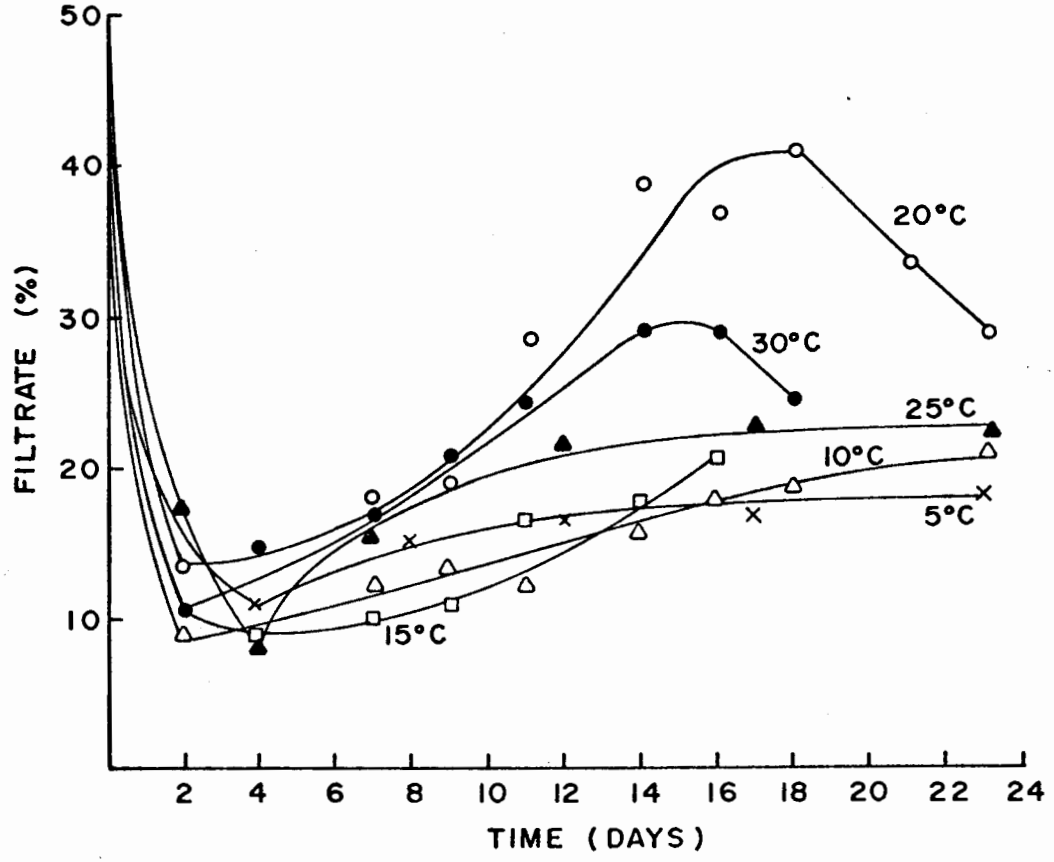

Figure 17

FILTERABILITY OF AEROBICALLY DIGESTED SLUDGE - PEASE AIR FORCE BASE

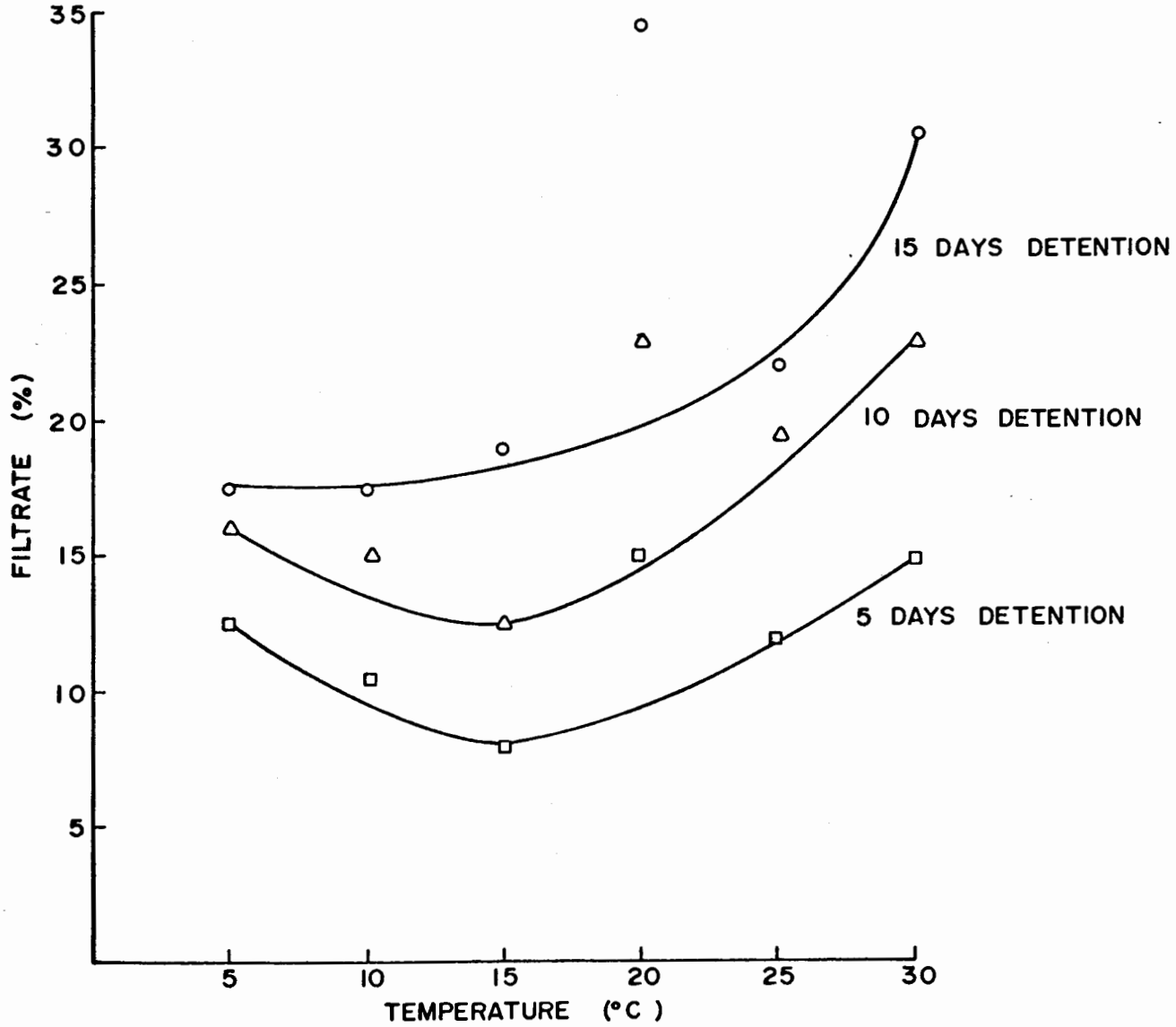

Figure 18

EFFECT OF DETENTION TIME ON FILTERABILITY - PEASE AIR FORCE BASE

of filtrate collected. Use of a different filter with larger pore openings and the addition of chemical treatment should improve the filterability of aerobically digested sludge considerably.

Again, it can be seen that higher temperatures are more desirable. Filterability was much better at higher temperatures than at lower ones.

No correlation could be established between the settleability of the digested sludge and the operating temperature. The only cases where the settling characteristics of the digested sludge approached those of the untreated sludge occurred in the 10°C and 20°C reactors after a digestion period of approximately 23 days. Here again, as mentioned earlier, no logical explanation can be presented to account for the results obtained at these two temperatures.

Two theories have been formulated to explain the poor settleability of the aerobically digested sludge, both of which are directly related to the analytical procedures followed during this determination. The procedure for the settleability test was taken from Graves et al (62), who reported noticeable improvement in the settling characteristics of an aerobically digested mixture of primary and secondary sludge, but stated that the settling characteristics of other sludges (primary and secondary separately) did not exhibit as noticeable an improvement. Due to the physical nature of the settleability apparatus plus the fact that the digesters were vigorously stirred prior to removing the sample to be tested, it is suspected that entrained air in conjunction with the side-wall adhesion effect in the graduate cylinder were the primary reasons for the poor results obtained. This theory was further substantiated by several observations of floating sludge and stratification in the graduated cylinder. These observations did not occur in any particular temperature or time range, and therefore cannot be attributed to either higher temperatures or escaping nitrogen gas which would occur under anaerobic conditions. Therefore

the poor settleability results are most likely due to the trapped air bubbles holding the sludge floc in suspension and the somewhat confined nature of the narrow cylinder used for the test. It would appear that oxygen levels in excess of those required to maintain aerobic conditions have definitely added to the problem and it is recommended that this situation be investigated in future research.

The overall settleability of the sludge did show minor improvement in the later stages of digestion; however, only in the two cases already mentioned did it compare favorably with the results obtained from the raw feed sludge.

At no time during digestion or filtration did the sludge present an objectionable odor. This was an additional indication of adequate aeration and stabilization.

Secondary Sludge

Aerobic digestion studies were also performed using waste activated sludge from Hooksett, N.H. A small study was also done using waste activated sludge from Somersworth, N.H. Both of these facilities use extended aeration, with no primary sedimentation.

These studies were performed using both batch feed and continuous feed modes of operation. Reduction of volatile suspended solids was used as an indicator of extent of digestion.

Batch-feed System

The results of the batch feed aerobic digestion studies are presented in Table 4. Digester 1 had a feed sludge containing 4740 mg/l VSS, while digester 2 contained 11,800 mg/l VSS. The reduction of suspended solids and volatile suspended solids as a function of detention time is shown graphically in Figure 19. It can be seen that the rate of SS and VSS reduction was highest

TABLE 4

BATCH DIGESTION DATA

Hooksett, N.H.

Detention Time (days)	SS (mg/l)	VSS (mg/l)	pH	Alkalinity (mg/l as $CaCO_3$)	Nitrogen (mg/l) TKN	Supernatant Org. N	NH_4^+ N	NO_3^- N	Phosphorus (mg/l) Total	Supernatant	D.O. (mg/l)
					Digester 1						
0.0	6390	4740	7.4	80	476	4.2	3.4	1.6	162	17.9	5.2
1.0	5610	4180	6.1	7	482	2.8	<1.0	21	169	10.8	7.1
2.0	5850	4470	5.3	4	465	3.9	5.0	36	142	16.8	7.1
3.1	4990	3730	5.5	4	437	5.0	12	45	151	19.5	7.4
4.2	5140	3910	5.5	4	431	3.9	20	50	135	25.2	7.5
5.9	4520	3370	5.4	5	420	5.9	39	71	124	31.3	7.9
8.0	4710	3490	5.6	8	386	0	57	84	143	46.8	-
					Digester 2						
0.0	15900	11800	7.2	97	1130	13	3.9	1.8	313	21.0	0.6
1.0	14300	10600	7.0	44	1060	6.2	<1.0	2.7	328	10.3	0.6
2.0	13400	9910	6.7	24	1040	2.8	<1.0	10	273	12.8	0.6
3.1	12500	9130	5.3	4	973	7.6	<1.0	41	265	23.7	6.1
4.2	11700	8320	5.4	4	952	9.5	10	52	294	27.8	6.2
5.9	11500	8400	5.3	5	917	9.2	37	75	263	37.9	6.7
8.0	11200	8170	5.2	6	840	14	49	94	280	63.0	-

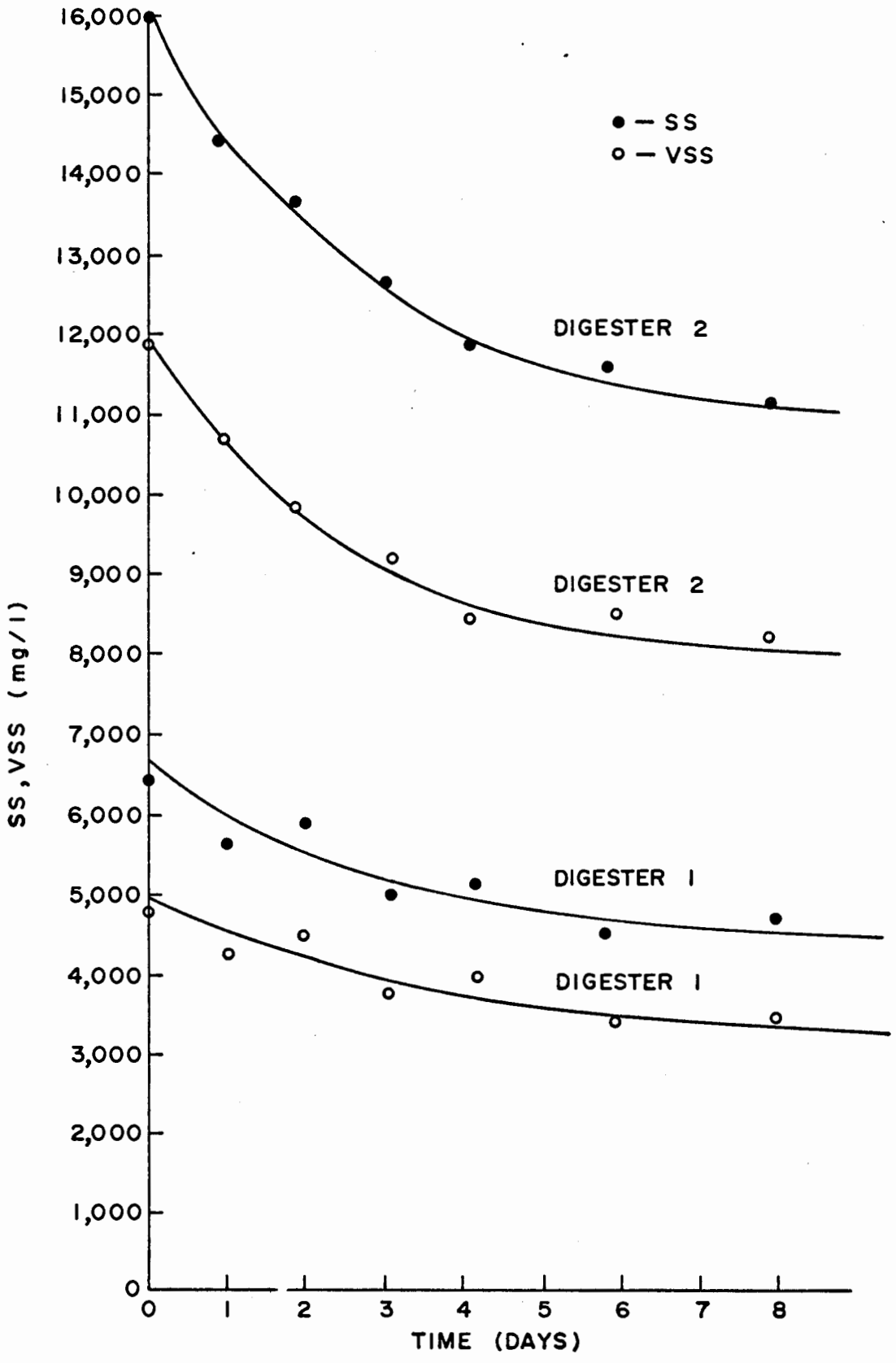

Figure 19

SOLIDS REDUCTION VS TIME - HOOKSETT, N.H.

during the first days of aerobic digestion and the reduction rate decreased with increasing detention time. This agrees with results reported by Eckenfelder (13), Adams et al. (33), and Reynolds (37), all of whom present data from studies of batch aerobic digestion of waste activated sludge. In general their findings were that the digestion rate decreased with increasing digestion time and that little VSS reduction occurred beyond 5 days detention time.

For digester 1, the SS reduction after 8 days of digestion time was about 30 percent and the VSS reduction was about 32 percent. For digester 2, the SS and VSS were reduced 31 and 32 percent, respectively. These values fall below the range of 40-60 per cent VSS reduction, which is generally considered normal for aerobic digestion of waste activated sludge. The feed sludge used in this investigation was taken from an extended aeration activated sludge system. Extended aeration plants, which operate at a low food-to-microorganism ratio, produce sludge which is more auto-oxidized than sludge from conventional or high rate activated sludge plants. Hence, the 32 percent VSS reduction obtained in this experiment seems compatible with previously reported results. It is concluded that digesters used in this experiment were operating in a normal manner.

The data concerning VSS reduction and detention time can be analyzed to determine the endogenous reaction rate constant. From Figure 19 the non-degradable VSS in digesters 1 and 2 were assumed to be 3300 and 7900 mg/l, respectively. By subtracting these values from the total VSS, the concentration of degradable VSS remaining at each detention time was calculated. The ratio of S_t/S_o is graphed as a function of detention time in Figure 20. (S_t is defined as the concentration of degradable VSS remaining at any time, and S_o is the initial concentration of degradable VSS.) By regression analysis the reaction rate constants were determined to be 0.32 and 0.34 $days^{-1}$ (base e) for digesters 1 and 2, respectively.

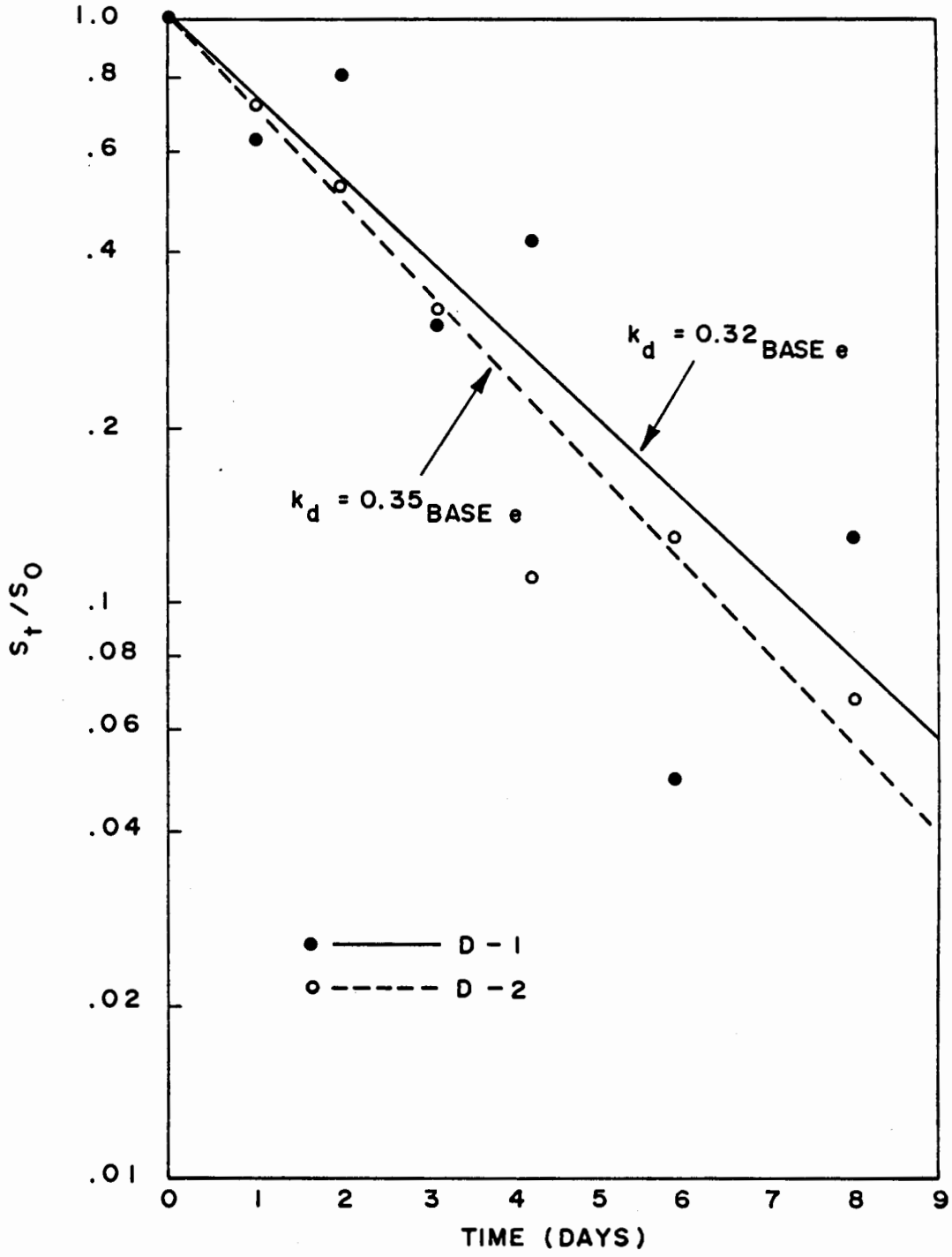

Figure 20

LOG ST/SO VS TIME

In a similar determination Adams et al. (33) found the batch reaction rate coefficient to be 0.325 $days^{-1}$ at 20°C. Activated sludge from a system treating an industrial waste was used in their study. Reynolds (37) reported four values for the endogenous rate constant ranging from 0.53 to 0.85 $days^{-1}$; temperature was not stated. He used activated sludge from a biosorption plant treating domestic sewage with a small portion of industrial waste (mainly from a slaughterhouse).

A similar analysis of the data in Tables 2 and 3 was used to determine reaction rate constants for aerobic digestion of the Pease and Portsmouth sludges. Constants were determined for the Pease sludge at three temperatures. The reaction rate constants were 0.03 day^{-1}, 0.07 day^{-1}, and 0.21 day^{-1} for 5°C, 15°C and 20°C, respectively. The reaction rate constant for the Portsmouth sludge was 0.36 day^{-1} at 20°C. These compare quite favorably to the results obtained with the Hooksett sludge.

The dissolved oxygen level in the low suspended solids digester remained high throughout the digestion time allowed. In contrast, the oxygen level in the high suspended solids digester was only 0.6 mg/l during the first 3 days of digestion, after which it remained above 6 mg/l. During the first few days of digestion, when the rates of VSS reduction and oxygen consumption were highest, the aeration system in digester 2 apparently was not able to transfer enough oxygen to maintain the dissolved oxygen concentration in the desired range of 1 to 2 mg/l.

It appears that this condition of low dissolved oxygen in digester 2 did not have a detrimental effect on VSS reduction. The oxygen demand of the digesting sludge was partially satisfied by reduction of nitrate. This is evidenced by the loss of nitrogen from the system during the first 3 days of digestion, which will be discussed later. By the fourth day, when the dissolved

oxygen level reached 6 mg/l, little or no dentification was occurring. The data in Table 8 show that between the third and sixth days of batch digestion the amount of nitrogen lost from digester 2 was minimal. Between the sixth and eighth days there was a significant increase in the amount of nitrogen lost (assumed to be through denitrification). This result was attributed to a period of about 12 hours during the sixth day when the air delivery system failed. Both digesters were without aeration during this time. The deficiency was corrected and the aeration system worked properly for the remainder of the investigation.

Batch feed studies were also performed on waste activated sludge from Somersworth, NH. Fresh samples were obtained from the return sludge feed line for each run.

The results of COD analyses performed on the waste during digestion are shown in Figure 21. As can be seen, COD removal increased with increasing temperature. At 20°C, COD was reduced approximately 50 percent, whereas at 10°C, it was reduced only 12 percent. Essentially all of this COD was associated with the particulate matter. The COD of sludge filtrate was only 60 to 150 mg/l indicating that the microorganisms were biodegrading all available soluble organic matter.

Continuous-feed System

Two separate studies were conducted using continuous-feed aerobic digesters. Both systems received waste activated sludge from Hooksett, N.H.

The first study utilized inverted 4-liter erlenmeyer flasks operating at 20°C. Results obtained from these studies are presented in Table 5. Examination of the data in Tables 4 and 5 show that a significant difference existed between the feed sludge used during continuous feed digestion and sludge used

TABLE 5

CONTINUOUS FEED DIGESTION DATA - Hooksett, N.H.

Description	Digester 1		Digester 2	
	Feed Sludge	Digested Sludge	Feed Sludge	Digested Sludge
SS (mg/l)	6850	5280	15100	10100
VSS (mg/l)	5240	3850	11700	7230
pH	6.5	5.1	6.9	5.1
Alkalinity (mg/l as $CaCO_3$)	251	5	325	9
Nitrogen (mg/lN)				
TKN	558	431	1100	782
Supernatant Organic N	17	7.3	33	23
NH_4^+ N	64	81	74	106
NO_3^- N	3.4	118	3.0	151
Phosphorus (mg/lP)				
Total	161	161	310	292
Supernatant	53.6	59.7	54.7	90.5

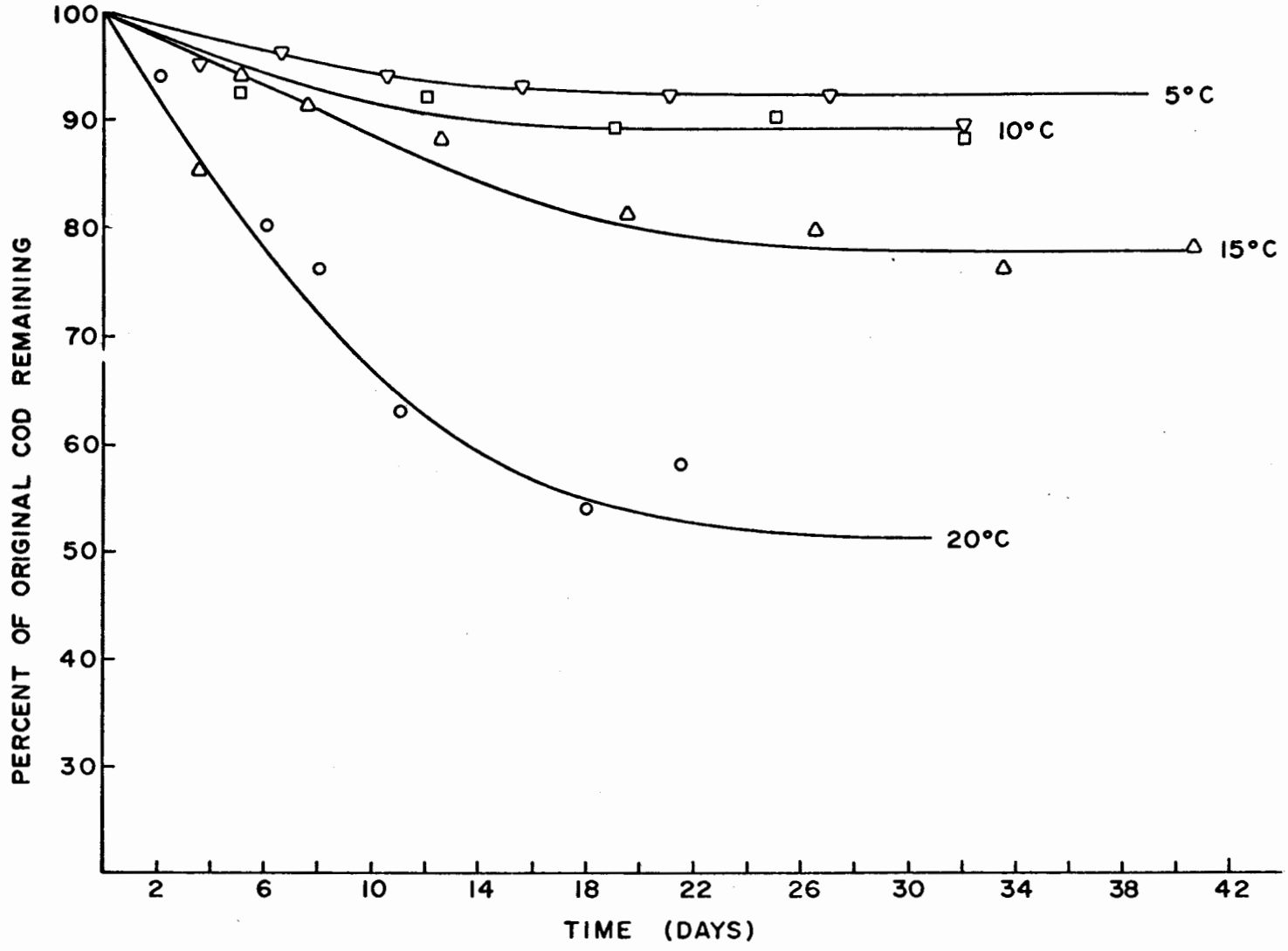

Figure 21

COD REDUCTION VS TIME - SOMERSWORTH, N.H.

for batch digestion. The sludge used in the batch experiments was relatively fresh, in that aeration of the feed sludge was begun within a day after the sludge was taken from the aeration system at the treatment plant. During the continuous studies, the feed sludge was stored at 4°C for up to 25 days before being added to the digesters. Anaerobic storage of the feed sludge apparently caused an increase in ammonium nitrogen, alkalinity, and liquid phase phosphorus.

During continuous feed digestion the oxygen level in each digester was monitored periodically. In digester 2, the dissolved oxygen concentration regularly dropped to less than 1 mg/l within 1 hour after the daily addition of feed sludge. Within 4 hours after feeding, the dissolved oxygen level generally recovered to about 3.0 mg/l. It is thought that this initially high oxygen uptake rate was caused by the rapid substrate utilization of dissolved volatile acids, which were present in the feed sludge as a result of storing the sludge prior to its addition to the digesters. The effect of this period of low oxygen concentration in the continuously fed digesters was probably to stimulate denitrification. A prolonged period of anoxic sludge storage before digestion would generally not occur in a sewage treatment plant. The procedure of sludge storage used for the continuous digestion experiments in this study (and in many others) changes the sludge characteristic to some extent. For this reason it would be desirable to have a ready supply of fresh activated sludge for continuous feed aerobic digestion experiments. In contrast to conditions observed in digester 2, the dissolved oxygen concentration in digester 1 was never observed below 2.0 mg/l, even after the addition of feed sludge.

In digester 1 the SS reduction averaged 23 per cent and VSS reduction averaged 27 per cent during continuous feed operation. In digester 2 the SS reduction was 33 percent and VSS reduction was 38 percent. The fraction of

VSS destroyed during digestion was greater than the fraction of SS destroyed, hence the volatile fraction of the sludge decreased slightly during aerobic stabilization.

The second study was primarily involved with determining the effect of loading rate on aerobic digestion. Loading rates employed were approximately 0.02, 0.06, 0.08 and 0.11 lb VS/ft^3/day. Digesters were operated until equilibrium was achieved, as evidence by a cessation of volatile solids reduction. The digesters were operated at 10°, 15° and 20°C.

At all temperatures, an increase in loading rate resulted in no significant increase in volatile solids destruction. This can be seen in Figure 22. Jaworski, et al. (11) in their continuous flow studies found that higher volatile solids removal was obtained at lower loading rates. The sludge used in their study was primary and waste activated and their low loading rate was similar to the higher loading rates of this study, which used an extended aeration sludge. Primary sludge has more raw food content providing a higher abundance of organics for the bacteria. This enables them to maintain a longer growth stage before endogenous respiration takes over. In their study the same concentration of feed sludge was used throughout the testing. The detention time was shortened or lengthened to adjust the volatile solids loading rate. In the study presented here one detention time with different concentrations of sludge was used to vary the volatile solids loading rates. Therefore, it was difficult to compare the two studies.

The high loading rate of 0.11 lbVS/ft^3/day showed a decrease in efficiency at 15°C, which was also noted by LePage (21). Operating with a loading rate of 0.08 lbVS/ft^3/day, the reactors exhibited a very small variation in efficiency with change in temperature. This was also true for the 0.06 lbVS/ft^3/day loading rate as shown in Figures 11 and 12, respectively.

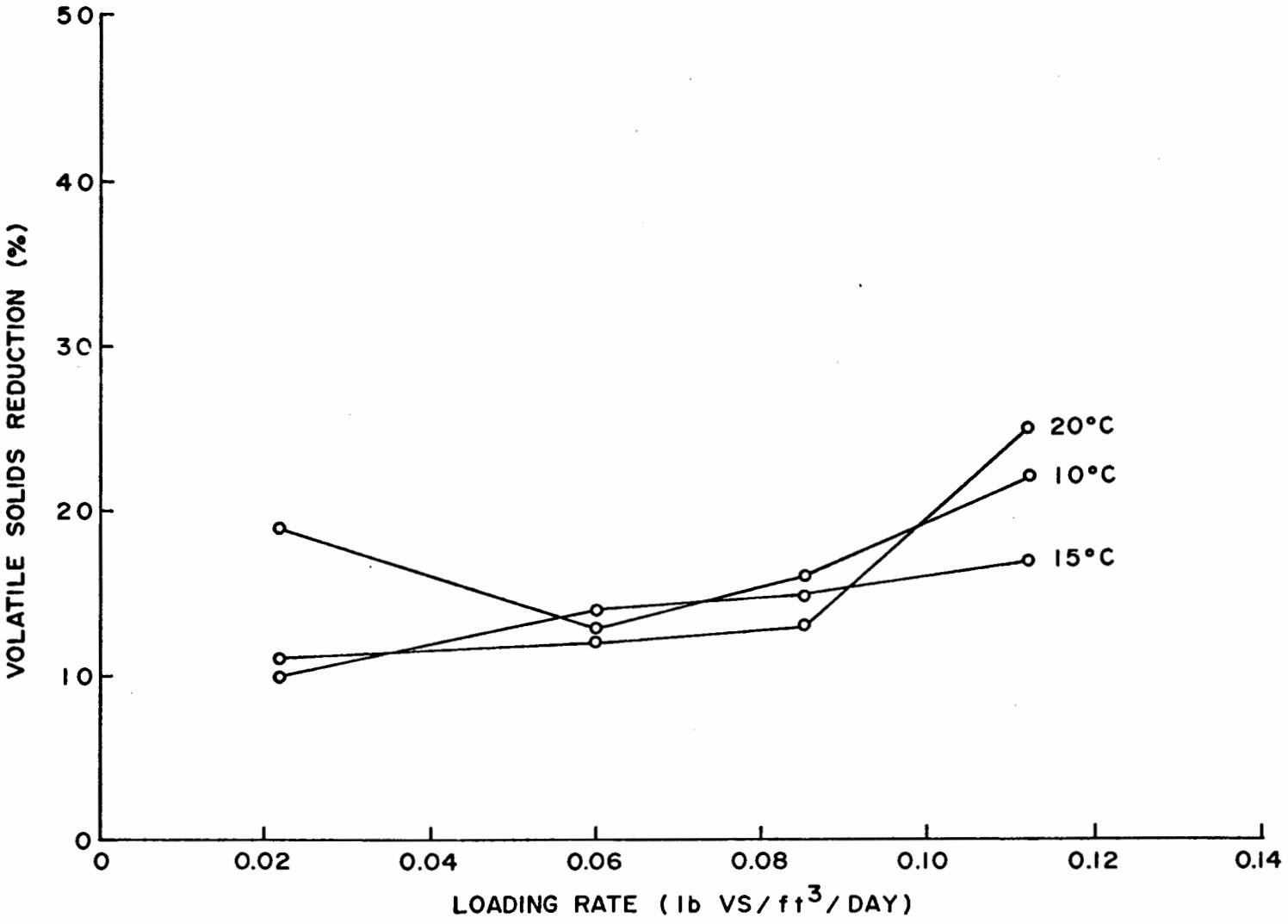

Figure 22

VOLATILE SOLIDS REDUCTION VS TEMPERATURE - HOOKSETT, N.H.

The lowest loading rate, 0.02 lbVS/ft^3/day, had an anomaly in the 10°C run when more than the expected amount of solids was removed. Otherwise a slight improvement was noted between the 15° and 20°C experiments, as shown in Figure 10.

These results indicate that the overall removal of volatile solids is not appreciably affected by temperature when the digester in operating at equilibrium. However, this statement can only be made for the temperature range of 10 to 20°C.

The COD of the sludge was monitored throughout each run as an indicator of the amount of substrate available to the bacteria. Figure 23 presents the results of this study. It is desirable to remove as many of the organics as possible for better stabilization of the sludge.

Loading rates of 0.02 and 0.122 lbVS/ft^3/day resulted in the best overall COD removals at all three temperatures evaluated. Percent removal at the other two loading rates were lower and essentially the same. COD removal at 10°C was about 40 percent of that at 20°C.

pH measurements were performed to insure that pH conditions which would inhibit bacteria did not occur, since extremes in acidity and alkalinity are not conducive to biological action.

The pH of the high loading rate reactor at all three temperatures, 10, 15, and 20°C, evidenced an increase to approximately pH 8 and remained there throughout the tests.

In the 20°C test the pH in the three lower loading rate reactors dropped after the first eight days to a pH of about 5 and then fluctuated from a low of 4.6 to a high of 5.4.

After seven days at 15°C, pH in the two lower loading rate reactors dropped and fluctuated between 4.9 and 6.6. The pH in the third reactor stayed closer to neutral and fluctuated between 6.2 and 7.1.

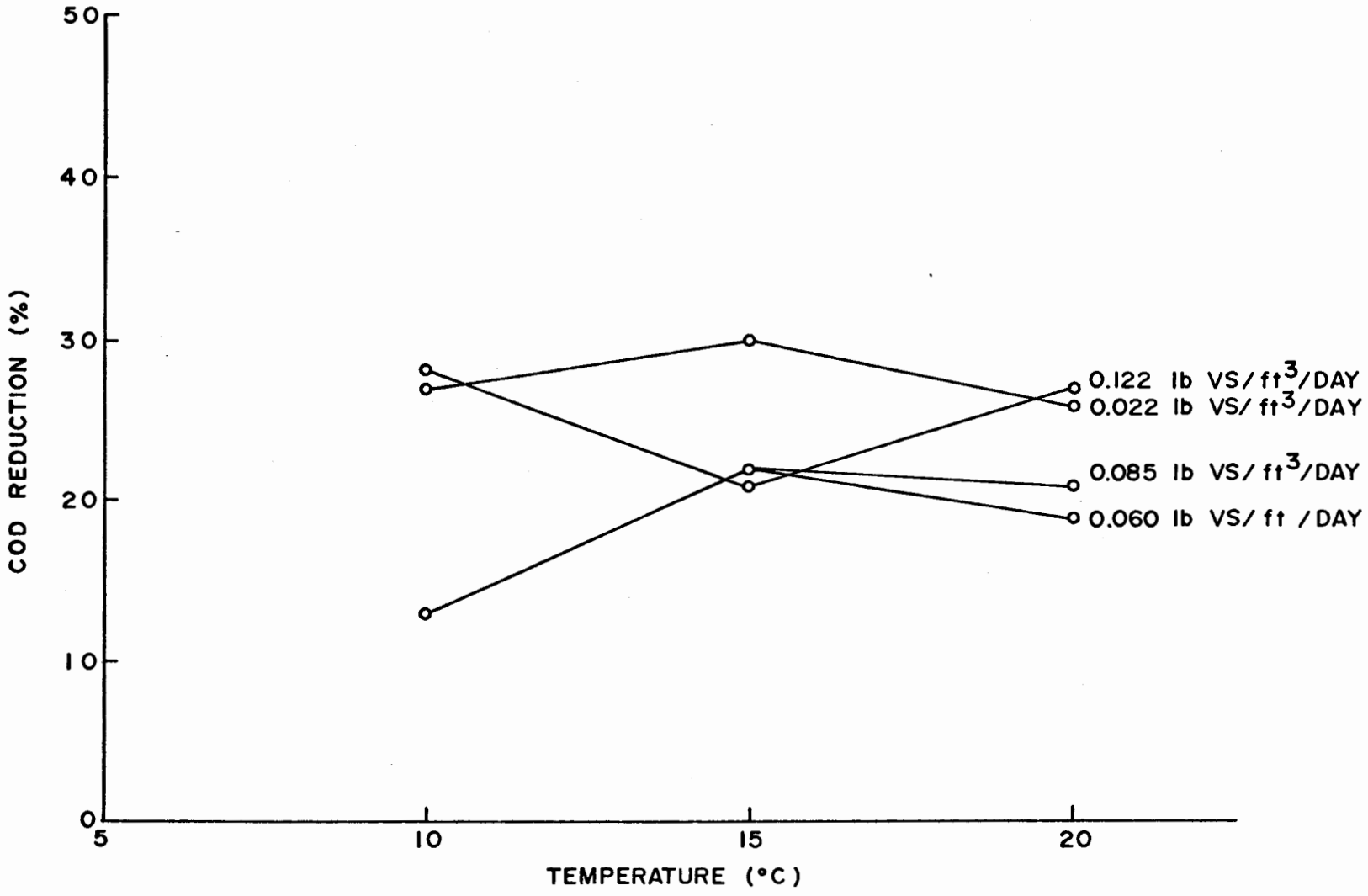

Figure 23

COD REDUCTION VS TEMPERATURE - HOOKSETT, N.H.

TABLE 6

pH DATA - Hooksett, N.H.

10°C

Day	Loading Rage $lbVS/ft^3/day$ 0.022	0.060	0.085	0.122
2	7.1	7.7	7.5	6.3
4	5.5	6.9	7.9	7.1
7	5.2	5.4	5.9	7.8
12	5.4	6.1	5.9	7.9
15	5.2	5.3	5.2	8.0
18	5.2	5.3	5.2	7.9
20	5.3	5.3	5.3	7.9
22	5.4	5.4	5.3	7.9
24	5.4	5.3	5.4	7.9
26	6.3	5.9	5.7	7.8
27	6.8	6.2	6.3	7.9
28	6.2	5.9	6.3	7.8

15°C

Day	Loading Rate $lbVS/ft^3/day$ 0.022	0.060	0.077	0.112
3	5.7	7.8	7.4	6.8
7	4.9	6.7	7.6	7.8
11	5.1	5.5	6.4	8.0
14	5.3	4.9	6.2	8.0
17	6.0	6.6	7.1	7.9

20°C

Day	Loading Rate $lbVS/ft^3/day$ 0.023	0.061	0.073	0.114
5	5.5	8.1	8.5	8.4
8	5.0	7.1	7.6	7.9
11	4.6	5.4	5.0	8.2
15	4.8	4.9	4.9	8.2
18	4.9	4.9	5.1	8.1

TABLE 7

SPECIFIC RESISTANCE DATA - Hooksett, NH

$r = (Sec^2/gram) \times 10^{10}$

10°C

Day	Loading Rate lbVS/ft³/day 0.022	0.060	0.085	0.122
2	4.44	2.99	2.72	2.86
4	2.77	2.95	2.79	2.93
7	3.38	2.43	2.31	3.88
12	2.34	2.23	2.69	4.83
15	2.91	2.54	2.67	3.2
18	2.62	2.33	3.07	6.06
20	2.42	2.06	2.69	6.06
22	2.46	2.21	2.72	4.68
24	5.44	4.06	4.31	6.73
26	4.01	4.08	5.09	10.3
28	5.72	3.91	4.46	10.9

15°C

Day	Loading Rate lbVS/ft³/day 0.022	0.060	0.077	0.112
3	0.858	2.15	3.47	3.47
7	1.19	2.61	1.06	5.85
11	1.72	2.05	2.42	5.06
14	1.47	1.89	2.09	5.2

20°C

Day	Loading Rate LbVS/ft³/day 0.023	0.061	0.073	0.114
5	1.52	4.57	5.8	9.56
8	1.86	4.82	7.64	14.9
11	2.38	2.92	5.3	11.9
15	5.43	3.14	4.45	12.0
18	6.88	2.73	4.19	12.5

In the 10°C run the pH of the first three reactors descended to a pH of approximately 5.3 after seven days. The pH then fluctuated from a low of 5.2 to a high of 6.1. After 24 days at 10°C, the pH in the first three reactors was chemically adjusted to neutral pH of 7 with sodium hydroxide and monitored for four days to discern whether there had been any inhibition from the low pH of 5. A check of the solids removal indicated that no such inhibition occured. This agrees with a study conducted by Lawton, et al. (12) who found that pH did not influence aerobic digestion. Other experimenters also found their pH to fluctuate during the digestion period (11, 23). Values of pH for each of the experiments is shown in Table 6.

Settleability was determined using a 100-ml graduated cylinder. Sludge from the three highest loading rate reactors did not settle, even after six hours under quiescent conditions. In some cases, the sludge from the low loading rate reactor floated. The poor settling may be due to the high solids content of the sludge or to gas entrapment by the sludge particles. Others have also reported similar findings (11).

Filterability, as represented by the specific resistence of the sludge, was also determined. At 10°C filterability of sludge from the first three reactors fluctuated between 2.0 and 5.7 x 10^{10} sec^2/gram. At 15°C the specific resistance in the first three reactors fluctuated between 0.8 and 3.4 x 10^{10} sec^2/gram. The fourth reactor, as usual, had a higher specific resistance averaging approximately 5 x 10^{10} sec^2/gram. At 20°C filterability of sludge in the first three reactors fluctuated between 1.3 and 7.6 x 10^{10} sec^2/gram. The fourth reactor reached a peak with an average of 12 x 10^{10} sec^2/gram. These specific resistance values can be seen in Table 7.

The 15°C run appeared to have the best filterability, as evidenced by its resistance value, while the high loading rate reactor in the 20°C run had the

poorest filterability. These resistance values compare favorably with typical values listed by Metcalf and Eddy (19). However, these values are higher than those found by other experimenters (20,21).

Oxygen uptake rates were measured to determine the amount of oxygen supply required for aerobic digestion. A marked increase in oxygen uptake rate was noted with warmer temperatures, as shown in Table 8. This is due to the higher biological activity at warmer temperatures. This finding was confirmed in a study done by Eikum, et al. (24).

Nutrient Transformations

Nutrient transformation studies were performed on sludge from Hooksett, N.H. to determine the fate of those nutrients during aerobic digestion.

Significant nitrification occurred during both batch and continuous feed aerobic digestion. Inspection of data concerning nitrate nitrogen, alkalinity, and pH (Tables 4 and 5) shows that nitrification consumed essentially all the alkalinity in both digesters under both batch and continuous feed operation. With the consumption of alkalinity, the pH dropped to the range of 5.0 to 5.5. When sufficient alkalinity was present in each digester to maintain pH above 5.5, nitrification of ammonium to nitrate appeared to be complete. This condition occurred in digester 1, the low loading rate digester, for only the first day of batch operation and in digester 2, the high loading rate digester, for the first 3 days of batch operation. During this time the ammonium nitrogen concentration was less than 1 mg/l and the nitrate nitrogen concentration steadily increased. At pH values below 5.5, nitrifying bacteria were apparently partially inhibited. This pH condition prevailed during continuous feed digestion and in the later days of batch digestion. In this environment nitrate concentrations in the batch aerobic digesters continued to increase

TABLE 8

OXYGEN UPTAKE - Hooksett, N.H.

mg O_2/mgVS/day

10°C

Day	Loading Rate lbVS/ft^3/day			
	0.022	0.060	0.085	0.122
2	0.212	0.498	0.40	0.2646
4	0.0535	0.1266	0.5772	0.2984
7	0.0927	1.567	0.2963	0.2806
12	0.2660	1.628	0.5930	0.4631
15	0.0455	0.028	0.2273	0.3795
18	0.0476	0.6790	0.4482	0.3113
20	0.0703	0.2667	0.6152	0.2476
22	1.798	1.6272	1.0632	0.6353
24	0.1068	0.6996	1.1792	0.6748
26	0.2647	1.3705	0.9235	0.9186
28	0.1938	0.8108	1.0112	0.6037

15°C

Day	Loading Rate lbVS/ft^3/day			
	0.022	0.060	0.077	0.112
3	2.4134	1.4824	1.6163	1.4115
7	2.5414	2.1798	2.1058	1.2316
11	1.1001	1.5373	1.5703	1.2893
14	0.1486	2.7658	3.1068	1.5678
17	5.1306	2.9212	2.3045	1.4319

20°C

Day	Loading Rate lbVS/ft^3/day			
	0.023	0.061	0.073	0.114
6	3.2944	2.2140	2.5426	1.8568
8	5.6908	2.8479	2.450	1.9558
11	1.7342	1.4159	2.3329	1.9791
15	6.1998	3.7232	2.6622	2.2569
18	4.2925	3.0169	2.7509	1.8750

with increasing aeration time, and large increases in sludge nitrate concentrations occurred in the continuously fed digesters. The accumulation of ammonium during this time indicates that the nitrifying bacteria were no longer able to oxidize ammonium nitrogen as fast as it was produced through mineralization. It is concluded that pH values in the range of 5.0 to 5.5 only partially inhibited nitrificiation, compared to pH above 6.0. Downing's equation (52) for nitrification rate as a function of pH, cited previously, predicts a zero growth rate at pH 6.0 and below. Nitrification consistently occurred in this study when pH was in the range of 5.0 to 5.5. On the basis of results obtained it must be concluded that Downing's equation is not accurate near pH 6.0 and below.

The alkalinity of digesting sludge will be changed by the biochemical reactions of ammonification (mineralization), nitrification and denitrification. Calculations were made to compare the experimentally measured changes in alkalinity during the continuous feed studies to theoretical predictions. The theoretical values of alkalinity yield for the ammonification reaction, and alkalinity consumption for the net reaction of ammonium oxidized to nitrate and reduced to gaseous nitrogen, were assumed for these calculations. For ammonification the theoretical value is 3.57 grams of alkalinity produced per gram of organic nitrogen released. For denitrification the theoretical value is 3.57 grams of alkalinity consumed per gram of ammonium nitrogen oxidized and reduced. These values and the experimentally determined changes in organic nitrogen, nitrate nitrogen, alkalinity, and loss of nitrogen through denitrification were used to calculate an experimental value for alkalinity consumption caused by nitrification. The calculated values were 6.2 and 6.5 grams of alkalinity consumed per gram of ammonium nitrogen oxidized to nitrate for digester 1 and 2, respectively. These figures are within the range of 6.0 to

7.4 grams of alkalinity per gram of ammonium nitrogen oxidized which other investigators have reported (34), however they are lower than the theoretical value of 7.14.

The results concerning nitrification, pH, and alkalinity agree with theorectical expectations and the findings of other researchers. The possibility of a low pH environment in aerobic digesters should be recognized because of its effect on sludge stabilization and nitrification kinetics.

PH is known to effect the growth rate of bacteria. The pH range of 6.5 to 7.5 is generally considered optimum for most activated sludge bacteria. Yet the possibility of changing pH, particularly in batch digesters, has not been mentioned in studies concerned with the determination of the endogenous rate coefficient (33,37). Decreasing digester pH could have been responsible for the discrepancy between values reported for the endogenous rate constant by different researchers.

The amount of nitrogen that was lost, assumed to be through denitrification, was related to the dissolved oxygen concentration. The data from this investigation were not sufficient to discern a qualitative relationship, however, general trends can be seen. Denitrification in digester 2 during batch digestion has been discussed. It was noted that denitrification occurred when the dissolved oxygen concentration was less than 1 mg/l. After 8 days of aeration about 17 per cent of the nitrogen in digester 2 was lost. In digester 1 no significant amount of denitrification occurred during the batch stabilization test, and the dissolved oxygen concentration remained high throughout the aeration period. During continuous feed digestion results followed the same pattern. In digester 2 the nitrogen lost from the system averaged 15 per cent of the total nitrogen in the feed sludge, and dissolved oxygen concentrations less than 1 mg/l were regularly observed. In digester 1 only about 2 per cent

TABLE 9

NITROGEN DATA FROM DIGESTER 1 - BATCH AEROBIC DIGESTION

Detention Time (days)	VSS Reduction (%)	Total N (mg/l)	Solid Phase N (mg/l)	Solid Phase N (% of SS)	Solid Phase N (% of VSS)	Liquid Phase N (mg/l)	Liquid Phase N (% of Total N)	Nitrogen Lost (% of Initial Total N in Digester)	Mineralized Nitrogen* (% of Initial Total N in Digester)
				Digester 1					
0.0	-	478	468	7.3	9.9	9.2	1.9	-	
1.0	12	503	478	8.5	11.4	25	5.0	-5.2	
2.0	6	501	456	7.8	10.2	45	9.0	-4.8	
3.1	21	482	420	8.4	11.3	62	13	-0.8	
4.2	18	481	407	7.9	10.4	74	15	-0.6	
5.9	29	491	375	8.3	11.1	116	24	-2.7	
8.0	26	470	320	6.8	9.2	150	32	1.7	
0.0	-	1132	1113	7.0	9.4	18.6	1.6	-	1.6
1.0	11	1060	1050	7.4	9.9	9.9	9.9	6.4	7.2
2.0	16	1053	1039	7.7	10.5	14	1.3	7.0	8.2
3.1	23	1014	964	7.7	10.6	50	4.9	10	15
4.2	30	1004	932	7.9	11.2	72	7.2	11	18
5.9	29	992	871	7.6	10.4	121	12	12	23
8.0	31	936	779	7.0	9.5	157	17	17	31

* Sum of Liquid Phase Nitrogen and Nitrogen Lost.

TABLE 10

PHOSPHORUS DATA FROM BATCH DIGESTION - Hooksett, N.H.

Detention Time (days)	VSS Reduction (%)	Solid Phase P (mg/l)	Solid Phase P (% of SS)	Solid Phase P (% of VSS)	Liquid Phase P (mg/l)	Liquid Phase P (% of total P)
			DIGESTER 1			
0.0	-	129	2.0	2.7	17.9	12
1.0	12	136	2.4	3.3	10.8	7.3
2.0	6	130	2.2	2.9	16.8	11
3.1	21	127	2.5	3.4	19.5	13
4.2	18	122	2.4	3.1	25.2	17
5.9	29	116	2.6	3.4	31.3	21
8.0	26	100	2.1	2.9	46.8	32
			DIGESTER 2			
0.0	-	267	1.7	2.3	21.0	7.3
1.0	11	278	1.9	2.6	10.3	3.4
2.0	16	275	2.0	2.8	12.8	4.4
3.1	23	264	2.1	2.9	23.7	8.2
4.2	30	260	2.2	3.1	27.8	9.7
5.9	29	250	2.2	3.0	37.9	13
8.0	31	225	2.0	2.8	63.0	22

TABLE 11

NITROGEN DATA FROM CONTINUOUS FEED DIGESTION - Hooksett, N.H.

Description	Digester 1		Digester 2	
	Feed Sludge	Digested Sludge	Feed Sludge	Digested Sludge
Total N (mg/l)	561	549	1100	933
Nitrogen Lost (mg/l)	-	2.1	-	15
Solid Phase N (mg/l)	477	343	991	653
Solid Phase N (% of SS)	7.0	6.5	6.5	6.5
Solid Phase N (% of VSS)	9.1	8.9	8.5	9.0
Liquid Phase N (mg/l)	84.4	206	110	280
Liquid Phase N (% of Total N)	15	38	10	30
Mineralized Nitrogen* (% of Feed Sludge Total N)	-	39	-	41

* Sum of Liquid Phase Nitrogen and Nitrogen Lost.

TABLE 12

PHOSPHORUS DATA FROM CONTINUOUS FEED DIGESTION - Hooksett, N.H.

Description	Digester 1		Digester 2	
	Feed Sludge	Digested Sludge	Feed Sludge	Digested Sludge
Solid Phase P (mg/l)	107	101	255	202
Solid Phase P (% of SS)	1.6	1.9	1.7	2.0
Solid Phase P (% of VSS)	2.0	2.6	2.2	2.8
Liquid Phase P (mg/l)	53.6	59.7	54.7	90.5
Liquid Phase P (% of Total P)	33	37	18	31

of the total amount of nitrogen added to the digester was lost; the oxygen level was never observed below 2 mg/l.

Experimental results were evaluated to determine how much nitrogen and phosphorus remained in the solid phase and how much were released to the liquid phase. The data in Tables 9 through 12 pertain to the amount of nitrogen and phosphorus in the liquid and solid phases. These data were generated from Tables 4 and 5. Data in the columns labeled "Liquid phase N - % of total N" were computed by dividing the supernatant nitrogen concentration by the total nitrogen concentration of the digested sludge. These computed values tell what percentage of nitrogen in the digester is in the supernatant. Data in columns labeled "N lost - % of Initial Total N in Digester" and "Mineralized Nitrogen - % of Initial Total N in Digester" were calculated by dividing the appropriate concentration of nitrogen lost, or nitrogen mineralized, by the total concentration of nitrogen in the feed sludge. These figures were included because they indicated the potential percentage of nitrogen that could have been in the supernatant phase had no denitrification occurred. In digester 2 (Table 9) 17 percent of the nitrogen initially in the sludge was lost by the eighth day of batch digestion. In digester 1 (Table 9) during batch digestion no significant loss of nitrogen occurred, as is indicated by the negative values generated under "Nitrogen Lost". It is assumed that these negative values were generated by errors involved in sampling procedures and analytical techniques.

During batch aerobic digestion, the percentage of nitrogen in the sludge solids remained fairly constant. In digester 1 the nitrogen content in the solid phase ranged from 7.0 to 7.9 per cent of the SS; in digester 2 the nitrogen content ranged from 6.8 to 8.5 per cent of the SS. The percentage of nitrogen released to the liquid phase was roughly equivalent to the percentage

of VSS destroyed; however, the release of nitrogen appeared to be a linear function of time, while the rate of VSS reduction decreased rapidly with time. This suggests that more organic nitrogen could have been mineralized if the batch digestion had continued beyond 8 days.

Similar results occurred during continuous feed digestion. The nitrogen content of the stabilized sludge averaged 6.5 per cent of the SS concentration. These results generally agree with the findings of Washington and Symons (43) who studied the buildup of inert volatile suspended solids in laboratory activated sludge units operated without sludge wasting. They reported that the inert solids which accumulated averaged 6.3 per cent nitrogen. It is concluded that aerobically stabilized waste activated sludge will contain a considerable amount of nitrogen, indicating that the inert residue contains protein material.

As mentioned previously, after 8 days of batch aerobic digestion the liquid phase nitrogen plus that lost through denitrification comprised about 31 per cent of the total nitrogen in the feed sludge. In the continuous feed digesters soluble nitrogen plus nitrogen lost through denitrification represented about 40 per cent of the nitrogen in the feed sludge. It is reasonable to expect even higher percentages of nitrogen to enter the liquid phase during aerobic digestion of activated sludge if the feed sludge were from a plant with a lower sludge age than that used in this experiment.

In order to correctly interpret the phosphorus data from this experiment it is important to realize the effect of an aerobic storage of activated sludge on sludge phosphorus content The bleedback of orthophosphate from activated sludge solids into solution when the sludge becomes anaerobic is well documented (21, 31, 59, 63). Eikum et al. (31) reported that phosphorus was always released from the solid to liquid phase during anoxic storage of primary sludge at different degrees of stability. Randall et al. (21) concluded

that the release of orthophosphate was highest during the first 90 minutes of storage. It is thought that the high supernatant phosphorus concentrations in the feed sludge used for the continuous flow digesters were not representative of soluble phosphorus levels that would occur in the feed sludge during normal operation of a plant scale aerobic digester. These high soluble phosphorus levels were probably the result of anoxic sludge storage.

With this in mind, the values of interest with respect to phosphorus are the percentages of total sludge phosphorus that are in the liquid phase after digestion. After 8 days of batch aerobic stabilization 32 and 22 per cent of the total sludge phosphorus remained in the supernatant in digesters 1 and 2, respectively. During continuous feed digestion 37 and 31 percent of the total sludge phosphorus was contained in the digested sludge supernatant in digesters 1 and 2, respectively. For all cases, the digester sludge solids contained about 2 percent phosphorus. This figure is similar to the phosphorus content of undigested waste activated sludge. This suggests that the mineralization of phosphorus during aerobic digestion can be predicted stoichiometrically with the first order reaction rate model.

At no time during this investigation was the solid phase phosphorus content greater than 3.0 percent of the SS concentration. The activated sludge used in this study showed no ability for luxury phosphorus uptake as described by proponents of the luxury biological uptake theory (57, 58, 59). These investigators suggest that high dissolved oxygen concentration will encourage luxury uptake. In spite of oxygen levels greater than 5.0 mg/l no luxury uptake of phosphorus occurred during any phase of this study.

Significant amounts of nitrogen and phosphorus were released to the supernatant during aerobic digestion. These findings are different from those reported by many other investigators. Irgens and Halvorson (5), Bruemmer

(29), and Eikum et al. (31) found that essentially no phosphate was present in digested primary sludge supernatant. All three of these investigations studied aerobic stabilization of primary sludge, while this study was concerned with waste activated sludge. With respect to nitrogen, Malina and Burton (3) reported that most of the nitrogen remained associated with the digested primary sludge solids. Irgens and Halvorson (5) found very little nitrogen in the digested sludge supernatant. However, about 14 percent of the sludge nitrogen was lost from the system. This 14 percent represents the nitrogen that would have been in the supernatant if conditions had not been favorable for denitrification.

Results from studies using primary sludge have provided most of the generally accepted knowledge about aerobic digestion. These results have shown that aerobic digestion of primary sludge can produce a supernatant that is generally low in total phosphorus and nitrogen when compared to anaerobic digestion. This is often stated as an advantage of aerobic stabilization over the anaerobic process, without recognizing that significant differences exist between primary sludge and waste activated sludge. Differences in the chemical content of activated and primary sludge can explain the different behavior during aerobic digestion. Primary sludge has relatively low levels of phosphorus and nitrogen compared to activated sludge. Metcalf and Eddy (19) give 0.9 percent phosphorus and 2.5 percent (of total solids) nitrogen as typical values for primary sewage sludge. Representative values for activated sludge are 2.5 percent phosphorus and 7.5 percent nitrogen (64). In activated sludge, nitrogen content is about 10 percent of the VSS. The 7.5 percent figure is arrived at by assuming that activated sludge is 75 percent volatile.

Aerobic digestion of primary sludge involves the breakdown of sludge solids by extracellular bacterial enzymes before degradation can occur.

Because of long detention times the predominant reaction during aerobic digestion of primary sludge is thought to be endogenous respiration (34). In this respect the process is like aerobic digestion of activated sludge. If it is assumed that the "typical" primary sludge mentioned above was aerobically digested with a total solids reduction of 50 percent, and all phosphorus and nitrogen was retained in the solid phase, then the digested sludge solids would contain 1.8 percent phosphorus and 5.0 percent nitrogen. These values are similar to ones reported for aerobically stabilized activated sludge. This suggests that with primary sludge the microorganisms in an aerobic digester will tend to retain most of the nitrogen and phosphorus, and little will be released to the digested sludge supernatant. The concentration of nitrogen and phosphorus in primary sludge is much less than in activated sludge. As observed in this investigation, aerobic digestion of waste activated sludge will lead to the mineralization of phosphorus and nitrogen. This comparison provides an explanation for the contrast between results reported here, using extended aeration activated sludge, and the results reported in the literature review, using primary sludge.

In summary, aerobic digestion of waste activated sludge can produce a supernatant with significant levels of nitrogen and phosphorus. In this investigation as much as 37 percent of the phosphorus and 41 percent of the nitrogen in the feed sludge remained in the supernatant of the digested sludge. The feed sludge used was taken from an extended aeration activated sludge plant so it was undoubtedly in a more stabilized state than sludge from a conventional or high rate activated sludge facility. Based on this observation, it is expected that still higher amounts of supernatant phosphorus and nitrogen, than observed in this study, could be obtained in an activated sludge aerobic digester.

SECTION V

CONCLUSIONS

Small-scale laboratory experiments were performed on primary and waste activated sludges to determine the effects of temperature on the aerobic digestion process. In addition nutrient transformations during the digestion process were evaluated. The digesters were operated using both batch and fill-and-draw modes of operation. Temperatures ranging from 5 to 30°C were used. It was concluded that:

1. Greater reductions in BOD, COD, Total Solids and Total Volatile Solids occur at higher temperatures.
2. Below 10°C, little sludge stabilization occurred.
3. Digestion for periods longer than 15 days produced little improvement in the degree of stabilization, except at temperatures below 10°C.
4. The effect of temperature on the degree of stabilization was more significant at shorter detention times.
5. Reaction rate constants for aerobic digestion varied between 0.21 and 0.36 day^{-1} (base e), based on volatile suspended solids reduction.
6. Higher loading rates improved volatile solids removal in the digesters at all temperatures within the range of loading rates evaluated.
7. Oxygen uptake increased with increasing temperatures.
8. The pH of the reactors loaded at 0.11 $lbVS/ft^3/day$ rose to 8.0 and maintained this pH throughout the testing at all three temperatures, while the pH of the reactors operating at lower loading rates fluctuated between 5 and 7.
9. No correlation between temperature and settleability of the sludge could be established. The settleability of the treated sludge was less than that of the raw feed sludge.

10. Higher temperatures of digestion resulted in improved filterability of the treated sludge. However, filterability of the treated sludge dropped off considerably after 15 days digestion at the higher temperatures.

11. Generally, the filterability of the digested sludge was not as good as the filterability of the raw sludge, even at the higher temperatures.

12. Aerobically digested sludge had no disagreeable odor either during aeration or filtration.

13. Solid phase organic nitrogen will be mineralized during aerobic digestion of waste activated sludge to nitrate, resulting in consumption of alkalinity and a drop in pH.

14. Nitrification still occurred in digesters operating in the pH range of 5.0 to 5.5.

15. Biological denitrification can also occur, but it is essentially absent if the dissolved oxygen concentration exceeds 2.0 mg/l.

16. Aerobic digestion results in the solublization of large quantities of nitrogen and phosphorus, with these nutrients than being present in the supernatant.

17. Phosphorus was released in approximately a stoichiometric ratio to the volatile solids destroyed, producing a stabilized sludge with a phosphorus content of about 2 per cent of the suspended solids.

18. The percentage of organic nitrogen that was mineralized was roughly equal to or slightly greater than the percentage of volatile suspended solids reduction.

19. Aerobically digested waste activated sludge will have a solid phase nitrogen content of about 6 to 7 per cent of the suspended solids.

20. The potential exists for the digested sludge supernatant to contain more than 50 per cent of the nitrogen and phosphorus in a digester treating only activated sludge.

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