AEROBIC DIGESTION OF TRICKLING FILTER HUMUS

by

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I. INTRODUCTION

As a result of the growing effort to instill in the masses a concern for the ecological dangers which confront the earth, terms such as 'water pollution' and 'sewage treatment' can no longer be considered obscure. The activities of the various regulatory agencies, as reported through the news media, have brought about exposure to even more technical terms, including 'suspended solids removal', 'BOD removal', and 'phosphate concentration'. Consequently, there is a growing public interest in the effluent quality of our municipal and industrial sewage treatment plants.

It is the responsibility of the design engineer and treatment plant operator to provide and maintain a facility which will remove the undesirable organic and inorganic material in the sewage and assure an acceptable level of effluent quality. However, maintenance of an acceptable effluent quality is only a part of their total responsibility. They must also provide facilities to render the organic and the inorganic solids removed from the waste flow amenable to disposal in an other than 'pollutional' state.

Although the actual volume of solids in the influent sewage is usually less than 1%, the treatment and disposal of these sewage solids and subsequent biological sludges is the most expensive problem in waste treatment today. Bacon (1) reported that the City of Chicago utilizes some 46% of its annual sewage treatment maintenance

and operating budget under the heading of solids disposal costs.

Two of the major operations in the sequence of solids disposal include the processes of sludge digestion, or stabilization, and sludge dewatering. The respective objectives of these processes are the reduction in volume and decomposition of highly putrescible organic matter to stable or inert organic material and the reduction in volume and moisture content of the sludges. Although both of these processes can be adapted to utilize primary, secondary, or tertiary sludges, as well as any mixture of these three, it is generally conceded that due to its unique chemical, physical, and biological characteristics the disposal of secondary or biological sludge is the most difficult and costly of the domestic sludges.

Traditionally, sludge stabilization has been accomplished in the absence of free oxygen by anaerobic organisms. For the sake of economics and convenience, sludge dewatering has been primarily achieved using gravity-type sand drying beds. However, due to the expansion of available technology, coupled with increases in waste flows, construction and operating costs, as well as more stringent effluent standards, it has become economically and technically feasible to investigate the use of other solids disposal methods. The aerobic digestion process is one of these methods.

Aerobic digestion can be used to stabilize primary or biological sludges or mixtures of the same. However, most applications of this process are in conjunction with activated sludge treatment facilities. It has been shown that the end products of aerobic diges-

tion have excellent dewatering characteristics (29).

The purpose of this investigation was to study the aerobic digestion of trickling filter humus and the relationship between digestion and the subsequent filterability of the sludge. The relative adaptability of this process to the stabilization of a biologically varying slime, such as that sloughed from a trickling filter, was determined through the monitoring of various water quality parameters. Also, a comparison of data collected under similar digestive conditions for waste activated sludge and primary-trickling filter mixtures was undertaken. It is hoped that the results of this work may serve as a base, guide, and possible impetus toward the development of working data for the application of the aerobic digestion process to trickling filter humus.

II. LITERATURE REVIEW

Biological Aspects of Aerobic Digestion

The aerobic stabilization process is similar in nature to the extended aeration variation of the activated sludge process. As available food substrate is depleted, the aerobic microorganisms will begin to consume their own protoplasm to obtain energy for cell maintenance. This then is the so-called phase of 'endogenous respiration'.

Norman (25) reported that most organisms involved in the aerobic stabilization of organic substrate are actually facultative heterotrophs which require an excess of oxygen to exist in an aerobic state.

Hoover (14) first developed the expression of biological cell mass as $C_5H_7NO_2$. Using the expression as a base, Eckenfelder (11) reported the following biochemical reaction as representative of the endogenous oxidation of cell tissue:

$$C_5H_7NO_2 + 5O_2 \longrightarrow 5CO_2 + 2H_2O + NH_3 - \Delta H$$

Other investigators (22) have estimated that between 75 to 80% of the cell tissue can be oxidized. The remaining 20 to 25% is composed of inert materials and non-biodegradable organic compounds.

In practice, the aerobic digestion facilities may be intermittently or continously fed with not only biological sludges, but also solids from the primary clarification units. Under these conditions the biological activity in the digester would be two-fold, encompassing (1) the direct oxidation of the organics in the primary sludge

and (2) endogenous oxidation of the sludge.

Metcalf & Eddy, Inc. (22) characterize the relative advantages of the aerobic process as follows:

(1) volatile solids reductions equal to those of anaerobic digestion,

(2) production of lower supernatant BOD_5 than anaerobic digestion,

(3) production of an odorless, humus-like, biological end product which can be easily disposed of,

(4) production of sludge with excellent dewatering characteristics,

(5) recovery of more of the fertilizer value in the sludge,

(6) fewer operational problems than with anaerobic digestion,

(7) low capital costs.

There are however, major disadvantages associated with the process. The most outstanding of which include:

(1) high power costs, in terms of aeration required,

(2) the useful by-product, CH_4, in not recovered as in anaerobic digestion.

Although the aerobic digestion process would seem to be easily understood in terms of reaction kinetics, in practice the design and operating parameters of this process are poorly understood and have not been investigated nearly as thoroughly as the anaerobic process. To further compound the problems of this investigation, extremely little research has been compiled utilizing aerobic digestion in conjunction with trickling filter humus. Following is a review of

the available literature which will attempt to define the scope of the aerobic digestion process. Also presented will be the results of other research which may be pertinent, in nature, to the methods used and results obtained in this investigation.

Aerobic Digestion Research and Practice

Although anaerobic digestion followed by sand drying beds is probably the most common method of sludge handling at most sewage treatment plants (29), aerobic digestion has received increased attention over the years.

As early as 1932, Rudolphs and Heukelekian (32) reported on the destruction of concentrated primary sewage sludges by both aerobic and anaerobic means. Their investigation showed that under properly seeded conditions the rates of decomposition of volatile matter, proteins, fats, and biochemical oxygen demand (BOD_5) were strikingly similar for both digestion processes. However, for unseeded material the rate of destruction was found to be much greater under aerobic conditions. After a 35 day aeration period, reductions in volatile solids, fats, and total nitrogen were found to be 50%, 99%, and 30% respectively.

In 1955, Coackley (7) reported the results of an investigation to determine the extent of aerobic stabilization possible for sludges previously subjected to anaerobic digestion. The sludges were aerated for periods of up to 70 days and at temperatures ranging from 18° to 37°C. An insignificant decrease in volatile solids

reduction was reported for the 18° C trial. However, the sludge that was aerobically digested at 37°C showed a significant volatile solids reduction (63% at 47 days) beyond that obtained during anaerobic digestion. The stability of the aerobically digested sludge was evidenced by the fact that anaerobic digestion could not be initiated in the aerobically digested sludge, even after inoculation with anaerobic organisms.

Kehr (17) initiated an investigation to determine the effectiveness of the aerobic stabilization process as an alternative to anaerobic digestion. Results of this work indicated that as much as a 77% reduction in total dry matter could be obtained for a biological sludge subjected to aerobic digestion. Primary sludge receiving aerobic stabilization exhibited a maximum of 60 to 70% reduction in organic matter. Based on this study, Kehr concluded that significant bacterial sludge stabilization could occur within a 3 to 5 day period under batch-type aerobic conditions. It was further recommended that aerobic digestion be considered economically feasible for wastewater facilities serving a population of 5,000 to 10,000.

Levis, et. al., (19) reported the choice of the aerobic digestion process over a comparable anaerobic system for the stabilization of waste activated sludge from a 1.0 mgd contact stabilization plant in Millersville, Pennsylvania. As well as being the more economical system, it was felt the aerobic digester could much more easily withstand the variation in solids loading due to the large portion of college students included in the town's population.

The aerobic stabilization of raw, or primary, sludges has been studied by several investigators. Malina and Burton (20) reported that primary sludges from domestic wastewaters can be effectively stabilized without the addition of seed material. At a loading rate of 0.14 pounds volatile solids per day per cubic foot, a 43% reduction in volatile solids was noted. However, for the same aeration period, only a 33% reduction was observed for a loading rate of 0.10 pounds volatile solids per day per cubic foot. The digested supernatant was found to be low in BOD_5 and ammonia nitrogen. A 75% reduction in the supernatant chemical oxygen demand (COD) was also observed. The digested sludge was found to have no disagreeable odor and an oxidation-reduction potential measurement of greater than 250 millivolts further suggested that the digested sludge was well stabilized.

Viraraghavan (37) concluded that, based on a study of primary wastewater sludges in India, a fairly high degree of sludge digestion can be obtained under aerobic conditions. Some other observations made during this study included:

(1) A significant reduction in BOD_5 occurred during digestion.

(2) No disagreeable odors accompanied the aerobically digested sludge.

(3) Nitrification of the sludge occurred during digestion.

Cook (9) reported on the aerobic stabilization of primary wastewater sludge at the Stillwater, Oklahoma municipal treatment

plant. Although the plant included a trickling filter process, only primary sludge was fed into the 180 gallon capacity pilot digesters. Based on trial digestion periods of 2, 4, 8, and 12 days, it was decided that the 4 day aeration period out-performed all the others in terms of producing the lowest values of BOD_5, COD, ammonia and phosphates. Further research at the Stillwater facility will determine if extension of the stabilization period to either 12, 18, 24, or 30 days may produce results that are more optimum than those obtained at 4 days.

Much of the research associated with the aerobic digestion of biological sludges has been in conjunction with the stabilization of waste activated sludges. Tebbutt (34) conducted a bench-scale digestion study of activated sludge at a solids concentration approximately equal to 2000 mg/l. From the results of this work, he concluded that it is possible to achieve a 50% reduction in volatile suspended solids in 8 to 16 days and that an increase in sludge concentration seemed to hamper stabilization at normal aeration rates (i.e. 0.5 to 0.75 liters of air per cubic meter per second).

At the request of some of his colleagues, Tebbutt (35) expanded his research of the digestion of waste activated sludge to include sludges of solids concentrations up to 12,000 mg/l. He also included the aerobic stabilization of primary sludge in this second investigation. Based on the observations made, Tebbutt concluded:

(1) Aeration rates sufficient to keep the sludge solids

in suspension will produce significant reductions in volatile solids concentrations.

(2) Primary sludges respond to aerobic stabilization in much the same manner as do activated sludges.

(3) In general, the dewatering characteristics of the sludges tend to deteriorate following aeration.

Norman (25) reported that for the aerobic digestion of a mixture of primary and waste activated sludge (1.75:1.0 ratio on a dry solids base) significant solids reductions were observed. At an initial mixed liquor total solids concentration of 32,000 mg/l, volatile solids reductions of 35% and 53% were noted for aeration periods of 15 and 30 days, respectively. Norman concluded that, in general an increase of temperature, solids loading and detention time produced an increase in solids reduction.

Lawton and Norman (18) studied the aerobic stabilization of waste activated sludge from a treatment plant receiving 67% domestic sewage and 33% pretreated meat packing waste, by volume. They concluded that aerobic digestion can produce sufficient volatile solids reduction (39 to 53% in 30 days at 20°C) to render a humus-like, biologically stable end-product. The BOD_5 of the digested sludge and supernatant were reportedly lower than those usually obtained during anaerobic digestion. It was concluded that volatile solids reduction showed a strong correlation with sludge age.

Jaworski, Lawton, and Rohlich (16) also compiled data on the aerobic digestion of waste activated sludge. They reported that

historically most aerobic digestion work was done at mixed liquor suspended solids (MLSS) concentrations of less than 1.0 percent. In an effort to review the literature, they cited the work of Akers. Upon investigating the factors which may influence the auto-oxidation rate (i.e. endogenous respiration) of biological sludges, Akers concluded that the rate of auto-oxidation is considerably less than the rate of change in sludge age (i.e. total pounds volatile suspended solids in the aeration system divided by the daily input in pounds of volatile suspended solids per day to the system). During this investigation, Jaworski, Lawton, and Rohlich reported that a mixture of primary and waste activated sludge (MLSS concentration of 3.0 percent) produced volatile solids reductions between 28 and 44% when aerated for 6 days at temperatures ranging from 15° to 25°C. Similar digestive conditions showed only an 11% decrease in volatile solids when the initial MLSS concentration was 0.75 percent. In general, the conclusions of their studies indicated:

(1) Reduction of volatile solids is a function of detention time, up to 15 days.

(2) The higher the temperature the greater the solids reduction. However, for long range digestion periods (e.g. 60 days), temperature appears to have little effect, as digestion is essentially complete at all temperatures.

(3) Higher volatile solids reductions were observed for

lower loading rates.

(4) Low supernatant BOD_5 concentrations (not greater than 100 mg/l after 5 days) and high nitrate concentrations (600 mg/l as NO_3-N at 60 days) can be expected upon prolonged aeration.

(5) Settling characteristics of digested sludges are poorer than those for undigested sludges.

(6) An odor-free, biologically stable end-product was produced.

Ritter (30) reported on the operation of aerobic digesters at three contact stabilization plants in Pennsylvania. Both primary and waste activated sludges were digested at all plants. Volatile solids reductions of 50 to 65% were noted. Some conclusions as to the effectiveness of and comments concerning the operational procedures for the three systems include:

(1) Aerobic digestion produces substantial solids reductions, resulting in a stable, readily dewatered sludge.

(2) Concentration of sludge prior to feeding increases residence time and results in an increase in volatile solids reduction.

(3) Operation of a continous feed system seems more optimal than a batch type operation.

(4) Cessation of aeration to facilitate the manual supernatant decanting contributes to higher effluent phosphate concentration. The decrease in

ORP (oxidation-reduction potential) and subsequent release of absorbed phosphate by sludge cells can be minimized through installation of an automatic decanting system.

(5) The design parameters of 0.08 - 0.14 cubic meters per capita and 0.02 cubic meters of air per minute per cubic meter digester capacity seem to be adequate.

(6) Operating costs of the three systems average less than $0.37 per capita per year.

As activated sludge and trickling filter units function biologically to achieve the common objective of oxidizing the organic matter in the sewage, it would seem logical to assume that the biological sludges from both processes would respond similarly to aerobic digestion. Drier (10) in fact, reported this to be the case. He cited the use of aerobic digestion at the Rockford, Illinois sewage treatment plant. At this facility, a mixture of primary and high-rate trickling filter sludges are subjected to aerobic stabilization. A 70 foot diameter by 15 foot deep digester is in use. The average loading rate to the digester is 0.10 pounds total suspended solids per cubic foot per day. After 30 days digestion, the sludge is pumped to a storage lagoon in an odor-free condition. All concerned have been pleased with the operation of the aerobic digester. As anaerobic digesters are in existence at the Rockford facility, the use of the aerobic digester as a final conditioning tank for the sludge from the anaerobic digesters, prior

to discharge to the lagoon is being contemplated.

Drier (10) also provided some design data for the aerobic digestion of trickling filter sludges. If both primary and biological sludges are to be stabilized, a design loading of 0.19 pounds total suspended solids per capita per day should be used. If only trickling filter sludge is to be stabilized, loadings of 0.06 to 0.07 pounds total suspended solids per capita per day would suffice. Metcalf and Eddy, Inc. (22) reported the required residence time for optimum solids reduction of trickling filter and primary sludge mixtures to be 18 to 22 days.

Most of the available literature on aerobic digestion is concerned with reviewing the results and effectiveness of the process. There are, however, instances where investigators have reported on the effect and interactions of specific operational parameters as they pertain to the effectiveness of the entire process. Barnhart (2) reported that, for any sludge, there is a definite limit of volatile solids reduction which can be obtained. Norman (25) suggested that this limit is a function of the relative quantities of cell material and biologically inert organic material present.

Several investigators (16, 18, 20, 25) have reported that volatile solids reduction is a function of the solids loading rate. However, there exists a disagreement among these investigators as to the exact nature of this relationship.

Based on aerobic digestion studies of waste activated sludge from a news and chip boardmill operation, Carpenter (5) reported

that the rate of volatile solids reduction is influenced by temperature in accordance with Arrhenius' principle. Other researchers (7, 15, 18, 20, 25) have also noted a direct relationship between temperature and solids reduction.

The oxygen requirements of the aerobic digestion process have been established by Eckenfelder (11). He reported these requirements as:

(1) 1.42 grams of oxygen per 1.0 gram of cell mass oxidized,

(2) 1.7 to 1.9 grams of oxygen per 1.0 gram of primary sludge BOD_5 oxidized,

(3) A minimum content of 2.0 mg/l dissolved oxygen in the aerobic system at all times.

Tebbutt (35) found the amount of aeration to be independent of solids reduction, given that sufficient aeration to maintain the sludge solids in suspension was available.

Metcalf and Eddy, Inc. (22) reported that the ultimate drop in pH during aerobic digestion is a function of the increased presence of nitrate ions in solution and the lowering of the buffering capacity of the system due to air stripping. Norman (25) found that the pH level may drop as low as 5.0 during aerobic digestion. He reported that solids reduction did not seem to be inhibited at this low pH. Randall and Koch (27) also reported drastic drops in pH to values as low as 4.5 during extensive aeration. However, they concluded that based on large increases

of orthophosphate and ammonia nitrogen in the supernatant, some toxic pH inhibition of microorganisms must have occurred. Randall, Saunders, and King (28) observed an apparent case of solids reduction inhibition, due to pH toxicity, during the aerobic digestion of waste activated sludge.

Moore (23) studied the effect of pH on the aerobic digestion of waste activated sludge. Based on the results of this investigation, he concluded:

(1) The aerobic digestion process is relatively insensitive to pH level, as significant solids reductions can be obtained over a pH range of 3.5 to 9.5.

(2) Improvement in the flocculation characteristics of the sludge may occur with digestion at low pH. Such reductions correspond to the development of very large predator populations, which ultimately enhance the settleability, filterability and drainability of the digested sludge.

Irgens and Halvorson (15) reported that during aerobic digestion studies of mixtures of primary and waste activated sludges, a near nutrient free supernatant was obtained. Supernatant BOD_5 and COD reductions ranging from 83% to 87% and 45% to 69% respectively, were noted. Examination of the sludge mixtures prior to digestion showed coliform counts averaging in excess of 8×10^6 per ml. The digested sludges were reexamined. Mixed liquor coliform counts varied from 0 to 100 per ml, while in no case were coliform organisms

observed in the digested supernatant. These observations suggest that the environment of the aerobic digester may also be unsuitable for the survival of pathogenic organisms.

Bruemmer (4) reported on supernatant nutrient data for both anaerobic and aerobic digestion. Typical ammonia nitrogen and orthophosphate concentrations of 413 mg/l and 213 mg/l respectively, are cited for the anaerobic process. The aerobic supernatant, based on the work of Irgens and Halvorson, was reported to contain less than 1.0 mg/l ammonia nitrogen and less than 10 mg/l of both orthophosphate and total nitrogen.

From the results of prolonged aerobic stabilization of activated sludge, Drier (10) suggested that a maximum supernatant nitrite-nitrogen (NO_2-N) concentration of 2.5 mg/l will be obtained at about 15 days aeration. Some 200 mg/l nitrate-nitrogen (NO_3-N) could be expected at 30 days. This data agrees well with the degree of nitrification observed by other investigators (18, 24).

Randall and Koch (27) also observed the build-up of orthophosphates, nitrates, sulfates, and chlorides in the digested supernatant.

Filterability of Aerobically Digested Sludge

A review of the literature indicates that the aerobic digestion process can greatly reduce sludge volume and produce a stable, humus-like end-product. The question then arises as to the filterability and dewaterability of the aerobically digested sludge.

Parker, Randall, and King (26) reported that the filtration char-

acteristics of an aerobically stabilized, biological sludge are dependent upon the degree of flocculation of that sludge. They further concluded that the mechanism of filtration improvement, which was observed during their investigation of waste activated sludge digestion, was biologically induced flocculation. When conditions for biological activity were good and the food to microorganisms (F:M) ratio of the system was low enough to promote endogenous respiration, sludge filterability improved or remained good. They also reported the filterability could be adversely affected by anaerobic storage, excessive mixing, chlorination and rapid changes in temperature.

In 1956, Coackley and Jones (8) developed a procedure for determining a parameter by which the relative filterabilities of various sludges could be compared. The data collected during this vacuum filtration procedure is used to compute the specific resistance of the sludge to filtration.

The equation for the calculation of the specific resistance (r) is as follows:

$$r = \frac{2PA^2n}{\mu C}$$

where: r = specific resistance (cm/g)

P = applied filter pressure (g (force)/cm^2)

A = filter area (cm^2)

μ = filtrate viscosity (poises)

C = total suspended solids concentration (g/ml)

n = slope of the plot of time/filtrate volume vs. filtrate volume (sec/ml^2)

Tebbutt (36) reported that the Water Pollution Research Laboratory of England currently expresses the units of specific resistance in terms of meters per kilogram (m/Kg) rather than centimeters per gram (cm/g). This conversion is readily made by multiplying the specific resistance in cm/g by 10 to obtain m/Kg.

Murphy (24) suggested that the relative success of any filtering operation depends upon:

(1) the quantity of sludge obtained per unit area,

(2) the degree of dewatering,

(3) the efficiency of solids-liquid separation,

(4) the cost of the operation.

Based on the results of a study measuring the effects of aeration at 15°C on the settleability and filterability of sewage sludges, Murphy concluded that a wide range of settleability and filterability rates exist for sludges of approximately the same concentration. Without chemical addition, the filterability of waste activated sludge was observed to increase up to an aeration period of 72 hours and them decrease through the end of the aeration period (144 hours). Murphy also reported that vigorous aeration reduced the filterability and settleability of the sludges; probably due to a shearing of the natural biological floc.

Carpenter (5) and Tebbutt (34) both reported a general degradation of the filtration characteristics of waste activated sludge during aerobic digestion. Rivera-Cordero (31) and Parker, Randall, and King (26) observed the specific resistance of activated sludge to

reach a minimum value after 1 to 5 days aeration. The specific resistance increases thereafter to values well over that of the undigested sludge.

Genter (12) has shown that a reduction in volatile solids will result in a savings in the amount of coagulant required to obtain a specific filter yield. Although anaerobic digestion can be used to reduce solids concentrations, the build-up of alkalinity in the anaerobic system increases the required coagulant dose to the extent where the effect of solids reductions is all but negated. Several investigators (22, 24, 25, 31) have shown that alkalinity is reduced quite rapidly and completely during aerobic digestion.

III. METHODS AND MATERIALS

In an effort to obtain a conclusive amount of data concerning the aerobic digestion of trickling filter humus, five individual batch-fed digestion runs were completed. In each case, ten liters of sludge was subjected to aeration. The composition of the sludges used and length of digestion runs are shown in Table I.

Digestion of the trickling filter sludges (Runs #1, 2, and 3) were of primary concern, in terms of obtaining conclusive data for this investigation. The mixture of primary and trickling filter sludges (Run #5) was designed to obtain digestion data under simulated digester conditions for a trickling filter treatment plant. Digestion of waste activated sludge (Run #4) was conducted to serve as a standard for comparison, as well as a source of correlation between this investigation and most of the published literature concerning aerobic digestion.

The digestion process was conducted within a constant temperature environment, where the experimental apparatus and sludge were maintained at a temperature of 20°C (± 1°C). No attempt was made to specifically control any other digestion parameters. However, additions of tap water were made as required to compensate for losses in digester volume due to evaporation. The only restrictions placed upon the air supply were (1) a minimum of 2.0 mg/l dissolved oxygen and (2) solids suspension be maintained in the digester at all times.

TABLE I

COMPOSITION AND LENGTH OF DIGESTION RUNS

Run No.	Nature of Sludge Digested	Dates	Length of Digestion Run
1	Trickling Filter	6/29-7/21	22 days
2	Trickling Filter	8/18-9/23	36 days
3	Trickling Filter	9/25-10/31	36 days
4	Waste Activated	7/23-8/14	22 days
5	70% Primary- 30% Trickling Filter*	8/02-9/09	38 days

* by volume

No chemical additions to adjust pH or improve sludge filterability were made.

Experimental Apparatus

Two separate aeration chambers were used during this investigation. Both were plexiglass cylinders, eight inches in diameter and eighteen inches high. Although the capacity of each digester was 13.5 liters, only 10 liters of sludge was digested in order to allow some freeboard. The chamber used in digestion runs #1, 2, 3, and 4 was equipped with a bottom sampling spout. The chamber used for run #5 had no sampling spout. Samples were obtained from this second digester through the modification and use of an ordinary kitchen-type ladle.

Air was dispersed into the digestion unit through a perforated ring of copper tubing positioned approximately one inch above the bottom of the aeration chamber. The addition of a clamp to the upright portion of the air supply tubing allowed for variation in the positioned depth of the aeration ring. This flexibility was an important element in insuring the proper mixing of the digester contents. The rubber tubing air supply line was equipped with a cotton filter to minimize contamination of the digester by oil or any other foreign matter which might come through the air supply. A sketch of the digestion unit is shown in Figure 1.

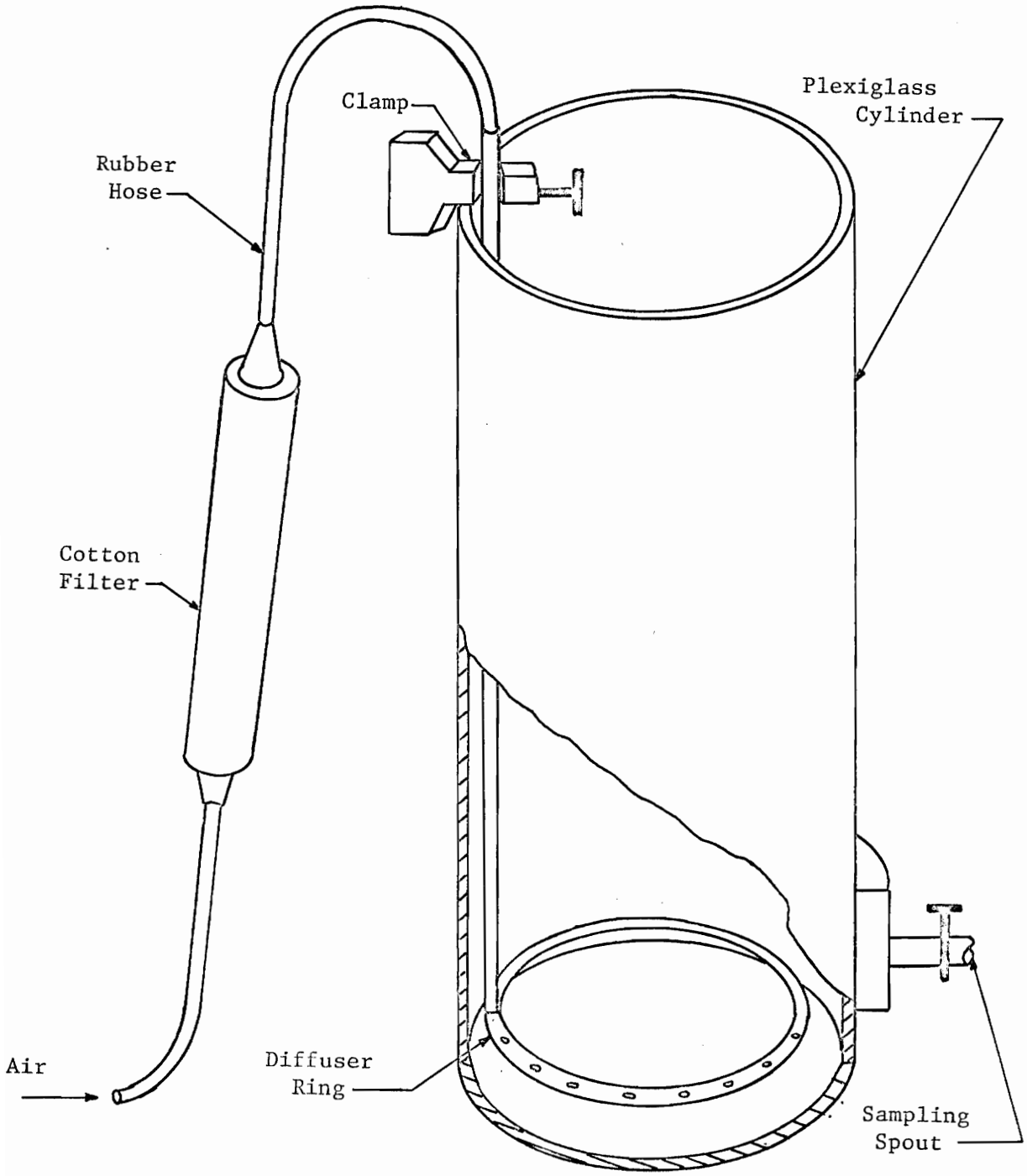

Figure 1: AEROBIC DIGESTION APPARATUS

Sludge Procurement and Handling

All trickling filter and primary sludges used in this investigation were procured from the Blacksburg - VPI Sanitation Authority Sewage Treatment Plant near Blacksburg, Virginia. This facility is a secondary treatment plant utilizing a high-rate trickling filter as its biological reactor. Most of the waste received by this plant is domestic sewage from the Town of Blacksburg and the VPI & SU campus. Average flow to this plant at the time of procurement of sludges for digestion runs #1, 2, and 5 was approximately 1.5 million gallons per day (mgd). The sludge for digestion run #3 was obtained about one week after the annual fall influx of students to the VPI & SU campus. Average flow at this time approached 2.5 mgd. The trickling filter and primary sludges were pumped directly from the secondary and primary clarifiers, respectively.

The waste activated sludge digested in run #4 was obtained from a Virginia Department of Highways sewage treatment plant located near Radford, Virginia on Interstate Route 81. This installation is a small extended aeration plant treating less than 10,000 gallons per day. Sludge was procured at the outlet of the return sludge line.

All sludges were allowed to settle and supernatant was decanted to increase solids concentrations. In order to provide a means of correlation with the results of other investigators (23, 28, 31), it was desired to achieve initial solids concentrations for the biological sludges of between 1.5 and 2.0 percent. The activated sludge was observed to float upon prolonged settling. Indications were that

the sludge may have already gone through some degree of nitrification-denitrification. Attempts to obtain a sludge that would settle from the plant in question proved futile. It was then decided to concentrate the floating sludge and subject it to aeration. After concentration, each of the sludges were thoroughly mixed prior to aeration. In no case did the elapsed time between the procurement of the sludge and beginning of aeration exceed 8 hours. The initial solids concentrations and percent volatility for each digestion run is shown in Table II.

A relatively low percentage of volatility was noted for the trickling filter sludge in digestion run #1. An explanation for this low value may be related to the fact that this sludge was obtained approximately one week after tropical storm Agnes had 'washed-through' the treatment plant. It is theorized that considerable amounts of inert material were carried into the system and that this period of time was not sufficient for bacterial cell synthesis to restore the normal balance of volatile and inert matter.

Sampling Procedures

On the days of sludge analysis, aeration was curtailed, the sides of the digester scraped, and water added to compensate for any evaporation losses. Aeration was then restarted to thoroughly mix the digester contents and two mixed liquor samples were collected. The first sample was of sufficient volume to perform the various mixed liquor chemical analyses. The second sample was of sufficient

TABLE II

INITIAL SOLIDS CONCENTRATIONS AND VOLATILITY

Run No.	Type of Sludge Digested	Total Suspended Solids (mg/l)	Percent Volatility
1	Trickling Filter	15,591	54.7
2	Trickling Filter	19,179	71.9
3	Trickling Filter	20,411	76.1
4	Waste Activated	16,450	75.4
5	Primary-Trickling Filter	35,695	64.6

size so that, after settling, it would produce an adequate amount of supernatant for chemical analyses. All unused sludge and supernatant samples, as well as those samples used in the analyses of pH, filterability, and oxygen uptake, were returned to the digester. Losses of sludge for the analyses of solids and BOD_5 were found to be minor. Supernatant samples for the determination of COD and total carbon content of digestion run #3 were sealed in a closed vessel and refrigerated until these tests could be performed.

A tentative sampling and testing schedule for sludge analysis was established. The various analytical tests were performed each day during the first five days of aeration. After that time, analyses were conducted every second day through fifteen days. From fifteen days through the completion of the digestion period, a random testing schedule was employed. However, in no case was the period between sludge analyses allowed to exceed 6 days. The average ratio of the number of 'test days' to the length of the aeration period for the five digestion runs was 0.50.

Analytical Methods

The following methods were used to analyze the mixed liquor and supernatant samples from the aerobic digesters:

1. Suspended Solids:

Total and volatile suspended solids concentrations were determined for all digestion runs. These determinations were made in accordance with the Gooch crucible method as described in Standard

Methods for the Examination of Water and Wastewater (33). However, the use of a glass fiber filter (Reeves-Angel, 2.1 cm) was substituted for the preparation and use of an asbestos mat filter. Three sludge samples of approximately 5 ml were filtered to insure the validity of the results obtained. An average of the results from these three samples was considered to be the actual solids concentration. In general, the variation in the results of the three samples was found to be mimimal throughout this investigation. However, when one sample was found to differ considerably from the others, it was rejected and the average value was calculated from the remaining data.

2. pH:

Both mixed liquor and supernatant pH values were determined for all digestion runs. A Leeds and Northrup line-operated laboratory model pH meter was used to obtain these measurements. Prior to each determination, the pH meter was calibrated with a standard buffer solution.

3. BOD_5:

A five-day biochemical oxygen demand analysis was performed on both the mixed liquor and supernatant for all digestion runs. Initially, sample dilutions of 1:1200, 1:600, 1:375 were used in determining the mixed liquor BOD_5. However, evaluation of the results obtained indicated that the 1:600 dilution provided the most reliable and reproducible data. A 1:300 dilution was used to prepare the supernatant samples. Supernatant BOD_5 values were obtained using settled supernatant. However, in an effort to reduce the degree of

fluctuation in the observed BOD_5 values, the supernatant for digestion run #3 was filtered through an 9 micron Millipore filter prior to dilution. The BOD_5 analysis was performed according to the procedures outlined in Standard Methods (33). Dissolved oxygen determinations for this analysis were made using a Yellow Spring Instrument Company Model 54 Dissolved Oxygen Meter equipped with a Clark type membrane covered polargraphic probe.

4. Alkalinity:

The total supernatant alkalinities for digestion runs #1, 2, 4, and 5 were determined. The procedure outlined in Standard Methods (33) was used for alkalinity determinations. However, sample sized of 25 to 50 ml were used in lieu of the recommended 100 ml.

5. Oxygen Uptake:

The oxygen uptake rates were measured for the sludges in digestion runs #2 through 5. A standard BOD bottle was filled with sludge and fitted with the dissolved oxygen metering equipment previously described. After the oxygen meter was allowed to stabilize an initial dissolved oxygen content was noted. The oxygen content of the sludge was then periodically noted, with the frequency of observation being a function of the relative rate of oxygen utilization. The slope of the best straight-line fit for the plot of oxygen utilization verses time was considered to be the oxygen uptake rate for that particular sample.

6. Filterability:

The filterability of the digested sludges was evaluated using

the concept of specific resistance developed by Coackley and Jones (8) and previously cited in this manuscript.

The filtration apparatus consisted of a 9.0 cm diameter Buchner funnel fitted with an 8.0 cm inside diameter plastic ring seating one piece of #40 ashless, Whatman filter paper. The stem of the funnel extended through a rubber stopper which firmly lodged into the top of a 100 ml plastic graduated cylinder. Through this stopper was also placed a piece of metal tubing which connected to a length of rubber tubing leading to a mercury manometer and a vacuum pump. A pinch clamp was used to seal the vacuum system ahead of the funnel apparatus. Manipulation of an adjustable hose clamp was used to produce the desired filtration pressure differential, as observed from the mercury manometer. The apparatus thus described is shown in Figure 2.

The filtration test was conducted in the following manner:

(1) The vacuum pressure was adjusted to the desired level. A 12.5 inch mercury differential was used during this investigation.

(2) A 25 ml volume of distilled water was filtered through the apparatus to insure proper seating of the filter paper.

(3) The vacuum was again checked.

(4) A 100 ml sample of sludge was placed in the funnel.

(5) The vacuum pressure was applied by slowly opening the pinch clamp.

(6) The volume of filtrate obtained at specific time

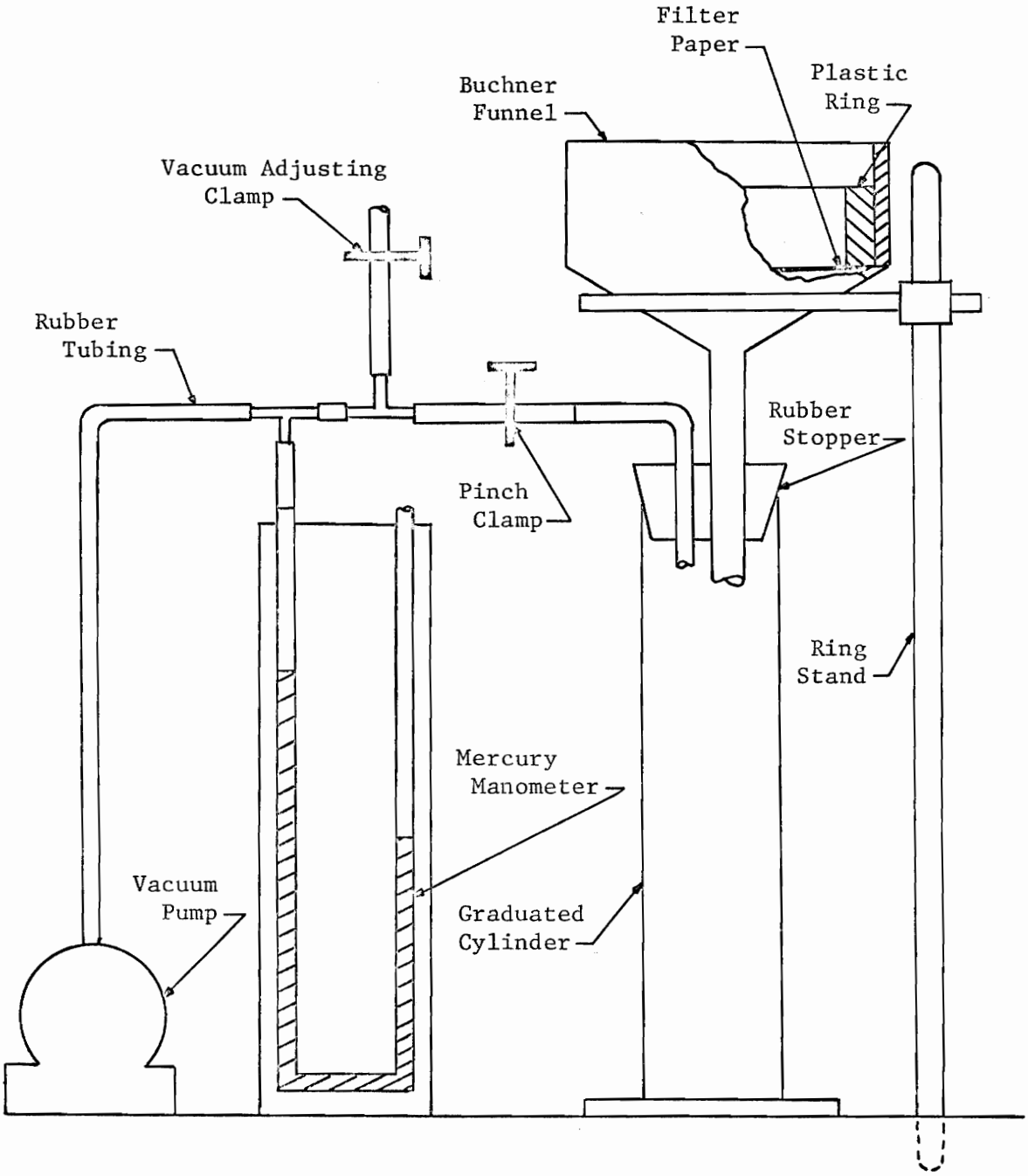

Figure 2: BUCHNER FUNNEL FILTRATION APPARATUS

intervals was noted. The time intervals generally used throughout this investigation were: 1, 2, 3, 4, 5, 6, 7, 8, 10, 12, 15, 18, 20, 22, 25, 28, and 30 minutes.

Substituting the data obtained into the Coackley and Jones equation, the specific resistance of the sludge to filtration was calculated. A review of this equation shows that:

$$r = \frac{2PA^2n}{\mu C}$$

where: r = specific resistance, cm/g x 10 = m/Kg

P = applied filter pressure, g(force)/cm^2

A = filter area, cm^2

μ = viscosity of filtrate, poises

C = suspended solids concentration, g/ml

n = slope of the plot T/V vs. V, sec/ml^2

During this investigation, the viscosity of the filtrate was assumed to be equal to that of water at the same temperature (20°C). The slope of the plot of T/V (time/filtrate volume) vs. V (filtrate volume) was determined from the best straight-line fit superimposed on the plot.

7. Chemical Oxygen Demand:

The COD of the supernatant from digestion run #3 was determined. The supernatant sample used in this analysis was filtered through an 8 micron Millipore filter. The COD determination was performed according to the procedures outlined in Standard Methods (33). A sample size of 20 ml was used.

8. Nitrite and Nitrate Nitrogen:

The nitrite and nitrate nitrogen supernatant concentrations were determined for digestion run #3. The supernatant was filtered as previously described for the COD analysis. These determinations were made using the Hach Chemical Company DR-EL Engineer's Laboratory field testing kit. As only relative determinations of nitrite and nitrate nitrogen concentrations were desired, it was felt that the results obtained using this equipment would be sufficiently accurate.

9. Total Carbon:

The total Carbon content of the supernatant from digestion run #3 was determined through the use of the total carbon channel of a Beckman Model 915 Total Organic Carbon Analyzer. The sample was filtered, as previously described for the COD analysis, and diluted to one-tenth strength with demineralized water prior to injection. A malfunction with the inorganic carbon channel of this same unit made it impossible to complete the determination of the total organic carbon content (TOC) of the supernatant.

IV. EXPERIMENTAL RESULTS

The experimental results obtained during the course of this investigation are presented and briefly discussed in this chapter. The data is presented in both graphical and tabular form. Each of the digestion parameters studied is discussed individually in regard to observations made during the various digestion runs.

Suspended Solids

The total and volatile suspended solids variations observed during the digestion of the trickling filter sludges are shown in Figures 3 and 4, respectively. The rate of solids destruction was noted to be approximately the same for trickling filter sludges 2 and 3 (i.e. trickling filter digestion runs #2 and 3). Trickling filter sludge 1 also exhibited a somewhat similar rate of solids destruction through approximately 10 days of digestion, after which a short period of solids build-up occurred followed by the rapid disappearance of virtually all volatile suspended solids. The percentage of solids reductions for the trickling filter sludges are shown in Figures 5 through 7. Total solids reductions in excess of 40% and 35% were noted at 30 days aeration for digestion runs #2 and 3, respectively. Digestion run #1 produced a 67% reduction in total solids through 22 days of digestion. Corresponding volatile suspended solids reductions for these sludges were observed to approach 50% and 40% for sludges 2

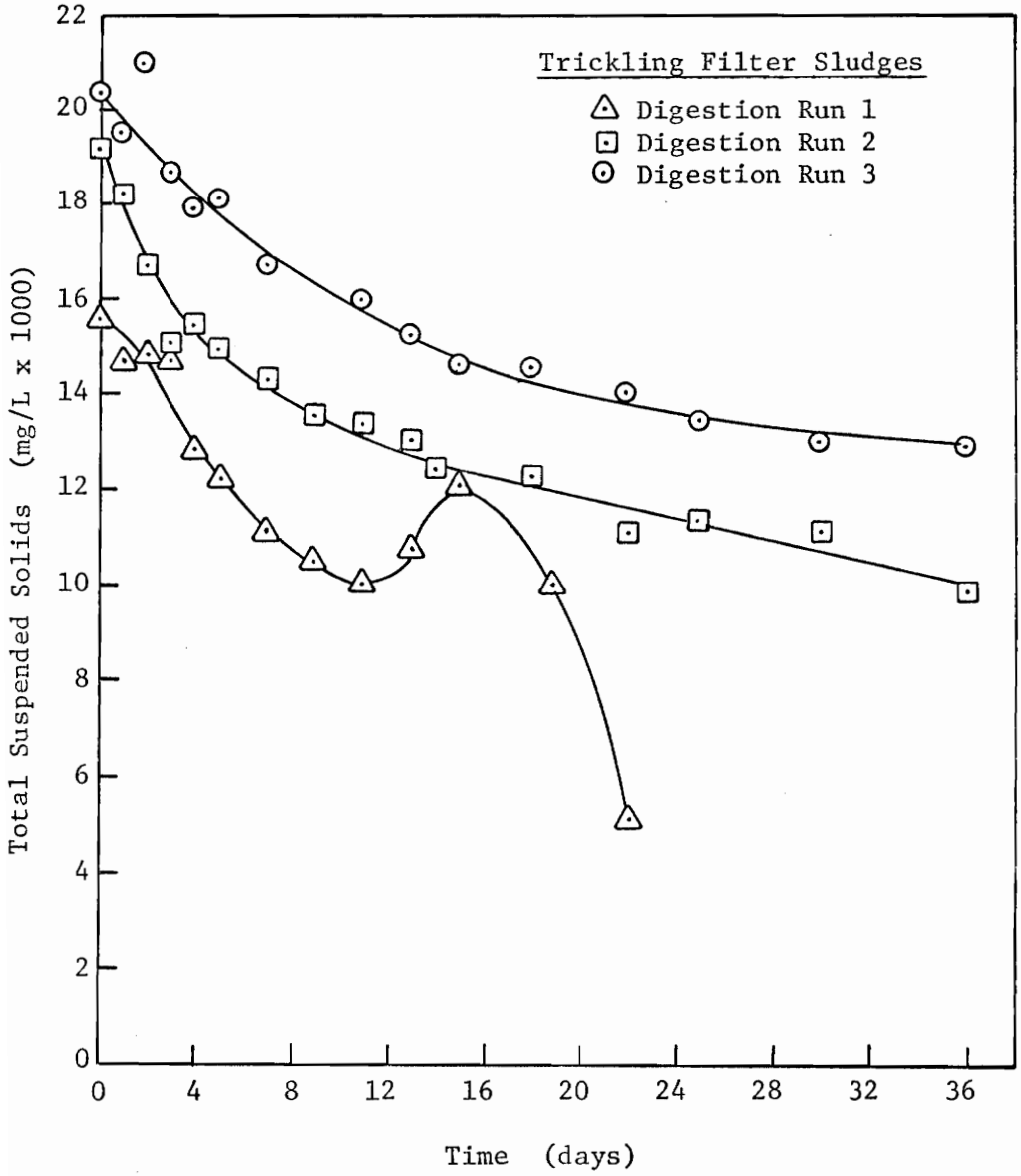

Figure 3: TOTAL SUSPENDED SOLIDS VARIATION FOR TRICKLING FILTER SLUDGES

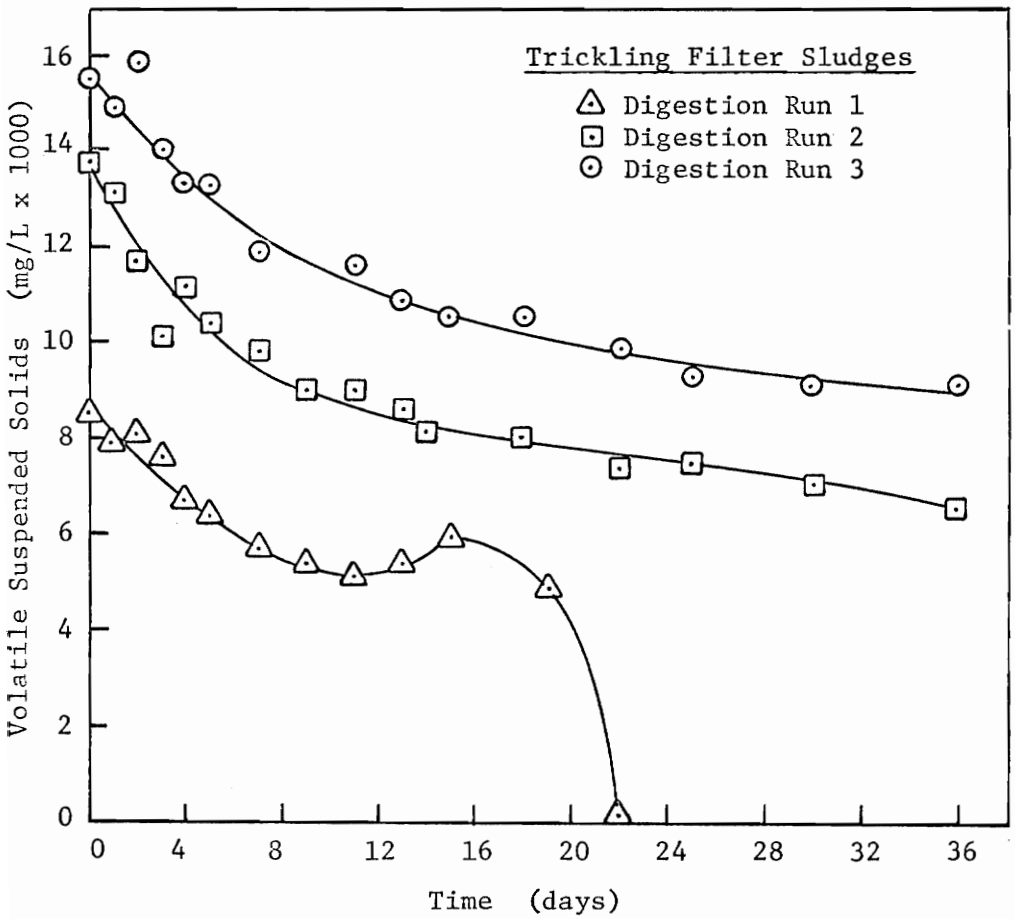

Figure 4: VOLATILE SUSPENDED SOLIDS VARIATION FOR TRICKLING FILTER SLUDGES

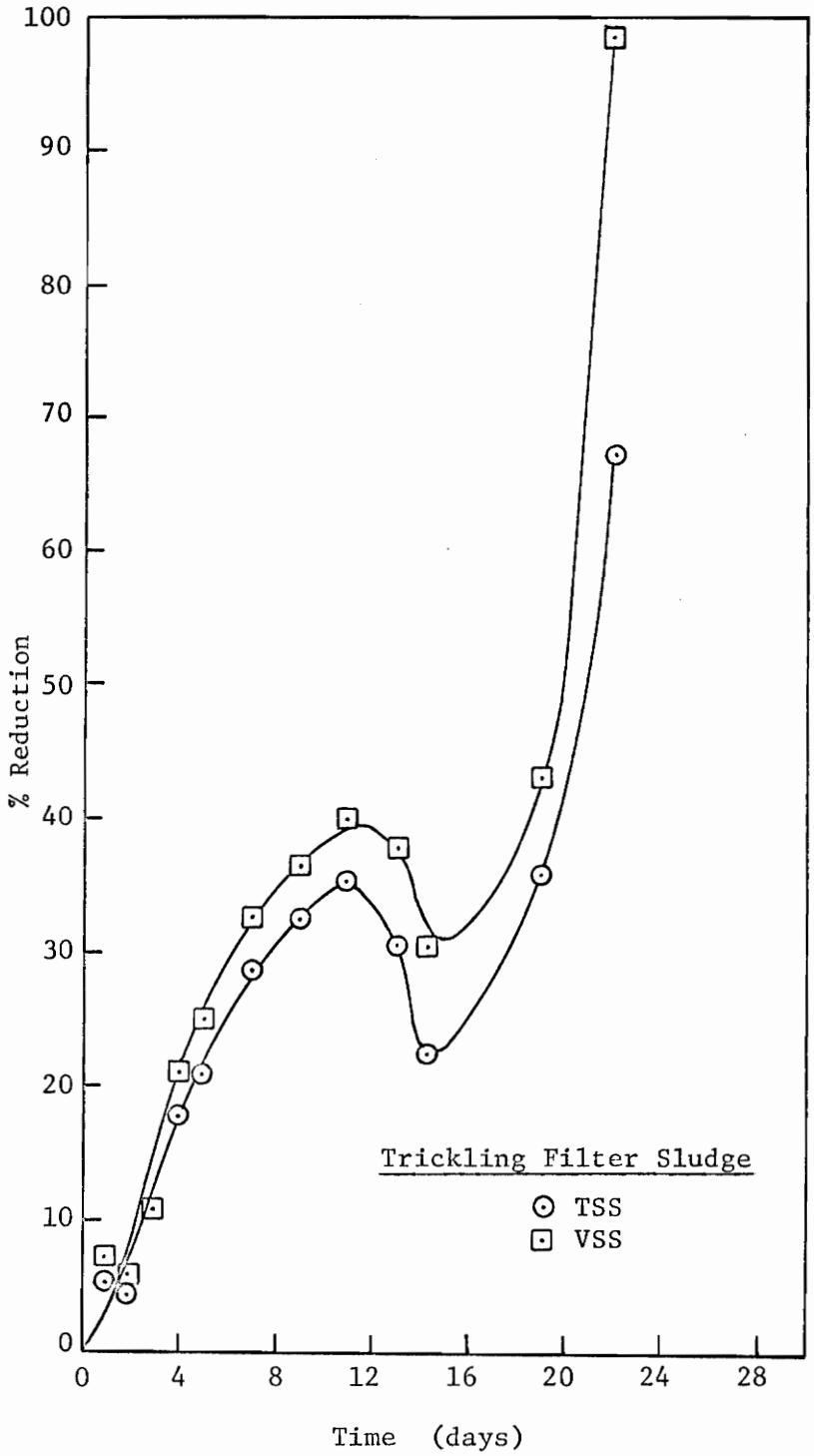

Figure 5: PERCENT SUSPENDED SOLIDS REDUCTION FOR TRICKLING FILTER SLUDGE - DIGESTION RUN 1

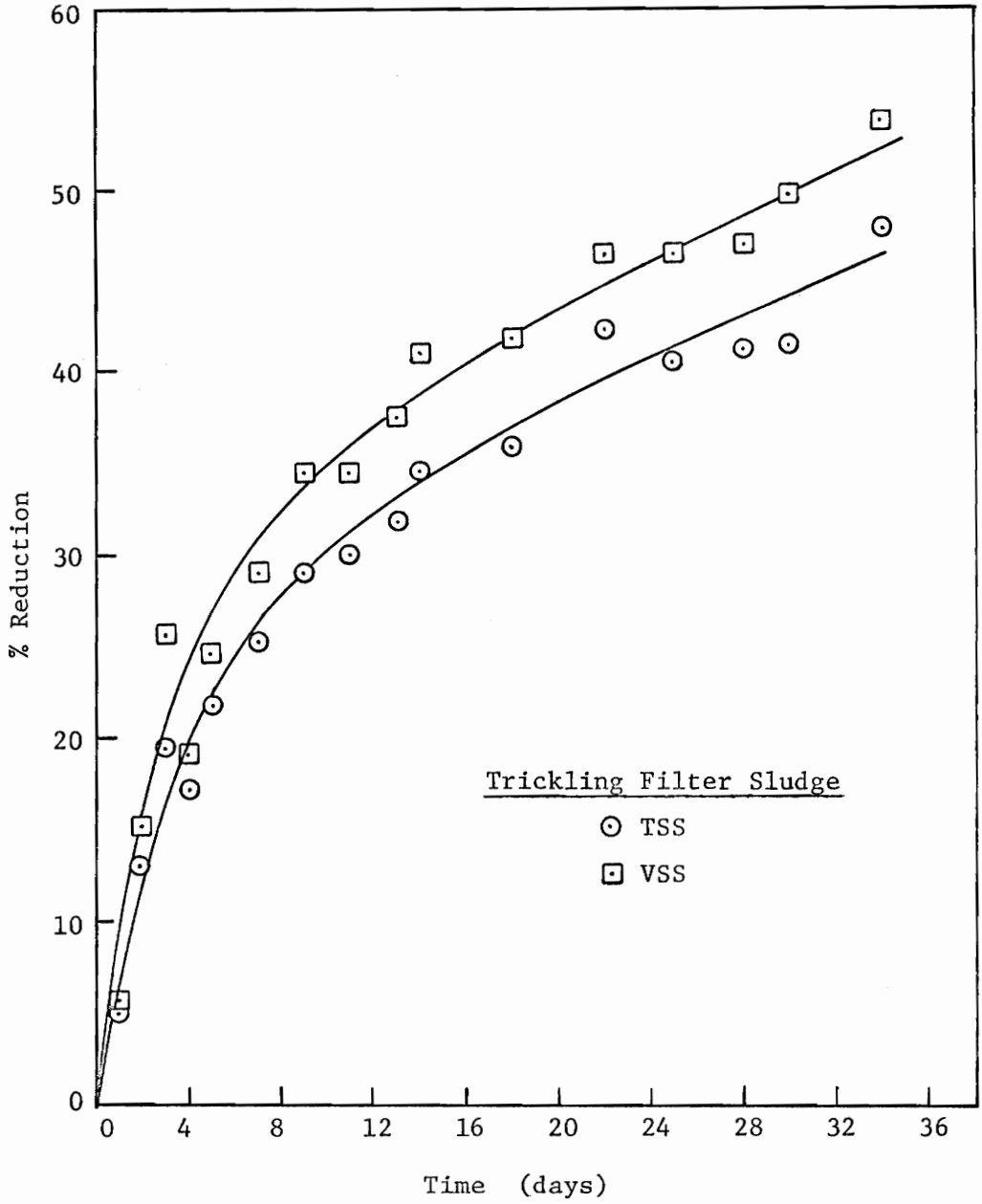

Figure 6: PERCENT SUSPENDED SOLIDS REDUCTION FOR TRICKLING FILTER SLUDGE - DIGESTION RUN 2

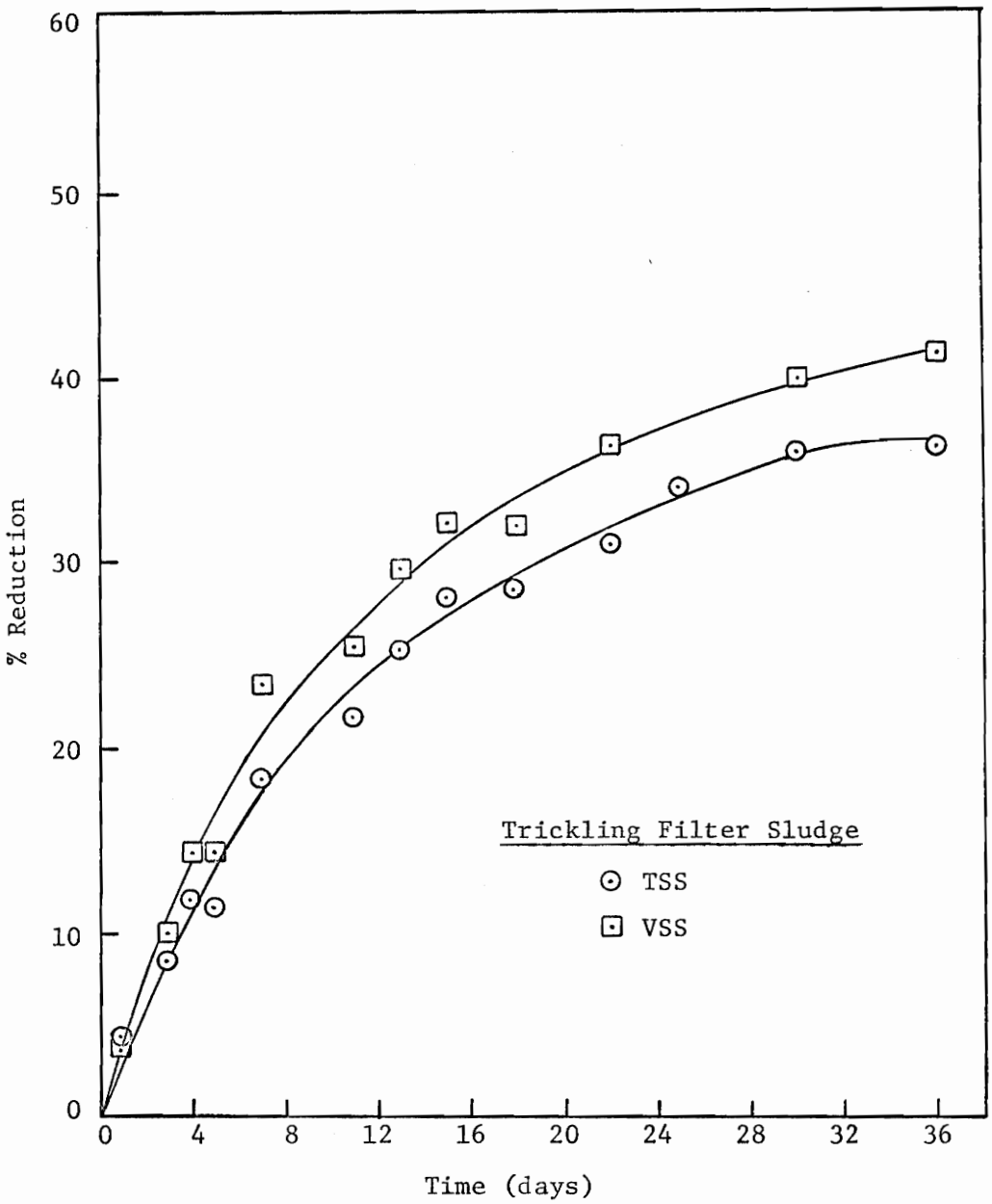

Figure 7: PERCENT SUSPENDED SOLIDS REDUCTION FOR TRICKLING FILTER SLUDGE - DIGESTION RUN 3

and 3 at 30 days, respectively, and 97% for sludge 1 at 22 days. It is interesting to note that during digestion, the greater percentage of solids reductions were observed for the trickling filter sludges of lesser initial solids concentrations. The exception to this trend was observed during the period of solids build-up in digestion run #1.

The digestion of the mixture of primary and trickling filter sludge produced significant solids destructions. The magnitude and percentage of the solids reductions are shown in Figures 8 and 9, respectively. The rate of solids destruction was observed to be significant throughout the 38 day digestion period. Through 30 days digestion, total and volatile suspended solids reductions were observed to exceed 55% and 65%, respectively.

The rate of solids reduction for the waste activated sludge was rather poor. However, due to the condition of the sludge prior to aeration (see Chapter III), this poor showing was not entirely unexpected. The solids data are shown in Figures 10 and 11. An initial increase in both total and volatile suspended solids was noted during digestion. A similar initial solids increase during the digestion of waste activated sludge has been reported by other investigators (23, 25). Moore (23) concluded that the increase was probably due to the unstabilized nature of the sludge at the time of sampling. This initial increase in solids was not observed for any of the other sludges digested during this investigation.

Table III is used to present the variation of volatility (i.e. the percentage ratio of volatile suspended solids to total suspended

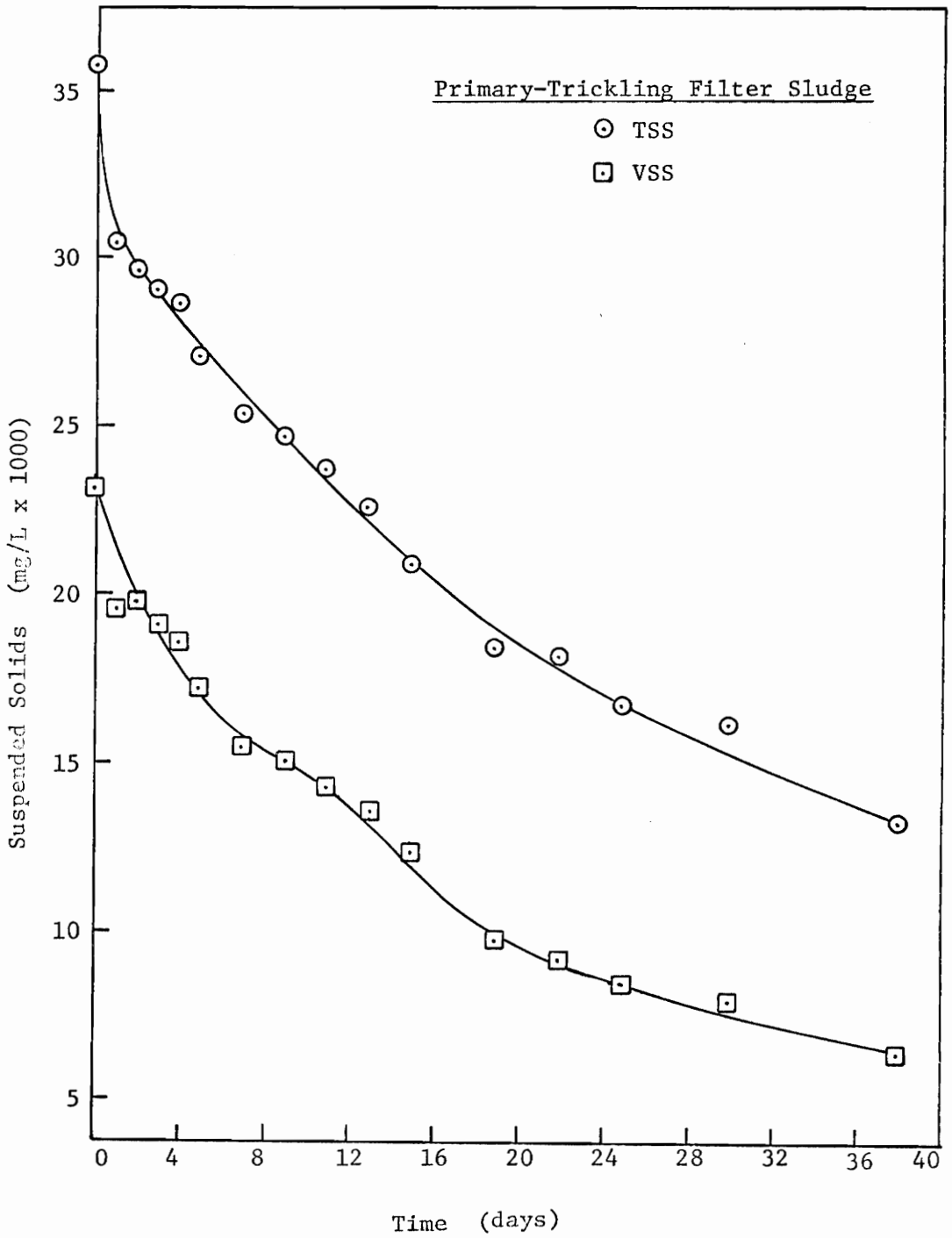

Figure 8: TOTAL AND VOLATILE SUSPENDED SOLIDS VARIATION FOR PRIMARY-TRICKLING FILTER SLUDGE MIXTURE

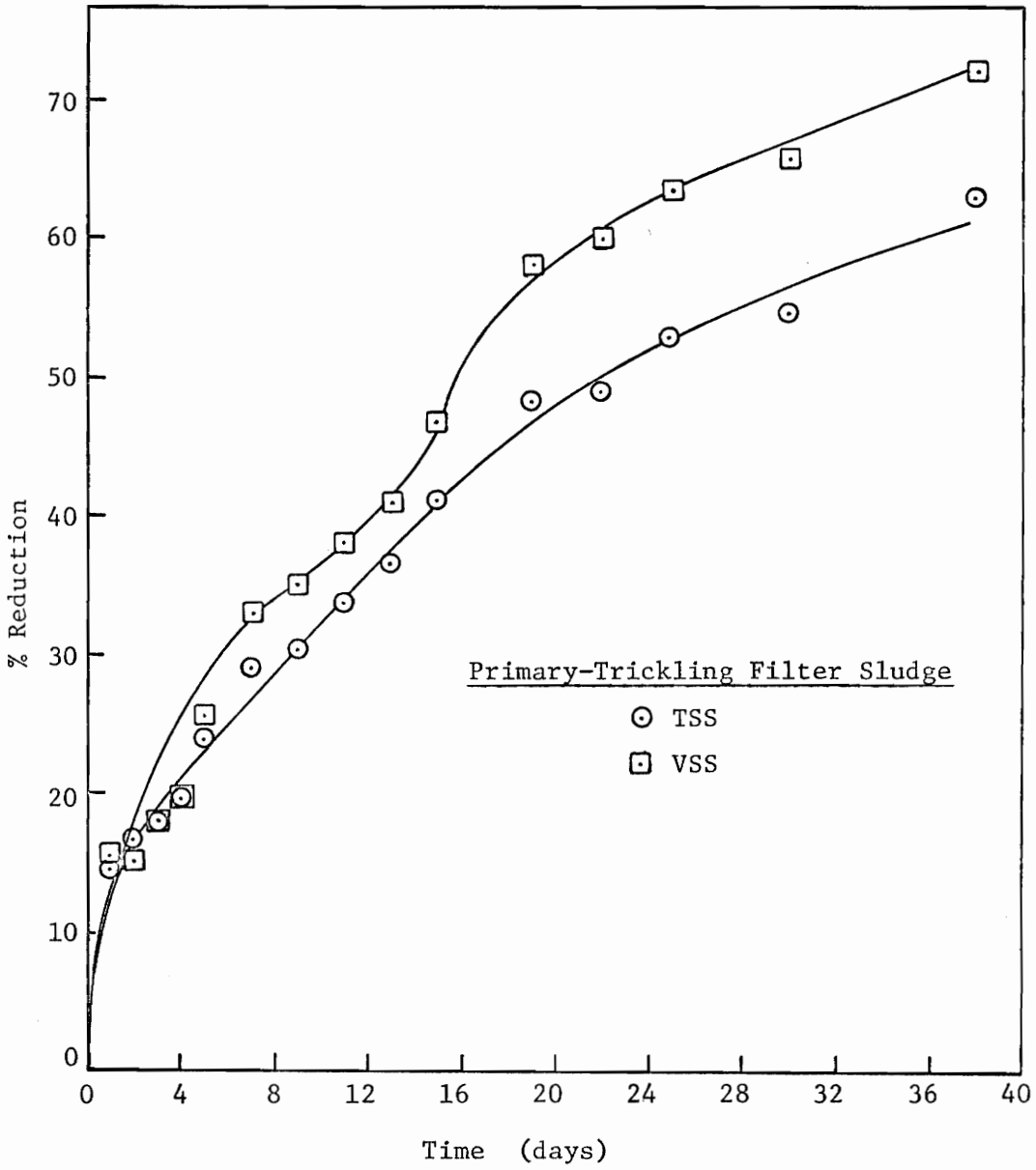

Figure 9: PERCENT SUSPENDED SOLIDS REDUCTION FOR PRIMARY-TRICKLING FILTER SLUDGE MIXTURE

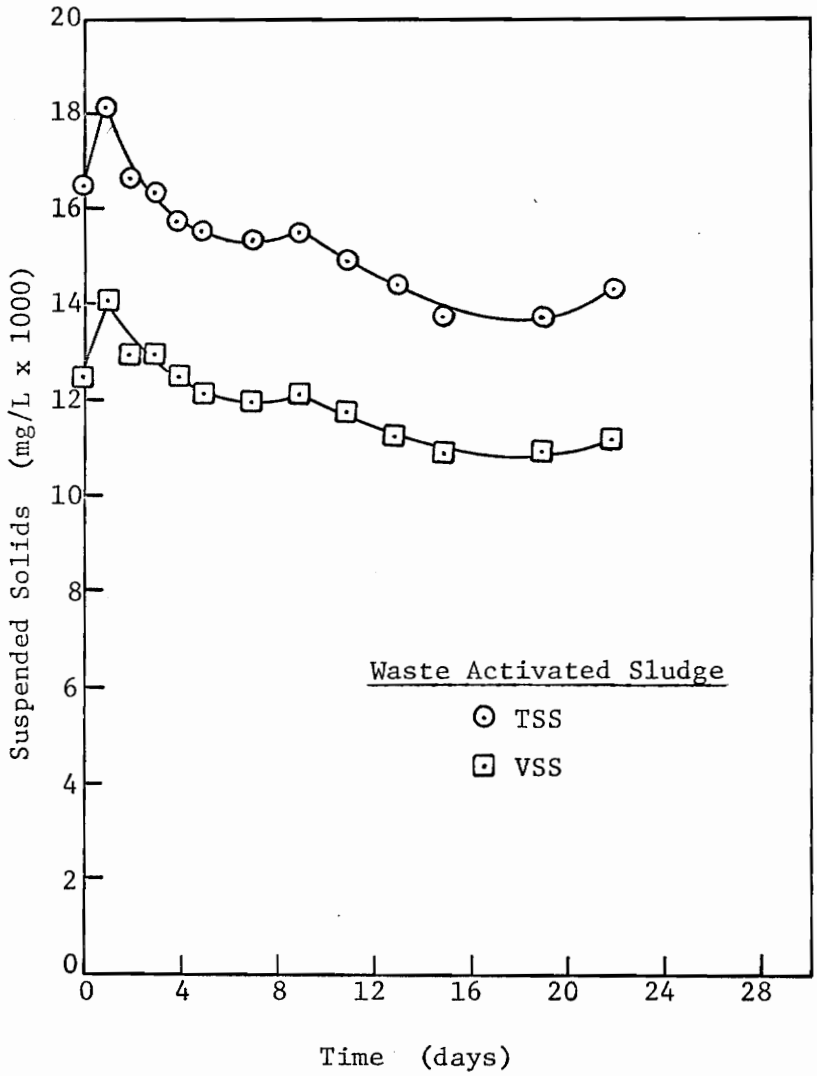

Figure 10: TOTAL AND VOLATILE SUSPENDED SOLIDS VARIATIONS FOR WASTE ACTIVATED SLUDGE

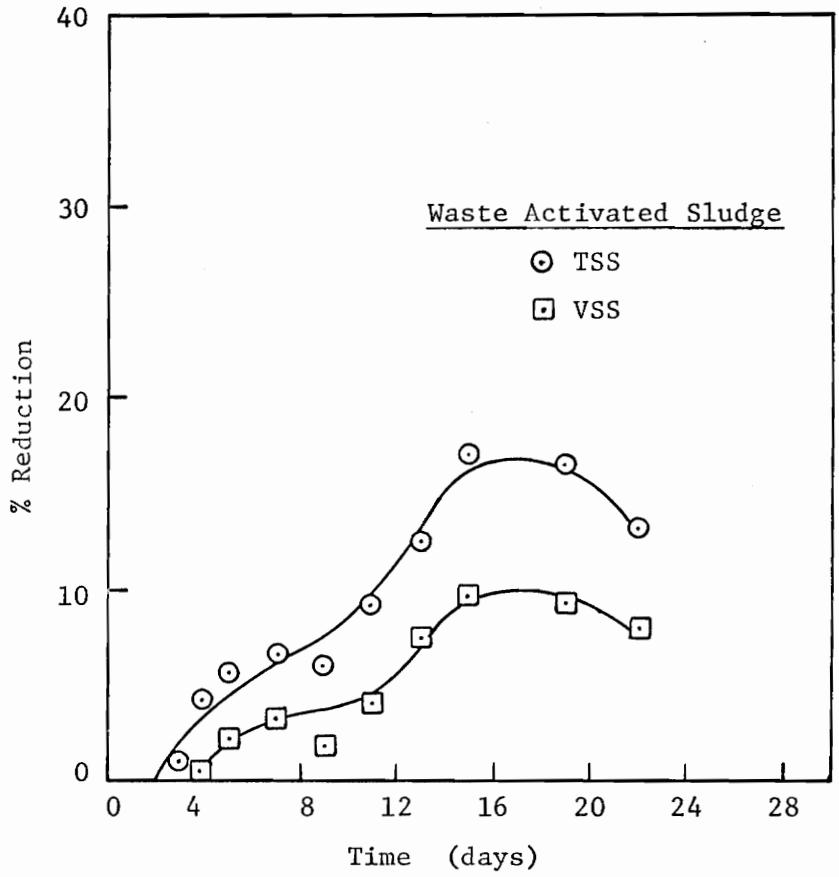

Figure 11: PERCENT SUSPENDED SOLIDS REDUCTION FOR WASTE ACTIVATED SLUDGE

TABLE III

VARIATION IN VOLATILITY DURING AEROBIC DIGESTION

Run No.	Sludge Digested	Digestion Period (days) 0	1	3	7	11	15	19	22	25	30	36
1	Trickling Filter	54.7	53.7	51.7	51.5	50.7	49.0	48.4	2.9	--	--	--
2	Trickling Filter	71.9	71.7	67.3	68.5	66.9	64.7	65.2	66.6	66.2	63.1	65.6
3	Trickling Filter	76.1	76.3	75.0	71.2	72.6	71.9	72.5	70.3	69.1	69.8	69.8
4	Waste Activated	74.5	77.5	79.5	77.5	79.0	79.6	79.3	78.0	--	--	--
5	70% Primary- 30% Trickling Filter*	64.6	63.6	64.8	60.7	60.1	58.5	52.4	50.3	50.3	48.8	--

* by volume

solids) observed for each sludge during the various digestion periods. As previously mentioned, the volatile portion of trickling filter sludge 1 almost completely disappeared. However, the volatility of the other trickling filter sludges exhibited only a slight decrease. Interestingly, the primary and trickling filter sludge mixture had the largest decrease in volatility during digestion. The waste activated sludge actually exhibited an increase in the percent of volatile matter. An apparent trend that might be concluded from the volatility data is: the sludges with the higher initial percent volatile matter had the least change in volatility during digestion.

pH and Alkalinity

The variations in mixed liquor and supernatant pH, as observed during the various digestion runs, are shown in Figures 12 through 15. It was noted that the difference in mixed liquor and supernatant pH for each of the sludges was found to be minimal throughout the digestion period. As might be expected, an initial increase in pH was observed for all sludges upon aeration. The only exception to this observation being the mixed liquor of trickling filter sludge 2. However, as the supernatant of this same sludge did exhibit an initial pH increase, the response of the mixed liquor remains unexplained and may possibly be the result of an experimental error. The magnitudes of these initial pH increases ranged from approximately 0.4 to 0.8 to 1.6 pH units for the waste activated, trickling filter, and primary-trickling filter sludges, respectively. Although the duration of the

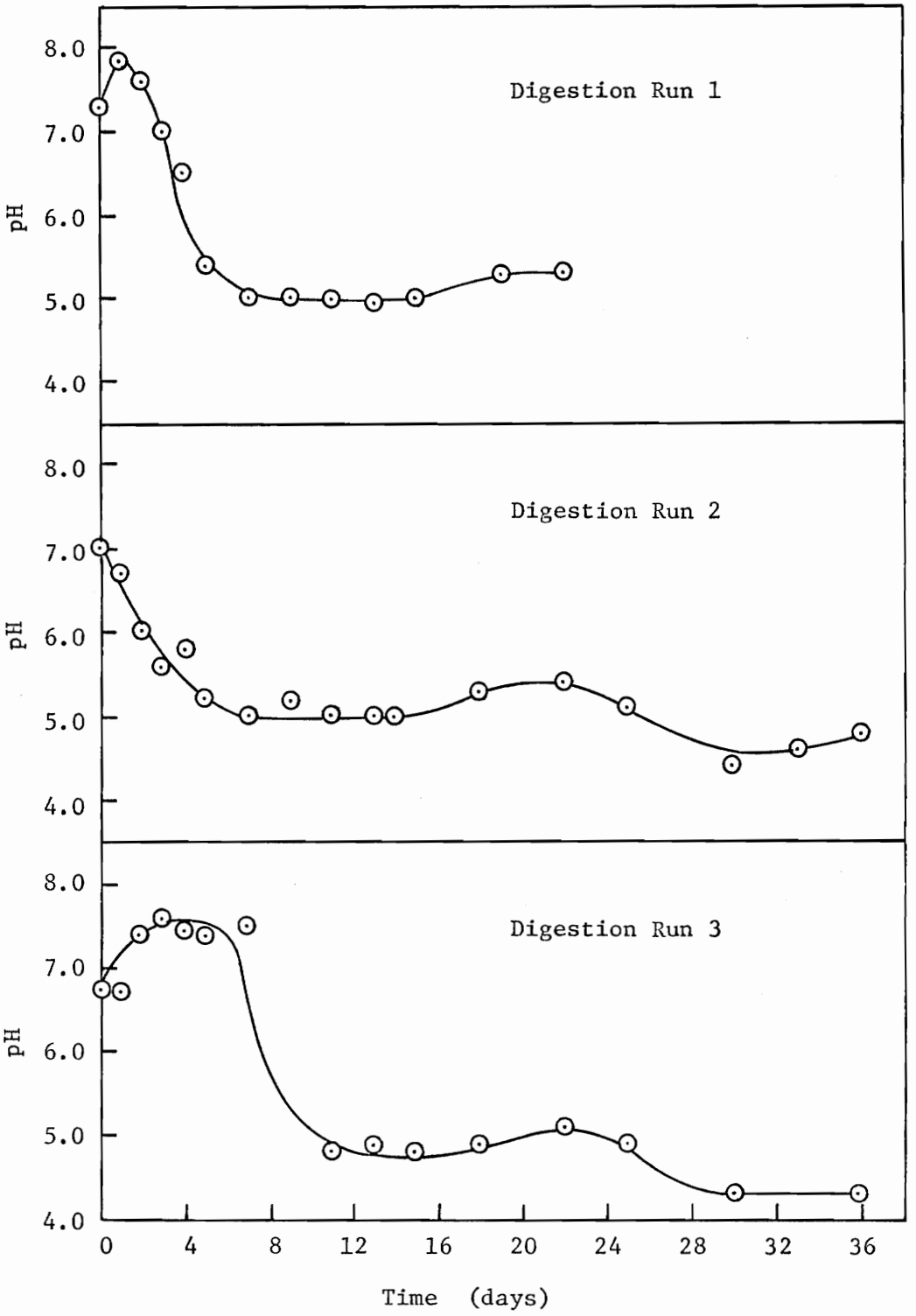

Figure 12: MIXED LIQUOR pH DATA FOR TRICKLING FILTER SLUDGES

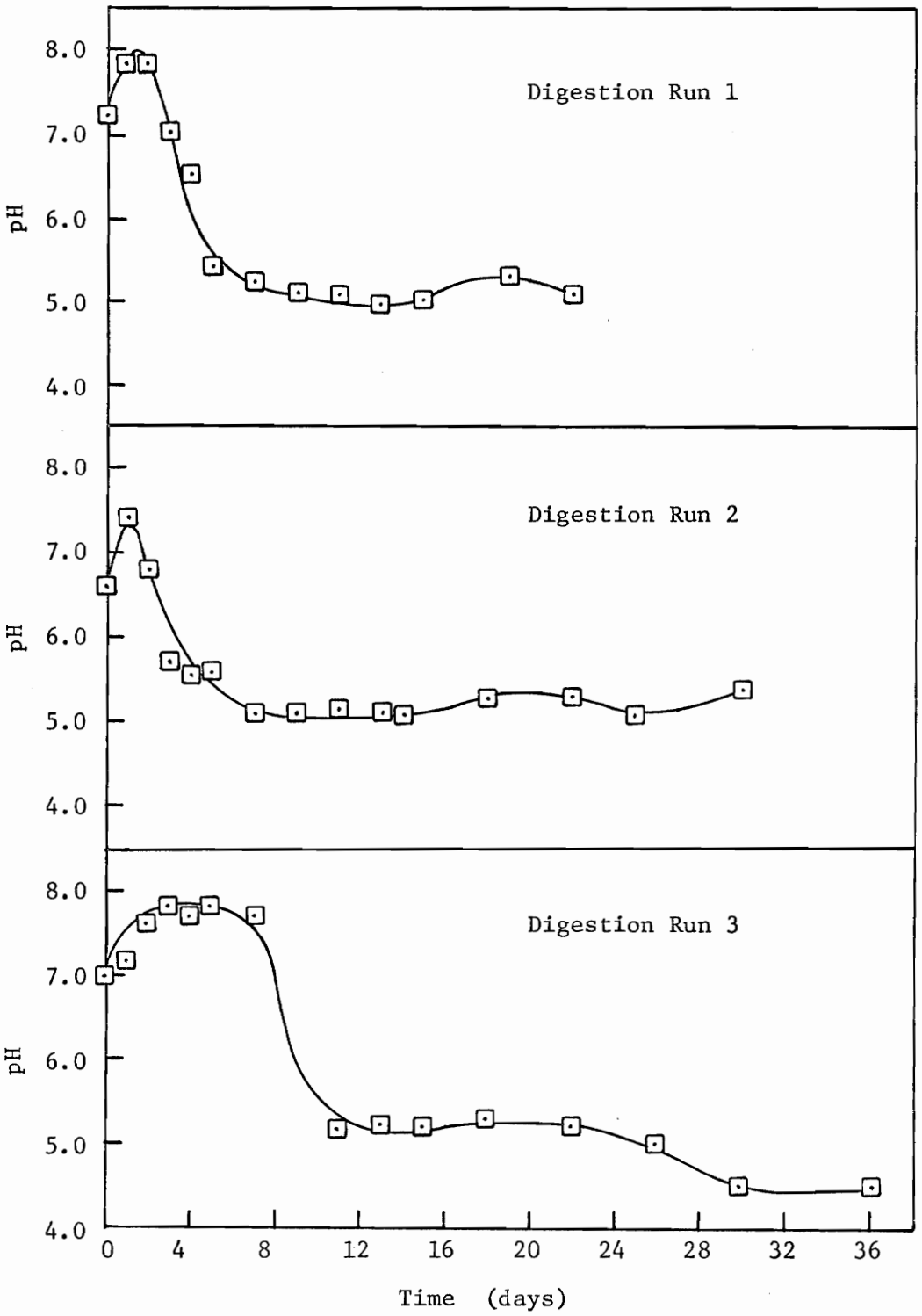

Figure 13: SUPERNATANT pH DATA FOR TRICKLING FILTER SLUDGES

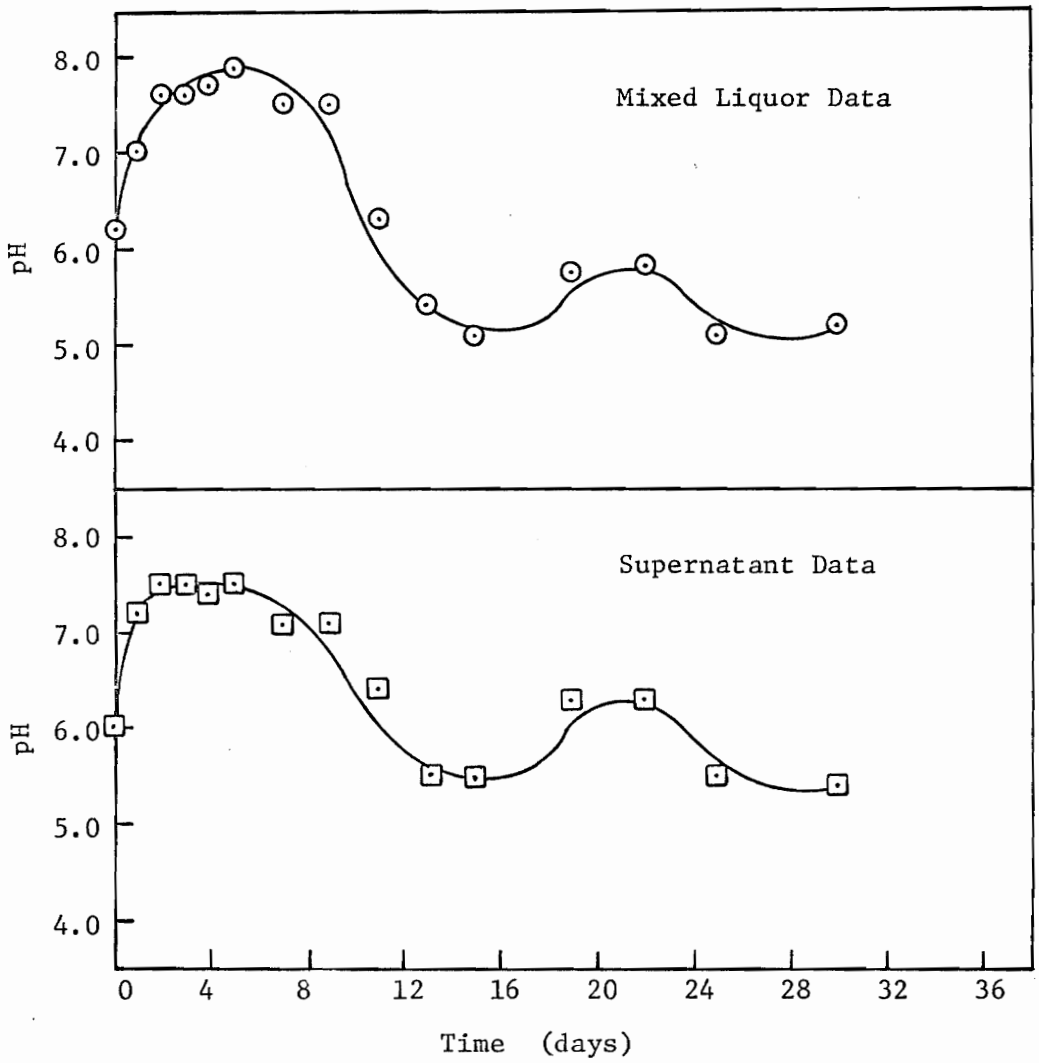

Figure 14: pH DATA FOR PRIMARY-TRICKLING FILTER SLUDGE MIXTURE

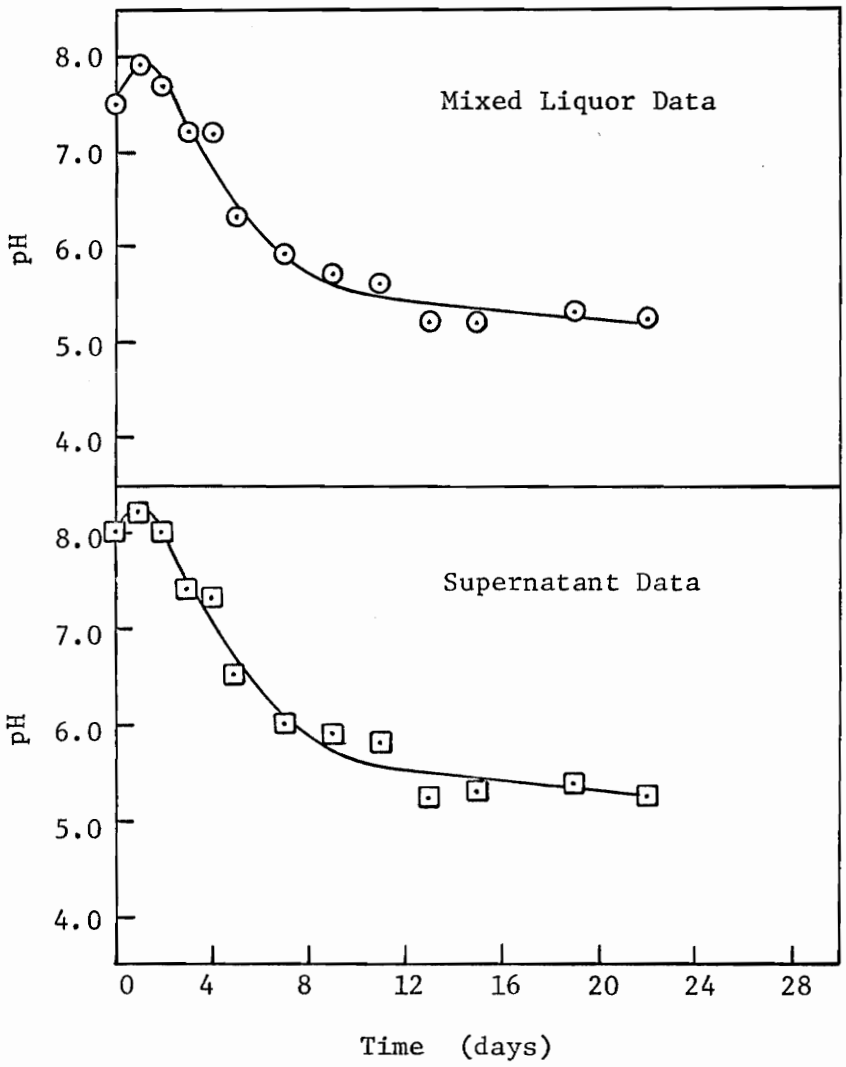

Figure 15: pH DATA FOR WASTE ACTIVATED SLUDGE

peak in pH was found to be short for the waste activated sludge and trickling filter sludges 1 and 2, the primary-trickling filter sludge mixture as well as trickling filter sludge 3 maintained a rather prolonged pH peak, indicating the presence of a strong buffering system in these sludges. After approximately 12 days aeration and significant pH reduction, the pH of most the sludges stabilized at various levels between pH 4.2 and 5.5. In no case did this low pH seem to severely inhibit solids reduction through the remainder of the digestion period.

The variation in supernatant alkalinity for digestion runs # 1, 2, 4, and 5 are shown in Figures 16 and 17. Trickling filter sludges 1 and 2, as well as the waste activated sludge all showed a rather complete reduction (greater than 95%) in total alkalinity in less than 8 days aeration. As would be expected from an analysis of the pH data, the mixture of primary and trickling filter sludges exhibited a higher initial alkalinity value and required some 30 days of aeration to achieve an alkalinity reduction comparable to the other sludges.

$\underline{BOD_5}$

The reductions in mixed liquor and supernatant biochemical oxygen demand during the digestion of the three trickling filter sludges are shown in Figures 18 and 19. respectively. Due to the high BOD_5 of the mixed liquors for sludges 2 and 3, their initial values could not be experimentally obtained using the prescribed sample dilution.

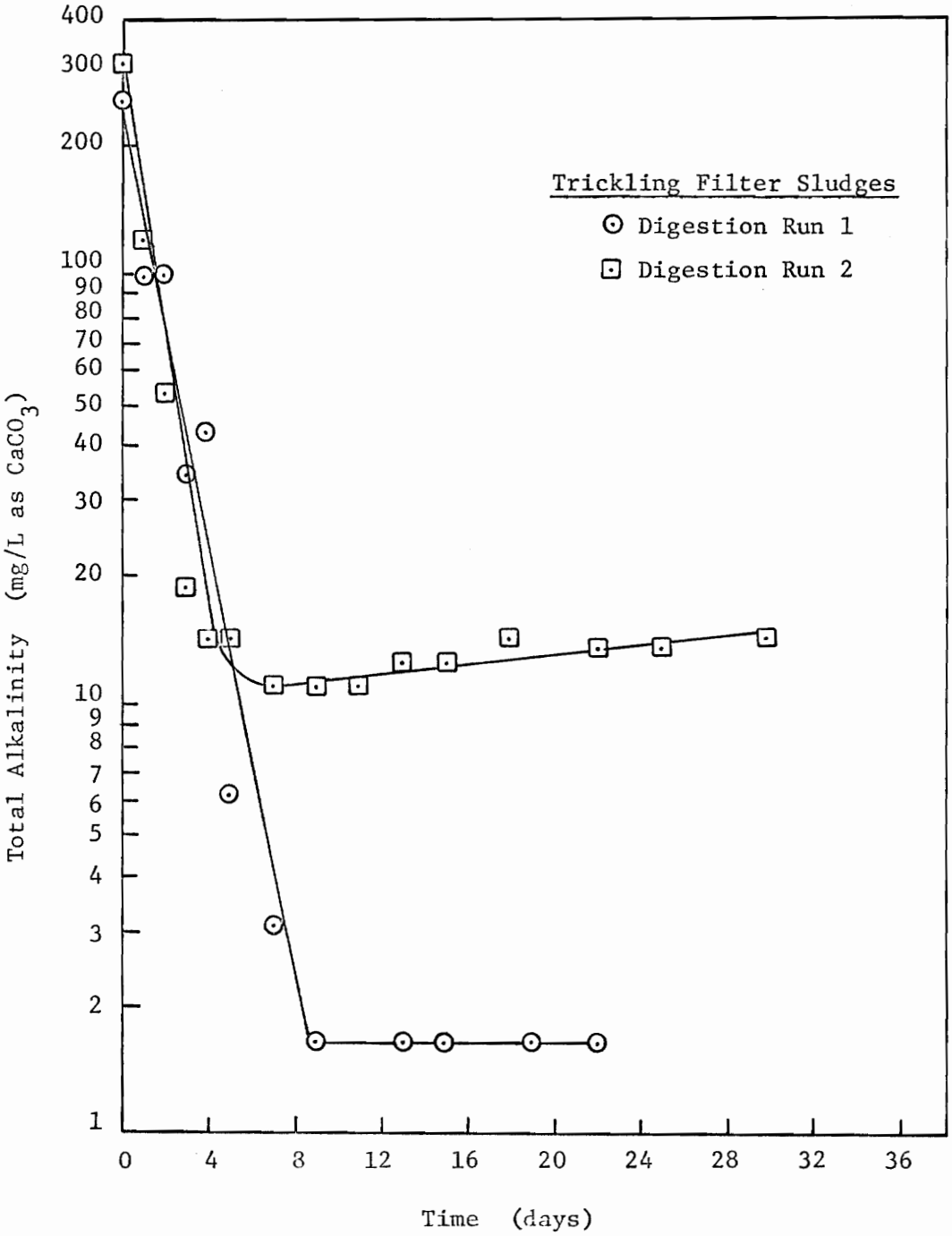

Figure 16: SUPERNATANT ALKALINITY DATA FOR TRICKLING FILTER SLUDGES

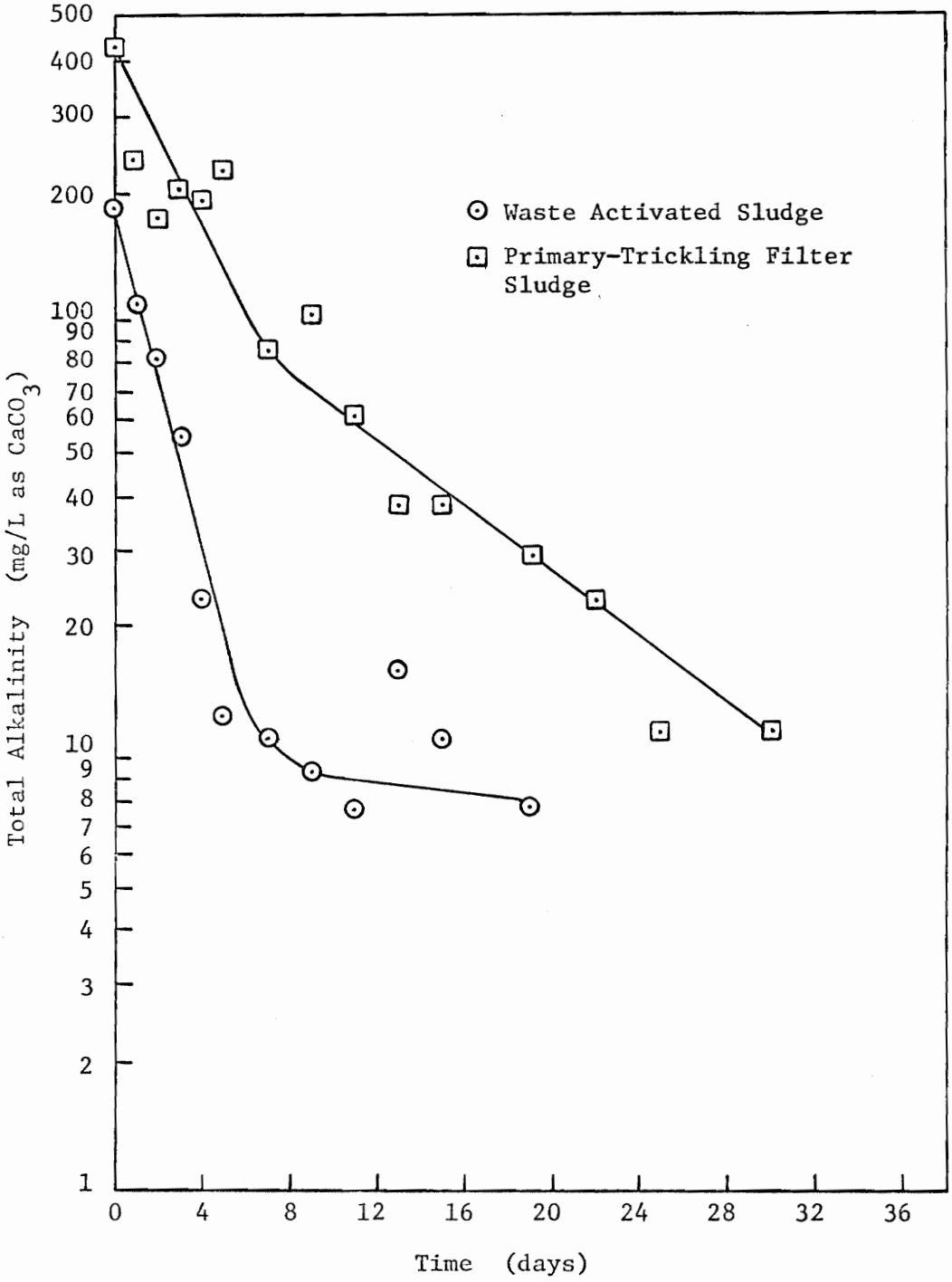

Figure 17: SUPERNATANT ALKALINITY DATA FOR DIGESTION RUNS 4 & 5

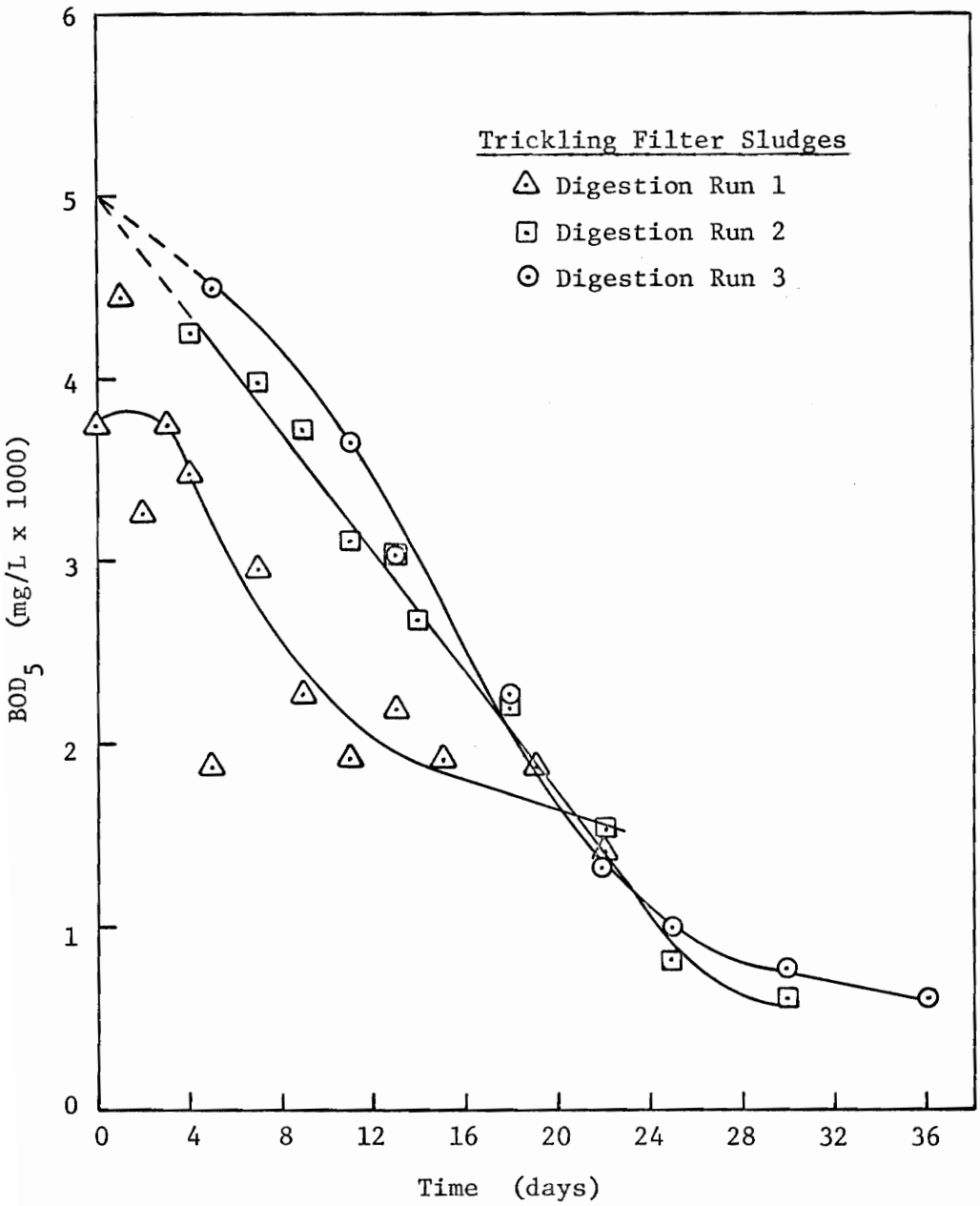

Figure 18: MIXED LIQUOR BOD_5 DATA FOR TRICKLING FILTER SLUDGES

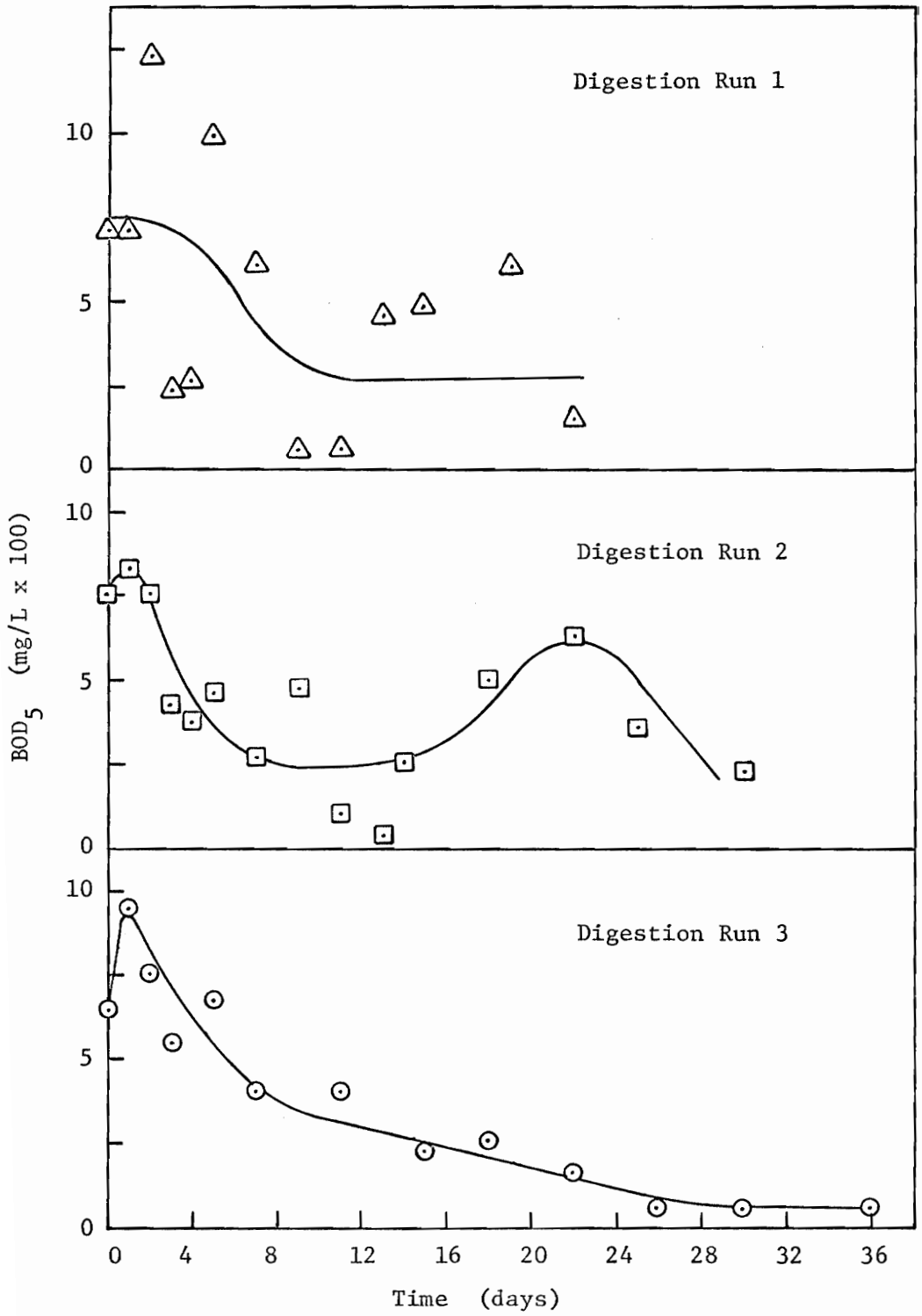

Figure 19: SUPERNATANT BOD_5 DATA FOR TRICKLING FILTER SLUDGES

Consequently, these values were obtained by extending the upper portion of the BOD_5 curves to intersect the ordinate of the graph. Using this assumption, the mixed liquor BOD_5 for sludges 2 and 3 was reduced 85% during 30 days of aeration. The supernatant BOD_5 values of sludges 1 and 2 were found to vary widely from day to day during digestion. However, a definite trend to decrease the BOD_5 supernatant value to less than 500 mg/l after 10 days aeration was noted. By filtering the supernatant of sludge 3 through an 8 micron Millipore filter prior to inoculation, these daily variations were virtually eliminated. Consequently, the supernatant BOD_5 values for sludge 3 were observed to decrease to less than 200 mg/l after 18 days digestion and eventually level off at about 60 mg/l after 26 days.

The mixture of primary and trickling filter sludge exhibited a rate of mixed liquor and supernatant BOD_5 reductions strikingly similar to those observed for trickling filter sludges 2 and 3. The data for this digestion run is shown in Figure 20. A mixed liquor BOD_5 reduction of approximately 70% was noted through 30 days of aeration.

The data shown in Figure 21 was used to compute a 62% reduction in mixed liquor BOD_5 during the 22 day digestion of waste activated sludge. As the initial supernatant BOD_5 for this sludge was found to be less than 400 mg/l, no appreciable reduction in supernatant BOD_5 occurred during digestion.

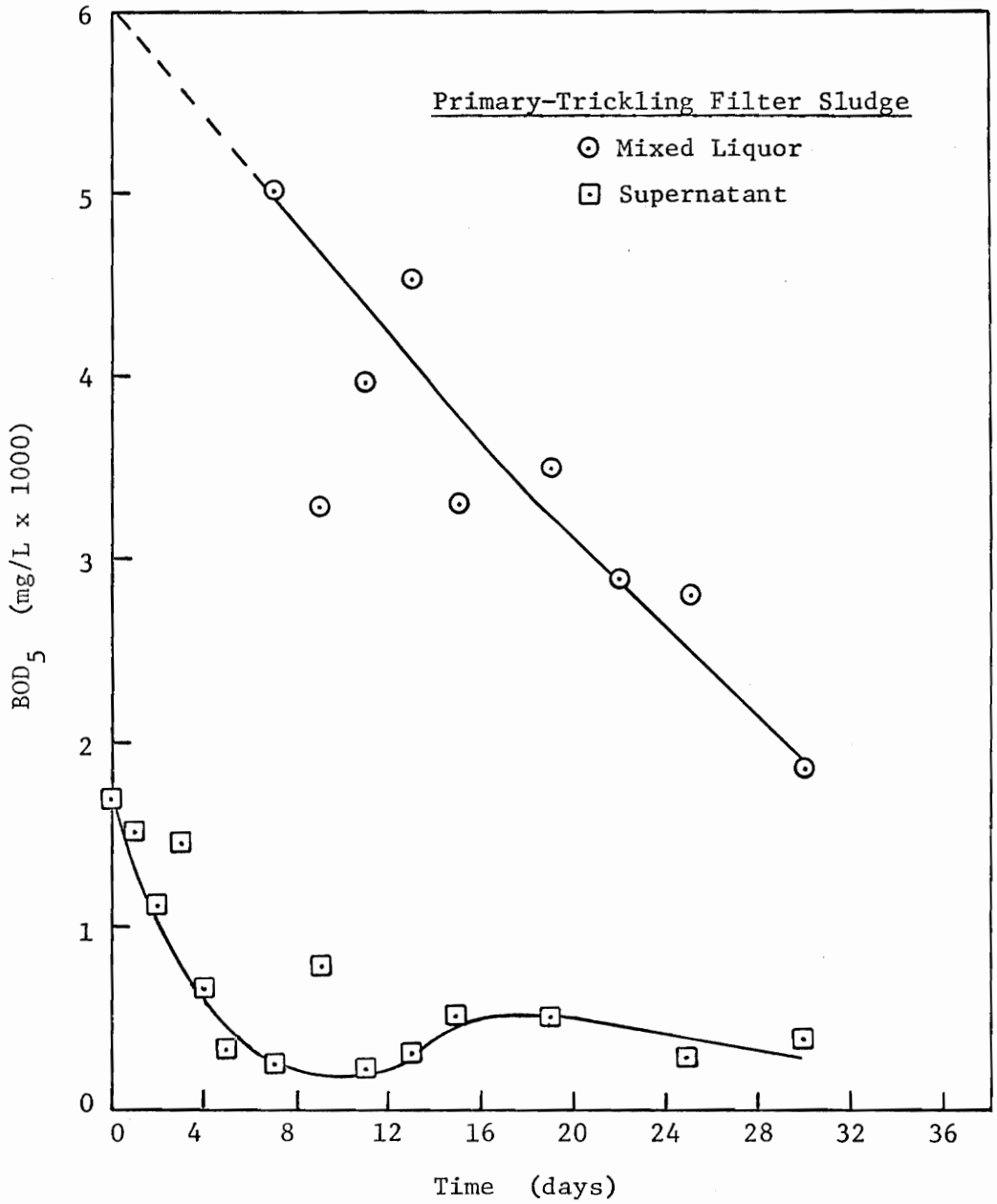

Figure 20: BOD_5 DATA FOR PRIMARY-TRICKLING FILTER SLUDGE MIXTURE

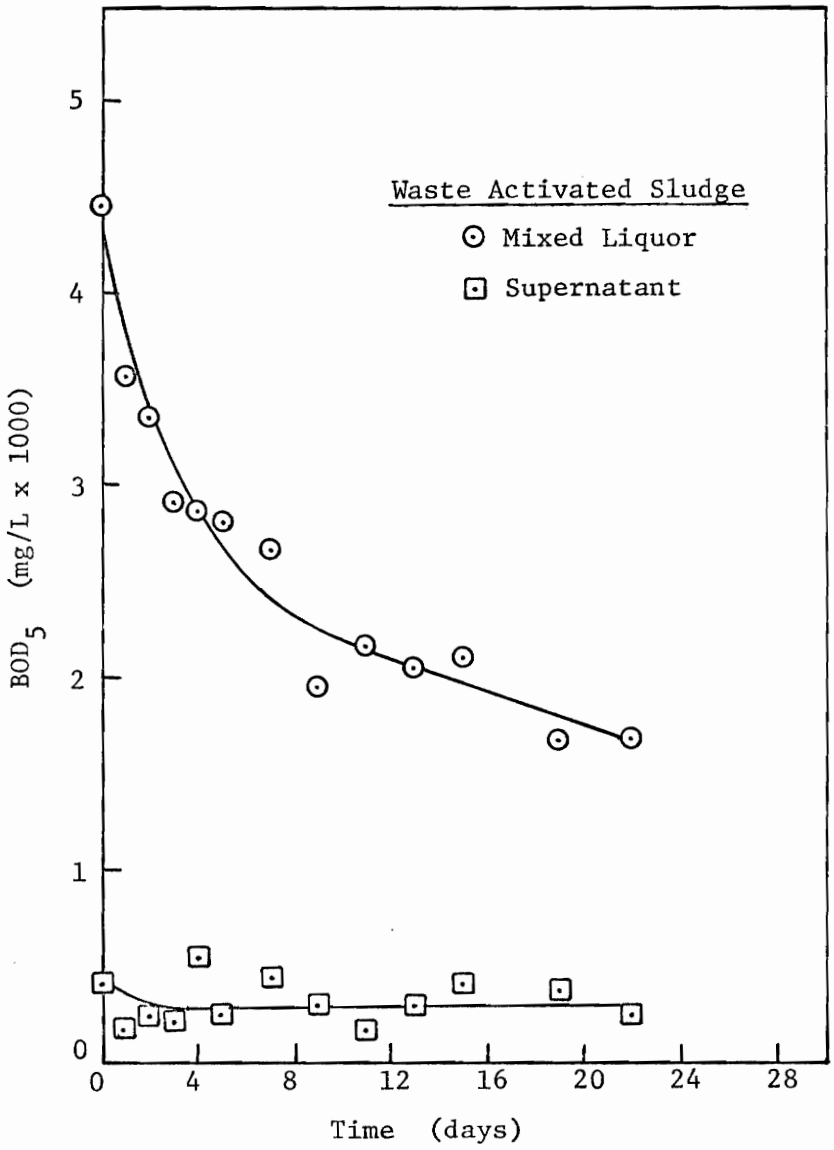

Figure 21: BOD_5 DATA FOR WASTE ACTIVATED SLUDGE

Oxygen Uptake Rate

The rate of oxygen utilization, or uptake, has been plotted against digestion time for trickling filter sludges 2 and 3, the mixture of primary and trickling filter sludge, and the waste activated sludge. This data is presented in Figures 22 and 23. Oxygen uptake in not a decisive digestion parameter by itself. However, when correlated with other parameters (e.g. solids reductions, BOD_5, filterability) it can be a valuable tool in determining the causes of various responses in these other parameters.

Figure 22 shows that trickling filter sludge does indeed progress to the state of endogenous respiration upon prolonged aeration. For sludge 2, a peak in oxygen utilization occurred after 1 day of digestion and endogeny was established at approximately 5 days. However, it is rather peculiar that sludge 3 did not exhibit an oxygen utilization peak until 7 days of digestion and endogenous respiration was not established until approximately 12 days. Although the exact reason for this response is unknown, it may possibly be related to the condition of the sludge at the time of procurement. The biological filter slime represented by sludge 3 may not have had time to fully adjust to the increase in loading caused by the return of VPI & SU students for the fall quarter (see Chapter III).

Figure 23 illustrates that although the mixture of primary and trickling filter sludge was only 30% biological sludge by volume, the system was still carried into the endogenous phase. The oxygen uptake peak was observed at 2 days and endogenous respiration was estab-

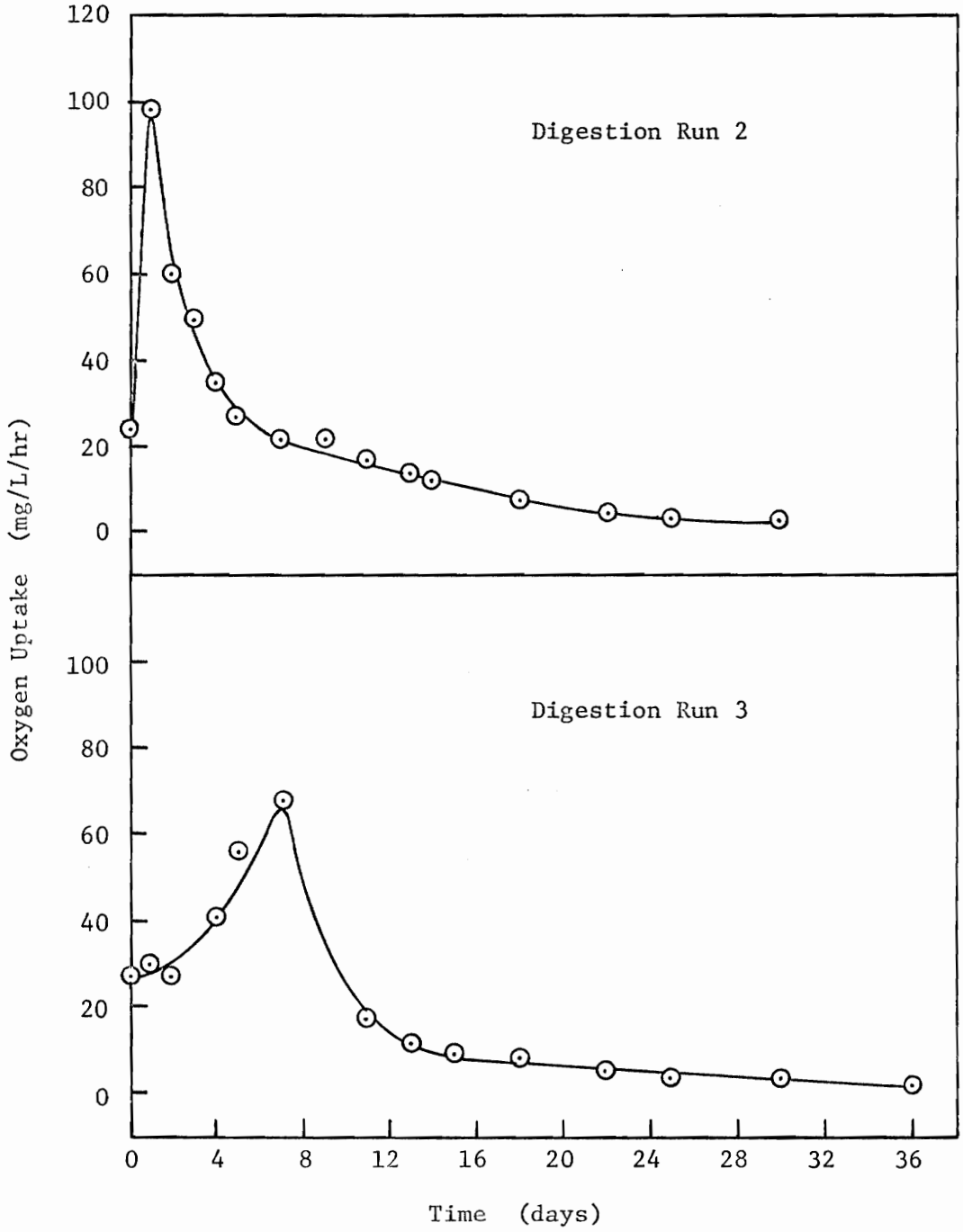

Figure 22: OXYGEN UPTAKE DATA FOR TRICKLING FILTER SLUDGES

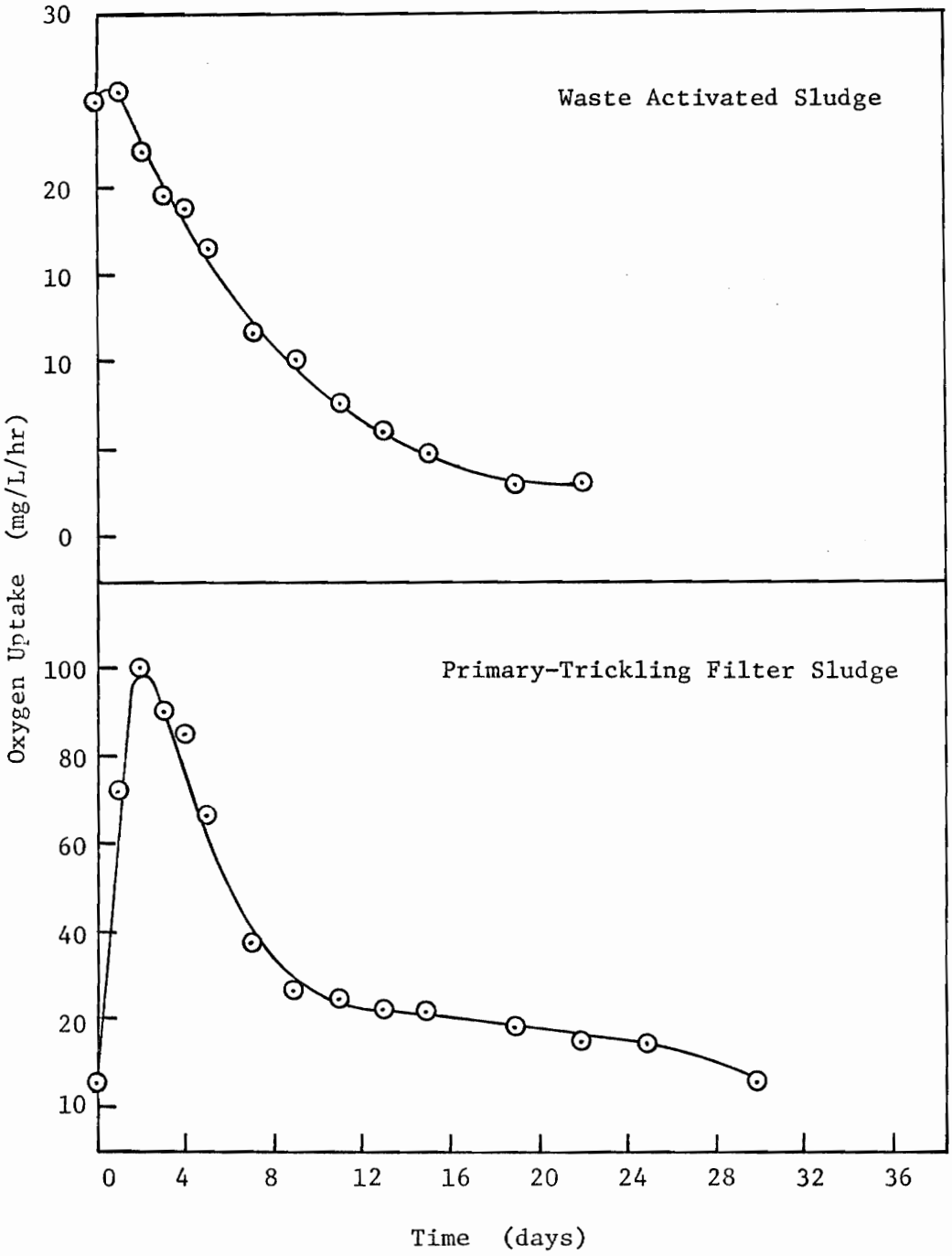

Figure 23: OXYGEN UPTAKE DATA FOR DIGESTION RUNS 4 & 5

lished within 10 days of digestion.

No real peak in oxygen utilization was noted for the waste activated sludge (Figure 23). The beginning of the endogenous phase is also graphically unclear. Due to the nature of the sludge at the time of procurement (see Chapter III), it is surmised that the sludge was in a state of endogeny throughout the entire digestion period and that the oxygen utilization was induced primarily to satisfy the BOD_5 of the sludge (see Figure 21).

Filterability

Specific resistance values for all the sludges were determined during the digestion periods and are shown in Figures 24 through 27. The actual experimental data which was collected and utilized to compute the specific resistance is given in the Appendix.

From Figure 24, it can be seen that the specific resistance of trickling filter sludge 1 was initially high, then decreased almost 85% through 2 days of aeration, after which it rose again to a level near the initial value. A second period of improved filterability occurred between 15 and 20 days digestion, with a minimum specific resistance value approximately one-half that of the initial, non-digested sludge.

Similarly, trickling filter sludge 2 (Figure 24) exhibited a slight decrease in specific resistance during the first few days of digestion. After this period, the filterability worsened and the specific resistance was never less than twice that of the undigested

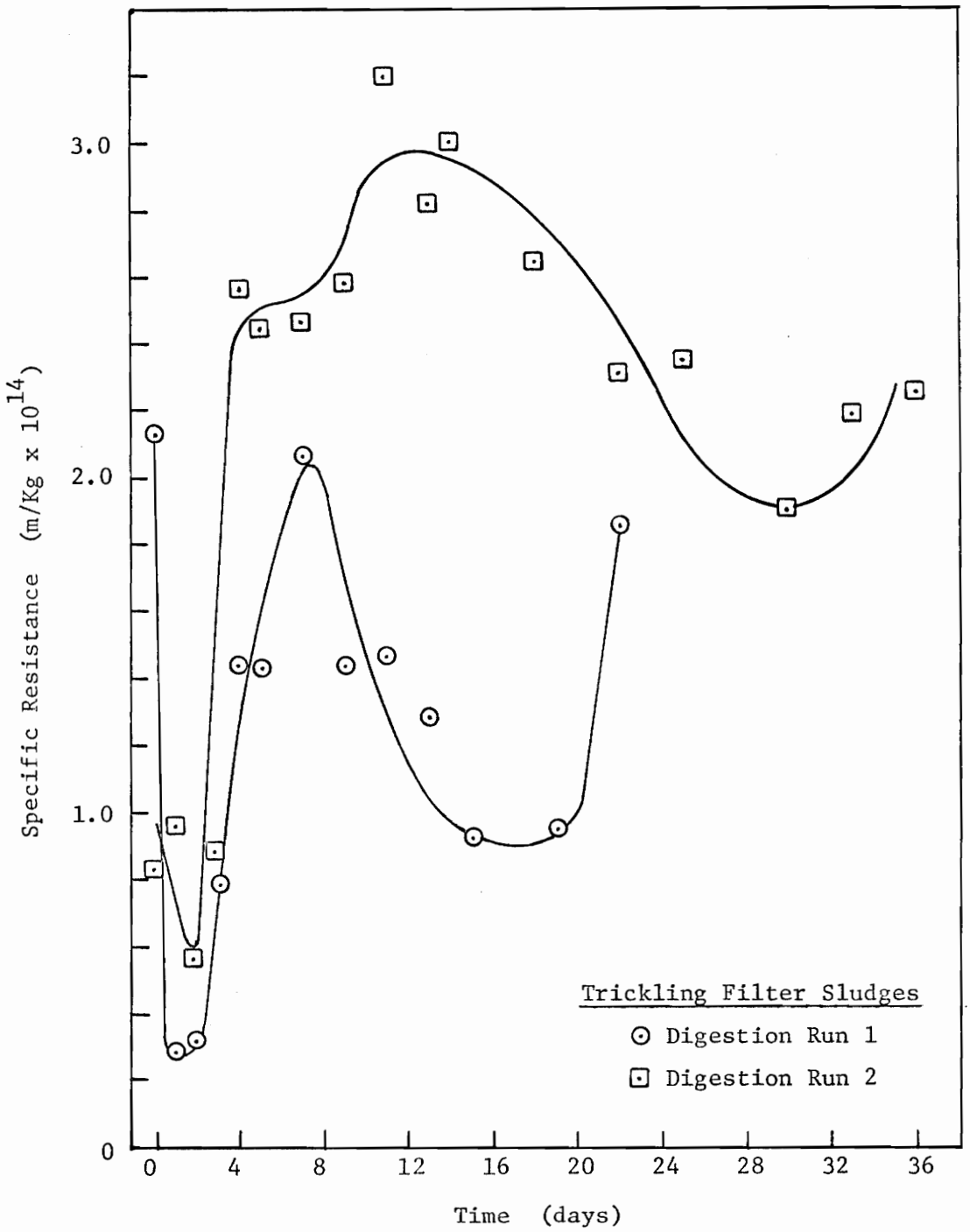

Figure 24: SPECIFIC RESISTANCE DATA FOR TRICKLING FILTER SLUDGES - DIGESTION RUNS 1 & 2

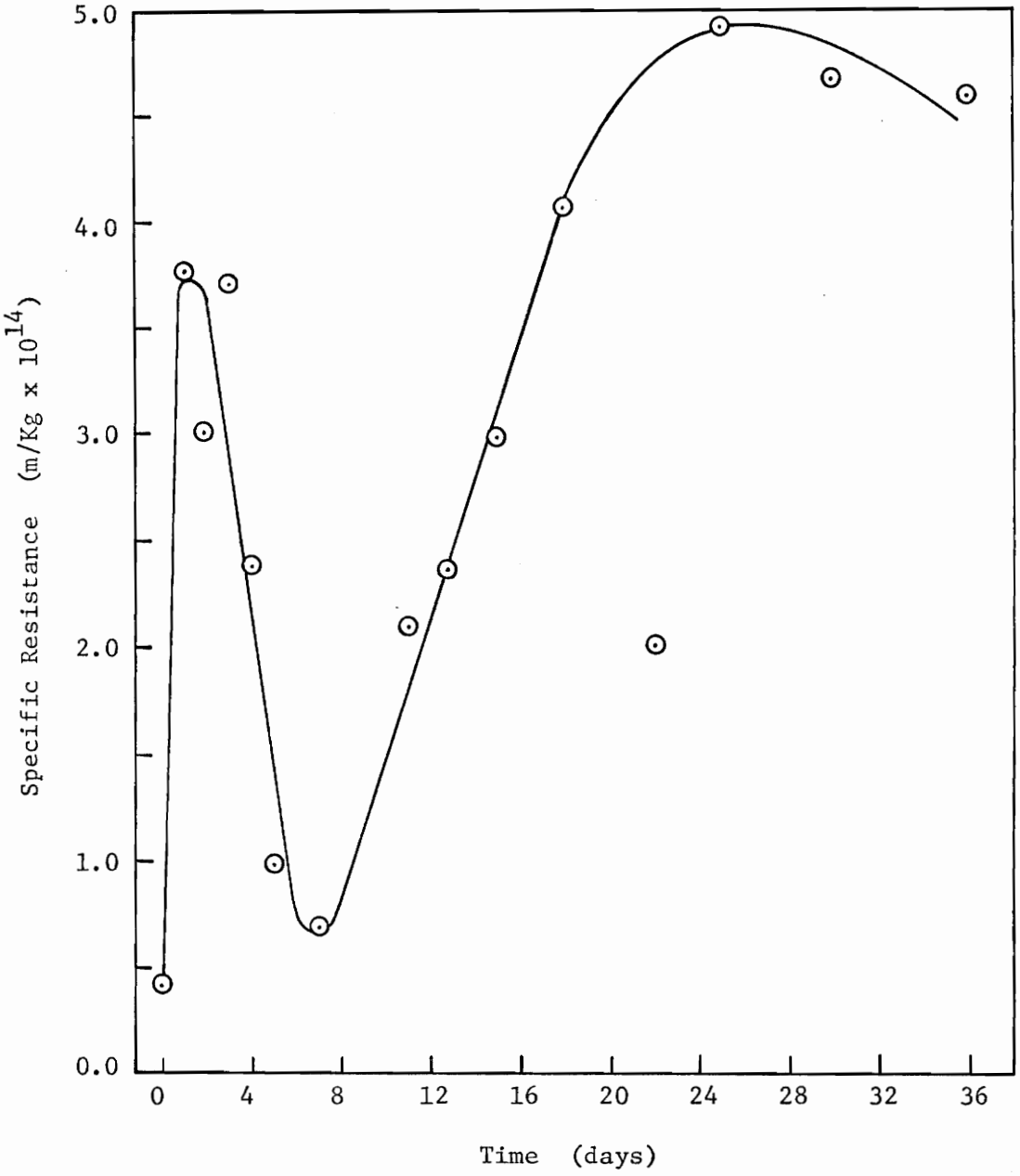

Figure 25: SPECIFIC RESISTANCE DATA FOR TRICKLING FILTER SLUDGE - DIGESTION RUN 3

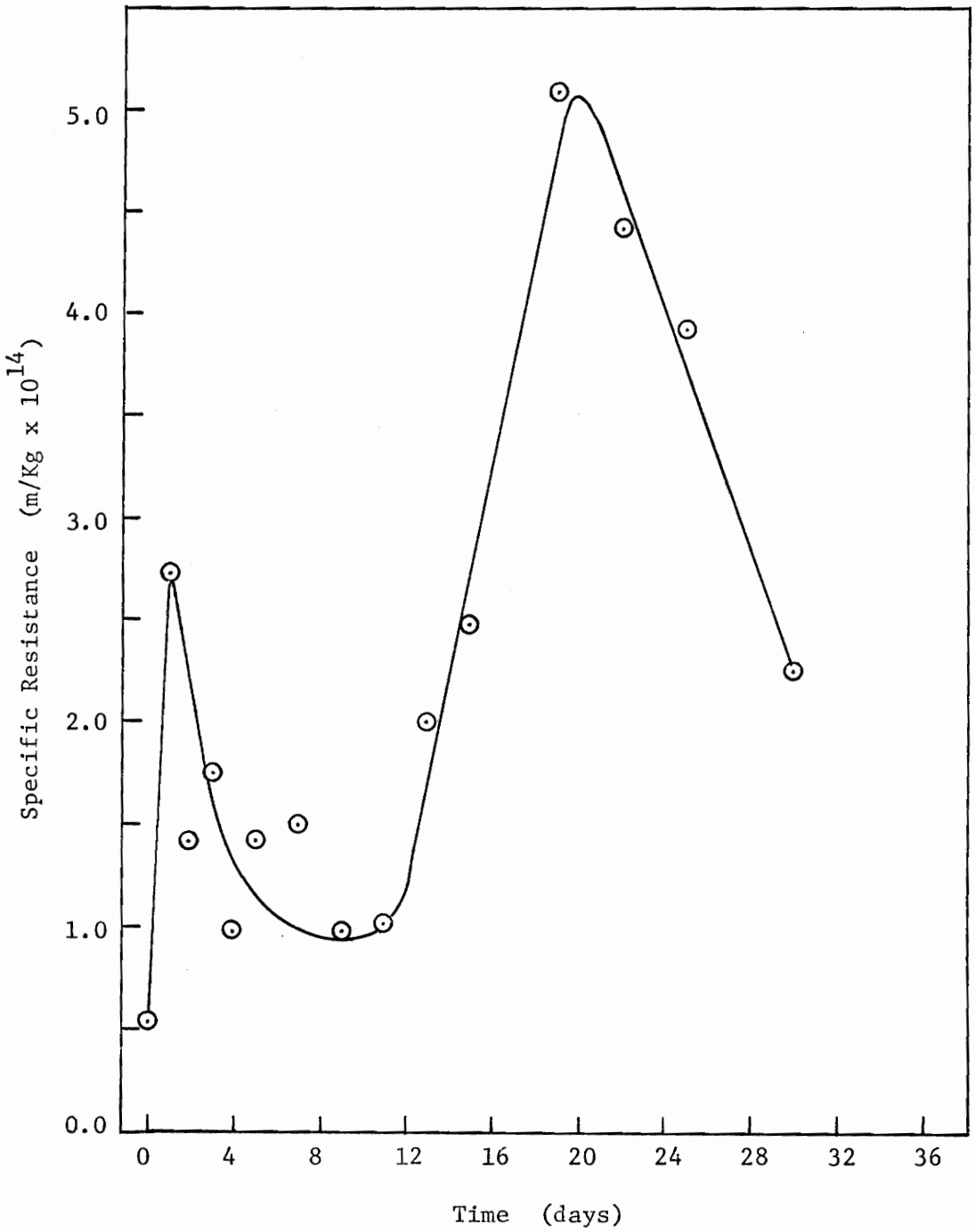

Figure 26: SPECIFIC RESISTANCE DATA FOR PRIMARY-TRICKLING FILTER SLUDGE MIXTURE

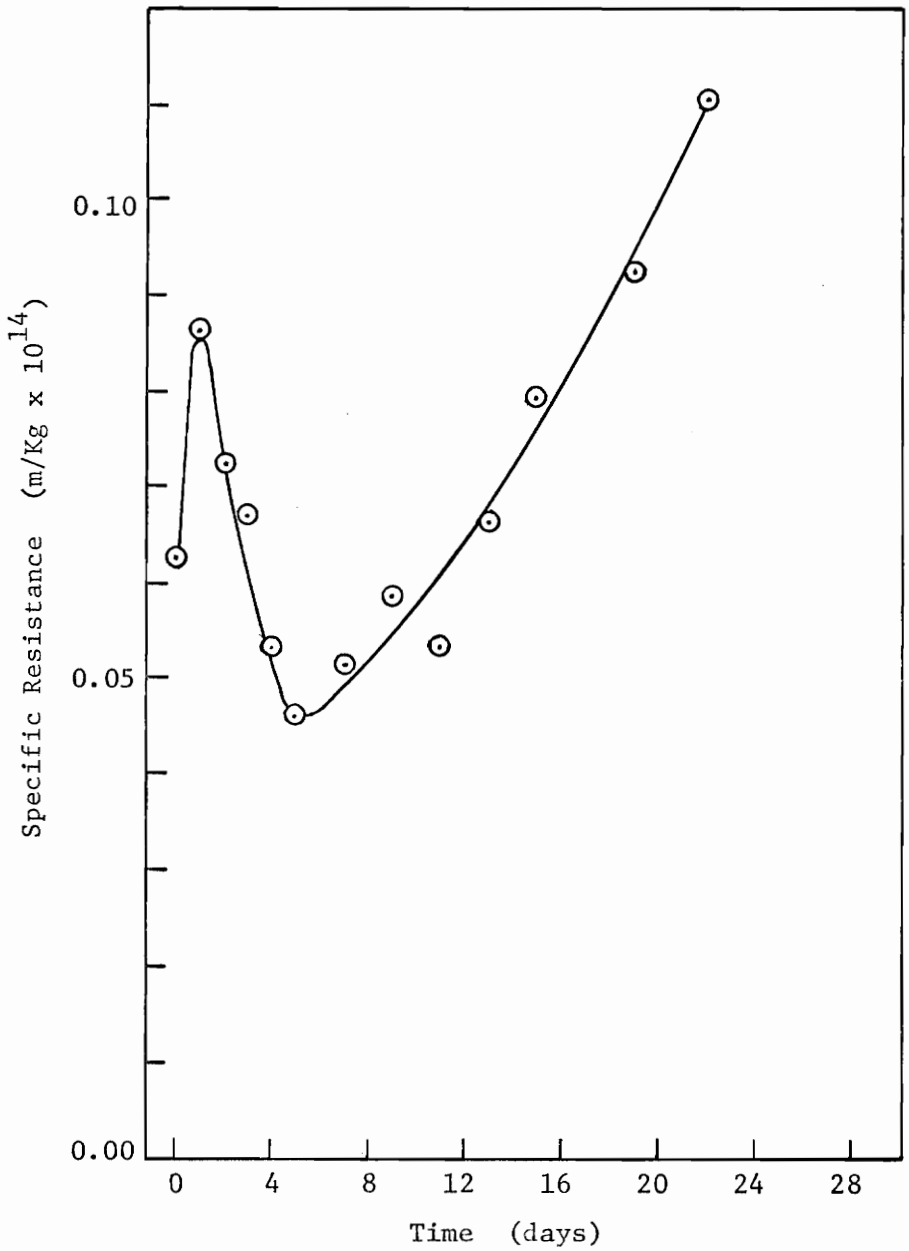

Figure 27: SPECIFIC RESISTANCE DATA FOR WASTE ACTIVATED SLUDGE

sludge. It is interesting to note that the period of best filterability coincided with the peak in oxygen utilization (Figure 22) for the sludge.

Figure 25 shows the specific resistance of trickling filter sludge 3. It increased sharply through the first few days of digestion and then dropped off sharply to a minimum value at 7 days. Beyond 7 days, the filterability worsened almost linearly with time through 25 days of digestion. Again, the region of best filterability during the digestion process was observed to coincide with the peak of oxygen utilization for the system (Figure 22).

Through 18 days of digestion, the specific resistance of the mixture of primary and trickling filter sludges exhibited changes similar to those of trickling filter sludge 3 (Figure 26). However, after 18 days the filterability of the sludge improved linearly through 30 days digestion. For this sludge the period of minimum specific resistance during the digestion process did not coincide with the peak of oxygen utilization. This was probably due to the fact that the point of maximum oxygen exertion was more dependent upon satisfaction of primary and secondary BOD_5 than energy for bacterial cell systhesis.

It should be noted that the relative filterability of the sludges previously described was poor, as indicated by the very large units of specific resistance ($m/Kg \times 10^{14}$). In no case was a readily disposable filter cake formed during filtration. In only two cases were more than 65 ml filtrate recovered from a 100 ml sample during the 30 minute filtration process.

However, during the digestion of the waste activated sludge as much as 89 ml filtrate was recovered in 15 minutes of vacuum filtration for the same sample size. As would be expected, this resulted in the formation of a firm filter cake and specific resistance values on the order of 20 to 50 times less than those obtained for any of the other sludges (Figure 27). During the digestion period, the specific resistance increased initially, decreased to a low value at approximately five days, and then increased almost linearly through the end of the period. The magnitude and response of these specific resistance values correlate well with the results obtained by Rivera-Cordero (31).

COD

The chemical oxygen demand of the supernatant of trickling filter sludge 3 was plotted against digestion time in Figure 28. An almost linear reduction of 75% of the initial COD occurred during 11 days of digestion, after which no substantial decrease was noted. The COD:BOD ratio of undigested sludge was calculated to be approximately 2.0:1.0. After 11 days digestion this ratio had decreased to a minimum value of 1.3:1.0. Due primarily to the low BOD_5 values, the ratio increased to 5.0:1.0 at 25 days digestion and stabilized at that level through the remainder of the digestion period.

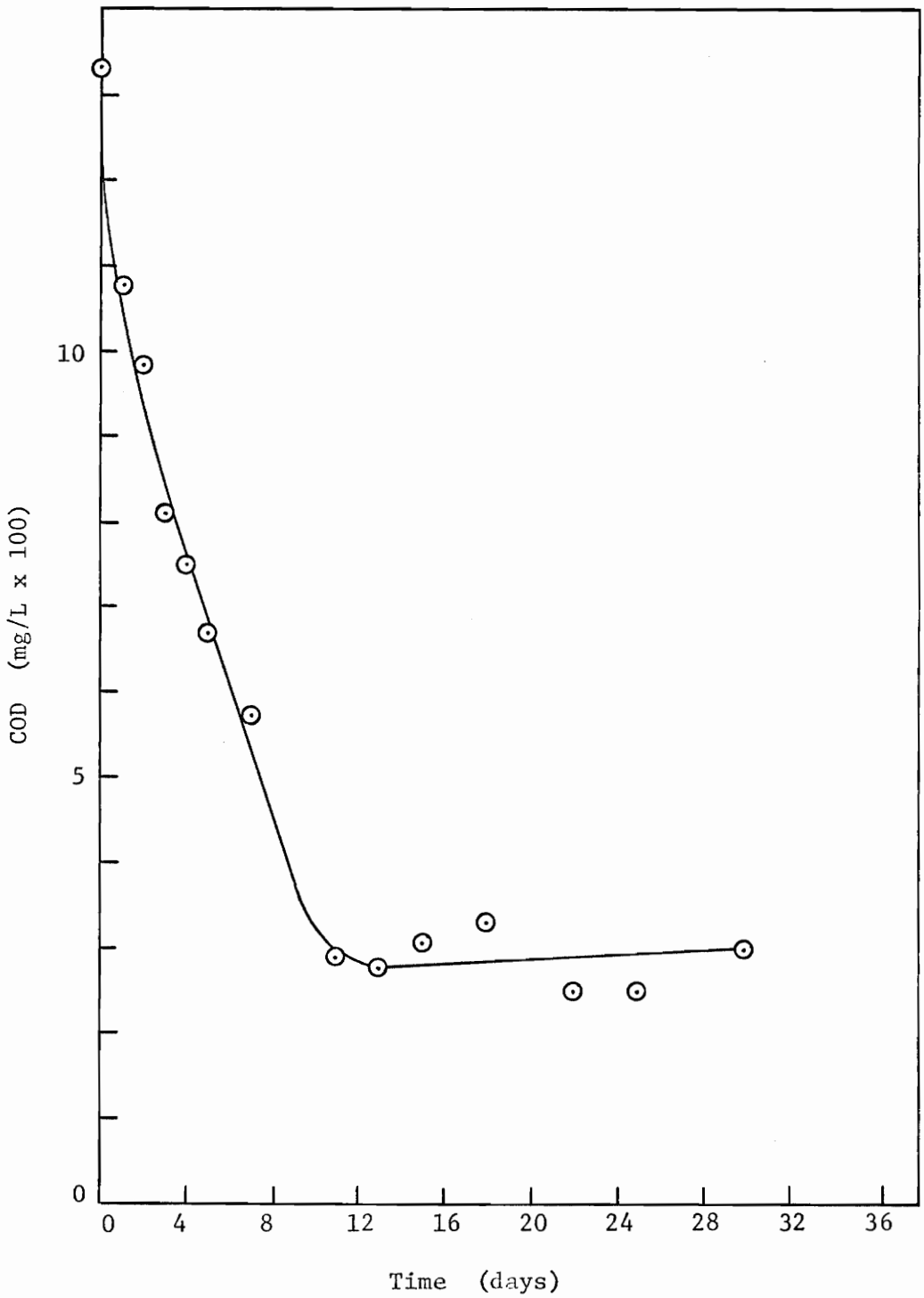

Figure 28: COD DATA FOR TRICKLING FILTER SLUDGE - DIGESTION RUN 3

Nitrification

Results from the nitrite-nitrate supernatant analyses for trickling filter digestion run #3 are shown in Figure 29. The analyses indicated that a high degree of nitrification was achieved during the digestion process. Nitrite concentrations exceeding 3.0 mg/l NO_2 as N were noted at 7 days digestion. Nitrate concentrations approaching 300 mg/l NO_3 as N were observed after 36 days digestion. The magnitude of these releases are slightly more than those reported by other investigators (10, 18, 23).

Total Carbon

Figure 30 shows the variation in total carbon content for the supernatant of trickling sludge 3. A maximum carbon content for the system was noted at 5 days digestion. Beyond this point, the carbon content dropped off rapidly through 11 days, after which endogenous respiration was established and the rate of carbon reduction slowed to a more stable level.

Frothing

Frothing occurred in all the digestion runs at various times, with no discernable pattern being established. When frothing occurred, the air rate was adjusted to help alleviate the problem and a glass cover was placed over the top of the digester to prevent loss of solids.

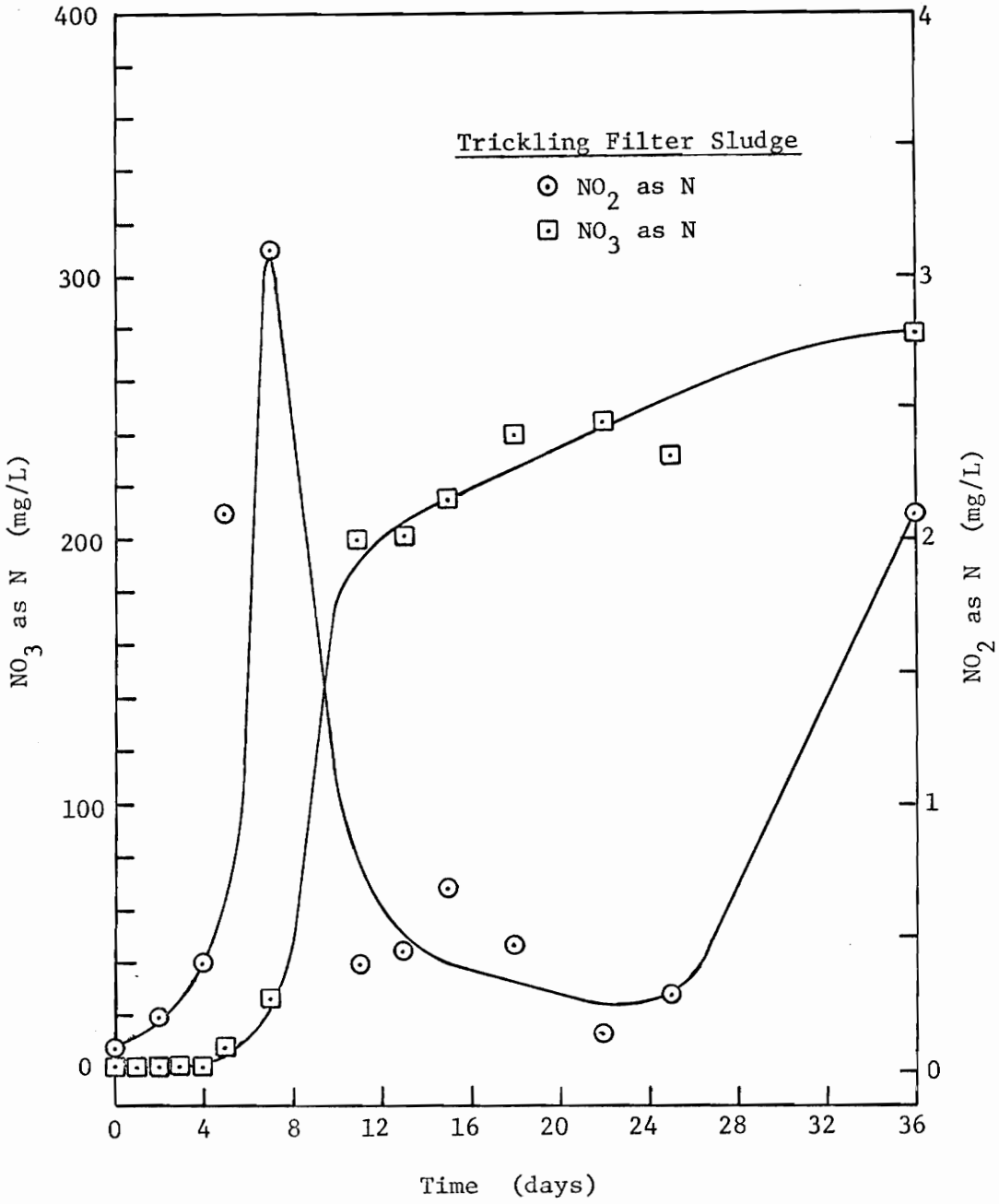

Figure 29: NITRIFICATION DATA FOR TRICKLING FILTER SLUDGE - DIGESTION RUN 3

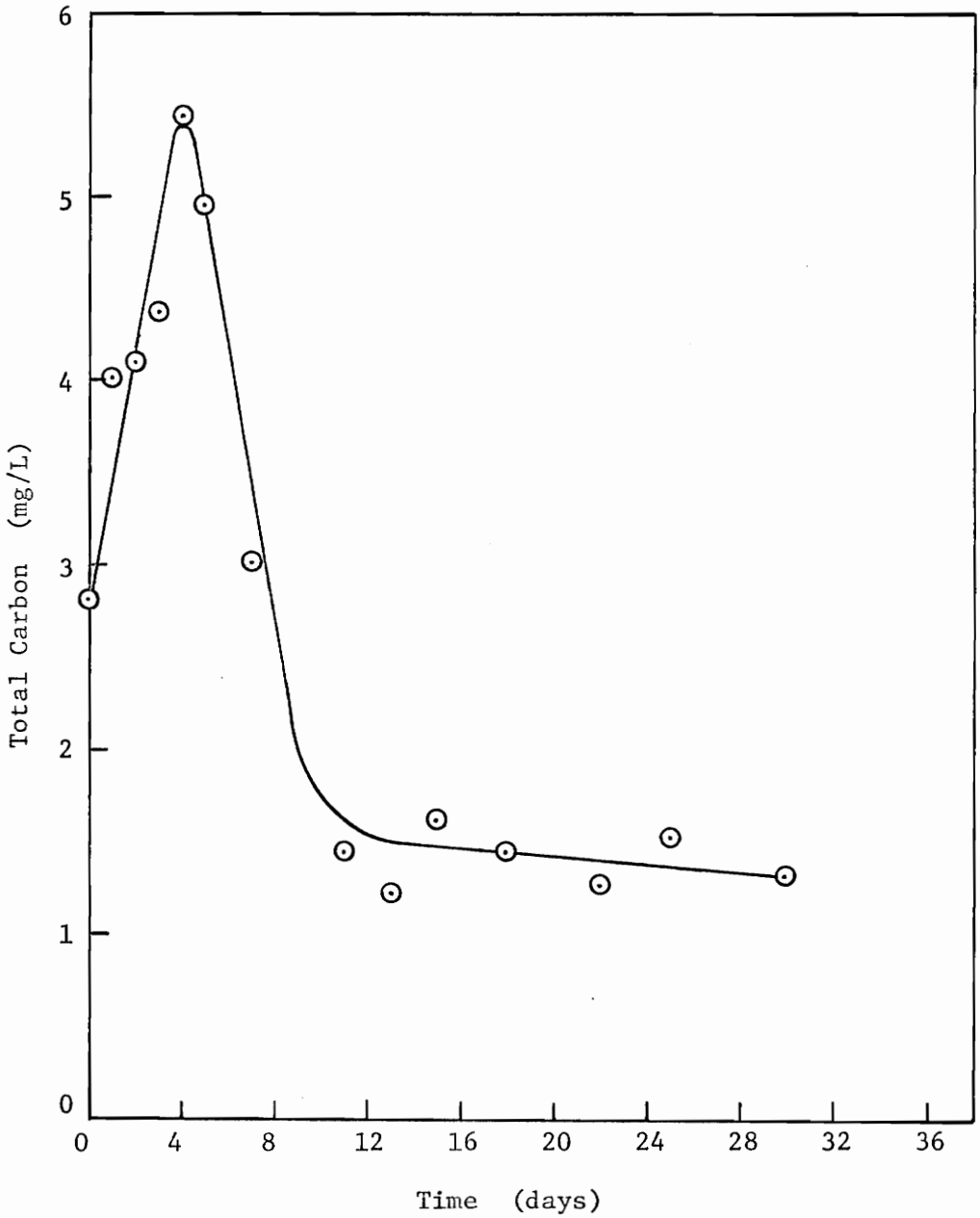

Figure 30: TOTAL CARBON DATA FOR TRICKLING FILTER SLUDGE - DIGESTION RUN 3

V. DISCUSSION OF RESULTS

As the purpose of this investigation was to study the aerobic digestion of trickling filter humus and the relationship between digestion and the subsequent filterability of the sludge, the discussion of the results of this experimentation will be presented primarily in terms of those objectives.

Perhaps one of the most obvious, but yet most important observations made during the course of this investigation was one based on the interpretation of the oxygen uptake data for trickling filter digestion runs #2 and 3 (Figure 22). This data indicates that the microorganisms associated with the biological trickling filter floc do, in fact, enter the endogenous growth phase. This fact is then the 'cornerstone' to any premise that trickling filter humus can be effectively stabilized through the aerobic digestion process.

The pathway by which the trickling filter humus enters the endogenous growth phase would seem more involved than that observed for waste activated sludge. This difference is undoubtedly due to the more varied biological composition of the humus. At the time it is 'sloughed' from the trickling filter media, the humus is rather unstable, as it contains significant populations of anaerobic, aerobic, and facultative bacteria, fungi, and protozoans existing in various conditions of anaerobic, endogenous, and aerobic growth. Consequently, it seems reasonable to assume that before endogeny can be established in the digestion system, a population shift away from the anaerobic

bacteria towards the facultative heterotrophs must be completed. For waste activated sludge, such an adjustment would not be required as the sludge floc constituents are essentially aerobic in nature. These facts suggest that trickling filter humus should require a longer period of metabolic activity before endogeny is established.

The results of this experimentation show that to be the case. During the aerobic digestion of trickling filter sludges 1 and 2, endogeny was not established until approximately 5 and 12 days, respectively. The 12 day period prior to the establishment of the endogenous phase for sludge 3 was felt to be unusually long and was probably due to the extremely unstable nature of the sludge resulting from a significant increase in hydraulic loading to the trickling filter prior to the time of procurement of the sludge (see Chapter III). The waste activated sludge (digestion run #4) appeared to be in a state of endogenous respiration throughout the entire digestion period. The quantity of oxygen consumed prior to the establishment of the endogenous phase indicated that significantly higher rates of metabolic activity occurred in the trickling filter humus. The peak in the oxygen utilization for the trickling filter sludges was generally about 3 to 5 times greater than that observed for the waste activated sludges both in this study (Figure 23) and by McDowell, et al. (21). As initial BOD_5 values for the trickling filter humus and waste activated sludge were not significantly different (Figures 20 and 21), indications were that the greater amount of metabolic activity in the trickling filter humus was not due to the presence of more initial avail-

able food substrate in the system, but rather to a population shift caused by the death of the anaerobic organisms. As would be expected, after endogeny had been established in the trickling filter humus, the rate of oxygen utilization was found to be comparable to that observed for the waste activated sludge.

The reductions in total and volatile suspended solids concentrations are generally considered the most important parameters for evaluating any sludge stabilization process. During the course of this investigation, significant solids reductions were noted for all three trickling filter sludges (Figures 5 to 7). However, due to the unusual extent of solids destruction observed for sludge 1 (i.e. virtually all volatile suspended solids were destroyed through 22 days of aeration), sludges 2 and 3 were considered to exhibit more typical results for solids discussion purposes. The solids reduction data for digestion run #1 will be discussed separately from these other trickling filter sludges.

The relative rates of solids destruction were observed to be greatest through about 12 days of aeration. After that time, substantial solids destruction continued to occur, but at rates which were generally one-third less than those obtained during the first 12 days. As these runs were batch-type in nature, it seems obvious that the decline in the rate of solids destruction was primarily related to the depletion of available nutrients (i.e. available carbon) in the system. This fact is well substantiated by the supernatant total carbon data for sludge 3 (Figure 30). Although it could

not be definitely determined whether the rates of solids destruction were particularly inhibited by the low pH which prevailed throughout much of the digestion period, it was noted that the change in the rate of solids destruction for sludges 2 and 3 approximately coincided with the time at which mixed liquor pH had dropped to the lower levels observed during much of the digestion period (Figure 13). Although this suggests that pH may have possibly affected the rate of solids reduction, in reality, the change in pH was probably more of a result of the near completion of stabilization than a factor affecting it.

Metcalf and Eddy, Inc. (22) have reported that about 18 to 22 days is considered the optimum digestion period for the stabilization of trickling filter humus. On a solids reduction basis, the results of this investigation showed that time period to be a fairly good approximation. Through 12, 22, and 36 days aeration, total solids reductions for sludges 2 and 3 were computed to average 28%, 36%, and 42%, respectively. Corresponding volatile solids reductions approached 32%, 42%, and 47%. Although solids reductions are shown to continue beyond a 22 day digestion period, the cost of aeration and required digester capacity may prove to negate the benefits of additional stabilization. The magnitude of the solids reductions observed for the trickling filter sludges compared well with those obtained by other investigators (7, 16, 18, 21, 23, 25) during aerobic digestion studies of waste activated sludge and are also generally comparable with solids reductions usually obtained during a similar period of anaerobic digestion of biological sludges (22).

As has been previously mentioned, an unusually high rate of volatile solids reduction (98% at 22 days) was observed for trickling filter sludge 1. Total solids reductions were also high (67% at 22 days). The reason for this peculiar response may be related to the low percentage of volatile matter noted in this sludge at the time of its procurement (see Chapter III). It is theorized that the normal aerobic and facultative bacteria population dominance in the humus had been upset by large storm flows which had 'washed-through' the trickling filter unit. Consequently it is theorized that larger populations of microorganisms which were more readily equipped to handle such an environmental stress (e.g. fungi) may have been present in the humus and when pH conditions in the digestion system dropped (at about 12 days), their metabolic activity was stimulated. However, the available food substrate in the system had already been significantly depleted and consequently a highly competitive phase of endogenous respiration occurred, resulting in the near total disappearance of organics in the system.

The reduction in BOD_5 is another parameter which can be used to evaluate the effectiveness of the stabilization process. As would be expected, significant reductions in mixed liquor and supernatant BOD_5 exertions were observed during the aerobic digestion of all three trickling filter sludges (Figures 18 and 19). The reductions in mixed liquor BOD_5 were found to be almost linear through the digestion periods, with residual concentrations of less than 3400 mg/l and 1400 mg/l noted after 12 and 22 days digestion, respectively. Except in the case of sludge 3, where samples were filtered through an 8 micron

Millipore filter, significant daily variations were observed for the supernatant BOD_5 values throughout the course of this investigation. However, in general, residual supernatant BOD_5 exertions were less than 375 mg/l after 12 days. The average magnitude of the mixed liquor BOD_5 reductions (approximately 67% at 22 days) was slightly less than those obtained by other investigators (15, 23, 24, 25) during the aerobic digestion of waste activated sludge. Although supernatant BOD_5 values fluctuated greatly during the digestion period, in general, they were considerably less than those which might be expected during the anaerobic process (10).

The COD to BOD ratio for the supernatant of trickling filter sludge 3 decreased from 2.0:1.0 for the undigested sludge, to 1.0:1.0 at about 12 days, and then increased to values in excess of 5.0:1.0 after 30 days digestion. The rate of COD reduction was significant through 12 days aeration, after which essentially no further reductions occurred. It is not surprising that this point of maximum COD stabilization corresponds to the beginning of the endogenous phase, reduction in the rate of solids destruction, and stabilization of the pH at lower levels in the digestion system. All of these factors indicate that a degree of stabilization had been achieved in the digestion system by that time.

The high degree of nitrification observed during the course of supernatant analysis for trickling filter sludge 3 (Figure 33) further attests to the degree of stabilization achieved during the initial 12 days of digestion. Nitrate concentrations (NO_3as N) rose sharply from

about 30 mg/l after 7 days to 200 mg/l at about 12 days. The concentration further increased to a high of 280 mg/l after 36 days aeration. The magnitudes of these supernatant nitrate releases were surprising and generally much higher than those observed during the digestion of waste activated sludge.

The results of the vacuum filtration analyses (i.e. specific resistance data) for the three trickling filter sludges showed the filterability to be poor throughout each of the digestion periods (Figures 24 and 25). In no case was an easy to handle or readily disposable filter cake formed. The magnitude of the specific resistance values (m/Kg x 10^{14}) was surprising and generally some 20 to 50 times greater than those obtained for the waste activated sludge studied during this investigation (digestion run #4) and by Rivera-Cordero (31).

Although the individual filtration characteristics of trickling filter sludges 1, 2, and 3 differed somewhat, an initial or delayed period of improved filterability (i.e. specific resistance values at that time were either less than, or approximately equal to, those obtained for the undigested sludges) occurred during the first week of each digestion run. After that time, specific resistance values rose sharply and fluctuated at higher levels throughout the remainder of the digestion period. A similar trend was noted by Graves, et al. (13) from capillary suction time (C.S.T.) filtration data collected during a 12 day aerobic digestion study of trickling filter humus and by Rivera-Cordero (31) and Parker, et al. (26) during the aerobic diges-

tion of waste activated sludge.

Only in the case of trickling filter sludge 1 was a net increase in filterability noted throughout the digestion period. This was probably due primarily to the exceptionally high initial specific resistance value of this sludge rather than to marked filtration improvement during the digestion process. The high initial value may indicate that filamentous forms were present in significant numbers at the time of its procurement (see Chapter III). The sludge was also low in the percentage of volatile matter and the resulting high proportion of mineral constituents in the sludge, coupled with an undoubtedly low level of biological activity possibly contributed to the poor initial filterability of the humus.

The oxygen uptake data for trickling filter sludges 2 and 3 can be used in conjunction with the vacuum filtration data to explain the changes in the relative filterabilities of these sludges throughout their digestion periods. Since the efficiency of the filtration process depends primarily upon the degree of natural biological flocculation achieved by the sludge, it was not surprising to note that the maximum filterability of these sludges consistently coincided with their periods of highest biological activity and reactive capacity (i.e. periods of maximum rate of oxygen uptake). These periods of high microbial activity resulted in the production of extracellular polymers that aided flocculation and improved filterability. However, extensive endogenous respiration produced a sludge of low biological activity and reactive capacity and eventually destroyed the cellular integrity of the system. This led to poor bioflocculation and resulted

in poor filterability throughout much of the digestion period. In addition to the detrimental effect of prolonged endogenous respiration upon the filterability of the sludges, it was also felt that the seemingly vigorous rates of aeration applied during the course of this investigation may have produced velocity gradients in the digesters which could have resulted in the shearing or inhibition of biological flocculation.

During the digestion of trickling filter sludges 1 and 2, supernatant alkalinities were observed to be almost completely removed within 9 days from the start of aeration (Figure 16). These reductions were undoubtedly related to the degree of nitrification occurring in the digestion system and should prove important in terms of reducing the coagulant dose of metallic salts required to produce a desired yield during the vacuum filtration of the aerobically digested sludge.

Based on the results of the various analyses for trickling filter sludges 1, 2, and 3, it would appear that significant stabilization of the humus can be accomplished during a 12 day digestion period. However, as extending the detention period to 22 days will result in further substantial reductions of total and volatile suspended solids and BOD_5 concentrations without significantly impairing the already poor filterability of the digested sludge, such an extension would be recommended. Oxygen requirements would essentially be the same throughout the 22 day period as other investigators (22, 35) have reported the required rate of aeration to be more dependent upon maintaining the sludge solids in suspension than satisfying the oxy-

gen demand of the system.

Analysis of the oxygen uptake data for the primary-trickling filter sludge mixture (digestion run #5) indicated that although the sludge was only 30% biological humus by volume, a state of endogenous respiration was established rather quickly (within 10 days aeration) and maintained throughout much of the digestion period. Apparently the addition of large volumes of primary sludge to the biological system does not significantly retard the aerobic digestion process efficiency. That is, the metabolism of the raw organic solids occurs so rapidly that the overall digestion process is not significantly prolonged. Surprisingly, the peak in oxygen utilization was approximately equal in magnitude to that exerted by the purely biological trickling filter sludges. However, the primary-trickling filter mixture was observed to utilize significantly more oxygen during the endogenous phase. This was probably due to a high initial food to microorganism ratio (F:M) and the subsequent generation of new, highly reactive cells.

The rates of total and volatile suspended solids reductions for the primary-trickling filter sludge mixture were observed to be significant throughout the 38 day digestion period (Figure 8). This sludge did not exhibit a marked decrease in the rate of solids reduction corresponding to the initiation of endogenous respiration in the system and in general produced solids reductions in excess of those noted for trickling filter sludges 2 and 3. Due to the high buffering capacity

of the sludge, pH was not observed to drop off as quickly or completely as in the case of the trickling filter sludges. Consequently, it is theorized that conditions more favorable to the propagation of facultative bacteria (i.e. those microorganisms primarily responsible for active organic substrate utilization) were maintained throughout much more of the digestion period and resulted in no real significant population shift toward the dominance of the more pH resistant microorganisms (e.g. fungi).

Both mixed liquor and supernatant BOD_5 values were greatly reduced during the digestion process. Residual supernatant BOD_5 exertions were on the order of those obtained for the trickling filter sludges.

In general, reductions in solids and BOD_5 increased linearly throughout the digestion period. However, after 30 days of digestion, the sludge seemed well stabilized, as total and volatile solids reductions approached 57% and 67%, respectively, and residual mixed liquor and supernatant BOD_5 exertions had been reduced to 1700 mg/l and 300 mg/l, respectively. For the 22 day period determined to be optimum for the trickling filter sludges, solids and BOD_5 reductions for the primary-trickling filter sludge mixture were generally on the order of 10% less than those observed at 30 days.

Unfortunately, the filtration characteristics of the primary-trickling filter sludge mixture were strikingly similar to those exhibited by trickling filter sludges 2 and 3. However, the maximum activity of the system did not coincide with the maximum filterability.

Since the large volume of raw sludge had to be converted to microbial cells first it follows that bioflocculation could not occur until the cells began to enter the endogenous phase and extracellular polymers were being produced in significant concentrations. The filterability of the sludge then improved until either the polymer was overproduced or cellular destruction began.

The digestion of the waste activated sludge (digestion run #4) produced disappointingly poor results in terms of solids reductions and therefore its usefulness as a source of comparison with the trickling filter sludge results was diminished. A negative-type correlation did exist with the results of Rivera-Cordero (31). In a previous investigation, he experienced the same type of poor reduction with waste activated sludge obtained from the same treatment plant. During the course of digestion run #4, biological activity and solids reductions in the system were poor (less than 10% volatile solids reductions in 18 days aeration). Although significant mixed liquor BOD_5 reductions were noted during the digestion, the supernatant values remained essentially constant throughout the digestion period. These results coupled with the oxygen uptake data of this digestion run, seemingly indicated that the undigested sludge had already been well stabilized.

The specific resistance values calculated during the digestion of waste activated sludge were higher than those normally obtained during the digestion of waste activated sludge, but are in line with those reported by Rivera-Cordero (31) for the sludge obtained from

this same plant. This seemingly confirms the consistency of the vacuum filtration analysis procedures used during the course of this investigation.

Perhaps the most noticeable physical change occurring as a direct result of the aerobic digestion of all the sludges studied during the course of this investigation was in terms of odor. At the time of procurement and especially for the first few hours of digestion, extremely offensive odors were produced by all the sludges. However, after a short period of sludge adjustment (generally less than 6 hours from the time of initial aeration) no further offensive odors were noticed throughout the duration of any of the digestion periods.

Based on the observations made during the course of this investigation, it is felt that the aerobic digestion of trickling filter humus is a feasible process and further study into the optimization of various operating parameters should be undertaken. Possible areas for studies could encompass continous as well as batch fed systems and include:

(1) effect of pH,

(2) effect of solids loading,

(3) effect of temperature,

(4) effect of the rate of aeration.

Another area in which further study could prove exceedingly valuable would include methods or procedures for improving the filterability of the aerobically digested trickling filter humus.

VI. CONCLUSIONS

Based on the results obtained during the course of investigation, the following conclusions have been made:

1. Trickling filter humus can be effectively stabilized by the aerobic digestion process. The humus adapts to the process in a manner analogous to waste activated sludge.

2. Significant total and volatile suspended solids reductions can be obtained during the aerobic digestion of trickling filter sludges for digestion periods in excess of 22 days.

3. Mixed liquor and supernatant BOD_5 values are significantly reduced during the aerobic digestion process.

4. A high degree of nitrification occurs during the aerobic digestion of trickling filter humus. Consequently, high concentrations of nitrates build up in the digester supernatant.

5. The filtration characteristics of the aerobically digested trickling filter humus are poor compared to similarly treated waste activated sludge. However, they improve during the early stages of digestion and generally deteriorate with further digestion, just as waste activated sludge does. The efficiency of the filtration process appears to be directly related to the biological activity of the sludge at the time of filtration.

6. Mixtures of primary and trickling filter sludges can be effectively stabilized by the aerobic digestion process. Total and volatile

suspended solids and mixed liquor BOD_5 reductions in excess of 57%, 67%, and 70%, respectively, were noted through 30 days of digestion. The filtration characteristics of the sludge mixture were generally poor and strikingly similar to those obtained during the digestion of trickling filter humus.

7. No offensive odors accompanied any of the aerobically digested sludges.

8. Future studies concerning the aerobic digestion of trickling filter humus should be made. Objectives of these studies should include the specific effects of various operational parameters (e.g. solids loading, pH, temperature, aeration rates) as well as methods for improving the filterability of the digested sludge.

VII. SUMMARY

This investigation was initiated for the purpose of studying the aerobic digestion of trickling filter humus and the subsequent relationship between digestion and the filterability of the sludge. To accomplish these objectives, five individual batch-type digestion studies were completed. In each case, ten liters of sludge were subjected to aeration within a 20°C temperature controlled environment. However, no attempts to specifically control any other digestion parameters, or chemical additions to improve the filterability of the digested sludges, were made. The composition and length of these digestion runs were as follows:

Run #1	Trickling filter humus	22 days
Run #2	Trickling filter humus	36 days
Run #3	Trickling filter humus	36 days
Run #4	Waste activated sludge	22 days
Run #5	70% Primary - 30% Trickling filter sludge mixture	38 days

All primary sludge and trickling filter humus digested during this investigation were obtained from the Blacksburg - VPI Sanitation Authority sewage treatment plant. The waste activated sludge was obtained from a small package-type extended aeration plant near Radford, Virginia. The digestion of this sludge was initiated to provide a means of comparison with the results obtained during the di-

gestion of the trickling filter humus, as well as a source of correlation with much of the published literature concerning the aerobic digestion process.

Water quality analyses conducted during the course of the various digestion runs included:

(1) Suspended solids,
(2) pH,
(3) Alkalinity,
(4) BOD_5,
(5) Oxygen uptake,
(6) COD,
(7) Nitrification,
(8) Total carbon.

The filterability of the sludges was evaluated by the use of the concept of specific resistance.

The results of the investigation showed that trickling filter humus responds to the aerobic digestion process in a manner similar to that of waste activated sludge. Both total and volatile suspended solids reductions were significant during digestion of the biological humus (36% and 42%, respectively, after 22 days) and compared well with those reductions typically reported in the literature for both the aerobic digestion of waste activated sludge and the anaerobic digestion of biological sludges. Significant and consistent mixed liquor BOD_5 reductions were also observed and residual supernatant BOD_5 concentrations (less than 250 mg/l after 22 days) were found to be less than those generally obtained during the anaerobic digestion process. However, high supernatant nitrate concentrations were observed to build up rapidly during the digestion of trickling filter humus (200 mg/l NO_3 as N at 12 days).

The filtration characteristics of the aerobically digested trickling filter humus were found to be poor compared to those of waste activated sludge. A period of improved filterability was noted during the early stages of the digestion process. However, filtration generally deteriorated with periods of prolonged endogenous respiration. Consequently, it was concluded that the efficiency of the vacuum filtration process was directly related to the biological activity of the sludge at the time of filtration.

Results observed during the digestion of the primary-trickling filter sludge mixture indicated that the addition of a large volume of primary sludge to the biological system did not significantly retard the efficiency of the aerobic digestion process. Total and volatile suspended solids and mixed liquor BOD_5 reductions in excess of 57%, 67%, and 70%, respectively, were noted through 30 days of digestion. Unfortunately, the filtration characteristics of the sludge mixture were generally poor and strikingly similar to those observed for the purely biological trickling filter humus.

The digestion of the waste activated sludge produced poor results in terms of solids reductions and exhibited specific resistance values in excess of those normally reported for the studies of waste activated sludges under similar digestive conditions. Consequently, its usefulness as a source of comparison for the results obtained for the trickling filter humus was diminished.

In general, the aerobic digestion of trickling filter humus appeared to be effective, in that it produced a well stabilized, odor-

free end-product. However, it is recommended that further studies be initiated to optimize the process by determining the effects of specific operating parameters (e.g. solids loading, pH, temperature, rates of aeration) as well as developing procedures to improve the filterability of the digested humus.

VIII. SELECTED BIBLIOGRAPHY

1. Bacon, V. W., and Dalton, F. E., "Chicago Metro Sanitary District Makes No Little Plans." Public Works, 97, 70 (November 1966).

2. Barnhart, E. L., "Application of Aerobic Digestion to Industrial Waste Treatment." Proceedings 16th Industrial Waste Conference, Purdue University, 612-619 (1961).

3. Busch, A. W., Aerobic Biological Treatment of Waste Waters. Oligodynamics Press, Houston, Texas, 151-173 (1971).

4. Bruemmer, J. H., "Use of Oxygen in Sludge Stabilization." Proceedings 21st Industrial Waste Conference, Purdue University, 544-558 (1966).

5. Carpenter, W. L., and Blesser, R. O., "Aerobic Decomposition of Secondary Papermill Sludges." Proceedings 17th Industrial Waste Conference, Purdue University, 126-135 (1962).

6. Clark, J. W., and Viessman, W., Water Supply and Pollution Control. International Textbook Company, Scranton, Pennsylvania (1970).

7. Coackley, P., "Laboratory Scale Filtration Experiments and their Application to Sewage Sludge Dewatering." Biological Treatment of Sewage and Industrial Waste, 2, Reinhold Publishing Company, New York (1956).

8. Coackley, P., and Jones, B. R. S., "Vacuum Sludge Filtration: Interpretation of Results by the Concept of Specific Resistance." Sewage and Industrial Wastes, 28, 963-975 (August 1956).

9. Cook, E. E., Graves, Q. B., and Scott, D., "Detention Time and Aerobic Sludge Digestion." Public Works, 102, 69-72 (November 1971).

10. Drier, D. E., "Aerobic Digestion of Solids." Proceedings 18th Industrial Waste Conference, Purdue University, 123-140 (1963).

11. Eckenfelder, W. W., Jr., "Kinetics of Biological Oxidation." Biological Treatment of Sewage and Industrial Waste, 1, Reinhold Publishing Company, New York, New York, 18-34 (1956).

12. Genter, A. L., "Computing Coagulant Requirements in Sludge Conditioning." Transactions of the American Society of Civil Engineers, 3, 635-678 (1946).

13. Graves, Q. B., et al., "Aerobic Digestion of Organic Waste Sludge." U. S. Environmental Protection Agency Water Pollution Control Research Series Publication No. 17070 DAU (December 1971).

14. Hoover, S. R., and Porges, N., "Assimilation of a Dairy Waste by Activated Sludge." Sewage and Industrial Wastes, 24, 306-312 (March 1952).

15. Irgens, R. L., and Halvorson, H. O., "Removal of Plant Nutrients by Means of Aerobic Stabilization of Sludge." Applied Microbiology, 13, 373-385 (May 1965).

16. Jaworski, N., Lawton, G. W., Rohlich, G. A., "Aerobic Sludge Digestion." International Journal of Air and Water Pollution, 4, 106-114 (June 1961).

17. Kehr, Dietrick, "Aerobic Sludge Stabilization in Sewage Treatment Plants." Journal of the Water Pollution Control Federation, 38, 354-356 (March 1966).

18. Lawton, G. W., and Norman, J. D., "Aerobic Sludge Digestion Studies." Journal of the Water Pollution Control Federation, 36, 495-504 (April 1964).

19. Levis, C. D., Miller, M. R., and Vosburg, L. E., "Design and Operation Experience Using Turbine Dispersion for Aerobic Digestion." Journal of the Water Pollution Control Federation, 43, 417-421 (March 1971).

20. Malina, J. F., and Burton, H. N., "Aerobic Stabilization of Primary Wastewater Sludge." Proceedings 19th Industrial Waste Conference, Purdue University, 716-723 (1964).

21. McDowell, M. A., et al., "Activated Sludge Processing." U. S. Environmental Protection Agency Water Pollution Control Research Series Publication No. 17050 DNW (February 1972).

22. Metcalf and Eddy, Inc., Wastewater Engineering. McGraw-Hill Book Company, New York, New York, 608-613 (1972).

23. Moore, H. R., "The Effect of pH on Aerobic Sludge Digestion." Unpublished Master's Thesis, Virginia Polytechnic Institute and State University (1970).

24. Murphy, K. L., "Sludge Conditioning by Aeration." Unpublished Master's Thesis, University of Wisconsin (1959).

25. Norman, J. D., "Aerobic Digestion of Waste Activated Sludge." Unpublished Master's Thesis, University of Wisconsin (1960).

26. Parker, D. G., Randall, C. W., and King, P. H., "Activated Sludge Dewatering: Biological Conditioning for Improved Filterability." Paper presented at the 44th Annual Water Pollution Control Federation Conference, San Francisco, California (October 1971).

27. Randall, C. W., and Koch, C. T., "Dewatering Characteristics of Aerobically Digested Sludge." Journal of the Water Pollution Control Federation, 41, R215-R238 (May 1969).

28. Randall, C. W., Saunders, F. M., and King, P. H., "Biological and Chemical Changes in Activated Sludge During Aerobic Digestion." Paper presented at the 18th Southern Water Resources and Pollution Control Conference, North Carolina State University, Raleigh, North Carolina (April 1969).

29. Randall, C. W., Turpin, J. K., and King, P. H., "Activated Sludge Dewatering: Factors Affecting Drainability." Journal of the Water Pollution Control Federation, 43, 102-122 (January 1971).

30. Ritter, L. E., "Design and Operating Experiences Using Diffused Aeration for Sludge Digestion." Journal of the Water Pollution Control Federation, 42, 1782-1791 (October 1970).

31. Rivera-Cordero, A., "Mechanisms of Change in Activated Sludge Dewaterability During Aerobic Digestion." Unpublished Doctoral Dissertation, Virginia Polytechnic Institute and State University (1972).

32. Rudolphs, W., and Heukelekian, H., "Aerobic Sludge Digestion Studies." Industrial and Engineering Chemistry, 24, 1312-1315 (November 1932).

33. Standard Methods for the Examination of Water and Wastewater. 14th Edition, American Public Health Association, Inc., New York, New York (1972).

34. Tebbutt, T. H. Y., "Some Studies of Aerobic Digestion." Institute of Public Health Engineers Journal, 69, 105-121 (1970).

35. Tebbutt, T. H. T., "Further Studies of Aerobic Digestion." Institute of Public Health Engineers Journal, 69, 223-232 (1971).

36. Tebbutt, T. H. Y., "A Note on the Units of Specific Resistance to Filtration." Water Pollution Control, 69, 694-695 (1970).

37. Viraraghavan, T., "Digesting Sludge by Aeration." Water Works and Wastes Engineering, 3, 86-89 (September 1965).

IX. APPENDIX

SPECIFIC RESISTANCE FILTRATION DATA

Filtrate Volume (ml) for Digestion Run 1 - Trickling Filter Sludge

Digestion Time (days)	Filtration Time (min.) 1	2	4	5	6	8	10	12	15	20	25	30	C (%)	r (m/Kg x 10^{14})
0	7	9	11	13	14	16	19	21	23	28	31	35	1.56	2.13
1	25	--	--	49	--	--	63	--	72	75	77	78	1.47	0.28
2	23	30	41	45	49	55	59	64	69	78	82	85	1.49	0.32
3	10	15	20	23	25	29	33	37	40	47	52	58	1.48	0.78
4	7	9	14	16	18	21	25	27	30	35	39	43	1.28	1.44
5	5	6	10	12	13	17	19	21	26	30	35	39	1.23	1.43
7	4	5	9	10	12	14	16	19	21	26	31	34	1.12	2.07
9	3	5	9	11	13	15	18	21	25	29	34	39	1.05	1.44
11	3	6	11	13	15	17	21	23	27	31	37	41	1.01	1.48
13	5	9	13	15	17	19	22	25	29	35	40	45	1.08	1.28
15	5	10	12	14	17	20	23	26	30	37	43	49	1.08	0.92
19	5	8	13	14	16	20	24	26	30	38	45	50	1.00	0.95
22	4	9	15	17	19	23	26	29	33	41	47	53	0.52	1.86

APPENDIX (continued)

SPECIFIC RESISTANCE FILTRATION DATA

Filtrate Volume (ml) for Digestion Run 2 - Trickling Filter Sludge

Digestion Time (days)	Filtration Time (min) 1	2	3	4	5	6	7	8	10	12	15	18	20	22	25	28	30	C (%)	r (m/Kg x 10^{14})
0	9	12	15	18	20	21	23	25	27	30	34	37	39	41	45	47	48	1.92	0.84
1	15	19	23	26	29	30	32	34	37	39	42	46	47	49	51	54	56	1.82	0.96
2	15	19	23	26	29	31	34	36	40	43	48	52	54	56	59	61	63	1.67	0.56
3	12	15	19	23	25	27	29	31	35	38	41	45	47	49	51	54	55	1.51	0.88
4	5	7	8	10	11	12	13	15	17	19	21	23	24	25	27	29	30	1.57	2.56
5	4	6	7	9	11	12	13	14	16	18	21	22	24	25	27	29	30	1.50	2.44
7	4	6	8	9	11	12	13	15	17	18	21	22	24	25	27	29	30	1.44	2.46
9	2	4	6	7	9	11	--	12	14	16	18	20	21	23	25	26	28	1.36	2.58
11	2	3	5	6	7	8	9	10	11	13	15	17	18	19	21	23	25	1.34	3.20
13	2	4	5	6	8	10	--	11	14	16	18	20	21	23	24	25	26	1.31	2.82
14	2	4	5	7	8	9	10	11	13	15	17	18	19	21	22	24	25	1.25	3.07
18	1	3	5	7	8	9	10	11	14	15	17	19	21	22	24	26	27	1.23	2.67
22	1	3	5	7	9	10	11	12	14	16	18	21	23	24	26	28	29	1.10	2.31
25	1	3	5	6	8	9	10	12	14	15	18	20	22	23	25	27	28	1.14	2.34
30	1	3	5	6	7	9	10	11	13	15	18	20	22	23	25	27	29	1.12	1.91
33	2	4	6	7	9	11	12	13	15	17	19	21	23	25	27	29	30	1.15	2.18
36	1	3	5	6	7	9	11	12	13	15	18	20	22	23	26	28	--	1.00	2.25

APPENDIX (continued)

SPECIFIC RESISTANCE FILTRATION DATA

Filtrate Volume (ml) for Digestion Run 3 - Trickling Filter Sludge

Digestion Time (days)	Filtration Time (min) 1	2	3	4	5	6	7	8	10	12	15	18	20	22	25	28	30	C (%)	r (m/Kg x 10^{14})
0	9	15	20	24	27	--	33	35	39	45	58	52	55	58	61	64	65	2.04	0.42
1	2	4	5	6	7	--	8	9	11	12	13	15	16	17	18	19	20	1.96	3.78
2	3	5	7	8	9	10	11	12	14	15	17	18	19	20	21	22	22	2.10	3.01
3	4	7	9	10	11	--	13	13	15	16	17	19	20	21	22	23	23	1.87	3.72
4	10	13	15	17	19	--	22	23	25	26	28	30	31	32	34	35	36	1.80	2.31
5	15	21	--	27	29	32	33	35	38	40	44	47	49	51	53	55	56	1.81	0.98
7	15	21	25	27	30	33	35	36	40	43	47	51	53	54	57	59	60	1.67	0.70
11	9	11	14	16	18	19	21	22	23	26	27	30	31	32	34	35	37	1.60	2.10
13	3	5	--	6	--	7	--	8	9	11	13	15	17	18	19	20	21	1.53	2.37
15	2	3	5	6	7	8	9	10	11	13	15	17	18	19	20	22	23	1.47	3.08
18	4	6	7	8	9	--	10	11	13	15	17	19	20	21	23	25	26	1.46	3.62
22	1	2	4	5	6	7	8	9	10	12	14	16	17	19	20	22	23	1.41	2.00
25	3	5	6	7	8	9	--	11	12	13	15	17	18	19	20	21	22	1.35	4.91
30	3	--	5	6	7	8	9	10	11	13	15	16	17	18	20	21	22	1.31	4.66
36	3	5	6	--	--	7	8	9	11	12	14	16	17	19	20	21	22	1.30	3.59

APPENDIX (continued)

SPECIFIC RESISTANCE FILTRATION DATA

Filtrate Volume (ml) for Digestion Run 4 - Waste Activated Sludge

Digestion Time (days)	Filtration Time (min) 1	2	3	4	5	6	8	10	12	15	C (%)	r (m/Kg x 10^{14})
0	41	57	65	68	69	70	71	72	73	73	1.64	0.062
1	36	48	55	58	60	61	62	62	63	63	1.81	0.086
2	38	52	62	70	76	78	80	82	83	84	1.66	0.072
3	38	52	64	72	77	80	82	83	84	84	1.63	0.067
4	43	60	70	75	77	78	80	80	80	--	1.57	0.053
5	43	60	73	79	82	83	84	--	--	--	1.55	0.046
7	42	58	71	77	79	80	81	82	82	--	1.54	0.052
9	43	58	70	76	78	79	80	81	81	--	1.55	0.058
11	42	60	70	76	79	80	81	82	82	--	1.49	0.054
13	40	54	66	76	81	83	86	87	88	--	1.44	0.067
15	35	49	60	70	76	81	85	86	87	89	1.37	0.080
19	34	47	58	65	72	76	80	82	83	84	1.37	0.111
22	25	37	46	54	60	66	74	79	80	82	1.43	0.076

APPENDIX (continued)

SPECIFIC RESISTANCE FILTRATION DATA

Filtrate Volume (ml) for Digestion Run 5 - Primary-Trickling Filter Sludge Mixture

Digestion Time (days)	Filtration Time (min) 1	2	3	4	5	6	8	10	12	15	18	20	22	25	28	30	C (%)	r (m/Kg x 10^{14})
0	5	9	11	13	15	17	19	23	25	29	31	34	35	38	41	43	3.57	0.50
1	--	--	--	1	2	2	4	5	5	7	8	9	10	11	11	12	3.05	2.73
2	--	--	1	1	2	3	4	5	5	7	8	9	10	11	12	13	2.97	1.40
3	--	--	1	3	3	4	5	6	7	9	10	11	11	13	15	15	2.93	1.76
4	--	1	3	3	4	5	7	8	9	11	13	14	15	16	17	19	2.87	0.98
5	3	5	7	8	9	10	12	15	17	19	21	22	23	25	27	28	2.71	1.42
7	3	5	7	8	9	10	12	15	17	19	21	22	23	25	27	28	2.53	1.50
9	1	2	3	5	6	7	9	11	13	15	17	19	20	22	24	25	2.48	0.97
11	1	3	3	5	6	7	9	11	13	15	17	19	20	22	24	25	2.37	1.01
13	1	2	3	5	5	7	9	10	11	13	15	17	18	19	22	23	2.26	2.00
15	--	1	2	3	5	5	7	9	10	12	13	14	15	17	19	20	2.09	2.49
19	1	2	3	4	6	7	8	9	11	12	13	14	15	16	17	18	1.85	5.07
22	4	6	7	8	9	10	12	13	14	16	17	18	19	20	21	22	1.82	4.42
25	3	4	6	7	9	10	11	12	14	15	17	18	19	21	22	22	1.67	3.92
30	1	3	5	6	7	9	11	13	15	18	19	21	22	24	25	26	1.62	2.23

VITA

William Stephen Young was born in Altoona, Pennsylvania, on July 18, 1948. He was educated in the public schools of the Altoona School District and received his diploma from the Altoona Senior High School in June 1966. In September 1966, he matriculated to the Pennsylvania State University and received an Associate in Enginneering degree in Drafting and Design Technology in June 1968.

Upon graduation, he was employed as an Engineering Technician in the Industrial Engineering Division of the Eastman Kodak Company in Rochester, New York.

In September 1969, he returned to his studies at the Pennsylvania State University and received a Bachelor of Technology degree in Water Resources Engineering Technology in June 1971.

At that time, he was employed as a Sanitary Design Engineer for the firm of Rummel, Klepper and Kahl, Consulting Engineers in Baltimore, Maryland. He maintained this position through March 1972, and has since been in the process of fulfillment of the requirements for the degree of Master of Science in Sanitary Engineering at the Virginia Polytechnic Institute and State University.

During his graduate studies at VPI & SU he was married to the former Linda M. Schultz of East Greenville, Pennsylvania.

He is a student member of the Water Pollution Control Federation and the American Water Works Association. He is also a member of the

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W. Stephen Young

AEROBIC DIGESTION OF TRICKLING FILTER HUMUS

by

William Stephen Young

(ABSTRACT)

The purpose of this investigation was to study the aerobic digestion of trickling filter humus and the relationship between digestion and the subsequent filterability of the sludge. Five individual, ten liter, batch-type digestion studies were conducted at 20°C for detention periods ranging from 22 to 38 days. Various water quality parameters were monitored during the three digestion runs of trickling filter humus, one digestion run of a 70% primary and 30% trickling filter sludge (by volume), and one digestion run of waste activated sludge completed during this investigation.

Results of the investigation showed that trickling filter humus responds to the aerobic digestion process in a manner similar to that of waste activated sludge. Significant total and volatile solids reductions along with consistent mixed liquor BOD_5 reductions were noted for digestion periods of 22 days. Using the concept of specific resistance, the filtration characteristics of the aerobically digested humus were determined to be poorer than those usually obtained for aerobically digested waste activated sludge. The degree of filterability was observed to deteriorate during periods of prolonged endogenous respiration. The primary-trickling filter sludge mixture was observed to adapt to the aerobic digestion process in a manner strikingly similar to that of the purely biological trickling filter humus.