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# Optimal Procedures for the Processing of Waste Activated Sludge

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# PREFACE

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

The processing for final disposal of sludges that accumulate during the treatment of wastewater is recognized as one of the most critical and most vexing problems of water pollution control. While the problem has always been a difficult one, its dimensions have increased rapidly in recent years, and they promise to increase at an even faster rate in the near future. Rapid population and industrial growth, combined with more rigorous pollution control requirements, have greatly increased the use of highly efficient waste treatment processes, particularly those that utilize microbial systems. The biological or secondary treatment processes not only convert a higher fraction of the waste impurities into sludges, they also produce sludges that are more voluminous and more difficult to concentrate and dewater.

Not only are large quantities of difficult-to-handle solids produced during secondary and advanced waste treatment, but the cost of sludge handling and disposal frequently exceeds both the capital and the operating costs of any other process in the treatment plant. Although the sludge volume seldom exceeds 1 percent of the plant influent flow, studies indicate that sludge handling costs for activated sludge plants are typically greater than 40 percent of the total capital and operating costs, and may account for as much as 65 percent. Some of the proposed advanced waste treatment processes will produce sludge volumes that approach 10 percent of the influent flows. Clearly, the development of techniques to reduce the cost of sludge handling and disposal are needed to maintain the economic viability of our pollution abatement programs.

Investigators are in almost unanimous agreement that of all sludges produced during waste treatment, waste activated sludge is the most difficult to handle and dewater. It is frequently difficult to concentrate by either sedimentation or flotation and it can be exceedingly troublesome when mechanical dewatering devices are used. The problem is often accentuated by the fact that the waste activated sludge is processed by techniques that were developed for the handling of primary solids. For example, numerous reports show that activated sludge is particularly difficult to dewater by any technique after it has been anaerobically digested. Indications are that some of the other procedures now in general use may also be detrimental. On the other hand, some promising methods are beginning to find acceptance.

The intent of this investigation was to prepare a guide, based on experimental evidence, that could be used by the wastewater plant designer or operator to improve the efficiency of sludge processing and thereby reduce the costs. The central aims were to define sludge handling procedures that are destructive to activated sludge dewaterability, to explore techniques that can improve filterability, and to determine the mechanisms that control changes in sludge dewatering characteristics.

Clifford W. Randall

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# ABSTRACT

An extensive study of changes that occur in waste activated sludge when it is processed by commonly used handling procedures and the effect of these changes on subsequent dewatering of the waste sludge is described. The purpose of the study was to identify the types of conditions that make activated sludge more difficult to dewater and to formulate procedures that would optimize dewatering. The principal handling procedures studied were: anaerobic storage, chlorination, aerobic digestion, polymer conditioning, and combinations of these techniques. The effect of operational parameters on the results were also studied. These include solids concentration, temperature, mixing, dissolved oxygen concentration, aeration rate, and digester flow conditions.

Dewatering changes were measured by vacuum filterability, specific resistance, and compressibility. Mechanisms of dewaterability change were elucidated and found to be related to microbial viability and floc size. A procedure for measuring floc size was developed. Aerobic digestion was found to be a beneficial sludge conditioning technique, but it must be properly used or it, too, can be detrimental to dewatering.

# INTRODUCTION

All conventional waste treatment processes produce large quantities of waste material in the form of dilute solids mixtures known as sludge. These sludges vary in make-up and solids content depending upon the characteristics of the raw waste flow and the treatment process that produced the sludge. For example, primary sludge consists of solid sewage particles of a predominately organic nature which are removed from the waste flow by sedimentation. On the other hand, secondary sludge consists of excess microbial cells produced during the biological removal of soluble and colloidal organics that remain in the waste flow following sedimentation. Raw sludges of both types consist mostly of water with a solids content of only 0.5 to 5.0 percent depending upon the origin of the solids and the method of removal.

Generally, raw sludge must undergo further processing after separation to reduce the volume and stabilize the organic material before final disposal. Two of the major operations in the sequence of solids disposal are the processes of sludge digestion, or stabilization, and sludge dewatering. The respective objectives of these processes are the reduction in volume and organic content by biological decomposition of the putrescible matter to more stable material, and the reduction in volume and moisture content of the sludges by some form of liquid-solids separation. Although both of these processes can be adapted to the handling of primary, secondary, or tertiary sludges, as well as any mixture of the three, it is generally conceded that because of its unique chemical, physical, and biological characteristics, secondary or biological sludge is the most difficult and costly of the wastewater sludges to process and dispose of.

Traditionally, sludge stabilization has been accomplished by anaerobic digestion; i.e., in the absence of free oxygen by methane producing bacteria. This process produces a stable sludge but much of the organic material is solubilized and the resulting supernatant, which is very high in nutrients and organic material, must be further treated before disposal. For the sake of economics and convenience, sludge dewatering has been primarily achieved using gravity-type sand drying beds. However, these techniques were originally designed for relatively small quantities of primary solids. In addition to changes in sludge composition, the expansion of available technology, coupled with increases in waste flows, degree of required treatment, and construction and operating costs, have made it both technically and economically necessary to investigate the use of other solids disposal methods.

Aerobic digestion is a recent development that is a possible alternative to anaerobic stabilization. This method uses the continuous aeration of sludge to achieve solids reduction, usually by maintaining the aerobic organisms in the endogenous phase of respiration. During endogenous respiration cellular material is destroyed and thus, solids are reduced and stabilized. The resulting supernatant has less pollution potential than anaerobic supernatant, the sludge is particularly amenable to aerobic digestion because the required aerobic organisms are already developed, and the dewatering problems commonly associated with anaerobically digested activated sludge do not develop.

Several processes have been developed for sludge dewatering. Besides gravity sand bed drying, widely used methods include vacuum filtration, centrifugation, filter presses, and lagooning. Vacuum filtration and sand bed drying are the most widely used with vacuum filtration generally confined to larger plants. Both methods involve a filtering action with the pressure differential being provided by gravity in the sand bed and by mechanical vacuum in the vacuum filter.

Of the newer methods of sludge handling, aerobic digestion and vacuum filtration appear to be the most promising. Vacuum filtration is being used to dewater many types of sludges and has been shown to be effective in dewatering chemically conditioned activated sludge. Aerobic digestion is an effective method for reducing waste solids without contributing to the pollution load on the plant or destroying the filterability of biological sludges. Since neither process is complete within itself for sludge processing, the joint application of the two seems warranted and logical for many situations, particularly those involving waste activated sludge disposal. However, technical data are lacking for aerobic digestion, and this process is not likely to be routinely included in sludge disposal evaluations by designers until more complete information is obtained and disseminated.

This research project was designed to evaluate the effect of several commonly used sludge handling procedures on subsequent activated sludge dewatering properties. Emphasis was placed on experiments involving aerobic digestion, and the major thrust of the study was the investigation of relationships between aerobic digestion and sludge filterability. It is believed that the resulting information will provide a better understanding of mechanisms that affect activated sludge dewatering, and it is hoped that it will stimulate the modification of present sludge handling design and operation.

# LITERATURE REVIEW

Water in sludges may be thought of as existing in four different forms: Free or drainable, capillary, floc or particle moisture, and chemically bound water [1].* The most important property of a sludge with respect to dewatering and drying is its drainability; i.e., the total amount of free water that is present, and the rapidity with which it can be removed [2]. The rate of drainage, or filterability, is more important when a mechanical dewatering process such as vacuum filtration is used because the controlling factor is cake form rate. On the other hand, when sludge beds are used, the total amount of water that can be removed by drainage is more important because the evaporation rate controls the overall rate of drying. The amount of water that cannot be removed by drainage must be removed by the slower process of evaporation.

The difficulty of dewatering secondary or biological sludges, particularly waste activated sludge, has been emphasized by several investigators. From plant-scale experiments of the chemical conditioning and subsequent vacuum filtration of waste sludges, Morris concluded that anaerobically digested activated sludge caused the greatest difficulties in handling [3], Albertson and Guidi [4] reported that waste activated sludge is the most difficult sludge to centrifuge, and Burke [5] noted that anaerobically digested activated sludge is the most difficult sludge to condition with polymers. Experiments at Chicago's southwest plant with sludge concentration by flotation showed that performance could be improved by decreasing the fraction of activated sludge in the mixture [6].

Factors influencing the filterability of mixed primary and activated sludge have been studied at the British Water Pollution Research Laboratory [7]. These studies showed that the fraction of activated sludge in the mixture profoundly increases the coagulant demand following anaerobic digestion. The results also indicated that the filterability of activated sludge is worsened by the following process operating conditions: Low waste retention time, low dissolved oxygen (DO) levels, low temperatures, and high waste biochemical oxygen demands (BOD). Ettelt and Kennedy [6] have demonstrated that storage of thickened activated sludge prior to chemical conditioning decreases dewatering efficiency. Reaeration of the thickened sludge shortly before conditioning was beneficial.

### I. Nature of the Problem

Secondary sludges are more difficult to concentrate and dispose of than other

*References are on page 83.

sludges because they consist almost entirely of microorganisms grown during the biological stabilization of soluble organics. These organisms are extremely small individually and thus, have a high clogging potential. In addition, they contain water internally and flocculate into a structure that has a high affinity for water and retains it tenaciously. This structure also tends to compress under pressure and further increase clogging. Thus, biological sludges are characterized by their gelatinous nature, their large volume of bound water, and their resistance to dewatering. As noted by the previous references, these detrimental characteristics are apparently increased by anaerobic digestion, and they become particularly critical when anaerobic digestion is followed by mechanical dewatering processes. Experience shows that, of the biological sludges, activated sludge is the most variable in nature and typically, the most troublesome to dewater.

A few studies have found that some undigested and anaerobically digested activated sludges have excellent dewatering properties [8, 9, 10]. However, while sludges that dewater well do exist, the studies reported by Randall and Koch [8] show that there is considerable variation even in sludges that have been processed similarly. Unfortunately, the properties that determine whether activated sludge is difficult or easy to dewater, the variation of these properties during various conditioning processes, and the relationship between these properties and typical biological sludge parameters, are neither well known nor adequately understood.

**II. State of Knowledge**

Tenney, et al. [9] showed that the primary sludge property affecting filterability is dispersion or bacterial flocculation. The more disperse the organisms are, the greater the amount of chemical required for conditioning, or the poorer the filterability. They also concluded that extracellular polysaccharides have a pronounced affect on dispersion and hence filterability.

Randall, et al. [2] have demonstrated that dispersion effects also apply to the gravity dewatering of waste activated sludge on sand beds. They used mixed-liquor BOD as a measure of dispersion, and found that there is an optimum level for best flocculation. Values below or above this level result in decreased drainability.

Their study also showed that solids concentration is the primary factor that affects both total drainage and drainage rate, but that above concentrations of 2.6 percent solids, additional effect is practically negligible. The results further demonstrated that drainability cannot be predicted by solids concentration alone, but that several other factors including mixed-liquor BOD (dispersion), alkalinity, carbohydrate (both total and extracellular), cellular protein, and the presence of rigid-celled protozoa, may have a strong effect on drainability.

In a separate publication, Randall, et al. [11] showed that aerobic digestion initially improves sand bed drainability and that further digestion beyond an optimum point increases dispersion, and therefore, decreases drainability, particularly drainage rate.

### III. Summary of Studies

In summary, studies have shown that anaerobically digested waste activated sludges have very poor dewatering characteristics. The observations indicate that sludge properties change in the absence of oxygen and that for activated sludge these changes are detrimental to dewatering. The exact nature of the changes that occur is not known. Although no research has been conducted on the subject, it is also thought that grease detrimentally affects sludge dewatering [12]. Studies with aerobically digested activated sludges have shown that sand bed drainability is related to several sludge and supernatant constituents, and that aerobic digestion initially improves dewatering properties. However, if digestion is continued, the improvement is overcome by the effects of dispersion. The results indicate that all constituents of the mixed liquor that affect drainage rate do so because they affect dispersion. The effect of such constituents on total drainage appears to be related to their ability to bind water.

### IV. Aerobic vs. Anaerobic Digestion

Despite the problems associated with the anaerobic digestion of waste activated sludge, most municipal plants still utilize the process for that purpose because they were designed according to experience gained from plants handling only primary sludge. Further, few efforts have been made to change design practices by developing alternate sludge handling procedures. Clearly though, new methods, or at least new modifications, must be developed if municipal plants are to keep pace with increasing standards and requirements. Aerobic digestion appears to be a particularly attractive alternative for many applications. Although technical data concerning aerobic digestion are not plentiful, several important advantages that justify attention by researchers and design engineers have been revealed. The advantages most often claimed for aerobic digestion have been listed by R. S. Burd [13]:

1. A humus-like, biologically stable end product is produced.

2. The stable end product has no odors; therefore, simple land disposal, such as in lagoons, is feasible.

3. When compared with anaerobic digestion and other schemes, capital costs for an aerobic system are low.

4. Aerobically digested sludge usually has good dewatering characteristics. When applied to sand drying beds, it drains well and redries quickly if rained upon.

5. Volatile solids reduction equal to anaerobic digestion is possible with aerobic systems.

6. Supernatant liquors from aerobic digestion have a lower BOD and nutrient concentration than those from anaerobic digestion. Most tests indicated that BOD would be less than 100 ppm. This advantage is important because the efficiency of many treatment plants is reduced as a result of recycling high BOD supernatant liquors. This is also true of nutrient removal efficiencies.

7. There are fewer operational problems with aerobic digestion than with the more complex anaerobic form because the system is more stable. As a result, less skillful labor can be used to operate the facility.

8. In comparison with anaerobic digestion, more of the sludge's basic fertilizer values are recovered.

Burd further states that the major disadvantage associated with aerobic digestion is high power costs. This factor is responsible for high operating costs in comparison with anaerobic digestion. At small waste treatment plants, the power costs may not be significant but they would be at large plants. This disadvantage is accentuated by the lack of methane gas production. Other disadvantages mentioned in Burd's report are:

1. Variable solids reduction efficiency with varying temperature changes.

2. Aerobically digested sludge does not always settle well in

subsequent thickening processes. This situation leads to a thickening tank decant having a high solids concentration.

3. Some types of sludge apparently do not dewater easily by vacuum filtration after being digested aerobically.

Burd concluded that, "While there is a difference in emphasis at municipal waste treatment plants as regards costs, it seems logical that aerobic digestion should be further evaluated, particularly for activated sludge facilities. Certainly more technical information on aerobic digestion is needed for a proper evaluation of the process. Acquiring additional data from existing systems would be a desirable first step. A considerable amount of new research in the process and in engineering design should be accomplished to improve existing technology. Very little information is currently available concerning loading rates, air requirements, rate of sludge oxidation, the effects of varying sludge characteristics, sludge amenability to subsequent handling and disposal steps, and cost-performance. Aerobic digestion will not be routinely included in sludge treatment evaluations by consulting engineers until more data are collected and disseminated."

### V. Filterability and Compressibility

A very important requirement for the development of improved methods of sludge disposal is a method of quantitatively describing the sludge dewatering properties in terms independent of the conditions of the measurement. A review of recent literature in the area of sludge conditioning shows that a variety of ways have been used to report filterability. Tenney and Cole [14] reported results in terms of time required to collect a given volume of filtrate using a Buchner funnel and specified pressures. They also expressed results as filtrate yield; i.e., the total volume of water collected during the filtration of a 200 ml sludge sample, as percent filtrate yield, which is that percentage of the original volume collected as filtrate, as thickness of the resulting filter cake, and as percent cake moisture. Scott and Cornwell [15] reported their sludge filterability results in terms of specific resistance using the filter leaf test, and as filter yield. Randall, Moore, and King [16] expressed filterability in terms of filtration rate. Several other methods have also been used to express sludge filterability and dewatering and this has complicated the comparison of experimental results.

Of the previously mentioned parameters, the concept of specific resistance has been the most widely accepted concept for describing the drainage of sludge. Developed from work by Carman [17, 18, 19] which was later refined by Coackley and Jones [20], it has proven to be a good comparative method for describing the dewatering of a compressible material by means of a vacuum filtration unit.

Particle compressibility is another important factor affecting sludge dewatering. Particles of compressible sludge tend to deform as pressure increases and the result is a tighter filter cake. For a given sludge, compressibility can be expressed as a constant function of pressure. When the factor of compressibility is 1.0, it means that the filtration rate will remain the same when the filtration pressure is increased. Sanders [21] and Nebiker, Sanders, and Adrian [22] have developed an equation for the determination of drainage time using the compressibility factor as a parameter.

# PROCEDURES

The experimental work conducted during this investigation was very extensive, consisting of two separate phases performed during two periods of time. During the first phase of this project, which encompassed the first year, a series of laboratory studies was conducted to evaluate the effect of typical waste treatment plant sludge handling procedures on the supernatant quality and the subsequent filterability of waste activated sludge. Areas of investigation included aerobic digestion, anaerobic conditions, mixed-liquor dissolved oxygen levels, mixing effects, temperature effects (including bacteriostatic conditions), sludge chlorination, suspended solids concentration effects, pH effects, and batch aerobic digestion versus continuous aerobic digestion. A further aim of the research was to identify the chemical and biological properties of sludge that affect filterability and the experiments were designed to yield such information.

The waste activated sludge samples used for all experiments during this phase were obtained from a 6000 gallon per day extended aeration waste treatment plant treating domestic sewage. The plant had a comminuter but no primary settling. Aeration was by means of a surface aerator and the sludge was recycled to the aeration tank by an airlift pump. The sludge samples were collected at the outlet of the return sludge line and transported directly to the laboratory. At the laboratory, the sludge was either used immediately or stored at 4°C for future use. Sludge concentrations used for the experiments were usually between 1.5 and 2.0 percent. Concentration of the sludge solids was accomplished by allowing the sludge to settle and removing the required amount of supernatant.

The experiments during the second year of the project were designed to define as completely as possible the mechanisms responsible for changes that occur in activated sludge dewaterability during aerobic digestion, and to evaluate the effects of the changes on subsequent polyelectrolyte conditioning. For this portion of the study, the waste activated sludge samples were obtained from the return sludge lines of three different full-scale waste treatment plants instead of one. Two separate extended aeration plants, the one utilized during the first phase and a 24,000 gallon per day unit, and a 10 MGD conventional activated sludge plant were sampled. Each sludge was simultaneously studied during most of the experiments, thus defining effects resulting from the variability of the sludges. Laboratory preparation of the samples was similar to that used during the first phase.

## I. Experimental Procedures

### a. First Year

Experimental design during the initial period of investigation consisted of separate experimental setups simulating the various methods of sludge handling under consideration. Most of the experiments involved aerobic digestion or were related to sludge aeration either before or after the handling procedure.

Aeration or aerobic digestion of sludge samples generally took place in what will be referred to as a standard aeration tube, shown in Figure 1. The unit illustrated is a test tube with a diameter of 2.5 inches and a volume of approximately 1.5 liters. Air was supplied through a I/8-inch inside diameter glass tube which extended to the bottom of the test tube. The air flow was metered at all times and maintained at a rate of 600 ml per minute unless otherwise specified. This air flow rate provided sufficient energy to keep solids in suspension, and enough air to maintain the dissolved oxygen level above 2 mg/l at all times. All aeration of sludge was done in a constant temperature room and except for temperature studies, the temperature was maintained at 20°C.

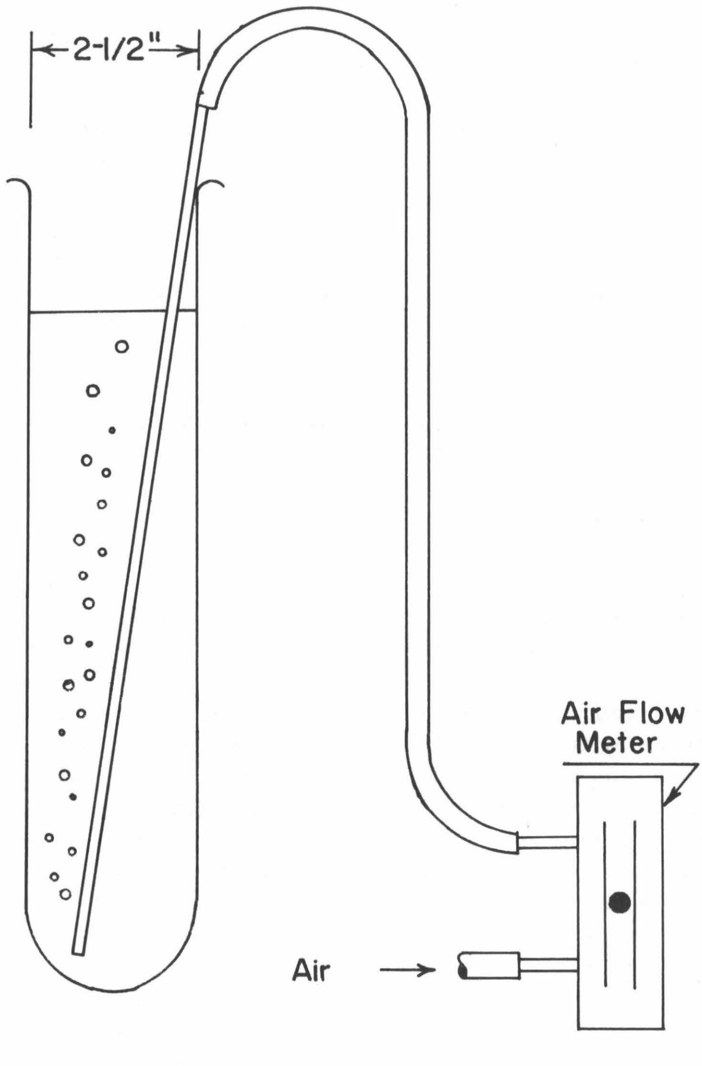
Each day before samples were removed, the sides of the aeration tubes were scraped and then rinsed with distilled water in order to resuspend any solids that had attached themselves to the sides. Any liquid lost by evaporation was replaced by adding distilled water to the aeration tubes to bring the level of liquid up to a mark made after the previous sample was removed.

The first experiment was designed to determine the general chemical and biological characteristics of the experimental activated sludge and to study the effect of aerobic digestion on these parameters. To accomplish this, a batch digestion experiment was conducted in which several sludge parameters were monitored throughout the aeration period. The factors studied during the experiment were suspended solids, BOD, chemical oxygen demand (COD), oxygen uptake, dehydrogenase enzyme activity, carbohydrate, protein, pH, and alkalinity. Filtration tests were conducted along with the other tests to determine what relationships exist between filtration rate and the other sludge characteristics.

For this initial experiment a large plexiglass cylinder was used as a batch digester so that a sufficient volume of sludge would be available for sampling. Aeration was provided through a perforated ring in the bottom of the

FIGURE 1

Standard Aeration Tube



cylinder. The air rate was sufficient to maintain a dissolved oxygen concentration above 6.0 mg/l and to provide constant mixing of the sludge contents throughout the aeration period.

Because the air flow and mixing rate in the large plexiglass cylinder could not be controlled, a second digester consisting of a standard aeration tube was run in parallel with the large digester. Filtration samples were taken from the standard aeration tube instead of the larger digester because it had been determined through preliminary study that the filtration characteristics of a digesting sludge can be affected by changing aeration and mixing conditions in the digester. It was assumed that the chemical and biological characteristics of the sludge in the two digesters were similar even though the physical conditions were different.

Three tests, the five-day BOD, the oxygen uptake, and the dehydrogenase enzyme test, were used to indicate the possible and actual biological activity of the sludge. The five-day BOD test was run on diluted samples of the sludge mixed liquor.

From a 0.45 micron millipore filter, the COD and TOC (total organic carbon) of the filtrate were measured to provide an estimate of the pollution potential of the dissolved material. Further, the carbohydrate and protein content of both the mixed liquor and the filtrate was monitored.

1. Zeta Potential and Specific Conductance: McKinney [23] has reported that a zeta potential below 20 to 30 millivolts is required for colloidal flocculation and that most bacteria have a potential below this range. Other investigators have indicated that zeta potential is of little consequence. An experiment designed to study the change in zeta potential and specific conductance with aerobic digestion was conducted. To study the zeta potential of the sludge under investigation and to determine how this parameter varies with time, the zeta potential, pH, specific conductance, and filtration rates were measured during a standard aeration tube run. Zeta potential and specific conductance were measured with a Zeta-Meter according to the instructions provided with the meter.

2. Dissolved Oxygen: The dissolved oxygen concentration has been found to have an effect on activated sludge bulking. For example, Bhatla [24] concluded that a high oxygen tension in the aeration tank could result in sludge bulking. It was felt that there was a possibility that the dissolved oxygen concentration in aerobic sludge digesters could have an effect on sludge dewatering. It has also been reported that sludge from the pure oxygen

activated sludge process settles better than conventional activated sludge [25], but this conclusion has been disputed [26].

Since aeration mixes the sludge and provides oxygen, a standard aeration tube arrangement was used that included mechanical mixing. Air rates in the various tubes remained constant throughout the study at 524, 148, 34 and 0 ml per minute. The mechanical mixing was varied to provide the same total power input in each tube. Aeration was continued for six days with periodic testing for dissolved oxygen and filtration rate conducted throughout the aeration time.

3. Aerobic and Anaerobic Sludge: While studying the effect of aeration rate and dissolved oxygen level on waste activated sludge filtration, it was found that zero dissolved oxygen caused a drastic degradation in filterability. From the results of the dissolved oxygen experiment it was not clear whether rapid mixing was causing the degradation in filtration by physically destroying the floc particles or whether the organisms themselves were reacting to the loss of oxygen in such a way as to cause the sludge to develop poor filtering characteristics. To further investigate the effects of zero dissolved oxygen, sludge was placed in two standard aeration tubes. One tube was aerated for one day and then aeration was stopped, no artificial mixing was provided. In this way, any degradation in filtering characteristics would be due to chemical and/or biological action and not to physical shearing forces caused by mixing. As a check on whether cellular material was staying intact, the total organic carbon concentration of the filtrate of each sludge was determined at the time that the filtration tests were run. Testing began when the aeration was stopped in the second tube.

4. Aerobic Conditioning of Anaerobic Sludge: Sludge may often be subjected to anaerobic conditions during normal operations in a sewage treatment plant. Such operations as pumping, settling, and storage cause sludge to become septic. The effect of lack of oxygen on the filterability of activated sludge was demonstrated by a previous experiment. In this experiment, the effect of aeration on sludge which had been subjected to anaerobic conditions was investigated.

Three standard aeration tubes were used in the study. Sludge was placed in all three tubes and one of the tubes was aerated. After two days had elapsed, aeration was started in the second tube, and after four days, aeration was started in the third tube.

5. Solids Concentration: It is well known that the suspended solids

concentration has an effect on sewage sludge dewatering [2, 20, 22, 27, 28]. There is also evidence that solids concentration has an effect on aerobic sludge digestion [11, 28]. Randall, et al. [11] reported in a laboratory study of aerobic digestion that the rate of solids destruction increased with the solids concentration, but that there were no significant differences in any of the measured parameters to account for the difference in solids destruction.

The purpose of this experiment was to determine the effect that suspended solids concentration has on the filtration rate of aerobically digested sewage sludge. Solids concentrations of 1.79, 1.59, 1.44 and 1.24 percent suspended solids were used. These concentrations were obtained by diluting settled sludge with the required amount of its own supernatant. The four concentrations were aerated in aeration tubes for 13 days. During aeration, the filtration rates and suspended solids concentrations were determined periodically.

6. Temperature: Temperature is a factor in all biological activity. Generally, the rate of biological reaction increases with increasing temperature within certain temperature ranges. Eckenfelder [29] has stated that the rate of aerobic biological reactions will increase with temperature to an optimum value of 30°C, but further increases in temperature result in a decrease in the respiration rate of mesophilic organisms. The magnitude of the temperature effect is, however, dependent on the nature of the process. Jaworski, et al. [28] found that temperature had an effect on aerobic digestion. In batch studies, it was found that volatile solids reduction during 6 days of aeration was increased from 28 percent at 15°C to 44 percent at 25°C. To test the effect of temperature on dewatering, sludge from the same batch was aerated at different temperatures. Temperatures were controlled at 20°, 25°, 30°, and 35°C by the use of a constant temperature room and water baths. The first series of filtration tests was conducted after the sludge was adjusted for the proper temperature, but before aeration was initiated. Due to the short time span between adjustment of temperature and filtration, the differing times were assumed to be a result of non-biological phenomena. In order to determine the effect of aeration on filtration time during this experiment, the test was repeated for 9 days on a batch of sludge which was a combination of the sludges aerated at 20° and 25°C.

7. Aeration at 6°C: The main purpose of aeration in aerobic treatment of sewage sludge is to provide oxygen to the aerobic organisms. This function generally is combined with the mixing of sludge. These operations can be combined in the form of subsurface diffused aeration, where the rising air causes the sludge to mix, or in the form of surface aeration where mechanical

mixing provides aeration. However, this combination of aeration and mixing causes other effects which may or may not be valuable in a particular treatment process.

In the case of sludge digestion, the mixing of sludge is good from the standpoint of biological growth, but may be bad from the standpoint of excessive shear forces which may tend to break up floc formations in the sludge. Further aeration provides oxygen to the organisms, but it also may tend to affect chemical reactions or act as a scrubber removing certain gases from solution. This experiment was designed to determine what non-biological effect aeration has on sludge digestion.

Three standard aeration tubes were filled with sludge from the same batch. Tubes A and B were aerated at a temperature of 19°C, while tube C was placed in a cold water bath and aerated at a temperature of 6°C. Aeration was continued for 22 hours, at which time tube B was also placed in the cold water bath. Aeration was continued for approximately seven more days. Cooling was used to inhibit biological activity without affecting chemical or physical characteristics other than those which are affected by temperature. In this way biological activity could be separated from the other factors affecting digestion.

Filtration tests were run on the sludges initially after placement of tube B in a cold water bath, and at other appropriate intervals throughout the experiment. TOC was also determined for the sludges at various intervals throughout the aeration period. The TOC of the filtrate was measured to detect any cell lysis or the release of any carbonaceous materials while the sludge cooled.

8. <u>Mixing Rate:</u> In order to further investigate the effects of mixing aerobic sludge on subsequent filtering, a mixing experiment was performed. Standard aeration tubes were used, but different air rates were employed to give different mixing characteristics. Rates of 2120, 1150, and 535 ml air per minute were used. Dissolved oxygen checks were made periodically and the dissolved oxygen level in all tubes was above 3.0 mg/l.

9. <u>pH:</u> The effect of pH on aerobic digestion has been reported in the literature. Lawton and Norman [10] reported that a pH as low as 5.0 did not appear to significantly affect digestion efficiency, and Moore [16] reported that solids reduction was relatively unaffected at a pH of 3.5. Moore did, however, find that filtration and settling characteristics were improved when the pH was held between 4.5 and 3.5.

Two experiments were run to test the effects of pH control on sludge filtration. The first experiment consisted of holding the pH of one tube of sludge at approximately pH 7.0, and allowing another tube to seek its own pH level. The pH in the controlled tube was allowed to rise originally, but was kept from falling below pH 7.0 by the addition of lime.

The second pH experiment was designed to investigate the findings of Moore that sludge which was aerated at a low controlled pH would filter better than uncontrolled sludge. Samples of the same sludge were placed in three aeration tubes. The first tube was uncontrolled, the second tube was controlled at pH 4.5, and the third tube was controlled at pH 3.5. The pH of each tube was checked daily and was controlled by the addition of concentrated sulfuric acid. The experiment was duplicated for increased reliability of the results.

10. Chlorination of Sludge: Chlorine is used in sewage treatment for a variety of purposes. In addition to disinfection, chlorine is used in sludge handling to improve sludge thickening and sedimentation, and to correct sludge bulking problems [30]. Heukelekian [31] studied the effects of chlorine on sludge bulking and bound water. He concluded that chlorination resulted in an immediate decrease in sludge volume index and bound water, and that the action was physical rather than biological.

The effect of chlorine on aerobic sludge filtration was studied by introducing concentrations of 250 mg/l and 750 mg/l calcium hypochlorite (HTH) with 70 percent available chlorine into two of three aeration tubes. After 15 minutes, the filtration tests were performed and a residual of 0.07 mg/l chlorine could be detected in the filtrate of the sludge with the original 750 mg/l HTH. After 18 hours, no chlorine could be detected in any of the three sludges. Filtration tests were conducted on all sludges for a period of 8 days.

In an attempt to determine how chlorine was affecting the sludge, another experiment was conducted. This time 500 mg/l HTH was mixed with the sludge, and after 15 minutes the filtrate of the sludge was collected and tested for TOC.

11. Batch Versus Continuous Digestion: Aerobic digesters can be operated on either a batch or a continuous basis. Batch digestion involves placing sludge in a digester, aerating the sludge for a specified length of time, removing the digested sludge and then repeating the procedure with a fresh batch of sludge. Continuous digestion usually involves aerating a completely mixed tank of sludge in which portions of digested sludge are periodically withdrawn and replaced with fresh sludge. There is some indication that the method of operation has an effect on digestion solids reduction efficiency [28].

To determine the relative effect of batch and continuous digesters on filtration, a comparative study was made. A batch digester was operated as a standard aeration tube. Sludge was placed in the tube, aerated, and samples were removed. No new sludge was introduced during digestion. The continuous feed digesters were standard aeration tubes also, but once a day a portion of the sludge was removed and the same quantity of fresh sludge was used to replace the quantity removed. The quantity removed each day was determined by the detention time desired. Detention time was calculated by dividing the total volume of sludge in the tube by the quantity of sludge replaced each day. Detention times tested were 3, 5, and 10 days. The sludge used throughout the experiment came from the same batch and was stored at 4°C until needed.

12. Dewatering by Sand Drying Bed: The most common method of sludge dewatering in small treatment plants is by sand drying beds. These beds employ two mechanisms of dewatering. First, water is drained from the sludge by gravity induced filtering using sand as the supporting filter medium Second, the remaining water is removed from the drained sludge by evaporation.

To measure the effect of aerobic digestion on sand bed sludge dewatering, a batch of aerated sludge was periodically dewatered by both the sand bed and vacuum filtration methods. To provide volumes of sludge required for the sand bed drainage test, a large plexiglass cylinder fitted with a perforated tube at the bottom to provide aeration was used as an aerobic digester. The vacuum filtration tests were conducted in accordance with the standard filtration procedures, but a special drainage cylinder was built to measure sand bed drainage.

b. Second Year

The experimental procedures used during the second year of the study were similar to those used during the first year. There were a few notable changes, however. For example, instead of standard aeration tubes, tubes with a capacity of 13.5 liters were used for the aerobic digestion experiments, and the experiments were generally run in triplicate with each sludge test sample being a composite from the three units. Also, instead of simple filtration tests, both specific resistance and the compressibility factor were measured for each sample with the exception of a few of the polyelectrolyte experiments. Further, two parameters not measured during the first year were added—natural exocellular polymer concentration and sludge floc median particle size.

The scope of the experimental procedures during this phase of the project may be divided into four general areas of investigation:

1. Measurement and comparison of chemical, physical, and biological changes that occur during the aerobic digestion of excess activated sludges from differing sources.

2. Measurement and comparison of changes in filtration characteristics that occur during the aerobic digestion of excess activated sludge from various sources.

3. Development and evaluation of a predictive test for sludge dewaterability.

4. Measurement of effects of snythetic polymer and chlorine on filtration characteristics of aerobically digested sludge at different stages of aeration.

Each sludge sample was aerobically digested for nine days or more, and the various parameters were measured at different intervals during the run and compared to the initial values and to the other sludges. Numerous experiments of this type were performed on each sludge to thoroughly define the significant aspects. Following these tests, the sludges were subjected to chlorination or polymer conditioning both prior to and during aerobic digestion, and the effect of the handling technique on specific resistance and compressibility was measured. A more detailed evaluation of the relationship between polymer addition and various handling procedures was then performed using five separate experiments, but only one sludge. Descriptions of the latter experiments follow.

Experiment One: Determination of Optimum Polymer and Optimum Polymer Dose

Prior to laboratory conditioning, each of the three sludges was treated with varying doses of each of four polymer solutions, two anionic and two cationic. The results were compared and the best polymer for the purpose was selected. In this experiment the vacuum was 30 cm Hg and times were recorded for up to 35 ml filtrate. From the data collected, it was possible to plot curves of time on the ordinate versus dosage on the abscissa, thereby obtaining a curve which could be used to find the optimum polymer dose for each polymer and each sludge.

Experiment Two: Extended Aeration

The purpose of this experiment was to see if aerobic digestion affects the polymer dosage required to produce good filtration rates. The two worst sludges from a conditioning standpoint were chosen for this experiment. Six liters of mixed liquor were placed in a plexiglass cylinder 14 cm in diameter and aerated using a 1/8-inch tube for a 40-day time period. Each sludge was aerated separately. Samples were taken from the cylinders initially and on days 1, 3, 4, 5, 6, 7, 8, 10, 12, 15, 19, 22, 26, 28, 31, 35, and 40 with two filtrations being run on each sample. One filtration was performed using no additive and another filtration was performed after adding the optimum dose of the selected cationic polymer. The time required to obtain a filtrate of 30 ml was recorded for all experiments except Sludge B with no additive; the time required to obtain 15 ml filtrate was used for that sludge because of the slow filtration rate.

Total solids were measured throughout the experiment, but pH and DO were not measured until the eleventh day. The reason for checking them at that time was to evaluate the extent of microbial activity and to see if pH was affecting filtration rates.

Experiment Three: Effects of Chlorination

Experiments were conducted to see if the addition of a heavy dose of chlorine would increase or decrease the specific resistance of the two sludges tested, and how this would affect polymer requirements. The type of chlorine used, HTH, was prepared in a 10,000 mg/l solution so that 10 ml of the solution would produce a concentration of 100 mg/l in a one liter volume. Polymer doses varied from 10 mg/l to 500 mg/l for Sludge A and from 100 mg/l to 800 mg/l for Sludge B. HTH concentrations of 0 mg/l, 100 mg/l, 250 mg/l, and 750 mg/l were added to both sludges and the resulting filtration rate was measured.

The results obtained from the sludge filtration rates upon the addition of polymer were used to obtain specific resistances at different polymer doses. These were compiled and plotted as specific resistance versus polymer dosage.

This series of tests was run in the constant temperature room utilizing 40 cm Hg of vacuum and completed all in one day so there was no influence on the resulting data from a storage period.

Experiment Four: Effect of Anaerobic Conditions

The purpose of this experiment was to see how anaerobic conditions affect the filtration rate of activated sludge and also to see how the optimum polymer dose varies with time of anaerobiosis. Data were collected initially, and 3 hours, 1 day, 2 days, and 4 days after anaerobiosis had begun. Polymer (Hercofloc 810) doses varied from 50 mg/l to 500 mg/l for Sludge A and from 100 mg/l to 700 mg/l for Sludge B.

For this experiment, total solids was the only variable measured. The filtration rates were determined using a vacuum of 40 cm Hg. Curves of specific resistance versus polymer dose were plotted from these data so that optimum polymer doses could be determined.

Experiment Five: Variation of Optimum Dose with Aeration Time

This experiment was set up to measure the variation in the optimum polymer dose with aerobic digestion time. In this experiment, two polyelectrolytes, Herofloc 810 (a cationic polymer) and Hercofloc 816 (an anionic polymer) were used. Optimum doses of the polymers were obtained initially and at 3, 5, 7, 10, and 20 days after aeration began.

The optimum doses were determined using the time to obtain 40 ml filtrate for the cationic Hercofloc 810 and only 10 ml filtrate for the anionic Hercofloc 816 treated sludge. These times were divided by the percent solids content to make the results more comparable between samples. These filtrations were run using a vacuum of 40 cm Hg.

Cationic polymer doses were varied from 25 mg/l to 250 mg/l, and anionic polymer doses were varied from 100 mg/l to 300 mg/l.

**II. Analytical Procedures**

The analytical testing procedures were chosen so that the tests measured the desired factors, gave consistent results, were reproducible, and were easy to run. With these criteria in mind, the tests chosen were not always in accordance with standard procedures. However, they were comparable to standard tests, and the results can be reproduced using the given descriptions. In general, the same analytical procedures were used during both years of the study. The main difference during the second year was the adoption of a more rigorous technique for measuring sludge dewatering properties. There were also some differences in the tests that were performed on the sludge and and sludge filtrate.

a. First Year

Sludge and filtrate changes were detected by measuring pH, COD, BOD, TOC, oxygen uptake, dehydrogenase enzyme activity, zeta potential, specific conductance, extracellular carbohydrates, cellular protein, and suspended solids.

The pH was determined by using a Leeds and Northrup stabilized pH meter assembly in conjunction with Leeds and Northrup glass-calomel electrodes. Before pH determinations, the meter was standardized against standard buffer solutions by Fisher Scientific.

Dissolved oxygen concentrations were determined using a PCL Dissolved Oxygen meter. The PCL Analyzer was calibrated at the temperature of the sample before each set of dissolved oxygen measurements was taken.

Oxygen uptake rates were measured by placing sludge in a standard BOD bottle which contained a magnetic stirring bar. The bottle was sealed with a self-stirring dissolved oxygen probe and the contents of the bottle were stirred while dissolved oxygen readings were recorded at periodic time intervals. The oxygen uptake rate was found by calculating the slope of the plot of dissolved oxygen with time. The rate was given in terms of mg/l DO depleted in one hour.

Total organic carbon was determined using a Beckman model 915 Total Organic Carbon Analyzer. Twenty-microliter samples were injected. Total carbon and inorganic carbon were determined, and the total organic carbon was found by subtraction. Samples were first filtered through a 0.45 micron millipore filter to remove all solids.

Zeta potential and specific conductance were measured with a Zeta-Meter and electrophoresis cell. Measurements were taken using the 8X objective.

Total available chlorine determinations were made using a Hach Model DR-EL Portable Water Engineer's Laboratory. The procedure used was basically an orthotolidine method.

All total and volatile suspended solids concentrations were determined by the Gooch crucible method as described in Standard Methods for the Examination of Water and Wastewater, [32]. Sample volumes used were 10 ml. Because of the high suspended solids, the drying time was 24 hours at 103°C, and the ignition time was 1 hour at 600°C.

The BOD and COD determinations were made in accordance with the procedure outlined in Standard Methods for the Examination of Water and Wastewater and using the simplifications as outlined in Simplified Laboratory Procedures for Wastewater Examinations [33], where applicable. Dissolved oxygen measurements for the BOD test were made using a self-stirring dissolved oxygen probe.

The procedure outlined in Standard Methods for the Examination of Water and Wastewater was used for alkalinity determinations with the exception that samples consisted of 10 ml filtrate instead of the recommended 100 ml.

The dehydrogenase enzyme activity measurements were made using the method presented by Ford, Young, and Eckenfelder [34]. The test is based on the ability of aerobic dehydrogenase enzymes to pass electrons to certain reducible dyestuffs as well as to oxygen. A tetrazolium salt (triphenzltetrazolium chloride or TTC) is used as a hydrogen acceptor and the oxidation of substrate results in the reduced form, triphenyl formazan (TF), which has a red color. The intensity of the red color is taken as a measure of dehydrogenase activity.

The test was conducted by placing 5 ml sludge and 5 ml tris-buffer (pH 8.4) in each of four tubes, bringing the solution to 37°C in a water bath, placing 1 ml TTC in three of the four tubes and 1 ml distilled water in the one remaining tube, incubating the mixture for 15 minutes at 37°C, and stopping the reaction with 39 ml ethyl alcohol. The contents of the tubes were then centrifuged to remove the solids, and the percentage transmission, as compared to the tube with distilled water, was read on a Bausch and Lomb Model 20 spectrophotometer at a wave length of 490 m$\mu$. The concentrations were then determined by comparing the absorbance with a standard curve of TF concentrations and absorbance, and reported as TF concentration.

Carbohydrate concentrations were determined by the Anthrone method described by Ramanathan, Gaudy, and Cook [35]. The carbohydrate concentration was measured for the sludge mixed liquor and for the sludge filtrate after filtering through a 0.45 micron millipore filter. The analysis consisted of placing 3 ml appropriate dilutions of the sample to be tested and known concentrations of dextrose in test tubes, cooling the contents of the tubes in ice water, injecting 9 ml anthrone reagent, placing the tubes in boiling water for exactly 15 minutes, and cooling the contents to room temperature. The percent transmission for the samples and known standards was then determined with a spectrophotometer at a wave length of 540 m$\mu$. Concentrations of carbohydrate were determined by comparison with the known standards.

The Folin-Ciocalteu method as described by Ramanathan, Gaudy, and Cook [35] was used to determine protein concentrations. The protein concentrations of the sludge mixed liquor and the sludge filtrate were measured after filtering the sample through an 0.45 micron millipore filter. The analysis consisted of placing 1.20 ml of appropriate dilution of sample to be treated and known concentrations of bovine serum albumin in test tubes, adding 6 ml freshly prepared alkaline copper solution to each tube and mixing. After 10 minutes, 0.3 ml Folin-Ciocalteu reagent was added and mixed, and 30 minutes were allowed for development. The percent transmission of the samples and known standards was then measured by the spectrophotometer at a wave length of 500 m$\mu$. Concentrations of protein were determined by comparison with the known standards.

Buchner funnel techniques were used to measure changes in filterability. The filtration apparatus used consisted of a 9 cm diameter Buchner funnel with 1 piece of Whatman number 40 filter paper. The funnel was connected to a rubber stopper by means of a rubber tube. The stopper contained a second tube for vacuum application and was seated in a 100 ml graduated cylinder as shown in Figure 2. The test results were obtained in the following manner:

1. Filter paper was placed in the Buchner funnel and 75 ml distilled water was filtered to wet and seat the paper;

2. The pinch clamp was closed, the cylinder emptied and replaced, and the vacuum was adjusted to 12 in. mercury;

3. 100 ml sludge was placed in the funnel;

4. The pinch clamp was opened and a stop watch was started;

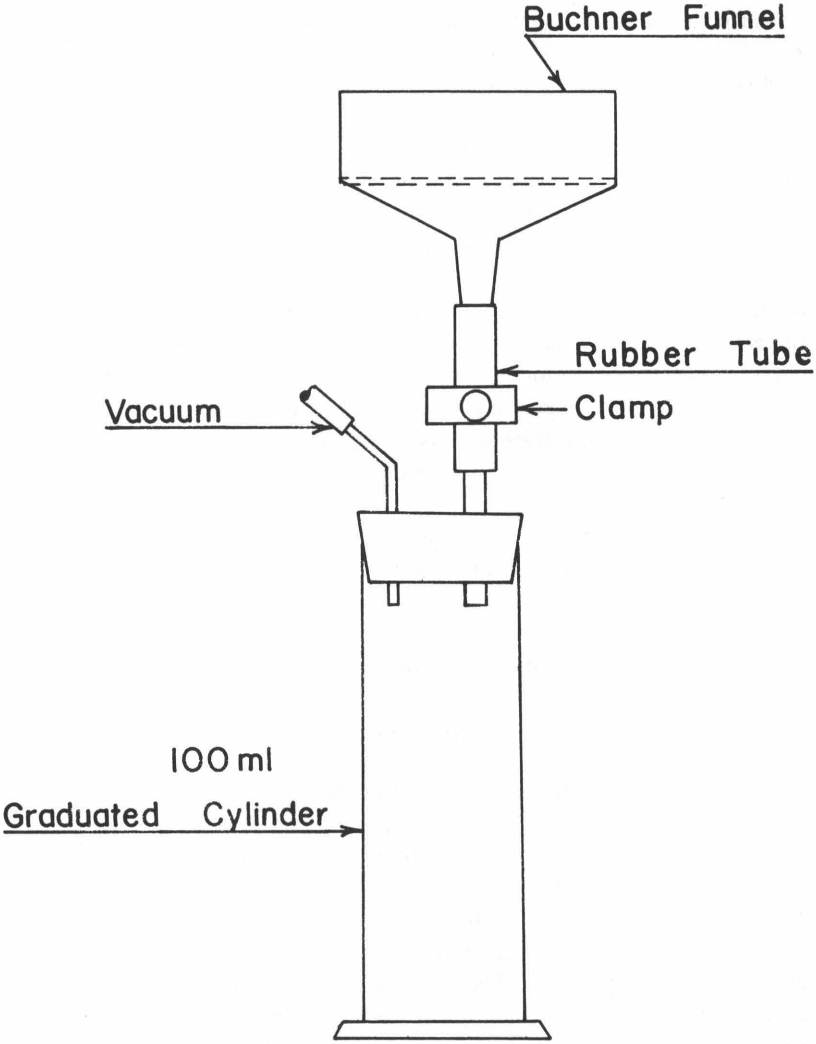
5. The time required to obtain 75 ml filtrate was recorded.

The filtration test used for this phase of the work was developed for its simplicity, its reproducibility, and its ability to be compared to the same test within a given experiment. The purpose of the test was to give an indication of the rate of filtration and it was used for comparison purposes within a given experiment to indicate the trends of filtering for a sludge subjected to various conditions.

Preliminary investigations were conducted to demonstrate that the specific

FIGURE 2

Vacuum Filtration Apparatus



resistance for a given sludge could be predicted by filtration time data from the test used in this investigation. Figure 3 shows the relationship between filtration time and specific resistance for a sludge which was aerated and tested over a nine-day period. This figure indicates a reasonable correlation between the two parameters.

A special sand drainage bed was constructed to test the effect of aerobic digestion on gravity dewatering. The sand bed consisted of a plexiglass cylinder with an inside diameter of 2-11/16 inches and a height of 12 inches. A column of sand was supported within the cylinder on a wire and cloth base 1-1/2 inches from the bottom. Drained fluid was collected by a funnel attached to the bottom of the cylinder and the volume of drainage was measured in a one liter graduated cylinder. A diagram of the setup is shown in Figure 4.

Before each drainage experiment, the cylinder was prepared by placing one inch of fresh sand on the wire base and then washing and wetting the sand by running three liters water through the cylinder. After the sand had thoroughly drained, one liter sludge was carefully placed in the cylinder and the cylinder was covered to prevent loss of water by evaporation. The time was recorded at the start of the test and the volume of filtrate was checked throughout the drainage period. After all drainage had ceased, the total drainable water was recorded.

b. <u>Second Year</u>

In addition to parameters used during the previous year, concentration of natural exocellular polymer and median particle size of activated sludge floc were measured and correlated to changes in filterability. To measure the naturally produced polymeric material, six 40-ml samples were removed from the aeration tank, centrifuged at 32,500g for 15 minutes and the supernatant was collected. The supernatant was then filtered through 0.45 micron millipore filters and added to 100 percent ethyl alcohol to a final volume ratio of 2 parts alcohol to 1 part supernatant. The mixture was then refrigerated at 4°C for 24 hours during which time a white fibrous precipitate formed. The polymer was then quantitatively measured by membrane filter separation and weighing.

As the study progressed, it became obvious that a physical parameter was needed in addition to the chemical and biological parameters, and a technique for measuring the median particle size of the sludge floc was developed. The

FIGURE 3

Relationship Between Filtration Time And Specific Resistance

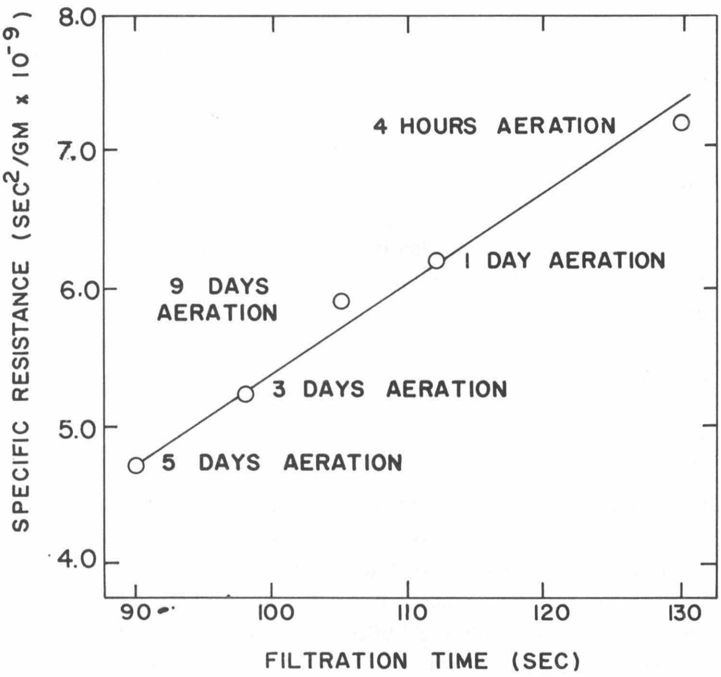
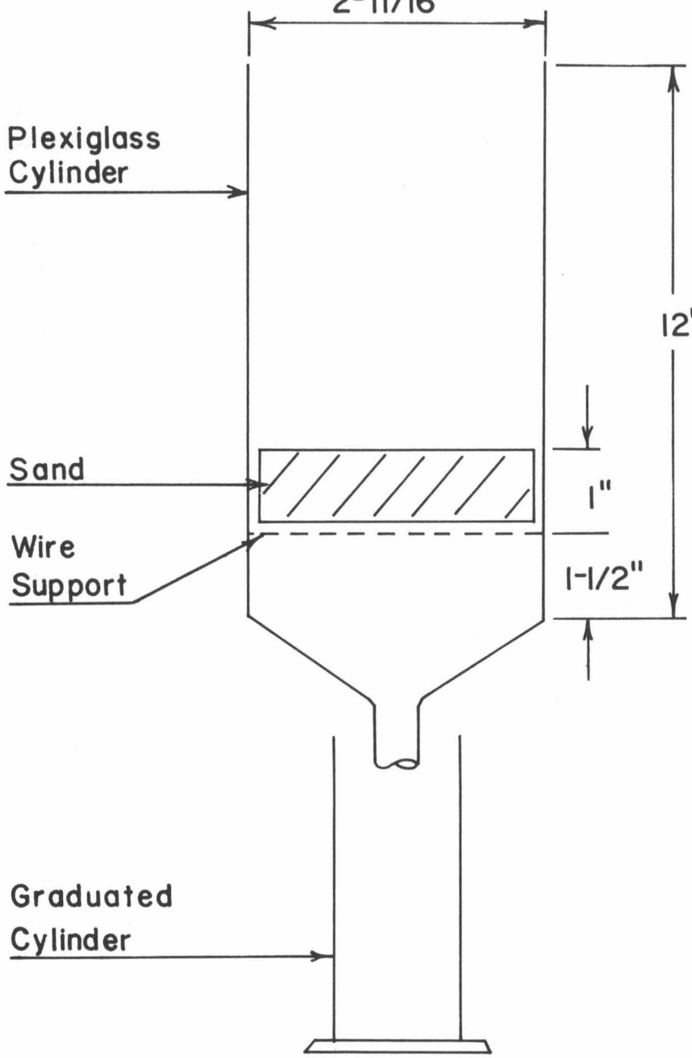


FIGURE 4

Sand Bed Drainage Cylinder



median particle size of the sludges was determined by wet sieving methods using special sieves, 3 inches in diameter, 4 inches deep with openings of 44, 75, 105 and 150 microns. The general procedure began by drying the sieve at $104^{\circ}C$ for about 2 hours, cooling it in a desicator and weighing it. For analysis, a sieve was partially submerged in distilled water, a volume of 50 ml sludge was transferred into it, and the sieve was shaken smoothly while remaining partially submerged. After about 10 minutes of shaking, the sieve was dried in the oven at $105^{\circ}C$, cooled in a desicator, and weighed to find the weight of solids retained by it. By repeating the procedure for the set of sieves, enough information was obtained for the determination of the median particle size.

The Buchner funnel setup used during this phase of the investigation was similar to that used previously except that a 4.5 cm inside diameter Buchner funnel was used instead of the 9 cm funnel. Using this apparatus, both specific resistance and compressibility factor were determined for each sludge sample except during some of the polyelectrolyte experiments. This more rigorous analysis was considered to be necessary for a reliable evaluation of mechanisms of dewaterability change. The following method was used to determine the two parameters:

1. The vacuum was adjusted to the desired pressure by closing or opening the pressure adjusting clamp;

2. The filter paper, Whatman No. 1, was wetted with distilled water and placed in the Buchner funnel and about 50 ml distilled water was filtered to seat the paper;

3. The pinch clamp was closed, and the attached 100 ml Mohr burette emptied;

4. 50 ml sludge was placed in the funnel;

5. The pinch clamp was opened and the stop watch started; when the level of filtrate reached the initial mark in the Mohr burette, the first time reading was taken;

6. The time to collect increments of 2 ml filtrate was recorded.

From the data obtained in the filtration test the specific resistance was calculated by the following equation:

$$r = \frac{2\,A^2 P\,b}{\mu\ C}$$

where $r$ = specific resistance, $sec^2/gm$

$P$ = pressure differential, gm/sq cm

$A$ = filter area, sq cm

$\mu$ = viscosity of filtrate, $9.82 \times 10^{-3}$ gm/cm-sec

$C$ = solids content of the feed sludge, gm solid/ml liquid

$b$ = slope of the plot T/V vs V, $sec/cm^6$

where $V$ = ml filtrate collected in T seconds.

The parameter C is the grams of dry solids deposited on the filter when one ml filtrate was obtained and it depends on the variables $C_o$ and $C_f$. $C_o$ is the solids content of the applied sludge, whereas $C_f$ is the solids content of the filter cake. The value of C was considered to be equal to the weight of solids per volume of liquid where the density of the liquid was one gram per cubic centimeter. This parameter was determined by the normal solids determination procedure. The slope of the best straight line of the T/V versus V plot was used as the parameter b. This expression is dimensionally incorrect because the pressure should be expressed as $Newtons/cm^2$ and the viscosity in $Newtons/seconds/cm^2$. The values obtained in the incorrect units $sec^2/gm$ can be converted to the correct units, by multiplying by $10^4$, with the new results in m/kg.

The second parameter describing the filtration characteristics of the sludge was the coefficient of compressibility. This value was obtained by plotting the logarithm of the specific resistance r, versus the logarithm of the applied vacuum P in $gm/cm^2$, the coefficient of compressibility being the slope of the straight line of best fit.

Both anionic and cationic polymers were investigated for synthetic polyelectrolyte conditioning effects. The polymer solution was prepared on a magnetic stirrer apparatus at a constant temperature of 20°C. After each day of experimentation, the used solution was discarded and a new one prepared and stirred for approximately 24 hours before being used.

Sludge samples of 50 ml in 100 ml beakers were used in these experiments. One station of a conventional multiple jar stirring device, with a 2 by 1-1/2-inch mixing blade, was employed to mix the samples. After adding the desired polyelectrolyte dose to the samples, they were mixed for 30 seconds at 100 rpm. A rapid mix was necessary to prevent the polymer solution and the solids (which had adsorbed large doses of polymer) from settling, and to obtain a homogeneous sample in which all solids present could adsorb polymer molecules equally. After mixing, the samples were gently stirred at 40 rpm for 90 seconds to promote floc formation and then were filtrated without any delay through the filter paper in the Buchner funnel. To expedite the screening evaluations, filtration time was used as the filterability parameter instead of specific resistance.

# RESULTS AND DISCUSSION

The results presented in this section are organized to show the relative effect and importance of the various factors, but the actual numbers should not be considered accurate enough for full-scale design purposes. However, it would be feasible to use the results as guidelines for pilot plant studies and the basic concepts could be applied to both plant operation and design.

**I. Sludge Handling**

It was found that handling procedures do have an effect on the filtering properties of waste activated sludge. During the first phase of the investigation, sludge collected from a rest stop along an interstate highway, hereinafter referred to as Sludge A, was subjected to separate experiments designed to evaluate the effects of sludge thickening, anaerobic conditions, chlorination, and aerobic digestion on subsequent filtration.

For the thickening experiment, solids concentration varied over a relatively small range, from 1.24 to 1.75 percent suspended solids. However, the results showed that increasing the solids concentration produced a nearly linear increase in filtration time. For the solids range studied, a 0.5 percent increase in solids concentration caused a 53 percent increase in filtration time. Thus, while sludge thickening is desirable to reduce the volume of sludge that must be subsequently handled, the advantage is partially offset by reduced filtration rate. Nevertheless, the overall effect of waste activated sludge thickening is positive since a substantial increase in filter yields can normally be obtained.

Randall, et al. [2] have previously reported that the solids concentration is the most important parameter affecting the rate of sludge gravity drainage and that this effect can be described by the equation:

$$y = \frac{1}{a + bx}$$

where

$y$ = drainage rate
$x$ = percent solids, and
$a$ & $b$ = curve fitting constants.

They reported that the intercept, a, was 0.27 and the slope, b, was 0.319 for gravity drainage. A similar analysis of the sludge filtration data previously mentioned yielded values of -0.085 and 0.264 respectively for a and b. The lower value obtained for the slope indicates that the solids concentration has a greater effect on sludge filtration than it has on gravity dewatering.

Sludges are often stored anaerobically for varying lengths of time, and anaerobic digestion is a common sludge treatment process. Figure 5 shows the results of anaerobic storage under quiescent conditions for 3 days. In this case the filtration time of the waste activated sludge increased by 45 percent. Typically, the increase is more linear with time than this figure indicates, and the change may be even greater. Any agitation of the sludge during anaerobic storage caused a further increase in the degradation of filtration properties. Mixing anaerobically stored sludge with sufficient power to keep the sludge solids from settling caused the filtration time to increase at a rate 10 times that of unmixed anaerobic sludge. It should be emphasized that these results apply only to waste activated sludge and not to primary sludge.

Chlorination is often used to improve the settleability of activated sludge. However, in this study chlorination was found to be detrimental to sludge filterability. Figure 6 shows that the addition of 175 and 525 mg/l chlorine to undigested activated sludge caused an increase in filtration time of 100 and 400 percent respectively.

The use of aerobic digestion as a sludge treatment method is increasing in popularity [36]. In contrast to thickening, anaerobic storage, and chlorination, it was found that aeration of waste activated sludge produced an initial improvement in filterability. Figure 7 shows the results of aeration of waste activated sludge for 14 days. Aeration produced an initial improvement in sludge filterability, with a maximum improvement in 4 to 6 days. As the figure shows, however, greater periods of aeration caused an increase in filtration time with the result that after 2 weeks of digestion the sludge was almost as difficult to dewater as it was prior to aerobic digestion. This general relationship has subsequently been reproduced several times with waste activated sludge from other plants.

During the anaerobic storage, chlorination, and sludge aeration studies, the total organic carbon of the sludge filtrate was periodically measured. Anaerobic storage caused a steady increase in filtrate organic carbon while aeration caused a decrease. The data, plotted in Figure 8, reveal a linear relationship between filtrate organic carbon and filtration time for sludge subjected to anaerobic storage and aerobic digestion. For the data shown, a

FIGURE 5

Variation in Filterability with Anaerobic Storage

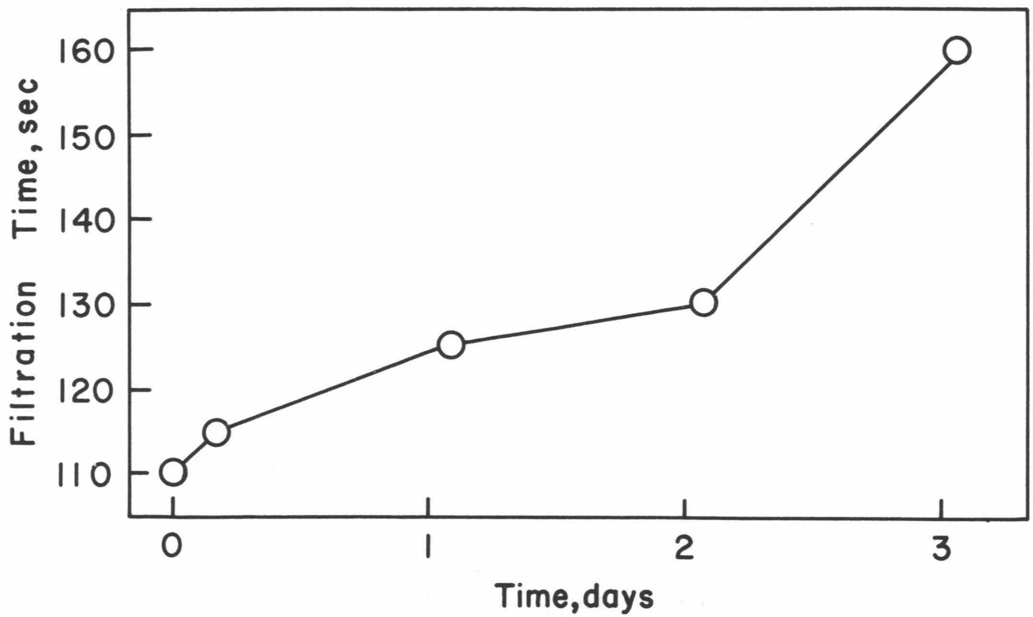


FIGURE 6

Variation in Filtration with Chlorine Concentration

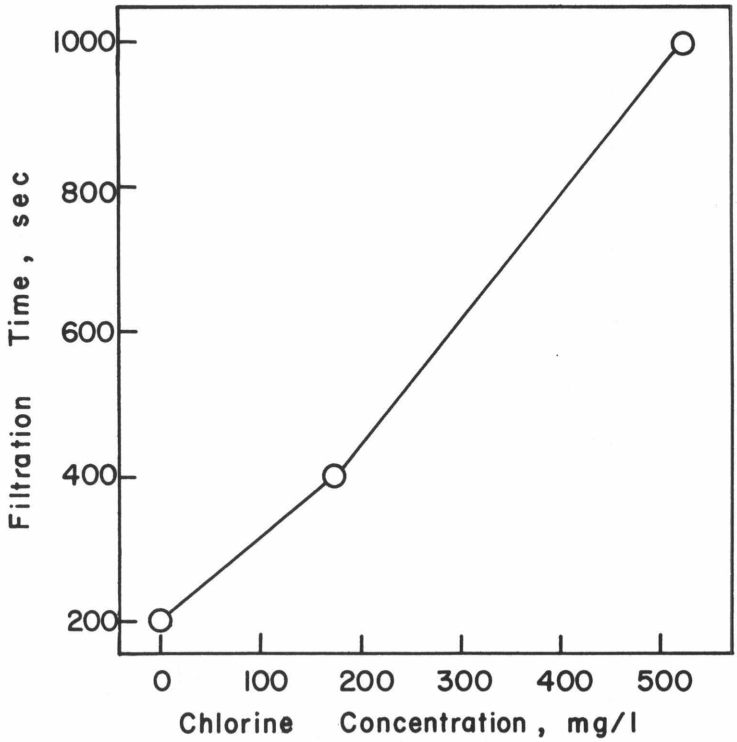


FIGURE 7

Variation in Filtration with Aeration

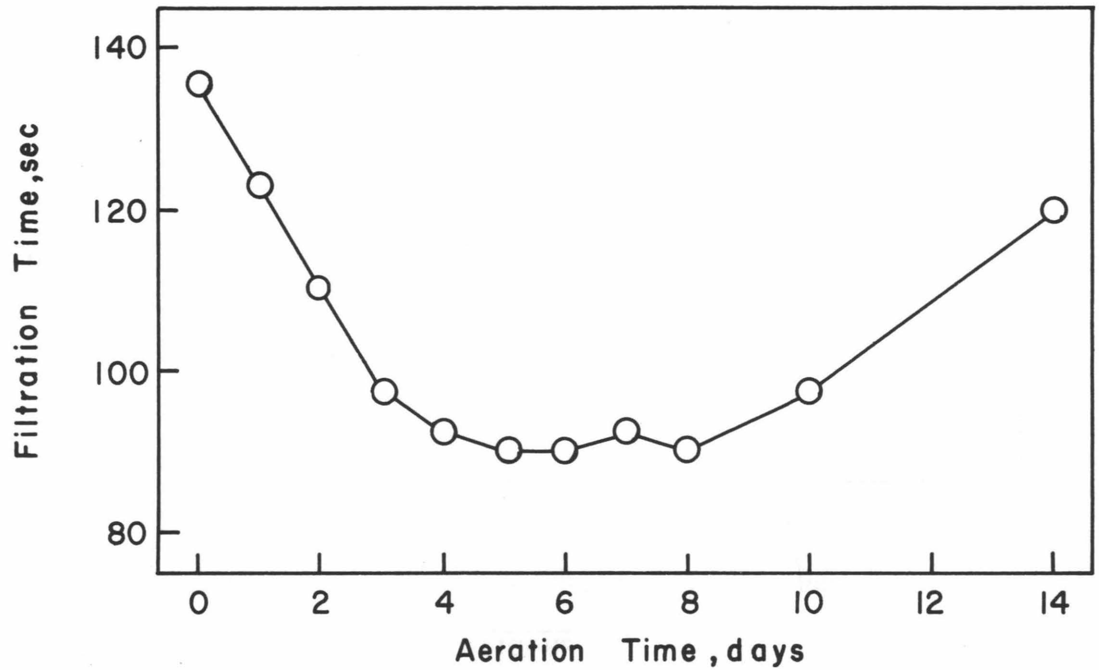
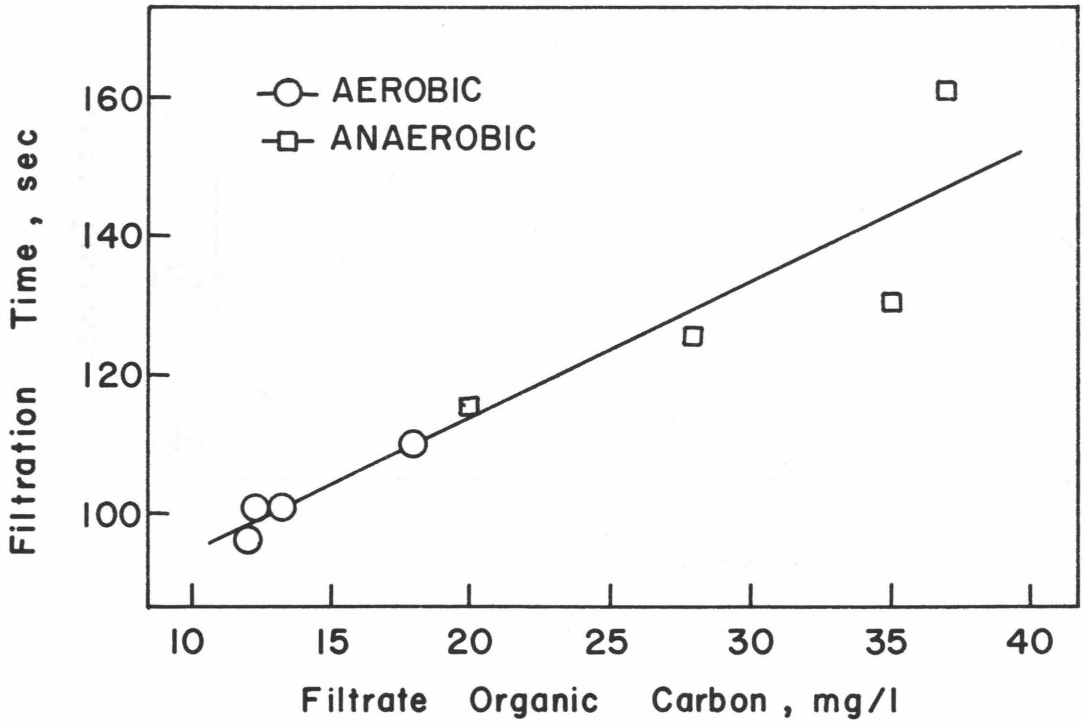


FIGURE 8

Variation in Filtration with Organic Carbon Concentration



10 mg/l increase in filtrate organic carbon corresponds to a 17 percent increase in filtration time. Chlorination of waste activated sludge also produced an increase in filtrate organic carbon that could be correlated with change in filterability, but the change in filtration time per unit increase in TOC was much greater. Chlorination with 350 mg/l chlorine caused the filtrate organic carbon to increase from 11 to 68 mg/l with an increase in filtration time of more than 60 percent per 10 mg/l increase in filtrate TOC.

The increase in filtrate organic carbon indicates some cellular destruction or leakage of cellular constituents from the activated sludge as a result of biological stresses imposed by the handling conditions. From correlation of the data, it seems clear that these stresses also worsen the filtering characteristics of the activated sludge. It appears that stressing or disrupting activated sludge to such an extent that organic carbon is released into solution results in a decrease in the filterability of the sludge. By contrast, since aerobic digestion initially maintains cellular integrity, normal microbial mechanisms control and dewatering enhancement is permitted to occur.

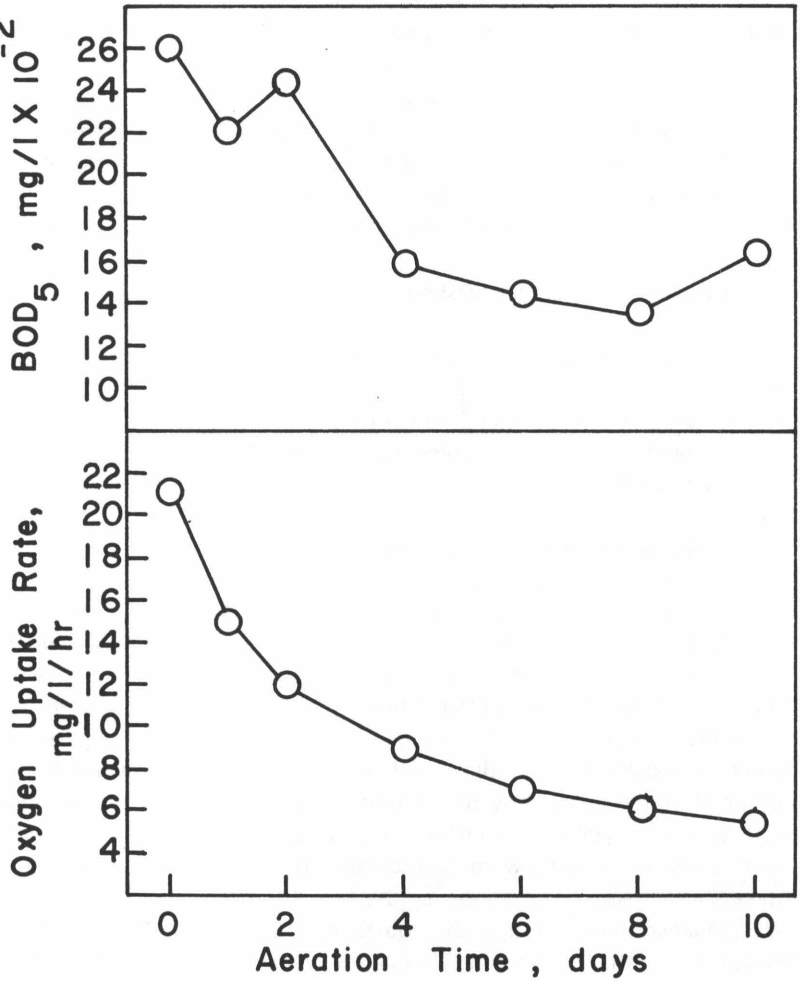
**II. Changes During Aerobic Digestion**

Since waste activated sludge undergoes both an initial improvement in filterability and then a considerable worsening, the changes that occur in several sludge properties and environmental parameters during aerobic digestion were carefully monitored in an attempt to define the important mechanisms of change.

Figure 9 shows the variation in mixed liquor 5-day BOD and oxygen uptake rate of the sludge during 10 days of aeration. The BOD decreased from 2600 mg/l to a low of 1300 mg/l. This change in BOD was considered to be primarily a result of endogenous respiration since the rate of BOD reduction during the aeration period remained steady. Further, supernatant COD and TOC dropped to low values after 6 hours of aeration and changed little the rest of the 10-day period. The oxygen uptake rate, which is a measure of aerobic biological activity, decreased rapidly at first and was less than 10 mg/l/hr after the fourth day of aeration. As a further measure of biological viability, the dehydrogenase enzyme activity was also measured. Changes in this measure of activity were considerably different from that of oxygen uptake. An increase occurred during the first day of the aeration period. The level remained high through the fourth day and then decreased steadily through the tenth day, reaching a value lower than the initial value after the sixth day. Thus, dehydrogenase enzyme activity changes were almost a mirror image of changes in filterability. This result is interesting because it indicates

FIGURE 9

Variation in BOD and Oxygen Uptake Rate
with Aeration



that the sludge was most viable during the period of improvement in filterability and that viability decreased prior to a worsening of the filtration rate.

Variation in the suspended solids, filtrate protein, and zeta potential of the sludge during 10 days of aeration is illustrated by Figure 10. The suspended solids decrease was very small, less than 10 percent of the original solids concentration. Filtrate protein decreased the first 4 days and then increased overall with additional aeration. With the exception of the measurement for the sixth day, filtrate protein concentration change paralleled the change in sludge filtration rate with aeration time. The absolute value of the zeta potential of the raw sludge was small; however, in 10 days, aeration did cause a slight decrease. The changes in suspended solids, filtrate protein, and zeta potential were considered to be too small during this experiment to significantly affect the variation in filtration rates that occurred during aeration although soluble protein probably reflects stress conditions.

Figure 11 shows the variation in filtrate carbohydrate concentrations during aeration. Carbohydrate was the only parameter that increased continually during the 10 days of aeration. The increase was substantially linear, from an original concentration of 6 mg/l to a final of 20 mg/l. It was concluded that this steady increase in filtrate carbohydrate was the result of extracellular polysaccharide production during endogenous respiration as described by Busch and Stumm [37]. It is obvious, however, that extracellular polymer did not control filterability during the latter stages of digestion.

## Reduced Biological Activity

To further evaluate the effect of reduced biological activity on sludge aeration, a sludge sample was divided into two portions; one portion aerated at 6°C, and the other aerated at 19°C. Figure 12 shows that the filtration time of the sludge aerated at 19°C decreased quickly during the first few days of aeration as expected from earlier observations. However, the filtration time of the sludge aerated at 6°C increased rapidly during the first 2 days of aeration, then decreased only gradually for the remainder of the aeration period. At no time during the 8 days of aeration did the filtration rate of the 6°C sludge drop below the original filtration rate. The 6°C temperature appears to have reduced the biological activity to such an extent that filtration improvement could not occur; thus indicating the importance of biological activity to the conditioning process.

FIGURE 10

Variation in Suspended Solids, Protein and Zeta Potential with Aeration

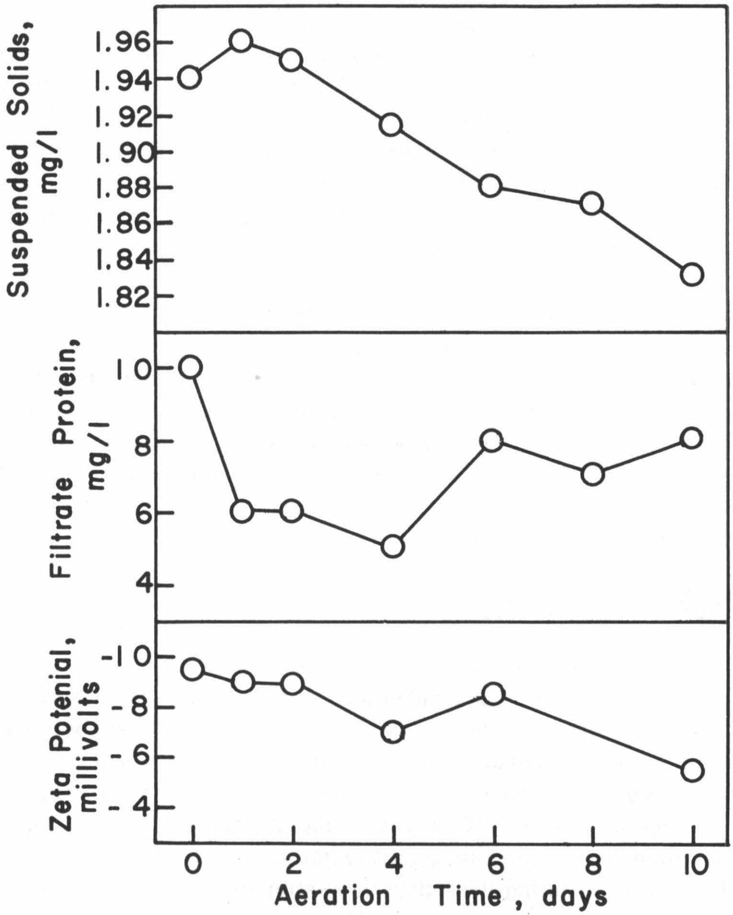
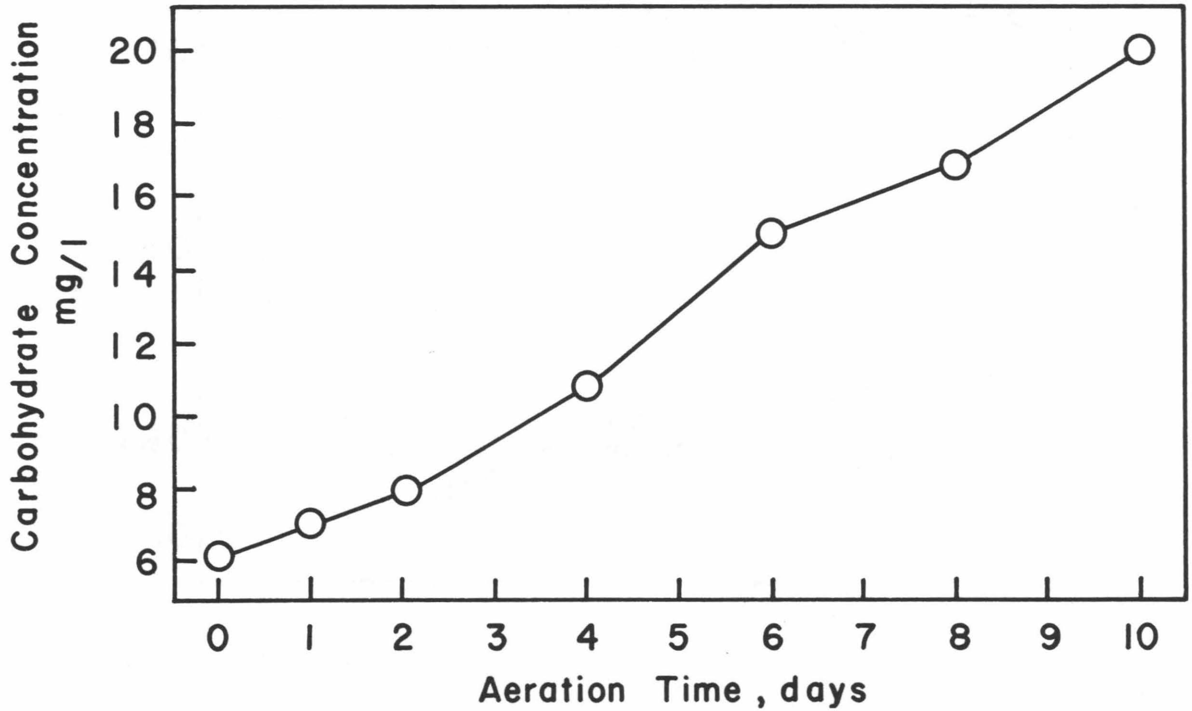


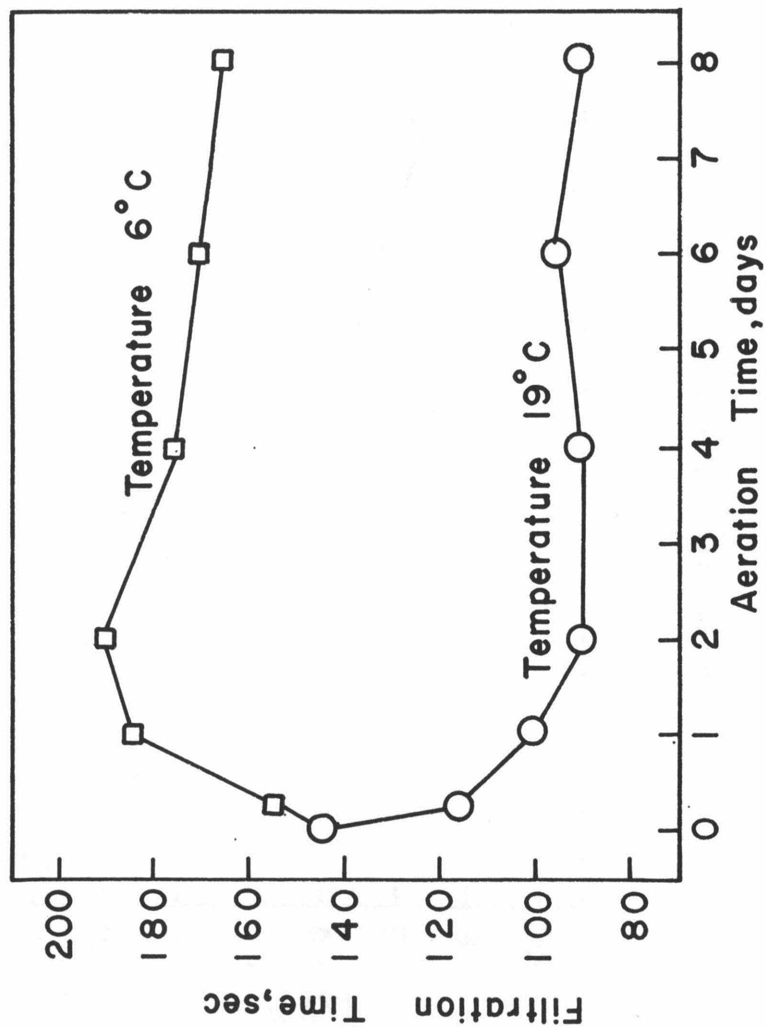
FIGURE 11

Variation in Carbohydrate with Aeration



## FIGURE 12

Variation in Filtration with Aeration at 6°C and 19°C



## IV. Biological Recovery of Filterability

Previous experiments had shown that anaerobiosis and chlorination degrade filterability. Additional experiments showed that filtration characteristics which had been degraded by anaerobic storage or chlorination could be recovered and improved by aeration. Figure 13 compares the filtration rates for three portions of sludge which were stored anaerobically for 0, 2, and 4 days, respectively, and then aerated. As previously described, the filtration properties of the sludges degraded steadily with anaerobic storage, more linearly in this case than illustrated by Figure 4. However, the filtration properties of all sludge portions started improving immediately upon aeration and all sludges reached a minimum filtration time of 90 seconds after from 4 to 6 days of aeration regardless of the length of time under anaerobic storage.

Figure 14 shows portions of sludge which were subject to 0, 175, and 525 mg/l chlorine, then aerated for 8 days. As previously noted, sludge filterability degraded in proportion to the concentration of chlorine added, but the filterability of all portions of sludge improved with aeration. However, unlike the anaerobic sludge, the filterability of chlorinated sludges did not completely recover with aeration.

## IV. Sludge Filterability versus Digester Operation

To investigate the operating parameters that affect aerobic biological conditioning, sludges were aerated at different suspended solids concentrations, aeration rates, and temperatures.

Figure 15 illustrates the results of aerating sludges at four different suspended solids concentrations. As indicated previously, the initial filtration rate varied linearly with suspended solids concentrations. Interestingly, the same linear relationship between filtration time and suspended solids concentration remained constant regardless of the time of aerobic digestion of the sludge. This consistent linear relationship indicates that the effect of suspended solids concentration on filterability is not affected by biological conditioning.

Figure 16 shows the effect of different aeration rates on sludge conditioning. Aeration at 535 ml/min resulted in filtration improvement as previously discussed. Aeration at 1150 ml/min resulted in a decrease in the improvement in filtration, and aeration at rates higher than 1150 ml/min resulted in no filtration improvement in 10 days of aeration. The results of this investigation show that while mild aeration improves sludge filterability, high aeration rates appear to cause excessive shear forces in the sludge flocs which counteract the

FIGURE 13

Variation in Filtration for Sludge Aerated After Anaerobic Storage

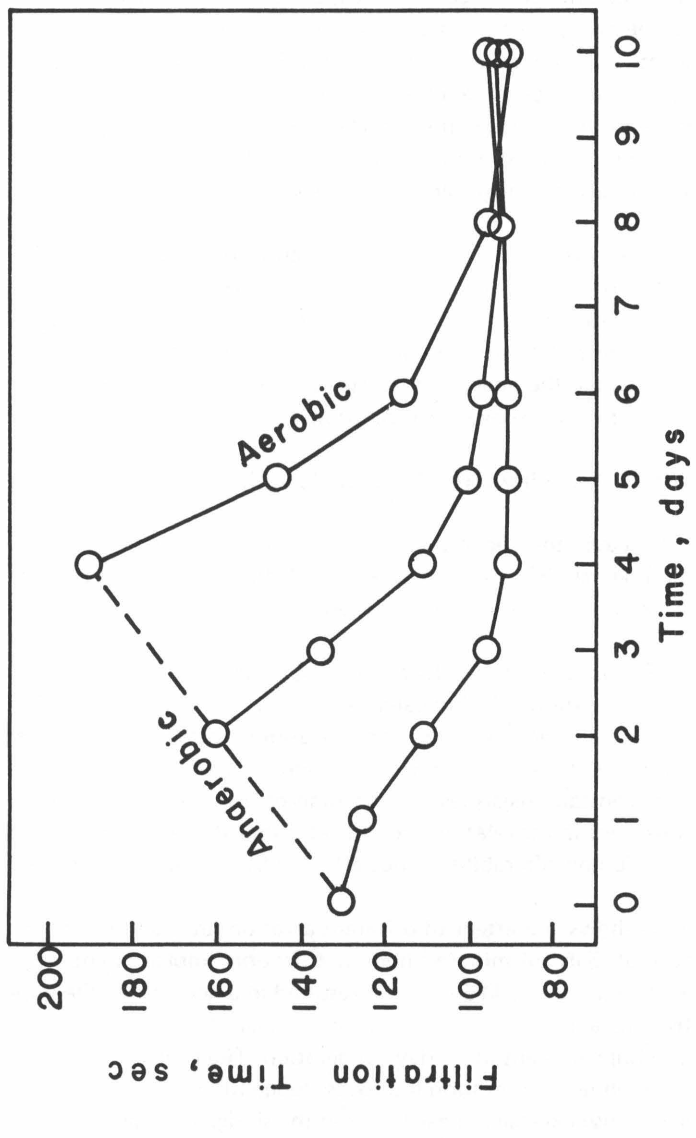


FIGURE 14

Variation in Filtration for Sludge Aerated After Chlorination

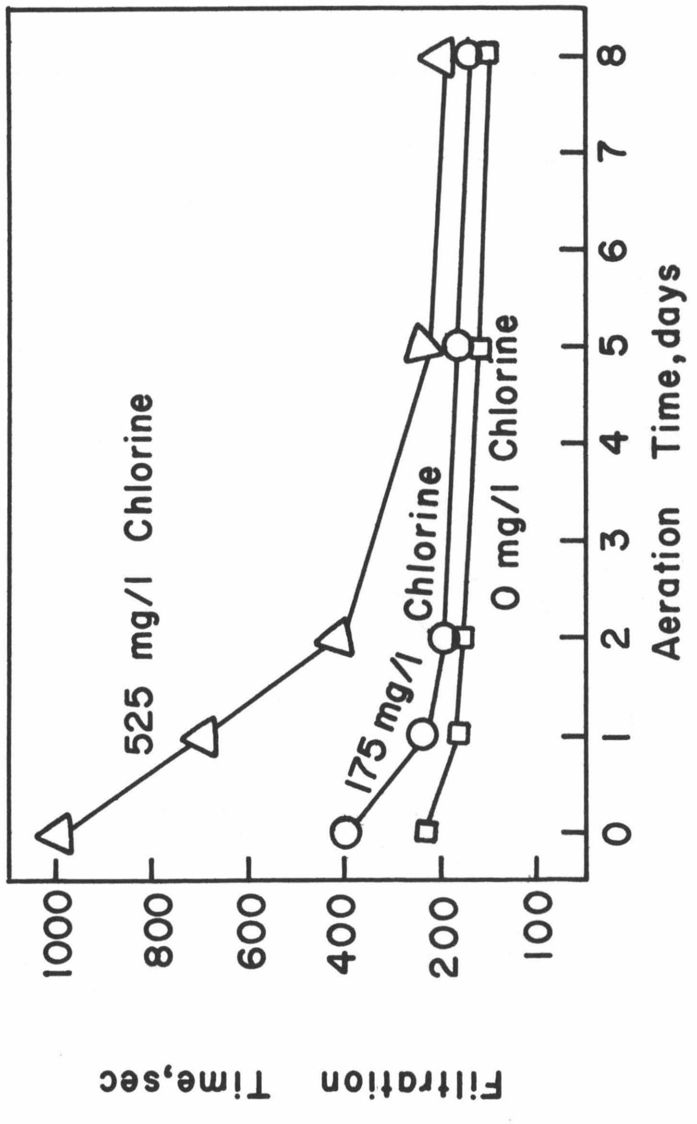


FIGURE 15

Variation in Filtration with Suspended
Solids Concentration

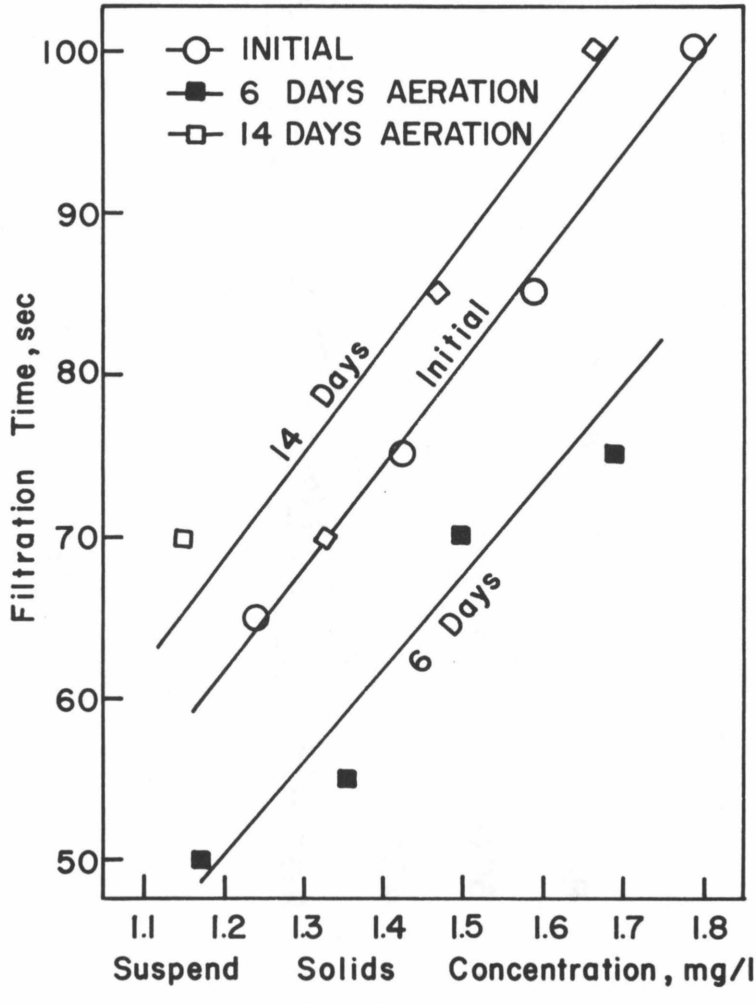
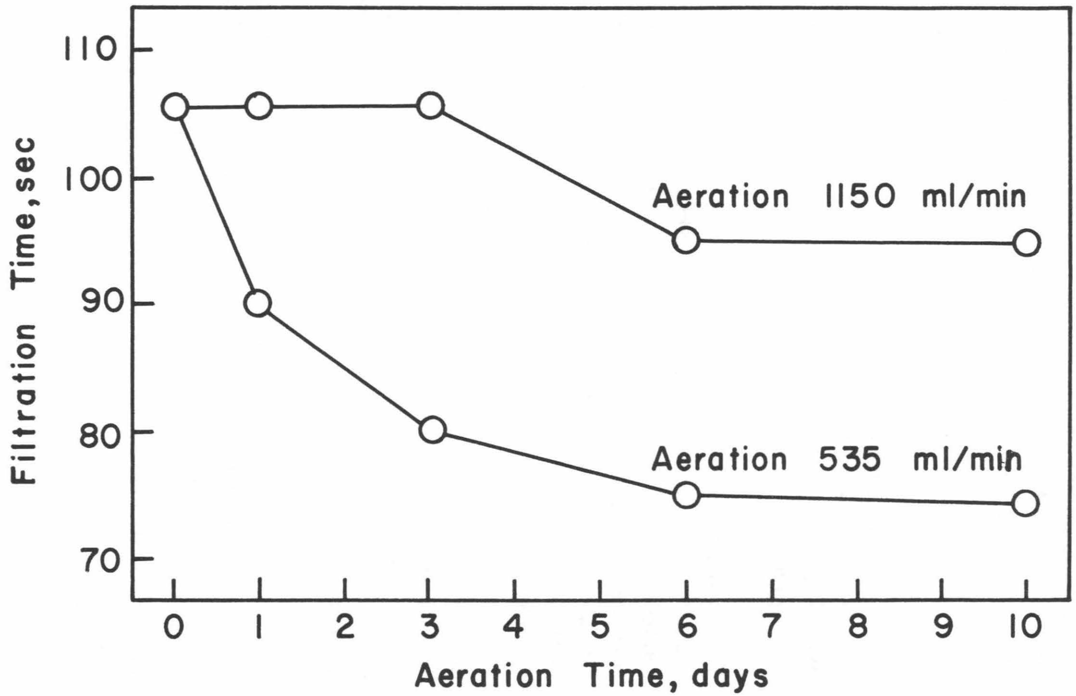


FIGURE 16

Variation in Filtration with Variation in Aeration Rate



improvement in filtration due to aerobic biological activity. The shear forces probably reduce the average size of the floc particles and the resulting fines make filtration more difficult.

To determine the effect of dissolved oxygen concentrations above zero on biological conditioning, three portions of the same sludge were mixed mechanically and aerated at 0, 34, and 148 ml/min. Figure 17 compares the variation in dissolved oxygen and filtration time for the sludges aerated at these rates. The sludge aerated at 148 ml/min maintained a dissolved oxygen concentration greater than 2.0 mg/l throughout the aeration period. The sludge which was not aerated had no detectible dissolved oxygen present during the entire testing period whereas the sludge aerated at 34 ml/min had no detectible dissolved oxygen for the first 2 days, approximately 0.2 mg/l at the end of the third day, and greater than 2.0 mg/l from the fourth through the sixth day.

The filterability of the sludge which was not aerated showed the typical linear decrease of an anaerobically stored sludge. The filterability of the sludge aerated at 148 ml/min showed the typical improvement of aerated sludge. However, the sludge aerated at only 34 ml/min showed a decrease in filterability when no dissolved oxygen was detectible and an improvement in filterability when some dissolved oxygen was present. A comparison of the dissolved oxygen and filtration time for the sludge aerated at 34 ml/min shows that some filtration improvement did occur starting after the first 1.5 days of aeration, but no dissolved oxygen was detected until the third day. This observation illustrates that when oxygen is supplied at a rate which partially satisfies the total oxygen demand, the filterability of the sludge will not degrade as rapidly as non-aerated sludge nor improve as rapidly as sludge aerated at a rate sufficient to supply excess oxygen.

The temperature of digestion was also found to have an effect on biological sludge conditioning but the relative difference was small. When portions of the same sludge sample were digested at temperatures of 20, 25, 30, and 35°C, the best filtration time was obtained with the 30°C sludge. It was only slightly better than the 25 and 35°C samples, but the results do clearly show that minimum filtration time occurs sooner and deteriorates more rapidly with an increase in temperature. Figure 18 compares the results of aeration at 20 and 30°C.

At the beginning of the temperature experiments it was noted that adjusting the temperature of the sludge samples produced a considerable change in filtration time. Filtration time had an immediate decrease of more than 30

FIGURE 17

Variation in Dissolved Oxygen and Filtration with Aeration at Different Aeration Rates

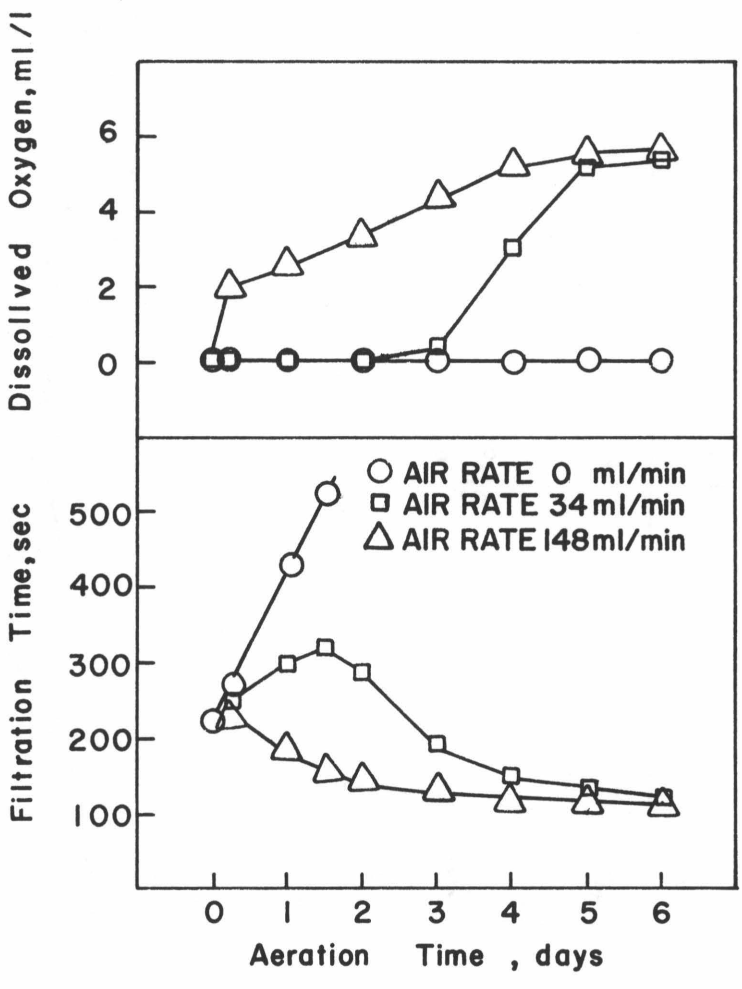
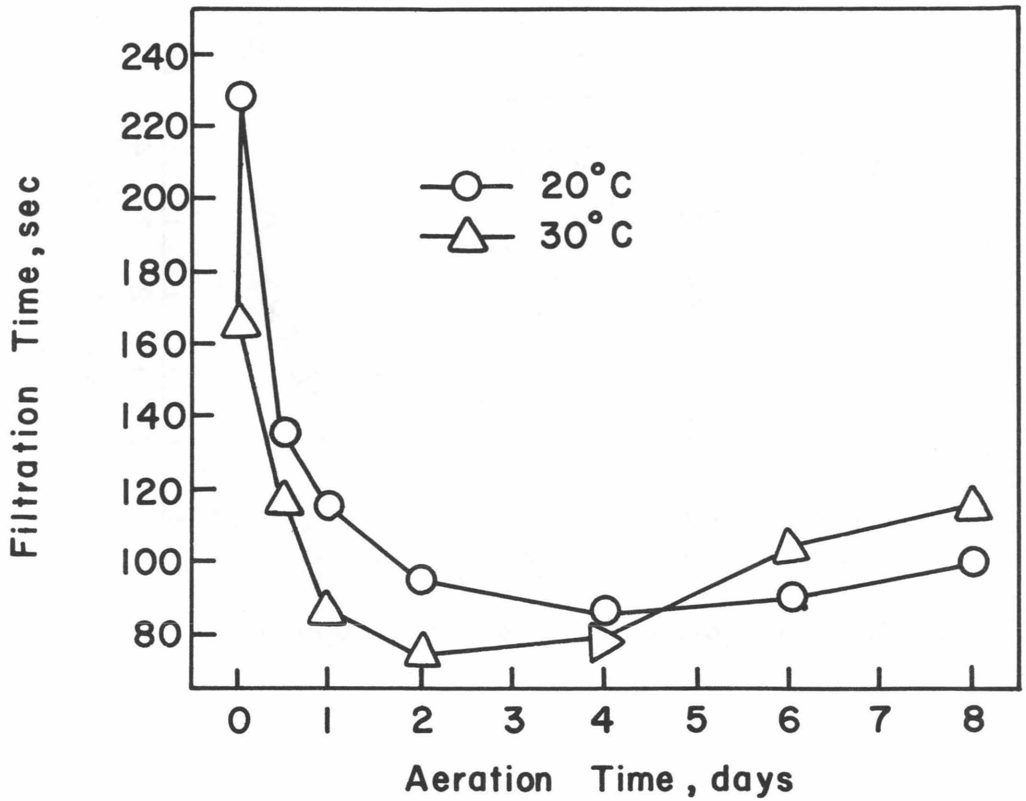


FIGURE 18

Variation in Filtration with Aeration at 20°C and 30°C



percent when the temperature was changed from 20 to 35°C. A similar comparison was made after 9 days of aeration, however, and the decrease for the same temperature change was only about 12 percent. Figure 19 compares the data. Since viscosity changes would have been the same, the 18 percent difference would have had to have been the result of changes in the sludge properties.

The comparative study of batch and continuous aerobic digestion showed that continuous digestion will produce a better conditioned sludge after several days of aeration, but the difference is small. There was no difference in the filtering rates of the sludges from the two systems after 3 days, about a 3 percent difference after 5 days, and a 12 percent difference after 10 days.

### V. Comparison of Gravity Dewatering and Filtration

Previous studies have shown that aerobic digestion will improve the gravity drainage of waste activated sludge. To make the studies more comparative, 10 liters of sludge were aerated for 6 days, and then at selected intervals 1 liter portions of the sludge were dewatered by sand bed drainage and 100 ml portions were vacuum filtered. Figure 20 shows that the total drainable water increased by only about 2 percent during the 6 days; however, the time required to drain 750 ml of a 1 liter sample decreased by 73 percent while the vacuum filtration time decreased by only 32 percent.

### VI. Discussion of the First Phase

In agreement with previous investigators [9, 23, 37, and 38], a conclusion of this phase of the research was that the filtration characteristics of a biological sludge are a function of the degree of flocculation of that sludge. However, there is less agreement with the precise mechanism of flocculation. Early investigations of bioflocculation [23] indicated that biological suspensions are similar to colloidal suspensions and that van der Waal's forces of attraction are the predominate forces of flocculation. The necessary conditions for bioflocculation were felt to be a sludge zeta potential below 20 millivolts and a low food to microorganism ratio (F/M). A low zeta potential was felt to be necessary so that van der Waal's forces of attraction could predominate, and a low F/M was necessary so that motile forms of bacteria would not predominate and overcome attractive forces.

More recent investigations [37, 38] indicate that van der Waal's attractive forces alone are not of sufficient strength to account for biological flocculation. It has been postulated that polymers of biological origin such as

FIGURE 19

Variation in Filtration Time with Temperature
at Two Stages of Digestion

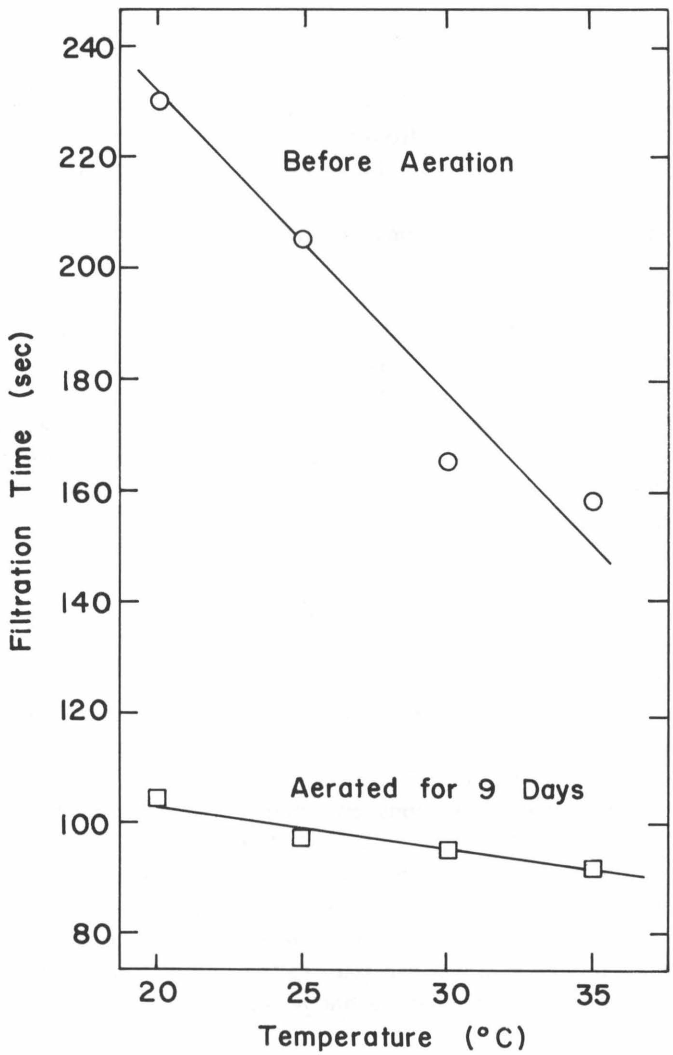
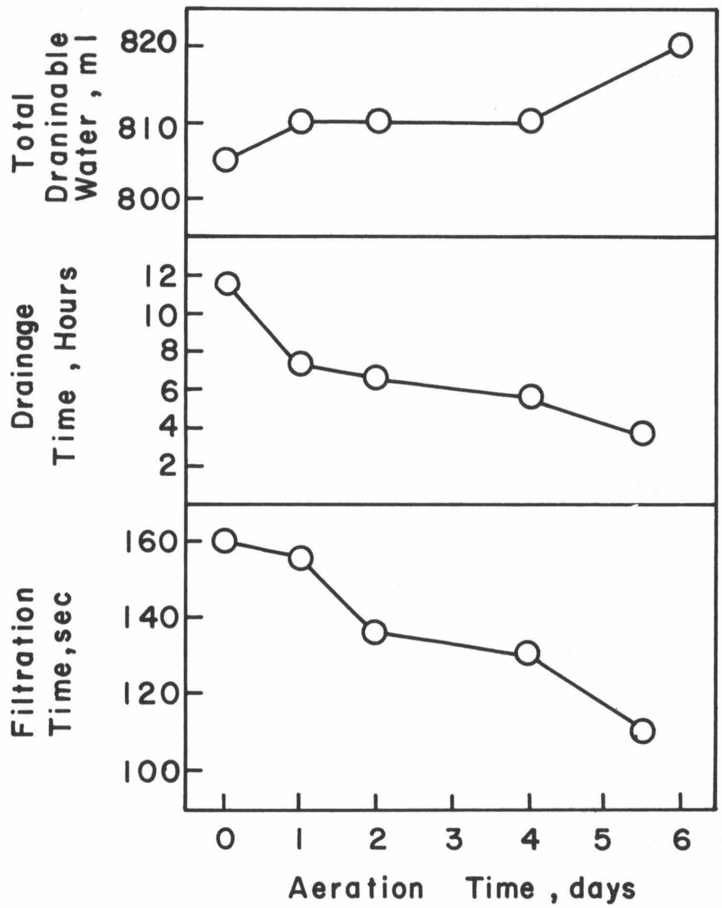


FIGURE 20

Variation in Dewaterability with Aeration



polysaccharides and polyamino acids, which are excreted or exposed at the surface of bacteria during the declining growth (or endogenous respiration) phase of metabolism are of sufficient length to form bridges between microbial particles. This bridging increases bioflocculation and improves sludge filterability.

During this investigation, zeta potential appeared to have little effect on filtration variation. The zeta potential for the raw sludge was low and changed only slightly during aeration. Perhaps the initial low potential was necessary for the other flocculation mechanism to function.

The presence of biologically produced polymers was not determined directly during this phase; however, the increase in filtrate carbohydrate indicated that polysaccharides were being excreted from bacteria and that these saccharides were accumulating in the filtrate. The steady increase in carbohydrate during aeration did indicate they were of biological origin. It was noted, though, that carbohydrates continued to increase during aerobic digestion even when the filterability was worsening.

The mechanism of filtration improvement during this investigation was almost certainly biologically induced flocculation. When conditions for biological activity were good and when the F/M ratio was low enough to promote endogenous respiration, sludge filterability either improved or remained at the same level. When biological activity was disrupted by chlorination, high temperatures, or lack of oxygen, the sludge filterability rapidly degenerated, and when biological activity was inhibited by cooling or by long aeration periods, filterability either stabilized or became progressively worse.

The results showed that the manner in which waste activated sludge is handled can have a significant effect on the rate of dewatering of the sludge by either vacuum filtration or by sand bed drainage. Filterability can be adversely affected by anaerobic storage, excessive mixing, chlorination, and changes in mixed liquor temperature. On the other hand, filtration properties can be restored or improved by aeration for extended periods of time. To achieve optimum conditioning, the biological organisms must be anaerobically maintained in the endogenous phase and be active and intact when subjected to dewatering procedures. Logically, the dewaterability of the sludge should be a strong function of the average floc particle size and of the percent fines in the mixed liquor. It seems likely that the filterability results reflected the effect of the handling procedure on the floc particles.

## VII. Sludge Variation

All the sludge samples used during the first phase of the investigation were collected from one source, and although numerous samples were used to compile the data, the applicability of the results to other sludges was a matter of concern. To evaluate this aspect, two other sources were selected and all three samples were subjected to comparative experiments. Each sample was also separately identified so that variation in sludge from the same source could be evaluated.

The initial characteristics of the various sludge samples used are given in Table 1. The Sludge A samples were obtained from the same Interstate rest stop unit that supplied the sludge for the first phase. Sludge B was obtained from another small extended aeration unit whereas Sludge C samples were obtained from an overloaded 10 MGD municipal plant. The alkalinity of Sludge B was extremely high because lime was added to the unit to aid settling. Thus, this sludge was chemically conditioned to a limited extent when collected.

Specific resistance was used as a measure of filterability instead of filtration time. By measuring specific resistance at several different pressures it was possible to measure the coefficient of compressibility. Since it was concluded from the previous work that the integrity and size of the floc particles determine sludge filterability, most of the characterization tests were omitted. Measurement of exocellular polymer was refined and continued to further evaluate the role of natural flocculation and part way through the experiments a test designed to measure the average particle size was introduced. With respect to the latter test, there seems to be little doubt that sludge dewatering is a function of the degree of flocculation, and hence the floc size. The question is; can a relatively simple test be devised to measure it? A successful test could conceivably be used to predict the dewaterability of sludge during biological conditioning.

The mixed liquor suspended solids concentrations of the samples and the solids reductions that occurred during aerobic digestion are listed in Table 2. In general, solids reduction was lower than expected and it was somewhat less than what had been obtained with sludge from the same sources during earlier studies [2, 11, and 16]. Solids reduction is often highly variable and is generally thought to reflect the initial stability of the sludge being digested. Such was seemingly not the case during these experiments, however, since Sludge C appeared to be poorly stabilized yet had low reductions. The sludge samples used during the first phase usually had reductions on the order of 8 percent as represented by Sludge A in the table.

TABLE 1

Initial Sludge Sample Characteristics

| | Characteristic | | | | | | |
|---|---|---|---|---|---|---|---|
| Sludge Sample | pH | Alkalinity mg/l | Natural Polymer mg/l | Polymer–Solids Ratio | Median Particle Size, $\mu$ | Specific Resistance m/kgx$10^{13}$ | Compressibility Factor |
| A–I | 6.0 | 195 | 60 | 0.0032 | – | 0.33 | 0.87 |
| A–II | 6.5 | 135 | 70 | 0.0057 | 30 | 0.53 | 0.81 |
| A–III | 6.1 | 130 | – | – | 41 | 0.35 | 0.95 |
| B–IA | 7.3 | 4600 | 75 | 0.0039 | 84 | 0.225 | 1.2 |
| B–IB | 7.2 | 4750 | 95 | 0.0053 | 61 | 0.225 | 0.93 |
| B–II | 7.6 | 2200 | – | – | – | 33 | 0.99 |
| C–I | 7.0 | 470 | 235 | 0.0242 | 41 | 16 | 1.05 |
| C–II | 7.0 | 600 | – | – | – | 11 | 1.03 |

## TABLE 2

Reduction in Total and Volatile Suspended Solids During Aerobic Digestion

| Sludge Sample | Initial Total Suspended Solids mg/l | Percent Initial Volatile Suspended Solids | *Percent Reduction in Total Suspended Solids | *Percent Reduction in Volatile Suspended Solids | Final pH |
|---|---|---|---|---|---|
| B–IA | 19,000 | 57 | 30 | 21 | 6.9 |
| B–IB | 17,600 | 60 | 33 | 26 | 6.9 |
| B–II | 11,300 | 62 | 7 | 7 | 7.8 |
| A–I | 18,800 | 74 | 13 | 11 | 4.8 |
| A–II | 12,300 | 76 | 6 | 5 | 4.5 |
| A–III | 12,200 | 75 | 3.2 | 2.1 | 5.0 |
| C–I | 11,600 | 75 | 14 | 15 | 6.1 |
| C–II | 10,250 | 79 | 6.3 | 13.5 | 6.0 |
| Sludge A | 19,400 | 67 | 7.5 | 7.5 | 5.8 |

*All values are after 11 days of aeration except for C–II which is after 9 days.

The filtration results showed that there is some variation with sludge origin in the conditioning effect of aerobic digestion on waste activated sludge, but the trends are similar. Typical specific resistance curves for each sludge are compared in Figure 21. In all instances shown, there was an initial improvement with a subsequent deterioration of filterability. The time at which deterioration began varied widely, however, and started after only one day of improvement with the most difficult to dewater sludge (Sludge C). Figure 22 shows that the variation with sludge origin in the conditioning effect on the coefficient of compressibility was even less.

There was actually more variation between samples from the same source than for different sludges as indicated by the previous figures. For example, Sludge A-I showed no improvement in filterability at all with aerobic digestion. Instead, the specific resistance steadily worsened with time of aeration. It was concluded that the sludge had been sufficiently digested in the plant until it had passed the point of improvement and that further digestion produced a worsening of filterability as would normally occur during the latter part of the digestion period. In general, however, the variation in samples from the same source was small.

Figure 23 reveals that there was a strong similarity in exocellular polymer production during aerobic digestion by the various sludges. The initial drop represents the metabolism of usable carbohydrate whereas the rest of the curve shows the build-up of biologically resistant polymer. As noted during the previous experiments, exocellular polymer continues to increase even when the sludge dewatering properties are getting worse. It can be concluded that either the exocellular polymer reaches an overdose point where a further build-up is antagonistic to dewatering or that it cannot overcome the effects of fragmentation during the latter stages of digestion. Figure 24, which compares the change in polymer-to-solids ratio for the three sludges, indicates that the rapid deterioration of Sludge C may have been caused by an overdose effect since the ratio is much higher than that of the other sludges.

The only significant decrease in natural polymer that occurred after the first day was during the last two days of Sludge A digestion. Since only one data point was involved, it would seem reasonable to consider it as a poor measurement except that a similar decrease occurred during the entire digestion period for the previously mentioned sample of Sludge A that showed no improvement in dewatering. Apparently, metabolism of the exocellular polymer can occur during the latter stages of digestion. Nevertheless, the interpretation that filterability worsened because of a decrease in polymer concentration does not seem warranted when all the data

FIGURE 21

Variation in Biological Conditioning with Sludge Origin: Filterability

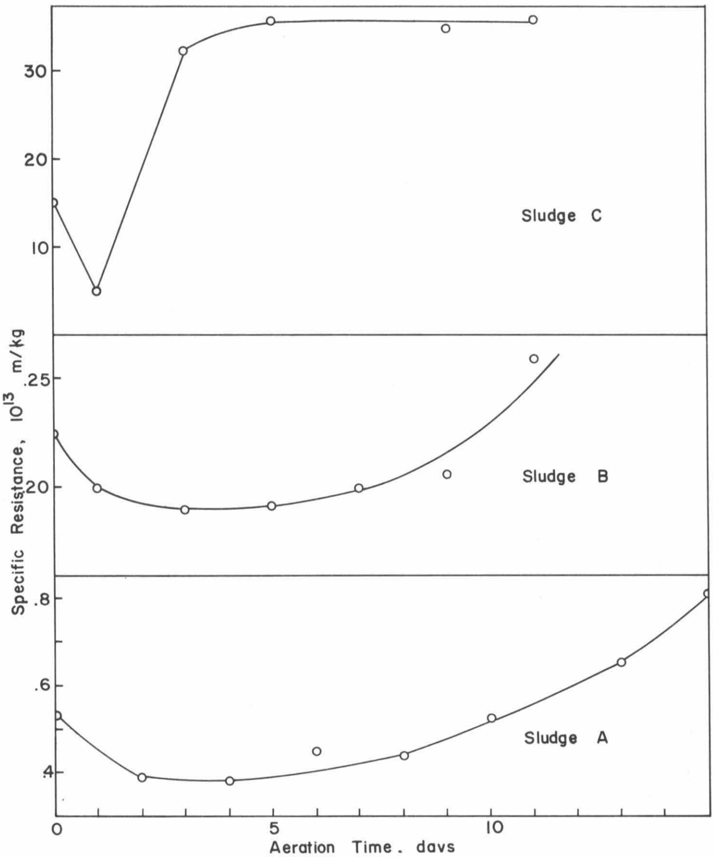
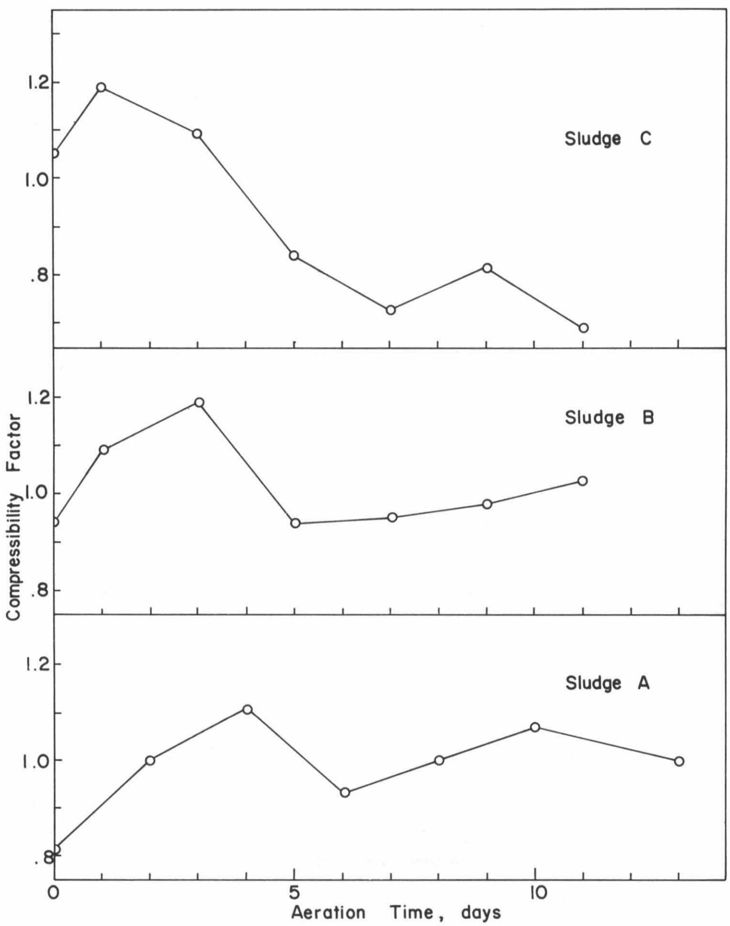


FIGURE 22

Variation in Biological Conditioning with Sludge Origin: Compressibility



## FIGURE 23

Variation in Polymer Production with Sludge Origin

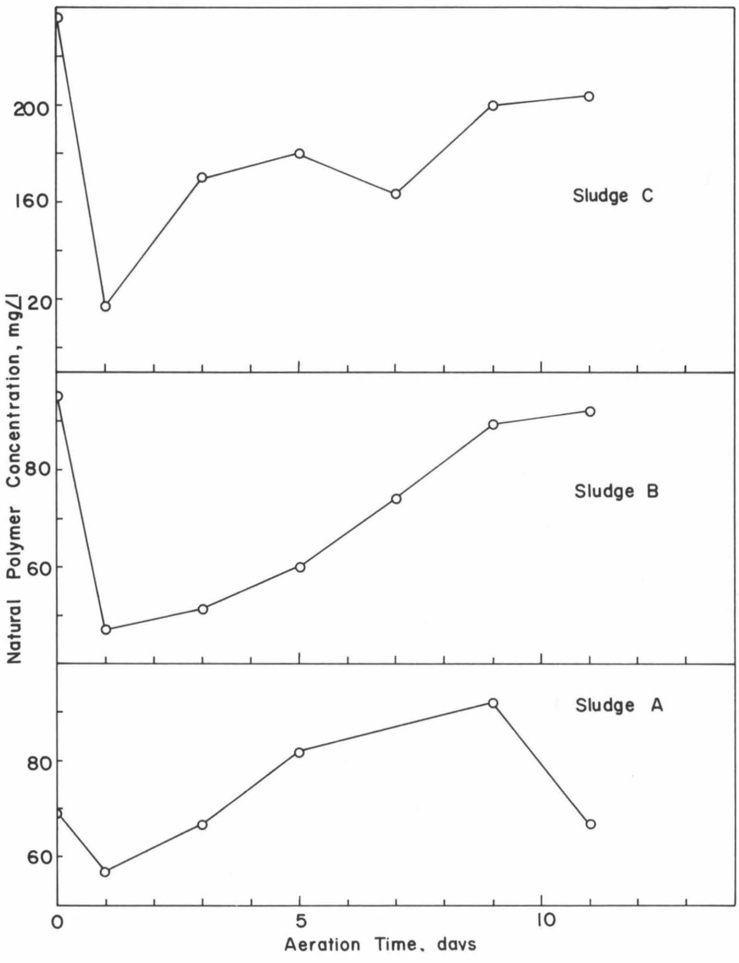
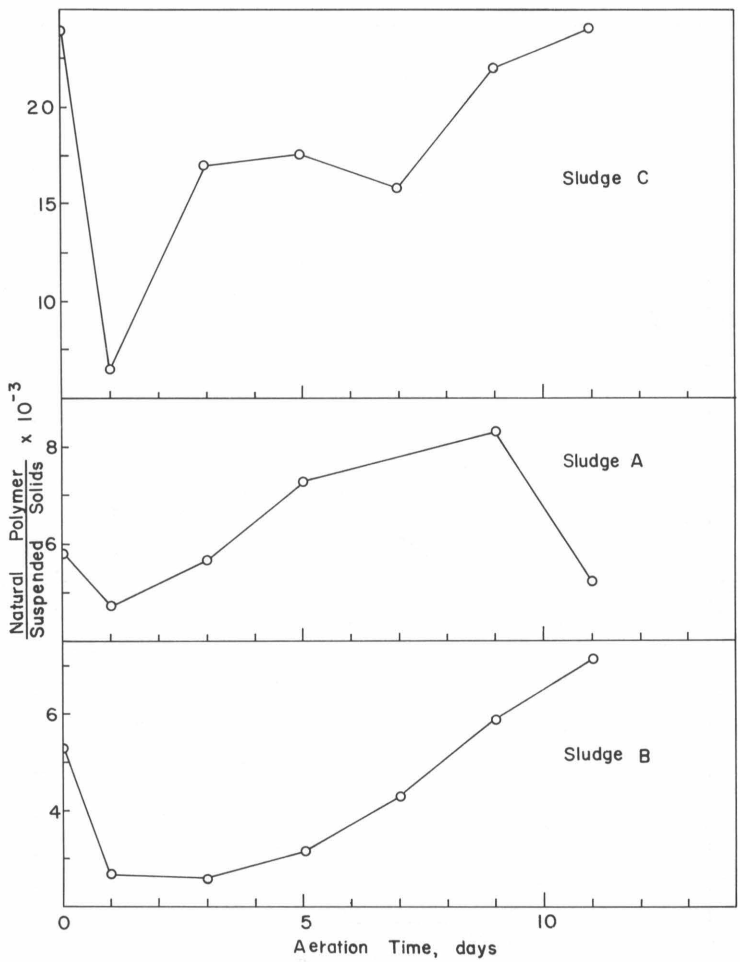


FIGURE 24

Variation in Polymer:Solids Ratio with Sludge Origin



are considered.

From the results of these experiments it was concluded that sludges are affected in a similar manner by biological conditioning regardless of their origin. Greater variation is likely to occur because of differences in sludge age than because of differences in origin.

**VIII. Median Particle Size**

Figure 25 shows the correlation between median particle size, as measured by the developed technique, and the compressibility factor for the three sludges. From the correlation it was concluded that the median particle size closely reflects changes in compressibility for all three sludges. Similar results were obtained from all samples and a reasonably good correlation between particle size and compressibility factor was established by combining all data points into one figure (Figure 26). It is quite likely that a considerable amount of the scatter of the data points was caused by the considerable variation in the suspended solids concentration of the sludge samples.

Specific resistance was found to be an inverse linear function of the median particle size, at least over the range of particle sizes observed. When a pressure of 30 cm Hg was used for filtering, the results were consistent for all sludge samples with one exception. The specific resistance of Sludge A-II decreased with an increase in median particle size to a minimum resistance value for a particle size between 45 and 50 microns, then increased with further increase in particle size.

A generalized plot of all data points confirmed that specific resistance could be correlated to median particle size (Figure 27). Good agreement was obtained from all points, except the three values of Sludge A-II for particle sizes above 50 microns.

On the basis of the previous correlations, it was concluded that activated sludge filterability is strongly affected by changes in the average floc size and that the median particle test developed during this study accurately reflects the changes that take place during aerobic digestion.

<u>Chlorine and Polymer Addition</u>

To further explore the usefulness of the test, a few sludge samples were conditioned separately by chlorine and synthetic polymer addition. The

<u>FIGURE 25</u>

Correlation of Floc Size and Compressibility

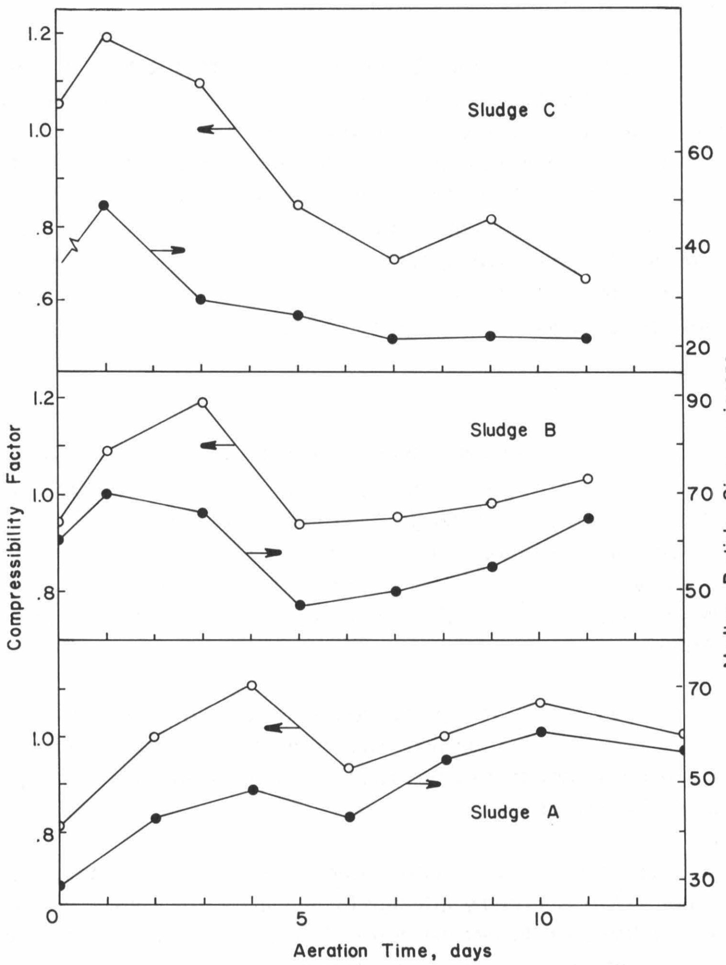
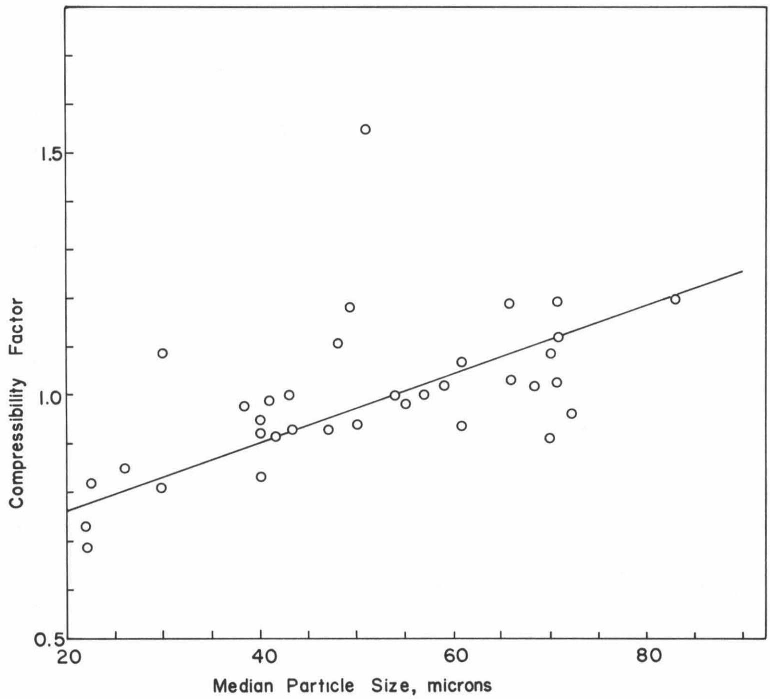


FIGURE 26

Compressibility Factor as a Function of Floc Size



## FIGURE 27

Specific Resistance as a Function of Floc Size

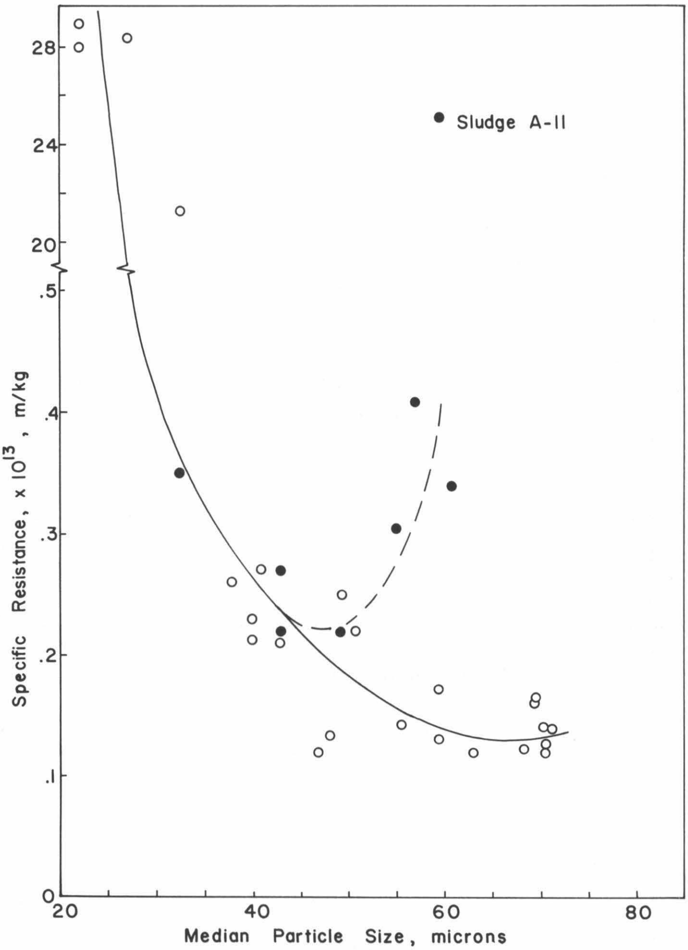



Figure 27. SPECIFIC RESISTANCE AS A FUNCTION OF FLOC SIZE

effect of chlorination was consistent with previous results. The filterability of samples of both sludges A and B were worsened by the chlorine addition. An HTH dose of 250 mg/l increased the specific resistance of Sludge B by a factor of 1.4 and produced a twofold increase in the specific resistance of Sludge A. The 750 mg/l dose caused a specific resistance increase of 1.7 and sevenfold, respectively, for the two sludges. Chlorination also worsened sludge compressibility with one exception. The 250 mg/l dose increased the compressibility coefficient of Sludge A and Sludge B by factors of 1.2 and 1.45, respectively. However, the 750 mg/l dose worsened the compressibility of Sludge B by the factor 1.35, and it improved the compressibility of Sludge A by 20 percent.

Both chlorine doses reduced the median particle size of the sludges and the magnitude of the change was proportional to the dose. The relationships between median particle size and specific resistance were inverse after chlorination as previously observed, but the direct linear relationship between compressibility factor and particle size was not observed after chlorination. In fact, a relationship between median particle size and compressibility factor could not be established after chlorine addition. Apparently, chlorination changed some characteristic other than floc size that affected the compressibility.

Samples of both sludges A and C were conditioned with doses of 10 and 30 mg/l polymer. The specific resistances of the sludges were reduced to values less than one-half the initial values by the 10 mg/l dose and to values less than one-tenth the initial values by the 30 mg/l dose. For Sludge C, the compressibility factor was 1.15, 0.89, and 1.16, respectively, for polymer doses of 0, 10, and 30 mg/l. The corresponding median particle sizes were found to be 49, 85, and >106 microns. For Sludge A, the compressibility factors were 0.89, 0.74, and 0.92 when the median particle sizes were 55, >106, and >106 microns, respectively, for the doses of 0, 10, and 30 mg/l polymer. Median particle sizes greater than 106 microns could not be determined because sufficient screens were not available.

Changes in specific resistance with polymer addition correlated very well with increases in floc size. However, changes in compressibility were more variable with respect to floc changes and did not correlate well. Apparently the addition of artificial polymer to sludges not only produces changes in particle size but also changes factors which affect the compressibility such as the particle density and shape, and the floc strength.

While the median particle size is a measure of both dewaterability parameters during aerobic digestion, when artificial methods of sludge conditioning such as chlorination and polymer addition are used, it indicates changes in specific resistance but is not a reliable parameter for changes in compressibility.

## IX. Synthetic Polymer Conditioning

Harrison [39] has stated that, "even though operating in the endogenous respiration zone dramatically improves settleability and dewaterability, the resultant sludge must still be chemically conditioned to permit vacuum filtration." Since chemical conditioning is always used to improve the mechanical dewatering of waste activated sludge, the relationship between the sludge handling procedures and the conditioning effect of the chemical is of concern. Because the use of synthetic polymers for sludge conditioning is rapidly increasing and because polymers seem to possess the potential for present and future use in mechanical-chemical sludge dewatering, this portion of the investigation was designed to evaluate the effects of some of the sludge handling procedures upon the subsequent polymer doses needed for efficient dewatering.

Since filtration rate was used for most of the experiments instead of specific resistance, the comparative filtration rates of the three sludges are shown by Figure 28. As previously indicated, Sludge C was extremely difficult to dewater compared to the other sludges. Also, during this series of tests, Sludge A was always easier to filter than Sludge B.

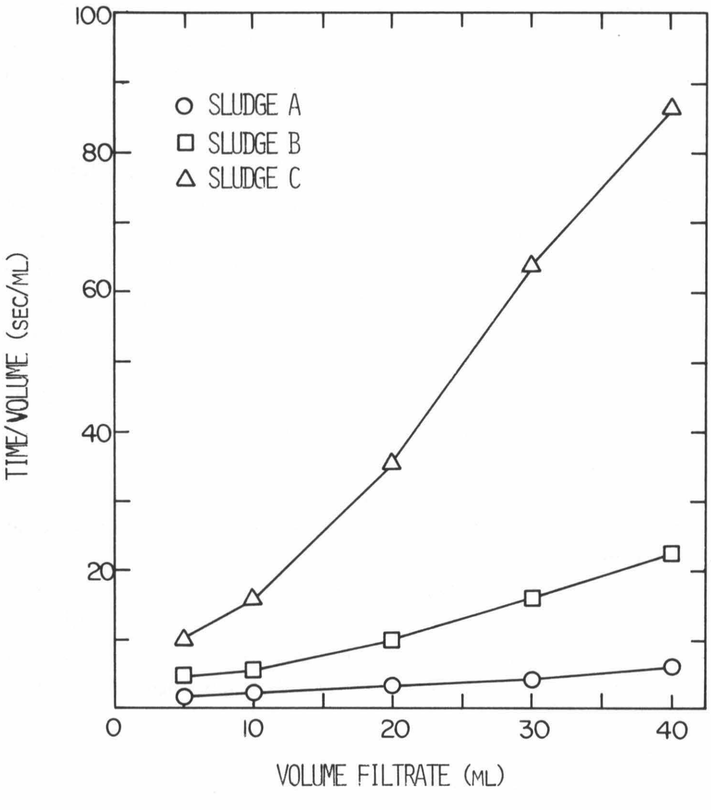
Both cationic and anionic polymers were tested for conditioning effectiveness on the sludges as initially obtained. In all cases the addition of anionic polymer greatly worsened the filtration rates whereas the cationic polymers improved filtration. Following these experiments, a specific cationic polymer, Hercofloc 810, was selected for use throughout the remaining sludge handling studies.

Major emphasis during the study was on defining the relationship between biological conditioning and subsequent polymer requirements. Since previous experiments with 10 to 30 mg/l cationic polymer had shown that the filterability of Sludge A is greatly improved by polymer addition, and since the purpose of the experiments was to detect differences in polymer conditioning following biological conditioning, Sludge A was eliminated from further study because any improvement would be difficult to detect.

The optimum polymer doses for the two sludges were found to be 130 mg/l and 240 mg/l, respectively, for Sludge B and Sludge C. Following deter-

Figure 28

Comparative Filtration Rates of Experimental Sludges



mination of the initial conditioning effects, each sludge was aerobically digested for an extended period of time; samples were removed at various intervals, and changes in the conditioning effectiveness of the previously determined optimum doses were measured. The results, illustrated by Figures 29 and 30, show that addition of the initial optimum dose always produced a large improvement in filtration except for days 5 through 8. During that period, for one day in each case, polymer addition actually decreased the filtration rate. After 10 days of aeration, filtration improvement with polymer addition was always very dramatic but it was never better than the initial filtration rate.

Another aerobic digestion study was conducted wherein the change in optimum polymer dose with aeration was determined. From the results it was determined that aerobic digestion greatly decreased the polymer dose required for optimum filtration during the first 7 days. From the initial optimum dose of 125 mg/l, there was a linear change with time to a low of 25 mg/l at 7 days. There was no further change in the optimum dose up to a digestion time of 20 days when the experiment was terminated. However, while the required dose decreased with aeration time, after 3 days the optimum filtration rate worsened with aeration time (Figure 31). The high peak at day 7 was confirmed by triplicate samples.

The effect of both sludge chlorination and anaerobic storage on the polymer conditioning requirements of the two raw sludges was studied. The addition of 100 mg/l HTH improved polymer conditioning of both sludges while greater doses (250 and 750 mg/l) decreased polymer effectiveness. Anaerobic storage results were less consistent, but in general, 3 hours of anaerobiosis decreased polymer effectiveness whereas one day of anaerobiosis improved it (Figures 32 and 33).

From the results, it was concluded that the conditioning effect of polymers on waste activated sludge can be strongly affected by the sludge handling procedure used prior to mechanical dewatering. From the standpoint of filtration rate, aerobic digestion is seldom beneficial when polymer is used for conditioning, and it will frequently cause a worsening of the polymer effectiveness. However, it may result in a considerable reduction in the amount of polymer required for optimum filtration.

FIGURE 29

Variation in Polymer Conditioning Effect with Digestion (Sludge B)

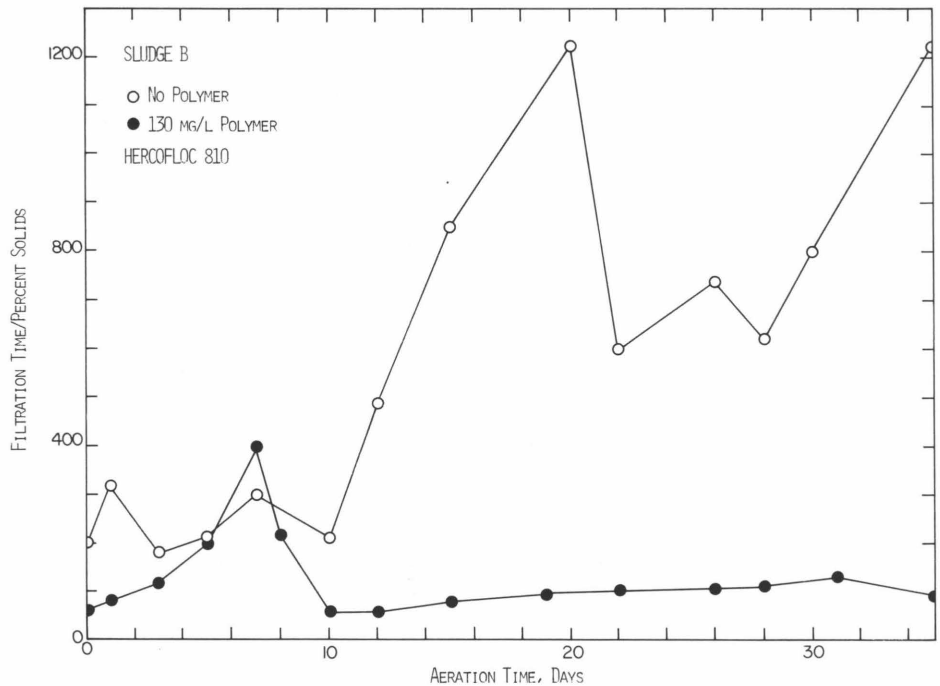


FIGURE 30

Variation in Polymer Conditioning Effect with Digestion (Sludge C)

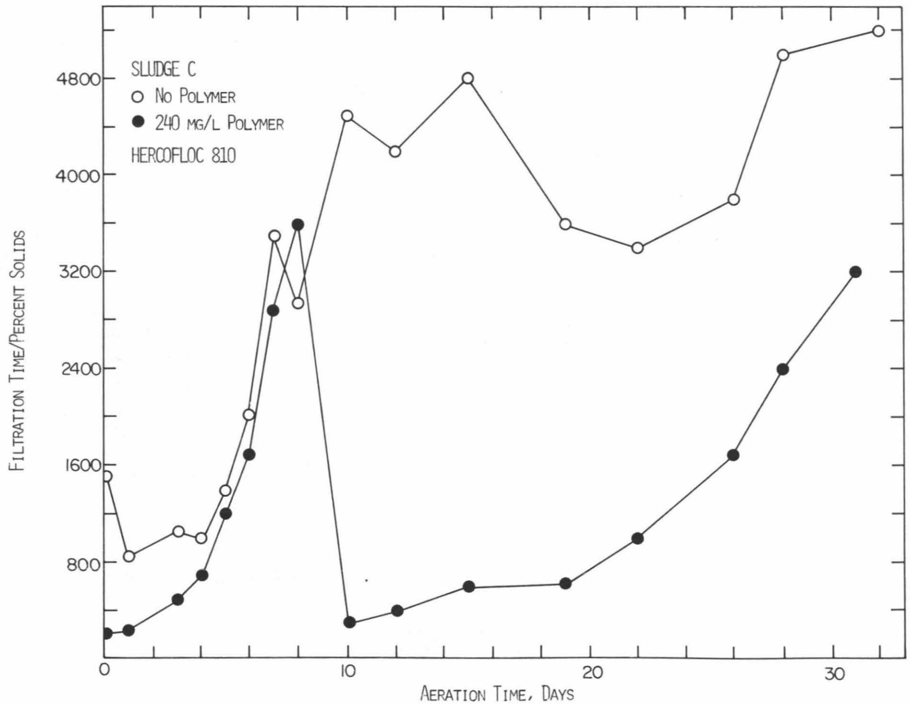
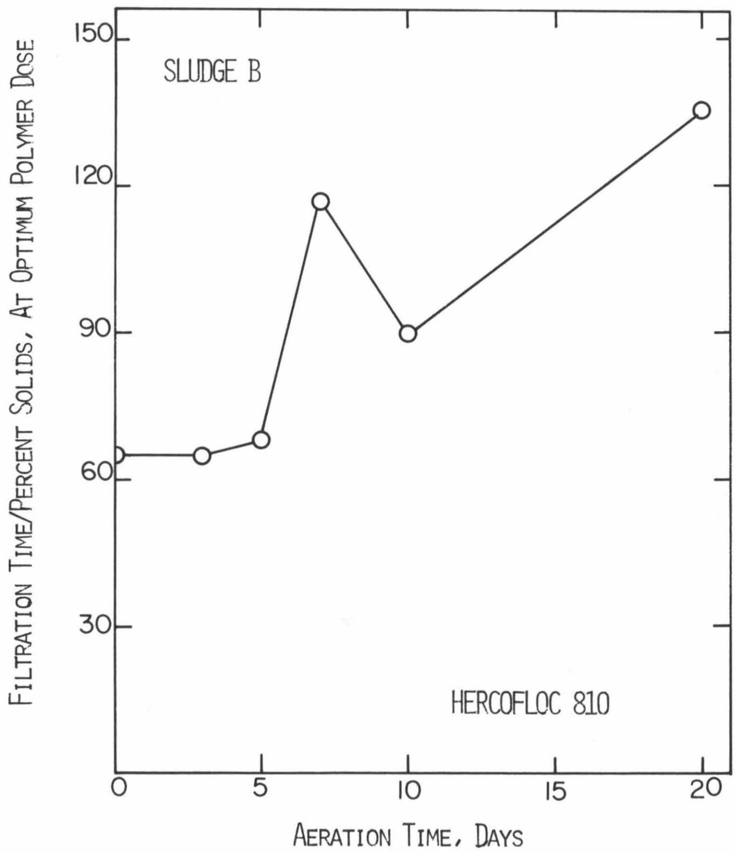


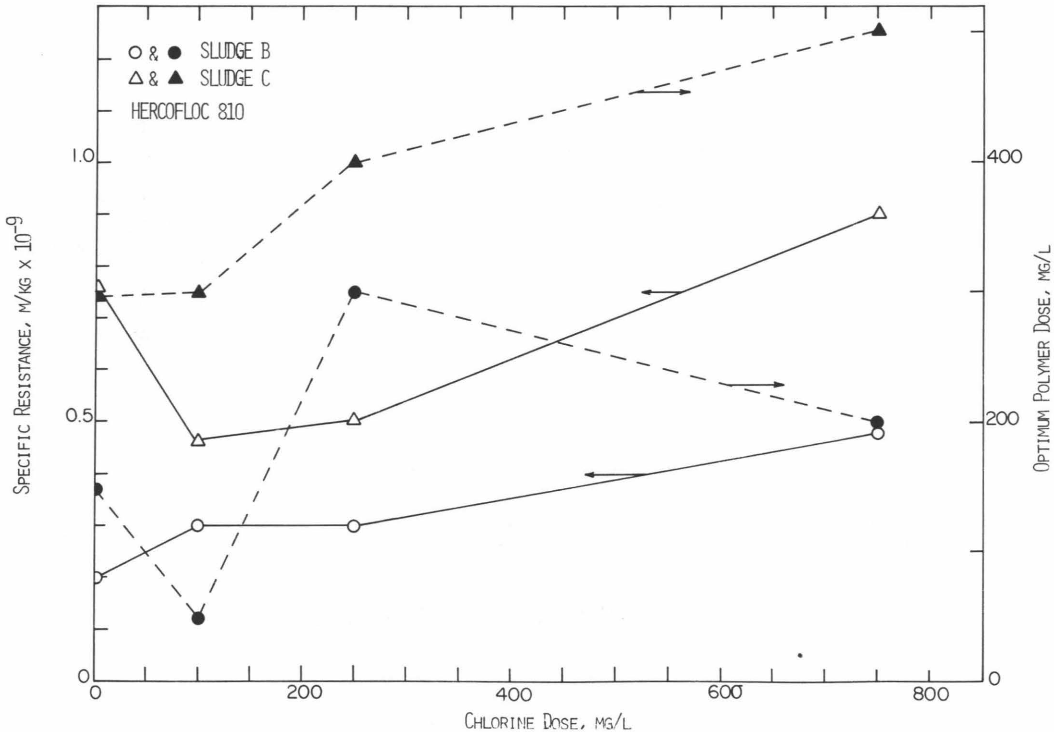
FIGURE 31

Variation in Filtration Rate at Optimum Cationic Polymer Dose
with Aerobic Digestion



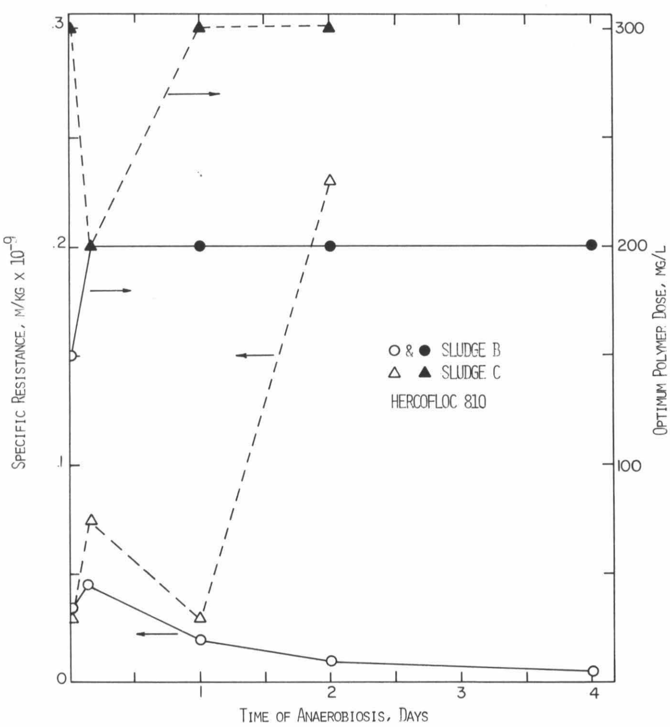
## FIGURE 32

Effect of Chlorination on Polymer Conditioning



## FIGURE 33

Effect of Anaerobic Conditions on Polymer Conditioning



# CONCLUSIONS

A thorough analysis of the experimental results of the total project produced the following conclusions:

1. The manner in which waste activated sludge is handled can have a considerable effect on subsequent dewaterability.

2. Anaerobic conditions are determintal to the filtration characteristics of activated sludge. When the aeration of activated sludge is stopped, the organisms quickly use up the available dissolved oxygen and the filtration characteristics then start to degrade. This degradation has been observed to make filtration continually worse for up to four days.

3. Chlorination of waste activated sludge drastically reduces the subsequent filtering rate. Specific resistance increases of more than 700 percent were observed. Chlorination generally has a detrimental effect on compressibility but some exceptions were noted when the dose was very high.

4. Aerobic digestion has a considerable effect on the filtration characteristics of waste activated sludge and it produces changes in both specific resistance and the compressibility factor.

5. During aerobic digestion, improvement in dewaterability occurs initially with maximum improvements after one to five days aeration. However, further aeration; i.e., aerobic digestion, will result in a worsening of dewatering properties and can produce conditions far worse than the initial state.

6. Filtration characteristics degraded during anaerobic conditions or by chlorination can be recovered by subsequent aeration. The period of aeration required for recovery is similar to that required to reach optimum filterability before biological stress occurs.

7. It is necessary to maintain activated sludge in a viable, cellularly intact condition to achieve favorable filtration rates and biological activity must occur for dewaterability to improve during aerobic digestion. The biological stress on the sludge flocs, and therefore the filterability of the sludge, is directly reflected by the organic matter of the

supernatant.

8. Temperature is a factor that affects sludge filtration. A given sludge at a higher temperature will filter faster than the same sludge at a lower temperature. Variation in sludge filtration due to temperature can be decreased by aerobic digestion which illustrates that this variation is caused by sludge properties as well as viscosity effects.

9. The degree of improvement in dewaterability that occurs during aerobic digestion is a function of the origin and nature of the fresh sludge, including its sludge age, the rate of aeration during digestion, the temperature of digestion, and the time of digestion. Filterability can usually be improved by 23 to 46 percent by biological conditioning and improvement as high as 62 percent was observed. However, the sludge must be filtered as soon as maximum improvement is attained in order to realize the benefit since further digestion will worsen filterability.

10. The rate at which sludge is mixed during aerobic digestion influences the dewatering properties of the sludge. It appears that the more rapidly a sludge is mixed the greater the forces placed on the sludge floc particles and the poorer the filtration rate. Mixing is also detrimental to the dewatering of non-aerobic sludge. Mechanical mixing of a sludge when dissolved oxygen is absent causes a more rapid degradation in sludge filtering properties than when the same sludge is not mixed.

11. Sludge digested in a continuously fed aerobic digester dewaters more readily than sludge digested in a batch aerobic digester but the difference is small.

12. Dissolved oxygen concentrations above 2 mg/l during digestion do not change the subsequent sludge filterability. Therefore, concentrations above this level are of no benefit.

13. Aerobic digestion affects the dewatering of sludge on sand drying beds. The time required to drain 750 ml from a one liter sample decreased by about 73 percent after 6 days of aeration and there was an increase in the total drainable water of 2 percent. Both of these effects would improve the overall dewatering of sludge on sand drying beds.

14. Changes in the filtration properties of activated sludge are related to

biological, chemical, and physical changes that occur during aerobic digestion but biological viability and cellular integrity control dewaterability, and parameters such as zeta potential have little or no effect.

15. The supernatant exocellular polymer concentration drops to a minimum value after one or two days of aerobic digestion, but then typically increases continually with additional digestion time. High ratios of natural polymer to microbial solids are apparently detrimental to dewatering. It appears that there is an optimum concentration of polymer per unit solids for best filterability and this value is easily exceeded during aerobic digestion. Under some conditions the normally biologically-resistant polymers can be metabolized.

16. Changes in specific resistance and compressibility factor are most reliably reflected by changes in median particle size as compared to the other parameters measured during this investigation. Specific resistance is inversely related to floc size, whereas compressibility factor is directly related to floc size.

17. Chlorine addition reduces median particle size and results in an increase in specific resistance. The magnitude of the change is related to the length of time the sludge was aerated prior to chlorination. Chlorination generally worsens compressibility but large doses may improve it for some sludges.

18. The addition of artificial polymers greatly increases the median particle size and drastically reduces the specific resistance. The extent of the effect is a function of the degree of aerobic digestion. Changes in compressibility factor with artificial polymer addition are highly variable but are also affected by aerobic digestion.

19. The dewatering properties of waste activated sludge vary with the origin and degree of stabilization of the particular sludge. The conditioning effect of artificial polymers is affected by the same variables.

20. Polymers are very specific in their conditioning action on waste activated sludge. Anionic polymers worsen filterability initially and after all periods of aerobic digestion. Cationic polymers improve filterability initially and almost all the time after periods of aerobic digestion. There is also considerable variation in the conditioning effect of different types of cationic polymers.

21. Generally speaking, polymer addition will prevent the extreme filtration times that occur after long periods of aerobic digestion. However, when the initial optimum polymer dose is used throughout the digestion period, virtually no improvement in filterability can be obtained with aerobic digestion, and a worsening of the filtration rate usually results. The worst results were obtained on the seventh day of digestion.

22. Aerobic digestion can produce a considerable decrease in polymer dose for sludge conditioning. An 80 percent decrease was obtained after 7 days of digestion, and the optimum dose remained constant after 7 days. The decreased optimum dose is partially offset by a worsening of the optimum filtration rate with digestion, particularly after 3 days of aeration.

23. In general, chlorination prior to polymer conditioning was detrimental to the dewatering process; however, with a poorly stabilized sludge, the addition of 250 mg/l or less HTH improved filterability. A dose of 750 mg/l to the same sludge worsened filterability.

24. Short periods of anaerobiosis are detrimental to polymer conditioning. On the other hand, anaerobic storage for one day or more tends to improve polymer conditioning although the effect may shift back with additional storage. It was further noted that anaerobiosis produces a shift in the optimum polymer dose.

In summary, for the most efficient handling, waste activated sludge should be maintained in a viable, aerobic condition prior to dewatering. Anaerobic conditions will destroy waste activated sludge filterability and produce a poorer quality supernatant, as will any other handling procedure that is destructive to the biological cells. The dewatering properties of the sludge can be very significantly improved by biological conditioning and/or synthetic polymer addition. The dewaterability of the activated sludge is most clearly reflected by the median particle size of the floc and measurements of this parameter could be used to control sludge handling processes.

## REFERENCES


1. Manchester University, "Sludge Drying," Effluent and Water Treatment Journal, 3 789, London, England (May, 1968).

2. Randall, C. W., Turpin, J. K., and King, P. H., "Activated Sludge Dewatering: Factors Affecting Drainability," Water Pollution Control Federation Journal, 43, 1, 102 (January, 1971).

3. Morris, R. H., "Polymer Conditioned Sludge Filtration," Water Works and Wastes Engineering, 2, 68 (1965).

4. Albertson, O. E., and Guidi, E. J., Jr., "Advances in the Centrifugal Dewatering of Sludges," Water & Sewage Works, 114, R-113 (1967).

5. Burke, J. T., "Organic Polymers in the Treatment of Industrial Wastes," Proceedings 21st Industrial Waste Conference, Purdue Univ. Ext. Ser. 121 303 (1966).

6. Ettelt, G. A., and Kenndey, T. J., "Research and Operational Experience in Sludge Dewatering at Chicago," Water Pollution Control Federation Journal, 38, 2, 248 (February, 1966).

7. Water Pollution Research Laboratory, "Water Pollution Research 1965," Her Majesty's Stationery Off, London, England (1966).

8. Randall, C. W., and Koch, C. T., "Dewatering Characteristics of Aerobically Digested Sludge," Water Pollution Control Federation Journal, 41, 5, R 215 (May, 1969).

9. Tenney, N. W., Echelberger, W. F., Jr., Coffey, J. J., and McAloon, T. J., "Chemical Conditioning of Biological Sludges for Vacuum Filtration," Proceedings, 24th Purdue Industrial Waste Conference, (May, 1969).

10. Lawton, G. W., and Norman, J., "Aerobic Sludge Digestion Studies," Water Pollution Control Federation Journal, 36, 4, 495 (April, 1964).

11. Randall, C. W., Saunders, F. M., and King, P. H., "Biological and Chemical Changes in Activated Sludge During Aerobic Digestion," Proceedings, 18th Southern Water Resources and Pollution Control

Conference, North Carolina State University, Raleigh, North Carolina (April, 1969).

12. Jackson, S., "Recent Research and Development Work at the Water Pollution Research Laboratory," Abstracts of Technical Papers, 42nd Annual Conference, Water Pollution Control Federation, Session No. 13, Dallas, Texas (1969).

13. Burd, R. S., "A Study of Sludge Handling and Disposal," Publication WP-20-4, U. S. Department of the Interior, Federal Water Pollution Control Administration, Office of Research and Development (May, 1968).

14. Tenney, N. W., and Cole, T. G., "The Use of Fly Ash in Conditioning Biological Sludges for Vacuum Filtration," Water Pollution Control Federation Journal, 40, R281-302 (August, 1968).

15. Scott, G. W., and Cornwall, G. M., "Filterability of Lime-Iron Sludges Generated in the Chemical Precipitation of Phosphorus from Treated Sewage," Sixth Canadian Symposium on Water Pollution Research, Waterloo, Ontario (February, 1971).

16. Randall, C. W., Moore, H. R., and King, P. H., "The Effect of pH on Aerobic Sludge Digestion," Proceedings, Mid-Atlantic Industrial Waste Conference, 5, pp. 431-457, Drexel University, Philadelphia, Pennsylvania (November, 1971).

17. Carman, P. C., "A Study of the Mechanism of Filtration Part I," Journal of the Society of Chemical Industry, 52, (1933).

18. ______ "A Study of the Mechanism of Filtration Part II: Experimental," Journal of the Society of Chemical Industry, 53 (1934).

19. _____ "A Study of the Mechanism of Filtration Part III," Journal of the Society of Chemical Industry, 53 (1934).

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

21. Sanders, T. G., "A Mathematical Model Describing the Gravity Dewatering of Wastewater Sludge on Sand Drainage Beds," Master's Thesis, University of Massachusetts, Amherst, Massachusetts (1968).

22. Nebiker, J. H., Sanders, T. G., and Adrian, D. D., "An Investigation of Sludge Dewatering Rates," Proceedings, Purdue Industrial Waste Conference, 23, Engr. Ext. Series 132, pp. 832-848 (1968).

23, McKinney, R. E., "Biological Flocculation," Biological Treatment of Sewage and Industrial Waste, I, New York: Reinhold (1956).

24. Bhatla, M. N., "Relationship of Activated Sludge Bulking to Oxygen Tension," Water Pollution Control Federation Journal, 39, (December, 1967).

25. Albertson, J. G., et al., Investigation of the Use of High Purity Oxygen Aeration in the Conventional Activated Sludge Process, Water Pollution Research Series 17050DNW05/70, Federal Water Quality Administration, U. S. Department of the Interior (May, 1970).

26. Ball, J. E., Humenick, M. J., and Speece, R. E., "The Kinetics and Settleability of Activated Sludge Developed Under Pure Oxygen Conditions," Technical Report EHE-72-18, CRWR-94, Civil Engineering Department, University of Texas, Austin, Texas (August 1972).

27. Jones, B. R. S., "Vacuum Sludge Filtration, Prediction of Filter Performance," Sewage and Industrial Wastes, 28, 9, (September, 1956).

28. Jaworski, N., Lawton, G. W., and Rohlich, G. A., "Aerobic Sludge Digestion," Conference on Biological Waste Treatment, Manhattan College, New York (April 20-22, 1960).

29. Eckenfelder, W. W., Industrial Water Pollution Control, New York: McGraw-Hill Book Company (1966).

30. Manual of Instruction for Sewage Treatment Plant Operators, Albany, New York: New York State Health Education Service.

31. Heukelekian, H., and Weisberg, E., "Bound Water and Activated Sludge Bulking," Sewage and Industrial Wastes, 28 (April, 1956).

32. Standard Methods for the Examination of Water and Wastewater, 12th Edition, New York: American Public Health Association (1965).

33. Simplified Laboratory Procedures for Wastewater Examination, W.P.C.F. Publication No. 18, Water Pollution Control Federation, Washington, D. C.

34. Ford, D. L., Yang, J. T., and Eckenfelder, W. W., "Dehydrogenase Enzyme as a Parameter of Activated Sludge Activities," Proceedings, 21st Industrial Waste Conference, Purdue University, Lafayette, Indiana, 534 (1966).

35. Ramanathan, M., Gaudy, A. F., Jr., and Cook, E. E., "Selected Analytical Methods for Research in Water Pollution Control," Center for Water Research in Engineering, Oklahoma State University, Stillwater, Oklahoma.

36. Cameron, John W., "Aerobic Digestion of Activated Sludge to Reduce Sludge Handling Costs," Abstracts of Technical Papers, Session 29, 45th Annual Conference Water Pollution Control Federation, Atlanta, Georgia (1972).

37. Busch, P. L. and Stumm, W., "Chemical Interaction in the Aggregation of Bacteria," Environmental Science and Technology, 2, 49 (1968).

38. Tenney, N. W., and Stumm, W., "Chemical Flocculation of Microorganisms in Biological Waste Treatment," Water Pollution Control Federation Journal, 37, 1370 (1966).

39. Harrison, J. H., "Solids Handling in Activated Sludge Plants," Report, Black, Crow & Eidness, Inc., Consulting Engineers, Wilmington, Delaware (September, 1971).

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