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A FEASIBILITY STUDY OF A LABORATORY MODEL

AS A RESEARCH TOOL FOR IMPOUNDMENT

WATER QUALITY INVESTIGATIONS

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CHAPTER I

INTRODUCTION

To meet the demands of an increasing population, there has been a corresponding increase in the construction of surface-water impoundments. Such impoundments are built primarily for the purpose of retaining a volume of water on the surface that normally would be "lost" to a less accessible phase of the hydrologic cycle. Coupled with increasing population and economic growth are the divergent, often conflicting, needs and desires of water consumers. Accordingly, these impoundments are usually multi-purpose systems designed to provide volumes of water for several different uses, such as domestic drinking water, irrigation, flood control, hydroelectric power, recreation, and to regulate downstream water quality and quantity. Therefore, when such a system is designed, space is allocated for each purpose as shown in Figure 1.

Although there have been allocation problems in such systems, when properly designed they usually function very well insofar as water quantity is concerned. However, there is a very close interrelation between water quantity and water quality. A study of the literature reveals numerous effects on water quality resulting from impoundment, many of which are detrimental.

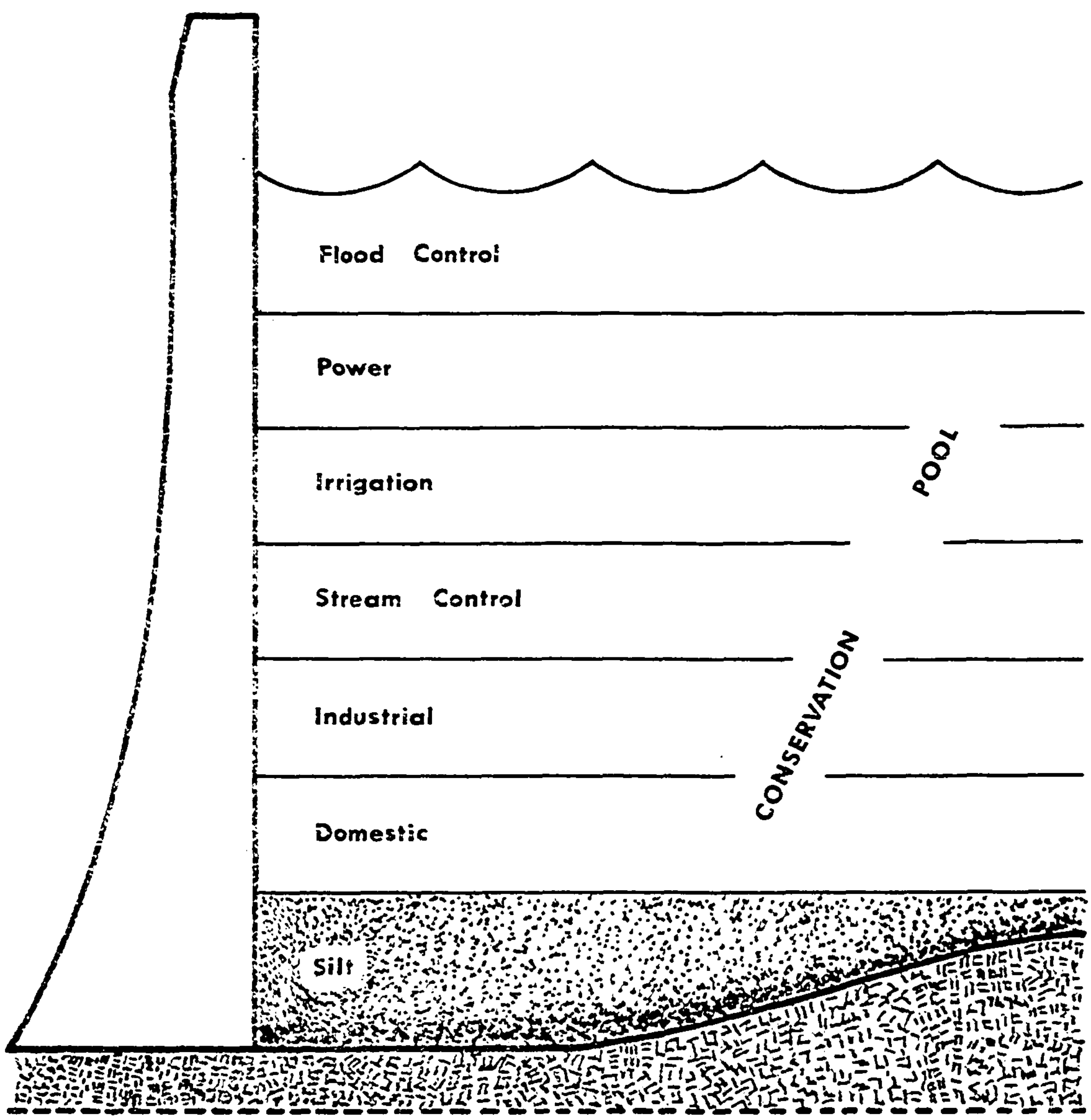

Figure 1. Multi-purpose Impoundment Storage Allocation
(After Oklahoma Water Resources Board, Pub. 18, 1968)

Symons, et al., (1) listed both the beneficial and detrimental effects of impoundment. A summary of these effects is shown in Table 1.

TABLE 1

IMPOUNDMENT EFFECTS ON WATER QUALITY

Possible Benefits	Possible Detriments
Turbidity Reduction	Less Mixing
Hardness Reduction	Less Reaeration
Organic Reduction	Backup of Pollutants
BOD Reduction	Algal Blooms
Color Reduction	Aesthetics
Coliform Reduction	Tastes and Odors
Smoothing Action	No Bottom Scour
	Thermal Stratification
	Low DO
	Iron and Manganese Increase
	H_2S Production
	CO_2 Increase
	pH Decrease
	Organic Persistance

Symons attributes reduced turbidity to low velocity and long detention times. Reduced hardness can occur due to algal consumption of carbon dioxide and subsequent precipitation of calcium carbonate. Impoundment usually causes a reduction in BOD and color due to biodegradation. Coliform reduction is attributed to long detention times allowing natural die-off. Due to large reservoir volumes, there tends to be a smoothing action on incoming pollutants. However, less mixing could allow wastes to accumulate in one region. There is usually less atmospheric reaeration due to lower velocities and increased depth. There are often increased algal problems due to nutrient buildup that can cause taste and odor problems and filter clogging. Because of increased depth there is usually no bottom scour, thus allowing organic sediment buildup. Also, with increased depth, thermal stratification often develops leading to the lowering of water quality in the hypolimnion due to reducing conditions. As can be seen, most of the effects in Table 1 can be attributed, directly or indirectly, to long detention times.

It should be pointed out that these effects shown in Table 1 are quite general. The literature reveals an almost countless number of articles dealing with the more specific effects on water quality attributable to impoundment. Symons (2) reviewed over 600 literature references in a compilation dealing with water quality behavior in reservoirs. Again, many of the works cited indicate that serious water quality problems can result from impoundment.

Since water is a reusable commodity, many of the water quality problems that develop in our impoundments are attributable to pollution from domestic and industrial wastes. Nonetheless, water quality problems have developed in impoundments that have not been exposed to man-made pollutants. Such problems, regardless of the cause, generally have the same end result--an increase in the cost of water, a decrease in the quantity of water available for a certain use, and a decrease in the aesthetic and recreational value of the water.

It would therefore seem desirable to incorporate consideration for water quality as well as quantity when designing an impoundment system. Watershed control by contour plowing and vegetative covering has been shown to be useful for improving both water quantity--due to increasing base level flows--and water quality--due to erosion and silt control. Selective discharge can often upgrade downstream water quality. Unfortunately, such discharge is usually not possible due to the quantity requirements demanded of multi-purpose systems and it usually has little beneficial effect on the water in the impoundment. Although these examples can and are being used in some cases to upgrade water quality, they do not represent predictive design techniques--techniques that grow out of a fundamental understanding of the impoundment system. Thus, although entire river basin planning techniques can adequately predict the cost/benefit ratio of an impoundment and provide volumes of water to meet expected consumer demands, there exists little predictive design capability insofar as water quality is concerned.

Symons (2) offers the following explanation for the lack of predictive design capability:

> "In spite of all the research, sufficient information is not available to permit the choice of one impoundment site from among several in a given geographical area that will have the minimum adverse effect on the quality of water and that will, therefore, provide the best quality water. Further, if an impoundment site is fixed, sufficient information is not available to permit prediction of how the incoming water will be changed as it stands in this impoundment. Finally, because of this lack of understanding, firm recommendations cannot be made relating to the outlet structure design, impoundment operating practices, or in-impoundment control, since the influence of changes in these items on discharged water quality is not known."

Symons (2) continues by listing three reasons why, with the tremendous amount of research in the appropriate areas, that the influence of impoundments on water quality cannot be predicted. First, he maintains that much of the data generally applicable to the understanding of impoundments has not been taken with impoundment behavior specifically in the researcher's mind. As an example, he notes that many limnologists and aquatic biologists have related growth of algae to light intensity and penetration without relating this to decreased turbidity due to impounding a turbid river. Secondly, many investigations lack the fundamental nature that would allow the observed findings to be projected into a new set of environmental conditions. Thirdly, impoundment environmental conditions differ from those of streams in many ways. He maintains that changes in water depth, increased detention time, and thermal stratification each influence the normal biological, physical, and chemical processes that lead to changes in water quality.

Silvey (3) suggests that part of the lack of solutions to water quality problems is one of basic philosophy; that is, that impoundments are being studied under the ground rules developed for the study of natural lakes. He compares, in a general manner, the differences and similarities between natural lakes and impoundments. He also compares the advantages and disadvantages of deep and shallow impoundments and concludes that reservoir shape and type of basin play an important role in controlling water quality. As an example of the ability of reservoir shape to control water quality, it is pointed out that when materials such as iron, manganese, and phosphate are lost from solution in a shallow impoundment, they are usually lost permanently because little reduction can occur in the aerobic zone.

Sylvester and Seabloom (4) also feel that basin type should be an important consideration when designing an impoundment. These authors suggest that attention should be given to the effect a particular impoundment site will have on the stored water quality. They state:

> "In selecting a site for water storage, attention should be given to the effect this particular impoundment site may have on the overlying water quality. Many of the

> site characteristics, such as future water depth, configuration, orientation to the prevailing wind direction, relative volume of inflow to storage capacity, quality of incoming water, geology of surrounding terrain, character of the original underlying soil, plus the nature and extent of vegetation and forestation may effect the impounded water quality. Climatic factors such as rainfall, temperature, wind and sunlight also may be important influences."

The authors further state "...it is necessary for engineers to adopt a more scientific approach to the location, design and operation of future impoundments, insofar as their effect on water quality is concerned."

From the preceding discussion, it seems reasonable to divide current research efforts into two broad categories. The first such category would pertain to fundamental research being done, for the most part, on already established impoundments. The goals of such research would be to obtain a better understanding of the behavior of impounded waters by investigating new relationships, reevaluating established relationships, and to relate such findings to the causative factors. The second category, which necessarily overlaps somewhat with the first, would be of an applied nature. The efforts here would be directed toward obtaining a predictive design capability for impoundment systems based on knowledge gained from the first category. It is hoped through such a dual research effort to aid the solution of problem areas in already established impoundments and also, through such techniques as preimpoundment site analysis, to detect and prevent many problems before the impoundment is constructed.

Due to the complexity of impoundment systems--the interactions of the multiple variables involved--research in both of the categories mentioned above has been painstakingly slow. As discussed earlier, there is almost no mention in the literature on the effect of various external environmental factors (light intensity, temperature, wind, etc.) on the various aspects of the impoundment ecosystem. Quantitative correlation studies between even one external environmental factor and some single aspect of an impoundment usually meet with frustration. It is because of this confusing complexity that research methods are being sought to simplify the system. Attempts are being made to control the variables involved with the hope of being able to apply the techniques of systems analysis.

Several research techniques are currently being utilized in an attempt to both simplify the system and to gain some control over the external environmental factors mentioned earlier. One method of approach employs what is usually referred to as a "physical model," "artificial ecosystem," "simulated natural system," or "laboratory model." These terms are not to be confused with such concepts as pure culture flasks. While culture studies have provided much useful information, it has become apparent that chemical and biological species respond differently within a community unit than in the simplified environment of the culture flask. McIntire (5) believes that the laboratory model is one of the most promising new research approaches, and if designed properly, should allow some control over the environment while retaining many of the properties of natural ecosystems. McIntire's work involved investigation of the productivity and bioenergetics of benthic algal communities, and an effort to relate these findings to controlled environmental factors such as light, carbon dioxide supply, DO, temperature, and current velocity. Although this work was concerned with stream laboratory models, the underlying philosophy is much the same as with impoundment models. In either case, the work involved does demonstrate the potential of laboratory models as another tool for the investigation of freshwater systems. The author concludes that laboratory models are best used to gain information that can supplement and help in understanding concurrent observations in the field.

In a very informative paper, Cairns (6) further expands McIntire's belief in laboratory models. Cairns feels the development of a capability for prediction is critical to the future role of biologists. He believes that the use of simulation techniques could allow study of alternative uses of the environment and that the consequences of each use could be estimated in the planning stages. It is also suggested that in the future, before implementation of some plan such as impoundment construction, that a model system be used as a sort of "proving ground"--hopefully a place to which we can restrict our environmental mistakes.

The literature contains very few descriptions of simulated natural systems, especially laboratory impoundment models. Even fewer articles pertain to the study of models in an attempt to determine their potential or feasibility as a research tool. It does seem apparent, however, that without emphasizing the modeling concept, that many investigators do incorporate models into their experimental design.

Gahler (7), in an investigation of sediment-water nutrient interchange, utilized both field and laboratory determinations of interchange. For the field studies of sediment-water nutrient interchange, four polyethylene pools were placed in the lake being investigated. Two of the pools were without bottoms, thus allowing comparison of the lake water trapped in the pools exposed to lake sediment with water trapped in the pools with bottoms. The physical, chemical, and biological variables of the water in the pools and that of the surrounding lake were observed for seven months. The author points out that although testing interchange under field conditions would be ideal, it is experimentally difficult because of the length of time involved, the installation expense, and the difficulties encountered in monitoring such *in situ* experiments. For the laboratory determinations, lake sediment was placed in the bottom of 6-inch diameter, 6-foot high glass columns. Lake water was then placed in the columns and monitored for any change in nutrient concentration that could be attributable to the underlying sediment. The water in the columns was maintained under both aerobic and anaerobic conditions. Further laboratory studies involved placing either lake or distilled water over sediment in aquaria. Mackenthun (8) also used aquaria "models" in much the same manner as Gahler. Using ^{32}P, Mackenthun found that the amount of phosphorus released to the overlying water was very small. Further laboratory studies revealed that ^{32}P placed only 1/4 inch below the mud surface was practically lost from the system. The aquaria experiments did reveal, however, that circulation of the water above the sediment, with the aid of air bubbles, did increase the phosphorus in solution.

In an attempt to determine the effect of impoundment-site soil on the water quality, Sylvester and Seabloom (4) also used large columns and water from the river that was to be impounded was placed in the columns. The columns were then subjected to various environmental conditions. The objectives were to obtain enough information to predict the effect this soil would have on the future water quality of the impoundment. The results indicated that water standing over organic soil for a period of several weeks became noticeably colored, assimilated enough nutrients to support large algal blooms, and was usually deficient in dissolved oxygen. They also found that algal growths produced changes in the overlying water quality which could exceed the quality changes due only to soil-water contact. An important aspect of this research revealed that a covering of mineral soil would suppress the detrimental effects of organic soil on the overlying water quality.

It is encouraging to find that the literature does contain some references that pertain primarily to the study and utilization of laboratory models. Strickland, et al., (9) made use of a deep tank to study the growth and composition of phytoplankton crops at low nutrient levels. The tank was described as a cylinder, 3 meters in diameter by 10 meters tall. The inside of the tank was coated with Laminar X 500, a black plastic finish that is inert toward growing phytoplankton. The outside of the tank was insulated with polyurethane to insure constant temperature. Photocells located within the tank at various locations measured the attenuation of light by the plankton as it grew. A 1,200-w Honovia Englehardt mercury arc tube could be lowered into the tank for sterilization purposes. Incoming water could be filtered and cooled. Three 5-kw heating elements could raise the water temperature by 1.5°C/day. Air bubbles could be introduced at the bottom of the tank. Artificial lighting could provide photosynthetic light to almost any depth. The authors also compared the advantages and disadvantages between laboratory models and *in situ* submerged, translucent spheres or cylinders. They concluded that, although the *in situ* experiments have many advantages over laboratory models, the *in situ* experiments suffered from many of the same drawbacks mentioned earlier by Gahler (7).

In order to investigate the fate of synthetic organics in stratified impoundments, DeMarco, *et al*. (10) designed a laboratory model impoundment to duplicate thermally and biologically a stratified impoundment. The 500 gallon tank was 3 feet in diameter by 12.5 feet high. The top water of the model could be heated and circulated and the bottom water kept cold and stagnant. It was possible to duplicate on a small scale the temperature and DO regime found in a typical stratified impoundment in the field. The tank was filled with water from a near-by river and make-up distilled water was added to account for evaporation losses. The synthetic organics under study were added to the tank and dispersed throughout with a high speed mixer. The tank was then thermally stratified and glucose was injected into the cold bottom waters to obtain zero DO conditions. The disappearance of the synthetic organics under study was periodically monitored by taking samples from the upper (aerobic) and lower (anaerobic) zones. Results indicated that the synthetic organics were broken down very slowly in the lower zone, as compared to the upper zone.

Although "models" are being used for a host of different studies ranging from eutrophication to bioassay, the current state of the art indicates that certain difficulties seem common to many cases. One of the most difficult problems encountered in laboratory models is the selection of a method to "feed" the model. "Feed" denotes a method for supplying water and constituents to the model in such a manner as to be representative of the sources assumed available to the natural system under study. According to Holm-Hansen (11), the nutrient supply to a natural system is continually being replenished by a variety of routes; therefore, an equilibrium is established between nutrient removal and nutrient replenishment. In laboratory tests that utilize natural waters, the nutrients are soon depleted and the routes of replenishment are cut off. Thus the dynamic processes that exist in nature must be incorporated into the design of model systems if such systems are to be maintained under laboratory conditions.

Undoubtedly, one of the most important sources of constituents to impoundments is the soil in the reservoir basin and in the surrounding watershed. As mentioned earlier by Silvey (3) and Sylvester and Seabloom (4), the shape and location of an impoundment will determine, in part, the behavior of the detained water. Sylvester and Seabloom state further that the soil of the intended basin and of the surrounding watershed plays an important role in determining water quality.

Below is a summary of the processes by which impounded water may have its quality altered when in contact with the soil, according to Sylvester and Seabloom (4).

1. Ion exchange through the clay and humic colloids in the soil.

2. Release of dissolved materials due to the microbiological degradation of organic materials, and carbon dioxide production that dissolves important nutrients such as phosphorus, calcium, and magnesium present in the mineral portion of the soil.

3. Leaching of organic and mineral substances from the soil or vegetation which may support algal growth, and a production of additional organic material with the added end products of decomposition.

4. Microbiological activity at the soil-water interface which depletes the dissolved oxygen, possibly causing anaerobiosis and a change in the products of decomposition.

The effects of soil on water due to the processes listed above are three-fold and are listed by the authors below:

1. Physical: by changing color, turbidity, and taste and odor characteristics of the water.

2. Chemical: whereby the pH is affected and the nature and amounts of the various dissolved solids and gases are influenced.

3. Biological: where nutrients are provided for growth of algae and other aquatic organisms.

Gjessing and Samdal (12) also feel the nature of the soil in the impoundment area should be considered. Humic color, usually attributed to plant decomposition compounds such as polyphenolics being leached from the soil, can result from new impoundments or by a changing water level in an established impoundment. Further observations indicate that humic color intensity and COD usually decrease with time, probably due to biologic processes.

Borchardt (13) states that soil in the surrounding impoundment watershed can be involved in a cycle leading to eutrophication. He maintains that an average lake retains 30-60 percent of the nutrients washed in from the watershed. This is coupled with the knowledge that Azotobacter, among the bacteria, and the blue-green algal forms have the ability to fix nitrogen from the atmosphere. Borchardt further states that carbon, as fixed by plant life, is an essential condition which begins the "fertility race." Decomposition of these plants produces organic acids that in turn release phosphorus and other elements from the mineral matter in the soil. It is argued, therefore, that the very presence of plants induces the generation of additional fertilizers which in turn enhance the development of more plant life and additional impoundment fertility.

Weiss (14) found that quantities of humic materials entering a lake were extremely stimulating to algal cultures; this growth stimulation was attributed to the chelating and buffering action of humates. He remarks that time and time again, algal cultures fail to grow unless extracts from soil or soil-water mixtures are added. Mackenthun (8) states: "In a freshwater environment, algal requirements are met by vitamins supplied in soil runoff, lake and streambed sediments, solutes in the water, and metabolites produced by actinomycetes, fungi, bacteria, and several algae."

Connors and Baker (15) point out that in some reservoirs, algal biomass is a minor contributor to the total organic matter present. It can often be the case that the total organic load, as determined by COD, is established by the character of the runoff-transformations that take place. The laboratory results of an interesting experiment demonstrated that soils placed in distilled water reached an equilibrium that showed the water to be a certain hardness. Decantation of this water and its replacement with fresh distilled water allowed the process to be repeated with essentially the same results, in that the second portion nearly always had the same hardness as the first.

Work done by Bricker, et al., (16) led to an integral component in the experimental design used in this dissertation. In order to investigate mineral-water interaction, a small (103 acre) watershed in which all waters draining from the basin originate as precipitation on the watershed was chosen. Two plexiglass columns 6 inches inside diameter by 3.5 feet long were filled with soil from the watershed. Distilled water was allowed to continuously percolate through the first column. The effluent was sampled and then discarded. In the second column, distilled water was initially used as the percolate: the effluent was sampled and then continuously recirculated through the column, no more distilled water being added. In a matter of days the effluent from both columns achieved a chemical composition similar to that of streamwater in the watershed. The effluent from the columns maintained this composition for a period of 19 weeks. During this time, a volume of water equivalent to six years rainfall was percolated through the columns. At this point, the soil in the columns was dried. A solution containing 50 ppm SiO_2 was circulated through column number 2. The silica concentration stabilized in a few hours at the same value observed previously (approximately 9 ppm). Distilled water percolated through column number 1 also achieved a stable silica concentration of 9 ppm in a matter of hours.

Soil water samples were collected by inserting a plastic pipe into an auger hole. Water from the soil was drawn through a 0.45μ membrane filter into the pipe by vacuum. Chemical analysis of the water showed it to be remarkably similar in composition to the stream water and thus the column effluent composition.

Objectives

The primary goal of this research project is to establish the feasibility of a laboratory model as a research tool for water quality investigations of an impoundment system. In particular, the research program is concerned with three areas.

The first area is concerned with the question: Can a laboratory model of Lake Thunderbird (to be described later) be established and maintained using percolation columns, as described by Bricker, *et al.*, (16), as the sole source of waters and constituents? This does not imply an attempt to dynamically duplicate Lake Thunderbird. Instead, it is an attempt to: (1) define the major source(s) of waters and constituents that give Lake Thunderbird its particular water quality and, (2) test a method of maintaining a laboratory model over relatively long periods of time.

The second area of study deals with responses of the model to changes in environmental conditions. An attempt is made to show what controlling influence such environmental factors have on various aspects of the model system. Such factors as aeration, DO, light, and evaporation are manipulated and the effects on various water quality parameters, as well as on the whole model, are monitored.

The third area of study pertains to a number of possible fundamental research applications of the model. These studies involve several different areas, overlap somewhat with the above-mentioned areas of concern, and are fairly general. One such application utilizes ^{45}Ca to determine if suspended clay particles under simulated natural conditions sorb dissolved minerals such as calcium. Another area of study compares topsoil from the Lake Thunderbird watershed, Lake Thunderbird sediment, and model sediment in order to gain information into the nature of sediments and their role in water quality. One other such study tries to ascertain whether or not available phosphates are released from model sediment under hypolimnetic conditions (no light and reduced oxygen concentrations).

It is hoped that the work presented here will contribute to the knowledge currently being established in the field of simulated natural systems.

CHAPTER II

THE LAKE THUNDERBIRD SYSTEM

The Lake

Lake Thunderbird is a relatively typical southwestern impoundment. It was built by the U. S. Bureau of Reclamation for flood control, recreation, and municipal and industrial supply for Norman, Del City, and Midwest City. The earth-fill dam was completed in 1965 across Little River, just downstream from the mouth of Hog Creek. The Central Oklahoma Master Conservancy District is the controlling agency of the lake and is composed of members from Del City, Norman, and Midwest City.

Lake storage capacities and basic morphometry figures are listed in Table 2. At the normal (conservation) elevation of 1039 feet above sea level, the reservoir has an area of 6070 acres and a capacity of 119,600 acre-feet. Storage allocations are 76,200 acre-feet for flood control, 85,000 acre-feet for industrial and municipal supply, and 35,000 acre-feet for minimum pool capacity and sediment accumulation.

The maximum depth of the conservation pool is 69 feet (sediment buildup not accounted for) in the area just northwest of the dam. However, the mean depth of the conservation pool is only 20 feet. It can be seen from Table 2 that the ratio $3(d_m/d_{mx})$ is fairly constant at the

TABLE 2

LAKE THUNDERBIRD MORPHOMETRY

	Maximum Pool	Flood Pool	Conservation Pool	Dead Storage Pool
Basin Elevation (msl in feet)	970	970	970	970
Pool Elevation (msl in feet)	1,065	1,049	1,039	1,010
d_{mx} (feet)	95	79	69	40
A (acres)	13,850	8,800	6,070	1,680
S (miles)	196	125	86	24
V (acre-feet)	367,500	196,200	119,600	13,700
d_m (feet)	27	22	20	8
Sd	11.9	9.5	7.9	4.2
Vd	0.84	0.81	0.84	0.60

msl = mean sea level

d_{mx} = Maximum depth

A = Surface area

S = Length of shoreline

V = Volume

d_m = Mean depth

Sd = Shoreline development = $2 \frac{S}{(\pi A)^{1/2}}$

Vd = Volume development = $3(d_m/d_x)$

various lake pools. This ratio is known as "volume development" (Vd) and compares the lake volume to the volume of a cone with a basal area equal to that of the lake and a height equal to the mean depth. Thus a ratio of 1 would indicate a cone-shaped basin. Therefore, the volume development for Thunderbird indicates: (1) a basin shape approaching that of a flat cone and (2) the basin shape is fairly constant for each of the lake pools.

"Shoreline development" (Sd) is the ratio of the length of shoreline to the length of circumference of a circle with an area equal to that of the lake. A ratio of 1 would indicate a circular-shaped lake. Figures from Table 2 indicate that Thunderbird is not circular-shaped at any of its storage pools and would be best classified as dendritic. Such figures do indicate the large amount of littoral area found in Thunderbird.

As can be seen in Figure 2, Lake Thunderbird is a somewhat u-shaped lake. Prevailing southerly winds keep the reservoir well mixed. Any summer stratification that has developed has been extremely transitory. Vertical temperature stratification in 40 feet of water has not exceeded 4-5 $^{\circ}$C. Chemical analyses reveal very little difference from one sample location to the next, with the possible occasional exception of the upper reaches of the Little River and Hog Creek arms. Suspended clay particles give the lake a muddy appearance although turbidity values as low as 5 Jackson turbidity units have been recorded. Water temperatures vary from a winter low near freezing to a summer high of 30 $^{\circ}$C. A partial ice cover was observed in the winter of 1968 and lasted about 2 days.

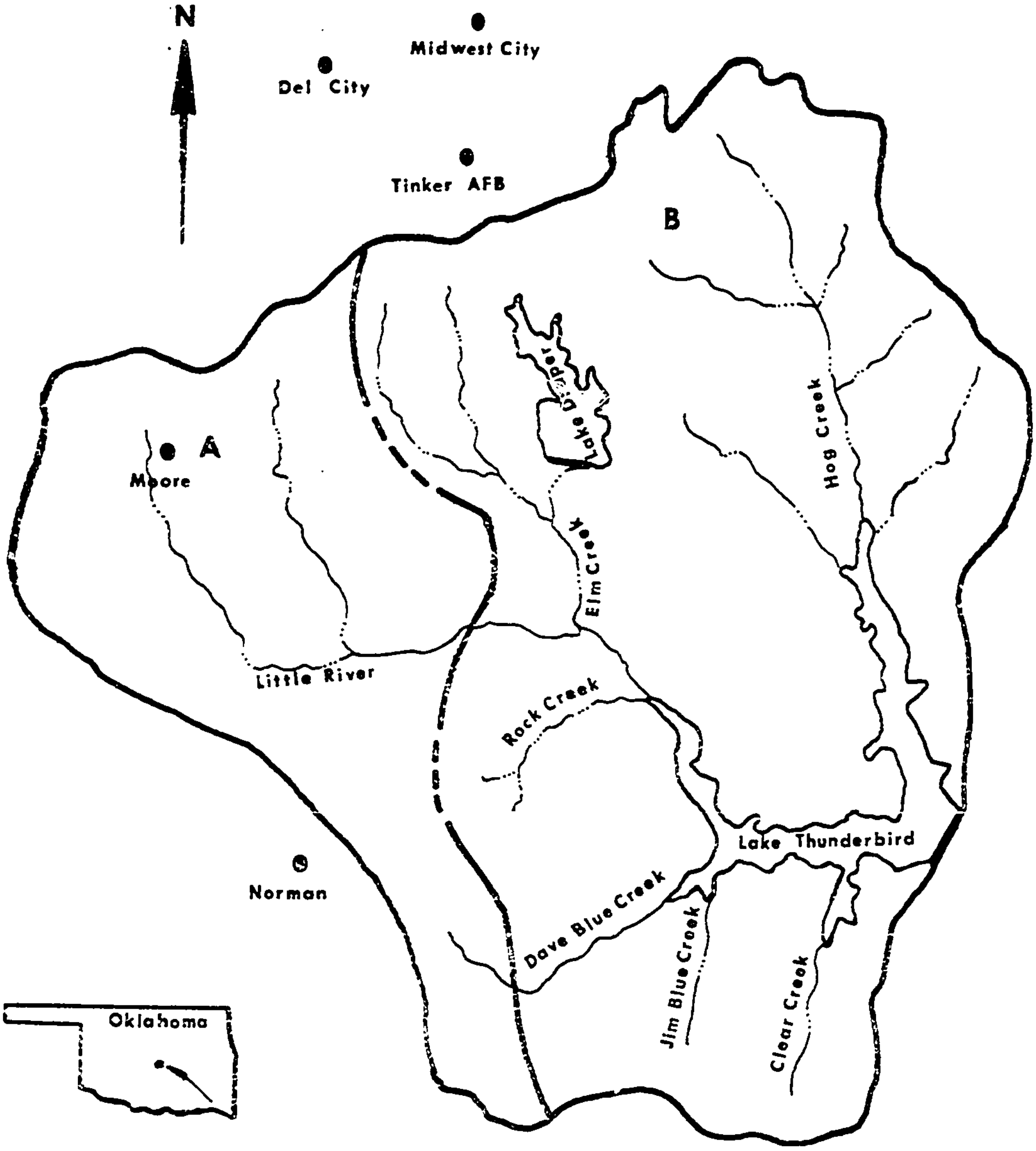

Figure 2. Lake Thunderbird Watershed

The water quality of Thunderbird is generally good. Table 3 lists approximate value ranges for some typical water quality parameters.

TABLE 3

LAKE THUNDERBIRD WATER QUALITY

D. O.	Near Saturation
pH	8.0-8.5
Turbidity	5-500 JTU
TDS	250-300 mg/l
Iron	0-0.10 mg/l
Manganese	0-0.25 mg/l
o-Phosphate	0.01-0.15 mg/l
T-Phosphate	0.20-0.30 mg/l
Alkalinity	180-200 mg/l
Total Hardness	170-230 mg/l
Calcium Hardness	90-120 mg/l
Magnesium Hardness	80-110 mg/l
5-Day BOD	2-3 mg/l
COD	15 mg/l

Although the dissolved oxygen concentration is usually near saturation, low lake bottom values have been recorded in the summer. Such low values (1-2 mg/l) usually persist for only a day or two. The low BOD and COD values indicate a low organic concentration and the absence of pollution in the area.

Climate

Lake Thunderbird is located in central Oklahoma, about 12 miles east of Norman, in Cleveland County. The climate of this region is controlled by the interaction of tropical and polar air masses, and most precipitation is brought about by the chilling of warm, moist Gulf air by cooler air of northerly origin. Most of the annual precipitation is due to rainfall, with very little attributed to hail, sleet, and/or snow.

Rain from regional cyclonic storms and local thunderstorms occurs throughout the year, but is greatest during the spring and summer months. The average annual precipitation at Norman is 33 inches. The average annual temperature is 60°F. However, as can be seen in Table 4, the central Oklahoma area is characterized by wide ranges in temperature and wide deviations from average precipitation. Although Table 4 does show the seasonal variation in central Oklahoma, such average figures do not reveal the tremendous range found in temperature and precipitation. An annual temperature range of 110°F is not unusual in this area. Likewise, rainfall intensities have been known to vary from a few hundredths of an inch per 24 hours to 10-12 inches per 24 hours.

Figure 3, redrawn and updated from Wood and Burton (17), illustrates the annual precipitation average for the period 1891-1969 and the year-to-year deviation from this average. Figure 4, redrawn and updated from Wood and Burton (17), shows the precipitation cumulative departure from average annual precipitation for the same period of time.

TABLE 4

AVERAGE TEMPERATURE AND PRECIPITATION IN CENTRAL OKLAHOMA

Month	Average Temperature (°F)	Average Precipitation (In.)
January	38.8	1.43
February	43.0	1.58
March	49.3	2.08
April	61.1	3.44
May	69.0	5.44
June	72.8	4.46
July	81.0	3.07
August	82.1	2.69
September	69.5	3.35
October	63.9	2.93
November	49.7	1.81
December	41.3	1.53
Annual	60.1	33.81

Source: Wood, P. R. and Burton, L. C., "Ground-water Resources in Cleveland and Oklahoma Counties, Oklahoma," Oklahoma Geological Survey, Circular 71, 1968

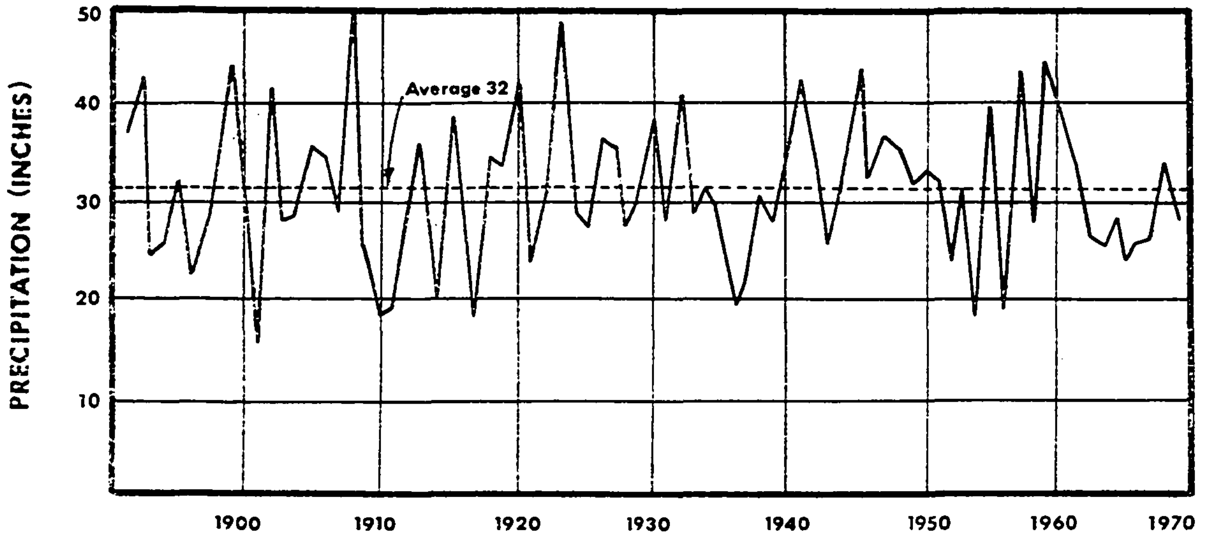

Figure 3. Annual precipitation at Oklahoma City (1891-1969)

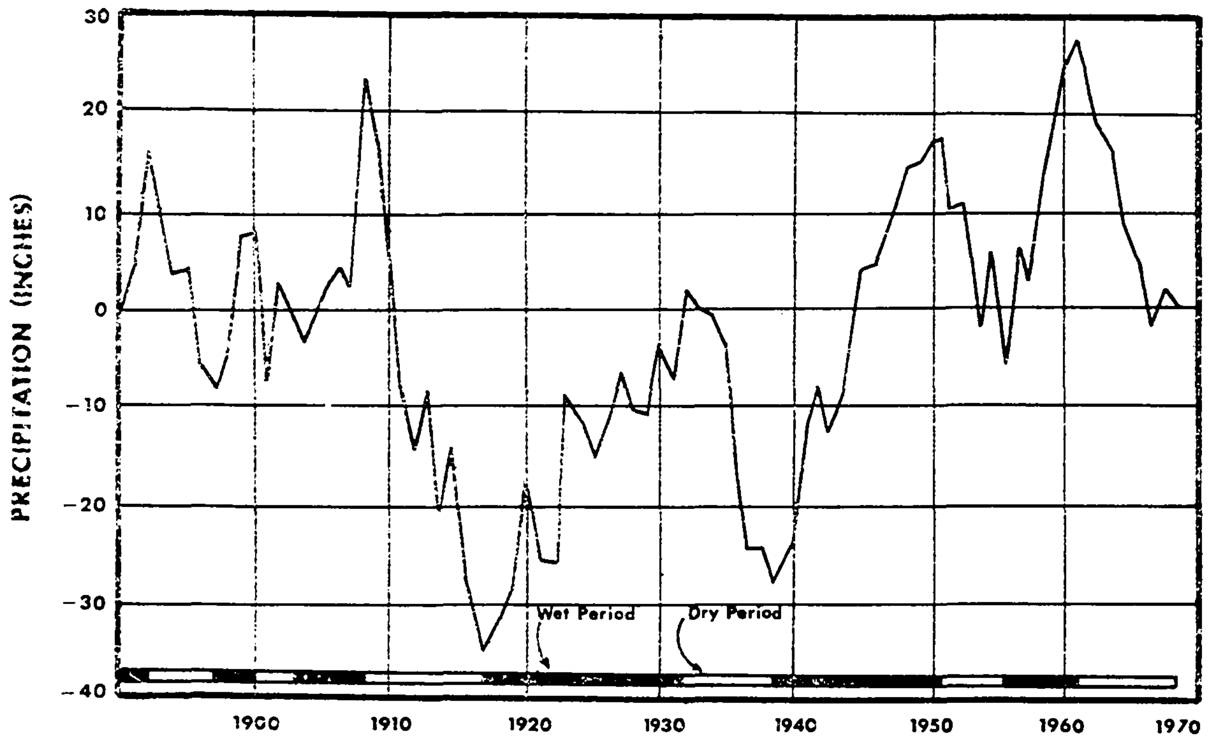

Figure 4. Cumulative departure from average precipitation at Oklahoma City (1891-1969)

The trend shown in Figure 4 indicates a cycling phenomenon. Upward curves indicate trends of greater than average precipitation and downward curves indicate trends of below-average precipitation. Such alternating wet and dry periods suggest dry periods, ranging in length from 5-9 years, and wet periods of 2-15 years. Both Figure 3 and Figure 4 are representative of precipitation trends at other locales in the great plains.

Figures 3 and 4 indicate that since completion of the Thunderbird dam in 1965, the region has been experiencing a dry period. The maximum reservoir elevation obtained (1,032 feet) occurred in the spring of 1969 and again in the spring of 1970. At the 1,032-foot elevation, the reservoir has a capacity of approximately 84,000 acre-feet. Thus the reservoir has yet to reach the conservation pool elevation of 1,039 feet. However, if past trends hold, a wet period can be forecast over the next few years. (Note: At the time of this writing, October 1970, unusually high rains have brought the reservoir level to an elevation of 1,036.7 feet.)

Hydrology

Lake Thunderbird is located in a well defined, small watershed of 256 square miles. All waters draining the watershed originate therein as precipitation. Little River is the principal stream in the basin. The gradient of Little River is about 12 feet per mile southwestward. According to Wood and Burton (17), Little River had an average annual rate of flow past a gaging station located just south of the present damsite of 45,700 acre-feet (63.2 cfs) for the 11-year period 1952-1963. Since closure of the dam in March 1965, an average daily flow of 0.5 cfs (362 acre-feet per year) has been maintained by leakage around the gates; the gates have not been opened to date. Due to clay-sediment accumulation, little loss due to seepage is thought to occur.

All the principal streams in the watershed are classified as intermittent. All streamflows are dependent on rainfall; groundwater is not thought to be a source of surface water anywhere in the region. Since closure, water has flooded over streambanks in many places and streamflows are very small. Volume calculations for the Thunderbird system for fiscal year 1968-1969 are shown in Table 5.

TABLE 5

VOLUME CALCULATIONS FOR THUNDERBIRD SYSTEM
FISCAL YEAR 1968-1969

Watershed Rainfall Volume (37 inches)	504,640 acre-feet
Measured Lake Volume Increase	31,640 acre-feet
Pumpage	7,079 acre-feet
Evaporation	11,971 acre-feet
Leakage Loss	362 acre-feet
Total Lake Volume Increase	51,052 acre-feet

Statistics provided by U. S. Geological Survey, U. S. Army Corps of Engineers, and Central Oklahoma Master Conservancy District.

Although the volume calculations in Table 5 are based on only one year, they are representative of the hydrology of the system. Evaporation in the area is high; the average annual evaporation-pan loss of 58 inches will cause a reservoir volume loss of about 29,000 acre-feet when the conservation pool elevation is reached.

It is further evident from Table 5 that of the 504,627 acre-feet potentially available to Thunderbird, only 11 percent (51,052 acre-feet) actually reached the impoundment. It has been estimated, Burton and Woods (17), that only 5 percent of the annual rainfall contributes to groundwater recharge. Therefore, approximately 84 percent of the annual rainfall appears to be lost to evapotranspiration.

The Watershed

Geologic, soil, and vegetation studies of the Thunderbird watershed revealed that it could be divided into two general regions. These regions are shown as "A" and "B" in Figure 2.

Region "A" is composed primarily of Hennessey Shale consisting of deep red clay with thin bands of sandstone, and Terrace Deposits of gravel, sand, and silt. The Hennessey Shale covers the western one-third of Cleveland County and is Permian in age. The formation is overlain with prairies that form a gently rolling, grass-covered plain. The prairies are largely barren of trees except in the valleys of the streams. Reddish-brown shale containing layers of siltstone and fine-grained sandstone dominate the Hennessey.

The soils of region "A" are moderately deep, and have been developed chiefly from calcareous clay shale and sandy shale. The soils are generally red to reddish-brown and vary in texture from clayey to loamy. Surface drainage in the region is slow to rapid, depending on local relief. Runoff rates are high and most sloping surfaces are susceptible to gully and sheet erosion.

Region "B" is composed almost entirely of the Permian Garber Sandstone and Wellington Formations. Much of the region is characterized by low, steep-sided sandstone hills formed by differential erosion of lenticular beds of red sandstone and shale. These hills are forested with small blackjack, scruboak, and other small slow-growing deciduous trees. Garber Sandstone and Wellington Formation are not easily distinguished because of the similarities in lithology and water-bearing characteristics: Wood and Burton (17) mapped them as a single unit.

The soils of region "B" are reddish-brown to brown and are shallow to moderately deep. They are generally loamy and sandy in nature. Surface drainage is rapid, runoff rates are relatively high, and the area is highly susceptible to both sheet and gully erosion.

With the exception of Moore, no urban areas exist in the watershed. Moore delivers its sanitary and storm sewage to the South Canadian River, located west of the watershed. The other urban areas in the vicinity--Norman, Del City, Tinker Air Force Base, and Midwest City--discharge their sanitary and storm wastes outside the watershed.

All of the land immediately surrounding Thunderbird has been set aside as a public area. The area includes boat ramps, picnic grounds, and camping grounds. In addition, the land surrounding the northern one-third of the Hog Creek arm of the lake has been designated as a wildlife area. The rest of the land in the watershed is either forested or grass-covered prairie. There is some general farming. However, according to the Cleveland County office of the Soil Conservation Service, there is no irrigation in the watershed and little or no use of fertilizers or pesticides; the trend over the past few years has been toward more pasture (prairie) land.

Lake Stanley Draper is the only other reservoir in the watershed. The lake is part of the municipal water-supply system of Oklahoma City and was built as a storage reservoir for water pumped from Lake Atoka, located in the southeastern part of the state. Other than the interception of an estimated 1,000 acre-feet per year from the Thunderbird watershed runoff, Draper has had no effect on Thunderbird. During the course of this study, the Draper release gates were not opened. However, due to a State Supreme Court ruling, water is now being released in an amount equivalent to the amount annually intercepted from the Thunderbird watershed.

Remarks

The preceding study of the Thunderbird system reveals that it is somewhat unique in its relative simplicity. As mentioned earlier, there is no evidence of any waste or groundwater inflow to the system. Furthermore, as Lake Draper released no water to Thunderbird during the course of study, it appears that watershed-rainfall-runoff is the major source of waters and constituents to Thunderbird.

The study of the watershed reveals that there are only two major geologic formations in the area and that these are quite similar. Soil in the watershed is derived from the weathering of the geologic formations and also is all quite similar, being composed primarily of clays, shales, sand, and silt. The high turbidity values found in Thunderbird, especially after storms, are probably due to the high runoff rates in the area and the susceptibility of the soil to sheet and gully erosion.

It is concluded that watershed topsoil plays a major role in determining the water quality of Thunderbird. It is on this conclusion that the following experimental design is based; namely the use of percolate from a column filled with watershed topsoil to establish a laboratory model of Thunderbird.

CHAPTER III

EXPERIMENTAL DESIGN

One of the more difficult design problems encountered in laboratory models is the development of a method for maintaining the model. Here, "maintaining the model" means keeping the model dynamic - maintaining a "living" system. Just as lake, stream, or impoundment systems should not be thought of as being static, neither should the laboratory models that attempt to simulate them. According to Holm-Hansen (11), in his discussion of nutrient enrichment experiments, when a water sample is placed in a bottle or container and the assumption is made that what is found in the bottle is true in the lake, the assumption is usually not valid. As mentioned in Chapter I, the nutrient supply to a natural system is continually being replenished by a number of routes. However, not only nutrients, but all constituents in a natural system such as an impoundment are continually being replenished, cycled, or changed in form. In the case of impoundments, the controlling effects of such factors as basin-type, climate, and location (to name just a few) must be considered. Such controlling factors not only can determine what particular reactions occur, they largely influence the rate at which they occur. To add to the complexity, consideration must be given to the nature of the source(s) available to the impoundment--sources responsible for a particular impoundment's biological and chemical identity--its water quality.

Figure 5 represents an attempt to simplify an impoundment system. Unlike many such diagrams, no attempt is made here to depict specifics. Instead, for purposes of simplification, broad, inclusive terminology has been used; hopefully such terminology will include any specific area of concern. Such simplification allows the entire impoundment system to be thought of in terms of subsystems. Thus as shown in Figure 5, the impoundment system is comprised of 4 subsystems: sources losses, impoundment, and controls.

The sources subsystem represents all waters and constituents available to the impoundment. Rainfall represents the major form of precipitation in many areas, especially in the southwest. Rainfall directly on the impoundment provides both water and any materials dissolved or physically trapped in the raindrops. Rainfall on the watershed can provide a tremendous variety of materials to the impoundment, both natural and man-made. Natural constituents in watershed-rainfall-runoff include suspended materials such as clay, sand, silt, and living organisms, and dissolved materials that can include virtually every element known. Man-made constituents can include fertilizers, pesticides, and a large variety of synthetic compounds. Groundwater can often be a source of iron, manganese, and other materials in the reduced state, due to the low redox potential of such waters. Wastewaters from municipalities, industries, and power generating stations have had disastrous effects on some impoundments.

The losses subsystem represents all waters and constituents that

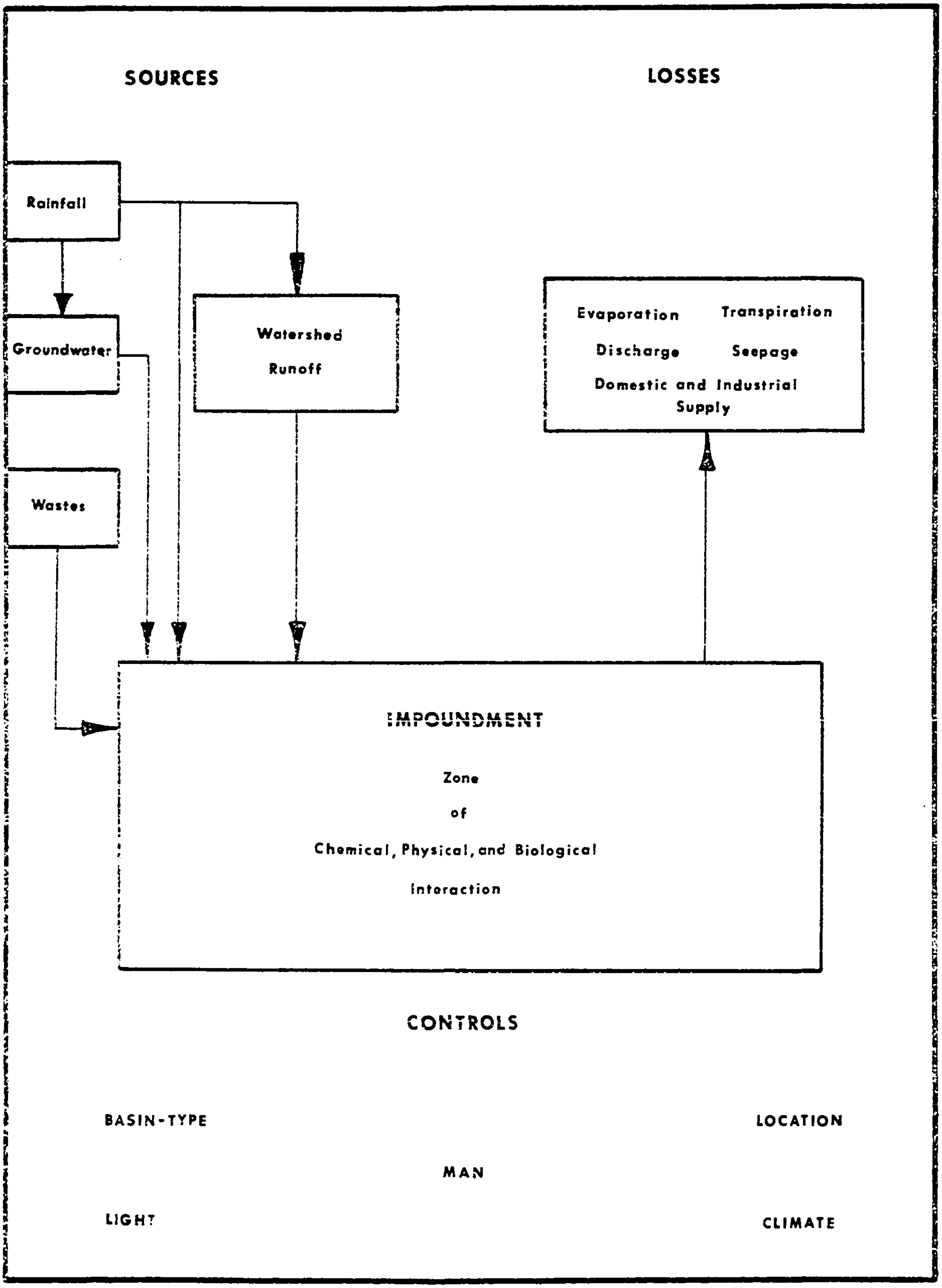

Figure 5. Impoundment System Schematic Diagram

can be lost from the impoundment. Losses from an impoundment can be divided into concentrating losses such as evaporation and transpiration, and non-concentrating losses such as discharge, seepage, and domestic and industrial supply.

In this diagram, the impoundment is viewed as a reactor on the sources available to it. As materials enter the impoundment they are involved in various chemical, physical, and biological interactions. These interactions involve all phases of the impoundment including the dissolved phase, the suspended phase, and the sediment phase. Although separate chemical, physical, and biological parameters are always changing, these parameters interact to produce an overall steady-state effect that changes slowly with time.

The impoundment system diagram also lists certain control factors. These factors represent once again an attempt at simplification--a broad categorization of all the external environmental factors that serve as controls over the whole impoundment system. Thus such factors not only directly control the nature and rate of reactions within the impoundment reactor but also indirectly, through their effect on the sources and losses subsystems. It should be pointed out that a control factor can be involved in various combinations with other control factors to produce an effect on the system. For example, climate may cause changes in the water elevation through its control over sources (rainfall) and losses (evaporation). Yet man can regulate surface elevation by regulating the rate of discharge. Furthermore, basin-type and shape control the amount of surface area change corresponding to change in surface elevation. Windborne materials from outside the watershed indicate the influence of climate on the nature of sources available to the impoundment.

In the case of the Lake Thunderbird system, it is possible to simplify Figure 5 even further. As mentioned in Chapter II, groundwater and wastewater are not known to be significant sources of input to Thunderbird. It was therefore concluded that rainfall-watershed-runoff is the major source of waters and constituents to the reservoir. Also, as was shown in Table 5, evaporation and domestic and industrial supply are the major losses from Thunderbird.

Utilizing the belief that rainwater can leach and wash a large variety of materials and living organisms from the soil, percolation column effluent was used to simulate Thunderbird rainfall-watershed-runoff. In contrast to the techniques of other investigators, no natural (lake) water was used in this experimental design. Instead, the effluent from the percolation column was retained, allowed to change with time, and certain water quality parameters compared with Thunderbird. Thus the concept of percolation columns as described by Bricker, et al. (16) was used as an integral component in this experimental design. In other words, the effluent from such columns was used to "feed" a model impoundment--to continuously supply what was believed to be a natural source of waters and constituents. Because soil is an important source of vitamins, nutrients, and other materials, and because the column effluent continuously replenishes these materials, it was hoped that the laboratory model of Thunderbird would be a dynamic system. It was felt that incorporation of the percolation column phenomenon into the experimental design would allow a laboratory model of

Thunderbird to be maintained over relatively long periods of time.

The experimental design used in this research was divided into two parts. The concept involved was essentially the same in each part.

The first part was a preliminary investigation of small percolation columns. There were two objectives involved in this investigation. The first objective was to determine if percolation columns filled with soil from the Lake Thunderbird watershed behaved like those described by Bricker. In other words, determine if the effluent from such columns was similar in quality to Thunderbird water, especially column effluent that had been retained over a period of time. The second objective attempted to determine whether or not a representative soil sample could be obtained from the Thunderbird watershed. That is, determine if a column packed with soil from one location in the watershed had an effluent similar in quality to the effluent of a column filled with soil from another watershed location.

Seven soil samples were collected from the Thunderbird watershed in July 1968 from locations shown in Figure 6. Samples 3 and 5 were collected from timbered areas, were dark brown in color, and were approximately 5 percent by weight organic material, as determined by loss of weight upon ignition at 600°C. The rest of the samples were red in color, about 1 percent by weight organic material, and were collected from grasslands, with the exception of sample number 2, which came from an eroded area. Approximately 53 grams of each sample were placed in small (4 inches long by 7/8 inch diameter) glass percolation columns. The columns were then leached with aerated distilled water (simulates rainfall) and the effluent (simulates rainfall-watershed-runoff)

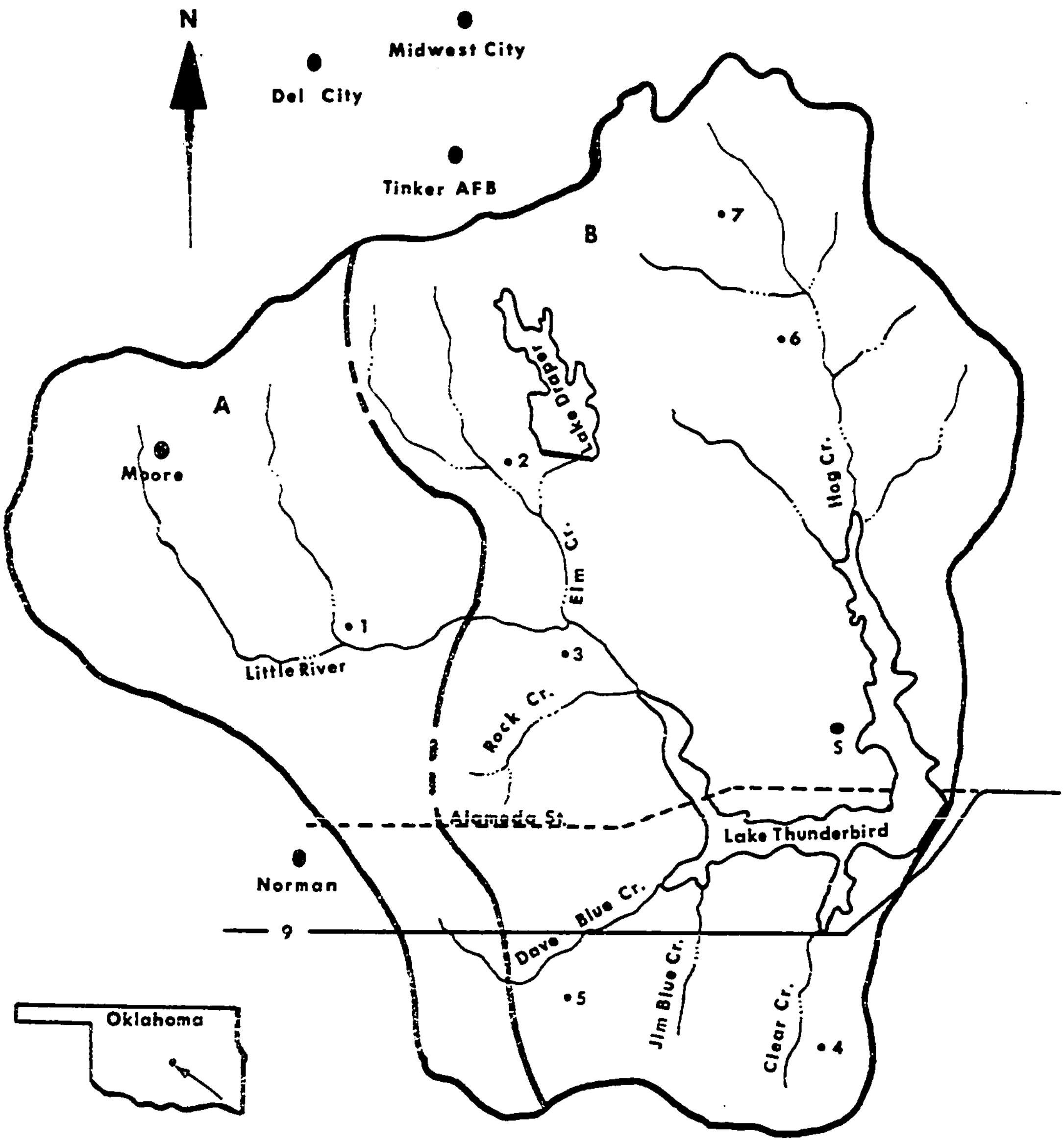

Figure 6. Soil Sample Locations.

collected in lighted, aerated, 2000 ml beakers (simulate impoundment). Column number 3 and its beaker were kept dark in order to rule out the effects of algal growth. With this one exception, all 7 columns were subjected to the same operational and environmental conditions.

The objective of the second part of the experimental design was to establish a column-fed, long term model of Lake Thunderbird. The experimental design used is shown in Figure 7. A 20-gallon aquarium contained continually aerated distilled water used to simulate rainwater. Continuous aeration maintained the pH of the distilled water at about 6. The tank was sealed with a plexiglas cover and then completely covered with aluminum foil to prevent algal growth. A port on top of the tank allowed additional distilled water to be added as needed. A peristaltic pump was connected to a 6-hour timer; the length of time that the pump remained on each 6 hours could be regulated. When on, the pump delivered water to the column at about 12 ml per minute. The compressed air used to aerate the water was filtered through a 5-inch cotton plug. A cotton-filled vent located on top of the tank allowed the compressed air to escape. Although the tank was not assumed to be completely free of contaminants, it was felt that such a design would largely reduce the amount of contamination from airborne dust and bacteria.

The percolation column used was made of plexiglas and was 4-1/2 feet long by 5-3/4 inches inside diameter. Soil for the column was collected in June 1969 from two sites in the vicinity of "S" shown in Figure 6. One soil was collected from a grass-covered prairie; this soil was red, contained a large percentage of sand, and was about

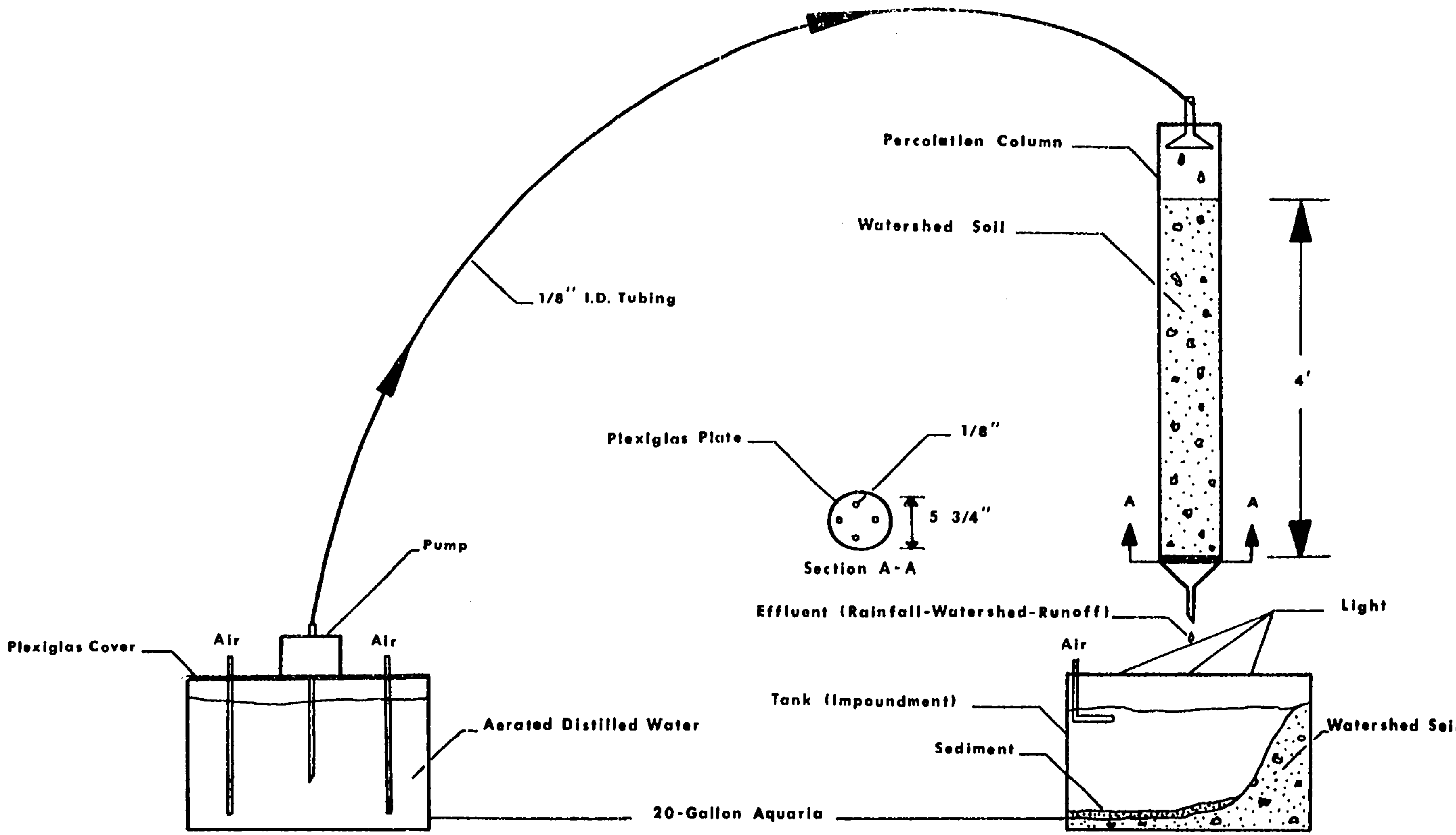

Figure 7. Experimental Design

1 percent by weight organic material. The other soil was collected from a timbered site; this soil was dark brown in color, appeared to be loamy, and was about 7 percent by weight organic material. An approximate 50-50 composite of these two soils was placed in the column to a depth of about 4 feet. The column was covered with aluminum foil to prevent excessive algal growth between the side of the column and the soil. An inverted funnel dispersed the drops of distilled water over the surface of the soil and grass at the top of the column. A plexiglas plate at the bottom of the column served to hold the soil in place. One-eighth inch diameter holes in the plate allowed the passage of effluent water and were also large enough to permit some soil washout, thus providing a source of turbidity and sediment to the receiving tank. Again, effluent from the column was used to simulate rainfall-watershed-runoff.

A 20-gallon receiving tank was used to simulate the impoundment. The tank was 24 inches long by 12-1/2 inches wide by 16 inches deep. Before receiving any column effluent, the bottom of the tank was covered with the same composite soil used in the column. The soil was used to shape the bottom of the tank as shown in Figure 7. Three sediment traps were placed on the bottom of the tank for future sediment analysis. Two 23-inch Westinghouse "Plant'Gro" fluorescent bulbs connected to a 12-hour timer were used to supply light to the tank. The bulbs were oriented along the long axis of the tank and provided an intensity of about 35 footcandles at the water surface when the tank was filled to a depth of 9 inches. A black plastic curtain was placed around the tank so that the "Plant-Gro" bulbs were the only light source to the tank. As water built up in the tank, it was aerated just under the surface along the long axis with filtered, compressed air.

Room temperature was held at approximately 24^{o}C during the course of study by central heating and cooling.

With such an experimental design, it was possible to periodically monitor the concentration of various chemical parameters in the column effluent (sources) and compare these to concentrations of these sources in the receiving tank. Thus it was possible to gain some knowledge of how these sources changed with time after "impoundment."

CHAPTER IV

EXPERIMENTAL METHODS AND RESULTS

The results of this research have been divided into several sections for the purposes of discussion. The objective(s), experimental methods, and results will therefore be presented on a section-by-section basis. The overall results will then be summarized and discussed in Chapter 5.

Small Percolation Columns

As mentioned earlier, the first part of the experimental design involved the study of small percolation columns filled with soil from 7 different Thunderbird watershed locations. The primary objective of this section was to gain some insight into the behavior of percolation columns in this region. It was felt that such a short-term preliminary investigation was needed before proceeding with the experimental design shown in Figure 7. In particular, this section of study was designed to:

1. Conduct a parameter-by-parameter comparison of column effluents from each of the 7 soil sample locations.
2. Determine if column effluent retained over a period of time in a lighted, aerated 2000-ml beaker that is continually being fed by additional column effluent approaches Lake Thunderbird in water quality.

Before further discussion, it should be pointed out that there are several problems associated with such small (4 inches long by 7/8 inch diameter) columns. Such a small sample size (53 grams of soil per column) cannot be interpreted as being representative. Secondly, slow percolation rates in small columns severely limit the numbers and types of analyses that can be performed. Furthermore, because clay particles tend to swell when in contact with water, thus obstructing flow through the column soil, it was almost impossible to maintain the same flow through all columns. Due to the swelling clay particles and the settling of soil in the columns, percolation could usually be maintained for only about 30 days. Despite these drawbacks, it was felt that useful information could be obtained from such a study.

The following parameters were chosen to periodically monitor both the column effluents and the 2000-ml beakers.

1. Total Dissolved Solids (TDS) - Myron TDS meter.
2. pH - Photovolt expanded scale pH meter.
3. Turbidity - Bausch and Lomb Spectronic 20.
4. Total Hardness - EDTA titration as in "Standard Methods." (18)
5. Dissolved ortho-phosphate (DOP) - Colorimetric, stannous chloride-ammonium molybdate as in "Standard Methods."
6. Alkalinity - Volumetric, as in "Standard Methods."
7. Calcium Hardness - EDTA titration as in "Standard Methods."

The above parameters were chosen for several reasons. First, these parameters are usually included in any attempt to identify chemical water quality. Secondly, certain of these parameters--especially pH, DOP, and alkalinity--are known to be indices of biological trends.

Finally, Lake Thunderbird data for all these parameters is available from the last three years. Klehr (19).

A drop-by-drop flow of aerated distilled water was started to all seven columns. Samples were then periodically collected immediately from the columns and from the 2000-ml receiving beakers. Analyses 1-5 were conducted on a routine basis; analyses 1-3 were conducted most frequently because they are non-destructive tests. Fifty-ml samples were used for total hardness determinations and 25-ml samples for DOP. Suspended material was removed from samples analyzed for DOP by centrifugation for 20 minutes at 15,000 rpm. Alkalinity and calcium hardness were determined on 50-ml samples, and in this phase of the work, analyses were conducted on a very infrequent basis. Magnesium hardness was determined by subtracting calcium hardness from total hardness.

Results from the 30-day study are shown in Figures 8, 9, 10, 11, and 12. These graphs were drawn so that all seven soil columns and receiving beakers could be compared on a parameter-by-parameter basis. "Beaker" (Impoundment) represents data taken from the 2000-ml beakers. "Column" represents data from samples taken from the column effluents.

All column, and therefore beaker, waters were initially highly turbid and urine-colored. This initial color made the first few colorimetric (DOP) analyses difficult. Blanks were used to help partially correct for the color interference. Column color had usually disappeared after 4-5 days. Color in the beakers had usually disappeared by 5-6 days. High turbidity values made all analyses difficult, including pH.

Aquatic organisms were noticed in all beakers except number 3 by day 13. As mentioned earlier, beaker number 3 and its column were kept under continual blackout conditions. Blue-green algae had grown primarily on the sides and bottoms of the beakers. Green algae, ciliates, and rotifers were the most abundant organisms seen.

Column and beaker TDS results for all soils are shown in Figure 8. An initial "slug" effect (up to 750 mg/l for soil number 7) is shown for all column effluents with the exception of soil number 1, from which no zero-day TDS data is available. With the exception of soil number 5, column effluent equilibrium usually had been achieved by the time the second analyses were performed, usually on day 1. This slug effect will be discussed further in a later section. As can be seen, column effluent data remained essentially constant for all columns from day 2 through day 30. As mentioned earlier, all beakers were aerated just under the water surface; however, all beakers were not aerated at the same rate. Thus concentration in the beakers due to evaporation is different for all seven beakers as shown in Figure 8. Table 6 shows the mean ($\overline{X}$) and the number of observations (N) on which these means were determined. The number of observations used depended upon the period of time for which equilibrium conditions were assumed: usually day 2 to day 30. Due to the limited number of observations, little significance can be attached to any statistical interpretation of the parameters. Nonetheless, Table 6 shows that mean TDS values in the beakers are all larger than mean values for the corresponding column effluents, thus reflecting the effect of evaporation.

Column and beaker pH values are shown in Figure 9. Highly erratic

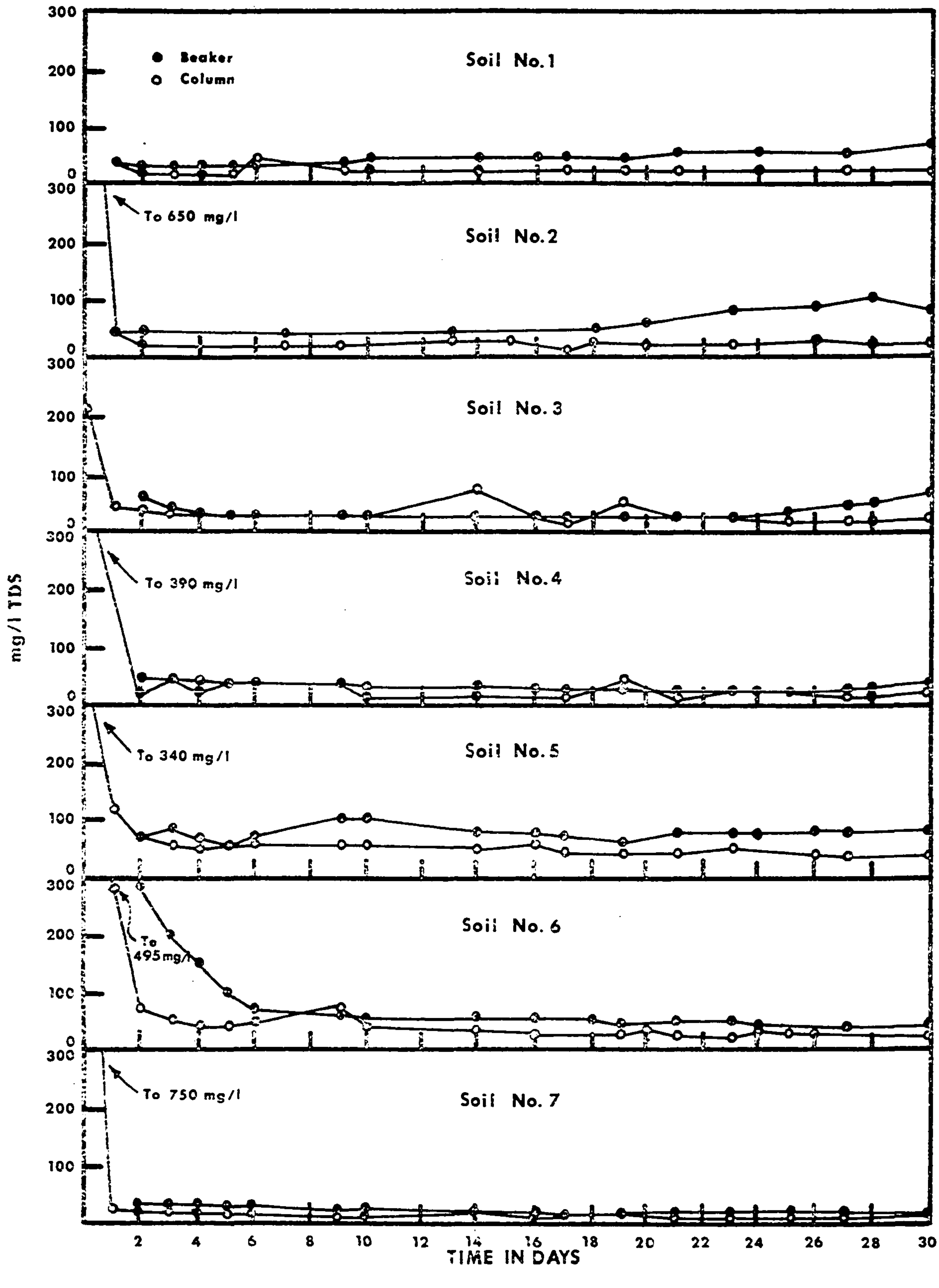

Figure 8. Beaker and Column Total Dissolved Solids Comparison for Seven Soils from Lake Thunderbird Watershed

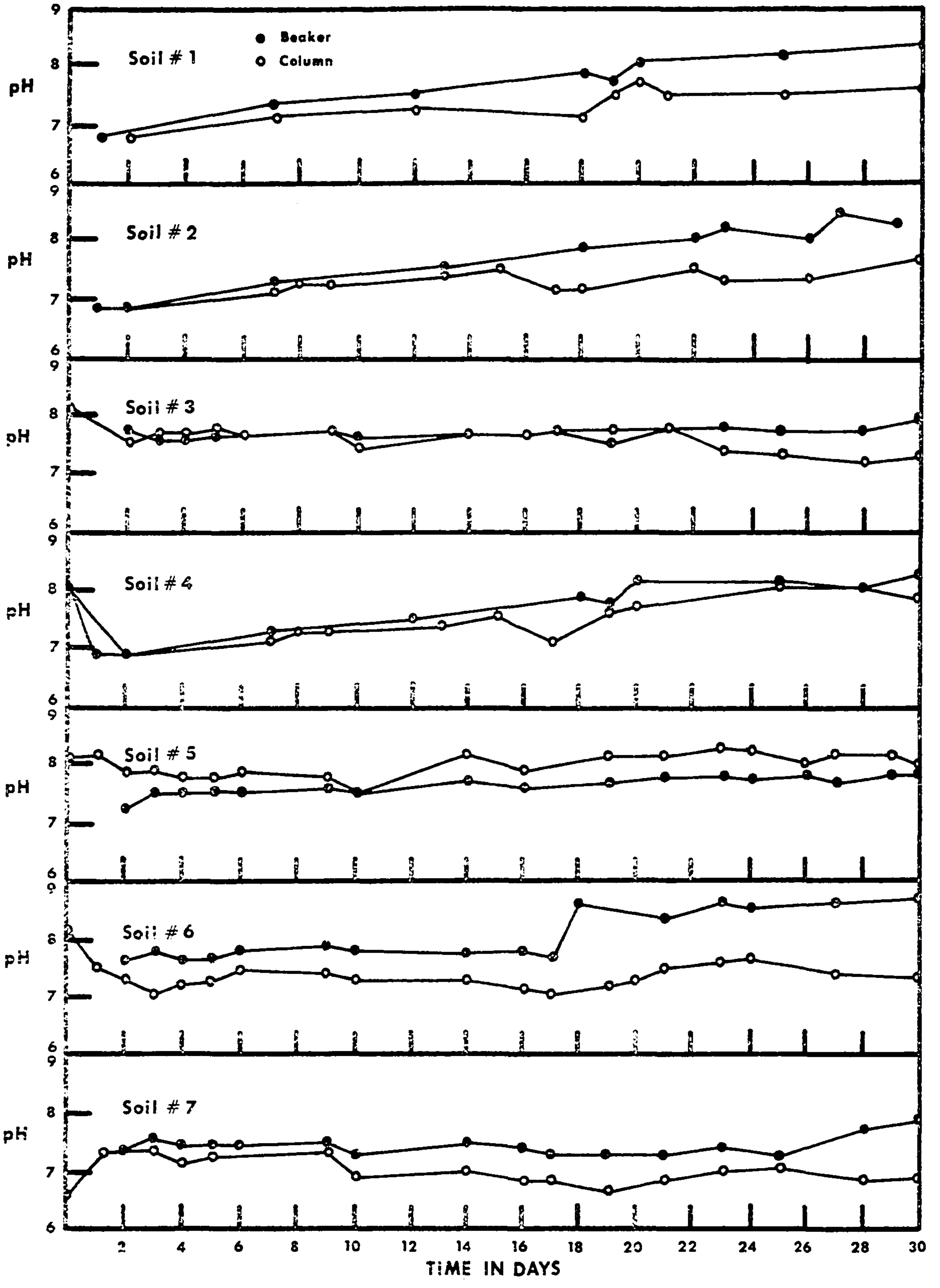

Figure 9. Beaker and Column pH Comparison for Seven Soils from Lake Thunderbird Watershed.

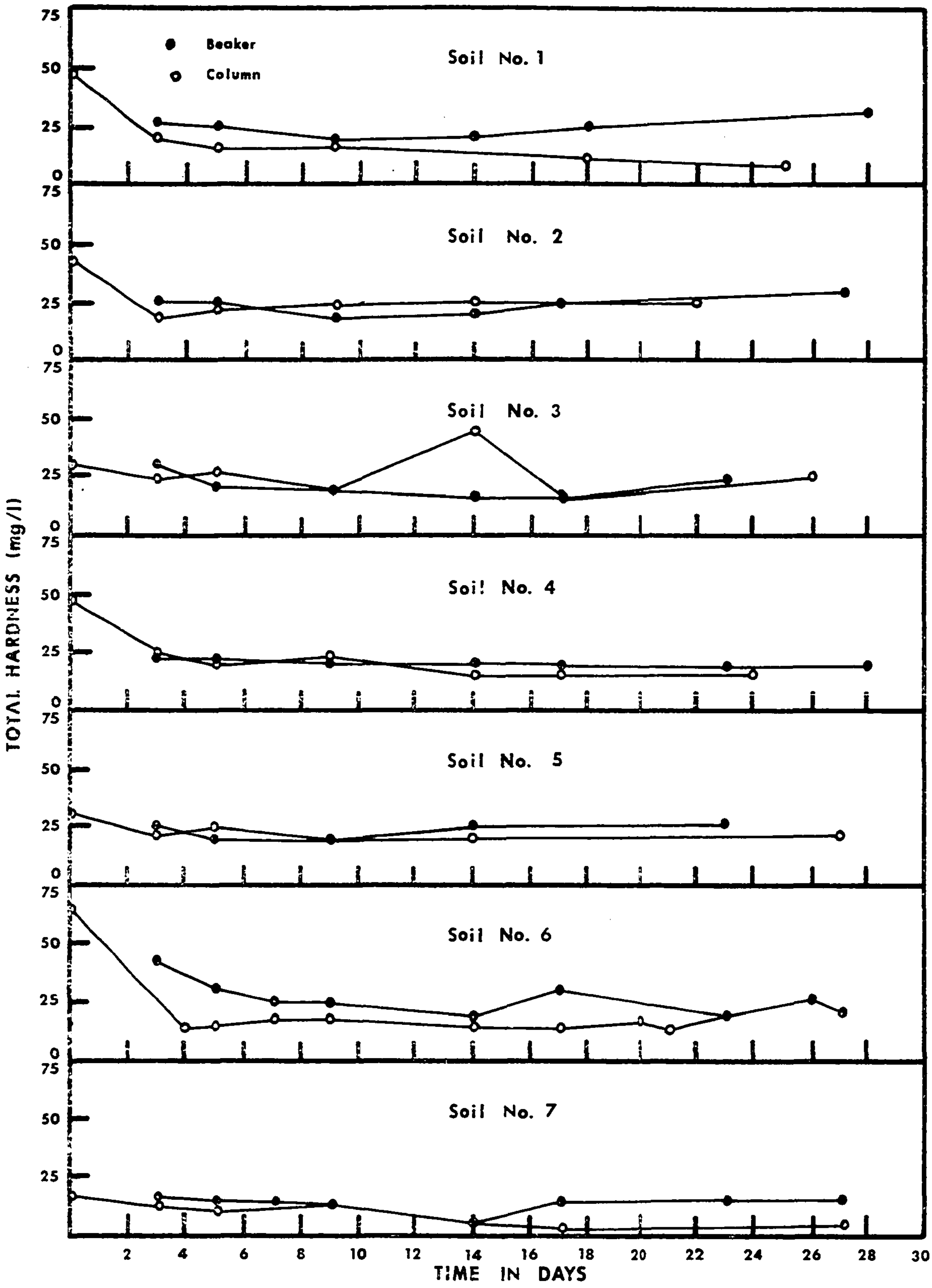

Figure 10. Beaker and Column Total Hardness Comparison for Seven Soils from Lake Thunderbird Watershed as mg/l $CaCO_3$.

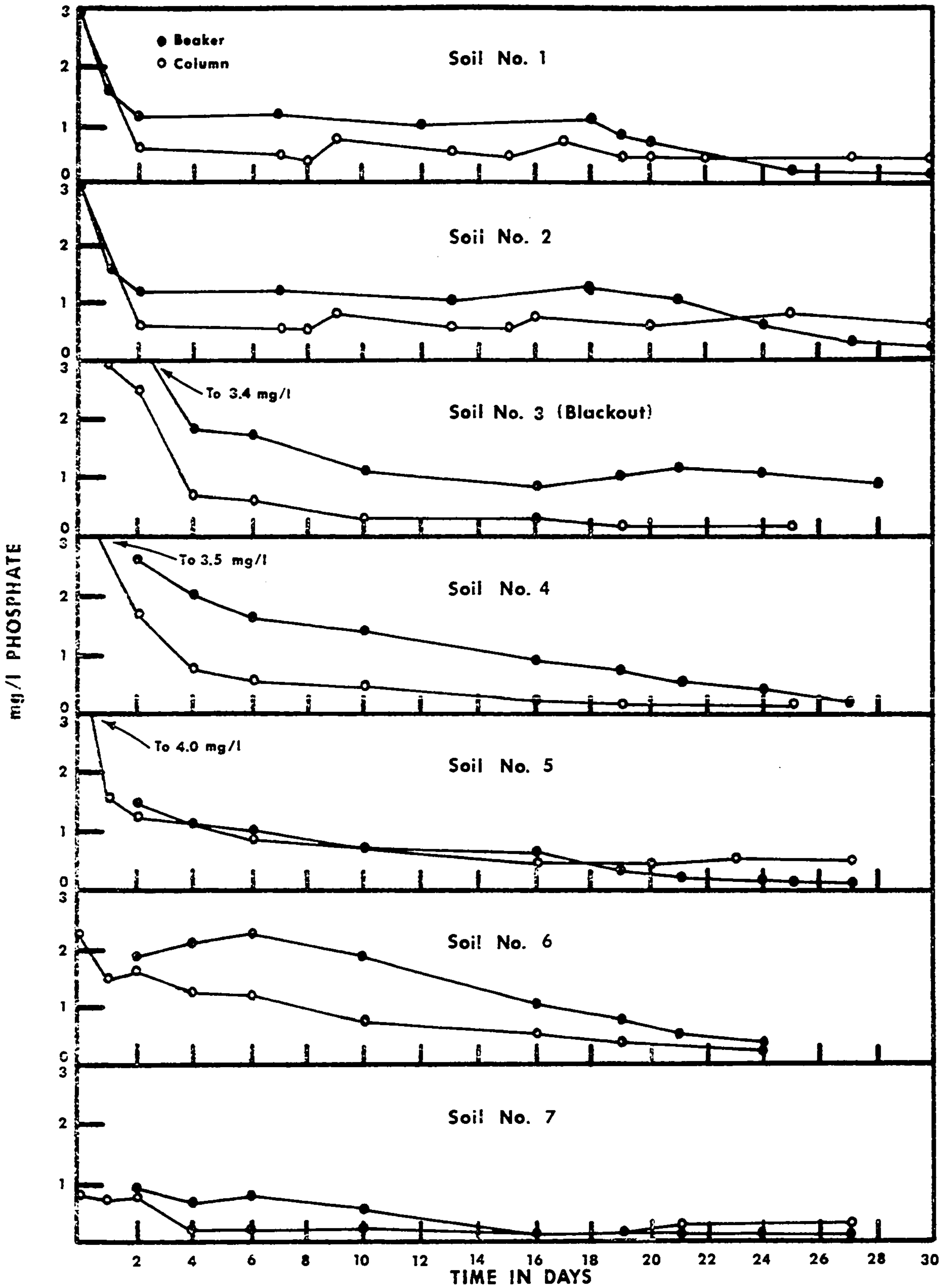

Figure 11. Beaker and Column Dissolved ortho-Phosphate Comparison for Seven Soils from Lake Thunderbird Watershed.

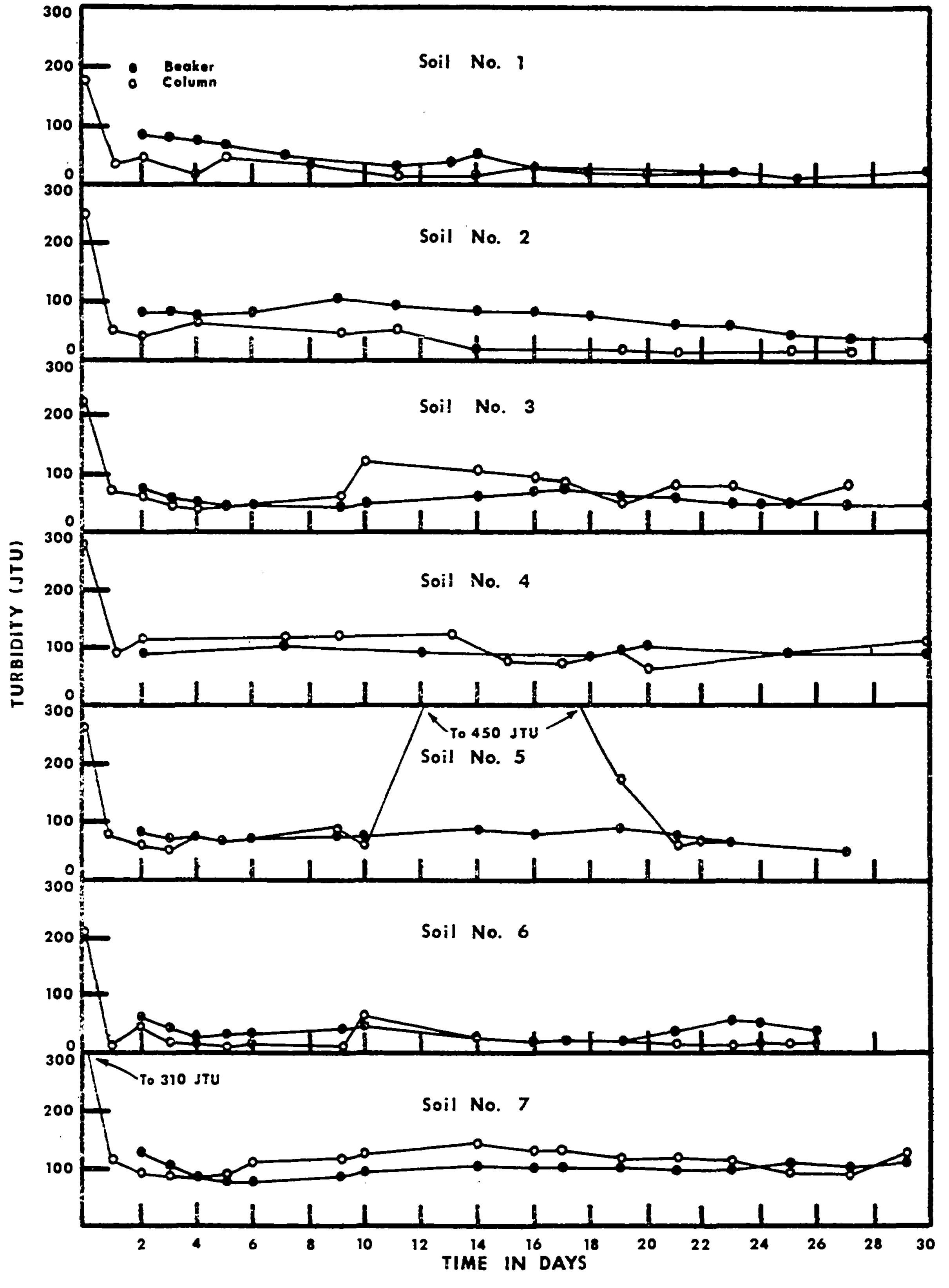

Figure 12. Beaker and Column Turbidity Comparison for Seven Soils from Lake Thunderbird Watershed in Jackson Turbidity Units.

TABLE 6

MEAN COLUMN AND BEAKER VALUES FOR SOILS FROM
SEVEN THUNDERBIRD WATERSHED LOCATIONS

Column									Beaker							
TDS		pH		Total Hardness		DOP			TDS		pH		Total Hardness		DOP	
N	$\bar{X}$	N	$\bar{X}$	N	$\bar{X}$	N	$\bar{X}$	Soil No.	N	$\bar{X}$	N	$\bar{X}$	N	$\bar{X}$	N	$\bar{X}$
15	26	8	7.4	5	15	12	0.56	1	16	46	8	7.7	6	23	8	0.88
13	23	11	7.3	5	24	10	0.38	2	10	66	10	7.7	6	24	8	0.87
18	34	16	7.6	6	25	6	0.39	3	17	40	16	7.7	6	20	6	1.03
17	29	12	7.5	6	20	6	0.42	4	10	45	11	7.7	7	21	7	0.85
16	51	18	8.0	5	21	6	0.58	5	17	80	17	7.6	5	23	10	0.60
17	42	18	7.4	9	16	4	0.50	6	13	63	16	8.1	9	26	8	1.37
17	15	15	7.1	6	10	8	0.23	7	16	23	16	7.4	8	13	9	0.38

readings prevented zero-day column pH determinations from being taken. Such interference could have been due to charged clay particles and/or a low buffer content of the effluent water. Column effluent from soil number 1 showed an increase with time from a value of 6.7 at day 2 to a value of 7.5 at day 30. Effluent from soil-column number 2 had a value of 6.8 at day 2 and steadily increased to 7.7 by day 30. Column number 3 had an initial value of 8.1; the pH stayed fairly constant thereafter, having a value of 7.5 at day 2 and a value of 7.2 at day 30. The initial column value of soil number 4 was 8.0 and dropped to 6.8 at day 2, and then steadily increased to a value of 7.9 at day 30. The effluent for soil number 5 was fairly constant, having an initial pH of 8.1, a day-2 pH of 7.8, and a day-30 pH of 8.0. The effluent from column number 6 also stayed relatively constant with time; the initial pH was 8.2, the value at day 2 was 7.3, and the pH at day 30 was 7.5. Unlike the other effluents, column number 7 increased from the zero-day reading of 6.5 to 7.4 at day 2; from day 2 the value declined to a pH of 7.0 on day 30.

The beaker pH value for soil number 1 climbed at a higher rate than the effluent, going from a pH of 6.7 at day 1 to a pH of 8.1 at day 30. The pH value of beaker number 2 also climbed at a higher rate than its corresponding column, going from a value of 6.8 at day 1 to a value of 8.3 by day 30. The reading for beaker number 3 was fairly constant; the value at day 2 was 7.8 and at day 30, 7.6. The pH for beaker number 4 climbed at about the same rate as its column, going from a value of 6.8 at day 1 to 8.2 at day 30. Beaker number 5 is the only case where the pH remained at a lower value than that of the effluent it was receiving. The pH for beaker number 5 was 7.3 at day 2 and 7.8 at day 30. Beaker number 6 had a pH of 7.6 at day 2 which remained fairly constant until day 17; after day 17 the value increased to 8.7 on day 18 and 8.8 by day 30. Beaker number 7 had a pH of 7.4 on day 2 and remained fairly constant until after day 25, where it increased to 7.8 by day 30. Table 6 shows the column and beaker mean pH values. As can be seen, in all setups except number 5, the beaker mean pH was greater than that of the corresponding column.

Figure 10 shows the results from total hardness determinations. Due to the large sample size required and because the test is destructive, relatively few determinations were made. Effluent results reveal that columns 1, 2, 4, and 6 exhibit a "slug" effect similar to that encountered with TDS. Most column effluents remained at a fairly constant total hardness after day 2. However, as in the case of TDS, most beaker values rose above the corresponding column values due to concentration by evaporation (see Table 6).

Dissolved ortho-phosphate (DOP) values for the 7 columns and beakers are shown in Figure 11. Column DOP values indicate a "slug" effect for this parameter. However, whereas this effect was of short duration for TDS and total hardness, the slug effect for DOP appeared to be of a longer duration. Column values after day 2 were fairly constant or showed a slight decrease with time. Unlike the trends established with the previous parameters, beaker DOP values appeared to decrease with time. Beaker values from soils 1, 2, 5, and 7 dropped below the DOP concentrations for the corresponding column effluents. That algal cells take up DOP is well documented and could account for the drop in DOP in the beakers. As can be seen in Figure 11, the number 3 beaker DOP concentration remained fairly constant after day 10. As noted earlier, no algal growth developed in beaker number 3, due to the continual blackout conditions; this could explain why the DOP concentration in the beaker did not decrease in this case. As can be seen in Figure 11 and Table 6, soil leached with distilled water provides considerable quantities of DOP.

Figure 12 shows turbidity results for the 7 columns and beakers in Jackson Turbidity Units (JTU). A study of the effluents reveals once again a "slug" effect. Further study of the effluent data for the various soils indicates that, although erratic, the turbidity readings approach a constant value with time. The flow of distilled water to column 5 was stopped for two days and then started again. As can be seen, the turbidity in the column effluent increased to 450 JTU after the flow to the column was interrupted. The beaker did not respond to this increased turbidity from the column, probably because the volume of effluent that reached the beaker after sampling was small. Figure 12 further indicates that most beaker JTU values remained constant or decreased with time. The cheesecloth that held the soil in place in the columns tended to rot as time passed, thus causing erratic column readings. Also, the need to change the flow rate to some of the columns because they were stopping up undoubtedly caused erratic turbidity readings. For these reasons, no mean turbidity values are given in Table 6. It should be pointed out, however, that the beaker turbidity values were usually less than 100 JTU despite mixing caused by continual aeration. This is in the range of turbidity values found in Lake Thunderbird.

Experimental work was terminated on all setups, with the exception of setup number 1, on day 30. Column number 1 continually had a greater percolation rate than the other columns and, consequently, there was more water in beaker number 1 by day 30. Therefore, the beaker phase from soil number 1 was allowed to concentrate by evaporation for an additional 35 days. During this time, the beaker was continually aerated just under the surface. A "Plant-Gro" bulb furnished light to the beaker 12 hours per day. No analyses were conducted during this period in order to conserve water. By day 65, the TDS had increased from 80 mg/l to 210 mg/l. Further analysis at this point yielded the results shown in Table 7.

TABLE 7

WATER QUALITY OF BEAKER NUMBER 1 AT DAY 65

pH	8.5
Alkalinity	160 mg/l as $CaCO_3$
Total Hardness	164 mg/l as $CaCO_3$
Calcium Hardness	84 mg/l as $CaCO_3$
Magnesium Hardness	86 mg/l as $CaCO_3$
DOP	0.12 mg/l as Phosphate

If the results of Table 7 are compared to the values found in Table 3, page 23, it can be seen that the pH and DOP values of beaker number 1 compare favorably with those of Thunderbird. Furthermore, if TDS is used as an index of comparison between the two systems, then the ratio of Thunderbird TDS to beaker TDS (250/210) multiplied by beaker values yields an alkalinity of 190 mg/l, a total hardness of 195 mg/l, a calcium hardness of 100 mg/l, and a magnesium hardness of 102 mg/l. Although such ratios must be used with a great deal of caution, it can be seen—at least in this instance--that the beaker values compare favorably with Table 3 after being modified by the TDS ratio. The use of TDS as a comparison index will be discussed in more detail in a later section.

There are several conclusions drawn from the small-column phase of this research. First, percolation columns in this area do behave as described by Bricker. That is TDS, total hardness, and DOP column effluent concentrations remained constant with time. Secondly, mean column values for each parameter appear remarkably similar considering only seven rather small soil samples were used from an area of 256 square miles. It also seemed apparent that effluent waters retained in beakers undergo many of the same changes that occur when stream waters are impounded. Beaker waters showed an increase in pH with time that seemed, at least partially, to begin with the establishment of algal growth. There was a reduction of color and turbidity in all beakers; TDS, calcium hardness, and magnesium hardness increased due to concentration by evaporation. There was also an increase in beaker alkalinity that could have been due in part to concentration by evaporation. There was an observed decrease in beaker DOP, possibly due to algal uptake or chemical precipitation. Finally, results of analyses of beaker number 1 on day 65 indicate that column fed systems do approach Lake Thunderbird in water quality—at least for the parameters measured.

On the basis of these results, work was begun on the large-column system shown in Figure 7.

Laboratory Impoundment Model

On June 19, 1969, the flow of distilled water to the column shown in Figure 7 was initiated. More frequent analyses and additional parameters were possible in this phase of the work due to the larger effluent and tank volumes. In addition to the 7 parameters listed on page 45, the following parameters were incorporated into the study.

1. Dissolved total-phosphate (DTP) - Hydrolysis achieved by placing centrifuged samples in autoclave at 121°C for 15 minutes. Samples then analyzed by same procedure used for DOP determinations.

2. Dissolved oxygen (DO) - Azide modification of Iodometric method as in "Standard Methods." Also, Precision Scientific Galvanic Cell Oxygen Analyzer.

3. Five-day Biochemical Oxygen Demand (BOD) - As in "Standard Methods."

4. Chloride - Mercuric Nitrate titration as in "Standard Methods."

5. Manganese - Periodate method as in "Standard Methods."

6. Iron - Phenanthroline method as in "Standard Methods."

7. Total Solids - Gravimetric as in "Standard Methods."

8. Carbon - F & M Model 185, C,H,N Analyzer.

9. Nitrogen - F & M Model 185 C,H,N Analyzer.

10. Total Bacteria - As described by Millipore. (20)

11. Coliform Bacteria - As described by Millipore.

12. Temperature - Electronic thermistor.

Some of the parameters listed above were determined only occasionally and some were used only for special studies. Again, results from Lake Thunderbird on all these parameters are available, thus facilitating comparisons.

Research was conducted on the model for a period of 450 days. For convenience, results from days 0-250 and from days 250-450 will be discussed separately. Likewise, results from some of the studies dealing with research applications of the model, mentioned on page 17, are discussed individually.

Days 0-250

The primary objective of this phase of the work was to establish and maintain a laboratory model of Lake Thunderbird. A second objective was to further study the slug effect in the column effluent mentioned earlier. A third objective was to compare on a parameter-by-parameter basis the column effluent and the receiving tank. A final objective during this period was to determine if dissolved ^{45}Ca is sorbed onto particulate (clay) materials.

The peristaltic pump was initially regulated to remain on for 30 minutes each 6 hours. At a pumping rate of 12.2 ml per minute, 366 ml of aerated, distilled water was delivered to the column each 6 hours--a simulated rainfall of approximately 0.8 inch per 6 hours, or a total of 3.2 inches per day. The first effluent was received from the column on June 21, hereafter referred to as day-zero.

The initial column effluent had an intense yellow color that persisted for about 17 days. By day 20, the tank contained a biological population that appeared to be comprised chiefly of blue-green algae, rotifers, and ciliates. Bacteriological analysis on the tank water on day 25 yielded a total bacteria count of 400 colonies per ml and a coliform count of 60 colonies per ml. On day 31, the total bacteria count was 290 colonies per ml and the coliform count was 25 colonies per ml in the tank. By day 30, _Euglena_, _Chlorella_, and several species of diatoms were growing in the tank waters. On day 31, a small amount of Lake Thunderbird water known to contain Cyclops, Daphnia, Bosmina, and Diaptomus was added to the tank. These organisms represent the majority of net-sized plankton in Thunderbird. The four genera survived for the duration of the study.

Column effluent samples were collected by periodically placing a 400-ml beaker directly under the column. Tank samples, except where noted, were always collected from the same tank location and just under the surface. Because the tank was well mixed, the tank samples were considered representative of the whole tank. No tank samples were collected before day 16.

Earlier investigations by the author suggested that the slug effect mentioned earlier might depend on the dryness of the soil sample. Accordingly, it was decided to periodically stop the flow of water to the column to let the soil dry. It should be pointed out that the soil when placed in the column was fairly wet, being collected shortly after recent rains.

Column effluent and tank TDS results for the first 250 days are shown in Figure 13. The effluent concentration exhibited no slug effect and remained fairly constant for the first 60 days at about 140-150 mg/l. The tank TDS value from days 16-60 also remained fairly constant at about 140 mg/l. To further investigate the relation between soil-dryness and the slug effect, flow to the column was stopped on day 58. Flow was started again on day 74, stopped on day 80, started on day 89, stopped on day 97, and finally, started again on day 115. As can be seen, the response was an increase in TDS in the effluent each time flow to the column was interrupted. Furthermore, there appeared to be a relation between the length of time flow was stopped and the relative increase in TDS. It can be seen that by day 130, the effluent TDS had decreased to a base level of approximately 140-150 mg/l, indicating that a steady state condition had been reached.

The tank TDS increased from a day-60 value of 140 mg/l to 250 mg/l by day 120. This increase is attributable to the column slug effect and, due to less column flow during this period, increased concentration due to evaporation. The tank TDS remained fairly constant from days 120-135.

By day 140, the column had almost ceased percolating. Therefore, on day 145, the column soil was replaced. The replacement soil was comprised of the same two soils used previously; however, to insure better percolation, the composite contained a much higher percentage of the red, sandy soil. It should be pointed out that this soil had been stored in the laboratory since June 1969 and was quite dry.

Figure 13 reveals that the initial TDS value of the column effluent on day 150 was 950 mg/l. This value rapidly dropped to a base level of approximately 40 mg/l. The tank responded to the column slug effect by increasing to a maximum TDS of 365 on day 150. The tank TDS decreased from day 150 to a value of 200 mg/l by day 190 and remained fairly constant thereafter.

On day 194, the flow to the column was reduced by 50 percent. Thus the volume of flow now was 183 ml every 6 hours, which corresponds to a simulated rainfall to the column of 0.4 inch every 6 hours or 1.6 inches per day. Figure 13 shows that this change in flow rate had no

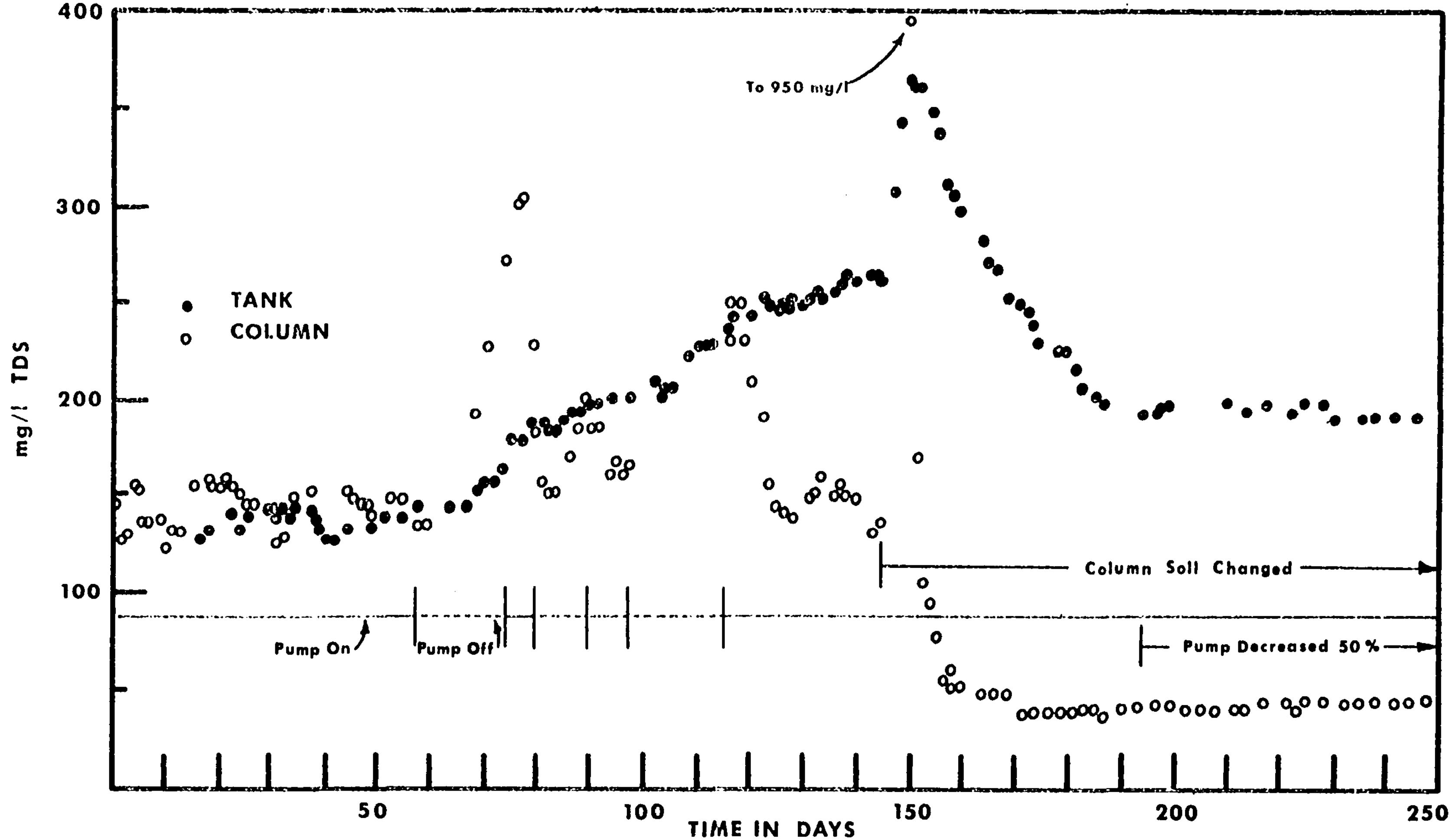

Figure 13. Comparison of Column Effluent and Tank Total Dissolved Solids.

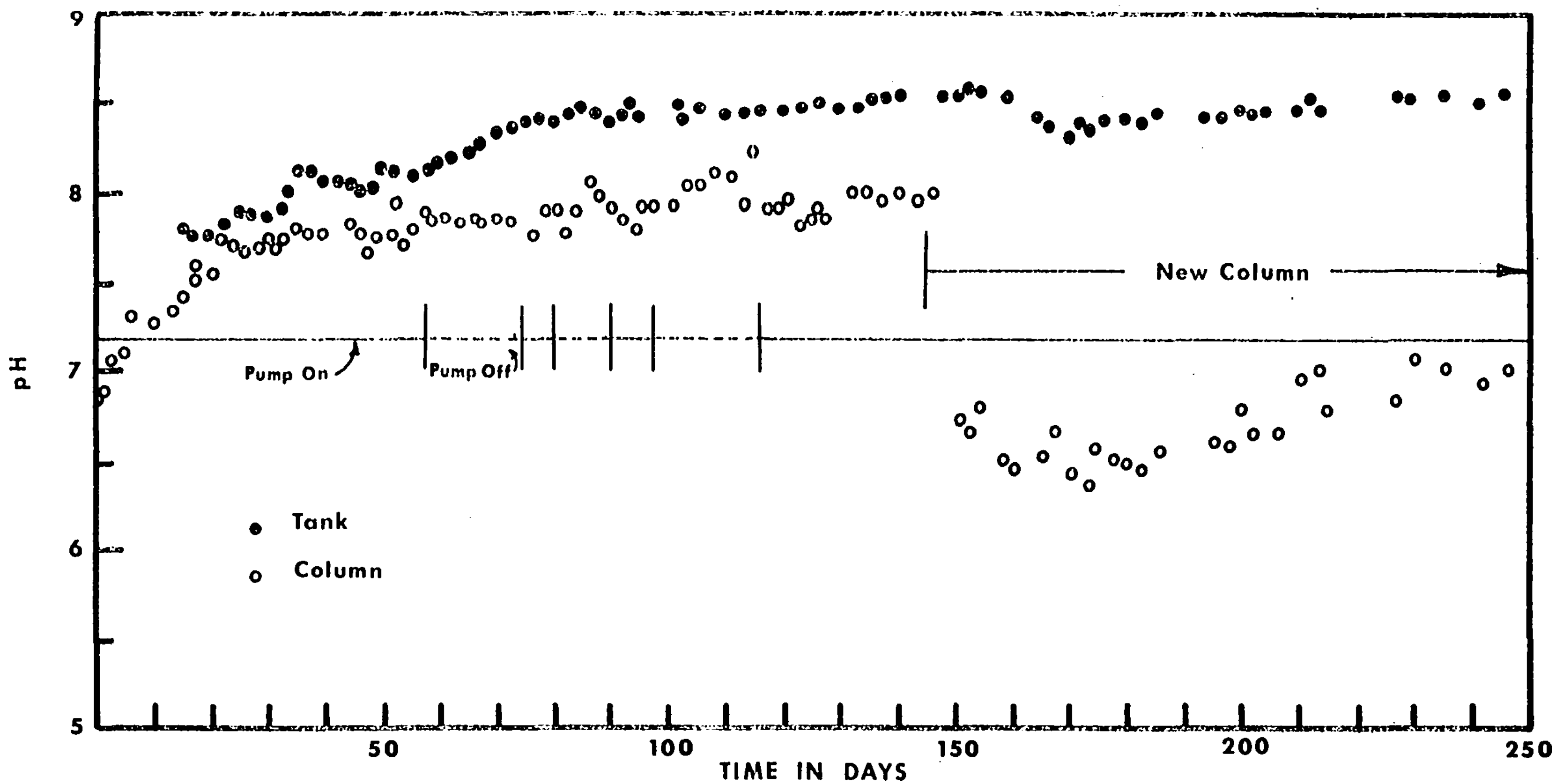

Figure 14. Comparison of Column Effluent and Tank pH.

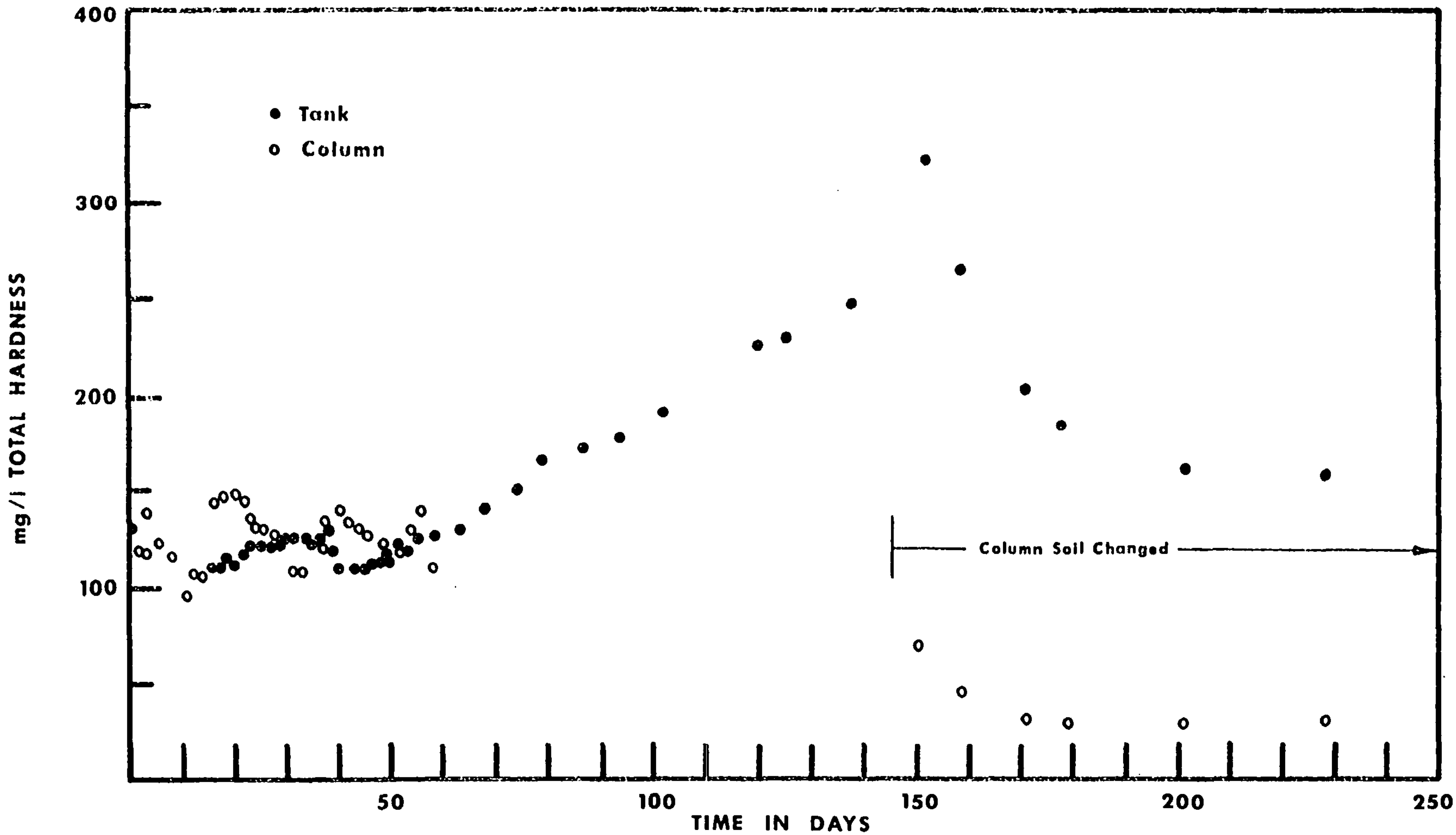

Figure 15. Comparison of Column Effluent and Tank Total Hardness as mg/l $CaCO_3$.

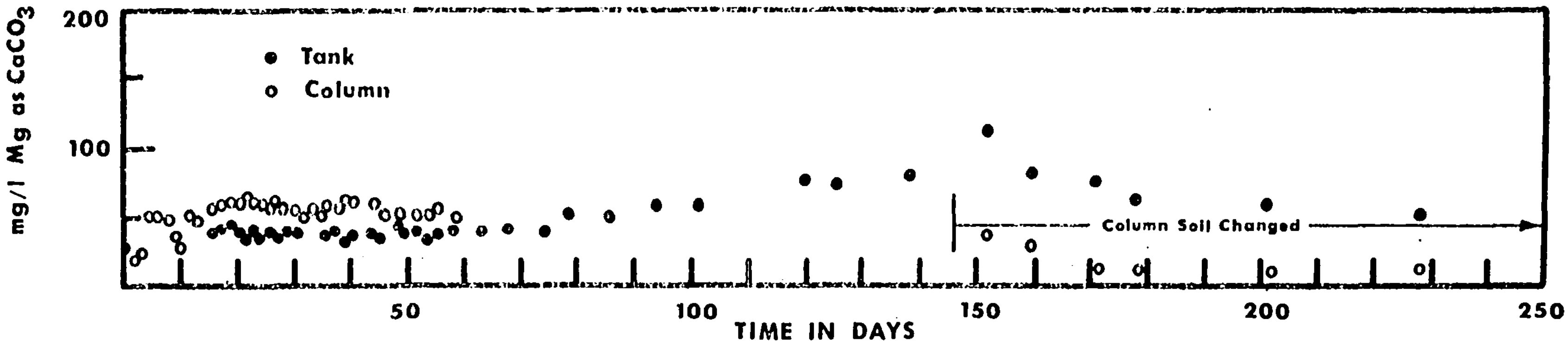

Figure 16. Comparison of Column Effluent and Tank Magnesium Hardness as mg/l $CaCO_3$.

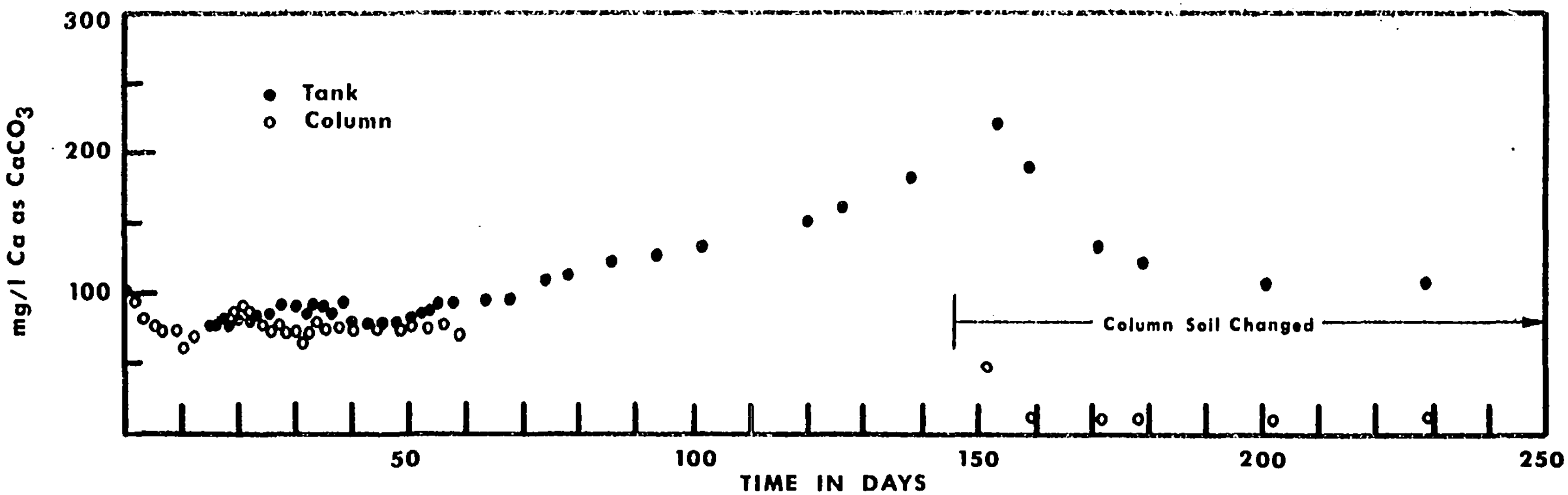

Figure 17. Comparison of Column Effluent and Tank Calcium Hardness as mg/l $CaCO_3$.

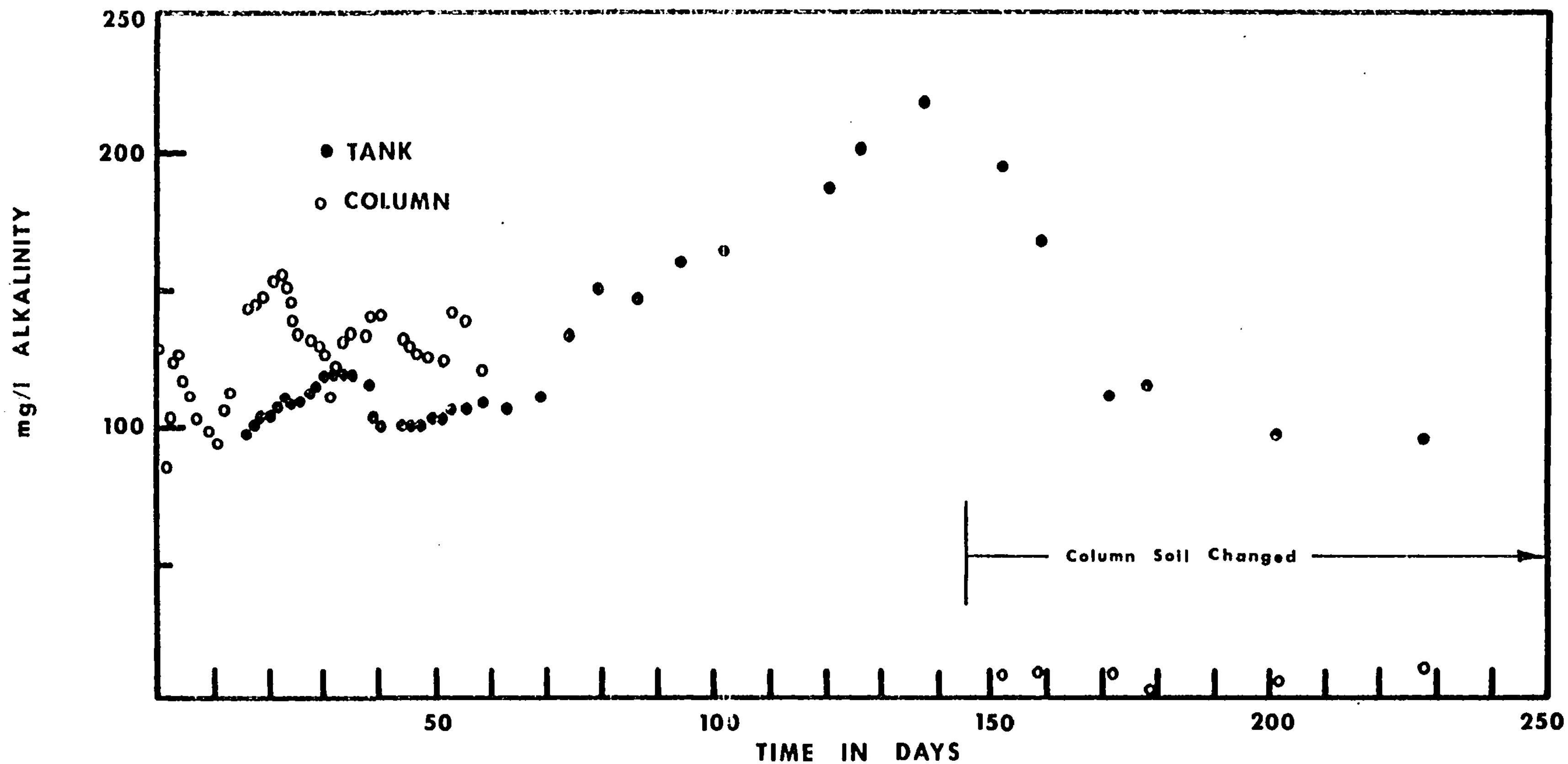

Figure 18. Comparison of Column Effluent and Tank Alkalinity as $CaCO_3$.

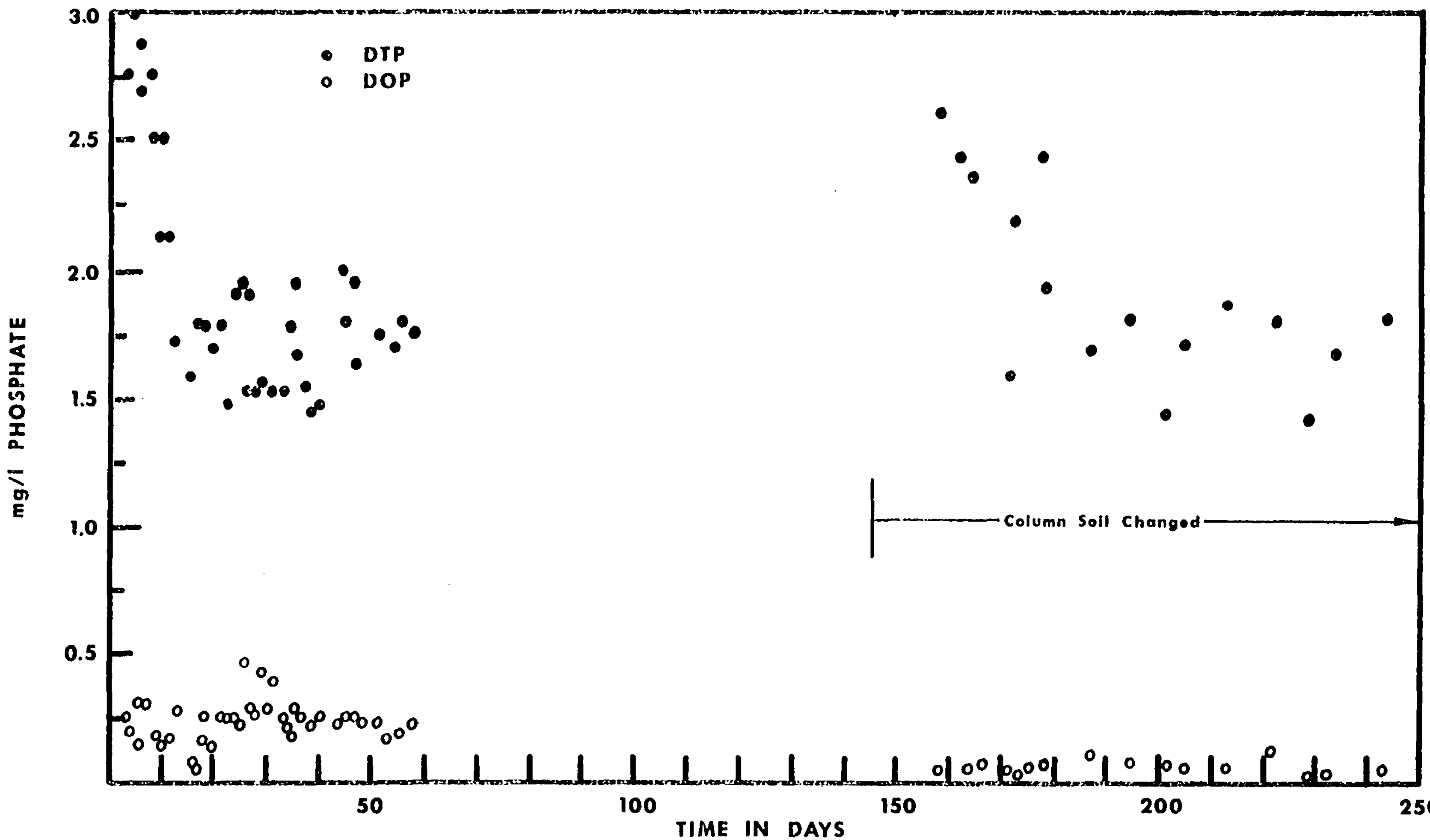

Figure 19. Column Effluent Dissolved Total Phosphate and Dissolved Ortho-Phosphate.

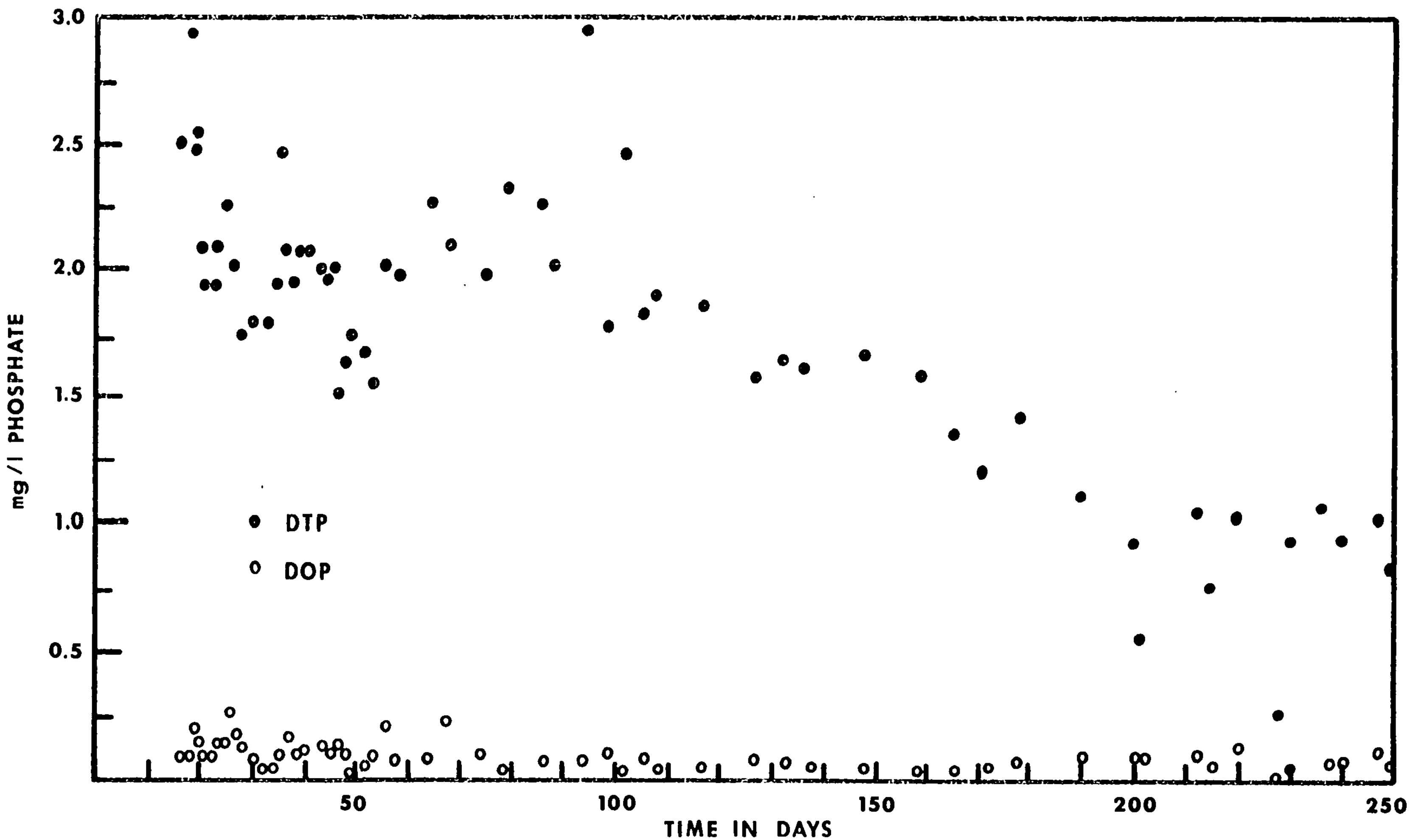

Figure 20. Tank Dissolved Total Phosphate and Dissolved Ortho-Phosphate.

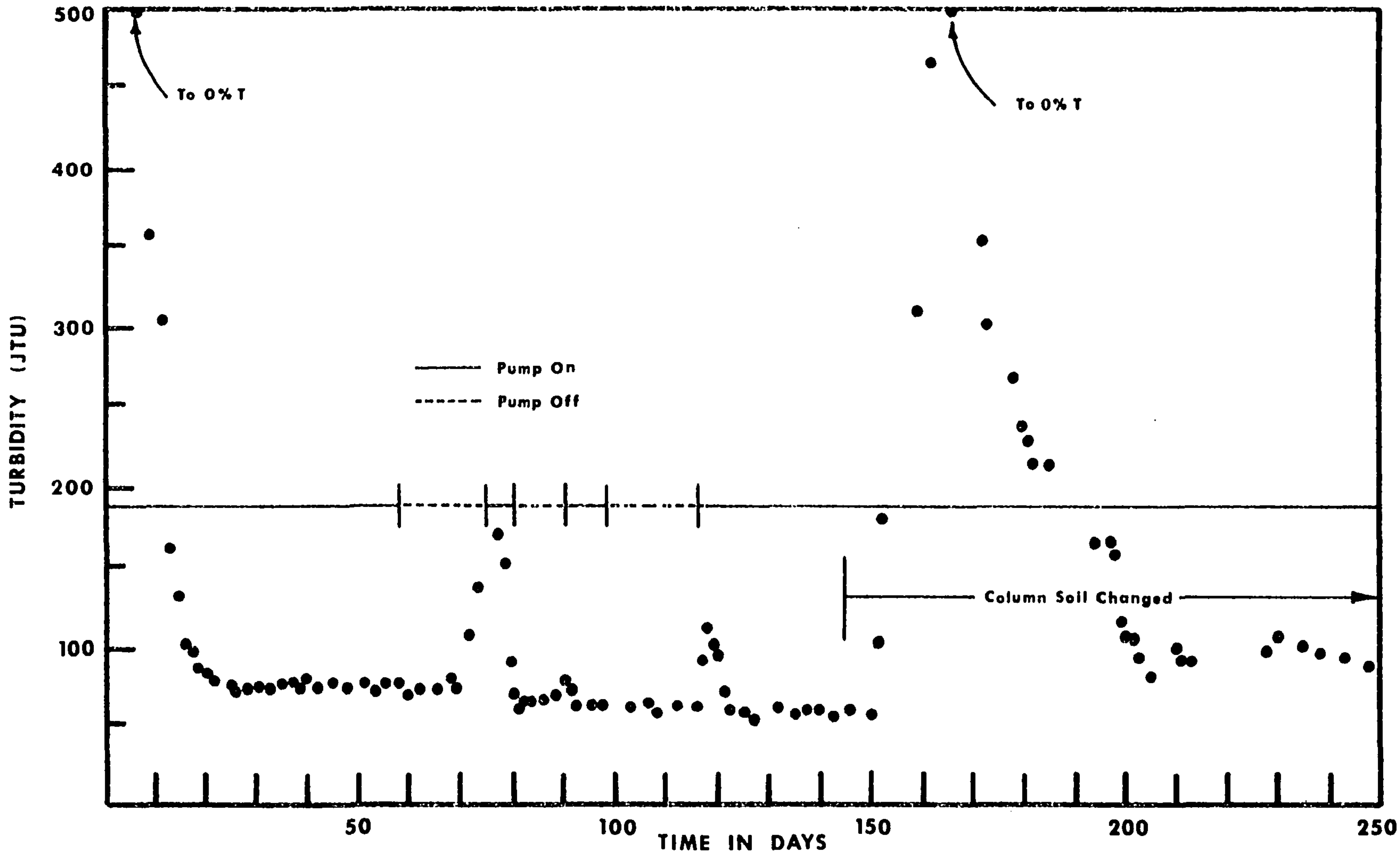

Figure 21. Column Effluent Turbidity in Jackson Turbidity Units.

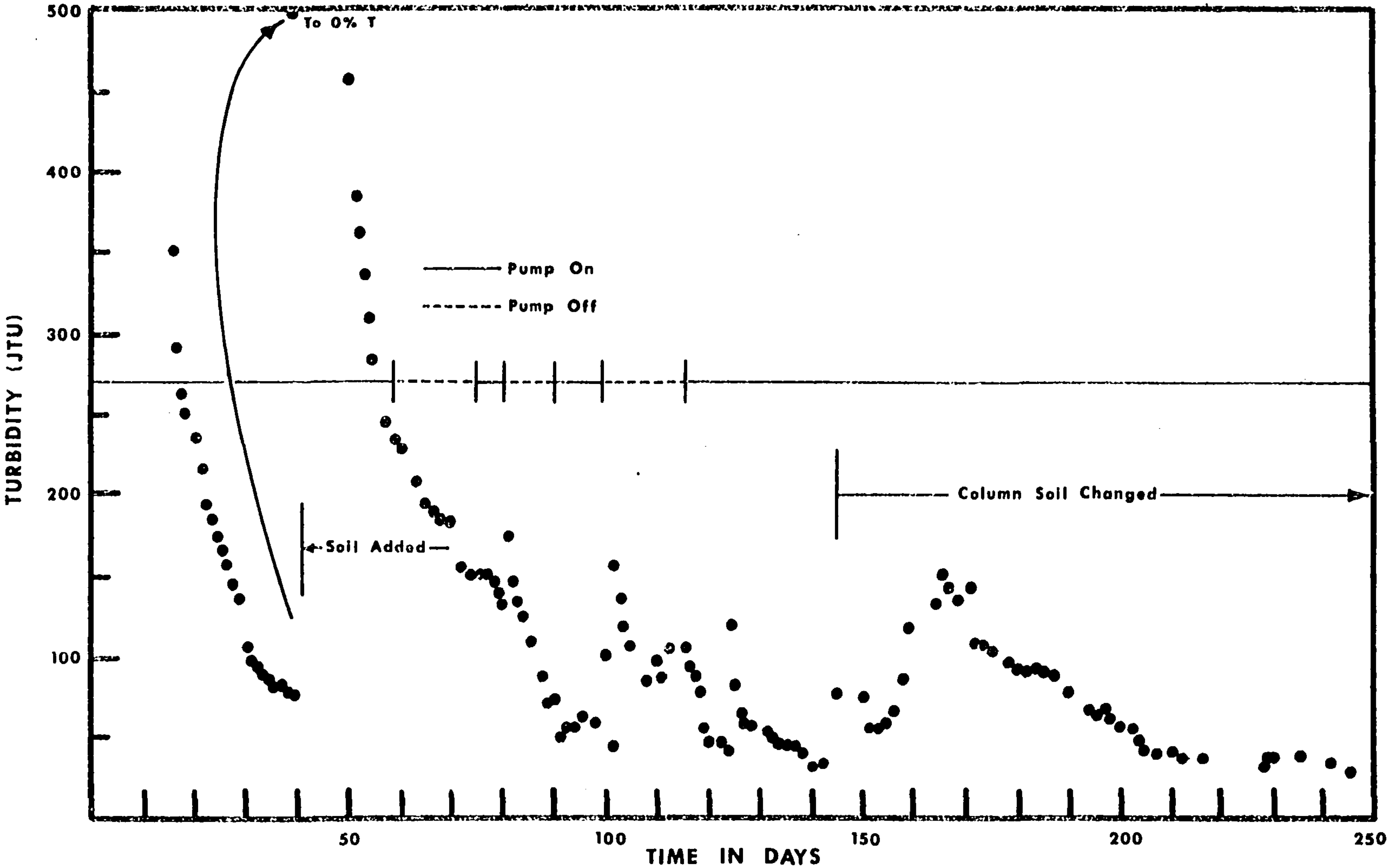

Figure 22. Tank Turbidity in Jackson Turbidity Units.

effect on effluent TDS. Since the column effluent had a diluting effect on the tank, it was possible, by regulating the volume of column effluent, to somewhat counteract the effect of evaporation on the tank. By day 190, the tank water depth fluctuated between 8 and 9 inches, corresponding to a volume of about 40-44 liters respectively.

Column effluent and tank pH results for the first 250 days are shown in Figure 14. As can be seen, the column effluent pH increased from 6.8 at day zero to 7.7 by day 20; from day 20, the effluent pH showed a slow increase to a value of 8.0 by day 145. It appeared that the interruption of flow to the column (pump on-pump off) had little effect on the effluent pH. The tank pH increased from a value of 7.8 at day 16 to 8.4 by day 83. The tank pH stayed relatively constant from day 83 to day 153 at a value of 8.4-8.6.

After the soil was replaced in the column, the initial effluent pH was 6.7 at day 150. The effluent pH dropped to about 6.5 at day 60 and remained fairly constant until day 190. At this point, the effluent pH climbed to a value of 6.9 at day 210 and remained fairly constant thereafter. The tank pH dropped from a value of 8.6 at day 153 to 8.3 at day 170; from day 170 the pH increased back to about 8.5 and remained fairly constant. That the tank did not respond more to the lower effluent pH indicates that the tank was well buffered.

Column effluent and tank total hardness results are shown in Figure 15. The effluent gave no indication of a slug effect and remained at approximately 130 mg/l from day 0 to day 60. Because of sample size, destructiveness of the test, and reduced column effluent volume due to the pump on-pump off cycle, no effluent total hardness data was taken from days 60 to 150. The tank total hardness was fairly constant from day 16 to day 60, with a value of approximately 125 mg/l. Tank total hardness increased from the value at day 60 to a value of 230 mg/l at day 135. The increase in tank total hardness is attributable to increased concentration due to evaporation because of less column flow, and any undetected slug effect that might have occurred. The effluent total hardness at day 150, after the column soil had been changed, was 75 mg/l and dropped to 35 mg/l by day 171; the total hardness after day 171 remained fairly constant. The tank had a maximum total hardness of 318 mg/l on day 152 and dropped to a base level of approximately 160 mg/l by day 200.

Figures 16 and 17 compare column effluent and tank magnesium and calcium hardness, respectively. Figure 16 shows that effluent magnesium hardness remained at about 50 mg/l for the first 60 days. For reasons mentioned earlier, no effluent data was obtained from days 60-150. As can be seen, the tank magnesium hardness had a base level value of about 45 mg/l and it increased from this value at day 60 to a value of 70 mg/l by day 135. Tank magnesium hardness reached a maximum of 105 mg/l at day 152 and then dropped to a base level of about 50 mg/l. The column effluent at day 150 had a magnesium hardness of 30 mg/l and dropped to a base level by day 171 of 15 mg/l.

The effluent calcium hardness remained at about 80 mg/l for the first 60 days. The tank calcium hardness was also about 80 mg/l during this period and then increased at a faster rate than did the tank magnesium hardness to a value at day 135 of 160 mg/l. The tank calcium hardness increased to a maximum at day 152 of 213 mg/l and then dropped to a base level of 110 mg/l. The effluent value at day 150 was 45 mg/l and then dropped to a base level of about 20 mg/l.

Figure 18 compares column effluent and tank total alkalinity values for the first 250 days. Effluent alkalinity was erratic for the first 25 days and then leveled off at about 130 mg/l through day 60. No effluent data is available from days 60-152. The effluent alkalinity on day 152, after the column soil had been changed, was about 10 mg/l and remained relatively constant thereafter. The tank alkalinity from days 16-60 averaged about 110 mg/l. The tank value increased from day 60 to a value of 185 mg/l at day 120; the maximum tank alkalinity was 216 mg/l at day 135. The tank alkalinity then dropped to a base level of approximately 100 mg/l. The increase in tank alkalinity from days 60-135 is attributed to increased concentration due to evaporation and any possible undetected effluent slug effects. Tank phenolphthalein alkalinity was usually not detectable during this period; in the few cases it was detectable, it ranged from a value of 4 mg/l on day 93 to a maximum of 10 mg/l on day 159. Therefore, most of the total alkalinity was in the bicarbonate form.

Figure 19 shows the column effluent results from the dissolved total-phosphate (DTP) and dissolved ortho-phosphate (DOP) analyses. The column effluent did exhibit somewhat of a slug effect for DTP, dropping from initial values of about 3 mg/l to an average value of 1.75 mg/l on day 12 that persisted through day 60. For the same reasons given earlier, no DTP or DOP data is available from days 60-150. DOP analyses revealed no slug effect and the readings averaged about 0.25 mg/l through day 60. DOP values averaged about 0.08 mg/l from day 159 onward. DTP results once again indicated a slug effect, after the column soil was changed, with a value of 2.60 mg/l on day 159 that dropped to a base level of about 1.70 mg/l by day 185.

DOP is the inorganic form of phosphate and this could explain why this parameter behaved in a manner similar to TDS, total hardness, and calcium and magnesium hardness, in that no slug effect was observed. DTP, however, appeared to be made up chiefly of meta-phosphate: meta-phosphate can be determined by subtracting DOP values from DTP values. It is possible, therefore, that much of the DTP was in the organic form and that the results shown in Figure 19 represent an initial flushing from the column, rather than leaching.

Results from analyses of tank waters for DTP and DOP are shown in Figure 20. As can be seen, the DTP concentration showed a slow decrease from a value of 2.50 mg/l at day 16 to a value of 0.90 mg/l at day 200, and then tended to level off. Tank DOP values averaged about 0.12 mg/l from days 16-60; from days 60-250, the DOP remained fairly constant at a value of about 0.06 mg/l. By comparing Figures 19 and 20, it can be seen that by day 200 the tank DTP (0.90 mg/l) had dropped to a lower value than the DTP of the column effluent (1.70 mg/l). It is apparent from Figure 20 that neither DTP nor DOP responded to evaporation or any possible slug effects as had the previous parameters discussed. It is concluded that the decrease in tank DTP is due primarily to algal uptake, chemical precipitation, and/or physical sorption onto particulate matter.

Column effluent turbidity results in Jackson Turbidity Units (JTU) are shown in Figure 21. As shown, the initial results were greater than 500 JTU (0% transmission of the sample compared to distilled water). The turbidity readings had dropped to a value of about 75 JTU by day 20 and remained fairly constant through day 70. As shown, interruptions of flow to the column were followed by increases in effluent turbidity. After the change in column soil, the turbidity once again increased to a value greater than 500 JTU, and then slowly decreased to an average value of 90 JTU by day 200.

Tank turbidity results are shown in Figure 22. The turbidity value at day 16 of 350 JTU dropped to a value of 75 JTU by day 40. At this point, approximately 200 grams of the same type soils used in the column were added to the tank to study the effects of a large increase in turbidity. As shown, the tank turbidity immediately increased to a value greater than 500 JTU and then rapidly dropped to a value of 150 JTU at day 80. The erratic turbidity values between days 80 and 140 are probably due to the slug effects received from the column effluent. In response to the effluent slug effect that followed the change in column soil, the tank turbidity increased from a value of 25 JTU at day 140 to a maximum of 155 JTU on day 165. After day 165, the tank turbidity decreased to a base level of approximately 35 JTU by day 210. Comparison of Figures 21 and 22 reveals that the tank turbidity after day 210 was lower than that of the column effluent for the same time period. This indicates that the particulate (clay) materials were settling to the bottom of the tank, thus forming a sediment layer.

The initial turbidity slug effect and the effect after changing the column soil (Figure 21) were probably due to the physical disturbance of the soil when placing it in the column.

It was felt that quantitative volume comparisons between the lake and the model were relatively meaningless insofar as water quality was concerned. For example, the ratios of lake volume to tank volume (120,000 acre-feet/40 liters) and of total volume available to the lake per year to total volume available to the tank per year (505,000 acre-feet/350 liters) were difficult to interpret in terms of water quality. Therefore, comparisons were made only in terms of water quality. TDS was used as a comparison index for total hardness, calcium hardness, magnesium hardness, and alkalinity, as these parameters behaved similar to TDS for the first 250 days. In other words, the ratio of TDS to total hardness was fairly constant, ranging from 1.1 to 1.3 through the four time periods shown in Table 8. The same was true for the ratios of TDS to calcium hardness, TDS to magnesium hardness, and TDS to alkalinity; the maximum ratio range occurred with TDS to magnesium hardness--ranging from a minimum of 3.1 to a maximum of 4.0 over the four time periods. Thus TDS has been used as an index of these minerological constituents of water quality.

The equilibrium concentrations of the effluent minerological constituents were found to be considerably less than the corresponding values in Thunderbird. Such results seem reasonable when consideration is given to the concentrating effect that the considerable evaporation losses from the lake must cause. It is believed, then, that the concentration of these materials in Thunderbird is regulated, in part, by the diluting effect of watershed-rainfall-runoff, temperature, surface area of lake, and intensity and duration of prevailing winds. Correspondingly, column effluent--excluding slug effects--provided dilution of tank waters. The rate of evaporation in the tank was chiefly a function of tank aeration, as the tank surface area and water and air temperature were essentially constant. Furthermore, sample withdrawal from the tank was considered roughly analogous to lake pumpage withdrawals and leakage through the dam in that these are non-concentrating losses. Therefore, using TDS as a guide, it was possible to increase the concentration of tank materials by regulating tank aeration and volume of column input. As shown in Figures 13 and 15-18, tank materails were concentrated to values above those encountered in Thunderbird and then decreased to values less than found in Thunderbird. Moreover, from day 200 it was possible to maintain approximate steady-state conditions by balancing tank evaporation and sampling losses with column input volume. The tank depth after day 200 fluctuated between 8 and 9 inches (about 40 l). Thus, although a tank TDS of 200 mg/l was less than the TDS of Thunderbird, it was felt that this tank depth and volume were necessary to maintain the model.

Table 8 compares tank results from 4 different time intervals during the first 250 days with Thunderbird. The actual tank results and the modified results obtained by multiplying the actual results by the ratio of lake TDS to tank TDS are shown for each time period. A TDS value of 250 mg/l was used for the lake. The results from days 16-60 represent the initial tank steady-state values described previously. The period from days 120-135 was chosen because the lake to tank TDS ratio was 1. The interval from days 150-152 represented the maximum values obtained during the first 250 days and consequently, the

TABLE 8

COMPARISON OF TANK (ACTUAL) AND TANK (TDS-MODIFIED) WITH LAKE THUNDERBIRD
Days 0-250

Parameter	Days 16-60 Tank Actual	Days 16-60 (Lake TDS/ Tank TDS) (250/140)	Days 120-135 Tank Actual	Days 120-135 (Lake TDS/ Tank TDS) (250/250)	Days 150-152 Tank Actual	Days 150-152 (Lake TDS/ Tank TDS) (250/365)	Days 200-250 Tank Actual	Days 200-250 (Lake TDS/ Tank TDS) (250/200)	Lake Thunderbird
TDS	140	250	250	250	365	250	200	250	250-300
Total Hardness	125	223	230	230	318	216	160	200	170-230
Calcium Hardness	80	142	160	160	213	145	110	138	90-120
Magnesium Hardness	45	80	70	70	105	71	50	63	80-110
Alkalinity	110	196	185	185	216	147	100	125	180-200

smallest TDS ratio. Finally, the interval from days 200-250 represented final tank steady-state conditions.

The modified total hardness values listed in Table 8 all fall within the total hardness range found in Lake Thunderbird. However, the ratio of calcium hardness to magnesium hardness in Thunderbird is about 1, whereas this ratio in the tank ranges from 1.8 for days 16-60 to 2.3 for days 120-135. This result is probably due to the fact that the effluent calcium to magnesium ratio is also greater than 1. Calculation of the solubility products of $CaCO_3$ and $Mg(OH)_2$ fail to explain the large tank ratio or the 1 to 1 ratio found in Thunderbird. In any case, as compared to Thunderbird, the modified calcium hardness values are all high and the modified magnesium hardness values are all low. Modified alkalinity values for days 16-60 and days 120-135 fall within the Thunderbird range, whereas the modified alkalinity values for the two remaining periods fall below the range found in Thunderbird.

Results shown in Figure 13 indicate that the column slug effect could be related to the relative dryness of the soil. As shown in Figure 13 and as mentioned earlier, when the column soil was wet, no slug effect was observed. Even when the flow to the column was decreased by 50 percent (Figure 13), the column soil was saturated with water and no slug effect was seen. This would indicate that the effluent concentration was independent of column flow as long as the soil remained wet. It is suggested that when the soil begins to dry, water evaporating from the soil interstitial spaces leaves behind a residue. This residue was composed of materials that previously had been dissolved (leached) from soil particles and therefore, was readily soluble in the next flow of water. It is further suggested that when the pump was off the column soil would begin to dry from the top down. Therefore, the longer the pump remained off the more the column soil dryed and the greater the resulting slug effect. Evidence from Lake Thunderbird indicates that slight increases in TDS and total hardness have occurred following late-summer rains that had been preceded by long periods of dryness. On the other hand, decreases in lake TDS and total hardness have been observed after spring rains, when the soil had been wet for some time. Thus although the column effluent (watershed-rainfall-runoff) does appear to have a diluting effect on the tank (impoundment), at times this effect can be quite the opposite and possibly, quite significant.

As mentioned in Chapter 1, soil is known to furnish all the nutrients, vitamins, and other materials needed to sustain biological growth. That such growth was sustained throughout the duration of this study was evidence of this fact. Furthermore, according to Alexander (21), soil is a source of a vast array of different organisms. No specific biological examinations such as species identification or species diversity were done; emphasis was placed on chemical quality. Furthermore, as the model obviously did not include the various habitats found in Thunderbird, any biological comparisons between the model and the lake would probably have been meaningless. However, because of the influence aquatic organisms have on water quality through their interactions with the aquatic environment, growth in the tank was considered essential.

As mentioned earlier, several of the parameters used in this study are known to be indices of biological trends. For example, the result that tank pH increased with time to a value of 8.5 can be attributed, at least partially, to the fact that algae removed CO_2 from solution--thus causing a shift in alkalinity forms from bicarbonate to carbonate, and from carbonate to hydroxide (Sawyer, 22). Likewise, the decrease of DTP and DOP with time can be explained by algal and bacterial uptake (Levin and Shapiro, 23). Further evidence that the tank-dissolved phosphate was incorporated into the biomass is presented in a later section. That there is a positive correlation between the color found in new impoundments and organic materials determined as COD has already been mentioned; according to Gjessing and Sandal (12), the decrease in color with time can be attributed to biologic processes. Therefore, the reduction of initial tank color could be considered as further evidence of biologic action.

Results from further analyses of tank waters at day 250 are shown in Table 9 and are compared to Lake Thunderbird. As can be seen in Table 9 all tank values with the exceptions of DTP and chloride fall within the ranges found in Thunderbird.

Results from the first 250 days indicated that a column-fed model of Thunderbird could be established that was remarkably similar to the lake in terms of chemical water quality. Results further indicated that the column was capable of continually supplying biological and chemical constituents to the tank, thus allowing the model to be maintained over relatively long periods of time. Concentrations of several tank materials--those defined as representing minerological constituents--appeared to be controlled primarily by the amount of

TABLE 9

COMPARISON OF TANK RESULTS AT DAY 250
WITH LAKE THUNDERBIRD

	Tank	Lake
pH	8.5	8.0-8.5
DOP (mg/l)	0.06	0.01-0.15
DTP (mg/l)	0.90	0.20-0.30
Turbidity (JTU)	35	low of 40
COD (mg/l)	12	10-15
5-day BOD (mg/l)	3	1-3
Manganese (mg/l)	0.05	0-0.25
Iron (mg/l)	0.03	0.0.10
Chloride (mg/l)	12	25
Total Bacteria*	200/ml	95-350/ml
Coliform Bacteria*	0-1.2/ml	0-1.5/ml

* Colonies/ml

used in this research was particularly suited for the use of radioactive tracers.

The soils in the Thunderbird region contain a high percentage of clay. Accordingly, most of the suspended solids and turbidity in Thunderbird, and in the model, are due to suspended clay particles. Furthermore, according to Thompson (24), Bailey (25), and McLean (26), clay particles possess the capacity for cation- and anion-exchange. Bailey (25) believes that there are a finite number of sites on the surfaces of individual clay particles that are involved in exchange. According to Marshall (27), these sites are charged and the number of positive sites per particle are equal to the number of negative sites.

It seems plausible, therefore, that there could exist a dynamic equilibrium between dissolved ions and ions attached to the surface of suspended clay particles. The objective of this study, then, was to determine if there was a shift of calcium-45 in the tank waters from the dissolved phase to the particulate phase.

On day 5, 0.11 mg calcium ($CaCl_2$), labeled with ^{45}Ca, was added to the 40 liters of aerated, distilled water being pumped to the column. It was hoped that the column effluent could be monitored to determine if the radioactive calcium was in the dissolved phase or the suspended phase. It was further hoped that the fate of the radioactive calcium, once it had entered the tank, could be determined. However, no radioactive calcium was ever detected in the column effluent. Subsequent investigation after 100 days indicated that the calcium-45 had not passed beyond the top inch of soil in the column.

On day 23, a chemically insignificant amount of calcium, 0.16 mg calcium ($CaCl_2$), labeled with ^{45}Ca, was mixed throughout the 40 liters of water in the tank. Samples were then periodically collected through day 103 from the top, middle, and bottom of the tank. Half of the sample from each depth was filtered through a 0.45μ membrane filter. Five-ml of filtrate and 5-ml of non-filtered sample were then placed in ribbed counting planchetts and brought to constant weight. After the residue weights were determined, each sample was counted with a G-M system. Each sample planchett was counted 4 times, being rotated 90 degrees for each count.

Statistical analysis of results from samples taken from the top of the tank indicated no significant difference between dissolved and total samples at the 5 percent level ($F_{1,70} = 0.49$). There was also no statistical difference at the 5 percent level between dissolved and total samples at the middle depth ($F_{1,63} = 0.71$). The bottom depth also showed no statistical difference between the two samples at the 5 percent level ($F_{1,64} = 0.78$). Further analysis of variance indicated that if the corresponding total and dissolved counts were combined, that there was no statistical difference at the 5 percent level between the top, middle, and bottom locations ($F_{2,200} = 0.87$). It should be pointed out that the suspended solids ranged from about 0.1 mg/5 ml to 40 mg/5 ml during the course of study. Furthermore, it was determined that self-absorption by the sample was not significant over the total solids range encountered.

The results of this study appeared somewhat paradoxical upon first study. Results indicated that all detectable radioactive calcium in the distilled water pumped to the column was lost from the dissolved phase. Yet results from the tank indicated that all radioactive calcium remained in the dissolved phase. The following explanation is suggested for these results.

It was assumed that at the top of the column the distilled water contained very few dissolved materials, mostly calcium and chloride ions. It is therefore suggested that there was a shift in materials from sites on the soil (clay) particles to the distilled water. However, as the water percolated downward through the saturated soil, materials that had been dissolved from the soil were exchanged with materials sorbed on the solid, soil phase. Thus, eventually a dynamic equilibrium was established that resulted in no further net increase of materials in the dissolved phase. In turn, each successive flow of distilled water dissolved materials from the top region of the column and simultaneously exposed additional soil particle surfaces containing potentially exchangeable (dissolvable) materials. It is further assumed that in the top portion of the column, the ratio of the non-radioactive cations attached to the soil (clay) surfaces to the calcium-45 in solution was large. Thus if equilibrium conditions were in effect, the probability of the dissolved calcium-45 --assuming it behaved like its non-radioactive counterpart--being exchanged with a non-radioactive cation would appear quite large. In other words, after dynamic equilibrium was established, the probability of any detectable calcium-45 appearing in the dissolved phase at any one time was quite low.

Much the same type of argument is suggested to explain the results found in the tank. That is, that the calcium-45 remained in the dissolved phase. In this case, it was assumed that the ratio of non-radioactive cations in the tank dissolved phase to dissolved calcium-45 was very large. Therefore, if dynamic equilibrium did exist between materials in the dissolved phase and materials attached to suspended clay particles, the probability of exchange involving a radioactive calcium ion would be quite small. It is concluded then, that some form of ion exchange involving suspended materials could have occurred in the tank, but that the amount of calcium-45 present in the suspended phase at any one time was too small to detect. Needless to say, more research is needed to substantiate the above suggestions.

Days 250-450

During this phase of study, research efforts were focused almost entirely on the tank. Column effluent analyses were conducted only periodically during this period to insure that the various source concentrations remained constant. Therefore, no column data has been presented for this section. Flow to the column and the column soil were left undisturbed. The objectives during this period were to determine if the model could continue to be maintained and to study the responses of the tank to alteration of various factors.

Routine tank dissolved oxygen (DO) analyses were begun on day 250. In order to avoid any possible diurnal fluctuations, all determinations were done at approximately 8:00 a.m. As shown in Figure 23, the DO concentration from days 250-277 remained at about 9.4 mg/l. Saturation at 21°C is about 9.0 mg/l. Thus the continual tank aeration kept the DO concentration slightly above saturation. Tank aeration was stopped from days 277-333. As can be seen, the DO dropped to a minimum value

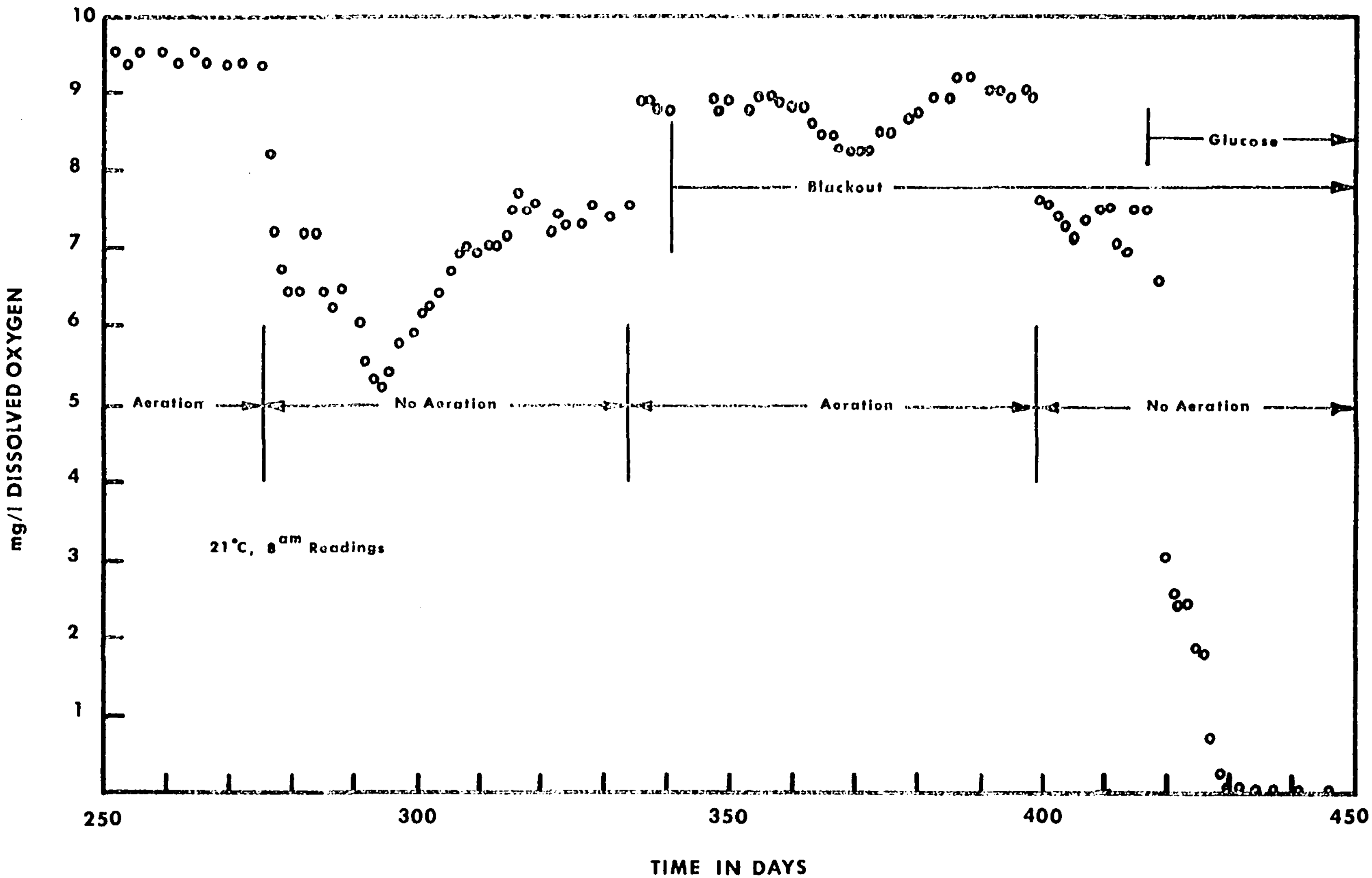

Figure 23. Effects of Aeration-No Aeration, Blackout, and Glucose Addition on Tank Dissolved Oxygen.

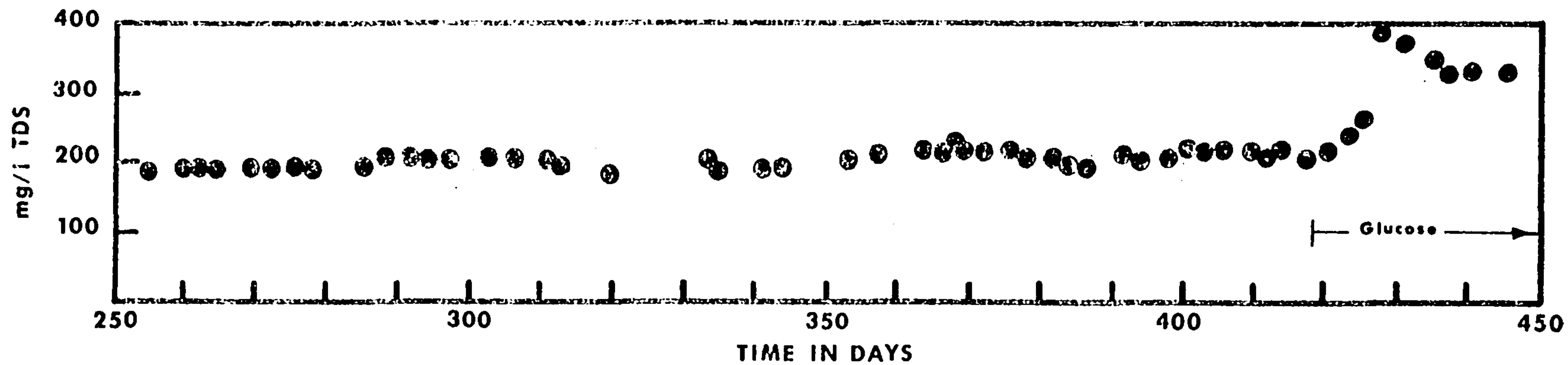

Figure 24. Effect of Glucose Addition on Tank Total Dissolved Solids.

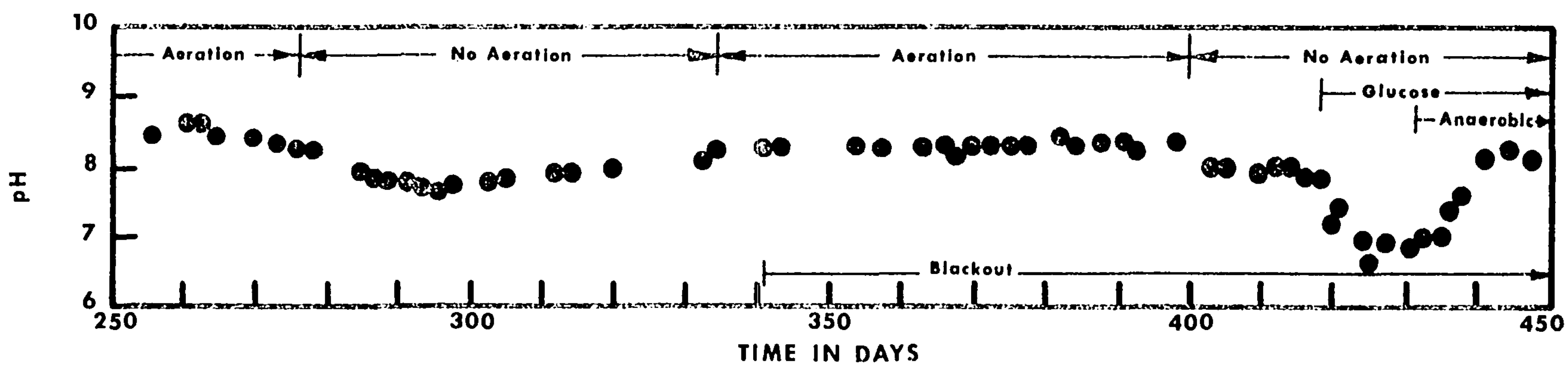

Figure 25. Effects of Aeration-No Aeration and Glucose Addition on Tank pH.

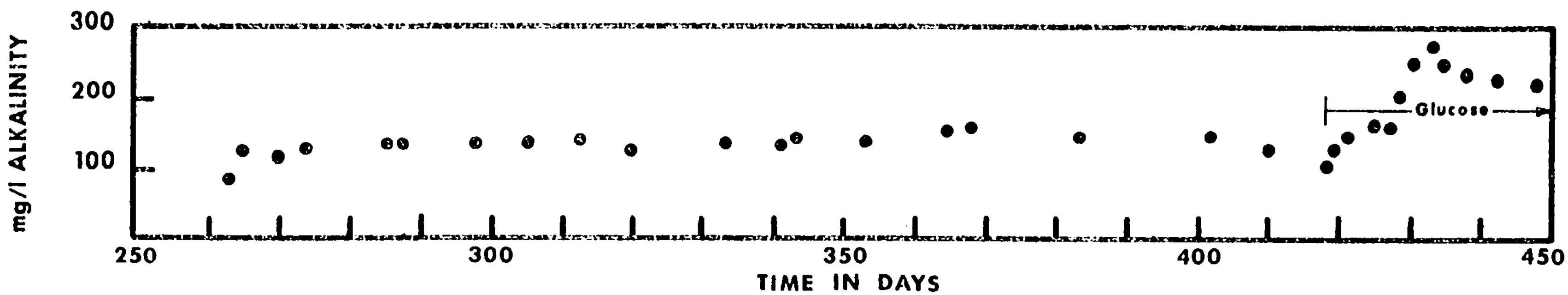

Figure 26. Effects of Glucose Addition on Tank Alkalinity as mg/l $CaCO_3$.

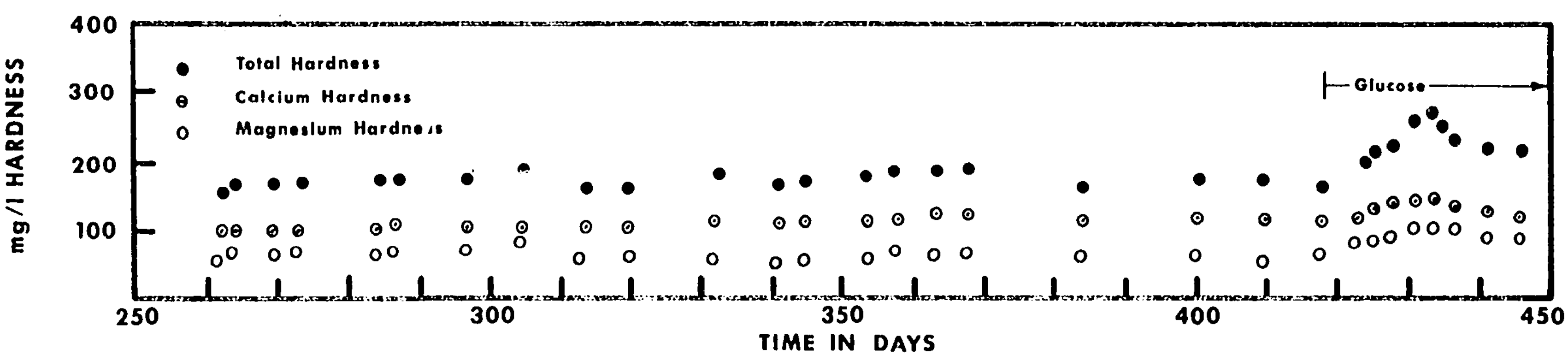

Figure 27. Effects of Glucose Addition on Tank Total, Calcium, and Magnesium Hardness as mg/l $CaCO_3$.

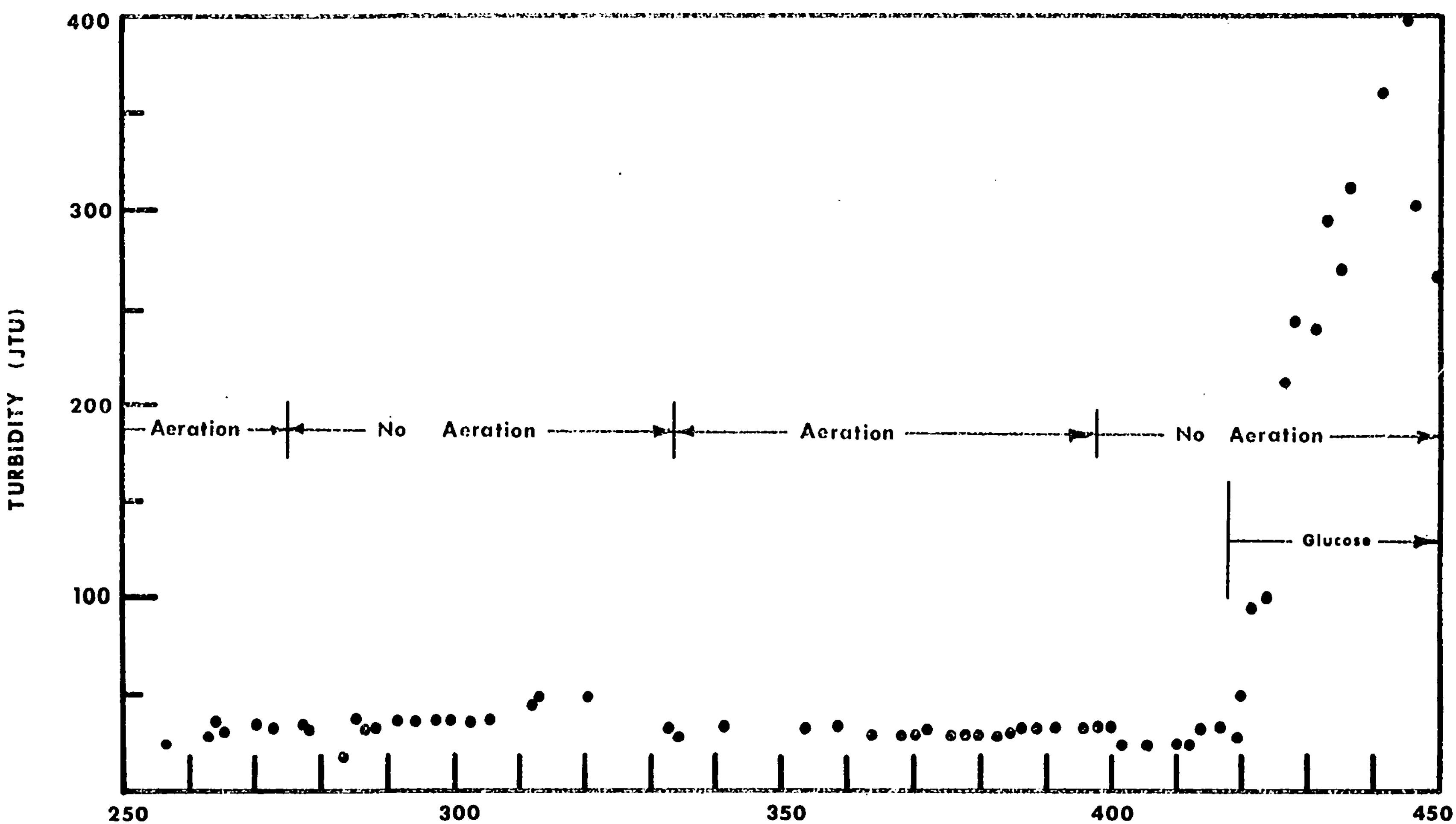

Figure 28. Effects of Aeration-No Aeration and Glucose Addition on Tank Turbidity. Results as Jackson Turbidity Units.

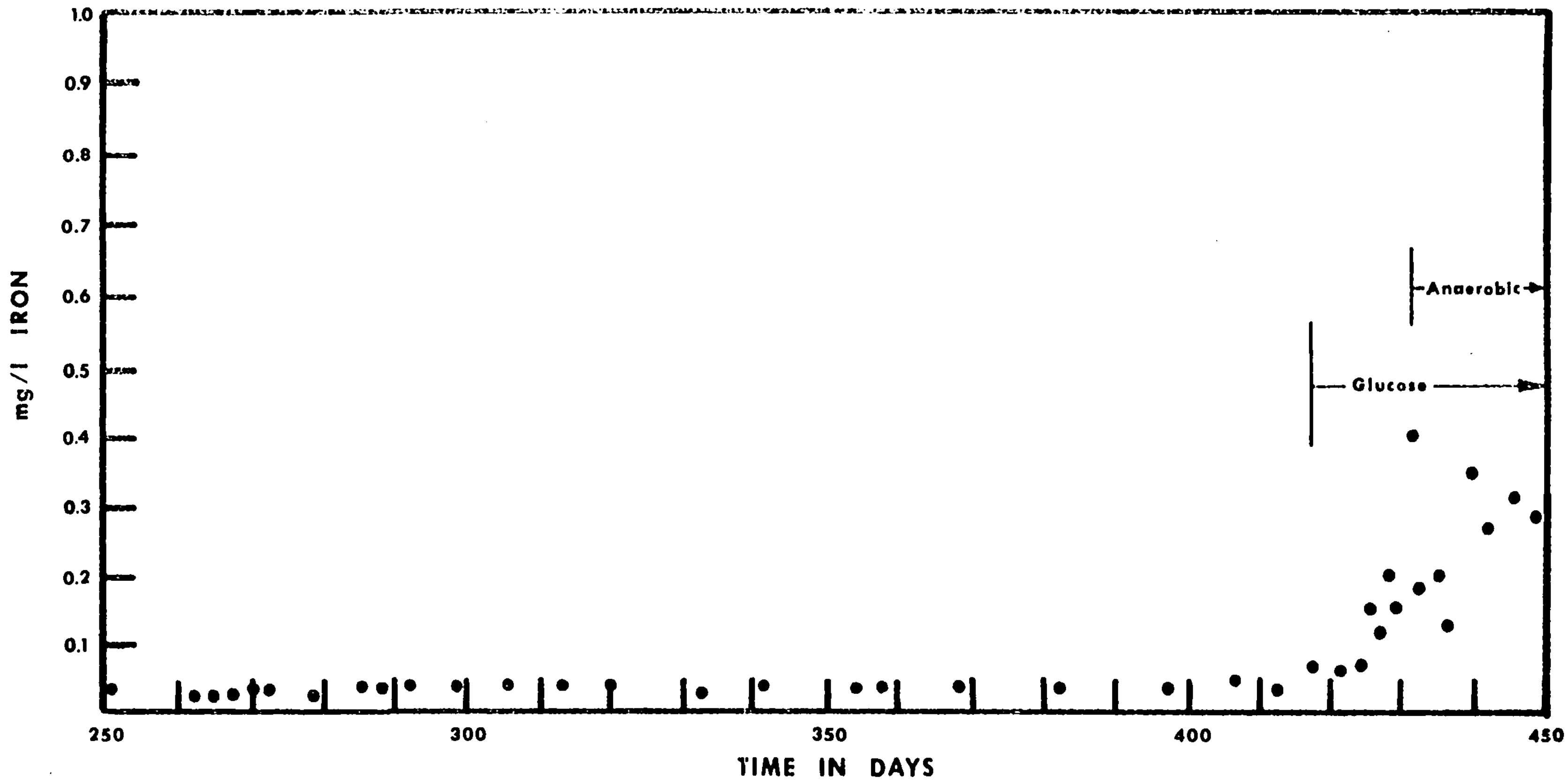

Figure 29. Effects of Glucose Addition on Tank Total Dissolved Iron.

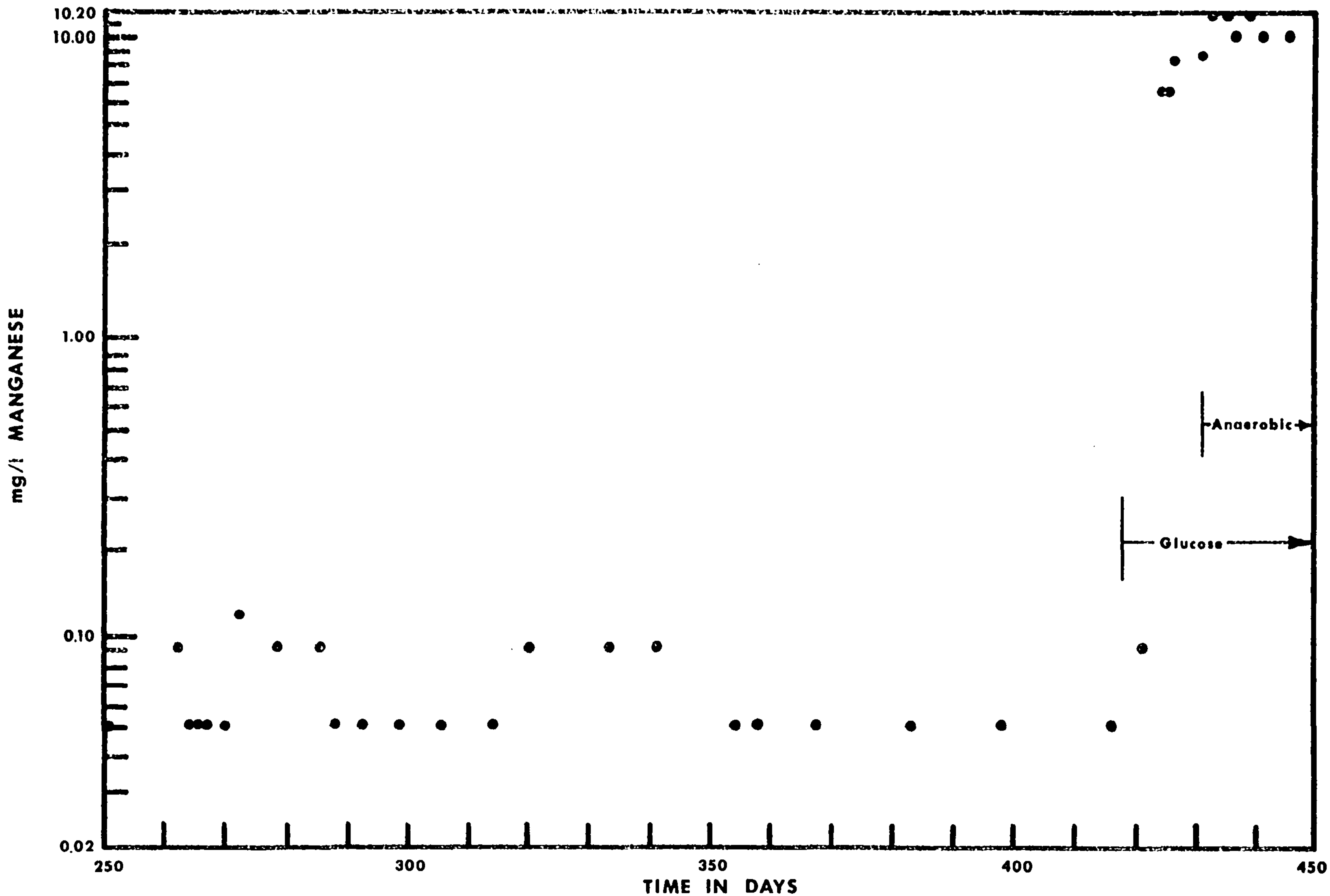

Figure 30. Effects of Glucose Addition on Tank Total Dissolved Manganese.

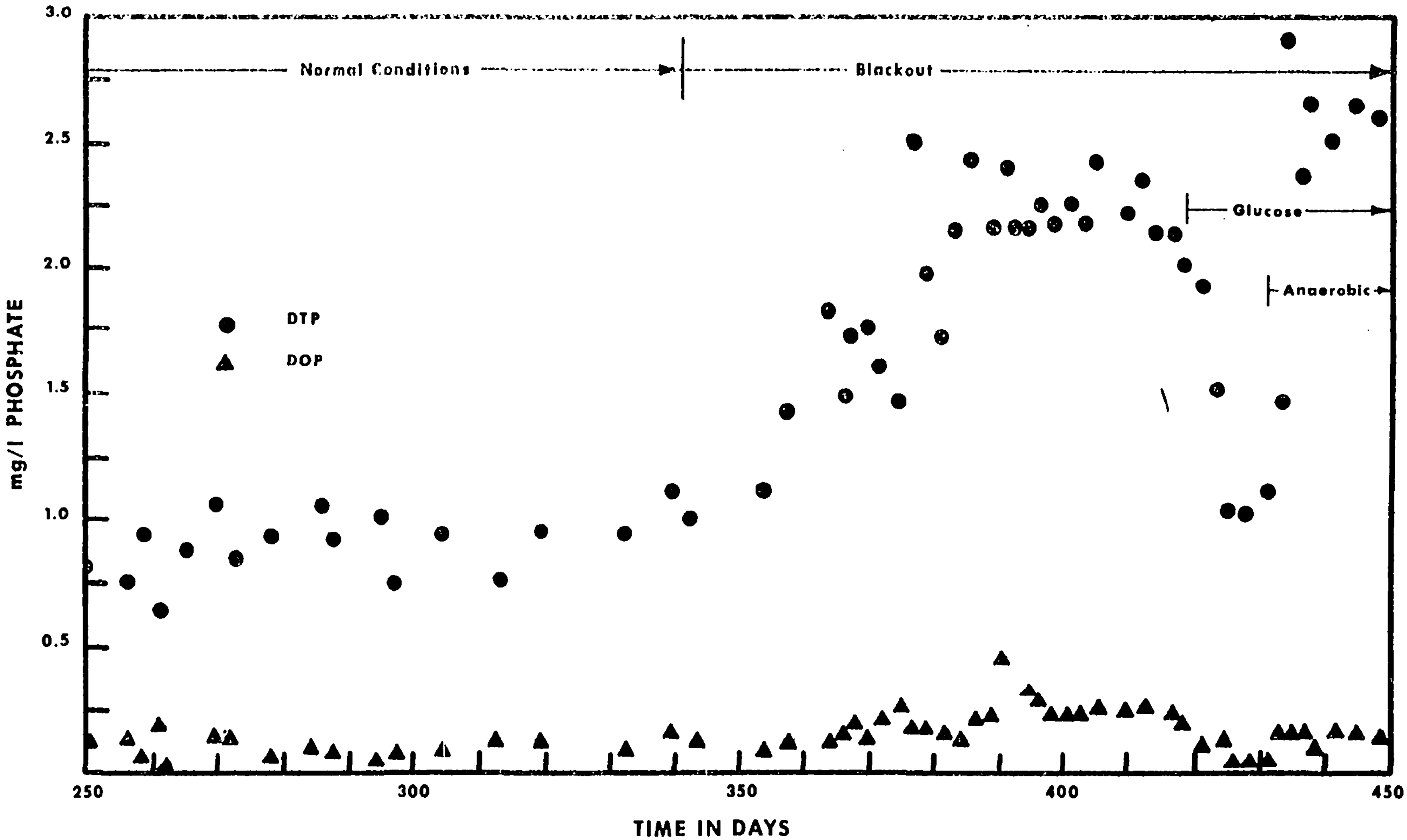

Figure 31. Effects of Blackout and Glucose Addition on Tank Dissolved Total Phosphate and Dissolved Ortho Phosphate.

of 5.2 mg/l and then increased to about 7.3 mg/l at day 314 and remained essentially constant to day 333. Aeration was started again on day 333 and, as shown, the DO increased immediately to about 9.0 mg/l. The "Plant-Gro" bulbs were shut off and the tank was enveloped in a black curtain on day 341. The response was a drop in the DO to a minimum value of 8.2 mg/l at day 370 followed by a DO increase back to about 9.0 mg/l by day 390. Aeration was stopped once again on day 399; the DO dropped immediately to a level of 7.3 mg/l and fluctuated about this point until day 415. Enough glucose solution to produce a theoretical ultimate BOD of about 300 mg/l was mixed into the tank on day 418. As shown, the DO decreased rapidly to day 420 and then, less rapidly to day 430, at which time no measurable DO was left.

Tank TDS results for days 250-450 are shown in Figure 24. As shown, the TDS level following day 250 remained at about 200 mg/l through day 420. Thus tank aeration, concentration of DO, and blackout conditions had no effect on TDS. There was only a slight increase in TDS immediately after the glucose addition on day 418. However, after day 420, TDS increased to a maximum value of 390 mg/l on day 428 and then dropped to a level of about 350 mg/l.

As shown in Figure 25, pH values remained at about 8.5 from days 250-399, with the exception of a decrease to 7.8 during the stopped-aeration period (days 277-333). After tank aeration was stopped on day 399, and under blackout conditions, tank pH dropped to a level of about 7.9 through day 418. After the glucose addition on day 418, tank pH decreased to a minimum value of 6.4 on day 426 and then increased to a value of 8.1 by day 447.

Figure 26 indicates that tank alkalinity values remained at about 130 mg/l through day 410. After the glucose addition on day 418, the alkalinity decreased to approximately 100 mg/l and then increased to a maximum value of 280 mg/l on day 431. The alkalinity then dropped to a value of approximately 220 mg/l. No phenolphthalein alkalinity was detectable during this period.

Tank total hardness, calcium hardness, and magnesium hardness results from days 250-450 are shown in Figure 27. These parameters behaved similar to TDS and alkalinity from days 250-410 in that they remained fairly constant. Total hardness remained at about 180 mg/l, calcium hardness at 110 mg/l, and magnesium hardness at about 70 mg/l. After the glucose was added, total hardness increased to a maximum of 260 mg/l, calcium hardness increased to 150 mg/l, and magnesium hardness increased to 110 mg/l. Total hardness then decreased to 200 mg/l, calcium hardness decreased to 110 mg/l, and magnesium hardness to 90 mg/l. It is interesting that at this point the calcium hardness/magnesium hardness dropped to a value of 1.2--close to the same ratio found in Thunderbird.

As shown in Figure 28, tank turbidity remained at about 35-40 JTU through day 420. After the glucose addition on day 418, tank turbidity increased to a maximum of 400 JTU on day 444. The turbidity then began to decrease and had dropped to a value of 260 JTU by the end of the study (day 450). As can be seen, tank aeration appeared to have no effect on the tank turbidity readings.

The results of tank iron analyses from days 250-450 are shown in Figure 29. As shown, the iron concentration remained fairly constant from day 250 to day 420, ranging from 0.03 mg/l to 0.05 mg/l. After the glucose addition and subsequent decrease in DO, the iron increased to approximately 0.3 mg/l by day 449. The minimum detectability of the iron test used was 0.02 mg/l.

Tank manganese concentrations for days 250-450 are shown in Figure 30. As shown, the manganese concentration fluctuated between 0.05 mg/l and 0.10 mg/l from days 250-418. Manganese concentrations of less than 0.05 mg/l could have occurred, as the minimal amount that could be detected was 0.05 mg/l. After the glucose addition and the subsequent anaerobic conditions that followed, there was a huge increase in the manganese concentration--to 10.2 mg/l by day 430.

Figure 31 shows the tank dissolved total phosphate (DTP) and dissolved ortho-phosphate (DOP) results from days 250-450. The DTP concentration averaged about 0.85 mg/l from day 250 to day 340. After the tank was subjected to blackout conditions on day 341, the DTP concentration increased to an average concentration of about 2.3 mg/l on day 380 and remained at this level through day 418. After the glucose addition, the DTP decreased to a minimum value of 1.0 mg/l on day 425. The DTP then increased to a level of approximately 2.6 mg/l by day 435. As shown, the DOP concentration remained fairly constant from days 250-360 at about 0.15 mg/l. The minimal concentration detectable with the DOP test was 0.01 mg/l. The DOP increased from the value at day 360 to 0.25 mg/l from days 360-370. The DOP reached a maximum of 0.50 mg/l on day 390 and then decreased back to 0.26 mg/l by day 400, remaining at this value through day 418. Following the glucose addition, the DOP decreased to 0.01 mg/l by day 425 and then increased to 0.18 mg/l by day 433 through day 449.

Results during this period indicated that steady-state conditions could be maintained over relatively long periods of time. With the exceptions of DTP and DOP, all parameters remained at the levels established about day 200 through about day 410. It is felt that if the tank light and aeration had not been interrupted, and if glucose had not been added, that such steady-state conditions could have been maintained over even longer periods of time. One of the main criticisms of simulated natural systems has been that they could not be utilized for long-term studies. That the experimental design used in this study made it possible to maintain steady-state conditions for over 200 days is considered one of the most important consequences of this research.

As mentioned earlier, one of the objectives during this period was to study the effect of external environmental factors (controls) on the model. Tank aeration (simulated wind) and artificial light (simulated sunlight) were the controls chosen for this study. No attempt was made in this study to determine the effects of varying degrees of light intensity or wind speed on the tank. Instead, effects on the tank due to aeration and no aeration with light and aeration and no aeration without light were investigated.

Glucose was added to the tank for several reasons. First, it was hoped that the overall response of the tank to a large increase in biodegradable material could be determined. Secondly, it was felt that the amount of glucose added would sufficiently increase the oxygen demand to a point where anaerobic conditions would result. Thirdly, it was hoped that in the absence of light, wind, mixing, and DO that the conditions in the tank would be similar to conditions known to exist in the hypolemnetic regions of stratified lakes and impoundments. Finally, it was hoped that if such conditions did form, the interaction of the sediments that accumulated in the tank with the overlying water could be studied.

Comparison of Figures 24, 26, and 27 reveals that once again TDS, alkalinity, and the various forms of hardness increased similarly. The presence or absence of light and aeration apparently had no effect on these parameters. However, these parameters all showed a substantial increase following the glucose addition. It is suggested that these increases were due, at least indirectly, to a large bacterial population that built up after addition of glucose. Visual (microscopic) examination of the tank waters indicated that the large increase in turbidity shown in Figure 28 was not due to suspended clay particles, but bacterial cells. Millipore plate counts revealed that the total bacteria had increased to over 2,500 colonies/ml by day 427. It is suggested that bacterial oxidation of the glucose to CO_2 and H_2O caused the low pH readings between days 420 and 435, shown in Figure 25. Furthermore, any anaerobic breakdown of the glucose could have resulted in the production of organic acids such as lactic acid; this also could have accounted for the low pH. It is further suggested that the increase in CO_2 due to oxidation of glucose—termed "aggressive" CO_2 by Hutchinson (28)—was responsible for the increases found in hardness, alkalinity, and consequently, TDS. Mechanisms accounting for the pH depression, increased alkalinity, and increased hardness forms are shown below.

$$CO_2 + H_2O \rightleftharpoons H_2CO_3 \rightleftharpoons HCO_3^- + H^+ \quad \text{(a)}$$

$$CO_3^{--} + H^+ \rightleftharpoons HCO_3^- \quad \text{(b)}$$

$$OH^- + H^+ \rightleftharpoons H_2O \quad \text{(c)}$$

As shown in equation (a), an increase in CO_2 would have the effect of shifting the equilibrium to the right, forming carbonic acid. At the pH levels encountered, the carbonic acid would dissociate forming bicarbonate and hydrogen ions, thus increasing the alkalinity and decreasing the pH. As shown in equation (b), an increased hydrogen-ion concentration would shift the equilibrium to the right, forming more bicarbonate. Thus any carbonates in the tank soil and/or sediment, such as $CaCO_3$, would be converted to the more soluble bicarbonate forms, with the release of the associated cations such as calcium. Calculations revealed that immediately prior to the glucose addition, the solubility product of $CaCO_3$ had been reached. Finally, as shown in equation (c), an increase in hydrogen-ion concentration would cause a shift in equilibrium to the right. Therefore, any metals in the hydroxide form in the tank soil and/or sediment, such as $Mg(OH)_2$, would be released to solution. However, this would not be likely at the pH ranges encountered. It is possible that the decrease in pH could cause some increase in calcium and magnesium due to solubilization of materials such as magnesium calcites; such materials are known to be precipitated by many organisms. Other materials not specifically tested for could have also gone into solution because of the shifts discussed in equations (a), (b), and (c); the cumulative increase in all such materials could have accounted for the TDS increase during this period.

The increases in iron and manganese (Figures 29 and 30) after the glucose addition can also be attributed, indirectly, to the increase in bacteria. The sharp drop in DO (Figure 23) after day 418 was undoubtedly due to oxidation of the glucose by bacteria. Equation (d) shows the mechanism involved.

$$C_6H_{12}O_6 + 6O_2 \rightarrow 6CO_2 + 6H_2O \qquad (d)$$

It is concluded that the decrease in DO after day 418 caused a lowering of the redox potential. Therefore, as the redox potential dropped, the relatively insoluble iron-III and manganese-IV found in the sediment would be reduced to the more soluble iron-II and manganese-II forms. By comparing Figures 29 and 30, it can be seen that the manganese had begun to increase by day 422, while the iron did not begin to increase until day 427. This would be expected if the redox potential was the controlling factor involved, as the potential for the manganese cycle is higher than that of the ferrous-ferric system. As can be seen, the DO was dropping from day 422 to 427, the concentration being 2.3 mg/l and 0.8 mg/l respectively.

The changes described earlier in DTP and DOP (Figure 31) also have been interpreted as being largely biologically mediated. According to studies done by Foree, et al. (29), algal cells subjected to blackout conditions die and release soluble organic phosphate to solution. Foree also found that if these cells had previously undergone "luxury uptake" of phosphate—defined as the incorporation of phosphate into the cells in quantities above that level required for optimum growth rate--that this excessive cellular phosphate would be released to solution as organic phosphate, independent of bacterial decomposition. The cellular phosphate needed for growth was found to be released much slower than the "luxury phosphate" and was dependent upon bacterial decomposition.

The increase in DTP following blackout conditions was attributed to the dying algal biomass. Visual examination of the tank waters, and the absence of any appreciable DO or pH diurnal fluctuations indicated that there had not been a large algal biomass. Furthermore, because of the rather large increase in DTP after day 341, it was concluded that the algal cells must have taken up considerable phosphate in excess of their growth needs. It is quite possible, then, that much of the DTP released could have been in the soluble organic form. If the DTP was in the soluble organic state, then bacteria could have begun to utilize it as a food source, releasing DOP. This could then account for the increase in DOP that began about 20 days after blackout. The large decrease in DTP and DOP following the glucose addition was attributed to uptake by the large bacterial population that developed. According to Mackenthun (8), bacteria can also be involved in excessive phosphate uptake, degrading the DTP and incorporating the DOP.

As described earlier, iron and manganese increased after the tank became anaerobic, and it was suggested that these materials were released from the sediment due to a low redox potential. Stumm (30) states that many phosphates found in lake sediments were formed by direct precipitation of the inorganic phosphate with calcium, aluminum, and iron. Stumm further states that redox potentials influence the affinity of sediments for phosphates. As an example, at high redox potentials iron-III reacts with inorganic phosphate to form the insoluble iron-III phosphate; at low redox potentials iron-III is reduced to iron-II, releasing iron and phosphate to solution. However, after the tank became anaerobic, there was no increase in inorganic phosphate (DOP). Yet as described earlier, there was a large increase in DTP shortly after anaerobic conditions developed. It was therefore concluded that the maximum concentration of DTP following blackout (2.3 mg/l) represented that amount that had been incorporated into the algal biomass; the maximum DTP concentration following the onset of anaerobic conditions (2.6 mg/l) represented that amount that had been incorporated into algae plus all other forms of aerobic life such as bacteria and protozoa.

Comparison of Lake and Model Systems in Terms of Carbon and Nitrogen

Carbon and nitrogen determinations were done on a periodic basis during the 450-day study period. It was felt that such determinations would allow additional means of comparing various components of the model with the Thunderbird system. The techniques used allowed comparison of the percent carbon and nitrogen content of the watershed topsoil, lake sediment, tank sediment, and total and dissolved residue from the lake and tank waters.

Sediments were collected from Lake Thunderbird with a six-inch Ekman dredge. The sediments were collected from a point approximately 200 feet northwest from the center of the Thunderbird dam, in about

40 feet of water (see Figure 6). Depth-sounding had shown this to be the deepest part of the lake; it was felt that there would be more accumulated sediments--including organic detritus--in this location because the deeper waters would allow more protection from wind-induced circulation. The water samples were collected from the same location, but just under the surface. Tank sediments were collected from the sediment traps mentioned earlier and tank waters were collected from just underneath the surface. The soil sample was from the same location as used in the column.

All samples were prepared for analysis in the same manner. The soil and sediment samples (approximately 50 grams dry weight) were air-dryed, pulverized, and placed in a drying oven at 103°C for 1 hour. Total residue was obtained from the lake and tank liquid samples by evaporating 100-ml samples to dryness. Dissolved residues were obtained by evaporating 100-ml portions of centrifugate (15,000 rpm for 20 minutes) to dryness. Total and dissolved residues were then placed in the drying oven at 103°C for 1 hour and brought to constant weight. All samples were then ground to a fine powder, mixed, and portions withdrawn for carbon and nitrogen analysis.

Carbon and nitrogen values were obtained by using a model 185 carbon, hydrogen, and nitrogen analyzer manufactured by F and M (Hewlett-Packard) Scientific Corporation. Basically, the instrument uses oxidation and reduction furnaces to convert all nitrogen to N_2 and all carbon to CO_2. A gas chromatographic system then separates the N_2 from the CO_2. The carrier gas then sweeps these materials individually to a thermal conductivity detector. The detector develops an electrical signal proportional to the concentration of the material in the carrier gas; this signal is then delivered to a potentiometric recorder, producing the chromatogram. The instrument was calibrated against a known sample according to the manufacturer's directions.

Results showing the percent carbon content of various phases of the model and Thunderbird are shown in Figure 32. The values shown are averages determined from 9 replicate analyses. As can be seen, there was very little difference in carbon content between watershed soil, lake sediment collected in the winter, and tank sediment collected on day 237. Total solids in the column effluent were determined to be 25 percent carbon. The lake (winter sample) and tank (day 237) total solids were found to be 55 and 63 percent carbon, respectively.

The percent nitrogen content of the same samples is shown in Figure 33. Once again, the 9-replicate averages indicated very little difference between watershed soil, lake sediment collected during the winter, and tank sediment collected on day 237. The total solids were determined to be 2.3 percent nitrogen in the column effluent, 0.6 percent in the lake, and 3.1 percent in the tank.

Figures 34 and 35 show the percent carbon and nitrogen found in lake sediments. Comparisons are also shown between lake total and lake dissolved residues, and between tank total and dissolved residues. As can be seen, there was very little of the carbon and nitrogen in the particulate phase. Both the total and dissolved lake residues contained about 55 percent carbon, whereas the tank total and dissolved residues were found to contain roughly 62 percent carbon. The percent nitrogen found in the lake total and dissolved residues was found to be

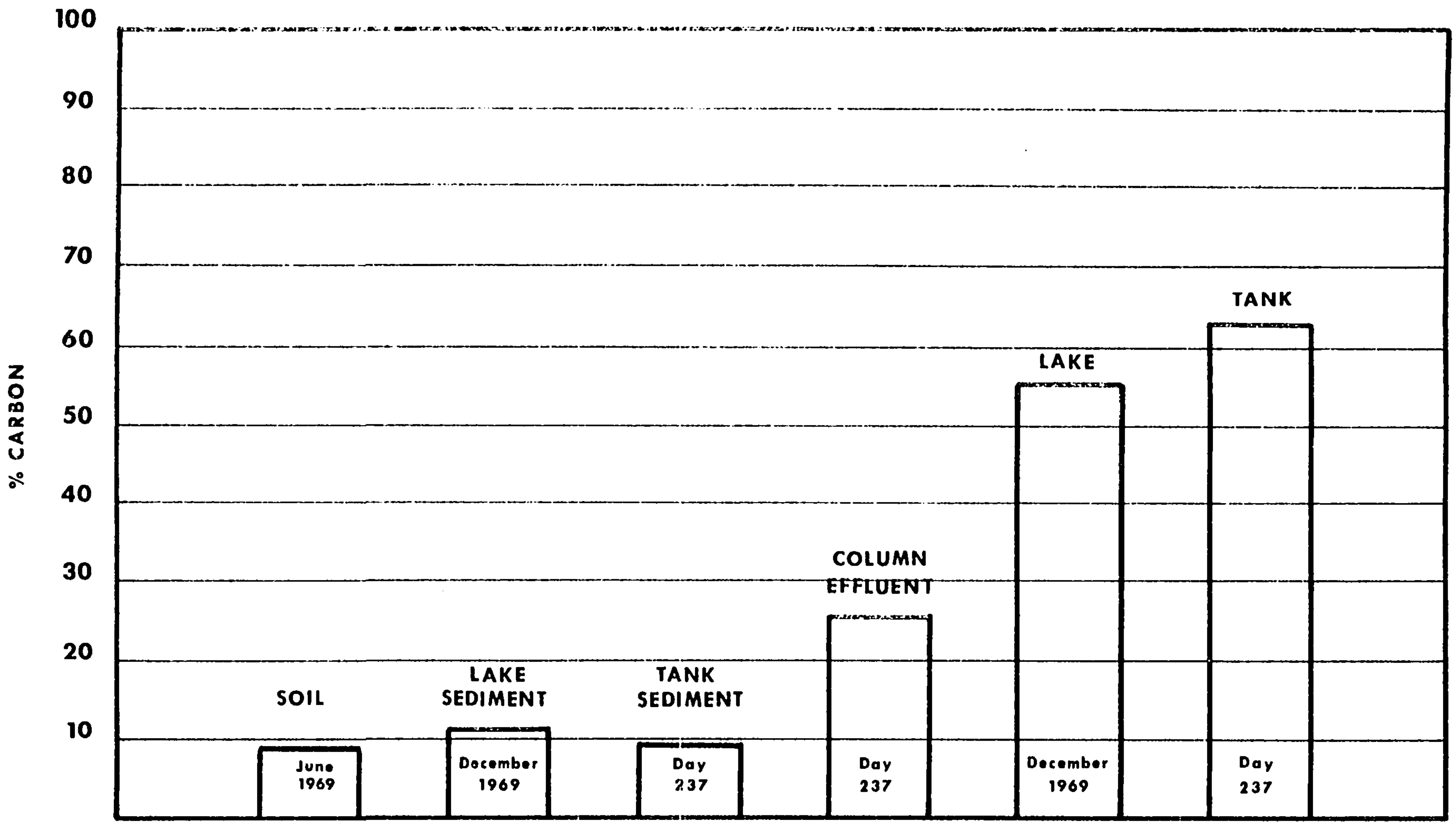

Figure 32. Comparison of Carbon Content in Different Phases of Lake and Model.

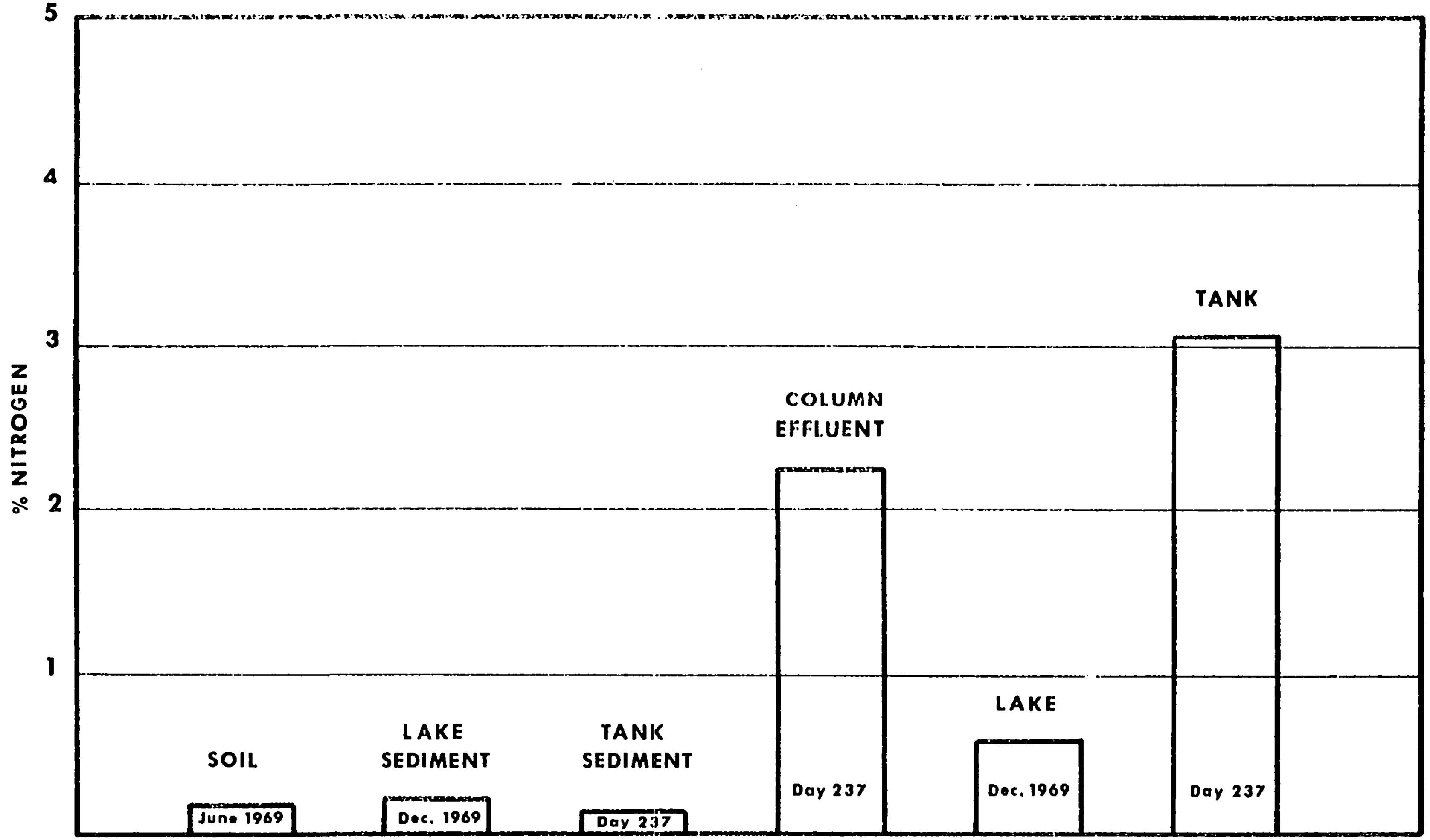

Figure 33. Comparison of Nitrogen Content in Different Phases of Lake and Model.

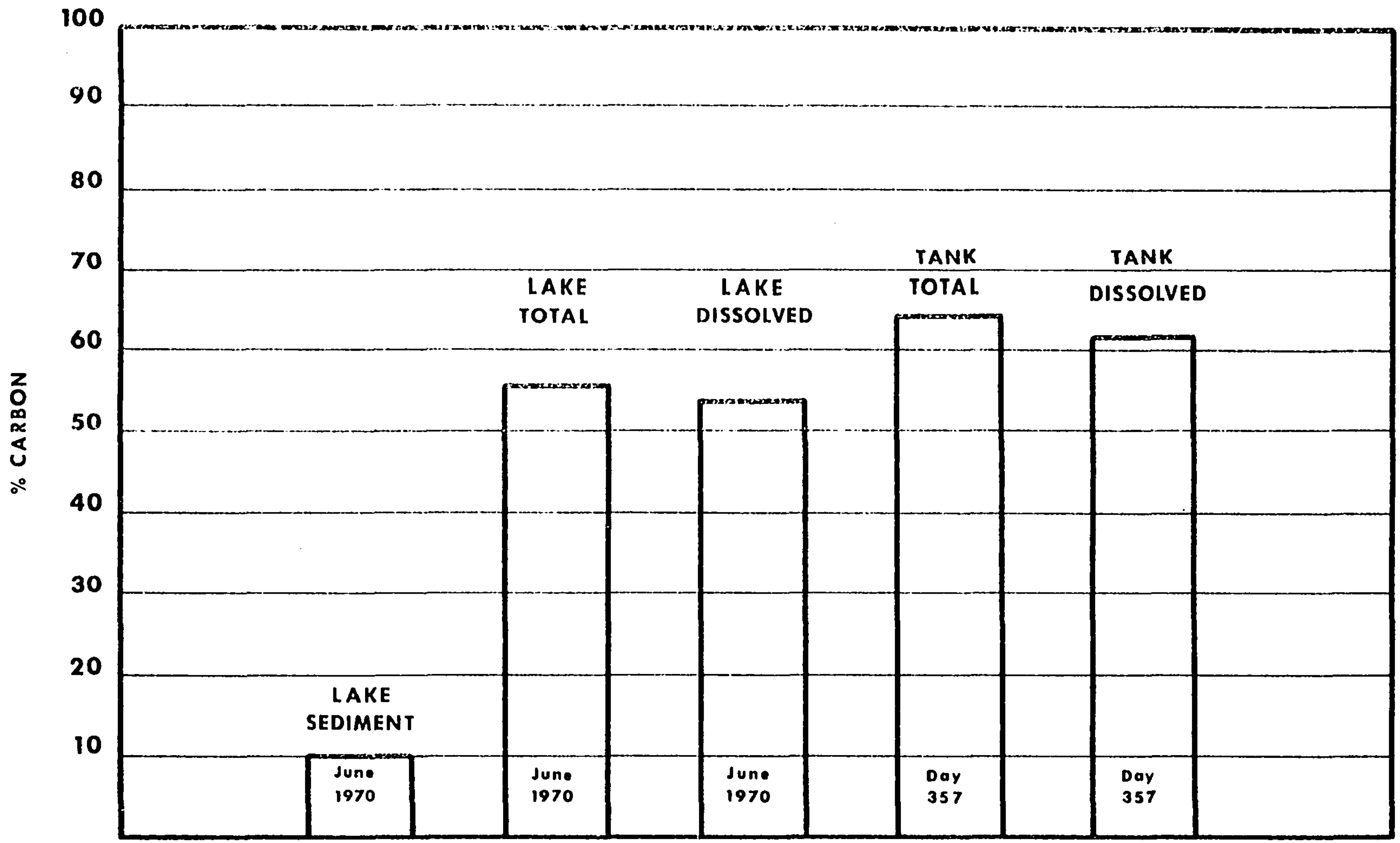

Figure 34. Comparison of Carbon Content in Different Phases of Lake and Model.

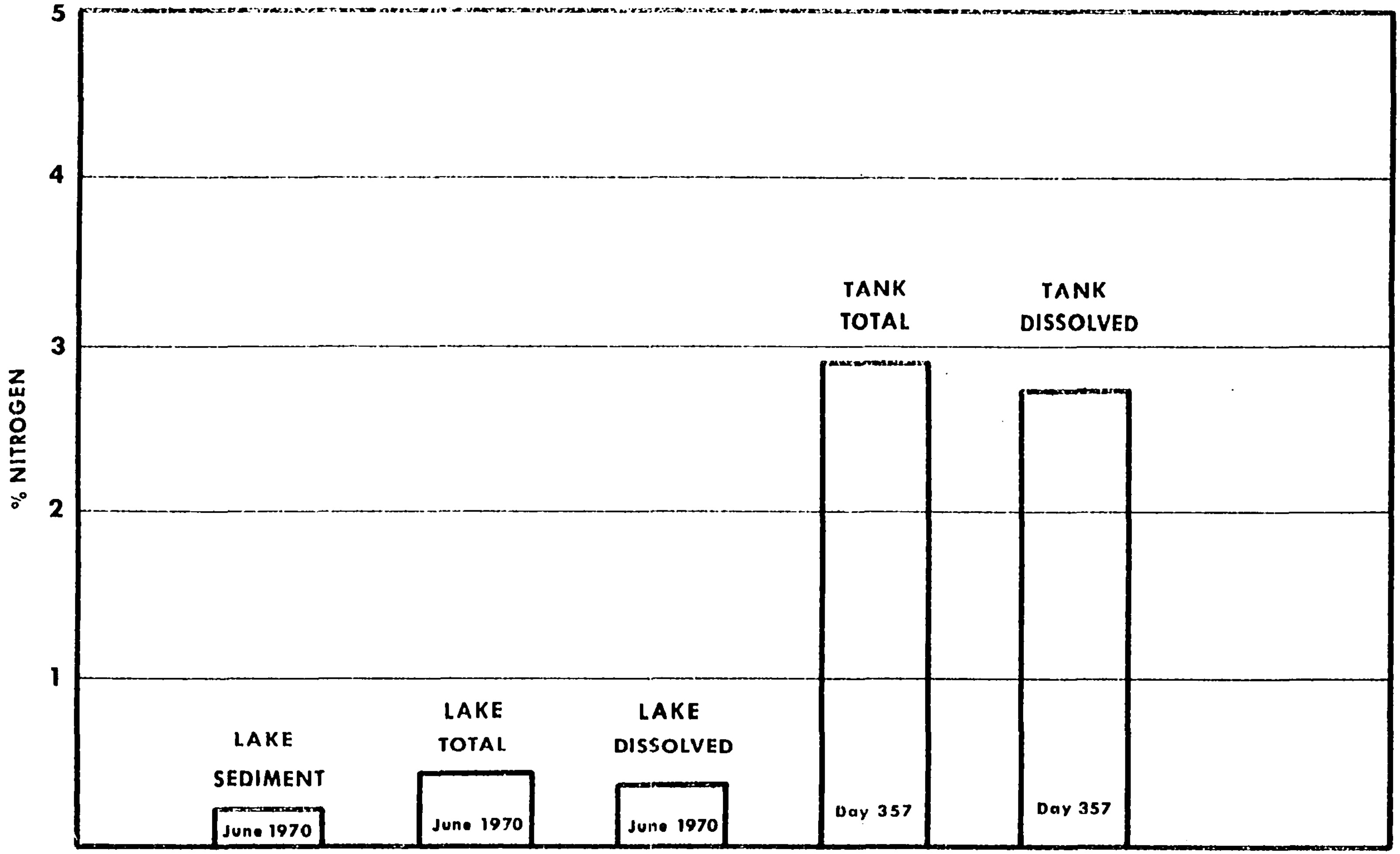

Figure 35. Comparison of Nitrogen Content in Different Phases of Lake and Model.

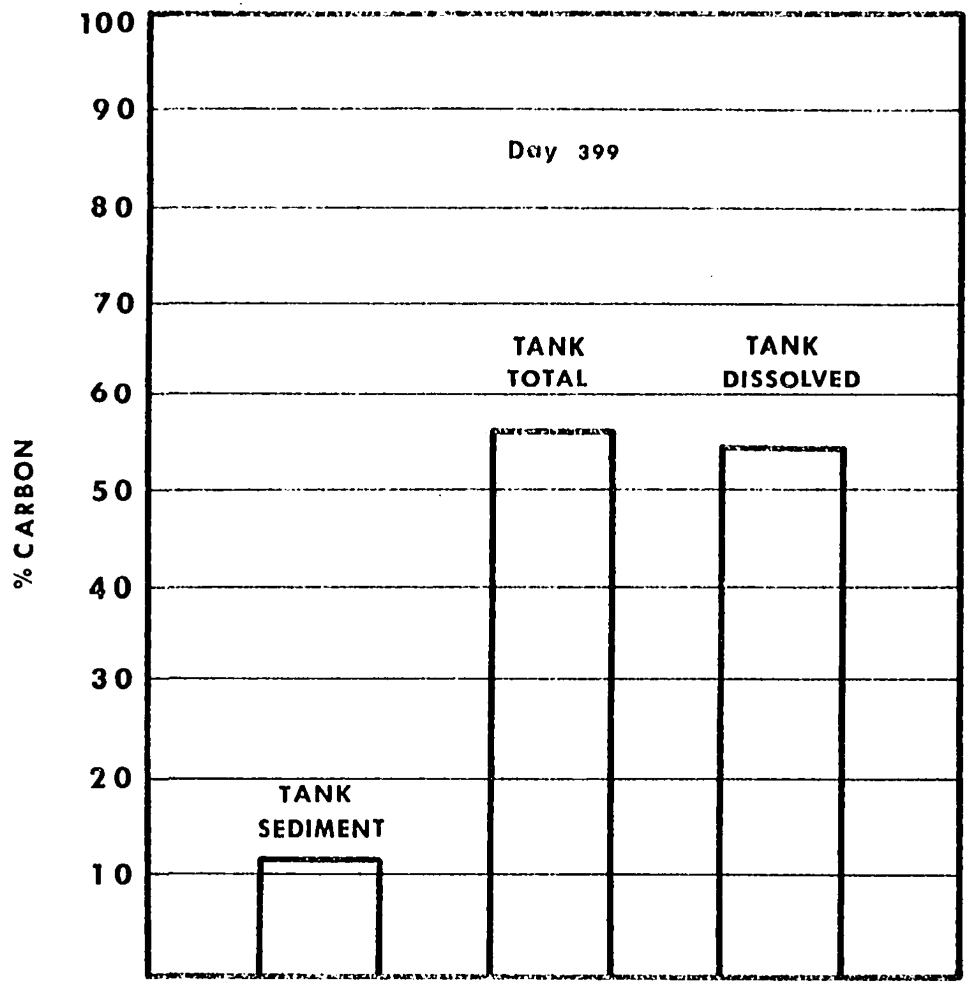

Figure 36. Comparison of Carbon Content in Different Phases of the Model.

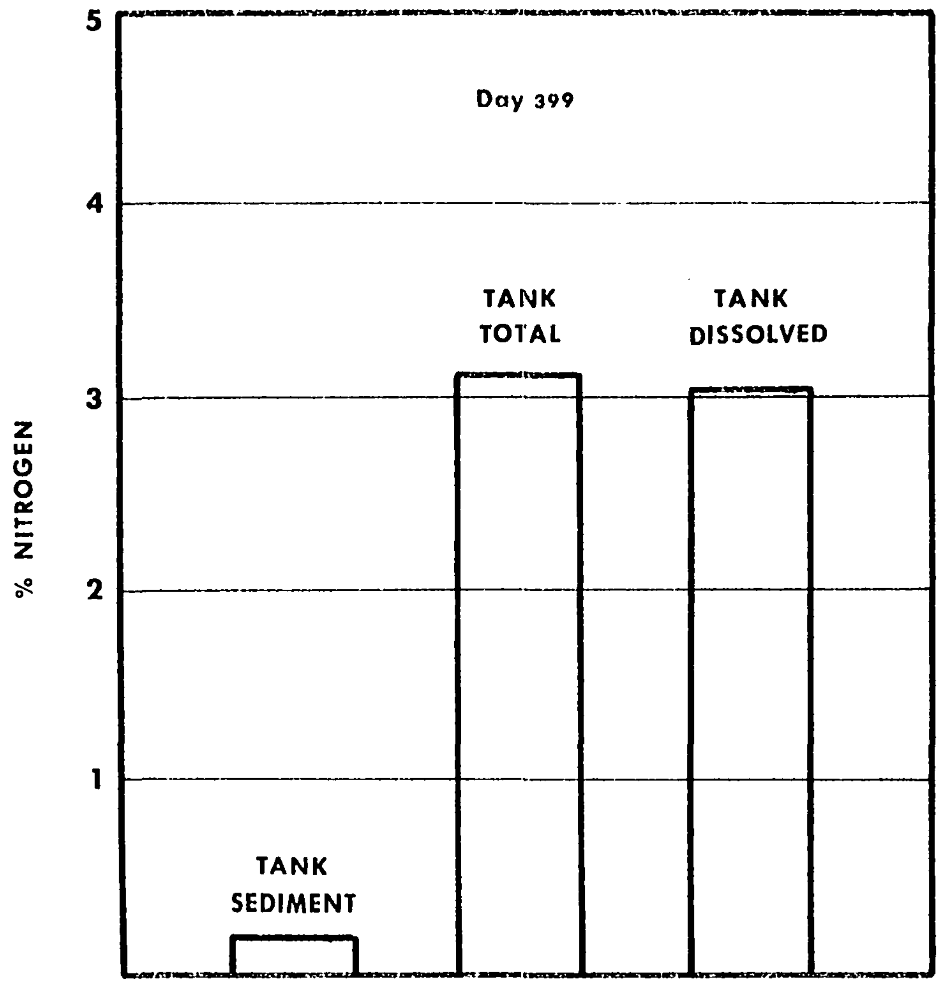

Figure 37. Comparison of Nitrogen Content in Different Phases of the Model.

about 0.5 percent; the tank total and dissolved residues were roughly 2.8 percent nitrogen. The lake samples were collected in June 1970, and the tank samples were collected on day 357.

Figures 36 and 37 show the percentage of carbon and nitrogen found in various phases of the tank on day 399. As before, the 9-replicate averages revealed very little difference in carbon and nitrogen content between the tank total residue and the tank dissolved residue. On this date, the tank residue was found to be roughly 55 percent carbon and 3-3.1 percent nitrogen.

Several conclusions have been drawn from the preceding results. That there was very little difference in carbon and nitrogen content in the soil, lake sediment, and tank sediment, and because the carbon and nitrogen percentages were relatively low, it was concluded that there had been no appreciable organic accumulation in either the lake or tank sediments. Furthermore, comparison of lake sediment values in Figures 32 and 33 with the values shown in Figures 34 and 35 indicated that there was no appreciable change in carbon and nitrogen content from December 1969 to June 1970. This apparent lack of any organic accumulation in the lake sediments has been attributed to the following:

1. Lake Thunderbird is a young impoundment.
2. The soils in the drainage area are relatively low in organic materials.
3. The lake is not known to have a large biomass.
4. The lake is almost continually mixed by prevailing winds, thus allowing decomposition of any suspended organics before they accumulate in the sediments.

The same reasons have been used to explain the similar results found in the tank sediment on day 237.

As was shown in Figures 34-37, there was very little difference found in carbon and nitrogen content between lake total and dissolved samples and between tank total and dissolved samples. Accordingly, most of the total solids in the lake and tank were found to be in the dissolved state. Therefore, assuming that alkalinity values represented most of the inorganic carbon present, it was possible to calculate the organic carbon concentration in the tank and lake. It is realized that the manner in which samples were prepared for analysis would have caused the loss of any carbon present as CO_2. However, such loss was assumed small due to the pH levels encountered: the pH in both systems was usually 8 or greater at the time of analysis. Nomographic calculation of the free CO_2 concentration, as prescribed in "Standard Methods" (18), indicated that under the conditions of temperature, pH, TDS, and alkalinity found in the lake and tank, 0.5 mg/l C was the maximum concentration present. All carbon values obtained were interpreted, therefore, as total carbon. The equation used to convert alkalinity cencentrations to inorganic carbon concentrations is shown below.

$$\frac{(\text{MW C})}{(\text{MW CaCO}_3)} (\text{mg/l alk. as CaCO}_3) = \text{mg/l inorganic carbon}$$

Total carbon percentages were converted to concentrations by multiplying percent carbon from total dissolved residue times total dissolved solids (TDS).

$$(\%\text{C})\ (\text{mg/l TDS}) = \text{mg/l total carbon}$$

Organic carbon concentrations were then estimated by subtraction.

Total Carbon - Inorganic Carbon = Organic Carbon

Calculations revealed that the tank total carbon concentrations ranged from 140 mg/l to 180 mg/l. Conversion of corresponding alkalinity concentrations to inorganic carbon concentrations revealed a range of about 15 mg/l to 25 mg/l. Organic carbon concentrations were then estimated by subtraction to range from 125 mg/l to 155 mg/l. Therefore, the total carbon in the tank was estimated to be 86 to 89 percent organic.

Similar calculations for Lake Thunderbird revealed that from December 1969 to June 1970 the total carbon ranged from 135 mg/l to 145 mg/l. Conversion of alkalinity values yielded an inorganic carbon range of 15 mg/l to 25 mg/l. The lake organic carbon concentration was then estimated to be about 120 mg/l. These calculations therefore estimated the total lake carbon to be about 83 to 89 percent organic.

The above calculations indicated that most (90%) of the carbon in the tank and lake aqueous phases was organic. Furthermore, for reasons given earlier, it is concluded that most of the carbon was in the dissolved state. This yields further evidence to the assumption made earlier that the tank DTP was of an organic nature. The low BOD_5 values obtained, accounting for only 25 percent of the measured COD, indicate that the organic material was only slightly biodegradable. Thus much of the organic material was probably a soil humus extract. The continuing supply of soil extract from the percolation column could account for the high nitrogen values, as compared to the lake, shown in Figures 33 and 35.

SUMMARY AND CONCLUSIONS

CHAPTER V

Throughout this study, the goal of determining the feasibility of a laboratory model as an additional science and engineering research tool for water quality studies was kept constantly in mind. Hopefully, model studies would give the engineer additional capabilities for pre-impoundment site selection, design of outlet structures, effects of anticipated pollutants, rate of eutrophication, and in general, the environmental impact of a proposed impoundment system. Model studies could also provide the scientific community additional fundamental research capabilities in the areas of nutrient cycling, sediment-nutrient exchange, bioassay, reaction rate studies, and many other areas of concern. However, the greatest value of the physical model studies could easily be the conceptual mathematical models that could develop parallel to them. Undoubtedly, mathematical models would be the end result of collaboration between the engineering and scientific communities, and would provide the predictive capabilities so desperately needed in the area of water resources.

The results obtained from this initial laboratory and field study were by necessity far more qualitative than quantitative. Consequently, no mathematical treatment of the data was offered. However, it was possible, at least for the parameters involved, to make several comments about the Lake Thunderbird system based on the results of the model studies.

That the chemical water quality of the model, under steady state conditions, was remarkably similar to that of Thunderbird was highly encouraging. This indicated that the percolation column was a source of the same type constituents as were available to Thunderbird via watershed-rainfall-runoff. However, much more significiance was attributed to the fact that the model responded to manipulation of various environmental control factors in a more or less predictable manner. That is, the model behaved similarly to known impoundment behavior. That the initial "impounded" tank water changed with time from a quality identical to the column effluent to a quality similar to Thunderbird indicated one type of response: the tank (impoundment) did perform as a reactor on the available sources. That long-term steady state conditions could be maintained by keeping the various environmental control factors constant was another type of response. Change in one or more control factors produced other types of responses. Finally, all such responses indicated that the model was a dynamic "living" system.

Results from percolation column studies indicated the presence of a slug effect. Studies of regional precipitation patterns indicated that the slug effect could be a significant factor in the Thunderbird system.

Other than a few bacteria counts, biological studies as such were not done. However, observations revealed various groups of plankton were present throughout most of the study. That the introduced species of Cladocera and Copepoda survived for the duration of the study suggests the culturing properties of the soil-water media. Several of the response studies done from days 250-450 suggested the influence of the biota on the aquatic media and vice versa.

Results obtained from studies of simulated hypolimnetic conditions indicated that if such conditions did develop in Thunderbird, domestic supply problems could arise due to increased concentrations of iron, manganese, and hardness.

Several inherent weaknesses in the experimental design used as the basis for this research became apparent during the course of study. The strong dependence of the model system on the percolation column, as well as some of the operational problems associated with the column, has already been pointed out. Probably the greatest weakness associated with the use of the percolation column was that in order to have a sufficient volume of water delivered to the tank, the soil in the column stayed almost completely saturated with water. In other words, based on an average annual precipitation of 33 inches per year, the amount of water pumped to the column during the 450-day study period was equivalent to approximately 25 years rainfall in the Thunderbird region. The reason for this is quite apparent in view of the fact that the ratio of the Thunderbird drainage area to the area of the reservoir is 27:1, while the ratio of the column surface area to the area of the tank was only 0.1:1. Closer approximation of this ratio in the future would allow more realistic attempts to be made at studies concerned with the effects of amount and intensity of rainfall on water quality.

It also became apparent during the course of study that several improvements were needed in the tank (impoundment) phase of the model. The glass sides of the tank resulted in a surface area to volume ratio that was highly unrealistic. One of the consequences of this was the growth of organisms (mostly blue-green algae) on the sides of the tank. Also, due to the shallow depth of the tank waters, no volume of water was representative of that found beneath the zone of light penetration in nature. It is suggested that any future research efforts in this area might include the use of topographical maps of the region of concern to "limnologically design" the receiving tank.

It was concluded from this research that a laboratory simulation of an impoundment system as a research tool is feasible. This conclusion was based on results that indicated that such a model could be established and dynamically maintained over long periods of time, at least in terms of the parameters studied. Man's understanding of the various physical, chemical, and biological processes found in lakes and impoundments is still limited. Yet it is entirely possible that man, as he has so often done in the past, by appropriate manipulation of known variables can predict and control certain natural systems without understanding the component parts and reactions found within any one system. It was to this end that this research was intended.

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