Analysis of Water Quality Problems in the VPI&SU Duck Ponds and Suggested Management Alternatives

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Thesis submitted to the Faculty of the

Virginia Polytechnic Institute and State University

in partial fulfillment of the requirements for the degree of

Master of Science

in

Environmental Science and Engineering

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November 1988

Blacksburg, Virginia

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(ABSTRACT)

Allochthonous nutrients were monitored during three storm events on one of the major tributaries entering the shallow VPI&SU Duck Ponds. Autochthonous nutrients were monitored for a period of ten months. During these storms, the stormwater runoff contributed large amounts of organic matter and fertilizer nutrients that settled in the ponds and during anoxic conditions, recycled to stimulate algal blooms. Alum was applied to one pond to reduce internal cycling of nutrients. A 25 mg/L dose of alum produced an aluminum hydroxide floc that settled to the bottom and afterwards, lowered othophosphate-phosphorus concentrations below 10 ug/L in the water column. The longevity of the one-time treatment in reducing the sediment-phosphate release rate is unknown because the monitoring program was not continued beyond July of 1988. A pond-treatment program involving copper sulfate was initiated to control algal blooms consisting mainly of the green alga, *Chlamydomonas*. Based on the complexing properties of the water, such as alkalinity and humics, a copper sulfate dose of 13.6 kg was determined to be a safe and effective dose that reduced algal densities but did not result in any visible adverse effects upon other aquatic life. Both of the pond management schemes were designed to aid managers of small urban ponds who have low operating budgets and a lack of technical equipment.

ACKNOWLEDGEMENTS

The author would like to express his appreciation to his graduate committee for their assistance in the completion of this project. A special thanks goes to Dr. Hoehn who freely gave his time, advice, and encouragement throughout the course of the study.

The author would also like to thank Dr. David Ford, , , and other personnel of the VPI&SU Physical Plant Department for the partial financial and technical support for this project and the Blacksburg Water Treatment Plant for providing, without charge, the aluminum sulfate used to treat the pond on one occasion.

Thanks goes also to my pals: , and all the others who helped with the late-night storm measurements and other Duck Pond adventures. The author would also like to express his gratitude to for the multitude of suggestions, of which the majority were unsolicitated, concerning the management of the ponds.

The author wishes to extend a special thanks to his family for their support, especially to his lovely wife, , who endured the many antics of a graduate student.

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INTRODUCTION

Small reservoirs and ponds have become attractive additions to many cities and residential areas. These small bodies of water may act as flood control systems, enhance the natural beauty of the surrounding area, provide a habitat for a variety of wildlife, and serve as a recreational site for children and adults. These multiple benefits may not always complement one another.

Ponds designed to reduce flooding and remove sediment, organic matter and nutrients are referred to as detention ponds. Engineers often utilize these ponds as best management practices to reduce nonpoint pollution from stormwater runoff. Sediment, organic matter, debris, and nutrients often accumualate quickly in urban detention ponds. These materials frequently produce massive algal blooms and odors, thus degrading the aesthetic qualities of the ponds.

Detention ponds, however, perform a vital role in reducing the pollutant loads of urban stormwater runoff to lakes, bays, and oceans. Many cities have eliminated public complaints by designing detention ponds with no public access areas. Nevertheless, many older ponds and reservoirs that were designed strictly as detention ponds have evolved into public park areas. A detention pond presents unique management problems because of its multiple use functions: an engineer views the pond as a detention facility, a child refers to the pond as a playground, and an ecologist perceives the area as a habitat for fish and wildlife.

Accumulation of excessive nutrients, particulate and dissolved organic matter, and inorganic matter perpetuate a natural process known as eutrophication. Urban areas frequently increase the rate of natural eutrophication due to increases in the impervious land surfaces, lawn fertilizers, faulty sewer systems, and other types of land disturbances related to urbanization. Increased rates of pond degradation in urban areas have been termed cultural eutrophication. Symptoms of eutrophication include dissolved oxygen fluctuations, algal blooms, fish kills, and decreased water depths.

The Virginia Tech Duck Pond area has evolved into a multi-use pond area. It has served as a quiet refuge and recreational area for students, faculty and the town of Blacksburg for over 100 years. The two ponds also function as stormwater detention basins for the town of Blacksburg, and, since their inception, they have periodically experienced massive algal blooms and produced foul odors. The ponds have been dredged several times to restore them to an acceptable quality, but algal blooms continue to persist.

The objectives of this study were to monitor major nutrient sources into the ponds, record the frequency of algal blooms, and to implement a pond-management scheme that would retard algal blooms. The management scheme was designed to aid managers and residents of small urban ponds who have low operating budgets and a lack of technical equipment.

LITERATURE REVIEW

Prior to the development of any pond management schemes, specific pond problems must be identified. Extensive monitoring programs should then be implemented to identify causal relationships and effects. Afterwards, proper control systems or procedures may be designed to address the identified problems. With this approach in mind, this section has been divided into four sections that discuss: 1) cultural eutrophication, 2) allocthonous sources and controls, 3) autochthonous sources and controls, and 4) pond treatment alternatives.

Cultural Eutrophication

The majority of researchers associate the term "cultural eutrophication" with increased productivity above that which would have naturally occurred in the absence of urbanization. Therefore, the factors contributing to cultural eutrophication should be understood before developing appropriate management schemes.

Liebig (1) postulated the concept of limiting nutrients which stated that biotic growth is limited by the required nutrient that is in shortest supply (2). Considerable controversy has periodically arisen over which nutrient is limiting: inorganic carbon, nitrogen, or phosphorus. The nutrient ratio required for photosynthesis to produce algal biomass is 106:16:1 for inorganic carbon, nitrogen, and phosphorus, respectively (3). Under normal light conditions with adequate supply of micronutrients, the majority of researchers recognize that the first nutrient to limit algal biomass in most environments is phosphorus (4).

Vollenweider (3) developed nutrient-loading models to estimate safe nutrient loading limits to reservoirs. Safe loadings were considered those that would not stimulate excess algal growths. These models focused lake management strategies toward reducing phosphorus loads.

Other researchers (5, 6, 7), however, quickly defended the inorganic carbon limitation concept. They reasoned that carbon can be the limiting nutrient in some instances because algae require more carbon than either nitrogen or phosphorus. The available sources of inorganic carbon were also a subject of debate for inorganic carbon limitations.

Ponds receiving excessive phosphorus loads are normally nitrogen limiting, and they often have blue-green algal species because green algae, unlike blue-green algae, are unable to assimilate atmospheric nitrogen sources (2).

Lakes are dynamic systems, and a number of algal nutrients periodically may become limited, especially if a lake is highly eutrophic. Despite a few exceptions, most lakes appear to be phosphorus limited (4); thus, most management schemes are designed to reduce phosphorus concentrations to a limiting level. Phosphorus is also the easiest of the three main algal nutrients to suppress because both nitrogen and inorganic carbon are in the atmosphere.

The effects of rapid accumulation of nutrients, sediment, and organic matter are: decreased lake volume, algal blooms, foul odors, depletion of dissolved oxygen, and fish kills. Some eutrophic algal species produce toxins that can harm animals and humans (8). Decomposing algae release nutrients and organic matter back into the water column, thus perpetuating the eutrophication process. Decaying algae settle to the bottom of a lake causing depletion of the dissolved oxygen in the hypolimnion. Water quality is reduced during anoxic conditions by the release of nutrients and the generation of sulfides, iron, and ammonia (2). These conditions often promote the development of nuisance fish populations such as carp and suckers (8). These combined effects decrease the aesthetic qualities and the usefulness of a pond. Although urbanization increases the rate of eutrophication, one must recognize that eutrophication is a natural process, and it cannot be totally eliminated. Hutchinson (9) eloquently stated: "Lakes seem, on the scale of years or of human life spans, permanent features of the landscape, but they are geologically transitory, usually born of catastrophes, to mature and die quietly and imperceptibly."

Allochthonous Sources and Controls of Algal Nutrients

The 1972 Amendments to the Federal Water Pollution Control Act (Public Law 92-500) recognized urban runoff as a possible pollutant source. Previously, water quality management had dealt only with point sources. In 1978, the Environmental Protection

Agency initiated the Nationwide Urban Runoff Program to examine the magnitude of urban runoff pollution. From these studies and others, it has become evident that nonpoint runoff, especially from urban areas, is a major source of algal nutrients and other pollutants (10). Reducing nonpoint nutrient loads is a complex problem because each rainfall event varies in magnitude, intensity, and duration (11); new data are extremely time-consuming and costly to obtain (12); and mitigation measures often require frequent maintenance and funding (11).

The composition of urban runoff is dependent upon the local geomorphology, types of impervious surfaces, soil types, and numerous other factors associated with urbanization of a watershed. Runoff pollutants normally accelerate the eutrophication process. Pollutants, other than algal nutrients, may also be present in urban runoff, but this paper is limited in that only algal nutrients were examined.

During a storm event, accumulated pollutants may become solubilized or suspended in solution and washed into streams or storm sewers. Pollutants are usually attached loosely to a surface and are normally concentrated in the first portion of a storm event. Griffin *et al.* (13) plotted cumulative flow versus cumulative pollutant load to illustrate a response referred to as the "first flush" effect, which describes the phenomenon.

Nutrient-loading graphs have helped quantify the impact of urban stormwater runoff on the recieving waters. Grizzard *et al.* (14) reported that urban areas contributed phosphorus and nitrogen levels from two to ten times greater than forested watersheds. Randall *et al.* (15) concluded that stormwater runoff from the urban Occoquan watershed in Virginia accounted for 85 percent of the nitrogen and phosphorus stream loads. Urban areas accounted for 28.9 percent of the annual phosphorus loads entering certain sections of Lake George in New York while contributing only 6.5 percent of the annual phosphorus load to other sections of the lake (16). These differences were attributed to the percentage of urbanization with the highest nutrient loadings correlating with the higher developed areas. These results reinforce the complexity of runoff studies.

Griffin *et al.* (13) reported that the flux of suspended solids during a runoff event exhibited a first-flush pattern, but the total soluble phosphorus flux did not. They postulated that the pollutant phase (soluble or particulate) influenced the first-flush response and described basic pollutant-removal processes that were involved. Insoluble pollutants are removed from land surfaces by physical mechanisms such as dislodgment by raindrops. Consequently, suspended solids should appear in the first portion of runoff during a storm event. In contrast, the concentration of soluble pollutants in stormwater runoff is dependent upon solubility equilibria, not physical mechanisms. Thus, the flux of soluble pollutants, such as orthophosphate, may not exhibit a first-flush effect. Other studies (16, 17), however, have reported orthophosphate first-flushes. Again, these discrepancies are typical of runoff study results.

The previously referenced studies emphasize that loading graphs are essential to one who wishes to design effective nutrient runoff control systems. The type of control systems that will be effective is also dependent upon the forms of nutrients in the runoff. Because the majority of the literature suggests that phosphorus is most often the limiting nutrient, this paper has been limited to phosphorus-control systems.

Phosphorus in stormwater runoff may occur in the forms of particulate inorganic and organic phosphates and dissolved inorganic phosphorus (2). Particulate phosphate forms include phosphorus in organisms and mineral phases in which phosphorus is adsorbed onto inorganic complexes such as clays, carbonates, and ferric hydroxides. Dissolved phosphorus forms include orthophosphate, polyphosphates mostly of detergent origin, and organic colloids. Orthophosphate forms of phosphorus are the most readily available form for biological uptake.

Dissolved inorganic phosphate and some particulate forms of phosphorus can be chemically inactivated within a stream. Harper *et al.* (18) integrated flow meters and injection pumps to dose aluminum sulfate (alum), based on flow, into storm sewers upstream of Lake Ella, Florida. The alum was adequately mixed in the storm sewers and produced a floc that settled to the bottom of the lake, thus reducing phosphorus loads by 90 percent. No adverse affects were observed upon the benthic organisms in Lake Ella.

At another Florida lake site, Harper *et al.*(19) routed stormwater through a filter medium consisting of a 50-50 mixture of alum sludge from a local water treatment plant and silica sand. Stormwater was collected in an underground basin, filtered through the alum-sand mixture, and discharged to the lake through drain pipes. Approximately 80 percent of the nutrient and sediment loads were removed. Another stormwater treatment facility was constructed above ground in the following manner: a small basin was excavated and partially filled with an alum-sand filtration media with underdrains flowing to the lake. The system was landscaped and covered with decorative rocks to resemble a rock garden. Stormwater flowed into the basin and then percolated through the rocks and filtration media. Again, nutrient removal efficiencies were in excess of 80 percent (19). Although both filtration systems were efficient in removing nutrients, no data were available on the length of time the system would remain effective.

Bannink and Vlugt (20) used flow-proportional additions of 10 milligrams per liter (mg/L) of ferric sulfate to reduce phosphorus levels in storage reservoirs on the Rhine

River in the Netherlands. Algae biomass was reduced while acceptable levels of water quality were maintained.

Hayes *et al.* (21) also used ferric sulfate to inactivate phosphorus in the Great Ouse River before it emptied into the Foxcote Reservoir in England. The Foxcote Reservoir is a storage reservoir for drinking water, but it had been untreatable for domestic consumption for up to six months each year because of dense algal blooms. A single application of ferric sulfate was applied to the reservoir. Afterwards, an inflow ferric sulfate dosage scheme was designed to treat the river water. The results of the reservoir ferric sulfate treatment will be presented in the next section. The in-line treatment scheme was mechanized, thereby requiring less time and money than a pond treatment scheme. Reduced external loadings and lake chlorophyll *a* levels resulted from the ferric sulfate treatment (21).

Calcium in the form of lime has reduced phosphorus in sewage effluents; however, it is of limited use in most lakes because effective phosphorus removal occurs at pH levels above those found in natural waters, as dictated by chemical equilibria (22).

Nutrient loads may also be controlled by constructing detention ponds. The dominate nutrient-removal processes within a detention pond are the uptake of nutrients by algae and sedimentation; therefore, an effective design for a detention pond should address the following:

- The design size should retain a predetermined percentage of runoff pollutants based on loading curves (13).

- To prevent anoxic conditions and subsequent release of nutrients, the pond depth should not exceed the mixing depth (23).

- The design depth should be close to the euphotic zone, because phosphorus removal is influenced by photoautotrophic processes (23).

A large detention pond constructed ahead of the Jesenice Reservoir in Czechoslovakia removed approximately 65 percent of the total phosphorus load (23). The phosphorus removal efficiency was reported to be an exponential function of the detention time. The detention pond was three meters (m) deep with a theoretical detention time of five days.

Yousef *et al.* (25) studied nutrient removal characteristics of two different aged detention ponds that received highway runoff. While both ponds had similar dimensions and nutrient loadings, one pond was more efficient in removing nutrients. The authors postulated that some of the removal mechanisms, such as biological uptake and sorption of phosphorus onto the sediments, in the one-year-old pond had not fully developed compared to the seven-year-old pond. The bottom sediment in the older pond functioned as a sink for phosphorus, removing 90 percent of the annual loads.

James *et al.* (26) stressed the importance of location and size of detention ponds. A series of small detention ponds located in the upstream portion of a watershed could only treat a small percentage of the total watershed area. Improperly sized detention ponds may require frequent maintenance and result in increased operation costs. Optimum detention pond sizing and location should be based on the degree of nutrient removal and flood control desired.

Kuo *et al.* (27) modeled the effectiveness of infiltration trenches, porus pavement, and detention ponds in reducing nutrient loads. An infiltration trench is a subsurface trench that temporarily stores runoff. The collected runoff slowly percolates through the stoned lined trench into the surrounding soil. Porus pavement is permeable asphalt that allows water to quickly pass through to a subsurface aggregate system for infiltration into the soil. Detention ponds with long detention times were determined to be the most cost-effective of the three nutrient control systems studied (27).

Cultural eutrophication may also be reduced by local ordinances that require runoff-control systems to be implemented in urban and rural areas. Currently, most states have erosion- and sediment-control laws designed to regulate land-disturbing activities. Although states may have laws controlling the initial building phase, few have laws specifically governing stormwater runoff (10). For a complete review of federal and state regulations concerning runoff, refer to Kuo *et al.* (27).

Despite removing large percentages of the annual external nutrient loads, some ponds and reservoirs continue to experience nutrient-related problems because of internal nutrient sources, which is the next topic.

Autochthonous Sources and Controls of Algal Nutrients

The previous section discussed the significant amounts of algal nutrients present in runoff and the immediate effects upon a waterbody. After entering a waterbody, these transported nutrients generally settle to the bottom of a pond or become incorporated into aquatic biomass. Numerous physical, chemical, and biological processes then determine whether the materials will cycle internally or remain incorporated within the sediment. Because lake sediments normally contain high phosphorus concentrations relative to those in the lake water, gross changes in sediment phosphorus need not occur for the sediment to significantly influence water chemistry and algae growth. Lake restoration methods to control internal cycling of phosphorus or the effect of internal phosphorus cycling will be reviewed also in this section.

The majority of phosphorus in reservoir sediments consists of inorganic phosphorus (2). Sedimenting organic phosphorus is decomposed and hydrolyzed to inorganic forms, which is generally adsorbed to clays and hydroxides of iron. Inorganic phosphorus may be contained also within discrete compounds, such as variscite ($AlPO_4$ and strengite ($FePO_4$) , and within the interstitial water of sediments. A complete review of phosphorus in lake sediments and water was compiled by Seyers *et al.* (28).

The interchange of phosphorus with the overlying water column is dependent upon the redox potential, biological activity above and within the sediment, and the morphometry of the waterbody. The oxygen concentration above the sediment is a major regulatory component that controls the release or sorption of phosphorus. The oxygen concentration in the hypolimnion is a function of the amount of organic matter loading and the morphometry of the lake basin.

According to Wetzel (2), organic material originates from three main sources: allochthonous sources, decaying plants and aquatic organisms produced within the waterbody, and excretions from lake fauna. The latter two are autochthonous sources. Bacterial metabolism consumes hypolimnetic oxygen during the decomposition of organic matter. As a waterbody thermally stratifies, the hypolimnetic oxygen, unlike the epilimnectic oxygen, is not renewed, because the hypolimnion is effectively isolated from the atmosphere. Bacterial decomposition of organic matter continues to utilize oxygen until all the hypolimnetic oxygen is consumed and the hypolimnion becomes anaerobic or anoxic.

Reduced hypolimnetic oxygen concentration has a pronounced affect on the sediment water interface redox (reduction and oxidation) chemistry, and redox reactions play an important role in regulating internal phosphorus cycling. Hypolimnetic oxygen oxidizes the top one to four centimeters of the sediment; this layer is called an oxidized microzone (27). When the microzone is oxidized, it acts as a seal by binding phosphorus within iron and manganese oxides, and these oxides also physically prevent the escape of phosphorus from the interstitial water within the sediment.

When the hypolimnion becomes anoxic, the redox potential is reduced, destroying the oxidized microzone (27). As the redox potential is decreased, iron and manganese compounds in the sediment are reduced to soluble ionic forms. As previously mentioned, a large percentage of the sediment phosphorus is complexed and adsorbed with iron compounds, so when the iron-phosphate compounds are reduced, soluble phosphate ions also are released into the water column.

The transfer rate of phosphorus at the sediment-water interface is also influenced by biological activity within and above the sediment, though biological factors are of lesser importance than the chemical factors. The burrowing activities of benthic organisms can release nutrients by altering the surface of the oxidized microzone. Bottom-feeding fish activities can displace nutrient-rich interstitial water and replace it with the overlying water. The displacement of the interstitial water also increases the oxygen content of the sediment, thereby allowing further biotic activity in the sediment.

Lake morphometry can also influence the phosphorus transfer process. Periodic mixing within a shallow lake may physically disturb the sediment and the microzone. Bates *et al.* (29) reported that mixing can promote the release of the interstitial water within the sediment, which may contain 50 times the amount of orthophosphate-phosphorus in the overlying water column. A deep lake, however, mixes infrequently. The effects of internal nutrient cycling are less pronounced in a deep lake, partially because the large water volume dilutes the phosphorus concentrations.

Sediment redox processes involved in phosphorus release are well documented, but they are usually influenced by other environmental factors; such as pH, types of sediment, temperature, allochthonous loading rates, flushing rates and depth of overlying water; thus making the degree of sediment-water interchange a dynamic and complex process. Nevertheless, nutrient-rich sediments in most eutrophic waterbodies can periodically act as substantial nutrient sources (10, 28, 29).

A number of lake-restoration methods have been designed to retard internal phosphorus cycling. One method utilizes chemical procedures employed extensively at water- and wastewater-treatment facilities. An aluminum salt, usually aluminum sulfate, is applied to a reservoir to inactivate phosphorus in the sediment and to precipitate the phosphorus in the water column. Inorganic and organic phosphates adsorb to aluminum, forming aluminum hydroxide precipitates and colloidal aluminum hydroxides. These precipitates form a floc that covers the bottom sediment and continue to adsorb phosphorus released from the bottom sediment, sometimes for several years (30). Unlike iron-phosphate compounds, aluminum compounds are not altered by changes in the redox potential; thus, phosphorus compounds bound to aluminum will not be solubilized under anaerobic or anoxic conditions.

The chemistry of aluminum has been extensively reviewed by Burrows (31). When aluminum is added to water, a number of solid and dissolved aluminum hydroxides form depending on the pH of the water treated. These hydroxides differ from most hydroxides in that they (31):

- can form complexes with other numerous ions such as phosphorus.
- can polymerize.
- are amphoteric.

Aluminum hydrolysis destroys a portion of the carbonate alkalinity, reducing the pH. A 1.0 ppm dose of alum consumes 0.5 ppm of the carbonate alkalinity (32). Insoluble aluminum complexes predominate between a pH range of 6 to 8. Thus, alkalinity and pH are two important water-quality parameters that regulate the amount of alum that can be added to a water body. Exceeding the pond alkalinity buffering capacity would result in lowering the pH and increasing the soluble aluminum concentration. Below pH 6.0, aluminum becomes increasingly more soluble. Aluminum, the most prevalent metal in the earth's crust (33), occurs naturally in insoluble, complex minerals, and in those forms does not pose a pollution problem. Soluble aluminum species, however, are toxic to many aquatic organisms. Similar to other metal toxicity reports, a large number of previous aluminum-toxicity studies did not consider the complex chemistry of the metal, and because of the ambiguity of the test conditions, these reports have not been reviewed here. Other studies were much better in that they considered aluminum chemistry. Some of them are reviewed here.

Everhart and Freeman (34) exposed various life stages of rainbow trout to aqueous aluminum complexes. A dissolved-aluminum concentration of 0.05 ppm had no measurable effects on growth or behavior on any life stages of rainbow trout. Similar to other metals in aquatic systems, some aluminum complexes are also toxic to aquatic organisms. In basic solutions, aluminum hydoxide, the phosphorus absorbing floc present at a pH range of six to eight, converts to the aluminate anion ($Al(OH)_4^-$), due to its amphoteric properties (31). Everhart and Freeman (34) demonstrated that larger concentrations of the aluminate anion, controlled by increasing the pH, were progressively more toxic to pH-acclimated rainbow trout fingerlings. They concluded that in most waters with a pH greater than 5.5, a 0.10 ppm concentration of either soluble aluminum or aluminate anion should not adversely affect rainbow trout.

Lamb and Bailey (35) examined the effects of alum on *Tanytarsus dissimilis* , a benthic invertebrate. During the 96-hour acute bioassays, no toxic responses were observed at alum doses ranging from 80 to 960 ppm. Chronic toxicity effects were observed at an alum dose of 480 ppm. This response could have been associated with the amount of floc material deposited on the invertebrates. For both the acute and chronic tests, the pH ranged from 6.6 to 7.7, and the alkalinity was approximately 30 mg/L. Dissolved aluminum concentrations for both the acute and chronic bioassays remained below 0.1 *u*g/L throughout the tests.

The earliest reported studies of lake treatments with alum had no established alum-dosage guidelines. Early objectives of alum treatments were to remove phosphorus from the water column. Today, the primary objective of whole-lake, alum applications is to retard phosphorus release from lake sediment. Kennedy (36) was the first to use alkalinity, pH, and aluminum relationships to determine a maximum-allowable alum dose. He developed a maximum-dose approach of 50 ug Al/L as a safe upper limit for a post-treatment dissolved aluminum concentration, based on the Everhart and Freeman (34) study previoulsy discussed. According to the maximum-dose approach, an aluminum dose producing a posttreatment pH of approximately 6.0 would ensure a soluble aluminum concentration below the established safe value of 50 ug/L.

Cooke and Kennedy (37) reviewed the results of 28 lake-restoration projects that involved the use of alum. The first recorded alum treatment was at Horseshoe Lake, Wisconsin in 1970 (38). Hypolimnetic phosphorus remained below pretreatment levels for approximately eight years. No detrimental effects, to either benthic organisms or fish, were reported. The majority of treatments of deep, dimictic lakes with alum have successfully reduced internal phosphorus cycling for at least five years with no adverse effects upon the aquatic fauna.

Alum may be dispensed into a lake either by a surface or hypolimnetic application. Hypolimnetic application, which is more expensive, further reduces the chances of any localized toxicity problems. Inadequate mixing during hypolimnetic alum application may prevent the formation of a uniform floc layer at the surface of the sediment, and an irregular floc cover may not significantly reduce phosphorus release from the sediment.

A similar, non-uniform floc layer could also result in polymictic lakes. Frequent mixings may destroy or concentrate the aluminum floc cover. Dosing polymictic lakes with alum for phosphorus control has had variable degrees of success. Jacoby *et al.* (39) reported that internal, sediment-phosphorus cycling in Long Lake (Washington) was reduced by an alum treatment. Long Lake has mean depth of 2.0 m and a maximum depth of 3.7 m. The alum floc continued to remain incorporated in the top layer of the sediment after two years, despite frequent lake mixing. In constrast, alum had little effect on internal phosphorus cycling in Pickerel Lake (Wisconsin) which also is a shallow, polymictic lake (40). Most of the aluminum floc was redeposited in the center of the lake after the lake completely mixed; thus, internal phosphorus cycling returned to pretreatment levels.

Ripl (41) developed another lake restoration method based on *in situ* oxidation of the sediment with nitrate. During anoxic periods, denitrifying bacteria utilize nitrate to biochemically oxidize organic matter in the upper-sediment surface. The redox potential for the reduction of nitrate is higher than that for the reduction of iron. Nitrate will be reduced, instead of iron, until the nitrate is depleted. Therefore, phosphorus combined with iron would continue to be immobilized despite anoxic conditions in the hypolimnion. Foy (42) reported that nitrate is effective in reducing phosphorus cycling, but it was 80 percent more expensive than aluminum salts.

Theis *et al.* (43) examined the effectiveness of fly ash from coal-burning, power plants for reducing internal-phosphorus cycling. Because fly ash is a waste product, incorporating it into a lake restoration program would represent a savings to both the power plant and the lake program. Although fly ash significantly reduced internal phosphorus cycling in Lake Charles East in Indiana, many bluegill were killed. They concluded that the fish kill resulted from the improper dosing of the lake with lime at the same time the fly ash was added. Numerous other studies (44, 45), however, have reported a wide range of adverse effects of fly ash on aquatic fauna. Presently, fly ash does not seem to be a viable lake restoration material, because it may produce more harm than benefits.

Artificial circulation, another lake-management technique (46), involves destratifing the entire lake by a mechanical device. Under-sizing the mixing device may result in incomplete destratification, and anoxic conditions may prevail in some regions of the lake.

Over-sizing often results in fish kills because nitrogen gas becomes supersaturated, causing the equivalent of bends in fish. Theoretically, artificial aeration should oxidize the sediment and prevent phosphorus release, but it may increase the rate of phosphorus release by increasing hypolimnetic water temperature and sediment-water exchange rates (47).

Hypolimnetic aeration, another lake restoration technique, has been used with mixed success. It has been proven to provide a habitat for cold water fish and to prevent fish kills in the winter caused by oxygen starvation (48). While information on reducing internal phosphorus cycling by hypolimnetic aeration is conflicting, hypolimnetic aeration has resulted in an increase in algal biomass in some cases.

Another lake-restoration technique involves the installation of a siphon near the dam to remove the anoxic and nutrient-rich water from the hypolimnion. This device is referred to as an Olszewski tube (30). This method is inexpensive and dependable because of the lack of mechanical parts. The lake morphometry dictates whether the siphon will be effective in removing the anoxic hypolimnetic water. The siphon would not function properly if depressions exist throughout the lake. Precautions would have to be taken to insure that the nutrient rich and anoxic water removed from the hypolimnion would not adversely affect the downstream water quality.

The final lake-restoration method to be discussed here is dredging, which is by far the most expensive and destructive. Dredging, however, is the only restoration method that closely returns a lake to its original state (49). Numerous devices have been designed to reduce the initial effect of dredging. For example, suction devices may be used instead of bucket type dredges to remove the sediment. Thus, the amount of resuspended material is reduced. Dredging to reduce the effects of eutrophication may also create another dilemma of how to dispose of the dredged sediment, that may contain toxic, synthetic organics and heavy metals.

In summary, many lake-restoration methods have been proposed, but most are still in the developmental stages. Their overall effectiveness and longevity are only speculative. Although dredging often results in improved water quality, disposal problems can arise. Each lake is unique, and, presently, there is no panacea for cultural eutrophication. Most of the documented lake-restoration programs have involved large lakes. Because there is a large number of small urban lakes, there is a demand for the development and evaluation of sound, inexpensive methods to slow the cultural eutrophication process.

Alternative Lake-Management Methods

The previous sections reviewed processes that reduced algal densities by controlling the availability of algal nutrients. This section contains a review of direct approaches for controlling algae.

For many decades, copper sulfate has been used extensively to control blooms of numerous species of algae and continues to be the most popular algicide (50). Copper added to a reservoir quickly precipitates to the sediment, but not so quickly that algal growth is not reduced. Because copper does not remain in the water column for extended periods, the level of toxicity to higher organisms is reduced. An algal-control program based solely on copper sulfate does not alleviate the cause of the bloom. Copper sulfate programs often require algal monitoring and routine doses of copper sulfate to adequately suppress algal blooms. Nevertheless, copper sulfate offers a viable alternative to lake managers who lack the technology and funds to set up expensive and time-consuming data-collection programs to identify and control the major nutrient sources.

It is generally accepted that the cupric ion, not the total copper concentration, is the important form of copper that is toxic to aquatic organisms. The exact mechanism that makes copper toxic is unknown. For instance, some researchers believe that additional copper species exhibit toxic effects upon aquatic organisms. Wageman and Barcia (51) attributed the algicidal effects of copper to a "toxic copper" concept, which holds that toxicity is caused by the cupric ion, $Cu(OH)_2$, and $CuOH^+$. Lauren and McDonald (52) presented evidence to partially confirm the total toxic copper concept. The test solution pH was lowered from 7.0 to 5.0. Increased cupric ion concentrations resulted, but the toxicity to juvenile rainbow trout remained the same (0.2 ppm). These results indicate that the dominate noncarbonate species $Cu(OH)_2$ at pH 7.0 is also toxic.

The algicidal effects of copper were first noticed in the mid-1800's (53). In 1959, Hale (54) compiled a safe, copper sulfate dose for several fish species. A copper sulfate dose of 0.14 ppm was deemed safe for trout. The report failed to mention the test conditions such as alkalinity and pH. Most of the early copper-toxicity studies are of little quantitative use today because the test conditions affecting copper speciation were vaguely described. McIntosh (55) reported that a copper sulfate manufacturer recommended 0.25 ppm copper sulfate for planktonic algae, while 0.5 ppm was recommended for filamentous algal growth. Again, no water quality data was taken into account in establishing the copper dose.

In a more recent study, Starodub *et al.* (56) reported that 100 *u*g Cu/L caused a 50 percent reduction in the photosynthetic activity of the green alga, *Scenedesmus* *quadricauda* . Mcknight (57) reported that the growth of *Ceratium hirundinella* , a green alga, was inhibited by 50 percent at a cupric ion activity of 6 *u*g/L.

An American Water Works Association Research Foundation report (58), citing Richey and Roseboom (59) reported that bluegill fry should not be exposed to more than 0.27 ppm soluble copper, while a level of 0.12 ppm soluble copper should not be exceeded for channel catfish. Studies of algae and fish toxicity indicate that unless copper sulfate is grossly overdosed, or if the alkalinity is low, copper sulfate should not be acutely toxic to fish. Thus, copper sulfate is a good algicide because at low concentrations it suppresses algal growth but does not cause acute toxicity in nontarget organisms under general lake water conditions (58). Richey and Roseboom (59) did not consider sublethal stress or the long-term environmental stress created by copper dosages.

Felts and Heath (60), citing Finney (61), reported a 96-hour, 50 percent lethal concentration of dissolved copper at 0.62 ppm in bluegill. Felts and Heath used Finney's probit analyses to determine a sublethal copper concentration. A value of 0.28 ppm cupric ion was determined to represent 98 percent survivability. This sublethal value caused a decrease in whole-body oxygen consumption in bluegills. Combined with increased water temperatures, copper could cause additional stress upon bluegill. The wide range of copper-toxicity levels requires one to understand the factors affecting copper availability before an optimal dose can be applied to a lake. Copper toxicity is influenced by numerous interactions between the metal and the lake water. Factors affecting copper speciation include bicarbonate alkalinity, pH, and organic complexing constituents. Hydrolysis and precipitation reactions involving inorganic constituents, such as carbonate and hydroxide, are important in controlling copper speciation. Copper doses in high alkalinity waters result in the formation of increased levels of the nontoxic, particulate copper complexes, such as malachite ($Cu_2 (OH)_2 CO_3$ and tenorite ($CuOs$)). Although alkalinity affects the amount of ionic copper in solution, Slyva (61) reported that pH had the greatest effect on the hydrolysis and precipitation reactions of copper complexes.

Organic chelating agents, which exist in all natural waters supporting life, also have an affinity for complexing metals. The complexations; involving organic substances such as humic, fulvic, and amino acids; are difficult to chemically model or distinguish, but their influence on copper chemistry should not be ignored (57).

Mcknight *et al.* (63) developed a copper sulfate dosage determination method that incorporates both the organic and inorganic complexing capacities. The cupric ion activity is measured by a selective ion electrode as a function of the total copper added to lake water in the laboratory. These data can then be compared to algal toxicity data, if any exist, to determine an optimal copper dose. This method should help lake managers decrease their copper sulfate doses.

In some cases, chelating agents, such as citric acid and triethanolamine, in high alkalinity waters have increased copper toxicity (63). Chelators decrease the supersaturation of malachite and tenorite; thus, soluble copper forms remain in the water column for longer periods of time. The citric acid-copper complexes may be toxic (64) or they may may release the copper slowly (58). The algicidal effects persist over a longer period of time if copper is slowly released from the citric acid complex. Like the cupric ion, the mechanism of toxicity is unknown for citric acid-copper complexes and other chelators. More studies are needed, but chelators can reduce the amount of algicidal copper needed in waters with high alkalinity.

Lake Monona in Wisconsin has been dosed with copper sulfate for over a 50 years (65). Dissolved copper in the water column has ranged from < 0.3 *ug*/L to 4 *ug*/L. Less dissolved copper in the anoxic hypolimnion was attributed to increased sulfide concentrations. Lake Monona continues to be a productive fishery. The top sediment layer contained 250 mg Cu/kg sediment, while the highest copper sediment concentration occurred at 60 centimeters below the sediment surface. No adverse effects to the benthic invertebrates were recorded.

The Fairmont Lakes in Minnesota have received over 1.5 million kilograms of copper sulfate over a 58 year period (66). Both short- and long-term effects of copper were observed. The adverse, short-term effects included dissolved oxygen depletion caused by decomposition of dead algae, accelerated phosphorus recycling, and fish kills directly related to copper toxicity and as a result of oxygen depletion. The long-term effects were an accumulation of copper in the sediment, a shift from green to blue-green algae, and the development of copper-tolerant species requiring higher copper doses. Copper sulfate doses ranged from 6,500 kilograms/square meter (kg/m^2) to 229,000 kg/m^2. During the first 25 years, no adverse effects were noticed. Rough-fish species also became dominant after approximately 25 years.

Some of the short-term, adverse effects of copper sulfate can be minimized by using an established method, such as McKnight's (63) to determine an optimum dose and by monitoring the growth of the algae by either direct algal counts or chlorophyll *a* measurements. Such a dosage method considers the water chemistry and reduces the possibility of overdosing. The timing of copper sulfate applications are important (63). If copper sulfate is applied after the algae bloom, the decomposition of the algae will deplete the oxygen and may result in fish kills. Algae growth data allow a lake manager to predict a bloom and dose the lake before the algae bloom, thus reducing the amount of oxygen consumed by algal decomposition.

McKnight (63) and Hansen and Stefan (66) gave a historical evaluation of adverse and positive aquatic responses related to copper doses. caused by copper toxicity. Mcknight (63) reported no major adverse effects of copper sulfate upon any of the lake biota except the algae. Hansen and Stefan (66) attributed several adverse effects to copper but failed to explain why a copper sulfate program was initiated. Algae blooms left untreated can kill fish by depleting the oxygen at night and may also favor a shift to rough fish species. New dosage methods now allow a manager to develop a dosage scheme based on the existing water characteristics. These methods should result in lower copper doses than those doses first used in the Fairmont Lakes.

Another lake management scheme is biorestoration. This type of management technique involves manipulating the food web in the lake to increase the algal grazer population of zooplankton. Schoenberg and Carlson (67) concluded that zooplankton can effectively decrease blue-green algal species by grazing (30). Increased zooplankton grazing on algae can be achieved by decreasing zooplanktivorous fish densities. Osgood (68) monitored Square Lake (Minnesota) to determine why the lake was so clear. The clarity of the water was determined to be a result of intense algal grazing by large-bodied *Daphnia*. He concluded that the success of biomanipulation depends on nutrient controls and the density of other biota in the lake.

METHODS AND MATERIALS

In this section, a historical background and a description of the study area will be discussed. Specific details of sampling and data analysis methods will also be covered.

Study Area History

Historical information was obtained from an interview with the retired Assistant Director of Buildings and Grounds, Mr. Howard Price (69), and from a 1986 seminar on the status of the ponds. Photographs indicate that the northeastern or the upper pond area was developed prior to the late 1860's. The southwestern or lower pond area was a mosquito-infested marsh-land. In the early 1900's, a committee proposed to convert the lower marsh land into a pond by building a small dam. The pond was to be used as a place for swimming, boating, and ice skating. One professor on the committee, however, predicted that the pond would quickly fill with sediment and organic matter from the Blacksburg watershed, thus converting the area back to the original marsh-land condition. The original pond design included a small detention pond on both of the major tributaries and one larger pond immediately downstream as shown in Figure 1.

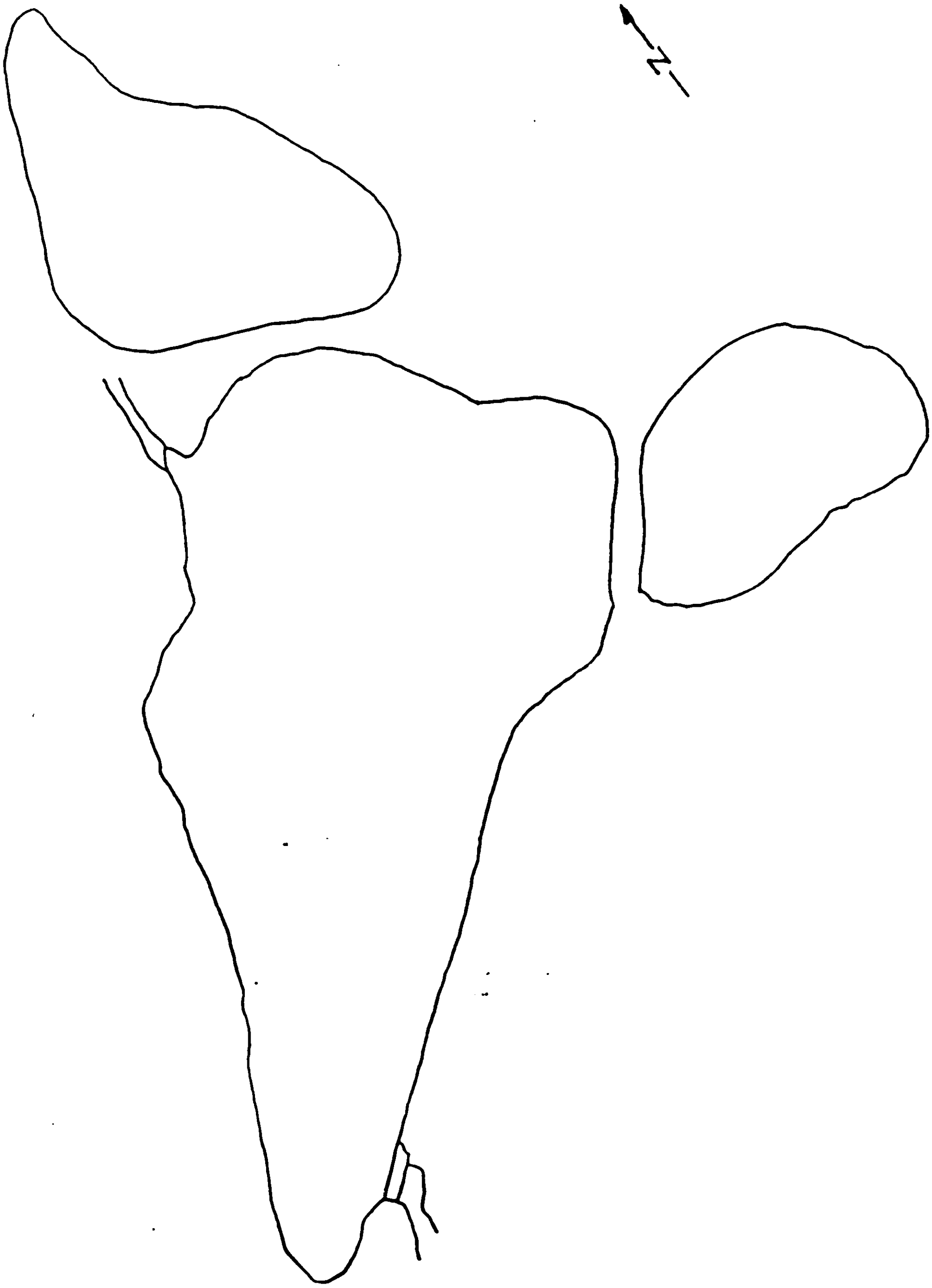

Figure 1. The original Virginia Tech Duck Pond design in the early 1900's.

The present-day form of the ponds was constructed in 1937 under a Work Progress Administration project. The Work Progress Administration was formed during the Great Depression of the 1930's to create jobs. The pond was to be used as a training facility for the Parks Department and for propagation of smallmouth bass to stock southwestern Virginia streams.

Since the 1930's, the ponds have continually experienced sedimentation problems. The ponds were dredged in the late 1950's and 1960's. The excavated depth of the lower pond was approximately 3.8 m and the upper pond approximately 2.2 m (70). Periodically, the ponds have been partially cleaned from the banks with small equipment such as back-hoes.

In the late 1970's, attention was focused on the Duck Pond area again. The State Water Control Board discovered that drains in a chemistry lab in Davidson Hall emptied into a tributary upstream of the ponds. Oil spills from the Virginia Tech power plant and nearby residential sites have further degraded the ponds. By the early 1980's, the ponds began to show evidence of sedimentation problems again. In 1985, the average depth of the lower pond was approximately 0.6 m while the average depth of the upper pond was approximately 0.3 m (70). Throughout the late 1970's and early 1980's, algal blooms were extensive in both ponds, thereby reducing the aesthetic qualities of the area. These conditions prompted another major dredging operation in 1986.

When the dredging was complete, rotenone was applied to the pond beds to exterminate any fishes. This was done to remove the undesirable fish, such as carp, that might have remained. During June and July of 1987, the ponds were stocked with 25 largemouth bass (5-7 centimeters), 15 adult largemouth bass, 265 bluegill, and 400 channel catfish fingerlings (71). In September of 1987, 48 grass carp were added to the ponds.

The pond area has evolved into an integral part of campus activities, providing a refuge and recreational area for alumni, students, faculty, and the town of Blacksburg. The ponds have also provided a habitat for a variety of wildlife, especially ducks and geese, and they have functioned as a stormwater detention facility, trapping sediment and nutrients, from the town of Blacksburg. These dual functions have historically resulted in eutrophic pond conditions requiring frequent maintenance.

Description of Study Area

The Virginia Tech Duck Pond watershed is located in Montgomery County, Virginia. Figure 2 shows the sampling locations during this study and depth contours of the Virginia Tech Duck Ponds. Throughout this paper, the larger pond will be referred to as the "lower pond" and the smaller pond as the "upper pond". The morphometric features of both ponds are summarized in Table 1. Together, these ponds receive drainage from a 634 hectare (ha) (2.5 mi^2) urban watershed. Both have drainage areas have similar land-use characteristics of single-family units, multi-family complexes, commercial properties, and institutional lands.

Five, major, geologic units have been mapped within the Duck Pond watershed: the Alluvium deposits of Quaternary age, the Rome Formation, Elbrook, and Copper Ridge Formations of Cambrian age, and the Knox Group of Ordovician age.

The Alluvium is made up of flood-plain deposits consisting of sand, silt and clay and is mostly confined to the Stroubles Creek watershed area. The geology unit of the northwestern portion of the Stroubles Creek watershed is the Rome Formation, which consists of interbedded and mottled mudstone, fine-grained sandstone and siltstone, and

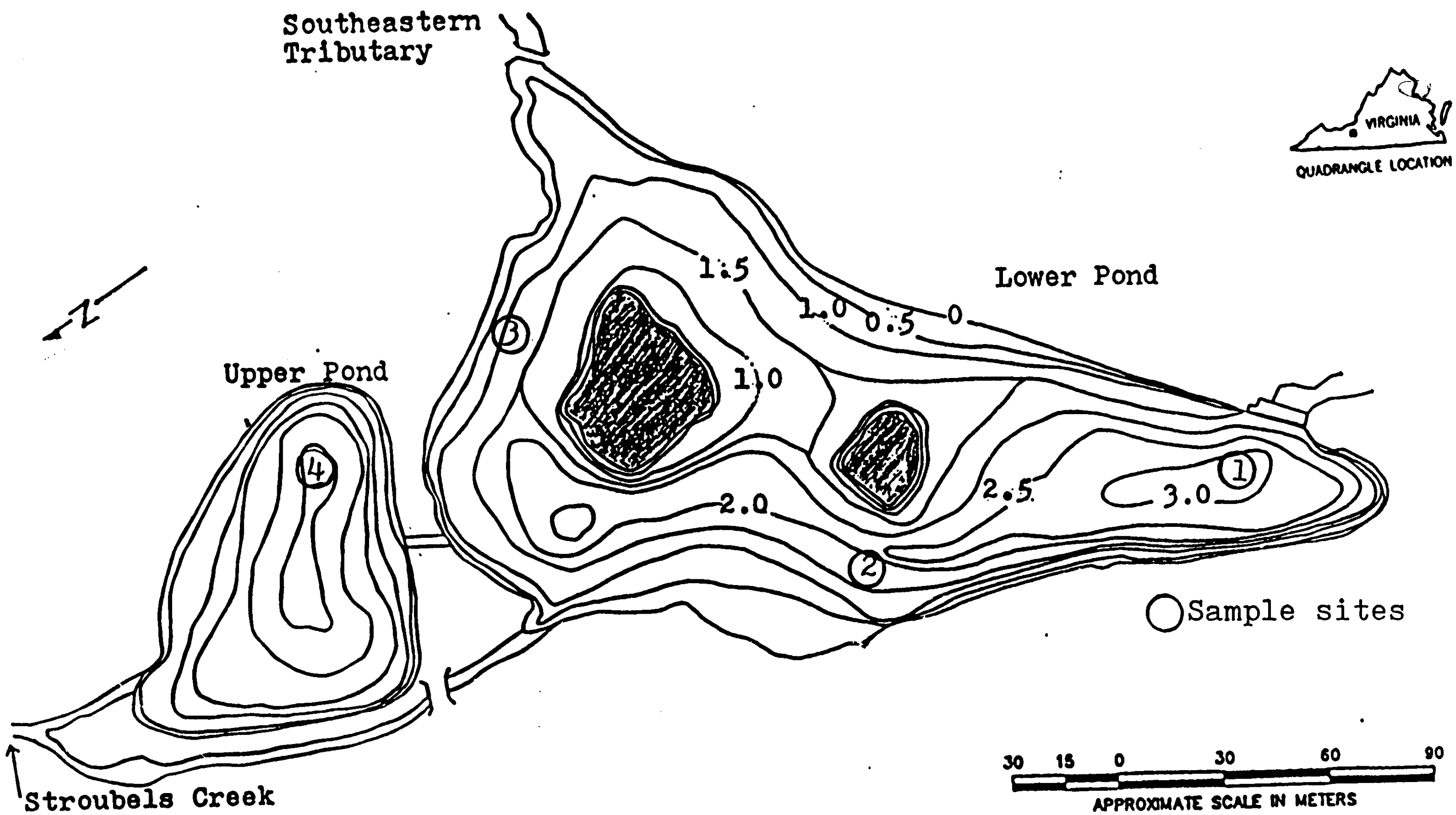

Figure 2. Location and depth contours (in meters) of the VPI&SU Duck Ponds.

Table 1. The morphological characteristics of the VPI&SU Duck Ponds.

MORPHOLOGICAL CHARACTERISTICS	LOWER POND	UPPER POND
Maximum Length	280 m	118 m
Breadth	158 m	80 m
Surface Area	23,429 $m^{\cancel{3}\,2}$	6,040 $m^{\cancel{3}\,2}$
Shoreline Length	760 m	305 m
Shoreline Development Index	1.4	1.1
Maximum Depth	3 m	3 m
Mean Depth	1.3m	1.5m
Volume	29,600 m^3	9,100 m^3
Drainage Area	303 ha (1.2 mi^2)	331 ha (1.3 mi^2)

dark-gray, fine-grained dolomite which is characterized by light-brown soil. The Elbrook Formation is the major geologic unit within the Duck Pond watershed. It contains interbedded sandy, fine-grained dolomite containing sandstone lenses and limestone bands. The soil of this formation is normally a light orangish-brown color. The geological unit of the ridge along the eastern portion of the southeastern tributary is the Knox Group. The Knox Group is characterized by thick beds of dolomite with interbeds of chert. The soil is light-brown to reddish-brown in color. Small pockets of the Copper Ridge Formation exist throughout both tributary watersheds. This formation is made up of dolomite and sandstone beds.

There are two main soil groups within the watershed: the Groseclose-Poplimento series and the Groseclose-Urban series. Small patches of the Duffield-Ernest Complex and Frederick-Vertrees Silt Loams also are located throughout the area. The majority of the soils are a brown loam, a yellowish-brown silt loam, or a fine sandy loam with a clay or loam subsoil. Slopes ranging from two to twenty-five percent exist within the watershed.

Description of Sampling Sites and Analyses

Samples were taken from three sites on the lower pond (Figure 2) in the late fall of 1987 and during the spring and summer of 1988. The following water-quality characteristics were quantified: Secchi disk transparency (SD), total phosphate-phosphorus (TP), orthophosphate phosphorus (OP), dissolved oxygen (DO), pH, temperature, alkalinity, total Kjeldahl nitrogen (TKN), algal counts, and total phosphate-phosphorus in the pond sediment.

Water samples for chemical analyses were collected with a Kemmerer water sampler (Wildlife Supply Co., Saginaw, Michigan). Sediment samples were collected with an Eckman (Forest Scientific and Mechanical Specialities Co., Chicago, Illinois) dredge. All samples, except for alkalinity samples, were transferred to Mason jars with Teflon lids that previously had been soaked in a 10 percent hydrochloric acid bath and rinsed with distilled/deionized water. Alkalinity samples were collected in brown, high-density polyethylene bottles that were acid-washed and rinsed with water.

Water transparency was determined with a 20 cm Secchi disk. Total phosphate-phosphorus samples were digested according to "Section 424 C-II, The Persulfate Digestion Method, *Standard Methods for the Examination of Water and Wastewater* (72)." After digestion, TP and OP samples were analyzed according to Section 424 F, The Ascorbic Acid Method (72), on a Klett-Summerson (Klett MFG, New York, New York) photoelectric colorimeter. TKN samples were analyzed according to an amended portion of Section 420-B, Semi-Micro-Kjeldahl Method (72). Ten milliliters (mL) of the TKN digestion reagent were added to a 150 mL water sample and concentrated by boiling to less than 50 mL. The remaining liquid was transferred to a round-bottom flask and heated on an electric heating unit until copious white fumes were produced. Immediately following the production of white fumes, the heater thermostat was set on high for 30 minutes. The remaining solution was washed into distillation flasks with several rinsings to bring the volume up to 75 mL. After ten mL of the sodium hydroxide-sodium thiosulfate reagent were added to a distillation flask, the solution was heated until 60 mL were distilled into 10 mL of boric acid. This solution was titrated with 0.02 N sulfuric acid according to Section 420-B of *Standard Methods*.

Dissolved oxygen was determined *in situ* at 30 cm depth intervals with a YSI (Yellow Springs Instrument Co., Yellow Springs, Ohio) Model 57 Oxygen Meter with a YSI Model 5739 submersible probe. The pH was determined with a Fisher pH meter/ion analyzer (Model 750) within one hour after collection. Alkalinity was determined according to Section 403, Alkalinity (72). Algae were enumerated in a Sedgwick-Rafter counting cell according to Section 1002-F-a, Algae (72). Algae were identified from information provided by three authors: Pennak (73), Needham and Needham (74) and Prescott (75).

A Fisher (Raleigh, N.C.) pH meter/ion analyzer with a reference electrode and an Orion cupric electrode (Model 94-29) were used to measure ionic copper concentrations. Aluminum concentrations were determined with a Perkin-Elmer (Norwalk, C.T.) atomic absorption spectrophotometer (AAS) (Model 703). The detection limit for aluminum was 0.1 mg Al/L.

Stormwater Collection and Analysis

The southeastern tributary (the tributary to the lower pond) was monitored during three storm events. A temporary staff gage was placed approximately 75 m upstream of the lower pond. A cross-section was taken at the gage location (Figure 3). Grab samples were taken as the water stage fluctuated throughout the storm event and were analyzed separately so that reliable estimates of the runoff-borne nutrient loads during the different periods of the storm could be determined. Stormwater flow was measured with a Gurley Pigmy (Troy, New York) meter every time a sample was taken. These measurements were used in calculation of the stormwater volume. The runoff samples were an-

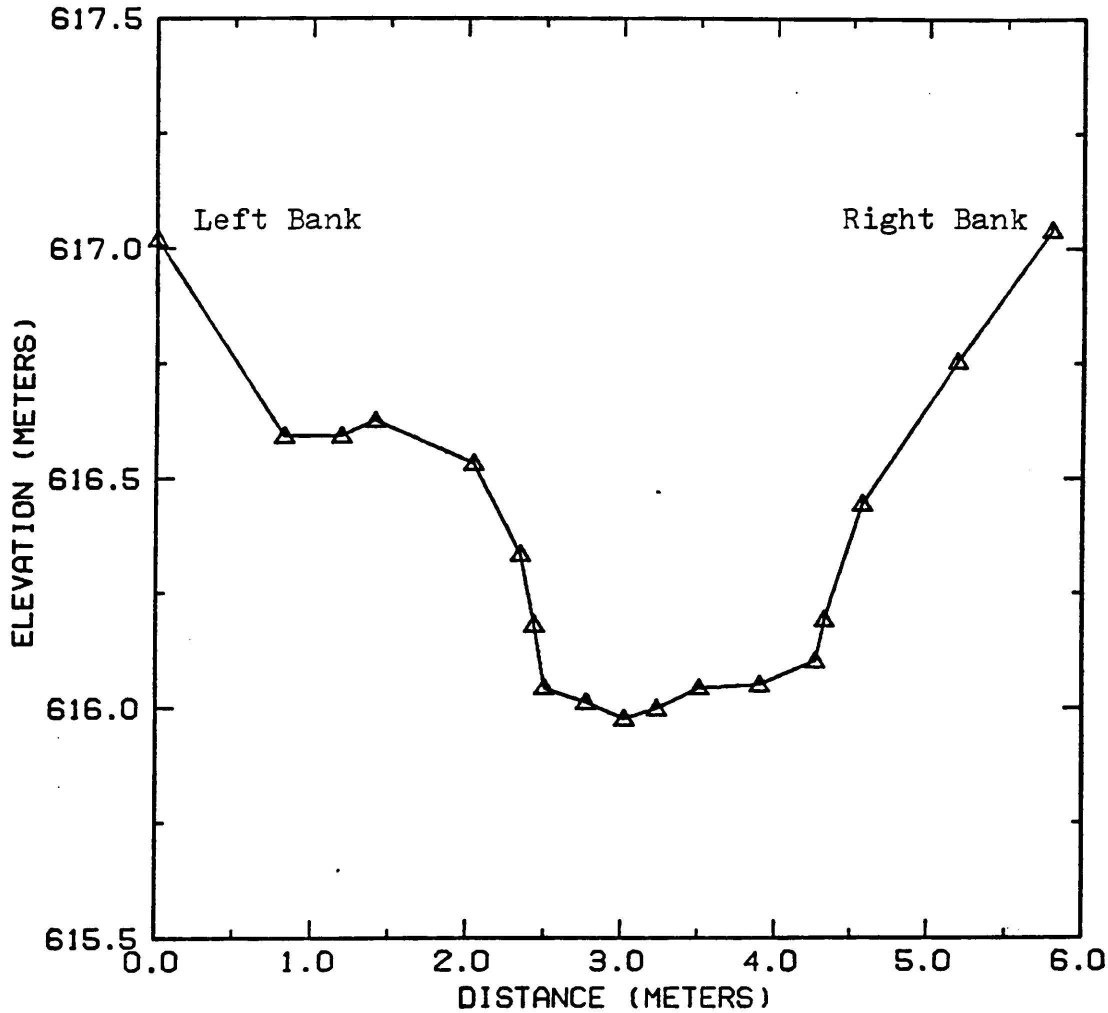

Figure 3. The cross-sectional area at the sampling site on the southeastern tributary.

alyzed for TP, OP, and TKN. The first and second storm samples were analyzed also for total suspended solids (TSS), according to Section 209 C, Total Suspended Solids (72). Only a few stormwater samples were taken on Stroubles Creek because of time and labor constraints.

Alum Dosage Procedure

A series of water samples were dosed with increasing amounts of alum (0, 10, 20, 25, 30, 40, 50 mg alum/L) as dictated by Kennedy (36). The mixture was stirred at 40 revolutions per minute (rpm) for two minutes to represent the rapid mixing of the boat propeller in the pond.

Liquid alum was obtained from the Blacksburg Water Treatment Plant in 190 liter (50 gallon) drums. A total of 1,135 liters (300 gallons) of alum were applied to the lower pond over a two day period (May 24 and 25). If this dose was mixed throughout the lower pond, it would produce a concentration of 26 mg alum/L. This dose does not represent the maximum dose as defined earlier by Kennedy (36). This dose represents one that produced a settable floc in the lower pond samples.

To keep the costs at a minimum, a simple alum dosage method was designed. The alum was transferred, by gravity, to a 95 liter (25 gallon) container that was in a 4.4 m (14 foot) boat. A Mini King (Attwood Corporation, Michigan) 360 Bilge Pump, with an outlet positioned near the boat propeller was used to mix the alum into the lower pond. The alum was distributed throughout the pond in a zig-zag fashion to ensure adequate mixing. After each 95 liter dose, the boat operator went back over the dosed area to further mix the alum, to aid in the formation of an aluminum hydroxide floc. The mixing equipment was simple and inexpensive, allowing the work to be performed by one person, thereby representing another savings.

Copper Sulfate Dosage Procedure

The copper sulfate dosage determination method followed was devised by Mcknight (57). Progressively higher doses of copper were added to pond samples. Ionic copper, considered to be the most toxic form, was measured after $CuSO_4$ was added and the mixture stirred for seven minutes or until the meter-reading stabilized. This method takes into account all the possible complexing properties, such as alkalinity and organics, of the sampled water, thus reducing the possibility of a copper overdose.

After determining the copper complexing properties, Mcknight (57) recommends that copper toxicity studies be performed on the algae most prominent in the water body. These tests were not performed in this study because it was not within the scope of this project, which was to design a dosage method that could be used by managers of small ponds or reservoirs with small amounts of money and equipment. Nevertheless, toxicity tests are valuable and should be performed whenever possible.

Copper toxicity levels to various fish (0.27 ppm for bluegill fry (59)) was then compared with the copper complexing properties of the pond water. From these comparisons, a dose of 13 kilograms (kg) (30 pounds) was determined to be a safe dose. A safe dose implies that it will not adversely affect non-target organisms such as fish.

The copper sulfate was distributed throughout the lower pond on June 15, 1988. As a precaution, this dose was assumed to mix only in the volume of water contained in the first meter of the pond. The maximum ionic copper concentration that could possibly result from the 13.6 kg dose in that volume of pond water was calculated to be less than 0.2 mg Cu/L. A 6.8 kg (15 pounds) dose of copper sulfate was applied on July 7, 1988 to the area surrounding site 3. Because an algal bloom was occurring only at site 3, no other areas of the lower pond were treated.

The first copper sulfate dose was applied over a four-hour period. The second copper sulfate dose was completed in less than two hours. The large copper sulfate crystals dissolved slowly, thus ensuring that no one area received an excessive dose of copper. The copper sulfate crystals were emptied into two burlap bags, which were then tied to the ends of a wooden 2 X 4 and placed across the front of the boat. Both ends of the board were tied separately to the front of the boat. This arrangement permitted the boat driver to distribute the copper sulfate throughout the pond without assistance. The copper sulfate was distributed into the pond by first circling the outer edges of the lower pond, then circling the edges of the islands, and then zig-zagging back across the pond until the entire lower pond was covered at which time the pattern was repeated until all the copper sulfate crystals had dissolved (Figure 4).

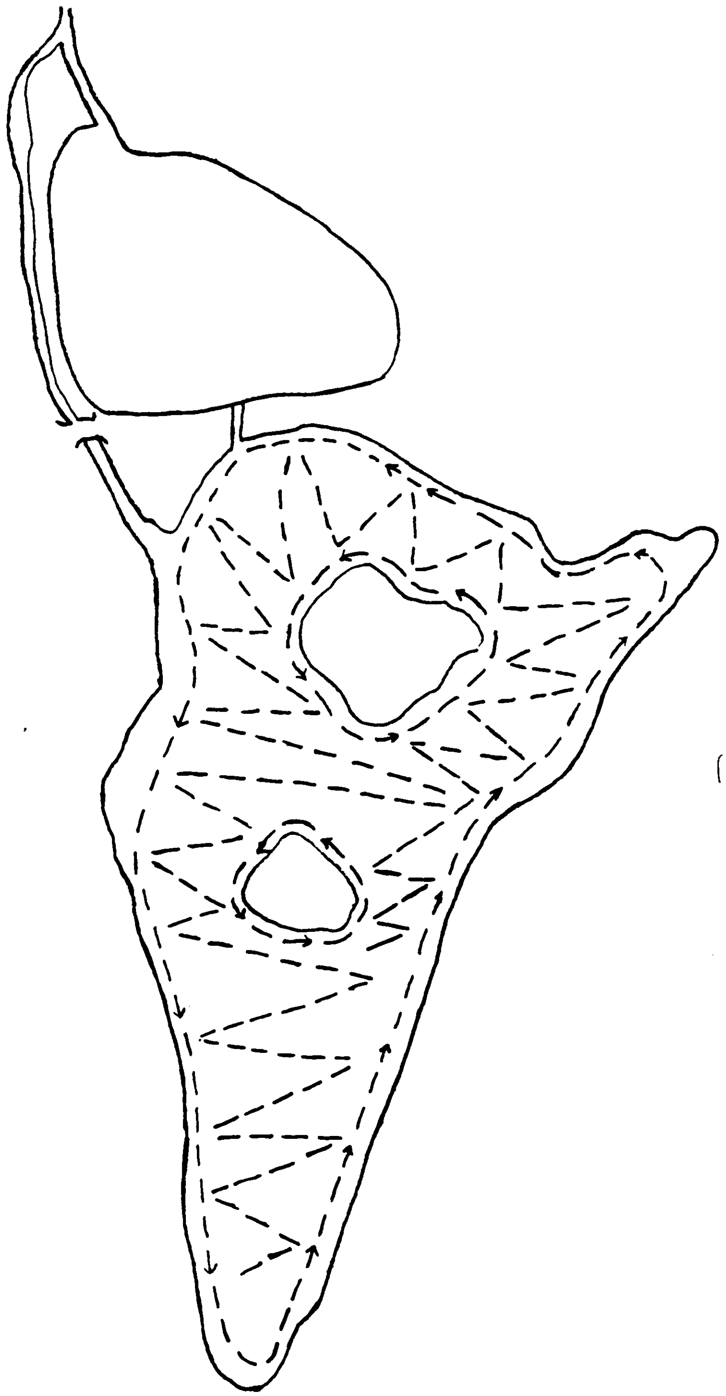

Figure 4. The copper sulfate dosage pattern that was followed in the lower duck pond.

RESULTS AND DISCUSSION

This section is divided into three sections: 1) Stormwater Quality, 2) Pond Water Quality, and 3) Management Schemes.

Stormwater Quality

The southeastern tributary was monitored because it emptied directly into the lower pond, which experienced algal blooms. Stroubles Creek, however, first drains into the upper pond and then overflows into the lower pond. Thus, the upper pond should remove the majority of runoff nutrients entering from Stroubles Creek by sedimentation processes before overflowing into the lower pond during low flow conditions. During a storm event, however, Stroubles Creek overflows into a bypass stream along side of the upper pond which empties directly into the lower pond. Therefore, nutrient loads from the Stroubles Creek watershed enter directly into the lower pond. Consequently, the upper pond is ineffective in removing nutrients during storm events. The data presented here do not include the contribution of nutrients present in Stroubles Creek or the nutrient contribution in the bypass stream.

The mass of phosphorus and TKN contained in the three storm events monitored on the southeastern tributary are shown in Table 2. The first storm monitored was on October 27, 1987. Rainfall was light during the first hours of the storm and later increased in intensity (Figure 5).

Figure 6 illustrates the percentage of runoff nutrients present during all stages of the storm event. If a cumulative nutrient curve lies above the 45 degree line, a flush has occurred (13). A flush indicates that the fraction of pollutant load removed is larger than the corresponding portion of runoff volume. Fifty percent of the OP nutrient load was contained in less than 40 percent of the total volume of the storm, thereby indicating a slight flush had occurred. Seventy-five percent of the OP nutrient load was contained in approximately 58 percent of the storm flow. TP loads were similar to OP results. If the OP in the stormwater was evenly distributed throughout the lower pond, the OP concentration would have been approximately 60 *u*g/L, which is well above the concentration that can stimulate algal growth (20 *u*g/L). Suspended solids concentrations were insignificant during this storm event; therefore, solids were not included in the loading curve.

TKN loads did not exhibit a flush during the October storm event. The TKN loading curve indicates that the pollutant removal phase was delayed. This type of removal phase could be a function of the form of nitrogen in the runoff or simply a function of the rainfall intensity. The Total Kjeldhal Nitrogen analysis includes organic forms of nitrogen and ammonia. Most inorganic nitrogen forms, nitrates and nitrites, are highly soluble, and, therefore, are more likely to be washed away rapidly. Organic nitrogen forms occuring mainly in the particulate form would not be flushed as rapidly as the more soluble inorganic nitrogen forms. Ammonia, the second component of TKN,

Table 2. Mass of phosphorus and Total Kjeldhal Nitrogen entering the lower pond in the southeastern tributary during three storms.

STORM EVENT	DATE	RUNOFF VOLUME m^3	GALLONS	TOTAL MASS DELIVERED, Kg and (lbs.) TP	OP	TKN
1	10/27/87	10,530	2,780	3.0 (6.7)	1.8 (4.0)	25.2 (55.5)
2	5/09/88	3,390	895	1.0 (2.2)	0.4 (0.8)	27.0 (59.4)
3	6/02/88	4,560	1,200	1.1 (2.4)	0.8 (1.7)	43.2 (95.0)

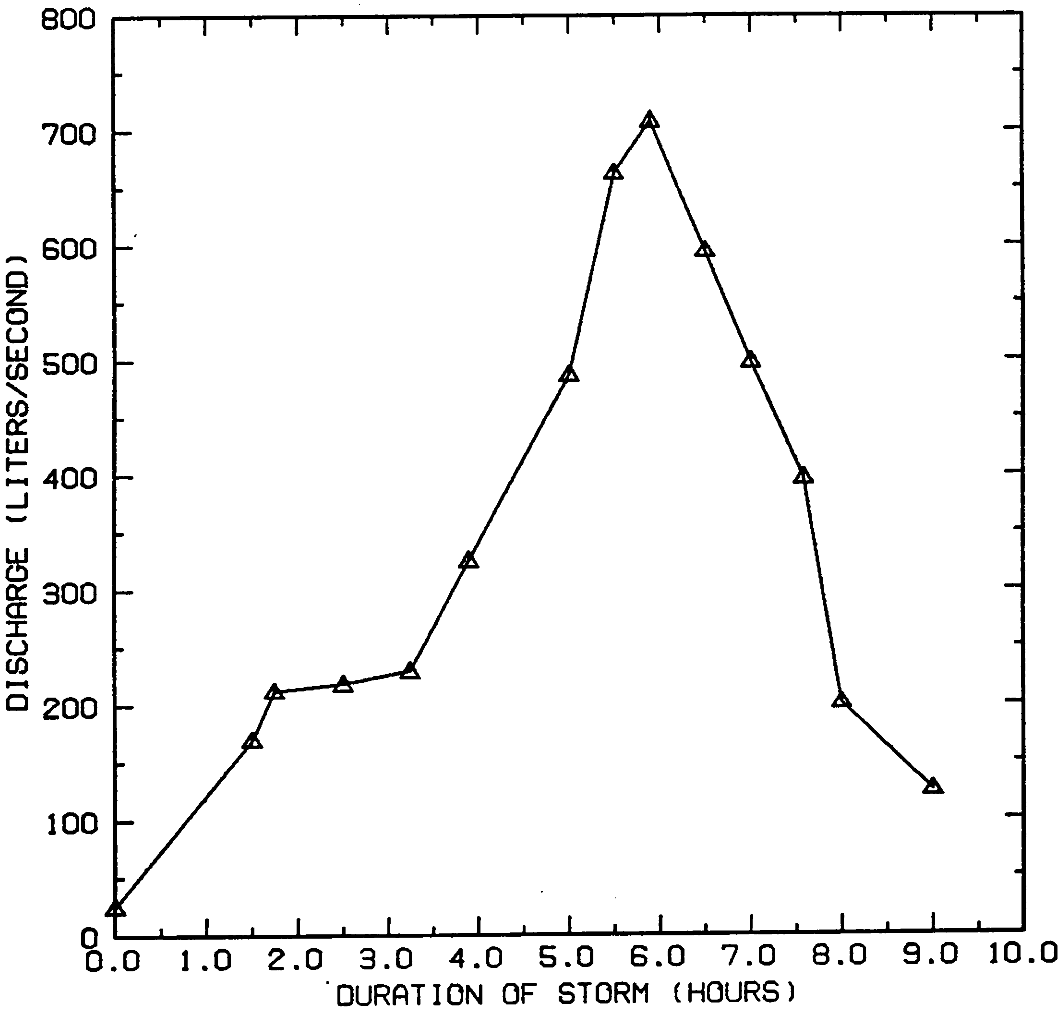

Figure 5. Hydrograph for the October 27, 1987 storm event on the southeastern tributary.

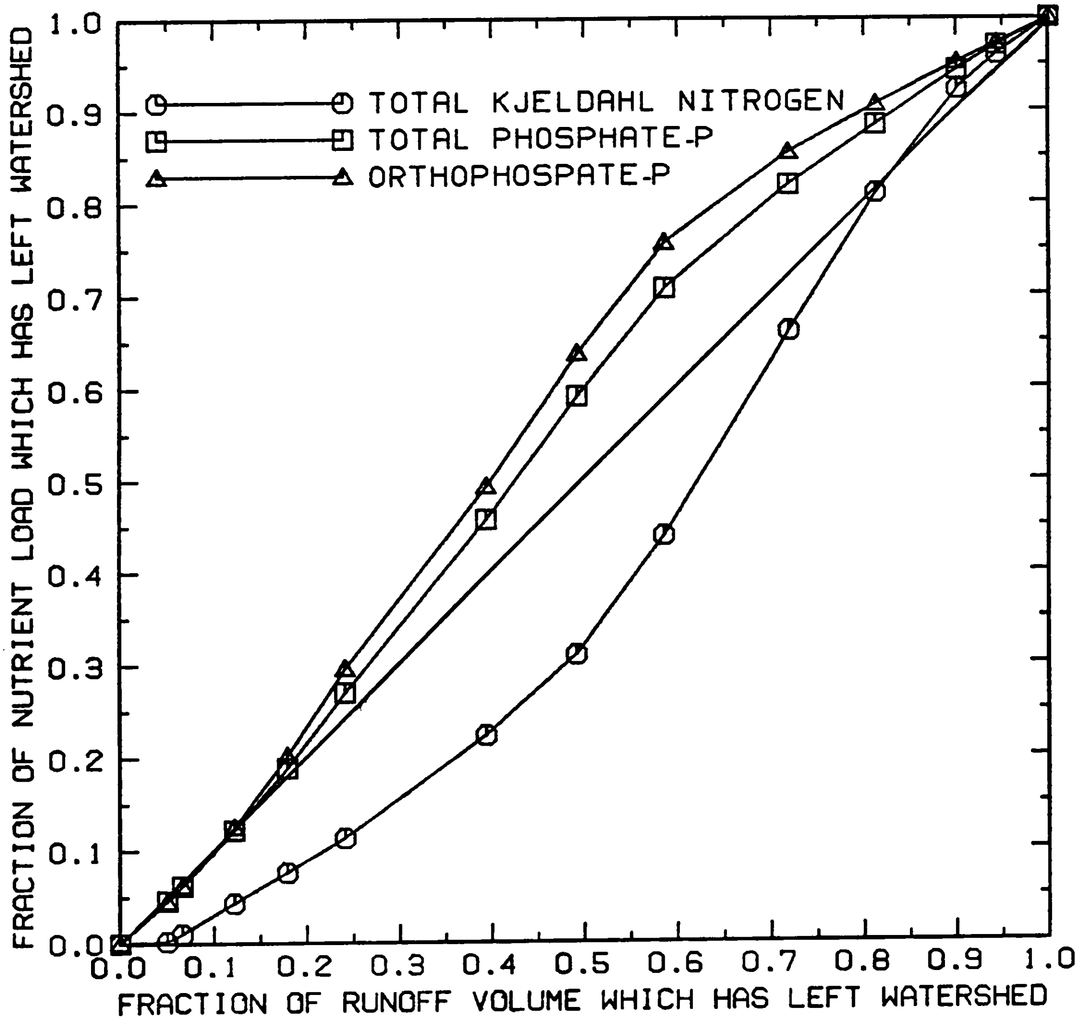

Figure 6. Nutrient loading graph for the October 27, 1988 storm event on the southeastern tributary.

however, is highly soluble and would react similar to the inorganic forms. Thus, the nitrogen in the runoff most likely was primarily organic. Because the early portion of the storm consisted of light showers, inorganic forms of nitrogen could have infiltrated into the ground instead of washing into nearby streams. Because the inorganic and organic nitrogen forms were not differentiated, the above description is only a postulation.

The second storm event was monitored on May 9, 1988; it was was typical of a spring event, starting quickly with heavy rainfall and ending quickly (Figure 7). Runoff samples were taken at intervals during a three-hour period.

All three nutrients monitored during this storm exhibited a small flush (Figure 8). Sixty-one percent of the total OP load was present in 51 percent of the total runoff volume. The largest TP flush occurred earlier in the storm when 41 percent of the total TP load was contained in 30 percent of the runoff volume. TKN loads were closely correlated with the total volume of the runoff. Suspended solids concentrations also exhibited a flush during this storm event.

The third storm event was monitored on June 2, 1988. The rainfall pattern was similar to that of the second event in that the rainfall was intense and lasted a short time (Figure 9). Runoff was monitored for 2.5 hours. The peak runoff volume for this storm was the largest of all the events monitored. Figure 10 illustrates that only OP exhibited a slight flush during the storm. Although no significant flush was observed during this storm event, the total mass of nutrients delivered to the lower pond was greater than the mass contained in the second storm.

Although each of the storms monitored exhibited various degrees of flushing, each storm contributed significant amounts of nutrients to the lower pond that promote eutrophic

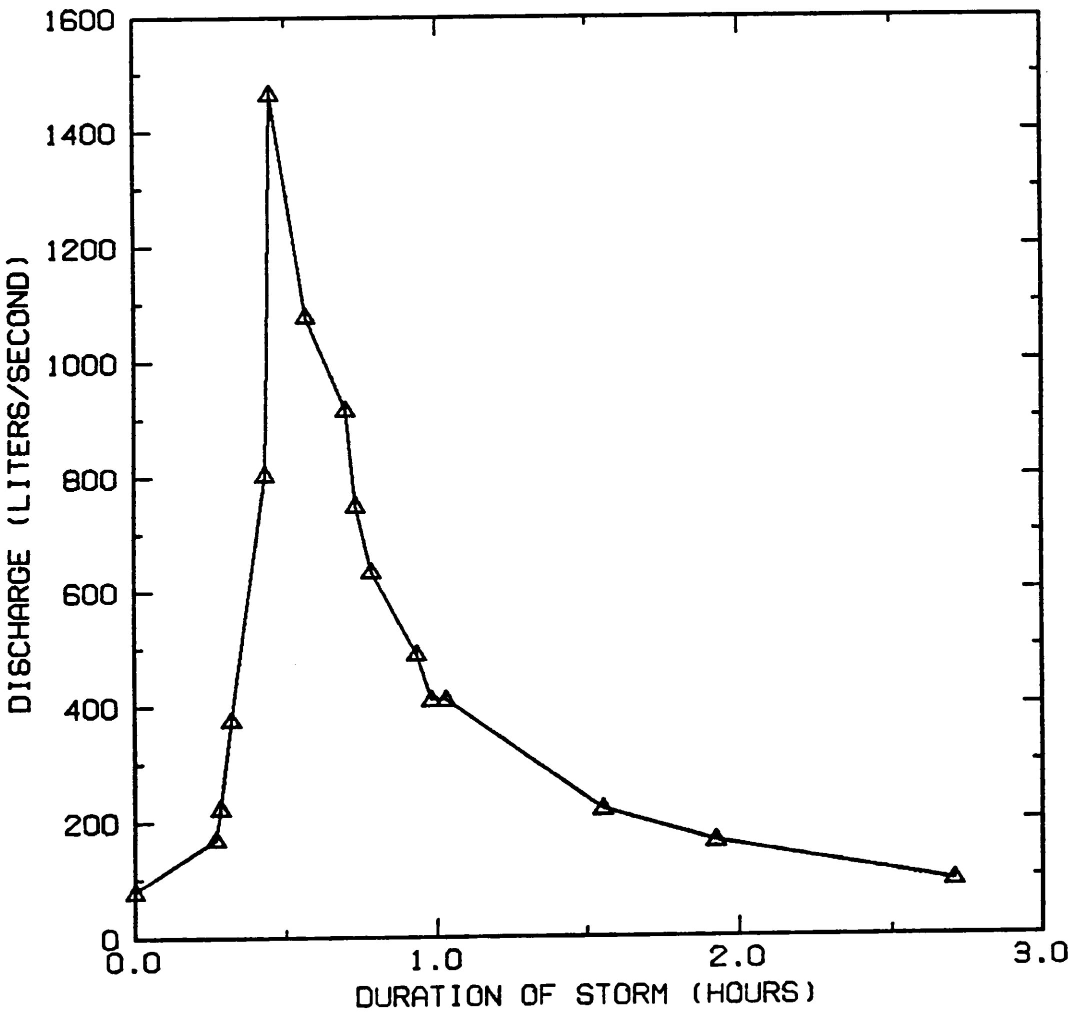

Figure 7. Hydrograph for the May 9, 1988 storm event on the southeastern tributary.

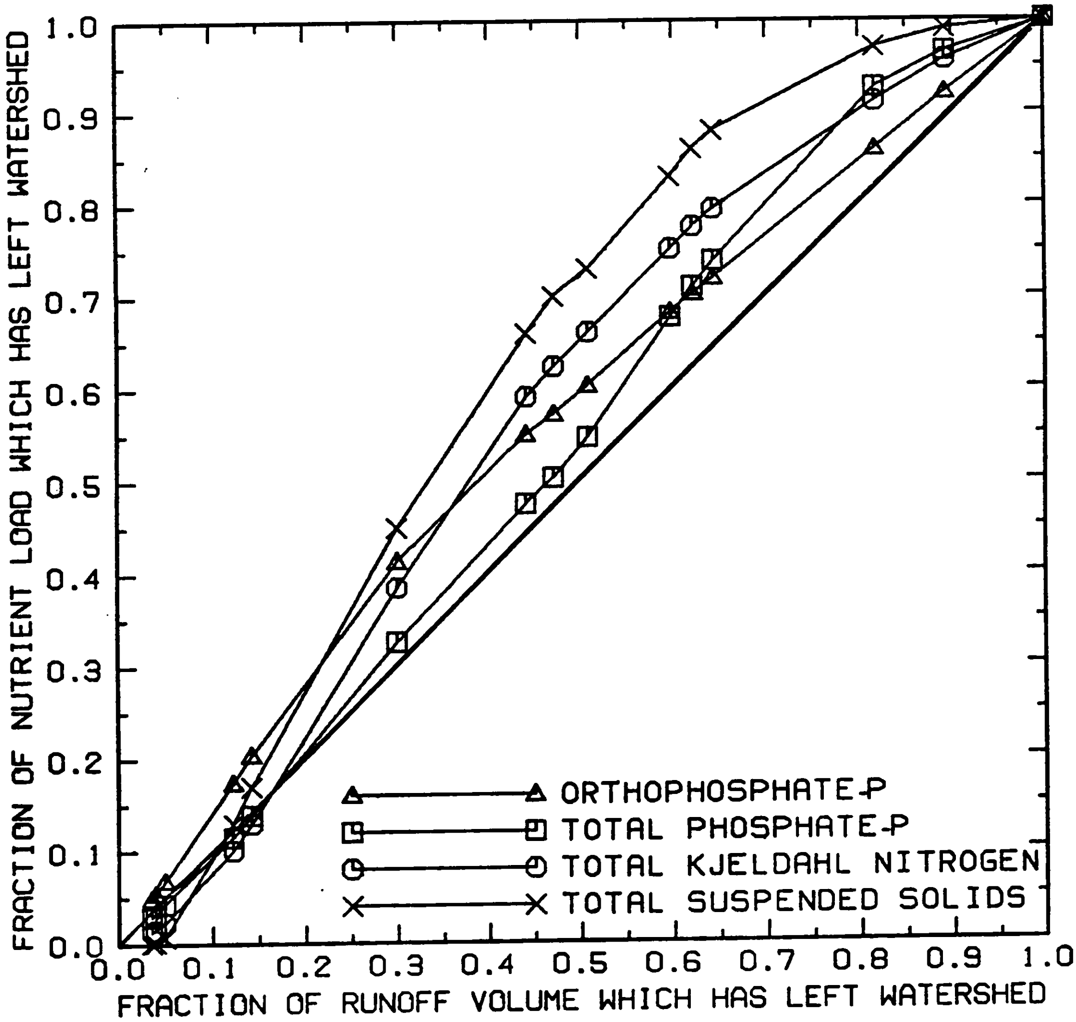

Figure 8. Nutrient loading graph for the May 9, 1988 storm event on the southeastern tributary.

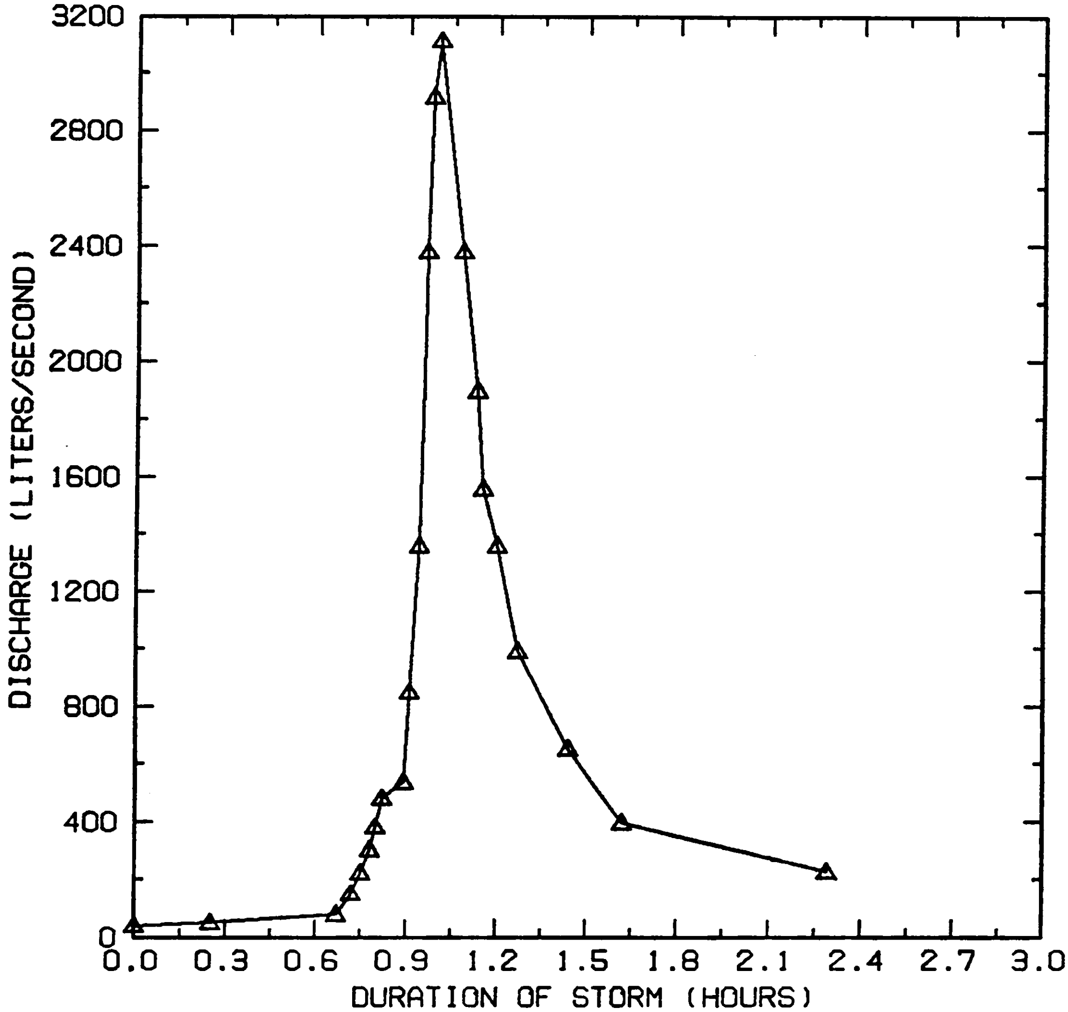

Figure 9. Hydrograph for the June 2, 1988 storm event on the southeastern tributary.

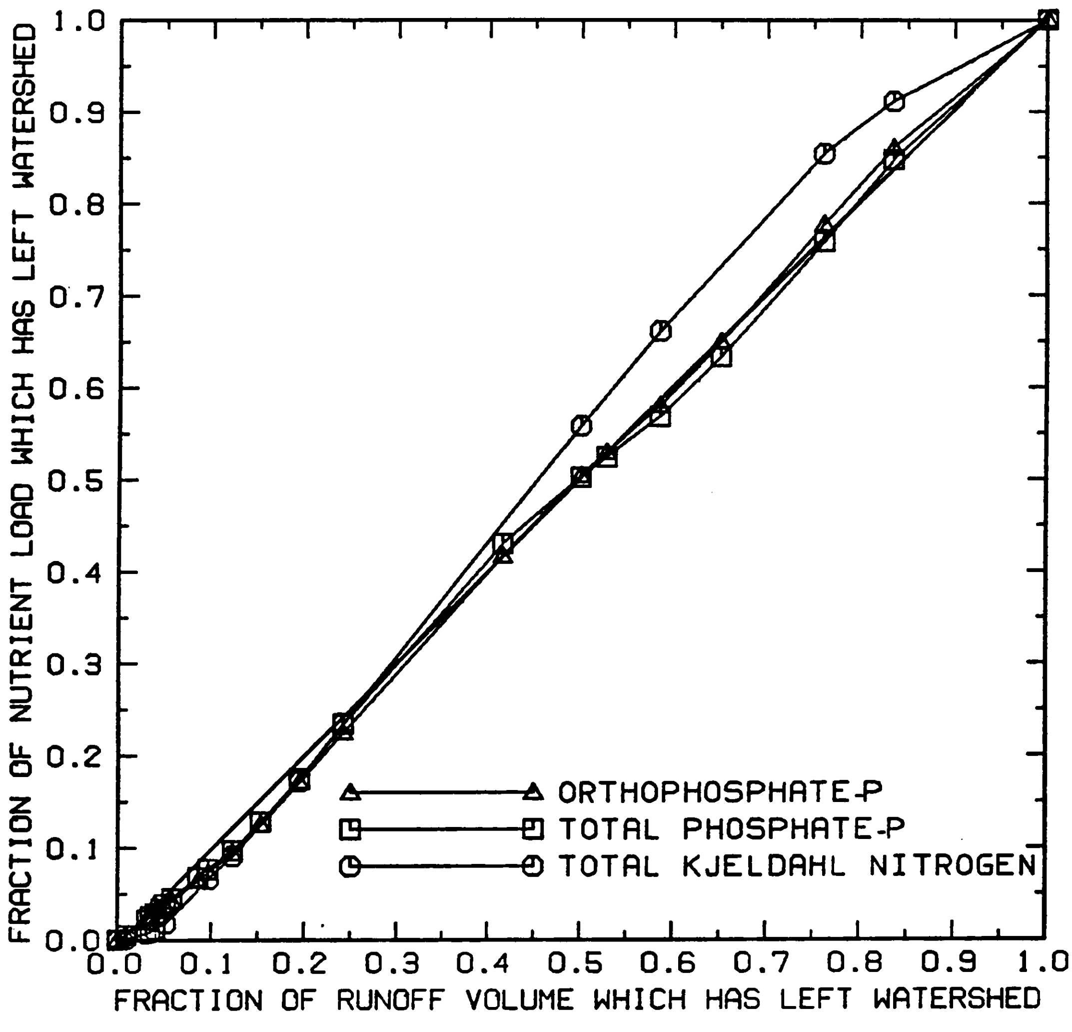

Figure 10. Nutrient loads for the June 2, 1988 storm event on the southeastern tributary.

conditions (Table 2). Together, the three storms contributed approximately 5,121 grams (g) of TP, 2,974 g of OP, and 95,202 g of TKN to the lower pond through the one tributary that was monitored. However, the streams may also contribute nutrient loads to the ponds during base-flow periods. Although base-flow nutrient loadings were consistently low (less than 10 *u*g/L), these loadings may represent a source of nutrients over a long period of time. The base-flow contribution of OP in the southeastern tributary was estimated to be 7.8 Kg/yr using 10 *u*g/L as a base-flow nutrient load. Again, it must be emphasized that only one tributary was monitored; therefore, the actual mass of nutrients entering the lower pond during storm events and baseflow conditions is probably larger based on obervations of Stroubles Creek during storm events.

Griffin *et al.* (13) reported that nutrient fluxes can dramatically differ from storm to storm in the same watershed due to rainfall intensity and duration, intervals between storm events, and the time of the year. Runoff and low-flow loads are also dependent on the day-by-day activities within the watershed. On more than one occasion, the tributaries would change colors because of some upstream activities. Once, the southeastern tributary turned milky white for about three hours. The white color might have been resulted from someone cleaning latex paint buckets and brushes in a pipe that emptied into the tributary. Due to the variability of storm events, the results of will be discussed collectively.

The southeastern tributary did not produce any major flushes in the early portions of the storm events monitored. One explanation for this response could be that the tributary to the lower pond is predominately a network of concrete drain pipes throughout the watershed. A discernible streambed only occasionally appears. At first, a pollutant flush during the early portion of a storm event would seem probable because storm drains are normally installed to remove runoff quickly. Nevertheless, a network of pipes may mask the results of a natural first-flush response. Runoff from areas within the watershed that would normally produce a flush may be mixed in small, upstream, settling wells before the water is released, thereby delaying and minimizing any first-flush responses of nutrients or other pollutants. Even though a large first-flush did not occur, cumulative runoff nutrient loads were significant sources of fertilizer nutrients to the lower pond.

The nutrient loading graph for the May 5, 1988 storm event indicated that 61 percent of the OP was contained in 51 percent of the total storm volume carried by one tributary. The implications of these data is that only 51 percent of the total storm volume delivered by this tributary would have to be contained in order to remove 61 percent of the runoff nutrients. Of course, a separate study would have to be conducted to obtain similar information regarding runoff delivered by the other tributary.

The lack of sediment in the southeastern tributary may be attributed to the concrete drainage system. Because the majority of the stream flows through concrete pipes, an alluvium bedload sediment source is not available. Thus, total suspended solids concentrations are negligible.

One aspect of water quality not shown in the runoff data is the presence of large quantities of organic matter in the form of leaves, grass clippings, limbs, and other plant debris. A large amount of urban debris, such as cans and paper, was also present in the runoff. During some storm events, velocity measurements were continually disturbed because large quantities of leaves, branches, and grass clippings collected on the cups of the velocity meter. Large amounts of allochthonous organic matter have already began to decrease the volume of the lower pond in the area of site 3 since the pond was last dredged.

Although Stroubles Creek, the tributary to the upper pond, was not monitored, it appeared to contribute more solids to the ponds than did the southeastern tributary. Unlike the southeastern tributary which is mostly a network of concrete pipes, Stroubles Creek has a distinguishable stream bed consisting of alluvial sediments; therefore, it has a readily available sediment source when stream velocities increase during storm events.

The Town of Blacksburg installed rip-rap along several upstream sections of Stoubles Creek in 1988 to prevent stream widening. As a result, total solids concentrations should be reduced. A small detention basin was built in 1988 on Stroubles Creek, in the Survey Park area of the VPI&SU campus, as a best management practice (BMP) to reduce solids loads to the upper pond. The pond is ineffective in removing solids because it is too shallow. As with most urban settings, the depth and size of a BMP is limited because of an extensive underground network of utilities surrounding the pond.

Pond Water Quality

As shown earlier in Figure 2, both ponds are shallow. General pond water quality data are summarized in Appendix B. In early April of 1988, the lower pond began to stratify thermally (Figure 11). When the pond became stratified, the oxygen in the colder bottom layers became depleted. Anoxic conditions promote the release of OP, which is the readily available and growth-limiting nutrient form for algae, from the bottom sediments. However, as will be shown, there were only slight differences between surface and bottom concentrations of phosphorus in the lower pond. The lower pond is easily destratified by wind action and runoff entering the pond as shown in Figure 12. In early

October of 1987, site 1 was thermally mixed and slightly mixed with respect to oxygen. By early November, site 1 had become stratified again with respect to oxygen.

Figures 13, 14, and 15 illustrate OP variations throughout the period of the study. The variations reflect periods of mixing and stratification. The fluctuations of TP and TKN in the lower pond during the study period may be found in Appendix B. These figures will be discussed further in the next subsection.

The algae growth-limiting OP concentration (0.02 mg/L) was frequently exceeded in the pond water column, and the data indicate that internal cycling of nutrients could provide enough phosphorus to support algal blooms. Table 3 lists the sediment phosphate concentrations at all sampling sites. The sediment at site 3, the area most often infested with nuisance algal blooms, had the largest sediment phosphate concentration. This could be attributed to both of the tributaries entering the lower pond in the area of site 3. Nutrients associated with particulate matter in runoff settle in the area of site 3 only to be released later during anoxic periods.

Hendricks and Silvey (76) reported that bottom sediments in shallow areas may increase productivity in those areas, though they often do not affect productivity in the deeper areas of the same lake. This was the case in the lower pond. Algal blooms were frequent in the shallow area of the lower pond (site 3), while lower algal densities were found continually in the deeper areas (sites 1 and 2).

Besides the sediments, other factors also increase productivity in shallow areas. For example, the temperature is usually greater in shallow areas, and higher temperatures increase the growth rate of some algal species. Algae rapidly assimilate available nutrients and experience periods of excessive growth often called algal blooms. When they

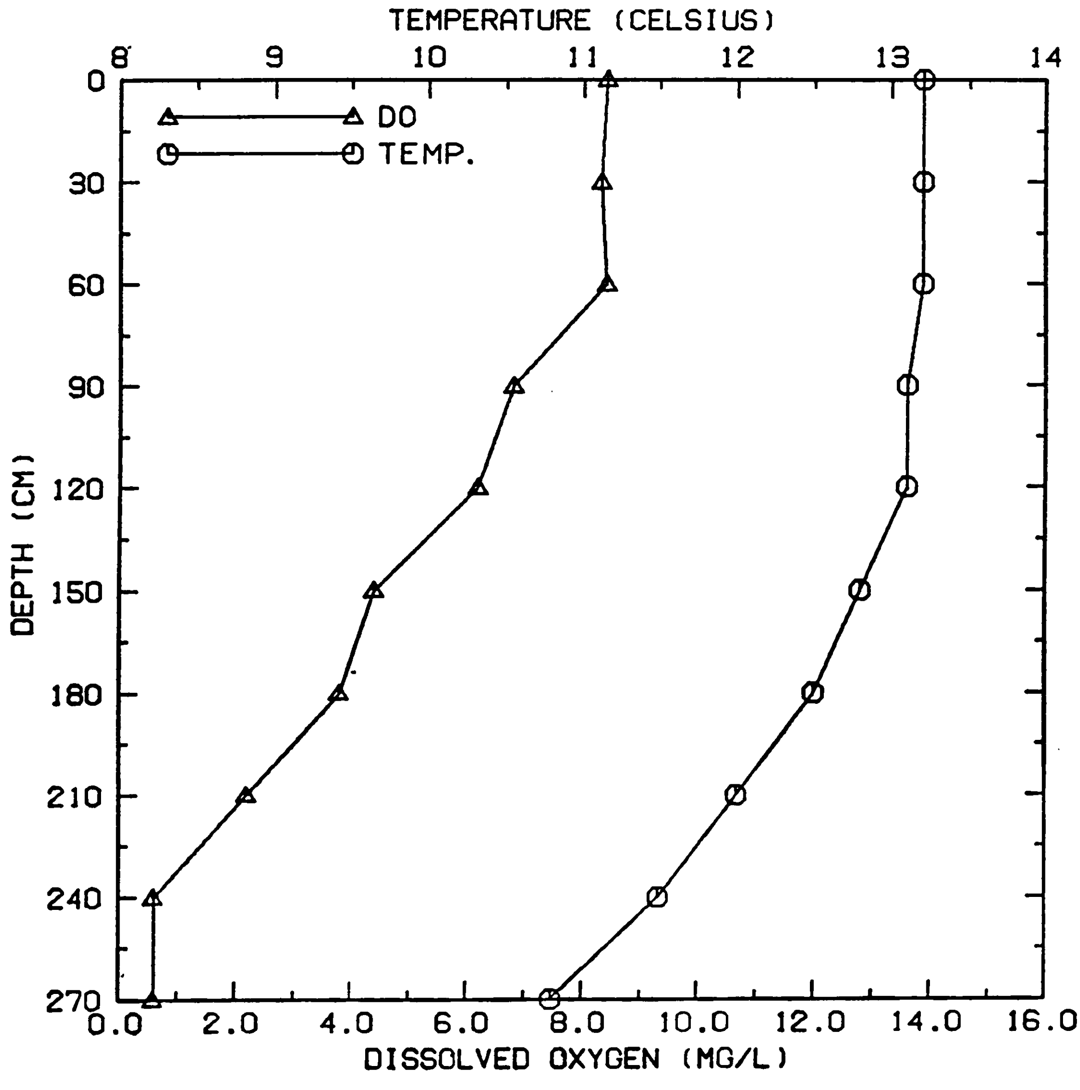

Figure 11. Oxygen and temperature profiles at site 1 in the lower pond on April 26, 1988.

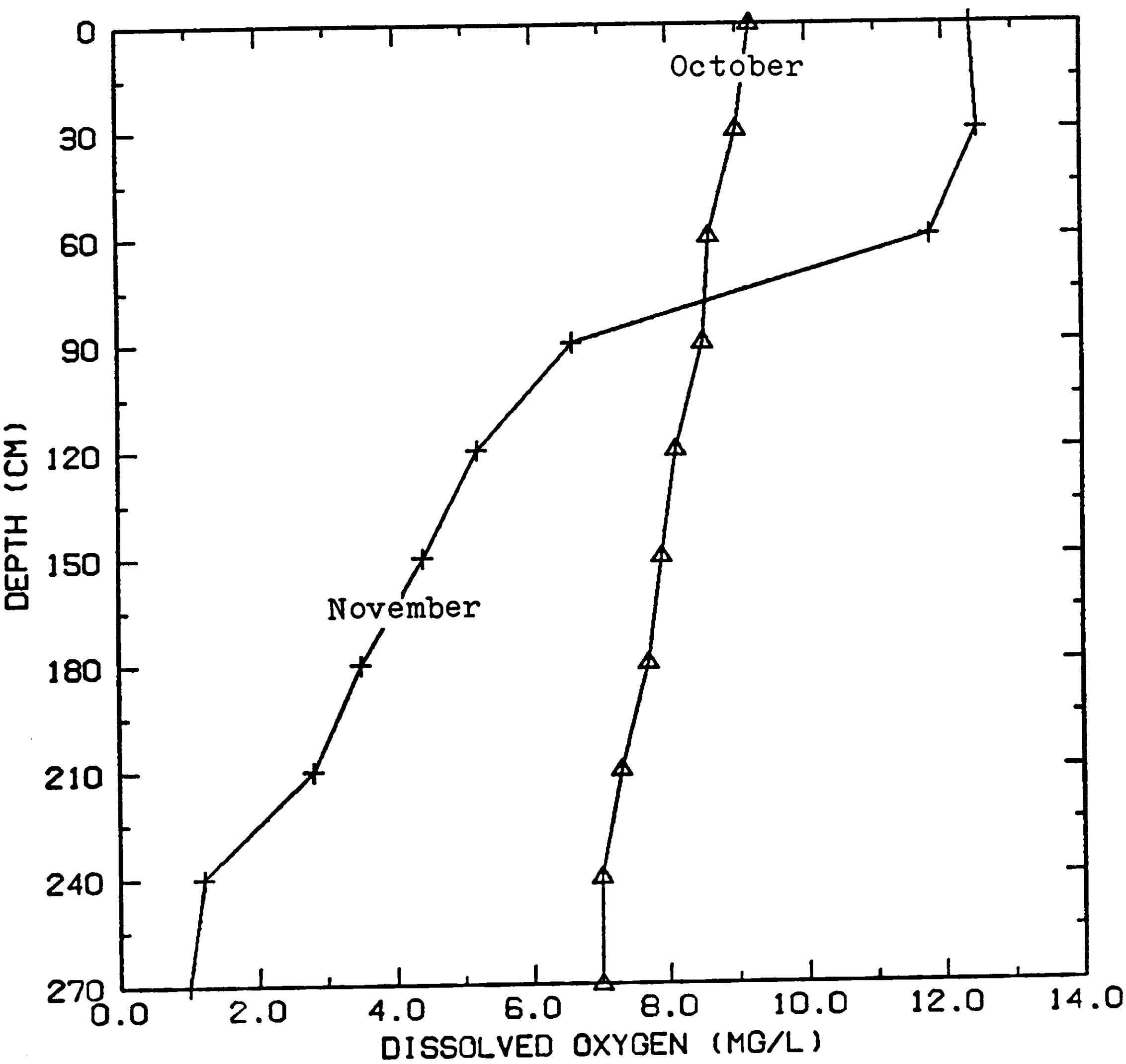

Figure 12. Dissolved oxygen variations with depth within a two month period.

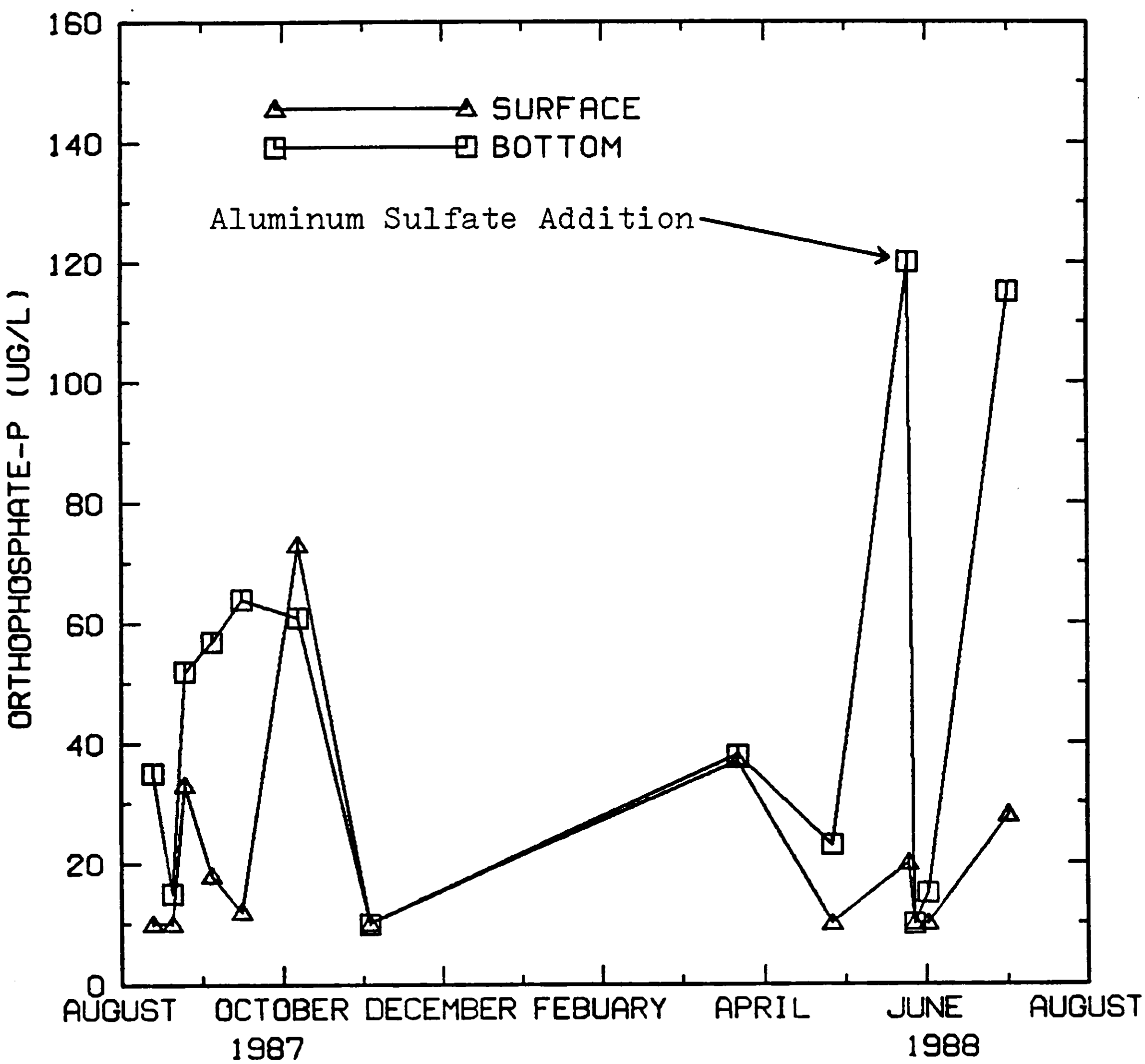

Figure 13. Variations in orthophosphate-phosphorus concentrations during the study period at site 1.

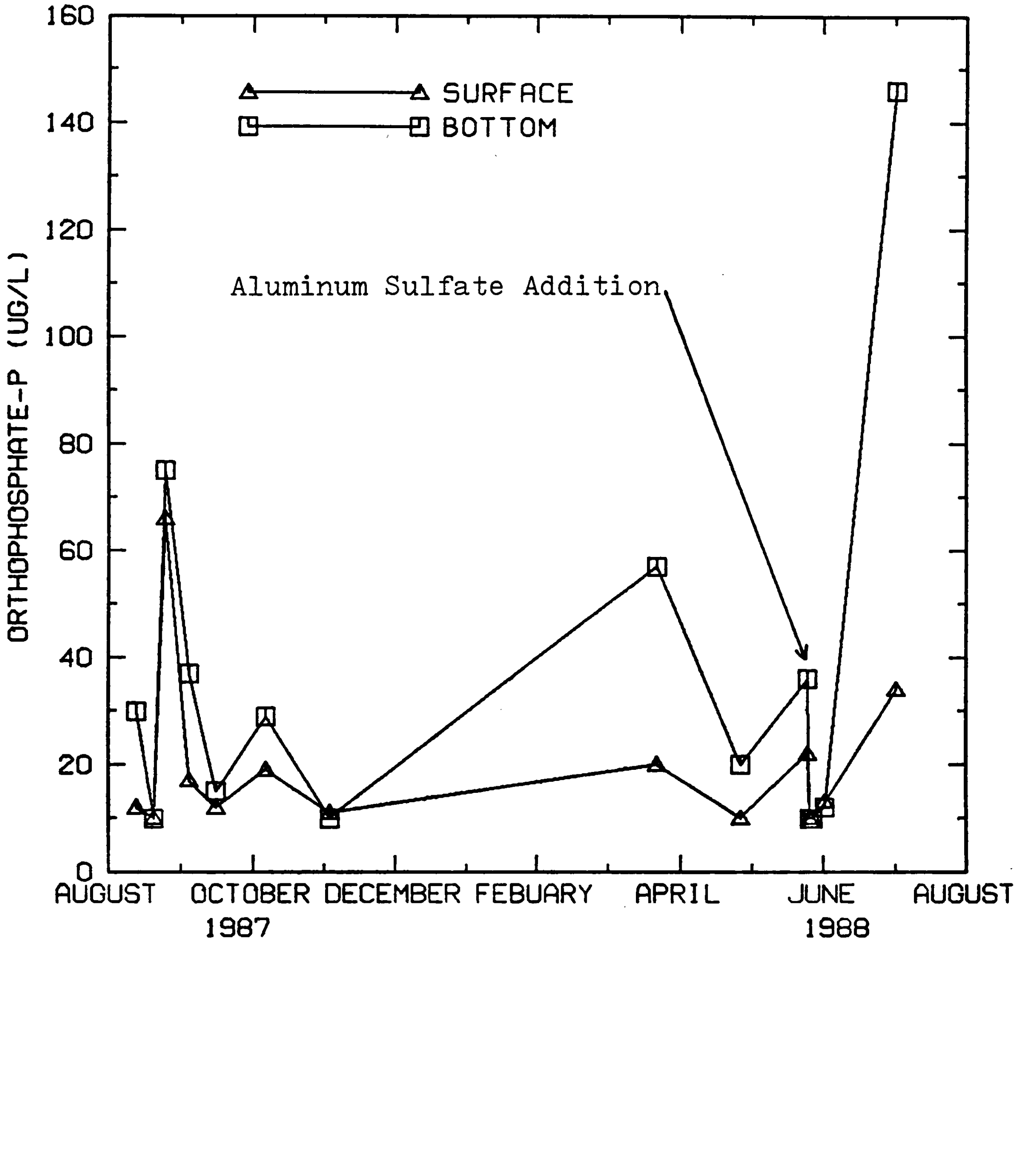

Figure 14. Variations in orthophosphate-phosphorus concentrations during the study period at site 2.

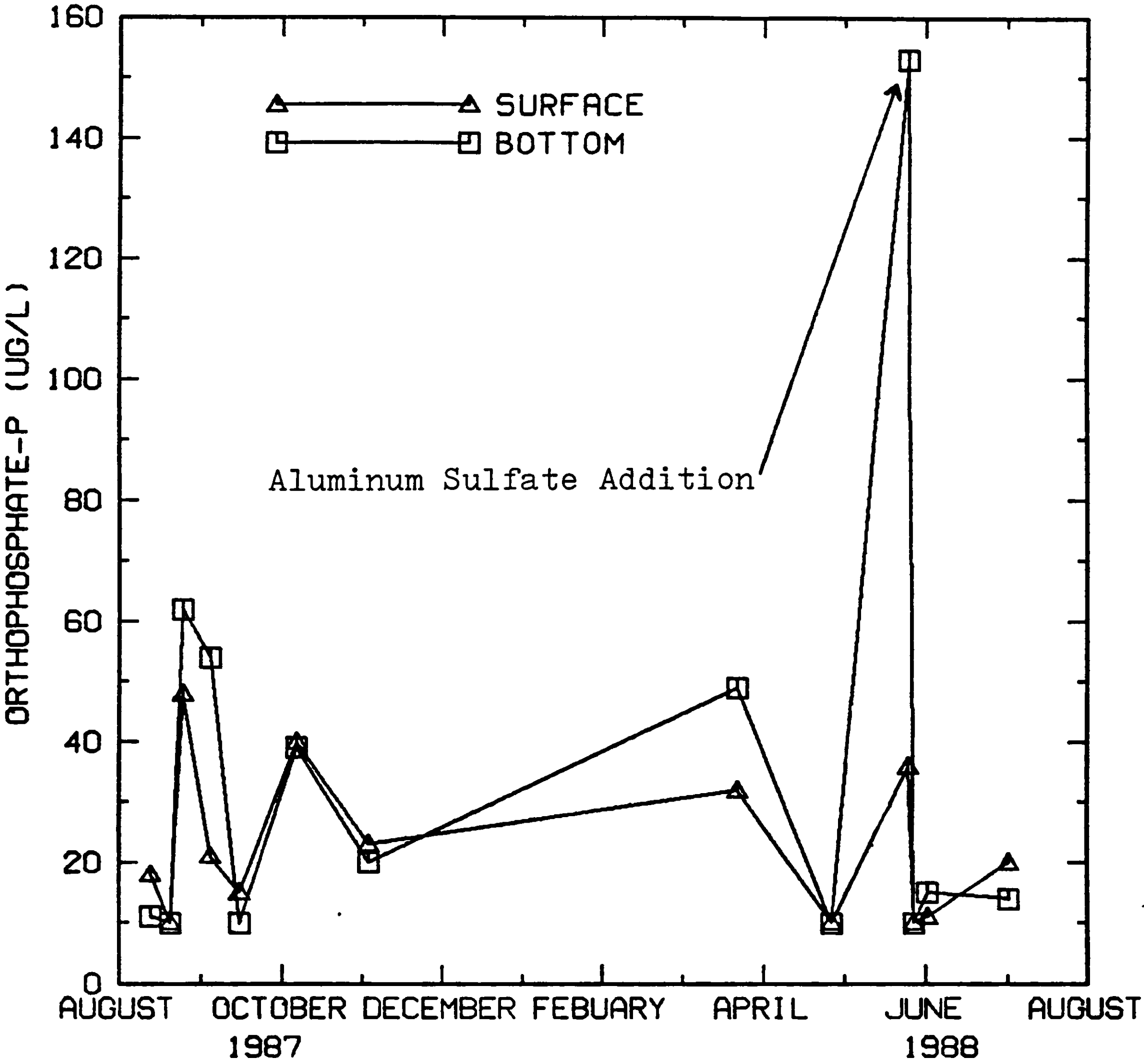

Figure 15. Variations in orthophosphate-phosphorus concentrations during the study period at site 3.

Table 3. Phosphate concentrations in the bottom sediments at four sampling sites.

DATE	PHOSPHORUS CONCENTRATION IN POND SEDIMENTS[a]			
	Site 1	Site 2	Site 3	Site 4
3/10/88	0.77	0.70	0.85	0.55
5/16/88	0.80	0.78	0.85	0.63

[a] Concentrations listed in mg P/gram of sediment.
See Figure 2 for location of sampling sites.

die, they settle settle to the bottom of the pond where they decompose. The nutrients are then recycled and continue to stimulate algal growth.

Algal blooms create large diurnal fluctuations of DO. Figure 16 illustrates the variations over a 24 hour period at site 3. The surface DO concentration was supersaturated by 99 percent in the late afternoon. The surface dissolved oxygen concentration had decreased to only 77 percent of saturation by early morning due to algal respiration.

The DO fluctuations were similar at Site 1 in most respects. Early morning DO values were only 63 percent of saturation, while late afternoon DO values were supersaturated by 68 percent. The DO at Site 1 differed from that at site 3 in one respect, namely that DO was nonexistent at a depth of 1.0 meter in the early morning (Figure 17). During this period, approximately 37 percent of the lower pond volume was void of oxygen. The remaining 63 percent contained oxygen but also was the warmest layer of the lake.

The large DO fluctuations in the lower pond were a result of large populations of the green alga, *Chlamydomonas*. Other algal species identified in the lower pond included *Ankistrodesmus, Euglena, Chlorella, Gonium,* and *Volvox*. Figure 18 shows the algal growth patterns and the effects of copper sulfate additions during the study period. The copper sulfate addition procedure will be discussed in the next subsection.

The large DO fluctuations could have a damaging effect on the fish population within the ponds. The Virginia Tech American Fisheries Society and the Wildlife Society conducted fish and duck population studies and they reported that the pond contained large numbers of bluegill, but only a few bass. Surprisingly, carp were found in large numbers.

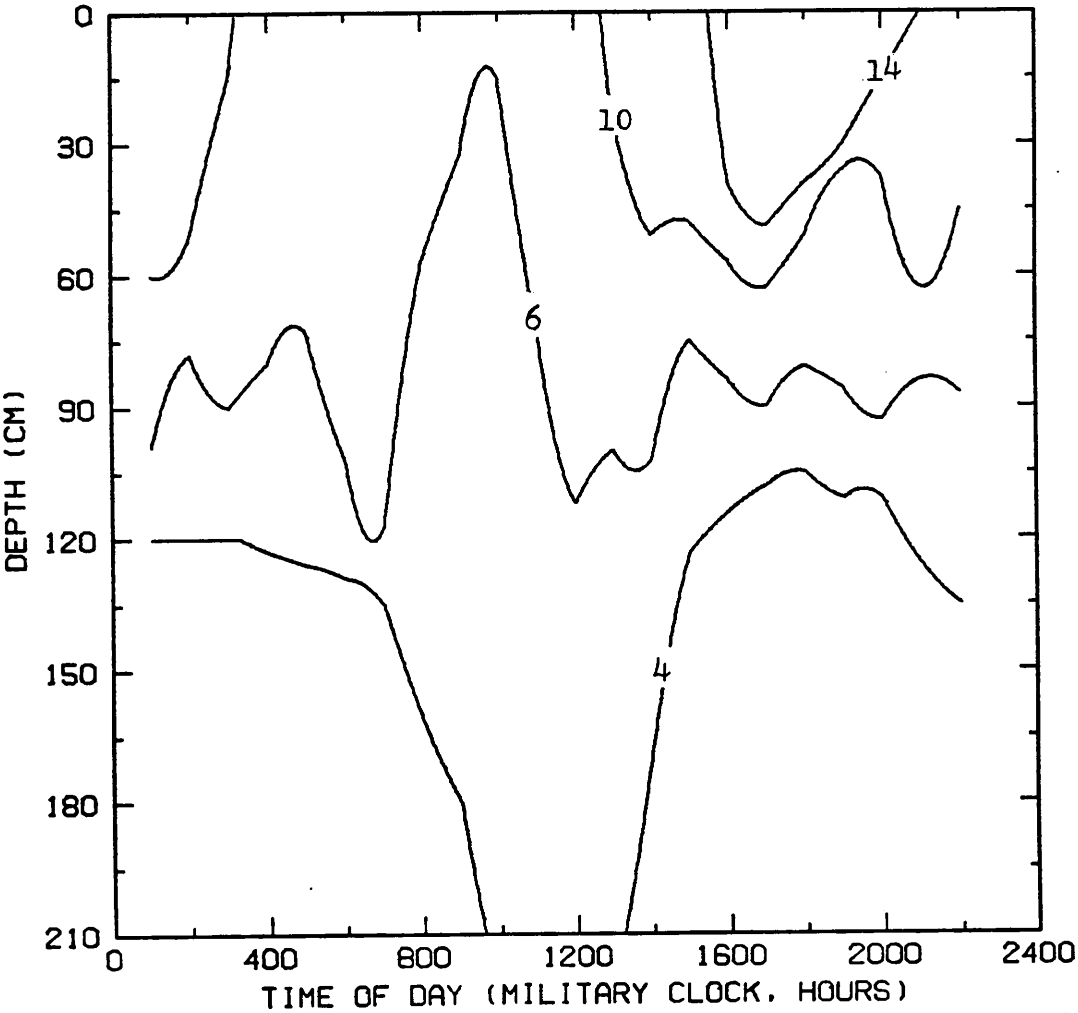

Figure 16. Diurnal variations of dissolved oxygen concentrations at site 3 on September 15, 1987.

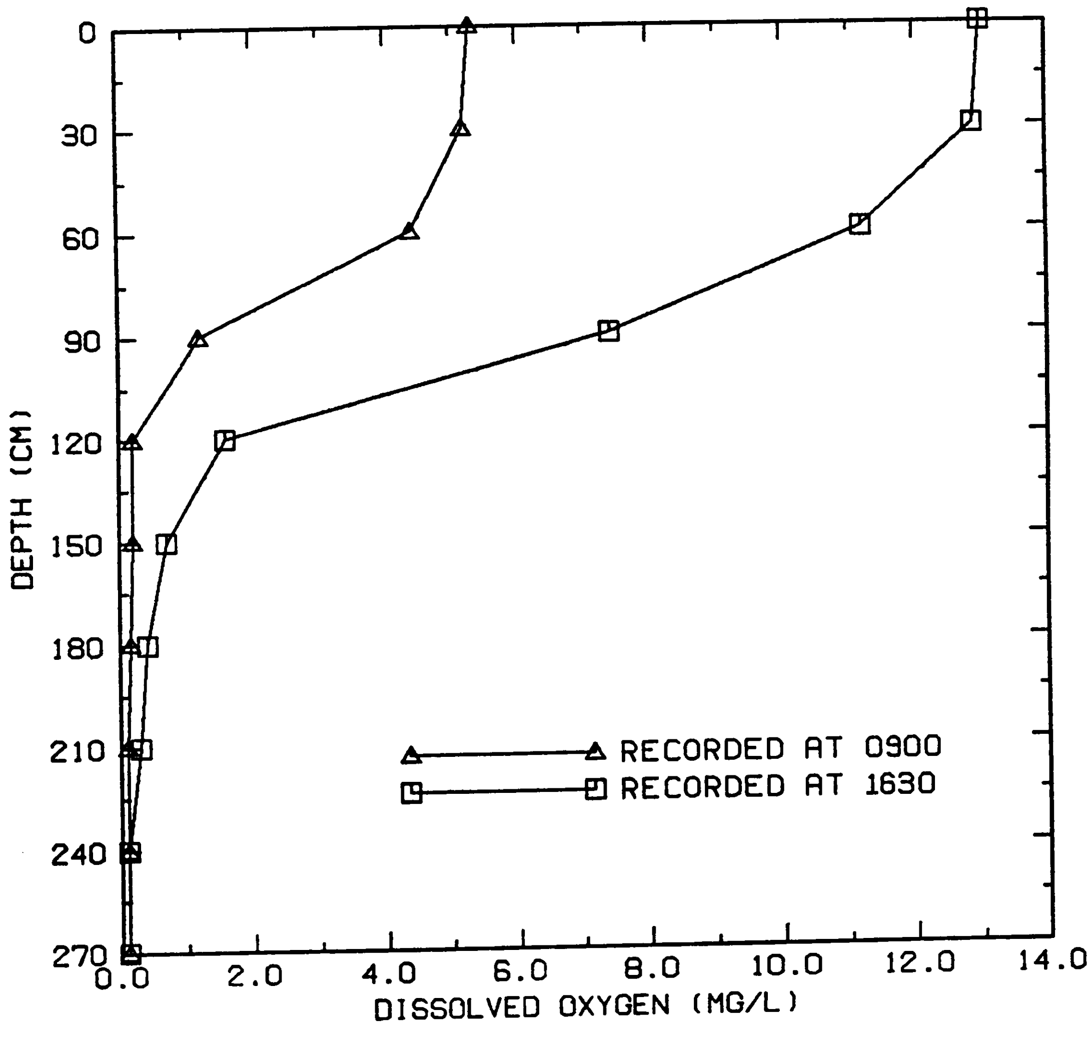

Figure 17. The highest and lowest diurnal dissolved oxygen profiles at site 1 on September 15, 1987.

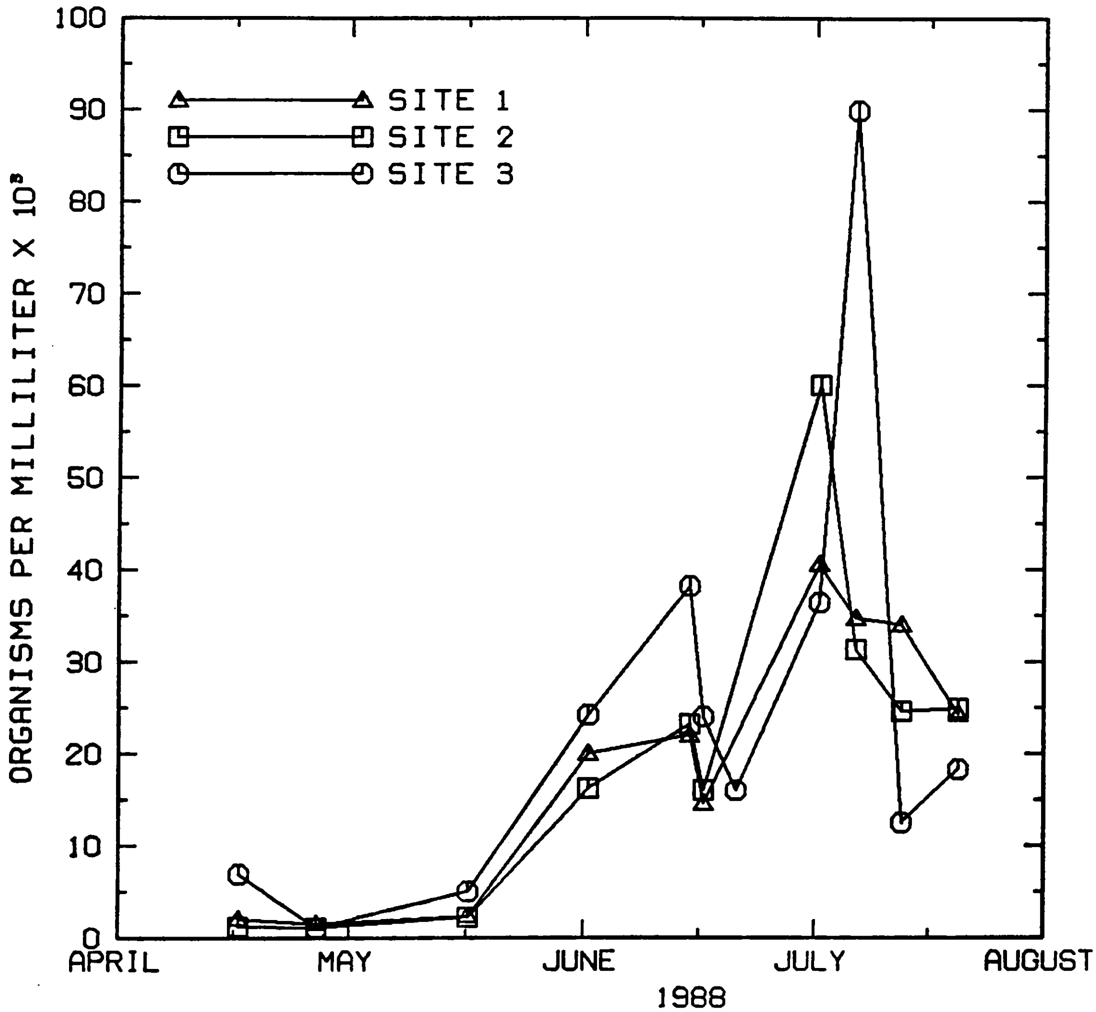

Figure 18. Algal populations in the lower pond during 1988. Thirty pounds of copper sulfate were added on June 15 and fifteen pounds on July 7.

Carp are known as a "rough" fish, which are capable of withstanding severe DO fluctuations and high water temperatures. Cooke *et al.* (30) reported that carp may result in reduced populations of game fish species because of interference with spawning or changes in game fish biomass. Grass carp have also been reported to increase nutrient recycling in some lakes (30). Nevertheless, grass carp have been effective in reducing aquatic vegetation in some lakes. The Duck Ponds, however, has had no documented problems with higher plants, and it is likely that the grass carp did not survive.

Total waterfowl, consisting mainly of mallards and Canadian geese, increased 67 percent over a two month period in 1988 (71). The total organic loads to the ponds, contributed by the waterfowl, is minor in that their organic waste composition is low (0.045 kg O_2 consumed/duck/day) (77). Thus, the organic load from the ducks most likely is inconsequential. The waterfowl cause aesthetic problems, however, because their feces covers the grounds surrounding the ponds and pose a possible contamination threat to small children who visit the ponds to feed the ducks.

Management Schemes Utilized

The OP, DO, and sediment phosphate data indicated that conditions in the lower pond were favorable for the internal cycling of phosphorus. Treatment with aluminum sulfate (alum) was chosen as the pond management technique to retard the internal cycling of phosphorus. As mentioned earlier, OP in the water column and internal cycling of OP from the sediments has been reduced in many lakes after adding aluminum sulfate (30).

Aluminum sulfate dosage procedures developed by Kennedy (36) were utilized to arrive at a safe, effective aluminum sulfate dose. Some researchers and lake managers are wary about adding a potentially toxic substance to a lake. Kennedy (36) addressed this problem by basing the alum dose on the alkalinity of the pond or lake instead of on the phosphate concentration in the lake. This type of dosage procedure was designed to produce a concentration of less than 0.05 mg Al/L, which is a level that has been shown to be harmless to aquatic life. Figure 19 illustrates the relationships of pH, alkalinity, and ionic aluminum as a function of the alum dose. Alum can be added to a lake without any adverse changes in the pH or ionic aluminum concentrations, the recognized toxic form, if the dose is based on the alkalinity of the water. This type of dosage procedure assures that the aluminum ion will not become soluble because the alkalinity will maintain pH values above a pH of 6.0, the pH at which aluminum becomes soluble.

A dose of 25 mg alum/L produced a floc that settled to the bottom of the jar within one hour. A sufficient floc size is desired so the floc will settle to the bottom of the pond and cover the sediment, thereby retarding OP release from the sediments. This dose lowered the pH from 8.1 to 7.7 in a water sample from site 1. Figure 20 shows the results of the alum doses in the lab. The alkalinity was lowered less than 15 mg/L at the 25 mg alum/L dose. Soluble aluminum remained below detectable limits of 0.1 mg Al/L in all the sample mixtures because the pH of the mixtures remained above a pH of 6.0, the pH where aluminum becomes soluble.

OP concentrations in the water column decreased below 20 ug/L at all sites monitored as shown in Figures 13, 14, and 15. TP in the water column was also lowered. The exact removal processes surrounding TP removal in the water column are unclear at the present time. Alkalinity in the treated pond remained above 160 mg/L throughout and after the treatment. As a result, soluble aluminum concentrations remained below the detectable limits of 0.1 mg/L. The increase in Secchi disk values, as shown in Figure 21,

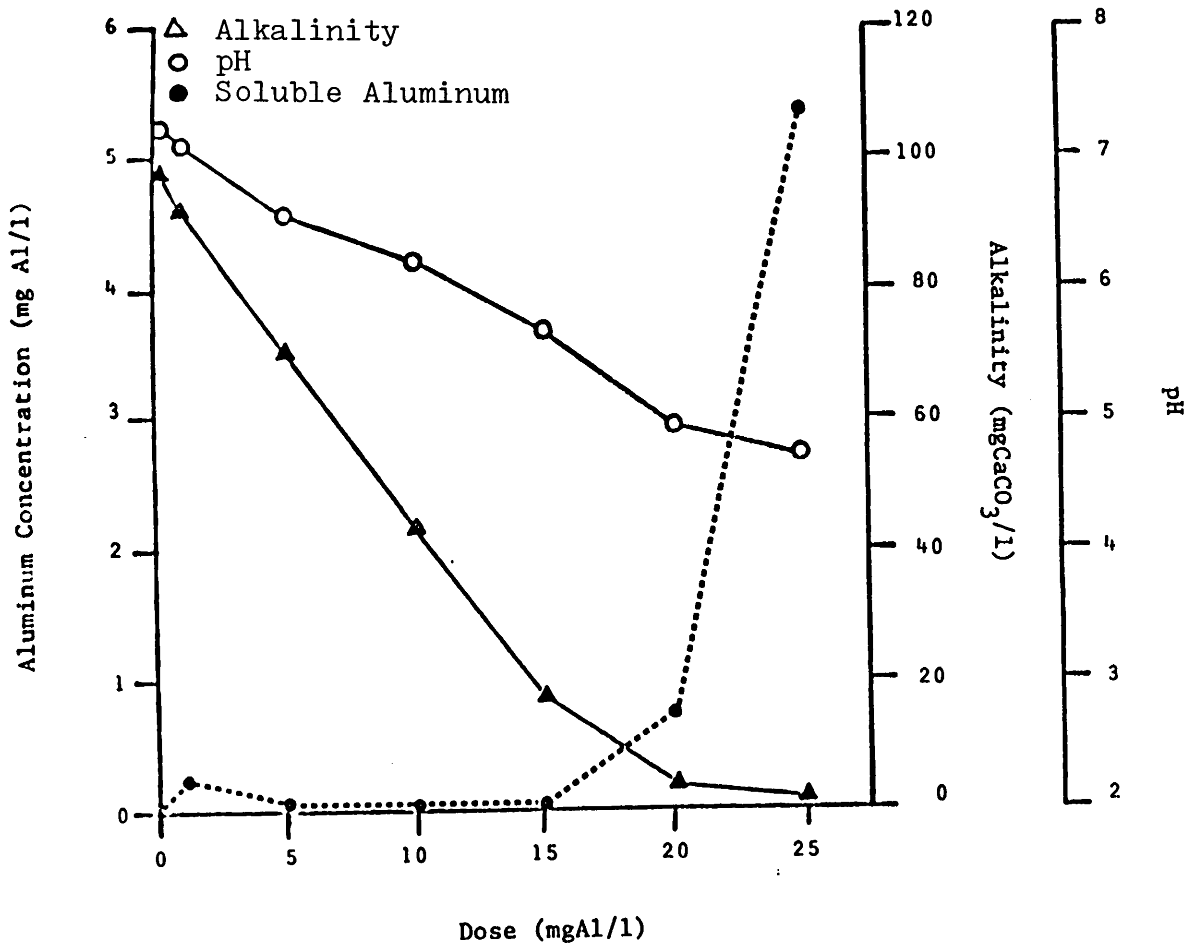

Figure 19. The effects of increased alum doses on alkalinity, pH, and ionic aluminum concentrations (from Kennedy (36)).

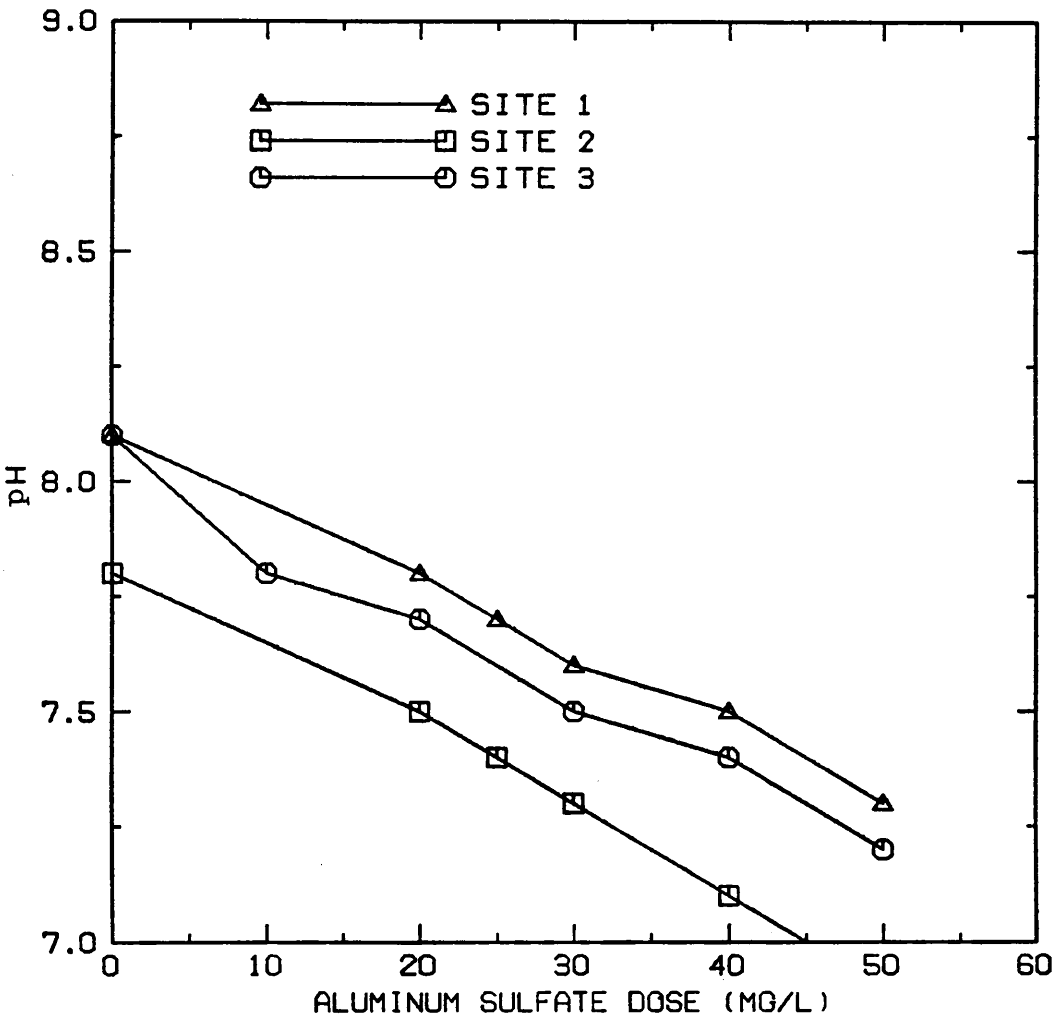

Figure 20. Changes in pH after alum was added to pond samples.

indicates the formation and subsequent settling of the aluminum hydroxide floc, producing a more transparent pond.

The long-term effectiveness of alum treatments in retarding OP release from sediments in shallow ponds may be lessened because the ponds frequently mix. The OP removal effectiveness depends on a well-distributed aluminum floc covering the bottom sediments. Therefore, if a pond mixes in a turbulent manner, the aluminum floc may be redistributed and concentrated in one area of the pond, thus reducing the effectiveness of the treatment. As is shown in Figure 13 and 14, the OP concentration in the bottom of the lower pond at sites 1 and 2 increased dramatically in early July. In contrast, the OP concentration at site 3 did not increase. Instead, the OP levels near the bottom at site 3 remained low for one month following the alum treatment, despite the lack of oxygen in the bottom water layers. As can be seen from Figure 13, bottom OP levels were markedly higher in the late part of June, possibly caused by the release of phosphorus from dead algal cells killed by the June 14 copper sulfate treatment. The extent of the long-term effectiveness of the one-time treatment is unknown because the monitoring program was not continued beyond July 1988.

The second management scheme was aimed at directly reducing algal densities by applying copper sulfate to the pond. The copper sulfate dosage determination method followed was devised by Mcknight (63).

The changes in Cu with increased doses of copper sulfate to pond samples in the lab are shown in Figure 22. Ionic copper, considered to be the most toxic form, was measured after copper sulfate was added and the mixture stirred for seven minutes or until the meter-reading stabilized. Ionic copper levels remained low because it reacted with the alkalinity to produce insoluble copper species such as $CuCO_3$ and $Cu(OH)_2$. Both

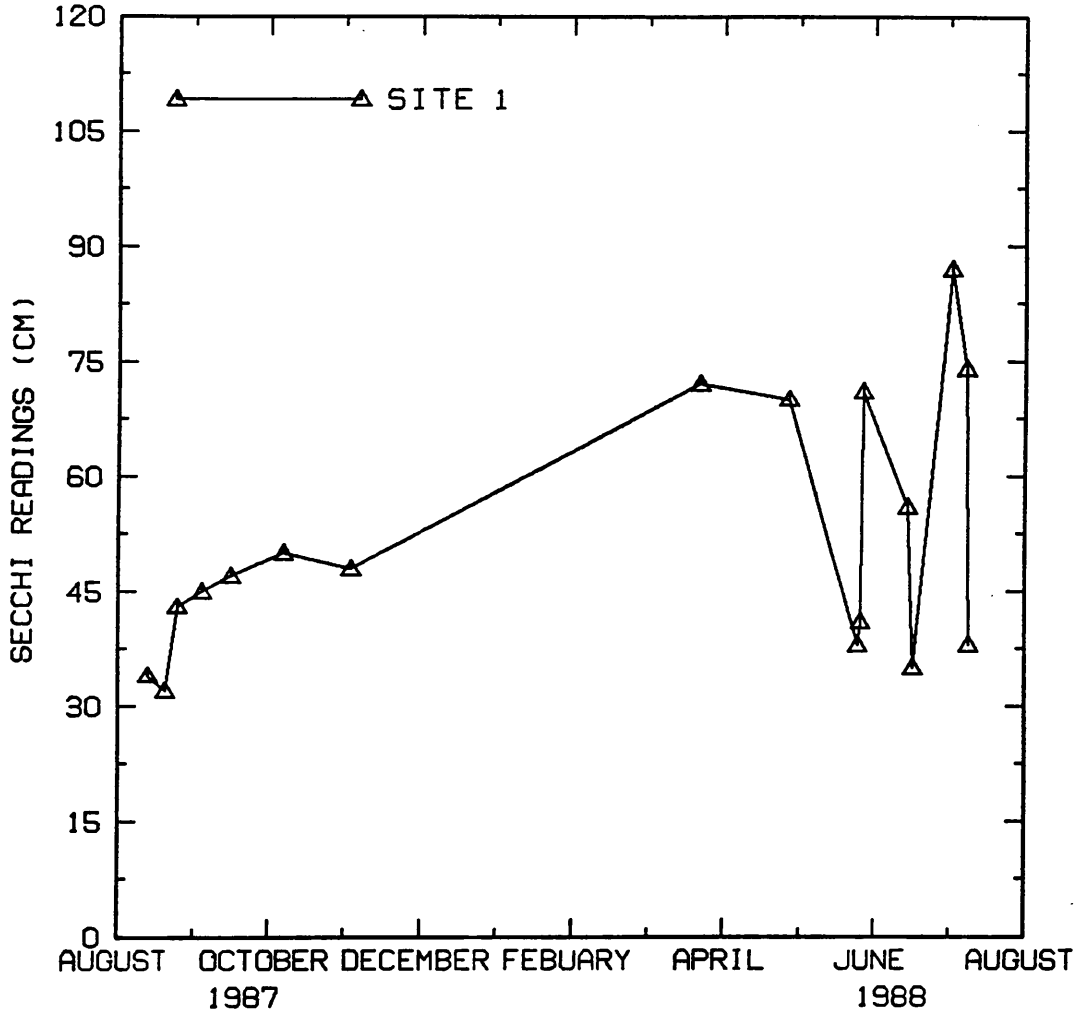

Figure 21. Secchi disk values during the study period at site 1. Aluminum sulfate was added on May 24 and 25.

of the copper sulfate treatments resulted in major reductions of algal populations (Figure 18).

Although copper dose determinations are important, proper timing of the dose is of equal importance. Applying copper sulfate in the midst of an algal bloom could result in the consumption of DO, potentially killing fish, as a result of decaying algae. Nevertheless, applying copper sulfate when algal populations are too low represents a waste of resources. For this reason, algal growth should be monitored during the spring, summer and early fall months to see if a growth pattern is evident; therefore, copper sulfate could be applied in the early growth stages of a algal bloom, preventing DO fluctuations with only a minimum amount of monitoring. Until a large algal growth data base is recorded, algae should be monitored weekly during the previously mentioned months and copper sulfate should be applied when data indicate the algal populations are increasing. The prescribed copper dose of 13.6 kg (30 pounds) to the lower pond was effective in reducing the green alga, *Chlamydomonas*. As the pond becomes enriched with nutrients, other algal species that are more resistant to copper may develop, thus requiring larger copper sulfate doses.

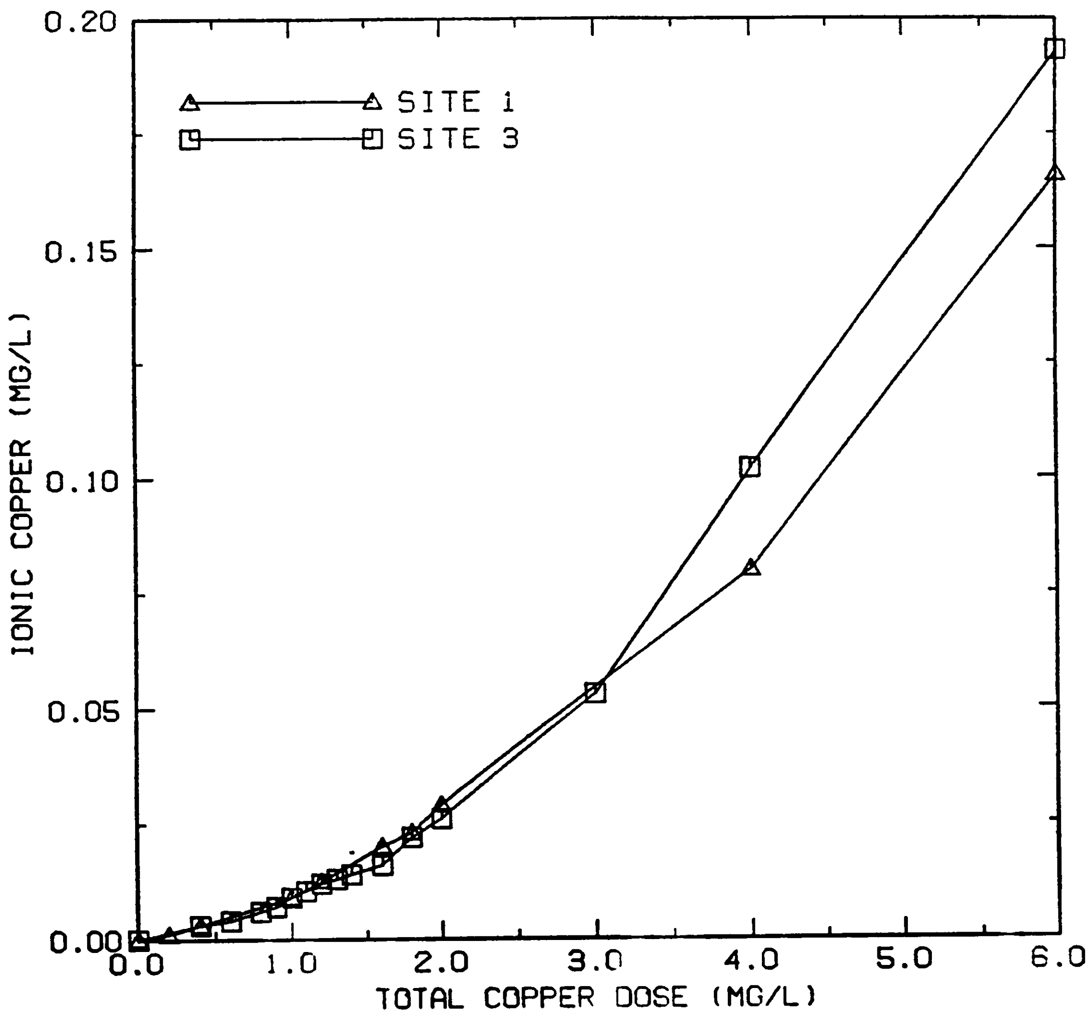

Figure 22. Variations in ionic copper concentrations with increasing copper sulfate doses in pond water from two surface sites.

SUMMARY AND CONCLUSIONS

The VPI&SU Duck Ponds were monitored for allocthonous and autochthonous nutrient loads. Nutrient loads in the southeastern tributary, the tributary to the lower pond, were monitored during three storm events. Nitrogen and phosphorus concentrations were also determined throughout the lower pond to evaluate the extent of internal nutrient cycling. A bathymetric map of both ponds was developed from which the volume of each pond was determined. Algae were monitored during the spring and summer of 1988 to determine the effectiveness of two management schemes: alum treatment to reduce phosphorus levels and copper sulfate to control algae.

Aluminum sulfate was added to the lower pond to complex orthophosphate-phosphorus in the water column. The aluminum sulfate floc also functions as a sealant over the bottom sediments, thus reducing the rate of phosphorus release from the sediments during anoxic periods. A copper-sulfate-dosing procedure was developed to reduce algal densities. No special equipment was required for either the alum or copper sulfate treatments, thereby keeping costs at a minimum.

The following conclusions seem warranted from an analysis of the data:

1. Runoff from the southernmost tributary contributes large amounts of organic matter and nutrients during storm events. Large percentages of these nutrients and organic matter could be reduced by containing the early portions of storm events in an upstream impoundment.

2. Runoff nutrient loads and internal nutrient cycling triggers frequent algal blooms which cause large fluctuations in dissolved oxygen and create aesthetic problems.

3. Algal counts suggests that copper sulfate treatment is an effective method for reducing algal densities in the lower pond without adversely affecting aquatic fauna when the dose is based on proper chemical considerations of alkalinity and pH.

4. The waterfowl (ducks and geese) most likely are not the major sources of organic pollution in the ponds. Leaves, grass, and other organic debris entering the ponds during storm events are likely the major organic sources.

Before any further pond management decisions are implemented, a fundamental question concerning the purposes of the ponds must be answered. Without a list of purposes, management decisions often conflict one another. This perpetuates the eutrophication process, thus wasting time and money.

Currently, the ponds function as a best management practice for treating the runoff from the town of Blacksburg and the Virginia Tech campus. Use of the ponds as a runoff treatment device is not compatible with their use as a fishery because they are too small. Until the runoff loads are reduced, the ponds will continue to infill with sediment and organic matter. If no management program is implemented, water quality will continue to deteriorate. Dense algal blooms will continue, thus further adding to the level of organic matter in the sediment and creating problems with acute fluctuations in dissolved oxygen levels in the water column.

RECOMMENDATIONS

This section has been subdivided into short-term and long-range recommendations because of their distinct differences in terms of costs and effectiveness.

Short Term Recommendations

The copper sulfate program was shown to be an effective algal control method for the lower Duck Pond. In order to ensure continued success, cell counts need to be performed on a weekly basis during the spring, summer, and early fall. An undergraduate biology student would be a logical choice for the task.

For *chlamydomonas* blooms, copper sulfate should be added when cell counts exceed approximately 30,000 cells/milliliter. This cell count is only applicable to *chlamydomonas* because other algal species may produce a bloom at lower cell counts. Thus, it is essential to monitor algae growth and add copper sulfate during the early growth stages. Copper sulfate should only be applied to the areas where rapid algal growth is occurring. For instance, the area surrounding site 3 experienced an algal bloom on July 6, 1988.

Fifteen pounds of copper sulfate were applied to the upper one-third of the lower pond. This dose reduced the algal density but did not result in drastic dissolved oxygen reductions. It is imperative to monitor the algal growth so that copper sulfate can be applied before the algae proliferate. A copper dose in the early stages of a bloom retard the bloom but do not result in dissolved oxygen reductions that could stress fish.

Nevertheless, if the only management scheme is copper sulfate, copper tolerant species may develop that require a higher dose of copper sulfate. Although copper sulfate has been used extensively throughout the United States to control algae, it is not a panacea.

Another short term alternative is to add aluminum sulfate or ferric aluminum sulfate directly into the tributaries. Either of the compounds would slowly dissolve, releasing aluminum ions that would bind the phosphorus in the tributaries and flocculate the water, thus removing incoming silt and clays. The transparency of the ponds would increase, thus increasing the likelyhood of algal growth, but if the phosphorus is removed by the treatment, which it should be, the algae should be phosphorus limited.

Long Term Recommendations

Proper watershed management should address the long-term problems of sedimentation and eutrophication. Both of the short-term recommendations are effective, but they do not address the main problems; instead, they only disguise the main problems. Long-range management schemes are often shoved aside because they are initially very costly. In the long range, however, the short-term costs often exceed the long term costs.

In order to properly manage the Duck Ponds, the Town of Blacksburg and the University should form a joint committee that will share not only in the management decisions regarding the ponds, but also in the costs of the processes, because both benefit from the Duck Pond as a recreational resource.

Long-term watershed management programs should concentrate on the installation of a series of detention ponds in the upstream areas of the watershed and development of strict land-use regulations that would properly reduce the sedimentation problems. Routine collection of grass and leaf matter by the Town would further reduce organic inputs to the Duck Pond. These inputs, not the waterfowl, are the greatest contributors of organic matter entering the ponds.

Another option includes rerouting both of the tributaries around the ponds. Water from the springs on Stroubles Creek could be routed directly to the ponds to maintain proper water levels. This option would definitely remove the runoff problems but likely would be very costly.

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Appendix A. Runoff Water Quality Data

Table 4. Runoff water quality data for the October 27, 1987 storm event.

Sample Time	Total Time, in Hours	Discharge Liter/sec	Total-P in ug/L	Loading rate, in g/hr	Ortho-P in ug/L	Loading rate, in g/hr	TKN in mg/L	Loading rate, in g/hr	Total Solids, mg/L	Solids Loading rate, in g/hr
0830	0	25.5	159	14.60	81	7.44	0.7	64.26	-	-
1000	1.5	169.9	279	170.65	168	102.76	2.8	1712.59	-	-
1045	1.75	212.4	301	230.16	193	147.58	2.9	2217.46	<0.1	15.29
1130	2.50	218.1	321	252.04	205	160.96	2.8	2198.45	<0.1	31.41
1215	3.25	229.4	350	289.04	260	214.72	3.4	2807.86	<0.1	41.29
1254	3.90	325.7	397	465.49	271	317.75	4.3	5041.84	0.1	117.25
1400	5.00	487.1	321	562.89	194	340.19	5.0	8767.80	0.3	526.07
1430	5.50	662.6	447	1066.26	299	713.22	6.8	16220.44	0.5	1192.68
1454	5.90	708.0	270	688.18	150	382.32	7.3	18606.24	0.5	1274.40
1530	6.50	594.7	207	443.17	101	216.23	7.0	14986.44	0.1	214.09
1600	7.00	498.4	190	340.91	94	168.66	5.4	9688.90	0.1	179.42
1635	7.58	396.5	192	274.06	79	112.76	3.2	4567.68	0.1	142.74
1700	8.00	201.1	181	131.04	98	70.95	2.8	2027.09	0.1	72.40
1800	9.00	125.0	100	45.00	71	31.95	1.3	585.00	0.1	45.00

Table 5. Runoff water quality data for the May 9, 1988 storm event.

Sample Time	Total Time, in Hours	Discharge liter/sec	Total-P in ug/L	Total-P Loading rate, in g/hr	Ortho-P in ug/L	Ortho-P Loading rate, in g/hr	TKN in mg/L	TKN Loading rate, in g/hr	Total Solids, in mg/L	Solids Loading rate, in g/hr
1433	0	79.3	136	38.82	100	28.55	0.5	142.73	<.1	0.57
1449	0.267	169.9	287	175.56	178	108.88	4.2	2569.19	0.1	61.17
1450	0.283	223.7	241	194.10	169	136.11	4.6	3704.93	0.2	136.92
1452	0.317	376.6	346	469.16	128	173.56	6.6	8949.35	0.8	1098.32
1459	0.433	804.3	265	767.29	175	506.70	9.3	26927.56	2.5	7238.59
1500	0.450	1467.0	360	1901.20	159	839.70	12.9	68126.36	2.4	12674.67
1507	0.576	1079.0	335	1301.26	128	497.20	12.8	49719.95	1.9	7380.31
1515	0.700	917.6	280	924.91	82	270.87	10.1	33362.77	1.6	5285.19
1517	0.733	750.5	281	759.19	85	229.65	7.7	20803.30	1.2	3242.07
1520	0.783	634.4	443	1011.69	99	226.09	8.2	18726.54	1.3	2968.84
1529	0.933	489.9	395	710.80	95	167.56	7.8	13757.40	1.4	2469.28
1532	0.983	410.6	390	576.54	89	131.57	8.2	12122.09	1.1	1626.13
1535	1.033	410.6	385	569.15	88	130.09	5.6	8278.50	0.7	1093.94
1606	1.553	220.9	187	148.71	88	69.98	4.8	3817.08	0.4	318.09
1628	1.923	164.3	100	59.13	90	53.22	4.5	2660.95	0.2	188.26
1715	2.706	94.9	90	30.74	70	23.91	1.2	409.84	0.1	34.15

Table 6. Runoff water quality data for the June 2, 1988 storm event.

Sample Time	Total Time, in Hours	Discharge Liter/sec	Total-P in ug/L	Loading rate, in g/hr	Ortho-P in ug/L	Loading rate, in g/hr	TKN in mg/L	Loading rate, in g/hr
1857	0	39.9	100	14.36	84	12.07	1.0	143.64
1912	0.25	51.0	152	27.91	115	21.11	1.8	330.48
1937	0.67	79.3	216	61.66	145	41.39	2.9	827.89
1940	0.72	151.0	217	117.96	122	66.32	3.4	1848.24
1942	0.75	223.7	166	133.68	100	80.53	4.8	3865.54
1944	0.78	302.7	179	195.06	112	122.05	-	-
1945	0.80	381.8	225	309.26	131	180.06	6.7	9209.02
1946	0.82	481.4	210	363.94	122	211.43	-	-
1950	0.89	538.1	190	368.06	135	261.52	-	-
1951	0.91	849.6	189	578.07	141	431.26	9.2	28138.75
1953	0.94	1359.4	187	915.15	143	699.82	9.0	44044.56
1954	0.96	2378.9	292	2500.70	190	1627.17	-	-
1956	0.98	2917.0	257	2698.81	181	1900.72	12.5	131265.0
1958	1.00	3115.2	334	3745.72	200	2242.94	12.3	137941.0
2003	1.08	2378.9	189	1618.60	168	1438.76	-	-
2006	1.13	1897.4	229	1564.22	177	1209.02	11.2	76503.16
2007	1.15	1557.6	152	852.32	130	728.96	-	-
2010	1.20	1359.4	225	1101.11	168	822.16	10.3	50406.55
2014	1.27	991.2	257	917.06	192	685.12	-	-
2024	1.44	651.4	298	698.82	199	466.66	8.1	18994.82
2035	1.62	396.5	267	381.12	170	242.66	6.0	8564.41
2115	2.29	226.6	150	122.36	100	77.50	3.4	2773.58

Appendix B. Pond Water Quality Data

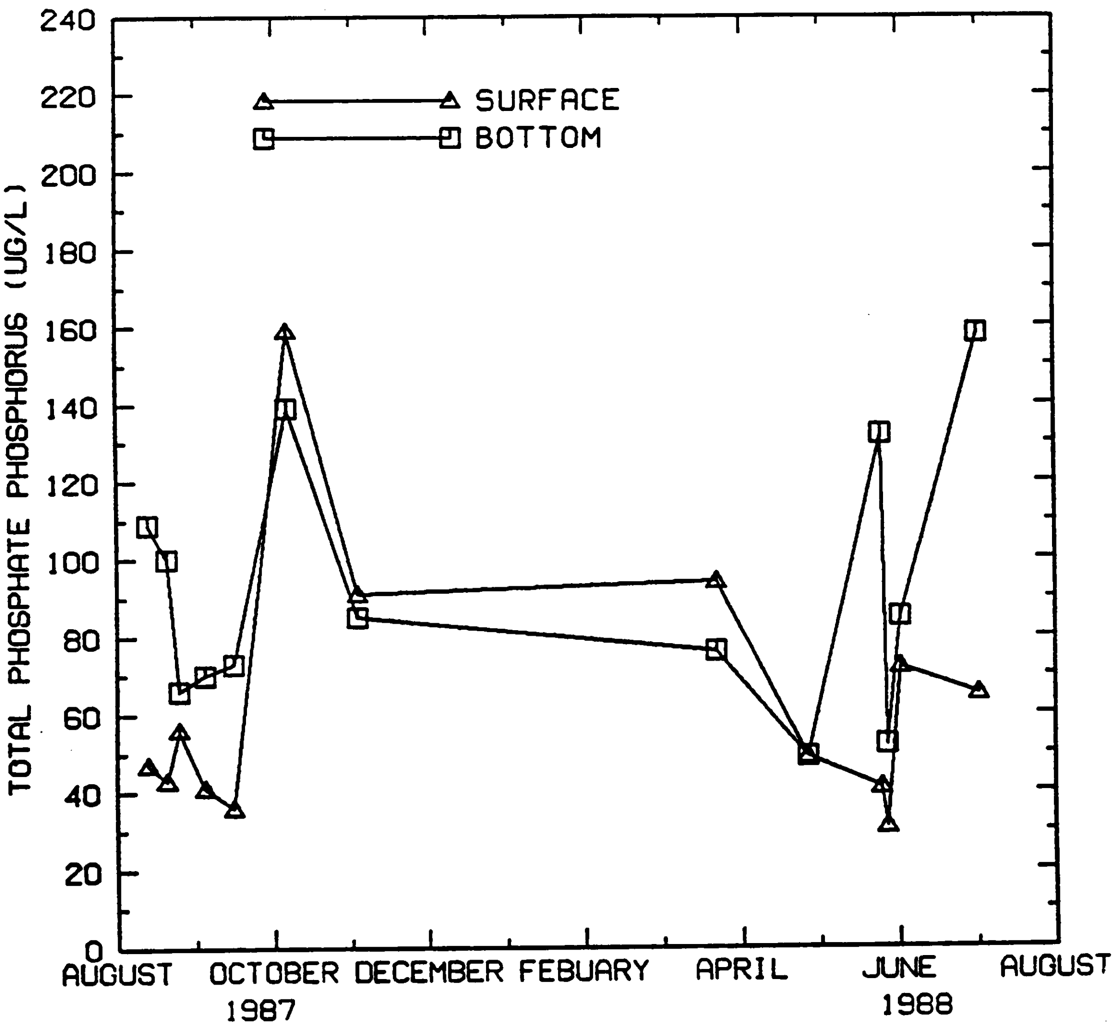

Figure 23. Total phosphate-phosphorus levels during the study period at site 1.

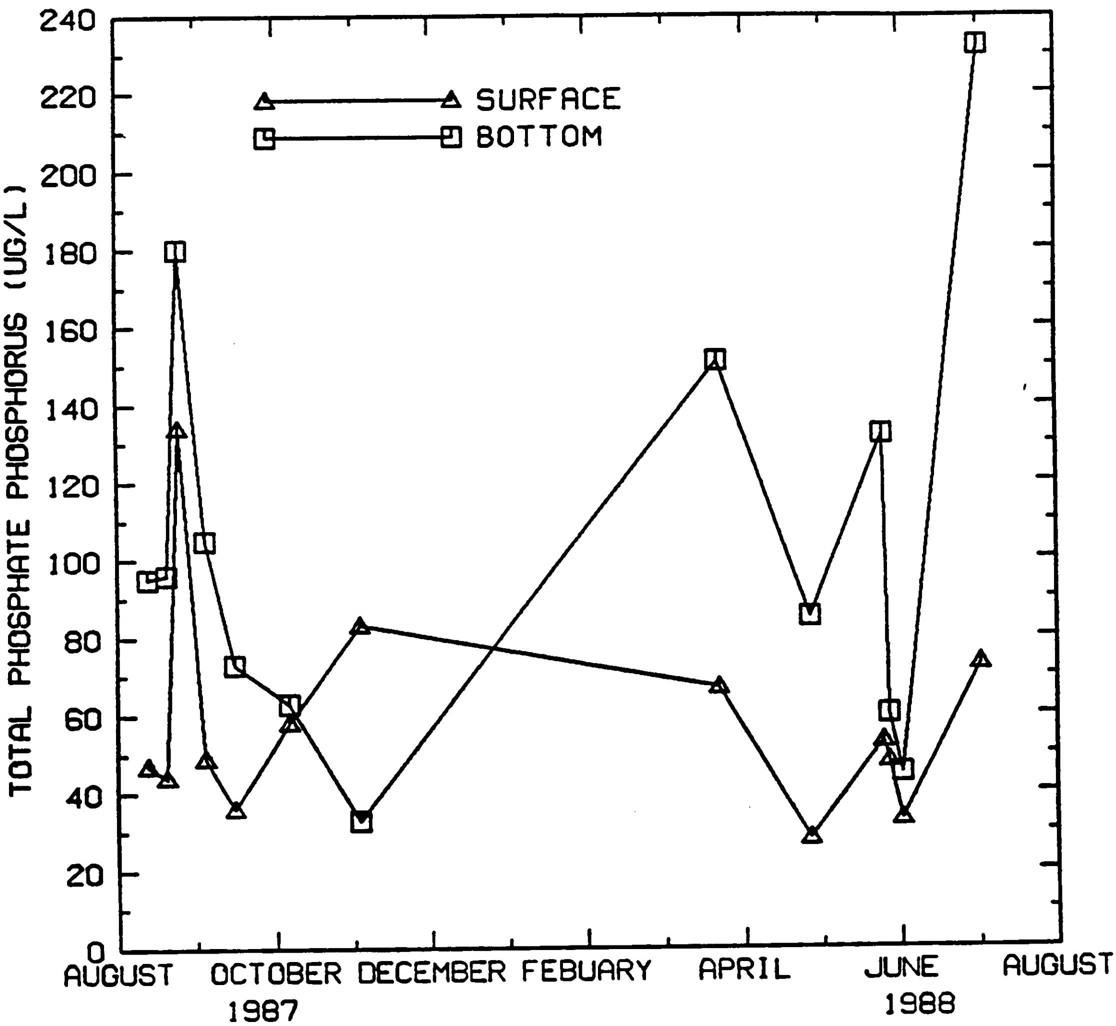

Figure 24. Total phosphate-phosphours levels during the study period at site 2.

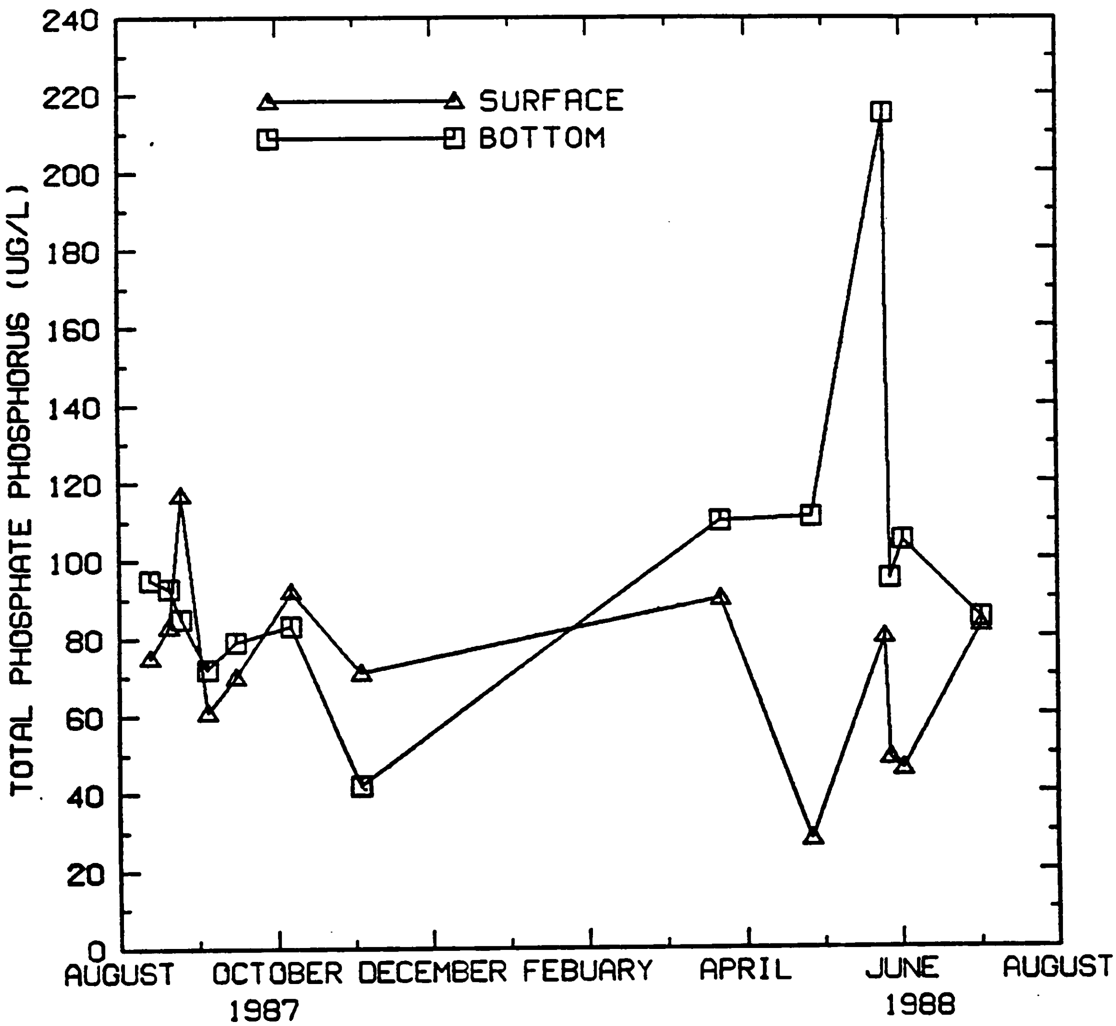

Figure 25. Total phosphate-phosphorus levels during the study period at site 3.

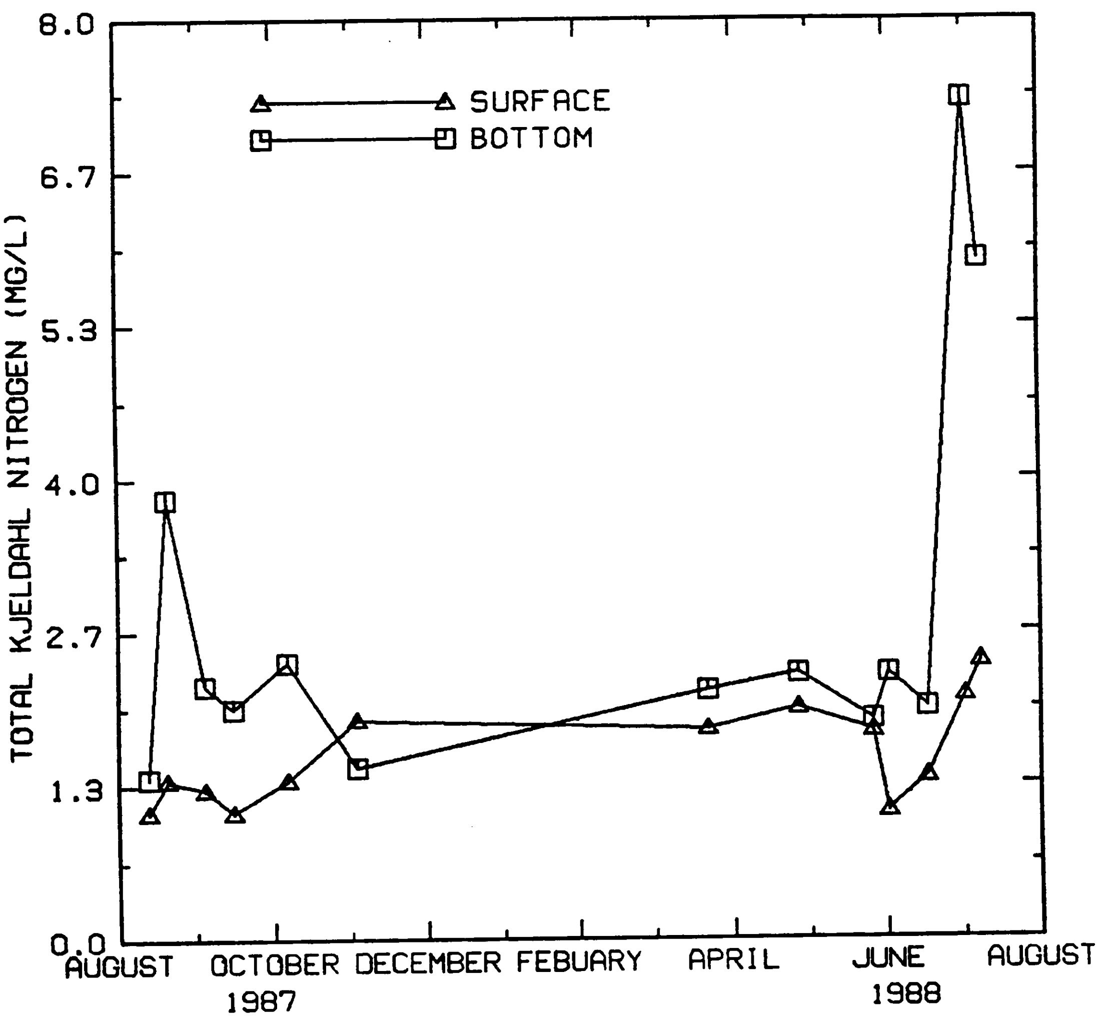

Figure 26. Total Kjeldahl Nitrogen levels during the study period at site 1.

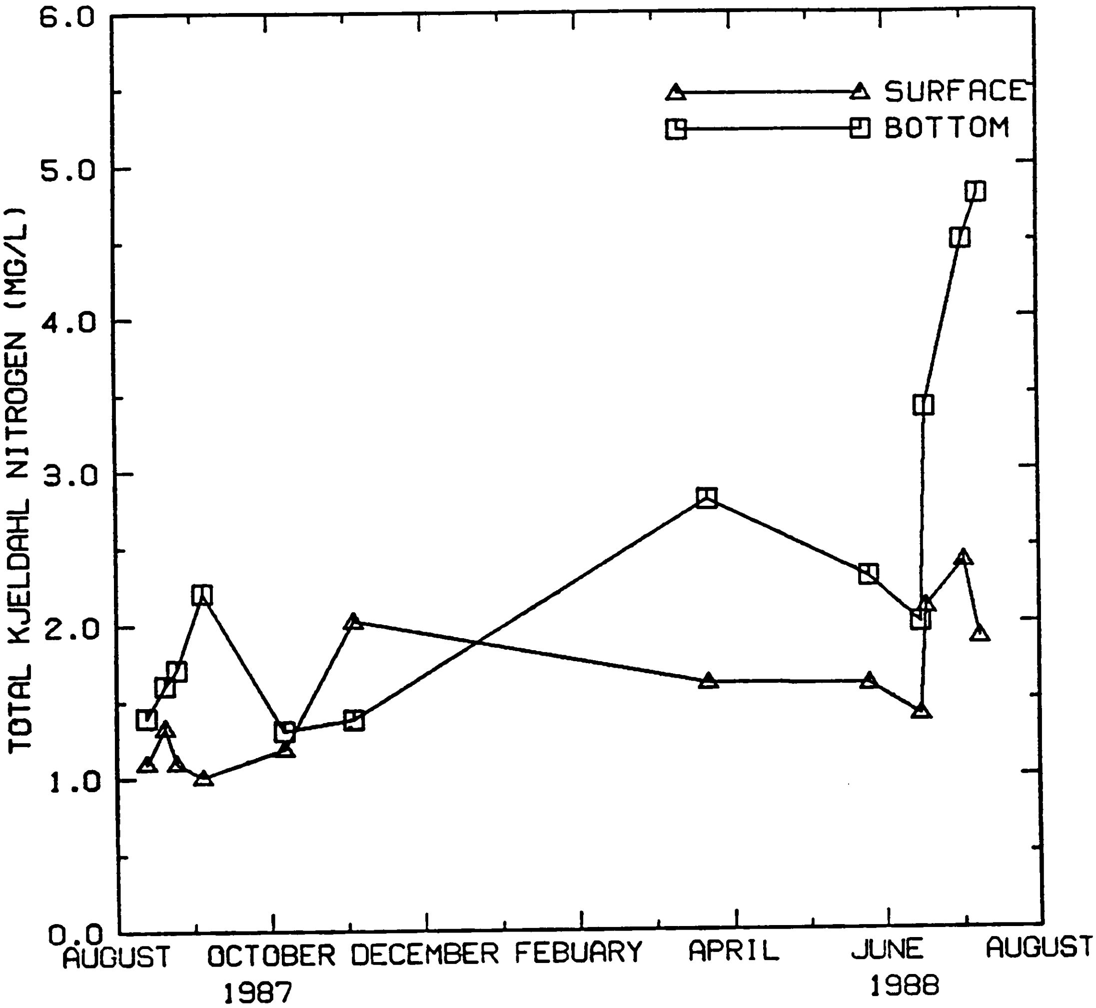

Figure 27. Total Kjeldahl Nitrogen levels during the study period at site 2.

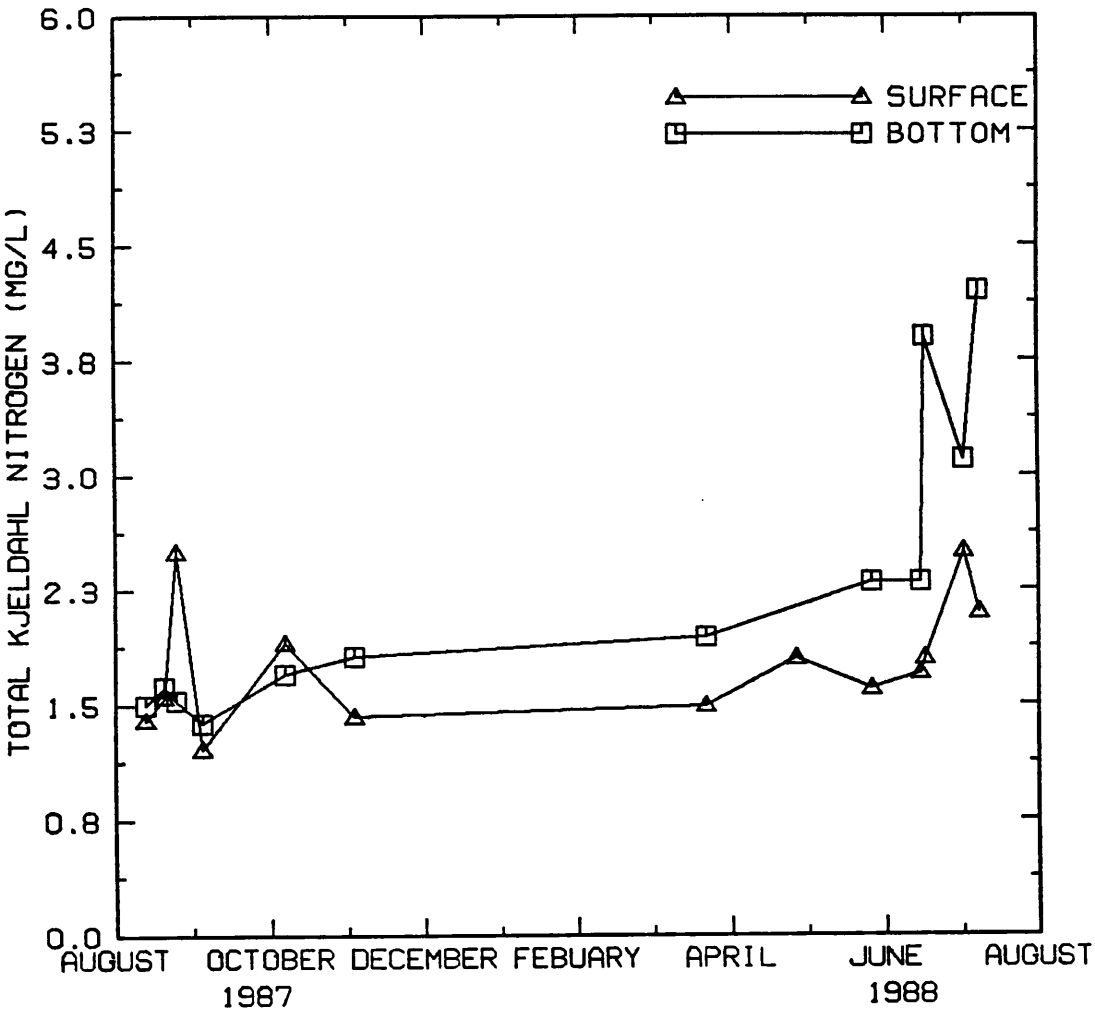

Figure 28. **Total Kjeldahl Nitrogen levels during the study period at site 3.**

Table 7. Water quality data at site 1.

Date	Sample Location	Total-P ug/L	Ortho-P, ug/L	Alkalinity mg/L	pH	Secchi, in cm	TKN, mg/L	Dissolved Oxygen mg/L
7-31-87	S	47	<10	160	-	-	-	12.6
	B	109	35	237	-		-	0.2
8-12-87	S	-	-	148	7.7	33.5	-	-
8-19-87	S	43	<10	171	6.8	31.8	1.5	10.1
	B	100	15	213	6.9		3.8	0.3
8-24-87	S	56	33	163	7.6	43.2	1.4	11.2
	B	66	52	207	6.7		1.9	0.1
9-04-87	S	41	18	-	-	45.1	1.3	-
	B	70	57	-	-		2.2	-
9-15-87	S	36	12	-	-	47	-	5.3
	B	73	64	-	-		-	0.1
10-06-87	S	165	-	199	7.8	50.0	1.4	9.2
	B	139	61	206	7.8		1.6	7.0
11-03-87	S	91	<10	-	6.8	48.0	1.9	12.4
	B	15	<10	-	6.7		1.5	1.0
3-21-88	S	94	37	160	7.9	72.0	-	8.2
	B	76	38	180	7.3		-	0.7
4-26-88	S	49	<10	-	-	70.0	2.0	8.4
	B	49	23	-	-		2.3	0.4

(continued)

Table 7. (continued)

Date	Sample Location	Total-P ug/L	Ortho-P, ug/L	Alkalinity mg/L	pH	Secchi, in cm	TKN, mg/L	Dissolved Oxygen mg/L
5-24-88	S	41	20	202	7.9	38.0	1.8	10.4
	B	132	120	234	7.2		1.9	0.2
5-26-88	S	31	<10	190	7.7	71.0	-	11.2
	B	52	<10	236	6.9		-	0.1
6-01-88	S	72	10	189	8.1	-	-	-
	B	85	15	195	7.2		-	-
6-14-88	S	-	-	211	7.9	56.0	-	13.2
	B	-	-	229	7.1		-	0.2
6-16-88	S	-	-	207	7.6	35.0	1.4	10.2
	B	-	-	235	7.0		2.0	0.2
7-01-88	S	65	28	215	7.5	87.0	2.1	2.1
	B	158	115	220	7.4		7.3	0.0
7-07-88	S	-	-	180	7.4	74.0	2.4	12.8
	B	-	-	225	6.9		5.9	0.0
7-08-88	S	-	-	176	7.5	38.0	-	8.2
	B	-	-	228	6.7		-	0.0

Table 8. Water quality data at site 2.

Date	Sample Location	Tot-P, ug/L	Ortho-P, ug/L	Alkalinity mg/L	pH	Secchi, in cm	TKN, mg/L	Dissolved Oxygen mg/L
7-31-87	S	47	<10	137	-	48.3	-	13.0
	B	95	30	210	-		1.4	0.1
8-12-87	S	-	-	156	7.7	34.3	-	-
8-19-87	S	44	<10	167	7.4	17.1	1.3	9.9
	B	96	<10	197	7.5		1.6	0.4
8-24-87	S	-	66	157	7.5	17.8	1.1	8.9
	B	-	75	182	7.0		1.7	0.2
9-04-87	S	49	17	-	-	-	1.0	-
	B	105	37	-	-		2.2	-
9-15-87	S	36	12	-	-	33.0	-	5.3
	B	73	15	-	-		-	1.2
10-06-87	S	58	19	193	7.9	60.0	1.2	9.4
	B	63	29	205	7.7		1.3	7.1
11-03-87	S	83	<10	-	8.0	45.0	2.0	9.8
	B	33	<10	-	7.6		1.4	0.6
3-21-88	S	67	20	159	7.7	71.0	1.6	-
	B	151	57	190	7.6		2.8	-
4-26-88	S	28	<10	-	-	65.0	1.9	8.5
	B	111	20	-	-		2.2	3.6

(continued)

Table 8. (continued)

Date	Sample Location	Tot-P, ug/L	Ortho-P, ug/L	Alkalinity mg/L	pH	Secchi, in cm	TKN, mg/L	Dissolved Oxygen mg/L
5-24-88	S	53	22	191	8.0	37.0	2.1	11.1
	B	132	36	229	6.9		3.3	0.1
5-26-88	S	30	<10	188	7.6	40.0	-	10.8
	B	50	<10	230	7.0		-	0.1
6-01-88	S	-	13	179	7.8	46.0	-	-
	B	-	12	200	7.4		-	-
6-14-88	S	-	-	195	8.0	75.0	-	13.2
	B	-	-	215	6.8		-	0.2
6-16-88	S	-	-	188	7.6	49.0	2.1	10.4
	B	-	-	209	6.7		3.4	0.2
7-01-88	S	73	34	-	8.0	60.0	2.4	2.5
	B	232	146	-	7.7		4.5	0.0
7-07-88	S	-	-	189	7.9	65.0	1.9	12.8
	B	-	-	229	6.9		4.8	0.0
7-08-88	S	-	-	180	7.6	48.0	-	8.4
	B	-	-	211	6.8		-	0.0

Table 9. Water quality data at site 3.

Date	Sample Location	Tot-P, ug/L	Ortho-P, ug/L	Alkalinity mg/L	pH	Secchi, in cm	TKN, mg/L	Dissolved Oxygen mg/L
7-31-87	S	11	11	141	-	-	-	14.0
	B	11	11	-	-		-	0.1
8-12-87	S	-	-	160	7.5	16.8	-	-
8-19-87	S	83	<10	177	-	19.3	1.6	11.1
	B	93	<10	164	-		1.6	0.2
8-24-87	S	117	-	163	7.2	18.0	2.5	10.5
	B	85	62	168	7.4		1.5	0.3
9-4-87	S	61	21	-	-	-	1.2	-
	B	72	54	-	-		1.4	-
9-15-87	S	70	15	-	-	30.5	-	6.5
	B	79	<10	-	-		-	4.9
10-06-87	S	92	40	204	7.6	-	1.9	9.8
	B	83	39	291	7.8		1.7	7.1
11-03-87	S	71	23	-	8.1	40.0	1.4	10.4
	B	42	20	-	7.8		1.8	1.8
3-21-88	S	90	32	139	7.8	49.0	1.5	-
	B	110	49	153	7.9		1.9	-
4-26-88	S	28	<10	-	-	50.0	1.8	8.1
	B	111	<10	-	-		2.1	6.8

(continued)

Table 9. (continued)

Date	Sample Location	Tot-P, ug/L	Ortho-P, ug/L	Alkalinity mg/L	pH	Secchi, in cm	TKN, mg/L	Dissolved Oxygen mg/L
5-24-88	S	80	36	180	7.8	38.0	1.7	10.3
	B	215	153	226	7.6		2.9	0.9
5-26-88	S	49	<10	167	7.7	50.0	-	9.9
	B	60	<10	220	7.6		-	1.2
6-01-88	S	-	11	191	7.7	34.0	-	-
	B	-	15	211	7.8		-	-
6-14-88	S	-	-	172	7.3	45.0	-	14.4
	B	-	-	199	6.9		-	3.2
6-16-88	S	-	-	168	7.4	55.0	1.8	10.8
	B	-	-	209	7.0		3.9	4.3
7-01-88	S	83	20	-	7.9	35.0	2.5	4.8
	B	85	14	-	7.9		3.1	1.4
7-07-88	S	-	-	179	8.4	10.0	2.1	12.0
	B	-	-	190	7.9		4.2	6.6
7-08-88	S	-	-	165	8.0	38.0	-	8.2
	B	-	-	187	7.7		-	0.0

The vita has been removed from the scanned document