THE IMPACT OF BANNING PHOSPHATE-CONTAINING DETERGENTS ON THE WATER QUALITY OF INLAND WISCONSIN LAKES

Prepared for The Soap and Detergent Association

by

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1. The statistical analysis was conducted by John W. Wilkinson, Ph. D., Rensselaer Polytechnic Institute, Troy, N.Y.

EXECUTIVE SUMMARY

Summary

of

"The Impact of Banning Phosphate-Containing Detergents

on the Water Quality of Inland Wisconsin Lakes"

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Objective

There are 15,000 inland lakes in the State of Wisconsin. The impact of the three-year Wisconsin detergent phosphate ban on Wisconsin lake water quality has been evaluated through research on Wisconsin lakes likely to be affected by changes in phosphate discharges. The ban was imposed on July 1, 1979 and ended June 30, 1982.

Methodology

"Test" lakes and "reference" lakes were used in the study. The test lakes were chosen as lakes likely to be affected by changes in phosphate discharges to them within the three-year time span of the ban. The reference lakes were lakes receiving negligible amounts of phosphate from sewage. Data from the reference lakes were used to help show what changes in the test lakes were due to the ban apart from natural effects.

The test lakes were: Swan Lake (Columbia County), Moss Lake (Vilas County), Townline Lake (Oneida), Enterprise Lake (Langlade County), Elk Lake (Price County, Balsam Lake (Washburn County), and Butternut Lake (Price County). The reference lakes were: Little Bearskin Lake (Oneida County) and Teal Lake (Sawyer County).

The lakes were studied during the summer of 1978 (before the ban) and during the summers of 1980, 1981 and 1982 to determine any effects of the ban.

The methodology involved measuring water quality parameters related to algal growth: algal concentration (measured by the concentration of chlorophyll *a*, the green plant pigment), total phosphate concentration (measured by the concentration of all phosphate species), water clarity (estimated by Secchi disc depth) and the enumeration and identification of phytoplankton.

Result

- The total amount of phosphorus entering the test lakes from all sources was estimated to have been reduced 0.1 to 12 percent as a result of the ban.

- No changes attributable to the ban in either chlorophyll *a* (algal concentration) or total phosphate concentration were observed in the test lakes. Secchi disc depth readings recorded during this study are inconsistent with chlorophyll *a* and total phosphate concentrations. This indicates that other factors (e.g., suspended solids, color) were affecting the Secchi disc depth readings.

- The dominance in the phytoplankton of blue-green algae (nuisance forms) has been on the increase since 1978 in both reference lakes and test lakes.

Conclusions

- There is little liklihood that a phosphate detergent ban could result in tangible water quality benefits for the vast majority of Wisconsin lakes.

- Based on the results of this study, diverse sources of phosphorus and other algal nutrients, plus factors such as light and temperature, are more important in determining water quality in Wisconsin lakes than the amount of phosphorus controllable by a phosphate detergent ban.

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I. INTRODUCTION

Reductions in the amount of phosphate contained in laundry detergents have been mandated in some areas as a means of improving water quality in lakes currently receiving phosphorus through municipal wastewater discharges or septic tank seepage. However, various reports suggest that such reductions in detergent-related phosphorus are ineffective in achieving perceptible water quality improvements because the relative reductions in the total load of phosphorus to the lakes are not significant. Studies were conducted in Wisconsin to determine any significant improvement in water quality due to a three year ban on phosphate-containing detergents.

Limitations prohibiting phosphorus in laundry detergents have been imposed in New York; Indiana; Vermont; Michigan; Minnesota; Wisconsin; Dade County, Florida; and the City of Chicago, Illinois. The State of Wisconsin legislated a phosphate detergent ban, effective July 1, 1979. The ban, originally enacted to be in effect to June 30, 1981, was extended to June 30, 1982. The purpose of imposing a short term ban was to allow an assessment of any impact the ban may have had on the water quality of Wisconsin lakes which might lead to a reevaluation of the need for the ban. The Soap and Detergent Association (SDA)-Wisconsin Lakes Study, of which the 1978 through 1981 sampling programs were a part, was designed to provide information which would contribute to the 1982 legislative evaluation of the Wisconsin phosphate detergent ban. The study would provide data which could allow the determination of any significant changes in water quality in inland Wisconsin lakes impacted by municipal wastewater treatment plant discharges or septic tank/tile field systems. To determine the variability in lake water quality that occurs naturally, several lakes which received neither septic tank system nor municipal treatment plant discharges were included in this program. A report evaluating the results of samples collected during the 1978-1981 time period was prepared by N. Clesceri (1981). No positive water quality improvement attributable to the detergent phosphate ban was observed in any of the study lakes. The Wisconsin Department of Natural Resources (1982) also could not detect any improvement in inland lake water quality. Therefore, the purpose of the ban was unsupported. From these studies, as well as consumer concern for effective laundry detergent products, the Wisconsin State Legislature did not extend the ban beyond June 30, 1982. The SDA-Wisconsin Lakes Study was continued in 1982 to allow for an analysis of any demonstrable improvement in inland lake water quality appearing three years after the imposition of the ban.

II. BACKGROUND

Our knowledge of the complexity of freshwater systems has greatly increased during the past 50 years, and efforts are continuing to advance further our understanding of lake processes. Although inland waters cover only two percent of the earth's surface, they play an important role in our society. Increasing population and demand for improved quality of life continue to place great demands on freshwater resources. Clean water supplies are required for drinking water, recreational activities, and industrial processes. Appropriate management techniques can be based on our understanding of the functioning of this precious and vital resource.

A. LAKE PHYLOGENY AND CULTURAL EUTROPHICATION

Aquatic ecosystems undergo changes, continually evolving as a result of their intimate relationship to the surrounding environment. Land use in the watershed; the age, type, and amount of soil and vegetation; wildlife habitation; and population density are all factors which can significantly influence the trophic status of a lake. Lakes generally progress from the oligotrophic state, characterized by clean, clear, unproductive water that is low in nutrients, to a eutrophic state in which nutrients are plentiful, plant and animal productivity is high, and water quality poor. Over long periods of time silt, minerals, nutrients, and organic materials from tributaries and other sources accumulate. Nutrients are necessary for plant and animal growth, which can eventually lead to a change in the biota of the lake. Under natural conditions a lake will age slowly, decreasing in depth due to the deposition of sediments, becoming a marsh or bog and finally a terrestrial ecosystem. This process, from start to finish, can take hundreds to thousands of years. This aging process is called eutrophication, a complex process which varies from lake to lake with regard to extent, rate, and causal factors. Cultural eutrophication is the accelerated aging of fresh water bodies caused by increased nutrient loadings due to human activities.

As awareness of the problems associated with lake eutrophication in the United States increased, numerous studies were undertaken to determine the causes and characteristics of the problem (e.g., Chen, 1970; Middlebrooks and Porcella, 1970; Porcella and Bishop, 1975; and Sawyer, 1947). Many remedial measures are currently being used or investigated. Some are simply stopgap measures which treat the symptoms rather than the cause of the water quality degradation. These include copper sulfate application to kill algae and reduce obnoxious algal blooms, alum additions to precipitate phosphorus from the lake water, and weed harvesting aimed at removing nutrients and the physical hindrance weeds pose to recreational activities. Other approaches reduce the loadings of nutrients to lakes in order to circumvent the problems which occur once the nutrients enter the lake. This has been accomplished by diverting wastewater flows around lakes, sewering lakeshore areas which were serviced by indequate septic tank/tile field systems, phosphate detergent bans, phosphorus removal at wastewater treatment plants, and a number of alternative wastewater treatment techniques. These latter techniques include using wastewater for irrigation and as a nutrient source for croplands and forests, or discharging it into natural and artificial wetlands which are capable of immobilizing large quantities of nutrients. Also, improved land management practices can reduce nutrient loadings from both agricultural (Woolhiser, 1975) and urban runoff.

B. NUTRIENT SOURCES

The sources of external (allochthonous) nutrient inputs to a lake or stream can be classified as either natural or anthropogenic, and point or non-point.

Point sources include:

- Treated wastes from municipal treatment plants and private, on-site waste disposal systems: domestic wastes, household food, industrial and commercial wastes discharged to municipal treatment plants, and water treatment chemicals.
- Industrial wastes discharged to waterways.
- Storm sewer discharges.

Non-point sources include:

- Land runoff: urban runoff and drainage, unchannelized stormwater flows, and runoff from agricultural lands (e.g., fertilizers, animal wastes, soils, etc.) and forested lands (e.g., decaying vegetation and wild animal wastes).
- Ground water.
- Atmosphere: dryfall (i.e., particulates and gases) and wetfall (i.e., rain and snow).

Natural sources of nutrients result from geochemical processes and circulate via the biogeochemical cycling of materials. Nutrients with a gaseous phase (e.g., carbon and nitrogen) have almost complete biogeochemical cycles, included in which is a direct transfer between the atmosphere and water, whereas nutrients without a prominent gaseous phase (e.g., phosphorus, calcium and silica) must be circulated by processes such as erosion, sedimentation, biological activity, and human sources, there being no gaseous atmospheric reservoir on which to draw.

C. LIMITING FACTOR CONCEPT

The logic of reducing the nutrient influx to natural waters, such that the nutrient concentration limits plant growth, is based on the functional relationship between plant productivity and nutrient availability. The concept of a limiting growth factor was first introduced in a concise form by Justus Liebig in 1840: "[the] growth of a plant is dependent on the amount of foodstuff which is presented to it in minimum quantity" in relation to its needs. This tenet has become known as "Liebig's 'Law' of the Minimum".

Although many factors such as light, temperature, turbulence and mixing, and nutrients may limit the extent of plant biomass in aquatic systems, only certain ones, particularly nutrients, are controllable by man in a practical and cost-effective manner. A limiting nutrient can be considered as such if, when its concentration in a waterbody is increased, it acts to stimulate plant growth and accelerate eutrophication.

D. NUTRIENT REQUIREMENTS OF AQUATIC PLANTS

Nutrients required for plant growth are generally separated into two categories, the macronutrients which are required in relatively large amounts, and the micronutrients which are needed only in minute quantities. The macronutrients include calcium, carbon, magnesium, nitrogen, phosphorus, potassium and sulfur. The micronutrients (trace elements) include boron, copper, iron, manganese, molybdenum and zinc.

Numerous investigations have shown that a wide variety of nutrients can become limiting to algal populations, but probably only two of the macronutrients, nitrogen and phosphorus, significantly control the development of algal blooms (Goldman, 1965; Hasler, 1974; and Hutchinson, 1957). The greatest attention has been focused on phosphorus since it is more often limiting than nitrogen, and has more readily controllable sources.

The principal sources of nitrogen for algae are the inorganic forms, i.e., nitrate and ammonia. Fixation of atmospheric nitrogen by blue-green algae and some bacteria, as well as bacterial degradation of organic nitrogen, can also serve as important nitrogen sources for aquatic ecosystems. Phosphorus is generally available to plants only as the orthophosphate (PO_4^{-3}) ion, an inorganic form.

Algal requirements for both nitrogen and phosphorus are species specific. A given concentration of nutrients in one lake may yield a significantly different algal standing crop than the same concentration in another lake due to numerous interacting factors. The general chemical composition of the water and its alkalinity, the nutrient turnover rate and rate of degradation of organic compounds, regeneration from sediments, and zooplankton and algal excretions, among other factors, all have some impact on the algal biomass produced in relation to nutrient concentration. A good example is the 'luxury' uptake of phosphorus by natural phytoplankton populations: "The pattern of phosphorus supply and utilization in natural populations of phytoplankton is characterized by highly variable rates of uptake and release which bear little apparent relation to the physiological needs of the plankton" (Richey, 1977 and 1979). Taft et al. (1975) reported phosphate uptake by Chesapeake Bay phytoplankton was as much as 100 times that required for photosynthesis. Several investigations have suggested that values of 0.01 mg P/L and 0.15 mg N/L are the upper concentration limits if an oligotrophic, aesthetically pleasing lake suitable for water-based recreation, propagation of cold water fisheries (such as trout), and with very high clarity is desired; concentrations approximately twice these are the conditions under which lakes become eutrophic (Chapra and Reckhow, 1979; Dillon and Rigler, 1975; and Vollenweider, 1968).

Typically, aquatic macrophyte and algal tissues contain a ratio of nitrogen to phosphorus of 15:1. In general, when the lake water N:P ratio is less than 10:1 nitrogen is limiting, and when it is greater than 14:1 phosphorus is limiting.

In conclusion, it can be stated that two macronutrients, nitrogen and phosphorus, generally limit algal growth. However, nutrient turnover, organism succession, and water chemistry all interact to make it impossible to determine specific limiting concentrations common to all water bodies, although general guidelines can be established. When excessive nutrients are present in a lake, plant growth may become limited by an extrinsic factor such as light.

E. PHOSPHORUS DYNAMICS IN LAKES

The many dynamic variables that directly influence and control aquatic productivity are intimately related. Not only are the material inputs to a lake a factor, but also the pathways they follow; materials recycle within a lake due to complex, interrelated physical, biological, and chemical processes. The major pathways of the phosphorus cycle are shown in Figure 1. All phosphorus originates from minerals which are weathered by natural processes or utilized by humans. Human-related activities may increase phosphorus inputs to surface waters over the natural level expected for a particular lake. The impact of any activity on a lake depends upon the geology, climate, morphology, and land use characteristics of the basin.

The role of lake sediments as a sink or source of phosphorus is a very important and complex process involving physical (e.g., turbulence and mixing), chemical (e.g., redox potential and oxygen content of the sediment-water interface, and the quantity and forms of iron and aluminum in the sediments), and biological factors (e.g., microbial mineralization of organic, phosphorus-containing compounds). Although numerous mechanisms for phosphorus release or uptake from sediments are postulated, the manner in which they interact, and the resulting net effect, are difficult to predict or interpret in natural waters. A few general statements are pertinent concerning the importance of lake sediments in lake phosphorus dynamics. The reader should refer to other sources (e.g., Wetzel, 1975) for a more complete discussion and list of references.

As an illustration of the complex nature of sediment phosphorus, let

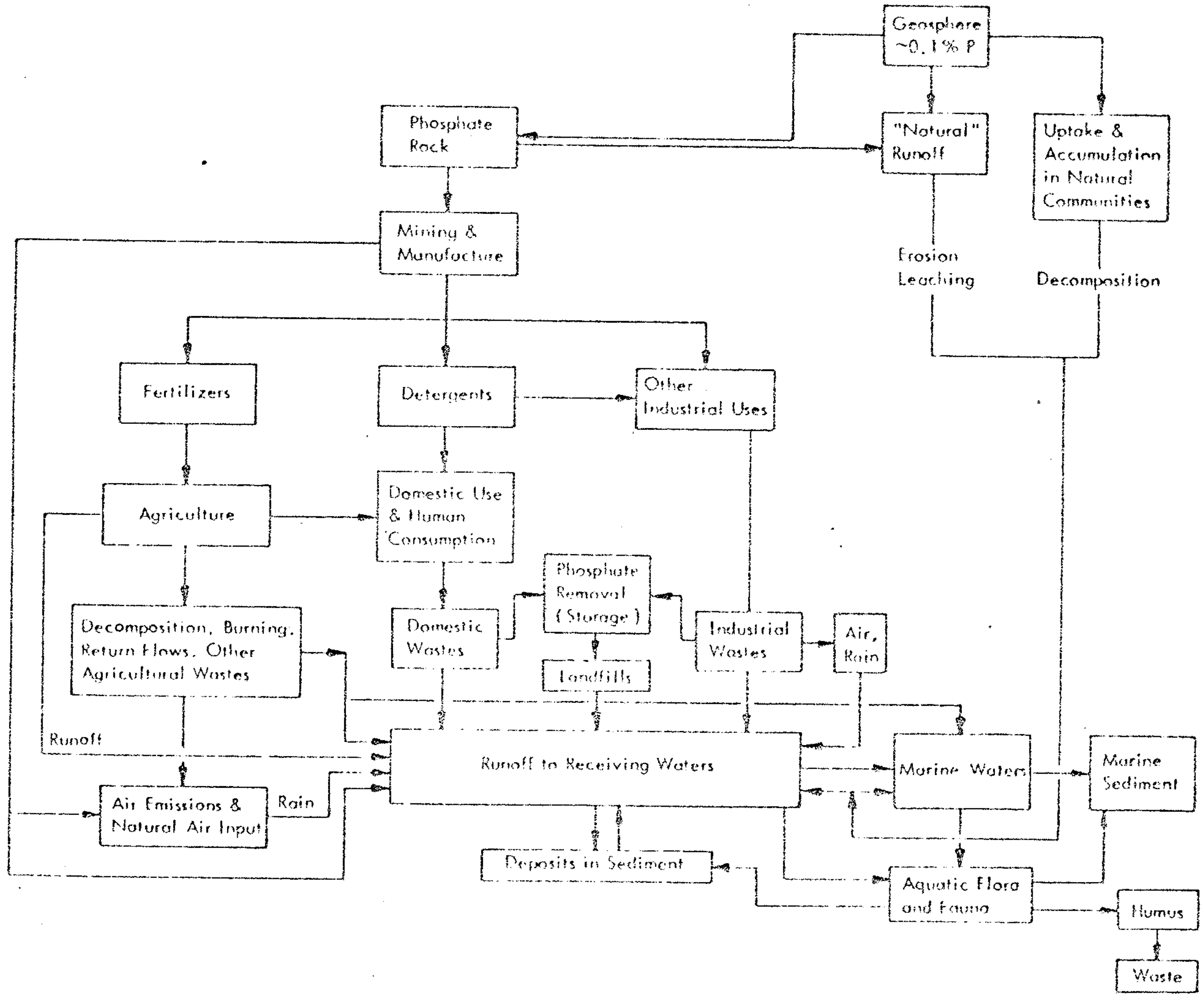

Figure 1: Phosphorus Pathways in the Environment (from Mumma et al., 1979)

us examine the impact of oxygen concentration on phosphorus solubility. Probably the most important point is that there is very little correlation between the phosphorus concentration in the sediments, and the phosphorus content and productivity of the overlying waters; factors other than simple concentration gradients and diffusion, control phosphorus exchange between lake sediments and lake waters. Therefore, the phosphorus concentration of sediments does not provide an indication of whether the sediments are acting as a source or a sink, or whether the phosphorus is available for algal utilization.

Many studies have pointed out the importance of the conditions at the water-sediment interface and the surface layer of sediments (microzone) in controlling phosphorus exchange (Mortimer 1941, 1942, and 1971). A major mechanism involved in sediment-phosphorus exchange is the covalent bonding and adsorption of apatite, organic phosphorus, and orthophosphate to hydrated iron or aluminum oxides (Williams et al., 1971). In the oxidized state, which occurs in the presence of oxygen, these compounds are only slightly soluble, whereas in the reduced state, which occurs under anaerobic conditions (absence of oxygen), they dissolve more readily in water. Once phosphorus is released the process is reversible, if oxygen is introduced or encountered. The oxygen levels can subsequently decrease, converting the oxides to their reduced form, and releasing the bound phosphorus. This is a cycle that can be encountered over time in a lake or within regions of a lake (i.e., movement of oxidized phosphorus to anaerobic regions or vice versa).

Therefore, if a lake is oxygenated the metal oxide-phosphorus complex is insoluble and therefore the sediments act as a phosphorus sink. For a lake which is thermally stratified in summer, if dissolved oxygen disappears in the hypolimnion, some phosphorus may be released into these hypolimnetic waters. But it is commonly held that this water is "trapped" in this section of the lake during the summer because of thermally-derived density differences. Because of this, hypolimnetic water, rich in phosphorus, does not come in contact with algae in the epilimnetic zone at a time when they are present in the highest population numbers and growing at their fastest rate in the summertime. When the lake cools in the fall and the fall turnover (mixing from lake bottom to top) occurs, conditions (e.g., lower temperatures, shorter periods of sunlight) are not conducive to extensive algal growths in spite of potentially higher phosphorus concentrations, enhanced by hypolimnetic waters. One controlling factor in maintaining phosphorus concentrations in lake water relates to the fact that as turnover occurs, oxygen, if it were absent, would be reintroduced into the lake water. This oxygen would cause the oxidation of the reduced metal-phosphorus complex to the metal oxide-phosphorus complex which is insoluble. Through this sequence, phosphorus released from sediment into overlying hypolimnetic waters could be returned to the sediments and perpetually trapped.

F. BIOAVAILABILITY OF PHOSPHORUS

All models utilizing phosphorus concentrations to estimate algal growths in lakes assume that phosphorus is the limiting nutrient and that changes in phosphorus concentrations are reflected by changes in algal growths. Historically, total phosphorus has been used as the index for such analytical models. The proposal of Schaffner and Oglesby (1978) to use a composite fraction they term biologically available phosphorus (BAP), which represents that portion of total phosphorus utilizable by algae, may improve the accuracy of models. By excluding the unusable (unavailable) phosphorus fractions (e.g., mineral apatite) a better relationship between phosphorus and algal growth should result, they argue. A further consideration was presented in a paper by Verhoff and Heffner (1979) describing a study on the rate of availability of total phosphorus in river waters: "The main conclusion to be obtained from this study is the rate of conversion of total phosphorus to phosphorus available to algae may be more important than the fraction which would eventually become available." The latter is roughly equivalent to Schaffner and Oglesby's BAP, and is often termed the ultimate BAP. A similar opinion was expressed by DePinto et al. (1981) in a study on suspended sediments in streams: "An extremely important factor in assessment of the biological effects of phosphorus from any source is the rate at which the phosphorus becomes available to aquatic biota within the receiving water. This is true, as the rate of conversion of potentially-available to actually-available phosphorus competes in time with the rate of other processes, for example: adsorption, precipitation, sedimentation, and dilution." Note that DePinto et al. have included the factors of the temporal-spatial distributions of phosphorus fractions, the largely unquantifiable physical-chemical-biological transformations of phosphorus species in natural waters, and the complex matter of stream and lake hydrology.

The most "accurate" model would utilize the actual quantity of phosphorus available to, and utilized by, algae at any given point in time.

However, from the standpoint of applied science, the state of the art of modeling and the limited knowledge of the influences referred to by DePinto et al., make such a comprehensive approach unrealistic at this time. This is true both with respect to the best available technology and analytical techniques for biologically available phosphorus, and budgeting considerations such as time, personnel, and funds which would be required to undertake such a study.

Numerous techniques have been developed for the analysis of phosphorus. Each of the methods discussed below measures a different portion of total phosphorus; unfortunately, no single analysis, or combination of analyses, adequately represents the phosphorus that is available to algae. Consequently, many investigations have been conducted in attempts to ascertain the best correlation between analytically defined phosphorus fractions and biologically available phosphorus.

Soluble phosphorus is generally considered to be that phosphorus which can be filtered through a membrane with a 0.45 μm pore size; this is often called "dissolved" phosphorus, although part of this fraction may be in fine particulates or colloids. Soluble phosphorus is usually divided into two subfractions, soluble unreactive phosphorus and soluble reactive phosphorus. Particulate phosphorus is that phosphorus which does not pass through a 0.45 μm filter.

Soluble reactive phosphorus (SRP), often measured by the method of Murphy and Riley (1962), or a modification thereof, is a fraction which is generally considered to be entirely BAP; a major percentage is usually orthophosphate (PO_4^{-3}). Orthophosphate is biologically the most readily used form of phosphorus. Experimentally, the availability of SRP has been shown by DePinto et al. (1980) and Paerl and Downes (1978). Paerl and Downes separated two fractions from lake water by gel filtration; a reactive low molecular weight portion and a reactive high molecular weight portion, the former being immediately available to algae whereas the latter generally took 48 to 96 hours to be utilized.

Soluble unreactive phosphorus (SUP) is most often calculated from the difference between total soluble phosphorus (TSP) and soluble reactive phosphorus (SRP) (i.e., SUP = TSP - SRP). At least some of the SUP appears to be available to algae (Berman, 1970; Peters, 1978; and Peters, 1979). The action of enzymes such as alkaline phosphatases can liberate phosphorus from organically bound fractions (Berman, 1970 and Fitzgerald and Nelson, 1966). Using a radioactive tracer in phosphorus kinetics studies, Peters (1978) suggested 50 to 100 percent of the SUP was BAP in the waters he studied. Furthermore, a complex exchange mechanism may exist in the euphotic zone of lakes involving particulate phosphorus, orthophosphate, an algal excreted organic phosphorus compound (XP), and a colloidal substance (Lean, 1973a; Lean, 1973b; Peters, 1978; and Peters, 1979); a portion of the excreted phosphorus may be in a refractory form unavailable to algae (Peters, 1979).

Numerous studies have been conducted to estimate the availability of phosphorus in suspended solids of rivers and lakes. A wide range of BAP concentrations has resulted, the variability often being attributed to the presence of apatite, a phosphorus rich mineral which analyzes as total phosphorus but is only minimally utilizable by algae. Cowen and Lee (1976) using <u>Selenastrum capricornutum</u> in batch bioassays lasting 19 to 22 days found that particulates from Madison, Wisconsin urban runoff released 8 to

55 percent of the total particulate phosphorus (TPP) with an average of 30 percent, when suspended in phosphorus-free Algal Assay Procedure (EPA, 1971) medium. DePinto et al. (1981) using their Dual Culture Diffusion Apparatus (DePinto, 1982) analyzed phosphorus availability in suspended sediments from Lower Great Lakes tributaries using Selenastrum capricornutum and Lake Erie water as a medium in harvest-reinoculation bioassays. The bioassays were continued until no further phosphorus uptake was observed, generally three to four weeks. A range of 5 to 31 percent of TPP was utilized, the difference being attributed to apatite concentrations. In studies using Selenastrum capricornutum in two-week long batch cultures with Provisional Algal Assay Procedure synthetic medium (USDI, 1969), Dorich et al. (1980) determined that 10 to 31 percent (mean = 21 percent) of TPP was BAP in sediments from the drainage water of Black Creek watershed, Allen County, Indiana.

Other work has been undertaken to relate BAP to chemically analyzed phosphorus fractions. The components of total sediment phosphorus with significant relationships to BAP are mainly associated with the reactive NaOH-extractable fraction (NaOH-P or base extractable, reactive) (DePinto et al., 1981; Williams et al., 1980; and Cowen and Lee, 1976). A citrate-dithionite-b carbonate extractable (CDB-P or reductant extractable) fraction has also been related to BAP, however, neither is a direct measurement of BAP. Portions of these fractions, 13 to 77 percent of NaOH-P and 14 to 24 percent of CDB-P, have been found to be BAP.

Data on the bioavailability of phosphorus (to algae and aquatic macrophytes) in wastewater treatment plant effluents is scarce. De Pinto et al. (1982) conducted algal bioassays using wastewater samples from several locations (untreated influent, intermediate effluents, and "final" effluent) in four treatment plants. These plants were located in areas where phosphate detergent bans were in effect (the Gates-Chili-Ogden Plant in Rochester, N.Y.; the Frank Van Lare Plant in Rochester, N.Y.; the Big Sister Creek Plant in Angola, N.Y.; and the Ely Plant in Ely, Mn.). The results of the Dual Culture Diffusion Apparatus (DePinto et al., 1982) bioassays suggested an average of 72 percent of the total phosphorus in the bioassay samples was BAP. A comparison of the samples from different locations in the treatment trains indicated that the proportion of total phosphorus which was BAP usually did not vary significantly as wastewater passed through a plant.

An indirect means to investigate the bioavailability of phosphorus in wastewater treatment plant effluents is to look at the degradation of various forms of phosphorus which are not readily used by algae. The extent to which treatment processes convert these phosphorus compounds to orthophosphate (which is available to algae) can be considered an indirect measure of bioavailability. One class of compounds generally considered to be unutilizable until hydrolyzed is condensed phosphates (Sutton and Larson, 1964 and Clesceri and Lee, 1965 a,b). The rate of hydrolytic degradation of condensed phosphates is influenced by many factors, including temperature, pH, enzymes, colloidal gels (e.g., hydrated oxides of iron and aluminum), complexing cations (e.g., calcium), concentration, the ionic environment of the solution, and biological activity. Studies on the kinetics of the hydrolysis of pyrophosphate and tripolyphosphate (condensed polyphosphates) in sterile and non-sterile lake water and algal culture media showed that the chemical hydrolysis of these compounds was a relatively slow process, while biochemical hydrolysis proceeded rapidly (Clesceri and Lee, 1965a). Other studies also have reported that the presence of organisms in solutions to which condensed phosphates were added caused an accelerated rate of hydrolysis of these phosphates (Karl-Kroupa et al., 1957; Sawyer, 1952; and Engelbrecht and Morgan, 1959). A phosphate balance performed on the Oxford, England activated sludge wastewater treatment plant (Lewin, 1973 and Perry et al., 1975) showed that more than 90 percent of the condensed phosphates were hydrolyzed to orthophosphate in the sewers. Only 1.5 percent remained in the settled sewage, and none was detectable in the final effluent. This plant was not equipped for chemical phosphorus removal.

Much is yet to be learned about the bioavailability of phosphorus in different waters. Further work must be undertaken to clarify which what forms of phosphorus are being measured by the various analytical techniques, which of these forms are immediately available to algae and other organisms and the time (and under what conditions) it takes others to degrade to a usable form, and how these various forms of phosphorus affect the ever-changing physical, chemical, and biotic environments of lakes and streams.

G. NATURE OF DETERGENT PHOSPHATE

A typical all-purpose household detergent is made up of the surfactant, the phosphate builder, and miscellaneous ingredients, such as brighteners, perfumes, and inhibitors. The surfactant portion was responsible for past foaming problems caused by the non-biodegradable ABS (branched chain alkyl benzene sulfonate). The ABS surfactant was replaced more than a decade ago with biodegradable LAS (straight chain, or linear alkyl benzene sulfonate).

Tripolyphosphate is the chemical of choice as a phosphate builder in the all-purpose household detergent. Phosphate effectively plays many diverse roles as a detergent builder. One function is to soften water by sequestering calcium and magnesium ions which would otherwise react with fatty soils to form insoluble curds, thereby preventing the surfactant portion of detergents to emulsify the fatty soils. Phosphates also sequester objectionable elements such as iron and manganese salts that can cause rust spots, yellowing, or other discoloration of laundered fabrics, i.e., iron and manganese are changed to an insoluble form which deposits on clothing. With phosphate, the iron and manganese remain in solution and are rinsed away. Phosphates also disperse and suspend dirt. Without phosphates, dirt particles get trapped in fabric; with phosphates, dirt particles with negatively charged phosphate ions attached repel each other and stay suspended in the wash water. Detergent phosphate builders also emulsify grease and promote micelle formation. Additionally, they provide alkalinity which is necessary for effective soil removal; perspiration, food residue, and other acid soils interfere with optimum detergency. Phosphates in detergents provide buffering capacity that maintains an ideal range of alkalinity which provides good cleaning action. Too high or too low alkalinity can cause damage to textile fibers, dyes, equipment, or skin. Good cleaning requires the maintenance of alkaline conditions. Tripolyphosphate has the attribute of simultaneously carrying out all these essential functions. At the same time, it fulfills the other requirements of a builder material in detergents in that it is (i) non-toxic, (ii) mild to skin, (iii) harmless to equipment, (iv) safe for textiles, (v) easily removable in wastewater treatment, (vi) compatible with other detergent ingredients, and (vii) economical to consumers.

H. EFFECTS OF THE CONTROL OF DETERGENT PHOSPHORUS INPUTS TO INLAND LAKES

The reasoning behind the control of phosphorus inputs to lakes is that any reduction should ultimately affect water quality, if the system is limited by phosphorus.

Schaffner and Oglesby studied 12 New York lakes to compare different methods of calculating phosphorus loadings and to develop relationships among phosphorus and chlorophyll *a* concentrations, and Secchi disc depths. (Schaffner and Oglesby, 1978 and Oglesby and Schaffner, 1978) Chlorophyll *a* data on six of the lakes from their study, covering pre- and post-ban periods for phosphate detergents in New York State, were used by Trautman et al. (1982) as an example of the use of their statistical methodology for determining data requirements for assessing lake restoration programs. In an analysis of each lake individually, the authors concluded that two years of pre-ban data were insufficient to demonstrate a change in post-ban chlorophyll *a* levels, "no matter how many years of post-ban data could be obtained." When Trautmann et al. employed their statistical analysis to the lakes as a group they concluded there was a probability of only 0.7 percent that a chlorophyll *a* drop at least as large as the one observed would have occurred by chance; they attributed this drop to the phosphate detergent ban. Cause and effect was not established, however. The data showed essentially no correlation (R^2 = 0.001) between the percentage drop in chlorophyll a and the authors' estimates of the percentage drop in the available phosphorus load for the six lakes studied.

Another study was conducted in Indiana by Bell and Spacie (1978). To study the effect of a 1973 detergent-phosphorus ban in Indiana, 15 lakes were chosen from the Environmental Protection Agency's (EPA) National Eutrophication Survey (NES) list. Data collected in 1977 were compared to 1973, but no significant difference in water quality changes could be observed between lakes that received wastewater and those that did not. The reason given was that the phosphorus contribution from detergents was small compared to other phosphorus sources. A similar conclusion was reached in a study of six Minnesota lakes (Runke, 1982).

Studies by Etzel et al. (1975a, b, c) also observed that the Indiana ban yielded little reduction in stream phosphorus. They concluded that the "remaining stream phosphorus levels [during the post-ban period] in regions receiving significant treated-sewage discharges are still far too high [for the ban] to be of any biological consequence".

Prior to implementation of the Wisconsin phosphate detergent ban, the EPA (1978) projected the impact of the ban on inland lakes of the state using data gathered in the NES. Twenty-one Wisconsin lakes included in the NES were selected that received wastewater that was not treated for phosphorus removal and for which nutrient loadings could be determined. All twenty-one lakes selected were eutrophic. To predict the impact of the phosphate laundry detergent ban on the lakes, a 50 percent reduction in the municipal wastewater treatment plant phosphorus load was assumed to result from a phosphate laundry detergent ban (approximately two times the present day average). Mathematical models were then used to predict the impact of such a phosphorus reduction on the water quality of the twenty-one lakes. The report projected that 18 of the 21 lakes evaluated would show no observable changes in water quality and would, therefore, remain eutrophic. It also stated that observable changes in water quality would be expected at two of the lakes (Lakes Delavan and Wapogasset). The effect of the ban on one lake (Butternut) could not be predicted since the primary productivity in the lake appeared to be limited by light or some factor other than phosphorus or nitrogen (Butternut Lake was one of the lakes in the SDA-Wisconsin Lakes Study).

III. METHODS

A. ESTABLISHMENT OF FIELD STUDY SITES

Lake Selection Criteria

Lakes receiving direct or indirect discharges of effluent from wastewater treatment plants or septic tank system seepage, and reference lakes receiving no point source wastewater discharges nor having high numbers of septic tank/tile field systems located on them, were studied. Descriptions of the lakes initially chosen for the study, some of which have subsequently been eliminated from the program, can be found in the protocol for the 1978 study (SDA, 1978). The selection of lakes was based on the best state-wide data available at the time the lakes were chosen.

In the first year of study (1978), water quality data were collected on twelve Wisconsin lakes. During the course of that sampling year, information was received which indicated that further sampling should be discontinued on two of the lakes because of the enactment of programs which would affect water quality and interfere with the interpretation of study results. These two were Shawano Lake in Shawano County, where the lake's shore was sewered, and Noquebay Lake in Marinette County, where an extensive weed harvesting program was conducted during 1978.

Due to water quality trends, and drainage basin and lake characteristics dissimilar from Swan Lake (e.g., aquatic weed dominance in Fish Lake versus phytoplankton dominance in Swan Lake), sampling of Fish Lake, a reference lake, was not continued after 1981. After analysis of the lake data, Little Bearskin Lake was substituted as a suitable reference lake for Swan Lake.

Although Delevan and Wapogasset Lakes were originally identified by the EPA as lakes most likely to change because of the ban (EPA, undated), they were excluded from this study because of impending plans to implement phosphorus removal at the municipal treatment systems at both sites during the study period. Several other candidate lakes were eliminated for the same reason.

Only lakes thought to have the potential to be impacted by phosphorus loadings from nearby municipal treatment plants or septic tank systems were chosen. Lakes receiving at least 30 percent of their annual phosphorus load from nearby municipal treatment plants, as reported by the NES, were considered for inclusion in the study; for lakes not included in the NES, discussions with Wisconsin Department of Natural Resources personnel (Research Bureau and Inland Lakes Renewal Program) were held to ascertain the suitability of candidate point source and septic tank system lakes.

Other lake selection criteria included the following:

1. Only natural lakes were selected; impoundments were excluded from consideration.
2. Because the phosphate detergent ban in Wisconsin was originally for two years, lakes with retention times no greater than two years were selected. The minimum acceptable retention time was 30 days.
3. In selecting lakes, consideration was given primarily to those lakes for which previously collected data were available from such studies as:

   a. The National Eutrophication Survey (NES)
   b. Wisconsin DNR Quarterly Monitoring Program

4. Selection was restricted to those lakes with depths of three meters or more.
5. Lakes were selected which had the public access required for a lake sampling program.
6. Preference was given to lakes which had at least one, but less than four significant tributaries. This criterion allowed for a reasonable monitoring program of the tributaries to be conducted.

Proponents of the Wisconsin phosphate detergent ban contended that the water quality of lakes in the northern part of the state would be protected by the reduced phosphorus input resulting from the ban on phosphate detergents. Lakes in the northern part of the state were reported to have good water quality and, theoretically, would be protected from eutrophication and water quality degradation by a ban on phosphate detergents. In consideration of this, most of the lakes chosen for study were in the northern region of the state.

Reference Versus Test Lakes

Observable water quality changes due to natural causes (e.g., climatic variations) are known to occur from year to year. Therefore, a means to distinguish between effects from natural causes and those of the phosphate detergent ban had to be established. The approach used by this study was to compare the water quality of test lakes, which received municipal wastewater effluent or septic tank seepage, to reference lakes which received a minimal phosphorus loading from these sources. By comparing water quality changes in these two types of lakes, shifts in the water quality of a lake receiving municipal or septic tank/tile field phosphorus loadings could be distinguished from natural changes. The project design was based on the assumption that natural changes (mostly climatic) affect both types of lakes in a similar manner. To enhance similarities in natural effects, reference lakes were chosen that were located in the same regions of the state as test lakes and had morphological and watershed characteristics similar to the test lakes.

Description of Lakes

Figure 2 shows the locations of the lakes selected for this study. Figures A1 through A9 (Appendix A) are maps of each lake. Tables 1, 2 and 3 summarize the pertinent limnological, morphological, and geological characteristics of the lakes, and wastewater treatment plants discharging into the lakes or their tributaries. The test lakes were Swan, Elk, Moss, Townline, Butternut, Balsam, and Enterprise; the reference lakes were Little Bearskin and Teal.

1. Swan Lake is located in Columbia County in the south central region of Wisconsin. The Fox River enters the lake at the eastern end and exits at the western end. The area surrounding Swan Lake is partially developed, although major portions are marshy and unsuitable for development. Shoreline development in 1981 consisted of 88 residences, four townhouses containing four units each, and a golf course. The major land use in the drainage basin is agriculture. The Pardeeville Municipal Wastewater Treatment Plant discharges into Spring Lake, about

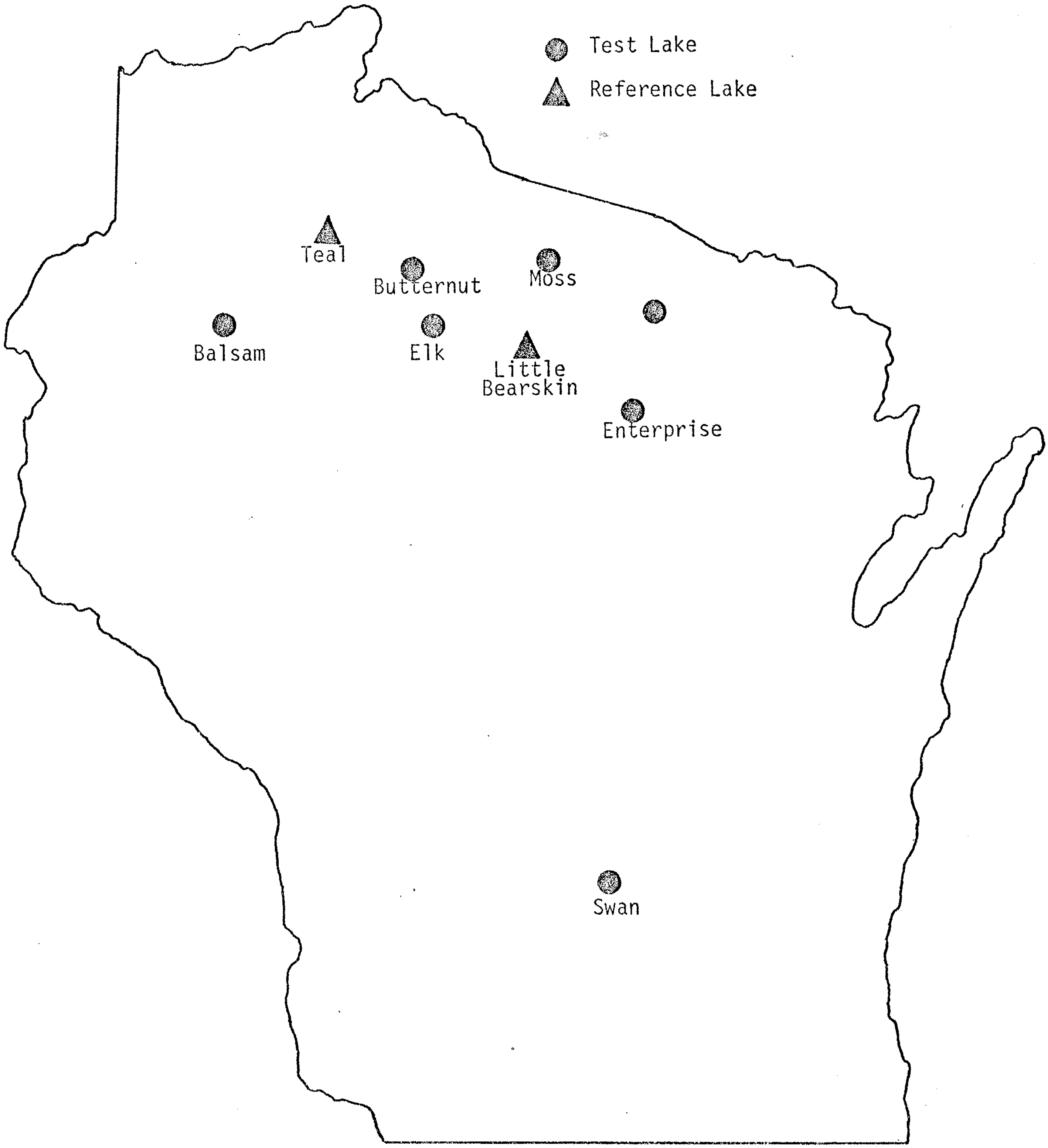

Figure 2: Location of Study lakes

Table 1: Limnological and morphological characteristics of the study lakes.

Lake	Surface Area (a) (ha)	Volume (b) (10**6 cu m)	Mean Depth (c) (m)	Maximum Depth (m)	Mean Hydraulic Residence Time (days) (d)	Tributaries (e) In	Tributaries (e) Out	Shoreline Length (km)
Balsam	107	8.74	8.2	15	67	2	1	11.9
Butternut	400	17.10	4.3	10	184	4	1	18.0
Elk	36	0.55	1.5	6	2	1	1	
Enterprise	194	7.26	3.7	9	600	1	1	9.8
Moss	76	2.36	3.1	9	860	0	1	5.3
Swan	163	16.03	9.8	25	178	1	1	10.5
Townline	61	2.15	3.5	6	270	2	1	3.4
Little Bearskin	63	1.57	2.5	8		1	1	6.8
Teal	405	16.15	4.0	9	300	2	1	16.9

(a): Surface areas were based on values from planimetry of U.S.G.S. topographic maps, data available from the Wisconsin DNR, and data presented on Clarkson Company maps (Kaukauna, Wisconsin).

(b): Volumes were estimated using bathymetric data provided on Clarkson Company maps.

(c): The mean depth equals the lake volume divided by the surface area.

(d): This was calculated using estimated flows for an average year (mean annual) precipitation and runoff (see nutrient budget methods), and lake volume estimates based on bathymetric data and surface areas.

(e): Intermittent streams are not listed as tributaries.

Table 2: Municipal wastewater treatment plant (WWTP) discharges to the study lakes.

Lake	County	Municipal WWTP	Type of Disposal for Treated Sewage
Balsam	Washburn	Birchwood	Land Disposal
Butternut	Price	Butternut	Indirect Discharge to Surface Water
Elk	Price	Phillips	Direct Discharge to Lake
Enterprise	Langlade	None	Septic Tank/ Tile Fields
Moss	Vilas	Lac du Flambeau	Land Disposal
Swan	Columbia	Pardeeville	Indirect Discharge to Surface Water
Townline	Oneida	Three Lakes	Indirect Discharge to Wetlands
Little Bearskin	Oneida	None	None
Teal	Sawyer	None	None

Table 3: Geological characteristics of the test lake drainage basins (a).

Lake	Soil Type	Surficial Geology	Bedrock	Base (BF) or Low (LF) Flow Runoff (cfs/sq mi)	Soil Permeability (in/hr) (b)
Balsam Lake	Sandy loam.	End moraine (till, sand, and gravel) and pitted outwash (sand and gravel); 100 to 150 feet thick.	Sandstone, undifferentiated.	LF: about 0.40	0.8 - 2.5
Butternut Lake	Butternut and Spiller Creek: silt loam. Mud and Schnur Lake Inlets, and Eastern shore: fine, sandy loam.	Ground moraine (clay, silt, sand, gravel, and boulders) about 100 feet thick.	Igneous and metamorphic, undifferentiated.	LF: 0.40 - 0.59	0.2 - 0.8
Elk Lake	Direct Drainage: fine, sandy loam. Inlet Drainage: Northern Region - fine sandy loam. Southern Region - silt loam.	Pitted outwash (sand and gravel); 100 feet thick.	Igneous and metamorphic, undifferentiated.	LF: 0.16 - 0.29	0.8 - 2.5
Enterprise Lake	Fine sandy loam.	All glacial drift 150 to 200 feet thick. Around lake: outwash and ice-contact deposits (sand, sand and gravel). Perimeter of basin: end moraine (till).	Igneous and metamorphic.	BF: <1.0	West Inlet: 0.05 - 0.20 Rest: 0.8 - 2.5
Moss Lake	Fine sand.	Pitted outwash (sand and gravel); about 100 feet thick.	Igneous and metamorphic, undifferentiated.	LF: 0.40 - 0.59	5 - 10
Swan Lake	Sandy loam and loamy sand, some silt loam.	Direct Drainage; glacial lake deposits, 0 to 100 feet thick. Fox River inlet Drainage: ground moraine, 100 to 200 feet thick.	Sandstone.	LF: <0.3	0.2 - 2.5
Townline Lake	Direct Drainage: sandy loam. Townline Cr.: peat, some fine sandy loam and fine sand. Maple Lake: fine sandy loam.	Stratified drift, outwash and ice-contact deposits (sand, sand and gravel) about 100 feet thick.	Igneous and metamorphic.	no data	2.5 - 5

(a) Sources: (1) Wisconsin Geological and Natural History Survey (1916 a,b; 1927; 1947; and 1959).
(2) United States Department of Agriculture (1978).

(b) Enterprise and Townline Lake values were measured under a 0.5 inch head (Oakes and Cotter, 1968).

three miles upstream on the Fox River.

Swan Lake was included in the National Eutrophication Survey (NES, 1974b) and was part of the Wisconsin DNR monitoring program during 1975, 1976 and 1977. Additional water quality data are available through the Inland Lake Renewal Program.

2. Elk Lake is located in Price County in the north central region of Wisconsin. At the southeastern corner of the lake is the lone inlet (from Lake Duroy), and at the northwestern end is the outlet (to Long Lake). The City of Phillips adjoins the lake along its entire southern shore. In 1981, there were seven residences, three industries, and a beach along the shoreline. Effluent from the Phillips Municipal Wastewater Treatment Plant is discharged directly to Elk Lake.

The Wisconsin DNR collected water quality data from Elk Lake twice during 1977. Elk Lake was included in the NES, but a final report was not prepared.

3. Moss Lake is located in Vilas County in the north central region of Wisconsin. The lake has no inlet, and one outlet, a culvert under the road separating Moss Lake from Long Interlaken Lake. The shoreline is partially developed with 40 residences evident in 1981. The land surrounding the lake is largely forested, although some swampy areas are present; agricultural activities are insignificant. Lac du Flambeau uses a stabilization pond, located approximately 0.4 km from Moss Lake, for

wastewater treatment. Presently, the pond is operating as a seepage cell.

The Wisconsin DNR collected water quality data on Moss Lake in 1975, 1976 and 1977. The lake was not included in the NES.

4. Townline Lake is located in Oneida County in the north central region of Wisconsin. The lake has two inlets, Townline Creek and the South Inlet; the former drains a large swamp into which the Three Lakes WWTP effluent is discharged, and the latter receives the flow from Maple Lake. There is an outlet to Planting Ground Lake along the northern shore. In 1981, there were 71 residences along the shoreline. The land immediately surrounding Townline Lake is largely forested.

Townline Lake receives effluent indirectly, via an intervening marsh, from the Three Lakes Wastewater Treatment Plant. The Three Lakes plant is new but does not have phosphorus removal capabilities.

No water quality data are currently available from the Wisconsin DNR, but the lake was included in the NES (NES, 1974c).

5. Butternut Lake is located in Price County in the north central region of Wisconsin. The lake has four inlets and one outlet. The Schnur's Lake inlet drains Schnur Lake which is nearly 100 percent developed; the Mud Lake inlet receives its water largely from several swamps; the Spiller Creek inlet is fed by runoff from forested and agricultural lands in its lower reaches, and swamps in the upper region; and Butternut Creek

receives runoff from forested, agricultural, and swampy lands. The Butternut Municipal Wastewater Treatment Plant discharges its effluent into Butternut Creek less than two miles upstream of the lake.

In 1981, the shoreline is over 50 percent developed with eight resorts and 185 residences. There are a few farms with pastures in close proximity to the lake, as well as several swampy regions.

An EPA report (EPA, undated) indicated that Butternut Lake might show observable water quality changes as a result of the phosphate detergent ban. The Wisconsin DNR collected water quality data on Butternut Lake during 1973, 1974 and 1975. The lake was studied under the (NES, 1974a).

6. Balsam Lake is located in Washburn County in the north-western region of Wisconsin. The lake has two inlets, one from Birch Lake at the northeastern end, and one from Mud Lake along the central portion of the eastern shoreline. The outlet to Red Cedar Lake is at the southern end of the lake. The shoreline is only sparsely developed with about 24 residences located on steep banks along the lake. The lake is surrounded by heavily wooded hills (birch, aspen, maple and a few conifers). The City of Birchwood Municipal Wastewater Treatment Plant seepage cell system is located in the Balsam Lake drainage basin; there is no direct discharge to any surface waters.

The Wisconsin DNR collected water quality data on Balsam

Lake during 1975, 1976 and 1977. The lake was not included in the NES.

7. Enterprise Lake is located in Langlade County in the northeastern region of Wisconsin. The lake has one tributary on the western end draining a sizable marsh and swamp, and an intermittent tributary draining another large swamp located near the southern shore of the lake. The outlet is on the northern side of the lake.

Approximately 50 percent of the shoreline is developed with a Boy Scout camp and 104 residences. In addition to the two large marsh and swamp areas already described, the surrounding land contains numerous swamps and marshes situated among the forested hills; agricultural land is virtually non-existent.

Enterprise Lake was chosen to represent northern lakes receiving phosphorus through septic tank seepage and receives no municipal wastewater effluent.

The lake is part of the Inland Lake Renewal Program and was sampled by the Wisconsin DNR during 1973, 1974 and 1975. It was not included in the NES.

8. Little Bearskin Lake is located in Oneida County in the north central region of Wisconsin. Bearskin Creek enters the lake on the northeastern side and exits at the eastern end. In 1981, there were 43 residences along the shoreline. The small drainage basin is largely forested with some areas of marsh and swamp. Little Bearskin Lake was used as a reference lake for Swan,

Enterprise, Townline and Moss lakes.

The Wisconsin DNR collected water quality data on Little Bearskin Lake during 1973, 1974 and 1975. The lake was not included in the NES.

9. Teal Lake is located in Sawyer County in the northwestern region of Wisconsin. The inlet from Lost Sand Lake is on the western end and the outlet, the Teal River, is on the southeastern side; a tributary, Lynch Creek, enters on the northern shore. In 1981, there were residences and four resort buildings along the shoreline but the lake receives no discharges from wastewater treatment plants. Teal Lake was used as a reference lake for Balsam, Butternut and Elk lakes.

The Wisconsin DNR collected water quality data on Teal Lake during 1975, 1976 and 1977. The lake was not part of the NES.

Wastewater Treatment Plants

Brief descriptions of the wastewater treatment plants discharging directly or indirectly to the test lakes are presented in Table 4. The process descriptions are accurate for the period from 1978 through 1982.

TABLE 4: Wastewater Treatment Plant Facilities
SDA-Wisconsin Lakes Study

<u>Pardeeville, Wisconsin</u> (Swan Lake)
primary settling
high-rate trickling filter
secondary settling
chlorination
(no process changes 1978-82)

<u>Three Lakes, Wisconsin</u> (Townline Lake)
primary settling
rotating biological contactor
secondary settling
anaerobic sludge digestion
sludge drying beds
chlorination
(no process changes 1978-82)

<u>Lac du Flambeau, Wisconsin</u> (Moss Lake)
stabilization pond (lagoon)
(original lagoon was 11 acres in 1978, and was expanded to three cells totaling 17 acres in 1979)

<u>Phillips, Wisconsin</u> (Elk Lake)
primary settling
low-rate trickling filter
secondary settling
chlorination
anaerobic sludge digestion
(no process changes 1978-82)

<u>Butternut, Wisconsin</u> (Butternut Lake)
primary settling
stabilization pond
chlorination
(no process changes 1978-82; is now aerated lagoon with seepage pits)

<u>Birchwood, Wisconsin</u> (Balsam Lake)
contact stabilization
secondary settling
seepage pits
(no process changes 1978-82)

B. FIELD OPERATIONS

Water Samples

Two or three sampling sites, one of which was the deepest site on a lake, were chosen for each lake. Lake sampling commenced as soon as feasible after ice out (within a week after spring turnover) and terminated at fall turnover. Eight sampling trips were made during this time period; twice per month in July and August when algal growths are generally the most rapid, and approximately once per month for the rest of the period. During all years an integrated, two-meter sample of the surface water was obtained at each site by means of a 1.5 inch diameter PVC pipe. The sample was taken by drawing a rubber stopper into the lower end of the pipe with a length of nylon rope, securing the stopper in position. Where the thermocline (identified by means of a temperature probe read every meter) was less than two meters, the integrated sample was taken only above the thermocline. In addition to these samples, a volume proportional sample was collected at the deepest site during 1978, 1980 and 1981. Based upon the calculation of water volume, and the percentage of the total water volume each lake stratum represented, water samples were collected at 10-foot intervals with a 2-liter Kemmerer water sampler, and composited to form a volume proportional sample. To insure that suspended sediments near the lake bottom did not contaminate the sample, the sampler was lowered no closer than one meter above the sediments.

Grab samples of discharges from the Pardeeville (Swan Lake), Three Lakes (Townline Lake), Phillips (Elk Lake) and Butternut (Butternut Lake) wastewater treatment plants were collected during each sampling trip.

Information on effluent flows was collected from records of the Wisconsin DNR and/or from the treatment plant operators. Since the phosphate loadings from septic tank systems around a lake are scattered and generally diffuse, individual septic tank systems were not sampled in this study.

Lake tributary and outlet samples were collected during many sampling trips. The site locations and the years sampled are presented in Table 5.

Additional samples were obtained during the weeks of July 19, August 2 and August 19, 1982, for further nutrient analyses and for use in the algal nutrient enrichment bioassays. These consisted of additional portions of the integrated surface sample at the deepest site of each lake.

All sample bottles were washed with phosphorus-free detergent, rinsed with distilled water, and indelibly labeled with an identification card which contained the location of the sample and the date. Bottles were used for this project only and were labeled with a project identification code. Water samples were preserved and stored from the time of collection until analyzed (Table 6).

Field Measurements and Observations

The date and time samples were collected at the deep site, the general weather conditions, the air temperature and any unusual conditions were recorded on field sheets during each sampling trip. Depth profiles of conductivity, water temperature and dissolved oxygen were taken at the deepest site on each lake at one meter intervals; Secchi disc depths were measured at all lake sites (Table 7).

Although Secchi disc measurements have been a traditional parameter in limnological field studies for many years, its usefulness as a surrogate

Table 5: Years during which inlets, outlets, and wastewater treatment plants were sampled.

Lake	Years Sampled
Swan Lake	
Fox River Inlet	1978, 1980, 1981, 1982
Fox River Outlet	1981, 1982
Pardeeville WWTP	1978, 1980, 1981, 1982
Elk Lake	
Inlet from Lake Duroy	1981, 1982
Outlet to Long Lake	1981
Phillips WWTP	1978, 1980, 1981, 1982
Townline Lake	
Townline Creek	1978, 1980, 1981, 1982
Southern Inlet	1981, 1982
Northern Outlet	1981, 1982
Three Lakes WWTP	1978, 1980, 1981, 1982
Balsam Lake	
Birch Creek Inlet	1978, 1980, 1981, 1982
Outlet to Red Cedar Lake	1981, 1982
Inlet from Mud Lake	1981
Butternut Lake	
Butternut Creek Inlet	1978, 1980, 1981, 1982
Butternut Creek Outlet	1981, 1982
Spiller Creek Inlet	1981, 1982
Butternut WWTP	1978, 1980, 1981, 1982
Mud Lake Inlet	1981
Schnur Lake Inlet	1981
Enterprise Lake	
Western Inlet	1981, 1982
Northern Outlet	1981, 1982
Moss Lake	
Outlet to Long Lake	1981, 1982
Teal Lake	
Lynch Creek Inlet	1981, 1982
Teal River Inlet	1981, 1982
Teal River Outlet	1981, 1982
Little Bearskin Lake	
Northwest Inlet	1981, 1982
Bearskin Creek Inlet	1981, 1982
Bearskin Creek Outlet	1981, 1982

Table 6: Sample handling and preservation procedures.

Parameter	Bottle Type	Handling and Preservation
Total Phosphorus	Nalgene (amber, 60 ml)	Stored at 4 deg. C on ice.
Nitrate+Nitrite-N Ammonia-N Total Kjeldahl-N Alkalinity	Nalgene (amber, 500 ml)	Stored at 4 deg. C on ice.
Total Filterable Phosphorus Filtered Ammonia-N Filtered Total Kjeldahl-N Filtered Nitrate+ Nitrite-N	Nalgene (amber, 500 ml)	Filtered in the field (0.45 μm membrane filter), stored at 4 deg. C on ice.
pH	Nalgene	Stored at 4 deg. C on ice.
Chlorophyll *a*	None	Filtered in the field, (0.45 μm membrane filter), filter stored at less than 0 deg. C on dry ice.
Algal Identification	Glass (amber, 100 ml)	Lugol's solution added in the field.

Table 7: Physical and chemical analyses performed at each lake.

Parameter	Deepest Site	Other Site(s)
Secchi Disc Depth	+	+
Surface Temperature	+	-
Surface Dissolved Oxygen	+	-
Dissolved Oxygen Profile	+	-
Temperature Profile	+	-
Conductivity Profile	+	-

measure of algal growth, chlorophyll a and trophic status remains a controversial subject (Carlson, 1980; Edmondson, 1980; Lorenzen, 1980; and Megard, 1980). The Secchi disc is designed to measure the transparency of water; Secchi disc depth decreases as algal growths increase. However, numerous other factors also affect the Secchi disc reading:

- The Secchi disc loses accuracy as algal growths become low.
- There is considerable variability among different observers.
- Light is attenuated by particulates other than algae.
- The size of the particles suspended in the water column varies, thereby affecting light attenuation.
- The natural coloration of lake water due to the presence of dissolved substances varies (e.g., brown lakes whose color originates from naturally-occurring humic substances).
- Surface light intensity.

In evaluating the results of Secchi disc readings it should be kept in mind that the chlorophyll a should have an inversely proportional relationship with Secchi disc depth. If this is not the case, then any observed increase (or decrease) in Secchi disc depth is most likely due to a decrease (or increase) in light attenuation due to one of the other factors mentioned above.

Secchi disc depths are usually recorded to the nearest foot or half-meter, thereby introducing more vagary into the measurement. A complete discussion of the use of Secchi discs can be found in Hutchinson (1957).

C. CHEMICAL ANALYSES

The analyses performed on the surface water sample collected at the deep site by means of the two-meter integrated water sampler was analyzed for the parameters listed in Table 8; the integrated samples collected at other sites on the lakes were analyzed for the same parameters, except for alkalinity and total Kjeldahl nitrogen. The volume proportional samples were analyzed for only total phosphorus. All the analytical techniques are presented in Table 9.

A portion of the integrated surface samples collected from the deep sites of the lakes during the fourth, fifth, and sixth sampling trips in 1982, was filtered in the field through a 0.45 μm membrane filter and analyzed for the following:

- Total filterable phosphorus
- Filtered ammonia nitrogen
- Filtered total Kjeldahl nitrogen (TKN)
- Filtered nitrate/nitrite nitrogen

These data were used in the nutrient enrichment bioassay studies.

D. LABORATORY BIOLOGICAL ANALYSES

Nutrient Enrichment Bioassays

Nutrient enrichment bioassays were conducted to ascertain which nutrient(s) limited the growth of algae in the study lakes during the peak productivity period of July and August, 1982. For these studies, an additional portion of the two-meter integrated sample was collected on the scheduled sampling trips during the weeks of July 19, August 2, and August

Table 8: Water samples collected for chemical analysis. Unless otherwise specified all lake samples were two-meter integrated samples, and tributary and wastewater treatment plant samples were grab samples.

Parameter	1978 to 1981 Deepest Lake Site	1978 to 1981 Other Lake Site(s)	1978 to 1981 Trib.	1978 to 1981 WWTP	1982 Deepest Lake Site	1982 Other Lake Site(s)	1982 Trib.	1982 WWTP
Total Phosphorus	+	+	+	+(a)	+	+	+	+
Total Phosphorus (volume prop.)	+	-	Not Applicable		-	-	Not Applicable	
Orthophosphate	+	+	+	+(a)	-	-	-	-
Nitrate+Nitrite-N	+	+	+	+(b)	+	+	+	-
Ammonia-N	+	+	+	+(b)	+	+	+	-
Total Kjeldahl-N	+	+	+	+(b)	+	+	+	-
Total Filtered Phosphorus	-	-	-	-	+(c)	-	-	-
Filtered Ammonia	-	-	-	-	+(c)	-	-	-
Filtered TKN	-	-	-	-	+(c)	-	-	-
Filtered Nitrate+ Nitrite-N	-	-	-	-	+(c)	-	-	-
Alkalinity	+	+(a)	-	-	+	+	-	-
Chlorophyll a	+	+	-	-	+	+	-	-
pH	+	+	-	-	+	+	-	-

(a): Not in 1978.
(b): Not in 1978 or 1980.
(c): Peak productivity samples. Only on the 4th, 5th and 6th sampling trips, 1982.

Table 9: Procedures for analyses performed on water samples.

Total Phosphorus	Standard Methods (1978) EPA Methods (1979-1982)	APHA et al. (1975) U.S. EPA (1979)
Orthophosphate Phosphorus	Strickland & Parsons: Murphy & Riley Technique (1978) EPA Methods (1979-1982)	Strickland & Parsons (1968) U.S. EPA (1979)
Nitrite + Nitrate Nitrogen	Strickland & Parsons: Cadmium-Reduction Method (1978) EPA Methods (1979-1982)	Strickland & Parsons (1968) U.S. EPA (1979)
Ammonia Nitrogen	Standard Methods: Indophenol Method (1979) EPA Methods (1979-1982)	APHA et al. (1975) U.S. EPA (1979)
Total Kjeldahl Nitrogen	EPA Methods (1978-1982)	U.S. EPA (1976 and 1979)
Alkalinity	Standard Methods: Titrimetric Methods	APHA et al. (1975)
pH	Standard Methods: Glass Electrode	APHA et al. (1975)
Dissolved Oxygen	Standard Methods: Winkler Method with Azide Modification	APHA et al. (1975)
Chlorophyll *a*	Standard Methods: Trichromatic Methods	APHA et al. (1975)
Phytoplankton Identification and Enumeration	Standard Methods: Sedgewick Rafter Cell	APHA et al. (1975)
Transparency	Secchi disc	

16. Triplicate cultures of Selenastrum capricornutum were established for each of the following additions for each lake:

1. Controls: no additions, lake water only.
2. Enriched with 50 μg P/L using KH_2PO_4.
3. Enriched with 300 μg N/L using ammonium nitrate.
4. Enriched with 50 μg P/L and 300 μg N/L.
5. Enriched with 2 μg/L SiO_2, using $Na_2SiO_3 \times H_2O$.
6. Enriched with 1 mL trace metal solution, no EDTA.
7. Enriched with 1 mL trace metal solution, EDTA added.
8. Enriched with 50 μg P/L, 300 μg N/L, 2 mg/L SiO_2, 1 mL trace metal solution, and EDTA.

A Turner Model 111 fluorometer was used to measure fluorescence. A given nutrient, or set of nutrients, was considered to be enriching if it produced at least 25 percent greater fluorescence than the controls during the first seven to ten days of the study. The three sets of experiments were conducted for 11, 12, and 20 days, respectively.

Algal Identification and Enumeration

One sample per lake, taken at the deepest sampling site, was collected during each sampling trip for identification and enumeration of the algal species. Samples were obtained from the lake surface by means of the two meter integrated water sampler and were preserved in Lugol's solution.

E. STATISTICAL ANALYSES[1]

Three forms of statistical analysis were performed; (i) covariance analysis for each test lake separately, (ii) combined covariance analyses for all test lakes, and (iii) multivariate analysis obtaining multiple comparison estimates for pre- and post-ban differences of interest. Comparisons were made between the test and reference lakes using the logarithms of the measurements of total phosphorus (TP), Secchi disc depth (SD), and chlorophyll *a* (Chl *a*). For each of these forms, an analysis was done for the data from Site 1 (the deepest portion of the lake) and the average of logarithms of Sites 1 and 2.

Covariance Analysis for Each Test Lake

For each test lake, a covariance analysis was performed using the model:

$$\log y_{ti} = \beta_0 + \beta_1 \log y_{ri} + \beta_2 B_i + E_i$$

where:

y_{ti} represents the i-th observation on the test lake

y_{ri} represents the corresponding i-th observation on the reference lake

B_i is a dummy variable with value 0 for pre-ban observations and value 1 for post-ban observations

In this analysis,the quantity of interest is β_2 which represents the post-ban shift in the test lake response after the modelled relationship with the reference lake has been considered.

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[1]The statistical analyses were conducted by John W. Wilkinson, Ph.D., Rensselaer Polytechnic Institute, Troy, N.Y.

Combined Covariance Analysis for All Test Lakes

For a simultaneous examination of all test lakes, a covariance analysis was performed using the model:

$$\log y_{ti} = \beta_o + \beta_1 \log y_{ri} + \sum_{j=1}^{6} \alpha_j D_{ji} + \sum_{j=1}^{6} \gamma_j D_{ji} \log y_{ri}$$

$$+ \delta B_i + \sum_{j=1}^{6} f_j D_{ji} B_i + E_i$$

where:

$$B_i = \begin{cases} 0 & \text{for pre-ban observations} \\ 1 & \text{for post-ban observations} \end{cases}$$

and

$$D_{ji} = \left.\begin{cases} 0 & \text{for pre-ban observations} \\ 1 & \text{for post-ban observations} \end{cases}\right\} \text{ as follows:}$$

	D_1	D_2	D_3	D_4	D_5	D_6
Swan	1	0	0	0	0	0
Balsam	0	1	0	0	0	0
Butternut	0	0	1	0	0	0
Elk	0	0	0	1	0	0
Enterprise	0	0	0	0	1	0
Moss	0	0	0	0	0	1
Townline	0	0	0	0	0	0

In the above model, looking at groups of terms from left to right, the interpretation is as follows. The log test lake response can be considered as the sum of a general intercept, a linear component relationship with the log reference lake response, an adjustment in the intercept for the specific test lake, an adjustment in the slope for the specific test lake, an adjustment in the general intercept for the pre/post-ban, and an adjustment in the intercept for a specific test lake for the pre/post-ban. An analysis partitioning the test lake response variability into assignable sources in the order listed was performed.

Multivariate Analysis/Multiple Comparisons[2]

A general multivariate analysis taking into account the covariance structure of the data was carried out. It complements the preceding two analyses by using statistical procedures which account for possible correlation of the measurements.

To this end, the measurements for total phosphorus for a given lake and year were considered to be a single multivariate variable, or vector, for purposes of analysis. For each post-ban year, the vector analyzed actually consisted of the differences from the corresponding sampling times for the single pre-ban year. In one analysis, the test lakes and the reference lakes were considered together. In another analysis, the test lakes were considered separately. In either case, the vectors of differences were analyzed in a two-way table in which the entries were identified by lake and by year. Simultaneous confidence intervals on the differences were also constructed. Several analyses were done for the logarithms (to

[2]We would like to acknowledge the assistance of A.S. Paulson and T.A. Delehanty with this multivariate analysis.

base 10) of Site 1 measurements, and for the average of the logged Site 1 and Site 2 measurements. The analysis was repeated for chlorophyll a and Secchi disc depth. A detailed description of these analyses is given in Appendix B.

A similar analysis was also performed for each test lake using the differences of the logarithms of the test lake measurements and the corresponding reference lake measurements -- in a sense, an analysis of the test lake data adjusted for potential relationship with its corresponding reference lake.

In addition to the above, simultaneous confidence intervals were constructed for differences of the yearly averages between the post-ban years and the pre-ban year.

F. PHOSPHORUS LOADING ESTIMATES

Phosphorus loadings to a lake can be segmented into tributary input, direct drainage and atmospheric input. The tributary input would result from any runoff from areas which eventually terminate into a stream draining into a lake. Wastewater-derived input would account for both sewered areas which have discharge points into the lake directly, or a tributary to the lake, as well as septic tank/tile field system seepage or system failure. Direct drainage would arise from areas directly adjoining the lake shoreline, areas which do not drain to a lake tributary, but rather drain to the lake itself. For atmospheric loadings, only the wetfall and dryfall which enter the lake surface area directly are included. Thus, the total of these inputs would comprise an estimate of the combined phosphorus loads to a lake from various sources. A complete description of the calculations involved in estimating phosphorus loads during this study are presented in Appendix C; the general procedures are described below. With these phosphorus loadings available for the decisions regarding water quality questions can be more adequately addressed.

Phosphorus loadings to a lake can be segmented into tributary input, direct drainage and atmospheric input. The tributary input would result from any runoff from areas which eventually terminate into a stream draining into a lake. Wastewater-derived input would account for both sewered areas which have discharge points into the lake directly, or a tributary to the lake, as well as septic tank/leach system seepage or system failure. Direct drainage would arise from areas directly adjoining the lake shoreline, areas which do not drain to a lake tributary, but rather drain to the lake itself. For atmospheric loadings, only the wetfall and dryfall which enter the lake surface area directly are included in this category. Thus, the total of these inputs would comprise an estimate of the combined phosphorus loads to a lake from various sources. A complete description of the calculations involved in estimating phosphorus loads during this study are presented in Appendix C.

The areas drained by streams and lakes, and land use areas, were delineated by surface morphology as provided in the most recently updated U.S.G.S. topographic maps.

Stream flows were determined utilizing data presented in the hydrologic budget sections of the U.S.G.S.-University of Wisconsin Geological and Natural History Survey hydrologic atlases (Olcott, 1968; Young and Hindall, 1972; and Oakes and Cotter, 1975), supplemented by normalized stream flows calculated by the U.S.G.S for the National Eutrophication Survey (NES) and National Oceanic and Atmospheric Administration (NOAA) monthly precipitation reports for Wisconsin.

The procedure for determining runoff quantities and flows involved calculating a runoff to precipitation ratio for each year. The value of this ratio for an average year's precipitation was obtained from the U.S.G.S.-University of Wisconsin Geological and Natural History Survey hydrologic atlases or from streamflows provided by the U.S.G.S. to the NES. To correct for the difference between the precipitation of a test year and that of an average year, an assumption was made that the proportion of runoff to precipitation could be modeled by the average basin data for dry, average, and wet years.

Lake tributary phosphorus loadings were estimated using flows and the mean yearly phosphorus concentrations of the tributaries. For streams receiving municipal wastewater treatment plant effluents, the loadings attributed to the streams were determined by subtracting the treatment plant loading from the total measured stream load.

Loadings to treatment plants were assumed to be 1.36 kg P/yr(3.0 lbs P/cap/yr) for untreated wastewater containing detergent phosphorus and 0.96 kg P/cap/yr(2.1 lbs P/cap/yr) for untreated wastewater without detergent phosphorus (SDA, unpublished data). Phosphorus removals due to treatment were considered to be 10, 20, and 90 percent for primary, secondary, and chemical phosphorus removal treatments, respectively, and 100 percent for seepage cells and lagoons.

No direct field measurements were made of septic tank/tile field loadings, therefore several assumptions were made. The results of the 1982 shoreline survey (SDA) were used to estimate the number of septic tank/tile field systems in use. The survey results are presented in the lake description section. All lakeshore dwellings (homes and cabins), resorts, and camps were assumed to have septic tanks (undoubtedly an overestimation). Resorts and camps were considered to be the equivalent of 10 and 20 dwellings, respectively, and to have seasonal occupancy only. Other dwellings were estimated to be 50 percent seasonal and 50 percent year-round residences. A year-round occupancy of 2.5 persons was assumed for each dwelling or dwelling equivalent; seasonal occupancy was considered to be four months per year. The per capita phosphorus load to septic tanks was assumed to be the same as for municipal treatment plants. (The value of 1.36 kg P/cap/yr is probably high for seasonal residences which may not have laundry facilities or other modern appliances). A phosphorus removal of 90 percent before the discharge entered a lake was used for non-failing septic tanks, whereas failing septic systems were considered to have no phosphorus removal capability. A septic tank failure rate of 10 percent was used.

For atmospheric phosphorus loadings (both wetfall and dryfall), the value of 0.175 kg P/ha/yr reported in the NES studies was used since it appeared to be a conservative value. More recent studies (Reckhow et al., 1980) conclude that the NES value probably underestimated the actual loadings. Weather stations were chosen on the basis of precipitation data provided in the U.S.G.S. - University of Wisconsin Geological and Natural History Surveys (Olcott, 1968; Young and Hindall, 1972; and Oakes and Cotter, 1975), and the average (year) precipitation data in NOAA climatological reports.

Direct drainage (the area surrounding a lake which contributes runoff directly to the lake rather than a tributary) loadings were calculated using export coefficients compiled by Reckhow et al. (1980) and the respective coefficients were applied to specific land use areas. The choice of coefficients was based on precipitation and runoff quantities, the areal extent of individual land use types, soil and geological data from the U.S.G.S.-University of Wisconsin Geological and Natural History Survey atlases (Olcott, 1968; Young and Hindall, 1972; and Oakes and Cotter, 1975), soil survey reports (Wisconsin Geological and Natural History Survey, 1916 a,b, 1927, 1947 and 1959; and U.S.D.A., 1978), types of vegetation, and field observations. The export coefficients were generally chosen from the lower end of the concentration range presented in the literature whenever data from more than one study were available, thereby yielding conservatively low estimates for direct runoff.

IV. Results

A. TOTAL PHOSPHORUS CONCENTRATIONS, CHLOROPHYLL A CONCENTRATIONS, AND SECCHI DISC DEPTHS

Tables E1 and E2 summarize the total phosphorus and chlorophyll a concentrations of samples collected in the field and Secchi disc depth measurements made in the field. Table E1 contains the average values for these parameters from Little Bearskin Lake, a reference lake, and its respective test lakes: Enterprise, Moss, Swan and Townline Lakes. Table E2 contains the same information on Teal Lake, a reference lake, and its respective test lakes: Balsam, Butternut, and Elk Lakes. Figures 1-3, A-E in Appendix E are graphical presentations of the monthly mean values of the three parameters. The following is an overview of the changes in these parameters.

Little Bearskin, Enterprise, Moss, Swan, and Townline

In Little Bearskin Lake annual mean total phosphorus concentrations were consistently higher (average of 40 percent higher) in the ban period (1980-82) than before the ban (1978). Chlorophyll a levels were in general, slightly higher during the ban period. Secchi disc depths were an average of about 20 percent greater.

In the ban period total phosphorus concentrations were much greater (more than doubled) in Enterprise Lake than in the pre-ban period. Moss, Swan, and Townline exhibited higher phosphorus concentrations in the first full ban year, 1980, but by 1982 all three exhibited reduced phosphorus concentrations compared to the pre-ban sampling year. This trend in these latter three lakes did not appear to occur in Little Bearskin Lake, the reference lake.

Chlorophyll a concentrations averaged about 50 percent higher in Enterprise Lake during the ban period compared to the pre-ban period. Chlorophyll a concentrations remained relatively unchanged in Moss Lake throughout the study. Chlorophyll a concentrations were higher in the Swan Lake during the first two ban years, 1980 and 1981, but were slightly lower in 1982 compared to the pre-ban period. In Townline Lake, a very slight increase in chlorophyll a was observed in the first two ban years, but a decline relative to the pre-ban period was observed in 1982, the third year of the ban.

Secchi disc depth measurements averaged about 20 percent greater in Little Bearskin Lake in the ban period compared to the pre-ban period. Similarly, Enterprise, Moss, and Townline Lakes also exhibited greater Secchi disc depth readings in all three ban years, 1980-82. Swan Lake, after exhibiting an increase in Secchi disc depth in the first full ban year, 1980, showed reduced readings the succeeding two ban years.

Teal, Balsam, Butternut, and Elk Lakes

Teal Lake, a reference lake, exhibited higher total phosphorus concentrations in all three of the ban years compared to the pre-ban period. Chlorophyll a concentrations were relatively the same in 1978 and 1980 in Teal Lake, but were higher in 1981 and 1982. Secchi disc depth readings were about 50 percent higher in 1980 compared to 1978, but declined to about 1978 levels by 1982.

Similar to Teal Lake, both Balsam and Butternut exhibited higher total phosphorus concentrations in the ban period compared to the pre-ban period. Elk Lake exhibited higher total phosphorus concentrations in 1980, but the concentrations in 1981 and 1982 were lower than pre-ban levels.

Both Balsam and Butternut exhibited higher chlorophyll *a* concentrations in 1981, but lower concentrations in 1980 and 1982 compared to pre-ban concentrations. Elk Lake exhibited chlorophyll *a* concentrations more than double the 1978 levels in all three ban years.

All three test lakes (Balsam, Butternut, and Elk) exhibited greater Secchi disc depth measurements in all three ban years compared to the pre-ban period.

The data obtained on the three parameters during the field study were statistically analyzed to determine if any improvements or deteriorations beyond those observed in the reference lakes occurred in the test lakes. The results of that analysis are presented in Section C.

B. ALGAL COUNTS AND NUTRIENT ENRICHMENT BIOASSAYS

Statistical Analysis of Algal Identification and Enumeration Data

Figures G1A through G1E (Appendix G) show the temporal variation in green algae over the course of the study. The monthly means of the counts of green algae are plotted. Figures G2A through G2E are similar plots for blue-green algae. It is readily apparent from these figures that both green and blue-green algae steadily increased in numbers in both test and reference lakes throughout the study.

Analyses were conducted on logarithms of blue-greens (ln BG), logarithms of greens (ln G), p = BG/BG+G) and ln [p/(1-p)]. The transformations, where used, were for obtaining better agreement with the distributional assumptions for the various analyses. The use of logistic transformation, ln [p/(1-p)], was of marginal value.

Tables G1 and G2 (Appendix G) represent covariance analyses like those done for total phosphorus, chlorophyll *a*, and Secchi disc depth. Observe in Table G1, for all measurements considered, that the apparent tracking of the test lakes by the reference lakes is accounting for a substantial amount of the total variability. Even so, except for the greens, the post-ban effect is statistically significant (although not dramatically so). In Table G2, you will observe that this post-ban effect is primarily associated with Elk Lake, with Butternut also appearing for the blue-greens. In both instances, the direction is associated with increases in the blue-greens that were greater than corresponding reference lakes.

Nutrient Enrichment Bioassays

A summary of the results of the 1982 nutrient enrichment bioassays conducted using samples from all test lakes is presented in Table 10. When interpreting the bioassay results, it should be kept in mind that if the P and P+N treatments (but not N) induced algal growth, then phosphorus was most likely the causative agent; similarly, whenever both the N and P+N treatments (but not P) showed algal growth stimulation, then the addition of nitrogen was most likely the causative factor.

In mid-July, stimulation was observed in only Balsam Lake and Elk Lake water samples. Both lakes supported increased algal biomass when enriched with silicate alone, or an addition of phosphorus and nitrogen. Additionally, phosphorus alone stimulated algal growth in Balsam Lake during the mid-July period.

In early August, phosphorus enrichment alone yielded larger standing crops in waters from Balsam, Butternut, Elk, and Enterprise Lakes; nitrogen additions alone increased growth in Townline Lake; simultaneous addition of phosphorus and nitrogen stimulated biomass levels in Balsam, Butternut, Enterprise, Moss, Swan and Townline Lake waters. In addition, trace metals appeared to be stimulatory in Butternut Lake water.

The results of the mid-August tests were very similar to the early August experiments. However, silicate alone stimulated growth in Elk Lake water and nitrogen alone in Townline Lake Water, and trace metals no longer were observed to yield greater algal biomass in Butternut Lake. Elk Lake water was stimulated by phosphorus as well as a phosphorus and nitrogen addition. Lastly, Townline Lake

Table 10: Results of nutrient enrichment bioassays conducted in 1982.

Key (a)

O :	No effect observed by any treatment.
P :	Stimulation with phosphorus enrichment.
N :	Stimulation with nitrogen enrichment.
P & N :	Stimulation with the simultaneous addition of phosphorus and nitrogen.
Si :	Stimulation with silicate enrichment.
TM :	Stimulation with trace metal enrichment.

Lake	Week of Sampling, 1982		
	July 19	August 2	August 16
Balsam	P, P & N, Si	P, P & N	P, P & N
Butternut	O	P, P & N, TM	P, P & N
Elk	P & N, Si	P	P, P & N, Si
Teal	O	P, P & N	P, P & N
Enterprise	O	P, P & N	P, P & N
Moss	O	P & N	P & N
Swan	O	P & N	P & N
Townline	O	N, P & N	P, N, P & N
Little Bearskin	P & N	P, N, P & N	P, N, P & N, Si

(a): A combination of P and P & N, or N and P & N, suggests phosphorus or nitrogen stimulation, respectively.

also responded to a phosphorus addition (as well as a nitrogen and phosphorus and nitrogen addition).

A comparison of the nutrient enrichment studies, algal counts, and chlorophyll *a* analyses provides valuable insights into algal community dynamics in these lakes. We are especially interested in Swan Lake since more than 10 percent of its 1978 phosphorus load could be attributed to detergent phosphates.

In Swan Lake, a bloom consisting of two diatoms (*Cyclotella* and *Fragilaria*) and a cryptophyte (*Cryptomonas*) occurred in early May, 1982, followed by a lesser growth of blue-greens (*Anabaena* and *Aphanizomenon*) and a green alga (*Ankistrodesmus*) in June. During the months of July, August and September, the lake's waters were dominated by blue-green algae, with peaks in late July of *Anabaena* and *Chroococcus*, and in mid-September of *Aphanizomenon* and *Chroococcus*; between these blooms, a significant decline in cell numbers was observed (mid-August). The nutrient enrichment bioassays, algal counts, and chlorophyll *a* analyses agreed well. The numbers of algal cells and chlorophyll *a* concentrations stabilized in late July and remained fairly constant throughout August. This was the period when bioassays indicated that an addition of both nitrogen and phosphorus was necessary for algal growth to be stimulated. It would be appropriate to conclude that both nitrogen and phosphorus are important nutrients for this lake.

C. STATISTICAL ANALYSES OF TOTAL PHOSPHORUS CONCENTRATIONS, CHLOROPHYLL A CONCENTRATIONS, AND SECCHI DISC DEPTHS[1]

Covariance Analysis for Each Test Lake

Table 11 lists the salient features of the covariance analyses for the

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[1]The statistical analyses were conducted by John W. Wilkinson, Ph. D., Rensselaer Polytechnic Institute, Troy, N.Y.

Table 11. Covariance analysis, Site 1, individual test lakes

		Intercept I	Post-Ban Change in I WI	Std Error of WI	Slope	Std Error of Slope	R^2_{R1} (1)	WR^2_{WI} (2)
Swan	TP	1.44	.01	.12	.17	.24	.02	.00
	SD	.89	-.01	.08	-.08	.24	.01	.00
	Chl a	.99	.26	.17	.02	.18	.00	.08
Balsam	TP	.73	-.04	.09	.61*	.12	.47	.00
	SD	.31	.07	.05	.53*	.14	.40	.04
	Chl a	1.08	-.20	.18	.24	.21	.05	.04
Butternut	TP	1.46	.13	.10	.17	.14	.08	.05
	SD	-.01	.10	.06	.62*	.17	.39	.06
	Chl a	1.46	.01	.21	-.14	.23	0.1	.00
Elk	TP	1.56	-.02	.08	.15	.11	.06	.00
	SD	.03	.13*	.05	.40*	.14	.33	.15
	Chl a	.85	.30	.16	.01	.18	.00	.11
Enterprise	TP	-.33	.20	.12	1.08*	.22	.44	.05
	SD	.06	.04	.06	.86*	.24	.32	.01
	Chl a	-.16	.13	.12	.95*	.17	.51	.02
Moss	TP	-.27	.03	.11	1.06*	.19	.53	.00
	SD	.15	.08	.05	.94*	.13	.63	.03
	Chl a	.24	.04	.11	.40*	.14	.21	.00
Townline	TP	.88	-.07	.08	.52*	.14	.30	.02
	SD	.33	.22*	.05	.22	.21	.12	.35
	Chl a	1.06	-.10	.12	.25	.13	.12	.02

(1) $R_{R1}{}^2$ represents proportion of variability explained by relationship with the covariate reference lake.

(2) WR^2_{WI} represents the increase in the proportion of variability explained by any shift introduced by the ban.

* Statistically significant at at least the 5% level.

logarithms of the responses for the individual test lakes for Site 1 data. The estimate for the change in the intercept associated with the post-ban period is the feature of greatest interest. Only for Elk and Townline Lakes for Secchi disc depth is this change statistically significant. Part of this may be attributed to the small sample sizes and the large variability, encouraging an examination of the test lakes simultaneously using a "dummy variable" approach. This analysis is described below.

Combined Covariance Analysis for All Test Lakes

When considering a covariance analysis of all test lakes simultaneously, additional degrees of freedom become available for statistical testing purposes, giving a greater capability of detecting differences. Certain additional assumptions are needed to use this method. However, an analysis of the residuals after fitting the model did not suggest that these additional assumptions were unreasonable.

Separate analyses were performed for the logarithms of Site 1 data only and for the averages of the logarithms for Site 1 and Site 2 data. The latter displayed slightly less variability than the former and for the sake of brevity and without loss of clarity, it is the only analysis that is reported.

Table 12 provides a summary of the analysis of variance for each of total phosphorus (TP), Secchi disc depth (SD) and chlorophyll a (Chl a). For the corrected total sum of squares, the variability was partitioned sequentially into the following components: reference lake, intercept adjustment for different test lakes, adjustment of slope of reference lake variables for different test lakes, and finally, intercept adjustment for

Table 12: Analysis of variance of the average of the logarithms of Site 1 and 2 data.

Lake	Measurement	S. S.	D. F.	Mean Square	Test Statistic	R^2	Test R^2
Reference Lake	TP	5.47	1	5.47	243.9*	.25	.25
	SD	1.14	1	1.14	67.1*	.10	.10
	Chl *a*	.85	1	.85	6.8*	.02	.02
Intercept Adjustment for Test	TP	5.43	6	.91	22.2*	.50	.25
	SD	5.63	6	.94	55.3*	.61	.51
	Chl *a*	9.66	6	1.61	12.9*	.27	.25
Slope Adjustment for Test Lakes	TP	2.26	6	.38	9.3*	.60	.10
	SD	.31	5	.06	3.5*	.64	.03
	Chl *a*	2.29	6	.38	3.0*	.33	.06
Intercept Adjustment for Post/Pre-Ban	TP	.45	7	.064	1.6	.62	.02
	SD	.46	7	.066	3.9*	.68	.04
	Chl *a*	1.02	7	.146	1.2	.35	.02
Residual	TP	8.27	203	.041			
	SD	3.49	203	.017			
	Chl *a*	25.37	203	.125			

*Statistically significant at at least the 5% level.

Table 12: Analysis of variance of the average of the logarithms of Site 1 and 2 data.

Lake	Measurement	S. S.	D. F.	Test Mean Square	Test Statistic	R^2	R^2
Reference Lake	TP	5.47	1	5.47	243.9*	.25	.25
	SD	1.14	1	1.14	67.1*	.10	.10
	Chl a	.85	1	.85	6.8*	.02	.02
Intercept Adjustment for Test	TP	5.43	6	.91	22.2*	.50	.25
	SD	5.63	6	.94	55.3*	.61	.51
	Chl a	9.66	6	1.61	12.9*	.27	.25
Slope Adjustment for Test Lakes	TP	2.26	6	.38	9.3*	.60	.10
	SD	.31	5	.06	3.5*	.64	.03
	Chl a	2.29	6	.38	3.0*	.33	.06
Intercept Adjustment for Post/ Pre-Ban	TP	.45	7	.064	1.6	.62	.02
	SD	.46	7	.066	3.9*	.68	.04
	Chl a	1.02	7	.146	1.2	.35	.02
Residual	TP	8.27	203	.041			
	SD	3.49	203	.017			
	Chl a	25.37	203	.125			

*Statistically significant at at least the 5% level.

post/pre-ban effect. Another way of expressing this is that one adjusts the total test lake response variability for potential relationship with the corresponding reference lake response and for individual test lake differences and then examines for the effect of imposition of the ban. Only the Secchi disc depth measurement showed a detectable variation between the pre- and post-ban values at a five percent level of significance. The reader should refer to the Methods Section (Field Measurements and Observations) for a discussion of the use of Secchi disc readings in estimating algal growths and a lake's trophic status.

Some additional information of potential interest that can be obtained from Table 12 is the proportion of the variability explained by various groups of terms in the model. Those are summarized in Table 13.

Also by inspection of the column under R^2 in Table 12, one can assess the proportion of variability in the data explained by the model. The model appears to do much better in this respect for total phosphorus (.62) and Secchi disc depth (.68) than it does for chlorophyll a (.35).

Table 14 lists each test lakes' estimates of the slope coefficients for the corresponding reference lake as well as estimates of the amount of shift in the model after the imposition of the ban. The estimated standard deviations of these estimates are listed in parenthesis. Asterisks (*) are used to indicate statistical significance at at least the five percent level.

A shift associated with the ban was detectable at the five percent level in only 4 of the 21 cases, namely for total phosphorus in Enterprise Lake, Secchi disc depth in Elk and Townline Lakes, and chlorophyll a in Elk Lake. In two of these cases, total phosphorus for Enterprise Lake and

Table 13: Proportion of variability explained by various sources.

Source	Measurement	Proportion of Variability Explained by Model	Proportion of Total Variability
Reference Lake Covariate	TP	0.40	0.25
	SD	0.15	0.10
	Chl *a*	0.06	0.02
Test Lake Differences	TP	0.57	0.35
	SD	0.79	0.54
	Chl *a*	0.86	0.30

Table 14: Estimates of the post/pre-ban shift and slope-coefficient for corresponding reference lake for the average of the logarithms of site 1 and site 2 data.

	TD		SD		Chl a	
Lake	Post/Pre Δ Intercept	Slope	Post/Pre Δ Intercept	Slope	Post/Pre Δ Intercept	Slope
Swan	-.08 (.09)	.35* (.16)	.05 (.07)	.11 (.19)	.14 (.15)	-0.1 (.19)
Balsam	-.01 (.09)	.55* (.11)	.10 (.06)	.52* (.17)	-.15 (.15)	.45* (.20)
Butternut	.06 (.09)	.17 (.11)	.10 (.06)	.63* (.17)	.01 (.15)	.09 (.20)
Elk	-.09 (.09)	.18 (.11)	.17* (.06)	.38* (.17)	.33* (.15)	.07 (.20)
Enterprise	.22* (.09)	.87* (.16)	.08 (.06)	1.02* (.17)	.08 (.15)	1.04* (.23)
Moss	-.07 (.09)	1.05* (.11)	.11 (.06)	.48* (.19)	-.03 (.15)	.25 (.24)
Townline	-.03 (.03)	.65* (.17)	.08* (.03)	.60* (.14)	-.01) (.06)	.59* (.18)

* Statistically significant at at least the five percent level

chlorophyll _a_ for Elk Lake, a positive direction in the post-ban shift is not something that could be attributable to the ban. Hence, from this analysis, the only effect that appears to be associated with the ban is for Secchi disc depth. The relative magnitude of this shift is approximately 10 percent, and, although statistically significant, a question could be raised about the meaningfulness of its significance. Again, the reader should refer to the Methods Section (Field Measurements and Observations) for a discussion of the accuracy of Secchi disc readings in estimating algal growth and a lake's trophic status.

The number of slope estimates that are statistically significant is an indicator that the relation of the reference lakes to the test lakes is accounting for a statistically significant proportion of the variability. These data were useful in making the analysis more sensitive. However, the amount of variability not explained by this relationship is larger still.

Multivariate Analysis/Multiple Comparisons

The estimated differences for corresponding dates between post- and pre-ban measurements and their simultaneous confidence intervals are best presented graphically. Figure 3 shows the results for the three analyses using Site 1, and Figure 4 the results for the three analyses combining Sites 1 and 2 for test and reference lakes combined. Figures 5 and 6 present similar analyses for test lakes only. For each response variable and year, the left curve is the lower confidence bound, the middle curve the estimated contrast value, and the right curve the upper confidence bound. A vertical "no effect" line passes through zero.

It is clear from Figures 3 and 4, and 5 and 6, that the ban has not

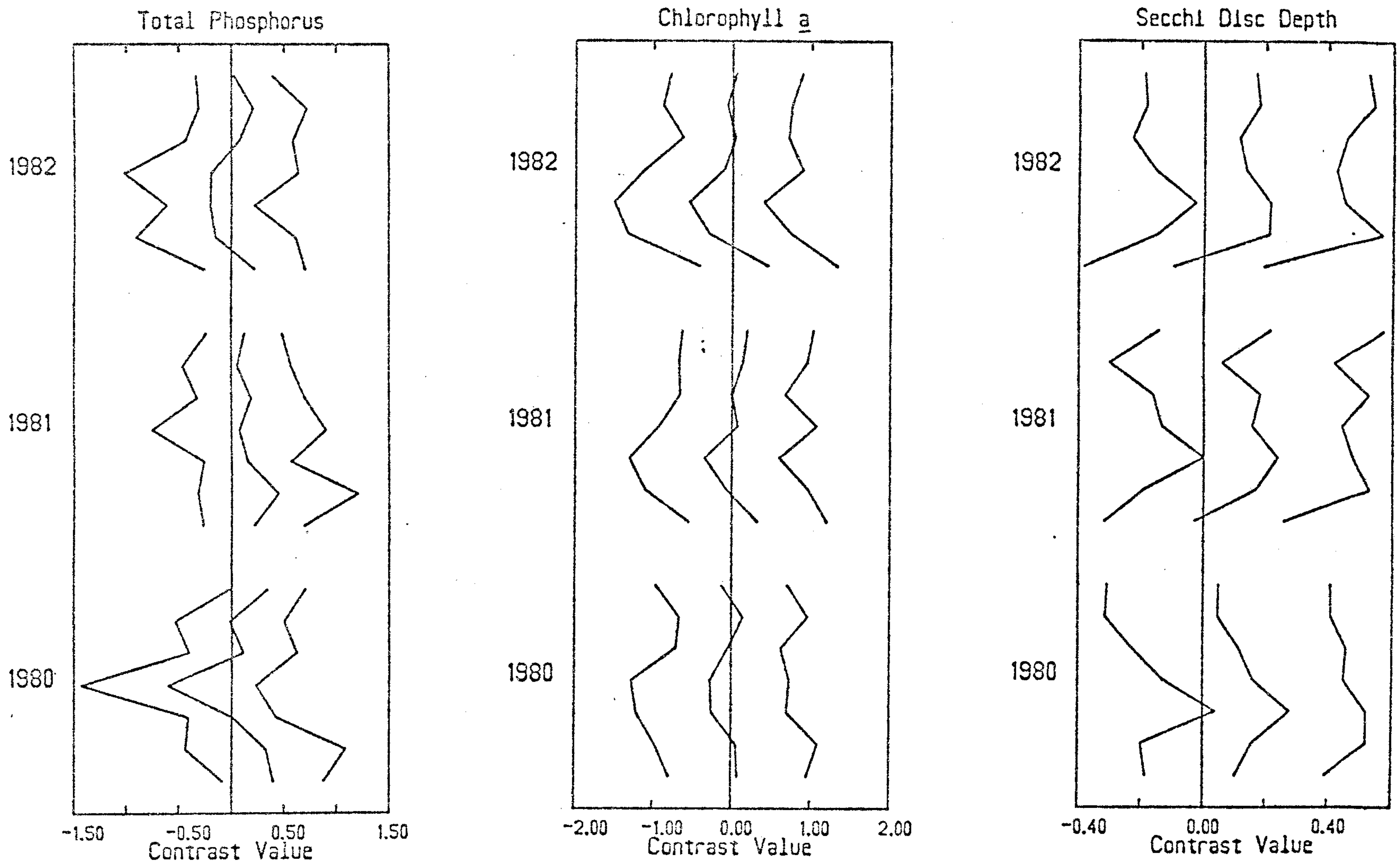

Figure 3: Estimated differences and 95% simultaneous confidence bounds, effects of post-ban years related to pre-ban. Data are log measurements from site 1.

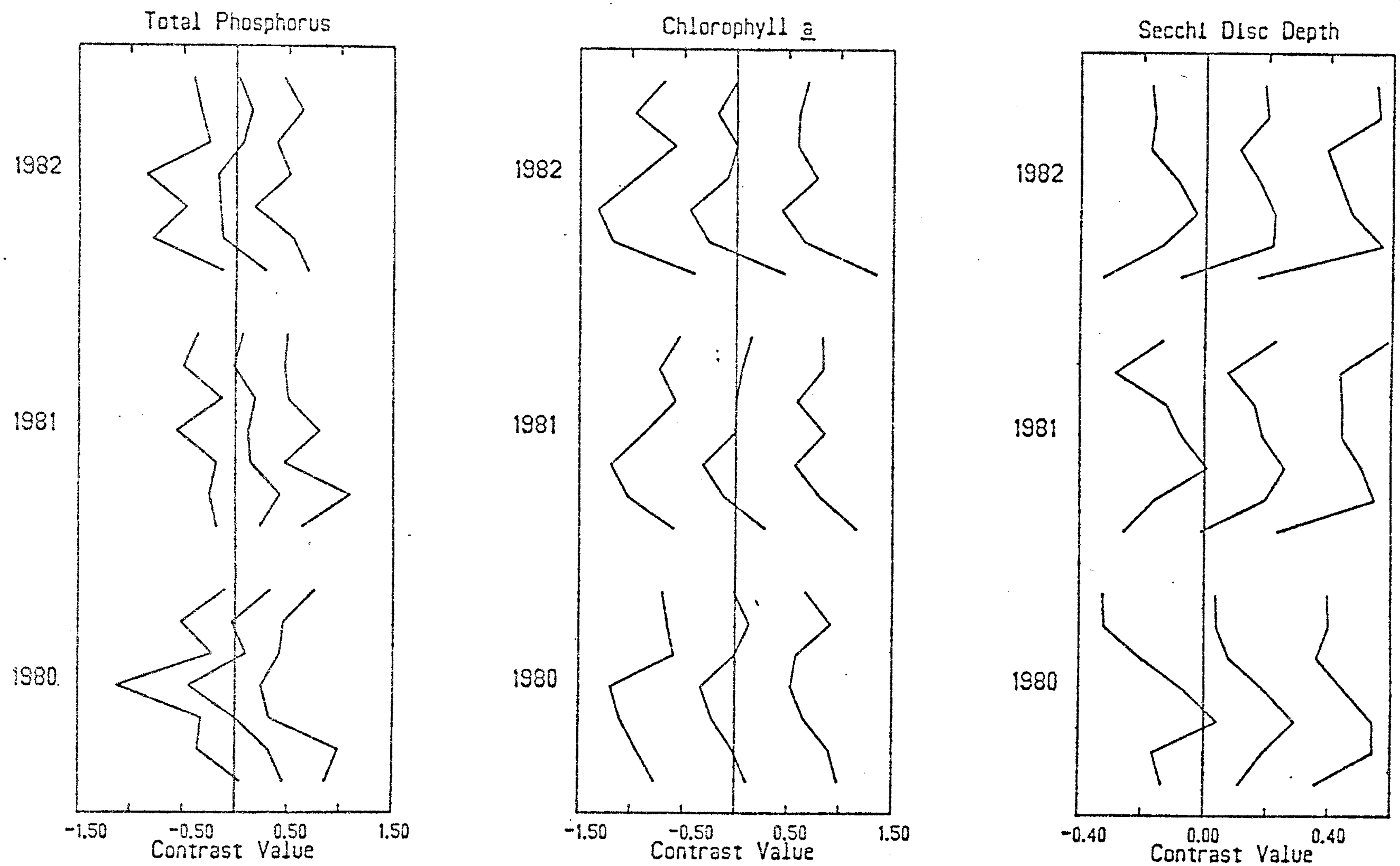

Figure 4: Estimated differences and 95% simultaneous confidence bounds, effects of post-ban years relative to pre-ban. Data are average of log measurements, sites 1 and 2.

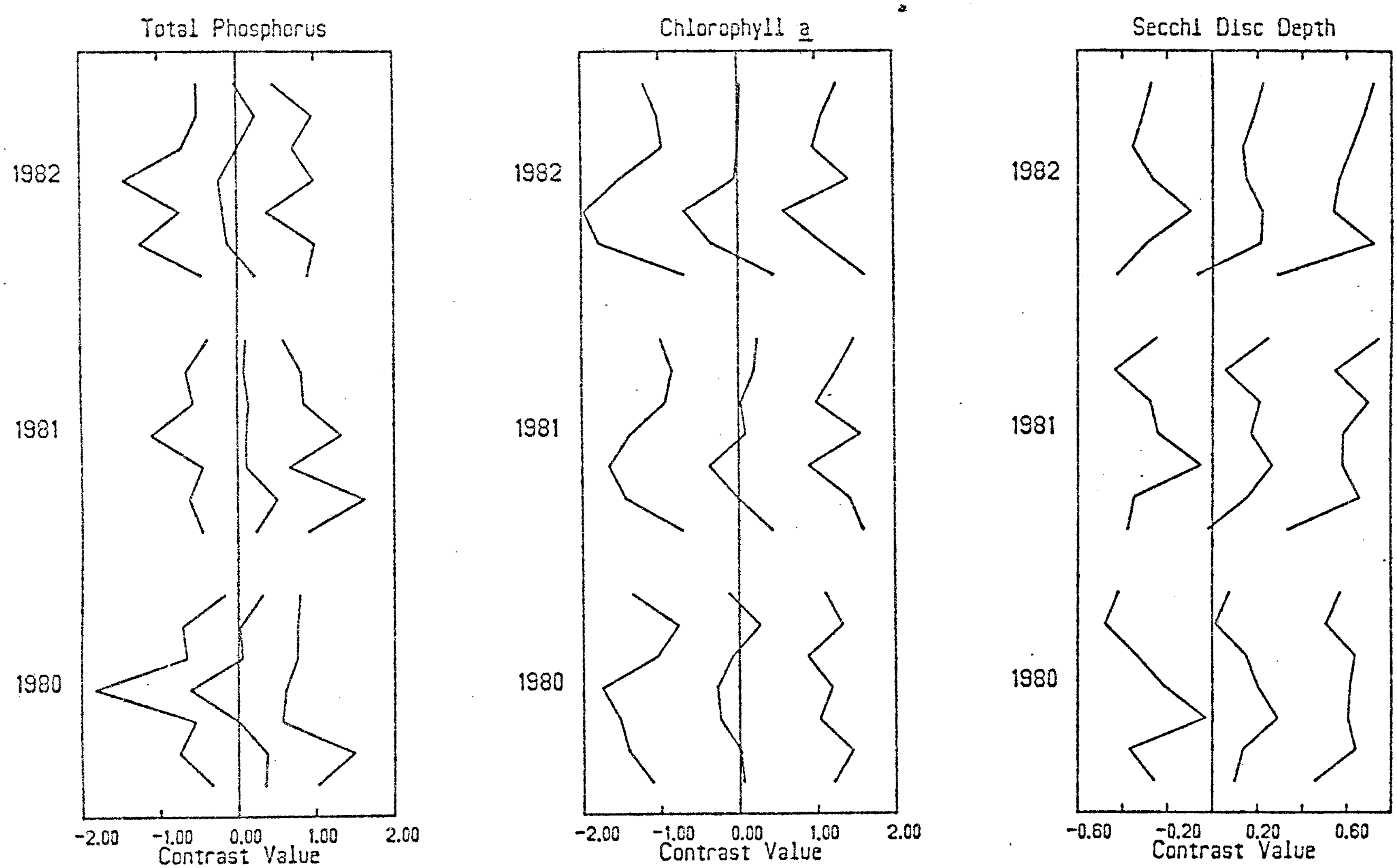

Figure 5: Estimated differences and 95% simultaneous confidence bounds for post-ban year effects. Analysis of test lakes only, log site 1 measurement.

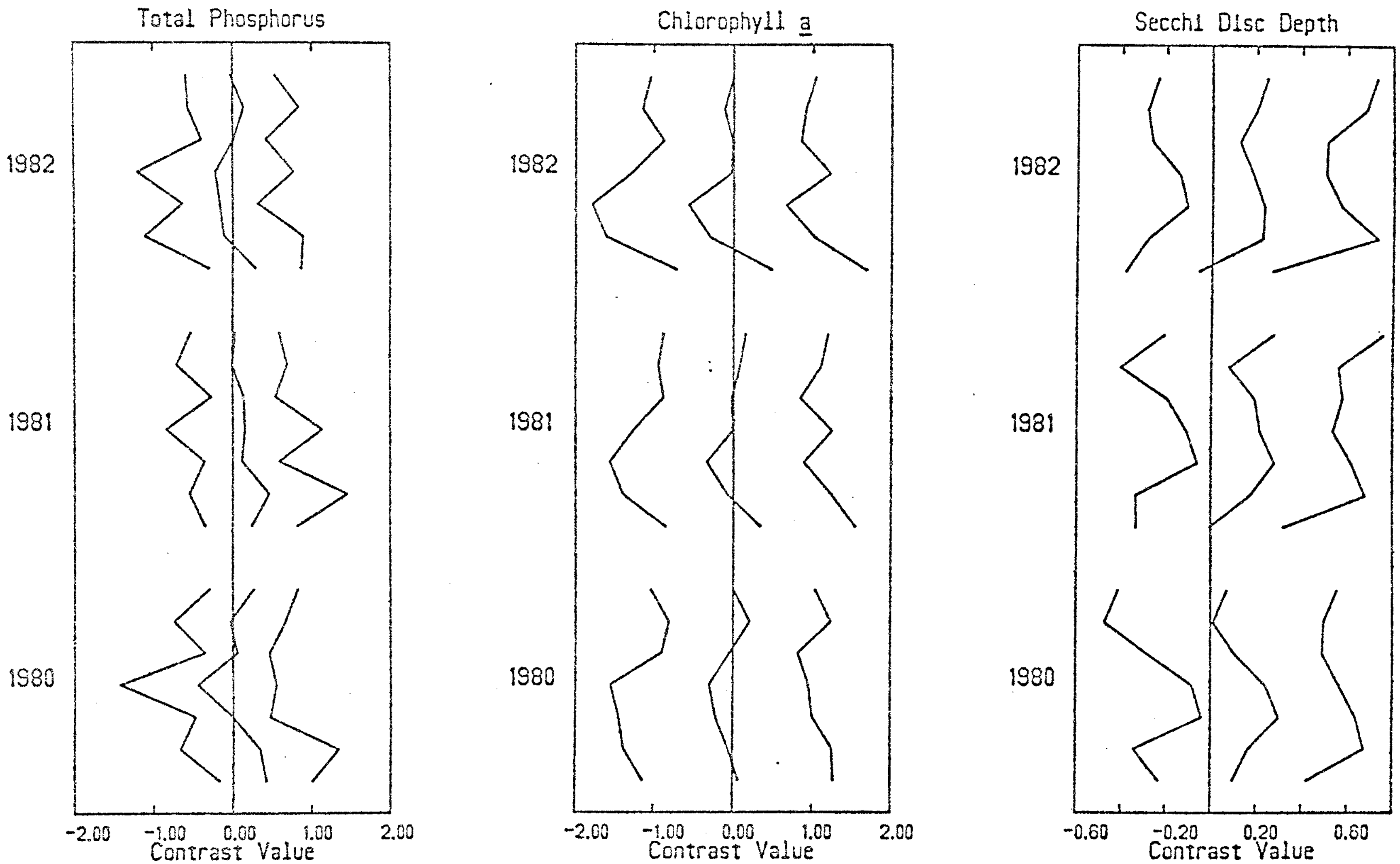

Figure 6: Estimated differences and 95% simultaneous confidence bounds for post-ban year effects. Analysis of test lakes only, log geometric mean of site 1 and 2 measurements.

had a statistically significant effect on total phosphorus, chlorophyll a, or Secchi disc depth, although the general positive nature of the estimate for the latter for all post-ban years may support an indication of some effect for Secchi disc depth.

Instead of obtaining interval estimates at each point in time during the post-ban years, narrower intervals with the same level of confidence are obtainable for the means for each of the years. These are summarized in Table 15. For total phosphorus and chlorophyll a, none of the differences between the post-ban years and the pre-ban year are statistically significant. Although only one difference is statistically significant at the five percent level for Secchi disc depth, all are pointing in the direction of Secchi disc depth being improved following the ban.

The multivariate analyses of these earlier sections were completely redone on data constructed from the differences of the log test lake responses and the corresponding log reference lake responses. Graphs of the simultaneous confidence intervals on the differences between post- and pre-ban years for each point in time that was sampled are given in Figure 7 for Site 1 only and in Figure 8 for the geometric mean (GM). In this analysis no effect of the ban is observable.

For the differences between test lake and corresponding reference lake responses, the year average differences betweeen the post-ban years and the pre-ban year were estimated and 95 percent simultaneous confidence intervals constructed. Those are summarized in Table 16. Here too, no post-ban effect is observable.

Table 15: Year average differences of the post-ban years with the pre-ban year and simultaneous 95 percent confidence intervals (C.I.).

Measurement	Model	C.I. Mid Points 1980	1981	1982	C.I. Half Widths
Total Phosphorus	all lakes, Site 1	.081	.179	-.001	.258
	all lakes, GM**	.101	.165	-.010	.259
	test lakes, Site 1	.076	.184	-.011	.377
	test lakes, GM	.086	.160	-.011	.379
Chlorophyll <u>a</u>	all lakes, Site 1	-.058	.046	-.070	.345
	all lakes, GM	-.049	.007	-.075	.336
	test lakes, Site 1	-.056	.088	-.079	.470
	test lakes, GM	-.049	.020	-.078	.498
Secchi Disc Depth	all lakes, Site 1	.131	.142	.132	.170
	all lakes, GM	.134	.155*	.145	.153
	test lakes, Site 1	.137	.156	.152	.226
	test lakes, GM	.142	.173	.164	.190

* Statistically significant at at least 5% level.

**GM stands for the logarithm of the geometric mean of Sites 1 and 2

Table 16: Year average differences of the post-ban years with the pre-ban year and 95 percent simultaneous confidence intervals C.I. for the differences between test lake and corresponding reference lake data.

Measurement	Model	C.I. Mid Points 1980	1981	1982	C.I. Half Widths
Total Phosphorus	Site 1	-.009	.022	-.042	.398
	GM*	-.061	-.020	-.088	.393
Chlorophyll a	Site 1	.020	.192	-.029	.542
	GM	.004	.059	.003	.549
Secchi Disc Depth	Site 1	.030	.066	.079	.282
	GM	.038	.082	.072	.262

* GM stands for the logarithm of the geometric mean of Sites 1 and 2

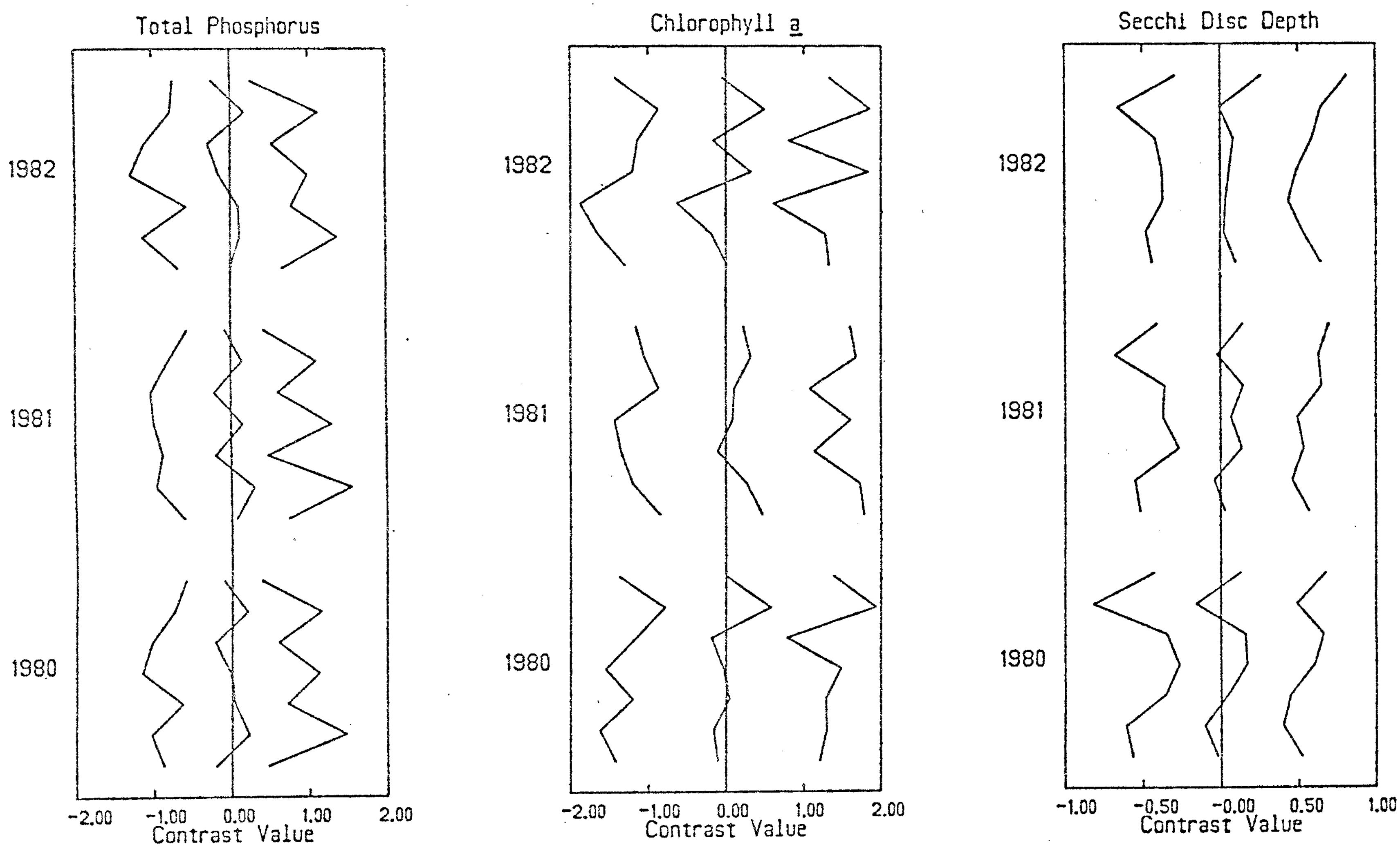

Figure 7: Post-Ban/Pre-Ban differences and 95% confidence bounds for difference of test lakes and reference lakes. Site 1 only.

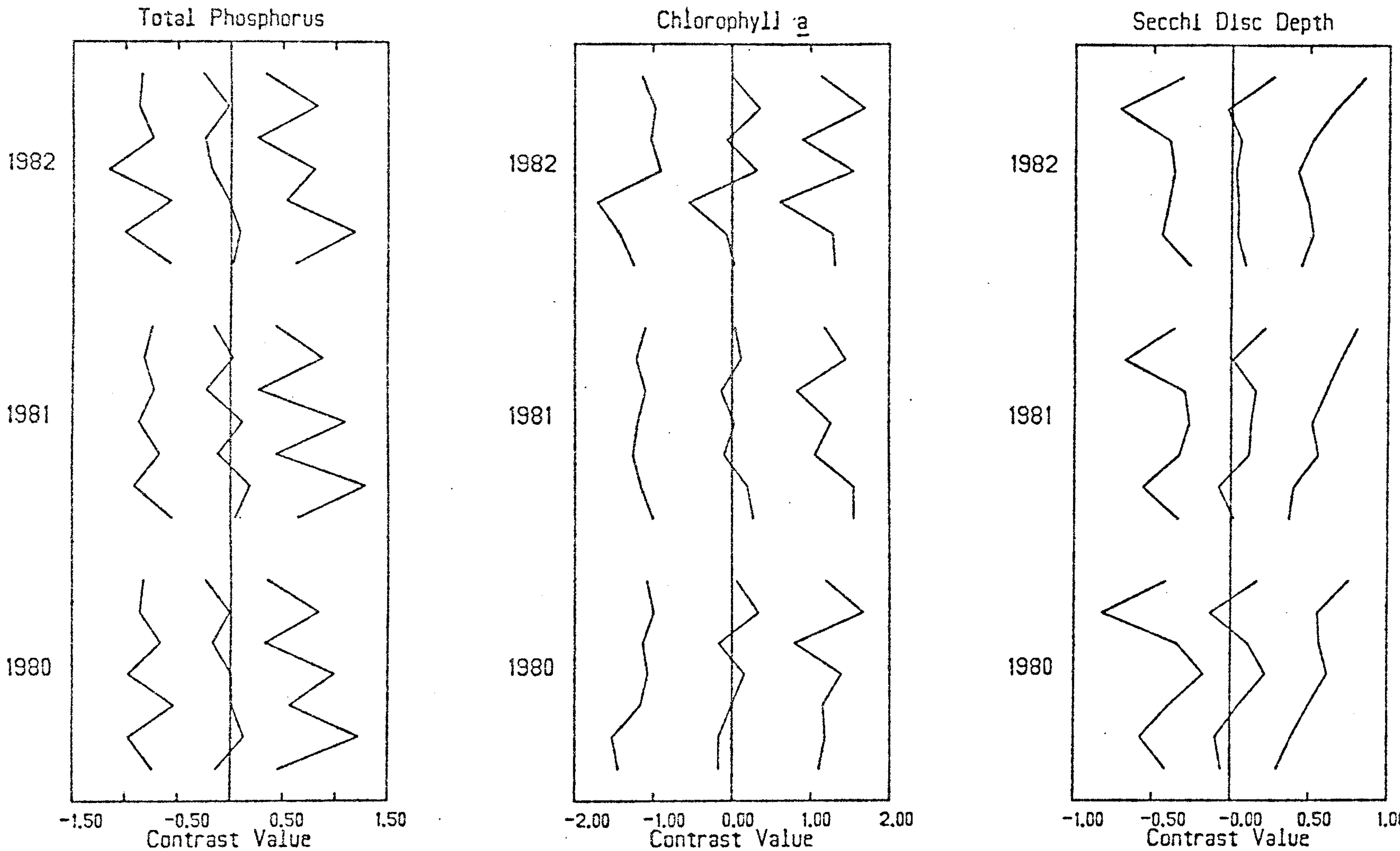

Figure 8: Post-Ban/Pre-Ban differences and 95% confidence bounds for difference of test lakes and reference lakes. Log-GM of sites 1 and 2.

D. PHOSPHORUS LOADINGS INTO TEST LAKES

The results of calculations for runoff and runoff rates, and the land use measurements for each lake are shown in Tables D1 through D21. The phosphorus nutrient budgets (Tables D22 through D28) show the various sources of phosphorus loads to each lake and their magnitude. Separate calculations were made to show the phosphorus budget for the case in which phosphate detergents were in use (i.e., the actual situation in 1978), and for the case in which phosphate detergents were not being used (i.e., if the phosphate detergent ban had been in effect). The changes in phosphorus loading which could be attributed to a cessation of the use of phosphate detergents, expressed as a percentage of the total load with phosphate detergents in use, are also exhibited. In reading the tables it is helpful to note that only the wastewater treatment plant and septic tank loadings were affected by detergent phosphates.

V. DISCUSSION

A. PHOSPHORUS LOADS AND THE EFFECTS OF DETERGENT PHOSPHORUS ON THE TROPHIC STATUS OF THE TEST LAKES

When considering the importance of detergent phosphorus, it should be remembered that detergent phosphorus represents 30 percent, or less, of the total phosphorus concentration in untreated, domestic wastewater. Another factor to be considered, especially for Enterprise and Moss Lakes, is that many of the dwellings with septic tanks probably do not have laundry facilities or other modern appliances, thereby reducing the impact of detergent phosphates as a nutrient source.

In 1978, the reduction in the yearly phosphorus loadings to the test lakes due to a phosphate detergent ban is estimated to have been less than 7 percent of the total loads, with the exception of Swan Lake where a reduction of 12 percent was estimated to have occurred (Tables D22-D28 and Table 17). The principal reason for the relative insignificance of phosphorus originating from phosphate detergents was the large, non-point loadings attributable to the tributaries and runoff from the direct drainage areas of the lakes.

When calculations of actual loadings with and without the presence of detergent phosphates are compared to the loading criteria proposed by Vollenweider (1975), it is readily seen that the removal of phosphorus from detergents would not have any significant effect on the trophic status of the lakes (Table 18 and Appendix H). With the exception of Enterprise and Moss Lakes, reductions in the total yearly phosphorus loads of 69 to 86 percent would be required to attain Vollenweider's "permissible" loading rate and none would meet Vollenweider's critical criterion even if 100 percent removal of phosphorus was attained at the wastewater treatment plants and in septic tank/tile field systems. Therefore, even a complete elimination of WWTP and septic tank phosphorus loads to these lakes would not be sufficient to meet Vollenweider's "permissible" loading rates.

Table 17: Total phosphorus loadings to test lakes due to wastewater and phosphate detergents.

Lake	County	Percent of Current Phosphorus Load Attributable to Wastewater	Percent of Current Phosphorus Load Attributable to Phosphate Detergents
Balsam Lake	Washburn	<1	<1
Butternut Lake	Price	23(a)	7
Elk Lake	Price	22	6
Enterprise Lake	Langlade	13	4
Moss Lake	Vilas	19	6
Swan Lake	Columbia	40(a)	12
Townline Lake	Oneida	12	4

(a): Implementation of phosphorus removal is planned.

Table 18: Comparison of the actual 1978 loading of phosphorus to the test lakes to calculated values of Vollenweider's (1975) permissible and critical loading value.

	Balsam lake det P with	Balsam lake det P w/o	Butternut Lake det P with	Butternut Lake det P w/o	Elk Lake det P with	Elk Lake det P w/o	Enterprise Lake det P with	Enterprise Lake det P w/o	Moss Lake det P with	Moss Lake det P w/o	Swan Lake det P with	Swan Lake det P w/o	Townline Lake det P with	Townline Lake det P w/o
Hydraulic Residence Time(yr)	0.17		0.50		0.004		1.09		2.08		0.36		0.61	
Actual Load (a) (g/sq m/yr)	3.00	3.00	0.62	0.57	21.2	19.9	0.14	0.13	0.08	0.08	2.74	2.42	1.08	1.04
Permissible Load (g/sq m/yr)	0.58		0.19		3.85		0.14		0.12		0.37		0.16	
Critical Load (g/sq m/yr)	1.16		0.37		7.70		0.27		0.23		0.74		0.31	
% Reduction in 1978 Load Required to Reach the Permissible Load.	81		69		82		0		0		86		85	

(a): A loading less than the permissible value suggests an oligotrophic lake, between the permissible and critical values a mesotrophic lake, and greater than the critical value a eutrophic lake.

Nutrient enrichment limitation bioassays provided additional evidence that even large reductions in phosphorus loads to Swan Lake might not result in observable improvements in water quality. In Swan Lake water, both phosphorus and nitrogen enrichment was required for algal growth stimulation to be observed in August. The mid-August bioassay results were supported by chlorophyll a analyses and algal counts of lake water samples which suggested that algal growth did not occur during at least the first half of August, 1982.

For Townline Lake, the nutrient enrichment bioassays indicated that nitrogen was required for increased algal biomass in early August, 1982, and phosphorus or nitrogen was required in mid-August. The chlorophyll a analyses and algal cell counts supported these results showing a decline in chlorophyll a concentrations and cell numbers starting in early August. Neither orthophosphate nor total dissolved phosphorus analyses were conducted routinely in 1982, but the filtered peak productivity samples collected on July 21 and August 4 were analyzed for total dissolved phosphorus (TDP) and total soluble inorganic nitrogen (TSIN, i.e. nitrate, nitrite, and ammonia nitrogen). Although N:P ratios are usually calculated as the ratio of TSIN to orthophosphate, the TSIN:TDP ratio which can be calculated from the available Townline Lake data can also be instructive. The TSIN and TDP concentrations on July 21 were 28 ug N/L and 4 ug P/L, yielding a TSIN:TDP ratio of 7:1; the concentrations on August 4 were less than 60 ug N/L and 48 µg P/L providing N:P ratio of less than 2:1. Similar observations were made during the NES study. Such low levels of total soluble inorganic nitrogen relative to total dissolved phosphorus are highly suggestive of nitrogen limitation. These observations could explain why Townline Lake has not experienced major water quality problems although Vollenweider's phosphorus loading criteria calculations would categorize Townline Lake as eutrophic. A reduction in the total phosphorus loading by 4 percent could not be expected to have a major effect on this Lake.

Algal growth was stimulated by phosphorus additions to Balsam Lake water on all bioassay dates. Therefore, a reduction in phosphorus in July and August might reduce the algal biomass. However, the phosphorus nutrient budget for 1978 showed detergent phosphorus accounted for less than 1 percent of the total load. Due to the very small reduction in phosphorus loads which would occur as a result of a detergent phosphorus ban, no observable effect would be expected.

Regarding hydraulic residence time, the situation at Elk Lake was different than expected from the data available prior to the study. The hydraulic residence time of the lake was found to be less than five days. This rapid turnover of lake water creates conditions under which a large, relatively constant supply of phosphorus is entering the lake; the mean total phosphorus concentrations in the inlet during 1981 and 1982 were 0.054 and 0.057 mg P/L, respectively, and the mean orthophosphate concentration was 0.015 mg P/L in 1981. The nutrient enrichment bioassays indicated stimulation with both nitrogen and phosphorus was necessary to increase algal biomass in mid-July, but phosphorus additions alone stimulated algal growth throughout August. Silicate also yielded increased standing crops in mid-July and mid-August; this is an important observation since diatoms, which require appreciable quantities of silicate, were an important component of the algal community in Elk Lake. Due to the rapid flushing of the lake, constant replenishment of phosphorus from the inlet, and possible silicate limitation, detergent phosphorus would not be expected to have a significant effect on the trophic status of Elk Lake.

Butternut Lake showed that phosphorus stimulated algal growth in August and, therefore, a reduction in the algal standing crop might result from a reduction in the phosphorus loads during those periods. However, as evidenced by Vollenweider's (1975) loading criteria, the improvement due to the removal of detergent phosphorus would not be significant (7 percent) and the lake would remain in a eutrophic state.

Both Enterprise Lake and Moss Lake had phosphorus loadings within Vollenweider's permissible range, suggesting oligotrophic conditions. Detergent phosphorus accounted for 0.01 g p/m^2/yr, or less, in both these lakes, a quantity of questionable importance relative to the total loadings (0.14 g p/m^2/yr at Enterprise and 0.08 g P/m^2/yr at Moss). Slight water quality improvements might occur at Enterprise Lake since phosphorus was shown to stimulate algal growth in the August nutrient bioassays. In Moss Lake water collected in August, increases in algal biomass occurred only when both phosphorus and nitrogen were added. Moss Lake chlorophyll a concentrations were consistently low throughout the study. Therefore little, if any, change in trophic status could be expected at Enterprise and Moss Lakes due to the removal of phosphorus from detergents.

B. EFFECTS OF DETERGENT PHOSPHORUS ON THE SPECIFIC WATER QUALITY PARAMETERS TOTAL PHOSPHORUS, CHLOROPHYLL A AND SECCHI DISC DEPTH

The employment of reference lakes having negligible influence from wastewater was useful in explaining some of the natural variability in the test lake data. However, the magnitudes of the amounts of the variability explained were small, ranging from 25 percent for total phosphorus to 2 percent for chlorophyll a.

All the analyses performed are consistent in their indication of a statistically significant shift in Secchi disc depth associated with improved water clarity after the ban. The estimated magnitude of this shift is reasonably consistent across the test lakes. The detectability of the shift, statistically, was aided by the apparent smaller relative variability of the Secchi disc measurements. This could be spurious due to inherent errors in Secchi disc measurements (see the Methods Section for a discussion of this subject). These possibilities need to be recognized in interpreting the meaningfulness of the apparent Secchi disc depth shift.

For example, the expected inverse relationship between Secchi disc depth and chlorophyll a appears to have been lacking in Townline Lake in 1978 (Figure F1). In this lake, one of two study lakes exhibiting an improvement in Secchi disc depth measurements between the pre- and ban periods, the expected relationship between the two variables is more observable in the ban period 1980-82 (Figures F2, F3 and F4), though many data points appear to be inconsistent. Similarly, in 1981, Elk Lake unexpectedly exhibited a direct linear relationship between Secchi disc depth measurements and chlorophyll a analyses (Figure F7). In 1980 no relationship appears to have existed between the two parameter (Figure F6). Only in 1978 and 1982 did Secchi disc depth measurements appear to vary inversely with chlorophyll a concentrations in Elk Lake (Figures F5 and F8).

In view of the lack of the inverse relationship between Secchi disc depth and chlorophyll a in these two lakes during certain years of the study, it can be concluded that factors other than algal concentrations, as measured by chlorophyll a, are affecting water clarity (i.e., suspended solids concentrations and sizes, natural coloration surface light intensity). Since these latter factors are unaffected by lake phosphorus concentrations and, therefore, detergent phosphorus, a more direct measure of the ban's effects is through analysis of total phosphorus and chlorophyll a concentrations in the lakes. Whether analyzed individually or as a group, the test lakes did not exhibit any statistically significant improvements in either lake total phosphorus concentration or chlorophyll a concentration.

The analysis using multivariate methods that took into account the potential covariances among the data generated inferences consistent with those obtained from the covariance analyses. This is gratifying in that it makes one more confident of insights that have been possible from examining the covariance mode. This is likely attributable to the fact that the magnitudes of the correlations in the data over time turned out not to be large relative to the inherent variability encountered for the sample sizes available. Also, all analyses were performed on the logarithms to the base 10 of the original data. Subsequent residual analysis supported the reasonableness of using this scale and assuming that the resulting distribution of the transformed measurements could be approximated by a Gaussian distribution.

The inherent variability in the data stemming from measurement and sampling variability combined was estimated for total phosphorus and Secchi disc depth to be sufficiently great such that a shift, or change, of approximately 10 percent would be needed to be able to detect a shift at the 5 percent level of significance. An even higher percent shift in chlorophyll a would be needed for detection with current variability estimates. If the magnitudes of the shifts, turn out to be smaller than this, then their potential detection will require more data or improved models or both. It can be inferred from this that changes in water quality that occur as a result of phosphorus load changes less than 10 percent are imperceptible to the scientist utilizing a reasonable monitoring program.

C. STATISTICAL ANALYSIS OF TEMPORAL VARIATIONS IN GREEN AND BLUE GREEN ALGAL COUNTS[1]

If a nutrient, such as phosphorus, is the factor controlling algal productivity in lakes, one would expect to observe a decrease in algal counts (biomass) if the phosphorus supply is reduced significantly. Figures G1A through E and G2A through E (Appendix G) clearly show there was a continual increase in both green and blue-green algal counts, respectively, throughout the study period and in all lakes. The monthly means of the counts are plotted.

A statistical analysis of these data confirms that no reductions in blue-green or green algal counts occurred in the test lakes relative to the reference lakes as a result of the ban. Analyses were conducted on $\mathrm{logarithm}_e$ of blue greens (ln BG), $\mathrm{logarithm}_e$ of greens (ln G), $p = BG/(BG+G)$, and $\ln [p/(1-p)]$. The transformations, where used, were for obtaining better agreement with the distributional assumptions for

[1]The statistical analysis was conducted by John W. Wilkinson, Ph. D., Rensselaer Polytechnic Institute, Troy, New York.

the various analyses. The use of the logistic transformation, ln P/(1-p), was of marginal value.

Tables G1 and G2 represent covariance analyses similar to those done for total phosphorus, chlorophyll a, and Secchi disc depth. For all measurements considered, the apparent tracing of the test lakes by the reference lakes is accounting for a substantial amount of the total variability (Table G1). Even so, except for the greens, the post-ban effect is statistically significant (although not dramatically so). This post-ban effect is primarily associated with Elk Lake, with Butternut also appearing for the blue greens (Table G2). In both instances, the direction is associated with increases in the blue greens that were greater than corresponding reference lakes. Although the causal factor(s) for these increases is not clear, the data do not support the hypothesis that a phosphate detergent ban would result in decreases in algal biomass.

Another possible indicator of general trends in lake water quality uses green and blue-green algae and the proportions of each in a lake's algal community. Generally, as water quality improves and the amounts of nutrients decrease, one might expect an overall shift in the algal community structure with green algae increasing in importance relative to blue-green algae. These changes must be monitored over a period of years, and should not be associated with the normal seasonal changes occurring over the course of a single year. From the statistical analysis (Tables G1 and G2) it is apparent that such a shift in the proportion of the two algal types did not occur as a result of the phosphate detergent ban, suggesting significant water quality improvements did not occur.

D. PHOSPHORUS RESIDENCE TIME AND LAKE RESPONSE

As an area of investigation, the exact nature of the internal recycling of phosphorus has received only limited study. Yet, a mathematical approach to the solution of the ultimate concentration of phosphorus in a lake in response to an input or change in input of that element offers some insight. As an alternative, one may calculate the length of time it would take to achieve the ultimate concentration given that change in input.

The development of such a mathematical model is described by Sonzogni et al. (1976). The basis for the model is equating a lake to a completely mixed reactor with constant chemical influx. Factors are then included to account for internal losses such as sedimentation (k) and stratification (α). Phosphorus residence time is used rather than hydraulic residence time since phosphorus is a nonconservative element.

The basic equation is therefore:

$$V\frac{dc}{dt} = Q_{c_i} - Q\ c - kVc$$

where:

V	=	volume of the lake (L^3)
Q	=	volumetric flow rate (L^3T^{-1})
c	=	concentration in the lake (ML^{-3})
c_i	=	constant influx concentration (ML^{-3})
k	=	internal loss rate constant (T^{-1})
	=	dimensionless proportionality factor that relates the mean annual outwash or surface water concentration to the mean annual concentration over the whole lake.

After substitutions and rearrangements, the equation can be expressed in terms of the ultimate steady-state concentration as:

$$\frac{(C - C_\infty)}{(C - C_\infty)} = C^{-t/R_p}$$

where: R_p = phosphorus residence time

$$= \frac{V}{Q\alpha + kV}$$

t = time

C_o = lake concentration before the step change in influx

C = concentration in lake

C_∞ = ultimate lake concentration after a step change in influx

According to Sonzogni et al. the time required to reach 95 percent of the expected change will be a time period equal to three phosphorus residence times. Estimates of the phosphorus residence times for the study lakes, calculated as the quotient of the mean annual content and annual input of phosphorus in 1978, are presented in Table 19. These data indicate that 95 percent of the expected change should have been observable in all the test lakes by the end of the ban.

Table 19: Number of Phosphorus and hydraulic residence times during ban and percent of expected change that should be observable.

	Yearly Mean [Total P] from 1978 volume proportional samples (mg/L)(a)	Phosphorus Residence Time (Yr)	Number of P Residence Times Between 7/1/79 and 7/1/82	Percent of Response	Hydraulic Residence Time (Yr)	Number of Hydraulic Residence Times Between 7/1/79 and 7/1/82
Balsam	0.061	0.17	18	(100%)	0.18	17
Butternut	0.071	0.49	6	(100%)	0.52	6
Elk	0.077	0.005	600	(100%)	0.004	750
Enterprise	0.024	0.65	5	(99%)	1.48	2
Moss	0.023	0.87	3	(95%)	2.49	2
Swan	0.112	0.40	8	(100%)	0.39	8
Townline	0.056	0.17	18	(100%	0.61	5

(a) Calculated as the lake phosphorus content (volume times mean yearly concentration) divided by the phosphorus loading (from Appendix D).

CONCLUSIONS

1. After a three-year time span without detergent phosphorus in wastewaters, the test lakes did not show any changes in chlorophyll a or total phosphate concentration attributable to the ban.

2. While statistical analysis indicated a statistically significant shift in Secchi disc depth associated with improved water clarity after the ban, this shift is inconsistent with the trends in chorophyll a and total phosphate concentrations. This indicates that other factors (e.g. suspended solids, color) were affecting the Secchi disc depth readings.

3. Nutrient enrichment bioassays demonstrated the importance of phosphorus and nitrogen in the nutrient dynamics of the test lakes, particularly as the summer season progressed.

4. The dominance of blue-green algae in the phytoplankton has been on the increase since 1978 in both the reference lakes and the test lakes. This occurrence is obviously unrelated to a detergent phosphate ban, indicating the importance of other factors (chemical, physical or biological) as regulating the ecosystem dynamics in these lakes.

5. Utilizing phosphorus loading estimation techniques, the total amount of phosphorus entering the test lakes from all sources was reduced by 0.1 to 11 percent as a result of the ban.

6. Non-point sources of phosphorus (those not derived from wastewaters) were the dominant loadings in the test lakes.

7. Treated wastewater discharges to the soil rather than to a lake or one of its tributaries prevents the wastewater impacts normally encountered in direct discharges.

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APPENDIX A

Maps of Study Lakes

Figure A1: BALSAM LAKE (Test)

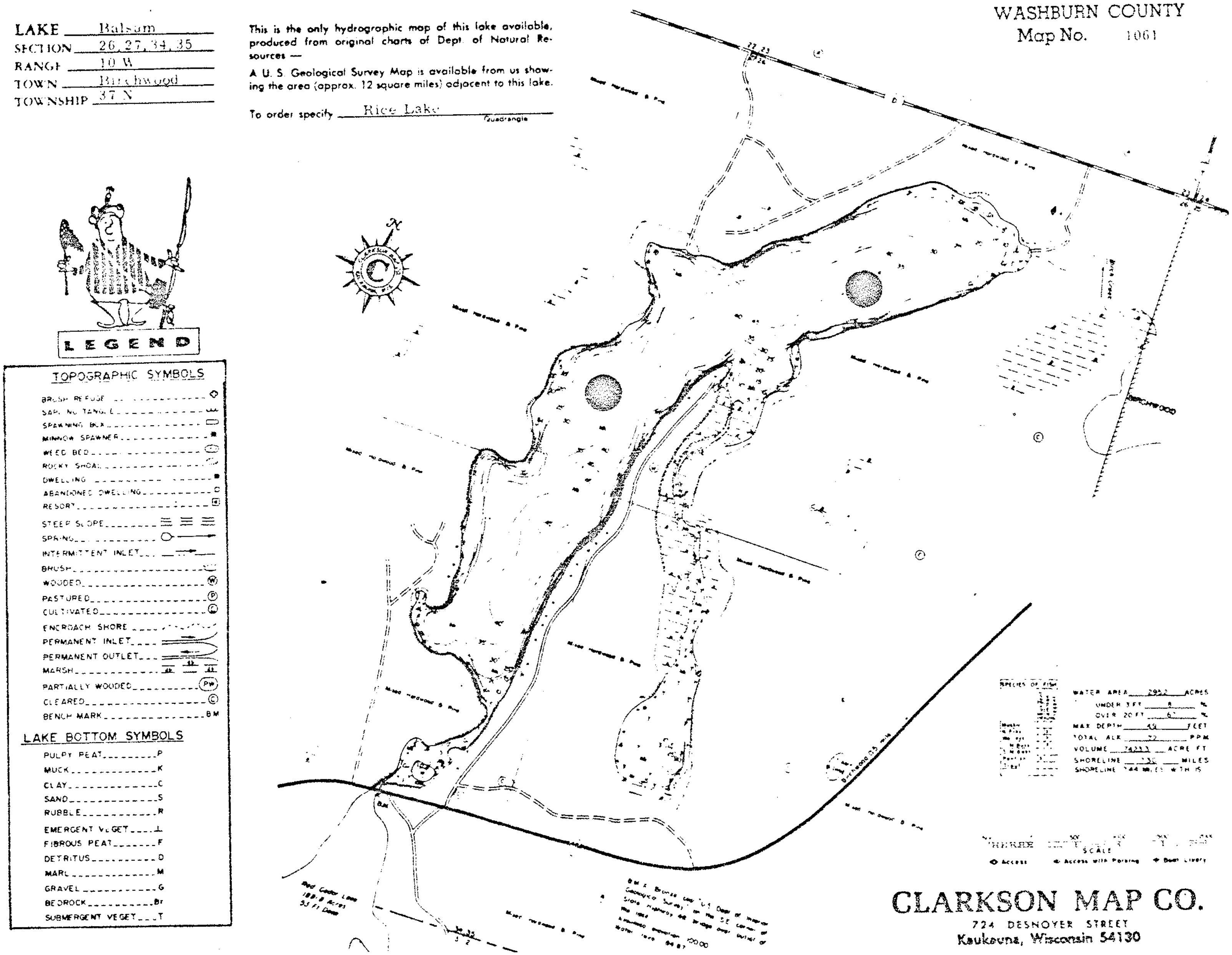


FIGURE A2: Butternut Lake (Test)

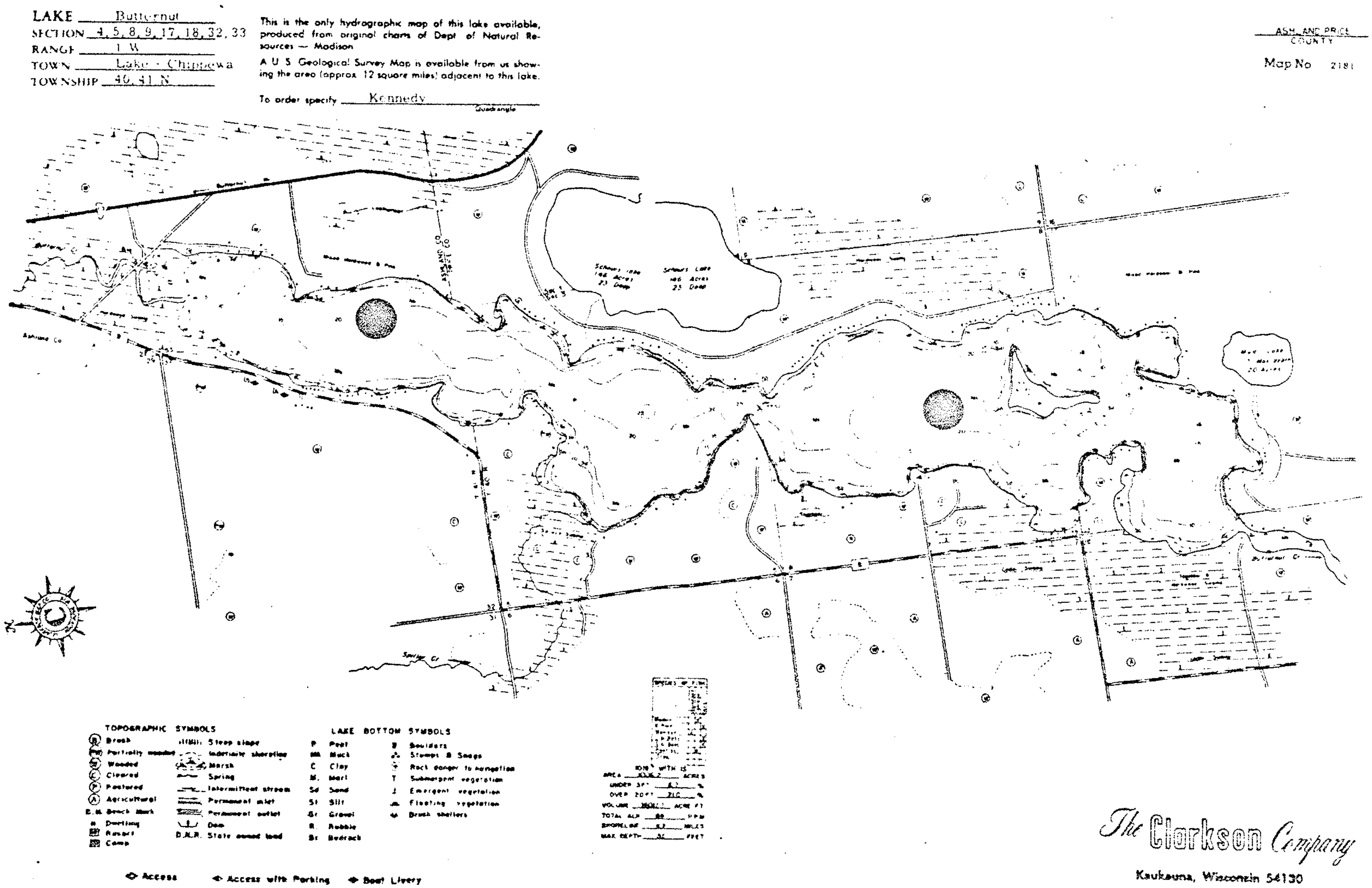


FIGURE A3: ELK LAKE (Test)

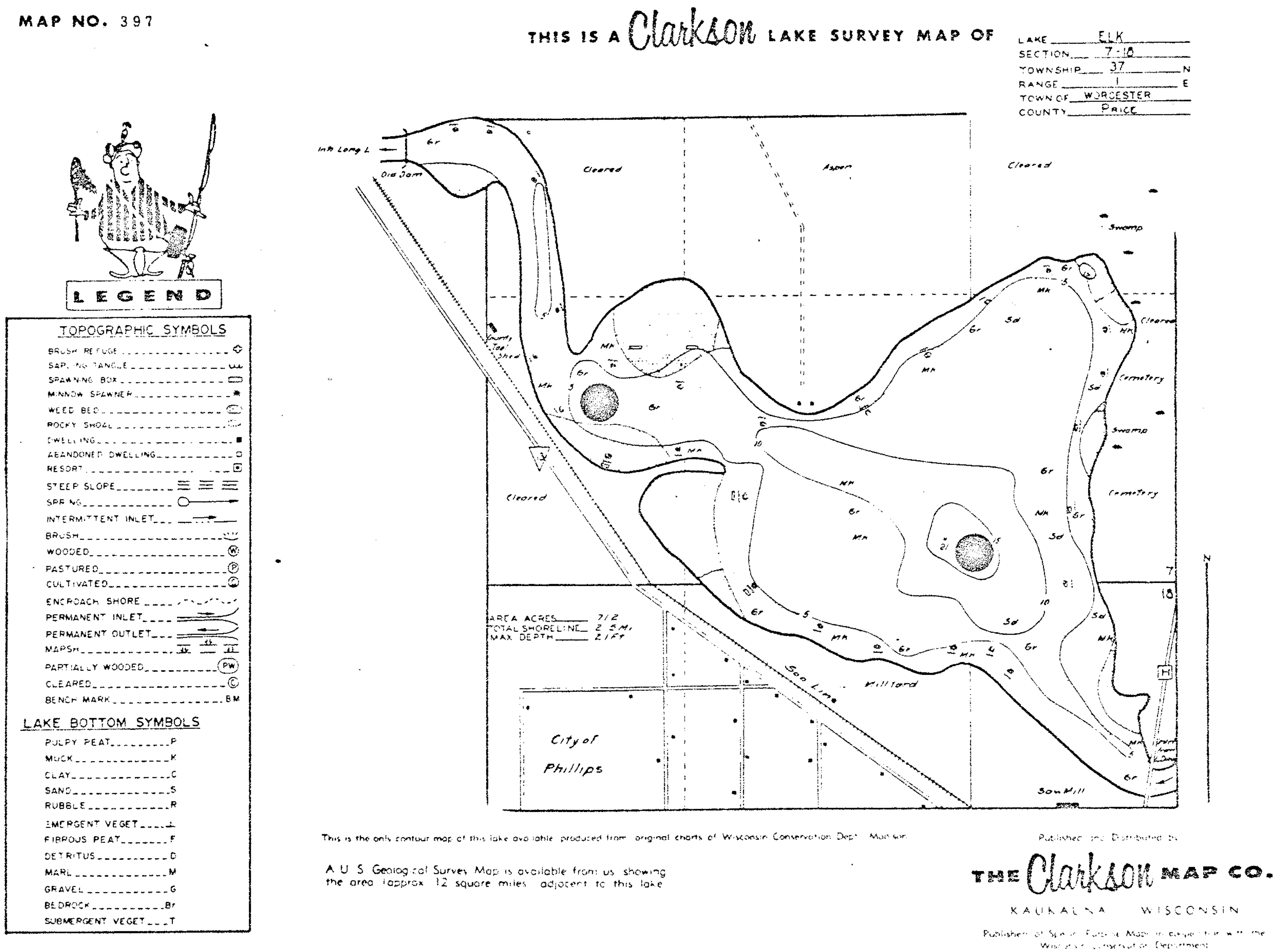


FIGURE A4: Enterprise Lake (Test)

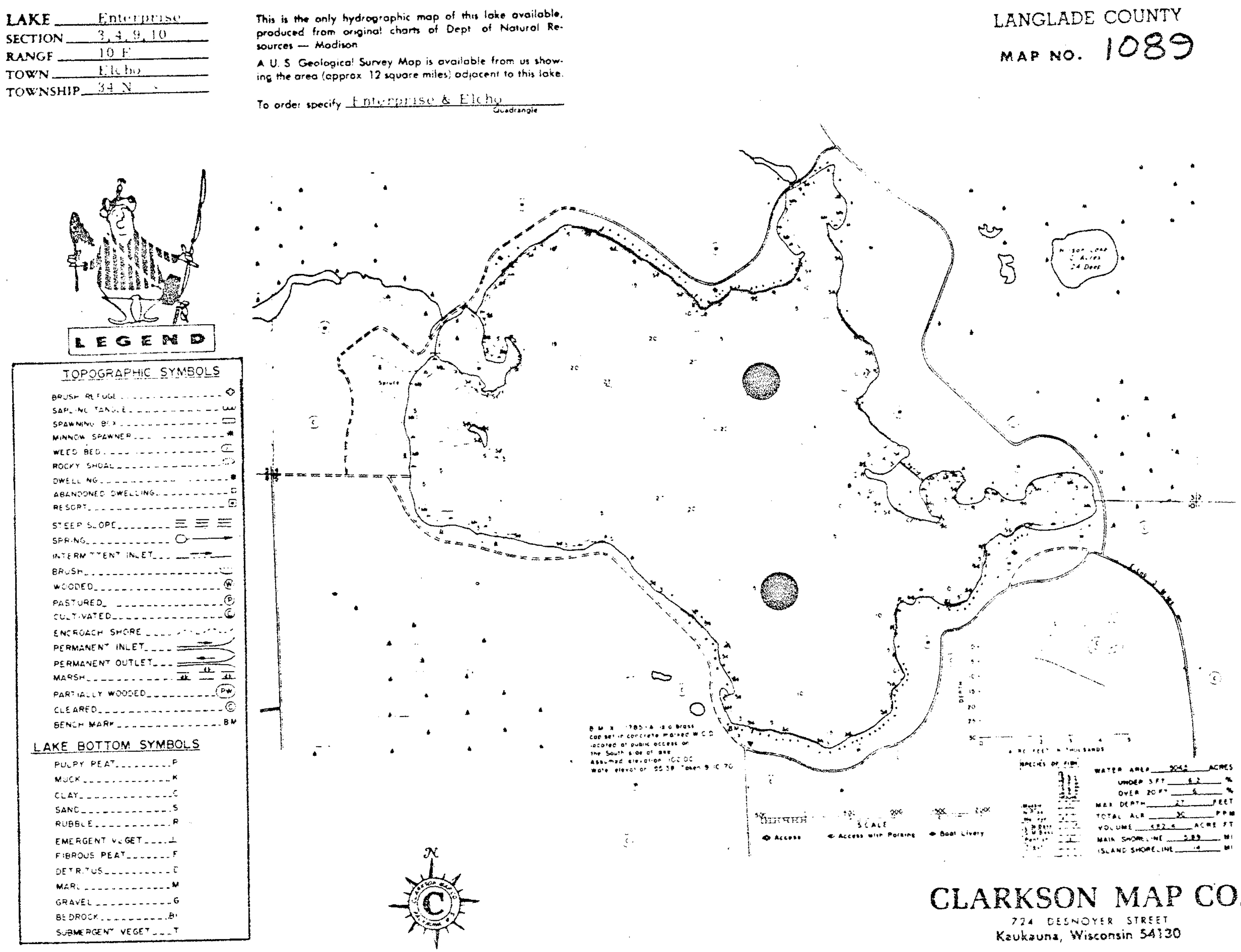


FIGURE A5: LITTLE BEARSKIN LAKE (Reference)

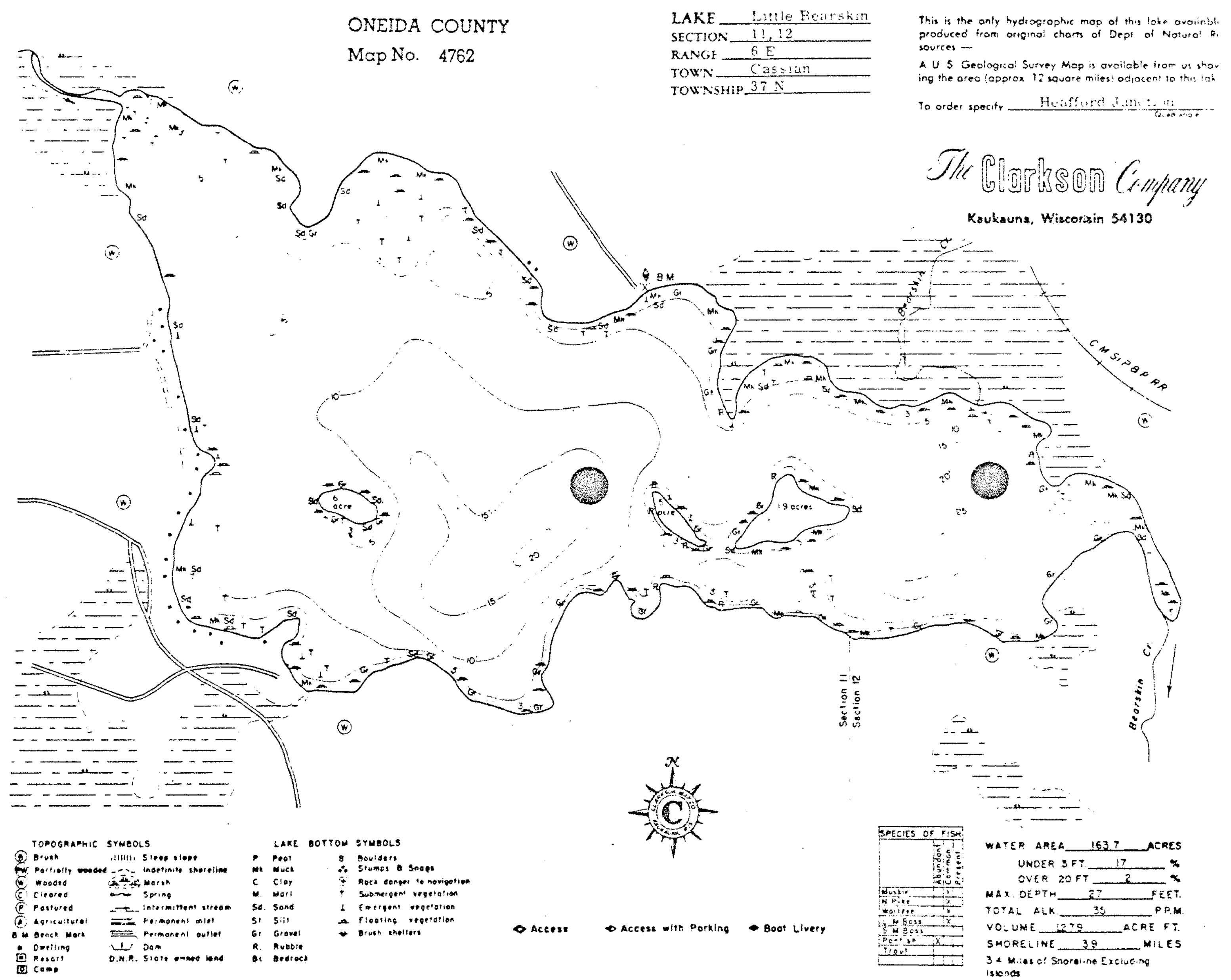


FIGURE A6: MOSS LAKE (Test)

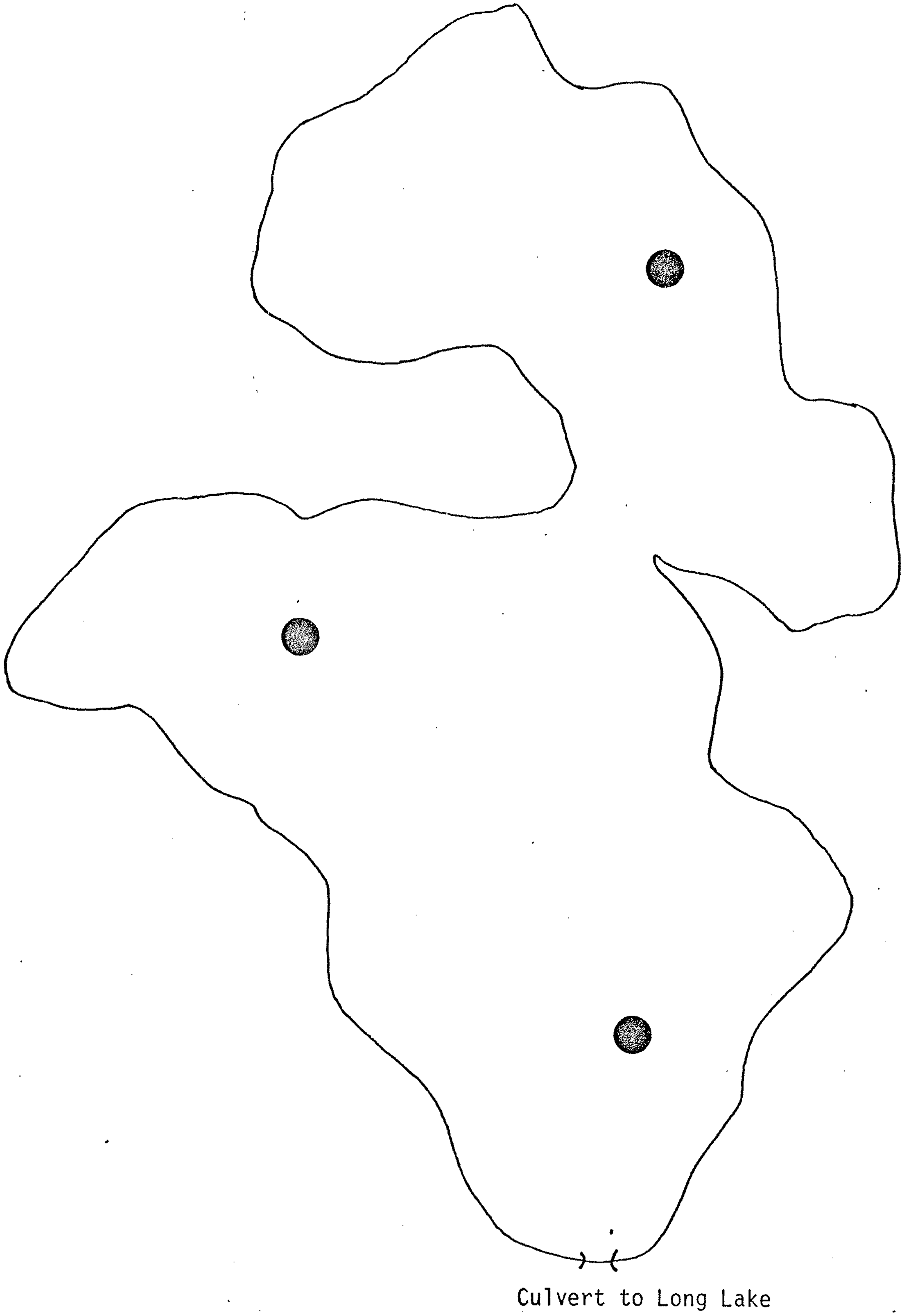
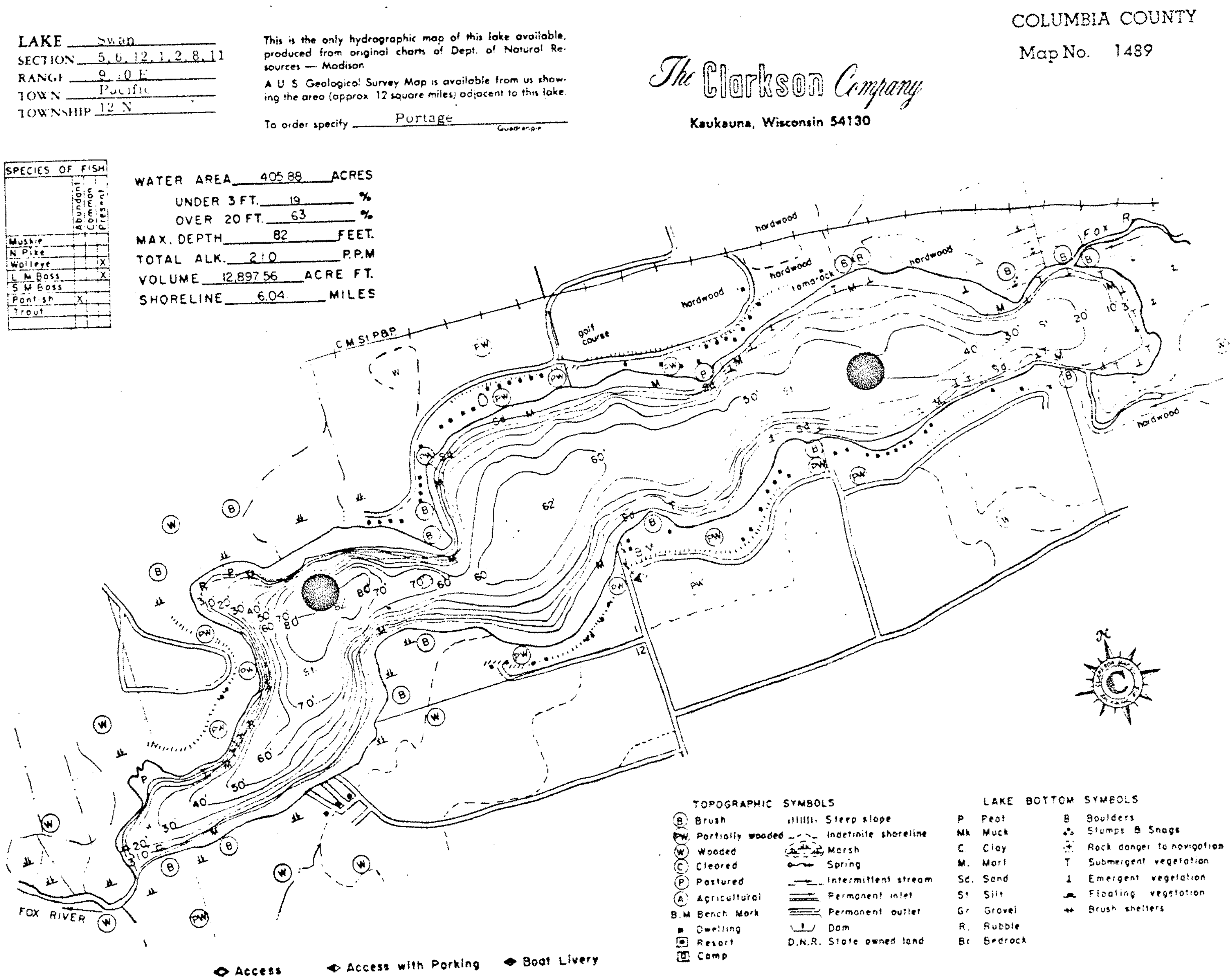


FIGURE A7: SWAN LAKE (Test)



-A7-

FIGURE A8: TEAL LAKE (Reference)

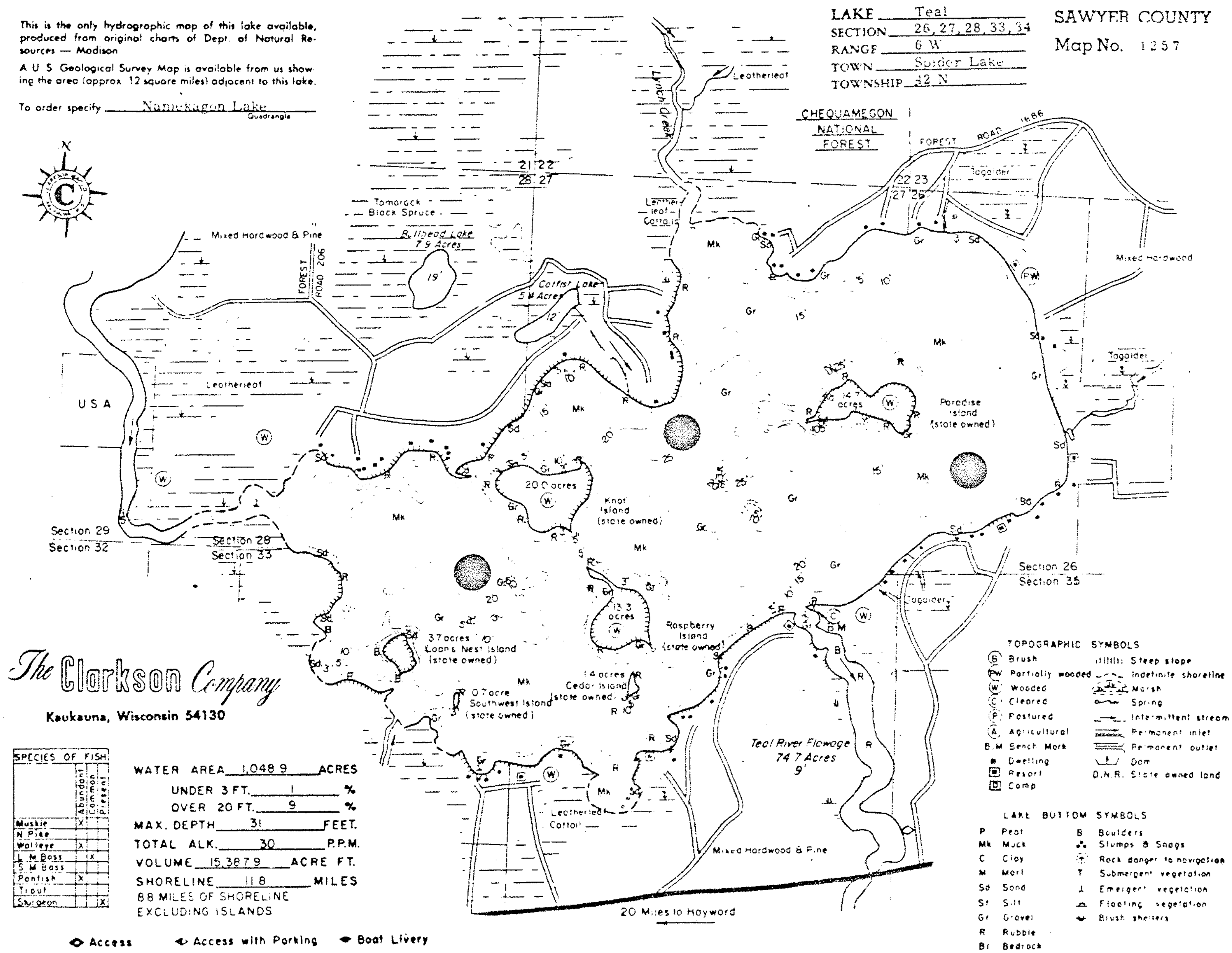
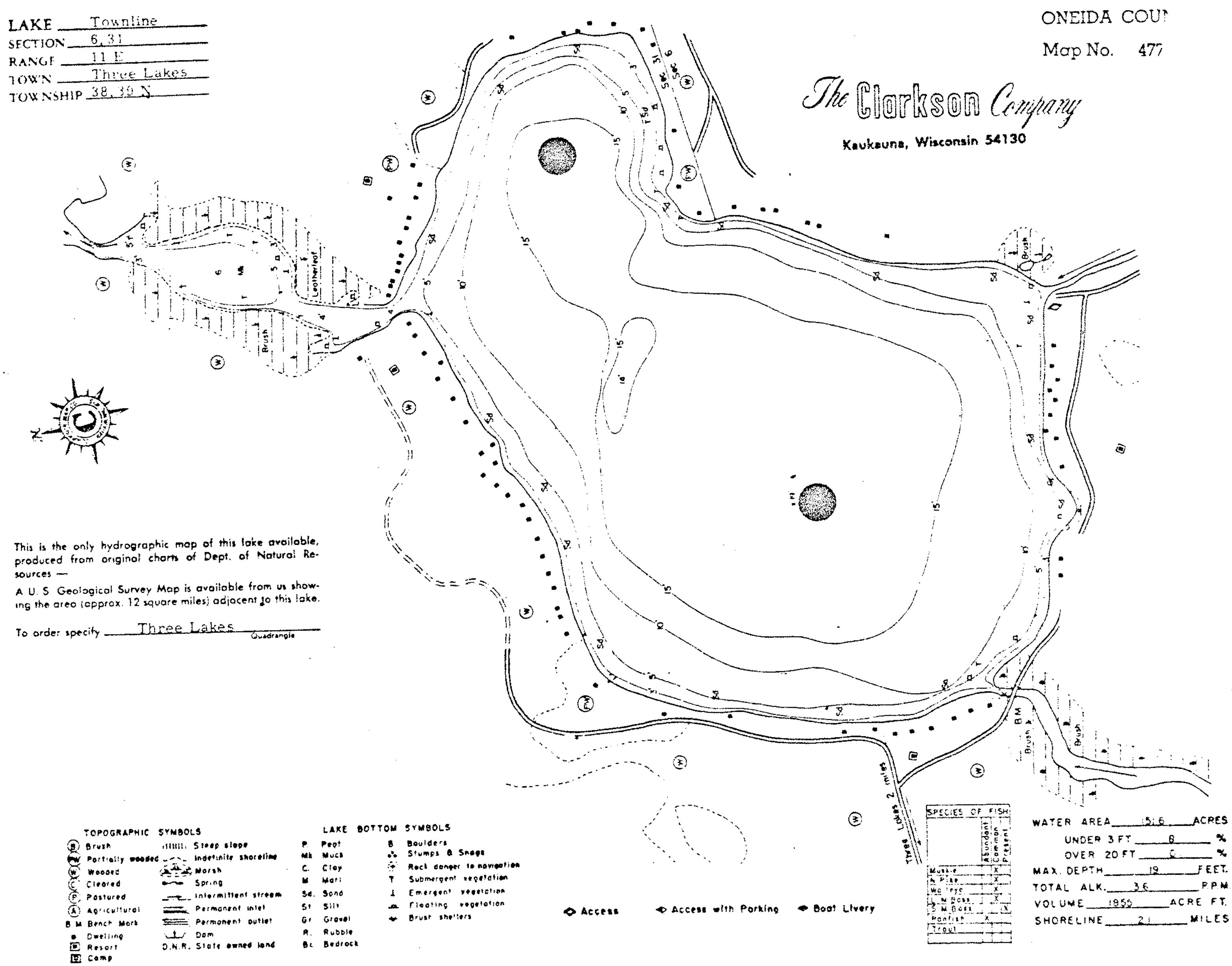


FIGURE A9: TOWNLINE LAKE (Test)



APPENDIX B

Description of the Multivariate/Multiple Comparison Analysis

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Description of the Multivariate/Multiple Comparison Analysis

The Statistical Model

The three response variables, total phosphorus, chlorophyll a and Secchi disc depth, were analyzed separately. Measurements of a variable for a lake in a year were analyzed as a vector response. To make the responses commensurate across years, measurements 1 through 7 for 1978, 1981 and 1982, and measurements 1 through 6 and 8 for 1980, were used. The responses are thus seven-dimensional. All responses were transformed by taking base 10 logarithms. Fish Lake was omitted, as 1982 data were not collected, so nine lakes were used.

The data on each response variable were analyzed as a multivariate two-way layout. Two analyses were performed for each response: one using only data from Site 1, the other using the average of the (logged) data from Sites 1 and 2.

The model can be written mathematically as

$$X_{ijk} = \mu_k + Y_{ik} + L_{jk} + e_{ijk}$$

(i = 1, 2, 3, 4, j = 1, 2, 3, 4, 5, 6, 7, 8, 9, k = 1, 2, 3, 4, 5, 6, 7), where X_{ijk} is the k^{th} observation on lake j in year i, μ_k is the k^{th} component of the grand mean, Y_{ik} is the k^{th} component of the effect of year i, and L_{jk} is the k^{th} component of the effect of lake j. The errors $\{{}^*e_{ij1}, e_{ij2}, e_{ij3}, e_{ij4}, e_{ij5}, e_{ij6}, e_{ij7})^T\}$ are assumed independent seven-variable Gaussian with zero mean and covariance matrix Σ. The advantages of this model are:

1. It takes account of the covariance structure of the data.
2. It is simple, allowing for differences between lakes and years without assuming a specific mathematical model for the differences.
3. If the simple analysis of variance model is unable to detect any effect due to the ban, more complicated models would not be worthwhile.

Parameter Estimation

The multivariate analysis of variance closely parallels its univariate counterpart. Maximum likelihood estimates of the effects are given by

$$\hat{\mu}_k = \overline{X}_{..k},$$

$$\hat{Y}_{ik} = \overline{X}_{i.k} - \overline{X}_{..k},$$

$$\hat{L}_{jk} = \overline{X}_{.jk} - \overline{X}_{..k},$$

where the dot and bar denote averaging over subscripts. The maximum likelihood estimate, $\hat{\Sigma}$, of the error covariance matrix, is proportional to the error sum of squares and cross products matrix E, where

$$E_{kl} = \sum_{i=1}^{4} \sum_{j=1}^{9} (X_{ijk} - \hat{\mu}_k - \hat{Y}_{ik} - \hat{L}_{jk})(X_{ijl} - \hat{\mu}_l - \hat{Y}_{il} - \hat{L}_{jl}).$$

Estimates of these give one an appreciation for the correlation structure as well as the variability reduction for the analysis based upon the combined Site 1 and Site 2 data.

Hypothesis Testing

The statistical tests of interest are multiple comparisons of contrasts $C_{ik} = Y_{ik} - Y_{1k}$, denoting the difference in measurement k between post-ban year i (1980, 1981, or 1982) and the pre ban year, 1978. The C_{ik} are estimated by $\hat{C}_{ik} = \hat{Y}_{ik} - \hat{Y}_{1k}$ ($i = 2, 3, 4$, $k = 1, 2, 3, 4, 5, 6, 7$). We shall follow the development of D.F. Morrison (1976).

By formula (8), pp. 200-201 of Morrison, the 100 $(1-\alpha)$ percent simultaneous confidence intervals on the $\{C_{ik}\}$ for all nine lakes are:

$$\hat{C}_{ik} - \left(\frac{2E_{kk}}{9}\frac{X_\alpha}{(1-X_\alpha)}\right)^{1/2} \leq C_{ik} \leq \hat{C}_{ik} + \left(\frac{2E_{kk}}{9}\frac{X_\alpha}{(1-X_\alpha)}\right)^{1/2}.$$

Here X_α is the upper 100_α percentage point of the greatest characteristic root distribution with parameters (in Morrison's notation) $s = 3$, $m = 3/2$ and $n = 8$. We take $\alpha = 0.05$, and find from Chart 11, p. 381 of Morrison that $X_\alpha = 0.625$. Similar expressions can be displayed for the other situations discussed in the Methods Section (Multivariate Analysis/Multiple Comparisons).

APPENDIX C

Description of the Approach Used to Develop Phosphorus Nutrient Loadings

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Description of the Approach Used to Develop Phosphorus Nutrient Budgets

Phosphorus loadings to a lake can be segmented into tributary input, direct drainage and atmospheric input. The tributary input would result from any runoff from areas which eventually terminate into a stream draining into a lake. Wastewater-derived input would account for both sewered areas which have discharge points into the lake directly, or a tributary to the lake, as well as septic tank/tile field system seepage or system failure. Direct drainage would arise from areas directly adjoining the lake shoreline, areas which do not drain to a lake tributary, but rather drain to the lake itself. For atmospheric loadings, only the wetfall and dryfall which enter the lake surface area directly are included in this category. Thus, the total of these inputs would comprise an estimate of the combined phosphorus loads to a lake from various sources.

The areas drained by streams and lakes, and land use areas, were delineated by surface morphology as provided in the most recently updated U.S.G.S. topographic maps.

Stream flows were determined utilizing data presented in the hydrologic budget sections of the U.S.G.S.-University of Wisconsin Geological and Natural History Survey hydrologic atlases (Olcott, 1968; Young and Hindall, 1972; and Oakes and Cotter, 1975), supplemented by normalized stream flows calculated by the U.S.G.S for the National Eutrophication Survey (NES) and National Oceanic and Atmospheric Administration (NOAA) monthly precipitation reports for Wisconsin.

The procedure for determining runoff quantities and flows involved calculating a runoff to precipitation ratio for each year. The value of this ratio for an average year's precipitation was obtained from the U.S.G.S.-University of Wisconsin Geological and Natural History Survey hydrologic atlases or from streamflows provided by the U.S.G.S. to the NES. To correct for the difference between the precipitation of a test year and that of an average year, an assumption was made that the proportion of runoff to precipitation could be modeled by the average basin data for dry, average, and wet years in the Upper Wisconsin River Basin (Enterprise and Townline Lakes) and the Chippewa River Basin (Balsam, Butternut, Elk, and Moss Lakes). For Swan Lake, located in the Fox-Wolf River Basin, the values for the Royalton-New London area were used since average basin values were not available.

To clarify the procedure, an example calculation of runoff (cm/yr) and runoff rate (m^3/yr) using data for Enterprise Lake is provided below:

1. The average year precipitation and runoff values for the lake were estimated from the U.S.G.S. atlas maps (Oakes and Cotter, 1975) which show lines of equal precipitation and runoff for the Upper Wisconsin River Basin in which Enterprise Lake is located:

    Lake average year precipitation = 72.6 cm/yr

    Lake average year runoff = 30.5 cm/yr

    Lake average year runoff to precipitation ratio = 0.42

2. The general relationship between runoff and precipitation in the Upper Wisconsin River Basin was characterized using mean basin data for precipitation and runoff during dry, average, and wet years provided in the U.S.G.S. atlas:

	Mean Basin Precipitation (cm/yr)	Mean Basin Runoff (cm/yr)	Runoff to Precipitation Ratio
Dry Year	60.2	17.0	0.28
Average Year	79.5	31.5	0.40
Wet Year	101.8	46.2	0.45

Through interpolation, a factor was obtained from this table, which, when added to the mean basin average-year runoff to precipitation ratio, provided an estimate of the runoff to precipitation ratio for the actual precipitation which occurred during a study year. The first and second terms of equation (a) represent the difference between the runoff to precipitation ratio for a dry and average year at Enterprise Lake. The lake average year precipitation in equation (a) is the average from the NOAA weather stations, not the U.S.G.S. atlases. These terms were then added to the third term (lake average-year runoff to precipitation ratio) to estimate the ratio for a study year at Enterprise Lake.

$$\text{(a)}\quad \begin{array}{c}\text{Dry}\\ \text{Year}\\ \text{Factor}\end{array} = \left[\frac{\begin{array}{c}\text{lake}\\ \text{avg. year}\\ \text{precip.}\end{array} - \begin{array}{c}\text{lake}\\ \text{study year}\\ \text{precip.}\end{array}}{\begin{array}{l}\text{mean basin}\\ \text{precip. for}\\ \text{avg. year}\end{array} - \begin{array}{l}\text{mean basin}\\ \text{precip. for}\\ \text{dry year}\end{array}} \right] \times$$

$$\left[\begin{array}{l}\text{mean basin}\\ \text{runoff to}\\ \text{precip. ratio}\\ \text{for dry year}\end{array} - \begin{array}{l}\text{mean basin}\\ \text{runoff to}\\ \text{precip. ratio}\\ \text{for avg. year}\end{array} \right] + \begin{array}{l}\text{lake runoff to}\\ \text{precip. ratio}\\ \text{for avg. year}\end{array}$$

The equation for the wet year factor was obtained by replacing "wet" for "dry" in equation (a). If NES values for the average-year runoff and precipitation were available, and the resulting runoff to precipitation ratio was less than that from Step 1, the NES ratio was used as the third term in equation (a) (i.e., the more conservative runoff estimate was calculated); NES values were not available for Enterprise Lake.

When the data for Enterprise Lake were used in equation (a) the following equations resulted:

$$\text{(b)}\quad \text{Dry Year Factor} = (-6.22\text{x}10^{-3}) \left[76.3 - \begin{array}{c}\text{lake}\\ \text{study year}\\ \text{precip.}\\ \text{(cm/yr)}\end{array} \right] + 0.42$$

(c) Average Year Factor = 0.42

(d) Wet Year Factor = $(2.24 \times 10^{-3}) \left[76.3 - \begin{matrix} \text{lake} \\ \text{study year} \\ \text{precip.} \\ \text{(cm/yr)} \end{matrix} \right] + 0.42$

Substitution of the precipitation (cm/yr) at the lake during a study year into equations (b) or (d) provided the factor for that year.

4. Multiplication of the factor derived in Step 3 by the precipitation (cm/yr) at the lake during the year for which the factor was calculated provided the runoff (cm/yr) for that year.

5. Runoff rates (m^3/yr) were calculated from the runoff (m/yr) value calculated in Step 4 and drainage areas (m^2).

Lake tributary phosphorus loadings were estimated using flows and the mean yearly phosphorus concentrations of the tributaries. For streams receiving municipal wastewater treatment plant effluents, the loadings attributed to the streams were determined by subtracting the treatment plant loading from the total measured stream load. For several tributaries, insufficient 1978 data, or no data at all, were available to calculate loadings. Table C1 presents the methodologies for estimating phosphorus loads from the unsampled tributaries.

Loadings to treatment plants were assumed to be 1.36 kg P/yr (3.0 lbs P/cap/yr) for untreated wastewater containing detergent phosphorus and 0.96 kg P/cap/yr (2.1 lbs P/cap/yr) for untreated wastewater without detergent phosphorus (SDA, unpublished data). Phosphorus removals due to treatment

Table C1: Inlet concentration estimates used in constructing the nutrient budgets for 1978. (a)

Lake	Stream	(mg P/L)	Source of Concentration
Balsam	Inlet from Birch Lake	0.064	Measured inlet concentration.
	Mud Lake Inlet	0.083	The Mud Lake Inlet was not sampled in 1978, therefore the ratio of the Mud Lake Inlet concentration to the Birch Lake Inlet concentration in 1981 was multiplied by the 1978 Birch Lake Inlet value to obtain this value.
Butternut	Spiller Creek	0.061	Spiller Creek was not sampled in 1978, therefore the ratio of the mean of the 1981 and 1982 Spiller creek concentrations to the mean of the 1981 and 1982 Butternut Creek concentrations was multiplied by the 1978 Butternut Creek concentration to obtain this value.
	Schnur Lake Inlet	0.051	The Schnur Lake Inlet was not sampled in 1978, therefore the ratio of the 1981 Schnur Lake Inlet concentration to the 1981 Butternut Creek concentration was multiplied by the 1978 Butternut Creek concentration to obtain this value.
	Mud Lake Inlet	0.062	The Mud Lake Inlet was not sampled in 1978, therefore the ratio of the 1981 Mud Lake Inlet concentration to the 1981 Butternut Creek concentration was multiplied by the 1978 Butternut Creek concentration to obtain this value.
	Butternut Creek	0.072	Measured 1978 Butternut Creek concentrations.
Elk	Inlet from Lake Duroy	0.055	Mean of 1981 and 1982 inlet data.
Enterprise	West Inlet	0.057	Mean of marsh-swamp values (Table C4).
Swan	Fox River Inlet	0.104	Measured 1978 Fox River Inlet concentrations.
Townline	Townline Creek	0.166	Measured 1978 Townline Creek concentrations.
	Maple Lake Inlet (b)	0.070	Mean of the 1981 and 1982 measured concentrations.

(a): These values represent the total loading of phosphorus in the tributaries and therefore include the loading from WWTP, when present.
(b): Samples were collected only in 1981 and 1982.

were considered to be 10, 20 and 90 percent for primary, secondary, and chemical phosphorus removal treatments, respectively, and 100 percent for the Birchwood (Balsam Lake) seepage cell and the Lac du Flambeau (Moss Lake) lagoon.

No direct field measurements were made of septic tank/tile field loadings, therefore several assumptions were made. The results of the 1982 shoreline survey (SDA) were used to estimate the number of septic tank/tile field systems in use. The survey results are presented in the lake description section. All lakeshore dwellings (homes and cabins), resorts, and camps were assumed to have septic tanks (undoubtedly an overestimation). Resorts and camps were considered to be the equivalent of 10 and 20 dwellings, respectively, and to have seasonal occupancy only. Other dwellings were estimated to be 50 percent seasonal and 50 percent year-round residences. A year-round occupancy of 2.5 persons was assumed for each dwelling or dwelling equivalent; seasonal occupancy was considered to be four months per year. The per capita phosphorus load to septic tanks was assumed to be the same as for municipal treatment plants. (The value of 1.36 kg P/cap/yr is probably high for seasonal residences which may not have laundry facilities or other modern appliances). A phosphorus removal of 90 percent before the discharge entered a lake was used for non-failing septic tanks, whereas failing septic systems were considered to have no phosphorus removal capability. A septic tank failure rate of 10 percent was used.

For atmospheric phosphorus loadings (both wetfall and dryfall), the value of 0.175 kg P/ha/yr reported in the NES studies was used since it appeared to be a conservative value. More recent studies (Reckhow et al., 1980) conclude that the NES value probably underestimated the actual loadings. The weather stations used to estimate precipitation at each lake are presented in Table C2; weather stations were chosen on the basis of precipitation data provided in the U.S.G.S. - University of Wisconsin Geological and Natural History Surveys (Olcott, 1968; Young and Hindall, 1972; and Oakes and Cotter, 1975), and the average (year) precipitation data in NOAA climatological reports.

Table C2: National Oceanographic and Atmospheric Administration weather stations used to estimate precipitation at the study lakes.

Lake	Weather Station	Average Yearly Precipitation (cm/yr)
Balsam	Couderay 7 W	not available
	Rice Lake	not available
	Spooner Experimental Farm	73.4 (a)
Butternut	Park Falls	83.7
Elk	Park Falls	83.7
	Prentice 1 N and Prentice 2	86.6
Enterprise	Rhinelander	76.3
Moss	Minocqua Dam	78.6
	Rest Lake	85.8
Swan	Portage	78.0
Townline	Rhinelander	76.3

(a): When referred to in the text, a value of 74.9 cm/yr was used for the average precipitation at Balsam Lake, based on the U.S.G.S. hydrologic budget (Young and Hindall, 1972).

Direct drainage (the area surrounding a lake which contributes runoff directly to the lake rather than a tributary) loadings were calculated using export coefficients compiled by Reckhow et al. (1980) and the respective coefficients applied to specific land use areas. The choice of coefficients was based on precipitation and runoff quantities, the areal extent of individual land use types, soil and geological data from the U.S.G.S.-University of Wisconsin Geological and Natural History Survey atlases (Olcott, 1968; Young and Hindall, 1972; and Oakes and Cotter, 1975), soil survey reports (Wisconsin Geological and Natural History Survey, 1916,a,b, 1927, 1947 and 1959; and U.S.D.A., 1978), types of vegetation, and field observations. The export coefficients were generally chosen from the lower end of the concentration range presented in the literature whenever data from more than one study were available, thereby yielding conservatively low estimates for direct runoff (Table C3).

Several deviations from the general procedure were required to account for special circumstances. For Swan Lake, an export concentration 0.057 mg P/L was used for the direct runoff. This was necessitated due to difficulties encountered in interpreting the effects of agricultural drainage into the marshes on the northern shore of the lake, the lack of information on agricultural fertilization, and the relatively great distance (five km) of parts of the direct drainage basin from the lake. The value represents the mean of concentrations from tributaries draining marsh and swamp areas (surrounded by forested lands) investigated during this study (Table C4).

At Enterprise Lake, the major potential sources of phosphorus were marsh and swamp drainage from the West Inlet, direct drainage loadings, and septic tanks. Since this was a test lake where runoff from the direct drainage area could be a major source of phosphorus, great care was taken in order not to overestimate its effect. The location of marshes and swamps, and the topography, suggested

Table C3: Export coefficients (kg P/ha/yr) used to calculate the 1978 direct drainage phosphorus loadings to the test lakes. All footnoted values are from data presented in Reckhow et al. (1980).

Lake	Agriculture	Marsh & Swamp	Forest	Urban and Shoreline Residential
Balsam	0.250(c)	0.157(d)	0.100(b)	Shore: 0.19(f)
Butternut	0.250(c)	See text, based on runoff.	0.047(a)	Shore: 0.19(f)
Elk	0.250(c)	0.157(d)	0.047(a)	Shore: 0.19(f) Urban: 1.1(e)
Enterprise	0.0	See text, based on runoff.	0.047(a) See text.	Shore: 0.19(f)
Moss	0.0	0.157(d)	0.047(a)	Shore: 0.19(f) Urban: 1.1(e)
Swan	A value of 0.057 mg P/L of runoff was used for the entire area (see text).			
Townline	0.250(c)	0.157(d)	0.047(a)	Shore: 0.19(f)

(a) Dillon, P. J. and W. B. Kirchner, 1975. For sandy soils over granitic and igneous bedrock.

(b) Dillon, P. J. and W. B. Kirchner, 1975; Schinder, D. W. and J. E. Nighswander, 1970; and Singer, M. J. and R. H. Rust, 1975. For silt, sandy, and clay loam soils; loam soils over sedimentary bedrock.

(c) Harms, L. L., J. N. Dornbush, and J. R. Anderson, 1974; and Schuman, G. E., R. G. Spomer, and R. F. Piest, 1973. For grazed and pastured agricultural lands.

(d) Verry, E. S., 1982. For forest composed of 70 % aspen, 30 % black spruce and alder.

(e) Kluesner, J. W. and G. F. Lee, 1974; and Landon, R. J., 1977. For urban watersheds.

(f) Landon, R.J., 1977. For low density residential subdivision with large lots covered with grass and trees.

Table C4: Mean yearly total phosphorus concentrations for inlets with considerable marsh and swamp drainage, and which do not receive discharges from a wastewater treatment plant.

<u>Lynch Creek (Teal Lake)</u>

1981: 0.068 mg/L (a)
1982: 0.037 mg/L (a)

<u>West Inlet (Enterprise Lake)</u>

1981: 0.132 mg/L (b)
1982: 0.061 mg/L (a)

<u>Mud Lake Inlet (Butternut Lake)</u>

1981: 0.062 mg/L (a)
1982: not sampled.

(a): The mean (0.057 mg P/L) of these values was used to estimate some tributary loadings in the 1978 phosphorus budgets (see Table C1).

(b): Only five samples were collected.

that some of the marsh-swamp areas (e.g., Hilson Lake) act as phosphorus sinks for a considerable portion of the forested region around the lake. Therefore, marsh-swamp areas with direct contact with the lake shore (either bordering it or having an intermittent tributary) were considered to contribute 0.057 mg P/L of runoff (as discussed for Swan Lake), and were assumed to receive drainage from the forested lands (1.9 km^2) surrounding them, whereas all other marshes and swamps were assumed to be phosphorus sinks which drained 50 percent of the remaining forested lands (2.9 km^2); the other 50 percent of these forests (2.9 km^2) were treated as contributing direct drainage runoff to the lake.

Modifications of the septic tank procedures were made for Butternut Lake, Moss Lake, Elk Lake and Enterprise Lake. At Butternut Lake, the large number of resorts (eight) suggested that some were year-round, therefore 50 percent were treated as year-round occupancy. For Elk Lake all seven dwellings were assumed to be year-round residences due to their proximity to the Town of Phillips. A failure rate of 3 percent was used for calculating septic tank loadings to Moss Lake. Since Enterprise Lake was chosen to estimate the impact of septic tanks, septic tank loadings were estimated for what were considered the most probably conditions. Langlade County, in which Enterprise Lake is located, is estimated by county sanitary code administrators to have a 3 percent septic tank failure rate, although sanitary surveys of private sewage disposal systems serving waterfront properties around 13 Wisconsin Lakes showed failure rates averaging 22 to 26 percent (W.N.R.B., 1977). These surveys also indicated that 70 percent of the properties were occupied seasonally. Based on these considerations, 50 percent of the dwellings and the camp were considered to be seasonal, and a septic tank failure rate of 3 percent was used.

APPENDIX D

1. Precipitation, Runoff Rates, and Land Use Areas for Test Lakes.

2. Phosphorus Nutrient Loads for the Test Lakes

Table D1: Balsam Lake precipitation and runoff (cm/yr).

	Dry Year	Avg. Year	Wet Year	1978	1980	1981	1982(a)
Precipitation (b)	54.1	74.9	114.5	80.0	80.2	77.2	78.4
Runoff (c)	16.3	26.7	38.8	28.3	28.4	27.4	27.8

(a): 1982 data were used for January through August, and average year values (NOAA) for the remaining months.
(b): The average year value is for the lake location (NOAA climatological reports). The dry and wet year values are Chippewa River Basin averages (Young and Hindall, 1972).
(c): These are calculated estimates for the lake location (see Appendix C).

Table D2: Balsam Lake runoff rates (cu m/s) (a).

Source	1978	1980	1981	1982
Direct Drainage	0.086	0.086	0.083	0.085
Birch Lake Inlet	1.481	1.486	1.434	1.454
Mud Lake Inlet	0.016	0.016	0.016	0.016
Total	1.583	1.588	1.533	1.555

(a): Lake surface areas were considered to have zero net yearly runoff.

Table D3: Balsam Lake land use areas.

Category	Area (sq km)	% of Category Area
Direct Drainage		
a. Marsh and Swamp	3.4	35.4
b. Forest	4.3	44.8
c. Agriculture	1.8	18.8
d. Shoreline Residental	0.1	1.0
Subtotal:	9.6	100.0
Mud Lake Inlet		
a. Mud Lake	0.2	10.0
b. Marsh and Swamp	0.1	5.0
c. Forest	1.2	60.0
d. Agriculture	0.5	25.0
Subtotal:	2.0	100.0
Birch Lake Inlet	165	100.0
Balsam Lake	1.1	100.0
Total	178	

Table D4: Butternut Lake precipitation and runoff (cm/yr).

	Dry Year	Avg. Year	Wet Year	1978	1980	1981	1982
Precipitation (a)	54.1	83.7	114.5	83.9	90.0	75.7	(c)
Runoff (b)	15.7	30.8	40.6	30.9	32.9	26.3	(c)

(a): The average year value is for the lake location (NOAA climatological reports). The dry and wet year values are Chippewa River Basin averages (Young and Hindall, 1972).
(b): These are calculated estimates for the lake location (see Appendix C).
(c): 1982 precipitation data were not available from the Park Falls station.

Table D5: Butternut Lake runoff rates (cu m/s) (a).

Source	1978	1980	1981	1982
Direct Drainage	0.083	0.089	0.071	(b)
Spiller Creek Inlet	0.204	0.217	0.173	(b)
Butternut Creek Inlet	0.710	0.756	0.605	(b)
Schnur Lake Inlet	0.044	0.047	0.037	(b)
Mud Lake Inlet	0.036	0.039	0.031	(b)
Total	1.077	1.148	0.917	(b)

(a): Lake surface areas were considered to have zero net yearly runoff.
(b): 1982 precipitation data were not available for the Park Falls station.

Table D6: Butternut Lake land use areas.

Category	Area (sq km)	% of Category Area
Direct Drainage		
a. Marsh and Swamp	1.4	16.5
b. Forest	3.4	40.0
c. Agriculture	2.9	34.1
d. Shoreline Residental	0.8	9.4
Subtotal:	8.5	100.0
Mud Lake Inlet		
a. Mud Lake	0.1	2.7
b. Marsh and Swamp	0.9	24.3
c. Forest	1.9	51.4
d. Agriculture	0.8	21.6
Subtotal:	3.7	100.0
Schnur Lake Inlet		
a. Schnur Lake	0.6	13.3
b. Marsh and Swamp	0.6	13.3
c. Forest	2.5	55.7
d. Agriculture	0.6	13.3
e. Shoreline Residential	0.2	4.4
Subtotal:	4.5	100.0
Spiller Creek Inlet		
a. Marsh and Swamp	7.3	35.1
b. Forest	5.9	28.4
c. Agriculture	7.6	36.5
Subtotal:	20.8	100.0
Butternut Creek Inlet	72.5	100.0
Butternut Lake	4.0	100.0
Total	114	

Table D7: Elk Lake precipitation and runoff (cm/yr).

	Dry Year	Avg. Year	Wet Year	1978	1980	1981	1982(a)
Precipitation (b)	54.1	85.2	114.5	84.9	87.7	77.5	83.1
Runoff (c)	15.3	31.0	40.3	30.8	31.8	26.6	29.8

(a): 1982 data were used for January through August, and average year values (NOAA) for the remaining months.
(b): The average year value is for the lake location (NOAA climatological reports). The dry and wet year values are Chippewa River Basin averages (Young and Hindall, 1972).
(c): These are calculated estimates for the lake location (see Appendix C).

Table D8: Elk Lake runoff rates (cu m/s) (a).

Source	1978	1980	1981	1982
Lake Duroy Inlet	4.268	4.406	3.686	4.129
Direct Drainage	0.037	0.038	0.032	0.036
Total	4.305	4.444	3.718	4.165

(a): Lake surface areas were considered to have zero net yearly runoff.

Table D9: Elk Lake land use areas.

Category	Area (sq km)	% of Category Area
Direct Drainage		
a. Marsh and Swamp	0.3	7.9
b. Forest	0.7	18.4
c. Agriculture	1.1	28.9
d. Urban and Airport	1.7	44.8
Subtotal:	3.8	100.0
Lake Duroy Inlet	437	100.0
Elk Lake	0.4	100.0
Total	441	

Table D10: Enterprise Lake precipitation and runoff (cm/yr).

	Dry Year	Avg. Year	Wet Year	1978	1980	1981	1982(a)
Precipitation (b)	60.2	76.3	101.8	101.4	81.4	72.6	75.0
Runoff (c)	19.3	32.0	48.6	48.3	35.1	28.8	30.9

(a): 1982 data were used for January through August, and average year values (NOAA) for the remaining months.
(b): The average year value is for the lake location (NOAA climatological reports). The dry and wet year values are Upper Wisconsin River Basin averages (Oakes and Cotter, 1968).
(c): These are calculated estimates for the lake location (see Appendix C).

Table D11: Enterprise Lake runoff rates (cu m/s) (a).

Source	1978	1980	1981	1982
West Inlet	0.044	0.032	0.026	0.028
Direct Drainage	0.167	0.121	0.099	0.107
Total	0.211	0.153	0.125	0.135

(a): Lake surface areas were considered to have zero net yearly runoff.

Table D12: Enterprise Lake land use areas.

Category	Area (sq km)	% of Category Area
Direct Drainage		
a. Marsh and Swamp	2.6	23.9
b. Forest	7.7	70.7
c. Agriculture	0.3	2.7
d. Shoreline Residential	0.3	2.7
Subtotal:	10.9	100.0
West Inlet		
a. Marsh and Swamp	1.9	65.5
b. Forest	1.0	34.5
Subtotal:	2.9	100.0
Enterprise Lake	1.9	100.0
Total	15.7	

Table D13: Moss Lake precipitation and runoff (cm/yr).

	Dry Year	Avg. Year	Wet Year	1978	1980	1981	1982(a)
Precipitation (b)	54.1	82.2	114.5	93.6	77.3	69.0	81.9
Runoff (c)	22.6	33.4	44.9	37.6	30.4	25.6	33.2

(a): 1982 data were used for January through August, and average year values (NOAA) for the remaining months.
(b): The average year value is for the lake location (NOAA climatological reports). The dry and wet year values are Chippewa River Basin averages (Young and Hindall, 1972).
(c): These are calculated estimates for the lake location (see Appendix C).

Table D14: Moss Lake runoff rates (cu m/s) (a).

	1978	1980	1981	1982
Direct Drainage	0.036	0.029	0.024	0.031

(a): Lake surface areas were considered to have zero net yearly runoff.

Table D15: Moss Lake land use areas.

Category	Area (sq km)	% of Category Area
Moss Lake	0.8	100.0
Direct Drainage		
a. Marsh and Swamp	0.5	16.8
b. Forest	2.1	70.0
c. Agriculture	0.1	3.3
d. Shoreline Residential	0.1	3.3
e. Urban - Residential	0.1	3.3
f. Wastewater Lagoon	0.1	3.3
Subtotal:	3.0	100.0
Total	3.8	

Table D16: Swan Lake precipitation and runoff (cm/yr).

	Dry Year	Avg. Year	Wet Year	1978	1980	1981	1982(a)
Precipitation (b)	48.3	78.0	113.4	96.1	103.5	83.9	81.6
Runoff (c)	11.4	18.8	35.4	26.7	30.3	21.2	20.3

(a): 1982 data were used for January through August, and average year values (NOAA) for the remaining months.
(b): The average year value is for the lake location (NOAA climatological reports). The dry and wet year values are Fox-Wolf River Basin averages (Olcott, 1968).
(c): These are calculated estimates for the lake location (see Appendix C).

Table D17: Swan Lake runoff rates (cu m/s) (a).

	1978	1980	1981	1982
Fox River Inlet	1.245	1.412	0.988	0.946
Direct Drainage	0.180	0.205	0.143	0.137
Total:	1.425	1.617	1.131	1.083

(a): Lake surface areas were considered to have zero net yearly runoff.

Table D18: Swan Lake Land Use Areas.

Category	Area (sq km)	% of Category Area
Swan Lake	1.6	100.0
Direct Drainage		
a. Marsh and Swamp	5.1	24.0
b. Forest	2.7	12.7
c. Agriculture	13.0	61.0
d. Shoreline Residential	0.5	2.3
Subtotal:	21.3	100.0
Fox River Inlet	147	100.0
Total	170	

Table D19: Townline Lake precipitation and runoff (cm/yr).

	Wet Year	Avg. Year	Dry Year	1978	1980	1981	1982(a)
Precipitation (b)	60.2	76.3	101.8	101.4	81.4	72.6	75.0
Runoff (c)	20.1	26.2	40.8	40.6	28.9	24.8	25.7

(a): 1982 data were used for January through August, and average year values (NOAA) for the remaining months.
(b): The average year value is for the lake location (NOAA climatological reports). The dry and wet year values are Upper Wisconsin River Basin averages (Oakes and Cotter, 1968).
(c): These are calculated estimates for the lake location (see Appendix C).

Table D20: Townline Lake runoff rates (cu m/s) (a).

Source	1978	1980	1981	1982
Townline Creek Inlet	0.109	0.078	0.067	0.069
Direct Drainage	0.022	0.016	0.013	0.014
Maple Lake Inlet	0.021	0.015	0.013	0.013
Total	0.152	0.109	0.093	0.096

(a): Lake surface areas were considered to have no net yearly runoff.

Table D21: Townline Lake land use areas.

Category	Area (sq km)	% of Category Area
Townline Lake	0.6	
Direct Drainage		
a. Marsh and Swamp	0.2	
b. Forest	1.1	
c. Agriculture	0.3	
d. Shoreline Residential	0.1	
Subtotal	1.7	
Townline Creek	8.5	
Maple Lake Inlet	1.6	
Total	12.4	

Table D22: Balsam Lake Phosphorus Loads.

Source	1978 With Det. P kg/yr	1978 With Det. P % of Total	1978 Without Det. P kg/yr	1978 Without Det. P % of Total	Change in Load kg/yr	Change in Load % of Total With Det. P
Direct Drainage						
a. Marsh and Swamp	55	2	55	2	0	0
b. Forest	45	1	45	1	0	0
c. Agriculture	45	1	45	1	0	0
d. Shoreline Residential	5	<1	5	<1	0	0
Subtotal:	150	5	150	5	0	0
Atmospheric	20	1	20	1	0	0
Septic Tanks	10(a)	<1	8(a)	<1	2	<1
WWTP (100% removal)	0	0	0	0	0	0
Tributaries						
a. Birch Lake Inlet	3000	93	3000	93	0	0
b. Mud Lake Inlet	40	1	40	1	0	0
Subtotal:	3040	94	3040	94	0	0
Total	3220	100	3218	100	2	<1

(a): This value was not rounded off in order to show the magnitude of the septic tank/tile field phosphorus load.

Table D23: Butternut Lake Phosphorus Loads.

Source	1978 With Det. P kg/yr	1978 With Det. P % of Total	1978 Without Det. P kg/yr	1978 Without Det. P % of Total	Change in Load kg/yr	Change in Load % of Total With Det. P
Direct Drainage						
a. Marsh and Swamp	35	1	35	2	0	0
b. Forest	15	1	15	1	0	0
c. Agriculture	75	3	75	3	0	0
d. Shoreline Residential	15	1	15	1	0	0
Subtotal:	140	6	140	7	0	0
Atmospheric	70	3	70	3	0	0
Septic Tanks	112(a)	4	80(a)	3	32	1
WWTP (20% removal)	480	19	340	15	140	6
Tributaries:						
a. Spiller Creek	390	16	390	17	0	0
b. Schnur Lake Inlet	70	3	70	3	0	0
c. Mud Lake Inlet	70	3	70	3	0	0
d. Butternut Creek	1130	46	1130	49	0	0
Subtotal:	1660	68	1660	72	0	0
Total	2462	100	2290	100	170	7

(a): This value was not rounded off in order to show the magnitude of the septic tank/tile field phosphorus load.

Table D24: Elk Lake Phosphorus Loads.

Source	1978 With Det.P kg/yr	1978 With Det.P % of Total	1978 Without Det.P kg/yr	1978 Without Det.P % of Total	Change in Load kg/yr	Change in Load % of Total With Det. P
Direct Drainage						
a. Marsh and Swamp	5	<1	5	<1	0	0
b. Forest	5	<1	5	<1	0	0
c. Agriculture	30	<1	30	<1	0	0
d. Urban & Airport	190	2	190	3	0	0
Subtotal:	230	3	230	3	0	0
Atmospheric	5	<1	5	<1	0	0
Septic Tanks	4(a)	<1	3(a)	<1	1	<1
WWTP: (20% removal)	1660	22	1170	16	490	6
Lake Duroy Inlet	5740	75	5740	80	0	0
Total	7639	100	7148	100	491	6

(a): This value was not rounded off in order to show the magnitude of the septic tank/tile field phosphorus load.

Table D25: Enterprise Lake Phosphorus Loads.

Source	1978 With Det.P kg/yr	1978 With Det.P % of Total	1978 Without Det. P kg/yr	1978 Without Det. P % of Total	Change in Load kg/yr	Change in Load % of Total With Det. P
Direct Drainage						
a. Shoreline Marshes	100	37	100	38	0	0
b. Forest	15	6	15	6	0	0
c. Agriculture	0	0	0	0	0	0
d. Shoreline Residential	5	2	5	2	0	0
Subtotal:	120	44	120	46	0	0
Atmospheric	35	13	35	13	0	0
Septic Tanks	35(a)	13	25(a)	10	10	4
West Inlet	80	30	80	31	0	0
Total	270	100	260	100	10	4

(a): This value was not rounded off in order to show the magnitude of the septic tank/tile field phosphorus load.

Table D26: Moss Lake Phosphorus Loads.

Source	1978 With Det. P kg/yr	1978 With Det. P % of Total	1978 Without Det. P kg/yr	1978 Without Det. P % of Total	Change in Load kg/yr	Change in Load % of Total With Det. P
Direct Drainage						
a. Marsh and Swamp	10	16	10	17	0	0
b. Forest	10	16	10	17	0	0
c. Agriculture	0	0	0	0	0	0
d. Urban - Residential	10	16	10	17	0	0
e. Shoreline Residential	5	8	5	9	0	0
Subtotal:	35	56	35	60	0	0
Atmospheric	15	25	15	26	0	0
Septic Tanks	12(a)	19	8(a)	14	4	6
WWTP (Lagoon)	0	0	0	0	0	0
Total	62	100	58	100	4	6

(a): This value was not rounded off in order to show the magnitude of the septic tank/tile field phosphorus load.

Table D27: Swan Lake Phosphorus Loads.

Source	1978 With Det. P kg/yr	1978 With Det. P % of Total	1978 Without Det. P kg/yr	1978 Without Det. P % of Total	Change in Load kg/yr	Change in Load % of Total With Det. P
Direct Drainage	320	7	320	8	0	0
Atmospheric	30	1	30	1	0	0
Septic Tanks	44(a)	1	31(a)	1	13	<1
WWTP (20% removal)	1730	39	1220	31	510	11
Fox River Inlet	2350	52	2350	59	0	0
Total	4474	100	3951	100	523	12

(a): This value was not rounded off in order to show the magnitude of the septic tank/tile field phosphorus load.

Table D28: Townline Lake Phosphorus Loads.

Source	1978 With Det. P kg/yr	1978 With Det. P % of Total	1978 Without Det. P kg/yr	1978 Without Det. P % of Total	Change in Load kg/yr	Change in Load % of Total With Det. P
Direct Drainage						
a. Marsh and Swamp	5	<1	5	<1	0	0
b. Forest	5	<1	5	<1	0	0
c. Agriculture	5	<1	5	<1	0	0
d. Shoreline Residential	5	<1	5	<1	0	0
Subtotal:	20	3	20	3	0	0
Atmospheric	10	2	10	2	0	0
Septic Tanks	30(a)	4	21(a)	3	9	1
WWTP (20% removal)	54	8	38	6	16	2
Tributaries:						
a. Maple Lake Inlet	25	4	25	4	0	0
b. Townline Creek	520	79	520	82	0	0
Subtotal:	545	83	545	86	0	0
Total	659	100	634	100	25	4

(a): This value was not rounded off in order to show the magnitude of the septic tank/tile field phosphorus load.

APPENDIX E

Temporal Variations in Total Phosphorus Concentrations,

Chlorophyll *a* Concentrations, and Secchi

Disc Depths in Study Lakes

Table E1: Mean pre-ban and post-ban total phosphorus concentrations, chlorophyll _a_ concentrations, and Secchi disc depths for Little Bearskin Lake and its test lakes: Enterprise, Moss, Swan, and Townline Lakes.

		Pre-ban	Ban							
		1978	1980		1981		1982		1980-1982	
Lake (Test/ Reference)	Pareameter	Mean(±s.d.)	Mean(±s.d.)	% Change from 1978	Mean(±s.d.)	% Change from 1978	Mean(±s.d.)	% Change from 1978	Mean(±s.d.)	% Change from 1978
Enterprise (Test)	Total P(ug/L)	23.0(±9.8)	48.5(±32.0)	+111	74.8(±68.0)	+225	47.3(±28.4)	+106	56.4(±44.8)	+145
	Chlorophyll _a_ (ug/L)	10.0(±6.0)	14.6(±13.9)	+46	14.1(±14.9)	+41	16.9(±15.1)	+69	15.3(±13.9)	+53
	Secchi disc depth (m)	1.9(±0.6)	2.1(±0.8)	+11	2.6(±0.9)	+37	2.0(±0.8)	+ 5	2.2(±0.8)	+16
Moss (Test)	Total P (ug/L)	26.4(±8.8)	38.8(±26.2)	+47	49.2(±20.7)	+86	22.0(±13.1)	-17	35.9(±22.4)	+36
	Chlorophyll _a_ (ug/L)	6.4(±2.8)	6.3(±2.8)	- 2	5.7(±3.4)	-11	6.7(±2.6)	+ 5	6.3(±2.8)	- 2
	Secchi disc depth (m)	2.6(±0.5)	3.3(±0.9)	+27	3.7(±0.9)	+42	3.5(±0.8)	+35	3.5(±0.8)	+35
Swan (Test)	Total P (ug/L)	77.4(±35.6)	87.9(±44.3)	+14	68.6(±16.8)	-11	54.4(±35.1)	-30	69.5(±35.1)	-10
	Chlorophyll _a_ (ug/L)	23.6(±18.3)	29.6(±31.3)	+25	37.1(±26.9)	+57	20.1(±13.16)	-15	28.5(±24.2)	+21
	Secchi disc depth (m)	1.9(±0.9)	2.2(±0.9)	+16	1.8(±0.8)	- 5	1.7(±0.6)	-11	1.8(±0.7)	-5
Townline (Test)	Total P (ug/L)	59.8(±23.9)	61.4(±32.2)	+ 3	51.6(±10.9)	-14	47.4(±27.3)	-21	53.2(±24.6)	-11
	Chlorophyll _a_ (ug/L)	21.3(±9.7)	22.2(±10.8)	+ 4	22.0(±8.7)	+ 3	14.6(±10.8)	-31	19.3(±10.3)	- 9
	Secchi disc depth (m)	1.0(±0.1)	1.5(±0.3)	+50	1.7(±0.5)	+70	2.0(±0.9)	+100	1.7(±0.6)	+70
Little Bearskin (Reference)	Total P (ug/L)	33.7(±3.6)	49.3(±32.8)	+46	51.5(±25.4)	+53	42.4(±22.2)	+26	47.5(±22.0)	+41
	Chlorophyll _a_ (ug/L)	14.6(±7.0)	12.7(±6.4)	-13	15.5(±10.5)	+ 6	12.6(±9.2)	-14	13.6(±8.5)	- 7
	Secchi disc depth (m)	2.1(±0.4)	2.6(±0.9)	+24	2.3(±0.6)	+10	2.7(±0.6)	+29	2.5(±0.7)	+19

Note: 1) Annual means are averages of monthly means of each year's sampling program: average of Sites 1 and 2: 1978(N=6), 1980(N=6), 1981(N=6), 1982(N=7), 1980-81(N=19).

2) s.d. = standard deviation.

Table E2: Mean pre-ban and post-ban total phosphorus concentrations, chlorophyll _a_ concentrations, and Secchi disc depths for Teal Lake and its test lakes: Balsam, Butternut and Elk Lakes.

		Pre-ban	Ban							
		1978	1980		1981		1982		1980-1982	
Lake (Test/ Reference)	Parameter	Mean(±s.d.)	Mean(±s.d)	% Change from 1978	Mean(±s.d.)	% Change from 1978	Mean(±s.d.)	% Change from 1978	Mean(±s.d.)	% Change from 1978
Balsam (Test)	Total P (ug/L)	41.3(±16.9)	48.2(±29.7)	+17	127.8(±139.0)[a]	+209	43.7(±22.3)	+ 6	71.7(±85.5)[b]	+74
	Chlorophyll _a_ (ug/L)	22.5(±13.2)	18.4(±14.5)	-18	25.8(±19.0)	+15	13.9(±10.3)	-38	20.3(±12.4)	-10
	Secchi disc depth (m)	1.8(±0.6)	2.4(±0.7)	+33	2.4(±0.5)	+33	2.0(±0.4)	+11	2.2(±0.6)	+11
Butternut (Test)	Total P (ug/L)	60.9(±29.3)	76.7(±34.4)	+26	77.8(±21.1)	+28	65.0(±25.5)	+7	72.7(±26.5)	+19
	Chlorophyll _a_ (ug/L)	27.6(±28.1)	23.7(±17.1)	-14	27.8(±21.4)	+ 1	27.2(±25.1)	-1	26.3(±20.5)	- 5
	Secchi disc depth (m)	1.1(±0.6)	1.5(±0.6)	+36	1.5(±0.4)	+36	1.3(±0.2)	+18	1.4(±0.4)	+27
Elk (Test)	Total P (ug/L)	72.4(±18.2)	80.9(±27.4)	+12	61.7(±21.9)	-15	54.3(±16.9)	-25	65.0(±23.8)	-10
	Chlorophyll _a_ (ug/L)	7.4(±2.7)	28.3(±29.5)	+282	19.0(±8.2)	+157	15.5(±12.1)	+109	20.6(±18.4)	+178
	Secchi disc depth (m)	0.7(±0.2)	0.9(±0.3)	+29	1.2(±0.3)	+71	1.2(±0.2)	+71	1.1(±0.3)	+57
Teal(Reference)	Total P (ug/L)	25.9(±3.8)	47.6(±36.6)	+84	98.7(±139.8)[c]	+281	35.7(±23.3)	+38	59.4(±82.2)[d]	+129
	Chlorophyll _a_ (ug/L)	13.5(±7.6)	13.2(±5.8)	- 2	20.3(±18.2)	+50	15.9(±9.6)	+18	16.4(±11.9)	+21
	Secchi disc depth (m)	1.9(±0.6)	2.8(±1.2)	+47	2.4(±0.9)	+26	1.8(±0.5)	-5	2.3(±0.9)	+21

Notes:
1) Annual means are averages of monthly means of each year's sampling program; average of sites 1 and 2: 1978(N=6), 1980(N=6), 1981(N=6), 1982(N=7), 1980-82(N=19).
2) s.d. = standard deviation

Footnotes:
(a) 1981 Mean is 71.9(±27.4), +74%, without exceptionally high value from the last summer sampling date in 1981.
(b) 1980-82 Mean is 53.0(±27.6), +28%, without exceptionally high value from the last summer sampling date in 1981.
(c) 1981 Mean is 41.9(±15.8), +62%, without exceptionally high value from the last summer sampling date in 1981.
(d) 1980-82 Mean is 41.4(±25.9), +60%, without exceptionally high value from the last summer sampling date in 1981.

FIGURE E1A: TEMPORAL VARIATION OF TOTAL PHOSPHORUS IN SWAN AND LITTLE BEARSKIN LAKES.

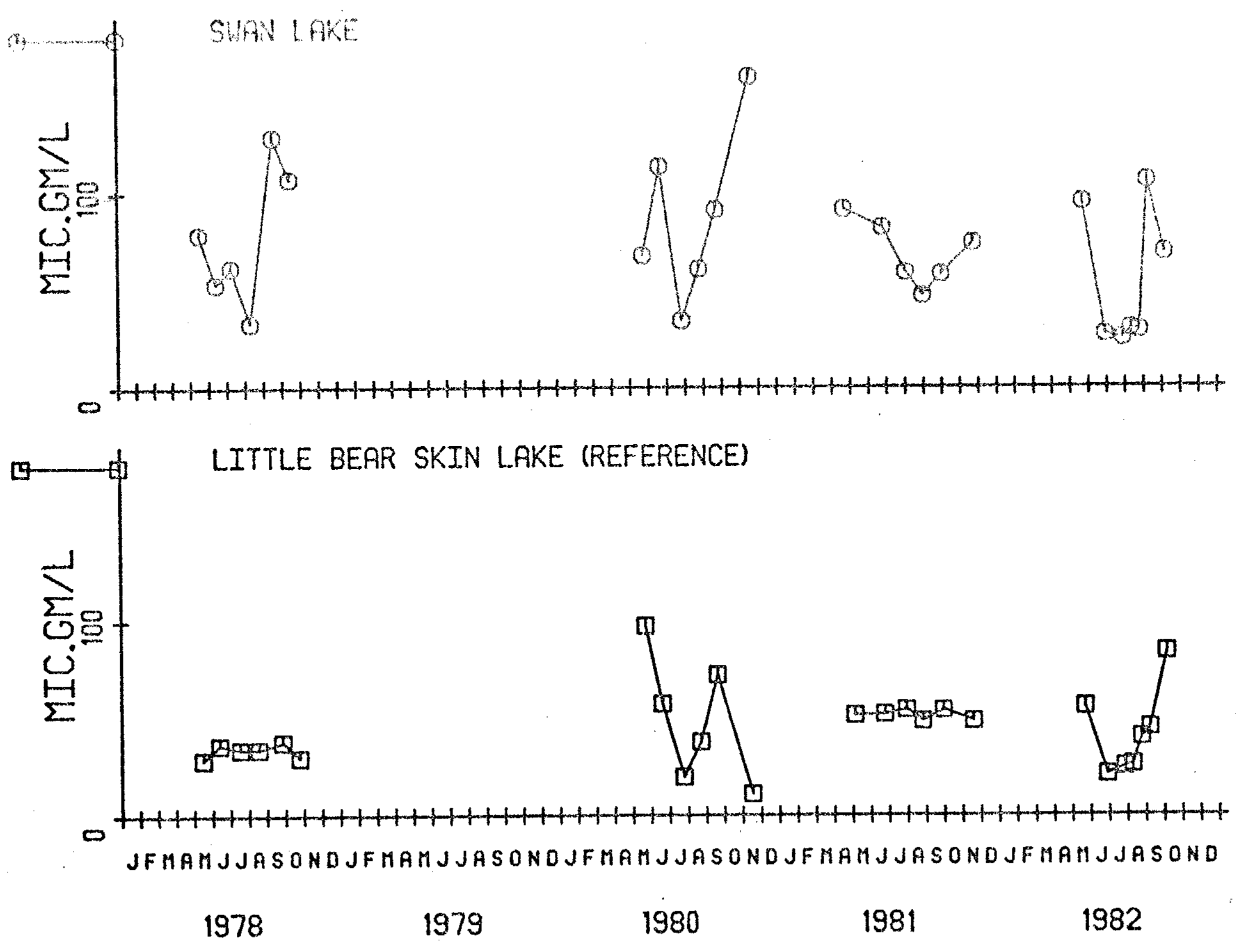
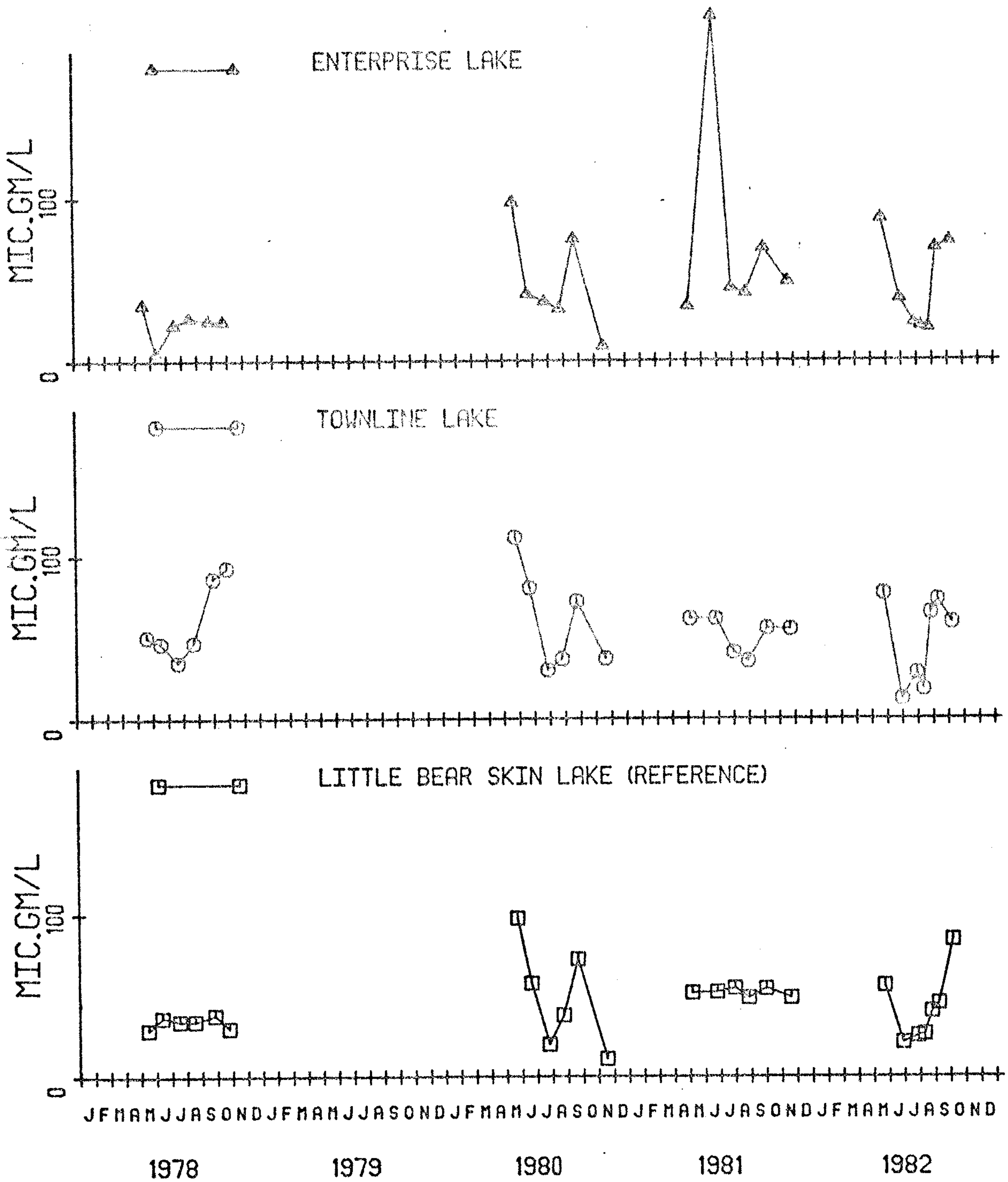
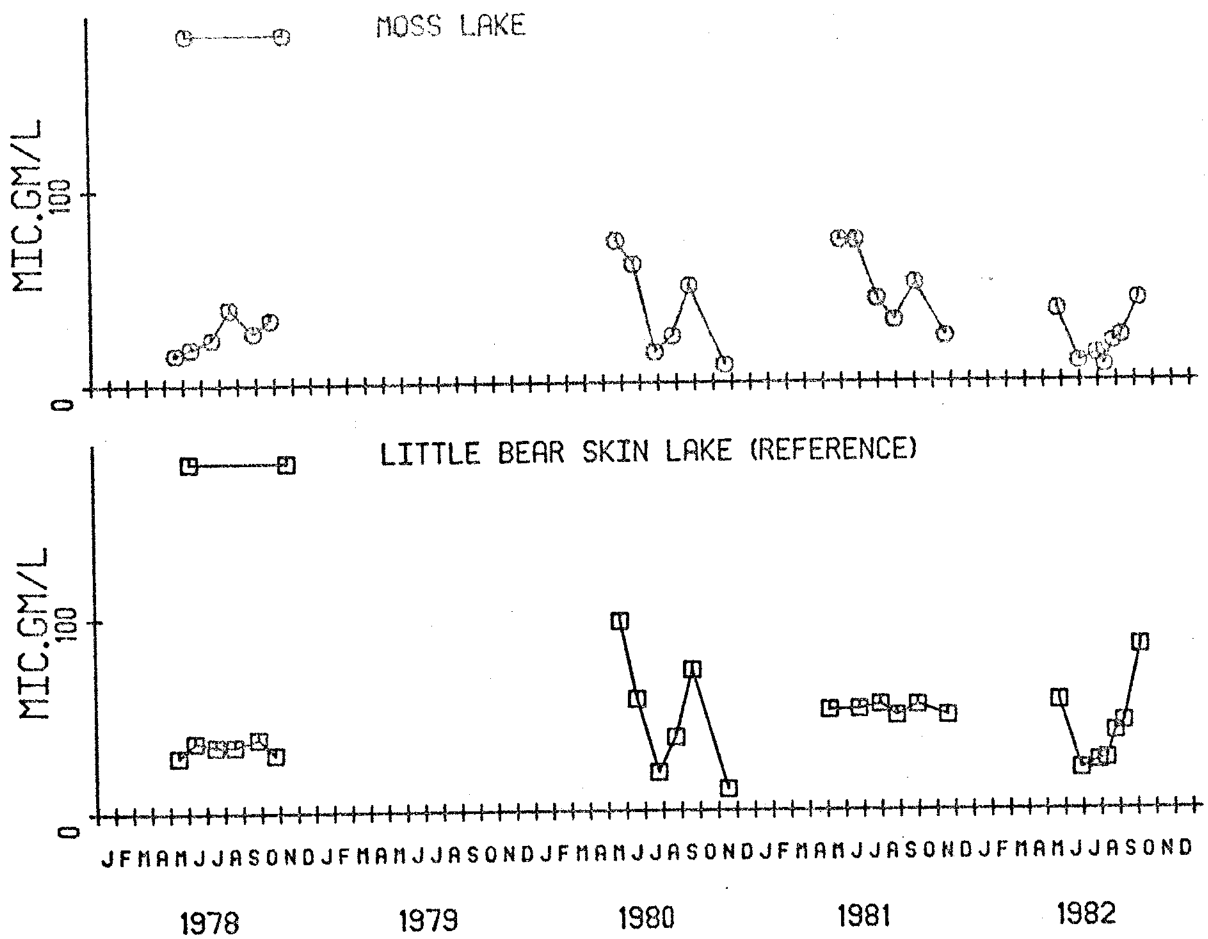


FIGURE E1B: TEMPORAL VARIATION OF TOTAL PHOSPHORUS IN ENTERPRISE, TOWNLINE, AND LITTLE BEARSKIN LAKES.



525991, Pen=L1, Scale=1., Plot No.=044

FIGURE E1C: TEMPORAL VARIATION OF TOTAL PHOSPHORUS IN MOSS AND LITTLE BEARSKIN LAKES.



525080, Pen=LI, Scale=1., Plot No.=015

FIGURE E1D: TEMPORAL VARIATION OF TOTAL PHOSPHORUS IN BALSAM, BUTTERNUT, AND TEAL LAKES.

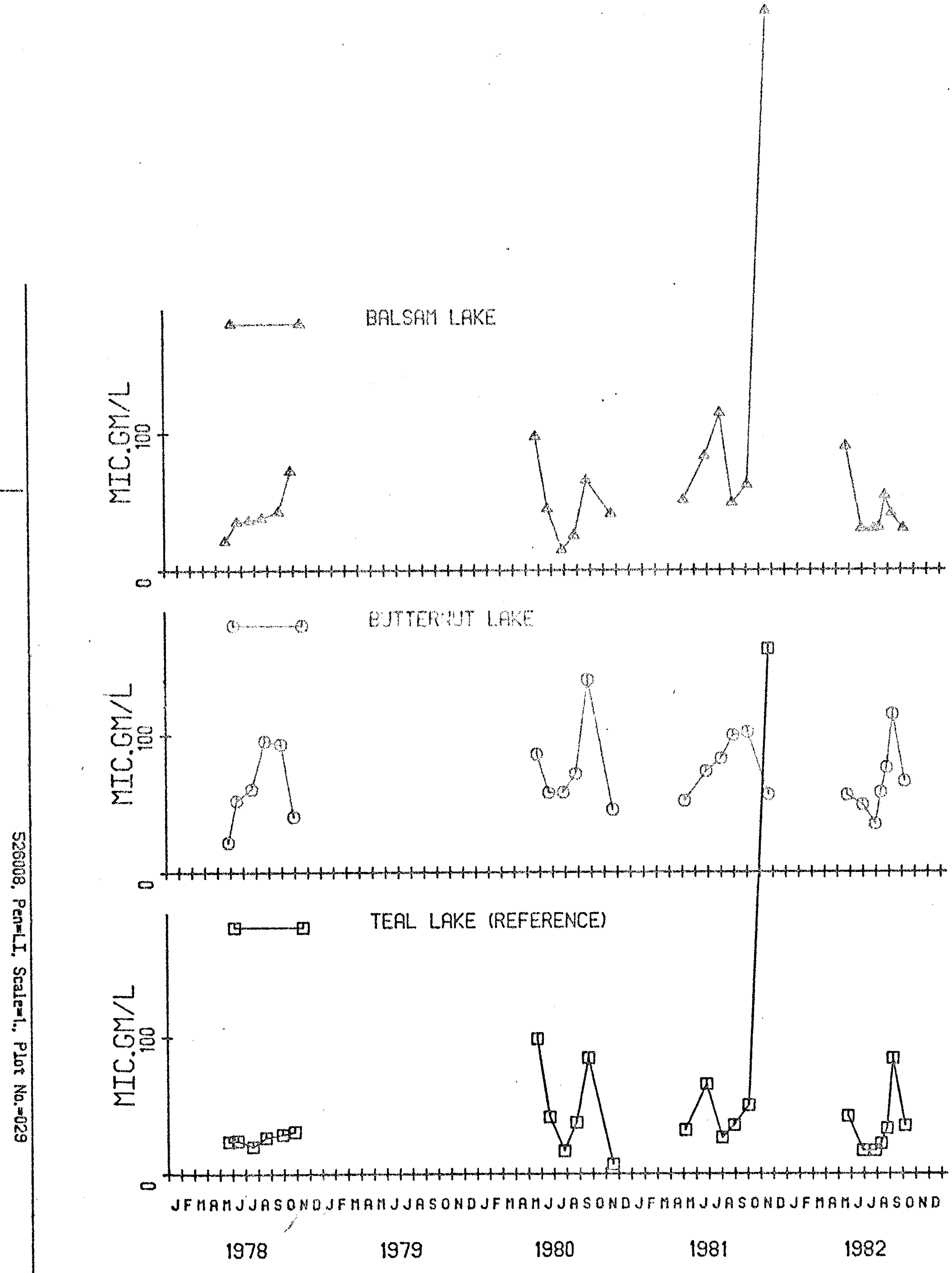
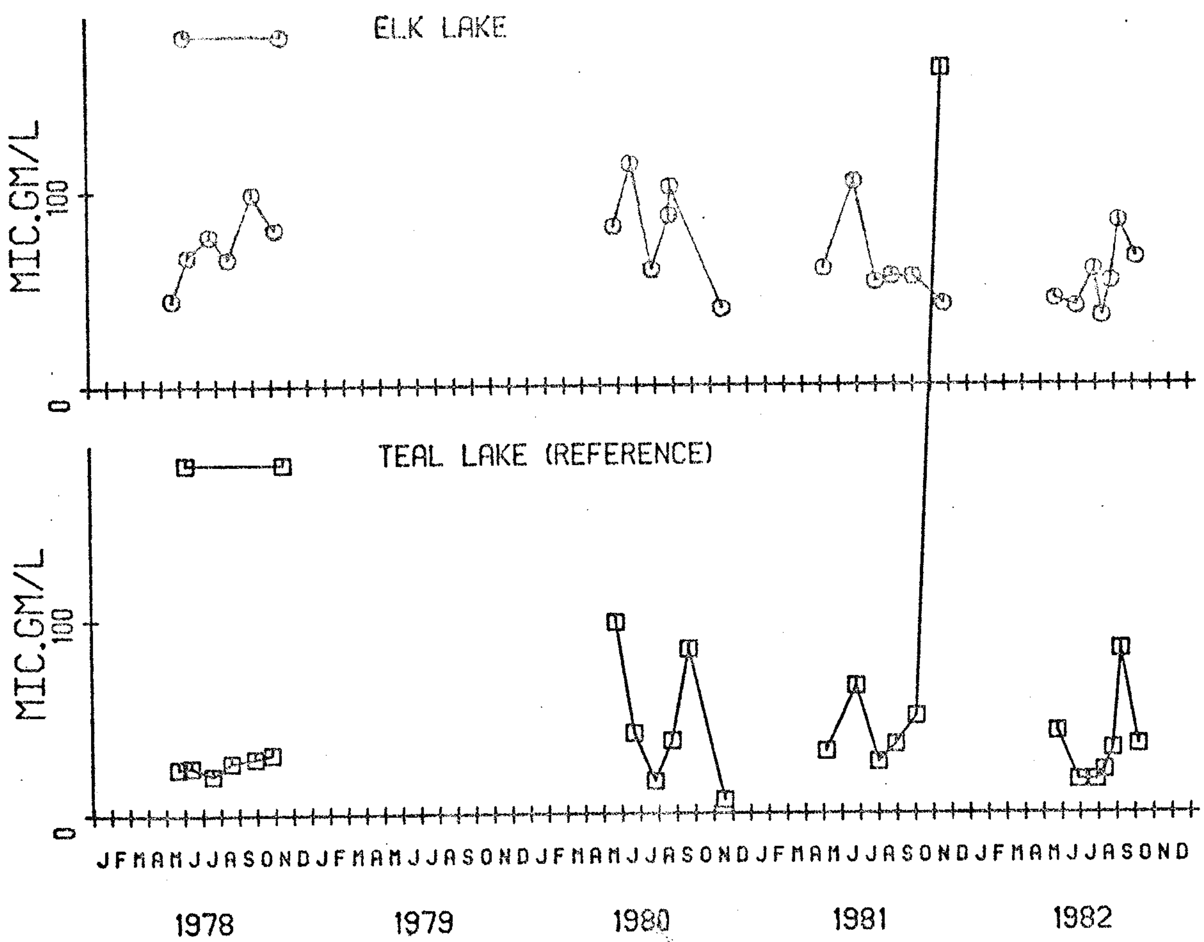


FIGURE E1E: TEMPORAL VARIATION OF TOTAL PHOSPHORUS IN ELK AND TEAL LAKES.



526021, Pen=LI, Scale=1., Plot No.=026

FIGURE E2A: TEMPORAL VARIATION OF CHLOROPHYLL A IN SWAN AND LITTLE BEARSKIN LAKES.

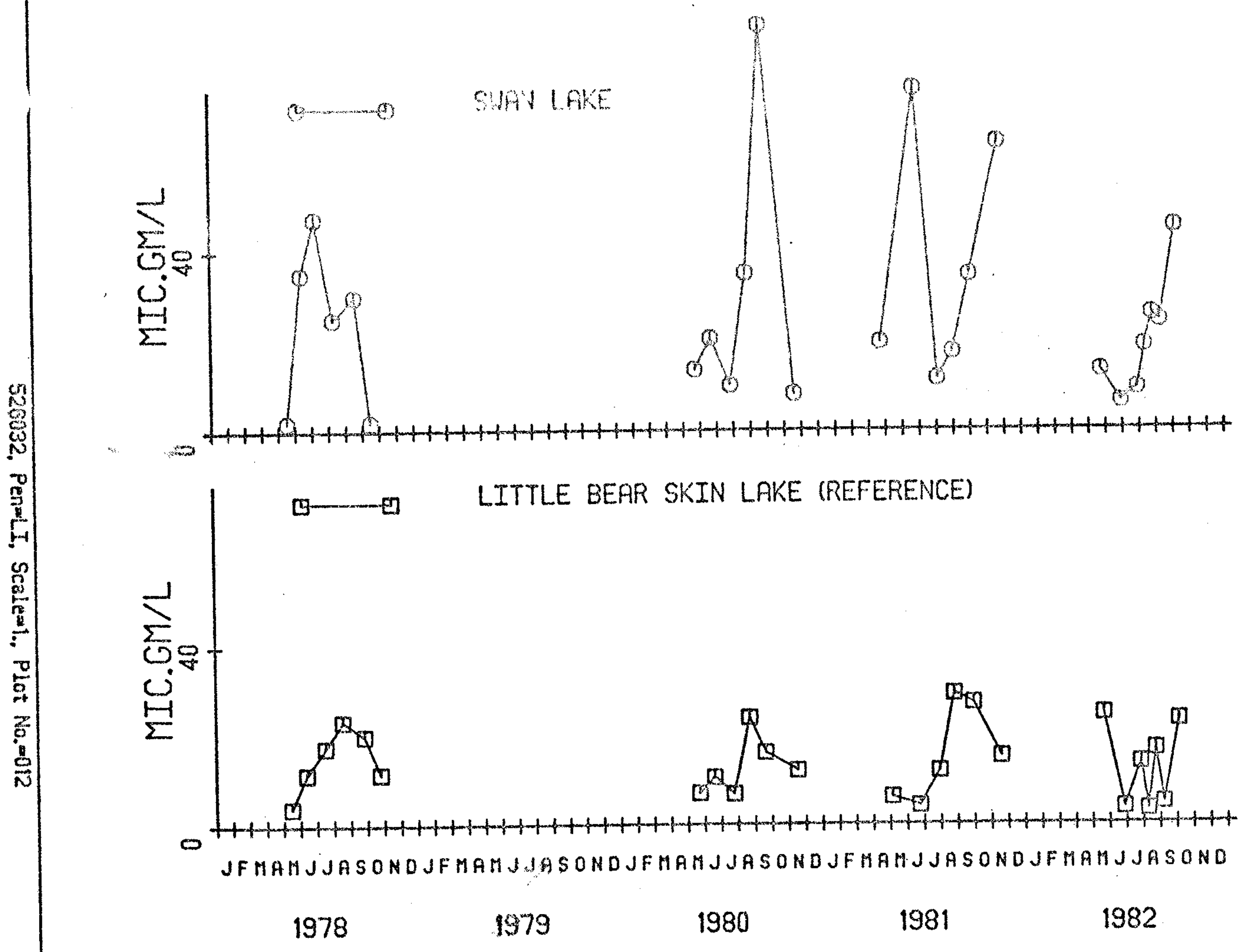


FIGURE E2B: TEMPORAL VARIATION OF CHLOROPHYLL-A IN ENTERPRISE, TOWNLINE, AND LITTLE BEARSKIN LAKES.

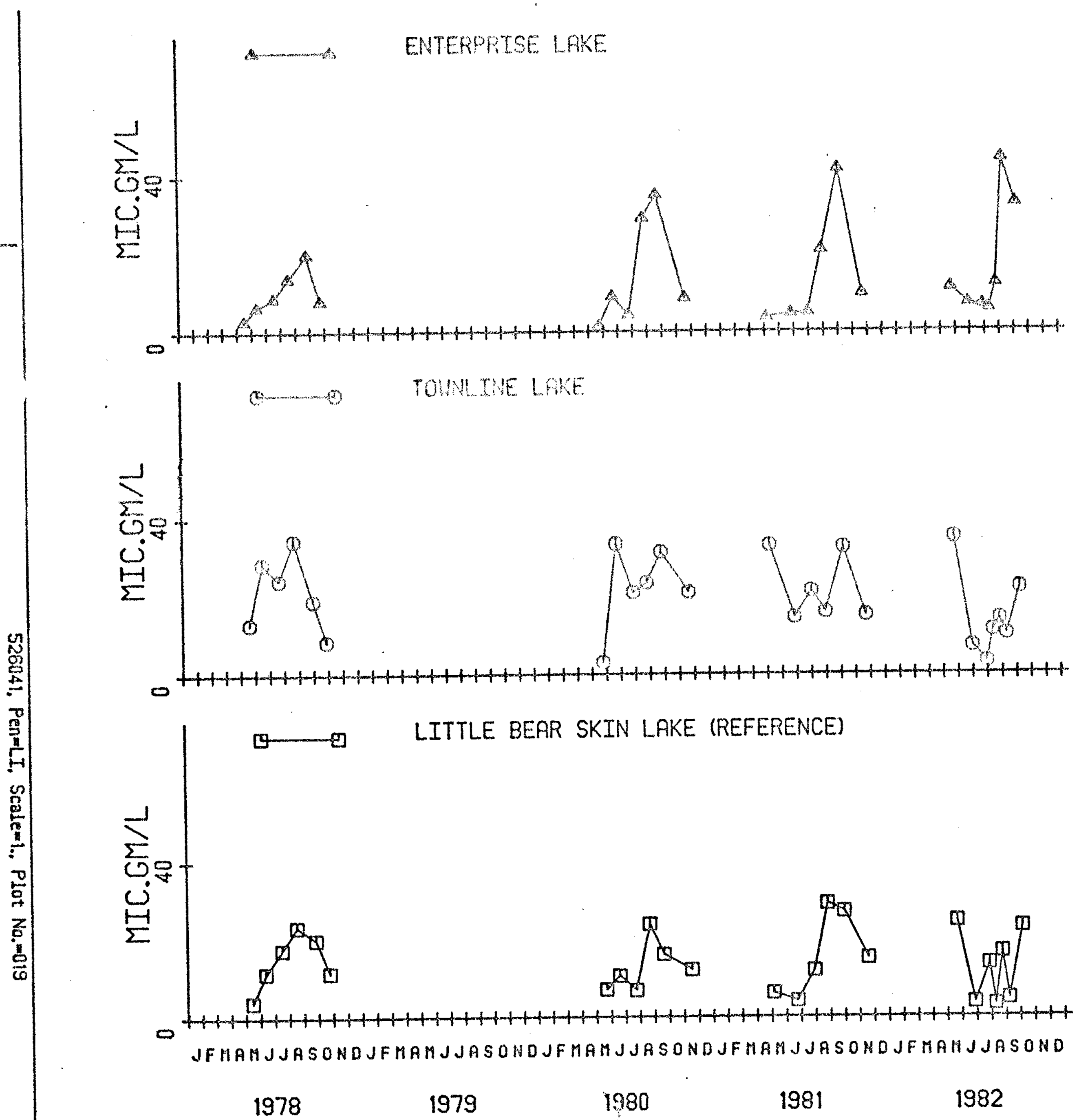
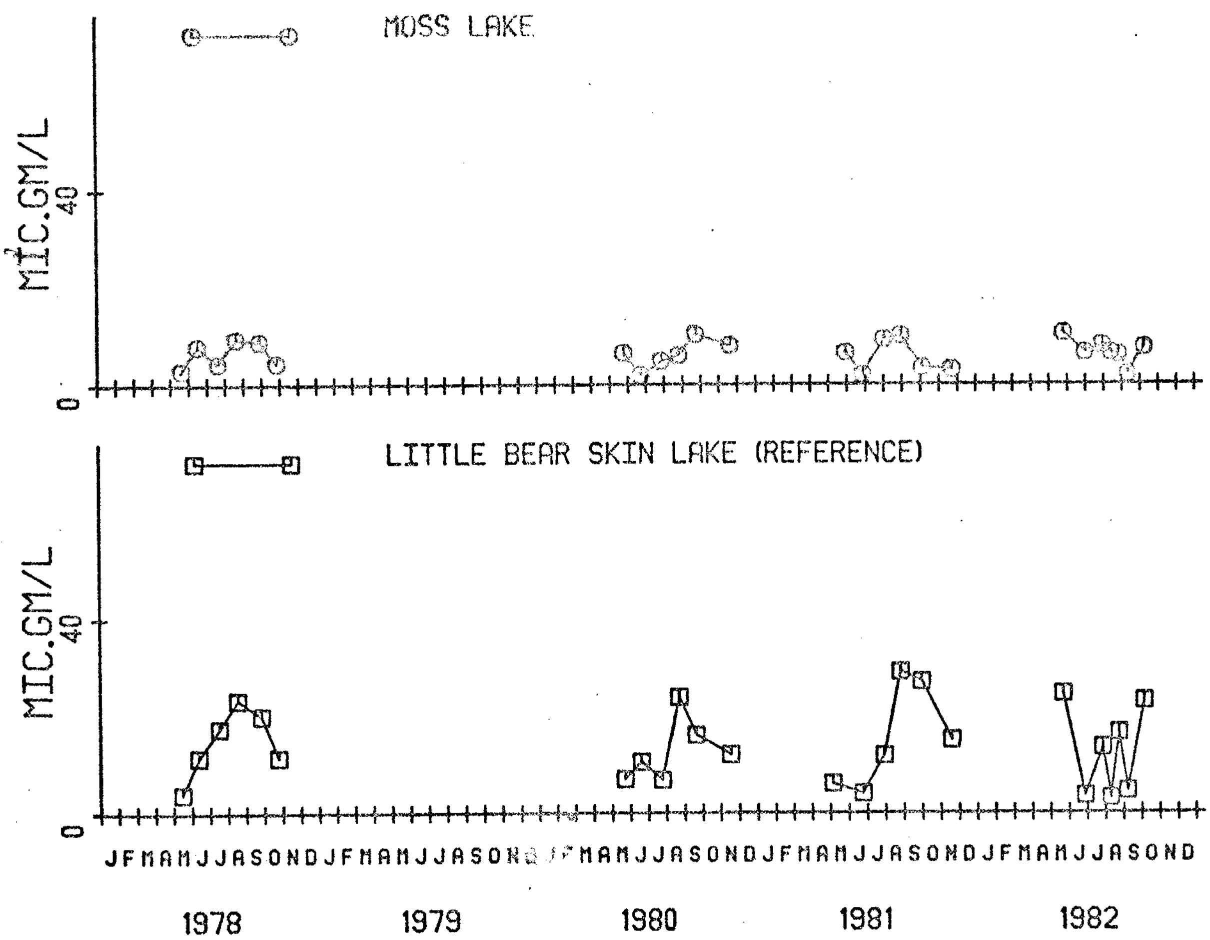
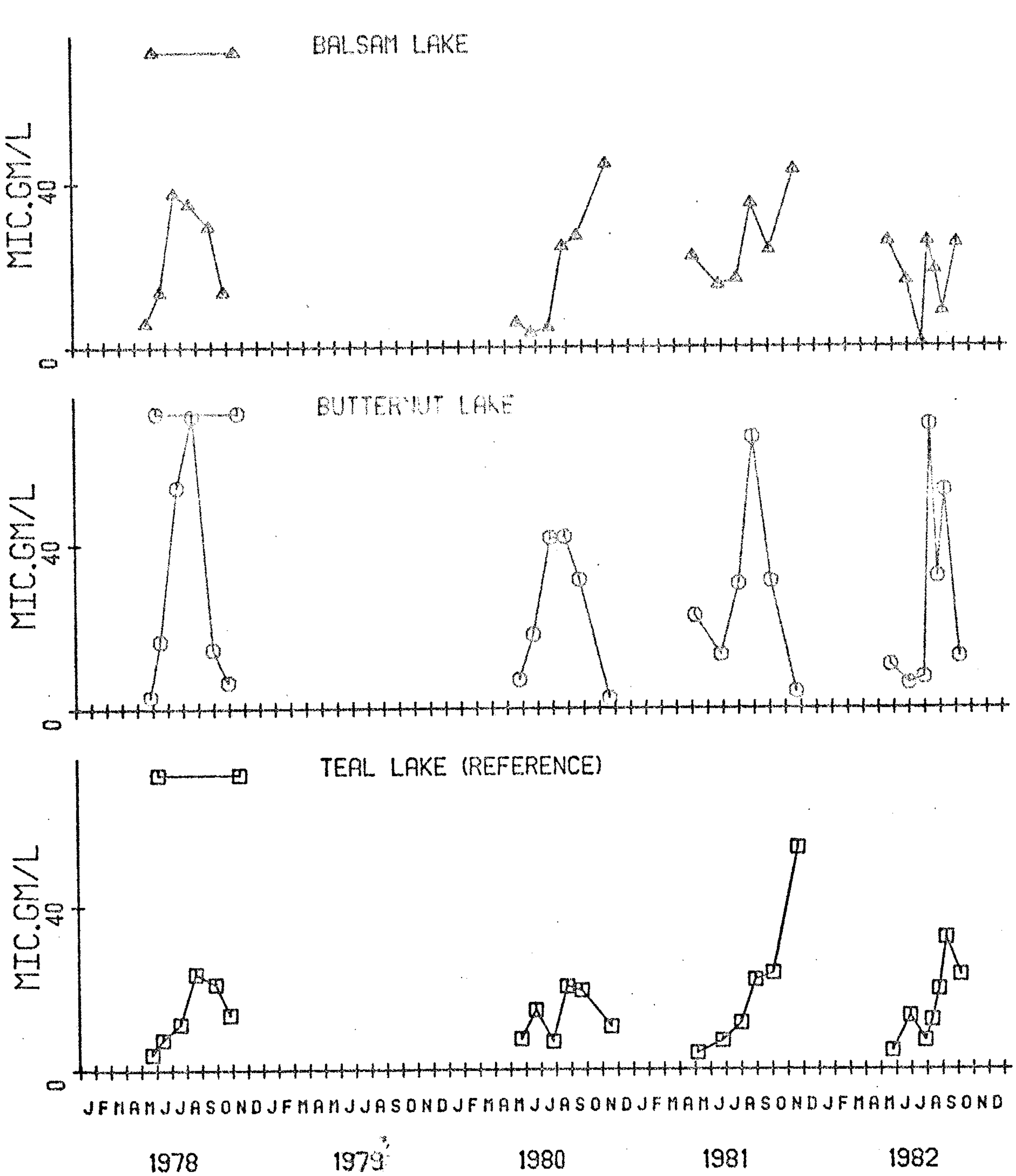


FIGURE E2C: TEMPORAL VARIATION OF CHLOROPHYLL A IN MOSS AND LITTLE BEARSKIN LAKES.



526053, Pen=LI, Scale=1, Plot No.=033

FIGURE E2D: TEMPORAL VARIATION OF CHLOROPHYLL A IN BALSAM, BUTTERNUT, AND TEAL LAKES.



52006S, Run=LI, Scale=1, Plot No.=011

FIGURE E2E: TEMPORAL VARIATION OF CHLOROPHYLL A IN ELK AND TEAL LAKES.

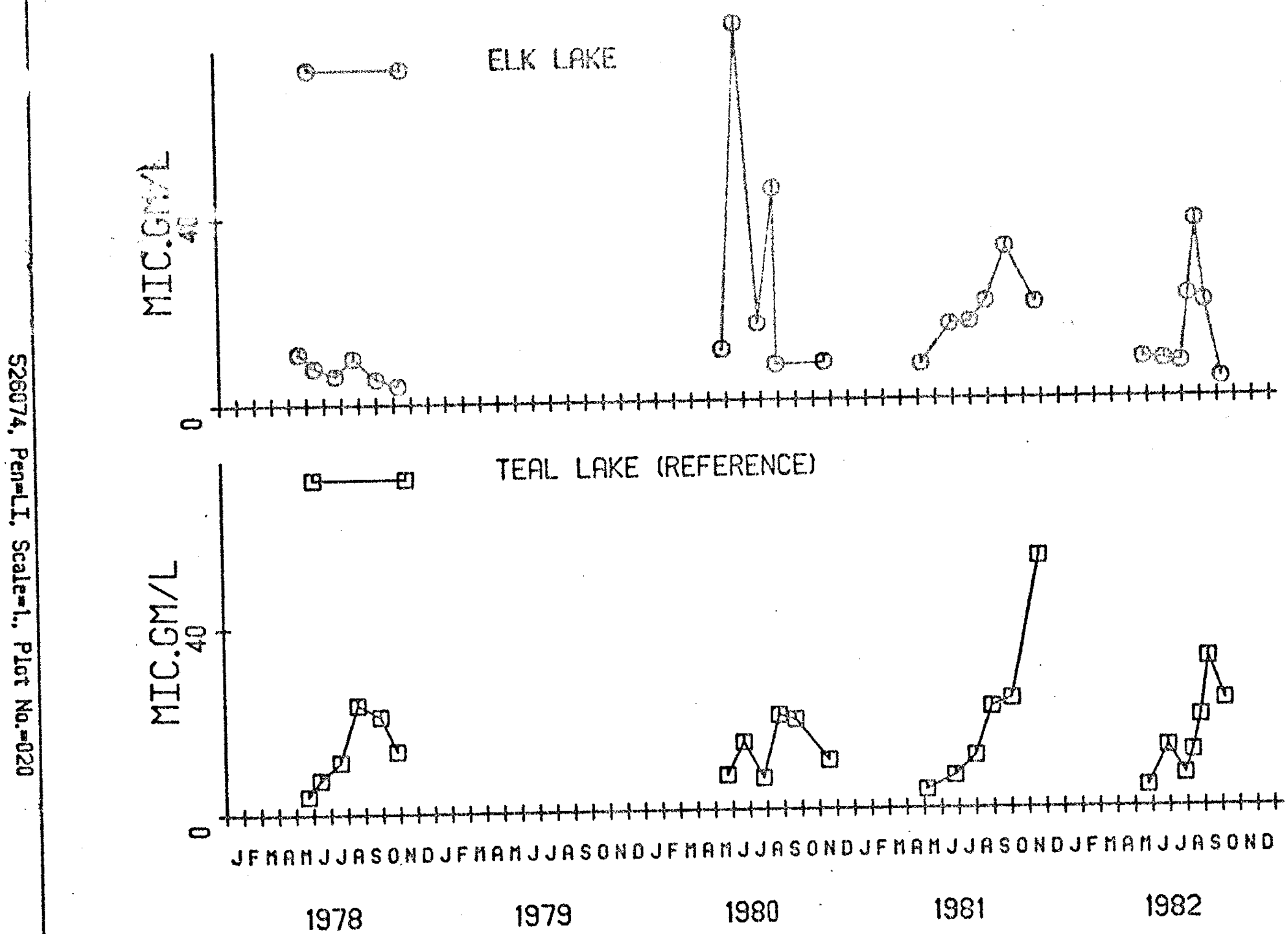
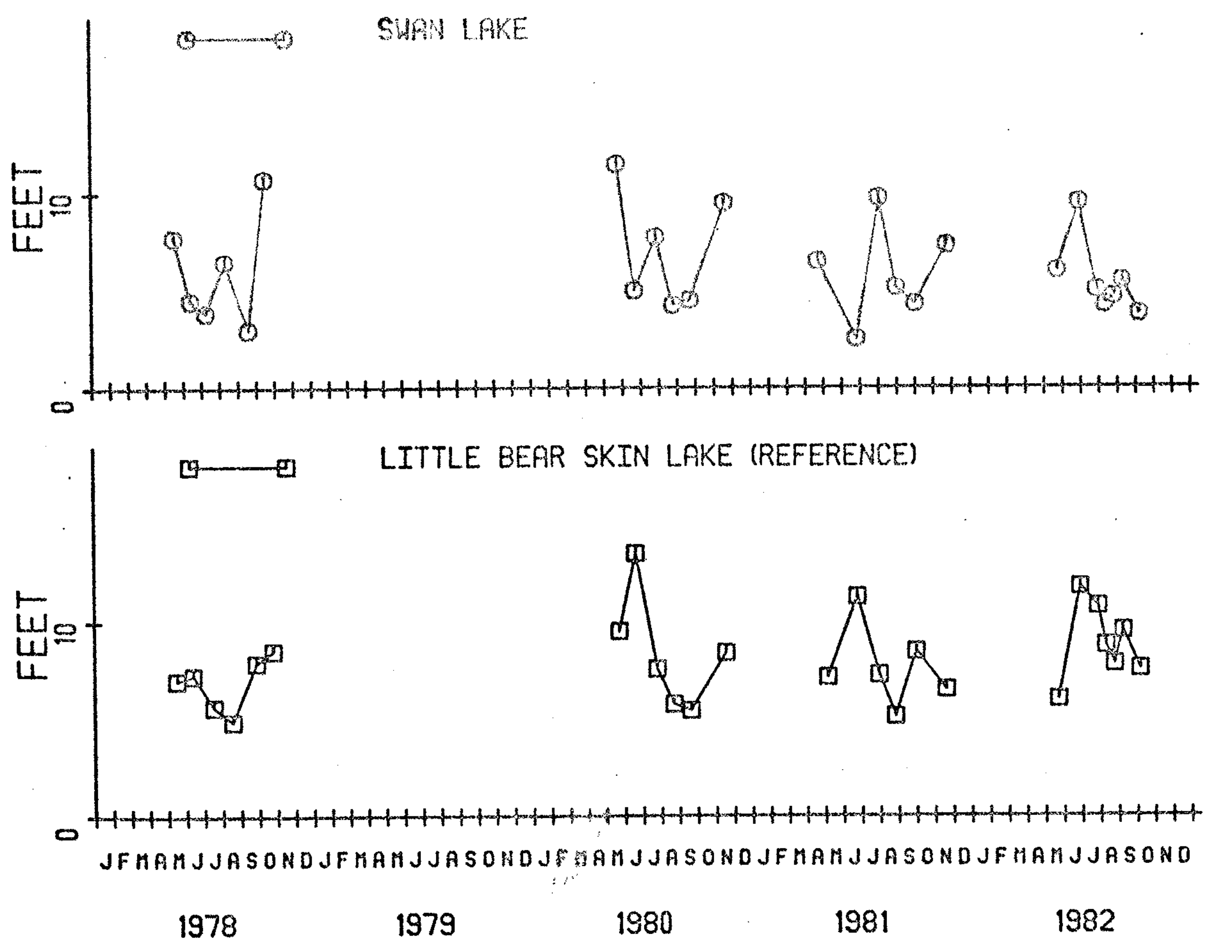
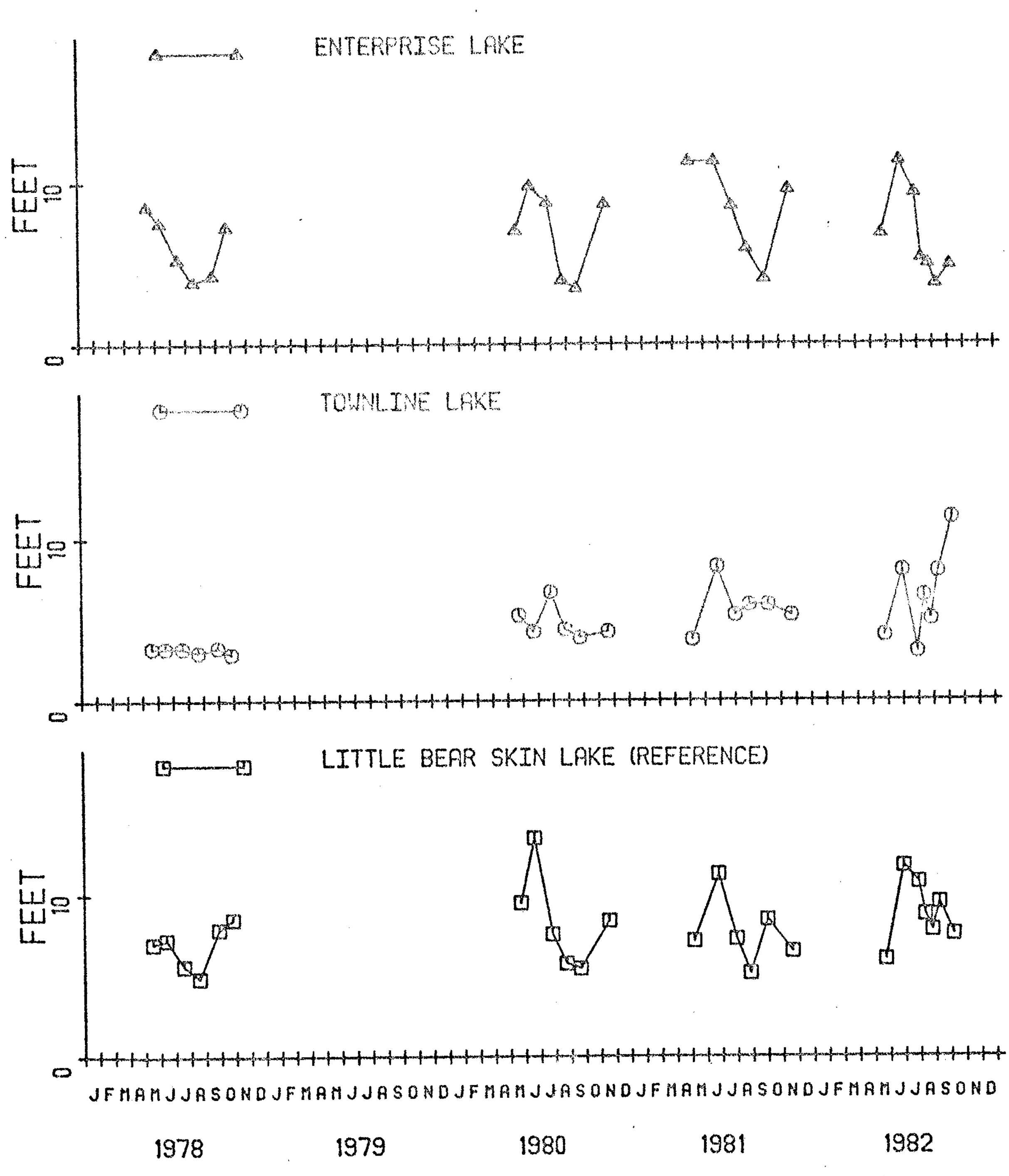


FIGURE E3A: TEMPORAL VARIATION OF SECCHI DISC DEPTH IN SWAN AND LITTLE BEARSKIN LAKES.



526087, Pen=LI, Scale=1, Plot No.=016

FIGURE E3B: TEMPORAL VARIATION OF SECCHI DISC DEPTH IN ENTERPRISE, TOWNLINE, AND LITTLE BEARSKIN LAKES.



526104, Pen=LI, Scale=1., Plot No.=002

FIGURE E3C: TEMPORAL VARIATION OF SECCHI DISC DEPTH IN MOSS AND LITTLE BEARSKIN LAKES.

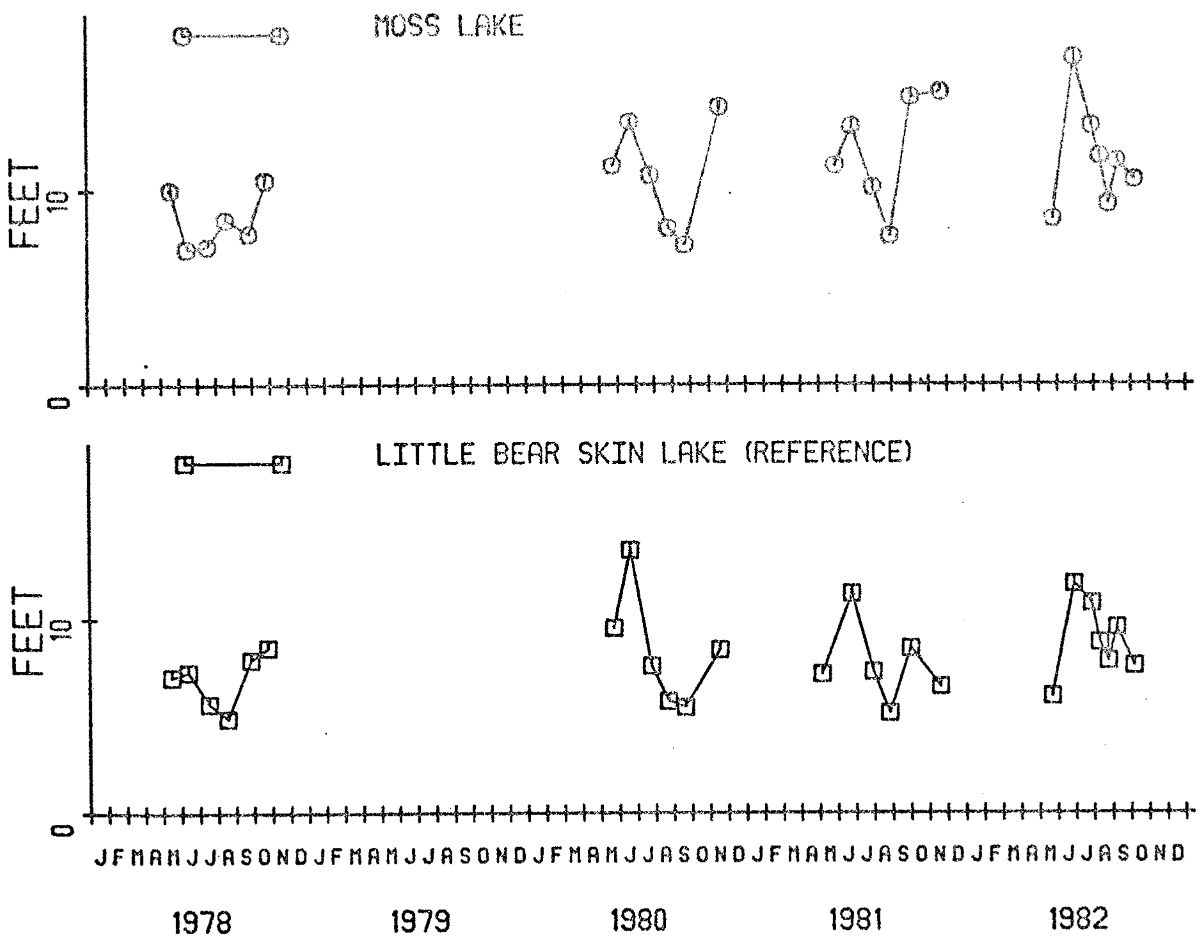


FIGURE E3D: TEMPORAL VARIATION OF SECCHI DISC DEPTH IN BALSAM, BUTTERNUT, AND TEAL LAKES.

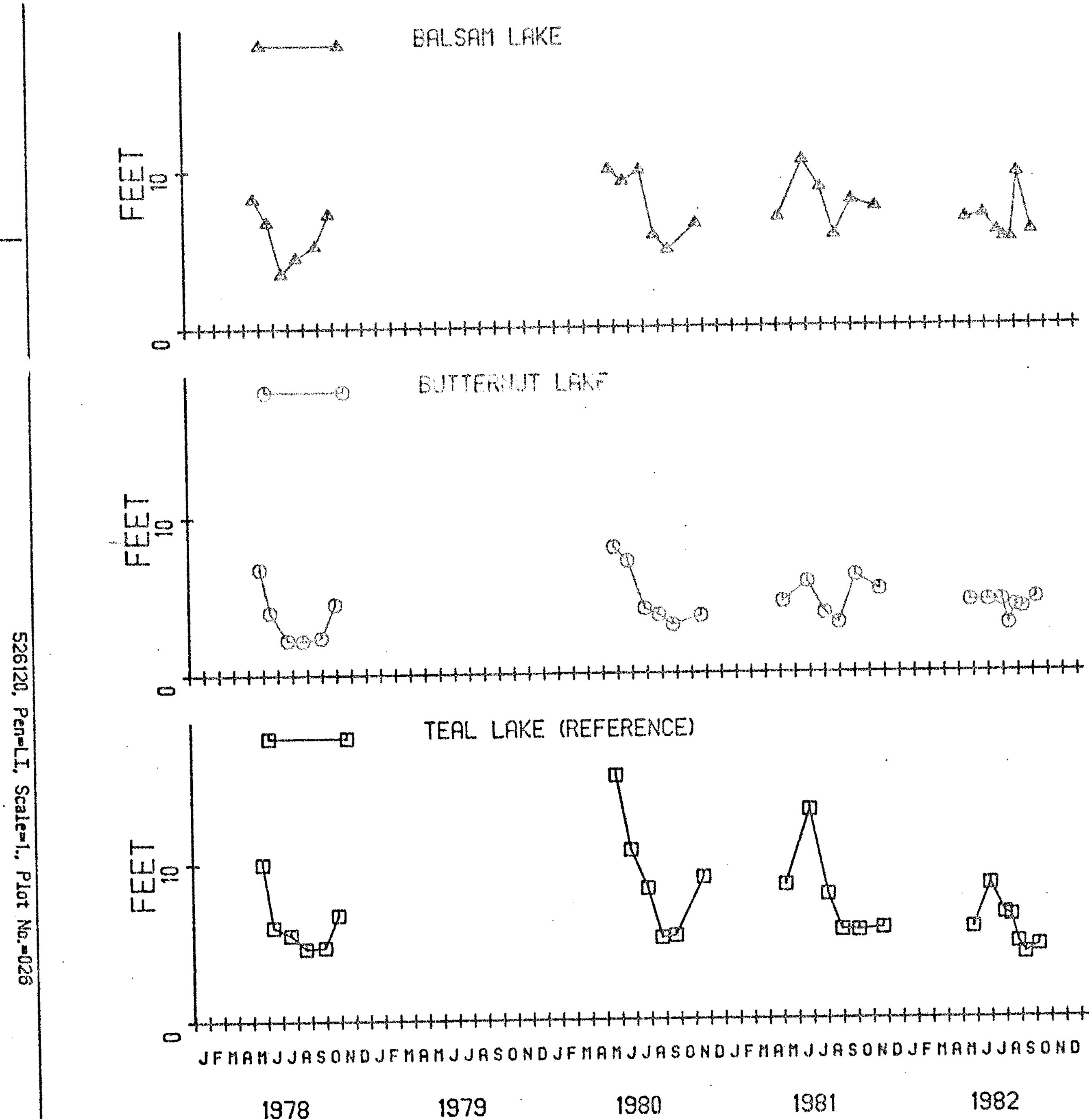
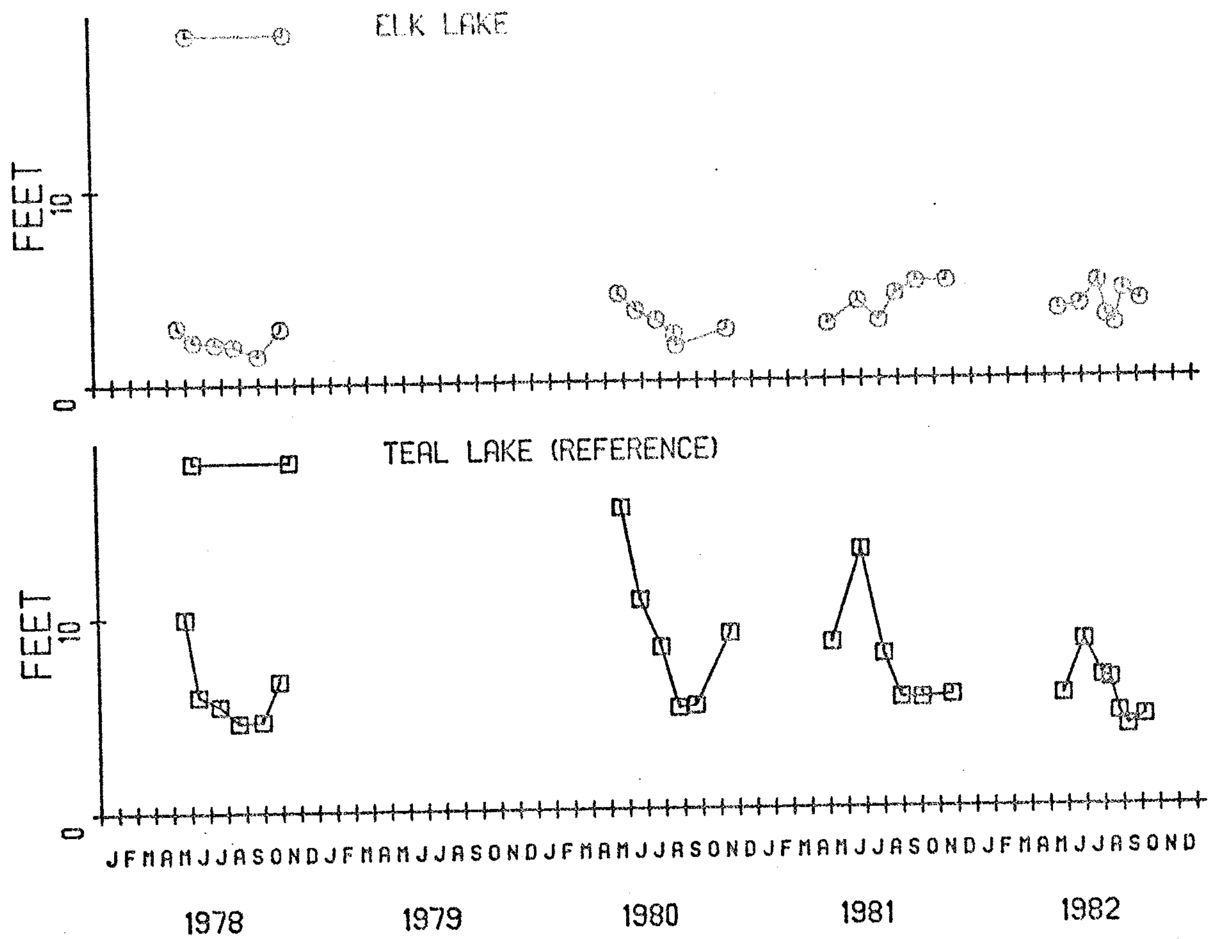


FIGURE E3E: TEMPORAL VARIATION OF SECCHI DISC DEPTH IN ELK AND TEAL LAKES.



527218, Pen=LI, Scale=1, Plot No.=010

APPENDIX F

Secchi Disc Depth and Chlorophyll a Relationships in Townline and Elk Lakes

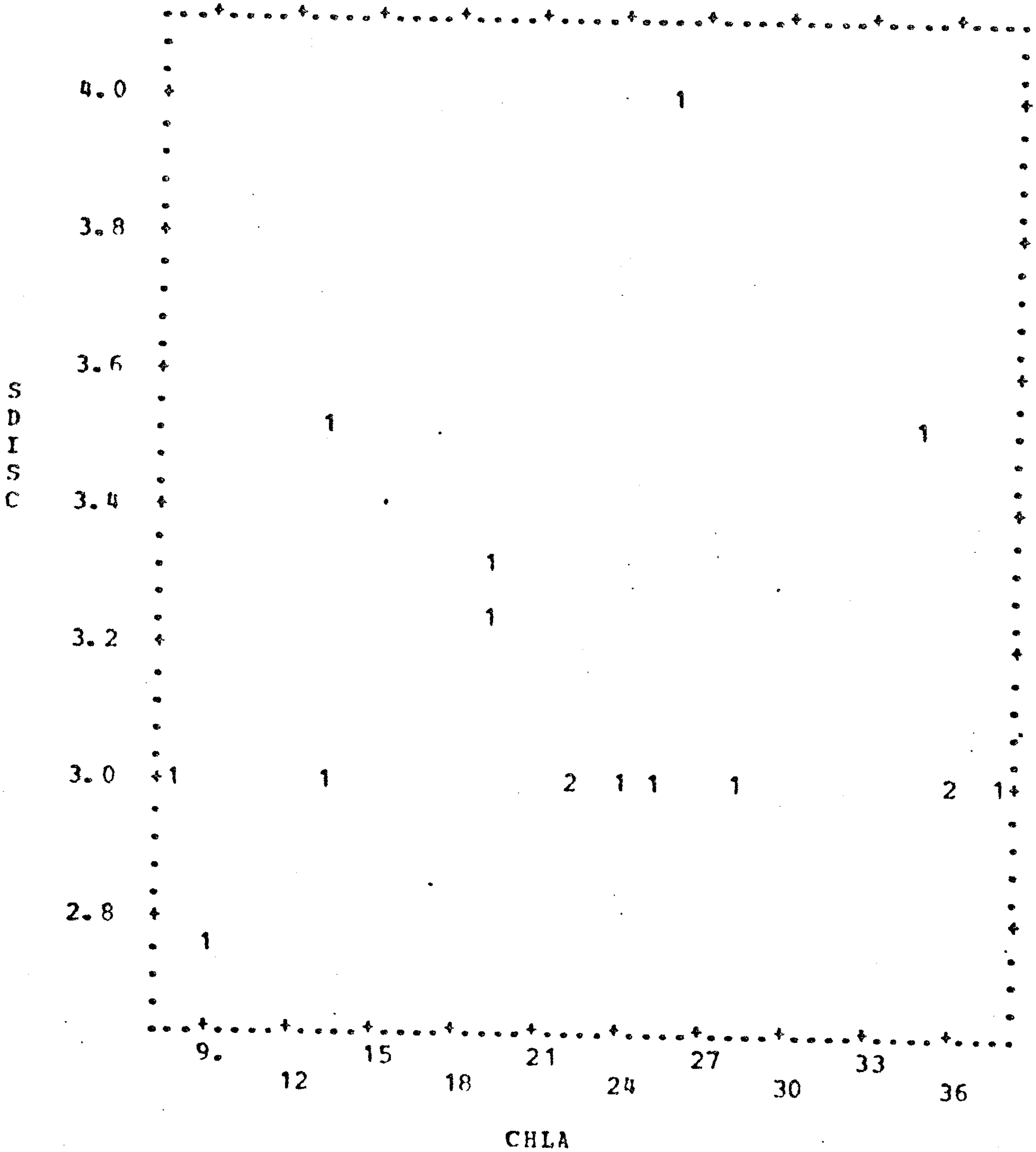


FIGURE F1: Townline Lake Secchi disc depth (ft) versus chlorophyll a (µg/L) 1978.

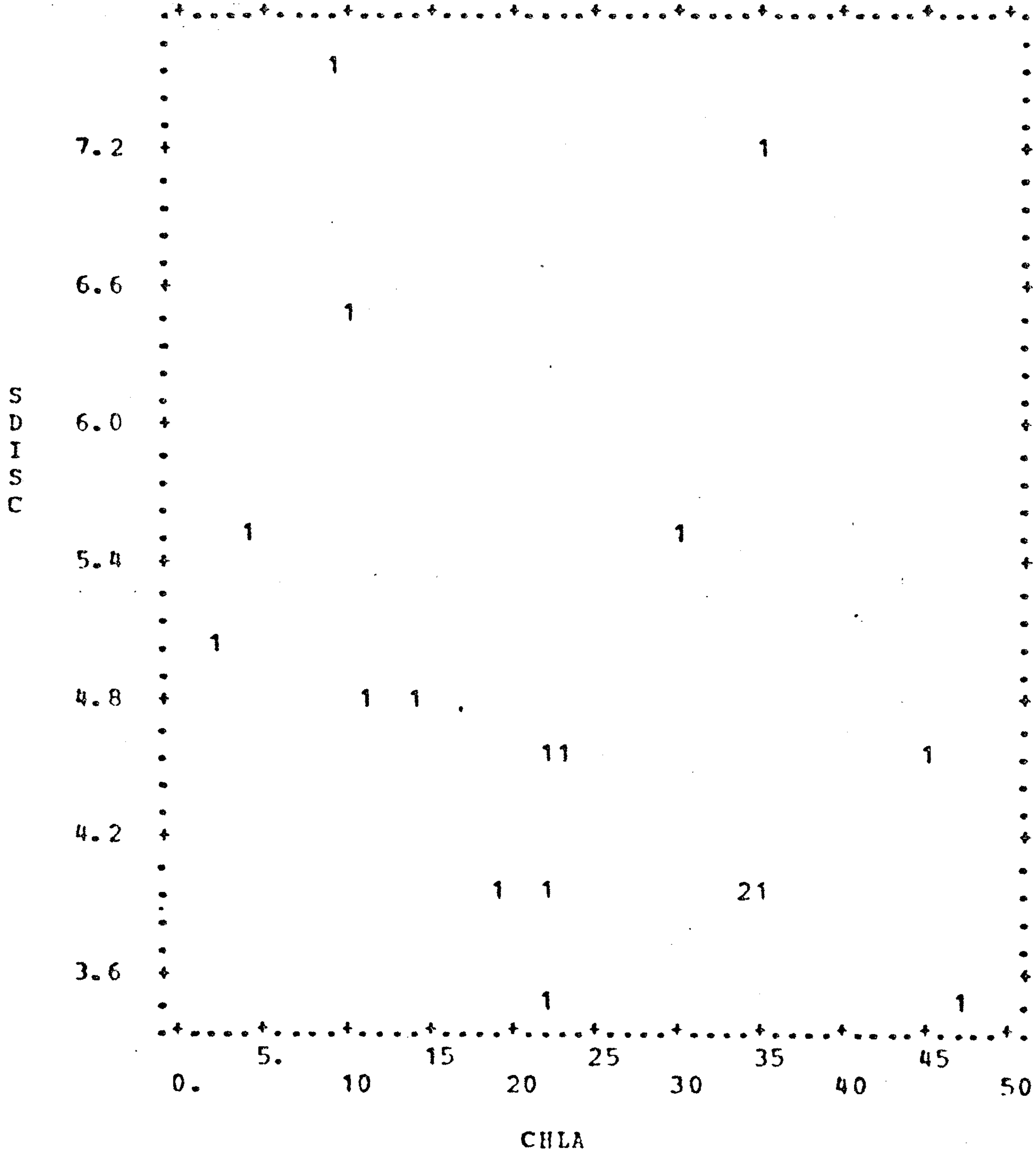


FIGURE F2: Townline Lake Secchi disc depth (ft) versus chlorophyll a (μg/L) 1980.

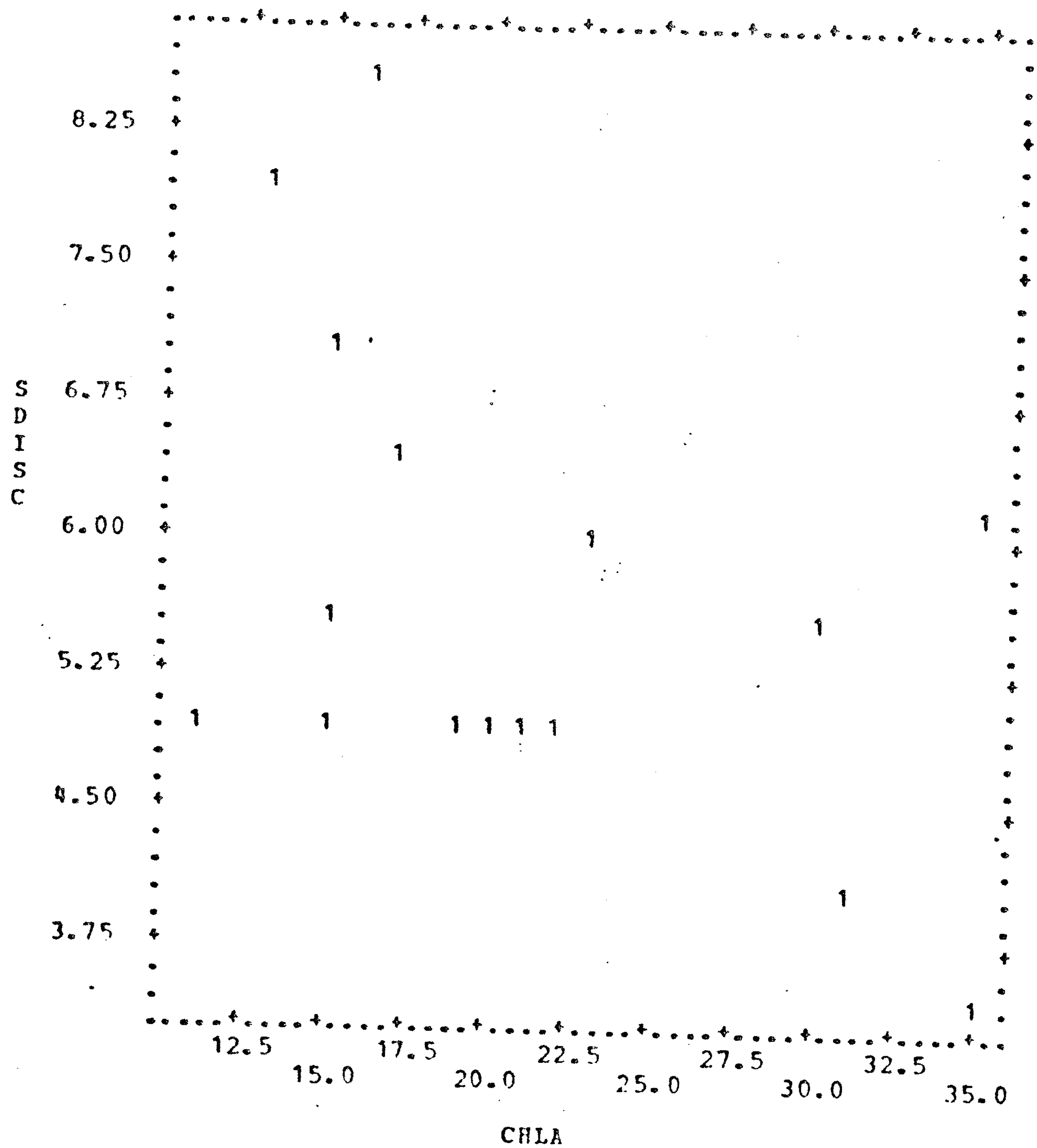


FIGURE F3: Townline Lake Secchi disc depth (ft) versus chlorophyll a (μg/L) 1981.

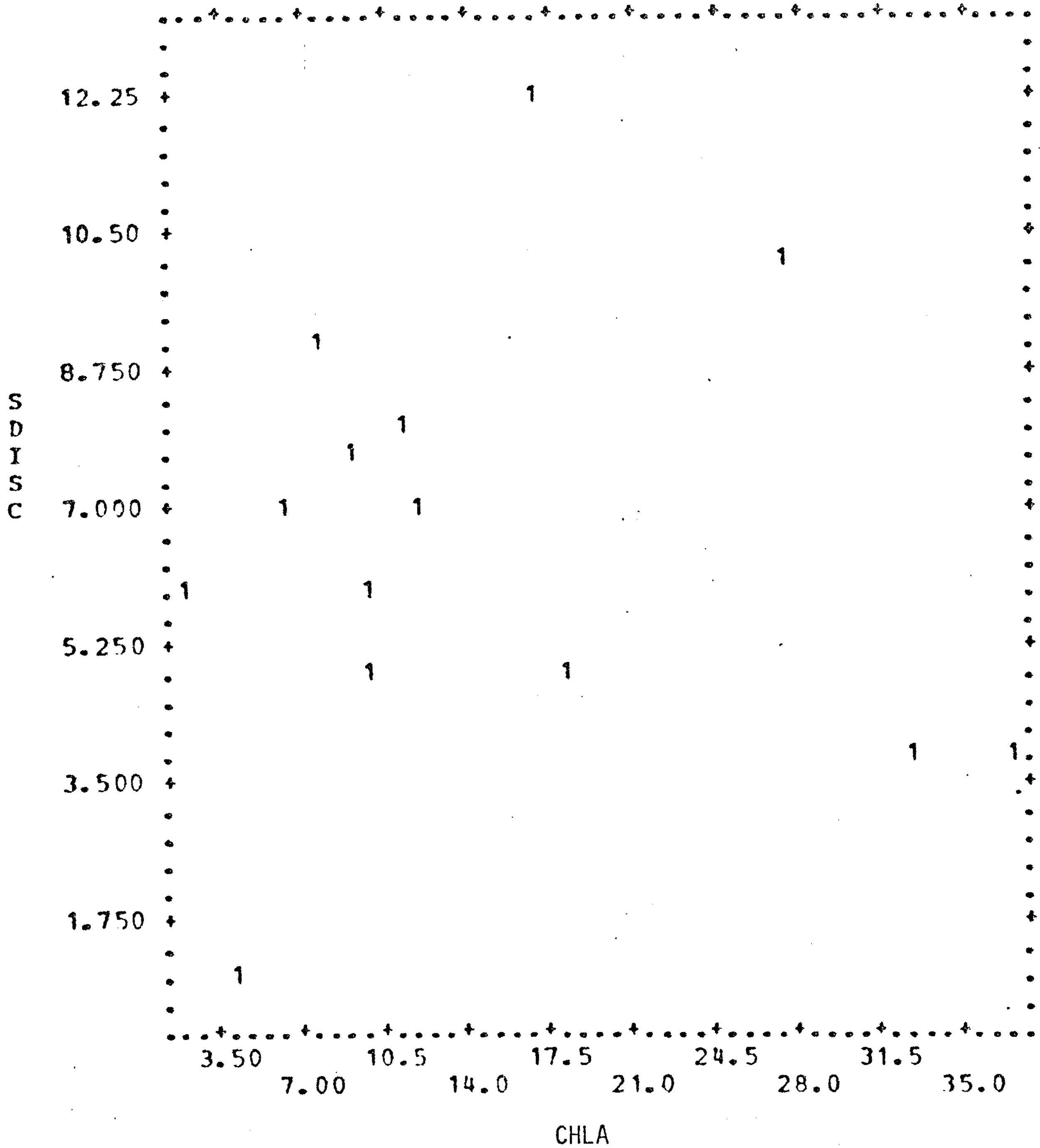


FIGURE F4: Townline Lake Secchi disc depth (ft) versus chlorophyll a (µg/L) 1982.

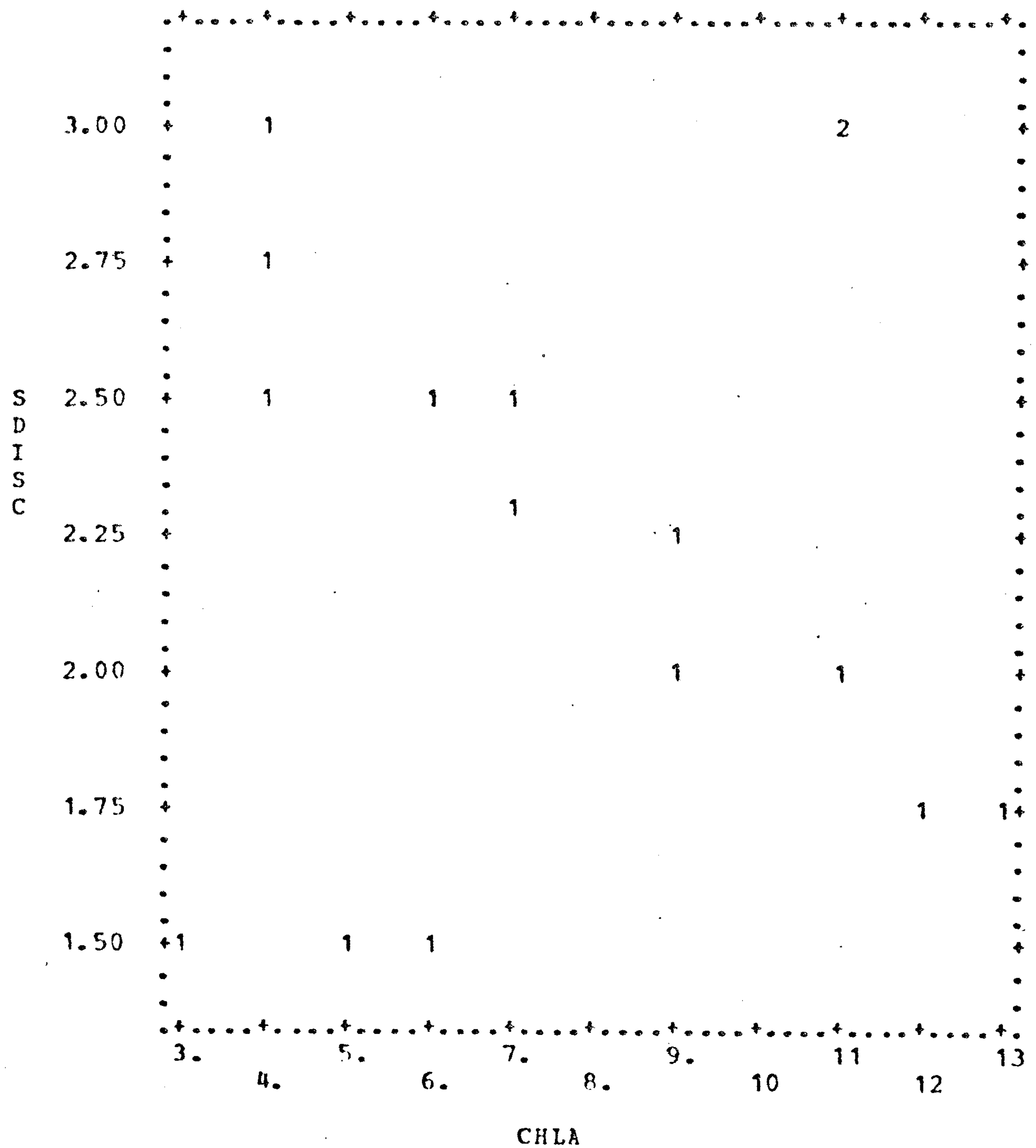


FIGURE F5: Elk Lake Secchi disc depth (ft) versus chlorophyll a (µg/L) 1978.

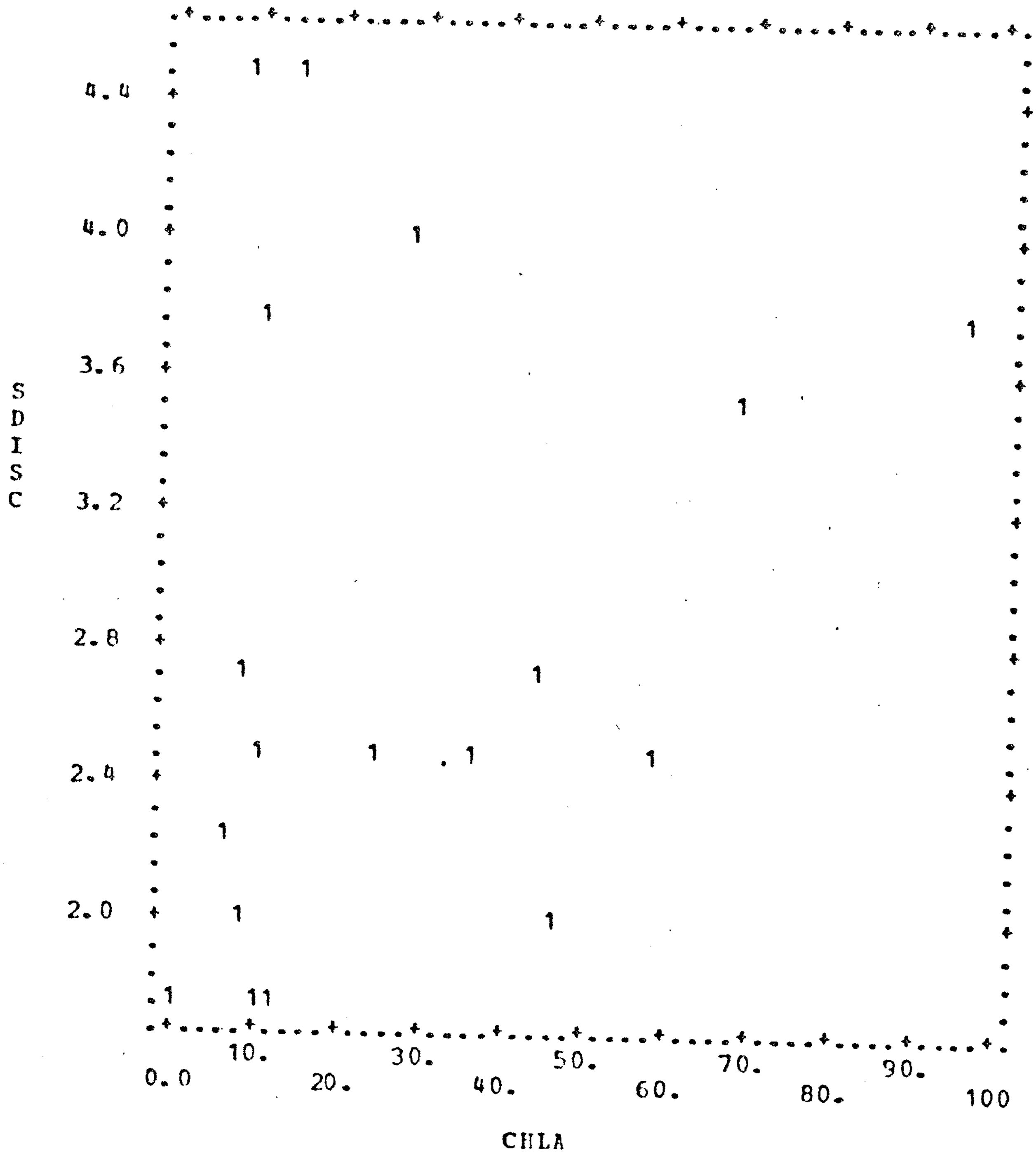


FIGURE F6: Elk Lake Secchi disc depth (ft) versus chlorophyll a (μg/L) 1980.

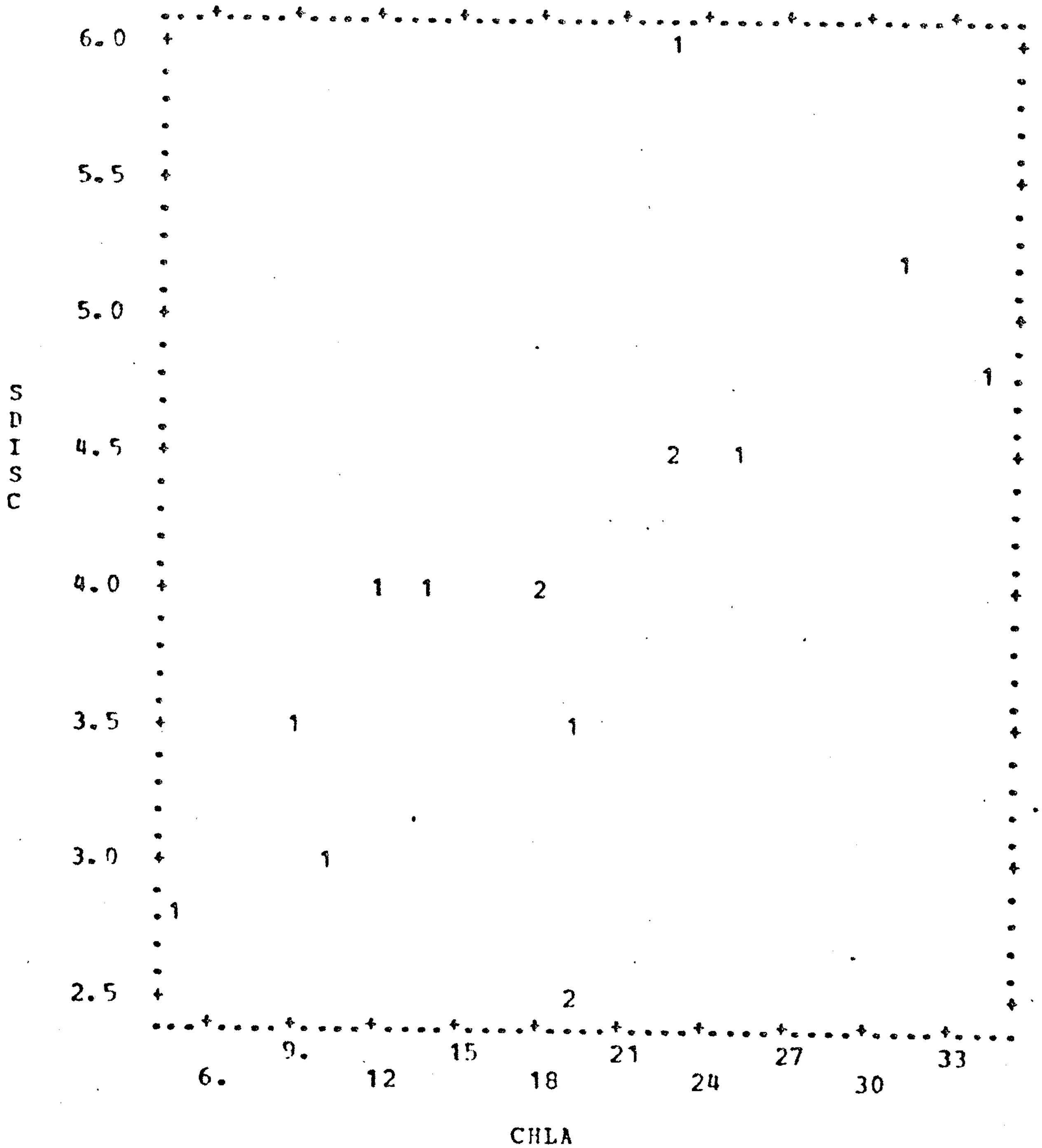


FIGURE F7: Elk Lake Secchi disc depth (ft) versus chlorophyll a (µg/L) 1981.

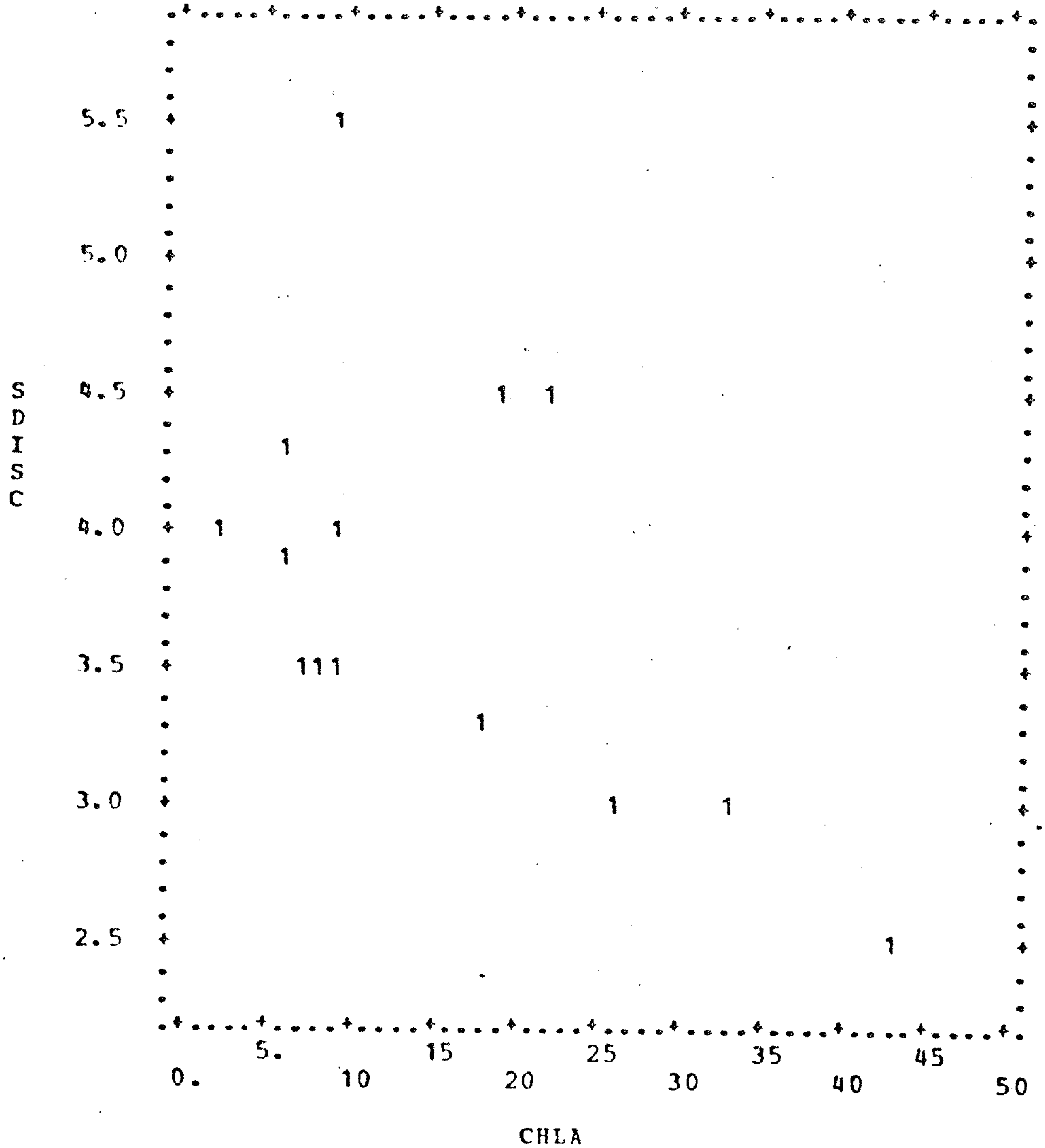


FIGURE F8: Elk Lake Secchi disc depth (ft) versus chlorophyll a (µg/L) 1982.

APPENDIX G

Temporal Variations in Green and Blue-Green Algal Counts

APPENDIX G

Analysis of Algal Identification and Enumeration Data

Figures 7A through 7E show the temporal variation in green algae over the course of the study. The monthly means of the counts of green algae are plotted. Figures 8A through 8E are similar plots for blue-green algae. It is readily apparent from these figures that both green and blue-green algae steadily increased in numbers in both test and reference lakes throughout the study.

Analyses were conducted on logarithm_e of blue greens (ln BG), logarithm_e of greens (ln G), $p = BG/(BG+G)$, and $\ln [p/(1-p)]$. The transformations, where used, were for obtaining better agreement with the distributional assumptions for the various analyses. The use of the logistic transformation, $\ln P/(1-p)$, was of marginal value.

Tables 1 and 2 represent covariance analyses similar to those done for total phosphorus, chlorophyll a, and Secchi disc depth. Observe in Table 1, for all measurements considered, that the apparent tracking of the test lakes by the reference lakes is accounting for a substantial amount of the total variability. Even so, except for the greens, the post-ban effect is statistically significant (although not dramatically so). In Table 2, you will observe that this post-ban effect is primarily associated with Elk Lake, with Butternut also appearing for the blue greens. In both instances, the direction is associated with increases in the blue greens that were greater than corresponding reference lakes.

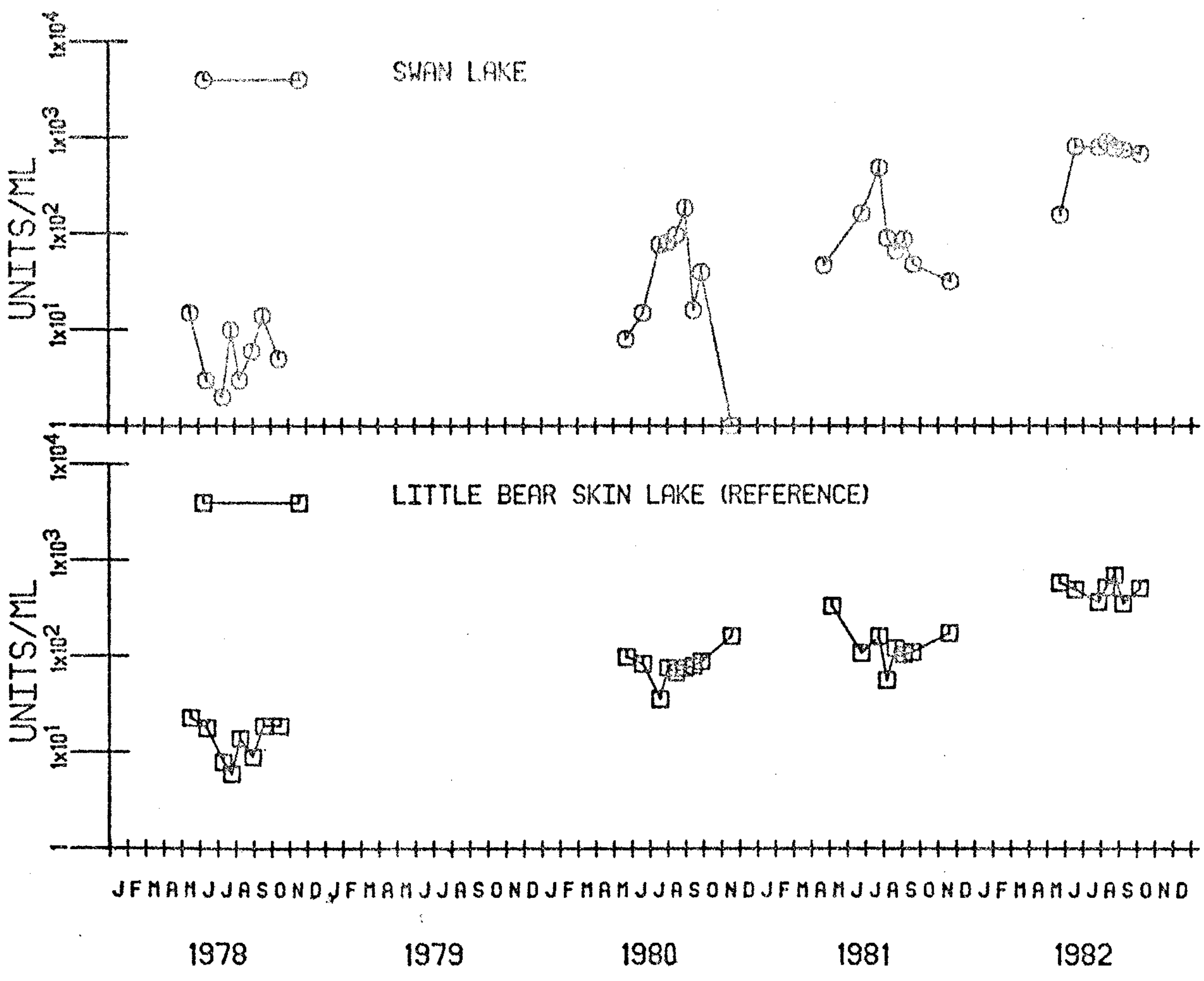


FIGURE G1A TEMPORAL VARIATION OF GREEN ALGAE

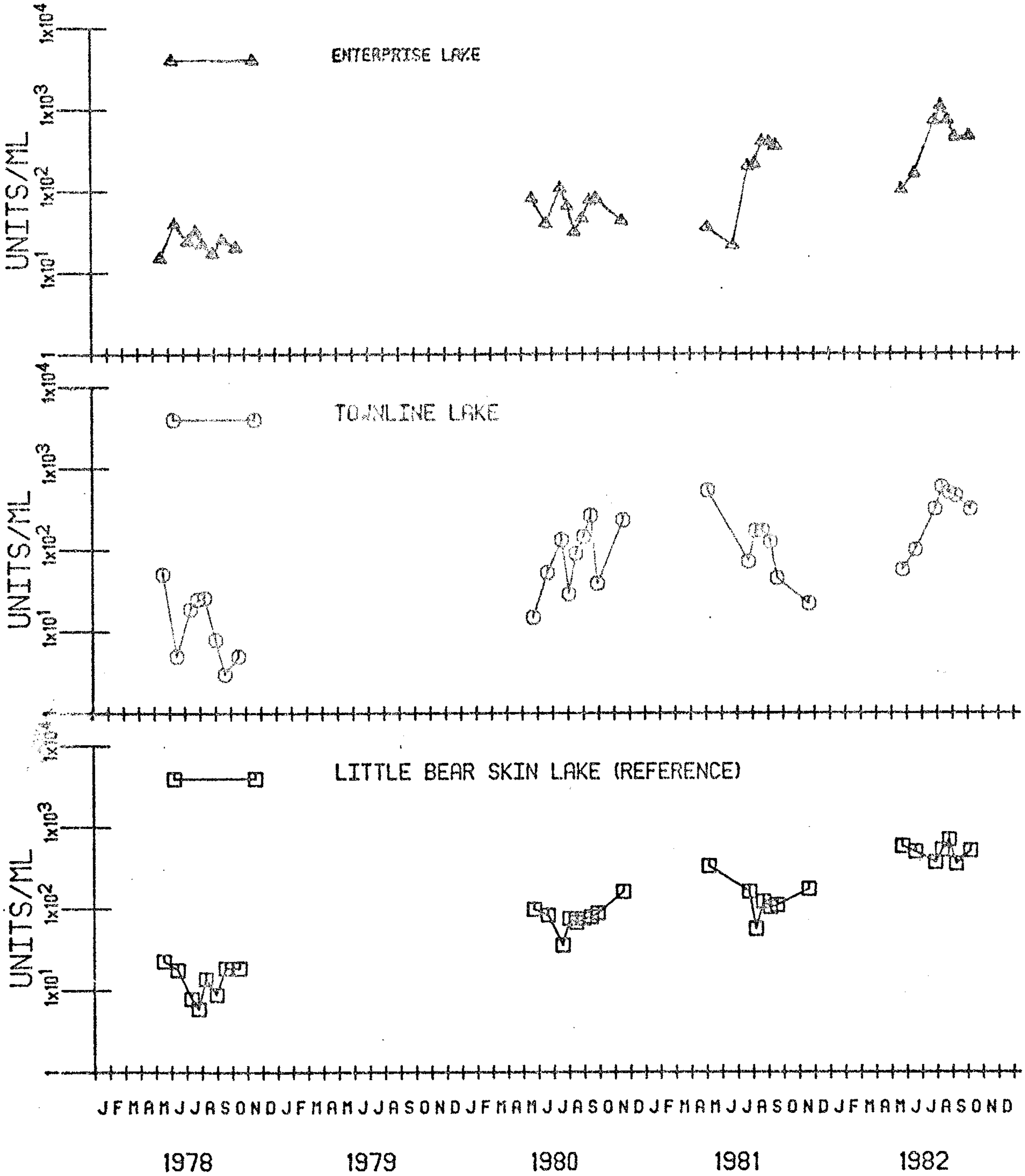


FIGURE G1B TEMPORAL VARIATION OF GREEN ALGAE

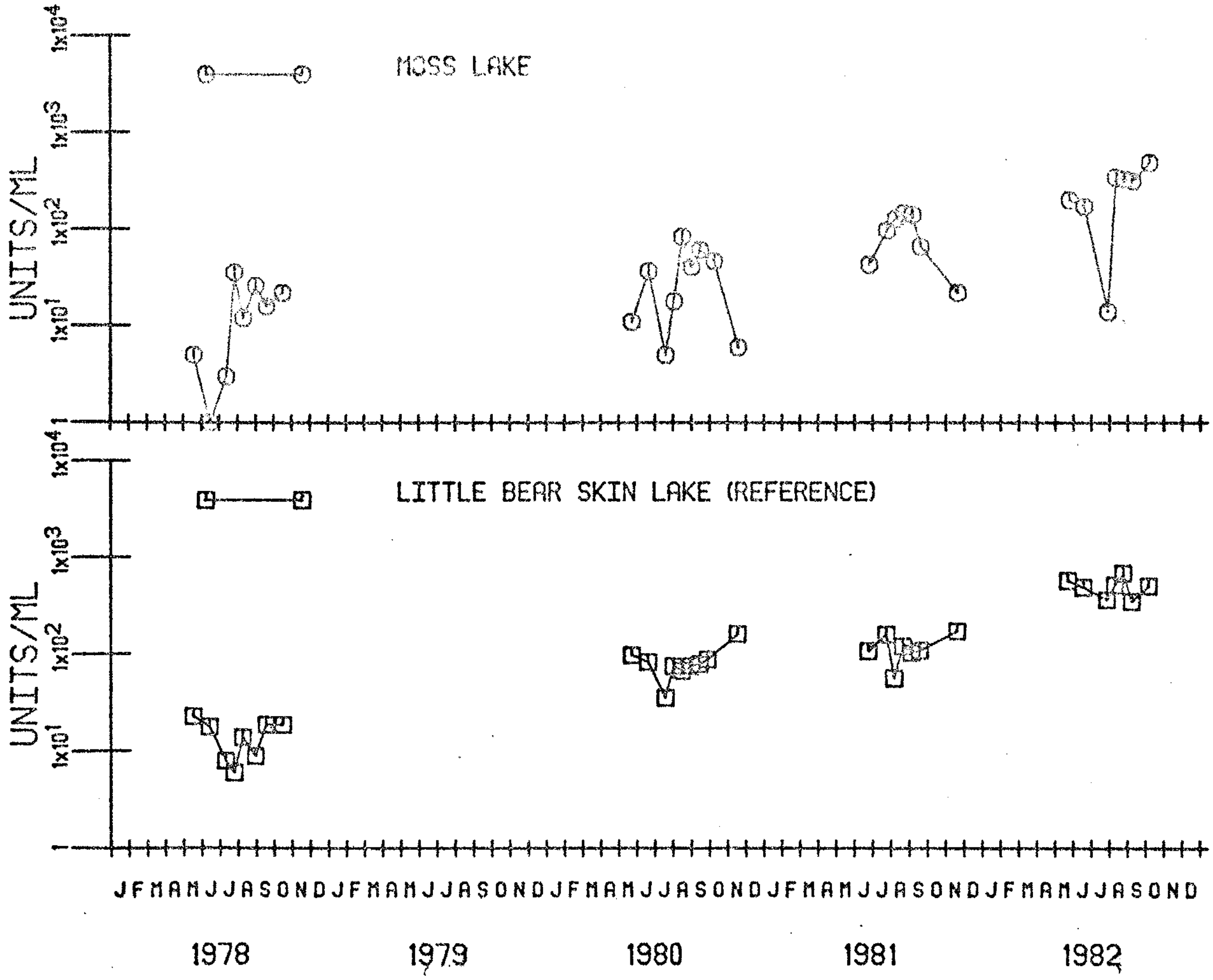


FIGURE G1C TEMPORAL VARIATION OF GREEN ALGAE

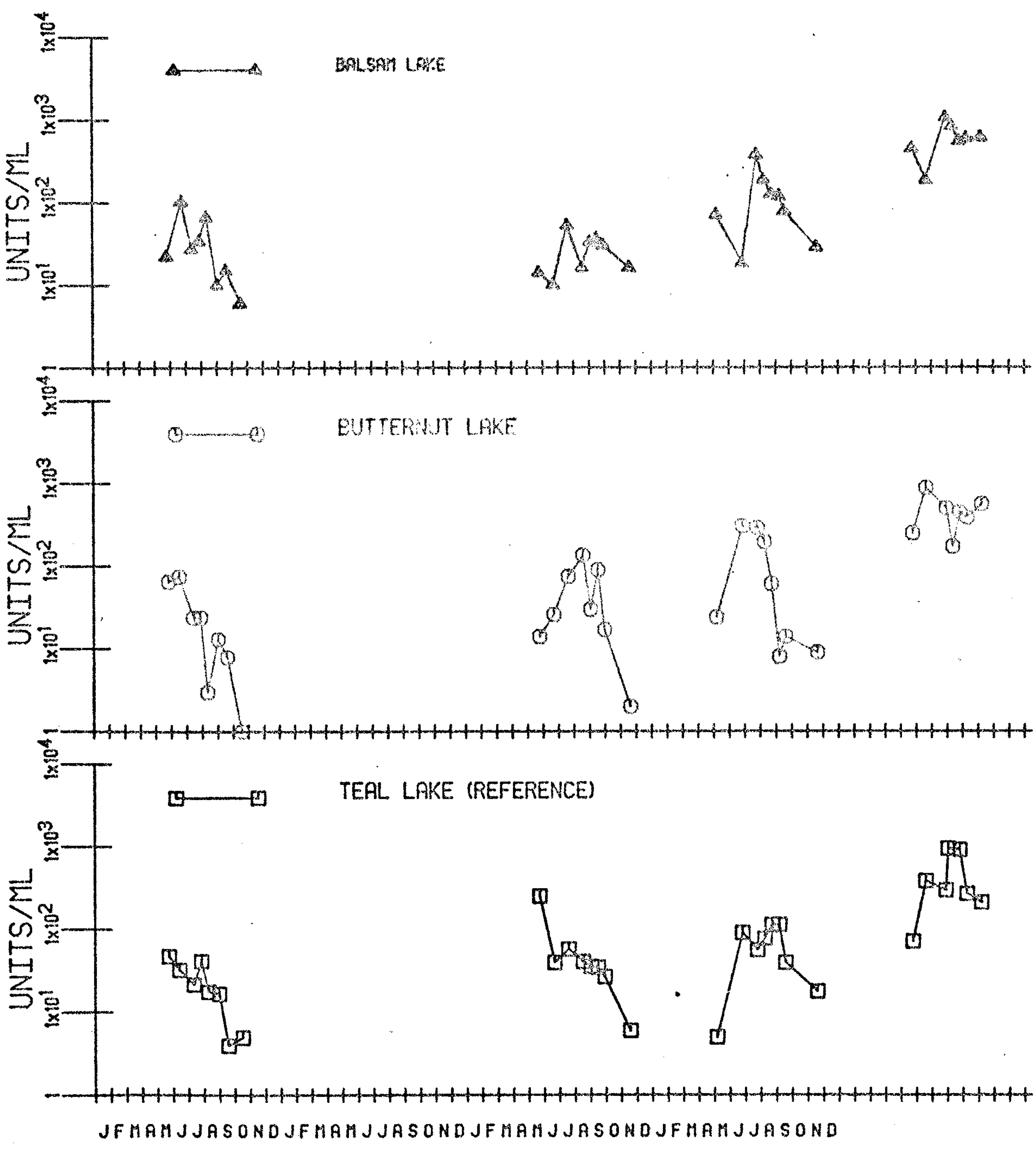


FIGURE G11 TEMPORAL VARIATION OF GREEN ALGAE

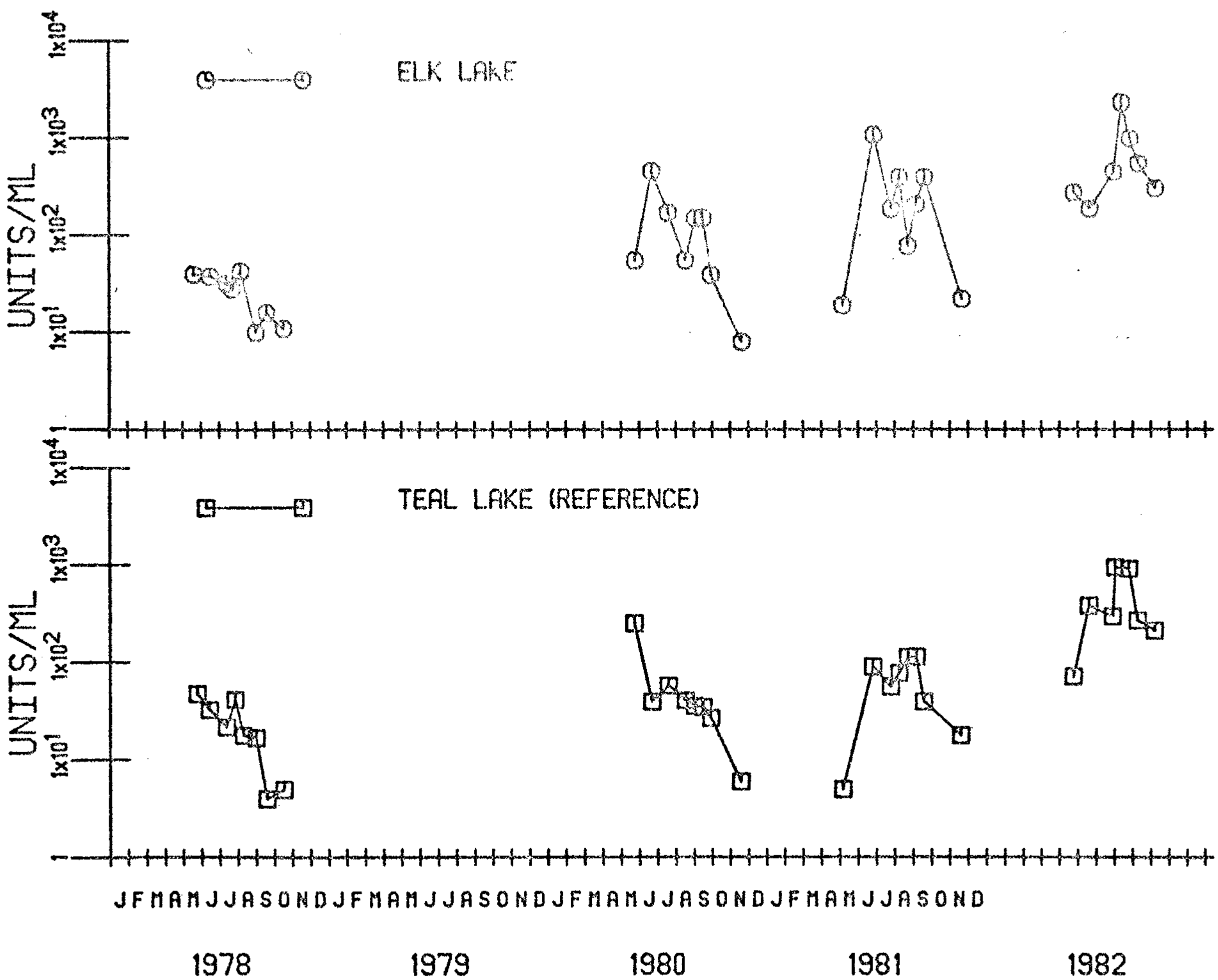


FIGURE G1E TEMPORAL VARIATION OF GREEN ALGAE

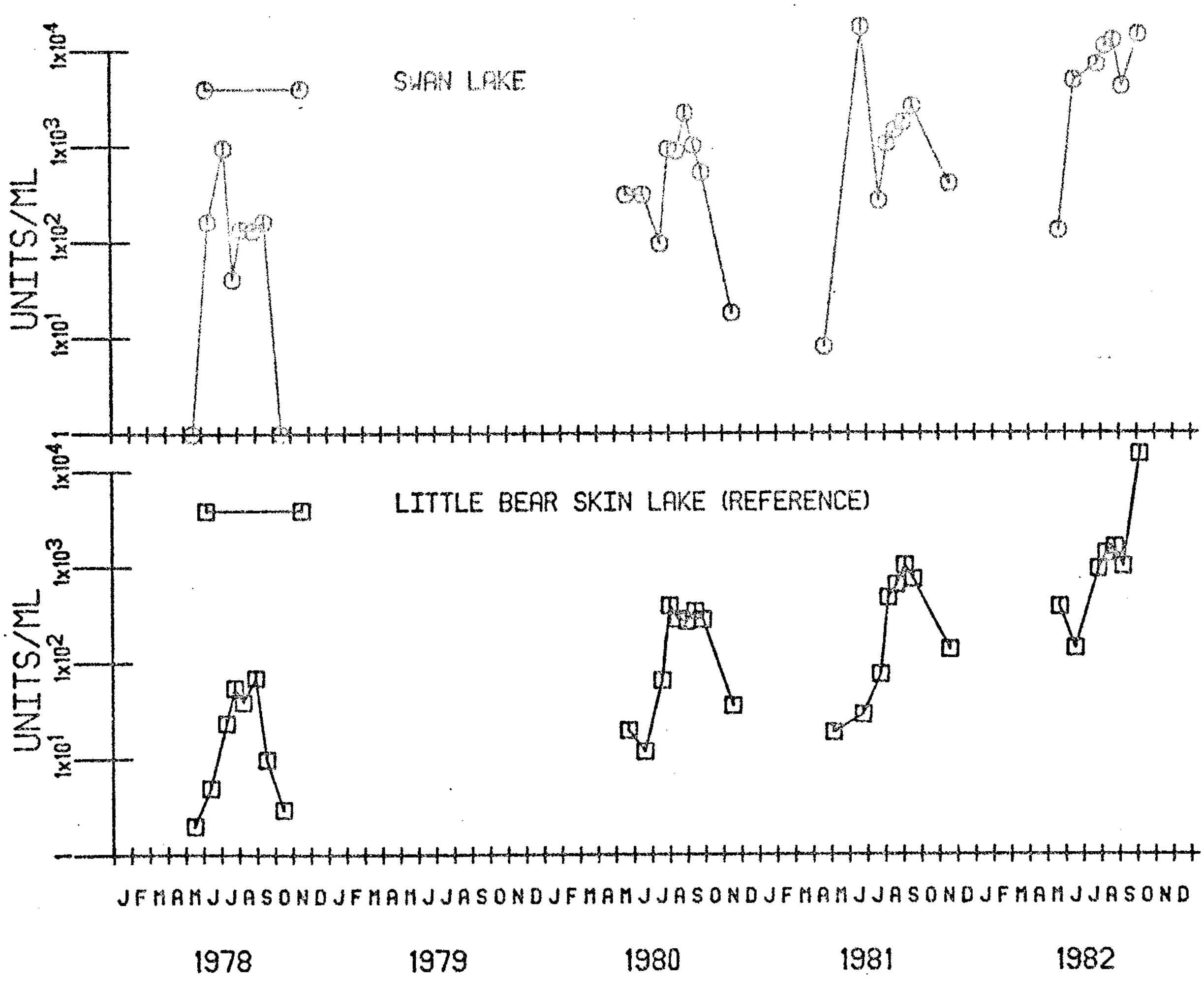


FIGURE G2A TEMPORAL VARIATION OF BLUE-GREEN ALGAE

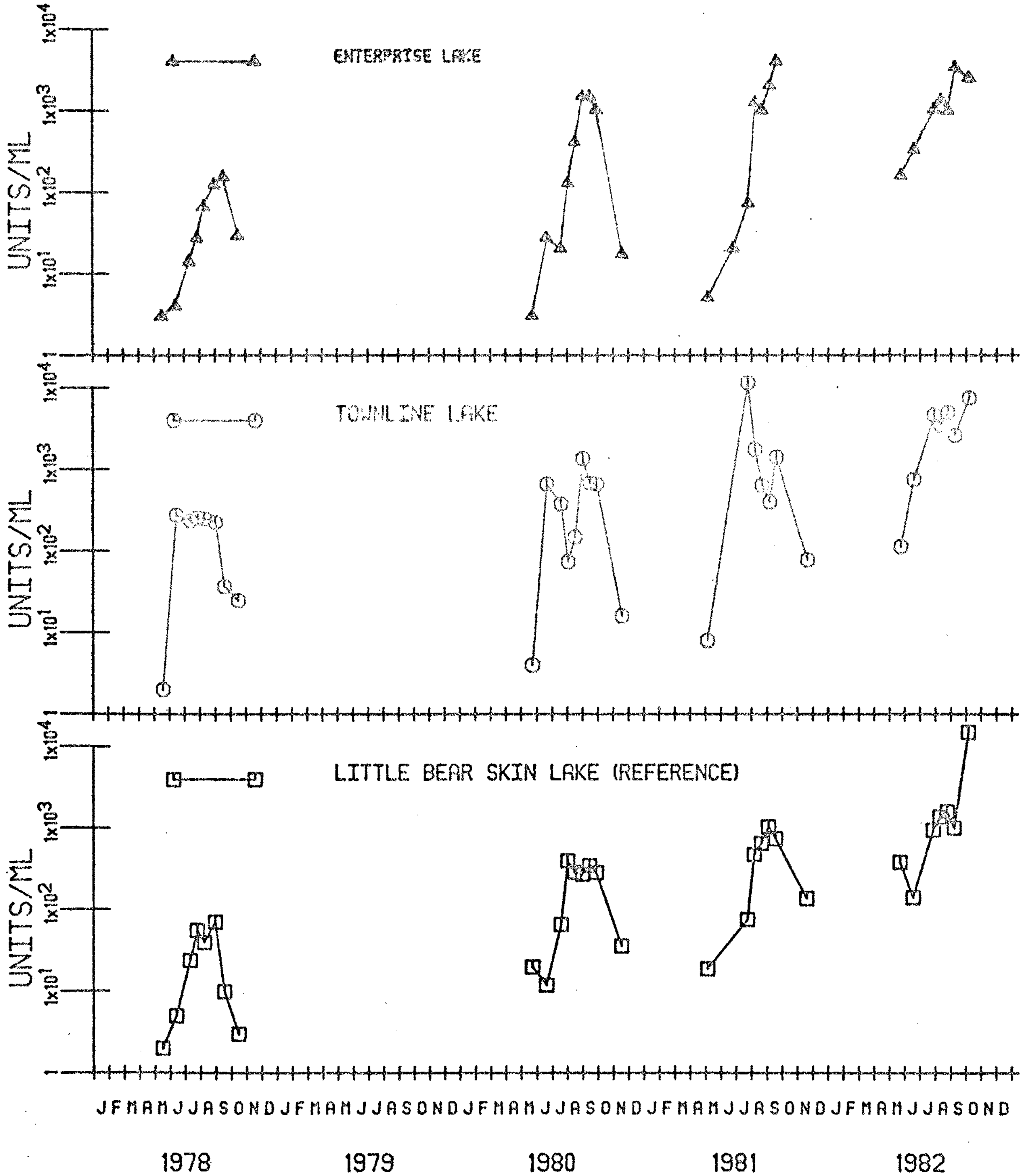


FIGURE 32B TEMPORAL VARIATION OF BLUE-GREEN ALGAE

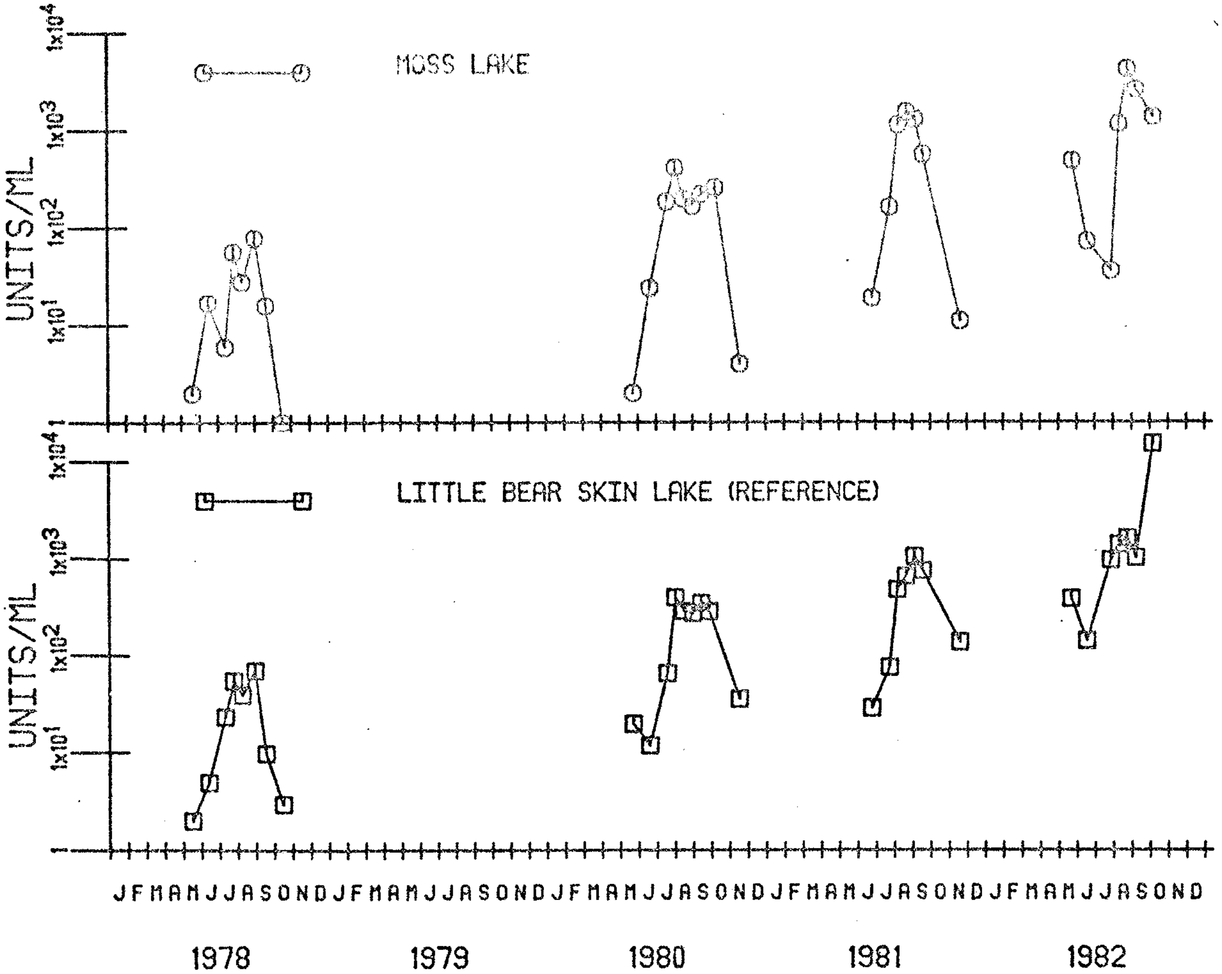


FIGURE G2C TEMPORAL VARIATION OF BLUE-GREEN

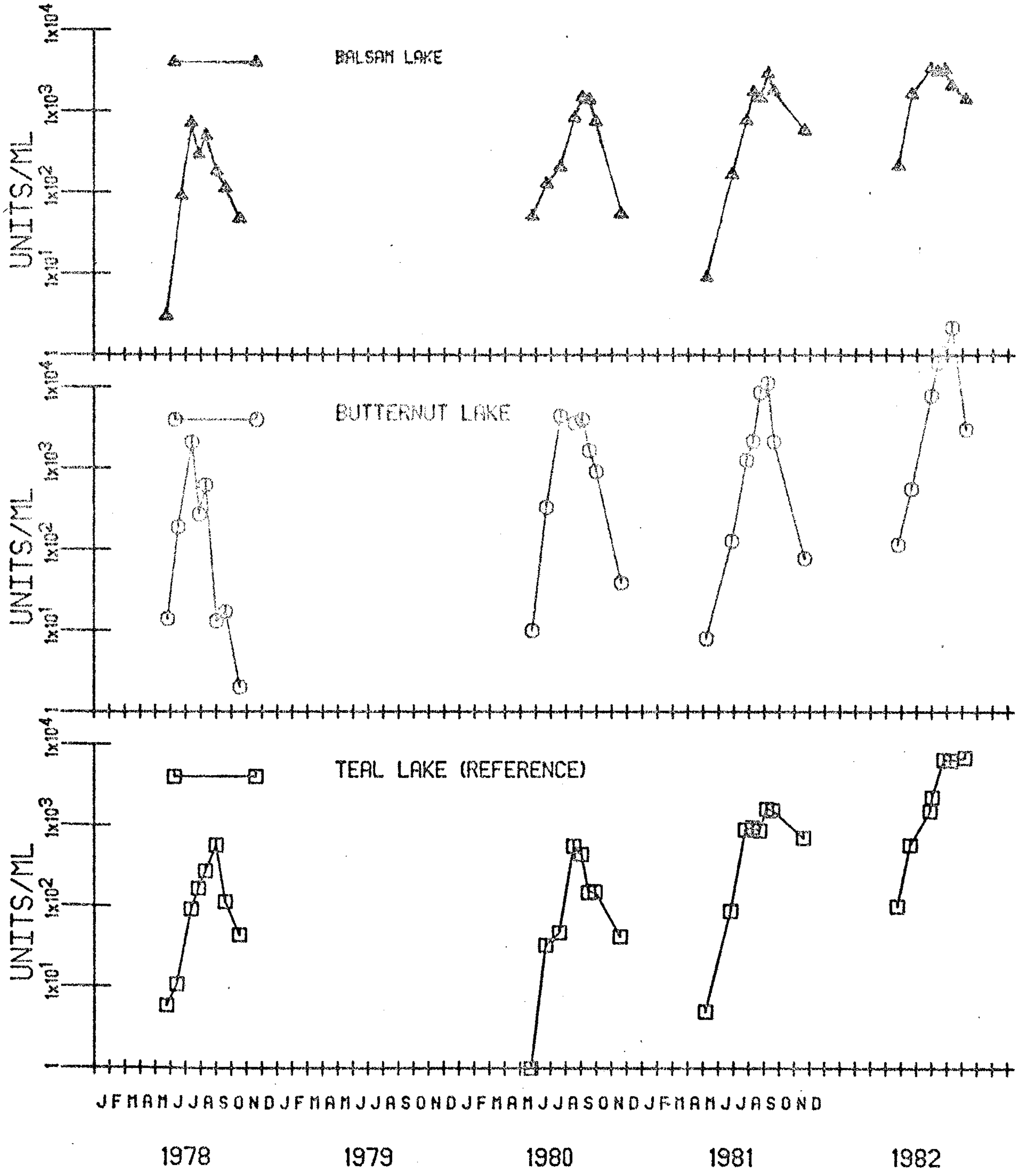


FIGURE G2D TEMPORAL VARIATION OF BLUE-GREEN ALGAE

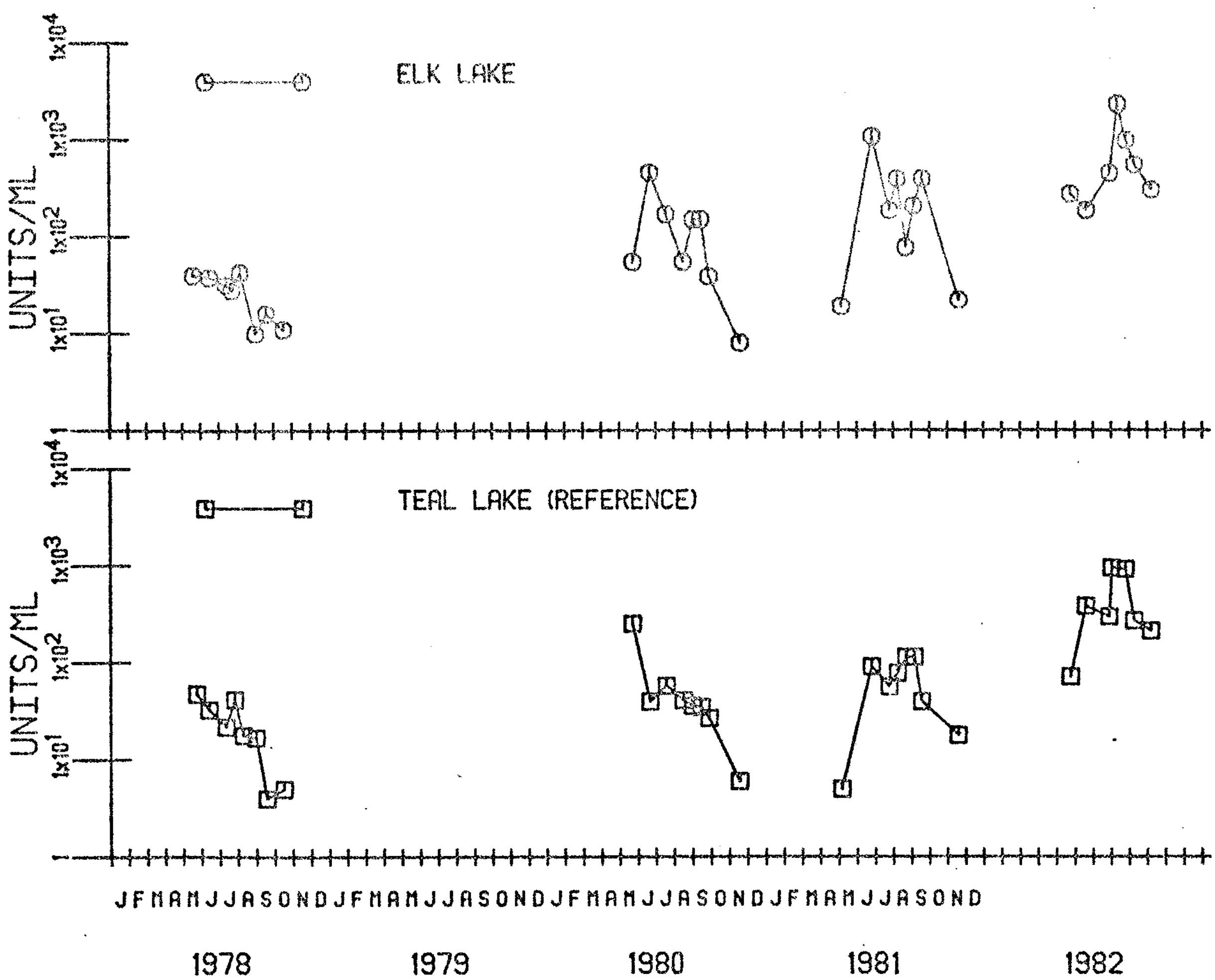


FIGURE G2E TEMPORAL VARIATION OF BLUE-GREEN ALGAE

TABLE G1: Analysis of covariance for a) ln BG; b) ln G; c) p = BG/(BG + G); d) ln[p/(1-p)].

	Source	Sum of Squares	Degrees of Freedom	Mean Square	Test Statistic	R^2	R^2
a)	R. F.	712.35	1	712.35	314.2*	.51	.51
	I.A.	140.53	6	23.42	10.33*	.62	.11
	S.A.	7.75	6	1.29	1	.62	.00
	B.	79.00	7	11.29	5.0*	.68	.06
	Residual	444.30	196	2.27			
b)	R.F.	242.33	1	242.33	158.4*	.40	.40
	I.A.	41.15	6	6.86	4.5*	.47	.07
	S.A.	9.92	6	1.65	1	.49	.02
	B.	7.59	7	1.08	1	.49	.01
	Residual	300.24	196	1.53			
c)	R.F.	4.41	1	4.41	88.2*	.22	.22
	I.A.	4.49	6	.75	15.0*	.45	.23
	S.A.	.13	6	.02	1	.46	.01
	B.	.83	7	.12	2.4*	.50	.04
	Residual	9.86	196	.05			
d)	R.F.	707.94	1	707.94	997.1*	.78	.78
	I.A.	34.11	6	5.69	8.0*	.82	.04
	S.A.	8.58	6	1.43	2.0	.83	.01
	B.	14.41	7	2.06	2.9*	.85	.02
	Residual	139.59	196	.71			

R.F.: Reference Lake
I.A.: Intercept adjustment for individual test lakes
S.A.: Slope adjustment for individual test lakes
B. : Pre/Post ban adjustment for intercept

*Statistically significant at at least 5% level.

TABLE G2: Estimates of pre/post ban effect on intercept and slope(s) for corresponding reference lake for ln BG, ln G, p = BG/(BG + G), and ln [p/(1-p)].

	ln BG		ln G		p		ln [p/(1-p)]	
Lake	Intercept	Slope	Intercept	Slope	Intercept	Slope	Intercept	Slope
Swan	.64	.83*	.26	1.03*	.07	.44*	.15	.98*
Balsam	.74	.65*	.34	.67	.04	.48*	.19	.70*
Butternut	1.61*	.81*	.47	.85*	.09	.60*	.29	.84*
Elk	3.08*	.62*	.91	.71*	.33*	.37*	1.47*	.70*
Enterprise	- .91	1.06*	.18	.57	.05	.39*	0	.94*
Moss	- .58	1.06	.52	.50	.07	.61*	.21	.86*
Townline	-1.23	.98*	.90	.51	-.11	.54*	-.25	1.11*
Approximate standard deviation	.80	.17	.90	.35	.09	.13	.34	.09

*Statistically significantly different from 0 at at least the 5% level.

APPENDIX H

Vollenweider Loading Criteria

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Vollenweider Loading Criteria

The phosphorus model based on mean depth and hydraulic residence time, developed by Vollenweider (1975), was used as a quantitative guideline by which to judge the magnitude of the effect of phosphate detergents on the test lakes. This model compares the actual areal loading of phosphorus (g P/m^2 lake surface/yr) to permissible and critical loading rates calculated from the model. The formula used for determining the critical loading rate was:

$$L_c = 0.02$$

where L_c = critical loading rate for phosphorus (g/m^2/yr)

Z = mean depth of lake (m)

= hydraulic residence time of lake (yr)

V_s = sedimentation velocity (m/yr)

For these lakes, the value of 10 m/yr for V_s was used (Vollenweider, 1975). The permissible criterion is one-half the critical loading criterion.