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ABSTRACT

Gross uptake rates of filterable reactive phosphorus were measured in surface waters of B. Everett Jordan Lake from December 1983 to January 1985. In conjunction, laboratory experiments examining the phosphorus adsorption/desorption potential of suspended sediments and bioavailability of suspended sediment-bound phosphorus were performed on suspended sediments obtained from the Haw River, the major tributary of Jordan Lake, during five periods of high flow from January 1985 to May 1985.

Phosphorus uptake rates were highly variable, ranging from 0.007 to 9.78 $\mu\text{g P l}^{-1} \text{ h}^{-1}$. The highest uptake rates were measured during the warm months in the New Hope Creek basin of the lake when filterable reactive phosphorus concentrations were low and algal biomass high, which when combined with high nitrogen to phosphorus ratios, indicate conditions of potential phosphorus limitation. Lower uptake rates were measured throughout the study in the Haw River basin of the lake and during the winter months in the New Hope Creek basin. Biotic uptake, which comprised most of the uptake during the warm months, was dominated by uptake in the small size fraction ($< 8.0 \mu\text{m}$). Abiotic uptake often comprised the majority of the total uptake in the winter months. Sampling sites near tributary inputs, especially in the Haw River arm, had the highest rates of abiotic uptake due to large inputs of suspended sediments during periods of high stream flow.

Sorption isotherms of suspended sediments obtained from the Haw River exhibited equilibrium phosphorus concentrations significantly below ambient stream filterable reactive phosphorus concentrations, indicating that the sediments were likely acting as a sink for soluble phosphorus present in the stream. Algae grown with suspended sediment-bound phosphorus under conditions otherwise optimal for growth were able to utilize a relatively small amount (mean=11.3%) of the sediment-bound phosphorus. Potential impacts of these results on lake management were discussed.

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INTRODUCTION

Phosphorus is generally considered to be the most important factor in the maintenance of biological cycles in freshwater systems (Schindler 1977). In comparison to other essential nutrients, phosphorus is present in relatively small amounts and usually limits the system's productivity (Wetzel 1983). Increased loadings of phosphorus into lakes and rivers often accelerates the natural evolutionary processes, prematurely creating eutrophic conditions and associated poor water quality. Understanding of the dynamics of phosphorus in these systems is important if we hope to develop effective strategies for their management.

PHOSPHORUS CYCLING IN THE EPILIMNION

Most models of phosphorus-microorganism (phytoplankton, bacteria, zooplankton) interactions confine themselves to epilimnetic waters, where steady-state conditions are assumed to exist, at least for short time periods. In reality, steady-state conditions are rarely found, however, since the cycling of phosphorus within the epilimnion is usually rapid relative to its movement through the epilimnion, the steady-state assumption does not greatly hinder interpretation (Rigler 1973).

Rigler (1956) initially proposed a simple two-compartment model of epilimnetic phosphorus cycling,

involving exchange between soluble and particulate phosphorus. Observation of uptake kinetics often not consistent with the two-compartment model lead to the development of a four-compartment model by Lean (1973), consisting of particulate phosphorus, orthophosphate, a low molecular weight organic phosphorus compound (XP), and a high molecular weight colloidal phosphorus compound. He proposed that the majority of the exchange occurs directly between particulate phosphorus and orthophosphate, but that a compound termed XP is also excreted which rapidly complexes with the large colloidal phosphorus compound, releasing small amounts of orthophosphate. During periods of phosphate deficiency phytoplankton and bacteria can develop phosphatase enzymes that allow for the utilization of dissolved organic phosphorus compounds which are analogous to the XP fraction (Kuenzler and Perras 1965; Paerl and Downes 1978).

Physiological adaptations of microorganisms to low ambient concentrations of phosphorus in the water column add complexity to the phosphorus cycle. Many phytoplankton have the ability to take up phosphorus in excess of their metabolic needs and store it within the cell as polyphosphate (Perry 1976). Low concentrations of phosphorus in the water may lead to assumptions of phosphorus limitation when, in reality, the algae may have sufficient stores of polyphosphate for significant additional growth. Lean et al. (1983) suggested that phytoplankton obtained from Lake Erie had enough stored

polyphosphates to sustain them for at least one month despite ambient nutrient concentrations that clearly indicated phosphorus limitation.

Fractionation of biotic phosphorus uptake in freshwater lakes generally indicate that the bacterial fraction, which, may also include nannoplankton if separation of biotic uptake is done by size fractionation (Lean 1984), dominates uptake (Rigler 1956; Berman and Stiller 1977; Kuenzler and Greer 1980; Currie and Kalff 1984a). Laboratory work by Rhee (1972) and Currie and Kalff (1984b) support bacterial superiority in competition for phosphorus, but Fuhs *et al.* (1972) found evidence indicating phytoplankton should outcompete bacteria at phosphorus concentrations typically found in nature. Field studies by Faust and Correll (1976), Kuenzler and Greer (1980) and Lean (1984) found bacterial domination of uptake for the majority of the year, although during periods of high algal biomass the relative fraction of algal uptake significantly increased.

Phosphorus regeneration by zooplankton can be an important mechanism in the phosphorus cycle, especially when ambient phosphate concentrations are low (Lehman 1980; Rigler 1973). Rigler (1973) estimated that direct release of phosphorus from ultraplankton and zooplankton excretion were equally important.

PHOSPHORUS UPTAKE KINETICS

The application of static measurements of phosphorus concentrations to the highly dynamic phosphorus cycle

contained within the epilimnion, capable of utilizing a phosphorus atom many times before it is leaves the system, is relatively limited. Periods of maximum biotic activity often correspond with periods of low phosphorus concentrations that should, if recycling is ignored, limit further biological activity. However, the continuation of biotic activity during these periods emphasizes the importance of knowledge of the flux of phosphorus through the system. Interpretations of static measurements are also hindered by the use of operationally defined phosphorus fractions that are often not accurate measurements of the forms of phosphorus that are actively being cycled (Kuenzler and Ketchum 1962; Rigler 1966). Consequently, studies emphasizing flux rates and pathways of phosphorus, in addition to measurements of concentrations, have evolved (Hutchinson and Bowen 1947; Rigler 1956).

Rates of phosphorus uptake are most often obtained using the techniques of Rigler (1956), which measure the rate of disappearance of radioactive phosphate (^{32}P) added to water samples in amounts small enough not to affect the assumption of steady-state. This technique measures gross rates of phosphorus uptake. Net uptake should be zero if steady-state conditions exist.

The rate of phosphorus uptake by micro-organisms is affected by many factors including: nutritional status (Fuhs 1969; Rhee 1972); presence or absence of light (Chis olm and Stross 1976; Kuenzler *et al.* 1979) and light intensity

(Nalewajko *et al.* 1981); pH (Healey 1973); cell size (Fuhs *et al.* 1972); temperature (Kuenzler *et al.* 1979); diurnal cycle (Richey *et al.* 1975; Currie and Kelff 1984b); and phosphate concentration (Fuhs *et al.* 1972; Halmann and Stiller 1974; Kuenzler *et al.* 1979). Nalewajko and Lean (1980) provide an extensive review of factors affecting phosphorus uptake kinetics and algal growth.

Distinct seasonal variation of phosphorus uptake has often been observed. During the winter months when algal and bacterial biomass and metabolic activity decrease, and phosphorus concentrations increase, phosphorus uptake rates are usually low (Halmann and Stiller 1974; Faust and Correll 1976; Kuenzler and Greer 1980). In summer months, rapid turnover times (1-8 minutes) are encountered in most lakes, apparently irrespective of trophic state (Rigler 1973). Algal blooms in the summer (Faust and Correll 1976) and the winter (Kuenzler *et al.* 1979) have been shown to greatly increase uptake rates.

SUSPENDED SEDIMENT-PHOSPHORUS DYNAMICS

The input of suspended sediments often has been implicated as being important to the phosphorus cycle of lakes (Hutchinson 1941; Kuenzler and Greer 1980; Jones and Redfield 1984), especially when the sediments are fine clays which generally have high phosphorus binding capacities and long retention times in the water column (Golterman 1973; Syers *et al.* 1973). Piedmont rivers and lakes are subject to large influxes of sediments from the surrounding watersheds

during heavy precipitation events. The soils of these watersheds consist primarily of fine clays (Daniels et al. 1984) which can remain suspended in the water column for long periods of time (Williams et al. 1980). However, the relatively small amount of research which has been conducted on the biological and chemical effects of the addition of sediment into lakes and rivers has resulted in low levels of understanding and consequently, hindered water resource management (Ongley et al. 1982).

The phosphorus desorption/adsorption capacity of suspended sediments, hereafter referred to simply as sorption capacity, must be understood to achieve an adequate understanding of the phosphorus cycle (Syers et al. 1973). Numerous studies have demonstrated the importance of sorption reactions between phosphorus and bottom sediments (Olsen 1964, Pomeroy et al. 1965, Harter 1968, Meyer 1979, Klotz 1985) and, while much fewer in number, the importance of suspended sediment-phosphorus exchange reactions in rivers (Taylor and Kunishi 1971; Green et al. 1978), in lakes (Kuenzler and Greer 1980), and in estuaries (Carritt and Goodgal 1954).

While most studies have found that the sediments act as a sink for phosphate (Syers et al. 1973), the high complexity of the sorption reaction combined with the large variability found within and between aquatic systems often make generalizations concerning the sediment's role difficult. The sorption mechanism is a function primarily of the solution phosphate concentration and the pH (Chen et

al. 1973a), but also of ionic strength, substances competing for the same reactive sites, temperature, organic substances, retention time, and particle size (Beek and vanRiemsdijk 1982).

The factors controlling the chemical reactions of phosphate ions with clay mineral surfaces are not well understood, especially with naturally occurring clays (Beek and vanRiemsdijk 1982). Reactions are clearly associated with the presence of aluminum hydroxides and iron oxides near the clay surface (Chen et al. 1973a; Edzwald et al. 1976), and are not necessarily based upon simple exchanges of the phosphate ion for the hydroxide ion bound to the aluminum and iron complexes of the clay surface, as has often been suggested (Golterman 1973). Phosphate adsorption isotherms, which relate the amount of phosphate adsorbed per gram dry matter to the concentration of phosphate in the water (Olsen 1964), are often constructed in attempts to evaluate the sorption mechanisms. These have shown the uptake of phosphate by the clay particles to be biphasic, consisting of a rapid initial adsorption of the phosphate onto the reactive sites of the clay followed by a much slower phase involving the formation of new solid phases (Carritt and Goodgal 1954; Pomeroy et al. 1965; Chen et al. 1973b; Edzwald et al. 1976). The initial adsorption step is usually complete within 12 hours (Edzwald et al. 1976; Huang et al. 1976; Green et al. 1978) and is readily desorbable (Carritt and Goodgal 1954; Ryden and Syers 1977). The

slower step can extend to over 60 days and is not easily reversed since it involves the formation of a solid phase (Chen et al. 1973b).

BIOAVAILABILITY OF SEDIMENT-BOUND PHOSPHORUS

Quantification of the amount of suspended sediment-bound phosphorus that is available for algal utilization may help explain the role of these sediments in the phosphorus cycle of lakes. Estimates of the relative availability of phosphorus associated with these sediments as they enter a water body are very important if we wish to determine the most cost effective control strategies for regulating phosphorus inputs (Sonzogni et al. 1982). For example, an extensive study on phosphorus management strategies for the Great Lakes (PLUARG 1978), which are subject to high loading of sediment-bound phosphorus, examined the bioavailability of various sources of inputs. While diffuse tributary inputs of phosphorus were found to comprise up to 53 percent of the total phosphorus entering the lake, they were found to be relatively unavailable (<40%). This heavily contributed to the decision that the most cost effective management strategy would be removal of municipal point source phosphorus. The sediment entering the lakes from eroding bluffs surrounding the lake was found to be essentially unavailable and considered unimportant for phosphorus management (Williams et al. 1980).

Estimates of the bioavailability of particulate phosphorus, hereafter referred to as algal available

phosphorus (AAP), are also important in attempts to refine empirical input/output models. These models attempt to predict algal biomass based on, among other factors, total phosphorus loadings, (Vollenweider 1968; Dillon and Rigler 1974). Nicholls and Dillon (1978) claim that the assumption behind the use of total phosphorus is that it is all potentially available for algal utilization. The limited success of these models is possibly related to a rough relationship between total phosphorus and AAP (Sonzogni et al. 1982). Increased incorporation of bioavailability values into these models, as was done by Schaffner and Oglesby (1978), should decrease some of the uncertainty associated with their predictions.

Laboratory experiments attempting to quantify the amount of sediment phosphorus available to algae, have primarily used modifications of standard algal bioassay methods. Basically, algae are grown with sediments as the sole source of phosphorus while all other conditions affecting growth are kept at optimum. Consequently, the bioavailability values obtained are usually considered to represent the maximum bioavailability obtainable under laboratory conditions (Lee et al. 1980).

The availability of phosphorus which ultimately results under natural conditions is a function of a multitude of factors including; the forms and amounts of the phosphorus in the particulate fraction, the residence time of the particle in the lake water, the abundance, species composition, and nutrient status of the algal population,

the solution phosphate concentration, and other factors controlling particulate phosphorus solubility such as pH and Eh (Armstrong et al. 1979). Other factors such as sediment resuspension (Williams et al. 1980) and retention time in the photic and/or mixing zone may also be important.

Measurements of AAP have shown variation between and within various sources. This is partially due to the large number of factors affecting the actual availability but also is related to variations of the methods used (Lee et al. 1980). Estimates of AAP have varied from 0 to 55 percent with an estimated mean AAP of 25 to 30 percent (Golterman et al. 1969; Fitzgerald 1970; Chiou and Boyd 1974; Golterman 1977; Cowen and Lee 1980; Williams et al. 1980; Klapwijk et al. 1982).

Inorganic sediment phosphorus consists of an apatite fraction and/or a non-apatite inorganic phosphorus (NAIP) fraction, which is mainly composed of phosphate adsorbed onto oxides of aluminum and iron, but also may include actual iron and aluminum minerals and non-apatite calcium phosphate (Sonzogni et al. 1982). Recently, attempts have been made to correlate AAP determined by bioassay to the presence of these fractions. Modifications of extraction techniques initially developed by Chang and Jackson (1956), which were developed in an attempt to relate the phosphorus fraction extracted to plant growth, have been used. The apatite fraction has often been found to be essentially unavailable due to its low solubility (Williams et al.

1980), however, some studies indicate small amounts may be available (Premazzi and Zanon 1984). In some experiments, an extraction using NaOH, which removes the majority of NAIP, has been shown to predict the AAP accurately, implicating the high availability of the NAIP (Williams et al. 1980; Young and DePinto 1982). However, Klapwick (1982) found that the NaOH extraction severely overestimated (2-10x) the AAP while Dorich et al. (1982) found the NaOH fraction comprised only 36 percent of the AAP.

METHODS

LAKE SAMPLING AND MEASUREMENTS

Sampling trips to Jordan Lake were made approximately monthly; from December 1983 at Station 10 (Figure 1) and Station 30, and February 1984 at Station 5, until January 1985. Additional intensive samplings were performed at Station 10 in March and May 1985. Most samples were taken between 0800 and 1400 hours. Water was collected from 0.5 m, assumed to be representative of epilimnetic water, using a nonmetallic pump (Jabsco Inc.) equipped with a plastic intake hose covered with 360 μ m netting. Water was collected from five depths throughout the water column during the intensive Station 10 samplings. Water was dispensed into large polyethylene carboys and shaded until use.

At each station; air temperature, weather conditions, and time of sampling were recorded. Water depth was measured using a Lowrance depth meter. In situ water column measurements of temperature, conductivity, and dissolved oxygen were made using a Hydrolab Surveyor Model 6D. Underwater light intensity of photosynthetically active radiation (400-700 nm wavelengths) was measured at several depths using a submarine photometer (Lambda Instruments Co.) from which the light extinction coefficient, $-K_d$, and the 1 percent light level were calculated. Secchi depth was also recorded at each station.

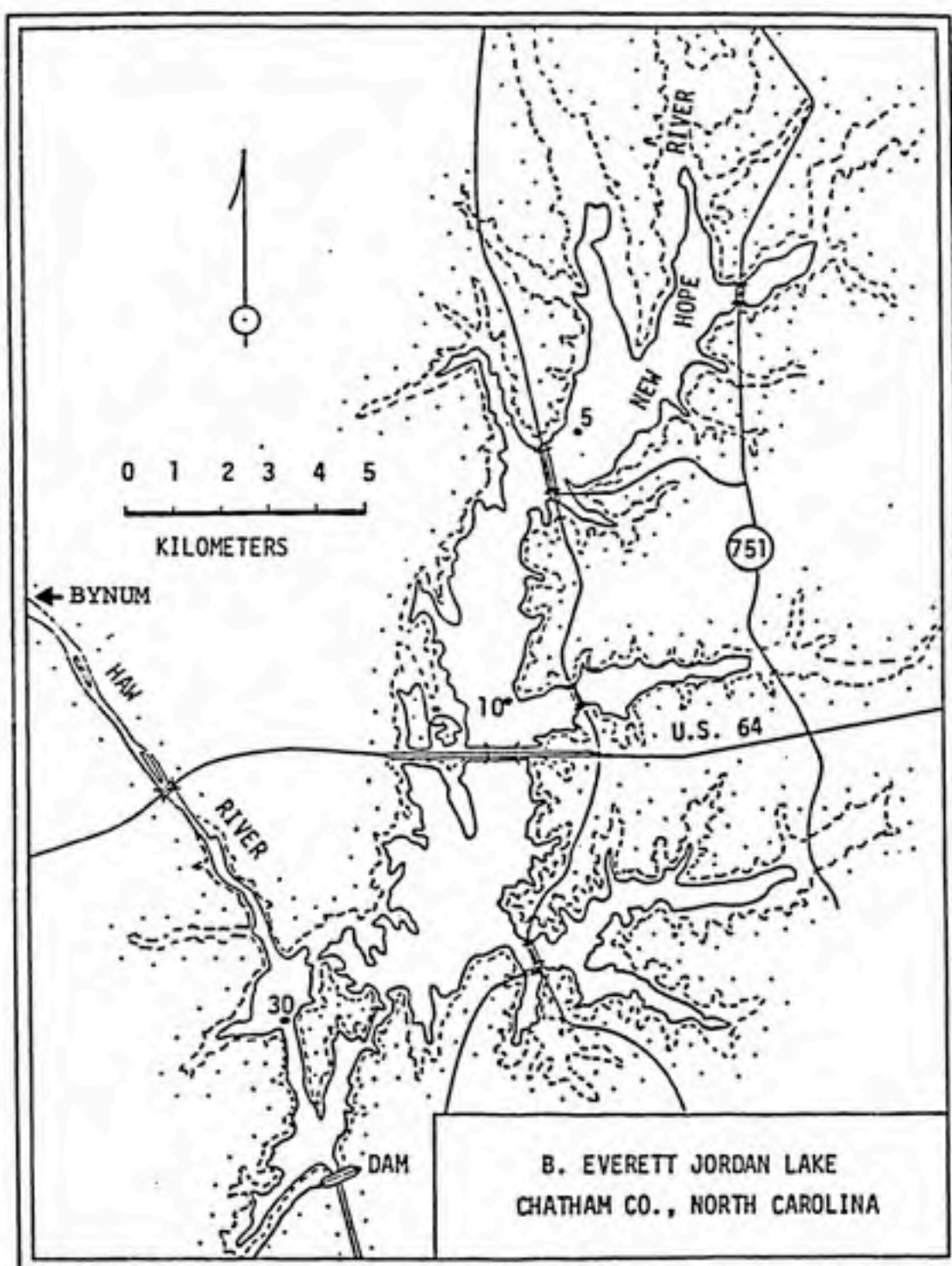


Fig. 1. Map of B. Everett Jordan Lake, North Carolina, showing location of sampling stations. Map shows lake when filled to top of conservation pool (Elev. 216 ft.); additional contour line around lake shows area when lake is filled to top of flood control pool (Elev. 240 ft.).

Filtrations for nutrient and pigment analyses (chlorophyll-a and phaeophytin) were done on the boat using acid-rinsed glass fiber filters (Whatman GF/F for phosphorus analyses, GF/C for nitrogen and pigment analyses). Filtered and unfiltered water samples were dispensed into acid-washed polyethylene bottles and stored on ice in the dark until return to the laboratory where they were refrigerated until analyses. Filters for pigment analyses were stored in a dessicator on ice until they could be frozen in the laboratory. Pigments were spectrophotometrically determined using an alkaline acetone extraction (Wetzel and Likens 1979).

PHOSPHATE UPTAKE

Gross phosphate uptake rates were measured aboard the boat via the use of carrier-free ^{32}P -labeled orthophosphate tracer (New England Nuclear). Uptake rates were measured using water samples from 0.5 m, however, during intensive samplings at station 10, in situ measurements of uptake rates throughout the entire water column were performed. Samples of lake water (100 ml) were dispensed into Pyrex bottles to which one drop (1 μCi) of tracer was added. Samples were then incubated in the dark at the approximate lake water temperature. Subsamples (5 ml) were removed after periods of about 5, 10, 15, 30, etc, minutes and gently filtered through 0.45 μm Gelman and 8.0 μm Nucleopore membrane filters to separate particulate matter from solution. Subsampling continued for at least 3 hours,

longer when uptake was slow. Both filters and filtrate from the 0.45 μ m filters were placed in plastic scintillation vials. Upon return to the laboratory, 5 ml of distilled-deionized water was added to the vials containing the filters and the radioactivity of all samples was counted using Cerenkov radiation (Heberer 1965) on a Packard Tri-carb Model 3320 liquid scintillation counter.

The rate coefficient for phosphate uptake, K , is described by the equation:

$$-K = \ln (A_t/A_0) / t$$

where A_0 = initial activity of filterable ^{32}P (determined from the sum of the filter and filtrate activity from the first subsampling), A_t = activity at time t , and t = elapsed time. This equation assumes that the removal of phosphate from the water is a first-order reaction and that a steady state exists between soluble and particulate phosphorus. The rate coefficient, K (h^{-1}), indicates the relative fraction of phosphate that is being transferred to particulate matter per unit time. K was estimated by calculating the slope of the regression line of the natural logarithm of filtrate activity vs time. Often K was not linear throughout the entire experiment and an asymptote was reached. This necessitated the use of the equation:

$$-K = \ln ((A_t - A_\infty)/(A_0 - A_\infty)) / t$$

where A_∞ is the activity present at the asymptote (Riggs 1963 in Stone 1982). From the uptake coefficient, the

phosphate turnover time ($1/-K$) can be generated. The turnover time is the time required for the amount of phosphate in the water to be completely replaced, assuming steady state conditions exist.

The gross phosphate uptake rate, v ($\mu\text{g P l}^{-1} \text{ h}^{-1}$), was calculated by multiplying the uptake rate coefficient by the ambient filterable reactive phosphorus (FRP) concentration:

$$v = -K \cdot (\text{FRP})$$

The phosphate uptake attributable to abiotic uptake was measured by poisoning water samples with formaldehyde (2.5 ml of 37 % formaldehyde into 100 ml sample). After 20-25 minutes, tracer was added and uptake measured as described above. Abiotic uptake was subtracted from uptake in untreated samples to estimate biotic uptake. Different pore-size filters (0.45 and 8.0 μm) were used to attempt to differentiate between "large" particle (algal) uptake, and "small" particle (bacterial) uptake. At Station 10, the biotic uptake was further fractionated into eucaryotic (algal) uptake and procaryotic (blue-green algal and bacterial) uptake by treating the samples with antibiotics (Gentamycin/Kanamycin at $300 \mu\text{g} \cdot \text{ml}^{-1}$). Eucaryotic uptake was calculated by subtracting the uptake in the killed sample from the uptake in the antibiotically treated sample; procaryotic uptake was calculated by subtracting uptake in the antibiotically treated sample from uptake in the ambient sample.

Measurement of phosphate uptake rates as a function of

phosphate concentration were performed twice at Station 10. The resulting values were analyzed using the Michaelis-Menton equation:

$$v = \frac{V_{max} \cdot S}{K_B + S}$$

where v = the measured uptake rate, V_{max} = the maximum uptake rate, and S = the phosphate (substrate) concentration, from which K_B , the half-saturation coefficient or the phosphate concentration where $v = 1/2 V_{max}$, was calculated.

Net phosphate uptake rates were occasionally obtained by measuring the change in FRP concentration over time. Lake water samples were incubated under conditions identical to those of the tracer experiment, including equal time, except no tracer was added. The FRP at the beginning and at the end of the incubation was measured; the difference between them corresponding to the net uptake or efflux by seston.

NUTRIENT ANALYSES-LAKE WATER

Measurements of total phosphorus (TP), total filterable phosphorus (TFP), and filterable reactive phosphorus (FRP), were spectrophotometrically determined using procedures based on the phosphomolybdate blue technique of Murphy and Riley (1962). FRP was determined within 24 hours using the method of Strickland and Parsons (1965). Turbidity corrections were often necessary when high suspended clay concentrations were present in the lake. Samples below the 1

ug l⁻¹ detection limit were assigned concentrations of 0.5 ug l⁻¹. TP (unfiltered water) and TFP (filtered water) were determined by digesting the samples with persulfate, autoclaving for 1/2 hour (Wetzel and Likens 1979) and allowing samples to sit overnight. P was then measured using the FRP technique described above. Particulate phosphorus (PP) was defined as the difference between TP and TFP. Filterable unreactive phosphorus (FUP) was defined as the difference between TFP and FRP. Nitrogen analyses were performed by other members of the Jordan Lake project using standard spectrophotometric techniques (Wetzel and Likens 1979).

STREAM SAMPLING AND SAMPLE PROCESSING

Water samples were collected from the Haw River at Bynum, N.C., during five major precipitation events from January 1985 to May 1985. Samples were collected at approximately the peak of the stream hydrograph. During the storms of January 1-2, 1985 and February 1-2, 1985, the stream was sampled twice in order to evaluate inter-event variability. On May 23, 1985 the upper Haw River arm of Jordan Lake was also sampled. Grab samples (20 l) were obtained with polyethylene carboys at 0.5 m depth.

Samples were immediately returned to the laboratory where they were immediately filtered through a 360 um plankton net screening to remove large particulate material and detritus.

Centrifugation of the samples to concentrate the

suspended sediments was generally begun within 12 hours, except for the May 16, 1985 sample which was centrifuged after 72 hours. A Sorvall RC-5B superspeed refrigerated centrifuge equipped with a continuous flow-through unit was run at 18,200 rpm at a flow of 180 ml/min. This was found to remove approximately 90 % (by weight) of the particulate material retained on a .2 um membrane filter. The resulting slurry was stored in glass beakers in the dark at 4 C.

Total residue (TR) and total centrifugable residue (TCR) (as opposed to total filterable residue) were measured by drying 100 ml samples of unfiltered and filtered stream water respectively, in evaporating dishes for 36 to 48 hours at 104 C. Repeated weighings were later made to ensure no further decreases in weight occurred. Total noncentrifugable residue (TNCR), which is closely related to the suspended solids concentration, was calculated by the difference between TR and TCR. Loss on ignition (LOI), an estimator of organic content, was obtained by ashing the residues for 3 hours at 550 C.

NUTRIENT ANALYSES-STREAM WATER

Total phosphorus, total centrifugable phosphorus (as opposed to total filterable phosphorus), and filterable reactive phosphorus were analyzed using methods described earlier.

Particulate phosphorus was directly measured using a modification of the digestion step of the total phosphorus analysis. 1 ml of sediment slurry was diluted to 10 ml with

4x persulfate in glass centrifuge tubes. The tubes, placed on their sides to maximize the sediment surface area in contact with the persulfate, were autoclaved for 1 hour at 15 psi. Samples were then placed on a rotary shaker for 18 to 24 hours, centrifuged, and a 2 ml aliquot of centrifugate was removed and diluted to 10 ml with distilled water. Samples were then analyzed for FRP. Standards and blanks were processed in the same manner. Spikes showed good recovery, generally 90 to 110 %.

Base extractable phosphorus was measured by extracting the sediment slurry with 0.1 N NaOH in 0.1 N NaCl for 18 hours in polyethylene centrifuge tubes on a rotary shaker (Williams *et al.* 1971; Armstrong *et al.* 1979). Sediment (15 mg dry weight) was diluted with extraction solution to 30 ml, resulting in a sediment to solution ratio of approximately 1:2000 by weight. Following extraction, the samples were centrifuged, an aliquot of centrifugate was removed and neutralized to pH 7, FRP was then measured.

DESORPTION-ADSORPTION CHARACTERISTICS OF SUSPENDED SEDIMENTS

Experiments to determine whether sediments were acting as a source or sink for soluble phosphorus in the river and lake were performed using methods similar to those of White and Beckett (1964), Taylor and Kunishi (1971), and Kuenzler and Greer (1980). 50 mg (dry weight) of sediment slurry was added to 40 ml of 0.01 M CaCl_2 which contained from 0 to 500 $\mu\text{g P l}^{-1}$. Flasks containing the sediment solution were equilibrated on a shaker table (140 oscillations/min) in the

dark for 12 hours at the approximate ambient temperature of the stream when sampled. Samples were then centrifuged, filtered through a .45 μ m Metrical TCM membrane filter and analyzed for FRP. The amount desorbed or adsorbed was calculated from changes of the initial phosphate concentrations in the flask. From the resulting curves, the equilibrium phosphate concentration (EPC), the concentration where no net desorption or adsorption occurs, was calculated. The amount of phosphate that could be adsorbed by the suspended sediments at the ambient stream FRP concentrations, termed the instantaneous sorption capacity index (ISCI), was calculated by measuring the height of the sorption curve above the x-axis (solution phosphate concentration). Samples from January and February 1985, were initially used within 3 to 5 days of collection, however, later realization that failure to maintain a constant ionic strength in all test flasks produced errant results, necessitated redoing the experiments on sediment slurries that had been stored frozen for several months. All other sorption isotherms were performed within 3 to 5 days of collection. On the May 3rd samples, the effects of biotic activity were measured by poisoning the flasks with 5 drops of chloroform.

BIOAVAILABILITY OF SEDIMENT-BOUND PHOSPHORUS

The amount of suspended sediment-bound phosphorus available for utilization by algae was measured via a modified version of the Selenastrum capricornutum Printz

Algal Assay Bottle Test (Miller *et al.* 1978). Selenastrum was grown with sediments as the sole source of phosphorus. The resulting growth was compared to a series of standards grown with known amounts of orthophosphate, assumed to be completely available. Eight- to ten- day old cultures of Selenastrum were concentrated and used as inoculum. Cultures of this age were deemed old enough to minimize carry-over of luxury phosphate but young enough to prevent any physiological damage that may occur when cells are severely phosphorus starved (Shiroyama *et al.* 1975). Sediment was resuspended in P-free media to give an initial total-P concentration of 317 ug P l⁻¹ (19 ug P/60 ml). Standards were prepared with known amounts of KH₂PO₄ (0-50-100-150-200-250 ug P l⁻¹). Three or more replicate flasks were inoculated to produce an initial algal concentration of 1x10⁴ cells/ml. Flasks were incubated at 24 C under continuous fluorescent lighting of approximately 400 foot-candles and rotated daily on the shaker table to ensure equal light exposure. Flasks were stoppered with cotton plugs and shaken at 110 oscillations per minute. Every three days, at least 300 cells were counted with hemacytometer counting chambers on a American Optical microscope. Clumping of sediments frequently occurred several days after incubation began. However, counts before and after gentle grinding with a tissue grinder showed no significant differences. Assays were terminated after the maximum number of cells per flask had been reached, usually

after 15 days.

Phosphorus limitation was ensured in the flasks by maintaining a N:P ratio of 17:1 by weight. In the sediment flasks, this ratio is maintained if the maximum phosphorus bioavailability is 80% or less. Malone et al. (1978) found that the minimum ratio to ensure phosphorus limitation for Selenastrum was 10:1. Flasks were also spiked with phosphorus at the end of the assays to confirm P limitation.

Sediment phosphorus bioavailability was calculated by comparing the average number of cells produced per unit P of the standards (100% available) to the number produced via the sediments.

RESULTS

PHYSICAL CHARACTERISTICS OF LAKE WATER

The surface (0.5 m) water temperature profile exhibited the characteristic sinusoidal curve (Figure 2). Slight spatial variability observed between stations can largely be attributed to time of day sampled. Additionally, the different origins of the water masses, Stations 5 and 10 came primarily from the New Hope River watershed under normal flow conditions while Station 30 water originates in the Haw River watershed, was likely important. The lowest

SURFACE TEMPERATURE

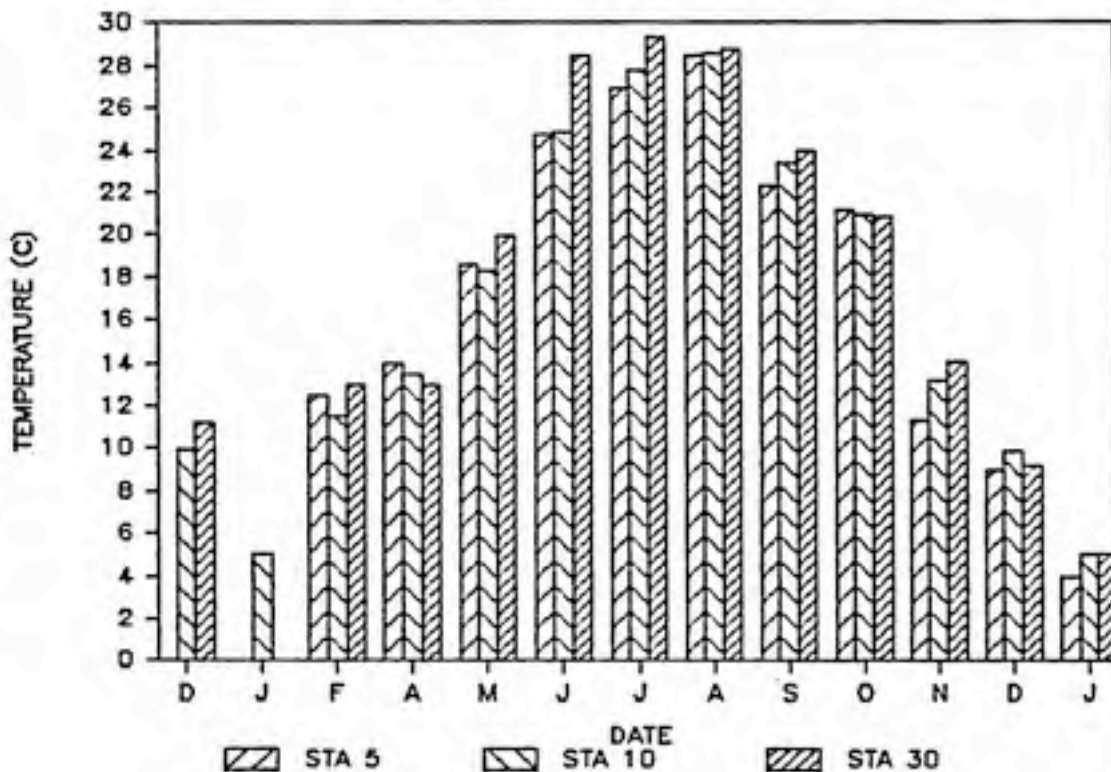


Fig. 2. Temperature (C) distribution in the surface waters of Jordan Lake, December 1983 to January 1985.

temperature (4.0 C) occurred in January 1985 at Station 5 while the highest temperature recorded (29.4 C) was at Station 30 in July 1984 after a period of prolonged warm weather. Ice was not observed in the main water channels of the lake during any winter samplings. Vertical temperature profiles (data not shown) usually exhibited little variation (0-3 C) between surface and bottom waters during the cool months. Stratification of varying intensity was evident from June to August 1984 during which time anoxic conditions were frequently observed in the hypolimnion.

Light attenuation through the water column was high as evidenced by the shallow depths of the photic zone, which, in this report is defined by the depth of the 1% light level (Figure 3). Station 5, and to a greater extent, Station 30, were subject to heavy point source loadings of suspended clays during the winter months, hence the shallow 1% light levels. The proximity of Stations 5 and 30 to tributary inputs cause them to be highly influenced by hydrologic events. Changes in the characteristics of the water at these two stations can occur rapidly. Consequently, while much of the data presented in this report is presented in the form of continuous data, however, it should be emphasized that significant deviations in the various parameters being measured could have occurred between the monthly samplings. Station 10, further removed from point source loadings, was consequently less responsive to heavy suspended sediment loadings and usually had the deepest photic zone. Spatial variability and overall light

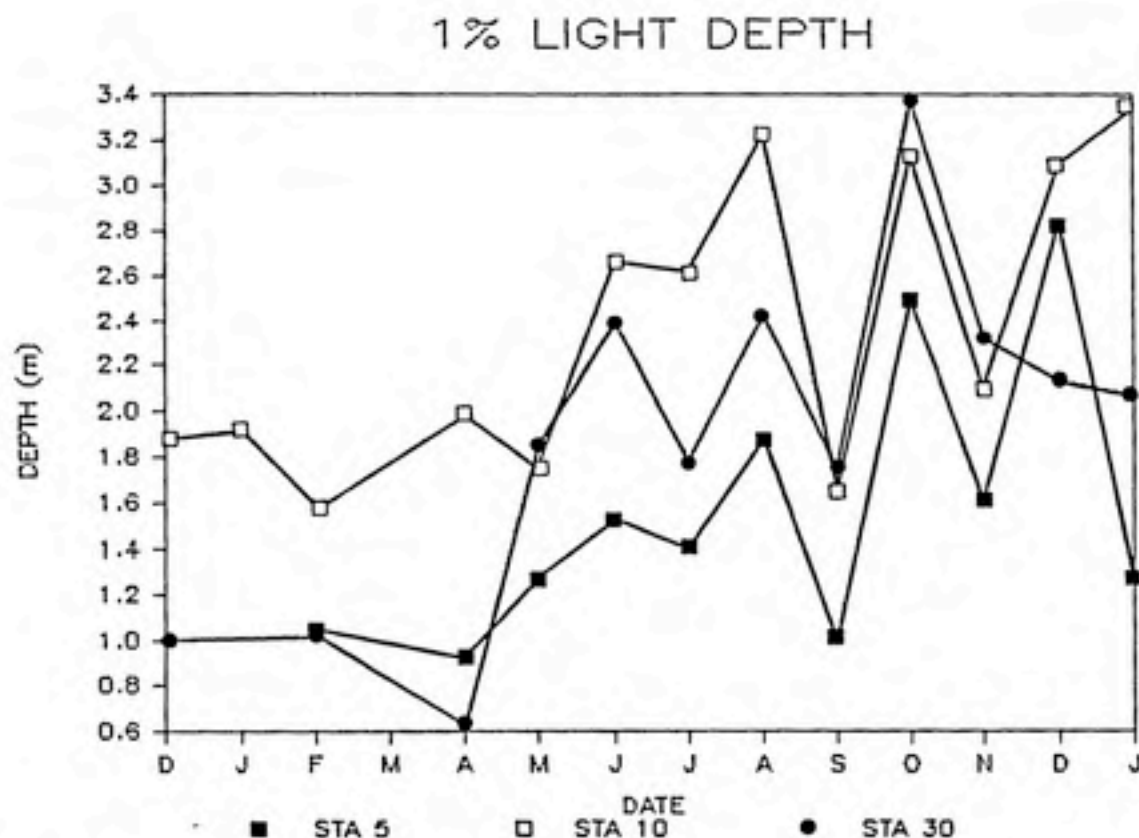


Fig. 3. Distribution of the 1% light level in Jordan Lake, December 1983 to January 1985.

attenuation decreased during the summer months when the amount of suspended clays in the water column appeared to decrease. Poor correlation between chlorophyll-a and the 1% light level ($r = -.138$), combined with visual observations of water samples, indicate that suspended clays, not phytoplankton, were the predominant source of turbidity for most of the year.

CHLOROPHYLL-a DISTRIBUTION

Chlorophyll-a concentrations, measured from April 1984 to January 1985 (Figure 4), are an indicator of algal

CHLOROPHYLL CONCENTRATIONS

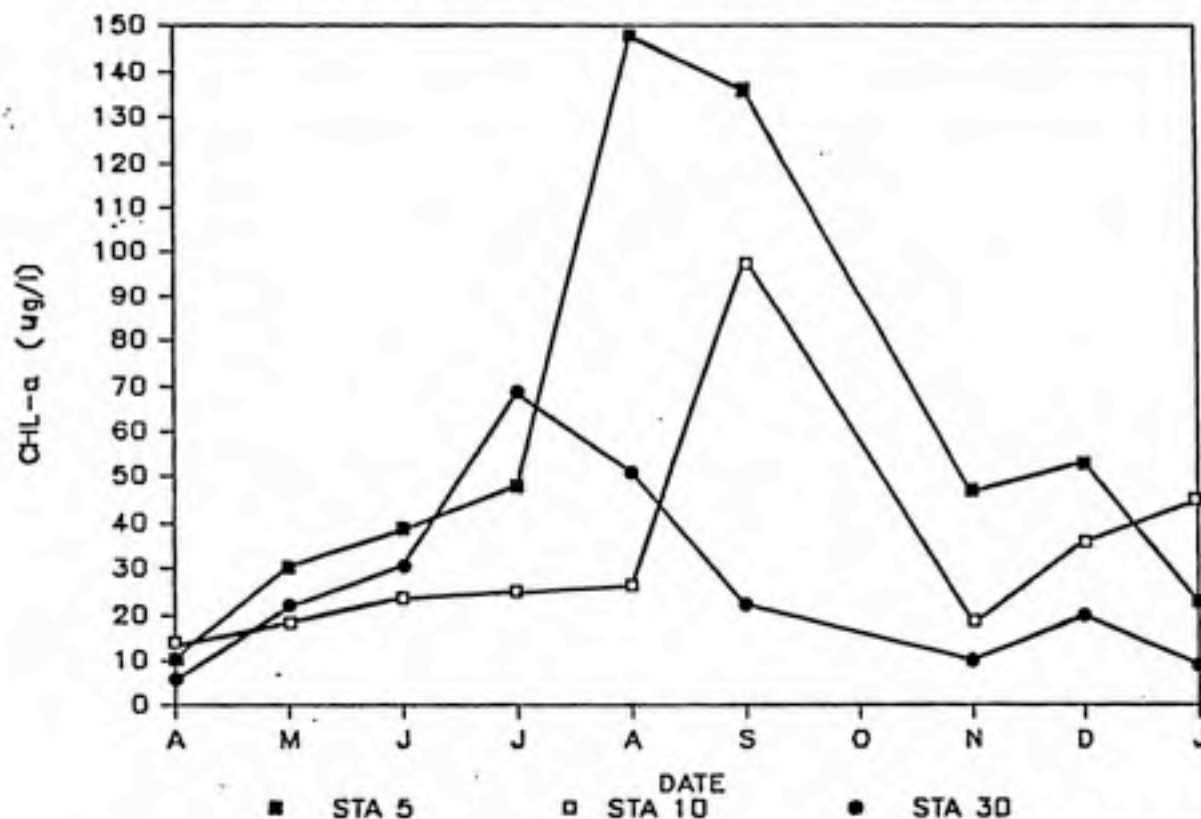


Fig. 4. Chlorophyll-a distributions in the surface waters of Jordan Lake, April 1984 to January 1985.

abundance. Chlorophyll-a concentrations were high throughout the year, with an overall three-station mean of 39.9 ug/l. Station 5 had the highest annual chlorophyll-a concentrations (mean chl-a = 59 ug l⁻¹; S.D. = 46.0), followed by Station 10 (mean = 34; S.D. = 24.7) and Station 30 (mean = 27; S.D. = 19.6). Highest concentrations occurred during the summer and fall periods, likely due, in part, to increased water retention times due to low flow conditions. Lower concentrations were observed during the winter months at all stations. This was likely due to decreased temperatures (Figure 2), increased light attenuation (Figure 3), and periodic flushing of the system.

during periods of heavy stream flow.

PHOSPHORUS DISTRIBUTIONS

The distribution of the phosphorus fractions in the lake appeared to be controlled largely by the hydrologic regime of the lake. During the cool season, periods of high stream flow following heavy precipitation events delivered nutrient-rich water into the lake, while during the warm season periods of low flow and long retention time likely allowed for biological uptake to reduce nutrient concentrations. Total phosphorus (TP) averaged 120.5 ug l^{-1} , clearly indicating the eutrophic status of the lake. Spatial variability of TP was very pronounced (Figure 5). Station 30 always had the highest TP concentrations (mean TP = 219 ug l^{-1} ; S.D. = 145.9), followed by Station 5 (mean = 118 ; S.D. = 88.4), and then by Station 10 (mean = 72 ; S.D. = 43.7) indicating a general trend of increasing TP with decreasing distance from major tributary inputs. Distribution patterns were dominated by high TP concentrations during the winter of 1984 when periods of high suspended sediment inputs occurred. Spatial and temporal variation was less pronounced from June through November 1984, during which time TP concentrations ranged from 31 to 171 ug l^{-1} . Particulate phosphorus (PP) concentrations (Figure 5) were high (22 to 404 ug l^{-1} , $\text{ave}=77.1 \text{ ug l}^{-1}$) and very closely correlated with TP ($r=.921$ $p=.001$). Both TP and PP were poorly correlated with chlorophyll-a ($r=-.191$ and $.228$ respectively). Filterable

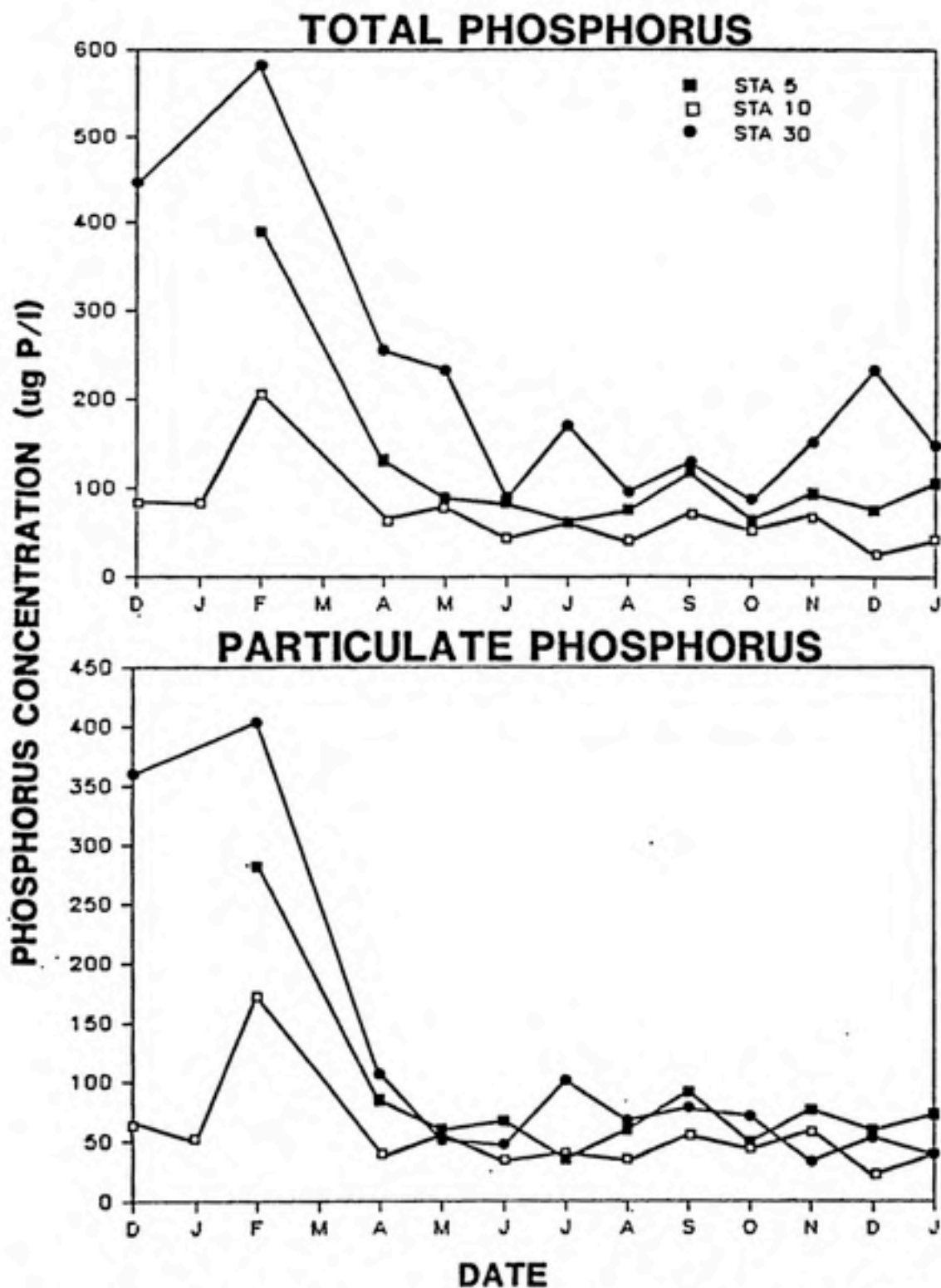


Fig. 5. Total phosphorus and particulate phosphorus distributions in the surface waters of Jordan Lake, December 1983 to January 1985.

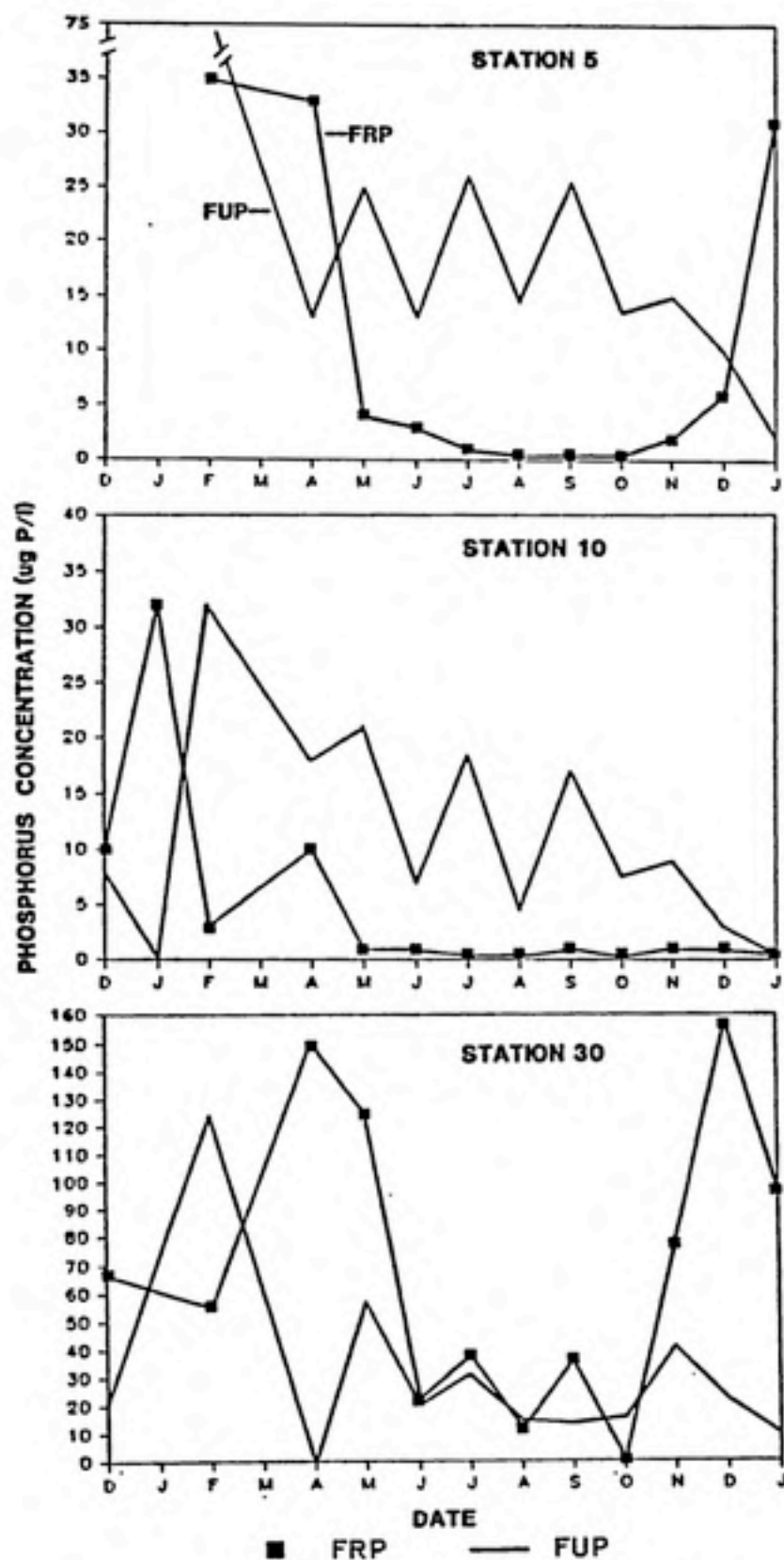


Fig. 6. Filterable reactive P and filterable unreactive P distributions in the surface waters of Jordan Lake, December 1983 to January 1985.

reactive phosphorus (FRP) concentrations (Figure 6) were high during the winter months of 1983-84 but often decreased to $1 \mu\text{g l}^{-1}$ or below at Stations 5 and 10 during the warm season. Station 30 had high FRP concentrations throughout the sampling period but did decrease to undetectable levels ($<1 \mu\text{g P l}^{-1}$) in October 1984. Filterable unreactive phosphorus (FUP) (Figure 7) usually comprised a greater portion of the total dissolved phosphorus component than did FRP and had less spatial variability. The cycle of FUP can not be easily explained.

FRP UPTAKE COEFFICIENTS AND TURNOVER TIMES

The relative rate of FRP uptake (K) exhibited distinctive spatial and temporal trends (Table 1) characterized by low values during the winter at all 3 stations and high values during the warmer months at Stations 5 and 10. The turnover time ($1/k$) of FRP in the lake showed large temporal and spatial variability (Figure 7). Long turnover times, 4 to 500 h, occurred throughout the lake from December 1983 to April 1984. Large decreases in turnover times occurred at stations 5 and 10 from May to December 1984, reaching a low of .06 h (3.6 min) at Station 10 in August 1984. Station 30 turnover times remained long throughout the study period, ranging from 10.8 to 143 h, except for one decrease to 1.2 h in October 1984 when FRP levels dropped to less than $1 \mu\text{g l}^{-1}$.

Gross uptake rates (v), equal to the uptake rate coefficient multiplied by the FRP concentration, varied from

Table 1. Phosphate uptake coefficient (K) distribution in surface waters of Jordan Lake. Units = h^{-1} .

=====			
DATE	STATION 5	STATION 10	STATION 30

18-DEC-83	---	.230	.068
20-JAN-84	---	.031	---
22-FEB-84	.002	.040	.017
06-APR-84	.039	.091	.012
05-MAY-84	1.29	2.99	.014
07-JUN-84	2.03	9.78	.074
12-JUL-84	2.59	3.09	.030
14-AUG-84	2.30	11.86	.095
17-SEP-84	1.82	7.40	.017
18-OCT-84	6.61	9.54	.838
17-NOV-84	1.55	.88	.007
16-DEC-84	1.59	6.03	.024
19-JAN-85	.034	1.82	.020

FRP TURNOVER TIME

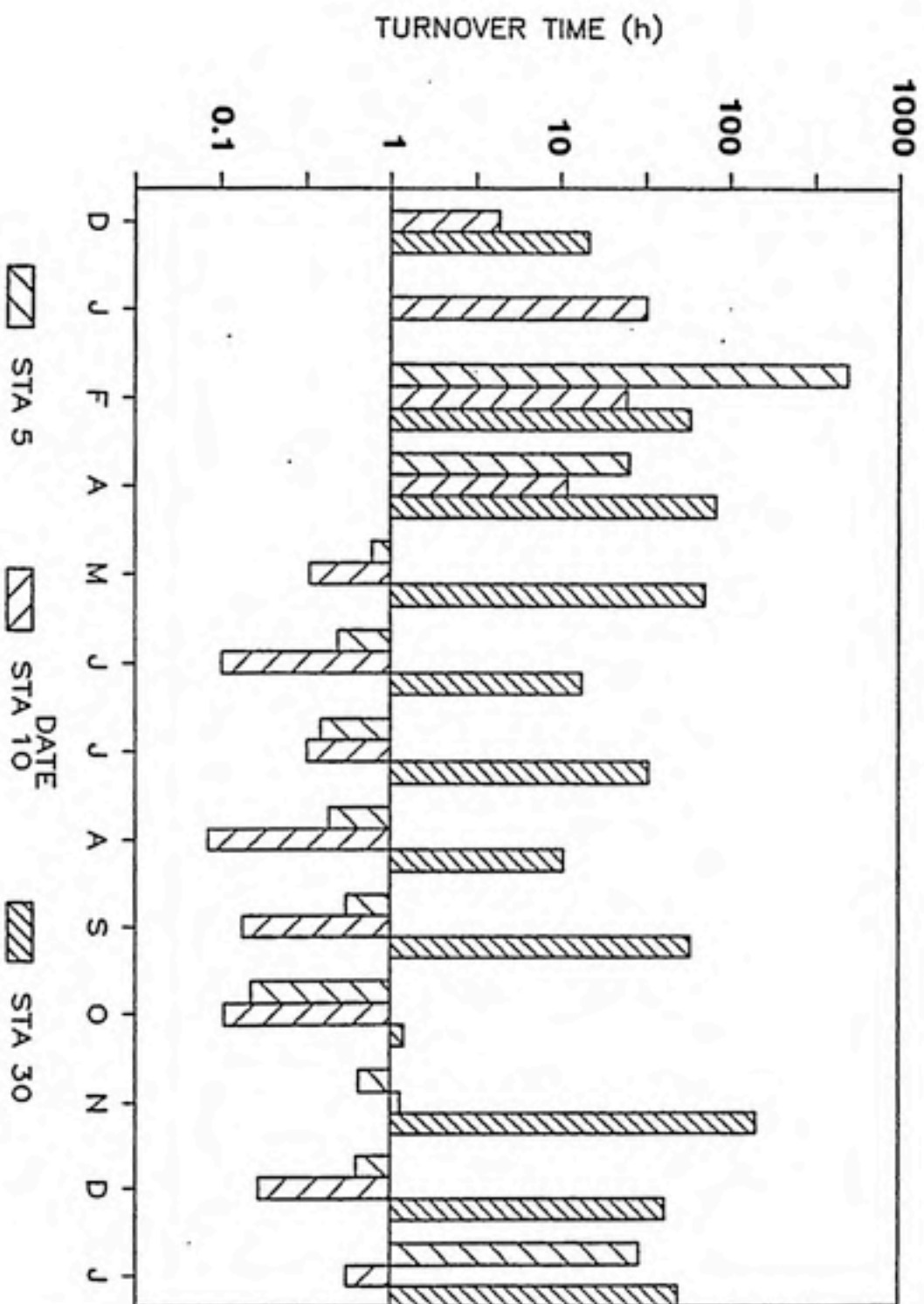


Fig. 7. Filterable reactive phosphorus turnover times (h) in surface waters of Jordan Lake, December 1983 to January 1985.

0.007 to 9.78 $\mu\text{g P l}^{-1} \text{ h}^{-1}$ (Figure 8). The lowest and most nearly constant uptake rates occurred at Station 30, even though TP concentrations were highest there. Rates at Stations 5 and 10 had significantly more variation with time and tended to be higher during the warmer months.

GROSS ABIOTIC AND BIOTIC UPTAKE RATES

Abiotic uptake rates, defined as the rate of uptake in formaldehyde-treated samples, were highest at Station 30 ($\bar{V}=1.08 \mu\text{g P l}^{-1} \text{ h}^{-1}$; S.D.=1.25) (Figure 8) likely due to the high concentrations of suspended clays and FRP that often occurred there. At Station 30, abiotic uptake was responsible for 100 percent of the total uptake in December 1983, February, May, November 1984, and January 1985 (Figure 9). Station 5 abiotic uptake ($\bar{V}=0.50 \mu\text{g P l}^{-1} \text{ h}^{-1}$; S.D.=0.87) was relatively constant and low, except for slight increases in April 1984 and January 1985 (Figure 8). Station 10 had the lowest abiotic uptake rates ($\bar{V}=0.09 \mu\text{g P l}^{-1} \text{ h}^{-1}$; S.D.=0.13). Abiotic uptake comprised less than 1 percent of the total uptake from June through October 1984, and only once, in February 1984, did it contribute more than 50 percent of the total uptake (Figure 9).

Biotic uptake was separated into two size fractions, a 0.45 to 8.0 μm fraction (small particle), assumed to represent bacterial uptake, and $> 8.0 \mu\text{m}$ fraction (large particle), assumed to represent algal uptake. Biotic uptake was dominated by the small size fraction (Figures 8,9), averaging 87.6 percent (S.D.=7.9) of total biotic uptake.

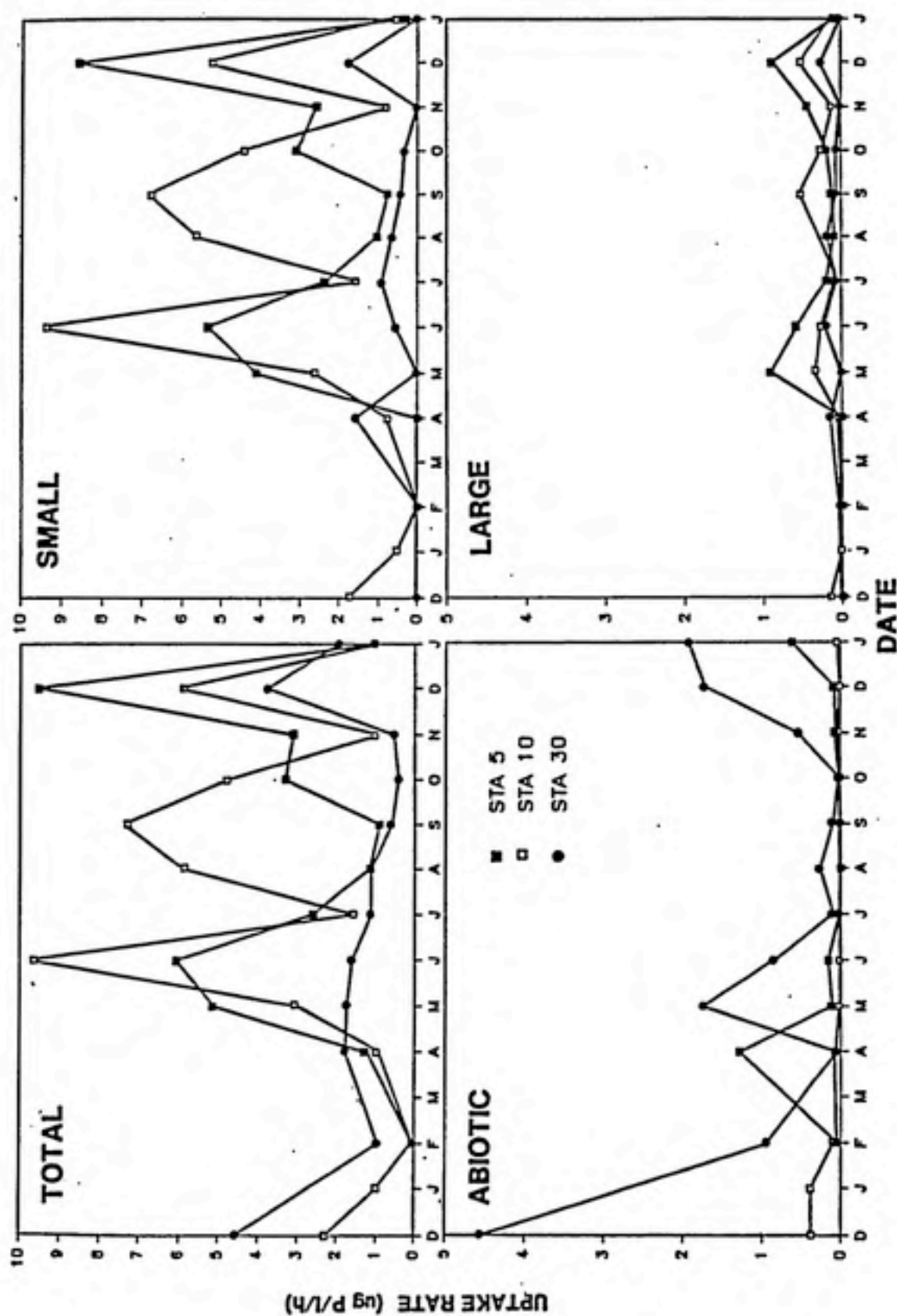


Fig. 8. Total and partitioned gross uptake rates in surface waters of Jordan Lake, December 1983 to January 1985.

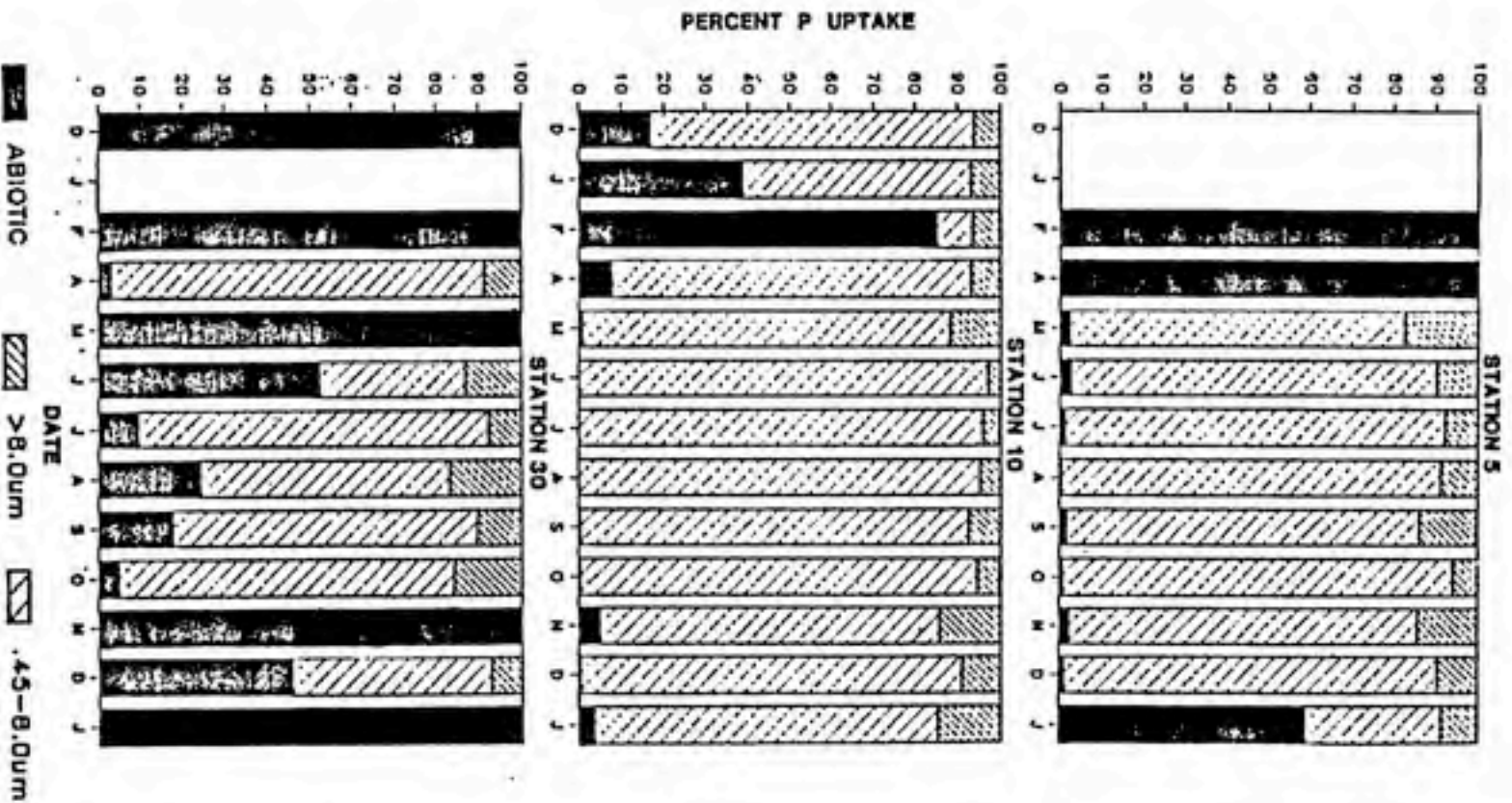


Fig. 9. Relative contribution of abiotic, 0.45-8.0 um biotic, and >8.0 um biotic fractions to total phosphate uptake in surface waters of Jordan Lake, December 1983 to January 1985.

Uptake attributable to the large fraction, while always measurable, never comprised more than 20 percent of total biotic uptake at any station, except in February 1984 at Station 10 when it comprised 42 percent of the biotic uptake during a period of very low total uptake.

At Station 10, antibiotic treatment of water samples was used in addition to size fractionation, to distinguish algal and bacterial uptake. However, since it is likely that antibiotic treatment inhibits the phosphorus uptake capabilities of blue-green algae as well as that of bacteria, a more correct differentiation of the uptake fractionation may be eucaryotic and procaryotic uptake. Eucaryotic uptake comprised from 3.5 to 99.9 percent of the total uptake and was always greater than the uptake in the large size fraction (Table 2). Eucaryotic uptake often comprised a large percentage of uptake in the small size fraction (Table 2) indicating either the large (8.0 μm) filter did not effectively retain most of the phytoplankton or the antibiotic was unsuccessful in eliminating all of the bacterial uptake.

NET PHOSPHORUS UPTAKE, VERTICAL PROFILES, AND EFFECT OF SUBSTRATE CONCENTRATION

In situ measurement of uptake rates throughout the water column were performed in March and May 1985 at Station 10 (Table 3). All uptake rates were low in the March sampling despite high chlorophyll-a concentrations. The rates varied nearly threefold throughout the column with the

Table 2. Effects of antibiotics and size fractionation on biotic uptake at Station 10 of Jordan Lake. K_b is the biotic uptake coefficient (h^{-1}).

DATE	TREATMENT	SIZE (μm)	K_b	% of BIOTIC UPTAKE	% UPTAKE	
					EUCARYOTIC	PROKARYOTIC
18 DEC	AMBIENT	>8	.014	7	10	90
		<8	.177	93	19	81
	ANTIBIOT	>8	.0014	4		
		<8	.0036	96		
20 JAN	AMBIENT	>8	.002	11	62	38
		<8	.017	89	105	0
	ANTIBIOT	>8	.001	7		
		<8	.018	93		
22 FEB	AMBIENT	>8	.003	42	16	84
		<8	.004	58	114	0
	ANTIBIOT	>8	.0004	7		
		<8	.006	93		
06 APR	AMBIENT	>8	.006	7	45	55
		<8	.078	93	104	0
	ANTIBIOT	>8	.003	3		
		<8	.081	97		
05 MAY	AMBIENT	>8	.347	12	4	96
		<8	2.62	88	4	96
	ANTIBIOT	>8	.013	10		
		<8	.102	90		
07 JUN	AMBIENT	>8	.264	3	27	73
		<8	9.52	97	4	96
	ANTIBIOT	>8	.071	14		
		<8	.428	86		
12 JUL	AMBIENT	>8	.124	4	160	0
		<8	2.97	96	97	3
	ANTIBIOT	>8	.198	6		
		<8	2.90	94		
14 AUG	AMBIENT	>8	.58	5	73	27
		<8	11.28	95	32	68
	ANTIBIOT	>8	.427	11		
		<8	3.56	89		
17 SEP	AMBIENT	>8	.544	7	14	86
		<8	6.831	93	22	78
	ANTIBIOT	>8	.078	5		
		<8	1.517	95		
18 OCT	AMBIENT	>8	.506	5	28	72
		<8	9.00	95	20	80
	ANTIBIOT	>8	.140	7		
		<8	1.78	93		
17 NOV	AMBIENT	>8	.125	15	163	0
		<8	.712	85	76	24
	ANTIBIOT	>8	.204	27		
		<8	.542	73		
16 DEC	AMBIENT	>8	.547	9	24	76
		<8	5.46	91	8	92
	ANTIBIOT	>8	.130	22		
		<8	.437	78		

Table 3. Distribution of filterable reactive P (FRP), phosphate uptake coefficients (K), phosphorus uptake rates (v), and chlorophyll-a, with depth at Station 10 of Jordan Lake. Units: FRP= $\mu\text{g P l}^{-1}$, $K=\text{h}^{-1}$, $v=\mu\text{g P l}^{-1} \text{ h}^{-1}$, chl-a= $\mu\text{g l}^{-1}$.

DATE	DEPTH	FRP	K	v	CHL-a
MARCH 1985	0.2	6	.069	.414	43
	0.5	5	.048	.240	69
	1.0	4	.049	.196	57
	2.0 *	5	.058	.290	35
	4.0	6	.023	.138	22
MAY 1985	0.2	4	13.9	55.6	24
	0.5	5	21.2	105.9	22
	1.4	2	14.7	29.4	25
	3.0 *	2	13.5	27.0	19
	5.0	2	2.9	5.8	10

*DENOTES APPROXIMATE 1% LIGHT LEVEL

highest rates occurring in the surface (0.2 m) sample ($v = .414 \text{ ug P l}^{-1} \text{ h}^{-1}$) and the lowest occurring in the deep (4 m) sample ($v = .138 \text{ ug P l}^{-1} \text{ h}^{-1}$). The May 1985 vertical profile was characterized by very high uptake rates, especially in the upper 0.5 m of the water column where the highest FRP concentrations were present. Below the metalimnion, present at 3 to 4 m, uptake rates decreased greatly. Size fractionation was not performed at the various depths, hence relative algal and bacterial uptake can not be estimated. The uptake rates generated from these vertical profile experiments should not be compared to other rates generated throughout the study because of different incubation conditions and a different method of calculating the uptake rate (using only two points, the initial and final filtrate activities, to derive the uptake rate coefficient). The uptake rates should not be critically compared between the vertical profile experiments since different incubation times were used, however, the order of magnitude increase observed in the rates in May 1985 as compared to those in March 1985, despite lower algal biomass and similar FRP concentrations, is probably a result of increased biological metabolism, not of the different incubation times.

The effect of additional phosphate substrate on uptake rates was measured in August 1984 and May 1985 at Station 10. Increments of 1 to 20 ug P l^{-1} were added to lake water and the resulting uptake rates were measured. The uptake rates were plotted using a Lineweaver-Burke transformation

of the Michaelis-Menton equation and the half saturation concentration (K_s), the concentration at which uptake is equal to one-half of the maximum uptake, was obtained. The K_s measured was 3.31 ug P l^{-1} in August 1984 and was 3.8 ug P l^{-1} in May 1985. Ambient FRP concentrations (0.5 and 2.0 ug P l^{-1} , respectively) were below the K_s values on both occasions, indicating that the rates of uptake by the biota were limited by the phosphorus available. Additional attempts at measuring K_s values in May and October 1984 were unsuccessful in that uptake rates at different phosphorus concentrations did not conform to the Michaelis-Menton equation and had no clearly discernable pattern in general.

Changes in the FRP concentrations over time (3-4 h) were occasionally measured to provide an indication of net phosphate uptake (Table 4). Low ambient FRP concentrations often made detection of changes in the concentration impossible and emphasizes the inherent difficulties involved

Table 4. Change of filterable reactive phosphorus concentrations during incubation of Jordan Lake water. Units are ug P l^{-1} .

DATE	STATION	INITIAL	FRP FINAL	CHANGE
MAY 84	10	1	1	0
AUG 84	30	36	40	+ 4
DEC 84	10	6	5	- 1
JAN 85	10	< 1	< 1	0
MAY 85	10	2	2	0

High phosphorus loads were apparent in the Haw River presumably due to the high phosphorus content of clays in combination with the numerous agricultural and industrial sources of phosphorus in the Haw River watershed. Total phosphorus (TP) (412 to 1236 $\mu\text{g P l}^{-1}$), and filterable reactive phosphorus (FRP) (87 to 479 $\mu\text{g P l}^{-1}$) (Table 6) were both high but not correlated with each other. FRP was inversely correlated with flow ($r = -.80$ $p = .05$).

SORPTION OF PHOSPHORUS BY SUSPENDED SEDIMENTS

All sediment samples exhibited considerable ability to adsorb phosphorus from solution (Figure 10). The equilibrium phosphate concentration (EPC), represented by the point where the sorption isotherm curve crosses the x-axis, ranged from 1 to 77 $\mu\text{g P l}^{-1}$ (Table 7). The EPC was always greater than the ambient stream FRP and the estimated lake FRP, indicating that the sediments were adsorbing phosphate, acting as a sink, on all occasions.

Direct comparison of the instantaneous sorption capacity (ISC) of the sediments, which estimates the amount of phosphate that can be adsorbed per gram dry weight of sediment at the ambient stream FRP concentration, can be made since a constant amount of sediment (50 mg/40 ml) was used in all experiments. The average ISC of the May stream samplings (140 $\mu\text{g P/g sed}$) was significantly higher ($p = .05$) than the average ISC of the January and February samplings (51 $\mu\text{g P/g sed}$). The ISC was inversely correlated with flow ($r = -.75$ $p = .10$). A new parameter, the net potential sorption

in trying to estimate uptake rates using chemical techniques. In two instances, discernable changes in the FRP concentration were measured; a decrease of 1 ug P l⁻¹ in December 1984 at Station 10 and an increase of 4 ug P l⁻¹ in August 1984 at Station 30.

PHYSICAL AND CHEMICAL CHARACTERISTICS OF STREAM WATER

Stream temperatures remained constant during the cool season samplings (January-February) at 5 C and increased during the warm season samplings (May) to range from 17 to 23 C (Table 5). Estimates of the lake water temperatures at the time of stream sampling, based on lake samplings generally within 10 days of the stream sampling, are 5 C and 18 to 21 C respectively, indicating that density (mainly temperature) differences between stream and lake water were likely too small to prevent some mixing of the water masses. Instantaneous flow rates of storm events (Table 5) were much higher during January and February (ave.=15,450 cfs) than in May (ave.=2963 cfs). Total residue (TR) and total nonfilterable residue (TNFR) were usually high and showed considerable variability (Table 5). Total filterable residues (TFR) remained relatively constant. The percentage of dry weight lost upon ignition (% LOI), an indication of organic content, ranged from 11 to 15 % on the sediment slurry samples used for experimentation (Table 5). % LOI was higher on TR samples (data not shown) than on the slurry samples, indicating a higher percentage of the organic content was contained in the TFR (colloidal-sized) fraction.

Table 5. Temperature (C), instantaneous stream flow (cfs), total and total nonfilterable residue (mg l⁻¹), and % loss on ignition of sediment slurry of samples from the Haw River at Bynum, NC, 1985.

DATE	TEMP	FLOW	TR	TNFR	% LOI
1-3	5	10,000	337	231	15
1-4	5	11,200	284	199	14
2-1	5	20,100	681	601	11
2-2	5	20,500	278	217	12
5-3	17	2,220	242	138	11
5-16	22	1,020	212	48	15
5-23	23	5,650	1333	1204	11
5-23(Lake)	23		957	820	11

Table 6. Total P, total filterable P, filterable reactive P, and particulate P (ug P l⁻¹) of water samples from the Haw River at Bynum, NC, 1985.

DATE	TP	TFP	FRP	PP
1-3	811	329	235	462
1-4	564	143	92	342
2-1	674	120	95	493
2-2	412	99	87	308
5-3	475	275	254	161
5-16	549	492	479	112
5-23	1236	268	195	1264
5-23(Lake)	932	300	232	853

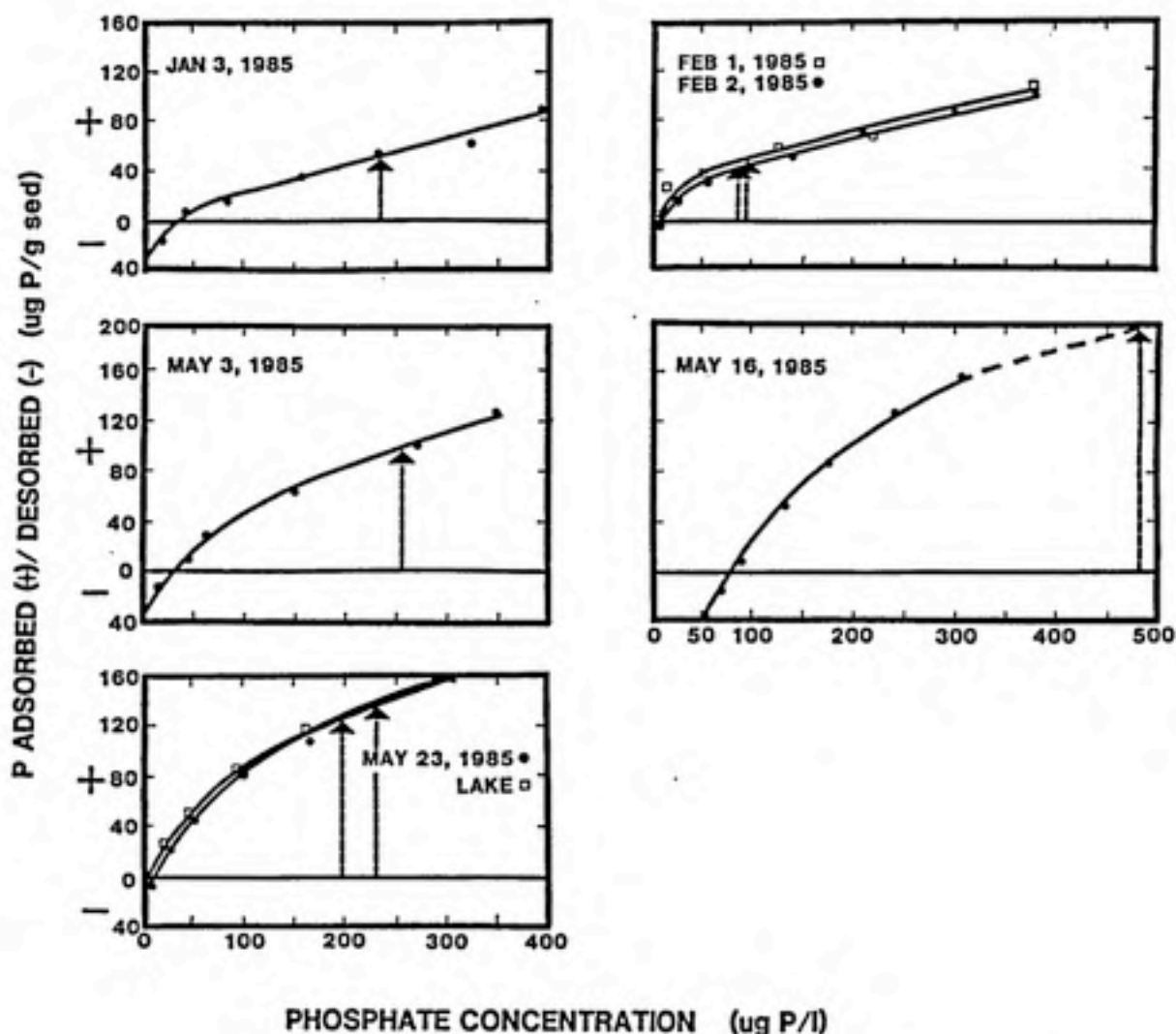


Fig. 10. Phosphate sorption isotherms of suspended sediment from the Haw River at Bynum, NC. The concentration at which the isotherm crosses the x-axis is the equilibrium phosphate concentration. Dashed vertical lines, drawn from the point on the x-axis equal to the ambient stream FRP concentration, represent the instantaneous sorption capacity of the sediment.

Table 7. Stream and estimated lake filterable reactive P (ug P l⁻¹), equilibrium phosphate concentration (ug P l⁻¹), instantaneous sorption capacity (ug P/g sed), and net potential sorption capacity (ug P m⁻¹) of samples from the Haw River at Bynum, NC.

=====

DATE	STREAM FRP	LAKE FRP	EPC	ISC	NPSI
03-JAN	235	200	37	52	340
01-FEB	95	125	1	52	1782
02-FEB	87	125	6	50	633
03-MAY	254	120	25	99	86
16-MAY	479	310	77	199	28
23-MAY	195	390	11	122	2351
23-MAY (Lake)	232	390	6	135	

index (NPSI), was developed to compare the instantaneous phosphorus sorption-potential of the stream during different samplings (Table 7). The NPSI was calculated using the equation:

$$\text{NPSI} = (\text{ISC}) (\text{TNFR}) (\text{FLOW}).$$

Units for NPSI are $\mu\text{g P (potentially sorbed) sec}^{-1}$. The high NPSI value on February 1 is largely due to high stream flow while the high NPSI on May 23 is largely due to high TNFR (Table 5). Low NPSI values occurred on May 3 and May 16 despite having high ISC values.

Two samples collected in February from the same storm event but at different periods showed relatively small differences between the EPC and ISC (Table 7). However, since the TNFR on February 1 was nearly three times the TNFR on February 2, and the streamflows were essentially equal, the NPSI of the February 1 sample was three times that of the February 2 sample. This emphasizes the large potential error that may occur when estimates of the net effect of suspended sediments are made using data from only one sampling during high flow conditions.

The May 23 sampling of both the stream and Haw River arm of Jordan Lake showed a relatively small decrease in the EPC (11 vs 6) and a small increase in the SCI (122 vs 139) (Table 7). TP and TNFR decreased from the stream to the lake indicating slight sedimentation of some of the larger particulates may have occurred, increasing the percentage of finer particulates which have higher sorptive capacities and consequently, higher ISC values. Such direct comparison of

samples must be done with the understanding that the samples taken from the lake resulted from an earlier parcel of stream water which may have contained sediments and water with considerably different sorption-related characteristics.

The addition of chloroform to the flasks in which the May 3rd sorption experiments were being performed resulted in no change in either EPC or ISC, indicating that biotic uptake of phosphate during the experiments was not significant.

BIOAVAILABILITY EXPERIMENT RESULTS

The percentage of algal available phosphorus (AAP) ranged from 5.1 to 18.6 percent of the total sediment phosphorus (Table 8). The mean AAP of the 5 stream samplings was 11.3 percent. Selenastrum's growth curves (Figure 11) using sediment-bound phosphorus were characterized by an initial phase of exponential growth, closely paralleling the growth of standards grown with orthophosphate, which is assumed to be 100% available. This initial rapid growth phase, which lasted approximately three to six days, was usually followed by one to three days of slower growth, after which little growth occurred. The maximum cell numbers were reached between nine and twelve days on all but two samples. Native algae present in the sediment slurry never comprised more than 5% of the total number of algae present and appeared to be outcompeted by the Selenastrum. Under phase-contrast microscopy, some

Table 8. Biomass produced with equal amounts of sediment-bound phosphorus and orthophosphate, percent sediment phosphorus bioavailable, and percent phosphorus removed via NaOH extraction. Standard deviations in parentheses.

=====				
DATE	cells/ug P (x10 ⁴) SEDIMENT	STANDARDS	PERCENT BIOAVAILABLE	PERCENT NaOH EXTR

03-JAN	7.27 (.58)	39.1 (.27)	18.6	52.6
04-JAN	6.06 (.88)	"	15.5	51.6
01-FEB	3.86 (.30)	"	9.9	52.0
02-FEB	2.92 (.41)	"	7.5	46.0
03-MAY	1.99 (.48)	"	5.1	43.2
16-MAY	5.07 (.97)	"	13.0	59.5
23-MAY	4.91 (.73)	"	12.6	46.1
23-MAY(L)	3.97 (.89)	"	10.2	45.6

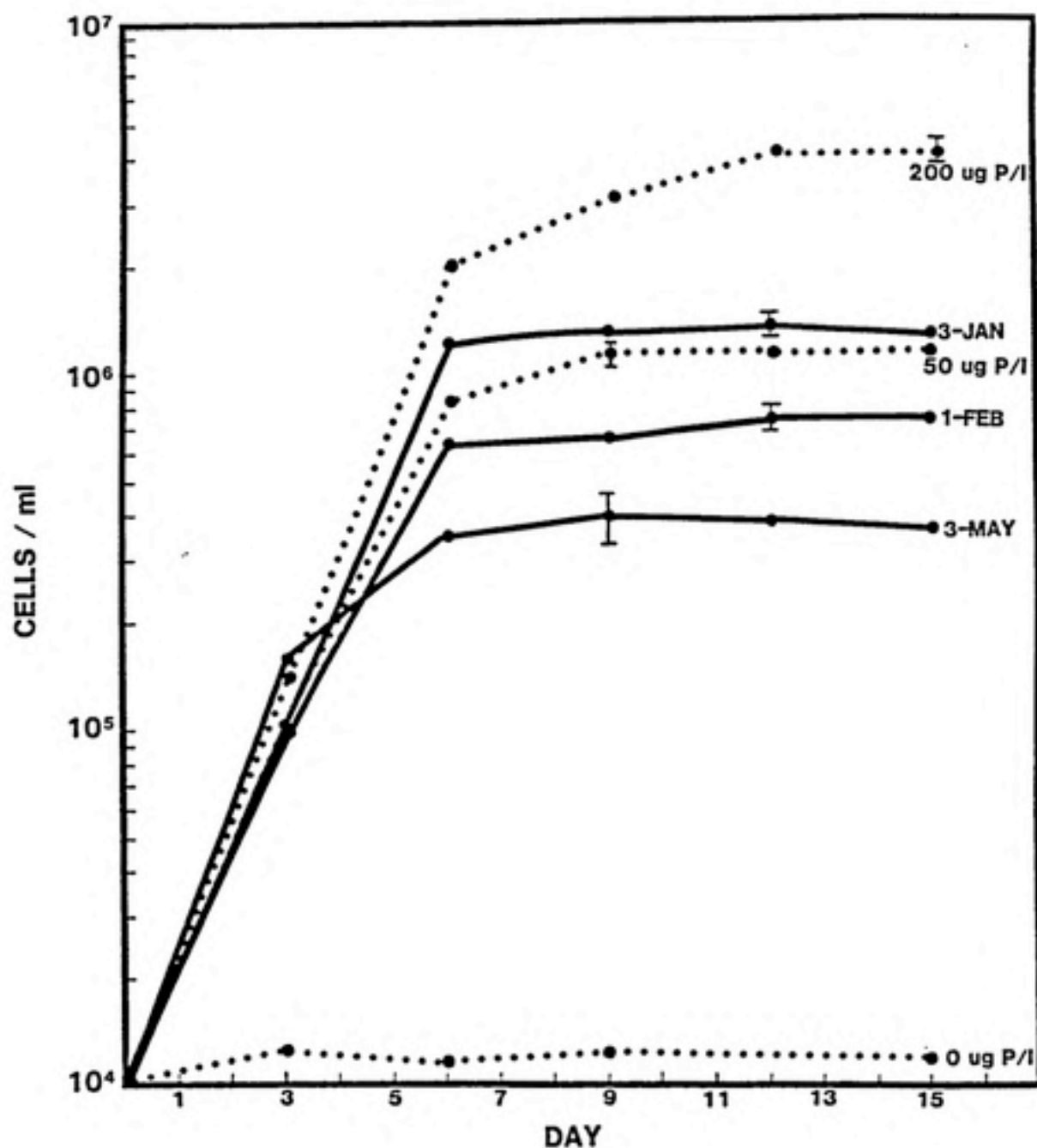


Fig. 11. Examples of growth curves of *Selenastrum capricornutum* grown with sediment (solid lines) and orthophosphate (dotted lines).

bacteria were visible, but they did not appear to flourish during the experiment. Their importance in algal utilization of the sediment P could not be quantified but is assumed to be minimal.

The 0.1M NaOH extraction removed from 43.2 to 59.5% of the sediment phosphorus (Table 8). This is significantly more than the 5.1 to 18.6% of the sediment phosphorus utilized by the algae indicating the NaOH extraction was a very poor indicator of AAP.

Some clumping of the sediments occurred, usually beginning after four to seven days of incubation. Careful pestling of the clumped samples did not significantly alter the algal counts, indicating that direct physical contact of the algae with the sediments may not have been an important factor in the resulting coagulation of the suspended sediments.

DISCUSSION

PHOSPHORUS DISTRIBUTION AND ALGAL BIOMASS

Phosphorus concentrations are high in Jordan Lake (Figures 5 and 6) and indicating eutrophic conditions. Using the trophic state index of Vollenweider (1968), which is based on epilimnetic total phosphorus (TP) concentrations, Station 10 (mean TP=72 $\mu\text{g l}^{-1}$) is classified as eutrophic while Station 5 (mean TP=118 $\mu\text{g l}^{-1}$) and Station 30 (mean TP=219 $\mu\text{g l}^{-1}$) are hyper-eutrophic. Weiss and Francisco (1984) using a trophic state index developed on North Carolina lakes (Weiss and Kuenzler 1976), also concluded that Jordan Lake is eutrophic, even when inputs during periods of heavy tributary stream flow are eliminated.

The high TP concentrations during the winter months at Station 5 and Station 30 are largely attributable to the inputs of phosphorus-rich suspended clays from the Haw River and New Hope River, respectively (Weiss and Francisco 1984). Station 10, being further removed from point source discharges, is less severely impacted by high point source flows, as is evidenced by decreased variability in TP concentrations compared to Stations 5 and 30 (Figure 5). Kuenzler and Greer (1980) found significantly lower values of TP in University Lake, which, while being in close proximity to Jordan Lake, is not subjected to the large amounts of industrial discharges and agricultural runoff that Jordan Lake receives.

During the summer months, increasing water residence times occur (Weiss and Francisco 1984), allowing for the development of greater algal biomass (Figure 4). The increase in biotic activity is likely the cause of reduction of FRP at Stations 5 and 10 to low levels; however, at Station 30, inputs of Haw River water appear to maintain significant levels of FRP throughout the warm season (Figure 6).

Mean annual chlorophyll-a concentrations were highest at Station 5 ($\bar{x}=59.3 \text{ ug l}^{-1}$) and lowest at Station 30 ($\bar{x}=26.5 \text{ ug l}^{-1}$), despite the abundance of phosphorus at this station. These patterns are similar to those found by Weiss and Francisco (1984) and indicate phosphorus limitation is unlikely at Station 30.

Particulate phosphorus and chlorophyll-a were poorly correlated throughout the year (Table 9), probably because the PP fraction was often composed of large amounts of suspended sediment-bound phosphorus and possibly also due to algal storage of polyphosphates. From May to September, when algal biomass was high (Figure 4) and suspended sediment concentrations decreased, the PP-chlorophyll correlation coefficient increased at all stations, probably because of an increase in the relative proportion of algal P per unit PP. Additionally, the slopes of the chlorophyll-PP regression line at Stations 5 and 30 increased in the May to September period, indicating more chlorophyll per unit PP. At Station 10, the similarity of the slopes between the

Table 9. Regression analysis of particulate P-chlorophyll relationships for surface waters of Jordan Lake from April 1984 to January 1985, and May 1984 to September 1984.

=====				
DATE	STATION	r	SLOPE	n

APR-JAN	5	.104	.49	9
	10	.200	1.67	9
	30	.379	.07	9
	ALL	.229	.14	27
MAY-SEP	5	.496	1.38	5
	10	.495	1.51	5
	30	.699	.65	5
	ALL	.432	.90	15

summer months (May to September) and the April to January period, indicates a relatively constant chlorophyll-PP relationship, likely because of the decreased importance of suspended solids at this station. Future studies of phosphorus dynamics in Jordan Lake and in other systems impacted by heavy inputs of inorganic particulate phosphorus should consider including measurements of algal phosphorus (see Kuenzler *et al* 1979) to allow for more critical analyses of phosphorus-algae dynamics.

N:P RATIOS

The Redfield ratio (Redfield 1954), based on the observation that N:P ratios of oceanic waters (7.2:1 by weight) are very similar to N:P ratios found in phytoplankton that grow in these water, are often used to estimate potential nutrient limitations. High N:P ratios imply P limitation while low N:P ratios imply nitrogen

limitation. This approach is complicated, however, by such factors as problems in accurately measuring orthophosphate (Rigler 1966), phosphorus storage capabilities (Reynolds 1984), and in variations between algal species in the optimum N:P ratios for growth (Rhee and Gotham 1980). Based on DIN:DIP ratios ($\text{NH}_4 + \text{NO}_3 + \text{NO}_2 / \text{FRP}$) calculated for Jordan Lake (Figure 12), phosphorus limitation frequently occurs at Station 10, occasionally at Station 5, and rarely at Station 30. During the winter months, high N:P ratios frequently occur implying a potential scarcity of phosphorus relative to nitrogen, however, high concentrations of phosphorus also prevail (Figure 6), often significantly above the

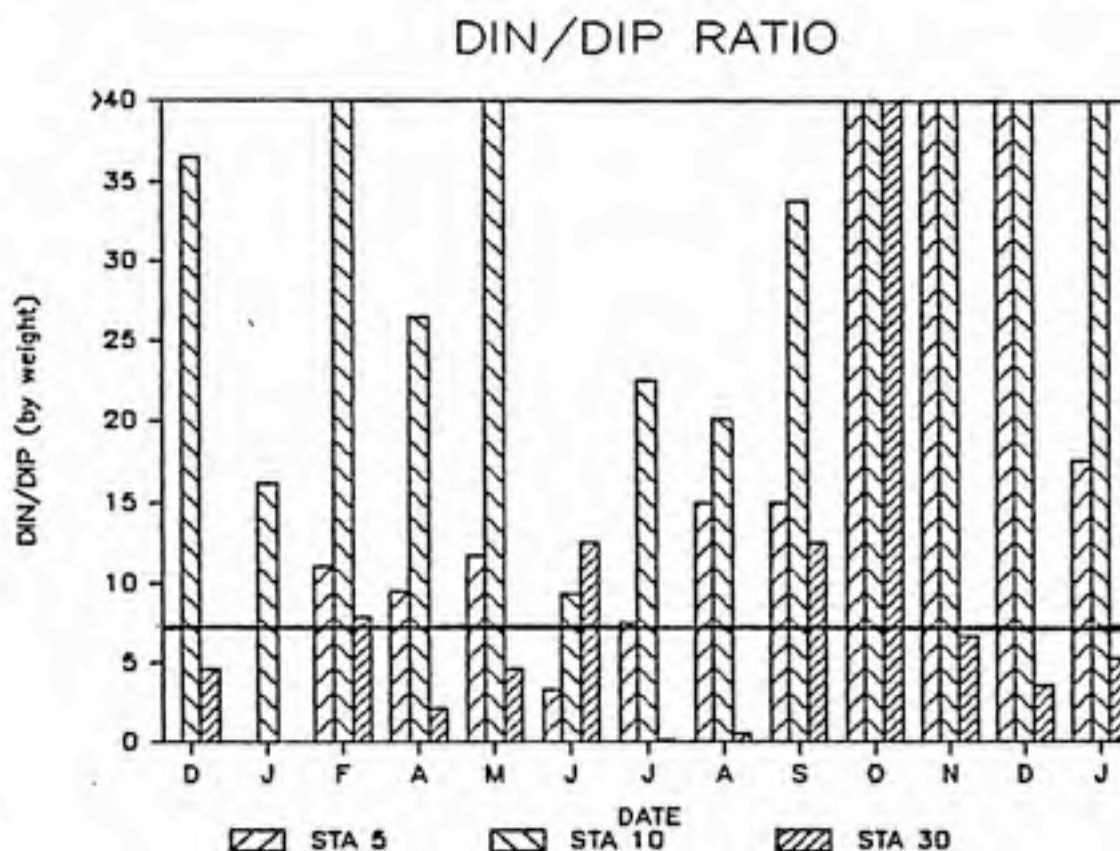


Fig. 12. Distribution of DIN:DIP ratios in surface waters of Jordan Lake, December 1983 to January 1985. Horizontal line drawn at DIN:DIP ratio of 7.2:1.

the K_s values (3.3 and 3.8 $\mu\text{g P l}^{-1}$) obtained in this study. It is therefore unlikely that phosphorus was limiting growth during these periods.

FRP TURNOVER TIMES

In Jordan Lake, short FRP turnover times often occurred during the summer and continued into the early winter of 1984 at Stations 5 and 10 (Figure 7) when FRP was scarce (Figure 6) and N:P ratios were high (Figure 12), thus implying phosphorus limitation. These turnover times, are similar to those found on other lakes during periods of summer stratification (Rigler 1973). Kuenzler and Greer (1980) observed seasonal uptake patterns similar to those found at Stations 5 and 10 but often measured shorter turnover times. Long turnover times measured throughout the sampling period at Station 30, combined with high FRP concentrations, indicate that phosphorus limitation was unlikely in this section of the lake.

ALGAL VS BACTERIAL PHOSPHORUS UPTAKE

Size fractionation of phosphorus uptake rates indicate that biotic phosphorus uptake by the small size fraction was much more important than uptake by the large size fraction (Figure 9). The dominance of phosphorus uptake rates by the small size fraction, composed mostly of bacteria, has been demonstrated in numerous studies of freshwater lakes (Rigler 1956; Lean 1973; Kuenzler and Greer 1980; Currie and Kalff 1984a).

Small particle uptake was well correlated with large particle uptake at Station 5 ($r=.88$) and Station 10 ($r=.73$) indicating that, if representative of bacterial and algal uptake respectively, increases in algal uptake were associated with increases in bacterial uptake, and that algae did not dominate uptake during periods of high biomass and phosphorus deficiency. This corresponds with the work of Hickman and Penn (1977) and Faust and Correll (1976) who found that high algal populations often had correspondingly high bacterial populations. Lean (1984) and Currie and Kalff (1984b) also found bacterial uptake rates in freshwater lakes were a relatively constant percentage of total biotic uptake throughout the year while Faust and Correll (1976) and Kuenzler and Greer (1980) found distinctive increases in the percentage of algal uptake during the warm season. Fairly constant percentages of bacterial and algal uptake rates correspond to the model proposed by Currie and Kalff (1984b) in which algal phosphorus uptake is dependent upon release of phosphorus from bacterial uptake and that the rates are consequently in phase with each other. Culture work by Fuhs (1972) supports superior algal utilization of phosphate at the low concentrations that typically occur during the summer months while Rhee (1972) suggested that bacteria are superior to algae in competition for available phosphate. Questions remain as to why bacteria can outcompete phytoplankton for the nutrient that is limiting their growth but yet, phytoplankton survive and manage to obtain phosphorus

(Currie and Kalff 1984b). Sufficient understanding of the ecological mechanisms controlling bacterial and algal competition for phosphorus utilization has not yet been achieved.

ANALYSIS OF SEPARATION OF ALGAL AND BACTERIAL UPTAKE

Samples treated with antibiotics at Station 10 exhibited significant decreases in phosphorus uptake (K_b) when compared to uptake in untreated samples (Table 2), indicating much of the uptake was procaryotic. Total eucaryotic uptake was usually significantly greater than algal "large particle" uptake (Table 2). Stone (1982) found similar results, whereas Kuenzler *et al.* (1979) and Kuenzler and Greer (1980) found that biotic P uptake in the large size fraction was somewhat greater than eucaryotic uptake. If the size fractionation scheme accurately separated bacterial uptake from algal uptake and the antibiotic was completely effective in preventing any P uptake by the procaryotic fraction, eucaryotic uptake should have been less than large particle biotic uptake since the antibiotic would likely decrease P uptake by the blue-green algae which have been found to be a significant percentage of the total algae present in Jordan Lake (Weiss and Francisco 1984). Analysis of the dynamics of the two size fractions in both the ambient and antibiotic treated samples (Table 2) showed significant uptake often remained in the small size fraction of the water samples after antibiotic treatment, indicating either the presence of eucaryotic algae in the small size

fraction and/or incomplete suppression of procaryotic P uptake. The possibility of algae passing through the 8.0 μ m filter is supported by the abundance of small, eucaryotic algae, such as *Cyclotella*, *Chlorella*, and *Chlamydomonas*, commonly found in Jordan Lake (Weiss and Francisco 1985, manuscript). These small cells often have greater relative uptake capacities than larger cells (Fuhs *et al.* 1972). Reasons for the potential ineffectiveness of the antibiotic treatment include: 1) High concentrations of suspended clays often present in the water samples were adsorbing the antibiotics or providing some type of physical protection for the bacteria. This possibility is supported by the frequent occurrence of high biotic uptake rates in the small size fraction during the winter months when concentrations of suspended clays tend to be high. 2) While gentamycin and kanamycin are broad-spectrum antibiotics and highly effective against many bacteria that are resistant to many other antibiotics (Davis *et al.* 1973), it is likely that some strains of organisms occurring in natural water samples were resistant to them and remained viable in the sample. 3) The length of incubation of the antibiotic and the dosage of antibiotic may not have been long enough to kill all bacteria before the beginning of the uptake experiments. Additionally, the accuracy of measurement of low biotic uptake rates in January, February, and March 1985, all of which had high eucaryotic uptake in the small fraction, may not be sufficient to allow for accurate fractionation of

rates. The uptake rate attributable to the large fraction often decreased after antibiotic treatment (Table 2). This phenomenon, similarly observed by Kuenzler and Greer (1980) and Kuenzler *et al* (1979), further exemplifies the inherent problems associated with using filters to separate algae from bacteria. It is likely that the large size fraction contained bacteria that were attached to algae (Paerl 1975) or to clay particles (Faust and Correll 1976), and blue-green algae, both of whose phosphorus uptake rates would be decreased by the antibiotic treatment, resulting in a decrease in the uptake of this fraction. It is also possible that the antibiotic affected the phosphorus uptake mechanism of the eucaryotic cells, however, Kuenzler *et al* (1979) found that the uptake of carbon, which does not necessarily involve the same uptake mechanism as phosphorus uptake, was not affected by the use of streptomycin or penicillin.

SUSPENDED SEDIMENT PHOSPHORUS SORPTION CAPACITY

Weiss and Francisco (1984) estimated that during 1981-82, 88% of the total nonfilterable residue and 75% of the total phosphorus that entered the lake system entered via the Haw River. Much of the phosphorus entering the Haw River arm enters over short periods during high flow conditions. For example, during a two-day period in February 1984, an extremely heavy flow of the Haw River delivered approximately 10% of the yearly total phosphorus load for all of Jordan Lake (Weiss and Francisco 1985,

manuscript). The sorption experiments performed in this study, using suspended sediments from 5 periods of high flow, showed that the sediment equilibrium phosphate concentrations (EPC) ($1-77 \text{ ug P l}^{-1}$) were always below that of the ambient stream FRP concentrations ($81-402 \text{ ug P l}^{-1}$) (Table 7). It is therefore assumed that the suspended sediments were acting as a sink for phosphate in river water during these periods of high flow. The single sorption experiment done using suspended sediments from the lake also showed the sediments to be acting as a sink. This is supported by uptake experiments in the Haw River Arm which demonstrated significant abiotic uptake of tracer (Figure 8). The apparent movement of phosphate onto the suspended sediments was observed despite considerable variability in TP, FRP, PP, TNFR, flow, and temperature. Kuenzler and Greer (1980) found EPC values ($1 \text{ to } 22 \text{ ug P l}^{-1}$) similar to those found in this study using suspended sediments obtained from University Lake. They concluded the sediments were acting as a sink for phosphate in the winter but, because of low lake FRP concentrations in the summer, were acting as a source of phosphate. However, the study of Kuenzler and Greer (1980) examined the role of suspended sediment that had already been in the lake for an undefined period of time, not of their potential effect before they entered the lake. Mayer and Gloss (1980) found EPC values of suspended sediment from the turbid Colorado River that were similar to those found in this study. However, since the FRP of the Colorado ranged from $1 \text{ to } 51 \text{ ug P l}^{-1}$, they concluded the

sediments were acting as a buffer, adsorbing P when concentrations were above the EPC and desorbing P when they are below. This concept of suspended sediments acting as a buffer for phosphate has also been proposed by, among others, Carritt and Goodgal (1954), Pomeroy *et al.* (1965), and Kunishi *et al.* (1972). Most of those, however, have much lower ambient stream FRP concentrations, which, despite similar EPC values, lead to different conclusions concerning the role of the suspended sediments.

Large differences between the EPC values of the suspended sediment and the ambient stream water FRP concentrations were evident (Table 7). The rate of adsorption of phosphate from solution onto the surface of the clay particle tends to be fairly rapid (<12 hrs) (Edzwald *et al.* 1976), and, when complete, should render a quasi-equilibrium close to the EPC. The possibility experimental design rendering erroneous EPC and ISC values must be ascertained. Studies have shown that the ionic composition, both species and concentration, of the contacting solution used in sorption experiments often affects the amount of phosphorus adsorbed (Ryden and Syers 1975; Ryden *et al.* 1977). A .01 M CaCl_2 contact solution, used in this study and commonly used in others (White and Beckett 1964; Taylor and Kunishi 1971; Meyer 1979;) has been shown to increase the amount of phosphorus adsorbed when phosphorus concentrations are greater than 100 ug l^{-1} , by 1.5 to 2.5 times than that adsorbed by the same soil when placed in

distilled water (Ryden and Syers 1975). The increased ionic strength and the presence of the calcium cation were both shown to be responsible for the increased sorption. The effect of .01 M CaCl_2 on sorption capacity was shown to be less at solution phosphorus concentrations below 100 ug P l^{-1} and decreased to essentially no difference at the EPC. It is therefore unlikely that the EPC values obtained in this study were significantly affected by the use of the .01 M CaCl_2 contact solution. The instantaneous sorption capacity (ISC) of the suspended sediment, which was usually measured at phosphorus concentrations greater than 100 ug P l^{-1} at which differences in amount P adsorbed using contact solutions of different ionic composition became significant, was likely overestimated since the ionic strength of Haw River water is probably much less than .01 M and hence, would exhibit less sorption capacity. However, since the differences in sorption capacity at different ionic compositions remain relatively constant at concentrations greater than 100 ug P l^{-1} the different ionic compositions would not greatly affect the relative differences between ISC obtained during different samplings.

The apparent lack of equilibrium may indicate that the residence time of the sediments in the water column is fairly short and consequently the sediments obtained may be of local origin. Initially they may have been from a distant source and been episodically carried down the stream. Much of the clay suspended in the water could have originated from the stream bottom due to the heavy scouring

action which is apparent at the Bynum sampling site during high flow conditions.

The instantaneous sorption capacity (ISC) (Table 7) of the sediments was significantly higher ($p=.05$) in the May samplings than in the winter samplings. This was due to both increased stream FRP values and higher absolute sorption capacity of the sediments in May. The ISC was inversely correlated with flow ($r=-.75$ $p=.10$). This was probably due to an increase in the percentage of large particles, with lower sorption capacities, that were suspended in the stream at periods of high flow.

ALGAL UTILIZATION OF SEDIMENT-BOUND PHOSPHORUS

Algal utilization of sediment-bound P ranged from 5.1 to 18.6% despite the observations from the sorption experiments that indicated the sediments were undersaturated with respect to phosphorus. The shape of the growth curves during the exponential growth phase of the algae grown with sediments was similar to those grown with a soluble phosphate source (Figure 11). Apparently the algae were able to assimilate the loosely-bound, easily desorbed sediment phosphorus rapidly, but the ensuing rate of desorption and/or algal utilization was insufficient to allow for significant additional growth. Additional studies comparing the total amount of phosphorus that can be desorbed from the sediment to the amount utilized by the algae would give additional insight to the mechanisms involved in algal utilization of sediment P. The 0.1 M NaOH

extraction of sediment-bound phosphorus was a poor estimator of AAP. This result is not surprising considering the high iron and aluminum content of Piedmont soils (Daniels *et al.* 1984) as opposed to the high calcium soils that were used to supply most of the information upon which the NaOH-AAP relationship was developed. For example, the often referred to study of Williams *et al.* (1980) found a strong correlation between NaOH extracted P and AAP in sediments in which 50% of the phosphorus was in the apatite fraction, 31% was in the nonapatite fraction, and 19% was in the organic P fraction, whereas two sediments that were poorly predicted by the NaOH extraction and consequently excluded from discussion, had nonapatite fractions of 50% and 65%. Huettl *et al.* (1979) suggest that the NaOH extraction may overestimate the AAP in soils with large quantities of iron and aluminum oxides having high capacity, low intensity sorption isotherms; characteristics which are found in the suspended sediments used in this study. Overall, the NaOH extraction has been empirically correlated to AAP using a small variety of soils with questionable methods of estimating AAP (see Lee *et al.* 1980) and may not be applicable to soils with different chemical characteristics than those upon which the correlation was based.

POSSIBLE MANAGEMENT IMPLICATIONS OF SEDIMENT STUDY RESULTS

The suspended sediment study has concentrated on the characterization of the adsorption/desorption and bioavailability of phosphorus bound to suspended sediments of

the Haw River, shortly before they reach Jordan Lake. The actual net effect of these sediments when they enter the lake is a function of many factors. The movement of the river water as it enters the lake is very important and is largely dependent on the density difference (primarily temperature difference) between the river and lake water and the rate and total volume of stream flow. Stream and estimated lake temperatures were similar on all samplings, but the lack of specific lake isopleths at the time of sampling makes conclusions regarding mixing difficult. Large particles entering the lake will tend settle out of the system fairly rapidly, but the clays can remain suspended for long periods of time. At a settling rate of 30 cm per day in calm waters (Williams et al. 1980) fine clays could remain in a 3 m photic zone for 10 days, longer if turbulent conditions exist (Reynolds 1984). Coagulation and encapsulation by algae (Faust and Corell 1975; Avnimelech 1982) may increase the settling rate. Severe restriction of the depth of the photic zone often results due to the high light scattering capacity of these clays and is an important factor when considering their overall impact.

It appears probable from the sorption experiments that these clays are scavenging phosphorus from the water column as they slowly settle towards the lake bottom. Given the fairly rapid rate of uptake of loosely-bound P, they should reach a state of quasi-equilibrium while they remain in the

trophogenic zone. The phosphorus adsorbed should be readily desorbable and highly bioavailable. However, the high AAP of these sediments is a direct result of the high lake and/or stream FRP concentrations, and if quantified at this stage, may lead to the erroneous conclusion that the suspended sediments are delivering highly available, easily desorbable phosphorus, which is contrary to the findings of this study. Immediately after high flow conditions, high ambient FRP concentrations and, depending on the degree of washout, low algal biomass, decrease the significance of the availability of sediment-bound phosphorus. However, as algal biomass and photic zone depth increase, and nutrient supplies decrease, the bioavailability of remaining clay particles becomes more important.

As the clay particles sink through the water column they may encounter anoxic conditions in the hypolimnion. The reducing conditions, which are common in Piedmont impoundments in summer months (Kuenzler and Greer 1980; Weiss and Francisco 1984), will reduce ferric hydroxides on the clay surface and release associated phosphate ions (Wetzel 1983). This phosphorus accumulated in the hypolimnion could become available to algae during periods of weak stratification and mixing of the epilimnion or during lake overturn. Upon reaching the lake bottom, sediments may be resuspended into the water column by turbulence generated by wind or boat traffic, or remain on the bottom where they still may play an important role in bottom sediment-phosphorus reactions.

The complex nature of suspended sediment-water interactions has been established. In the stream, sediment is likely acting as a sink for phosphate. Thus, it is likely important in the complexation of soluble phosphate that enters from both diffuse and point sources. Management practices focusing on the removal of sediment inputs into the stream without concurrent measures to decrease the inputs of soluble phosphorus may therefore result in negligible changes in soluble P concentrations delivered to the lake and may actually increase the soluble P delivered due the lack of complexation. Such a reduction may also considerably increase the depth of the photic zone in the Haw River arm of the lake where it is likely that light is often one of the limiting factors. This could possibly result in a considerable increase of algal biomass in the Haw River arm.

CONCLUSIONS

- 1) Total P concentrations were high throughout the lake, averaging 121 $\mu\text{g l}^{-1}$. Highest concentrations occurred during the winter months when large amounts of suspended sediments were present in the lake. Filterable reactive P concentrations were high in the winter at all stations but often decreased to 1 $\mu\text{g l}^{-1}$ or less during the warm months in the New Hope River arm of the lake.
- 2) Nitrogen-to-phosphorus ratios indicative of P-limitation were observed in the New Hope River arm during the warm months. High nutrient concentrations throughout the year in the New River arm and during the cool months in the New Hope River arm indicate N or P limitation was unlikely during these periods.
- 3) FRP turnover times were long during the cool months at all stations, but greatly decreased in the New Hope River arm of the lake during the warm months when FRP concentrations decreased and algal biomass increased. Short turnover times, high N:P ratios, and low FRP concentrations, indicate P limitation in the New Hope River arm of the lake during the warm months.
- 4) The small size fraction dominated biotic uptake (Ave = 87.6%) throughout the year at all stations. At Station 10, procaryotic organisms were responsible for most of the uptake in the small fraction.

- 5) Abiotic uptake often comprised most of the uptake at Stations 5 and 30 during the winter months when large quantities of suspended sediments were present in the water column. Abiotic uptake at Station 10 was usually low.
- 6) Suspended sediments obtained from the Haw River during periods of high flow were found to be removing phosphorus from solution.
- 7) Algae grown with suspended sediment-bound phosphorus under conditions otherwise optimal for growth were able to utilize a relatively small amount (mean = 11.3%) of the sediment-bound phosphorus.

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