Final Report

Assessment of internal nutrient loading to Crowley Lake, Mono County

(SWRCB # 00-196-160-0)

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INTRODUCTION

The most common impairment of surface waters in the United States is eutrophication caused by excessive inputs of phosphorus (P) and nitrogen (N). Impaired waters are defined as those that are not suitable for designated uses such as drinking, irrigation, industry, recreation, or fishing. Crowley Lake (Long Valley Reservoir) is a valuable aquatic resource identified as impaired by nutrients by the CA Water Resources Control Board. The lake is eutrophic and is characterized by an ample supply of nutrients and significant summer algal blooms (EPA 1978, Melack and Lesack 1982). Adverse impacts of increased eutrophication at Crowley Lake have included de-oxygenation of the hypolimnion and downstream fish kills (Milliron 1997), and decreased water quality as indicated by taste, odor, and large areas of floating algal mats.

The California Regional Water Quality Control Board, Lahontan Region (RWQCB), is the State agency responsible for protection of water quality within the Lahontan Region of California. The jurisdiction of the RWQCB extends from the Oregon border to the northern Mojave Desert and includes all of California east of the Sierra crest. The RWQCB implements the goals of the federal Clean Water Act to restore and maintain the physical, chemical, and biological integrity of the nation's waters. This includes the development of Total Maximum Daily Loads (TMDLs) for water bodies that do not currently meet State standards. Crowley Lake (Long Valley Reservoir) is listed as impaired pursuant to Section 303(d) of the Clean Water Act, and the RWQCB plans to develop TMDLs for the reservoir. In order to develop TMDLs, the RWQCB will need information on nutrient inputs and dynamics.

A 2-yr study of nutrient loading to Crowley Lake found high nitrogen and phosphorus loading rates to Crowley Lake (California Water Control Board Award #9-175-256-0; Restoration to Riparian Habitat and Assessment of Riparian Corridor Fencing and Other Watershed Best Management Practices on Nutrient Load and Eutrophication of Crowley Lake, California). Measured phosphorus inputs were approximately equal to reservoir outflows suggesting very little retention by lake sediments. Nitrogen outflows from the reservoir were 3-4 times the measured inputs in precipitation and tributary inflows, suggesting the sediments or nitrogen fixation are significant sources of nitrogen. In preparation for designing and implementing TMDL's for Crowley Lake and its tributaries, the Lahontan RWQCB awarded this supplemental research grant to further assess nutrient budgets of Crowley Lake. The goals of this research were to measure summer changes in water quality and other ecological variables within Crowley Lake and to assess internal nutrient loading to the reservoir through measurements of pelagic nitrogen fixation and sediment-water fluxes. Here, we present the results of this study.

ACKNOWLEDGEMENTS

This research was funded under a California Regional Water Quality Control Board (Lahontan Region) contract (#00-196-160-0) awarded to the University of California (co-PIs Robert Jellison and John M. Melack, University of California, Santa Barbara (UCSB)). Field and laboratory work were performed at the Sierra Nevada

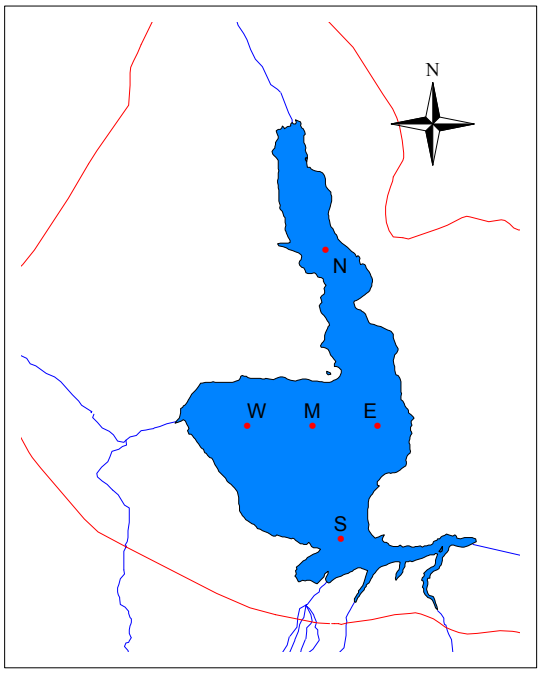
Aquatic Research Laboratory (SNARL) of the UC Natural Reserve System. Particulate carbon and nitrogen analyses were performed by the Marine Science Analytical Laboratory, UCSB. We thank the Los Angeles Department of Water and Power for providing access to the lake and Wayne Hopper (LADWP, Bishop) for providing Long Valley hydrological data.

METHODS

Field Sampling

Eight 2-day lakewide surveys were conducted at approximately biweekly intervals from 20 June 2002 to 25 September 2002 to assess limnological conditions and collect pelagic plankton samples for a variety of analyses including the measurement of nitrogen fixation rates. Five pelagic stations were chosen to represent the major sectors of the lake (Fig. 1). The N station is located midway up the long narrow portion of the lake, is relatively shallow (5-6 m), and is influenced by inflows from the Owens River, the dominant inflow and source of nutrients (see Jellison et al 2003, SWRCB #9-175-256-0). The W, M, and E stations lie along a west-east transect at the widest portion of the lake and are ~8, 15, and 18 m deep, respectively. The W station is located near the inflows from McGee Creek the second largest inflow and source of nutrients. The S station is in the central portion of the deep (23-24 m) southern portion of the lake. Although we had intended to begin the biweekly sampling immediately following our 3 April 2002 lakewide survey conducted as part of a cooperative nutrient loading study with the Los Angeles Department of Water and Power (SWRCB #9-175-256-0), they denied permission to continue sampling until review of the supplemental research included in this contract. Thus we were not able to sample again until mid-June. However, to assess the changes from several weeks after ice-off to the first sampling in mid-June we include the results of the 3 April survey in this report.

Fig. 1 Pelagic sampling stations on Crowley Lake



On the first day of the survey, temperature, dissolved oxygen, and nutrient profiles were determined to assess hypolimnetic nutrient accumulation and vertical mixing. Temperature was collected with a high-precision, conductivity-temperature-depth profiler (CTD) (Seabird Electronics, Model Seacat 19) at the two deep stations (S and E)(Fig. 1) and at two additional deep stations (one near the dam and one approximately 2 km west of station E. The two additional stations were included to lessen errors in estimating changes in the lakewide heat budget. Dissolved oxygen concentration was measured at the S and E stations at 1-m

intervals with a Yellow Springs Instruments temperature-oxygen meter (YSI, model 58) and probe (YSI, model 5739). The oxygen meter was calibrated in water-saturated air prior to each use. At the E and S stations, samples for the determination of ammonia (NH_4) and soluble reactive phosphorus (SRP) were collected with a Van Dorn water sampler at 1-m intervals from 11 m to near the bottom. Water samples were immediately filtered and kept cool and in the dark during transport to the laboratory.

For the purposes of this project, DIW is used to refer to filtered, deionized, reverse osmosis treated water. This is our primary washing and rinse water with a specific conductance of approximately 5 $\mu S\ cm^{-1}$. For reagent and standard preparation this water is further polished by ion exchange to a specific conductance of approximately 0.5 $\mu S\ cm^{-1}$. All bottles used in water sampling were soaked in deionized water and then rinsed 3 times with DIW. Sample collection bottles were rinsed with 10% HCl before DIW soaking and rinsing. Filtered samples were filtered in the field with plastic syringes fitted with Gelman A/E filters which were rinsed with at least 150 ml of DIW or sample water.

On the second day of each survey a suite of physical, chemical, and biological characteristics were measured at each of the five pelagic stations (Fig. 1) and water samples collected for determination of nitrogen fixation rates. Temperature profiles were taken with a high-precision, conductivity-temperature-depth profiler (CTD) (Seabird Electronics, Model Seacat 19) equipped with a cosine-corrected photosynthetically available radiation (PAR) sensor (LiCor 191S). Both CTD and LiCor sensors are calibrated annually by the manufacturer. The CTD records at 0.5 s intervals and was deployed by hand-lowering at ~0.2 m s^{-1}. On two dates when the CTD was not available (20 June and 1 July) temperature was measured at 1-m intervals with a Yellow Springs Instruments temperature-oxygen meter (YSI, model 58) and probe (YSI, model 5739). Transparency was measured at all five stations with a 20-cm white Secchi disk.

A 5-m integrated sample was collected at all five stations for analysis of water quality, phytoplankton, and nitrogen fixation. Samples were collected with a one-inch (inner diameter) Tygon tube (#R3603) lowered into the water and retrieved by raising the lower end to the surface before draining the tube. Duplicate 60-ml subsamples for the determination of ammonia (NH_4), nitrate (NO_3) and soluble reactive phosphorus (SRP) were immediately filtered. Duplicate 60-ml subsamples for total nitrogen (TN) and total phosphorus (TP) were determined from unfiltered samples. Particulate nitrogen (PN), particulate carbon (PC), particulate phosphorus (PP) and chlorophyll a (Chl a) were determined from samples filtered in the laboratory (25 mm, ashed Whatman glass microfiber filters for PN, PC and PP; 47 mm Whatman glass microfiber filters for Chl a). Additional, duplicate 60-ml subsamples were preserved with formalin for phytoplankton identification and quantification. The remaining portion (1-2 liter) was used to assess nitrogenase activity via acetylene reduction rates during incubations conducted immediately on return to the laboratory.

Sediments were collected during autumn 2001 for use in laboratory benthic chamber experiments. Grab samples were obtained with an Ekman dredge. Gravity

corers were also tried, but sediments could not be collected in the corers' tubes because of their high water content.

Four additional freeze cores were collected in September 2002 to provide information on sediment ammonia flux and historical changes in burial of carbon, nitrogen, and phosphorus. A single core was collected at the E and M stations and two cores were collected at the S station. Cores were collected with a 2 m, rectangular freeze-corer (see Crusius and Anderson 1991 for detailed description of method). The corer was quickly lowered to just above the sediment surface and then lowered gently into the sediments. The coring device was held vertical with the retrieval line during the first several minutes of freezing to assure it remained upright and then line tension relaxed to prevent sideways pressure due to boat movements during the remaining 12 minutes of freezing. The 15-min freezing period resulted in a flat frozen slab approximately 20 cm wide and 2.5 cm thick. Length varied depending on the sediment characteristics at the site.

Analytical Procedures and Analysis

Water quality and plankton enumeration

Samples for the determination of NH_4 and SRP were analyzed on the same day as collection. Remaining filtered field samples were frozen and analyzed for NO_3 within two months of collection. Unfiltered samples for TN and TP were frozen upon return to the laboratory and kept frozen until analysis within two months of collection. Filtrations for Chla, PP, PC, and PN were performed in the laboratory on the same day as collection. Chla filters were frozen until analysis within two weeks. PP, PC, and PN were dried at 50°C for 48 h in a Fisher Scientific Isotemp oven and stored in a desiccator until shipped to the Marine Science Institute Analytical Laboratory, UCSB. Analytical methods and detection limits are listed in Table 1. With the exception of PC and PN above analyses were performed at the SNARL.

The plankton communities were characterized through species identification and enumeration. For phytoplankton analysis, 10 ml of well mixed 5-meter integrated sample was allowed to settle for 24 h in a 10 ml Hydro-Bios Utermohl chamber. A few drops of Lugol's Solution were added to aid in settling. The samples were analyzed on a Carl Zeiss inverted microscope at 16X magnification. To ensure capture of rare species, the entire sample was scanned and each organism identified to genus. For a more accurate estimate of the larger species, 60 ml was analyzed under a Wild Heerbrugg dissecting scope at 12X magnification. A portion of the total settling area was counted and extrapolated to the total surface area (Wetzel and Likens 1979). Biovolume was calculated using the methods of Hillebrand et al. (1999). Identification keys included Prescott, 1978; Dillard, 1999; and Canter-Lund and Lund (1995).

Table 1 Analytical chemistry methods

Species	Method	Reference:	Detection Limit (μM)
NH_3 + NH_4^+	phenol-hypochlorite colorimetric	Strickland and Parsons, 1972 Wetzel and Likens, 1991	0.30
SRP	phospho-molybdate colorimetric	Strickland and Parsons, 1972 Wetzel and Likens, 1991	0.06
NO_3	Cd reduction followed by azo dye colorimetric	Strickland and Parsons, 1972 Wetzel and Likens, 1991	0.20
TP/PP	Valderrama (oxidation/phospho-molybdate)	Valderrama, 1981	0.4
TN	Valderrama (oxidation/Cd reduction/azo dye)	Valderrama, 1981	0.4
PC, PN	Automated Organic Elemental Analyzer (Model CEC440HA), Dumas combustion method.	Marine Science Analytical Laboratory UCSB	
As	As(V) As(III) reduction/phospho-molybdate	Johnson, 1971	0.02

Chlorophyll *a* concentrations were also used to characterize the phytoplankton. Chlorophyll *a* was extracted in 90% ethanol using the method of Sartory and Grobbelaar (1984). Following clarification by centrifugation, absorption was measured at 750 and 665 nm on a spectrophotometer (Milton Roy, model Spectronics 301), calibrated once a year by Milton Roy Company. The sample was then acidified in the cuvette, and absorption was again determined at the same wavelengths to correct for phaeopigments. During periods of low phytoplankton concentrations (<5 μg chl *a* l^{-1}), the fluorescence of extracted pigments was measured on a fluorometer (Sequoia-Turner, model 450) which was calibrated against the spectrophotometer using fresh lettuce.

Sediment Chambers

Two types of sediment chambers were used in laboratory experiments designed to measure sediment-water exchange.

Bench-top chambers consisted of 3 inch diameter, 15 inch long gray PVC pipes threaded on one end. The end without threads was covered by a black rubber cap. The threaded end had a PVC cap with 3 holes that allowed for sampling, venting, and bubbler tubing to be inserted. Nitrogen or air was bubbled into the chambers to create anoxic or oxic conditions.

Sediments collected by Ekman dredge were transported to the laboratory in a 5 gallon bucket, homogenized and then settled, which separated the sediment into a heavier portion and a suspended portion. To each chamber, 400 ml of heavier sediment, 400 ml of suspended sediment, and 400 ml of bottom water were added. All six chambers were placed in a cold room at 15º C, which was approximately the temperature of the bottom water at the time of the experiments. Samples for nutrient chemistry were taken each day for the next 4 days. 50 ml of water from 2 cm above the sediment surface was obtained

using a 100 ml plastic syringe, fitted with a glass fiber filter and Tygon tubing which was threaded through the chamber caps. Samples were placed directly into acid washed plastic test tubes to be used for analysis. Ammonia and SRP determinations were done immediately. Nitrate samples were immediately frozen for analysis later.

Benthic chambers are clear Plexiglas chambers approximately 20 cm high and 50 cm in diameter. They are equipped with pumps that circulate water within the chambers. The chambers were mounted to plywood to permit their use in the laboratory. About 3 cm sediment and a half and half mixture of lake water and deionized water were placed in the chambers. The sediments were homogenized by circulating the water at sufficienly high speed and then settled for two days. Samples for ammonium, SRP and nitrate were collected every hour for eight hours using the technique described above.

Laboratory experiments designed to determine sediment exchange rates indicated that the sediments were a sink over the course of several days, but a very small source over several hours to one day. Although the bench-top chambers had a range of dissolved oxygen concentrations (Table 2), the changes over the experiments were similar in all six chambers. Nitrate concentrations varied little over time (0.02 mg l^{-1} to 0.18 mg l^{-1}), and decreased after an initial increase on day one (Fig. 2). Ammonium decreased slightly from about 1.6 mg l^{-1} to about 1.4 mg l^{-1} (Fig. 3). Phosphate decreased to low concentrations after a very slight increase (Fig. 4).

In the benthic chambers, nitrate levels rose from about 0.1 mg l^{-1} to approximately 0.15 mg l^{-1} between hours four and five, indicating some release of nitrate from the sediment (Fig. 5). Ammonium remained constant over the eight hour experiments. A very small increase in phosphate concentration was detected in the benthic chamber waters over eight hours (Fig. 6).

Table 2 Concentration of dissolved oxygen (DO) in bench-top chambers

Chamber		DO reading (mg/L)
1	N_2 bubbles	0.25
2	N_2 bubbles	0.25
3	No bubbles	0.26
4	No bubbles	3.03
5	Air bubbles	5.84
6	Air bubbles	6.15

Following these preliminary analyses, we decided further experiments were not warranted given the high porosity of the sediment and difficulty in interpreting these types of measurements. Therefore we focused our efforts on collecting and analyzing several freeze cores.

Fig. 2 Nitrate changes in bench-top chambers.

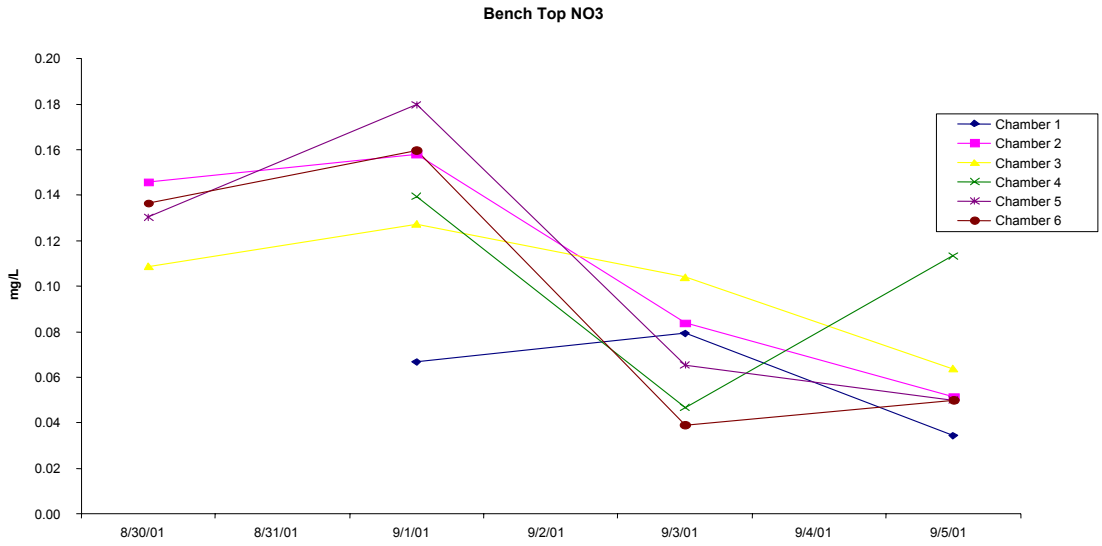


Fig. 3 Ammonia changes in bench-top chambers.

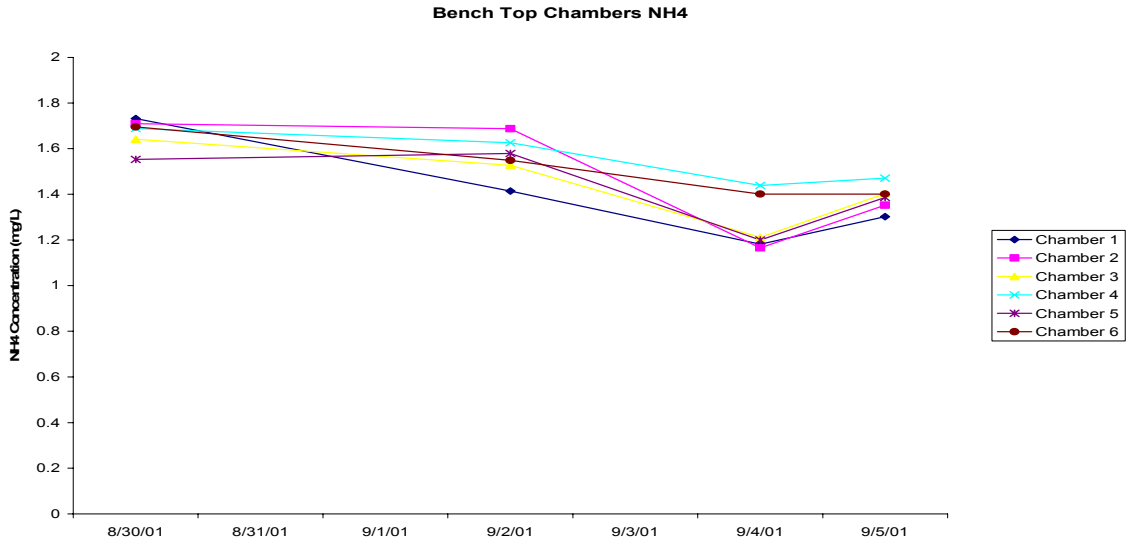


Fig. 4 Phosphate changes in bench-top chambers.

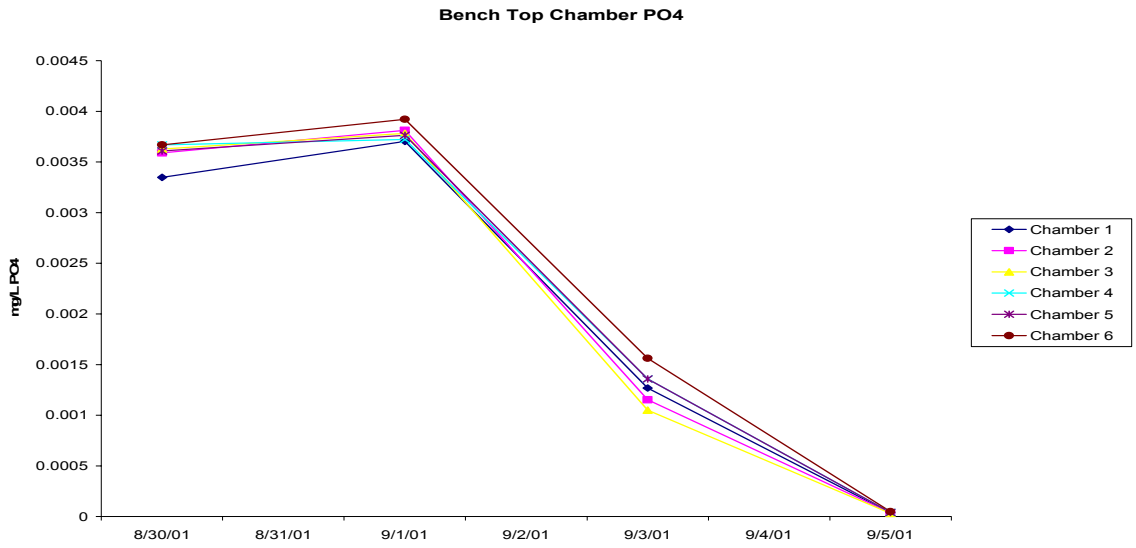


Fig. 5 Changes in nitrate in benthic chamber with slow circulation.

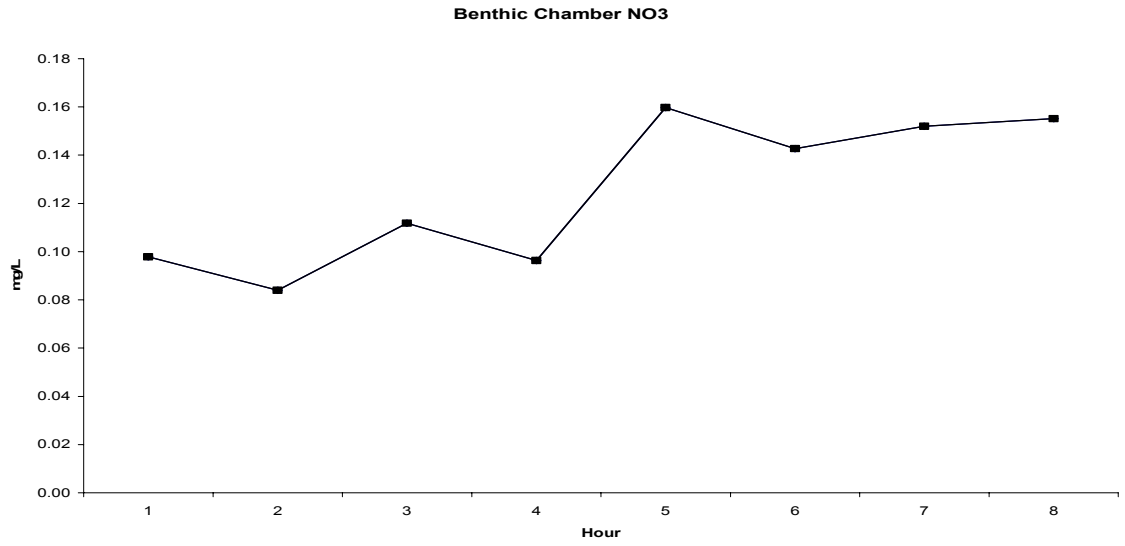
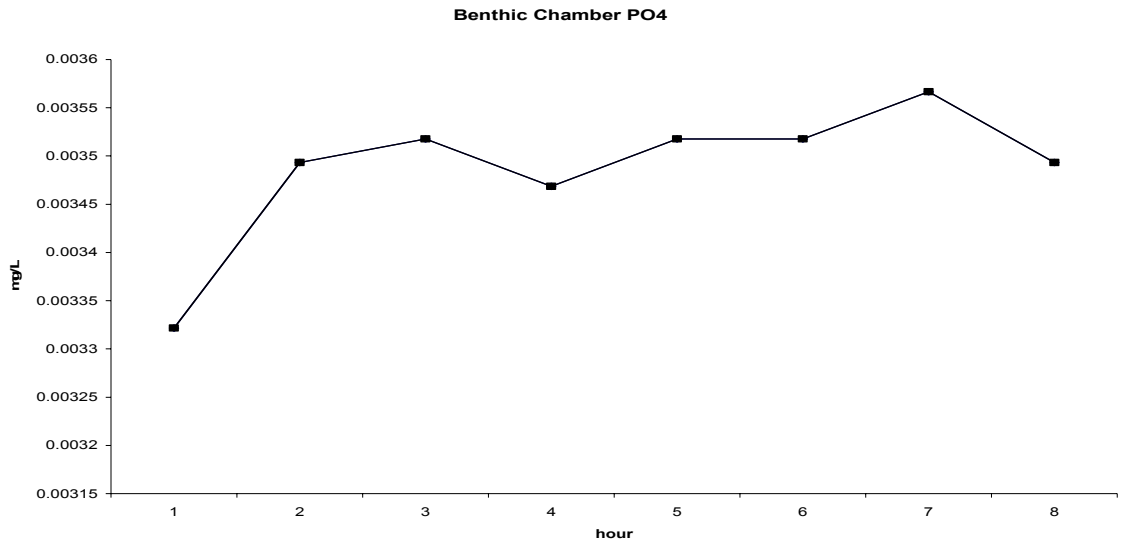


Fig. 6 Phosphate changes in benthic chambers.



Sediment freeze cores

Preparation of the cores took place in the SNARL cold room at a temperature of 5°C. The cores were allowed to warm enough to facilitate sectioning without losing their integrity. Each core was sectioned lengthwise and one half archived. The remaining section was subsampled each centimeter for the first upper 10 cm of the core and every 2nd cm to the bottom end. Each subsample consisted of four replicate 1-cm^2 sections of one cm depth.

Two subsamples were analyzed for porewater NH_4. Samples were centrifuged and supernatant removed, diluted 50X, and analyzed with the phenol-hypochlorite colorimetric method. An 800 μm standard was prepared and diluted as a quality control measure of dilution accuracy.

The remaining two subsamples were placed in acid-washed, rinsed, dried, and pre-weighed glass scintillation vials and used for determination of porosity, non-apatite inorganic phosphorus (NAI-P) dried, TP, TC, and TN. They were weighed immediately (wet weight) and then dried at approximately 50°C to a constant weight. Porosity was calculated as

$$\varphi = \frac{W_{wet}}{\left(W_{wet} + W_{dry} \times D\right)_w}$$, where D, the bulk density, was assumed to be 2.2 g cm^{-3}.

Note that while bulk densities for various sediment constituents range from 1.4 for humus to 5.0 for various heavy minerals average bulk densities are typically 2.2 – 2.8 and even assuming bulk densities as low or high as 1.4 or 3.0 would have little effect on our calculations due to the high porosity of these sediments.

The dried sediment in each vial was then ground by hand using a small metal spatula and portioned for NAI-P, TP, PC, and PN analysis.

NAI-P was measured using the method of Schelske and Hodell (1995). Sediment was weighed and leached with 5.0 ml of 0.1 N NaOH for a 17 h period. The tubes were centrifuged and the supernatant analyzed for SRP using the ammonium molybdate method corrected for As interference with the sodium metabisulfite and sodium thiosulfate method. Internal standards of 5 and 10 μm PO_4 were prepared on three random samples during each analysis.

A second portion of dry sediment was weighed, diluted with 25 ml DDW and digested with persulfate solution at 250 ^{0}F for 30 minutes in an electric pressure steam sterilizer. Samples and standards were then analyzed for TP using the ammonium molybdate method as above.

The remaining dry sediment was reweighed and acidified using 1N HCl. The samples were re-dried and weighed prior to shipping to the Marine Science Institute Analytical Laboratory, UCSB, for PC and PN analysis.

Pore-water concentration profiles of ammonia were modeled after Klump and Martens (1981) to determine concentration gradients at the sediment-water interface. Exponential equations were fit by least-squares minimization from the bottom of the core to the sediment-water interface (bottom water concentrations) with

$$C_z = (C_\infty - C_0)(-e^{-az}) + C_\infty$$

where C_z, C_∞, and C_0 are the concentrations at depth z, infinity, and the sediment-water interface. C_∞ and a were calculated; C_0 was fixed at the bottom-water concentration. The gradients were calculated at z = 0. The flux of ammonia out of the sediment was then calculated from the best-fit concentration gradient with Fick's first law of diffusion:

$$Flux = -\varphi D \frac{\partial C}{\partial z}\Big|_z$$

where φ is porosity at the interface, D is the sediment diffusion coefficient corrected for tortuosity, θ, where

$D = \frac{D_f}{\theta^2}$ and $\theta^2 \approx 1 - 2\ln(\varphi)$ (Boudreau 1996).

Nitrogen fixation

Nitrogen fixation was measured by the acetylene reduction method (Flett et al. 1976). Experiments consisted of four light treatments for 5-meter integrated samples from each of the five Crowley Lake sampling stations. Light treatments consisted of two placed in direct sunlight in Convict Creek on the property of the Sierra Nevada Aquatic

Research Laboratory, one wrapped in a neutral density gray screen, and two in a laboratory water bath, one under artificial fluorescent lights (90-100 μE m^{-2} s^{-1}) and one placed in dark bag. Temperature was recorded in both the stream and water bath. Photosynthetic available radiation (PAR) was recorded daily by sensors located on the SNARL property. Hourly values were averaged over the incubation period. PAR measurements were also taken in the stream and water bath during each experiment using a Licor 185 Light meter.

Incubations consisted of 50 ml of lake water in a 60 ml serum bottle. Five ml of air was replaced with 5 ml of acetylene gas. Bottles were shaken for 30 seconds and incubated for a 4 h period. After the incubation period, 1 ml of gas was removed and injected into a Shimadzu GC-8A gas chromatograph. A set of ethylene gas standards (prepared from Scott Specialty Gases (Scotty mix 849 – 1% (by mole) ethylene in nitrogen) were run at the beginning of each experiment to provide a standard curve. A ratio of 3 moles of acetylene reduction (ethylene produced) to one mole of N_2 fixed was assumed (Flett et al. 1976). Note rates are reported in units of moles of N (not N_2) fixed for comparison with loading fluxes.

Lakewide estimates of planktonic nitrogen fixation were made using a numerical interpolative model which combines hourly insolation, in situ PAR attenuation, and nitrogen fixation versus light rates. We assumed the fixation versus light intensity rates measured in the 5-m integrated samples were representative of the euphotic zone. As no significant persistent differences in nitrogen fixation rates were noted among the five lakewide stations, a lakewide average for each day was used. The fixation versus light intensity relationship was represented as an initial slope and a maximum rate. Further refinement was not warranted based on the variability of the rate measurements. Hourly insolation values from SNARL 7 km west of the center of the lake and a lakewide average of in situ PAR attenuation was used.

Hypolimnetic Nutrient Accumulation and Eddy Diffusivity

Lakewide hypolimnetic nutrient (SRP and NH_4) accumulation was calculated by averaging data from the two deep stations and then linearly interpolating to 0.25 m from 11 m to the bottom. The deepest sample at the shallower E station often was elevated relative to the S station most likely reflecting close proximity to the sediments and was not used. A volume-weighted sum was then calculated based on hypsographic data provided by the Los Angeles Dept. of Water and Power.

Eddy conductivities were calculated using the flux-gradient heat method modified for solar heating (Jassby and Powell 1975).

$$K_z = -\frac{1}{\frac{\delta\theta}{\delta z}}\left[\frac{1}{A_z}\frac{d}{dt}\int_z^{z_m} A_u\theta_u\,du - \frac{1}{\rho c}R_z\right]$$

where K_z is the coefficient of vertical eddy conductivity at depth z, z_m is the maximum depth of the lake, A_z is area at depth z, u and z are depths positive downwards, R_z is irradiance at depth z, θ is temperature, ρ is density, c is thermal capacity, and t is time. The temperature gradients were estimated as 1 m central differences. Depths, areas, and volumes were changed to correspond to changes in the lake level. The heat integral was evaluated at 1 m intervals using lakewide mean temperatures and area-capacity curves. Eddy diffusivities were assumed equal to eddy conductivities after being corrected for molecular conductivity (0.13×10^{-6} m^{-2} s^{-1}; Chemistry and physics handbook 1977, Table E-11).

Solar heating was estimated from continuous measurements of incident PAR, calculated albedoes, and light attenuation within the water column. Comparison of measurements with an Eppley pyranometer (285–2,800 nm) and PAR (400–700 nm) collected at SNARL, indicated PAR comprised 44.6% of the total solar irradiance assuming a conversion of 4.57 μEinst = 1 joule (McCree 1972). This is close to findings in other studies (45%, Gates 1966; 41%, Jassby and Powell 1975). The PAR data were converted to total solar input using this ratio. Albedoes were calculated assuming all radiation was direct; this assumption introduces only a small error (Jassby and Powell 1975). Attenuation within the water column was divided into seven wavelength bands. Visible light attenuation was measured as PAR attenuation. The attenuation of infra-red light for five intervals was obtained from Hale and Querry (1973): 1.1 m-1, 700–800 nm; 3.4 m-1, 825–900 nm; 26 m-1, 925–1,000 nm; 870 m-1, 1,200–1,800 nm; and 7,800 m-1 from 2,000–2,400 nm. The effect of solar heating at the chosen depth of 12 m was insignificant (<1%) throughout the period.

Results

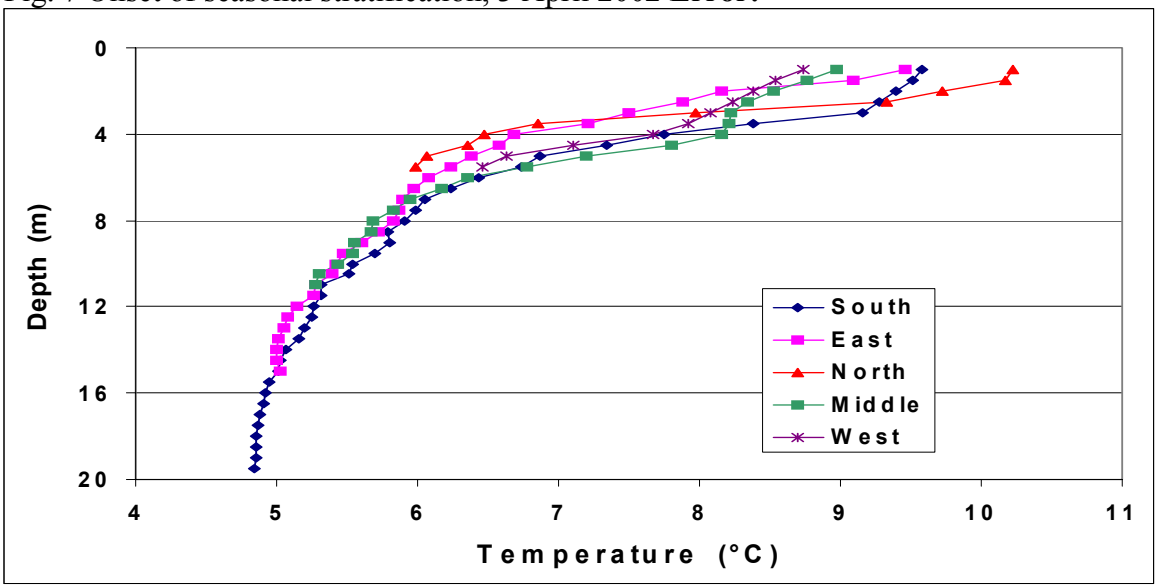
Eight 2-day lakewide surveys were conducted at approximately biweekly intervals from 20 June 2002 to 25 September 2002 to assess limnological conditions and collect pelagic plankton samples for a variety of analyses and the measurement of nitrogen fixation rates. On the first day of each survey, temperature, dissolved oxygen, and nutrient profiles were determined to assess hypolimnetic nutrient accumulation and vertical mixing. On the second day of each survey a suite of physical, chemical, and biological characteristics were measured at each of the five pelagic stations and water samples collected for determination of nitrogenase activity via the acetylene reduction measurements in laboratory incubations. To view the seasonal development, we include the results of a survey conducted 3 April 2002 a couple weeks after ice-off. Also, four sediment cores were collected in September 2002 to provide information on sediment ammonia flux and historical changes in burial of carbon, nitrogen, and phosphorus.

Physical, chemical, and phytoplankton conditions in Crowley Lake during 2002

Seasonal Thermal stratification

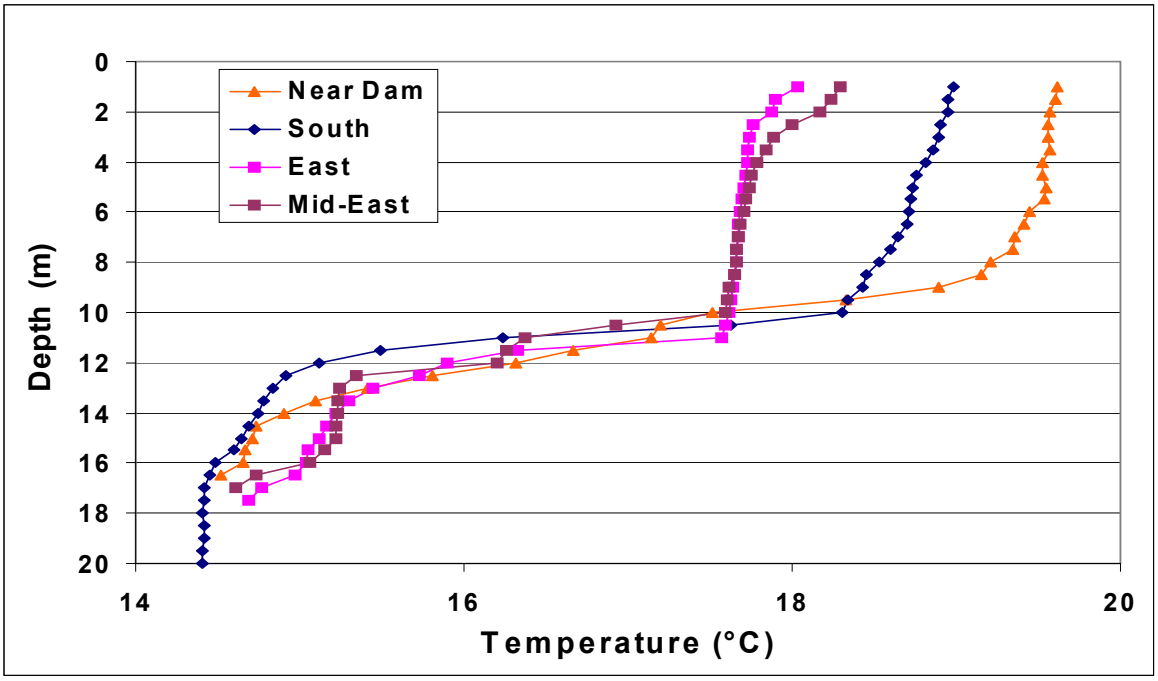
A lakewide survey was conducted on 3 April 2002 following ice-off in mid-March as part of a separate cooperative study of nutrient loading with the City of Los Angeles (SWRCB #9-175-256-0). Seasonal thermal stratification had already been initiated with temperatures nearly isothermal below 12 m at near 5°C, increasing gradually to 6°C between 12 and 8 m and then more rapidly to 8-10°C in the upper water column (<4 m) (Fig. 7). Near surface waters were slightly warmer at the N station which is influenced by the Owens River, and slightly cooler at the W and M stations which are more strongly influenced by McGee Creek inflows.

Fig. 7 Onset of seasonal stratification, 3 April 2002 **Error!**



By 19 June, the epilimnion (upper mixed layer) had warmed to 17.5-20°C while the hypolimnion had increased to 14-15°C (Fig. 8). There was a significant longitudinal gradient in which epilimnetic waters warmed from north to south. Epilimnetic temperatures were ~18.0°C at the E and Mideast station, ~18.9°C at the S station, and ~19.6°C at the Dam station. There was a pronounced thermocline between 10 and 12 m at all four stations.

Fig. 8 Early summer thermal stratification in Crowley Lake.



Epilimnetic temperatures continued to increase, reaching their annual maximum of 21-22°C in mid-August. The pronounced thermocline observed at 10-12 m was mixed downward through the summer and by mid-August only a gradual thermal gradient from 21-22°C near the surface to 18-19°C near the bottom existed indicating active mixing through this period (Fig. 9). Further mixing and seasonal cooling of the upper water column resulted in less than 1°C gradient between upper and lower water temperatures during September when the field sampling ended.

Seasonal and lakewide variation in dissolved oxygen

Dissolved oxygen concentrations showed marked seasonal variation typical of eutrophic temperate lakes (Fig. 10). During the 3 April 2002 survey the entire water column was well-oxygenated at the S station with values only ranging from 9.5 to 10.5 mg O_2 l^{-1}. By the next sampling on 19 June, the hypolimnion was anoxic (<1 mg O_2 l^{-1}). The hypolimnion remained anoxic until late August and was not fully oxygenated until the 10 September survey.

Fig. 9 Annual thermal stratification in Crowley Lake, 2002 (S Station). Tic marks show sample dates and isotherms in °C.

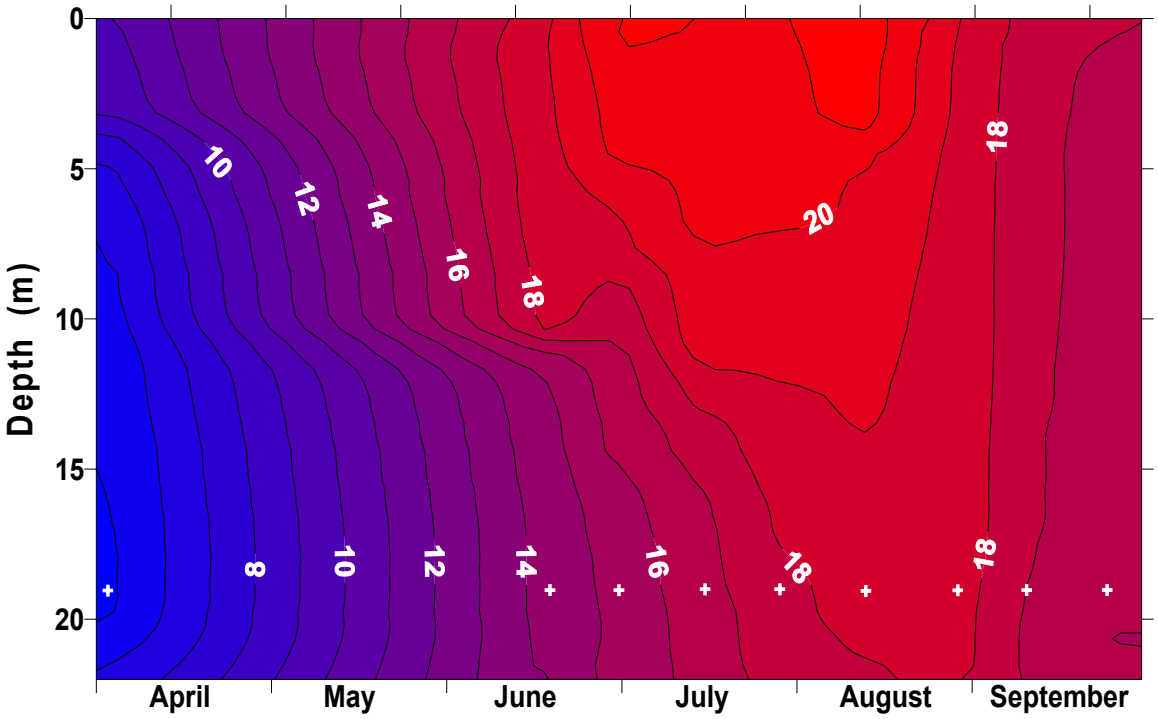
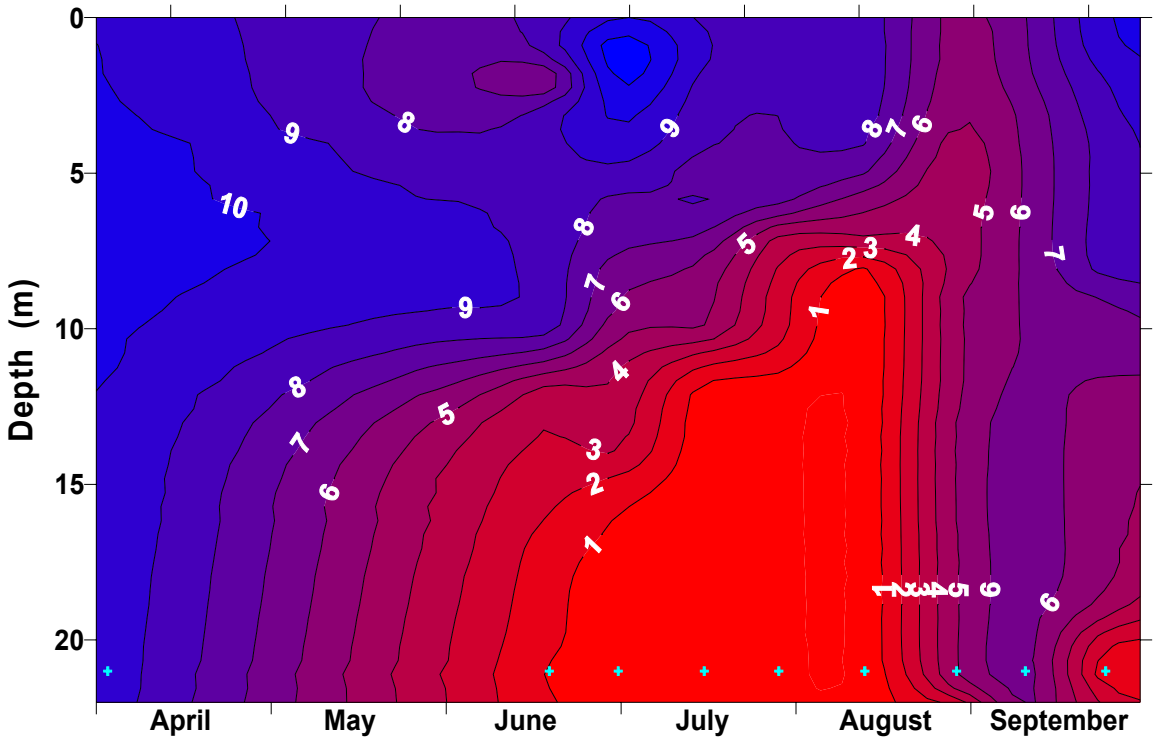
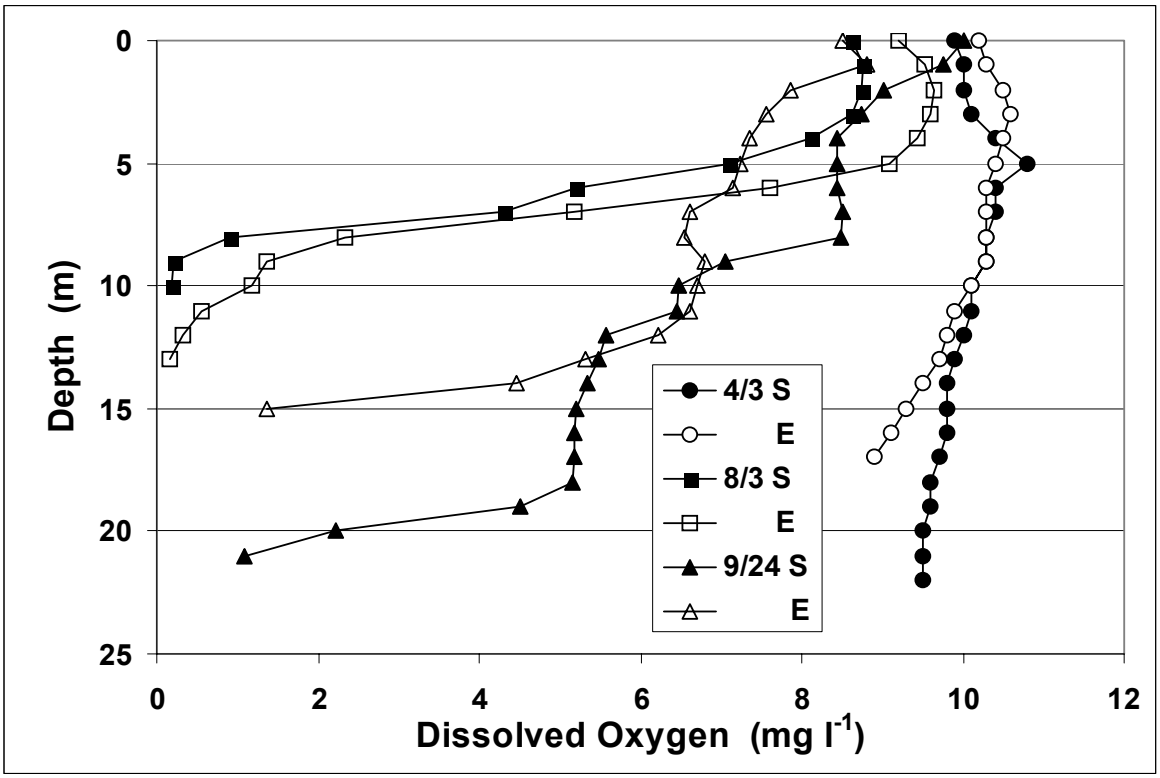


Fig. 10 Annual variation in dissolved oxygen (mg l^{-1}) in Crowley Lake, 2002 (S Station). Tic marks show sample dates.



Dissolved oxygen profiles were generally similar between the two deep stations, S and E (Fig. 11) except late in the year. On these dates, the decline in dissolved oxygen concentrations higher in the water column is most likely due to proximity with the bottom at 15 m. The E station is significantly shallower than the S station and the reservoir was significantly drawn down in autumn. Above 15 m, the dissolved oxygen profiles at the two stations were roughly similar.

Fig. 11 Comparison of dissolved oxygen at S and E stations during 2002.



Seasonal and lakewide variation in transparency

Secchi depth provides a readily collected and widely used measure of transparency (Fig. 12). On 3 April 2002, Secchi depth was 3.7-4.2 at N, W, M, and E stations. At the S station it was 6.0 m. By the next sampling on 19 June, the transparency at the S station had decreased to 4.0 m and the other 4 stations ranged from 3.2-3.8 m. Transparencies decreased further to 1.5-2.6 through July before increasing to near 4 m at all but the N station in mid-August. Transparencies then declined to 1.8-2.3 during September during an autumn bloom. Thus, the general trend in transparency reflects the presence of spring and autumn algal blooms. The overall mean transparency from 19 June through 24 September was 2.7 m (1 SE, 0.1; n, 56).

Photosynthetically available radiation (400-700nm) was also collected at all five stations. The euphotic zone depth as defined by the mean lakewide 1% light level varied from 4.5 to 9.5 m through the season (Fig. 13). On 3 April 2002, the 1% light level was between 8.5-9.5 m and depth attenuation was fairly uniform at the five stations (Fig. 14).

The euphotic zone decreased to 4.2-4.7 m by 2 July 2002 and was still nearly uniform at all five stations (Fig. 15). During the autumn algal bloom (25 September), the euphotic zone varied from 3-6 m and showed marked variation from north to south with the depth of the euphotic zone being less at the southern station (Fig. 16).

Fig. 12 Seasonal and lakewide variation in Secchi depth during summer 2002.

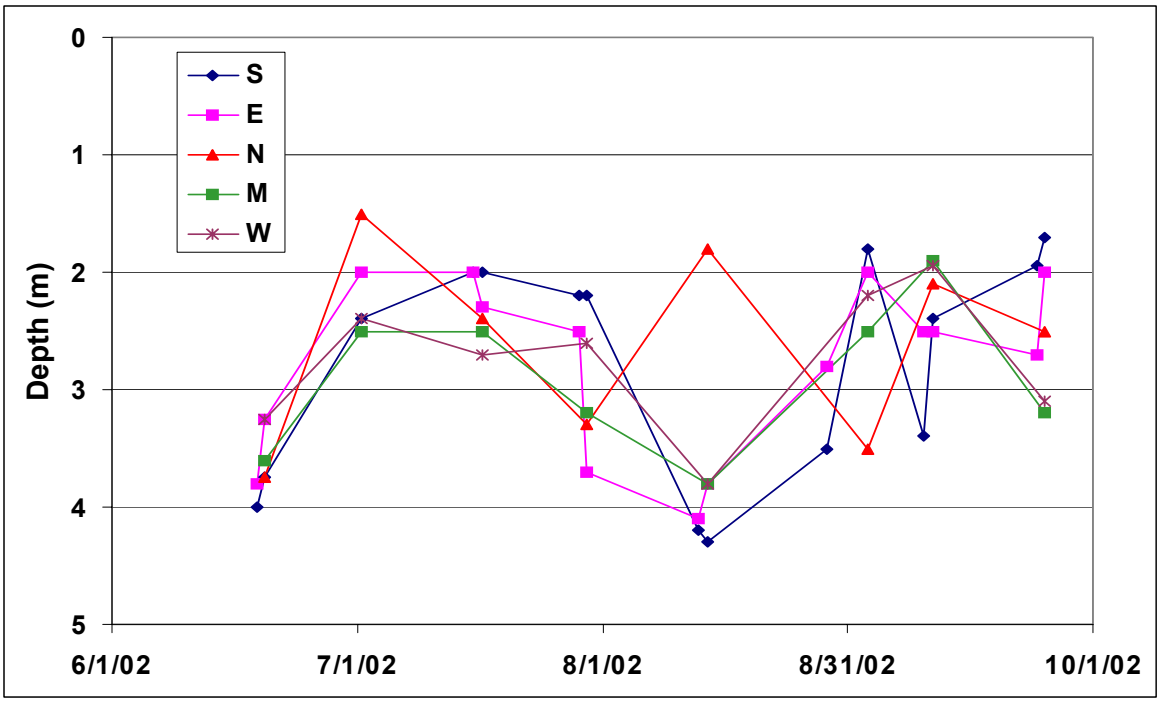


Fig. 13 Attenuation of PAR (fraction of surface) during 2002 (S station)

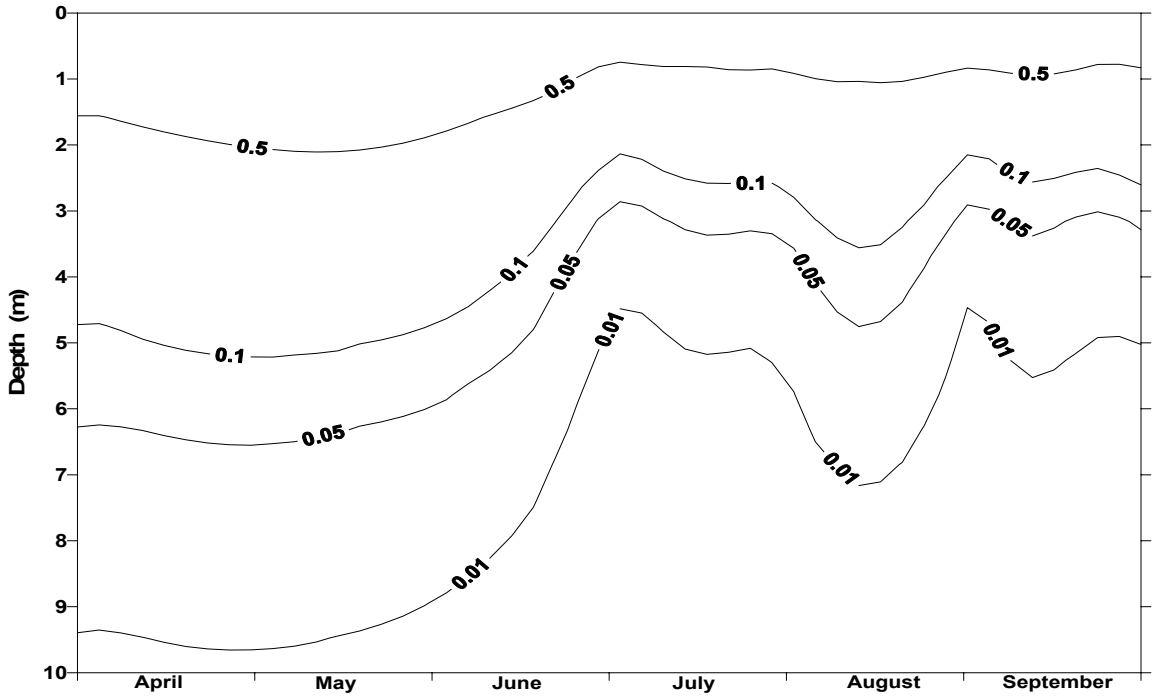


Fig. 14 Spring lakewide comparison of PAR attenuation

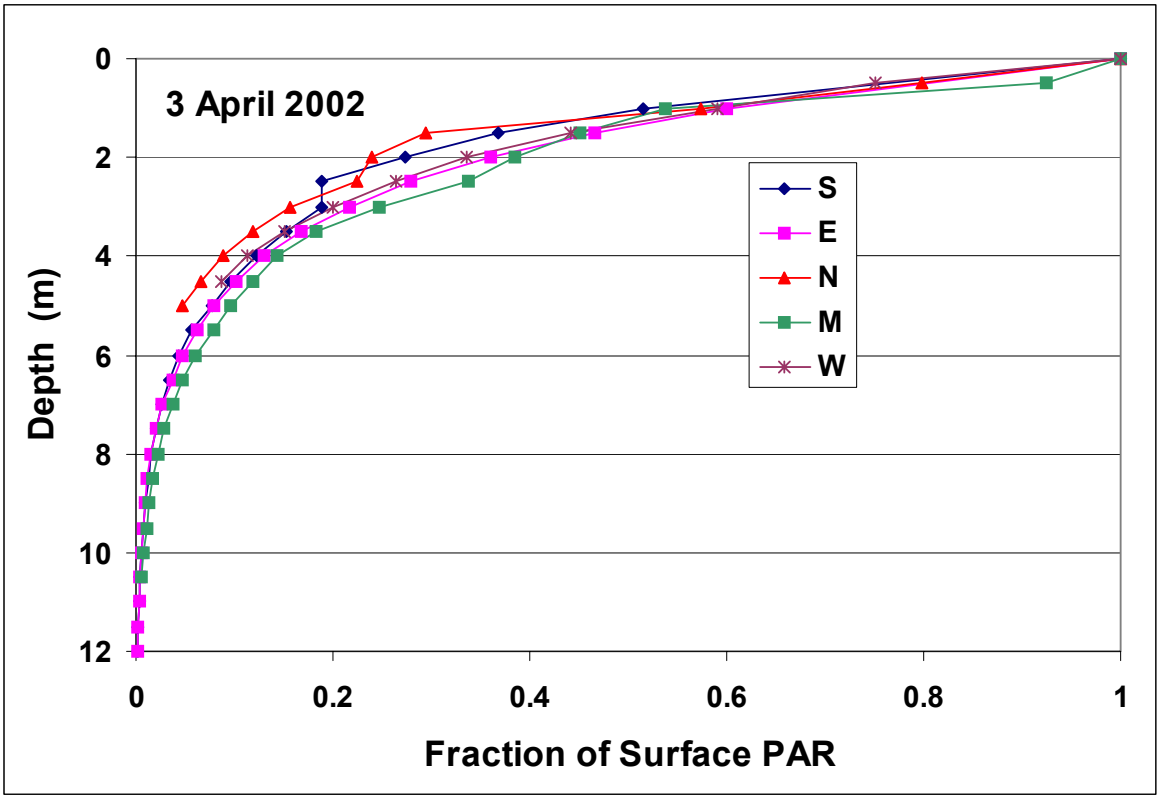


Fig. 15 Summer lakewide comparison of PAR attenuation

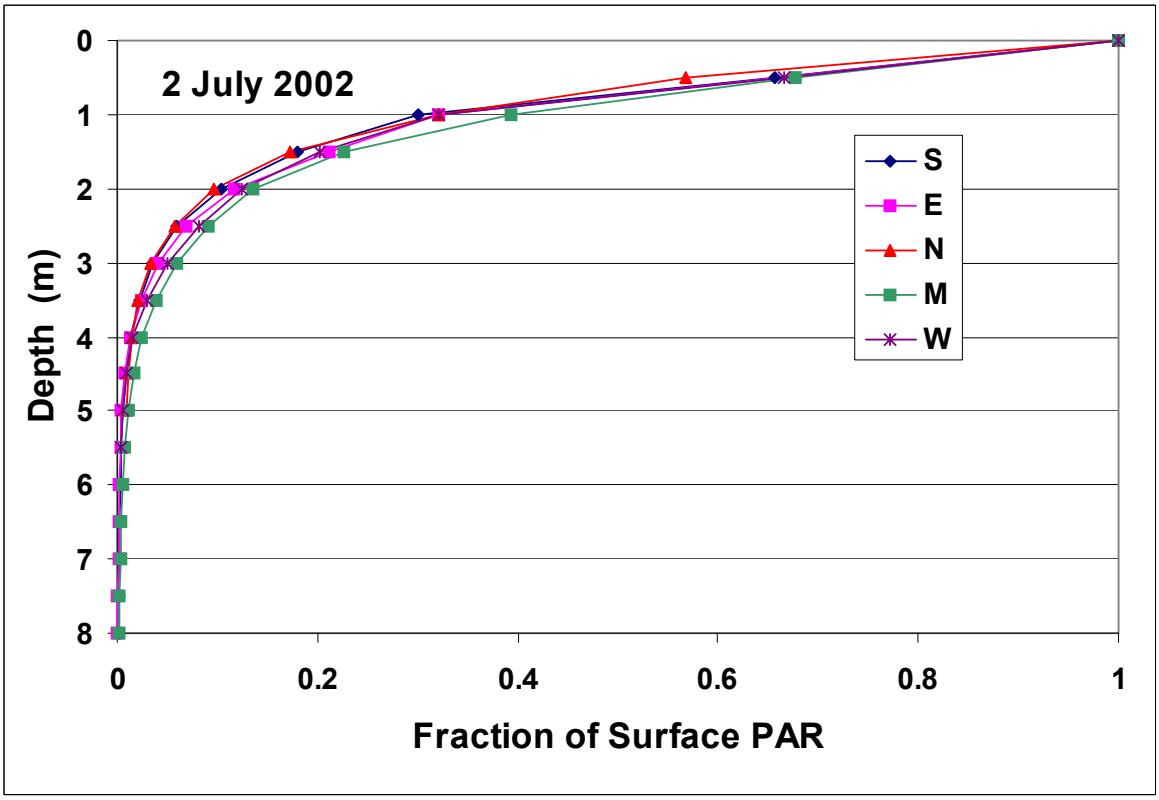
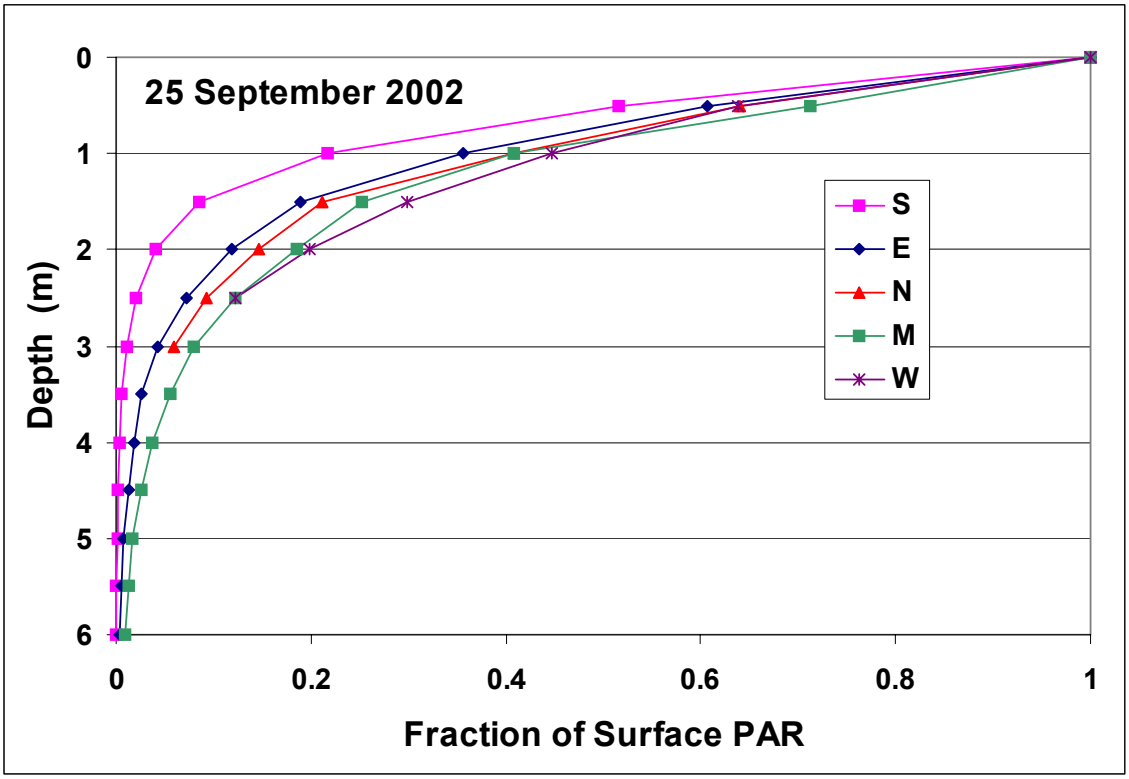


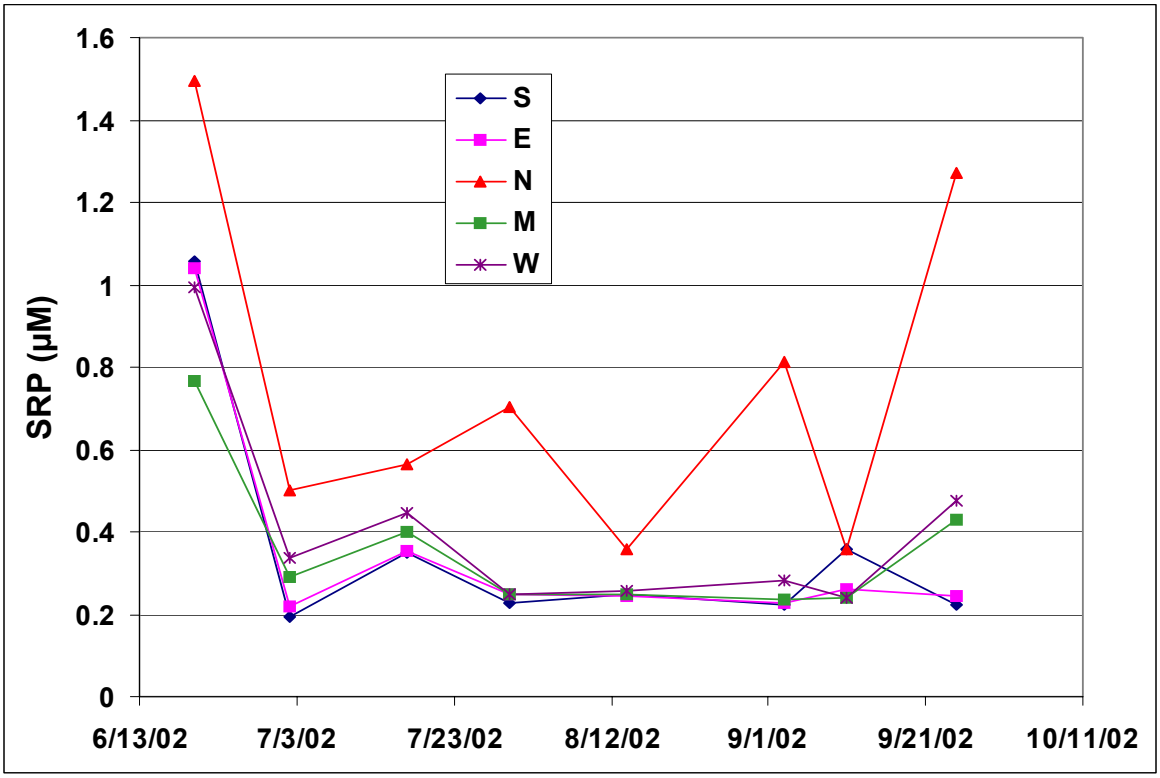
Fig. 16 Autumn lakewide comparison of PAR attenuation



Epilimnetic SRP and dissolved inorganic nitrogen (NH_4, NO_3)

Epilimnetic SRP concentrations in the upper 5-m integrated samples of the water column ranged from 0.2 to 1.5 µM (Fig. 17). On all but one date, the northern station was significantly higher than any of the other four stations. The northern station is in the center of the long narrow northern portion of the lake which receives inflows of the Owens River. The other four stations had concentrations below 0.5 µM throughout the summer except for on the first date, 19 June, when they were 0.8–1.1 µM.

Fig. 17 Epilimnetic (0-5 m integrated) SRP concentrations



Epilimnetic ammonium concentrations were low (<1.2 µM) throughout the summer except for a single sample on 3 September at the N station and on the last sample date (Fig. 18). The 5.5 µM reading at the N station appears to be an outlier and may have resulted from the integrated sampler coming into contact or disturbing the sediments. The reservoir had been drawn down significantly in late summer and the N station was just over 5 m deep. The increase at M, N, and W stations on 25 September likely resulted from deepening of the mixed layer and entrainment just prior to autumn overturn.

Epilimnetic nitrate was even lower than ammonium through most of the summer when it ranged from 0.1 to 0.3 µM (Fig. 19). As with ammonium, NO_3 was slightly higher on the first and last sample dates, 19 June and 25 August. A significantly higher value at the N station on 3 September also occurs and may be due to disturbing sediments (see above).

Fig. 18 Epilimnetic (0-5 m integrated) NH_4 concentrations

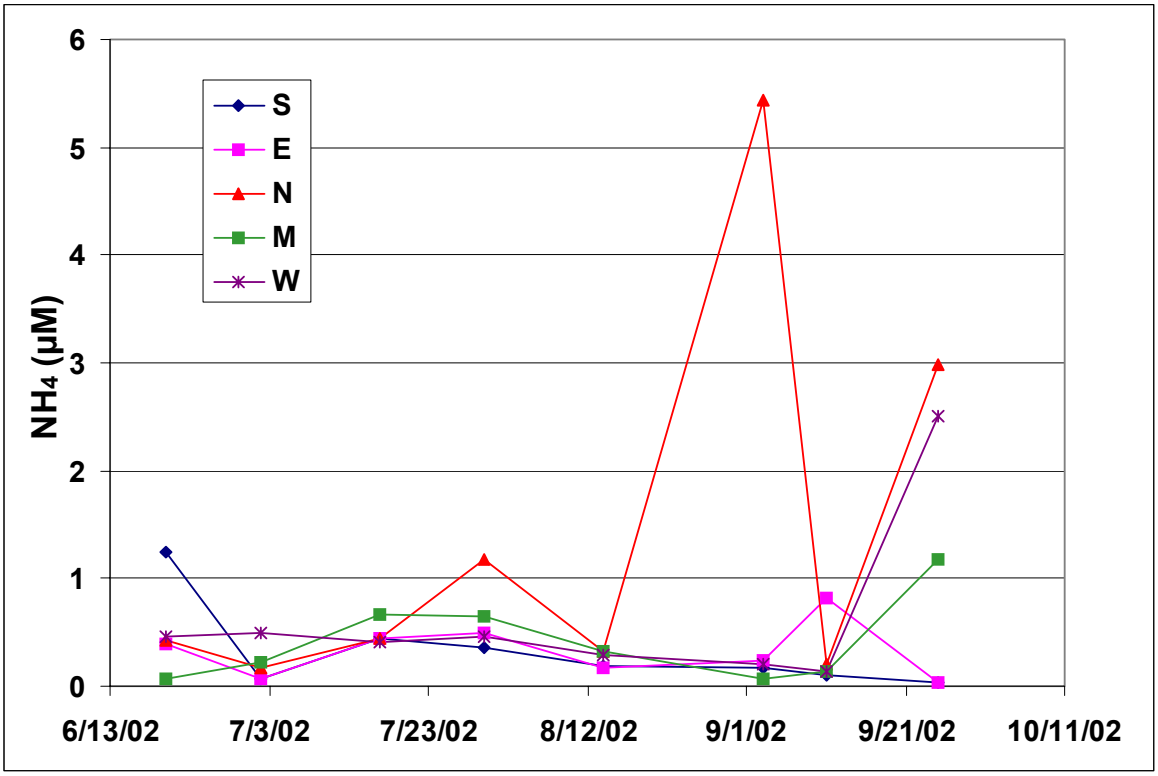
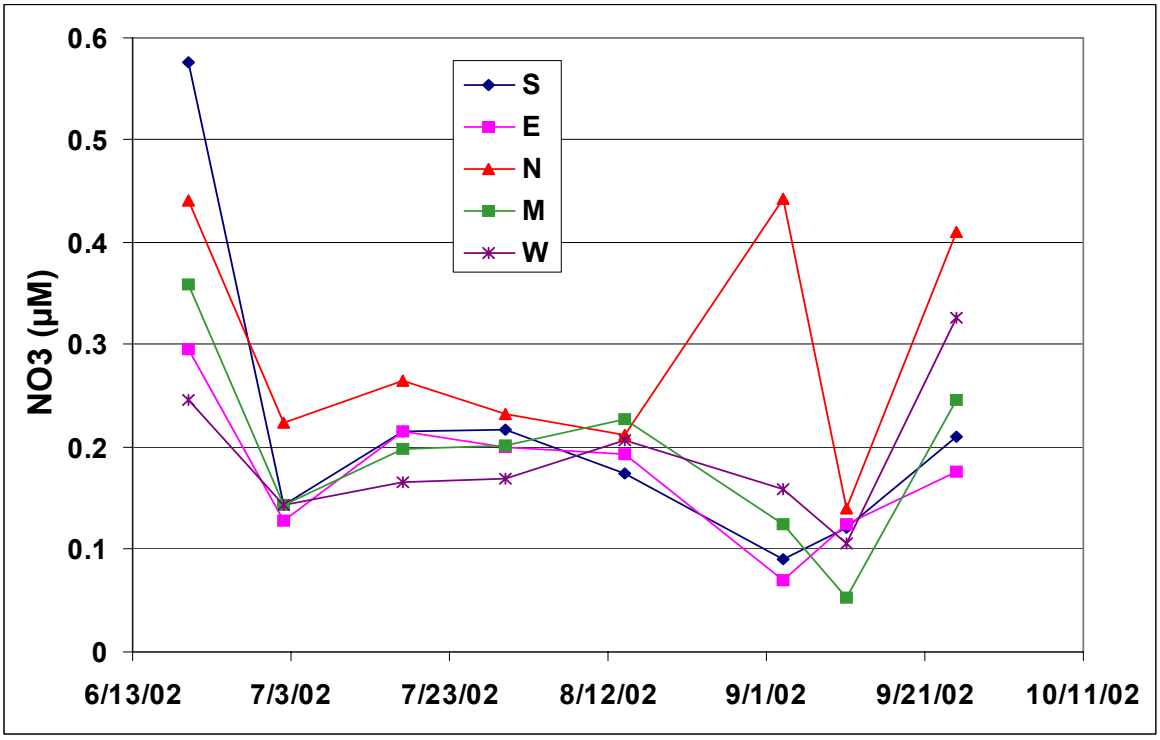


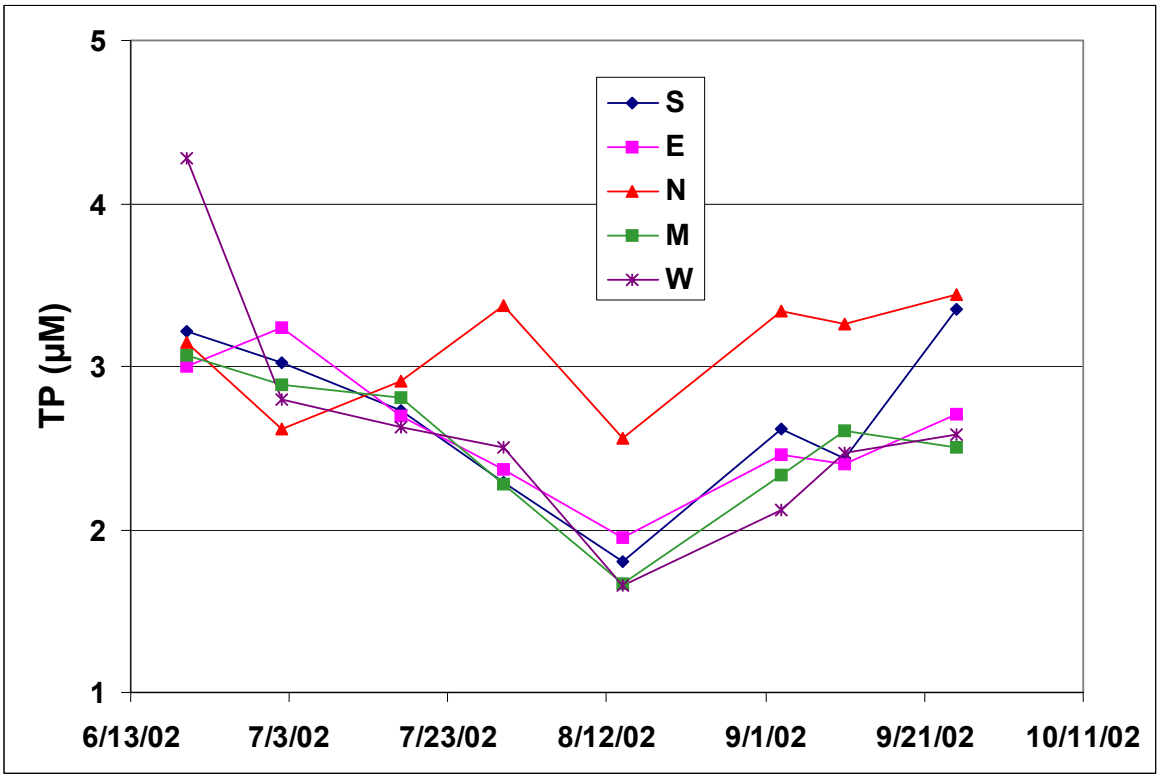
Fig. 19 Epilimnetic (0-5 m integrated) NO_3 concentrations



Epilimnetic total phosphorus (TP) and total nitrogen (TN)

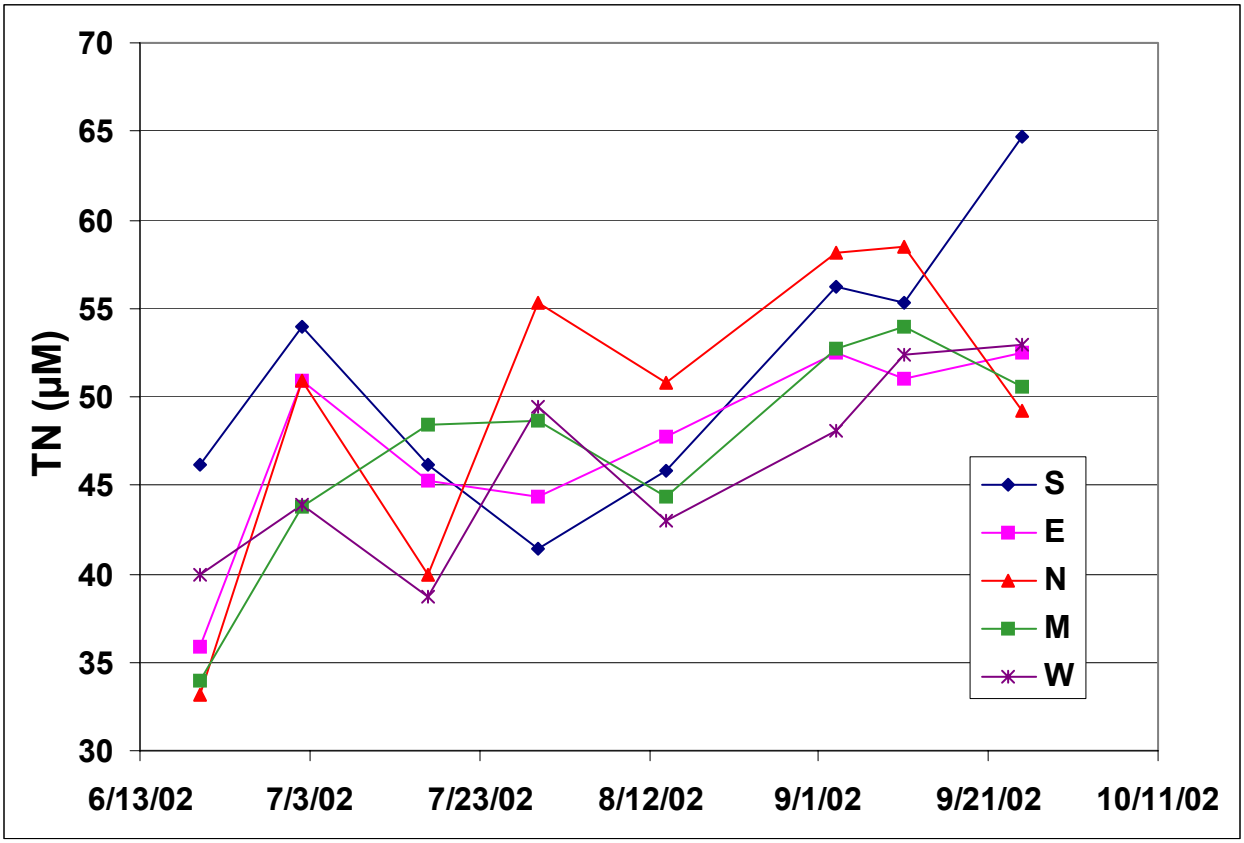
Summer epiliminetic total phosphorus (TP) concentrations ranged from 1.5 to 3.5 µM, except for a slightly higher value (4.3 µM) at the W station on 19 June 2002 (Fig. 20). The TP concentrations were very similar among W, M, E, and S stations through the entire period, while TP at the N station was 0.5-1.0 µM higher beginning on 30 July. The N station lies midway up the long, narrow portion of the lake which receives inputs from the Owens River. The slightly higher concentrations of TP, undoubtedly reflect the influence of the Owens River from which most of the TP loading to the lake comes. The general seasonal trend was peak average lakewide concentrations (3.3 µM) at the beginning of summer (19 June) declining to a minimum (1.9 µM) in mid-August, followed by a gradual rise to higher values (2.9 µM) by the end of September.

Fig. 20 Epilimnetic (0-5 m integrated) TP concentrations



Summer epilimnetic total nitrogen (TN) concentrations ranged from 33 to 65 µM (Fig. 21). There was a clear seasonal trend of gradually increasing concentrations through the period from a lakewide mean of 37.8 µM on 19 June to 54.0 µM on 25 September. On 25 September, the S station value (64.7) was 13 µM higher than the mean of the other four stations (51.8 µM). However, even without this value, the consistent seasonal increase is still present. As with TP, the N station was consistently higher beginning on 30 July and continuing through 11 September. However, N station values were only 5-10µM higher than the average of the other 4 stations and thus on a relative basis less pronounced than the TP values. The overall summer mean concentration was 48.3 µM (±1.1, 1 SE).

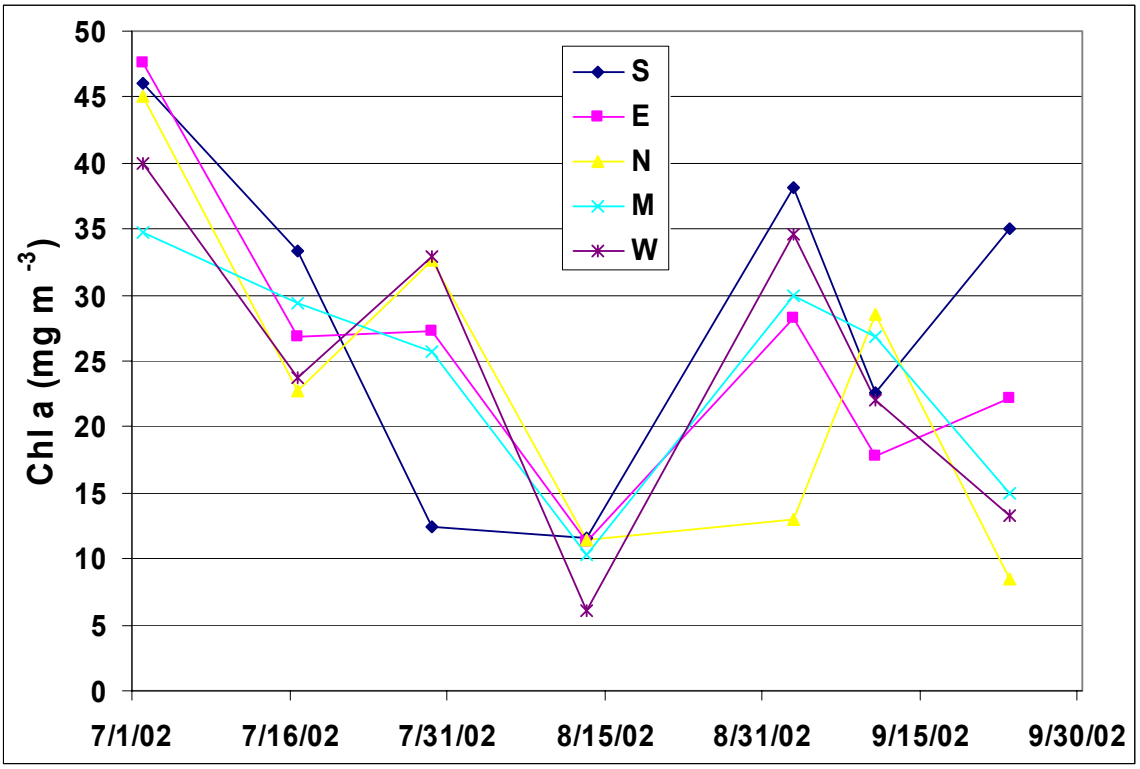
Fig. 21 Epilimnetic (0-5 m integrated) TN concentrations



Chlorophyll concentration and phytoplankton community

Summer epilimnetic chlorophyll *a* concentration ranged from 6 to 48 µg Chl *a* l^{-1} with the highest observed 3 July (mean, 43 µg Chl *a* l^{-1}) (Fig. 22). Following the peak concentrations observed during the spring algal bloom, chlorophyll *a* decreased to 10 µg Chl *a* l^{-1} in mid-August and was followed by a second smaller (mean, 29 µg Chl *a* l^{-1}) autumn bloom. Concentrations decreased slightly by mid-September and then at stations N, M, and W decreased further by the last sampling on 25 September. There was a slight increase at E and S stations between the last two sample dates. As these are the two deepest stations, this likely resulted from entrainment of nutrients during deep autumnal mixing.

Fig. 22 Epilimnetic (0-5 m integrated) Chl *a* concentrations



Phytoplankton samples for species enumeration were collected from all stations on each sample date to determine the species composition. General trends can be described using the South station as a typical representation of other stations in the lake. On 20 June, the cyanophytes dominate throughout the lake (Fig. 23) and continue to dominate through the summer (Fig. 24). The phyrrhophyte *Ceratium* was abundant at all stations, June through August when the biovolume of all phytoplankton decreased. At this time, we start seeing an increase in the relative biovolume of Bacillariophyceae (Fig 24). N station typically had a higher Bacillariophyceae (diatom) population than the other stations, increasing to 70% relative abundance on the 25 September sampling date. As the cyanophytes are the important nitrogen fixers in the lake, they will be discussed in more detail under the heading Nitrogen Fixation.

Fig. 23. Relative abundance of phytoplankton at each station on 20 June, 2002.

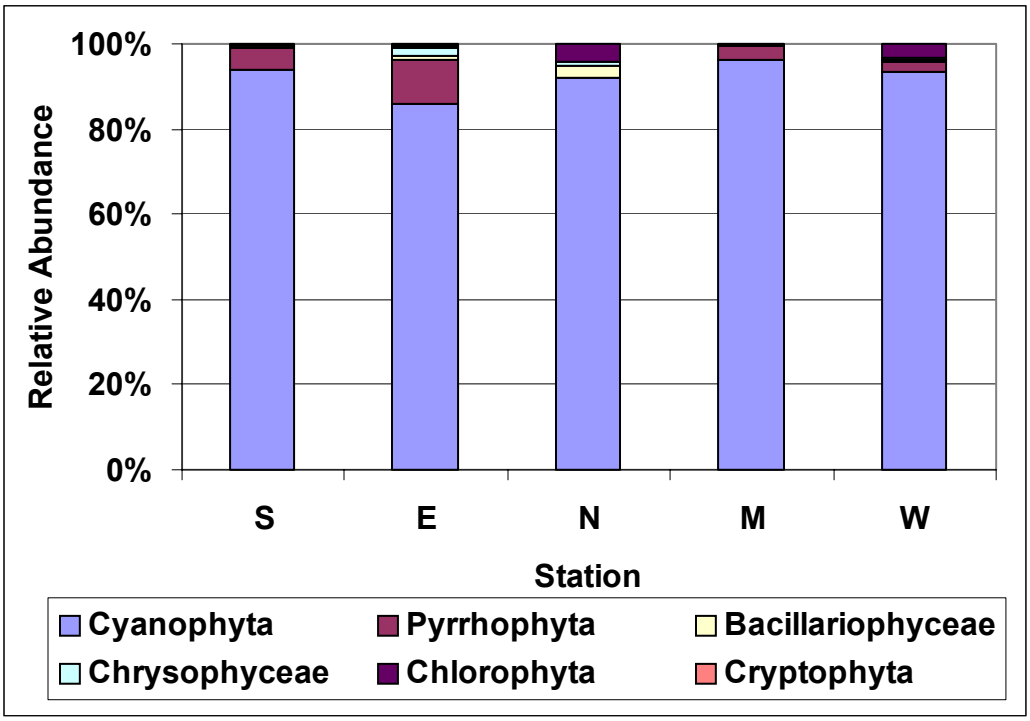
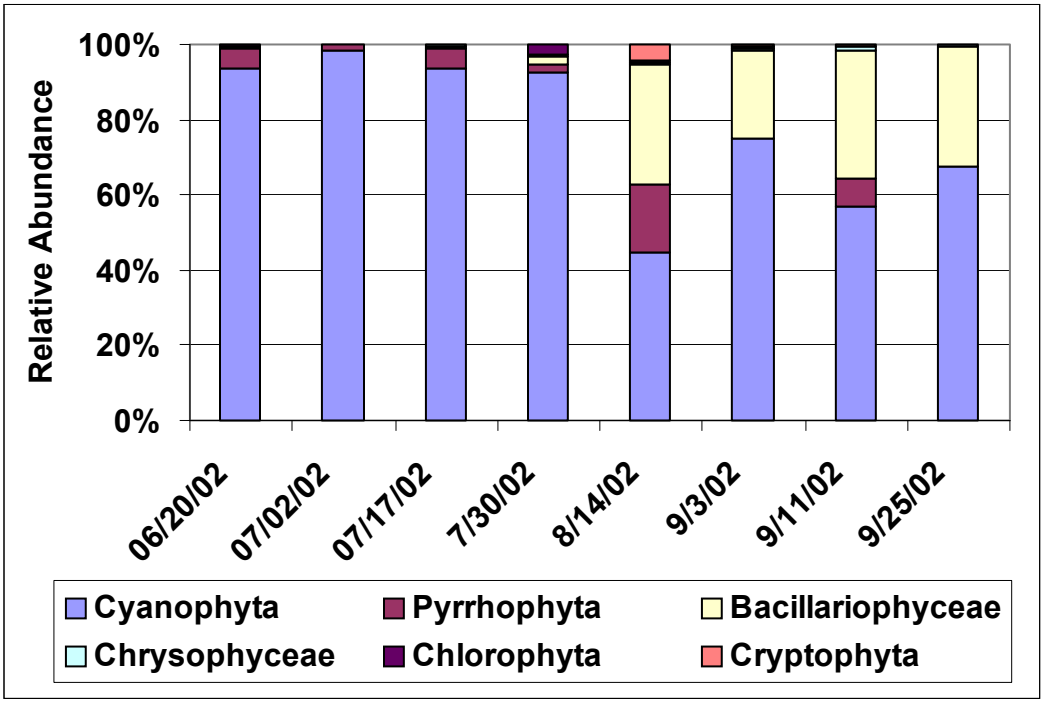


Fig. 24. Relative abundance of Phytoplankton at South Station during 2002.



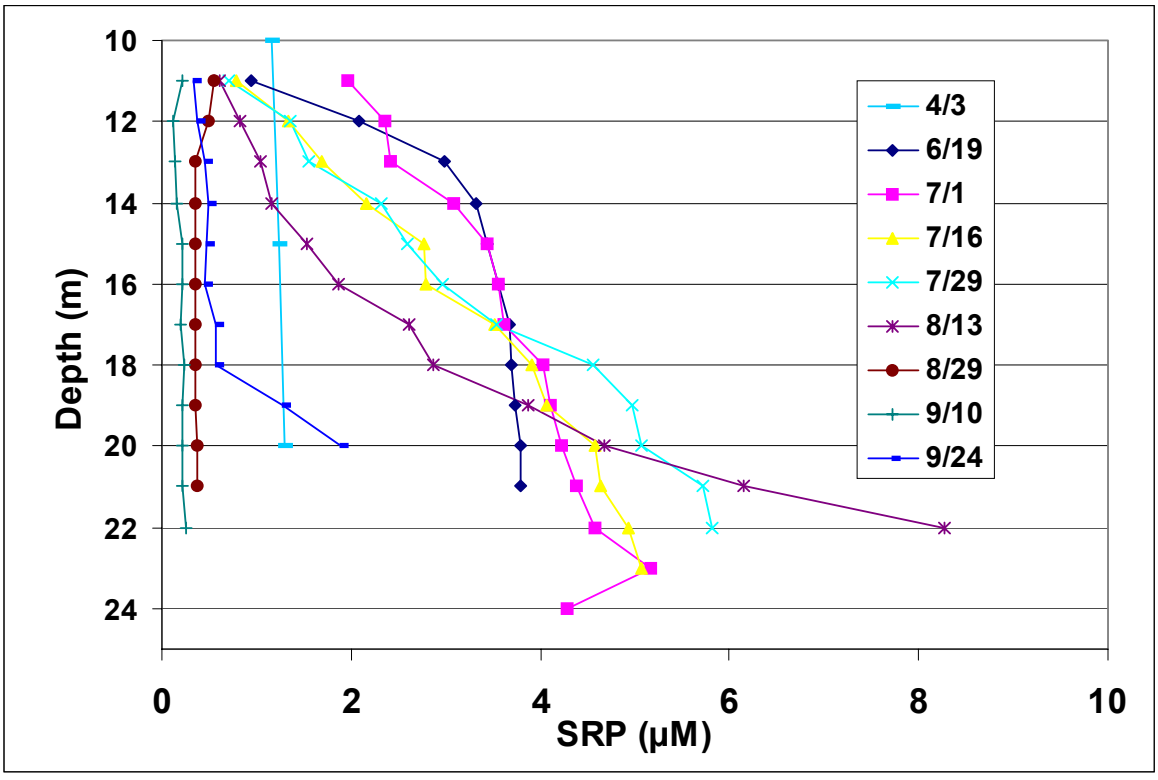
Hypolimnetic ammonium, nitrate, and phosphorus accumulation

Nutrients accumulate beneath the seasonal thermocline in eutrophic temperate lakes. During 2002 we measured the accumulation of nutrients beneath the seasonal thermocline in Crowley Lake by analyzing NH_4, SRP, and NO_3 collected at 1-m intervals from 11 m depth to the bottom at the deep E and S stations.

In 2002, ice-off occurred about the third week of March, several weeks earlier than normal. On the first survey date, 3 April 2002, SRP concentrations at the deep, S station were nearly uniform throughout the water column ranging from 1.2 µM in the upper 5-m integrated sample to 1.3 µM at 20 m (Fig. 25). SRP accumulated beneath the seasonal thermocline during April-June and by the 19 June sampling had reached concentrations between 3-4 µM beneath 13 m. Hypolimnetic concentrations continued to increase and by 13 August exceeded 8 µM at 22 m. The nutricline (region of strong nutrient gradient) descended through the summer following the thermocline as the mixed layer deepened.

On 29 August, SRP concentrations were nearly uniform with depth at the S station ranging only from 0.34-0.56 µM indicating that mixing throughout the water

Fig. 25 Hypolimnetic accumulation of SRP at S station during 2002



column had occurred since the 13 August sampling. The SRP profile on 10 September was also nearly uniform with somewhat lower concentrations ranging from 0.14-0.25. On the final survey, 24 September, mixed-layer concentrations were slightly higher at

0.3-0.6 µM and increased to 1.9 µM at 20 m indicating continued rapid release from the sediments.

A similar seasonal pattern of hypolimnetic SRP accumulation was observed at the other deep station (E) (Fig. 26). On 19 June, SRP concentrations from the two stations were nearly identical (Fig. 27). While upper water column concentrations were similar between the two stations on other dates, the lowest depth at E was often significantly higher than comparable depths at S. This most likely reflect high concentration gradients in proximity to the sediments at the shallower E station and is parallel to the lower dissolved oxygen concentrations observed at the lower depths of the E station.

Fig. 26 Hypolimnetic accumulation of SRP at E station during 2002

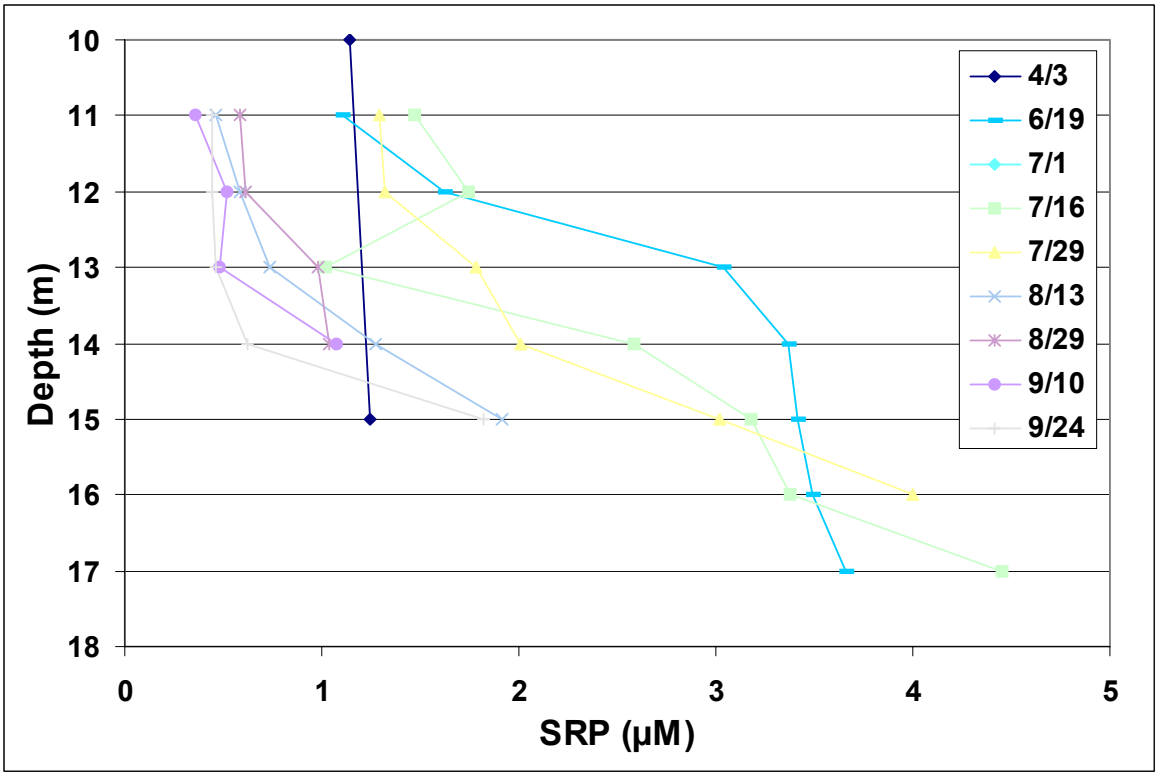
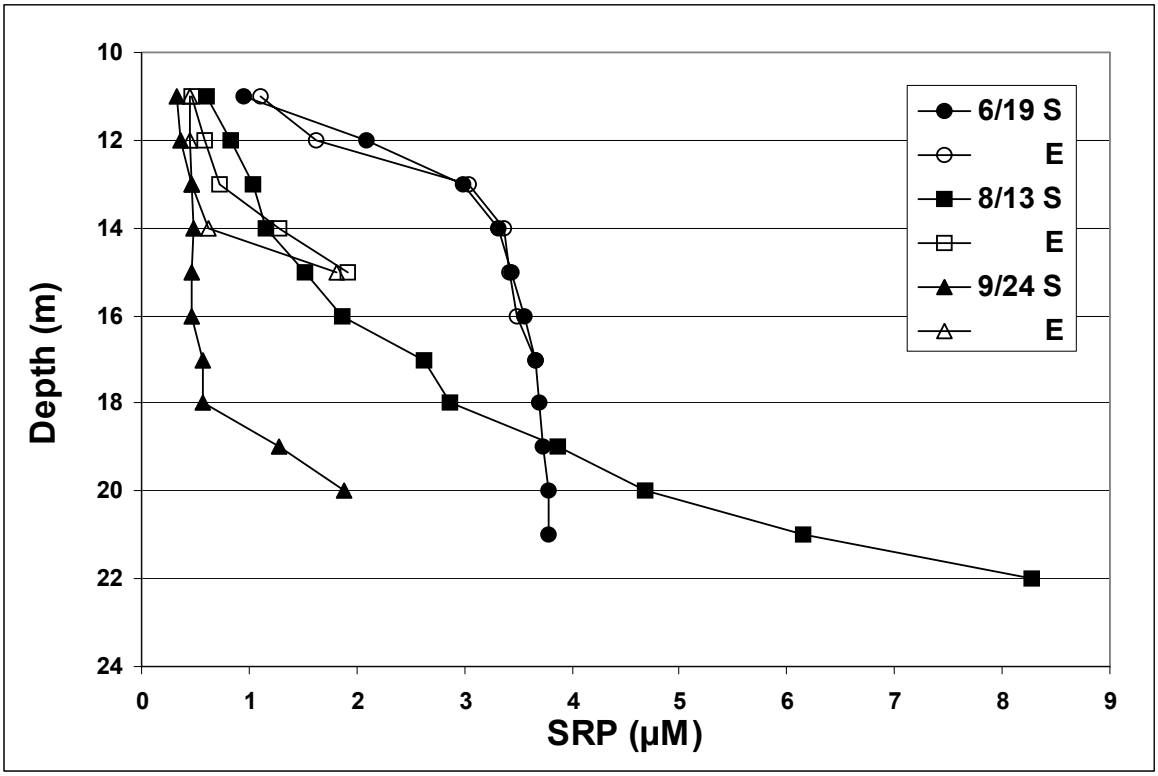


Fig. 27 Comparison of hypolimnetic SRP accumulation at E and S station



On 3 April, ammonium already displayed a depth gradient, increasing from 0.4 µM at 5 m to 4.4 µM at 20 m (Fig. 28). Ammonium concentrations increased to 13-16 µM below 13 m by 19 June. Hypolimnetic concentrations continued to increase reaching greater than 60 µM by 29 July. Peak concentrations were observed on 13 August when they exceed 80 µM at 22 m. As with SRP concentrations, hypolimnetic concentrations were uniform on 29 August only varying between 1.8-2.8 µM between 13 and 21 m. These decreased to 0.3-1.3 µM by 10 September before increasing to 1.5-5.0 from 11 to 18 m and further to 16 µM at 20 m on 24 September. This pattern is identical to that observed for SRP.

Fig. 28 Hypolimnetic accumulation of NH_4 at S station during 2002

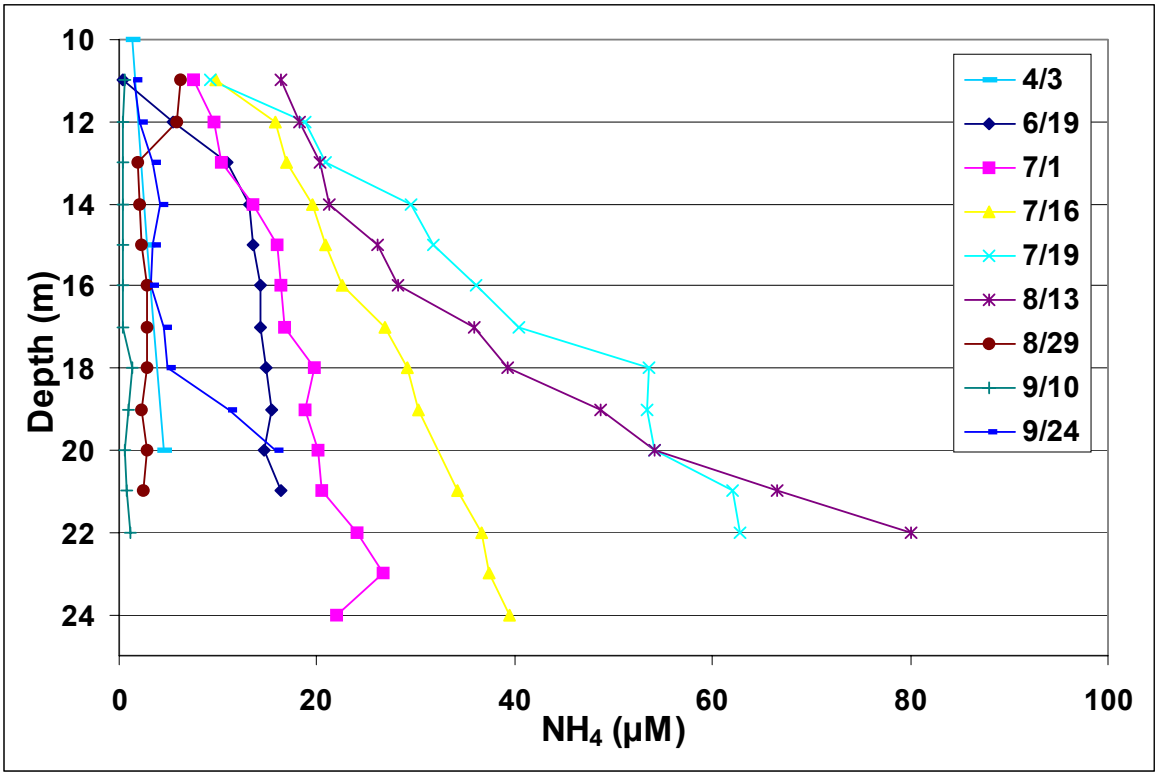


Fig. 29 Hypolimnetic accumulation of NH_4 at E station during 2002

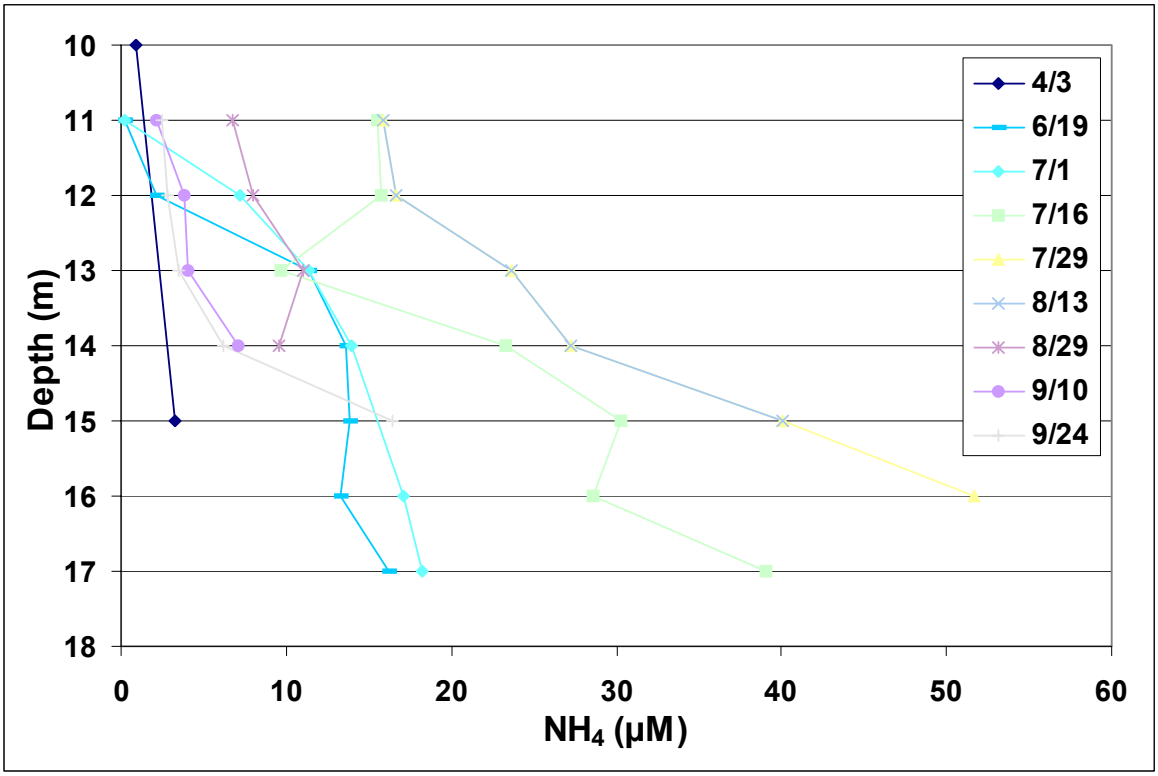
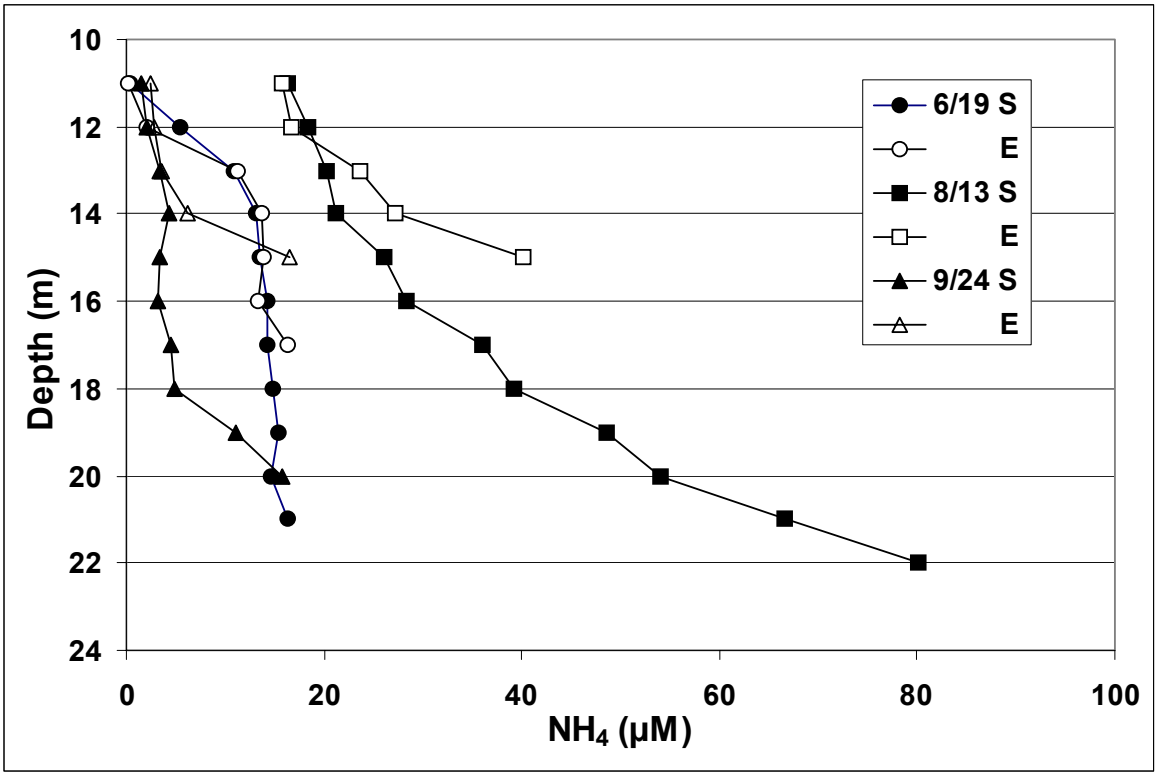


Fig. 30 Comparison of hypolimnetic ammonia accumulation at E and S station



Hypolimnetic accumulation of ammonia at the E station similarly increased through the season with concentrations peaking at 51.6 on 29 July at 16 m (Fig. 29). As with SRP, the ammonia accumulated rates were similar between the two stations (Fig. 30). The two stations were nearly identical 19 June but diverged later in the year with the deep depths at E being higher than comparable depths at S.

Nitrate profiles at the two stations were low and variable and showed no significant seasonal, depth, or station to station differences. The overall mean concentration of samples collected from 11 to 22 m at the two stations was 0.23 µM (Std Dev, 0.08) with individual samples ranging from 0.09 to 0.44 µM.

The total accumulation of SRP and NH_4^+ beneath 11 m was estimated by interpolating the nutrient concentrations to 0.25 m intervals and multiplying by the appropriate incremental volumes (Table 3). The concentrations at the two stations were averaged excluding the lowest depth at the east station which was not included. During the 77-day period from 3 April to 19 June, lakewide SRP beneath 11 m increased at 619 moles d^{-1} while NH_4 increased at 3,688 moles d^{-1}. The N:P molar ratio of accumulation was ~6. As the area beneath 11 m during this period was ~6.7 km^2, this yields accumulation rates of 92 and 550 µmoles m^{-2} d^{-1}, for SRP and NH4, respectively.

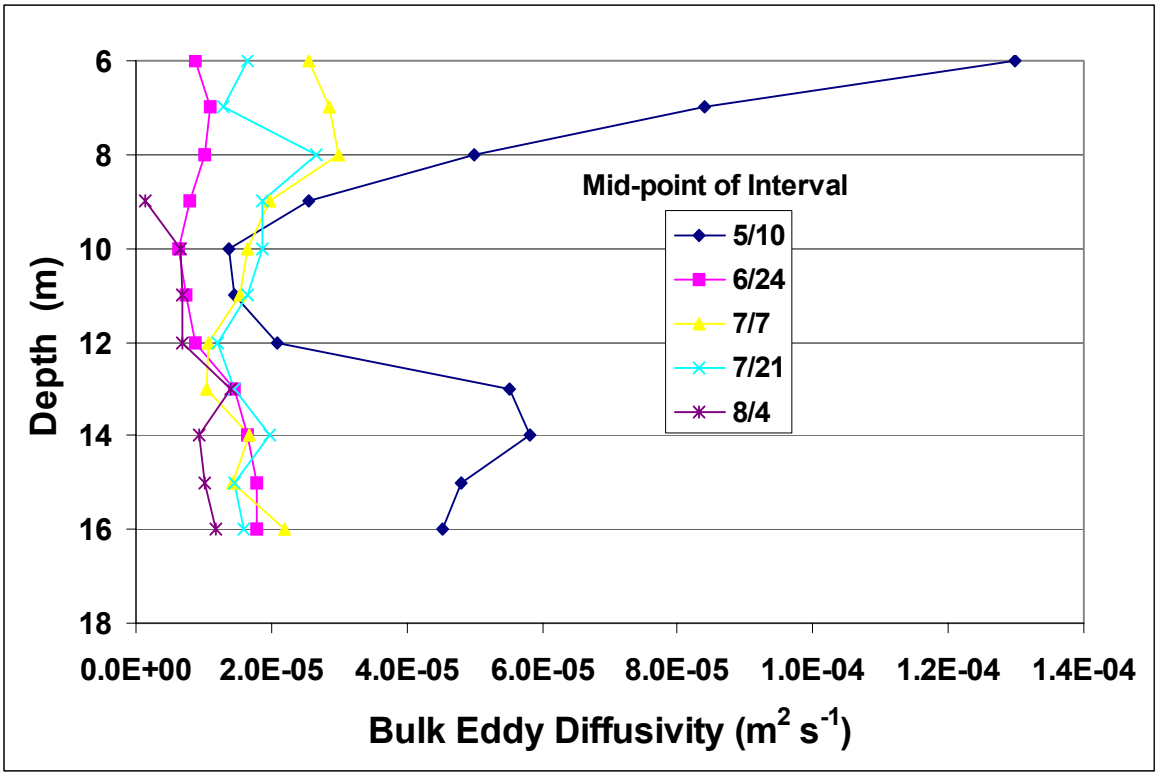
Table 3 Hypolimnetic accumulation of SRP and NH_4 beneath 11 m depth.

Date	SRP (moles)	NH4 (moles)
4/3/2002	43945	52155
6/19/2002	91597	336144
7/1/2002	90701	406263
7/16/2002	63352	543748
7/29/2002	57840	704888
8/13/2002	32253	529985
8/29/2002	6917	58392
9/10/2002	3050	8073
9/24/2002	8178	59047

Eddy diffusivity and upward nutrient fluxes

The flux-gradient heat method of estimating bulk eddy diffusivities uses heat as a tracer of mixing during periods in which heat is being mixed downward in the lake. Eddy diffusivities were estimated from 6 to 18 m depth for the period 3 April through 29 August (Fig. 31). Values ranged from 1.4 x 10^{-6} m^2 s^{-1} (~10 times molecular conductivity) to 1.3 x 10^{-4} m^2 s^{-1} (~1000 molecular conductivity). Minimum values were located between 10 and 12 m at the region of maximum thermal stratification.

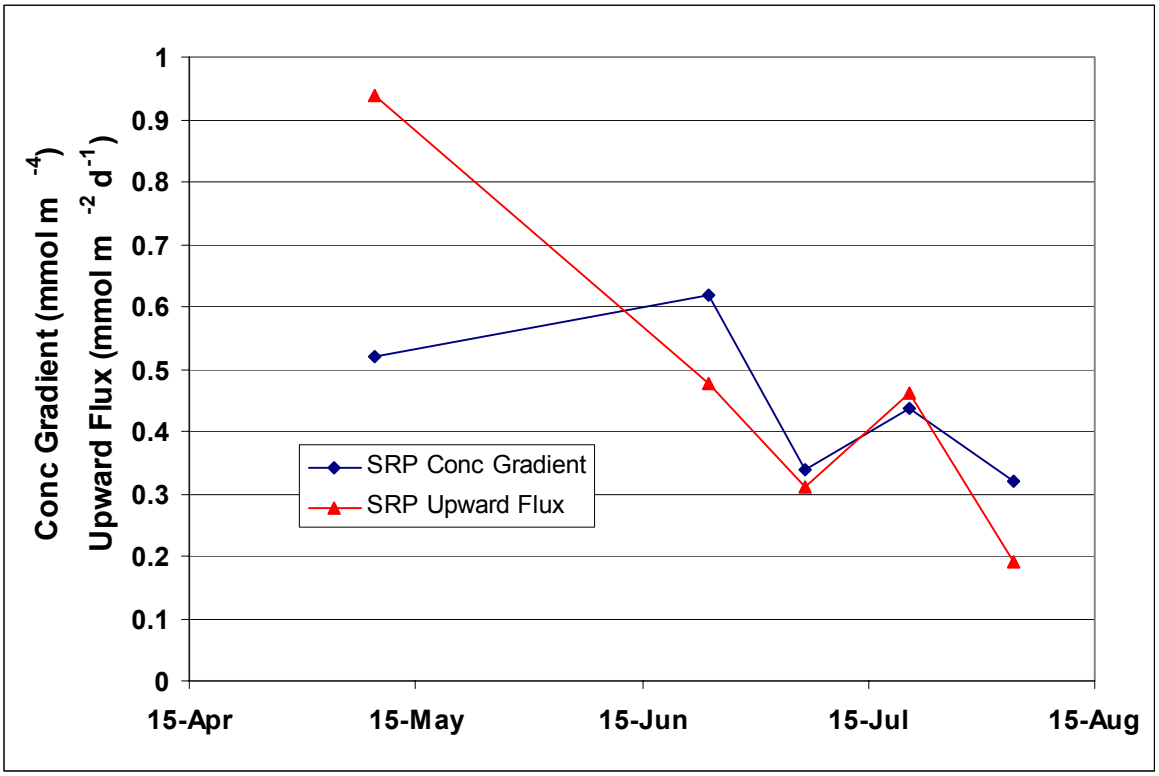
Fig. 31 Bulk eddy diffusivities estimated from the flux-gradient heat method.



The average SRP concentration gradients at 12 m over the 5 periods ranged from about 0.3 to 0.6 mmole m^{-4} (Fig. 32). Combining these with the estimated eddy

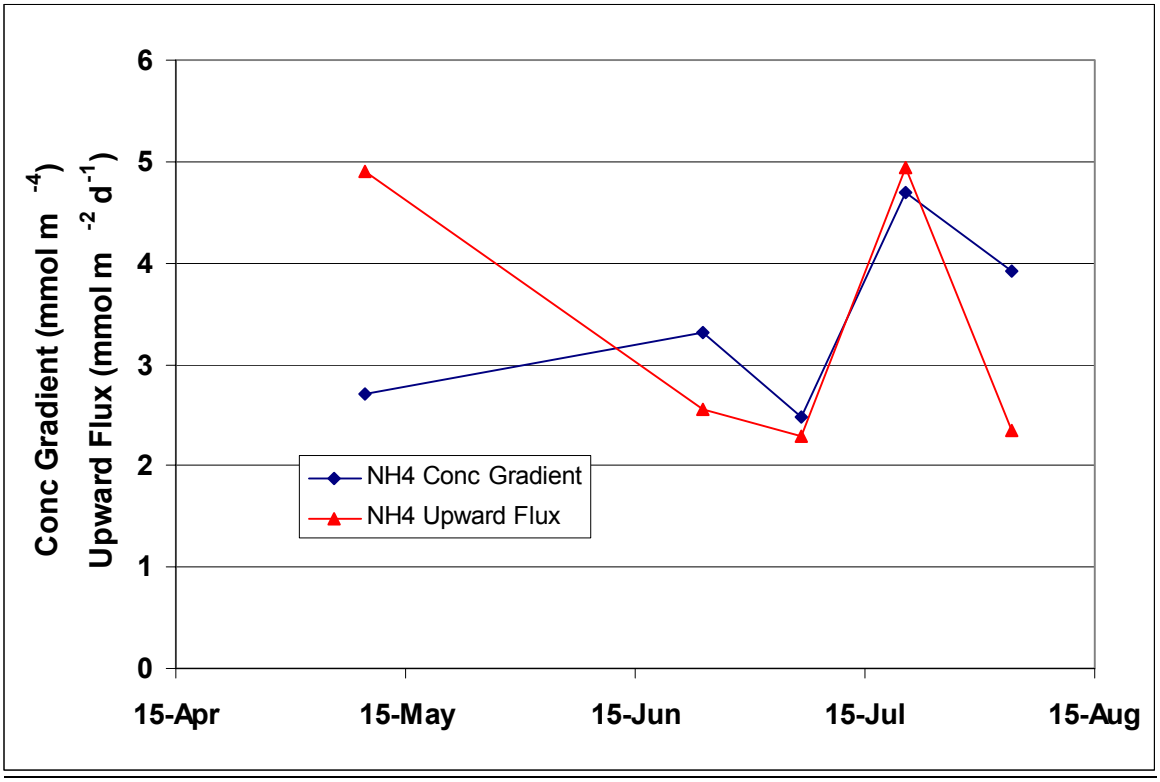
diffusivities yields upward SRP fluxes ranging from 0.94 mmole m^{-2} d^{-1} for the 3 April to 19 June to a minimum of 0.2 mmole m^{-2} d^{-1} for the period, 29 July to 14 August.

Fig. 32 SRP concentration gradients and estimated upward fluxes at 12 m



The average NH_4 concentration gradients at 12 m over the same periods ranged from about 2.4 to 4.7 mmole m^{-4} (Fig. 33). Combining these with the estimated eddy diffusivities yields upward NH_4 fluxes ranging from 2.2 to 5 mmole m^{-2} d^{-1}. In contrast to SRP fluxes, the estimated ammonia flux was high for the first period, declined during the next two and increased in mid summer.

Fig. 33 NH_4 concentration gradients and estimated upward fluxes at 12 m



Nitrogen Fixation

Planktonic nitrogen fixation was estimated from measurements of nitrogenase activity based on the acetylene reduction technique. Acetylene reduction was measured on 8 dates in freshly collected plankton samples from 5 stations. A plankton sample was collected from the Crowley Lake marina dock on 29 May for time course and saturation tests. Acetylene reduction was linear over the 6 hr test period and the average nitrogen fixation rate for three replicates was 34.6 µmol N m^{-3} h^{-1}. This was higher than any other rates measured on the eight surveys. Results from all eight surveys are shown below (Fig. 34).

On the June survey, rates at the highest light level of 95 µE m^{-2} s^{-1} ranged from 3.9 to 10.2 µmol N m^{-3} h^{-1} or only about one fifth that measured on 29 May. Rates were much lower and variable on the following survey (2 July). As rates appeared to not be saturated at the highest light levels on the first two dates, the remaining surveys included two treatments in which samples were incubated in the stream and exposed to much higher (1000-1800 µmol N m^{-3} h-1) light intensities. On these six remaining surveys, several exhibited inhibition at the highest light level, and most exhibited a maximum at the 400-800 µE m^{-2} s^{-1} treatment.

On the next two dates in July rates dropped even lower. In August and September rates increase. Rates were slightly higher on the 15 August survey and increased further on 3 September. The last two September dates were slightly lower but still higher than

June and July values. On three of the four August and September surveys, one station was higher than the rest and indicative of pronounced spatial variability.

Fig. 34 Nitrogen fixation estimates based on acetylene reduction measurements

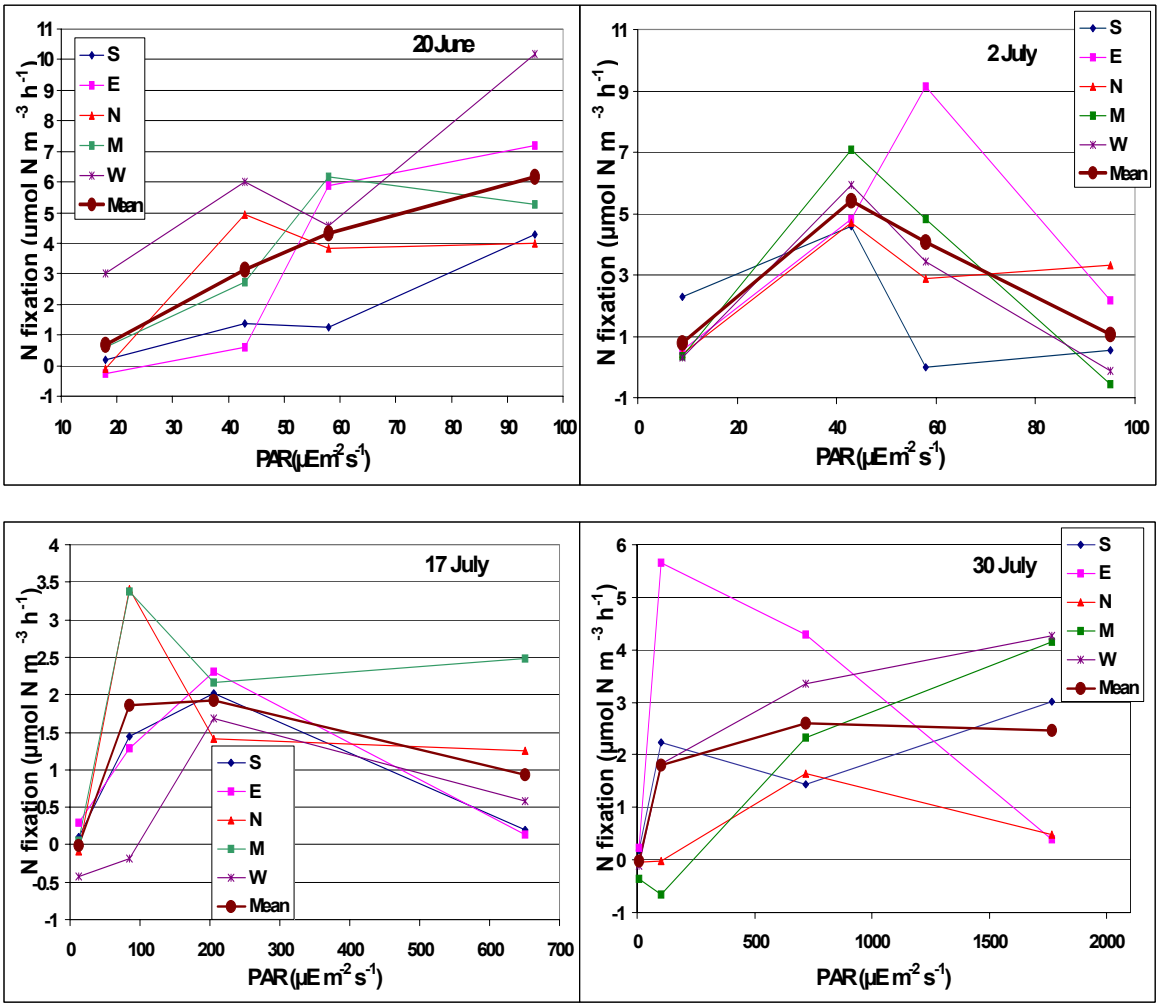
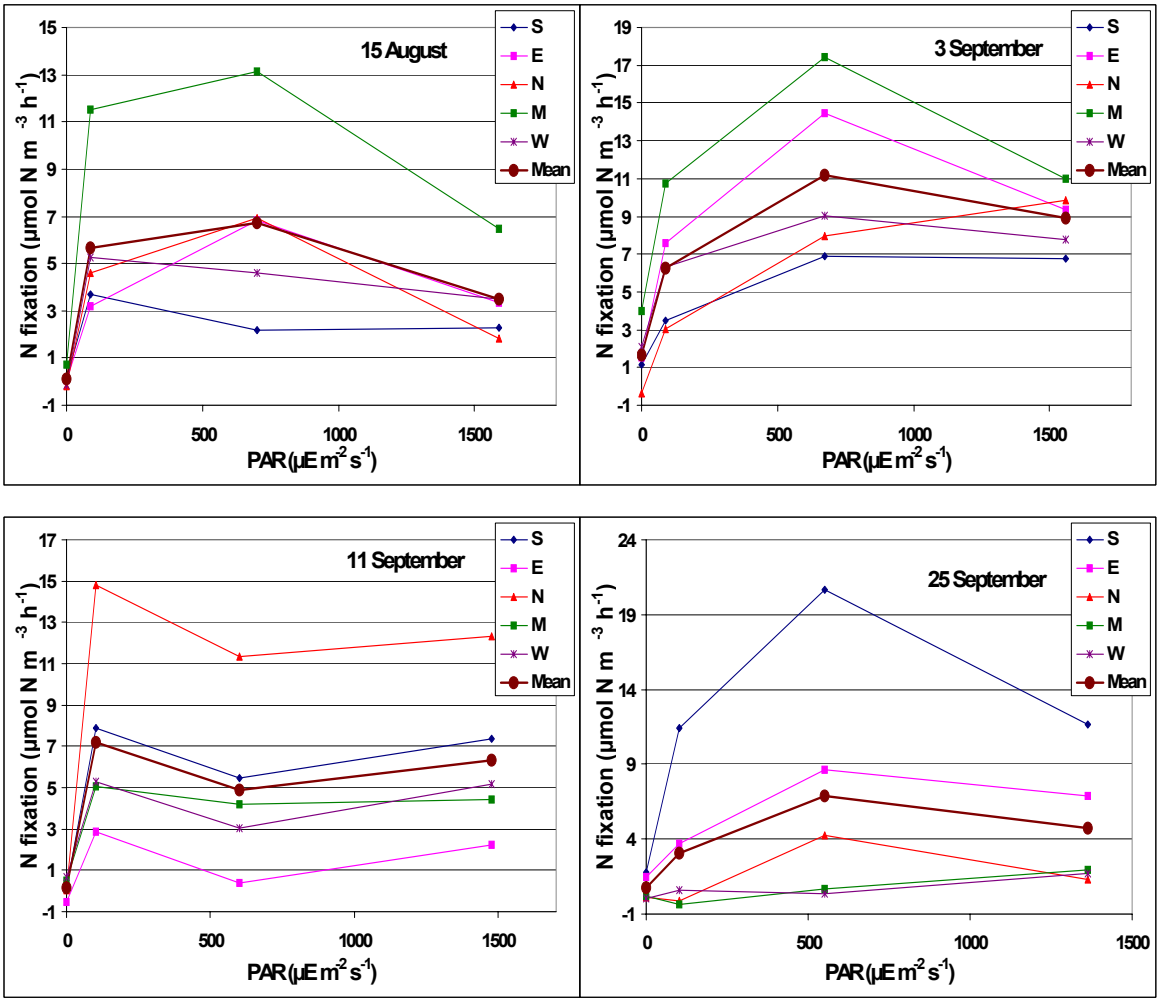


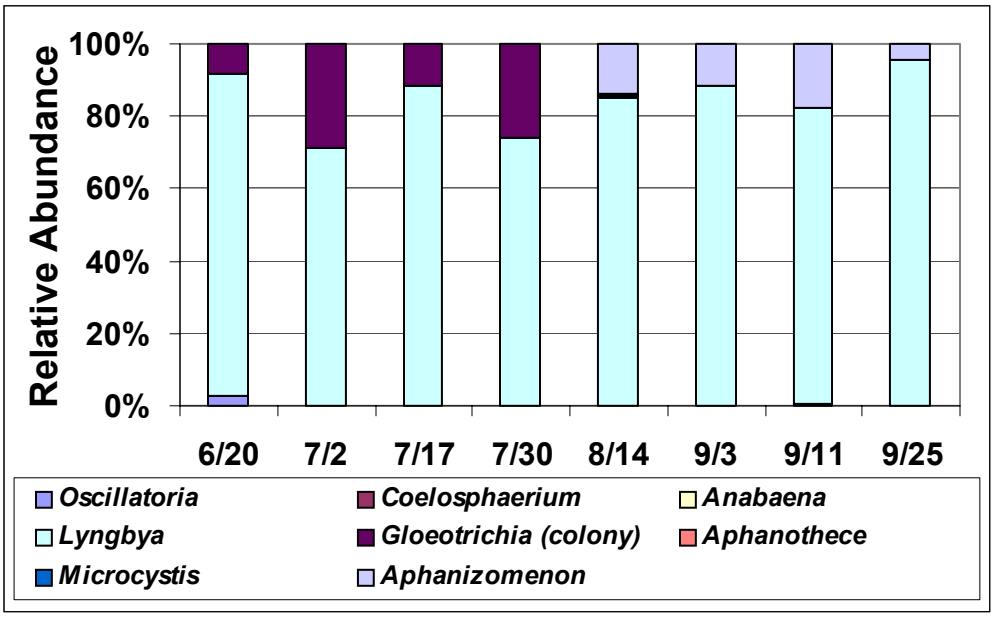
Fig. 34 (cont) Nitrogen fixation estimates based on acetylene reduction measurements



Nitrogen fixation was quite variable both spatially and temporally throughout the summer. Rates appeared to be fairly well correlated with the abundance of the cyanophyte, *Gloeotrichia*, during the months of June and July and with *Aphanizomenon* during September.

General trends can be described using the South station as a typical representation of other stations in the lake. *Lyngbya* was found throughout the lake in fairly high abundance relative to other cyanophytes throughout the summer. *Gloeotrichia*, present in June and July, was replaced by *Aphanizomenon* in August through September (Fig 35).

Fig. 35 Relative abundance of Cyanophytes at South Station during 2002



While overall abundance can vary substantially between stations on any given date, the general trend was similar to that found at the S station (Fig. 36) and paralleled the pattern of chlorophyll concentration (Fig. 22). Abundance was highest on 2 July, and then declined to a minimum on 14 August. Abundance then increased through September until the 25th, after which all stations except S, started to decline. Higher fixation rates at S (Fig. 37) during August and September corresponded with the appearance of *Aphanizomenon* (Fig. 36). *Aphanizomenon* was most abundant on 11 September, and decreases (in order) on 3 and 25 September. The highest fixation rates occurred on 24 September and decreased in order from 11 and 3 September. *Gloeotrichia* was present in June and July and most abundant on 2 July. Nitrogen fixation rates were higher on 2 July than other dates in June and July. However, it should be noted that the first two sets of incubations on June 20 and July 2 were done inside under artificial light and PAR was well below normal values observed in the lake. Beginning 17 July, two sets of incubations were done with one in Convict Creek under higher more natural light.

Fig. 36 Abundance of Cyanophytes at South Station during 2002

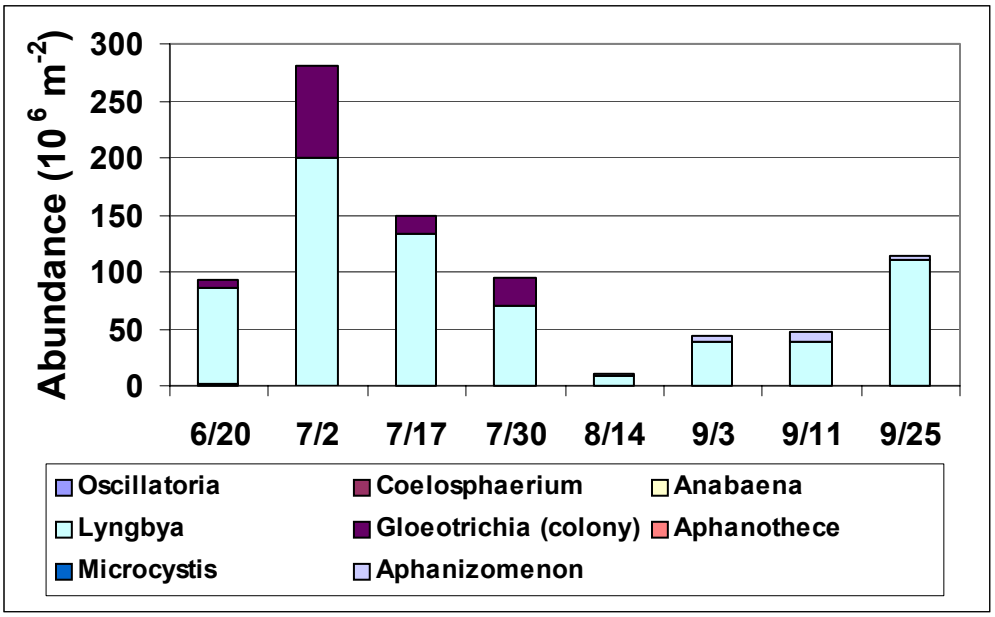
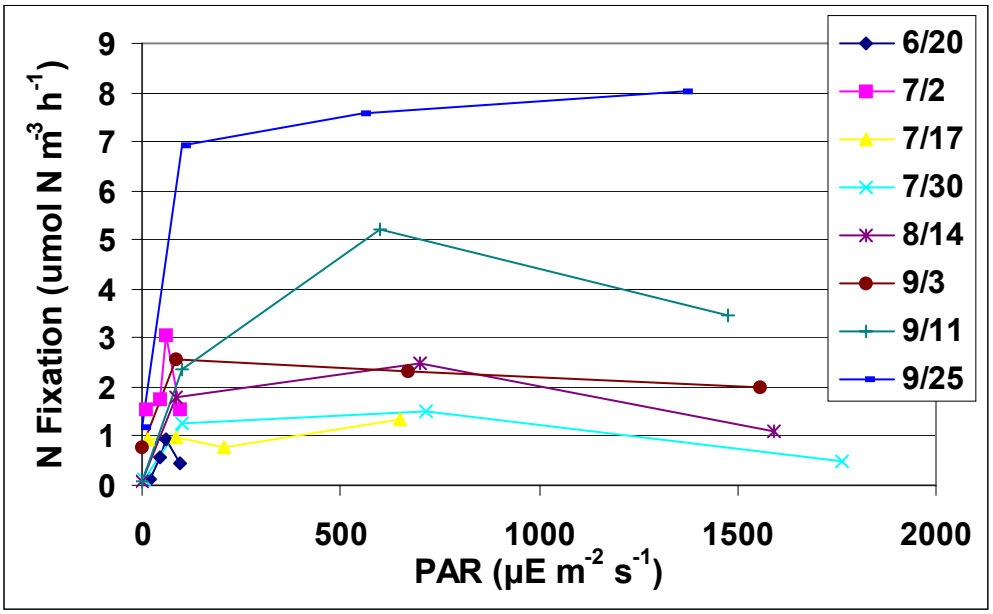


Fig 37 Nitrogen fixation at South station during 2002.



Fixation rates obtained on specific dates correspond well to Cyanophyte abundance at each of the five stations. The highest rates we obtained occurred during incubations on September 3 (Fig. 38). If we compare these rates to the amount of *Aphanizomenon* at each location (Fig. 39), we find that M has the highest fixation rates and the highest abundance of *Aphanizomenon*. Both fixation rates and abundance of *Aphanizomenon* decrease in order from E, W, S and N stations.

Fig.38 Nitrogen fixation rates obtained on September 3, 2002 (from Fig. 34)

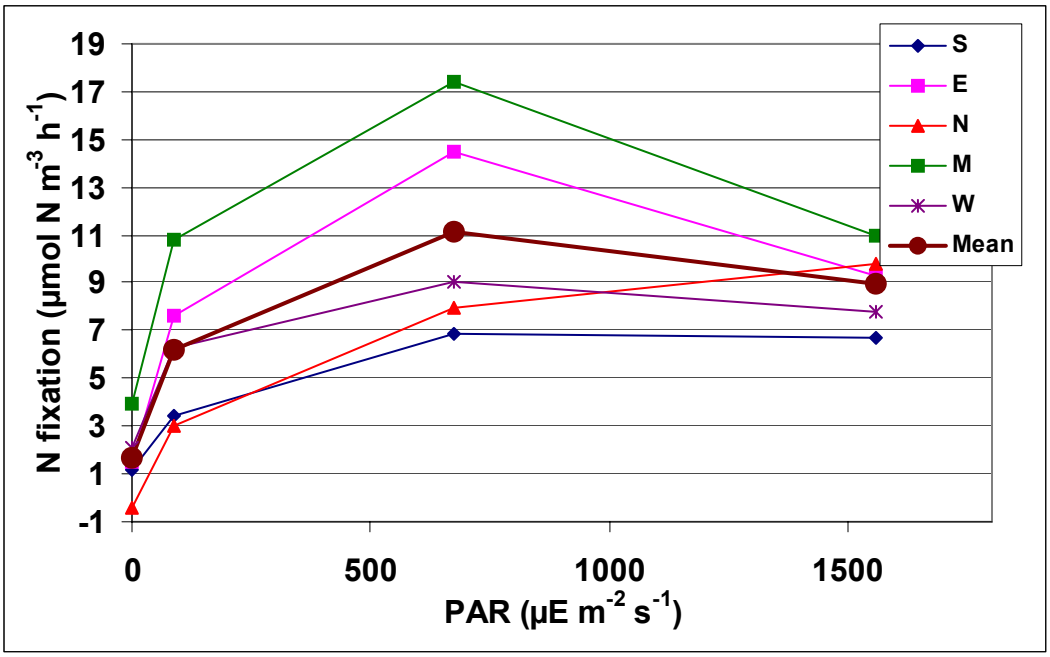
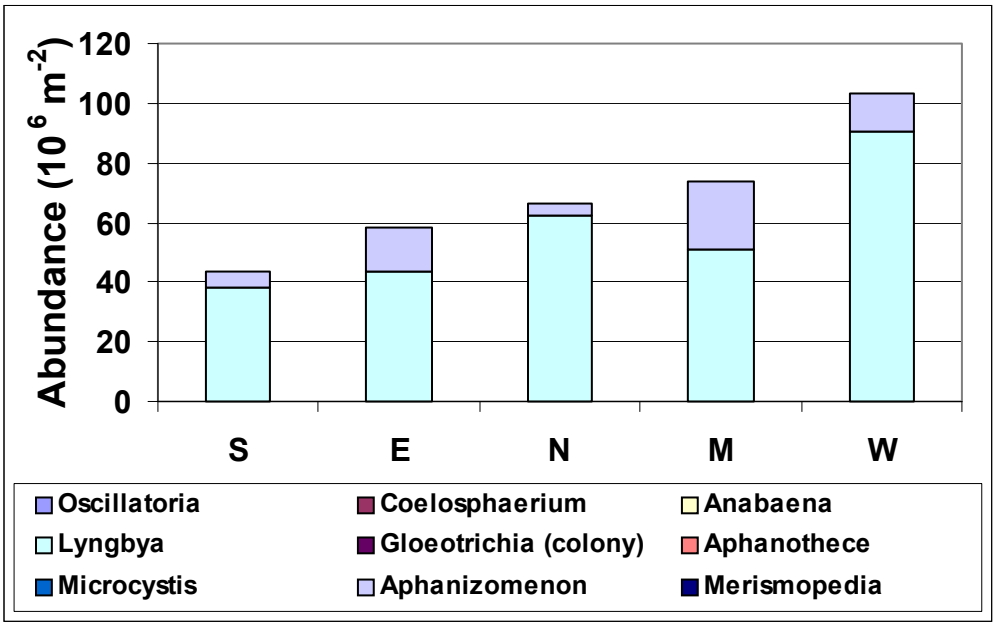


Fig. 39 Cyanophyte abundance on September 3, 2002.



The same comparison on a date with a low fixation rate, indicated a similar trend; 20 June had the 6th ranking fixation rate out of 8 trials and is very close to values obtained on July 17 and 30. Values up to 10 µmol N fixed m^{-3} h^{-1} were obtained at W and descended in order of M, N, E, S (Fig 40). At this time of year, *Aphanizomenon* was not present, but colonies of *Gloeotrichia* were abundant around the lake. Abundance of

this genera of cyanophyte also correlated with fixation rates. *Gloeotrichia* was most abundant at M followed by W, N, E, and S (Fig. 41).

Fig. 40 Nitrogen fixation rates on June 20, 2002 (from Fig. 34).

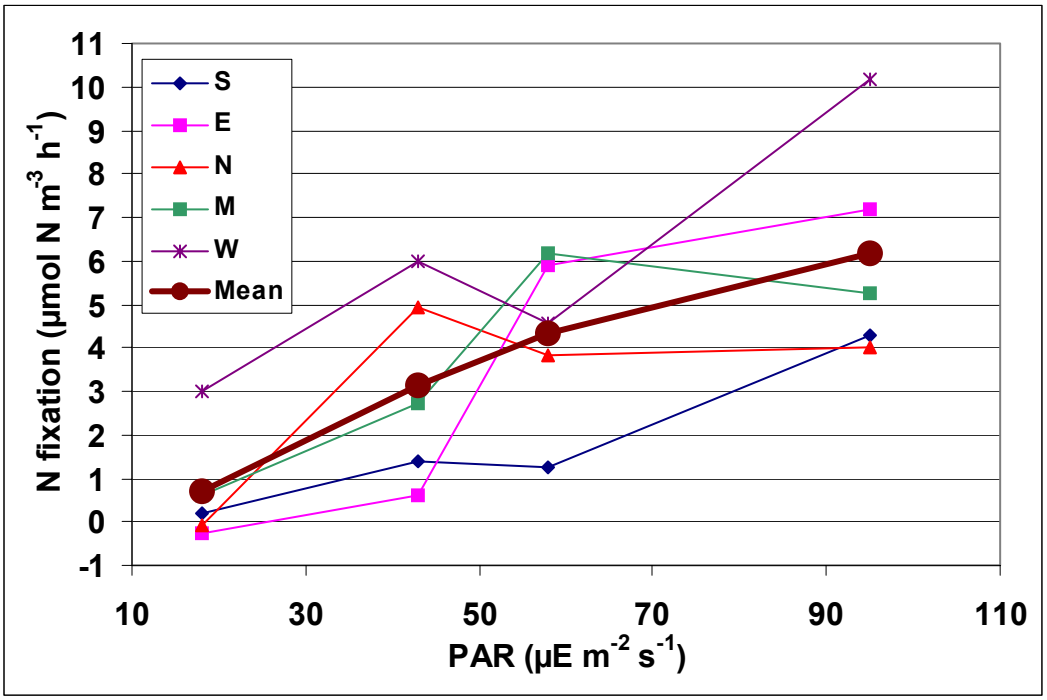
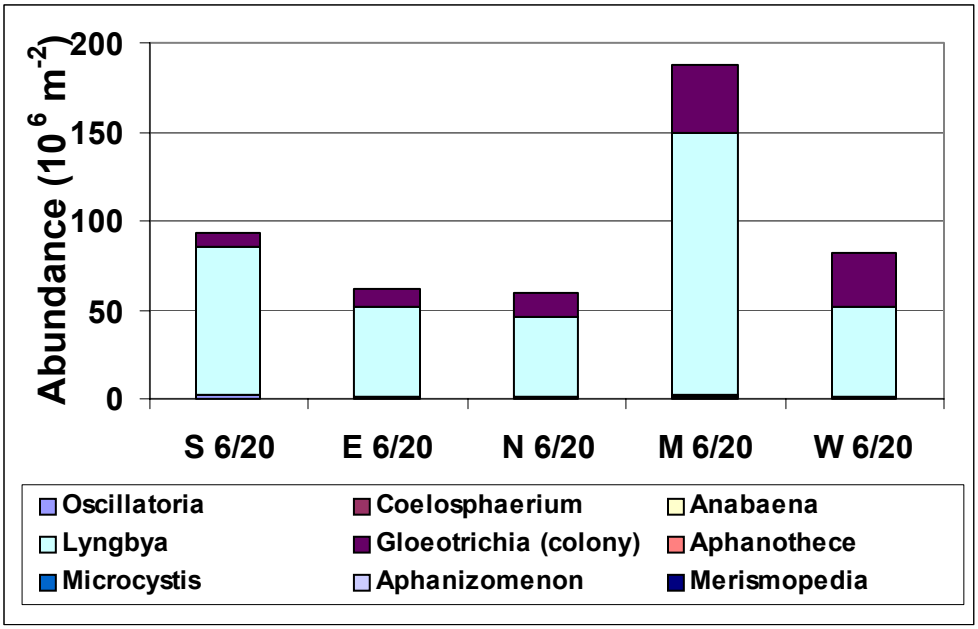


Fig. 41 Cyanophyte abundance on June 20, 2002.



Phytoplankton abundance was lowest on the 14 August sampling date. At this time, a mixture of *Gloeotrichia* and *Aphanizomenon* occured at stations within the lake. *Aphanizomenon* was present at S, E, M and W, while *Gloeotrichia* was present at E, M and W (Fig. 42). Fixation rates were highest at the M station (Fig.43) which had the highest abundance of *Aphanizomenon* and the 2nd highest abundance of *Gloeotrichia*. West had the 2nd highest fixation rates and the highest abundance of *Gloeotrichia* and the 2nd highest abundance of *Aphanizomenon*. E had the 3rd highest abundance of both genera and was approximately 3rd highest in measured nitrogen fixation rates. However, E shows a higher fixation rate than W at the mid PAR range. North shows a rather inexplicably high fixation rate as neither *Gloeotrichia* nor *Aphanizomenon* was counted in the sample. However, the nitrogen fixer, *Anabaena,* was present at N, as well as S and M, and may contribute substantially to fixation rates. The fixation at South was low and can be explained by the low abundance of *Aphanizomenon* present there.

Fig.42 Cyanophyte abundance on August 14, 2002.

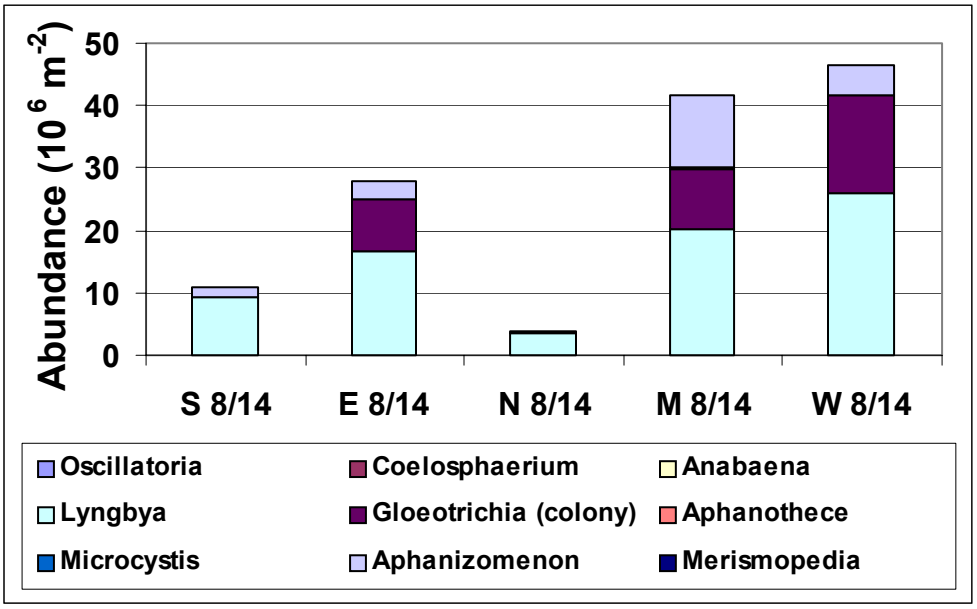
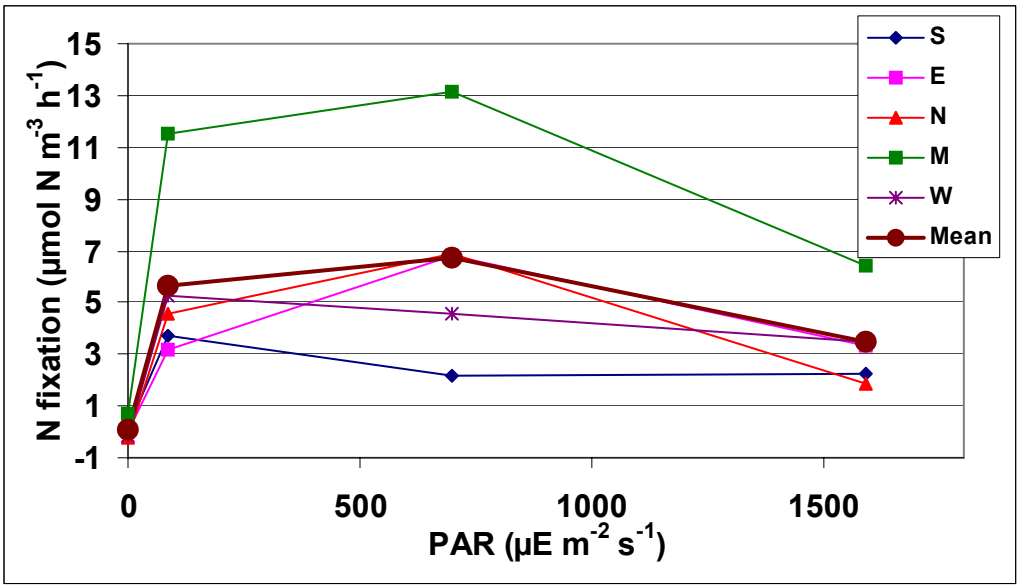


Fig. 43 Nitrogen Fixation rates on August 14, 2002 (from Fig. 34).

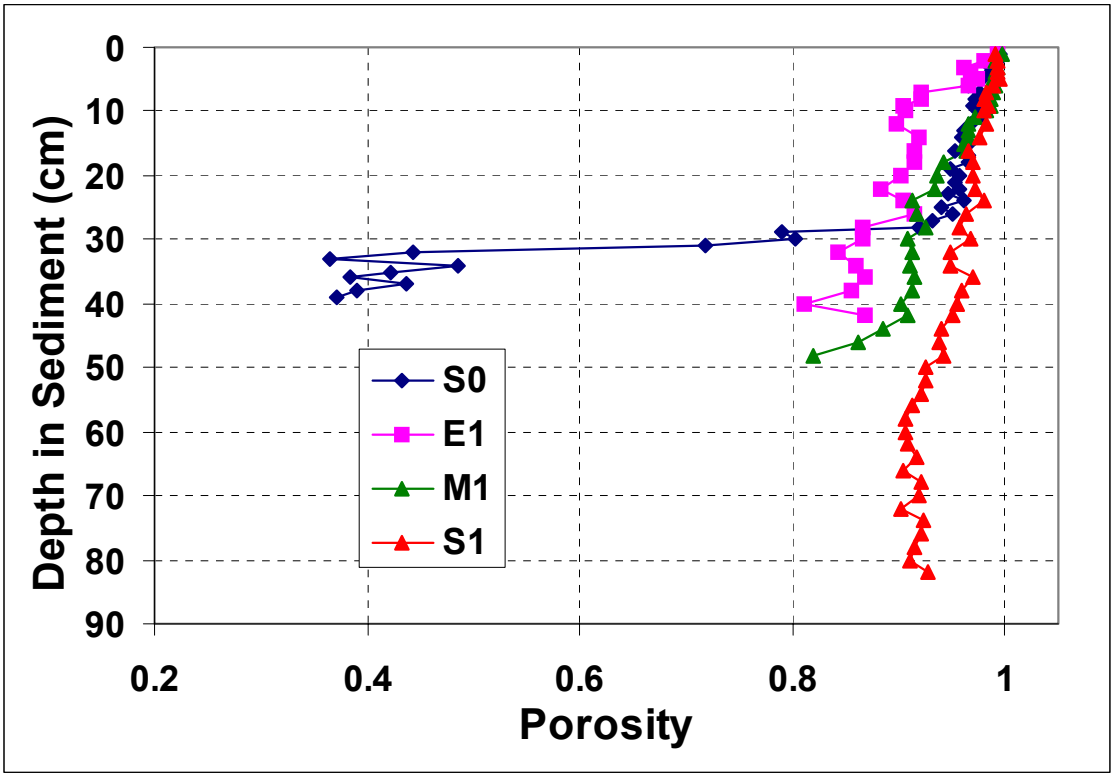


Sediment cores

Four sediment cores were collected from Crowley Lake during September 2002. One core was collected at the S station on 15 September (S0, 40 cm length) and one core from each of stations S (S1, 82 cm length), E (E1, 42 cm length) and M (M1, 48 cm length) was collected on 30 September. The sediments were amorphous, dark-green to black in color characteristic of gyttja. There was little evidence of lamination and they were highly porous.

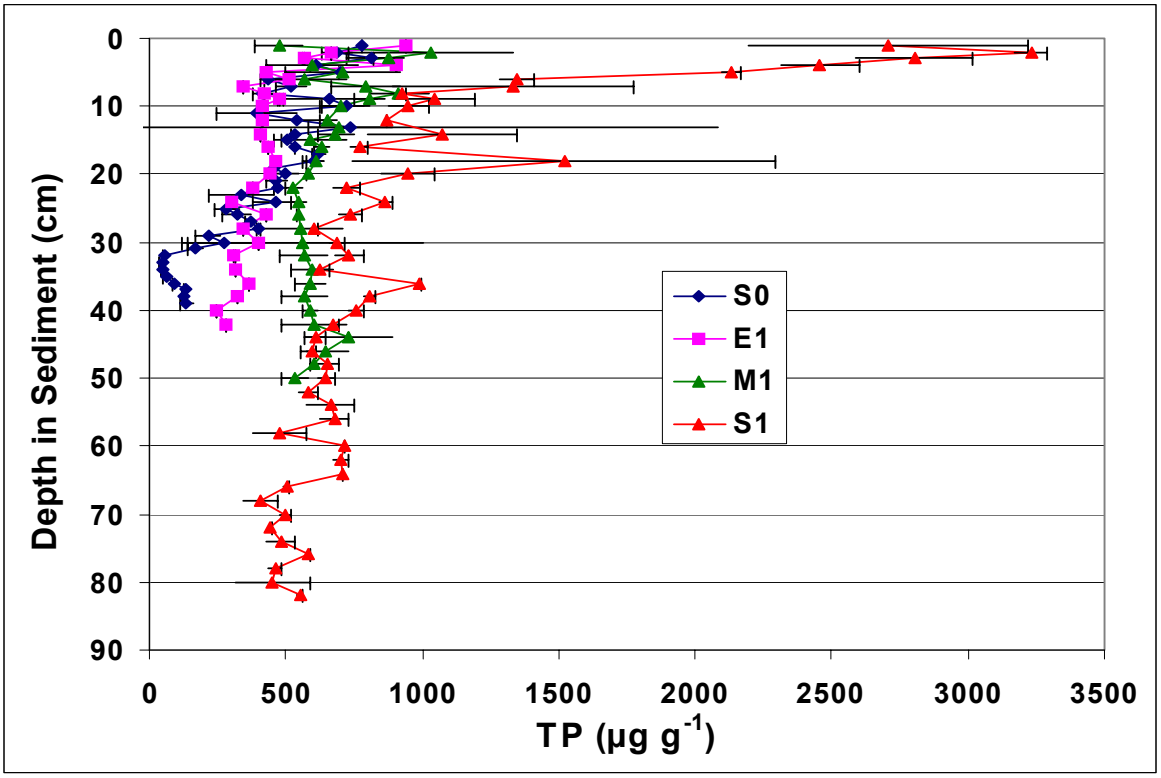
Porosity ranged from 80 to 99.5 (Fig. 44) in all but the bottom 10 cm of the first core (S0) taken at the S station. Below 27 m, core S0 showed a distinct drop in porosity. This was reflected in the texture of the dry sediment, which became quite gravelly at this point whereas all other cores remained quite fine. E1 was noticeably less porous, dropping below 90% at 7m where porosity continued to be approximately 10% lower than that of other cores. The bottom 4 cm of M1 decreased in porosity from ~0.9 to 0.81. In contrast, porosity in the bottom 30 cm of the longest core (S1) remained near 0.9.

Fig. 44 Sediment porosity



Total phosphorus content of the core generally ranged from 300-1000 µg g^{-1} (Fig. 45). It was significantly lower (50-140 µg g^{-1}) in the bottom 8 cm of S0, corresponding to the low porosities observed there. TP was markedly higher (1300-2700 µg g^{-1}) in the upper 7 cm of S1. Except for these two exceptions, the TP profiles were relatively constant with depth. A slight general decrease with depth below 10 cm (7.2 µg g^{-1} cm^{-1}, r^2=0.53) is apparent within S1.

Fig. 45 Total phosphorus (TP) in Crowley Lake sediment cores.



Non-apatite inorganic phosphorus (NAIP) generally ranged from 50-450 µg g^{-1}, except that as with TP it was lower in the bottom 10 cm of S0 and higher in the upper portion of S1 (Fig. 46). At the E station (E1) it was nearly uniform at 50-70 µg g^{-1} except for the upper 7 cm where it increased to 220-230 µg g^{-1}. This increase in the upper 7 cm was more pronounced in S1 where values increased to over 1500 µg g^{-1}. Both southern cores and the core from station M showed a slight but general decrease with depth. The mean ratio of NAIP:TP was 0.36 (1SE, 0.13) and showed no trend with depth.

NAIP is generally considered the biologically active sediment phosphorus. Plotted by volume (5-pt running mean), it displays only a slight increasing trend with depth (0.09 µg cm^{-4}, r^2=0.41) in S1, while TP increases more sharply with depth (1.2 µg cm^{-4}, r^2=0.85)(Fig. 47). The mean NAIP:TP ratio in S1 was 0.2, generally decreasing with depth.

Porewater ammonia differed markedly among the three cores (Fig. 48). In S1, the longest and most porous core, NH_4 concentrations increased from 600 µM in the top few cm to 1600-1750 µM in the bottom 20 cm. In the other south station core, S0, the values only increased to near 400 µM near the bottom. Core M1 concentrations were similar to S0 except that the upper 10 cm gradients were distinctly different. As with porosity and

Fig. 46 Non-apatite inorganic phosphorus (NAIP) in Crowley Lake sediment cores

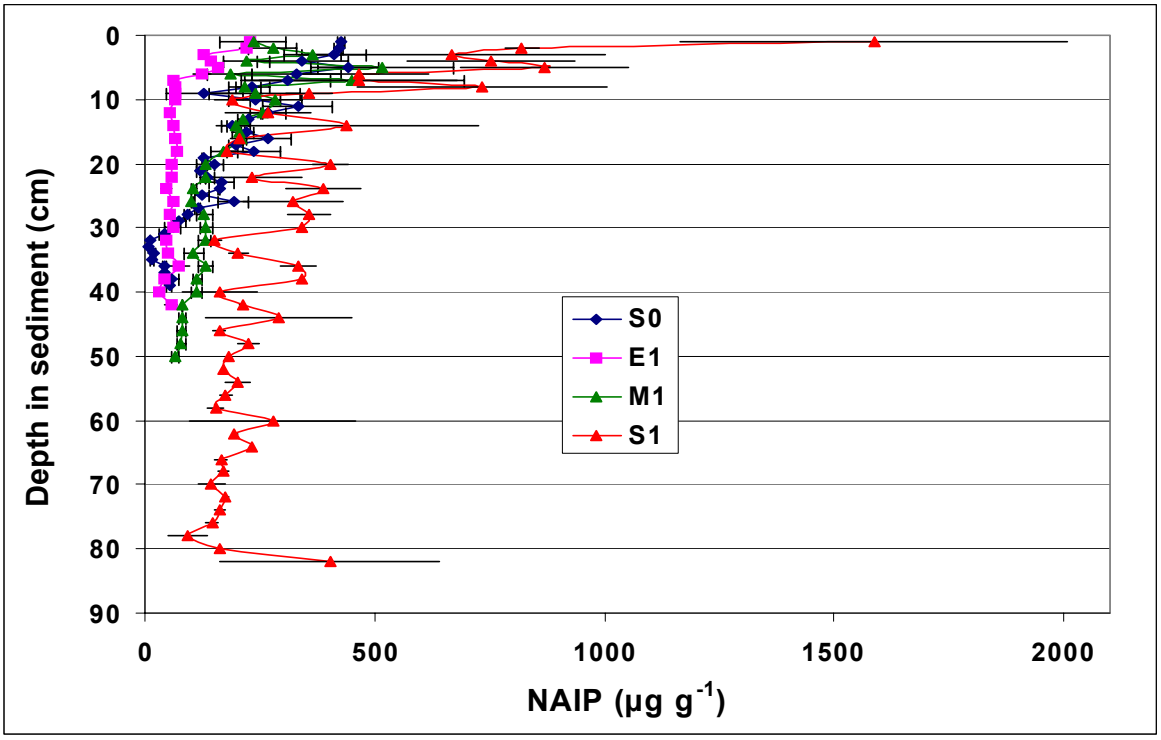


Fig. 47 NAIP and TP in Crowley Lake sediment cores (by volume). Five point mean.

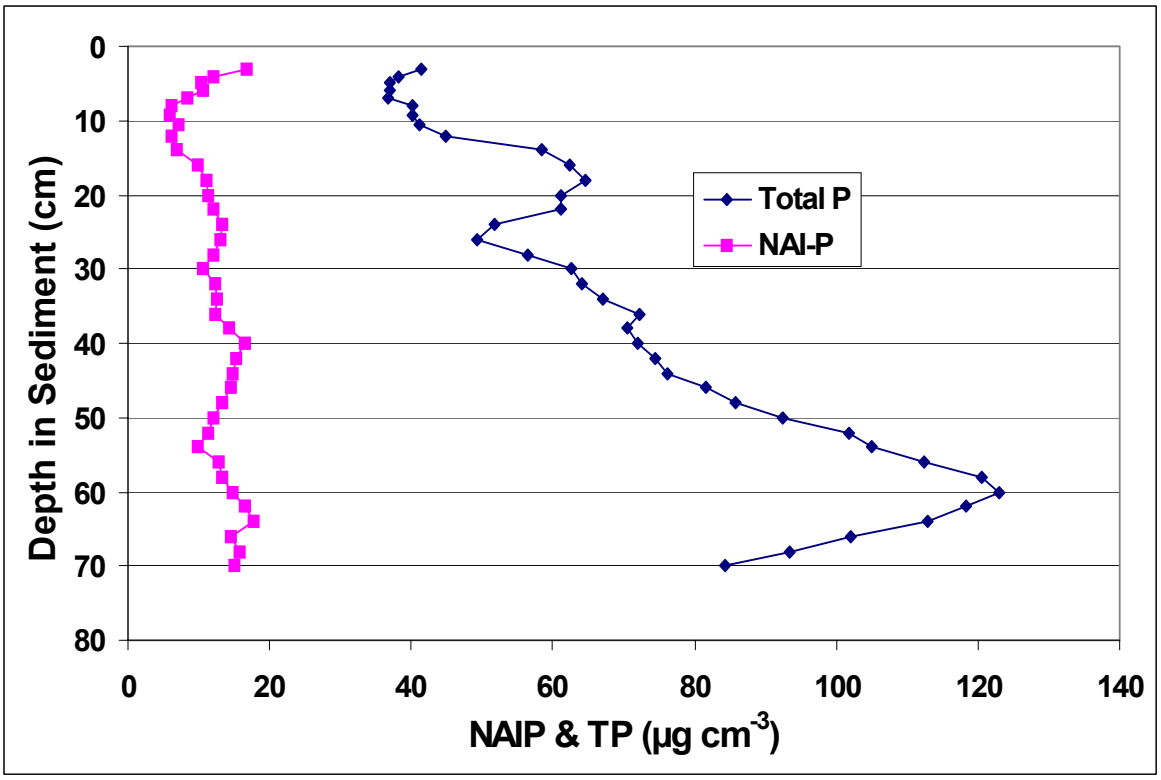
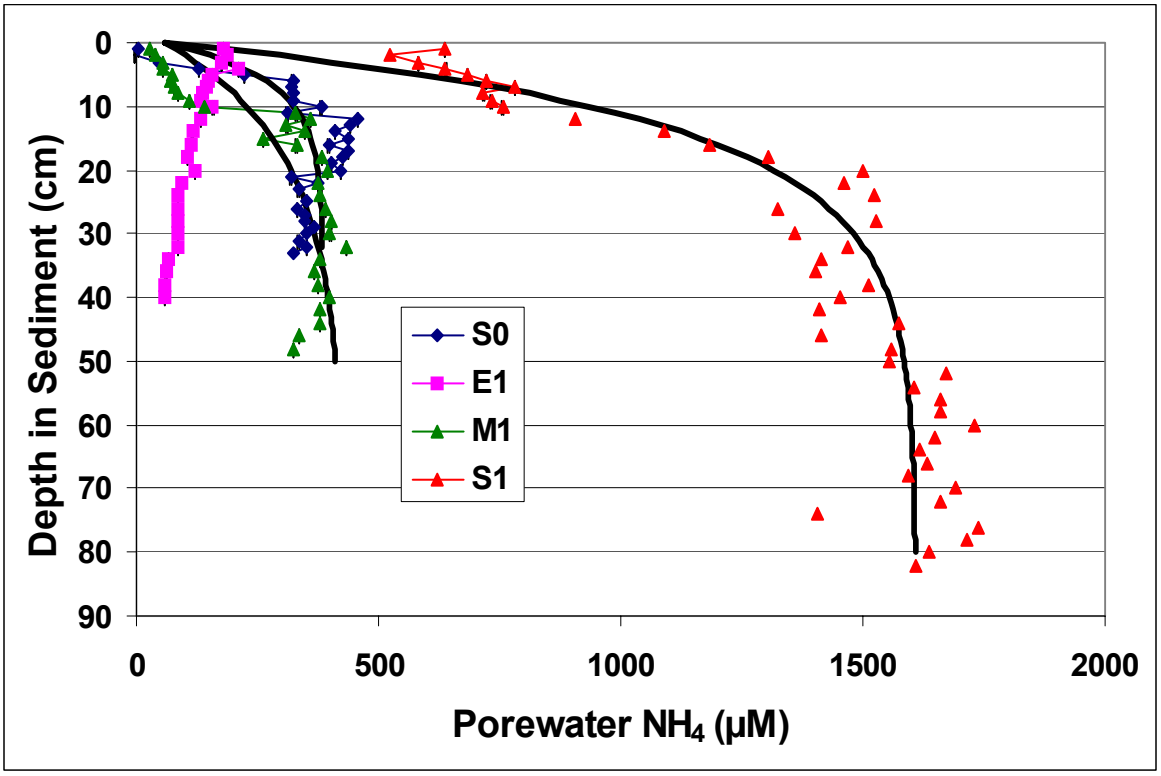


Fig. 48 Porewater ammonia profiles and fitted concentrations



TP, porewater NH_4 at E1 was distinctly different from the other cores and decreased from near 200 µM near the surface to about 60 µM at 40 cm. As this core appears anomalous, it is not further considered here.

The porewater concentrations were modeled to calculate gradients at the sediment water interface (Fig. 48). Estimated gradients ranged from 56 to 225 µM cm^{-1} (Table 4). S0, S1, and M1 the porewater profiles and fitted lines suggest a marked change in the upper 10 cm that is poorly modeled. Therefore, these porewater concentrations in these three profiles were also fitted to just the top 10 cm concentrations (Fig 49.) resulting in estimated concentration gradients of 15 to 78 µM cm^{-1}at the sediment-water interface. Although the concentration gradients derived from the upper 10 cm probably provide more accurate estimates, we calculated the estimated flux rates for both as an indication of the uncertainty in the estimates. Estimated NH_4 fluxes out of the sediments based on concentration gradients using the upper 10 cm samples ranged from 0.2 in M1 to 1.1 mmol m^{-2} d^{-1} at S0 with S1 lying in the middle of the range at 0.7 mmol m^{-2} d^{-1}.

Fig. 49 Porewater ammonia profiles and concentrations in the upper 10 cm fitted

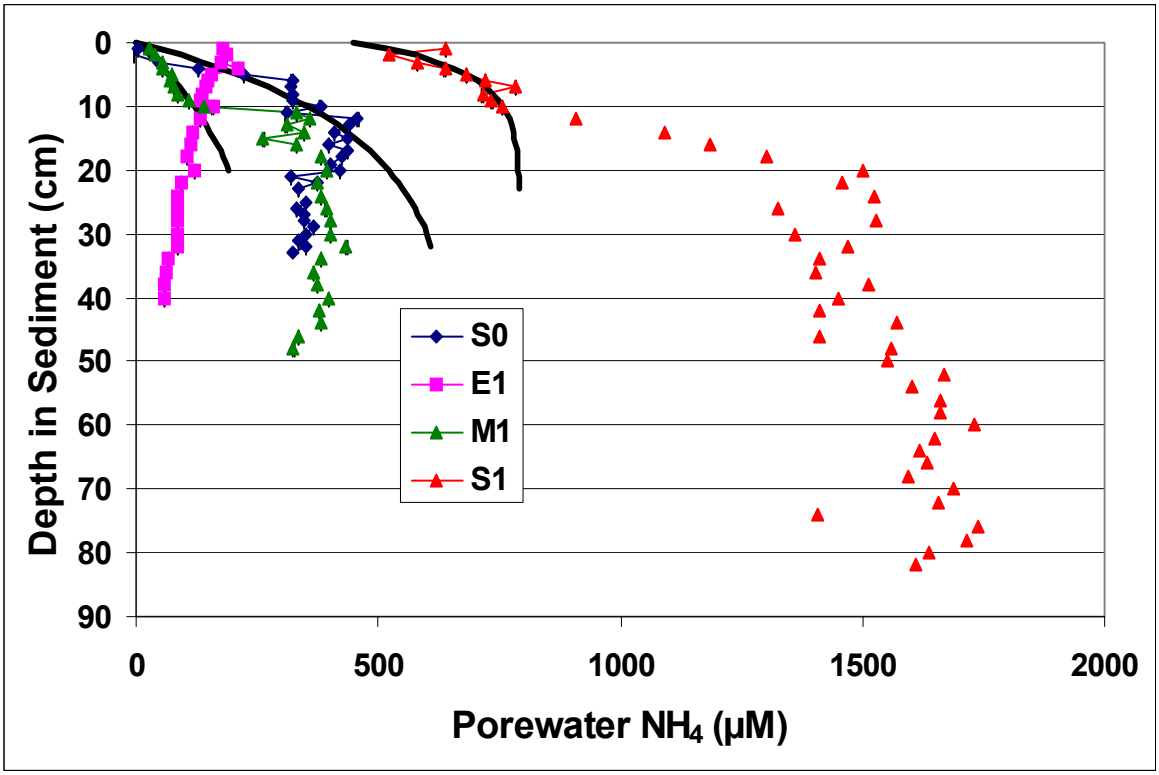
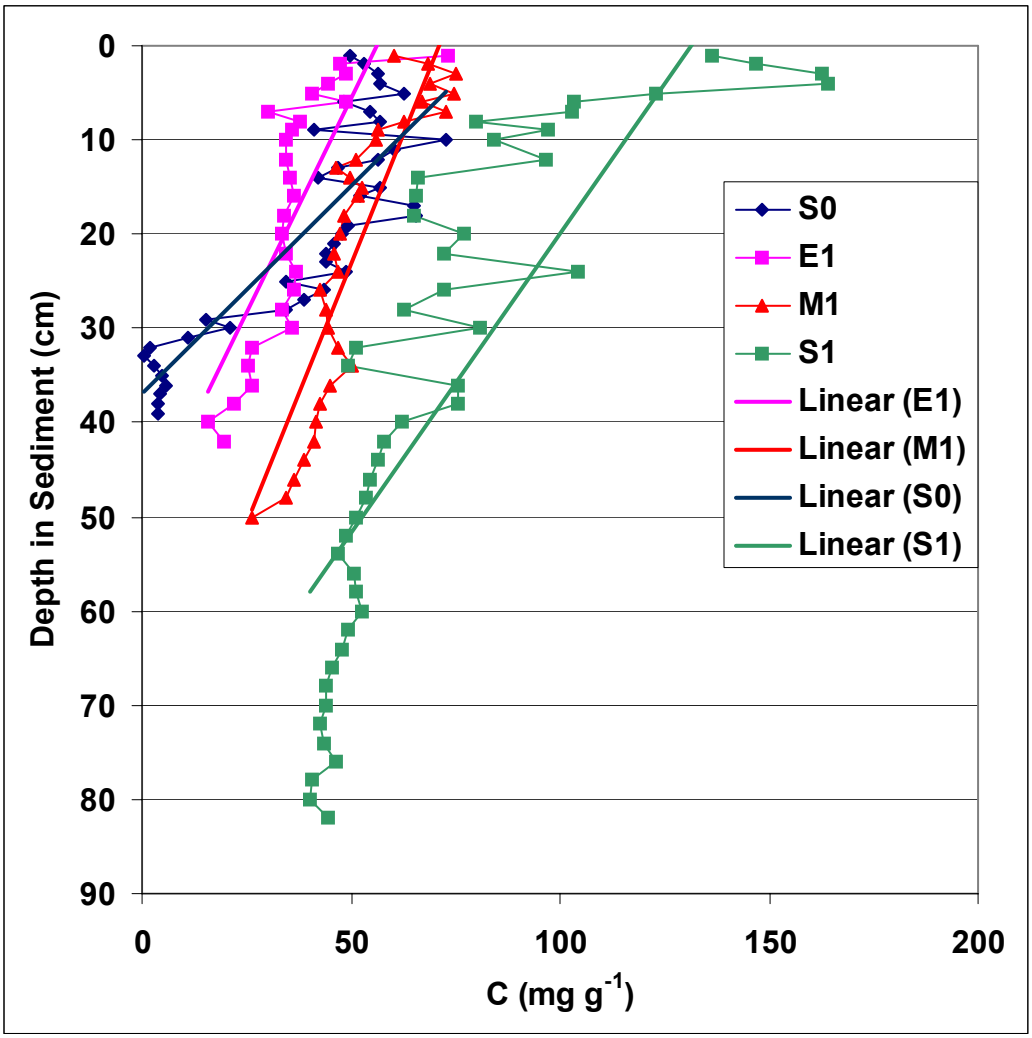


Table 4 Estimated sediment-water ammonia fluxes

Core	Porosity (upper 10 cm)	NH4 gradient µM cm^{-1}	NH4 Flux mmol m^{-2} d^{-1}
Based on sediment-water gradients using only the upper 10 cm value			
S0	0.98	77.8	1.1
S1	0.99	50.3	0.7
M1	0.98	15.3	0.2
Based on sediment-water gradients using all data			
S0	0.98	56	0.8
S1	0.99	129	1.8
M1	0.98	225	3.2

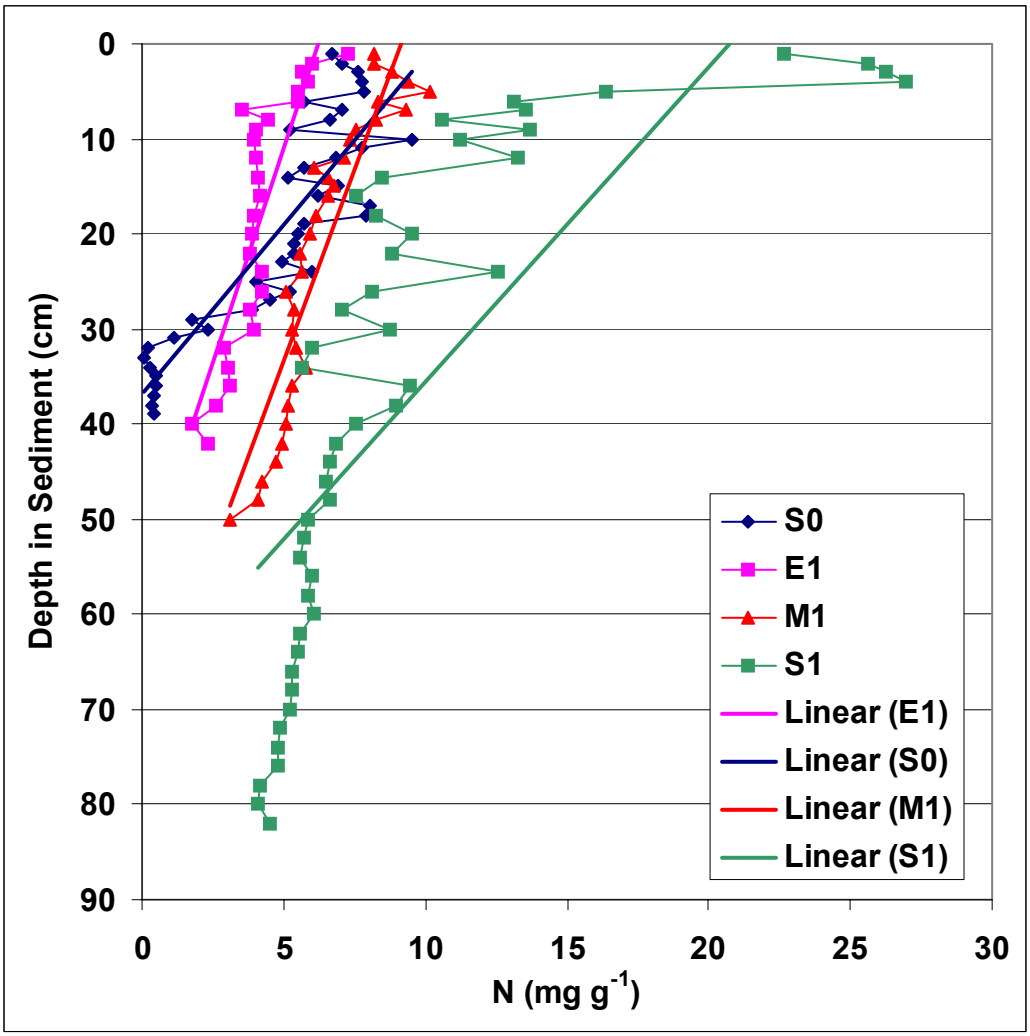
Sediment particulate inorganic carbon ranged from 11.1 - 164.2 mg g^{-1} with the exception of the bottom 8 cm of S0 where values declined to 0.69 mg g^{-1} (Fig. 50). As seen with TP and NAI-P, there is a higher PC content in the upper few cm of S1. There was an overall trend for carbon to decrease with depth (r^2 = 0.6106 – 0.7757) (Fig. 50).

Fig. 50 Particulate carbon content in sediment cores.



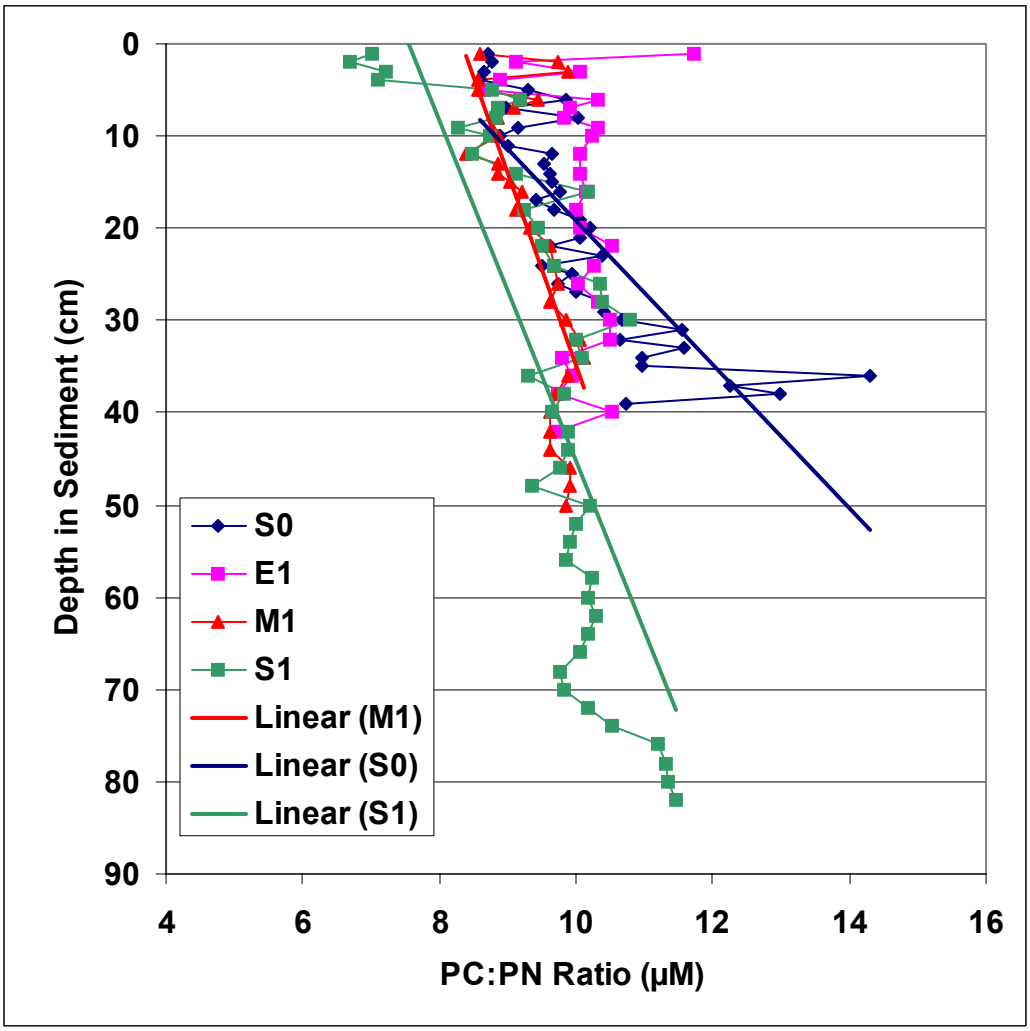
Sediment inorganic nitrogen ranged from .07 to 27 mg g^{-1} sediment with the characteristic decrease at the bottom of S0 and increase in the top of S1 (Fig. 51). Nitrogen also showed a decreasing trend with depth (r^2 = 0.58 – 0.83). Removing the top 5 cm of S1 increases the r^2 = 0.58 to r^2 = 0.75.

Fig. 51 Particulate nitrogen content in sediment cores.



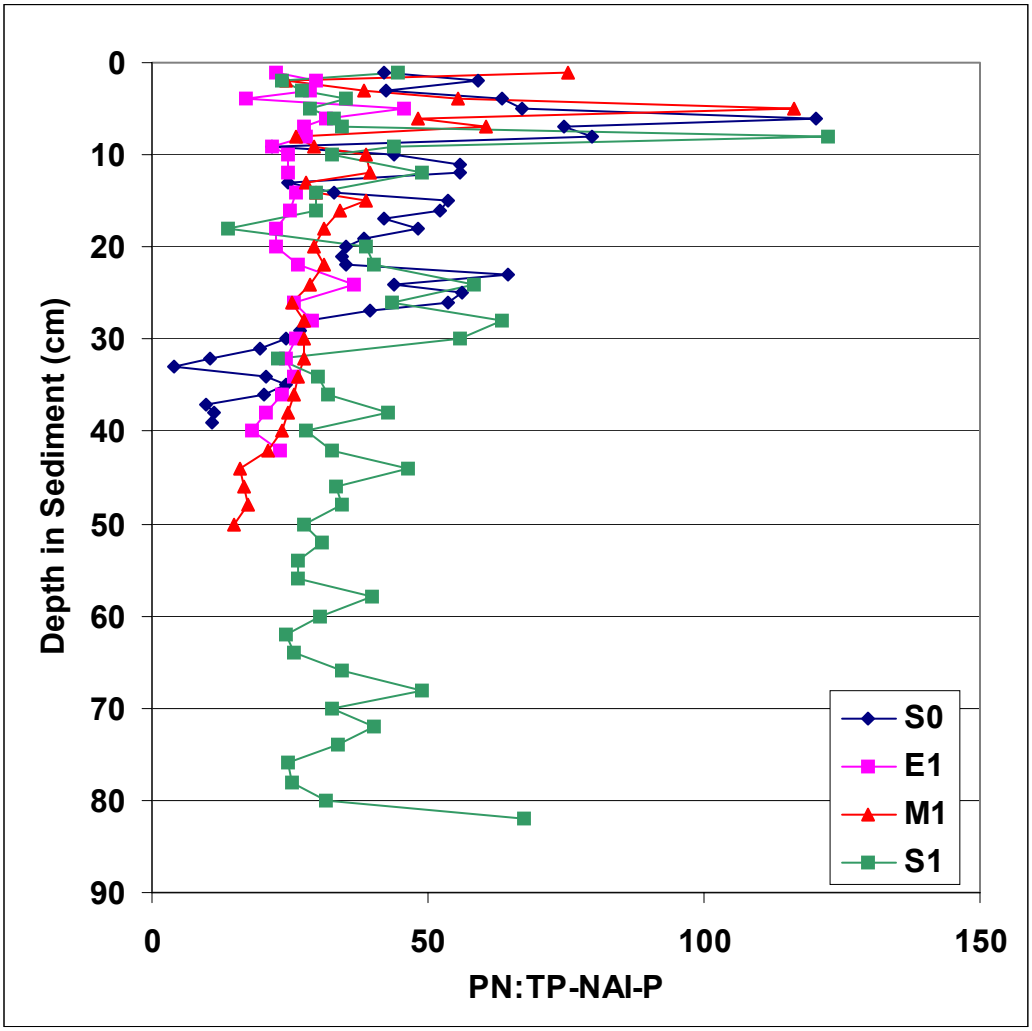
The Sediment C:N ratio varied from 6.69 to 14.32 (Fig. 52). Mean values ranged from 9.38 to 10.11 with an overall mean of 9.79 (1SE, 0.08), which is slightly higher than the Redfield ratio of 6.6 (Wetzel, 2001). Ratios are relatively constant (possibly a slight increase in S0, M1, S1) with depth indicating that nitrogen is decreasing with depth at a faster rate than carbon.

Fig. 52 Molar ratio of particulate carbon to particulate nitrogen in sediment cores.



The sediment inorganic N:TP-NAI-P ratio varied from 3.89 – 122.47 (Fig.53). The mean ratios ranged from 26.04 – 40.76 with an overall average of 35.55 (1SE, 1.56) which is more than double the redfield ratio of 16 (Wetzel, 2001). There is no trend below 10 cm.

Fig. 53 Sediment nitrogen to total phosphorus less non-apatite inorganic phosphorus for sediment cores.



DISCUSSION

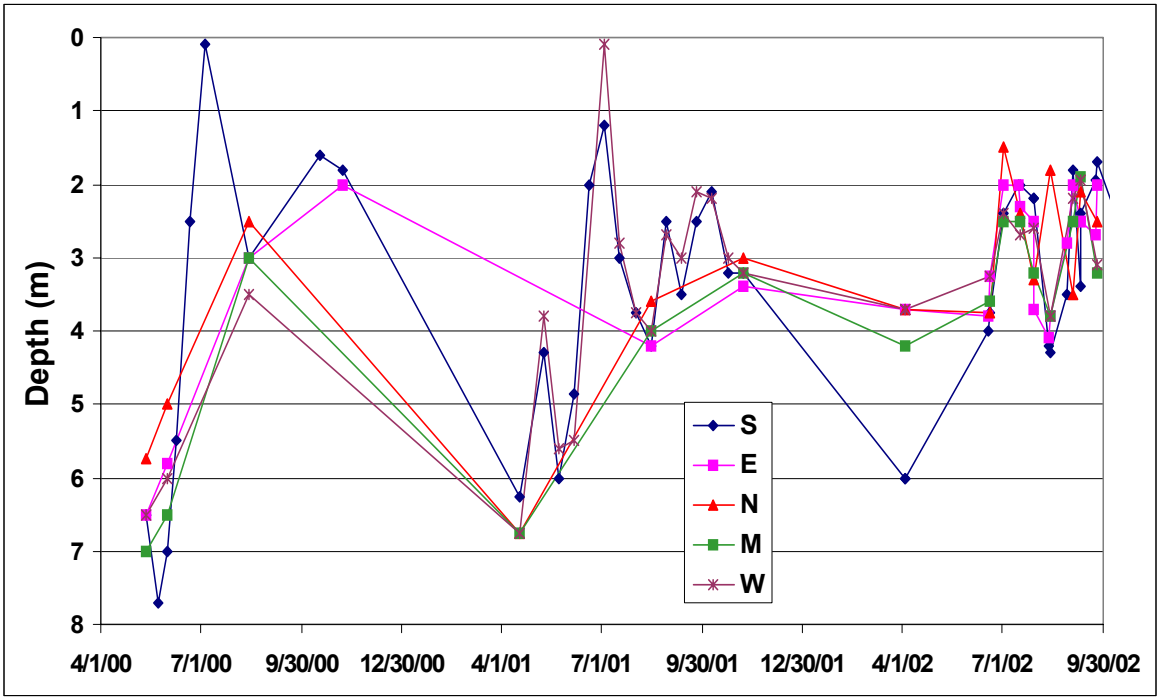
Crowley Lake (Long Valley Reservoir) is located in southern Mono County in the Long Valley Caldera at an elevation of ~2062 m. Created in 1941 with construction of the Long Valley Dam, it is a moderately sized reservoir with an area of 15.6 km^2, volume of 0.135 km^3 and a mean depth of 8.6 m. The lake undergoes a regular seasonal pattern of thermal stratification typical of temperate, dimictic lakes. Ice cover disappears in late March – mid-April resulting in spring turnover and a brief period when the entire water column is well-mixed and near 4°C. Warming air temperatures and increasing insolation result in heating and the rapid onset of seasonal thermal stratification during April - early May. The epilimnion (upper mixed layer) warms rapidly during May through July, while

the hypolimnion warms somewhat more slowly. With cooler air temperatures and decreased insolation, the epilimnion begins to cool in late August. Further cooling in autumn results in a period of mixing prior to becoming ice-covered in late December. Associated with this dimictic seasonal mixing regime are changes in the supply rates and availability of nutrients with consequent changes in phytoplankton community.

Plankton characteristics during summer 2002

During June through September, 2002, chlorophyll *a,* the rapid development of an anoxic hypolimnion, and Secchi depths were all indicative of the eutrophic status of Crowley Lake. The phytoplankton abundance, as measured by chlorophyll *a*, peaked in early July at 35 to 48 µg Chl *a* l^{-1}, followed by a mid-August decrease to 5-12 followed by an early September peak of 28-38 µg Chl *a* l^{-1}. This summer pattern and magnitude of Chl *a* concentration is remarkably similar to that observed at a deep water station in 1964 (Warner 1965). Secchi depths were generally between 2 to 4 m throughout the summer (Fig.54). An accurate comparison of transparency to 2000 and 2001 is not possible due to the sparse sampling in those years and the rapid temporal changes in transparency. The 2002 summer transparencies were generally within the range observed during 2000 and 2001. The 2002 spring decrease appears to have occurred somewhat earlier in 2002. Ice-off was quite early (mid-March) in 2002 and may account for this difference. Also, during 2002, no algal blooms in which Secchi depths were reduced to near zero as observed in 2000 and 2001 were encountered. However, blooms are spatially and temporally highly variable and no confidence can be placed in the significance of differences between only a few stations.

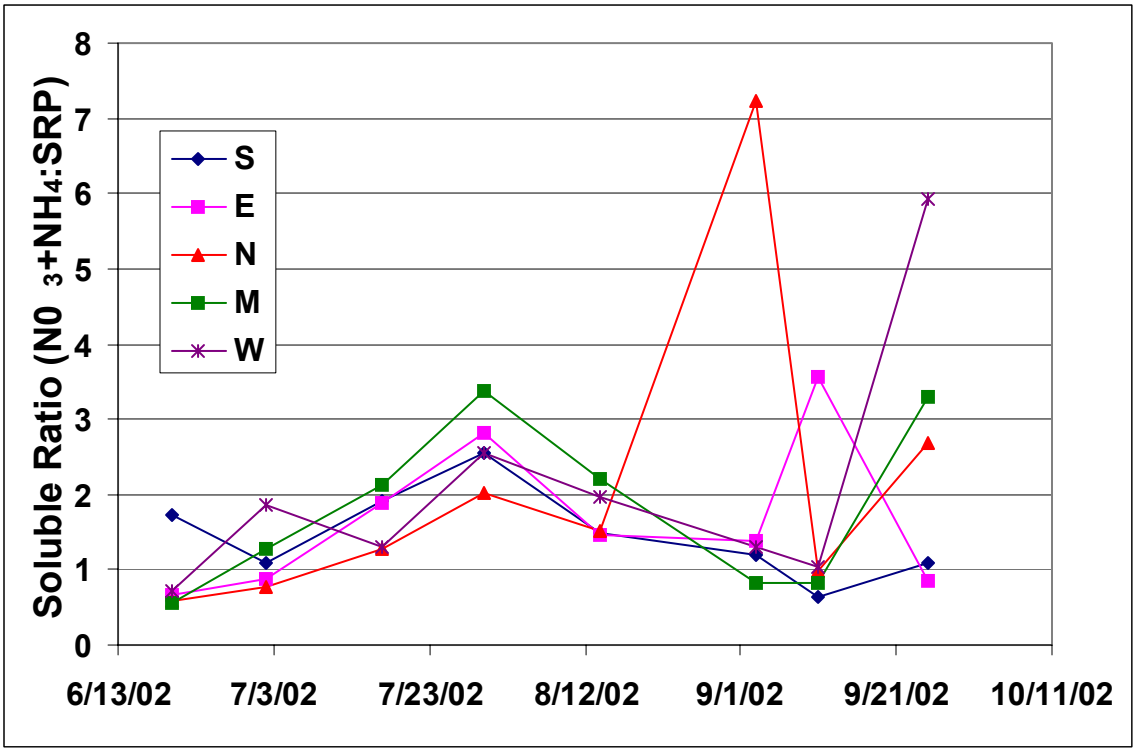
Fig. 54 Comparison of Secchi depths between this study and 2000 and 2001 surveys.



Marine phytoplankton show a relatively constant ratio of C:N:P of 106:16:1. This is known as the Redfield Ratio and is attributed to the nutrient sufficient growth conditions of marine plankton and the homogeneous and stable nature of the oceans. In freshwater, N:P ratios in plankton are strongly correlated with N:P loading rates and deviations from the Redfield Ratio provide an indication of the type and extent of nutrient limitation (Wetzel, 2001).

The total phosphorus in the upper 5 m of the water column was ~3 µM (0.09 mg l^{-1}) early in June and July, decreased slightly in August and then increased to ~2.5 µM in September. While these values lie within the range of those observed during a 1982 study (Melack and Lesack 1982), we did not observe any of the elevated mixed-layer concentrations (8-10 µM) noted in that study. Ammonia was generally less than 1µM except on the final survey on 25 September and NO_3 was generally ~0.2 µM. Thus, dissolved inorganic nitrogen to phosphorus ratio was well below the molar Redfield ratio of 16, (Wetzel, 2001) throughout the study period (Fig. 55). The molar ratio of dissolved inorganic N (NO_3 + NH_4) to SRP was low (<3) throughout the summer except for the outlier on 3 September at the N station and an increase observed on the last two sample dates. The ratio increased from ~1 in mid-June to a peak of 2-3.5 at the end of July before decreasing back to ~1 by mid-September.

Fig. 55 Dissolved inorganic N:P ratio in upper 5 m of the water column

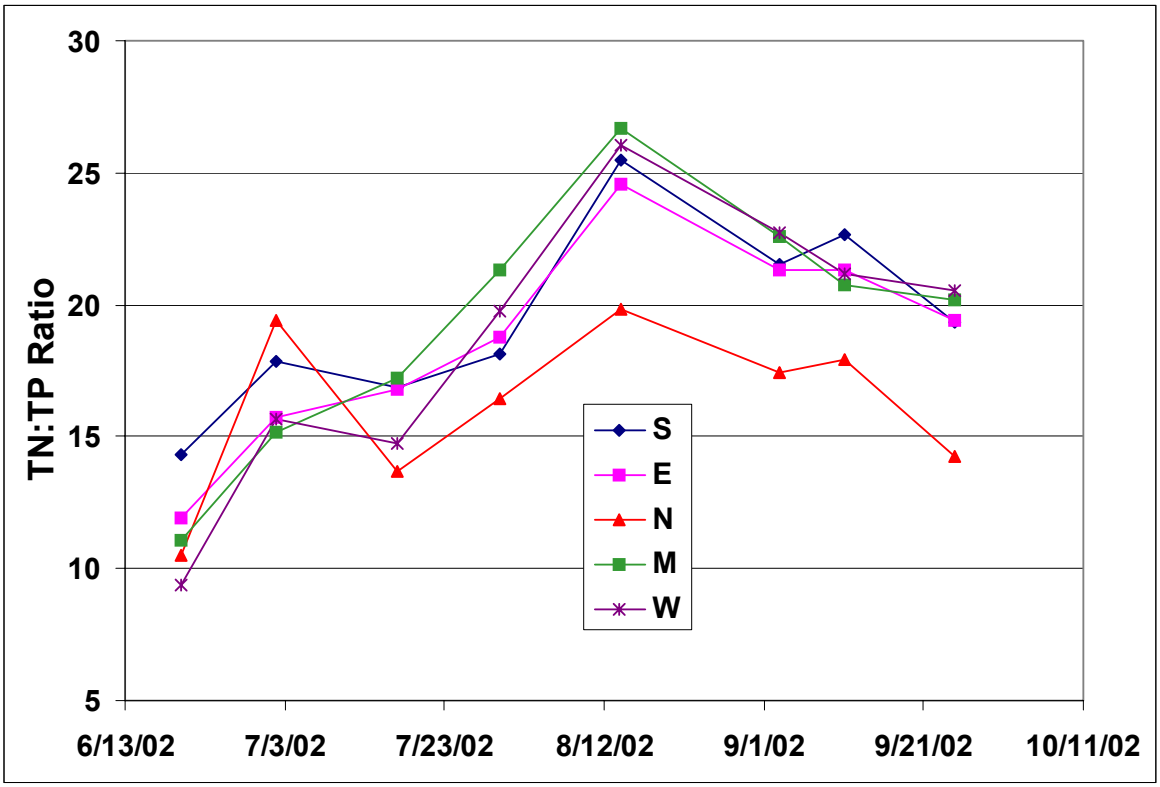


Given the overall rates of phosphorus loading to Crowley Lake, the molar ratio of TN to TP was surprisingly high through most of the summer with the lakewide mean value ranging from 9.4 to 26.7 (Fig. 56). The seasonal trend of the lakewide mean was a minimum value of 11.5 on 19 June, increasing to 24.5 in mid-August and then declining

to 18.7 by the end of September. Only on the June date were values significantly below the Redfield ratio of 16. As with TN and TP, the N station differed from the other four stations through the period beginning in late July. Consistently lower ratios at the N station reflect the high phosphorus loading associated with Owens River inflows. The overall summer mean concentration was 18.5 (±0.7, 1 SE). However, 2002 ratios are much higher compared to the ratios over a similar period in 2001. The ratios reported in 2001 indicate that the lake was most likely N limited throughout most that summer.

Because various pools of inorganic, organic, and particulate phosphorus and nitrogen may be recycled at different rates, dissolved inorganic pools may not accurately reflect the relative availability of nitrogen versus phosphorus. The use of particulate elemental ratios as a measure of nutrient limitation was developed with both marine (Goldman, 1980) and freshwater phytoplankton (Healey & Hendzel, 1980) and has become widely used. Molar carbon to nitrogen ratios of summer seston (planktonic particulates) in the upper 5 m ranged from 5 to 7 and were thus near the Redfield ratio of 6.6 (Fig. 57). There was little variation among stations and only a slight seasonal decrease from a lakewide mean of 6.7 on 19 June to 5.9 on 25 September. The overall summer mean was 6.2 (SE, 0.1; n, 40).

Fig. 56 Ratio of total N to total P in upper 5 m of the water column



The molar nitrogen to phosphorus ratio of summer seston ranged from 14.5 to 83.3 and would suggest phosphorus-limited growth by the plankton during most of the summer (Fig. 58). Only at the N station on 19 June and on the 15 August sampling date did sestonic N:P ratios approach the Redfield ratio of 16. A marked increase from

lakewide average of 21-31 during June-August occurred following the 15 August survey as September values climbed to 50-83. The overall mean of June-August samples was 28 (SE, 3) while the September mean of three sample dates was 66 (SE, 2). Both the C:N and N:P ratios are surprising given the low N:P loading ratios. They suggest no nitrogen deficiency and severe phosphorus limitation during September.

Fig. 57 C:N ratio of seston in upper 5 m of water column

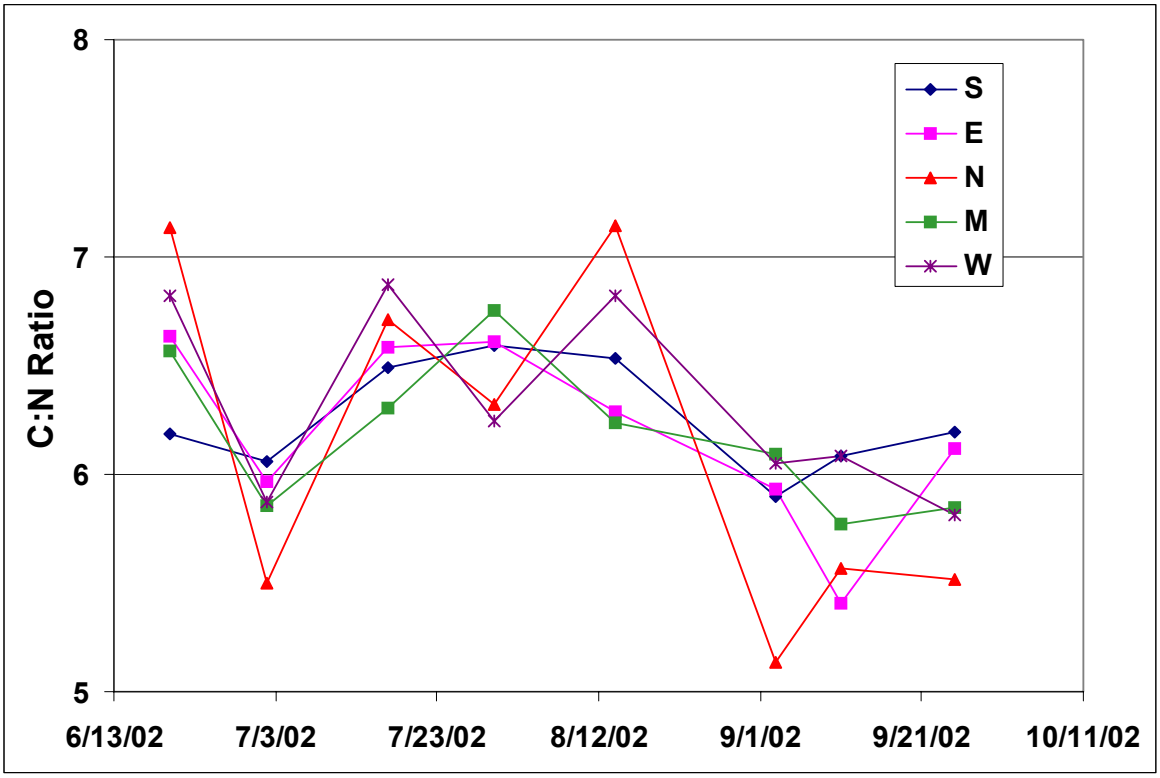
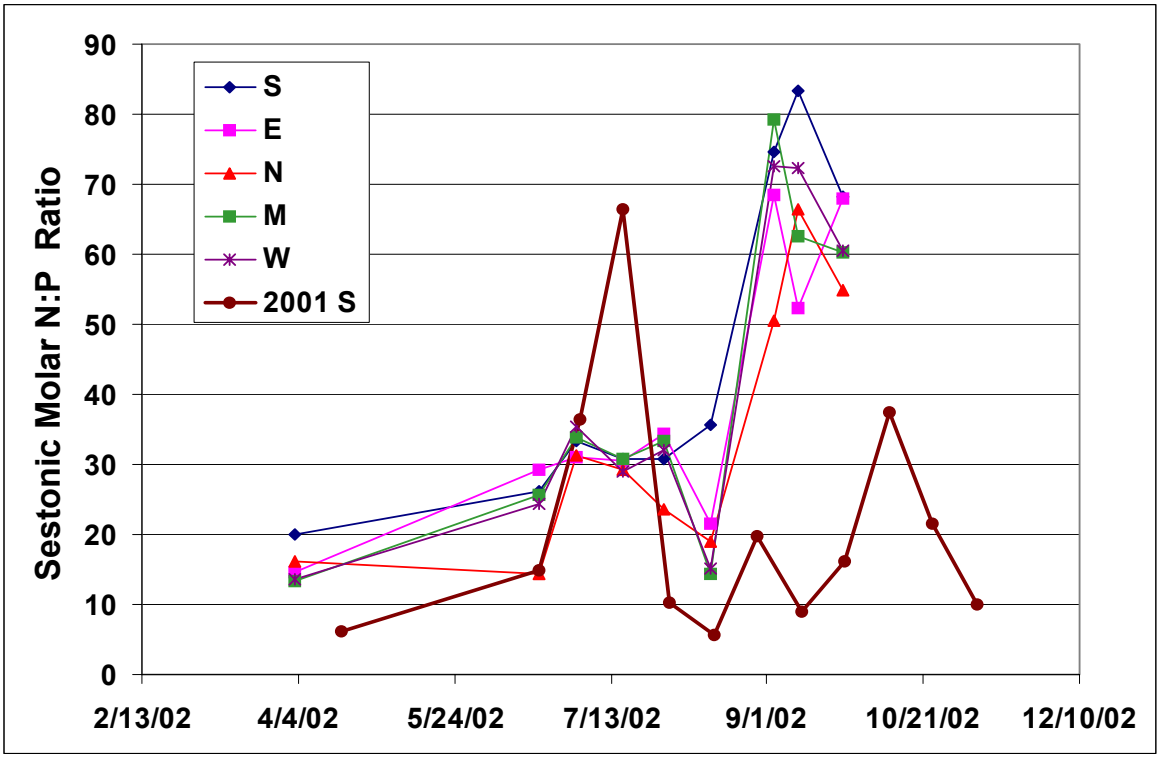


Fig. 58 N:P ratio of seston in upper 5 m of water column



Internal nitrogen loading and pelagic nitrogen fixation

Freshwater lakes are generally limited by phosphorus, in part, because nitrogen fixation by cyanobacteria (blue-green algae) is often able to relieve nitrogen limitation. Many factors, including nutrient availability, pH, light penetration, turbulence, temperature and zooplankton community structure, play a role in cyanophyte abundance and in turn, nitrogen fixation (MacKay & Elser, 1998; Elser, 1999; Paterson et al, 2002). As a result timing, intensity, predictability and species composition of blooms vary substantially (Elser, 1999). The presence of heterocystic cyanobacteria in Crowley Lake, the recurring algal blooms, and the low N:P loading ratios all suggest nitrogen fixation is an important part of the overall nitrogen budget. However, nitrogen fixation rates measured in pelagic samples during summer 2002 were low except for a slight increase to ~15 µM N m^{-3} h^{-1} in September. This increase coincided with the appearance of the nitrogen fixer, *Aphanizomenon flos-aquae*. *Aphanizomenon* has been reported previously in the lake as appearing late in the season (Melack and Lesack 1982, Warner 1965, EPA 1978). The nitrogen fixer, *Gloeotrichia*, was present during the early summer. Measured rates on the 20 June and 2 July may not fully represent the nitrogen fixing capacity of this cyanophyte due to the low PAR under which the samples were incubated. Studies show that cyanophytes will utilize ammonia, urea and nitrate from the water column before fixing nitrogen from the atmosphere (Presing et al, 2001). During midsummer, sestonic ratios suggest phosphorus rather than nitrogen limitation. *Aphanizomenon* appears in Crowley late in the season. The appearance of *Aphanizomenon* in autumn coincides with

lower N:P ratios. These low ratios favor cyanophytes not only due to the ability to fix nitrogen from the atmosphere but also due to their higher storage capacity of nitrogen compared to other algal species (MacKay & Elser, 1998).

The estimated lakewide nitrogen fixation from 3 April to 25 September was only ~0.6 g N m^{-2} y^{-1} or ~1.0 g N m^{-2} y^{-1} if you include the higher rate measured from a sample collected at the dock on 29 May. These annual rates lie in the median of 17 eutrophic lakes reported by Howarth et al. (1988). The surplus of measured exports over inputs during 2000-2001 runoff years was 4.8 and 5.8 g N m^{-2} y^{-1}, respectively. Thus, the measured nitrogen fixation during 2002 could only account for a fifth of the imbalance observed during the two previous years.

Nutrient cycling can be linked to food web structure (Paterson et al, 2002). *Daphnia*, a freshwater planktonic crustacean, has a low N:P body ratio and retains P while releasing nitrogen to the surrounding water. Therefore, an increase in *Daphnia* populations can increase N:P ratios reducing the cyanophyte advantage (Elser, 1999). MacKay and Elser (1999), show evidence from the Experimental Lakes area in Ontario, Canada that *Daphnia* were not present during cyanophyte blooms and that cyanophytes were not able to establish themselves in enclosures containing *Daphnia*. The zooplankton community was not considered during this study, but cascading trophic level effects may accompany fish kills or the timing and magnitude of stocking events in Crowley Lake.

While sediments usually act as a net sink, they may act as a source on seasonal time scales and over longer periods if the sediments are in disequilibrium due to long-term changes in loading or release rates. High ammonia content in pore water is indicative of nitrification, the decomposition of organic material by heterotrophic bacteria. Denitrifying bacteria are well adapted to rapidly changing conditions in sediments. They are abundant and can have a large effect on nitrogen turnover (Hakanson and Jansson, 2002). While bacteria do not create new biomass or fix new energy, they act as a link returning nitrogen back into the water column. When external loading is reduced, sediments can release nutrients back into the water column in sufficient concentration to support algal growth for extended periods thus delaying improvement in water quality (Hu et al, 2001).

While we were unable to measure sediment ammonia release with benthic chambers, ammonia release estimates were estimated based on porewater profiles. The estimated release rates ranged from 0.2 to 1.1 mmol N m^{-2} d^{-1} or 1.0 - 5.6 g m^{-2} y^{-1}. While this is of similar magnitude to the "missing" nitrogen source, in terms of the lakewide nitrogen budget, it will be offset by any deposition and burial occurring. This can only act as a net source if there has been a significant change in the retentive properties of the sediments or a large decrease in deposition so that current high rates of sediment release are due to high rates of deposition in the past. The porewater profiles of three of the four cores all showed a significant discontinuity at approximately 10 cm depth. This could be a result of either of these two possibilities.

It is informative to examine hypolimnetic accumulation and upward fluxes of nitrogen compared to estimated sediment release rates. The upward flux estimates and hypolimnetic accumulation are much larger than the estimated sediment release rates (Table 5). This implies depositional fluxes and remineralization rates within the hypolimnion are large relative to sediment-water interface fluxes. The estimated upward fluxes are large relative to the imbalance in the lakewide nitrogen budget. Thus, remineralization of decaying algal matter from one or two years previous may be contributing significantly to these fluxes. In this case, the nitrogen imbalance observed in 2000 and 2001 may just represent temporal time lags. However, the thermal regimes were nearly identical among all three years and all were below normal runoff years (68, 57, and 51% of normal for April-September).

Table 5 Hypolimnetic accumulation, diffusive fluxes, and sediment release

	Hypolimnetic accumulation (mmol m-2 d-1)	Upward Diffusive Flux at 12 m (mmol m-2 d-1)	Estimated sediment release based on porewater profile
P	0.6	0.2-0.9	NA
N	3.7	2.2 – 5	0.2-1.1

SUMMARY AND CONCLUSIONS

Summer (June – August) 2002 nitrogen fixation rates were generally low and, surprisingly, sestonic particulate ratios throughout the summer (including September) suggested P rather than N limitation. Estimates of lakewide nitrogen fixation were within the range observed in other eutrophic lakes, but could only account for one fifth of the imbalance between measured inputs and outputs of nitrogen during 2000 and 2001. However, no massive algal blooms, as have been observed during past years, were present at any of the stations sampled on the eight summer surveys during 2002. Also, sestonic molar N:P ratios were higher in 2002 compared to 2001, most likely indicating less nitrogen limitation. Nitrogen fixation may also continue into October-November and rates may have been higher during spring 2002 as evidenced by higher rates in a sample collected at the dock in late May. However, it is unlikely that these factors could account for the entire imbalance. Both sediment release rates and upward ammonia fluxes were of similar magnitude to the imbalance in the N budget observed during 2000 and 2001 but they do not necessarily constitute net sources. The discontinuity at ~10 cm in sediment ammonia porewater raises the intriguing possibility that either overall nutrient loading rates have decreased, possibly due to range best management practices initiated over the past ten years, or the sediment environment has changed in a manner which has reduced its retentive capacity. We are not able to distinguish between these two possibilities with the present limited data. However, sedimentation rates are fairly high (>1 cm yr^{-1}) in Crowley Lake and revisiting the sediments in several years would allow distinguishing between these two possibilities.

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