

INTERNAL LOADING AND NUTRIENT CYCLING IN CANYON LAKE

Final Report Submitted to:

Santa Ana Regional Water Quality Control Board
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EXECUTIVE SUMMARY

Canyon Lake is a small, multiple use drinking water reservoir located in Riverside County, CA. Canyon Lake was constructed in 1928 as the Railroad Canyon Reservoir to provide long-term storage of water in a region facing growing water demands. The State of California has placed Canyon Lake on its list of 'impaired' water bodies due to excessive nutrients, as authorized by Section 303(d) of the Clean Water Act. As a result, the Regional Water Quality Control Board is required to develop a nutrient TMDL for the lake. The research summarized in this report was conducted from July 2001 – August 2002 to support development of the nutrient TMDL for Canyon Lake.

The study included characterization of the sediments, measurement of the rate of nutrient release from the sediments to the water column, measurement of the rate of particulate-bound nutrient deposition from the water column to the sediments, and development of a nutrient budget for the lake. The nutrient budget, in turn, provided parameter values needed for the implementation of the BATHTUB water quality model.

Canyon Lake was found to be monomictic, and turnover occurred by late fall. Ratios of total nitrogen:total phosphorus as well as algal growth experiments indicated that algal production at Canyon Lake was predominantly controlled by nitrogen. Furthermore, Canyon Lake could be separated into a small north basin that receives San Jacinto River flow and two southern basins – the narrow, channelized East Bay, which had high turbidity and low transparency, and the broader, deeper main body that showed a lower suspended solids concentration and greater transparency. Other differences between the two southern basins were: 1) the main body displayed thermal stratification for approximately 8 months of the year, while East Bay was usually well-mixed, and 2) the main body had an anoxic hypolimnion during stratified periods while East Bay was always well aerated.

Sediments were characterized for a number of properties, including particle size, solid-phase and porewater nutrients, and carbonate contents. Distribution of these properties varied depending on texture of the sediments, where higher contents of C, N, P, S and CaCO₃ were found in the finest sediments (>60% clay-sized particles) relative to coarser sediments. The distribution of coarse and fine grained particles also varied within the lake bed, where sands were generally found close to the shoreline and in shallow embayments, clays were found in the deeper portions of the main body, and silts were mostly concentrated in the northern portion and East Bay. Sandy sediments were

classified as Type I, silts as Type II, and fine, organic sediments as Type III. The Type II sediments occupied an estimated 48% of the total area in the lake, while Types I and III took up 21% and 31% of the lake bottom.

The sediments were found to release nutrients at high rates, with the P flux averaging 6.3, 15.1 and 6.5 mg/m²/d for the Type I, II and III sediments, respectively. The cooler temperatures associated with the hypolimnion limited summer P release from the Type III sediments relative to the shallower, warmer Type II sediments. Internal recycling of N proceeded at an annual average rate approximately 4x that of P (22.7-34.8 mg/m²/d). Good agreement was found between core-flux and hypolimnetic mass balance methods, although peepers significantly underpredicted SRP and NH₄-N release.

Sedimentation rates of particulate nitrogen and phosphorus measured using sediment traps were very high for each sampling period (as high as ~300 mg N/m²/d and ~80 mg P/m²/d for some sites). Evidence of mechanical resuspension was strong for East Bay; consequently particulate matter was overtrapped relative to amounts expected based upon primary productivity. The main body, however, also exhibited high trapping rates, although no direct evidence of resuspension was found. Nevertheless, the properties of trapped material (*i.e.*, total C, N and P) were in adequate agreement with the Redfield ratio indicating that the particulate material in the water column of Canyon Lake was mostly derived from phytoplankton. In addition, mean calcium carbonate contents of the trapped material ranged from 13.5% to 19%, suggesting the formation of *in situ* calcium carbonate resulting from the removal of CO₂ by photosynthesizing algae and from evapoconcentration effects.

Given the limited watershed runoff of 2001-2002, internal processes, specifically nutrient recycling and resuspension, were the dominant source of nutrients to the lake over the study period. It should be noted, however, that the contributions of nuisance runoff and other local sources of nutrients were not quantified in this study. Such runoff, found to be an important source of bacteria to Canyon Lake, may also be an important part of the nutrient budget, especially during dry years.

Using the nutrient budget results, the BATHTUB surface water quality model was able to reproduce the main features of the average water quality in the lake using a 2nd order, fixed sedimentation model, although the model did a poor job of reproducing the observed TP levels in the main body of the lake. Notwithstanding, preliminary simulations suggest that substantial reductions (>75%) in both N and P loading are

required to measurably improve water quality in the lake. Even greater reductions will be needed during periods of substantial nutrient inputs during high runoff events.

The findings also raise the possibility that destratification of the main body of the lake, suggested as an in-lake nutrient control measure, may minimally influence or even degrade overall water quality; destratification will, by design, break the thermal barrier to mixing and also increase the temperature of the deep water (Type III) sediments. While delivery of DO to the deeper waters of the lake will lower dissolved concentrations of Fe^{2+} , Mn^{2+} , H_2S and other reduced chemical species, core-flux measurements indicate that SRP and $\text{NH}_4\text{-N}$ recycling in the lake is not strongly affected by DO status. Rather, internal loading appears to be more influenced by temperature and the delivery of fresh organic matter to the sediments. The effectiveness of such a destratification system, then, appears to depend upon its ability to limit the residence time of phytoplankton in the upper, well-lit portion of the water column.

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1.0 INTRODUCTION

Canyon Lake, constructed in 1928 as the Railroad Canyon Reservoir, is located in Riverside County, CA about 1 mile upstream of Lake Elsinore. It is owned and operated by Elsinore Valley Municipal Water District and serves multiple uses, including drinking water supply, agricultural supply, and recreational activity. The area around the reservoir is highly developed, where much of the land use in the area is residential. During wet periods, the lake collects runoff from the San Jacinto River (which enters into the northernmost portion of the lake) and from Salt Creek (which meets East Bay). The San Jacinto River drains a 750 square mile watershed comprised of urban, agricultural and wild lands that can be a significant source of nutrients to Canyon Lake during periods of heavy rainfall. Salt Creek collects agricultural and urban runoff and may be a significant source of nutrients to East Bay. The Railroad Canyon Dam, recently upgraded in 1997, holds nearly 12,000 acre-feet of water. The stored surface water is conventionally treated, blended with groundwater and imported water, and transported to local residents. The lake currently has a mean depth of 5.9 m and a maximum depth of approximately 13 m. The average temperature is 21 °C and the annual evaporative losses generally do not exceed 1.4 m/yr.

As a result of its geographical and climatic setting, morphology of the lake basin, external and internal nutrient sources and other factors, water quality at Canyon Lake has deteriorated. The Santa Ana RWQCB has listed the lake as 'impaired' by excessive nutrients and pathogens, as authorized in Section 303(d) of the Clean Water Act. The RWQCB is thus required to develop a nutrient TMDL for the lake. This research has been conducted for the RWQCB in support of the TMDL development effort. The main objectives of this study were to quantify internal nutrient loading and nutrient cycling within the lake.

In order to achieve the above objectives, a series of laboratory and field measurements were conducted from July 2001 to August 2002, including:

- Measurement of surface water quality
- Characterization of sediments and their distribution in the lake
- Quantification of nutrient release from sediments
- Measurement of sedimentation rates and the rate of particulate N, and P deposition to the sediments
- Development of nutrient budget

2.0 SURFACE WATER QUALITY

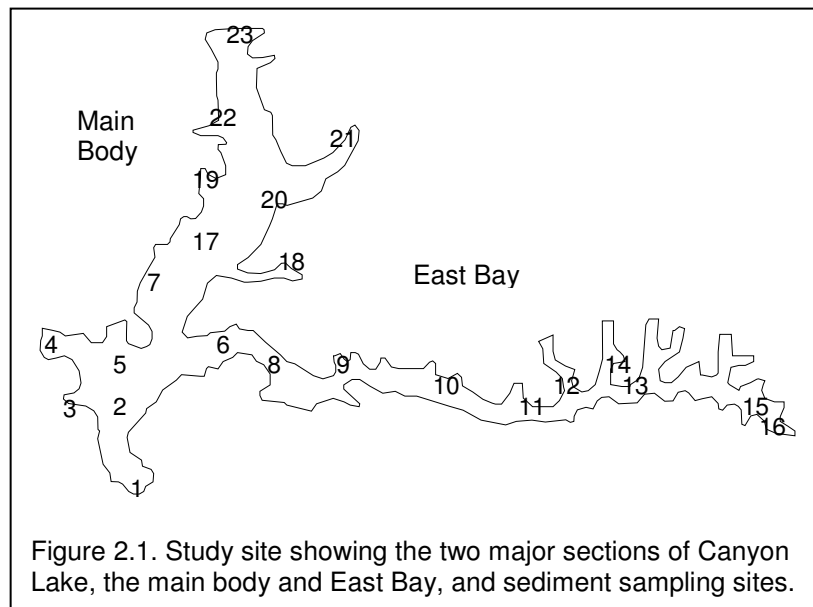
A number of common water quality parameters (*e.g.*, DO, temperature, EC, and pH) were measured over the year to determine the general limnological conditions in the lake.

2.1 Methods

A Hydrolab Quanta and DataSonde4 multiparameter water quality monitoring probes were used to make field measurements at various sites in Canyon Lake. The immersible probe was fitted with a pressure transducer, temperature sensor, DO probe, pH electrode and EC electrode that was connected to a data display/controller via a 50 m waterproof cable. The DO probe was calibrated approximately monthly against an O₂-saturated water sample prepared by sparging with lab air. The pH electrode was periodically calibrated against pH 7 and 10 buffers (Fisher Scientific). The pressure transducer, used to determine depth of the multiparameter probe, was not calibrated, although it was re-zeroed in the field as needed to properly record depths. Temperature and EC were not calibrated, although they were confirmed as working properly and within specification by the factory upon delivery. The transparency of the lake was measured using a Secchi disk. Chlorophyll measurements were also made using a SCUFA fluorometer at selected sites from June-August, 2002. Water samples were collected at several depths of the water column using a van Dorn sampler, stored in 125 mL polypropylene bottles and stored on ice until returned to the lab. The samples were then filtered and acidified to pH<2 for dissolved nutrient analyses using an Alpkem autoanalyzer. Separate subsamples were digested and analyzed for total N and P following Valderama (1981).

2.2 Results

Measurements were made at Canyon Lake on a number of different dates between July 2001 and August 2002. Hydrolab casts made at site 2 and other sites in the main body of the lake (Fig. 2.1) show that Canyon Lake is monomictic, *i.e.*, it is stratified during the summer and mixes during the winter (Fig. 2.2).



The warm monomictic character of the lake can be more visibly seen in deeper areas, where temperature and DO exhibited steep gradients and changed seasonally (Fig. 2.2). When the lake was stratified, both temperature and DO showed a sharp decrease at the thermocline, which is generally present at about 6-7 m in this lake (Fig. 2.2a). DO levels dropped from about 8 mg/L at the surface to 0.2 mg/L at bottom, while temperature decreased from 28°C to 14°C. During this stratified period (late spring to fall) the lake showed a progressive seasonal separation of nutrients within the water column, where large quantities of nutrients were trapped at or below the thermocline (Fig. 2.2b). Note that SRP levels were higher than $\text{NH}_4\text{-N}$ levels up to about 8 m, after which $\text{NH}_4\text{-N}$ concentrations increased to reach maximum concentrations of 1.6 mg/L at the bottom.

Destratification of the lake occurred in late fall, with relatively well-mixed conditions present throughout the winter. Uniform temperatures (around 11-12 °C) were maintained within the water body and the hypolimnion became modestly oxygenated (DO ~ 2.5 mg/L) (Fig. 2.2a). Mixing of the lake also resulted in redistribution of nutrients within the water column, creating a uniform vertical profile (Fig. 2.2b). Nutrients were thus redistributed within the water column with the ongoing disappearance and emergence of the thermocline.

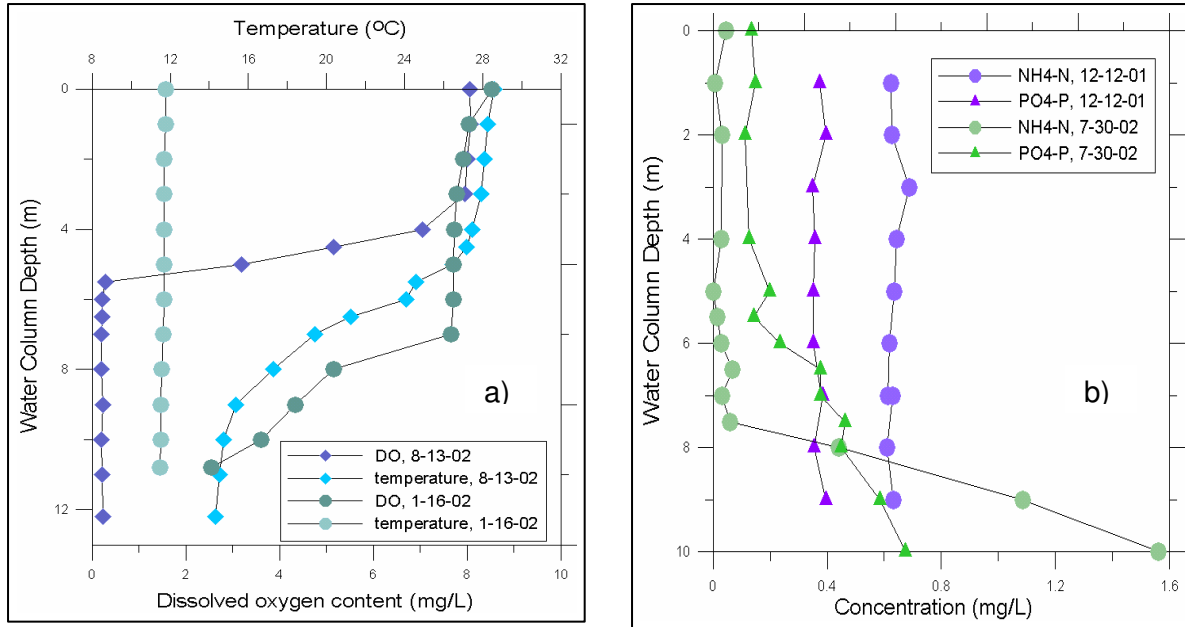


Figure 2.2. Water column (a) temperature and DO, and (b) nutrient profiles for summer and winter at a deep site (site 2) near the dam.

The distribution of algal biomass within the water column at site2 in the main body and a shallow East Bay site (site 11) is shown in Figure 2.3. Chlorophyll levels taken from June-August, 2002 indicate high algal populations present at the thermocline of the deep site, where concentrations approached 180 $\mu\text{g/L}$ at 6 m depth (Fig. 2.3a). Shallow East Bay yielded chlorophyll concentrations as high as 140 $\mu\text{g/L}$ (Fig. 2.3b).

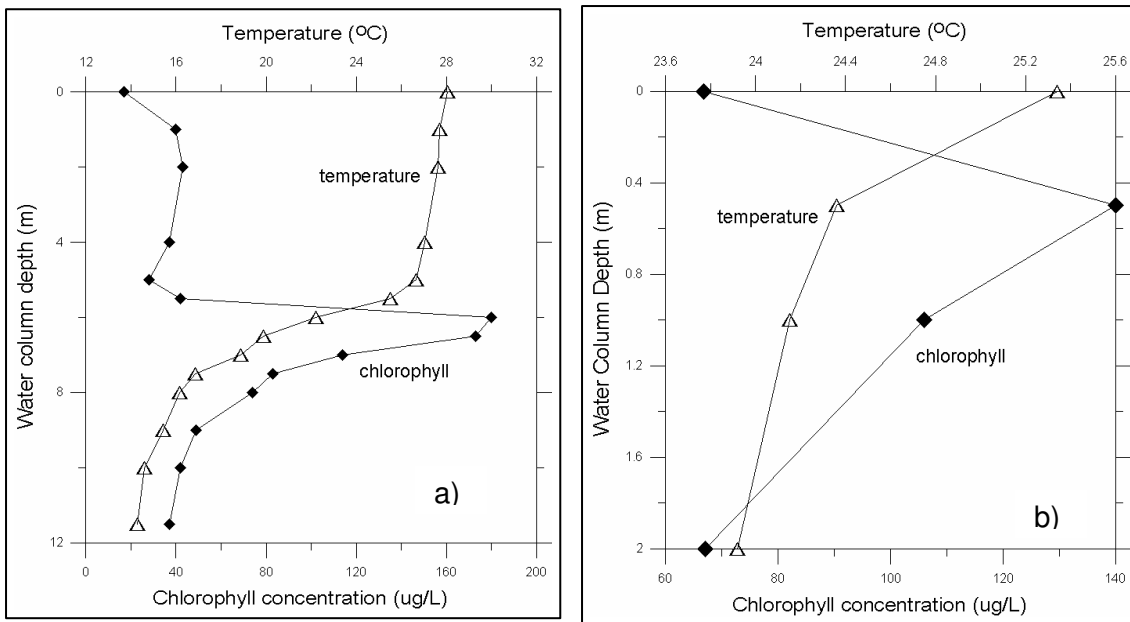


Figure 2.3. Chlorophyll levels at (a) site 2 on 7/30/02 and (b) East Bay on 6/10/02.

Table 2.1 summarizes average water quality data for the main body and East Bay collected over the year. The mean annual temperatures were 19°C at the main body and approximately 22°C for East Bay (Table 2.1). Temperatures varied seasonally and ranged from 9°C to 30°C during the year. Mean electrical conductivity remained fairly uniform (1-1.2 $\mu\text{s}/\text{cm}$) throughout the lake while mean dissolved oxygen contents were almost twice as high in East Bay relative to the main body. Maximum DO contents of 15.9 and 17.3 mg/L reflected the top 1 m of the water column, and minimum DO contents were representative of the hypolimnion. East Bay displayed a slightly higher mean pH than the main body (8.3 vs. 7.8), but the ranges were similar. Mean turbidity at East Bay was 15.6 NTU, much higher than that found in the main body (8.7 NTU). Dissolved organic carbon was measured for the main body only, and ranged from 7.9-17.9 mg/L during the year. Transparency values, although not collected frequently, were generally less than 2.0 m.

| Parameter | Units | Main Body | | | East Bay | | |
|--------------------|-----------------------------|-----------------|-------------|----|-------------------|-------------|----|
| | | Mean \pm S.E. | Range | n | Mean \pm S.E. | Range | n |
| Temp. | (°C) | 19.0 \pm 5.9 | 8.8-30.3 | 42 | 21.8 \pm 7.1 | 9.6-30.8 | 20 |
| E.C. | ($\mu\text{s}/\text{cm}$) | 1.1 \pm 0.1 | 0.8-1.9 | 42 | 1.2 \pm 0.2 | 1.0-1.4 | 20 |
| DO | (mg/L) | 3.9 \pm 3.4 | 0.05-15.9 | 42 | 7.7 \pm 3.6 | 0.3-17.3 | 20 |
| pH | | 7.8 \pm 0.5 | 6.8-9.0 | 42 | 8.3 \pm 0.7 | 6.7-9.2 | 20 |
| Turbidity | (NTU) | 8.7 \pm 16.3 | 0.0-222.8 | 42 | 15.6 \pm 15.5 | 0.0-119.1 | 20 |
| DOC | mg/L | 11.4 \pm 2.3 | 7.9-17.9 | 12 | NA | NA | NA |
| Secchi Depth | (m) | 1.34 \pm 0.4 | 0.85-2 | 9 | 0.81 \pm 0.3 | 0.46-1.2 | 5 |
| SRP | mg/L | 0.33 \pm 0.19 | 0.1 - 1.04 | 8 | 0.185 \pm 0.047 | 0.143-0.245 | 2 |
| NH ₄ -N | mg/L | 0.41 \pm 0.46 | 0 - 2.11 | 8 | 0.125 \pm 0.010 | 0.093-0.115 | 2 |
| NO ₃ -N | mg/L | 0.06 \pm 0.05 | 0 - 0.5 | 8 | 0.101 \pm 0.044 | 0.064-0.168 | 2 |
| Total P | mg/L | 0.51 \pm 0.42 | 0.07 - 1.98 | 3 | 0.22 \pm 0.18 | 0.1-0.4 | 2 |
| Total N | mg/L | 1.86 \pm 1.65 | 0.63 - 8.61 | 3 | 1.81 \pm 0.37 | 1.4-2.1 | 2 |

n is the number of sampling dates on which data was collected.

Soluble reactive phosphorus (SRP) in the water column of the main body averaged 0.33 mg/L while mean total phosphorus contents were 0.51 mg/L. The difference in total P and SRP was attributed to particulate organic phosphorus, which accounted for about 40% of total P. Ammonium constituted most of the total inorganic nitrogen present, whereas nitrate (as nitrogen) was almost depleted in the water column. Inorganic nitrogen (as NH₄-N and NO₃-N) represented roughly 25% of the total nitrogen present in the lake, which averaged 1.86 \pm 1.65 mg/L.

Existing nutrient data strongly indicate that phosphorus is relatively abundant and that nitrogen is the limiting factor for phytoplankton at Canyon Lake over much of the year. To assess nutrient limitation in Canyon Lake, mass ratios of total nitrogen to total phosphorus were calculated for June, July and August, 2002 (Table 2.2). Ratios of the biologically available forms of N and P are also given in Table 2.2. According to Fleming (1940) and Redfield (1934), N:P mass ratios of less than 7:1 indicate nitrogen limitation. Mean TN/TP ratios measured across Canyon Lake during June and July ranged between 2-4, indicating nitrogen limitation. Mean TN/TP ratios measured in August however, did not show serious N or P limitation in the main body (sites 2, 8, and 22) while site 11 in East Bay displayed a high TN:TP ratio of 15. Total nitrogen and phosphorus data collected by the Regional Board confirm the high ratios found in August; *e.g.*, TN/TP ratios in East Bay were between 12-18 and ratios in the main body were between 6-9. Such conditions are not uncommon, and in fact, Ryding and Rast (1989) have reported that nutrient concentrations in a lake can change over an annual cycle. The main implication is that Canyon Lake appears to have an adequate supply of nitrogen and phosphorus during the late summer, but is limited by nitrogen earlier in the summer.

| Date | Site | Mean TN:TP | Mean NO ₃ +NH ₄ :SRP |
|---------|------|------------|--|
| 6-25-02 | 2 | 2.0 | 1.1 |
| | 8 | 2.7 | N/A |
| | 22 | 1.8 | 1.4 |
| 7-9-02 | 2 | 3.5 | 1.4 |
| | 11 | 4 | N/A |
| | 22 | 3.3 | 1.5 |
| 8-8-02 | 2 | 6.2 | 1.3 |
| | 8 | 10 | N/A |
| | 11 | 15 | N/A |
| | 22 | 5.9 | 1.0 |

In addition to the use of nitrogen:phosphorus ratios to determine the degree of nutrient limitation, a series of algal bioassays were conducted. Nutrient spikes of 0.5, 1.0, and 2.0 mg/L NO₃-N and NH₄-N as well as 0.1, 0.2, and 0.4 mg/L PO₄-P solutions were added to separate sub-samples of lake water collected in August, 2002. The lake water was kept under optimal light and temperature conditions in a growth chamber and

chlorophyll concentrations were measured daily for five days. Results showed strong algal growth response with the addition of 1 and 2 mg/L $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ solutions, whereas no response was detected with variable addition of $\text{PO}_4\text{-P}$ (Fig. 2.4a). Additions of 2 mg/L of ammonium and nitrate solutions produced a substantial increase in algal biomass over a 5 day period, suggesting that the algal species inhabiting Canyon Lake are nitrogen-limited. Figure 2.4b shows an increasing algal response to increasing nitrate concentrations. A similar response was seen with $\text{NH}_4\text{-N}$ additions (data not shown).

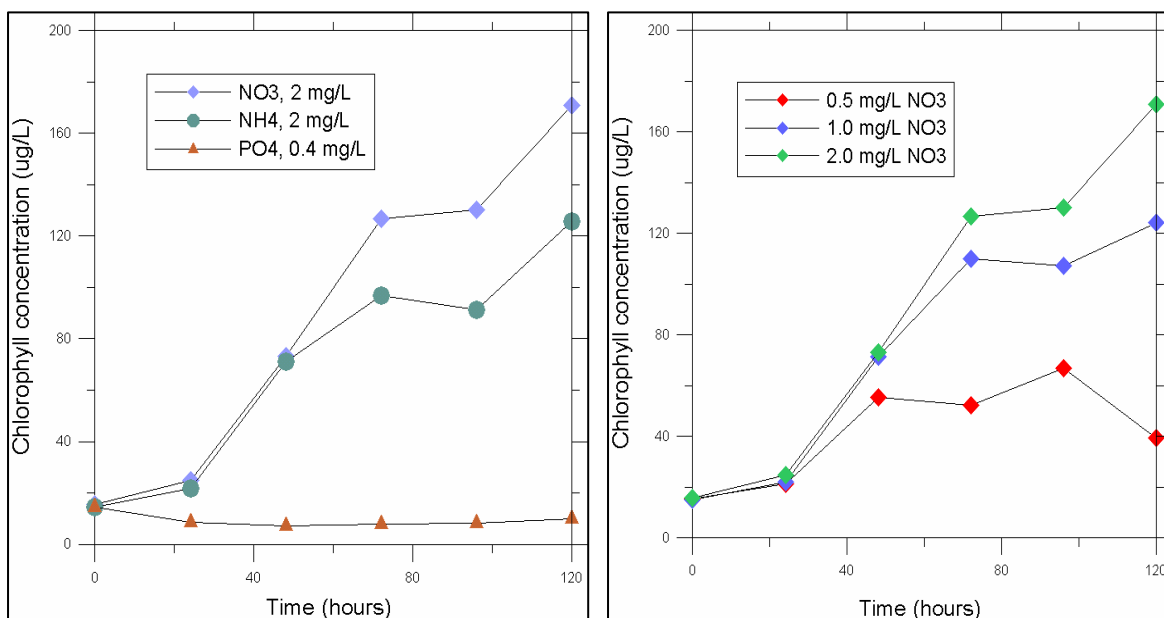


Figure 2.4. Algal bioassays conducted in August, 2002 showing algal growth response to (a) individual ammonium, nitrate, and phosphate additions and (b) increasing nitrate concentrations.

3.0 CHARACTERIZATION OF SEDIMENTS AND THEIR DISTRIBUTION

In order to quantify internal nutrient loading from the sediments to the water column, it was necessary to evaluate the properties and distribution of sediments in Canyon Lake.

3.1 Methods

A sampling grid was developed for sediment grab sampling. A total of 23 sites were sampled using a Ponar sampler (see Figure 2.1 for site locations). Locations of sampling sites were recorded using GPS; depth to bottom was also recorded.

Sediment grab samples were packed into 500 mL glass jars with screw-cap lids and stored on ice until return to the lab. The sediments were then completely mixed and sub-sampled for porewater nutrients, particle size, and other properties. Porewater was extracted by centrifugation, and promptly filtered and acidified. Particle size was determined on untreated sediment using the hydrometer method (Gee and Bauder, 1986). Total carbon, nitrogen, and sulfur were determined on dry sediment using a Carlo-Erba CNS analyzer (Nelson and Sommers, 1982). Inorganic C and CaCO_3 were determined manometrically following Loeppert and Suarez (1996). Organic C was taken as the difference between total C and inorganic C. Organic nitrogen was essentially total nitrogen, since inorganic nitrogen was negligible. Total and inorganic P was determined following Aspila *et al.* (1976). Organic P was taken as the difference between total and inorganic P (Aspila *et al.*, 1976; Berner and Rao, 1994). Total Fe and Fe(II) contents of the sediments were determined on 1 M HCl extracts using the o-phenanthroline method. Porewater $\text{NH}_4\text{-N}$ and SRP and extracted SRP were analyzed using an Alpkem autoanalyzer following standard methods (APHA, 1989).

In addition to the sediment characterization described above, a study to further characterize sediments in East Bay was carried out. Intact sediment cores 40 cm in length were taken from sites 11, 13 and 16 using a Wildco corer, and frozen for later analysis. The cores were extruded vertically in 2.5 cm slices and each slice was homogenized and analyzed for total C, N, S, total P, inorganic P, and CaCO_3 following procedures outlined above. Porewater nutrient and particle size analyses were not conducted due to limited sample size.

Supplementing the above sediment characterization, a bathymetric map of Canyon Lake was constructed. The map for the lake was constructed by linking a survey

of the shoreline with electronically measured depths and GPS coordinates (Fig. 3.1). The survey was taken between December 2001 and February 2002. The data was compiled into a grid file that was then used to develop a contour map using SURFER 7.0 (Golden Software, Golden, CO).

3.2 Results

The bathymetric map developed from GPS and depth data is provided in Fig. 3.1.

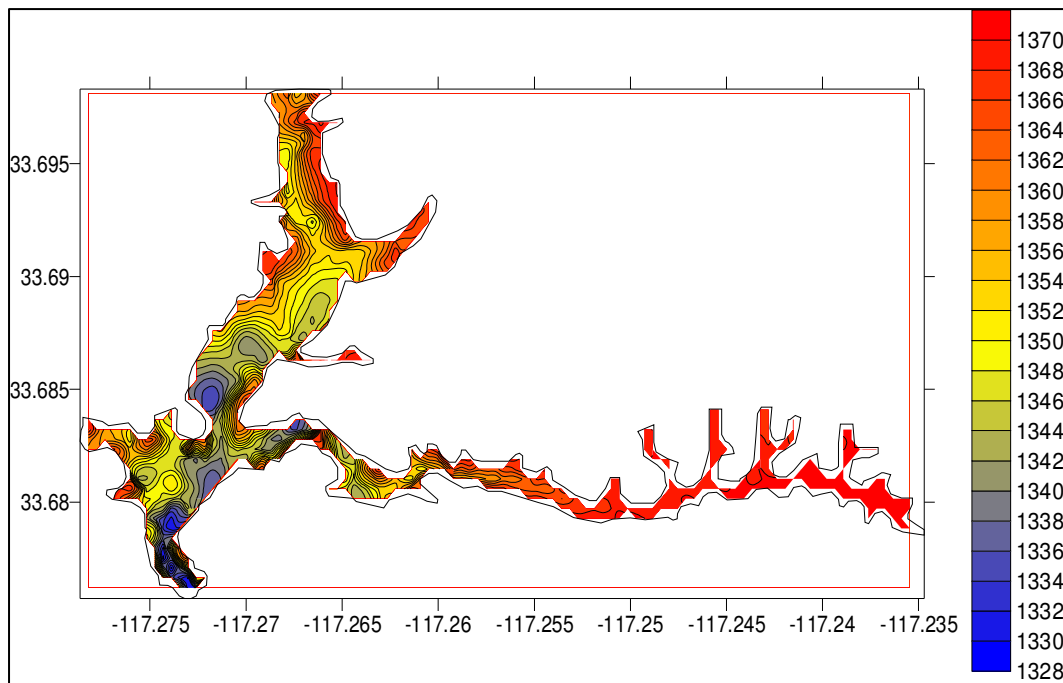


Figure 3.1. Bathymetric survey of Canyon Lake, Riverside County, CA .

The map provides lake bottom elevation in relation to sea level. The spillway elevation is 1381.8 ft. Lake levels have fluctuated from 1376.9' (8/20/01) to 1373.7' (10/02/02) above sea level. As one can see, East Bay is quite shallow (<6 ft.) compared to the main body, which reaches a maximum depth of 48 feet (current for 2001-2002). The mean depth of 5.9 m (or 19.4 ft.) denoted by the yellow/orange areas (Fig. 3.1) is seen in the central portions of the main body. In addition to Fig. 3.1, Table 3.1 provides some morphometric characteristics of the lake.

Mean and maximum depths at Canyon Lake were 5.9 m and 13 m, respectively. The surface area of the reservoir (1.21 km² or 0.467 mi²) is extremely small compared to the size of the drainage basin (760 mi²) and the extensiveness of the drainage basin is seen in the drainage area:surface area ratio. The high ratio suggests that external flows of nutrients can substantially affect water quality at Canyon Lake, given appropriate climatic conditions. The high shoreline development number gives an indication of Canyon Lake's complex shape (e.g., circular water bodies have shoreline development numbers of close to 1).

| | |
|------------------------------------|-----------------------|
| Surface area (km ²) | 1.21 |
| Volume (m ³) | 7.2 × 10 ⁶ |
| Length of shoreline (km) | 24 |
| Shoreline development # | 6.1 |
| Drainage area:surface area ratio | 1606 |
| Mean Depth (m) | 5.9 |
| Max. Depth (m) | 13 |
| Average Temp. (°F) | 70 |
| Primary Inflow | San Jacinto River |
| Storage capacity (m ³) | 1.5 × 10 ⁷ |

A summary of the sediment properties is given in Table 3.2.

| Property | Units | Mean | Std Dev | Median | Range |
|--------------------|-------|-------|---------|--------|-------------|
| <i>Sediment</i> | | | | | |
| Sand | % | 16.5 | 21.8 | 5.9 | 0.0 – 70.9 |
| Silt | % | 42.0 | 11.5 | 44.1 | 16.1 – 61.5 |
| Clay | % | 41.8 | 20.2 | 45.6 | 9.3 – 70.3 |
| Total C | % | 3.51 | 1.06 | 3.67 | 0.61 – 4.80 |
| Organic C | % | 2.83 | 1.16 | 3.26 | 0 – 4.09 |
| Inorganic C | % | 0.67 | 0.46 | 0.53 | 0 – 5.17 |
| CaCO ₃ | % | 3.72 | 3.83 | 3.95 | 0 – 12.7 |
| Total N | % | 0.40 | 0.20 | 0.44 | 0.20 – 0.56 |
| Total S | % | 0.67 | 0.50 | 0.69 | 0 – 1.17 |
| Total Fe | mg/g | 30.8 | 13.5 | 32.8 | 10.5-52.9 |
| Fe (II) | mg/g | 28.9 | 13.4 | 31.5 | 8.3-51.9 |
| Total P | mg/kg | 747 | 249 | 743 | 257 – 1523 |
| Inorganic P | mg/kg | 551 | 163 | 570 | 213 – 929 |
| Organic P | mg/kg | 196 | 146 | 178 | 0 – 615 |
| <i>Porewater</i> | | | | | |
| SRP | mg/L | 2.81 | 1.14 | 2.76 | 0.42 – 5.52 |
| NH ₄ -N | mg/L | 15.90 | 7.54 | 14.34 | 6.86 – 43.3 |

Canyon Lake sediments appear to be dominated by fine particles (> 40% of each silt and clay), with limited sand contents found at most sites. Greater contents of silts and clays were found at East Bay and the main body, while greater sand fractions were concentrated in littoral zones and embayments. Average total carbon, nitrogen, and sulfur contents were 3.51%, 0.40%, and 0.67%, respectively. CaCO₃ contents in the sediments ranged from 0-12.7%, but the mean CaCO₃ content of 3.7% was fairly low. Total iron constituted between 10 and 53 mg per gram of sediment, while total P ranged from 257-1523 mg/kg throughout the lake. Most of the mean properties at Canyon Lake were very similar to those of Lake Elsinore sediments, with the exception of higher mean sand contents (*e.g.*, 27.6% at Lake Elsinore compared to 16.5% at Canyon Lake), higher mean CaCO₃ (7.08%) and higher mean total S (0.78%) contents found at Lake Elsinore.

Sand contents were greatest in shallow areas close to shore, however a deeper site near the dam consisted of 71% sand (Fig. 3.2). Although most of East Bay and the main body contained limited sand in its sediments (Fig. 3.2), the two areas differed in their clay contents (Fig. 3.3). The highest clay contents (>60%) were located in the main/central body, whereas somewhat lower values (40-55% clay-sized particles) were found within the sediments of East Bay (Fig. 3.3).

Little variation was present in total and organic carbon in the sediments (Fig 3.4). Both averaged about 3% across the lake, with lower concentrations found only at those sites with a significant amount of sand



Fig. 3.2. Sand content of sediment (%).

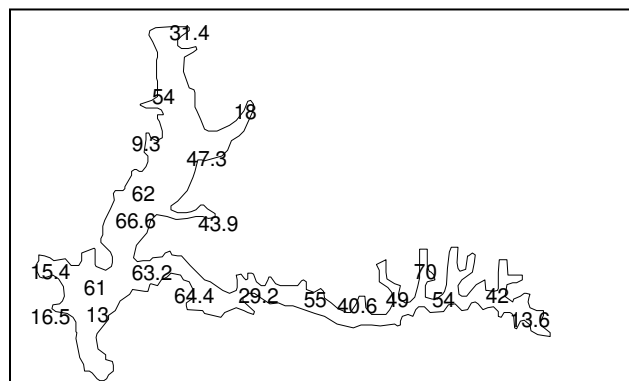
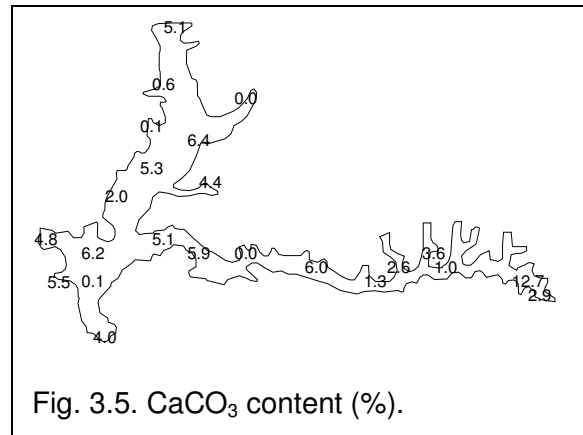
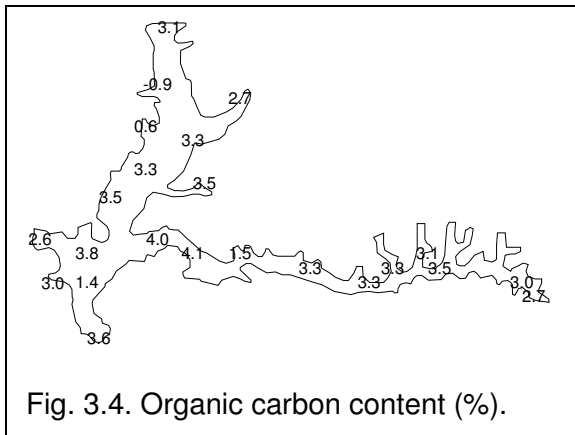


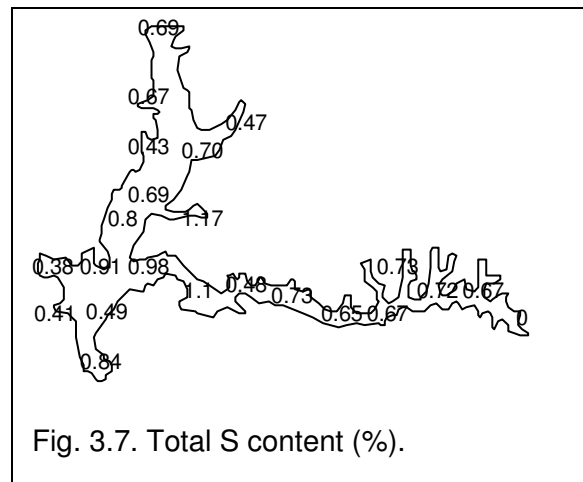
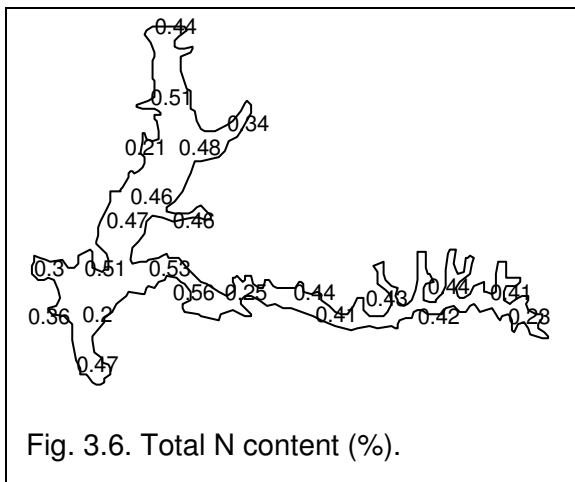
Fig. 3.3. Clay content of sediments (%).

(Fig. 3.2). Total carbon was negatively correlated to sand ($R^2 = -0.64$), however both total and organic carbon were poorly correlated with clay ($R^2 = 0.5$ and 0.2 , respectively).

Inorganic carbon constituted a small fraction of the total carbon present, while the carbonate content varied from 0 - 12.7% throughout the lake (Fig. 3.5).



Total N (Fig. 3.6) averaged around 0.4 - 0.5 % in the finer sediment types (with lower concentrations found at sandy sites), while total S (Fig. 3.7) was present in greater quantities among the deeper sites in the main body. Total N correlated well with clay content ($R^2=0.74$) whereas total S correlated less strongly with clay ($R^2=0.56$).



The total P content of the sediments ranged from 257 - 1523 mg/kg (Fig. 3.8) and was correlated with both clay and total C contents ($R = 0.90$ and 0.75 , respectively) and negatively correlated with sand content (-0.87). Total P was highest (reaching 1000 mg/kg) in the deep sites around the main body and at the very north end of the reservoir.

The highest TP content was found at the dam (Fig. 3.8). Over 70% of the total P in the sediments was present in an inorganic form.

Porewater SRP concentration was highest near the confluence with the upper part of the lake and the San Jacinto River inlet (Fig. 3.9). The concentration of SRP was very poorly correlated with total P content in the

sediments, however ($R=0.01$), indicating that other processes play a role in the release of dissolved phosphorus. Porewater $\text{NH}_4\text{-N}$ concentrations showed different spatial patterns and were highest near the culvert separating East Bay from the main body of the reservoir (Fig. 3.10). $\text{NH}_4\text{-N}$ was not well correlated to total N contents in the reservoir, again suggesting that other physico-chemical processes also determine porewater $\text{NH}_4\text{-N}$ concentrations in the lake.

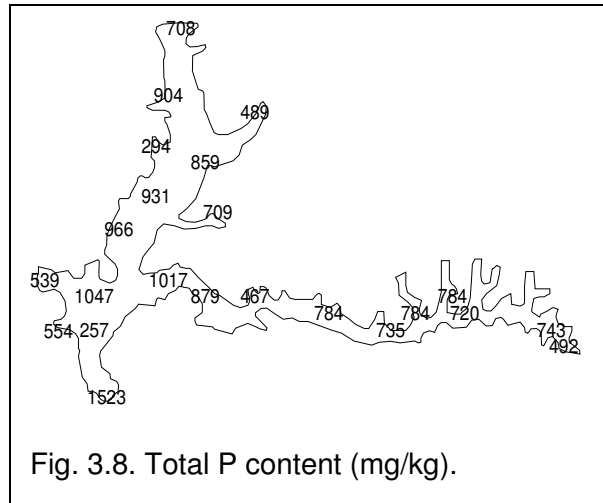


Fig. 3.8. Total P content (mg/kg).

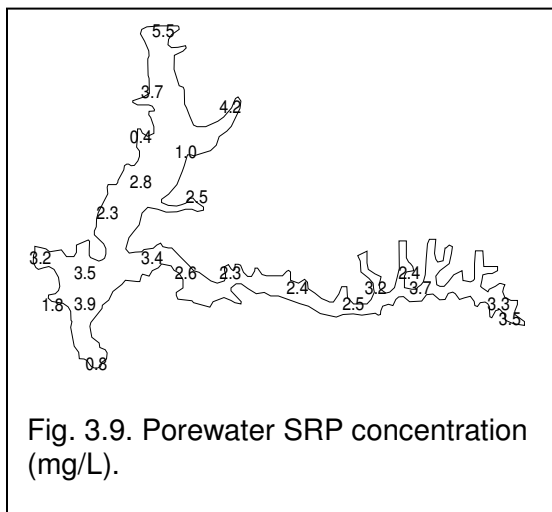


Fig. 3.9. Porewater SRP concentration (mg/L).

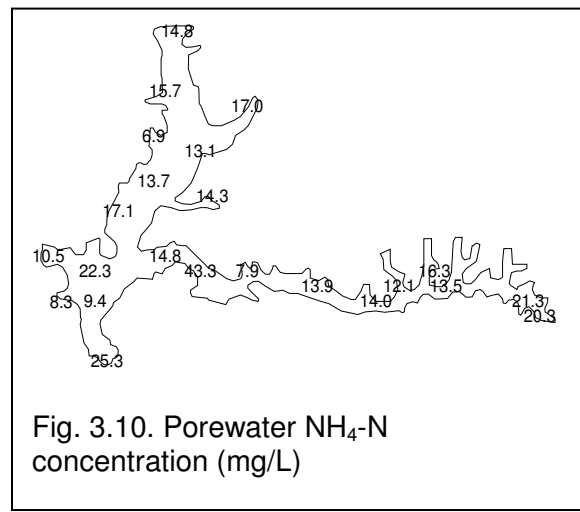
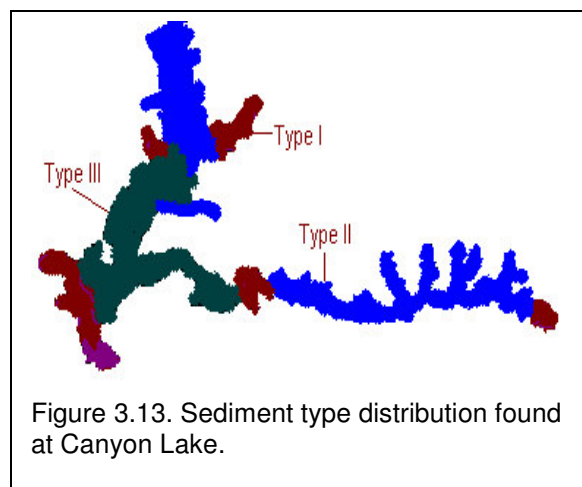
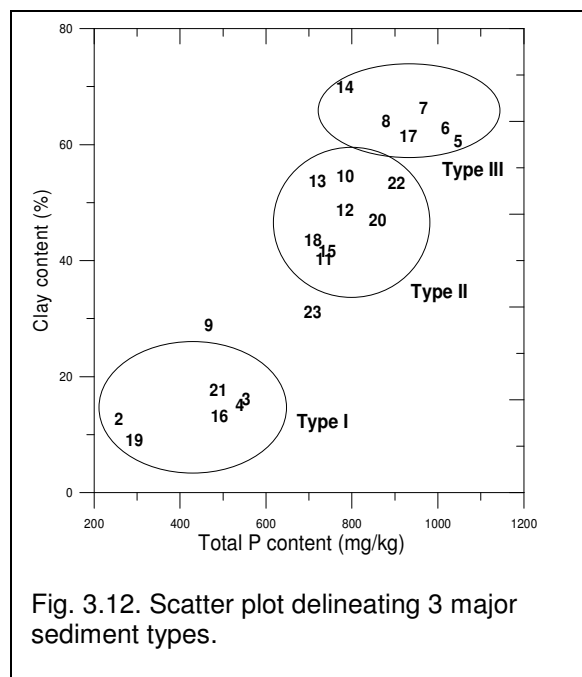
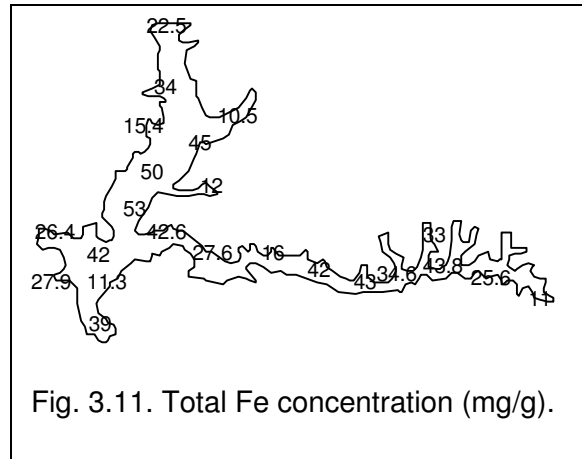


Fig. 3.10. Porewater $\text{NH}_4\text{-N}$ concentration (mg/L).

A substantial amount of iron was found in the sediments (Fig. 3.11), with an average of 93% of total iron in the reduced form. Total iron was modestly correlated to inorganic phosphorus and clay ($R^2 = 0.54$ and 0.56 , respectively). Sites with high amounts of total iron (majority of which was present as Fe II) also contained high inorganic phosphorus contents. The highest amount of total Fe was present in the central-most portion of the high speed zone and near the dam (Fig. 3.11).

To establish major groups or sediment types at Canyon Lake, a simple graphical cluster analysis of sediment properties was conducted. The analysis included creating a series of scatter plots and examining them for clusters of sediment properties. For example, Fig. 3.12 shows a scatter plot of total P vs. clay content. The label points represent sampling sites in the lake, which appear to be grouped into 3 types of sediment. In this classification, 26% of the sites fell into Type I sediment, 35% into Type II, and 26% fell into the Type III group. Two sites, 9 and 23 did not fall into either of the categories (Fig. 3.12). However, because Site 23 was monitored as part of our study, it is important to denote this site as intermediate to Type I and II sediments. Scatter plots using other independent variables (e.g., total nitrogen, total sulfur) resulted in similar classifications, however, outliers varied between 2 and 4 sites (including 9, 23, 16, and 19). (See Figure 1.1 for site identification.)

The Type I sediments were comprised of coarse textured material, with lower organic carbon, total P, total N, and total S contents relative to other sites within the reservoir. Generally the Type I sediments encompassed the shallow embayments of Canyon Lake, with the exception of a deeper site close to the dam (site 2). The Type III group were finely textured and



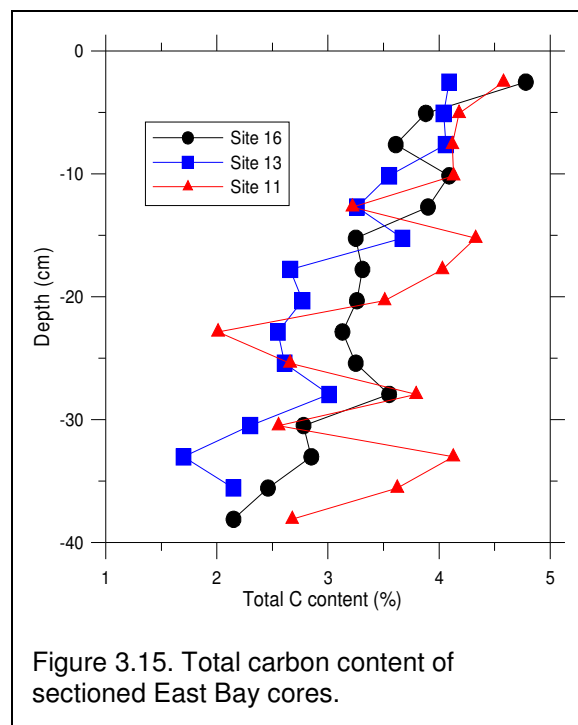
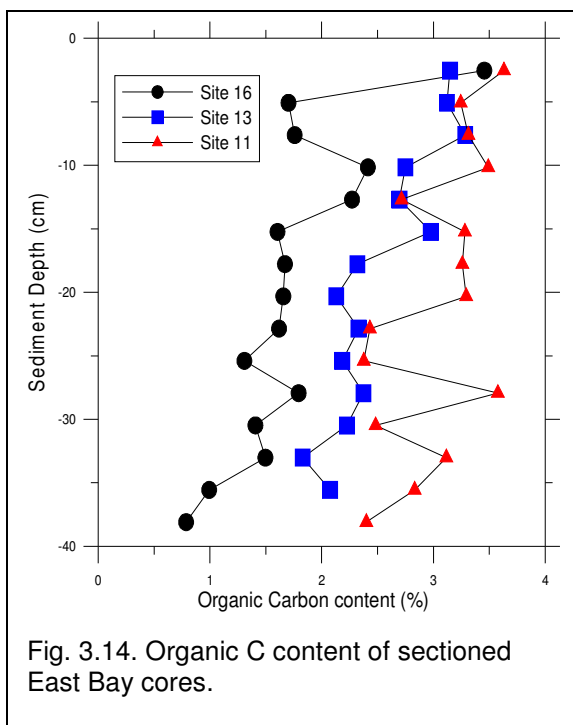
found mainly at the deeper sites located around the main/central portion. Type III sediments were characterized by higher organic contents (total P, total N, total S, organic carbon). Type II was transitional between Types I and III, located in East Bay and the north portion of the reservoir. An approximate spatial distribution of the 3 sediment types in the lake is shown in Fig. 3.13. A summary of the mean properties of each type is provided in Table 3.3. Two sites, 9 and 23, did not fall under this classification system.

The estimated total surface area of Types I, II, and III were 61.3 acres, 143.3 acres, and 93.9 acres, respectively. The amount of organic C, total N and other constituents appear to increase with increasing clay contents, which in turn increases with depth. This phenomenon can be explained by particle focusing in deeper areas, where net delivery of organic matter is higher than at littoral or shallow zones. Table 3.3 shows this distinction, where sediment total S and total P more than doubles, average porewater $\text{NH}_4\text{-N}$ concentration doubles, and sediment organic P more than quadruples in quantity from Type I to Type III sediments.

| Mean Property | Units | Type I | Type II | Type III |
|--------------------|-------|-------------|------------|-------------|
| <i>Sediment</i> | | | | |
| Area | acres | 61.3 | 143.3 | 93.9 |
| Depth | ft. | 13.7 ± 9.3 | 12.8 ± 7.8 | 28.5 ± 10.1 |
| Sand | % | 45.1 ± 19.1 | 2.7 ± 2.9 | 2.8 ± 3.7 |
| Silt | % | 40.6 ± 17.1 | 49.1 ± 4.5 | 33.8 ± 4.6 |
| Clay | % | 14.3 ± 3.0 | 48.2 ± 5.7 | 64.5 ± 3.5 |
| Total C | % | 2.4 ± 1.2 | 4.0 ± 0.4 | 4.2 ± 0.5 |
| Organic C | % | 2.2 ± 0.9 | 2.8 ± 1.5 | 3.6 ± 0.4 |
| CaCO ₃ | % | 2.2 ± 2.5 | 4.4 ± 4.0 | 4.7 ± 1.6 |
| Total N | % | 0.3 ± 0.1 | 0.4 ± 0.0 | 0.5 ± 0.0 |
| Total S | % | 0.4 ± 0.2 | 0.7 ± 0.2 | 0.9 ± 0.2 |
| Total P | mg/kg | 437 ± 128 | 780 ± 69 | 937 ± 96 |
| Inorganic P | mg/kg | 382 ± 165 | 578 ± 44 | 672 ± 155 |
| Organic P | mg/kg | 55 ± 98 | 202 ± 60 | 265 ± 111 |
| <i>Porewater</i> | | | | |
| SRP | mg/L | 2.61 ± 1.4 | 2.8 ± 0.9 | 3.0 ± 0.5 |
| NH ₄ -N | mg/L | 11.18 ± 4.0 | 14.9 ± 2.5 | 22.0 ± 11.0 |

In addition to the sediment characterization described above, intact cores from sites 11, 13 and 16 within the primary channel of East Bay were collected and analyzed. Site

16 is adjacent to the entry point of Salt Creek, site 13 is about 500 m into East Bay, and site 11 is a few hundred m past site 11 near the middle of East Bay (Fig. 2.1). A number of differences were seen between the sites. Organic C was lowest at all depths throughout the core from site 16 (closest to the confluence with Salt Creek), and increased with increasing distance into East Bay in the order site16<site 13<site 11 (Fig. 3.14). Site 16 was categorized under the coarser, Type I sediments while both Sites 11 and 13 fell into the transitional, Type II group. Organic carbon content decreased slightly with depth within the sediment at all three sites (Fig. 3.14). Total carbon contents were similar for all 3 sites and, like organic C, also decreased with depth, from about 4% at the surface to 2-3% 35-40 cm beneath the sediment surface (Fig. 3.15).



Total nitrogen contents within the sediment cores showed similar vertical and spatial gradients as organic carbon, where concentrations were lowest at site 16 and highest at site 11 (*i.e.*, increased with distance from Salt Creek further into East Bay) and decreased with increasing depth into the sediments (Fig. 3.16). It is interesting to note that calculated organic carbon:total nitrogen ratios were close to 10 for both sites 11 and 13 within the vertical profile, while organic C:N ratios for site 16 increased from 7.2 at the bottom of the profile to 11.9 in the top 2.5 cm sediment layer. Ratios near 6 have been reported for phytoplankton, with substantial deviation from this value

indicating preferential removal of N from the sediments and/or non-algal inputs of organic matter to the sediments.

Canyon Lake's record indicates often limited amount of inflow from the Salt Creek tributary due to lack of rainfall, strongly suggesting that accumulation of organic matter at the reservoir is primarily autochthonous (produced within the lake). According to Meyers and Ishiwatari (1993), autochthonous organic matter typically have low C:N ratios (<10), whereas allochthonous terrestrial organic matter is enriched in humic, high molecular weight C-rich compounds and C/N ratios are generally between 20-30.

Calcium carbonate contents were found to be low for sites 11 and 13 when compared to site 16 (Fig. 3.17). CaCO_3 contents averaged ~13% for site 16 and 5 - 8% for sites 11 and 13. A gradual accumulation of CaCO_3 in recently deposited (*i.e.*, near surface) sediments was found at all three sites (Fig. 3.17).

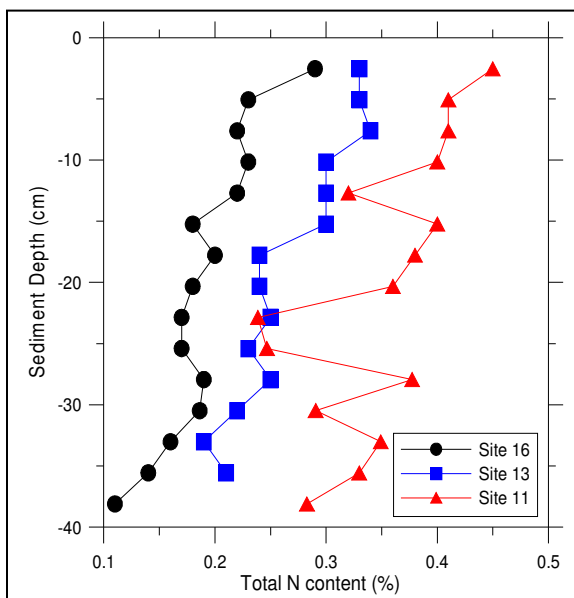


Figure 3.16. Total nitrogen content of East Bay cores.

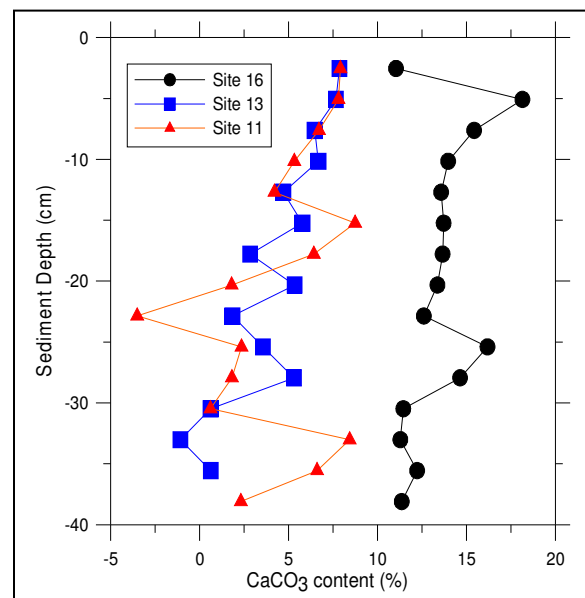
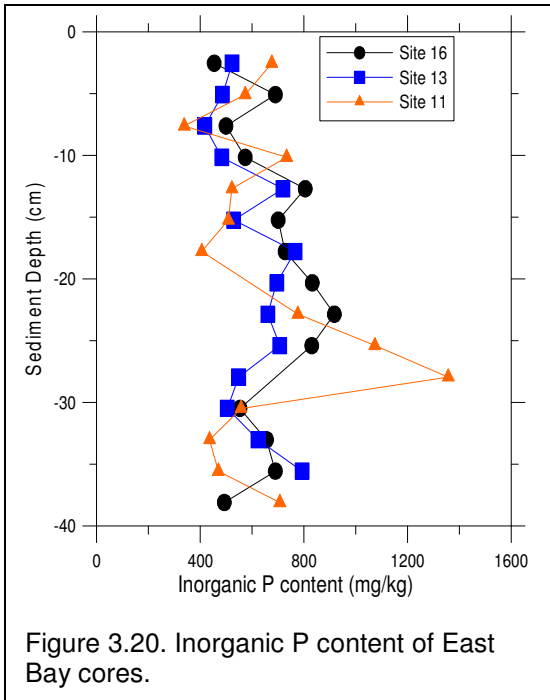
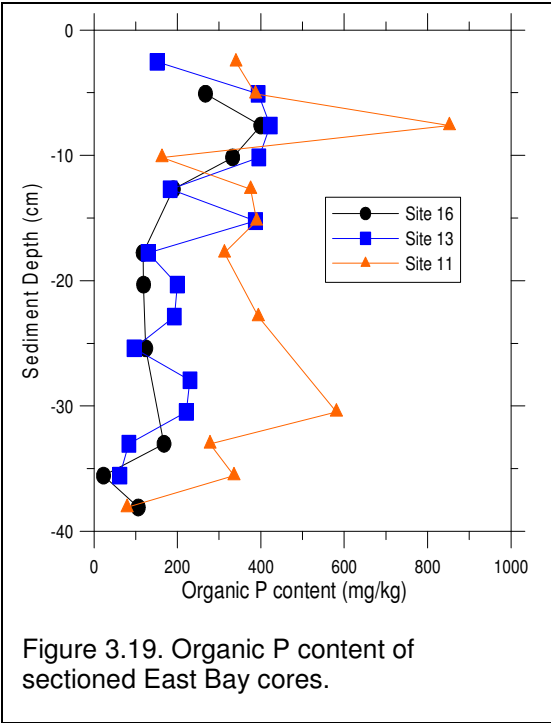
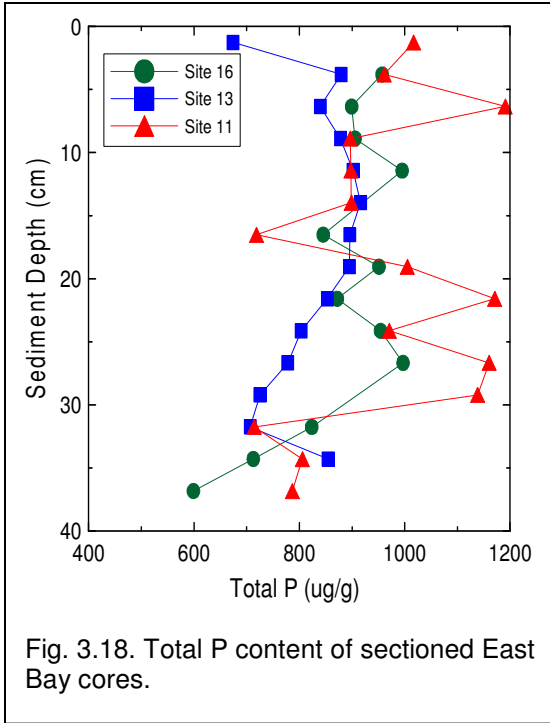


Figure 3.17. CaCO_3 contents of East Bay cores.

The composition of total, organic, and inorganic phosphorus was quite variable within the sediment profiles of East Bay (Figs. 3.18 – 3.20). The distribution of phosphorus fractions showed that the operationally-defined inorganic P fraction extractable with 1 M HCl dominated the total P in the sediments.



4.0 NUTRIENT RELEASE FROM SEDIMENTS

In order to quantify direct loading of nutrients from the sediments to the water column at Canyon Lake, laboratory studies as well as *in situ* measurements of porewater concentration gradients were conducted.

4.1 Methods

4.1.1 Core-flux studies

Cores were collected following Beutel (2000) and Anderson (2001). An Ekman dredge was used to collect a grab sample, which was then subsampled by carefully inserting a 30.5 cm by 6.3 cm diameter Lucite tube approximately 10 cm into the sediment. The presence of a layer of fine organic matter at the top of the sediment surface represented fresh detrital material and was generally indicative of a good core. The bottom of the core was then sealed using a rubber stopper and secured. The core was then carefully topped off with bottom water sampled using a van Dorn sampler and stoppered with zero headspace. Triplicate cores were collected at each site with sites selected to represent the principal sediment types.

Cores were then incubated in the dark at the temperature and DO levels measured at the time of sampling. Approximately 10 mL of water was removed daily, filtered and analyzed for soluble $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$ and SRP using an Alpkem autoanalyzer following standard methods (APHA, 1989). Dissolved oxygen and temperature of each core was measured using a YSI Model 55 DO meter. The water was briefly sparged with N_2 or lab air as needed to maintain DO and to very gently mix the water column within the core. Care was taken to not disturb the sediment when sampling or sparging. The measured change in concentration was used in conjunction with water volume and sediment-water interfacial area to calculate a mass flux rate.

4.1.2 Peeper studies

Field-based rates of nutrient release from sediments were also determined from *in situ* pore water profiles collected using peepers. Duplicate peepers were placed at four locations around the lake within the major sediment zones of the basin. Peepers were allowed to equilibrate 3-4 weeks and were then removed and analyzed for $\text{NO}_3\text{-N}$, $\text{NH}_4\text{-}$

N, and SRP using the Alpkem Autoanalyzer as above. Diffusive flux from the sediments was then calculated by substituting the nutrient gradients derived from linear regression of the peeper data into Fick's first law, which states that flux is proportional to the concentration gradient; that is:

$$J = -\phi^n D (dC / dz) \quad (1)$$

where J is the diffusive flux in mass per unit area per unit time; ϕ is the porosity; n is a correction for sediment tortuosity (~ 2); D is the aqueous diffusion coefficient, and dC/dz is the pore water concentration gradient (Hesslein, 1976).

4.1.3 Hypolimnetic mass balance

In addition to the flux estimates made using the core-flux and peeper methods, nutrient flux from the deep (principally type III sediments) was estimated from the accumulation of dissolved nutrients within the hypolimnion over the summer-stratified period. That is, since thermal stratification effectively isolates the colder hypolimnetic water from the warmer, well-lit epilimnion, nutrients and other dissolved constituents accumulate below the thermocline (*e.g.*, Fig. 2.2.b). Thus, with information about the volume of the hypolimnion, the approximate sediment area, and the change in nutrient concentration over time, one can estimate the average nutrient release rate (Cooke *et al.*, 1986).

4.2 Results

4.2.1 Core flux experiments

Triplicated intact cores were taken from a total of 7 sites at Canyon Lake, with the seventh site added during the third monitoring period. Cores were collected and monitored for a total of five sampling periods. Sites 2, 4, 7, 8, 11, and 22 were selected based on the 'sediment type' classification (Fig. 3.11). With the addition of Site 23 (the point of convergence of the main body and the upper basin), it was possible to broaden our study. Site 23 did not fall directly into the sediment type classification, however as mentioned previously this site consisted of properties intermediate to the Type I and Type II group. Sites 2, 7, and 8 were below the thermocline and thus were exposed to

much lower temperatures and lower DO than shallower sites (4 and 11). Site 22 was located within the thermocline and transitional in temperature and DO.

Results of the core-flux experiments indicate that the sediments serve as a major source of SRP and $\text{NH}_4\text{-N}$ to the water column of Canyon Lake. Figure 4.1 shows calculated mean SRP flux rates from September 2001 to August 2002. Dissolved oxygen contents, temperature measurements, and flux rate values are given in Table 4.1.

Some seasonal trends influencing rates of SRP release (Fig. 4.1) in the reservoir are clear. During the summer (Sept. 2001 and May 2002 sampling), lower SRP flux rates were observed for the colder, deeper and often anoxic sites (2, 7, and 8) when compared to warmer, shallower sites that were often relatively well-aerated (sites 4, 11, 22, 23). The lake was completely stratified during this time period. It appears that high temperature plays a more important role in stimulating phosphorus release than simply low DO levels when the lake is stratified. Also, site 2 displayed a slight negative flux in May, indicating a net deposition of dissolved phosphorus into the sediments.

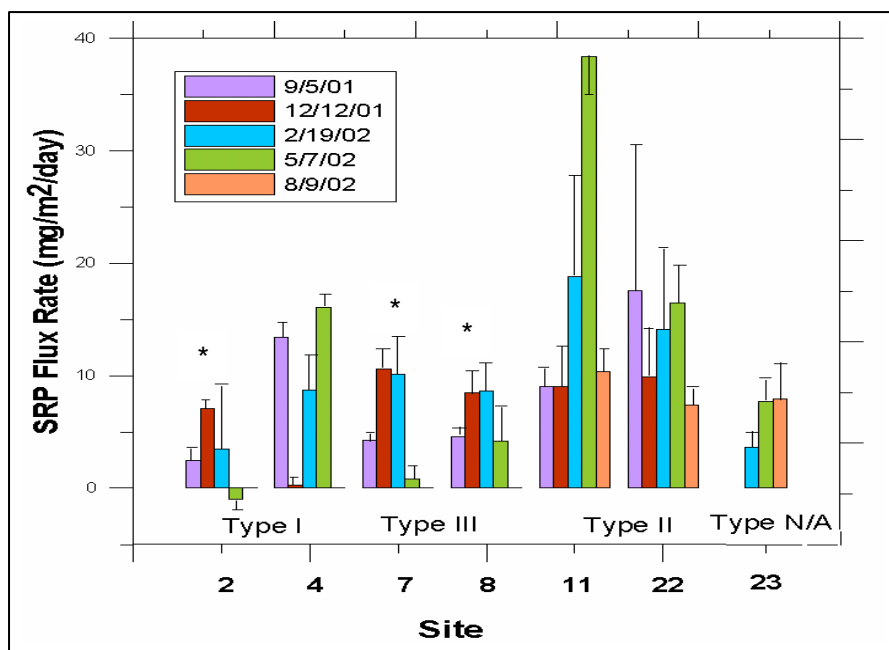


Figure 4.1. Average SRP flux rates from sediments for 7 sites in Canyon Lake (* indicates sites below thermocline).

Flux rates calculated for August 2002 are shown for sites 11, 22, and 23 only. A white precipitate formed in the overlying water of cores from sites 2, 7 and 8 (the deepwater sites), apparently as a result of CO_2 outgassing and/or flux of HCO_3^- into the

overlying water. These cores all yielded anomalously low or negative SRP and NH₄-N flux rates. Previous studies have shown that phosphate is strongly adsorbed on the surface of calcite and aragonite (Cole *et. Al.*, 1953, de Kanel and Morse, 1978, Gaudetter, 1980).

Table 4.1. Mean nutrient flux rates calculated using cores.

| Site | Type | Sampling Date | Bottom Temp (°C) | Hypolimnetic DO (mg/L) | Depth (m) | SRP Flux (mg/m ² /d) | NH ₄ -N Flux (mg/m ² /d) | NO ₃ -N Flux (mg/m ² /d) |
|------|------|---------------|------------------|------------------------|-----------|---------------------------------|--|--|
| 2 | I | 9/5/01 | 13 | 0.1 | 13 | 2.5 ± 1.4 | 24.2 ± 7.1 | N/A |
| | | 12/12/01 | 13 | 2.7 | | 7.1 ± 0.5 | 27.6 ± 3.2 | N/A |
| | | 2/20/02 | 10.4 | 3.7 | | 3.5 ± 5.3 | 32.2 ± 10.7 | -1.7 ± 0.9 |
| | | 5/7/02 | 12.8 | 0.4 | | -1.0 ± 0.9 | 21.5 ± 13.5 | -0.3 ± 0.5 |
| | | 8/9/02 | 14 | 0.3 | | N/A | N/A | N/A |
| 4 | I | 9/5/01 | 26.2 | 2.0 | 4.3 | 13.4 ± 0.9 | 39.6 ± 5.3 | N/A |
| | | 12/12/01 | 13 | 3.0 | | 0.3 ± 0.1 | 6.2 ± 1.9 | N/A |
| | | 2/20/02 | 11 | 6.4 | | 8.7 ± 4.1 | 11.3 ± 1.8 | 4.0 ± 0.4 |
| | | 5/7/02 | 16.2 | 0.5 | | 16.1 ± 1.1 | 19.2 ± 4.8 | -0.1 ± 0.1 |
| | | 8/9/02 | 26.7 | 7.4 | | N/A | N/A | N/A |
| 7 | III | 9/5/01 | 13 | 0.1 | 11.6 | 4.3 ± 0.6 | 21.8 ± 10.3 | N/A |
| | | 12/12/01 | 13 | 3.9 | | 10.6 ± 0.9 | 40.6 ± 0.7 | N/A |
| | | 2/20/02 | 10.4 | 5.1 | | 10.1 ± 4.7 | 16.2 ± 5.4 | -1.7 ± 1.4 |
| | | 5/7/02 | 12.9 | 0.4 | | 0.8 ± 1.4 | 19.9 ± 1.4 | 0.4 ± 0.5 |
| | | 8/9/02 | 15.5 | 0.3 | | N/A | N/A | N/A |
| 8 | III | 9/5/01 | 13 | 0.1 | 10.2 | 4.6 ± 0.1 | 45.8 ± 26.6 | N/A |
| | | 12/12/01 | 12 | 6.2 | | 8.5 ± 3.0 | 39.9 ± 0.6 | N/A |
| | | 2/20/02 | 10.2 | 5.4 | | 8.7 ± 1.7 | 23.4 ± 6.2 | -2.1 ± 0.5 |
| | | 5/7/02 | 11.8 | 0.6 | | 4.2 ± 2.1 | 31.1 ± 10.7 | -1.4 ± 0.6 |
| | | 8/9/02 | 13.9 | 0.3 | | N/A | N/A | N/A |
| 11 | II | 9/5/01 | 27.5 | 7.4 | 3 | 9.0 ± 1.7 | 35 ± 5.1 | N/A |
| | | 12/12/01 | 10 | 9.7 | | 9.0 ± 3.7 | 29.4 ± 6.5 | N/A |
| | | 2/20/02 | 11 | 6.6 | | 18.8 ± 8.6 | 25.1 ± 6.9 | -0.7 ± 0.3 |
| | | 5/7/02 | 20.5 | 5.7 | | 38.4 ± 3.7 | 52.2 ± 16.1 | 0.8 ± 2.5 |
| | | 8/9/02 | 25.4 | 7.7 | | 10.4 ± 2.8 | 56.6 ± 8.5 | N/A |
| 22 | II | 9/5/01 | 16 | 0.1 | 6.8 | 17.6 ± 12.1 | 62 ± 73.8 | N/A |
| | | 12/12/01 | 12 | 10.1 | | 9.9 ± 5.8 | 25.3 ± 9.8 | N/A |
| | | 2/20/02 | 10.6 | 4.4 | | 14.1 ± 8.8 | 5.6 ± 1.0 | 2.8 ± 3.6 |
| | | 5/7/02 | 16.7 | 0.5 | | 16.5 ± 3.8 | 30.2 ± 9.9 | 0.01 ± |
| | | 8/9/02 | 22.8 | 0.3 | | 7.4 ± 1.8 | 27.0 ± 20.5 | N/A |
| 23 | N/A | 2/20/02 | 11.6 | 7.9 | 2.5 | 3.7 ± 1.9 | 20.1 ± 6.2 | 1.8 ± 0.6 |
| | | 5/7/02 | 19.9 | 4.3 | | 7.7 ± 2.5 | 16.0 ± 7.0 | -0.5 ± 0.6 |
| | | 8/9/02 | 26.2 | 5.1 | | 8 ± 4.4 | 13.5 ± 1.0 | N/A |

During the December 2001 measurements approximately 2½ weeks following lake turnover, SRP flux rates were fairly uniform across the lake (~9 mg/m²/d) with the exception of Site 4 (0.3 mg/m²/d). Although DO levels were relatively high (2.7-10 mg/L), low temperatures were maintained throughout the lake (Table 4.1); nevertheless, SRP release was equal to or greater than that measured earlier in September, 2001. It seems

likely that mixing resulted in increased availability of nutrients that fostered elevated chlorophyll levels and, subsequently, also greater delivery of fresh particulate matter to the sediments. Increased fresh organic matter thus allowed microbial mineralization to take place. SRP flux rates for the deeper sites (2, 7, 8) were seen to double from September (stratified conditions) to December 2001 (following mixing). SRP flux rates in February (during which the lake was still destratified) were also high at most sites despite lower temperatures and high DO levels (Fig. 4.1). It is evident that low temperature and high oxygen concentration do not inhibit phosphorus release from the sediments following mixing.

Modest rates of $\text{NH}_4\text{-N}$ release were seen during the year and throughout the reservoir (Fig. 4.2). In general, calculated $\text{NH}_4\text{-N}$ flux rates were only about 4 times higher than SRP flux rates (Table 4.1). Summer flux rates (September 2001, May 2002) were greater than $15 \text{ mg/m}^2/\text{day}$ for all sites.

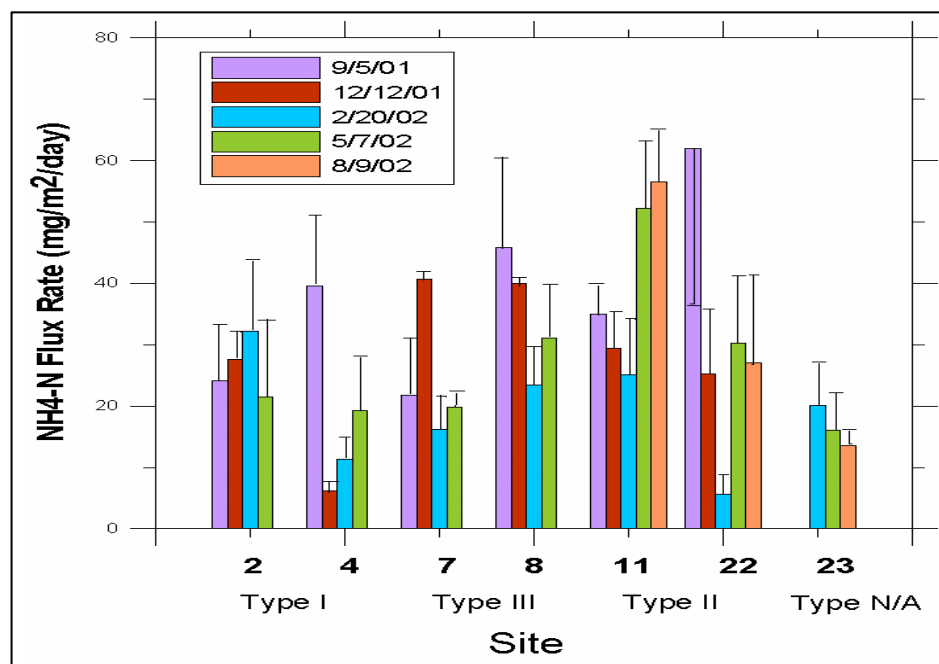


Figure 4.2. Average SRP flux rates from sediments for 7 sites in Canyon Lake.

Most sites continued to exhibit rates of $\geq 25 \text{ mg/m}^2/\text{d}$ $\text{NH}_4\text{-N}$ during well-oxygenated, unstratified conditions (post-turnover in December). Measured flux rates for sites 7 and 8 were roughly $40 \text{ mg/m}^2/\text{day}$, where site 7 exceeded its September internal loading rate

almost 2 times over. Site 2 also displayed a higher flux rate in the winter relative to September.

Some removal of $\text{NO}_3\text{-N}$ was observed (Table 4.1), although the low levels of $\text{NO}_3\text{-N}$ in the hypolimnion of the lake (often near the detection limit of 0.04 mg/L) limited our ability to routinely provide accurate measurements of loss rates.

The data from Table 4.1 were then used to calculate the annual average SRP and $\text{NH}_4\text{-N}$ flux rates for the 3 sediment types (Table 4.2). The transitional Type II (Sites 11 and 22) sediments served as the greatest source of SRP and $\text{NH}_4\text{-N}$ to the water column.

The annual average SRP release rates for the Type I and III sediments were roughly 6 $\text{mg/m}^2\text{/day}$, while $\text{NH}_4\text{-N}$ release was slightly higher in the Type III sediments (30 $\text{mg/m}^2\text{/day}$) when compared to Type II (~23 $\text{mg/m}^2\text{/day}$). Site 23, which fell in between the Type I and II category exhibited mean SRP and $\text{NH}_4\text{-N}$ flux rates of 6.4 $\text{mg/m}^2\text{/day}$ and 16.6 $\text{mg/m}^2\text{/day}$, respectively (data not shown in table).

Table 4.2. Mean SRP and $\text{NH}_4\text{-N}$ flux rates for the 3 sediment types at Canyon Lake.

| Sediment Type | Mean SRP Flux ($\text{mg/m}^2\text{/day}$) | Mean $\text{NH}_4\text{-N}$ Flux ($\text{mg/m}^2\text{/day}$) |
|----------------------|--|--|
| Type I | 6.3 | 22.7 |
| Type II | 15.1 | 34.8 |
| Type III | 6.5 | 29.8 |

4.2.2 Peeper studies

Peepers were used at Canyon Lake in order to examine vertical trends in porewater chemistry and provide an *in situ* estimate of nutrient flux. Peepers were deployed in April and August of 2002, allowed to equilibrate about 4 weeks, and then retrieved. Four sets of peepers were deployed at a total of four sites, however, we were unable to successfully recover peepers at Sites 22 and 23 for both sampling events. Nevertheless, data from peepers obtained at Sites 2 and 11 provided us with useful information about composition of solutes within the sediment porewater.

Dissolved nitrogen and phosphorus profiles exhibited similar concentration gradients in both April and August of 2002 for Site 2, with very good reproducibility between duplicates (Fig. 4.3). Duplicate peepers from Site 11 also showed close correspondence in August (Fig. 4.4), however April's measurements of duplicates were not in as good agreement.

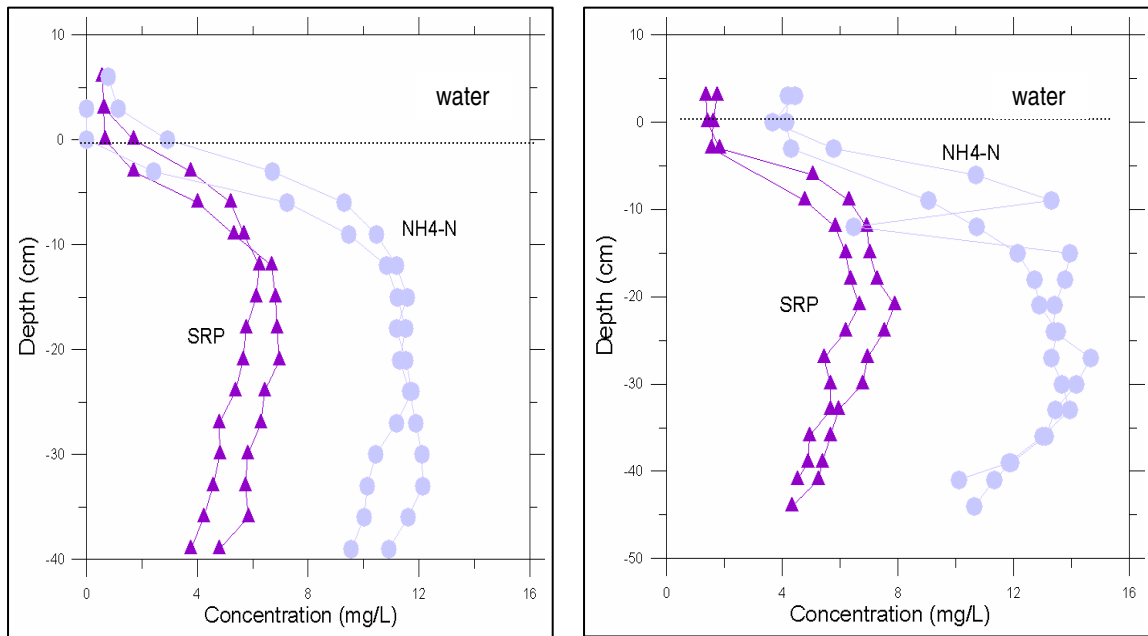


Figure 4.3. Replicate porewater nutrient profiles for Site 2 using 'peepers' on (a) 4-10-02 and (b) 8-15-02. Note that the sediments are below the dashed line.

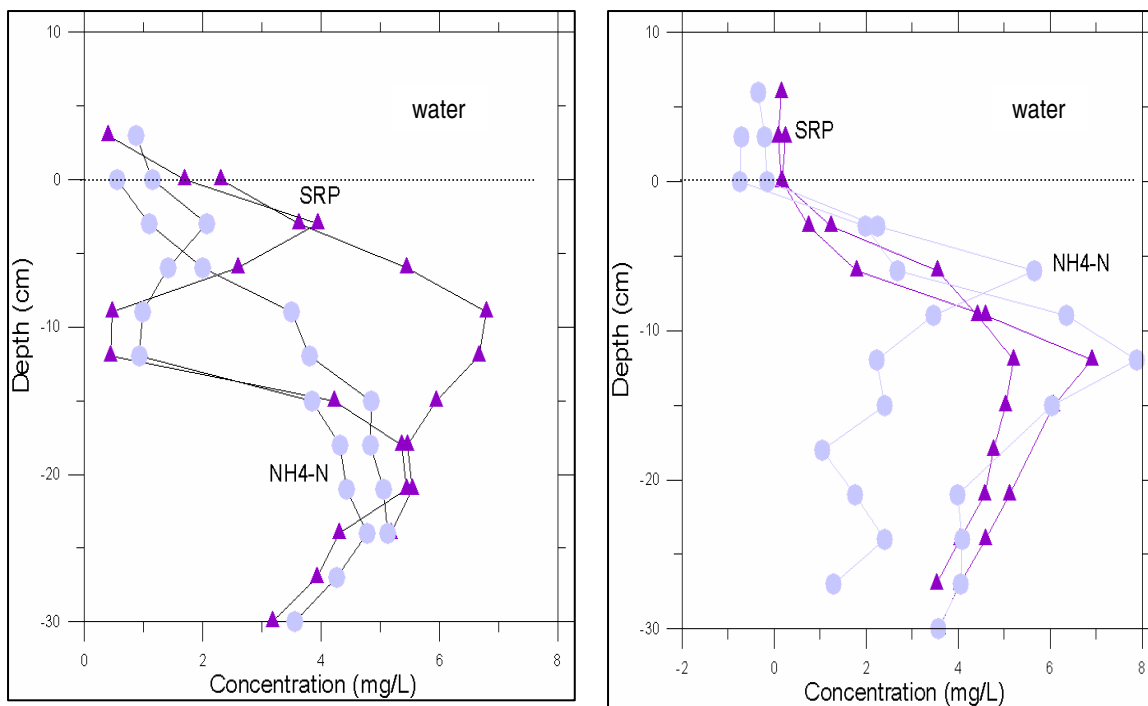


Figure 4.4. Replicate nutrient porewater profiles for site 11 on (a) 4-10-02 and (b) 8-15-02. Note that the sediments are below the dashed line.

Concentrations of $\text{NH}_4\text{-N}$ and SRP increased approximately linearly for both dates within the uppermost 10 cm of the sediment surface to concentrations of about 6-7 mg/L dissolved phosphorus and 12-14 mg/L $\text{NH}_4\text{-N}$ (Fig. 4.3). Concentrations were slightly lower in April 2002 than in August 2002, however. Concentrations of $\text{NH}_4\text{-N}$ and SRP at site 11 were lower (5-6 mg/L SRP and 6-8 mg/L $\text{NH}_4\text{-N}$) and more irregular than Site 2 (Fig. 4.4), although the measurements below ~18 cm depth were in good agreement. The reason for the variability in the April 2002 sampling is not clear.

Diffusive flux rates for nutrients calculated for April and August are given in Table 4.4. Flux rates were calculated by substituting the porewater concentration gradient (dC/dz) into Fick's 1st Law (eq. 1). Porewater SRP flux remained close to 2 mg/m²/day for both sites, whereas estimated $\text{NH}_4\text{-N}$ flux varied between 1.8-8.6 mg/m²/day (Table. 4.4). The $\text{NO}_3\text{-N}$ concentrations were low (near detection limit of 0.04 mg/L) and variable during both seasons and were observed to decrease slightly from the sediments to the water column. Due to the variability and low concentration range, flux rates for $\text{NO}_3\text{-N}$ were not calculated, although they would clearly be low (<0.05 mg/m²/day).

| Date | Site | Mean SRP flux (mg/m ² /day) | Mean $\text{NH}_4\text{-N}$ flux (mg/m ² /day) |
|---------|------|--|---|
| 4-10-02 | 2 | 1.9 ± 0.2 | 8.6 ± 0.9 |
| 8-15-02 | 2 | 2.4 ± 0.8 | 6.9 ± 3.1 |
| 4-10-02 | 11 | 1.9 ± N/A | 1.8 ± N/A |
| 8-15-02 | 11 | 1.6 ± 0.8 | 6.0 ± 1.8 |

As previously noted for Lake Elsinore, SRP and $\text{NH}_4\text{-N}$ flux rates measured using peepers were considerably lower than flux rates measured using intact sediment cores (Anderson, 2001). The differences seen between the two methods have been attributed to mineralization of surficial sediments not measured using peepers (Anderson, 2001). Other researchers (*e.g.*, Urban *et al.*, 1997) have also concluded that solute fluxes measured using peepers are more representative of processes occurring within the sediments and may fail to provide true estimates of overall internal loading.

In addition to the porewater nutrient profiles, strong vertical gradients were also found for pH, sulfide, and alkalinity. Sulfide and pH porewater profiles are shown in Figure 4.5 while alkalinity is shown in Figure 4.6. Concentrations of sulfide (mainly as HS^-) reached a maximum of 12.6 mg/L within 3-6 cm beneath the sediment surface at the deepwater, main body site 2, and between 4-6 mg/L at the shallow, East Bay site 11.

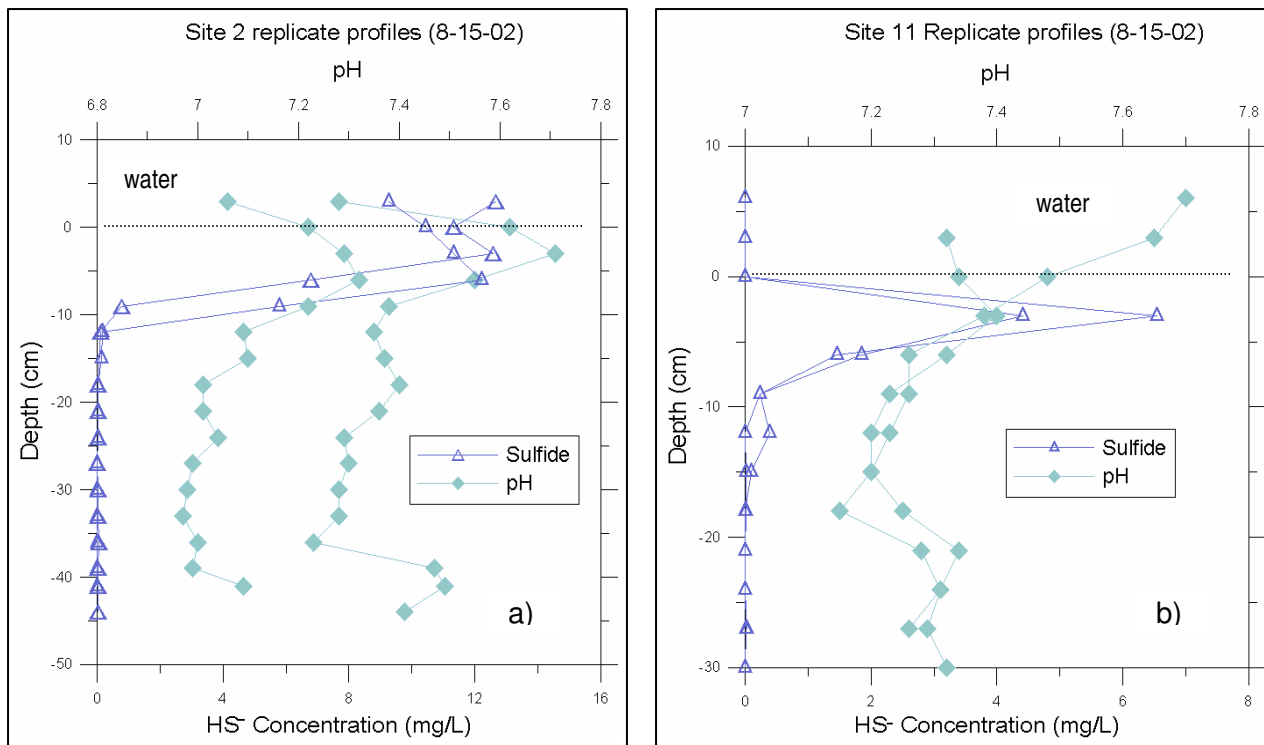


Figure 4.5. Sulfide and pH porewater profiles at (a) site 2 and (b) site 11 (8-15-02). Note that the sediments are below the dashed line.

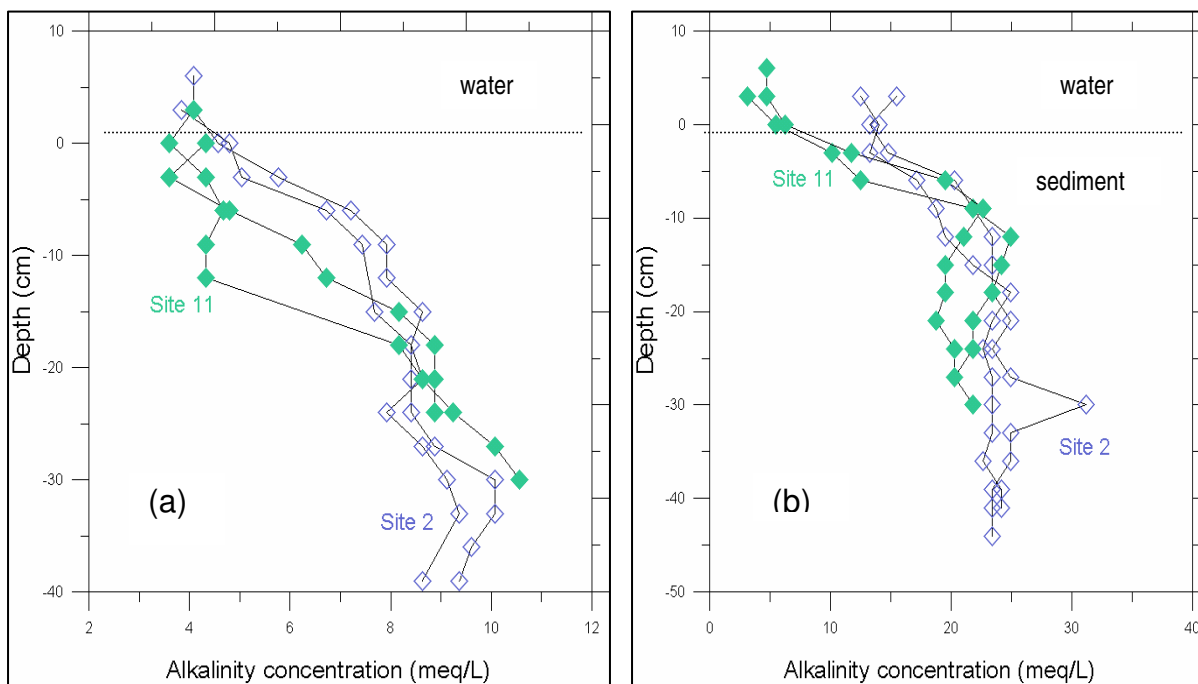


Figure 4.6. Alkalinity porewater profile at sites 2 and 11 on (s) 4-10-02 and (b) (8-15-02).

Sulfide at site 2 appeared to be high in the water column as well during late summer, between 9-12 mg/L (Fig. 4.5a), while no detectable HS^- was found in the water column of East Bay (Fig. 4.5b). pH levels ranged from 7 – 7.6 and were lowest at depth (>15 cm into the sediment) (Fig. 4.5).

Alkalinity levels increased with decreasing pH and were observed to more than double from April (maximum of 10-11 meq/L) to August (~25 meq/L) (Fig. 4.6). The increase in alkalinity, sulfide and ammonium levels from spring (April) to summer (August) is evidence of increased rate of sulfate reduction in the summer.

Dissolved iron and manganese concentrations were also measured in the sediment porewaters. Porewater Mn concentrations were higher at site 2 than site 11, and increased with depth reaching concentrations of 5 and 1.5 mg/L, respectively (Fig. 4.7a).

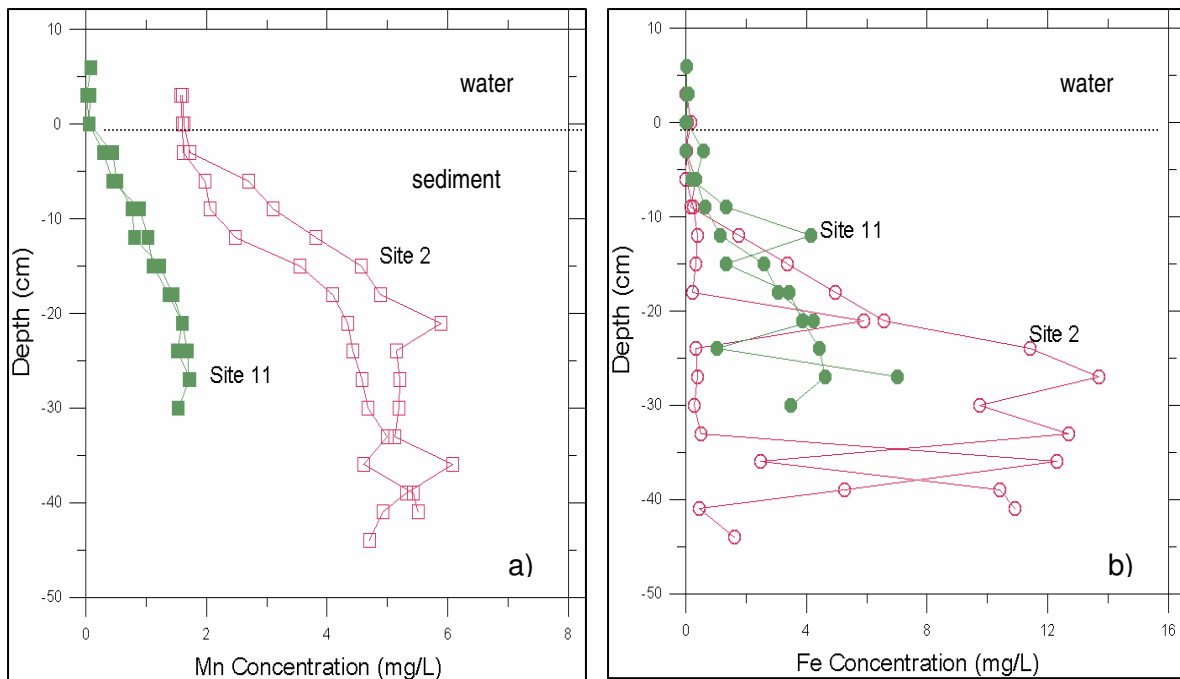


Figure 4.7. Replicate porewater (a) manganese and (b) iron concentrations at sites 2 and 11 on 8-15-02. Note that sediments are below the dashed line.

Mn profiles exhibited good reproducibility between duplicate peepers and remained largely unchanged between April and August (profiles not shown for April). Iron concentrations exhibited high variability at site 2. The high variability observed in iron is most likely caused by iron oxyhydroxide precipitation. Studies have shown that the rate of oxygen diffusion from the atmosphere is high ($4 \mu\text{M}/\text{min}$) and that Fe (II) is oxidized rapidly at ambient pH ($t_{1/2}=2.34 \text{ min}$ at pH 8; $t_{1/2}=234 \text{ min}$ at pH 7) (Carnigan, 1984;

Davison and De Vitre, 1992). Porewater pH was near 7-7.5 and subsampling of peeper chambers took up to ~10-15 min, so some oxidation of Fe^{2+} may have occurred during processing.

An overview of the peeper data indicates a strong vertical distribution of SRP, $\text{NH}_4\text{-N}$, HS^- , pH, alkalinity, Fe, and Mn within the sediment porewater at Canyon Lake. The concentrations of each species appear to be controlled by a combination of mineral dissolution and desorption processes. As seen with Lake Elsinore, nutrient flux rates measured using peepers were typically only 10-30% of the rate measured using the core-flux method, clearly indicating that mineralization within the first few mm of surface sediments is not measured using peepers.

4.2.3 Hypolimnetic mass balance

The average rate of internal loading from profundal sediments over the summer was calculated using the hypolimnetic mass balance method. Since the project period ran from the summer of 2001 to the summer of 2002, it was not possible to sample a complete cycle from spring stratification to fall mixing. Thus, spring 2002 data was used in conjunction with fall 2001 data to represent the summer-stratified condition.

For these calculations, the hypolimnion was considered to include all water in the main body of the lake found at depths >7 m. Seven m was taken as the summer-average depth, recognizing that the thermocline tends to deepen over the course of the summer due to hypolimnetic entrainment (*e.g.*, from about 6.5 m shortly after stratification in the spring to ~8 m in the fall). Lake elevation and bathymetric data (Fig. 3.1) were used to estimate the surface area of the profundal sediments (124 acres or $5.02 \times 10^5 \text{ m}^2$) and the mean depth of the hypolimnion (3.25 m). The average dissolved nutrient concentrations from the hypolimnion from site 2 in the main body of the lake were used for calculations made using UCR data (Table 4.4). Data collected by the RWQCB (Cindy Li, pers. comm.) were also used for comparison. Here, the average of the middle and bottom concentrations at a site near the dam were used (Table 4.4). This is necessary since a strong nutrient concentration gradient was present within the hypolimnion (*e.g.*, Fig. 2.2b). Use of just the bottom concentration would result in a substantial overestimate of SRP and $\text{NH}_4\text{-N}$ accumulation within the hypolimnion.

Results of these calculations indicate that approximately 200 kg of SRP was released from the profundal sediments over a period of 7 months. Moreover, good

agreement was found between calculations made using UCR and RWQCB data (Table 4.4). The differences between the absolute SRP concentrations used for the calculations (UCR vs. RWQCB) appear to be due to the different sites used, although the relative differences were quite similar between the two data sets. Mass balance calculations indicate about 10x higher loading of $\text{NH}_4\text{-N}$ to the hypolimnion when compared with SRP (1425 – 2310 kg $\text{NH}_4\text{-N}$ vs. 182 – 214 kg SRP, respectively).

Table 4.4. Estimates of internal loading using hypolimnetic mass balance method.

| | Concentration: April 2002 (mg/L) | Concentration: October 2001 (mg/L) | Change in Concentration (mg/L) | TP Mass (kg) | Flux Rate (mg/m²/d) |
|--|---|---|---|-------------------------|---|
| <i>SRP</i> | | | | | |
| UCR | 0.57 | 0.68 | 0.11 | 182 | 1.7 |
| RWQCB | 1.44 | 1.57 | 0.13 | 214 | 2.0 |
| <i>NH₄-N+NO₃-N</i> | | | | | |
| UCR | 0.64 | 1.50 | 0.86 | 1425 | 13.3 |
| RWQCB | 0.30 | 1.70 | 1.40 | 2310 | 21.6 |

It is instructive to compare the internal loading rates estimated by the hypolimnetic mass balance with the core-flux method. A UCR-estimated internal loading rate of 182 kg corresponds to an areally-averaged internal loading rate of 1.7 mg/m²/d for this period, while the RWQCB data points to a slightly higher nutrient flux rate of 2.0 mg/m²/d (Table 4.4). Using core-flux measurements over this same period (September 2001 and May 2002) from the 2 deepwater sediment sites in the main body (sites 2 and 7) yields an average SRP release rate of 1.6 mg/m²/d. Thus, the 2 methods (core-flux and hypolimnetic mass balance) are in good agreement with each other.

Similarly, the average of the $\text{NH}_4\text{-N}$ core-flux rates from sites 2 and 7 for September 2001 and May 2002 (21.8 mg/m²/d) is in fair to good accord with the hypolimnetic mass balance estimates of 13.3 and 21.6 mg/m²/d using the UCR and RWQCB data sets, respectively (Table 4.4). The general agreement between the 2 methods is heartening and provides an increased measure of confidence in the internal loading estimates.

5.0 SEDIMENTATION AND PARTICULATE-BOUND NUTRIENT DEPOSITION

In order to understand the cycling and turnover of nutrients within Canyon Lake, it is necessary to measure both their *release from* the sediments (Section 4.0) and their *deposition to* the sediments. The relative rates of these two processes help define the efficiency of nutrient use within Canyon Lake and the potential for burial and removal of N and P from the system. This study assessed the downward vertical flux of particles in the water column through use of sediment traps.

5.1 Methods

A series of cylindrical sediment traps with a 20 cm diameter and 60 cm height were constructed out of PVC pipe fitted with an endcap. It has been previously reported that a height/width ratio of 3.0 most accurately traps suspended material in waters with local velocities less than 10-20 cm/s (Gardner, 1980). Traps were deployed at 7 sites using subsurface buoys at the sites used for nutrient flux measurements (the site north of the causeway was replaced by the site at the dam). Mid-depth and bottom traps were positioned at sites 2 and 7; bottom traps were deployed at the remaining sites (Fig. 5.1). Traps were deployed on 3 different dates spanning the time period of November, 2001 – August, 2002. Following retrieval of the sediment traps, the top ~80% of the water was slowly siphoned out, reducing the volume from 18.8 L to only about 4 L per trap.

The suspended solids within the trap sample are allowed to settle overnight, the upper ~80% of the water again carefully siphoned off, and the remainder was then transferred to a series of 250 mL centrifuge bottles. The suspended material was then collected, dried to 70 °C, weighed, and analyzed for total CNS, CaCO₃, and total, organic and inorganic P as described in Section 3.1.

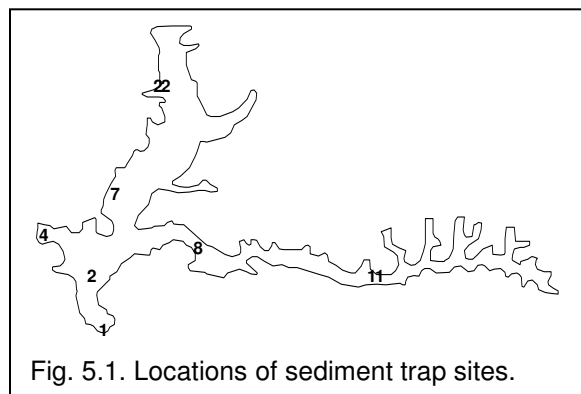


Fig. 5.1. Locations of sediment trap sites.

5.2 Results

The material recovered in the sediment traps was high in total and organic C, CaCO₃, and total N and P (Table 5.1). The traps deployed at the end of November, 2001 were in place after mixing and yielded the highest relative concentration of total C, total N and total P (mean values of 15.7%, 2.3%, and 1635 µg/g, respectively) (Table 5.1). Insufficient material was available to complete all analyses, however (N/A). The lake was stratified by May and lower total C, N, and P contents were observed. Particulate sulfur contents were seen to increase from 0.4% to 0.8% between late fall and summer.

Analysis of sediment trap material point toward seasonally differing compositions of sedimentary materials. The properties of trapped material indicate that phytoplankton played a major role in the formation of particles in Canyon Lake. Average C:N:P mass ratios were 30:3.5:1, which, when compared with the Redfield mass ratio of 40:7:1, suggests nitrogen limitation in algal cells. Changes in the concentration of particulate matter were in tune with changes in algal biomass, where higher contents of total C, N, P, and organic C were observed in late fall. CaCO₃ was highest in May, during which the consumption of CO₂ in the euphotic zone by phytoplankton most likely led to formation of calcium carbonate in the water column (Table 5.1). The precipitated calcite was ultimately deposited to the sediments, where it could potentially undergo some dissolution owing to the high CO₂ and lower pH environment at the sediment-water interface. CaCO₃ contents averaged about 16% in the trapped material. Among the limited CaCO₃ measurements conducted (due to sample size constraints), available values indicated that the production of lacustrine calcite was greatest in the main body (sites 2, 4, and 7). Some CaCO₃ values for sites 2 and 4 are not shown in Table 5.1, however, since analysis of material from these 2 sites yielded CaCO₃ levels that were in exceedance of manometer detection limits. Thus values were calculated based on maximum detection limits of the manometer (*e.g.*, >22% calcite present in sites 2 and 4 in May). Organic P comprised about 50% of the total P found in the trapped particulate matter.

| Date | Site | Total C (%) | Organic C (%) | CaCO ₃ (%) | Total N (%) | Total S (%) | Total P (µg/g) | Organic P (µg/g) |
|--------------------|--------------------|-------------|---------------|-----------------------|-------------|-------------|----------------|------------------|
| 11-30-01 | 2- middle | 26.3 | N/A | N/A | 4.03 | 1.13 | N/A | N/A |
| | 2- bottom | 27.4 | N/A | N/A | 4.68 | 0.44 | N/A | N/A |
| | 4- bottom | 13.4 | N/A | >18.5 | 0.61 | 0.00 | 1017 | 413 |
| | 7- middle | 22.7 | N/A | N/A | 3.91 | 0.47 | 7459 | 4933 |
| | 7- bottom | 14.6 | 13.0 | 13.6 | 2.17 | 0.48 | 3691 | 1903 |
| | 8- bottom | 10.2 | 9.3 | 7.2 | 1.66 | 0.58 | 2995 | 1526 |
| | 11-bottom | 3.4 | 1.8 | 13.5 | 0.30 | 0.00 | 836 | 159 |
| | 22-bottom | 7.8 | 5.9 | 15.2 | 0.93 | 0.33 | 2602 | 873 |
| <i>Mean values</i> | | 15.7 | 7.5 | >13.5 | 2.29 | 0.43 | 3100 | 1634 |
| 5-13-02 | 1-bottom | 10.4 | N/A | N/A | 1.43 | 0.95 | 3213 | 1985 |
| | 2-middle | 10.2 | N/A | N/A | 1.55 | 0.82 | 4100 | 2206 |
| | 2- bottom | 12.8 | N/A | >22 | 0.98 | 0.16 | 2346 | 1083 |
| | 4 bottom | 7.6 | N/A | >22 | 0.94 | 0.51 | 2944 | 1077 |
| | 7 middle | - | N/A | N/A | N/A | N/A | 2578 | 1412 |
| | 7 bottom | 8.5 | 4.6 | 32.3 | 0.80 | 0.48 | 1726 | 901 |
| | 8 bottom | 9.9 | 8.8 | 9.2 | 1.47 | 0.75 | 3061 | 2008 |
| | 11 bottom | 3.8 | 2.0 | 15.1 | 0.36 | 0.40 | 4916 | 5191 |
| | 22 bottom | 5.8 | 4.3 | 12.8 | 0.71 | 0.47 | 1798 | 756 |
| | <i>Mean values</i> | | 8.6 | 4.9 | >18.9 | 1.03 | 0.57 | 2721 |
| 8-30-02 | 1-bottom | 15.6 | N/A | N/A | 1.80 | 1.06 | 4537 | N/A |
| | 2-middle | 12.9 | N/A | N/A | 1.70 | 1.43 | 5150 | N/A |
| | 2-bottom | 16.4 | N/A | N/A | 2.14 | 1.32 | 5744 | N/A |
| | 4-bottom | 9.1 | N/A | 26.0 | 0.79 | 0.09 | 2817 | 918 |
| | 7-middle | 5.6 | 4.2 | 11.8 | 0.65 | 0.42 | 2149 | 847 |
| | 7-bottom | 15.3 | N/A | >26 | 1.02 | 0.10 | 2280 | 950 |
| | 8-middle | 13.2 | N/A | N/A | 1.88 | 1.30 | 3562 | N/A |
| | 8-bottom | 6.5 | 4.7 | 14.8 | 0.72 | 1.15 | 2521 | 1163 |
| | 11-bottom | 6.0 | 4.5 | 11.9 | 0.73 | .59 | 1883 | 972 |
| | 22-bottom | 9.2 | 7.4 | 15.5 | 1.20 | .83 | 2647 | 1312 |
| <i>Mean values</i> | | 11.0 | 5.2 | 16.0 | 1.26 | 0.83 | 3329 | 1027 |

Note: Calcium carbonate values with a (>) sign were a result of samples exceeding manometer detection limits, as described in the text.

Sedimentation rates showed a great deal of seasonal and spatial variability (Fig. 5.2). Sedimentation rates were generally slightly higher for bottom traps versus for those deployed at middle depths. Site 11 (East Bay) displayed an unusually high particulate sedimentation rate on May 13, 2002 (737 g/m²/d), indicating heavy disturbance of the sediment layer; as a result, data for this site is not shown in Fig. 5.2. Sedimentation rates measured for Site 11 in November 2001 and August 2002 were 50 and 40 g/m²/day, respectively. The very high value found in May 2002 suggests that mechanical resuspension of bottom sediments does occasionally occur, and significantly affects particulate deposition rates at East Bay. As a result, such values can not be used as a

direct measure of natural sedimentation processes. Sedimentation rates in parts of the main body (sites 4, 7, 8) were highest in August, while sites 1 and 2 (near the dam) displayed higher flux rates in May (Fig. 5.2). In general, the maximum downward flux of particles in the main body took place during the stratified period (May and August), although sedimentation rates were relatively high following turnover in November in the north portion of the lake and at parts of the main body. Sedimentation rates in the deep sites (2, 7, 8) ranged from 1.16 - 20.9 g/m²/day; shallower sites ranged from 7.36 - 50 g/m²/day (excluding the May 13, 2002 measurement at site 11).

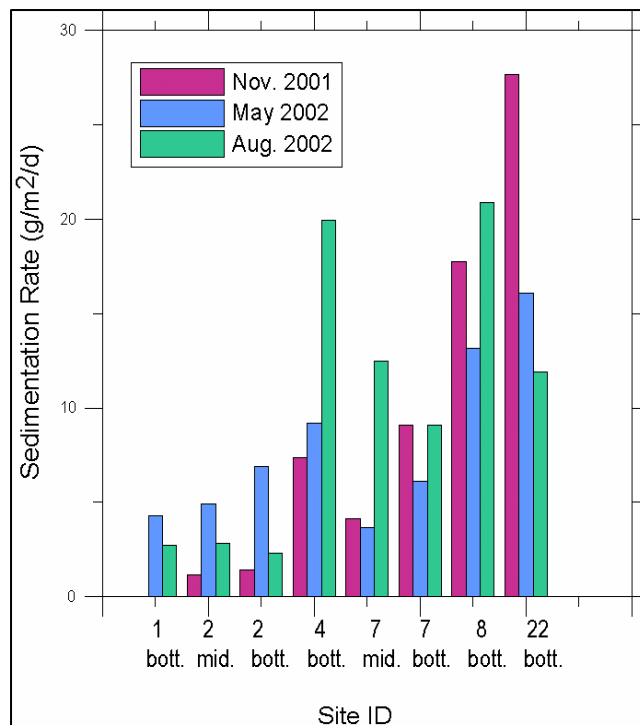


Figure 5.2. Mass sedimentation rates measured using sediment traps. Note that sediment traps were not deployed at site 1 in November. Results for site 11 are not shown in the above graph as explained in the text.

Particulate nutrient sedimentation rates measured for total nitrogen and total phosphorus yielded similar seasonal and spatial trends (Fig. 5.3). In general, the rate of deposition of particulate nitrogen and phosphorus was proportional to the rate of deposition of total particulate material at Canyon Lake. However, the rate of particulate N and P deposition in mid-depth and bottom traps deployed at deep sites (*e.g.*, sites 2 and 7) differed from late fall to summer. While bottom traps collected more nitrogen and phosphorus in November 2001 (after mixing), stratification led to similar or lower

trapping rates measured in bottom traps relative to mid-depth traps. Note that data for site 11 in May 2002 was not included in Figure 5.3 because of unusually high values (2617 mg/m²/day total N and 3620 mg/m²/day total P) resulting from resuspension processes.

The particulate N sedimentation rate was about 4x that of P for both deep and shallow sites. This general trend is consistent with core-flux results, where NH₄-N release from the sediments was on average 4.3x times higher than SRP release. Sediment trap results also indicate that mixing of the lake (in November) delivered a large quantity of nitrogen and phosphorus to the Type II and III sediments (> 150 mg/m²/day nitrogen and > 30 mg/m²/day phosphorus). In addition, sites that were at or above the thermocline (e.g., 4, 11, and 22) displayed greater particulate sedimentation rates for nitrogen and phosphorus during the summer (May and August, 2002) compared to deeper sites below the thermocline. Site 8, located near the culvert of East Bay also displayed high sedimentation rates in all three sampling periods. Higher productivity and/or boat activities in East Bay most probably transported some resuspended material into this site, resulting in higher N and P trapping rates compared to deeper sites in the main body.

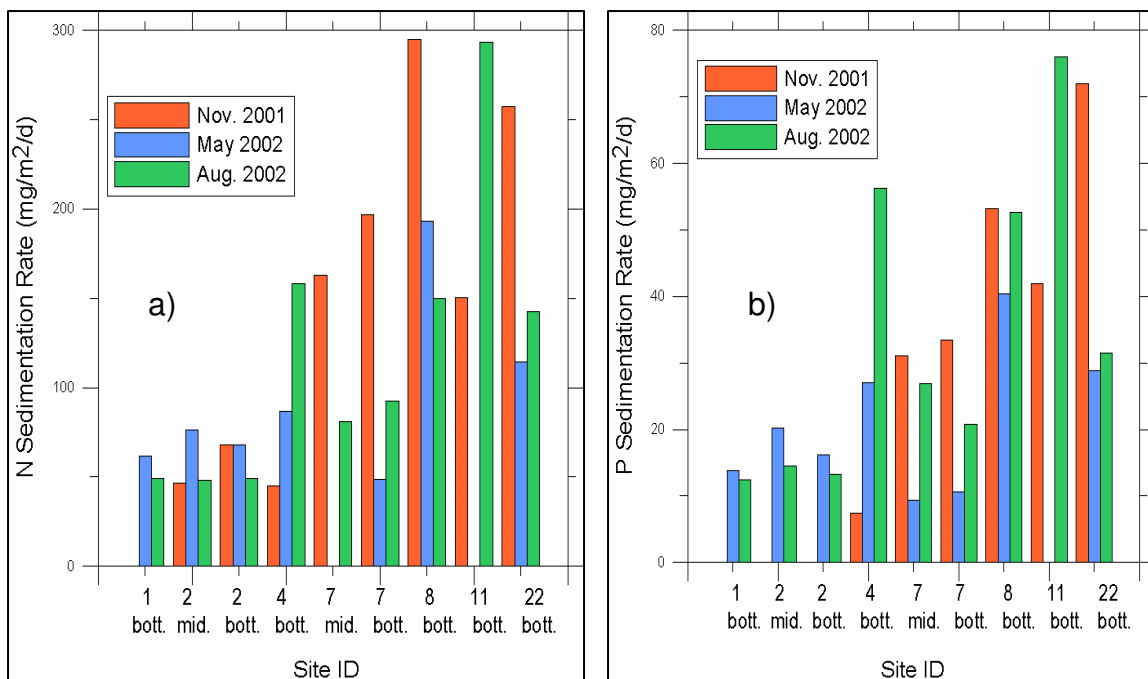


Figure 5.3. Sedimentation rates calculated for total nitrogen (a) and total phosphorus (b) using sediment traps. Data not presented in the above graphs was either due to a limited sample size or in the case of site 11, unusually high values (described in text).

6.0 NUTRIENT BUDGET

The nutrient flux, sedimentation, and other studies were used to develop an inventory of nitrogen and phosphorus inputs and outputs within Canyon Lake.

6.1 Methods

The nutrient budget was developed assuming no significant annual net change in N or P mass within the water column of the lake. Thus, inputs should be reasonably balanced with losses from the water column. Inputs were attributed to up to 5 different sources: i) atmospheric deposition, ii) runoff, iii) internal loading, iv) resuspension and, for N, v) N-fixation by blue-green algae. Nutrient losses included i) sedimentation, ii) outflow (specifically water delivered to the treatment plant for municipal use) and, for N, iii) denitrification. While evidence for resuspension was recorded in sediment traps deployed in East Bay in May 2002, direct measurements of the rate and extent of sediment resuspension were not possible. Additionally, local watershed inputs due to nuisance runoff and winter storms were not quantified, so these sources remain inputs of unknown magnitude to the lake. Finally, N-fixation by cyanobacteria is a potentially important source of N to the water column in this N-limited system, although direct measurements were also not possible. Thus, the sum of inputs due to sediment resuspension, local nuisance runoff and, for N, N-fixation, will be taken as the difference between measured inputs and losses.

External sources of nutrients to the lake included atmospheric deposition and runoff. Wet deposition of N and P to the lake surface was estimated from National Atmospheric Deposition Program (NADP) data for the region. Dry deposition is an additional source of nutrients to the lake, although sufficient controversy surrounds measurements of dry deposition that data is not provided as part of the NADP program. As a result, this source is not included in our nutrient budget calculations, although such inputs are thought to exceed wet deposition. Runoff to the lake includes runoff from the upper watershed, nuisance runoff, local watershed runoff following rain events, and water purchased and delivered through the San Jacinto River in April 2002. Since no runoff from the upper watershed was recorded for the 2001-2002 study period, runoff inputs were restricted to runoff from the local watershed immediately adjacent to the lake (both summer nuisance runoff and winter storm flows) and that associated with purchased

water delivered through the San Jacinto River in April 2002. As previously noted, local watershed inputs due to nuisance runoff and winter storms were not quantified. Estimates of internal loading and sedimentation were made as previously described (e.g., Sections 4.0 and 5.0).

6.2 Results

6.2.1 External loading

Direct wet deposition of N to East Bay and the main body of the lake was estimated based upon the 2001 NADP data for the region (NADP, 2002) that indicates that about 1.9 kg/ha/yr total inorganic nitrogen was deposited (as wet deposition). Using the wet deposition rate and areas of 0.79 and 0.42 km² for the main body and East Bay, respectively, one estimates that 201 and 107 kg N were deposited directly onto the surface of the main body and East Bay, respectively. Similar calculations indicate that approximately 144 and 77 kg P were deposited to the main body and East Bay, respectively.

Summer nuisance runoff was previously shown to be an important source of enterococcus and *E. coli* bacteria to East Bay and, to a somewhat lesser extent, the main body of the lake (Anderson *et al.*, 2002). Such runoff would also serve as a source of nutrients, although the magnitude of this input is not known; some hypothetical calculations suggest it may be fairly small relative to internal loading. The limited rainfall for the 2001-2002 study period eliminated natural runoff from upstream sources, but natural runoff from the local watershed following rain events would still be a potentially important source of nutrients to the lake. The importance of this source and the related summer nuisance runoff needs to be more carefully assessed.

The delivery of approximately 600 af of water to Canyon Lake that entered the reservoir north of the causeway in April of 2002 also represents an external input of nutrients. The added water would pick up nutrients, pathogens, and other contaminants along its pathway to the reservoir, therefore samples were collected and analyzed from north of the causeway closest to Canyon Lake to represent the potential nutrient inputs associated with this water delivery. The nutrient concentrations of inflow were measured to be 1.05 mg/L TN and 0.24 mg/L TP (Table 6.1). The total estimated loads were then estimated at 777 kg total N and 178 kg total P.

| Inflow | Total flow (af) | Total N (mg/L) | Total P (mg/L) | Total N (kg) | Total P (kg) |
|--------|-----------------|----------------|----------------|--------------|--------------|
| | 600 | 1.05 | 0.24 | 777 | 178 |

Although runoff from the upper watershed did not contribute nutrients to Canyon Lake during 2001-2002, it is nevertheless instructive to quantify the potential inputs during wet years. To quantify this potential source of nutrients to the lake, samples from the Perris Valley storm drain (denoted PVS) were taken and analyzed for total nitrogen and phosphorus contents (Figure 6.1). The Perris Valley storm drain, which receives urban runoff, delivers flow to the San Jacinto River at Sun City (denoted SJS), which then flows into the north portion of Canyon Lake. Flow from the gaging station was used in conjunction with the measured average nutrient concentrations to estimate a potential a nutrient load to the lake (Table 6.2).

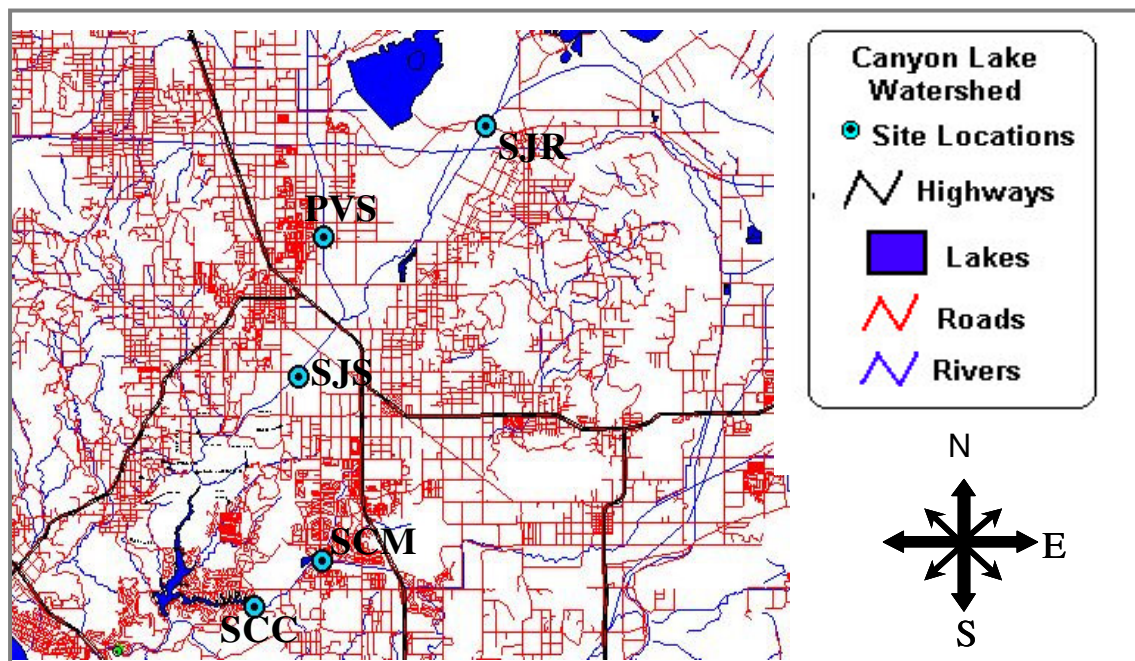


Figure 6.1. The Canyon Lake watershed. Major tributaries are denoted SJS (San Jacinto at Sun City), SCC (Salt Creek at Canyon Lake), SCM (Salt Creek at Murrieta), PVS (Perris Valley Storm drain), and SJR (San Jacinto River).

| Discharge | Total flow (af) | Total N (mg/L) | Total P (mg/L) | Total N (kg) | Total P (kg) |
|-----------|-----------------|----------------|----------------|--------------|--------------|
| PVS | 3662 | 1.48 | 0.39 | 6686 | 1762 |

Runoff from PVS was moderately high in nitrogen and phosphorus and estimated loads that could have been delivered to the lake from this single source were 6686 kg TN and 1762 kg TP, although much greater loads from this source and others within the upper watershed would be expected during periods of heavy rainfall.

Totaling the projected known external inputs, one estimates that about 506 kg N and 116 kg P were delivered to the main body of the lake by the purchased water flow, and an additional 201 kg N and 144 kg P were delivered by direct atmospheric deposition onto the lake surface. Analogously, a total of 286 kg N (179 kg from runoff and 107 kg from atmospheric deposition) and 136 kg P (59 kg from runoff and 77 kg from atmospheric deposition) was delivered to East Bay. Nevertheless, a number of potentially significant inputs were not included to due data limitations; thus the above loadings are thought to under-represent the total external loading of nutrients to Canyon Lake.

6.2.2 Internal loading

Internal loading of N and P varied among sediment type as well as between the main body and East Bay. The overall contributions of each sediment type to the internal nutrient loading in both parts of the lake depended on the area each sediment type covered. For example, the type I and III sediments comprised a small proportion of the sediment area in East Bay (14.4 and 14.8 acres, respectively), and thus contributed relatively little N (Table 6.3) and P (Table 6.4) to East Bay. The total internal N load (as $\text{NH}_4\text{-N}$) to the lake was estimated at 13,549 kg/year, with ~63% contributed by the main body and 37% by East Bay (Table 6.3). Some organic N was released from the sediments, although $\text{NH}_4\text{-N}$ generally comprised >80-90 % of the total N flux. Since total N was not routinely measured on core-flux samples, $\text{NH}_4\text{-N}$ flux was used for estimates of internal N loading.

Table 6.3. Annual internal nitrogen loading to Canyon Lake (2001-2002).

| Sediment | Flux ($\text{mg}/\text{m}^2/\text{d}$) | Main Body | | East Bay | | Total (kg) |
|--------------|---|-----------------|--------------|-----------------|--------------|---------------|
| | | Area (acres) | Mass (kg) | Area (acres) | Mass (kg) | |
| Type I | 22.7 | 47.8 | 1607 | 14.4 | 483 | 2090 |
| Type II | 34.8 | 64.8 | 3328 | 74.5 | 3836 | 7164 |
| Type III | 29.8 | 82.6 | 3643 | 14.8 | 652 | 4295 |
| <i>Total</i> | | <i>195.2</i> | <i>8578</i> | <i>103.7</i> | <i>4971</i> | <i>13549</i> |

The total phosphorus load to the lake was estimated at about 4625 kg/year, with the main body and East Bay contributing roughly 58 and 42%, respectively, to the total P load (Table 6.4). Acidification of core-flux and other samples below pH 2 promoted rapid hydrolysis of the organic phosphate esters, so that total dissolved P (TDP) and SRP concentrations were essentially identical.

Table 6.4. Annual internal phosphorus loading to Canyon Lake (2001-2002).

| Sediment | Flux (mg/m ² /d) | Main Body | | East Bay | | Total (kg) |
|--------------|-----------------------------|--------------|-------------|--------------|-------------|-------------|
| | | Area (acres) | Mass (kg) | Area (acres) | Mass (kg) | |
| Type I | 6.3 | 47.8 | 446 | 14.4 | 134 | 580 |
| Type II | 15.1 | 64.8 | 1444 | 74.5 | 1664 | 3108 |
| Type III | 6.5 | 82.6 | 795 | 14.8 | 142 | 937 |
| <i>Total</i> | | <i>195.2</i> | <i>2685</i> | <i>103.7</i> | <i>1940</i> | <i>4625</i> |

Areal-averaged nutrient release rates were higher in East Bay than in the main body of the lake (Table 6.5). Mean SRP and NH₄-N flux rates were 12.7 and 32.5 mg/m²/day for East Bay and 9.3 and 29.7 mg/m²/day for the main body, respectively.

Table 6.5. Mean SRP and NH₄-N flux rates in the main body and East Bay.

| | Mean SRP Flux (mg/m ² /day) | Mean NH ₄ -N Flux (mg/m ² /day) |
|-----------|--|---|
| Main Body | 9.3 | 29.7 |
| East Bay | 12.7 | 32.5 |

6.2.3 Other inputs

Other possible inputs of nutrients to the lake include subsurface discharge of storm drains and residential drains, groundwater flow and N-fixation. Numerous small drains were observed near the water surface that drained yards and landscaping. Collectively, such drains and groundwater flow may deliver an unknown amount of nutrients to the lake. Nitrogen-fixation is also a potentially important source of N to the this generally N-limited lake. Although not measured in this study, N-fixation rates have been reported to reach 0.2 – 9.2 g N/m²/dy in eutrophic lakes (Howarth *et al.*, 1988a,b). Thus it is possible that up to 3865 kg N may be added to East Bay and up to 7270 kg N may be delivered

to the main body by this biological process. Not enough is known about the algal ecology, however, to speculate further about the abundance of the N-fixing cyanobacteria and the importance of this process.

6.2.4 Particulate flux to sediments

Significant amounts of material were collected in sediment traps as noted in Section 5.2. Given the limited sampling frequency, seasonal trends in the rates of particulate N and P flux were not clear, so annual average flux rates were calculated for East Bay and the main body based upon the measured sedimentation rates within each basin. The average N and P sedimentation rates in East Bay were about 25% higher than those for the main body, consistent with the higher productivity and greater potential for mechanical resuspension of bottom sediments in East Bay (Table 6.6). Assuming total areas of 0.42 and 0.79 km² for East Bay and the main body, respectively, one calculates about 38,400 kg N/yr and 9,400 kg P/yr sedimented out of the water column in the main body, and 24,400 kg N/yr and 6,500 kg/yr P were removed from the water column in East Bay.

| Table 6.6. Average N and P sedimentation rates and total annual mass removed from water column by sedimentation process. | | | | |
|--|---|---|----------------------------|----------------------------|
| | Average N Sedimentation Rate (mg/m ² /d) | Average P Sedimentation Rate (mg/m ² /d) | Total N Deposition (kg/yr) | Total P Deposition (kg/yr) |
| Main Body | 133 | 34 | 38,443 | 9,406 |
| East Bay | 159 | 42 | 24,397 | 6,482 |

6.2.5 Other losses

Denitrification is a bacterial process whereby NO₃⁻ is reduced to N₂ and thus potentially removed from the water column. Although denitrification was not measured in this study, it is possible that denitrification is relatively unimportant in the N budget for Canyon Lake since denitrifying bacteria would be in direct competition with phytoplankton for NO₃-N in this N-limited system. Nevertheless, the importance of this process is not known.

The final process serving to remove nutrients from Canyon Lake is that due to outflow. Since Canyon Lake did not discharge water to Lake Elsinore during the study

period, discharge was limited to water taken by EVMWD for treatment and municipal use. The amount of water taken for municipal use is not known, but is assumed to be small since little runoff from the San Jacinto River recharged the lake during the study period.

6.2.6 Proposed N and P budgets.

Based upon the above considerations and caveats, nitrogen and phosphorus budgets were developed for the main body and East Bay of Canyon Lake (Figs. 6.2 and 6.3). Internal loading greatly exceeded known external inputs of N and P for the study period (2001-2002). Sedimentation of particulate material was found to remove substantially more N and P from the water column than known inputs, however (Figs. 6.2 and 6.3). As a result, additional sources need to be invoked to reasonably balance inputs and losses. As previously discussed, such sources include sediment resuspension, summer nuisance runoff and winter storm runoff from the local watershed, subsurface drains/groundwater inputs and, for N, N-fixation by blue-green algae. The potential for N-fixation across the lake is thought to be relatively high since the lake is typically N-limited; N-fixation would give blue-green algae a strong competitive advantage over other types of phytoplankton. Sediment resuspension due to mechanical disturbance by prop wash is known to be important for the shallow East Bay, although it is no doubt also important for other shallow embayments off of the main body of the lake. Prop wash may also enhance the release of dissolved nutrients, including both inorganic and organic forms, as well, since such turbulence would enhance the effective diffusive flux of N and P from the uppermost layer of sediments. Finally, summer nuisance runoff and winter storm runoff from the local watershed, as well as groundwater and subsurface drains could also deliver nutrients to the lake. In light of the uncertainty of the relative magnitude of inputs from these additional sources of nutrients to the lake, the budget residual was assigned as the sum of these processes (Figs. 6.2 and 6.3).

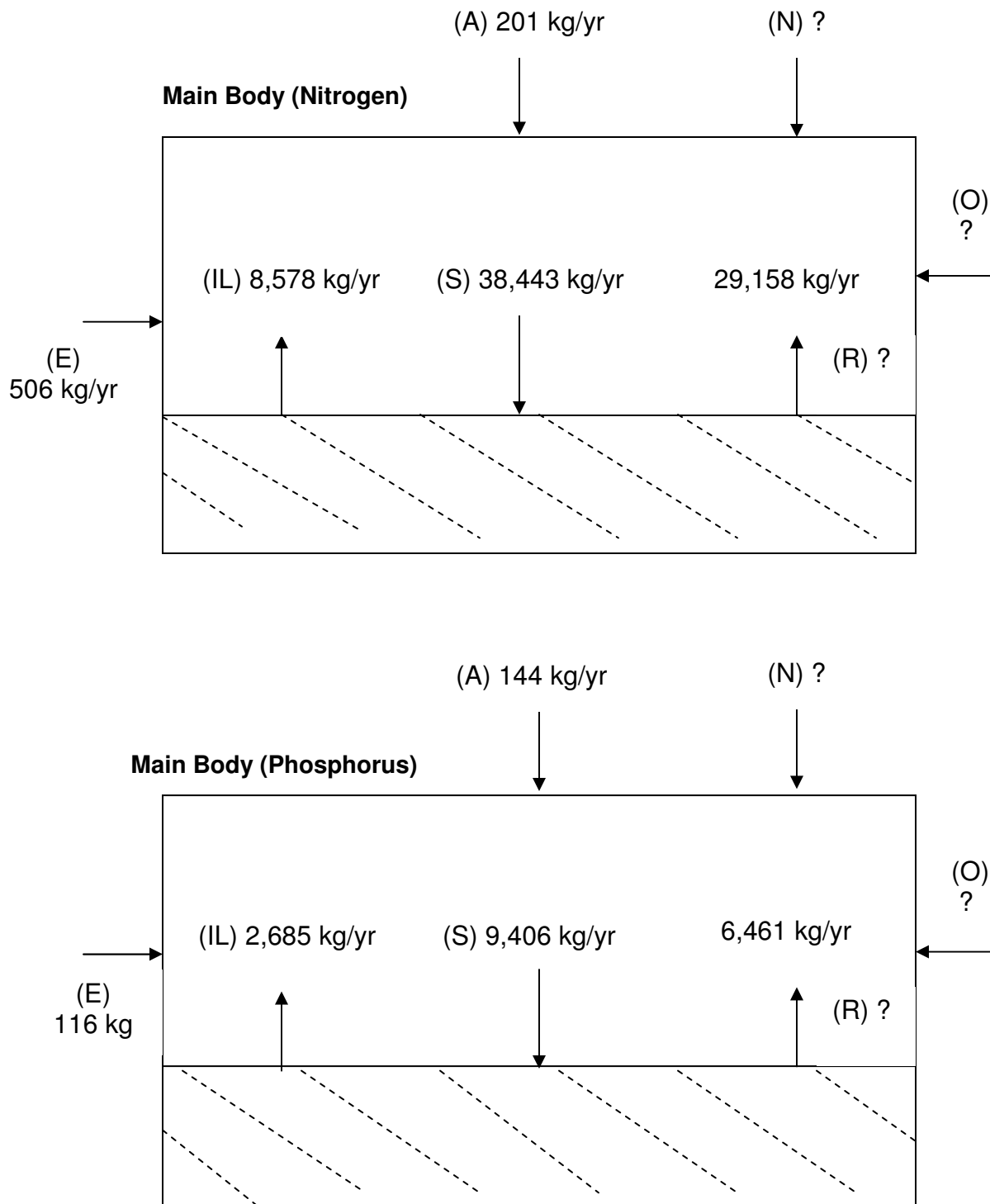


Figure 6.2. (a) Nitrogen and (b) phosphorus mass balance for the main body of Canyon Lake, showing external inputs (E), internal loading (IL), atmospheric deposition (A), particulate sedimentation losses (S), and the unquantified sediment resuspension (R), N-fixation (N) and nuisance runoff and other unquantified sources (O). The sum of $N+O+R$ is taken as 6,461 kg P/yr and 29,158 kg N/yr for P and N, respectively.

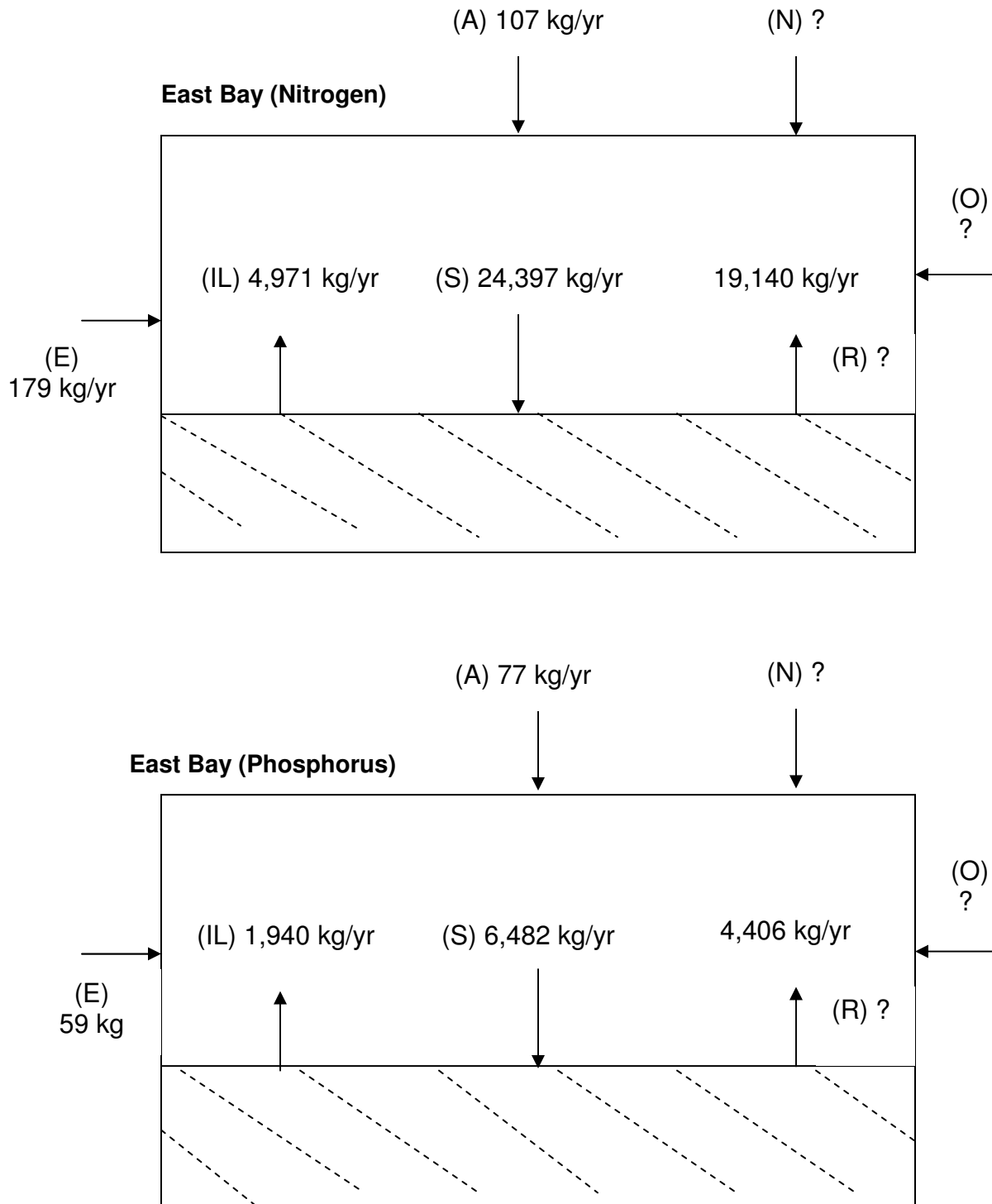


Figure 6.3. (a) Nitrogen and (b) phosphorus mass balance for the East Bay of Canyon Lake, showing external inputs (E), internal loading (IL), atmospheric deposition (A), particulate sedimentation losses (S), and the unquantified sediment resuspension (R), N-fixation (N), and nuisance runoff and other unquantified sources (O). The sum of $N+O+R$ is taken as 4,406 kg P/yr and 19,140 kg N/yr for P and N, respectively.

7.0 WATER QUALITY MODELING

The data obtained from our sampling at Canyon Lake was used with the BATHTUB water quality model to predict annual average water quality in the lake. With reasonable agreement between observed and predicted values, the model could also be used with some confidence in predicting water quality changes with reduced external and internal nutrient loads. Separate simulations were conducted using a 2-dimensional depth-averaged finite element hydrodynamic model to provide information about typical circulation patterns in the main body and East Bay of the lake.

7.1. Methods

BATHTUB is an empirical lake water quality model developed by Walker (1986) that was used in this phase of the project. This model was developed through an extensive assessment of 299 reservoirs operated by the US Army Corps of Engineers throughout the country. The model can be adapted to morphometrically complex reservoirs such as Canyon Lake, and its main use is in predicting annual average water quality based on a number of parameters, including external, atmospheric, and internal inputs, lake morphometry (mean depth, surface area, etc.), evaporation, precipitation, and other parameters. Canyon Lake was modeled as two hydraulically linked segments – the main body and East Bay – and water quality predictions were made for each segment. Input parameters used for both segments in the BATHTUB simulations are provided in Table 7.1.

The BATHTUB program includes a number of sub-models that have been used to predict chlorophyll a, Secchi depth, and TP and TN concentrations in lakes. Chlorophyll concentrations were predicted using the P, N, light, turbidity model, while Secchi depth was estimated using predicted chlorophyll and turbidity (Table 7.2). Previous experience has shown that model results are strongly dependent upon the N and P balance model used. Model simulations were conducted using 5 different N and P balance models that differ in the mathematical formalism used to represent nutrient sedimentation processes (Table 7.2). The default parameter values provided in the model were used in all simulations.

| Input Parameter | Main Body | East Bay |
|--|-----------|----------|
| Surface Area (km ²) | 0.79 | 0.42 |
| Mean Depth (m) | 7.6 | 2.8 |
| Mixed Layer Depth (m) | 4.7 | 2.8 |
| Precipitation (m) | 0.12 | 0.12 |
| Evaporation (m) | 1.4 | 1.4 |
| Non-Algal Turbidity (m ⁻¹) | 0.2 | 0.4 |
| Atmospheric N Loading (kg N) | 201 | 107 |
| Atmospheric P Loading (kg P) | 144 | 77 |
| External N Loading (kg N) | 506 | 179 |
| External P Loading (kg P) | 116 | 59 |
| Internal N Loading (kg N) | 8,135 | 5,462 |
| Internal P Loading (kg N) | 2,482 | 2,158 |
| Other N Loading (kg N) | 29,223 | 17,850 |
| Other P Loading (kg P) | 6,525 | 3,925 |

| Water Quality Parameter | Model |
|-------------------------|--|
| Chlorophyll a | P, N, light, turbidity |
| Secchi depth | Chlorophyll a and Turbidity |
| P Balance | 2 nd Order, Available 2 nd Order, Decay 2 nd Order, Fixed 1 st Order Canfield and Bachmann |
| N Balance | 2 nd Order, Available 2 nd Order, Decay 2 nd Order, Fixed 1 st Order Bachmann Vol. Load |

Hydrodynamic simulations of epilimnetic flow under summer-stratified conditions were conducted using a 2-D depth-averaged finite element model assuming a 5 m/s wind out of the west-southwest and a 250 node finite element mesh. Supporting field measurements of water velocities using acoustic Doppler current profiling (ADCP) were not conducted due to equipment problems.

7.2 Results

7.2.1 BATHTUB simulations results

BATHTUB-predicted TN and TP concentrations for Canyon Lake were strongly dependent upon the nutrient balance model used (Table 7.3 and 7.4), while predicted chlorophyll concentrations and Secchi depths varied less dramatically. For the main body of the lake, the 2nd order, fixed nutrient balance model most closely reproduced the observed average Secchi depth and chlorophyll and TN concentrations, but underpredicted by a rather large margin the measured TP levels in the main body of the lake (Table 7.3). The 2nd order, fixed model also did a better job of reproducing the overall water quality in East Bay than the other nutrient balance (*i.e.*, nutrient settling) models (Table 7.4). The 2nd order, fixed model was also found to most closely reproduce measured water quality in Lake Elsinore (Anderson, 2001).

| Variable | Observed Values | 2 nd Order, Available | 2 nd Order, Decay | 2 nd Order, Fixed | 1 st Order | Canfield, Bachman |
|---------------------------------|-------------------|----------------------------------|------------------------------|------------------------------|-----------------------|-------------------|
| Chlorophyll ($\mu\text{g/L}$) | 24.5 ^a | 37.6 | 40.4 | 30.8 | 51.3 | 35.9 |
| Secchi (m) | 1.3 ^b | 0.9 | 0.8 | 1.0 | 0.7 | 0.9 |
| TP (mg/L) | 0.51 ^b | 0.201 | 0.227 | 0.125 | 2.076 | 0.155 |
| TN (mg/L) | 1.86 ^b | 2.046 | 2.584 | 1.452 | 8.432 | 2.077 |
| | | | | | | |

^aRWQCB data, ^bUCR data

| Variable | Observed Values | 2 nd Order, Available | 2 nd Order, Decay | 2 nd Order, Fixed | 1 st Order | Canfield, Bachman |
|---------------------------------|--------------------|----------------------------------|------------------------------|------------------------------|-----------------------|-------------------|
| Chlorophyll ($\mu\text{g/L}$) | 76.5 ^a | 67.0 | 70.9 | 57.2 | 86.5 | 75.5 |
| Secchi (m) | 0.8 ^b | 0.5 | 0.5 | 0.5 | 0.4 | 0.4 |
| TP (mg/L) | 0.220 ^b | 0.377 | 0.426 | 0.235 | 6.060 | 0.536 |
| TN (mg/L) | 1.81 ^b | 3.654 | 4.613 | 2.594 | 23.136 | 6.287 |
| | | | | | | |

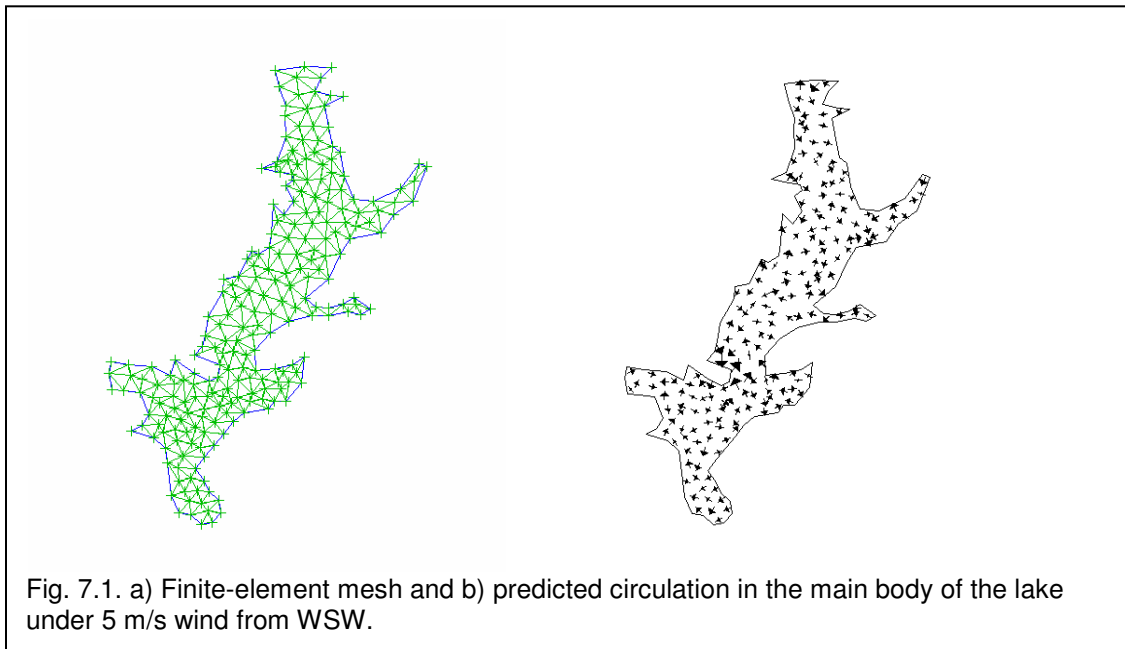
^aRWQCB data, ^bUCR data

Although a rigorous evaluation of the effects of different lake restoration efforts on predicted water quality in the lake was not conducted, preliminary simulations indicate

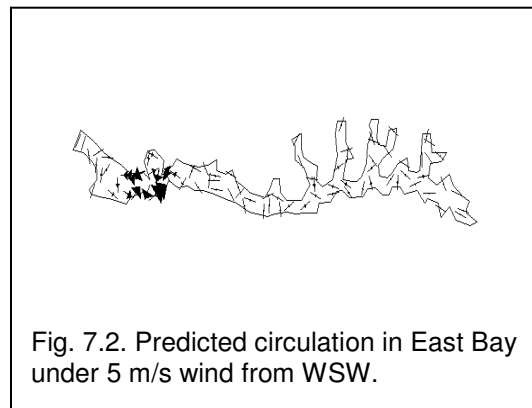
that significant reductions (>75%) of both N and P loading will be necessary to have a measurable impact on water quality in the lake.

7.2.2 Hydrodynamic simulation results

Simulations of flow under summer-stratified conditions in the epilimnion of the main body of the lake indicate a complex flow field with generally very low velocities ($\ll 1$ cm/s). No strong circulation pattern is evident, although a weak clockwise gyre in the southern part of the lake may set up. The relatively high degree of sheltering combined with short fetch limits the amount of wind-energy imparted to the water column (Fig. 7.1).



Weak currents were also predicted for the East Bay, although velocities were predicted to increase substantially (to ~ 1 cm/s) where the deeper water grades to relatively shallow water (2-3 m depth). Limits to the number of nodes used in the simulations precluded modeling of both the main body and East Bay as one unit, although the narrow culvert and low velocities found in the lake limit exchange between the two portions of the reservoir.



8.0 DISCUSSION

These studies evaluated the role sediments play in the supply and cycling of nutrients in Canyon Lake. Before discussing the contribution of sediments to water quality, it is helpful to comment on other factors that can influence nutrient cycling and surface water quality in the lake. Firstly, the lake is small, with a mean depth of 5.9 m and a surface area of about 300 acres. While small in size, it has a very large watershed (almost 500,000 acres), yielding a watershed:lake area ratio of approximately 1600. A large ratio (*e.g.*, 100) is often characteristic of eutrophic systems (Horne and Goldman, 1994). Likens and Bormann (1974) point out that the terrestrial and aquatic portions of the watershed are intimately linked by movement of materials from land to water. Although measured external inputs to the lake were relatively minor during this study period, inputs from the watershed during wet years are known to be substantial.

Moreover, the lake is situated in a warm, semi-arid region with high solar flux over most of the year. Straskraba (1980) developed a climate-productivity relationship in which he found that variables such as solar radiation and temperature can have a substantial effect on phytoplankton productivity in water bodies.

Surface Water Quality

Characteristics typical of relatively high productivity were observed at Canyon Lake. The lake has frequent algal blooms, a relatively low transparency, anoxic bottom waters in the main body during stratified periods, and an overall rather poor water quality. For example, transparency in the main body of the lake averaged 1.3 m and did not exceed 2 m. Average transparency in East Bay, which is much shallower (<3 m), was 0.8 m. Dissolved oxygen contents in the hypolimnion were as low as 0.1 mg/L in the main body and 0.3 mg/L in East Bay, although East Bay generally remained well-aerated. The lake also had high dissolved ammonium and phosphorus concentrations (mean $\text{NH}_4\text{-N}$ was 0.41 mg/L and mean SRP was 0.33 mg/L) although dissolved nitrate concentrations were generally low (mean $\text{NO}_3\text{-N}$ was 0.06 mg/L).

Algal growth at Canyon Lake was limited by nitrogen during most of the spring and summer (TN:TP ratio of ~2 – 4), but did not appear to be limited by either nitrogen or phosphorus in late summer (TN:TP ratio of 6 – 15). For comparison, studies conducted in eutrophic lakes in western Canada found minimum N:P ratios as low as 1 for one week (United Nations Environmental Programme). However, such short-lived spring

N:P ratios can trigger the onset of nitrogen fixing cyanobacteria blooms. Nitrogen fixation then restores N:P ratios to 15 and higher. In this way, a nitrogen deficit may trigger the appearance of nitrogen fixing species and change in-lake nutrient ratios. The observed increase in TN:TP ratios over the summer thus provides anecdotal evidence for N-fixation in Canyon Lake.

An interesting point is that Lake Elsinore was found to be limiting in phosphorus, with TN:TP ratios often greater than 20:1. Since Canyon Lake supplies water to Lake Elsinore during wet years, one might expect similar TN:TP ratios for the two water bodies. While low TN:TP ratios in Lake Elsinore have been reported following large water inputs from Canyon Lake, the infrequent discharge of water to Lake Elsinore, combined with more effective lake P-removal processes relative to N, results in a higher N:P ratio in Lake Elsinore much of the time. The differences may also be a function of phytoplankton ecology and resource competition. For instance, Tilman *et al.* (1982) demonstrated that diatoms outcompete other algae for phosphorus but are inferior competitors for nitrogen. Thus diatoms grow better under low phosphorus concentrations than other algal species. In the case of Lake Elsinore, diatoms appear to be one of the major algal groups in the lake during the cool winter-spring months. Another study by V.H. Smith (1983) reported that low TN:TP ratios favor blue-green algal dominance in natural lakes in temperate zones. N-fixing blue-green algae were rare in lakes where the TN:TP mass ratio was >29:1. Algal species have not been studied at Canyon Lake, so it is unclear whether species composition is significant in influencing nutrient availability.

Sediment Composition and Distribution

Understanding the bathymetry of the lake basin was important in understanding sediment distribution, nutrient release, and sedimentation rates at Canyon Lake. Therefore a bathymetric map was created and used in conjunction with data generated from sediment characterization to reveal that three distinct sediment types were present in the reservoir. The lake bottom was divided into 1) a littoral zone with coarse grained sediments (classified as Type I), 2) a transitional zone, which was deeper than the littoral zone and had finer sediments (classified as Type II), and 3) an accumulation zone, generally the middle of the lake where the finest sediments were 'focused' (classified as Type III). The mean sediment properties varied successively from Type I to Type III sediments, with greater concentrations of total C, N, and P found in the finer sediments. The Type II sediments were located mostly in East Bay and the north portion of the main

body, and served as the greatest source of SRP and $\text{NH}_4\text{-N}$ to the water column. The Type III sediments were concentrated in the deep portion of the main body and served as an intermediate source of SRP and $\text{NH}_4\text{-N}$ to the water column. The Type I sediments were mostly present in shallow embayments and in an area near the dam (Fig. 3.2). The reason for the coarse-textured sediment in the deep water near the dam is not clear, although water withdrawal by the EVMWD may have scoured fines out of the surficial sediment. Additional sediment sampling would help define the sediment properties here.

In addition to the identification of three sediment types present in the lake basin, East Bay and the main body could be considered two separate lake basins with rather limited exchange under most conditions. Figure 3.1 showed that East Bay was much shallower than the main body, and surface water quality data suggested higher temperatures, higher DO contents, and lower transparency present in East Bay. Sediment characterization showed no key differences in sediment composition between the two parts of the lake, however. In general, organic and total reserves of carbon, nitrogen and phosphorus did not differ significantly between East Bay and the main body. Nevertheless, differences in rates of SRP and $\text{NH}_4\text{-N}$ release did exist, where release of SRP and $\text{NH}_4\text{-N}$ (averaged over the year) was much greater at East Bay when compared with the main body of the lake. This situation is common in shallow, nutrient enriched lakes in which warm water temperatures, sediment resuspension and other factors result in high rates of internal loading. Studies on the effects of sediment resuspension caused by recreational activity (*e.g.*, boat propellers) appear to be site-specific, however. Aspund *et al.* (1995) surveyed 10 Wisconsin lakes to assess whether water quality changed during periods of heavy boat use. He found that for all lakes there were significant increases in motorboat activities on weekends that resulted in a corresponding decrease in water clarity and an increase in turbidity. There was no significant change in chlorophyll concentrations, however. Moreover, he found that the changes in water quality that occurred over one weekend were much smaller than the seasonal changes that occurred over the summer (Asplund *et al.*, 1995). Overall, his findings suggested that motor boats may have a temporary impact on water clarity through sediment resuspension during times of heavy boat use, but that long term effects on nutrient regeneration and algal stimulation were not important for his lake systems or were minimal relative to other factors. In contrast, a study performed by Nedohin and Elefsiniotis (1997) indicated that motorboat activity created enough disturbance on the bottom sediment to enhance the release of phosphorus into the

overlying water. In the case of Canyon Lake, resuspension of bottom sediments most likely increased both turbidity and net internal loading of nitrogen and phosphorus, although the extent of resuspension could not be resolved from N-fixation and other inputs of nutrients.

Nutrient Release and Internal Loading

Internal loading, found to be an important source of SRP and $\text{NH}_4\text{-N}$ to the water column of Canyon Lake, was greatest for the Type II and Type III sediments. Holdren and Armstrong (1980) investigated four Wisconsin lakes to determine the effects of various environmental parameters on phosphorus release. They found: 1) high water temperatures led to increased release rates of P, while lower temperatures removed P from the water column, 2) temperature effects were most pronounced in the calcareous sediments, while redox played a greater role in P release in non-calcareous sediments, 3) P release increased with greater mixing intensity, and 4) bioturbation was found to have the greatest effect on P release rates. The above findings appeared to hold for Canyon Lake. Firstly, during the summer when the lake was stratified, SRP release rates were strongly influenced by temperature and the vertical thermal structure of the lake. Deeper sites with colder water temperatures and anoxic hypolimnia released significantly less SRP than their warmer, shallower counterparts. Secondly, it was found that during the winter when the lake was completely mixed, SRP was released at higher rates in the deeper sites. Recirculation of nutrients and mineralization of deposited particulate matter were most likely responsible for the significant increase in SRP release after mixing. Also, shallow sites (*e.g.*, like those found in East Bay) that were warm, had high DO, and showed high resuspension almost always displayed high SRP flux rates. Effects of bioturbation were not investigated in our studies. Overall, it appears that warmer temperatures and mixing promotes greater phosphorus release from the sediments of Canyon Lake.

$\text{NH}_4\text{-N}$ released from the sediments represented a moderate input of nitrogen to Canyon Lake, and the contribution of Type II sediments was higher ($\sim 35 \text{ mg/m}^2/\text{d}$) than Types I and II. $\text{NH}_4\text{-N}$ (mainly a product of organic nitrogen hydrolysis/remineralization) released from the sediments served chiefly to stimulate algal growth rather than to produce nitrate (since mean $\text{NO}_3\text{-N}$ concentrations were only 0.06 mg/L in the water column). The $\text{NH}_4\text{-N}$ used by algae was then incorporated into particulate form and deposited back into the sediments. Measured $\text{NO}_3\text{-N}$ flux rates from the sediments were

usually negative, so the sediments generally took up that nitrate present in the water column. Core-flux measurements, then, provide evidence for denitrification, although its relative importance for N-cycling in the lake is not clear.

In comparing core-flux results from Canyon Lake and Lake Elsinore, a large difference was seen in rates of $\text{NH}_4\text{-N}$ release, while SRP release rates were generally similar. Summertime $\text{NH}_4\text{-N}$ flux rates at Elsinore were $>100 \text{ mg/m}^2/\text{d}$ in both Type II and III sediments compared to $30\text{-}40 \text{ mg/m}^2/\text{d}$ at Canyon Lake, although flux rates of $50\text{-}60 \text{ mg/m}^2/\text{d}$ were seen in East Bay during summer. SRP flux rates during the summer were $10\text{-}12 \text{ mg/m}^2/\text{d}$ for Elsinore and somewhat higher for Canyon Lake, although generally similar. These differences may be due to the difference in the N:P ratio of material delivered to the sediments (Table 5.1). Material captured in sediment traps bore TN:TP ratios of approximately 4:1, comparable to the average N:P flux ratios measured in core-flux experiments (Tables 4.2, 4.3) and peeper studies (Table 4.4). The N:P ratios of sediment trap material in Lake Elsinore were closer to 6:1, consistent with the higher flux of N relative to P. Thus, the observed N:P flux ratio appears to be directly related to the composition of material delivered to the sediments.

While core-flux results quantified nutrient release and provided the basis for loading estimates, peepers provided nutrient flux estimates and, more importantly, characterized porewater chemistry. Peepers revealed that the sediment surface was a zone of intense microbial activity especially in the summer, as sulfate and nitrate reduction followed oxygen depletion. Evidence for sulfate reduction was seen in the increase in sulfide concentration near the sediment surface, with an associated drop in pH. Porewater profiles at the deep site (site 2) showed an overall increase in concentrations of Fe, Mn, and SRP. Alkalinity and $\text{NH}_4\text{-N}$ levels also increased with depth in the sediments. These results indicated that mineral dissolution reactions and desorption processes were most likely responsible for the elevated concentrations of metals and nutrients found in the porewaters.

An important finding at Lake Elsinore was that the release of SRP was strongly dependent on the biogeochemistry of the sediments. Anderson (2001) found that high sulfide concentrations (summer levels close to 50 mg/L) in porewaters resulted in FeS_2 formation, thus allowing greater release of SRP that would otherwise be bound to Fe (III). The lower porewater sulfide concentrations at Canyon Lake (maximum of 12 mg/L) yielded higher Fe^{2+} concentrations, although Fe(III) control of SRP release appears limited for much of the lake, since core-flux results demonstrated that higher SRP

release rates were found at sites with warmer water temperatures and high DO levels. These conditions are optimal for microbial growth and thus enhance organic matter mineralization.

Sedimentation and Nutrient Deposition

Sedimentation served as the main loss process for nutrients at Canyon Lake. Sediment trap results revealed that a significant amount of particulate nitrogen and phosphorus were delivered to the sediments, with rates of sedimentation exceeding known inputs of N and P by a wide margin. These high rates of nutrient deposition relative to known nutrient inputs suggests that resuspension and other unquantified input processes (*e.g.*, N-fixation, unquantified surface and subsurface flows) are important parts of the nutrient budget in Canyon Lake.

Nevertheless, some possible limitations in the trap flux estimates may also help account for the high observed nitrogen and phosphorus sedimentation rates observed relative to known inputs. Firstly, studies have shown that certain trap designs and hydrodynamic flows around a trap may cause particles to concentrate and thus over-trap (Gardner, 1980). However, the average efficiency of cylindrical traps we used in this study was found to be closer to 100% compared to other trap designs, and, as shown in other studies, most overtrapping caused by hydrodynamic effects around the traps occurs in high current environments, which do not exist at Canyon Lake. Therefore it is reasonable to rule out the effects of trap designs. Secondly, many researchers have evidenced that traps deployed close to lake bottoms are collecting primary and secondary flux, so that settled particles are resuspended several times before final burial in bottom sediments (Hans-Peter Kozerski, 1994). Strong evidence of mechanically-induced resuspension at East Bay existed and is most likely the main cause of high N and P trapping rates measured throughout the year. However, evidence of resuspension in the main body was not as strong. Resuspension is commonly identified when bottom traps collect more material than mid-depth traps. Mid-depth and bottom traps deployed at two deep sites in the main body however, revealed minimal differences in rates of nutrient deposition (Fig. 5.3); in some cases lower rates of nutrient sedimentation were seen in bottom traps. It appears that resuspended material from relatively shallow sediments near the margins and shallow embayments of the main body of the lake was laterally transported and focused into the deeper waters.

Nutrient Budget

Internal loading was the principal known source of nutrients for the 2001-2002 study period, contributing an estimated 8,578 kg of N and 2,685 kg P to the main body of the lake. Atmospheric deposition and inputs associated with the delivery of purchased water upper part of the lake contributed <10% of the known N and P inputs to the lake. The imbalance in known N and P inputs and losses indicate that unquantified inputs due to sediment resuspension, N-fixation and other sources were an important part of the nutrient budget in the lake. For example, local nuisance runoff was found to be a significant source of indicator bacteria to East Bay (Anderson *et al.*, 2002). Although nutrients were not measured in this nuisance runoff, previous researchers have found such runoff to contain high levels of nutrients, metals and other contaminants.

Water Quality Modeling

A second-order, fixed nutrient balance model, combined with a chlorophyll model using N, P, turbidity and light as model parameters, was found to best reproduce measured water quality in Canyon Lake. The overall agreement between observed and predicted values was somewhat better for the main body compared to East Bay, although the model underpredicted by a fairly large margin the TP in the main body of the lake. The model did a better job with N, predicting total N concentrations within about 25% of the average observed values in the main body (Table 7.3), while very good agreement was found between observed and predicted TP in East Bay (Table 7.4).

A preliminary assessment was conducted of the predicted impacts of internal load reductions on water quality in the lake. Reductions in loading by 50% or less did not yield a significant improvement in water quality. In fact, reductions of at least 75% in the total nutrient inputs (internal loading, resuspension, etc.) were predicted to be necessary to measurably improve water quality in the lake. Thus, the model suggests that substantial nutrient control measures will need to be implemented to have a significant impact on water quality in the lake.

Implications for Lake Management

The findings of the study have some implications for management efforts aimed at improving water quality in the lake. First of all, internal recycling, resuspension and other unquantified inputs are important sources of nutrients to the lake under conditions of

limited watershed inputs. Resuspension due to motorboat activity was particularly important for East Bay and apparently also in the shallow embayments on the main body of the lake. Hydrodynamic simulations indicate low potential for wind-induced resuspension in the lake due to the limited fetch and high degree of sheltering. Local runoff may also be an important part of the nutrient budget under dry conditions, although its contribution was not quantified in this study. Assessment is thus needed of the importance of nuisance runoff and other inputs of nutrients to the lake during dry-weather conditions. Nutrient input during high runoff events in the upper watershed also need further characterization, since very limited runoff to Canyon Lake took place over the 2001-2002 study period.

Temperature and sediment texture appeared to have greater effects on internal recycling of nutrients from sediments than redox state, since the highest rates of release were found in relatively shallow, warm, well-aerated fine sediments. This observation has implications for proposed destratification of the main body of the lake. Specifically, the findings raise the possibility that destratification of the main body of the lake may have little influence or even degrade overall water quality; destratification will increase the temperature and, thus, most likely also increase the rate of internal recycling of the deep water (Type III) sediments. The effectiveness of such an effort for controlling algae appears to depend upon limiting the residence time phytoplankton spend in the upper, well-lit portion of the water column, although such a system will beneficially reduce the concentrations of $\text{NH}_4\text{-N}$, H_2S and other reduced species and increase DO in the lower portion of the water column.

9.0 CONCLUSIONS

Water quality in Canyon Lake varied rather strongly between the main body and East Bay, with higher chlorophyll and lower transparencies found in East Bay relative to the main body of the lake. The warm, shallow conditions in East Bay, combined with high levels of internal recycling, sediment resuspension by boat action, and potential local runoff inputs of nutrients maintained poor water quality throughout the year. The main body of the lake was stratified during much of the year, with lower rates of internal recycling found in the deeper, cooler hypolimnetic water relative to shallower, warmer waters.

Given the limited watershed runoff of 2001-2002, internal nutrient recycling was the dominant quantified source of nutrients to the lake over the study period, although the contributions from resuspension, N-fixation, nuisance runoff and other local sources of nutrients were not directly quantified in this study.

Water quality in the lake was reproduced with fair success using the nutrient budget data and the BATHTUB surface water quality model. Preliminary simulations suggest that substantial reductions (>75%) in both N and P loading are required to measurably improve water quality in the lake. Even greater reductions will be needed during periods of substantial nutrient inputs during high runoff events.

The study findings also have implications for in-lake efforts at improving water quality. For example, findings raise the possibility that destratification of the main body of the lake, as an in-lake nutrient control measure, may minimally influence or even degrade overall water quality; destratification will increase the temperature and rate of particulate delivery of the deep water (Type III) sediments during the summer. While delivery of DO to the deeper waters of the lake will lower dissolved concentrations of Fe^{2+} , Mn^{2+} , H_2S and other reduced chemical species, core-flux measurements indicate that SRP and $\text{NH}_4\text{-N}$ recycling in the lake is not strongly affected by DO status. Rather, rates of internal loading in the lake appear to be more influenced by temperature and the delivery of fresh organic matter to the sediments. The effectiveness of such a destratification system, then, appears to depend upon its ability to limit the residence time of phytoplankton in the upper, well-lit portion of the water column.

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