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Columbia Environmental Research Center

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Final Report To

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Great Plains Region
U.S. Bureau of Reclamation
Billings, MT

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EXECUTIVE SUMMARY

A study was conducted to determine the sources, fate, and effects of nutrients and bacteria in Fort Cobb Reservoir, OK. A total of 26 sampling sites were studied on a bimonthly basis from June 2000 to June 2002. Results indicated that Fort Cobb Reservoir is highly eutrophic based on Carlson Trophic State Indices (TSI) of total phosphorus (TSI = 67), algal biomass (TSI = 61), and water clarity (TSI 67). This trophic classification indicates that the reservoir contains excessive concentrations of nutrients that are resulting in high concentrations of algal biomass; furthermore, water clarity of the reservoir is low. Collectively, these observations indicate that water quality is reduced, with emerging problems associated with taste, odor, and recreational values. Concern over these water quality conditions were frequently expressed by recreational users during the course of the study.

The reservoir is well mixed, but shows some signs of late summer stratification based on dissolved oxygen and pH depth profiles; however, there is no evidence of a thermocline. Dissolved oxygen and pH stratification occurs during periods of maximum temperature and low mixing. Lack of mixing, paired with light limitation, leads to a decrease in primary productivity and resultant oxygen depletion and CO_2 increase due to respiratory demands at greater depths. Continued water quality degradation could ultimately lead to greater dissolved oxygen depletions and impacts on fish and wildlife.

Water quality is poorest in the upper end of the reservoir near the tributary inflows but improves somewhat towards the dam. Highest algal biomass occurs in the upper reservoir and embayments. The lake is dominated by cyanobacteria (blue-green algae), which comprise over 90% of the phytoplankton numbers. Primary algal genera in order of occurrence include *Microsystis*, *Wollea*, *Anabaena*, *Oscilliatoria, Merismopedia*, *Anabaenopsis*, and *Aphanizomenon spp*. The algal toxin microcystin is highest during mid to late summer and reaches approximately 15% of the World Health Organization's concentration of concern of 1 µg/L.

The Cobb Creek, Lake Creek, and Willow Creek sub-watersheds comprise 62%, 26%, and 11% of total acreage of the Fort Cobb Watershed, respectively. All tributaries contribute elevated nutrients and bacteria to the reservoir. However, Cobb Creek contributes proportionally

more discharge (and hence total nutrient loading) to the reservoir compared to Lake Creek or Willow Creek.

Total coliform and *Escherichia coli* bacteria are elevated in all tributaries. *E. coli* concentrations are highest in the tributaries and upper end of the reservoir after periods of high runoff in early spring. In general, *E. coli* numbers are rapidly attenuated down-reservoir towards the dam. One exception to this, however, is the December to February period when over-wintering populations of waterfowl contribute high concentrations of *E. coli* and ammonia to the mid-lake and near-dam areas.

Currently, there is a 319 Demonstration Project in the Lake Creek Watershed that is implementing educational programs and land-use changes to improve water quality in the reservoir. However, it is unclear how successful this will be in reversing water quality declines for two reasons: 1) Lake Creek comprises only 26% of the total watershed, and 2) there is a large internal load of sediment-associated nutrients that are available to the water column via wind action, wave action, and bioturbation. Quarterly monitoring of a suite of water quality variables (depth-integrated measures of chlorophyll *a*, total nitrogen, total phosphorus, Secchi depth, and *E. coli*; and depth profiles of temperature, dissolved oxygen, pH, and turbidity) at a minimum of three sites (Sites 0, 3, and 6) is recommended to monitor the trajectory of water quality conditions in Fort Cobb Reservoir and to determine the success of management efforts to reduce non-point source pollution inputs and impacts.

INTRODUCTION

Fort Cobb Reservoir is a 4,100 acre reservoir located in a 314 square mile watershed in Caddo County, southwestern Oklahoma (Oklahoma Water Resources Board 1990) (Figure 1). The reservoir is managed by the U.S. Bureau of Reclamation (USBOR) for drinking water supplies, irrigation sources, flood control, recreation, and fish and wildlife habitat.

Fort Cobb Reservoir lies within a watershed dominated by sandy loam soils. Land use in the watershed is approximately 87% cropland, 9% rangeland, 2% water, 2% forest, and 1% urban (Table 1; Figure 2; Paul Yue, OKDEQ, personal communication). Therefore, agriculture is the primary land use. Primary row crops include peanuts, wheat, and cotton. Livestock operations are dominated by pasture grazing of cattle and several large confined animal feeding

operations (CAFOs) used for hog production. Martin (2002) cited U.S. Department of Agriculture statistics and indicated that Caddo County contains approximately 130,000 head of cattle and 12,000 hogs; however, these were county-based statistics that do not necessarily relate directly to numbers of livestock in the Fort Cobb Watershed.

Fort Cobb Reservoir has been listed by the State of Oklahoma as impaired based on the 305(b) Report to Congress (OKDEQ 2000). The reservoir has been listed as impaired due to excessive inputs of nutrients, sedimentation, and pesticides from row cropping and livestock production. Confined animal feeding operations are of particular concern, because they produce huge amounts of animal waste products containing significant amounts of nitrogen and phosphorus. These confinement facilities have little capacity for primary or secondary waste treatment and primarily store wastes in on-site retention lagoons. Excess waste from retention lagoons is frequently applied to soils over a relatively limited spatial area. Associated nutrients and bacteria can enter local streams via surface runoff, or can directly percolate into groundwater via the sandy soil matrix. Ultimately, these nutrients are transported to Fort Cobb Reservoir. Thus, the quality of reservoir waters for future human consumption is a major concern. Likewise, the U.S. Fish and Wildlife Service (USFWS) is concerned about impacts of animal waste on migratory birds in the area.

This report presents the results of a 2-yr study of the sources, fate, and effects of nutrients and bacteria in the Fort Cobb Reservoir and its watershed. The U.S. Bureau of Reclamation (USBOR) requested this study to determine the current biological and chemical conditions of the reservoir to determine existing and future threats to the water quality of the system.

OBJECTIVES

There were four objectives for this study:

1) Determine sources of nutrients and other contaminants that may be entering the reservoir;
2) Determine the current quality of water resources within Fort Cobb Reservoir;
3) Determine the health concerns related to coliform bacteria and algal cyanotoxins; and
4) Provide these data to the U.S. Bureau of Reclamation and other Federal and State agencies for help in development of watershed protection plans for Fort Cobb Reservoir.

MATERIALS AND METHODS

Bi-monthly reservoir assessments were conducted over a 2-yr period from 26 sites (16 in-lake and 10 stream sites) (Figure 3). Sample sites were located to evaluate a longitudinal water quality gradient extending from the upper end of the reservoir to the dam. Sites were also located to determine the sources of inputs among the three major tributaries and sub-basins: Cobb Creek (sub-basins 1-6); Lake Creek (sub-basins 7 and 8); and Willow Creek (sub-basin 10) (Figures 2 and 4; Table 1). Site numbers, locations, GPS coordinates, and notes are provided in Table 2. A list of measurement parameters and procedures is presented in Table 3.

Water quality monitoring and analysis

Depth profiles of dissolved oxygen, temperature, pH, and conductivity were determined using a YSI logging unit to determine seasonal stratification patterns. Water samples for laboratory analysis were collected as a surface composite using a 4-m long depth-integrating hose sampler and analyzed for nutrients (TN, TP, NO_2-NO_3, NH_3, SRP) and physical chemistry (pH, alkalinity, hardness, turbidity, conductivity, and total dissolved solids). Algal biomass was estimated using in-vivo fluorescence (both chlorophyll *a* and cyanin) and extracted chlorophyll *a* (in-vitro assessment of biomass) using a Turner Designs Fluorometer. In addition, particulate organic carbon (POC) was measured using a Coulometrics Model 5020 Carbon Analyzer (UIC, Inc.; Joliet, IL).

Discharge was measured in each tributary based on incremental-width methodology. A Swoffler-type current meter was lowered at equally spaced intervals across the stream to determine current velocity at multiple locations. Depth at each location was also measured. Discharge was calculated as the sum of the width x depth x velocity increments. Accuracy of this method was determined by comparison to the U.S. Geological Survey (USGS) gaging site located above Fort Cobb Reservoir (Cobb Creek near Eakly, OK; Station 07325800). The 2-yr discharge records are presented in Figure 5.

Phytoplankton community analysis

Phytoplankton samples for algal taxonomy were collected as a sub-sample from the water quality sample described above. A 30-mL sample of unfiltered water was poured into a glass

vial and preserved using 1 mL of Lugol's solution. Samples were counted using a 0.1 mL counting chamber (divided into 100 fields) under 100x magnification. A minimum of 10 fields were counted depending on density of the sample. Algae were identified to genus or species when possible. Data were used to calculate total numbers, species richness, and Simpson's Dominance Index.

Cyanotoxin analysis

The plankton community of eutrophic reservoirs is frequently dominated by cyanobacteria. Cyanobacteria are considered a threat to water quality for several reasons including taste and odor problems. In addition, cyanobacteria can produce numerous chemical substances such as microcystin that can be toxic to mammals and birds. Water samples for analysis of cyanotoxins were collected using a composite of two 4-m plankton tows (63-µm pore net) which collected a total of 90 L of water. Samples were rinsed into a 60-mL brown polypropylene bottle, placed on ice, and shipped overnight to the Columbia Environmental Research Center (CERC), U.S. Geological Survey, Columbia, MO. On receipt, the samples were freeze-dried and stored at -20°C prior to analysis.

All samples were screened for microcystin using the Enzyme-Linked Immunosorbent Assay (ELISA). If the ELISA test indicated the presence of cyanotoxins, the sample was subsequently analyzed by high pressure liquid chromatography (HPLC) in order to determine structural analogs and to better interpret the potential for toxicity (Park and others 1993). The molecular structure of microcystin is presented in Figure 6.

The HPLC methods were based on a modification of the methods derived from Harada and others (1988). Freeze-dried algal samples were weighed (10 to100 mg) using an analytical balance. Using HPLC grade water and solvents, samples were extracted three times with 5 mL of 50% methanol/50% water for 5 minutes by ultrasonication. Each extraction was decanted into a 50-mL centrifuge tube and the combined solvent/algal extracts were centrifuged at 10,000 rpm for 15 minutes. After centrifugation, the supernatant was decanted and filtered using either a Whatman 0.45-µm syringe filter or a Whatman Uniprep™ 0.45-µm GMF filter (5 mL volume). Methanol was removed from the extracts by rotoevaporation. The sample extract was transferred to a calibrated 10-mL culture tube and brought to volume with distilled, deionized water. The extract was diluted by transferring 100 µl to a calibrated 10-mL culture tube and filling to 10 mL

with distilled, deionized water. This portion was analyzed by the ELISA method. The remaining 9.9 mL were destined for further cleanup by solid phase extraction (SPE) cartridges. SPE cartridges (Bakerbond ODS SPE cartridges; 0.2 g, 3 mL) were preconditioned with 10 mL of methanol and 10 mL of water. After applying the extracts, the cartridges were washed with 10 mL of water and 10 mL of 9:1 water-methanol. The fraction containing microcystin (MC) toxin was eluted from the column with 20 mL of methanol and collected in a 125-mL round flask. The methanol fractions containing the MC toxins were concentrated by rotoevaporation to ~1 mL methanol, transferred and diluted to 2 mL with MeOH before HPLC analysis. Extracts were kept in a refrigerator or freezer at less than 4 °C.

The ten-fold dilutions of the algal extracts were analyzed using the EnviroGard™ ELISA kit from Strategic Diagnostics Inc. If a sample's concentration fell outside the upper bound of the calibration range (0.1 to 1.5 µg/L of MC-LR), it was re-diluted and reanalyzed.

HPLC analyses for four microcystin analogs (MC-LR, -RR, -YR, -LA, see Figure 6) were performed using a Dionex Summit HPLC system consisting of a quaternary low-pressure pump, autosampler, and PDA or an Agilent Series 1100 HPLC system with a binary high pressure pump, autosampler, and variable wavelength detector. The HPLC column used was a 250-mm x 4.6-mm Vydac (201SP54) column (5-µm particle size) with a C18 guard column. The solvent program had a total run time of 85 minutes with a flow rate of 1 mL/min. The HPLC grade solvents were filtered using 0.45-µm nylon filters and then sparged with helium. A phosphate buffer was used (pH 2.6; 25 mM). Solvent A was 90% methanol:10% phosphate buffer; solvent B was 25% methanol:75% phosphate buffer. Microcystins were quantified by UV absorbance at 238 nm, with confirmatory spectral analysis (190-600 nm).

HPLC data were collected and archived using the Chromeleon 6.2 software or collected with TotalChrom 6.2. The data were transferred to Turbochrom or TotalChrom chromatography software, and then four concentrations of a five-component MC standard were used to calibrate the method. The MCs in the standard were MC-LR, MC-YR, MC-RR, MC-LA and nodularin (Calbiochem, La Jolla, CA) at the following concentrations: 0.06, 0.2, 0.8, and 8 µg/mL. The MCs were quantified using 4 µg of angiotensin I (a small peptide) spiked into 200 µL of sample as the internal standard.

Procedure blanks and matrix blanks (*Spirulena platensis*) were processed with each batch of samples. A recovery spike was processed through the SPE cleanup step and the percent

recoveries of RR, LR, YR and LA were 114, 123, 70, and 117%, respectively. Detection limits for the microcystins were calculated based upon responses of the procedure blanks and the lowest calibration standards using three times the signal-to-noise ratio. Detection limits were determined to be 0.1 ng/mg by HPLC and 0.1 μg/L for the ELISA.

Bacterial analysis

Water samples were collected for bacterial analysis using surface grab samples. Samples were collected in sterile polycarbonate bottles, iced, and immediately shipped to the CERC for measurement of total coliform and *E. coli* bacteria using the Enzyme Substrate Coliform Test (APHA 1998, Edberg and others 1990). This assay uses duo-substrates with specific chromogenic and fluorogenic markers for the simultaneous detection of total coliform bacteria and *Escherichia coli* enzymes (Colilert®, IDEXX Laboratories 2002). This test defines the total coliform group as all bacteria which possess the enzyme b-D-galactosidase that cleaves the chromogenic substrate; *Escherichia coli* (fecal coliform) are defined as bacteria which give a positive total coliform response and that possess the enzyme b-glucuronidase that cleaves the fluorogenic substrate. Specific protocols were used to provide qualitative and quantitative analysis. This enzyme test used the multi-well format (Colilert® Protocol, IDEXX Laboratories 2002) with sterilized disposable packets (Quant-Tray, IDEXX Laboratories 2002). Water samples (1.0 and 50 mL) were added to 100-mL disposable bottles with DST® substrate and brought to 100-mL final volume with sterile 0.1 M KCl; samples were shaken vigorously and poured immediately into trays. The tray sealer dispensed the sample and sealed the wells. The trays were incubated in the dark at 35 ± 5 ^{o}C for 12-18 hours. Total coliforms were read under normal light (yellow color) and *E. coli* was read under long-wavelength UV (366 nm) fluorescence. Tables were used to convert these data into MPN values (Software Package, IDEXX Laboratories 2002).

Statistical analysis

Data were analyzed using the Statistical Analysis System (SAS 1990). Reservoir data and tributary data were analyzed separately. Each dataset was analyzed using a two-way analysis of variance for main effects (location and season) and their interactions. Data categories

for the main effect of season (growing season, May to September; senescent season, October to April) were similar for both reservoir and tributary data and were based on differences in water/air temperatures known to affect biological processes. Data categories for location for the reservoir data included uplake (Sites 3, 4, 6, 12, 16, and 17) and downlake (Sites 0, 1, 2, 7, 8, 9, 10, and 11), which reflected a longitudinal gradient expected to differ by depth and distance from tributary confluence. Differences among tributaries (locations at the reservoir confluence for Cobb Creek, Lake Creek, and Willow Creek) were tested using a two-way analysis of variance using tributary and season as main effects. Data was tested prior to ANOVAs using Proc Univariate and the Shapiro Wilk's statistic in SAS. Only pH was normally distributed. All other variables were transformed prior to analysis using a log10 transformation. Actual data presented in tables were not transformed for ease of visualization. Data were further explored using bivariate correlation and step-wise multiple regression in SAS. However, these analyses were not transformed. All significance levels were maintained at $p \leq 0.05$.

RESULTS AND DISCUSSION

Assessment of trophic status

Carlson (1977) proposed a Trophic State Index (TSI) based on a number of commonly measured limnological parameters including chlorophyll *a*, Secchi depth, and total phosphorus (Table 4). The actual indices are weighted using the following algorithms to give the approximate trophic classifications as represented in Table 4:

$$TSI(SD) = 60 - 14.41 \ln(SD)$$

$$TSI(CHL) = 9.81 \ln(CHL) + 30.6$$

$$TSI(TP) = 14.42 \ln(TP) + 4.15$$

where:

TSI=Trophic State Index,

SD= Secchi depth reading (m),

CHL= Chlorophyll *a* (µg/L), and

TP= Total Phosphorus (µg/L).

The State of Oklahoma relies primarily on chlorophyll *a* readings to evaluate reservoir trophic status (Table 5) due to concerns over the influence of suspended sediments, which can bias Secchi readings downward (i.e., mineral turbidity) and total phosphorus upward (i.e., phosphorus adsorbed to sediments) (OKDEQ 2000). However, for the purposes of comparison we evaluated the trophic status of Fort Cobb Reservoir using all three parameters.

Fort Cobb Reservoir is eutrophic/hypereutrophic based on all three endpoints proposed by Carlson (1977): chlorophyll *a*, Secchi depth, and total phosphorus. The grand means of Trophic State Indices over the course of the study were 61, 67, and 67 for TSI-Chl, TSI-TP, and TSI-Secchi, respectively (Table 6). An analysis of variance, comparing the effect of reservoir location (uplake or downlake) and season (growing season versus the senescent season) indicated that both location and season were significant main effects ($p \leq 0.01$) for TSI-Chl, TSI-TP, and TSI-Secchi; the interactions of main effects were not significant (Table 7). Trophic State Indices were significantly higher for the growing season (total mean of 63, 70, and 70 for TSI-Chl, TSI-TP, and TSI-Secchi, respectively), compared to the senescent season (total mean 59, 62, and 61 for TSI-Chl, TSI-TP, and TSI-Secchi, respectively) (Table 6). Trophic State Indices were also significantly higher in the uplake location (combined total uplake mean of 63, 68, and 72 for TSI-Chl, TSI-TP, and TSI-Secchi, respectively) compared to the downlake location (combined total downlake mean of 59, 65, and 62 for TSI-Chl, TSI-TP, and TSI-Secchi, respectively) (Table 6).

Temporal changes at three sites (Sites 0, 3, and 6) are presented in Figure 7. The TSI-Chl ranged from 39 to 85 among sites over the course of the study. Results indicate that the highest Trophic State Indices occurred during the spring/summer growing season, whereas lowest values were observed during the fall/winter senescent season. It should be noted that Site 6 was consistently higher in TSI values compared to the other sites due to two factors: 1) this was a shallow site (<1 m depth) and subject to wind-mixing and bioturbation by common carp (*Cyprinus carpio*), and 2) this is near the point of entry of the two major tributaries (Cobb Creek and Lake Creek) to the reservoir. The influence of tributary entry was diminished at sites located near the dam as biological processes became more influential on observed water quality parameters. Similar observations were made by Martin (2002).

The hypereutrophic conditions observed in Fort Cobb Reservoir indicate that substantial water quality degradation has occurred. The State of Oklahoma relies primarily on the

chlorophyll-based Trophic State Index and narrative provided in Table 5. Excessively high nutrients, reduced clarity, and nuisance algae are commonly observed in the reservoir as described in the sections below. During this study dissolved oxygen concentrations were not reduced to levels of concern for fish communities. However, the overall aesthetics of the lake are degraded to a point of recreational concern. For example, during the conduct of these studies we spoke with numerous recreational users who were selecting campsites based on their perceptions of water clarity and associated health concerns in the reservoir. Recreational users avoided use of the upper end of the reservoir due to the high levels of blue-green algae and turbidity. Although such observations are collected with some bias, they do indicate the public's awareness and concern for water quality in the reservoir.

Reservoir water quality: Spatial and temporal trends

Both season and reservoir location had significant effects on algal biomass and nutrient concentrations in Fort Cobb Reservoir (Table 7). Chlorophyll *a*, reflecting algal biomass, was significantly affected by both location ($p=0.0021$) and season ($p=0.0085$) (Table 7) and averaged 34 µg/L across all reservoir sites and dates (Table 6). Chlorophyll *a* values were significantly higher in the upper ends of the reservoir (combined total mean 46 µg/L) compared to the lower end of the reservoir (combined total mean 24 µg/L) (Table 6). Similarly, chlorophyll *a* was significantly higher during the growing season (total mean 38 µg/L) compared to the senescent season (total mean 30 µg/L) (Table 6). Highest chlorophyll *a* concentrations occurred in late summer in both 2000 and 2001 for Sites 0 and 3 (Figure 8). Chlorophyll concentrations were higher in the upper end of the reservoir at Site 6 on all sample dates due to the close proximity of nutrient input and the shallow depth (<1 m). Highest concentrations of chlorophyll *a* measured at Site 6 reached 250 µg/L in late February 2001. This high concentration was greatly in excess of concentrations in the lower reservoir, however, perhaps due to algal re-suspension. Chlorophyll concentrations decreased towards the dam during all seasons. The highest concentration of chlorophyll at the dam (Site 0) was approximately 70 µg/L in July of 2001 (Figure 8). Average annual concentrations of chlorophyll are presented in Figure 9. The data indicate that chlorophyll concentrations, reflective of algal biomass, decrease as one proceeds from the upper end of the lake towards the dam. No spikes in average chlorophyll

concentrations were observed in any embayment which indicates a lack of localized influence of nutrients associated with cottages or other form of localized reservoir development.

Particulate organic carbon reflects both living algal biomass and dead organic material including algae, bacteria, and detritus. Trends in particulate organic carbon (POC) were generally similar to those of chlorophyll and were significantly affected by both reservoir location ($p=0.0222$) and season ($p=0.0001$). However, unlike the chlorophyll data, we observed a significant season x location interaction ($p=0.0336$), which statistically confounds interpretation of the data (Table 7) (Snedecor and Cochran 1967). Particulate organic carbon averaged 2.5 mg/L across all locations and dates (Table 6); in general POC values were highest during the growing season and in the upper end of the reservoir but the degree of difference varied within season. For example, location differences (uplake versus downlake) in POC concentrations were much greater in the senescent season than during the growing season (Table 6). Highest concentrations of POC were observed at the upper end of the reservoir at Site 6 where a maximum of 9.6 mg/L POC was observed in June 2000 (Figure 8). Notably, this value did not correspond with chlorophyll on this date, which likely reflects an increase in suspended detrital material. Concentrations of POC and chlorophyll varied with season but were similar at Sites 0 and 3 when compared on a specific date. Departures of Site 6 from the other sites are likely due to the re-suspension of bottom materials that is reflected in other parameters including turbidity (discussed below).

Limnologists frequently use the ratio of total nitrogen to total phosphorus to assess nutrient limitation. Most freshwater aquatic systems are phosphorus limited, whereas marine systems are frequently nitrogen limited (Wetzel 1983, Rabalais and others 2000). Nitrogen:phosphorus (N:P) ratios of approximately 16 (range from 10 to 20) are considered optimum for phytoplankton production; numbers exceeding 20 are usually viewed as phosphorus limited, whereas ratios below 10 are considered nitrogen limited (Wetzel 1983, Geider and LaRoche 2002). However, regional departures can occur, such as in cases where light is limiting due to turbidity or other factors. In this study the N:P ratio averaged 17.6 over all dates and sites (Table 6). Both location ($p=0.0002$) and season ($p=0.0001$) had significant main effects on the N:P ratio (Table 7). Overall N:P ratios averaged across the study were 15.7 and 19.2 for uplake and downlake strata, respectively (Table 6). The N:P ratio during the growing seasons averaged 10.7 whereas the average increased to 25.8 during the senescent season (Table 6). Temporal

trends in N:P ratio are illustrated in Figure 8. These data indicate that N:P ratios average within the optimum range for maximum primary productivity. However, during the growing season the lake approaches nitrogen limited conditions, which are known to favor the growth of cyanobacteria that can "fix" atmospheric nitrogen within cells and thus out-compete other phytoplankton species for available phosphorus (Wetzel 1983). These findings are discussed in detail under the section entitled "Phytoplankton Community Dynamics".

Total phosphorus concentrations varied significantly by location ($p=0.0001$) and season ($p=0.0001$) (Table 7) and averaged 83 µg/L (combined total mean) across all sites and dates (Table 6). The combined total mean for total phosphorus was 96 and 72 µg/L for the uplake and downlake locations, respectively. During the growing season total phosphorus averaged 118 (uplake locations) and 88 µg/L (downlake locations) (Table 6). Total phosphorus decreased during the senescent season to an average of 69 (uplake locations) and 52 µg/L (downlake locations) (Table 6). Seasonal and spatial total phosphorus concentrations are presented in Figure 10. Concentrations of total phosphorus were highest at the upper end of the reservoir at Site 6 compared to downlake Sites 0 and 3 (Figure 10). Total phosphorus concentrations peaked during summer in association with maximum chlorophyll and particulate organic carbon concentrations (Figure 10, Table 6). Average annual concentrations of total phosphorus for 17 reservoir sites are presented in Figure 11. The data parallel the chlorophyll data in two significant ways: 1) there is a longitudinal decrease from the upper to the lower end of the reservoir, and 2) there are no peaks associated with any particular embayment, which indicates a lack of localized point sources of nutrients.

Soluble reactive phosphorus concentrations averaged 12 µg/L across all dates and sites (Table 6). There were no significant differences in soluble reactive phosphorus by location or season; however, the location x season interaction was significant ($p=0.0037$) (Table 7). Seasonal and spatial differences in soluble reactive phosphorus were less than those for total phosphorus with the exception of two dates (September 2001 and June 2002) when soluble reactive phosphorus was significantly higher at the dam (Site 0) compared to Sites 3 and 6 (Figure 10).

Total nitrogen, ammonia, and nitrate-nitrite averaged 1.18, 0.20, and 0.25 mg/L across all dates and sites (Table 6). There were significant effects ($p \leq 0.0184$) of both location and season on total nitrogen; however, the location x season interaction ($p=0.0004$) was also significant.

Total nitrogen concentrations averaged 1.10 (uplake locations) and 0.95 mg/L (downlake locations) during the growing season. Average total nitrogen increased, however, during the senescent season to 1.35 (uplake locations) and 1.39 mg/L (downlake locations). Thus, season had a greater effect on total nitrogen than location, which influences the statistical interaction.

Ammonia concentrations also differed significantly ($p \leq 0.0001$) by location, season, and the location x season interaction (Table 7). Ammonia averaged 0.20 mg/L (combined total mean over all seasons and dates) and 0.15 and 0.24 mg/L for the uplake and downlake areas, respectively (average over all sites and dates). During the growing season ammonia was low (0.07-0.08 mg/L) at both uplake and downlake sites. In contrast, ammonia averaged 0.24 mg/L (uplake locations) and 0.42 mg/L (downlake locations) during the winter, senescent season (Table 6). Thus, ammonia greatly increased in the senescent season due to increases in inputs (from sediments and waterfowl) combined with decreased biological uptake. Highest ammonia concentrations in the reservoir (0.5 mg/L; temperature=5 ^{o}C; pH=8.5; Site 0; February 2001) were well below those known to be of risk to aquatic life (e.g., continuous chronic criterion of 1.77 mg/L, temperature=5^{o}C, pH=8.5; USEPA 1999).

Trends in nitrate-nitrite in the reservoir were similar to those for ammonia; however, only season had a significant effect ($p \leq 0.0001$) on trends (Table 7). Nitrate-nitrite averaged 0.25 mg/L (combined mean across all dates) and was similar in uplake and downlake locations (all data combined) (Table 6). However, there were strong seasonal differences. Nitrate-nitrite was similar in uplake and downlake locations during the growing season (0.10-0.12 mg/L); however, nitrate-nitrite increased during the senescent season with averages of 0.44 and 0.39 mg/L for uplake and downlake locations, respectively. Thus, both ammonia and nitrate-nitrite increased during the cold winter period due to decreases in biological uptake.

Seasonal concentrations of total nitrogen, ammonia, and nitrate-nitrite at Sites 0, 3, and 6 are presented in Figure 12. Concentrations of total nitrogen ranged from 0.6 to 2.1 mg/L across Sites 0, 3, and 6. However, differences among sites were much less than those observed for chlorophyll, particulate organic carbon, and total phosphorus over the annual cycle. Results illustrate the strong seasonality in nitrogen as dictated by the annual cycle of growth and senescence. There were no localized sources of ammonia in the reservoir as indicated by the annual means for 17 sites presented in Figure 13; a generalized, longitudinal trend of increasing

ammonia was observed in a down-lake trend towards the dam. This trend was driven largely by the influence of ammonia release during the winter (Figure 13, Table 6).

Temperature data, used to divide the dataset into two seasonal datasets (senescent and growing), averaged 17.7 ^{o}C over all sites and dates (Table 6). The assignment of seasons was supported by the analysis of variance, which indicated that temperature indeed varied significantly ($p \leq 0.0001$) among seasons (Table 7). Overall, the average temperature was similar in the uplake (17.9 ^{o}C) and downlake (17.6 ^{o}C) designations (Table 6). Strong seasonal differences were noted, however, with a total average of 24.7 ^{o}C during the growing season and 9.2 ^{o}C during the senescent season. Temperatures were quite similar at the surface within the reservoir at Sites 0, 3, and 6 on a particular sampling date (Figure 14).

Secchi depth, or the visual water clarity of the reservoir, was significantly affected by both location and season ($p \leq 0.0001$), and averaged 0.7 m across all sites and dates (Table 6). Secchi depth averaged 0.40 and 0.72 m at the uplake and downlake locations, respectively, during the growing season; water clarity increased toward the dam. Secchi depth increased during the senescent season in both the uplake (0.68 m) and downlake (1.32 m) locations. Seasonal variation of Secchi transparency is visually presented in Figure 14. Highest Secchi depth occurred at Site 0 (dam location) on all dates (Figure 14). The upper end of the reservoir at Site 6 had low clarity and typically was less than 0.2 m (Figure 14). Highest water clarity occurred during the spring clear-water phase (February-May) due to classical reductions in phytoplankton biomass from zooplankton grazing (Wetzel 1983). Water clarity decreased in late summer/fall as phytoplankton biomass increased. A decrease in water clarity was also observed at Site 6 due to wind mixing and perhaps bioturbation by carp species. Average annual Secchi readings for each of the 17 reservoir locations are presented in Figure 15. Transparency increases downlake towards the dam with no particular deviation within any embayment or other location.

Turbidity, which is measured as light scatter by particulates, was essentially the reversal of trends of Secchi transparency and was significantly affected by main effects of location and season ($p \leq 0.0003$). Turbidity averaged 11 NTUs over all sites and dates and was much higher in the uplake regions (combined total average 16.5 NTUs) compared to the downlake region (combined total average 6.2 NTUs) due to the combination of phytoplankton biomass and re-suspension of sediments (Table 6, Figure 14).

Seasonal and spatial analysis of dissolved oxygen concentrations indicated that season had a stronger effect than spatial location. Reservoir dissolved oxygen concentrations averaged 10.3 mg/L across all sites and locations (Table 6). Highest average concentrations of dissolved oxygen occurred during the senescent season (12.0 mg/L) and were near saturation. However, lowest concentrations of average dissolved oxygen in surface waters were only 8.8 mg/L in the upper end of the reservoir during the growing season (Figure 16). Thus, in spite of the hypereutrophic conditions, dissolved oxygen concentrations usually remained well above levels of concern for fish (i.e., 5 mg/L), even during morning hours when lowest dissolved oxygen is expected. It is probable that relatively high dissolved oxygen is maintained due to a combination of wind mixing of the reservoir and super-saturation of algal cells at dusk.

The pH of the reservoir averaged 8.44 over the entire study (Table 6). Both location ($p=0.0261$ and season ($p=0.0454$) had significant effects on pH (Table 7). Average pH was higher during the growing season (total average pH=8.47) compared to the senescent season (total average pH=8.39). During both seasons pH tended to be higher in the uplake areas compared to the downlake areas. Temporal changes in pH at Sites 0, 3, and 6 are shown in Figure 16. Comparing these three sites it is evident that pH can range from 8.0 to 9.0 depending on the season and site. However, the average conditions were more similar when considering means by location and season and averaged from 8.35-8.51 (Table 6). This narrow range is due, in part, to the high buffering capacity of the reservoir and the fact that in-situ pH was generally measured in early morning. The pH is known to vary diurnally in aquatic systems due to carbon metabolism (Wetzel 1983). However, diurnal variations in pH were not evaluated in this study.

Conductivity, reflecting the ionic composition of water, varied little either spatially or temporally in the reservoir (Table 6). Conductivity was not significantly influenced by either location ($p=0.3971$) or season ($p=0.8489$) (Table 7). Slightly higher values of conductivity were observed at Site 6 (near the Cobb Creek Tributary confluence, 400-600 μS/cm) compared to down-lake locations (Figure 16); however, variations are not considered of ecological significance to reservoir water quality.

Both location ($p=0.0019$) and season ($p=0.0041$) had significant effects on alkalinity (Table 7). Alkalinity averaged 118 mg/L (combined total average) and was higher at uplake locations (total mean 123 mg/L) compared to downlake locations (total mean 114 mg/L). Spatial differences in alkalinity were greater during the senescent season (129 mg/L uplake; 115 mg/L

downlake) compared to the growing season where spatial differences were slight. Temporal differences in alkalinity at Sites 0, 3, and 6 are presented in Figure 17. Hardness was significantly different due to the main effects of location (p=0.0001), season (p=0.0015) and the location x season interaction (p=0.0157) (Table 7). Hardness averaged 189 mg/L across all sites and dates and was consistently higher in uplake compared to downlake locations. Differences among uplake and downlake locations were greatest during the senescent period (similar to alkalinity), which implies that both variables were driven by increased concentrations of divalent cations contributed by tributaries. These differences are readily apparent in trends in hardness plotted for Sites 0, 3, and 6 (Figure 17).

Reservoir water quality: Vertical trends and assessment of stratification

Reservoirs frequently exhibit vertical stratification in terms of dissolved oxygen, temperature, and pH. Vertical stratification of these parameters can profoundly influence reservoir dynamics due to changes in dissolved oxygen availability and subsequent nutrient exchange/dynamics with sediments (Wetzel 1983). Therefore, we monitored vertical stratification of the reservoir at regular intervals.

A marked thermocline was only observed on one occasion in the reservoir over the course of this study (Site 0 at the dam on July 12, 2000) (Figure 18). On this occasion we observed a 4 ^{o}C differential in temperature at approximately 10-14 m depth. A thermocline was never observed on other dates at Site 0 or at any time at other locations (Figure 18).

In contrast, strong vertical differences in dissolved oxygen and pH were observed during midsummer in deeper portions of the reservoir (Figures 19 and 20). Dissolved oxygen concentrations and pH frequently decreased with depth during the growing season when primary production was greatest. For example, significant decreases in dissolved oxygen and pH were observed in July 2000, April 2001, June 2001, and September 2001 (Figures 19 and 20). In July 2000, dissolved oxygen decreased to nearly zero near the dam at the depth interval from 12-14 m. However, such severe depletions were only observed on one date and dissolved oxygen remained in abundance in the upper 10 m of the reservoir. Decreases in dissolved oxygen and pH are due to a combination of several factors. The primary factor is light limitation, when high concentrations of algal biomass decrease the depth of the photic zone, therefore diminishing oxygen replenishment by primary producers at greater depths. Concomitant with decreased

primary productivity is the continued respiratory demand, which depletes remaining oxygen and decreases pH due to cellular respiration. These conditions are further exacerbated during cloudy weather periods with low wind mixing. Such conditions likely combined to decrease even surface dissolved oxygen and pH at the dam in September 2001. However, overall, oxyclines (and associated low dissolved oxygen and pH) are not a major problem in the reservoir due to the large oxygen supply available at the surface as indicated by Table 6 and Figure 16. However, under cloudy, windless conditions during the summer there are occasional dissolved oxygen depletions due to hypereutrophic conditions that should be monitored in the future should conditions worsen.

The high degree of reservoir uniformity can be compared using depth-specific comparisons of turbidity in the reservoir (Figure 21). Although turbidity concentrations varied seasonally and horizontally, as indicated by the data provided in Table 6 and Figure 14, there was relatively little vertical variation in turbidity concentrations. For example, at Sites 0 and 3 there was relatively little variation in turbidity with depth (Figure 21). In some cases there was a sudden increase in turbidity measured at the bottom; however, this was likely due to accidental physical disturbance by the YSI instrument during deployment.

Reservoir water quality: Phytoplankton community dynamics

A total of 76 phytoplankton taxa were identified during the study (Table 8). The phytoplankton community was dominated by the cyanobacteria (Phylum Cyanophycota) at all reservoir sites (Figure 22). The remainder of the community was dominated by Bacillariophyta, Chlorophycota, Chrysophyta, Cryptophycophyta, Euglenophyta, and the Pyrrophycophyta, however, these phyla were of low proportion compared to the cyanobacteria (Figure 22). Primary cyanobacterial genera in order of occurrence were *Microsystis*, *Wollea*, *Anabaena*, *Oscilliatoria, Merismopedia*, *Anabaenopsis*, and *Aphanizomenon spp.* (Figure 23). The species composition observed is consistent with that of hypereutrophic reservoirs (Wetzel 1983).

Total number of algal cells averaged 53,363 cells/mL when averaged across all dates (Table 6). Location ($p=0.0099$), season ($p< 0.0001$), and the location x season interaction ($p=0.0110$) had significant effects on algal numbers (Table 7). Total algal numbers were higher during the growing season compared to the senescent season. During the growing season algal numbers were much higher in the upper end of the reservoir compared to the lower end, but

numbers were highly variable. There was relatively little differences in uplake:downlake comparisons during the senescent season.

Simpson's dominance index of algae averaged 0.42 over the entire study (Table 6) and did not vary by location (p=0.4385) or season (p=0.6011) (Table 7).

Total number of algal species averaged 12.2 over the entire study (Table 6) and was significantly affected by season (p=0.0015) but not by location (p=0.2895) (Table 7). For example, highest algal species richness (16 species) was observed in the uplake portion of the reservoir during the growing season; however, species numbers dropped to 10.5 throughout the reservoir during the senescent season (Table 6).

Seasonal comparisons of the number of species, total numbers of algal cells, and dominance of phytoplankton by site are presented in Figure 24. Maximum numbers of species (38-42) were observed near the dam (Sites 0 and 1) and at the upper end of the reservoir (Sites 5 and 6); intermediate numbers of species were observed at mid-reservoir locations (Sites 2, 3, and 5). Highest cell numbers (up to 1.65 million cells per mL) were observed at the upper end of the reservoir at Sites 4, 5, and 6. However, maximum dominance values (0.65 to 0.85) were similar across sites.

Reservoir water quality: Algal toxins

The dominance of cyanobacteria in the phytoplankton community is a major concern for water quality in Fort Cobb Reservoir. Cyanobacteria blooms are frequently associated with taste and odor problems in reservoirs. In addition, many species of cyanobacteria, including *Microcystis sp*., can produce hepatotoxins and neurotoxins that are harmful to mammals (Carmichael 1992, 1997; Kotak and others 1993, 1995). *Microcystis aeruginosa* was the most commonly observed species of cyanobacteria observed in Fort Cob Reservoir (Figure 23).

We measured the spatial and temporal concentrations of the algal toxin microcystin in Fort Cobb Reservoir. Total microcystin concentrations in water were significantly affected by season (p< 0.0001) but not by location (p=0.2426) (Table 7). Microcystin concentrations averaged 15.5 ng/L over the course of the study (Table 6). Microcystin concentrations were higher in the growing season (28.7 and 19.0 ng/L in uplake and downlake areas, respectively) (Table 6). Concentrations were significantly lower during the senescent season and averaged 5.4

ng/L (Table 6). Seasonal variations in total microcystin and the four isomers at Sites 0, 3, and 6 are presented in Figures 25 and 26. Highest concentrations of total microcystin at Sites 0, 3, and 6 occurred on two dates: June 2000 and September 2001 (Figures 25 and 26). Concentrations ranged from 0 to a high of 120 ng/L total microcystin with highest concentrations being observed at Sites 3 and 6 in September 2001. Maximum concentrations of total microcystin were observed in September 2001 at Site 5 (148 ng/L); this value was approximately 15% of the World Health Organization's maximum for drinking water of 1 μg/L (Chorus and Bartram 1999). The relative concentrations of various isomers of microcystin varied. For example, highest concentrations of microcystin-lr occurred in June 2000 in association with a high runoff event; however, the relative proportion of microcystin-lr was low in September 2001 (Figure 25). In contrast, the proportions of microcystin-rr and microcystin-la were highest in September 2001 compared to other dates (Figures 25 and 26). Microcystin-yr generally comprised the lowest proportion of the four isomers (Figure 26).

Reservoir water quality: Coliform bacteria

Coliform bacteria can be contributed by both non-point sources (e.g., free-ranging domestic animals and wildlife; natural soil communities) and point sources (e.g., confined animal feeding operations and domestic sewage systems). Both total coliform bacteria (a diverse group of naturally-occurring soil bacteria) and *E. coli* (derived from the digestive tracts of warm-blooded vertebrates) were studied.

Concentrations of total coliform bacteria averaged 288 cells/mL) over all sites and dates in the reservoir (Table 6). However, total coliform numbers varied widely (range 2 – 4,839 cells/mL) and there were no significant effects of location ($p=0.6551$) or season ($p=0.2069$) (Table 7). There was a significant effect of season on numbers of *E. coli* ($p=0.0001$), but no effect of location due to high variation. Seasonal variations in total coliform and *E. coli* bacteria at Sites 0, 3, and 6 are presented in Figure 27. Seasonal variation in total coliform (soil-associated bacteria) and *E. coli* (digestive tract-associated bacteria) did not correspond due to their different origins. Highest numbers of *E. coli* were observed under two conditions: 1) during increased periods of precipitation which led to increased runoff (e.g., June 2000); and 2) when high numbers of waterfowl occupied the reservoir during over-wintering (e.g., February and December 2001). These sources are depicted in the average annual numbers of *E. coli*

presented for each of 17 reservoir sites, which shows higher average levels at the upper end (due to spring runoff) and lower end (due to wintering waterfowl) of the reservoir (Figure 28).

The causes and implications of these data vary. For example, storm runoff led to extremely high concentrations of *E. coli* in the upper ends of the reservoir due to the influence of tributaries. This usually occurred in late spring following rainfall. However, numbers of *E. coli* were rapidly attenuated as one moved downlake towards the dam. This attenuation is due to two reasons. First, bacteria were diluted by increased reservoir volume in downlake areas. Second, *E. coli* is not very competitive once outside the digestive tract of warm-blooded animals. Secondary peaks in *E. coli*, observed during late winter in the reservoir, occurred due to the presence of waterfowl. Peak numbers were much lower than those observed following the June 2000 storm event, but are still of concern. For example, in February 2001 E. coli exceeded the recreational contact criterion of 200 cells/100 mL (OKDEQ 2000). This exceedence of criteria is of particular concern because these observations occurred near the dam and the water intake for the city of Anadarko, OK. Additional samples detected low numbers of *E. coli* in water at the Anadarko water plant. However, they were not found in finished water after the water treatment process.

The State of Oklahoma has specific standards for total coliform bacteria and *E. coli* in reservoirs classified as recreational waters with potential dermal contact (OKDEQ 2000). Total coliform in water cannot exceed a monthly geometric mean of 5,000 cells/100 mL water at the point of intake of a public water supply. *E. coli* cannot exceed a monthly geometric mean of 200 cells/100 mL water. These are similar to national criteria for these bacteria. Note, however, that the criteria for total coliform bacteria, including *E. coli*, in finished drinking water is zero (Anderson and Davidson 1997, World Health Organization 1998).

There are many species of coliform naturally found in soils that are not of fecal origin. However, *E. coli* is derived from waste material from numerous animal species and is an indication of fecal contamination of water via direct input or runoff. Total coliform and *E. coli* are not necessarily primary concerns for human health in all situations. However, they are considered indicators of other potential pathogens in drinking water including *Salmonella sp.*, *Giardia sp*, *Cryptosporidium sp.,* and numerous other pathogenic organisms (Anderson and Davidson 1997). Thus, wintering waterfowl were observed to be the greatest source of bacteria at the drinking water intake located at the dam at Fort Cobb Reservoir. Future monitoring may

need to be done to determine if management actions are needed to reduce *E. coli* concentrations near the dam/water intake when high numbers of waterfowl are present.

Reservoir water quality: Evaluating relationships using correlation analysis

To gain further insight into the relationships among reservoir water quality indicators we conducted bivariate correlation analysis to determine associations among major limnological variables. These analyses do not necessarily reveal cause and effect relationships, but can be used to determine associations and in some cases to identify variables that can be cost-effective surrogates for mapping and monitoring of water quality trends. Bivariate correlations were only conducted on reservoir sites during the growing season dataset because these data most likely reflect the biological and chemical interactions that drive primary productivity and resulting eutrophication processes. Results of the bivariate correlations are presented in Table 9. Correlation coefficients (*r*) are presented in the upper right hand portion of the table; *p* values are presented in the lower left hand portion of the matrix with significant probabilities (i.e., $p \leq 0.05$) shaded.

The primary variables used to assess trophic status are chlorophyll *a*, total phosphorus, and Secchi transparency as described above under "Assessment of trophic status". Chlorophyll was most highly positively correlated with dissolved oxygen (r=0.600; p<0.001), followed by total P (r=0.438; 0.001) and pH (r=0.411; p<0.001); chlorophyll was negatively correlated with Secchi transparency (r=-0.432; p<0.001) and nitrate-nitrite (r=-0.324; p<0.002). Total phosphorus was negatively correlated with Secchi transparency (r=-0.695; p<0.001) and positively correlated with turbidity (i.e., opposite of Secchi transparency) (r=0.797; p=0.001). Thus, the three variables used for assessment of trophic status are highly correlated (both positively and negatively) and are therefore somewhat redundant in interpretation. Secchi transparency is the easiest and cheapest of the three assessment variables to measure, however it can be biased by the presence of inorganic turbidity. To successfully use Secchi transparency it would need to be measured concurrently with suspended solids and volatile suspended solids, which increases the overall cost of interpretation.

Secchi transparency was negatively correlated with dissolved oxygen (r= -0.0227; p= 0.031). This negative correlation is of particular concern because low dissolved oxygen is one of

the primary stressors that can impair fish production. However, as described above, dissolved oxygen concentrations are *not* currently a concern in Fort Cobb Reservoir.

Tributary water quality: Spatial and temporal trends in relation to nutrient loading

One of the major objectives of this study was to evaluate the influence of discharge on nutrient loading to Fort Cobb Reservoir. This relationship is important because in many systems a majority of nutrient loading can occur during a single, large flow event. This evaluation was problematic for our study because rainfall events were sporadic and discharge increased and decreased rapidly. In fact, the only major rain event that could be sampled occurred on June 17, 2000, on the first sampling date.

It should be noted that these estimates of nutrient loading are only the dates on which we measured water quality and do not reflect conditions under peak flows. Peak flows of over 1000 cfs are common but infrequent and only occur for a period of a few days (Figure 5). Peak flow events are significant for two reasons: 1) the majority of loads of nutrients and sediments can occur during large storm events as opposed to loads measured under average, low-flow conditions, and 2) we were only able to sample discharges and loads under a single storm event in this study. Thus, estimates of total annual nutrient loads must be modeled or actually measured during storm events for best accuracy. The data presented herein are not modeled and are thus only relative comparisons of loading. Highest loading of nitrogen and phosphorus tend to occur in early spring in association with peak discharge, although as indicated this loading is strongly flow-dependant. Other data has shown that peak loads of sediments and materials sorbed to particulates (such as phosphorus) typically occur on the ascending limb of a hydrograph when loose particles are subject to the greatest amount of erosive energy; accordingly, sediment loads tend to decrease on the descending limb of a hydrograph (Richards and Baker 1993). In contrast, dissolved substances such as nitrate-nitrite do not sorb to particles and actually increase during the descending limb of the hydrograph due to time lags from leaching processes (Richards and Baker 1993). Thus, the timing of sampling in relation to the hydrograph is critical in estimating true, total nutrient loads.

Seasonal discharge, total nitrogen loads, and total phosphorus loads measured at the Cobb Creek gaging station near Eakly, OK (i.e., Site 21) are presented in Figure 29. Peak discharge was measured on June 17, 2000 and was approximately 650 cfs (Figure 29). Other peak

discharges occurred, however we could not logistically sample nutrients on these dates. For example, on June 28, 2000, just 11 days following our sampling, the gaging station measured discharge in Cobb Creek at 2,800 cfs, or four-fold over that measured on June 17, 2000 (Figure 5). On June 17, 2000 total nitrogen and total phosphorus loading were 22,000 and 5,600 mg/sec, respectively (Figure 29). Discharge on all other dates was less than 30 cfs and nutrient loading declined accordingly.

To evaluate relative trends in nutrient loading among tributaries we compared inputs from Cobb Creek, Lake Creek, and Willow Creek during the period from April 1, 2001 to June 15, 2002. Note that this time period represents a subset of the previous information shown for Cobb Creek over the entire course of the study (Figure 30); it does not capture high flow events and is therefore useful for relative comparisons, only. Cobb Creek represented the greatest source of inflow to the reservoir with annual average discharge of 22.2 cfs, compared to 4.9, and 2.5 cfs for Lake Creek and Willow Creek, respectively, based on the dates we sampled (Table 10). There were significant main effects of both location ($p=0.0005$) and season ($p=0.0106$) on discharge (Table 11). Discharge of all tributaries decreased during the growing season and increased during the senescent season (Table 10). Seasonal discharges of each tributary are presented in Figure 30. Cobb Creek consistently delivers 75% of discharge to the reservoir. Actual amount of discharge varies seasonally with peaks occurring in spring during the period March to June (Figure 5 and Figure 30). Note, however, that the data presented in Figure 30 only reflect discharges on days of water sampling and therefore do not reflect peak flows. These are depicted in Figure 5, which demonstrates the abrupt trends of storm-water discharge in Cobb Creek.

Nutrient loading to Fort Cobb (total nitrogen and total phosphorus) responded differently by tributary and season. Total phosphorus loading was significantly different among tributaries ($p=0.0023$) but not by season ($p=0.0712$) (Table 11). Total phosphorus loading from Cobb Creek (total mean 139 mg/s) was significantly greater than that of Lake Creek 15 (mg/s) or Willow Creek (7 mg/s) (Table 10); thus, Cobb Creek, which comprises 62% of the watershed (Figure 2), delivers 86% of the total phosphorus load to the reservoir which is greater than anticipated simply based on the size of the watershed. Total phosphorus loading was slightly higher during the senescent season compared to the growing season but was not significantly different across seasons. To some degree this lack of significance across seasons is an artifact of

the assignment of seasons which is based on temperature as opposed to rainfall. Highest rainfall and discharge occurs from March to June, which overlaps the two season classes (growing versus senescent) and results in a high correlation between discharge and total nitrogen load among the three tributaries (Figure 30).

Total nitrogen loading was significantly affected by both tributary (p=0.0005) and season (p=0.0165) (Table 11). Total nitrogen loading from Cobb Creek (1199 mg/s) was significantly greater than either Lake Creek (194 mg/s) or Willow Creek (63 mg/sec) (Table 10). Thus, Cobb Creek delivered 82% of total nitrogen load during this study, which is similar in proportion to total phosphorus and greater than the actual watershed percentage (62%). Total nitrogen loading was significantly greater (approximately a two-fold increase) during the growing season (May to September), which again reflects greater overall discharge during this period due to the influence of spring rainfall events. Seasonal trends of total nitrogen loading for the three tributaries followed the same relative trends as for discharge and total phosphorus loading (Figure 30).

Tributary water quality: Spatial and temporal trends regarding concentrations

The highest total phosphorus concentrations in tributary streams occurred on June 17, 2000 (Figure 30) during a high-flow runoff event in which discharge reached approximately 650 cfs (Figure 5). June 17, 2000 was the highest discharge event sampled for nutrients and occurred on the first trip to the study area. Discharge decreased to approximately 50 cfs within 2 days after this event but increased to 2800 cfs after another runoff event on June 28 just 11 days after our sampling date. This dramatic range of discharge illustrates the dynamic nature of hydrology in the watershed.

On June 17, 2000 total phosphorus concentrations reached 420, 330, and 261 µg/L in Cobb Creek, Lake Creek, and Willow Creek, respectively; these measures were three-fold higher than any other date during the study (Figure 31). Total phosphorus concentrations were significantly affected by season (p=0.0105) but not by tributary (p=0.3496) (Table 11). The lack of effect of tributary on total phosphorus indicates that the sub-watersheds do not vary in terms of terrestrial erosion and transport processes that contribute to total phosphorus concentrations in streams. Total phosphorus concentrations averaged 139, 112, and 95 µg/L across all dates in Cobb Creek, Lake Creek, and Willow Creek, respectively (Table 10). Average total phosphorus

concentrations were greater during the growing season, which is related largely to higher discharge during April, May, and June.

Soluble reactive phosphorus concentrations did not vary significantly by season or tributary (Table 11) and averaged 57, 33, and 30 µg/L over all dates in Cobb Creek, Lake Creek, and Willow Creek, respectively (Table 10). Higher concentrations of soluble reactive phosphorus were observed during the growing season but varied on a monthly basis (Figure 31). Soluble reactive phosphorus was less affected by discharge compared to total phosphorus because much of the total phosphorus is highly bound to eroded and suspended particulates.

Total nitrogen concentrations varied significantly due to both tributary ($p=0.0001$) and season ($p=0.0391$) (Table 11). Total nitrogen averaged 1.77, 1.33, and 0.84 mg/L over all dates in Cobb Creek, Lake Creek, and Willow Creek, respectively (Table 10); further, each tributary differed significantly from each other ($p \leq 0.05$). Total nitrogen was significantly higher during the senescent season compared to the growing season due to the influence of high nitrate-nitrite concentrations, which increased due to plant senescence and decreased plant uptake (Table 10, Figure 32) (Richards and Baker 1993).

Nitrate-nitrite varied significantly by tributary ($p=0.0188$) and season ($p=0.0488$) (Table 11). Nitrate-nitrite averaged 1.35, 0.71, and 0.50 mg/L, respectively, in Cobb Creek, Lake Creek, and Willow Creek in accordance with stream size (Table 10, Figure 32). Nitrate-nitrite varied much less seasonally in Cobb Creek compared to the other tributaries, which may imply a greater influence of groundwater in this tributary.

Ammonia concentrations did not differ significantly across tributaries or season (Table 11). Ammonia averaged 0.09, 0.11, and 0.07 mg/L over all dates for Cobb Creek, Lake Creek, and Willow Creek, respectively, and varied little among seasons (Table 10). Peak concentrations of ammonia in tributaries (0.27 mg/L total ammonia; Lake Creek, June 2000) were below national water quality criteria (continuous chronic criterion of 0.52 mg/L total ammonia, temperature=25 ^{o}C, pH=8.5; USEPA 1999).

The ratio of total nitrogen to total phosphorus (N:P ratio) was significantly influenced by both tributary ($p=0.0093$) and season ($p=0.0001$) (Table 11). The N:P ratios averaged across all dates were 17, 15, and 10 for Cobb Creek, Lake Creek, and Willow Creek, respectively (Table 10). The N:P ratio was significantly higher during the senescent season (22, 21, and 13 for Cobb Creek, Lake Creek, and Willow Creek, respectively) compared to the growing season (Table 10)

due to the influence of high nitrate concentrations (Figures 32 and 33). Overall, N:P ratios ranged from 5 (nitrogen limitation) to 35 (phosphorus limitation) over the course of the study (Figure 33). During most of the study, however, the N:P ratio ranged from 10 to 20, which indicates optimum ratios for primary productivity.

Particulate organic carbon was significantly related to season (p=0.0262) but not to tributary location (p=0.5785) (Table 11). POC concentrations averaged across all dates were 2.64, 2.70, and 1.46 mg/L for Cobb Creek, Lake Creek, and Willow Creek, respectively (Table 10). Significantly higher values occurred during the growing period perhaps due to transport of dead organic matter associated with higher discharge conditions in year 2000 (Figure 33).

In contrast, chlorophyll was significantly higher in the senescent season in tributaries (p=0.0282) but did not differ by tributary (p=0.9475) (Table 11). The chlorophyll concentration pattern is the opposite trend of particulate organic carbon. Peak amounts of chlorophyll were observed in February 2001 when sunlight and low flow conditions maximized production and transport of algae in the tributaries (Figure 33). Lower concentrations of chlorophyll were observed in summer due to a combination of nutrient and light limitation.

Turbidity of tributary streams did not vary significantly by tributary or season (Table 11). Turbidity values averaged across all dates were 13, 17, and 10 NTU's for Cobb Creek, Lake Creek, and Willow Creek, respectively (Table 10), which are similar to those observed in the reservoir (Table 6). Turbidity was consistently higher in Lake Creek compared to the other tributaries but relative concentrations varied over the course of the study (Figure 34).

Dissolved oxygen was significantly greater during the senescent season compared to the growing season (p=0.0001) but did not vary by tributary (Table 11). Dissolved oxygen concentrations were lowest in midsummer of each year but were not at concentrations known to affect aquatic organisms (Figure 34).

Temperature of streams varied significantly by season (p=0.0001) but not by tributary (p=0.7929) (Table 11). Overall average temperatures were 18, 18, and 16 °C for Cobb Creek, Lake Creek, and Willow Creek, respectively (Table 10). Temperatures averaged 21-23 °C during the growing season and 10-12 °C during the senescent season, which indicates that the seasonal allocation of dates used in these analyses were reasonable. Seasonal changes in temperature among tributaries are presented in Figure 34.

Data for pH, conductivity, and hardness are presented in Figure 35. Neither tributary nor season had significant effects on pH of streams (Table 11). Overall averages for pH were 8.3, 8.3, and 8.2 for Cobb Creek, Lake Creek, and Willow Creek, respectively (Table 10) and were similar regardless of season (Table 10, Figure 35). In contrast, conductivity was significantly different among tributaries ($p=0.0450$; Table 11) with highest total average values observed in Cobb Creek (1021 µS/cm) compared to Lake Creek (745 µS/cm) and Willow Creek (644 µS/cm) (Table 10, Figure 35). Similar trends were observed with hardness, where Cobb Creek was significantly greater in hardness ($p=0.0001$) compared to either Lake or Willow Creek (Figure 35). Hardness also varied significantly with season ($p=0.0030$). For example, average hardness in Cobb Creek was 376 mg/L during the senescent season compared to 320 mg/L during the growing season. Alkalinity exhibited overall average values of 214, 233, and 208 mg/L in Cobb Creek, Lake Creek, and Willow Creek, respectively (Table 10) and was significantly greater in the senescent period compared to the growing season (Tables 10 and 11).

Tributary water quality: Bacteria

All three tributaries of Fort Cobb Reservoir were significant sources of bacteria to the reservoir (Figure 36). Total coliform and *E.coli* bacteria varied throughout the study and were not significantly different among the tributaries or seasons (Tables 10 and 11). Highest concentrations of total coliforms occurred in Cobb Creek in June of 2001 (250,000 cells/100 mL) and March of 2002 (170,000 cells/100 mg) (Figure 36). In contrast, the highest concentrations of *E. coli* were observed in all tributaries during the high water event of June 2000 (2,500 cells/mL) and in Lake Creek in May 2001 (2,800 cells/100 mL). Thus, temporal dynamics of bacterial classes varied with local conditions. High concentrations of *E. coli* were associated with the high water event, presumably due to erosion of manure by surface runoff. In contrast, total coliform concentrations were low under high discharge conditions. Elevated concentrations of *E. coli* observed in May 2001 in both Lake Creek and Cobb Creek occurred in spite of increased discharge, which implies some other mechanism of entry of bacteria such as wading livestock or entry of sewage. Peak numbers of total coliform occurred at the same time as peaks in the reservoir (Figure 36, Figure 27). In contrast, *E. coli* concentrations in the tributaries did not correspond to periods of high numbers in the reservoir (Figure 36, Figure 27). This difference

indicates that there are multiple sources of *E.coli* to the reservoir and that sources vary seasonally (i.e., waterfowl in winter; terrestrial runoff to tributaries in late spring and summer).

Tributary water quality: Metals

Metals in tributaries were measured on September 18, 2000, to determine anthropogenic and natural influences on cations and metals in the Fort Cobb Watershed (Table 12). Eight sites were sampled: 2 sites in Lake Creek (Sites 18 and 20); 2 sites in Willow Creek (Sites 13 and 15); and 4 sites in Cobb Creek and its tributaries (Sites 21, 22, 25, and 26) (Table 2, Figure 3). The data were unremarkable with the exception of strontium, which was higher in the main stem of Cobb Creek compared to Cobb Creek tributaries or either Lake or Willow Creek. Similarly elevated concentrations of strontium were observed in Cobb Creek by Martin (2002). Elevated strontium is related to a geologic outcropping in this sub-watershed (Peter Becker, Oklahoma Water Resources Division, personal communication). Overall observations of metal concentrations were similar to those of Martin (2002) and indicate that elemental concentrations are not of concern to wildlife or humans and that nutrients and bacteria are the primary environmental concern for water quality in Fort Cobb Reservoir.

Tributary water quality: Point-source evaluations of sources of nutrients and bacteria

Longitudinal changes in several water quality constituents (total nitrogen, nitrate-nitrite, ammonia, total phosphorus, soluble reactive phosphorus, and *E. coli* bacteria) were compared on three dates (April 23, 2001; June 21, 2001; and September 16, 2001) to determine if there were point sources or major patterns in nutrient changes among the major tributaries to Fort Cobb Reservoir. Five-Mile Creek, a major tributary entering Cobb Creek near Colony, OK, was evaluated separately (Figure 4). Note that the lower sites near the reservoir are repeated in this comparison from previous discussions to increase the resolution of the longitudinal perspective.

There were no significant point sources of total nitrogen observed in any tributary to Fort Cobb Reservoir. The strongest longitudinal variations in total nitrogen were observed between Sites 27 and 28 on the West Fork of Cobb Creek (Figure 37). Total nitrogen was reduced by over 50% between these two sites, which are located above and below Crowder Lake (Figure 4). A comparison of total nitrogen to nitrate-nitrite data (Figure 38) revealed that the majority of

total nitrogen was in the oxidized, dissolved state. Actual nitrogen losses were not studied specifically at these two sites, but losses probably occurred due to biological uptake in Crowder Lake during the growing season (i.e. low dissolved concentrations) similar to that observed in Fort Cobb Reservoir (Figure 12). Nitrate-nitrite and total nitrogen tended to increase gradually in Five-Mile Creek (entering as the eastern tributary to Cobb Creek), but were variable in the Lake Creek and Willow Creek tributaries (Figures 37 and 38).

Longitudinal ammonia concentrations varied more than total nitrogen and total nitrate in both temporal and spatial comparisons (Figure 39). Highest ammonia concentrations occurred in Lake Creek reaching 0.26 mg/L on April 23, 2001, and ammonia increased downstream toward Fort Cobb Reservoir. Cattle grazing was common in the riparian area of Lake Creek during this study and was the most likely source of ammonia. Lower ammonia concentrations were observed in the other tributaries. The greatest spatial differences seemed to be the significant increase in ammonia at Site 25 on Cobb Creek; this area was also heavily grazed by cattle which can contribute ammonia to surface waters. Lowest concentrations of ammonia occurred in September in all tributaries due to the lack of runoff and low stream flow conditions (Figure 39). Fencing of cattle is currently being evaluated in the Lake Creek NonPoint 303 Project in order to reduce inputs of ammonia to Fort Cobb Reservoir.

Longitudinal total phosphorus concentrations tended to increase with distance downstream at all sites (Figure 40) and were largely composed of the soluble reactive phosphorus form (Figure 41). The major point source contribution occurred at Site 14 on a small tributary to Willow Creek, where the soluble reactive phosphorus concentration was 200 µg/L; this value was 3-fold higher than that measured at Site 13 on the Willow Creek mainstem. The elevated concentration of soluble reactive phosphorus was consistent over time, which implies a continuous point source. However, the total amount contributed to the reservoir is not a significant amount of the total, because Site 14 is on a semi-permanent stream with relatively low flow.

Longitudinal evaluation of *E. coli* concentrations in tributaries revealed several point sources of input (Figure 42). Highest concentrations of *E. coli* were measured in April 2001 during periods of higher stream flow and runoff, and peaked in the Lake Creek tributary at 3,100 cells per 100 mL. Concentrations increased in the Lake Creek watershed at Site 32 in April and remained high at all downstream locations. High concentrations of *E. coli* were also observed at

Site 29 in Five Mile Creek in September, 2001; this unusual peak is most likely due to localized cattle grazing or direct entry of cattle into the stream. Consistent, high concentrations of *E. coli* occurred at Site 14 in Willow Creek, which is similar to patterns observed for soluble reactive phosphorus. This pattern would indicate a chronic, point source of manure or domestic sewage at this tributary. This input should be further evaluated in future studies.

CONCLUSIONS

A 2-year study of the trophic status of Fort Cobb Reservoir, OK, revealed that the reservoir is highly eutrophic and is dominated by blue-green algae. Blue-green algae are at high enough concentrations to create aesthetic concerns for recreational users. However, associated algal toxins are less than 15% of concentrations of concern to human health as established by the World Health Organization. Concentrations of *E. coli* bacteria reach concentrations of concern during the winter when high populations of waterfowl use the reservoir as winter staging habitat. Concentrations are particularly high at times near the dam water intake. *E. coli* has been detected in pre-treated water at the Anadarko Water Treatment Plant. Therefore, water managers should be aware of this condition and make plans to reduce bacterial concentrations in treated drinking water. Nutrient concentrations, algal biomass, bacteria, and turbidity were highest in the upper end of the reservoir (Site 6) near the confluence of the tributaries; water quality improves at the mid-lake and dam area due to the combination of biological uptake and deposition of nutrients and suspended materials. The upper end of the reservoir is quite shallow (<1 m) which exacerbates the effects of wind-mixing, boat wake, and bioturbation in re-suspension processes. Consideration should be given to the construction of a physical barrier in the upper end of the reservoir near Site 6 which might aid in sediment retention, establishment of emergent aquatic plant communities, and retention of nutrients prior to entry into the reservoir.

Each of the three main tributaries to Fort Cobb Reservoir contributes significant concentrations of nutrients and bacteria to the lake. High concentrations of nutrients and bacteria move during periods of high runoff, which results in surface and erosional losses in runoff. Cobb Creek drains the largest portion of the watershed and therefore delivers the majority of water and associated nutrients. Nutrient loading in Cobb Creek is greater than that indicated by the proportion of drainage, which may indicate differences in land use or merely be a function of

scale. The only consistent point source of bacteria and nutrients was at Site 14 in the Willow Creek Watershed, which may indicate the influence of livestock or domestic sewage.

Currently, there is a U.S. Environmental Protection Agency-sponsored 319 Non-Point Pollution Prevention Project in the Lake Creek watershed. This project is providing cost-share incentives and education to landowners in order to reduce nutrient loading to Fort Cobb Reservoir. Our results indicate that nutrient reduction is needed if water quality of the reservoir is to be maintained or improved. Erosion reduction, improved nutrient management plans, and fencing of livestock are potential management strategies that may improve water quality conditions. Continued monitoring of water quality in the reservoir and watershed are needed to measure the success of nutrient reduction programs.

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LITERATURE CITED

American Public Health Association (APHA). 1998. Enzyme substrate coliform test, i*n* Standard Methods for Examination of Water and Wastewater (20th ed.). APHA, Baltimore, MD, p. 9-68–9-70.

Anderson, K.A. and Davidson, P.M. 1997. Drinking water and recreational water quality: Microbiological criteria. Report No. CIS1069. University of Idaho Extension Service, Moscow, ID. 4 pp.

Carlson, R.E. 1977. A trophic state index for lakes. Limnol. Ocean. 22:361-369.

Carmichael, W.W. 1992. A status report on planktonic cyanobacteria (blue-green algae) and their toxins. U.S. Environmental Protection Agency, EPA/600/R-92/079, Cincinnati, OH, 141 pp.

Carmichael, W.W. 1997. The cyanotoxins, *in* Callows, J. A., ed., Advances in Botanical Research, Vol. 27, Academic Press Ltd., London, p. 211-256.

Chorus, I. and Bartram, J., eds. 1999. Toxic Cyanobacteria in Water: A Guide to their Public Health Consequences, Monitoring, and Management. E & FN Spon, London, 416 pp.

Edberg, S.C., Allen, M.J., Smith, D.B., and Kriz, N.J. 1990. Enumeration of total coliforms and *Eschericheria coli* from source water by the defined substrate technology. Appl. Environ. Microbiol. 56:366.

Geider, R.J. and LaRoche, J. 2002. Redfield revisited: Variability of C:N:P in marine microalgae and its biochemical basis. Eur. J. Phycol. 37:1-17.

Harada, K., Matsuura, K., Suzuki, M., Oka, H., Watanabe, M. F., Oishi, S., Dahlem, A. M., Beasley, V. R., and Carmichael, W. W. 1988. Analysis and purification of toxic peptides

from cyanobacteria by reversed-phase high-performance liquid chromatography. J. Chromatogr. Sci. A. 448:265-274.

IDEXX Laboratories. 2002. Standard method for total coliform detection in water with Define Substrate Technology®. IDEXX Laboratories, Westbrook, ME.

Kotak, B. G., Kenefick, S. L., Fritz, D. L., Rousseaux, C. G. Prepas, E. E., and Hrudey, S. E. 1993. Occurrence and toxicological evaluation of cyanobacterial toxins in Alberta lakes and farm dugouts. Wat. Res. 27:495-506.

Kotak, B. G., Lam, A.K.Y., Prepas, E. E., Kenefick, S. L., and Hrudey, S. E. 1995. Variability of the hepatotoxin microcystin-LR in hypereutrophic drinking water lakes. J. Phycol. 312: 248-263.

Martin, D. 2002. Fort Cobb Reservoir Water Quality Investigations. U.S. Fish and Wildlife Service Report to the U.S. Bureau of Reclamation and Fort Cobb Master Conservancy District. Tulsa, OK. 84 pp.

Oklahoma Water Resources Board. 1990. Oklahoma Water Atlas. Oklahoma Water Resources Board Publication 135. Oklahoma City. 96 pp.

Oklahoma Division of Environmental Quality (OKDEQ). 2000. The State of Oklahoma Water Quality Assessment Report to Congress. Section 305(b) Report, Oklahoma Division of Environmental Quality, Okahoma City, OK. 122 pp.

Park, H. D., Watanabe, M. F., Harada, K., Nagai, H., Suzuki, M., Watanabe, M., and Hayashi, H. 1993. Hepatotoxin (Microcystin) and neurotoxin (Anatoxin-a) contained in natural blooms and strains of cyanobacteria from Japanese freshwaters. Nat. Toxins 1:353-360.

Rabalais, N.N., R.E. Turner, and W.J. Wiseman, Jr. 2000. Hypoxia in the Gulf of Mexico. J. Env. Qual. 30:320-329.

Richards, R.P. and Baker, D.B. 1993. Pesticide concentration patterns in agricultural drainage networks in the Lake Erie basin. Environ. Toxicol. Chem. 12:13-26.

Snedecor, G.W. and W.G. Cochran. 1967. Statistical Methods, 6th ed. Iowa State University Press, Ames, IA, 593 pp.

Statistical Analysis System Institute. 1990. SAS Institute Inc. SAS/STAT Guide for Personal Computers, Version 6, 4th ed., Volumes 1 & 2. SAS Institute Inc., Cary, NC, 1685 pp.

U.S. Environmental Protection Agency (USEPA). 1999. 1999 Update of Ambient Water Quality Criteria for Ammonia. EPA 882-R-99-014, Cincinnati, OH.

Wetzel, R.G. 1983. Limnology, 2nd ed. Saunders College Publishing, Philadelphia, PA, 767 pp.

World Health Organization. 1998. Guidelines for drinking-water quality, 2nd Edition, Vol. 2: Health criteria and other supporting information, Geneva, Switzerland, pp. 940-949.

Table 1. Land use in the Fort Cobb Watershed by category and sub-watershed.

	Acres by Category							
Sub-watershed	**Agriculture**	**Forest**	**Range**	**Water**	**Urban**	**Barren**	**Total (acres)**	**Total (%)**
1	19,903	0	8,535	198	66	0	28,702	15
2	16,640	0	573	56	0	0	17,269	9
3	7,018	0	1,664	0	157	0	8,839	5
4	12,736	3	2,397	0	0	43	15,179	8
5	10,749	59	442	11	0	0	11,261	6
6	22,992	55	719	0	45	0	23,811	12
7	12,217	664	256	0	149	0	13,286	7
8	29,023	146	2,189	0	62	0	31,420	16
9	19,786	1,683	675	3,503	601	0	26,248	12
10	17,509	627	584	0	29	0	18,749	10
Total (acres)	168,573	3,237	18,034	3,768	1,109	43	194,764	100
Total (%)	87	2	9	2	1	0	100	

Table 2. Sampling locations used during study.

Station	Station Name	Location/Comments	North GPS	West GPS
0	Dam	0 mi NW from dam	35° 9.905'	98° 27.054'
1	1 mi marker	1 mi NW from dam	35° 10.606'	98° 27.734'
2	2 mi marker	2 mi NW from dam	35° 11.342'	98° 28.251'
3	3 mi marker	3 mi NW from dam	35° 11.893'	98° 29.171'
4	4 mi marker	4 mi NW from dam	35° 12.573'	98° 29.808'
5	5 mi marker	5 NW from dam	35° 13.155'	98° 30.571'
6	6 mi marker	6 mi NW from dam	35° 13.824'	98° 31.176'
7	Marina Cove	0.5 mi NE of dam, 6 mi from major inflow 36 ft deep	35° 10.325'	98° 27.015'
8	Farmers Slough	0.4 mi NW of dam, 6 mi from major inflow 10 ft deep	35° 10.018'	98° 28.063'
9	Carnegie Cove	1.8 mi NW of dam, 4.5 mi from major flow 11 ft deep	35° 10.786'	98° 28.629'
10	Kardokas Slough	Cove on E side of island 2.3 mi NE of dam, 4.5 mi from major inflow 21 ft deep	35° 11.816'	98° 28.157'
11	West Shore	Cove on W side W of 2.5 mi marker, 4 mi from major inflow 11 ft deep	35° 11.301'	98° 29.140'
12	West Shore	Cove on W side W of 3 mi marker, 3.5 mi from major inflow 13 ft deep	35° 11.904'	98° 29.524'
13	Willow Creek upstream	County Rd E1210, E branch of Willow Creek 2.5 mi NNE of reservoir	35° 15.716'	98° 27.073'
14	Willow Creek tributary	County Rd E1210, W branch of Willow Ck. 2.5 mi NNE of reservoir	35° 15.709'	98° 27.486'
15	Willow Creek downstream	County Rd N2540, below Sites 13 & 14, 0.5 mi NNE of reservoir	35° 13.982'	98° 28.002'
16	Willow Creek cove	Cove on E side, near inflow of Willow Creek 3.8 mi from dam, 1 mi from major inflow 15 ft deep	35° 13.124'	98° 28.938'
17	West Shore	Cove on W. side, boat ramp, park boundary 4.2 mi from dam, 2.5 mi from major inflow 9 ft deep	35° 12.474'	98° 30.307'
18	Lake Creek upstream	E Branch, State Rd 152 , 3.5 mi N of reservoir	35° 17.441'	98° 31.778'
19	Lake Creek tributary	W Branch, State Rd 152, 3.5 mi N of reservoir	35° 17.440'	98° 31.939'
20	Lake Creek downstream	Below Sites 18 & 19, County Rd E1220 0.5 mi N of reservoir	35° 16.573'	98° 31.835'

Table 2. Sampling locations used during study.—Continued

Station	Station Name	Location/Comments	North GPS	West GPS
21	Cobb Creek upstream	Near Eakly; State Rd 58 & 152 6 mi NW of reservoir, USGS Gage Station	35° 17.442'	98° 35.643'
22	Crooked Creek	Cobb Ck tributary, State Rd 58 5 mi WNW of reservoir	35° 15.277'	98° 36.376'
23	Camp Creek upstream	Cobb Ck tributary, State Rd 58 & N2460 4 mi W of reservoir, usually has no flow	35° 13.503'	98° 36.371'
24	Camp Creek downstream	Cobb Ck tributary, County Rd E1230 2 mi E of Site 23, 2 mi W of reservoir	35° 13.988'	98° 34.184'
25	Cobb Creek downstream	Cobb Ck above reservoir, County Rd N2490 1 mi W of reservoir	35° 14.082'	98° 33.192'
Dam 26	Below Dam	Below Dam outflow sites and above Site 26	35° 9.774'	98° 27.006'
26	Cobb Creek	Cobb Ck below reservoir, County Rd N2550 2 mi SSE of dam, USGS Gage Station	35° 8.250'	98° 25.835'
27	Cobb Creek	Upper Watershed, County Rd E1100 20 mi NW of reservoir	35° 25.266'	98° 43.173'
28	Cobb Creek	Upper Watershed County Rd E1140 & N2420, 15 mi NW of reservoir	35° 21.741'	98° 40.577'
29	Five Mile Creek	Upper watershed, Cobb Ck tributary County Rd E1100 15 mi NNW of reservoir	35° 25.255'	98° 36.668'
30	Five Mile Creek	Upper watershed, Cobb Ck tributary County Rd E1140, 10 mi NNW of reservoir	35° 21.784'	98° 36.204'
31	Lake Creek	Upper watershed, Cobb Ck tributary County Rd E1100, 13 mi N of reservoir	35° 25.248'	98° 31.358'
32	Lake Creek	Upper watershed, Cobb Ck tributary, County Rd E1140, 9 mi N of reservoir	35° 15.277'	98° 36.376'
Add1	Small tributary	Inflow is in Cove at Site 10	35° 12.251'	98° 27.629'
Add2	Small wetland	May be connected to the reservoir near Site 16	35° 13.938'	98° 27.880'
RC	Raw city water	City of Anadarko water treatment plant		
FC	Finished city water	City of Anadarko water treatment plant		
EDO	East dam outflow	Below dam from water collection and channeling structures	35° 15.277'	98° 36.376'
MDO	Middle dam outflow	Below dam from water collection and channeling structures	35° 13.503'	98° 36.371'
WDO	West dam outflow	Below dam from water collection and channeling structures	35° 13.988'	98° 34.184'

Table 3. List of measurement parameters and methods used for study[1].

Matrix	Parameter	Analytical Method
Water	Particulate organic carbon	Oxidation/coulometric
Water	Nitrate/nitrite	Colorimetric (sulfanilamide with cadmium reduction)
Water	Ammonia	Colorimetric (Salycilate/Nitroprusside)
Water	Soluble reactive phosphorus	Colorimetic (Ascorbic Acid)
Water	Total nitrogen/phosphorus	Colorimetric (Persulfate digestion)
Water	Turbidity	YSI Probe
Water	pH	YSI Probe
Water	Alkalinity	Titrimetric
Water	Hardness	Titrimetric
Water	Conductivity	YSI Probe
Water	Dissolved oxygen	YSI Probe
Water	Temperature	YSI Probe
Water	Metals	ICP/MS; EPA methods
Phytoplankton	Numbers	Counts (100X magnification)
Phytoplankton	Taxonomic composition	Keys and references
Phytoplankton	Biomass (Chlorophyll *a*)	Fluorometry
Phytoplankton	Biomass (Cyanin)	Fluorometry
Phytoplankton	Microcystin	HPLC
Bacteria	Total coliform; E. coli	IDEX enzyme:specific color

[1]SOPs available on request.

Table 4. Trophic State Index (TSI) proposed by Carlson (1977)[1].

		Oligotrophic	Mestotrophic	Eutrophic		Hypereutrophic
TSI		30	40	50	60	70
Transparency (Secchi depth (m))		8	4	2	1	0.5
Chlorophyll *a* (µg/L)		0.95	1.6	7	20	55
Total Phosphorus (µg/L)		0	12	24	40	90

[1]Note: Classifications are for relative comparison only and must be adjusted for site-specific conditions as described by Carlson (1997). For further information refer to web page at http:// dipin.kent.edu/tsi.htm.

Table 5. Oklahoma's classification of trophic status based on the chlorophyll index component of Carlson's Trophic State Index (1977)[1].

Chl TSI (range)	Trophic State	Narrative Reservoir Conditions
<40	Oligotrophic	Reservoir typified by low nutrients, low productivity, high clarity, and good water quality.
41 to 50	Mesotrophic	Reservoir with increased levels of nutrients and productivity.
51 to 60	Eutrophic	Reservoir with elevated nutrients, sedimentation, productivity, and decreased clarity.
>60	Hypereutrophic	Reservoir with very high levels of nutrients, productivity, and decreased clarity. Nuisance algae, low dissolved oxygen, and fish kills likely or common leading to loss of recreational values.

[1]Information from OKDEQ (2000).

Table 6. Means of reservoir water quality variables in relation to season and location[1,2].

Parameter	Sampling Season and Reservoir Location								
	Growing			Senescent			Combined Total		
	Uplake (n=42)	Downlake (n=48)	Total (n=90)	Uplake (n=35)	Downlake (n=40)	Total n=(75)	Up lake (n=77)	Down lake (n=88)	Total (n=165)
TSI-Chl *a*	65.0 (8.6)	60.9 (8.5)	62.8 (8.8)	61.6 (12.0)	56.7 (6.1)	59.0 (9.6)	63.4 (10.5)	59.0 (7.7)	61.0 (9.3)
TSI-TP	72.2 (4.6)	68.4 (2.8)	70.2 (4.2)	63.9 (5.6)	60.8 (3.5)	62.3 (4.8)	68.4 (6.5)	65.0 (4.9)	66.6 (6.0)
TSI-Secchi	74.6 (7.2)	65.2 (3.4)	69.6 (7.2)	67.2 (7.9)	56.4 (3.4)	61.4 (8.0)	72.1 (8.2)	62.3 (5.4)	66.9 (8.4)
Chl *a* (µg/L) lab	46.8 (38.2)	30.9 (25.1)	38.2 (32.5)	45.0 (53.1)	17.0 (9.4)	30.0 (39.3)	45.9 (45.6)	24.4 (20.5)	34.3 (36.0)
POC (mg/L)	3.63 (2.12)	3.06 (1.99)	3.32 (2.06)	1.81 (1.51)	1.12 (0.85)	1.45 (1.25)	2.80 (2.06)	2.20 (1.86)	2.48 (1.97)
N:P ratio	10.0 (2.9)	11.3 (3.3)	10.7 (3.2)	22.6 (8.0)	28.6 (8.9)	25.8 (9.0)	15.7 (8.5)	19.2 (10.8)	17.6 (9.9)
TP (µg/L)	118.3 (44.3)	87.7 (16.8)	102.0 (35.9)	68.8 (33.7)	52.4 (13.2)	60.0 (26.1)	95.8 (46.7)	71.7 (23.3)	82.9 (38.0)
SRP (µg/L)	11.8 (8.2)	12.4 (17.2)	12.1 (13.7)	9.1 (12.6)	13.5 (15.4)	11.5 (14.3)	10.6 (10.5)	12.9 (16.3)	11.8 (13.9)
TN (mg/L)	1.10 (0.2)	0.95 (0.14)	1.02 (0.19)	1.35 (0.20)	1.39 (0.17)	1.37 (0.18)	1.22 (0.24)	1.15 (0.27)	1.18 (0.26)
NH_3 (mg/L)	0.07 (0.08)	0.08 (0.10)	0.07 (0.09)	0.24 (0.13)	0.42 (0.06)	0.34 (0.13)	0.15 (0.13)	0.24 (0.19)	0.20 (0.17)
NO_3NO_2 (mg/L)	0.10 (0.12)	0.12 (0.13)	0.11 (0.12)	0.44 (0.13)	0.39 (0.11)	0.41 (0.12)	0.26 (0.21)	0.25 (0.18)	0.25 (0.19)
Temp (^oC) field	24.8 (2.6)	24.7 (2.8)	24.7 (2.7)	9.5 (5.0)	9.0 (4.8)	9.2 (4.9)	17.9 (8.6)	17.6 (8.8)	17.7 (8.6)
Turbidity (NTU)	19.6 (17.1)	7.1 (2.3)	13.0 (13.3)	14.1 (12.9)	5.4 (1.6)	9.5 (9.8)	16.5 (15.1)	6.2 (2.1)	11.0 (11.6)
Secchi depth (m)	0.40 (0.17)	0.72 (0.17)	0.57 (0.23)	0.68 (0.27)	1.32 (0.32)	1.02 (0.44)	0.50 (0.24)	0.91 (0.37)	0.72 (0.38)
Dissolved oxygen (mg/L)	8.8 (1.6)	9.1 (1.9)	8.9 (1.8)	12.2 (1.3)	11.8 (1.1)	12.0 (1.2)	10.3 (2.2)	10.3 (2.1)	10.3 (2.6)
pH	8.51 (0.24)	8.44 (0.32)	8.47 (0.29)	8.44 (0.22)	8.35 (0.16)	8.39 (0.20)	8.48 (0.23)	8.40 (0.26)	8.44 (0.25)
Conductivity (µS/cm)	580 (181)	570 (175)	575 (177)	585 (187)	471 (26)	568 (185)	582 (182)	562 (179)	572 (180)
Alkalinity (mg/L)	117 (19)	113 (16)	115 (17)	129 (20)	115 (11)	122 (17)	123 (21)	114 (14)	118 (18)
Hardness (mg/L)	187 (26)	182 (19)	184 (22)	205 (21)	184 (13)	194 (20)	195 (25)	183 (17)	189 (22)
Total coliforms (#/ml)[3]	340 (786)	347 (772)	344 (773)	242 (496)	215 (355)	228 (425)	292 (656)	285 (611)	288 (630)
E. coli (#/100 ml) [3]	80 (420)	8 (13)	42 (286)	38 (73)	51 (66)	45 (70)	60 302)	29 (51)	43 (210)
Total microcystin (ng/L)[4]	28.7 (38.7)	19.0 (27.3)	23.5 (33.2)	5.6 (6.0)	5.2 (5.1)	5.4 (5.5)	18.5 (31.1)	12.9 (21.7)	15.5 (26.6)
Algae (# cells/ml)	169759 (305935)	53847 (47972)	107939 (218187)	8344 (12050)	7480 (8387)	7883 (10209)	81717 (220215)	28556 (40109)	53363 (155030)
Simpson's Dom. algae	0.42 (0.23)	0.40 (0.19)	0.41 (0.20)	0.39 (0.19)	0.46 (0.23)	0.43 (0.21)	0.40 (0.20)	0.43 (0.21)	0.42 (0.21)
Total # algal species	15.6 (9.0)	13.0 (1.2)	14.2 (8.1)	10.4 (6.6)	10.5 (7.7)	10.5 (7.2)	12.8 (8.2)	11.6 (7.5)	12.2 (7.8)

[1]Growing season May to September; senescent season October to April.
[2]Locations refer to uplake (Sites 3, 4, 5, 6, 12, 16, and 17; northwest tributary end) or downlake (Sites 0, 1, 2, 7, 8, 9, 10, and 11; southeast dam end).
[3]Note difference in units of expression for total coliform and *E. coli* numbers for ease of presentation.
[4]Total microcystin represents sum of four isomers.

Table 7. Two-way analysis of variance of reservoir water quality data.

Parameter	Main Effects and Interaction ($p \leq X$)[1]		
	Location[2]	Season[3]	Season x Location
TSI-Secchi	0.0001	0.0001	0.1906
TSI-Chl *a*	0.0059	0.0069	0.9431
TSI-TP	0.0001	0.0001	0.8046
Chl *a*	0.0021	0.0085	0.7751
POC	0.0222	0.0001	0.0336
N:P ratio	0.0002	0.0001	0.2110
TP	0.0001	0.0001	0.6068
SRP	0.8212	0.8580	0.0037
TN	0.0184	0.0001	0.0004
NH_3	0.0001	0.0001	0.0001
NO_3NO_2	0.4775	0.0001	0.6920
Temperature	0.4780	0.0001	0.5297
Turbidity	0.0001	0.0003	0.5265
Dissolved oxygen	0.9192	0.0001	0.2843
pH	0.0261	0.0454	0.7841
Conductivity	0.3971	0.8489	0.6413
Alkalinity	0.0019	0.0041	0.0688
Hardness	0.0001	0.0015	0.0157
Total coliform bacteria	0.6551	0.2069	0.6018
E coli bacteria	0.5939	0.0001	0.824
Total microcystin	0.2426	0.0001	0.2873
Total # algal cells	0.0099	<0.0001	0.0110
Simpson's Dominance algae	0.4385	0.6011	0.1711
Total # algal species	0.2895	0.0015	0.2549

[1]Associated probability significant if $p \leq 0.05$ (shaded).
[2]Locations refer to uplake (Sites 3, 4, 5, 6, 12, 16, and 17; northwest tributary end) or downlake (Sites 0, 1, 2, 7, 8, 9, 10, and 11; southeast dam end).
[3]Growing season May to September; senescent season October to April.

Table 8. Species list of phytoplankton sampling, June 2000 through May 2002.

Phylum	Class	Order	Family	Genus
Chlorophycota	Chlorophyceae	Chlorococcales	Characiaceae	*Characium*
			Chlorococcaceae	*Tetraedron*
			Coccomyxaceae	*Gloeocystis*
			Coelastraceae	*Coelastrum*
			Dictyosphaeriaceae	*Botryococcus*
			Hydrodictyaceae	*Pediastrum*
			Micractiniaceae	*Micractinium*
			Oocystaceae	*Ankistrodesmus*
				Closteriopsis
				Franceia
				Glaucocystis
				Kirchneriella
				Oocystis
				Radiococcus
			Scenedesmaceae	*Actinastrum*
				Crucigenia
				Scenedesmus
		Klebsormidiales	Elakatotrichaceae	*Elakatothrix*
		Tetrasporales	Palmellopsidaceae	*Sphaerocystis*
		Ulotrichales	Ulotrichaceae	*Dactylococcopsis*
		Volvocales	Chlamydomonadaceae	*Carteria*
				Chlamydomonas
			Haematococcaceae	*Haematococcus*
			Volvocaceae	*Eudorina*
				Pandorina
		Zygnematales	Desmidiaceae	*Arthrodesmus*
				Closterium
				Cosmarium
				Staurastrum
Chrysophyta	Chrysophyceae	Chromalinales	Chrysococcaceae	*Chrysococcus*
		Chrysamoebidales	Stylococcaceae	*Chrysopyxis*
		Ochromonadales	Ochromonadaceae	*Uroglenopsis*
			Synuraceae	*Mallomonas*
				Synura
		Rhizochrysidales	Rhizochrysidaceae	*Chrysamoeba*
Cryptophycophyta	Cryptophyceae	Cryptomonadales	Cryptomonadaceae	*Chilomonas*
				Chroomonas
				Cryptomonas
				Rhodomonas
Euglenophycota	Euglenophyceae	Euglenales	Euglenaceae	*Euglena*
				Lepocinclis
				Phacus
				Trachelomonas
Pyrrophycophyta	Dinophyceae	Gonyaulacales	Gonyaulacaceae	*Gonyaulax*
		Peridiniales	Peridiniaceae	*Peridinium*
Xanthophyta	Xanthophyceae	Mischococcales	Characiopsidaceae	*Peroniella*

Table 8. Species list of phytoplankton sampling, June 2000 through May 2002.—Continued

Phylum	Class	Order	Family	Genus
Bacillariophyta	Bacillariophyceae	Achnanthales	Achnanthaceae	*Achnanthes*
			Cocconeidaceae	*Cocconeis*
		Bacillariales	Bacillariaceae	*Nitzschia*
		Cymbellales	Cymbellaceae	*Cymbella*
			Gomphonemataceae	*Gomphonema*
		Naviculales	Pinnulariaceae	*Caloneis*
			Pleurosigmataceae	*Gyrosigma*
			Naviculaceae	*Navicula*
			Stauroneidaceae	*Stauroneis*
		Thalassiophysales	Catenulaceae	*Amphora*
	Coscinodiscophyceae	Rhizosoleniales	Rhizosoleniaceae	*Rhizosolenia*
		Thalassiosirales	Stephanodiscaceae	*Cyclotella*
				Stephanodiscus
	Fragilariophyceae	Fragilariales	Fragilariaceae	*Asterionella*
				Fragilaria
				Synedra
Cyanophycota	Cyanophyceae	Chroococcales	Chroococcaceae	*Aphanocapsa*
				Aphanothece
				Chroococcus
				Gloeocapsa
				Merismopedia
				Microcystis
				Synechococcus
		Nostocales	Nostocaceae	*Anabaena*
				Anabaenopsis
				Aphanizomenon
				Wollea
			Oscillatoriaceae	*Arthrospira*
				Lyngbya
				Oscillatoria
				Phormidium

Table 9. Bivariate correlation analysis of reservoir data for all dates during growing season (May to September).

		Regression Coefficient, *r*																				
		Diss. Oxygen	pH	Alkalinty	Hardness	Conductivity	Temperature	Secchi	Turb.	TN	TP	NO_3NO_2	Ammonia	SRP	Chl *a*	In vivo Cyanin	Total Micro-cystin	*E. coli*	Total Coliform	Total # algae	Algal dominance	# algal species
Probability, *p*	**Diss. Oxygen**	1	0.586	-0.186	-0.128	0.359	0.323	-0.227	-0.013	-0.093	-0.002	-0.387	-0.349	-0.447	0.600	-0.299	-0.237	0.006	0.106	0.016	0.029	-0.001
	pH	<.001	1	-0.524	-0.484	-0.090	0.433	-0.391	0.043	0.151	0.127	-0.612	-0.481	-0.568	0.411	-0.395	-0.399	0.006	0.523	0.082	-0.076	0.101
	Alkalinity	0.079	<.001	1	0.921	-0.112	-0.452	0.187	0.481	0.109	-0.043	0.715	0.681	0.226	0.048	-0.689	0.050	0.086	-0.478	0.129	0.228	-0.059
	Hardness	0.230	<.001	<.001	1	0.095	-0.557	0.209	0.430	0.206	-0.085	0.665	0.649	0.336	0.028	-0.591	0.261	-0.045	-0.514	0.006	-0.009	0.203
	Conductivity	0.001	0.400	0.294	0.374	1	0.047	-0.053	-0.107	-0.143	-0.032	-0.247	-0.305	0.286	0.315	0.545	0.483	-0.113	-0.300	-0.144	-0.139	0.297
	Temperature	0.002	<.001	<.001	<.001	0.663	1	-0.277	0.190	-0.257	0.285	-0.732	-0.757	-0.406	0.412	0.266	-0.505	-0.228	0.026	0.149	0.394	-0.433
	Secchi	0.031	0.001	0.078	0.049	0.619	0.008	1	-0.632	-0.406	-0.695	0.321	0.449	0.296	-0.432	-0.173	-0.085	-0.261	-0.201	-0.295	-0.201	-0.065
	Turbidity	0.924	0.746	0.001	0.001	0.417	0.146	<.001	1	0.616	0.797	-0.004	-0.135	-0.047	0.392	-0.190	-0.124	0.240	-0.067	0.717	0.378	-0.005
	TN	0.385	0.157	0.307	0.051	0.178	0.014	<.001	<.001	1	0.578	0.056	0.138	0.256	0.199	-0.326	0.017	0.139	0.345	0.170	-0.084	0.370
	TP	0.986	0.232	0.685	0.424	0.763	0.007	<.001	<.001	<.001	1	-0.195	-0.369	-0.001	0.438	0.286	0.006	0.171	0.163	0.362	0.299	-0.036
	NO_3NO_2	0.001	<.001	<.001	<.001	0.020	<.001	0.002	0.977	0.605	0.067	1	0.719	0.290	-0.324	-0.506	0.186	0.263	-0.284	0.007	0.016	0.100
	Ammonia	0.001	<.001	<.001	<.001	0.004	<.001	<.001	0.305	0.198	0.001	<.001	1	0.329	-0.339	-0.522	0.102	0.206	-0.330	-0.136	-0.144	0.082
	SRP	<.001	<.001	0.032	0.001	0.006	<.001	0.005	0.721	0.015	0.997	0.006	0.002	1	-0.257	0.295	0.597	0.120	-0.277	0.007	-0.175	0.349
	Chl *a*	<.001	<.001	0.655	0.791	0.003	<.001	<.001	0.002	0.060	<.001	0.002	0.001	0.014	1	-0.390	-0.208	0.022	0.059	0.269	0.348	-0.061
	In vivo Cyanin	0.046	0.007	<.001	<.001	0.001	0.077	0.256	0.210	0.029	0.057	0.001	0.001	0.049	0.008	1	0.556	-0.711	-0.129	-0.197	-0.158	-0.043
	Tot. Microcystin	0.041	0.001	0.668	0.024	<.001	<.001	0.467	0.415	0.886	0.961	0.112	0.390	<.001	0.073	0.001	1	0.189	-0.254	0.094	-0.195	0.473
	E. coli	0.963	0.963	0.477	0.710	0.348	0.056	0.028	0.112	0.249	0.154	0.028	0.087	0.318	0.854	<.001	0.114	1	-0.043	-0.002	0.073	-0.049
	Total Coliform	0.376	<.001	<.001	<.001	0.011	0.830	0.090	0.661	0.003	0.171	0.016	0.005	0.019	0.629	0.498	0.031	0.723	1	-0.009	-0.219	0.242
	Total # algae	0.884	0.445	0.226	0.952	0.176	0.162	0.005	<.001	0.108	0.001	.0947	0.202	0.945	0.010	0.194	0.420	0.984	0.940	1	0.316	-0.118
	Algal dominance	0.789	0.477	0.310	0.933	0.192	0.001	0.057	0.003	0.434	0.004	0.878	0.177	0.099	0.001	0.299	0.093	0.545	0.065	0.002	1	-0.422
	# Algal species	0.996	0.346	0.586	0.055	0.004	<.001	0.544	0.970	0.001	0.738	0.353	0.448	0.001	0.567	0.781	<.001	0.684	0.041	0.267	<.001	1

Table 10. Means of tributary water quality variables in relation to season.[1,2]

Parameter	Season and Tributary								
	Growing			Senescent			All Dates		
	Willow (n=6)	Lake (n=6)	Cobb (n=6)	Willow (n=5)	Lake (n=5)	Cobb (n=5)	Willow (n=11)	Lake (n=11)	Cobb (n=11)
Discharge (cfs)	1.5 (0.8)	3.4 (2.6)	15.8 (10.1)	3.9 (0.3)	7.1 (1.0)	31.9 (1.7)	2.5 (1.4)	4.9 (2.8)	22.2 (11.3)
TP Load (mg/s)	4 (2)	10 (10)	71 (59)	11 (4)	21 (13)	90 (57)	7 (6)	15 (11)	139 (104)
TN Load (mg/s)	34.2 (14.7)	102.7 (90.8)	883 (646)	106 (29)	330 (197)	1672 (830)	63 (43)	194 (172)	1199 (753)
N:P ratio	8 (3)	10 (3)	12 (5)	13 (4)	21 (8)	22 (4)	10 (4)	15 (8)	17 (7)
POC (mg/L)	1.74 (0.89)	3.948 (4.44)	3.59 (4.31)	1.13 (0.55)	1.23 (0.70)	1.50 (0.79)	1.46 (0.78)	2.70 (3.47)	2.64 (3.28)
Chl *a* (µg/L)	5.0 (4.3)	6.5 (4.8)	5.0 (5.0)	14.1 (16.8)	14.7 (14.2)	10.0 (6.3)	9.5 (12.5)	10.6 (10.9)	7.5 (6.0)
TP (µg/L)	113 (74)	136 (104)	184 (124)	73 (25)	84 (37)	86 (34)	95 (59)	112 (82).	139 (104)
SRP (µg/L)	33.1 (23.4)	40.3 (29.5)	68.4 (30.2)	25.7 (20.6)	24.1 (24.8)	42.5 (25.6)	29.7 (21.5)	32.9 (27.4)	56.6 (30.0)
TN (mg/L)	0.77 (0.22)	1.12 (0.39)	1.72 (0.28)	0.93 (0.19)	1.59 (0.40)	1.82 (0.42)	0.84 (0.21)	1.33 (0.45)	1.77 (0.34)
NO_3NO_2 (mg/L)	0.39 (0.19)	0.50 (0.33)	1.33 (0.44)	0.64 (0.11)	1.02 (0.29)	1.37 (0.28)	0.50 (0.20)	0.71 (0.40)	1.35 (0.35)
NH_3 (mg/L)	0.05 (0.03)	0.10 (0.09)	0.08 (0.05)	0.09 (0.05)	0.12 (0.09)	0.10 (0.07)	0.07 (0.04)	0.11 (0.09)	0.09 (0.06)
Turbidity (NTU)	9.4 (3.7)	18.0 (6.6)	14.0 (4.3)	10.0 (3.0)	15.3 (6.3)	12.8 (9.4)	9.7 (3.1)	16.5 (6.2)	13.3 (7.2)
Dissolved oxygen (mg/L)	7.9 (1.1)	7.8 (0.7)	8.7 (1.3)	11.6 (0.2)	15.0 (4.5)	13.4 (1.8)	9.5 (2.1)	11.1 (4.7)	10.9 (2.9)
Temp (°C)	20.7 (3.1)	22.8 (3.7)	22.2 (3.2)	10.4 (3.2)	11.4 (6.1)	11.9 (5.2)	16.0 (6.2)	17.6 (7.5)	17.6 (6.7)
pH	8.2 (0.1)	8.3 (0.15)	8.2 (0.3)	8.2 (0.1)	8.3 (0.2)	8.3 (0.3)	8.2 (0.11)	8.3 (0.2)	8.3 (0.20)
Conductivity (µS/cm)	659 (268)	767 (343)	1079 (412)	626 (321)	620 (40)	952 (362)	644 (278)	745 (302)	1021 (377)
Hardness (mg/L as $CaCO_3$)	227 (48)	216 (47)	320 (76)	266 (10)	274 (26)	376 (18)	244 (40)	242 (49)	348 (60)
Alkalinity (mg/L as $CaCO_3$)	194 (45)	211 (52)	204 (56)	225 (10)	260 (21)	225 (10)	208 (36)	233 (47)	214 (40)
Total coliform (#/ml.)[3]	236 (131)	567 (586)	647 (997)	324 (498)	452 (720)	814 (959)	280 (347)	509 (62)	731 (926)
E. coli (#/100 ml) [3]	786 (964)	525 (1059)	604 (1019)	100 (92)	735 (1183)	421 (499)	443 (740)	630 (1064)	512 (763)

[1]Growing season May to September; senescent season October to April.
[2]Tibutary locations used include Cobb Creek (Site 25), Lake Creek (Site 20) and Willow Creek (Site 15). Refer to Figure 3 and Table 2 for specific locations.
[3]Note difference in units of expression for total coliform and *E. coli* numbers for ease of presentation.

Table 11. Two-way analysis of variance of tributary water quality data.

Parameter	Main Effects and Interaction ($p \leq X$)[1]		
	Tributary[2]	Season[3]	Tributary x Location
Discharge	0.0005	0.0106	0.9754
TP Load	0.0023	0.0712	0.7421
TN Load	0.0005	0.0165	0.8465
N:P ratio	0.0093	0.0001	0.3487
POC	0.5785	0.0262	0.8037
Chl *a*	0.9457	0.0282	0.9806
TP	0.3496	0.0105	0.7051
SRP	0.1296	0.2183	0.8937
TN	0.0001	0.0391	0.3762
NH_3	0.5286	0.1640	0.7144
NO_3NO_2	0.0188	0.0488	0.4545
Turbidity	0.1051	0.5524	0.7489
Dissolved oxygen	0.1270	0.0001	0.1946
Temperature	0.7929	0.0001	0.9235
pH	0.4134	0.2143	0.5696
Conductivity	0.0450	0.6550	0.9687
Hardness	0.0001	0.0030	0.8470
Alkalinity	0.4444	0.0245	0.8811
Total coliform bacteria	0.6076	0.9029	0.8794
E. coli	0.8338	0.7433	0.0921

[1]Associated probability significant if $p \leq 0.05$ (shaded).
[2]Tibutary locations used include Cobb Creek (Site 25), Lake Creek (Site 20) and Willow Creek (Site 15). Refer to Figure 3 and Table 2 for specific locations.
[3]Growing season May to September; senescent season October to April.

Table 12. September 18, 2000 concentrations of elements in Fort Cobb surface water determined by semi-quantitative scan. Units expressed as ng/ml unless otherwise specified.

	Tributary and Site							
	Willow Creek		**Lake Creek**		**Cobb Creek**			
	13	**15**	**18**	**20**	**21**	**22**	**25**	**26**
Ag	0.42	0.19	0.24	0.21	< 0.1	0.14	0.8	0.31
Al	31	39	43	58	27	37	27	33
As	8.1	8.7	9.4	9.5	4.8	2.3	5.9	3.9
Au	0.53	< 0.1	0.16	< 0.1	< 0.1	< 0.1	0.45	< 0.1
Ba	170	160	150	150	150	140	140	130
Be	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Ca[1]	52	59	54	47	110	60	110	63
Cd	< 0.1	0.1	< 0.1	3.8	< 0.1	< 0.1	< 0.1	< 0.1
Ce	< 0.1	< 0.1	< 0.1	0.11	< 0.1	< 0.1	< 0.1	< 0.1
Co	0.11	0.12	0.22	0.25	< 0.1	0.11	0.14	0.1
Cr	1.5	1.6	2.1	1.9	1.3	3.1	1.6	1.7
Cu	< 1	< 1	< 1	2.4	< 1	< 1	< 1	< 1
Fe	< 10	34	< 10	25	< 10	< 10	< 10	33
Ga	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Ge	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
In	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
K[1]	0.82	1	1.4	1.8	1.1	0.61	1.1	1.7
La	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Li	16	18	9.6	9.8	10	12	12	12
Mg[1]	9.8	11	20	20	24	13	24	19
Mn	25	11	17	6.5	26	6.9	37	33
Mo	0.85	0.96	1.2	1.3	0.82	0.56	1.7	1.5
Na[1]	16	19	27	31	26	29	26	24
Ni	< 1	< 1	< 1	2.6	< 1	< 1	< 1	< 1
Pb	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Pd	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Pt	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Rb	0.45	0.49	0.54	0.73	0.49	0.32	0.51	0.76
Ru	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Sb	0.17	0.12	0.14	0.24	< 0.1	< 0.1	0.19	0.13
Se	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Sn	1.1	0.19	0.37	0.93	< 0.1	0.81	< 0.1	< 0.1
Sr	340	400	520	520	3600	260	2800	610
Ta	0.13	0.11	< 0.1	< 0.1	< 0.1	< 0.1	0.2	0.13
Ti	2.1	1.8	2.6	2.5	1.7	1.8	1.7	1.8
Tl	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
U	1.4	1.4	2.2	2.6	2.4	1.5	2.2	1.8
V	6.9	7.6	6.1	10	5.8	5.3	6.1	3.4
Zn	13	10	6.3	28	< 1	16	2.9	15

[1]Concentration units µg/ml. For site locations refer to Figure 3 and Table 2.

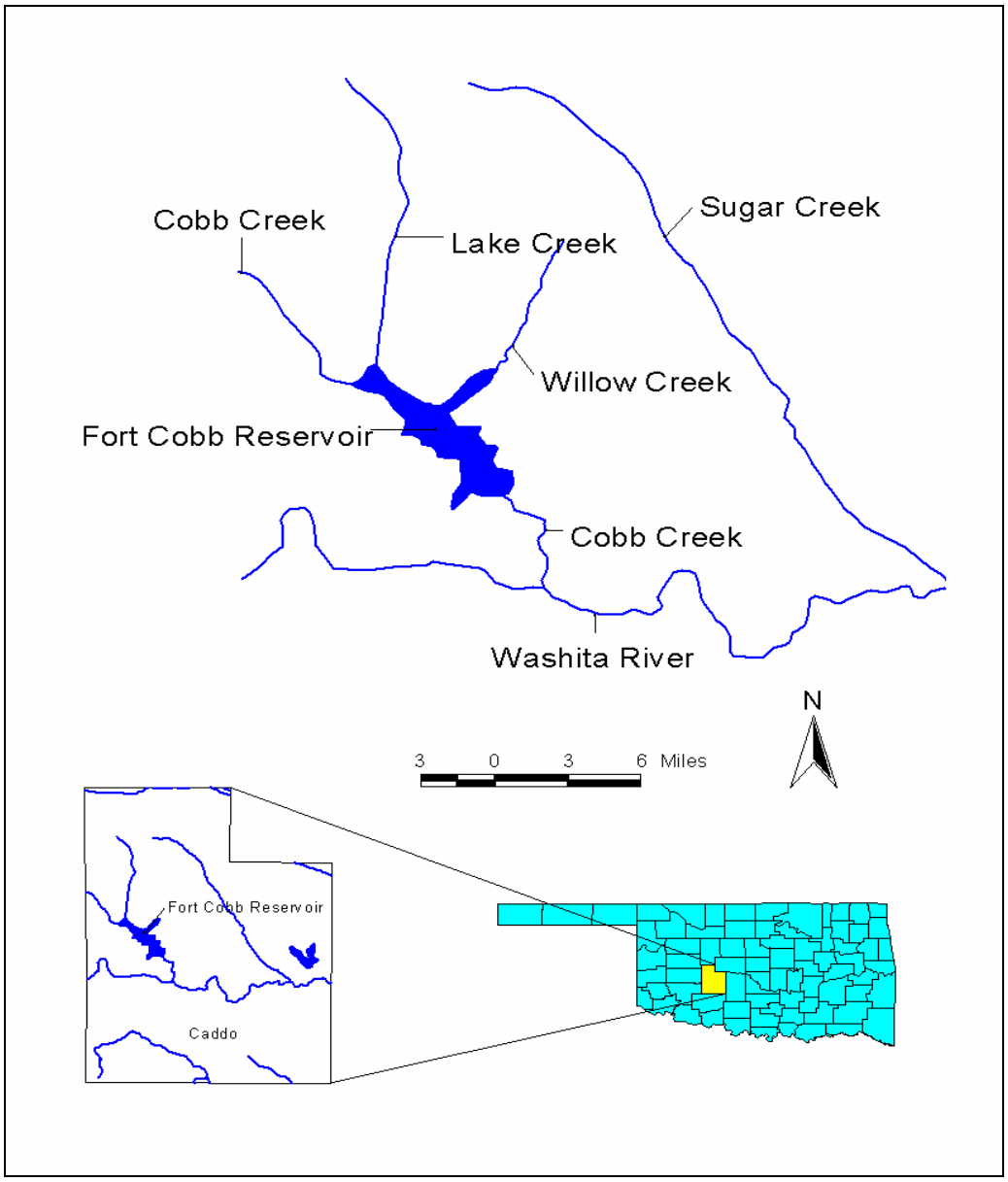


Figure 1. Map of Fort Cobb Watershed located in Caddo County, OK.

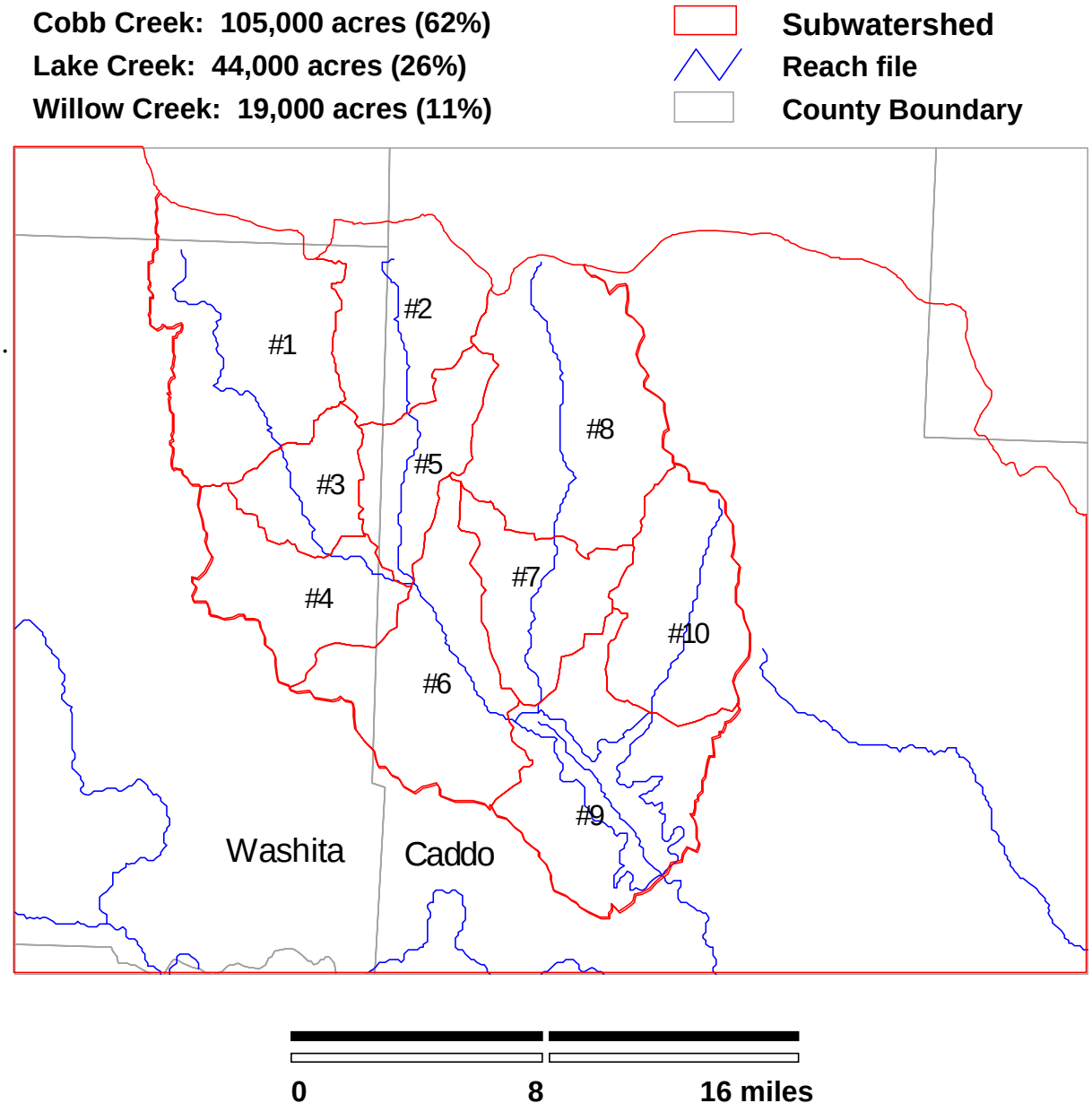


Figure 2. Map of Fort Cobb Watershed divided into sub-basins. Map courtesy of Paul Yue, Oklahoma DEQ.

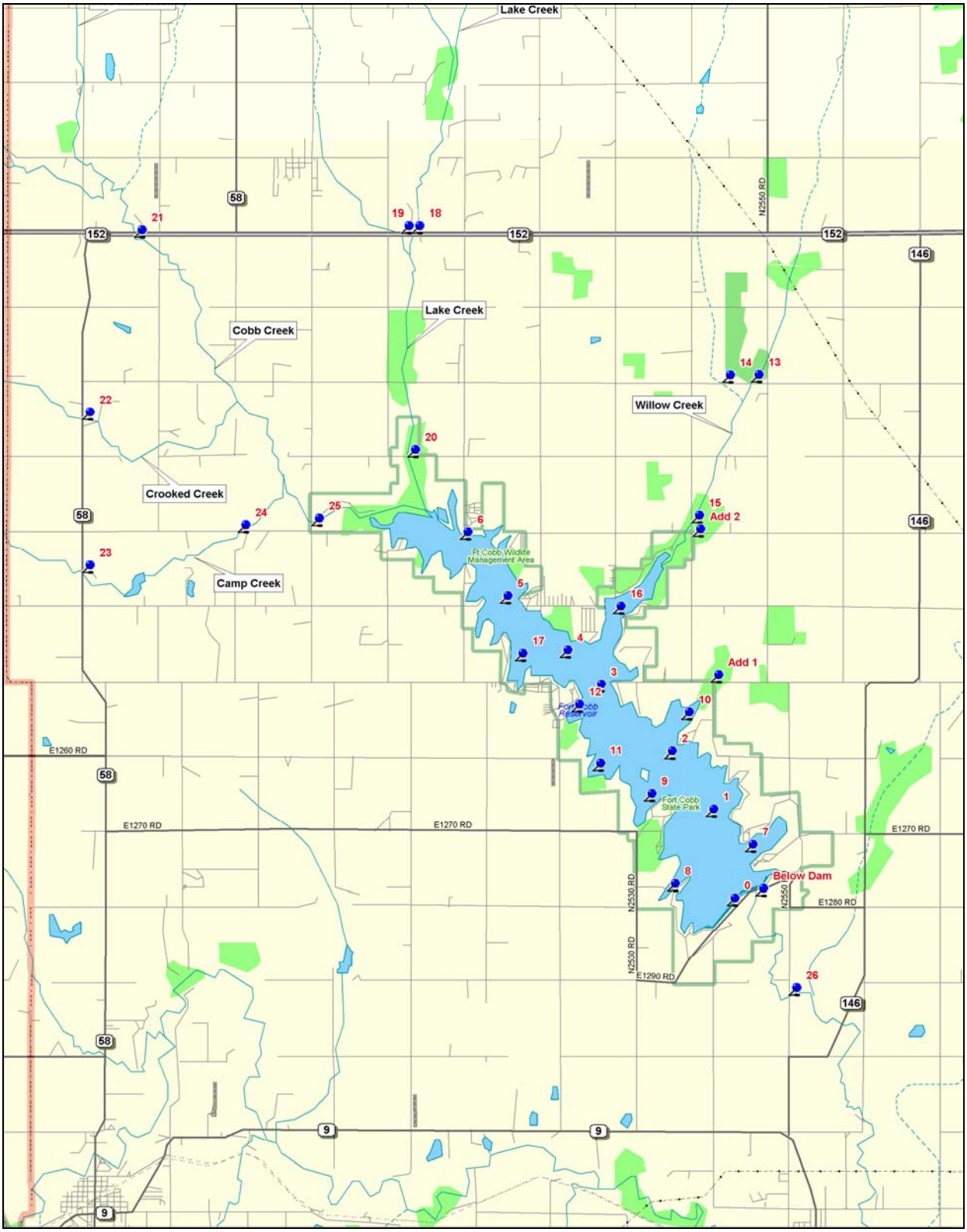


Figure 3. Localized map of field sampling locations/sites used in studies at Fort Cobb Reservoir.

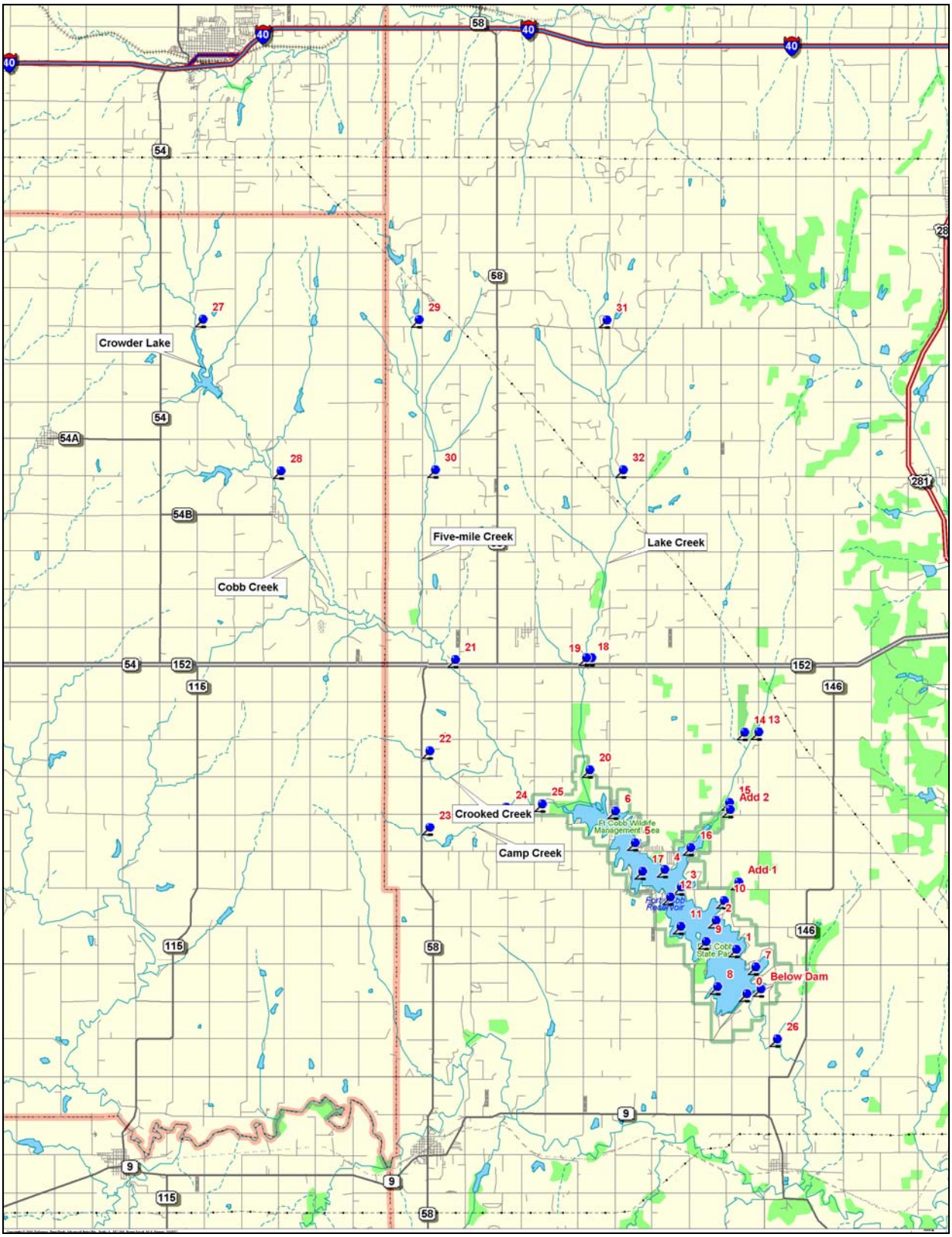


Figure 4. Expanded map of field sampling locations/sites used in studies at Fort Cobb Reservoir.

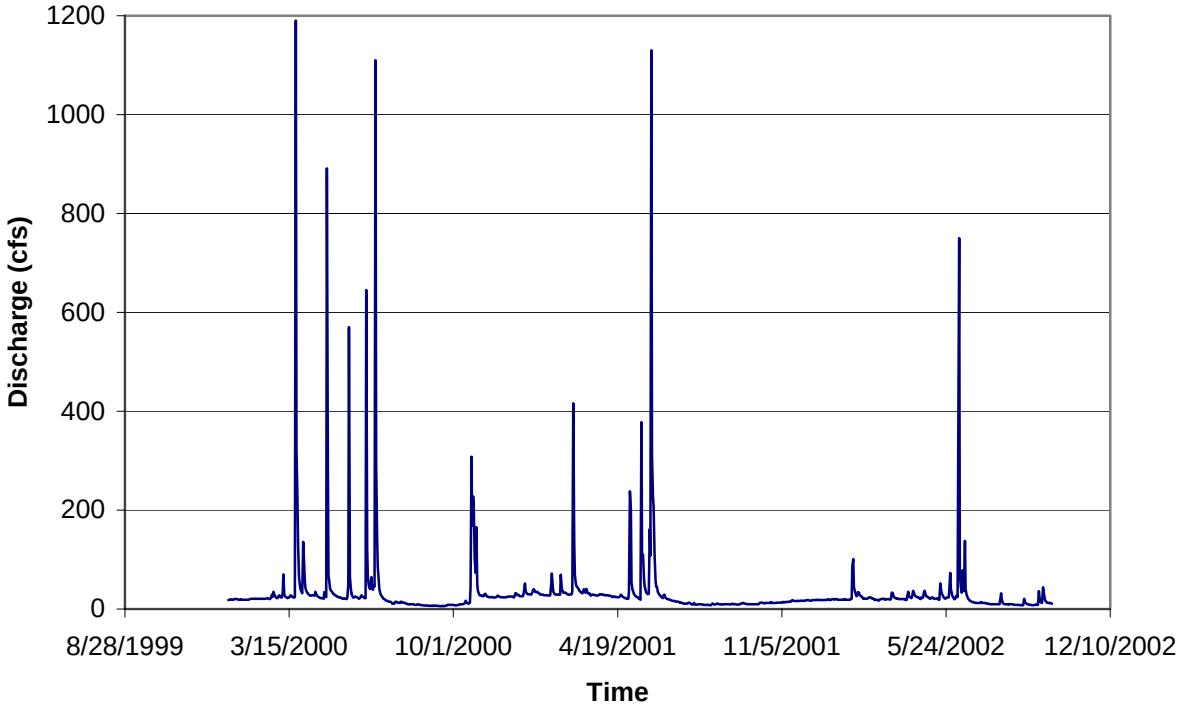


Figure 5. USGS daily discharge records for Cobb Creek (Station 07325800). Available at: http://nwis.waterdata.usgs.gov/usa/nwis/discharge

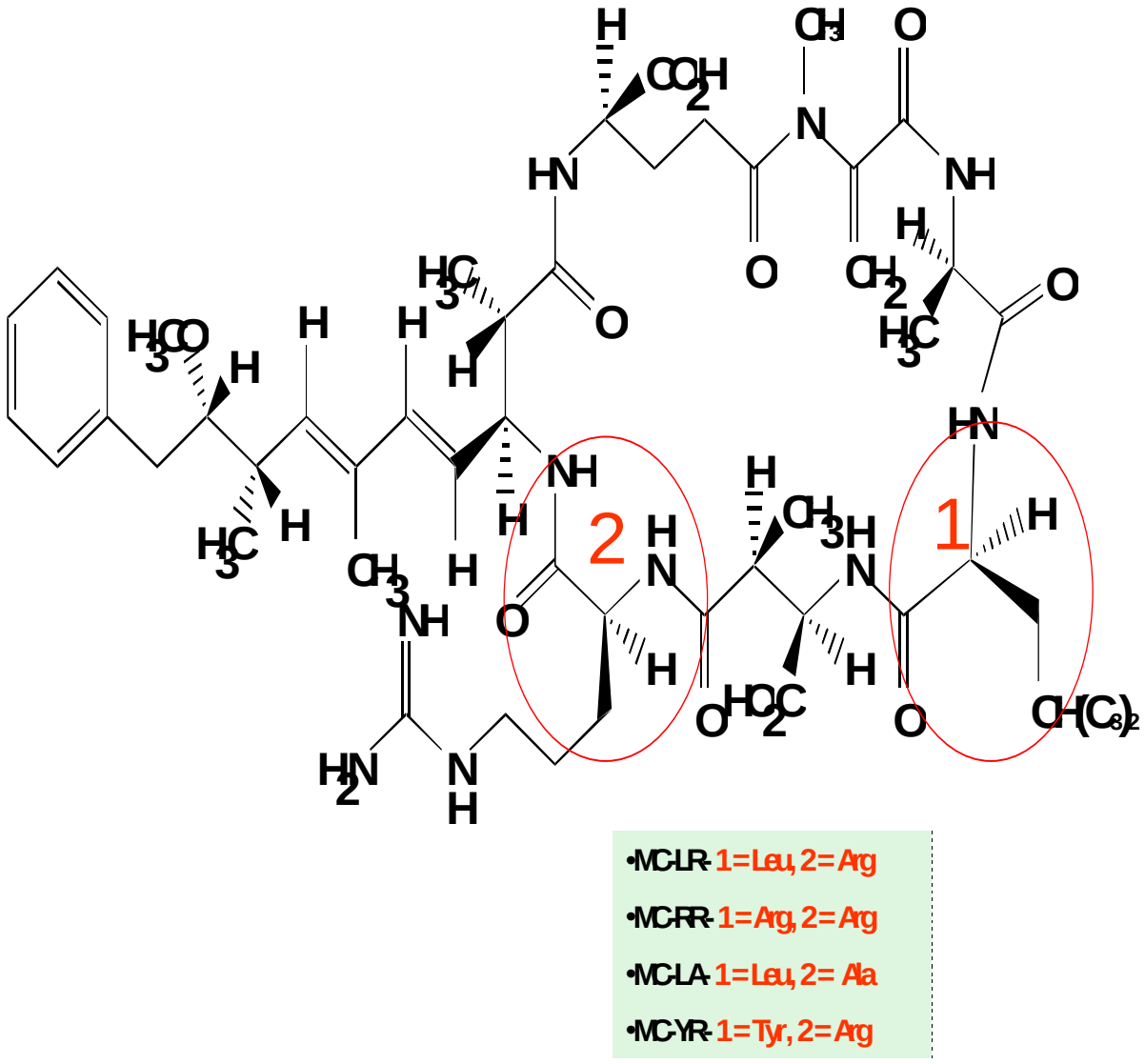
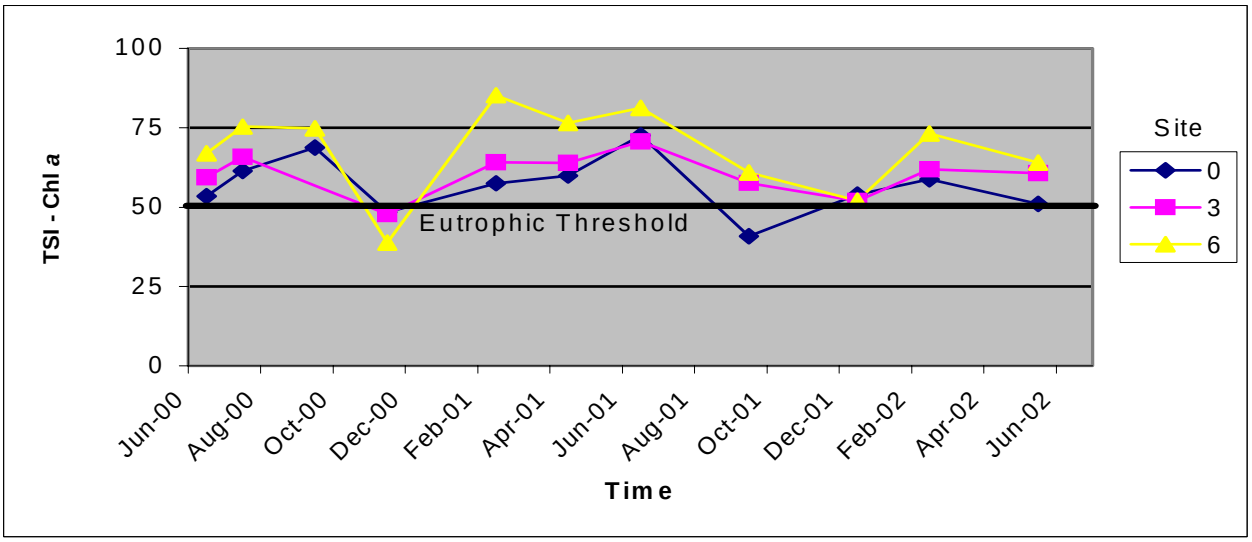
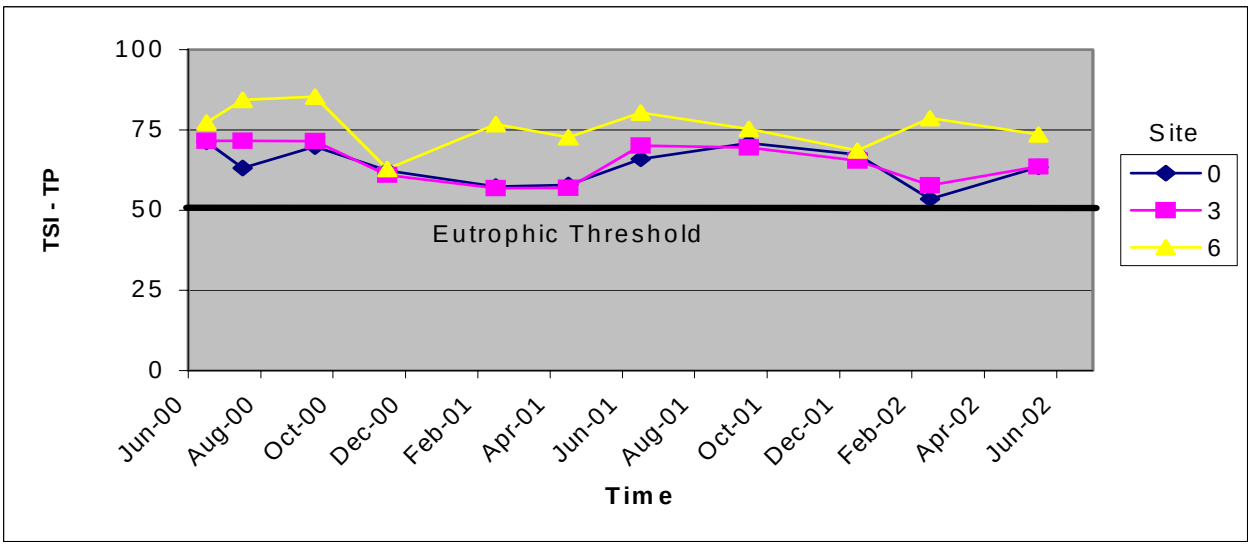


Figure 6. Structure of four analogs of microcystin. The red circles illustrate the significant chemical moities that control toxicity.





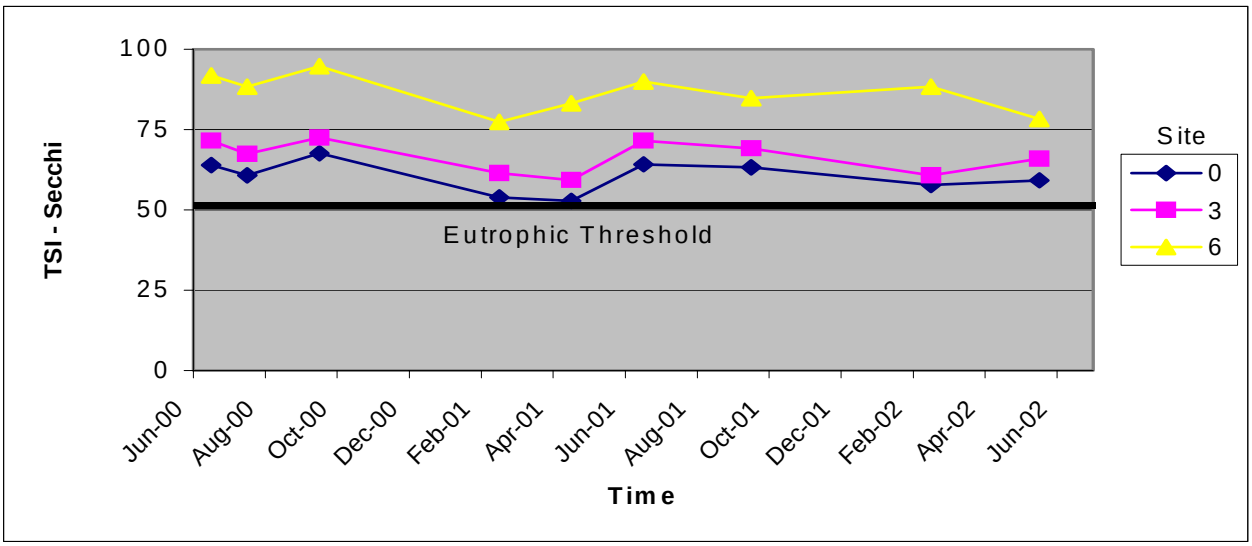


Figure 7. Spatial and temporal trends in Trophic State Indices for Fort Cobb Reservoir.

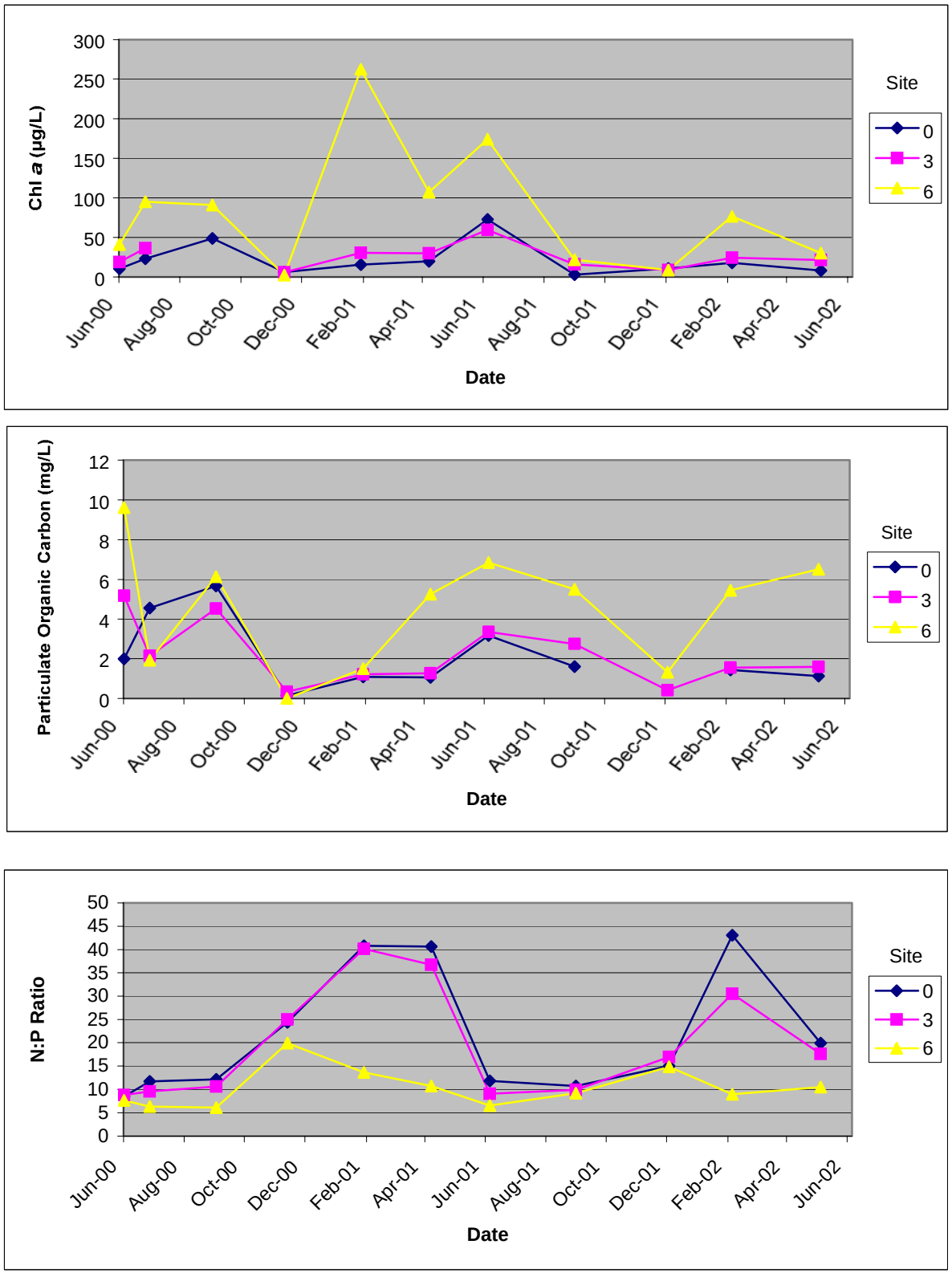
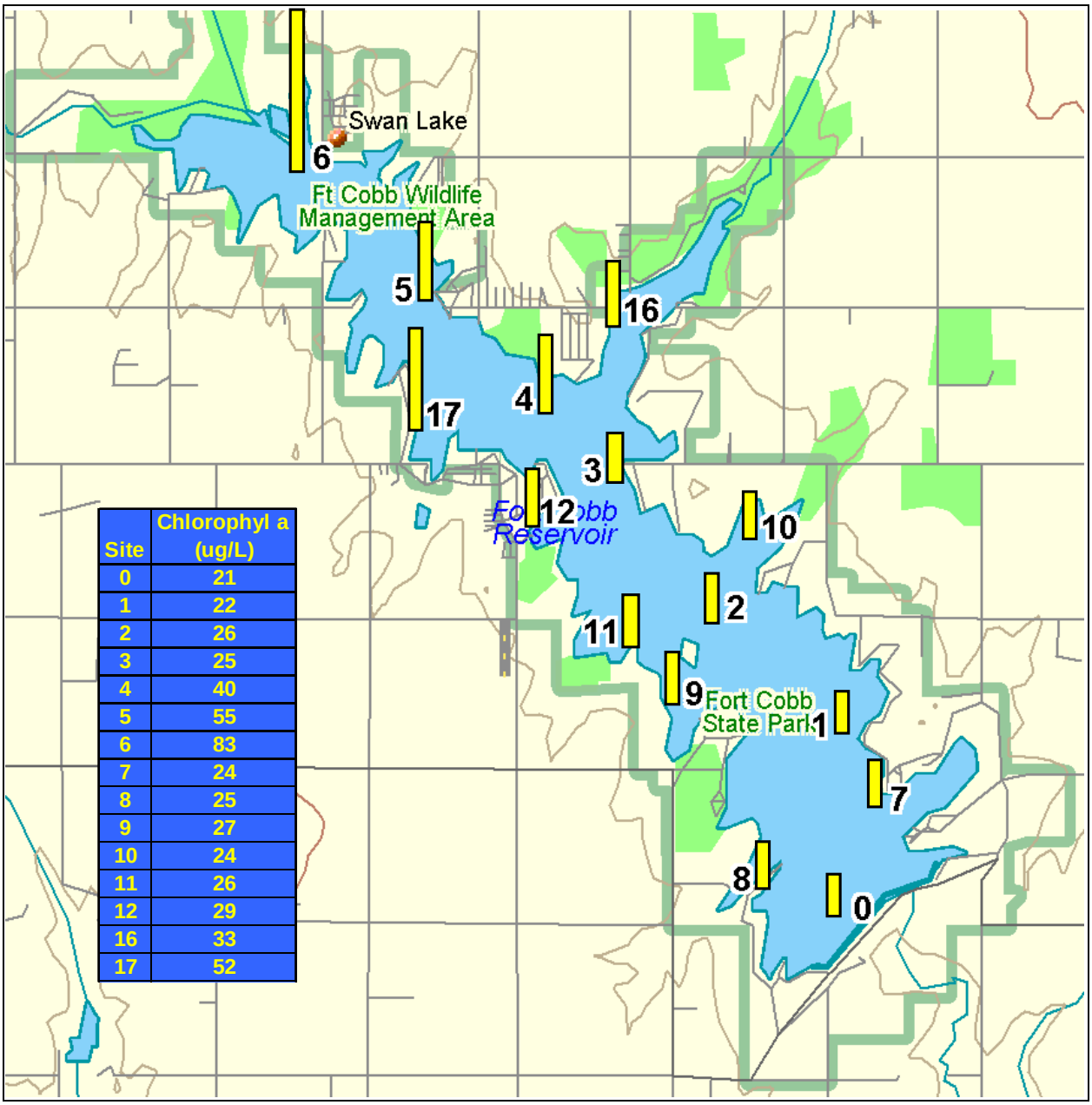
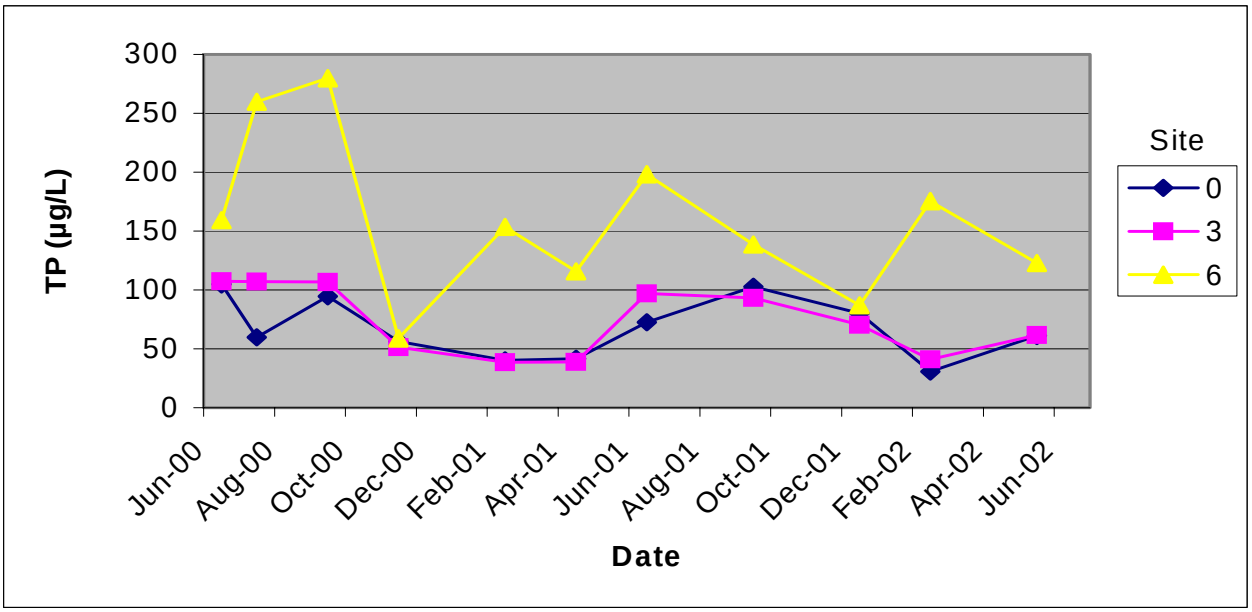


Figure 8. Spatial and temporal trends in chlorophyll *a*, particulate organic carbon, and N:P ratio in Fort Cobb Reservoir.



Site	Chlorophyl a (ug/L)
0	21
1	22
2	26
3	25
4	40
5	55
6	83
7	24
8	25
9	27
10	24
11	26
12	29
16	33
17	52

Figure 9. Average annual concentrations of chlorophyll at 17 reservoir stations in Fort Cobb Reservoir over the course of the 2-yr study. Numbers on the figure represent sites as described in Table 2. Height of the bars indicate relative chlorophyll concentrations shown in legend.



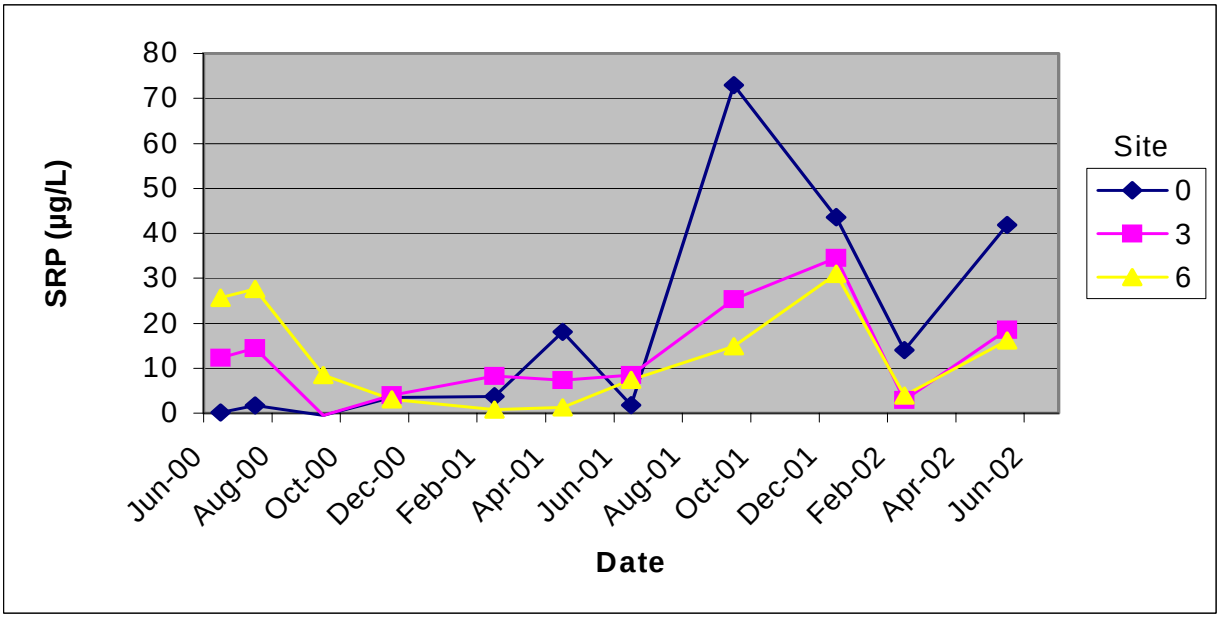
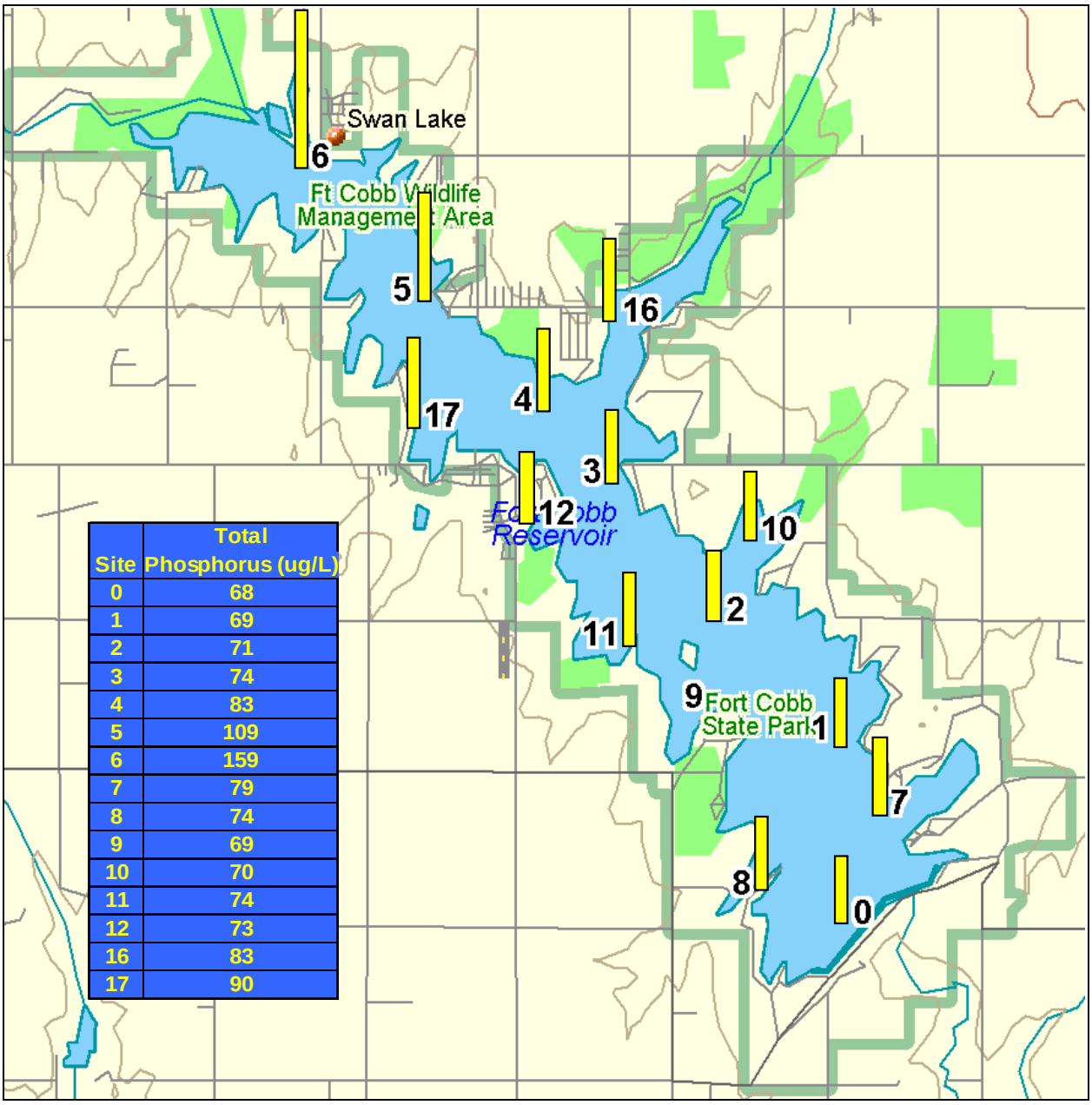


Figure 10. Spatial and temporal trends in total and soluble reactive phosphorus in Fort Cobb Reservoir.



Site	Total Phosphorus (ug/L)
0	68
1	69
2	71
3	74
4	83
5	109
6	159
7	79
8	74
9	69
10	70
11	74
12	73
16	83
17	90

Figure 11. Average annual concentrations of total phosphorus at 17 reservoir sites in Fort Cobb Reservoir measured over the course of the 2-yr study. Numbers on the figure represent sites as described in Table 2. Height of the bars indicate relative total phosphorus concentrations shown in legend.

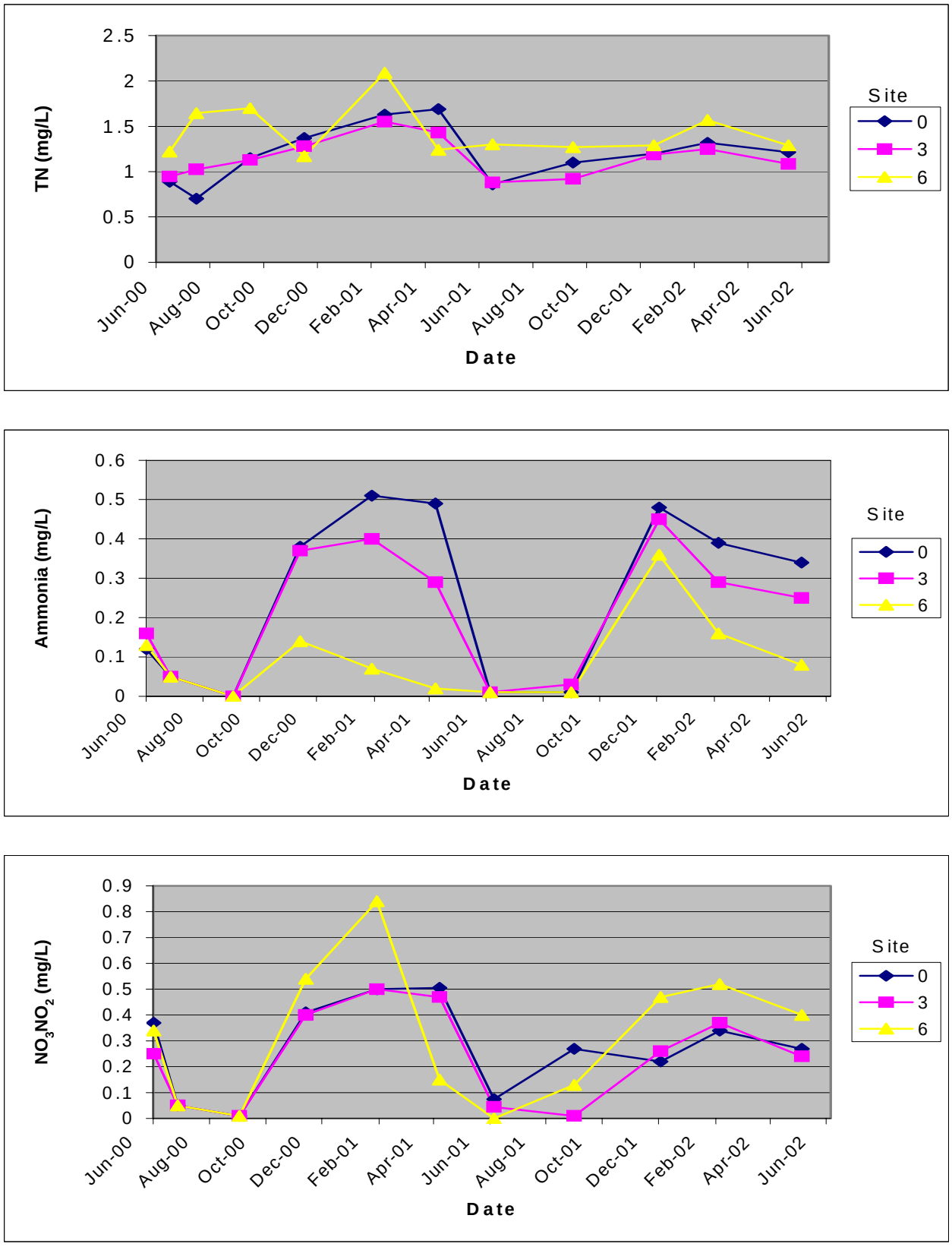
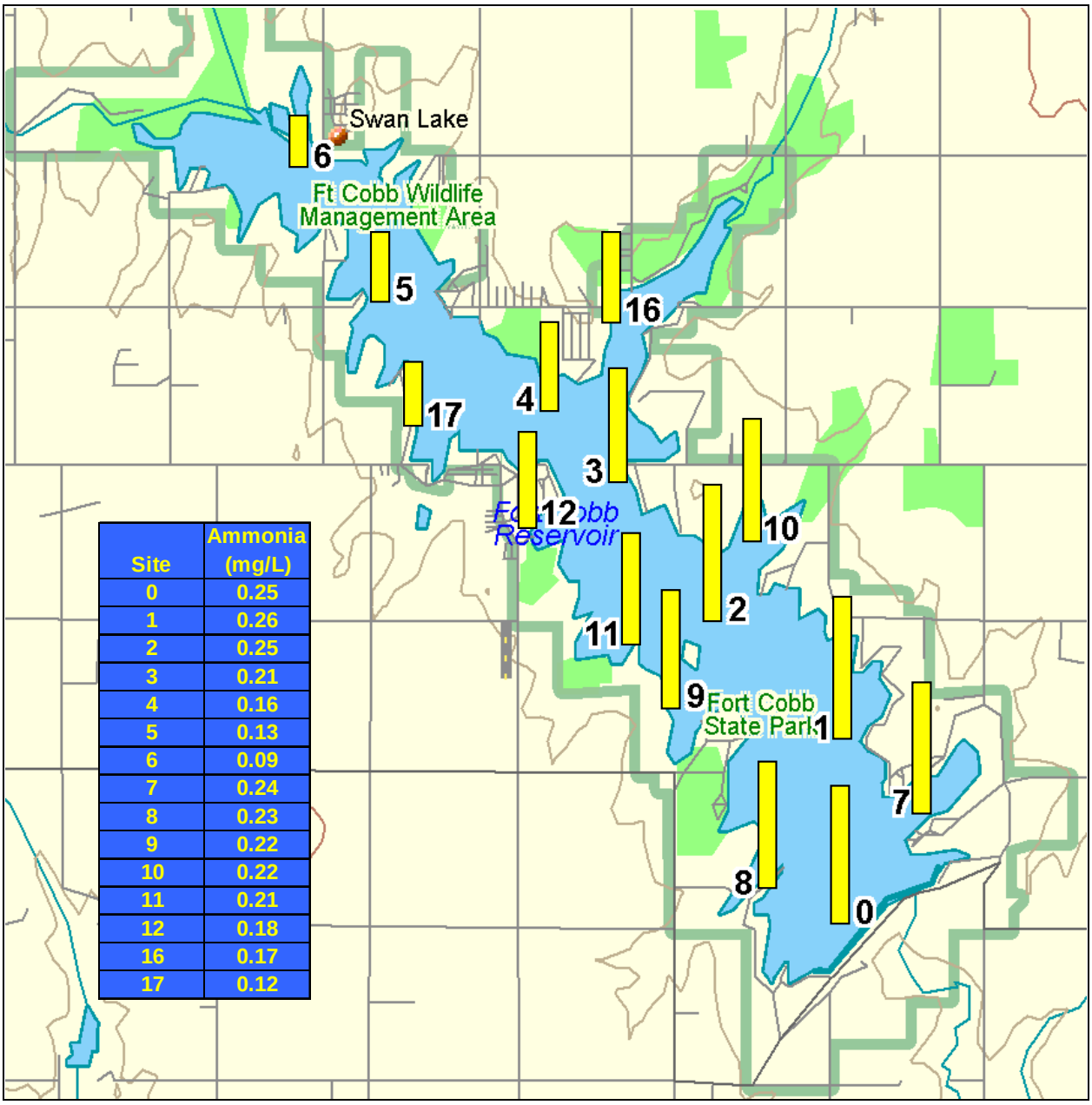
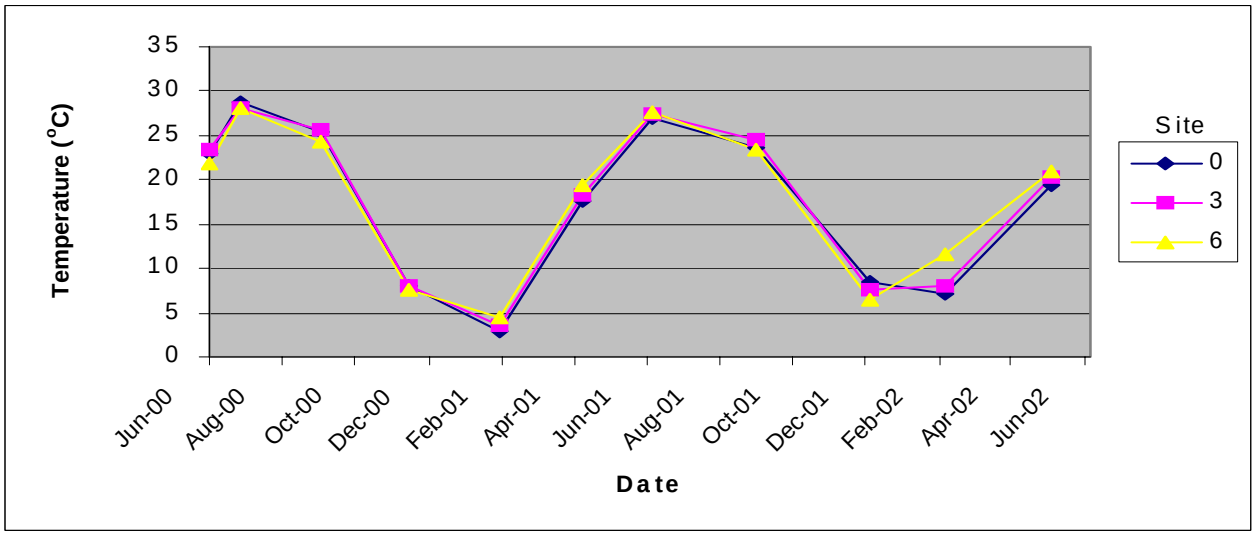


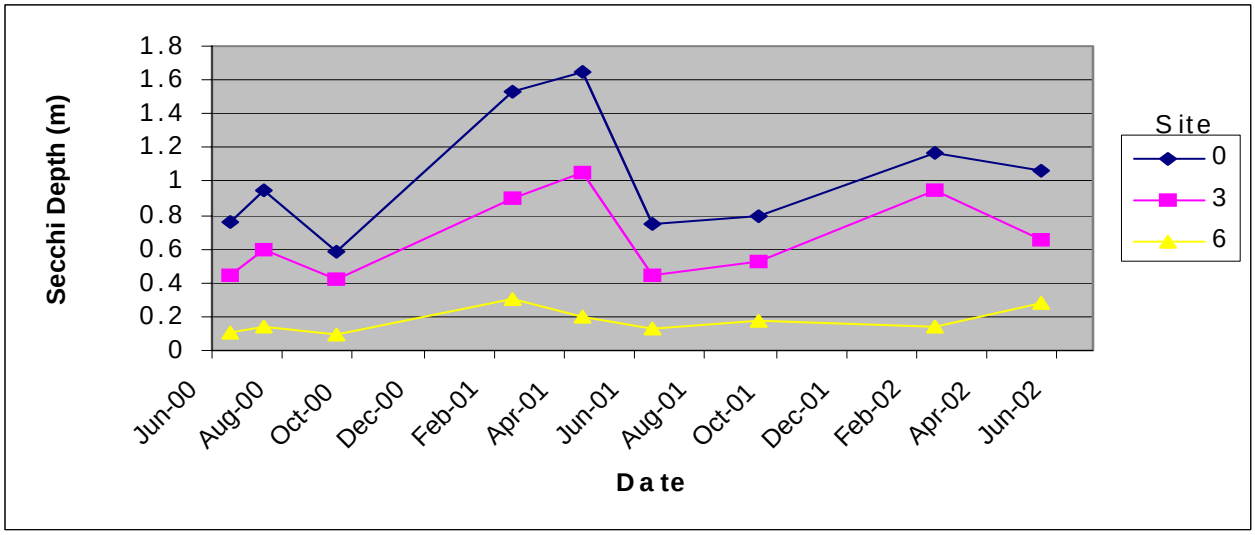
Figure 12. Spatial and temporal trends in total nitrogen, ammonia, and nitrate/nitrite in Fort Cobb Reservoir.



Site	Ammonia (mg/L)
0	0.25
1	0.26
2	0.25
3	0.21
4	0.16
5	0.13
6	0.09
7	0.24
8	0.23
9	0.22
10	0.22
11	0.21
12	0.18
16	0.17
17	0.12

Figure 13. Average annual concentrations of ammonia at 17 reservoir stations in Fort Cobb reservoir measured over the course of the 2-yr study. Numbers on the figure represent sites as described in Table 2. Height of the bars indicate relative ammonia concentrations shown in legend.





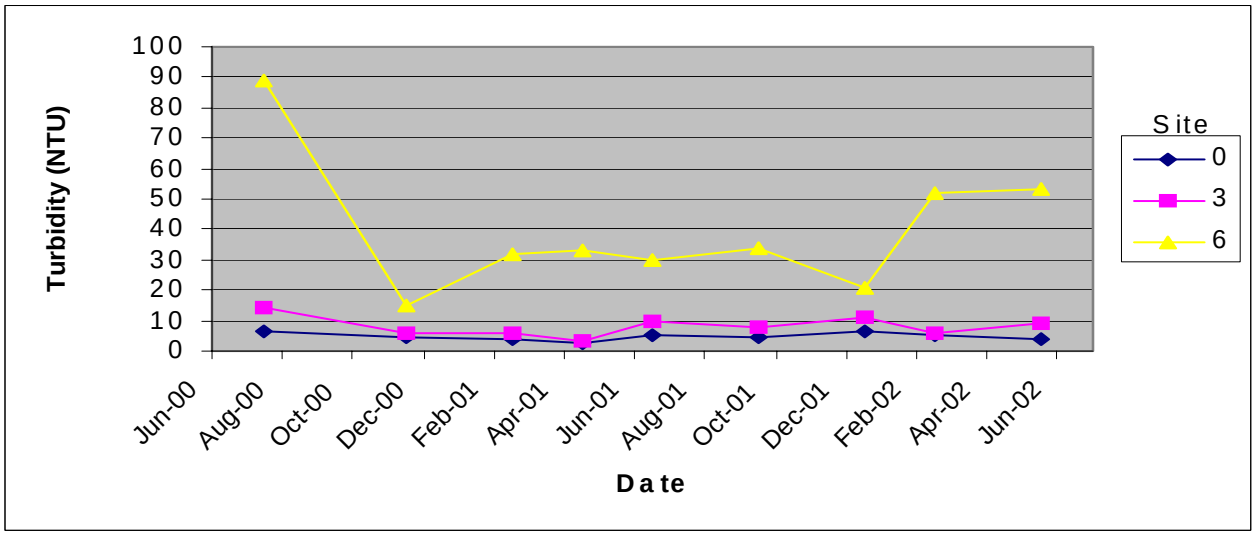


Figure 14. Spatial and temporal trends in temperature, Secchi depth, and turbidity in Fort Cobb Reservoir.

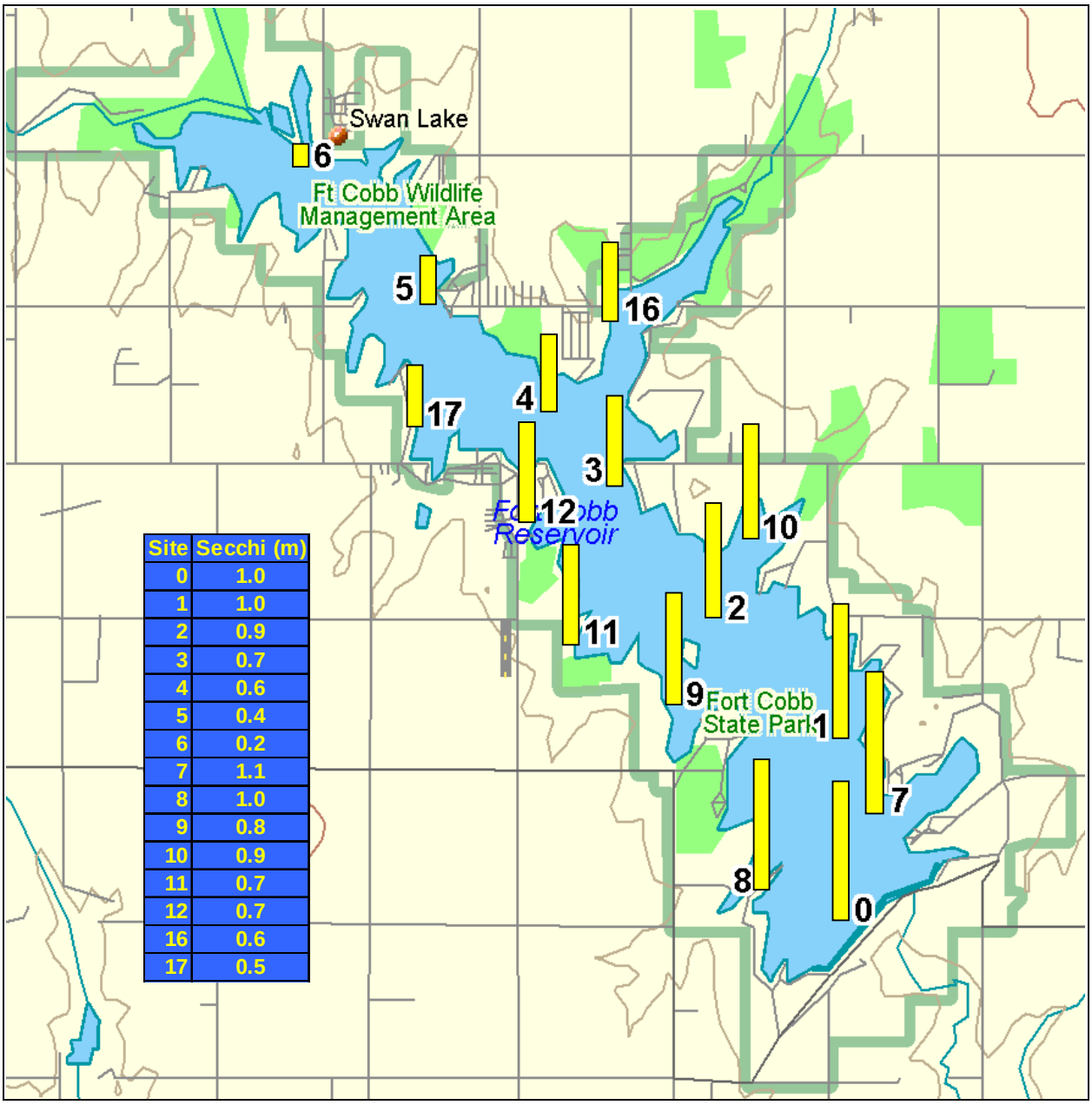
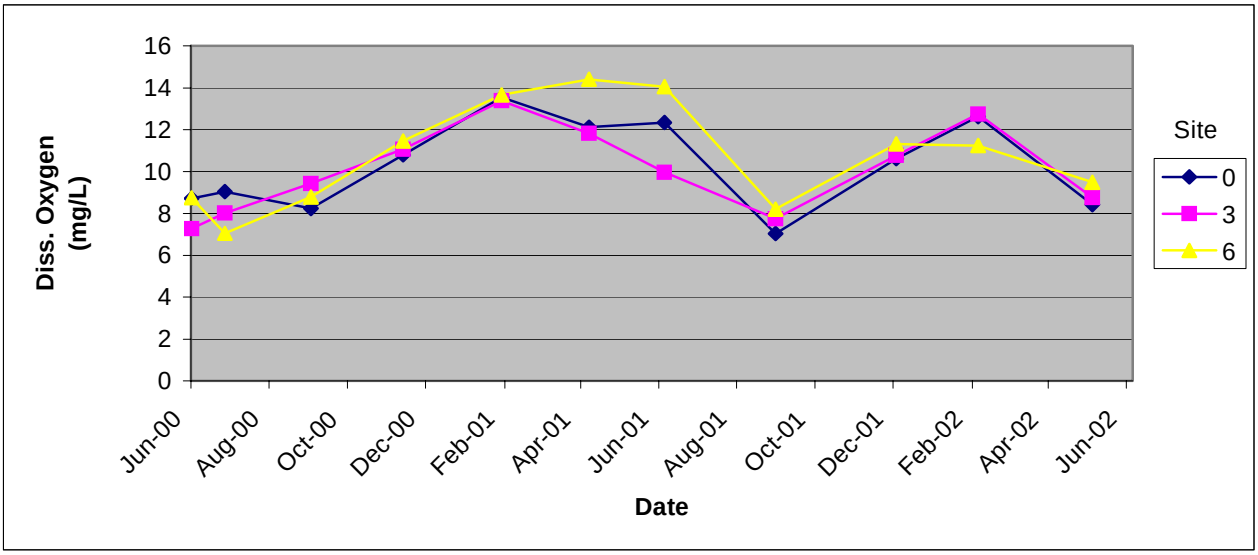
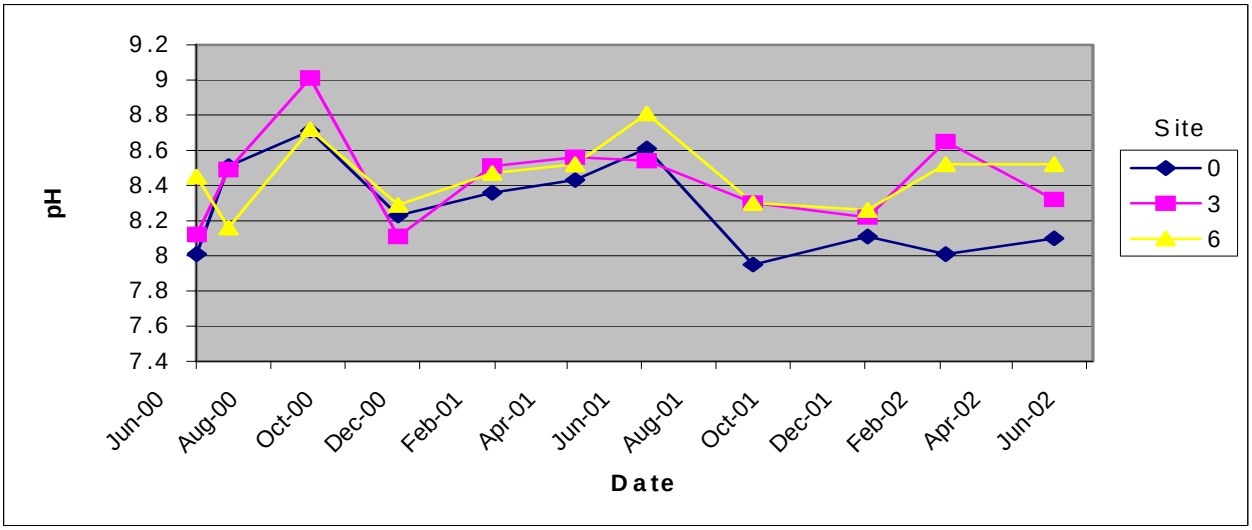


Figure 15. Average annual Secchi transparencies at 17 reservoir stations in Fort Cobb Reservoir measured over the course of the 2-yr study. Numbers on the figure represent sites as described in Table 2. Height of the bars indicate relative Secchi transparencies shown in legend.





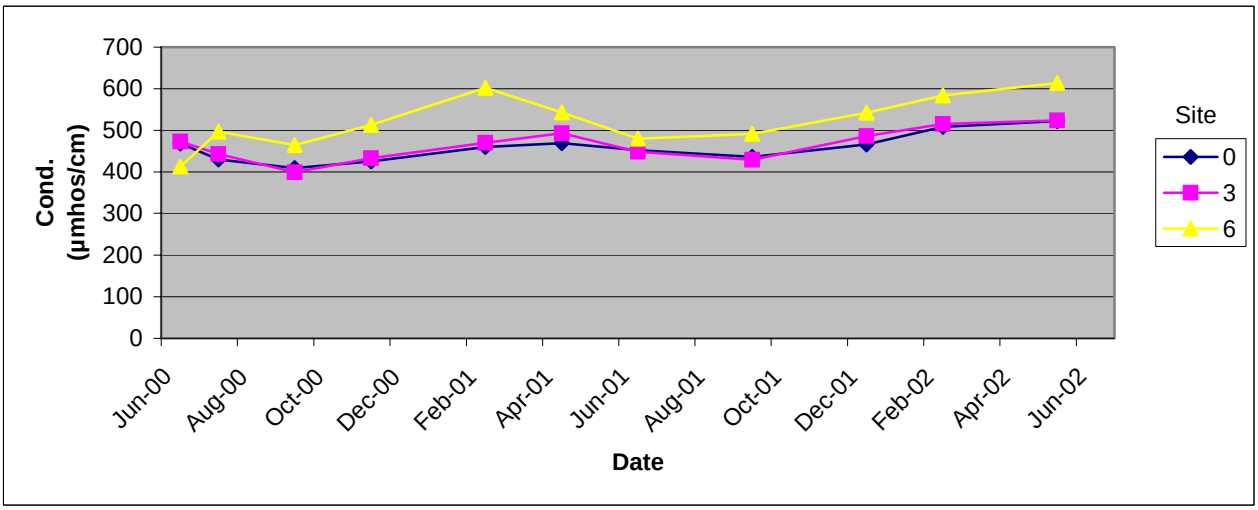
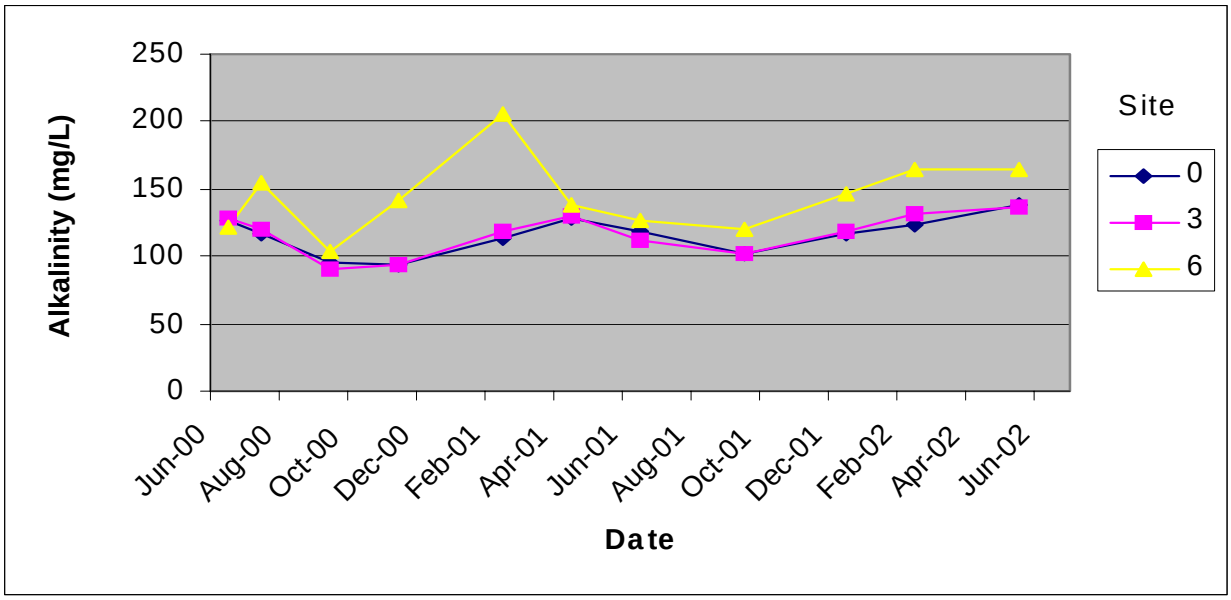


Figure 16. Spatial and temporal trends in dissolved oxygen, pH, and conductivity in Fort Cobb Reservoir.



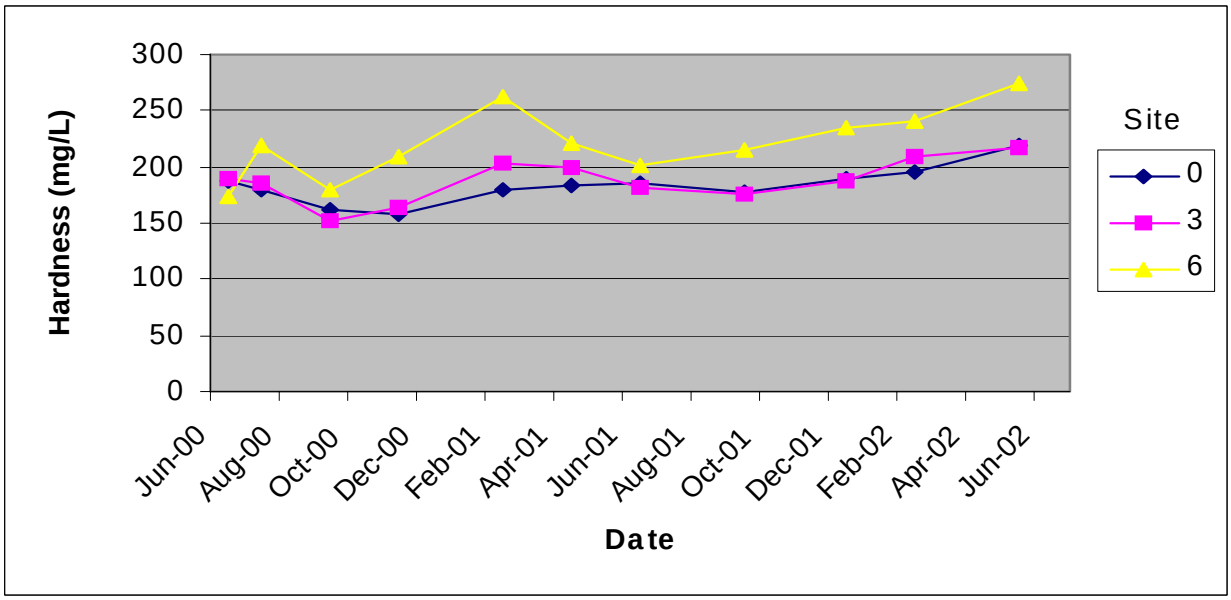


Figure 17. Spatial and temporal trends in alkalinity and hardness in Fort Cobb Reservoir.

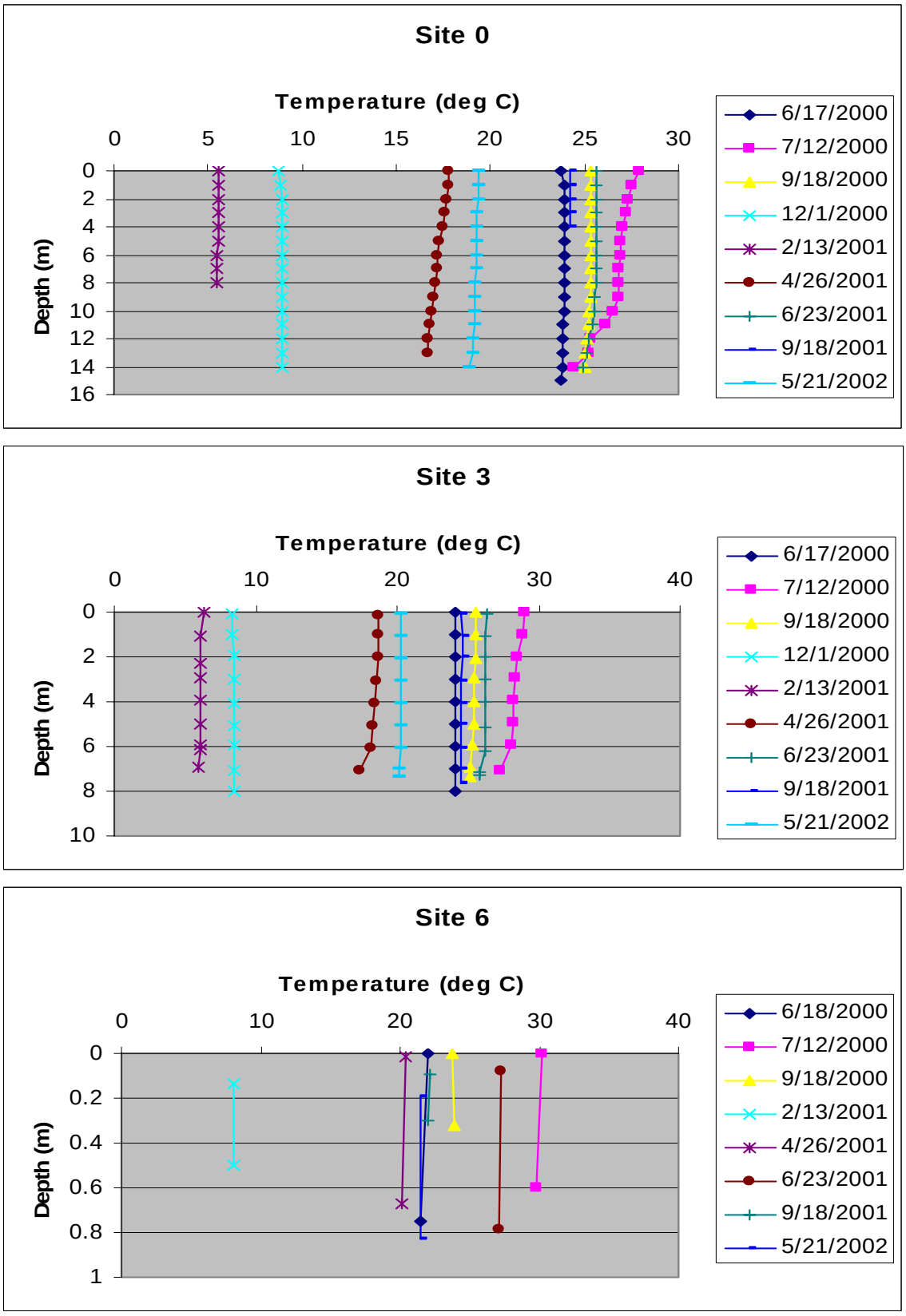


Figure 18. Spatial and temporal profiles of temperature by depth in Fort Cobb Reservoir.

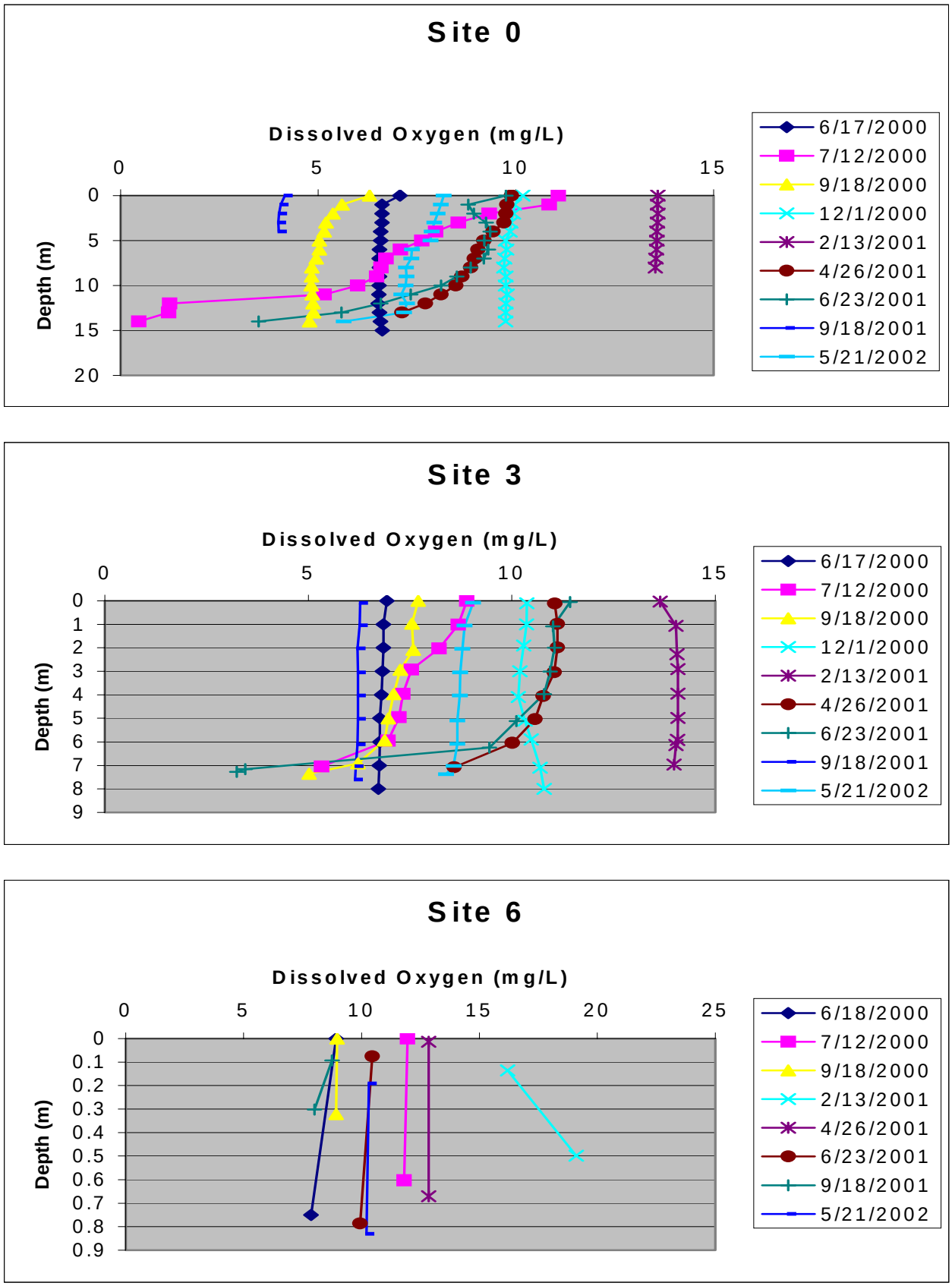
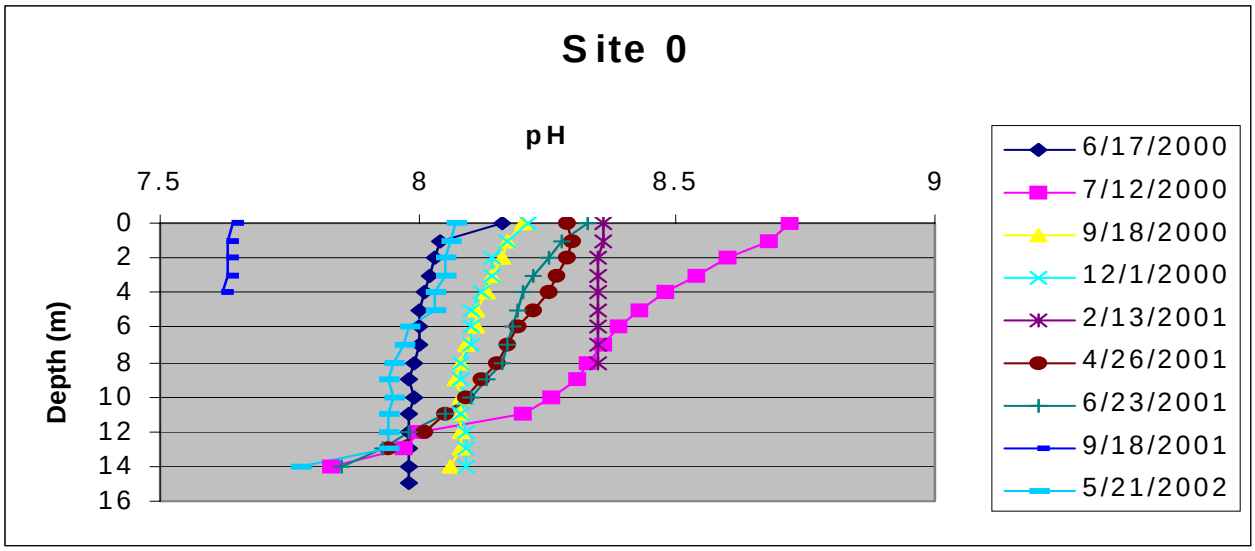
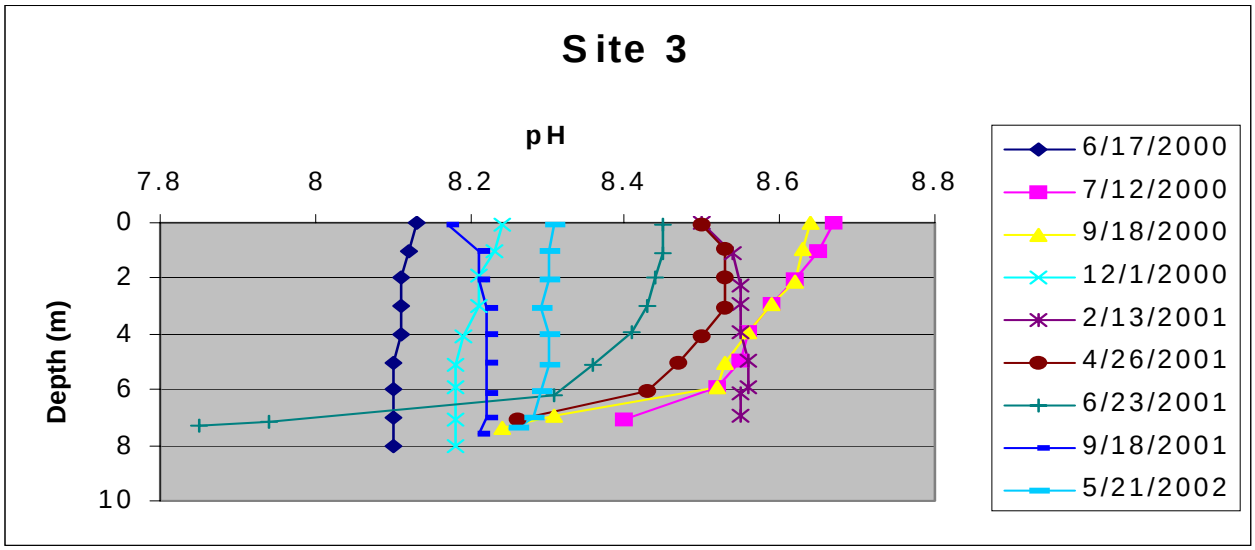


Figure 19. Spatial and temporal trends of dissolved oxygen by depth in Fort Cobb Reservoir.





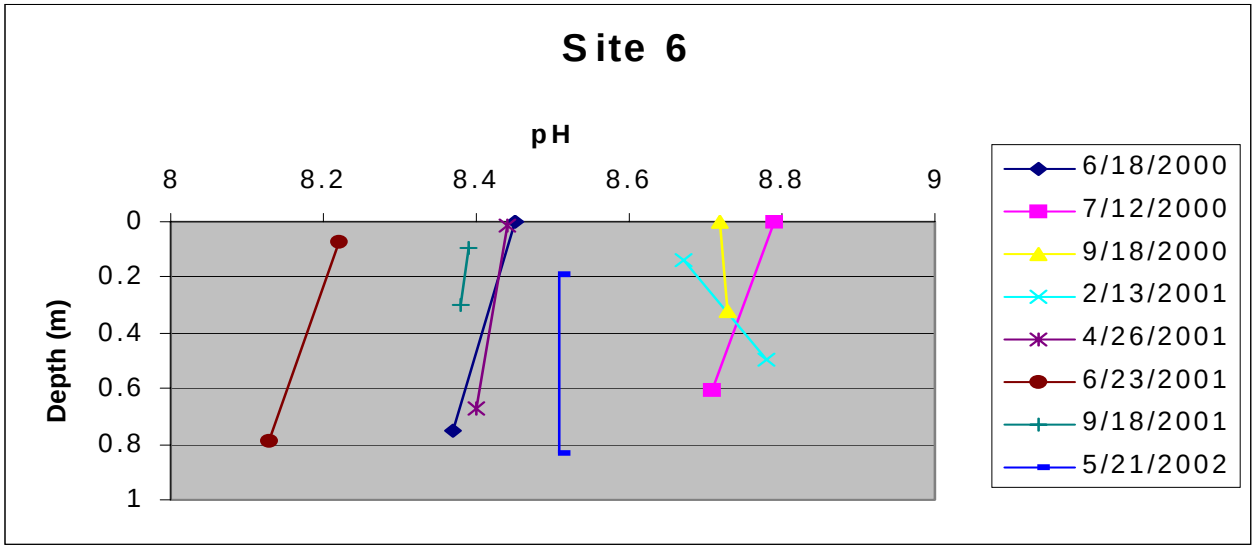


Figure 20. Spatial and temporal trends of pH by depth in Fort Cobb Reservoir.

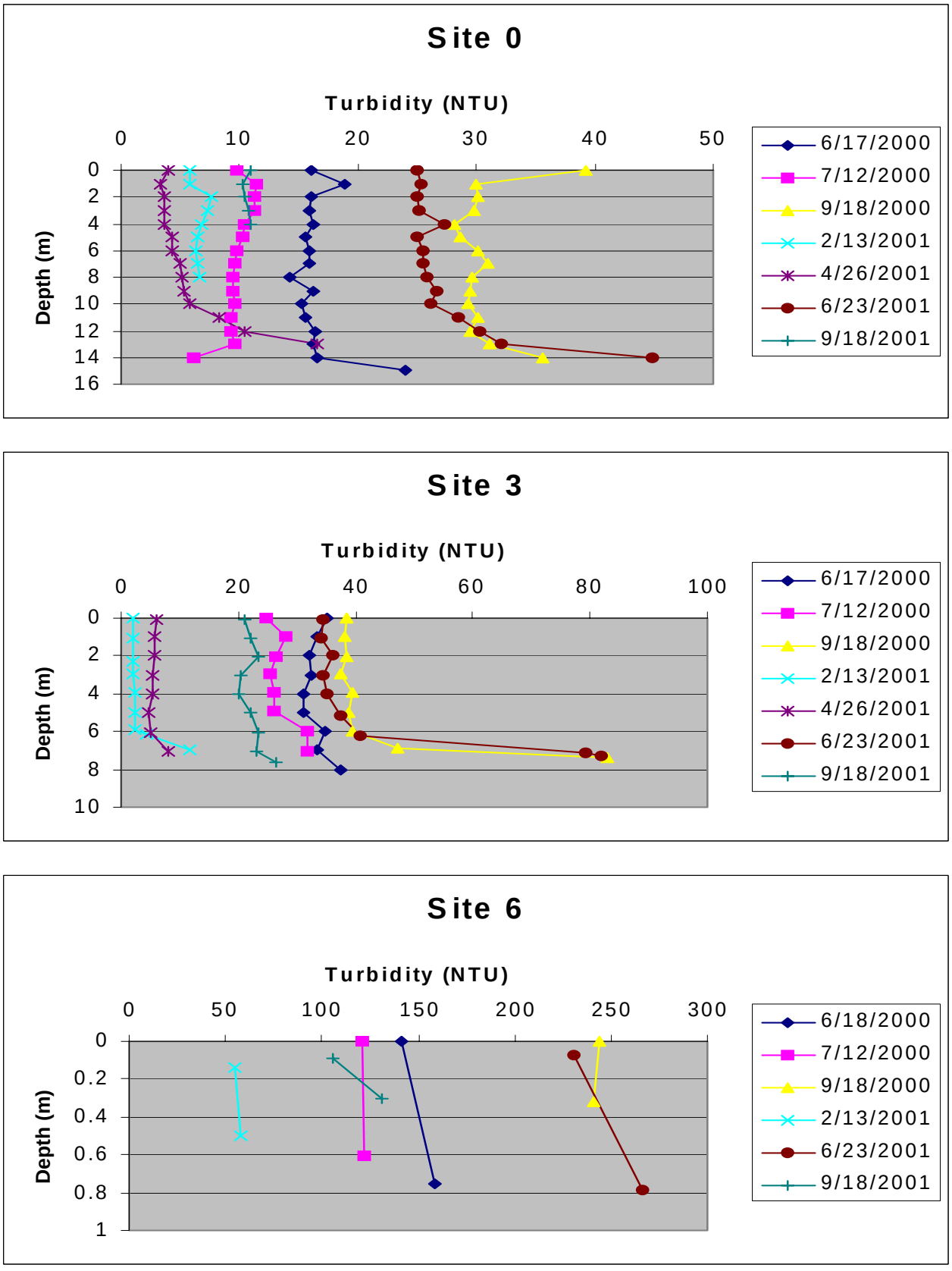


Figure 21. Spatial and temporal trends of turbidity by depth in Fort Cobb Reservoir.

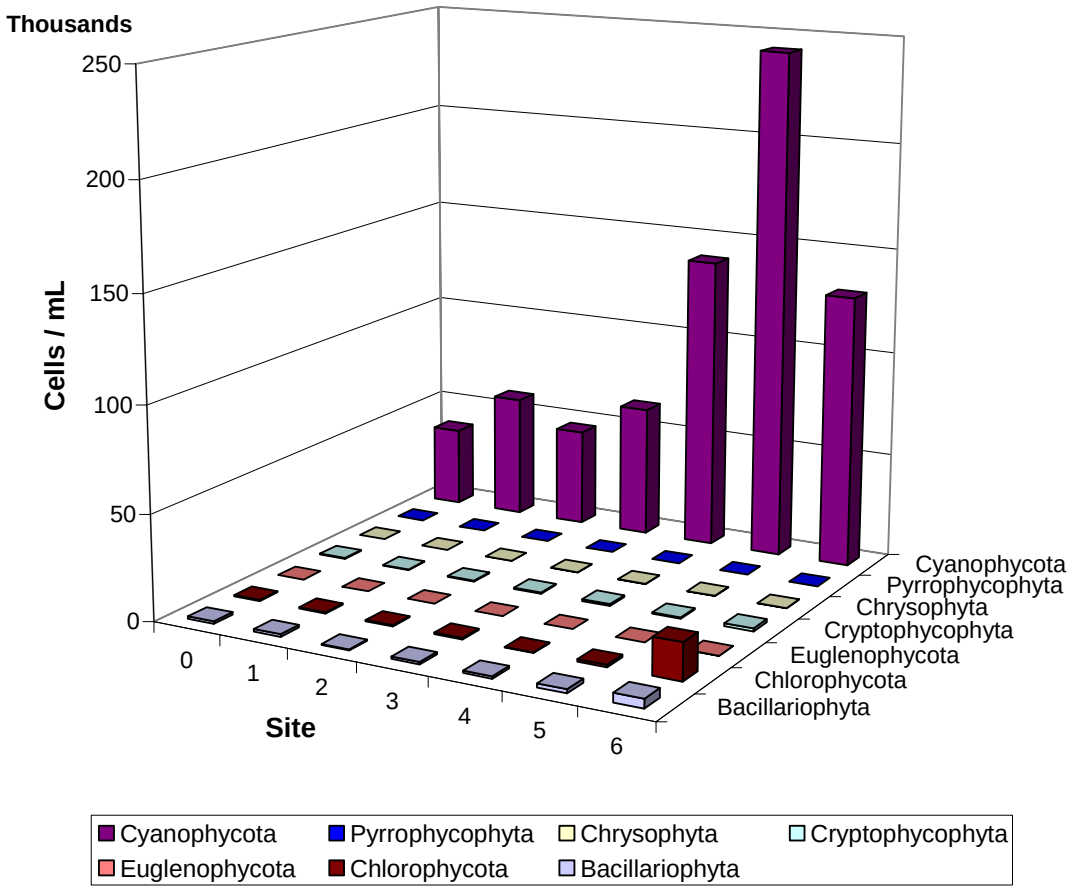


Figure 22. Spatial comparison of numbers of cells of major phyla of phytoplankton averaged over entire study (n=11 dates) at six sites in Fort Cobb Reservoir.

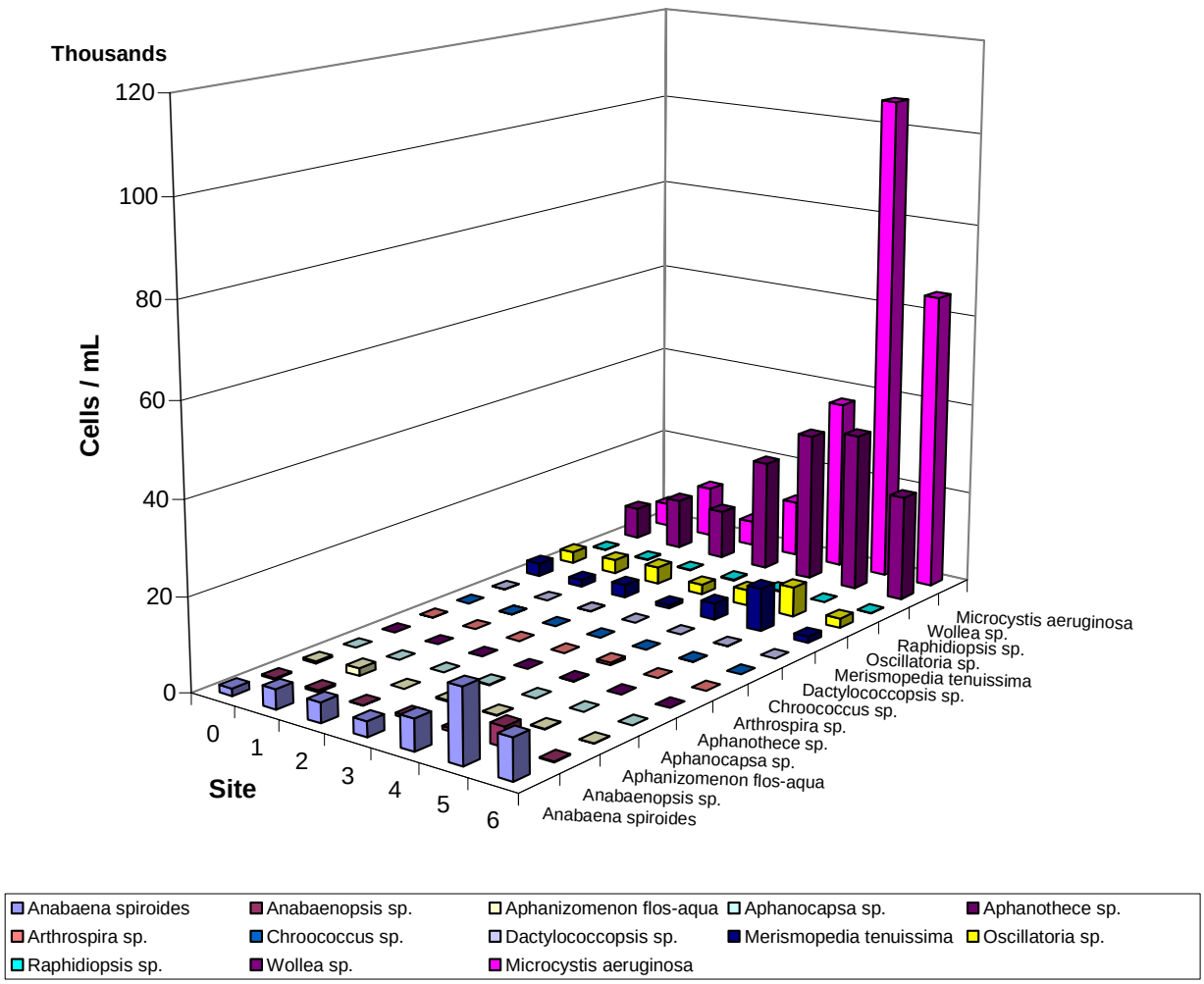


Figure 23. Spatial comparison of the total numbers of cells of cyanobacterial species averaged over entire study (n=11 dates) at six sites in Fort Cobb Reservoir.

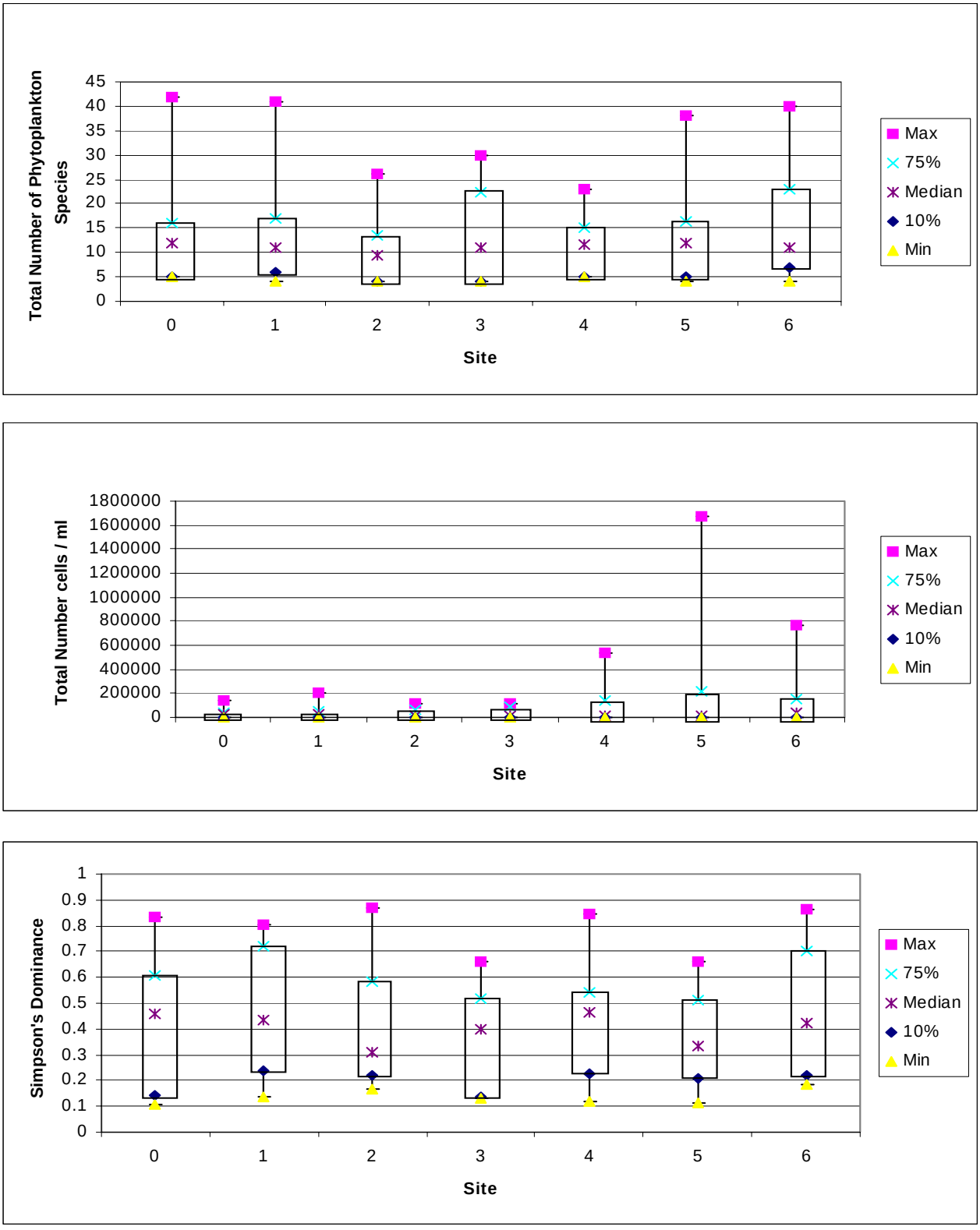


Figure 24. Box plots of algal community structure of phytoplankton averaged over the entire study (n=11 dates) in Fort Cobb Reservoir.

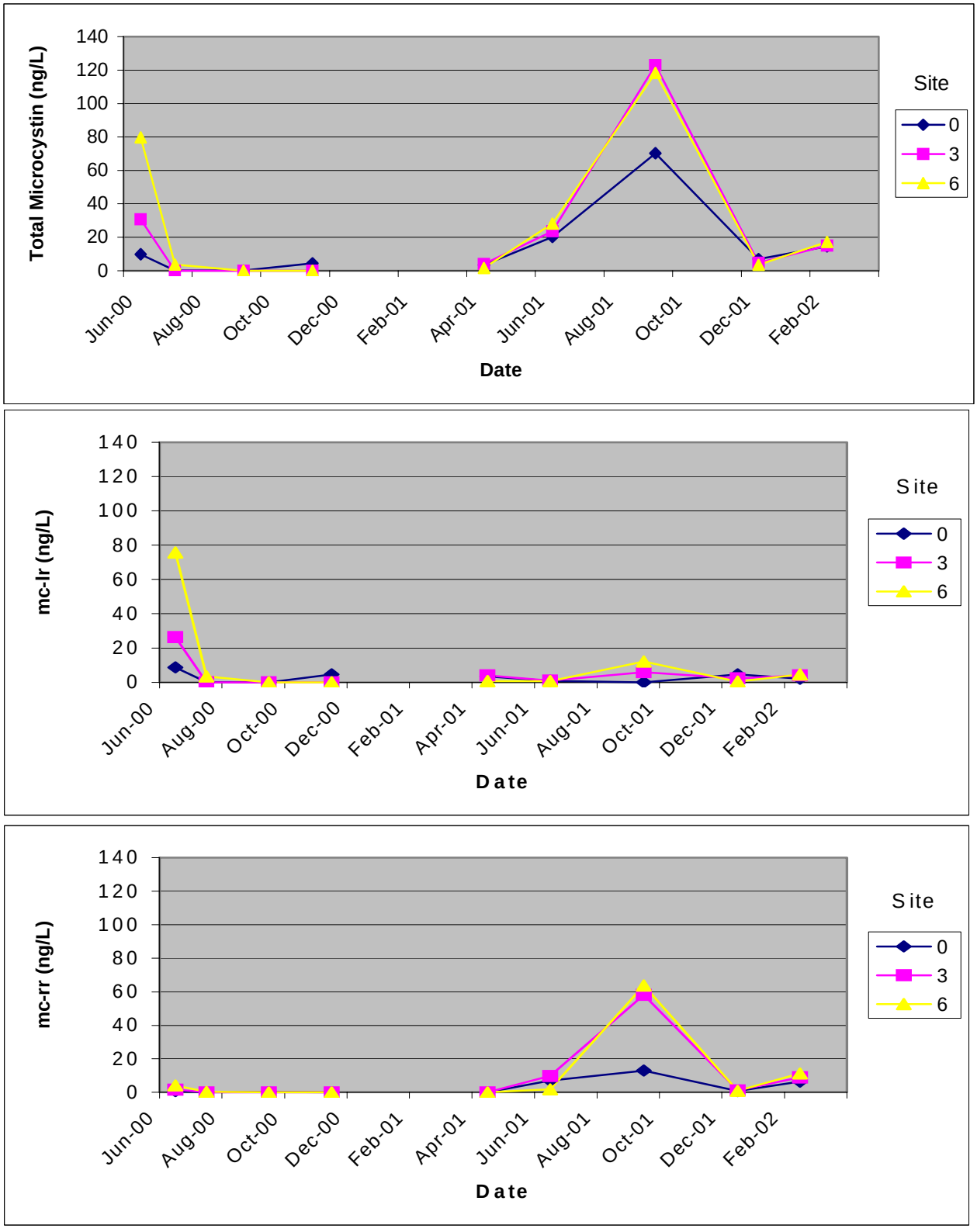


Figure 25. Spatial and temporal trends of total, mc-lr, and mc-rr microcystin in Fort Cobb Reservoir.

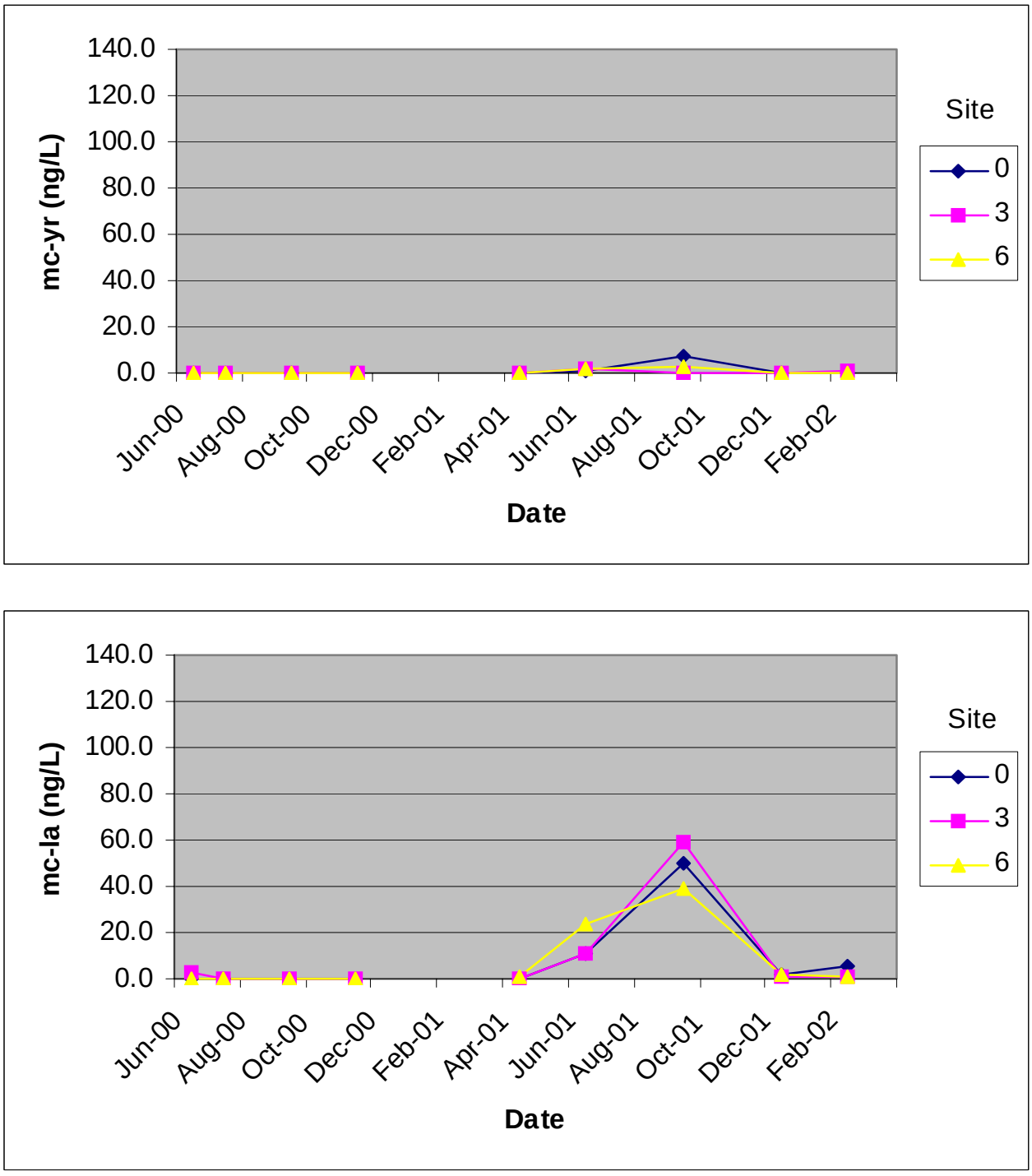
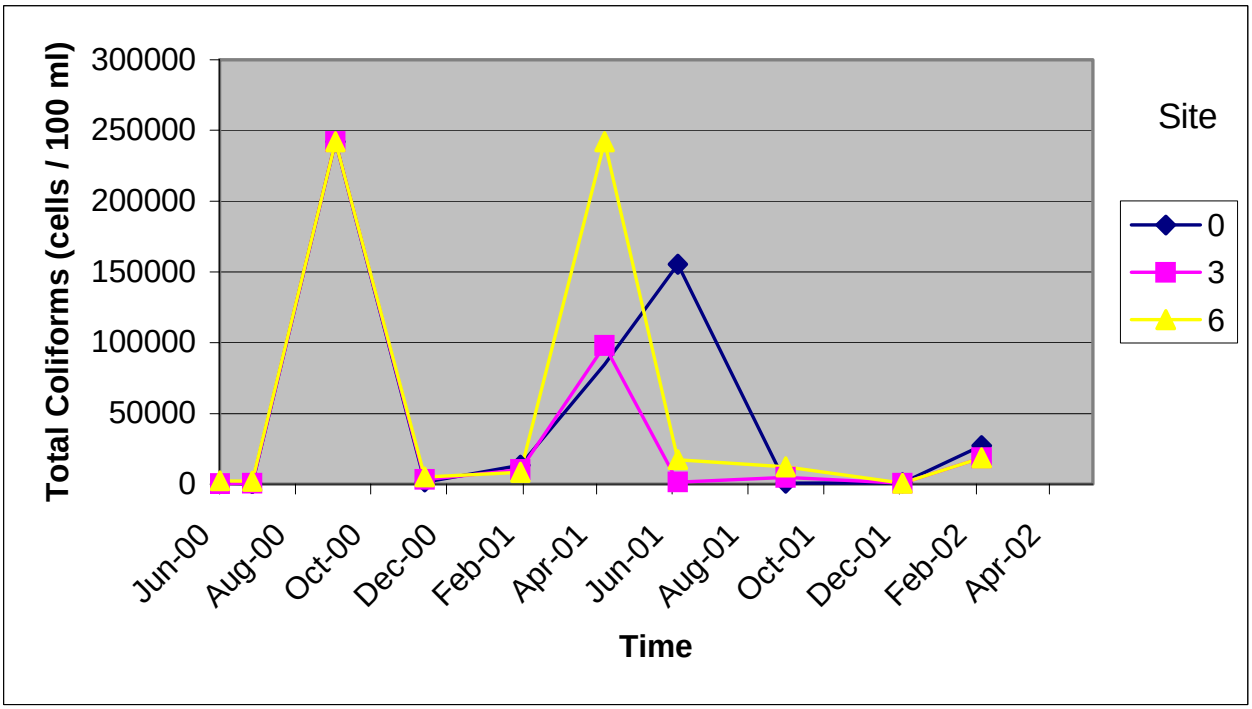


Figure 26. Spatial and temporal trends of mc-yr and mc-la microcystin in Fort Cobb Reservoir.



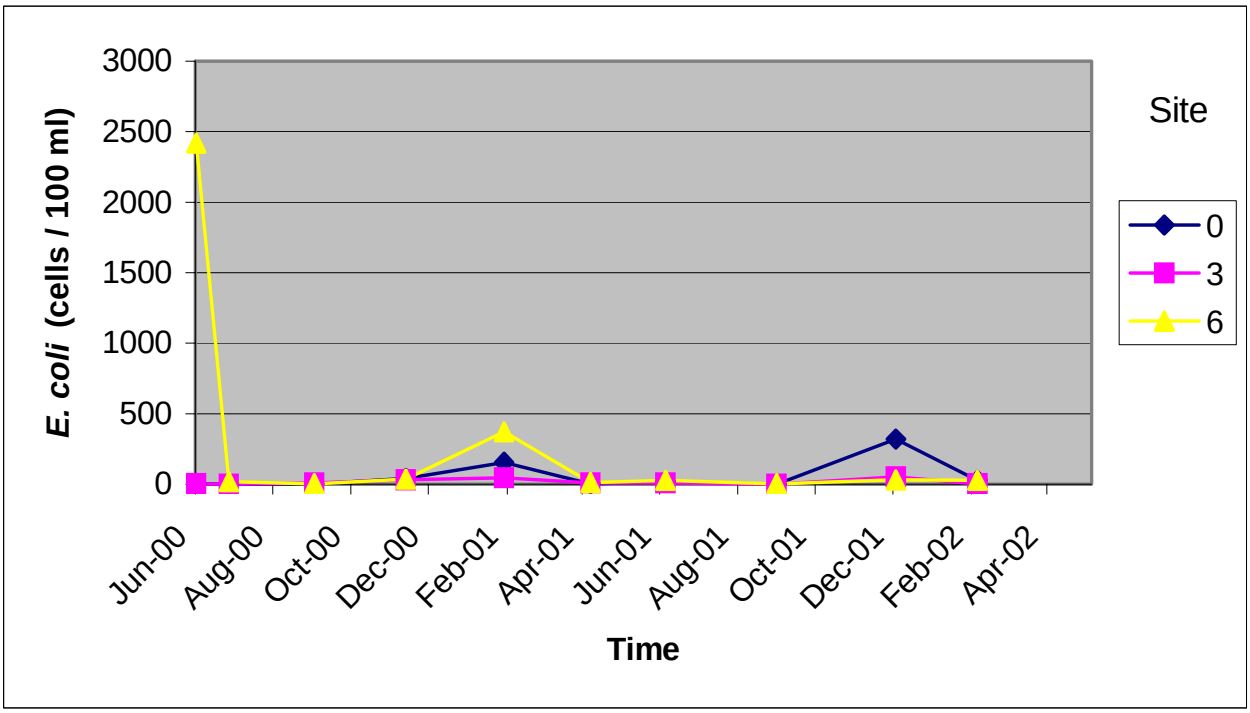
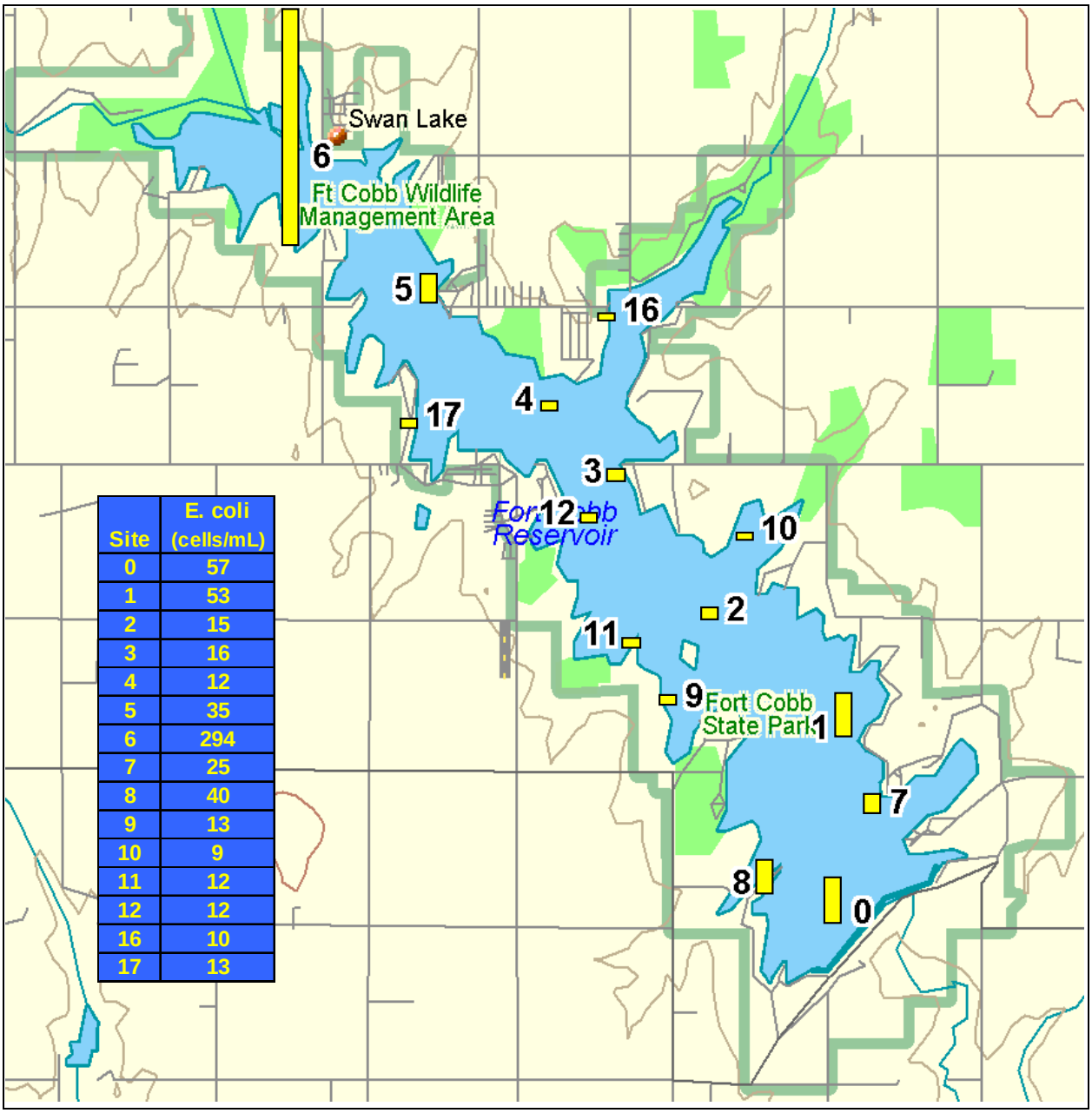
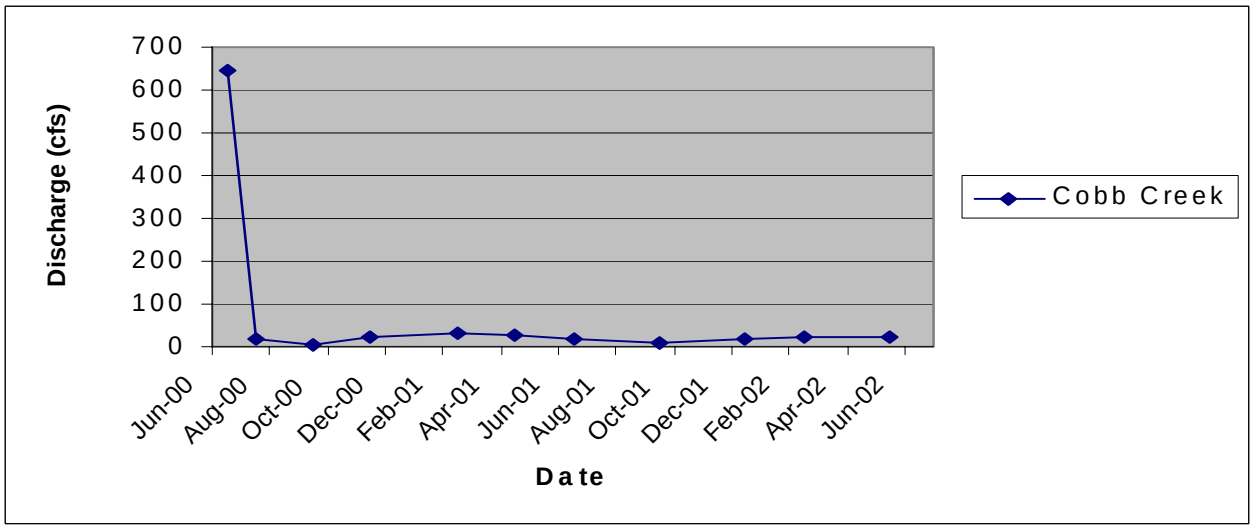


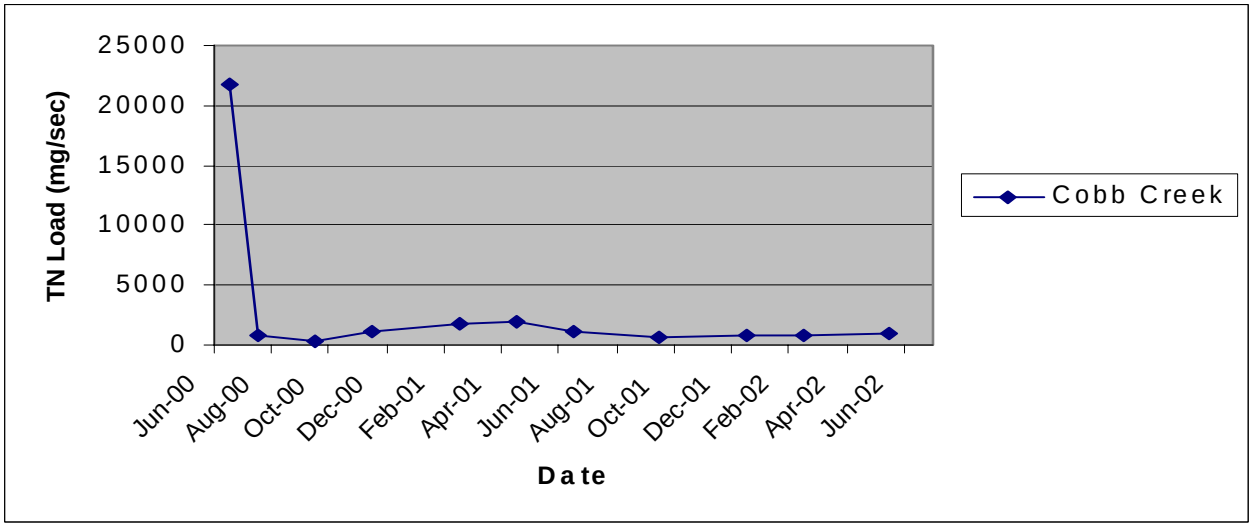
Figure 27. Spatial and temporal trends in total coliform and *E. coli* bacteria in Fort Cobb Reservoir.



Site	E. coli (cells/mL)
0	57
1	53
2	15
3	16
4	12
5	35
6	294
7	25
8	40
9	13
10	9
11	12
12	12
16	10
17	13

Figure 28. Average annual concentrations of *E. coli* at 17 reservoir stations in Fort Cobb Reservoir measured over the course of the 2-yr study. Numbers on the figure represent sites as described in Table 2. Height of the bars indicate relative *E. coli* numbers (cells/100 mL) shown in legend.





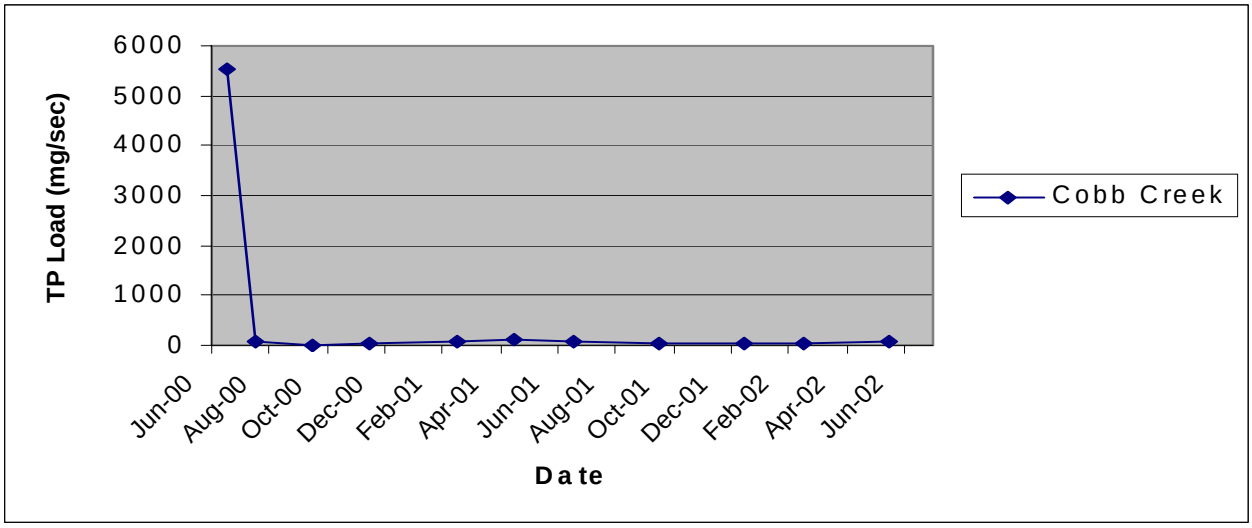
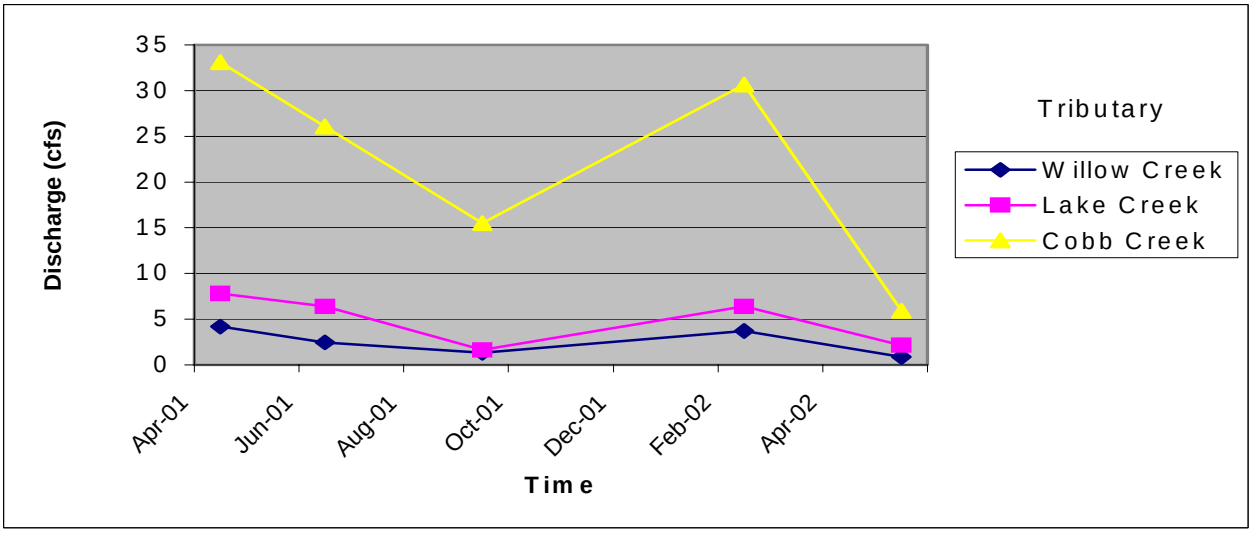
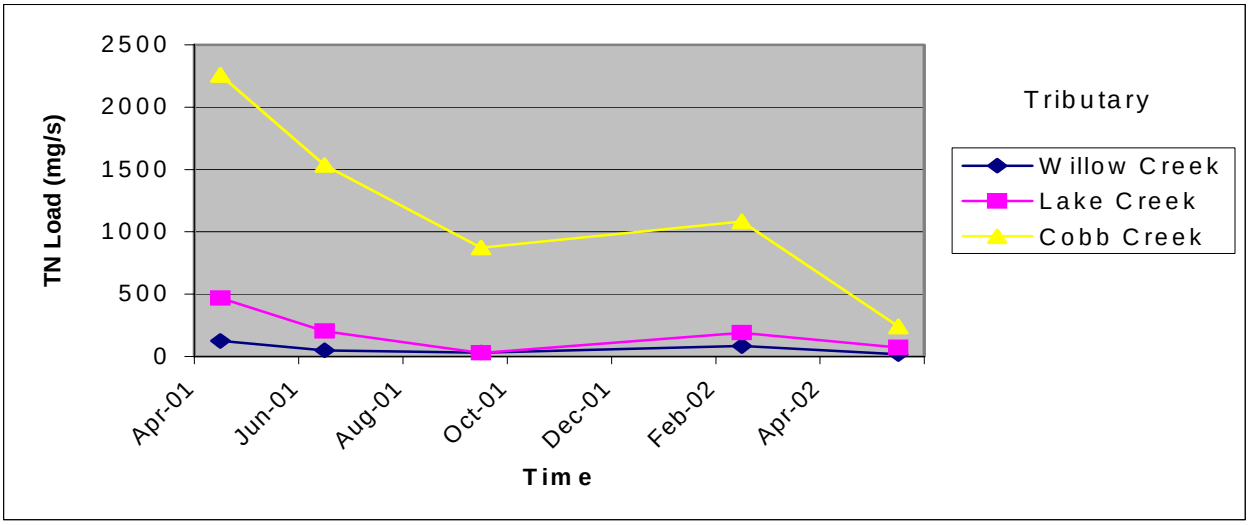


Figure 29. Temporal trends in discharge and nutrient loading in Cobb Creek. Data is based on discharge records from USGS Cobb Creek gaging station. Note influence of high discharge event on loading.





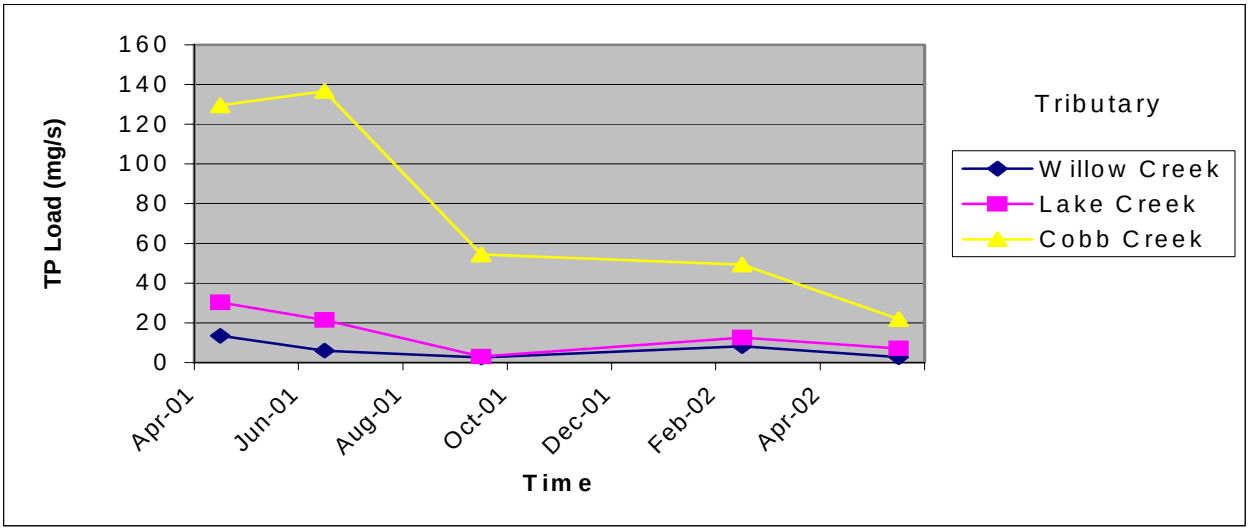
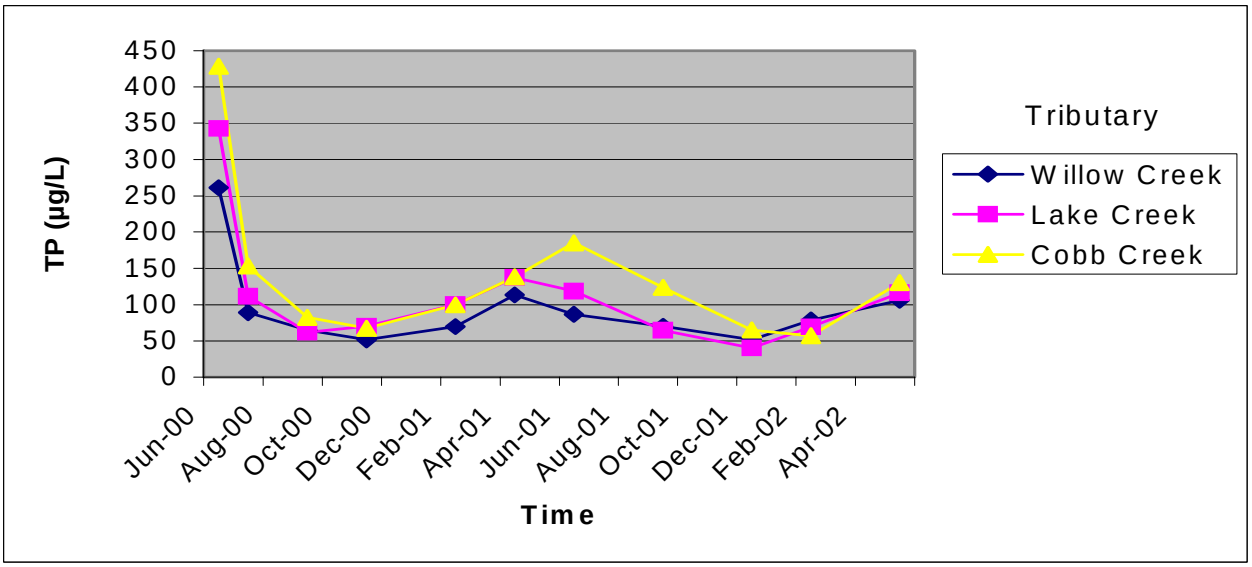


Figure 30. Temporal trends in discharge and loading of total nitrogen and phosphorus in tributaries to Fort Cobb Reservoir.



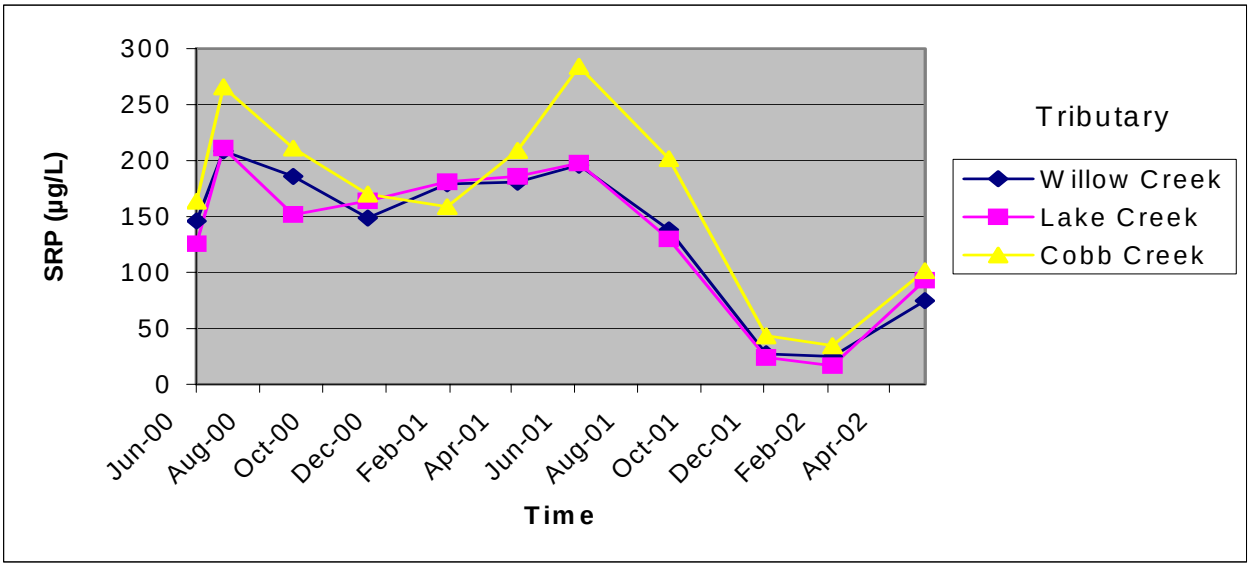


Figure 31. Temporal trends in total and soluble reactive phosphorus in tributaries to Fort Cobb Reservoir.

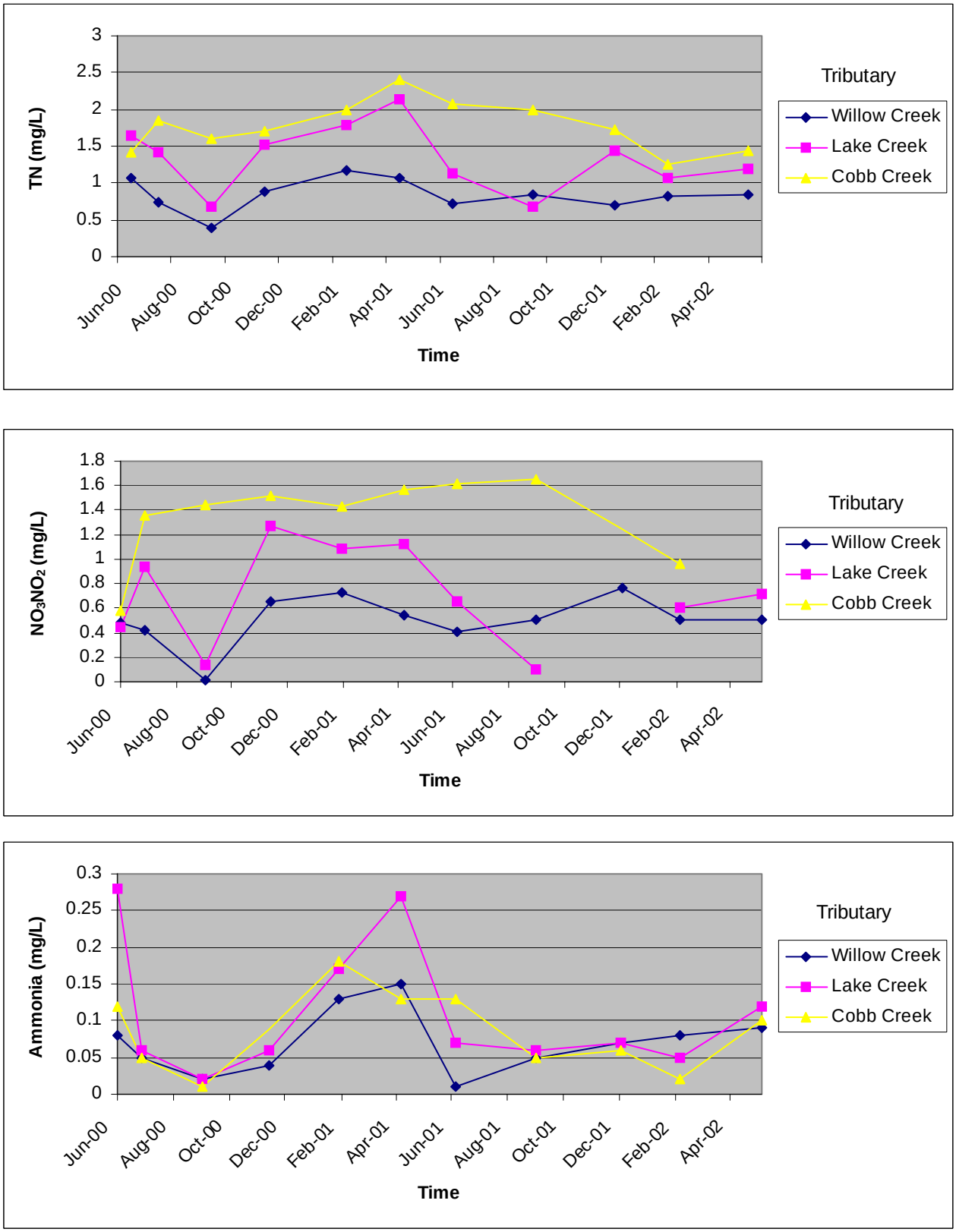
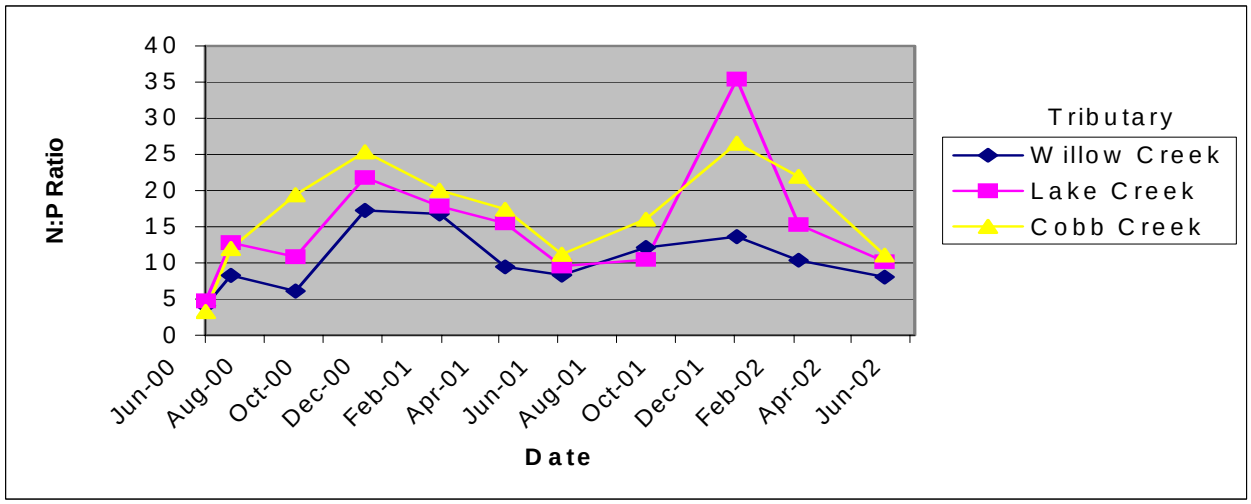
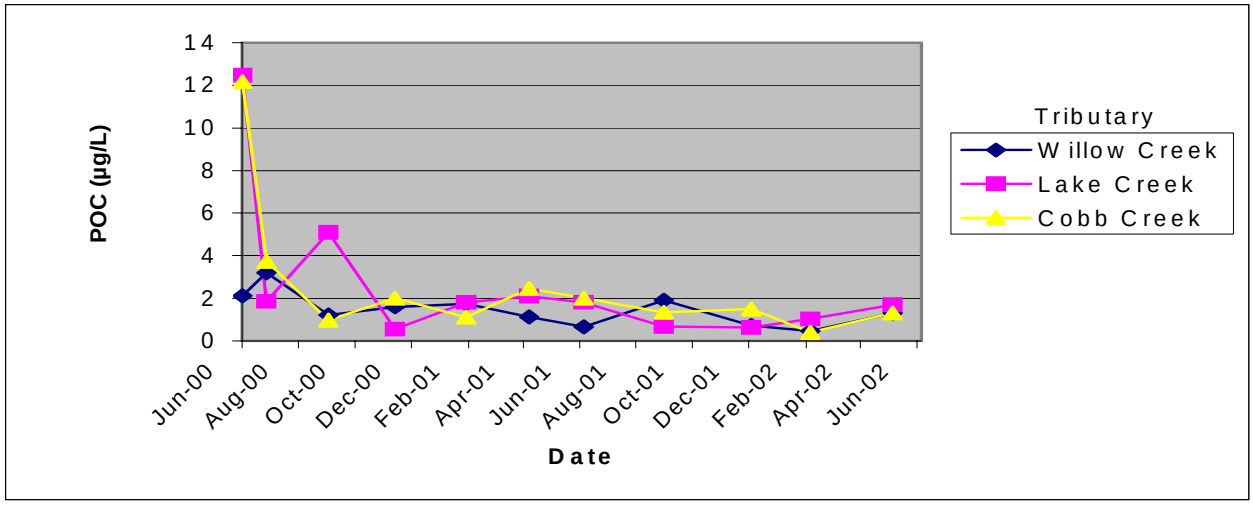


Figure 32. Temporal trends in total nitrogen, nitrate/nitrite, and ammonia in tributaries to Fort Cobb Reservoir.





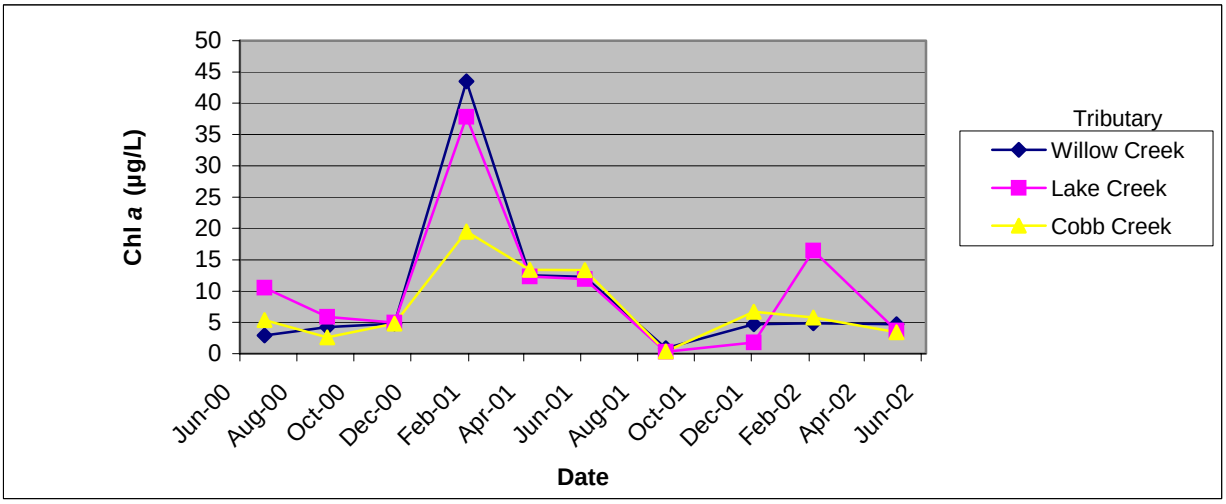


Figure 33. Temporal trends of N:P ratio, particulate organic carbon, and chlorophyll *a* in tributaries to Fort Cobb Reservoir.

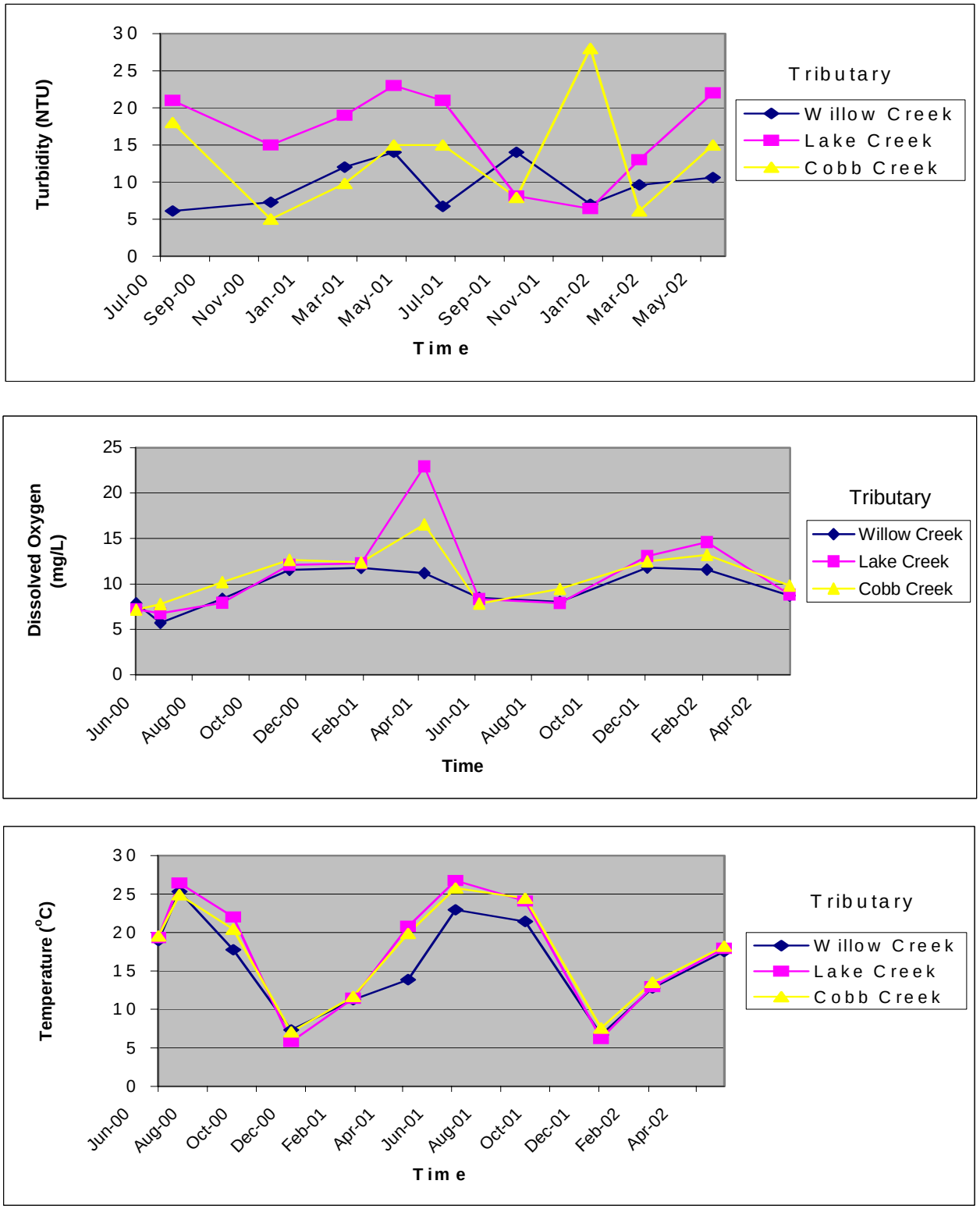


Figure 34**.** Temporal trends in turbidity, dissolved oxygen, and temperature in tributaries to Fort Cobb Reservoir.

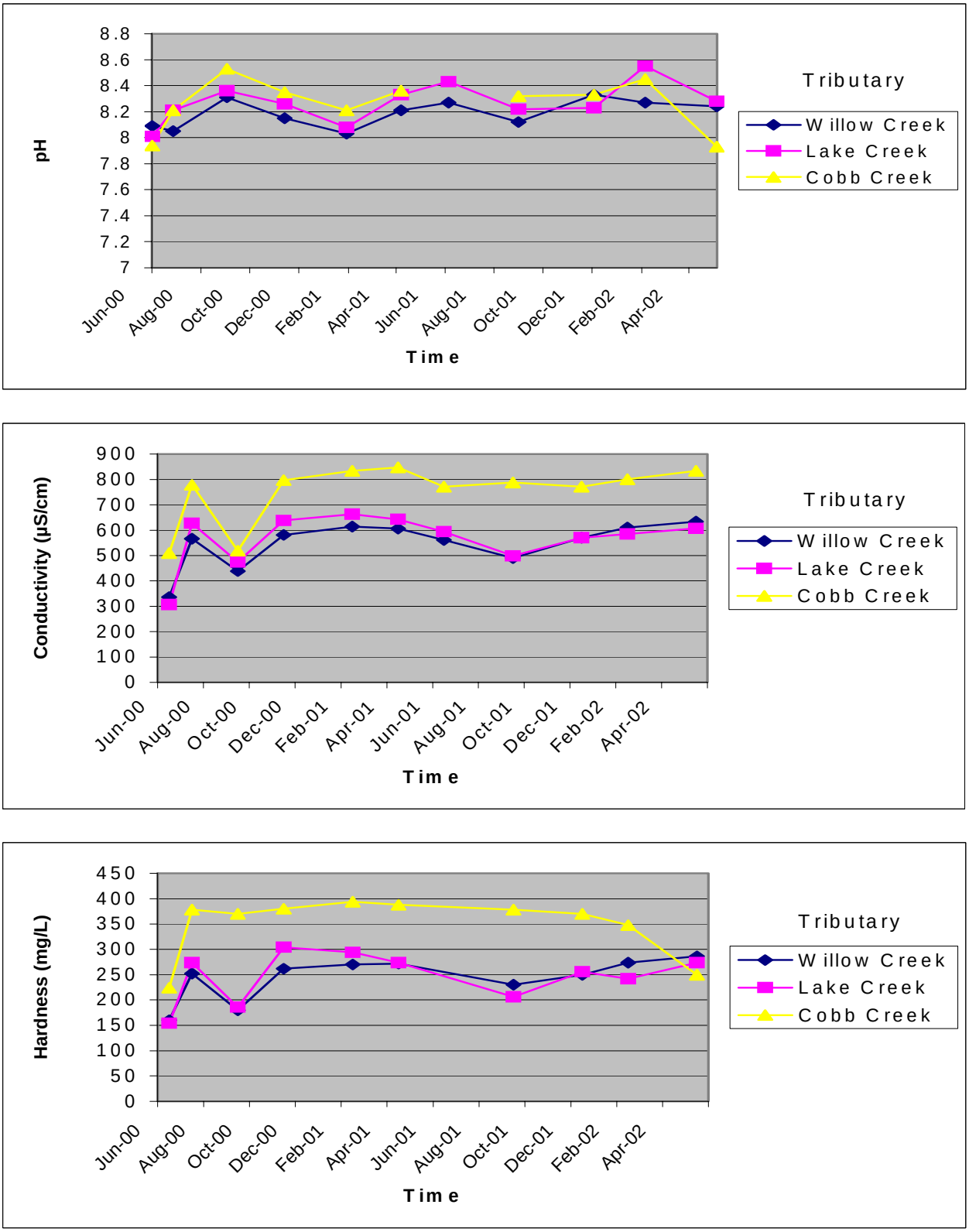
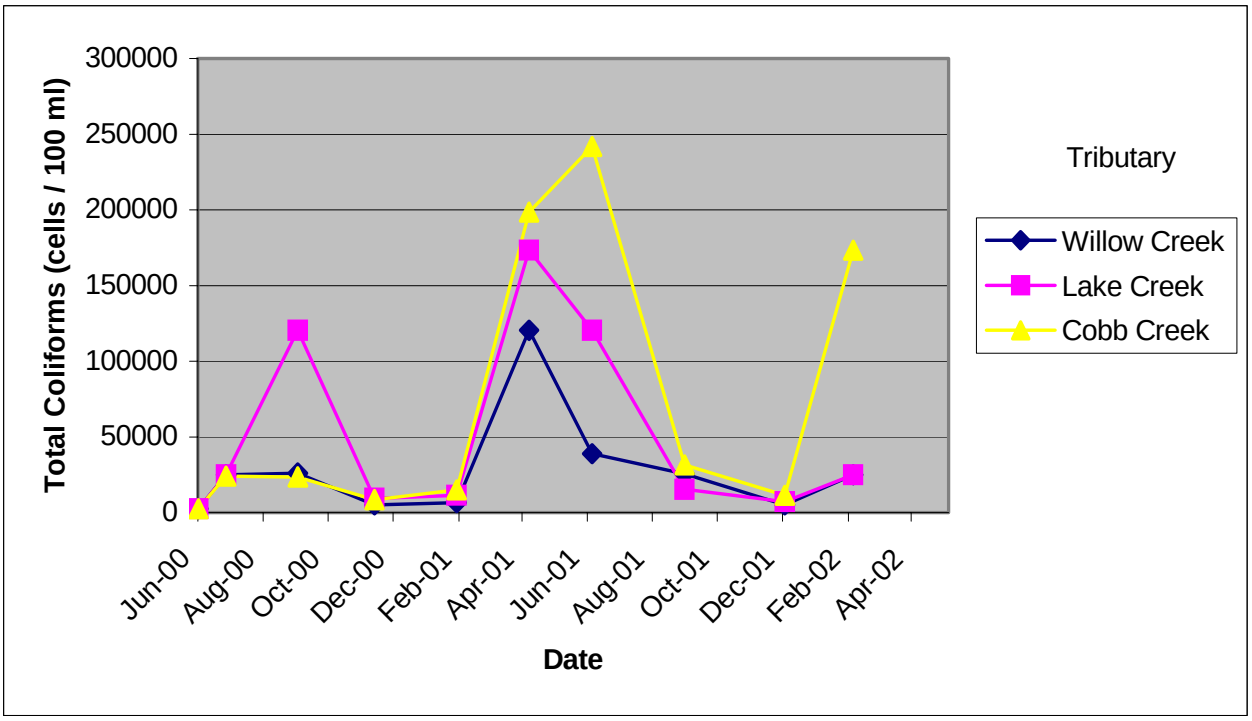


Figure 35. Temporal trends in pH, conductivity, and hardness in tributaries to Fort Cobb Reservoir.



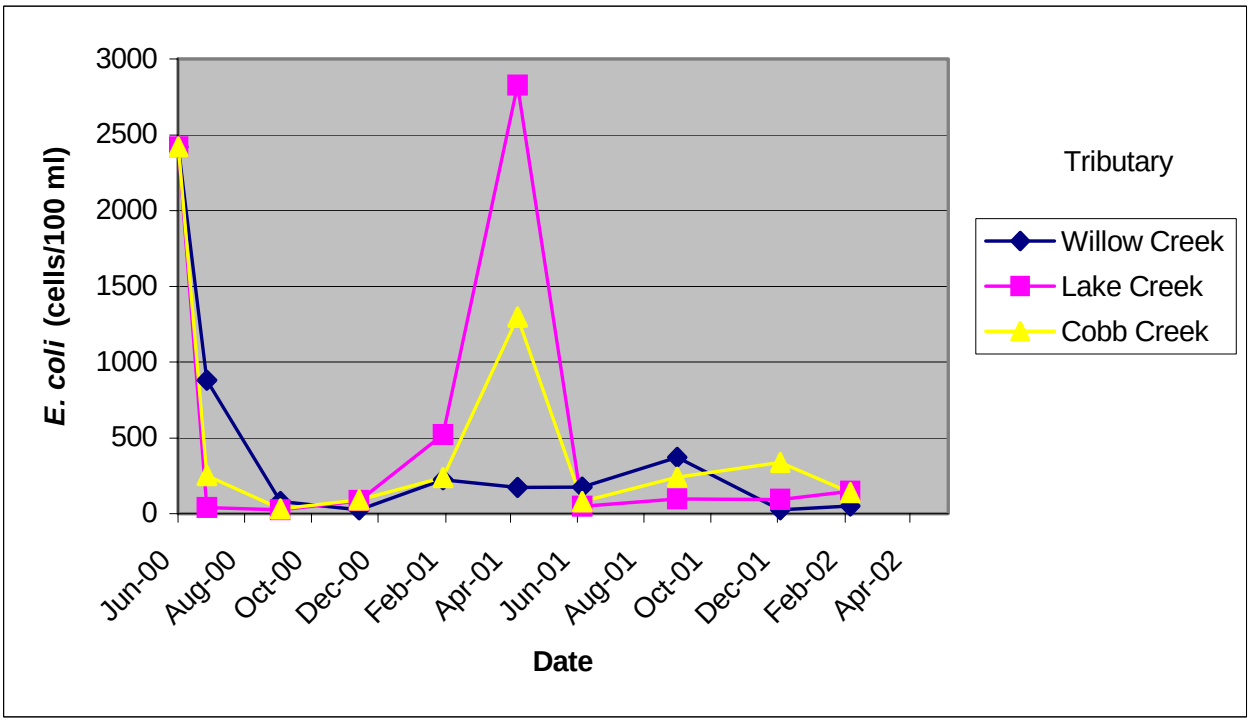
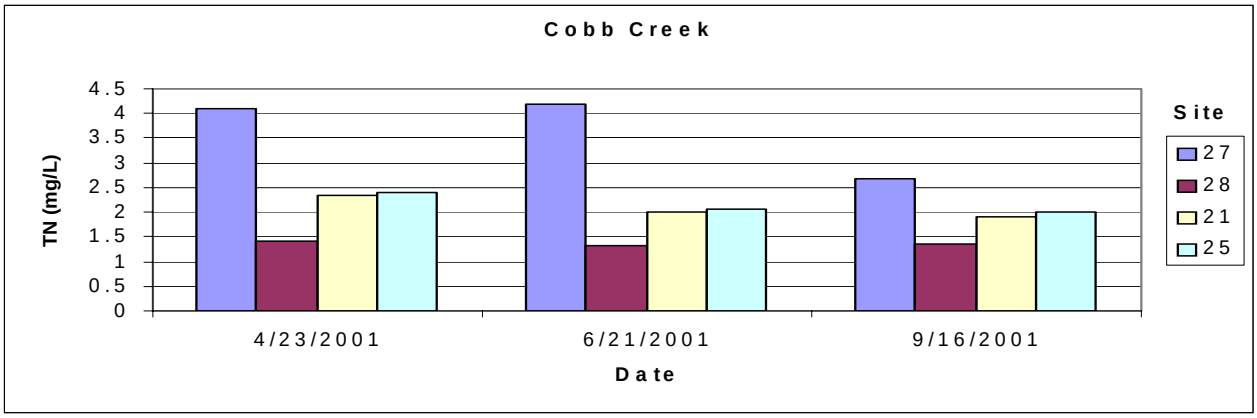
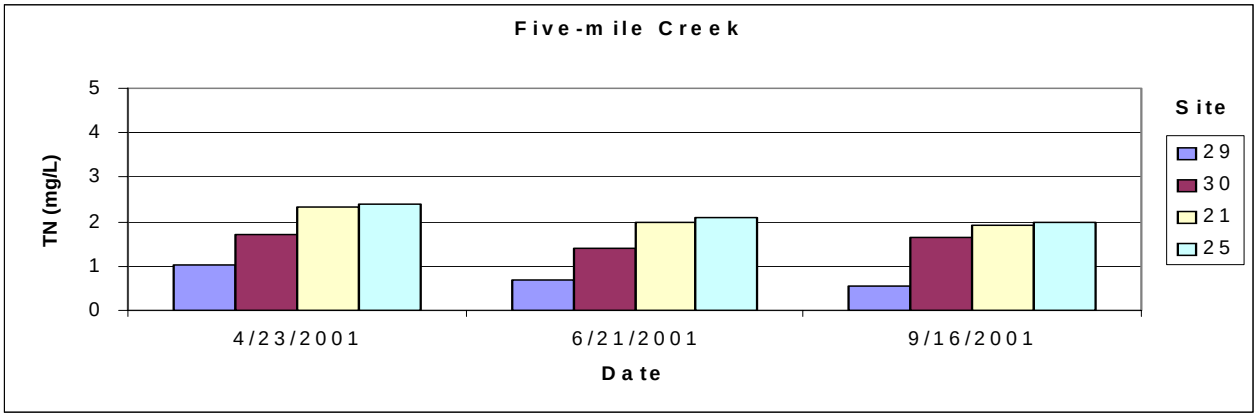
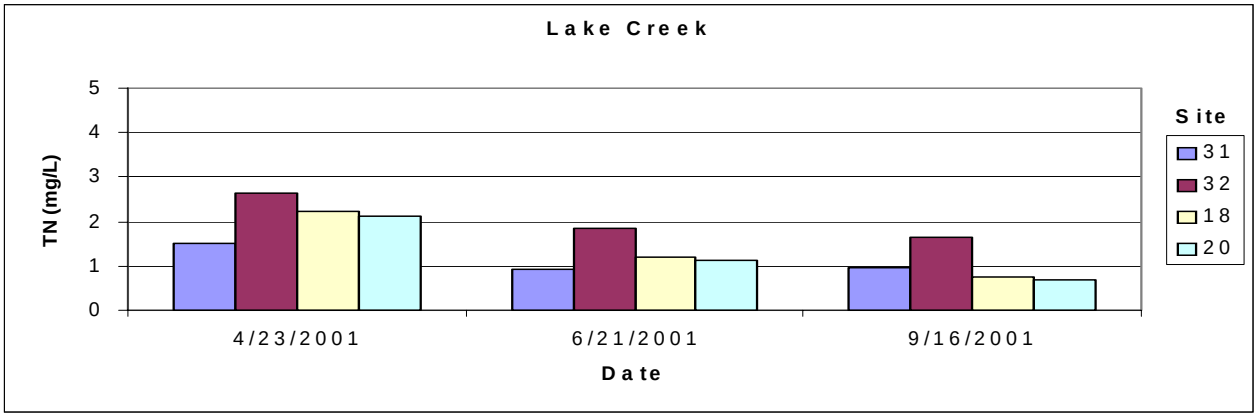


Figure 36. Temporal trends of total coliform and *E. coli* in tributaries to Fort Cobb Reservoir.







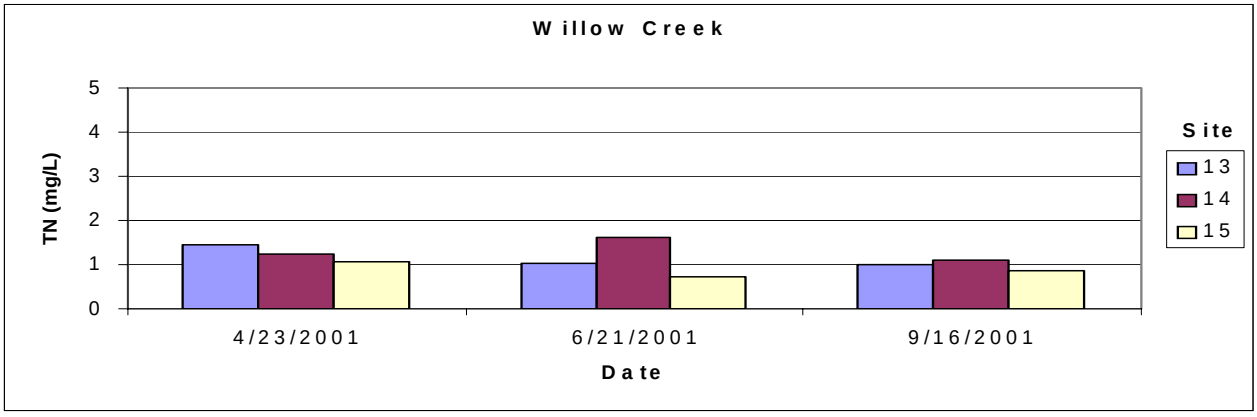


Figure 37. Longitudinal trends in total nitrogen in tributaries to Fort Cobb Reservoir.

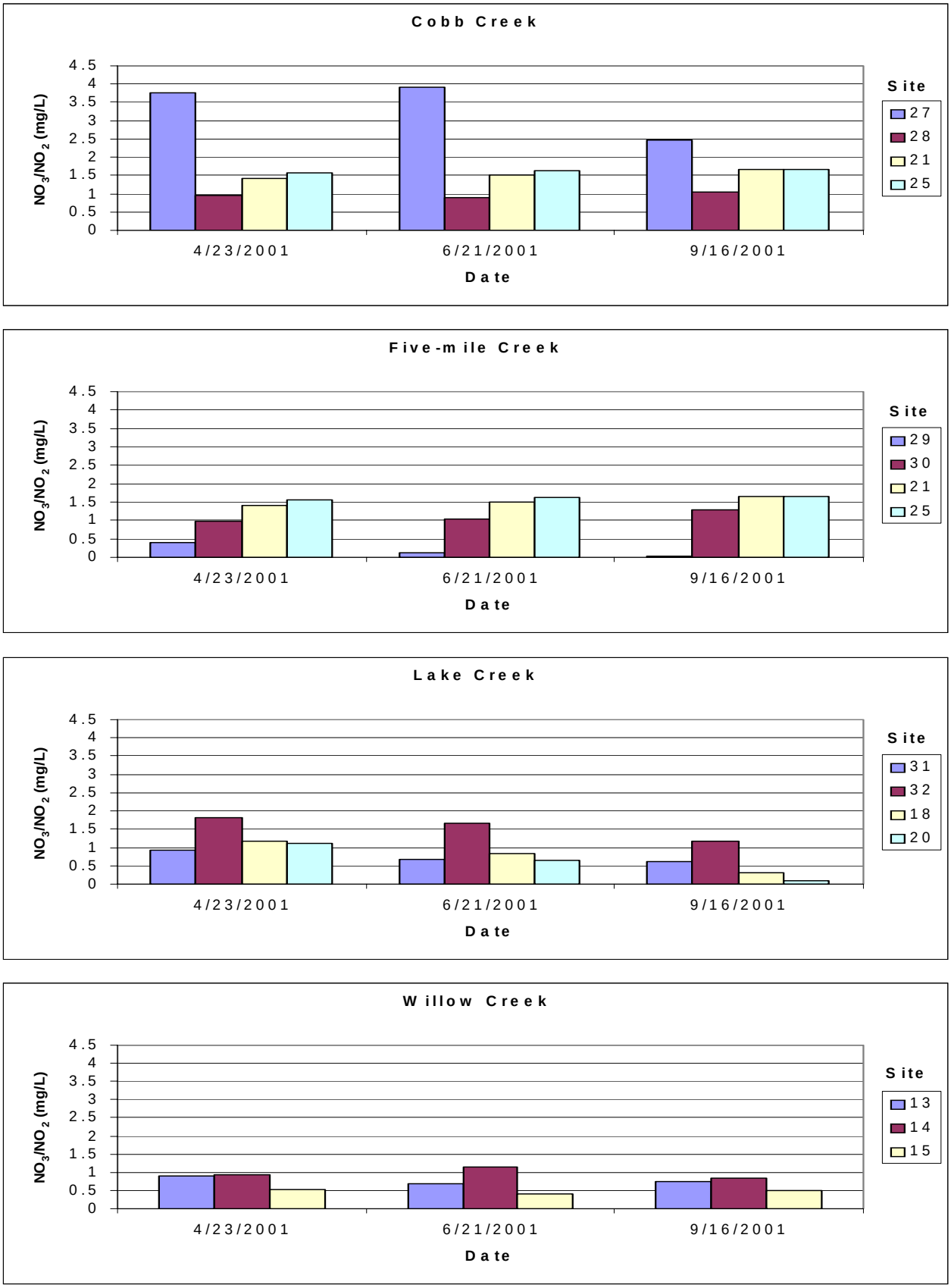
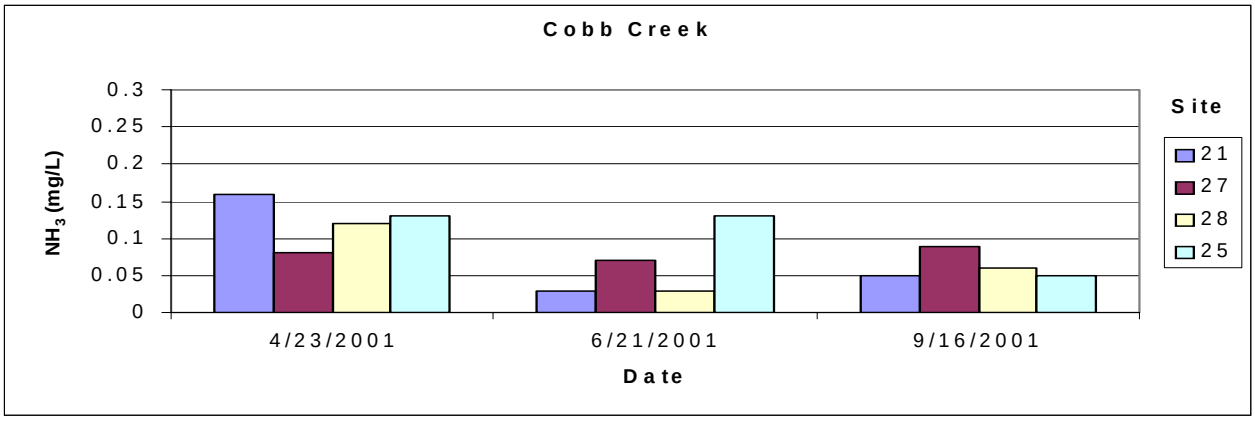
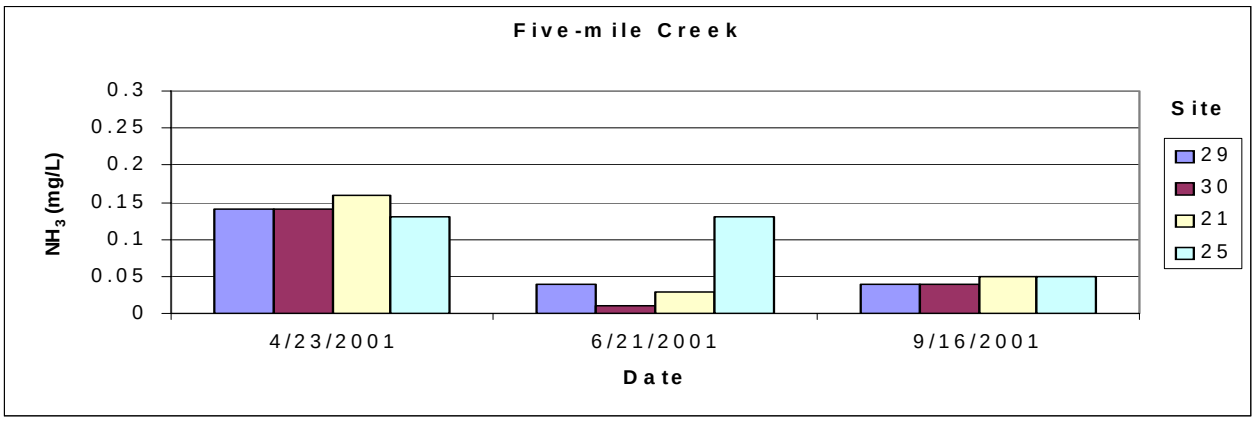
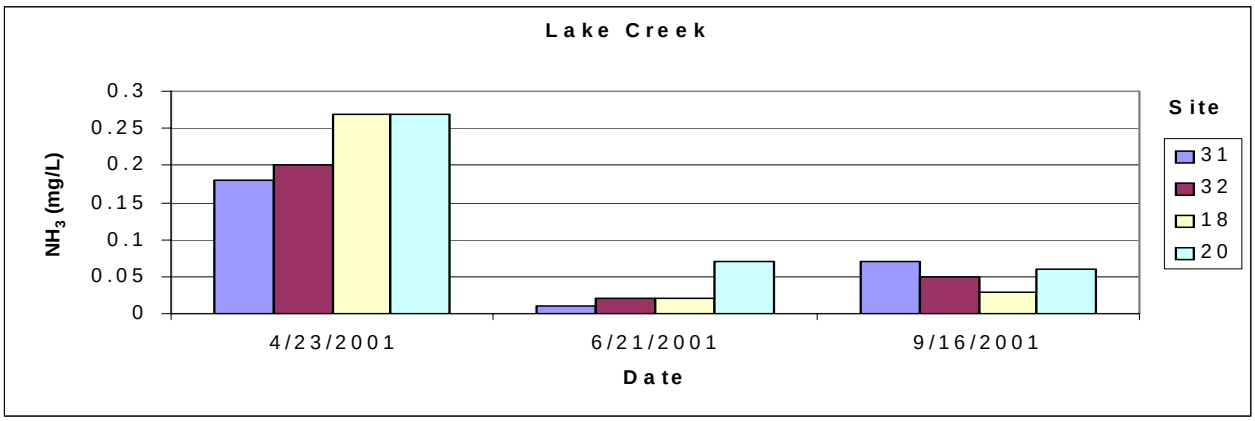


Figure 38. Longitudinal trends in total nitrate/nitrite in tributaries to Fort Cobb Reservoir.







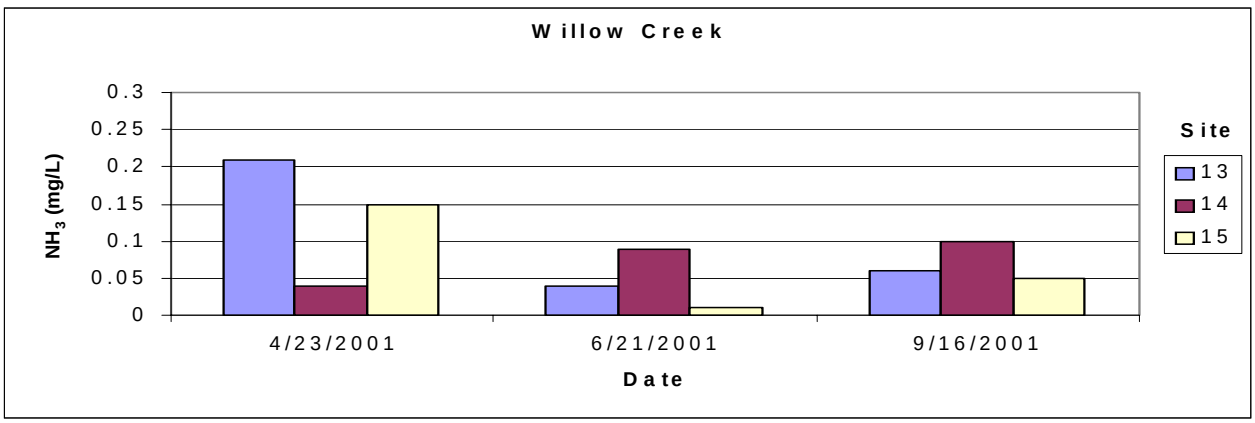


Figure 39. Longitudinal trends in ammonia in tributaries to Fort Cobb Reservoir.

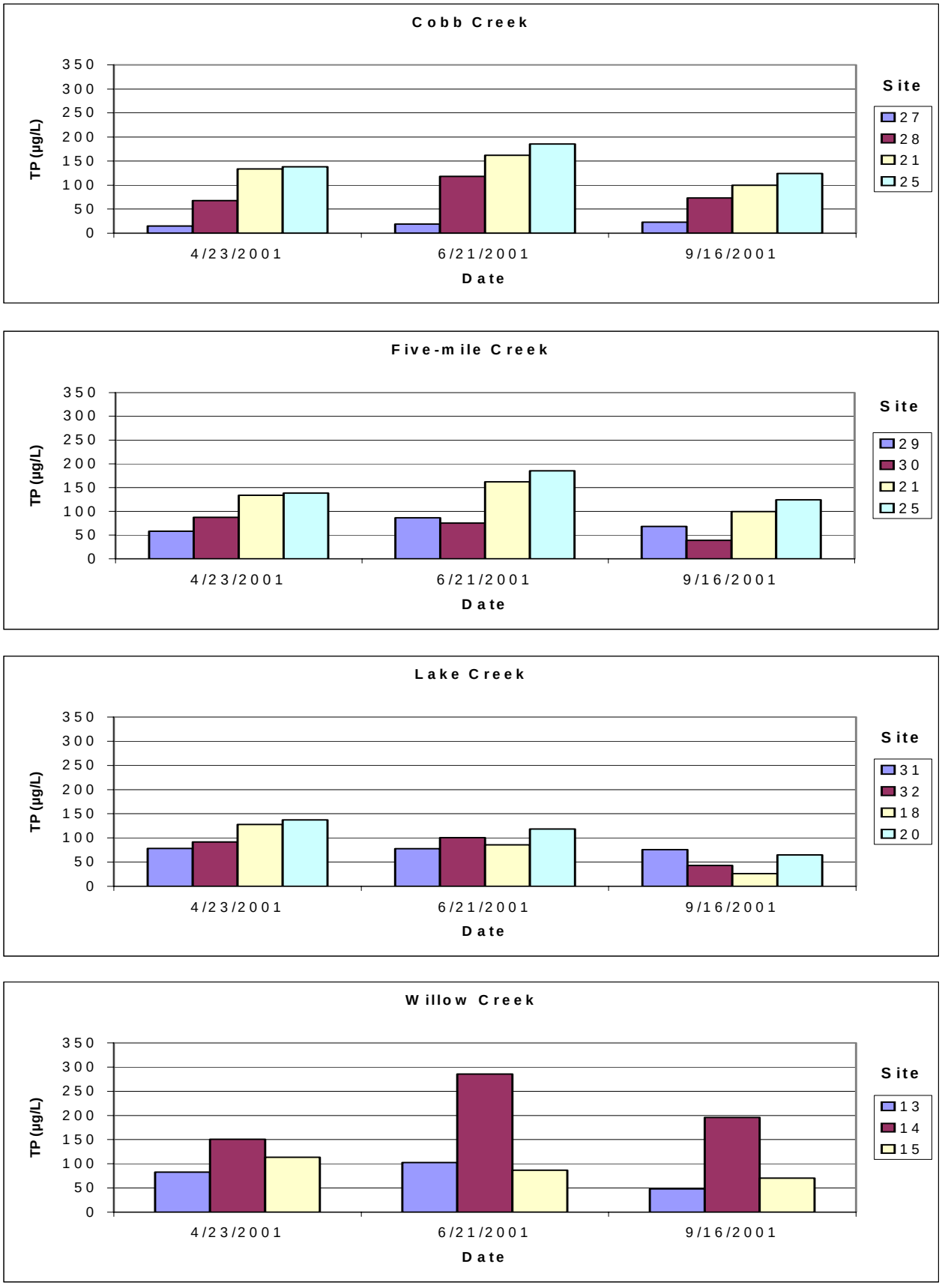


Figure 40. Longitudinal trends in total phosphorus in tributaries to Fort Cobb Reservoir.

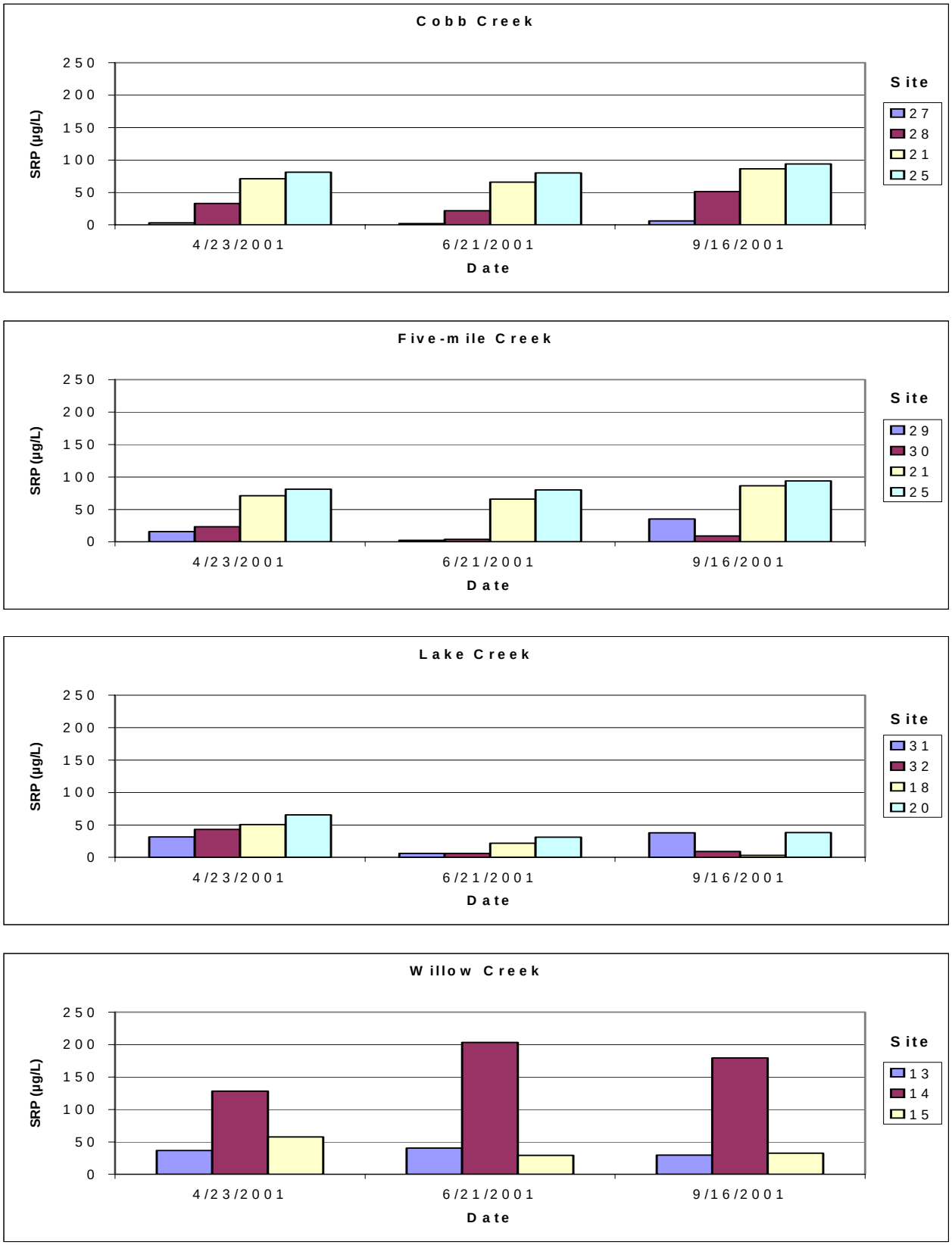
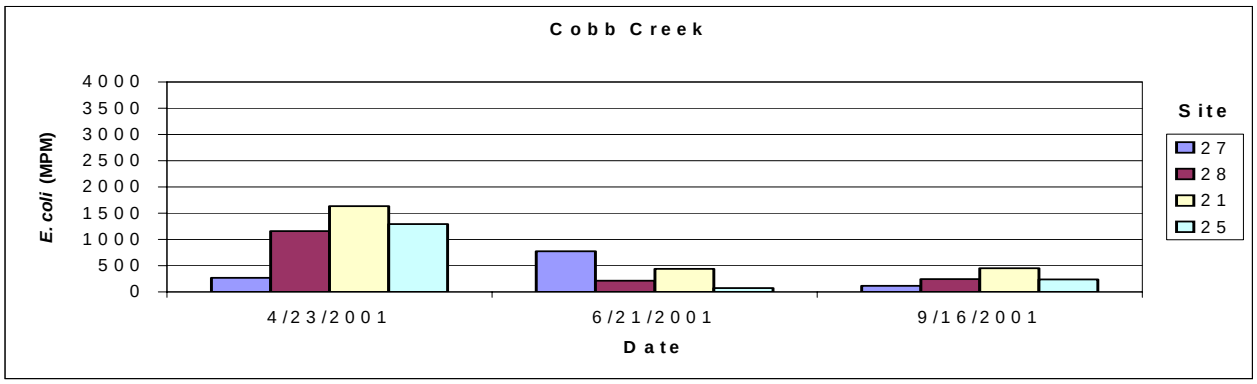
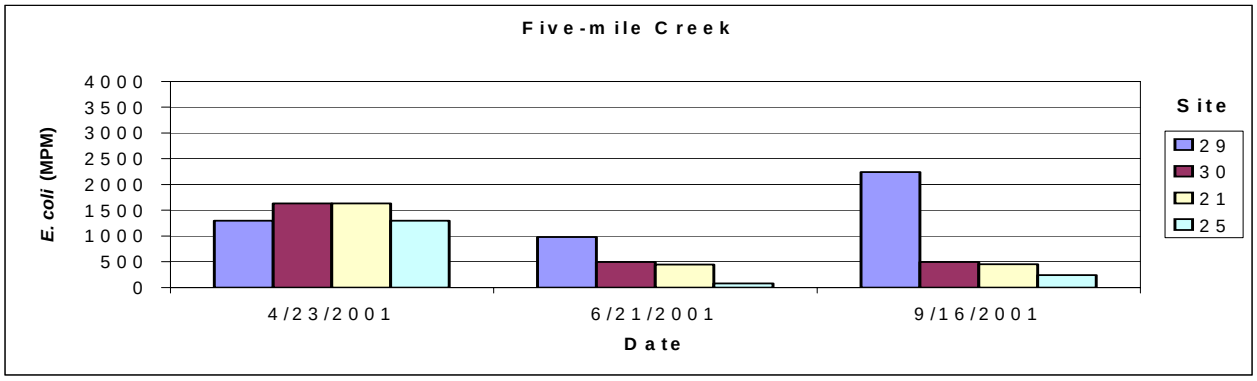
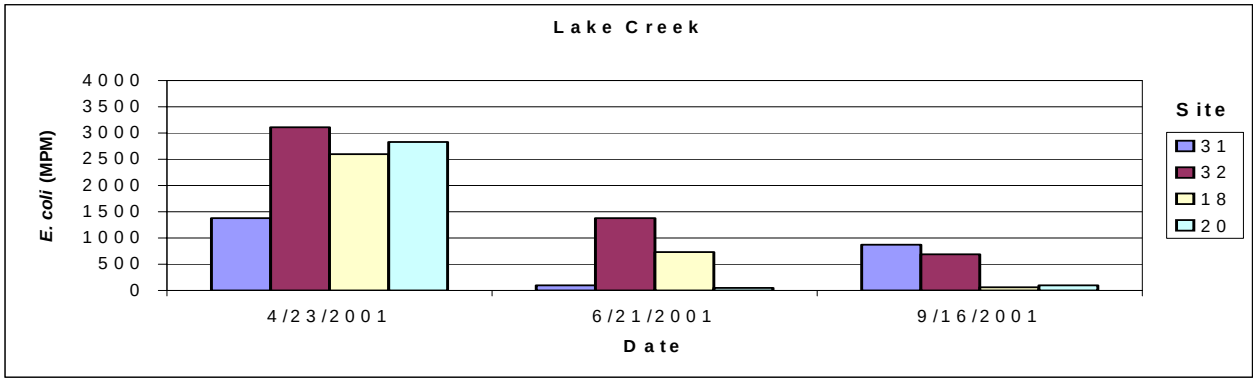


Figure 41. Longitudinal trends in soluble reactive phosphorus in tributaries to Fort Cobb Reservoir.







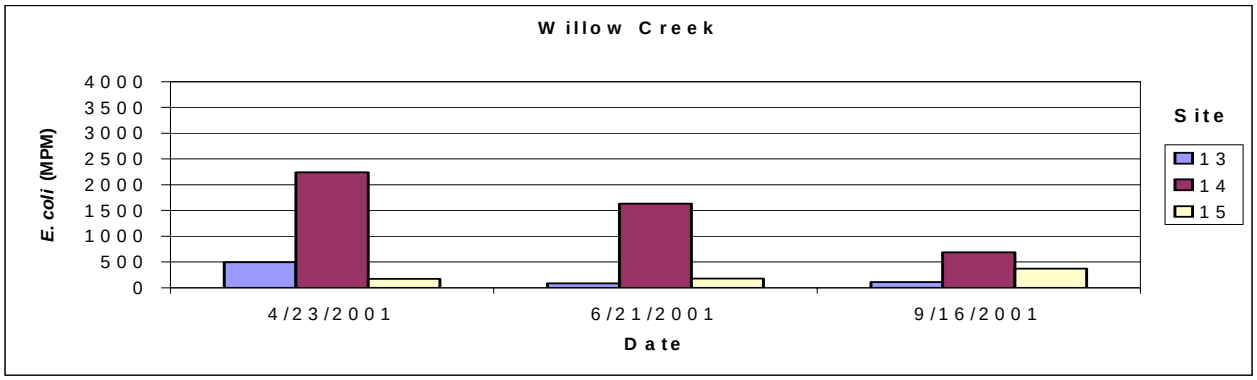


Figure 42. Longitudinal trends in *E. coli*. bacteria in tributaries to Fort Cobb Reservoir.