OKLAHOMA WATER RESOURCES BOARD

UNITED STATES ARMY CORPS OF ENGINEERS

AND

OKLAHOMA STATE UNIVERSITY

# FINAL REPORT

# FOR

# COOPERATIVE "CLEAN-LAKES" PROJECT

# PHASE I

# DIAGNOSTIC AND FEASIBILITY STUDY

# ON

# TENKILLER LAKE, OKLAHOMA

SPONSORED BY

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY REGION VI
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## Abstract

DIAGNOSTIC-FEASIBILITY STUDY OF WATER QUALITY IN LAKE TENKILLER, OKLAHOMA

**Scope and Method of Study:** A Phase I Diagnostic-Feasibility study was performed to determine the levels of nutrients, metals, and pesticides within the water column and metals in the sediment of Lake Tenkiller. The relative inputs of nutrients from point and nonpoint sources into the Illinois River were determined by retrieval of water quality data from agency archival databases and by analysis of nutrient concentrations in runoff samples from selected tributaries reflecting dominant agricultural landuse practices. Selected samples collected during runoff events were used to calibrate an event-based runoff model to extrapolate/predict total quantity of nitrogen and phosphorus exported from nonpoint sources in the Illinois River Basin. Several empirical models were used to predict total annual loads of nutrients transported by the river to its confluence with Lake Tenkiller. Nitrogen, phosphorus, and other physical chemical parameters were analyzed in samples collected from stations distributed from upper to lower end of the reservoir to ascertain any spatial and temporal gradients. The annual loading of phosphorus and nitrogen versus mean retention time of water and mean annual concentration of phosphorus, nitrogen, Secchi disk, and chlorophyll *a* in the reservoir were used as indices of the trophic status of the reservoir. Nutrient dynamics within Lake Tenkiller were modeled with a combination of empirical methods and lake models to predict trophic status along a gradient from upper to lower end of lake. Outputs from the models were combined to predict potential effects upon Lake Tenkiller if nutrients carried by the Illinois River exceeded a predicted total maximum daily load (TMDL) necessary to maintain the desired trophic status. The TMDL's were intended to be used as water quality goals for evaluating alternative management practices within the basin and lake to restore and maintain desirable trophic status of Lake Tenkiller.

**Findings and Conclusions:** The mean annual concentrations of phosphorus, nitrogen, and chlorophyll *a* measured throughout Lake Tenkiller were indicative of eutrophic conditions. A gradient in trophic status was evident in the epilimnetic strata of the lake, i.e., from eutrophic at the upper end to meso-eutrophic at the lower end. The entire lake was affected by eutrophication as evidenced by the presence of anoxic conditions in the hypolimnion during summer stratification. The principal investigators recommend the eutrophication process be controlled or reversed by reducing phosphorus input to the lake from both point and nonpoint sources. The recommendations included a short-term goal of 30-40% reduction of influent total phosphorus within the next five years, followed by a further reduction of up to 70-80% of current total phosphorus loading. Predictions of reductions in total phosphorus loads that would yield hypolimnetic oxia were cost-prohibitive and probably not technically feasible. Therefore, we further recommend re-areation devices be installed in the tailrace for protection of the downstream trout fishery.

**Acknowledgements**

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The staff of the Oklahoma Water Resources Board were extremely helpful and we wish to extend our appreciation to Shon Simpson, Derek Smithee, Bill Cauthron, and Paul Koenig.

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Land use data and maps for Craig, Delaware, Ottawa, and Mayes Counties in Oklahoma were provided by Ron Treat and Darrel Hammond of the USDA Soil Conservation Service office in Stillwater, OK.

**Executive Summary:**

## Introduction

Lake Tenkiller, located in eastern Oklahoma has been one of the clearest lakes in Oklahoma. Due to its exceptional clarity, the lake serves many recreational purposes in addition to its authorized purposes (e.g., flood control, public water supply, hydroelectric power). These recreational opportunities include SCUBA diving, which is rare for Oklahoma lakes, and have fostered development of a recreational economy for the surrounding region. However, users have noted a degradation in water quality and possibly-impaired uses. In response to these indications, the United States Environmental Protection Agency funded a Clean Lakes study on the lake. The Oklahoma Water Resources Board, United States Army Corps of Engineers, and Oklahoma State University were the investigative agencies. The project objectives were to evaluate past and current trends of water quality in the lake and watershed. Based upon these trends, the investigators evaluated any potential problems in the lake and the feasibility of recommended restorative measures.

## Findings and Conclusions

This study determined that Lake Tenkiller is currently showing signs of water quality degradation. These symptoms included periodic algal blooms, excessive algal growths, and hypolimnetic anoxia throughout the stratified period. The lake was classifed as eutrophic based upon nitrogen and phosphorus loads which were excessive when compared to published criteria. These loads were derived predominantly from non-point sources during high flows. During low flow periods, point and nonpoint source contributions were approximately equal. Furthermore, these nutrient loads, especially the nonpoint fractions, have increased significantly since 1974 but may have temporarily stabilized since 1985-86. The excessive current loads ostensibly have increased algal growth and thus compromised water clarity throughout the lake. Nutrient limitation analysis indicated that the lake was phosphorus limited in the lower end (i.e., near the dam), dual (N and P) limited in the midreaches, and probably light limited in the extreme upper end. Based upon these results, we concluded that source control of impinging phosphorus loads is the optimum management alternative.

Accumulation of toxics in the lake and its resident fish did not appear to be a problem at this time.

**Recommended Remediation Efforts**

**Eutrophication**: Three alternative P-control measures are:

1. no action,
2. maintain current condition of the lake by preventing further increases in nutrient loads, and,

3. reverse the accelerated eutrophication by more stringent P-control measures.

Obviously, the above three alternatives are not discrete options but represent a continuum of management. After considering the feasibility and effectiveness of control measures, **we recommend a 30-40% reduction in the headwater P loads** be implemented as a short-term goal and 70-80% reduction as a long-term goal. Since both of these goals still indicated a significant risk of hypolimnetic anoxia, we further recommend that re-areation devices be installed in the tailrace to protect the downstream trout fishery.

We recommend the following programs be initiated to attempt to reduce phosphorus contamination within the basin:

1. voluntary switch to non-phosphate detergents by all lake side residents and the cities of Tahlequah and Watts, OK, and Rogers and Springdale, AR,
2. implementation of best management practices upstream from Lake Tenkiller to minimize contributions of phosphorus in surface water runoff from agricultural fertilizer and waste and poultry-litter applications,
3. continue to work with point source dischargers, to the extent possible within the watershed to minimize discharges of nutrients including phosphorus, and
4. establish a citizen's monitoring group for basic water quality analysis and evaluation thus affording a more robust assessment of management effectiveness.

**RATIONALE**

If the concentration of phosphorus was reduced in Lake Tenkiller, how would this affect water quality?

We used Reckhow's empirical equations relating concentrations of phosphorus and nitrogen to chlorophyll *a* concentrations to calculate similar relationships for Tenkiller (Reckhow 1988). The concentration of chlorophyll *a* was used as our primary index of trophic status and as a tool for predicting future algal density if phosphorus was reduced.

The growth of algae is generally increased by an increased level of phosphorus in the water. In fact, high concentrations of phosphorus coupled with relatively lower levels of nitrogen often lead to extensive growths of blue-green algae, an undesirable, odor-producing form of algae. Although present, extensive growths of blue-green algae presently do not exist in Lake Tenkiller and our recommendations included taking measures to prevent development of nuisance blue-green algal blooms.

The measurement of chlorophyll *a*, a green pigment found in the algae, provides an index of the density of algae. Low concentrations of chlorophyll *a* indicate a low density of algae and vice versa. Since the growth of algae is increased by phosphorus, we predict a reduction in phosphorus would reduce the density of algae and chlorophyll *a*.

One of the problems associated with algal growths is a reduction in the clarity of the water. Thus, the aesthetic quality of the water is decreased and the lake is less attractive to visitors and residents. Many of the more popular recreational lakes in the northcentral United

States have low concentrations of phosphorus and therefore low densities of algae. In contrast, some enriched lakes are extremely green and are not as attractive for some recreational activities. Tenkiller Lake is currently exhibiting symptoms of accelerated eutrophication in the upper end and in some coves. Our recommendations included implementation of measures that would prevent further deterioration in water quality.

The clarity of water is easily measured by lowering a circular disk into the water and recording the depth at which the disk disappears from view. The disk is called a Secchi disk after the limnologist who developed the method. In two Michigan lakes, summer Secchi disk depths were generally less than 1.5 meters (about 5 feet) in an enriched lake and generally exceeded 5 meters (over 16 feet) in a nonenriched, hardwater lake. We predict the annual average Secchi disk depth at the lower end of Lake Tenkiller would increase in response to a reduction in impinging phosphorus loads. This increase would reflect increased light penetration and thus an increase in clarity, which preserves the aesthetic quality of the lake and its associated recreational assets.

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3. Will the proposed project be likely to lead to a change in established land use patterns, such as increased development pressure near the lake? To what extent and how will this change be controlled through land use planning, zoning, or through other methods?

4. Will the proposed project adversely affect a significant amount of prime agricultural land or agricultural operations on such land?

5. Will the proposed project result in a significant adverse effect on parkland, other public land, or lands or recognized scenic value?

6. Has the State Historical Society of State Historical Preservation Officer been contacted? Has he responded, and, if so, what was the nature of that response? Will the proposed project result in a significant adversely effect on lands or structures of historic, architectural, archaeological or cultural value?

7. Will the proposed project lead to a significant long-range increase in energy demands?

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12. Does the project proposal contain all information that EPA requires in order to determine whether the project complies with Executive Order 11990 on wetlands? Will the proposed project have a significant adverse effect on fish and wildlife, or on wetlands or any other wildlife habitat, especially those of endangered species? How significant is this impact in relation to the local or regional critical habitat needs? Have actions to mitigate habitat destruction been incorporated into the project? Has the recipient properly consulted with appropriate Sate and Federal fish, game and wildlife agencies and with the U.S. Fish and Wildlife Service? What were their replies?

13. Describe any feasible alternatives to the proposed project in terms of environmental impacts, commitment of resources, public interest and costs and why they were proposed.

14. Describe other measures not discussed previously that are necessary to mitigate adverse environmental impacts resulting from the implementation of the proposed project.

# 1. IDENTIFICATION OF LAKE

a. Name: Tenkiller Ferry Lake - - operated by the U. S. Army Corps of Engineers, Tulsa District, Tulsa, OK. Construction of the lake, which was authorized under the Flood Control Act of 1938 and River and Harbor Act of 1946, began in 1947 with full flood control operation by 1953. The lake was built by the United States Army Corps of Engineers. The rolled earthfill dam is approximately 923 m long and 61 m above the streambed. Outlet works include ten 15.4 X 7.7 m tainter gates, a 5.8 m diameter hypolimnetic conduit, and a 5.8 m diameter hypolimnetic intake to the powerhouse. Lake morphometric data are given in Table I.

Table I. Lake Tenkiller Morphometry.

| Parameter | Value |
|---|---|
| Elevation (NGVD) | |
| @Conservation pool | 632.0 |
| @Flood pool | 667.0 |
| Capacity ($km^3$) | |
| @Conservation pool | 0.81 |
| @Flood pool | 1.52 |
| Area ($km^2$) | |
| @Conservation pool | 52.2 |
| @Flood pool | 84.2 |
| Depth (m) | |
| Mean | 15.5 |
| Maximum | 46.3 |
| Relative | 0.57 |
| Shoreline length (km) | 209 |
| Shoreline development | 8.17 |
| Volume development | 1.00 |
| Average hydraulic residence time (yr) | 1.76 |

b. Location: Lake Tenkiller is located in eastern Oklahoma in Cherokee and Sequoyah counties. The dam is loacated at T13N, R21E in Sequoyah County, Oklahoma (latitude 35°35' 42" and longitude 95°02'48"). The lake is part of the Arkansas River drainage and the dam lies on the Illinois River at river kilometer 20.6 (approximate) about 11.3 km northeast of Gore and 35.4 km southeast of Muskogee, Oklahoma. The entire lake is located in eastern Oklahoma. Approximately 55 % of the drainage basin is located in eastern Oklahoma with the remainder in western Arkansas. Numerous tributaries are located in the basin. Discussion of these tributaries is deferred to later sections on watershed assessment.

c. General Hydrologic Relationship to Associated Upstream and Downstream Waters: The drainage area consists of 4170 sq. kilometers (1610 sq. miles) (Figure 1). The major tributary into and out of Tenkiller Ferry Lake is the Illinois River. The average discharge at gaging station 07198000 below the dam, completely regulated since 1954, was 1,558 cfs for the period 1940-1990. The maximum was 180,000 cfs on May 11, 1950 and the minimum was 2.0 cfs on Sept. 16, 1959 (R.L. Blaze et al. 1991). Minor tributaries that directly converge with the lake include Caney, Dry, Elk, Sixshooter, Terrapin, Chicken, Snake, Cato, Pine, Salt, Dogwood, Burnt Cabin, Sisemore, and Pettit creeks (Figure 2). Numerous additional unnamed tributaries also directly converge with the lake. The Illinois River flows downstream of the lake for approximately 20.6 km (12.8 mi) where it converges with the Arkansas River. Although many small impoundments are located upstream, no major reservoir exists in the basin. Previously, Lake Frances, near the Oklahoma/Arkansas border on the Illinois River, was the only reservoir located on the mainstem. However, the Lake Frances dam denuded from excessive runoff ca. 1990. Lake Frances emptied and, at the time of this report has not been reconstructed. The United States Environmental Protection Agency (USEPA) has studied sediment stability of the lakebed (Frances) and its effect on suspended sediments in the river (Hill personal communication). These results are discussed in the soils section. Since the elimination of Lake Frances, the inflows into Lake Tenkiller predominantly have been unregulated, while the outflows have been completely regulated via hydraulic releases from the outlet works, since the lake (Tenkiller) began regulated storage in 1952. Hydraulic releases from Lake Tenkiller are determined, in part, from current flows in the Arkansas River as part of the McClellan-Kerr Arkansas River Navigation System. The releases from Lake Tenkiller augment releases from Lake Keystone (located at the confluence of the Arkansas and Cimarron rivers).

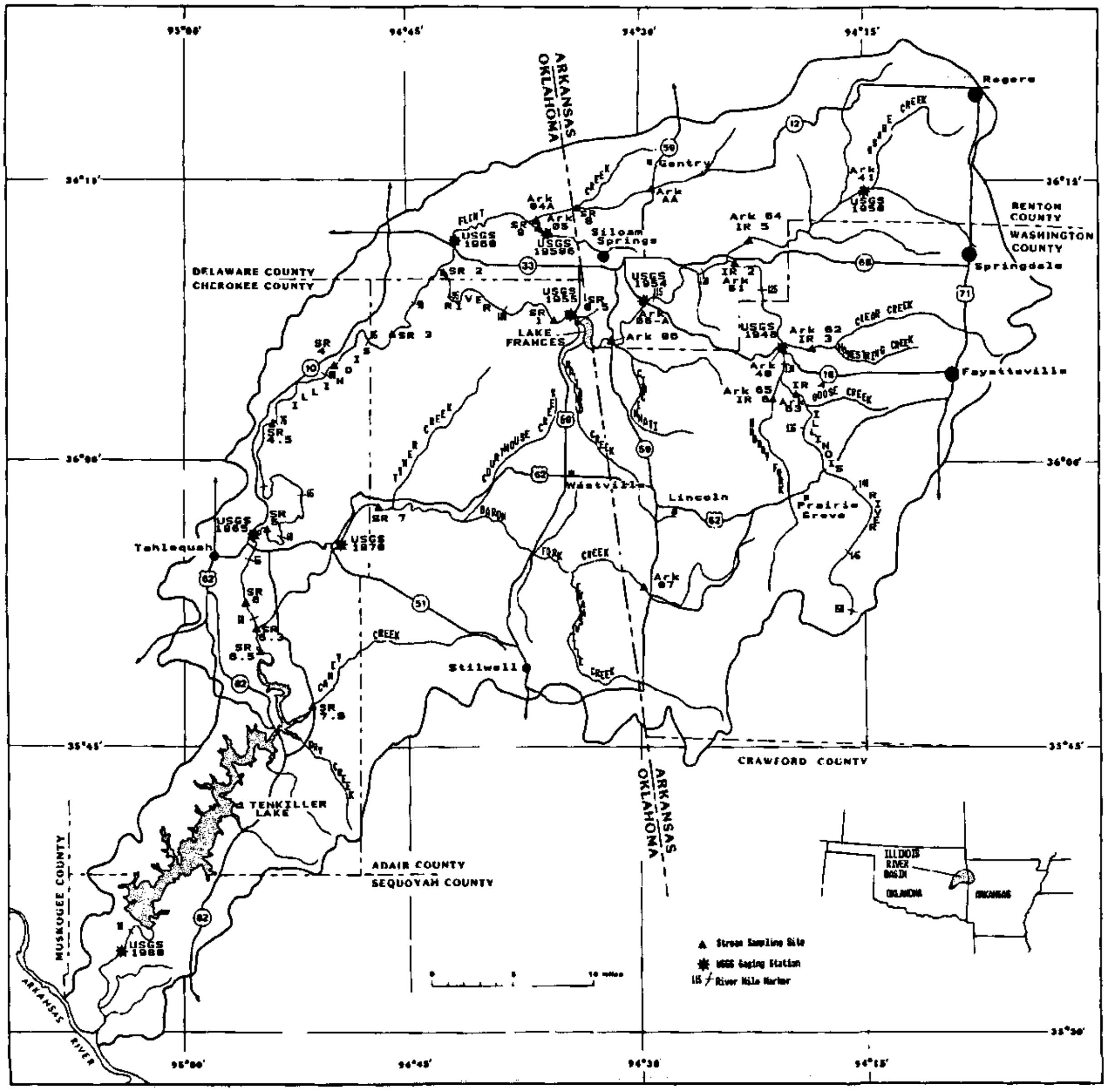



**Figure 1.** Map of the Illinois River Drainage Basin Showing River Mile Locations of Sampling Stations Upstream from Confluence with the Arkansas River.

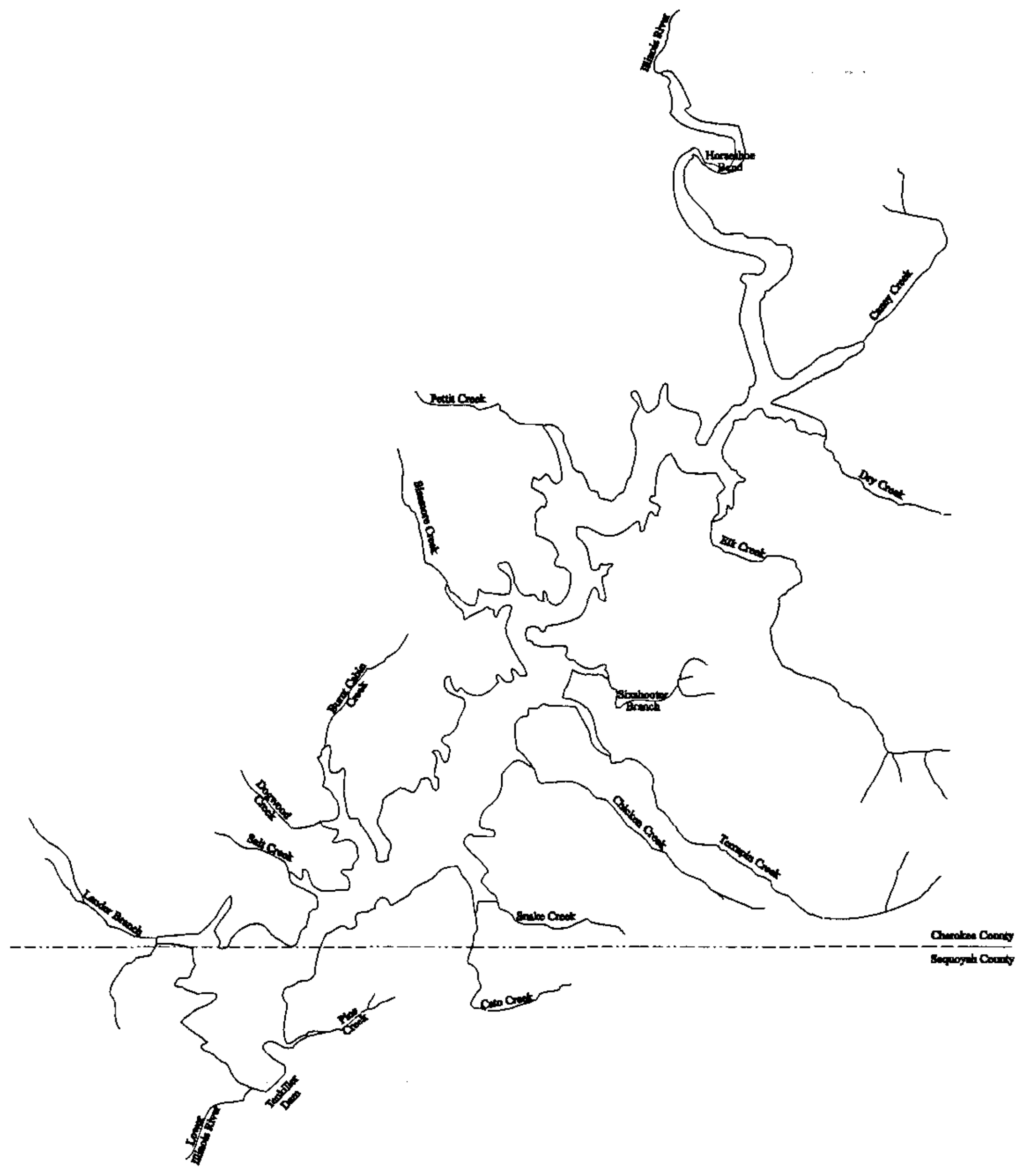



**Figure 2** Lake Tenkiller and Associated Confluent Tributaries.

d. Approved State Water Quality Standards for the Lake are:

(1) "For lakes.....the following beneficial uses are designated:"

- (A) Fish and Wildlife Propagation (Warm Water Aquatic Community) (785:45-5-12).
- (B) Agriculture (785:45-5-13).
- (C) Industrial and Municipal Process and Cooling Water (785:45-5-15).
- (D) Primary Body Contact Recreation (785:45-5-16).
- (E) Aesthetics (785:45-5-19).

(2) "The beneficial use of Public and Private Water Supplies (785:45-5-10) is specifically designated for certain lakes.....otherwise the beneficial uses designated in this paragraph take control.....".

## BENEFICIAL USES AND CRITERIA

**785:45-5-10. Public and Private Water Supplies**

The following criteria apply to surface waters of the state having the designated beneficial use of Public and Private Water Supplies:

(1) **Raw Water Numerical Criteria.** For surface water designated as public and private water supplies, the numerical criteria for substances listed in Table 2 shall not be exceeded.

(2) **Radioactive Materials.**

- (A) There shall be no discharge of radioactive materials in excess of the criteria found in Oklahoma Radiation Protection Regulations, 1969, or its latest revision.
- (B) The concentration of gross alpha particles shall not exceed the criteria specified in (i) through (iv) of this subparagraph, or the naturally occurring concentration, whichever is higher.
  - (i) The combined dissolved concentration of Radium-226 and Radium-228, and Strontium-90, shall not exceed 5 picocuries/liter, and 8 picocuries/liter, respectively.
  - (ii) Gross alpha particle concentrations, including Radium-226 but excluding radon and uranium, shall not exceed 15 picocuries/liter.
  - (iii) The gross beta concentration shall not exceed 50 picocuries/liter.
  - (iv) The average annual concentration of beta particle and photon radioactivity from man-made

Table II. Raw Water Numerical Criteria.

| SUBSTANCES (Total) | NUMERICAL CRITERIA (mg/L) |
|---|---|
| Inorganic Elements: | |
| Arsenic | 0.10 |
| Barium | 1.00 |
| Cadmium | 0.020 |
| Chromium | 0.050 |
| Copper | 1.000 |
| Cyanide | 0.200 |
| Fluoride (at 90°F) | 4.0 |
| Lead | 0.100 |
| Mercury | 0.002 |
| Nitrates (as N) | 10.000 |
| Selenium | 0.010 |
| Silver | 0.050 |
| Zinc | 5.000 |
| | |
| Organic Elements: | |
| Benzidine | 0.001 |
| Detergents (total | 0.200 |
| Methylene blue active substances | 0.500 |
| Phthalate esters (except butylbenzyl) | 0.003 |
| Butylbenzyl | 0.150 |
| 2,4-D | 0.100 |
| 2,4,5-TP Silvex | 0.010 |
| Endrin | 0.0002 |
| Lindane | 0.004 |
| Methoxychlor | 0.100 |
| Toxaphene | 0.005 |

radionuclides in waters having the designated use of Public and Private Water supply shall not produce an annual dose equivalent to the total body or any internal organ greater than 4 millirem/year.

(3) **Coliform Bacteria.**

(A) The bacteria of the total coliform group shall not exceed a monthly geometric mean of 5,000/100 ml at a point of intake for public or private water supply.

(B) The geometric mean will be determined by multiple tube fermentation or membrane filter procedures based on a minimum of not less than five (5) samples taken over a period of not more than thirty (30) days.

(C) Further, in no more than 5% of the total samples during any thirty (30) day period shall the bacteria of the total coliform group exceed 20,000/100 ml.

(D) In cases where both public and private water supply and primary body contact recreation uses are designated, the primary body contact criteria will apply.

(4) **Oil and Grease (Petroleum and Non-Petroleum Related).** For Public and Private Water Supplies, surface waters of the State shall be maintained free from oil and grease and taste and odors.

(5) **General Criteria.**

(A) The quality of the surface waters of the state which are designated as public and private water supplies shall be protected, maintained, and improved when feasible, so that the waters can be used as sources of public and private raw water supplies.

(B) These waters shall be maintained so that they will not be toxic, carcinogenic, mutagenic, or teratogenic to humans.

(6) **Water Column Criteria to Protect for the Consumption of Fish Flesh and Water.**

(A) Surface waters of the State with the designated beneficial use of Public and Private Water Supply shall be protected to allow for the consumption of fish, shellfish and water.

(B) The water column numerical criteria listed in Table 3 to protect human health for the consumption of fish flesh and water shall apply to all surface waters designated with the beneficial use of Public and Private Water Supply.

Table III. Water Column Numerical Criteria to Protect Human Health for the Consumption of Fish Flesh and Water.

| Substances (Total Recoverable) | Criteria ($\mu g/l$) |
|---|---|
| | |
| Aldrin | 0.001273 |
| Arsenic | 0.175 |
| Benzene | 11.87 |
| Chlordane | 0.00575 |
| Dieldrin | 0.001352 |
| DDT | 0.005876 |
| Gamma BHC (Lindane) | 0.1458 |
| Heptachlor | 0.00208 |
| Hexachlorobenzene | 0.009026 |
| Carbon Tetrachloride | 2.538 |
| Chloroform | 56.69 |
| PCB | 0.00079 |
| 2,3,7,8-TCDD (Dioxin) | 0.00000013 |
| 1-1-1 TCE | 3094.0 |
| Cadmium | 14.49 |
| Chromium (Total) | 166.3 |
| Endrin | 0.7553 |
| Ethylbenzene | 3120.0 |
| Lead | 5.0 |
| Mercury | 0.5563 |
| Nickel | 607.2 |
| Pentachlorophenol | 1014.0 |
| Phenol | 20900.0 |
| Silver | 104.8 |
| Toluene | 10150.0 |

Water column criteria to protect human health for the consumption of fish flesh only may be found in 785:45-5-12(9).

**785:45-5-11. Emergency Public and Private Water Supplies**

(a) During emergencies, those waters designated Emergency Public and Private Water Supplies may be put to use.

(b) Each emergency will be handled on a case-by-case basis, and be thoroughly evaluated by the appropriate State agencies and/or local health authorities.

**785:45-5-12. Fish and Wildlife Propagation**

(a) "The narrative and numerical criteria (Table III) in this section are designated to promote fish and wildlife propagation for the fishery classifications of....., Warm Water Aquatic Community......".

(d) Warm Water Aquatic Community means a subcategory of the beneficial use category "Fish and Wildlife Propagation" where the water quality and habitat are adequate to support intolerant climax fish communities and includes an environment suitable for the full range of warm water benthos.

(e) The narrative and numerical criteria shall include:

(1) **Dissolved Oxygen.**

(A) Dissolved oxygen (DO) criteria are designed to protect the diverse aquatic communities of Oklahoma.

(B) Allowable loadings are designed to attain these criteria. For streams with sufficient historical data, the allowable load shall be based on meeting the dissolved oxygen concentration standard at the seven-day, two-year low flow and the appropriate seasonal temperatures. For streams lacking sufficient historical data, or when the appropriate flow is less than one (1) cubic foot per second (cfs), the allowable load shall be based on meeting the dissolved oxygen concentration standard at one (1) cfs and the appropriate seasonal temperature.

(C) Except for naturally occurring conditions, the dissolved oxygen criteria are set forth in Table IV.

(2) **Temperature.**

(A) At no time shall heat be added to any surface water in excess of the amount that will raise the

Table IV. Dissolved Oxygen Criteria.

| Fishery Class | Dates Applicable | D.O. Criteria (Minimum) (mg/L) | Seasonal Temp. (°C) |
|---|---|---|---|
| Warm Water Aquatic Community | | | |
| Early Life Stages | 4/1 - 6/15 | 6.0[2] | 25[3] |
| Other Life Stages | | | |
| Summer Cond. | 6/16 - 10/15 | 5.0[2] | 32 |
| Winter Cond. | 10/16 - 3/31 | 5.0 | 18 |

1 For use in calculation of the allowable load.
2 Because of natural diurnal dissolved oxygen fluctuation, a 1.0 mg/l dissolved oxygen concentration deficit shall be allowed for not more than eight (8) hours during any twenty-four (24) hour period.
3 Discharge limits necessary to meet summer conditions will apply from June 1 of each year. However, where discharge limits based on Early Life Stage (spring) conditions are more restrictive, those limits may be extended to July 1.

temperature of the receiving water more than 2.8°C.

(B) The normal daily and seasonal variations that were present before the addition of heat from other than natural sources shall be maintained.

(C) In streams, temperature determinations shall be made by averaging representative temperature measurements of the cross sectional area of the stream at the end of the mixing zone.

(D) In lakes, the temperature of the water column and/or epilimnion, if thermal stratification exists, shall not be raised more than 1.7°C above that which existed before the addition of heat of artificial origin, based upon the average of temperatures taken from the surface to the bottom of the lake, or surface to the bottom of the epilimnion if the lake is stratified.

(E) No heat of artificial origin shall be added that causes the receiving stream water temperature to exceed the maximums specified below:

(i) The critical temperature plus 2.8°C in warm water and habitat limited aquatic community streams and lakes except in the segment of the Arkansas River from Red Rock Creek to the headwaters of Keystone Reservoir where the maximum temperature shall not exceed 34.4°C.

(F) Water in privately-owned lakes and reservoirs used in the process of cooling water for industrial purposes is not classified as "waters of the state" (see 785:45-1-2), and is exempt from these temperature restrictions, provided the water released from any such lake or reservoir into a stream system shall meet the water quality standards of the receiving stream.

(3) **pH (Hydrogen Ion Activity).** The pH values shall be between 6.5 and 9.0 in waters designed for fish and wildlife propagation; unless pH values outside that range are due to natural conditions.

(4) **Oil and Grease (Petroleum and Non-Petroleum Related).**

(A) All waters having the designated beneficial use of any subcategory of fish and wildlife propagation shall be maintained free of oil and grease to prevent a visible sheen of oil or globules of oil or grease on or in the water.

(B) Oil and grease shall not be present in quantities that adhere to stream banks and coat bottoms of water courses or which cause deleterious effects to the biota.

(5) **Biological Criteria.**

(A) Aquatic life in all waterbodies designated Fish and Wildlife Propagation (excluding waters designated "Trout, put-and-take") shall not exhibit degraded conditions as indicated by one or both of the following:

(1) comparative regional reference data from a station of reasonably similar watershed size or flow, habitat type and Fish and Wildlife beneficial use subcategory designation.

or

(2) by comparison with historical data from the waterbody being evaluated.

(B) Compliance with this criterion shall be based upon, but not limited to such measures as diversity, similarity, community structure, species tolerance, trophic structure, dominant species, indices of biotic integrity (IBI's), indices of well being (IWB's), or other measures.

(6) **Toxic Substances (for Protection of Fish and Wildlife).**

(A) Surface waters of the state shall not exhibit acute toxicity and shall not exhibit chronic toxicity outside the mixing zone.

(B) Procedures to implement these narrative criteria are found in Oklahoma's Continuing Planning Process document, adopted by the Pollution Control Coordinating Board.

(C) Toxicants for which there are specific numerical criteria are listed after (G) of this paragraph.

(D) For toxicants not specified in the table following (G) of this paragraph, concentrations of toxic substances with bio-concentration factors of 5 or less shall not exceed 0.1 of published $LC_{50}$ value(s) for sensitive representative species using standard testing methods, giving consideration to site specific water quality characteristics.

(E) Concentrations of toxic substances with bio-concentration factors greater than 5 shall not exceed 0.01 of published $LC_{50}$ value(s) for sensitive representative species using standard testing methods, giving consideration to site specific water quality characteristics.

(F) Permit limits to prevent toxicity caused by discharge of chlorine and ammonia are determined pursuant to the narrative criteria contained within (A) and (B) of this paragraph.

(G) The following acute and chronic numerical criteria listed in Table V apply to all waters of the state designed with any of the beneficial use sub-categories of Fish and Wildlife Propagation. Equations are presented for those substances whose toxicity varies with water chemistry. Metals listed inTable V are measured as total metals in the water column.

(7) **Fish Tissue Levels.**

(A) Surface waters of the state shall be maintained to prevent bio-concentration of toxic substances in fish, shellfish, or other aquatic organisms.

(B) Concentrations of substances in fish tissue (fillets) in excess of the listed concern levels listed in Table VI shall be cause for further investigation by the appropriate regulatory agency.

(C) Concentrations of substances in fish tissue (fillets) in excess of the listed alert levels listed in Table VI shall be cause for evaluation of discharge permits to determine if point source discharges are causing or contributing to the alert level exceedance.

(D) Waste discharge permit limits shall be modified or established as necessary to restrict the discharge of the exceeded substance where an evaluation determines that point source discharge(s) are causing or contributing to the alert level exceedance.

(E) Non-point sources of these substances should be restricted by application of best management practices in areas where concern or alert levels are exceeded.

(8) **Water Column Criteria to Protect for the Consumption of Fish Flesh.**

(A) Surface waters of the State with the designated beneficial use of Warm Water Aquatic Community, Cool Water Aquatic Community or Trout Fishery shall be protected to allow for the consumption of fish and shellfish.

(B) The water column numerical criteria listed in Table 5 to protect human health for the consumption of fish, shellfish and aquatic life shall apply to all surface waters designated with the beneficial use of Warm Water Aquatic Community, Cool Water Aquatic Community or Trout Fishery.

(9) **Turbidity.**

(A) Turbidity from other than natural sources shall be restricted to not exceed the following numerical limits:

(i) Lakes.................25 Nephelometric Turbidity Units

(B) In waters where background turbidity exceeds these values, turbidity from point sources shall be restricted to not exceed ambient levels.

(C) Numerical criteria listed above apply only to normal stream flow conditions.

(D) Elevated turbidity levels may be expected during, and for several days after, a runoff event.

(E) Nephelometric turbidity unit (NTU) is the method based upon a comparison of the intensity of light scattered by the sample under defined conditions with the intensity of light scattered by a standard reference suspension (formazin). The higher the intensity of scattered light, the higher the turbidity. Readings in NTUs are considered comparable to the previously reported Jackson Turbidity Units (JTU).

**785:45-5-13. Agriculture: Livestock and Irrigation**

(a) The surface waters of the State shall be maintained so that toxicity does not inhibit continued ingestion by livestock or irrigation of crops.

(b) Highly saline water should be used with best management practices as outlined in "Diagnosis and Reclamation of Saline Soils," United States Department of Agriculture Handbook No. 60 (1958).

(c) Guidelines for suitability of water quality for livestock and irrigation purposes are provided in Appendix C of Oklahoma State Water Quality Standards.

(d) For chlorides, sulfates and total dissolved solids at 180°C (see Standard Methods), the arithmetic mean of the concentration of the samples taken for a year in a particular segment shall not exceed the historical "yearly mean standard" determined from the table following subsection (g) of this and 785:45-1-2 calculated for that segment. Furthermore, not more than one (1) in twenty (20) samples randomly collected at a site shall exceed the historical value of the "sample standard" calculated for that segment.

(e) Increased mineralization from other elements such as calcium, magnesium, sodium and their associated anions shall be maintained at or below a level that will not restrict any beneficial use.

(f) The data from sampling stations in each segment are averaged, and the mean chloride, sulfate, and total dissolved solids at 180°C are presented in Oklahoma Water Quality Standards following (g) of this Section. Segment averages shall be used unless more appropriate data are available.

Table V. Numerical Criteria For Toxic Substances ($\mu$g/l).

| Substance | Acute | Chronic |
|---|---|---|
| Aldrin | 3.0 | —— |
| Arsenic | 360.0 | 190.0 |
| Benzene | —— | 2,200.0 |
| Cadmium | e(1.128[ln(hardness)]-1.6774) | e(.7852[ln(hardness)]-3.490) |
| Chlordane | 2.4 | 0.0043 |
| Chlorpyrifos (Dursban) | 0.083 | 0.041 |
| Chromium (Total) | —— | 50.0 |
| Copper | e(0.9422[ln(hardness)]-1.3844) | e(0.8545[ln(hardness)]-1.386) |
| Cyanide | 45.93 | 10.72 |
| DDT | 1.1 | 0.001 |
| Demeton | —— | 0.1 |
| Dieldrin | 2.5 | 0.0019 |
| Endosulfan | 0.22 | 0.056 |
| Endrin | 0.18 | 0.0023 |
| Guthion | —— | 0.01 |
| Heptachlor | 0.52 | 0.0038 |
| Hexachlorocyclohexane (Lindane) | 2.0 | 0.08 |
| Lead | e(1.273[ln(hardness)]-1.460) | e(1.273[ln(hardness)]-4.705) |
| Malathion | —— | 0.10 |
| Mercury | 2.4 | 0.012 |
| Methoxychlor | —— | 0.03 |
| Mirex | —— | 0.001 |
| Nickel | e(.8460[ln(hardness)]+3.3612) | e(.8460[ln(hardness)+1.1645) |
| PCB's (Total) | —— | 0.044 |
| Parathion | 0.065 | 0.013 |
| Pentachlorophenol | e[1.005(pH)-4.830] | e[1.005(pH)-5.290] |
| Selenium | 20 | 5.0 |
| Silver | e(1.72[ln(hardness)]-6.52) | 0.49 |
| 2,4,5-TP Silvex | —— | 10.0 |
| Toluene | —— | 875.0 |
| Toxaphene | 0.78 | 0.0002 |
| Zinc | e(.8473[ln(hardness)+.8604) | e(.8473[ln(hardness)]+.7614) |

Table VI. Alert and Concern Levels in Fish Tissue.

| Substance | Alert Level (mg/kg) | Concern Level (mg/kg) |
|---|---|---|
| Aldrin | 0.3 | 0.15 |
| Chlordane | 0.3 | 0.15 |
| DDT | 5.0 | 2.5 |
| Dieldrin | 0.3 | 0.15 |
| Endrin | 0.3 | 0.15 |
| Heptachlor | 0.3 | 0.15 |
| Mercury | 1.0 | 0.5 |
| PCB's | 2.0 | 1.0 |
| Toxaphene | 5.0 | 2.5 |

(g) The standards document also contains statistical values from historical water quality data of mineral constituents. In cases where mineral content varies within a segment, the most pertinent data available should be used.

**785:45-5-16. Primary Body Contact Recreation**

(a) Primary Body Contact Recreation involves direct body contact with the water where a possibility of ingestion exists. In these cases the water shall not contain chemical, physical or biological substances in concentrations that are irritating to skin or sense organs or are toxic or cause illness upon ingestion by human beings.

(b) In waters designated for Primary Body Contact Recreation the following limits for bacteria set forth in (c) shall apply only during the recreation period of May 1 to September 30. The criteria for Secondary Body Contact Recreation will apply during the remainder of the year.

(c) Compliance with 785:45-5-16 shall be based upon meeting the requirements of one of the three (3) options specified below for bacteria. Upon selection of one (1) group or test method, said method shall be used exclusively over that thirty (30) day period.

(1) Coliform Bacteria: The bacteria of the fecal coliform group shall not exceed a monthly geometric mean of 200/100 ml, as determined by multiple-tube fermentation or membrane filter procedures based on a minimum of not less than five (5) samples collected over a period of not more than thirty (30) days.

Table VII. Water Column Numerical Criteria to Protect Human Health for the Consumption of Fish Flesh.

| Substances (Total Recoverable) | Criteria ($\mu g/l$) |
|---|---|
| Aldrin | 0.001356 |
| Arsenic | 1.399 |
| Benzene | 714.1 |
| Chlordane | 0.00587 |
| Dieldrin | 0.00144 |
| DDT | 0.0059 |
| Gamma BHC (Lindane) | 0.4908 |
| Heptachlor | 0.00214 |
| Hexachlorobenzene | 0.009346 |
| Carbon Tetrachloride | 44.18 |
| Chloroform | 4708.0 |
| PCB | 0.00079 |
| 2,3,7,8-TCDD (Dioxin) | 0.000000138 |
| 1-1-1 TCE | 173100.0 |
| Cadmium | 84.13 |
| Chromium (Total) | 3365.0 |
| Endrin | 0.814 |
| Ethylbenzene | 28720.0 |
| Lead | 25.0 |
| Mercury | 0.5874 |
| Nickel | 4583.0 |
| Pentachlorophenol | 29370.0 |
| Phenol | 4615000.0 |
| Silver | 64620.0 |
| Toluene | 301900.0 |

Further, in no more than 10% of the total samples during any thirty (30) day period shall the bacteria of the fecal coliform group exceed 400/100 ml.

(2) *Escherichia coli* (*E. coli*): *E. coli* shall not exceed a monthly geometric mean of 126/100 ml based upon a minimum of not less than five (5) samples collected over a period of not more than thirty (30) days. No sample shall exceed a 75% one-sided confidence level of 235/100 ml in lakes and high use waterbodies and the 90% one-sided confidence level of 406/100 ml in all other Primary Recreation beneficial use areas. These values are based upon all collected samples. Analysis procedures shall follow EPA-600/4-85/076, "Test Methods for *Escherichia coli* and Enterococci in Water by the Membrane Filter Procedure."

(3) Enterococci: Enterococci shall not exceed a monthly geometric mean of 33/100 ml based upon a minimum of not less than five (5) samples collected over a period of not more than thirty (30) days. No sample shall exceed a 75% one-sided confidence level of 61/100 ml in lakes and high use waterbodies and the 90% one-sided confidence level of 108/100 ml in all other Primary Recreation beneficial use areas. These values are based upon all collected samples. Analysis procedures shall follow EPA-600/4-85/076, "Test Methods for *Escherichia coli* and Enterococci in Water by the Membrane Filter Procedure."

**785:45-5-19. Aesthetics**

(a) To be aesthetically enjoyable, the surface water of the State must be free from floating materials and suspended substances that produce objectionable color and turbidity.

(b) The water must also be free from noxious odors and tastes, from materials that settle to form objectionable deposits, and discharges that produce undesirable effects or is a nuisance to aquatic life.

(c) The following criteria apply to protect this use:

(1) **Color.** Surface waters of the State shall be virtually free from all coloring materials which produce an aesthetically unpleasant appearance. Color producing substances, from other than natural sources, shall be limited to concentrations equivalent to 70 Platinum-cobalt color units.

(2) **Nutrients.** Nutrients from point source discharges or other sources shall not cause excessive growth of periphyton, phytoplankton, or aquatic macrophyte communities which impairs any existing or designated beneficial use.

(3) **Solids (Suspended and/or Settleable).** The surface waters of the State shall be maintained so as to be essentially free of floating debris, bottom deposits, scum, foam and other materials, including suspended substances of a persistent nature, from other than natural sources.

(4) **Taste and Odor.** Taste and odor producing substances from other than natural origin shall be limited to concentrations that will not interfere

with the production of a potable water supply by modern treatment methods or produce abnormal flavors, colors, tastes and odors in fish flesh or other edible wildlife, or result in offensive odors in the vicinity of the water, or otherwise interfere with beneficial uses.

## 2. GEOLOGIC DESCRIPTION OF THE DRAINAGE BASIN

a. Soil Types - Soil types and coverages were downloaded from GRASS 4.0 data files and are presented in TableVIII.

Table VIII. Major Soils in Illinois River Basin.

| Category | $m^2$ | Acres | % Cover |
|---|---|---|---|
| Bodine very cherty silt loam, 1-8% slopes | 2.36e+08 | 5.82e+04 | 3.84e+00 |
| Bodine stony silt loam, 5-15% slopes | 6.53e+07 | 1.61e+04 | 1.06e+00 |
| Bodine stony silt loam, steep | 3.47e+08 | 8.57e+04 | 5.66e+00 |
| Craig cherty silt loam, 1-5% slopes | 5.20e+06 | 1.28e+03 | 8.48e-02 |
| Dickson silt loam, 1-3% slopes | 5.43e+07 | 1.34e+04 | 8.86e-01 |
| Dickson cherty silt loam, 0-3% slopes | 9.11e+07 | 2.25e+04 | 1.49e+00 |
| Etowah silt loam, 0-1% slopes | 6.81e+06 | 1.68e+03 | 1.11e-01 |
| Etowah silt loam, 1-3% slopes | 2.91e+07 | 7.18e+03 | 4.74e-01 |
| Etowah gravelly silt loam, 1-3% slopes | 4.56e+07 | 1.13e+04 | 7.43e-01 |
| Etowah and Greendale soils, 3-8% slopes | 7.38e+07 | 1.82e+04 | 1.20e+00 |
| Gravelly alluvial land | 4.18e+07 | 1.03e+04 | 6.82e-01 |
| Hector complex | 1.50e+08 | 3.71e+04 | 2.44e+00 |
| Hector-Linker fine sandy loams, 1-5% slopes | 3.17e+07 | 7.84e+03 | 5.17e-01 |
| Huntington silt loam | 7.01e+06 | 1.73e+03 | 1.14e-01 |
| Huntington gravelly loam | 1.27e+07 | 3.14e+03 | 2.07e-01 |
| Jay silt loam, 0-2% slopes | 1.61e+07 | 3.97e+03 | 2.62e-01 |
| Lawrence silt loam | 2.43e+06 | 6.00e+02 | 3.96e-02 |
| Linker fine sandy loam, 1-5% slopes | 8.72e+06 | 2.15e+03 | 1.42e-01 |
| Linker fine sandy loam, 3-5% slopes, eroded | 1.99e+06 | 4.91e+02 | 3.24e-02 |
| Linker loam, 3-5% slopes | 6.19e+06 | 1.53e+03 | 1.01e-01 |
| Linker loam, 3-5% slopes, eroded | 2.29e+06 | 5.65e+02 | 3.73e-02 |
| Osage clay loam | 1.98e+06 | 4.88e+02 | 3.22e-02 |
| Parsons silt loam, 0-1% slopes | 2.72e+06 | 6.72e+02 | 4.44e-02 |
| Sogn soils | 7.20e+06 | 1.78e+03 | 1.17e-01 |
| Summit silty clay loam, 0-1% slopes | 3.30e+06 | 8.15e+02 | 5.38e-02 |
| Summit silty clay loam, 1-3% slopes | 4.76e+06 | 1.18e+03 | 7.77e-02 |
| Summit silty clay loam, 3-5% slopes | 2.12e+06 | 5.24e+02 | 3.46e-02 |
| Summit silty clay loam, 3-5% slopes, eroded | 1.07e+06 | 2.65e+02 | 1.75e-02 |
| Taft silt loam | 8.40e+06 | 2.08e+03 | 1.37e-01 |
| Taloka silt loam, 0-1% slopes | 1.07e+06 | 2.64e+02 | 1.74e-02 |
| Borrow Pits | 3.62e+05 | 8.94e+01 | 5.90e-03 |
| Gravel Pits | 3.42e+05 | 8.45e+01 | 5.58e-03 |
| Pits, Quarries | 6.84e+04 | 1.69e+01 | 1.12e-03 |
| Quarries | 3.62e+05 | 8.94e+01 | 5.90e-03 |
| water | 8.89e+07 | 2.20e+04 | 1.45e+00 |
| Baxter silt loam, 1-3% slopes | 1.67e+07 | 4.13e+03 | 2.73e-01 |

| | | | |
|---|---|---|---|
| Baxter cherty silt loam, 1-3% slopes | 2.25e+07 | 5.57e+03 | 3.67e-01 |
| Baxter-Locust complex, 3-5% slopes | 3.35e+07 | 8.29e+03 | 5.47e-01 |
| Captina silt loam, 1-3% slopes | 4.80e+07 | 1.19e+04 | 7.83e-01 |
| Clarksville very cherty silt loam, 1-8% slopes | 1.55e+08 | 3.83e+04 | 2.53e+00 |
| Clarksville stony silt loam, 5-20% slopes | 9.06e+07 | 2.24e+04 | 1.48e+00 |
| Clarksville stony silt loam, 20-50% slopes | 3.49e+08 | 8.63e+04 | 5.69e+00 |
| Collinsville fine sandy loam, 2-5% slopes | 6.35e+05 | 1.57e+02 | 1.03e-02 |
| Dennis silt loam, 1-3% slopes | 8.01e+05 | 1.98e+02 | 1.31e-02 |
| Eldorado silt loam, 1-3% slopes | 1.05e+06 | 2.61e+02 | 1.72e-02 |
| Eldorado silt loam, 3-5% slopes | 1.28e+07 | 3.17e+03 | 2.09e-01 |
| Eldorado soils, 3-12% slopes | 4.99e+06 | 1.23e+03 | 8.14e-02 |
| Elsah soils | 5.26e+07 | 1.30e+04 | 8.58e-01 |
| Hector fine sandy loam, 2-5% slopes | 3.95e+07 | 9.76e+03 | 6.44e-01 |
| Hector-Linker association, hilly | 1.80e+08 | 4.45e+04 | 2.93e+00 |
| Jay silt loam, 0-2% slopes | 1.71e+07 | 4.23e+03 | 2.79e-01 |
| Linker fine sandy loam, 2-5% slopes | 1.19e+07 | 2.95e+03 | 1.94e-01 |
| Locust cherty silt loam, 1-3% slopes | 6.01e+07 | 1.48e+04 | 9.80e-01 |
| Newtonia silt loam, 0-1% slopes | 7.97e+05 | 1.97e+02 | 1.30e-02 |
| Newtonia silt loam, 1-3% slopes | 1.98e+07 | 4.90e+03 | 3.23e-01 |
| Newtonia silt loam, 3-5% slopes | 6.11e+06 | 1.51e+03 | 9.97e-02 |
| Newtonia silt loam, 2-5% slopes, eroded | 1.05e+06 | 2.60e+02 | 1.72e-02 |
| Okemah silty clay loam, 0-1% slopes | 7.51e+06 | 1.85e+03 | 1.22e-01 |
| Okemah silty clay loam, 1-3% slopes | 1.30e+07 | 3.21e+03 | 2.12e-01 |
| Okemah silty clay loam, 3-5% slopes | 4.45e+06 | 1.10e+03 | 7.26e-02 |
| Osage clay | 4.35e+06 | 1.08e+03 | 7.09e-02 |
| Rough stony land | 2.89e+07 | 7.15e+03 | 4.72e-01 |
| Sallisaw silt loam, 0-1% slopes | 3.97e+06 | 9.80e+02 | 6.46e-02 |
| Sallisaw silt loam, 1-3% slopes | 2.08e+07 | 5.14e+03 | 3.39e-01 |
| Sallisaw gravelly silt loam, 1-3% slopes | 3.02e+07 | 7.45e+03 | 4.92e-01 |
| Sallisaw gravelly silt loam, 3-8% slopes | 6.25e+07 | 1.54e+04 | 1.02e+00 |
| Staser silt loam | 1.36e+07 | 3.36e+03 | 2.22e-01 |
| Staser gravelly loam | 3.95e+07 | 9.75e+03 | 6.43e-01 |
| Stigler silt loam, 0-1% slopes | 1.47e+07 | 3.62e+03 | 2.39e-01 |
| Summit silty clay loam, 2-5% slopes, eroded | 4.03e+06 | 9.96e+02 | 6.57e-02 |
| Taloka silt loam, 0-1% slopes | 8.00e+06 | 1.98e+03 | 1.30e-01 |
| Talpa-Rock outcrop complex, 2-8% slopes | 1.68e+07 | 4.16e+03 | 2.75e-01 |
| Talpa-Rock outcrop complex, 15-50% slopes | 5.96e+07 | 1.47e+04 | 9.71e-01 |
| Verdigris silt loam | 2.13e+05 | 5.27e+01 | 3.48e-03 |
| Verdigris soils, frequently flooded | 2.03e+05 | 5.00e+01 | 3.30e-03 |
| Gravel Bar | 6.03e+04 | 1.49e+01 | 9.83e-04 |
| Bodine stony silt loam, steep | 2.94e+06 | 7.26e+02 | 4.79e-02 |
| Cleora fine sandy loam | 2.20e+05 | 5.43e+01 | 3.58e-03 |
| Hector-Linker-Enders complex, 5-40% slopes | 1.18e+08 | 2.92e+04 | 1.93e+00 |
| Linker-Hector complex, 2-5% slopes | 1.57e+07 | 3.88e+03 | 2.56e-01 |

| | | | |
|---|---|---|---|
| Linker-Hector complex, 5-8% slopes | 7.35e+05 | 1.82e+02 | 1.20e-02 |
| Linker and Stigler soils, 2-8 % slopes, severely eroded | 5.92e+05 | 1.46e+02 | 9.65e-03 |
| Mason silt loam | 3.20e+06 | 7.92e+02 | 5.22e-02 |
| Pickwick loam, 1-3% slopes | 3.27e+06 | 8.09e+02 | 5.34e-02 |
| Pickwick loam, 3-5% slopes | 4.39e+06 | 1.09e+03 | 7.16e-02 |
| Pickwick loam, 2-5% slopes, eroded | 6.09e+05 | 1.51e+02 | 9.93e-03 |
| Razort fine sandy loam | 6.29e+05 | 1.55e+02 | 1.03e-02 |
| Rosebloom silt loam, occasionally flooded | 2.18e+05 | 5.38e+01 | 3.55e-03 |
| Rosebloom and Ennis soils, broken | 3.58e+06 | 8.84e+02 | 5.83e-02 |
| Sallisaw complex, 8-30% slopes | 1.49e+05 | 3.69e+01 | 2.44e-03 |
| Sallisaw loam, 1-3% slopes | 2.04e+06 | 5.03e+02 | 3.32e-02 |
| Sallisaw loam, 3-5% slopes | 5.96e+05 | 1.47e+02 | 9.71e-03 |
| Sallisaw loam, 2-5% slopes, eroded | 3.44e+05 | 8.50e+01 | 5.61e-03 |
| Sogn complex, 10-25% slopes | 8.12e+06 | 2.01e+03 | 1.32e-01 |
| Stigler-Wrightsville silt loams, 0-1% slopes | 1.24e+06 | 3.07e+02 | 2.03e-02 |
| Stigler silt loam, 1-3% slopes | 4.22e+06 | 1.04e+03 | 6.87e-02 |
| Stigler silt loam, 2-5% slopes, eroded | 7.38e+04 | 1.82e+01 | 1.20e-03 |
| Summit silty clay loam, 1-3% slopes | 8.77e+05 | 2.17e+02 | 1.43e-02 |
| Summit silty clay loam, 3-5% slopes | 1.89e+06 | 4.68e+02 | 3.09e-02 |
| Baxter cherty silt loam, 3-8% slopes | 1.18e+06 | 2.93e+02 | 1.93e-02 |
| Baxter cherty silt loam, 8-12% slopes | 3.48e+06 | 8.60e+02 | 5.67e-02 |
| Baxter cherty silt loam, 12-20% slopes | 2.86e+06 | 7.07e+02 | 4.66e-02 |
| Baxter cherty silt loam, 20-45% slopes | 2.34e+07 | 5.77e+03 | 3.81e-01 |
| Britwater gravelly silt loam, 3-8% slopes | 2.07e+07 | 5.12e+03 | 3.38e-01 |
| Britwater gravelly silt loam, 8-12% slopes | 7.03e+05 | 1.74e+02 | 1.15e-02 |
| Captina silt loam, 1-3% slopes | 2.07e+08 | 5.11e+04 | 3.37e+00 |
| Captina silt loam, 3-6% slopes | 1.73e+07 | 4.27e+03 | 2.81e-01 |
| Captina silt loam, 3-6% slopes, eroded | 5.86e+07 | 1.45e+04 | 9.55e-01 |
| Craytown silt loam | 2.27e+06 | 5.62e+02 | 3.70e-02 |
| Clarksville cherty silt loam, 12-50% slopes | 2.23e+08 | 5.52e+04 | 3.64e+00 |
| Clarksville cherty silt loam, 12-60% slopes | 1.28e+08 | 3.16e+04 | 2.09e+00 |
| Elsah soils | 3.53e+07 | 8.73e+03 | 5.76e-01 |
| Fatima silt loam, occasionally flooded | 5.63e+06 | 1.39e+03 | 9.17e-02 |
| Guin cherty silt loam, 3-8% slopes | 1.19e+07 | 2.94e+03 | 1.94e-01 |
| Healing silt loam | 5.02e+06 | 1.24e+03 | 8.18e-02 |
| Healing silt loam, occasionally flooded | 2.00e+07 | 4.94e+03 | 3.26e-01 |
| Jay silt loam, 1-3% slopes | 4.59e+07 | 1.14e+04 | 7.49e-01 |
| Jay silt loam, 3-8% slopes | 9.92e+06 | 2.45e+03 | 1.62e-01 |
| Johnsburg silt loam | 4.96e+07 | 1.23e+04 | 8.09e-01 |
| Johnsburg complex, mounded | 4.37e+06 | 1.08e+03 | 7.13e-02 |
| Leaf silt loam | 1.50e+07 | 3.70e+03 | 2.44e-01 |
| Leaf complex, mounded | 5.96e+06 | 1.47e+03 | 9.71e-02 |
| Limestone Outcrop | 2.26e+05 | 5.58e+01 | 3.68e-03 |
| Mayes silty clay loam | 2.76e+06 | 6.82e+02 | 4.50e-02 |

| | | | |
|---|---|---|---|
| Newtonia silt loam, 1-3% slopes | 5.07e+06 | 1.25e+03 | 8.27e-02 |
| Nixa cherty silt loam, 3-8% slopes | 2.98e+08 | 7.36e+04 | 4.86e+00 |
| Nixa cherty silt loam, 8-12% slopes | 8.67e+07 | 2.14e+04 | 1.41e+00 |
| Nixa very cherty silt loam, 3-8% slopes | 2.88e+04 | 7.12e+00 | 4.70e-04 |
| Noark very cherty silt loam, 8-12% slopes | 5.28e+06 | 1.31e+03 | 8.62e-02 |
| Noark very cherty silt loam, 12-20% slopes | 1.25e+07 | 3.09e+03 | 2.04e-01 |
| Noark very cherty silt loam, 20-45% slopes | 3.65e+07 | 9.01e+03 | 5.94e-01 |
| Pembroke silt loam, 1-3% slopes | 7.77e+06 | 1.92e+03 | 1.27e-01 |
| Pembroke silt loam, 3-6% slopes, eroded | 1.21e+07 | 2.98e+03 | 1.97e-01 |
| Pembroke gravelly silt loam, 3-8% slopes, eroded | 6.38e+06 | 1.58e+03 | 1.04e-01 |
| Peridge silt loam, 1-3% slopes | 2.87e+07 | 7.08e+03 | 4.67e-01 |
| Peridge silt loam, 3-8% slopes | 3.09e+07 | 7.63e+03 | 5.04e-01 |
| Pickwick silt loam, 1-3% slopes | 9.01e+06 | 2.23e+03 | 1.47e-01 |
| Pickwick silt loam, 3-8% slopes, eroded | 6.21e+07 | 1.53e+04 | 1.01e+00 |
| Pickwick gravelly loam, 3-8% slopes , eroded | 3.32e+06 | 8.21e+02 | 5.42e-02 |
| Pickwick gravelly loam, 8-12% slopes, eroded | 2.10e+06 | 5.18e+02 | 3.42e-02 |
| Razort loam | 1.38e+07 | 3.42e+03 | 2.26e-01 |
| Razort silt loam, occasionally flooded | 1.77e+07 | 4.36e+03 | 2.88e-01 |
| Razort gravelly silt loam, occasionally flooded | 2.26e+07 | 5.58e+03 | 3.68e-01 |
| Rock land | 6.11e+06 | 1.51e+03 | 9.96e-02 |
| Secesh gravelly silt loam, occasionally flooded | 5.23e+07 | 1.29e+04 | 8.53e-01 |
| Sloan silt loam | 2.65e+07 | 6.54e+03 | 4.31e-01 |
| Sogn rocky silt loam | 5.87e+06 | 1.45e+03 | 9.57e-02 |
| Sogn rocky silt loam, 12 to 40% slopes | 3.07e+05 | 7.58e+01 | 5.00e-03 |
| Sogn-Clarison Complex, 8 to 20% slopes | 1.71e+05 | 4.23e+01 | 2.79e-03 |
| Summit silty clay, 0-1% slopes | 1.86e+07 | 4.59e+03 | 3.03e-01 |
| Summit silty clay, 1-3% slopes | 3.37e+06 | 8.33e+02 | 5.50e-02 |
| Summit silty clay, 3-8% slopes, eroded | 6.66e+06 | 1.64e+03 | 1.09e-01 |
| Summit silty clay, 3-15% slopes, eroded | 2.58e+05 | 6.38e+01 | 4.21e-03 |
| Summit silty clay, 8-12% slopes, eroded | 2.04e+06 | 5.04e+02 | 3.33e-02 |
| Summit stony silty clay, 3-12% slopes, eroded | 4.22e+06 | 1.04e+03 | 6.88e-02 |
| Summit stony silty clay, 12-25% slopes, eroded | 1.72e+06 | 4.26e+02 | 2.81e-02 |
| Summit complex, mounded | 1.07e+06 | 2.65e+02 | 1.75e-02 |
| Taloka silt loam, 0-1% slopes | 4.58e+07 | 1.13e+04 | 7.47e-01 |
| Taloka silt loam, 1-3% slopes | 7.84e+06 | 1.94e+03 | 1.28e-01 |
| Taloka complex, mounded | 6.21e+06 | 1.53e+03 | 1.01e-01 |
| Tonti cherty silt loam, 3-8% slopes | 1.05e+08 | 2.59e+04 | 1.71e+00 |
| Ventris stoney silt loam, 15 to 40% slopes | 3.36e+05 | 8.30e+01 | 5.47e-03 |
| Waben very cherty silt loam, 3-8% slopes | 1.00e+07 | 2.48e+03 | 1.63e-01 |
| Waben very cherty silt loam, 8-12% slopes | 1.51e+06 | 3.73e+02 | 2.46e-02 |
| Allegheny gravelly loam, 3-8% slopes | 2.37e+06 | 5.86e+02 | 3.87e-02 |
| Allegheny gravelly loam, 3-8% slopes, eroded | 3.26e+06 | 8.06e+02 | 5.32e-02 |
| Allegheny gravelly loam, 8-12% slopes, eroded | 2.69e+06 | 6.65e+02 | 4.39e-02 |
| Allegheny stony loam, 8-12% slopes | 4.23e+06 | 1.04e+03 | 6.89e-02 |

| | | | |
|---|---|---|---|
| Allegheny stony loam, 12-40% slopes | 1.01e+07 | 2.50e+03 | 1.65e-01 |
| Allen loam, 3-8% slopes, eroded | 3.68e+06 | 9.10e+02 | 6.00e-02 |
| Allen loam, 8-12% slopes, eroded | 4.12e+06 | 1.02e+03 | 6.72e-02 |
| Allen loam, 12-20% slopes, eroded | 2.04e+06 | 5.05e+02 | 3.33e-02 |
| Allen stony loam, 12-35% slopes | 3.27e+06 | 8.08e+02 | 5.33e-02 |
| Allen soils, 8-20% slopes | 6.18e+06 | 1.53e+03 | 1.01e-01 |
| Allen-Hector complex, 20-40% slopes | 3.64e+07 | 8.99e+03 | 5.93e-01 |
| Allen-Hector complex, 40-55% slopes | 2.44e+06 | 6.02e+02 | 3.97e-02 |
| Apison loam, 1-3% slopes | 1.21e+06 | 2.98e+02 | 1.97e-02 |
| Apison loam, 3-8% slopes, eroded | 1.61e+07 | 3.97e+03 | 2.62e-01 |
| Apison gravelly loam, 3-8% slopes, eroded | 2.99e+06 | 7.38e+02 | 4.87e-02 |
| Cane loam, 3-8% slopes | 1.67e+06 | 4.14e+02 | 2.73e-02 |
| Cherokee silt loam | 2.35e+07 | 5.82e+03 | 3.84e-01 |
| Cherokee complex, mounded | 5.04e+06 | 1.24e+03 | 8.21e-02 |
| Cleora fine sandy loam | 3.77e+07 | 9.32e+03 | 6.15e-01 |
| Elsah gravelly soils | 1.77e+07 | 4.38e+03 | 2.89e-01 |
| Elsah cobbly soils | 1.89e+07 | 4.67e+03 | 3.08e-01 |
| Enders gravelly fine sandy loam, 8-20% slopes | 4.33e+06 | 1.07e+03 | 7.07e-02 |
| Enders gravelly loam, 3-8% slopes | 2.40e+06 | 5.94e+02 | 3.92e-02 |
| Enders gravelly loam, 3-8% slopes, eroded | 1.14e+07 | 2.82e+03 | 1.86e-01 |
| Enders gravelly loam, 3-12% slopes | 5.81e+06 | 1.43e+03 | 9.47e-02 |
| Enders gravelly loam, 8-12% slopes | 2.96e+06 | 7.31e+02 | 4.82e-02 |
| Enders gravelly loam, 8-12% slopes, eroded | 4.32e+06 | 1.07e+03 | 7.05e-02 |
| Enders stony loam, 3-12% slopes | 3.47e+07 | 8.58e+03 | 5.66e-01 |
| Enders stony loam, 12-30% slopes | 1.75e+06 | 4.32e+02 | 2.85e-02 |
| Enders-Allegheny Complex, 8-20% slopes | 2.04e+08 | 5.03e+04 | 3.32e+00 |
| Enders-Allegheny Complex, 20-40% slopes | 2.82e+08 | 6.97e+04 | 4.60e+00 |
| Fayetteville fine sandy loam, 3-8% slopes, eroded | 2.84e+07 | 7.01e+03 | 4.63e-01 |
| Fayetteville fine sandy loam, 8-12% slopes, eroded | 7.40e+06 | 1.83e+03 | 1.21e-01 |
| Fayetteville fine sandy loam, 12-20% slopes, eroded | 2.75e+06 | 6.80e+02 | 4.48e-02 |
| Fayetteville stony fine sandy loam, 12-35% slopes | 4.36e+06 | 1.08e+03 | 7.11e-02 |
| Fayetteville-Hector complex, 20-40% slopes | 1.55e+07 | 3.83e+03 | 2.53e-01 |
| Hector-Mountainburg gravelly fine sandy loams, 3-8% slopes | 1.61e+07 | 3.98e+03 | 2.62e-01 |
| Hector-Mountainburg gravelly fine sandy loams, 8-12% slopes | 5.87e+06 | 1.45e+03 | 9.58e-02 |
| Hector-Mountainburg stony fine sandy loams, 3-40% slopes | 1.14e+08 | 2.81e+04 | 1.85e+00 |
| Linker fine sandy loam, 3-8% slopes, eroded | 1.23e+07 | 3.04e+03 | 2.01e-01 |
| Linker loam, 1-3% slopes | 3.20e+06 | 7.90e+02 | 5.21e-02 |
| Linker loam, 3-8% slopes, eroded | 4.94e+07 | 1.22e+04 | 8.06e-01 |
| Linker gravelly loam, 3-8% slopes, eroded | 1.55e+07 | 3.83e+03 | 2.53e-01 |
| Linker gravelly loam, 8-12% slopes | 1.34e+06 | 3.31e+02 | 2.18e-02 |
| Linker-Mountainburg Association, undulating | 3.86e+06 | 9.53e+02 | 6.29e-02 |

| | | | |
|---|---|---|---|
| Montevallo soils, 3-12% slopes | 5.22e+06 | 1.29e+03 | 8.51e-02 |
| Montevallo soils, 12-25% slopes | 7.60e+05 | 1.88e+02 | 1.24e-02 |
| Mountainburg stony sandy loam, 3-12% slopes | 8.22e+05 | 2.03e+02 | 1.34e-02 |
| Mountainburg stony sandy loam, 12-40% slopes | 8.07e+05 | 1.99e+02 | 1.32e-02 |
| Nella-Enders Association, very steep | 9.89e+06 | 2.44e+03 | 1.61e-01 |
| Nella-Mountainburg Association, rolling | 3.20e+06 | 7.90e+02 | 5.21e-02 |
| Samba silt loam | 9.66e+06 | 2.39e+03 | 1.58e-01 |
| Samba complex, mounded | 1.30e+06 | 3.20e+02 | 2.11e-02 |
| Savannah fine sandy loam, 1-3% slopes | 1.18e+07 | 2.93e+03 | 1.93e-01 |
| Savannah fine sandy loam, 3-8% slopes, eroded | 6.69e+07 | 1.65e+04 | 1.09e+00 |
| Enders fine sandy loam, 3-8% slopes | 2.33e+05 | 5.76e+01 | 3.80e-03 |
| Enders stony fine sandy loam, 12-45% slopes | 2.04e+06 | 5.04e+02 | 3.33e-02 |
| Enders-Mountainburg Association, rolling | 1.57e+07 | 3.89e+03 | 2.57e-01 |
| Enders-Mountainburg Association, steep | 1.32e+06 | 3.26e+02 | 2.15e-02 |
| Leadvale silt loam, 1-3% slopes | 1.55e+06 | 3.82e+02 | 2.52e-02 |
| Leadvale silt loam, 3-8% slopes | 4.73e+06 | 1.17e+03 | 7.70e-02 |
| Linker fine sandy loam, 3-8% slopes | 1.77e+06 | 4.37e+02 | 2.88e-02 |
| Mountainburg gravelly fine sandy loam, 3-8% slopes | 2.18e+06 | 5.39e+02 | 3.56e-02 |
| Mountainburg gravelly fine sandy loam, 8-12% slopes | 2.38e+05 | 5.87e+01 | 3.87e-03 |
| Mountainburg stony fine sandy loam, 3-12% slopes | 1.64e+06 | 4.04e+02 | 2.67e-02 |
| Nella gravelly fine sandy loam, 3-8% slopes | 7.74e+04 | 1.91e+01 | 1.26e-03 |
| Nella-Enders Association, rolling | 3.04e+06 | 7.50e+02 | 4.95e-02 |
| Nella-Enders Association, steep | 7.98e+07 | 1.97e+04 | 1.30e+00 |
| Nella-Mountainburg Association, steep | 2.62e+06 | 6.47e+02 | 4.27e-02 |
| Spadra fine sandy loam, occasionally flooded | 1.24e+07 | 3.08e+03 | 2.03e-01 |
| Totals | 6.13e+09 | 1.52e+06 | 1.00e+02 |

b. Soil Loss to Lake Stream Tributaries - Soil losses were evaluated based upon STORET data collected by the United State Geological Survey monitoring stations in the Illinois River basin. The parameters were STORET codes 530 (residue, total nonfilterable in mg/l) and 70300 (solids, residue evaporated at 180 °C in mg/l). The highest value of total nonfilterable residue was at USGS1955 located just below the Lake Frances lakebed (Figure 3). This statistic probably was influenced by the effects of the previous impoundment because a large portion of the data was collected when the lake was intact. The second largest concentration of total nonfilterable residue was at USGS1948, located on the Illinois River in Arkansas. Although mass loadings from the individual sub-watersheds were not calculated, relative comparison of the concentrations indicated the Baron Fork (USGS1970) diluted the total nonfilterable residue loading from the Illinois River (Figure 3). The downlake station (USGS1980) indicated a higher load than any upstream station which could indicate a larger load escaping the lake than entering

(Figure 3). However, flows downstream from Lake Tenkiller are entirely regulated by reservoir releases which are primarily hypolimnetic. Turbidity trends in the hypolimnion of the reservoir indicated sediment instability surrounding the intake structures (this trend is discussed further in Task 10) and in part explain the high total nonfilterable residues at USGS1980.

For residual dissolved solids (parameter code 70300), USGS 19586, located on a tributary in the Flint Creek subwatershed, indicated the highest concentration, while the lowest mainstem concentration was at USGS 1960, located on Flint Creek just above its confluence with the Illinois River (Figure 3). Osage Creek (USGS1950) and a tributary on the Flint Creek (USGS19586) also indicated higher dissolved solids than the Illinois River mainstem (Figure 3). No dissolved solids data for the Baron Fork (USGS 1970) were available. Finally, unlike total nonfilterable residues, USGS1980 (below Tenkiller Dam) indicated a slightly lower dissolved solids concentration than most of the upstream station but was similar to the concentration at the Illinois River station immediately above the lake (USGS1965).

Based upon these data, estimated soil losses would be biased for Illinois River mainstem stations below the former Lake Frances dam and hence would be of limited use. However, data indicated that Lake Tenkiller transported most of the dissolved solids and induced higher total nonfilterable residues.

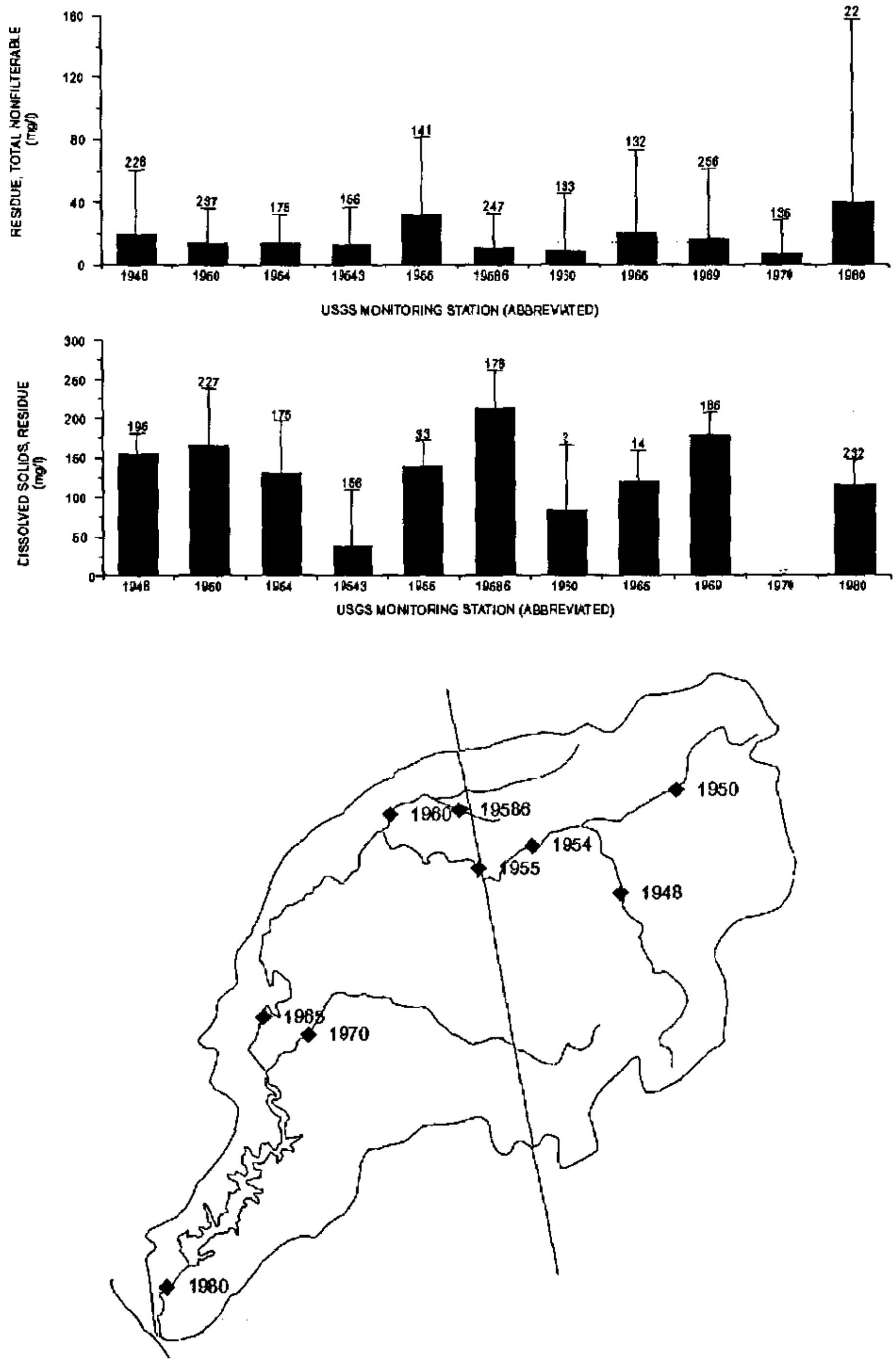



Figure 3. Total Dissolved Solids and Total Nonfilterable Residue at Selected USGS Monitoring Stations in the Illinois River Basin (bars = mean; error bars = 1 sd; numbers above error bars = n).

## 3. PUBLIC ACCESS DESCRIPTION

a. Type of Public Transportation to Access Points: U.S. Highways 62 and 64 and State Highways 51, 82, 10, 10a, and 100 are the major transportation access routes to Tenkiller Lake (Figure 4). Interstate 40 passes south of the lake and provides access for populations from the east and the west, while the Muskogee Turnpike provides access for populations from the west and from the north. The Muskogee Turnpike connects the nearest lake access to the metropolitan area of Tulsa, Oklahoma. Interstate 40 connects Oklahoma City from the western side to southern lake access highways and eastern cities in Oklahoma and western Arkansas. County roads that branch off at many points along the major highways provide access to almost all reaches of the lake. Public transportation to the area, other than that provided by privately-owned vehicles, exists.

b. Amount of Public Transportation to Access Points: The U.S. Army Corps of Engineers and the State of Oklahoma provide public access to the lake in conjunction with recreation areas. The lake has 20 recreation areas. These areas include the overlook, dam site, Tenkiller State Park, Blackgum Landing, Cato Creek Landing, Snake Creek Cove, Chicken Creek, Sixshooter Camp, Cookson Bend, Carlisle Cove, Standing Rock Landing, Elk Creek Landing, Etta Bend, Horseshoe Bend, Carter's Landing, Cherokee Landing, Pettit Bay, Sizemore Cove, Burnt Cabin Ridge, and Strayhorn Landing. The recreation areas include numerous swimming beaches, marinas, boat ramps, fishing docks, and shoreline. Table IX lists the number of facilities available at Tenkiller Lake by type of recreation, and Figure 4 displays the locations of the recreation areas at the lake.

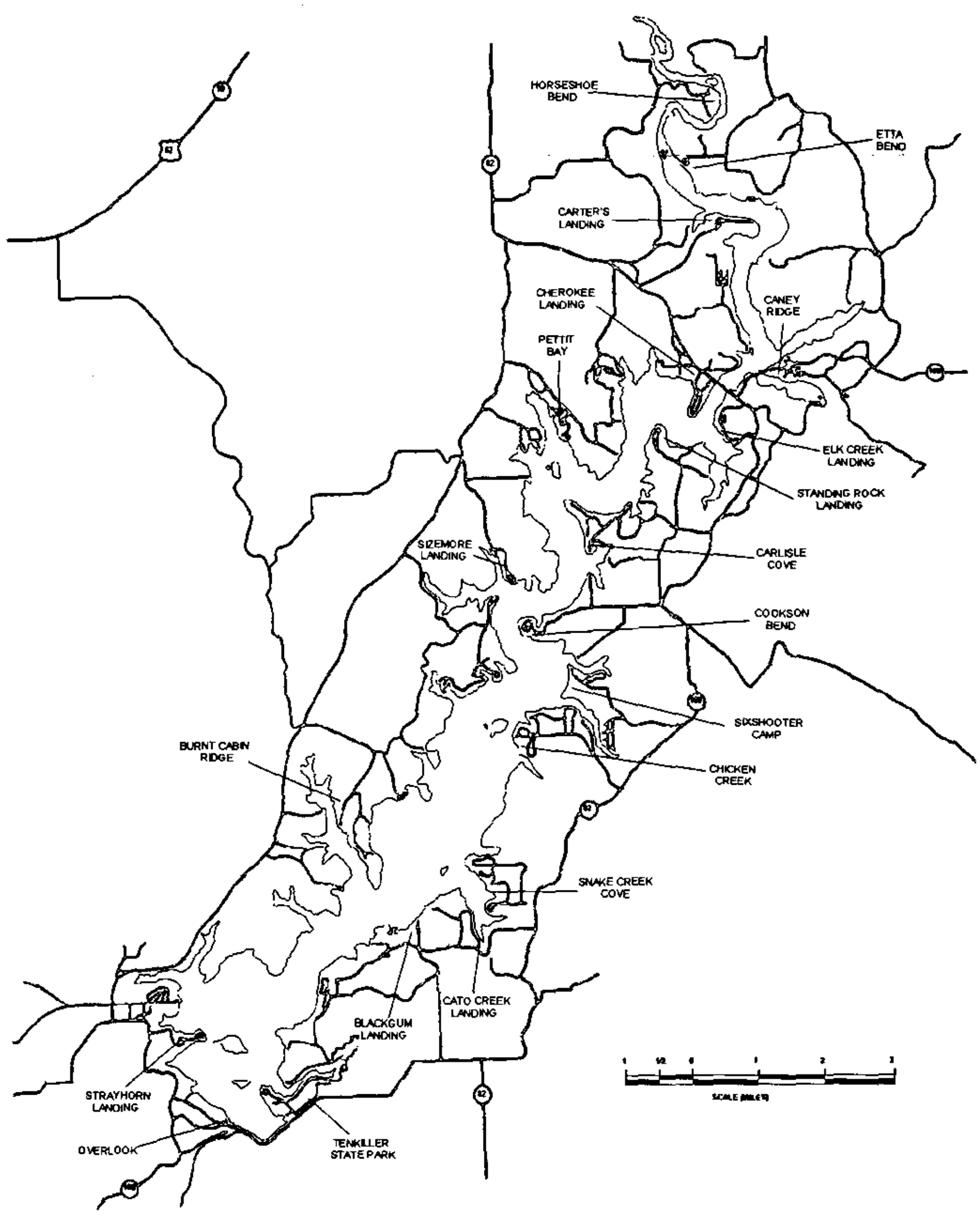



Figure 4. Public Access Points and Routes to Lake Tenkiller (adapted from OWRB 1990).

Table IX. Number and Type of Public Access Facilities at Lake Tenkiller.

| Facilities | Number |
|---|---|
| Concessions Services | 10 |
| Public Camp Grounds | 13 |
| Picnic Areas | 7 |
| Boat Launching Ramps | 16 |
| Nature Trails | 2 |
| Campsite Areas | 13 |
| Campsite Areas with Electrical Connections | 8 |
| Restrooms | 20 |
| Showers | 8 |
| Swimming Beaches | 9 |
| Drinking Water Taps | 19 |
| Trailer Dump Stations | 8 |
| Marina Dump Station | 1 |
| Picnic Shelters | 12 |
| Heated Fishing Docks | 5 |
| Marinas | 10 |

a. Adjacent Population: Tenkiller Lake is located in the southeast part of Cherokee County and the northwest part of Sequoyah County in Oklahoma. Areas surrounding the lake are lightly populated and primarily rural. Individuals living in these areas include retirees who chose to live in areas which have easy access to the recreational opportunities provided by Tenkiller Lake. Other individuals are engaged in a variety of agricultural activities or commute to places of employment in nearby communities such as Tahlequah, Stilwell, Sallisaw, and Muskogee. The largest metropolitan communities near the lake are Fort Smith, 40 miles to the east, Tulsa, 50 miles to the northwest, Oklahoma City, 168 miles to the west, and McAlester, 95 miles to the southwest.

Table X displays the number of persons living in the four counties and metropolitan areas surrounding Tenkiller Lake. The table also provides historic

Table X. Poulation of Counties and Metropolitan Areas Surrounding Lake Tenkiller.

| Surrounding Counties | Year | | |
|---|---|---|---|
| | 1980 | 1990 | 1995 (Est.) |
| Adair | 18,575 | 18,421 | 19,900 |
| Cherokee | 30,684 | 34,049 | 35,800 |
| Muskogee | 67,033 | 68,078 | 71,400 |
| Sequoyah | 30,749 | 33,828 | 35,400 |
| Four-County Area | 147,041 | 154,376 | 162,500 |
| State of Oklahoma | 3,025,487 | 3,145,585 | 3,299,200 |
| Tulsa Metropolitan Area | 657,173 | 708,954 | 750,400 |
| Fort Smith Metropolitan Area | 162,813 | 175,911 | 184,100 |

and projected populations for the surrounding counties. The people living in these areas compose the vast majority of users of Tenkiller Lake; consequently, they are most directly affected by water quality changes.

## 5. HISTORICAL LAKE USES

a. Recreational Uses Up to Present Time: As authorized by Congress, the U.S. Army Corps of Engineers constructed Tenkiller Lake for flood control and hydroelectric power generation. Of the lake's total 1.2 million acre-feet of storage capacity, 346 thousand acre-feet are allocated to hydroelectric power generation, 577 thousand acre-feet are allocated to flood control, and 283 thousand acre-feet are allocated to permanent storage. In addition, 25 thousand acre-feet of the lake are dedicated to water supply under the discretionary authority of the Assistant Secretary of the U.S. Army. Water supply users currently contract with the U.S. Army Corps of Engineers for 20 thousand acre-feet of the lake's storage.

Since its construction, average annual flood control benefits have been $58.1 million. The facility has provided protection for properties along the Illinois River as well as for properties along the Arkansas River. Annually, the generators at Tenkiller dam generate an average of 109 million kilowatt hours of electricity. Power is marketed through the Department of Energy, Southwestern Power Administration to rural and municipal electric cooperatives. Water supply users include municipal, industrial, and agricultural users.

Table XI. Historical Recreational Use of Lake Tenkiller.

| Year | Visitors |
|---|---|
| 1988 | 2,315,800 |
| 1989 | 1,121,000 |
| 1990 | 1,607,000 |
| 1991 | 1,248,000 |
| 1992 | 1,650,400 |
| 1993 | 1,891,500 |
| 1994 | 1,505,600 |

Although recreation is not a project purpose, Tenkiller Lake historically has been a popular recreation area. People from surrounding counties use the lake for a variety of water-related recreational activities. In

1994, recreation visitors spent an estimated 26.6 million visitor hours, or 2.2 million visitor days, at the lake. Because of the lack of consistent methods of measurement, historical trends in visitation at the lake is difficult to assess. For example, the data collected prior to 1989 is not comparable to data collected after that date. However, data since 1989 indicate recent trends in visitation. Table XI displays the visitation at Tenkiller Lake since 1989.

As indicated by Table XI, there are swings in the number of people visiting the lake. A number of factors are related to visitation, including such diverse variables as the economic activity of the area and the weather. The linkages between changes in water quality and visitation is difficult to quantitatively address without extensive study. The relatively high quality of water at Tenkiller Lake is one factor that visitors have mentioned as being a unique feature which attracts them to the lake. Yet, visitation data do not clearly show a simple relationship between variation in lake turbidity and visitation.

Camping, boating, fishing, skiing, sightseeing, and swimming are the primary recreational activities at Tenkiller Lake. According to U.S. Army Corps of Engineer site survey data, 74 % of all visitors to Tenkiller Lake recreation areas are involved in water-related activities. Of all visits, 23.7 % were swimming related and 17.8 % were tied to fishing. Also, Tenkiller Lake is popular among SCUBA divers and for snorkeling because it offers water clarity unsurpassed by any lake in the state of Oklahoma. No statistics are maintained for these specialized recreation activities. However, few lakes in the state of Oklahoma maintain a supply station for accessories and air for SCUBA divers. Lake Tenkiller has such a station (Gene's Aqua Pro Shop, Strayhorn Landing/Marina).

Alterations in water quality will have an impact on recreation activities at Tenkiller Lake. Fort Gibson Lake, Robert S. Kerr Reservoir, Grand Lake, and the Arkansas River Navigation Channel offer water-based recreation opportunities. However, Tenkiller Lake's high water quality, relative to other surrounding bodies of water, has made it unique to the region. If recreationists perceive water quality degradation at Tenkiller Lake, there is a likelihood that use of competing recreation resources, such as those provided by surrounding lakes, would increase and use of Tenkiller Lake would decrease.

Alterations in water quality will impact hydroelectric generation. As quality declines, more demands will be placed on making water releases from Tenkiller dam to accommodate down-stream water quality. Increased nutrients in the lake water equates to less dissolved oxygen in the water. Hydroelectric releases are made from the lower lake strata, a strata that is low in dissolved oxygen during summer stratification. As releases from the lower strata contain less dissolved oxygen, more upper strata water has to be released and less lower strata water needs to be released to balance the standard dissolved oxygen levels mandated by State and Federal water quality standards. Consequently, agencies will require that less hydroelectric power be generated. The alternative is to develop costly structural solutions to ensure that water released from the dam meets dissolved oxygen standards. However, if upper strata are used for releases, the temperature of the tailwaters will be higher and compromise the put-and-take trout fishery. It is imperative that cooler water with adequate dissolved oxygen be released to maintain the downstream fishery.

Alterations will also have a direct impact on water supply users, as decreases in water quality will require more treatment. As there are a number of alternate water supply sources in the area, the decline in water quality may result in users seeking alternate sources. The impact in either situation is more costs associated with treatment or development of alternate water supply sources. Federal agencies and State officials are examining changing hydropower generation to accommodate decreases in dissolved oxygen that have occurred below the dam.

6. AN EXPLANATION IF A PARTICULAR SEGMENT OF THE LAKE USER POPULATION IS OR WILL BE MORE ADVERSELY IMPACTED BY LAKE DEGRADATION

Further degradation would directly impact water supply users, users of hydroelectric power, and recreationists. Water supply users would have to bear the cost of additional treatment. Users of hydroelectric power would have to bear the cost of foregone generation. Because Tenkiller Lake offers unique recreation opportunities within the region, water based recreationists would experience a decline in quality of the recreation experience. Further degradation would indirectly impact those who have an economic interest in the recreation industry in the areas surrounding Tenkiller Lake. Users would most likely seek out alternate recreation areas and activities, and those individuals and businesses providing goods and services to Tenkiller Lake recreationists would experience a loss in income as fewer people visited the lake.

7. WATER USE OF LAKE COMPARED TO OTHER LAKES WITHIN 80 KILOMETER RADIUS

Lakes within 80 km of Lake Tenkiller include Lake Eufaula, Lake Hudson, Lake Eucha, Webber's Falls Reservoir, Robert S. Kerr Reservoir, Greenleaf Lake, Lake Fort Gibson, Lake Sallisaw, Lake Spavinaw, Lake Wister, and

Lake Frances (prior to 1990) in Oklahoma and Lake Wedington, Prairie Grove Lake, Lincoln Lake, Lake Fort Smith, and Hollis Lake in Arkansas. Most of these lakes are small city lakes and do represent a competitive resource to Lake Tenkiller. The major lakes with comparable uses are depicted in Figure 5. Most of the following discussion was taken from the Oklahoma Water Atlas (OWRB 1990).

Lake Eufaula (Oklahoma) is located in the Arkansas River Basin on the Canadian River at mile 27.0. The lake was constructed by the United States Corps of Engineers under the River and Harbor Act of 1946. Construction began in 1956 and was complete in 1964. The lake has a shoreline length of 600 miles and a drainage area of 47,522 square miles. Authorized purposes include flood control, water supply, hydroelectric power, and sediment control. Although not an authorized purpose, recreation is a major use. The USACE and state of Oklahoma maintain 22 recreational areas (Table XII).

Lake Hudson (Oklahoma), also known as Markham Ferry Reservoir, is located on the Grand (Neosho) River at river mile 47 in Mayes County, Oklahoma. The reservoir lies approximately 2 miles northwest of Locust Grove, Oklahoma and 8 miles southeast of Pryor, Oklahoma. Authorized under the Flood Control Act in 1941, the lake was built in 1961 by the Grand River Dam Authority and completed by 1964. Authorized purposes were flood control and hydroelectric power, albeit recreation is a major use. Only two designated public recreational areas exist.

Lake Eucha (Oklahoma) is located in Delaware County, Oklahoma northeast of Tulsa. The lake was built in 1952 and is owned by the city of Tulsa. The lake has 9 recreational areas for public use.

Webber's Falls Reservoir (Oklahoma) is located on the Arkansas River at navigation mile 369 about 5 miles northwest of Webber's Falls, Oklahoma in Muskogee County. The USACE constructed the lock and dam beginning in 1965 which became operational in 1970. Authorized under the River and Harbor Act in 1946, the reservoir and lock structures were built for navigation and hydroelectric power. The lake has 7 recreational areas.

Robert S. Kerr Reservoir (Oklahoma) is located on the Arkansas River at navigation mile 336 in Sequoyah, Haskell, and LeFlore Counties about 8 miles south of Sallisaw. The reservoir is just below Webber's Falls. The Robert S. Kerr lock and dam was built by the USACE in 1964 with completion in 1970. The construction was authorized under the River and Harbor Act in 1946 with navigation, hydroelectric power, and recreation as authorized purposes. The lake has 11 recreational areas.

Greenleaf Lake (Oklahoma) is located immediately west of the south end of Lake Tenkiller and east of Webber's Falls Reservoir, in Muskogee County. The lake was built in 1939 and has one designated recreational area, Greenleaf State Park.

Lake Fort Gibson (Oklahoma) is located on the Grand (Neosho) River at river mile 7.7, in Cherokee and Wagoner Counties. A small portion of the headwaters is located in Mayes County. Construction of the lake was authorized by the Flood Control Act in 1941 and incorporated into the Arkansas River Multiple-purpose Plan by the River and Harbor Act in 1946. The USACE began construction in 1942 and was completed by 1953. Authorized purposes were flood control and hydroelectric power. The lake has 23 recreational areas.

Lake Spavinaw (Oklahoma) is located just downstream from Lake Eucha northeast of Tulsa. The lake was constructed in 1924 and is owned by the city of Tulsa. The lake has 5 recreational areas, Spavinaw State Park, the Permit Office, Picnic Hollow, Beaty Cove, and Owlsey Slough.

Lake Wister (Oklahoma) is located in the Arkansas River drainage on the Poteau River at its confluence with the Fourche Maline in LeFlore County about 2 miles south of Wister, Oklahoma. The lake was constructed by the USACE in 1946-49 under the authorization of the Flood Control Act in 1938. Authorized purposes include flood control, water supply, low flow augmentation, water conservation, and sedimentation. Recreation is an unauthorized use, albeit a valuable economic base. The lake has 6 recreational areas, the overlook, Potts Creek, Victor Area, Wister Ridge, Quarry Island, and a dam site.

Grand Lake O' the Cherokees (Oklahoma) is located on the Grand (Neosho) River at river mile 77 which is just downstream from its confluence with the Spring River. The dam (Pensacola Dam) is located in Mayes County, Oklahoma, about 13 miles southeast of Vinita and is the nation's longest multiple arch structure. Pensacola dam was constructed by the Grand River Dam Authority in 1940 under the authority of the Flood Control Act. Authorized purposes include flood control and hydroelectric power. Grand Lake has the largest shoreline development ratio among all large lakes in the state of Oklahoma. Therefore, it should afford the greatest recreational access per areal unit. However, there are only 5 state parks on the lake, Bernice, Cherokee, Honey Creek, Little Blue/Disney, and Twin Bridges. While most of these parks are quite large and can accommodate many visitors, a large portion of the lake has undergone private development into resorts and homes.

In general, Lake Tenkiller compares favorably with the major lakes and reservoirs within an 80 km radius. Its uses include flood control, hydroelectric power, navigation (indirectly), and recreation. The lake, however, does have a unique use, SCUBA and snorkel diving. The only other lakes in the state that maintain an air station for SCUBA divers are Lake Broken Bow, Lake of the Arbuckles, Lake Murray, and Grand Lake. Only Grand Lake is within 80 km of Lake Tenkiller and it lies at the extreme periphery and does not have the "public access" of Lake Tenkiller. The lakes within the 80 km radius that have comparable accessible facilities include Lake Eufaula, Fort Gibson, and Robert S. Kerr Reservoir. The latter two are in close proximity to Lake Tenkiller and could attract the users of Lake Tenkiller should the water quality continue its decline and the aesthetic quality continue to degrade. Also, recreational use of the Illinois River (e.g., canoe floats) will be impacted by further degradation. Recreational benefits of the lake and the river are inextricably linked and any degradation that affects one will affect the other.

The lake also functions as a water supply for surrounding communities. Continued degradation of water quality will induce higher treatment costs for compliance of drinking water standards. In this instance, unlike its recreational use, the user will be forced to pay the added cost. Few alternatives with comparable costs exist.

The reduction in volume from accelerated sedimentation also bears cost to the user. A smaller lake volume equates to a reduced flood control capacity and reduced quantities for hydroelectric power generation. While reservoirs were not designed to last forever and Tenkiller is no exception, an increased sedimentation from suspended sediment loads and eutrophication will decrease the lifespan of the reservoir and thus decrease its effective uses.

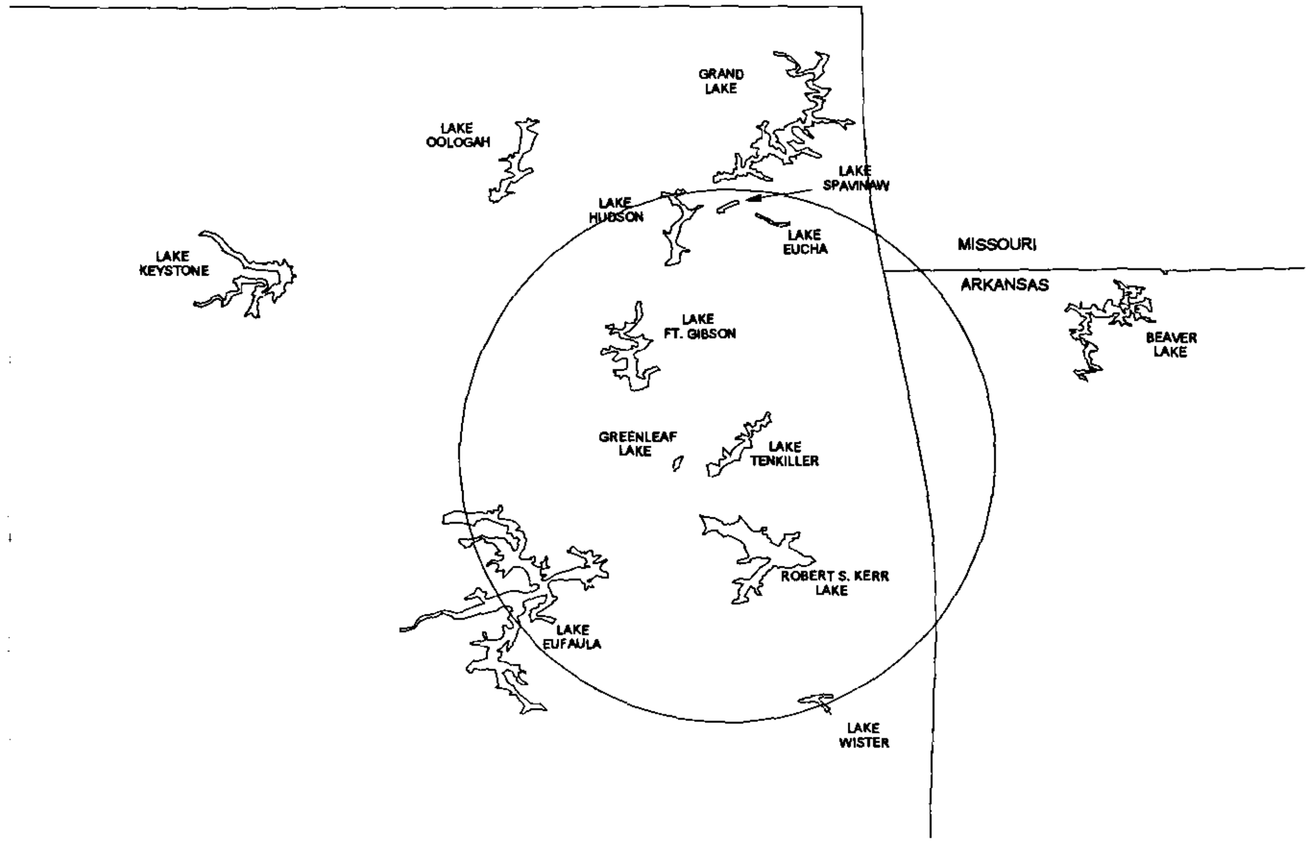



Figure 5. Major Lakes and Reservoirs within 80 km (50 mi) of Lake Tenkiller, Oklahoma (source USGS 1:2,000,000 DLG).

Table XII. Number of Recreational Areas with Listed Facilities at Lakes within 80 km of Lake Tenkiller, Oklahoma (source OWRB 1990).

| Facilities | Eufaula | Eucha | Hudson | Webbers F. | R. S. Kerr | Greenleaf | Ft. Gibson | Spavinaw | Wister | Grand |
|---|---|---|---|---|---|---|---|---|---|---|
| Boat Ramps | 22 | 6 | 2 | 6 | 10 | 1 | 21 | 3 | 5 | 5 |
| Picnic Areas | 19 | 7 | 2 | 6 | 9 | 1 | 18 | 4 | 5 | 5 |
| Designated Campsites | 16 | 5 | 2 | 4 | 7 | 1 | 17 | 1 | 3 | 5 |
| Drinking Water Sources | 17 | 3 | 0 | 6 | 8 | 1 | 16 | 2 | 4 | 1 |
| Group Shelters | 12 | 2 | 0 | 1 | 5 | 1 | 11 | 1 | 4 | 3 |
| Restrooms | 18 | 6 | 2 | 7 | 11 | 1 | 17 | 3 | 5 | 3 |
| Showers | 10 | 1 | 0 | 4 | 0 | 1 | 8 | 1 | 3 | 3 |
| Swimming Beaches | 13 | - | 0 | 1 | 5 | 1 | 6 | - | 0 | 2 |
| Change Houses | 8 | - | 0 | 1 | 1 | 1 | 0 | - | 0 | 2 |
| Trailer Dump Stations | 5 | - | 1 | 4 | 6 | 1 | 10 | 1 | 2 | 4 |
| Electric Outlets | 6 | 1 | 1 | 2 | 2 | 1 | 5 | 1 | 3 | 5 |
| Concession Services | 7 | 1 | 0 | 0 | 1 | 1 | 9 | 0 | 3 | 1 |
| Playgrounds | 0 | 6 | 1 | 0 | 0 | 1 | 0 | 3 | 2 | 4 |
| Nature Trails | 3 | 1 | 0 | 0 | 0 | 1 | 3 | 1 | 0 | 0 |
| Marinas | 2 | 1 | 0 | 0 | 0 | 1 | 1 | 1 | 0 | 0 |
| Golf Courses | 2 | - | 0 | 0 | 0 | - | 1 | - | 0 | 0 |
| Cabins | 0 | - | 0 | 0 | 0 | 1 | 1 | - | 2 | 0 |
| Enclosed Fishing Docks | 2 | 3 | 0 | 0 | 0 | 1 | 1 | 2 | 0 | 2 |
| Boat Rentals | 0 | - | 0 | 0 | 0 | - | 0 | - | 0 | 2 |
| Swimming Pools | 0 | 1 | 0 | 0 | 0 | 1 | 1 | - | 1 | 1 |
| Total Areas | 22 | 9 | 2 | 7 | 11 | 1 | 23 | 5 | 6 | 5 |

## 8. KNOWN POINT SOURCE POLLUTION DISCHARGES WHICH HAVE AFFECTED THE LAKE WATER QUALITY OVER THE LAST 5 YEARS

a. Itemized Inventory: A total of 14 National Pollutant Discharge Elimination System (NPDES) permits with 20 discharges are currently in effect in the Illinois River Basin (Table XIII). These permits are issued through the Oklahoma Department of Environmental Quality (ODEQ) and the Arkansas Department of Pollution Control and Ecology (ADPC&E). Some permits have multiple discharges (Table XIII).

Table XIII. Permitted Discharges in the Illinois River Basin.

| NPDES Permit ID | Name | Discharge ID |
|---|---|---|
| OK0001198 | Cavenham Forest Industries | SUMA |
| OK0027456 | Cherokee Nation of Oklahoma | 001A |
| OK0034070 | Cherokee Nation of Oklahoma (Sequoyah High School) | 001A |
| OK0030341 | Stilwell Area Development Authority | 001A |
| OK0026964 | City of Tahlequah | 001A |
| OK0028126 | City of Westville | 001A |
| AR0020010 | City of Fayetteville | 001A<br>001N<br>002A |
| AR0033910 | USDAFS - Lake Wedington Recreation Area | 001A<br>001N<br>001Q<br>001Z |
| AR0035246 | City of Lincoln | 001A |
| AR0022098 | City of Prairie Grove | 001A |
| AR0043397 | City of Rogers | 001A |
| AR0022063 | City of Springdale | 001A<br>001N |
| AR0020273 | City of Siloam Springs | 001A |
| AR0020184 | City of Gentry | 001A |

b. Abatement Actions Taken or in Progress: The proximity of the Tahlequah discharge to the headwaters of Lake Tenkiller prompted concern on the impact of discharged nutrients upon the lake. Consequently, the City of Tahlequah upgraded its wastewater treatment plant (WWTP) and began operation ca. late 1991-91. This upgrade reduced total P in the effluent (Figure 6). Therefore, calculations on point source contributions from this source were based upon post-implementation data (i.e., 1992-1993).

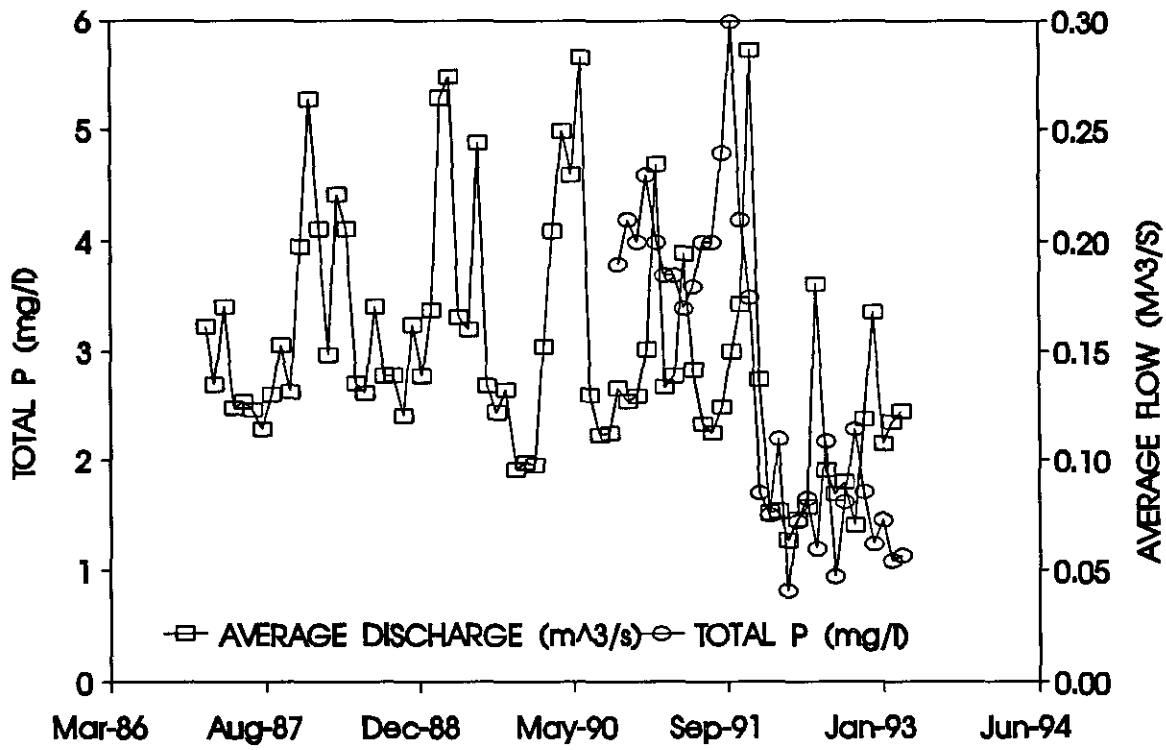



Figure 6. Total P Concentration and Flow for Tahlequah WWTP (OK0026964).

c. Corrective Action Contemplated in the Future: It is anticipated phosphorus controls using best available technologies or BATs will be included in future NPDES permitting in the Illinois River Basin. Suggested reductions and time periods are discussed in later sections of this report.

## 9. LAND USES IN LAKE WATERSHED

a. Each Land Use Classification - Percentage of Whole Watershed: Current land uses in the Illinois River for this study (see Table XIV) were obtained from work sponsored by the U.S. Environmental Protection Agency. Data from 1985 maps were derived from photointerpretation of 1:24,000 scale color infrared aerial film positives flown from August 30 - September 1, 1985. This information was transferred to clear mylar overlays based on USGS 7.5 minute (1:24,000 scale) quadrangles. Digitization and polygon attribution was performed based on these mylar overlays. Land uses derived from this information were reclassed into seven broad categories.

Table XIV. Land Uses for the Illinois River Basin Based on 1985 Data.

| Land Use | Illinois River Basin | Oklahoma Portion | Arkansas Portion |
|---|---|---|---|
| | Area Hectares (%) | Area Hectares (%) | Area Hectares (%) |
| Crop | 5713.20 (1.33) | 1675.44 (0.72) | 4037.76 (2.06) |
| Confined Animal | 1647.99 (0.38) | 232.65 (0.10) | 1415.34 (0.72) |
| Forest | 186199.20 (43.48) | 128955.69 (55.49) | 57243.51 (29.24) |
| Pasture/Range | 211521.87 (49.40) | 91679.76 (39.45) | 119842.11 (61.21) |
| Roads & ROW | 1227.15 (0.29) | 572.40 (0.25) | 654.75 (0.33) |
| Urban | 14980.77 (3.50) | 3028.23 (1.30) | 11952.54 (6.10) |
| Water | 6910.29 (1.61) | 6258.15 (2.69) | 652.14 (0.33) |
| Totals | 428200.47 (100.00) | 232402.32 (100.00) | 195798.15 (100.00) |
| | | (54.27% of Total) | (45.73% of Total) |

Pasture/range and forest lands account for 92.88% of the Illinois River watershed above the Lake Tenkiller Dam. Forest lands (43.48%) generally contribute relatively small amounts of nutrient loading due to their having the best protection from sediment and pollution losses. Low hydraulic activity due to high surface water storage and good permeability because of significant depths of mulch and organic debris tend to discourage excessive surface runoff from forested land (Novotny and Chesters 1988). Pasture and rangelands (49.40%) can be limited pollution hazards when cattle are allowed close to watercourses. In the Illinois River Basin much of the litter produced by confined animal operations (poultry, cattle, and hogs) is distributed on pasture lands as a fertilizer to enhance forage growth and is then susceptible to transport to waterways during runoff events. Croplands, which can be significant sources of nutrient pollution due to transport in runoff, comprise about 1.33% of the basin and tend to be of limited concern in this basin.

Confined animal operations, which includes all poultry rearing, dairy, and hog rearing operations comprise a relatively small percentage of the total basin area (0.38%), but the quantities of litter and waste produced by the animals present significant sources of nutrients which become susceptible to transport to waterways.

A 1989 Soil Conservation Service inventory estimated there were approximately 230 million poultry and 300,000 cattle and swine residing in the basin during an average year. These animals produce an estimated 8.8 billion pounds of manure per year (SCS 1989). Most of this manure remains in the basin and is used as fertilizer on pasture and range lands to enhance forage production. Significant fractions of this fertilizer likely are not incorporated into herbaceous or animal biomass and are susceptible to transport to surface streams during runoff events.

b. Each Land Use Classification - Amount of Nonpoint Pollutant Loading Produced: Nutrient export relationships for the Illinois River Basin were derived from literature values. Beaulac and Reckhow (1982) conducted an extensive literature review of nutrient export studies. They reviewed and evaluated the studies based on sampling design criteria and compiled the results according to land use for both total phosphorus and total nitrogen. Land uses identified in their review included row and non row crops, pasture, mixed agriculture, urban, forest, and feedlots. Ranges of export coefficients for total phosphorus and total nitrogen were determined (Table XV).

Table XV. Quartile Ranges of Land Use/Nutrient Export Coefficients From Beaulac and Reckhow (1982).

| Land Use | Total Phosphorus kg/ha/yr | Total Nitrogen kg/ha/yr |
|---|---|---|
| Row Crops | 0.9 - 5.3 | 4.0 - 21.8 |
| Non Row Crops | 0.6 - 1.5 | 4.1 - 6.5 |
| Pasture | 0.2 - 2.6 | 2.4 - 10.9 |
| Mixed Agriculture | 0.5 - 1.4 | 9.4 - 25.5 |
| Urban | 0.6 - 2.7 | 4.0 - 11.2 |
| Forest | 0.1 - 0.3 | 2.2 - 3.3 |
| Feedlot | 170 - 425 | 1580 - 3425 |

Vaithiyanathan and Correll (1992) examined the effects of land use on phosphorus transport in forested and agricultural soils of the Rhode River watershed, MD. Total phosphorus export coefficients for forest and agriculture were 0.31 ± 0.07 and 2.41 ± 0.85 kg/ha/yr, respectively.

Sharpley et al. (1992) determined phosphorus transport from cropped and uncropped (grass) watersheds in the Southern Plains of the U.S. over a five year period. Export coefficients for grass watersheds ranged from 0.02 to 0.31 kg P/ha/yr, and from 0.29 to 14.90 kg P/ha/yr for cropped watersheds.

Dillon and Kirchner (1975) measured export of total phosphorus from 34 watersheds in Southern Ontario over a 20 month period. Export coefficients for total phosphorus ranged from 0.025 to 0.145 kg/ha/yr for forested areas and from 0.080 to 0.37 kg/ha/yr for areas of mixed forest and pasture.

Clesceri et al. (1986) derived total phosphorus and total nitrogen export coefficients for land uses of forest, mixed forest and agriculture, and agriculture from data for 17 watersheds in Wisconsin. Forest export coefficients ranged from 0.09 to 0.13 kg P/ha/yr and 3.50 to 3.93 kg N/ha/yr. Mixed land use coefficients ranged from 0.14 to 0.24 kg P/ha/yr and 2.50 to 5.61 kg N/ha/yr. Coefficients for agricultural land uses ranged from 0.11 to 0.30 kg P/ha/yr and 3.24 to 16.39 kg N/ha/yr.

Omernick (1977), summarizing data from the National Eutrophication Survey (1972 - 1975), provides some information specific to the Lake Tenkiller area for that period. Over the broad range of land uses in the Illinois River Basin above Lake Tenkiller, mean export coefficients of 0.10 kg/ha/yr for total phosphorus, and 3.63 kg/ha/yr for total nitrogen were calculated. More general values for the Central Region of the U.S. were determined by combining data from watersheds. Regional export coefficients for forest and agricultural land uses were calculated as 0.124 and 0.19 kg P/ha/yr, respectively, and 3.11 and 4.40 kg N/ha/yr, respectively.

In the Final Report for Beaver Lake - Phase I Clean Lakes Feasibility Study (FTN Associates, Ltd. 1992) phosphorus and nitrogen export coefficients were developed for the Beaver Lake Reservoir. Literature including Reckhow (1980) and Omernick (1977) was reviewed in obtaining the estimates of export coefficients for land uses including crop, pasture, urban, forest, and other. Coefficients in this study were calibrated using discharge monitoring data from various sites around the lake. The export coefficient for the land use category of pasture was adjusted to equilibrate the results of the unit area loading method with the loadings calculated from discharge data. A summarization of the export coefficients used in that report follows in Table XVI.

Table XVI. Land Use/Nutrient Export Coefficients Used For Beaver Lake Watershed (AR) (FTN Associates Ltd. 1992).

| Land Use | Total Phosphorus kg/ha/yr | Total Nitrogen kg/ha/yr |
|---|---|---|
| Crop | 0.6 | 4.22 |
| Pasture | 0.2 - 0.65 | 4.0 - 9.5 |
| Urban | 0.6 - 2.0 | 3.0 - 8.0 |
| Forest | 0.025 | 2 |
| Other | 0.045 - 0.05 | 1.2 |

Tables XVII and XVIII show the results of unit area loading estimates of total phosphorus and total nitrogen for the Illinois River Basin above the Lake Tenkiller dam. Unlike the Beaver Lake Report, export coefficients for pasture/range land use were held constant across the entire watershed for this estimate of the sources of nonpoint nutrient pollution. The land use category of confined animal was used to calibrate the unit area loading results with monitoring station water quality data. Here it is assumed that all wastes produced at these locations are susceptible to transport due to rainfall events and erosion. An algorithm for distributing this waste across pasture lands in the immediate vicinity of the confined animal operations was not used. The end result was, however, similar without having to make assumptions of how the potential fertilizer is distributed.

It is important to keep in mind that the estimates of nutrient loading are for the entire Illinois River Basin. Estimates of nutrient loading to the Horseshoe Bend Area of Lake Tenkiller were based primarily on data from USGS gaging stations 07196500 and 07197000 which account for about 75% of the Illinois River Basin. The remaining 25% of the basin not included in that estimate is defined by the Caney Creek watershed and numerous subwatersheds lateral to Lake Tenkiller itself.

Export values were chosen which were comparable to those used in the Beaver Lake Clean Lakes Report based on the assumption that land uses and soil types are relatively similar. Weighting the confined animal operations more heavily (120 kg P/ha/yr and 750 kg N/ha/yr) resulted in showing this land use as the most significant contributor to nutrient transport to Lake Tenkiller despite the limited area it occupies in the basin. Pasture and range lands become the second leading contributor to nutrient transport due to the area this land use occupies within the basin.

The unit area loading results compare favorably to the results of the calculated average annual nonpoint source loading quantities calculated at the Horseshoe Bend area of Lake Tenkiller. Approximately 75% of the Illinois River Basin drains to the Horseshoe Bend area of Lake Tenkiller, and the calculated average annual nonpoint phosphorus load calculated there is 190,078 kg/yr. The total phosphorus load calculated for the entire basin is about 36% larger at 257,748 kg/yr. It is estimated that about 1.7 million kg/yr nitrogen enter the Horseshoe Bend area of Lake Tenkiller from nonpoint sources, and the unit area loading method for the entire basin estimates 2.2 million kg/yr nitrogen.

Unit area loading methods suggest that 76.73% of the nonpoint phosphorus load reaching Lake Tenkiller are a result of manures produced by animal operations within the basin. Extrapolation of this result suggests that a similar percentage of the phosphorus load entering Lake Tenkiller at Horseshoe Bend are from the same sources (76.73% * 190,078 average nonpoint source kg P/yr = 145,847 kg P/yr from animal litter and manures produced within the basin).

Unit area loading methods suggest that 56.37% of the nitrogen load reaching Lake Tenkiller is a result of animal manures produced within the basin, and again, extrapolation suggests that 952,078 kg N/yr of the total nonpoint source nitrogen load entering Lake Tenkiller at the Horseshoe Bend area can be attributed to animal manures produced within the basin.

Table XVII. Calculation of Total Phosphorus Nonpoint Source Loading to Lake Tenkiller From the Illinois River Basin Using Unit Area Loading Methods.

| Land Use | Unit Area Loading Factor kg P/ha/yr | Calculated Load | | | % Total |
|---|---|---|---|---|---|
| | | Oklahoma kg P/yr | Arkansas kg P/yr | Total kg P/yr | |
| Crop | 0.6 | 1005 | 2423 | 3428 | 1331 |
| Confined Animal | 120 | 27918 | 169841 | 197759 | 76.73 |
| Forest | 0.025 | 3224 | 1431 | 4655 | 1.81 |
| Pasture/Range | 0.2 | 18336 | 23968 | 42304 | 16.41 |
| Roads & ROW | 0.5 | 286 | 327 | 614 | 0.24 |
| Urban | 0.6 | 1817 | 7172 | 8988 | 3.49 |
| Water | 0 | 0 | 0 | 0 | 0.00 |
| Totals | | 52586 | 205162 | 257748 | 100.00 |

Table XVIII. Calculation of Total Nitrogen Nonpoint Source Loading to Lake Tenkiller From the Illinois River Basin Using Unit Area Loading Methods.

| Land Use | Unit Area Loading Factor kg N/ha/yr | Calculated Load | | | % Total |
|---|---|---|---|---|---|
| | | Oklahoma kg N/ha/yr | Arkansas kg N/ha/yr | Total kg N/ha/yr | |
| Crop | 5 | 8377 | 20189 | 28566 | 1.30 |
| Confined Animal | 750 | 174488 | 1061505 | 1235993 | 56.37 |
| Forest | 2 | 257911 | 114487 | 372398 | 16.98 |
| Pasture/Range | 2.4 | 220031 | 287621 | 507652 | 23.15 |
| Roads & ROW | 2.5 | 1431 | 1637 | 3068 | 0.14 |
| Urban | 3 | 9085 | 35858 | 44942 | 2.05 |
| Water | 0 | 0 | 0 | 0 | 0.00 |
| Totals | | 671323 | 1521296 | 2192620 | 100.00 |

c. Nutrient Loading Calculations at Horseshoe Bend: A statistical software package called FLUX (Walker, 1987) was used to determine nutrient loads at USGS 07196500 and 07197000 based on data from water years (WTRs) 1980 - 1993. The data from these two USGS gaging stations combined, USGS 07196500 on the Illinois River near Tahlequah, OK, and USGS 07197000 on the Baron Fork Creek near Eldon, OK, are assumed to represent fairly accurately the quality of water entering Lake Tenkiller at Horseshoe Bend. The drainage area for these two streams accounts for approximately 75% of the Illinois River watershed above the Lake Tenkiller dam.

Data were retrieved from the EPA STORET database for WTRs 1980 through 1993. There were, respectively, 139 and 127 total phosphorus observations, and 42 and 38 total nitrogen observations, at each of these sites. Instantaneous stream discharge values were not available for a majority of the concentration observations, thus, mean daily stream discharge data, corresponding to the collection date of the concentration observations, were obtained from the USGS ADAPS database. Total nitrogen observations were either measured total nitrogen, or calculated by summing total Kjeldahl nitrogen and nitrite plus nitrate analysis data for a given sample collection date.

The matched concentration and discharge data were divided into three strata based on stream discharge values for the entire period of record at each gaging station. The first stratum (low flow) included all discharges less than one-half of the long-term mean discharge ($Q_\mu/2$). The second stratum (medium flow) included all those observations with discharge equal to or greater than $Q_\mu/2$ to less than two times mean discharge ($Q_\mu$*2) . The third stratum (high flow) included all those observations with discharge equal to or greater than $Q_\mu$*2.

A flow-weighted mean concentration method was used to calculate average annual loading. Flow-weighted concentration is calculated as:

$$c_{fw}=c_i*(Q_\mu/q_i)$$

where

| | | |
|---|---|---|
| $c_{fw}$ | = | flow-weighted concentration, |
| $c_i$ | = | $i^{th}$ observed concentration, |
| $Q_u$ | = | mean discharge, and |
| $q_i$ | = | $i^{th}$ observed discharge. |

Flow-weighted concentrations were calculated for all observations within a stratum using the mean discharge ($Q_u$) for that stratum. The mean flow-weighted concentration for each stratum was calculated as the sum of all flow-weighted concentrations in the stratum divided by the number of observations. The average annual load contributed by each stratum was then calculated as the mean flow-weighted concentration for stratum *i* multiplied by the mean discharge for stratum *i* multiplied by the percent frequency of observed flows within stratum *i*. Total average annual load was calculated as the sum of the annuals loads of each stratum.

The average annual total phosphorus load for the period was calculated to be 197,932 kg P/yr at USGS 07196500 (standard deviation = 23,007 kg/yr), and 72,625 kg P/yr at USGS 07197000 (standard deviation = 26,650 kg/yr), giving a total combined load of 270,557 kg P/yr (standard deviation = 24,812 kg/yr) potentially entering Lake Tenkiller at Horseshoe Bend from these two tributaries. A 95% confidence interval of total phosphorus load entering Lake Tenkiller from these two combined sources ranges from 229,741 to 311,373 kg P/yr for WTRs 1980-1993.

In addition to this total, there are three point discharges, below the confluence of the Illinois River and Baron Fork Creek and above the Horseshoe Bend area of Lake Tenkiller, from the Tahlequah, OK. wastewater treatment plant, the Cherokee Nation of Oklahoma discharge, and the Cherokee Nation Sequoyah School discharge. Data from EPA's Permit Compliance System (PCS) database for Tahlequah's discharge from 1992 - 1993 show an average annual total phosphorus load of 4670 kg P/yr (standard deviation = 1540 kg/yr). The combined discharge from the Cherokee Nation and School is estimated as 530 kg P/yr.

The average annual total nitrogen load was calculated to be 1,947,795 kg N/yr (standard deviation = 106,563 kg/yr) at USGS 07196500 and 631,183 kg N/yr (standard deviation = 57,280 kg/yr) at USGS 07197000 giving a combined total of 2,578,978 kg N/yr (standard deviation = 86,732 kg/yr) potentially entering Lake Tenkiller from these two sources. A 95% confidence interval for average total nitrogen load from these two sources for the period 1980 - 1993 then is 2,436,304 to 2,721,652 kg N/yr.

Additional total nitrogen loading from Tahlequah's treatment plant was estimated from ammonia data in the PCS report for the period 1991 - 1993 as 17,010 kg N/yr (standard deviation = 15,975 kg/yr). The combined average total nitrogen load from the Cherokee Nation and School was estimated as 1400 kg N/yr.

High flows contributed 66.3% of the annual total phosphorus load at USGS 07196500 and 82.0% at USGS 07197000. Data from other gaging stations within the Illinois River Basin also indicate that approximately 70% of the annual load of total phosphorus is transported during high flows, 25% during medium flows, and 5% during low flows.

High flows contributed 52.9% of the annual total nitrogen load at USGS 07196500 and 60.8% at 07197000. Data from other gaging stations within the Illinois River Basin also indicate that approximately 55% of the annual load of total nitrogen is transported during high flows, 37.5% during medium flows, and 7.5% during low flows.

An EPA screening method (Mills et al., 1985) was used to approximate background nutrient loads which represent the "chemical ... composition of surface waters which would result from natural causes and factors". While this concept may be somewhat artificial in that few, if any, waterbodies in the U.S. remain unaffected by human activity, it can provide an estimate of a "baseline level of pollution which cannot be eliminated by local or area-wide water quality management" (Mills et al. 1985). Background pollution levels could be determined by measuring water quality levels in areas which are free of human activity. In the absence of such data, approximations can be made using information from the USGS Hydrological Benchmark Network (McElroy et al. 1976). These data represent data from monitoring stations considered free of human disturbance. Also, concentration data from the U.S. National Eutrophication Survey (Omernick 1997) for areas of "≥90% Forest" category can be used to estimate background concentrations for three major regions of the U.S.

United States Geological Survey Water Resources Data for Oklahoma (Blazs et al. 1991) was used to obtain the drainage areas of the Illinois River Basin watershed above USGS 07196500 and 07197000, as well as average annual streamflow for these two gaging stations. Average annual runoff was determined by dividing average annual streamflow by the drainage area. Estimates of background concentration were obtained from Omernick (1977) for the Central U.S. Region category of "≥90% forest". Background concentration of total nitrogen is estimated as 0.501 mg as N/l, and 0.020 mg P/l for total phosphorus.

Background loads were determined by multiplying the average annual runoff by the background concentration. The combined background load for gaging stations 07196500 and 07197000 was estimated as 25,000 kg P/yr total phosphorus, and 550,000 kg N/yr total nitrogen.

Over a period of time and stream distance, average nutrient concentrations should decrease from an upstream point to a downstream point, assuming no additional nutrient input between the points, due to biological uptake of nutrients and settling. The rate for this removal can be calculated if instream data are available and an assumption is made that no significant additional sources of nutrient pollution enter the stream between the two points.

First order kinetic rates for instream removal of nutrients due to biological uptake and settling were calculated from differences in total phosphorus and total nitrogen concentrations at two sets of water quality monitoring sites on the Illinois River in Oklahoma. Water quality data retrieved from STORET for 1986 to 1993 for monitoring stations SR1 and SR2, and SR3 and SR4 were paired by date, and USGS ADAPS data were used to determine flow regimes for the dates with corresponding data. It was assumed that no significant nutrient inputs occurred between the upstream and downstream stations.

The kinetic decay rate was then calculated as:

$$k = \frac{\ln\left(\frac{lc_s}{lc_o}\right)}{s}$$

where

$k$ = instream first order kinetic decay rate,

$c_o$ = initial average concentration at the upstream station,

$c_s$ = average concentration at the downstream station, and

$s$ = the distance between the stations in miles (Nemerow 1985).

First order kinetic decay rates were calculated for three flow regimes discussed earlier for both total phosphorus and for total nitrogen concentration observations. Kinetic decay rates calculated from the two pairs of water quality monitoring stations for low, medium, and high flows were averaged to obtain an estimate of the rate for the Illinois

River and its tributaries as a whole. First order kinetic decay rates for total phosphorus were calculated as -0.036 for low flows, -0.026 for medium flows, and -0.017 for high flows. First order kinetic decay rates for total nitrogen were calculated as -0.028 for low flows, -0.014 for medium flows, and -0.010 for high flows. The development of the kinetic decay rate then allows for estimates of the contribution of point source nutrients to total nutrient loads at some point downstream given varying flow using the equation:

$$L_s = L_o e^{-ks}$$

where

$L_s$ = load remaining at some distance downstream,
$L_o$ = the initial load,
$k$ = the first order kinetic decay rate, and
$s$ = the distance downstream.

The rates calculated above were used to estimate the annual quantity of nutrient loads from municipal point source dischargers in the Illinois River Basin arriving at the Horseshoe Bend area of Lake Tenkiller (see Tables XIX and XX). The method provides an estimate of 12,547 kg P/yr and 61,605 kg N/yr entering the Horseshoe Bend area of Lake Tenkiller from point sources. Estimates of municipal point source nutrient loads were obtained from personal communications with William Keith of the Arkansas Department of Pollution Control and Ecology (ADPCE) and Mark Derichsweiler of the Oklahoma Department of Environmental Quality (ODEQ), and PCS output for dischargers in the Illinois River Basin. The ADPCE data were estimates for calendar years 1990 to 1991 or later. Data from ODEQ were estimates based on data from 1992 to 1993. Data from PCS for 1992 to 1993 were used for Tahlequah since their newer system went on-line in 1991. Data from PCS estimates for Fayetteville's discharge were made for the same period. The 1993 Curtis Report (OK Dept. of Ag., and Plant Industry and Consumer Services) was used to obtain estimates for nutrient discharges from plant nurseries in the Oklahoma portion of the Illinois River Basin. Point source discharges were distributed between low, medium, and high flows based on average frequency percent of flows in these strata of USGS gaging stations within the Illinois River Basin (07194800, 07195000, 07195400, 07195500, 07196000, 07196500, 07196900, and 07197000) calculated as 58.1% low flow, 31.5 % medium flow, and 10.4% high flow.

These rates were also used to estimate the instream decay of nutrient loads calculated at USGS gaging stations 07196500 and 07197000, and point discharges from Tahlequah and Cherokee Nation facilities, which are several miles above the Horseshoe Bend area of Lake Tenkiller (see Tables XXI and XXII). Estimates of total annual adjusted loads entering Lake Tenkiller at Horseshoe Bend were 227,625 kg P/yr and 2,300,585 kg N/yr.

Tables XXIII and XXIV summarize the estimated distribution of nutrient loads between background, point and nonpoint sources. For all flow regimes combined, nonpoint sources of nitrogen are estimated to represent 73.4% of the total nitrogen entering Lake Tenkiller at the Horseshoe Bend area. Point sources are estimated to contribute 2.7% of the total. Most stream flows fall within the 'Medium Flow' stratum and here the results are similar with 73.9% from nonpoint sources, 2.2% from point sources, and 23.9% as background (Table XXIII). As would be expected, the point source contribution is greatest at low flows and least under high flow conditions.

The estimates for total phosphorus loads at the Horseshoe Bend area of Lake Tenkiller were similar. Considering all flow regimes combined, point sources contributed 5.5%, and nonpoint sources 83.5% of the total phosphorus load. Under low flow conditions it was estimated that point sources contribute 44.1% of the total phosphorus load, and this percent contribution declines dramatically with increasing stream flow rates.

Table XIX. Estimates of Point Source Discharge Quantities of Total Phosphorus to the Horseshoe Bend Area of Lake Tenkiller (1991 to 1993 data).

| Discharger | Estimated Load at Source (kg P/yr) | Distance to Horseshoe Bend (mi) | Estimated Corrected Load at Horseshoe Bend | | | Estimated Annual Total Load (kg P/yr) |
|---|---|---|---|---|---|---|
| | | | Low Flow (kg P/yr) | Medium Flow (kg P/yr) | High Flow (kg P/yr) | |
| Prairie Grove | 1200 | 100 | 19 | 28 | 23 | 70 |
| Rogers | 21600 | 99 | 355 | 519 | 417 | 1292 |
| Fayetteville | 4500 | 97 | 80 | 114 | 90 | 283 |
| Springdale | 43150 | 95 | 820 | 1150 | 893 | 2862 |
| Lincoln | 1200 | 81 | 38 | 46 | 31 | 115 |
| Gentry | 1700 | 68 | 85 | 91 | 56 | 232 |
| Siloam Springs | 10000 | 62 | 623 | 628 | 362 | 1614 |
| Watts | 500 | 62 | 31 | 31 | 18 | 81 |
| Westville | 2900 | 28 | 615 | 441 | 187 | 1243 |
| Midwestern Nursery | 600 | 14 | 211 | 131 | 49 | 391 |
| Tahlequah | 4700 | 6 | 2200 | 1267 | 441 | 3908 |
| Cherokee Nation | 530 | 5 | 257 | 147 | 51 | 454 |
| Total | 92580 | | 5335 | 4593 | 2619 | 12547 |

Table XX. Estimates of Point Source Quantities of Total Nitrogen to the Horseshoe Bend Area of Lake Tenkiller (1991 - 1993 Data).

| Discharger | Estimated Load (kg N/yr) | Distance to Horseshoe Bend (mi) | Estimated Corrected Load at Horseshoe Bend | | | Estimated Annual Total Load (kg N/yr) |
|---|---|---|---|---|---|---|
| | | | Low Flow (kg N/yr) | Medium Flow (kg N/yr) | High Flow (kg N/yr) | |
| Prairie Grove | 3000 | 100 | 106 | 233 | 115 | 454 |
| Rogers | 54000 | 99 | 1962 | 4254 | 2087 | 8303 |
| Fayetteville | 11250 | 97 | 432 | 911 | 444 | 1787 |
| Springdale | 107875 | 95 | 4384 | 8987 | 4339 | 17710 |
| Lincoln | 3000 | 81 | 180 | 304 | 139 | 623 |
| Gentry | 4250 | 68 | 368 | 517 | 224 | 1108 |
| Siloam Springs | 25000 | 62 | 2560 | 3306 | 1399 | 7264 |
| Watts | 1300 | 62 | 133 | 172 | 73 | 378 |
| Westville | 7300 | 28 | 1936 | 1554 | 574 | 4064 |
| Midwestern Nursery | 5000 | 14 | 1963 | 1295 | 452 | 3710 |
| Tahlequah | 17010 | 6 | 8354 | 4926 | 1666 | 14947 |
| Cherokee Nation | 1400 | 5 | 707 | 411 | 138 | 1257 |
| Total | 240385 | | 23086 | 26870 | 11648 | 61605 |

Table XXI. Calculated Estimated Quantities for Sources of Total Phosphorus Loads to the Horseshoe Bend Area of Lake Tenkiller.

| Discharger | Estimated Load at Source (kg P/yr) | Distance to Horseshoe Bend (mi) | Estimated Corrected Load at Horseshoe Bend | | | Estimated Annual Total Load (kg P/yr) |
|---|---|---|---|---|---|---|
| | | | Low Flow (kg P/yr) | Medium Flow (kg P/yr) | High Flow (kg P/yr) | |
| Gage 07196500 | 197932 | 9.7 | 12005 | 38607 | 111279 | 161891 |
| Gage 07197000 | 72625 | 8.9 | 2056 | 8125 | 51191 | 61371 |
| Tahlequah | 4700 | 6 | 2200 | 1267 | 441 | 3908 |
| Cherokee Nation | 530 | 5 | 257 | 147 | 51 | 454 |
| *Totals* | 275787 | | 16518 | 48145 | 162962 | 227625 |

Table XXII. Calculated Estimated Quantities of Total Nitrogen Loads to the Horseshoe Bend Area of Lake Tenkiller.

| Discharger | Estimated Load (kg N/yr) | Distance to Horseshoe Bend (mi) | Estimated Corrected Load at Horseshoe Bend | | | Estimated Annual Total Load (kg N/yr) |
|---|---|---|---|---|---|---|
| | | | Low Flow (kg N/yr) | Medium Flow (kg N/yr) | High Flow (kg N/yr) | |
| Gage 07196500 | 1947795 | 9.7 | 109855 | 685285 | 924524 | 1719664 |
| Gage 07197000 | 631183 | 8.9 | 35421 | 181103 | 348193 | 564717 |
| Tahlequah | 17010 | 6 | 8354 | 4926 | 1666 | 14947 |
| Cherokee Nation | 1400 | 5 | 707 | 411 | 138 | 1257 |
| Totals | 2597388 | | 154338 | 871725 | 1274522 | 2300585 |

Table XXIII. Estimated Distribution of Total Nitrogen Load Between Background, Point, and Nonpoint Sources at the Horseshoe Bend Area of Lake Tenkiller.

| Source | Estimated Average Total Nitrogen Load at Horseshoe Bend kg N/yr (%) | Estimated Low Flow Contribution at Horseshoe Bend kg N/yr (%) | Estimated Medium Flow Contribution at Horseshoe Bend kg N/yr (%) | Estimated High Flow Contribution at Horseshoe Bend kg N/yr (%) |
|---|---|---|---|---|
| Background | 550000 (23.9) | 35200 (22.8) | 208450 (23.9) | 306350 (24.0) |
| Point Source | 61605 (2.7) | 35793 (23.2) | 19406 (2.2) | 6407 (0.5) |
| Nonpoint Source | 1688980 (73.4) | 83345 (54.0) | 643869 (73.9) | 961795 (75.5) |
| Total | 2300585 | 154338 (6.71% of Total) | 871725 (37.89% of Total) | 1274552 (55.40% of Total) |

Table XXIV. Estimated Distribution of Total Phosphorus Load Between Background, Point, and Nonpoint Sources at the Horseshoe Bend Area of Lake Tenkiller.

| Source | Estimated Average Total Phosphorus Load at Horseshoe Bend kg P/yr (%) | Estimated Low Flow Contribution at Horseshoe Bend kg P/yr (%) | Estimated Medium Flow Contribution at Horseshoe Bend kg P/yr (%) | Estimated High Flow Contribution at Horseshoe Bend kg P/yr (%) |
|---|---|---|---|---|
| Background | 25000 (11.0) | 1600 (9.7) | 5225 (10.9) | 18175 (11.2) |
| Point Source | 12547 (5.5) | 7290 (44.1) | 3952 (8.2) | 1305 (0.8) |
| Nonpoint Source | 190078 (83.5) | 7628 (46.2) | 38968 (80.9) | 143482 (88.0) |
| Total | 227625 | 16518 (7.26% of Total) | 48145 (21.15% of Total) | 162962 (71.59% of Total) |

## 10. LIMNOLOGICAL DATA DISCUSSION AND ANALYSIS

### a. Historical Baseline Limnological Data:

Historical water quality data on the Illinois River basin and Lake Tenkiller is divided into two categories; 1) the watershed and 2) the lake. Historical data in the watershed has recently been summarized in Burks et al. (1991) and provided the basis for the following discussion of historical trends in water quality of the Illinois River. Historical lake data primarily were derived from three sources; 1) an EPA National Eutrophication Survey (NES) study in 1974 (EPA 1974), 2) a USACE study in 1985-86 (USACE 1988), and 3) this EPA Clean Lakes Study of 1992-93.

Historical Water Quality Trends in the Illinois River Basin.

Historical trends in water quality in the Illinois River basin were analyzed and summarized in Burks et al. (1991). The following synopsis was taken (with permission) with slight modification from Burks et al. (1991).

Phosphorus

The annual loading rate for total phosphorus (P) was calculated by multiplying concentration (mg/l) for a particular day by daily discharge (cfs) and converting the product to annual loading (kg/yr). The annual loading value for each sampling period was then used in WQSTAT to calculate seasonal means, medians, 25%, and 75% distribution for trend analyses. The summary statistics for annual loadings indicate a decrease from USGS 07195500, below Lake Frances, when compared to USGS 07196500 located several miles downstream (Table XXV). This decrease occurred even after additional loading was contributed from the Flint Creek drainage basin, USGS 07196000.

The sampling stations and point source discharges with adequate records were used to calculate quartile distributions of annual loadings of total phosphorus (Table XXVI). The calculated annual loadings were then used to calculate the annual loading at Horseshoe Bend area, located just above Lake Tenkiller. This was assumed to be equal to the sum of the loading at USGS 07196500 plus the contribution from Tahlequah WWTP plus the loading from Baron Fork (USGS 07197000), minus a correction factor for the average loss of phosphorus per mile of river flow (0.0017 mg/l/mile).

Raw data were not readily available for the three Arkansas WWTPs included in Table XXVI so we used total P loading data calculated by Walker (1987) for these point source discharges. Walker derived loading values based on monitoring data for 1986. These loading values were adjusted using the correction factor described above.

The calculated annual loadings for each sampling period were also subjected to trend analysis in WQSTAT to determine if there were longterm temporal tendencies. Quarterly averages were calculated from the monthly sample annual loadings and subjected to the Kendall Tau Test. The data were also corrected for seasonal trends to determine if there were significant temporal changes over time. The upstream stations (USGS 071964800 and USGS 07915400) above Lake Frances, showed significant increases over the period of record (Table XXVII). The stations on Flint Creek (USGS 07196000) and Baron Fork Creek showed a significant increase in annual loading rate of total P. Both mainstem Illinois River gaging stations in Oklahoma (USGS 07195500 and USGS 07196500) showed highly significant increases in total P loading.

Analysis of the annual loading of phosphorus for long term trends indicated a significant increase when comparing the earliest with the recent records (Table XXVIII). For example, the median annual phosphorus loading at USGS 07194800 during the period from 1975-80 was significantly ($p=0.05$) less than the period from 1981-86, estimated difference in medians was 1,408 kg/yr. Similar trends of increase in annual loading rates were detected at USGS 07196000, i.e., a significant increase of 9,177 kg/yr in median annual loading when comparing the period of 1979-82 and 1983-86 (Table XXVIII). The USGS 07196500 records also showed a significant increase from 1977-81 vs 1982-86 of 34,775 kg/yr of annual phosphorus (P) loading.

The calculated annual loading values for the Illinois River at the Horseshoe Bend area (river mile 46.1) were used to calculate the appropriate total phosphorus (P) loading for graphical illustration of the eutrophication potential as per Vollenweider's Index (Figures 7-9). The mean hydraulic residence time was based upon 0.25, 0.50, and 0.75 quartiles distribution of lake level; i.e., lake volume, for each year calculated. The quartile distributions of the discharge were used as measures of the annual inflow for the respective lake levels noted above, i.e., 0.25, 0.50, and 0.75 quartiles of discharge were matched with the appropriate lake levels.

The phosphorus loadings to Lake Tenkiller in 1986 were obviously ranked in the eutrophic categories by the Vollenweider index (Figure 7). The fact that the 0.25 quartile distribution of the phosphorus loading also was ranked as eutrophic would indicate that the phosphorus loading would have to be rated as carrying excessive loadings more than 75% of the time during 1986. Similar loading of phosphorus occurred all previous years for which we had records (Figures 7-9). There does seem to be a trend for the phosphorus loading to increase over time, with a greater percentage distribution exceeding eutrophication boundary conditions.

During the period from 1980-83, some of the 0.25 quartile distribution of phosphorus would have been classified as mesotrophic, i.e., still excessive but not as deleterious to overall lake water quality as the eutrophic classification. A similar trend was also occurring in the period from 1976-79, at least some of the annual loadings would be classified as mesotrophic. However, the calculated loadings for greater than 50% of the time from 1976 through 1986 would be classified as eutrophic loading conditions. We can obtain an estimate of the necessary reduction in annual phosphorus loading that would be required to initiate restoration of Lake Tenkiller towards oligotrophic classification, by using the 0.25 quartile loadings from 1979. The loadings would have to be reduced by a minimum of 50% and as much as 80%, in order for the lower quartile of the annual loadings to be classified as mesotrophic. Obviously, the median and 0.75 quartile loadings would have to be reduced even more. Since 1983, the 0.75 quartile distribution of annual phosphorus loadings have exceeded 100,000 kg/yr (Table XXIX). Therefore, the projected reductions for these years would be 90% or greater. There seems to be overwhelming evidence that the annual loading of phosphorus to the upper end of Lake Tenkiller is excessive. The consequences of a continued increase in loading rates to Lake Tenkiller would be deleterious to long-term water quality conditions.

There has been some debate concerning the applicability of Vollenweider's Index of Eutrophication to reservoirs and lakes occurring at southern temperate latitudes. However, Lake Tenkiller is a relatively clear and deep reservoir and thus may be more analogous to the deep-clear northern temperate zone lakes that Vollenweider used to develop his relationships than to most of the southern reservoirs that tend to be shallow and relatively turbid. We believe the Vollenweider Index is an acceptable index of eutrophication loading in Lake Tenkiller.

Table XXV. Summary Statistics for Calculated Annual Phosphorus Loadings in kg/yr as P for Entire Period of Record.

| Source | Annual Phosphorus (P) loadings, kg/yr | | | |
|---|---|---|---|---|
| | N | Mean | Median | SD |
| GS48 | 108 | 14,853 | 858 | 54,385 |
| GS50 | 77 | 89,051 | 0* | 162,133 |
| GS54 | 48 | 146,535 | 50,800 | 218,942 |
| GS55 | 170 | 159,464 | 42,000 | 504,577 |
| GS60 | 99 | 23,961 | 5505 | 99,803 |
| GS65 | 127 | 127,702 | 29,300 | 371,305 |
| GS70 | 126 | 32,163 | 3,640 | 153,355 |
| Tahl STP | 14 | 17,520 | 17,225 | 5,908 |

* missing several years of data

Table XXVI. Calculated Annual Total Phosphorus (P) Loadings (kg/yr) for Illinois River for CY 1986.

| | | Percentile Distribution | | | |
|---|---|---|---|---|---|
| Location | Minimum | 25% | 50% Median | 75% | Maximum |
| USGS 07194800 | 540 | 3,000 | 4,200 | 19,000 | 160,000 |
| Rogers, Ark. STP | | | 18,000 | | 53,000 |
| Springdale, Ark. STP | | | 32,000 | | 58,000 |
| USGS 07195400 | 48,000 | 55,000 | 75,000 | 150,000 | 930,000 |
| USGS 07195500 | 31,000 | 51,000 | 80,000 | 180,000 | 9,600,000 |
| Siloam Springs, Ark STP | 0.0 | | 5,300 | | 16,000 |
| USGS 07196000 | 2,8000 | 12,000 | 14,000 | 26,000 | 3,800,000 |
| USGS 07196500 | 20,000 | 51,000 | 91,000 | 260,000 | 5,800,000 |
| Tahlequah, Ok STP | 6 | 17,000 | 19,000 | 23,000 | 44,000 |
| USGS 07197000 | 94 | 7,800 | 15,000 | 31,000 | 1,700,000 |
| Horseshoe Bend of Tenkiller* | 9048 | 49,677 | 92,901 | 215,865 | 6,575,389 |

* estimated as sum of USGS 07196500 + Tahlequah STP + USGS 07197000 minus correction in loss/mile of flow, calculated as 0.0017 mg/l/mile

Table XXVII. Trend Tests, Total Phosphorus (As P) Annual Sample Loading.

| Station | Kendall Tau Test Statistic | Seasonal Kendall Test Statistic | Seasonal Kendall Sen Slope Estimate (kg/yr)/yr |
|---|---|---|---|
| USGS 07194800 | 2.444*** | 2.640*** | 611 |
| USGS 07195000 | 1.392* | 0.857 | 3181 |
| USGS 07195400 | 1.910** | 1.242 | 6162 |
| USGS 07195500 | 2.035*** | 2.333*** | 2575 |
| USGS 07196000 | 2.402*** | 3.777*** | 1804 |
| USGS 07196500 | 3.510*** | 3.542*** | 9102 |
| USGS 07197000 | 2.478*** | 2.297*** | 1013 |

* = significant at the 80% confidence level
** = significant at the 90% confidence level
*** = significant at the 95% confidence level

Quarterly averages used to calculate all statistics. The Kendall Tau Test was performed on deseasonalized data.

Table XXVIII. Comparison of Upstream vs Downstream Median Loading Total Phosphorus (as kg/yr P).

| Stations Compared Upstream vs Downstream or (Intervals) | Wilcoxon Signed Rank Test Test Statistic | Seasonal Hodges-Lehmann Est. of Differences in Medians Total P (as P) (kg/yr) |
|---|---|---|
| GS48 vs GS54 | -5.655*** | 60212 |
| GS48 (75-80 vs 81-86) | -2.784*** | 1408 |
| GS54 vs GS55 | -1.302* | -13600 |
| GS54 (82-84 vs 85-87) | -1.988*** | 22400 |
| GS60 (79-82 vs 83-86) | -4.918*** | 9177 |
| GS65 (77-81 vs 82-86) | -4.594*** | 34775 |
| GS55 vs GS65 | 4.168*** | -15000 |

* = significant at 80% confidence level
** = significant at 90% confidence level
*** = significant at 95% confidence level

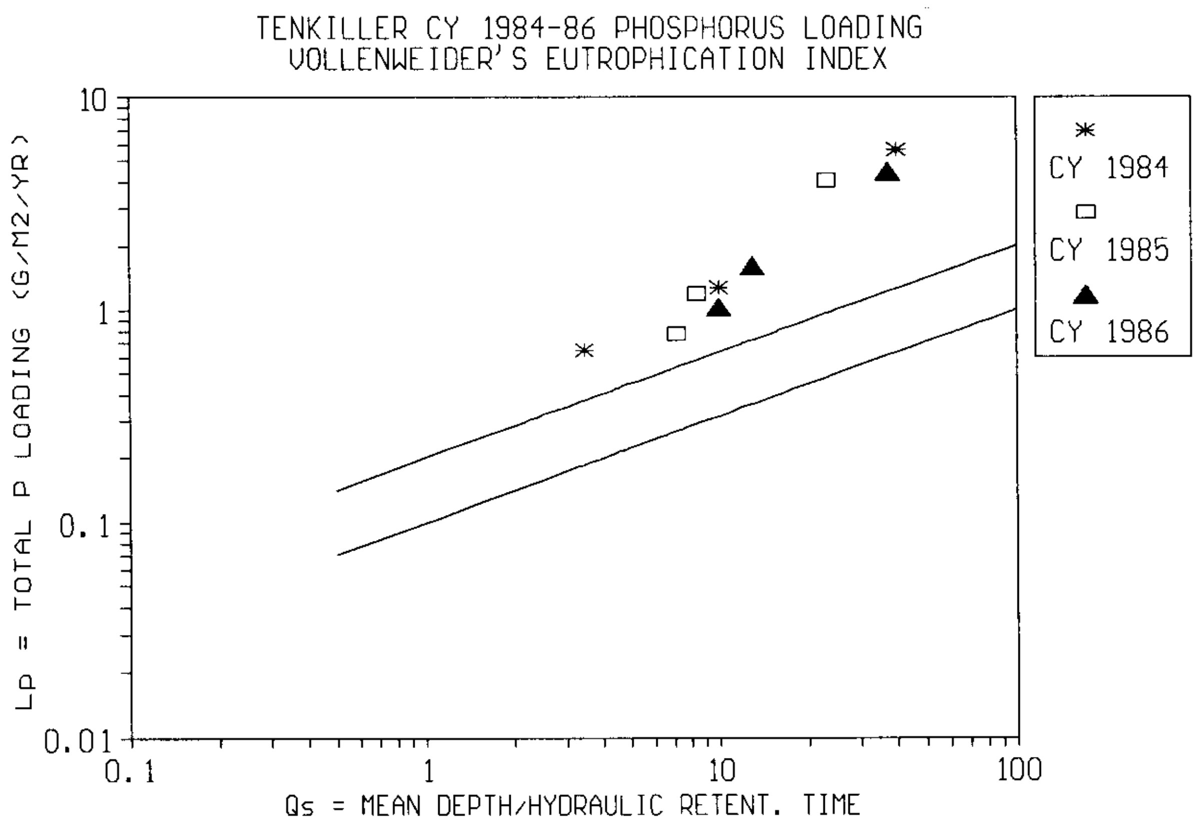



Figure 7. Vollenweider's Eutrophication Index for Lake Tenkiller for 1984-86 Calculated for the 0.25, 0.50, and 0.75 Quartile Distribution of Annual Phosphorus (P) Loadings and Volume Inflows.

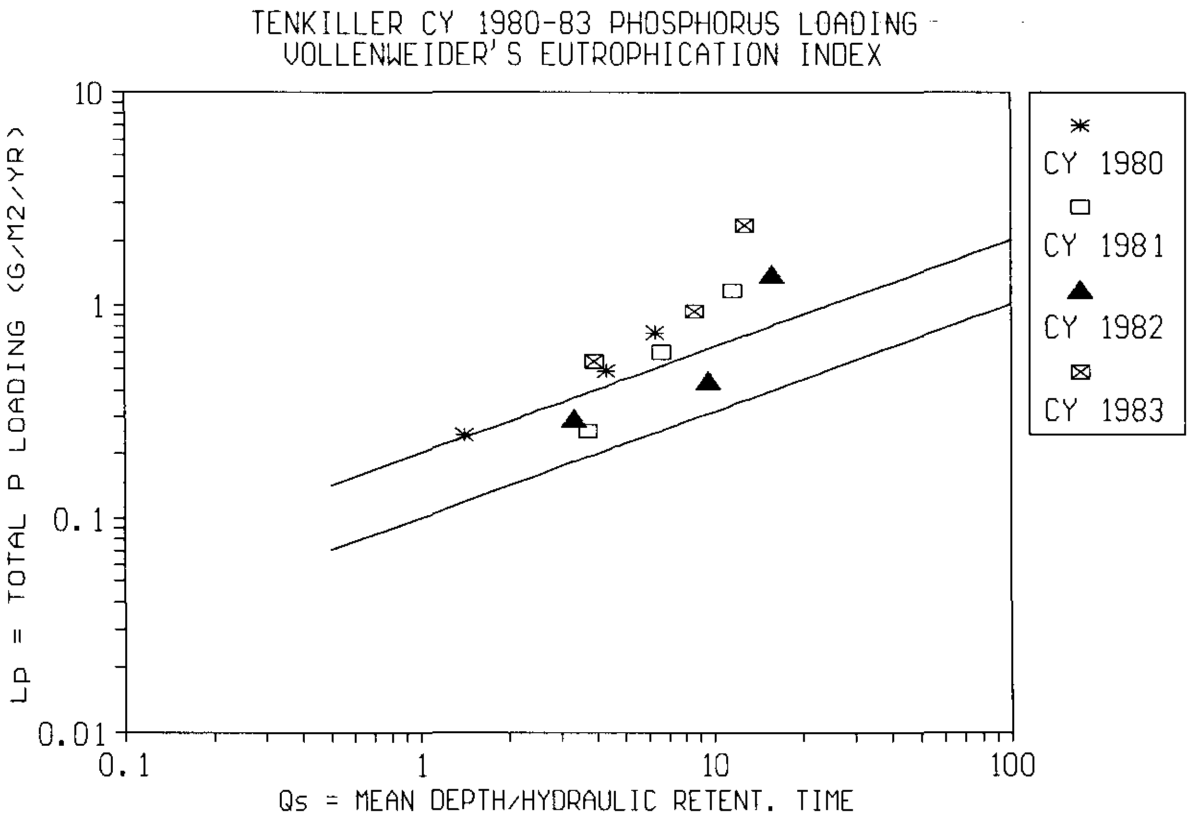



Figure 8. Vollenweider's Eutrophication Index for Lake Tenkiller for 1980-83 Calculated for the 0.25, 0.50, and 0.75 Quartile Distribution of Annual Phosphorus (P) Loadings and Volume Inflows.

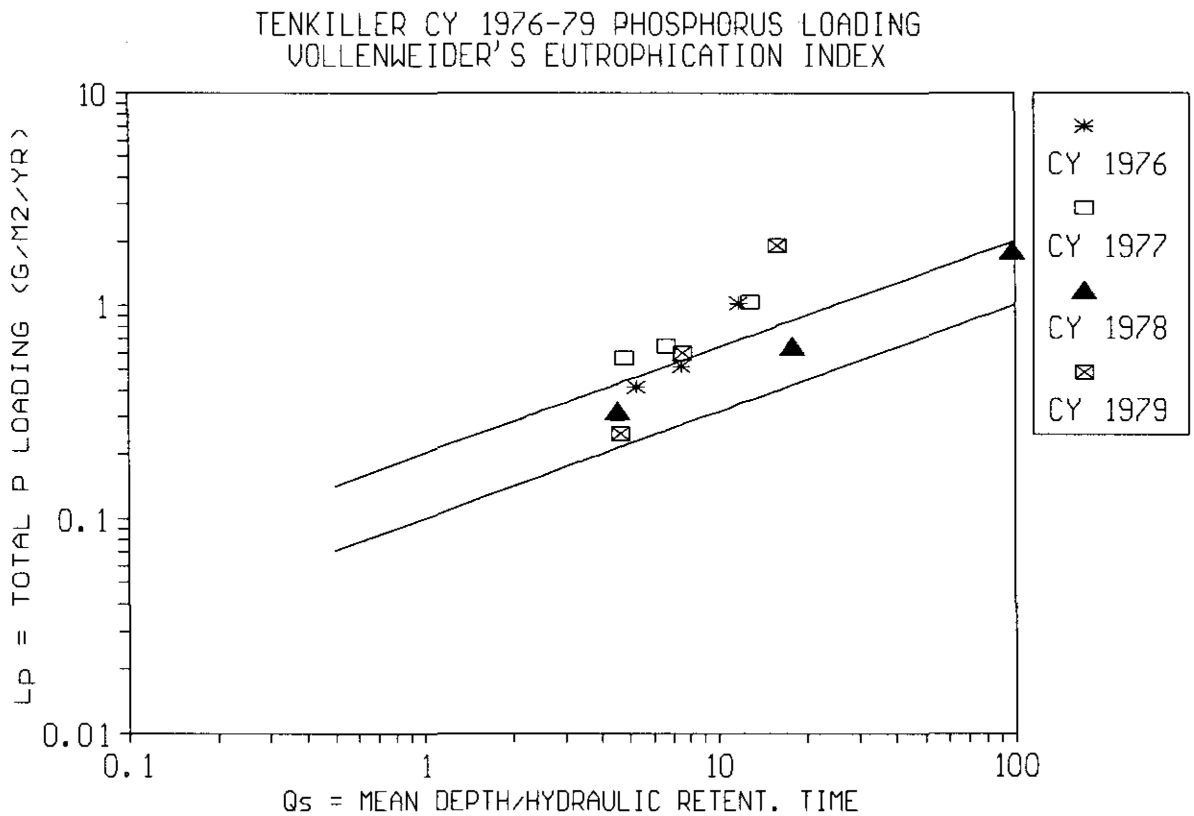



Figure 9. Vollenweider's Eutrophication Index for Lake Tenkiller for 1976-79 Calculated for the 0.25, 0.50, and 0.75 Quartile Distribution of Annual Phosphorus (P) Loadings and Volume Inflows.

Table XXIX. Calculated Annual Loadings of Phosphorus as P at Horseshoe Bend Area of Lake Tenkiller, Corrected for Loss in Miles of Flow.

| Calendar Year | Quartile Distribution of Loading kg/yr as P | | |
|---|---|---|---|
| | 0.25 | 0.50 | 0.75 |
| 1977 | 23321 | 32091 | 52524 |
| 1978 | 14852 | 32430 | 95476 |
| 1979 | 11822 | 30856 | 99432 |
| 1980 | 11370 | 23097 | 37297 |
| 1981 | 11841 | 29283 | 59643 |
| 1982 | 13787 | 22392 | 74119 |
| 1983 | 25431 | 48369 | 124744 |
| 1984 | 30962 | 66750 | 308260 |
| 1985 | 40569 | 64715 | 233113 |
| 1986 | 53887 | 84656 | 246561 |

## Nitrogen

The annual loading for a specific sampling period was calculated by multiplying the mg/l concentration for a specific sampling date by the daily discharge (cfs) and an appropriate conversion factor to obtain kg/day. This value was then converted to an annual loading value by multiplying by 365 days/year. The sampling period annual loading value was then used in WQSTAT to obtain summary statistics of mean, median, and standard deviation of the sampling period annual loading values for the entire period of record. The mean annual loading of nitrite/nitrate as N in kg/yr showed an increase from the upper end (USGS 07194800) of 123,190 kg/yr to a peak concentration of 884,020 kg/yr at USGS 07195500, just below Lake Frances (Table XXX). There was a decrease in mean annual loading of N from below Lake Frances to USGS 07196500.

There is no current agreement upon loading rates of inorganic nitrogen which are considered eutrophic. Inorganic nitrogen definitely contributes to growth of primary producers; however, the level that is considered excessive or that would promote development of undesirable species or densities of growth cannot be accurately defined. Sawyer (1947) suggested that concentrations of greater than 0.300 mg/l of total nitrogen and 0.010 mg/l of phosphorus at spring mixing would be conducive to development of eutrophic conditions. Vollenweider calculated and plotted the mean annual loadings of nitrogen in his classic study of over 200 lakes, but never attempted to develop an overall index based upon nitrogen loading. This is primarily because of the complications of interactions of nitrogen and phosphorus in promotion of algal growth. Although Vollenweider never attempted to develop a eutrophic index based upon nitrogen loading, his data may be useful for comparisons. Wetzel (1983) summarized Vollenweider's data into two major categories, permissible and dangerous loading for both nitrogen and phosphorus (Table XXXI).

We calculated the annual loading values projected to occur at the Horseshoe Bend area of Lake Tenkiller. The loading values were calculated by summing the annual loadings at USGS 07196500, USGS 07197000, and from Tahlequah STP and subtracting an average loss of 0.00907 mg/l/mile of flow. The annual NO2/NO3(N) loadings were calculated for the quartile distributions of both concentration and discharge for each year of record. The 0.25, 0.50, and 0.75 quartile distributions of nitrogen load were then graphed with respect to the projected mean lake level, i.e., power pool minus 25% volume lake level elevation (618 ft MSL) for the 0.25 quartile of discharge and nitrogen loading, power pool lake level elevation (632 ft MSL) for the 0.50 quartile distribution of discharge and nitrogen loading, and flood stage lake level elevation (667 ft MSL) for the 0.75 quartile distribution of discharge and nitrogen loading. Obviously, we had to make some arbitrary assumptions that greater discharge would transport higher loadings of nitrogen and would result in higher lake elevations and the converse of this relationship.

Table XXX. Summary Statistics for Calculated Annual Nitrite/nitrate Loadings for Entire Period of Record.

| Source | Annual NO2/NO3(N) loadings, (kg/yr) | | | |
|---|---|---|---|---|
| | N | Mean | Median | SD |
| USGS 07194800 | 96 | 163000 | 20100 | 246139 |
| USGS 07195000 | 33 | 377724 | 0* | 224096 |
| USGS 07195400 | 55 | 1030478 | 63800 | 1050163 |
| USGS 07195500 | 110 | 766045 | 169000 | 1644380 |
| USGS 07196000 | 82 | 123207 | 30800 | 361764 |
| USGS 07196500 | 96 | 807803 | 214000 | 1484658 |
| USGS 07197000 | 98 | 246966 | 45700 | 444291 |
| * missing several years of data | | | | |

Table XXXI. Provisional Permissible Loading Levels for Total Nitrogen and Total Phosphorus in $g/m^2/yr$.

| | Permissible Loading | | Dangerous Loading | |
|---|---|---|---|---|
| Mean Depth (m) | Total Nitrogen | Total Phosphorus | Total Nitrogen | Total Phosphorus |
| 5 | 1.0 | 0.07 | 2.0 | 0.13 |
| 10 | 1.5 | 0.10 | 3.0 | 0.20 |
| 50 | 4.0 | 0.25 | 8.0 | 0.50 |
| 100 | 6.0 | 0.40 | 12.0 | 0.80 |

Modified from Wetzel (1983)

The projected loadings with respect to mean depth (m) provided an index of the relative potential of nitrogen to stimulate algal growth in Lake Tenkiller. The projected annual nitrogen loading during the four years from 1977 to 1980 indicated that nitrogen loadings were rated excessive for all of the quartile distributions in 1977 (Figure 10). The plots indicated that more than half the nitrogen loading in 1978 and 1979 also were rated as excessive. The loadings in 1980 were not as excessive, with only the upper quartile exceeding the dangerous level suggested by Wetzel (1983).

During the periods from 1981-84, the calculated annual NO2/NO3(N) loadings exceeded the permissible levels at all quartile distributions, except for the 0.25 distribution in 1981-1983 (Figure 11). The highest loadings occurred at the 0.75 quartile distribution during 1984. The annual NO2/NO3(N) loadings for 1985-1986 and for the entire period of record indicated a similar trend (Figure 12). However, due to high levels of discharge, the calculated loadings were much higher in 1985-1986 than in the previous years. When considered over the entire period of record from 1977-1986, the quartile distributions greater than 0.25 exceeded the suggested permissible loadings by factors ranging from 2X to 10X.

Based upon the indices of nitrogen loadings suggested by Wetzel (1983), the Illinois River transported more than enough NO2/NO3(N) to stimulate excessive algal growth in Lake Tenkiller. We did not attempt to include ammonia or Kjeldahl nitrogen in the total nitrogen loading calculations, since there was a limited number of samples. However, if these additional forms of nitrogen were added to the calculated NO2/NO3(N), it would have increased the total annual loading even more.

More than 50% of the calculated annual loadings of both phosphorus and nitrogen exceeded established limits for development of eutrophic conditions in Lake Tenkiller. There can be little doubt concerning the long-term prospects for water quality in Lake Tenkiller, if this trend is not reversed.

Many authors have attempted to rate the relative contribution of nitrogen to development of eutrophic conditions on the basis of N/P ratios, since it has been well established that Liebig's Law of the Minimum describes controlling factors for algal growth. The N/P ratio is often used to determine which macronutrient may be the "limiting" factor for algal productivity. If N/P ratios are greater than 10 to 15, phosphorus is often cited as the macronutrient limiting primary productivity. Conversely, N/P ratios less than ten may be interpreted to mean that nitrogen is probably the limiting nutrient.

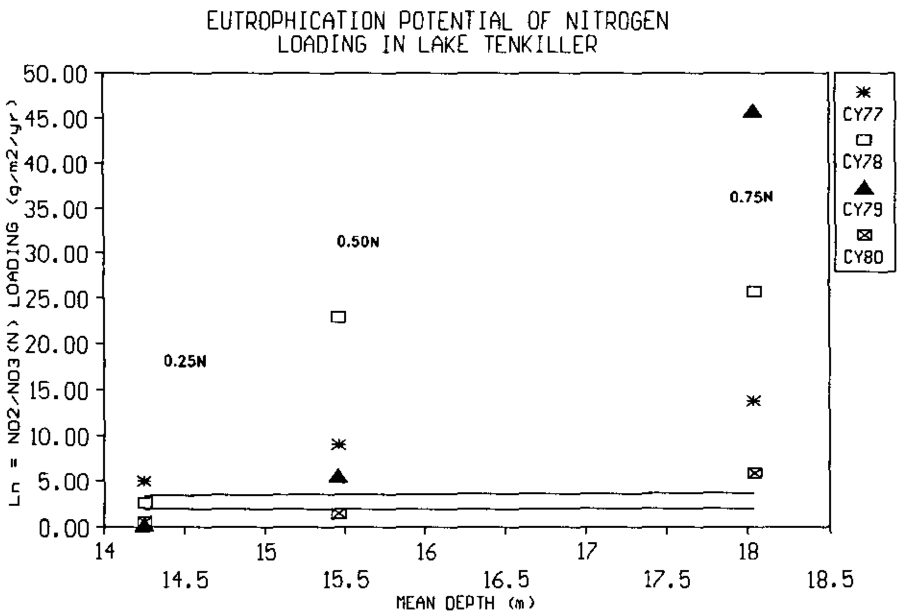



Figure 10. Comparison of the Quartile Distributions of Annual Nitrogen (N) Loadings to Lake Tenkiller in CY 1977-80 with Loadings Suggested by Wetzel (1983) as Dangerous or Permissible.

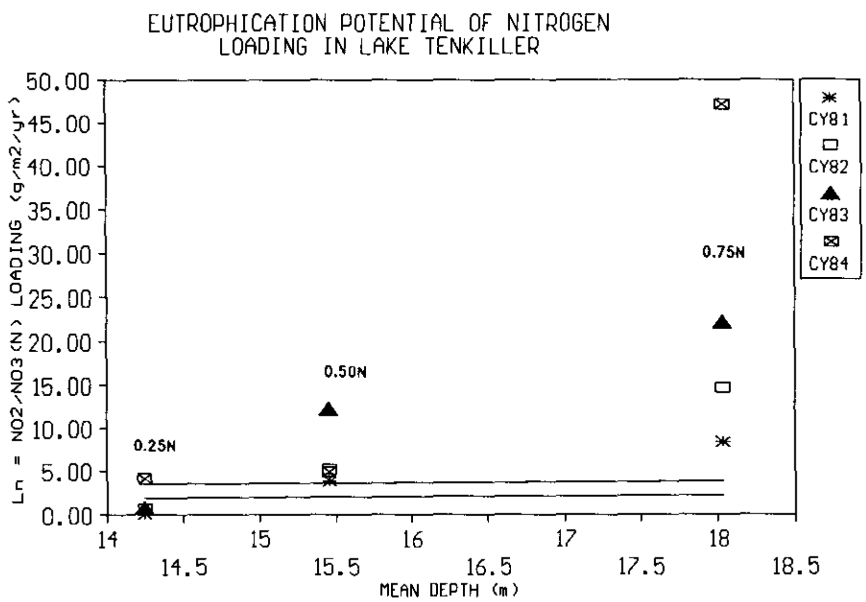



Figure 11. Comparison of Quartile Distributions of Nitrogen (N) Loadings to Lake Tenkiller in CY 1981-84 with Suggested Dangerous or Permissible Loadings as per Wetzel (1983).

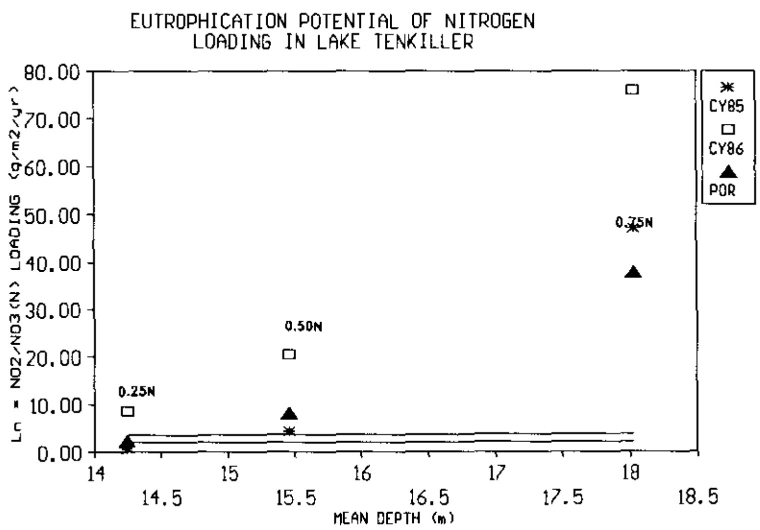



Figure 12. Comparison of the Quartile Distribution of Annual Nitrogen (N) to Lake Tenkiller in CY 1985-86 and Entire Period of Record with Dangerous and Permissible Loadings as Suggested by Wetzel (1983).

We calculated the N/P ratios at two locations in the upper end of Lake Frances and at Horseshoe Bend, which is on the upper end of Lake Tenkiller (Table XXXII). Based upon these ratios, it appeared that phosphorus may be the limiting nutrient in Lake Frances, whereas nitrogen appears to be the limiting nutrient in Lake Tenkiller headwaters and phosphorus limited elsewhere in Lake Tenkiller. However, these ratios obviously vary with the level of discharge and seasons of the year. The ratios should not be the primary factor in determining whether to focus all control measures on either nutrient alone. The concentrations of both nitrogen and phosphorus were excessive and should be reduced. If the concentrations of both nutrients could be reduced significantly, then perhaps a higher priority could be placed upon the nutrient most easily controlled and in the most economically feasible manner.

The long term trends in nitrogen loading at USGS 07194800 showed a significant increase of 14,604 kg/yr/yr over the period of record (Table XXXIII). The increase in loading rate was even greater at USGS 07195400, showing a significant trend of 49,167 kg/yr/yr. The overall trend for most of the mainstem sampling stations was an increase for the period of record. These long-term trends could be interpreted to indicate that while there is an increase in nitrogen discharge to the upper portions of the Illinois River, the biota along the length of the river has assimilated most of the increased nitrogen.

Table XXXII. Calculated N/P Ratios for the Quartile Distributions of Annual Loadings at the Upper Ends of Lake Frances and Lake Tenkiller.

| Calendar Year | Lake Frances N/P ratio | | | Lake Tenkiller N/P ratio | | |
|---|---|---|---|---|---|---|
| | .25 | .50 | .75 | .25 | .50 | .75 |
| 1977 | --- | --- | --- | 6.9 | 11.1 | 12.3 |
| 1978 | --- | --- | --- | 6.7 | 28.2 | 22.0 |
| 1979 | --- | --- | --- | 3.3 | 8.3 | 19.0 |
| 1980 | --- | --- | --- | 2.1 | 3.7 | 7.5 |
| 1981 | 4.6 | 5.8 | 7.7 | 2.4 | 6.4 | 7.2 |
| 1982 | 4.2 | 3.6 | 9.7 | 2.9 | 11.6 | 9.7 |
| 1983 | 4.6 | 5.7 | 7.1 | 1.9 | 9.9 | 7.7 |
| 1984 | 4.3 | 17.3 | 5.1 | 4.2 | 3.8 | 7.2 |
| 1985 | 6.2 | 16.9 | 11.0 | 1.8 | 3.4 | 8.7 |
| 1986 | 7.4 | 12.5 | 11.5 | 5.4 | 9.4 | 13.3 |
| 1987 | 7.4 | 8.5 | 17.6 | --- | --- | --- |

Table XXXIII. Trend Tests, NO2+NO3-N Annual Sample Loading.

| Station | Kendall Tau Test Statistic | Seasonal Kendall Test Statistic | Seasonal Kendall Sen Slope Estimate (kg/yr)/yr |
|---|---|---|---|
| USGS 07194800 | 3.266*** | 2.526*** | 14604 |
| USGS 07195000 | 0.183 | 0.000 | -208 |
| USGS 07195400 | 2.059*** | 2.130*** | 49167 |
| USGS 07195500 | 0.955 | 0.912 | 22477 |
| USGS 07196000 | 2.588*** | 2.814*** | 18473 |
| USGS 07196500 | 1.906** | 1.867** | 36058 |
| USGS 07197000 | 1.186 | 0.733 | 4605 |
| Quarterly averages used to calculate all statistics. The Kendall Tau Test was performed on deseasonalized data. | | | |
| * = significant at the 80% confidence level<br>** = significant at the 90% confidence level<br>*** = significant at the 95% confidence level | | | |

## Historical Lake Data

Historical lake data (biological and chemical) were derived from the following sources (only includes those with original data or notable significance):

Gakstatter, J. And A. Katko. 1986. An intensive survey of the Illinois River (Arkansas and Oklahoma) in August 1985. USEPA, Duluth MN.

Nolen, S., J. Carroll, D. Combs, J. Staves, and J. Veenstra. 1989. Limnology of Tenkiller Ferry Lake, Oklahoma, 1985-1986. Proc. Okla. Acad. Sci., 69:45-55. (USACE 1985-86 data)

United States Army Corps of Engineers. 1988. Water Quality Report - Tenkiller Ferry Lake, Oklahoma 1986-1986. USACE, Tulsa District.

United States Environmental Protection Agency. 1977. Report on Tenkiller Ferry Reservoir, Cherokee and Sequoyah Counties, Oklahoma. Working Paper No. 593. USEPA.

Finnell, J. C. 1954. Dissolved oxygen and temperature profiles of Tenkiller Reservoir and tailwaters with consideration of these waters as a possible habitat for rainbow trout. Proc. Okla. Acad. Sci., 34:65-72.

Hall, G. 1952. Observations on the fishes of the Fort Gibson and Tenkiller Reservoir areas, 1952. Proc. Okla. Acad. Sci., 33:55-63.

Hall, G. 1953. Preliminary observations on the presence of stream-inhabiting fishes in Tenkiller Reservoir, a new Oklahoma impoundment. Proc. Okla. Acad. Sci. 34:34-40.

Hern, S., V. Lambou, M. Morris, W. Taylor, and L. Williams. 1979. Distribution of phytoplankton in Oklahoma lakes. EPA-600/3-79-068. USEPA, Las Vegas NV.

Jenkins, R., E. Leonard, and G. Hall. 1952. An investigation of the fisheries resources of the Illinois River and pre-impoundment study of Tenkiller Reservoir, Oklahoma. Oklahoma Fisheries Research Lab, Norman OK.

Jenkins, R. 1953. Continued fisheries investigation of Tenkiller Reservoir, Oklahoma, during its first year of impoundment. Oklahoma Fisheries Research Lab, Norman OK.

Lewis, S. And J. Mense. 1976. Bibliography of literature on Oklahoma waters. Oklahoma Fisheries Research Lab, Norman OK. (No data, but many historical references).

Lindsay, H., P. Buck, and T. Buckley. 1973. A biological and historical inventory and assessment of the Tenkiller Ferry project. USACE, Tulsa District. (No original data, but summarizes other biological data sources on Tenkiller).

Morris, W. 1979. Tenkiller Ferry Reservoir and lower Illinois River 1979 temperature and dissolved oxygen investigation. Publication 93, Oklahoma Water Resources Board, Water Quality Division, Oklahoma City OK.

Oklahoma State Department of Health. 1978. Water quality of the Illinois River and Tenkiller Reservoir, June 1975 - October 1977. State Water Quality Lab, Oklahoma City OK.

Oklahoma State Department of Health. 1981. Toxics monitoring survey of Oklahoma reservoirs 1980-1981. OSDH, Environmental Health Services, State Environmental Laboratory Service, Oklahoma City OK.

Oklahoma State Department of Health. 1987. Toxics monitoring survey of Oklahoma reservoirs, 1985 Final Report. OSDH, Environmental Health Services, State Environmental Laboratory Service, Oklahoma City OK.

Smith, J. 1987. Fish management survey and recommedations for Tenkiller. Federal Aid Project No. F-44-D-2. Oklahoma Department of Wildlife Conservation, Oklahoma City, OK.

Summers, G. 1978. Sportfishing statistics of Oklahoma reservoirs. Oklahoma Fishery Research Lab, Oklahoma City, OK.

The historical trends presented here are cursory because much of the discussion of historical trends of eutrophication are contained in the next section on current limnological data of the lake.

Robust trend tests as were done on the Illinois River above were not possible because much of the data were not collected with consistent methods. However, some conlcusions were afforded. In summary, the following were concluded based upon the historical records:

- Lake Tenkiller developed an anoxic hypolimnion soon after impoundment but whether this condition was maintained during every stratification season throughout the reservoir's life could not be ascertained;
- the lake developed an anoxic hypolimnion during the 1974 EPA-NES and the 1985-86 USACE studies;
- the total phosphorus and nitrogen loads impinging the lake increased significantly from 1974 to 1985-86 and increased the concentrations in the epilimnion of the entire lake, this trend is further supported by a steady increase in sedimentary total phosphorus in the mid-reaches of the reservoir thalweg;

- angling success has increased and ostensibly is better now than it ever has been (Smith personal communication);
- chlordane in channel catfish (*Ictalurus punctatus*) and in smallmouth buffalo (*Ictiobus bubalus*) tissue extracts exceeded Oklahoma State Department of Health concern levels in 1980 and in smallmouth buffalo in 1983, also in 1983, polychlorinated biphenyls (PCBs) were detected in several species but were below OSDH concern levels, no violations or exceedances were detected in the 1987 survey, albeit low levels of dichlorodiphenyltrichloroethane (DDT), chlordane, and mercury residues were detected in most fish species collected.

Overall, Lake Tenkiller historically has exhibited exceptionally good water quality for its designated uses. One exception noted here is the high levels of polychorinated biphenyls (PCBs) and chlordane residues found in fish tissue extracts during the toxics monitoring survey conducted by the Oklahoma State Department of Health (OSDH 1987). However, since chlorinated hydrocarbon pesticides have been banned and residue levels appear to be decreasing, future problems of this contamination are not anticipated.

## b. One Year Current Limnological Data

The following discussion on one year of current limnological data is based upon the original data collected during this project. Although the heading indicates one year these current data represents two years of monitoring. In some cases, the data are presented in conjuction with previous studies, primarily the 1974 EPA-NES and the 1985-86 USACE studies, to illustrate temporal changes in water quality. These studies are chosen because they are consistent with the methods used in the current study, the referenced sampling sites are on or near those chosen for the current study (Figure 13), and incorporated replicated sampling episodes. The discussion presents standard physicochemical data first with modelling results and conclusions last.

1. Monitoring Schedule: The monitoring data collected during this study reflects monthly sampling (occasionally biweekly and bimonthly) for two years. We chose to sample the lake based upon seasonal and/or hydraulic events in lieu of calender periodicity. We sampled the reservoir more often immediately pre-, immediately post-, and during summer stratification. On each sampling occurrence, data were collected in accordance with 40 CFR Part 35 § 35-1650 Appendix A subpart 10.

   Sampling sites for the EPA-NES 1974 study included 4 stations, the USACE 1985-86 survey included 14 stations, and the EPA-CLP of 1992-93 survey (this study) included 8 stations, one of which was located in the tailwaters (Figure 13).

2. Present Trophic Condition: The present trophic status of Lake Tenkiller is classified as eutrophic. This classification is based upon excessive levels of nitrogen and phosphorus concentrations in the lake, nitrogen and phosphorus loads impinging the lake, and resultant increased algal standing crop and hypolimnetic oxygen depletion. More detailed explanation follows in the ensuing subsections.

3. Surface Area (Hectares): (see Table I)

4. Depth (Meters) Maximum and Average: (see Table I)

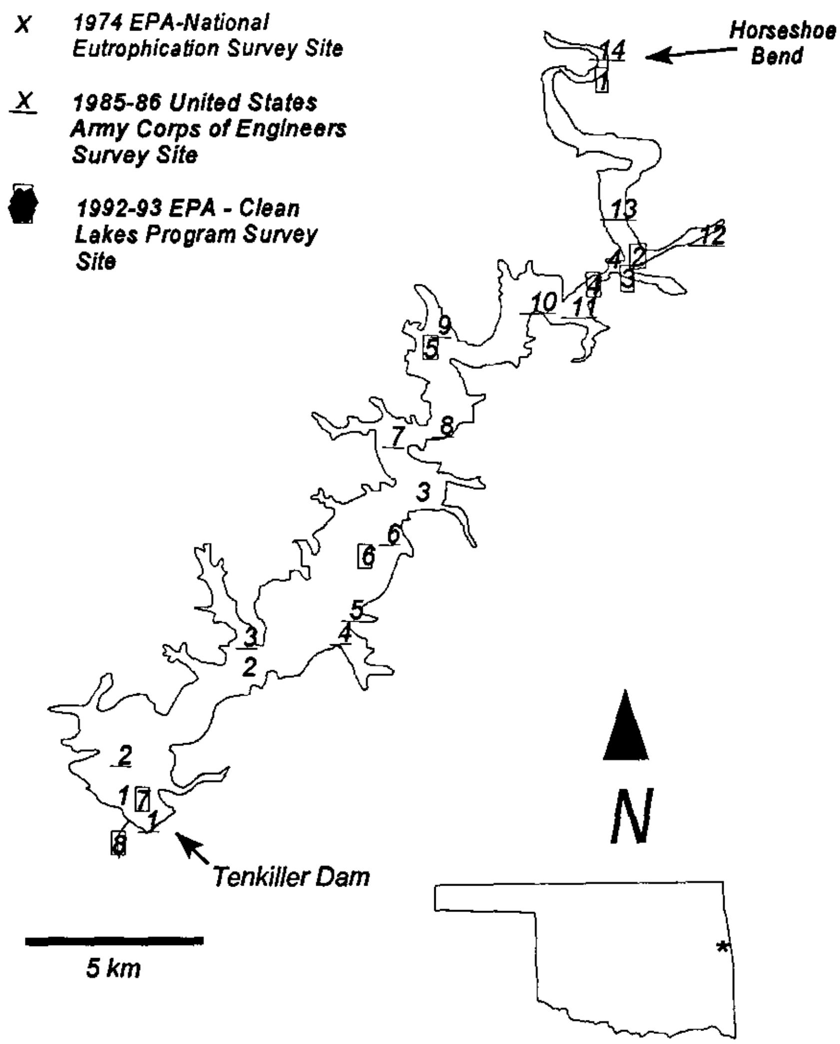



**Figure 13.** Sampling Sites on Lake Tenkiller, Oklahoma.

5. Hydraulic Residence Time: (see Table I)

6. Watershed Area (Hectares): (see Table I)

7. Physical, Chemical and Biological Quality of the Lake and Major Tributaries:

Profiles of Temperature, Dissolved Oxygen, and Conductivity

Within Lake Tenkiller, profiles of temperature indicated the onset of summer stratification (thermal) occurred circa April to May and complete mixis began circa September to October (Figures 14-19; upper panels). Maximum surface temperatures approached 30°C and minimum surface temperatures measured were approximately 8°C, albeit we did not sample intensively during the winter. Bottom temperatures did not exceed 22° at the lower end but often reflected surface temperatures in the upper end. Mean temperatures for stations 1, 2, 3, 4, 5, 6, 7, and 8 were 20.4, 20.8, 22.4, 22.1, 21.5, 21.3, 19.2, and 16.5 °C, respectively. Generally, the mean temperature increased as the dam is approached. While the calculated means given above indicate a decrease beginning at station 6, hypolimnetic waters which are colder influence the mean temperature more (due to the increased hypolimnetic stratum) and skew the mean lower. Station 8 (tailwaters) obviously reflected the hypolimnetic discharge. The stratification period was longer in the deeper waters of the lower reaches of the reservoir.

Following the onset of thermal stratification, rapid depletion of hypolimnetic dissolved oxygen occurred and led to hypolimnetic anoxia which persisted throughout the period of thermal stratification (Figures 14-19; middle panels). Current thought in reservoir limnology indicates that hypolimnetic oxygen depletion can be due to excessive allochthonous organic loads. Epilimnetic dissolved oxygen often exceeded 100% saturation during the growing season but not during the winter. This phenomena excessive photosynthesis. Mean values were not computed because extremes (i.e., surficial supersaturation and hypolimnetic anoxia) were deemed most descriptive of the lake's current condition. Little information is gained from the means.

Chemical stratification was also observed with hypolimnetic conductivities higher than surface conductivities (Figures 14-19; lower panels). However, this phenomena could be due to incoming waters with higher conductivities "plunging" into the hypolimnion and chemical reactions that occur in an unmixed, oxygen depleted hypolimnion. Mean conductivities stations 1, 2, 3, 4, 5, 6, 7, and 8 were 187, 181, 194, 185, 185, 185, 186, and 190 $\mu\mho$/cm @ 25°C, respectively.

Comparison of these data with data collected immediately after impoundment, in 1974, and in 1985-86 indicated the same trend. All these data indicated hypolimnetic anoxia. The rates of depletion of hypolimnetic dissolved oxygen among the studies, however, could not be determined due to insufficient data.

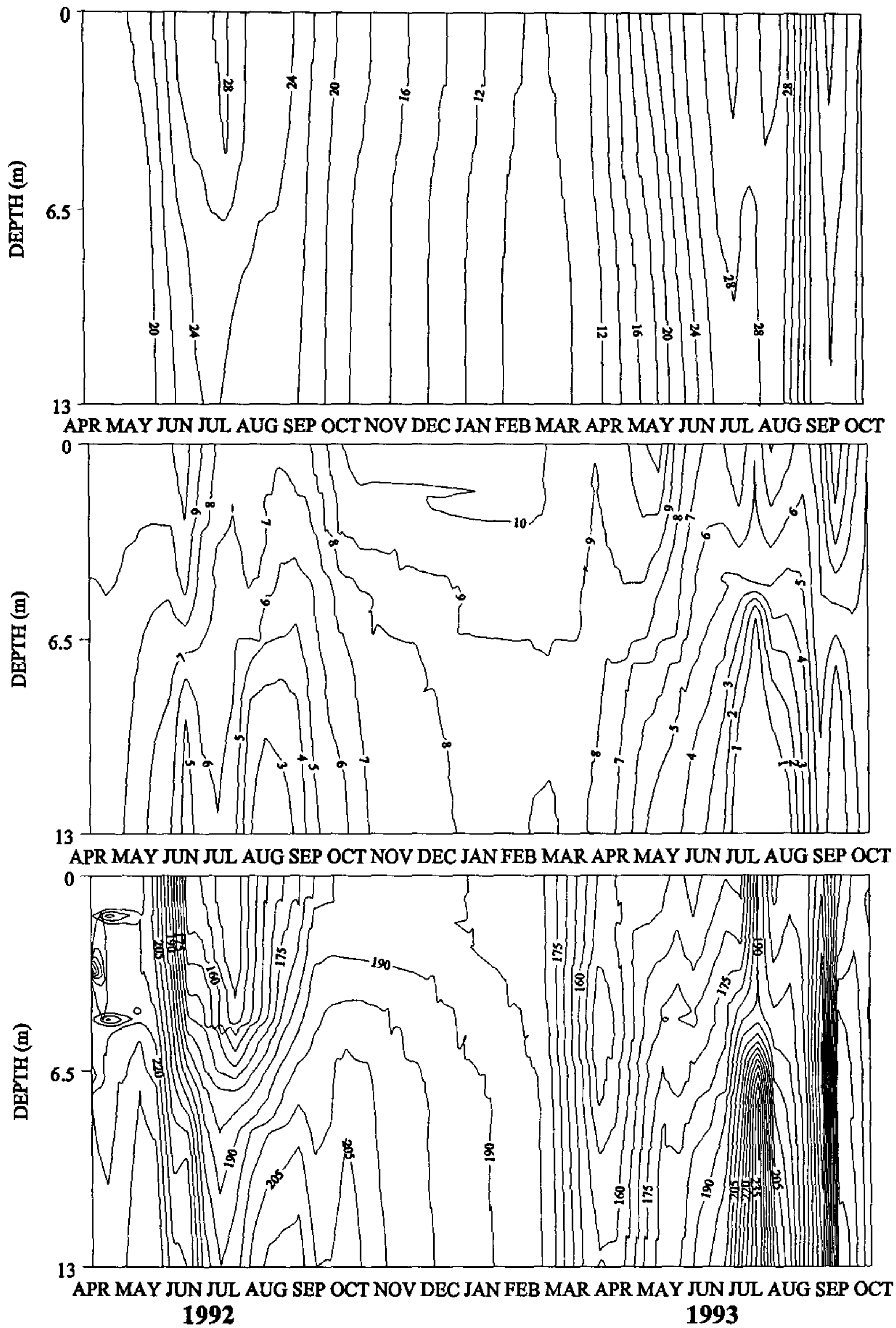



**Figure 14** Lake Tenkiller Station 2 isopleths for temperature (°C, top), dissolved oxygen (mg/l, middle), and conductivity ($\mu$℧/cm at 25°C, bottom) for CY 92-93.

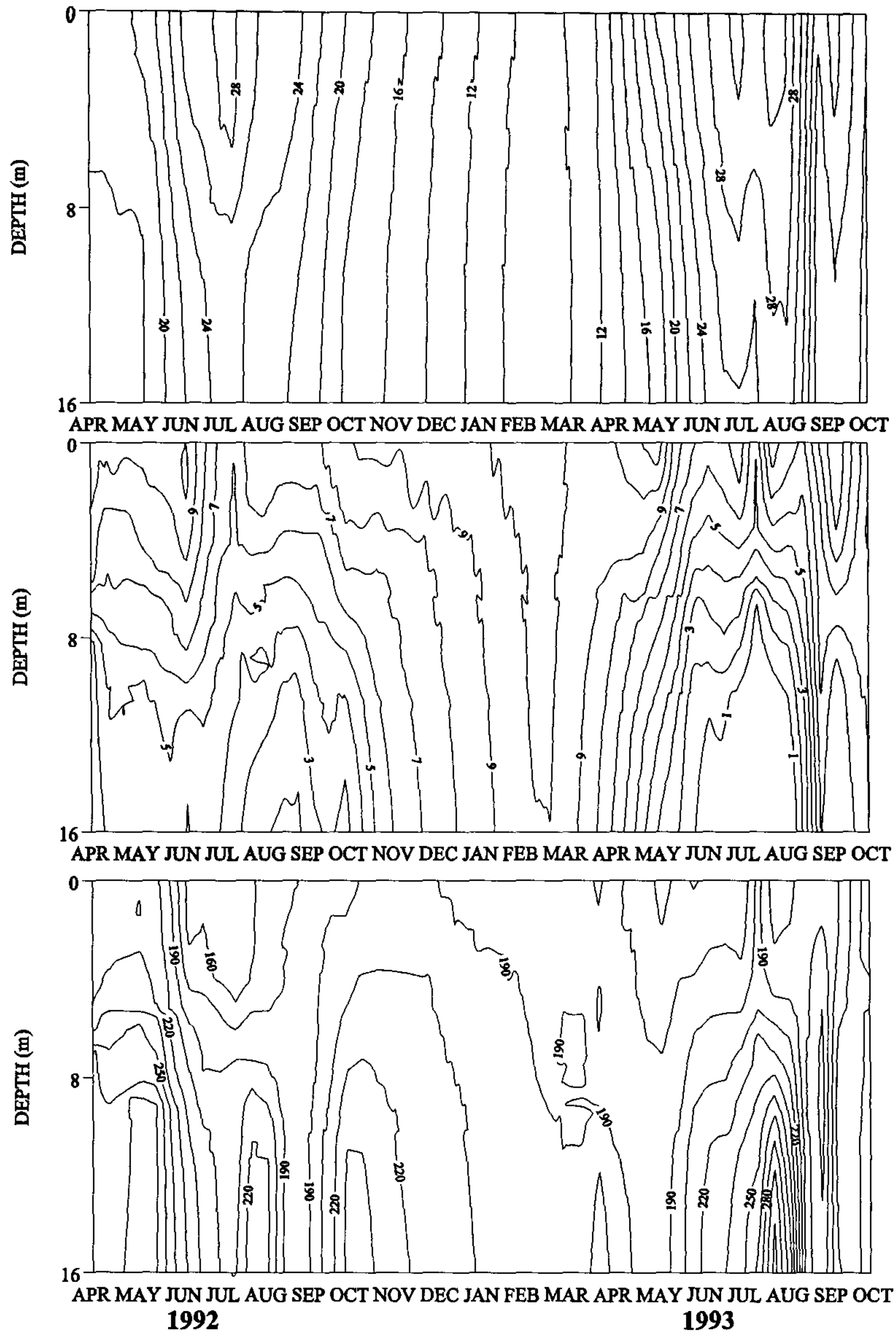



**Figure 15** Lake Tenkiller Station 3 isopleths for temperature (°C, top), dissolved oxygen (mg/l, middle), and conductivity (μ℧/cm at 25°C, bottom) for CY 92-93.

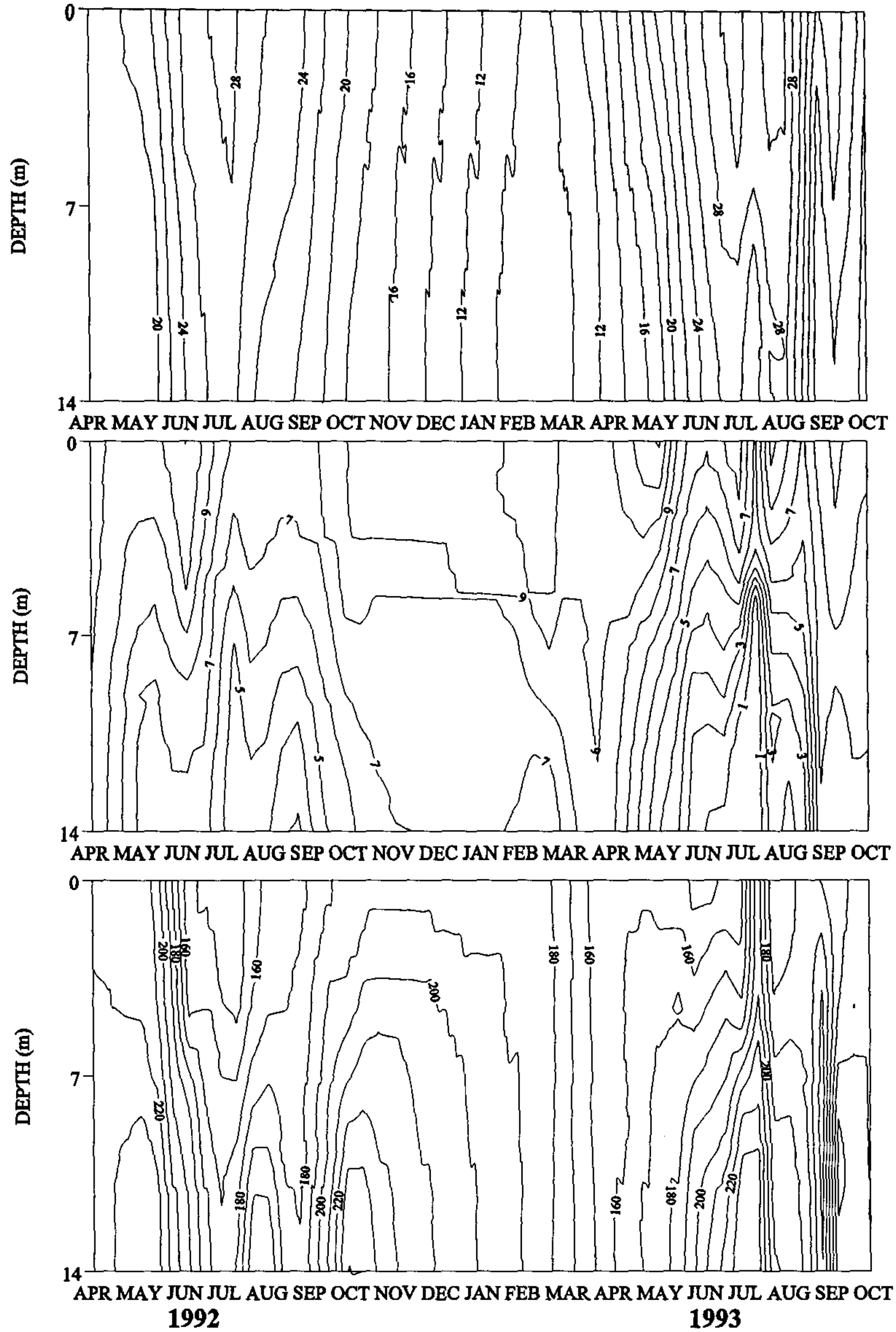



**Figure 16** Lake Tenkiller Station 4 isopleths for temperature (°C, top), dissolved oxygen (mg/l, middle), and conductivity (μ℧/cm at 25°C, bottom) for CY 92-93.

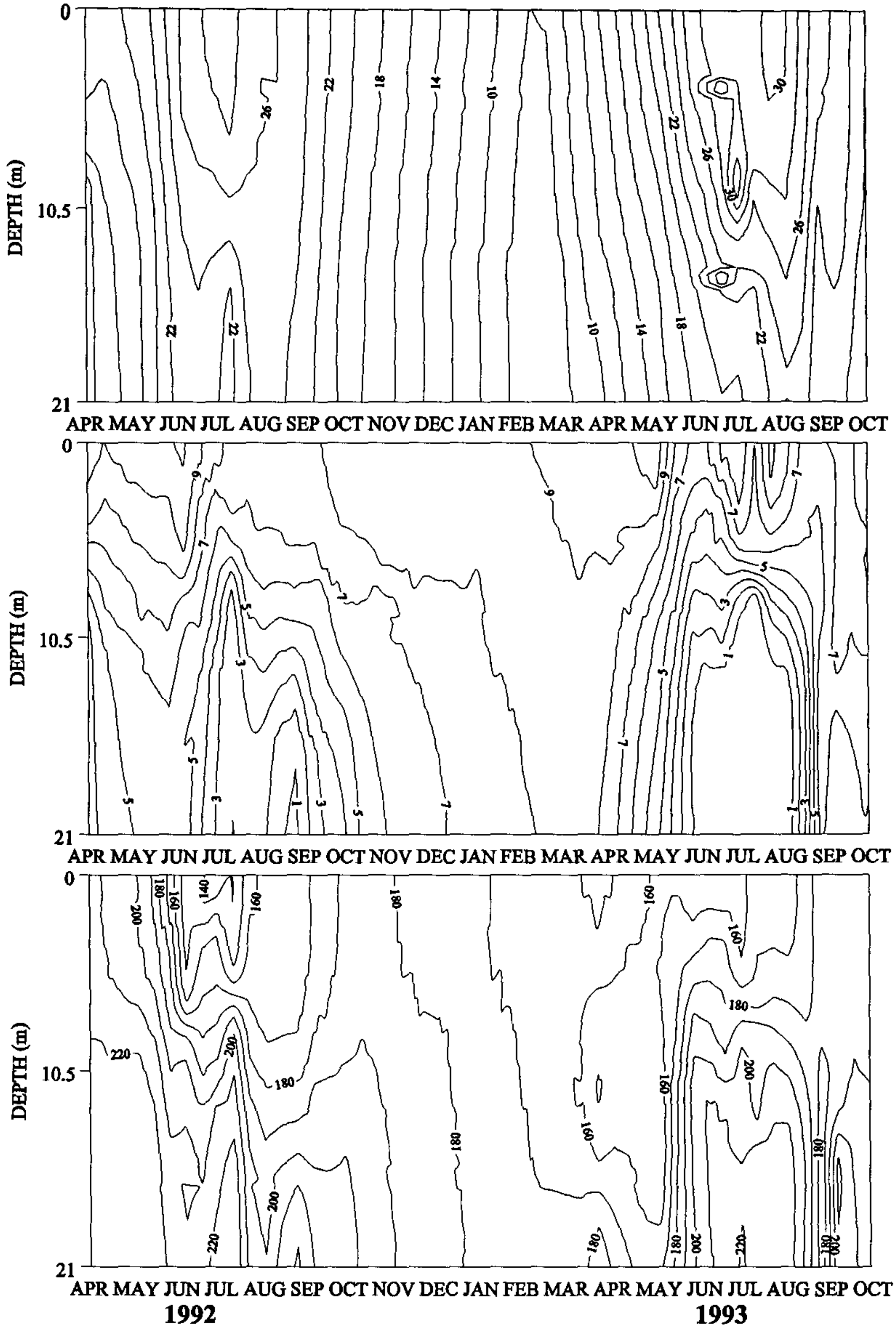



**Figure 17** Lake Tenkiller Station 5 isopleths for temperature (°C, top), dissolved oxygen (mg/l, middle), and conductivity (μ℧/cm at 25°C, bottom) for CY 92-93.

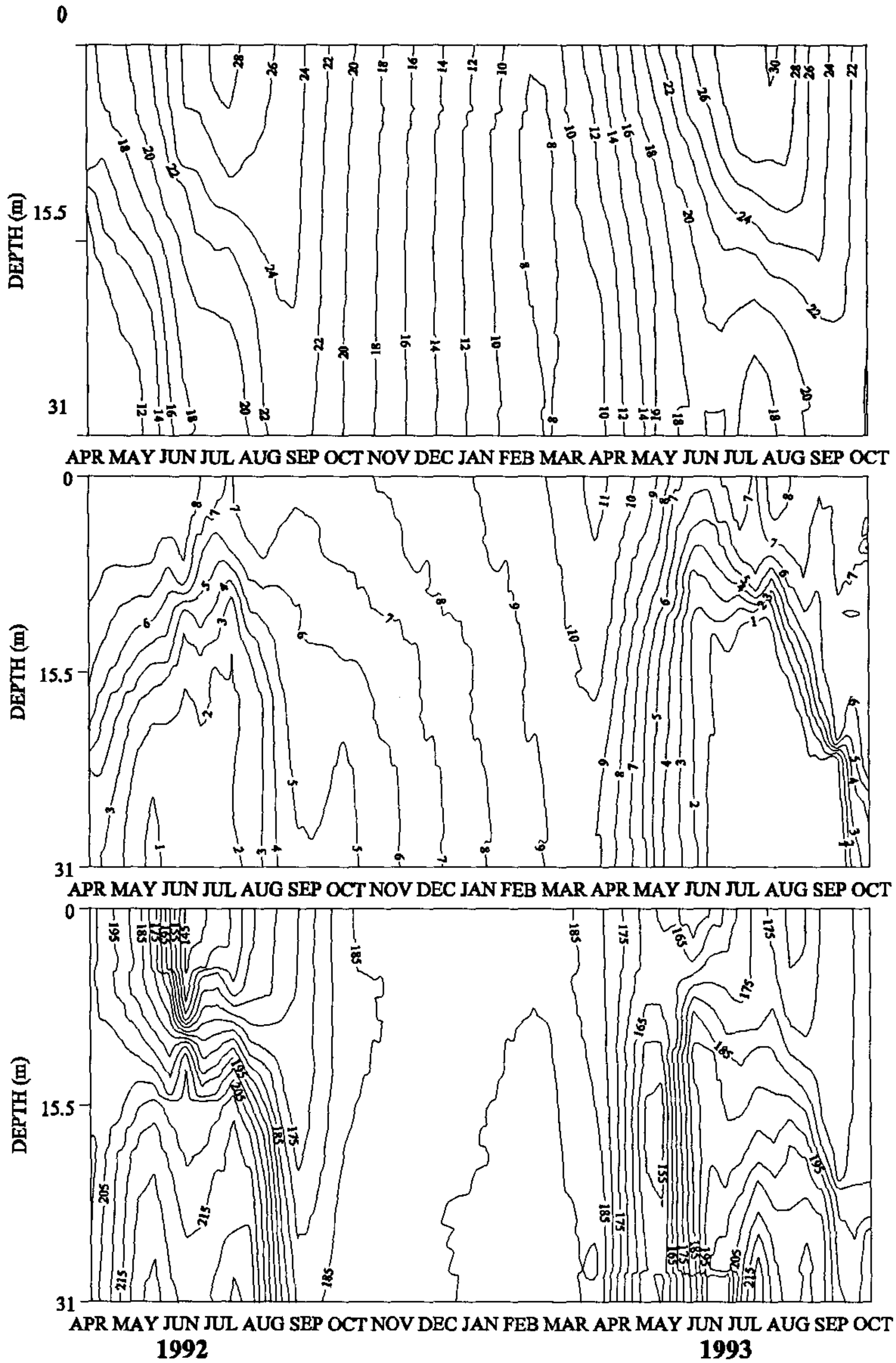



**Figure 18** Lake Tenkiller Station 6 isopleths for temperature (°C, top), dissolved oxygen (mg/l, middle), and conductivity ($\mu$℧/cm at 25°C, bottom) for CY 92-93.

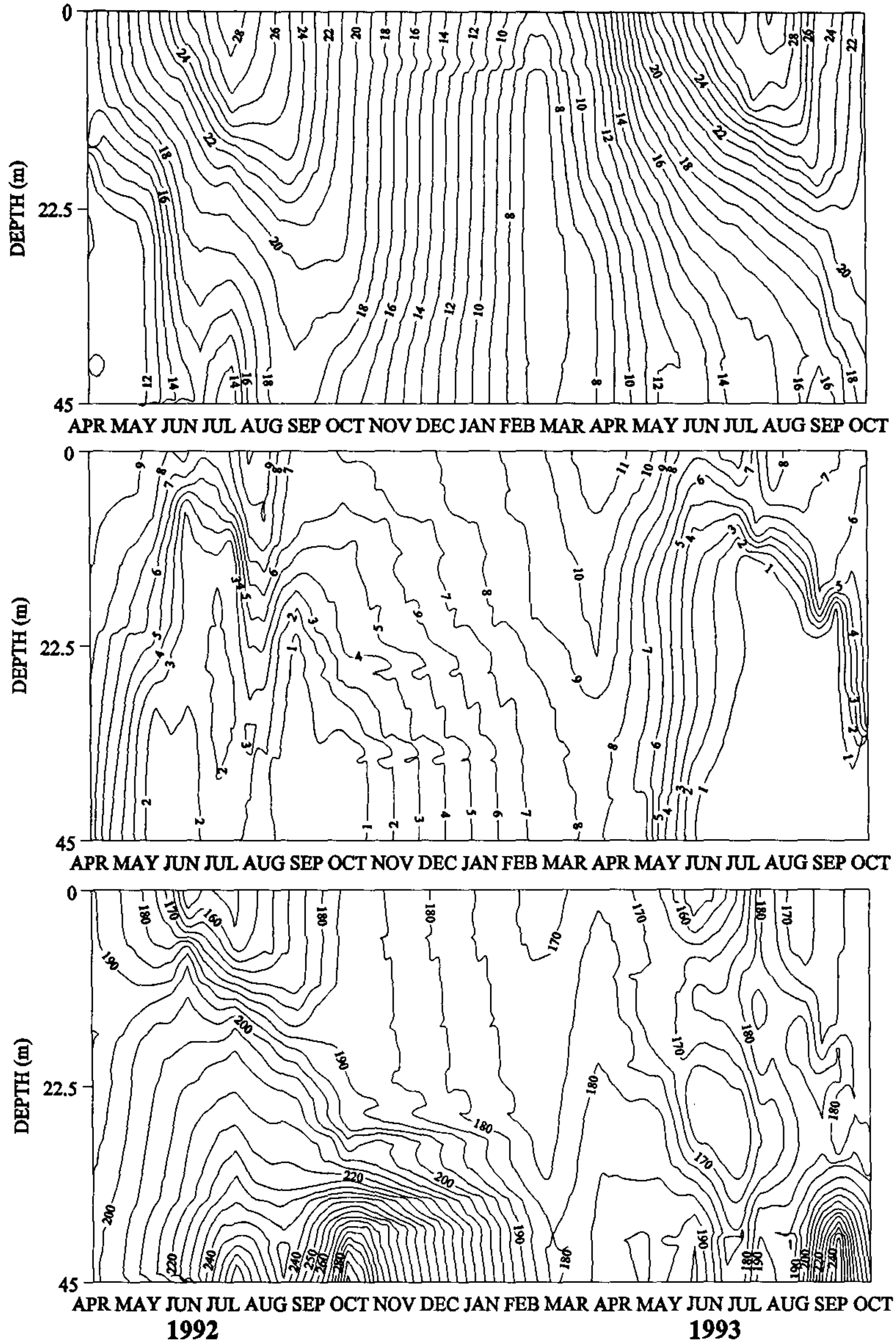
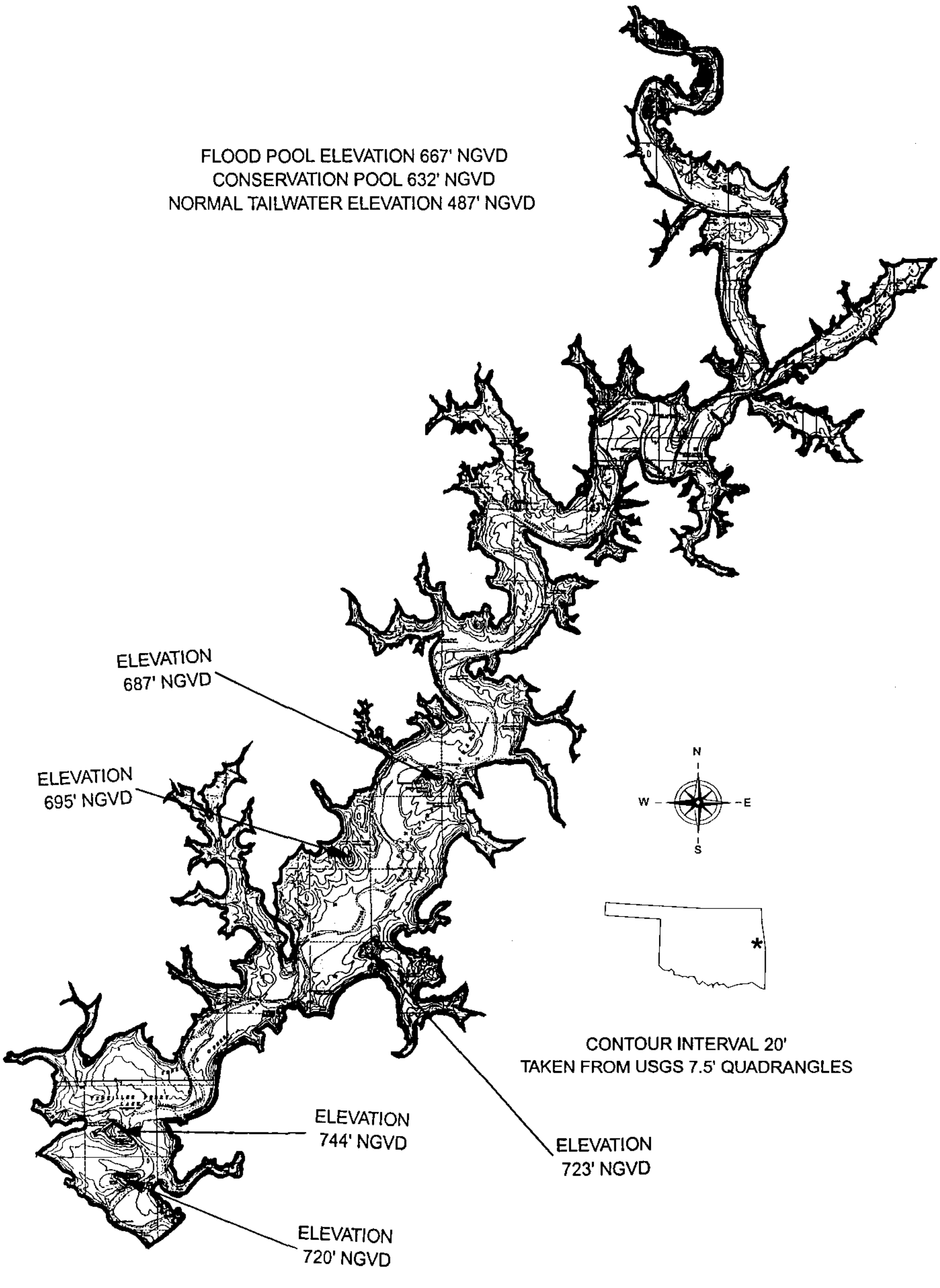



**Figure 19** Lake Tenkiller Station 7 isopleths for temperature (°C, top), dissolved oxygen (mg/l, middle), and conductivity ($\mu$℧/cm at 25°C, bottom) for CY 92-93.

8. Bathymetric Map

(see insert - next page)


FLOOD POOL ELEVATION 667' NGVD
CONSERVATION POOL 632' NGVD
NORMAL TAILWATER ELEVATION 487' NGVD
ELEVATION 687' NGVD
ELEVATION 695' NGVD
N
W
E
S
CONTOUR INTERVAL 20'
TAKEN FROM USGS 7.5' QUADRANGLES
ELEVATION 744' NGVD
ELEVATION 723' NGVD
ELEVATION 720' NGVD

# 9. Sediment Analytical Technique and Analyses

As per the CFR 40 part 35 subpart H appendix A requires monitoring metals contamination, we initially scanned the water and sediment for heavy metals. However, no water quality standards or USGS alert or warning levels were violated. Therefore, we focused on sedimentary processes and how they might relate to historical conditions of the lake. This phase was used as a thesis and is taken (with slight modification) from Wagner (1995) which was part of this "clean lakes" project. Understanding of the indices referenced in this section warrants a brief review of pertinent literature and thus follows.

Paleolimnological studies are important for effective ecosystem management. Effective ecosystem management requires long term data so that baseline conditions, natural variability, and the point in time that the system changed can be established. The main sources of data are direct historical measurements, comparisons to similar but unaffected ecosystems, hindcasts using computer models, and paleolimnological reconstructions. However, direct historical measurements are rarely available, continuous data collection on all lakes is not economically feasible, and computer models contain much uncertainty. Thus, paleolimnological studies are important in unraveling lake history archived in the sediments and defining the ecosystem that existed prior to human impact (Smol 1992).

Trace element chemistry in lakes is influenced by biological productivity and the development of an anoxic hypolimnion (Kuhn et al. 1994). As primary productivity increases, hypolimnetic dissolved oxygen consumption increases (Gachter and Meyer 1990). Elevated levels of productivity and organic input can result in the development of anoxic conditions in the hypolimnion during summer thermal stratification. During this critical stage, the redox of the hypolimnion changes dramatically resulting in greater recycling of iron, manganese, and other trace elements (Wetzel 1983). Therefore, the degree of oxygenation of the hypolimnion could reflect the productivity of the lake with less productivity indicated by more hypolimnetic oxygen and increasing productivity indicated by hypolimnetic oxygen depletion. Increasing and decreasing degrees of oxygenation are then manifested in varying redox conditions.

Changes in organic input results in changes in redox (Horowitz 1991). Therefore, paleoredox conditions can be useful indicators of historic productivity. Several methods have been developed for determining paleoredox conditions. The precipitation and solubility of iron and manganese can be controlled by redox conditions. Since changes in redox conditions can alter the amounts of iron and manganese flushed from the lake and the amounts retained in the sediments, sedimentary iron and manganese concentrations are useful indicators of the paleoredox conditions in lakes (Engstrom and Wright 1984). Mackereth (1966) used the iron to manganese ratio (Fe:Mn), along with iron and manganese concentration-depth profiles, in sediment cores to determine the paleoredox conditions of several lakes in the English Lakes District. However, Engstrom and Wright (1984) said that the Fe:Mn should be used with caution when reconstructing paleoredox conditions of lakes and suggested that supporting evidence be used when possible.

Hallberg (1972) suggested that differing mobilities of copper and zinc (i.e., $R=Cu/Zn$) could be used to indicate paleoredox conditions. During reducing conditions metals are precipitated as metal sulfides. The solubility products of metal sulfides indicate that precipitation of copper is more favored than zinc during reducing conditions. During oxidizing conditions the solubilities of zinc and copper are similar. Therefore, R drops under oxidizing conditions and rises under reducing conditions. This paleoredox indicator was later expanded to RP=(Cu+Mo)/Zn by Hallberg to calibrate it to more natural conditions. Frevert and Sollmann (1987) determined the paleoredox conditions of Lake Kinneret (Israel), a regularly stratified and mixed lake, using the paleoredox indicator RP.

Brooks et al. (1969) and Powell and McKirdy (1973) postulated that the pristane to phytane ratios (Pr:Ph) of ancient sediments and oils reflected paleoenvironmental conditions. Pristane and phytane are both products of chlorophyll decomposition. Pristane formation is dominant in aerobic environments and phytane formation is dominant in anaerobic environments. Therefore, a high Pr:Ph may indicate an aerobic environment and a low Pr:Ph may indicate an anaerobic environment. Didyk et al. (1978) used the Pr:Ph to determine the paleoredox conditions of the Cariaco Trench and the Black Sea, both of which are presently anoxic. Didyk emphasized, however, that the Pr:Ph should not be regarded as the definitive indicator of paleo-environmental conditions. After further search of the literature, it was concluded that this method did not provide an adequate indicator of paleoredox conditions.

In addition to the paleoredox indicators, other indicators of historical productivity were used. Calcium should provide a good indicator of historic trends in the productivity of Lake Tenkiller. Increased biological productivity generally results in increased calcium carbonate concentrations in the sediments (Hickman and Klarer 1981; Flannery et al. 1982). If $CO_2$ is removed from a solution where $Ca(HCO_3)_2$, $CO_2$, $H_2CO_3$, and $CO_3$ are in equilibrium, $CaCO_3$ will precipitate until equilibrium is reestablished. When $CO_2$ is removed from the water by photosynthetic organisms, large amounts of $CaCO_3$ are precipitated (Wetzel 1983). Therefore, increasing productivity should result in increasing calcium concentrations in the sediment.

Phosphorous is an essential nutrient for phytoplankton (Cole 1975). Its concentrations are generally closely related to productivity in natural systems (Engstrom and Wright 1983). Because of this association, it has been used by many researchers to predict trophic state (Leach and Herron 1992). Because phosphorous is generally efficiently fixed in sediments, stratigraphical trends in sedimentary phosphorous may be useful in determining the historical trends in trophic state (Engstrom and Wright 1984).

Sodium was used as an indicator of past erosion (Anderson and Rippey 1988). When erosion occurs, unweathered minerals containing high concentrations of sodium are transported to the lake and sedimented. Because these minerals and their components are rarely altered in aquatic systems, their distribution in sediments is useful for determining weathering and erosion (Engstrom and Wright 1984). Increasing sedimentary sodium concentrations generally indicate increasing erosion (Sekar et al. 1992) while declining sedimentary sodium concentrations indicate decreasing erosion (Hickman and Klarer 1981; Sekar et al. 1992).

Cesium-137, which was analyzed by the USDA-ARS in Durant, Oklahoma, was used to date the sediments and determine the historical sedimentation rate of the lake.

The objectives of this part were to evaluate the potential of trace element concentration-depth profiles and ratios for determining the paleoredox conditions and historical trends in the productivity of Lake Tenkiller. The specific tasks were the following:

1) measure phosphorous, zinc, copper, molybdenum, calcium, sodium, iron, and manganese concentration-depth profiles from a sediment core taken from Lake Tenkiller and calculate the Fe:Mn ratio, R=Cu/Zn, and RP=(Cu+Mo)/Zn,

2) date the sediments using Cesium-137,

3) use the Fe:Mn ratio, R=Cu/Zn, RP=(Cu+Mo)/Zn, iron, manganese, molybdenum, calcium, and phosphorous to determine the historic productivity, and

4) use the sodium profile to determine past erosion intensity.

## Trophic Classification Using Hypolimnetic Dissolved Oxygen

Trophic classification based on hypolimnetic dissolved oxygen concentrations is not a new concept. In the early 1900s, Thienemann proposed hypolimnetic dissolved oxygen depletion as an indicator of trophic state. Later, Hutchinson (1938) and Hutchinson and Mortimer (Hutchinson 1957) expanded on Thienemann's work suggesting that areal hypolimnetic oxygen depletion rates provided a useful indicator. In 1976, Burns suggested that both areal and volummetric hypolimnetic oxygen depletion rates were needed for correct trophic classification. In addition to these researchers, trophic classification based on hypolimnetic oxygen has also been used by Lueschow and others in 1971 as a Trophic Index Number (TIN) in EPA's National Eutrophication Survey (1974), and in Uttormark and Wall's (1975) Lake Classification Index (LCI). Walker (1979) expanded Carlson's (1977) TSI using hypolimnetic oxygen as it applies to trophic state. Chapra and Dobson (1981) observed that measurement of the hypolimnetic dissolved oxygen deficit was essential in trophic classification (Leach and Herron 1992). Additional models have been developed for determining volummetric hypolimnetic oxygen depletion (Vollenweider and Janus 1982, Coffey et al. 1989) and the probability of having an oxic hypolimnion (Reckow 1988, Coffey et al. 1989).

## Relationship Between Dissolved Oxygen and Redox

According to Wetzel (1983) redox potential in water is affected by oxygen concentrations according to the reaction,

$$H_2O = \tfrac{1}{2}\, O_2 + 2H^+ + 2e^-.$$

However, clinograde oxygen profiles do not necessarily result in clinograde redox profiles (Hutchinson 1957). Even though a clinograde oxygen profile is present, the redox remains positive and high as long as the water is not near anoxia (Wetzel 1983). However, in most lakes with clinograde oxygen profiles and extreme oxygen deficits, clinograde redox profiles exist (Hutchinson 1957). This is because redox quickly decreases as dissolved oxygen nears zero and anoxia occurs in the lower hypolimnion and sediments (Wetzel 1983). In most cases, measurement of dissolved oxygen is sufficient to estimate redox (Goldman and Horne 1983).

Organisms act as redox catalysts by mediating redox reactions and the transfer of electrons (Wetzel 1983). The balance between photosynthesis and respiration is primarily responsible for regulating lake redox conditions. Products of photosynthesis settling into the hypolimnion and sediment act as reductants by supplying electrons (Stumm and Baccini 1978). Most redox reactions are carried out by bacteria (Goldman and Horne 1983).

## Iron and Manganese Cycling in Aquatic Systems

Much research has been conducted on the cycling of iron and manganese in aquatic environments. Iron and manganese are closely associated (Hsiung and Tisue 1994). Most iron and manganese transported into lakes is associated with mineral particles resulting from erosion or resulted from rock or soil weathering (Mackereth 1966). Acid rain can increase transport of iron and manganese. In addition, organic acids can vastly increase the mobility of iron and manganese in water by lowering the pH, enhancing reducing conditions, and forming organometallic complexes. Correlations between iron and manganese and sedimentary organics have been noted in lake sediments. Anoxic groundwater may also be important in transporting iron and manganese to lakes.

Iron and manganese compounds are generally insoluble under oxidizing conditions (Engstrom and Wright 1984). In oxygenated waters, iron is primarily found as insoluble iron hydroxide ($Fe(OH)_3$) or associated with colloidal or suspended particles, with only low concentrations found in ionic form (Engstrom and Wright 1984, Vuorinen et al. 1986). Manganese is more soluble than iron, however, it is more complexed with sulfates, carbonates, and organic acids (Engstrom and Wright 1984). In natural waters, manganese can be reduced by reaction with $Fe^{2+}$, readily oxidizable organics, and reduced sulfur compounds. In addition, manganese reduction by organic matter is promoted by sunlight. Microbial mediations dominate manganese biogeochemistry (Hsuing and Tisue 1994).

Most iron and manganese flowing into lakes is transported to the sediments (Davison et al. 1982). In oxic waters, iron and manganese are deposited as hydrated oxides and with coagulates of humic organics. Manganese may also be precipitated by sorbing onto iron oxides (Engstrom and Wright 1984). Because iron and manganese oxide precipitates are fine (Engstrom and Wright 1984, Horowitz 1991), wave and current action transports them to more protected areas. Therefore, iron and manganese concentrations generally increase with increasing depth of water (Engstrom and Wright 1984).

In oxic environments, iron and manganese become enriched in the sediments because of the efficient precipitation mechanisms (Mackereth 1966). Once in the sediments, microbial degradation of organic matter releases iron and manganese which are subsequently transformed to immobile hydrous oxides. Most iron and manganese in oxygenated surface sediments exists as immobile hydrous oxides. However, large concentrations of iron and manganese in sediments can also be found bound to humic acids or in ferromanganese nodules and crusts. The stability of iron and manganese is increased by interaction with organic and sulfur compounds as well as with equilibrium with enriched interstitial waters (Engstrom and Wright 1984). Kjensmo (1988) found that the redox sensitive elements (Iron and Mn) accumulate in organic sediments. The concentrations of iron and manganese oxides in sediment are important, because they contribute to the metal-sorption potential of sediment and greatly influence the quantities of other trace metals present (Jenne and Zachara 1987).

In most lake sediments, only a thin layer at the surface of the sediments is oxygenated. Below this layer, microbial activity and diagenesis of inorganic chemicals deplete the sediments of oxygen and cause reducing conditions (Engstrom and Wright 1984). Most redox reactions do not occur at significant rates unless they are microbially mediated (Kuhn et al. 1994). Degradation of organic compounds by microorganisms results in the sequential consumption of oxygen, $Mn^{4+}$, nitrate, $Fe^{3+}$, sulfate, and bicarbonate (Carlton and Klug 1990, Adams et al. 1990). This consumption of oxygen causes a sharp separation between the oxidizing and reducing environments (Peiffer 1994), which is known as the redox-cline.

As sediments are moved across the redox-cline due to progressive burial, iron and manganese are reduced causing them to enter the interstitial water (Engstrom and Wright 1984). Reduction of oxidized manganese occurs more rapidly than the reduction of oxidized iron (Hsuing and Tisue 1994) to its ionic form $Fe^{2+}$ (Vuorinen et al. 1986). Upward diffusion may occur until the iron and manganese are reprecipitated at the redox-cline as ferric and manganic compounds (Engstrom and Wright 1984, Sakata 1985, Anderson and Rippey 1988, Peiffer 1994). This can produce iron and manganese concentration peaks at the top of the sediment profile. However, surface enrichment of iron usually can not be recognized, because the sedimentary flux is greater than the diffusive flux (Sakata 1985). Because of this post-depositional migration, trends at the top of cores must be carefully interpreted.

If surface sediments remain oxidized, then iron and manganese remain relatively immobile even below the redox-cline (Engstrom and Wright 1984). Iron is generally retained by the sediments with minimal loss (Davison et al. 1982), however, it alternates between mobile and immobile states (Cole 1975).

Diagenetic effects on iron and manganese in lake sediments are pronounced in eutrophic systems, where seasonal anoxia, organic accumulation, and changes in redox are substantial (Anderson and Rippey 1988). In the hypolimnion of eutrophic lakes, microorganisms remove oxygen and reduce iron and manganese hydroxides (Kuhn et al. 1994). When the hypolimnion becomes anoxic, iron and manganese are reduced and released from the sediments into the hypolimnion (Engstrom and Wright 1984, Sakata 1985, Anderson and Rippey 1988, Kuhn et al. 1994). However, iron and manganese may begin dissolving before complete deoxygenation of the hypolimnion (Davison et al. 1982).

Iron and manganese diffuse into anoxic water from sediments when the redox potential drops to 0.20 isovolts (Eh) above the sediment interface. Manganese is released from the sediment before iron because of its greater solubility and manganese is selectively depleted and released moderately anaerobic hypolimnion (Engstrom and Wright 1984). Loss of iron from sediments results mainly from microbial reduction of ferric oxides (Peiffer 1994, Kuhn et al. 1994) and occurs with low redox and extreme hypolimnetic anoxia (Anderson and Rippey 1988). Therefore, if anoxia becomes severe, both iron and manganese will be removed from the sediment. Continued loss of iron and manganese from the sediments occurs as iron and manganese are desorbed from the solid phases and as hydrous oxides dissolve (Engstrom and Wright 1984).

The release of metals from the sediment results in increased metal concentrations in the hypolimnion. High concentrations of dissolved iron accumulates in the hypolimnion during summer anoxia (Davison et al. 1982). The sedimentation of phytoplankton from the epilimnion also carries metals associated with the algae into the hypolimnion and sediments (Kuhn et al. 1994). Substantial dissolution of manganese and some dissolution of iron occurs in lower anoxic water of lakes (Davison et al. 1982). Davison (1981) found that most iron accumulated in an anoxic hypolimnion was derived from the sediment, while most manganese was supplied by dissolution from particulate matter sedimenting through the water column. This contradicts previous studies which attribute the hypolimnetic manganese to diffusion from sediment alone. Regardless of whether manganese diffuses from the sediments, dissolves from particulates, or both, less manganese will be reaching the sediment.

Manganese is generally rapidly reduced even when the hypolimnion is oxic. Most manganese entering a seasonally anoxic lake will be washed out, generally following one redox cycle so that less than 10% is retained by the sediments. The residence time of manganese is generally less than 1 year (Davison et al. 1982). A recent study by Hsuing and Tisue (1994) indicates that more manganese may be released from sediments when the hypolimnion is oxygenated then when it is anoxic. The mobilization and accumulation of manganese in the hypolimnion was found to be linked to the degradation of organic matter, which occurs more rapidly under aerobic conditions. Anoxia may inhibit the release of manganese associated with organic matter from the sediments, because degradation is decreased. Therefore, aerobic degradation of organic matter by microorganisms may be more important in releasing manganese from sediments than dissolution of manganese oxides during anoxia (Hsuing and Tisue 1994).

When the redox potential goes below 100 mv, $H_2S$ is formed through $SO_4$ reduction. Since iron is released before hydrogen sulfide (200-300 mv), iron and hydrogen sulfide will quickly form insoluble iron sulfide (FeS) when the redox potential goes below 100 mv (Engstrom and Wright 1984). In eutrophic lakes, reoxidation of $Fe^{2+}$ and sulfide does not occur causing an accumulation of iron sulfide in the sediments (Peiffer 1994). However, this recycling of iron between sediment and water through dissolution and precipitation as FeS is generally small compared with other iron fluxes (Davison et al. 1982). Manganese sulfide (MnS) is more soluble than iron sulfide (FeS), therefore, hydrogen sulfide does not inhibit the release of manganese from the sediments. Iron sulfide may also form below the redox-cline in sediments under oxic waters (Engstrom and Wright 1984).

Precipitation and sedimentation of both iron and manganese occurs at fall mixing (Davison et al. 1982). The reoxidation of $Fe^{2+}$ is rapid while the reoxidation of $Mn^{2+}$ is slow (Kuhn et al. 1994, Hsiung and Tisue 1994, Peiffer 1994). The oxidation of manganese can be catalyzed by adsorption onto oxide surfaces. However, most manganese oxidation is microbially mediated (Kuhn et al. 1994). The oxidation of $Mn^{2+}$ is initiated by bacteria after which abiotic processes complete the oxidation (Hsiung and Tisue 1994). Precipitation of iron and manganese oxides at overturn can carry significant amounts of trace metals to the sediments (Horowitz 1991, Kuhn et al. 1994).

The precipitation and solubility of iron and manganese is primarily controlled by the ionic composition, pH, and redox condition of the water. Since long-term changes in redox conditions could alter the amount of iron and manganese being flushed from the lake and the amount being retained in the sediments, iron and manganese concentrations in the sediments can be used to determine the paleoredox conditions in lakes (Engstrom and Wright 1984). Because paleoredox conditions are controlled by organic input, it can be used to determine the paleo-productivity of the lake.

### Iron and Manganese as Paleoredox Indicators

Iron and manganese can be useful in paleolimnological studies, because their concentrations indicate the oxygenation of the hypolimnion (Kjensmo 1988) and paleoredox conditions. However, profiles are difficult to interpret because changes in the rate of supply and limnological conditions determine the iron and manganese concentration in the sediment (Mackereth 1966, Engstrom and Wright 1984).

Changes in the supply of iron and manganese can be difficult to distinguish in sediment concentration-depth profiles from changes produced by the redox potential of the hypolimnion (Engstrom and Wright 1984). If the iron and manganese concentrations result from erosion, the sediment Fe:Mn will be equal to that of the soils of the watershed or the Fe:Mn profile will follow the profile of an erosion indicator such as sodium (Mackereth 1966). A high Fe:Mn in lake sediments results from oxidizing conditions in the soil. However, if iron and the Fe:Mn are positively correlated, then it is caused by changes in soil redox (Engstrom and Wright 1984). A low Fe:Mn results from reducing conditions in the soil, because in mildly reducing conditions manganese is transported preferentially (Mackereth 1966, Engstrom and Wright 1984). If iron concentrations are consistent with erosion indicators and manganese increases in the sediment, then it can be concluded that mildly reducing conditions in the watershed are causing preferential removal of manganese which is then precipitated to the sediments in oxic lake environments (Mackereth 1966).

When iron maxima corresponds with Fe:Mn maxima, the iron concentration in the sediment is produced by varying rates of supply from the watershed. If the Fe:Mn is consistent throughout the sediment profile, then it can be concluded that redox conditions have remained at constant levels (Mackereth 1966).

When an anoxic hypolimnion develops, a significant decrease in sedimentary manganese can be observed in the manganese profile (Mackereth 1966). Loss of manganese from lake sediment by diffusion and its subsequent washout at overturn is indicated by a decrease in sediment concentration and reflects the onset of eutrophication (Anderson and Rippey 1988). The position of the marked fall in the manganese concentration-depth profile indicates the development of the anoxic hypolimnion and eutrophic conditions (Mackereth 1966). The drop in sedimentary manganese concentration is an indicator of redox-related cycling (Anderson and Rippey 1988). This release of manganese causes the Fe:Mn to increase. Therefore, increasing Fe:Mn can indicate a change from oxidizing to reducing environments (Cole 1975). If a peak in the Fe:Mn correlates with a minima in the iron concentration, then its caused by changes in the hypolimnetic redox conditions (Engstrom and Wright 1984).

Maximum oxidation results in the correspondence of a iron maxima with a Fe:Mn minima which is preceded and followed by a Fe:Mn maxima (Mackereth 1966). The final stages of anoxia are characterized by increased iron and sulfide content in the sediment and an increased Fe:Mn (Engstrom and Wright 1984). However, overturn of many eutrophic lakes terminates stratification before this stage occurs (Cole 1975).

If watershed soils are reduced and the hypolimnion is anaerobic, then the sedimentary manganese concentration will be low and Fe:Mn will be high (Mackereth 1966).

The Fe:Mn may vary spatially in lake sediments due to the greater solubility of manganese. Thus the Fe:Mn decreases with distance from a source such as a river. In single-core studies, such variations are minor relative to changes in overall inputs and preservation (Engstrom and Wright 1984).

Post-depositional migration below the redox-cline may alter the initial Fe:Mn and result in poor correspondence between the Fe:Mn and other evidence for hypolimnetic oxygen conditions (Engstrom and Wright 1984). Mechanical disturbance of sediments can also alter original sediment profiles (Mackereth 1966). Because several factors control the flux of iron and manganese from the sediments, interpretation of profiles for paleolimnological purposes should be done with caution, and supporting evidence should be used when possible (Engstrom and Wright 1984).

Cycling of Zinc, Molybdenum, and Copper

While redox is significant in iron and manganese cycling, living organisms are chiefly responsible for the cycling of zinc, copper, and molybdenum. Whereas iron and manganese are easily released from anoxic sediment, zinc, molybdenum, and copper are less mobile

(Goldman and Horne 1983). The cycling of zinc, copper, and molybdenum is primarily controlled by uptake by organisms and subsequent release by decay from the sediments (Mackereth 1966, Goldman and Horne 1983).

In most lakes, zinc is supplied by anthropogenic sources and the flow of trace element loaded sediment caused by erosion (Goldman and Horne 1983). Acid rain can also result in high rates of leaching of zinc from poorly buffered soils (Wetzel 1983).

In freshwater, zinc is present in many forms: in its ionic form as a divalent cation; bound to siliceous matter, iron oxyhydroxides, diatoms, and sulfate; complexed in organic materials; absorbed on solids; and incorporated in crystalline structures (Wetzel 1983, Vuorinen et al. 1986). Most zinc is transported as crystalline solids or absorbed on solids with little zinc present in its ionic form (Wetzel 1983). In basic waters, zinc is primarily absorbed on colloidal particles. High zinc concentrations are associated with low pH. In acidic waters, approximately 50% of the zinc is present in its noncolloidal inorganic form (Nriagu 1980).

During summer stratification, algal uptake and sedimenting detritus plays an important role in the cycling of zinc (Wetzel 1983). Zinc is also removed from the water column by precipitation with clay, hydrous iron and manganese oxides, and calcium carbonate (Mackereth 1966, Cole 1975, Nriagu 1980). The zinc bound to biomass or manganese oxides is then transported to the sediments (Kuhn et al. 1994). Zinc is insoluble in oxidized states and in intensely reducing environments where sulfides are formed (Cole 1975). Zinc also precipitates at high alkalinities (Nriagu 1980). Sulfide increases the precipitation and sedimentation rate of zinc (Cole 1975, Kuhn et al. 1994).

Sedimentary zinc concentrations are indirectly influenced by redox conditions, because anoxic conditions favor its retention. Zinc is generally efficiently retained by sediments (Mackereth 1966, Kuhn et al. 1994). In reducing sediments, zinc is held as an insoluble sulfide (Mackereth 1966). Degradation of sedimentary organic detritus is the primary source of hypolimnetic zinc (Wetzel 1983).

Molybdenum occurs in seven stable and five radioactive isotopes. In freshwater, the molybdenum cycle is similar to iron's. Because molybdenum is a micronutrient, it is concentrated in plankton which eventually carry it to the sediments. Molybdenum concentrations in the water column are generally low while the sediments are oxidized. However, during summer thermal stratification, molybdenum may accumulate in the hypolimnion. Molybdenum is most abundant in water when phosphorous is released from the sediments. However, molybdenum concentrations decrease when $Fe(OH)_3$ and FeS precipitate (Cole 1975).

Copper exists in ionic, organic, and sestonic forms in water. Most copper is associated with colloidal organic matter. In aquatic systems, copper is virtually immobile. Copper uptake by plants and its subsequent release from decaying plants is responsible for most of its cycling (Hutchinson 1957).

Copper is insoluble in oxic environments and in intensely reducing environments where sulfides are formed (Cole 1975). At normal pH, most copper is lost from solution by precipitation, sorption, or chelation by organic matter (Goldman and Horne 1983). Copper also coprecipitates with iron and manganese (Mackereth 1966).

Once in the sediment, copper is firmly bound by sulfides (Mackereth 1966, Vuorinen et al. 1986) and organic matter (Vuorinen et al. 1986). Some copper is released under anoxic conditions from the sediments by dissolution of ferromanganese oxides and hydroxides, and organic degradation. However, copper is usually not released in appreciable amounts indicating that diagenetic remobilization is negligible (Sakata 1985). Bacterial decomposition of organic matter is primarily responsible for the release of copper. Increasing copper concentrations near the bottom may be observed because of the decomposition of copper containing seston or diffusion of copper containing organic compounds from the sediments. When the surface sediments are reduced, copper is generally present as CuS and immobile. Copper is more mobile when oxidizing conditions are present (Hutchinson 1957).

Copper, Molybdenum, and Zinc as Paleoredox Indicators

After sedimentation, debris is attacked by chemical and biochemical processes. However, these processes differ depending on redox conditions. Reducing environments created in the absence of oxygen are characterized by hydrogen sulfide ($H_2S$) production. Hydrogen sulfide causes metals to precipitate in quantities dependent on their sulfide solubility product. Therefore, sulfide solubility products are important for metal precipitation in reducing environments.

In sediments, microbial decomposition produces intermediate compounds which act as metal chelators. As a result, competition occurs between $H_2S$ and the chelating compounds for metals. However, chelating ability differs for different metals. Chelated metals can not be trapped in the sediment and fixed as sulfides. Thus, metals are generally either fixed in the sediment as sulfides, or in solution as metal chelates.

Chelated metals are located above the redoxcline, and sulfides are located below. If the redoxcline is above the sediment surface, the metal chelates are released from the sediment. Conversely, little metal is released when the redoxcline is in the sediment. Metal sulfides are stable and remain in the sediment.

Differences between copper and zinc concentrations in oxic environments is primarily a factor of chelating ability (Hallberg 1972). Changes from oxic to anoxic conditions during diagenesis in the sediment leads to copper chelation and higher mobility of copper than zinc (Frevert and Sollman 1987). Copper has a higher chelating ability than zinc and is not as readily fixed as a sulfide in the sediments relative to zinc when the redoxcline is below the sediment surface (oxic environment). Copper has a greater tendency to chelate than to form a sulfide in oxic sediments. Zinc is fixed in oxic sediments and chelating compounds have little effect on its sedimentary concentrations.

If copper is precipitated directly as a sulfide in a reducing environment, then it may become more enriched in the sediment than if it was precipitated in an oxic environment. This is due to CuS being less soluble than oxidized or chelated copper (Hallberg 1972). The formation of copper sulfides is more favored than zinc sulfide formation in reducing environments. Therefore, Cu/Zn decreases under oxidizing conditions and increases under reducing conditions (Hallberg 1972, Frevert and Sollman 1987).

The paleoredox variable $R = Cu/Zn$ was later expanded to $RP = (Cu+Mo)/Zn$ by Hallberg. Molybdenum, like copper, is more abundant in the sediment in strictly reducing environments. However, copper is generally high in areas which are chemically reduced, while molybdenum is generally high in areas with high $H_2S$ levels.

Frevert and Sollman (1987) found that the paleoredox variable RP was also applicable to lakes experiencing seasonal anoxia if copper and zinc inputs were similar and constant, sedimentation rate was greater than or equal to the penetration of dissolved oxygen into the sediment, no resuspension of sediment occurred, sulfate was sufficient for microbial hydrogen sulfide reduction, and chelating mobilization of copper was negligible.

In Lake Kinneret, RP indicated the length of reducing and oxidizing periods in the hypolimnion during seasonal anoxia. Periods of increasing oxygen (shorter periods of anoxia) resulted in lower RP values. In contrast, periods of decreasing oxygen (longer periods of anoxia) resulted in a higher RP variable. Therefore, RP may provide a useful indicator of the relative duration of annual reducing and oxidizing periods if the five assumptions are true (Frevert and Sollman 1987).

Pristane:Phytane as Paleoredox Indicators

Terpenoid hydrocarbons, also known as isoprenoid hydrocarbons, are some of the most ubiquitous natural products (Blumer and Snyder 1965) and have attracted much attention from petroleum chemists since pristane was first isolated by Bendoraitis and others in 1962 (Trusell 1979). Isoprenoid hydrocarbons, such as phytane (2,6,10,14-tetramethylhexadecane) and pristane (2,6,10,14-tetramethylpentadecane), are naturally occurring compounds considered to be derived from the isoprene molecule (2-methyl-1, 3-butadiene) (Adlard 1979). Pristane and phytane are considered to originate from phytol the esterified side-chain of chlorophyll a (Didyk et al. 1978).

Pristane occurs in some terrestrial plants and animals. It is also abundant in marine crustaceans (copepods) which derive pristane from phytol in their food (Blumer and Snyder 1965). Both pristane and phytane are found in crude oil, oil shale, coal, and ancient sediments (Blumer and Snyder 1965, Brooks et al. 1969, Powell and McKirdy 1973). In recent sediments, pristane alone is generally found suggesting that phytane is a post-depositional product (Blumer and Snyder 1965, Brooks et al. 1969). Phytane is likely formed by the conversion of sedimentary phytol, which is too slow to generate detectable quantities of phytane in the uppermost sediment layer (Blumer and Snyder 1965).

Pristane appears at the brown coal stage of diagenesis suggesting that isoprenoid hydrocarbons could be derived from esters or by similar reaction routes. Pristane is always the major isoprenoid hydrocarbon in coal extracts or in artificial diagenesis. Therefore, phytol cannot be the principal precursor. The dominance of pristane suggests that it is formed by the decarboxylation of a $C_{20}$ isoprenoid acid, such as phytanic acid (Brooks et al. 1969, Powell and McKirdy 1973). Phytane is formed by dehydration and hydrogenation of phytol (Powell and McKirdy 1973). Phytanic acid (precursor of pristane) formation is less likely in the more anaerobic conditions of aquatic environments than it is in the more aerobic conditions on land during the decay of plant material. Therefore, a pristane:phytane ratio (Pr:Ph) may reflect the amount of oxygen present during the early stages of chlorophyll decomposition, with a low Pr:Ph indicating origination from an aquatic environment and a high Pr:Ph indicating land plant origin (Brooks et al. 1969, Powell and McKirdy 1973).

These organic geochemical indicators can be useful in paleoredox determinations, because they resist diagenesis and their abundance patterns often reflect the conditions of the environment in which they were deposited (Didyk et al. 1978). The relative abundance of pristane and phytane depend on the redox conditions and the acidity of the environment in which they are formed. Phytane is the principle product during anoxic conditions, while pristane is the main product during oxic conditions (Wakeham 1993).

The Pr:Ph of sediment can reflect paleo-environmental conditions. Sediment deposited during anoxic conditions has low Pr:Ph values. Sediment deposited during alternating anoxic/oxic conditions has Pr:Ph values near 1. Oxic conditions produce sediment Pr:Ph values greater than 1 (Didyk et al. 1978).

Several organic geochemistry studies and geological constraints suggest that the Pr:Ph is not a valid indicator of paleoredox conditions (ten Haven et al. 1987).

1. Terpenoid hydrocarbons are among the most ubiquitous natural products and reenter the present-day environment from petroleum and derived products (Blumer and Snyder 1965, ten Haven et al. 1987).

2. Partly decomposed terrestrial organic material washed into lakes can skew results, because the early decomposition occurred in an oxic environment (Brooks et al. 1969).

3. Archaebacterial lipids and pristane originating from tocopherols can provide additional sources of these isoprenoids.

4. Pristane and pristenes have been detected in zooplankton.

5. An analytical problem of precise determination of Pr:Ph arises from the identification of 2,6,10-trimethyl-7-(3-methylbutyl)-dodecane in sediments, which coelutes with pristane on most capillary columns. This raises doubt about the validity of Pr:Ph previously published.

6. All sediments are anoxic below a surface oxic layer.

7. Extraction of organic rich sediments from the eastern Mediterranean revealed that most of the phytol encountered (>95%) was present in an esterified form and in abundant quantities, while pristane and phytane were absent or present in low concentrations. When phytol is eventually degraded in these sediments, this process will take place under reducing conditions, giving rise to low Pr:Ph. Therefore, Pr:Ph reflects the result of reactions taking place well after reducing conditions have been established, not the oxicity of the environment of deposition.

It is virtually impossible to draw valid conclusions from Pr:Ph with respect to the oxicity of the environment of deposition (ten Haven et al. 1987).

### Calcium as a Paleolimnological Indicator

Calcium is generally considered an essential nutrient for most algae (Cole 1975, Wetzel 1983, Goldman and Horne 1983). However, in aquatic environments it is rarely deficient, because typical inland waters are basically a solution of $CaCO_3$. Calcium is abundant in the earth's crust. Calcium carbonate is found in nature as calcite and aragonite. Calcium carbonate is also abundant in soils; however, it is only weakly soluble (Cole 1975). Other sources of calcium are gypsum, dolomite, anhydrite, flourite, plagioclase, pyroxene, and amphibole (Hounslow 1993). Calcium carbonate is generally insoluble in water, except in the presence of acid (specifically carbonic acid), where it becomes $Ca(HCO_3)_2$ (Cole 1975). Calcium bicarbonate is relatively soluble in water (Wetzel 1983).

Calcium is present in aquatic systems as suspended particulates (mainly $CaCO_3$) and in its ionic form. Calcium levels, bicarbonate, pH, and conductivity are all correlated in lake water (Goldman and Horne 1983). In many lakes, specific conductance follows changes in $Ca^{2+}$ and $HCO_3^-$ concentrations in lake water (Wetzel 1983). Calcium carbonate solubility, which is also temperature dependent, decreases as temperature rises from O° to 35° C (Cole 1975). The concentration of calcium in lakes is primarily controlled by precipitation of $CaCO_3$ during photosynthesis and the solution of calcium by rainwater in the watershed (Goldman and Horne 1983).

The most common sinks for calcium in aquatic systems is as calcite and gypsum (Hounslow 1993). If $CO_2$ is removed from a solution where $Ca(HCO_3)_2$, $CO_2$, $H_2CO_3$, and $CO_3$ are in equilibrium, $CaCO_3$ will precipitate until equilibrium is reestablished. The general equation (Wetzel 1983) for this process is:

$$Ca(HCO_3)_2 \longleftrightarrow CaCO_3 + H_2O + CO_2$$

Loss of $CO_2$ results in a shift in the reaction to the right causing massive precipitation of calcium carbonate (Cole 1975, Wetzel 1983). The rate of calcium carbonate precipitation is slow unless induced by metabolic reactions, such as photosynthesis (Wetzel 1983). Photosynthetic organisms trigger this precipitation by absorbing $CO_2$ (Cole 1975, Goldman and Horne 1983). Plant cells serve as centers for the formation of particulate calcium carbonate (Wetzel 1983). The reaction to $CaCO_3$ is basically irreversible in lakes, because carbonic acid is generally lost soon after $CO_2$ uptake by plants (Goldman and Horne 1983).

During photosynthesis, large amounts of calcium carbonate are precipitated by algae and macrophytes. Decreasing epilimnetic and metalimnetic calcium concentrations have been directly related to increased photosynthetic utilization of $CO_2$. The precipitation of calcium carbonate is also responsible for the removal of nutrients from the water column, because phosphates coprecipitate with carbonates (Wetzel 1983). Calcium is also abundantly sedimented during intense erosion. However, in productive lakes, calcium is associated more with organic matter than erosion products, except for during periods of intense erosion (Mackereth 1966).

Carbon dioxide released from the decay of organic matter may hinder the deposition of $CaCO_3$ (Cole 1975). As precipitating $CaCO_3$ moves through the hypolimnion, some is resolubilized; however, most is permanently incorporated into the sediments. Under oxic conditions, $Ca^{2+}$ is incorporated directly into the sediments. In addition, association with organic detritus reduces the dissolution rate of sedimenting calcium carbonate allowing it to become incorporated permanently into the sediments. However, when the hypolimnion becomes anoxic, some $Ca^{2+}$ is released from the sediments (Wetzel 1983).

Calcium concentrations generally followed carbonate concentrations in a sediment profile from Lake Isle indicating that most calcium was deposited as calcium carbonate (Hickman and Klarer 1981). Sediment calcium concentrations have also been found to be closely associated with organic matter (Mackereth 1966, Flannery et al. 1982). High calcium concentrations in surface sediments were observed in Florida lakes with high TSI values. This sedimentary calcium enrichment was attributed to increased biogenic calcium sedimentation resulting from increased primary productivity (Flannery et al. 1982). Therefore, stratigraphical analysis of calcium concentration-depth profiles should provide an accurate assessment of historical productivity, because increased biological productivity generally results in increased $CaCO_3$ concentrations in the sediments (Hickman and Klarer 1981).

## Phosphorus as a Paleolimnological Indicator

Phosphorus is an essential nutrient (Cole 1975), therefore, a close relation exists between its concentration and primary productivity (Engstrom and Wright 1984). Because of this, it has been used to predict algal biomass and trophic state. Phosphorus has been used to predict trophic state by Carlson (1977), Vollenweider (1968), the EPA (1974), and many others. Hakanson (1984) used sedimentary nitrogen, phosphorus, carbon, and loss on ignition to indicate trophic state (Leach and Herron 1992). Phosphorus may also provide a possible indicator of paleo-productivity. However, deviations are possible due to limnological factors controlling phosphorus incorporation in the sediments (Engstrom and Wright 1984).

Igneous rock was the original source of phosphorus. However, now it is mined and used extensively by industry and agriculture (Cole 1975). Most phosphorous in fresh water is bound in organic phosphates, cellular constituents of organisms, and adsorbed to organic colloids. Orthophosphate, which is directly used by organisms, is generally present in small concentrations due to rapid uptake and sedimentation (Wetzel 1983).

Phosphorus is sedimented by biological uptake and subsequent deposition (Mackereth 1966, Cole 1975, Engstrom and Wright 1984), sorption by humic complexes and iron oxides (Mackereth 1966, Engstrom and Wright 1984, Jones et al. 1993), precipitation as iron phosphates, and coprecipitation with carbonates (Engstrom and Wright 1984). Phosphorous is also coprecipitated with calcium (Cole 1975, Goldman and Horne 1983). Dissolved phosphorus is removed from the water so rapidly that generally the water is depleted and the sediment is enriched with phosphorus (Engstrom and Wright 1984).

Sedimenting organic phosphorus is either incorporated into the sediments or degraded in the hypolimnion or surface sediment by microbes and released as orthophosphate or soluble organic compounds (Cole 1975, Engstrom and Wright 1984).

The sedimentary phosphorus concentration depends on rate of supply, efficiency of precipitation mechanisms (biological and chemical), sedimentation rate, and rate of phosphorus loss from sediment. Biological precipitation of phosphorus, which operates only part of the year, is inefficient compared to the more continuous coprecipitation of phosphorus with iron and manganese. However, high efficiency of phosphorus sedimentation is achieved by the combined mechanisms of biological precipitation and coprecipitation with iron and manganese (Mackereth 1966).

The sediments provide an effective sink for phosphorus (Engstrom and Wright 1984). In oligotrophic lakes, where productivity and nutrients are low and hypolimnetic dissolved oxygen is high, most phosphorus is permanently buried in the sediments (Gachter and Meyer 1990). This is because phosphorus is efficiently retained in oxidized sediments (Engstrom and Wright 1984). However, when the sediment surface becomes anoxic, phosphorus is released leading to accelerated eutrophication (Mackereth 1966, Cole 1975, Wetzel 1983, Engstrom and Wright 1984, Gachter and Meyer 1990).

Phosphorus is fixed in the sediments primarily by sorption to hydrated ferric oxides or complexation with organics. Variations in iron content and redox influence phosphorus concentrations in the sediment (Engstrom and Wright 1984). When iron is lost from the sediment, phosphorus is also lost (Mackereth 1966). Therefore, changes in phosphorus retention may be responsible for sediment stratigraphy. Because of this, interpretation of past trophic state is possible only if redox remains constant over time. If redox is not constant, then sedimentary phosphorus concentrations may not reflect the actual phosphorus deposition. Although sedimentary phosphorus is generally uncorrelated with measures of lake productivity, several studies found good agreement between sedimentary phosphorus and historic trophic development (Engstrom and Wright 1984). High phosphorus indicated high organic material and high productivity (high TSI) in the deep water sediments of Florida lakes. Phosphorus abundance was unrelated to iron ($r^2=0.024$). Therefore, biogenic sedimentation was considered the main process for the phosphorus sedimentation (Flannery et al. 1982).

If iron and other sedimentological conditions remain constant, changing phosphorus levels may be preserved in the sediments. However, comparisons of sedimentary phosphorus trends with historical records of phosphorus loading is beneficial (Engstrom and Wright 1984).

### Sodium as an Indicator of Erosion

Sodium is the sixth most abundant element and is reactive and soluble. When it is leached from rocks, it generally stays in solution. Because of this, it is at least the third most abundant element in lakes and streams (Cole 1975). The main sources of sodium are halite, silicates, and natural ion exchange. The only common sink for sodium is reverse ion exchange which occurs only in highly saline waters such as brines (Hounslow 1993).

Sodium is conservative in aquatic systems. In most lakes, sodium concentrations are uniformally distributed throughout and experience only small seasonal fluctuations (Wetzel 1983). Sodium is also unaffected by redox conditions (Moore 1994).

Sodium provides a geochemical indicator of erosion (Anderson and Rippey 1988). Sodium is generally associated with the mineral fraction of the sediment (Mackereth 1966, Vuorinen et al. 1986). Variations in the sediment mineral content are primarily brought about by variations in erosion. Therefore, variations in sodium should represent erosion intensity, because when erosion occurs, unweathered minerals carrying sodium are transported to the lake and sedimented. Because these minerals and their components are rarely altered in aquatic systems, their distribution in sediments is useful for assessing weathering and erosion in the watershed. In contrast, dissolved sodium from areas which are not actively eroding is not appreciably sedimented (Engstrom and Wright 1984). Therefore, increasing sodium concentrations in sediment profiles indicate decreasing soil stabilization and increasing erosion (Sekar et al. 1992) and declining sodium concentrations indicate soil stabilization and decreased erosion (Hickman and Klarer 1981, Sekar et al. 1992). However, dilution of minerals in sediment by organic matter can be responsible for sodium trends instead of erosion (Engstrom and Wright 1984).

### Cesium-137 Dating

Cesium-137 (Cs-137) has been present in the atmosphere since 1954 due to atomic bomb testing (Engstrom and Wright 1984). Between 1958 and 1967, nuclear testing created significant quantities of cesium-137 which was deposited as fallout. The annual deposition of cesium-137 follows the frequency of the nuclear testing. The maximum deposition of cesium-137 occurred in the northern hemisphere between 30° and 60° latitude. The peak fallout rate occurred between 1962 and 1964. Cesium-137 was distributed throughout the northern hemisphere after atmospheric testing and is now used to estimate the age of sediments deposited since 1958 within plus or minus 2 years (Vuorinen et al. 1986).

The method assumes that cesium-137 fallout with rain becomes attached to particles which are quickly (< 1 year) transported from the drainage basin to lake sediments. Cesium-137 falling directly upon the lake surface is adsorbed onto suspended particulate matter and sedimented. It is also assumed that disturbance of the sediment stratigraphy is small. In some cases, disturbance by water movements and redistribution of sediments by benthic macroinvertebrates can obscure dating chronology. However, careful interpretation of cesium-137 dating results, combined with other paleolimnological data, provides much insight into the reconstruction of recent lake events (Wetzel 1983).

## Paleolimnological Methods

The samples were taken in 21.6 m of water. High chlorophyll density has been observed in this area (discussed later). Eutrophication is pronounced at Station 5 as indicated by high chlorophyll *a* levels and is located in the transition zone. It correlates with a preexisting sampling station; therefore, historical data can be referenced.

### Review of Historical Data

Historical data from STORET and USGS were reviewed to determine historical trends in metal and nutrient supply to Lake Tenkiller. The National Eutrophication Survey and other limnological studies on Lake Tenkiller were also reviewed to determine the historic trophic state and hypolimnetic oxygen conditions of the lake.

### Limnological Methods

At Station 5, dissolved oxygen, temperature, conductivity, and pH, were measured *in situ*. Dissolved oxygen and temperature profiles were determined using a YSI Dissolved Oxygen/Temperature meter. Conductivity profiles were determined using a YSI Combination Salinity-Conductivity-Temperature meter. The pH was determined using an Orion pH meter in samples collected at 0.5 m from the surface and bottom using a van Dorn sampler.

### Metals Analyses (Water)

Metals were analyzed four times in the water on 4 Jun 1992 and 18 Apr, 26 May, and 19 Aug 1993. The samples (100 ml) were collected at 0.5 m from the surface and 0.5 m from the bottom using a van Dorn sampler. The water samples were then acidified with 3 ml of concentrated nitric acid, placed on ice, and returned to the lab. In the lab, the water samples were digested and then refluxed with concentrated nitric acid. Finally, the samples were decanted into a 100 ml graduated flask and diluted to 100 ml with Type II water (EPA 1987). Metals were analyzed with a Perkin-Elmer 5000 Atomic Absorption Spectrophotometer using the settings listed in Table XXXIV. Metals were reported as total metals (mg/l).

**Table XXXIV** Atomic Absorption Spectrophotometer Settings for Analysis of Metals from Lake Tenkiller Water.

| Metal | Wavelength | Slit Width | FAA or HGA |
|---|---|---|---|
| Iron | 248.3 | 0.2 High | FAA |
| Manganese | 279.5 | 0.2 Low | HGA |
| Copper | 324.8 | 0.7 Low | HGA |
| Zinc | 213.9 | 0.7 High | FAA |
| Calcium | 422.7 | 0.7 High | FAA |

Sediment Collection and Preservation

Sampling was completed on 18 October 1993. Sediment cores were collected using a Ballchek sediment corer holding liner tubes two inches in diameter and thirty inches long. The liner tubes containing the sediment were removed from the corer after each sampling and were capped on both ends. The capped liner tubes were stored in an upright position and kept cool until returned to the lab. Upon arrival at the lab, the liner tubes containing the sediments were drained of excess water and frozen until analysis. Before analysis the sediment cores were sectioned while frozen. The top 10 cm of the sediment core was cut into 1 cm sections, and the rest was cut into 2.5 cm sections. Each section was then placed in a labeled glass sample bottle for storage until analysis. Each section was allowed to melt and then was mixed thoroughly before subsamples were taken for the individual digestions and analysis.

Metals Analyses (Sediment)

The tare weights of beakers were obtained by drying and weighing until two consistent weights were recorded. Three 1 to 2 g (wet weight) subsamples were then taken from each section of the sediment core and placed into the tared beakers. The samples were then dried until two consistent weights were recorded. The dry weight of the sample was determined by subtracting the weight of the tared beaker from the weight of the dried sample and beaker.

Sediment samples were prepared for metals analyses using the digestion procedure described in EPA Method 3050. The sediment samples were digested in nitric acid and hydrogen peroxide. The digestates were then refluxed with hydrochloric acid. Finally, the samples were decanted into a 100 ml graduated flask and diluted to 100 ml with Type II water (EPA 1986). Metals were analyzed with a Perkin-Elmer 5000 Atomic Absorption Spectrophotometer using the settings listed in Table XXXV. Metals were reported as mg/g of dried sediment.

**Table XXXV** Atomic Absorption Spectrophotometer Settings for Analyses of Metals from Lake Tenkiller Sediment.

| Metal | Wavelength | Slit Width | FAA or HGA |
|---|---|---|---|
| Iron | 248.3 | 0.2 High | FAA |
| Manganese | 279.5 | 0.2 High | FAA |
| Copper | 324.8 | 0.7 Low | HGA |
| Molybdenum | 313.3 | 0.7 Low | HGA |
| Zinc | 213.9 | 0.7 High | FAA |
| Calcium | 422.7 | 0.7 High | FAA |
| Sodium | 589.0 | 0.4 High | FAA |

Sedimentary Phosphorus Analysis

Sediment cores were mixed thoroughly after they were sectioned and thawed. An aliquot of mixed sediment was taken from each section and dried. After the sediment was dried, duplicate subsamples weighing between 0.0123 and 0.03924 g were taken from each section and placed in 250 ml Erlemeyer flasks. To each subsample, 50 ml of water and 8 ml of 5% potassium persulfate were added. The samples were then capped with aluminum foil and digested in an autoclave at 15 psi for 30 min at 121°C. Then, 8 ml of mixed reagent were added to each digested sample. After 10 min, the absorbance of each sample was measured at 880 and 690 nm in a spectrophotometer. The phosphorus concentrations (mg/l) were then determined from a standard phosphorus calibration curve (Lind 1985). These concentrations (mg/l) were then converted to mg/g of oven dried sediment.

Sediment Dating

Sediment cores were also collected by the USDA on 18 October 1993 for use in dating of the sediments. The sediment cores were dated in conjunction with the USDA lab in Durant, Oklahoma using the cesium-137 dating method. The sediment cores were divided into 5 cm sections beginning at the sediment surface. Core sections of like depth were composited and placed into labeled plastic bags for transporting to the laboratory. In the laboratory, sediment samples were oven dried at 105°C and then ground with a mortar and pestle until they passed a 6 mm screen. Samples were then analyzed for cesium-137 with a multichannel analyzer using a lithium-drifted germanium detector (McHenry et al. 1980). Cesium-137 activity from each sample was counted twice and then averaged. Dating the sediment was possible because cesium-137 produced identifiable markers when it first became detectable in 1954 from atmospheric nuclear tests and peaked in activity in 1963 (Larsen 1984).

Accuracy was assessed by using EPA reference standards and percent recoveries. A universal blank matrix does not exist for solid samples, therefore, no matrix blank was used (EPA 1987). A duplicate sample from each batch of samples was spiked with a predetermined quantity of stock solution prior to digestion and was run with each batch of samples. Percent recoveries, calculated from the spikes, were used to determine accuracy and matrix effects. Precision was assessed by analyzing triplicate samples. The standard deviations of the replicates were used to determine precision (Horowitz 1991). Standard curves were used to determine the concentrations of the samples. The standard curves were verified using an initial calibration verification standard (an EPA reference standard) before analysis of any samples. After initial verification, one EPA reference standard was run for every ten samples and if results were beyond the EPA predetermined acceptable limits, corrective action was taken. All important dates, such as collection, digestion, and analysis dates, were documented. Holding times were no longer than 6 months after digestion (EPA 1987).

### Data Analysis

For sedimentary metals analysis, the absorbance of the blank was first subtracted from the absorbance of the sample. Then the corrected absorbance was multiplied by the slope of the regression line after which the y-intercept was added. This gave the concentration in mg/l. This concentration was then multiplied by 0.1 l and then divided by the dry weight of the sample to give the concentration in mg/g.

Sedimentary depth profiles were then established. These profiles were correlated with the dated sediment profiles and the data were analyzed to determine paleolimnological significance. Large shifts in the chemical concentrations of each element's profile was interpretted to determine its paleolimnological significance. Correlation coefficients also were determined between each sedimentary trace element profile. Trends were determined by performing a regression of depth and concentration. The resulting regressions were plotted.

## Paleolimnological Results and Discussion

### Historical Conditions of the Basin

The combined mean daily discharge from the Illinois River (USGS 1965) and Baron Fork (USGS Station 07197000) averaged 1257 cubic feet per second (cfs) (standard deviation 695 cfs). Above average flows (Figure 20) occurred from 1973-74, 1985-86, and 1990. Below average flows occurred from 1963-64, 1967, 1977, and 1980-81 (USGS 1965-1993).

Nutrient levels in the Illinois River have increased with time. This increase in nutrient loading to Lake Tenkiller has resulted in an increasing rate of eutrophication. In addition, Burks et al. (1991) reported that an upward trend in hydraulic discharge also had occurred, presumable due to the loss of Lake Frances. However, the increased nutrient load is based upon the increased concentration on nutrients coupled with the increased water influx.

This area of Oklahoma has seen a major growth in agriculture. The number of chickens raised in the Illinois River Basin (Figure 21) peaked in the late 1960s and has increased since the mid-1970s. The number of broilers in the basin (Figure 21) peaked in the mid-1960s and has experienced phenomenal growth since the early-1980s.

The number of hogs and pigs (Figure 22) being raised in the Illinois River Basin has increased continuously since the mid-1970s. These growing industries have surely had an impact on Lake Tenkiller due to the additional nutrient sources present in the basin.

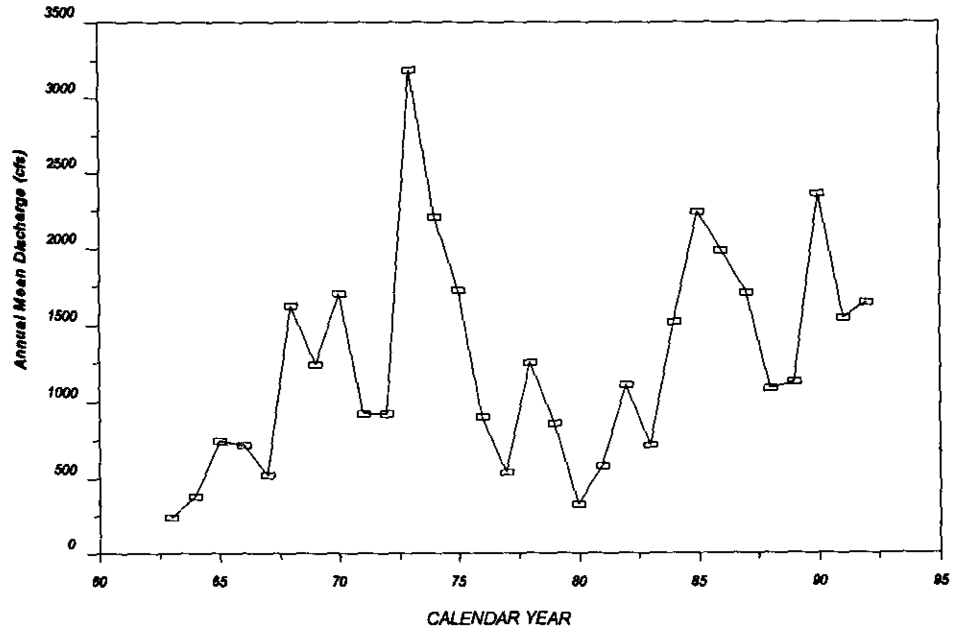



**Figure 20** Annual Mean Discharge from 1963-92 Based on the Combined Daily Discharge of the Baron Fork (07197000) and the Illinois River (07196500).

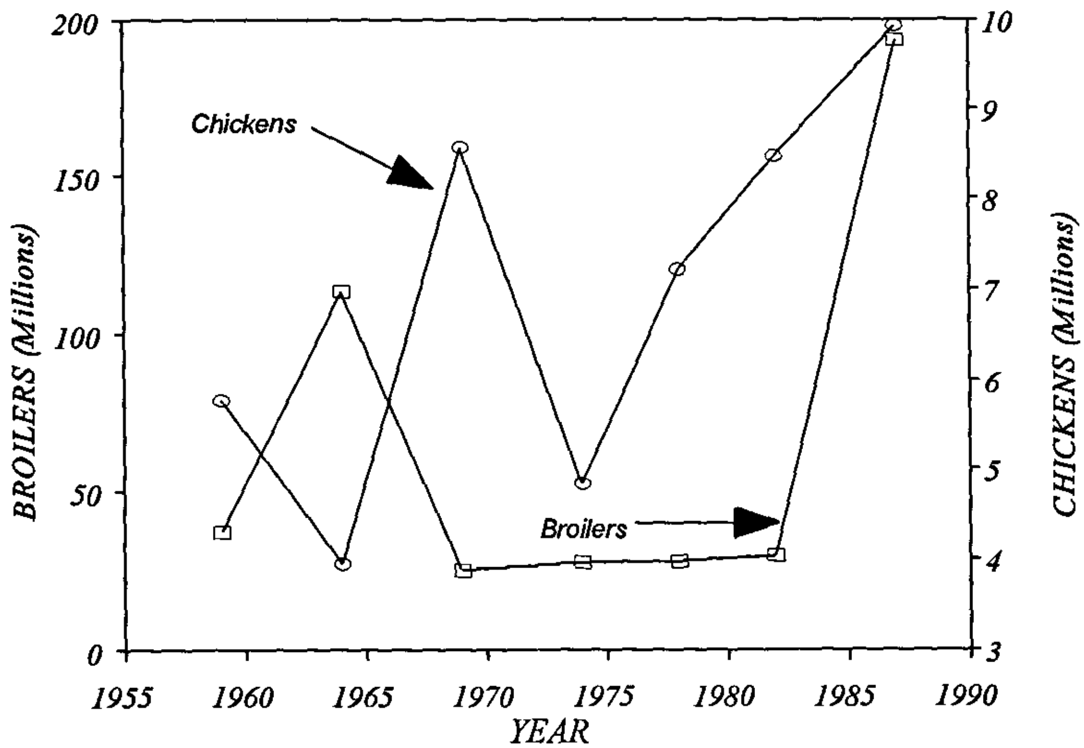



**Figure 21** Number of Broilers and Three Month Old or Older Chickens in the Illinois River Basin.

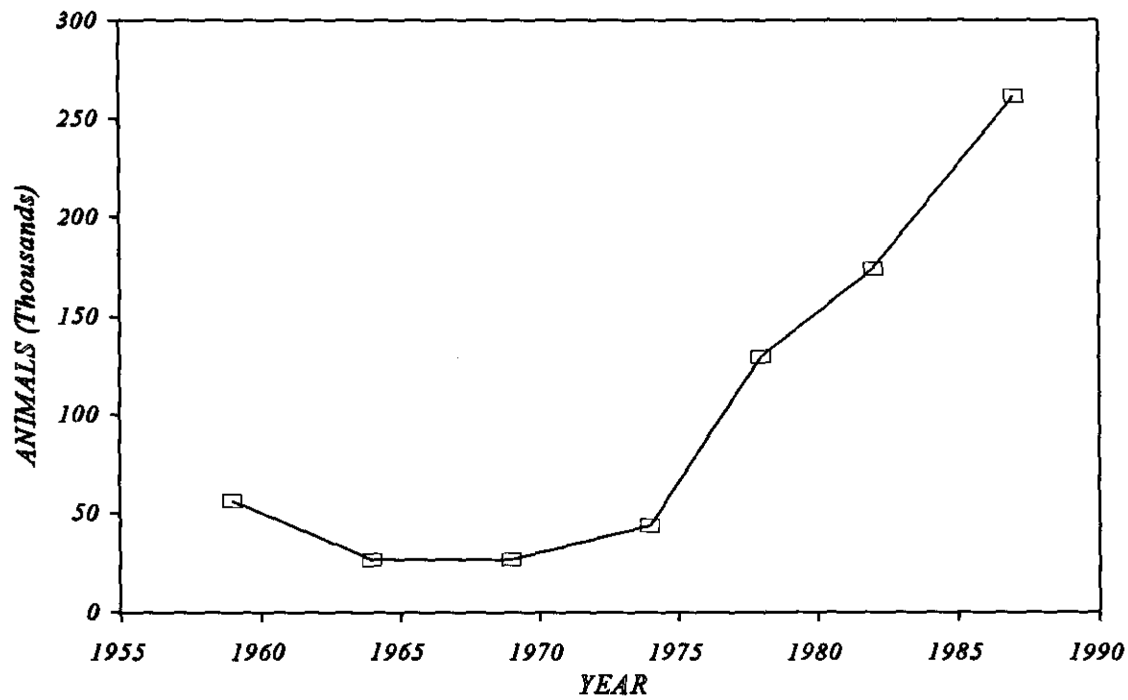



**Figure 22** Number of Hogs and Pigs in the Illinois River Basin.

## Historical Conditions of the Lake

Numerous studies have been conducted on Lake Tenkiller since its impoundment began in 1952. Jenkins (1953) studied fish of Lake Tenkiller during its first year of impoundment. He measured temperature and dissolved oxygen profiles. A thermocline was present at the dam at a depth of 6 to 12 m throughout the summer. Dissolved oxygen was negligible below the thermocline. Dissolved oxygen ranged from 8.1 mg/l at the surface to zero at the bottom. Dissolved oxygen was generally less than 1.5 mg/l below 11 m. The thermocline at Standing Rock Bridge (near the Phase I study's station 4) was present at a depth of 6 to 9 m (Jenkins 1953). Finnell (1953) also studied Lake Tenkiller from 16 Jun to 22 Jul 1953 to determine if its tailwaters provided suitable temperatures and dissolved oxygen concentrations to sustain a rainbow trout fishery. Dissolved oxygen levels at the dam were generally less than 1.5 mg/l below 11 m and was zero at the bottom. The dissolved oxygen concentration at the bottom of Station II (near Phase I study's station 4) was 1.2 mg/l (Finnell 1953). However, the cause of hypolimnetic anoxia was not determined. Hypolimnetic anoxia often occurs in newly constructed reservoirs due to the decomposition of the terrestrial biomass left in the lake. This could have resulted in the observed clinograde oxygen profile.

From Feb 1960 to Jan 1961, the Oklahoma Department of Wildlife Conservation (ODWC) conducted a limnological study on Lake Tenkiller to determine the success of walleye stocking proposed for the spring of 1961. The lake was found to be thermally stratified during summer. During stratification, average hypolimnetic dissolved oxygen concentrations ranged from zero to 3.5 mg/l. The ODWC did not determine the trophic state (Summers 1961).

In 1974, four sites on Lake Tenkiller were sampled quarterly as a part of EPA's National Eutrophication Survey (NES). On 30 Aug 1974 the hypolimnetic dissolved oxygen concentrations at 3 stations were zero. Based on the data collected, the NES classified Lake Tenkiller as eutrophic (EPA 1978). Trophic state in 1974 ranged from eutrophic (TSI=56.8) in the upper reaches of the lake (near Phase I Station 3) to mesotrophic (TSI=46.8) near the dam (Phase I Station 7) according to Carlson's TSI-chlorophyll a (Carlson 1977). Station 5 was also eutrophic. However, due to the small sample size, much uncertainty exists.

The Oklahoma State Department of Health (OSDH) studied Lake Tenkiller from Jun to Nov 1975. The lake was thermally stratified from June to Oct between 5-7 m in the headwaters of the lake, and between 11-16 m in the lower regions. Dissolved oxygen in the hypolimnion were generally less than 3 mg/l. At one station , the dissolved oxygen concentration was near zero at the bottom on 18 Jun 1975. The lake was classified as mesotrophic (OSDH 1977). However, based on surface phosphorus concentrations the trophic status of the lake ranged from eutrophic near Phase I Station 3 to mesotrophic near the dam (Carlson 1977). Since the sample size was small, ranging from only 3 to 5 samples, there is much uncertainity in this classification.

Morris (1979) investigated the temperature and dissolved oxygen near the dam of Lake Tenkiller on 9, 15, 28 Aug and 5 Sep 1979. Lake Tenkiller was again thermally stratified. Dissolved oxygen concentrations in the hypolimnion were less than 1 mg/l during the entire study (Morris 1979).

From Oct 1985 to Nov 1986, the U.S. Army Corps of Engineers studied the limnology of Lake Tenkiller. The lake was stratified from May 1986 through late Sep. Lake Tenkiller exhibited a clinograde dissolved oxygen profile, and dissolved oxygen was generally depleted below 10 m. Based on TSI-chlorophyll a (Carlson 1977), Lake Tenkiller was classified as eutrophic. Total manganese concentrations in the hypolimnion were approximately twice those found in the epilimnion. However, total iron was greatest in epilimnetic samples (Nolen et al. 1988). Harton (1989) found that nonpoint source phosphorus loading was the cause of the eutrophication in Lake Tenkiller.

In conclusion, the studies showed that the hypolimnion of Lake Tenkiller experienced dissolved oxygen depletions in 1953, 1961, 1974, 1975, 1979, 1985, and 1986. It can be assumed that the hypolimnion has experienced dissolved oxygen depletion during thermal stratification since impoundment. The studies also showed that Lake Tenkiller has experienced eutrophic conditions in its headwaters since the mid-1970s. While the results may appear tenuous due to a small sample size, these conclusions are consistent with with many reservoir studies. Chlorophyll a concentrations had at least doubled between 1975 and 1985. In addition, manganese was found to be accumulating in the hypolimnion in 1986, indicating its release from the anoxic sediments or dissolution from particulate matter falling through the anoxic hypolimnion. Nonpoint source phosphorus loading was found to be the cause of the eutrophication of Lake Tenkiller.

## Limnological Results

During the Phase I study (1992-93), Station 5 was thermally stratified from mid-June to late-September in 1992, and from Jul to Sep 1993 (Figure 17). During thermal stratification, a clinograde dissolved oxygen profile was exhibited. In 1992, hypolimnetic dissolved oxygen (DO) concentrations were generally less than 2 mg/l from July through September (Figure 17). In 1993, hypolimnetic dissolved oxygen concentrations were generally less than 1 mg/l from late June to mid-August.

## Metals in the Water Column

Iron and manganese concentrations, which were measured four times in the water, were consistently higher in the hypolimnion than in the epilimnion. All measurements of iron and manganese were completed between April and August. The increase of hypolimnetic manganese (Figure 23) as seen in Lake Tenkiller shows that manganese is being released from sediments and/or dissolved from particulate matter falling through the hypolimnion. The increased iron concentrations in the hypolimnion (Figure 24) indicates that it is being released from the sediment. Sodium and molybdenum were not measured in the water. Copper and zinc concentrations were below detection limit in all water samples from station 5.

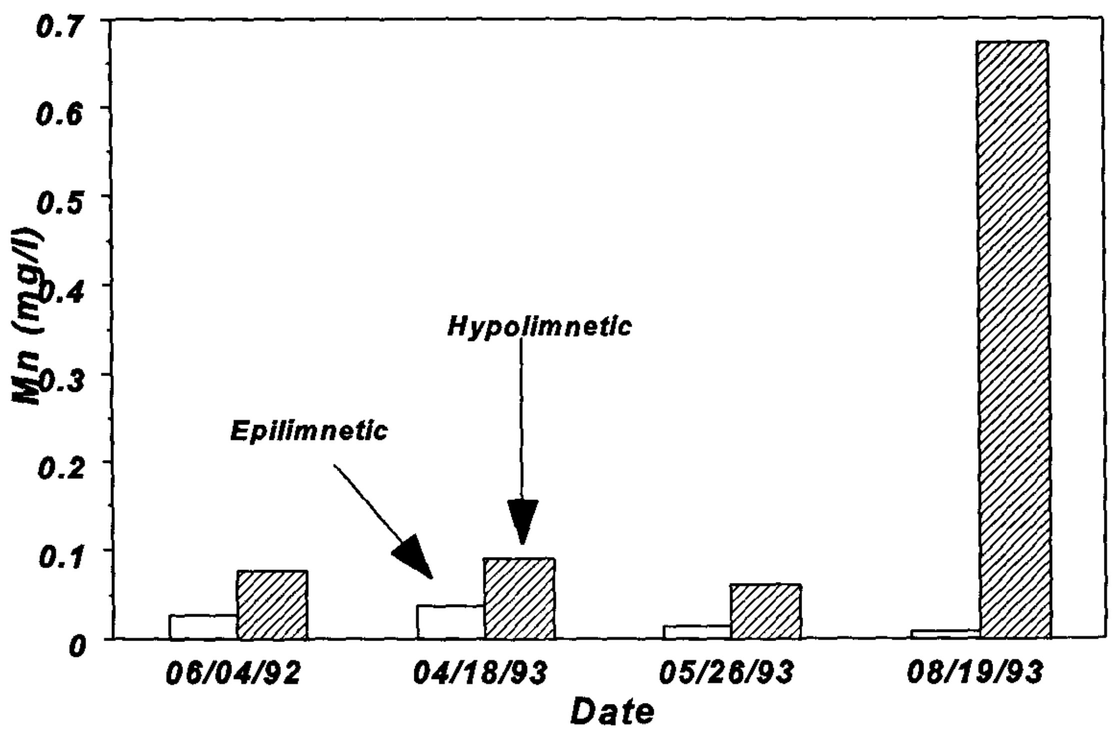



**Figure 23** Epilimnetic and Hypolimnetic Manganese Concentrations Measured at Station 5 in Lake Tenkiller, 1992-93.

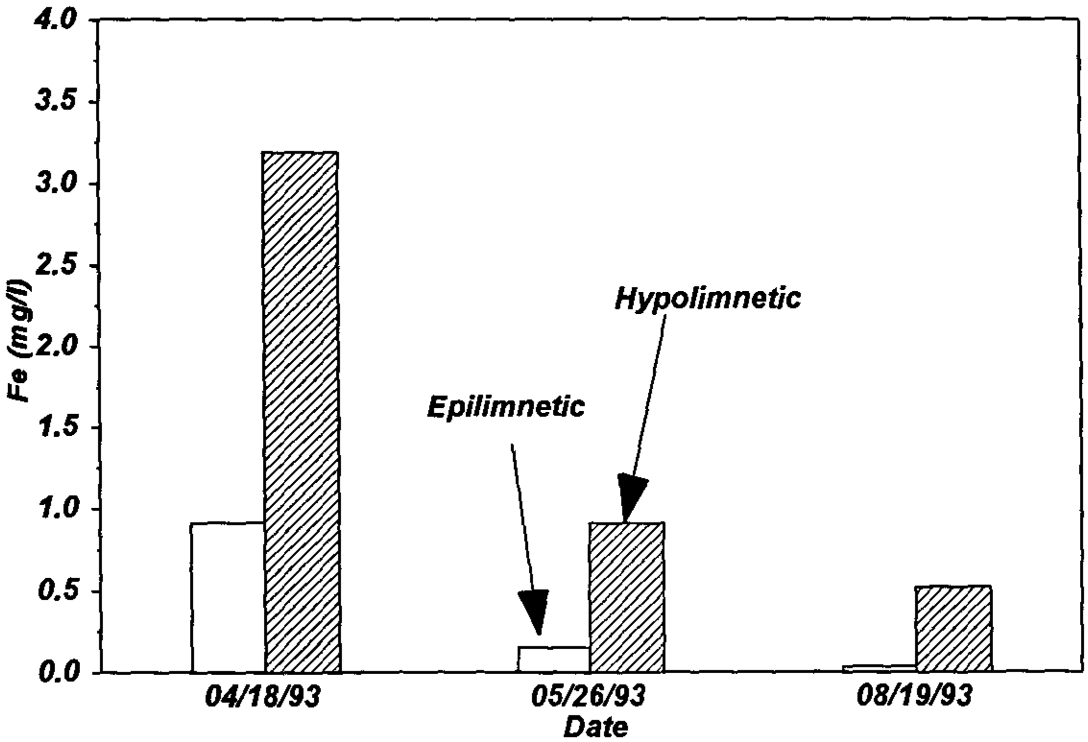



**Figure 24** Epilimnetic and Hypolimnetic Iron Concentrations Measured at Station 5 in Lake Tenkiller, 1992-93.

Sediment Dating

The cesium-137 dating indicated the sedimentation rate at Station 5 was approximately 1.5 cm/year from 1954 to 1963 and 1.9 cm/year from 1964 to 1993. The depth of 55 cm corresponds with the year 1964 (Figure 26). Sedimentation rate has remained low throughout the lake's history increasing only slightly in the thalweg.

Sedimentary Sodium

Sedimentary sodium concentrations ranged from 0.0675 to 0.1348 mg/g. There was little variation of sodium concentration with depth (Figure 27) indicating that the watershed soils were stable and little erosion had occurred. This agreed with the findings of the cesium-137 dating, which showed that sedimentation rates were low. However, peaks occurred at 1, 22.5, and 32.5 cm. The peak at 1 cm likely resulted from the above average flow of the Illinois River during 1993, while the peak at 32.5 cm likely resulted from the high flow years of 1973-74. With the exception of the peaks at 1, 22.5 and 32.5 cm, sodium indicated a stable erosion rate. Because the sedimentary sodium concentration-depth profile indicated a stable erosion rate, increasing and decreasing metal concentrations probably resulted from processes other than erosion.

Sedimentary Iron and Manganese

Lake Tenkiller has experienced seasonal anoxia since impoundment as evidenced by previous studies. Therefore, the sedimentary iron and manganese concentrations in Lake Tenkiller were controlled by changes in supply, variations in the hypolimnetic redox, and post-depositional migration.

The iron (Figure 28) and Fe:Mn (Figure 25) profiles were not positively correlated ($r^2=0.000321$) indicating that the iron concentrations did not result from changes in the soil redox (Engstrom and Wright 1984). In addition, the Fe:Mn data were not correlated with sodium ($r^2=0.008$) indicating that the iron and manganese concentrations did not result from erosion (Mackereth 1966). The high Fe:Mn (mean=25.3) indicated that oxidizing conditions prevailed in watershed soils (Engstrom and Wright 1984). Iron maxima (with the exception of the maxima at 25 cm) generally did not correspond with Fe:Mn maxima indicating that iron concentrations in the sediment were not produced by varying rates of supply from the watershed (Mackereth 1966). No major peaks in the manganese profile, with the exception of the peak at 1 cm, corresponded with high flow years indicating that sedimentary manganese concentrations were not produced by varying rates of supply. All data indicated that the iron and manganese concentrations in Lake Tenkiller sediment did not result from changes in supply from the watershed. Iron maxima did not correspond with high flow years, with the exception of peaks at 1 and 6 cm, indicating that supply was not primarily responsible for the observed sedimentary iron profile. Therefore, changing iron and manganese concentrations in the sediment probably resulted from either post-depositional migration or redox changes from hypolimnetic anoxia as driven by organic input from epilimnetic primary productivity.

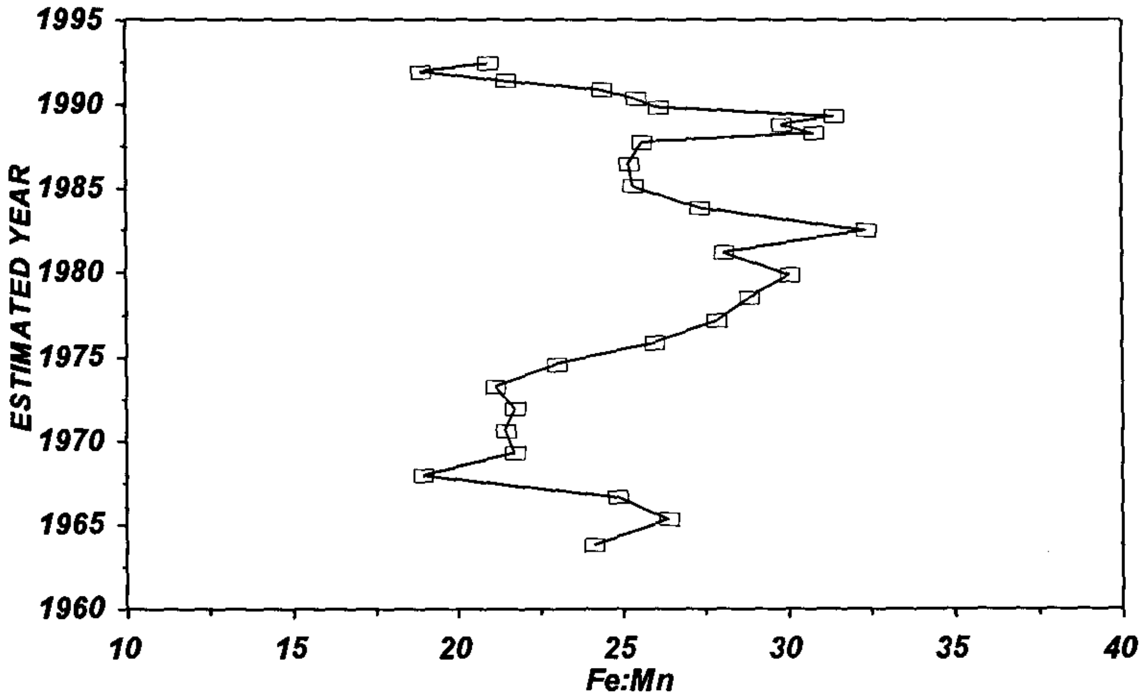



**Figure 25** Fe:Mn Profile Calculated from Fe and Mn Concentrations Measured in Lake Tenkiller Sediment Collected from Station 5 in October 1993.

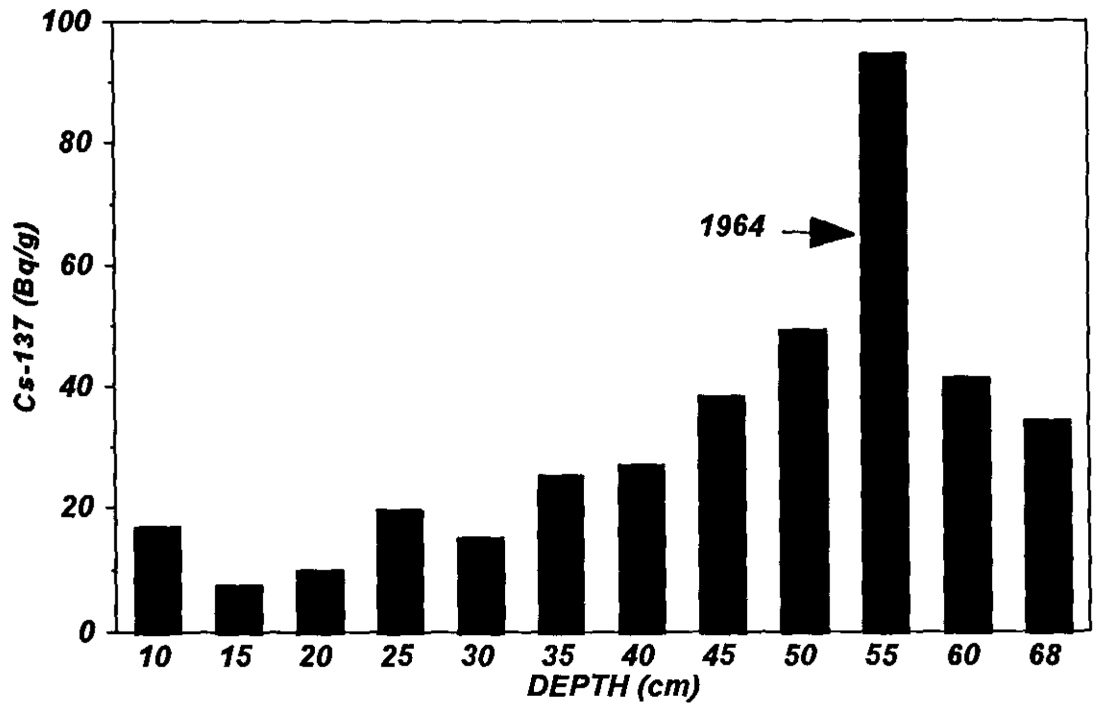



**Figure 26** Cesium-137 Activity Measured in Lake Tenkiller Sediment Collected from Station 5 in October 1993

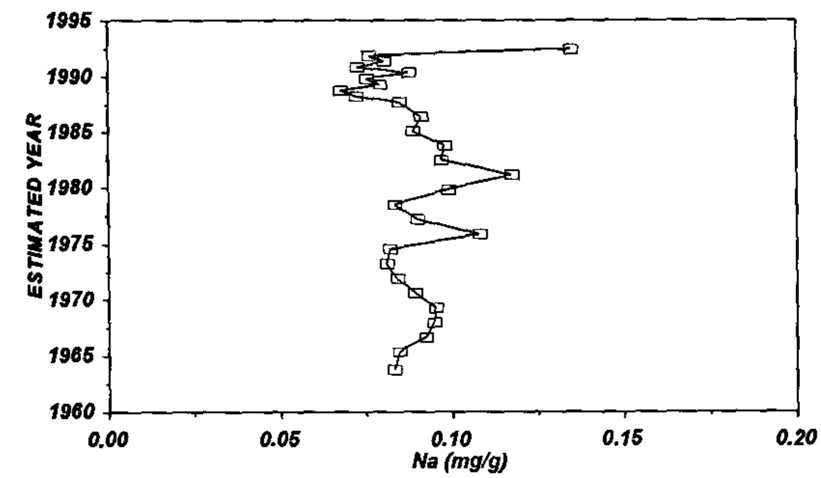



**Figure 27** Concentration-depth Profile of Sodium Measured in Lake Tenkiller Sediment Collected from Station 5 in October 1993.

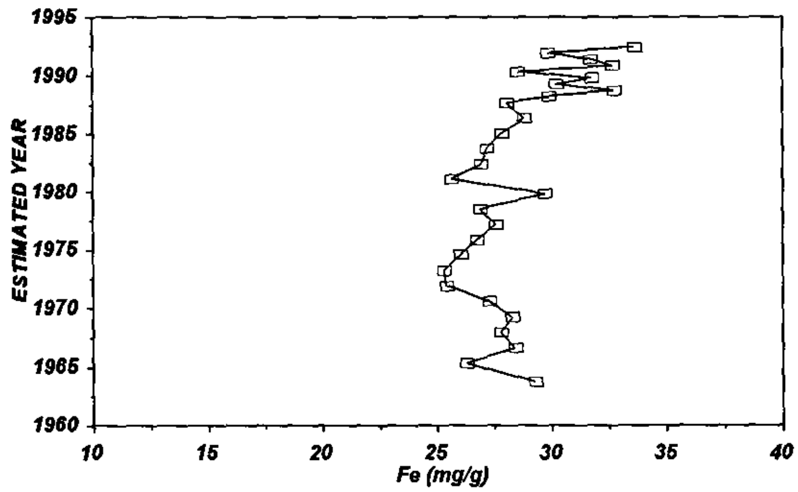



**Figure 28** Concentration-depth Profile of Iron Measured in Lake Tenkiller Sediment Collected from Station 5 in October 1993

Post-depositional migration of iron and manganese below the sediment redox cline as described by Engstrom and Wright (1984) can result in poor correlation between the Fe:Mn and other paleoredox indicators. Both iron and manganese concentrations increased near the sediment surface. The sedimentary manganese concentration (Figure 29) steadily increased from 5 cm to the sediment surface indicating that post-depositional migration of manganese had likely occurred. This resulted in the observed decreasing Fe:Mn from 5 cm to the surface of the sediment core. Iron did not exhibit the same increase indicating that post-depositional migration was insignificant.

The Fe:Mn, which ranged from 18.92 to 31.42, was not consistent throughout the sedimentary profile indicating that redox conditions have not remained constant (Mackereth 1966). Because Lake Tenkiller discharges from its hypolimnion, release of metals from the sediments should result in an obvious decrease in the sediment concentration-depth profile. However, this was not the case in the iron and manganese concentration-depth profiles from Lake Tenkiller. The manganese concentration decreased 0.64 mg/g from 47.5 cm (1968) to 20 cm, after which it increased 0.78 mg/g from 20 cm to the sediment surface. The decreasing manganese concentrations from 47.5 cm to 20 cm would be expected as redox conditions become more reducing in a lake experiencing eutrophication. However, the increasing manganese concentrations from 20 cm to the sediment surface is not typical of a lake undergoing eutrophication. The iron concentrations ranged from 25.31 to 33.63 mg/g at the sediment surface. The highest iron concentrations occurred in the top 10 cm of the sediment core. The iron concentration peaks at 1 and 6 cm are likely caused by the recent flood events of 1990 and 1993, respectively. The iron peaks at 4 and 8 cm could have resulted from post-depositional migration and deposition of iron during extremely anaerobic periods as FeS. There was some evidence that hypolimnetic redox conditions control the iron and manganese concentrations. The Fe:Mn maxima at 7 and 52.5 cm corresponded with iron minima indicating hypolimnetic redox changes (Engstrom and Wright 1984).

The validity of the Fe:Mn and manganese as paleoredox indicators is questionable. The manganese peak at the surface of the sediment core showed that post-depositional migration had occurred which could change the original concentrations of manganese lain down in the sediment. The recent study by Hsuing and Tisue (1994) which indicated that more manganese is released during oxic than anoxic periods adds further doubt and uncertainty to its usage. They also concluded that anoxia may inhibit the release of reduced manganese from sediments (Hsuing and Tisue 1994). However, this is unlikely in Lake Tenkiller as evidenced by the differences between the epilimnetic and hypolimnetic manganese concentrations. In Lake Tenkiller, the hypolimnetic manganese concentration was much greater than the epilimnetic manganese concentration during stratification indicating that manganese was being released from sediments or from particulate matter in the hypolimnion. Because manganese can be rapidly reduced even when the hypolimnion is oxic (Davison et al. 1982) and is released under both oxic and anoxic conditions, it should not be used as a paleoredox indicator nor an indicator of paleo-productivity. Because manganese is not a valid paleoredox indicator, the Fe:Mn ratio is also invalid. Due to new developments and based on recent investigations, manganese and the Fe:Mn should not be used as paleolimnological indicators.

Iron alone may provide a good indication of paleoredox conditions and past productivity. It is released from sediments when sediments become moderately reducing and is not easily reduced in aerobic environments. In eutrophic lakes, FeS accumulates when extremely reducing conditions occur. However, it is difficult to determine if increasing iron is indicating increasing hypolimnetic oxygen (decreasing productivity) or lower redox conditions (increasing productivity) which resulted in FeS deposition. In addition, Davison and others (1982) determined that the recycling of iron was generally small when compared with other iron fluxes. The trend of increasing iron concentrations from 40 cm to the sediment surface may indicate the final stages of anoxia when FeS was precipitated (Engstrom and Wright 1984). If this is the case, then primary productivity reached a rate of high productivity in the mid-1970s causing the lower redox and subsequent FeS deposition.

Because many trace elements coprecipitate with iron and manganese, their concentration-depth profiles could correlate with those of iron and manganese. Iron was most correlated with molybdenum ($r^2$=0.49), calcium ($r^2$=0.37), phosphorus ($r^2$=0.36)and manganese ($r^2$=0.23) profiles. The correlation of molybdenum with iron is due to the fact that the molybdenum cycle in freshwater is similar to iron's (Cole 1975). The correlation of phosphorus and manganese with iron is due to their coprecipitation with iron (Mackereth 1966; Engstrom and Wright 1984; Jones et al. 1993). Calcium's correlation with iron is likely due to increasing primary productivity which resulted in greater precipitation of calcium and increased sedimentary retention of iron as FeS. Iron was not correlated with the Fe:Mn, sodium, Cu/Zn, zinc, (Cu+Mo)/Zn, or copper. Manganese was highly correlated with the Fe:Mn ($r^2$=0.73) indicating that the manganese concentration controlled the Fe:Mn ratio. The manganese profile was also somewhat correlated with the iron profile ($r^2$=0.23). This correlation is likely due to both elements being redox sensitive elements. The manganese profile was not correlated with calcium, copper, molybdenum, sodium, zinc, (Cu+Mo)/Zn, Cu/Zn, and phosphorus.

Cu/Zn and (Cu+Mo)/Zn as Paleoredox Indices

Copper concentrations ranged only from 0.0120 to 0.0178 mg/g and exhibited a slight upward trend. The copper profile (Figure 30) fluctuated repeatedly. The copper peaks at 1, 2, 5, 9-10, and 35 cm likely correspond to floods during 1993, 1990, 1985-86, and 1973-74. The copper peak at 20 cm corresponds to a peak in the calcium profile. The peak at 25 cm corresponds to peaks in the phosphorus and iron profiles.

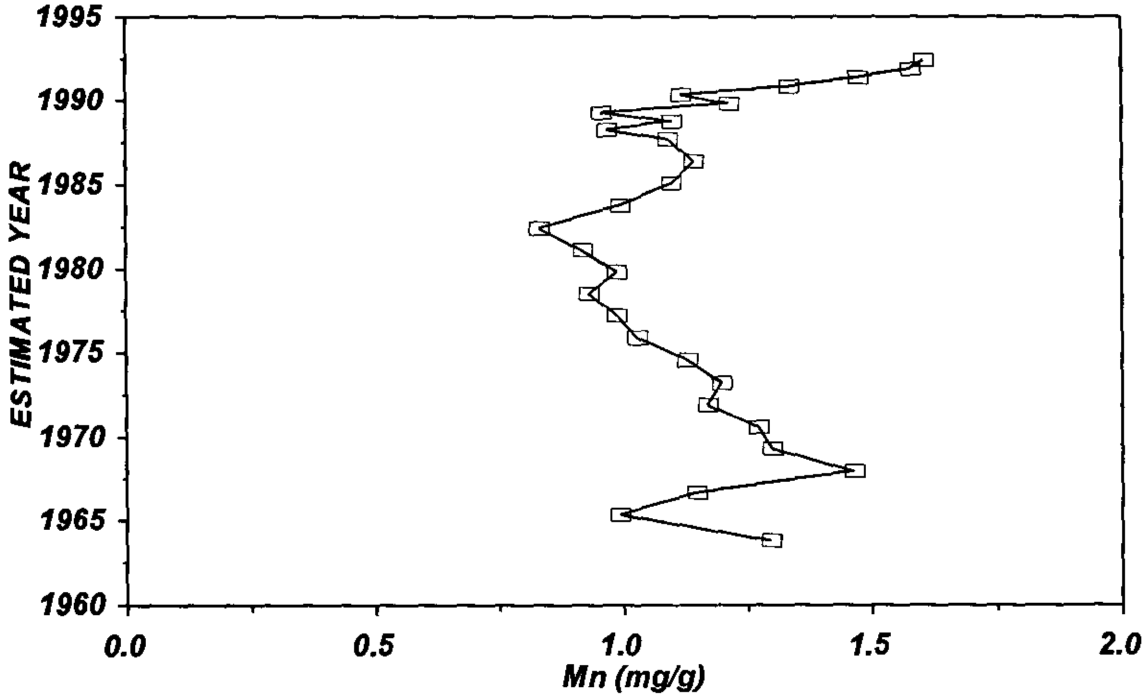



**Figure 29** Concentration-depth Profile of Manganese Measured in Lake Tenkiller Sediment Collected from Station 5 in October 1993.

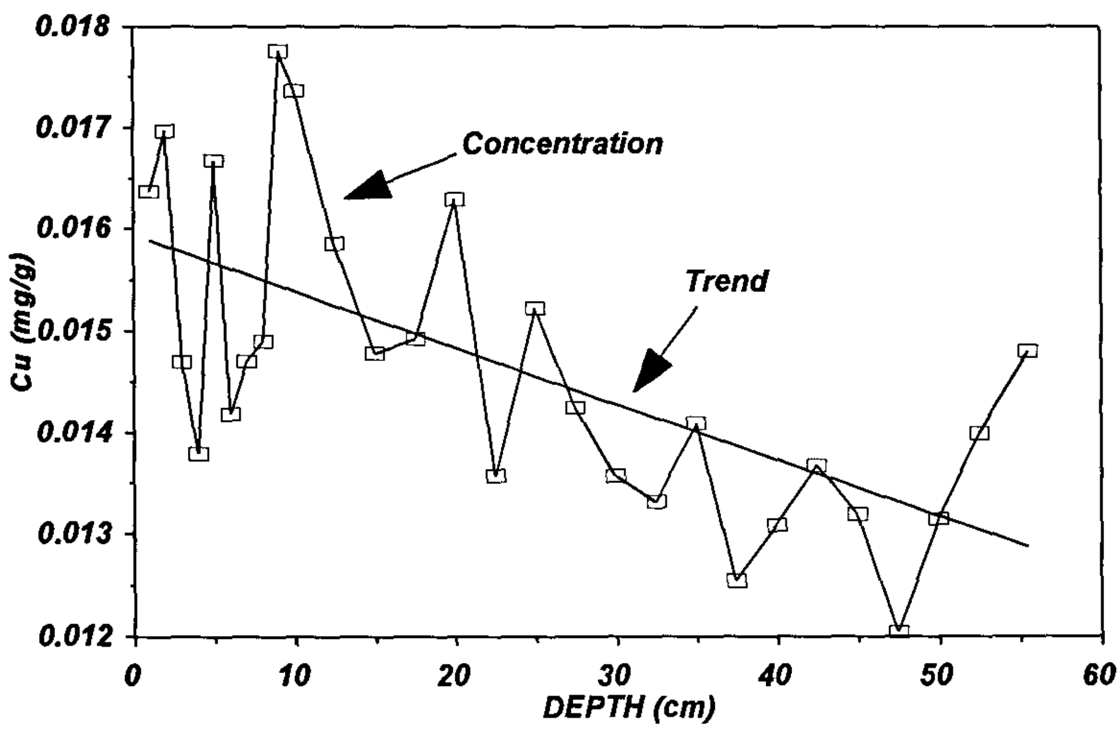



**Figure 30** Concentration-depth Profile of Copper Measured in Lake Tenkiller Sediment Collected from Station 5 in October 1993.

Copper was correlated with the Cu/Zn ($r^2$=0.81), (Cu+Mo)/Zn ($r^2$=0.67), and calcium ($r^2$=0.51) profiles. The high correlation of copper with both the Cu/Zn and (Cu+Mo)/Zn profiles indicates that the copper concentration controls these indices. The correlation between calcium and copper is likely caused by both being associated with primary productivity. Copper uptake and subsequent deposition by algae is responsible for most sedimentation of copper (Hutchinson 1957). Calcium precipitates as productivity increases due to the uptake of carbon dioxide (Goldman and Horne 1983). Copper was also somewhat correlated with phosphorus and molybdenum. Copper was not correlated with the iron, manganese, sodium, zinc, and Fe:Mn profiles. Because sodium and copper are not correlated, it can be assumed that changes in copper profile were not primarily due to erosion.

Zinc concentrations varied little and ranged from 0.0901 to 0.115 mg/g. Significant peaks occur at depths of 3, 5, 12.5, 27.5, and 32.5 cm in the zinc profile (Figure 31). The zinc peaks at 5, 12.5 and 32.5 cm likely correspond with the flood events during the years 1990, 1985-86, and 1973-74, respectively. Although post-depositional migration of zinc has not been suggested in previous studies, the maxima at 3 cm could potentially be related to post-depositional migration. The zinc profile was not correlated with any of the parameters measured.

Molybdenum concentrations ranged from 0.0006 to 0.0170 mg/g. Little variation occurred in the molybdenum profile (Figure 32) with the exception of the peak at 8-9 cm. This peak likely resulted from flooding during the mid-1980s. Molybdenum was correlated with the iron ($r^2$=0.49), (Cu+Mo)/Zn ($r^2$=0.42), phosphorus ($r^2$=0.31), calcium ($r^2$=0.30), Cu/Zn ($r^2$=0.22), and copper ($r^2$=0.21) profiles. The correlation of molybdenum and iron were discussed in the previous section. The correlation between molybdenum and (Cu+Mo)/Zn indicates the influence of molybdenum on the paleoredox index. The correlation between molybdenum and phosphorus likely due to molybdenum being released from the sediments when phosphorus is released (Cole 1975). Molybdenum was not correlated with the manganese, sodium, and zinc profiles.

The Cu/Zn and (Cu+Mo)/Zn profiles (Figure 33) fluctuated little indicating that redox conditions in the lake have changed little since 1964. This is contrary to the Fe:Mn which indicated major fluctuations in the redox of the lake. All major peaks in the Cu/Zn profile correspond with peaks in the copper profile. All peaks in the (Cu+Mo)/Zn profile also correspond with peaks in the copper profile, with the exception of the peak at 7 cm.

As indicated by the correlation coefficient, variations in the Cu/Zn and (Cu+Mo)Zn were primarily controlled by variations in the copper profile. The Cu/Zn and (Cu+Mo)/Zn were also highly correlated ($r^2$=0.79) with each other. The (Cu+Mo)/Zn profile followed the Cu/Zn profile closely and provided no additional distinction. Despite increasing productivity and the associated decreasing dissolved oxygen and redox, the Cu/Zn and (Cu+Mo)/Zn changed little and remained relatively constant. Thus, they were not useful in determining the length of seasonal anoxia in the hypolimnion and the paleo-productivity of Lake Tenkiller.

These indicators may not apply to the lake due to infringement of several assumptions. The sedimentation rate (1.9 cm/yr) may not exceed the penetration of dissolved oxygen into the sediment. Cesium-137 dating of some cores from Station 5 indicated possible resuspension of sediment. No quantitative data that sufficient $SO_4$ is present for microbial $H_2S$ reduction existed; however, the smell of $H_2S$ in the tailwaters provided qualitative evidence.

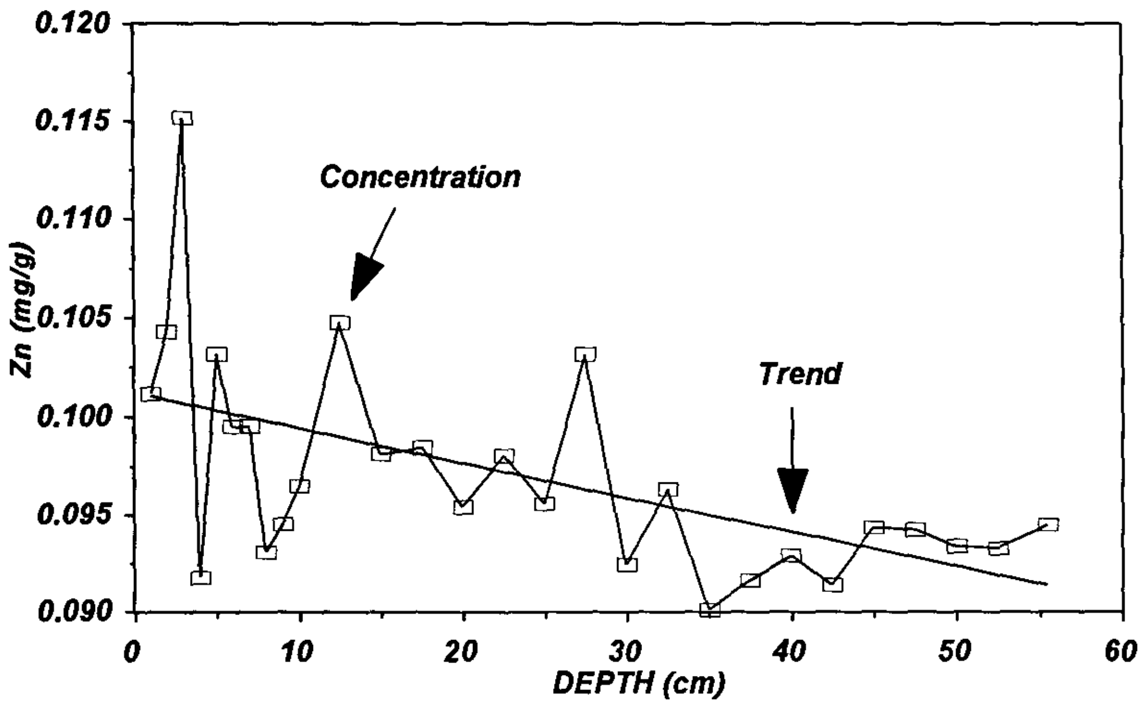



**Figure 31** Concentration-depth Profile of Zinc Measured in Lake Tenkiller Sediment Collected from Station 5 in October 1993.

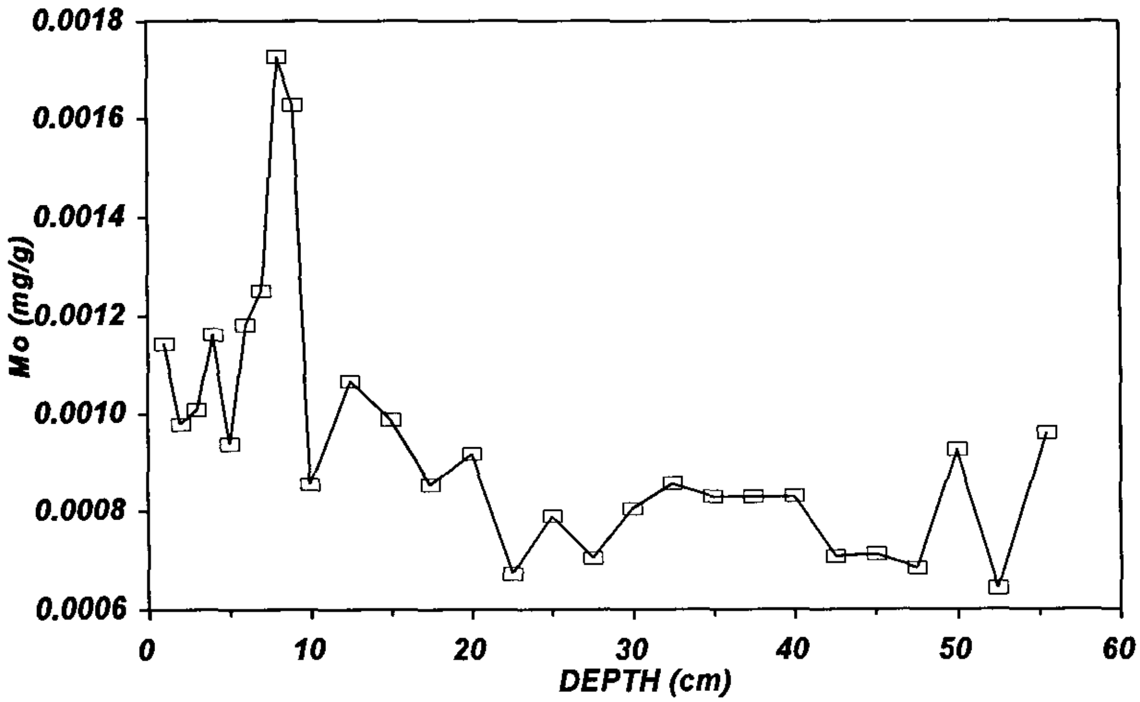



**Figure 32** Concentration-depth Profile of Molybdenum Measured in Lake Tenkiller Sediment Collected from Station 5 in October 1993.

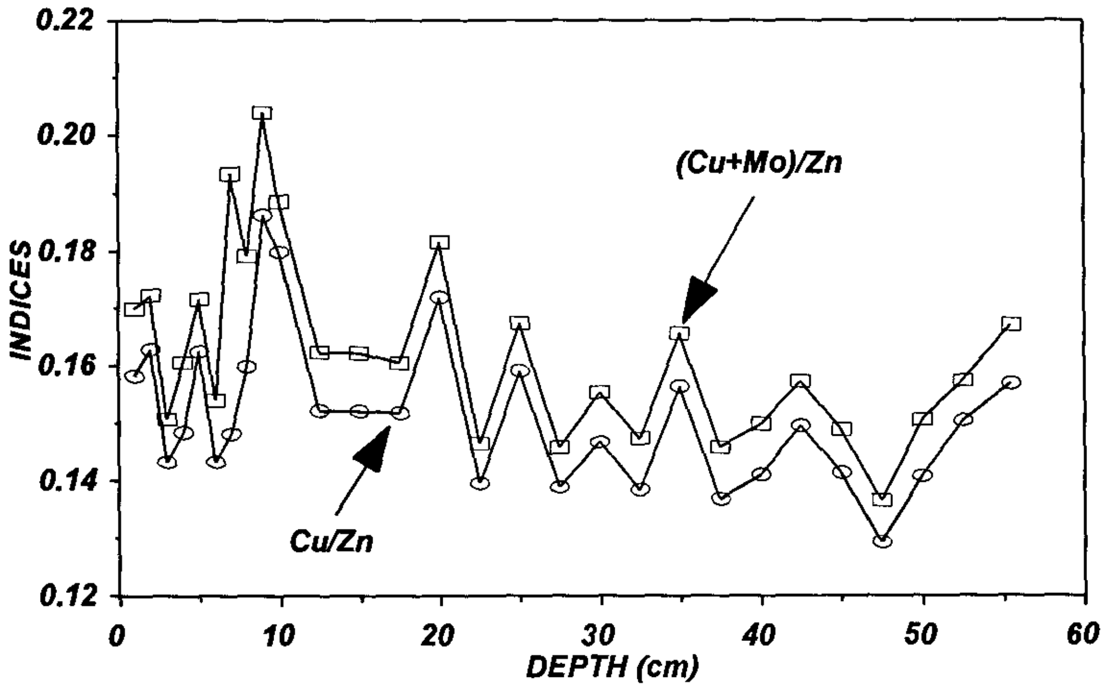



**Figure 33** R=Cu/Zn and Rp=(Cu+Mo)/Zn Profiles Calculated from Copper, Zinc, and Molybdenum Concentrations Measured in Lake Tenkiller Sediment Collected from Station 5 in October 1993.

## Sedimentary Calcium

Calcium concentrations generally increased from less than 1 mg/g at the bottom of the sediment profile (1964) to greater than 3 mg/g at the top (1993) of the profile (Figure 34). Calcium was not correlated with sodium ($r^2=0.00$) indicating that erosion was not the major influence on the sedimentary calcium concentration. Therefore, increased productivity must result in the observed trend of increasing calcium concentrations. Major calcium peaks occurred at 20, 9, and 2 cm. The calcium peaks at 9 and 2 cm were likely caused by the floods during 1985-86 and 1993, respectively. The calcium peak at 20 cm corresponds to a minima in the phosphorus profile and a period of low flow (1980-81). High productivity and stagnation may have caused increased deposition of calcium and the observed peak at 20 cm.

The calcium profile varies little from 55.5 to 27.5 cm indicating that little change in productivity occurred from 1964 to the late-1970s. From 27.5 to 20 cm, the calcium profile increased a great deal. The peak at 20 cm was discussed above. Calcium concentrations dropped dramatically following the peak at 20 cm and remained stable from 17.5 to 12.5 cm. From 12.5 cm to the sediment surface, the calcium concentrations increased dramatically again. This increase in the calcium profile is likely in response to the increased phosphorus loading (discussed in the next section) and the resulting increased primary productivity. The calcium profile indicated that primary productivity began increasing as late as the mid-1980s or as early as the late 1970s.

Calcium is correlated with iron ($r^2=0.51$), phosphorus ($r^2=0.38$), copper ($r^2=0.37$), Cu/Zn ($r^2=0.35$), (Cu+Mo)/Zn ($r^2=0.34$), and molybdenum ($r^2=0.30$). Calcium's correlation with iron and copper was discussed in a previous section. The correlation of calcium and phosphorus is discussed in the next section. Calcium's correlation with Cu/Zn, (Cu+Mo)/Zn, and molybdenum is likely due to its correlation with copper. Calcium was not correlated with the manganese, sodium, zinc, and Fe:Mn profiles.

The calcium profile was affected by high flow as observed at 1-2 and 8-9 cm. The calcium peak at 20 cm corresponds to a period of low flow and possible stagnation of the lake. During this period, increased productivity could have resulted in the observed peak. Little variation occurred in the profile from 55.5 to 27.5 cm. The major increases from 27.5 to 20 cm and from 12.5 cm to the sediment surface were likely caused by increased primary productivity in response to stagnation or increased phosphorus loading.

## Sedimentary Phosphorus

Phosphorus concentrations generally increased from 0.67 mg/g at the bottom of the sediment profile (1964) to 1.5 mg/g at the top (1993) (Figure 35). The trend when applied to the estimated year (Figure 36) agreed with the two to three fold increase found by Nolen et al. (1988). The phosphorus and sodium profiles were not correlated ($r^2=0.03$), thus the phosphorus concentrations were not primarily controlled by erosion.

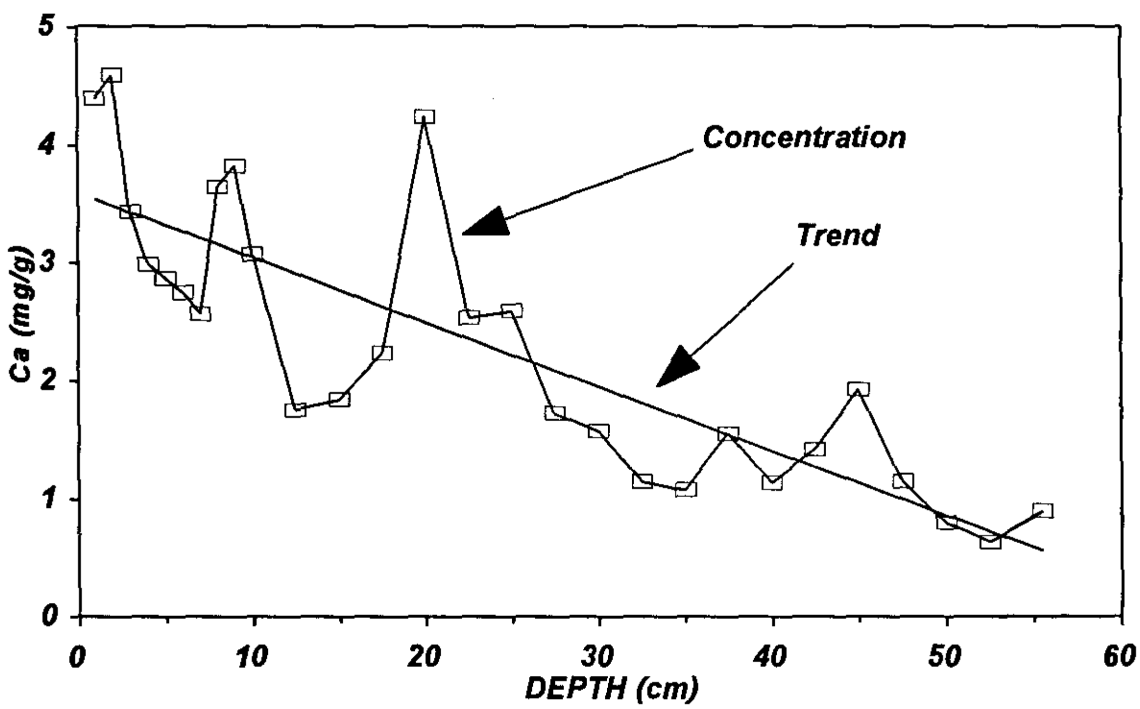



**Figure 34** Concentration-depth Profile of Calcium Measured in Lake Tenkiller Sediment Collected from Station 5 in October 1993.

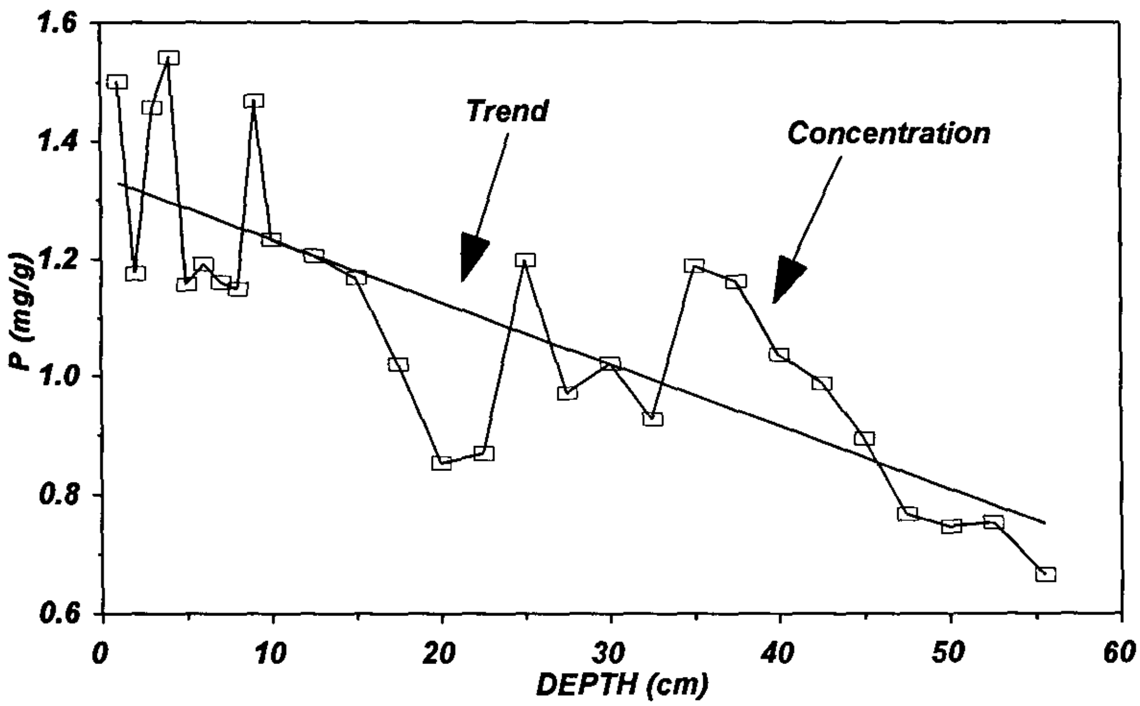



**Figure 35** Concentration-depth Profile of Phosphorus Measured in Lake Tenkiller Sediment Collected from Station 5 in October 1993.

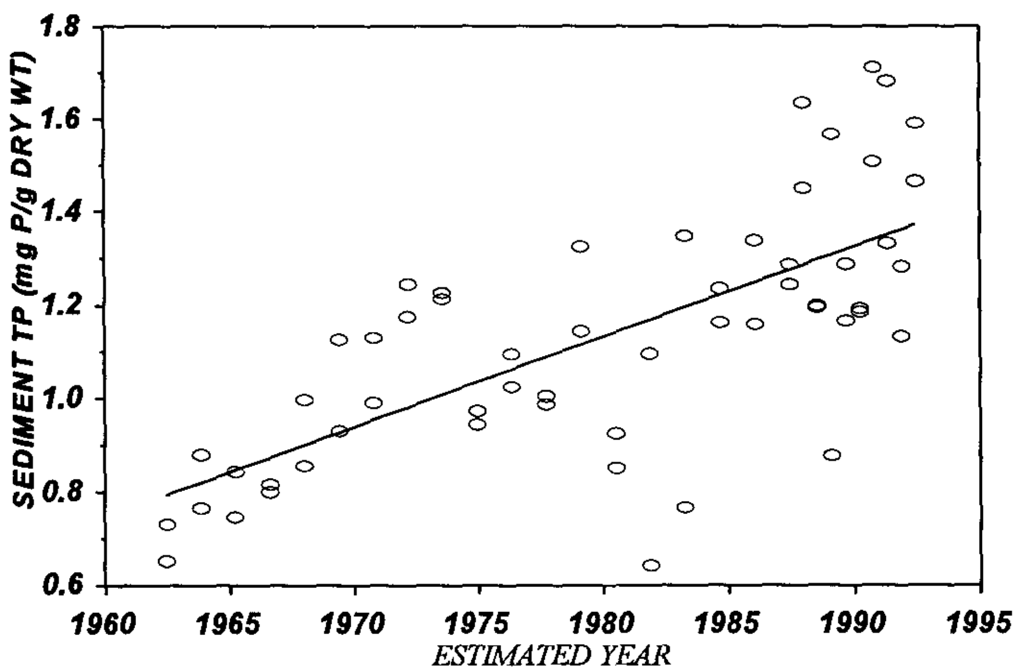



**Figure 36** Sedimentary Phosphorus in Lake Tenkiller at Station 5 with Estimated Year Based Upon Ce-137 Dating of Depth Profile.

Peaks in the phosphorus profile occured at 35, 25, 9, 4, and 1 cm. The phosphorus peaks at 35, 9, 4, and 1 cm were likely caused by floods of 1973-74, 1985-86, 1990, and 1993, respectively. The phosphorus peak at 25 cm does not correspond to any years with above average flow, however, it does correspond to a similar peak in the iron profile. The phosphorus minima at 20-22.5 cm likely correspond to the low flow years of 1980-81. The low phosphorus concentrations between 47.5 and 55.5 cm is probably due to the below average flow during the years of 1963-67.

The phosphorus profile exhibits an increase in the late 1960s and again in the early 1980s. In the late 1960s, the number of chickens (Figure 21) being raised in the Illinois River basin increased until the early 1970s, when the number decreased a great deal. In the early 1980s, the broiler (Figure 21) and hog and pig (Figure 22) industries expanded in the Illinois River Basin. Unlike the increase which occurred in the late 1960s, the phosphorus levels since the early 1980s have not experienced a minima or returned to the lower concentrations observed before the increase. However, the overall the phosphorus trend indicated a statistically significant increase since the 1960s (Figure 35).

The phosphorus profile was most correlated with the calcium profile ($r^2$=0.38); however, it was also somewhat correlated with iron ($r^2$=0.36), molybdenum ($r^2$=0.31), and copper ($r^2$=0.22). Phosphorus is sedimented by sorption by iron oxides (Mackereth 1966, Engstrom and Wright 1984, Jones et al. 1993), precipitation as iron phosphates, and coprecipitation with carbonates (Wetzel 1983, Engstrom and Wright 1984). Therefore, the correlation of phosphorus with calcium and iron is likely due to it coprecipitating with iron and $CaCO_3$. This coprecipitation with iron was the likely cause of the corresponding iron and phosphorus peaks at 25 cm. Molybdenum is most abundant in water when phosphorus is released from the sediments (Cole 1975). Therefore, the correlation between phosphorus and molybdenum is likely caused from both phosphorus and molybdenum being released from the sediments at the same time. It is likely that phosphorus and copper are somewhat correlated, because both phosphorus and copper are sedimented by biological uptake and deposition and coprecipitation with iron (Hutchinson 1957, Mackereth 1966, Cole 1975, Engstrom and Wright 1984). The phosphorus profile was not correlated with manganese, zinc, Fe:Mn, Cu/Zn, and (Cu+Mo)/Zn.

It is obvious that the phosphorus profile was affected by flow resulting in peaks at 1, 4, 9, and 35 cm and increasing input from developing agricultural enterprises in the basin. It is also obvious that phosphorus coprecipitating with calcium and iron may have affected the profile, especially the phosphorus peak at 25 cm which corresponded with the iron peak at 25 cm.

## QA Results

The overall QA results were good. No contamination was found in the blanks, and the percent recoveries from the spikes were generally acceptable (Table XXXVI).

**Table XXXVI** Percent Recovery of Spiked Sediment Samples from Lake Tenkiller.

| Metal | Range of Percent Recovery | Mean Percent Recovery |
|---|---|---|
| Calcium | 95 - 106 | 100 |
| Copper | 124 - 169 | 146 |
| Molybdenum | 170 - 233 | 202 |
| Sodium | 109 - 135 | 122 |
| Zinc | 150 - 175 | 162 |
| Manganese | 99 - 147 | 116 |
| Iron | 98 - 140 | 113 |
| Phosphorus | 117 - 124 | 120 |

High percent recoveries for zinc, copper, and molybdenum were caused by chemical interferences. The high copper concentrations in the spikes interfered with molybdenum and zinc analysis, and likewise, high molybdenum concentrations in the spikes interfered with copper analysis causing high percent recoveries (EPA 1986). The percent recoveries indicated that the analysis was accurate. Standard deviations were generally low, indicating that precision was good.

## Chlorinated Pesticide Residues

Chlorinated hydrocarbon pesticide residue analyses were performed on the water at Stations 1 and 2, on tissue extracts from fish (*Ictalurus punctatus*) collected at stations 2, 5, and 6, and thalweg sediments collected from stations 1, 2, 4, 5, and 6. Although residues were present none were in violation of alert, warning, or concern levels. For specific analytical and statistical methods, the reader is referred to Gartman (1995). Water and sedimentary data are presented in Tables XXXVII-XXXVIII; fish residues are presented in the Public Health Monitoring section.

**Table XXXVII** Insecticide Concentrations (in the water) Entering Lake Tenkiller.

| Insecticide | Station 1 (μg/l) | Station 2 (μg/l) |
|---|---:|---:|
| Lindane | 1.5071 | 2.4708 |
| Heptachlor | 2.9842 | ND |
| Aldrin | ND | 0.7445 |
| DDD | ND | 0.4822 |

**Table XXXVIII** Insecticide Concentrations of Lake Tenkiller Sediments (mg/kg).

| ID-Replicate | pH | Lindane | Heptachlor | Aldrin | Dieldrin | Endrin | DDD | DDT | Methoxychlor |
|---|---|---|---|---|---|---|---|---|---|
| 1 - 1 | 5.3 | 0.0009 | ND | 0.0008 | 0.0042 | ND | ND | 0.0154 | ND |
| 1 - 2 | 7.1 | 0.0010 | ND | 0.0015 | 0.0079 | ND | ND | ND | ND |
| 1 - 3 | 6.7 | ND | 0.0010 | 0.0014 | 0.0161 | ND | ND | ND | ND |
| 2 - 1 | 6.6 | ND | 0.0038 | 0.0065 | ND | ND | ND | 0.2430 | ND |
| 2 - 2 | 6.2 | 0.0022 | 0.0009 | 0.0022 | 0.0038 | ND | 0.0002 | 0.0016 | ND |
| 2 - 3 | 6.4 | ND | 0.0021 | 0.0038 | ND | 0.0075 | 0.0003 | 0.0018 | ND |
| 4 - 1 | 6.6 | 0.0005 | 0.0004 | 0.0005 | 0.0003 | ND | ND | ND | ND |
| 4 - 2 | 6.2 | 0.0010 | 0.0005 | 0.0013 | 0.0016 | ND | ND | ND | ND |
| 4 - 3a | 6.4 | 0.0011 | 0.0006 | 0.0008 | 0.0005 | ND | ND | ND | ND |
| 4 - 3b | 6.3 | 0.0030 | 0.0016 | 0.0027 | 0.0026 | ND | 0.0018 | ND | ND |
| 4 - 3c | 6.1 | 0.0005 | 0.0004 | 0.0005 | 0.0003 | ND | ND | ND | ND |
| 5 - 1 | 6.5 | 0.0012 | ND | 0.0026 | 0.0234 | ND | ND | ND | ND |
| 5 - 2 | 6.0 | ND | 0.0017 | 0.0023 | 0.0041 | ND | ND | ND | ND |
| 5 - 3 | 7.0 | 0.0015 | ND | 0.0022 | 0.0056 | ND | 0.0385 | ND | ND |
| 6 - 1 | 7.1 | ND | ND | 0.0044 | 0.0264 | ND | ND | 0.0634 | ND |
| 6 - 2 | 6.3 | 0.0044 | 0.0027 | ND | 0.0048 | ND | 0.0054 | 0.0168 | ND |
| 6 - 3 | 6.4 | 0.0026 | ND | 0.0024 | 0.0054 | ND | ND | 0.0020 | ND |

## 10. Limiting Nutrient (Nitrogen and Phosphorus) Inflow and Outflow

Evaluation of the limiting nutrient is discussed in the algal assay bottle tests described below. Briefly, limiting nutrient assays illustrated that Lake Tenkiller was phosphorus limited in the lower reaches, variable (N, P, or N and P) in the midreaches, and probably light-limited in the headwaters. Also, initial evaluation indicated that about 75% of the total phosphorus load impinging Lake Tenkiller is decayed (sedimentation, biological uptake, etc.) in the upper 1/3 of the reservoir while nitrogen did not significantly decay throughout the length of the reservoir. Therefore, regulation of algal growth probably was influenced by the interaction of light availability (e.g., turbidity and Secchi disk depth), phosphorus, and nitrogen, all of which are strongly influenced by hydraulic loads. Therefore, we chose to emphasize the longitudinal transport of the nutrients in conjunction with the hydraulic loads. The following discussion describes the development of the model and was taken from Jobe (1995). A brief review of available literature also is included.

The objectives of this study were to:

1) identify and characterize statistical distributions of spatial trends in phosphorus, nitrogen, turbidity, and transparency in Lake Tenkiller,

2) calculate a hydraulic budget of Lake Tenkiller

3) develop a risk-model that describes the intrinsic risk of eutrophication of Lake Tenkiller by evaluating the reservoir ecological risk factors (RERF)

4) integrate the reservoir ecological risk factors to evaluate trends of trophic status of Lake Tenkiller,

5) evaluate trends in trophic status of Lake Tenkiller under various nutrient load reduction scenarios and predict concomitant eutrophication responses

### Quantitative Analysis of Eutrophication

Phosphorus

Most of the traditional methods have been based on or modified from Vollenweider's (1968) nutrient loading concept and attempted to predict inlake P concentrations under observed allochthonous nutrient loads. Since the models proposed in this project do not rely on inherent assumptions common to the traditional loading models, only a brief discussion of these models is presented (for a more detailed description of traditional models or their derivations consult the original citations or Jobe 1991).

Originally proposed for lakes by Biffi (1963), Vollenweider's (1968) loading model is based upon a mass balance equation of P inputs and outputs,

$$V \frac{dP}{dt} = M_p - Q P - \sigma P V \qquad \text{Equation 3.}$$

where P is the inlake P concentration, t is time, V is the lake volume, $M_P$ is the impinging annual P load, Q is the annual hydraulic load (outflow volume), and $\sigma$ is the net sedimentation coefficient. To predict inlake P concentration, the derivative must be set to zero (this is the steady state assumption or $dP/dt=0$) and solve for P,

$$P = \frac{L}{z_a(\sigma + \rho)}$$ Equation 4.

where P is inlake P concentration, $z_a$ is the mean depth, L is the areal loading (i.e., $M_p$/Area), $\sigma$ is the net sedimentation coefficient, and $\rho$ is the hydraulic flushing rate (Q/V and often denoted as $\tau_w^{-1}$). Vollenweider (1968) used this approach to delineate trophic status boundaries of lakes on a plot of L versus $z_a/\tau_w$, while recognizing the only parameter that can be managed was L. Therefore, acceptable limits of P loads in a given lake would be determined by its $z_a$ and hydraulic load. The reason for including $z_a/\tau_w$ was that lakes with larger volumes and smaller residence times could tolerate larger P loads without eutrophication symptoms.

The major criticism of Vollenweider's model was the estimation of $\sigma$. Therefore, with the model given in equation 2 as the theoretical construct, some investigators empirically estimated $\sigma$, while others attempted to estimate inlake P from measured external loads by choosing to estimate a P retention coefficient (Dillon and Rigler 1974a, Larsen and Mercier 1976, Canfield and Bachmann 1981, Reckhow 1988), which describes the fraction of inflowing P retained by the lake (Table XXXIX). Some of these models differentiated natural and artificial lakes, while others were developed exclusively for natural lakes. Ironically, as suggested by the results of the Lake Washington and ELA studies, a lake's response has been found to be dependent on the magnitude of the P flux, and not the concentration. Since, criteria for trophic status have been based upon measured inlake P concentration, the goal of these models was to estimate the resulting inlake P concentration.

These models, having the mass balance equation as the theoretical basis, assume the lake is a continuously stirred tank reactor (CSTR) and is in steady state (i.e., $dP/dt = 0$). Although long time frames in natural lakes may approximate this assumption and short-lived reservoirs do not, these models have been applied to artifical lakes with some success (Larsen and Mercier 1976, Canfield and Bachmann 1981, Reckhow 1988). Large run-of-the-river reservoirs violate the CSTR assumption by differential P sedimentation rates in the headwaters when compared to the lower reaches (Thornton et al. 1990). This differential sedimentation rates are influenced by seasonal hydraulic regimes (Thornton et al. 1990). The reservoir ecological risk factor (RERF) model conserves the steady state assumption but does not assume a CSTR. It is this advantage that provides a better comparative indicator among and within reservoirs while incorporating an assessment of intrinsic risk.

## Nitrogen

During the peak of eutrophication research ca. late 1960 to ca. late 1970s, controversy surrounded the acceptance of phosphorus as the primary cause of accelerated eutrophication. Most of this controversy stemmed from data that suggested algal growth as being controlled by carbon or nitrogen in advanced stages of eutrophication (Bowen 1970, Anonymous 1971). The resulting debate ended with dispelling the carbon school of thought while the nitrogen versus phosphorus continues today. Thus, many nitrogen indices of eutrophication have been proposed and merit discussion.

Most nitrogen-based trophic status indices have been based upon inlake nitrogen

Table XXXIX Traditional Phosphorus Models in Lakes.

| Reference | Model Equation | CSTR Assumption |
|---|---|---|
| Vollenweider (1968) | inlake $P = L/(z_a(\sigma+\rho))$ | Yes |
| Dillon and Rigler (1974a) | inlake $P = L(1-R_{exp})/(z_a\rho)$ | Yes |
| Vollenweider (1975) | $\sigma = 10/z_a$ | Yes |
| Kirchner and Dillon (1975) | $R_P = 0.426e^{-.271q} + 0.574\ e^{-0.00949q}$ | Yes |
| Larsen and Mercier (1976) | $R_{exp} = 1/(1+\rho_w)$ | Yes |
| Canfield and Bachmann (1981) | $\sigma_{natural\ lakes} = 0.162(L/z_a)^{0.458}$ | Yes |
| | $\sigma_{artificial\ lakes} = 0.114(L/z_a)^{0.589}$ | Yes |
| | inlake $P = fL/(z_a(\sigma+\rho))$ | Yes |
| Reckhow (1988) | $\log(P) = \log[P_i/(1+k_P\tau_w)]$ | Yes |
| | $k_P = 3.0P_i^{0.53}\tau_w^{-0.75}z_a^{0.58}$ | |
| | $\log(N) = \log[N_i/(1+k_N\tau_w)]$ | Yes |
| | $k_N = 0.67\tau_w$ | |
| | $\log(ch)_{max} = 1.314+\log(P^{0.321}N^{0.384}n_c^{0.45}\tau_w^{0.136})$ | |
| | $\log(Secchi) = -0.47+\log(P^{-0.364}\tau_w^{0.102}z_a^{0.137})$ | |

concentrations instead of a loading model. However, exceptions exist. Baker et al. (1985) proposed a simple modification to Vollenweider's (1968) P loading model by incorporating nitrogen loading as the primary nutrient in the mass balance equation for Florida lakes. Using this model, Baker et al. (1985) predicted more accurate chlorophyll *a* levels in the Florida lakes as well as 101 National Eutrophication Survey (NES) lakes. They proposed the decision of which model to use (i.e., the N or P) should depend upon which gives the lowest chlorophyll *a* prediction (Baker et al. 1985). I assume the rationale for such a decision was that lower values reflected more extreme limitation and thus more closely followed the lake's trend in eutrophication.

Trophic State Indices

To simplify comparison among lakes, assess current trophic status, and monitor effectiveness of restoration schemes, several empirical "inlake" indices have been developed. The salient strengths and weaknesses of each are discussed below.

Carlson's (1977) trophic state index (TSI) traditionally has been the most popular and is based upon the regression of natural log transformed values of spring surface total phosphorus ($\mu$g P/l), Secchi disk depth (m), and chlorophyll *a* ($\mu$g/l). Three TSIs can be calculated from these three data, 1) TSI(CHL), 2) TSI(TP), and 3) TSI(SD). He scaled Secchi disk limits of 0 and 64 m to TSI values of 100 and 0, respectively, where increasing TSI values denoted increasing eutrophy (Carlson 1977). A TSI increase of 10 units equated to a halving of the Secchi disk and doubling of surface P. Chlorophyll *a* was the dependent variable and thus the model did not impose such linearity. Computational forms of each TSI are:

$$TSI\ (CHL) = 10\left(6 - \frac{2.04 - 0.68 \ln (chl)}{\ln 2}\right) \qquad \text{Equation 5.}$$

$$TSI\ (TP) = 10\left(6 - \frac{\ln \frac{48}{TP}}{\ln 2}\right) \qquad \text{Equation 6.}$$

$$TSI\ (SD) = 10\left(6 - \frac{\ln SD}{\ln 2}\right) \qquad \text{Equation 7.}$$

Carlson's (1977) models have been used widely and apply mostly to natural lakes. These TSIs assume P-limited algal growth and algal biomass controls transparency. Reservoirs may temporarily exhibit the first but varies from transparency controlling biomass (headwaters, i.e., light limited growth) to algal biomass controlling transparency in the lacustrine zone (Thornton et al. 1990). This trend presents a problem in correlation studies and denotes uncertainty in directionality. Thus, whole-lake correlation studies in reservoirs provide little insight into the system process. Significant deviations among Carlson's (1977) three TSI's from the headwaters to the lacustrine zone in reservoirs most likely result from this phenomena. Therefore, a biased assessment of reservoir trophic status results from using such whole-lake indices of homogeneity. However, Jobe (1991) illustrated that these deviations could be minimized if the headwater stations were eliminated and only zones where the algae were not light-limited were included. The weakness of this approach is the resulting assessment only describes the lacustrine zone.

Carlson (1980b) and Osgood (1982b) have proposed using TSI deviations on natural lakes to describe lake typology (e.g., argillotrophic, dystrophic), but warns the TSI is an index not a definition of trophic status, i.e., the index and its deviations reflect trophic structural conditions not a functional process like accelerated eutrophication. Carlson (pers. com.) further believes the trophic status of a lake, by definition, is a manifestation of lake function or response of a lake not its structure, *per se*. The RERF integrates all zones of the reservoir as a continuum and compensates for the spatial transition from light to nutrient limitation.

Canfield (1983) proposed a similar model that predicts chlorophyll *a* concentrations from observed inlake nitrogen concentrations. This index is more applicable to southern lakes that are highly eutrophic. Canfield (1983) proposed the nitrogen-based predictor was more appropriate for eutrophic systems because algal growths in these lakes were characteristically N-limited. Predicted chlorophyll *a* in this model was given by:

$$\log (Chl\ a) = -0.15 + 0.744 \log (TP), \quad r^2 = 0.59 \qquad \text{Equation 8.}$$

$$\log (Chl\ a) = -2.99 + 1.38 \log (TN), \quad r^2 = 0.77 \qquad \text{Equation 9.}$$

$$\log (Chl\ a) = -2.49 + 0.269 \log (TP) + 1.06\ (TN),\ r^2 = 0.81 \qquad \text{Equation 10.}$$

The best correlation from the above parameters was obtained from inclusion of TN and TP.

Kratzer and Brezonik (1981) also developed a trophic state index based on nitrogen for several Florida lakes. Their computational form was:

$$TSI\ (TN) = 54.45 + 14.43 \ln (TN) \qquad \text{Equation 11.}$$

where TN was total nitrogen (mg N/l). Kratzer and Brezonik (1981) suggested, similar to Canfield's (1983) approach, to consider the TSI(TN) with Carlson's (1977) TSI(TP) and average the lower of the two with Carlson's TSI(SD) and TSI(CHL) to derive a TSI(AVG). This averaging protocol was criticized by Osgood (1982a) and Lambou (1982), while Kratzer and Brezonik (1982), Brezonik and Kratzer (1982), and Baker et al. (1985) defended the TSI(TN) usage for the same reasons. Although both sides agreed the study lakes on which the TSI(TN) was developed were N limited, arguing for or against incorporation of the N parameter in trophic state indices parallels the nitrogen versus phosphorus debate.

Shannon and Brezonik (1972b) used a multivariate analysis (cluster analysis) to develop two multiparameter TSI's that included a variety of eutrophication-related parameters. This index categorized colored and clear lakes. The rationale for a multivariate approach was given by Brezonik (1984). Computational forms, respectively, were:

$$TSI_{Colored}$$

$$= 0.848 \frac{1}{SD} + 0.809\ COND + 0.887\ TON + 0.768\ TP + 0.930\ PP + 0.780\ CHA + 0.893 \frac{1}{CR} + 9.33 \qquad \text{Equation 12.}$$

$$TSI_{Clear}$$

$$= 0.936 \frac{1}{SD} + 0.827\ COND + 0.907\ TON + 0.748\ TP + 0.938\ PP + 0.982\ CHA + 0.579 \frac{1}{CR} + 4.76 \qquad \text{Equation 13.}$$

where SD is Secchi disk transparency (m), COND is conductivity ($\mu$℧/cm), TON is total organic nitrogen (mg N/l), TP is total phosphorus (mg P/l), PP is primary productivity (mg C/$m^3$-hr), CHA is chlorophyll *a* (mg/$m^3$), and CR is the cation ratio (Na + K)/(Ca + Mg) (Shannon and Brezonik 1972b). The TSI boundaries that denoted transitions from oligotrophy to mesotrophy to eutrophy were $\approx$ 1.2 and $\approx$ 5, respectively. Although the Shannon and Brezonik (1972b) TSIs are useful from an empirical perspective, ecological oversimplification coupled with the cost of obtaining the necessary data has hindered its widespread application. Also, this TSI, like most single indices, reduces the analysis of trophic status to a single "whole lake" criterion based solely on structural properties; it describes nothing of a lake's functional responses.

Porcella et al. (1980) proposed a similar multi-parameter index that incorporated scalar transformations of Carlson's (1977) TSI's among other trophic state parameters and coined it the lake evaluation index (LEI). The LEI was developed to evaluate restoration effectiveness and was calculated as the arithmetic average of the individual scalars where the TP parameter or TN scalar were mutually exclusive according to which yielded the lowest LEI. As argued by Baker et al. (1985), Kratzer and Brezonik (1982), and Brezonik and Krazter (1982), the reason for choosing the lower was that the lower parameter was in lesser supply and thus most likely denoted the limiting nutrient, thereby reflecting the eutrophication trend. The computational form was:

$$LEI = 0.25\ [0.5\ (XCA + XMAC) + XDO + XSD + XTP] \qquad \text{Equation 14.}$$

where

$$XCA = 30.6 + 9.81 \ln(CA), \qquad XCA \leq 100 \qquad \text{Equation 15.}$$

$$XMAC = PMAC, \qquad XMAC \leq 100 \qquad \text{Equation 16.}$$

$$XDO = 10 \frac{\left( \sum_{i=0}^{i=Z_{MAX}} |EDO - CDO| \; \Delta V_i \right.}{V} \qquad \text{Equation 17.}$$

$$XSD = 60 - 14.427 \ln ( SD ), \qquad XSD \leq 100 \qquad \text{Equation 18.}$$

$$XTP = 4.15 + 14.427 \ln ( TP ), \qquad XTP \leq 100 \qquad \text{Equation 19.}$$

$$XTN = 14.427 \ln ( TN ) - 23.8, \qquad XTN \leq 100 \qquad \text{Equation 20.}$$

and EDO and CDO represented equilibrium dissolved oxygen concentrations predicted from temperature profiles and current (or measured) DO concentrations for stratum z, respectively. The XDO scalar was omitted from the LEI equation in the original published equation, but the accompanying text describes its incorporation. Due to the original textual description and since the multiplier was 0.25 (implying an arithmetic average of $n = 4$), I assume the exclusion of XDO was merely an oversight and thus have included it in the above LEI formulation. Analogous to Carlson's (1977) scale, the LEI ranges from 0 (denoting extreme oligotrophy) to 100 (denoting extreme eutrophy). Weaknesses in the LEI, some of which Porcella et al. (1980) recognized, include:

1) individual scalars for a given lake may exhibit large discrepancies,
2) arithmetic averaging will result in the same LEI if the discrepancies of the individual scalars offset one another,
3) applies a single index of homogeneity and thus denotes a "whole" lake structural descriptor, and
4) gives equal weights to a multitude of eutrophication symptoms when fewer actually may exist or exceed one another in significance (e.g., deep eutrophic lakes are not likely to have a macrophyte problem thus inclusion of XMAC is knowingly biased).

The LEI and the other trophic state indices discussed above can be used, however, to evaluate functional aspects of lakes. Some authors proposed using changes or deviations in the indices over time to assess trophic status rather than absolute values (Carlson 1980b, 1983). Unfortunately, reservoirs present both functional responses in time and a strong spatial heterogeneity that imposes unique considerations in using such singular "whole-lake" indices of homogeneity or their deviations.

Finally, two functional models that have been used as trophic state indicators are areal and volumetric hypolimnetic oxygen depletion rates, AHOD and VHOD. However, these models assume hypolimentic oxygen depletion is caused by community respiration (mostly bacterial) of endogenic organic material, material which was solely produced in the epilimnion or at least a constant fraction of it (Hutchinson 1938). Significant variance was observed when these models were applied to natural lakes, but most problems seem to have stemmed from differing fractional hypolimnetic volumes. Reservoirs may approximate natural lakes in the constancy of fractional hypolimnetic volumes but have hypolimnetic oxygen dynamics that are highly dependent on inflowing oxygen demand and hydraulic regimes (i.e., short hypolimnetic $\tau_w$). These influences can decrease the AHOD, while allochthonous organic loads can increase the AHOD. Clearly, these phenomena influence the impact of epilimnetic productivity (or a constant fraction of it) on AHOD and VHOD rates. For these reasons, AHOD rates have been of little use in assessing trophic status of reservoirs. I am not suggesting hypolimnetic oxygen levels lack diagnostic significance. On the contrary, hypolimnetic anoxia not only indicates increasing eutrophy, it exacerbates the problem by P recycling. I merely suggest numerical criteria for reservoir AHOD and VHOD are of little use. For further reviews on AHOD and VHOD model details, computation, and interpretation, the reader is referred to Hutchinson (1938), Cornett and Rigler (1979), Welch (1979), and Cornett and Rigler (1980).

## Reservoirs

Reservoirs have introduced an array of unique questions on accelerated eutrophication to limnologists. Traditional approaches have applied classical limnological concepts to reservoirs which are usually qualitatively valid. However, when quantitative methods are used, large errors often result. These problems have led contemporary thought to differentiate reservoirs and natural lakes as distinct ecosystems.

The differences between the two types of ecosystems ostensibly are derived from the external influences upon the ecosystem's behavior and the system's internal response patterns. Typical glaciated lakes are influenced by diffuse allochthonous nutrient loads but have a high degree of autochthonous control mechanisms with a large degree of homogeneity. In contrast, reservoirs are driven by less diffuse allochthonous loads (i.e., point of entry is the headwaters) and maintain allochthonous control in the headwaters but exhibit a gradation to autochthonous control near the lower end (dam) indicating more heterogeneity. Reservoirs exhibit characteristics similar to rivers in the headwaters and similar to glaciated lakes near the dam, but not enough of each to be classified as either type of system (Table XL). Assessment of eutrophication-related processes in large mainstem reservoirs must accommodate these phenomena. Accurate modelling of these processes must account for the uniqueness of these "hybrid" ecosystems.

Table XL Comparison of a River, Reservoir, and Glacial Lake (modified from Thornton et al. 1990).

| Attribute | River | Reservoir | Glacial Lake |
|---|---|---|---|
| Watershed influence | greater | intermediate | lesser |
| Shoreline | elongate | astatic | circular |
| Water level | variable | variable | natural |
| Flushing rate | rapid | intermediate | slow |
| Ionic composition | variable | intermediate | stable |
| Sedimentation | low | high | low |
| Turbidity | high | intermediate | low |
| Organic accumulation | low | rapid | slow |
| Nutrient supply | allochthonous | both | autochthonous |
| Nutrient loss | advection | both | sedimentation |
| Growth selection | rapid (r) | intermediate (r & K) | homeostatic (K) |
| Immigration/ extinction | rapid | rapid | slow |
| Spatial structure | longitudinal gradients | both | vertical gradients |

Longitudinal Gradients

The major abiotic influences on reservoirs originate in the watershed from point source and nonpoint (diffuse) sources. The stream network serves to focus the loads and impinges upon the reservoir at the headwaters. These loads vary most during the spring and fall when runoff occurs more frequently and with greater variation. These events tend to be processed in a "plug-flow" manner instead of a diffusive homogenization throughout the ecosystem as in natural lakes.

During summer, the runoff, lake level, and incoming nutrient loads stabilize and form a longitudinal gradient in which a water quality trend is established along a continuum. Along this continuum, the headwaters are dominated by qualities of the incoming hydraulic load (river) which shifts to lacustrine conditions near the dam (Table III).

The gradient is determined by hydraulic flow characteristics in the reservoir. In the headwaters, flow velocity is sufficient to keep inflowing sediments in suspension in the water and few differences between river water and lake water exist. The velocity of the influent water decreases as the width and depth of the reservoir increases causing a clarification as suspended solids sediment. As depth of the euphotic zone increases and available nutrients are still relatively high, a peak in algal growth occurs. This is the zone of transition (Thornton et al. 1990). The velocity of water continues to decrease which allows finer particles to settle and gives way to a quiescent, less variable body of water. This zone maintains characteristics most similar to natural lakes and hence has been coined the lacustrine zone (Thornton et al. 1990). A comparative summary is given in Table XLI.

## Modelling The Trend

Quantification of the spatial trend consists of describing the longitudinal gradient as a continuum of change along the thalweg of the reservoir. Intrinsic parameters that drive the system appear to be the availability of light (i.e., turbidity or transparency) which increases and phosphorus which decays with thalweg distance. The endpoint used in this model is chlorophyll *a* density, although other biotic variables, e.g., algal biomass, diversity, algal cell counts, could be used and should be investigated.

Driving Variables. While few models have been fit to the trend in light availability, total phosphorus has been shown to exponentially decay with distance (Thornton et al. 1990). Motulsky (1987) gives the generalized exponential,

$$TP = A\, e^{(-TD)} + B \qquad \text{Equation 21.}$$

where, TP is the total phosphorus concentration at thalweg distance TD, A is the decay coefficient, and B is an asymptote at which TP decay is 0. While the model's simplicity is attractive, it forces the inital decay of TP to occur at the upper bound of the reservoir (i.e., thalweg distance = 0). Data from Lake Tenkiller illustrate that TP decay may start as far as 30% downstream along the thalweg. Therefore, because reservoirs are known to change the TP/turbidity decay according to seasonal hydraulic regimes (i.e., spring or fall runoff), a more appropriate model for accommodating such trends is a sigmoidal equation which can evaluate various decay rates at any location of the transition zone.

This precept is supported further by the length/cross-sectional area relationship of a pyramid. If a large mainstem reservoir approximates a pyramid laying on its side the cross-

Table XLI Characteristics of the Longitudinal Gradient in Large Mainstem Reservoirs (modified from Thornton et al. 1990).

| Factor | Riverine | Transition | Lacustrine |
|---|---|---|---|
| Basin | narrow | intermediate | broad, deep |
| Flow Velocity | high | reduced | low |
| Transparency | low | intermediate | high |
| Turbidity | high | intermediate | low |
| Light Availability | $z_{photic} < z_{mix}$ | increased | $z_{photic} > z_{mix}$ |
| Nutrients | high | intermediate | low |
| Primary Productivity | light limited | high | nutrient limited |
| Organic Matter Supply | external | intermediate | internal |
| Trophic Status Trend | eu- | eu/meso- | oligo- |

sectional area increases with distance (height) as a point moves from its apex to base. It follows that because velocity is a quotient of flow divided by cross-sectional area and flow is constant (or nearly so) in a mainstem reservoir, the increase in cross-sectional area should equate to a decrease in velocity. At some point inherent to the morphometry of the reservoir (i.e., dimensions of the pyramid), velocity slows to a point at which the suspensoids begin to settle; further decreases in velocity would have little effect. Hence, a sigmoidal trend in environmental correlates of suspended particles, e.g., P coprecipitates, can be expected. Applying Motulsky's (1987) sigmoidal equation to TP on distance yields,

$$TP_i = MIN + \frac{MAX - MIN}{1 + e^{SF(EL_{50} - TD)}} \qquad \text{Equation 22.}$$

where, $TP_i$ is total phosphorus concentration ($\mu$g P/l) at thalweg distance $i$, MAX is maximum TP observed for the given data, MIN is minimum TP observed, SF is the slope factor which describes the rate at which TP decays, EL50 is the effective length (thalweg distance) at which TP decay is 50% complete, and TD is the thalweg distance.

A different approach is suggested for turbidity. Unlike phosphorus, a decreasing turbidity increases algal growth until light is replete (the transition zone). Therefore, an inverse relationship of turbidity and algal growth is expected. Applying Motulsky's (1987) sigmoidal equation to turbidity on distance yields,

$$TB_i = MIN + \frac{MAX - MIN}{1 + e^{SF(EL_{50} - TD)}} \qquad \text{Equation 23.}$$

For light-related measurements that indicate transparency (e.g., Secchi disk), the same model can be applied. If Secchi disk is applied, a transformation of the depth to decimeters is suggested because the typical Secchi disk depths in reservoirs usually are 10 m or less. Scaling the Secchi disk depths to decimeters establishes a range that approximates that of turbidity in nephelometric turbidity units (NTUs) and TP in $\mu$g P/l. This scaling affords direct comparison of model parameters and coefficients. Applying Motulsky's (1987) sigmoidal equation to Secchi disk depth on distance yields,

$$SD_i = MIN + \frac{MAX - MIN}{1 + e^{SF(EL_{50} - TD)}} \qquad \text{Equation 24.}$$

where SD is the Secchi disk depth (dm) and the other factors are as previously described relative to the spatial trend in Secchi disk. It is worth noting that much controversy has been generated on the relationship between algal biomass and Secchi disk depths (Tyler 1968, Bannister 1975, Lorenzen 1980, Megard et al. 1980, Carlson 1980a, Edmondson 1980). However, this controversy has been directed towards the effect of chlorophyll and/or algal biomass upon the transparency. In the above model, the opposite is being simulated where the transparency controls the algal biomass in the headwaters via light limitation and undergoes a transition to nutrient limitation upon decay. The point of contention for past controversy is insignificant in this model.

Clearly, the sigmoidal model assumes the highest TP and turdidity will occur at the headwaters and the minimum occurs downstream from the maximum, while opposite transparency trends exist. If this trend is not observed in a given snapshot of data (i.e., a given day), the reservoir doesn't approximate equilibrium and thus invalidates the model. In such a case, the reservoir is probably processing a "plug" of "flow" as described above and hence is in a state of transition.

Response Variables. The longitudinal gradient in chlorophyll density occurs as a maximum at the transition zone. I postulate that the zone of maximal chlorophyll density occurs at a point where light availability increases to a point of saturation for the algae and another environmental factor becomes restrictive (i.e., nutrient limitation). It is this linkage upon which the risk model is based.

Alternatively, a strong argument for hydraulic control could be made because flow regimes could regulate mixing depth and thus suspension of algal cells, thereby optimizing enviromental conditions for algal cells by preventing settling. However, my hypothesis is based on the close correlation of the occurrence of the chlorophyll maximum with the turbidity and TP decays. It is not likely that the algal community would be adapted to velocity requirements that are optimized so near the range of that for suspensoid and coprecipitate settling, especially given the variety of buoyancy adaptations of algae.

If the assumed linkage is true, a relationship between the driving variables and the endpoint will exist. I propose that the maximum chlorophyll density can be predicted by an equation in which light and TP are optimized. Such a relationship can be evaluated by a variety of equations but I chose a maxima function as given by Spain (1982) of chlorophyll *a* density on thalweg distance. I acknowledge the weakness of this approach is a lack of parametric coefficients that have a theoretical basis. However, the estimated coefficients are evaluated on a relative basis and are not characterized as representing a physical process. The maxima function is represented by,

$$CA = CRF \times TD \times e^{(PF \times TD)} \qquad \text{Equation 25.}$$

where CA is chlorophyll *a* density ($\mu$g/l), CRF is an empirical chlorophyll response factor, TD is thalweg distance (decimal fraction), and PF is an empirical power factor. If the hypothesis is true, a close correlation between kinetics of transparency and phosphorus decay with the chlorophyll maximum should exist.

## MATERIALS AND METHODS FOR MODELLING LIMITING NUTRIENT INFLOWS AND OUTFLOWS

### Water Quality Analyses

Each of the sampling sites were sampled approximately monthly from Apr 92 to Oct 93. During each sampling time, I collected epilimnetic (0.5 m below the surface) and hypolimnetic (0.5 m above sediment) water samples in clear acid-washed high-density polyethylene (HDPE) or polypropylene (PP) bottles and one epilimnetic sample in a non-acid-washed opaque HDPE or PP bottles at each site. The samples were placed on ice and returned to Oklahoma State University for analyses. The analyses were conducted within 48 hr and included total phosphorus and ortho-P, total and phenolphthalein alkalinity, and total hardness as per Lind (1985), total nitrogen as per Bachmann and Canfield (1990), and chlorophyll *a* as per Standard

Methods (APHA 1989). Additional lab analyses included nitrate-N, chloride, and sulfate by liquid chromatography (EPA Method 300.0, Pfaff et al. 1989).

Secchi disk depth, turbidity, pH, and depth profiles of dissolved oxygen, temperature, and conductivity were collected. Depth profiles were measured with Yellow Springs Instrument meters and turbidity was measured by a Hach model 16800 turbidimeter. Secchi disk depth was measured with a standard 20 cm diameter Secchi disk.

Statistical summaries of water quality parameters were computed and presented as notched box and whisker plots as described by McGill et al. (1978). Although sample size usually is indicated by the width of the box, I chose to use a standard box width and present sample sizes in tabular format (Table XXLII). These plots illustrate the upper and lower limits, first, second, and third quartiles, and an approximate statistical domain at $\alpha \approx 0.05$ (McGill et al. 1978). Statistical domain is used here to indicate the range at which statistical significance is indicated when two or more domains (i.e., notches) do not overlap.

## Hydraulic Budget

A hydraulic budget was estimated based upon 24 hr averages of hourly lake elevation data provided by USACE. General models for elevation/capacity and elevation/area were constructed by nonlinear least squares regression to an exponential model of the form:

$$CAPACITY(ac\text{-}ft) = A*ELEVATION*e^{B*ELEVATION} + E*ELEVA \qquad \text{Equation 26.}$$

and

$$AREA(acres) = A*ELEVATION*e^{B*ELEVATION} + E*ELEVATION \qquad \text{Equation 27.}$$

where, A and B are estimated coefficients and E is an estimated asymptotic coefficient. Capacity data used in the regression were provided by USACE and areal data given in USACE (1993). Hydraulic residence time was computed by dividing the sum of flows at two gauging stations nearest the lake (USGS07196500 - Illinois River near Tahlequah and USGS07197000 - Baron Fork at Eldon) by the estimated capacity for each day. The flows for the stations were downloaded from STORET, while the capacity was estimated based on a 24 hr average of hourly elevation data provided by USACE. Hydraulic balance was assessed by subtracting the inflows (USGS07197000 + USGS07196500) from the outflow (USGS07198000 - Illinois River near Gore).

## Risk Model Description

The reservoir ecological risk factor model is based on the longitudinal water quality gradient in mainstem reservoirs as described above. I fit existing data for total phosphorus, turbidity, and Secchi disk depths to a sigmoidal function (Equations 22, 23, and 24) using least squares nonlinear regression in SYSTAT©. The spatial trend in chlorophyll density was fit via the same method to a maxima function (Equation 25) also using SYSTAT©. Data were fit to the equations using a decimal-scaled thalweg distance as the independent variable. The purpose of scaling the distance was to standardize for reservoir size, thus enabling comparative analysis with other reservoirs of differing sizes. The model was run on the cumulative data set and separately for each sampling event (i.e., selected by date). The model parameters EL50, MIN, and MAX of Equations 22, 23, and 24 were used to infer intrinsic properties of phosphorus fate, transport, and impact. A correlation matrix was created among the abiotic model parameters (i.e., EL50, MIN, and MAX), estimated chlorophyll maximum and its location, and hydraulic conditions (i.e., inflow, outflow, capacity, and instantaneous residence time ($\tau_w$). From these results, a general model that predicts the spatial gradient from hydraulic conditions was constructed.

The maxima function was fit to the longitudinal trend in chlorophyll *a* density and the resulting estimated parameters, CRF and PF, were evaluated. These model parameters with the estimated errors were used as the reference for trophic status. Inference of the trophic status from the model parameters was used because the chlorophyll *a* densities at the various stations indicated the entire range of trophy could be identified. For example, many dates indicated chlorophyll densities > 30 $\mu$g/l (eutrophy) at the mid stations while the lower stations indicated < 5 $\mu$g/l (oligotrophy). Using the model parameters allows a correction for the maxima peak (intensity) and area distribution of trophic status.

Since the PF induces a higher peak, CRF influences the peak breadth, and the effect of one is relative to the other, I opted to integrate both parameters in the overall assessment of reservoir trophic status (Figure 37). The dotted lines represent equivalent chlorophyll *a* maxima and have been drawn at 5, 10, 20, 40, and 80 $\mu$g/l. I chose these values because the traditional boundary of oligotrophy and mesotrophy is approximately 5 $\mu$g/l. The remaining lines of Figure 37 represent a doubling of the chlorophyll maxima. The axes limits on Figure 37 were not intended to be absolute. However, if the power factor is 0 or greater the maxima trend cannot be observed within the 1.0 scaled thalweg distance and hence invalidates the model anyway. Chlorophyll densities greater than 80 $\mu$g/l can produce self-shading and hence induce another inhibitory factor not modelled here (see Talling 1960). If the traditional reference of trophic status is to be inferred, increasing eutrophy is indicated by proceeding right and upward in Figure 37. Boundaries of trophic status depend on subjective judgment.

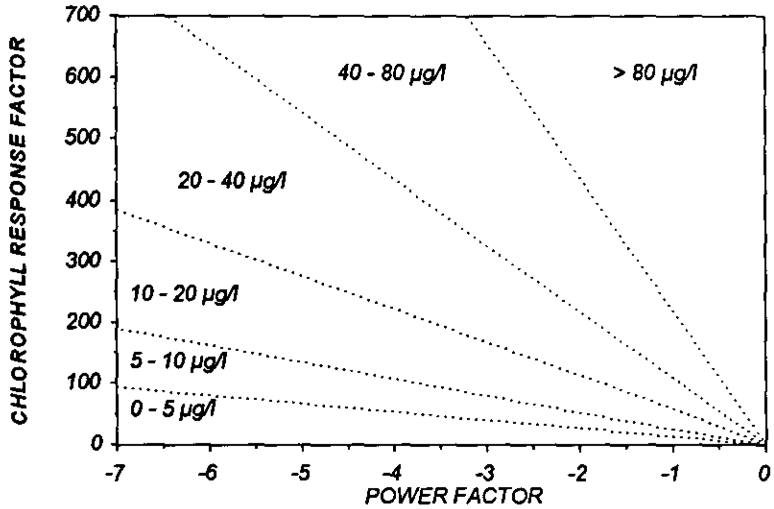



Figure 37. Integration of CRF and PF into Trophic Status Characterization (lines represent isoclines of chlorophyll maxima).

Total Maximum Daily Load

While the model developed here describes the sensitivity of a reservoir to allochthonous loads of phosphorus, external loads must be managed according to the intrinsic sensitivity. Historically, external loads have been calculated on an annual basis. However, recent management strategies have targeted daily loads in lieu of annual loads. I calculated the daily load by dividing the estimated annual load by 365. This total maximum daily load (TMDL) of phosphorus for Lake Tenkiller was calculated by four methods, Vollenweider's (1968, 1975) critical loads, a morphoedaphic index (MEI) method proposed by Vighi and Chiadauni (1985), Reckhow's (1988) model, and applying the assimilative capacity inferred from the RERF where lacustrine conditions were targeted at 0.01 mg P/l for an upper oligotrophic boundary and 0.02 mg P/l the upper mesotrophic boundary ( > 20 $\mu$g P/l indicate eutrophy).

Vollenweider's (1968) model as refined by Vollenweider in (1975, 1976) defined critical levels of areal P loading which he called "dangerous" for the meso/eutrophic boundary and "permissible" for the oligo/mesotrophic boundary in lakes based upon the observed areal hydraulic loading divided by the mean depth. Vollenweider (1976) empirically derived:

$$L\,(P)_{critical} \;=\; A * q_s * \left(1 + \sqrt{\frac{z_{avg}}{q_s}}\right) \qquad \text{Equation 28.}$$

where L(P) is the critical areal loading (g P/m$^2$/yr), A is a coefficient defined by the trophic boundary (10 for oligo/mesotrophic boundary and 20 for meso/eutrophic boundary, $q_s$ is the areal hydraulic loading (m/yr), and $z_{avg}$ is the mean depth. The observed values were calculated and compared to estimated "critical" L(P) values for the observed hydraulic conditions of each sampling event.

A second reference point was calculated based upon the regression of two MEIs (alkalinity and conductivity) on observed inlake P concentration under "no anthropogenic influence" (Vighi and Chiaudani 1985). The equations were:

$$\log[P] = 1.44 + 0.33(\pm 0.10) \log MEI_{alk} \qquad r = 0.87 \qquad \text{Equation 29.}$$

and

$$\log[P] = 0.71 + 0.26(\pm 0.11) \log MEI_{con} \qquad r = 0.72, \qquad \text{Equation 30.}$$

where $MEI_{alk}$ and $MEI_{con}$ are total alkalinity (meq/l)/$z_{avg}$ and conductivity ($\mu$S)/$z_{avg}$, respectively. One can argue that no such background P concentration exists for reservoirs because the "anthropogenic construction" of the reservoir implies influence by default. However, no acceptable alternative method of estimating nonanthropogenic background loadings for reservoirs has been evaluated. Therefore, methods given by Vighi and Chiaudani (1985) were calculated and used for comparative purposes only.

Reckhow's (1988) model of trophic state for southeastern reservoirs was used to simulate nutrient reduction effects. In the process, the percent reduction required for a downstream total P concentration of 10 $\mu$g P/l was used as an oligotrophic target. From this percent reduction, a distribution of TMDLs was calculated as estimated percent reduction in current daily load. Whether or not the reduction in load equates to a similar reduction in downstream concentration is unknown and should be monitored.

Finally, a total maximum daily load was estimated according to the lake's ability to assimilate the load. In the sigmoidal equation (Equation 22), the difference in the upper and lower asymptotes provided an estimate of the reservoir's assimilative capacity in the transition zone. Based upon this assumption, I calculated a TMDL for total P at the headwaters of Lake Tenkiller that would yield a lacustrine (i.e., station 7) concentration of 10 and 20 $\mu$g P/l for the upper boundaries of oligotrophic and mesotrophic conditions, respectively. These limits are in accordance with currently accepted boundaries as applied to P-limited phytoplankton in natural lakes. I exclusively applied the limits to the lacustrine zone of the reservoir because it most emulates the natural lake ecosystem, most frequently P-limited (Haraughty 1995), and represents the portion of the reservoir most sensitive to P-availability. Additionally, Jobe (1991) found an improved agreement among Carlson's (1977) TSIs in Grand Lake, Oklahoma when only data from the lacustrine zone were used (i.e., data from the riverine and transition zones introduced more error among the TSIs than did lacustrine data).

## Results and Discussion of Limiting Nutrient Inflows and Outflows

### Current Conditions in Water Quality

Statistical summaries of water quality conditions from the 1974 EPA-NES study (USEPA 1977), 1985-86 USACE study (USACE 1988), and the current 1992-93 Clean Lakes Program (CLP) study were compared for temporal and spatial trends. Observed trends indicated a temporal increase in total P from 1974 to 1985 and then a decrease in 1992. Hypolimnetic total P has also increased since 1974, but it is difficult to determine if this is due to an artifact of underflow of higher headwater loads or P recirculation resulting from the apparent accelerated eutrophication. Differences in epilimnetic total N and nitrate N were not statistically significant. Chlorophyll *a* in the midlake region has increased by almost three times since 1974, but apparently did not affect the Secchi disk tranparencies.

Epilimnetic total P indicated the expected sigmoidal trend in the 1985-86 USACE data and the 1992-93 CLP data, but could not be ascertained from the 1974 NES data (Figure 38). Two possiblities exist. The NES study did not include a "critical" headwater station as did the later two studies and did not have as many samples (Table XLII). However, an apparent decrease in total P at Horseshoe Bend has occurred since 1985-86 from a median of 209 $\mu$g P/l in 1985-86 to 122 $\mu$g P/l in 1992-93 which was statistically significant (Figure 38). The most plausible explanation is the city of Tahlequah (located just above the headwaters) implemented a P control sewage treatment plant in the interim.

Also, a comparison of epilimnetic total P among the EPA-NES station 4, USACE station 13, and CLP station 2 (the locations were approximately the same) showed a similar trend. For these stations, the median of 48 $\mu$g P/l in 1974 increased to 173 $\mu$g P/l in 1985-86 and then decreased to 98 $\mu$g P/l in 1992-93. All were statistically significant (Figure 38).

Finally, epilimnetic total P trends at the most downstream station (i.e., EPA-NES station 1, USACE station 1, and CLP station 7) showed a slightly different trend. These data, located near the dam, showed an increase from a median value of 42 $\mu$g P/l in 1974 to 98 $\mu$g P/l in 1985-86 and then decreased to 24 $\mu$g P/l in 1992-93. All were statistically significant (Figure 38).

The sample size of the EPA-NES study was only four. Yet, these four data represented both low-flow and above average flow conditions and thus should have approximated the true range. All median values for all three studies exceeded the suggested surface values of 20 $\mu$g P/l during the summer.

Hypolimnetic total P trends showed a similar temporal trend, but also showed more spatial variation, especially at the midlake stations (Figure 39). I hypothesize the highest values at the midlake region is a result of P recirculation during stratification. Current dogma contends that anoxia develops first in the transition zone because it has the highest organic load and a lower hypolimnetic volume (Thornton et al. 1990). Therefore, the duration when redox potentials are favorable to P recirculation is longer in this region and thus allows for more accumulation (Thornton et al. 1990). However, none of the midlake stations were significantly different than the headwater station (i.e., Horseshoe Bend).

Table XLII. Sample Sizes of Referenced Studies on Lake Tenkiller.

| | | EPILIMNION | | | | | | HYPOLIMNION | |
|---|---|---|---|---|---|---|---|---|---|
| Study | Site | TP | TURB | TN | $NO_3$ | SD | CA | TP | TURB |
| NES | 1 | 4 | ND | 4 | 4 | 4 | 4 | 4 | ND |
| | 2 | 4 | ND | 4 | 4 | 4 | 4 | 4 | ND |
| | 3 | 4 | ND | 4 | 4 | 4 | 4 | 4 | ND |
| | 4 | 4 | ND | 4 | 4 | 4 | 4 | 4 | ND |
| USACE | 1 | 10 | 16 | ND | 10 | 13 | 14 | 8 | 9 |
| | 2 | 10 | 16 | ND | 10 | 14 | 14 | 7 | 8 |
| | 3 | 10 | 16 | ND | 10 | 15 | 14 | 8 | 9 |
| | 4 | 10 | 16 | ND | 10 | 13 | 14 | 7 | 8 |
| | 5 | 10 | 16 | ND | 10 | 14 | 14 | 7 | 8 |
| | 6 | 10 | 16 | ND | 10 | 14 | 14 | 7 | 8 |
| | 7 | 10 | 15 | ND | 10 | 14 | 14 | 7 | 7 |
| | 8 | 10 | 15 | ND | 1 | 15 | 13 | 5 | 6 |
| | 9 | 9 | 15 | ND | 1 | 15 | 13 | 6 | 7 |
| | 10 | 9 | 16 | ND | 9 | 15 | 12 | 2 | 2 |
| | 11 | 9 | 15 | ND | 9 | 14 | 13 | 3 | 4 |
| | 13 | 9 | 15 | ND | 9 | 14 | 13 | 9 | 2 |
| | 14 | 9 | 15 | ND | 9 | 14 | 13 | ND | ND |
| CLP | 1 | 16 | 14 | 16 | 16 | 1 | 16 | --[1] | --[1] |
| | 2 | 17 | 14 | 17 | 17 | 18 | 22 | 17 | 14 |
| | 3 | 17 | 14 | 17 | 18 | 18 | 18 | 17 | 14 |
| | 4 | 17 | 14 | 17 | 18 | 18 | 18 | 17 | 14 |
| | 5 | 17 | 14 | 17 | 18 | 18 | 22 | 17 | 14 |
| | 6 | 17 | 14 | 17 | 18 | 18 | 22 | 17 | 14 |
| | 7 | 17 | 14 | 17 | 18 | 19 | 18 | 17 | 14 |

ND = no data available.
[1] = No stratification observed at Station 1 thus limnetic layers were not differentiated.

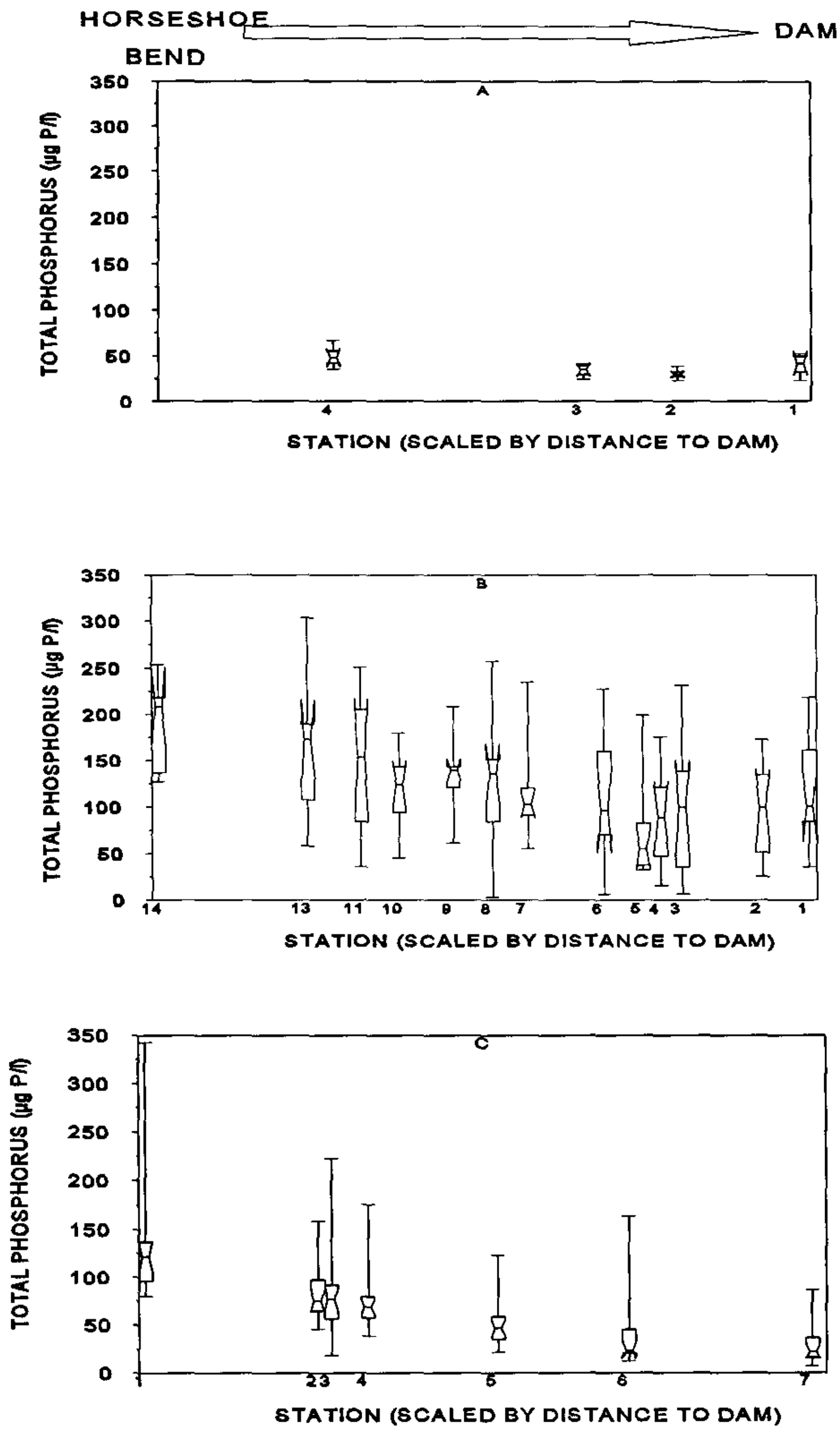



Figure 38. Epilimnetic Total P Concentrations in Lake Tenkiller for (A) EPA-NES in 1974, (B) USACE in 1985-86, and (C) CLP Study 1992-93.

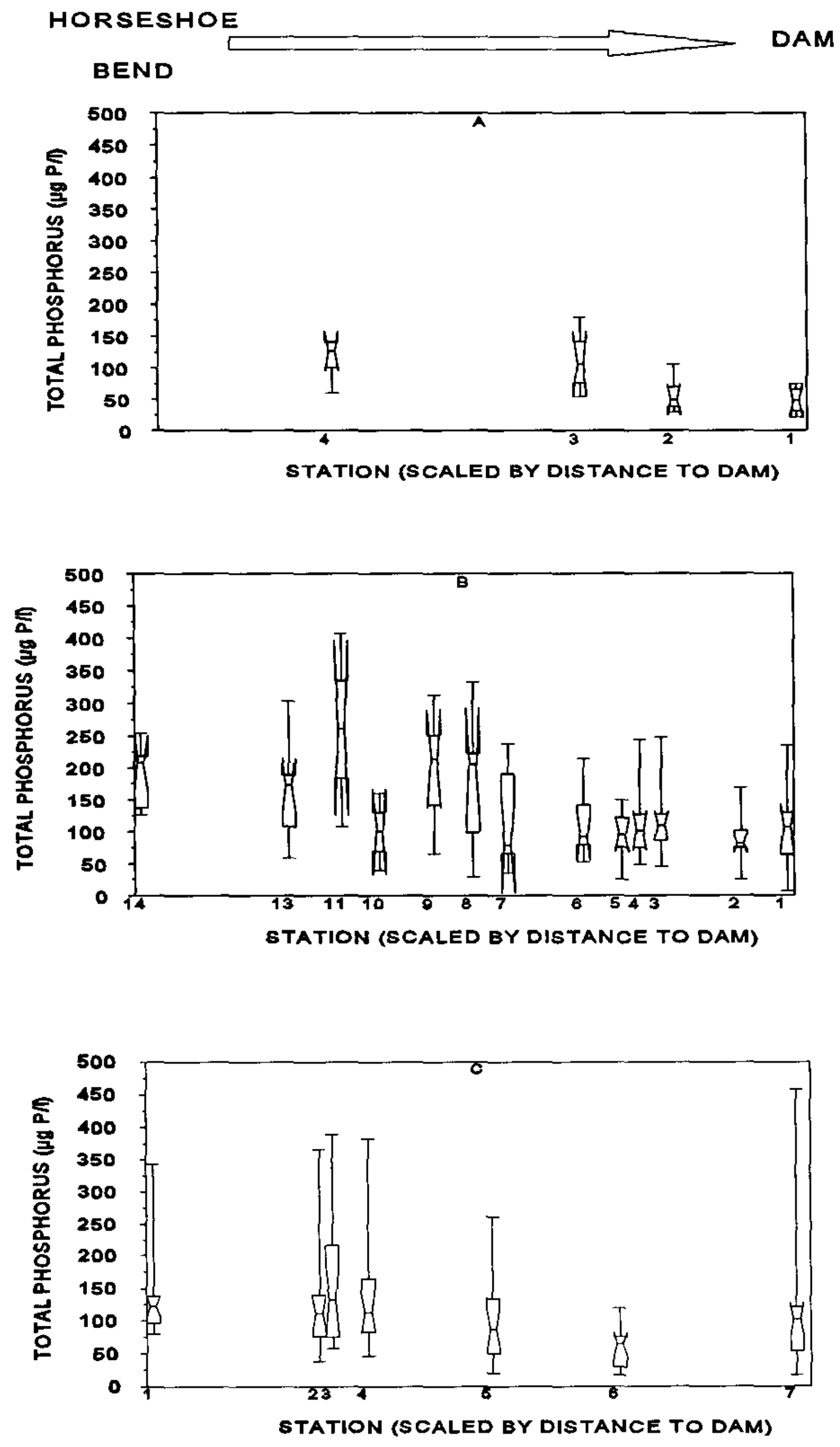



Figure 39. Hypolimnetic Total P Concentrations in Lake Tenkiller for (A) EPA-NES in 1974, (B) USACE in 1985-86, and (C) CLP in 1992-93.

Although nitrogen fractions were not incorporated into the actual model, the relevance of nitrogen to eutrophication of lakes cannot be overlooked. The common assumption to most, if not all, phosphorus-based eutrophication models and indices is phosphorus-limited algal growth. Sakamoto (1966) found this to be true only for values of TN:TP greater than approximately 12. Lakes usually show symptoms of nitrogen-limitation as a result of being P replete and will revert to phosphorus-limitation if the phosphorus supply is reduced sufficiently. Critical concentrations have been proposed by Sawyer (1947) as 300 and 600 $\mu$g N/l for oligo/mesotrophic and meso/eutrophic boundaries, respectively. Therefore, while nitrogen was not incorporated into the RERF model, the data were evaluated and discussed.

In the CLP study, inorganic nitrogen (nitrate-N) in the epilimnion showed an initial decrease in concentration from a median of 1170 $\mu$g N/l at station 1 to 461 $\mu$g N/l at station 2 with little decrease through the remaining stations (Figure 40). The USACE study in 1985-86 showed a different trend where the median nitrate N at Horseshoe Bend was 355 $\mu$g N/l and increased to 1000 $\mu$g N/l at the next downstream station. Following this increase, nitrate levels showed an overall decrease with a relatively high variability. The EPA-NES study in 1974 showed epilimnetic nitrate N concentrations similar to to that observed in 1992-93 (Figure 40).

The only comparisons of total N trends were between the EPA-NES 1974 study and the current CLP 1992-93 study; total N data for the USACE study were not available for comparison. Total N trends, like nitrate N, showed a similar trend in 1992-93 decreasing from a median of 2180 $\mu$g N/l at station 1 to 1160 $\mu$g N/l at station 2 (Figure 41). Median values of epilimnetic total N continued to decrease to station 5 and stabilized thereafter. These 1992-93 median values were not significantly different than those observed in the 1974 EPA-NES study. All median values observed exceeded the eutrophic criteria proposed by Sawyer (1947).

The implication of total N content in lake eutrophication is relative to the amount of phosphorus (see Sakamoto 1966). The philosophy of using the atomic ratio total N:total P was discussed earlier. The 1992-93 median total N:toal P trend observed in Lake Tenkiller exceeded threshold criteria proposed by Vollenweider (1968) and Sakamoto (1966) and thus suggests P-limitation (Figure 42). Phosphorus limitation in Lake Tenkiller during 1992-93 also was indicated by algal assays (Haraughty 1995). Therefore, nitrogen-limitation was not implied, nor was it detected in these nutrient limitation assays. Close examination of Figure 42 illustrates that at times Lake Tenkiller may exhibit total N:total P ratios $\leq$ approximately 15 as far downstream as station 6. Therefore, the lake could exhibit undesirable conditions of blugreen dominance if P loads continue to increase and shift these distributions downwards.

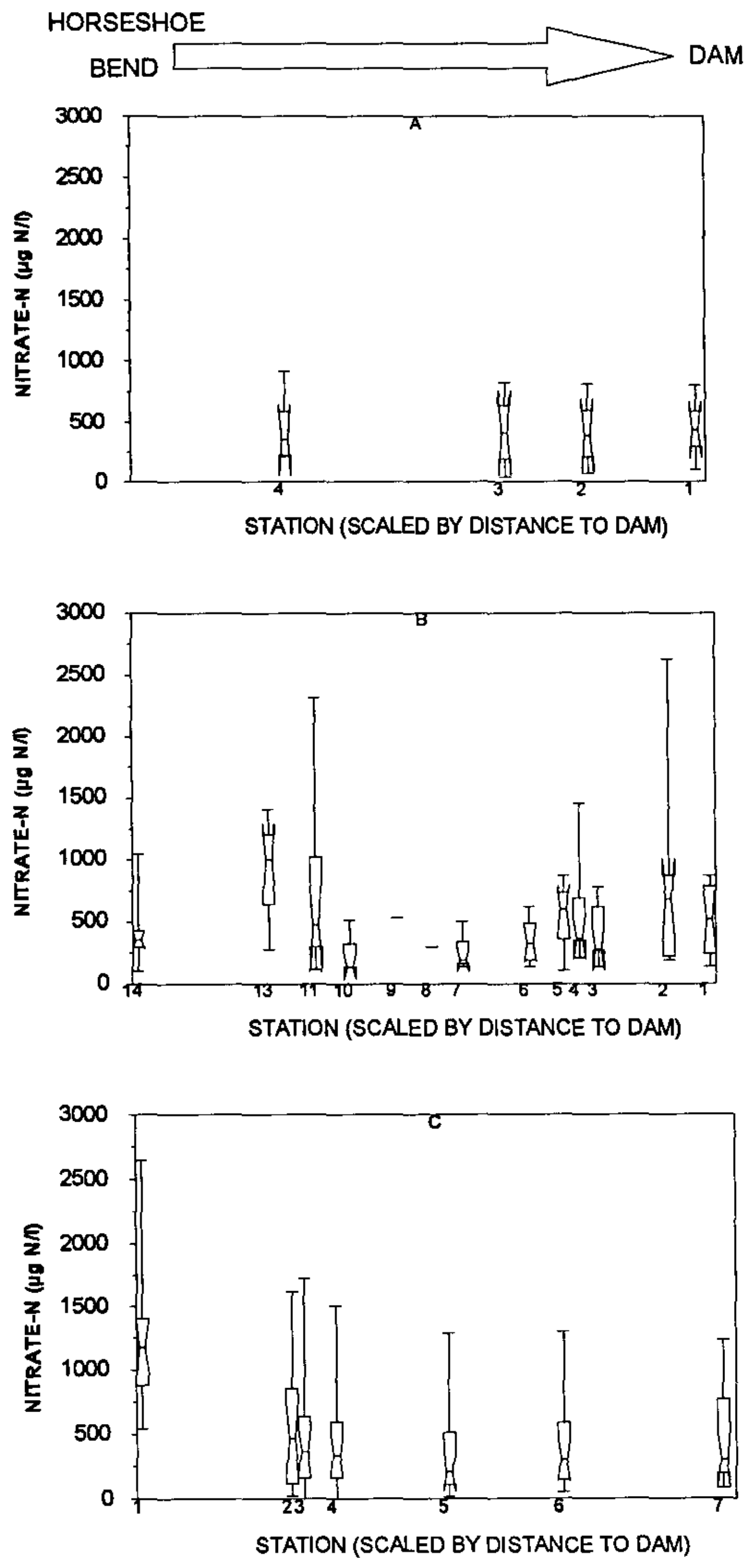



Figure 40. Epilimnetic Nitrate N Concentrations in Lake Tenkiller for (A) EPA-NES in 1974, (B) USACE in 1985-86, and (C) CLP in 1992-1993.

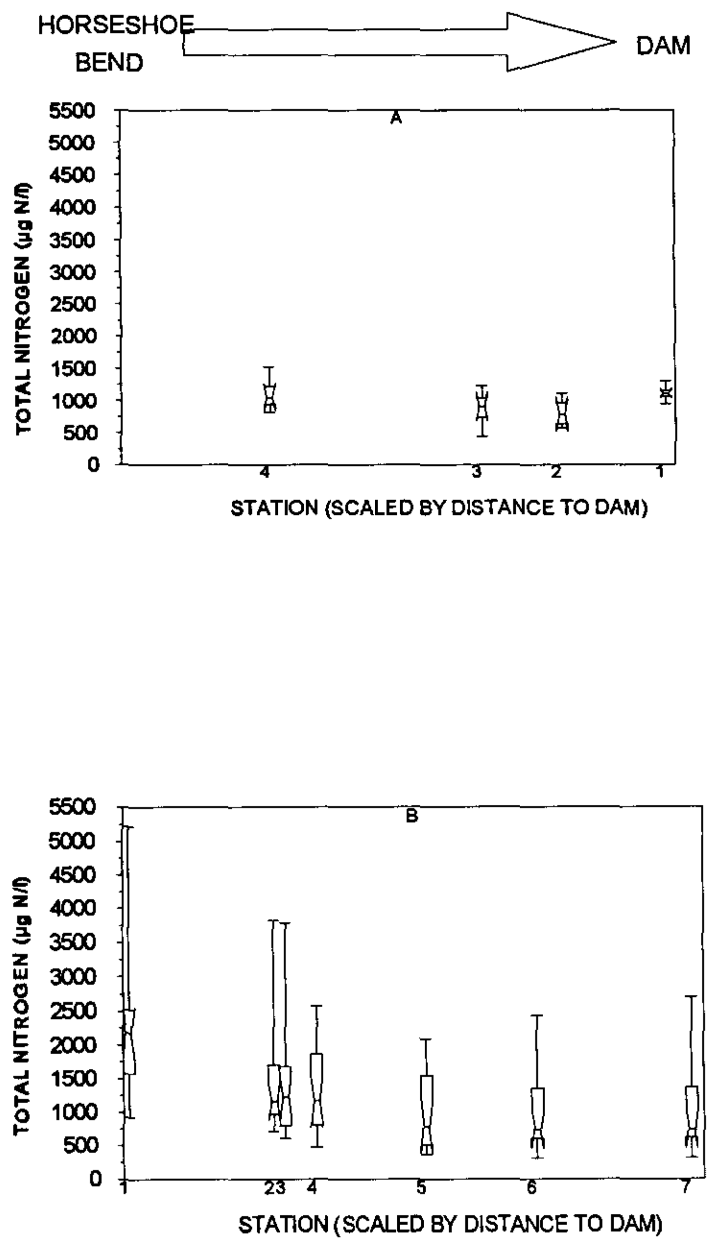



Figure 41. Epilimnetic Total N Concentrations in Lake Tenkiller for (A) EPA-NES in 1974 and (B) CLP in 1992-93.

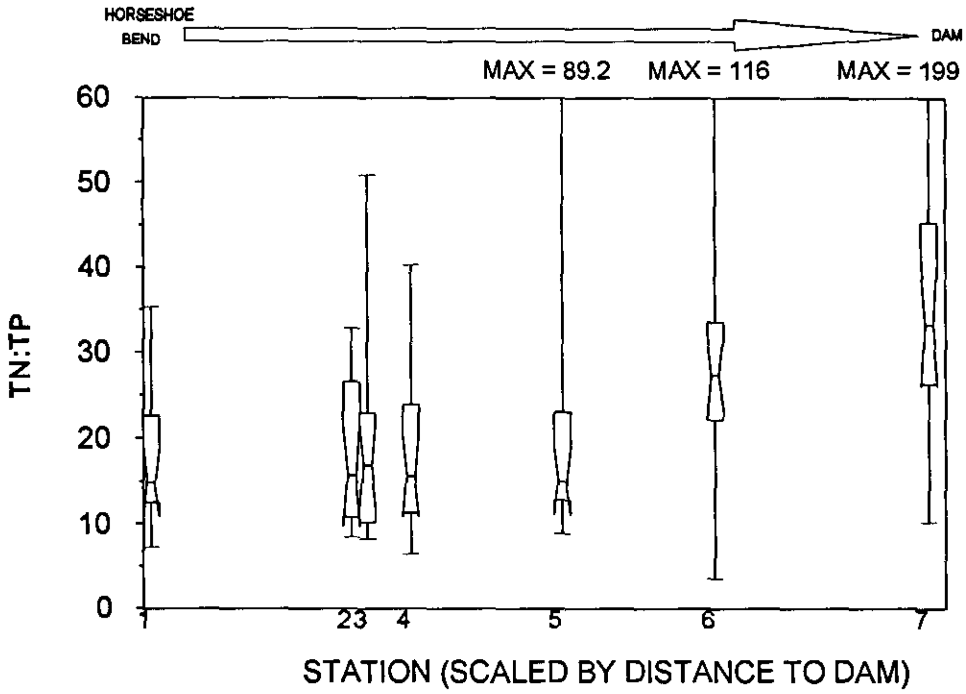



Figure 42. Total N:Total P Trends in Lake Tenkiller for CLP Study of 1992-93.

Turbidity trends were compared for USACE 1985-86 and CLP 1992-93 studies only because no comparable turbidity data for the EPA-NES 1974 study was available. The epilimnetic turbidity trend for both studies were similar. These trends exhibited a sigmoidal spatial trend in which the maximum turbidity was at Horseshoe Bend and had a maximum decay at approximately 1/3 to 1/2 the thalweg distance (i.e., CLP stations 2-4 and USACE stations 11-8) (Figure 43). Absolute epilimnetic turbidities were not significantly different between the two studies.

Hypolimnetic turbidities exhibited a strong contrast between the two studies. The spatial trend in the USACE 1985-86 study indicated a sigmoidal trend as was detected in the epilimnion while the CLP 1992-93 study showed an opposite trend in which turbidity increased with thalweg distance (Figure 44). I speculate the extreme turbidity at station 7 of the CLP study is due partially to effects of hypolimnetic withdrawal. The increase in turbidity along the thalweg at the other stations might be due to underflow. If underflow is the predominant direction of influent water, these flows coupled with hypolimnetic withdrawal conceivably could increase turbidity as it approaches the dam because hypolimnetic sediments become less consolidated with thalweg distance (these are the fine silts).

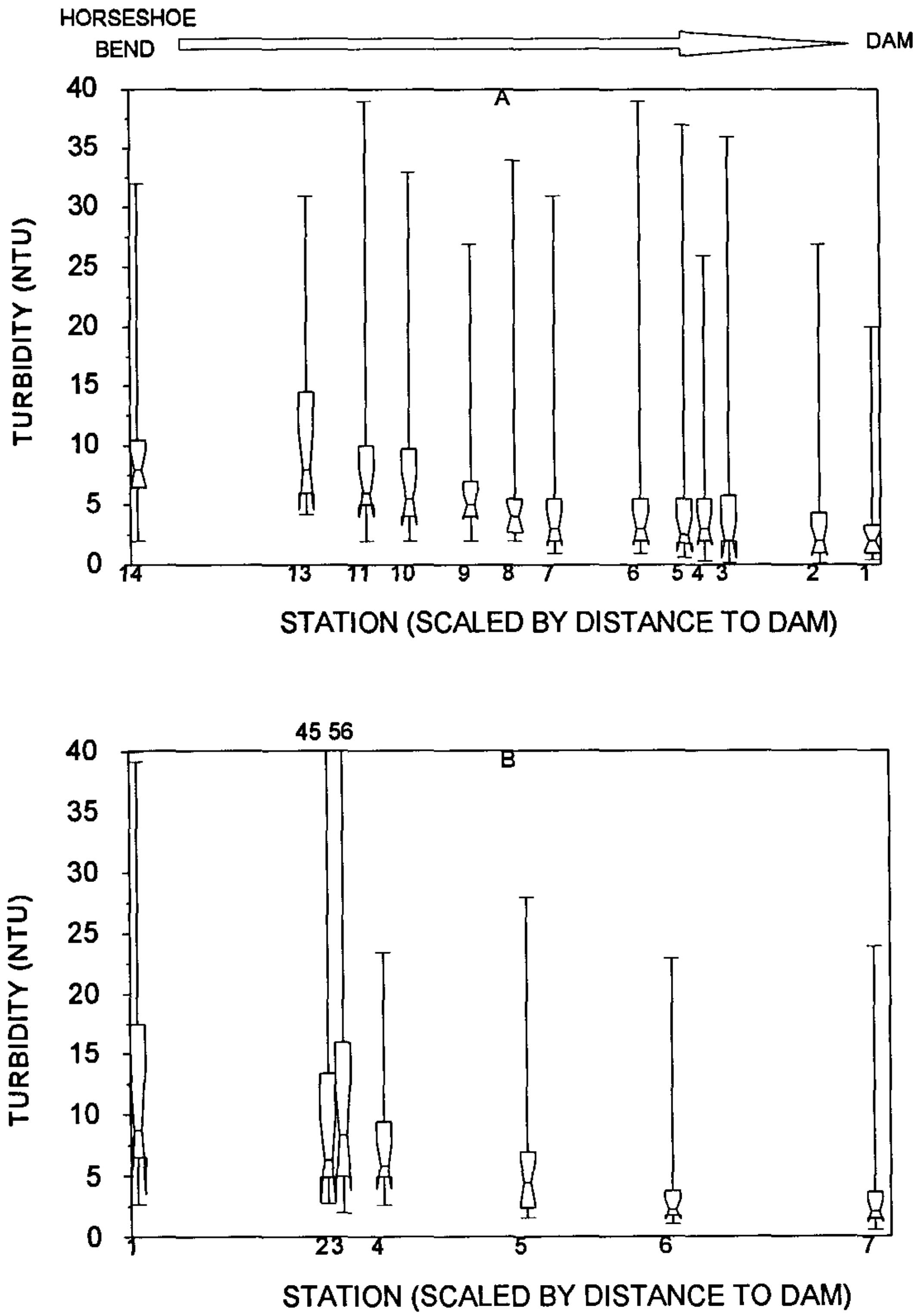



Figure 43. Epilimnetic Turbidity Trends in Lake Tenkiller for (A) USACE in 1985-86 and (B) CLP 1992-93.

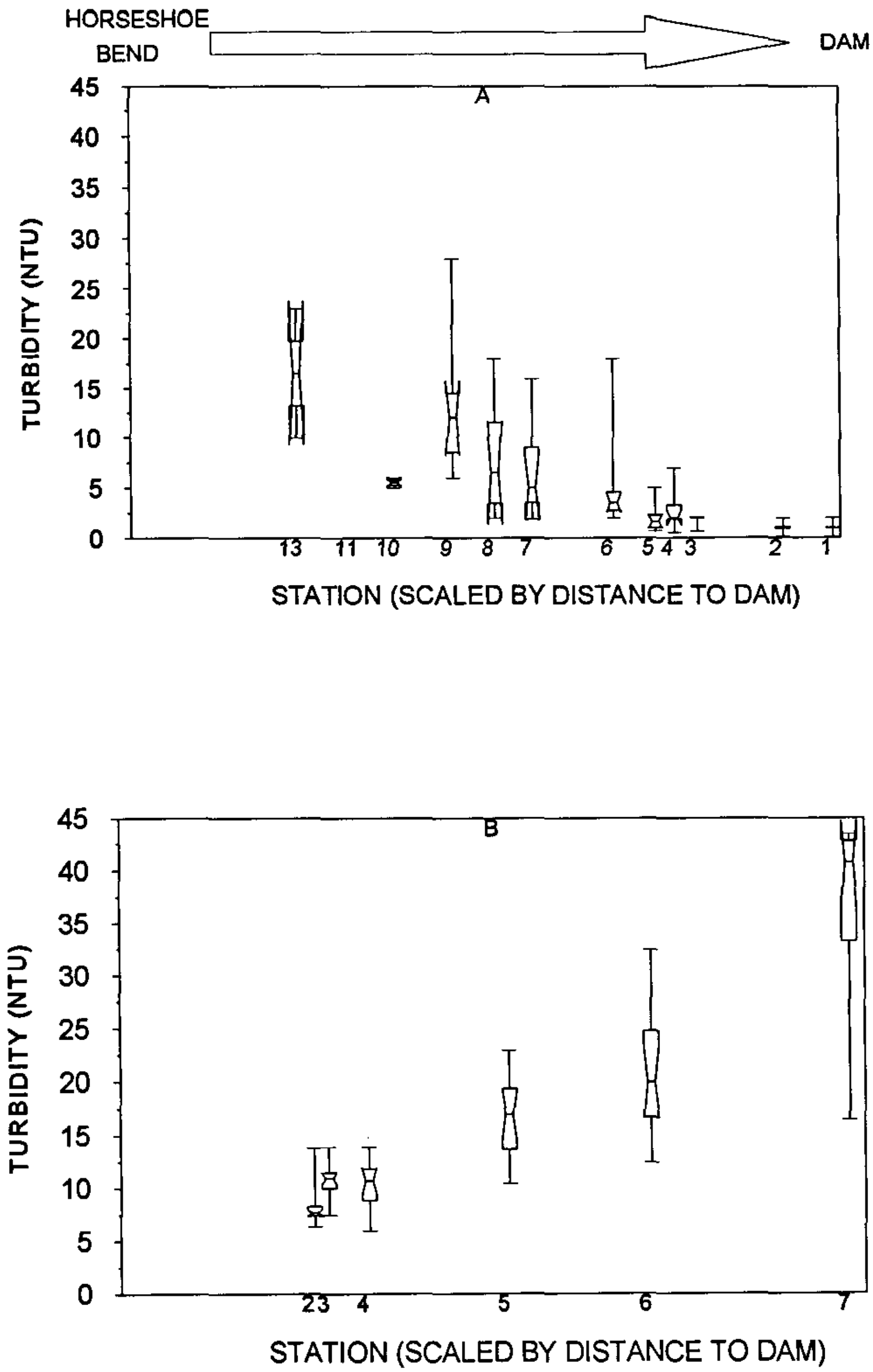



Figure 44. Hypolimnetic Turbidity in Lake Tenkiller for (A) USACE in 1985-86 and (B) CLP in 1992-93.

Secchi disk depths showed a spatial sigmoidal trend for all three studies where the lowest Secchi disk depths were observed in the headwaters and increased to a maximum near the dam (Figure 45). The largest increases in depth along the thalweg were slightly downstream from the largest decrease in turbidity (cf. Figures 43 and 45). No significant temporal trend was detected; in fact the median Secchi disk depths were generally larger in 1992-93 and 1985-86 than in 1974 (Figure 45). However, the variability and the sample size of the EPA-NES 1974 study might limit the value of such generalizations.

The observed trends in Secchi disk depths given in Figure 45 are not surprising when you consider the close association of transparency (i.e., Secchi disk depth) and turbidity. Both have been evaluated here because; 1) although closely related, transparency and turbidity are not measures of the same property and 2) for reduction in model error a model parameter descriptive of light availability with less variability was desired.

In 1992-93, chlorophyll *a* exhibited a maxima relationship with a median concentration of 28.6 $\mu$g/l at station 2. Because station 2 was located in the mouth of Caney Creek (Figure 13), the peak may have been due to a "cove effect" of the tributary and thus did not represent a main thalweg effect. However, station 4 was in the mainstem thalweg (Figure 13) and had a median chlorophyll *a* of 28.7 $\mu$g/l which was not significantly different than station 2 or 3 (Figure 46). Station 1 showed the lowest median chlorophyll *a* density of all stations (excluding station 8, the tailrace) with a median density of 2.5 $\mu$g/l (Figure 46).

In 1985-86, chlorophyll *a* trends were similar to that observed in 1992-93 where comparable stations were slightly higher in 1985-86, but the differences were not statistically significant (Figure 46). A statistically significant increase in chlorophyll *a* density at the point of maximum density and near the dam has occurred in Lake Tenkiller since the 1974 EPA-NES study (Figure 46). Interestingly, an increase in variance of the chlorophyll densities also has occurred (Figure 46). I believe this is due to the increased amplitude of total P loads associated with runoff (cf. Figures 38 and 46). Also, a characteristic of increasing stress is an increased variance in biotic structure (Barrett et al. 1976, Odum 1975, Odum 1979).

In summary, Lake Tenkiller exhibited symptoms of accelerated eutrophication because:

1) total P loads have significantly increased since 1974 while corresponding increases in N fractions were not detected, and

2) chlorophyll *a* densities along the thalweg have increased significantly since 1974.

Evaluating this trend quantitatively and incorporating an element of risk follows with construction of the hydraulic budget and RERF model development.

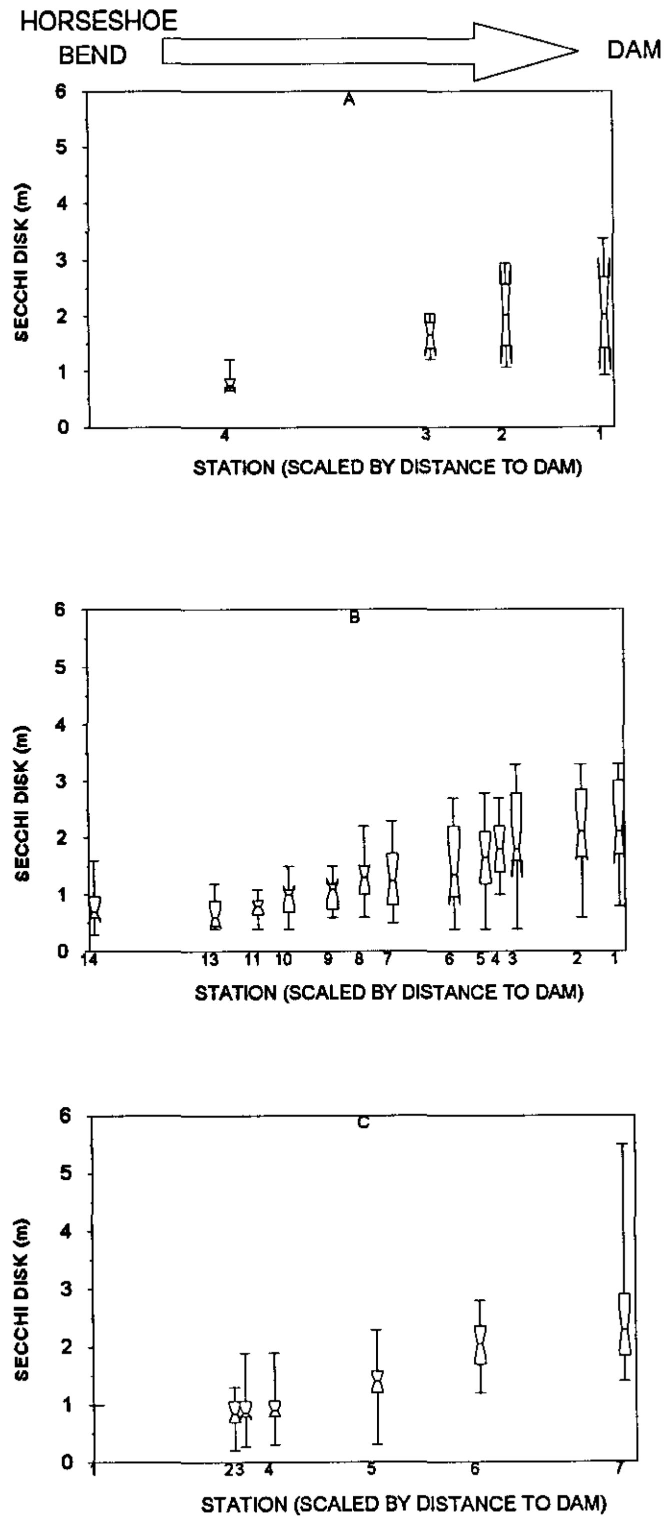



Figure 45. Secchi Disk Depths in Lake Tenkiller for (A) EPA-NES in 1974, (B) USACE in 1985-86, and (C) CLP in 1992-93.

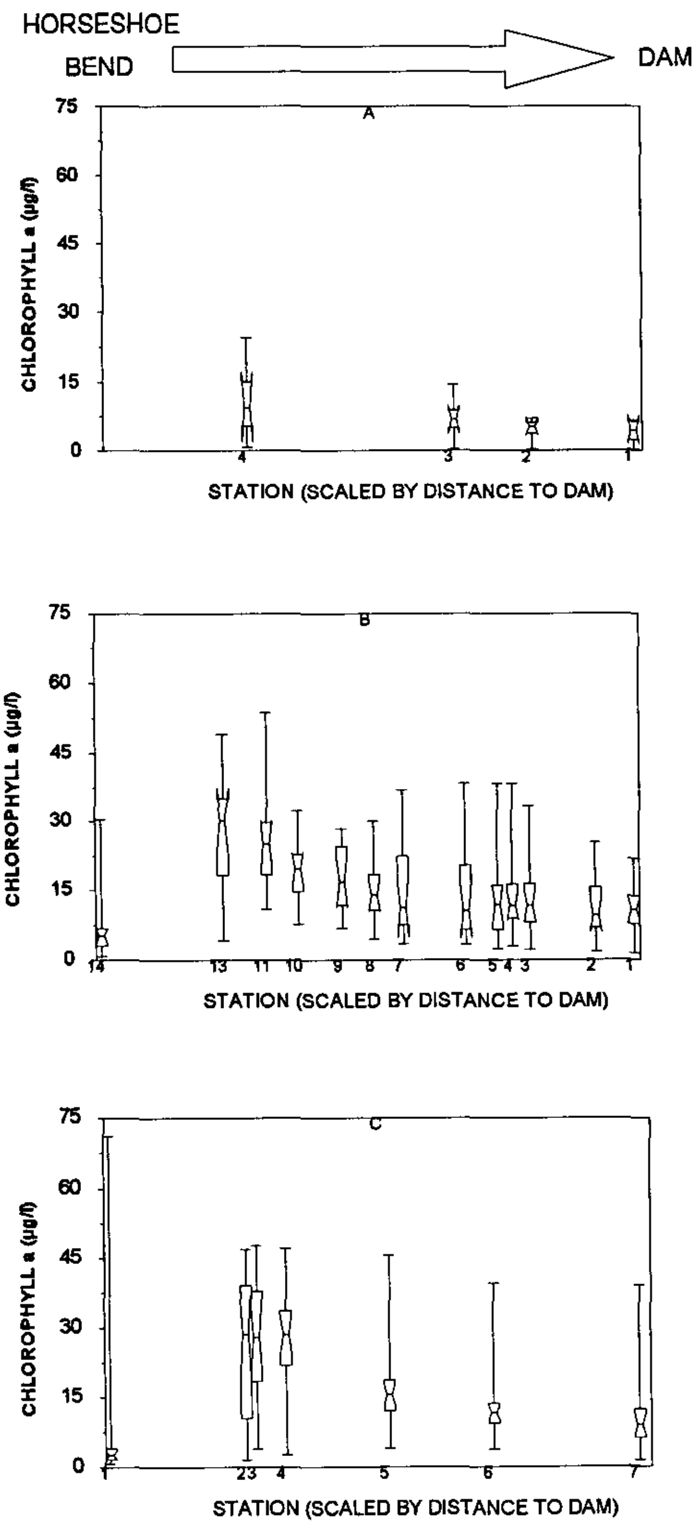



Figure 46. Chlorophyll *a* Densities in Lake Tenkiller for (A) EPA-NES in 1974, (B) USACE in 1985-86, and (C) CLP in 1992-93.

## Hydraulic Budget

An empirical analysis indicated that capacity and area could be estimated from lake elevation. The empirical relationship derived was based upon data provided by USACE. Nonlinear regression for capacity on elevation yielded:

$$CAPACITY = 197.2 * e^{0.135 * ELEVATION} - 527.8 * ELEVATION \qquad \text{Equation 31.}$$

where CAPACITY is in ac-ft and ELEVATION is ft msl. Area was regressed on elevation and yielded:

$$AREA = 2.195 * e^{0.0137 * ELEVATION} \qquad \text{Equation 32.}$$

where AREA is in acres and ELEVATION is ft msl.

Hydraulic residence time varied from 0.033 to 3.4 yr with a mean of 0.76 yr (Figure 47). The maximum residence times were observed during ca. Aug - Sep 92 and Aug 93, while the minimum residence times were observed in early 1993 (Figure 47). Undoubtedly, the observed trend was due to seasonality of runoff and the 1993 spring flood. The implication of the variability in residence time (more than trebled) to the development of a longitudinal transport model (RERF) is that decreased residence time indicates increased flow through the reservoir and thus would be expected to "push" the transition zone towards the dam. The daily hydraulic balance approximated equilibrium conditions except for ca. May - Jun 92 and ca. Dec 92 - Jun 93 (Figure 48). An individual peak in Figure 48 did not necessarily indicate the reservoir was in a state of disequilibrium because dam discharge schedules may dictate no flow downstream for 24 hr. The breadth of the peaks in Figure 48 is more indicative of the reservoir as being "filled" or "emptied". However, comparison of Figures 48 and 49 indicated that the four largest peaks of inbalance (Figure 48) immediately preceded the largest increases in lake elevation. This trend is discussed further in evaluation of the RERF model.

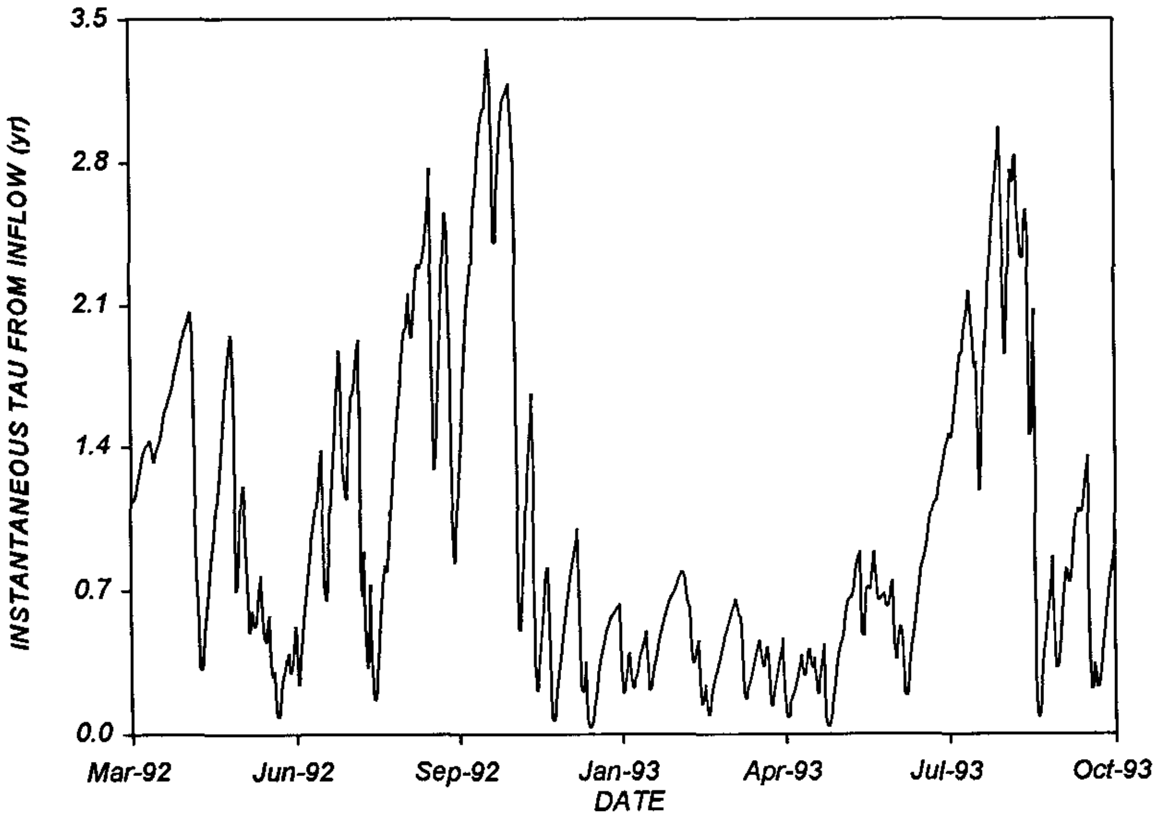



Figure 47. Daily Hydraulic Residence Time for Lake Tenkiller, CY 92-93.

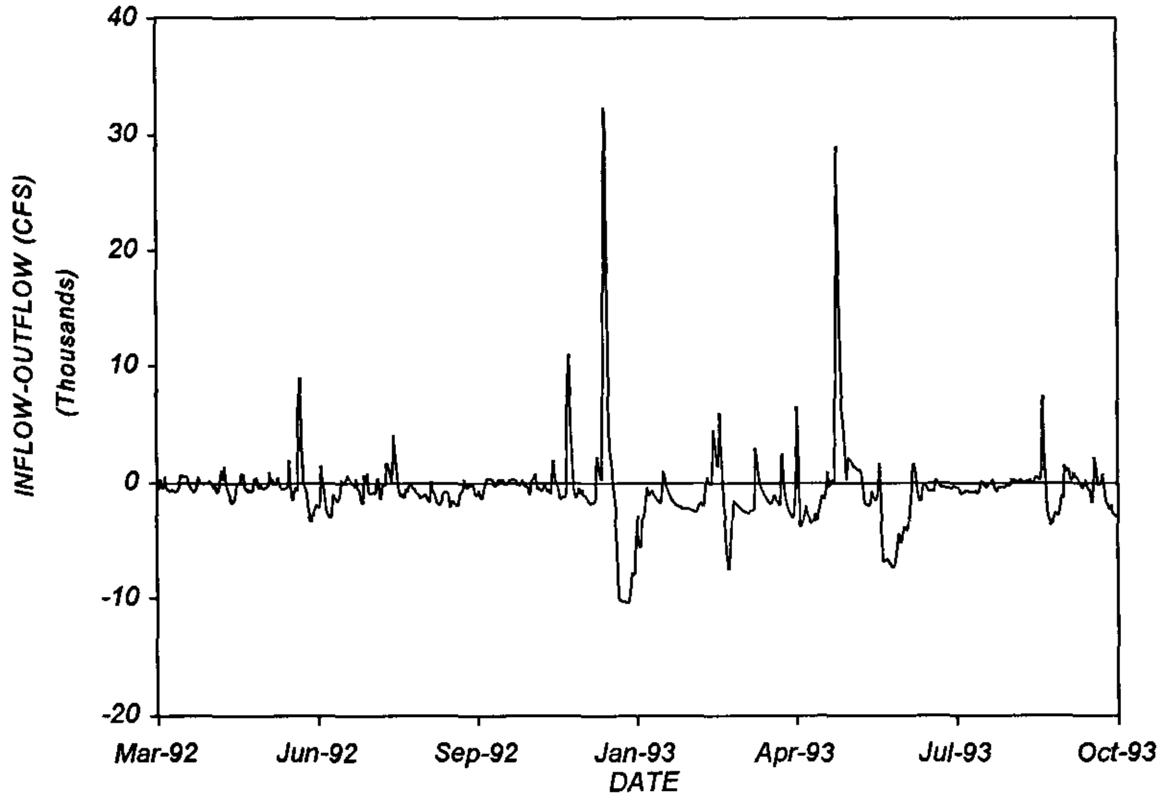



Figure 48. Hydraulic Balance of Lake Tenkiller for CY 92-93.

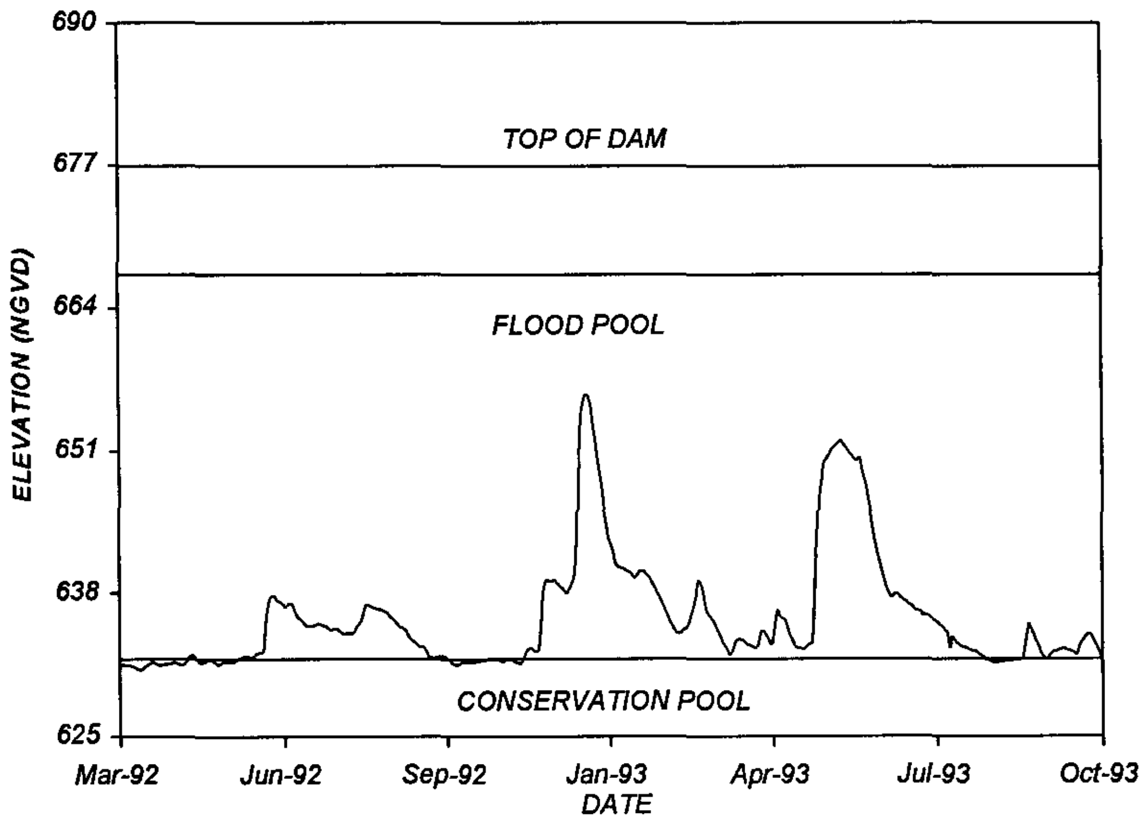



Figure 49. Daily Elevations (NGVD) of Lake Tenkiller for CY 92-93 (24 hr means based upon hourly data provided by USACE).

## Reservoir Ecological Risk Factor Model

### Nonlinear Regression Results

The TP sigmoid equations fit to the data collected during the CLP project (CY 1992-93) yielded an EL50 of 29% (95% confidence interval = 21 to 38%) of the thalweg distance, a slope factor of -7.04 (95% confidence interval = -11.8 to -2.3), and an estimated MIN of 33 $\mu$g P/l (95% confidence interval = 16.3 to 49.8) (Figure 50). The variance in measured TP decreases nearer the dam (Figure xx). The smaller model variance in the upper end is an artifact of setting the median TP at station 1 as the absolute MAX in the equation. This constraint was necessary because initial best fit equations estimated a MAX TP much greater than any observed in the lake or in the immediately-upstream gauging stations (USGS07196500 or USGS07197000) and thus was considered to be unrealistic.

Nonlinear regression results for the USACE 1985-86 data yielded a MIN = 93.3 $\mu$g P/l (95% confidence interval = 65.8 to 120), a slope factor = - 5.339 (95% confidence interval = -9.78 to -0.893), and an EL50 = 35.7% of the thalweg distance (95% confidence interval = 20.8 to 50.6) (Figure 50). The higher minimum estimated for the 1985-86 data indicated a decrease in total P at the lower end of the lake by 1992-93. However, the model maximum for 1985-86 data was set at 211 $\mu$g P/l (median at Horseshoe Bend) which is almost twice that during 1992-93 (130 $\mu$g P/l). The relationship illustrates that total P levels near the dam will respond to decreases in headwater loads. This is an important result for recommended management in Lake Tenkiller.

Turbidity data from the USACE 1985-86 study fit to the sigmoidal equation yielded a MIN = 2.223 nephelometric turbidity units or NTU (95% confidence interval = 11.9 to 0 NTU), a slope factor of -8.438 (95% confidence interval = -23.9 to -7.0), and an EL50 = 80% (95% confidence interval = 29.3 to >100%) of the thalweg distance. The MAX was set to 8 NTU (the median of the headwater station). These best fit results given in Figure 51 - panel A appear to overestimate the true trend indicated by the error bars (quartiles). However, comparison of Figure 51 - panel A with Figure 43 illustrates that extreme outliers (maximum turbidities) during the USACE study biased the estimates of the regression. The RERF is based on chlorophyll *a* trends and not quantitative turbidity; thus data "culling" was not performed.

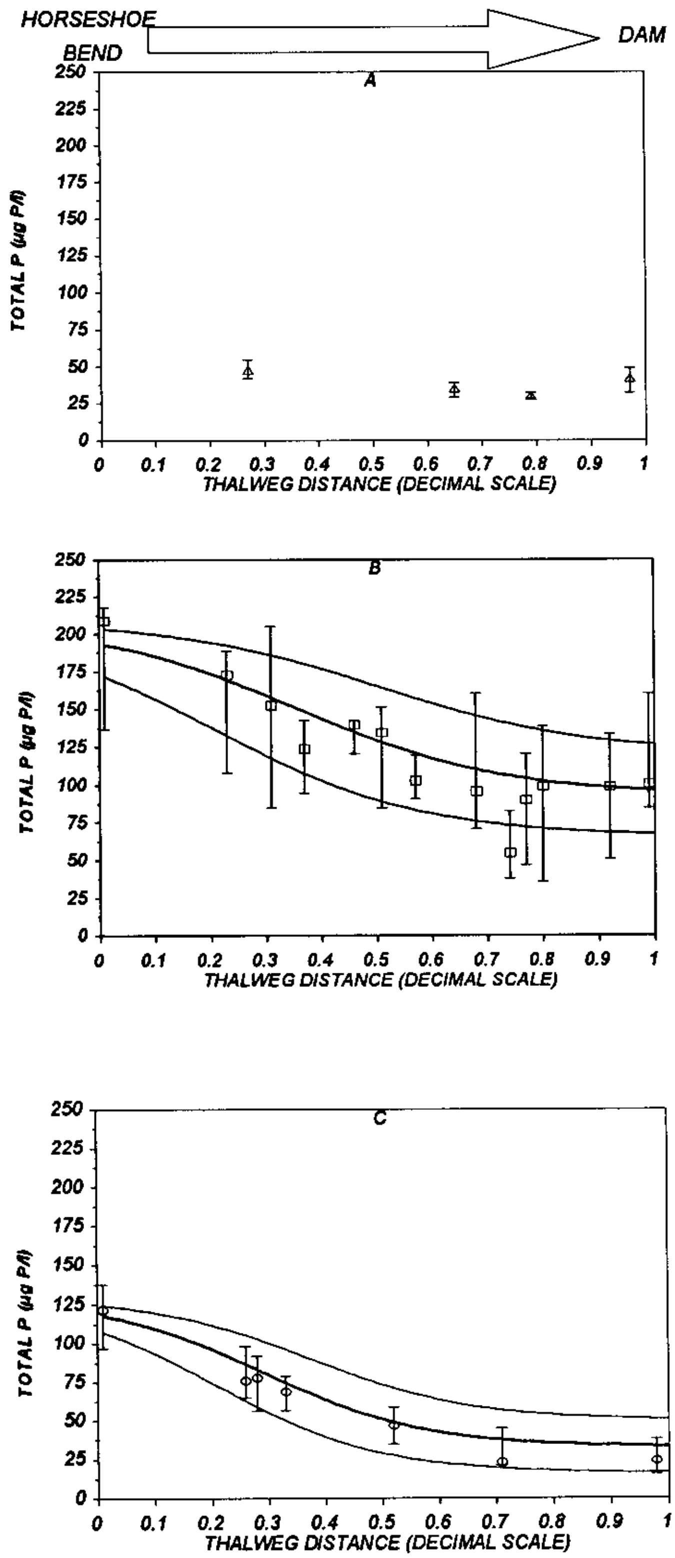



Figure 50. Nonlinear Regression Results for Epilimnetic Total P; (A) 1974 (data only), (B) 1985-86, (C) 1992-93 (lines = ±2 SE of model; error bars = quartiles).

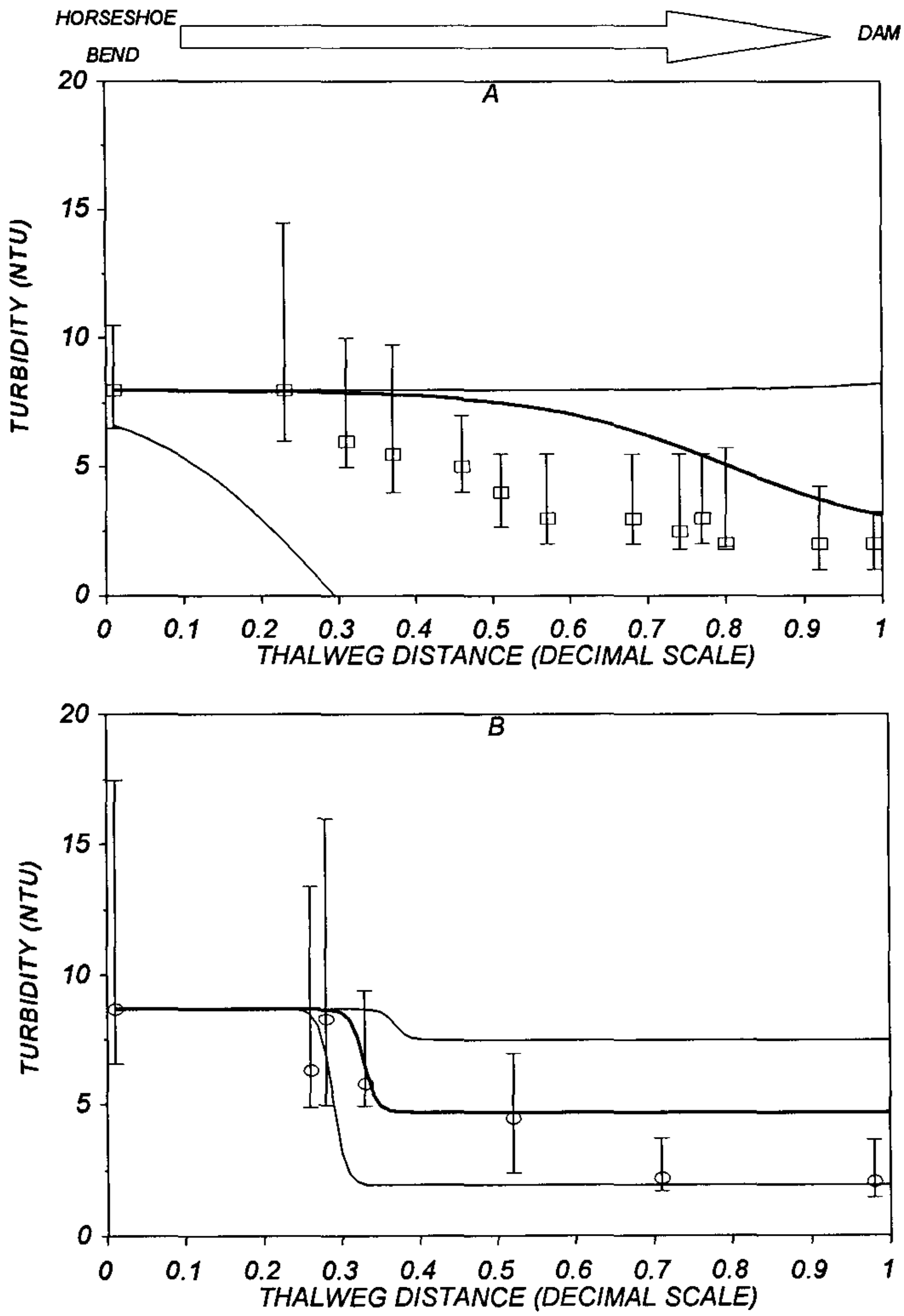



Figure 51. Nonlinear Regression Results for Turbidity; (A) 1985-86, (B) 1992-93; (lines = $\pm 2$ SEM of model; error bars = quartiles).

Turbidity data for the CLP 1992-93 study fit to the sigmoidal equation yielded an EL50 of 33% (95% confidence interval = 29 to 37%), a slope factor of -122 (95% confidence interval = -1260 to 1020), and a MIN of 4.73 nephelometric turbidity units or NTU (95% confidence interval = 1.99 to 7.47 NTU; Figure 51). The slope factor's confidence interval may seem wide but, considering the parameter approaches infinity as the slope approaches a vertical line, the estimate is realistic. Therefore, the high slope factor merely indicates a rapid decline in turbidity near the EL50 (Figure 51). For this reason, the error estimates of the slope factors were not included in the sigmoidal graphs.

Overall, the trend in turbidity has not changed significantly since the USACE 1985-86 study. Both studies illustrated higher turbidities and variances in the headwaters giving way to a lower and more stable turbidity near the dam. The only turbidity assumption the RERF requires is a sufficient decay so that light availability changes from limitation to saturation.

Secchi disk data from the EPA-NES 1974 study yielded a MAX = 21.4 dm (95% confidence interval = 12.5 to 30.3 dm), a slope factor = 9.56 (95% confidence interval = -22.9 to 42), and an EL50 = 58% (95% confidence interval = 23.9 to 92.8%) of the thalweg distance (Figure 52). The USACE 1985-86 data yielded a MAX 24.9 dm (95% confidence interval = 18.9 to 31.0 dm), a slope factor = 6.714 (95% confidence interval = 3.57 to 9.86), and an EL50 = 68.7% (95% confidence interval = 55.2 to 82.3%) of the thalweg distance. The Secchi disk data for the CLP 1992-93 study yielded an EL50 at 63% (95% confidence interval = 54 to 71%), a slope factor of 9.13 (95% confidence interval = 4.61 to 13.6), and an estimated MAX of 25.8 dm (95% confidence interval = 22.3 to 29.3) (Figure 52). Again, decimeters were used in lieu of the standard meter to approximate more closely the range (absolute) of TP, turbidity, and chlorophyll. Also, Secchi disk EL50s were greater than turbidity EL50s and thus implied the former was more sensitive at the lower end of the lake (i.e., higher transparencies). This is plausible because turbidity measures scattering of light while the Secchi disk measures transparency or penetration depths. The increased variance of the Secchi disk nearer the dam (Figure 52) and vice versa for turbidity (Figure 51) tends to support this conclusion.

Overall comparison of the trends illustrated in Figures 51 and 52 indicated no significant change in turbidity since 1985-86 or Secchi disk since 1974. This is antithetical to traditional concepts of accelerated eutrophication which leads to increased turbidity and decreased transparency. Hence, I propose chlorophyll *a* is the primary response variable to be modelled as an indicator of trophic status, and thus temporal trends in turbidity are irrelevant assuming the transition from light limitation to saturation.

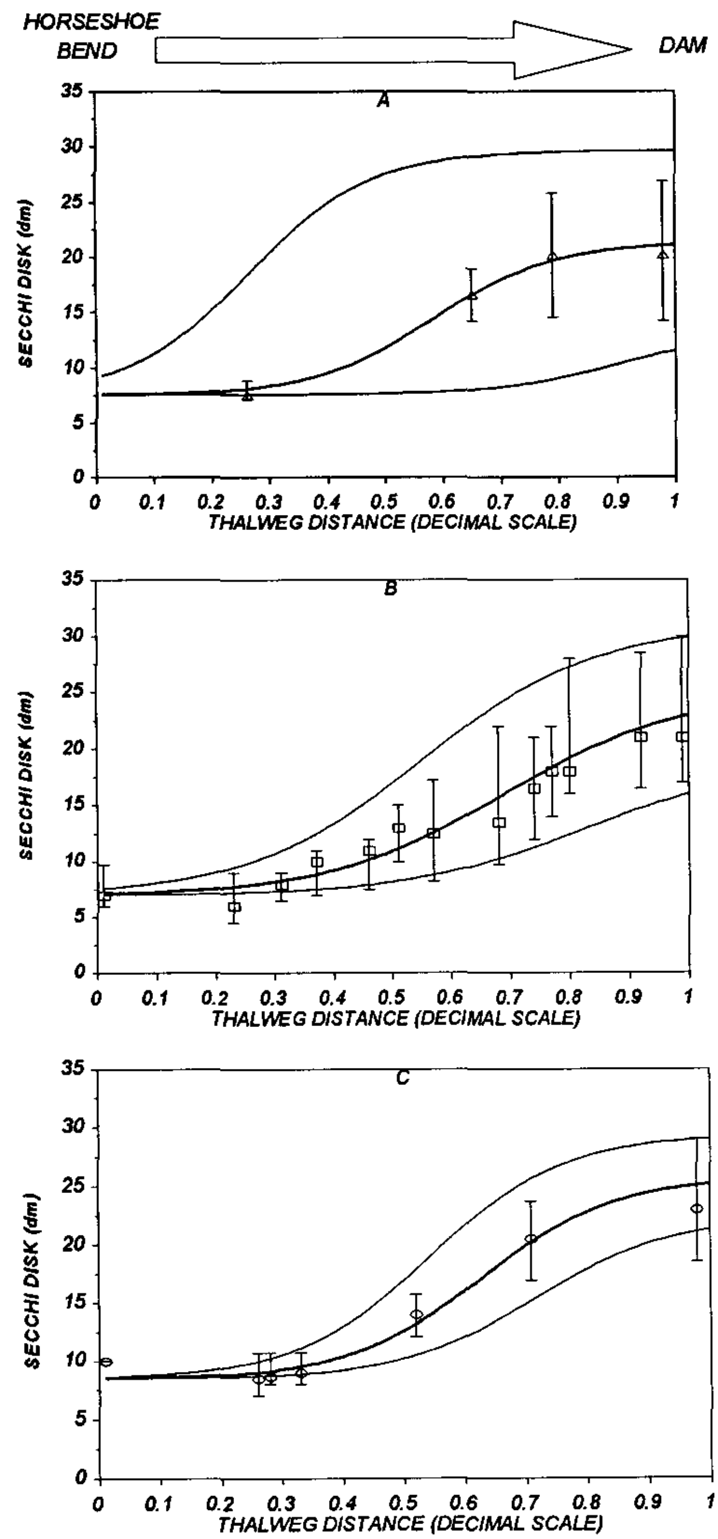



Figure 52. Nonlinear Regression Results for Secchi Disk; (A) 1974, (B) 1985-86, (C) 1992-93 (lines = ±2 SE of model; error bars = quartiles).

Chlorophyll *a* data from the EPA-NES 1974 study fit to the maxima function resulted in a CRF = 105 (95% confidence interval = 7.13 to 203) and a PF = -3.50 (95% confidence interval -5.29 to -1.72). The USACE 1985-86 study resulted in a CRF = 209 (95% confidence interval = 154 to 263) and a PF = -3.32 (95% confidence interval = -3.82 to -2.82). The CLP 1992-93 study yielded a CRF = 254 (95% confidence interval = 177 to 331) and a PF = -3.48 (95% confidence interval = -4.22 to -2.73). Unlike the TP, turbidity, and Secchi disk depth models, initial chlorophyll *a* value is set as 0 $\mu$g/l as a model constraint (a default for the maxima function). While this is obviously untrue, the constraint is irrelevant to model interpretation because impacts of nutrients do not occur until the transition zone and beyond.

These coefficients indicated a statistically significant increase in chlorophyll density at the maxima (i.e., transition zone) from 1974 to 1985-86 and no significant change since 1985-86 (Figure 53). Model results shown in Figure 53 indicated the largest variation was in the transition zone which graded to a more stable population downstream and upstream from the maximum occurrence (peak). Also, the feature in comparing panels A, B, and C in Figure 53 is an obvious temporal increase in variance in chlorophyll *a* densities at the peaks. This is a classic sign of a "stressed" ecosystem (Odum 1975, 1979).

In summary, the nonlinear regression efforts resulted in the following conclusions:

1) no statistically significant temporal changes in turbidity or Secchi disk transparencies could be detected, however, spatial trends were statistically detected;

2) a statistically significant temporal increase in total P levels was detected at headwaters and transition zones; and

3) a statistically significant temporal increase in chlorophyll *a* densities at the transition zone was detected.

Maxima Function Interpretation

For each sampling date during the CLP 1992-93 and USACE 1985-86 studies, the nonlinear fit was performed and resulting parameters evaluated. Lake Tenkiller exhibited accelerated eutrophication (beyond mesotrophy as delineated by chlorophyll *a* densities > 10 $\mu$g/l) on all dates (Figure 54). Lake Tenkiller was not definitively classified according to trophic status. The boundaries of mesotrophy, eutrophy, hypereutrophy, etc. were not delineated (Figures 17 or 54). I conclude the reservoir was in advanced stages of eutrophication the further the points lie to the upper right of the figure (i.e., higher CRF and PF). Two sampling dates for the CLP study are not shown in Figure 20, because they were off the scale. On 18 Apr 93, the model yielded CRF = 5.19 and PF = 1.80, and on 26 May 93, the CRF estimate was 8070 and PF = -14.7. The 18 Apr 93 results reflected a

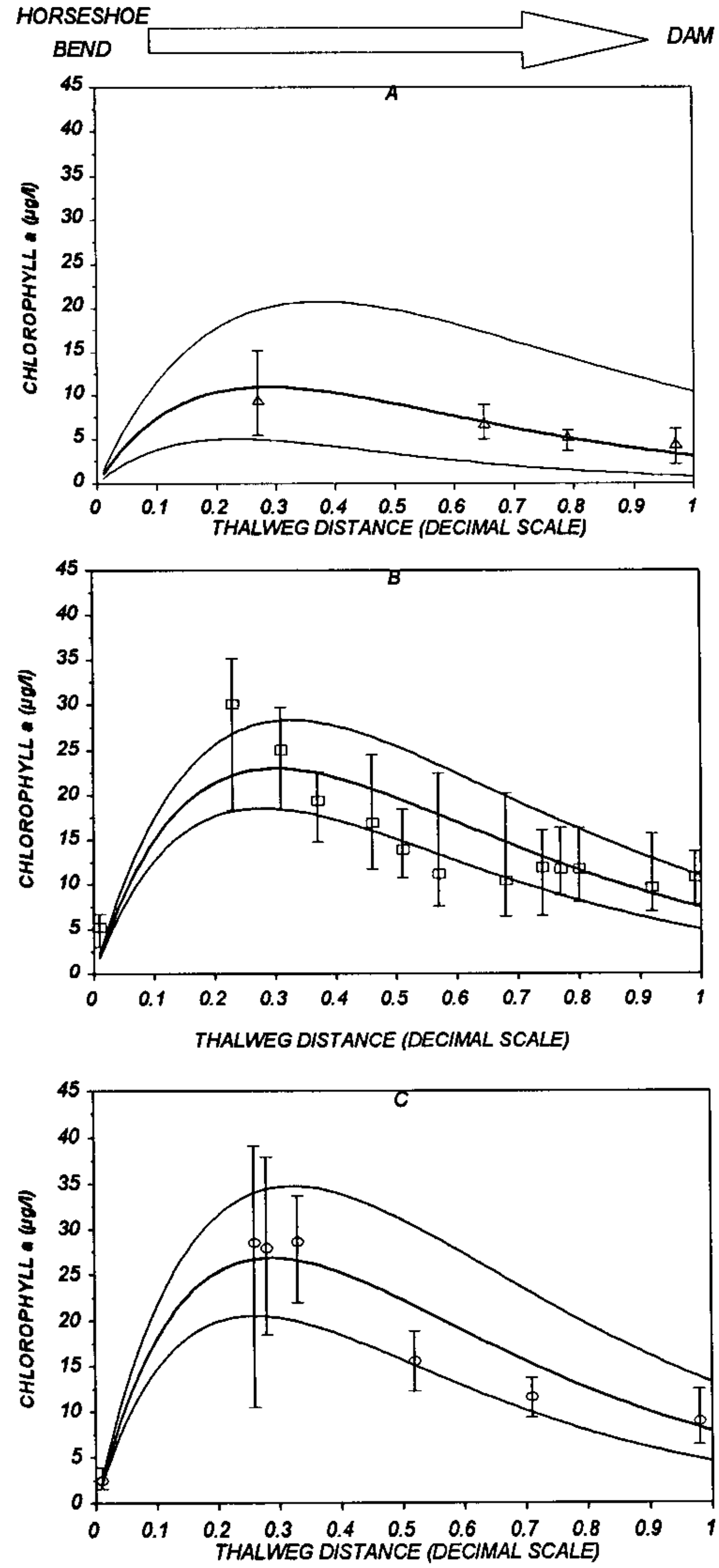



Figure 53. Nonlinear Regression Results for Chlorophyll *a*; (A) 1974, (B) 1985-86, (C) 1992-93 (lines = ±2 SE of model; error bars = quartiles).

chlorophyll *a* trend where the maximum was at station 7 (31 $\mu$g/l) while the upstream stations were 11 $\mu$g/l or less. As an indication of how fast the reservoir conditions can change, the other outlier was on 26 May 93 when the opposite was observed. On this date, station 1 showed 71 $\mu$g/l while the remaining stations steadily decreased to approximately 8 $\mu$g/l at station 7 (Appendix A). Interpretation of the confidence intervals of the model parameters also indicated a significant trend of accelerated eutrophication in Lake Tenkiller since 1974 (Figure 54).

Model Formulation

One of my hypotheses was that hydraulic phenomena affected the impact of the nutrient load. Therefore, to gain insight on these effects, I constructed a correlation matrix of model parameters on hydraulic conditions during the CLP 1992-93 study (Table XXLIII). Initial evaluation of the matrices indicated that the capacity, $z_{avg}$, and area were closely correlated with CRF. Actually, this is misleading. Area and capacity were calculated from the same datum, elevation and $z_{avg}$ is capacity/area. Therefore, any correlation of these to CRF PF can only be inferred from one of these morphometrics.

The peak height (CAMAX) was significantly correlated with mean depth, while the distance at which the peak occurred was better correlated with the hydraulic balance and $z_{avg}/\tau_w$, albeit not significantly (cf. Tables XLIII and XLIV). Peak height was also negatively correlated with the model parameter TPMAXRED ($r = -0.49$, $p < 0.05$) and positively correlated with turbidity minimum ($r = 0.62$, $p < 0.05$). This relationship is plausible because as TP reduction is hampered and turbidity is minimized, algal growth would be expected to increase. The RERF model is based upon this relationship. The significant correlation between the turbidity maximum and $L_P$ (areal P loading) exemplifies the close association between runoff (which translate to high P loads) and increased headwater turbidity. However, turbidity minima were not correlated with $L_P$ and only weakly correlated with $z_{avg}$.

The correlation and probability matrices indicated the CRF could be predicted from $z_{avg}$ and PF can be predicted from the CRF. However, as described by Thornton, et al. (1990) the flow regime dictates the occurrence of the longitudinal trend when inter-reservoir comparisons are made. Therefore, I performed a stepwise multiple regression analysis of CRF versus the hydraulic parameters in the correlation matrix. The result of the regression indicated that most of the variance in CRF could be explained by a combination of $\tau_w$, capacity, hydraulic balance, and inflow/$z_{avg}$. The resulting equation was:

$$\ln(CRF) = -1.08 + 0.352(\pm 0.188)\tau_w + 8.50(\pm 1.83)CAPACITY + 0.206(\pm 0.029)BALANCE - 0.243(\pm 0.0414)\frac{Q_{INFLOW}}{Z_{AVG}} \quad \text{Equation 33.}$$

where $\tau_w$ is hydraulic residence time in years, CAPACITY is km$^3$, balance is m$^3$/s, and $Q_{inflow}/z_{avg}$ is m$^2$/s ($r^2 = 0.72$).

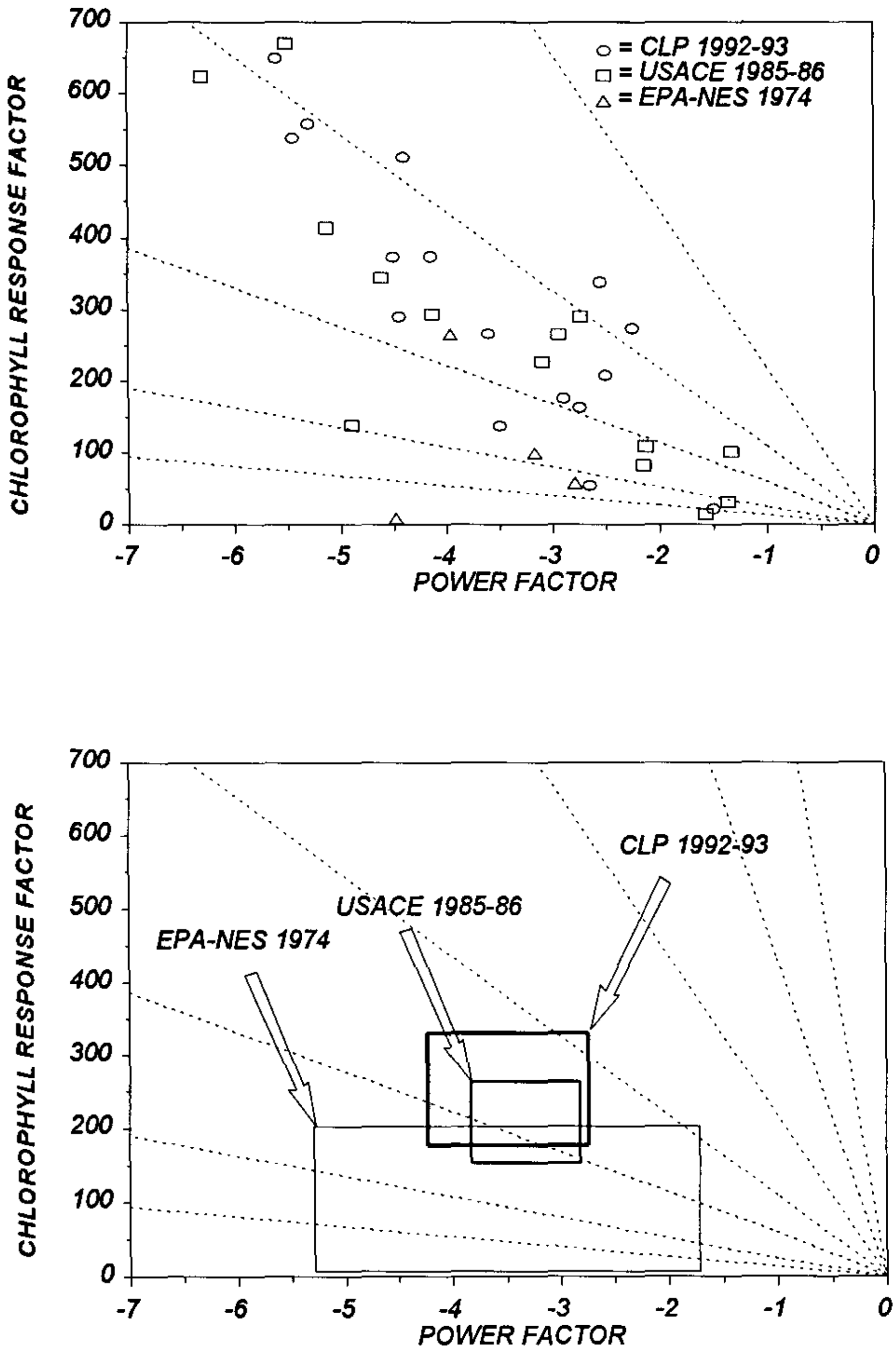



Figure 54. Integration of Maxima Model Coefficients in Lake Tenkiller for Each Sampling Event (upper) and 95% Confidence Intervals of Model Parameters (lower).

Table XLIII. Pearson Correlation Coefficients for Hydraulic and RERF Model Parameters (CAP=capacity; other parameters as identified in the text).

| | TAU (YR) | $Q_{in}$ (CFS) | CAP (ACFT) | ZAVG | BAL (CFS) | AREA ($M^2$) | $Z_{AVG}$+ TAU | $Q_{IN}$+ $Z_{AVG}$ | AREA+ $Q_{IN}$ | L(P) $g/m^2$/YR |
|---|---|---|---|---|---|---|---|---|---|---|
| CRF | -0.22 | -0.07 | 0.92 | 0.89 | 0.08 | 0.92 | 0.32 | -0.09 | -0.09 | -0.12 |
| PF | 0.03 | 0.28 | -0.68 | -0.64 | -0.30 | -0.98 | 0.03 | 0.29 | -0.08 | 0.37 |
| CAMAX | -0.16 | -0.10 | 0.90 | 0.87 | 0.06 | 0.90 | 0.27 | -0.11 | -0.08 | -0.11 |
| MAXD | -0.32 | 0.41 | -0.13 | -0.08 | -0.55 | -0.13 | 0.58 | 0.41 | -0.28 | 0.54 |
| TPMAX | 0.07 | 0.53 | 0.02 | 0.05 | 0.08 | 0.02 | 0.14 | 0.53 | -0.41 | 0.81 |
| TPMIN | -0.26 | 0.02 | 0.41 | 0.43 | -0.58 | 0.41 | 0.43 | 0.01 | -0.02 | 0.18 |
| TPRED | 0.11 | 0.35 | -0.49 | -0.50 | 0.39 | -0.49 | -0.14 | 0.36 | -0.25 | 0.31 |
| TPEL50 | 0.04 | -0.02 | -0.25 | 0.25 | -0.04 | -0.25 | -0.33 | -0.02 | 0.21 | -0.17 |
| TBMAX | NA | 0.81 | 0.28 | 0.29 | NA | NA | NA | NA | NA | 0.82 |
| TBMIN | NA | -0.07 | 0.76 | 0.76 | NA | NA | NA | NA | NA | -0.11 |
| TBEL50 | NA | -0.25 | -0.32 | -0.34 | NA | NA | NA | NA | NA | -0.24 |
| SDMAX | NA | -0.15 | -0.14 | -0.17 | -0.09 | -0.14 | 0.02 | -0.14 | 0.29 | -0.18 |
| SDEL50 | NA | -0.18 | -0.28 | -0.33 | 0.13 | -0.28 | -0.28 | -0.17 | 0.36 | -0.27 |

NA = NOT ANALYZED; TPRED = TOTAL P REDUCTION i.e., (TPMAX-TPMIN)/TPMAX;
MAXD = DECIMAL DISTANCE TO CHLOROPHYLL PEAK

Table XLIV. Bonferroni Adjusted Probabilities for Correlation Coefficients Given in the Previos Table.

| | TAU (YR) | $Q_{IN}$ (CFS) | CAP (ACFT) | $Z_{AVG}$ (m) | BAL (CFS) | AREA ($m^2$) | ZAVG÷ TAU | $Q_{IN}$÷ $Z_{AVG}$ | AREA÷ $Q_{IN}$ | L(P) $g/m^2/yr$ |
|---|---|---|---|---|---|---|---|---|---|---|
| CRF | 1.00 | 1.00 | <0.01 | <0.01 | 1.00 | <0.01 | 1.00 | 1.00 | 1.00 | 1.00 |
| PF | 1.00 | 1.00 | 0.16 | 0.42 | 1.00 | 0.16 | 1.00 | 1.00 | 1.00 | 1.00 |
| CAMAX | 1.00 | 1.00 | <0.01 | <0.01 | 1.00 | <0.01 | 1.00 | 1.00 | 1.00 | 1.00 |
| MAXD | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| TPMAX | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | <0.01 |
| TPMIN | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| TPRED | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| TPEL50 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| TBMAX | NA | 0.03 | 1.00 | 1.00 | NA | NA | NA | NA | NA | 0.03 |
| TBMIN | NA | 1.00 | 0.09 | 0.09 | NA | NA | NA | NA | NA | 1.00 |
| TBEL50 | NA | 1.00 | 1.00 | 1.00 | NA | NA | NA | NA | NA | 1.00 |
| SDMAX | NA | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| SDEL50 | NA | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |

Analysis of variance of the regression indicated statistical significance (Table XLV).

Table XLV. Analysis of Variance for Multiple Regression of Chlorophyll Response Factor on Various Hydraulic Parameters in Lake Tenkiller.

| Parameter | df | SS | MS | F | Significance |
|---|---|---|---|---|---|
| Regression | 4 | 40.2 | 10.0 | 17.7 | $P < 0.01$ |
| Residual | 27 | 15.3 | 0.568 | | |
| Total | 31 | 55.6 | | | |

Prediction of the actual chlorophyll trend requires estimation of the CRF and PF. Therefore, a similar regression analysis was performed with the PF. A higher correlation was obtained from regressing the PF on the CRF alone than on the hydraulic parameters. The resulting equation was:

$$PF = -0.00156 \times CRF - 2.90, \quad r^2 = 0.69 \qquad \text{Equation 34.}$$

The analysis of variance of this regression also indicated signficance (Table XLVI).

Table XLVI. Analysis of Variance for Regression of Power Factor on the Chlorophyll Response Factor in Lake Tenkiller.

| Parameter | df | SS | MS | F | Significance |
|---|---|---|---|---|---|
| Regression | 1 | 147 | 147 | 68.0 | $P < 0.01$ |
| Residual | 30 | 64.7 | 2.16 | | |
| Total | 31 | 211 | | | |

These regressions were performed on pooled data from the CLP 1992-93 and USACE 1985-86 studies. If the nutrient loads change, say from management, a new regression must be computed. In other words, although the hydraulics are being used to calculate a distribution of chlorophyll model parameters, hydraulics do not cause eutrophication. Only the current nutrient loads apply to these regressions.

## Eutrophication Risk Assessment

Assuming hydraulics effected light availability, nutrient fate and transport, and respective biotic linkages, I performed a standard frequency analysis based upon daily values of predicted CRFs (period of record for hydraulics = 1 Jan 81 - 31 Oct 93). Such a series is often called a full series and should not be confused with the annual series used in most flood-frequency anaylsis (Hewlett 1982). From the resulting probability distribution, I predicted the distribution of the CRF and PF. This probability distribution is the intrinsic risk due to morphometric modifiers; the realized risk is the current total P load and is addressed further with TMDLs.

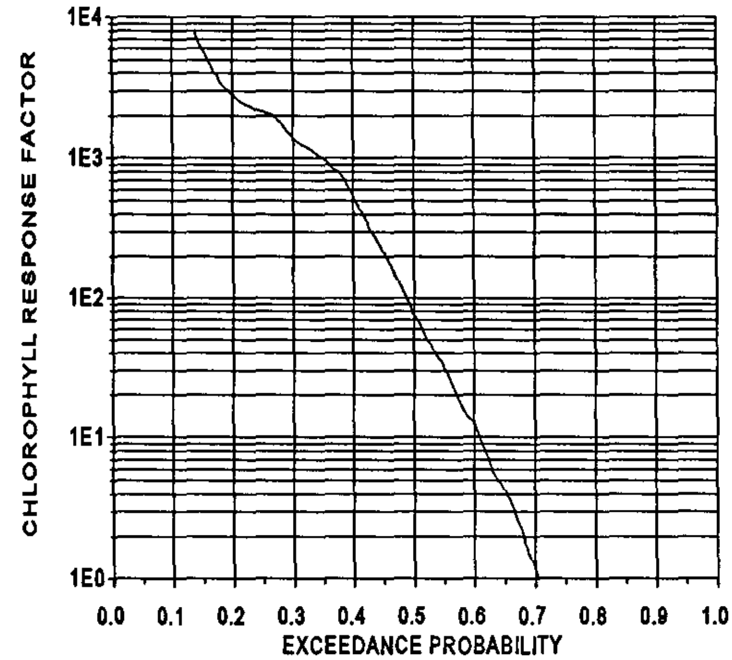
The resulting frequency analysis indicated that the maxima model could describe the chlorophyll *a* trend about 70% of the time, albeit the accuracy declines at ca. 15% exceedance probability (Figure 55). Otherwise stated, the model predicted the lake was in a state of transition such that the hydraulic conditions were not favorable to a maxima trend in chlorophyll *a* for about 30% of the time (Figure 55). I speculate these are conditons of such high flows and low capacities that residence times of N and P fractions are not sufficient for maximum biotic impact, i.e., high chlorophyll densities.

I formulated the model such that the response domain (in Figure 55; x range 0.15 - 0.70) would shift towards the upper right for reservoirs with increasing risks of eutrophy (i.e., higher CRFs with higher probabilities) and towards the lower left for reservoirs with lower risks (i.e., lower CRFs and lower probabilities). Limits of the ordinate were not intended as absolute. Unfortunately, a comparative analysis with other mainstem reservoirs that range in trophic status is yet to be performed and thus I cannot ascertain the relative risk of Lake Tenkiller.

Uncertainty

Every modelling exercise has uncertainty. This one is no exception. Uncertainty that can be quantified describes the risk while that which cannot represents limits to the meaning of the model (see Talcott 1992). In the RERF model, two types of uncertainty can be described, model and parameter.

Model uncertainty is that uncertainty that involves model selection. The sigmoid equation can be justified by the pyramid analogy, while the maxima function is purely empirical. The CRFs were used as a "yardstick" of the chlorophyll *a* trend and do not possess a theoretical foundation (i.e., physical meaning). During inital modelling efforts, I attempted to use a model that substracts turbidity from total P and a ratio of total P to turbidity. This approach could describe an "optimality" model with real meaning in which the estimated biomass optimizes the parameters. However, initial development using these equations suggested the greatest biomass should occur farther along the thalweg than was observed. The observed maxima occurred at the point of inital decay of turbidity and total P.



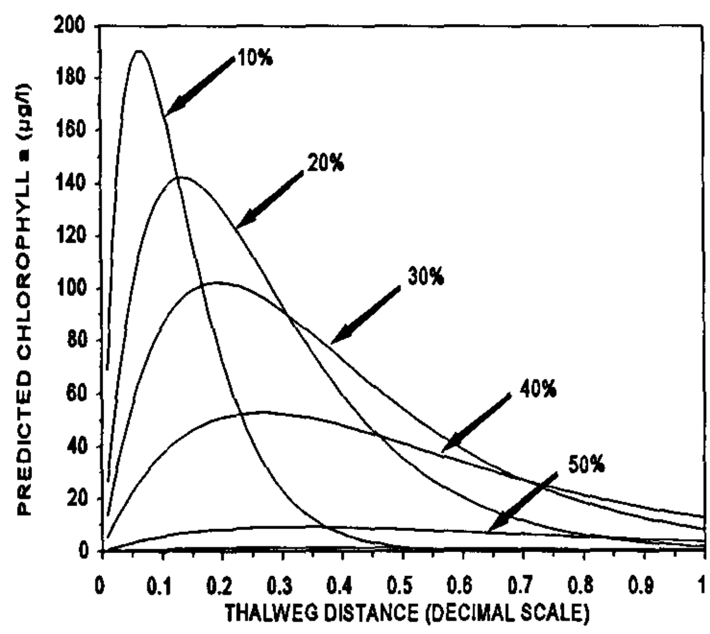




Figure 55. Eutrophication Risk Assesment in Lake Tenkiller from Frequency Analysis of Predicted CRF (top) and Predicted Chlorophyll *a* Trends (bottom).

The observed trend might have been a result of a "threshold" relationship in which the algae were light-limited until the threshold was attained (i.e., initial turbidity decay) and nutrient levels began to impart growth regulation. The maxima function, although empirical, yielded a better fit and thus provided a more accurate, though less real, predictor. Model uncertainty also was introduced during application of the multiple regression analysis of hydraulics on natural log transformed CRFs. In this type of uncertainty, quantification of the uncertainty cannot be performed; the relevance is whether the model selected was appropriate.

Parameter uncertainty begins with data collection. Data collection in this study was subjected to a quality assurance/control procedure under which limits of accuracy were set forth. However, the contribution of uncertainty is real. The actual value cannot be evaluated because logistics did not afford multiple replicates of each analysis for every parameter on every sampling event.

Secondly, model parameter uncertainty also was induced from the multiple regressions. This uncertainty can be estimated by the standard errors of the coefficients (Table XLVII). These uncertainties were not preserved in the final evaluation.

Table XLVII. Estimated Parameter Uncertainty for Multiple Regressions in the RERF Model.

| Parameter | Coefficient | Standard Error | P-value |
|---|---|---|---|
| Model Hydraulic Function | | | |
| Constant | -1.09 | 1.58 | 0.497 |
| $\tau_w$ | 0.352 | 0.188 | 0.070 |
| Capacity | 8.50 | 1.83 | <0.001 |
| Balance | 0.206 | 0.029 | <0.001 |
| Inflow | -0.243 | 0.041 | <0.001 |
| Maxima Function | | | |
| Constant | -2.90 | 0.278 | <0.001 |
| CRF | -1.56E-3 | 1.90E-4 | <0.001 |

Total Maximum Daily Load for Phosphorus

As was discussed previously, I believe reservoirs have intrinsic and extrinsic risk domains. The intrinsic domain is described by the reservoir's response to allochthonous hydraulic conditions. The extrinsic domain is the current stressor load. In eutrophication, the extrinsic domain is the excessive P loads. This extrinsic domain is the manageable variable and requires numeric limitations that promote ecosystem health or prevent further degradation.

Whenever a pollutant limitation is evaluated, as is the case in total maximum daily load (TMDL), an acceptable target value must be put forth. Inherent in this target value is an assumption the ecosystem can tolerate some level of pollutant without undesirable effects which usually include qualitative properties such as aesthetic appearance and/or quantitative properties such as chlorophyll *a* density, species diversity, etc. Odum (1979) alluded to these properties as the performance of an ecosystem. I can understand the applicability of this approach to a subsidizing input such as phosphorus. Indeed, aquatic ecosystems require P to exist while too much will cause undesirable effects, i.e., eutrophication. In other words, too little or too much P will cause harm. For a good review on the philosophy of the "stress-subsidy gradient" the reader is referred to Odum (1979).

The pertinent question is "What is the maximum level and when does the subsidy become a stressor?" This level becomes the TMDL. However, to answer this question a target ecological endpoint(s) must be defined, managers call these goals or management objectives. This is a difficult task for multipurpose reservoirs. For example, the recreational user who appreciates water clarity will desire maximum clarity which requires minimizing algal growth in the lacustrine zone. In contrast, fisheries managers desire more algal growth to support a larger warmwater fishery. Hence, conflicts arise. No intention of compromising the various goals is made here. It is my conviction that reversing the accelerated eutrophication of Lake Tenkiller should be a goal. With this in mind, I calculated a statistical distribution of TMDLs for phosphorus entering Lake Tenkiller using traditional oligotrophy and mesotrophy as target conditions for the TMDL. I compared these "suggested" values with the daily loads observed during the 1974 NES, 1985-86 USACE, and 1992-93 CLP studies (Table XLVIII).

The model proposed by Vollenweider (1968, 1975, 1976) evaluates permissible and dangerous P loads by inference from a given lake's mean depth divided by the hydraulic retention time ($z_a/\tau_w$,). For Lake Tenkiller, the current loads exceeded the critical levels proposed by Vollenweider (1968, 1976) and denoted eutrophy on all sampling events (Table XLVIII, Figures 63 & 64).

I compared the current P loading with computed permissible and dangerous P loadings as suggested by Vollenweider (1968, 1975) for each sampling event. Because the hydraulic budget analysis indicated large variations of $z_{avg}$ and $\tau_w$, I included observed hydraulic conditions into the calculations. The estimated TMDL for this method was derived by dividing each statistic of the annual load by 365 (Table XLVIII, Figure 63). I discuss the obvious weakness of this approach later.

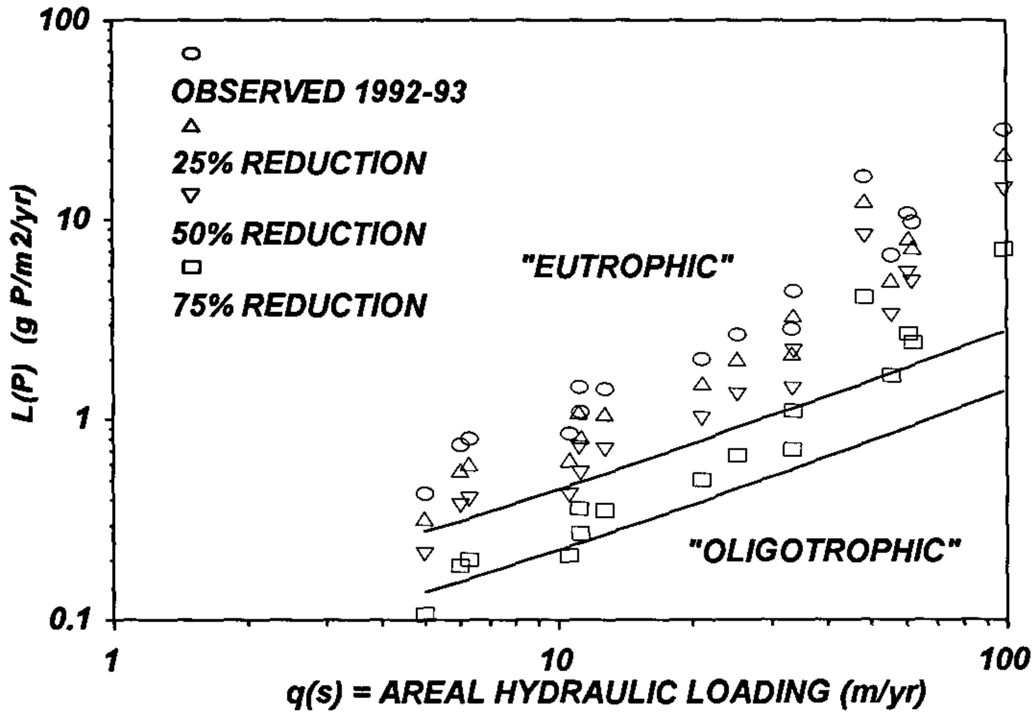



Figure 56. Estimated Trophic Classification as per Vollenweider's (1969) Model Resulting from Influent Total Phosphorus Reductions in Lake Tenkiller.

Using empirical equations developed by Vighi and Chiaudani (1985) that correlated background (non-anthropogenic) P concentrations with two morphoedaphic indices (conductivity and total alkalinity), I back-calculated an annual load based upon the observed flow and predicted total P concentration derived from Vighi and Chiaudani (1985) for each sampling event during the 1992-93 study (Table XLVIII, Figure 63).

Lake Tenkiller simulations using the model developed by Reckhow (1988) indicated approximately 70% reduction of influent TP would be required to achieve 0.01 $\mu$g P/l at the downstream station (Figure 57). I assumed a 70% reduction in influent total P concentration equated to a 70% reduction in annual load. Based upon this approach, the resulting TMDL for P was somewhat higher than those estimated from methods proposed by Vollenweider (1968, 1975) and Vighi and Chiaudani (1985), although the order of magnitudes were similar (Table XLVIII). Resulting trophic conditions of TP reductions and a comparison with nitrogen reductions can be seen in Figures 56-62.

Discussion of the TMDL. A total maximum daily load for phosphorus entering Lake Tenkiller has some inherent weaknesses and should be considered prior to implementing numeric quantities. First, a TMDL assumes that some critical level of phosphorus load is protective of the lake at all times. At times when the lake elevation is low and $\tau_w$ is short, the TMDL could be relaxed without significant impacts. In contrast, if the lake elevation is high and a low inflow imparts a long $\tau_w$, the TMDL must be more restrictive for an equally protective effect. Recall, the RERF model predicted these conditions exacerbate eutrophy. Secondly, seasonal effects influence the impact of impinging P loads. During the fall overturn, accumulated hypolimnetic P is transported to the photic zone and a peak in algal growth is usually observed. The implication of this phenomena is dichotomous. First, the fall peak is a period of transition away from P limitation and thus restrictive TMDLs for P is over-protective. In contrast, a more restrictive TMDL could be promoted because this is a period when excess P is inducing ecological release and having its greatest impact (i.e., more P will likely exacerbate eutrophication because a larger biomass is available for uptake).

Thirdly, a TMDL for a specific contaminant does not address factor interactions. Many limnologists agree that the impact of P is highly dependent upon many environmental factors (e.g., available light, micronutrients) and the most important factor for evaluating a phosphorus TMDL is concurrent N concentrations. As was discussed earlier, ratios of TN:TP have ecological significance and thus impart a significant interaction with P, i.e., a P TMDL is useful under conditions of TN:TP > ca. 15-20. A lower ratio will magnify the effect of P by promoting blue-green dominance (Shapiro 1973). Should the ratio decrease below critical thresholds ($\approx$ 15), more restrictive TMDLs for P will be required for equal protection. In reality, the ratio is being monitored while TP is being managed. One can argue the inverse of this scenario is higher nitrogen imparts a protective effect. While low nitrogen levels can foster undesirable blue-green dominance, excessive nitrogen imparts P control of algal growth. Under these conditions, biomass is proportional to P thus implying P control in biomass reduction.

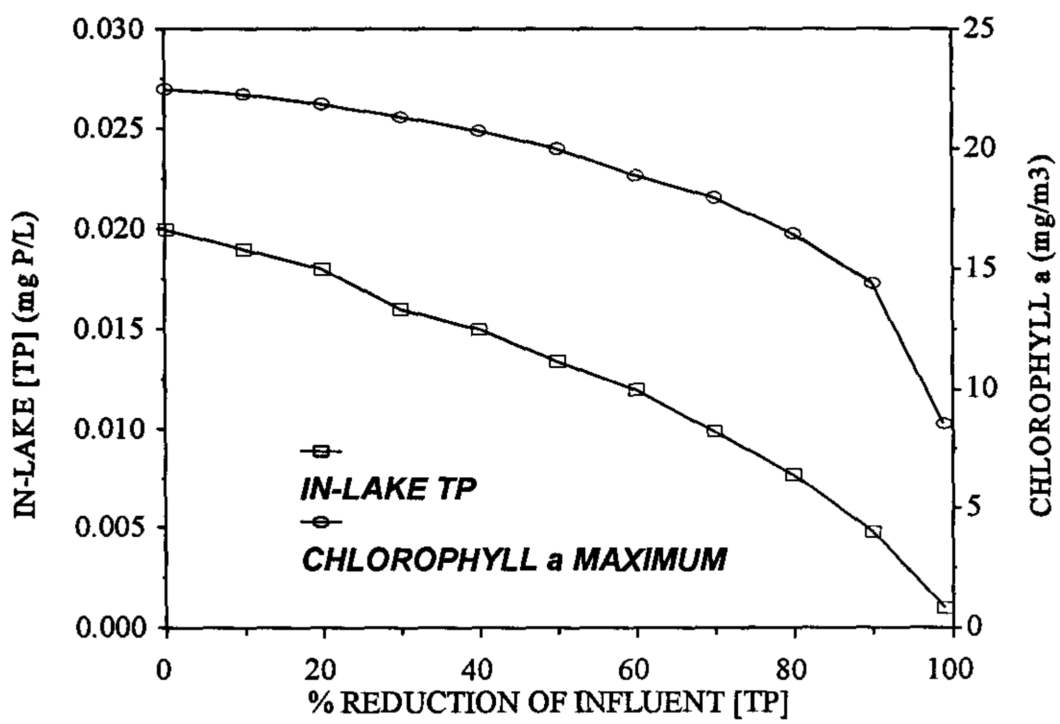



Figure 57. Predicted Summer Chlorophyll *a* Maximum with Influent Total Phosphorus Reductions as per Reckhow's (1988) Model.

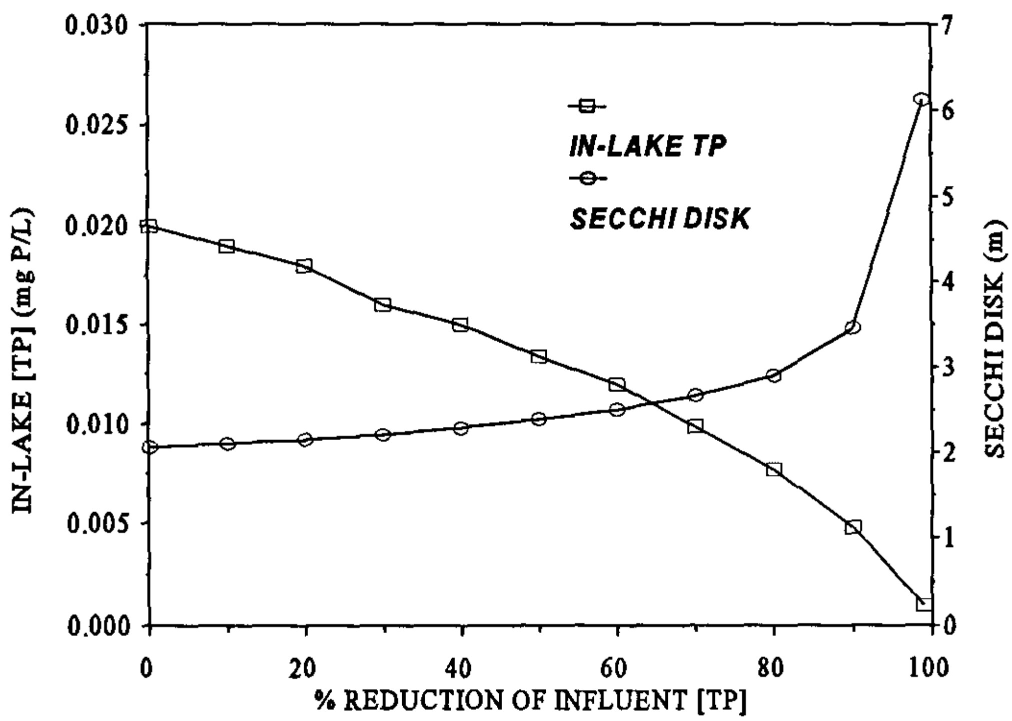



Figure 58. Predicted Secchi Disk Transparency with Influent Total Phosphorus Reductions as per Reckhow's (1988) Model.

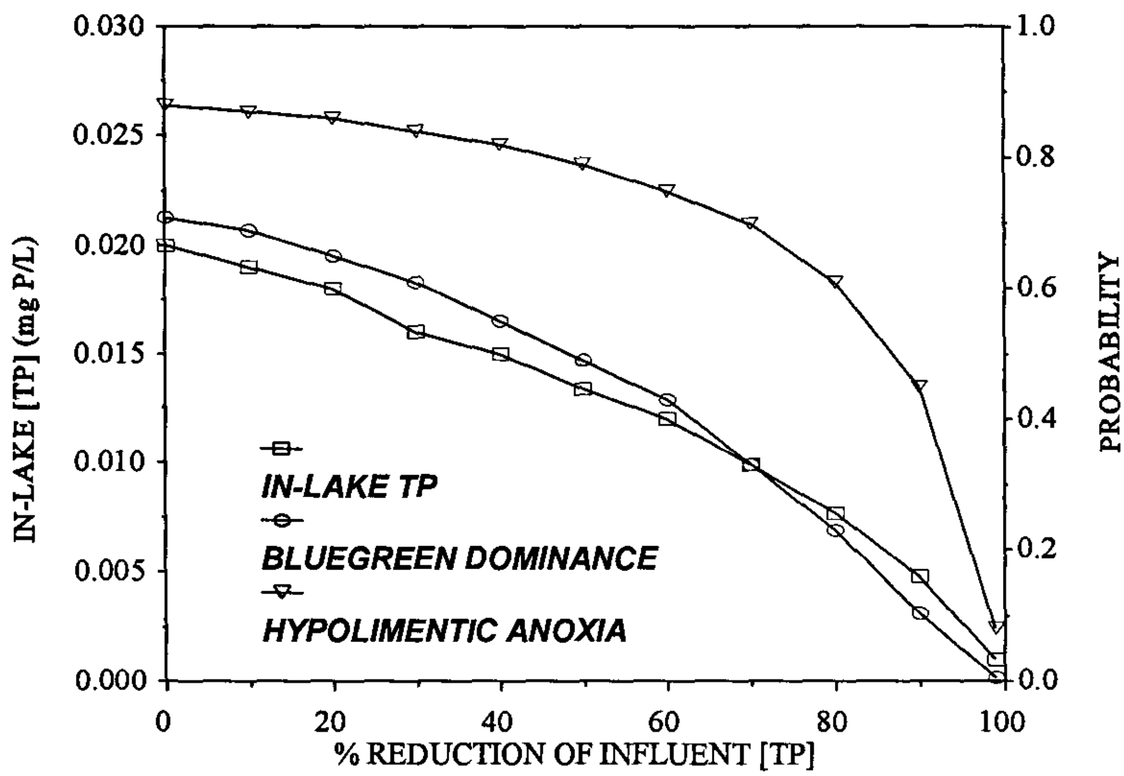



Figure 59. Predicted Probability of Bluegreen Algal Dominance and Hypolimnetic Anoxia with Influent Total Phosphorus Reductions as per Reckhow's (1988) Model.

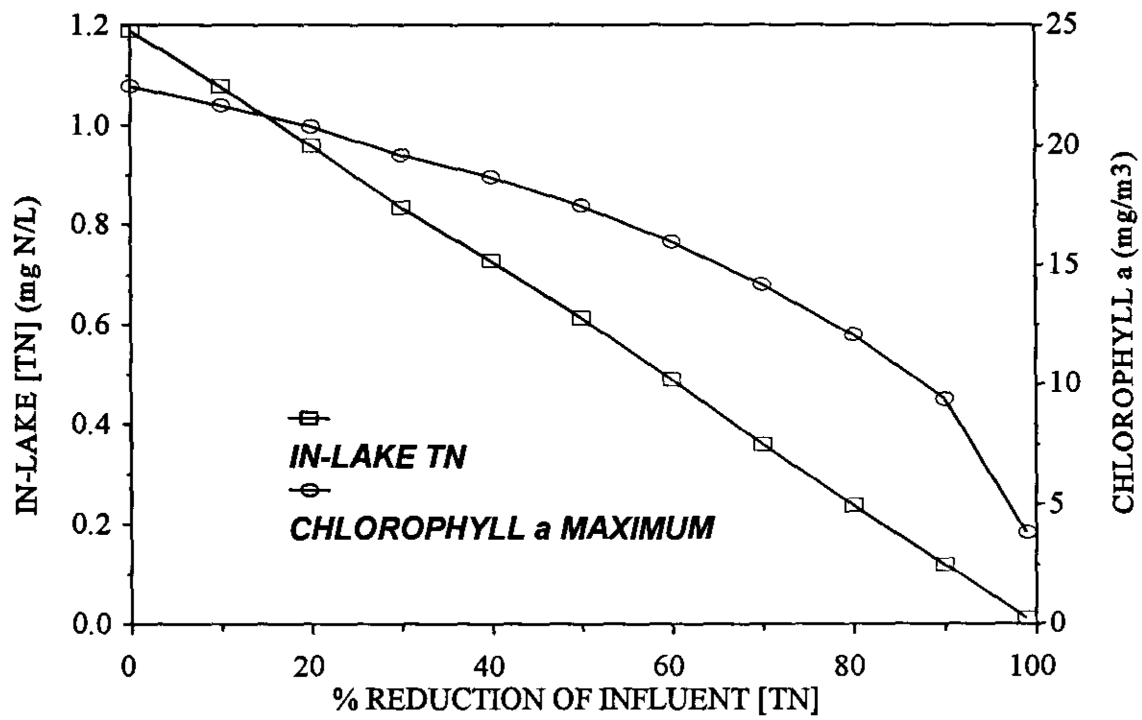



Figure 60. Predicted Summer Chlorophyll *a* Maximum with Influent Total Nitrogen Reductions as per Reckhow's (1988) Model.

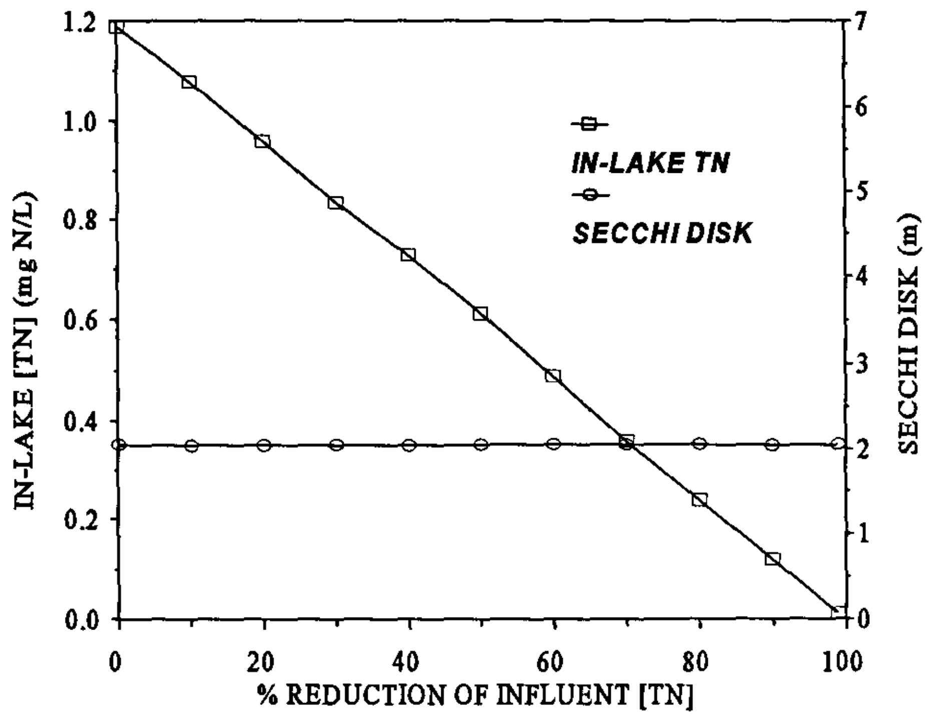



Figure 61. Predicted Secchi Disk Transparency with Influent Total Nitrogen Reductions as per Reckhow's (1988) Model.

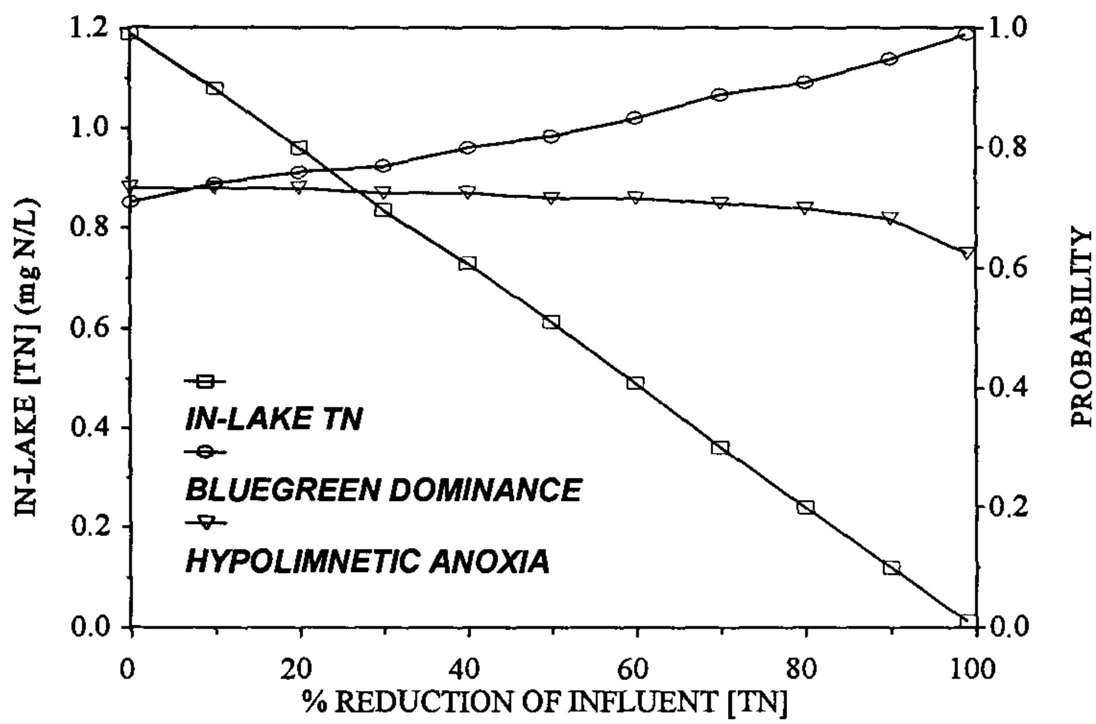



Figure 62. Predicted Probability of Bluegreen Algal Dominance and Hypolimnetic Anoxia with Influent Total Nitrogen Reductions as per Reckhow's (1988) Model.

Finally, I cautiously put forth initial trials of TMDL management on a subsidizing pollutant such as P. Almost any ecologist will agree that ecosystems need a baseline input of P for existence. Thus, a TMDL approach seems appropriate. However, when managment of TMDLs is applied to exclusive stressors (e.g., toxic organics that bioacculmulate), I question the applicability of the approach. We may have studies that indicate an ecosystem has an innate capacity to accommodate such a toxic insult, but have chronic effects on the evolution of the ecosystem been assessed adequately? The adage "time will tell" seems appropriate.

Table XLVIII. Estimated Total Maximum Daily Loads of Total Phosphorus for Lake Tenkiller Headwaters.

| | Statistic of TMDL (kg P/d) | | | | |
|---|---|---|---|---|---|
| Model | Min | 0.25P | 0.50P | 0.75P | Max |
| **Observed** | | | | | |
| EPA-NES 1974 | 67.5 | 75.3 | 113 | 199 | 352 |
| USACE 1985-86 | 109 | 255 | 478 | 645 | 918 |
| CLP 1992-1993 | 61.0 | 150 | 357 | 1070 | 4290 |
| **Suggested** | | | | | |
| Vollenweider (1968, 1976) | | | | | |
| Oligotrophy | | | | | |
| Mesotrophy | 19.7 | 35.6 | 70.4 | 115 | 210 |
| | 39.3 | 71.2 | 141 | 230 | 419 |
| Vighi and Chiadauni (1985) | | | | | |
| Conductivity | 8.33 | 13.8 | 29.6 | 59.7 | 212 |
| Alkalinity | 10.6 | 20.7 | 42.0 | 92.1 | 262 |
| Reckhow (1988) | | | | | |
| *Oligotrophy | 18.3 | 45.0 | 107 | 321 | 1287 |

* = Oligotrophy as TP = 10 $\mu$g P/l at station 7.

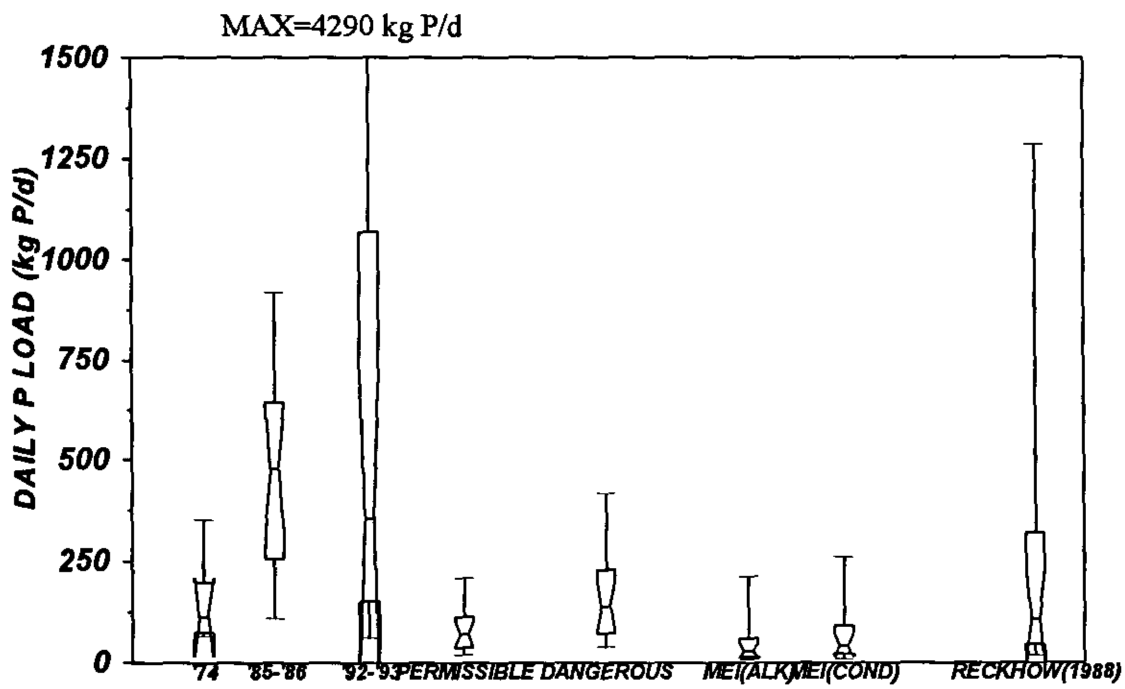



Figure 63. Estimated TMDLs for Total Phosphorus Entering Lake Tenkiller Headwaters.

Evaluation of Recommended TMDL of Phosphorus for Lake Tenkiller

Many methods of establishing a TMDL have been and can be put forth. However, all have a commonality, the TMDL must be a result of a goal or desired condition of the lake. Therefore, the following represent possible management plans based upon inlake goals.

Scenario I. Establish pre-1974 Trophic Status - If the water quality goal for Lake Tenkiller is to establish pre-1974 trophic status conditions, approximately 50-70% reduction of influent total phosphorus would be required according to Vollenweider's (1968, 1976) model. However, when using Reckhow's (1988) model predicted probabilities of hypolimnetic anoxia and bluegreen algal dominance would be approximately 75% and 35%, respectively. Therefore, although an oxic hypolimnion (during summer stratification) may not be achievable, this scenario would mimimize oxygen demand load to the hypolimnion. This, in turn, would augment the DO demand in the tailwaters, and hence, would not be as deleterious to the downstream "put and take" trout fishery.

Scenario II. Establish a Goal which Affords a Measurable Improvement - This is a difficult goal to establish *a priori*. Many uncertainties are part of the modelling process. Many of the variables have large variations and thus present a difficult response to document improvement. However, if a long-term monitoring program and phosphorus reductions are implemented, improvement should be observable within 5-10 years. Clearly, the specifics of the monitoring program and phosphorus reductions will dictate if the improvement can be documented. While phosphorus reduction may not be feasible, control should be enforced as a minimum. Improvement of lake water quality is not the only endpoint we should consider. It has been known for many decades that eutrophication shortens the lifespan of a lake. Therefore, if the lifespan of Lake Tenkiller is to be maximized, the eutrophication process must be controlled, preferrably reversed. The lifespan of the reservoir obviously cannot be measured but certainly should be considered in designing a management plan.

Scenario III. Establish Oligotrophy at the Dam - The state of "oligotrophy" denotes "nutrient poor" but connotes low productivity, high clarity, and an oxic hypolimnion during summer stratification. It has been stated above that Lake Tenkiller ostensibly has experienced hypolimnetic anoxia since impoundment. By this standard, oligotrophic conditions may prove unachievable. However, productivity (and consequently water clarity at the dam) might be controlled by phosphorus reductions. Reckhow's (1988) model predicted that about 70-80% reduction of influent phosphorus would be required to achieve a state of "literal oligotrophy" at the dam. I use "literal oligotrophy" as meaning a total phosphorus concentration of 10 $\mu$g P/l, exclusively, a commonly-used criterion.

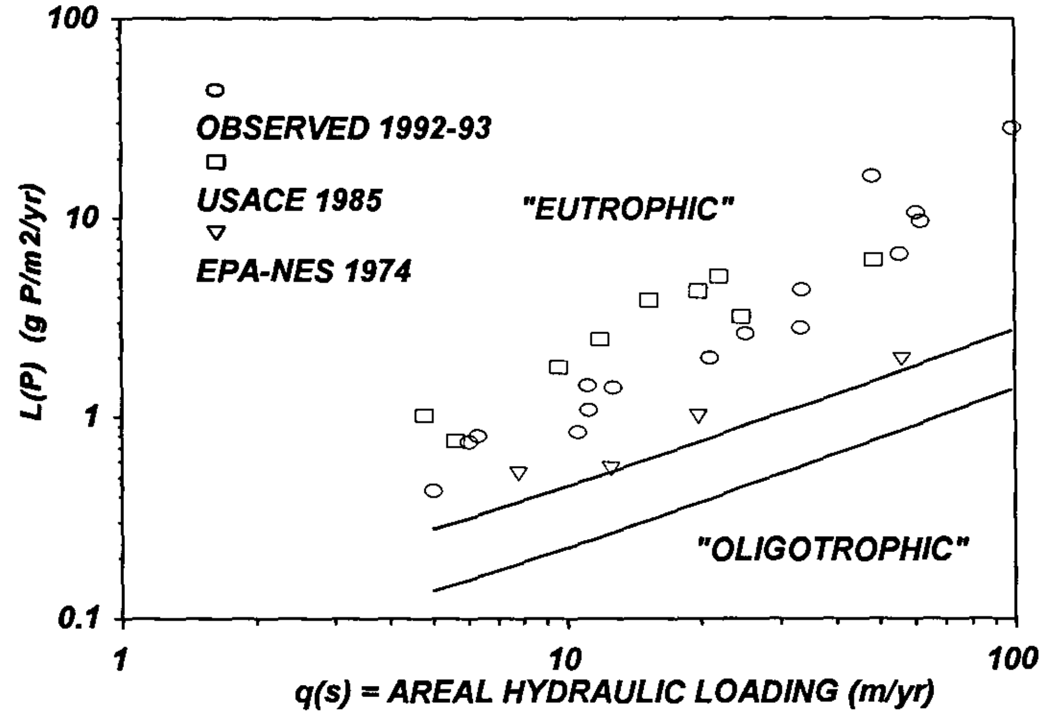



Figure 64. Plot of Lake Tenkiller in 1974, 1986, and 1992-93 as per Vollenweider's (1968, 1976) Model.

Scenario IV. Establish a 30-40% TP Reduction from Best Management Practices of NPS - While differentiation of NPS versus point source impacts is difficult, the data collected in this project afford a few inferences.

First, it appeared that NPS loading was largest during high flow conditions. Historical data indicated these events occurred primarily during the spring runoff season. The data also indicated that pooling of the reservoir also occurred primarily during the same season. The coupling of these two events should have exacerbated eutrophication by increasing the residence time of the largest influx of phosphorus. The data seemed to confirm such a relationship. Therefore, control of NPS is desirable. It follows that if 30-40% reduction of influent total phosphorus is implemented, benefits would be twofold. This approach is highly recommended. However, an intensive monitoring schedule should be coupled with this management and designed in accordance with the effectiveness of the control. In other words, if management is implemented over 3-5 years, a 10 year monitoring schedule should be designed. The monitoring schedule should be in effect during the implementation phase and extend at least 5 years post management. The phosphorus control strategy should be updated in accordance with the results of the monitoring plan.

Scenario V. Establish a 30-40% TP Reduction with BMPs from NPS and Best Available Technologies (BATs) for Point Sources - Assuming BATs afford a 1 mg P/l at point sources, the results from the models have the highest confidence (relative). This higher confidence is due to a certainty of 30-40% reduction at the headwaters of the lake. In contrast, 30-40% reduction of NPS alone (Scenario IV) does not guarantee a 30-40% reduction at the headwaters of the lake, albeit it would nearly be as such during high flow conditions. While this scenario is a reduction in total phosphorus at the headwaters of the lake, as NPS is controlled, point sources will become more signficant in the total phosphorus load to the lake. Unfortunately, the significance of point source contributions depends upon the degree and effectiveness of the NPS control. Therefore, if NPS is controlled effectively, point source controls are anticipated for more lake protection. The inverse is not true, however. If NPS is NOT controlled, point sources will remain obscured.

Scenario VI. Establish Point Source Control with BAT (i.e., effluent limitation of 1 mg P/l) - Exclusive point source control would be partially effective and only during low flow conditions. During high flow conditions, NPS dominated the total phosphorus load impinging the headwaters of Lake Tenkiller. Assuming the data reflects reality, high flow conditions occur primarily during the spring runoff season simultaneously with reservoir "pooling". Therefore, exclusive point source control would not be optimal. However, as NPS control is implemented, point source control is anticipated. Therefore, it should be implemented as part of the management plan and incorporated as a contingency on the effectiveness of the NPS control. The degree of point source control should be calculated based upon the effectiveness of the NPS control.

## Suggested TMDL of Phosphorus for Lake Tenkiller

### Immediate Goal

We recommend a target reduction of influent total phosphorus by **30-40% during the next 5 years**. Additionally, we recommend a monitoring schedule in conjunction with this management goal. Monitoring should begin with the management implementation and continue for at least **5 years** post-implementation.

### Long-Term Goal

We recommend a long-term goal of **70-80% reduction of current influent total phosphorus loads**. The accompanying monitoring schedule should be contingent upon the effectiveness of the short-term goal. If measurable effects of the short-term goal are observed, the monitoring schedule for the long-term goal can be relaxed to approximately quarterly sampling. However, if effects of the short-term goal are not observed, the management and monitoring schedules may need modified accordingly.

## 11. Hydraulic Budget, Including Groundwater

The hydraulic budget of Lake Tenkiller includes inflows from the Illinois River (the major inflow) and the minor tributaries Caney, Elk, Dry, Sixshooter, Terrapin, Chicken, Snake, Cato, Pine, Salt, Dogwood, Burnt Cabin, Sisemore, and Pettit Creeks. The lake has one major eflux, the Illinois River. A USGS gauging station (USGS 07198000) is located below the lake and two stations (USGS 07197000 and USGS 07196500) are located on the Baron Fork near Eldon, OK and Illinois River above the Baron Fork confluence. Analysis of these stations (discussed and presented in the Limiting Nutrient Inflow and Outflow) indicated the minor tributaries did not signficantly contribute to the lake's hydraulic budget. Other influences include groundwater, precipitation, and evaporation. Lake Tenkiller lies outside the southern edge of the Boone formation (eastern Oklahoma groundwater basin). While this formation contributes to area stream flows, its contribution to the hydraulic budget of the lake would be through increasing flow in the Illinois River which is accounted in data from stations 07196500 and 07197000. Annual precipitation averages 43 inches (OWRB 1990). Annual lake evaporation averages about 60 inches (OWRB 1983). A net loss of about 13 inches is indicated. A comparison with the average annual runoff (≈ 15 inches) indicates that atmospheric contributions and losses approximately are approximately balanced. As noted later, macrophytes are primarily non-existent in Lake Tenkiller and hence evapotranspiration is not a significant loss to the hydraulic budget of the lake. Overall, the hydraulic budget of Lake Tenkiller is described by the flows within the Illinois River (see Hydraulic Budget in Limiting Nutrient Inflow and Outflow).

## 12. Alkalinity and Hardness

Epilimentic total alkalinity ranged from a median value of 89 (as mg $CaCO_3$/l) at station 1 to 75 at station 7 (Figure 65). Hypolimnetic total alkalinity ranged from a median of 84.9 (as mg $CaCO_3$/l) at station 2 to 78 at station 7 (Figure 65). Phenolphthalein alkalinities were observed in the epilimnion at all inlake stations. Generally, these alkalinities were 0 except on 1 Aug, 12 Sep, and 24 Oct 92 and 18 Apr, 26 May, 25 Jun, 22 Jul, 4 Aug, 19 Aug, 2 Sep, 16 Sep, 30 Sep, and 21 Oct 93. Although these values were generally low (approx. $<10$ as mg $CaCO_3$/l), the presence of atypical phenolphthalein alkalinity generally indicates excessive photosynthesis.

Epilmnetic total hardness ranged from a median of 93 (as mg $CaCO_3$/l) at station 1 to 72 at station 7 (Figure 66). Hypolimnetic total hardness ranged from a median of 88 (as mg $CaCO_3$/l) at station 2 to 80 at station 7 (Figure 66). The median total hardness at Station 8 (tailrace) was 85. Based upon these data, Lake Tenkiller is classified as hard water (Wetzel 1983).

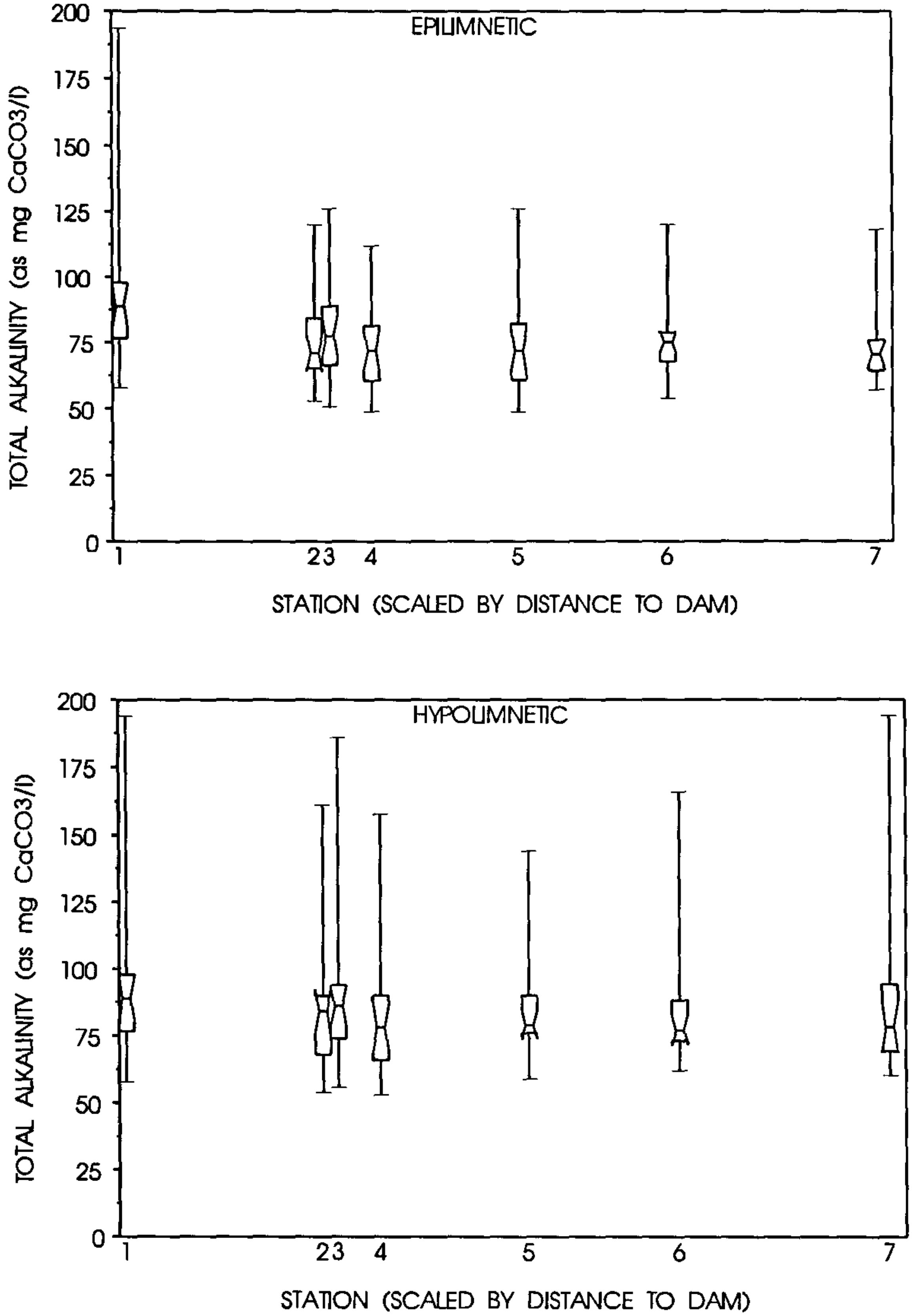



Figure 65. Epilimnetic and Hypolimnetic Total Alkalinities in Lake Tenkiller for CY 1992-93.

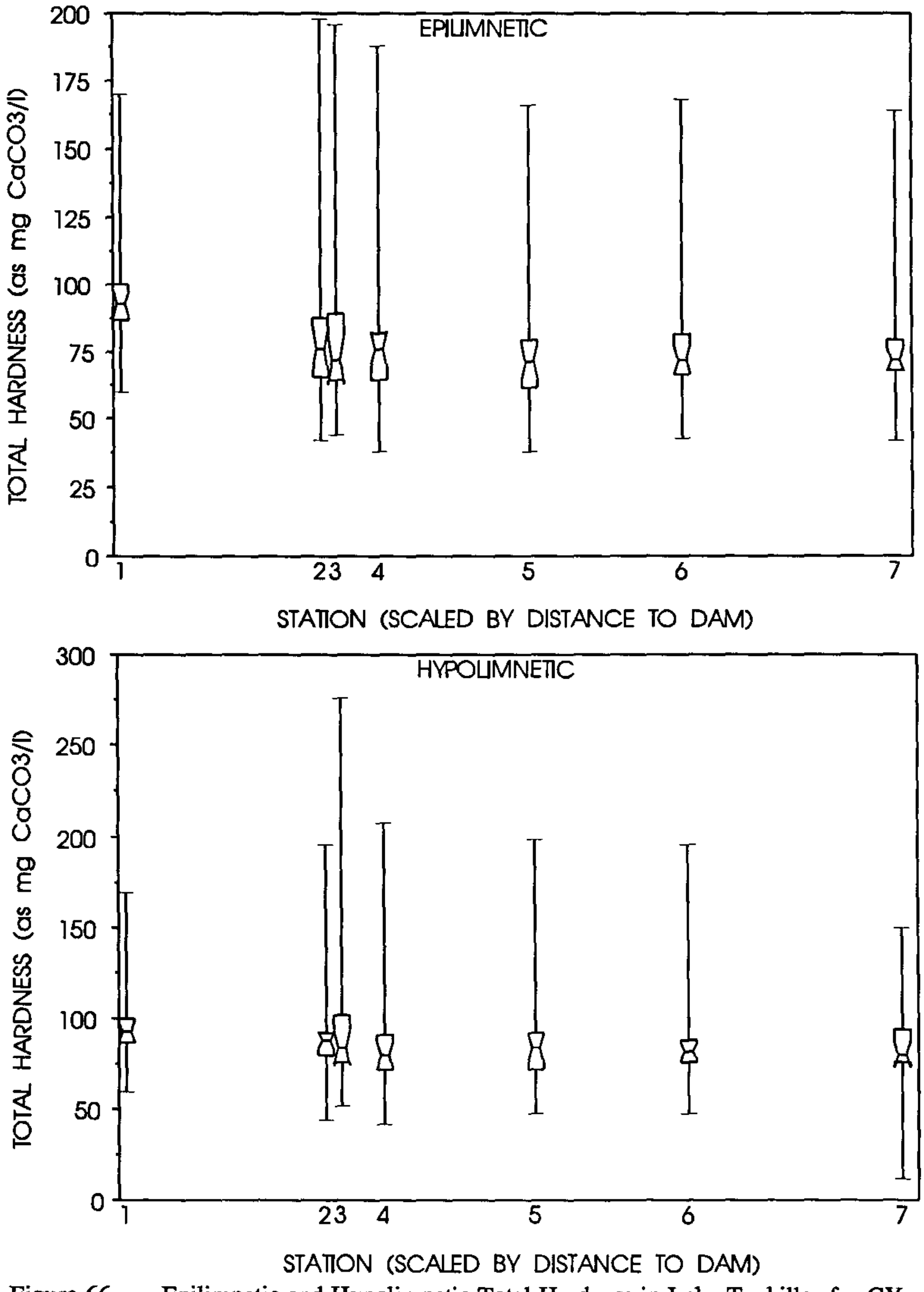



Figure 66. Epilimnetic and Hypolimnetic Total Hardness in Lake Tenkiller for CY 1992-93.

## 13. Algal Assay Bottle Tests and Total N to Total P Ratios Used to Define the Growth Limiting Nutrient and Algal Composition

This section of trophic status evaluation based upon limiting nutrient assays and algal assemblages was taken from Haraughty (1995).

Trophic status was estimated using Carlson's trophic state index (TSI) (Carlson 1977). A TSI was calculated for stations one through seven using chlorophyll *a* values collected during summer stratification. Use of biological data, such as chlorophyll *a*, was recommended by Carlson to provide data which was most free from interferences such as turbidity or high humic acid content. Carlson also suggested using data collected during summer stratification to reduce the variability in chlorophyll *a* caused by spring and fall mixing. Differences between the TSI's at different stations were detected based upon quartile distributions.

In the interests of quality assurance/quality control (QA/QC) procedures, triplicates of at least one sample were analyzed for each of the laboratory parameters. In addition, EPA lab certification standards and HACH standards of known concentrations were tested as unknowns. Field blanks of double deionized water in appropriate HDPE bottles (acid washed or non-acid washed for chlorophyll *a*) were transported to the field, stored on ice,and returned for analysis. Laboratory blanks of double deionized water were also analyzed as unknowns.

Nutrient Limitation Assays. Samples from the above stations at 0.5 m below the surface were collected in the manner previously described, stored in non-acid washed 1 ℓ HDPE bottles on ice, and transported to the lab for use in the Printz Algal Assay: Bottle Test (Miller et al. 1978) on 4 August, 3 September, and 1 October 1993. These dates were chosen because late summer is a period of stabilization in planktonic communities and physico-chemical conditions. Coupled with a date in October when conditions first began to destabilize, these dates should provide the best estimate of nutrient limitation as it relates to indigenous phytoplankton communities.

Axenic cultures of *Selenastrum capricornutum* obtained from Dr. Richard Starr at the University of Texas at Austin were grown to log-growth phase and used to inoculate samples. Samples were spiked with additions of nitrogen, phosphorus, disodium ethylenedinitrilo tetraacetate (EDTA), and nitrogen + phosphorus (**Table XLIX**). The EDTA is a chelator which insures that trace minerals in water are available for algae. Twenty-five ml samples were cultured for 14 days under constant temperature and light intensity in 125 mℓ Erlenmeyer flasks with foam stoppers. Correction for variable light and temperature within the constant temperature room and regulation of $CO_2$ availability and pH were maintained by shaking and rotating samples at least four times daily. Samples from one randomly chosen station were cultured in triplicate and results compared to estimate standard deviations. In addition field blanks as described above were cultured as blank controls.

On days 7, 9, 11, 13, and 14, sample turbidity was measured at 678 and 750 nm

**Table XLIX** Algal Assay Bottle Test Sample Collection Dates.

| Sampling date | Algae sample | AA:BT water | Sampling date | Algae sample | AA:BT water |
|---|---|---|---|---|---|
| 14 FEB 92 | + | - | 26 MAY 93 | + | - |
| 25 APR 92 | + | - | 25 JUN 93 | + | - |
| 04 JUN 92 | + | - | 25 JUL 93 | + | - |
| 04 JUL 92 | + | - | 04 AUG 93 | + | + |
| 01 AUG 92 | + | - | 18 AUG 93 | + | - |
| 19 AUG 92 | + | - | 02 SEP 93 | + | + |
| 12 SEP 92 | + | - | 15 SEP 93 | + | - |
| 24 OCT 92 | + | - | 01 OCT 93 | + | + |
| 08 MAR 93 | + | - | 21 OCT 93 | + | - |
| 18 APR 93 | + | - | | | |

**Table L** Definition of Nutrient Additions.

| Sample ID |
|---|
| Control |
| Control + 0.05 mg P/l as $K_2HPO_4$ |
| Control + 1.00 mg N/l as $NaNO_3$ |
| Control + 1.00 mg EDTA/l |
| Control + 0.05 mg P/l as $K_2HPO_4$ + 1.00 mg N/l as $NaNO_3$ |

using a Secomam S.1000G UV-visible spectrophotometer and used to estimate growth curves. Ocular counts of a least four samples were conducted on day 14 according to Lind's (1985) counting method using a Palmer-Maloney cell. A linear regression was performed using the software package QUATTRO-PRO® to correlate ocular counts with turbidity and estimate cells/mℓ. According to Miller et al. (1978), growth rate should not be used as a growth parameter as it is indirectly related to external nutrient concentrations. Therefore, maximum standing crop (MSC) expressed as cells/mℓ is the growth parameter reported. The nutrient whose addition resulted in the greatest increase in MSC was termed the limiting nutrient.

Phytoplankton Community Structure

Phytoplankton grab samples were collected from mainstem stations concurrent with water samples from 0.5 m below the surface. They were preserved with Lugol's solution as described by Lind (1985) and returned to the lab to be stored at 4° C in the dark until they could be identified and enumerated. Samples were concentrated by centrifugation then analyzed in triplicate by the field method using a Palmer Maloney cell (Lind 1985). Dominant genera were identified and counted.

Species composition was compared with algal assay results to determine whether nutrient limitation drives species composition. Annual trends in species composition were established for riverine, transition and lacustrine zones based on samples from stations 2, 5, and 7.

## Statistical Analysis

Statistical methods were conducted according to procedures outlined in Steele and Torrie (1980) using QUATTRO-PRO® and SYSTAT® software. Longitudinal zonation as depicted by physico-chemical parameters was evaluated using the Mann-Whitney Test as described by Zar (1974). Nutrient limitation was verified by treatment culture growths which were significantly different ($\alpha = 0.05$) from growth of control cultures. Longitudinal zonation of phytoplankton communities as measured by community indices such as total cell count, species diversity, ratio of pennate to centric diatoms and percent blue-green algae was also tested for using the Mann-Whitney test. Phytoplankton community indices were related to physico-chemical parameters via canonical correspondence analysis of transformed data using SYSTAT® software. The data were transformed to account for differences in the magnitude of values. Nutrient limitation was also related to phytoplankton community indices in such a manner.

## Results and Discussion

Lake Tenkiller displayed decreasing values of epilimnetic orthophosphate, total phosphorus and turbidity from stations 1 to 7 (**Table LI**). This trend was also exhibited by nitrogen fractions; however, mean total nitrogen values were slightly higher at station 7 than station 6 and mean nitrate nitrogen values were greater at stations 6 and 7 than station 5. These values were to be expected given the morphometric characteristics of the reservoir. Increases in depth and width of the reservoir between stations 2 and 4 resulted in decreases in water velocity which in turn allowed suspended particles to settle out. The dilution factor of nutrients further increased with proximity to the dam due to increases in lake basin width and depth. This trend was in accordance with Thornton et al.'s (1990) explanation of the longitudinal zonation of reservoirs.

Mean chlorophyll *a* concentrations peaked around stations 3 and 4, then decreased toward station 7. This trend was also in agreement with Thornton et al. (1990) who suggested phytoplankton are light limited in the more turbid headwaters of a reservoir. However, as particles settle out yet nutrient concentrations remain relatively high, primary productivity peaks. Finally, as dilution continues to decrease nutrient concentrations, decreases in chlorophyll *a* follow.

## Longitudinal Zonation

### Riverine Zone

Orthophosphate, total phosphorus, nitrate, total nitrogen, and chlorophyll *a* concentrations at station 1 were significantly higher ($\alpha = 0.01$) than other in-lake stations (**Table LI**). Nephelometric turbidity measurements indicated no significant difference in turbidity at stations 1, 2, 3 or 4. Secchi depths were not measured at station 1 as data was collected from the shore, rather than in the pelagic zone. The ratio of total nitrogen to total phosphorus (TN:TP) was not significantly different among stations 1, 2, 3, 4, or 5.

Turbidity was similar from station 1 through 4, and, along with Secchi depth measurements, placed stations 1 through 4 in the riverine zone. However, as Secchi depths were not measured at station 1, Secchi depth was not weighed as heavily in determining longitudinal zonation as other variables. In addition, because statistical analysis of nitrate and TN:TP did not divide lake stations into at least three groups of stations which were significantly different from one another, those parameters were not given equal weight in determination of lake zonation. Thus, because phosphorus, nitrogen, and chlorophyll *a* concentration were significantly higher at station 1 than other stations, I concluded that the riverine zone included station 1 but generally terminated before reaching stations 2, 3 or 4.

**Table LI** Epilimnetic Nutrient Concentration Statistics of Lake Tenkiller.

| PARAMETER | STATION | MEAN | MEDIAN | S | n |
|---|---|---|---|---|---|
| o-PHOSPHATE | 1 | 0.11 | 0.09 | 0.05 | 16 |
| (mg/ℓ) | 2 | 0.05 | 0.04 | 0.03 | 18 |
| | 3 | 0.04 | 0.03 | 0.03 | 18 |
| | 4 | 0.04 | 0.03 | 0.03 | 18 |
| | 5 | 0.03 | 0.02 | 0.03 | 18 |
| | 6 | 0.02 | 0.01 | 0.02 | 18 |
| | 7 | 0.02 | 0.01 | 0.02 | 18 |
| TOTAL | 1 | 0.14 | 0.12 | 0.07 | 16 |
| PHOSPHORUS | 2 | 0.08 | 0.08 | 0.03 | 18 |
| (mg/ℓ) | 3 | 0.08 | 0.08 | 0.04 | 18 |
| | 4 | 0.08 | 0.07 | 0.04 | 18 |
| | 5 | 0.05 | 0.05 | 0.03 | 18 |
| | 6 | 0.04 | 0.02 | 0.04 | 18 |
| | 7 | 0.03 | 0.02 | 0.04 | 18 |
| NITRATE | 1 | 1.27 | 1.18 | 0.56 | 16 |
| (mg/ℓ) | 2 | 0.53 | 0.46 | 0.44 | 17 |
| | 3 | 0.49 | 0.36 | 0.45 | 18 |
| | 4 | 0.46 | 0.34 | 0.42 | 18 |
| | 5 | 0.38 | 0.21 | 0.38 | 18 |
| | 6 | 0.44 | 0.30 | 0.40 | 18 |
| | 7 | 0.47 | 0.30 | 0.36 | 18 |
| TOTAL | 1 | 2.25 | 2.18 | 1.00 | 16 |
| NITROGEN | 2 | 1.45 | 1.16 | 0.75 | 17 |
| (mg/ℓ) | 3 | 1.40 | 1.23 | 0.77 | 17 |
| | 4 | 1.34 | 1.17 | 0.66 | 17 |
| | 5 | 1.06 | 0.79 | 0.60 | 17 |
| | 6 | 0.97 | 0.74 | 0.59 | 17 |
| | 7 | 1.01 | 0.74 | 0.64 | 17 |

S = Standard Deviation; n = sample size

**Table LII** Continued.

| PARAMETER | STATION | MEAN | MEDIAN | S | n |
|---|---|---|---|---|---|
| TN:TP | 1 | 17.95 | 14.86 | 8.60 | 16 |
| | 2 | 18.66 | 15.74 | 8.70 | 17 |
| | 3 | 19.58 | 16.95 | 10.95 | 17 |
| | 4 | 18.75 | 15.64 | 9.53 | 17 |
| | 5 | 21.23 | 15.06 | 15.71 | 17 |
| | 6 | 31.34 | 27.47 | 21.69 | 17 |
| | 7 | 44.04 | 26.40 | 39.72 | 17 |
| CHLOROPHYLL-a | 1 | 8.16 | 2.55 | 16.97 | 16 |
| (μg/ℓ) | 2 | 25.82 | 28.60 | 15.41 | 22 |
| | 3 | 27.51 | 28.01 | 13.64 | 18 |
| | 4 | 26.23 | 28.66 | 11.35 | 18 |
| | 5 | 17.63 | 15.62 | 9.87 | 22 |
| | 6 | 13.42 | 11.62 | 8.22 | 22 |
| | 7 | 12.60 | 8.95 | 10.38 | 18 |
| PHAEOPHYTIN | 1 | 1.27 | 1.06 | 1.41 | 15 |
| (μg/ℓ) | 2 | 1.15 | 0.60 | 1.39 | 21 |
| | 3 | 1.67 | 1.30 | 1.66 | 17 |
| | 4 | 2.16 | 1.43 | 2.35 | 17 |
| | 5 | 1.15 | 0.52 | 1.68 | 21 |
| | 6 | 0.76 | 0.13 | 1.41 | 21 |
| | 7 | 1.04 | 0.07 | 1.70 | 17 |
| TURBIDITY | 1 | 13.67 | 8.70 | 10.36 | 11 |
| (NTU) | 2 | 11.11 | 6.30 | 11.04 | 15 |
| | 3 | 14.26 | 8.30 | 14.63 | 15 |
| | 4 | 8.03 | 5.80 | 5.31 | 14 |
| | 5 | 6.22 | 4.50 | 6.39 | 15 |
| | 6 | 4.18 | 2.25 | 5.30 | 15 |
| | 7 | 3.81 | 2.10 | 5.54 | 15 |

S = Standard Deviation; n = sample size

Transition Zone

Orthophosphate, total phosphorus and total nitrogen concentrations between stations 2, 3 and 4 were not significantly different. Nitrate concentrations were not significantly different among stations 2 through 7, indicating that nitrate concentrations were not a useful tool in delimiting zonation in Lake Tenkiller.

Orthophosphorus and total phosphorus concentrations as well as turbidity at stations 2, 3, and 4 were significantly higher ($\alpha = 0.05$) than stations 5, 6, and 7. Total nitrogen concentrations at station 2 were not significantly different from those at stations 3, 4 or 6. No significant difference existed between chlorophyll *a* concentrations at station 2, 3, or 4 though concentrations at only stations 3 and 4 were significantly higher than station 5. No significant difference between nephelometric turbidity or Secchi depth was determined for stations 2, 3, and 4. Secchi depth at station 2 was not significantly different from those at stations 3 or 4 but was significantly ($\alpha = 0.01$) less than those at stations 5, 6, and 7.

Assuming the riverine zone included only station 1, use of phosphorus, nitrogen and, chlorophyll *a* concentration and Secchi depth gradients to determine longitudinal zonation of Lake Tenkiller would include stations 2, 3, and 4 in the transition zone. Nephelometric turbidity gradients placed station 5 in the transition zone. As the majority of the parameters grouped stations 2, 3, and 4 as not being significantly different from each other, the transition zone was likely to include these stations for most of the year. Given the migratory nature of longitudinal zonation (Thornton et al. 1990), station 5 could sometimes be included in the transition zone.

**Table LIII** Statistical Significance of Epilimnetic Nutrient Concentrations Using the Mann-whitney Test.

| NUTRIENT | | ST1 | ST 2 | ST 3 | ST 4 | ST 5 | ST 6 |
|---|---|---|---|---|---|---|---|
| Orthophosphate | 2 | ** | -- | -- | -- | -- | -- |
| | 3 | ** | NS | -- | -- | -- | -- |
| (mg/ℓ) | 4 | ** | NS | NS | -- | -- | -- |
| | 5 | ** | ** | ** | * | -- | -- |
| | 6 | ** | ** | ** | ** | NS | -- |
| | 7 | ** | * | ** | ** | * | NS |
| Total Phosphorus | 2 | ** | -- | -- | -- | -- | -- |
| | 3 | ** | NS | -- | -- | -- | -- |
| (mg/ℓ) | 4 | ** | NS | NS | -- | -- | -- |
| | 5 | ** | ** | ** | * | -- | -- |
| | 6 | ** | ** | ** | ** | * | -- |
| | 7 | ** | ** | ** | ** | ** | NS |
| Nitrate | 2 | ** | -- | -- | -- | -- | -- |
| | 3 | ** | NS | -- | -- | -- | -- |
| (mg/ℓ) | 4 | ** | NS | NS | -- | -- | -- |
| | 5 | ** | NS | NS | NS | -- | -- |
| | 6 | ** | NS | NS | NS | NS | -- |
| | 7 | ** | NS | NS | NS | NS | NS |
| Total Nitrogen | 2 | ** | -- | -- | -- | -- | -- |
| | 3 | ** | NS | -- | -- | -- | -- |
| (mg/ℓ) | 4 | ** | NS | NS | -- | -- | -- |
| | 5 | ** | * | NS | NS | -- | -- |
| | 6 | ** | NS | * | * | NS | -- |
| | 7 | ** | ** | * | ** | NS | NS |

* = significant ($\alpha$ = 0.05); ** = highly significant ($\alpha$ = 0.01)
NS = not significant

**Table LIV** Continued.

| PARAMETER | | STATION 1 | 2 | 3 | 4 | 5 | 6 |
|---|---|---|---|---|---|---|---|
| TN:TP | 2 | NS | -- | -- | -- | -- | -- |
| | 3 | NS | NS | -- | -- | -- | -- |
| | 4 | NS | NS | NS | -- | -- | -- |
| | 5 | NS | NS | NS | NS | -- | -- |
| | 6 | ** | ** | ** | ** | ** | -- |
| | 7 | ** | ** | ** | ** | ** | NS |
| CHLOROPHYLL-a | 2 | ** | -- | -- | -- | -- | -- |
| | 3 | ** | NS | -- | -- | -- | -- |
| (μg/ℓ) | 4 | ** | NS | NS | -- | -- | -- |
| | 5 | ** | NS | * | ** | -- | -- |
| | 6 | ** | * | ** | ** | * | -- |
| | 7 | ** | ** | ** | ** | ** | NS |
| TURBIDITY | 2 | NS | -- | -- | -- | -- | -- |
| | 3 | NS | NS | -- | -- | -- | -- |
| (NTU) | 4 | NS | * | NS | -- | -- | -- |
| | 5 | ** | ** | * | * | -- | -- |
| | 6 | ** | ** | ** | ** | * | -- |
| | 7 | ** | ** | ** | ** | ** | NS |
| SECCHI DEPTH | 2 | -- | -- | -- | -- | -- | -- |
| | 3 | -- | NS | -- | -- | -- | -- |
| (Meters) | 4 | -- | NS | NS | -- | -- | -- |
| | 5 | -- | ** | ** | ** | -- | -- |
| | 6 | -- | ** | ** | ** | ** | -- |
| | 7 | -- | ** | ** | ** | ** | NS |

* = significant ($\alpha = 0.05$); ** = highly significant ($\alpha = 0.01$)
NS = not significant

Lacustrine Zone

Though orthophosphate concentrations did not differ significantly among stations 5, 6, and 7, total phosphorus was significantly higher ($\alpha = 0.05$) at station 5 than stations 6 and 7. No significant difference existed between total phosphorus concentrations at stations 6 and 7. Though station 5 did not differ significantly in total nitrogen concentrations from stations 3, 4, 6 or 7, station 4 concentrations were significantly ($\alpha = 0.05$) greater than stations 6 and 7. Chlorophyll *a* concentrations and turbidity at station 5 were significantly ($\alpha = 0.01$) greater than those at 6 and 7. No significant difference existed between concentrations or turbidity at station 6 and 7. Secchi depth at station 5 was significantly less than at stations 6 and 7 but no significant difference existed between secchi depth at 6 and 7.

Use of phosphorus, total nitrogen, and chlorophyll *a* concentration gradients as well as turbidity and Secchi depth to determine longitudinal zonation of Lake Tenkiller generally placed stations 5, 6, and 7 in the lacustrine zone. Though station 5 differed significantly from stations 6 and 7 in to total phosphorus, nitrogen species, chlorophyll *a*, Secchi depth, and turbidity, significant differences exist with regard to several parameters between station 5 and stations 2, 3, and 4, leading to the conclusion that station 5 was near the gradient between the transition and lacustrine zone. I assumed that though station 5 sometimes exhibited characteristics of the transition zone, it was more often associated with the lacustrine zone. Finally, the longitudinal zonation of reservoirs is seasonally dynamic; *i. e.*, no abrupt boundaries exist between zones but rather zone delineation is temporarily and spatially variable. Although a station may not always fall into the same reservoir zone, for the purposes of this study, stations are assumed to fall within the same zone year-round.

Trophic Status of Reservoir Zones

The trophic structure of an aquatic system is defined by the qualitative and quantitative aspects of energy transfer (Lindeman 1942). Calculation of a trophic state index (TSI) produces a simple measure of these energy transfers. TSI values, illlustrated in Box and Whisker format (**Figure 67**) from Lake Tenkiller support the distribution of lake zonation as established by comparison of physico-chemical parameters from within the lake. Box and Whisker plots illustrate an entire data range: error bars represent minimum and maximum values; top and bottom of the box represent upper and lower quartiles, respectively; median values are illustrated by the midline in the box; the notch represents the approximate statistical domain; and box width is representative of sample size. The TSI's for Tenkiller could have been used to classify station 1 as mesotrophic, stations 2 - 4 as hypereutrophic, and stations 5 - 7 as eutrophic (Carlson 1979). The classification of station 1 as mesotrophic was most likely due to higher turbidity which inhibited phytoplankton productivity. Trophic status based upon total phosphorus (Carlson 1979) would classify stations 1- 4 as hypereutrophic and stations 5, 6, and 7 as eutrophic (**Figure 68**).

Nutrient Limitation Assays

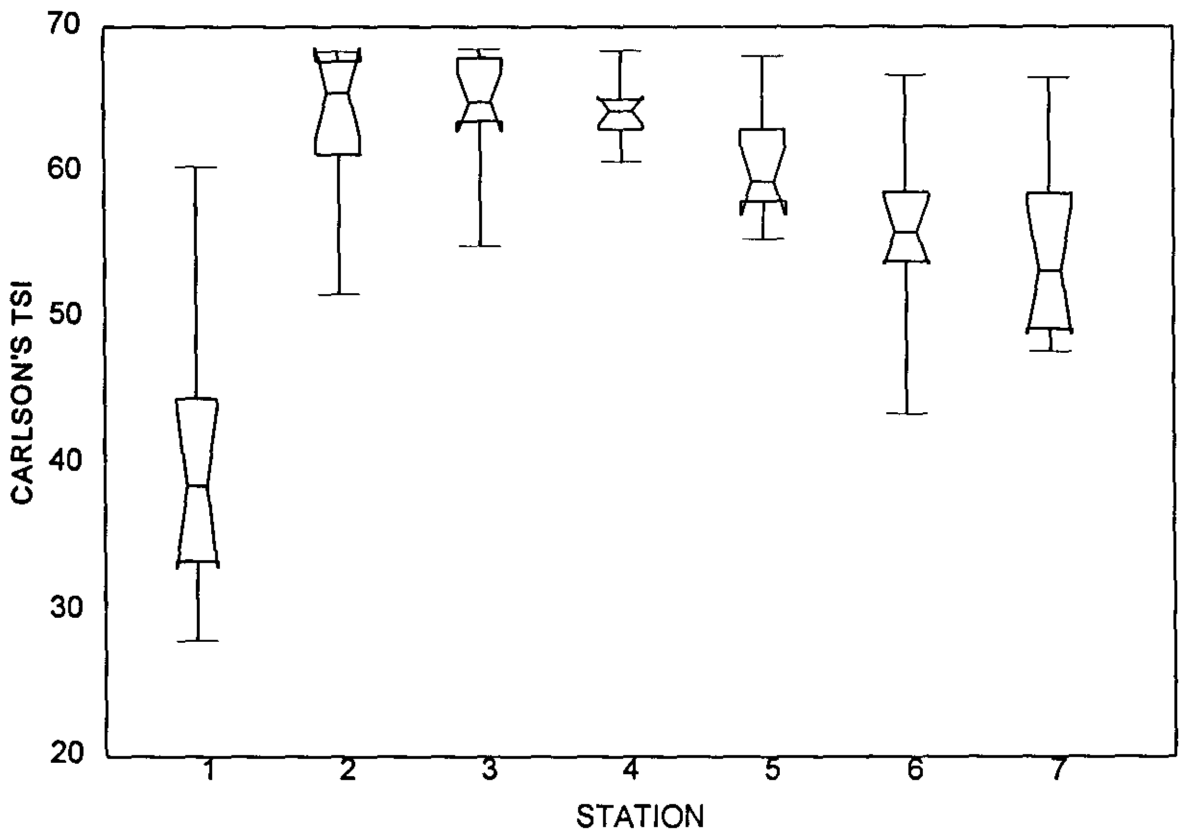



**Figure 67** Carlson's Trophic State Index (TSI-chlorophyll *a*) for Lake Stations.

In nutrient limitation assays, potential nutrient limitation is defined by significant differences in biomass between control and treatments (**Table LV**). Statistically significant ($\alpha$

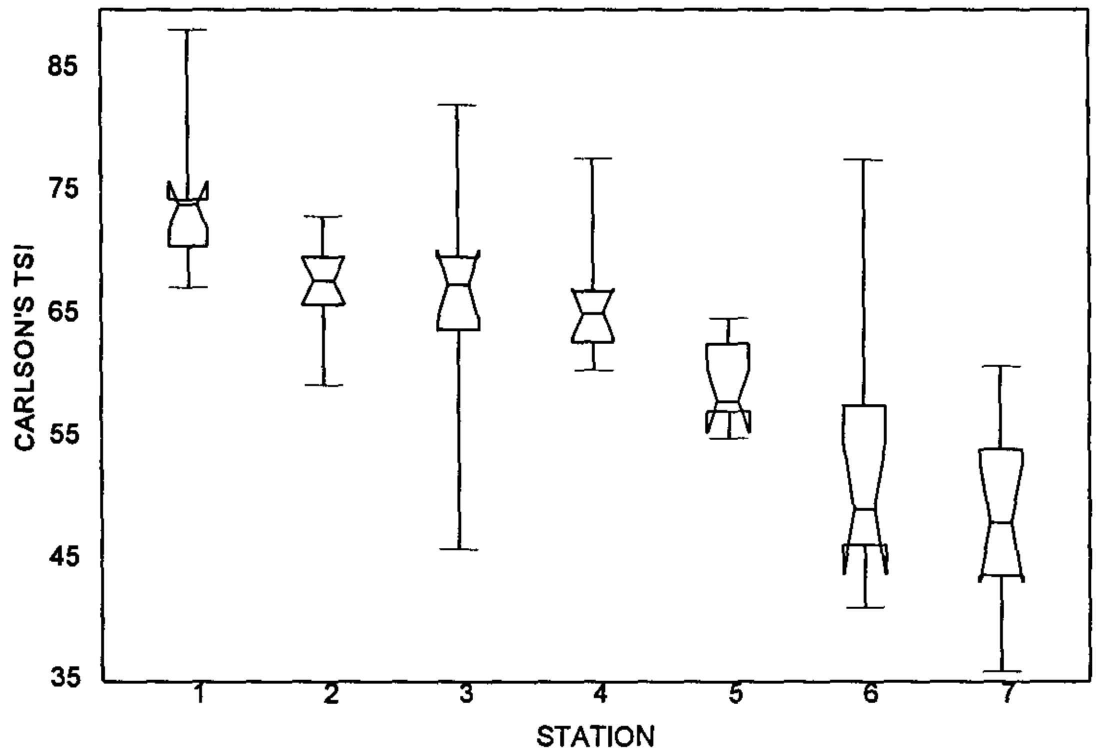



**Figure 68** Carlson's Trophic State Index - Total Phosphorus.

= 0.05) differences between phosphorus and phosphorus plus nitrogenspiked treatments and controls for 4 Aug 1993 indicated phosphorus limitation at all in-lake stations (**Figure 69**). Results from 2 Sep 1993 displayed more variable limitation (**Figure 69**). Station 2 exhibited nitrogen limitation; stations 1, 3, 4, and 5 displayed dual limitation, both phosphorus and nitrogen were needed; and results from stations 6 and 7 indicated phosphorus limitation. On 1 Oct 1993, stations 2, 3, 5, 6, and 7 displayed phosphorus limitation (**Figure 69**). Stations 1 and 4 exhibited dual limitation of nitrogen and phosphorus.

Longitudinal Zonation of Nutrient Limitation

Nutrient limitation was variable between the longitudinal zones of the reservoir as determined from physico-chemical parameters. Results from the riverine zone,

**Table LV** Interpretation of Nutrient Limitation Assay Results (Adapted from Page et al. 1985).

| NUTRIENT SPIKE | | | | | RESULT |
|---|---|---|---|---|---|
| P | N | P + N | EDTA | C | |
| * | NS | * | NS | 0 | Phosphorus Limited |
| NS | * | * | NS | 0 | Nitrogen Limited |
| NS | NS | * | NS | 0 | Dual Limitation |
| NS | NS | NS | * | 0 | Trace Element Limitation |
| * | * | * | * | 0 | Dual Limitation |

P = phosphorus enriched, N = nitrogen enriched, EDTA = EDTA enriched, C = control, * = Significant ($\alpha = 0.05$) difference in growth over controls, NS = no significant growth over that of controls.

though limited, indicated that both nitrogen and phosphorus were limiting. Results in the transition zone were more variable, as expected given the definitive characteristics of the zone, where nitrogen, phosphorus, and dual limitation were indicated (**Figure 69**). Nutrient limitation in the lacustrine zone was exclusively phosphorus limitation with the exception of station 5 on 2 Sept. 93.

Although results from the riverine zone did indicate potential nutrient limitation, it was likely that those results were due to weaknesses in the method, rather than actual nutrient limitation. Given the high turbidities at station 1, it was likely that phytoplankton at station 1 were light limited, rather than nutrient limited.

Results of nitrogen limitation in the transition zone on 2 Sep 93 were probably due to low nitrate concentrations on that day (0.04 mg/ℓ at station 2, 0.09 mg/ℓ at station 3, and 0.04 mg/ℓ at station 4). The variability of assay results from transition zone stations alludes to the heterogeneity of that zone.

Gakstatter and Katko (1986) found sites on the Illinois River to be primarily phosphorus limited (excluding sites with influence from point source discharges). In addition, some sites were limited by some unknown factor, presumably a trace element which was not identified. This phosphorus limitation in the river and reservoir was due more to high nitrogen concentrations than to low phosphorus concentrations. That high nitrogen to phosphorus ratio was actually beneficial for the reservoir. Had the ratio been skewed in the other direction, nitrogen limitation may have resulted, which in turn could have resulted in phytoplankton communities dominated by blue-green algae for a longer portion of the year, given the ability of many blue-green algae to fix atmospheric nitrogen and thus out-compete other algae when nitrogen is the limiting nutrient (Shapiro 1973).

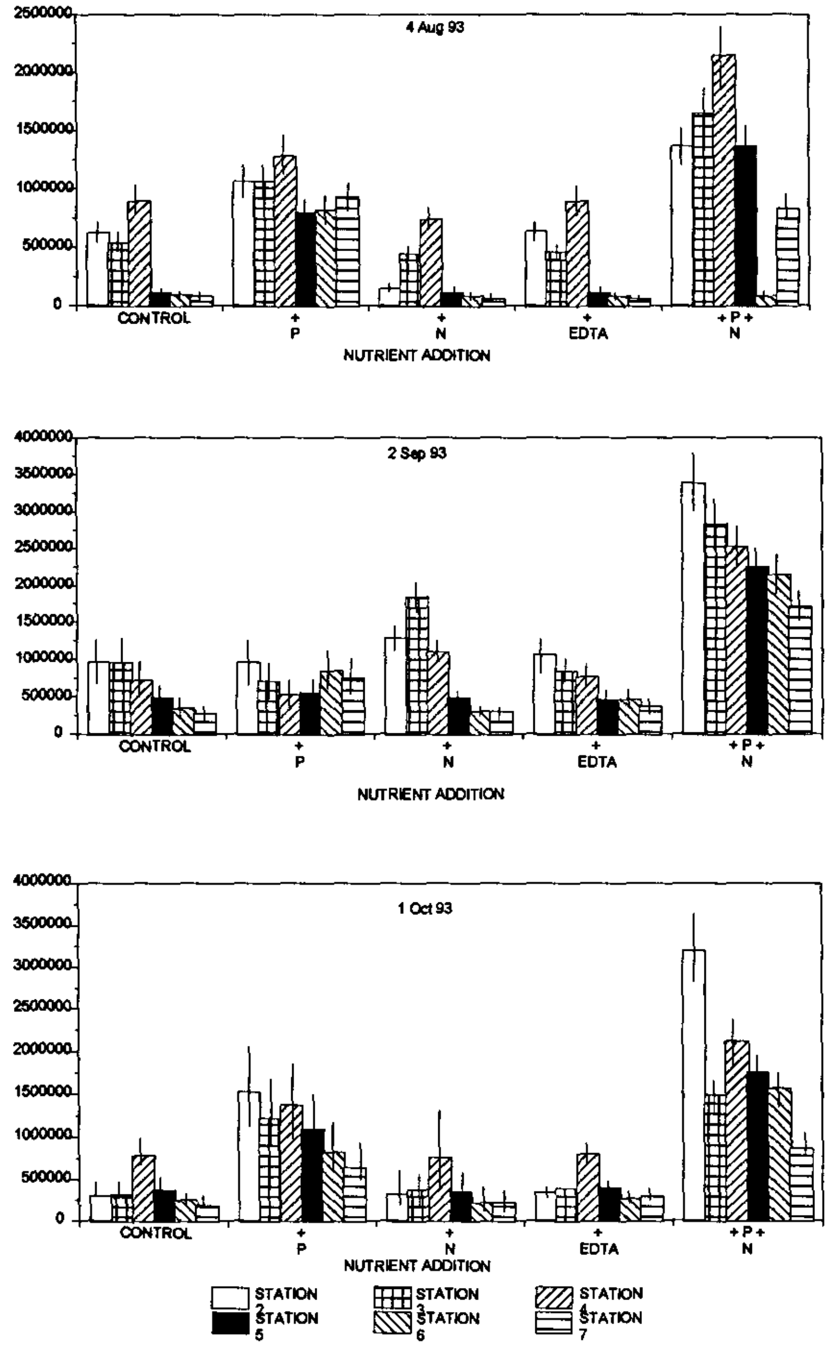



**Figure 69** Algal Assay : Bottle Test Results for Lake Tenkiller.

## Phytoplankton Distribution

### Phytoplankton Community Structure

Phytoplankton assemblages were temporally and spatially variable (**Table LVI**, **Figure 70**, Appendix B). Most genera were collected throughout the pelagic zone of the lake. Notable anomalies in the phytoplankton communities included dinoflagellate blooms of *Peridinium* spp. recorded in the Caney Creek Cove on 14 Feb 1992 and in the Sixshooter Creek Cove on 8 Mar 1993. Conspicuous dinoflagellate blooms occurred at stations 2, 3, and 4 on 25 Jun 1993 (**Figure 70**).

Dinoflagellate blooms are most common under calm, stratified conditions (Harris 1986). Binary fission is their most common form of reproduction and optimal cell division occurs predominantly nocturnally in the calm epilimnion of hard waters with high calcium content. (Harris 1986). These blooms are typical in the summer populations of productive systems (Reynolds 1984). This preference for calm water explains the locations of the blooms; coves are more protected from the wind. This also explains why dinoflagellate cells were more concentrated in the transition zone on 26 Jun 1993 as the upper end of the reservoir is narrower and thus often less wind-whipped than at lower stations as well as having the higher nutrient concentrations which are favored by dinoflagellate blooms.

All but 16 of the 47 genera found were ranked as organic pollution tolerant (Palmer 1969). Seven of the genera were ranked as clean water algae (Clesceri et al. 1989). The annual maximum biomass occurred in August of both years (**Figure 70**). All but one of the 47 genera were found during the summer, 30 were found in the spring, and 37 were found during the fall. Twenty-six of the genera were reported previously in a national eutrophication survey (Hern et al. 1978) and 16 were reported in an ecological investigation report by the Oklahoma Department of Wildlife Conservation (Summers 1961). Mean cell densities ranged from 32.6 cells/ml at station 2 on 8 Mar 93 to 14839 cells/ml at station 5 on 19 Aug 92. Although some average cell densities were surprisingly high, results were similar to average cell densities reported by Gakstatter and Katko (1986) in their Aug 85 assessment of the Illinois River and Tenkiller Ferry Lake. The greatest average cell densities occurred when blue-green algae were dominant. However, given the small cell size of most blue-green taxa, an increase in cell counts per mℓ may not necessarily correlate with an increase in biomass.

### Seasonal Trends in Community Structure

Phytoplankton community structure at all stations followed expected seasonal trends with spring blooms dominated by diatoms, and an early summer community composed primarily of green algae (**Figure 70**). Late summer and early fall communities were dominated by blue-green algae. The onset of cooler temperatures and the breakdown of stratification were followed by decreases in blue-green abundance and subsequent increases in green algae and diatom populations.

**Table LVI** Phytoplankton Genera Collected from Lake Stations.

| Phylum | Genera | Station |
|---|---|---|
| Chlorophyta | *Actinastrum spp.* | 2, 3, 4, 5, 6, 7 |
| | *Ankistrodesmus spp.* | 2, 3, 4, 5, 6, 7 |
| | *Chlamydomonas spp.* | 2, 3, 4, 5, 6, 7 |
| | *Chlorella spp.* | 2, 3, 4, 5, 6, 7 |
| | *Closterium spp.* | 2, 3, 5, 6, 7 |
| | *Coelastrum spp.* | 2, 3, 4, 5, 6, 7 |
| | *Cosmarium spp.* | 2, 4, 5, 6, 7 |
| | *Crucigenia spp.* | 2, 3, 4, 5, 6, 7 |
| | *Gleocystis spp.* | 2, 3, 4, 5, 6, 7 |
| | *Gonium spp.* | 2, 3, 4, 5, 6, 7 |
| | *Kirchnerella spp.* | 7 |
| | *Mougeotia spp.* | 4, 5 |
| | *Oedogonium spp.* | 2, 3, 4, 5, 6, 7 |
| | *Oocystis spp.* | 2, 3, 4, 5, 6, 7 |
| | *Pandorina spp.* | 2, 3, 4, 5, 6, 7 |
| | *Pediastrum spp.* | 2, 3, 4, 5, 6, 7 |
| | *Platydorina spp.* | 2, 3, 4, 5, 6, 7 |
| | *Rhizoclonium spp.* | 4, 7 |
| | *Richterella spp.* | 5, 6 |
| | *Scenedesmus spp.* | 2, 3, 4, 5, 6, 7 |
| | *Staurastrum spp.* | 2, 3, 4, 5, 6, 7 |
| | *Stephanoon spp.* | 3 |
| | *Tetraedron spp.* | 3, 4, 5, 6, 7 |
| | *Ulothrix spp.* | 2, 3, 4, 5, 6, 7 |
| Chrysophyta | *Asterionella spp.* | 2, 3, 4 |
| | *Cyclotella spp.* | 2, 3, 4, 5, 6, 7 |
| | *Cymbella spp.* | 2, 3, 5, 7 |
| | *Dinobryon spp.* | 3, 4 |
| | *Gomphonema spp.* | 2, 3, 5 |
| | *Mallomonas spp.* | 2, 3, 4, 5, 6, 7 |
| | *Melosira spp.* | 2, 3, 4, 5, 6, 7 |
| | *Navicula spp.* | 2, 3, 4, 5, 6, 7 |
| | *Synedra spp.* | 2, 3, 4, 5, 6, 7 |
| Cryptophyta | *Cryptomonas spp.* | 2, 3, 4, 5, 6, 7 |
| Cyanophyta | *Anabaena spp.* | 2, 3, 4, 5, 6, 7 |
| | *Aphanocapsa spp.* | 2, 3, 4, 5, 6, 7 |
| | *Chroococcus spp.* | 3, 4 |
| | *Lyngbya/Oscillatoria spp.* | 2, 3, 4, 5, 6, 7 |
| | *Merismopedia spp.* | 2, 3, 4, 5, 6, 7 |
| | *Microcystis spp.* | 2, 3, 4, 5, 6, 7 |
| | *Microspora spp.* | 2 |
| | *Sphaerocystis spp.* | 2, 3, 4, 5, 6, 7 |
| | *Spirulina spp.* | 2, 3, 4, 5, 6, 7 |
| Euglenophyta | *Euglena spp.* | 2, 3, 4, 5, 6, 7 |
| Pyrrhophyta | *Ceratium spp.* | 2, 3, 4, 5, 6, 7 |
| | *Gymnodinium spp.* | 2, 4, 5 |
| | *Peridinium spp.* | 2, 3, 4, 6, 7 |

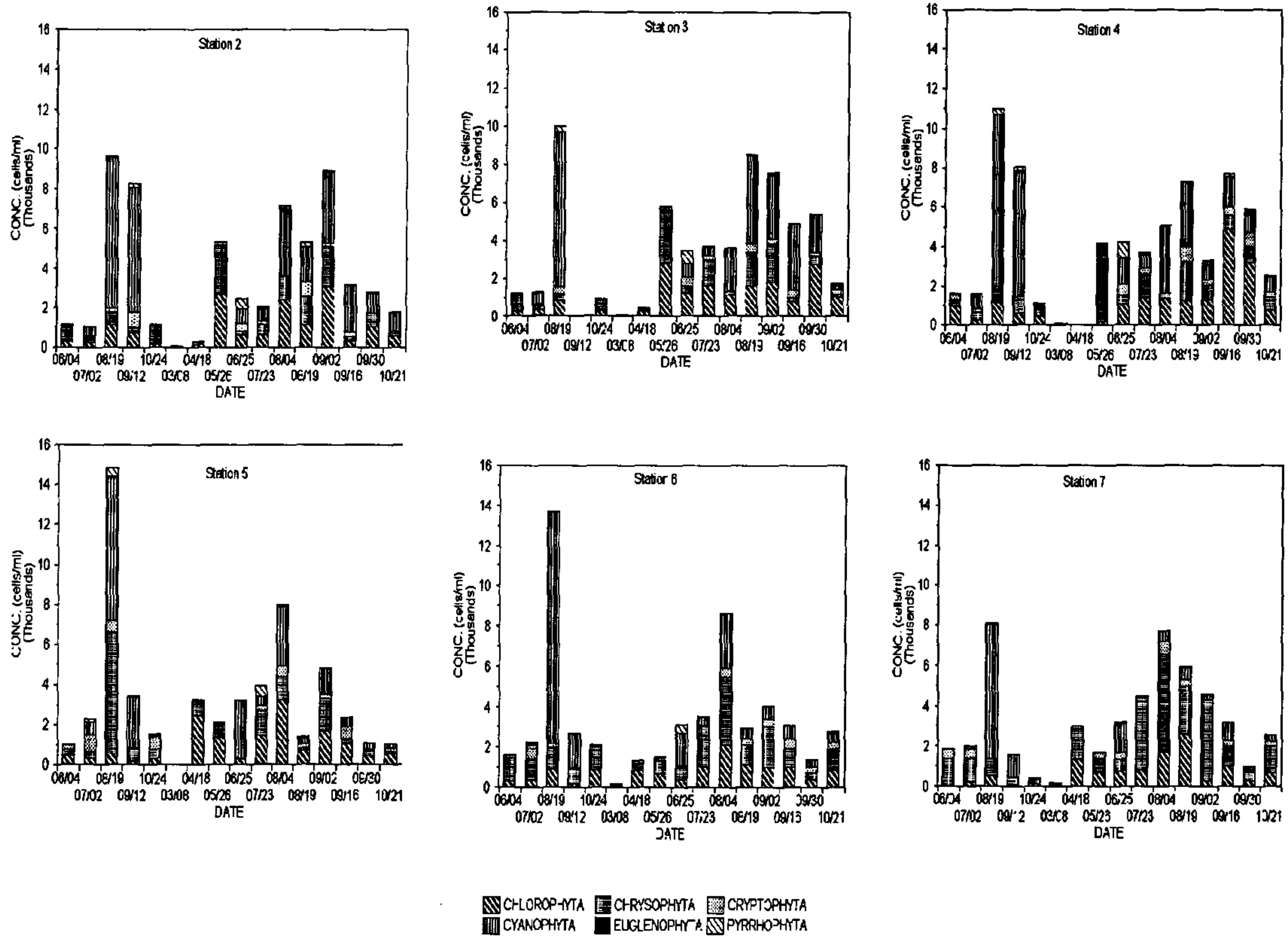



**Figure 70** Spatial and Temporal Variation in Tenkiller Phytoplankton Communities.

## Longitudinal Zonation of Phytoplankton Communities

Although the phytoplankton community structure at the upper end of the lake was typically different from that near the dam, no significant differences ($\alpha = 0.05$) were observed between average total cell count, species diversity (Shannon-Weaver 1949), ratio of centric to pennate diatoms, or percent blue-green algae between any of the six stations where phytoplankton were collected. It was suspected that the seasonal oscillations in the communities may have overshadowed the differences between stations and thus data were corrected for seasonal variation (Phillips et al. 1989) and reanalyzed. No statistically significant differences ($\alpha = 0.05$) were found between total cell count, species diversity, ratio of centric to pennate diatoms or percent blue-green algae between the six in-lake stations.

However, because statistical and ecological significance are not always coincidental, qualitative differences between phytoplankton communities in the transition and lacustrine zone were noted. Observable differences occurred in the predominant phyla among the different reservoir zones (determined as per physico-chemical parameters). In the transitional zone, blue-green algae were most often the dominant taxa (**Figure 70**). *Lyngbya/Oscillatoria, Spirulina, Microcystis* and *Merismopedia* accounted for 43.6, 43.7, and 48.1 % of summer collections and 34.5, 30.5, and 15.1 % of fall collections at stations 2, 3, and 4 respectively. Diatoms dominated most often in the lacustrine zone of the reservoir. *Cyclotella, Navicula, Synedra, Melosira* and *Cymbella* accounted for 26.3, 29.5, and 47.1 % of summer communities, 21.9, 31.5, and 48.7 % of spring communities, and 15.9, 34.3, and 43.2 percent of fall communities at stations 5, 6 and 7, respectively.

## Phytoplanktonic Community Structure as Related to Physico-chemical Parameters

Use of canonical correspondence analysis (CCA) to relate water quality parameters to phytoplankton community indices such as species diversity, chlorophyll *a* concentrations, total cell counts, ratio of centric to pennate diatoms, and percent blue-green algae abundance indicated that turbidity and Secchi depth correlated best with the fore-mentioned indices for stations 2 - 6 (**Table LVII**).

To test for effects of longitudinal zonation of physico-chemical parameters on phytoplankton community structure, CCA was performed on data from within the transitional zone and the lacustrine zone (**Table LVIII**). In the transition zone, the univariate F value for Secchi depth was not statistically significant so that parameter could not be correlated to algal community indices. Phytoplankton communities correlated best to trends in turbidity; however, nutrient trends (total phosphorus in particular) were also correlated with phytoplankton community indices.

**Table LVII** Canonical Loadings for Water Quality Parameters.

| PARAMETER | UNIVAR. F VALUE | CANONICAL CORRELATION | | | |
|---|---|---|---|---|---|
| | | 0.733 | 0.568 | 0.442 | 0.150 |
| Total Nitrogen | 3.065* | 0.317 | 0.597 | 0.613 | 0.409 |
| Total Phosphorus | 0.008** | -0.397 | 0.654 | 0.501 | -0.405 |
| Secchi Depth | 0.001** | 0.579 | -0.434 | -0.612 | -0.320 |
| Turbidity | 0.002** | -0.524 | 0.797 | -0.063 | 0.295 |

*: $\alpha = 0.05$ ; **: $\alpha = 0.01$; Multivariate F (largest root criterion) = 3.857**

This correlation suggested that turbidity influences phytoplankton productivity in the transition zone, rather than phytoplankton productivity influencing turbidity. That directionality of influence was supported by Thornton's definition of the transition zone where productivity is impacted by decreases in turbidity without excessive decreases in nutrient availability (Thornton et al. 1990).

Lacustrine phytoplankton communities were best correlated to trends in Secchi disk readings (**Table LVIII**). Again this correlation was expected given the mechanics of longitudinal zonation, suggesting that phytoplankton productivity influences Secchi depth, rather than Secchi depth directing phytoplankton productivity. Total nitrogen concentrations trends also were well correlated to phytoplankton community indices.

Phytoplankton Community Structure as Related to Nutrient Limitation

There was no statistical significance in either multivariate (largest root criterion F = 0.827, P = 0.911) F values or univariate F values for CCA comparing nutrient limitation assay results to phytoplankton community indices (**Table LIX**). Therefore, CCA did not show any significant correlation between nutrient limitation and any of the phytoplankton community indices.

However, monthly nutrient limitation results still can be compared to corresponding in-lake phytoplankton community structure. Limiting factors define phytoplankton community structure by allowing certain taxa to become most common based upon differential light, temperature, current, and nutrient requirements. The ability of blue-green algae to fix atmospheric nitrogen allows them to out-compete other taxa under conditions of nitrogen limitation, given adequate phosphorus concentrations. Given high nutrient concentrations, if phosphorus is the limiting nutrient, green algae are expected to dominate. If both nitrogen and phosphorus are limiting, co-dominance of green and blue-green algae can be expected (Miller et al. 1978). Correlation between nutrient limitation assay results and in-lake phytoplankton communities suggests nutrient limitation of phytoplankton productivity rather than limitation by some other factor.

**Table LVIII** Canonical Loadings for Zoned Water Quality Parameters.

| PARAMETER | UNIVAR. F VALUE | ZONE | CANONICAL CORRELATION | | | |
|---|---|---|---|---|---|---|
| | | | 0.850 | 0.767 | 0.674 | 0.340 |
| Total Nitrogen | 0.041* | TRANS | 0.523 | 0.186 | 0.747 | -0.366 |
| Total Phosphorus | 0.017* | TRANS | -0.560 | -0.339 | 0.747 | 0.120 |
| Secchi Depth | 0.087 | TRANS | 0.254 | 0.416 | 0.746 | 0.453 |
| Turbidity | 0.008** | TRANS | -0.701 | 0.394 | 0.558 | -0.205 |
| | | | CANONICAL CORRELATION | | | |
| | | | 0.773 | 0.632 | 0.563 | 0.182 |
| Total Nitrogen | 0.002** | LACUS | 0.062 | 0.959 | -0.190 | 0.200 |
| Total Phosphorus | 0.023* | LACUS | -0.471 | 0.537 | 0.251 | -0.654 |
| Secchi Depth | 0.001** | LACUS | 0.867 | -0.493 | 0.054 | 0.047 |
| Turbidity | 0.000** | LACUS | -0.552 | 0.428 | 0.686 | 0.203 |

*: $\alpha = 0.05$ ; **: $\alpha = 0.01$: Transitional Multivariate F (largest root criterion) = 3.070**, Lacustrine Multivariate F = 4.316**

**Table LIX** Canonical Loadings for Nutrient Limitation Assay Results..

| PARAMETER | UNIVARIATE F VALUE | CANONICAL CORRELATION |
|---|---|---|
| | | 0.327 |
| Total Cells | 0.061 | 0.189 |
| Chlorophyll *a* | 0.560 | -0.563 |
| Species Diversity | 0.008 | 0.067 |
| Centric : Pennate | 0.487 | 0.526 |
| % Blue-green | 0.051 | 0.173 |

Correlation between nutrient limitation results (**Figure 69**) and phytoplankton community structure (**Figure 70**) were better in the lacustrine zone than in the transition zone. Given nutrient limitation results, dominant taxa were as expected for stations 5 and 7 on 4 Aug 93, 6 and 7 on 2 Sep 93 and 5, 6, and 7 on 1 Oct 93. Dominant taxa were as expected for stations 2 and 4 on 2 Sep 93 and station 3 on 1 Oct 93. These results suggested that nutrients were the predominant factor controlling phytoplankton productivity at the lacustrine stations, but that other factors such as light and turbulence may have been equally important in limiting phytoplankton productivity in the transition zone. These results supported Thornton's definition of the reservoir zones (Thornton et al. 1990).

## Degradation of Tenkiller Ferry Lake

Changes in environmental conditions which are perceived by the public are not always supported by documentation. The EPA Clean Lakes Study on Beaver Lake, Arkansas, indicated that there had not been significant changes in the water quality of Beaver Lake between the 1974 EPA-NES and the 1992-93 EPA Clean Lakes (CL) Phase I Study (FTN 1992), contrary to public opinion. In fact, the NES report for Tenkiller indicated eutrophic conditions existed in 1974 (USEPA 1977). However, a comparison between median values from the NES report (n = 4), the 1985-86 USACE study (n = 9 - 16) (USACE 1988) and the 1992-93 CL Phase I Study (n = 11 - 22) indicated that significant changes have occurred in the water quality of Tenkiller Ferry Lake (**Table LX**).

Though no significant differences existed among the quartile distributions for Secchi disk, Turbidity, Total Nitrogen or Nitrate-Nitrogen of the 3 different studies, significant differences were detected among chlorophyll *a* and total phosphorus distributions in certain areas of the lake. Total phosphorus and chlorophyll *a* concentrations were significantly higher in 1985-86 than in 1974. Though these concentrations decreased somewhat between 1985-86 and 1992-93 at some CL stations (likely due to implementation of tertiary treatment at Tahlequah, OK waste water treatment plant and best management practices in the basin), total phosphorus concentrations remained significantly greater at CL stations 2 - 4 in 1992-93 than in 1974. These increases were manifested in significantly higher chlorophyll *a* concentrations from CL stations 3 - 7 between 1974 and 1992-93. Thus, Lake Tenkiller exhibited signs of degradation in the 1992-93 CL study.

## Conclusions and Discussion of Phytoplankton Analysis

Tenkiller Ferry Lake is a reservoir with high ambient nutrient concentrations primarily resulting from non-point source pollution. The reservoir was divided longitudinally into three zones as defined by physico-chemical parameters such as nutrient concentrations and turbidity. Use of biotic parameters to define longitudinal zonation met with limited success. Though chlorophyll *a* concentrations could be used to delimit reservoir zones, various community indices such as species diversity and percent blue-green algae could not be used to define zones. Other detectable differences in biotic parameters between zones included dominant phyla and occurrence of certain genera.

**Table LX** Median values for Limnological Parameters From Studies on Tenkiller Ferry Lake.

| **1974 USEPA NES Station** | **4** | **3** | **2** | **1** |
|---|---|---|---|---|
| Relative Distance[b] | 0.27 | 0.65 | 0.79 | 0.98 |
| Secchi Disk (meters) | 0.76 | 1.67 | 2.03 | 2.03 |
| Nitrate-N (mg/ℓ) | 0.35 | 0.41 | 0.39 | 0.43 |
| Total Nitrogen (mg/ℓ) | 1.05 | 0.91 | 0.79 | 1.11 |
| Total Phosphorus (mg/ℓ) | 0.05 | 0.04 | 0.03 | 0.04 |
| Chlorophyll *a* (µg/ℓ) | 9.50 | 6.85 | 5.30 | 4.50 |

| **'85-86 USACE Station** | **14** | **13** | **11** | **10** | **9** | **8** | **7** | **6** | **5** | **4** | **3** | **2** | **1** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Relative Distance[b] | 0.01 | 0.24 | 0.32 | 0.37 | 0.46 | 0.52 | 0.57 | 0.68 | 0.74 | 0.77 | 0.80 | 0.92 | 0.99 |
| Secchi Disk (meters) | 0.70 | 0.60 | 0.80 | 1.00 | 1.10 | 1.30 | 1.25 | 1.35 | 1.65 | 1.80 | 1.80 | 2.10 | 2.10 |
| Nitrate-N (mg/ℓ) | 0.36 | 1.00 | 0.48 | 0.14 | 0.53 | 0.30 | 0.19 | 0.33 | 0.60 | 0.37 | 0.28 | 0.68 | 0.52 |
| Total Phosphorus | 0.21* | 0.17* | 0.15* | 0.12* | 0.14* | 0.14* | 0.10* | 0.10* | 0.06 | 0.09* | 0.10* | 0.10* | 0.10* |
| Chlorophyll *a* (µg/ℓ) | 5.30** | 30.20* | 25.10* | 19.55 | 17.00 | 14.00* | 11.25 | 10.50 | 12.00* | 11.80* | 11.85* | 9.70* | 10.90* |
| Turbidity (NTU) | 8.0 | 8.0 | 6.0 | 5.5 | 4.0 | 4.0 | 3.0 | 3.0 | 2.5 | 3.0 | 2.0 | 2.0 | 2.0 |

| **'92-93 USEPA CL Station** | **1** | **2** | **3** | **4** | **5** | **6** | **7** |
|---|---|---|---|---|---|---|---|
| Relative Distance[b] | 0.01 | 0.26 | 0.27 | 0.33 | 0.52 | 0.72 | 0.97 |
| Secchi Disk (meters) | --- | 0.85 | 0.86 | 0.90 | 1.40 | 2.05 | 2.30 |
| Nitrate-N (mg/ℓ) | 1.18[c];** | 0.46 | 0.36 | 0.34 | 0.21 | 0.30 | 0.30 |
| Total Nitrogen (mg/ℓ) | 2.18[c] | 1.16 | 1.23 | 1.17 | 0.79 | 0.74 | 0.74 |
| Total Phosphorus (mg/ℓ) | 0.12[c];* | 0.08*;** | 0.08* | 0.07* | 0.05 | 0.02 | 0.02 |
| Chlorophyll *a* (µg/ℓ) | 2.55 | 28.60 | 28.01* | 28.66* | 15.62* | 11.62* | 8.95* |
| Turbidity (NTU) | 8.7 | 6.3 | 8.3 | 5.8 | 4.5 | 2.3 | 2.1 |

[b]: Relative distance from dam calculated as % of total thalweg length with dam = 1.00; [c]: signif. > than any value from NES study, however no comparable station between NES and CL study; *: signif. > than NES study; **: signif. different from USACE study

Trophic classification based upon nutrient concentrations and chlorophyll *a* concentrations categorized Tenkiller as a eutrophic system. However, the degree of eutrophy was variable among zones, with the lacustrine zone being less eutrophic than the transition and riverine zones. Eutrophy of the transition and riverine zone was variable depending upon which parameters were used. The riverine zone was less eutrophic than the transition zone based upon biotic parameters and the transition zone was less eutrophic than the riverine zone based upon abiotic parameters. Phytoplankton seasonal succession patterns were also most like those of eutrophic systems. Blue-green algae were relatively abundant in the phytoplankton community during summer and fall.

Potential nutrient limitation also was different among reservoir zones. Nutrient limitation was more variable in the transition zone, where primarily both nitrogen and phosphorus were limiting. However, phosphorus was the primary limiting nutrient in the lacustrine zone. Primary limiting factors also differed between the two zones. Nutrients were the primary limiting factor in the lacustrine zone, but differences between algal assay results and phytoplankton community structure indicates that factors such as light and turbulence may be equally if not more important than nutrients in limiting productivity in the transition zone.

Given the high ambient nutrient concentrations under the influence of primarily non-point source pollution, the limiting nutrients and the current phytoplankton community patterns, it has become essential that action be taken to slow the degradation of the reservoir. This could be done most effectively by controlling phosphorus discharge into the watershed through the use of best management practices. Though nitrogen concentrations were also high, nitrogen is less easily manipulated and thus nitrogen control is a less feasible option. Should control of nitrogen sources into the basin such as animal wastes and fertilizer runoff be targeted, it is essential that phosphorus control also be exercised lest the phosphorus to nitrogen ratio become reduced to a level which would stimulate dominance of blue-green algae.

## 15. Secchi Disk Depth and Suspended Solids Data

The Secchi disk transparency of Lake Tenkiller has been discussed in the previous sections. Therefore, only the data and basic descriptive statistics are presented here. During the CLP 1992-93 study, Secchi disk transparency ranged from a minimum of 0.20 m at station 2 to a maximum of 5.5 m at station 7. Median values were 0.85, 0.86, 0.90, 1.4, 2.1, and 2.3 m at stations 2, 3, 4, 5, 6, and 7, respectively (Figure 71). Clearly, the headwater sediment load affects the upstream estimates and gives way to a higher, more biologically-mediated Secchi disk measurement in the lacustrine zone.

Secchi disk has not changed significantly since the EPA-NES 1974 study (Figure 71). However, equivalent trophic status can not be assumed. It was illustrated that phosphorus and nitrogen fractions have increased with an increase in chlorophyll density in the transition and lacustrine zones of the reservoir. Implying trophic classification based upon Secchi disk transparency to a system that has high inorganic turbidity would be erroneous, and thus no inferences are made here.

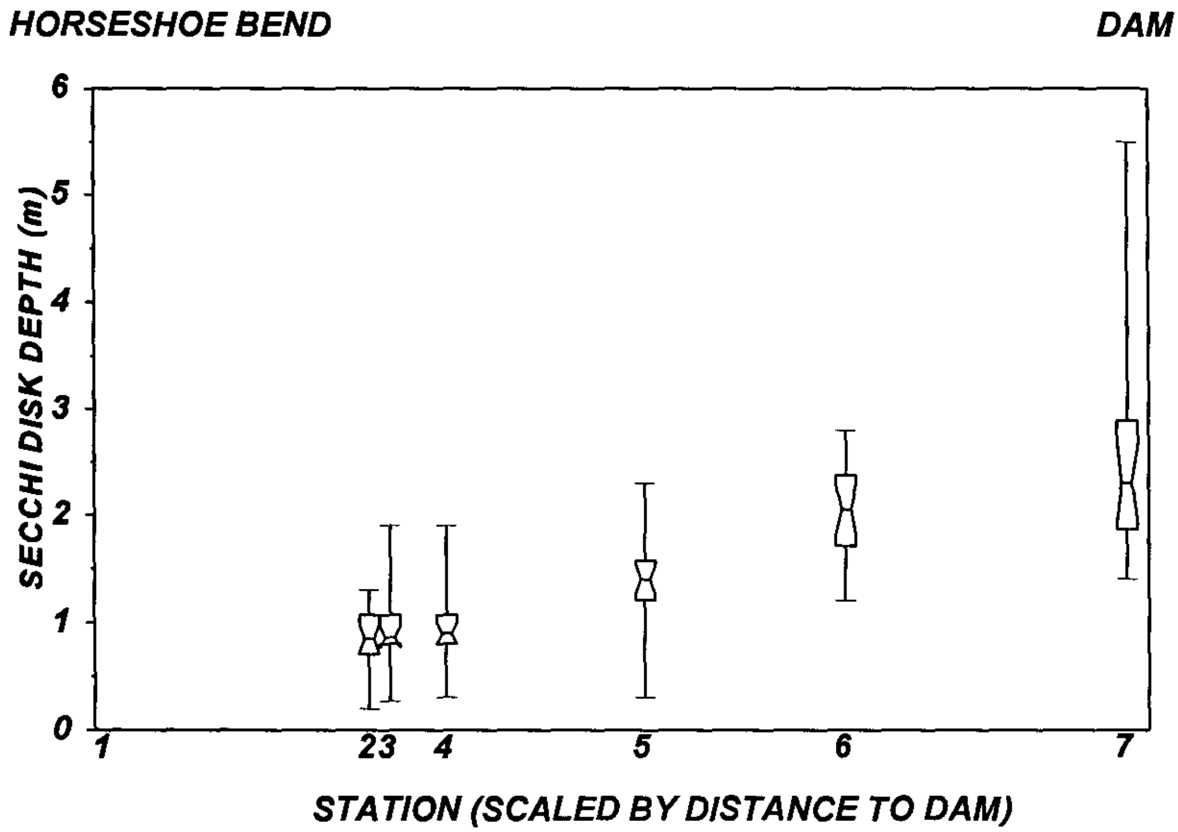



**Figure 71** Secchi Disk Transparency of Lake Tenkiller for CLP 1992-93.

### 16. Estimate of Shoreline and Bottom Impacted by Vascular Plants

On all observations during this study, macrophytic coverage was virtually nonexistent. Neither the EPA-NES 1974 study nor the USACE 1985-86 survey mentioned any macrophyte problem. No other study or data source used in this report indicated any excessive macrophyte growths. Lake Tenkiller has relatively steep sides and rocky substrates in the littoral zone. Presumably, this arrangement coupled with fluctuating water levels ostensibly has not afforded macrophytic bed development.

### 17. Identification of Predominant Vascular Plant Species

As stated in the previous section, no vascular plant dominated Lake Tenkiller during this study. However, on one occassion we did observe *Wolffia* sp. along the shoreline near station 4. Previous studies did not report any significant macrophyte developments. For a list of common plants refer to the biological and historical prepared for USACE (1973).

### 18. Public Health Monitoring Data

#### i. Standard Bacteriological Analyses

Fecal coliform bacteria densities are presented in Figure 72. While some of the maximum densities measured during the USACE 1985-86 survey exceeded state health criteria, the data collection methods were approximately monthly. The criteria set forth for surface waters were formulated based upon a 30 day period of sampling (see standards section in part 1). Therefore, no conclusions of standard violations can be made. All samples had minima of less than 10 colonies per 100 ml. The lower portions of the lake did not appear in violation of any standard criteria at any time samples were collected and analyzed. The only ostensible violation was at the extreme headwaters (near Horseshoe Bend) and on a sporadic basis.

#### ii. Fish Flesh

Fish tissue analysis was conducted on channel catfish (*Ictalurus punctatus*) collected from stations 2, 5, and 6. Results from the chlorinated hydrocarbon pesticide analysis are presented in Table LXI with lipid content data presented in Table LXII. While historical residues of these contaminants have been documented by OSDH (see historical data), body burdens appear to be decreasing and do not represent an ecological threat at this time.

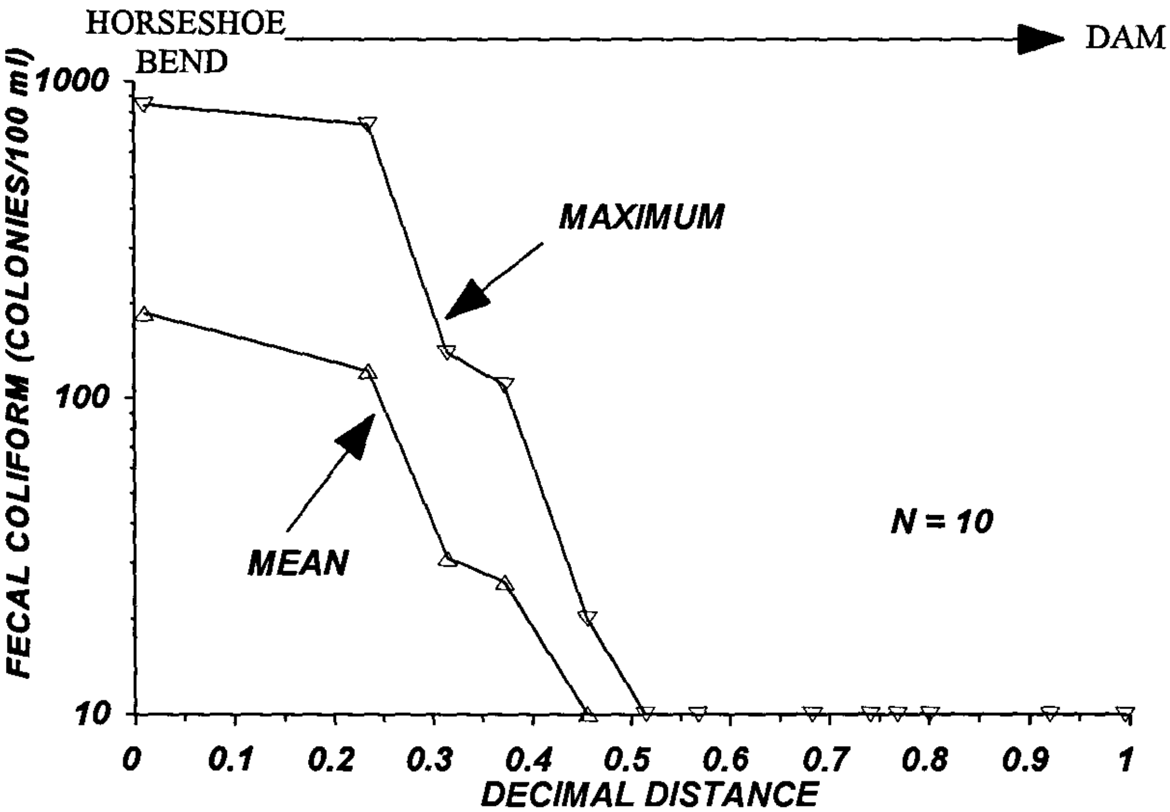



**Figure 72** Fecal Coliform Bacteria for Lake Tenkiller in 1985-86.

**Table LXI** Fish Tissue Pesticide Residue Results (mg/kg).

| Sample | Lindane | Heptachlor | Aldrin | Dieldrin | Endrin | DDD | DDT | Methoxychlor |
|---|---|---|---|---|---|---|---|---|
| 1 Fish 1 | 0.0043 | 0.0045 | ND | ND | ND | 0.0287 | 0.1079 | 0.1073 |
| 1 Fish 2 | ND | ND | 0.0017 | ND | ND | ND | 0.0676 | 0.0622 |
| 1 Fish 3 | 0.0030 | 0.0035 | 0.0042 | ND | ND | 0.0186 | 0.0298 | 0.3915 |
| x | 0.0036 | 0.0040 | 0.0029 | 0 | 0 | 0.0236 | 0.0684 | 0.1870 |
| s | (0.002) | (0.002) | (0.002) | 0 | 0 | (0.039) | (0.001) | (0.001) |
| 2 Fish 1 | 0.0279 | 0.0102 | 0.0100 | 0.0303 | ND | ND | 0.0162 | ND |
| 2 Fish 2 | 0.0146 | 0.0110 | 0.0099 | 0.0387 | ND | ND | 0.0096 | ND |
| 2 Fish 3 | 0.0151 | 0.0099 | ND | 0.0295 | ND | ND | ND | 0.1004 |
| x | 0.0192 | 0.0104 | 0.0099 | 0.0328 | 0 | 0 | 0.0129 | 0 |
| s | (0.007) | (0.000) | (0.006) | (0.005) | 0 | 0 | (0.008) | 0 |
| 3 Fish 1 | 0.0305 | 0.0089 | 0.0055 | 0.0749 | ND | ND | ND | 0.4085 |
| 3 Fish 2 | 0.0106 | 0.0116 | 0.0071 | 0.0478 | ND | ND | 0.0127 | 0.0613 |
| 3 Fish 3 | 0.0215 | 0.0097 | 0.0064 | 0.0584 | ND | ND | 0.0955 | 0.1286 |
| x | 0.0208 | 0.0100 | 0.0063 | 0.0604 | 0 | 0 | 0.0541 | 0.1994 |
| s | (0.009) | (0.000) | (0.000) | (0.014) | 0 | 0 | (0.052) | (0.184) |
| 4 Fish 1 | 0.0264 | 0.0131 | 0.0115 | ND | ND | ND | ND | ND |
| 4 Fish 2 | 0.0355 | 0.0136 | 0.0195 | ND | ND | ND | ND | ND |
| 4 Fish 3 | 0.0414 | 0.0136 | 0.0321 | ND | ND | ND | ND | ND |
| x | 0.0334 | 0.0134 | 0.0211 | 0 | 0 | 0 | 0 | 0 |
| s | (0.007) | (0.000) | (0.010) | 0 | 0 | 0 | 0 | 0 |
| 5 Fish 1 | 0.0206 | 0.0068 | 0.0113 | ND | ND | ND | 0.0110 | ND |
| 5 Fish 2 | 0.0061 | 0.0035 | 0.0133 | ND | ND | ND | 0.0005 | ND |
| 5 Fish 3 | 0.0695 | 0.0096 | 0.0134 | ND | ND | ND | ND | ND |
| x | 0.0321 | 0.0066 | 0.0126 | 0 | 0 | 0 | 0.0057 | 0 |
| s | (0.033) | (0.003) | (0.001) | 0 | 0 | 0 | (0.006) | 0 |
| 6 Fish 1 | 0.0174 | 0.0218 | ND | 0.0348 | ND | ND | 0.2576 | 0.0543 |
| 6 Fish 2 | 0.0081 | 0.0349 | 0.0214 | 0.0493 | ND | 0.0235 | 0.2128 | 0.0751 |
| 6 Fish 3 | 0.0102 | 0.0255 | 0.0379 | 0.0246 | ND | 0.0013 | 0.1576 | 0.0348 |
| x | 0.0119 | 0.0274 | 0.0296 | 0.0246 | 0 | 0.0083 | 0.2093 | 0.0547 |
| s | (0.005) | (0.007) | (0.019) | (0.024) | 0 | (0.013) | (0.003) | (0.037) |

**Table LXII** Lipid Content and Sex of Fish Specimens (*Ictalurus punctatus*) for Pesticide Residue Analysis.

| Fish ID | Station | Length (mm) | Sex | % Lipid |
|---|---|---|---|---|
| 1 | 5 | 153 | M | 1.61 |
| 2 | 2 | 305 | M | 4.17 |
| 3 | 5 | 385 | M | 3.6 |
| 4 | 6 | 396 | M | 4.57 |
| 5 | 6 | 285 | M | 5.3 |
| 6 | 2 | 462 | F | 6.12 |

## 11. BIOLOGICAL RESOURCES OF THE LAKE - IDENTIFICATION AND DISCUSSION - Fish Population and Ecological Relationships

Ecological structure and function is combined with the fish population analysis in the following section. Phytoplankton and abiotic conditions have been discussed earlier and hence the actual data are not presented here. However, these data are occasionally referenced.

### Sampling Design

Data from three of the sampling stations, station 2, station 5, and station 6, (chosen to represent the reservoir's zones) were used for this portion of the study. Transects across these stations were established. The data collected for this study incorporated six, two-day sampling events on the following dates: 18-19 Aug, 2-3 Sep, 16-17 Sep, 30 Sep-1 Oct, 18-19 Oct, and 21-22 Oct 1993. These dates were chosen to avoid seasonal bias in pelagic fish assemblages and related distributions (longitudinally) due to reproductive migrations which occur in spring (Howell 1945, Walburg et al. 1971, Adornato 1990). Each sampling event entailed two replicate days of biological sampling and one water quality monitoring event (except 18-19 October 93).

### Plankton

Four net-zooplankton samples were collected randomly along each of the three station transects within the pelagic zone. A 12.0 cm (mouth diamater) Wisconsin Plankton Net with number 20 nylon bolting cloth was used to make a bottom to surface vertical tow at an anchored position. Samples were fixed in neutral formalin (Lind 1985) and placed in cold storage until analysis.

Three of the four zooplankton samples collected for each transect were analyzed. Samples were identified (Edmondson 1958, Pennak 1978) and enumerated via random strip counts made on sample aliquots in a Sedgewick-Rafter cell using a compound, binocular microscope at 100X magnification (Lind 1985). Counts were made for each aliquot until changes in moving averages were insignificant or until approximately 300 individuals were counted. Longitudinal trends in density and relative frequency of major net zooplankton were determined from replicate averages for all stations and dates. Generic trends were derived using dates only for which a capture was recorded to alleviate potential effects of more significant temporal variation.

### Fish

To assess semi-instantaneous pelagic fish abundance, sonar transects were conducted using a Lowrance X-15 recording sonar device (Thorne 1983, O'Brien et al. 1984, Wilde and Paulson 1989). The sonar was operated at the factory set frequency of 192 KHz with a beam angle of 20°. Care was taken to ensure that operation of the sonar (i.e., control settings, transducer position) and boat speed (app. 7 km/h) were standardized throughout the study. Station transects were echosounded once within 1.5 hr of each other for each sampling date between 1500 and 1900 hr. Echograms were analyzed by counting all discernable targets and expressed as a function of linear distance across the respective transect.

In conjunction with the sonar, gillnetting was conducted to directly assess the pelagic fish assemblage. Efficient experimental design necessitated the development of a portable gillnet system which could be randomly used along station transects and afford standardized, inter-transect comparison of catch/effort (C/f) irrespective of heterogeneity in vertical distribution. The resulting design emulates the traditional horizontal, experimental gillnet stretched in a vertical plane. Use of this design afforded investigation of vertical (i.e., entire water column) and horizontal trends in fish distribution.

The vertical "diving" gillnet consisted of a series of four, adjacent, vertical panels of 1, 1 1/4, 1 1/2, and 1 3/4 inch bar mesh monofilament netting (size #69); all netting was depth stretched to accommodate maximum achievable dorso-ventral extension and was bordered with 3/8 inch polypropylene line. The flotation was constructed of 1 inch schedule 40 pvc pipe equipped with 5 six inch pvc sponge floats; 3/8 inch re-bar was used for the leadline. Additional lateral anchors of variable lengths of weighted polypropylene line were included.

One vertical net was randomly used (accommodating damaging structure) along each of three study transects (all within 1 hr) between 1200 to 1800 hr. Nets were allowed a minimum 4-hr soak-time before retrieval. Upon retrieval, fish were identified and length, weight, and vertical position recorded. Net captures (both as number and wet weight) were converted to standardized catch per unit efforts (C/f) to facilitate inter-transect comparison. Total and species-specific gillnet yields were analyzed for longitudinal tendencies.

Statistical Analyses

Differences among sample stations for all data were assessed with the Kruskal-Wallis one-way analysis of variance (Zar 1974). This test is distribution-free and often applicable where assumptions for the parametric ANOVA are violated (e.g., non-normal data, unequal variances) or unknown . However, if data meet assumptions for parametric analysis, the test is only 95.5 % as powerful (i.e., more conservative) and will be less likely to show significance (Sokal and Rohlf 1973, Zar 1974).

Echogram (all), gillnet (all), and selected net zooplankton data (total and general taxa) were analyzed for all sample dates (i.e., n=12 for each station except n=11 for Station 2 for both zooplankton and echogram data). A value of "0" was assigned in instances when capture was not recorded for a given station and date if sampling effort occurred; this was almost exclusive to species-specific gillnet data. This approach is conservative and may fail to yield significance when temporally truncated data (replicated only for dates on which captures were recorded for any station) will. The latter technique was used when analyzing trends in zooplankton data as significant temporal (i.e., monthly) fluctuations in these parameters have been demonstrated and were expected (Wetzel 1983). All ANOVA's and plots (except profiles) were performed with the computer statistical program SYSTAT™ (Wilkinson 1991)

Upon determination of significant trends, stations were analyzed for pairwise differences at an imposed alpha of 0.05. In instances of equal replication, a nonparametric analogue of the Newman-Keuls multiple range test (Zar 1974) was employed. For unequally replicated data, the Newman-Keuls test with modified standard error calculation (Zar 1974) was implemented.

Results and Discussion

Plankton

Longitudinal tendency in total net zooplankton was evident, although not statistically significant (P = 0.11). Total density peaked at station 5 with similar variability exhibited among stations (**Figure 73**). This trend supports the notion of highest densities at transitional localities and may, in part, account for low chlorophyll at this station.

Densities of general net zooplankton groups were variable among stations (**Figure 74**). A total of 38 net zooplankton taxa were collected over the study period (**Table LXIII**). Rotifers were the most abundant and peaked at station 5; however, the trend was not significant (P = 0.07). Copepoda and other net zooplankters also peaked at Station 5, while Cladocerans increased downlake. In all cases, trends were not significant (P = 0.36, P =

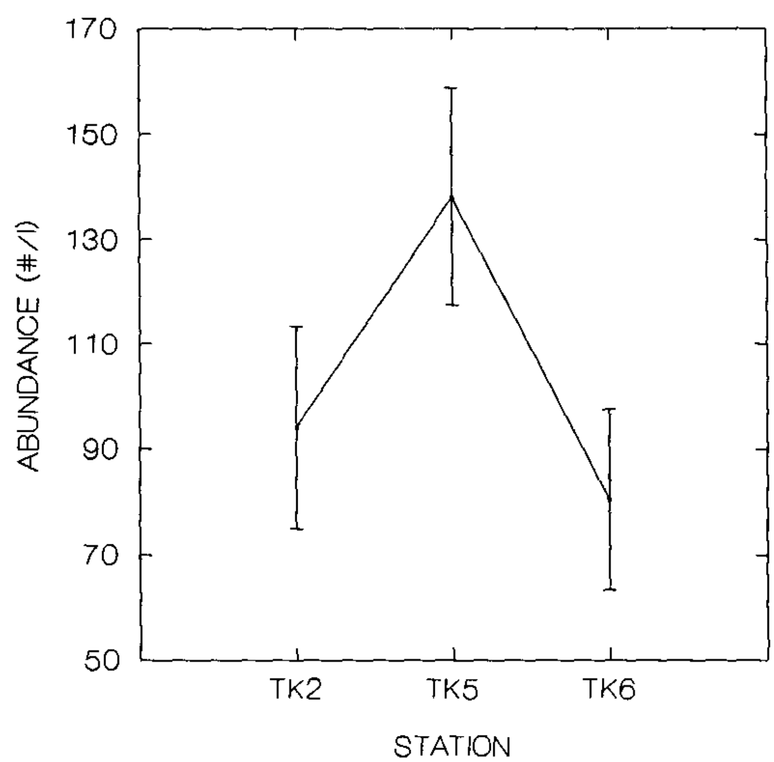



**Figure 73.** Longitudinal Trend in Total Net Zooplankton Density (average +/- standard error of the mean).

0.49, and P = 0.09, respectively).

As total density does not completely reflect changes in community structure, relative frequencies were computed for the various taxonomic groups at sample stations (**Figure 75**). Rotifers dominated zooplankton assemblages lakewide declining downlake in relative frequency. This trend was significant (P = 0.02) with Station 6 being significantly different

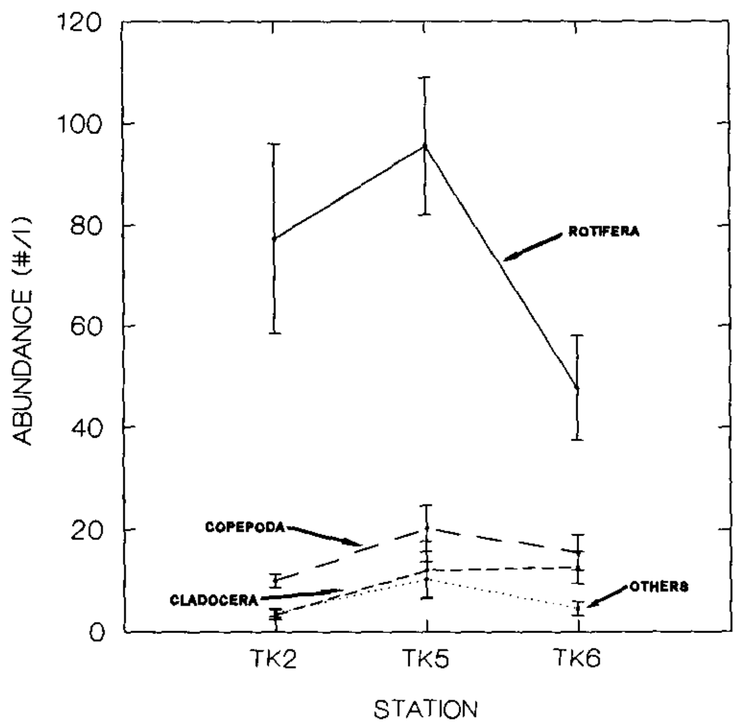



**Figure 74.** Longitudinal Trends in Densities of Major Net Zooplankton Groups (average +/- SEM).

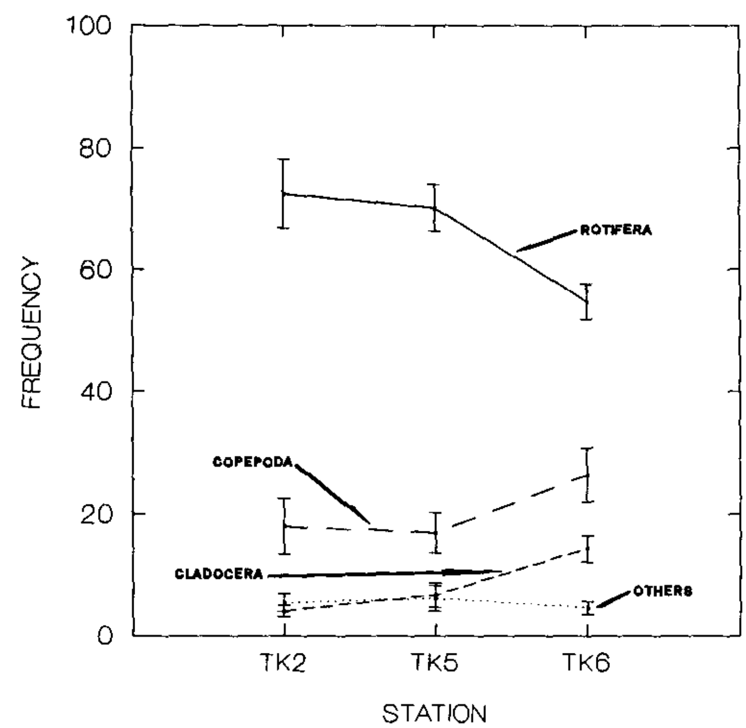



**Figure 75.** Longitudinal Trends in Relative Frequency of Major Zooplankton Taxa (average +/- SEM).

**Table LXIII.** Net Zooplankton Taxa Collected in Lake Tenkiller.

| ROTIFERA | CLADOCERA | COPEPODA | OTHER |
|---|---|---|---|
| *Ascomorpha* | *Bosmina* | Calanoida | Chaoborus |
| *Asplachna* | *Ceriodaphnia* | Cyclopoida | Ciliata |
| *Brachionus* | *Daphnia* | *Ergasilus* | Difflugia |
| *Colurella* | *Diaphanosoma* | juveniles | Gyrinidae |
| *Filinia* | *Latona* | | Nematoda |
| *Gastropus* | *Moina* | | Ostracoda |
| *Hexarthra* | *Pleuroxus* | | *Tanypus* |
| *Kellicotia* | *Polyphemus* | | |
| *Keratella* | | | |
| *Lecane* | | | |
| *Monostyla* | | | |
| *Platyias* | | | |
| *Ploesoma* | | | |
| *Polyarthra* | | | |
| *Syncheata* | | | |
| *Testudinella* | | | |
| *Trichocerca* | | | |
| Rotifer sp. 1 | | | |
| Rotifer sp. 2 | | | |

from the remaining two stations. Copepods were the next most frequent taxa of net zooplankton comprising approximately 18, 17, and 26 percent of samples from headwaters downlake, respectively. Although percent occurrence increased at station 6, the trend was not statistically significant ($P = 0.18$). Cladocerans exhibited an overall increase in relative frequency downlake. This trend was highly significant ($P < 0.01$) with station 6 being significantly different from the remaining two stations. Remaining taxa showed no appreciable longitudinal tendency in percent occurrence ($P = 0.93$).

Variation existed in the longitudinal trends of density of specific rotifers (**Figure 76** and **Figure 77**). Several of the taxa either decreased downlake or peaked at station 5. Statistically significant trends included those for *Hexarthra* ($P = 0.02$; station 2 station 5 station 6), *Syncheata* ($P = 0.02$; station 2 station 5 station 6), *Filinia* ($P = 0.02$; erroneous comparison results) *Asplanchna* ($P = 0.02$; erroneous comparison results), and *Kellicottia* ($P < 0.01$; station 2 station 5 station 6).

Copepoda densities also varied among stations (**Figure 78**). Juveniles were most abundant overall and peaked at station 5 ($P = 0.75$). Cyclopoida increased significantly ($P = 0.01$) to station 5, while little change occurred to station 6; station 2 was significantly different from the remaining two stations. The parasitic cyclopoid *Ergasilus* peaked slightly at station 5, but the trend was not significant ($P = 0.23$). Calanoida were almost invariable among stations ($P = 0.73$).

Strong horizontal tendency in cladocerans were apparent (**Figure 79**). Most abundant, *Bosmina* density increased significantly ($P < 0.01$); all stations were significantly different from each other. *Diaphanosoma* and *Daphnia* both exhibited non-significant ($P = 0.11$ and $P = 0.88$, respectively) peaks in density at station 5. *Moina* densities declined non-significantly ($P = 0.33$) downlake. *Ceriodaphnia* densities were not significantly different among stations ($P = 0.22$).

Densities of other net zooplankton were also heterogenous among stations (**Figure 80**). *Difflugia* and ciliates peaked in density at station 5, but neither trend was statistically significant ($P = 0.25$ and $P = 0.12$, respectively). The downlake decline in *Chaoborus* density was significant ($P = 0.02$), but erroneous comparison results preclude report of exact differences. *Tanypus* exhibited a similar and significant ($P = 0.03$) trend with station 2 different from the remaining stations.

## Fish

Total pelagic target density (TPTD) declined to lowest value at station 5 (**Figure 81**). This trend was highly significant ($P = 0.01$) with station 2 significantly different from the remaining two stations. As echograms potentially are a more holistic means of transect assessment than discreet net samples, these results are suggested to be the more reliable reflection of the longitudinal trend in total (i.e., species undiscernible) fish density. It must be noted, however, target strength (and thus discernability) is dependent on target size. Therefore, results are reflective of trends in individuals of targetable size and density possibly excluding small species and juvenile stages.

Total C/f declined downlake (**Figure 82**), although the trend was not significant ($P = 0.47$).

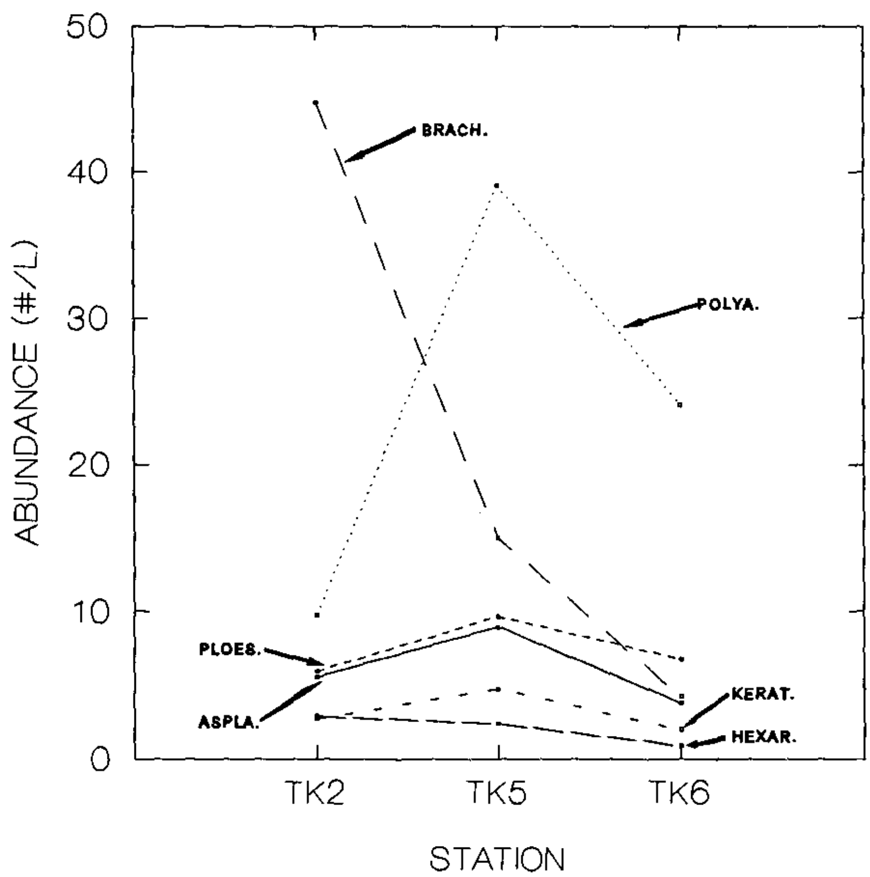



**Figure 76.** Longitudinal Trends in Rotifer Densities.

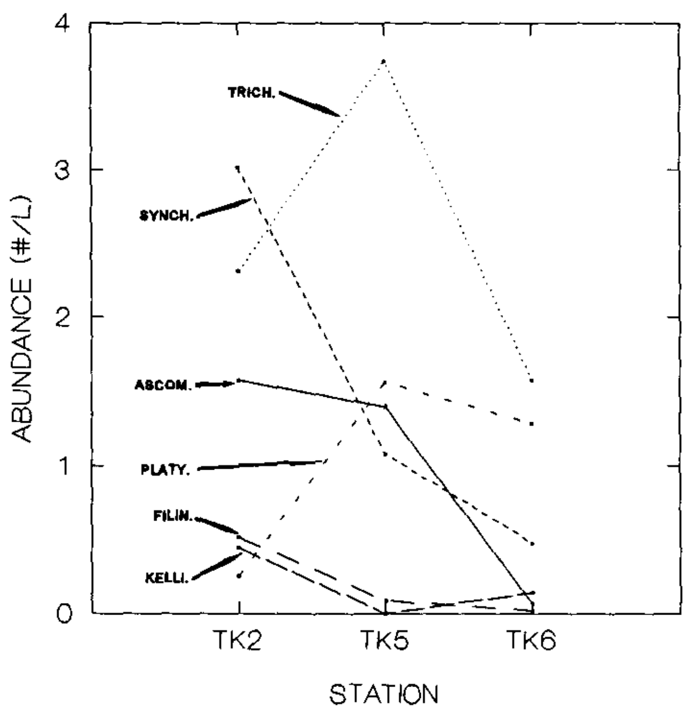



**Figure 77.** Longitudinal Trends in Rotifer Densities.

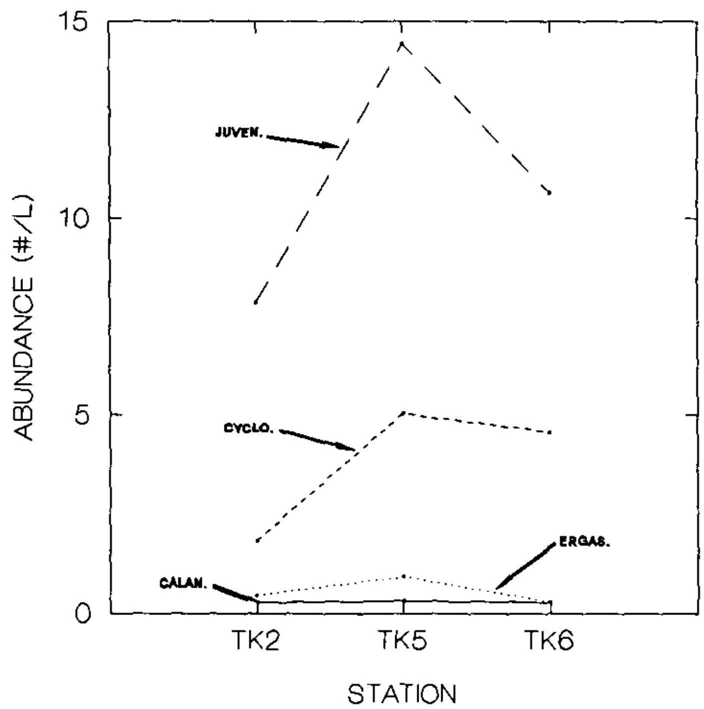



**Figure 78.** Longitudinal Trends in Copepod Densities.

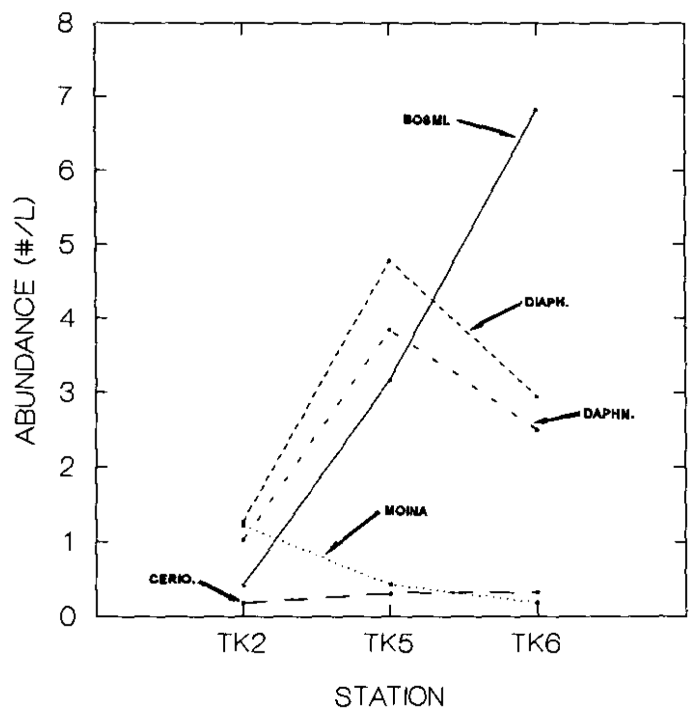



**Figure 79.** Longitudinal Trends in Cladoceran Densities.

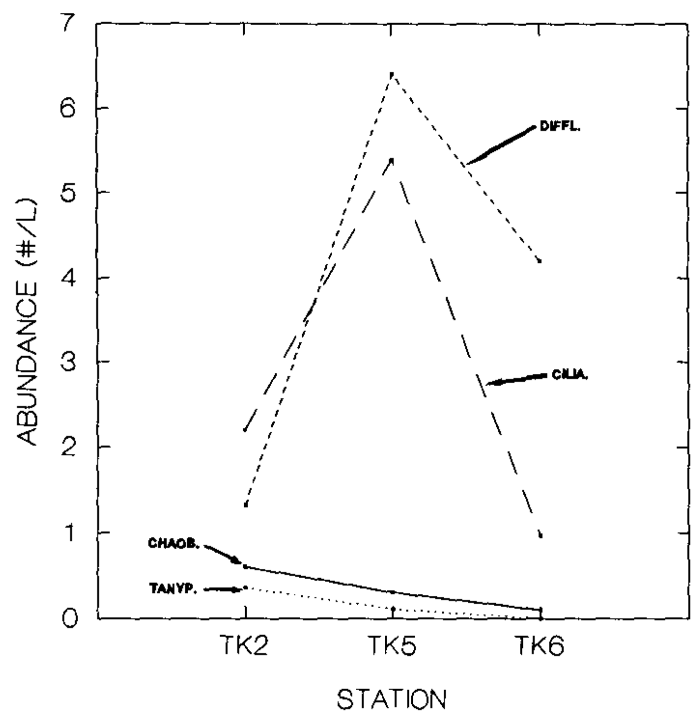



**Figure 80.** Longitudinal Trends in Remaining Net Zooplankton Densities (average +/- SEM).

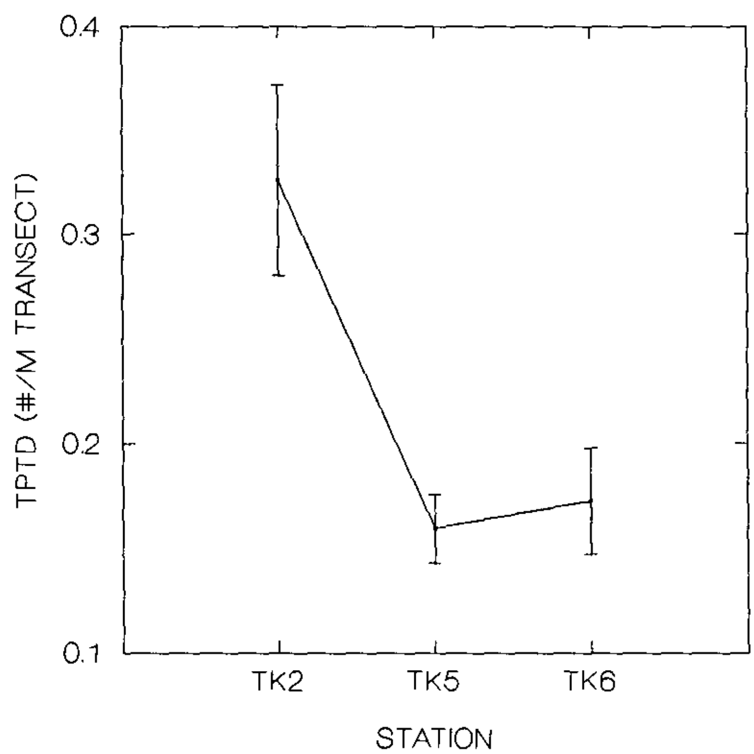



**Figure 81.** Longitudinal Trend in Total Pelagic Target Density (TPTD) (average +/- SEM).

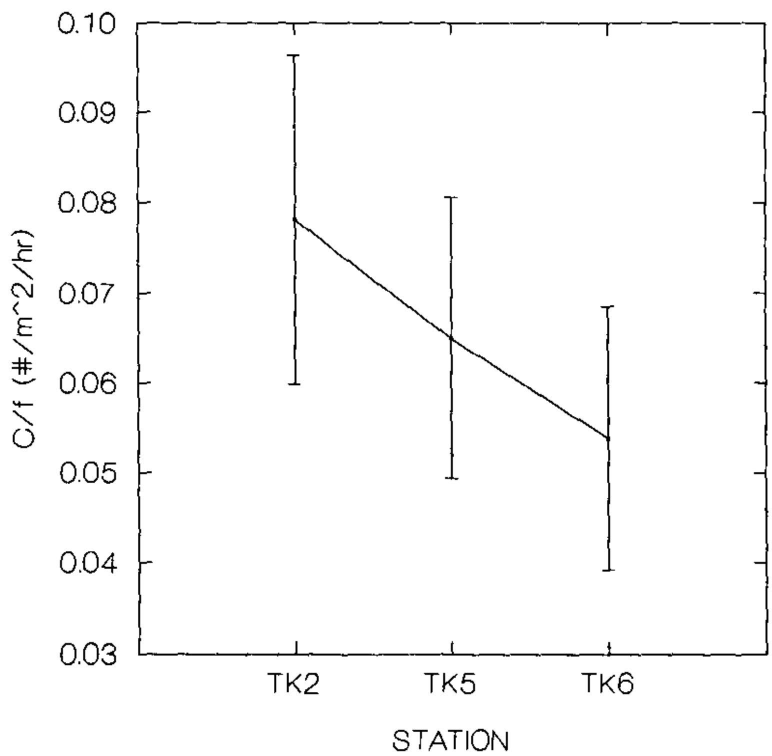



**Figure 82.** Longitudinal Trend in Gillnet Catch (as number) per Effort (average +/- SEM).

Total net yield expressed as grams wet weight decreased to station 5, then increased to station 6, but the trend was not significant (P = 0.58) (**Figure 83**). Vertical trends in fish distribution were evident (**Figures 86-98**) but not statistically evaluated. In general, fish captures were more variable in depth following destratification, and most fish were located near or just above the thermocline during the stratified period.

Catches were variable among stations for most species (**Figure 84**). A total of 10 species and 527 individuals were captured over the six sampling periods (**Table LXIV**). White bass dominated overall catch in numbers and varied slightly among stations. Gizzard shad decreased significantly (P = 0.02) downlake; station 2 was significantly different from the remaining two stations. Channel catfish yield decreased from station 2 and stabilized at station 5, but the trend was not significant (P = 0.14). Freshwater drum exhibited a non-significant (P = 0.10) decline throughout.

Longitudinal trends in wet weight yields for select species were also evident (**Figure 85**). Captured weight of white bass increased from station 5 but not significantly (P = 0.83). The downlake decline in gizzard shad weight was significant (P = 0.02) with station 2 being different from the remaining stations. Captured weight of channel catfish declined non-significantly (P = 0.44) to station 5 and stabilized. Freshwater drum catch as weight decreased non-significantly (P = 0.09) downlake.

## Conclusions

The majority of water quality, pelagial fish and planktonic parameters investigated in Lake Tenkiller demonstrated longitudinal spatial tendencies. Of the exhibited trends, many showed statistical significance. Statistical comparisons resulted in inference of only two zones in most cases with station 5 and station 6 exhibiting statistical similarity while different from station 2 in many parameters. Again, this result indicates station 5 to be late-transitional in location exhibiting characteristics more equitable with lacustrine reaches than the reservoir transition proper.

Lake Tenkiller plankton and fish exhibited several longitudinal trends. Total zooplankton, rotifers, and copepods peaked in density at station 5, while cladocerans increased along the thalweg. Relative frequency of rotifers decreased along the thalweg and that of copepods and cladocerans increased. Two plausible explanations for the changes in relative densities include the decreasing trend in turbidity with thalweg distance and less planktivory by large gizzard shad which exhibited greater catches at the upper station. Also, TPTD decreased with thalweg distance. This result is consistent with site-specific MEI predictions based upon R-T-L zonation. Additionally, total fish, gizzard shad, drum, and channel catfish abundance yield decreased downlake, while that of white bass was relatively invariable among stations. Total catch as wet weight decreased to station 5 then increased to station 6. Wet weight catch of gizzard shad and drum decreased downlake while that of channel catfish decreased to station 5 then stabilized. White bass catch as wet weight exhibited a marked increase from station 5 to station 6.

Experimental design may have affected the results necessitating suggestions for improvement. Although longitudinal, spatial trends were determined for several variables, the entirety of the stratified period should have been incorporated. Seasonal transition occurring during the latter three sampling periods may have overwhelmed spatial trends of some variables (e.g., chlorophyll). The addition of one or two more sampling stations is suggested to better represent the transition zone and refine trends. Replicate vertical net sets (two per transect) are also recommended in order to garner a more robust data set for statistical tests. Although each of the three nets were used randomly along their respective transects, intra-transect patchiness in distribution is suggested to have influenced tests for longitudinal trends. Multiple nets per transect and/or increased sample size should decrease the effect of outlying biases which possibly may have masked trends.

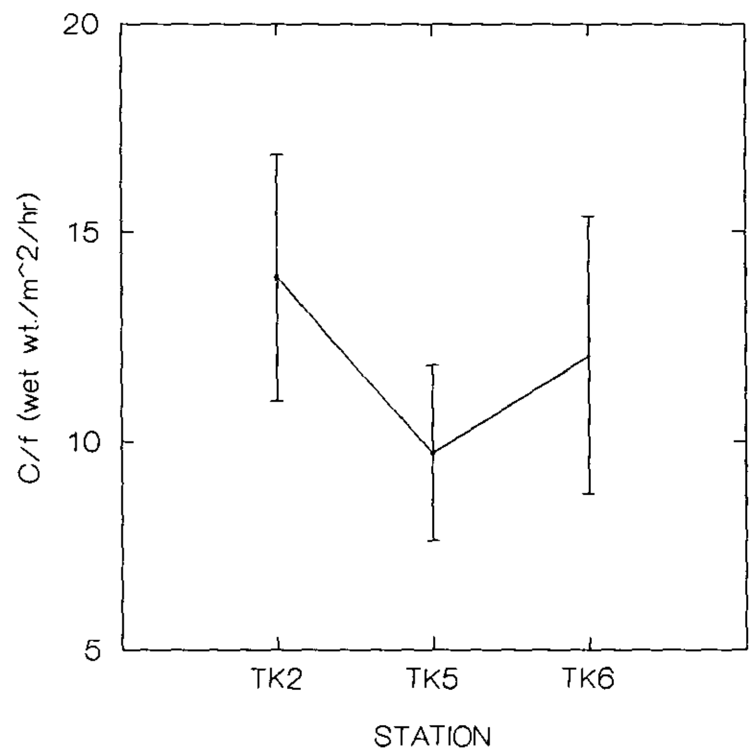



**Figure 83.** Longitudinal Trend in Gillnet Catch (as wet weight) per Effort (average +/- SEM).

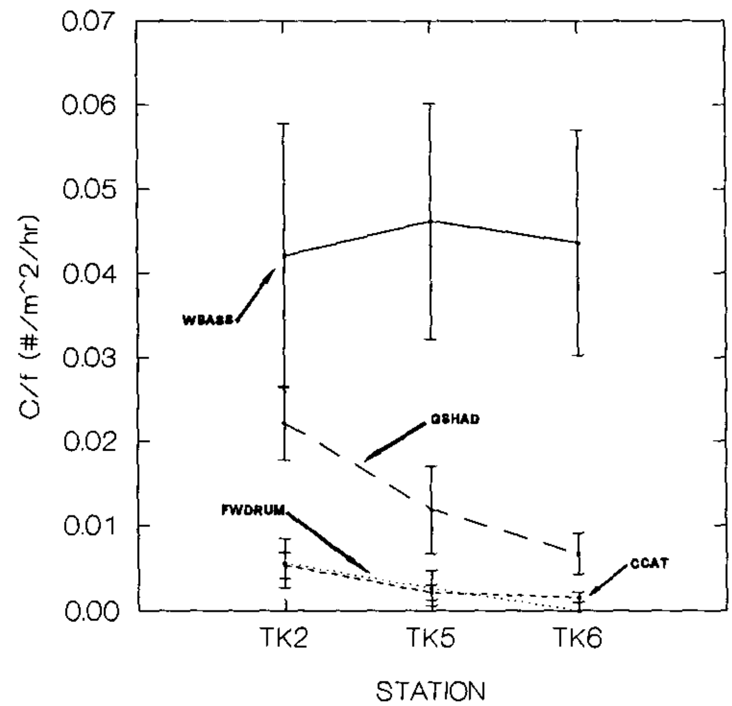



**Figure 84.** Longitudinal Trends in C/f as Number (average +/- SEM) for Select Fish Species.

**Table LXIV.** Fish Species and Abundance Collected in Lake Tenkiller.

| Species | Total | |
|---|---|---|
| | Number | % |
| White bass (*Morone chrysops*) | 371 | 70.4 |
| Gizzard shad (*Dorosoma cepedianum*) | 98 | 18.6 |
| Channel catfish (*Ictalurus punctatus*) | 21 | 3.98 |
| Freshwater drum (*Aplodinotus grunniens*) | 16 | 3.03 |
| Largemouth bass (*M. salmoides*) | 9 | 1.71 |
| Bluegill (*Lepomis macrochirus*) | 5 | 0.95 |
| Spotted bass (*Micropterus punctulatus*) | 3 | 0.57 |
| White crappie (*Pomoxis annularis*) | 2 | 0.38 |
| River carpsucker (*Carpiodes carpio*) | 1 | 0.19 |
| Smallmouth buffalo (*Ictiobus bubalus*) | 1 | 0.19 |
| TOTALS | 527 | 100.00 |

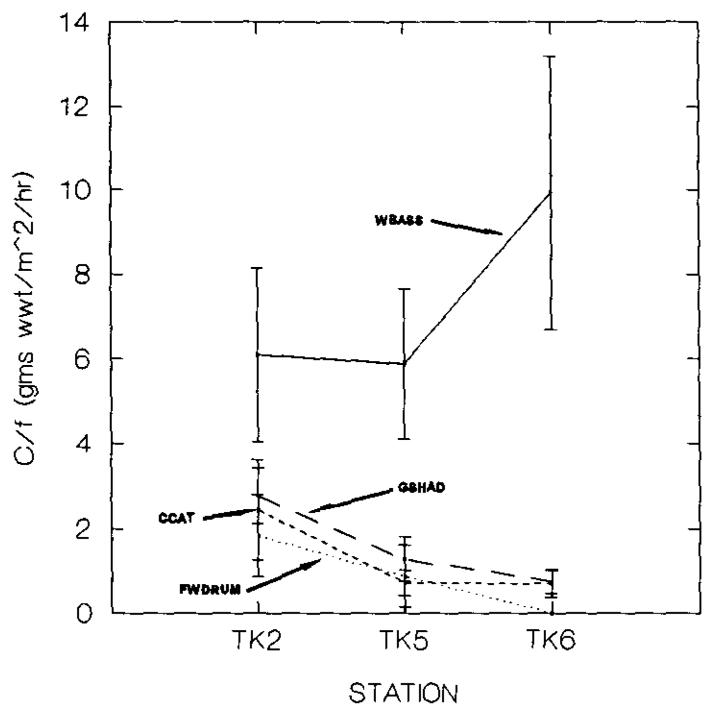



**Figure 85.** Longitudinal Trends in C/f as Wet Weight (average +/- SEM) for Selected Fish Species.

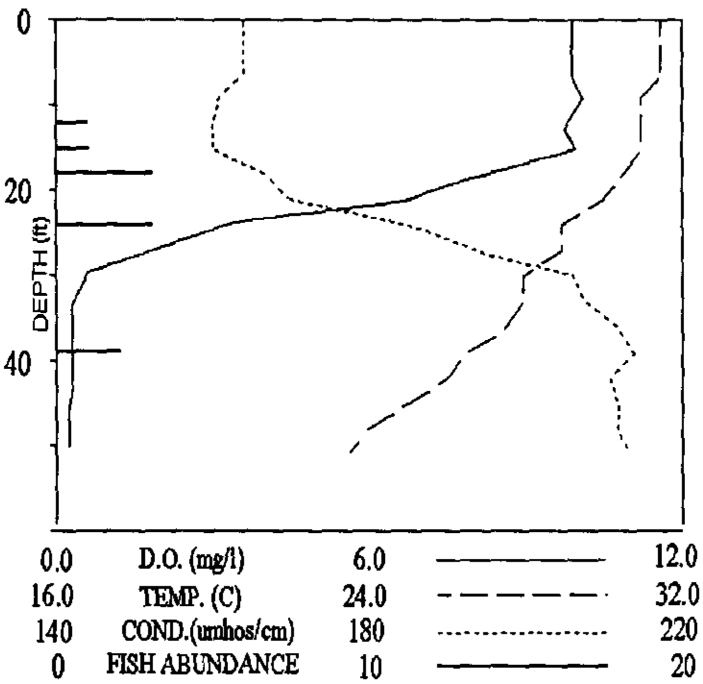



Figure 86. Vertical Profile of Water Quality and Fish Catch for Station 5, 16 Aug 93.

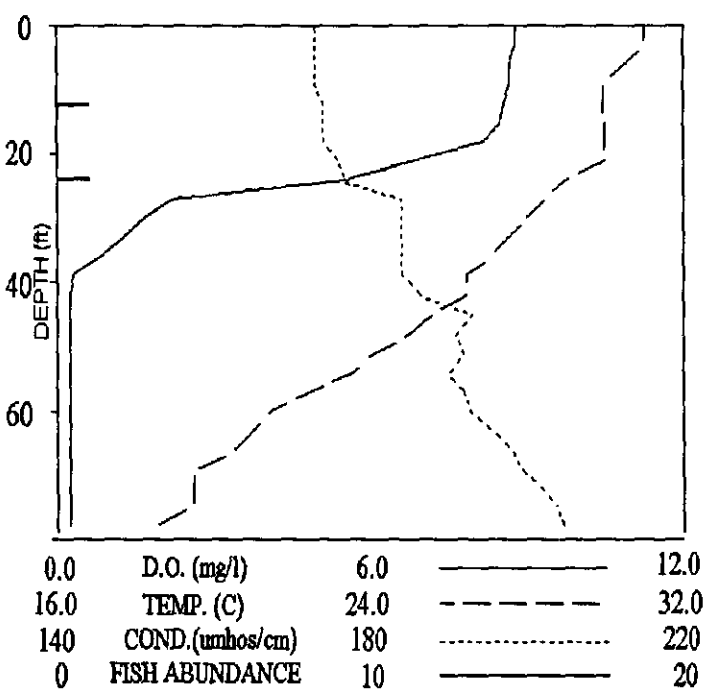



Figure 87. Vertical Profile of Water Quality and Fish Catch for Station 6, 16 Aug 93.

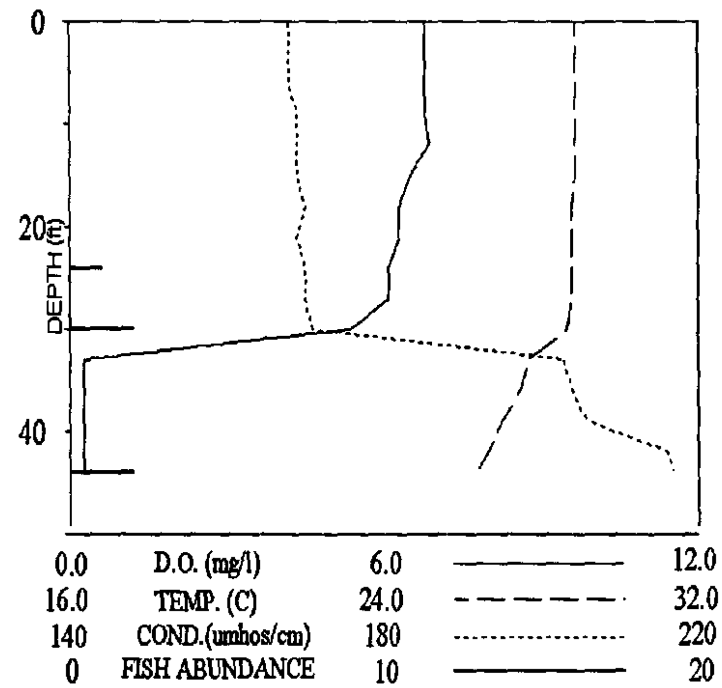



**Figure 88.** Vertical Profile of Water Quality and Fish Catch for Station 5, 2 Sep 93.

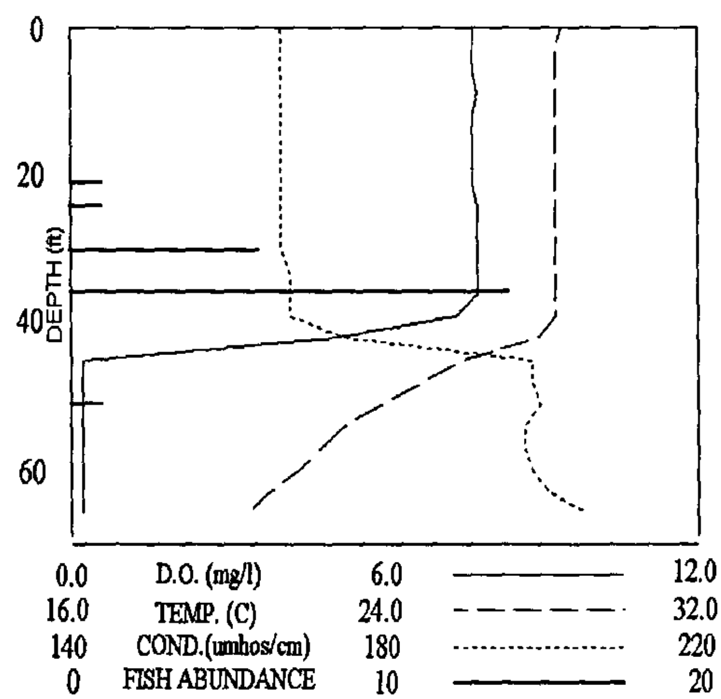



**Figure 89.** Vertical Profile of Water Quality and Fish Catch for Station 6, 2 Sep 93.

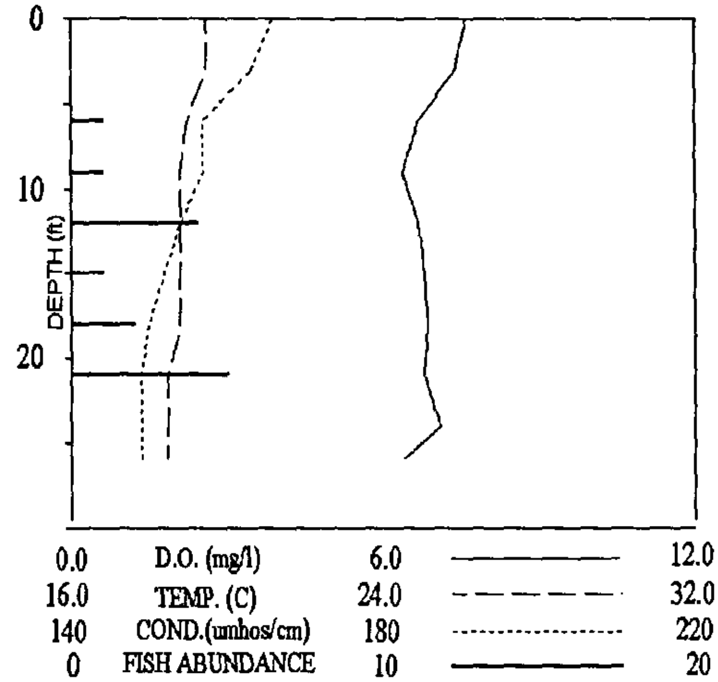



**Figure 90.** Vertical Profile of Water Quality and Fish Catch for Station 2, 16 Sep 93.

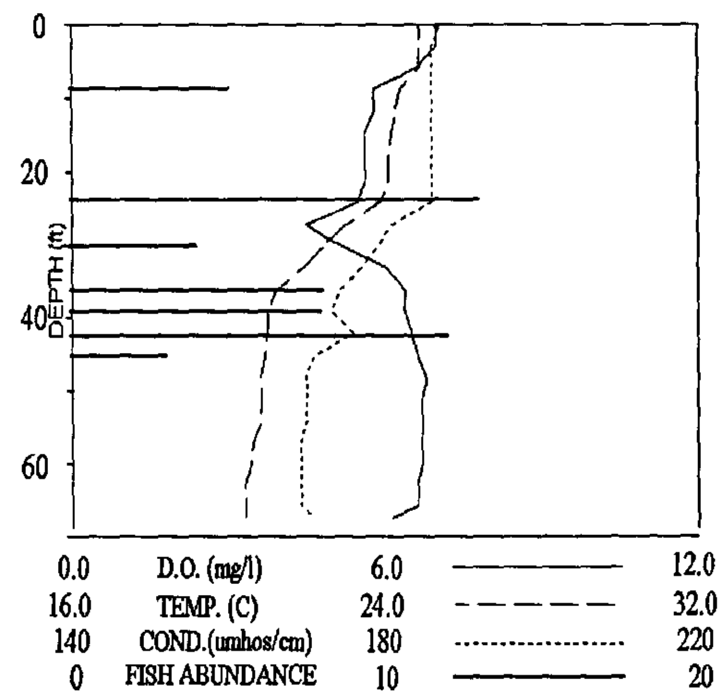
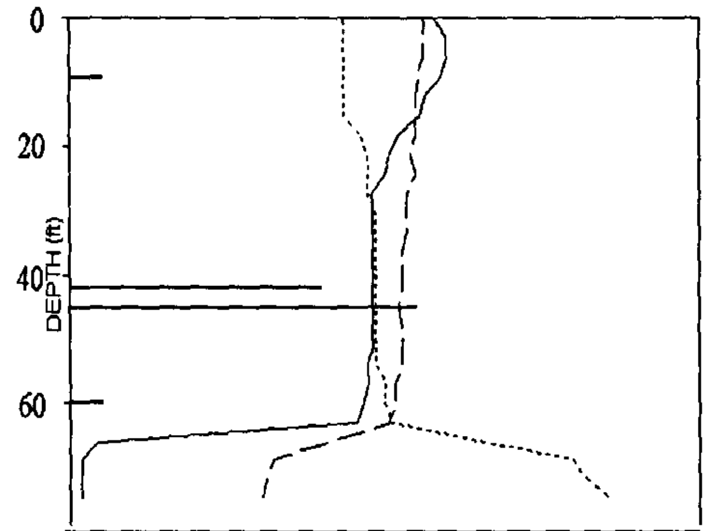
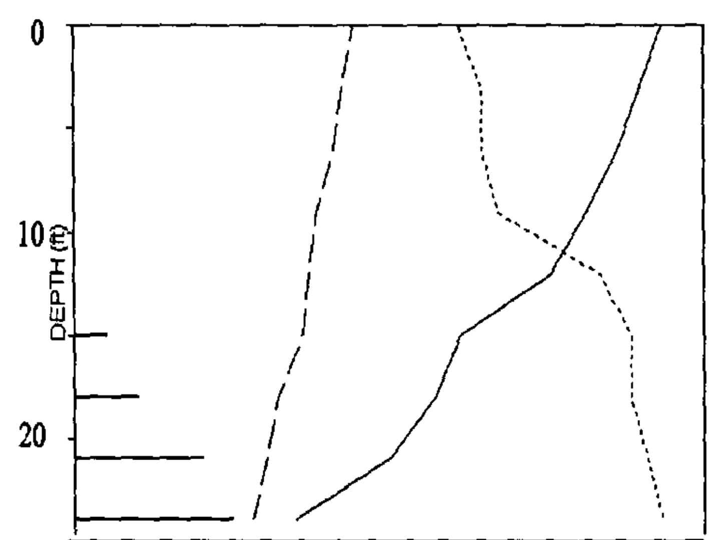



**Figure 91.** Vertical Profile of Water Quality and Fish Catch for Station 5, 16 Sep 93.




| | | | | |
|---|---|---|---|---|
| 0.0 | D.O. (mg/l) | 6.0 | ———— | 12.0 |
| 16.0 | TEMP. (C) | 24.0 | — — — — | 32.0 |
| 140 | COND.(umhos/cm) | 180 | ........ | 220 |
| 0 | FISH ABUNDANCE | 10 | ———— | 20 |

**Figure 92.** Vertical Profile of Water Quality and Fish Catch for Station 6, 16 Sep 93.




| | | | | |
|---|---|---|---|---|
| 0.0 | D.O. (mg/l) | 6.0 | ———— | 12.0 |
| 16.0 | TEMP. (C) | 24.0 | — — — — | 32.0 |
| 140 | COND.(umhos/cm) | 180 | ........ | 220 |
| 0 | FISH ABUNDANCE | 10 | ———— | 20 |

**Figure 93.** Vertical Profiles of Water Quality and Fish Catch for Station 2, 30 Sep 93.

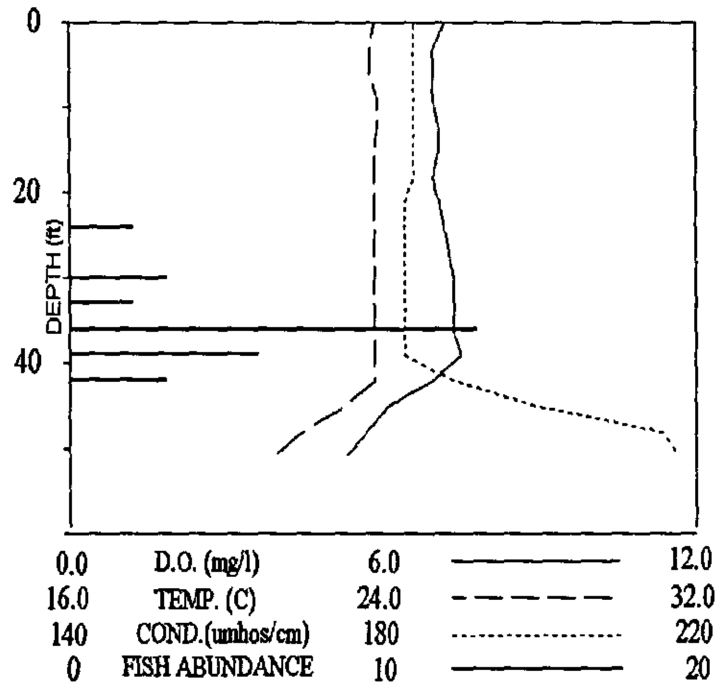



**Figure 94.** Vertical Profile of Water Quality and Fish Catch for Station 5, 30 Sep 93.

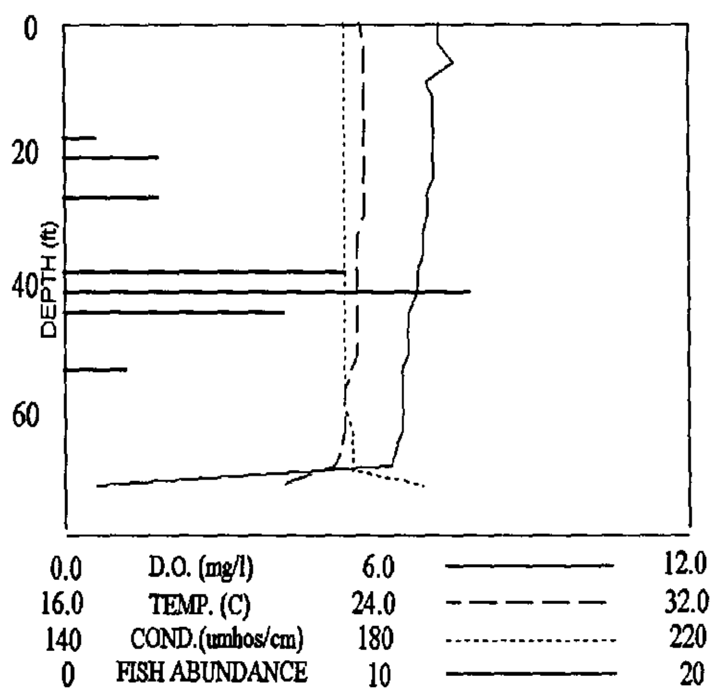



**Figure 95.** Vertical Profile of Water Quality and Fish Catch for Station 6, 30 Sep 93.

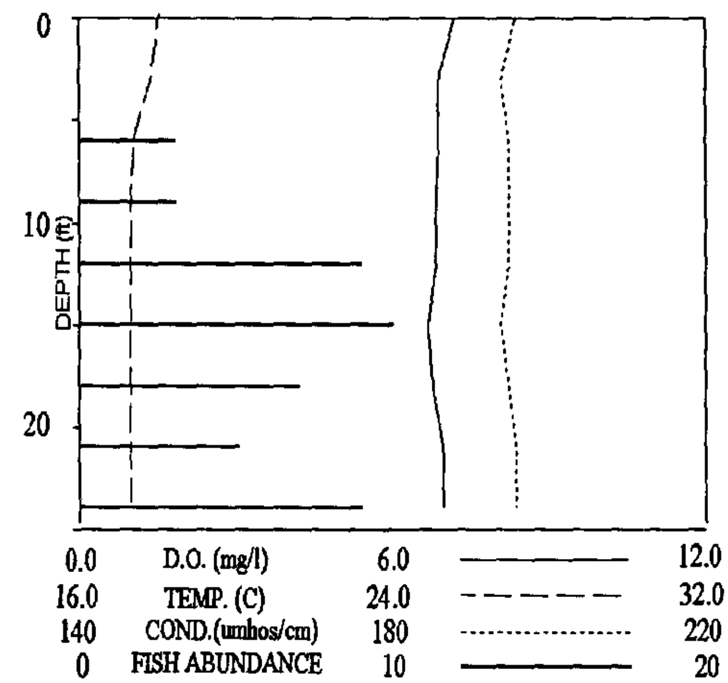



**Figure 96.** Vertical Profile of Water Quality and Fish Catch for Station 2, 21 Oct 93.

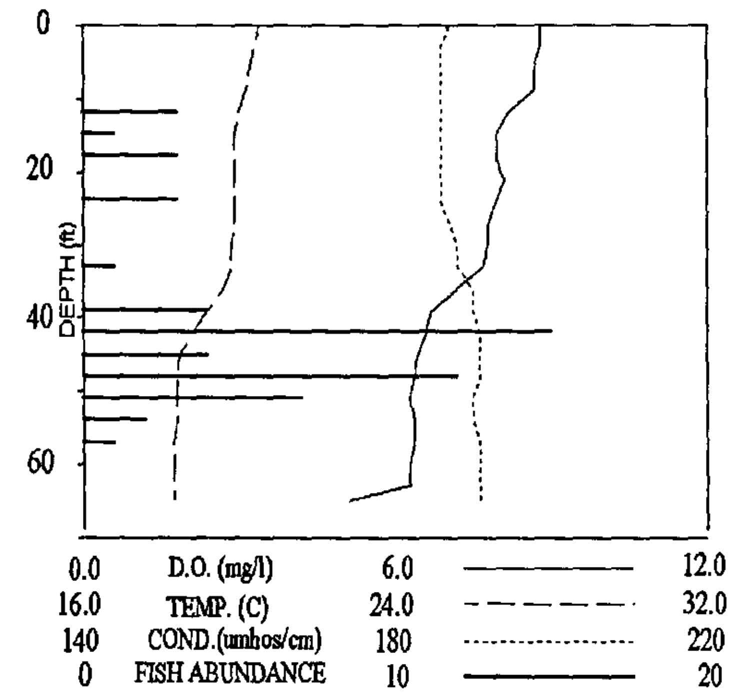



**Figure 97.** Vertical Profile of Water Quality and Fish Catch for Station 5, 21 Oct 93.

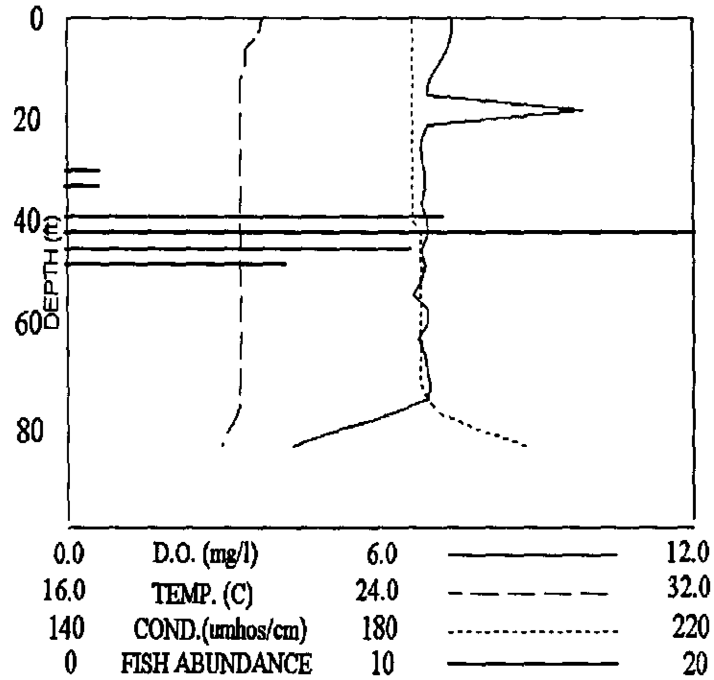



**Figure 98.** Vertical Profile of Water Quality and Fish Catch for Station 6, 21 Oct 93.

## B. FEASIBILITY STUDY

### 1. IDENTIFICATION AND DISCUSSION OF POLLUTION CONTROL AND LAKE RESTORATION ALTERNATIVES CONSIDERED AND SELECTED

#### a. Identification and Justification of Each Selected Alternative

Source control of total P loadings to the headwaters of Lake Tenkiller and a tailrace re-aeration device were recommended for the following reasons:

* represents a long term solution,
* is protective of the scenic Illinois River and Lake Tenkiller, and
* in-lake treatments would be cost-prohibitive and are only symptomatic correctives.

##### I. Expected Water Quality Improvement

Reductions in total P loadings have been shown to control and in some cases reverse the eutrophication process. Expected improvements in water quality include but are not limited to:

* increased water clarity,
* decreased frequency of algal blooms and thus reduced risk of fish kills from dissolved oxygen depletions, and
* increased recreational use.

##### ii. Technical Feasibility

After discussion of the feasibility of non-point source control of P runoff with officials from Natural Resource Conservation Service and Oklahoma Water Resources Board, we concluded that about 30-40% reduction in total P loads is feasible.

##### iii. Estimated Cost

Few published estimates of costs of total P removal are available. One publication cites that TP removal costs about $6.50-$6.65 per kg (Seip 1994). Assuming a 30-40% reduction in the median value equates to approximately $260,000-$340,000 per year. However, these costs probably reflect a cost of point source removal not NPS control. As was stated earlier, the recommended management will require both.

##### iv. Detailed Description - Exact Activities (How and Where Implemented), Engineering Specifications with Drawings, Anticipated Pollution Control Effectiveness

Control of NPS total P will require site-specific recommendations. Since most of these incorporate BMPs, exact activities and drawings are variable. The re-aeration weir in the tailrace is tentatively planned by the USACE and will be designed by such.

b. Identification and Justification of Each Considered Alternative

Three broad categories of alternatives are (with a few examples):

* source control

1. site treatment (e.g., BMPs) - Management of the watershed to protect the Illinois River and Lake Tenkiller will require implementation of BMPs including but not limited to restricted application of animal wastes to open fields, road designs and access points that minimize erosional losses during runoff events.

2. diversion of sources - Since point sources account for less than 50% of the total phosphorus load during low flows and much less during medium and high flows and NPDES regulatory agencies are implementing total phosphorus limits in the basin, diversion of sources is judged prohibitive. Additionally, nonpoint sources are difficult, if not impossible to divert and thus negates the high cost of point source diversion, albeit sedimentation basins have been used for nutrient traps.

3. watershed management - This approach was recommended because the main tributary to Lake Tenkiller is the Illinois River, which has been designated as a Scenic River by the State of Oklahoma and thus is afforded stringent protective measures. Implementing watershed management not only protects the lake but affords added river protection also.

* in-lake treatment

1. dredging - Sedimentation is not a problem is Lake Tenkiller and thus dredging would not only be cost-prohibitive but ineffective also.

2. shoreline modification - Lake Tenkiller does not possess gently sloping shorelines nor does it exhibit extensive macrophyte or wind-induced turbidity problems. Therefore, shoreline modification was not recommended.

3. dilution - The technique of lake flushing and dilution has been somewhat successful on smaller lakes but requires extensive costs and a large source of freshwater with less than ambient total phosphorus (or the

pollutant of interest) concentrations. No such source is readily available for Lake Tenkiller and thus this technique was deemed inappropriate.

4. nutrient precipitation/inactivation (e.g., alum application) - Nutrient precipitation/activation would be cost prohibitive because of the large size and volume of Lake Tenkiller. Additionally, the large allochthonous total phosphorus loads would likely render the technique ineffective after a few years or require continual application.

5. lake level drawdown - Lake Tenkiller, as can be seen from the bathymetric map has relatively steep shorelines and virtually no macrophyte problem. Therefore, lake level drawdown was not recommended and would only cost the users access and authorities compromised functions (e.g., flood control).

* problem treatment

1. physical (e.g., biological) - The use of algacides, e.g., endothall, copper sulfate, or biological controls, e.g., bighead carp, were not recommended because continual application of an algacide would be required and might accumulate toxicity in the higher organisms and/or sediments. Biological controls were not recommended because these techniques, while environmentally appealling, have not been well-documented in large reservoirs.

2. hypolimnetic aeration - The cost of hypolimnetic aeration prohibits its application is such a large reservoir. Additionally, like all in-lake treatments, hypolimnetic aearation does not impart any protective effect on the Illinois River.

Clearly, these alternatives are not mutually exclusive.

I. Expected Water Quality Improvement

Expected water quality improvement includes that discussed in the preceding section. The overall goal is to slow or reverse the eutrophication process in the lake, restore and protect the river, and as such help to protect the tailwater fishery. Improvement and protection of these resources will prolong the life of the reservoir and should increase its beneficial uses.

ii. Technical Feasibility

The feasibility of the other alternatives are available. However, most are cost-prohibitive for installation, operation, and maintenance. Some such as in-lake chemical treatment would be ineffective because the short residence time of the reservoir would force frequent treatments.

iii. Estimated Cost

Most of the alternatives would be cost-prohibitive and require extensive personnel and management for operation and maintenance. The in-lake treatments further have the disadvantages of neither protecting nor restoring the scenic Illinois River.

iv. Detailed Description - Exact Activities (How and Where Implemented), Engineering Specifications with Drawings, Anticipated Pollution Control Effectiveness

The exact details of the considered alternatives (excluding those recommended) were not evaluated in depth because the disadvantages nullify the possible implementation. Should the proposed recommendations not effectively meet the water quality goals, the alternatives may be considered in additions to the source control.

## 2. DISCUSSION OF EXPECTED BENEFITS OF PROJECT

Benefits of water quality restoration include current authorized and unathorized uses. Authorized uses include flood control, public water supply, and hydroelectric power generation. Additionally, though recreation was not an "authorized" purpose, a significant economic base has developed from the recreational uses of the river and lake.

a. New Public Water Uses from Enhanced Water Quality

Current uses are in place but few users would disagree these have been impaired. Therefore, although new uses may be few, the recommedations should enhance current recreational uses.

b. Other Benefits

Current benefits include those discussed earlier. The USACE estimated that average annual benefits of Lake Tenkiller from flood control are about $58.1 million. Recreational benefits include those from the Illinois River and Lake Tenkiller. Water treatment costs would also be less.

## 3. WATER QUALITY SAMPLING SCHEDULE OF PHASE 2 MONITORING PROGRAM

Large reservoirs typically do not conform to a typical Phase II project because of the different entities involved in a basin-wide watershed management approach. Such an approach was proposed in this Phase I study and thus a detailed proposal for Phase II funding was not prepared. However, a monitoring schedule was suggested for evaluating the effectiveness of the point source (e.g., NPDES limits) and non-point source (e.g., BMPs) controls. The monitoring schedule is given below.

a. Monthly/Biweekly Single In-Lake Site Data - The in-lake data to be collected should, as a minimum, coincide with congruent sampling sites of the 1974 NES, 1985-86 USACE, and 1992-93 CLP studies. Sampling should emphasize the stratified period and coincide with biological (e.g., spring/fall algal peaks) and hydraulic events (e.g., changes in lake elevation). Epilimnetic and hypolimnetic samples should be collected for water quality analyses. Profiles of temperature, dissolved oxygen, conductivity, and pH should be collected during each sampling trip.

b. Additional Site Data - Tributary sites should be included provided funding is available. The major coves do not represent significant portions of the basin but the contribution to the total nutrient load to the lake was assumed in this study and hence remains uncertain. Watershed sites should be maintained and coordinated with the Oklahoma Scenic Rivers Commission and United States Geological Survey.

c. Analytes - Total and Soluble Reactive Phosphorus; Nitrite, Nitrate, Ammonia and Organic Nitrogen; pH; Temperature; Dissolved Oxygen; Alkalinities; Chlorophyll *a*; Algal Biomass; Secchi Disk; Suspended Solids; Surface Area Covered by Macrophytes; and Any Specific Measurements Deemed Necessary - These parameters should be measured along with notes of any unusual occurrence (e.g., algal blooms, odors, fish kills). There is no need to monitor macrophytes at this time. This study (1992-93) did not reveal any macrophyte problems and concluded the absence of macrophytes was due to steep shorelines and rocky substrates along the majority of the lake shoreline.

4. PROPOSED MILESTONE WORK SCHEDULE

a. Project Completion Schedule - The total maximum daily load evaluation and total phosphorus reductions were proposed on a short-term (5-10 year) basis and a long-term basis (no timeline intended) as technology develops. Since a Phase II application is not submitted here, *per se*, a typical milestone schedule was not developed.

b. Budget - A budent was not prepared because the Phase II application was not proposed.

c. Payment Schedule - A payment schedule was not prepared because a Phase II application was not proposed.

5. DESCRIPTION OF METHOD OF OBTAINING NON-FEDERAL FUNDS - The non-federal match as required by the Clean Lakes Program (Section 314) is not applicable because a Phase II was not proposed. However,

many non federal agencies are currently conducting studies and implemetation projects in the Illinois River basin. These agencies include the Oklahoma Scenic Rivers Commission, Oklahoma Conservation Commission (NPS), Oklahoma Water Resources Board (TMDL and standards), and the Oklahoma Department of Environmental Quality (NPDES). The multilevel Arkansas/Oklahoma Arkansas River Compact Commission also is involved in the process of formulating and implementing a watershed management plan. The Lake Tenkiller Association also has been involved in this project and should continue to be consulted on future activities such as a watershed management plan or any management activity effecting Lake Tenkiller.

6. DESCRIPTION OF RELATIONSHIP OF PROPOSED PROJECT TO LOCAL, STATE, REGIONAL AND/OR FEDERAL PROGRAMS RELATED TO THE PROJECT - No project was proposed. However, agencies that are related to any activity in the Illinois River basin and effecting Lake Tenkiller were listed in the preceding section.

7. SUMMARY OF PUBLIC PARTICIPATION IN DEVELOPING AND ASSESSING THE PROPOSED PROJECT (MAY USE PART 25.8 RESPONSIVENESS SUMMARIES)

a. Matters Brought Before The Public - The Oklahoma Water Resources Board, in cooperation with the Office of the Secretary of the Environment, USEPA, Oklahoma State University - Water Quality Research Laboratory, and the Oklahoma Scenic Rivers Commission, held an informal public meeting concerning the Lake Tenkiller Clean Lakes Phase I Project and Total Maximum Daily Load study at 7:00 pm on 28 March 1996 at the First United Methodist Church Gymnasium in Tahlequah, Oklahoma. Several state and federal agencies participated in the meeting by presenting current and futrue activities taking place in the watershed of Lake Tenkiller. Five master's and two doctoral candidates produced research for the project. Master's recipient, Shanon Haraughty and doctoral recipient Noble Jobe, both of Oklahoma State University, authored and accompanied a poster presentation which was on display immediately before and during the meeting.

Results of the Clean Lakes Study were presented by Dr. Sterling (Bud) Burks, Water Quality and Ecotoxicology Assessments, Inc. and Dr. Noble Jobe, Oklahoma State University. Two question were addressed from the audience; the first conerned the 40% reduction in total phosphorus recommendation and where did this value come from? Dr. Burks responded that the value came from a reduction of current modeled numerical calculations. Burks also commented that a 40% reduction in phosphorus will probably only maintain the lake in its current condition, there will be no future accelerated degradation. The second comment was informative from Arkansas. Arkansas has implemented serveral point source Best Management Practices in the watershed that have resulted in a phosphorus reduction of 30% since 1986. Dr. Burks responded that a more precise estimate of Arkansas' contribution will be available upon completion of the other doctoral candidate's thesis.

b. Measures Taken by the Reporting Agency to Meet its Responsibilities Under Part 25 and Related Provisions Elsewhere in Chapter 35

c. The Public Response and Agency's Response to Significant Comments - A list of 10 questions faxed to the Oklahoma Scenic Rivers Commission by a consultant in Arkansas was reviewed by the principal investigators and responses were prepared. At this writing, the formal response is being prepared to be sent to the consultant and other agencies provided as a cc list on the original fax.

8. OPERATION AND MAINTENANCE PLAN AND TIME FRAME FOR THE STATE TO FOLLOW - Since a formal Phase II proposal does not coincide with large reservoir watershed studies and was not proposed, an O&M plan and time frame was not applicable.

9. COPIES OF ALL PERMITS OR PENDING PERMITS NECESSARY TO SATISFY THE REQUIREMENTS OF SECTION 404 OF THE ACT - The recommendations set forth in this study did not necessitate any section 404 permits. However, the rearation weirs in the tailrace currently under consideration may require such permits.

## C. ENVIRONMENTAL EVALUATION TO ANSWER FOLLOWING QUESTIONS

1. Will the proposed project displace any people?

The remediation recommendations put forth here should not displace people but will impact lifestyles and cost of living expenses of many people. For example, changes in agricultural practices will impact the poultry industry while implementing point source criteria could and likely will cost the discharge contributors more.

2. Will the proposed project deface existing residences or residential areas? What mitigative actions such as landscaping, screening, or buffer zones have been considered? Are they included?

The proposed remediation should not deface existing residences or residential areas. The recommendations put forth included working with state agencies to implement buffer zones along the riparian areas along the Illinois River and associated tributaries. These zones are designed to reduce erosion and promote water clarity in the river and reduce turbidity loading to the lake. However, as the Illinois River riparian zones are complex, each buffer zone needs customized to existing conditions on a site-specific basis. Therefore, mitigative actions are unforeseen.

3. Will the proposed project be likely to lead to a change in established land use patterns, such as increased development pressure near the lake? To what extent and how will this change be controlled through land use planning, zoning, or through other methods?

The remediation proposed in this report undoubtedly will require a change in land use practices, especially for the agricultural industry. Implementation of BMPs will require extensive involvement by the state conservation agencies (e.g., Natural Resources Conservation Service, Department of Agriculture, OWRB, Oklahoma Conservation Commission, Department of Environmental Quality).

4. Will the proposed project adversely affect a significant amount of prime agricultural land or agricultural operations on such land?

The change in agricultural practices will affect a significant portion of land. However, the remediation does not include a reduction in production from the land, thus adverse affects are not anticipated. However, operation costs may increase due to restrictions on land application of wastes and/or new costs of transport and disposal of such wastes.

5. Will the proposed project result in a significant adverse effect on parkland, other public land, or lands or recognized scenic value?

The changes recommended here should not affect these land categories. Indeed, these are the categories being targeted for protection. The Illinois River is noted for its exceptional clarity and scenic value.

6. Has the State Historical Society of State Historical Preservation Officer been contacted? Has he responded, and, if so, what was the nature of that response? Will the proposed project result in a significantly adverse effect on lands or structures of historic, architectural, archaeological, or cultural value?

While the State Historical Society was not contacted, we assumed that historical inventories conducted during the pre-impoundment study and other CLP studies in the basin (Threlkeld 1983) were appropriate. Additionally, the only new construction requires minor disturbance and is targeted in the tailrace of the lake, where no historical site would be impacted. For further information on regional historical sites consult (Lindsay et al. 1973).

7. Will the proposed project lead to a significant long-range increase in energy demands?

The recommendations should not lead to a significant increase in energy demands.

8. Will the proposed project result in significant and long-range adverse changes in ambient air quality or noise levels? Short term?

Construction of the re-aeration weir may have a signficant short term increase in noise levels of the immediate vicinity of the tailrace. However, after construction, noise levels should not be significantly affected. Air quality could be improved if the re-aeration weir oxygenates the water such that ammonia and volatile sulfides are reduced.

9. If the proposed project involves the use of in-lake chemical treatment, what long and short-term adverse effects can be expected from that treatment? How will the project recipient mitigate these effects?

No in-lake treatment was recommended, therefore adverse effects are not implied.

10. Does the proposal contain all the information that EPA requires in order to determine whether the project complies with Executive Order 11988 on floodplains? Is the proposed project located in a floodplain? If so, will the project involve construction of structures in the floodplain? What steps will be taken to reduce the possible effects of flood damage to the project?

The project does not include construction of devices in the floodplain that would impact current flood control capacities. The re-aeration weir in the tailrace, though in the floodplain, would be near Tenkiller Dam and its impact would be negligible when compared to the capacity of Tenkiller Dam to regulate floods.

11. If the project involves physically modifying the lake shore or its bed or its watershed, by dredging, for example, what steps will be taken to minimize any immediate and long-term adverse effects of such activities? When dredging is employed, where will the dredging material be deposited, what can be expected and what measures will the recipient employ to minimize any significant adverse impacts from its deposition?

No dredging was recommended. However, some disturbance will be necessary to install the re-aeration weir in the tailrace. This is a tentative plan of the USACE and thus feasibility and handling of this portion should be tended by such.

12. Does the project proposal contain all information that EPA requires in order to determine whether the project complies with Executive Order 11990 on wetlands? Will the proposed project have a significant adverse effect on fish and wildlife, or on wetlands or any other wildlife habitat, especially those of endangered species? How significant is this impact in relation to the local or regional critical habitat needs? Have actions to mitigate habitat destruction been incorporated into the project? Has the recipient properly consulted with appropriate Sate and Federal fish, game and wildlife agencies and with the U.S. Fish and Wildlife Service? What were their replies?

The recommended management targeted land use changes and reduction of P loads. In this respect it impact on wetlands should be negligible. However, installation of sedimentation ponds for nutrient retention could effect current wetlands or create new ones. These implications will be on a site-specific basis. Appropriate agencies should be contacted should this occur.

13. Describe any feasible alternatives to the proposed project in terms of environmental impacts, commitment of resources, public interest and costs and why they were proposed.

We believe that the proposed management is the optimum based upon cost-effectiveness, environmental impacts, commitment of resources, public interest and costs. Other treatments, such as in-lake treatment, would be cost-prohibitve and could require addition of large quantities of chemicals (e.g., biocides, alum). Furthermore, such treatments would be short term symptomatic approaches and would not be protective of the Illinois River.

14. Describe other measures not discussed previously that are necessary to mitigate adverse environmental impacts resulting from the implementation of the proposed project.

Economic analyses need to be conducted to monitor the economic impact of land use changes. This aspect is important for an accurate cost/benefit analysis and monitoring.

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# APPENDIX A

## CLP STUDY 1992-93 LAKE TENKILLER WATER QUALITY DATA

## USED FOR RERF DEVELOPMENT

| DATE | STATION | Total N mg N/l | Total P mg P/l | TN:TP | Chlr a µg/l | Turb NTU | Secchi m |
|---|---|---|---|---|---|---|---|
| 25-Apr-92 | 2 | ND | 0.070 | ND | 25.3 | 4.3 | 0.90 |
| | 3 | ND | 0.052 | ND | 23.1 | 4.2 | 0.88 |
| | 4 | ND | 0.074 | ND | 19.3 | 4.8 | 0.88 |
| | 5 | ND | 0.022 | ND | 17.1 | 2.3 | 1.40 |
| | 6 | ND | 0.013 | ND | 7.2 | 1.5 | 2.50 |
| | 7 | ND | 0.007 | ND | 3.5 | 0.6 | 5.50 |
| 04-Jun-92 | 1 | 2.38 | 0.119 | 20.0 | 2.8 | ND | ND |
| | 2 | 1.99 | 0.061 | 32.8 | 28.4 | ND | 0.95 |
| | 3 | 1.86 | 0.066 | 28.1 | 22.3 | ND | 1.00 |
| | 4 | 2.46 | 0.061 | 40.4 | 34.3 | ND | 1.00 |
| | 5 | 2.09 | 0.027 | 77.3 | 15.7 | ND | 1.60 |
| | 6 | 2.71 | 0.023 | 116.0 | 16.5 | ND | 1.70 |
| | 7 | 3.15 | 0.016 | 199.0 | 9.4 | ND | 2.40 |
| 02-Jul-92 | 1 | 2.50 | 0.111 | 22.4 | 10.0 | ND | ND |
| | 2 | 2.28 | 0.069 | 33.0 | 45.2 | 4.9 | 0.90 |
| | 3 | 2.13 | 0.064 | 33.0 | 47.7 | 5.1 | 0.85 |
| | 4 | 2.08 | 0.058 | 35.9 | 47.2 | 4.9 | 0.70 |
| | 5 | 2.00 | 0.056 | 35.7 | 45.6 | 4.0 | 1.30 |
| | 6 | 1.73 | 0.051 | 33.8 | 39.6 | 1.1 | 1.60 |
| | 7 | 1.99 | 0.031 | 65.0 | 28.0 | 2.2 | 1.60 |
| 01-Aug-92 | 1 | 2.52 | 0.343 | 7.3 | 4.2 | 18.0 | ND |
| | 2 | 1.09 | 0.100 | 10.9 | 44.5 | 2.8 | 0.70 |
| | 3 | 0.92 | 0.018 | 50.9 | 46.2 | 2.4 | 0.80 |
| | 4 | 0.97 | 0.069 | 14.0 | 31.0 | 2.7 | 0.70 |
| | 5 | 0.51 | 0.034 | 15.1 | 28.0 | 1.6 | 1.23 |
| | 6 | 0.70 | 0.020 | 34.9 | 13.3 | 1.2 | 1.65 |
| | 7 | 0.63 | 0.014 | 45.3 | 12.1 | 1.2 | 1.70 |
| 19-Aug-92 | 1 | 1.21 | 0.080 | 15.1 | 3.9 | 6.6 | ND |
| | 2 | 0.77 | 0.063 | 12.3 | 32.7 | 5.9 | 1.28 |
| | 3 | 0.76 | 0.081 | 9.4 | 32.9 | 5.1 | 0.90 |
| | 4 | 0.81 | 0.069 | 11.8 | 33.4 | 5.2 | 0.90 |
| | 5 | 0.57 | 0.057 | 9.9 | 18.8 | 2.6 | 1.50 |
| | 6 | 0.31 | 0.042 | 7.5 | 13.4 | 2.3 | 1.70 |
| | 7 | 0.39 | 0.038 | 10.3 | 10.1 | 1.5 | 2.20 |
| 12-Sep-92 | 1 | 1.33 | 0.097 | 13.7 | 1.1 | 8.7 | 1.00 |
| | 2 | 0.99 | 0.084 | 11.8 | 43.4 | 6.6 | 0.80 |
| | 3 | 0.72 | 0.081 | 8.8 | 33.0 | 13.5 | 0.80 |
| | 4 | 0.59 | 0.057 | 10.4 | 29.2 | 10.1 | 1.00 |
| | 5 | 0.37 | 0.041 | 9.0 | 15.8 | 5.4 | 1.60 |
| | 6 | 0.47 | 0.017 | 27.5 | 11.0 | 4.4 | 1.80 |
| | 7 | 0.31 | 0.025 | 12.5 | 6.3 | 3.0 | 2.30 |
| 24-Oct-92 | 1 | 1.58 | 0.086 | 18.4 | 1.9 | ND | ND |
| | 2 | 1.16 | 0.048 | 24.2 | 40.3 | 3.5 | 1.30 |
| | 3 | 0.97 | 0.031 | 31.4 | 26.1 | 2.0 | 1.90 |
| | 4 | 1.17 | 0.039 | 30.0 | 23.4 | 5.3 | 1.90 |
| | 5 | 0.75 | 0.025 | 30.0 | 11.8 | 2.0 | 2.30 |
| | 6 | 0.76 | 0.023 | 32.8 | 14.2 | 1.7 | 2.60 |
| | 7 | 0.61 | 0.023 | 26.6 | 12.6 | 1.3 | 4.30 |
| 08-Mar-93 | 1 | 2.86 | 0.085 | 33.7 | 1.0 | ND | ND |
| | 2 | 1.47 | 0.073 | 20.1 | 2.7 | ND | 0.70 |
| | 3 | 1.69 | 0.079 | 21.4 | 3.9 | ND | 0.75 |
| | 4 | 1.86 | 0.081 | 23.0 | 4.3 | ND | 0.80 |
| | 5 | 1.94 | 0.118 | 16.5 | 4.1 | ND | 0.30 |
| | 6 | 1.80 | 0.085 | 21.2 | 9.4 | ND | 1.70 |
| | 7 | 1.42 | 0.087 | 16.3 | 2.5 | ND | 1.45 |
| 18-Apr-93 | 1 | 2.36 | 0.287 | 8.2 | 2.6 | ND | ND |
| | 2 | 1.71 | 0.159 | 10.8 | 1.6 | ND | 0.30 |
| | 3 | 1.50 | 0.146 | 10.3 | 8.9 | ND | 0.27 |
| | 4 | 2.00 | 0.176 | 11.4 | 2.7 | ND | 0.30 |
| | 5 | 1.83 | 0.124 | 14.8 | 5.8 | ND | 0.48 |
| | 6 | 1.66 | 0.078 | 21.3 | 11.9 | ND | 1.20 |
| | 7 | 1.93 | 0.067 | 28.8 | 30.6 | ND | 1.40 |
| 26-May-93 | 1 | 2.52 | 0.106 | 23.8 | 71.1 | 17.0 | ND |

(cont.)

| DATE | STATION | Total N mg N/l | Total P mg P/l | TN:TP | Chlr a µg/l | Turb NTU | Secchi m |
|---|---|---|---|---|---|---|---|
| 26-May-93 | 2 | 1.36 | 0.048 | 28.3 | 44.4 | 13.0 | 1.10 |
| | 3 | 1.24 | 0.054 | 23.0 | 29.4 | 15.0 | 1.30 |
| | 4 | 1.51 | 0.056 | 26.9 | 36.4 | 7.4 | 1.20 |
| | 5 | 1.39 | 0.056 | 25.0 | 15.5 | 8.5 | 1.40 |
| | 6 | 1.36 | 0.046 | 29.5 | 5.8 | 8.0 | 2.00 |
| | 7 | 1.37 | 0.037 | 37.0 | 8.5 | 4.8 | 2.20 |
| 25-Jun-93 | 1 | 1.92 | 0.131 | 14.7 | 1.7 | 39.0 | ND |
| | 2 | 1.08 | 0.102 | 10.6 | 46.8 | 28.0 | 1.20 |
| | 3 | 1.01 | 0.097 | 10.4 | 45.0 | 56.0 | 1.10 |
| | 4 | 0.90 | 0.081 | 11.1 | 38.0 | ND | 0.95 |
| | 5 | 0.79 | 0.064 | 12.3 | 40.3 | 28.0 | 1.15 |
| | 6 | 0.91 | 0.040 | 22.8 | 32.4 | 23.0 | 1.55 |
| | 7 | 0.87 | 0.033 | 26.4 | 39.1 | 24.0 | 1.40 |
| 22-Jul-93 | 1 | 1.67 | 0.130 | 12.8 | 2.5 | 6.5 | ND |
| | 2 | 0.81 | 0.046 | 17.6 | 10.6 | 6.3 | 1.20 |
| | 3 | 0.80 | 0.051 | 15.7 | 12.0 | 8.3 | 1.10 |
| | 4 | 0.72 | 0.059 | 12.2 | 21.7 | 6.3 | 1.20 |
| | 5 | 0.49 | 0.038 | 12.9 | 12.6 | 4.9 | 1.50 |
| | 6 | 0.74 | 0.022 | 33.6 | 3.7 | 2.9 | 2.10 |
| | 7 | 0.74 | 0.020 | 37.0 | 6.4 | 4.7 | 2.00 |
| 04-Aug-93 | 2 | 0.97 | 0.076 | 12.7 | 20.0 | 14.0 | 0.70 |
| | 3 | 1.23 | 0.076 | 16.1 | 17.3 | 18.0 | 0.90 |
| | 4 | 1.34 | 0.076 | 17.6 | 22.9 | 15.0 | 0.85 |
| | 5 | 0.61 | 0.042 | 14.4 | 26.5 | 10.0 | 1.20 |
| | 6 | 0.64 | 0.023 | 28.1 | 13.8 | 4.3 | 2.10 |
| | 7 | 0.77 | 0.016 | 48.3 | 8.2 | 4.4 | 3.00 |
| 19-Aug-93 | 1 | 1.57 | 0.129 | 12.2 | 21.0 | 2.7 | ND |
| | 2 | 0.83 | 0.090 | 9.2 | 35.4 | 4.9 | 1.00 |
| | 3 | 0.61 | 0.061 | 10.0 | 26.6 | 4.9 | 1.10 |
| | 4 | 0.47 | 0.050 | 9.4 | 31.1 | 3.9 | 1.30 |
| | 5 | 0.50 | 0.034 | 14.7 | 16.9 | 2.2 | 1.95 |
| | 6 | 0.62 | 0.028 | 22.3 | 10.5 | 2.9 | 2.50 |
| | 7 | 0.70 | 0.021 | 33.3 | 7.0 | 1.5 | 2.80 |
| 02-Sep-93 | 1 | 0.92 | 0.125 | 7.4 | 1.2 | 7.0 | ND |
| | 2 | 0.71 | 0.083 | 8.6 | 25.7 | 7.4 | 0.80 |
| | 3 | 0.77 | 0.093 | 8.3 | 39.6 | 7.5 | 0.85 |
| | 4 | 0.56 | 0.086 | 6.5 | 28.1 | 7.2 | 0.80 |
| | 5 | 0.45 | 0.041 | 11.0 | 18.9 | 2.6 | 1.40 |
| | 6 | 0.35 | 0.015 | 23.1 | 22.3 | 2.2 | 2.30 |
| | 7 | 0.41 | 0.015 | 27.3 | 24.9 | 1.6 | 2.00 |
| 16-Sep-93 | 1 | 5.22 | 0.178 | 29.3 | 2.0 | 25.0 | ND |
| | 2 | 3.82 | 0.119 | 32.1 | 8.6 | 45.2 | 0.20 |
| | 3 | 2.17 | 0.103 | 21.1 | 30.9 | 17.0 | 0.50 |
| | 4 | 2.58 | 0.165 | 15.6 | 26.0 | 23.5 | 0.45 |
| | 5 | 1.55 | 0.067 | 23.1 | 23.5 | 9.5 | 1.00 |
| | 6 | 0.78 | 0.013 | 60.0 | 11.6 | 1.8 | 2.40 |
| | 7 | 1.03 | 0.009 | 114.4 | 10.7 | 2.4 | 3.10 |
| 30-Sep-93 | 1 | 3.40 | 0.096 | 35.4 | 0.8 | 4.4 | ND |
| | 2 | 2.00 | 0.075 | 26.7 | 35.2 | 6.1 | 0.80 |
| | 3 | 3.78 | 0.223 | 17.0 | 44.5 | 41.0 | 0.80 |
| | 4 | 1.80 | 0.075 | 24.0 | 33.9 | 11.0 | 0.90 |
| | 5 | 1.33 | 0.059 | 22.5 | 13.7 | 4.5 | 1.40 |
| | 6 | 0.58 | 0.164 | 3.5 | 8.9 | 2.2 | 2.80 |
| | 7 | 0.67 | 0.051 | 13.1 | 5.7 | 2.1 | 3.30 |
| 21-Oct-93 | 1 | 2.00 | 0.157 | 12.7 | 2.9 | 15.5 | ND |
| | 2 | 1.59 | 0.101 | 15.7 | 3.8 | 13.8 | 0.60 |
| | 3 | 1.61 | 0.090 | 17.9 | 5.6 | 13.9 | 0.70 |
| | 4 | 1.03 | 0.056 | 18.4 | 9.5 | 5.3 | 1.10 |
| | 5 | 0.85 | 0.051 | 16.7 | 11.4 | 5.1 | 2.20 |
| | 6 | 0.61 | 0.023 | 26.5 | 5.3 | 3.2 | 2.10 |
| | 7 | 0.61 | 0.017 | 35.9 | 1.3 | 1.8 | 2.80 |

APPENDIX B - PHYTOPLANKTON COUNTS

| | Grid Count | Total Count | #/ml |
|---|---|---|---|
| STATION 2 | | | |
| 4 JUN 92 | | | |
| centrales | | 3108 | 684.58 |
| pennales | | 664 | 146.26 |
| *Scenedesmus* | | 107 | 23.57 |
| *Spirulina* | | 104 | 22.91 |
| *Crucigenia* | | 18 | 3.96 |
| *Gonium* | | 290 | 63.88 |
| *Pediastrum* | | 825 | 181.72 |
| *Actinastrum* | | 8 | 1.76 |
| *Cryptomonas* | | 6 | 1.32 |
| TOTAL | | 5130 | 1129.96 |
| 2 JUL 92 | | | |
| *Scenedesmus* | | 603 | 217.08 |
| *Oedogonium* | | 11 | 3.96 |
| centrales | | 27 | 9.72 |
| *Anabaena* | | 6 | 2.16 |
| *Chlorella* | | 34 | 12.24 |
| *Microcystis* | | 1330 | 478.80 |
| pennales | | 587 | 211.32 |
| *Peridinium* | | 18 | 6.48 |
| *Pediastrum* | | 11 | 3.96 |
| *Ceratium* | | 9 | 3.24 |
| *Closterium* | | 2 | 0.72 |
| *Cryptomonas* | | 284 | 102.24 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Actinastrum* | | 6 | 2.16 |
| TOTAL | | 2928 | 1054.07 |
| 19 AUG 92 | | | |
| *Gleocystis* | 17 | 6054.68 | 605.47 |
| *Peridinium* | 26 | 260 | 26.00 |
| *Cryptomonas* | 6 | 2136.94 | 213.69 |
| *Lyngbya/Oscillatoria* | 122 | 43451.20 | 4345.12 |
| *pennales* | 10 | 3561.57 | 356.16 |
| *Microcystis* | 21 | 7479.31 | 747.93 |
| *Spirulina* | 44 | 15670.93 | 1567.09 |
| centrales | 8 | 2849.26 | 284.93 |
| *Aphanocapsa* | 22 | 7835.46 | 783.55 |
| *Scenedesmus* | 14 | 4986.20 | 498.62 |
| *Crucigenia* | 6 | 60 | 6.00 |
| *Mallomonas* | 4 | 40 | 4.00 |
| *Merismopedia* | 5 | 1780.79 | 178.08 |
| *Actinastrum* | 6 | 60 | 6.00 |
| *Ceratium* | 5 | 50 | 5.00 |
| *Pediastrum* | 2 | 20 | 2.00 |
| TOTAL | 318 | | 9629.63 |
| 12 SEP 92 | | | |
| *Gleocystis* | 7 | 3739.65 | 277.01 |
| *Peridinium* | 3 | 1602.71 | 118.72 |
| *Cryptomonas* | 20 | 10684.72 | 791.46 |
| *Lyngbya/Oscillatoria* | 94 | 50218.19 | 3719.87 |
| pennales | 2 | 1068.47 | 79.15 |
| *Microcystis* | 15 | 8013.54 | 593.60 |
| *Spirulina* | 33 | 17629.79 | 1305.91 |
| centrales | 4 | 2136.94 | 158.29 |
| *Platydorina* | 1 | 534.24 | 39.57 |
| *Scenedesmus* | 4 | 2136.94 | 158.29 |
| *Crucigenia* | 4 | 2136.94 | 158.29 |
| *Mallomonas* | 1 | 534.24 | 39.57 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Merismopedia* | 17 | 9082.01 | 672.74 |
| *Actinastrum* | 1 | 534.24 | 39.57 |
| *Ceratium* | 1 | 534.24 | 39.57 |
| *Pandorina* | 1 | 534.24 | 39.57 |
| *Anabaena* | 1 | 534.24 | 39.57 |
| TOTAL | 209 | | 8270.77 |
| 24 OCT 92 | | | |
| *Melosira* | 34 | | 93.62 |
| centrales | 214 | | 589.28 |
| pennales | 97 | | 267.10 |
| *Platydorina* | 4 | | 11.01 |
| *Pandorina* | 3 | | 8.26 |
| *Chlorella* | 26 | | 71.59 |
| *Closterium* | 3 | | 8.26 |
| *Gleocystis* | 14 | | 38.55 |
| *Scenedesmus* | 3 | | 8.26 |
| *Lyngbya/Oscillatoria* | 6 | | 16.52 |
| *Microcystis* | 4 | | 11.01 |
| TOTAL | 408 | | 1123.48 |
| 8 MAR 93 | | | |
| pennales | 5 | 50 | 5.26 |
| *Melosira* | 2 | 20 | 2.10 |
| *Scenedesmus* | 3 | 30 | 3.16 |
| *Cryptomonas* | 4 | 40 | 4.21 |
| centrales | 17 | 170 | 17.89 |
| TOTAL | 31 | | 32.63 |
| 18 APR 93 | | | |
| centrales | 8 | 80 | 22.86 |
| pennales | 32 | 320 | 91.43 |
| *Gomphonema* | 7 | 70 | 20.00 |
| *Gleocystis* | 1 | 10 | 2.86 |
| *Chlorella* | 50 | 500 | 142.86 |
| *Cryptomonas* | 3 | 30 | 8.57 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| TOTAL | 101 | | 288.57 |
| 26 MAY 93 | | | |
| *Asterionella* | 6 | 1227.61 | 129.22 |
| centrales | 1080 | | 2160.00 |
| *Chlorella* | 109 | 22301.52 | 2347.53 |
| pennales | 4 | 818.40 | 86.15 |
| *Oedogonium* | 5 | 1023.01 | 107.68 |
| *Cryptomonas* | 6 | 1227.61 | 129.22 |
| *Pediastrum* | 1 | 204.60 | 21.54 |
| *Anabaena* | 2 | 409.20 | 43.07 |
| *Actinastrum* | 3 | 613.80 | 64.61 |
| *Scenedesmus* | 4 | 818.40 | 86.15 |
| *Gleocystis* | 2 | 409.20 | 43.07 |
| *Crucigenia* | 1 | 204.60 | 21.54 |
| *Sphaerocystis* | 1 | 204.60 | 21.54 |
| *Melosira* | 4 | 818.40 | 86.15 |
| TOTAL | 122 | | 5347.47 |
| 25 JUN 93 | | | |
| *Pediastrum* | 1 | 234.54 | 18.61 |
| *Peridinium* | 26 | 6098.11 | 483.98 |
| *Anabaena* | 23 | 5394.48 | 428.13 |
| *Pandorina* | 10 | 2345.43 | 186.15 |
| *Aphanocapsa* | 9 | 2110.88 | 167.53 |
| *Oocystis* | 8 | 1876.34 | 148.92 |
| *Cryptomonas* | 18 | 4221.77 | 335.06 |
| *Gonium* | 2 | 469.09 | 37.23 |
| *Gleocystis* | 6 | 1407.26 | 111.69 |
| *Lyngbya/Oscillatoria* | 12 | 2814.51 | 223.37 |
| *Mallomonas* | 4 | 938.17 | 74.46 |
| *Scenedesmus* | 3 | 703.63 | 55.84 |
| *Sphaerocystis* | 2 | 469.09 | 37.23 |
| *Coelastrum* | 3 | 703.63 | 55.84 |
| pennales | 6 | 1407.26 | 111.69 |

| Continued | Grid | Total Count | #/ml |
|---|---|---|---|
| TOTAL | 133 | | 2475.73 |
| 23 JUL 93 | | | |
| *Ulothrix* | 4 | 836.20 | 64.32 |
| *Peridinium* | 6 | 1254.29 | 96.48 |
| *Oocystis* | 6 | 1254.29 | 96.48 |
| *Gleocystis* | 14 | 2926.68 | 225.13 |
| *Cryptomonas* | 10 | 2090.49 | 160.81 |
| *Staurastrum* | 3 | 627.15 | 48.24 |
| *Aphanocapsa* | 28 | 5853.37 | 450.26 |
| *Pandorina* | 2 | 418.10 | 32.16 |
| *Mallomonas* | 2 | 418.10 | 32.16 |
| pennales | 31 | 6480.52 | 498.50 |
| *Scenedesmus* | 10 | 2090.49 | 160.81 |
| *Microcystis* | 10 | 2090.49 | 160.81 |
| *Spirulina* | 2 | 418.10 | 32.16 |
| *Ceratium* | 1 | 209.05 | 16.08 |
| TOTAL | 129 | | 2074.41 |
| 4 AUG 93 | | | |
| *Gonium* | 12 | 2195.83 | 731.94 |
| *Scenedesmus* | 17 | 3110.77 | 1036.92 |
| pennales | 19 | 3476.74 | 1158.91 |
| *Microcystis* | 28 | 5123.61 | 1707.87 |
| *Aphanocapsa* | 21 | 3842.71 | 1280.90 |
| *Chlorella* | 3 | 548.96 | 182.99 |
| *Oocystis* | 1 | 182.99 | 61.00 |
| *Gleocystis* | 6 | 1097.92 | 365.97 |
| *Spirulina* | 2 | 365.97 | 121.99 |
| *Platydorina* | 1 | 182.99 | 61.00 |
| *Anabaena* | 8 | 1463.89 | 487.96 |
| TOTAL | | | 7197.46 |
| 19 AUG 93 | | | |
| *Gleocystis* | 5 | 2003.39 | 139.61 |
| pennales | 48 | 19232.50 | 1340.24 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Spirulina* | 36 | 14424.38 | 1005.18 |
| *Cryptomonas* | 25 | 10016.93 | 698.04 |
| *Aphanocapsa* | 20 | 8013.54 | 558.43 |
| *Merismopedia* | 4 | 1602.71 | 111.69 |
| *Peridinium* | 6 | 2404.06 | 167.53 |
| *Mallomonas* | 3 | 1202.03 | 83.77 |
| *Lyngbya/Oscillatoria* | 1 | 400.68 | 27.92 |
| *Chlamydomonas* | 18 | 7212.19 | 502.59 |
| *Ceratium* | 3 | 1202.03 | 83.77 |
| *Platydorina* | 4 | 1602.71 | 111.69 |
| *Pediastrum* | 2 | 801.35 | 55.84 |
| *Crucigenia* | 5 | 2003.39 | 139.61 |
| *Anabaena* | 3 | 1202.03 | 83.77 |
| *Scenedesmus* | 4 | 1602.71 | 111.69 |
| *Pandorina* | 3 | 1202.03 | 83.77 |
| TOTAL | | | 5305.13 |
| 2 SEPT 93 | | | |
| *Scenedesmus* | 5 | 5306.60 | 363.47 |
| pennales | 22 | 23349.04 | 1599.25 |
| centrales | 6 | 6367.92 | 436.16 |
| *Merismopedia* | 25 | 26533.00 | 1817.33 |
| *Pediastrum* | 1 | 1061.32 | 72.69 |
| *Ankistrodesmus* | 6 | 6367.92 | 436.16 |
| *Spirulina* | 8 | 8490.56 | 581.55 |
| *Chlamydomonas* | 9 | 9551.88 | 654.24 |
| *Anabaena* | 8 | 8490.56 | 581.55 |
| *Gonium* | 7 | 7429.24 | 508.85 |
| *Microcystis* | 9 | 9551.88 | 654.24 |
| *Gleocystis* | 9 | 9551.88 | 654.24 |
| *Cryptomonas* | 2 | 2122.64 | 145.39 |
| *Coelastrum* | 2 | 2122.64 | 145.39 |
| *Cosmarium* | 2 | 2122.64 | 145.39 |
| *Staurastrum* | 1 | 1061.32 | 72.69 |

| Continued | Gierid | Total Count | #/ml |
|---|---|---|---|
| *Euglena* | 1 | 1061.32 | 72.69 |
| TOTAL | 122 | | 8868.56 |
| 16 SEPT 93 | | | |
| *Aphanocapsa* | 27 | 5524.23 | 753.30 |
| *Microcystis* | 31 | 6342.63 | 864.90 |
| *Lyngbya/Oscillatoria* | 8 | 1636.81 | 223.20 |
| pennales | 5 | 1023.01 | 139.50 |
| *Spirulina* | 3 | 613.80 | 83.70 |
| *Gleocystis* | 7 | 1432.21 | 195.30 |
| *centrales* | 3 | 613.80 | 83.70 |
| *Merismopedia* | 17 | 3478.22 | 474.30 |
| *Cryptomonas* | 8 | 1636.81 | 223.20 |
| *Pediastrum* | 1 | 204.60 | 27.90 |
| *Anabaena* | 1 | 204.60 | 27.90 |
| *Microspora* | 1 | 204.60 | 27.90 |
| *Actinastrum* | 1 | 204.60 | 27.90 |
| TOTAL | 113 | | 3152.72 |
| 30 SEPT 93 | | | |
| *Scenedesmus* | 7.00 | 2417.80 | 185.98 |
| pennales | 24.33 | 8404.73 | 646.52 |
| centrales | 1.00 | 345.40 | 26.57 |
| *Merismopedia* | 14.67 | 5065.87 | 389.68 |
| *Pediastrum* | 1.67 | 575.67 | 44.28 |
| *Actinastrum* | 1.33 | 460.53 | 35.43 |
| *Gleocystis* | 4.67 | 1611.87 | 123.99 |
| *Ankistrodesmus* | 6.33 | 2187.53 | 168.27 |
| *Spirulina* | 14.33 | 4950.73 | 380.83 |
| *Microcystis* | 9.00 | 3108.60 | 239.12 |
| *Gonium* | 10.00 | 3454.00 | 265.69 |
| *Chlamydomonas* | 8.67 | 2993.47 | 230.27 |
| *Peridinium* | 2.67 | 921.07 | 70.85 |
| TOTAL | 105.67 | | 2807.48 |
| 21 OCT 93 | | | |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Gleocystis* | 24 | 5367.21 | 365.12 |
| *Spirulina* | 4 | 894.53 | 60.85 |
| *Aphanocapsa* | 40 | 8945.35 | 608.53 |
| *Microspora* | 12 | 2683.60 | 182.56 |
| *Cryptomonas* | 8 | 1789.07 | 121.71 |
| pennales | 4 | 894.53 | 60.85 |
| centrales | 4 | 894.53 | 60.85 |
| *Merismopedia* | 20 | 4472.67 | 304.26 |
| TOTAL | 116 | | 1764.73 |
| STATION 3 | | | |
| 4 JUN 92 | | | |
| centrales | | 3328 | 698.8807 |
| pennales | | 808 | 169.6802 |
| *Gleocystis* | | 621 | 130.4101 |
| *Pediastrum* | | 8 | 1.680002 |
| *Anabaena* | | 5 | 1.050001 |
| *Gonium* | | 537 | 112.7701 |
| *Actinastrum* | | 8 | 1.68 |
| *Melosira* | | 7 | 1.47 |
| *Chlorella* | | 8 | 1.68 |
| *Cryptomonas* | | 499 | 104.79 |
| *Scenedesmus* | | 19 | 3.99 |
| *Euglena* | | 4 | 0.84 |
| *Dinobryon* | | 4 | 0.84 |
| *Spirulina* | | 4 | 0.84 |
| TOTAL | | 5860 | 1230.601 |
| 2 JUL 92 | | | |
| *Scenedesmus* | | 763 | 274.68 |
| *Microcystis* | | 1626 | 585.36 |
| pennales | | 650 | 233.99 |
| *Cryptomonas* | | 291 | 104.76 |
| *Melosira* | | 1 | 0.36 |
| *Ceratium* | | 14 | 5.04 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Peridinium* | | 25 | 8.99 |
| *Chlorella* | | 17 | 6.12 |
| *Pediastrum* | | 9 | 3.24 |
| centrales | | 38 | 13.68 |
| *Actinastrum* | | 7 | 2.52 |
| *Oedogonium* | | 12 | 4.32 |
| TOTAL | | 3453 | 1243.07 |
| 19 AUG 92 | | | |
| *Mallomonas* | 1 | 331.59 | 55.27 |
| *Lyngbya/Oscillatoria* | 97 | 32164.70 | 5360.78 |
| *Spirulina* | 26 | 8621.47 | 1436.91 |
| *Microcystis* | 19 | 6300.30 | 1050.05 |
| *Cryptomonas* | 7 | 2321.16 | 386.86 |
| *Aphanocapsa* | 4 | 1326.38 | 221.06 |
| *Gleocystis* | 7 | 2321.16 | 386.86 |
| *Actinastrum* | 2 | 663.19 | 110.53 |
| *Scenedesmus* | 4 | 1326.38 | 221.06 |
| pennales | 5 | 1657.97 | 276.33 |
| *Ceratium* | 1 | 331.59 | 55.27 |
| *Platydorina* | 1 | 331.59 | 55.27 |
| *Pediastrum* | 1 | 331.59 | 55.27 |
| *Merismopedia* | 2 | 663.19 | 110.53 |
| TOTAL | 181 | | 10003.11 |
| 12 SEP 92 | | | |
| CONTAMINATED | | | |
| 24 OCT 92 | | | |
| *Gleocystis* | 30 | | 60 |
| *Melosira* | 30 | | 60 |
| *Chlorella* | 48 | | 96 |
| *Oocystis* | 18 | | 36 |
| *Peridinium* | 32 | | 64 |
| *Cryptomonas* | 78 | | 156 |
| *Gonium* | 35 | | 70 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| pennales | 75 | | 150 |
| centrales | 68 | | 136 |
| *Scenedesmus* | 7 | | 14 |
| *Microcystis* | 2 | | 4 |
| *Pediastrum* | 3 | | 6 |
| *Pandorina* | 10 | | 20 |
| *Platydorina* | 5 | | 10 |
| *Ceratium* | 1 | | 2 |
| *Actinastrum* | 3 | | 6 |
| *Lyngbya/Oscillatoria* | 3 | | 6 |
| *Crucigenia* | 3 | | 6 |
| TOTAL | 451 | | 902 |
| 8 MAR 93 | | | |
| *Scenedesmus* | 2 | 20 | 2.74 |
| centrales | 14 | 140 | 19.18 |
| pennales | 26 | 260 | 35.62 |
| *Coelastrum* | 4 | 40 | 5.48 |
| *Cryptomonas* | 4 | 40 | 5.48 |
| *Oedogonium* | 2 | 20 | 2.74 |
| *Melosira* | 2 | 20 | 2.74 |
| TOTAL | 54 | | 73.97 |
| 18 APR 93 | | | |
| *Spirulina* | 2 | 20 | 4.55 |
| *Cryptomonas* | 3 | 30 | 6.82 |
| centrales | 18 | 180 | 40.91 |
| *Chlorella* | 107 | 1070 | 243.18 |
| pennales | 44 | 440 | 100.00 |
| *Gomphonema* | 2 | 20 | 4.55 |
| *Closterium* | 4 | 40 | 9.09 |
| *Melosira* | 4 | 40 | 9.09 |
| *Scenedesmus* | 7 | 70 | 15.90 |
| *Oedogonium* | 2 | 20 | 4.55 |
| TOTAL | 193 | | 438.64 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| 26 MAY 93 | | | |
| centrales | 1160 | | 2562.79 |
| *Chlorella* | 124 | 27100.34 | 3151.20 |
| *Anabaena* | 7 | 1529.86 | 177.89 |
| pennales | 3 | 655.65 | 76.24 |
| *Cryptomonas* | 1 | 218.55 | 25.41 |
| *Oedogonium* | 9 | 1966.96 | 228.72 |
| *Asterionella* | 3 | 655.65 | 76.24 |
| *Scenedesmus* | 2 | 437.10 | 50.83 |
| *Actinastrum* | 3 | 655.65 | 76.24 |
| *Coelastrum* | 1 | 218.55 | 25.41 |
| *Melosira* | 18 | | 39.77 |
| *Gleocystis* | 1 | 218.55 | 25.41 |
| TOTAL | 1332 | | 6516.15 |
| 25 JUN 93 | | | |
| *Peridinium* | 21 | 5939.45 | 539.95 |
| *Pandorina* | 20 | 5656.62 | 514.24 |
| *Anabaena* | 15 | 4242.46 | 385.68 |
| *Gleocystis* | 8 | 2262.65 | 205.69 |
| *Cryptomonas* | 19 | 5373.79 | 488.53 |
| *Aphanocapsa* | 6 | 1696.99 | 154.27 |
| *Oocystis* | 5 | 1414.15 | 128.56 |
| *Ceratium* | 4 | 1131.32 | 102.85 |
| *Sphaerocystis* | 3 | 848.49 | 77.14 |
| *Pediastrum* | 1 | 282.83 | 25.71 |
| *Coelastrum* | 3 | 848.49 | 77.14 |
| *Chlamydomonas* | 5 | 1414.15 | 128.56 |
| *Lyngbya/Oscillatoria* | 7 | 1979.82 | 179.98 |
| *Scenedesmus* | 1 | 282.83 | 25.71 |
| *Mallomonas* | 7 | 1979.82 | 179.98 |
| centrales | 8 | 2262.65 | 205.69 |
| *Gonium* | 1 | 282.83 | 25.71 |
| TOTAL | 134 | | 3445.39 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| 23 JUL 93 | | | |
| *Actinastrum* | 1 | 343.44 | 28.15 |
| *Scenedesmus* | 10 | 3434.38 | 281.50 |
| pennales | 45 | 15454.69 | 1266.78 |
| *Microcystis* | 6 | 2060.63 | 168.90 |
| *Oocystis* | 24 | 8242.50 | 675.61 |
| *Aphanocapsa* | 6 | 2060.63 | 168.90 |
| *Gleocystis* | 16 | 5495.00 | 450.41 |
| *Mallomonas* | 2 | 686.88 | 56.30 |
| *Pandorina* | 2 | 686.88 | 56.30 |
| *Cryptomonas* | 6 | 2060.63 | 168.90 |
| *Platydorina* | 2 | 686.88 | 56.30 |
| *Merismopedia* | 4 | 1373.75 | 112.60 |
| *Staurastrum* | 1 | 343.44 | 28.15 |
| *Ulothrix* | 1 | 343.44 | 28.15 |
| *Peridinium* | 2 | 686.88 | 56.30 |
| *Lyngbya/Oscillatoria* | 1 | 343.44 | 28.15 |
| *Euglena* | 1 | 343.44 | 28.15 |
| *Gonium* | 2 | 686.88 | 56.30 |
| TOTAL | 132 | | 3715.88 |
| 4 AUG 93 | | | |
| *Microcystis* | | | 1497.3 |
| *Chroococcus* | | | 294.7 |
| *Chlorella* | | | 796.4 |
| *Lyngbya/Oscillatoria* | | | 495.4 |
| *Tetraedron* | | | 8 |
| *Navicula* | | | 111.4 |
| *Scenedesmus* | | | 191.1 |
| *Cyclotella* | | | 23.9 |
| *Cymbella* | | | 8 |
| *Melosira* | | | 15.9 |
| *Pediastrum* | | | 127.4 |
| TOTAL | | | 3609.3 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| 19 AUG 93 | | | |
| *Stephanoon* | 2 | 1748.41 | 134.49 |
| pennales | 26 | 22729.32 | 1748.41 |
| *Crucigenia* | 3 | 2622.61 | 201.74 |
| *Aphanocapsa* | 15 | 13113.07 | 1008.70 |
| *Gleocystis* | 10 | 8742.05 | 672.47 |
| *Spirulina* | 43 | 37590.80 | 2891.60 |
| *Scenedesmus* | 3 | 2622.61 | 201.74 |
| *Merismopedia* | 12 | 10490.45 | 806.96 |
| *Chlamydomonas* | 6 | 5245.23 | 403.48 |
| *Cryptomonas* | 7 | 6119.43 | 470.73 |
| TOTAL | | | 8540.31 |
| 2 SEPT 93 | | | |
| *Pediastrum* | 3 | 1923.25 | 184.93 |
| *Crucigenia* | 5 | 3205.42 | 308.21 |
| *Merismopedia* | 17 | 10898.42 | 1047.92 |
| *Gleocystis* | 17 | 10898.42 | 1047.92 |
| *Spirulina* | 14 | 8975.17 | 863.00 |
| pennales | 26 | 16668.17 | 1602.71 |
| *Aphanocapsa* | 17 | 10898.42 | 1047.92 |
| *Cryptomonas* | 4 | 2564.33 | 246.57 |
| *Scenedesmus* | 5 | 3205.42 | 308.21 |
| centrales | 6 | 3846.50 | 369.86 |
| *Lyngbya/Oscillatoria* | 9 | 5769.75 | 554.78 |
| TOTAL | 123 | | 7582.04 |
| 16 SEPT 93 | | | |
| *Gleocystis* | 5 | 2185.51 | 203.30 |
| *Microcystis* | 4 | 1748.41 | 162.64 |
| *Merismopedia* | 20 | 8742.05 | 813.21 |
| *Cryptomonas* | 10 | 4371.02 | 406.61 |
| *Aphanocapsa* | 26 | 11364.66 | 1057.18 |
| *Crucigenia* | 7 | 3059.72 | 284.62 |
| *Lyngbya/Oscillatoria* | 23 | 10053.35 | 935.20 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Anabaena* | 6 | 2622.61 | 243.96 |
| pennales | 5 | 2185.51 | 203.30 |
| *Scenedesmus* | 7 | 3059.72 | 284.62 |
| *Spirulina* | 7 | 3059.72 | 284.62 |
| TOTAL | 120 | | 4879.28 |
| 30 SEPT 93 | | | |
| *Crucigenia* | 21 | 8780.05 | 888.67 |
| *Merismopedia* | 7 | 2926.68 | 296.22 |
| *Scenedesmus* | 7 | 2926.68 | 296.22 |
| *Microcystis* | 8 | 3344.78 | 338.54 |
| *Cryptomonas* | 5 | 2090.49 | 211.59 |
| *Lyngbya/Oscillatoria* | 16 | 6689.57 | 677.08 |
| *Gleocystis* | 16 | 6689.57 | 677.08 |
| *Chlamydomonas* | 21 | 8780.05 | 888.67 |
| *Ankistrodesmus* | 1 | 418.10 | 42.32 |
| *Aphanocapsa* | 14 | 5853.37 | 592.45 |
| pennales | 9 | 3762.88 | 380.86 |
| *Spirulina* | 3 | 1254.29 | 126.95 |
| *Anabaena* | 3 | 1254.29 | 126.95 |
| TOTAL | 131 | | 5543.61 |
| 21 OCT 93 | | | |
| *Ulothrix* | 16 | 2903.02 | 227.69 |
| pennales | 11 | 1995.83 | 156.54 |
| *Gleocystis* | 39 | 7076.11 | 554.99 |
| *Cryptomonas* | 16 | 2903.02 | 227.69 |
| *Pandorina* | 8 | 1451.51 | 113.84 |
| *Spirulina* | 11 | 1995.83 | 156.54 |
| *Scenedesmus* | 9 | 1632.95 | 128.07 |
| *Merismopedia* | 13 | 2358.70 | 185.00 |
| TOTAL | 123 | | 1750.35 |
| STATION 4 | | | |
| 4 JUN 92 | | | |
| *Spirulina* | | 183 | 38.43 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Scenedesmus* | | 1427 | 299.67 |
| *Pediastrum* | | 20 | 4.20 |
| *Chlorella* | | 2221 | 466.41 |
| *Actinastrum* | | 30 | 6.30 |
| *Cryptomonas* | | 1486 | 312.06 |
| centrales | | 1368 | 287.28 |
| pennales | | 212 | 44.52 |
| *Gonium* | | 533 | 111.93 |
| *Dinobryon* | | 2 | 0.42 |
| *Platydorina* | | 338 | 70.98 |
| *Oedogonium* | | 3 | 0.63 |
| *Mougeotia* | | 13 | 2.73 |
| *Euglena* | | 1 | 0.21 |
| *Coelastrum* | | 4 | 0.84 |
| TOTAL | | 7841 | 1646.61 |
| 2 JUL 92 | | | |
| *Scenedesmus* | | 632 | 240.16 |
| *Microcystis* | | 1377 | 523.26 |
| pennales | | 818 | 310.84 |
| *Cryptomonas* | | 529 | 201.02 |
| *Ceratium* | | 18 | 6.84 |
| *Pediastrum* | | 10 | 3.80 |
| *Anabaena* | | 8 | 3.04 |
| *Melosira* | | 12 | 4.56 |
| centrales | | 257 | 97.66 |
| *Oedogonium* | | 16 | 6.08 |
| *Spirulina* | | 372 | 141.36 |
| *Actinastrum* | | 5 | 1.90 |
| *Peridinium* | | 15 | 5.70 |
| *Lyngbya/Oscillatoria* | | 60 | 22.80 |
| TOTAL | | 4129 | 1569.02 |
| 19 AUG 92 | | | |
| *Peridinium* | 6 | 2747.50 | 209.73 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Mallomonas* | 1 | 457.92 | 34.96 |
| *Lyngbya/Oscillatoria* | 107 | 48997.08 | 3740.24 |
| *Spirulina* | 88 | 40296.67 | 3076.08 |
| *Microcystis* | 20 | 9158.33 | 699.11 |
| *Cryptomonas* | 4 | 1831.67 | 139.82 |
| *Aphanocapsa* | 23 | 10532.08 | 803.98 |
| *Gleocystis* | 16 | 7326.67 | 559.29 |
| *Actinastrum* | 2 | 915.83 | 69.91 |
| *Scenedesmus* | 6 | 2747.50 | 209.73 |
| pennales | 18 | 8242.50 | 629.20 |
| *Coelastrum* | 3 | 1373.75 | 104.87 |
| *Cosmarium* | 2 | 915.83 | 69.91 |
| *Pediastrum* | 1 | 457.92 | 34.96 |
| *Merismopedia* | 11 | 5037.08 | 384.51 |
| *Crucigenia* | 3 | 1373.75 | 104.87 |
| *Ceratium* | 2 | 915.83 | 69.91 |
| *Staurastrum* | 1 | 457.92 | 34.96 |
| TOTAL | 314 | | 10976.02 |
| 12 SEP 92 | | | |
| *Peridinium* | 3 | 848.49 | 121.21 |
| *Mallomonas* | 3 | 848.49 | 121.21 |
| *Lyngbya/Oscillatoria* | 52 | 14707.21 | 2101.03 |
| *Spirulina* | 56 | 15838.53 | 2262.65 |
| *Microcystis* | 20 | 5656.62 | 808.09 |
| *Cryptomonas* | 4 | 1131.32 | 161.62 |
| *Aphanocapsa* | 14 | 3959.63 | 565.66 |
| *Gleocystis* | 6 | 1696.99 | 242.43 |
| *Actinastrum* | 1 | 282.83 | 40.40 |
| *Scenedesmus* | 3 | 848.49 | 121.21 |
| pennales | 11 | 3111.14 | 444.45 |
| *Gonium* | 1 | 282.83 | 40.40 |
| *Cosmarium* | 1 | 282.83 | 40.40 |
| *Anabaena* | 1 | 282.83 | 40.40 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Merismopedia* | 13 | 3676.80 | 525.26 |
| *Crucigenia* | 3 | 848.49 | 121.21 |
| *Ceratium* | 2 | 565.66 | 80.81 |
| *Euglena* | 2 | 565.66 | 80.81 |
| centrales | 3 | 848.49 | 121.21 |
| TOTAL | 199 | | 8040.478 |
| 24 OCT 92 | | | |
| *Peridinium* | 13 | 260.00 | 52.00 |
| *Mallomonas* | 14 | 280.00 | 56.00 |
| *Spirulina* | 7 | 140.00 | 28.00 |
| *Microcystis* | 10 | 200.00 | 40.00 |
| *Cryptomonas* | 12 | 240.00 | 48.00 |
| *Pandorina* | 42 | 840.00 | 168.00 |
| *Gleocystis* | 28 | 560.00 | 112.00 |
| *Melosira* | 19 | 380.00 | 76.00 |
| *Scenedesmus* | 11 | 220.00 | 44.00 |
| pennales | 40 | 800.00 | 160.00 |
| *Actinastrum* | 13 | 260.00 | 52.00 |
| *Chlamydomonas* | 18 | 360.00 | 72.00 |
| *Staurastrum* | 2 | 40.00 | 8.00 |
| *Merismopedia* | 8 | 160.00 | 32.00 |
| *Crucigenia* | 2 | 40.00 | 8.00 |
| *Pediastrum* | 3 | 60.00 | 12.00 |
| centrales | 34 | 680.00 | 136.00 |
| TOTAL | 276 | | 1104 |
| 8 MAR 93 | | | |
| *Peridinium* | 1 | 10.00 | 3.23 |
| *Cryptomonas* | 4 | 40.00 | 8.00 |
| *Platydorina* | 1 | 10.00 | 2.00 |
| *Oocystis* | 1 | 10.00 | 2.00 |
| *Gonium* | 1 | 10.00 | 2.00 |
| *Scenedesmus* | 3 | 30.00 | 6.00 |
| pennales | 8 | 80.00 | 16.00 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| centrales | 22 | 220.00 | 44.00 |
| TOTAL | 41 | | 83.23 |
| 18 APR 93 | | | |
| CONTAMINATED | | | |
| 26 MAY 93 | | | |
| centrales | 1838 | | 3492.2 |
| *Melosira* | 32 | | 60.8 |
| *Cryptomonas* | 81 | | 153.9 |
| *Pediastrum* | 2 | | 3.8 |
| *Actinastrum* | 22 | | 41.8 |
| pennales | 128 | | 243.2 |
| *Gleocystis* | 38 | | 72.2 |
| *Sphaerocystis* | 3 | | 5.7 |
| *Chlorella* | 36 | | 68.7 |
| *Scenedesmus* | 2 | | 3.8 |
| *Microcystis* | 2 | | 3.8 |
| TOTAL | 2184 | | 4149.6 |
| 25 JUN 93 | | | |
| *Pediastrum* | 2 | | 30.16 |
| *Peridinium* | 52 | | 784.13 |
| *Anabaena* | 48 | | 723.81 |
| *Pandorina* | 21 | | 316.67 |
| *Aphanocapsa* | 18 | | 271.43 |
| *Oocystis* | 16 | | 241.27 |
| *Cryptomonas* | 36 | | 542.86 |
| *Gonium* | 4 | | 60.32 |
| *Gleocystis* | 12 | | 180.95 |
| *Lyngbya/Oscillatoria* | 24 | | 361.90 |
| *Mallomonas* | 8 | | 120.63 |
| *Scenedesmus* | 6 | | 90.48 |
| *Sphaerocystis* | 4 | | 60.32 |
| *Coelastrum* | 6 | | 90.48 |
| pennales | 12 | | 180.95 |

| Continued | Grid Count | Total | #/ml |
|---|---|---|---|
| centrales | 10 | | 150.79 |
| TOTAL | 279 | | 4207.14 |
| 23 JUL 93 | | | |
| *Coelastrum* | 4 | 1326.38 | 111.46 |
| pennales | 38 | 12600.60 | 1058.87 |
| *Aphanocapsa* | 18 | 5968.71 | 501.57 |
| *Mallomonas* | 6 | 1989.57 | 167.19 |
| *Tetraedron* | 1 | 331.59 | 27.86 |
| *Gleocystis* | 12 | 3979.14 | 334.38 |
| *Spirulina* | 5 | 1657.97 | 139.33 |
| *Oocystis* | 19 | 6300.30 | 529.44 |
| *Cryptomonas* | 11 | 3647.54 | 306.52 |
| *Scenedesmus* | 5 | 1657.97 | 139.33 |
| *Peridinium* | 3 | 994.78 | 83.59 |
| *Microcystis* | 2 | 663.19 | 55.73 |
| *Pandorina* | 3 | 994.78 | 83.59 |
| *Gonium* | 2 | 663.19 | 55.73 |
| *Actinastrum* | 1 | 331.59 | 27.86 |
| *Crucigenia* | 2 | 663.19 | 55.73 |
| *Staurastrum* | 1 | 331.59 | 27.86 |
| TOTAL | 133 | | 3706.06 |
| 4 AUG 93 | | | |
| *Synedra* | | | 39.8 |
| *Microcystis* | | | 2269.8 |
| *Chlroococcus* | | | 294.7 |
| *Chlorella* | | | 1115 |
| *Lyngbya/Oscillatoria* | | | 886.4 |
| *Melosira* | | | 143.4 |
| *Scenedesmus* | | | 111.5 |
| *Cyclotella* | | | 39.8 |
| *Navicula* | | | 39.8 |
| *Asterionella* | | | 15.9 |
| *Pediastrum* | | | 127.4 |

| Continued | Grid Conunt | Total Count | #/ml |
|---|---|---|---|
| TOTAL | | | 5083.5 |
| 19 AUG 93 | | | |
| pennales | 31 | 21293.13 | 1980.76 |
| *Spirulina* | 32 | 21980.00 | 2044.65 |
| *Merismopedia* | 12 | 8242.50 | 766.74 |
| *Cryptomonas* | 11 | 7555.63 | 702.85 |
| *Ankistrodesmus* | 3 | 2060.63 | 191.69 |
| *Microcystis* | 3 | 2060.63 | 191.69 |
| *Scenedesmus* | 5 | 3434.38 | 319.48 |
| *Aphanocapsa* | 5 | 3434.38 | 319.48 |
| *Gleocystis* | 5 | 3434.38 | 319.48 |
| *Chlamydomonas* | 6 | 4121.25 | 383.37 |
| *Pediastrum* | 1 | 686.88 | 63.90 |
| TOTAL | | | 7284.07 |
| 2 SEPT 93 | | | |
| *Scenedesmus* | 12 | 2747.50 | 345.49 |
| pennales | 24 | 5495.00 | 690.99 |
| *Spirulina* | 22 | 5037.08 | 633.41 |
| *Gleocystis* | 13 | 2976.46 | 374.29 |
| *Microcystis* | 11 | 2518.54 | 316.70 |
| *Pediastrum* | 3 | 686.88 | 86.37 |
| *Cryptomonas* | 9 | 2060.63 | 259.12 |
| *Crucigenia* | 7 | 1602.71 | 201.54 |
| *Chlamydomonas* | 5 | 1144.79 | 143.96 |
| centrales | 1 | 228.96 | 28.79 |
| *Pandorina* | 5 | 1144.79 | 143.96 |
| *Merismopedia* | 2 | 457.92 | 57.58 |
| *Actinastrum* | 1 | 228.96 | 28.79 |
| TOTAL | 115 | | 3310.984 |
| 16 SEPT 93 | | | |
| *Anabaena* | 3 | 1602.71 | 178.08 |
| *Crucigenia* | 3 | 1602.71 | 178.08 |
| pennales | 11 | 5876.60 | 652.96 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Pandorina* | 7 | 3739.65 | 415.52 |
| *Gonium* | 2 | 1068.47 | 118.72 |
| *Rhizoclonium* | 12 | 6410.83 | 712.31 |
| *Pediastrum* | 2 | 1068.47 | 118.72 |
| *Gleocystis* | 13 | 6945.07 | 771.67 |
| *Microcystis* | 10 | 5342.36 | 593.60 |
| *Peridinium* | 2 | 1068.47 | 118.72 |
| *Chlamydomonas* | 33 | 17629.79 | 1958.87 |
| *Merismopedia* | 4 | 2136.94 | 237.44 |
| *Scenedesmus* | 5 | 2671.18 | 296.80 |
| *Spirulina* | 8 | 4273.89 | 474.88 |
| *Aphanocapsa* | 2 | 1068.47 | 118.72 |
| *Cryptomonas* | 7 | 3739.65 | 415.52 |
| *Actinastrum* | 6 | 3205.42 | 356.16 |
| TOTAL | 130 | | 7716.744 |
| 30 SEP 93 | | | |
| *Actinastrum* | 2 | 1479.42 | 97.01 |
| *Pediastrum* | 3 | 2219.13 | 145.52 |
| *Merismopedia* | 3 | 2219.13 | 145.52 |
| *Crucigenia* | 7 | 5177.98 | 339.54 |
| *Gleocystis* | 9 | 6657.40 | 436.55 |
| *Spirulina* | 8 | 5917.69 | 388.05 |
| pennales | 17 | 12575.10 | 824.60 |
| *Chlamydomonas* | 16 | 11835.38 | 776.09 |
| *Cryptomonas* | 15 | 11095.67 | 727.59 |
| *Anabaena* | 6 | 4438.27 | 291.03 |
| *Scenedesmus* | 8 | 5917.69 | 388.05 |
| *Pandorina* | 7 | 5177.98 | 339.54 |
| *Gonium* | 12 | 8876.54 | 582.07 |
| *Aphanocapsa* | 7 | 5177.98 | 339.54 |
| *Staurastrum* | 1 | 739.71 | 48.51 |
| *Ulothrix* | 1 | 739.71 | 48.51 |
| TOTAL | 122 | | 5917.69 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| 21 OCT 93 | | | |
| *Staurastrum* | 5 | 1265.30 | 89.42 |
| *Gleocystis* | 12 | 3036.71 | 214.61 |
| *Chlamydomonas* | 4 | 1012.24 | 71.54 |
| pennales | 36 | 9110.13 | 643.83 |
| *Ankistrodesmus* | 8 | 2024.47 | 143.07 |
| *Scenedesmus* | 7 | 1771.41 | 125.19 |
| *Spirulina* | 32 | 8097.89 | 572.29 |
| *Aphanocapsa* | 9 | 2277.53 | 160.96 |
| *Cryptomonas* | 11 | 2783.65 | 196.72 |
| *Lyngbya/Oscillatoria* | 2 | 506.12 | 35.77 |
| centrales | 3 | 759.18 | 53.65 |
| *Actinastrum* | 1 | 253.06 | 17.88 |
| *Merismopedia* | 3 | 759.18 | 53.65 |
| *Ulothrix* | 1 | 253.06 | 17.88 |
| *Peridinium* | 4 | 1012.24 | 71.54 |
| *Crucigenia* | 4 | 1012.24 | 71.54 |
| *Pediastrum* | 1 | 253.06 | 17.88 |
| TOTAL | 143 | | 2557.418 |
| STATION 5 | | | |
| 4 JUN 92 | | | |
| centrales | | 762 | 167.64 |
| pennales | | 528 | 116.16 |
| *Spirulina* | | 259 | 56.98 |
| *Scenedesmus* | | 756 | 166.32 |
| *Pediastrum* | | 22 | 4.84 |
| *Actinastrum* | | 4 | 0.88 |
| *Cryptomonas* | | 1008 | 221.76 |
| *Gonium* | | 659 | 144.98 |
| *Platydorina* | | 223 | 49.06 |
| *Oedogonium* | | 1 | 0.22 |
| *Chlorella* | | 434 | 95.48 |
| *Peridinium* | | 71 | 15.62 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Coelastrum* | | 57 | 12.54 |
| *Melosira* | | 1 | 0.22 |
| *Ceratium* | | 10 | 2.20 |
| *Euglena* | | 22 | 4.84 |
| TOTAL | | 4817 | 1059.74 |
| 2 JUL 92 | | | |
| *Microcystis* | | 1560 | 592.8 |
| pennales | | 511 | 194.18 |
| *Cryptomonas* | | 2098 | 797.24 |
| *Ceratium* | | 297 | 112.86 |
| *Peridinium* | | 11 | 4.18 |
| *Scenedesmus* | | 799 | 303.62 |
| centrales | | 428 | 162.64 |
| *Pediastrum* | | 11 | 4.18 |
| *Actinastrum* | | 4 | 1.52 |
| *Oedogonium* | | 11 | 4.18 |
| *Spirulina* | | 238 | 90.44 |
| *Anabaena* | | 8 | 3.04 |
| *Melosira* | | 4 | 1.52 |
| *Chlorella* | | 26 | 9.88 |
| *Lyngbya/Oscillatoria* | | 21 | 7.98 |
| TOTAL | | 6027 | 2290.26 |
| 19 AUG 92 | | | |
| pennales | 258 | | 4902 |
| *Ceratium* | 4 | | 76 |
| *Peridinium* | 17 | | 323 |
| *Gonium* | 2 | | 38 |
| *Cryptomonas* | 32 | | 608 |
| *Lyngbya/Oscillatoria* | 22 | | 418 |
| *Spirulina* | 344 | | 6536 |
| *Chlorella* | 8 | | 152 |
| *Gleocystis* | 3 | | 57 |
| *Actinastrum* | 2 | | 38 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Staurastrum* | 2 | | 38 |
| *Scenedesmus* | 5 | | 95 |
| *Microcystis* | 2 | | 38 |
| centrales | 54 | | 1026 |
| *Oocystis* | 6 | | 114 |
| *Merismopedia* | 2 | | 38 |
| *Pediastrum* | 1 | | 19 |
| *Anabaena* | 5 | | 95 |
| *Euglena* | 7 | | 133 |
| *Pandorina* | 2 | | 38 |
| *Melosira* | 3 | | 57 |
| TOTAL | 781 | | 14839 |
| 12 SEP 92 | | | |
| *Peridinium* | 2 | 40.00 | 26.67 |
| *Pandorina* | 1 | 20.00 | 13.33 |
| *Lyngbya/Oscillatoria* | 29 | 580.00 | 386.67 |
| *Spirulina* | 105 | 2100.00 | 1400.00 |
| *Microcystis* | 7 | 140.00 | 93.33 |
| *Cryptomonas* | 6 | 120.00 | 80.00 |
| *Aphanocapsa* | 22 | 440.00 | 293.33 |
| *Gleocystis* | 4 | 80.00 | 53.33 |
| *Scenedesmus* | 7 | 140.00 | 93.33 |
| pennales | 28 | 560.00 | 373.33 |
| *Staurastrum* | 1 | 20.00 | 13.33 |
| *Pediastrum* | 1 | 20.00 | 13.33 |
| *Merismopedia* | 27 | 540.00 | 360.00 |
| *Ceratium* | 1 | 20.00 | 13.33 |
| centrales | 17 | 340.00 | 226.67 |
| TOTAL | 258 | | 3440 |
| 24 OCT 92 | | | |
| *Melosira* | 50 | | 197.92 |
| *Peridinium* | 8 | | 31.67 |
| *Cryptomonas* | 154 | | 609.58 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Pandorina* | 8 | | 31.67 |
| *Actinastrum* | 4 | | 15.83 |
| centrales | 44 | | 174.17 |
| *Scenedesmus* | 18 | | 71.25 |
| *Oocystis* | 4 | | 15.83 |
| *Chlorella* | 16 | | 63.33 |
| *Gleocystis* | 16 | | 63.33 |
| *Crucigenia* | 4 | | 15.83 |
| *Spirulina* | 14 | | 55.42 |
| *Pediastrum* | 6 | | 23.75 |
| pennales | 28 | | 110.83 |
| *Staurastrum* | 6 | | 23.75 |
| *Microcystis* | 14 | | 55.42 |
| TOTAL | 394 | | 1559.58 |
| 8 MAR 93 | | | |
| NO VISIBLE CELLS | | | |
| 18 APR 93 | | | |
| pennales | 9 | 1730.93 | 288.49 |
| centrales | 12 | 2307.90 | 384.65 |
| *Chlorella* | 44 | 8462.30 | 1410.38 |
| *Melosira* | 3 | 576.98 | 96.16 |
| *Scenedesmus* | 27 | 5192.78 | 865.46 |
| *Closterium* | 2 | 384.65 | 64.11 |
| *Oedogonium* | 2 | 384.65 | 64.11 |
| *Gleocystis* | 2 | 384.65 | 64.11 |
| TOTAL | 101 | | 3237.47 |
| 26 MAY 93 | | | |
| centrales | 251 | | 476.90 |
| *Chlorella* | 83 | 11912.67 | 1267.31 |
| *Melosira* | 5 | 717.63 | 76.34 |
| *Mallomonas* | 2 | 287.05 | 30.54 |
| *Scenedesmus* | 6 | 861.16 | 91.61 |
| *Cryptomonas* | 5 | 717.63 | 76.34 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Asterionella* | 1 | 143.53 | 15.27 |
| *Sphaerocystis* | 1 | 143.53 | 15.27 |
| pennales | 1 | 143.53 | 15.27 |
| *Lyngbya/Oscillatoria* | 1 | 143.53 | 15.27 |
| TOTAL | 356 | | 2080.12 |
| 25 JUN 93 | | | |
| *Anabaena* | | | 876.10 |
| *Lyngbya/Oscillatoria* | | | 1871.70 |
| *Ceratium* | | | 10.00 |
| *Navicula* | | | 11.90 |
| *Cyclotella* | | | 6.00 |
| *Chlorella* | | | 23.90 |
| *Microcystis* | | | 21.90 |
| *Mougeotia* | | | 2.00 |
| *Sphaerocystis* | | | 234.90 |
| *Peridinium* | | | 2.00 |
| *Synedra* | | | 15.90 |
| *Eucapsis* | | | 79.60 |
| *Chroococcus* | | | 19.90 |
| *Synechococcus* | | | 10.00 |
| *Chlorococcus* | | | 17.90 |
| *Gomphonema* | | | 2.00 |
| *Melosira* | | | 4.00 |
| TOTAL | | | 3209.70 |
| 23 JUL 93 | | | |
| *Ceratium* | 2 | 400.68 | 64.63 |
| *Gleocystis* | 19 | 3806.43 | 613.94 |
| pennales | 35 | 7011.85 | 1130.94 |
| *Mallomonas* | 3 | 601.02 | 96.94 |
| *Anabaena* | 2 | 400.68 | 64.63 |
| *Sphaerocystis* | 3 | 601.02 | 96.94 |
| *Pandorina* | 6 | 1202.03 | 193.88 |
| *Staurastrum* | 1 | 200.34 | 32.31 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Cryptomonas* | 8 | 1602.71 | 258.50 |
| *Aphanocapsa* | 5 | 1001.69 | 161.56 |
| *Peridinium* | 14 | 2804.74 | 452.38 |
| *Tetraedron* | 5 | 1001.69 | 161.56 |
| *Scenedesmus* | 7 | 1402.37 | 226.19 |
| *Lyngbya/Oscillatoria* | 7 | 1402.37 | 226.19 |
| *Pediastrum* | 1 | 200.34 | 32.31 |
| *Ulothrix* | 3 | 601.02 | 96.94 |
| TOTAL | 12 | | 3909.83 |
| 4 AUG 93 | | | |
| *Oocystis* | 7 | 1615.05 | 556.91 |
| *Scenedesmus* | 9 | 2076.50 | 716.03 |
| *Chlorella* | 2 | 461.44 | 159.12 |
| *Gleocystis* | 1 | 230.72 | 79.56 |
| *Pediastrum* | 2 | 461.44 | 159.12 |
| *Ankistrodesmus* | 19 | 4383.71 | 1511.63 |
| *Staurastrum* | 1 | 230.72 | 79.56 |
| pennales | 11 | 2537.94 | 875.15 |
| centrales | 3 | 692.17 | 238.68 |
| *Cryptomonas* | 7 | 1615.05 | 556.91 |
| *Microcystis* | 19 | 4383.71 | 1511.63 |
| *Merismopedia* | 2 | 461.44 | 159.12 |
| *Spirulina* | 7 | 1615.05 | 556.91 |
| *Aphanocapsa* | 9 | 2076.50 | 716.03 |
| *Euglena* | 1 | 230.72 | 79.56 |
| TOTAL | 100 | | 7955.92 |
| 19 AUG 93 | | | |
| pennales | 12 | 3036.71 | 117.25 |
| *Ulothrix* | 32 | 8097.89 | 312.66 |
| *Chlamydomonas* | 12 | 3036.71 | 117.25 |
| *Gleocystis* | 13 | 3289.77 | 127.02 |
| *Microcystis* | 27 | 6832.60 | 263.81 |
| *Cryptomonas* | 18 | 4555.07 | 175.87 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Pediastrum* | 6 | 1518.36 | 58.62 |
| centrales | 4 | 1012.24 | 39.08 |
| *Scenedesmus* | 8 | 2024.47 | 78.17 |
| *Lyngbya/Oscillatoria* | 11 | 2783.65 | 107.48 |
| *Euglena* | 1 | 253.06 | 9.77 |
| *Merismopedia* | 7 | 1771.41 | 68.39 |
| TOTAL | 151 | | 1475.37 |
| 2 SEPT 93 | | | |
| *Pediastrum* | 3 | 3183.96 | 108.30 |
| pennales | 45 | 47759.40 | 1624.47 |
| *Gonium* | 8 | 8490.56 | 288.79 |
| *Merismopedia* | 8 | 8490.56 | 288.79 |
| *Scenedesmus* | 10 | 10613.20 | 360.99 |
| *Gleocystis* | 10 | 10613.20 | 360.99 |
| *Spirulina* | 17 | 18042.44 | 613.69 |
| *Anabaena* | 3 | 3183.96 | 108.30 |
| *Cosmarium* | 1 | 1061.32 | 36.10 |
| *Peridinium* | 6 | 60.00 | 2.04 |
| *Microcystis* | 7 | 7429.24 | 252.70 |
| *Actinastrum* | 1 | 1061.32 | 36.10 |
| *Cryptomonas* | 6 | 6367.92 | 216.60 |
| *Pandorina* | 3 | 3183.96 | 108.30 |
| *Chlamydomonas* | 11 | 11674.52 | 397.09 |
| *Euglena* | 1 | 1061.32 | 36.10 |
| *Mallomonas* | 3 | 30.00 | 1.02 |
| TOTAL | 143 | | 4840.37 |
| 16 SEPT 93 | | | |
| *Gleocystis* | 15 | 3898.48 | 271.67 |
| *Gonium* | 2 | 519.80 | 36.22 |
| *Microcystis* | 7 | 1819.29 | 126.78 |
| *Staurastrum* | 1 | 259.90 | 18.11 |
| *Spirulina* | 5 | 1299.49 | 90.56 |
| *Cryptomonas* | 34 | 8836.55 | 615.79 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Chlamydomonas* | 12 | 3118.78 | 217.34 |
| *Chodatella* | 2 | 519.80 | 36.22 |
| *Scenedesmus* | 12 | 3118.78 | 217.34 |
| pennales | 9 | 2339.09 | 163.00 |
| *Actinastrum* | 8 | 2079.19 | 144.89 |
| *Merismopedia* | 11 | 2858.89 | 199.23 |
| centrales | 1 | 259.90 | 18.11 |
| *Tetraedron* | 5 | 1299.49 | 90.56 |
| *Anabaena* | 3 | 779.70 | 54.33 |
| *Ankistrodesmus* | 3 | 779.70 | 54.33 |
| *Ulothrix* | 1 | 259.90 | 18.11 |
| TOTAL | 131 | | 2372.59 |
| 30 SEPT 93 | | | |
| *Gonium* | 9 | 3183.96 | 98.27 |
| *Scenedesmus* | 2 | 707.55 | 21.84 |
| *Chlamydomonas* | 11 | 3891.51 | 120.11 |
| *Gleocystis* | 5 | 1768.87 | 54.59 |
| *Pediastrum* | 2 | 707.55 | 21.84 |
| *Actinastrum* | 1 | 353.77 | 10.92 |
| *Coelastrum* | 1 | 353.77 | 10.92 |
| *Ulothrix* | 6 | 2122.64 | 65.51 |
| *Ankistrodesmus* | 2 | 707.55 | 21.84 |
| *Platydorina* | 5 | 1768.87 | 54.59 |
| pennales | 17 | 6014.15 | 185.62 |
| centrales | 2 | 707.55 | 21.84 |
| Cryptomonas | 9 | 3183.96 | 98.27 |
| *Microcystis* | 19 | 6721.69 | 207.46 |
| *Merismopedia* | 2 | 707.55 | 21.84 |
| *Spirulina* | 2 | 707.55 | 21.84 |
| *Anabaena* | 4 | 1415.09 | 43.68 |
| TOTAL | 99 | | 1080.97 |
| 21 OCT 93 | | | |
| *Ulothrix* | 37 | 5310.47 | 323.81 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| pennales | 8 | 1148.21 | 70.01 |
| *Cryptomonas* | 17 | 2439.94 | 148.78 |
| *Microcystis* | 23 | 3301.10 | 201.29 |
| centrales | 3 | 430.58 | 26.25 |
| *Gleocystis* | 16 | 2296.42 | 140.03 |
| *Chlamydomonas* | 5 | 717.63 | 43.76 |
| *Spirulina* | 2 | 287.05 | 17.50 |
| *Scenedesmus* | 6 | 861.16 | 52.51 |
| *Gonium* | 3 | 430.58 | 26.25 |
| TOTAL | 120 | | 1050.19 |
| STATION 6 | | | |
| 4 JUN 92 | | | |
| *Scenedesmus* | | 865 | 155.69 |
| *Pediastrum* | | 13 | 2.34 |
| *Cryptomonas* | | 577 | 103.85 |
| centrales | | 232 | 41.76 |
| pennales | | 6204 | 1116.63 |
| *Gonium* | | 370 | 66.59 |
| *Platydorina* | | 146 | 26.28 |
| *Oedogonium* | | 10 | 1.80 |
| *Chlorella* | | 239 | 43.02 |
| *Coelastrum* | | 116 | 20.88 |
| *Peridinium* | | 57 | 10.26 |
| *Ceratium* | | 5 | 0.90 |
| *Melosira* | | 3 | 0.54 |
| *Richterella* | | 6 | 1.08 |
| *Closterium* | | 4 | 0.72 |
| TOTAL | | 8847 | 1592.33 |
| 2 JUL 92 | | | |
| *Cryptomonas* | | 1864 | 708.32 |
| pennales | | 2558 | 972.04 |
| *Microcystis* | | 200 | 76.00 |
| *Scenedesmus* | | 788 | 299.44 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Ceratium* | | 19 | 7.22 |
| centrales | | 135 | 51.30 |
| *Pediastrum* | | 5 | 1.90 |
| *Peridinium* | | 14 | 5.32 |
| *Chlorella* | | 120 | 45.60 |
| *Oedogonium* | | 15 | 5.70 |
| *Spirulina* | | 126 | 47.88 |
| *Lyngbya/Oscillatoria* | | 82 | 31.16 |
| *Anabaena* | | 13 | 4.94 |
| *Richterella* | | 2 | 0.76 |
| *Melosira* | | 17 | 6.46 |
| TOTAL | | 5958 | 2264.04 |
| 19 AUG 92 | | | |
| *Lyngbya/Oscillatoria* | 63 | 11650.46 | 5825.23 |
| *Spirulina* | 43 | 7951.90 | 3975.95 |
| *Cryptomonas* | 2 | 369.86 | 184.93 |
| *Aphanocapsa* | 11 | 2034.21 | 1017.10 |
| *Gleocystis* | 4 | 739.71 | 369.86 |
| *Scenedesmus* | 5 | 924.64 | 462.32 |
| pennales | 9 | 1664.35 | 832.18 |
| *Melosira* | 2 | 369.86 | 184.93 |
| *Merismopedia* | 8 | 1479.42 | 739.71 |
| *Crucigenia* | 1 | 200.34 | 100.17 |
| TOTAL | 148 | | 13692.37 |
| 12 SEP 92 | | | |
| *Lyngbya/Oscillatoria* | 81 | 1620.00 | 265.57 |
| *Spirulina* | 299 | 5980.00 | 980.33 |
| *Cryptomonas* | 9 | 180.00 | 29.51 |
| *Aphanocapsa* | 18 | 360.00 | 59.02 |
| *Gleocystis* | 24 | 480.00 | 78.69 |
| *Scenedesmus* | 21 | 420.00 | 68.85 |
| pennales | 162 | 3240.00 | 531.15 |
| *Pandorina* | 1 | 20.00 | 3.28 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Merismopedia* | 88 | 1760.00 | 288.52 |
| *Crucigenia* | 3 | 60.00 | 9.84 |
| centrales | 58 | 1160.00 | 190.16 |
| *Pediastrum* | 2 | 40.00 | 6.56 |
| *Microcystis* | 25 | 500.00 | 81.97 |
| *Chlamydomonas* | 6 | 120.00 | 19.67 |
| *Peridinium* | 2 | 40.00 | 6.56 |
| *Mallomonas* | 3 | 60.00 | 9.84 |
| *Staurastrum* | 2 | 40.00 | 6.56 |
| *Actinastrum* | 1 | 20.00 | 3.28 |
| TOTAL | 805 | | 2639.34 |
| 24 OCT 92 | | | |
| *Melosira* | 130 | | 425.86 |
| *Peridinium* | 4 | | 13.10 |
| *Cryptomonas* | 64 | | 209.65 |
| *Gonium* | 4 | | 13.10 |
| centrales | 70 | | 229.31 |
| *Oocystis* | 118 | | 386.55 |
| *Chlorella* | 104 | | 340.69 |
| *Ceratium* | 2 | | 6.55 |
| *Gleocystis* | 12 | | 39.31 |
| *Spirulina* | 16 | | 52.41 |
| pennales | 90 | | 294.83 |
| *Scenedesmus* | 20 | | 65.52 |
| *Staurastrum* | 1 | | 3.28 |
| *Pandorina* | 2 | | 6.55 |
| *Microcystis* | 3 | | 9.83 |
| *Coelastrum* | 2 | | 6.55 |
| *Pediastrum* | 1 | | 3.28 |
| TOTAL | 643 | | 2106.38 |
| 8 MAR 93 | | | |
| *Peridinium* | 18 | | 34.2 |
| centrales | 58 | | 110.2 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| pennales | 7 | | 13.3 |
| *Melosira* | 2 | | 3.8 |
| *Gleocystis* | 1 | | 1.9 |
| TOTAL | 86 | | 163.4 |
| 18 APR 93 | | | |
| *Melosira* | 3 | 506.12 | 38.93 |
| centrales | 14 | 2361.89 | 181.68 |
| *Chlorella* | 48 | 8097.89 | 622.92 |
| pennales | 6 | 1012.24 | 77.86 |
| *Peridinium* | 11 | 1855.77 | 142.75 |
| *Scenedesmus* | 5 | 843.53 | 64.89 |
| *Gleocystis* | 6 | 1012.24 | 77.86 |
| *Cryptomonas* | 1 | 168.71 | 12.98 |
| *Oedogonium* | 6 | 1012.24 | 77.86 |
| *Closterium* | 2 | 337.41 | 25.95 |
| TOTAL | 102 | | 1323.69 |
| 26 MAY 93 | | | |
| *Mallomonas* | 4 | 369.86 | 38.93 |
| centrales | 226 | | 429.40 |
| *Chlorella* | 52 | 4808.13 | 506.12 |
| *Coelastrum* | 8 | 739.71 | 77.86 |
| *Pandorina* | 2 | 184.93 | 19.47 |
| *Gleocystis* | 7 | 647.25 | 68.13 |
| *Cryptomonas* | 16 | 1479.42 | 155.73 |
| *Oedogonium* | 4 | 369.86 | 38.93 |
| *Scenedesmus* | 3 | 277.39 | 29.19 |
| pennales | 11 | 1017.10 | 107.06 |
| *Lyngbya/Oscillatoria* | 3 | 277.39 | 29.19 |
| TOTAL | 336 | | 1500.04 |
| 25 JUN 93 | | | |
| *Ceratium* | 3 | 848.49 | 62.85 |
| *Anabaena* | 27 | 7636.43 | 565.66 |
| *Lyngbya/Oscillatoria* | 42 | 11878.90 | 879.92 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| pennales | 17 | 4808.13 | 356.16 |
| *Peridinium* | 17 | 4808.13 | 356.16 |
| *Mallomonas* | 7 | 1979.82 | 146.65 |
| *Coelastrum* | 3 | 848.49 | 62.85 |
| *Gleocystis* | 7 | 1979.82 | 146.65 |
| *Cryptomonas* | 6 | 1696.99 | 125.70 |
| *Aphanocapsa* | 7 | 1979.82 | 146.65 |
| *Sphaerocystis* | 7 | 1979.82 | 146.65 |
| *Melosira* | 2 | | 3.80 |
| centrales | 29 | | 55.10 |
| *Euglena* | 1 | | 1.90 |
| *Pediastrum* | 3 | 848.49 | 62.85 |
| TOTAL | 178 | | 3119.56 |
| 23 JUL 93 | | | |
| *Pandorina* | 4 | 1039.59 | 105.01 |
| *Gleocystis* | 14 | 3638.58 | 367.53 |
| pennales | 68 | 17673.11 | 1785.16 |
| *Coelastrum* | 5 | 1299.49 | 131.26 |
| *Lyngbya/Oscillatoria* | 5 | 1299.49 | 131.26 |
| *Oocystis* | 6 | 1559.39 | 157.51 |
| *Mallomonas* | 2 | 519.80 | 52.50 |
| *Aphanocapsa* | 6 | 1559.39 | 157.51 |
| *Crucigenia* | 2 | 519.80 | 52.50 |
| *Melosira* | 4 | 1039.59 | 105.01 |
| *Peridinium* | 1 | 259.90 | 26.25 |
| *Anabaena* | 4 | 1039.59 | 105.01 |
| *Tetraedron* | 1 | 259.90 | 26.25 |
| *Spirulina* | 2 | 519.80 | 52.50 |
| *Pediastrum* | 2 | 519.80 | 52.50 |
| *Actinastrum* | 1 | 259.90 | 26.25 |
| *Cryptomonas* | 2 | 519.80 | 52.50 |
| *Staurastrum* | 1 | 259.90 | 26.25 |
| *Scenedesmus* | 4 | 1039.59 | 105.01 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| TOTAL | 134 | | 3517.82 |
| 4 AUG 93 | | | |
| pennales | 45.33 | 12028.29 | 3341.19 |
| *Merismopedia* | 6.67 | 1768.87 | 491.35 |
| *Oocystis* | 10.67 | 2830.19 | 786.16 |
| *Aphanocapsa* | 7.33 | 1945.75 | 540.49 |
| *Cryptomonas* | 6.67 | 1768.87 | 491.35 |
| *Microcystis* | 6.67 | 1768.87 | 491.35 |
| *Ankistrodesmus* | 8.33 | 2211.08 | 614.19 |
| *Spirulina* | 9.00 | 2387.97 | 663.33 |
| *Anabaena* | 7.33 | 1945.75 | 540.49 |
| *Scenedesmus* | 6.00 | 1591.98 | 442.22 |
| *Actinastrum* | 0.67 | 176.89 | 49.14 |
| *Staurastrum* | 1.67 | 442.22 | 122.84 |
| *Gleocystis* | 1.00 | 265.33 | 73.70 |
| TOTAL | 117.33 | | 8647.79 |
| 19 AUG 93 | | | |
| *Gleocystis* | 6 | 1989.57 | 150.72 |
| pennales | 36 | 11937.41 | 904.35 |
| *Mallomonas* | 1 | 331.59 | 25.12 |
| *Chlamydomonas* | 7 | 2321.16 | 175.85 |
| *Cryptomonas* | 12 | 3979.14 | 301.45 |
| *Microcystis* | 7 | 2321.16 | 175.85 |
| centrales | 1 | 331.59 | 25.12 |
| *Ankistrodesmus* | 23 | 7626.68 | 577.78 |
| *Ulothrix* | 2 | 663.19 | 50.24 |
| *Scenedesmus* | 5 | 1657.97 | 125.60 |
| *Spirulina* | 12 | 3979.14 | 301.45 |
| *Merismopedia* | 2 | 663.19 | 50.24 |
| *Staurastrum* | 3 | 994.78 | 75.36 |
| TOTAL | | | 2939.14 |
| 2 SEPT 93 | | | |
| pennales | 62 | 22081.76 | 2007.43 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Chlamydomonas* | 6 | 2136.94 | 194.27 |
| *Cryptomonas* | 11 | 3917.73 | 356.16 |
| *Scenedesmus* | 2 | 712.31 | 64.76 |
| *Gleocystis* | 7 | 2493.10 | 226.65 |
| *Microcystis* | 4 | 1424.63 | 129.51 |
| *Spirulina* | 6 | 2136.94 | 194.27 |
| *Lyngbya/Oscillatoria* | 3 | 1068.47 | 97.13 |
| *Gonium* | 3 | 1068.47 | 97.13 |
| *Aphanocapsa* | 5 | 1780.79 | 161.89 |
| *Ankistrodesmus* | 12 | 4273.89 | 388.54 |
| *Actinastrum* | 1 | 356.16 | 32.38 |
| *Merismopedia* | 2 | 712.31 | 64.76 |
| TOTAL | 124 | | 4014.87 |
| 16 SEPT 93 | | | |
| pennales | 36 | 12363.75 | 852.67 |
| *Gleocystis* | 6 | 2060.63 | 142.11 |
| *Lyngbya/Oscillatoria* | 5 | 1717.19 | 118.43 |
| *Cryptomonas* | 21 | 7212.19 | 497.39 |
| *Microcystis* | 3 | 1030.31 | 71.06 |
| *Spirulina* | 15 | 5151.56 | 355.28 |
| *Ankistrodesmus* | 22 | 7555.63 | 521.08 |
| *Aphanocapsa* | 1 | 343.44 | 23.69 |
| centrales | 3 | 1030.31 | 71.06 |
| *Merismopedia* | 3 | 1030.31 | 71.06 |
| *Actinastrum* | 2 | 686.88 | 47.37 |
| *Scenedesmus* | 4 | 1373.75 | 94.74 |
| *Staurastrum* | 1 | 343.44 | 23.69 |
| *Chlamydomonas* | 7 | 2404.06 | 165.80 |
| TOTAL | 129 | | 3055.41 |
| 30 SEPT 93 | | | |
| pennales | 22 | 8338.94 | 297.82 |
| *Ulothrix* | 10 | 3790.43 | 135.37 |
| *Microcystis* | 15 | 5685.64 | 203.06 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Chlamydomonas* | 2 | 758.09 | 27.07 |
| *Pediastrum* | 4 | 1516.17 | 54.15 |
| *Gonium* | 9 | 3411.39 | 121.84 |
| centrales | 4 | 1516.17 | 54.15 |
| *Spirulina* | 2 | 758.09 | 27.07 |
| *Cryptomonas* | 21 | 7959.90 | 284.28 |
| *Scenedesmus* | 3 | 1137.13 | 40.61 |
| *Aphanocapsa* | 8 | 3032.34 | 108.30 |
| *Cosmarium* | 1 | 379.04 | 13.54 |
| TOTAL | 101 | | 1367.26 |
| 22 OCT 93 | | | |
| pennales | 32 | 8316.76 | 803.55 |
| *Chlamydomonas* | 10 | 2598.99 | 251.11 |
| *Cryptomonas* | 16 | 4158.38 | 401.78 |
| *Gleocystis* | 17 | 4418.28 | 426.89 |
| *Tetraedron* | 1 | 259.90 | 25.11 |
| *Ulothrix* | 5 | 1299.49 | 125.55 |
| *Scenedesmus* | 1 | 259.90 | 25.11 |
| *Aphanocapsa* | 8 | 2079.19 | 200.89 |
| centrales | 7 | 1819.29 | 175.78 |
| *Mallomonas* | 1 | 259.90 | 25.11 |
| *Microcystis* | 5 | 1299.49 | 125.55 |
| *Lyngbya/Oscillatoria* | 3 | 779.70 | 75.33 |
| *Spirulina* | 6 | 1559.39 | 150.67 |
| TOTAL | 112 | | 2812.43 |
| STATION 7 | | | |
| 4 JUN 92 | | | |
| *Ceratium* | | 127 | 24.13 |
| *Cryptomonas* | | 2521 | 478.99 |
| *Peridinium* | | 53 | 10.07 |
| *Coelastrum* | | 99 | 18.81 |
| *Scenedesmus* | | 165 | 31.35 |
| centrales | | 101 | 19.19 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| pennales | | 6696 | 1272.24 |
| *Staurastrum* | | 15 | 2.85 |
| *Pediastrum* | | 6 | 1.14 |
| *Closterium* | | 4 | 0.76 |
| *Melosira* | | 32 | 6.08 |
| TOTAL | | 9819 | 1865.61 |
| 2 JUL 92 | | | |
| *Cryptomonas* | | 1181 | 448.78 |
| pennales | | 2954 | 1122.52 |
| centrales | | 146 | 55.48 |
| *Scenedesmus* | | 576 | 218.88 |
| *Melosira* | | 20 | 7.6 |
| *Oedogonium* | | 23 | 8.74 |
| *Spirulina* | | 160 | 60.8 |
| *Ceratium* | | 13 | 4.94 |
| *Microcystis* | | 225 | 85.5 |
| *Peridinium* | | 15 | 5.7 |
| *Anabaena* | | 16 | 6.08 |
| TOTAL | | 5329 | 2025.02 |
| 19 AUG 92 | | | |
| *Peridinium* | 3 | 335.45 | 26.41 |
| *Mallomonas* | 2 | 223.63 | 17.61 |
| *Lyngbya/Oscillatoria* | 287 | 32091.44 | 2526.88 |
| *Spirulina* | 428 | 47857.62 | 3768.32 |
| centrales | 9 | 1006.35 | 79.24 |
| *Cryptomonas* | 3 | 335.45 | 26.41 |
| *Aphanocapsa* | 37 | 4137.22 | 325.77 |
| *Gleocystis* | 32 | 3578.14 | 281.74 |
| *Scenedesmus* | 18 | 2012.70 | 158.48 |
| pennales | 77 | 8609.90 | 677.94 |
| *Coelastrum* | 3 | 335.45 | 26.41 |
| *Pediastrum* | 1 | 111.82 | 8.80 |
| *Merismopedia* | 12 | 1341.80 | 105.65 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Crucigenia* | 2 | 223.63 | 17.61 |
| *Staurastrum* | 6 | 670.90 | 52.83 |
| TOTAL | 920 | | 8100.12 |
| 12 SEP 92 | | | |
| *Peridinium* | 2 | 40.00 | 4.21 |
| *Mallomonas* | 1 | 20.00 | 2.11 |
| *Lyngbya/Oscillatoria* | 115 | 2300.00 | 242.11 |
| *Spirulina* | 317 | 6340.00 | 667.37 |
| *centrales* | 8 | 160.00 | 16.84 |
| *Cryptomonas* | 6 | 120.00 | 12.63 |
| *Aphanocapsa* | 14 | 280.00 | 29.47 |
| *Gleocystis* | 10 | 200.00 | 21.05 |
| *Scenedesmus* | 14 | 280.00 | 29.47 |
| pennales | 168 | 3360.00 | 353.68 |
| *Coelastrum* | 3 | 60.00 | 6.32 |
| *Pediastrum* | 1 | 20.00 | 2.11 |
| *Merismopedia* | 76 | 1520.00 | 160.00 |
| *Crucigenia* | 2 | 40.00 | 4.21 |
| *Staurastrum* | 3 | 60.00 | 6.32 |
| *Chlamydomonas* | 6 | 120.00 | 12.63 |
| *Ceratium* | 1 | 20.00 | 2.11 |
| *Microcystis* | 20 | 400.00 | 42.11 |
| *Oedogonium* | 1 | 20.00 | 2.11 |
| TOTAL | 768 | | 1616.84 |
| 24 OCT 92 | | | |
| *Mallomonas* | 2 | 40.00 | 0.42 |
| *Lyngbya/Oscillatoria* | 2 | 40.00 | 4.21 |
| *Spirulina* | 4 | 80.00 | 8.42 |
| centrales | 8 | 160.00 | 16.84 |
| *Cryptomonas* | 14 | 280.00 | 29.47 |
| *Gleocystis* | 13 | 260.00 | 27.37 |
| *Scenedesmus* | 7 | 140.00 | 14.74 |
| pennales | 52 | 1040.00 | 109.47 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Coelastrum* | 2 | 40.00 | 4.21 |
| *Pediastrum* | 5 | 100.00 | 10.53 |
| *Melosira* | 44 | | 83.60 |
| *Crucigenia* | 2 | 40.00 | 4.21 |
| *Actinastrum* | 3 | 60.00 | 6.32 |
| *Chlamydomonas* | 7 | 140.00 | 14.74 |
| *Oedogonium* | 27 | 540.00 | 56.84 |
| TOTAL | 192 | | 391.39 |
| 8 MAR 93 | | | |
| *Cryptomonas* | 2 | 20 | 3.08 |
| *Oedogonium* | 25 | 250 | 38.46 |
| centrales | 50 | | 95.00 |
| pennales | 13 | 130 | 20.00 |
| *Peridinium* | 2 | 20 | 3.08 |
| *Gleocystis* | 2 | 20 | 3.08 |
| *Melosira* | 3 | | 5.70 |
| *Microcystis* | 1 | 10 | 1.54 |
| *Platydorina* | 1 | 10 | 1.54 |
| TOTAL | 99 | | 171.47 |
| 18 APR 93 | | | |
| *Oedogonium* | 27 | 4636.41 | 482.96 |
| centrales | 650 | | 1235.00 |
| *Chlorella* | 46 | 7899.06 | 822.82 |
| pennales | 6 | 1030.31 | 107.32 |
| *Cryptomonas* | 9 | 1545.47 | 160.99 |
| *Scenedesmus* | 2 | 343.44 | 35.77 |
| *Melosira* | 8 | 1373.75 | 143.10 |
| TOTAL | 748 | | 2987.96 |
| 26 MAY 93 | | | |
| centrales | 7 | 747.93 | 82.19 |
| *Chlorella* | 40 | 4273.89 | 469.66 |
| *Coelastrum* | 5 | 534.24 | 58.71 |
| pennales | 26 | 2778.03 | 305.28 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Cryptomonas* | 21 | 2243.79 | 246.57 |
| *Sphaerocystis* | 3 | 320.54 | 35.22 |
| *Gleocystis* | 2 | 213.69 | 23.48 |
| *Chlamydomonas* | 6 | 641.08 | 70.45 |
| centrales | 190 | | 361.00 |
| *Platydorina* | 1 | 106.85 | 11.74 |
| *Cosmarium* | 1 | 106.85 | 11.74 |
| TOTAL | 302 | | 1676.04 |
| 25 JUN 93 | | | |
| *Pediastrum* | 5 | 1265.30 | 105.44 |
| *Sphaerocystis* | 4 | 1012.24 | 84.35 |
| *Mallomonas* | 2 | 506.12 | 42.18 |
| *Scenedesmus* | 2 | 506.12 | 42.18 |
| *Cryptomonas* | 16 | 4048.95 | 337.41 |
| pennales | 11 | 2783.65 | 231.97 |
| *Anabaena* | 26 | 6579.54 | 548.29 |
| *Cosmarium* | 1 | 253.06 | 21.09 |
| *Gleocystis* | 14 | 3542.83 | 295.24 |
| *Lyngbya/Oscillatoria* | 41 | 10375.43 | 864.62 |
| *Peridinium* | 2 | 506.12 | 42.18 |
| *Oocystis* | 10 | 2530.59 | 210.88 |
| *Ceratium* | 1 | 253.06 | 21.09 |
| TOTAL | 135 | | 2846.92 |
| 23 JUL 93 | | | |
| pennales | 91 | 31252.81 | 3156.85 |
| *Melosira* | 4 | 1373.75 | 138.76 |
| *Staurastrum* | 2 | 686.88 | 69.38 |
| *Cryptomonas* | 4 | 1373.75 | 138.76 |
| *Gleocystis* | 12 | 4121.25 | 416.29 |
| *Kirchnerella* | 2 | 686.88 | 69.38 |
| *Tetraedron* | 3 | 1030.31 | 104.07 |
| *Mallomonas* | 1 | 343.44 | 34.69 |
| *Scenedesmus* | 3 | 1030.31 | 104.07 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Pandorina* | 1 | 343.44 | 34.69 |
| *Oocystis* | 1 | 343.44 | 34.69 |
| *Spirulina* | 1 | 343.44 | 34.69 |
| *Merismopedia* | 1 | 343.44 | 34.69 |
| centrales | 2 | 686.88 | 69.38 |
| *Aphanocapsa* | 1 | 343.44 | 34.69 |
| TOTAL | 129 | | 4475.09 |
| 4 AUG 93 | | | |
| *Actinastrum* | 2 | 369.86 | 123.29 |
| *Spirulina* | 7 | 1294.50 | 431.50 |
| *Ankistrodesmus* | 6 | 1109.57 | 369.86 |
| *Gleocystis* | 12 | 2219.13 | 739.71 |
| *Cryptomonas* | 10 | 1849.28 | 616.43 |
| *Staurastrum* | 1 | 184.93 | 61.64 |
| *Chlamydomonas* | 3 | 554.78 | 184.93 |
| *Anabaena* | 1 | 184.93 | 61.64 |
| *Scenedesmus* | 4 | 739.71 | 246.57 |
| TOTAL | 125 | | 7705.33 |
| 19 AUG 93 | | | |
| *Cryptomonas* | 7 | 3365.69 | 323.62 |
| *Chlamydomonas* | 11 | 5288.94 | 508.55 |
| *Ankistrodesmus* | 21 | 10097.06 | 970.87 |
| *Spirulina* | 3 | 1442.44 | 138.70 |
| pennales | 51 | 24521.44 | 2357.83 |
| *Scenedesmus* | 4 | 1923.25 | 184.93 |
| *Gleocystis* | 10 | 4808.13 | 462.32 |
| *Lyngbya/Oscillatoria* | 1 | 480.81 | 46.23 |
| *Aphanocapsa* | 6 | 2884.88 | 277.39 |
| *Merismopedia* | 3 | 1442.44 | 138.70 |
| *Staurastrum* | 6 | 2884.88 | 277.39 |
| *Tetraedron* | 3 | 1442.44 | 138.70 |
| *Ulothrix* | 1 | 480.81 | 46.23 |
| *Euglena* | 1 | 480.81 | 46.23 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Mallomonas* | 1 | 480.81 | 46.23 |
| TOTAL | 129 | | 5963.92 |
| 2 SEPT 93 | | | |
| pennales | 146 | 96845.45 | 3889.38 |
| *Gonium* | 3 | 1989.98 | 79.92 |
| *Scenedesmus* | 4 | 2653.30 | 106.56 |
| *Spirulina* | 8 | 5306.60 | 213.12 |
| centrales | 4 | 2653.30 | 106.56 |
| *Gleocystis* | 4 | 2653.30 | 106.56 |
| *Microcystis* | 2 | 1326.65 | 53.28 |
| *Peridinium* | 5 | 50 | 2.01 |
| *Staurastrum* | 9 | 90 | 3.61 |
| *Cosmarium* | 2 | 20 | 0.80 |
| *Chlamydomonas* | 3 | 30 | 1.20 |
| *Anabaena* | 3 | 30 | 1.20 |
| *Pediastrum* | 4 | 40 | 1.61 |
| TOTAL | 197 | | 4565.81 |
| 16 SEPT 93 | | | |
| centrales | 8.00 | 5128.67 | 197.26 |
| *Merismopedia* | 8.33 | 5342.36 | 205.48 |
| *Eutetramonas* | 1.00 | 641.08 | 24.66 |
| *Staurastrum* | 1.33 | 854.78 | 32.88 |
| pennales | 35.00 | 22437.92 | 863.00 |
| *Kirchnerella* | 6.67 | 4273.89 | 164.38 |
| *Scenedesmus* | 4.67 | 2991.72 | 115.07 |
| *Spirulina* | 11.67 | 7479.31 | 287.67 |
| *Ankistrodesmus* | 14.67 | 9402.56 | 361.64 |
| *Cryptomonas* | 9.67 | 6197.14 | 238.35 |
| *Aphanocapsa* | 4.33 | 2778.03 | 106.85 |
| *Gleocystis* | 9.67 | 6197.14 | 238.35 |
| *Lyngbya/Oscillatoria* | 10.33 | 6624.53 | 254.79 |
| *Tetraedron* | 0.31 | 195.89 | 7.53 |
| *Ulothrix* | 0.33 | 213.69 | 8.22 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Chlamydomonas* | 0.67 | 427.39 | 16.44 |
| *Pediastrum* | 0.33 | 213.69 | 8.22 |
| *Cosmarium* | 1.00 | 641.08 | 24.66 |
| TOTAL | 127.97 | | 3155.42 |
| 30 SEPT 93 | | | |
| pennales | 47 | 12470.51 | 395.89 |
| centrales | 7 | 1857.31 | 58.96 |
| *Cryptomonas* | 10 | 2653.30 | 84.23 |
| *Merismopedia* | 6 | 1591.98 | 50.54 |
| *Gleocystis* | 3 | 795.99 | 25.27 |
| *Microcystis* | 4 | 1061.32 | 33.69 |
| *Spirulina* | 5 | 1326.65 | 42.12 |
| *Scenedesmus* | 9 | 2387.97 | 75.81 |
| *Pediastrum* | 2 | 530.66 | 16.85 |
| *Staurastrum* | 3 | 795.99 | 25.27 |
| *Aphanocapsa* | 5 | 1326.65 | 42.12 |
| *Gonium* | 18 | 4775.94 | 151.62 |
| TOTAL | 119 | | 1002.36 |
| 22 OCT 93 | | | |
| *Pandorina* | 6.33 | 1646.02 | 82.30 |
| *Melosira* | 16.33 | 4245.01 | 212.25 |
| *Pediastrum* | 4.33 | 1126.23 | 56.31 |
| *Ulothrix* | 5.67 | 1472.76 | 73.64 |
| *Scenedesmus* | 10.67 | 2772.25 | 138.61 |
| *Cryptomonas* | 16.33 | 4245.01 | 212.25 |
| *Chlamydomonas* | 16.33 | 4245.01 | 212.25 |
| *Rhizoclonium* | 4.33 | 1126.23 | 56.31 |
| pennales | 71.67 | 18626.07 | 931.30 |
| centrales | 16.67 | 4331.64 | 216.58 |
| *Aphanocapsa* | 10.00 | 2598.99 | 129.95 |
| *Microcystis* | 4.33 | 1126.23 | 56.31 |
| *Lyngbya/Oscillatoria* | 12.00 | 3118.78 | 155.94 |
| *Staurastrum* | 3.00 | 779.70 | 38.98 |

| Continued | Grid Count | Total Count | #/ml |
|---|---|---|---|
| *Sphaerocystis* | 0.67 | 173.27 | 8.66 |
| *Crucigenia* | 0.33 | 86.63 | 4.33 |
| TOTAL | 199 | | 2585.99 |