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ARCADIA LAKE WATER-QUALITY EVALUATION

by

R. W. Hall, Jr., R. H. Plumb, Jr., K. W. Thornton
R. L. Eley, A. S. Lessem, D. L. Robey
B. Loftis, P. E. Saunders

Environmental Effects Laboratory
U. S. Army Engineer Waterways Experiment Station
P. O. Box 631, Vicksburg, Miss. 39180

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Final Report

Approved For Public Release; Distribution Unlimited



Prepared for U. S. Army Engineer District, Tulsa
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The water quality of the proposed Arcadia Lake, Oklahoma, was evaluated relative to water-quality criteria and standards appropriate for the project purposes. Study procedures included evaluation of existing data and previous studies of streams and lakes in the project area; collection and evaluation of additional field data; determination of available and limiting nutrients through algal bioassays; establishment of relationships between stream discharges and loadings of nutrients, metals, and pesticides based on stream concentrations and land-use patterns; application of various mathematical models; and comparison of predicted or measured results with existing or proposed water-quality criteria.

A comparison of average values for 70 water-quality parameters with the most stringent standard or criterion revealed that ammonia, manganese, mercury, DDT,

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dieldrin, aldrin, chlordane, lindane, heptachlor, PCB, phenols, and fecal coliforms equaled or exceeded permissible or recommended levels at least part of the time. Only coliform bacteria, ammonia, and manganese exceeded present Oklahoma standards applicable to proposed reservoir uses. Ammonia would not be expected to reach toxic concentrations in the hypolimnion of Arcadia Lake or interfere with project purposes.

Nutrient evaluations based on concentrations and loading indicated that the proposed impoundment would be eutrophic and that algal blooms were likely to occur during the late spring and summer months. Algal bioassays and ecological model simulations indicated that not all of the available nutrients would be used because of light limitation. Algal blooms are expected to minimally interfere with recreational purposes of the lake because surrounding nutrient-rich Oklahoma reservoirs presently receive heavy recreational use and alternative, less eutrophic recreational reservoirs do not exist in the project area. It would not be feasible through watershed management practices or reservoir operational strategies to reduce in-lake nutrient concentrations sufficiently to limit algal growth. Routine chemical treatments are not feasible nor compatible with all project purposes. A capability for selective withdrawal of municipal and industrial water supply releases would be of benefit in minimizing treatment costs.

Manganese concentrations and occasionally iron concentrations are expected to exceed drinking water standards in the hypolimnion and headwaters of the proposed impoundment. Iron and manganese would be less likely to exceed standards in the epilimnion near the dam. Excessive iron and manganese would not be a problem in finished water supplies if the potential problem is recognized in the design of the water treatment plant.

Average mercury concentration computed over all samples collected near Arcadia was ten times less than the public water supply criterion but exceeded the criterion for the protection of freshwater aquatic life. Results of analyses of the mercury content of fish collected in the Deep Fork River indicated body-burden concentrations less than Food and Drug Administration (FDA) and Environmental Protection Agency (EPA) limits.

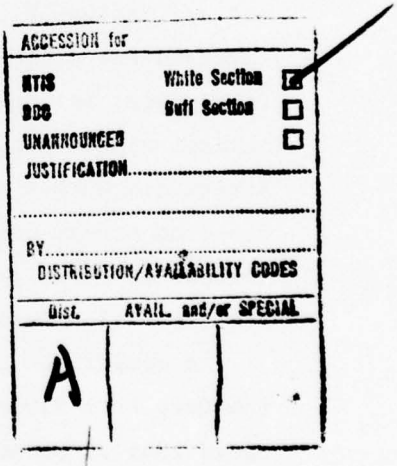
Pesticides would not be expected to be a water-quality problem in Arcadia Lake because sorption and precipitation would reduce concentrations significantly and restricted use of some of the pesticides has been implemented or is proposed by the EPA. Heptachlor epoxide concentrations in fish exceeded FDA administrative guidelines. However, the rare occurrence of heptachlor, the failure to detect heptachlor epoxide in Deep Fork River water samples, the failure to detect heptachlor or heptachlor epoxide in river sediments, and the fact that the EPA has banned the distribution and use of the pesticide suggest that heptachlor and its degradation products will not exceed criteria in the proposed impoundment. Detectable phenol concentrations would be expected to occur only in the headwaters because of rapid decay and dilution.

Coliform bacterial contamination would be limited to the headwaters of the proposed impoundment during base flow. Coliform bacteria might occasionally exceed standards in the lower portions of the pool, following major storm events in the watershed. But even then dilution probably would prevent concentrations from exceeding standards.

The impoundment would be expected to exhibit weak thermal stratification during the late spring and summer months. Wind-mixing would determine the degree of stratification. Downstream temperature objectives could be met by project releases if selective withdrawal were practiced. It is expected that dissolved oxygen of project releases will be approximately 80 to 90 percent saturation due to reaeration as flows pass through the outlet works and stilling basin.

If Arcadia Lake is constructed, water-quality data collection should continue through pre- and postimpoundment in order to provide a basis for lake management to meet intended project purposes.

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EXECUTIVE SUMMARY

The proposed Arcadia Lake would be formed by impoundment of the Deep Fork River in central Oklahoma east of Oklahoma City. Authorized project purposes are flood control, municipal and industrial water supply, and recreation. In the past, the Deep Fork River received effluents from sewage treatment plants, lagoons, industry, several small businesses, and nonpoint source contaminants from urban land. Oklahoma State University (OSU) evaluated the water quality in 1973 and reported that 10 constituents often exceeded surface water criteria; coliform counts exceeded recommended standards; and nutrient concentrations might stimulate algal blooms. After publication of the OSU report, various point sources of contaminants were identified and diverted into sanitary sewer systems, and other diversions are planned. The present study was conducted to investigate in detail possible water-quality problems that might interfere with project purposes in view of recent and proposed changes in the watershed.

Most procedures that have been used previously to evaluate the water quality of proposed impoundments possess serious deficiencies. To provide more reliable predictions, a number of approaches were used in this study to address the same basic questions. These procedures included (a) evaluation of existing data and previous studies of streams and lakes in the project area; (b) collection and evaluation of additional field data; (c) determination of available and limiting nutrients through algal bioassays; (d) establishment of relationships between stream discharges and loadings of nutrients, metals, and pesticides based on stream concentrations and land-use patterns; (e) application of various mathematical models; and (f) comparison of predicted or measured results with existing or proposed water-quality criteria.

A comparison of average values for 70 water-quality parameters in the Deep Fork River with the most stringent standard or criterion indicated that 12 parameters in the Deep Fork River equaled or exceeded permissible or recommended levels at least part of the time. These parameters were ammonia, manganese, mercury, DDT, aldrin, dieldrin,

chlordane, heptachlor, lindane, PCB, phenols, and fecal coliforms. Only three parameters in this list, coliform bacteria, ammonia, and manganese, exceeded present Oklahoma standards applicable to the use classification of the Deep Fork River or to proposed reservoir uses. Ammonia would not be expected to reach toxic concentrations in the epilimnion of Arcadia Lake or to interfere with project purposes.

Nutrient evaluations, using several approaches, indicated that the proposed impoundment would exceed eutrophication criteria based on concentrations and loadings and that algal blooms would occur during the late spring and summer months. However, both algal bioassays and ecological model simulations indicated that not all of the available nutrients would be used because of light limitation. Turbidity, and therefore light availability, in Arcadia Lake would not be expected to be different from conditions in several existing Oklahoma reservoirs. Therefore, algal blooms in Arcadia Lake would not be expected to be significantly different in frequency or intensity than those occurring in surrounding eutrophic waters. Future land-use changes would not be expected to affect lake eutrophication since light rather than nutrients would be the limiting factor. Based on a limited evaluation, it does not appear feasible to reduce the eutrophication potential of Arcadia Lake significantly by either watershed or existing in-lake management techniques.

Iron and manganese concentrations in the hypolimnion of Arcadia Lake probably would exceed public water supply standards during summer stratification. These constituents also would be expected to exceed standards occasionally in the headwaters of the impoundment. Toxic conditions are not expected. Iron and manganese would be less likely to exceed standards in the epilimnion near the dam. The selective withdrawal capability of the proposed project would enable project operation to minimize iron and manganese concentrations in water supply withdrawals and downstream releases. Furthermore, iron and manganese would not be a problem in finished water supplies if the potential problem were recognized in the design of the water treatment plant.

Mercury concentrations in the Deep Fork River were less than public

water supply standards but occasionally exceeded Environmental Protection Agency (EPA) proposed aquatic life criteria. Analysis of fish collected near the proposed damsite indicated that mercury in fish tissue was less than Food and Drug Administration (FDA) limits for fish flesh mercury content and the EPA's proposed criterion for the protection of freshwater aquatic life.

Earlier studies of the Deep Fork River concluded that lead would be a water-quality problem in the proposed impoundment. This conclusion was based on a technically invalid comparison of total lead concentrations with standards derived from studies using dissolved lead. A comparison using dissolved lead concentrations indicates that soluble lead is 10 times lower than the standard. It is concluded that lead would not be a water-quality problem in the proposed impoundment at Arcadia. Analysis of lead content in fish flesh substantiates this conclusion.

Comparisons of trace metal concentrations in surrounding impoundments indicated that the impoundments had similar or lower concentrations than their tributary waters. This suggests that accumulation of metals would not occur in the water column of Arcadia Lake.

Pesticides would not be anticipated to be a water-quality problem in Arcadia Lake for the following reasons: (a) sorption and precipitation would be expected to reduce concentrations significantly; (b) surrounding reservoirs do not show buildup in the water column, providing some support to the hypothesis that significant concentrations would not occur in Arcadia Lake; and (c) restricted use of some of the pesticides has been implemented or is proposed by the EPA. Heptachlor epoxide concentrations in fish exceeded FDA administrative guidelines. However, the rare occurrence of heptachlor, the failure to detect heptachlor epoxide in Deep Fork River water samples, the failure to detect heptachlor or heptachlor epoxide in river sediments, and the fact that the EPA has banned the distribution and use of the pesticide suggest that heptachlor and its degradation products will not exceed criteria in the proposed impoundment.

Because heptachlor epoxide or other chemical compounds with a similar retention time were measured in significant concentrations in fish

from the Deep Fork River, it is recommended that additional water, sediment, and fish samples be analyzed. If heptachlor or heptachlor epoxide are again detected in significant concentrations, confirmatory analysis by mass spectrometry should be conducted. It is paradoxical that fish would have high body burdens of a contaminant that is undetectable in water and sediment analyses, even if a significant potential for biomagnification existed. Additional data should assist in understanding the situation, but in any case heptachlor or heptachlor epoxide are not expected to be water-quality problems in the proposed Arcadia Lake.

Phenol concentrations decay very rapidly in natural waters and would not be expected to be a problem in the impoundment. Concentrations of coliform bacteria in the headwaters of the impoundment would exceed standards for primary body contact recreation. Concentrations of coliform bacteria near the dam would rarely exceed standards. Coliform bacteria decay rapidly in natural waters and bacterial contamination would be restricted to the headwaters except following major storms. Even then, dilution probably would result in concentrations below standards. Also, body contact recreational use of the reservoir generally would be expected to be low immediately following major storms. However, it is recommended that recreational areas with swimming beaches be located in the lower one-half of the impoundment and that coliform bacteria concentrations in these areas be monitored on a regular basis.

The impoundment would be expected to exhibit weak thermal stratification during the late spring and summer months. Wind-mixing probably would determine the degree of stratification. Downstream temperature and dissolved oxygen objectives could be met by project releases if selective withdrawal were practiced.

Water-quality standards probably will be frequently exceeded in Arcadia Lake by manganese and total dissolved solids and occasionally by iron. Standards for these three constituents are based upon aesthetic rather than toxicological effects. Therefore, the possible water-quality problems associated with Arcadia Lake are predicted to be based primarily on aesthetic considerations related to water supply uses,

including possible staining, taste, and odor problems. Algal blooms would occur during various periods of the year. These problems probably would not interfere with recreational purposes of the lake to any greater extent than occurs in other nutrient-rich Oklahoma reservoirs presently receiving heavy recreational use. During periods of thermal stratification and algal blooms, water supply treatment costs could be increased. A capability for selective withdrawal of municipal and industrial water supply releases would be of significant benefit in minimizing treatment problems.

The conclusions of this study are that the water quality to be expected following impoundment would be suitable for water supply purposes. Toxic conditions would not be expected to occur. However, taste, odor, and staining problems probably would occur occasionally and should be considered in water supply treatment plans. During most of the year, bacteriological contamination from inflowing waters would limit primary contact recreation to approximately the lower two-thirds of the reservoir. However, following major summer storms, bacteriological concentrations might occasionally exceed standards near the damsite. If the project is constructed, water quality of the impoundment is not expected to prohibit the meeting of authorized project purposes.

If Arcadia Lake is constructed, water-quality data collection should continue through pre- and postimpoundment to provide a basis for lake management to meet intended project purposes.

PREFACE

The work described in this report was performed by the U. S. Army Engineer Waterways Experiment Station (WES), Vicksburg, Mississippi, for the U. S. Army Engineer District, Tulsa. The project was authorized by Intra-Army Order for Reimbursable Services No. DOL 740015 dated 2 August 1974 and amended 11 February 1975.

This report is an evaluation of the water quality expected in the proposed Arcadia Lake relative to water-quality criteria and standards appropriate for the project purposes.

In the preliminary draft reports, existing and expected water quality were compared with the Environmental Protection Agency's (EPA's) 1973 Proposed Criteria for Water Quality. During September 1976, the EPA published a revision of the 1973 criteria entitled "Quality Criteria for Water." The revised 1976 criteria are referenced in this report.

The project was undertaken as a joint effort by the Environmental Effects (EEL) and Hydraulics (HL) Laboratories at the WES. The research was conducted under the direct supervision of Dr. Rex L. Eley, Chief, Ecosystem Research and Simulation Division, and the general supervision of Dr. John Harrison, Chief, EEL. Mr. Ross W. Hall, Jr., served as principal investigator. Drs. Russel H. Plumb, Jr., Kent W. Thornton, and Allan S. Lessem participated in the study and assisted in the preparation of this report. Mr. Donald L. Robey, Chief, Ecosystem Modeling Branch, assisted in the review and revision of the draft report.

Messrs. Bruce Loftis and Peter E. Saunders conducted supporting studies and prepared Part V of the report under the supervision of Mr. Joseph P. Bohan, Chief, Spillways and Channels Branch, Mr. John L. Grace, Chief, Structures Division, and Mr. Henry B. Simmons, Chief, HL.

Messrs. James Irwin, Vernon Sours, and Willard Mills and Ms. Jo Ann Kirkland of the U. S. Geological Survey; Mr. Joe Hall of the U. S. Bureau of Reclamation; Mr. Mark Coleman of the Oklahoma State Department of Health; Messrs. Forest Nelson, Zack Williams, and Bill Potter of the Oklahoma Water Resources Board; Mr. R. L. Peterson of the Oklahoma Department of Pollution Control; Dr. S. L. Burks of Oklahoma State

University; Dr. Larry Canter of the University of Oklahoma; Mr. Preston Carter of the Oklahoma City-County Health Department; Mr. Tom Carpenter of the Water Pollution Control Division of the Oklahoma City Department of Public Works; and Mr. Jimmie Pigg of Moore High School, Moore, Oklahoma, provided assistance in data compilation. Drs. Ed Klehr and John Fletcher of the University of Oklahoma provided facilities for conducting bioassay analyses. Dr. Jack A. Stanford of North Texas State University supervised the analyses of pesticides, mercury, and lead in fish. Mr. Robert M. Jenkins, Director, National Reservoir Research Center, U. S. Fish and Wildlife Service, provided an estimate of the quality and quantity of the lake fishery (Appendix C).

Appendix D of the report was written by Dr. Jack A. Stanford and Ms. Sandra A. Zimmerman, Department of Biological Sciences, North Texas State University, Denton, Texas.

The project was monitored by Mr. Ed Lindsey, Tulsa District, under the supervision of Mr. J. L. Mathews, Chief, Red River Planning Section.

Directors of WES during the preparation and publication of this report were COL G. H. Hilt, CE, and COL J. L. Cannon, CE. Technical Director was Mr. F. R. Brown.

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CONVERSION FACTORS, U. S. CUSTOMARY TO METRIC (SI)
UNITS OF MEASUREMENT

U. S. customary units of measurement used in this report can be converted to metric (SI) units as follows:

Multiply	By	To Obtain
feet	0.3048	metres
miles (U. S. statute)	1.609344	kilometres
acres	4046.856	square metres
acre-feet	1233.482	cubic metres
pounds (mass)	0.4535924	kilograms

ARCADIA LAKE WATER-QUALITY EVALUATION

PART I: INTRODUCTION

The Problem

1. The proposed Arcadia Lake would be formed by impoundment of the Deep Fork River in central Oklahoma east of Oklahoma City. The river originates in northwest Oklahoma City, flows through the metropolitan area for a distance of 9 km, and then flows 17 km through predominantly pastures and woodlands to the proposed damsite. The Arcadia Lake watershed covers an area of 273 km^2 of which 39 percent is open, 33 percent is urbanized, and the remaining 28 percent is woodland, pasture, or cultivated.[1] At conservation pool elevation, the proposed impoundment would have a surface area of 1280 ha and a volume of 8.55×10^7 m^3. Average depth would be 6.7 m and maximum depth at the structure would be 22 m.

2. During much of the time during which Deep Fork River chemical data were collected, the effluents of up to six sewage treatment plants and lagoons contributed over 64,780 m^3/day or 28 percent of the average discharge measured near the proposed damsite. For many extended periods, domestic sewage plant effluents constituted base flow. During the same period, industry and many small businesses were discharging contaminants directly into the river or indirectly through the storm sewer system. The influx of nonpoint contaminants from the urban land and the direct discharge of municipal and industrial wastes have degraded the water quality in the upper Deep Fork River for most beneficial uses.

3. The Arcadia Lake project would consist of a multiple-purpose lake for flood control, municipal and industrial water supply, and recreation. This type of project would satisfy the desires of local interests who have expressed a primary need for water supply and water-oriented recreation.

4. Oklahoma State University (OSU) was contracted by the Tulsa District (TD) in 1972 to evaluate the existing water quality of the Deep Fork River and to predict expected water quality in Arcadia Lake. Not

surprisingly, the study concluded that ammonia, nitrate, chloride, iron, manganese, lead, aldrin, lindane, DDT, and endrin concentrations often exceeded surface water criteria for public water supplies; that coliform counts exceeded recommended standards for primary contact recreational use; and that existing concentrations of nitrogen and phosphorus might stimulate blooms of nuisance algae.[2,3] Primarily on the basis of the OSU report, the Environmental Protection Agency (EPA) in a letter to TD, dated 10 October 1973, recommended that public and industrial water supply and recreation not be included as project purposes.* More recent investigations in the Arcadia Lake watershed reported excessive lead concentrations and stated that lead would be expected to be a problem in Arcadia Lake.[4]

5. The OSU appraisal of the possible water quality of the proposed Arcadia Lake was inconclusive. While the data compilation was useful, possible technical errors in data interpretation created questions of the tenability of the study's results. Furthermore, the scope of the OSU study was not sufficient for adequate evaluation of the uncertainties associated with the predictions.

6. Since publication of the OSU report, one sewage treatment plant was relocated outside the Arcadia watershed; a second plant relocation was planned; and two sewage lagoon operations were discontinued. In addition, major industrial sources of contaminants were identified, and their effluents diverted through sanitary sewer systems.

Study Objectives

7. The purpose of the Waterways Experiment Station (WES) study was a more thorough investigation of possible water-quality problems identified in earlier studies. The study was to define the possible problems

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* Letter from Mr. Charles H. Hembree, Acting Chief, Federal Assistance Branch, Region VI, U. S. Environmental Protection Agency, Dallas, Texas, to U. S. Army Engineer District, Tulsa, Attention: Mr. M. W. DeGeer, Chief, Engineering Division; subject: Review of Report, "Water Quality in the Upper Deep Fork River Basin and Arcadia Lake."

more clearly in view of recent and proposed changes in the watershed and to evaluate these problems using several available methodologies.

8. Initial discussions with TD and subsequent meetings with the Corps of Engineers Southwestern Division office (SWD), EPA, and the Oklahoma State Health Department resulted in the formulation of the following study objectives:

a. Compile existing data.

b. Evaluate previous studies.

c. Establish a limited data-collection program in coordination with TD to obtain some additional data deemed critical for project evaluation.

d. Calculate loadings and estimate inflows of nutrients, metals, and pesticides using existing information.

e. Use the WESTEX model to predict lake temperatures and dissolved oxygen profiles.

f. Make comparative water-quality predictions based on water quality of existing lakes.

g. Simulate seasonal water quality with a mathematical ecological model.

h. Evaluate the effects of projected urbanization on the water quality of the Deep Fork River and Arcadia Lake using existing information.

i. Extrapolate existing water-quality data over all years of synthetic flow data at Arcadia gage and calculate water-quality constituent concentration-frequency curves.

j. Evaluate effects that future changes in gasoline lead concentrations may have on the water quality of Arcadia Lake.

9. Agreements on research procedures and assumptions that emerged during discussions with TD, SWD, EPA, and Oklahoma State Health Department are as follows:

a. Use the Association of Central Oklahoma Government's (ACOG) projections of land-use changes to evaluate the effects of urbanization on the water quality of Arcadia Lake.

b. EPA personnel stated that it should be assumed for purposes of the WES study that Public Law (PL) 92-500 will be implemented and that 1977, 1983, and 1985 deadlines for water-quality improvements will be met.

c. Based on the results of meetings with representatives of EPA and the Oklahoma State Health Department, it was concluded that these agencies would accept final conclusions and

recommendations resulting from a water-quality evaluation made by the WES using the proposed research approaches as reliable.

General Study Approach

10. A variety of procedures were used in this study to address the same question in order to provide more reliable predictions. If the various procedures provided equivalent predictions, then conclusions could be postulated with increased confidence.

11. The following approaches were used to achieve the study objectives.

Collection of existing data

12. Government agencies and educational institutions that have conducted water-quality studies within the Arcadia Lake watershed or nearby existing impoundments and their watersheds were contacted in order to collect existing water-quality data and other information that might assist in data interpretation or provide insights during the WES study.

13. Government agencies (Oklahoma and Federal) possessing regulatory powers over water quality and usage were contacted; the proposed study was discussed; and modifications were made where necessary to ensure concurrence of study adequacy.

14. Previous studies were evaluated as to parameters measured, frequency and location of sampling, and adequacy of analytical procedures and conclusions to ensure consistency of results and to assist in data interpretation.

Additional sampling programs

15. The pathways and rates of material transfer were not adequately understood to develop reliable models for predicting the fate of metals and pesticides in Arcadia Lake. Therefore, at the request of WES, the U. S. Geological Survey (USGS), in coordination with TD, established a sampling program on two existing impoundments, Lakes Thunderbird and Eufaula, to observe directly the distribution of materials in the tributary stream, impoundment water column, and bottom sediments.

16. The Northside Sewage Treatment Plant (STP) is the major point

source of contaminants in the proposed Arcadia Lake watershed. Because this source would be diverted below the proposed Arcadia Lake and very limited data on the quality of this effluent were available, at the request of WES, TD contracted OSU to sample nutrients, metals, pesticides, and other constituents in the final effluent of the STP. These data were used to estimate water-quality constituent concentrations following treatment plant relocation.

17. The WES conducted algal bioassay studies using established EPA methodologies to assess the availability of nitrogen and phosphorus compounds and the nutritional state of algae in the Deep Fork River system and surrounding waters. These studies were initiated because observed nutrient concentrations greatly exceeded levels that are generally accepted as adequate to support nuisance algal blooms. The purpose of the studies was to evaluate if algal growth limitation would be expected to occur due to nutrients in the Deep Fork River being in an unavailable form or due to limitation by factors other than nitrogen and/or phosphorus.

18. During the course of this study, data collected by USGS indicated that mercury might be a potential problem. The paucity of existing mercury data precluded reliable evaluation of the possible water quality of Arcadia Lake with respect to this constituent. OSU, in coordination with WES, conducted additional mercury sampling in the Deep Fork River near Arcadia to supplement the very limited USGS data.

19. The TD provided continuity in mercury sampling at the Deep Fork River by contracting the Oklahoma State Health Department to conduct sampling of both the river and the final effluent of the Northside STP at weekly intervals during the period 29 March through 1 June 1976.

20. During a meeting held in Dallas during February 1976, representatives from the Dallas Regional Office of EPA recommended that fish be collected from the Deep Fork River near the proposed damsite and analyzed for mercury content. The results of the analysis could directly determine the suitability of the fish as a food source by comparing body burden concentrations to Food and Drug Administration (FDA) limits and determine the existence of possible adverse environmental effects of

mercury contamination by comparison to EPA criteria. Furthermore, the procedures of fish flesh analysis would provide adequate sensitivity that is difficult to attain in the analysis of water samples. The study recommended by EPA was expanded to include lead and all chlorinated pesticides that were considered possible problem constituents in the proposed impoundment.

Water-quality parameters warranting further analysis

21. Existing water-quality data were surveyed to determine those water-quality constituents whose values approached or exceeded appropriate standards. The most stringent criterion or standard selected from Oklahoma Water Quality Standards[5] or from EPA's 1976 Quality Criteria for Water[6] was used to ensure a thorough evaluation.

Establishment of relationships between stream discharges and loadings of nutrients, metals, and pesticides

22. Using existing data, mass loadings of important water-quality constituents were calculated and the significance of major point sources of pollution evaluated. The establishment of relationships between discharge and loadings for some parameters permitted the simulation of reservoir behavior for time periods for which quality data were not available. Maximum and probable reservoir and release concentrations were calculated based on loadings, precipitation, evaporation, biological activity, and withdrawals.

23. A land-use approach to estimate phosphorus and nitrogen loadings to the proposed Arcadia Lake was conducted. This procedure allowed the evaluation of changes in tributary water quality as a result of increased urbanization and diversion of municipal point-source pollution that may occur during the life of the project. Resultant loadings calculated from land use were compared to loadings predicted from existing stream water-quality data.

Comparison with existing reservoirs

24. When appropriate data were available, a comparison of

predicted water quality of the proposed Arcadia Lake with nearby existing reservoirs permitted an evaluation of the predictive procedures and provided additional information about the likelihood of possible water-quality problems in Arcadia Lake.

Ecologic model application

25. An ecologic model was used to simulate the limnological conditions expected to occur at Arcadia Lake. The thermal portion of the model was calibrated using data from surrounding reservoirs. The model was used to predict physicochemical stratification, to evaluate the potential for algal blooms, and to provide one means of evaluating the potential problem of coliform contamination. Predictions of thermal and dissolved oxygen stratification made with the ecologic model were compared to predictions made by the WESTEX model.

WESTEX model application

26. The seasonal thermal and dissolved oxygen stratification that would be expected to occur in Arcadia Lake was simulated using the WESTEX model. Temperature profiles from surrounding reservoirs were used to calibrate the model. Because of the significance of meteorological conditions on the behavior of southern Great Plains reservoirs, a factorial arrangement of simulations was made for dry, wet, and average flow conditions under both seasonally hot and cold conditions. The WESTEX model was also used to evaluate selective withdrawal on downstream temperature and dissolved oxygen.

Lake fishery estimates

27. Mr. Robert M. Jenkins, Director, National Reservoir Research Program, U. S. Fish and Wildlife Service, provided an estimate of the quality and quantity of the lake fishery using regression equations appropriate for an Oklahoma reservoir with the expected morphometrical, limnological, and operational characteristics of the proposed Arcadia Lake.

PART II: WATERSHED DESCRIPTION AND MAJOR POINT SOURCES OF CONTAMINANTS

28. The Arcadia Lake watershed covers an area of 273 km^2 of which 39 percent is open, 33 percent is urbanized, 19 percent is woodlands, 8 percent is pasture, and 1 percent is cultivated.[1] Existing data indicate water-quality contamination from both point and diffuse sources. Of particular concern in this study was the pollution of the upper Deep Fork River from diffuse sources, especially from the relatively large urbanized area.

Industrial Effluents

29. The OSU environmental study identified three major point sources of industrial effluents: a door and window manufacturing plant, a plastics plant, and a power generating plant.[2] The study reported that these sources did not appear to be causing any degradation of water quality; however, since their effluents would be flowing into the proposed impoundment, their significance was evaluated.

30. Using industrial waste load allocations developed by the Oklahoma State Department of Pollution Control for Basin Five, an "order of magnitude" estimate of mass contribution can be derived.[7] Table 1 lists maximum waste load allocations as a percentage of load measured at Arcadia. The USGS water-quality sampling near the Arcadia damsite provided estimates of many water-quality concentrations based on both grab and composite samples (Table 1). Grab samples represent the quality of water at the time of sample collection, and composite samples represent a mixture of stream water collected over intervals ranging from 1 to 20 days. A very significant difference existed between the methods of sample handling prior to analysis. Grab samples were preserved and analyzed within a short time period. Composite samples were not preserved immediately but allowed to sit for time periods of roughly 30 days prior to analysis.

31. Losses to the stream bed would be expected to decrease the

percentage contributions tabulated in Table 1. Even assuming no losses, industrial point-source pollution from these three plants appears insignificant relative to measured loadings at Arcadia. The nutrient concentrations in the industrial effluents reflect high ambient concentrations in the influent. Data collected prior to 1974 reflect to varying degrees industrial effluents discharged into storm sewers or directly into the upper Deep Fork River. However, major sources of industrial contaminants were identified by 1974, and their effluents redirected through sanitary sewer systems.

Municipal Effluents

32. The OSU environmental study identified the municipal effluents of Oklahoma City and Edmond as major point sources of contaminants to the upper Deep Fork River.[2] Data used to evaluate the significance of sewage treatment facilities in the watershed are presented in Table 2. A comparison of loadings reveals that the Edmond Southeast and Northside STP's contribute 92 percent of the nitrogen and phosphorus attributable to municipal effluents (Table 3). Significantly, the Edmond Southeast STP was relocated outside of the Arcadia Lake watershed in August 1974, and the Northside plant is scheduled to be removed from the watershed.

33. Data measured during base flow conditions at Arcadia prior to August 1974 should directly reflect the Edmond STP discharge of contaminants. Little phosphorus or nitrogen data are available near Arcadia following sewage effluent diversion, which precluded the calculation of reliable reductions in loadings at Arcadia. However, using loading estimates in Table 3, upper bounds for phosphate and nitrogen reduction at Arcadia are 32 to 37 and 25 to 47 percent, respectively, as a result of relocating the Edmond STP discharge. In reality, effects of diversion of the discharge were less significant than these percentages indicate because of losses in the stream due to chemical reactions, biological utilization, and precipitation.

34. For the same reasons, estimates of 49- to 91- and 68- to

77-percent reduction in loadings of nitrogen and phosphorus, respectively, at Arcadia following Northside STP diversion are high. However, this analysis does show that the Northside discharge is the most significant point source of nitrogen and phosphorus in the upper Deep Fork basin, and the diversion of this discharge below the proposed Arcadia damsite should sharply reduce the stream load of contaminants at Arcadia.

PART III: SURVEY OF EXISTING DATA, EVALUATION OF THE OSU STUDY, AND INITIATION OF ADDITIONAL STUDIES

35. A major effort in the preparation of this report was the evaluation of data previously collected by other agencies. This approach provided information on the number and duration of past studies that could be used for predicting the water quality of the proposed Arcadia Lake. It also indicated those constituents that had been measured and the frequency of past measurements and provided some estimate of the variation in water quality that could be expected in the Deep Fork River. In addition, consideration was given to the methodologies used in previous studies as this will influence the chemical species measured and may influence the resulting data interpretation.

36. Besides indicating what information was available, the evaluation of previous studies indicated what additional data were needed to predict water quality in the proposed impoundment. Thus, the approach taken also served as a planning tool to initiate and conduct studies necessary to assess water quality in the proposed Arcadia Lake.

Water-Sampling Program

37. Seven agencies, institutions, or individuals have conducted sampling in the Arcadia Lake watershed: USGS, Oklahoma Water Resources Board (OWRB), Oklahoma State Health Department, OSU, Oklahoma City-County Health Department, Water Pollution Control Division of the Oklahoma City Department of Public Works, and Mr. Jimmie Pigg of Moore High School, Moore, Oklahoma. A summarization of the parameters measured, the frequency of measurement, and duration of the studies is presented in Table 4. Water-quality constituents were classified as nutrients, metals, pesticides, and other water-quality parameters for convenience of generic discussion and report organization.

38. USGS data were used most extensively in the water-quality analyses conducted as a part of the WES study because these data were the most comprehensive; samples were collected at regular intervals; and

corresponding river discharge data were available. The USGS data provided estimates of over 70 water-quality constituents measured over a 5-yr period. Grab samples were collected at monthly intervals while composite samples, on the average, were collected weekly. Data collected by agencies other than the USGS provided estimates of the expected ranges of specific water-quality parameters and enabled the identification of major sources of pollutants and an evaluation of their significance. Data collected by the Oklahoma City Department of Public Works provided frequent measurements of nutrient concentrations near Arcadia between 1966 and 1974.

39. Differences among agencies in location of sampling stations, sample collection and preservation, and analysis resulted in inconsistent results. The evaluation of the causes and significance of the inconsistencies is discussed in the following sections of the report.

OSU Environmental Study

40. OSU prepared a report for TD on the water quality of the Deep Fork River during 1972-1973.[2] Time and funding constraints prevented OSU from conducting a thorough water-quality evaluation. While the data compilation was useful, the technical merits of some portions of the data interpretation must be questioned. For example, it is stated on page 5 of the report that an impoundment evaluation "is very objective." This is true only if the chemical species in the impoundment are in the same form as those used in the bioassay studies used to establish the criteria applied in an evaluation of water quality. As the chemical forms become more divergent, the evaluation becomes more subjective. This point is important because it was reported in the OSU study that lead would exceed water-quality criteria. However, lead concentrations in the study were defined as being soluble at pH ≤ 2; this definition is not comparable to bioassay conditions used to establish the applied water-quality criteria. Thus, there is no established relationship between the lead concentration results reported in the OSU study and the environmental effects of lead. Other results, to be discussed in more

detail elsewhere in this report, show that most lead is in the particulate form and that dissolved lead concentrations are usually an order of magnitude below the lead criterion.

41. The eutrophication discussion in the OSU report should also be interpreted with caution. This section leaves the unsupportable impression that a knowledge of algal chemical composition and the ratio of nutrients to critical concentrations allows a prediction of expected algal populations. Algal cell composition is not a measure of growth potential because it fails to consider the chemical form of the potential nutrients in the water column. This point is important because all chemical forms are not equally available to algae and nutrients cannot be stimulatory if they are not in a form that can be used.

42. The ratio of actual nutrient concentrations to critical nutrient concentrations is also a poor method of estimating an algal response because it fails to consider the limiting nutrient. The limiting nutrient concept states that the nutrient in least supply relative to actual need will control algal response. If a nutrient is present and available in excess of actual needs, the abundance is not a measure of algal response. The method presented by OSU also fails to consider the possibilities of temperature, light, or one or more toxicants acting to limit the algal population.

43. A bioassay procedure to define nutrient availability and limiting nutrient(s) in the Deep Fork River is discussed elsewhere in this report.

Northside STP Sampling Program

44. Because the Northside STP discharge is large relative to other known point sources and since the plant will be removed from the watershed, additional studies were conducted to characterize the composition and contribution of the plant effluent. In addition, data were needed to determine the significance of STP relocation on instream concentrations of critical parameters other than nitrogen and phosphorus.

45. The TD contracted OSU to conduct additional sampling of the

Northside STP. The objective of the study was to estimate the concentrations of water-quality constituents in the plant effluent previously identified as possible problems. These data were subsequently used to predict instream concentrations after STP relocation. The sampling program was limited to conform with time constraints of the WES study.

Procedure

46. Four samples were collected below the final effluent outfall at 1-week intervals on 12, 19, and 26 March and 2 April 1975. Each sample consisted of two replicates. The multiple samples permitted estimates of weekly variation, while the replicates permitted estimates of analytical error. The methods used for chemical analyses were American Public Health Association (APHA) Standard Methods[10] or EPA Methods[11] except for the analyses of the chlorophenoxy herbicides, which were based on a tentative procedure from an American Society of Testing and Material (ASTM) committee.[13]

Results

47. The water-quality parameters that were measured and the means of their concentrations during the study are listed in Table 5. The data are within the ranges of values measured at Arcadia. Results suggest that the Northside STP is a significant source of phenol, total nitrogen, lindane, dieldrin, diazinon, 2,4-D, silvex, 2,4,5-T, and possibly mercury.

Modification of Existing Data to Reflect STP Relocation

48. Existing data on concentrations and flows measured at Arcadia in conjunction with estimates of concentrations at the Northside STP permitted modification of instream Deep Fork River water-quality constituent concentrations to reflect STP relocation.

Assumptions

49. The concentrations of selected constituents measured in the final effluent of the STP during March and April 1975 were assumed to be characteristic of effluent concentrations during the period 1969-1974

for which data exist at the Arcadia gage on the Deep Fork River. Additional assumptions were that no losses or gains in flow or constituent mass occurred between the point of sewage treatment discharge and the Arcadia gage. The significance of errors resulting from the extrapolation of concentrations measured over a 1-month period to the period of record cannot be evaluated. The sampling program could not detect periodicities or trends. The assumption of zero losses or gains probably resulted in overestimation of the mass contribution of the STP because losses of material in the stream will occur due to chemical reaction, biological utilization, and precipitation. However, the results probably provide conservative estimates of concentrations to be expected following STP diversion. Existing Deep Fork River data used in the calculations reflect contaminants from the Edmond Southeast STP and industries no longer discharging into the Arcadia Lake watershed.

Procedure

50. Mass loadings of water-quality constituents at the STP were calculated at a monthly resolution to account for both seasonal and annual variations in treatment plant effluent volumes. Loadings calculated based upon records available at Arcadia were reduced by the amount equal to that discharged by the treatment plant. In addition, flows at Arcadia were reduced to reflect diversion of flows that originated at the treatment plant. When necessary, the modified flows and modified loadings were used to calculate concentrations that would be expected after STP relocation.

Lakes Thunderbird and Eufaula Sampling Program

51. At the request of WES, the USGS in coordination with the TD established a sampling program on two existing impoundments, Lakes Thunderbird and Eufaula, to observe directly the distribution of materials in the tributary stream, impoundment water column, and bottom sediments.

52. Lake Thunderbird was selected for comparative analysis due to its close proximity and similar watershed characteristics to the

proposed Arcadia Lake. The Deep Fork River arm of Lake Eufaula was chosen because the major tributary was the Deep Fork River and a water-quality gaging station existed upstream near Beggs, Oklahoma.

53. Six sampling sites were established, three in the Little River arm of Lake Thunderbird and three in the Deep Fork River arm of Lake Eufaula. Site 1 was located within the lakes at a point such that the cumulative volume of the arm was equal to the volume of Arcadia Lake. Site 3 in each lake was located upstream at the point where tributary flow was first observed. Site 2 was located at the midpoint between Sites 1 and 3.

54. At each site, sampling was conducted 1 m below the water surface and 1 m above the lake bottom. In addition, the top 10 cm of sediment at each site was sampled for metals and pesticides. Sampling was conducted on 27 September 1974 at Lake Eufaula and on 30 September 1974 at Lake Thunderbird.

55. A tabulation of parameters measured and the results of the analyses are presented in Appendix A.

PART IV: EVALUATION OF CRITICAL PARAMETERS

56. The State of Oklahoma has designated the Deep Fork River for the following present and potential uses:

a. Public and private water supplies.

b. Fish and wildlife propagation.

c. Agriculture (including livestock watering and irrigation).

d. Industrial and municipal cooling water.

e. Receiving, transporting, and/or assimilation of adequately treated waste.

f. Primary body-contact recreation (including recreational uses where the human body may come in direct contact with the water to the point of complete body submergence).

g. Secondary body-contact recreation (including recreational uses, such as fishing, wading, and boating, where ingestion of water is not probable).

h. Aesthetics.

57. Water-use classifications are developed with the intention of providing the same level of protection for all waters of the same beneficial use designation. Protection of a particular beneficial use is accomplished through the enforcement of uniform water-quality standards which apply to that use. A water-quality standard is an authoritatively established rule that permits regulation of the limits of a water-quality constituent to ensure compatibility with the intended use or uses of the water. In contrast, a water-quality criterion is a limit of variation of a water-quality constituent judged, based on scientific evidence, not to have an adverse effect on man or organisms inhabiting the water.[13] Water-quality standards are established by a regulatory agency through the selection of a subset of water-quality criteria. The choice depends upon the beneficial use of the water and the degree of impairment judged acceptable as a matter of public policy.

58. An evaluation of existing information was made to identify potentially critical water-quality parameters which should receive more detailed study. One consideration in this evaluation was to determine if downstream project releases from the proposed Arcadia Lake would

result in impairment of the designated Deep Fork water uses. A second consideration was to determine if the water quality in the proposed Arcadia Lake would comply with standards appropriate to the reservoir project purposes.

59. For both considerations, the selected methodology was the comparison of predicted impoundment and downstream water quality with the appropriate Oklahoma State water-quality standards.

Procedure

60. In anticipation of more stringent standards resulting from the implementation of the "Federal Water Pollution Control Act Amendments of 1972 (PL 92-500)," [14] all known existing water-quality data collected in the Arcadia Lake watershed were surveyed to identify toxic or harmful constituents for which numerical criteria or standards exist in order to determine potentially critical parameters not identified in previous studies. Although the survey was quite comprehensive, this does not preclude the unknown existence of potentially toxic or harmful materials in the upper Deep Fork River. However, recent legislation provides for more stream surveillance, more stringent control of toxic materials, and a timetable for the elimination of the discharge of pollutants into the Deep Fork River. This legislation is directed to the elimination of pollution from point sources and minimization of man-induced contamination from nonpoint sources.

61. Standards and criteria used from evaluation were selected from Oklahoma Water Quality Standards[5] or EPA Quality Criteria for Water.[6] The evaluation of trace metal data was based on dissolved or soluble concentration data because metals in this form, as opposed to particulate form, would more closely approximate the chemical species used in bioassay experiments that were the basis for recommended criteria. As discussed by Lee et al., toxicity can vary markedly with the form of a chemical, and the application factor used to relate acute lethal to chronic sublethal effects has questionable applicability to soluble and particulate forms of the same species.[15] Therefore, with the exception

of mercury and pesticides, only dissolved chemical data were compared to criteria. Criteria for mercury concentrations were based on total mercury because of possible interconversion of mercury forms.

Results

62. Data representing a total of 70 water-quality constituents measured by the USGS near the proposed damsite were evaluated. Twenty-five water-quality parameters were found to have at least one sample value equal to or exceeding one-tenth of the most stringent criterion or standard. In Table 6, these parameters are listed with the associated standard or criterion used for evaluation and selected statistics to summarize sample distribution. Extensive nutrient data and limited metal data collected within the Arcadia Lake watershed were within the ranges measured at the Arcadia damsite.

63. A comparison of the average values of the constituents tabulated in Table 6 with the most stringent standard or criterion revealed that concentrations of 12 water-quality parameters equaled or exceeded recommended or permissible levels: ammonia; manganese; mercury; DDT; aldrin; dieldrin; chlordane; heptachlor; lindane; PCB; phenols; and fecal coliforms.

64. Based upon the data surveyed, fecal coliform bacteria, ammonia nitrogen, and manganese were the only water-quality constituents found to exceed present State standards applicable to the Deep Fork River use classification or proposed reservoir purposes. For purposes of this evaluation, all water-quality parameters that were found to have at least one sample value equal to the most stringent criterion or standard received further analysis.

65. The State of Oklahoma has as part of their water-quality standards the requirement that a cumulative relationship value (CRV) not exceed 1.0.

$$CRV = \frac{C_1}{L_1} + \frac{C_2}{L_2} + \cdots + \frac{C_n}{L_n} \qquad (1)$$

where

C = the measured concentration of toxic materials 1 through n

L = the established water-quality standard for the toxic materials 1 through n

The series of toxic materials specified by the State of Oklahoma to be considered in calculating the CRV are ammonia, cadmium, hexavalent chromium, trivalent chromium, copper, cyanide, lead, mercury, nickle, selenium, silver, and zinc.

66. The purpose of the CRV standard as implied in Oklahoma's Water-Quality Standards[5] was to provide a measurement of the cumulative and synergistic effects of the toxicants. However, the CRV standard was derived from a concept originally intended for use when two or more contaminants with significant toxicities were present in concentrations approaching values known to cause acute toxicity and where the cumulative effect of these contaminants might produce toxicity.

67. In applying this concept, Sprague[16] recommended that $C_{1....n}$ in equation 1 be the concentration of the available form of chemical used in bioassays to establish the respective LD_{50} values ($L_{1....n}$) and that only contaminants known to produce synergistic effects be included. Oklahoma's interpretation of the CRV does not recognize the necessity of considering only the available form of a contaminant; the error in using a water-quality standard that is several orders of magnitude lower than the LD_{50} because of safety factors applied in establishing the standard; or the necessity of only considering contaminants with demonstrated synergism with respect to toxicity.

68. The use of the CRV specified in Oklahoma's Water-Quality Standards[5] for measuring cumulative chronic effects of the contaminants is without valid scientific or technical basis. Furthermore, the CRV as proposed by the State of Oklahoma is a result of misinterpretation or disagreement with guidance provided by the EPA and the technical literature that served as a basis of EPA's guidance.[17] If applied as stated in Oklahoma's Water-Quality Standards,[5] the CRV would overestimate the value intended by the developers of the concept by orders of magnitude, and many safe natural water supplies in the U. S. would fail the

CRV standard. The rationale for this conclusion was based on a personal communication* and information in reference 16.

69. It is concluded that the application of the CRV to observed concentrations of the specified contaminants in the Deep Fork River would be technically invalid and without basis in scientific fact. Furthermore, such a comparison would not be of value in predicting synergistic effects in Arcadia Lake since, based on data from surrounding lakes and their inflowing tributaries, the concentrations of these contaminants would be lower than observed in the Deep Fork River.

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* Telephone conversation between Dr. Rex L. Eley, WES, and Dr. Bill Brungs, EPA Environmental Research Laboratory, Duluth, Minnesota.

PART V: USE OF THE WESTEX MODEL FOR PREDICTING TEMPERATURE AND DISSOLVED OXYGEN REGIMES IN ARCADIA LAKE AND DOWNSTREAM RELEASES

70. The purpose of this phase of the study was to simulate the temperature and dissolved oxygen regime to be expected within the proposed Arcadia Lake and to determine a selective withdrawal intake configuration that would allow project operation to satisfy the release temperature objective. The computer model used in conjunction with this investigation was developed at WES and will be identified as "WESTEX."

Approach

71. The use of a numerical simulation model provided the capability for assessing the effect of historical data on the lake for long periods of record. The approach involved the selection of several study years and simulation of project operation and lake response for each of these years. Study years selected had combinations of hydrometeorological conditions that could be expected to occur at the proposed project. The data required to conduct the simulations were lake inflows and outflows, inflow stream temperatures, meteorological data for each of the study years, geometry of the lake, and geometry of the intake structure.

72. The heat transfer into and out of the lake was computed, and the heat was distributed within the lake. A heat budget was maintained throughout the simulation period. An objective temperature was specified for each simulation day, and an operating scheme was determined. The operation for any day was the combination of open ports that minimized the difference between the downstream objective temperature and the predicted release temperature. The output from the simulation included a comparison between objective and release temperature in graphical form through the simulation period as well as tabular summaries for each day and plotted profiles of temperature within the reservoir at specified times of the year.

Model Description

73. The WESTEX model provides a procedure for examining the balance of thermal energy imposed on an impoundment. This energy balance and lake hydrodynamic phenomena are used to map vertical profiles of temperature and dissolved oxygen in the time domain. The model includes computational methods for simulating heat transfer at the air-water interface, advected heat due to inflow and outflow, and the internal dispersion of thermal energy. The model is conceptually based on the division of the impoundment into discrete horizontal layers. Fundamental assumptions include the following:

a. Isotherms are parallel to the water surface both laterally and longitudinally.

b. The water in each discrete layer is isotropic and physically homogeneous.

c. Internal advection and heat transfer occur only in the vertical direction.

d. External advection (inflow and outflow) occurs as a uniform horizontal distribution within each layer.

e. Internal dispersion (between layers) of thermal energy is accomplished by a diffusion mechanism that combines the effects of molecular diffusion, turbulent diffusion, and thermal convection.

The surface heat exchange, internal mixing, and inflow and outflow processes are simulated separately, and their effects are introduced sequentially at daily intervals.

74. The WESTEX model employs an approach to the evaluation of surface heat transfer developed by Edinger and Geyer.[18] This method formulates equilibrium temperatures and coefficients of surface heat exchange. Equilibrium temperature is defined as that temperature at which the net rate of heat exchange between the water surface and the atmosphere is zero. The coefficient of surface heat exchange is the rate at which the heat transfer process will occur. The equation describing this relationship is:

$$H = K (E - \theta) \qquad (2)$$

where

H = net rate of surface heat transfer (Langleys/day)

K = coefficient of surface heat exchange (Langleys/day/^{o}C)

E = equilibrium temperature (^{o}C)

θ = surface temperature (^{o}C)

The computation of equilibrium temperature and heat exchange coefficient is based on meteorological data and is outlined in the literature by Edinger, Duttweiler, and Geyer.[19]

75. The net heat exchange at the surface is composed of seven heat exchange processes:

a. Short-wave solar radiation.

b. Reflected short-wave radiation.

c. Long-wave atmospheric radiation.

d. Reflected long-wave radiation.

e. Heat transfer due to conduction.

f. Back radiation from the water surface.

g. Heat loss due to evaporation.

For every day of meteorological data, the seven heat exchange terms can be evaluated and the net heat exchange expressed in terms of an equilibrium temperature and an exchange coefficient.

76. All of the surface heat exchange processes, with the exception of short-wave radiation, affect only approximately the top metre of the lake. Short-wave radiation penetrates the water surface and increases the temperature at greater depths. Based on laboratory investigations, Dake and Harleman[20] have suggested an exponential decay with depth for describing the heat flux due to short-wave penetration.

77. The surface heat exchange concepts are implemented in the WESTEX model by the exponential penetration of a percentage of the incoming short-wave radiation and the placement of the effect of all other sources of surface heat exchange into the surface layer. This can be expressed mathematically by the following two equations:

$$H_s = K\ (E - \theta) - (1 - \beta)\ S \qquad (3)$$

$$H_i = (1 - \beta)\, Se^{-\lambda z_i} \qquad (4)$$

where

H_s = heat transfer rate into or out of surface layer (Langleys/day)

β = fraction of short-wave radiation absorbed in the surface layer

S = total incoming short-wave radiation (Langleys/day)

H_i = rate of heat absorbed in layer (i) (Langleys/day)

e = natural logarithmic base (2.7183)

λ = absorption coefficient (m^{-1})

z_i = depth below surface (m)

78. The process of inflow into a lake is simulated in WESTEX by the placement of inflow quantity and quality at that layer where the density of the lake corresponds most nearly to the density of the inflow. Research efforts and physical model studies at WES have indicated the existence of entrainment-induced density currents that flow upstream along the surface into the turbulent mixing zone caused by the inflow. Entrainment is implemented in the model by augmenting the inflow quantity with a volume from the surface layer. Characteristics of inflow and the entrained flow are averaged, and mixed values of density, temperature, and other water-quality parameters are determined. The mixed density is used to determine placement of the total quantity and mixed quality. Simulation of the inflow process displaces upward a volume equal to the total inflow quantity. This upward displacement is reflected in the model by an increase in the water surface. A corresponding decrease in water surface occurs as a result of the outflow simulation process.

79. The volume of the entrained current is generally expressed as a percentage of the inflow quantity. Prior flume studies indicated that this percentage ranges from 0 to nearly 200 percent. The percentage is thought to be a function of slope, width, flow quantity, density of inflow, and density within the lake; but such analytical relationships have not been determined.

80. The internal dispersion process is represented by an internal mixing scheme based on a simple diffusion analogy. Internal mixing transfers heat and other water-quality constituents between adjacent layers.

The magnitude of the transfer between two layers is a percentage of the total transfer required to mix the two layers completely. This percentage is a mixing coefficient that is defined for every layer. Data input includes values of the mixing coefficient at the top and at the bottom of the lake. An exponential fit between the two extreme values is used to determine the appropriate coefficient at each layer.

81. The outflow component of the model incorporates the selective withdrawal techniques developed at WES.[21] Transcendental equations defining the zero velocity limits of the withdrawal zone are solved with a half-interval search method. With knowledge of the withdrawal limits, the velocity profile due to outflow can be determined. The flow from each layer is then the product of the velocity in the layer, the width of the layer, and the thickness of the layer. A flow-weighted average is applied to water-quality profiles to determine the value of the release content of each parameter for each time step.

82. The lake regulation algorithms have been developed to realistically simulate the field operation of a selective withdrawal system. The selective withdrawal system is assumed to be configured with an arbitrary number of selective withdrawal intakes located in each of two wet wells with a separate floodgate. Maximum flows and minimum flows from each intake and from the floodgate must be specified. Also, the maximum flow for the selective withdrawal system is specified. The algorithms attempt to withdraw water numerically at or near the objective temperature. Withdrawal will be from either one intake level, two adjacent intake levels, and/or the flood control intake, depending upon the objective temperature, the temperature profile, the intake capacities, and the amount of flow to be released.

83. The WESTEX model contains a simple method for routing dissolved oxygen (DO) and biochemical oxygen demand (BOD) based on the work of Bella,[22] Carroll and Fruh,[23] and Markofsky and Harleman.[24] The DO and BOD content of the inflow into the lake and the outflow from the lake are evaluated and used to adjust profiles within the lake. The surface layers are assumed saturated with DO. The saturation condition is extended from the surface down to the depth at which a 1°C temperature

difference exists from the temperature of the water surface. After saturation of the surface layers, the process of mixing between adjacent layers occurs. A depletion term for DO is then applied to every layer below the saturation zone. The DO depletion rate is of the form:

$$D = \left[K(T)\right] L \tag{5}$$

where

D = DO depletion rate (mg/ℓ/day)

K(T) = temperature dependent deoxygenation coefficient (day^{-1})

L = existing BOD (mg/ℓ)

84. The temperature-dependent deoxygenation coefficient is found in the literature as:

$$K(T) = K(20)\ (1.047)^{T-20} \tag{6}$$

where

K(20) = deoxygenation coefficient at 20°C (day^{-1})

T = temperature within the layer (°C)

85. The BOD depletion rate has the form:

$$B = K_b\ L \qquad \text{(Reference 20)} \tag{7}$$

where

B = BOD depletion rate (mg/ℓ/day)

K_b = decay coefficient (day^{-1})

L = BOD (mg/ℓ)

Subtracting the DO and BOD depletions from the existing DO and BOD in each layer yields the final DO and BOD content in every layer for the time step. When data required for the routing of BOD proved to be inadequate, a constant value of BOD in the lake was assumed.

86. Dissolved oxygen is an unconservative parameter whose temporal and spatial dynamics are influenced by biological, chemical, and

physical processes. The simple approach used in the WESTEX model can provide useful information concerning relative effects of reservoir hydrodynamics and project operations on in-lake DO profiles and downstream releases. However, in making predictions of dissolved oxygen concentrations and in analyzing effects of varying organic and nutrient loadings, other approaches are desirable. Dissolved oxygen simulations made using the WESTEX model are compared elsewhere in this report with ecologic simulations and with data from existing reservoirs in the area.

Selection of Study Years

87. For the selection of study years, statistical analyses of mean monthly streamflow and mean monthly dry bulb temperature were conducted for the period of record 1944-71 (Plate 1). Study years were limited to this period due to lack of adequate meteorological data prior to 1944 and streamflow records after 1971. Only records from March through October were considered in the selection of study years because experience has shown that this is the period in which density stratification in the lake is most affected by hydrology and meteorology. Emphasis was given to the characteristics of the spring months due to the particular importance of these months in fish reproductive cycles.

88. Combinations of above average, average, and below average hydrologic and meteorologic conditions were considered in the selection of study years. The nine years discussed below were selected for the Arcadia Lake water-quality investigation.

a. 1947 - Runoff was less than average for the period January to March and was well above normal for the remainder of the year. Air temperatures were slightly colder than average through July and much warmer than average from August through October.

b. 1953 - Runoff was slightly greater than average for the stratification period. Air temperatures were near average for most of the year, although they were much warmer than average in June.

c. 1954 - Runoff was well below average throughout the year. Air temperatures were relatively cold in May and warm in April and from July through September.

d. 1954 - Runoff was well below average throughout the year. Air temperatures were near average throughout the year.

e. 1957 - Runoff was greater than average during the stratification period. Air temperatures were colder than average over most of the year.

f. 1958 - Runoff was near average throughout the year. Air temperature was near average throughout the stratification period but colder before and after the stratification period.

g. 1963 - Runoff was less than average throughout the year. Air temperature was average for most of the stratification period but was warmer in April and October.

h. 1969 - Runoff was average for the first half of the year and less than average for the second half. Air temperature was colder than average in March, June, and October but was average the rest of the year.

i. 1970 - Runoff was slightly less than average throughout most of the year and cold during October.

Data Requirements

Meteorology

89. Meteorological data from the Oklahoma City Weather Station were used for this study. The weather station is located approximately 16 km southwest of the proposed project. The required data consist of dry bulb temperature, dew point temperature, wind speed, and cloud cover. These data were obtained from the National Climatic Center in Asheville, North Carolina. Eight observations were furnished for each day. Daily average values were computed and used to determine equilibrium temperatures, surface heat exchange coefficients, and daily average net solar radiation for all study years.

Hydrology

90. Mean daily lake inflow and outflow quantities are shown in Plate 2. Hydrologic routings were based on the proposed plan of operation of the project.

Stream temperature

91. Stream temperature records for each of the nine study years were not available. Observed stream temperatures on the Deep Fork at

Oklahoma City existed for the period 1970 through 1973. A total of 45 grab samples were available for stream temperature analysis. These data were used in the development of a regression equation relating air temperature to observed stream temperature. The following regression model was used:

$$\theta_t = \alpha + \beta_1 T_t + \beta_2 T_{t-1} + \beta_3 T_{t-2} + \beta_4 T_{t-3} \quad (8)$$

where

θ = stream temperature (^oC)

t = Julian day

T = dry bulb temperature (^oC)

α and β are regression coefficients as follows:

α = 4.076

β_1 = 0.4271

β_2 = 0.1845

β_3 = 0.0391

β_4 = 0.1433

Equation 8 is considered to describe the daily average natural stream temperature at the project inflow. The computed stream temperatures (Plate 3) were used as input for the model simulations.

Objective temperature

92. A least-square analysis was used to fit a harmonic curve to the predicted stream temperatures for the nine study years. The curve represents the average annual natural stream temperature variation to be expected during a year. The following regression model was used:

$$\theta_t = A \sin (Bt + C) + D \quad (9)$$

where

θ = average stream temperature (^oC)

t = Julian day

The coefficient B is a unit conversion from days to radians.

The coefficients A, C, and D were determined by solution of Equation 8 with the Newton-Raphson technique and were computed to be the following:

A = -9.87°C

B = 0.01721 radians/day

C = 1.230 radians

D = 16.16°C

Equation 9 was used to define the project release temperature objective.

Dissolved oxygen

93. Measurements of DO and BOD in 22 grab samples from the Deep Fork River were provided by USGS. A regression analysis was performed relating observed DO to saturated DO in the stream. The regression coefficients were used to generate daily values of DO content of the flow entering the lake for all of the study years. As shown in the literature, the expression for saturated DO in a stream as a function of stream temperature is:

$$D_S = \frac{1}{A + B\theta} \tag{10}$$

where

D_S = saturated dissolved oxygen value (mg/ℓ)

θ = stream temperature (°C)

A and B are constants as follows:

A = 0.0677

B = 0.00208

94. A linear regression of observed DO with saturated DO used the following regression equation:

$$D_i = \alpha D_S + \beta \tag{11}$$

where

D_i = inflow DO (mg/ℓ)

D_S = saturated DO (mg/ℓ)

α and β were computed to be the following:

α = 1.04

β = -2.53

These relations provided daily values of inflow dissolved oxygen to be used as input to the model.

95. It was ascertained that insufficient data exist for the generation of daily input values of BOD. Simulations were therefore conducted with an oxygen demand independent of time and depth. No attempt was made to route BOD; rather the constant demand was maintained throughout the simulation period. Markofsky and Harleman[24] suggest that oxygen decay rates that are reasonable in a stream environment may be inappropriate for use in a lake. A 5-day BOD value will not be a good estimate of ultimate BOD in a lake, and the decay rate may proceed at a slower rate than in a stream. Markofsky and Harleman suggest using a decay coefficient on the order of 0.01 and increasing the 5-day BOD values to represent a larger ultimate demand.[24] Accurate estimation of ultimate BOD in Arcadia Lake is, however, not possible with existing data.

96. It has been observed[22,23] that the overall oxygen depletion rate in a relatively unpolluted lake is on the order of 0.1 mg/ℓ/day. This value of an oxygen-depletion rate was verified and used for the Richard B. Russell Water Quality Study.[25] For the Arcadia Lake investigation, simulations were conducted with a deoxygenation coefficient of 0.01 and constant oxygen demands of 2 and 45 mg/ℓ. The value of 2 mg/ℓ represents relatively unpolluted conditions. The value of 45 mg/ℓ is the largest value of 5-day BOD observed in the Deep Fork River and thus represents an extreme degree of pollution. These two conditions give average oxygen depletions of 0.02 and 0.45 mg/ℓ/day, respectively. These two values of oxygen depletion should bracket the depletion that would exist in Arcadia Lake with current and future BOD loadings.

Model Calibration

97. As has been discussed previously, the WESTEX model requires the determination of coefficients of surface heat exchange distribution and internal mixing. For Arcadia Lake these coefficients were determined by conducting simulations with 1970 hydrologic and meteorologic data. Coefficients were adjusted and simulation was repeated until the

predicted temperature profiles corresponded in shape and range to those observed in three existing Oklahoma lakes. The lakes used for this purpose were Arbuckle, Oologah, and Keystone. Profiles of temperature and DO were obtained from TD (Plate 4).

98. The following coefficients were determined from this analysis:

β = fraction of incoming short-wave radiation absorbed in the surface layer = 0.6

λ = absorption coefficient = 0.66 m^{-1}

α_1 = mixing coefficient at surface = 0.6

α_2 = mixing coefficient at bottom = 0.3

99. The mixing coefficients for Arcadia Lake simulations were considerably larger than the coefficients used in previous studies of other lakes by WES. This was done to simulate mixing which will occur due to wind shear at the lake surface. Large magnitudes of wind speed have been consistently recorded at the Oklahoma City Weather Station (Plate 5). Profiles from Arbuckle Lake show considerable mixing.

Simulations

Location of selective withdrawal intakes

100. Many intake configurations were considered in the analysis of location of selective withdrawal intakes. Two of these configurations were found to provide the best capability for satisfying downstream temperature objectives. One configuration consisted of two wet wells and a separate floodgate. Each wet well contained two levels of selective withdrawal intakes. The second configuration consisted of seven intakes in a single wet well and a separate floodg te.

101. The configuration with two wet wells, hereafter referred to as the "2-port operation," had intake levels at 12.2 and 16.8 m above the base of the dam and a floodgate with center line at 3.0 m above the bottom. This operation allowed passage of flow through one intake in each wet well. Blending of flows with two selective withdrawal intakes or one intake and the floodgate could be used to achieve a downstream objective temperature.

102. The configuration with seven intakes in one wet well will be identified as the "7-port operation." Intakes were located above the bottom at 9.1, 10.7, 12.2, 13.7, 15.2, 16.8, and 18.3 m. The floodgate center line was again located at 3.0 m above the bottom. With seven-level operation, the intake structure is simpler, but blending is allowed only between one selective withdrawal intake and the floodgate.

103. All simulations were conducted with a maximum flow capacity of 1.4 m^3/sec for each of the selective withdrawal intakes. This capacity was adequate for all of the study years except for a few days in 1947 when relatively high flows occurred in the spring months. Minimum flow capacities of 0.06 m^3/sec were used for selective withdrawal intakes and the floodgate.

104. The 7-port operation has been described as containing seven intakes into a single wet well. This concept could be implemented with a pipe at each intake level including the floodgate. Control of low flows could be accomplished with a butterfly valve at each intake. Thus many intakes could be provided economically and control of low flow at various elevations could be accomplished.

Temperature

105. Simulations were conducted for both of the selected intake configurations. Plates 6 and 7 show the capability of the two systems for meeting an objective temperature. Also shown in these plates are the port level hydrographs for operation to achieve the objective temperature.

106. Acceptable release temperatures can be achieved for all combinations of hydrology and meteorology investigated. It can be noted in Plate 7 that with 7-port operation, each of the lower ports was used during every study year. The top intake was not used during the years 1956, 1969, and 1970 because the pool depth remained below the elevation of the intake. Two or three of the intakes in the middle of the wet well could possibly be eliminated without adversely affecting release temperatures. The structure of the temperature within the lake during the year can be seen in the isotherm plots presented in Plates 8 and 9. Due to small outflow quantities, very little difference in temperature

structure can be observed for the two intake configurations.

Dissolved oxygen

107. Simulations were conducted for both intake configurations with a constant BOD of 2 mg/ℓ. Very little difference was observed in DO profiles and release DO content for the two configurations. Release DO was computed by the model to be the DO entering the intake structure and was above 3 mg/ℓ throughout the year. Additional simulations were conducted with 7-port operation and a constant oxygen demand of 45 mg/ℓ. Although some predicted values of DO release were quite low, most releases were 2 mg/ℓ or greater because downstream flow requirements were small and use of the low-level floodgate was not required. Flow was taken from the upper intake levels where the DO content was higher than at the bottom. Release DO contents for the two BOD conditions are shown in Plate 10. The upper band of points represents a constant BOD of 2 mg/ℓ. The lower band of points reflects a BOD of 45 mg/ℓ. Plate 11 shows isogram plots of DO within the lake for the year 1970. Reaeration, which occurs as flow passes through the outlet works and through a properly designed stilling basin, will increase the DO content of the downstream release to possibly 80- to 90-percent saturation.[26]

Results

108. The computed temperature releases for the proposed Arcadia Lake were well within the range of natural stream temperatures. Although there were some rapid temperature changes, these were not greater than the changes in natural stream temperatures. The temperature of the Arcadia Lake release should be adequate to meet water-quality standards.

109. Mathematical simulations of the lake indicated that isothermal conditions could occur as early as the beginning of September. This late summer overturn could cause problems in meeting a temperature objective, particularly during a period of low pool elevation. However, the release temperatures for the study years were reasonably close to target temperatures.

110. The simulations with a heavy BOD loading indicated anaerobic

conditions within the hypolimnion. However, the release water was drawn mostly from the upper ports, resulting in a release DO usually above 2 mg/ℓ. Oxygen uptake through the stilling basin or natural reoxygenation downstream could easily raise this value to well above 5 mg/ℓ. The temperature and DO content of project releases should be satisfactory.

111. There were no significant differences between the 2-level and the 7-level operation schemes for temperature or DO releases. The determination of which port arrangement to construct should be based on operational or other water-quality considerations.

PART VI: NUTRIENTS AND EUTROPHICATION POTENTIAL

112. Eutrophication is a natural process that is a consequence of high loadings of available nutrients that results, at least some time during the year, in a large standing biomass of algae. Emphasis was placed on nitrogen and phosphorus since the evidence is overwhelming that these are the major nutrients controlling algal productivity in water bodies; Deep Fork River data indicate that carbon could not be a limiting nutrient. Excessive eutrophication would be expected when the rate of supply of required nutrients in available form is in excess of nutritional needs necessary to maintain acceptable algal biomass and when other factors such as light availability, temperature, or toxic materials are not limiting algal growth.

113. Most procedures presently available to predict the water quality of a proposed impoundment possess some deficiencies. The significance of the deficiencies is dependent upon the adequacy of the data and site-specific characteristics of the impoundment. The relationships among available data, specific characteristics of the impoundment, and the most appropriate procedure to use are not entirely understood, often resulting in predictions that may be justifiably questioned.

114. In order to provide defensible predictions, a number of procedures were used in this study: loading analyses, mass-balance calculations, mathematical model simulations, and bioassays. Multiple approaches were employed because conclusions could be postulated with increased reliability if the various procedures provided equivalent predictions.

Existing and Predicted Nutrient Concentrations and Loadings in the Deep Fork River

Phosphorus

115. An anomaly in the phosphorus data for composite samples from Deep Fork River was detected. Apparently the magnitude of a sample value is strongly dependent upon the interval length represented by the

composite sample. The anomaly was observed not only for phosphorus but for all composite samples for which an interval length-concentration analysis was done: total nitrate, total dissolved solids, chloride, and sulfate. The relationship between interval length and concentration for phosphorus is presented in Table 7. Composite samples collected over intervals of 1 to 4 days (56 percent of the composite samples) have a mean value 53 percent less than the mean of the remaining composite samples collected over intervals ranging from 5 to 20 days. The large variability of sample values within intervals of a given length preclude the detection of differences between interval means. Therefore, one must conclude that all composite samples estimate the same quantity. Additional studies need to be conducted to determine if composite sampling is a valid procedure to characterize the water quality of natural waters.

116. The mean phosphorus concentration for grab samples over the period 1969-74 was 5.5 mg P/ℓ (n = 241). A paired T-test on samples collected at corresponding times revealed that the mean difference between grab samples (5.1 mg P/ℓ) and composite samples (5.6 mg P/ℓ) was not significantly different from zero ($P>0.4$, n = 32). Therefore, one must conclude that the two sample procedures estimate the same quantity and that the apparent differences between procedures represent the inherent variability of the phosphorus data.

117. The existing data record reveals considerable irregularity in the time elapsing between sampling for most water-quality constituents. For purposes of comparison between procedures, a weighted mean over the study period was calculated. Often the weighted mean differed slightly from the averages presented in Table 6 in which each sample value was weighted equally. Weighted means were calculated at a monthly resolution in which each month was weighted equally regardless of the number of observations made in a month. Weighted means over the period 1969-74 were calculated by weighting each year proportionally to the number of months in the year in which at least one observation was made.

118. Using this weighting procedure, the average composite phosphorus concentration over the interval 1969-74 was 5.0 mg P/ℓ, and the average phosphorus concentration estimated with grab sample values was 5.5 mg P/ℓ. Phosphorus concentrations decreased over the study

period but phosphorus loadings generally remained uniform over the period (Table 8).

119. Examination of phosphorus data near Arcadia compiled by agencies other than the USGS reveals similar concentrations and decreases over time (Table 9).

120. Average phosphorus concentration measured in the final effluent of the Northside STP was 1.33 mg P/ℓ (Table 5), considerably less than sample values measured near Arcadia. Since the final effluent is contributing considerably more flow than mass of phosphorus relative to the river discharge near Arcadia, calculations indicate that relocation of the STP will result in 64- to 66-percent increase in instream concentrations and a 2- to 26-percent reduction in loadings (Table 10). Previous sampling of the STP effluent showed concentrations varying between 0.27 and 45.7 mg/ℓ with an overall mean of 12 mg/ℓ (Table 2). If average phosphorus concentration in the effluent were 12 mg/ℓ, relocation of the plant would decrease loadings 68 to 77 percent. However, because the variability of the various phosphorus data precludes the detection of extremes in the data, the estimate of 1.33 mg/ℓ was used to calculate loading reductions due to STP relocation. Therefore, calculated reductions should be considered conservative.

Nitrogen

121. Both composite samples and grab samples were collected and analyzed for nitrogen. Composite samples were analyzed for total nitrate nitrogen. Presumedly, the total nitrate values reflect the total nitrogen content of the sample, the rationale being that the storage period of composite samples was of sufficient length for most nitrogen forms to be converted into nitrate. Grab samples were analyzed for total nitrate, nitrite, and ammonia plus organic nitrogen. The sum of the nitrogen forms provides an estimate of total nitrogen.

122. Composite nitrogen samples displayed anomalies similar to that of phosphorus; sample values were dependent upon the interval length represented by the sample (Table 11). Composite nitrogen samples collected over intervals of 1 to 4 days (54 percent of the composite samples) had a mean value 56 percent less than the mean of the remaining

composite samples collected over intervals ranging from 5 to 20 days. Large variability of sample values within intervals of a given length precluded the detection of differences between interval means.

123. Average total nitrogen concentration over the period 1969-74 estimated by grab samples was 12 mg N/ℓ (n = 43). Average nitrogen concentration estimated by composite nitrate nitrogen was 7.1 mg N/ℓ (n = 308). A comparison of composite and grab estimates of total nitrogen using the paired T-test with time of collection as the basis for pairing revealed that the mean difference between the two estimates is significantly different from zero ($P<0.001$, n = 42). Based on these results one may conclude that the two procedures estimate different quantities. However, a decision was not made as to which estimate was more reliable. Throughout the report, calculations were made and conclusions based on both estimates. The disparity of results based upon the two estimates provides an indication of the uncertainty of the predictions.

124. Loading estimates based upon composite samples averaged approximately 40 percent less than estimates based upon grab samples. Loading rates over the period 1969 to 1974 averaged 336,000 and 620,000 kg/yr for composite and grab samples, respectively (Table 12). Examination of Table 12 demonstrates a rather consistent decrease in composite nitrogen concentrations over the period of record while total grab sample nitrogen concentrations generally remained constant.

125. Comparison of data collected near Arcadia and compiled by agencies other than the USGS reveals a decrease with time of nitrate concentrations and a rather constant ammonia concentration (Table 9).

126. Average total nitrogen concentration measured in the final effluent of the Northside STP was 12.7 mg N/ℓ, which is greater than average composite nitrogen concentration but approximately equal to total nitrogen estimates based upon grab samples from the Deep Fork River near Arcadia. The calculated result of diverting the sewage effluent is summarized in Table 10. Composite sample concentrations are decreased 17 percent and loadings are decreased 54 percent. In contrast, grab sample concentrations increase 3 percent, while loadings decrease 28 percent.

Predicted nutrient loadings
using a land-use approach

127. The purpose of using a land-use approach in this study was to estimate how phosphorus and nitrogen loadings to the proposed Arcadia impoundment may change with time. Existing watershed models with water-quality subroutines have not been adequately developed and verified to be applied for predictive purposes with confidence unless rather extensive on-site calibration and verification are possible. In cases where an extensive effort is not possible, equally or perhaps more reliable predictions can be made from anticipated land-use changes. This required a calculation based on present land use, a comparison with present loadings estimated from concentration-discharge analyses, and a calculation of how the land use and loads may change with time.

128. A calculation of loadings based on land use requires a knowledge of how much land is used for a particular purpose within a basin. The loading from a particular use is considered to be equal to the area of land committed to that use times the amount of material that can be derived from runoff per unit of area. When these values are summed over all uses within a watershed, the result is an estimate of the total loading for the watershed.

129. The present land use for the Deep Fork River basin was determined using information provided in the General Design Memorandum for Arcadia Lake (Figure 1).[1] The categories considered were pasture, cultivated, wooded, urban, and open land. Actual urban land area was calculated as equal to percent urbanization times land area designated as urban. The remaining urban land area was designated open land.

With the exception of phosphorus export from urban land, phosphorus and nitrogen yields per unit area used in the calculations were based on values listed in a paper by Shannon and Brezonik.[28] The value used to determine the amount of phosphorus derived from urban activity was taken from a study of urban runoff in Tulsa, Oklahoma.[29] The export values used are tabulated in the yield columns of Tables 13 and 15.

130. Little consistency can be found in land-use nutrient-export

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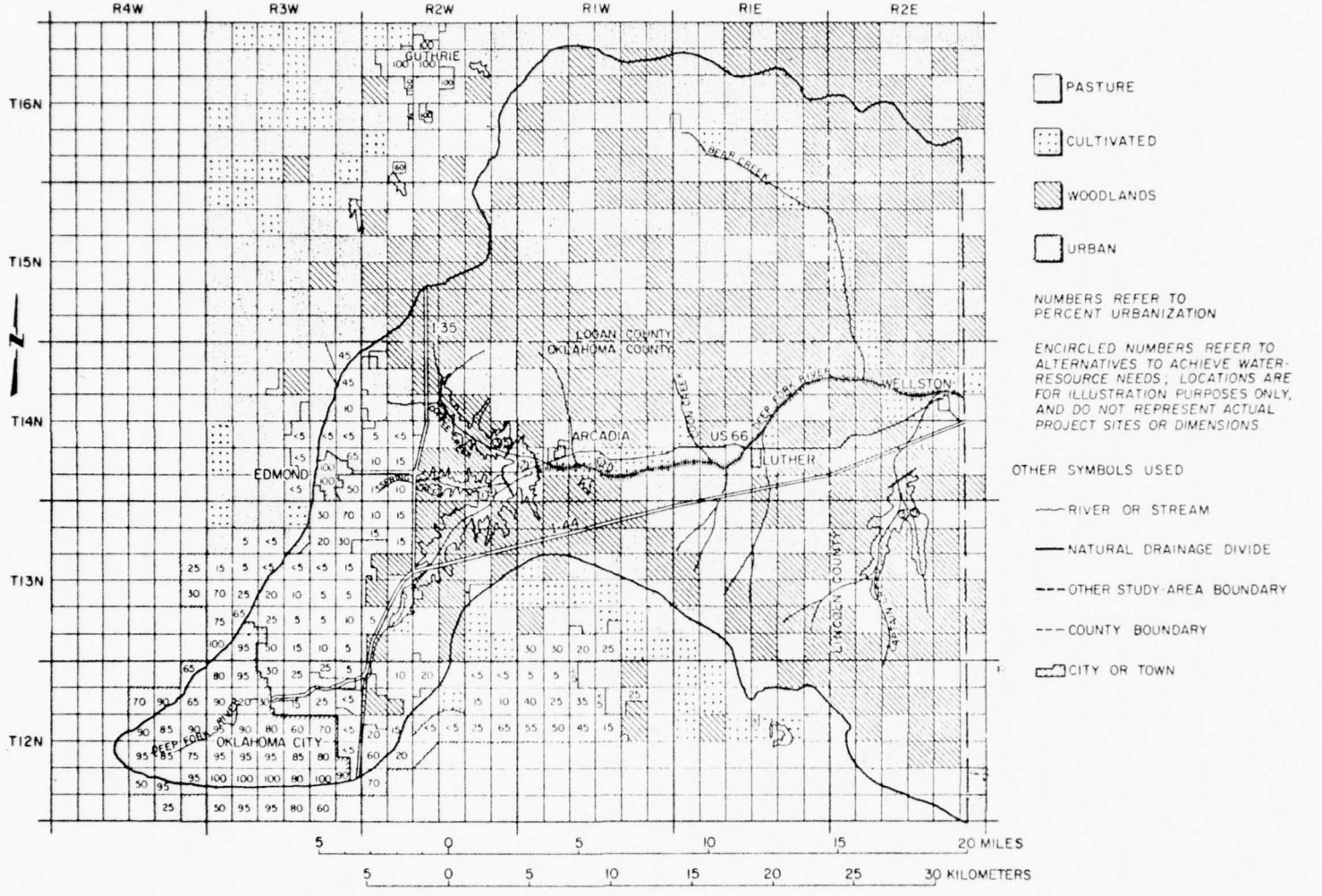


Figure 1. Detailed land use in the upper Deep Fork River basin [1]

relationships reported in the literature from one study to another. Examination of two of the more recent and comprehensive reviews, one by Loehr[30] and the second by the EPA,[31] indicates that little basis exists for selecting one yield value over another. The paucity of studies for the southern Great Plains makes selection based on similar climatological conditions questionable. Similarly, most studies were conducted in the eastern and northeastern United States, making hazardous the extrapolation of values obtained by averaging over separate study values. Time constraints for this study did not permit the field data collection required to develop land-use loading relationships for the Arcadia watershed.

131. Phosphorus loadings. The phosphorus loading at Arcadia calculated for the present land-use pattern is 143,000 kg/yr. Allowing for minor land-use changes due to reservoir construction and diversion of the domestic sewage effluent, the loading after impoundment would be reduced to 26,000 kg/yr (Table 13).

132. An estimated loading for the Year 2000 was calculated using the following assumptions. It was assumed that 80 percent of the present population in the Deep Fork basin is in the urban area with a density of 13 persons/ha. It was assumed that 80 percent of the projected population would also be located in the urban area and that the density would remain the same. Finally, it was assumed that the urban area would grow at the expense of the open area. Projected population in the Arcadia Lake watershed was obtained from the OSU water-quality study.[3] The Regional Land Use Plan for Central Oklahoma prepared by the ACOG provided little information that could be used to estimate land use or population distribution within the watershed.[32] The Regional Land Use Plan applies to an area of 751,400 ha and provides insufficient resolution to project conditions in the 27,200-ha Arcadia Lake watershed.

133. Based on the assumptions outlined above, the projected phosphorus loading to Arcadia Lake for the Year 2000 is 46,000 kg/yr. This value is approximately one-third of the present calculated phosphorus loading rate and double the estimated phosphorus loading rate following construction of a dam at Arcadia and diversion of the sewage treatment plant discharge.

135. Nitrogen loadings. The loading of nitrogen at Arcadia calculated from present land-use practices is 690,000 kg/yr. It is estimated that the loading would drop to 125,000 kg/yr following construction of the impoundment at Arcadia, largely due to sewage diversion, and then increase to 174,000 kg/yr by the Year 2000 (Table 14). This is essentially the same pattern predicted for phosphorus except that the phosphorus loading increased faster than the nitrogen loadings between impoundment and the Year 2000.

136. Comparison with measured loadings. The calculated loadings of nitrogen and phosphorus obtained using the land-use method were compared to loadings derived from flow and concentration data (Table 15). An examination of phosphorus data revealed that the land-use method provided loading estimates 2 to 10 times lower than flow-concentration estimates. The land-use estimate of the present (preconstruction) loading rate was 39 to 46 percent less than estimates calculated using flow-concentration data, while the land-use estimate of loading rate following impoundment construction was 85 to 90 percent less than flow-concentration predictions. The ranges are due to differences in estimates based on grab and composite samples. The greater disparity in estimates of loadings following impoundment construction may be due to the fact that the land-use calculations assume all sewage effluents were diverted from the watershed, while the flow-concentration calculations only assume the Northside STP effluent will be diverted from the watershed.

137. In contrast, land-use nitrogen load estimates are 12 to 105 percent greater than flow-concentration estimates based on preconstruction data and are 28 to 72 percent less following construction. The greater reduction in nitrogen loadings following sewage effluent diversion demonstrated by the land-use data may be interpreted similarly to that for phosphorus. Land-use calculations assume total diversion of all domestic sewage while flow-concentration calculations assume relocation of only the Northside STP.

138. The disparity of phosphorus loading rates calculated by the land-use and concentration-flow methods as well as the large difference in nitrogen loading rate estimates calculated using flow-concentrations

and based on grab and composite samples make uncertain the estimation of reliable nutrient loading rates. However, the land-use method provides an indication of trends to be expected following impoundment and demonstrates the significance of domestic sewage loading and urban land as a diffuse source of nutrients. Fortunately, the existence of independent estimates of disparate nutrient loading rates explicitly embodies the uncertainties associated with the evaluation of the eutrophication potential of Arcadia Lake and suggests cautious reliance upon the results of any single procedure used alone.

Evalution of Arcadia Lake Eutrophy Based Upon Loadings

Critical concentrations

139. Numerous attempts have been made to correlate algal productivity to mean nitrogen and phosphorus concentrations. These attempts were motivated by the observation that eutrophic waters are generally characterized by higher nutrient concentrations than oligotrophic waters. In 1947 Sawyer presented critical limits for nitrogen and phosphorus that are most often referenced when relating nutrient concentrations to expected trophic conditions.[33] Work on Wisconsin lakes led Sawyer to hypothesize that aquatic blooms are likely to develop in lakes during the summer when nitrogen and phosphorus concentrations exceed 0.3 and 0.01 mg/ℓ, respectively, during the ice-out period. Although not directly applicable for reasons to be discussed elsewhere in this section, the predicted average nitrogen and phosphorus concentrations in the Deep Fork River of 9 to 12 and 5 mg/ℓ, respectively, suggest that potential eutrophication problems may exist due to nutrient abundance.

Loading rates

140. Vollenweider's criterion. In an attempt to make the prediction of the eutrophic status of water bodies more precise, investigators have incorporated consideration of other characteristics of lakes in their predictive procedures. In 1968, Vollenweider published a comprehensive review of the eutrophication literature and presented a

procedure to evaluate the eutrophy of a lake based on lake mean depth and loading rates of nitrogen and phosphorus.[34] Until recently Vollenweider's work has been the most widely used procedure to predict the trophic status of water bodies based on nutrient loadings.

141. The basis for Vollenweider's procedure is that knowledge of the rate of supply of nutrients to lakes embodies more information relating to the essentially rate-limited process of algal productivity than knowledge of static concentrations of nutrients. Furthermore, specification of the mean depth of a lake implicitly provides general specification of other important lake characteristics such as sediment-to-volume ratios and possible nutrient transport limitations as might occur between the trophogenic and tropholytic zones of deep lakes.

142. Vollenweider's procedure to establish permissible loadings of phosphorus was to examine 30 lakes for which appropriate data were available, classify the lakes according to mean depth, and establish boundaries on loadings for lakes known to be eutrophic, oligotrophic, and transitional. The boundary delimiting eutrophic and transitional lakes was designated the dangerous loading level, and the boundary delimiting oligotrophic and transitional was designated the permissible loading level. Corresponding levels for nitrogen were derived by assuming a N/P ratio of 15:1 (by weight) and transforming phosphorus values to nitrogen equivalents.

143. In Vollenweider's procedure, surface units were selected as a reference basis with the specific annual loadings expressed in terms of grams per square metre of lake surface. Examination of Arcadia data revealed average annual phosphorus surface loadings of 18 and 20 g/m^2 for composite and grab samples, respectively, over the period 1969-70. Calculated phosphorus loadings over the same period, had the Northside STP not been discharging into the Deep Fork River, were estimated to be 14 and 20 g/m^2. The dangerous annual loading phosphorus level based on Vollenweider's procedure for Arcadia Lake is 0.15 g/m^2. Annual projected loadings for Arcadia Lake far exceed dangerous loading levels established by Vollenweider.

144. Average annual nitrogen loading rates to Arcadia are 26 and

48 g/m^2, respectively, for composite and grab samples prior to sewage treatment plant relocation and 12 and 36 g/m^2 following relocation. The corresponding dangerous loading level for nitrogen is 2.34 g/m^2. Nitrogen loadings at Arcadia are similar to phosphorus in that they greatly exceed dangerous levels with respect to the 1968 Vollenweider criterion.

145. Shannon and Brezonik's procedure. Shannon and Brezonik presented an interesting study of the relationship between lake trophic state and nitrogen and phosphorus loading rates.[28] Shannon and Brezonik measured seven indicators of trophic status in 55 lakes in north-central Florida over a 1-yr period. The indicators were primary production, chlorophyll-a, total phosphorus, total organic nitrogen, Secchi disk transparency, specific conductivity, and Pearsall's cation ratio (Na + K)/(Ca + Mg). Inverse transformations were made on Secchi disk transparency and Pearsall's cation ratio so that numerical increases in any of the seven variables would be in the direction of increased eutrophy. The first principal component extracted from the correlation matrix of the seven indicators was named the Trophic State Index (TSI). Comparison of the calculated TSI's with each of the corresponding 55 lakes indicated that eutrophic lakes had a TSI greater than 7 while oligotrophic lakes had a TSI less than 4.

146. Concurrently with the preceding analyses, Shannon and Brezonik estimated nitrogen and phosphorus loadings to each lake using a land-use procedure. A multiple-regression analysis of TSI on various functions of nitrogen and phosphorus loading rates revealed significant relationships.

147. Shannon and Brezonik then inserted a TSI value of 7 into one of the significant relations and solved for nitrogen and phosphorus loading rates subject to the constraint of a molar N/P ratio of 16:1 (7:1 by weight) assumed in algal cells. The resulting nitrogen and phosphorus loadings of 1.51 and 0.22 $g/m^3/yr$ for N and P, respectively, were hypothesized as critical loading levels. Similarly, the insertion of a TSI value of 4 resulted in the nitrogen and phosphorus loadings of 0.86 and 0.12 $g/m^3/yr$, for N and P, termed permissible.

148. Volumetric loading rates calculated for Arcadia Lake by

various procedures for different conditions are tabulated in Table 16. Phosphorus loading rates calculated using USGS data collected near the proposed damsite exceed the Shannon and Brezonik critical levels both prior to and after construction by an order of magnitude. In contrast, estimated nitrogen loadings after impoundment construction are greater than the critical limit by a factor of 1.2 based on composite samples and 3.4 based on grab samples.

149. More meaningful is the comparison of predicted Arcadia loadings based on the land-use method. Shannon and Brezonik used an equivalent procedure to derive their critical limits. The yield factors used by Shannon and Brezonik and in the calculation of Arcadia loadings were the same except for phosphorus yields from urban areas. Arcadia calculations used a value of 2.8 kg/ha/yr while the Florida lakes study used a value of 1.1 kg/ha/yr.

150. Predicted preconstruction land-use loadings at Arcadia exceed the critical limit, and postconstruction loadings are near the limit. Phosphorus loadings exceed postulated critical levels by a factor of 1.4, and nitrogen loadings are equal to the limit. By the Year 2000, nitrogen and phosphorus loadings exceed the critical limit by factors of 1.3 and 2.5, respectively.

151. Dillon's model. Dillon has presented a phosphorus loading rate procedure relating trophic status to an additional variable reflecting the flushing rate of a lake (reciprocal of hydraulic residence time).[35] Dillon essentially modified observed phosphorus loadings by multiplying by a factor equal to the product of hydraulic residence time and the fraction of phosphorus not retained in the lake. A plot of modified phosphorus loading rates against mean depth of lakes of known trophic status provided a separation of eutrophic, transitional, and oligotrophic lakes.

152. *A priori*, it can be concluded that application of Dillon's procedure would result in conclusions similar to Vollenweider's approach. Thirteen of 27 lakes used by Dillon to establish permissible loading rates were used by Vollenweider. In addition, 10 of the 13 lakes had theoretical residence times nearly equal to that of Arcadia. For this

reason and because of the necessity to estimate phosphorus retention in order to apply the approach, Dillon's procedure was not directly applied to Arcadia Lake.

Evaluation and summary

153. Several serious deficiencies exist in the application of Sawyer's critical limits to impoundments generally and to Arcadia specifically. Critical concentration limits apply to concentrations in the water column observed prior to the vegetative growth period in the spring. Nutrient measurements near Arcadia pertain to tributary concentrations that provide a very obtuse indication of concentrations to be expected in the water column. Losses to the sediments through chemical precipitation, sedimentation, and other processes would decrease water column concentrations to values lower than the tributaries.

154. Sawyer's critical limits and Vollenweider's and Dillon's loading rate procedures were hypothesized based on observations on natural lakes. Natural lakes generally have longer hydraulic residence times relative to man-made impoundments that may completely exchange water masses in a few days during major storm events. For both mainstem and upstream reservoirs, water circulation and withdrawal characteristics are generally different from natural lakes. Lakes overflow from the surface or through seepage, while withdrawal from reservoirs is often from below the surface. Another difference between reservoirs and natural lakes is the maturity of sediments resulting in differences in sediment-water column nutrient exchange. In the case of Arcadia, two additional factors that make extrapolation quite indefensible are the great differences in climatological conditions and watershed characteristics between lakes used to develop the available methods of analysis and the Deep Fork River basin.

155. Both the critical concentration concept and loading rate analyses imply nutrient limitation. Unfortunately, previous measurements of nutrients near Arcadia failed to consider the chemical form of the potential nutrients. All chemical forms are not equally available to algae, and nutrients cannot be stimulatory if they are not in a form that can be used. Furthermore, neither procedure considers light or

other factors that may dominantly control algal productivity in Arcadia Lake.

156. In summary, the application of Sawyer's critical concentration concept and Vollenweider's loading rate procedure indicates that a potential of eutrophication exists. Loading rate analysis using the Shannon and Brezonik procedure suggests that while nitrogen and phosphorus loadings immediately following construction would be marginal, loadings prior to construction and by Year 2000 may be sufficient to stimulate nuisance algal blooms. However, the marginal loadings were based on the assumption that all domestic sewage would be diverted from the Arcadia Lake watershed.

157. Loading rate analyses used alone are inadequate to evaluate properly the eutrophication potential of Arcadia Lake since these procedures were developed to characterize natural lakes much different hydraulically, geographically, and in watershed characteristics from the Arcadia impoundment.

Mass-balance calculations

158. The TD requested that WES prepare frequency duration curves based on daily concentrations calculated using daily flow data and regression equations relating flow and concentrations.

159. An examination of plots of concentration versus discharge for all the critical water-quality constituents indicated that for at least some of the parameters, a relationship between discharge and concentration could be described using statistical models. Using the models, daily concentrations were generated over the period 1 January 1938 through 31 December 1970 for measured or synthetic flow records at Arcadia. Flow and mass routings through the reservoir were made; simulated reservoir concentrations were recorded; and duration curves of daily concentrations were plotted.

160. Procedures and assumptions. The following models were fit to concentration data modified to account for relocation of the Northside STP.

$$\text{Model I:} \quad C = a + b \frac{1}{\log_e (D + 1)}$$

$$\text{Model II:} \quad C = a \exp \frac{b}{D + 1} - 1$$

$$\text{Model III:} \quad C = \frac{1}{a + b \log_e (D + 1)} - 1$$

$$\text{Model IV:} \quad C = a + bD$$

$$\text{Model V:} \quad C = a + \frac{b}{D + 1}$$

where:

C = concentration, mg/ℓ or µg/ℓ

D = discharge, m^3/sec

a,b = coefficients

The constant 1 was included to preclude calculating logarithms or reciprocals of zero.

161. For each constituent, appropriate logarithmic transformations were made when necessary to linearize the hypothesized relationship. Pertinent statistics were computed using the ordinary least-squares procedure. The best model for a particular constituent was selected on the basis of the maximum coefficient of determination. After transformation, residuals were assumed normally and homoscedastically distributed in order to make probability statements about the model. A model was assumed satisfactory if the hypothesis of zero coefficients was found untenable using the F-test with a critical region of size 0.05. Only those constituents whose best model was found satisfactory with respect to this criterion were simulated using mass-balance calculations.

162. Predicted concentrations were generated by using the relationship implied by the model. Such predictions are expected mean values and do not reflect the uncertainty demonstrated by the scatter of points around the regression curve.

163. Flow and mass routings through Arcadia Lake were conducted under the assumptions:

a. The lake is thoroughly and instantaneously mixed.

b. The only source of material is through tributary inflow.

c. The sources of water to the lake are tributary inflow and direct precipitation on the surface.

d. The losses of water from the lake are downstream releases, water supply withdrawals, and evaporation.

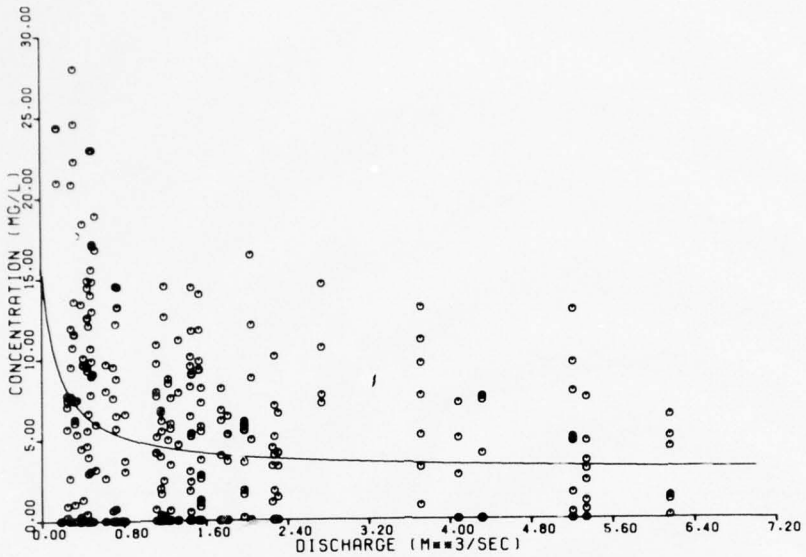
164. The rule curve employed for lake operation was to maintain lake elevations at the top of the conservation pool except when minimum releases and withdrawals exceeded tributary inflow or when flood storage releases exceeded downstream channel capacity.

165. For each constituent considered, simulations were conducted to evaluate the significance of degradation and sedimentation. The first set of simulations was conducted under the assumption of conservative behavior. The second set of simulations incorporated decay where appropriate. The third set of simulations provided for a 30-percent loss to the sediments over the average theoretical residence time of 1.8 yr; the fourth set of simulations provided for a 60-percent loss to the sediments. The estimates of 30- and 60-percent loss to sediments were arbitrary. The range should include expected losses of phosphorus[36] and provide conservative estimates of pesticide losses. The validity of these assumptions is discussed elsewhere in the report as the individual constituents are considered.

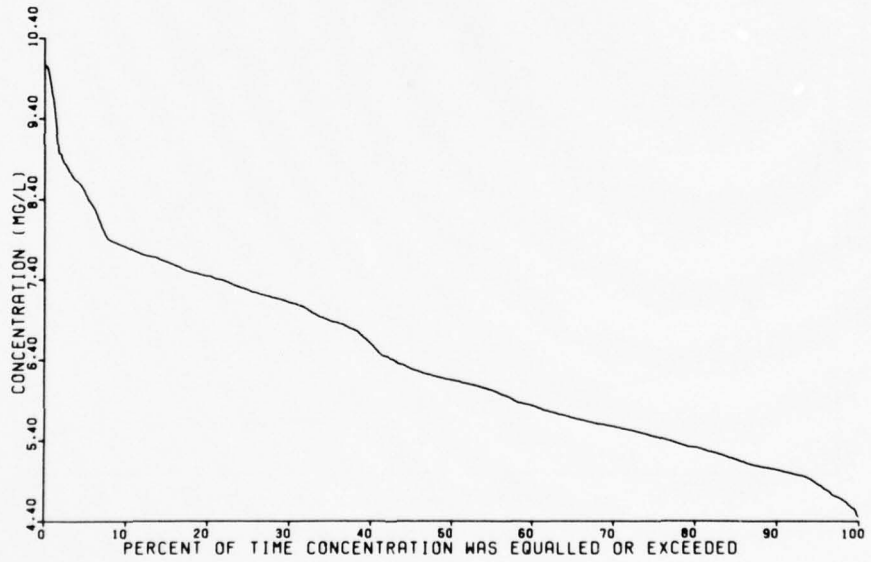
166. Results. A tabulation of parameters for which models could reasonably estimate concentrations given discharge is presented in Table 17. The coefficient of determination R^2 may be interpreted as the fraction of the total variance of concentration observations accounted for by the model. The large residual variance suggests that expected mean concentrations may be misleading. The expected range of concentrations at a given discharge is large based on existing data.

167. An examination of daily concentration-duration curves for the nutrients nitrate (Figure 2), phosphate (Figures 3 and 4), and ammonia-organic-nitrogen (Figure 5) reveals that median simulated concentrations assuming conservative behavior are within 20 percent of average expected stream concentrations (Table 10).

168. Simulations conducted under the assumption of 30- and 60-percent losses of nutrients to the sediments demonstrate a decrease in impoundment concentrations. Nitrate concentrations decrease 35 and 55 percent with 30- and 60-percent loss, respectively. Phosphate and

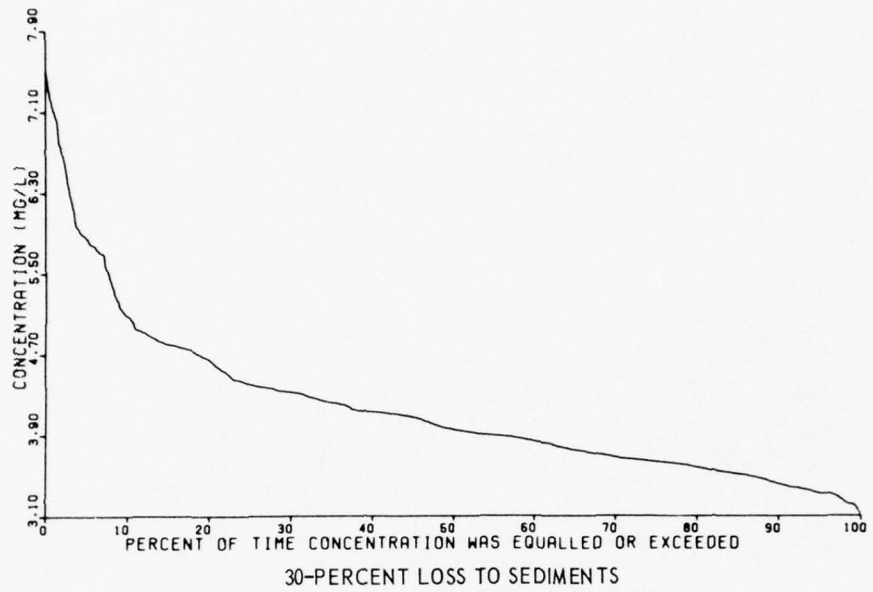


CONCENTRATION-DISCHARGE RELATIONSHIP



CONSERVATIVE BEHAVIOR

Figure 2. Nitrate duration curves based on daily concentrations, from composite NO_3-N samples



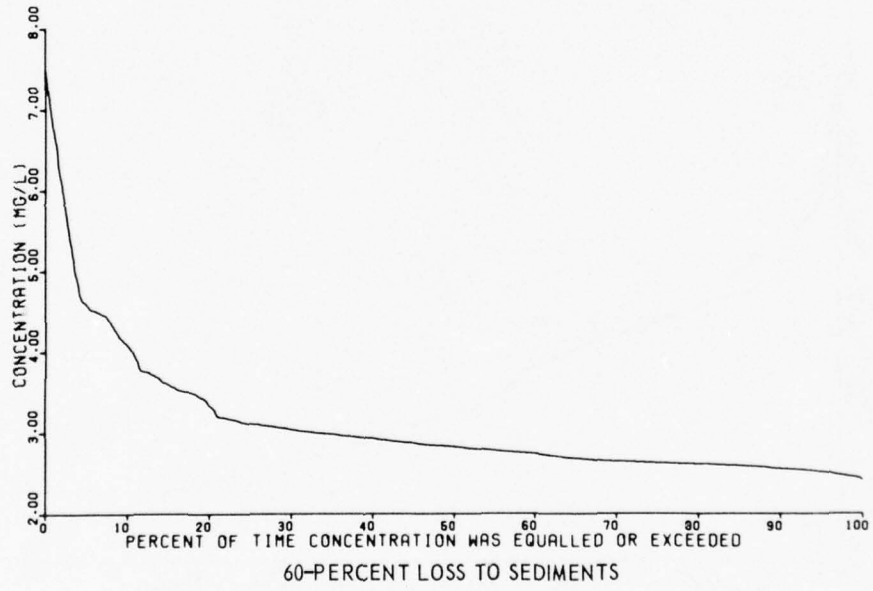
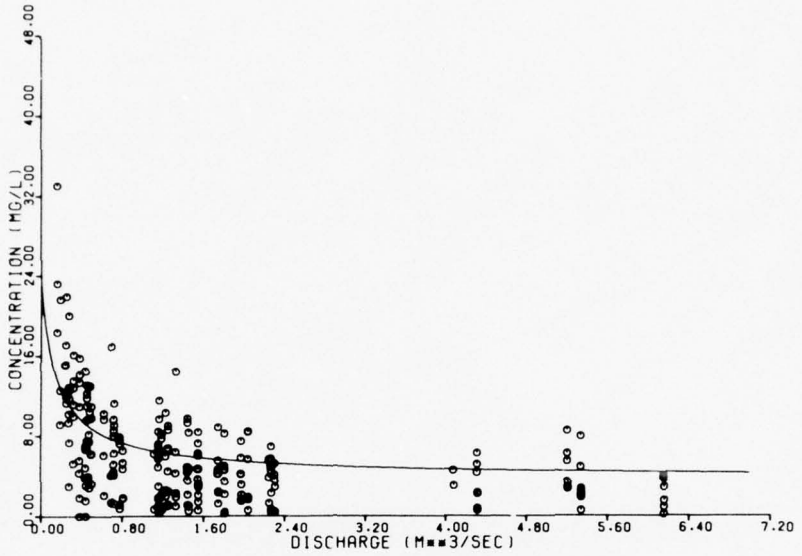


Figure 2. (Concluded)



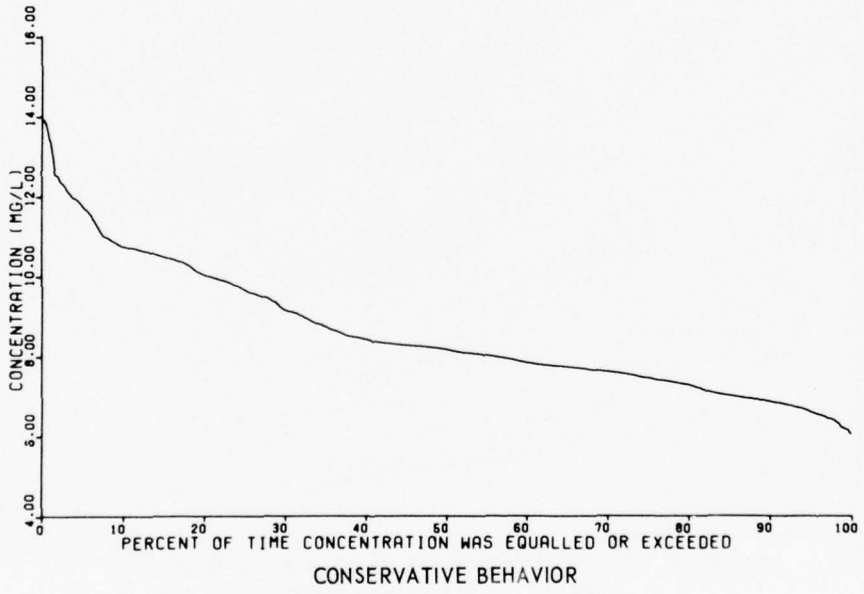
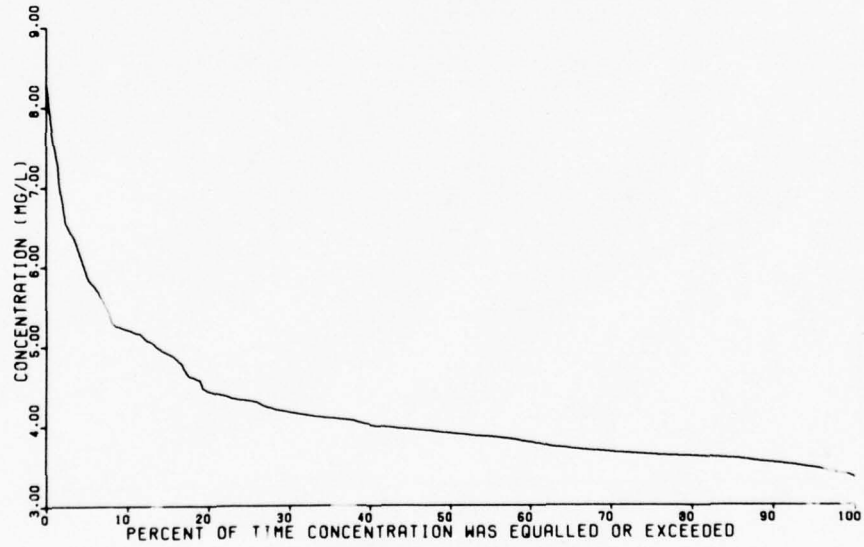


Figure 3. Phosphorus duration curves based on daily concentrations, from composite PO_4-P samples

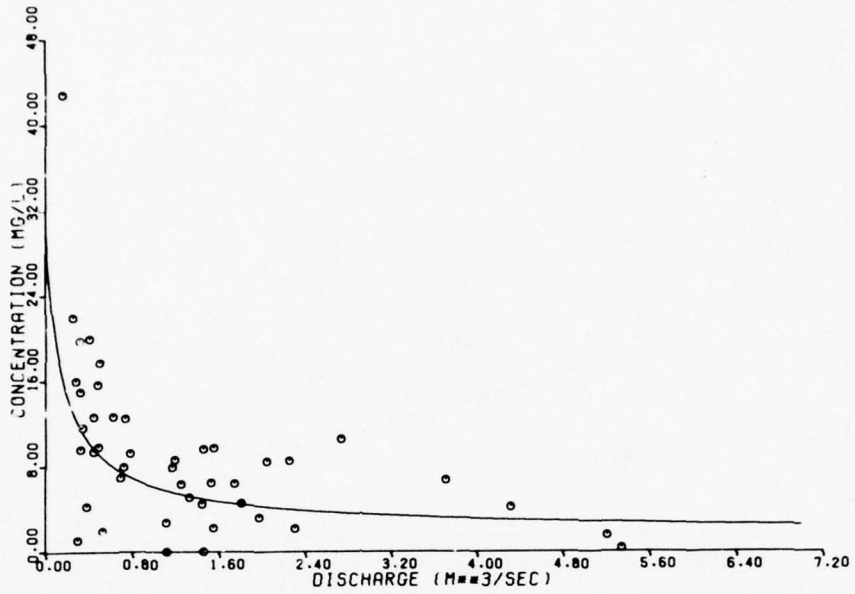


30-PERCENT LOSS TO SEDIMENTS

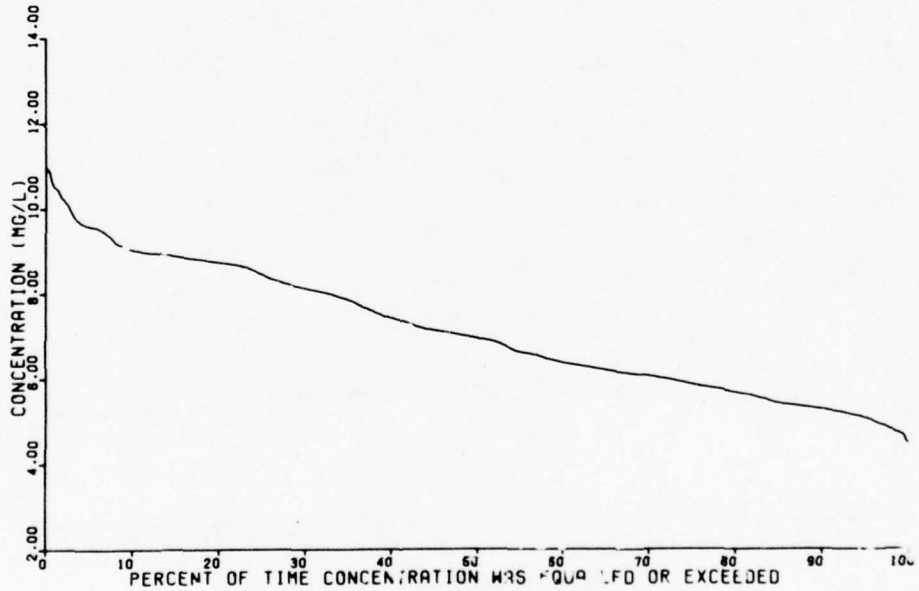


60-PERCENT LOSS TO SEDIMENTS

Figure 3. (Concluded)

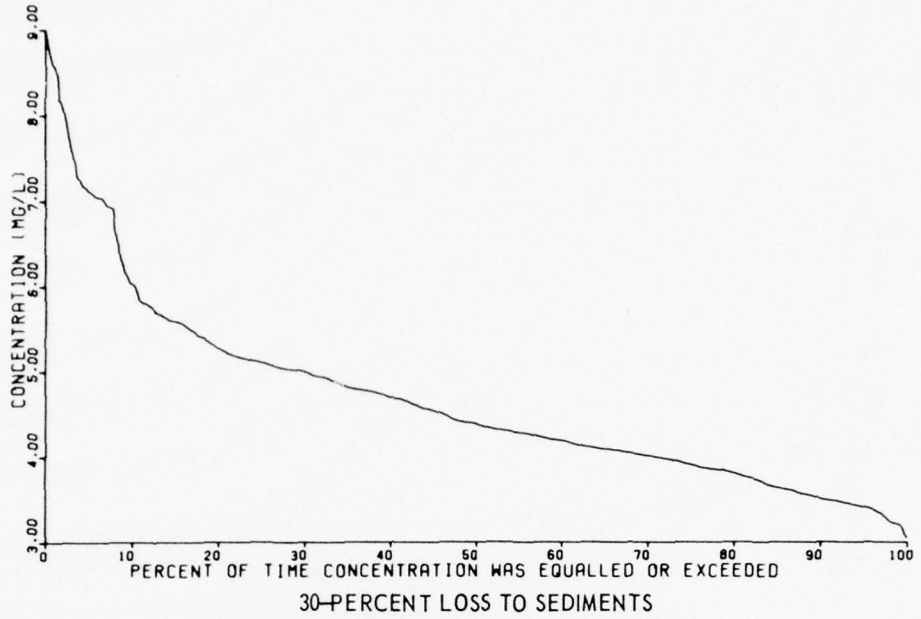


CONCENTRATION-DISCHARGE RELATIONSHIP



CONSERVATIVE BEHAVIOR

Figure 4. Phosphorus duration curves based on daily concentrations, from grab PO_4-P samples



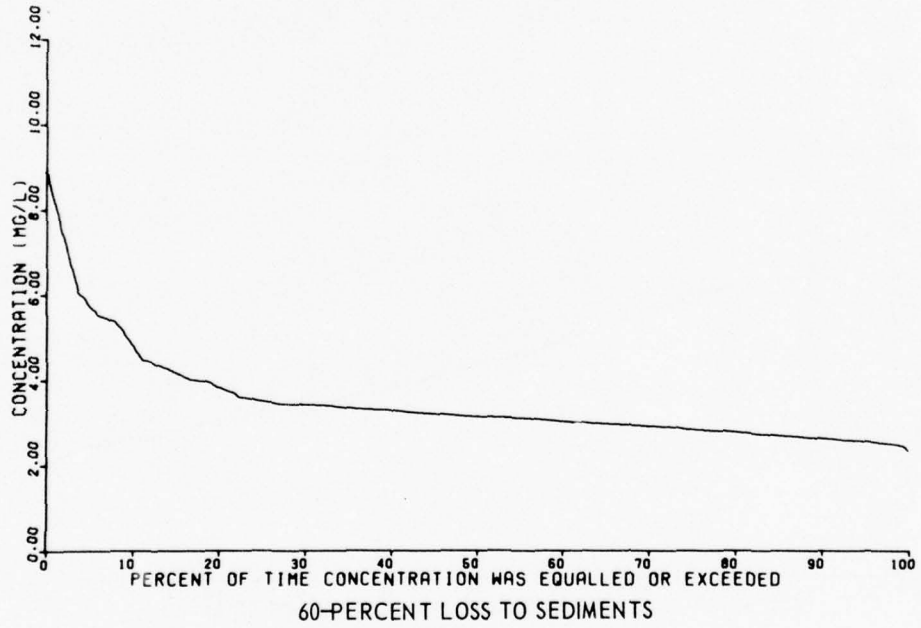
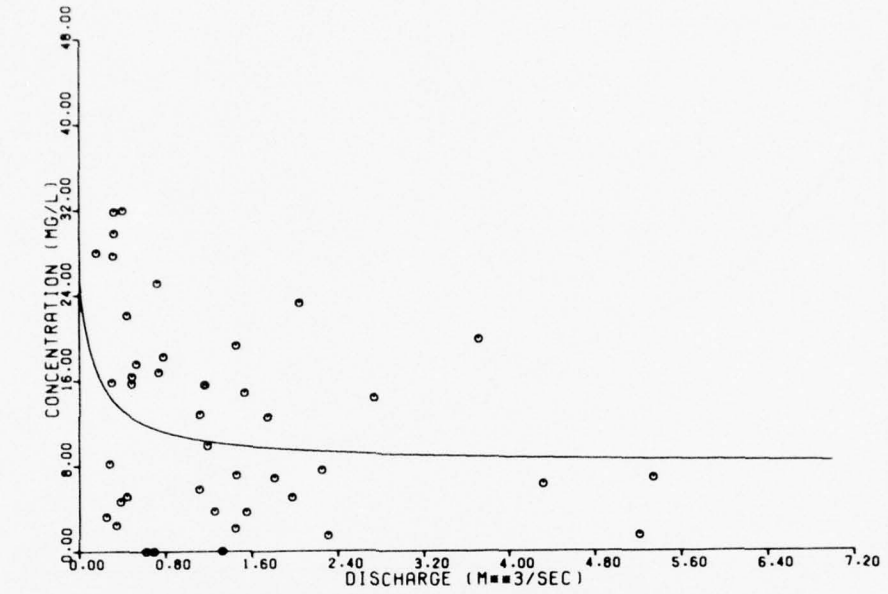
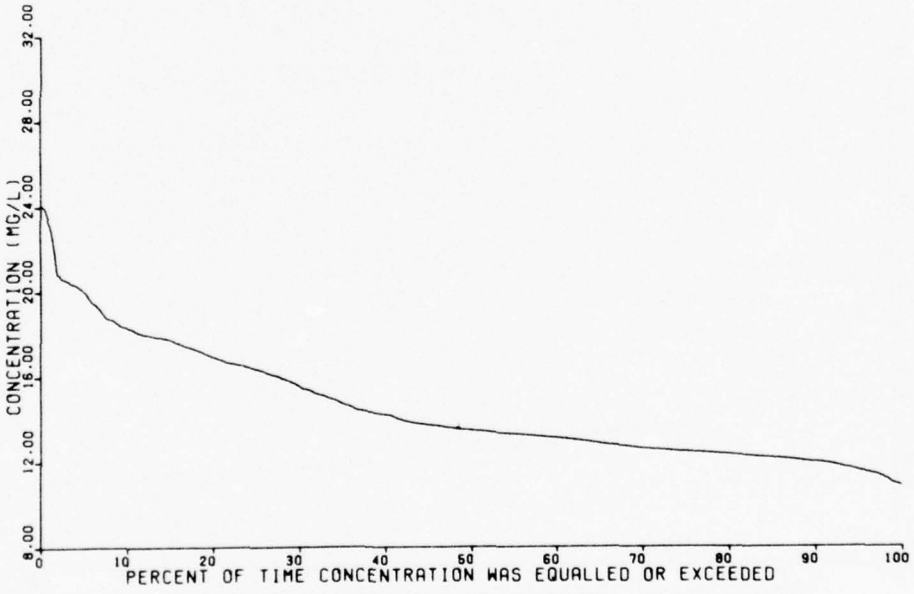


Figure 4. (Concluded)

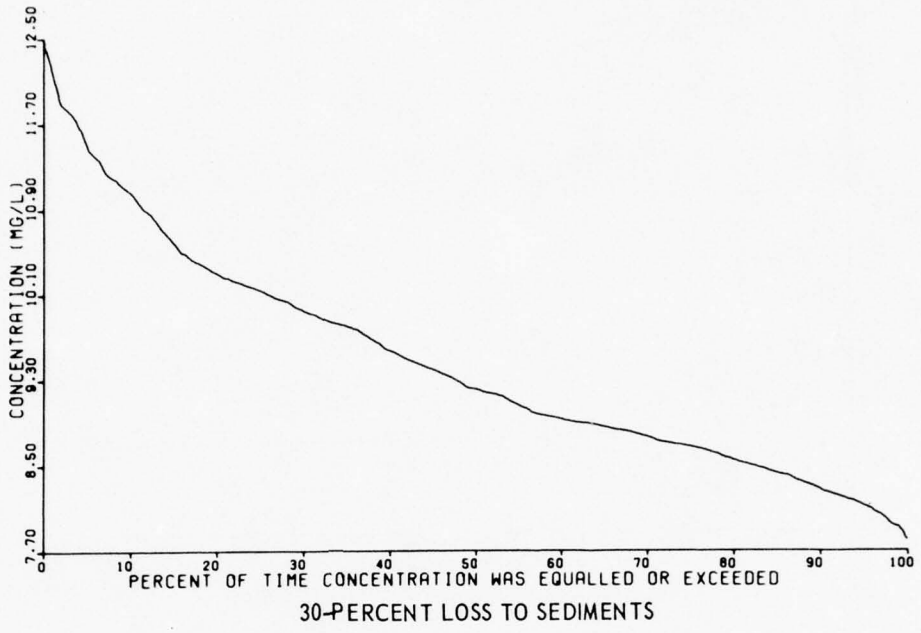


CONCENTRATION-DISCHARGE RELATIONSHIP



CONSERVATIVE BEHAVIOR

Figure 5. Ammonia-organic-nitrogen duration curves based on daily concentrations



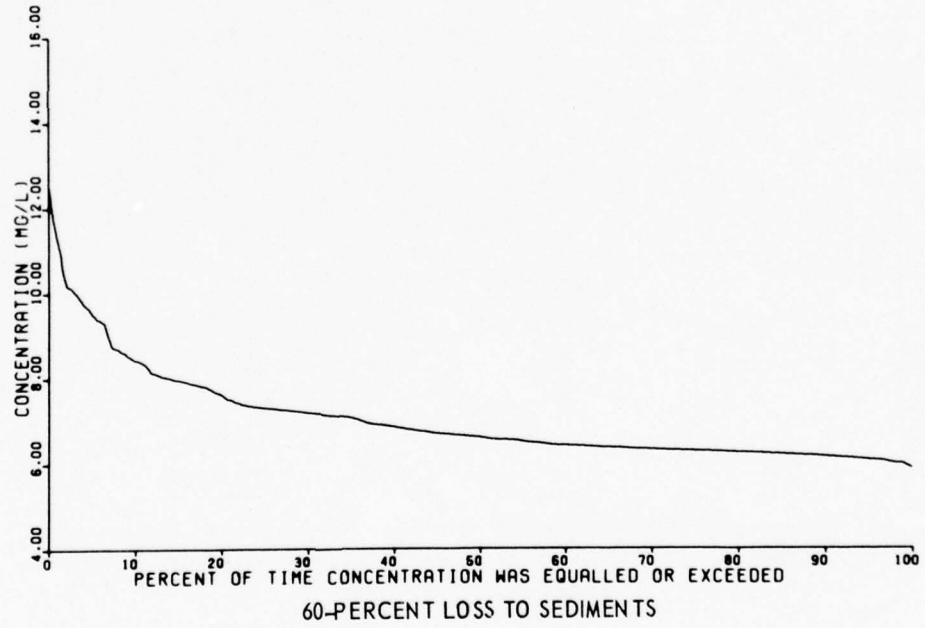


Figure 5. (Concluded)

ammonia-organic-nitrogen concentrations demonstrate similar patterns with nearly equivalent reductions.

169. Phosphorus retention in lakes has been estimated to vary between 20 and 50 percent, increasing with the hydraulic residence time.[36] For phosphorus, mass-balance calculations provide an indication of the trend to be expected following impoundment. Reservoir concentrations are generally less than tributary concentrations. However, for nitrogen significant exchanges may occur between the lake and atmosphere through denitrification and nitrogen fixation. These processes were not included in the mass-balance calculations.

Results of Algal Bioassay Studies on the Deep Fork River and Nearby Waters

Introduction

170. Based on preliminary analyses there was concern that excessive algal populations would occur in Arcadia Lake because predicted concentrations of nitrogen and phosphorus exceed levels generally accepted as sufficient to promote nuisance algal blooms. This situation does not necessarily mean that nuisance algal blooms would occur in Arcadia Lake because algal utilization of the nutrients would be dependent on availability (chemical form) and relative abundance of the nutrients. However, the presence of high nutrient concentrations indicates a potential problem that deserves careful consideration. The purposes of this phase of the water-quality evaluation were to assess the biological availability of nitrogen and phosphorus compounds and to assess the limiting nutrient(s) in water samples collected from the Deep Fork River and surrounding waters. The results were used to assess the significance of nitrogen and phosphorus entering the proposed impoundment and to evaluate the use of nutrient concentrations to predict algal growth problems.

Sampling and procedures

171. The first set of samples collected during this study consisted of five 4-ℓ grab samples collected from Lake Eufaula (Highway 69 bridge), Deep Fork River at the Arcadia gaging station (Post Road

bridge), Deep Fork River at the Eastern Avenue bridge, Lake Thunderbird at the Alemeda Avenue bridge (old Highway 77), and Little River at the Franklin Street bridge. These samples were returned to the laboratory at the University of Oklahoma, Norman, and filtered through 0.45-μ pore size Millipore filters prior to use in the nutrient availability study. The experiment was set up as indicated in Table 18. Each flask contained 800 ml of water to be tested and was inoculated with 2 ml of a stock *Selenastrum capricornutum* suspension. The standards were prepared in Algal Assay Procedure-Phosphorus (AAP-P) or AAP-Nitrogen (AAP-N) media depending on the stock algal culture to be added. The phosphate and nitrogen concentrations used in the standard cultures were selected because an evaluation of USGS data suggested average concentrations in the Deep Fork River were approximately 5 mg P/ℓ and 12 mg N/ℓ. The Lake Eufaula and Little River samples were prepared in duplicate because of difficulties in filtering the volume of water necessary. All other cultures were prepared in triplicate.

172. Laboratory cultured *Selenastrum capricornutum* was used as the test organism in this experiment. One stock culture was grown in phosphorus-deficient AAP medium and was transferred once a week for 4 weeks prior to actual testing of samples from the Arcadia area. This was done to obtain a viable inoculum that was sensitive to phosphorus. A second stock culture was grown in nitrogen-deficient AAP medium to obtain a viable inoculum sensitive to nitrogen. These organisms are referred to as P-deficient or N-deficient *Selenastrum* in Table 18.

173. A second set of grab samples collected in this study consisted of 4-ℓ samples obtained at the Arcadia and Eastern Avenue stations. These samples were returned to the laboratory for use in the nutrient spiking study using procedures described by Plumb[37] and Plumb and Lee.[38] The initial rate of carbon fixation was determined on the sample using carbon-14 methodology. Subsamples were then cultured in the laboratory after known nutrient additions. Following a 2-week incubation period, the rate of carbon fixation was redetermined to identify the additions that had an effect on the natural algal population. This experiment considered the possible effects of phosphate,

nitrate-nitrogen, ammonia-nitrogen, carbonate, trace metals, and complete AAP medium and is summarized in Table 19.

174. The procedure used to follow the response of algae during the nutrient availability and nutrient spiking experiments was carbon-14 incorporations. On selected sample days, six 25-ml subsamples from a culture received 5 μ C from a basic sodium bicarbonate solution (5 μ C/ml). Four replicates were incubated under a light intensity of approximately 4,300 lux for a 4-hr period. The remaining two samples were incubated in the dark during the same 4-hr period. Following incubation, the samples were filtered through presoaked 0.45-μ pore size Millipore filters using a pressure differential of 15 to 20 cm of mercury. The filters were rinsed with five washes of water and one wash of 1-percent HCl, exposed to HCl fumes in a desiccator for approximately 30 min, and then dried in a desiccator.[37,39] The filters were then dissolved in a dioxane-based scintillation cocktail containing naphthalene PPO, dimethyl POPOP, cellosolve, and water[37] and counted on a Beckman Scintillation Spectrophotometer. The algal activity in the sample was then calculated as the average light bottle uptake minus the average dark bottle uptake. More specific details of the procedures can be found elsewhere.[37,38,39,40]

Results

175. The results of the phosphorus portion of the availability study are presented in Table 20. The replicate data were averaged and the net carbon fixation (average light bottle activity minus average dark bottle activity) are presented as the log of the amount of radioactive carbon, expressed in counts per minute, incorporated by the *Selenastrum* cultures during the 4-hr incubation period. The data from the standards indicate that the test population was viable and would respond to phosphorus additions. However, a good response curve was not obtained because of the elevated phosphate concentrations used (high phosphate concentrations of 0-1.4 mg PO_4-P/ℓ were used to cover the range of concentrations that was expected in the Deep Fork River based on USGS data).[27] Nevertheless, the results obtained with the standard cultures are useful because they demonstrate that the continual addition of a nutrient does not mean *a priori* that a proportionate algal response will occur.

176. The rate of carbon-14 fixation in the Eastern Avenue sample continually decreased between day 3 and day 14 (Table 20). The final response was equivalent to approximately 0.09 mg PO_4-P/ℓ. Since the addition of phosphate to the Eastern Avenue sample did not produce a proportionate response, this would suggest that phosphate is not a critical element in that reach of the river.

177. The response of the P-deficient *Selenastrum* inoculum to the Arcadia water sample was equivalent to an available phosphorus concentration of 0.30 mg PO_4-P/ℓ or 3.3 times the Eastern Avenue response (Table 20). This is undoubtedly due to the Northside STP contribution below Eastern Avenue. The fact that the test population responded to phosphorus additions to Arcadia site water would indicate that the water was nontoxic to the organism and that phosphorus is a critical nutrient.

178. *Selenastrum* originally cultured in P-deficient AAP medium produced a lower response to filtered Lake Eufaula water compared to Arcadia water. This would suggest that part of the available phosphorus at Arcadia is lost in the Deep Fork River and Lake Eufaula. The mechanisms for this loss are probably sorption and precipitation and possibly algal utilization. An important point with respect to the Arcadia data evaluation is that the available phosphorus in the lake (Eufaula) is lower than the tributary (Deep Fork). This trend was also observed in the Little River-Lake Thunderbird data sets and can probably be expected in the proposed impoundment.

179. Nitrogen standards were prepared at 0 to 21 mg N/ℓ to cover the concentration range expected based on an evaluation of USGS data.[27] However, essentially the same response was obtained from the 5-, 10-, and 21-mg N/ℓ cultures. While this makes it impossible to prepare a concentration-response curve, it again demonstrates that algae will not necessarily be stimulated by the continual addition of a nutrient.

180. The response of nitrogen-sensitized *Selenastrum* to filtered Deep Fork River water collected at Arcadia was equivalent to approximately 5 mg N/ℓ (Tables 21 and 22). This was the greatest response observed in this portion of the study and is undoubtedly due to the proximity of the STP effluent. Another similarity to the phosphorus

portion of the study is that lower concentrations of available nitrogen were present in the lakes (Eufaula and Thunderbird) than in the tributaries to the lakes (Deep Fork and Little River).

181. The response to ammonia additions (Table 22) showed the same trends and was the same order of magnitude as nitrate additions (Table 21). Since the *Selenastrum* cultures responded similarly to nitrate and ammonia additions at the same nitrogen concentrations, it is not possible to state whether *Selenastrum* was using ammonia-N or nitrate-N from the water samples.

182. A nutrient spiking study was conducted to determine the limiting algal nutrient. Results obtained with Deep Fork River water collected at Arcadia and Eastern Avenue are presented in Table 23. The greater rate of carbon-14 fixation in the initial Arcadia sample would indicate a higher standing algal population, which agrees with the greater availability of nitrogen and phosphorus at the Arcadia site compared to Eastern Avenue. It is interesting to note that the incubation of these samples without additional nutrients produced an increased rate of carbon-14 fixation (raw water vs. initial water, Table 23). This would strongly indicate that nutrient concentrations are not regulating the algal population in the Deep Fork River but that some other factor, quite probably light, is more important.

183. All nutrient additions increased the rate of carbon-14 incorporation above that measured for the initial sample collected at the Arcadia gage site (Table 23). However, only nitrate, ammonia, and complete AAP media increased the rate of carbon-14 incorporation above that for the incubated water sample without additions (raw water). This suggests that nitrogen is the limiting nutrient in the Deep Fork River near Arcadia at the present time and that phosphorus, carbon, and heavy metal additions would not be stimulatory to the natural algae population.

184. All nutrient additions also increased the rate of carbon-14 incorporation above the initial Eastern Avenue sample, but only phosphorus and AAP media significantly increased the rate of carbon-14 incorporation above the raw water samples (Table 23). Phosphorus produced an 86-fold increase and AAP media produced a 7-fold increase. The fact

that the natural population responded to AAP media demonstrates that the population is viable. The response to phosphate but not nitrate, ammonia, carbon, or metals suggests that phosphorus is the limiting nutrient in the upper Deep Fork River.

Discussion

185. The Deep Fork River contains phosphorus and nitrogen concentrations in excess of levels generally considered necessary to support nuisance algal populations. The availability study using *Selenastrum* was run to determine how much of the phosphorus and nitrogen is in a chemical form that is readily available to algae. The results indicate that available phosphorus at Arcadia is approximately 0.30 mg PO_4-P/ℓ and that available phosphorus at other sampling stations ranged from 0.09 to 0.24 mg PO_4-P/ℓ (Table 20). Available nitrogen at the Arcadia sampling station was equivalent to more than 5.2 mg/ℓ as NO_3^--N or 4.9 mg/ℓ as NH_3-N. These values compare to 0.6 to 2.5 mg NO_3^--N/ℓ (Table 22) and 0.5 to 2.1 mg NH_3-N/ℓ (Table 22) that were determined for other sampling locations in the study. It must be cautioned that these results are based on a single experiment, but they do suggest that available nutrient concentrations are lower than the average phosphorus and nitrogen concentrations, 5.5 mg/ℓ and 12.4 mg/ℓ, respectively, based on 5 yr of USGS data at Arcadia.

186. Another fact pertinent to evaluating the importance of nutrient concentrations in the Deep Fork River is obtainable from the standard culture data in the availability experiment. Phosphorus concentrations of 0.92 mg PO_4-P/ℓ and 1.85 mg PO_4-P/ℓ resulted in the same response as 0.46 mg PO_4-P/ℓ. Also, 10.5 mg N/ℓ and 21 mg N/ℓ, as either nitrate or ammonia, resulted in essentially the same algal response as 5.2 mg N/ℓ. This demonstrates that increased concentrations of nutrients will not always produce a corresponding increase in the algae population. This is caused by the fact that algae require a number of nutrients and only the nutrient or other environmental requirements present in lowest supply relative to need will stimulate an algal response. Thus, the algal response to nutrients in the Deep Fork River will not be determined by the presence or total concentration of the nutrients in the river but

by the nutritional status of the algae and physical factors such as temperature and light.

187. In order to assess the nutritional status of algae in the Deep Fork River, a nutrient spiking study was conducted with samples collected at the Eastern Avenue bridge and the Arcadia gaging station. Initial carbon fixation rates for these two samples would suggest a higher algal population at Arcadia than Eastern Avenue, which is in agreement with the fact that a higher concentration of available phosphorus and nitrogen occur at Arcadia. However, the ratios for available phosphorus and nitrogen at the Arcadia and Eastern Avenue sampling stations were 3.3 and 9.7, respectively, compared to the algal activity ratio that was determined to be 2.2. Since the algal activity ratio was less than the calculated available nutrient ratio, this would suggest that the nutrients are not being completely utilized and that some other factor is more important as a regulator of the algae population. Because of the fact that the rate of carbon-14 fixation increased when samples without nutrient addition were incubated in the laboratory, this would indicate that light is probably the other factor. Certainly, the results demonstrate that available nutrients are present in the Deep Fork River that are not being used.

188. Nutrient spiking results with the Arcadia water sample show that the algae population only responded to nitrate and ammonia additions. This would be interpreted to mean that if light, temperature, and other physical factors were satisfactory, then nitrogen would be the limiting nutrient and that the other potential nutrients tested (phosphate, carbon, and trace metals) would not control the algal population in the Deep Fork River at Arcadia. Nutrient spiking results with the eastern Avenue sample show that phosphorus is more important than nitrogen in this reach of the river. If an impoundment is constructed at Arcadia and if the Northside STP effluent is diverted, it would be expected that influent water to the impoundment would be more similar to existing conditions in the river at Eastern Avenue that at Arcadia. Based on the nutrient spiking results, the diversion would have the effect of changing the element present in least supply relative to

need from nitrogen to phosphorus. The results also suggest that a physical factor, probably light, would be more important than nutrient concentrations in controlling the algal population in the proposed impoundment.

189. Results from the availability study suggest that available nutrient concentrations in the proposed Arcadia impoundment would be lower than the concentration in the influent water. This is based on the fact that available phosphorus and available nitrogen as determined by a *Selenastrum* bioassay were lower in Lake Eufaula than the Deep Fork tributary and lower in Lake Thunderbird than in the Little River tributary. This behavior would also be anticipated based on chemical data showing a general reduction in concentration in impoundments relative to influent concentrations. Discussion of the chemical data is presented elsewhere in the report.

190. An inconsistency occurred in the results of this study. The availability study suggested that nitrogen is limiting at Eastern Avenue and phosphorus is limiting at Arcadia while the nutrient spiking results suggested the opposite. However, it must be remembered that a filtered water sample was used in the availability study and an unfiltered sample was used in the nutrient spiking study. Also, the availability study used a single algal species while the spiking study utilized whatever organisms were present in the river at the time of sampling. Although *Selenastrum* can be used to assess nutrient availability, the use of a natural population provides a better estimate of algal nutritional status in the sample of interest. The inconsistency in the results is therefore probably due to the use of different samples, different sample treatments, and different test organisms.

Conclusions

191. Results obtained through the algal bioassay experiments demonstrate that available phosphorus and available nitrogen exist in the Deep Fork River. At the time of sampling, this amounted to 0.09 to 0.30 mg PO_4-P/ℓ and 0.5 to >5 mg N/ℓ. The results indicated that all of the available nutrients were not being used by the algae. Since the population was viable, as demonstrated by the fact that they responded

positively to the addition of AAP medium and productivity increased during laboratory incubation, light is probably controlling the algal population in the Deep Fork River.

192. Nutrient spiking results showed that nitrogen concentrations are lowest relative to need at Arcadia, but phosphorus concentrations are lowest relative to need at Eastern Avenue. This would indicate that a change in limiting nutrients would be expected following STP diversion prior to impoundment of the Deep Fork River. It would also be anticipated that the process of impoundment would cause a reduction in available nutrient concentrations. It is not possible to use the past record of nutrient concentrations in the Deep Fork River at Arcadia to predict accurately the algal population in the proposed impoundment because light or some other physical factor is more important in controlling the algal population in the Deep Fork River, available nutrients are not completely used at the present time, available nutrient concentrations will probably decrease due to impoundment, and the relative abundance of nutrients can be expected to change on impoundment.

193. Nitrogen and phosphorus concentrations in the proposed impoundment will exceed levels presently thought to promote excessive algal populations, but algal use of this material will probably be dependent on light availability rather than the presence of nutrients.

Eutrophication Potential and Mathematical Models

Mathematical ecosystem models

194. The water quality of a reservoir or lake is affected by many dynamic processes occurring both in the impoundment and in the watershed. Mathematical ecosystem models represent a relatively new approach for relating and coupling many of these processes to analyze and predict environmental changes resulting from alternative management practices. It is important to identify differences between ecological models and thermal models. Most thermal models are developed from a theoretical construct based on a set of physical axioms or laws such as Beer's law

for the extinction of light, Fick's law for diffusion, and the Stefan-Boltzman radiation law. Ecological models usually have a thermal model as a basis but the biological and chemical relationships are not developed from a set of similar ecological axioms or laws. Model formulations are based on empirical relations observed among components in the field or laboratory and, in many instances, are based on poorly understood processes such as the regeneration or loss of nutrients to the sediments.

195. Present ecological models should not be interpreted as predicting absolute values. However, based on order of magnitude arguments, these models permit experienced investigators to evaluate general trends resulting from various management alternatives such as changing flow regimes and/or nutrient loadings. Ecological models are useful tools when interpretations are made with a thorough understanding of model limitations and assumptions. This requires a multidisciplinary team approach. Present ecological models provide a means for interpreting voluminous amounts of water-quality data. The greatest advantages of mathematical ecosystem models are their coupling and simultaneous consideration of a large number of complex interactive processes and the ability to simulate water quality over relatively long time periods. This permits the evaluation of temporal effects of changing land-use patterns or similar occurrences on water quality.

Model description

196. The model selected for use in evaluating the eutrophication potential in Arcadia Lake was the Water Quality for River-Reservoir Systems (WQRRS) model. The model was originally developed by Water Resources Engineers (WRE) under a Title II contract from the Office of Water Resources Research[41] and subsequently modified for the U. S. Army Engineer Hydrologic Engineering Center (HEC).[42] During the past few years, numerous modifications and improvements have been made to the model by the Environmental Effects Laboratory at WES. Considering the hydrodynamics, biology, chemistry, and data requirements, the WQRRS model is presently one of the most comprehensive and realistic lake-riverine ecological models available for practical applications.

197. In addition to the five conditions previously stated for the

WESTEX model, the model assumptions are as follows:

a. A reservoir or impoundment can be represented by a series of one-dimensional horizontal slices. This implies that only the vertical dimension is retained during computation. This assumption is generally satisfactory for impoundments with long residence times.[42]

b. Each horizontal layer is assumed to be completely homogeneous and instantaneously mixed for all components.

c. The dynamics of each chemical and biological component can be expressed by the Law of Conservation of Mass and the Kinetic Principle.[41]

198. The computer approximates the numerical solution of the mass-balance differential equations in finite difference form using an implicit solution technique. The model is programmed in FORTRAN and has been run on IBM, CDC, UNIVAC, and Honeywell systems.

Data input

199. The data requirements for the model are rather extensive, so the sources and/or calculations of required inputs and initial conditions are discussed briefly.

200. Daily meteorological values of cloud cover, dry bulb and dew point temperatures, barometric pressure, and wind speed were obtained from the National Climatic Center in Asheville, North Carolina, for the Class A weather station 16 km southwest of the damsite. This information was used to update daily meteorological conditions in the model.

201. Stream flows, temperature, and chemical constituents were obtained from the 1969-1974 USGS records for their gaging station at Arcadia. Stream constituents entering the proposed impoundment were updated on a monthly basis in the model. The purpose of the simulations was to investigate nutrient loading and light penetration effects on water quality rather than daily operational characteristics of the impoundment. The stream update information was weighted to reflect conditions before and after diversion of the Northside STP.

202. Detritus and total coliform concentrations were computed from an urban runoff report for Tulsa, Oklahoma.[29] These values were also weighted as a function of stream flow and updated on a monthly basis. The Northside STP was assumed to contribute total coliform concentrations

in proportion to its contribution to the total stream flow. The total loading of coliforms to the lake following diversion of the STP was, therefore, reduced proportionally to the total stream flow reduction.

203. Stream algae concentrations were calculated from chlorophyll-a concentrations measured below an outfall of domestic and industrial wastes in Skeleton Creek, Oklahoma.[43] A conversion factor[44] was used to compute algae dry weight values; the data were weighted as a function of flow; and the concentrations were updated on a monthly interval.

204. Zooplankton values were taken from a study of the lower Illinois River[45] and weighted to reflect Deep Fork River flows. Stream zooplankton values are generally quite low and have little effect on reservoir concentrations.

205. All simulations were begun using January dates because most impoundments in Oklahoma are isothermal and well mixed during the winter period. This simplified the data requirements for the initial conditions since only one value near the bottom and one near the surface were necessary to specify most of the chemical and biological constituents. Initial concentrations were originally taken from literature on Oklahoma impoundments,[46-51] and a simulation was performed using updates for an average year. This permitted the transient responses in the model output to decay. If the initial conditions deviated sufficiently from the values predicted at the end of a year, the initial conditions were modified to reflect these yearly values.

206. Rate coefficients used in the simulations were obtained from the literature,[52-67] based on personal experiences, and personal communication with Mr. Donald Smith of Tetra Tech, Inc., Lafayette, California, one of the original developers of WQRRS.

207. The model was run with daily meteorological updates and monthly recorded USGS stream flow, physical, and chemical data for 1970 and 1973. These two years represented below and above average flow years, respectively, for the period from 1969 to 1974.

208. The algal bioassays indicated light would probably be the factor limiting or controlling algae production with secondary control exerted by nitrogen or phosphate if sufficient light were available.

To predict the potential effects of these limiting factors on water quality, update values 50 percent higher and lower than the mean annual light penetration and mean monthly values for ammonium and phosphate concentrations were used in the simulations.

209. Light penetration in the model is a function of a model variable titled "Secchi disc." The Secchi disc value specified in the model is equivalent to the depth of penetration of 10 percent of the light impinging on the surface. The larger the value for the Secchi disc variable, the deeper light penetrates in the impoundment. Three values of light penetration were used in the simulations: 1, 2, and 3 m, where 2 m represents the base case; 1 m represents the least light penetration; and 3 m represents the greatest penetration. Data on Oklahoma impoundments indicate these Secchi disc values are reasonable. During 1970, Lake Carl Blackwell had an average Secchi disc value of approximately 0.7 m,[51] while in 1972 the average value was near 2 m.[47] An average annual Secchi disc value of 2 m was also recorded for Lake Thunderbird, an impoundment on the outskirts of Oklahoma City.[50] Since values greater than 2 m have been recorded during periods of the year in Oklahoma impoundments,[47,50] 3 m was selected to investigate the effects of increased light penetration on water quality. Update values 50 percent higher and lower than the mean monthly coliform concentrations were simulated to evaluate potential health problems. In all cases where actual data from the Arcadia basin did not exist, conservative estimates were calculated to simulate the "worst-case" conditions. Since data existed for 1970,[51] the model was calibrated for thermal similitude on Lake Carl Blackwell, which is a 1200-ha reservoir approximately 50 km north of Oklahoma City. The reservoir is relatively shallow, and wind-mixing is a dominant force controlling stratification of the impoundment. A wind-mixing algorithm was incorporated in the model and lake profiles simulated by the model were similar to those in Lake Carl Blackwell for 1970 (Plate 12).

Model simulations

210. Model constraints. Three constraints concerning reservoir operations were placed on the model simulations. First, minimum

monthly downstream water releases were to be maintained to meet various downstream water rights. Releases never exceeded stream channel capacity. Secondly, the release temperatures had to be similar to annual preimpoundment stream temperatures. Finally, the water level in the reservoir was to be maintained at a nearly constant level at the top of the conservation pool. These three constraints were sufficiently satisfied (Plates 13, 14, and 15).

211. Sample output. A portion of the simulation for the 1970 data set following STP diversion provides an example of the type of numerical output generated (Appendix B). The first set of numbers provides information on the frequency of output, the frequency of input, and other information relating to the actual mechanics of the simulation. The next several blocks of numbers define some of the physical coefficients used during the simulation, such as the evaporation coefficient and heat exchange coefficients. The following several pages list the required biological and chemical coefficients and the initial conditions of the ecosystem. Since the reservoir was isothermal, one surface and bottom initial condition was specified and an interpolation algorithm calculated initial conditions for every metre in depth from the surface of the impoundment to the bottom.

212. The remainder of the output represents predicted values for 10-day intervals from 1 January to 26 December. Since this output is presented in graphical form elsewhere in the text and is listed here only for illustrative purposes, only the printouts for 30-day intervals are included. The simulation day is indicated at the top of each page followed by two columns. The first column represents the meteorological conditions used for this particular day, while the second column provides general information about the elevation of surface waters, the total inflow to the reservoir, and the residence time. The next two lines of data represent inflow and outflow water-quality constituents such as temperature, pH, various nutrient concentrations, and TDS. The outflow distribution is further delineated in the next two lines. The first represents the outlet through which release occurs; while the second line indicates the quantity of flow in cubic metres per second (cms).

Three types of fish can be simulated in the model--cold-water fish, warm-water fish such as the gizzard shad, and benthos feeders such as suckers. Cold-water fish will not exist in Arcadia Lake and were therefore not considered. In this study fish were used primarily to control zooplankton and benthos populations. Since the model could not simulate the dynamics of carnivorous fish such as the black bass or crappie, predictions of the lake fishery were based on an analysis provided by Mr. Robert M. Jenkins* (Appendix C).

213. The majority of each output page has 19 columns that represent, in vertical profile, the following predicted water-quality constituents abbreviated as shown:

a. Temperature - TEMP
b. Dissolved Oxygen - DO
c. Daily Oxygen Demand - DOD
d. Total Alkalinity - ALKA
e. pH
f. Carbon Dioxide - CO2C
g. Ammonia Nitrogen - NH3N
h. Nitrate Nitrogen - NO3N
i. Nitrite Nitrogen - NO2N
j. Phosphate Phosphorus - PO4P
k. Total Coliforms - COLIFM
l. Nannophytoplankton - ALGA 1
m. Net Phytoplankton - ALGA 2
n. Zooplankton - ZOO
o. Organic Detritus - DETRI
p. Total Dissolved Solids - TDS
q. Organic Sediment - SEDM
r. Benthic Organisms - BENTH

214. The extensive output generated during the simulations has been summarized in the following text and plates.

215. Temperature. Many of the processes occurring within the

* Director, Reservoir Fisheries Research Program, U. S. Fish and Wildlife Service, Fayetteville, Arkansas.

impoundment are either directly or indirectly influenced by the thermal properties of the stream and impoundment. Since the density of water is a function of its energy state as indicated by temperature, temperature profiles can provide information on the movement of constituents through the impoundment.

216. From early April until early September, Deep Fork River inflows would, in general, be 3 to 4°C cooler than the reservoir surface waters (Plate 16). During this period, the Deep Fork River inflows, after some inflow mixing, would sink or plunge below the surface to a level of similar density in the lake. The chemical and biological constituents in the inflowing stream waters would be correspondingly distributed below the surface during this period. From the middle of September until the beginning of April, the reservoir surface waters would be 4 to 5°C cooler than the inflowing waters. During this period, the stream water and its chemical and biological constituents would remain at or near the surface of the reservoir.

217. Hydrometeorological conditions influence the temperature differences that may exist between inflowing and reservoir surface waters. The temperature differences were less during a wet, high-flow year (e.g. 1973) than during a dry, low-flow year (e.g. 1970) (Plate 16). This flow influence was also exhibited by observing temperature isopleths of the reservoir for the base case when the annual mean light penetration equaled 2 m (Plate 17). During the dry year, for both pre- and post-STP diversion conditions, the reservoir heated and cooled more rapidly than during the corresponding periods in the wet year. The reservoir reached similar maximum temperatures during both years, but the maximum temperatures were sustained for longer periods during the wet years. This same pattern also occurred when the annual mean light penetration averaged 1 or 3 m (Plates 18 and 19, respectively).

218. Diversion of the STP flow had a moderate effect during the low-flow year (Plates 17 and 19). When the light penetration equaled 2 m, the reservoir heated slightly more rapidly and cooled slightly more quickly following diversion of the flow. The reservoir also heated slightly more rapidly following diversion in the high-flow year, but

the temperature profiles were similar for the rest of the year. With an average light penetration of 1 m, the reservoir heated more rapidly but cooled slightly slower following diversion in the dry year. Temperature profiles were similar before and after diversion in the wet year. An increase in light penetration to 3 m resulted in similar temperature profiles preceding and following diversion in 1970 and 1973. Increased light penetration in the reservoir probably resulted in a sufficient increase in the internal heat budget to damp the influence of stream flow reductions following diversion. This same response is observed if comparisons are made of the three light penetration values within a given period such as 1970 prediversion or during the 1973 postdiversion period. The deeper light penetration resulted in more rapid heating during the spring and slower cooling during the fall. The impoundment begins to stratify during April and remains weakly stratified until middle August. The thermal gradient during the summer is only 2 to 4°C from surface to bottom but is stable. The lake begins to mix in mid to late August and remains well mixed until the following April.

219. Dissolved oxygen. In all cases, DO near the bottom of the impoundment was reduced to zero during some portion of the year (Plates 20 through 22). The periods of oxygen depletion, in general, occurred earlier in the dry year than in the wet year.

220. For the base case (i.e., 2-m annual mean light penetration), diversion of the effluent from the STP resulted in higher DO values during the dry year than during the wet year (Plate 20). While the hypolimnion was still anoxic for certain periods, the total duration of periods when the DO concentration was less than 4 mg/ℓ was reduced. Diversion did not result in similar improvements during the wet year (Plate 20). The anoxic period during late spring was reduced by half following diversion, but a subsequent anoxic period occurred during the summer. The total duration of periods with less than 4 mg/ℓ of DO was reduced slightly following diversion.

221. The DO concentrations were generally greater before diversion when the mean annual light penetration average was 1 m (Plate 21). While the DO concentrations appear to be lower before diversion in the

wet year, the periods of anoxia and periods with less than 4 mg/ℓ are greater following, rather than preceding, diversion. Since the stream inflows are nearly saturated with DO, the higher inflow-outflow relations before diversion result in higher DO concentrations within the impoundment.

222. With a mean annual light penetration of 3 m (Plate 22), STP diversion generally resulted in increased DO concentrations. While the anoxic periods were similar before and after diversion in the dry year, the periods of less than 4 mg/ℓ DO were reduced following diversion (Plate 22). Diversion resulted in a significant improvement in the DO concentrations during the wet year (Plate 22). The anoxic periods were significantly reduced with similar periods of less than 4 mg/ℓ DO.

223. Nutrients - PO_4-P, NO_3-N, and NH_3-N. Phosphate concentrations, in general, decreased in the wet year (Plates 23-25). The flow through the reservoir was greater during the wet year (1973) than during the dry year (1970), resulting in lower concentrations although the total loadings were greater (see Part VI, Table 10). Epilimnetic phosphate resembles a conservative substance in its response since phosphate concentrations are in excess of algae needs (Plates 23-25). Postdiversion phosphate concentrations increased slightly during both the dry and wet years since the flow through the reservoir is decreased slightly. Variations in average light penetration had a negligible effect on epilimnetic phosphate concentrations.

224. Fluctuations in the hypolimnetic phosphate concentrations correspond with anoxic periods (Plates 23-25). When DO concentrations reached minimal levels, phosphate uptake by algae no longer occurred and phosphate concentrations began to increase since release still continued. Increased oxygen concentrations resulted in renewed uptake of phosphate, and hypolimnetic phosphate concentrations were again similar to epilimnetic concentrations. Variations in light penetration had a minimal effect on reservoir phosphate concentrations since phosphorus is not limiting to algal populations.

225. Nitrate concentrations are a function of flow rates and

ammonia concentrations. Increased flow rates during the wet year, in general, resulted in decreased nitrate concentrations compared to the dry year concentrations (Plates 23-25). Increases or decreases in nitrate concentrations were also generally correlated with inverse decreases or increases in ammonia concentrations. This response can be seen by comparing wet year with dry year data: nitrate concentrations were generally lower in the wet year than in the dry year.

226. The general inverse relation between surface concentrations of nitrate and ammonia and between bottom nitrate and ammonia concentrations is also apparent in Plates 23-25. Diversion of the STP effluent resulted in less fluctuation in both nitrate and ammonia concentrations during the dry year. Nitrate concentrations were slightly higher following diversion due to reduced inflow. The changes in bottom concentrations of nitrate and ammonia correspond to anoxic conditions in the hypolimnion. Ammonia concentrations increased as DO concentrations became minimal since oxygen is required to convert ammonia through nitrite to nitrate. Nitrate concentrations decreased as the conversion of ammonia diminished. The return of aerobic conditions resulted in the reestablishment of the steady-state isotropic concentrations of nitrate and ammonia throughout the reservoir. Diversion also resulted in reduced fluctuations and concentrations of nitrate and ammonia in the wet year. Similar trends in nitrate and ammonia concentrations occurred with light penetration values of 1 and 3 m.

227. Variations in the annual light penetration means had minimal effects except for the wet year postdiversion period (Plates 23-25). The difference in the response of ammonia and nitrate at a light penetration of 1 m versus that at 2 and 3 m was again a function of the hypolimnion DO concentration. The bottom was anoxic at the end of May resulting in an increase of ammonia and a decrease in nitrate. The concentrations were similar for the rest of the wet-year period.

228. Increases and decreases of 50 percent in the update ammonia concentrations resulted in increases and decreases in the ammonia and nitrate concentrations. The majority of the increase or decrease occurred in the nitrate concentrations (Table 24). There was an inverse

relation between ammonia and DO concentrations. A 50-percent increase or decrease in the ammonia concentration resulted in a corresponding decrease or increase of approximately 0.5 mg/ℓ DO, respectively.

229. Algae. Algae growth and decay functions as an integrator for many of the physicochemical responses, such as nutrient interactions, temperature effects, and changes in DO concentrations, that contribute to water quality. The model has two compartments to represent the various algae species. These two compartments, designated ALGA 1 and ALGA 2, represent nannophytoplankton and net phytoplankton, respectively. Nannoplankton are considered to be less than 63 μ in size and will pass through a No. 20 plankton net.[68]

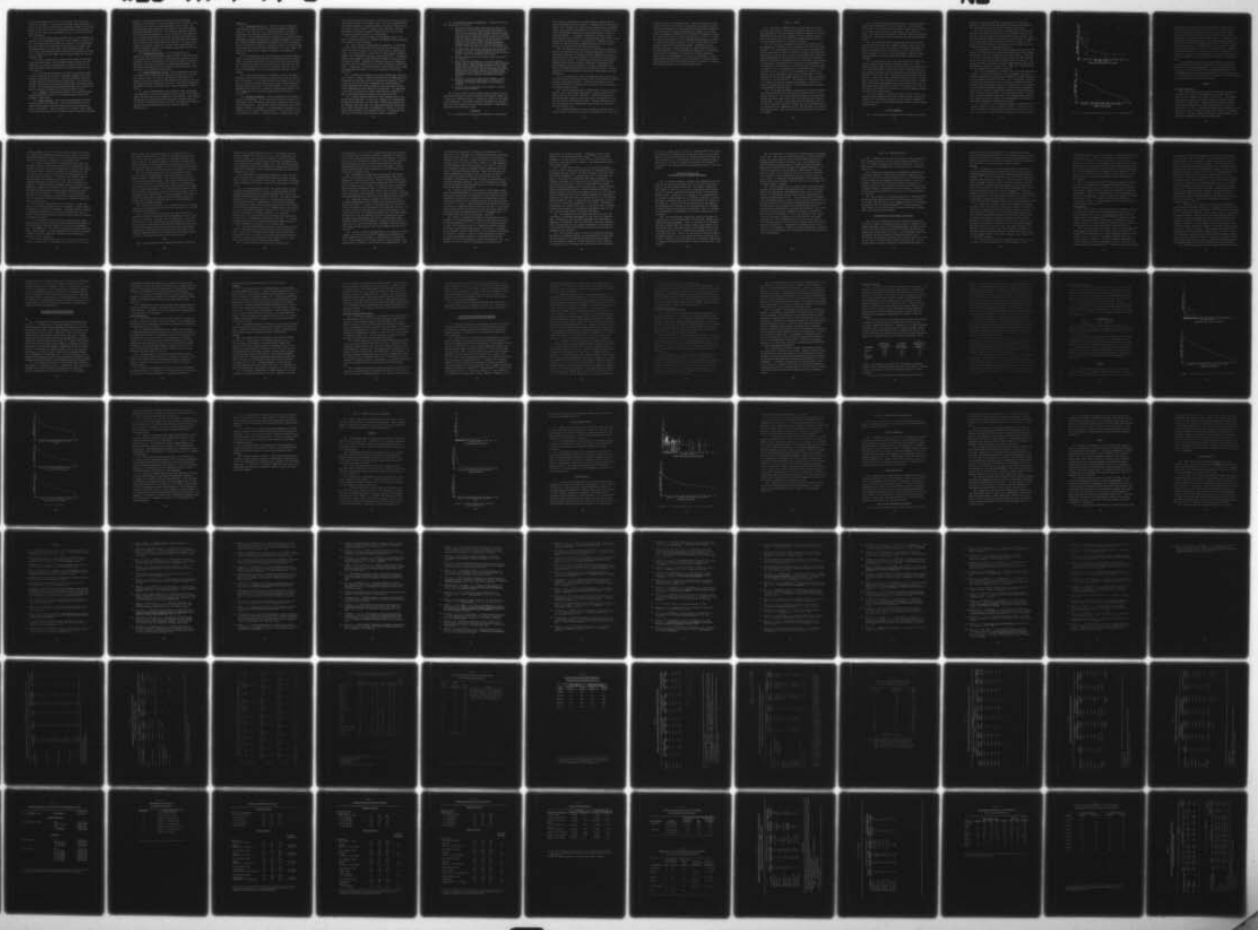
230. The predominant algae type predicted by the model under all conditions was the nannoplankton, ALGA 1 (Plates 26-28). For most conditions, there were two periods when large blooms were predicted: midspring, late April or early May, and late summer, late August or early September. This general trend did not apply to the wet-year postdiversion conditions. The model predicted a bloom of nannoplankton to occur in early spring, late spring, and late summer-autumn under the wet-year postdiversion conditions.

231. In the base case the dry year pre- and postdiversion responses differed in the intensity and composition of the late summer bloom (Plate 26). The prediversion bloom had a lower intensity than the postdiversion bloom but was composed of nearly equal concentrations of net and nannoplankton. The postdiversion bloom consisted almost entirely of nannoplankton. The wet year pre- and postdiversion conditions differ in time phasing, intensity, and duration of the nannoplankton bloom (Plate 26). The first prediversion bloom occurred during midspring, while the postdiversion bloom occurred in early spring followed by another bloom in late spring. Both pre- and postdiversion data sets exhibited a bimodal curve representing a late summer bloom, but the postdiversion bloom was more intense and lasted until late fall.

232. For an annual mean light penetration of 1 m, the dry year pre- and postdiversion algal curves were similar (Plate 27). The shape of the nannoplankton curve for the postdiversion case was less pronounced

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and occurred slightly later than the prediversion case but still occurred in late summer. In the wet year decreased light penetration resulted in both pre- and postdiversion nannoplankton peaks occurring later during the year (Plate 27). The prediversion bloom occurred during late spring and fall while in the postdiversion case, the fall bloom lasted until early winter.

233. When the annual light penetration was 3 m, the intensity and duration of the nannoplankton curves were quite similar for the dry year pre- and postdiversion cases (Plate 28). However, in the postdiversion case, net plankton also contributed substantially to the summer bloom. The bimodal shape of the late summer curve was again evident in the wet year pre- and postdiversion data (Plate 28); the intensity and duration of postdiversion late summer peak were again greater than the prediversion peak.

234. Algal concentrations, in general, were higher with greater light penetration values. While minimal differences among algae concentrations occurred for light penetration values of 2 and 3 m, the differences among algae concentrations were pronounced between 1 and 2 or 1 and 3 m.

235. Increases and decreases by 50 percent in the update ammonia concentrations generally resulted in negligible changes in the nannoplankton concentrations (Plate 29). The only significant difference occurred during the 1970 prediversion summer bloom. The bloom was intensified and preceded the increased nitrogen bloom period approximately 30 days. The bloom attributed to decreased nitrogen was over about the time the increased nitrogen bloom would have reached its peak.

236. Phosphate. Increases and decreases in the update phosphate concentrations had no effect on any component except the reservoir phosphate concentration.

237. Total coliforms. Total coliform concentrations correlated almost perfectly with flow rates: total coliform concentrations increased with increased flow rates and decreased with decreased flow rates (Plates 26-28). Total coliforms were, in general, higher during the wet year than the dry year and higher preceding rather than following

diversion. The peak in coliform concentrations occurred during September for two reasons. First, the highest flows of record for both the dry and wet years occurred during September and, secondly, the reservoir surface water temperature was similar to the stream temperature during September (Plate 16). The isothermal conditions resulted in the stream inflow remaining on or near the surface of the reservoir. Total coliform concentrations, on several occasions, were higher than those recorded for the top 1-m water layer but increased concentrations were located at greater depths within the reservoir. Increases and decreases of 50 percent in the total coliform update concentrations resulted in 50-percent increases or decreases, respectively, in the reservoir concentrations (Plate 30).

238. Total dissolved solids. Total dissolved solids are treated as a conservative substance within the model. This implies the concentrations are affected only by dilution and not by decay processes. In general, the TDS concentration in the reservoir remains constant regardless of variations in diversions or light penetration means. The TDS update concentrations were not changed following diversion, so dilution effects are negligible (Plates 26-28).

239. Outflow temperatures and DO. Outflow temperatures generally were similar for the same year but varied from the dry year to the wet year (Plates 31-33). The outflow temperatures in the wet year, in general, were cooler than the dry year during the first half of the year and warmer during the second half of the year. The outflow temperatures were within the range of existing natural stream temperatures (Plates 3, 31-33).

240. Zero DO concentrations were predicted at least once during every simulation. A selective withdrawal option was used in the simulations that considered only the temperature of the outflow. After passing through the outlet structure and stilling basin, the outflow would be at least 80- to 90-percent saturated with DO. The dashed lines indicate expected DO concentrations during actual reservoir operation (Plates 31-33).

Discussion

241. Physicochemical factors. Simulated reservoir temperature isopleths were similar to those recorded for Oklahoma impoundments (Plate 12). The importance of including wind-mixing in the model is indicated by the relatively high wind velocities for 1970 and 1973 (Plate 34). Since the WESTEX model was calibrated on a southern Oklahoma impoundment surrounded by high-relief topography and WQRRS was calibrated on a northern Oklahoma impoundment surrounded by low-relief topography, Lake of the Arbuckles would not be expected to exhibit as extensive upper layer mixing due to wind as Lake Carl Blackwell. The actual temperature profile at Arcadia may lie somewhere between the two predictions (Table 26).

242. DO concentrations for selected dates on Lake Carl Blackwell[51] (Plate 12) correspond reasonably well with simulated values for similar dates. The model predicted less DO near the bottom than that measured in the field on day 200 but DO concentrations for other days were similar. This similitude may be fortuitous, however, since the flows, nutrient concentrations, etc., are not expected to be similar in both systems.

243. Nutrient concentrations of phosphate, nitrate, and ammonia predicted by the model are probably too high since no loss to the sediment is considered. In the model, nutrients are released from the sediment in proportion to their concentration in the biological organisms and detritus that settle out to form the sediment. It is difficult to determine what proportion of the nutrient concentrations should be lost to the sediments, but it has been documented that such losses do occur.[69,70]

244. Biological components. Algae concentrations are probably too high in the simulations. To be conservative, algae were assumed to be controlled by only one limiting factor with no interaction among potential limiting factors. The possible limiting factors were light, carbon dioxide, nitrogen, or phosphate. A Michaelis-Menten formulation was established for all possible limiting factors,[71-74] and the factor in the lowest proportion relative to need was assumed to limit growth during

that particular integration interval. During the next iteration, another factor might control or limit algae growth. The actual growth-limiting relationship probably lies somewhere between the product formulation and the single limiting concept.[73] However, since algae are a potential water-quality problem, the single limiting factor was used to ensure simulating the maximum potential for algae growth.

245. While no loss of nutrients to the sediment occurs in the model, a 50-percent reduction in ammonia and phosphate showed negligible or no effects on algae responses.

246. Total coliform concentrations were assumed to be 10 percent of total coliform concentrations 0.5 miles below the STP outfall. The travel time from the outfall to the Arcadia Lake site is approximately 2 days. For small streams, a travel time of 2 days results in approximately a 98- to 99-percent reduction in total coliforms.[75] A 90-percent reduction was assumed to ensure sufficient latitude in assessing potential health problems. Assuming this reduction implies the loadings to the stream near the outfall were on the order of 16,000 to 1,150,000 per 100 ml. Furthermore, the decay rate based on this reduction was 1.4/day, which was within the range of 0.5 to 2.46 indicated in the literature.[42,75]

247. Standing crop estimates for the lake fishery were calculated by Mr. Robert M. Jenkins. Fishery predictions from the WQRRS model simulations are not considered valid for evaluating the potential fishing of Arcadia Lake since the fish compartments were used primarily to control zooplankton and benthos populations and carnivorous fish such as the largemouth bass and crappie were not simulated. The improvement of the model to provide valid fishery simulations is an objective of part of the ongoing modeling research being conducted at WES.

248. Jenkins' equations indicated a total standing crop of fish between 200 and 290 kg/ha. Of this total standing crop, approximately 64 kg/ha were predicted to be sport fish; 74 kg/ha were clupeids; and the remainder were rough fish. The annual angler harvest was predicted to be roughly 60 percent of the top predators, 25 percent of the panfishes, and 20 percent of the catfishes and carp. The regression equations used to make these predictions are listed in Appendix C.

249. Interpretations from the simulations. Interpretations from the simulations are as follows:

a. The reservoir will be weakly stratified from mid to late spring until late summer or early fall with a thermal gradient of 2 to 4°C. During the first two-thirds of the year, stream constituents such as heavy metals, pesticides, nutrients, and coliforms will probably enter the reservoir below the surface. During the latter one-third of the year, these constituents will enter at or near the surface of the reservoir. Wind, inflow mixing, and withdrawal will probably be the dominant forces distributing these constituents within the reservoir. Outflow temperatures should be within the range of natural stream temperatures.

b. The simulations indicate the hypolimnion will become anoxic, possibly as high as 10 m from the bottom during various periods of the year. Outflows should be at least 80 percent saturated with DO even during anoxic periods.

c. Nutrient concentrations will probably remain quite high in the reservoir.

d. Algal blooms will probably be an occasional problem. A general rule of thumb indicates 1.5 mg/ℓ of algae represents visible levels. During several periods of the year, predicted values exceed this level by a factor of 3. Light appears to be the critical factor controlling algal growth. During periods of low wind velocity or increased light penetration, algal blooms may be expected since nutrient concentrations do not appear to be potential limiting factors.

e. Diversion of the STP effluent does not appear to reduce nutrient concentrations sufficiently to preclude the occurrence of nuisance algal blooms in the proposed impoundment.

f. Total coliform concentrations do not appear to indicate potential health problems.

250. General trends from model studies provide insight into possible system response to various alternative environmental changes, but a comparison of absolute values or a statistical analysis of the output is not recommended. These results should be interpreted by comparing them with results obtained from other study approaches, recognizing the limitations and assumptions of both.

Discussion

251. The evaluation of nitrogen and phosphorus in the Deep Fork

River based on concentrations and loading analyses suggests that the proposed impoundment would be eutrophic. Mathematical model simulations suggest that occasional algal blooms would be expected after impoundment. However, algal response was relatively insensitive to variations in nitrogen and phosphorus loadings but responded noticeably to changes in light availability. A reasonable interpretation of these results is that nutrients would be available in excess of algal needs and that algal growth would be light limited. Bioassay results corroborate the hypothesis of light limitation by demonstrating that all the available nutrients in the Deep Fork River were not being used.

252. The existing state-of-the-art is insufficiently developed to permit a precise prediction of the timing and magnitude of algal blooms. Predictions based upon the mathematical model simulations may be misleading because of the model assumption of uniformly and instantaneously mixed horizontal layers. Limited horizontal mixing in the proposed impoundment would result in incomplete dispersion of developing algal blooms allowing localized algal growth to attain concentrations exceeding the average concentration over the horizontal layer. Localized algal blooms may attain densities much greater than predicted by model simulations.

253. Algae contribute to taste and odors and other problems such as high particulate concentrations and variable water chemistry requiring treatment plant control measures. Excessive algae densities may be visually displeasing, and decomposition of massive algal accumulations may liberate obnoxious odors.

254. During extended periods of cloudy weather in which high algal densities occur, daily oxygen requirements for algal respiration may exceed production and surface reaeration. Surface DO concentrations could be depressed during early morning hours to levels sufficient to cause the death of low DO-sensitive fish. Transient depressions of DO were not simulated in the mathematical model because the time resolution selected was 1 day.

255. Management measures to control problems associated with excessive algae may possibly be applied to the impoundment or, in the case

of water supply, at the treatment plant. Selective withdrawal of water supply water does offer potential for minimizing water-quality problems. A cursory examination of a recent publication surveying lake rehabilitation techniques for reducing algae populations suggests that reliable reservoir management strategies that are effective, economically reasonable, and compatible with all project purposes do not exist.[76] It probably would not be feasible, either through watershed management practices or reservoir operational strategies, to reduce in-lake nutrient concentrations sufficiently to limit algal growth. Furthermore, routine chemical treatment of large water bodies to control algal blooms does not appear feasible. It is recommended that sampling of the tributaries to the proposed impoundment be continued and a sampling program be established in the impoundment if it were to be constructed. The sampling program would provide more precise estimates of nutrient loadings and concentrations that would enable evaluation of procedures to mitigate algal problems specific to the proposed Arcadia Lake.

PART VII: METALS

256. This section summarizes the trace metal data from the Deep Fork River with regard to potential water-quality problems in the proposed impoundment. The study approach involved a comparison of reported concentrations with recently recommended EPA water-quality criteria.[6] Recent EPA criteria were selected because Oklahoma standards[5] did not provide numerical limits for the metals of interest.

257. The first data considered were the data on dissolved metal concentrations in the Deep Fork River compiled by the USGS between 1969 and 1974.[27] With the exception of arsenic and mercury, the evaluation of trace metal data was based on dissolved or soluble concentration data because metals in this form, as opposed to particulate form, would more closely approximate the chemical species used in bioassay experiments that were the basis for recommended criteria. High total concentrations of water-quality constituents may be indications of contamination, but a comparison of total concentrations to criteria based on soluble chemical species is not a good indication of the potential effects. As discussed by Lee et al., toxicity can vary markedly with the form of a chemical, and the application factor used to relate lethal to chronic sublethal effects has questionable applicability to soluble and particulate forms of the same species.[15] Total arsenic was evaluated because of the absence of data for dissolved concentrations. Criteria for mercury concentrations were based on total mercury because of possible interconversion of mercury forms.

258. The minimum and maximum concentrations of 25 metals measured by the USGS in the Deep Fork River during the period 1969-74 are summarized in Table 26. Since many analytical results were less than some reported detection limits, two average concentrations were calculated for the table. The first average was based on the assumption that all so-called "less-than" values were actually zero, and the second average was calculated on the assumption that all less-than values were equal to the detection limit. The actual average concentration should be within this range.

259. The average and maximum dissolved metal concentrations were compared to the EPA numerical criteria (Table 26). This comparison revealed that 2 metals exceeded the criteria; 10 metals were less than the criteria; and no criteria were specified for the remaining 13 metals. The two metals that exceeded the proposed limits were manganese and mercury. Average iron concentrations were near maximum permissible levels.

260. The 10 dissolved metal concentrations that were less than proposed EPA criteria were iron, cadmium, chromium, copper, lead, zinc, barium, beryllium, arsenic, and silver. The 13 metal concentrations about which nothing can be said because of a lack of information to develop standards are nickle, cobalt, molybdenum, vanadium, aluminum, lithium, bismuth, tin, gallium, germanium, titanium, strontium, and zirconium.

261. An examination of other data from the Deep Fork River near Arcadia yielded the same general conclusions (Table 27). Silver, cadmium, chromium, copper, and zinc were less than the EPA criteria, and iron, lead, and manganese exceeded the criteria. Nothing definite can be said about the mercury results because while the mercury content was less than the detection limit, the detection limit was greater than the criteria. Average total lead concentration equaled the limit for public water supply in the Deep Fork River near the proposed damsite. However, based on the present understanding of the water chemistry of lead, the high lead concentrations shown in Table 27 are probably due to particulate species that should not be compared to toxicity standards developed from tests using soluble species.

262. Iron, manganese, and mercury (metals that were found to approach or exceed numerical limits) were further studied to assess their potential for water-quality degradation in the proposed impoundment. Lead data were analyzed in more detail because previous studies reported excessive levels.

Iron and Manganese

263. Average manganese concentration for 26 samples collected by

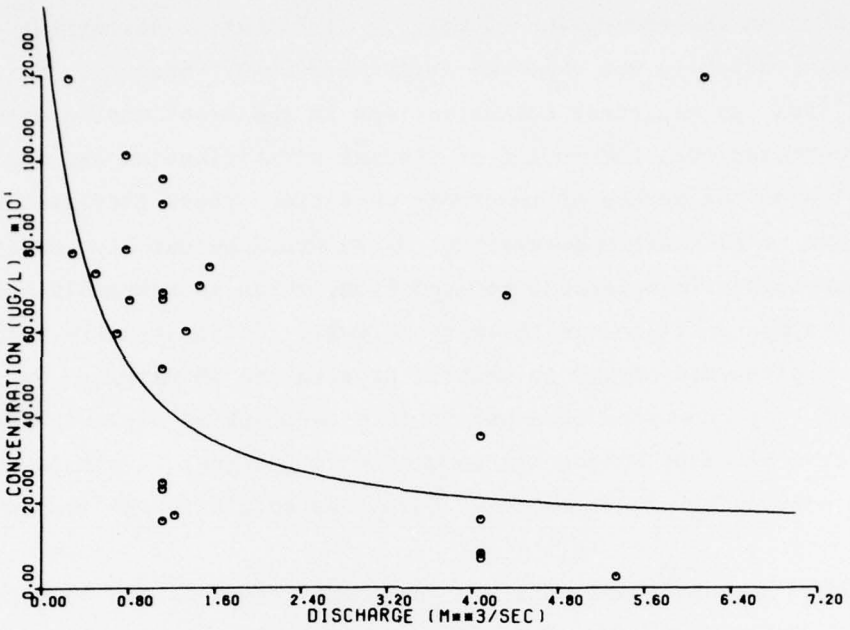
the USGS near Arcadia was 400 µg/ℓ; the range was 30 to 1100 µg/ℓ (Table 26). Both the average and maximum concentrations exceeded the most stringent EPA criterion (50 µg/ℓ for public water supply).

264. The average concentration of manganese measured in the final effluent of the Northside STP was 75 µg/ℓ, considerably less than the average of 400 µg/ℓ measured near Arcadia between 1969 and 1974. Since the final effluent supplied considerably more flow than mass of manganese, calculations suggest that relocation of the STP will result in a 60-percent increase in manganese concentrations near the damsite with an insignificant decrease in loadings (Table 10).

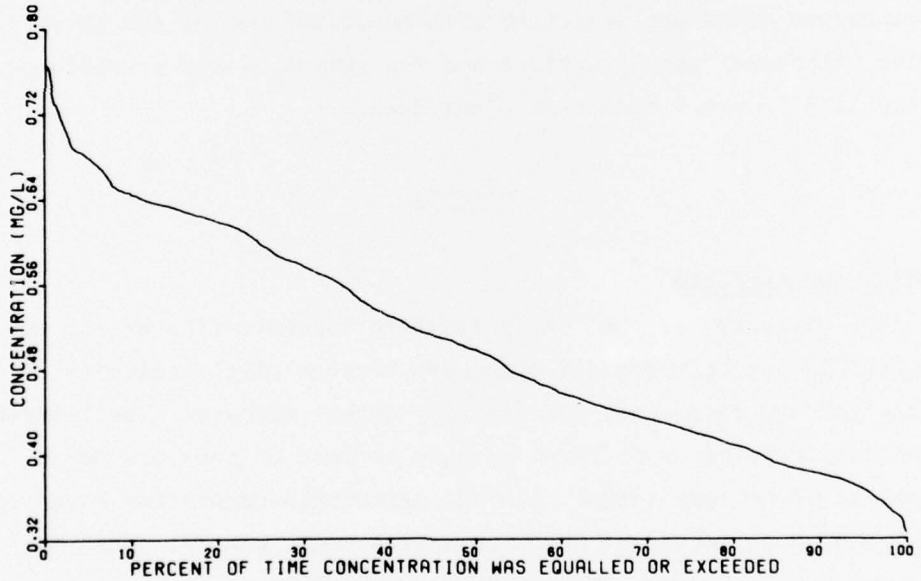
265. Examination of data collected by agencies other than the USGS (Table 27) reveals that iron concentrations exceeded the most stringent EPA criterion, 300 µg/ℓ, proposed for public water supply. The average of the iron concentrations measured near Arcadia by the USGS (Table 26) was 285 µg/ℓ, less than the most stringent criterion of 300 µg/ℓ. The average was based on 26 samples collected between 1969 and 1974 and was substantially inflated by a single value of 3800 µg/ℓ. Fifteen percent of the 26 sample values exceeded the criterion.

266. Iron and manganese are different from mercury and other metals in Table 26 in that the standards for iron and manganese are based on aesthetic considerations and other metal standards are based on toxicity. Iron and manganese in excess of 300 and 50 µg/ℓ, respectively, can cause "red water" and "black water" that will result in unsightly stainings. Dissolved iron and manganese would not be expected to be a toxicological or health problem following impoundment. Excessive iron and manganese would not be a problem in finished water supplies if the potential problem is recognized in the design of the water treatment plant. Furthermore, selective withdrawal capability of the proposed project would enable project operation to minimize iron and manganese concentrations in water supply withdrawals and downstream releases.

267. Mass-balance calculations were employed to prepare daily concentration duration curves for manganese (Figure 6). The median simulated concentration assuming conservative behavior was 500 µg/ℓ, 23 percent less than the average tributary concentrations. The calculations



CONCENTRATION-DISCHARGE RELATIONSHIP



CONSERVATIVE BEHAVIOR

Figure 6. Manganese duration curves based on daily concentrations

were based on the assumption of uniform distribution throughout the reservoir, which is not expected in Arcadia Lake. Seasonal increases in the iron and manganese concentrations in the hypolimnetic waters would coincide with the period of thermal stratification and anoxia. Depending on the method of reservoir operation, these increases could be withdrawn or discharged downstream. Iron would be quickly converted to the particulate form because reduced iron, which is responsible for the concentration increase, is thermodynamically and kinetically incompatible with dissolved oxygen in neutral or alkaline pH waters. Manganese would also be converted to a particulate form but at a slower rate. The conversion of soluble iron and manganese to particulate species would have a scavenging effect on other dissolved metals in the impoundment or river.

268. Manganese and possibly iron concentrations are expected to exceed drinking water standards, particularly in the hypolimnion and headwaters of the proposed impoundment. Iron and manganese would be less likely to exceed standards in the epilimnion near the dam. Iron and manganese would not interfere with municipal use of the water if selective withdrawal were practiced and the potential for possible problems was realized in water treatment plant design.

Mercury

Limiting concentration

269. Mercury has the lowest limiting concentration of the metals specified in the criteria and standards because of the toxicity of organic mercury compounds, specifically methyl mercury. The limiting concentration is based on total mercury because of possible interconversion of mercury forms. Average mercury concentration based on eight samples collected by the USGS near Arcadia was 1.1 µg/ℓ; the range was 0 to 2.6 µg/ℓ. The maximum recorded mercury concentration and the average mercury concentration exceeded EPA's value for freshwater aquatic life. However, average Deep Fork River concentrations are less than EPA's recommended level for public water supply.

Total mercury analyses

270. Because mercury was found in concentrations exceeding some criteria and since only eight measurements were made for the element of which seven were made in the same month, additional sampling was conducted on the Deep Fork River near Arcadia. OSU was contracted by WES to conduct additional sampling and mercury analyses on the Deep Fork River near the Arcadia damsite. Samples were collected on 5, 7, and 14 June 1975. On 7 June the river was turbid due to moderately high flow transporting suspended particulates. The remaining samples were collected during low flows. On each date, three water samples were collected and separately analyzed for dissolved and total mercury. No mercury was detected in the samples (detection limit = 0.3 µg/ℓ).

271. A water-quality survey conducted by the Oklahoma State Health Department 21-22 April 1971 for the upper Deep Fork River Basin from above Belle Isle Lake to below Arcadia revealed undetectable mercury concentrations (number of samples = 8; detection limit = 5.0 µg/ℓ).

272. Weekly duplicate sampling of the final effluent of the Northside STP on 12, 19, and 26 March and 2 April 1975 by OSU revealed an average total mercury concentration of 0.7 µg/ℓ; individual samples values varied from <0.1 to 2.8 µg/ℓ. Only samples collected on 19 March 1975 had detectable concentrations of mercury (detection limit = 0.1 µg/ℓ).

273. Storm water runoff sampling for mercury in the upper Deep Fork River above Belle Isle Lake was conducted by ACOG during October 1976. As shown in the following tabulation, the average mercury concentration was 1.68 µg/ℓ; individual sample values varied from 0.66 to 3.8 µg/ℓ.

	Total Mercury Concentration, µg/ℓ		
Date Sampled	Sample 1	Sample 2	Sample 3
5 October 1976	0.69	0.66	1.0
29 October 1976	2.7	3.8	1.2

274. The Oklahoma State Health Department conducted weekly sampling for mercury in both the Deep Fork River and the final effluent of the Northside STP during the period 29 March through 1 June 1976. No mercury was detected in the Deep Fork River, while only samples collected 19 April 1976 in the Northside STP had detectable concentrations (4.0 and 4.5 µg/ℓ). The total number of samples collected at each sampling site was 20 and the detection limit was 2 µg/ℓ (Table 28).

275. A total of 79 analyses of total mercury were performed between 1971 and 1976. These data indicate that mercury has a low probability of degrading water supply (one purpose of the proposed project) because the mean mercury concentration computed over all samples collected near Arcadia is 0.2 µg/ℓ, which is less than the public water supply criterion of 2 µg/ℓ. It is further anticipated that mercury concentrations in the proposed impoundment would decrease relative to input in much the same way that trace metal concentrations in surrounding impoundments are less than their respective input concentrations.

Evaluation of effect on aquatic life

276. The data on total mercury concentrations do not provide a basis for a conclusive evaluation of the possible adverse effects on freshwater aquatic life. Some of the reported mercury concentrations exceed the proposed criterion for protection of aquatic life. The seriousness of this potential problem could not be evaluated in a quantitative manner with the available data base because the majority of the samples had mercury concentrations less than the analytical detection limit, but the analytical detection limit is greater than the proposed criterion. Two additional factors make definitive predictions of possible environmental effects of mercury in the proposed impoundment difficult based upon EPA guidelines. First, analytical procedures for mercury analysis as outlined in EPA's Manual of Methods for Chemical Analysis of Water and Wastes recommend that mercury concentrations below 0.2 µg/ℓ determined by their standard cold vapor technique be reported as <0.2 µg/ℓ.[11] This detection limit is four times greater than the recommended criterion. This causes a single detectable measurement to greatly inflate the average concentration relative to the criterion.

Secondly, proposed criteria are based on chronic exposure levels which were largely and conservatively extrapolated from lethal toxicity data obtained from bioassays; therefore, it is not known what environmental effects may result as a consequence of exceeding this value.

277. During a meeting held in Dallas during February 1976, representatives from the Dallas Regional Office of EPA recommended that, in order to provide a conservative approach for evaluating the potential for adverse effects of mercury on aquatic life in Arcadia Lake, fish be collected from the Deep Fork River near the proposed damsite and analyzed for mercury content. The results of the analysis could directly determine the suitability of the fish as a food source by comparing body burden concentrations to FDA limits and determine the existence of possible adverse environmental effects of mercury contamination by comparison to 1973 EPA criteria. EPA published a maximum body burden concentration for mercury in fish tissue that is equal to the FDA limit of 0.5 mg/kg.[17] The rationale of the EPA-published criterion is that maintenance of fish tissue concentrations below this level is sufficient to ensure the protection of freshwater aquatic life from excessive mercury.

278. The study recommended by EPA was expanded to include lead and all chlorinated pesticides that were considered as possible problem constituents in the proposed impoundment. Following a search of analytical laboratories capable of the accuracy and precision the study required, North Texas State University (NTSU), Denton, Texas, was selected to perform the analysis.

279. A collection of fish consisting of green sunfish, orange spotted sunfish, bluegill sunfish, and black bullhead was obtained by TD by electrofishing in the Deep Fork River at the proposed impoundment site. The samples were sent to NTSU for analysis to determine metal and trace organic contents. A report prepared by NTSU detailing the methods of sample collection, preparation, and analysis and presentation of results is given in Appendix D.

280. Merury concentrations in the fish flesh samples ranged from 0.05 to 0.76 mg/kg, while corresponding concentrations in fish viscera

samples ranged from less than detectable to 0.04 mg/kg (Table 29). The sampling design provides an estimate of the population mean based upon 20 fish, but only 11 observations are available to estimate fish-to-fish variance. An unbiased estimate of the population mean concentration of mercury in fish flesh obtained by weighting the sample observations proportional to the number of fish in the composite is 0.26 mg/kg. The estimate of mean mercury concentration in fish viscera is 0.11 mg/kg.

281. An unbiased estimate for fish-to-fish variance of flesh mercury content is 0.0629 mg^2/kg^2; the variance estimate of viscera mercury content is 0.0288 mg^2/kg^2. These estimates were used to construct one-sided confidence limits. This statistic was chosen because *a priori* the fish were to be analyzed to determine if the population mean concentration exceeded the FDA limit or EPA criterion of 0.5 mg/kg, apart from some assigned chance. The upper 99-percent confidence limit on the population mean mercury content in flesh is 0.42 mg/kg. The corresponding upper confidence limit for mercury content in fish viscera is 0.22 mg/kg. Based upon the sample values, the hypothesis that fish mercury content equals or exceeds the FDA limit or EPA criterion of 0.5 mg/kg is considered untenable.

282. Based on these results, it is concluded that the true mercury content of fish inhabiting the Deep Fork River near Arcadia is less than published FDA and EPA limits.

283. The fish species collected from the Deep Fork River and analyzed for metals and pesticides feed predominantly on insects, other fish, and crustaceans. Thus these fish are positioned in a food chain whose origin is in the sediments and associated particulate materials. Since the data collected on the Deep Fork River indicate that most trace metals and pesticides are associated with particulates, the evidence suggests that concentrations of trace metals and pesticides in fish following impoundment would not be greater than, and probably would be less than, concentrations observed in the Deep Fork River.

Lead

284. Average dissolved lead concentration measured by USGS near

Arcadia over the period 1969-74 was 3.9 µg/ℓ, considerably less than the most stringent EPA criterion of 50 µg/ℓ. Lead concentrations reported by agencies other than the USGS (Table 27) exceed the maximum concentration permitted in Oklahoma waters and EPA's recommended level for the protection of freshwater aquatic life. The high lead concentrations tabulated in Table 27 are probably due to particulate species that should not be compared to toxicity standards developed from tests using soluble species of lead.

285. Dissolved lead concentrations modified to reflect STP relocation remained equal to prediversion values with an average concentration of 5 µg/ℓ over the period of 1969 to 1974. This value is less than the most stringent EPA criterion of 30 µg/ℓ for the protection of aquatic life.

286. In a report prepared by OSU,[2] lead concentrations in the proposed Arcadia impoundment were listed as a potential problem because measured lead concentrations in the Deep Fork River were close to and sometimes exceeded the EPA criterion of 0.05 mg/ℓ. The range of reported values was <0.01 to 0.18 mg/ℓ with 13 of 51 observations equaling or exceeding the criterion.[2] Additional data collected by the Oklahoma State Department of Health along the upper Deep Fork River and presented in the OSU report demonstrated that 6 of 8 samples taken on one particular day also exceeded the present criteria for lead. The most logical source for this lead is automobile emissions. In a report by Newton et al.,[77] it was calculated that the average lead concentration in runoff from the Oklahoma City area would be 0.23 mg/ℓ. In a limited sampling program, actual total lead concentrations ranged from 0.09 to 8.5 mg/ℓ, depending on the location of the sampling site.

287. The lead concentrations presented in the OSU report[2] and discussed as dissolved lead are actually something other than dissolved lead. At the time of collection, water samples were preserved with nitric acid to a pH less than 2. The analysis thus measured dissolved lead plus any particulate lead that may have dissolved in the dilute nitric acid solution between collection and analysis.

288. A recent study of the Deep Fork River reported lead

concentrations of 1 to 73 µg/ℓ.[78] Although actual data were not given, it was reported that most lead was present in suspension as shown by the fact that the lead was removed from the water by filtration. In addition, total lead concentrations increased in those sections of the Deep Fork River where the current velocity increased and the total lead concentrations decreased as the stream velocity decreased. This is a further indication that lead is transported in the particulate form rather than the dissolved form.

289. An examination of USGS water-quality data[27] from the Deep Fork River near Arcadia, Oklahoma, further demonstrates that lead is transported in the particulate form. During the period January through May 1974, the total lead concentration ranged from <0.015 to 0.900 mg/ℓ with an average value of 0.26 mg/ℓ. The dissolved lead concentrations during this period ranged from 0.001 to <0.016 mg/ℓ with an average value of 0.007 mg/ℓ. Based on these results, only 2.7 percent of the total lead present (on the average) is in the dissolved form. Additional USGS unpublished data for dissolved and total lead follow this pattern.[79]

290. Additional evidence to support the position that lead is transported through natural water systems in particulate form was presented by Peterson.[80] A study of 16 Wisconsin lakes and 2 streams revealed that dissolved lead concentrations were generally less than 5 µg/ℓ and that the ratio of dissolved lead to total lead was usually in the range of 0.2 to 0.5. The mechanisms responsible for the observed phase distribution of lead are probably sorption and coprecipitation since the waters were undersaturated with respect to lead based on known solubility products for lead compounds. Laboratory studies also confirmed that the lead in the water would be reduced to less than equilibrium levels even when placed in contact with sediments with high lead contents.[80]

291. The fate of lead in natural waters is to accumulate in sinks. A survey of the technical literature[80] suggested that lacustrine and marine sediments act as lead sinks, as demonstrated by the increased rate of lead accumulation in the upper layers of these sediments. The effectiveness of sediments as lead sinks can be demonstrated by a study

of an urban-rural watershed in Illinois, which showed that 97 to 98 percent of the lead input over a 6-month period was not transmitted through the water system.[81] The time necessary for lead movement to sediments can be very short as shown by the fact that the calculated residence time for lead in Monona Bay, a portion of a lake influenced by urban drainage from Madison, Wisconsin, is 0.5 days.[80] Also, once in the sediments, there is no apparent chemical mechanism for the release of lead from sedimentary material. Under oxic conditions, the aqueous environmental chemistry of lead is controlled by sorption-coprecipitation reactions rather than solubility and less than detectable concentrations of sulfide will prevent the solubilization of lead under anaerobic conditions. Also, lead concentrations are apparently not magnified through the food chain as are other trace contaminants in the aquatic environment.[82]

292. Although total lead concentrations in urban runoff reached several thousand micrograms per litre and lead concentrations in urban precipitation reached tens of micrograms per litre, dissolved lead concentrations in lake water only reached a few micrograms per litre because of the dominant role of sorption reactions in the water chemistry of lead.[80] As a result, it was concluded that lead derived from the use of the metal as a gasoline additive would not constitute a serious water-quality hazard and that the process of clarification would reduce lead concentrations below recommended standards if the Madison lakes (or even urban runoff) were used as water supply sources.[80]

293. There is an apparent discrepancy in the dissolved lead concentration in the Deep Fork River as reported by OSU[2] and those reported by other investigators.[27,78,79] The cause of the discrepancy appears to result from an improper choice of a sample preservation technique. The method of acidification chosen by OSU[2] will result in the measurement of dissolved lead plus some fraction of the particulate lead present in each sample. However, a review of selected references concerning the aqueous environmental chemistry of lead demonstrates that the majority of lead in natural waters is present in particulate form. This suggests that the OSU analysis[2] overestimated the dissolved lead

concentration in the Deep Fork River. Consequently, based on other sampling near the proposed impoundment site[27,78,79] and the present knowledge of the water chemistry of lead,[80] this element would not be expected to cause water-quality problems if the impoundment were used as a water supply source.

294. A recent study by Davies et al. of the acute and chronic toxicity of lead to rainbow trout demonstrates the differential toxicity of dissolved and total lead.[83] Two static bioassays in hard water gave 96-hr LC50's (estimates of the concentrations of a toxicant which are lethal to 50 percent of the organisms tested under the specified conditions and times) of 1.32 and 1.47 mg/ℓ for dissolved lead and 542 and 471 mg/ℓ for total lead. LC50's for total lead were over three orders of magnitude greater than for dissolved lead. Maximum acceptable toxicant concentrations (MATC's) of lead for rainbow trout based on chronic bioassays in hard water were between 18.2 and 31.7 µg/ℓ dissolved lead and 120-360 µg/ℓ total lead. Average dissolved lead concentration measured in the Deep Fork River near Arcadia of 3.9 µg/ℓ is less than the calculated MATC for trout. Average total lead concentration of 220 µg/ℓ is near but less than the average MATC for total lead. Significantly, the bioassays were conducted with trout, a fish species that is generally considered more sensitive to environmental perturbations than the warm-water fish that would inhabit the proposed impoundment.

295. The preceding study, the observation that trace metal concentrations in surrounding impoundments are less than their respective input concentrations, and the present knowledge of the environmental chemistry of lead indicate that lead would not be expected to adversely affect fish populations in the proposed impoundment. Recent analyses of lead content in fish collected from the Deep Fork River (Table 29) substantiate this conclusion.

296. Since the major source of lead in the Arcadia watershed was attributed to automobile emissions, an initial objective of this study was the evaluation of the significance of possible future changes in the lead content of gasoline. The EPA published an initial schedule for phasing lead out of gasoline that required lead to be reduced from

0.45 g/ℓ on 1 January 1975 to 0.13 g/ℓ on 1 January 1979.[84] This action was successfully challenged on the grounds that the data did not demonstrate that use of leaded gasoline in vehicles produced any harmful effects on human health. On 20 December 1974, these regulations were set aside by the U. S. Court of Appeals for the District of Columbia.[85] As a result of this decision, EPA has suspended enforcement of the regulations.

Trace Metal Loadings and Concentrations in the Proposed Impoundment

297. No attempt was made to calculate total loading for most trace metals at the Arcadia site because of the lack of data and the variability of the data available (Table 26). For example, although the data period for Table 26 is 5 yr, only six observations were obtained for 14 of the metals and these were obtained during a 1-month period (February 1974). Therefore, it is not known if the concentration can be expected to change seasonally. In addition, variability is high in some cases as demonstrated by the iron concentrations that ranged from 0 to 3800 µg/ℓ. Finally, the state-of-the-art has not progressed to the point where metal loadings can be calculated based on a knowledge of land-use patterns and population distribution. Thus, the influence of the impoundment or urbanization on trace metal loadings cannot be calculated.

298. Even though present loading rates cannot be reasonably approximated, it can be anticipated that the trace metal loadings will be reduced following impoundment. First, the diversion of the Northside STP effluent should reduce the loading of metals below present rates because many metals have been detected in the effluent (Table 27) and the sewage outfall represents a significant percentage of total flow in the Deep Fork River. Second, the conversion from leaded gasoline to nonleaded gasoline, if it occurs, should specifically reduce the flux of lead. A reduction in gas consumption due to shortages, higher prices, or changing driving habits would tend to further lower the rate of lead loading.

299. The significance of reduced metal loadings cannot be estimated at this time. Since there are insufficient data to calculate the present loading, the percent reduction due to sewage diversion and the use of nonleaded gasoline cannot be calculated. Also, the distribution of metals from these sources between the soluble and particulate phase is unknown, which would further complicate any attempted prediction. As can be seen in Table 30, the percentage of soluble metal in the Deep Fork River ranges from 1 percent for iron to 77 percent for strontium. Thus a reduction in total loading would not have a uniform effect on all the metals. In addition, the redistribution of the residual metals between soluble and particulate forms that may occur if the present rate of loading changes is unknown.

300. Another approach taken to evaluate future trace metal concentrations in the proposed Arcadia impoundment was to collect and analyze samples from Lakes Eufaula and Thunderbird (Tables 31 and 32, respectively). In Lake Eufaula, average dissolved metal concentrations were less than recommended EPA criteria. Total concentrations for cadmium, iron, and manganese appear to exceed criteria, but the comparison is not valid because the total concentration includes particulate metal species, but the criteria were established using dissolved metal species.[15] Dissolved metal concentrations in the Deep Fork tributary to Lake Eufaula were less than EPA criteria, suggesting no buildup in concentration would occur after impoundment. In fact, dissolved iron, total copper, and total zinc appear to decrease in the lake relative to tributary concentrations. Lake Thunderbird data are similar in that dissolved metal concentrations are usually low with occasional high total iron and manganese concentrations. A comparison of tributary and lake data in Tables 31 and 32 suggests a reduction in dissolved manganese, a reduction in total iron and manganese, and no significant change in other dissolved metal concentrations.

PART VIII: PESTICIDES AND PCB

301. A comparison of the 1976 EPA criteria with all pesticide and PCB's measured in the Deep Fork River near Arcadia is summarized in Table 33. Lindane, chlordane, DDT, dieldrin, heptachlor, aldrin, and PCB exceeded the recommended limits for the protection of freshwater aquatic life.

302. Because of the persistence, bioaccumulative properties, and carcinogenic potential of chlordane, DDT, dieldrin, heptachlor, and aldrin, the EPA did not establish numerical limits for these pesticides for water supply but recommended minimum human exposure to these constituents. Significantly the EPA has suspended the production or restricted the use of each of these pesticides, which should result in a gradual decrease in concentrations in the environment. However, each of these pesticides was detected at least once in the Deep Fork River near Arcadia.

303. Data reported by the Oklahoma Departments of Agriculture and Pollution Control suggest that pesticide contamination is restricted to the upper Deep Fork River or that residues are either not transported downstream or they are rapidly degraded.[86,87] Monthly sediment and water samples collected from the Deep Fork River near Beggs, Oklahoma, had no detectable pesticide residues.

Water Chemistry of Pesticides: An Overview

304. The use of organic pesticides has increased exponentially since 1940. Because of widespread increased usage, the question is not whether pesticides occur in a particular environment but rather at what level do they occur in the environment.[88] Pesticide residues are introduced into surface waters as a result of (a) atmospheric deposition, groundwater, and runoff losses from treated agricultural areas; (b) municipal and industrial effluents; and (c) direct addition to lakes for the control of aquatic plants, insects, and rough fish. Once in surface waters, the aquatic chemistry of chlorinated hydrocarbons is

strongly influenced by the properties of the individual pesticides. Since most chlorinated hydrocarbons have low solubilities and are usually nonpolar, the dominant reactions are (a) sorption on suspended solids and/or precipitation with resultant incorporation into the sediments and (b) accumulation in the fatty tissue of living organisms.

Sorption

305. The importance of the sorption/precipitation process for removing chlorinated hydrocarbon compounds from surface waters has been repeatedly demonstrated by the rapid and large accumulation observed in sediments.[89-95] The phase distribution ratio for pesticides in water and suspended sediments has been as great as 1:1000 for dieldrin, DDT, and aldrin and between 1:100 and 1:1000 for aldrin, heptachlor, and heptachlor epoxide.[94] The most plausible mechanisms for the sorption processes are van der Waal's forces and hydrogen bonding because of the low polarity and slight ionization of pesticide compounds.[92] Important variables that can influence the sorption equilibrium are (a) suspended solids concentrations, (b) organic matter content of the solids, (c) type of pesticide, (d) pesticide concentration, (e) clay content of the solids, (f) pesticide-to-sediment ratio, (g) pH, and (h) temperature.[92,93]

306. Sorption also becomes a process for transporting pesticides from one portion of the environment (the surface waters) to another (the sediments) as suspended solids settle under the influence of gravity. Thus, sorption is an indirect form of environmental dissipation since the chlorinated hydrocarbon pesticide is still present chemically, but the degree of availability to aquatic organisms is usually reduced markedly.[93] The process of sorption may also act as a catalyst in the degradation of pesticides as shown by the fact that adsorbed parathion will hydrolyze two to three times faster than soluble parathion.[96] In addition, sorption may increase the flux of pesticide residues into anaerobic sediments where the degradation of γBHC (a high-energy form of benzene hexachloride),[97] DDT, endrin, lindane, aldrin, heptachlor,[94] and dieldrin[98] is enhanced.

307. Association of pesticides with suspended solids has the general effect of reducing the mobility of these compounds. After

reviewing the literature, Pionke and Chesters concluded that chlorinated hydrocarbons were not amenable to leaching at concentrations as high as 1000 µg/g.[94] However, this effect will depend on the pesticide in question and the nature of the adsorbent. For example, Boucher and Lee observed that 70 percent of the lindane sorbed on natural aquifer sands was leached by three washes of distilled water, but less than 20 percent of the sorbed dieldrin was released under the same conditions.[99] Also, Huang and Liao[100] observed that 25 percent of the dieldrin but little DDT or heptachlor was desorbed from montmorillonite washed with distilled water.

308. In a separate study, Lee[101] observed the apparent release of DDT, endrin, and methoxychlor from Houston Ship Channel sediments in quantities sufficient to exceed EPA criteria. (The pesticide content per kilogram of sediment was 46.5 mg DDT, 29.3 mg endrin, and 30.0 mg methoxychlor.) Also, the continued presence of PCB's in Escambia Bay, Florida, was presumed to be caused by leaching from sediments containing 486 ppm Aroclor 1254.[102] Sediments from the area containing 1.4 to 61.0 ppm PCB's produced a concentration of 3.5 ppb in the effluent from a flow-through bioassay tank.[95]

Bioaccumulation

309. Another method of pesticide movement in the aquatic environment that has received considerable attention is bioaccumulation. This is the phenomenon that results in organisms having higher pesticide residues than the surrounding water,[89,103-106] and organisms at higher trophic levels having higher concentrations than organisms at lower trophic levels.[107] This distribution of pesticides within a community is frequently attributed to magnification within the food chain.

310. It has been shown that predators could be acutely affected by endrin-containing organisms. For example, Rosato and Ferguson[108] and Ferguson[109] have demonstrated that endrin-containing mosquito fish were acutely toxic to 11 species of vertebrates including fish, frogs, turtles, snakes, and birds. In one case, an endrin-contaminated fish released enough pesticide into the water to be toxic to mosquito fish that were previously uncontaminated.[110] However, there is some question

as to whether the observed accumulation of pesticides in the food web is due to ingestion of lower species. Ferguson and Goodyear[111] determined mortality of black bullheads in endrin solutions after tying off the gut in the region of the upper esophagus. Since there was no difference in mortality of treated and control fish, it was concluded that endrin ingested with food is inconsequential and that the gills were the predominant mechanism of entry. Reinert et al.[112] compared dieldrin and DDT uptake in lake trout from water containing 0.006 to 0.010 ppb insecticides and food pellets containing 1700 to 2300 ppb. The accumulated levels after 152 days were 350 to 650 ppb, but it was not possible to determine how much of the accumulated material was due to the food source since 0.003 to 0.004 ppb of insecticide leached into the water. Nimmo et al.[95] observed that crab and shrimp accumulated Aroclor 1254 but could not distinguish between ingestion of sediment particles or adsorbing PCB's leached from the sediments. However, the results of these studies suggest that predatory species can concentrate pesticides directly from the water. Pesticides in sediments are probably not readily available to fish.[113]

311. Regardless of the mechanism of chlorinated hydrocarbon insecticide uptake, organisms generally do not continue to concentrate the compounds until toxic levels are reached. Meeks[89] followed the distribution of isotopically labeled DDT in a Lake Erie marsh for 15 months after a single application. Samples of water, suspended solids, sediment, and 40 species of organisms ranging from plants to muskrats and birds were collected and analyzed for DDT residues in as many as 16 tissues or organs per specimen. The black crappie was the only species to accumulate DDT continually through the duration of the study, reaching concentrations of 0.7 ppm at 14 months. Other organisms had higher whole body concentrations that reached their maximum value in 3 days to 3 months after application, but these levels steadily declined although the sediments contained a constant residual of 0.3 ppm. This suggests that many species can either degrade or excrete chlorinated hydrocarbon residues and sedimentary DDT is not readily available to the marsh organisms.

312. Mrak[90] reviewed several studies showing oysters and shellfish

could cleanse themselves of pesticide residues. Shellfish excreted 0.3 to 1.5 ppm of chlorinated hydrocarbon in 10 to 20 days, depending on the compound. Another study demonstrated that endrin concentrations in submerged plants, clams, and fish varied with the seasonal use of the compound. PCB concentrations in Escambia Bay, Florida, oysters also fluctuated with changes in the water concentration.[102] These results demonstrate that short-term pesticide contamination does not imply a permanent residual concentration in aquatic organisms. The significance of this accumulation is not understood at this time, although excretion of residuals[110] and predation on contaminated organisms[108] have caused mortality to other species in some cases.

Toxicity

313. An important aspect of the water chemistry of pesticides that is influenced by the sorption and bioaccumulation mechanisms discussed above is the apparent toxicity of the residue. Sorption may catalyze the degration of a pesticide[96] or may carry the residue to the sediments where degradation may be favored.[94,97,98] In addition, incorporation of pesticide residues into the sediments may decrease biological magnification factors by several orders of magnitude.[114]

314. The process of biological magnification also tends to reduce apparent toxicity of pesticides as shown by the higher mean tolerance limit of test animals with a past exposure history and a measurable body burden residue. In one case, previously exposed mosquito fish had a 36-hr TLm_{50} (median tolerance level) of 1000 µg/kg endrin, and nonexposed mosquito fish had a 36-hr TLm_{50} of 1 µg/kg endrin.[110] In another case, susceptible golden shiners were killed in 75 min when exposed to endrin for the first time, but tolerant shiners had only 80-percent mortality after 40 hr, even though their blood content was 64 times higher and the total body residue was 82 times higher than the susceptible specimens.[115] The increased tolerance was passed on genetically through at least four generations. This form of acclimatization has also been observed for DDT, toxaphene, and methyl parathion.[116] In addition, previously treated lake sediments degrade 2,4-D faster than untreated lake sediments, which further suggests biological acclimatization.[90] However, as discussed by Ferguson et al., an increased tolerance

may not be completely satisfactory since this may allow a greater biological accumulation that will result in a tolerant specimen being toxic to susceptible specimens of the same species at some later date.[117]

314. Studies also indicate that it may be necessary to distinguish between toxicity and the analytically determined pesticide concentration. Hughes has shown that toxaphene recovered from lake sediments is not as toxic as the original formulation.[91] Toxaphene is a multicomponent mixture containing 20 to 30 compounds, and one of these compounds that apparently contributes to toxicity was selectively degraded in the sediments. The same component was degraded in fish tissue, suggesting that accumulated residues had lost some of their toxicity. Another example is the isomerization of BHC (benzene hexachloride). A high-energy form, γBHC, is usually applied to the environment. However, this form is transformed into lower energy α, β, and δ forms by bond rotation. This reduces BHC toxicity from 125 mg BHC/kg body weight for the γ form to 6000 mg BHC/kg body weight, even though the BHC structure remains intact.[97]

316. A point of caution that should be mentioned is that all chlorinated hydrocarbon pesticide decay products are not known. It is thus possible that degradation may increase the toxicity of pesticide residues in some cases.[93] As an example (where the products are known), the degradation product of aldrin (dieldrin) is more toxic to some species than the parent compound.[118]

317. The toxicity of pesticide residues in water can be reduced by physical processes of volatilization[97] and codistillation.[88] These processes do not reduce toxicity by changing chemical structure or biological availability but by physically transporting chlorinated hydrocarbons across the water-air interface and out of the water phase.

Summary

318. The environmental chemistry of chlorinated hydrocarbons is exceedingly complex, as illustrated by the information discussed above. Although pesticides may persist for long times chemically, it is necessary to distinguish between the chemical and biological half-life. Processes such as sorption and sedimentation can reduce the biological

half-life relative to chemical half-life (i.e. reduce the availability). In addition, the toxicity-time relationship for a pesticide can be markedly changed in the environment. This can be attributed to an aging or selective degradation of the residue and to an increased tolerance that can be developed and passed on genetically. This adaptation may lower the apparent toxicity of pesticides by three orders of magnitude and further reduce the significance of chemical persistence. The variable adaptability of individual organisms also makes it impossible to establish precise critical levels for bioaccumulated chlorinated hydrocarbon insecticides at this time.[115]

Existing and Predicted Concentrations and Loadings in the Deep Fork River

DDT

319. The average DDT concentration over the period 1969-74 was 0.006 µg/ℓ. Concentrations of 64 samples ranged from 0 to 0.06 µg/ℓ with 30 percent of sample values exceeding the freshwater aquatic life criterion of 0.001 µg/ℓ. Average DDT concentration in Deep Fork River bottom deposits was 0.75 µg/kg. Inspection of Table 34 reveals that average annual concentrations decreased during the study. Loadings during the same period decreased from 1969 through 1972 but increased during 1973 to 1974 to prior levels (Table 35). However, 1973 and 1974 were exceptionally wet years and DDT loading is correlated with flow. While part of the increased loading is undoubtedly related to increased runoff as a consequence of increased rainfall, part of the increased loading may also be due to greater quantities of suspended sediments being transported at higher flow rates. It would be expected that the organochlorine pesticides would be found associated (adsorbed) with the suspended stream load because of the hydrophobic nature of pesticides.[98]

320. The average concentrations of the DDT degradation products, DDD and DDE, were 0.008 and 0.002 µg/ℓ, respectively. DDD concentration decreased from a high of 0.028 µg/ℓ during water year 1969-70 to 0.001 µg/ℓ during water years 1971-72 and 1972-73. Two detectable

concentrations measured May and August 1974 inflated the average DDD concentration for water year 1973-74 to 0.006 µg/ℓ. One sample collected in May 1974 had a detectable DDE concentration. Neither DDD or DDE were detected in bottom deposits of the Deep Fork River near Arcadia.

321. Modification of DDT sample values to reflect STP relocation resulted in no reduction in loading but a 16-percent increase in concentration. The sewage treatment plant is not a major source of DDT (Table 10).

322. Significantly, almost all uses of DDT were canceled under the Federal Insecticide, Fungicide, and Rodenticide Act, 7 USC 136 et seq., on 18 March 1971. The effect of this action should be a sharp reduction of DDT additions to the watershed.

Aldrin and dieldrin

323. The 1976 EPA criteria specifies that criteria for aldrin and dieldrin be based on the sum of the two pesticides. This recommendation was motivated by the observation that aldrin is metabolically converted to dieldrin by aquatic organisms.

324. Average aldrin concentration in 64 samples was 0.002 µg/ℓ; corresponding average dieldrin concentration in 64 samples was 0.038 µg/ℓ. The average sum of aldrin and dieldrin was 0.040 µg/ℓ, which exceeds the recommended EPA limit of 0.003 µg/ℓ for the protection of freshwater aquatic life. No aldrin was detected in bottom deposits while average dieldrin concentration was 0.95 µg/kg in the bottom deposits. Annual concentration of aldrin-dieldrin remained constant (0.03 to 0.04 µg/ℓ) over the study period, but annual loadings reflected increased flows during 1973 and 1974 (Tables 34 and 35).

325. The data suggest that the STP contributes flow and aldrin-dieldrin in the same relative proportion as measured near Arcadia. The sample values modified after considering STP diversion represent no change in instream concentrations but show a 19-percent decrease in average loadings.

326. The EPA has suspended the major uses of aldrin and dieldrin. The expected effect of the suspension is restriction of use to structural application by licensed exterminators with subsequent declines in

instream concentrations and loadings in the Deep Fork River.

Chlordane

327. The average value of 65 chlordane samples collected during the period 1969-74 was 0.15 µg/ℓ. Sample values ranged from 0 to 1.7 µg/ℓ with 52 percent of the values exceeding the EPA criterion of 0.01 µg/ℓ for the protection of freshwater aquatic life. Chlordane concentrations appeared to decrease during the first half of the study period to below the criterion and then increase during the remainder of the period (Table 34). The high concentration observed during water year 1973-74 is due to two exceptionally high values of 1.7 µg/ℓ measured on 21 and 22 May 1974. The same pattern was observed for annual loadings (Table 35) with the large loading calculated for 1973-74 due to the high sample values. Chlordane loadings are strongly related to flows. The average chlordane concentration in the sediments of the Deep Fork River near Arcadia was 5.5 µg/kg.

328. Calculations suggest that relocation of the STP will increase chlordane concentrations by 33 percent, while loadings will remain constant. The EPA has canceled many of the major uses of chlordane, which should reduce chemical loadings and probably concentrations.

Lindane

329. The average lindane concentration over the period 1969-74 was 0.07 µg/ℓ. Concentrations of 64 samples ranged from 0 to 3 µg/ℓ with 39 percent of the samples exceeding the most stringent EPA criterion of 0.01 µg/ℓ for the protection of freshwater aquatic life. However, the average lindane concentration was 50 times less than the recommended public water supply criterion. Average annual concentrations decreased during the study period to values below recommended levels (Table 34). The peak concentration observed during water year 1970-71 was due to one exceptionally high sample value collected on 22 January 1971 of 3 µg/ℓ. Loadings displayed a similar pattern, decreasing throughout the study period (Table 35). A continuation of this trend should reduce the concern for lindane in the Deep Fork River.

330. The average concentration of lindane measured in the final effluent of the treatment plant was 0.04 µg/ℓ, considerably less than

0.07 µg/ℓ measured near Arcadia between 1969 and 1974. Since the final effluent supplied considerably more flow than mass of lindane relative to instream values, calculations suggest that relocation of the treatment plant will result in a 71-percent increase in concentration and a 9-percent decrease in loading (Table 10). However, these calculations were based on the average lindane concentration over the period of data collection. Since the annual concentration at Arcadia has decreased to 0.009 µg/ℓ in the last 2 yr (Table 34), this would suggest that diversion of the Northside effluent would significantly reduce concentrations and loadings of lindane. No lindane was detected in Deep Fork River sediments near Arcadia.

Heptachlor and heptachlor epoxide

331. Sixty-three measurements of heptachlor and 64 measurements of its degradation product heptachlor epoxide were made in the Deep Fork River near Arcadia between September 1969 and December 1974. Only samples collected on 22 May 1974 had detectable heptachlor concentrations. No heptachlor epoxide was detected in water samples. The average heptachlor concentration was 0.002 µg/ℓ. Three percent of the samples exceeded the recommended EPA criterion of 0.001 µg/ℓ for the protection of freshwater aquatic life. Neither heptachlor or heptachlor epoxide were detected in Deep Fork River sediments near Arcadia

332. The ACOG conducted storm water runoff sampling for pesticides in the upper Deep Fork River above Belle Isle Lake. Heptachlor concentrations for single samples collected 5 and 29 October 1976 were 0.0002 and 0.000083 µg/ℓ, respectively. Heptachlor epoxide concentration of a sample collected 29 October 1976 was 0.000119 µg/ℓ. Both heptachlor and heptachlor epoxide concentrations were less than recommended EPA limits for the protection of freshwater aquatic life. Significantly, the EPA suspended the production and use of heptachlor on July 1975, which should result in a gradual decrease in concentrations in the environment.

PCB

333. Average PCB concentration measured in the Deep Fork River was 0.003 µg/ℓ, which exceeds the recommended 1976 EPA limit of 0.001 µg/ℓ for the protection of freshwater aquatic life. Only one sample collected

on 24 August 1974 had a detectable PCB content that was equal to the detection limit of 0.1 µg/ℓ. A total of 28 PCB analyses were conducted between 1969 and 1975. No PCB's were detected in the sediments of the Deep Fork River near Arcadia. Because of the rare occurrence of PCB's in the Deep Fork River and the phasing out of the use of PCB's as a dielectric in electrical equipment, PCB's are not expected to exceed criteria in the proposed impoundment.

334. Sampling of the Northside STP revealed that the STP was a significant source of one of the PCB's, Aroclor 1254. Diversion of the STP effluent should further decrease concern for these constituents.

An Evaluation of Possible Water-Quality Problems in Arcadia Lake due to Pesticides

335. Following STP diversion, calculations indicate that pesticide loading would decrease while pesticide concentrations would increase. An evaluation of organic pesticides in Arcadia Lake and downstream releases requires a consideration of the environmental chemistry of these constituents.

336. The calculation of average yearly concentrations of selected organic compounds in the Deep Fork River was based on the assumption that these compounds would behave conservatively. Extrapolation of these values to Arcadia Lake is in error because pesticides and related compounds generally exhibit sorption tendencies.[119] Meeks observed that 20 percent of the DDT in the water phase was associated with suspended material 1 hr after application of the pesticide to a marsh on Lake Erie.[89] Within 5 days, more than 80 percent of the material in the water was associated with suspended material. In another case, measurable quantities of DDT associated with suspended solids were transported 68 miles downstream from the point of application.[90] Other workers have shown that toxaphene,[91] lindane,[92,120] and parathion[90,121] concentrations are reduced by sorption. In addition, Robeck et al. have shown that activated carbon can reduce DDT, lindane, parathion, dieldrin, 2,4,5-T,

and endrin concentrations by as much as 99 percent, depending on the carbon dose.[122] The sorption of pesticides to particulate matter will reduce the amount available to organisms and increase their rate of removal from the water column.[93]

337. Additional data reported by the Oklahoma Department of Agriculture and the Oklahoma Department of Pollution Control also suggest that the calculated concentrations may be high. Sediment and water samples collected monthly during November 1972 through October 1973 from the Deep Fork River near Beggs, Oklahoma, had no detectable (detection limit = 0.02 µg/ℓ) pesticide residues.[86] Also, bass from Lake Thunderbird contained 0.08 to 0.115 mg/kg DDT compared to the recommended standard of 5.0 mg/kg.[86] A less-frequent sampling program conducted during 1971 and 1972 reported less than 0.02 µg/ℓ DDT, aldrin, dieldrin, heptachlor, lindane, and toxaphene in the Deep Fork River.[87] DDT residues of less than 0.7 mg/kg were reported in bass collected from Lakes Eufaula and Thunderbird (Table 36). Analysis of samples recently collected from Lake Eufaula and Lake Thunderbird and their tributaries indicates that the concentrations of organic compounds are not building up (accumulating) in lakes in the vicinity of Oklahoma City.[123] Three samples in the survey were positive for 2,4,5-T and one sample was positive for 2,4-D, but excessive concentrations were not reported.

338. An analysis of fish collected in Oklahoma reservoirs as part of a National Pesticide Survey demonstrated that excessive bioaccumulation is not occurring. Results from Lake Eufaula specimens for the period 1967-73 ranged from 0.1 to 0.3 mg/kg wet weight for DDT-related compounds and PCB's and were 10 times lower for dieldrin and BHC (Table 37). The dieldrin results are significant because this compound frequently exceeded recommended EPA standards for aquatic life but was not accumulated in the food chain. This suggests dieldrin is rapidly degraded or it is in a form that is unavailable to organisms. Either of these possibilities would diminish the potential problems of dieldrin residues in the proposed impoundment. The average pesticide concentration in fish in Lake Eufaula, which is fed by the Deep Fork River, is 15 to 500 times less than the FDA guideline of 5.0 mg/kg, depending on

the species of fish and the particular pesticide.

339. STP effluent diversion may have an additional indirect effect on the chlorinated hydrocarbon loading of the Deep Fork River at Arcadia. The diversion will reduce the flow in the river, which may reduce the suspended solids carrying capacity and scouring of the bottom deposits. Since pesticide residues entering water are frequently tightly bound to soil particles and residues are rapidly incorporated into the sediments, the total chlorinated hydrocarbon loading at Arcadia could be expected to be reduced due to sewage effluent diversion.[124]

Clorinated hydrocarbon
pesticides and public water supply

340. Because of the persistence, bioaccumulative properties, and carcinogenic potential of chlordane, DDT, dieldrin, heptachlor, and aldrin, the EPA did not establish numerical limits for these pesticides for water supply but recommended minimum human exposure to these constituents. An extremely conservative interpretation of EPA's recommendation is that concentrations of these pesticides should be less than detection limits. However, each of these pesticides was detected at least once in the Deep Fork River near Arcadia.

341. Significantly, the EPA has suspended the production and use or severely restricted the use of each of these pesticides. Permissible uses of pesticides that were not totally banned include termite control, nursery root dips, and other uses in which washoff and subsequent contamination of waterways are minimized. The EPA suspensions and restrictions should result in a gradual decrease in concentrations in the environment.

342. The pattern of occurrence, as well as the magnitude of contamination from restricted pesticide residuals in the Arcadia Lake watershed, depends upon the rates of pesticide degradation and land use activity. Reported degradation times defined as the time required for 75-percent degradation in soils varies from 2 yr for heptachlor and aldrin to 5 yr for chlordane.[94] Land-use activity, especially construction, substantially increases the washoff of particulates with adsorbed contaminants from the watershed.

343. The existing data do not permit precise quantification of the time required for pesticide residuals to degrade to below detection limits in the proposed Arcadia Lake watershed. However, a conservative estimate for the most persistent and most frequently occurring pesticide chlordane may be made. Assuming watershed soil concentrations equal to measured stream sediment concentrations, no losses except through degradation at a rate reported above, and a detection limit equal to 0.1 µg/kg (one-tenth of the chlordane detection limit of the analytical procedures used in this study), 14 yr would be required for chlordane concentrations in watershed soils to decrease to below detection limits.

344. This analysis overestimates the persistence of chlordane and provides no information on the possible ecological effects of chlordane in the environment. The assumption of equal concentrations in stream sediments and watershed soils results in an overestimation of soil concentrations. Quantitative relationships between soil and stream sediment concentrations are not known. They most likely depend upon land-use activity, watershed characteristics, and meteorological conditions. Sediment concentrations are expected to exceed watershed soil concentrations because the stream sediments represent an accumulation of the more easily erodible surface soils. The assumption of degradation as the only process operating to decrease residual concentrations likewise results in an overestimation of persistence. Erosion with subsequent transport out of the system represents a significant process operating to decrease watershed concentrations.

345. As previously discussed, sediment concentrations provide a poor indication of possible adverse effects. Adverse ecological effects are dependent upon the bioavailability of the pesticide while concentrations in raw water supplies are dependent upon the phase distribution of the pesticide and particulate sedimentation characteristics of the impoundment. A reasonable interpretation of this discussion is that a ban on the use of a pesticide will result in decreasing environmental concentrations and that if chlordane were to be totally banned, residual concentrations might be detected in sediment sinks for approximately a decade.

Fish tissue analysis

346. To provide a more definitive approach to the assessment of the magnitude of organochlorine pesticide contamination of the upper Deep Fork River, fish inhabiting the river were collected and their flesh analyzed for pesticides. A collection of fish consisting of green sunfish, orange spotted sunfish, bluegill sunfish, and black bullhead was obtained by TD by electrofishing in the Deep Fork River at the proposed impoundment site. The samples were sent to NTSU for analysis. Whole-fish samples for pesticide analysis were ground, digested, extracted, and trace organic content determined by gas chromatography. A report prepared by NTSU detailing the methods of sample collection, preparation, and analysis and presentation of results is given in Appendix D.

347. The estimates of the population mean pesticide concentrations for dieldrin, DDT, and heptachlor epoxide were 0.100, 0.013, and 1.23 mg/kg, respectively (Table 29). Comparison of the upper 99-percent confidence limit on the population mean for these pesticides indicates that the population mean concentrations of dieldrin and DDT were less than FDA administrative guidelines while heptachlor epoxide exceeded the guideline. No known limits exist for lindane and chlordane in fish

Pesticide	Population Mean Estimate mg/kg	99% Upper Confidence Limit, mg/kg	FDA Administrative Guideline mg/kg
Dieldrin	0.100	0.165	0.3
DDT	0.013	0.024	5.0
Heptachlor epoxide	1.23	2.11	0.3

tissue. Concentrations of dieldrin, DDT, chlordane, and heptachlor epoxide in fish inhabiting the Deep Fork River near Arcadia are expected to decrease because of recent EPA restrictions on the use of these pesticides.

348. The fish flesh analyses are considered inconclusive with

respect to heptachlor. Gas chromatographic analyses for pesticides provide a screening methodology that indicates possible pesticide contamination when eluted peaks from samples have similar retention times to known pesticide standards. The lack of eluted peaks with retention times equal to that of known pesticide standards does conclusively indicate pesticide absence or concentrations less than detection. However, if detectable peaks are eluted with retention times equal to that of known pesticide standards, confirmation of the identity of the contaminant requires additional analysis such as mass spectrometry. The conclusion that can be made about heptachlor in fish tissue is that heptachlor epoxide or other chemical compounds with a similar retention time was detected in the tissues of fish collected in the Deep Fork River.

349. The rare occurrence of heptachlor in the Deep Fork River, the failure to detect heptachlor epoxide in Deep Fork water samples, the fact that the EPA has banned the distribution and use of the pesticide, and the observation that heptachlor strongly adsorbs to settling particulates suggest that heptachlor and its degradation products are not expected to exceed criteria in the water column of the proposed impoundment. Significantly, no heptachlor or heptachlor epoxide were detected in Deep Fork River sediments.

350. However, because of the measurement of heptachlor epoxide or other chemical compounds with a similar retention time in significant concentrations in the fish, it is recommended that additional water, sediment, and fish samples be analyzed. If heptachlor or heptachlor epoxide were again detected in significant concentrations, confirmatory analysis by mass spectrometry should be conducted.

351. The fish species collected from the Deep Fork River and analyzed for pesticides feed predominantly on insects, other fish, and crustaceans. Thus these fish are positioned in a food chain whose origin is in the sediments and associated particulate materials. Since pesticides and related compounds generally exhibit sorption tendencies with particulates, the evidence suggests that concentrations of pesticides in fish in the proposed impoundment would be less than observed in the Deep Fork River.

Mass-balance calculations

352. Mass-balance calculations were used to prepare concentration duration curves of pesticides in Arcadia Lake. Simulations were made assuming (a) conservative behavior, (b) degradation, and (c) degradation and loss to sediments. Assuming conservative behavior, median simulated impoundment concentrations of DDT and chlordane were equal or greater than average measured stream concentrations (Figures 7 and 8).

353. Degradation rates for the pesticides were taken from a paper by Pionke and Chesters.[94] Persistence time was defined as the time required for 75-percent chemical degradation. Transformation of persistence times of 4 yr for DDT and 5 yr for chlordane provided the following daily decay rates:

	Decay Rate
DDT	9.4952×10^{-4} day^{-1}
Chlordane	7.5961×10^{-4} day^{-1}

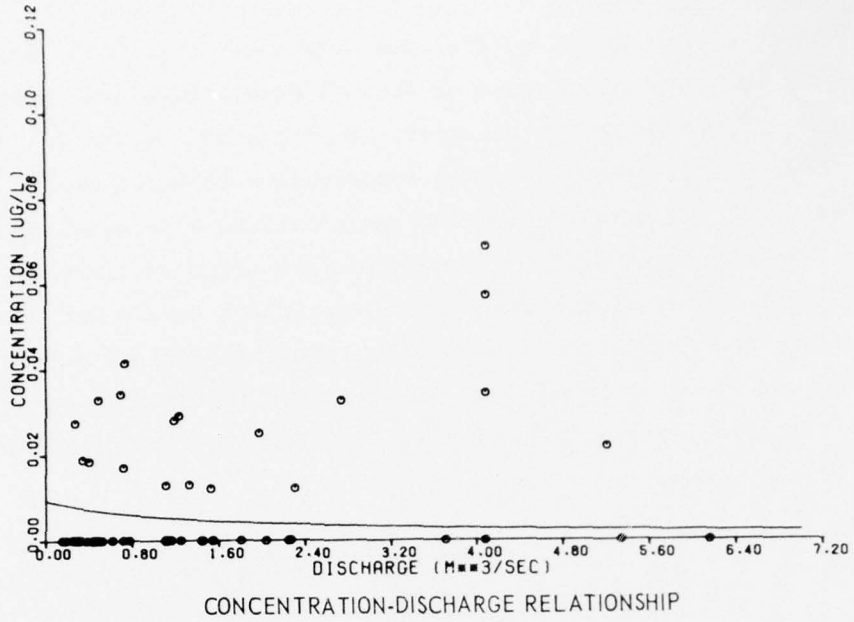
354. The incorporation of decay resulted in a decrease of the median simulated DDT concentration from 0.005 to 0.002 µg/ℓ, a 60-percent decrease. Corresponding decrease for chlordane was from 3 to 1 µg/ℓ, a 67-percent decrease.

355. Simulations were conducted assuming 30- and 60-percent losses to sediments in addition to degradation. Simulated DDT median concentrations were <0.002 and 0.001 µg/ℓ assuming 30- and 60-percent losses, respectively. Respective chlordane concentrations were <1 and 0.82 µg/ℓ.

356. Mass-balance calculations probably do not provide reliable quantitative estimates of expected concentrations. The simulations do suggest that, using the assumptions presented previously, sedimentation is a significant process decreasing water column concentrations of pesticides.

Summary

357. The USGS data for concentration of pesticides in the Deep Fork River indicate that a majority of the compounds exist at levels below the limits of EPA criteria. In addition, data from sources other



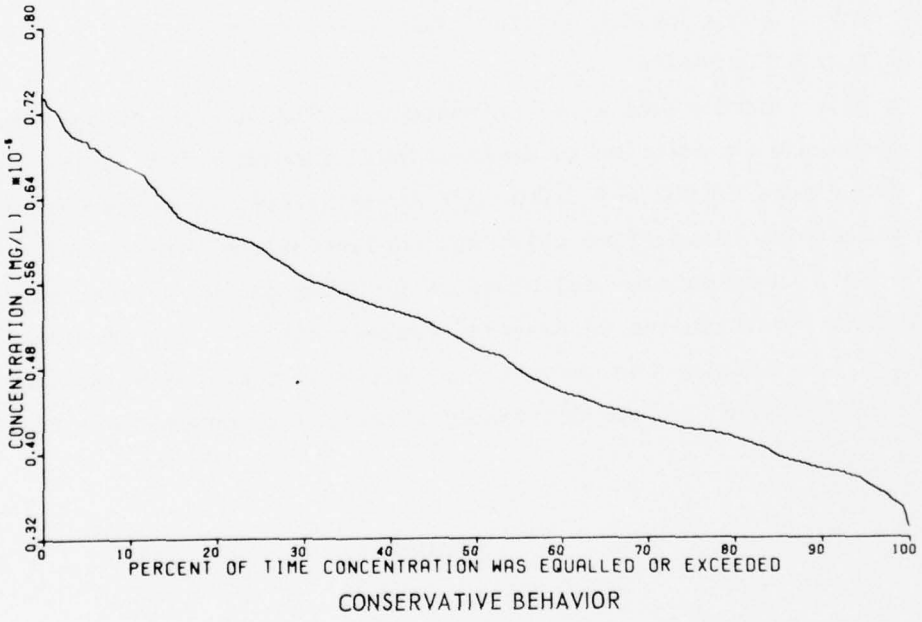
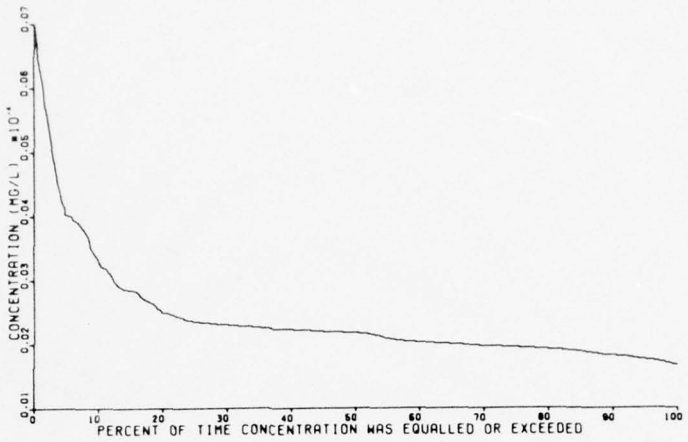
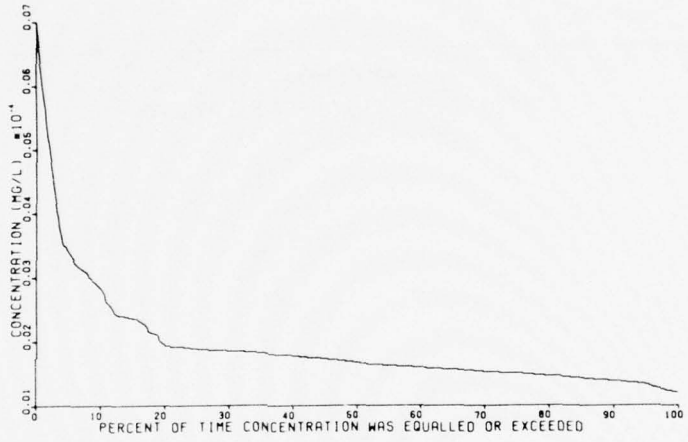


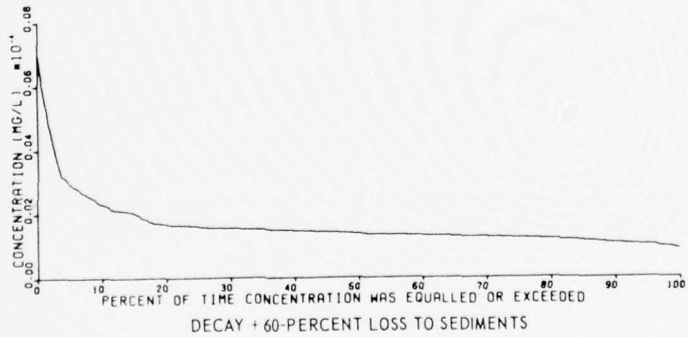
Figure 7. DDT duration curves based on daily concentrations



DECAY

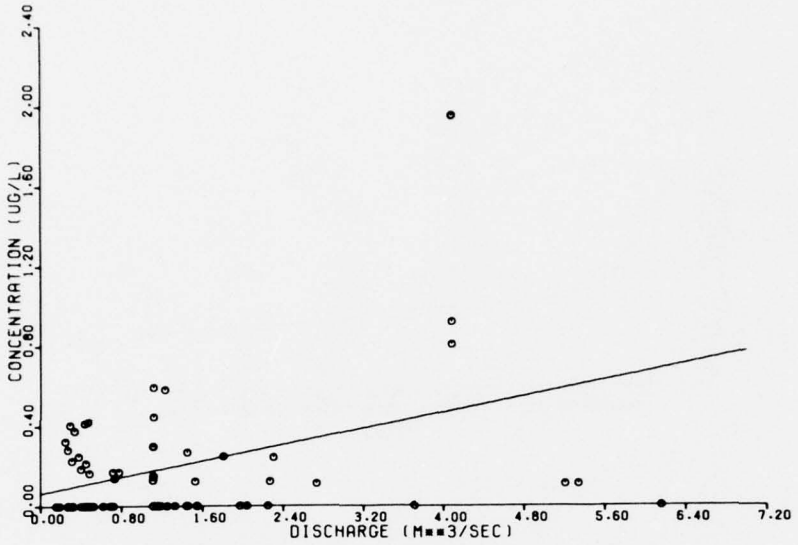


DECAY + 30-PERCENT LOSS TO SEDIMENTS

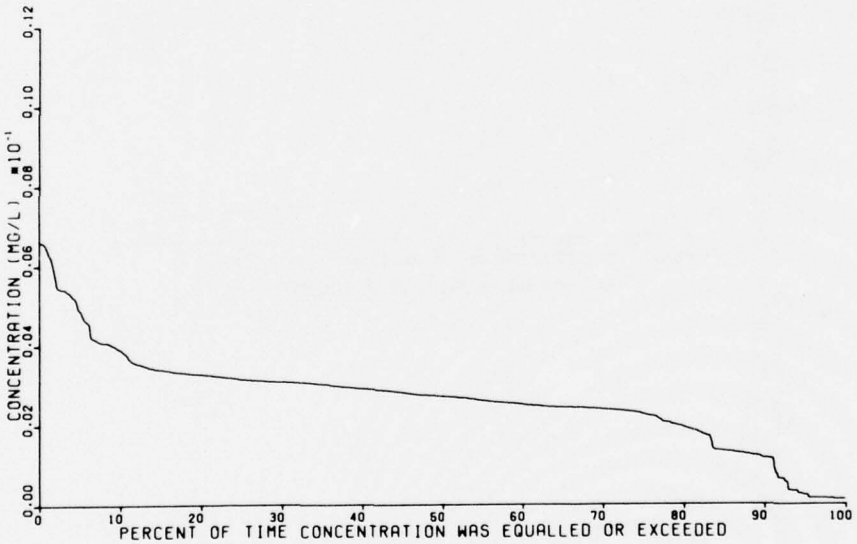


DECAY + 60-PERCENT LOSS TO SEDIMENTS

Figure 7. (Concluded)



CONCENTRATION-DISCHARGE RELATIONSHIP



CONSERVATIVE BEHAVIOR

Figure 8. Chlordane duration curves based on daily concentrations

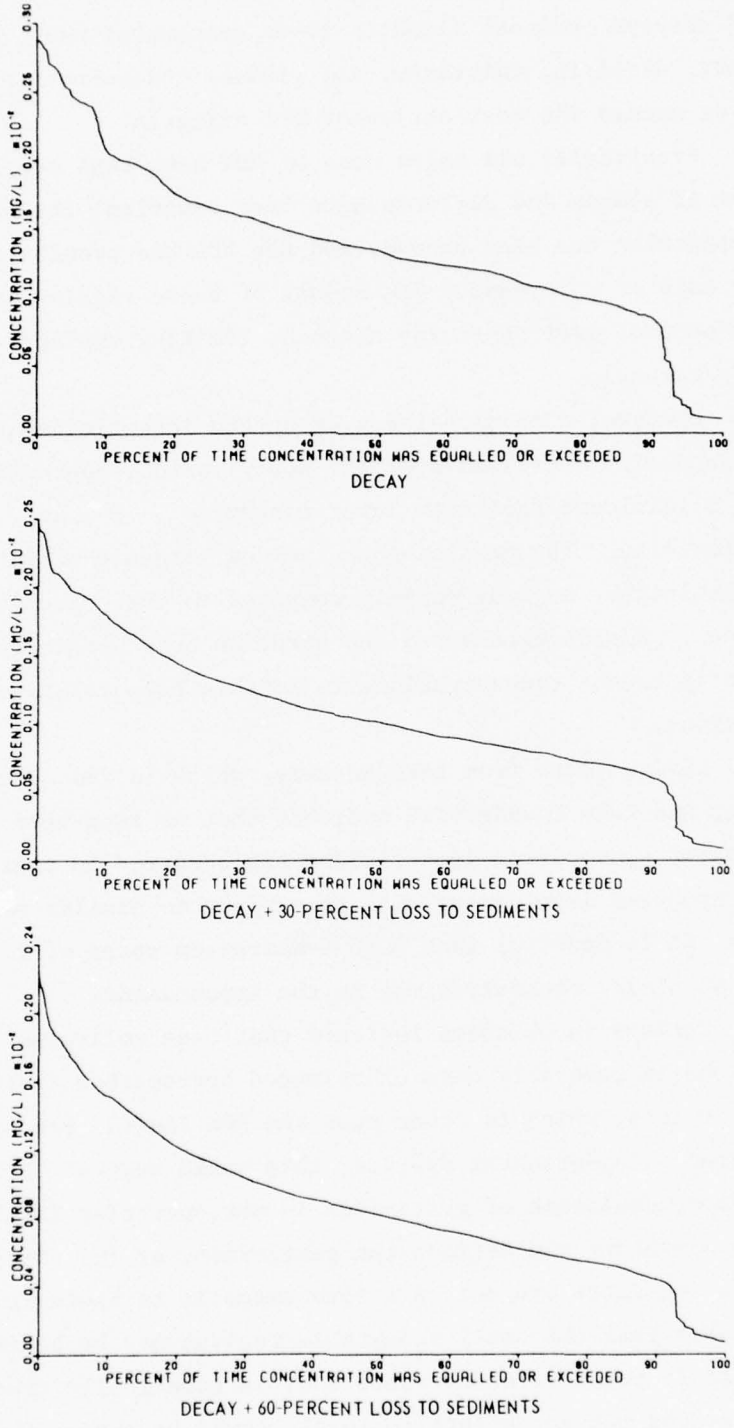


Figure 8. (Concluded)

than USGS surveys indicate slightly lower concentrations. However, average DDT, dieldrin, chlordane, and lindane concentrations were found to equal or exceed the most stringent EPA criteria.

358. Practically all major uses of DDT have been suspended; most major uses of aldrin and dieldrin have been canceled; the production and use of heptachlor has been banned; and the EPA has prohibited many of the major uses of chlordane. The result of these regulations will be increased control over pesticide usage by limiting application to licensed personnel.

359. Lindane, a chlorinated hydrocarbon insecticide whose concentrations generally decreased over the study period, degrades at a potentially significant rate into lower energy α, β, and δ forms, reducing toxicity. The remaining pesticides, organophosphates and phenoxy herbicides, degrade rapidly compared to the organochlorine insecticides. Rapid degradation and sorption onto particulates should significantly reduce concentrations in Arcadia Lake relative to tributary concentrations.

360. Limited data from Lake Eufaula, which is fed by the Deep Fork River, and Lake Thunderbird indicate that no long-term accumulation of chlorinated hydrocarbon insecticides has occurred in these waters. Since the proposed impoundment will be subject to similar meteorological conditions, it is doubtful that high evaporation rates will result in increased pesticide concentrations in the impoundment.

361. Surveys in Oklahoma indicate that bass collected in the Deep Fork River basin generally have chlorinated hydrocarbon residues of 0.5 mg/kg or less, which is lower than the FDA limit. Since bass would be considered a top-predator species, this would suggest that (a) significant bioaccumulation of pesticides is not occurring in the food web, or (b) the organisms can degrade the pesticides, or (c) the pesticides measured in the water are not in a form amenable to biologic uptake. It is conceivable that the small measurable residue may be beneficial to the organisms if it produces a tolerance that is genetically transmitted and the residue does not accumulate to levels toxic to susceptible fish-eating organisms.

362. It is doubtful that the pesticide levels under consideration would be a direct hazard to the general public. Chesters and Konrad have reported that no evidence exists demonstrating that long-term consumption of water with low pesticide levels has created a public health hazard.[93]

363. A review of the technical literature suggests that sorption and bioaccumulation are the dominant aspects of the water chemistry of pesticides. Both of these reactions tend to remove pesticides from the water and increase the rate of degradation. This has the general effect of reducing the biological availability and toxicity of chlorinated hydrocarbon residues.

364. Pesticide residues are apparently not an acute problem in the Deep Fork River at this time and a review of the literature would suggest that present conditions would not cause long-term chronic problems.

365. Because heptachlor epoxide or other chemical compounds with a similar retention time were measured in significant concentrations in fish, it is recommended that additional water, sediment, and fish samples be analyzed. If heptachlor or heptachlor epoxide are again detected in significant concentrations, confirmatory analysis by mass spectrometry should be conducted.

PART IX: OTHER WATER-QUALITY PARAMETERS

366. Phenols, TDS, and coliform bacteria were identified as possible water-quality problems in the proposed impoundment. These parameters cannot be conveniently classified into the generic classes of nutrients, metals, or pesticides and thus are discussed separately.

Phenols

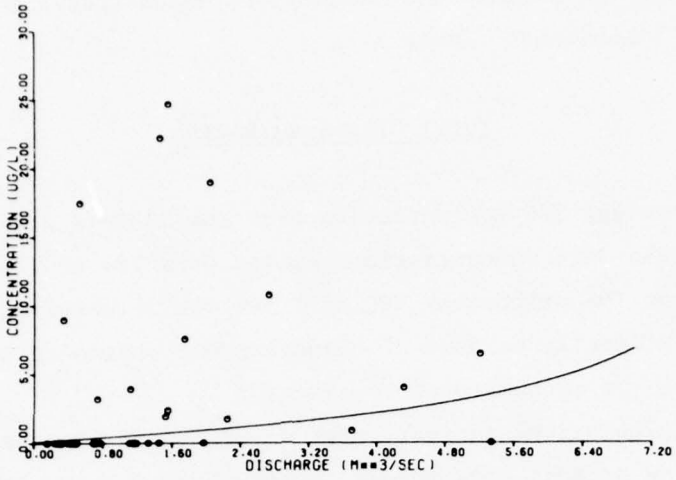
367. The average of phenol concentrations measured over the period 1969-74 was 5 µg/ℓ (Table 38). Concentrations in 39 samples ranged from 0 to 22 µg/ℓ with 74 percent of the sample values exceeding the most stringent criterion of 1 µg/ℓ, which was established based on taste and odor considerations.

368. Average phenol concentration measured in the final effluent of the Northside STP was 13.3 µg/ℓ, considerably greater than most measured instream concentrations near Arcadia. The calculated effect of STP diversion would be to reduce phenol concentrations 20 percent and decrease loadings 40 percent.

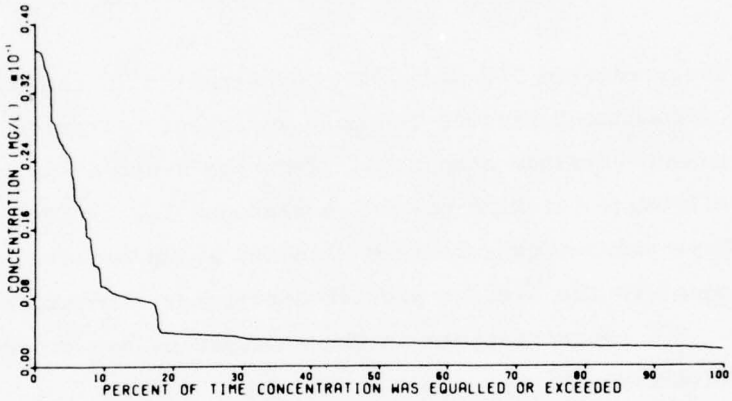
369. Phenol concentrations in Arcadia Lake are expected to be much lower than predicted stream concentrations. Phenol is a very labile compound in natural waters with reported persistence times varying between 2 to 8 days.[125]

370. Mass-balance calculations were used to develop daily concentration duration curves (Figure 9). Median simulated phenol concentration assuming conservative behavior was 3 µg/ℓ, which is 25 percent less than predicted stream concentrations.

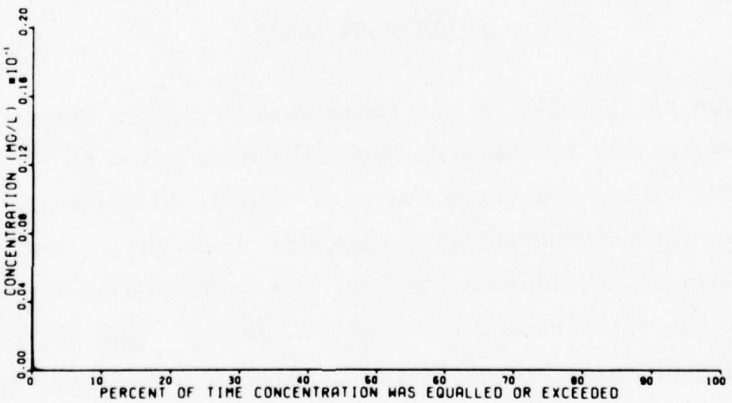
371. Assuming a persistence time of 8 days for 99-percent decay, the daily decay rate of phenol was 0.5756. The incorporation of this value into the daily simulations resulted in a decrease of the median concentration to zero (Figure 9). Although the mass-balance calculations may not provide a precise quantitative estimate of expected concentrations, the simulation does demonstrate the significance of decay on phenol concentrations. Examination of phenol concentration data



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CONSERVATIVE BEHAVIOR



DECAY

Figure 9. Phenol duration curves based on daily concentrations

collected at Lakes Eufaula and Thunderbird demonstrates a reduction in phenol with impoundment (Table A4).

Total Dissolved Solids

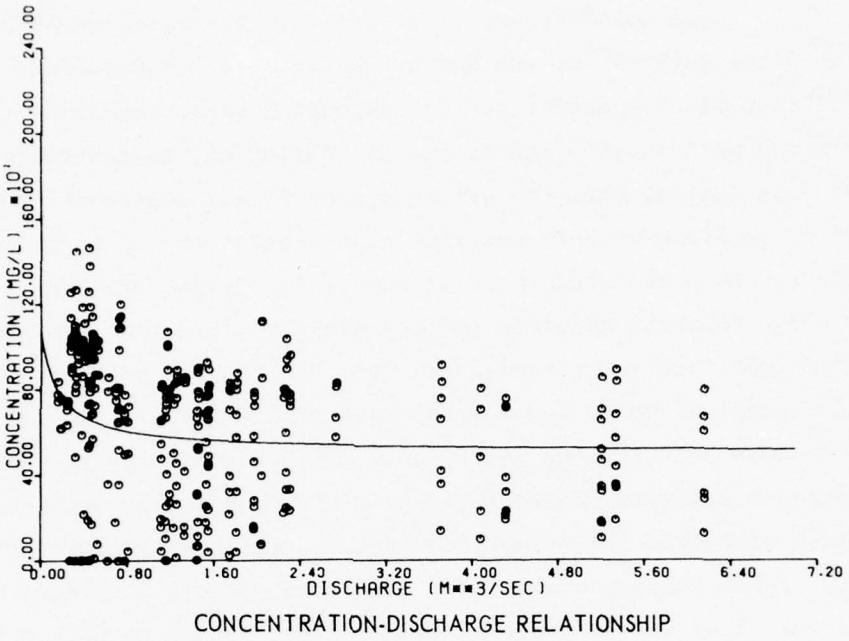
372. Average TDS concentration over the 1969-74 study period was 794 mg/ℓ (Table 10); concentrations varied from 184 to 1140 mg/ℓ. The most stringent TDS criterion, 500 mg/ℓ for public water supplies, was established primarily because of objectionable physiological effects, mineral taste, or economics of treatment.

373. Calculations indicate that diversion of the sewage treatment effluent below Arcadia will result in less than a 1-percent reduction in stream concentrations and a 42-percent reduction in loadings (Table 10).

374. The predicted 792 mg/ℓ TDS concentration in the Deep Fork River after impoundment exceeds the most stringent criterion, but it is less than present Oklahoma standards. Oklahoma's standards explicitly provide an allowance for high natural background levels for this constituent. Mass-balance calculations resulted in an expected median concentration equal to the average predicted tributary concentration (Figure 10). TDS concentrations in the impoundment are expected to behave conservatively.

Coliform Bacteria

375. Two total coliform and three fecal coliform sample values measured near Arcadia by the USGS were available prior to the initiation of this study. Total coliforms averaged 64,000/100 ml, while fecal coliforms averaged 44,000/100 ml. Although these values are reasonable for urban drainage, additional sampling was initiated by the TD to characterize the distribution of coliform numbers. The Oklahoma State Health Department collected duplicate samples of water from the Deep Fork River near Arcadia and the final effluent of the Northside STP and analyzed them for both total and fecal coliforms. The sampling was



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Figure 11. TDS duration curves based on daily concentrations

conducted weekly from 29 March through 1 June 1976.

376. Total coliform and fecal coliform bacterial numbers measured in the final effluent of the Northside STP were 3,400,000/100 ml and 330,000/100 ml, respectively. Corresponding measurements in the Deep Fork River were 200,000/100 ml and 34,000/100 ml, respectively (Table 39). These data suggest that the STP is a significant source of bacteria. However, coliform numbers measured near Arcadia are of a similar magnitude to those measured in urban runoff for Tulsa, Oklahoma.[29]

377. Coliform bacteria rapidly die when isolated from their source. Reported coliform decay rates vary from 0.5 to 2.46 per day. Using the WQRRS ecological model and a decay rate of 1.4 per day, it was estimated that sources contributing 16,000 to 1,150,000 coliforms per 100 ml would not produce coliform concentrations in the proposed impoundment that exceeded standards for primary contact recreation. However, the WQRRS model assumes homogeneously mixed horizontal layers of water, which would not occur in the proposed impoundment. Based on this characteristic nonhomogeneity and observations of the coliform distribution in Lakes Eufaula and Thunderbird, the conclusions are that coliform concentrations in the upper reaches of the reservoir would exceed primary contact recreational standards, but waters near the dam would be suitable for primary body contact recreation with the possible exception of short periods following major storm events. Selection of recreational areas should reflect the expected distribution of coliform bacteria, and routine bacteriological monitoring should be conducted to detect possible bacterial contamination of recreational areas.

378. Examination of fecal coliform and fecal streptococcus data collected on Lakes Thunderbird and Eufaula demonstrates a rapid decrease in bacterial numbers along the length of the impoundments (Table A4).

PART X: CONCLUSIONS AND RECOMMENDATIONS

379. Based on the results of several study approaches, the following conclusions concerning the water quality in the proposed Arcadia Lake are considered justified.

Critical Parameters

380. A comparison of average values for 70 parameters from previous studies with the most stringent standard or criterion indicated that 12 water-quality parameters in the Deep Fork River equaled or exceeded the permissible or recommended levels. These parameters were ammonia, manganese, mercury, DDT, aldrin, dieldrin, chlordane, heptachlor, lindane, PCB, phenols, and fecal coliforms. Only three parameters in this list, coliform bacteria, ammonia, and manganese, exceeded present Oklahoma standards applicable to the use classification of the Deep Fork River or proposed reservoir uses. Ammonia would not be expected to reach toxic concentrations in the epilimnion of Arcadia Lake or to interfere with project purposes because of oxidation to nitrate.

Temperature and DO

381. Modeling efforts suggested that the reservoir would be weakly stratified from mid to late spring through late summer or early autumn. A selective withdrawal structure would be required for the proposed project for water-quality management purposes. Arcadia Lake release temperatures from a selective withdrawal structure would be adequate to meet the annual variation of natural stream temperatures downstream. The hypolimnion would become anoxic during various periods of stratification. However, DO concentrations in releases should be satisfactory because of expected 80- to 90-percent reaeration as the water passes through the outlet works into the stilling basin.

Nutrients and Eutrophication Potential

382. Nutrients are abundant in the Deep Fork River with average

concentrations of 4.0 to 5.5 mg P/ℓ and 7.0 to 12.0 mg N/ℓ depending on the method of sample collection. Sewage treatment plant effluent diversion would reduce phosphorus loadings 2 to 26 percent and nitrogen loadings 28 to 54 percent depending on the data source used in calculations.

383. An evaluation of nitrogen and phosphorus in the Deep Fork River based on concentrations and loading analyses suggested that the proposed impoundment would be eutrophic. Bioassay results suggested that all the available nutrients in the Deep Fork River were apparently not being used because of light limitation. The mathematical model simulations also suggested that the algal response in the proposed impoundment would be insensitive to nutrient fluctuations.

384. A nutrient availability study indicated that available phosphorus and nitrogen concentrations at Arcadia are 0.3 mg P/ℓ and 5.0 mg N/ℓ, respectively. This suggests that available nutrients are much lower than average phosphorus and nitrogen concentrations, 4.0 to 5.5 mg P/ℓ and 7.0 to 12.0 mg N/ℓ, at Arcadia. The availability study also indicated lower available nutrient concentrations in Lake Eufaula relative to the Deep Fork tributary and in Lake Thunderbird relative to the Little River tributary. It can thus be anticipated that nutrient concentrations in the proposed impoundment would be lower than historical data for the Deep Fork River at Arcadia. However, nutrient concentrations would not be sufficiently low to limit algal growth.

385. Nutrient spiking results indicate that the relative abundance of nitrogen and phosphorus would be affected by impoundment of the Deep Fork River and sewage effluent diversion. At present nitrogen concentrations are lowest relative to need at Arcadia, but phosphorus concentrations are lowest relative to need above the Northside STP.

386. Mathematical model simulations suggest that occasional nuisance algal blooms would be expected after impoundment. However, the algal population changes in model simulations were controlled by the degree of light penetration and water temperature rather than nitrogen or phosphorus concentrations. This is in good agreement with the bioassay results.

387. Algal blooms in Arcadia Lake would not be expected to be significantly different in frequency or intensity than those occurring in surrounding eutrophic waters. Future land-use changes would not be expected to affect lake eutrophication since light rather than nutrients would be the limiting factor. Based on a limited evaluation, it does not appear feasible to reduce the eutrophication potential of Arcadia Lake significantly by either watershed or existing in-lake management techniques.

Metals

388. Manganese concentrations, and occasionally iron concentrations, can be expected to exceed drinking water standards, particularly in the hypolimnion and headwaters of the proposed impoundment. Iron and manganese would be less likely to exceed standards in the epilimnion near the dam. The selective withdrawal capability of the proposed project would enable project operation to minimize iron and manganese concentrations in water supply withdrawals and downstream releases. Furthermore, excessive iron and manganese would not be a problem in finished water supplies if the potential problem is recognized in the design of the water treatment plant. It should be pointed out that the standards for iron and manganese are based on aesthetics. Toxic conditions would not be expected.

389. Earlier studies of the Deep Fork River concluded that lead would be a water-quality problem in the proposed impoundment. This conclusion was based on a technically invalid comparison of total lead concentrations with standards derived from studies using dissolved lead. A comparison using dissolved lead concentrations indicated that soluble lead in the Deep Fork River is 10 times lower than the standard for water supply and the protection of freshwater aquatic life. It was concluded that lead would not be a water-quality problem in the proposed impoundment at Arcadia. Analysis of fish flesh content of lead substantiated this conclusion.

390. The average mercury concentration of 73 water samples analyzed

between 1971 and 1976 is 0.2 µg/ℓ. This value is 10 times lower than recommended limits for public water supply. It is further anticipated that mercury concentrations in the proposed impoundment would decrease relative to input in much the same way that trace metal concentrations in surrounding impoundments are less than their respective input concentrations. These data indicate that excessive mercury concentrations would not interfere with water supply purposes of the proposed impoundment. Mercury concentration in fish collected in the Deep Fork River near Arcadia was less than FDA limits for food source and EPA limits for the protection of freshwater aquatic life. Because the food chain of the fishes that were examined originates in the sediments and associated particulates, concentration of mercury in fish in the impoundment is expected to be less than values measured in fish in the stream prior to impoundment.

Pesticides and PCB

391. Using USGS data from the Arcadia gaging station, DDT, dieldrin, aldrin, chlordane, heptachlor, lindane, and PCB were found to exceed the most stringent EPA criteria for the protection of freshwater aquatic life. Data from other sources suggest that pesticide contamination is restricted to the upper Deep Fork River.

392. Average yearly lindane concentrations have been decreasing, which should lessen the concern for this parameter. Also, recent action by EPA to restrict the use of DDT, dieldrin, aldrin, heptachlor, PCB, and chlordane should also lessen the potential problems associated with these materials. A comparison of fish-flesh concentrations of pesticides in fish collected in the Deep Fork River near Arcadia revealed that measured concentrations of DDT and dieldrin were less than published numerical FDA limits while heptachlor epoxide exceeded the guideline. No limits for chlordane and lindane in fish tissue are known to exist. Concentrations of pesticides in reservoir fish following impoundment are expected to be less than those observed in the Deep Fork River.

393. Heptachlor epoxide concentrations in fish exceeded FDA

administrative guidelines. However, the rare occurrence of heptachlor, the failure to detect heptachlor epoxide in Deep Fork River water, the failure to detect heptachlor or heptachlor epoxide in river sediments, and the fact that the EPA has banned the distribution and use of the pesticide suggest that heptachlor and its degradation products are not expected to exceed criteria in the proposed impoundment.

394. Because heptachlor epoxide or other chemical compounds with a similar retention time were measured in significant concentrations in fish, it is recommended that additional water, sediment, and fish samples be analyzed. If heptachlor or heptachlor epoxide are again detected in significant concentrations, confirmatory analysis by mass spectrometry should be conducted.

Other Parameters

395. Based on an analysis of Deep Fork River data, phenols, TDS, and coliform bacteria are potential problems in the proposed impoundment. Phenol concentrations should decrease due to sewage plant effluent diversion and decay, but at times phenol concentrations in the reservoir's headwaters might still exceed the most stringent standard. TDS would exceed desirable criteria but would still meet acceptable standards. Concentrations of coliform bacteria in the headwaters of the impoundment would exceed primary contact recreational standards, but waters in the lower two-thirds of the impoundment should be suitable for primary body contact recreation. Selection of recreational areas should reflect the expected distribution of coliform bacteria and be restricted to the lower one-half of the impoundment for safety reasons. Routine bacteriological monitoring should be conducted to detect possible bacterial contamination of recreational bathing areas.

396. An evaluation of all water-quality data for the Deep Fork River suggested that toxicological problems would not be expected in the proposed Arcadia impoundment. Iron, manganese, phenol, and TDS presently exceed criteria established on aesthetics (possible staining, odor, and taste). Also, nutrient levels would be high enough to support algal

blooms during times when sufficient light is available. Because of the marginal acceptability of these constituents, it is recommended that sampling of water-quality constituents be continued in the Deep Fork River near the proposed damsite. Continued sampling of inflows and in-lake conditions would permit evaluation of reductions in contaminant loadings due to municipal and industrial effluent diversions and would provide the data base for operation of the proposed project using model evaluations of various management strategies.

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Table 1

Maximum Concentrations, Loadings, and Relative Contribution of Major Industrial Effluents in the Arcadia Lake Watershed Permitted Under Oklahoma's Permit Management Plan

Source	Water Quality Constituent	Waste Load Allocation* Concentration mg/ℓ	Waste Load Allocation* Loading kg/day	Loading at Arcadia** 1969-1974 kg/day Grab	Loading at Arcadia** 1969-1974 kg/day Composite	Percent of Loading at Arcadia Grab	Percent of Loading at Arcadia Composite
Dowell, Div. of Dow Chemical	P	3.2	0.024	730	650	<1	<1
	Pb	0.08	<0.001	1.3	--	<1	--
	Hg	0.002	<0.001	†	--	--	--
	Mn	0.08	<0.001	110	--	<1	--
	NH_4-N	1.8	0.01	†	--	--	--
	NO_2-N	0.2	0.002	69	--	<1	--
	NO_3-N	0.2	0.002	130	--	<1	--
Macklanburg - Duncan Co.	NO_3-N	1.7	0.93	130	--	<1	--
	P	140	76.26	730	650	12	12
	Pb	0.19	0.10	1.3	--	8	--
	Mn	0.20	0.11	110	--	<1	--
O.G.&E. Belle Isle Plant	NH_4-N	0.12	4.54	--	--	--	--
	P	0.26	9.84	730	650	2	--
	NO_3-N	0.46	17.4	130	--	13	--

* From Reference 6.

** Weighted average over years; annual weights proportional to relative number of samples collected in a year.

† Insufficient data to calculate realistic loadings.

Table 2

Summary of Municipal Sewage Effluent Concentrations of Selected Water-Quality Constituents

Source	Reference	Flow m^3/day	Concentration of Constituents* Organic-N mg/ℓ	NH_4-N mg/ℓ	NO_2-N mg/ℓ	NO_3-N mg/ℓ	Total-N mg/ℓ	Total-P mg/ℓ	TDS mg/ℓ
Northside Sewage Treatment Plant (STP)		4.2×10^4	--	--	--	--	--	--	--
Plant operations charts, 1969-1974	8	--	9.4 (52)*	18.7 (53)	0.11 (34)	0.7 (45)	--	--	--
Effluent, 21 April 1971	9	--	--	23.5	0.02	0.3	--	45.7	900
Effluent, April 1972	9	--	--	--	--	1.6	11.0	11.7	--
Below STP, 1972-1974	9	--	--	--	--	1.3 (12)	--	10.3 (21)	770 (16)
Average		--	9.4	18.8	0.11	1.23	11.0	12.0	780
Nichols Hills STP		1.9×10^3	--	--	--	--	--	--	--
Effluent, 4 March 1971	9	--	0.1	2.4	0.28	35.6	--	22.8	900
Effluent, 21 April 1971	9	--	--	9.9	0.27	10.8	--	25.0	--
300 ft below STP, 16 March 1971	9	--	--	--	--	17.6	--	--	--
Average		--	0.1	6.2	0.28	21.3	--	23.9	900
Edmond Southeast STP**		1.9×10^4	--	--	--	--	--	--	--
Effluent, 22 April 1971	9	--	--	20.0	0.32	6.7	--	44.5	700
Effluent, 8 June 1972	9	--	--	--	--	--	--	8.5	900
3/4 mile below STP, 1971-1974	9	--	--	--	--	2.4 (16)	--	10.9 (18)	700 (18)
Average		--	--	20.0	0.32	2.6	--	12.5	710
Turnpike Park Lagoon		1.5×10^3	--	--	--	--	--	--	--
Effluent, 21 April 1971	9	--	--	36.0	0.16	0.9	--	42.0	1200
1/3 mile below lagoon, 1971-1974	9	--	--	--	--	7.2 (16)	--	10.8 (17)+	830 (18)
Average		--	--	36.0	0.16	6.8	--	12.5	850
Broadway Park Lagoon**	9	--	--	--	--	--	--	--	--
Smiling Hills Lagoon**		3.8×10^2	--	--	--	--	--	--	--
1/8 mile below lagoon, 1971-1974	9	--	--	--	--	1.5 (14)	--	18.0 (13)	540 (16)

* Numbers in parentheses are the number of observations. Estimates used in the calculation of the averages were weighted in proportion to the number of observations in the estimate.

** Plant presently not in operation.

+ One value of 228 mg-P/ℓ measured on 5 May 1973 not included in estimate.

Table 3

Concentration, Loadings, and Relative Contribution of Municipal Effluents in the Arcadia Lake Watershed

Source	Water-Quality Constituent	Concentration mg/ℓ	Loading kg/day	Loading at Arcadia 1969-1974, kg/day		Percent of Loading at Arcadia	
				Grab	Composite	Grab	Composite
Northside STP	NH_4^+-N	18.8	780	1500**	--	52	--
	NO_2-N	0.11	5	69	--	6	--
	NO_3-N	1.23	51	130	--	39	--
	Total-N*	20.1	840	1700†	920	49	91
	Total-P	12.0	500	730	650	68	77
Nichols Hills STP	NH_4-N	6.2	12	1500**	--	1	--
	NO_2-N	0.28	0.52	69	--	1	--
	NO_3-N	21.3	40	130	--	30	--
	Total-N*	27.8	52	1700†	920	3	5
	Total-P	23.9	45	730	650	6	7
Edmond Southeast STP	NH_4-N	20.0	380	1500**	--	25	--
	NO_2-N	0.32	6.0	69	--	8	--
	NO_3-N	2.6	49	130	--	37	--
	Total-N*	22.9	430	1700**	920	25	47
	Total-P	12.5	240	730	650	32	37
Turnpike Park Lagoon	NH_4-N	36.0	52	1500**	--	3	--
	NO_2-N	0.16	0.23	69	--	<1	--
	NO_3-N	6.8	9.9	130	--	7	--
	Total-N*	43.0	63	1700†	920	4	7
	Total-P	12.5	18	730	650	2	3

* Total-N = ΣNH_4-N, NO_2-N, NO_3-N.

** Value used is average NH_4-organic-N.

† Total-N = ΣNO_2-N, NO_3-N, NH_4-organic-N.

Table 4

Sampling Programs Conducted Within the Arcadia Lake Watershed

on Which the Analyses in this Report are Based

Agency, Institution, or Individual	Nutrients		Pesticides		Metals	
	Years of Record	Frequency	Years of Record	Frequency	Years of Record	Frequency
USGS	5 (1969-74)	1/month*	5 (1969-74)	1/month	5 (1970-74)	1/month
OWRB	1 (1973-74)	0.1-1/month	--	--	1 (1973)	0.25-1/month
OK State Health Dept.	5 (1970-74)	0.125-1/month	--	--	0.1 (1971)	1 sample
OSU	--	--	0.1 (1972)	1 sample	0.25 (1972-73)	4/month
OK City-County Health Dept.	3 (1971-74)	0.25-1/month	--	--	--	--
Water Pollution Control Div., Dept. Public Works (Tom Carpenter)	8 (1966-74)	1/month**	--	--	--	--
Jimmie Pigg	--	--	--	--	0.1 (1974)	1 sample

* Composite samples for "total-N" and "total-PO4," 4/month.

** Sampling more frequent, data summarized monthly.

Table 5

Sample Means of Water-Quality Parameters Measured in the Final Effluent of the Northside Sewage Treatment Plant*

Date	pH	Cl mg/ℓ	NO_2-N mg/ℓ	NO_3-N mg/ℓ	NH_4-N mg/ℓ	Total-N mg/ℓ	SO_4 mg/ℓ	PO_4-P Dissolved mg/ℓ	PO_4-P Total mg/ℓ	Phenol µg/ℓ	Specific Conductance µmhos	Temperature °C
3-12-75	7.5	260	0.17	0.23	5.29	7.7	191.0	1.51	1.49	7.0	1190	12.7
3-19-75	7.6	280	0.01	0.07	10.6	14.9	198.9	1.82	2.04	9.6	1090	15.0
3-26-75	7.3	243	0.16	0.09	10.9	12.5	147.5	1.42	1.79	28.8	1890	15.1
4-2-75	8.1	230	0.01	0.04	14.3	15.7	153.8	0.02	0.01	7.8	1390	15.0

	Mn		Fe		Pb		Hg					
	Dissolved mg/ℓ	Suspended mg/ℓ	Dissolved mg/ℓ	Suspended mg/ℓ	Dissolved mg/ℓ	Suspended mg/ℓ	Dissolved µg/ℓ	Suspended µg/ℓ	Lindane ng/ℓ	Heptachlor ng/ℓ	Aldrin ng/ℓ	Dieldrin ng/ℓ
3-12-75	0.10	<0.05	0.10	0.90	0.003	0.006	<0.1	<0.1	21	ND**	ND	25
3-19-75	0.10	<0.05	0.10	0.25	0.010	0.009	1.8	1.0	38	ND	ND	32
3-26-75	0.10	<0.05	0.05	0.30	0.004	0.008	<0.1	<0.1	36	ND	ND	21
4-2-75	<0.05	0.10	0.10	0.45	0.011	0.017	<0.1	<0.1	60	ND	ND	19

	p,p'-DDE ng/ℓ	Endrin ng/ℓ	p,p'-DDT ng/ℓ	Arochlor 1254 ng/ℓ	Chlordane ng/ℓ	Diazinon ng/ℓ	Malathion ng/ℓ	Parathion ng/ℓ	Methyl Parathion ng/ℓ	Ronnel ng/ℓ	2,4-D ng/ℓ	Silvex ng/ℓ	2,4,5-T ng/ℓ
3-12-75	ND	ND	ND	ND	ND	287	ND	ND	ND	ND	ND	442	ND
3-19-75	ND	ND	ND	153	ND	665	ND	ND	ND	ND	181	174	14
3-26-75	ND	ND	ND	252	ND	146	ND	ND	ND	ND	158	82	38
4-2-75	ND	ND	ND	90	ND	1306	ND	ND	ND	ND	584	154	134

* Sampling conducted by OSU.
** ND means nondetectable (lowest detection limit equaled 10 ng/ℓ).

Table 6

Water-Quality Parameters Measured in the Deep Fork River Whose Maximum Value Equals or Exceeds One-Tenth of the Most Stringent Criteria or Standards[27]

Parameter	Most Stringent Criterion or Standard Value	Source*	No. of Samples	Mean Value	Sample Standard Deviation	Range Maximum	Range Minimum	Percent of Samples Exceeding Criterion or Standard
Nutrients	(mg/ℓ)			(mg/ℓ)	(mg/ℓ)	(mg/ℓ)	(mg/ℓ)	
Nitrate (NO_3-N)	10.0	c	44	1.0	2.6	14.0	0.04	4
Nitrite (NO_2-N)	1.0	c	44	0.28	0.36	1.9	0.0	4
Ammonia (NH_4-N)	0.02	b	1	13.0	--	13.0	13.0	100
Metals	(µg/ℓ)			(µg/ℓ)	(µg/ℓ)	(µg/ℓ)	(µg/ℓ)	
Arsenic (As)	50	c	2	22	3	24	22	0
Barium (Ba)	1000	c	6	275	105	400	160	0
Cadmium (Cd)	10	c	28	1	2	8	0	0
Chromium (Cr)	50	c	26	2	6	30	0	0
Iron (Fe)	300	c	26	285	731	3,800	0	15
Lead (Pb)	50	c	25	4	4	14	0	0
Manganese (Mn)	50	c	26	399	242	1,100	30	96
Mercury (Hg)	0.05	b	8	1.1	1.0	2.6	0	75
Pesticides and PCB	(µg/ℓ)			(µg/ℓ)	(µg/ℓ)	(µg/ℓ)	(µg/ℓ)	
Aldrin	**	c	64	0.002	0.008	0.05	0.0	5
DDT	**	c	64	0.006	0.012	0.06	0.0	30
Dieldrin	**	c	64	0.038	0.027	0.13	0.0	95
Chlordane	**	c	65	0.15	0.32	1.7	0.0	52
Endrin	0.004	b	64	0.0003	0.0025	0.02	0.0	2
Heptachlor	**	c	63	0.002	0.01	0.08	0.0	3
Lindane	0.1	b	64	0.07	0.37	3.0	0.0	38
Malathion	0.1	b	37	0.005	0.017	0.09	0.0	14
PCB	0.001	b	28	0.003	0.019	0.1	0.0	4
Other Water-Quality Parameters	(mg/ℓ)			(mg/ℓ)	(mg/ℓ)	(mg/ℓ)	(mg/ℓ)	
Chloride (Cl_2)	250.0	c	308	177.0	84.0	400.0	12.0	17
Total dissolved solids (TDS)	949	a	306	676.0	239.0	1140.0	184.0	72
Phenols	0.001	c	39	0.005	0.006	0.022	0.0	74
Sulfates (SO_4)	162.0	a	308	112.0	43.0	180.0	16.0	6
Fecal coliforms	200/100 ml	a,d	3	44,000	69,000	123,000	3,000	100

* Source of criterion or standard indicated as follows:

a - Oklahoma's Water Quality Standards[5]
EPA's Quality Criteria for Water:[6]
b - Freshwater Aquatic Life
c - Public Water Supply
d - Recreational Waters

**No numerical standard. Reference 6 recommends "minimum exposure."

Table 7

Relationship Between Composite Interval Length and Phosphorus Concentration[27]

Interval Length days	Number of Observations	Mean mg/ℓ	Notes
1	56	2.3	Table based on all composite samples collected between October 1969 and May 1974 (a total of 221 observations). The period of record extended from October 1969 through September 1974 for a total of 241 observations.
2	29	2.5	
3	21	4.0	
4	17	4.5	
5	11	4.6	
6	9	4.2	
7	6	5.1	
8	7	4.4	
9	8	5.6	
10	28	6.2	
11	14	6.3	
12	3	4.8	
13	3	5.4	
14	2	5.0	
15	4	7.5	
16	1	5.2	
17	0	--	
18	0	--	
19	1	6.6	
20	1	1.3	
	Grand Mean =	4.1	

Table 8

Annual Loadings and Average Phosphorus Concentrations at Arcadia, 1969-74[18]

Water Year*	Grab Samples		Composite Samples	
	Concentration mg/ℓ	Loading 10^3 kg	Concentration mg/ℓ	Loading 10^3 kg
1969-70	6.4	300	--	--
1970-71	5.5	200	6.2	280
1971-72	6.0	210	5.2	130
1972-73	4.6	320	4.5	280
1973-74	4.8	280	4.0	250

* Water year refers to the USGS system of records wherein a water year covers the period from 1 October of one year to 30 September of the following year.

Table 9

Summarization of Nutrient Data Compiled by Agencies Other than the USGS

Parameter	EPA-STORET Arcadia*			Deep Fork-Arcadia** Jan 66-Jul 72			Deep Fork-Arcadia† Jul 72-Jan 74			Deep Fork-Arcadia†† May 73-Sep 73		
	Max	Min	Avg	Max	Min	Avg	Max	Min	Avg	Max	Min	Avg
Total-P	31	0.85	5.5	--	--	--	--	--	--	--	--	--
PO_4-P	--	--	--	22	0.0	7.1	7.9	0.61	4.6	1.6	0.78	2.0
NH_4-N	--	--	--	18	5.6	11	21	3.4	11	1.6	0.10	0.70
NO_3-N	19	0.04	8.2	4.1	0.0	1.4	0.35	0.00	0.21	0.8	0.3	0.55

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Note: Concentrations expressed in mg/ℓ.

* From Reference 10. Concentrations transformed to elemental weights to maintain report consistency.

** From Reference 10. Concentration of o-PO_4 transformed to elemental weights. Data collected by the Oklahoma City Department of Public Works, Industrial Pollution Control.

† Oklahoma City Department of Public Works, Industrial Pollution Control, Monthly Laboratory Reports, July 1972-June 1974.

†† Oklahoma Water Resources Board, Water Quality Division, Stream Monitoring Stations, Station 55-00308-0, Printout date: 01/02/74.

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Table 10

Comparison of Existing and Predicted Concentrations and Loadings of Selected Water-Quality Constituents in the Deep Fork River, 1969-74

Parameter	Concentration*			Loading, kg		
	Pre-Diversion	Post-Diversion	Percent Change	Pre-Diversion	Post-Diversion	Percent Change
Nutrients						
Total Nitrogen: Composite samples	9.0	7.5	-17	336,000	156,000	-54
Grab samples	12.4	12.8	3	620,000	447,000	-28
Phosphorus: Composite samples	5.0	8.30	66	236,000	175,100	-26
Grab samples	5.5	9.04	64	265,000	260,000	-2
Metals						
Manganese	450	650	44	39,000	39,000	0
Mercury	1.1	--	--	--	--	--
Lead	5.0	5.5	11	490	490	0
Pesticides						
DDT	0.006	0.007	16	0.40	0.40	0
Dieldrin	0.04	0.04	0	2.06	1.67	-19
Chlordane	0.09	0.12	33	6.87	6.87	0
Lindane	0.08	0.12	50	3.22	2.92	-9
Diazinon	0.20	0.09	-55	11.03	4.20	-62
Other Water-Quality Constituents						
Phenol	5	4	-20	315	190	-40
Total Dissolved Solids	794	792	0.1	3.3×10^7	1.9×10^7	-42
Coliforms	64,000	--	--	--	--	--

* Annual estimates weighted in proportion to the number of months for which observations exist and each month for which an observation exists weighted equally regardless of the number of observations in that month. Concentrations expressed by the following units: Nitrogen and total dissolved solids = mg/ℓ; metals, pesticides, and phenol = µg/ℓ; coliforms = MPN/100 ml.

Table 11

Relationship Between Composite Interval Length and Nitrogen Concentration[27]

Interval Length days	Number of Observations	Mean mg/ℓ
1	77	4.0
2	37	6.1
3	22	6.9
4	19	7.0
5	13	6.8
6	12	8.1
7	6	7.0
8	9	10.1
9	11	8.8
10	36	10.6
11	19	10.0
12	4	9.9
13	4	9.1
14	4	11.7
15	8	12.7
16	1	4.3
17	1	10.4
18	0	--
19	1	10.0
20	1	3.4

Grand Mean = 7.2 mg/ℓ

Note: Table based on all composite samples collected between October 1969 and May 1974 (a total of 285 observations). The period of record extends from October 1969 through September 1974 for a total of 308 observations.

Table 12

Annual Loadings and Average Nitrogen Concentrations at Arcadia, 1969-74[27]

Water Year	Grab Samples								Composite Samples	
	NO_3-N		NO_2-N		NH_4-Organic-N		ΣN			
	Concentration mg/ℓ	Loading 10^3 kg	Concentration mg/ℓ	Loading 10^3 kg	Concentration mg/ℓ	Loading 10^3 kg	Concentration mg/ℓ	Loading 10^3 kg	Concentration mg/ℓ	Loading 10^3 kg
1969-70	0.17	6.8	0.42	20	14.0	670	15	700	12.0	430
1970-71	2.9	120.0	0.33	16	7.5	280	11	410	10.1	310
1971-72	0.32	11.0	0.16	7	13.0	430	13	450	8.2	230
1972-73	0.31	33.0	0.15	16	11.0	850	11	900	7.9	390
1973-74	0.79	38.0	0.27	17	10.0	580	12	640	6.6	320

Table 13

Calculated Phosporus Loadings Using Land-Use Approach

Type of Use	Yield kg/ha/yr	Preconstruction of Impoundment		Postconstruction of Impoundment		Year 2000	
		Area ha	P-Loading kg/yr	Area ha	P-Loading kg/yr	Area ha	P-Loading kg/yr
Pasture	0.18	2,200	400	1,700	300	1,700	300
Cultivated	0.75	260	200	260	200	260	200
Woodlands	0.08	5,200	420	4,100	330	4,100	330
Urban	2.8	8,900	25,000	8,900	25,000	16,000	45,000
Open	0.06	11,000	660	11,000	660	3,900	230
Domestic sewage	0.80*	145,000**	116,000	145,000**	†	250,000††	†
			143,000		26,000		46,000

* kg P/person/yr.

** Persons.

† Domestic sewage contribution will be diverted below impoundment.

†† Projected population for Year 2000.[2]

Table 14

Calculated Nitrogen Loadings Using Land-Use Approach

Type of Use	Yield kg/ha/yr	Preconstruction of Impoundment		Postconstruction of Impoundment		Year 2000	
		Area ha	N-Loading kg/yr	Area ha	N-Loading kg/yr	Area ha	N-Loading kg/yr
Pasture	8.5	2,200	19,000	1,700	14,000	1,700	14,000
Cultivated	12.0	260	3,100	260	3,100	260	3,100
Woodlands	2.4	5,200	12,000	4,100	10,000	4,100	10,000
Urban	8.8	8,900	78,000	8,900	78,000	16,000	140,000
Open	1.8	11,000	20,000	11,000	20,000	3,900	7,000
Domestic sewage	3.9*	145,000**	560,000	145,000**	†	250,000††	†
			690,000		125,000		174,000

* kg N/person/yr.
** Persons.
† Domestic sewage contribution will be diverted below impoundment.
†† Projected population for Year 2000.[2]

Table 15

Comparison of Nitrogen and Phosphorus Loadings Calculated Using Flow-Concentration Data and Land-Use Values

Time Period	Flow-Concentration Loadings, kg/yr*				Land-Use Loadings, kg/yr	
	Composite Samples		Grab Samples			
	Phosphorus	Nitrogen	Phosphorus	Nitrogen	Nitrogen	Phosphorus
Preconstruction of impoundment	236,000	336,000	265,000	620,000	690,000	143,000
Postconstruction of impoundment	175,000	156,000	260,000	447,000	125,000	26,000
Year 2000	--	--	--	--	174,000	46,000

* Loadings based on 1969-74 USGS samples.

Table 16

Volumetric Loading Rates of Nitrogen and Phosphorus to Arcadia Lake

Time Period	Flow-Concentration Loadings, $g/m^3/yr$*				Land-Use Loadings $g/m^3/yr$	
	Composite Samples		Grab Samples			
	Phosphorus	Nitrogen	Phosphorus	Nitrogen	Nitrogen	Phosphorus
Preconstruction of impoundment	2.8	3.9	3.1	7.2	8.1	1.67
Postconstruction of impoundment	2.0	1.8	3.0	5.2	1.5	0.30
Year 2000	--	--	--	--	2.0	0.54

* Loadings based on 1969-74 USGS samples.

Table 17

Summary of Parameters, Models, and Selected Statistics for Which Discharge-Concentration Relations Were Found

Parameter	Model Selected	Coefficient of Determination R^2	Intercept a	Slope b
Total-N (composite samples)	I	0.0890	2.18529	1.99736*
Total-P (composite samples)	I	0.3278	2.82582	3.00407*
TDS	I	0.1008	474.66405	87.67761*
NH_4-Org-N	I	0.1520	7.119187	2.71444*
Total-P	I	0.5784	0.41797	4.34824*
Mn	II	0.2892	4.80644	2.85757*
Phenols	III	0.2091	0.98923	-0.41016*
Chlordane	IV	0.1371	0.06398	0.10069*
DDT	V	0.0887	0.00092	0.00982*

* $Pr\ (|t_b| \geq t_{.01}) < 0.01$.

** $Pr\ (|t_b| \geq t_{.05}) < 0.05$.

Table 18

Experimental Setup for Arcadia Nutrient Availability Study

Sample*	Additions	*Selenastrum* inoculum
	Sample Conditions	
Filtered water sample	None	P-deficient
	0.37 mg PO_4-P	P-deficient
	None	N-deficient
	4.2 mg NO_3^--N	N-deficient
	4.2 mg NH_3-N	N-deficient
	Standards	
AAP-P medium	None	P-deficient
	0.37 mg PO_4-P	P-deficient
	0.74 mg PO_4-P	P-deficient
	1.12 mg PO_4-P	P-deficient
AAP-N medium	None	N-deficient
	4.2 mg NO_3^--N	N-deficient
	8.4 mg NO_3^--N	N-deficient
	16.8 mg NO_3^--N	N-deficient
	4.2 mg NH_3-N	N-deficient
	8.4 mg NH_3-N	N-deficient
	16.8 mg NH_3-N	N-deficient

* 800 ml of filtered water or standard medium used for each test.

Table 19

Experimental Conditions for

Nutrient Spiking Study

Replicates	Sample Condition
6	Initial water sample
6	Incubated water sample
6	Sample + 0.71 mg PO_4-P/ℓ
6	Sample + 1.6 mg NO_3^--N/ℓ
6	Sample + 1.6 mg NH_3-N/ℓ
6	Sample + 23 mg CO_3 as $CaCO_3$/ℓ
6	Sample + AAP trace metals
6	Sample + AAP medium

Table 20

Phosphorus Availability Results*

Standard Cultures				
Phosphorus concentration: Test duration, days	2	6	13	
0.0 mg PO_4-P/ℓ	0.5	3.4	2.4	
0.46 mg PO_4-P/ℓ	3.6	5.8	6.1	
0.92 mg PO_4-P/ℓ	3.6	5.7	6.2	
1.85 mg PO_4-P/ℓ	3.6	5.8	6.2	
Sample Cultures				
				Available Phosphorus
Conditions: Test duration, days	3	7	14	
Eastern	4.1	3.7	3.1	0.09 mg/ℓ
Eastern + 0.37 mg PO_4-P	4.3	3.8	3.1	0.09 mg/ℓ
Test duration, days	4	8	15	
Arcadia	3.7	4.4	4.8	0.30 mg/ℓ
Arcadia + 0.37 mg PO_4-P	3.5	4.4	6.1	0.46 mg/ℓ
Test duration, days	3	7	14	
Eufaula	4.2	4.4	4.1	0.21 mg/ℓ
Eufaula + 0.37 mg PO_4-P	4.2	4.6	4.4	0.25 mg/ℓ
Test duration, days	4	8	15	
Little River	4.2	5.1	4.3	0.24 mg/ℓ
Little River + 0.37 mg PO_4-P	4.7	4.9	4.0	0.20 mg/ℓ
Test duration, days	5	9	16	
Thunderbird	3.7	3.8	3.2	0.10 mg/ℓ
Thunderbird + 0.37 mg PO_4-P	4.4	4.2	4.0	0.20 mg/ℓ

* Response is reported as log (counts per minute) of carbon-14 incorporated by algae during 4-hr incubation period.

Table 21

Nitrate-Nitrogen Availability Results*

Standard Cultures				
Nitrate-nitrogen concentration:				
Test duration, days	2	5	12	
0.0 mg NO_3^--N/ℓ	3.1	2.9	2.6	
5.2 mg NO_3^--N/ℓ	4.0	5.9	5.9	
10.5 mg NO_3^--N/ℓ	3.9	5.7	6.1	
21.0 mg NO_3^--N/ℓ	4.0	5.7	6.0	
Sample Cultures				
				Available Nitrogen
Conditions:				
Test duration, days	3	7	14	
Eastern	4.0	3.5	3.0	0.6
Eastern + 4.2 mg NO_3^--N	4.2	4.2	3.6	1.6
Test duration, days	4	8	15	
Arcadia	3.8	4.6	6.3	>5.2
Arcadia + 4.2 mg NO_3^--N	4.1	4.6	6.4	>5.2
Test duration, days	3	7	14	
Eufaula	4.2	4.6	4.2	2.5
Eufaula + 4.2 mg NO_3^--N	4.1	4.3	4.0	2.2
Test duration, days	4	8	15	
Little River	4.5	4.8	4.1	2.4
Little River + 4.2 mg NO_3^--N	4.7	5.4	4.3	2.7
Test duration, days	5	9	16	
Thunderbird	4.0	3.8	3.6	1.6
Thunderbird + 4.2 mg NO_3^--N	3.7	3.6	3.4	1.3

* Response is reported as log (counts per minute) carbon-14 incorporated by algae during 4-hr incubation period.

Table 22

Ammonia-Nitrogen Availability Results*

Standard Cultures			
Ammonia-nitrogen concentration: Test duration, days	2	6	12
0.0 mg NH_3-N/ℓ	3.1	2.9	2.6
5.2 mg NH_3-N/ℓ	3.7	6.0	6.5
10.5 mg NH_3-N/ℓ	3.6	6.2	6.7
21.0 mg NH_3-N/ℓ	3.7	6.2	6.6

Sample Cultures				Available Nitrogen
Conditions: Test duration, days	3	7	14	
Eastern	4.0	3.5	3.0	0.5
Eastern + 4.2 mg NH_3-N	4.1	4.1	3.6	1.3
Test duration, days	4	8	15	
Arcadia	3.8	4.6	6.3	4.9
Arcadia + 4.2 mg NH_3-N	5.4	5.6	6.1	4.7
Test duration, days	3	7	14	
Eufaula	4.2	4.6	4.2	2.1
Eufaula + 4.2 mg NH_3-N	4.2	4.5	3.9	1.7
Test duration, days	4	8	15	
Little River	4.5	4.8	4.1	2.0
Little River + 4.2 mg NH_3-N	4.5	5.5	4.5	2.5
Test duration, days	5	9	16	
Thunderbird	4.0	3.8	3.6	1.3
Thunderbird + 4.2 mg NH_3-N	3.8	3.4	3.6	1.3

* Response is reported as log (counts per minute) carbon-14 incorporated by algae during 4-hr incubation period.

Table 23

Nutrient Spiking Results

Experimental Condition	Arcadia		Eastern Avenue	
	Activity*	Enrichment**	Activity*	Enrichment**
Initial sample	10,200	--	4,600	--
Incubated (raw water)	51,700	1.0	20,300	1.00
Sample + 0.7 mg P/ℓ	56,000	1.08	1,755,600	86.5
Sample + 1.6 mg NO_3^--N/ℓ	103,900	2.01	22,700	1.11
Sample + 1.6 mg NH_3-N/ℓ	112,400	2.17	20,700	1.02
Sample + 23.0 mg $CO_3^=$/ℓ	50,700	0.98	27,400	1.35
Sample + trace metals	40,400	0.78	25,300	1.25
Sample + AAP medium	1,186,400	22.9	147,900	7.28

* Net rate of carbon-14 fixation after 4-hr incubation period, counts per min/4 hr.

** Ratio of sample response to incubated (raw water) response.

Table 24

Effects of Variations of Inflow of Ammonia

On Lake Nitrate Concentrations

Year and Type	Status	Lake Nitrate Concentration, mg/ℓ		
		50% Ammonia Increase	Base	50% Ammonia Decrease
1970-Dry	Prediversion	10.87	8.48	5.95
	Postdiversion	11.15	8.60	6.12
1973-Wet	Prediversion	9.66	7.03	4.43
	Postdiversion	7.57	5.59	3.71

Table 25

Comparison of Selected Characteristics of Reservoirs

Used in the Water-Quality Evaluation

Reservoir	Conservation Pool Depth, m		Theoretical Residence Time years	Outlet(s) Depth from Bottom, m	Location with Relation to Project
	Mean	At Structure			
Thunderbird	6.0	21.3	0.08	8.2	43 km SSE
Arbuckle	9.4	28.7	0.9	3.4, 6.7, 14.6	130 km SSE
Keystone	7.7	22.7	0.25	2.3, 2.3	120 km NE
Oologah	5.7	24.7	0.30	4.6, 8.1, 18.9	170 km NE
Carl Blackwell	5.0	13.5	2.45	0, 3.6, 7.3, 11.0	60 km NNE
Arcadia	6.7	22.0	1.83		

Table 26

Comparison of Dissolved Metal Concentrations Measured in the Deep Fork River, 1969-74, with Proposed EPA Water-Quality Criteria

Metal	Number of Samples	Dissolved Metal Concentration µg/ℓ Minimum	Maximum	Average*	EPA Criteria,[6] µg/ℓ** Freshwater Aquatic Life	Public Water Supply
Cadmium (Cd)	28	0	<9	1 - <2.2	12††	10
Chromium (Cr)	26	0	30	2.0 - <3.5	100	50
Copper (Cu)	26	0	37	11	Bioassay	1000
Iron (Fe)	--	--	--	--	1000	300†
$Fe^{+2} + Fe^{+3}$	26	0	3800	285	--	--
Lead (Pb)	25	0	<18	3.9 - <6.5	Bioassay	50
Manganese (Mn)	26	30	1100	400	--	50
Zinc (Zn)	15	0	80	33	Bioassay	5000
Nickel (Ni)	8	5	<18	4.6 - <10.6	Bioassay	--
Barium (Ba)	6	160	400	275	--	1000
Beryllium (Be)	6	<2	<4	0 - <3	1100	--
Bismuth (Bi)	6	<6	<18	0 - <11.8	--	--
Cobalt (Co)	6	<6	<18	0 - <11.5	--	--
Molybdenum (Mo)	6	<2	< 4	0.7 - <3	--	--

(Continued)

* Where two average values are given, the first was calculated assuming that all less-than values were zero and the second average was calculated assuming that all less-than values were equal to the reported concentration, which was the detection limit.

** Entries are abbreviated as follows:
Bioassay - Bioassay test must be run to establish limit
NL - No limit established
-- - No information provided

† Criterion expressed as Fe.

†† 1.2 µg/ℓ for clodocerans and salmonid fishes; 12.0 µg/ℓ for other less sensitive aquatic life

Table 26 (Concluded)

Metal	Number of Samples	Dissolved Metal Concentration[27] µg/ℓ			EPA Criteria,[8] µg/ℓ	
		Minimum	Maximum	Average	Freshwater Aquatic Life	Public Water Supply
Silver (Ag)	5	0	<2	0 - <1.0	Bioassay	50
Vanadium (V)	6	<8	<15	6.5 - 10.7	--	--
Tin (Sn)	6	<6	<18	0 - <11.7	--	--
Aluminum (Al)	6	30	830	413	--	--
Gallium (Ga)	6	<2	<6	0 - <4.2	--	--
Germanium (Ge)	6	<8	<25	0 - <17.5	--	--
Lithium (Li)	6	4	18	11	--	--
Titanium (Ti)	6	<8	24	10.7 - <14.8	--	--
Zirconium (Zr)	6	<8	<25	0 - <17.5	--	--
Mercury (Hg)	8	0.0	2.6	1.1	0.05	2
Arsenic (As)	2	20	24	22	--	50
Strontium (St)	6	660	2500	1400	--	--

Table 27

Trace Metal Concentrations in the Deep Fork River, as Reported by OSU[2]

Metal	Average	EPA STORET-Arcadia Max	Min	Avg	Arcadia	Deep Fork River 1/71	4/72	Northside Effluent
Chromium	15	30	0.0	15	<50	<50	4	<20
Copper	5	10	0.0	5	<50	<50	3	<1000
Iron	112	160	0.0	112	390	390	1810	700
Lead	30	0.0	0.0	0.0	50	40	<1	90
Manganese	360	480	270	360	200	200	240	<60
Zinc	10	20	0.0	0.8	<5	<50	51	<5000
Cadmium	--	4	0.0	0.8	9	9	<1	<10
Silver	--	--	--	--	25	25	<1	<50
Mercury	--	--	--	--	<5	<5	--	<5

Note: All concentrations in µg/ℓ.

Table 28

Mercury Concentration Measured in Final Effluent of Northside STP and Deep Fork River near Arcadia* **

Date	Northside STP		Arcadia Lake	
	Sample 1	Sample 2	Sample 1	Sample 2
3-29-76	<2	<2	<2	<2
4-5-76	<2	<2	<2	<2
4-12-76	<2	<2	<3	<2
4-19-76	4	4.5	<2	<2
4-26-76	<2	<2	<2	<2
5-3-76	<2	<2	<2	<2
5-10-76	<2	<2	<2	<2
5-17-76	<2	<2	<2	<2
5-24-76	<2	<2	<2	<2
6-1-76	<2	<2	<2	<2

* Sampling conducted by the Oklahoma State Department of Health.
**Concentrations in µg/ℓ.

Table 29

Selected Metal and Pesticide Concentrations in Fish Collected in the Deep Fork River near Arcadia*

Metal Analysis	Limit	Avg.** Conc.	Samples: Green Sunfish 1	2	3	4	5	6	7	8	9	10	Black Bullhead 11
Hg - Tissue	0.5	0.26	0.76	0.26	0.11	0.30	0.30	0.11	0.41	0.31	0.13	0.05	0.45
Viscera		0.11	0.24	0.08	0.22	0.40	†	0.14	0.07	†	†	0.05	0.16
Pb - Tissue	‡	0.39	0.30	0.30	0.28	0.22	0.2	0.29	0.24	0.28	0.20	1.12	0.42
Viscera		1.58	1.76	2.08	2.01	0.93	1.18	1.03	1.04	1.92	0.94	2.59	2.87

Pesticide Analysis (Whole Fish Analysis)	Limit	Avg.** Conc.	Samples: Green Sunfish 1	2	3	4	5	6	7	Bluegill Sunfish 8	Orange Spotted Sunfish 9
DDT	1.0	0.013	0.007	0.022	0.005	0.005	0.039	0.011	0.027	0.003	0.008
Dieldrin	0.3	0.100	0.070	0.140	0.110	0.215	0.200	0.080	0.150	0.003	0.040
Chlordane	‡	5.17	2.80	10.00	2.30	6.00	11.00	6.30	10.30	0.28	2.00
Lindane	0.3	0.016	0.006	0.019	0.018	0.030	0.043	0.015	0.040	†	0.003
Heptachlor epoxide	‡	1.23	0.74	1.20	1.50	2.70	2.40	1.10	2.40	0.18	0.33

* All concentrations in mg/kg.
** Weighted in proportion to the number of fish per sample.
† None detected.
‡ No limit established.

AD-A039 492 ARMY ENGINEER WATERWAYS EXPERIMENT STATION VICKSBURG MISS F/G 13/2
ARCADIA LAKE WATER-QUALITY EVALUATION.(U)
APR 77 R W HALL, R H PLUMB, K W THORNTON
UNCLASSIFIED WES-TR-Y-77-2 NL

3 OF 4
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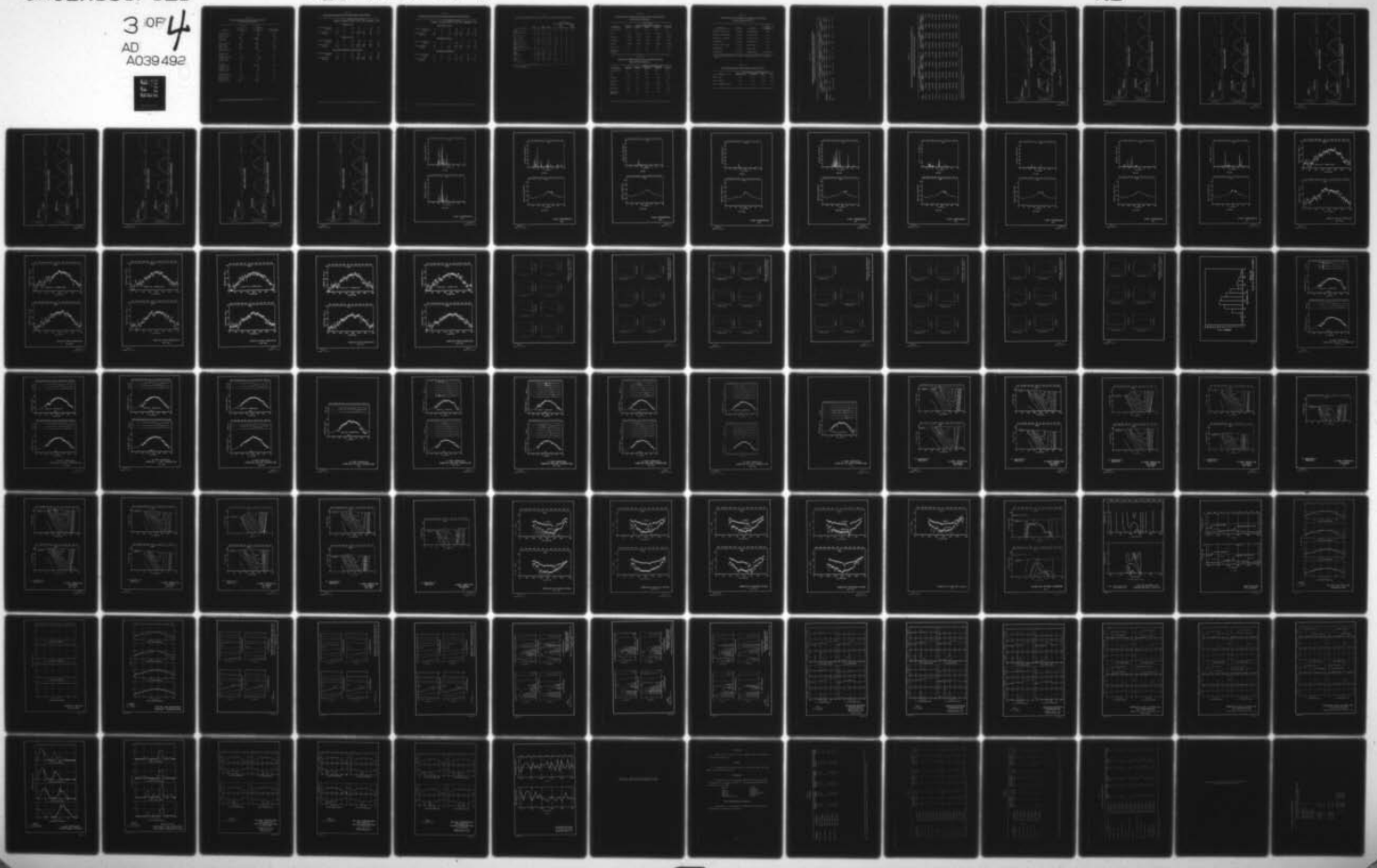


Table 30

Soluble and Total Concentrations of Metals in the Deep Fork River [27]

Metal	Soluble Concentration µg/ℓ	Total Concentration µg/ℓ	Percentage*
Cadmium (Cd)	0.93	6.1	16
Chromium (Cr)	1.7	50	3.4
Copper (Cu)	11	72	15
Iron (Fe)	284	26,000	1.1
Lead (Pb)	3.9	220	1.8
Manganese (Mn)	399	2,100	19
Zinc (Zn)	33	283	12
Nickel (Ni)	4.6	51	9.0
Barium (Ba)	105	1,300	8.0
Beryllium (Be)	0	2.0	0
Bismuth (Bi)	0	0	
Cobalt (Co)	0	17	0
Molybdenum (Mo)	0.67	1.6	42
Silver (Ag)	0	5.8	0
Vanadium (V)	6.5	50	13
Tin (Sn)	0	0	
Aluminum (Al)	413	26,000	1.6
Gallium (Ga)	0	6.0	0
Germanium (Ge)	0	0	
Lithium (Li)	11	46	24
Titanium (Ti)	11	83	13
Zirconium (Zr)	0	0	
Mercury (Hg)	0.66	1.1	60
Arsenic (As)	--	22	
Strontium (St)	1,400	1,800	77

* (Soluble Concentration/Total Concentration) × 100.

Table 31

Trace Metal Concentrations in Lake Eufaula and Tributary

	Concentration, µg/ℓ						
	Cadmium	Chromium	Copper	Iron	Lead	Manganese	Zinc
Eufaula Site 1							
Dissolved-Surface	1	0	14	40	2	30	30
-Bottom	0	0	34	70	3	20	40
Total-Surface	20	0	20	1400	<100	80	40
-Bottom	<10	0	50	1900	<100	190	110
Eufaula Site 2							
Dissolved-Surface	0	0	6	70	1	20	0
-Bottom	0	0	10	70	2	10	0
Total-Surface	<10	0	20	9000	<100	930	100
-Bottom	<10	0	10	5300	<100	370	60
Deep Fork Tributary to Lake Eufaula							
Dissolved-Surface	0	0	8	100	2	0	30
-Bottom	0	0	10	140	2	0	30
Total-Surface	<10	0	40	6700	<100	410	120
-Bottom	10	0	50	7900	<100	460	160

Table 32

Trace Metal Concentrations in Lake Thunderbird and Tributary

	Concentration, µg/ℓ						
	Cadmium	Chromium	Copper	Iron	Lead	Manganese	Zinc
Thunderbird Site 1							
Dissolved-Surface	0	0	6	30	3	0	10
-Bottom	0	0	6	20	1	20	10
Total-Surface	<10	0	<10	270	<100	20	20
-Bottom	<10	0	10	670	<100	110	30
Thunderbird Site 2							
Dissolved-Surface	0	0	5	20	2	0	30
-Bottom	0	0	7	20	1	0	0
Total-Surface	20	0	<10	300	<100	40	30
-Bottom	<10	0	<10	440	<100	20	30
Thunderbird Tributary							
Dissolved-Surface	0	0	2	20	3	60	0
-Bottom	--	--	--	--	--	--	--
Total-Surface	<10	0	<10	820	<100	160	10
-Bottom	--	--	--	--	--	--	--

Table 33

Comparison of Trace Organics Measured in the Deep Fork River near Arcadia with EPA's 1976 Criteria for Water Quality*

Toxic Organic	Number Samples	Mean	Sample Standard Deviation	Range Min	Range Max	1976 Quality Criteria for Water Protection: Aquatic Life Criterion	Aquatic Life Percent Samples Exceeding Criterion	Water Supply Criterion	Water Supply Percent Samples Exceeding Criterion
Lindane	64	0.07	0.37	0	3.0	0.01	39	4	0
Lindane in bottom deposits	2	0	0	0	0	--	--	--	--
Chlordane	65	0.15	0.32	0	1.7	0.01	52	**	--
Chlordane in bottom deposits	2	5.5	--	2	9	--	--	--	--
DDD	62	0.008	0.016	0	0.08	--	--	--	--
DDD in bottom deposits	2	0	0	0	0	--	--	--	--
DDE	64	0.002	--	0	0.01	--	--	--	--
DDE in bottom deposits	2	0	0	0	0	--	--	--	--
DDT	64	0.006	0.012	0	0.06	0.001	30	**	--
DDT in bottom deposits	2	0.75	--	0	1.5	--	--	--	--
Dieldrin	64	0.038	0.027	0	0.13	0.003†	95	**	--
Dieldrin in bottom deposits	2	0.95	--	0.6	1.3	--	--	--	--
Endrin	64	0.0003	0.002	0	0.02	0.004	2	0.2	0
Endrin in bottom deposits	2	0	0	0	0	--	--	--	--
Heptachlor	63	0.002	0.010	0	0.08	0.001	3	**	--
Heptachlor in bottom deposits	2	0	0	0	0	--	--	--	--
Heptachlor epoxide	64	0	0	0	0	--	--	--	--
Heptachlor epoxide in bottom deposits	2	0	0	0	0	--	--	--	--
Aldrin	64	0.002	0.008	0	0.05	0.003†	6	**	--
Aldrin in bottom deposits	2	0	0	0	0	--	--	--	--
Diazinon	38	0.208	0.236	0	1.2	--	--	--	--
Malathion	37	0.005	0.017	0	0.09	0.1	0	--	--
Parathion	37	0.0003	0.002	0	0.01	0.04	0	--	--
Methlyl parathion	36	0.0006	0.003	0	0.02	--	--	--	--
2,4-D	65	0.12	0.19	0	0.66	100	0	--	--
2,4,5-T	65	0.04	0.08	0	0.51	--	--	--	--
2,4,5-TP (Silvex)	57	0.03	0.07	0	0.30	10	0	--	--
Toxaphene	1	0	--	0	0	0.001	0	--	--
PCB	28	0.003	0.019	0	0.1	0.001	4	--	--
PCB in bottom deposits	2	0	0	0	0	--	--	--	--

* Units: µg/ℓ in aqueous phase, µg/kg in sediments.

** Minimum human exposure.

† Sum of aldrin and dieldrin.

Table 34

Concentration of Organic Pesticides in the Deep Fork River near Arcadia, Oklahoma[27]

Parameter	Water Year-Average Concentration, µg/ℓ				
	1969-70	1970-71	1971-72	1972-73	1973-74
Flow (10^6 m^3)	43	42	32	88	75
DDT	0.011	0.007	0.001	0.004	0.006
Dieldrin	0.031	0.028	0.045	0.038	0.031
Chlordane	0.10	0.00	0.04	0.11	0.20
Lindane	0.056	0.324	0.027	0.009	0.009
Heptachlor	0.0	0.0	0.0	0.0	0.005
Heptachlor epoxide	0.0	0.0	0.0	0.0	0.0

Table 35

Loadings of Organic Pesticides in the Deep Fork River near Arcadia, Oklahoma[27]

Parameter	Water Year-Loading, kg				
	1969-70	1970-71	1971-72	1972-73	1973-74
Flow (10^6 m^3)	43	42	32	88	75
DDT	0.52	0.38	0.02	0.43	0.72
Dieldrin	1.3	1.3	2.0	2.9	2.8
Chlordane	3.4	0.0	1.2	9.0	22
Lindane	2.7	12	0.62	0.51	0.53
Heptachlor	0.0	0.0	0.0	0.0	0.31
Heptachlor epoxide	0.0	0.0	0.0	0.0	0.0

Table 36

DDT Residue in Bass Samples Collected in Lake Eufaula and Lake Thunderbird[84,85]

Station	Date	Fish	DDT Concentration* mg/kg
Eufaula, Deep Fork	1/16/72	Cleaned bass	0.69
Eufaula, Poram Land	1/19/72	Cleaned bass	0.45
Eufaula, N. Canadian	2/01/72	Cleaned bass	0.18
Thunderbird	1/20/72	Cleaned bass	0.02
Eufaula, Porum Land	1/19/72	Whole bass	0.56
Thunderbird	1/24/72	Whole bass	0.18
Thunderbird	11/15/72	Cleaned bass	0.08-0.115

* No detectable residue for aldrin, dieldrin, heptachlor, lindane, or PCB.

Table 37

Pesticide Residues in Lake Eufaula Fish, 1967-73[121]

Fish	Concentration, mg/kg (Wet Wt)			
	Metabolites	Dieldrin	BHC	PCB
Whole carp	0.27	0.03	0.01	0.29
Whole bluegill	0.16	<0.01	<0.01	0.15
Whole largemouth bass	0.35	<0.01	<0.01	0.07

Table 38

Average Concentrations and Annual Loadings of Phenols and Total Dissolved Solids in the Deep Fork River near Arcadia [38]

	Water Year-Average Concentrations and Annual Loadings									
	1969-70		1970-71		1971-72		1972-73		1973-74	
Parameter	Concentration mg	Loading kg	Concentration mg	Loading kg	Concentration mg	Loading kg	Concentration mg	Loading kg	Concentration mg	Loading kg
Phenols	0.0057	240	0.0039	160	0.0025	88	0.0055	520	0.0081	460
TDS	880	3.0×10^7	770	2.6×10^7	850	2.4×10^7	780	4.6×10^7	690	3.9×10^7

Table 39

Total Coliform and Fecal Coliform Bacteria Measured in Final Effluent of Northside STP and Deep Fork River near Arcadia* **

	Northside STP				Arcadia Lake			
	Total Coliform		Fecal Coliform		Total Coliform		Fecal Coliform	
Date	Sample 1	Sample 2	Sample 1	Sample 2	Sample 1	Sample 2	Sample 1	Sample 2
3-29-76	340,000	450,000	10,000	11,000	47,000	46,000	4,000	7,400
4-5-76	3,700,000	4,200,000	90,000	110,000	900,000	700,000	8,000	7,000
4-12-76	520,000	410,000	33,000	16,000	80,000	80,000	19,000	16,000
4-19-76	18,000,000	26,000,000	2,100,000	2,000,000	280,000	310,000	190,000	160,000
4-26-76	4,700,000	4,500,000	900,000	800,000	220,000	150,000	18,000	16,000
5-3-76	330,000	340,000	60,000	70,000	160,000	250,000	40,000	38,000
5-10-76	900,000	1,400,000	100,000	150,000	40,000	50,000	12,000	13,000
5-17-76	340,000	350,000	26,000	28,000	150,000	110,000	10,000	12,000
5-24-76	170,000	190,000	30,000	20,000	130,000	140,000	20,000	20,000
6-1-76	380,000	360,000	60,000	80,000	60,000	110,000	28,000	40,000

* Sampling conducted by the Oklahoma State Department of Health.
**Units are in colonies/100 ml.

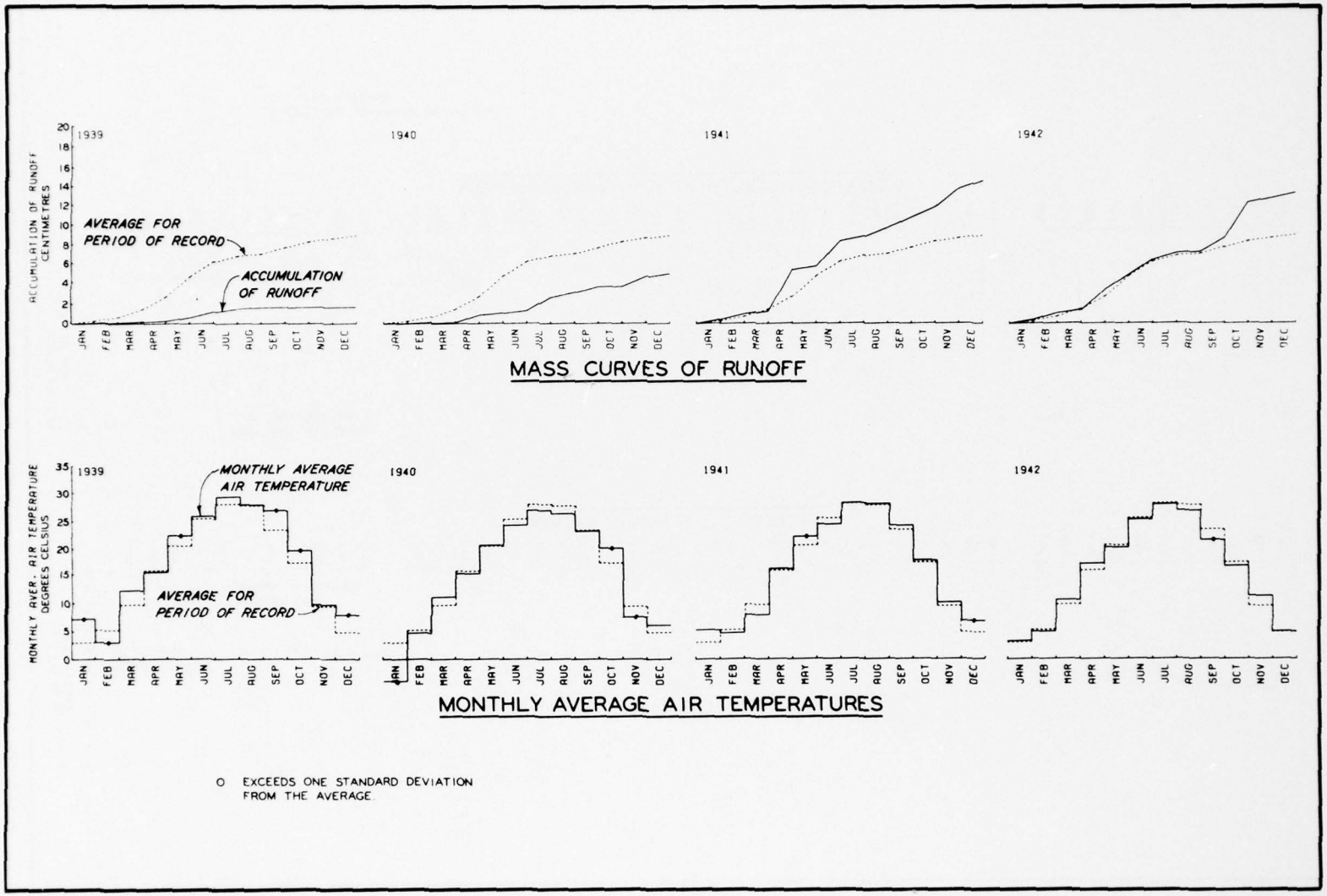
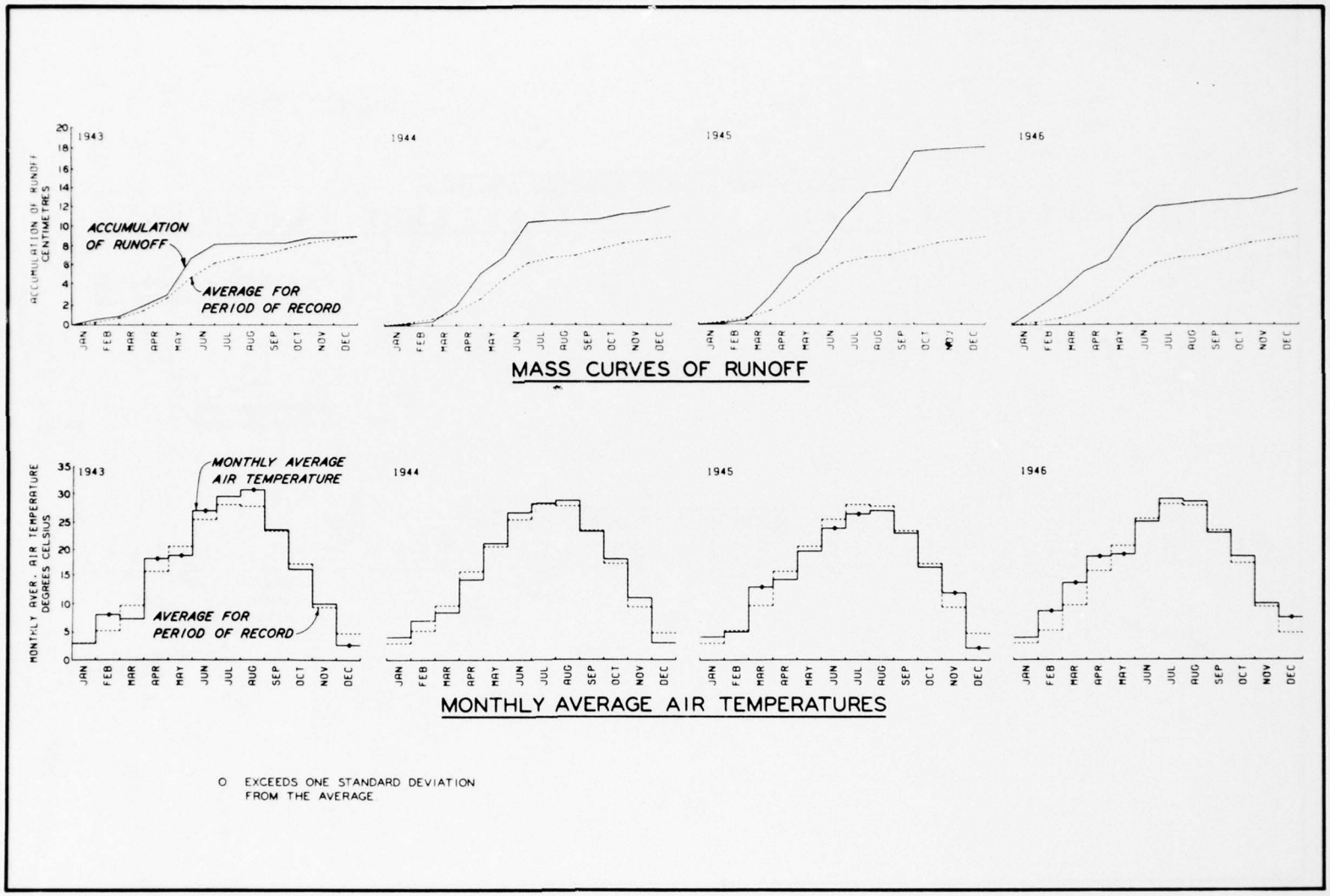


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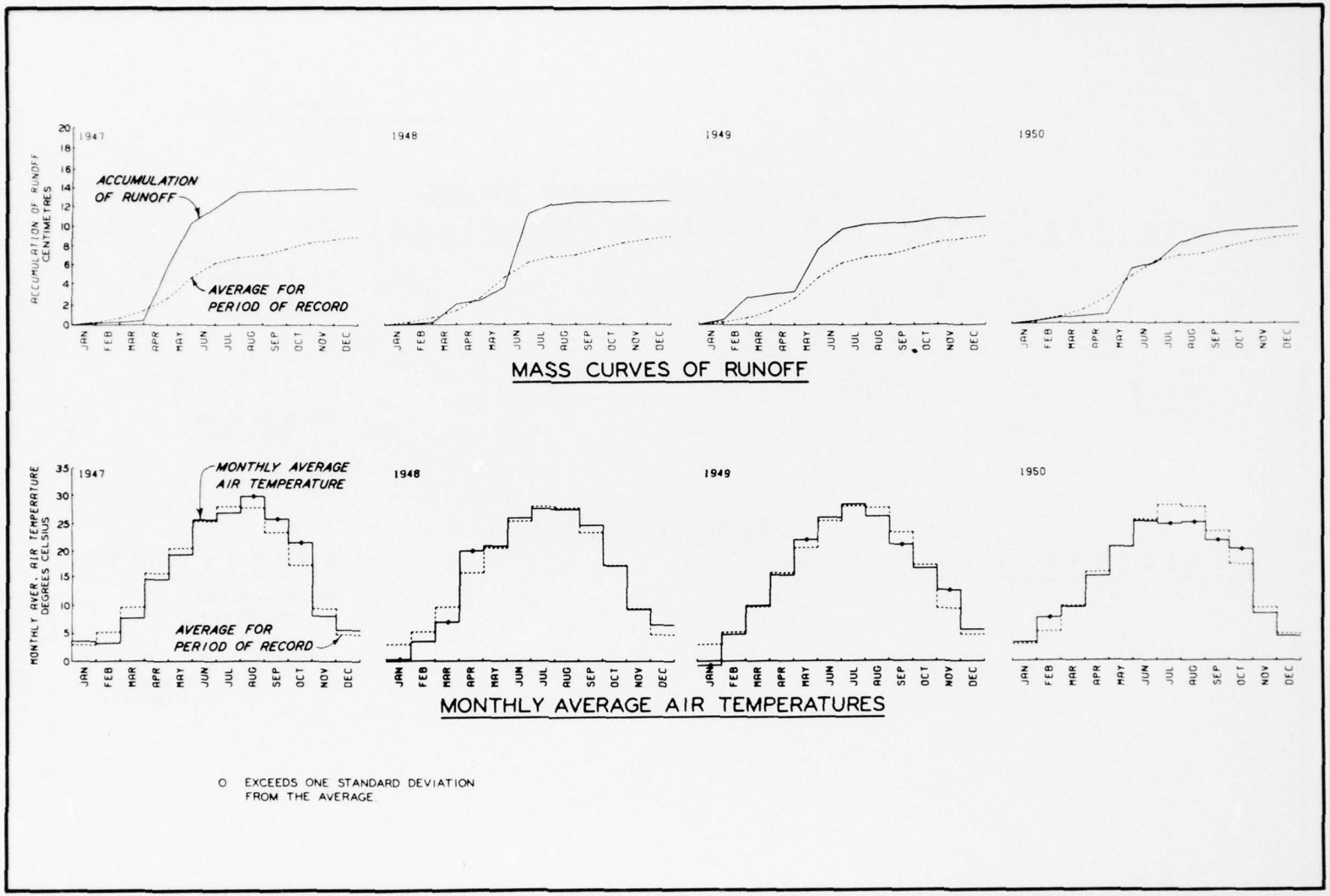
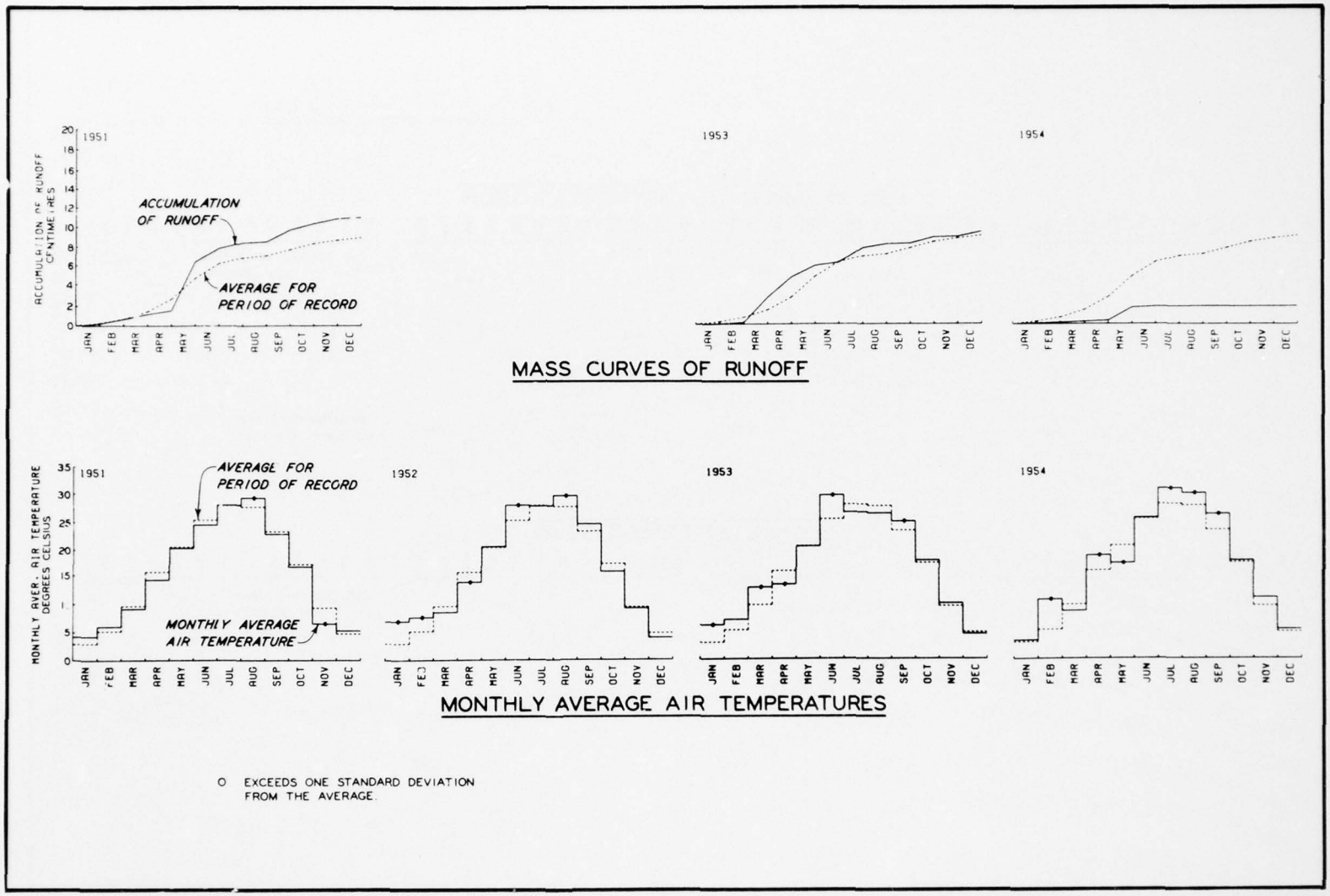
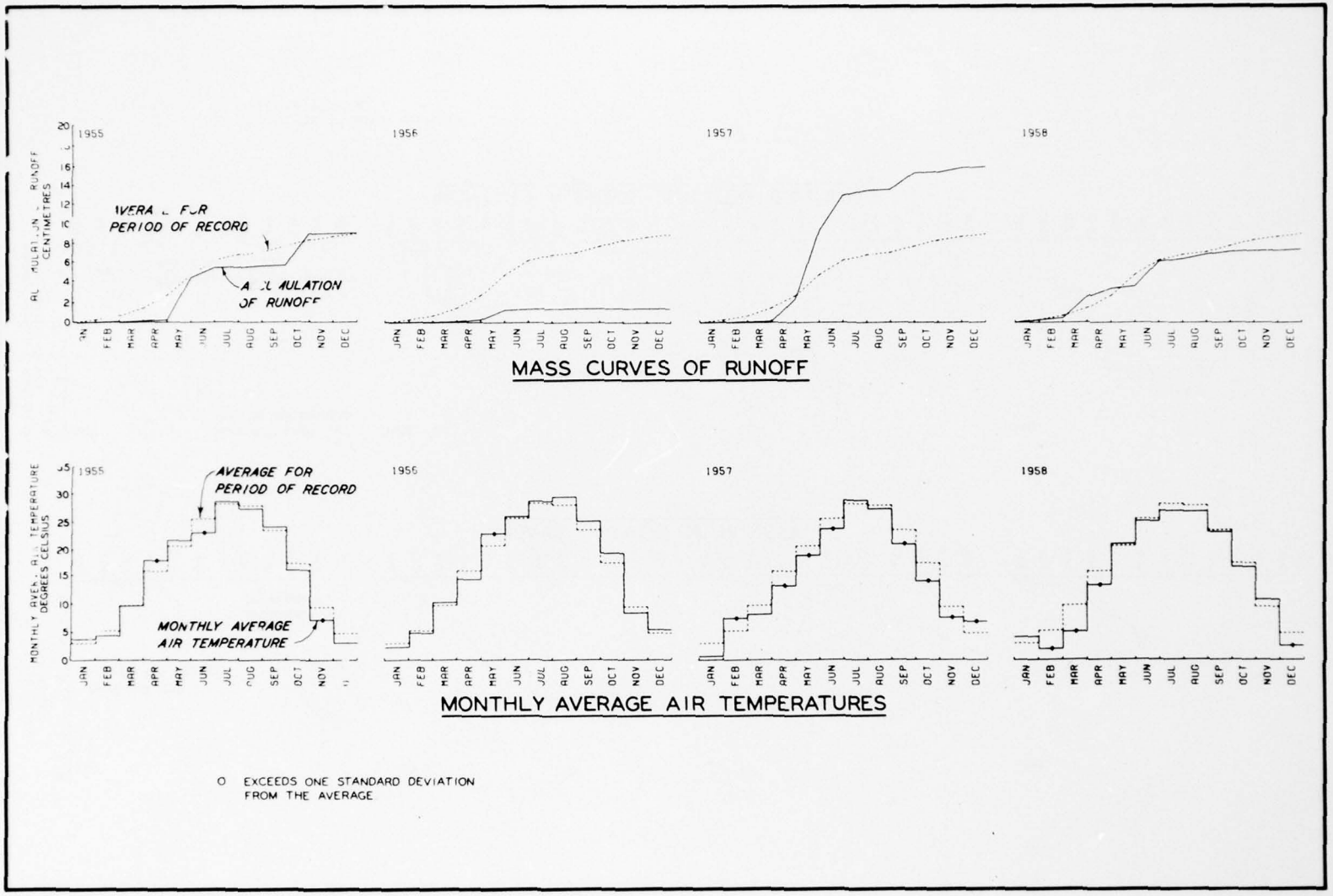


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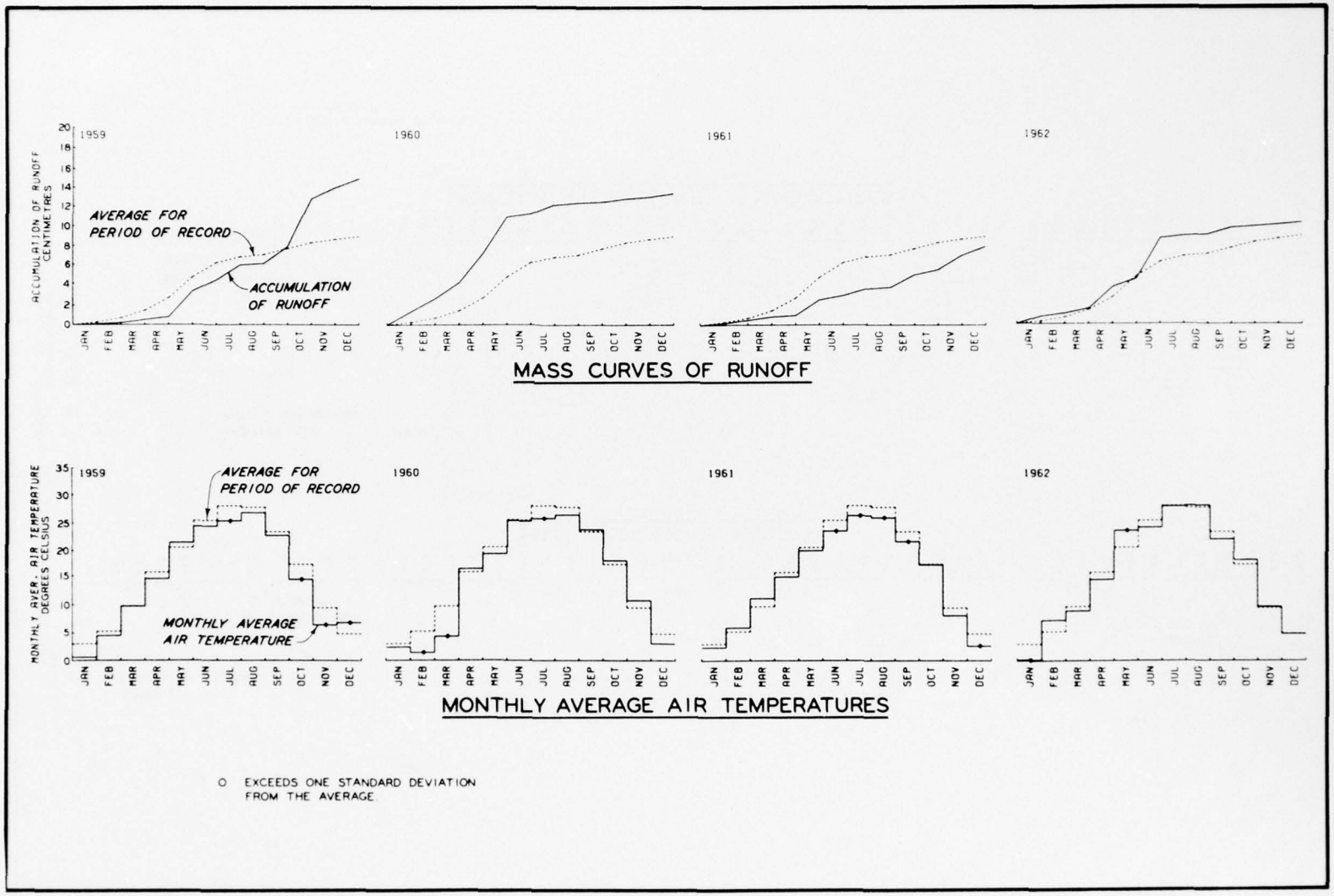




O EXCEEDS ONE STANDARD DEVIATION FROM THE AVERAGE

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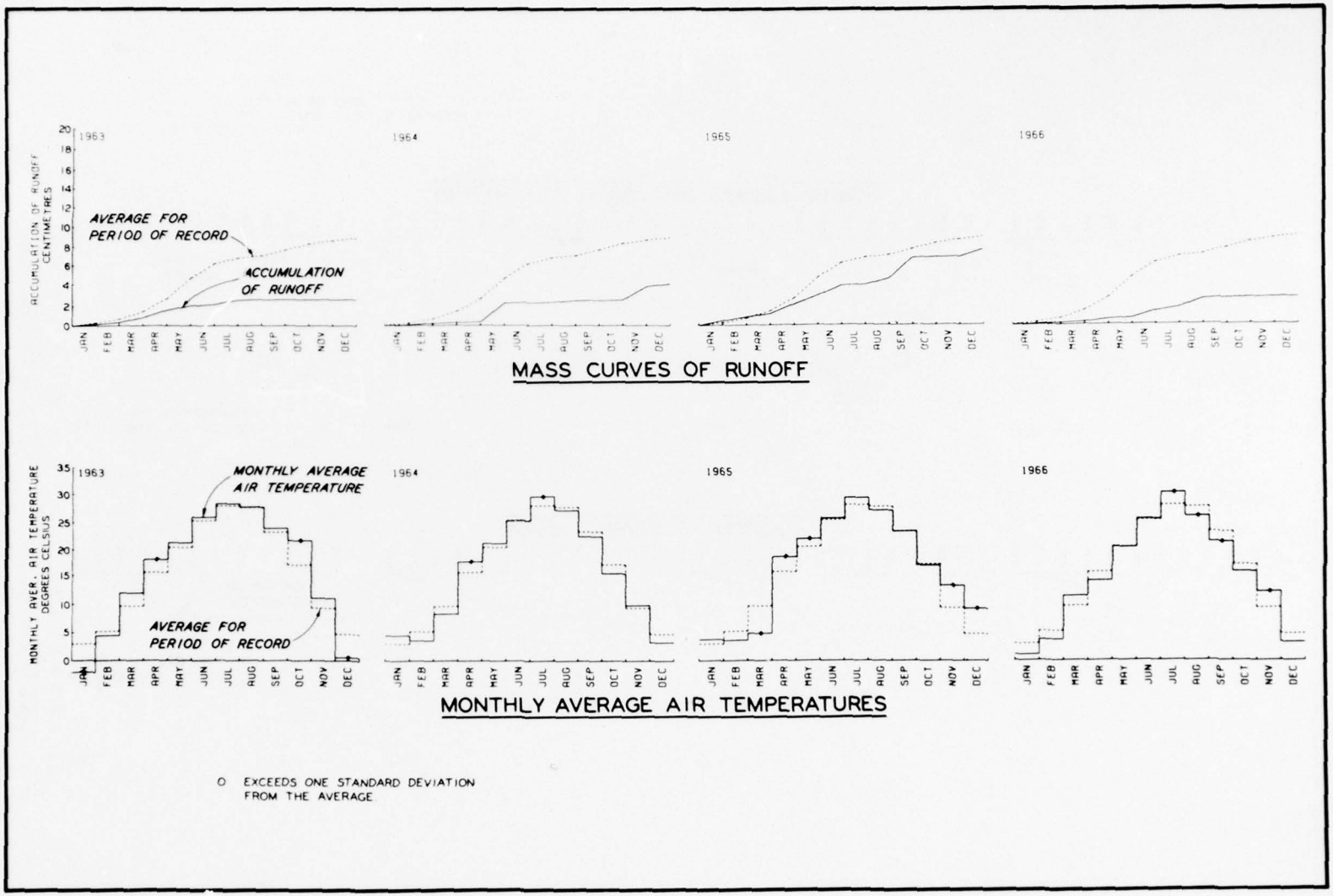
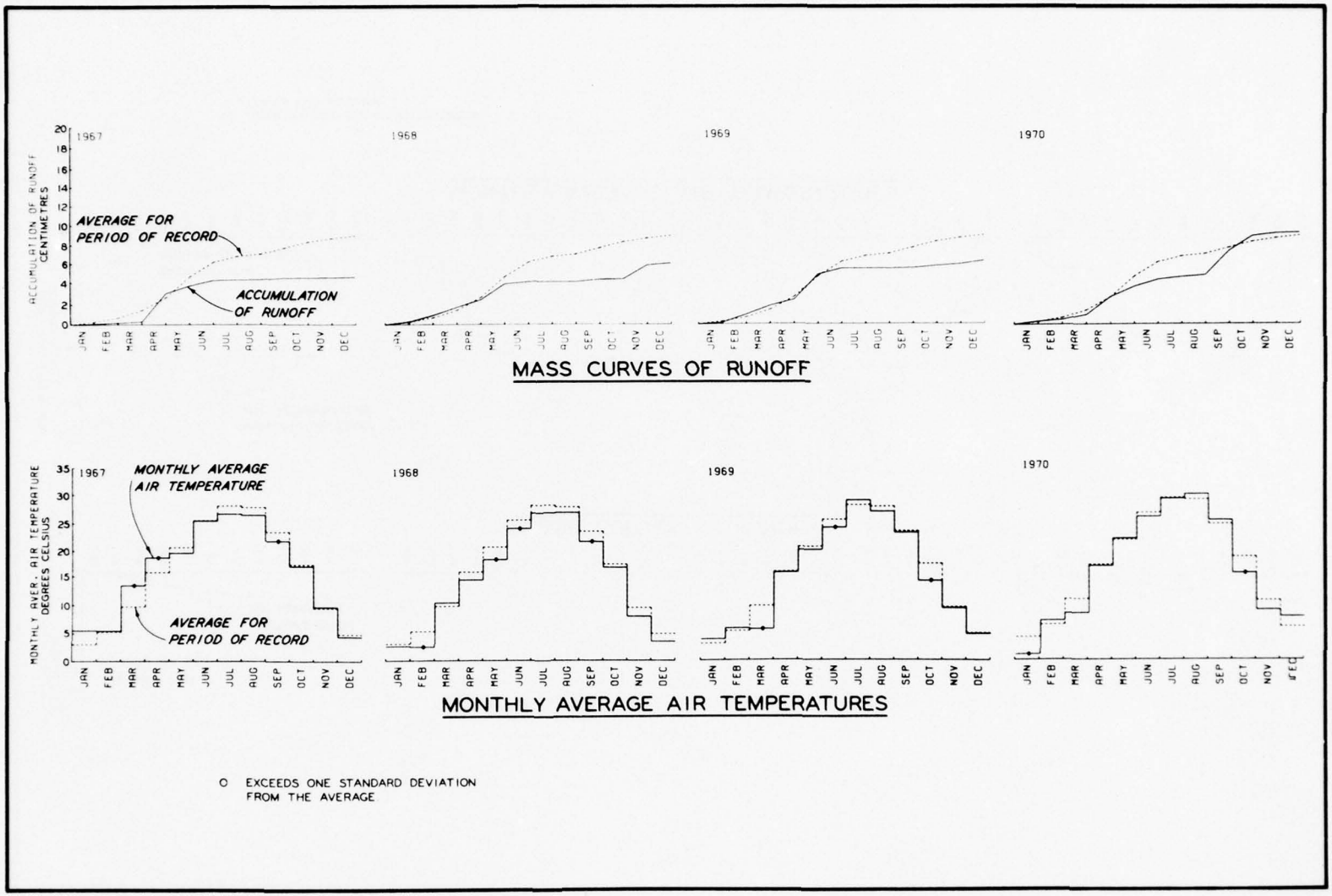
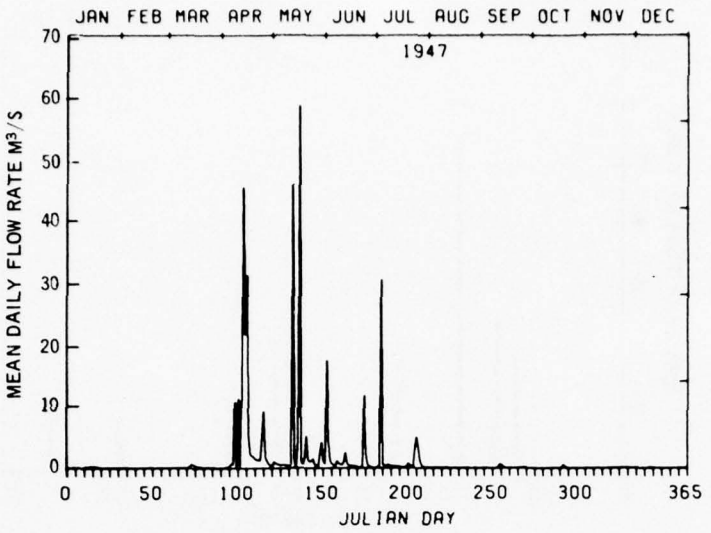


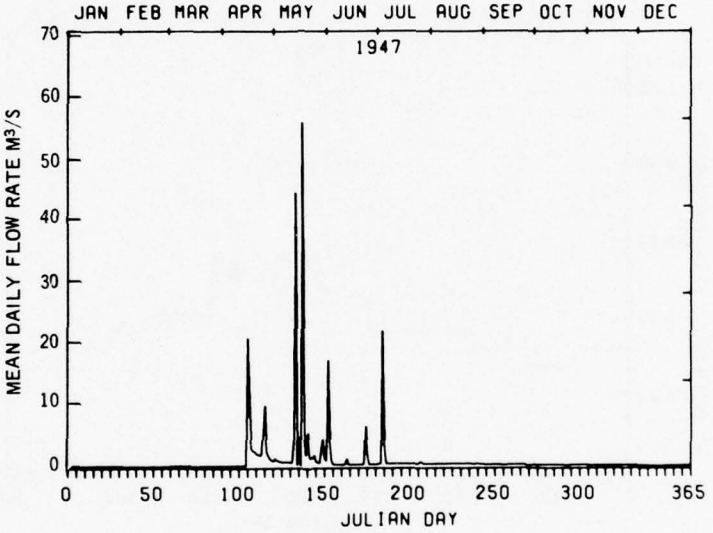
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INFLOW

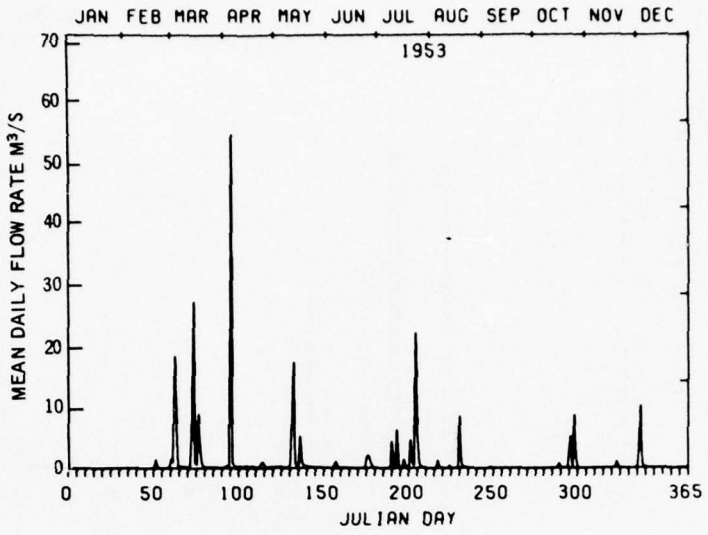


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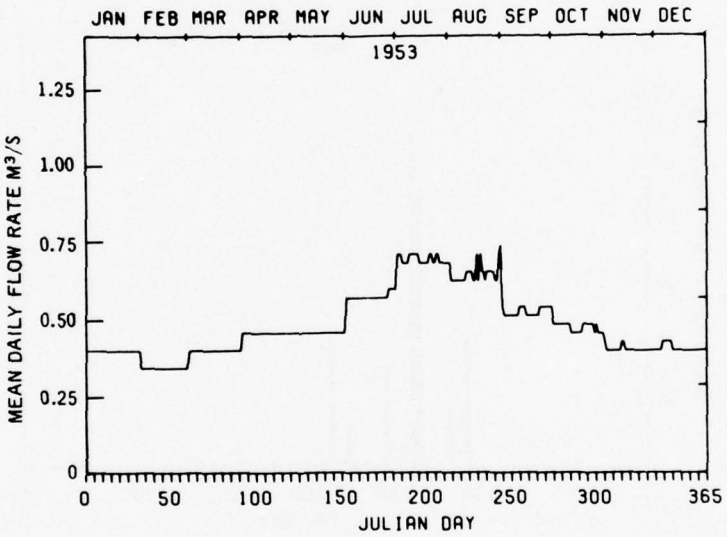
FLOW HYDROGRAPHS

1947

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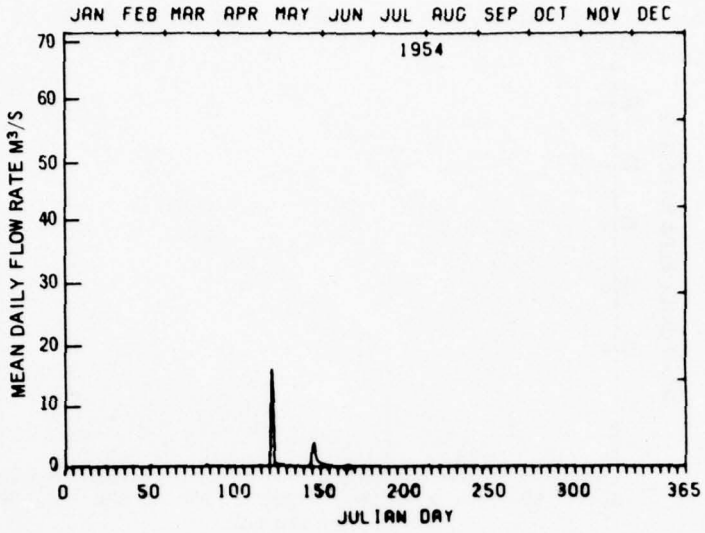


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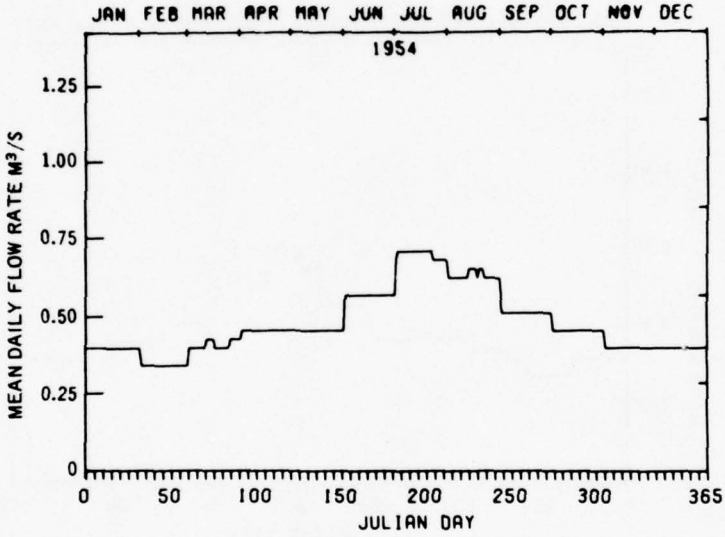
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1953

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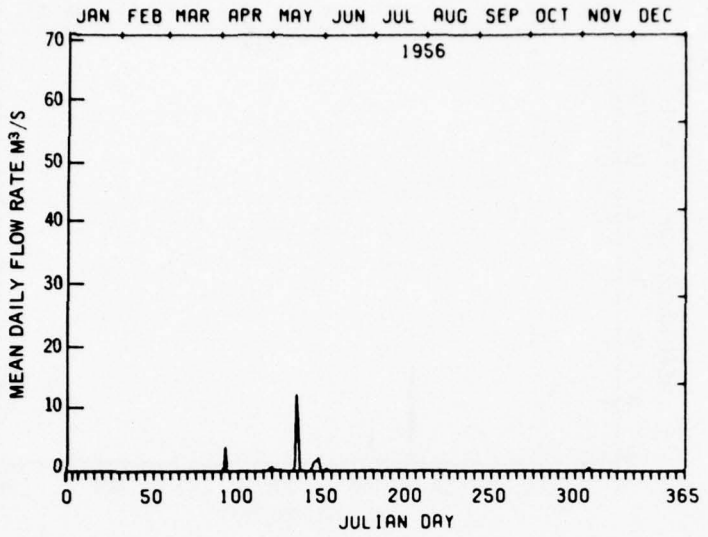


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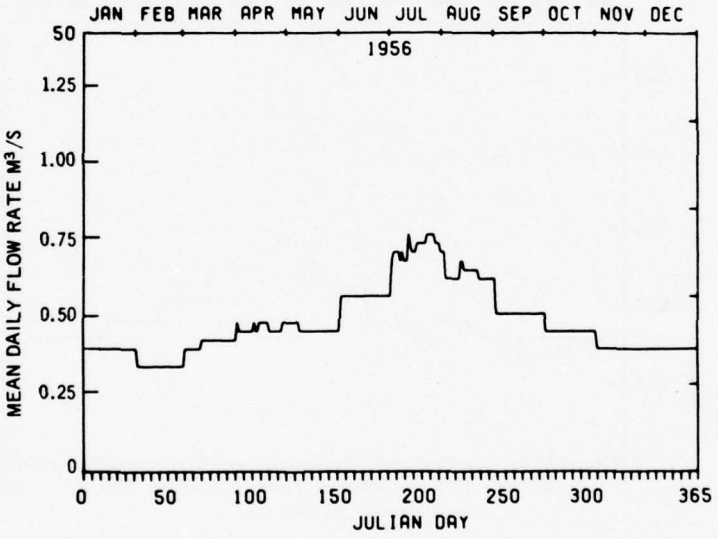
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1954

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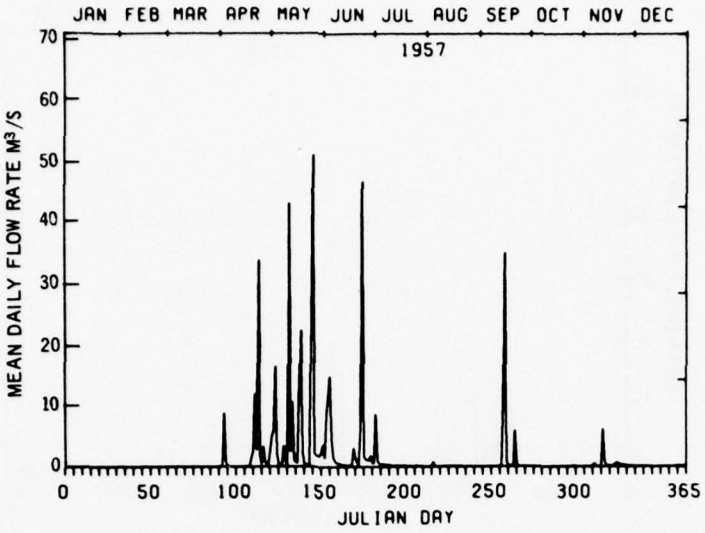


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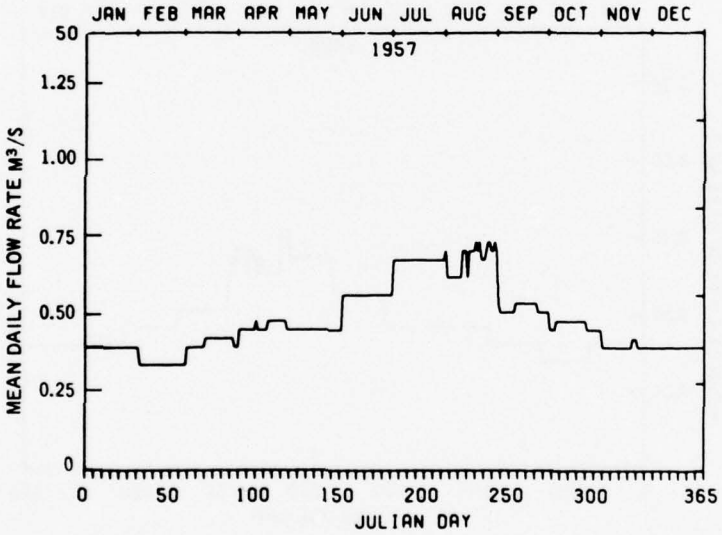
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1956

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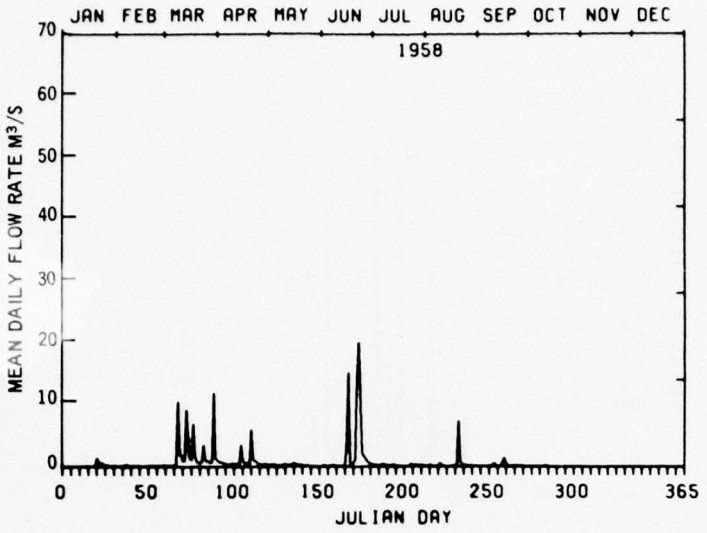


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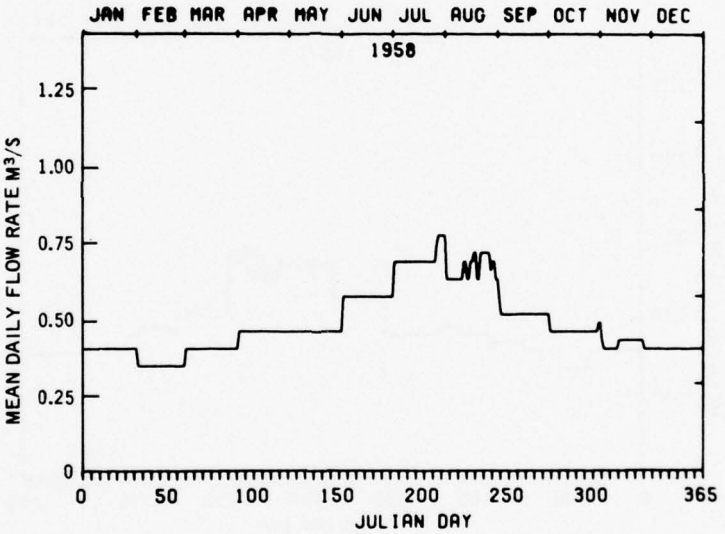
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1957

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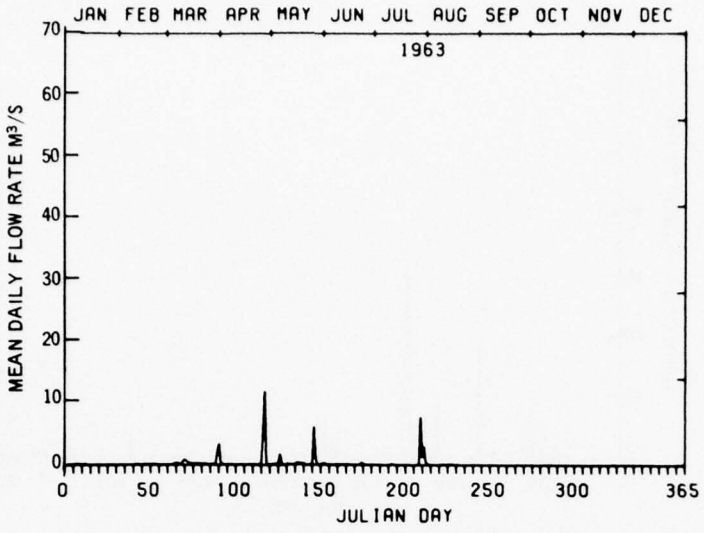
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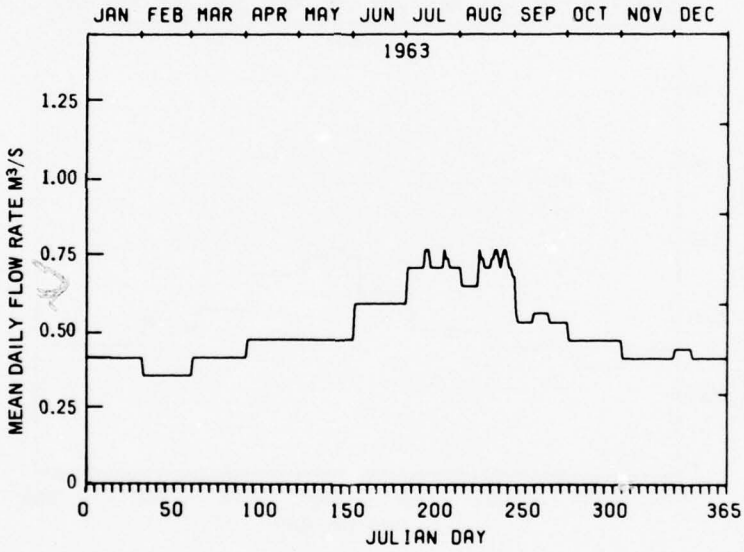
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1958

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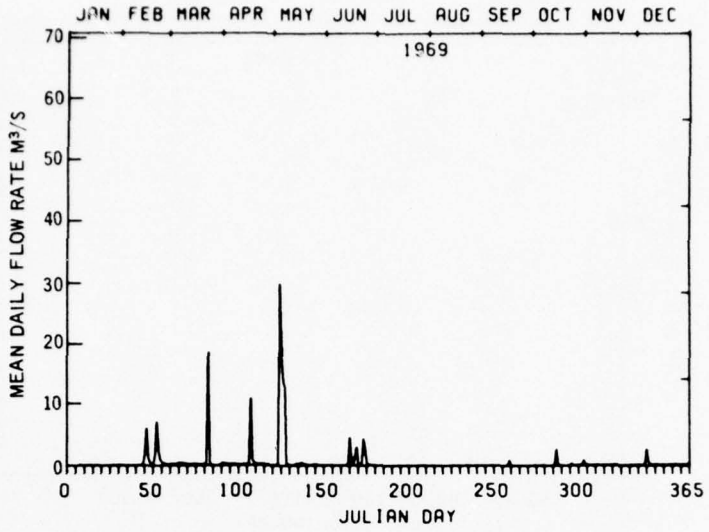


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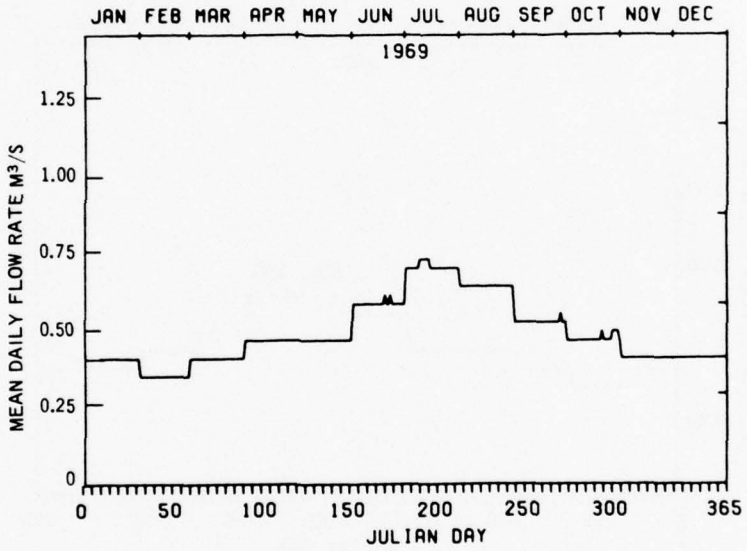
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1963

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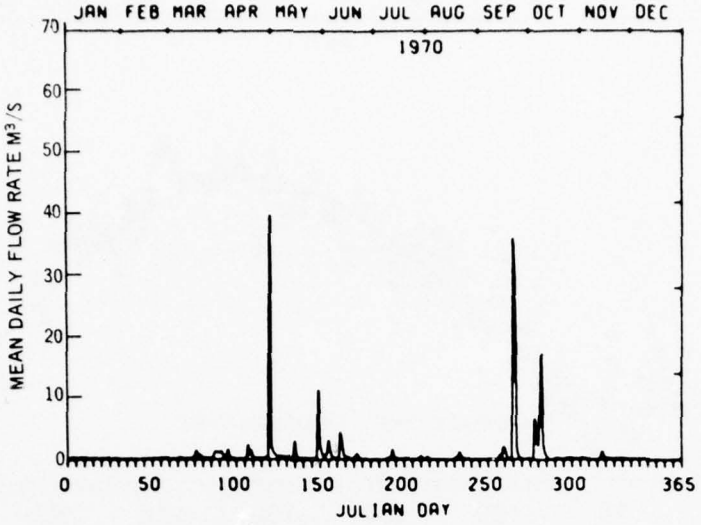


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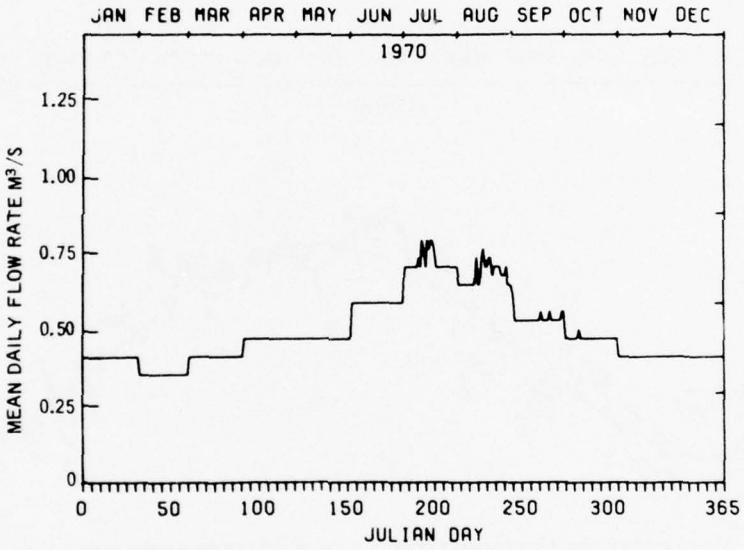
FLOW HYDROGRAPHS

1969

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INFLOW



OUTFLOW

FLOW HYDROGRAPHS

1970

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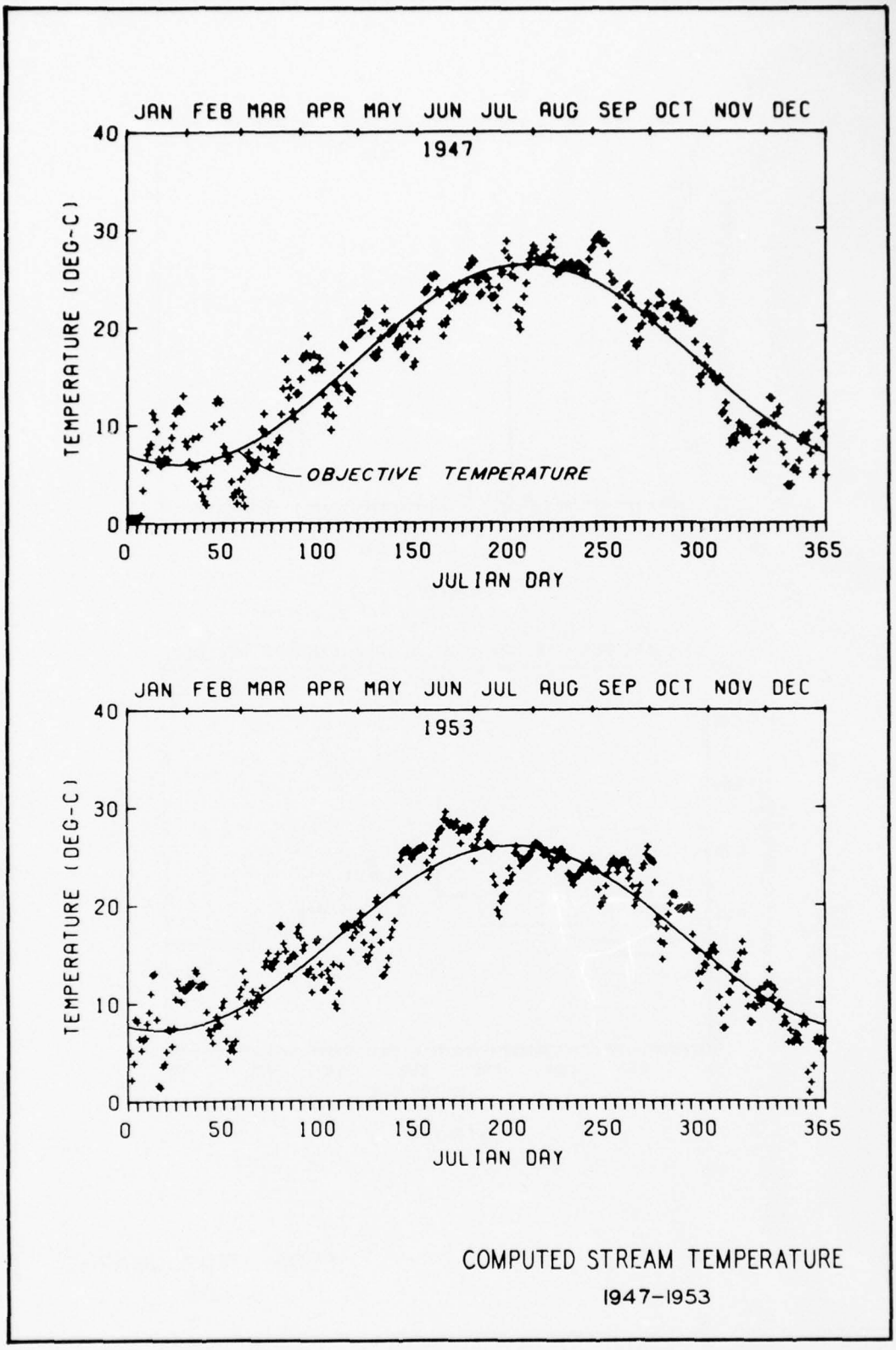
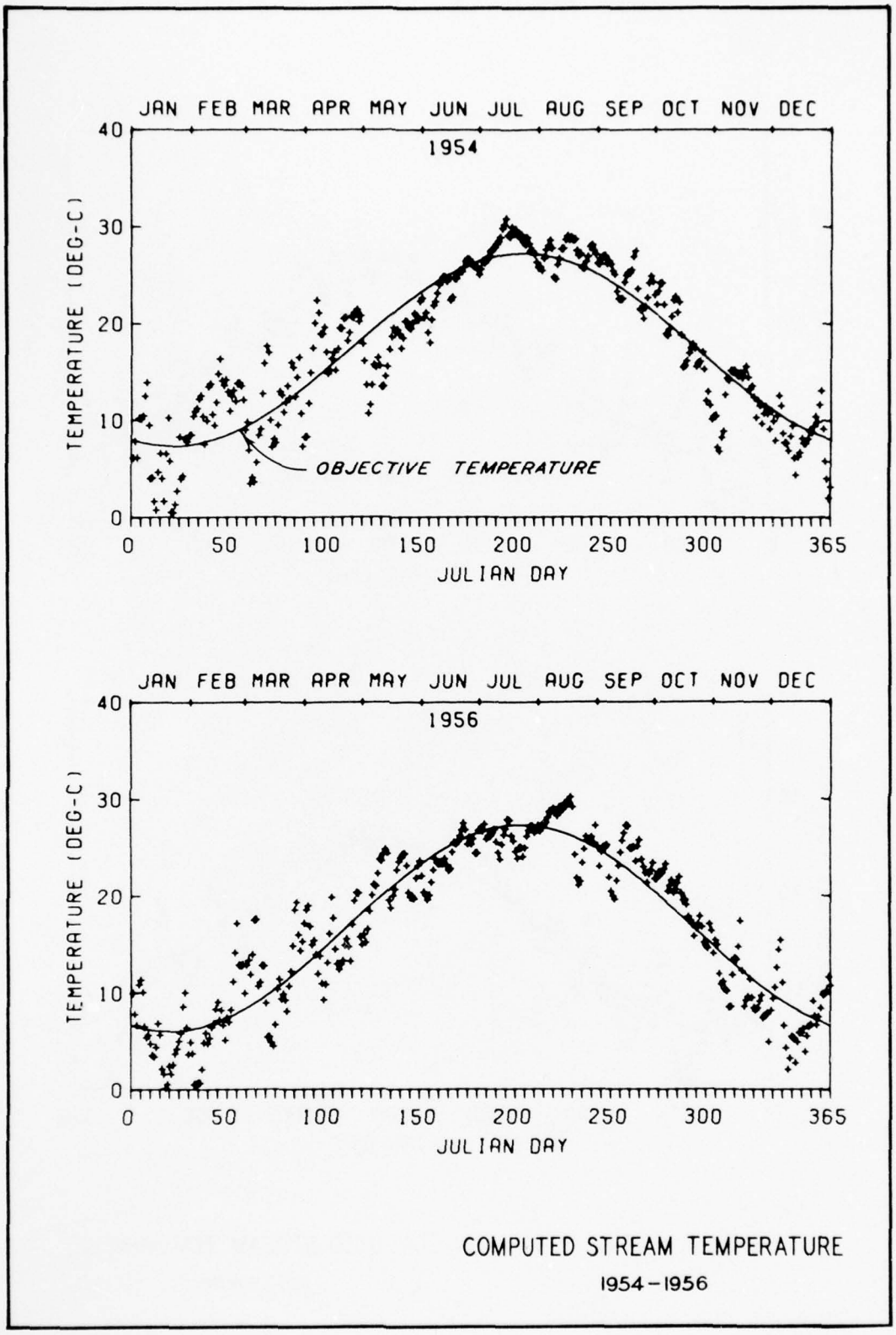


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COMPUTED STREAM TEMPERATURE

1954–1956

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(Sheet 2 of 6)

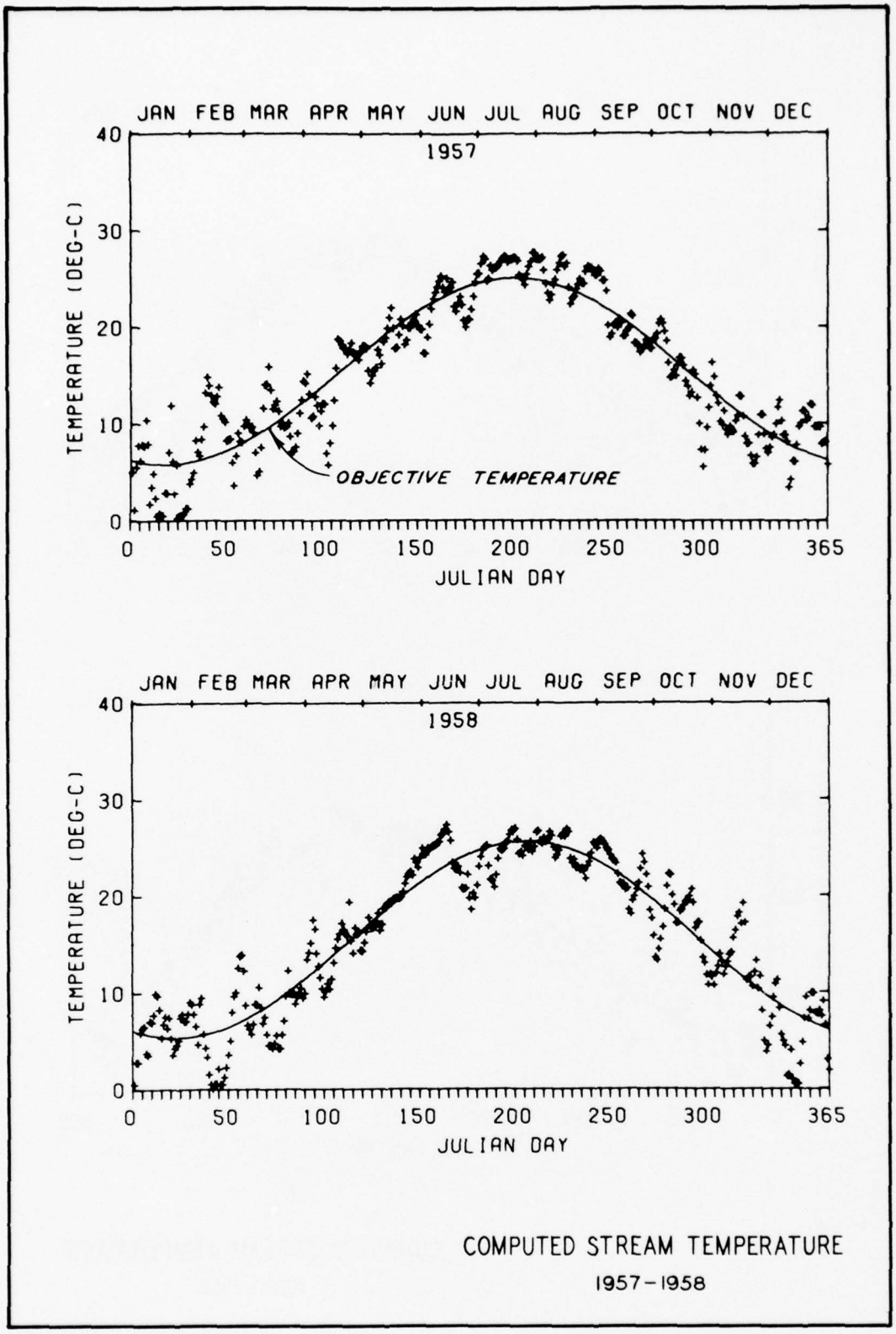
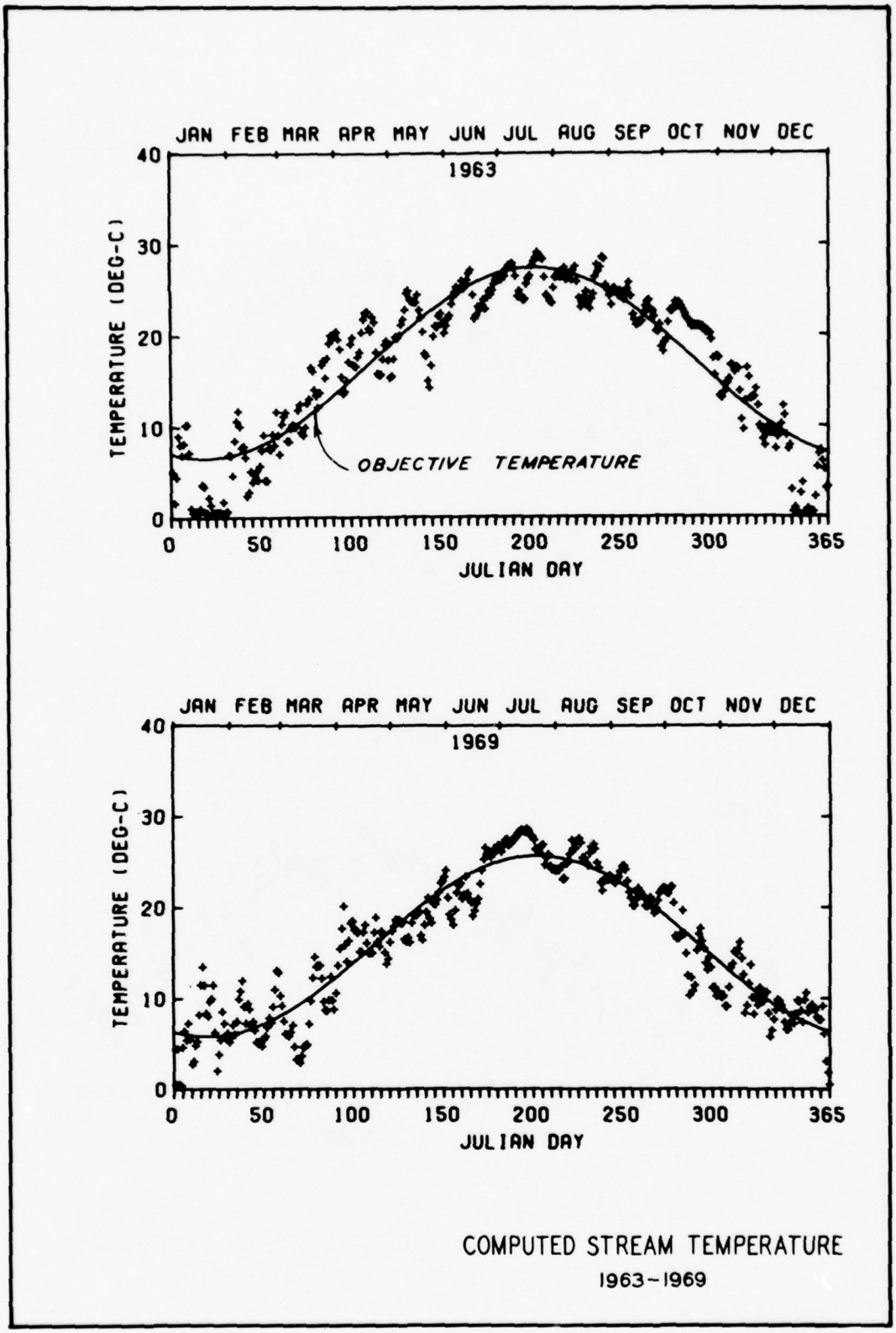


PLATE 3
(Sheet 3 of 6)



COMPUTED STREAM TEMPERATURE
1963-1969

PLATE 3
(Sheet 4 of 6)

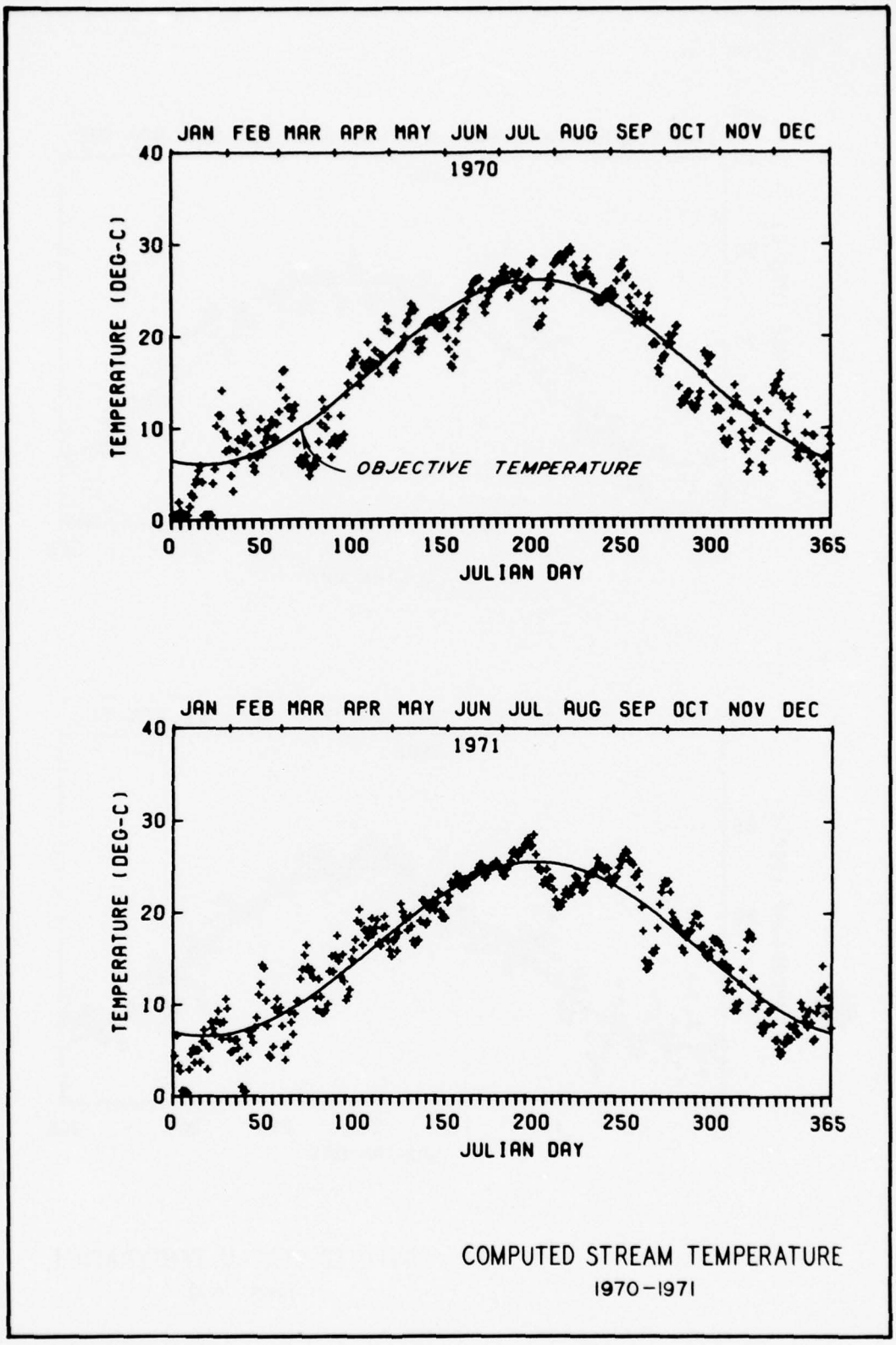
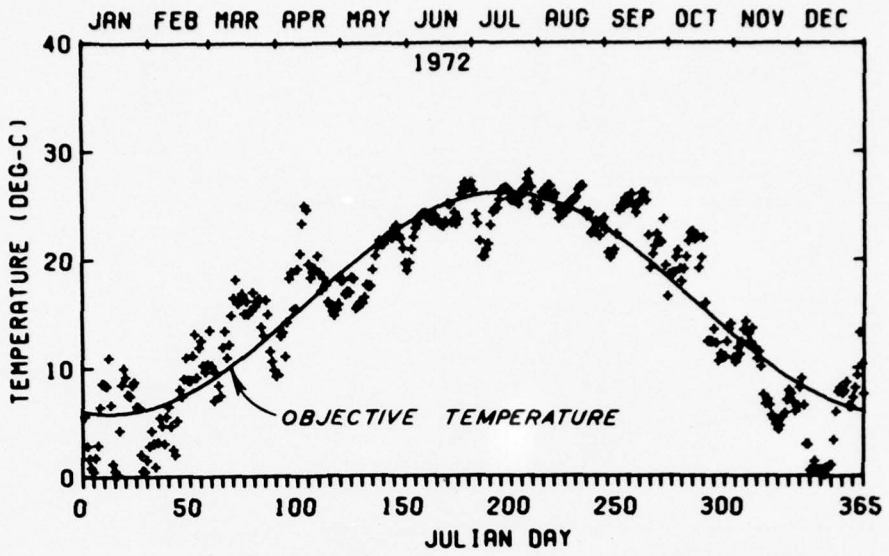
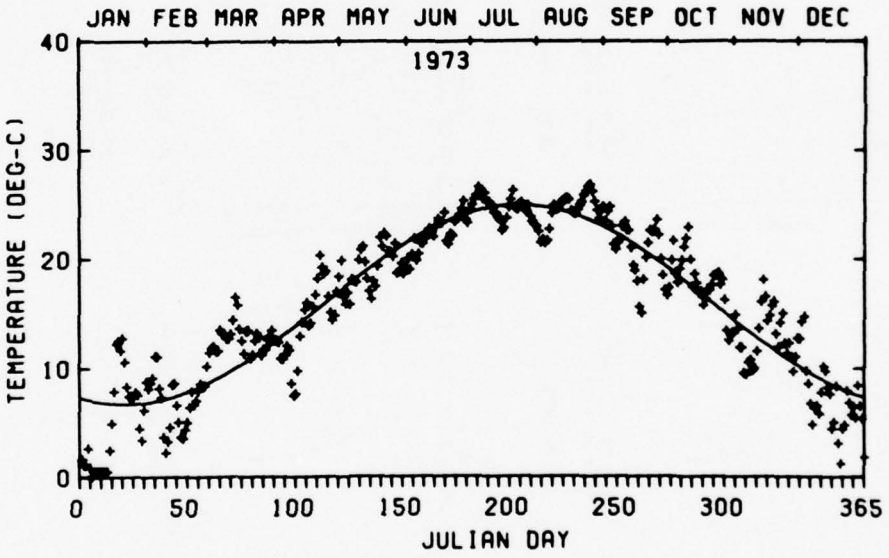


PLATE 3
(Sheet 5 of 6)

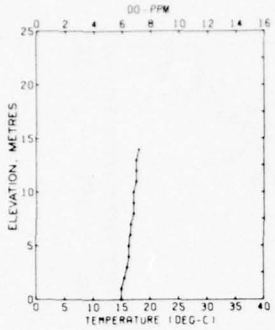




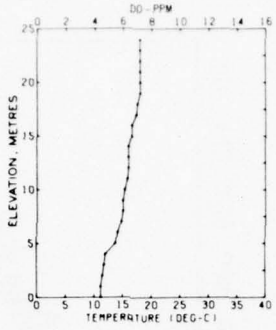
COMPUTED STREAM TEMPERATURE
1972 - 1973

PLATE 3
(Sheet 6 of 6)

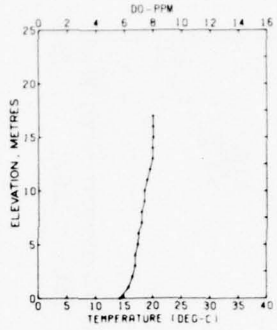
PLATE 4
(Sheet 1 of 7)



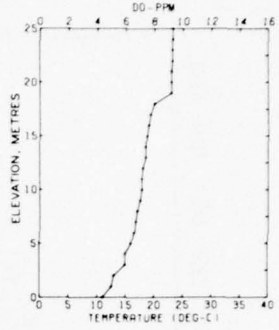
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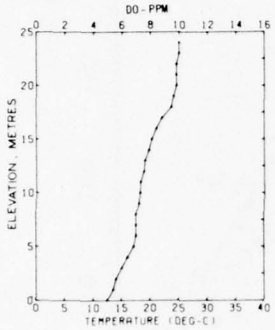
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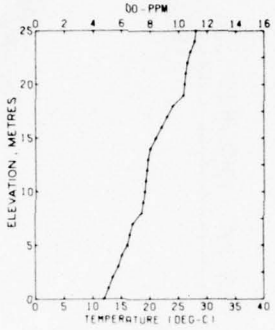
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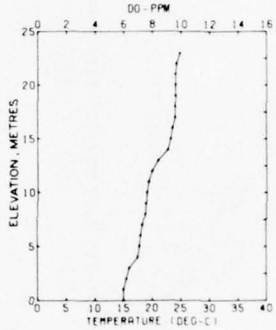
20 JUN 1973



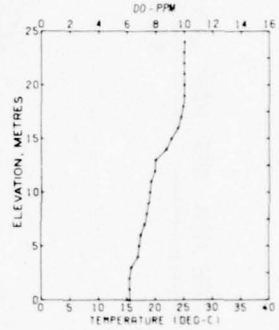
10 JUL 1973



25 JUL 1973



7 AUG 1973



23 AUG 1973

ARBUCKLE LAKE PROFILES

3 MAY 1973 – 23 AUG 1973

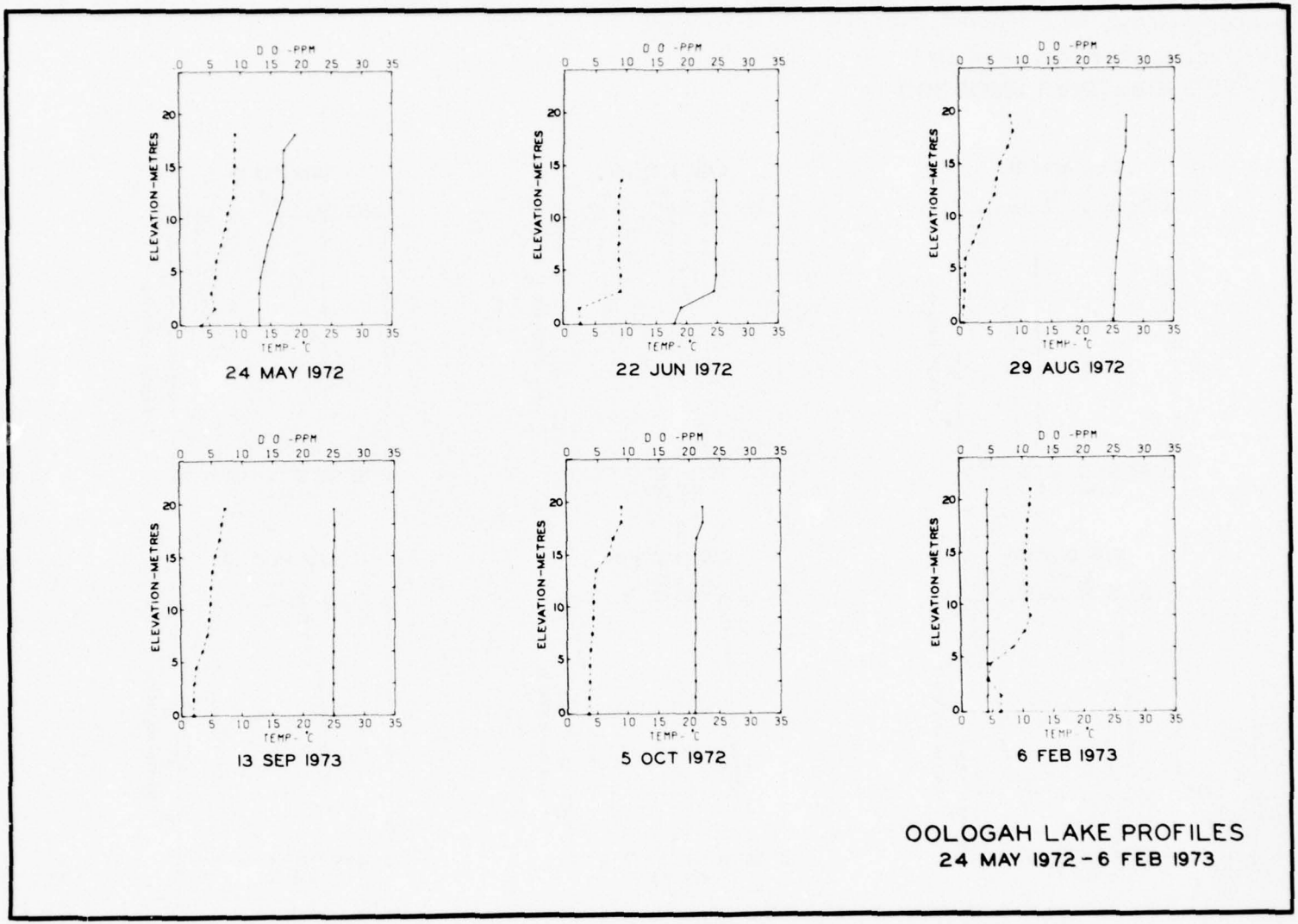


PLATE 4
(Sheet 2 of 7)

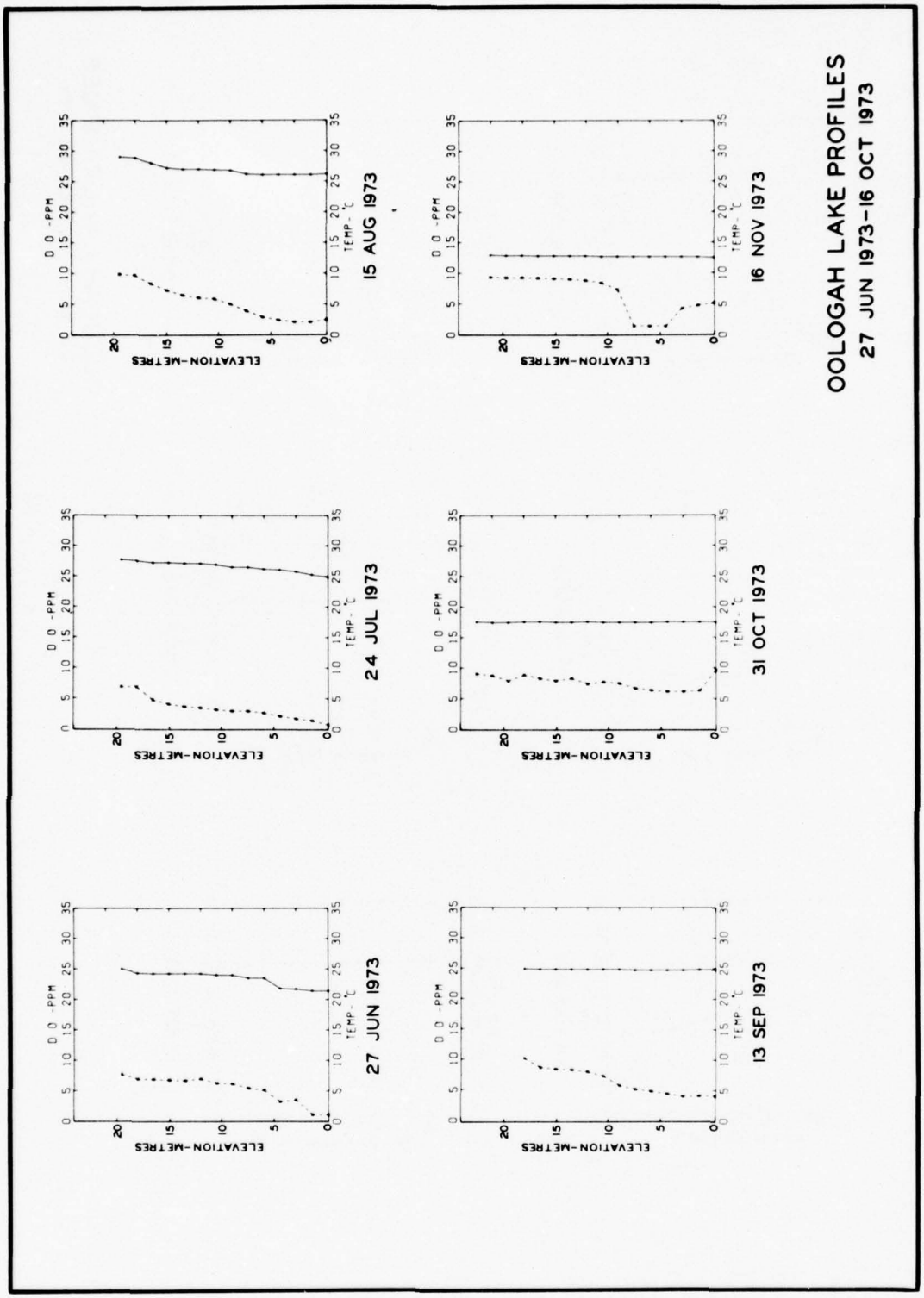


PLATE 4
(Sheet 3 of 7)

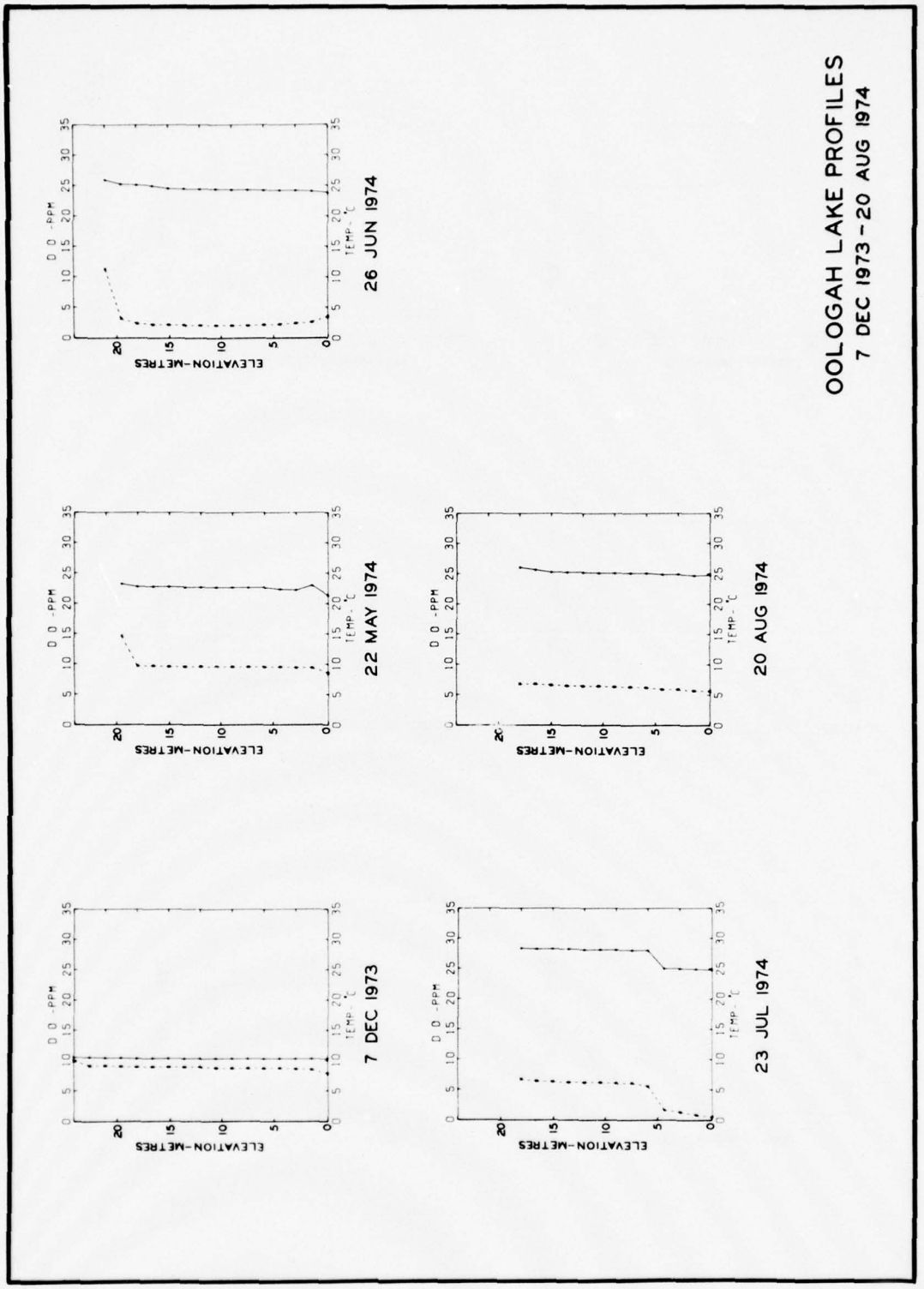


PLATE 4
(Sheet 4 of 7)

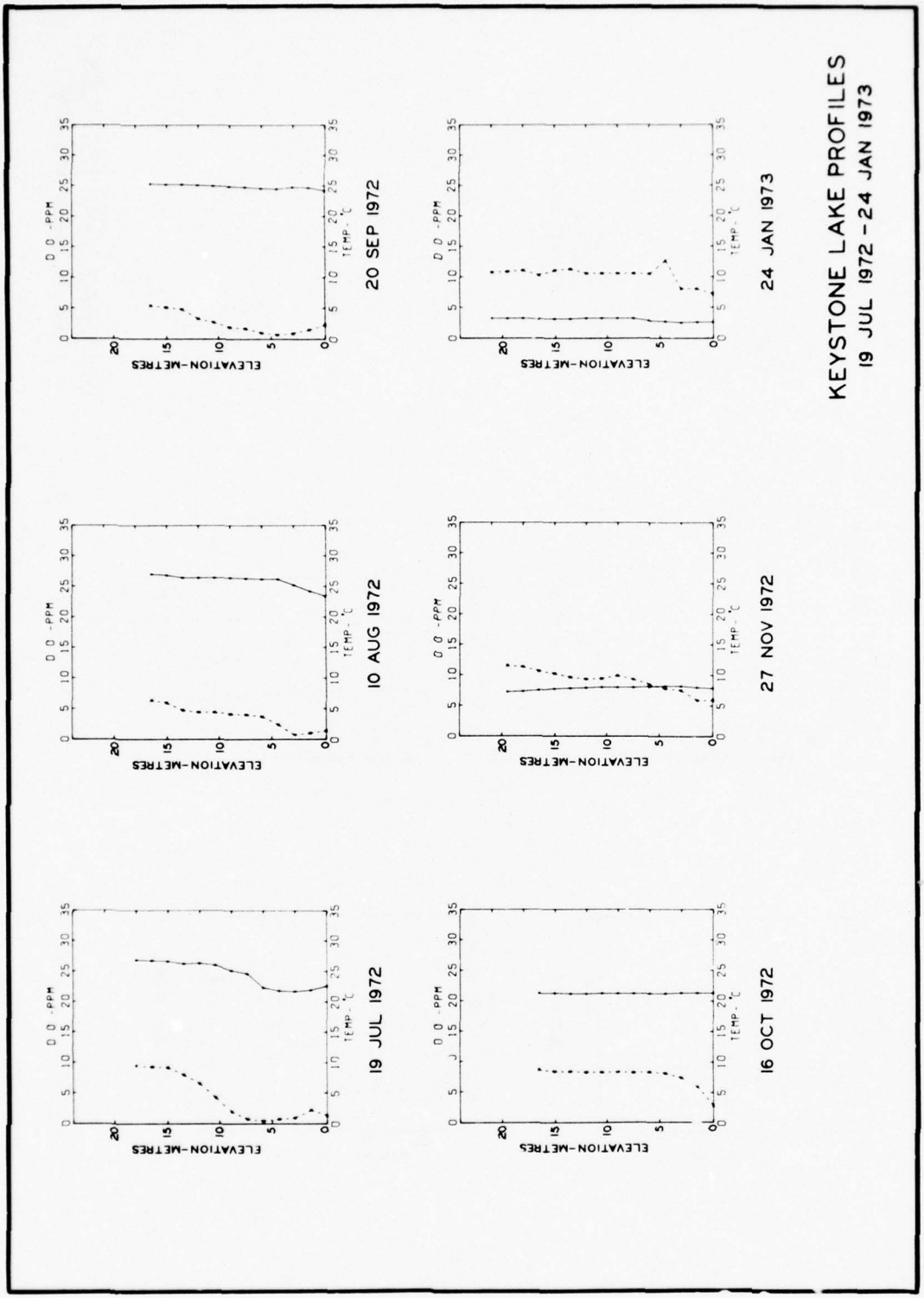
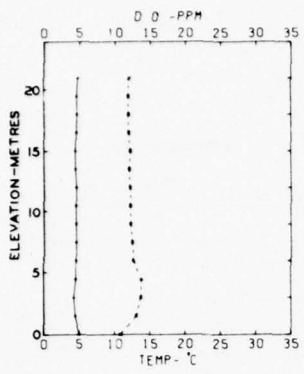
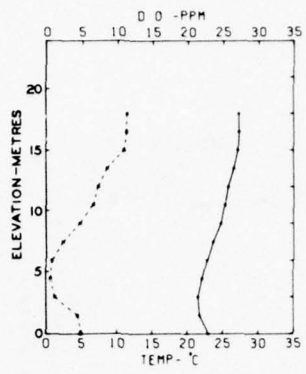


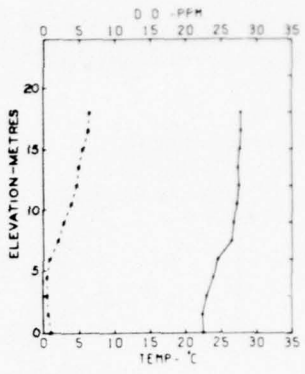
PLATE 4
(Sheet 5 of 7)



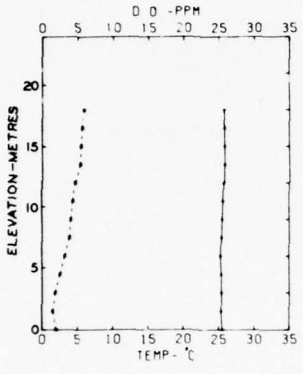
5 FEB 1973



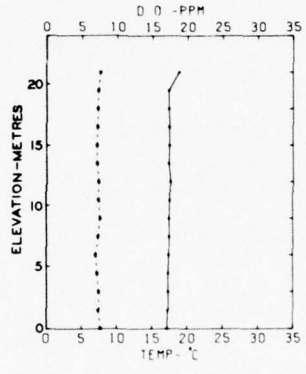
28 JUN 1973



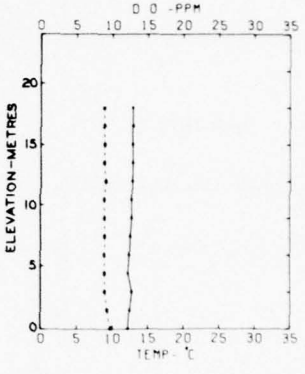
23 JUL 1973



11 SEP 1973



29 OCT 1973



15 NOV 1973

KEYSTONE LAKE PROFILES
5 FEB 1973-15 NOV 1973

PLATE 4
(Sheet 6 of 7)

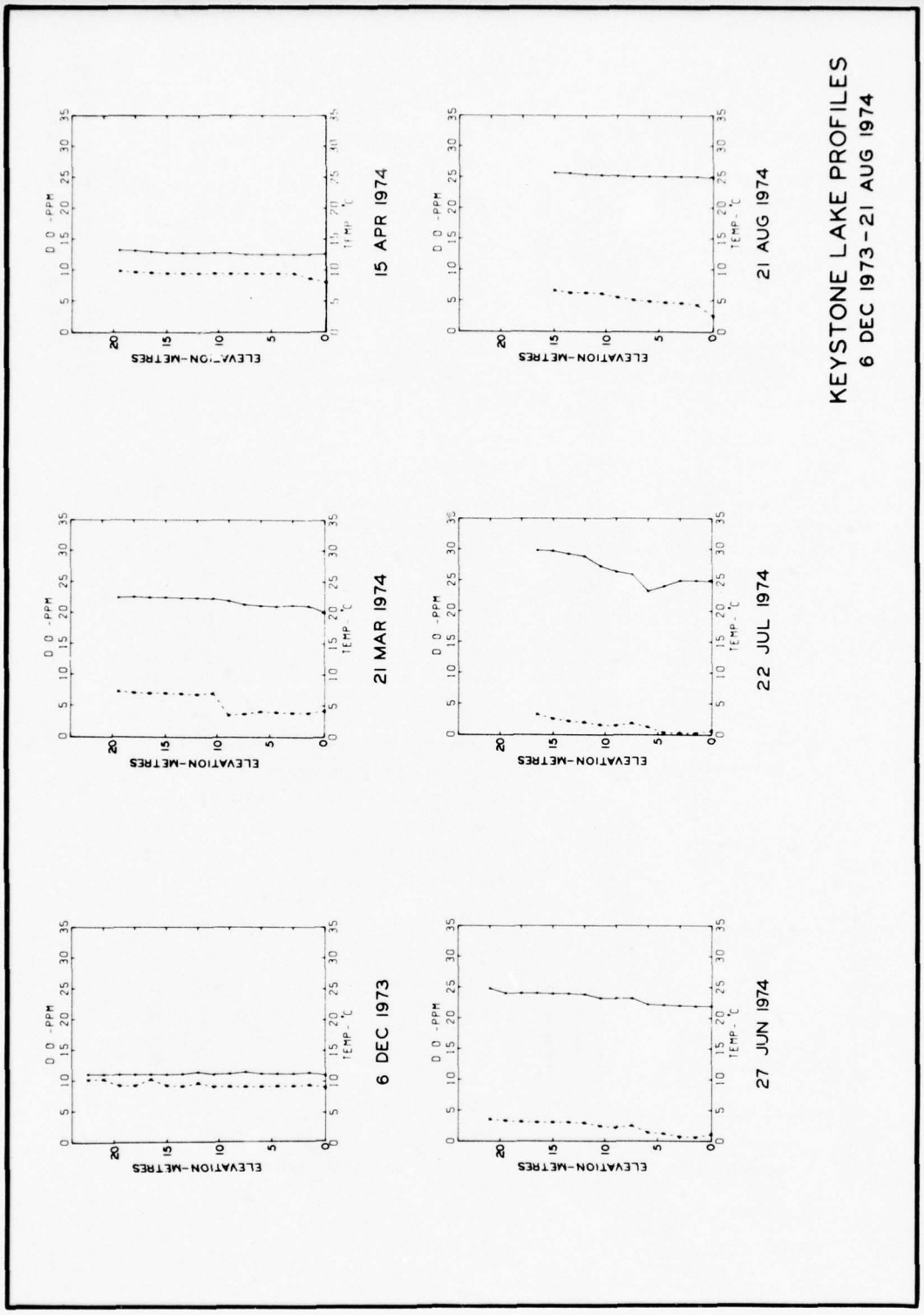


PLATE 4
(Sheet 7 of 7)

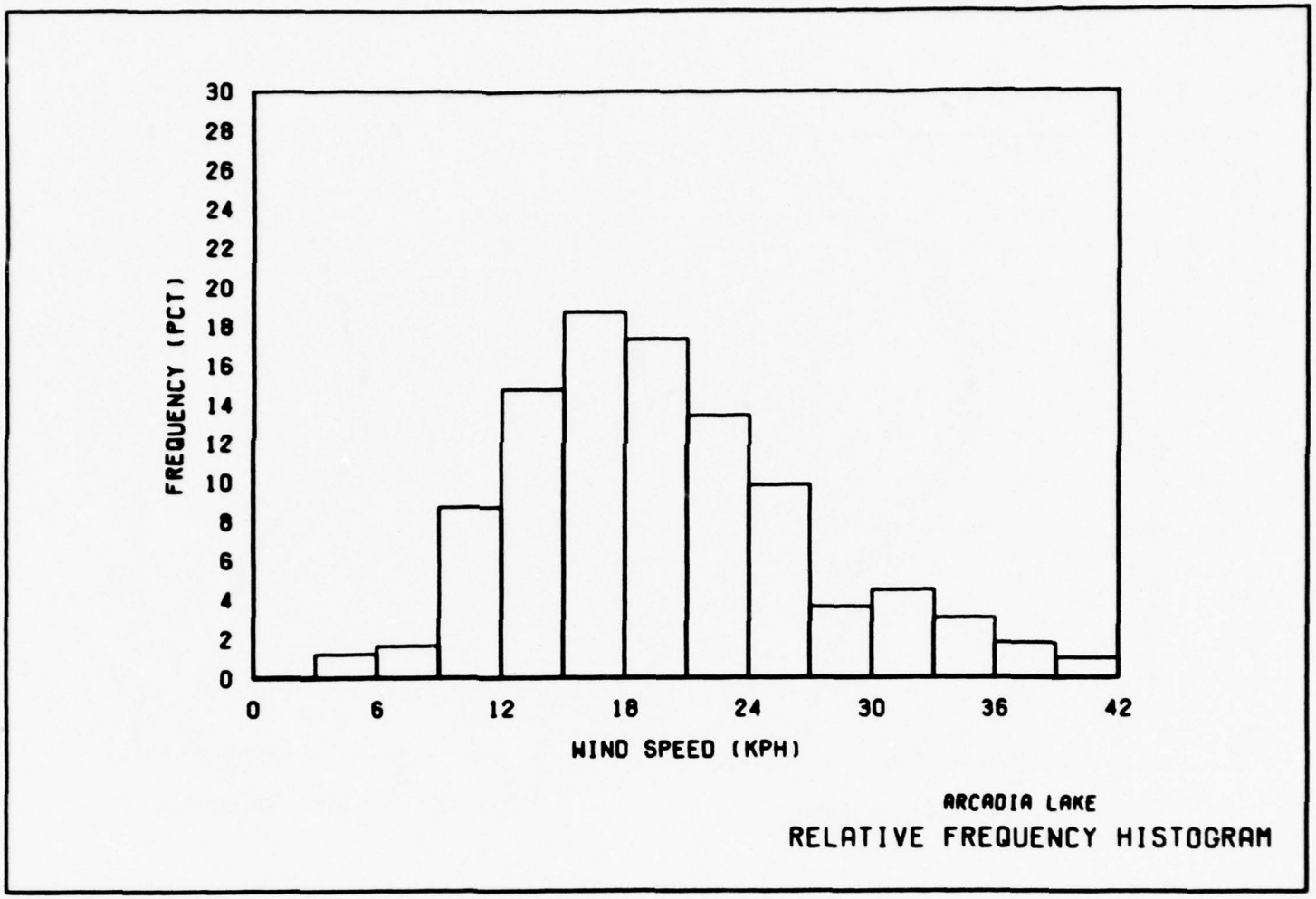
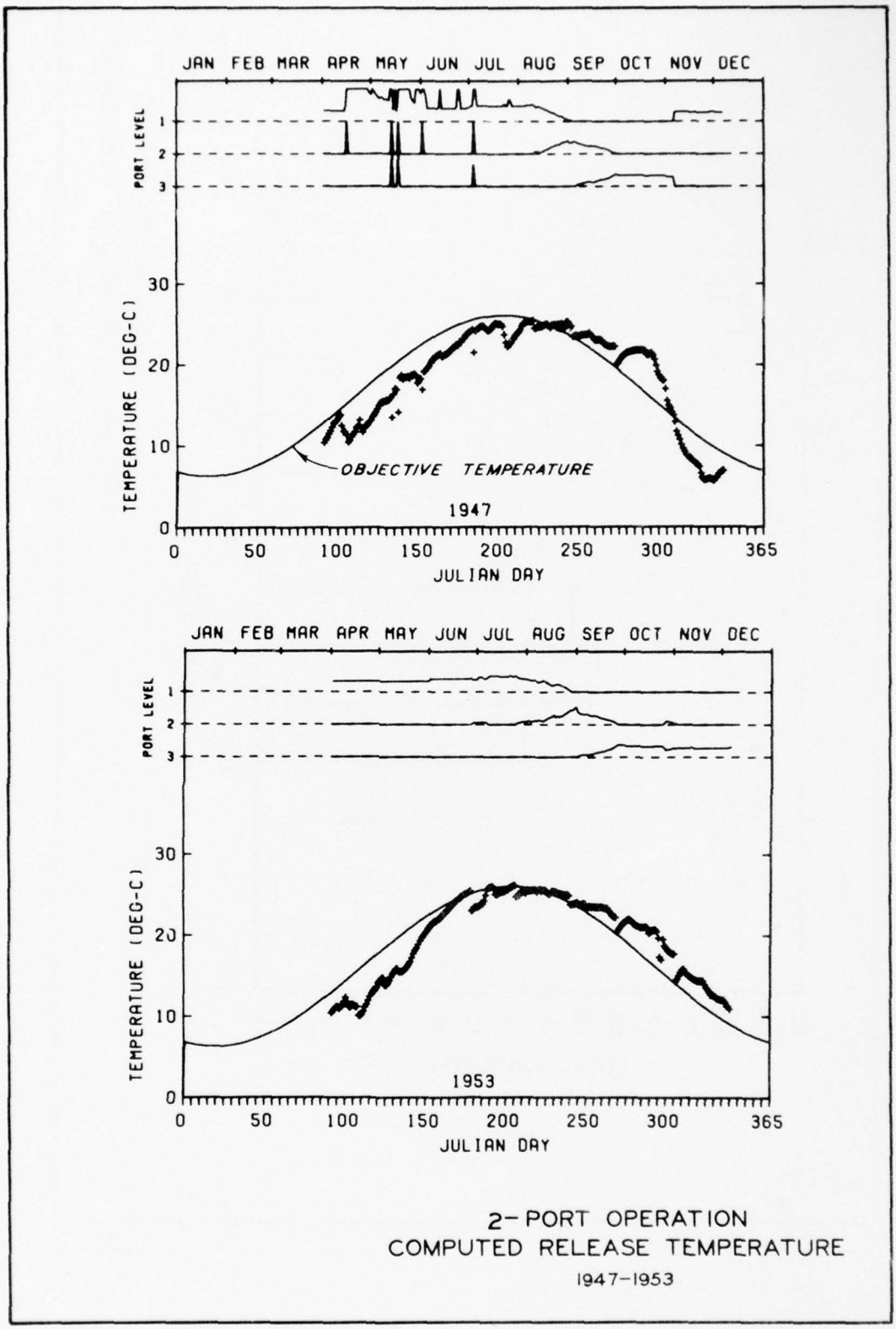
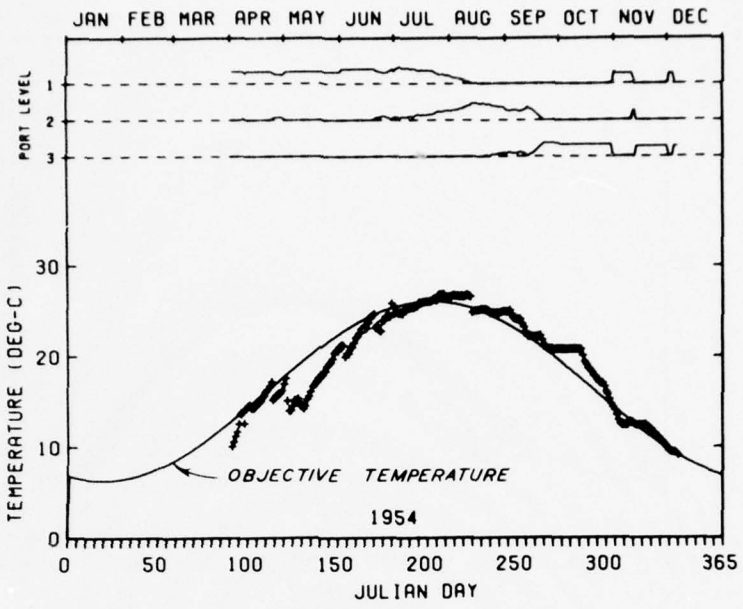

30
28
26
24
22
20
18
16
14
12
10
8
6
4
2
0
FREQUENCY (PCT)
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6
12
18
24
30
36
42
WIND SPEED (KPH)
ARCADIA LAKE
RELATIVE FREQUENCY HISTOGRAM

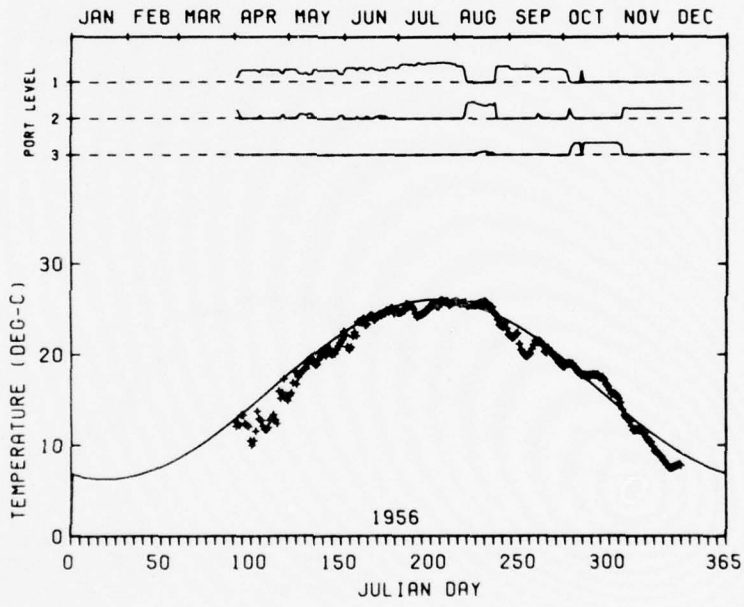
PLATE 5



2- PORT OPERATION
COMPUTED RELEASE TEMPERATURE
1947-1953

PLATE 6
(Sheet 1 of 5)





2-PORT OPERATION
COMPUTED RELEASE TEMPERATURE

1954-1956

PLATE 6
(Sheet 4 of 5)

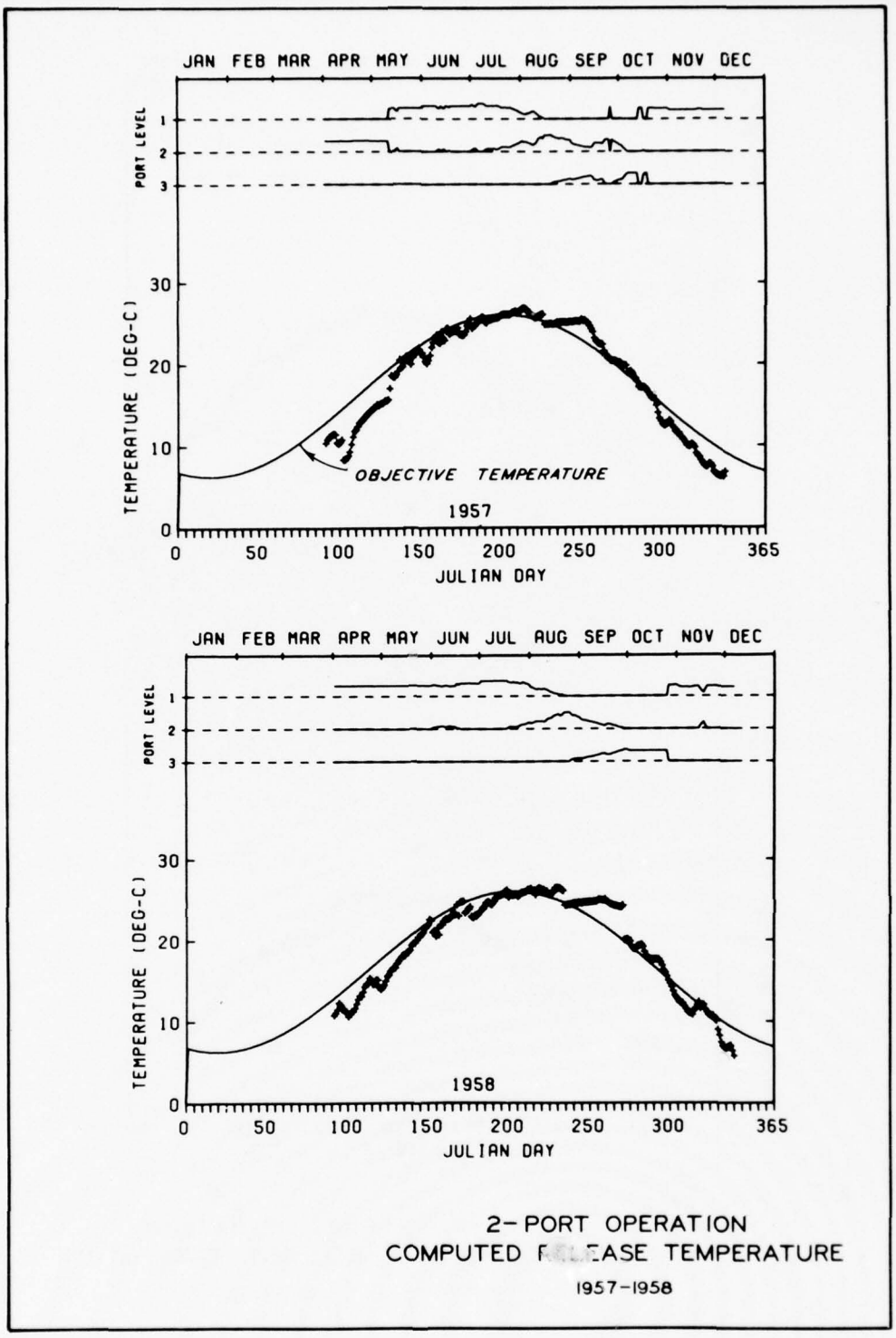
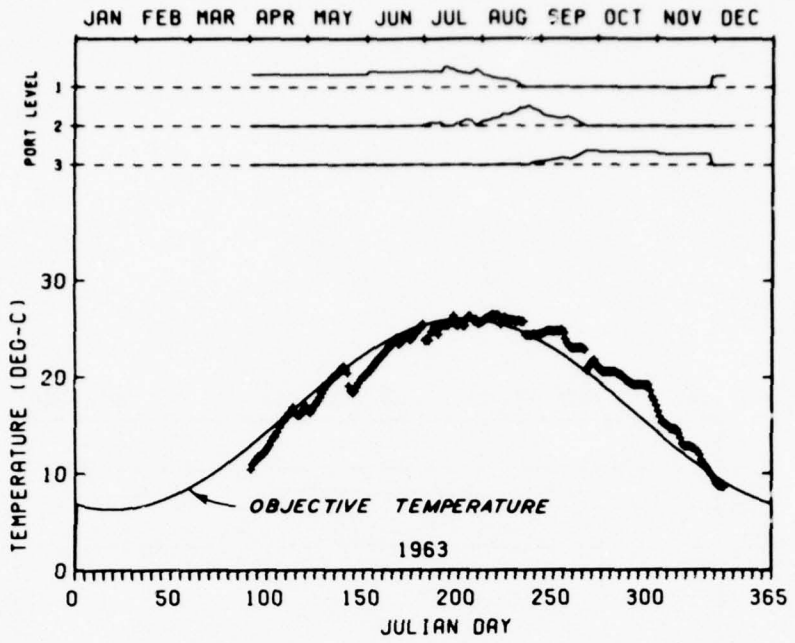
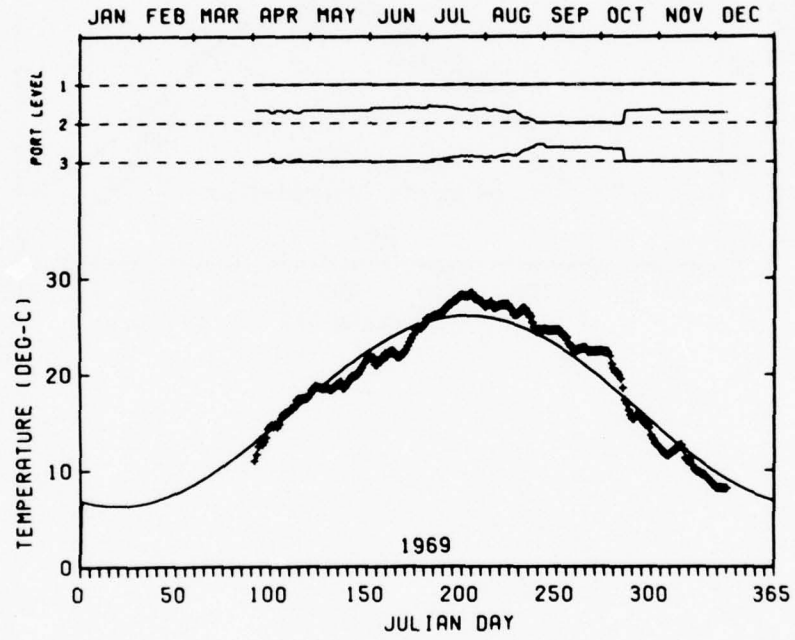


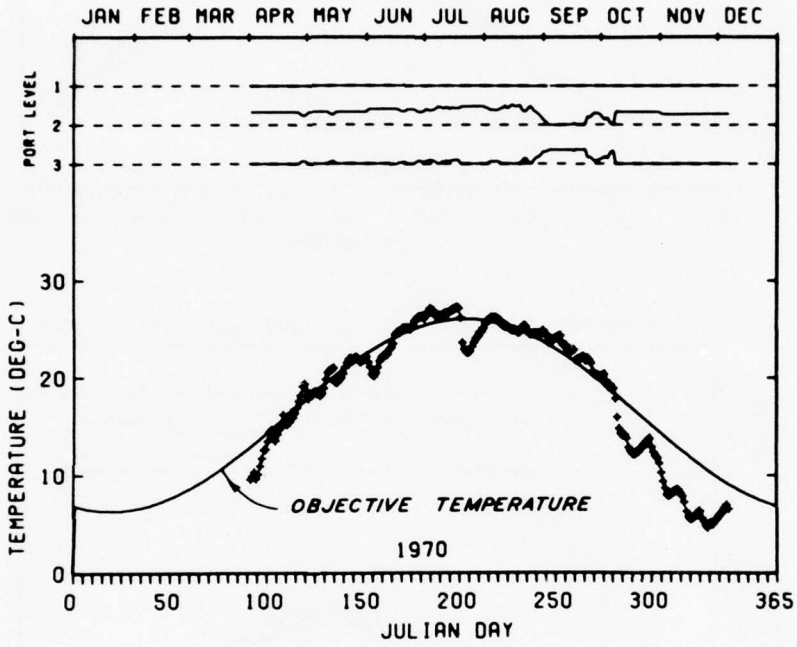
PLATE 6
(Sheet 3 of 5)





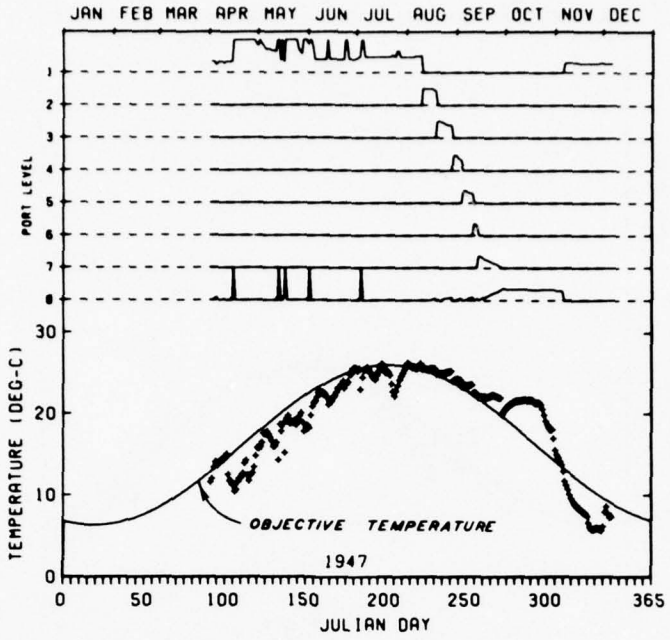
2-PORT OPERATION
COMPUTED RELEASE TEMPERATURE
1963-1969

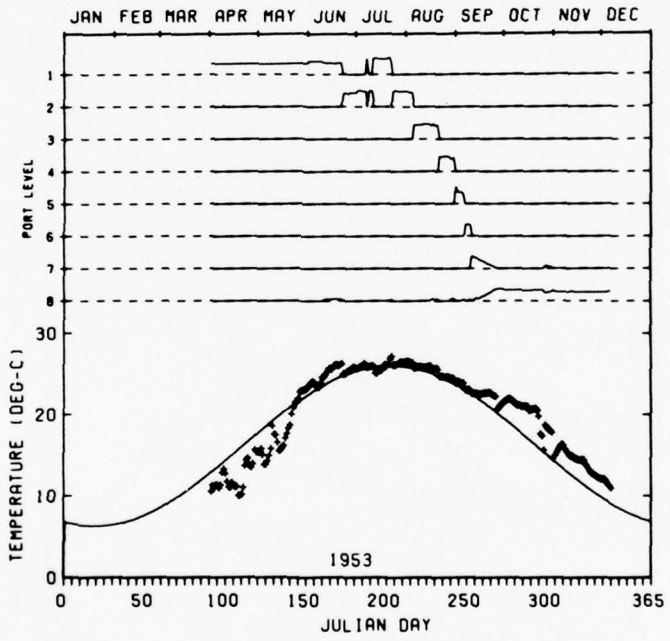
PLATE 6
(Sheet 4 of 5)



2-PORT OPERATION
COMPUTED RELEASE TEMPERATURE
1970

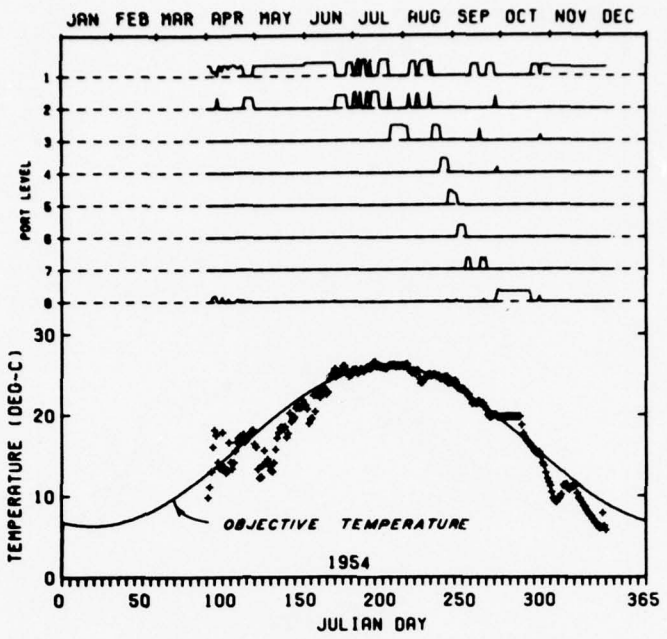
PLATE 6
(Sheet 5 of 5)

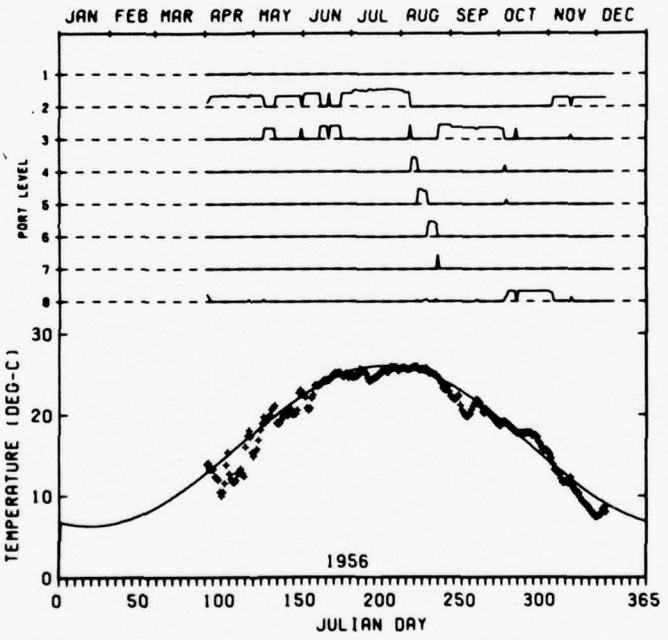




7 - PORT OPERATION
COMPUTED RELEASE TEMPERATURE
1947 - 1953

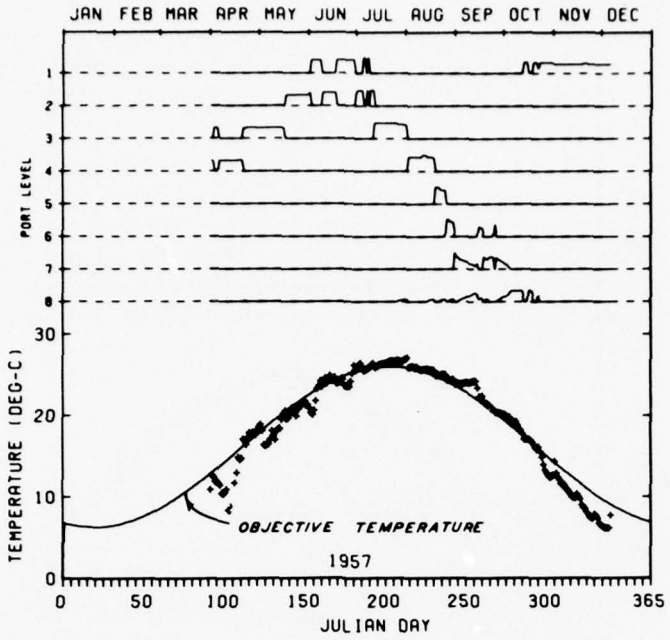
PLATE 7
(Sheet 1 of 5)

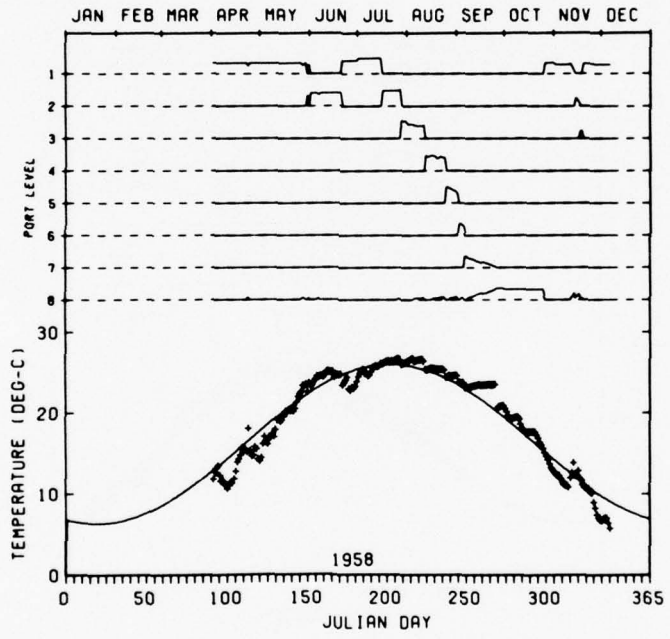




7 - PORT OPERATION
COMPUTED RELEASE TEMPERATURE
1954 - 1956

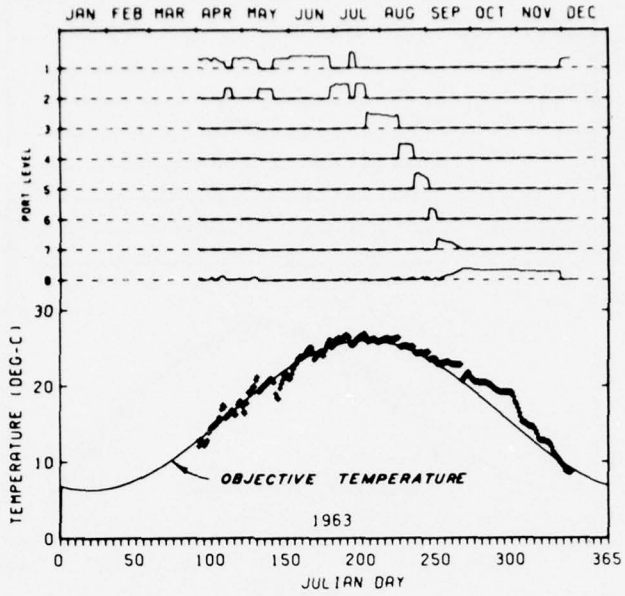
PLATE 7
(Sheet 2 of 5)

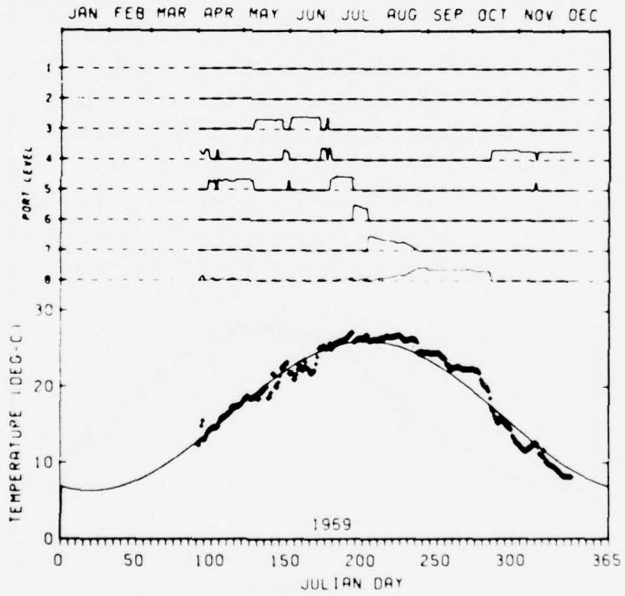




7 - PORT OPERATION
COMPUTED RELEASE TEMPERATURE
1957 - 1958

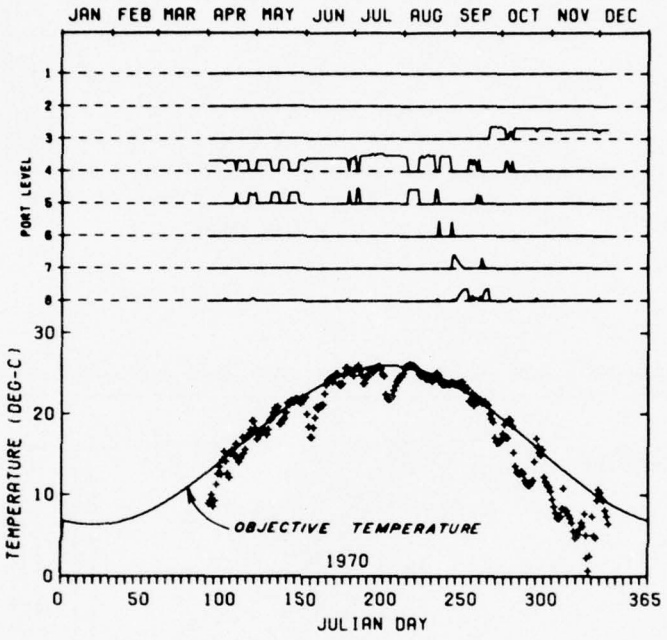
PLATE 7
(Sheet 3 of 5)





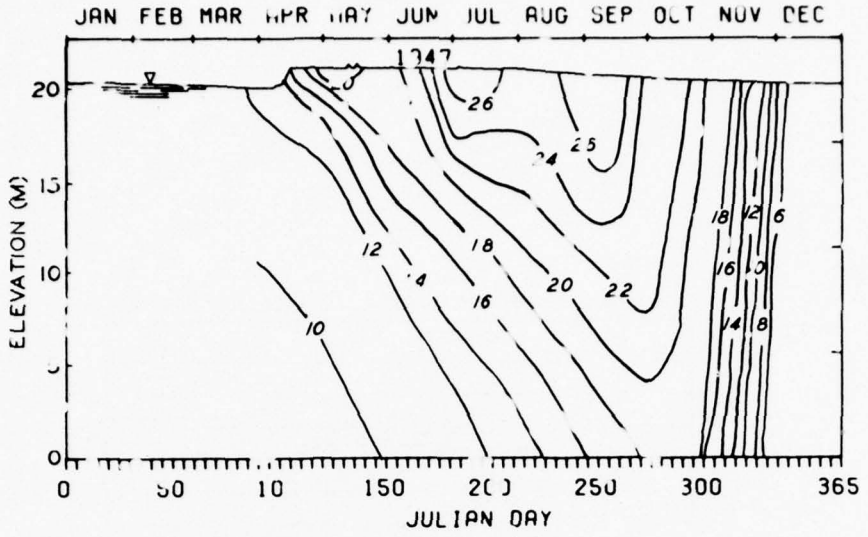
7- PORT OPERATION
COMPUTED RELEASE TEMPERATURE
1963-1969

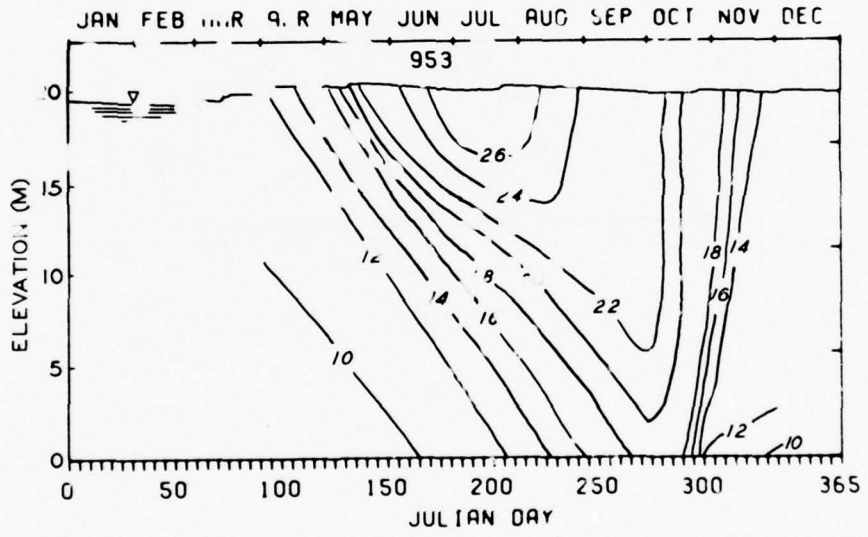
PLATE 7
(Sheet 4 of 5)



7 - PORT OPERATION
COMPUTED RELEASE TEMPERATURE
1970

PLATE 7
(Sheet 5 of 5)

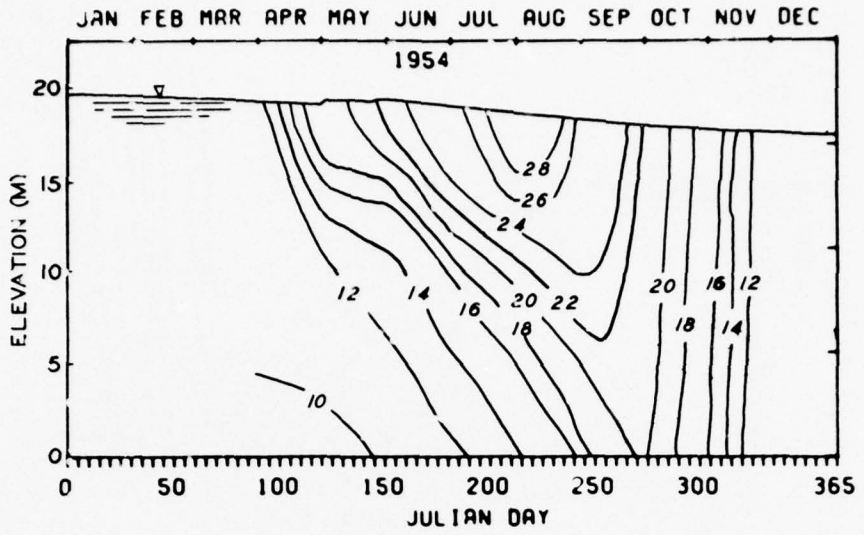


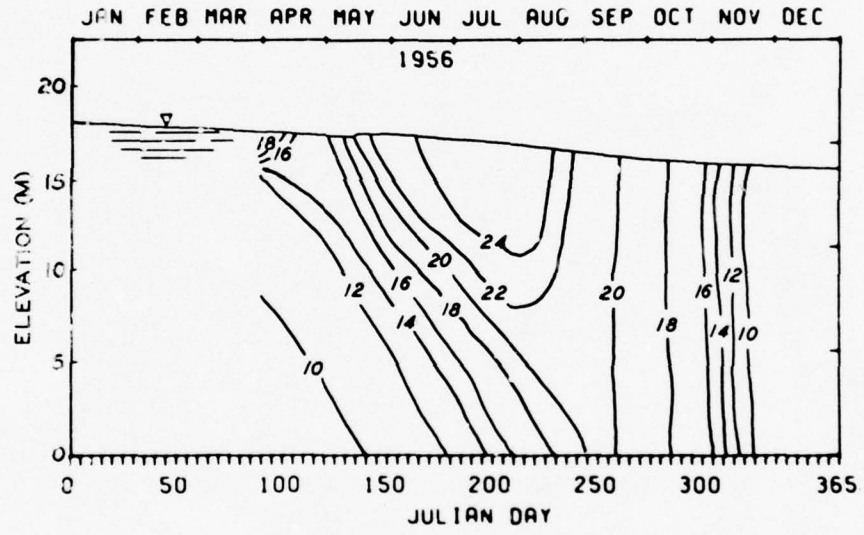


NOTE: CONTOURS SHOWN ARE TEMPERATURES IN DEGREES CELSIUS.

2-PORT OPERATION
ISOTHERMS
1947-1953

PLATE 8
(Sheet 1 of 5)

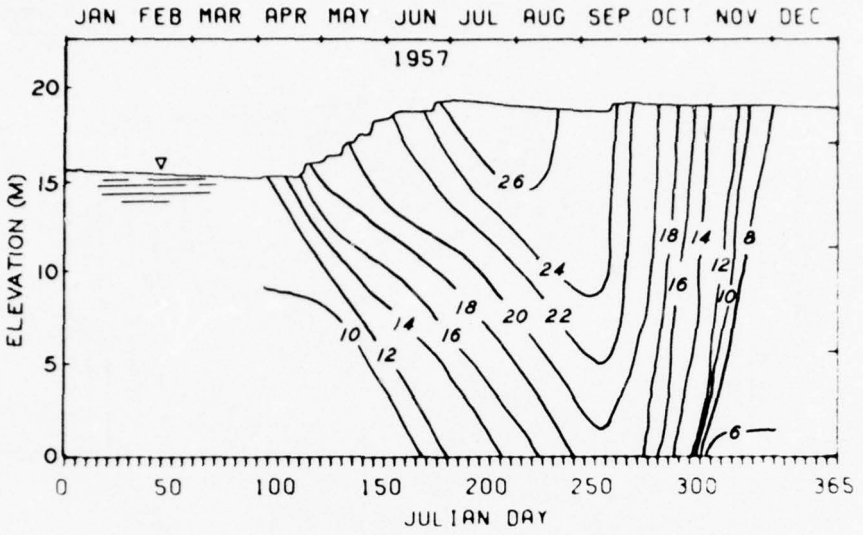


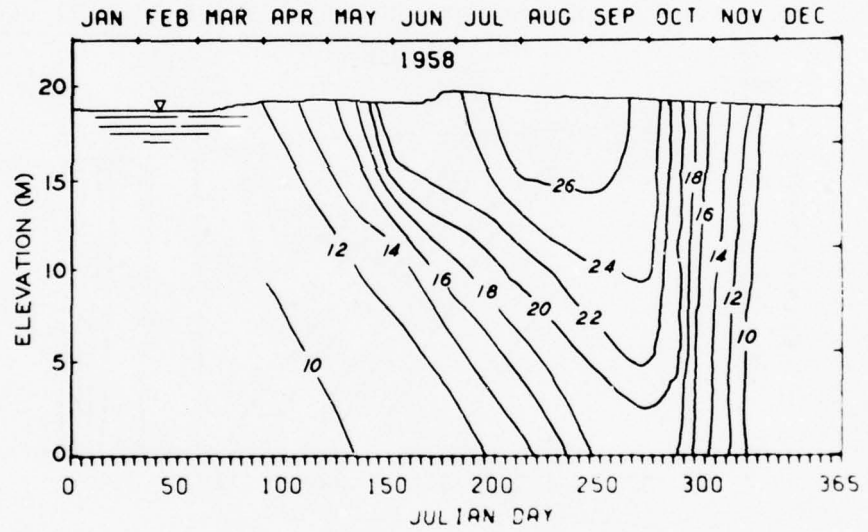


NOTE: CONTOURS SHOWN ARE TEMPERATURES IN DEGREES CELSIUS.

2-PORT OPERATION
ISOTHERMS
1954-1956

PLATE 8
(Sheet 2 of 5)

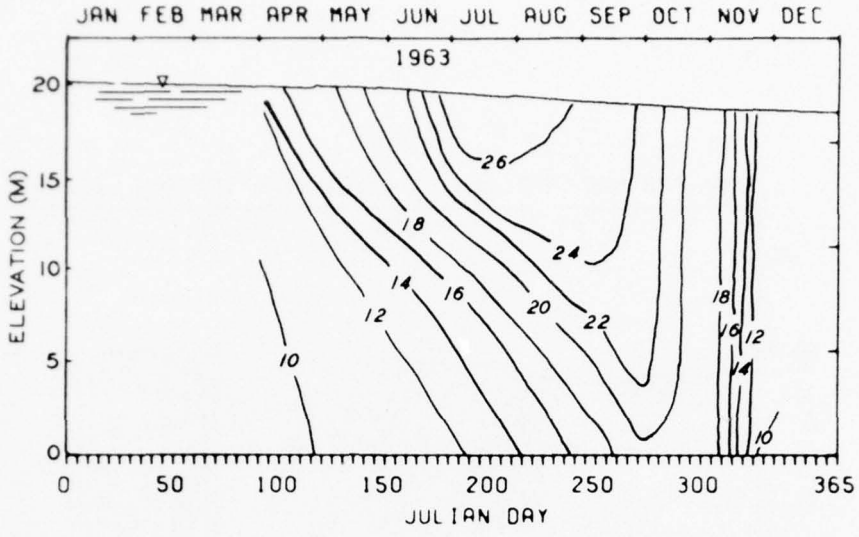


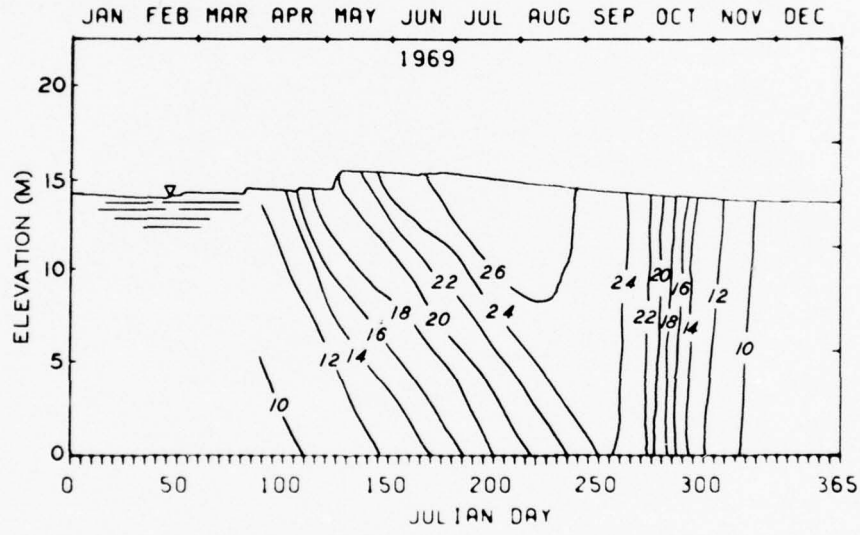


NOTE: CONTOURS SHOWN ARE TEMPERATURES IN DEGREES CELSIUS.

2 - PORT OPERATION
ISOTHERMS
1957 - 1958

PLATE 8
(Sheet 3 of 5)

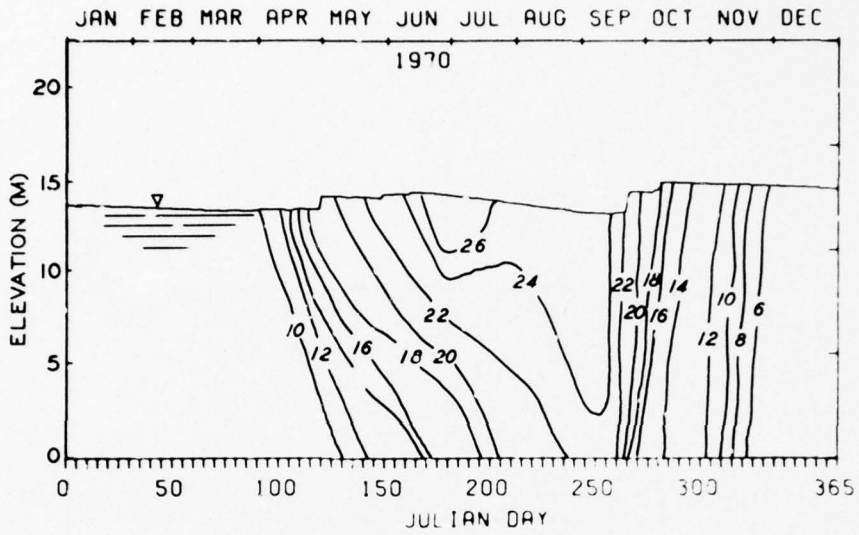




NOTE: CONTOURS SHOWN ARE TEMPERATURES IN DEGREES CELSIUS.

2-PORT OPERATION
ISOTHERMS
1963 - 1969

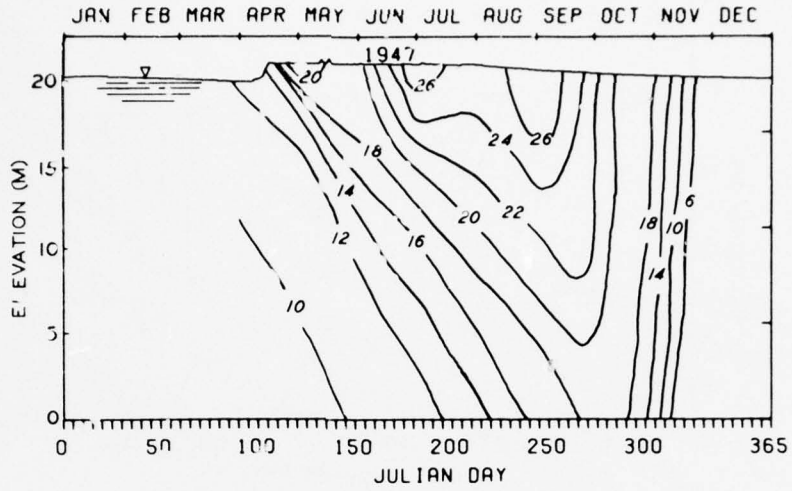
PLATE 8
(Sheet 4 of 5)

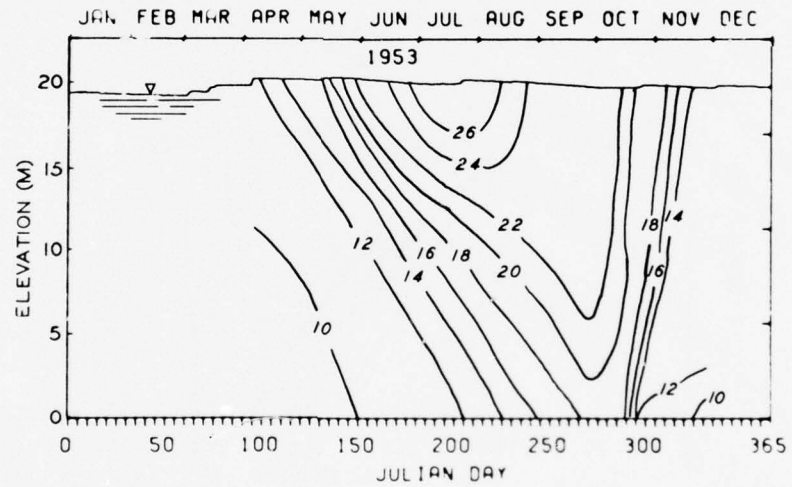


NOTE: CONTOURS SHOWN ARE TEMPERATURES IN DEGREES CELSIUS.

2-PORT OPERATION
ISOTHERMS
1970

PLATE 8
(Sheet 5 of 5)

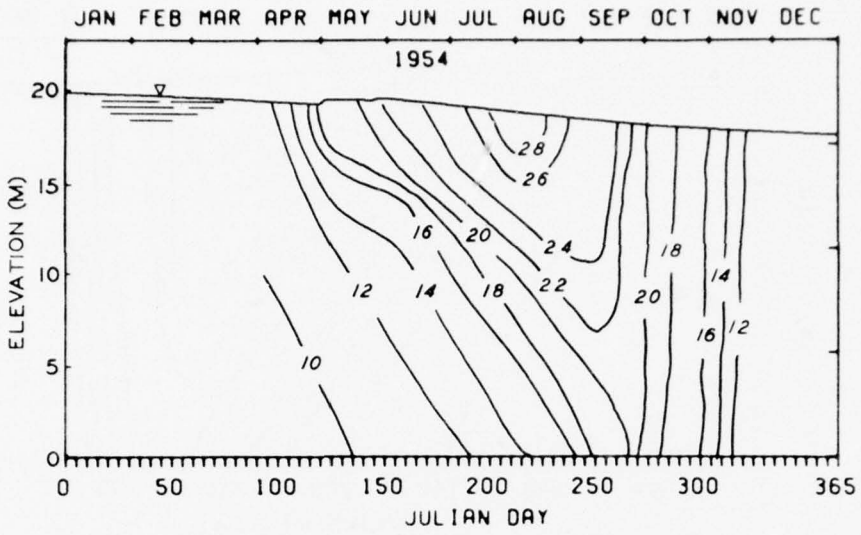


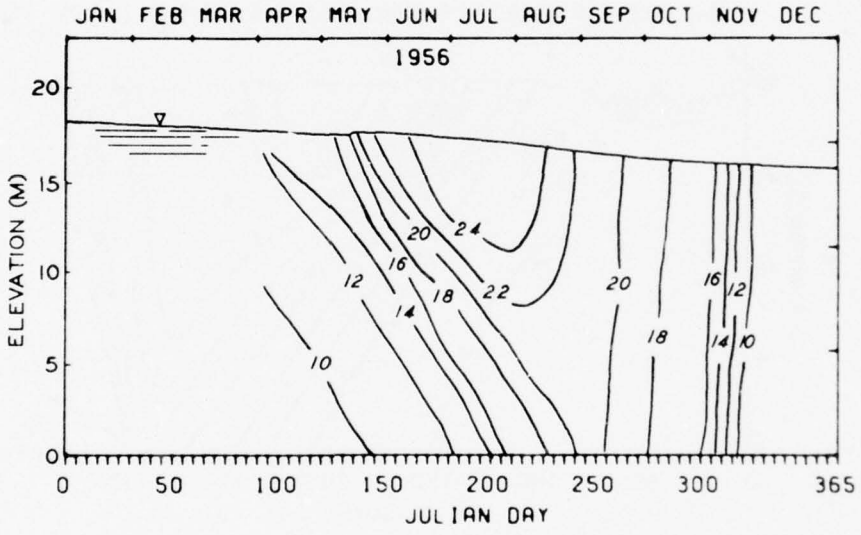


NOTE: CONTOURS SHOWN ARE TEMPERATURES IN DEGREES CELSIUS

7-PORT OPERATION
ISOTHERMS
1947-1953

PLATE 9
(Sheet 1 of 5)

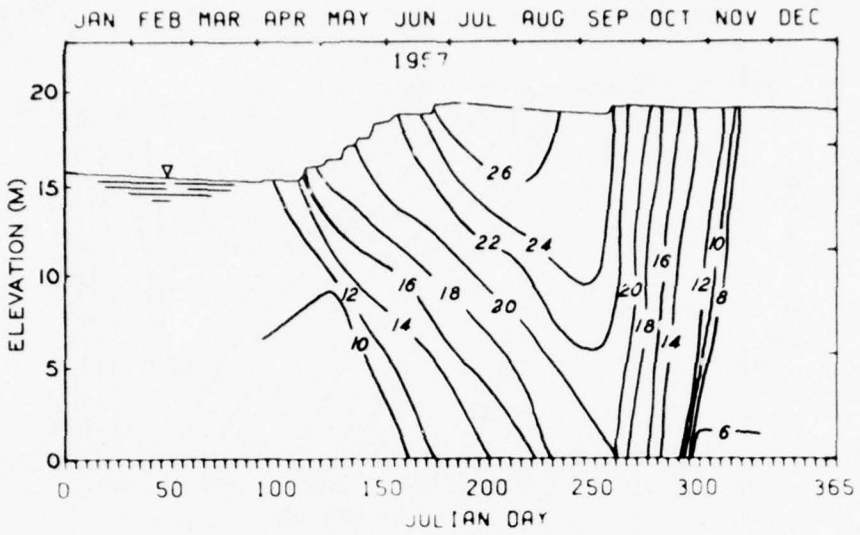


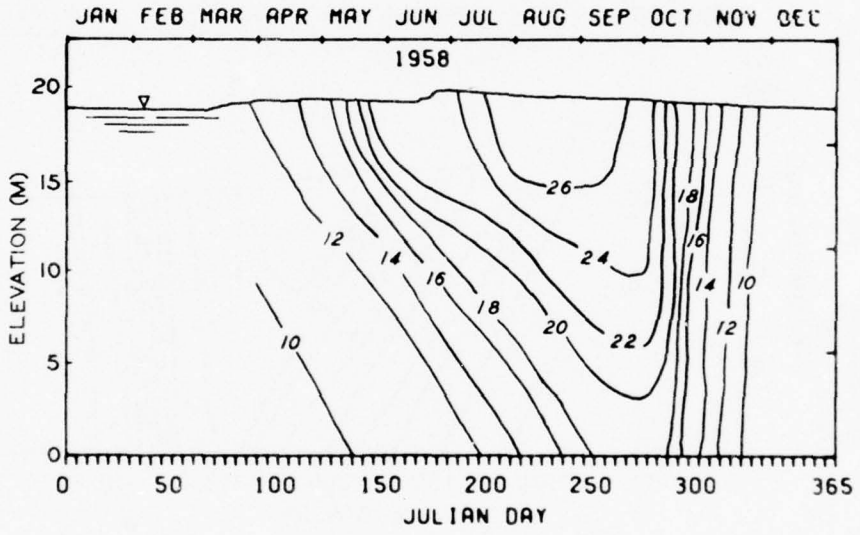


NOTE: CONTOURS SHOWN ARE TEMPERATURES IN DEGREES CELSIUS.

7-PORT OPERATION
ISOTHERMS
1954 - 1956

PLATE 9
(Sheet 2 of 5)

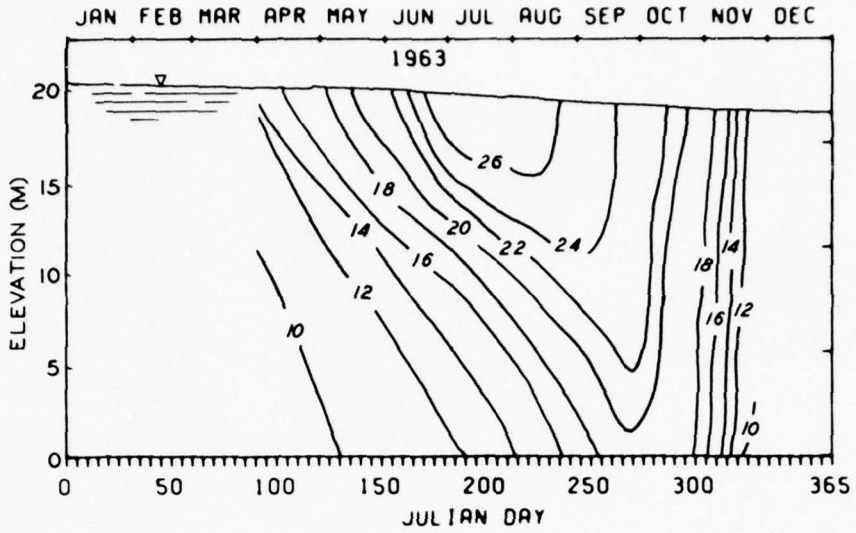


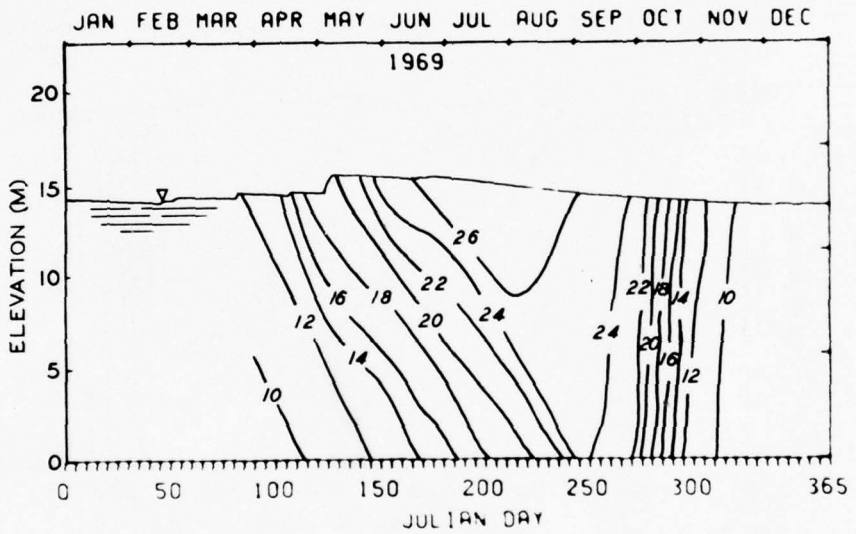


NOTE: CONTOURS SHOWN ARE TEMPERATURES IN DEGREES CELSIUS

7-PORT OPERATION
ISOTHERMS
1957-1958

PLATE 9
(Sheet 3 of 5)

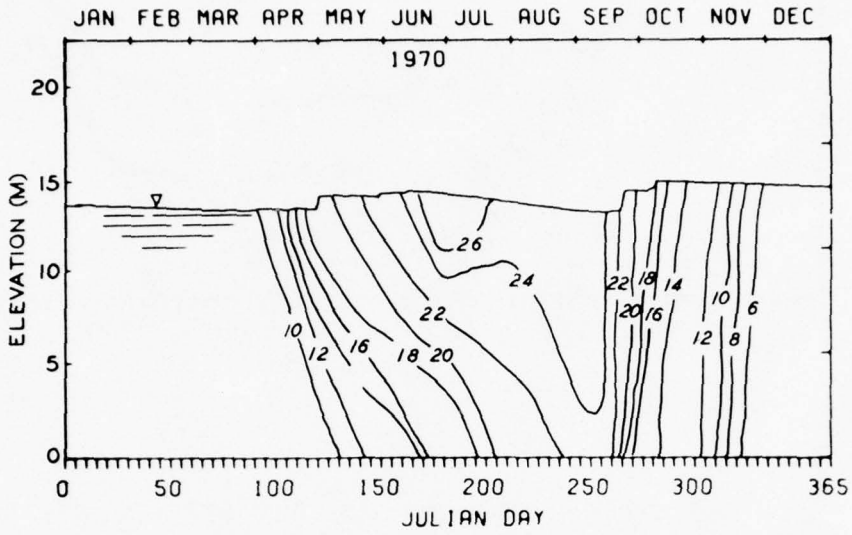




NOTE: CONTOURS SHOWN ARE TEMPERATURES IN DEGREES CELSIUS.

7-PORT OPERATION
ISOTHERMS
1963 - 1969

PLATE 9
(Sheet 4 of 5)



NOTE: CONTOURS SHOWN ARE TEMPERATURES IN DEGREES CELSIUS.

7 – PORT OPERATION
ISOTHERMS
1970

PLATE 9
(Sheet 5 of 5)

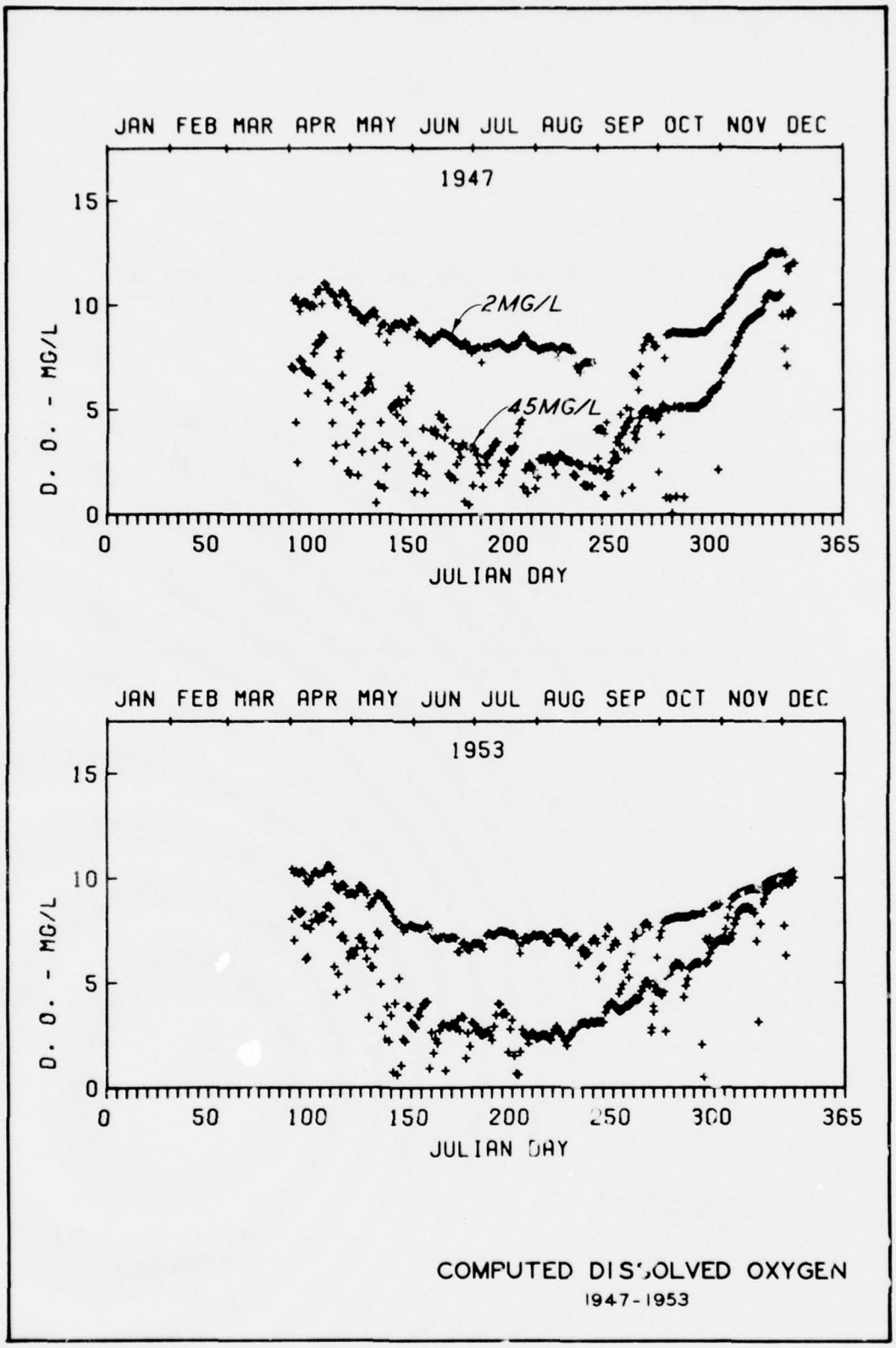
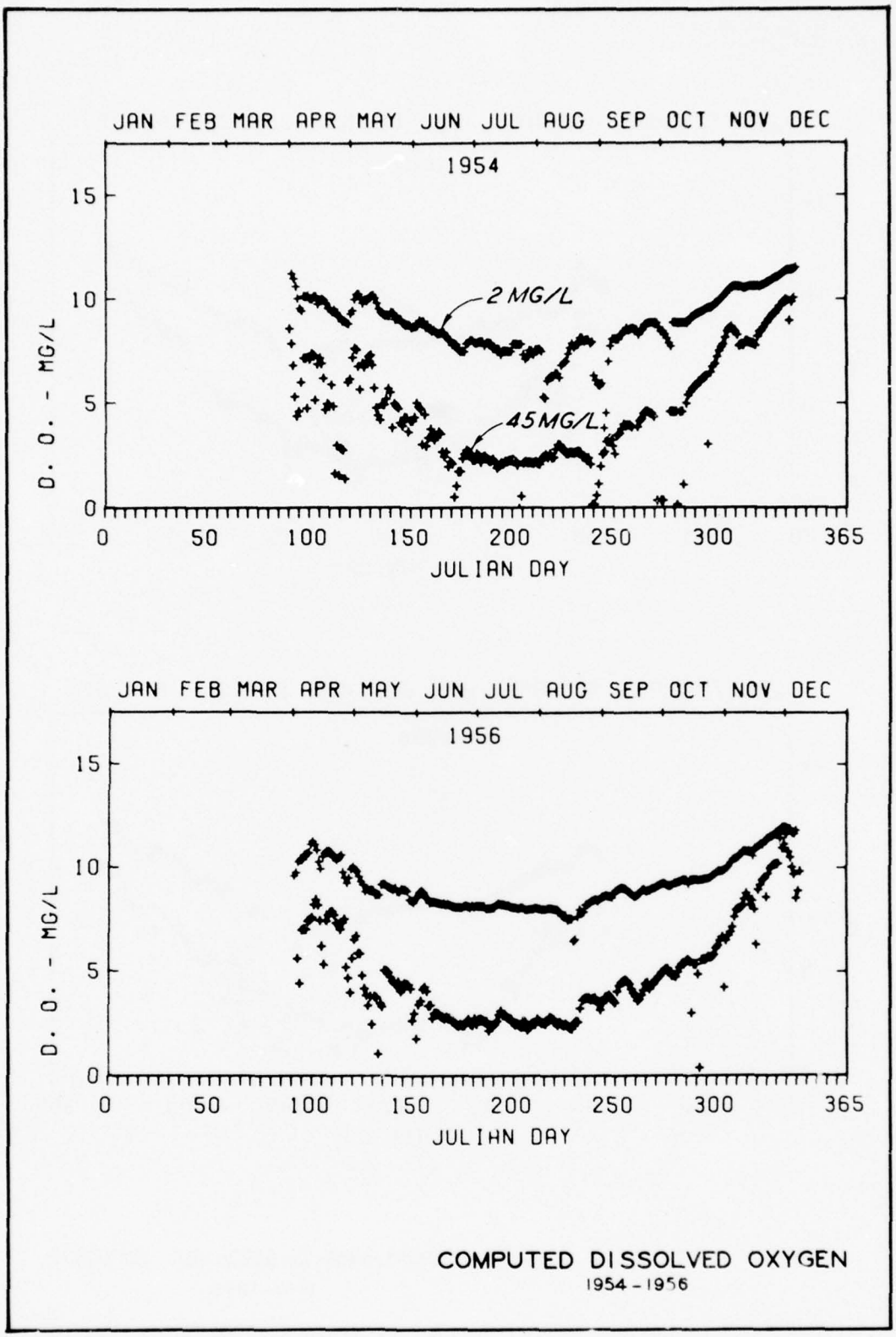


PLATE 10
(Sheet 1 of 5)



COMPUTED DISSOLVED OXYGEN
1954 - 1956

PLATE 10
(Sheet 2 of 5)

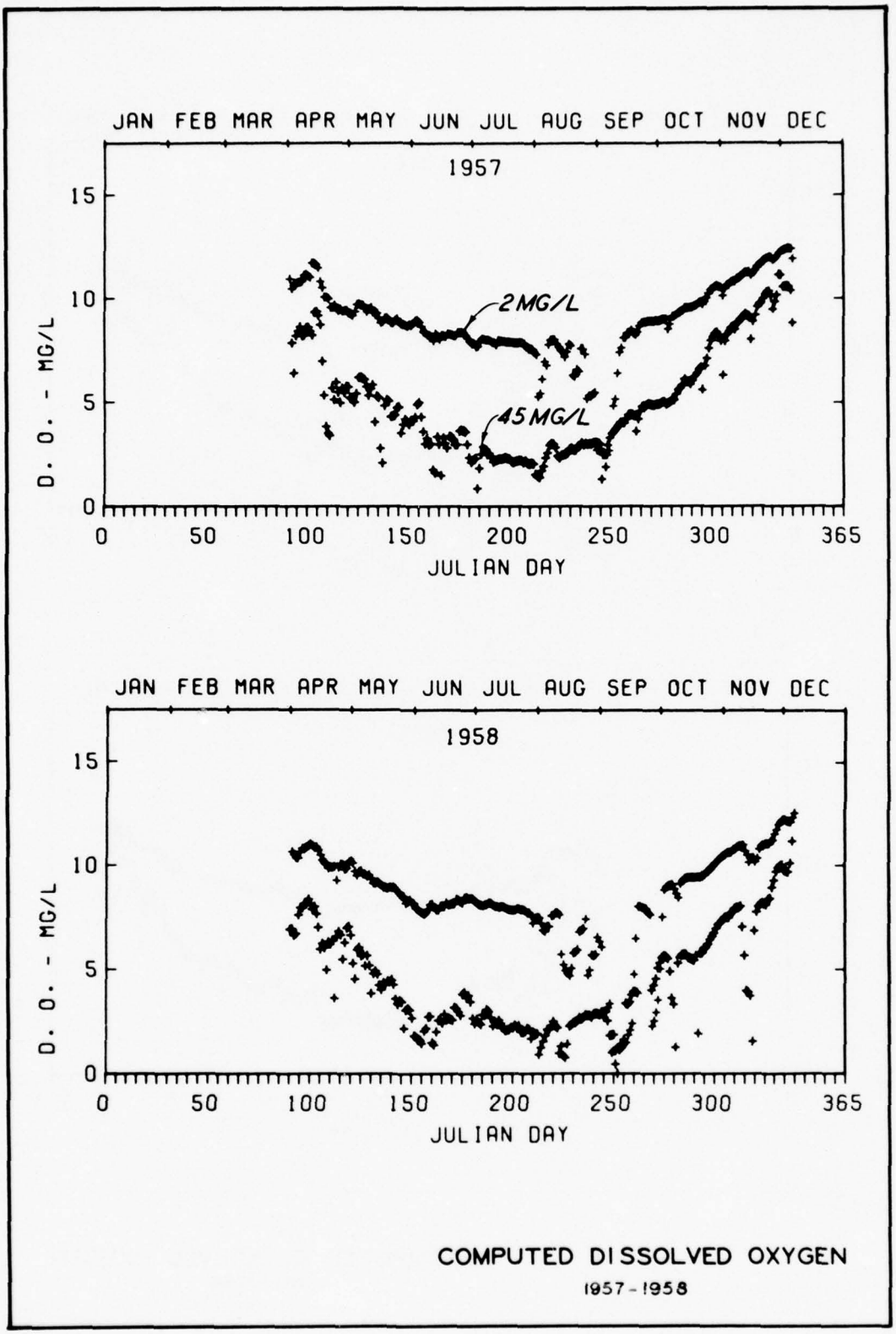
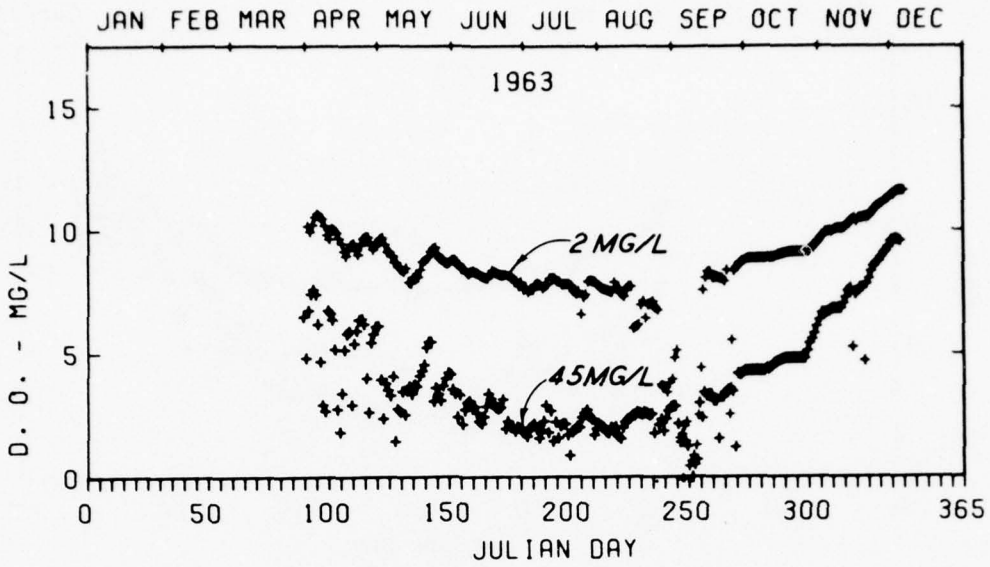
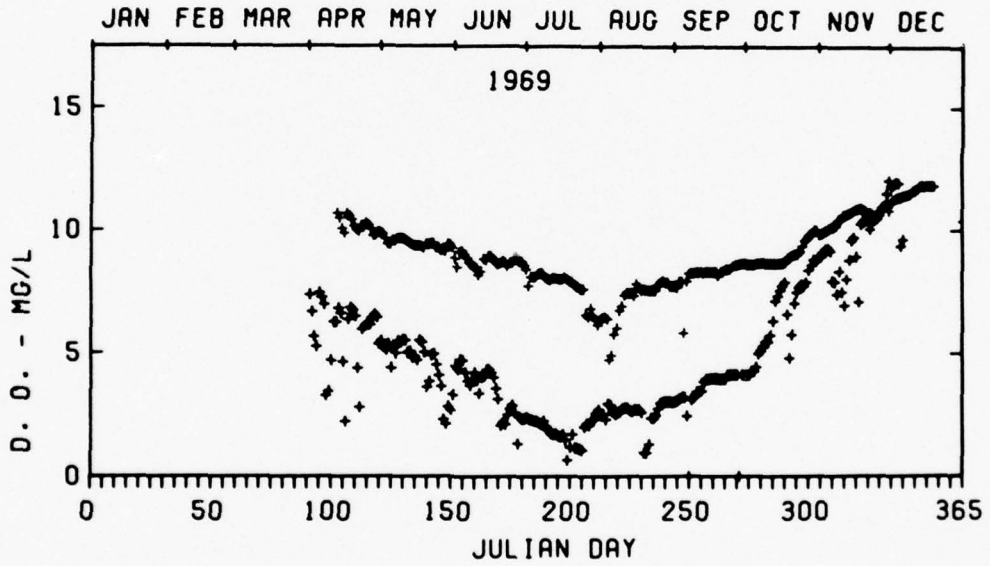


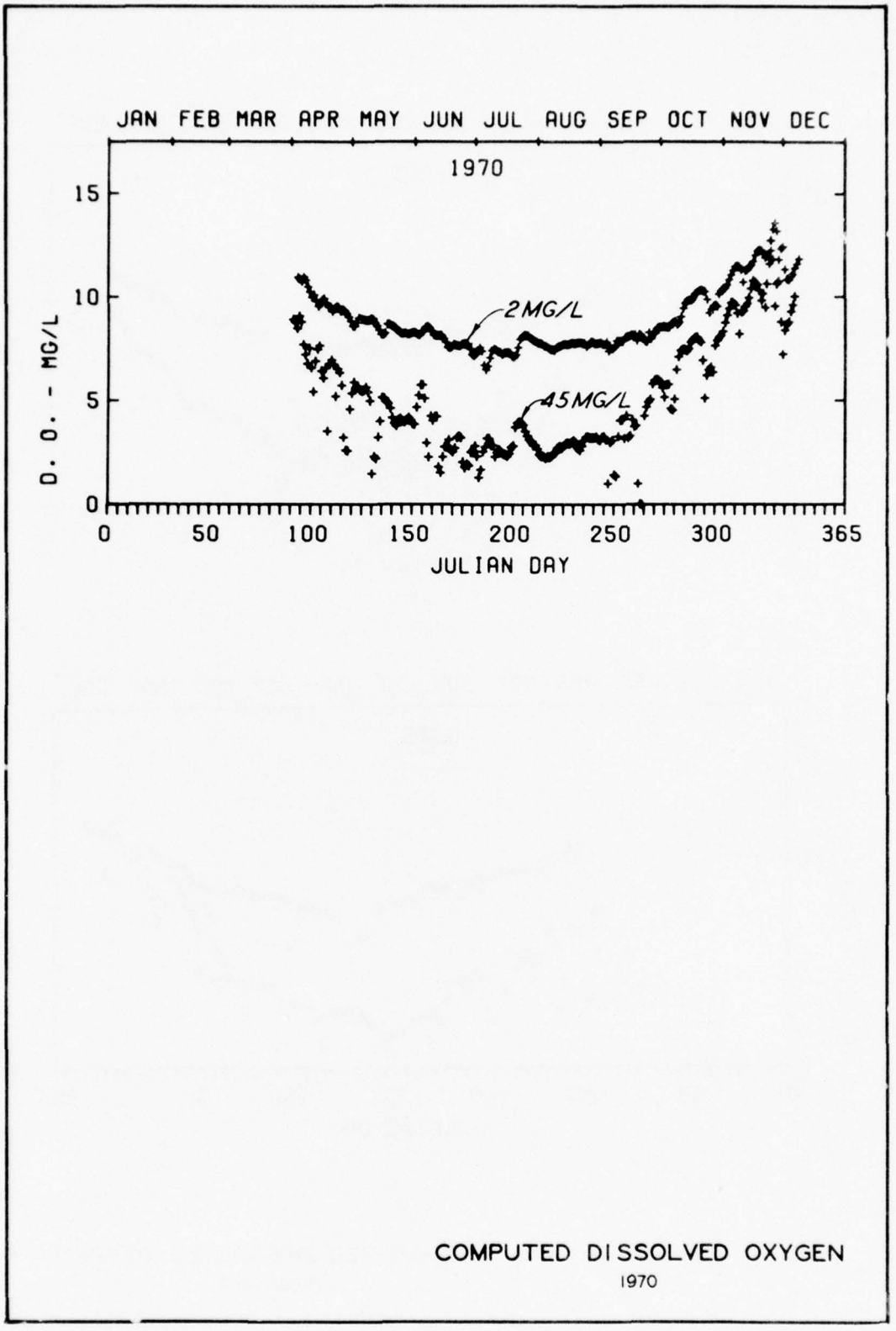
PLATE 10
(Sheet 3 of 5)





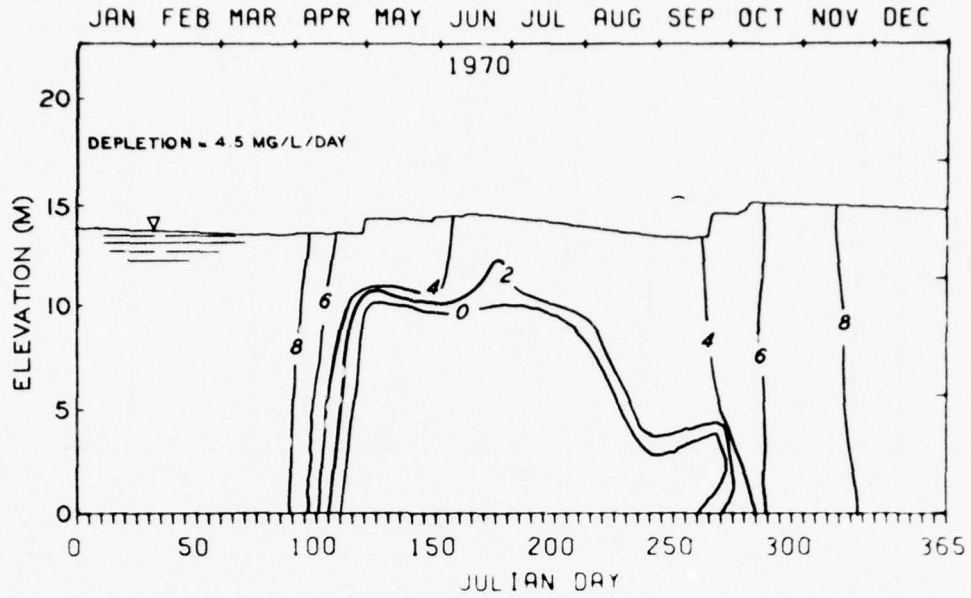
COMPUTED DISSOLVED OXYGEN
1963 - 1969

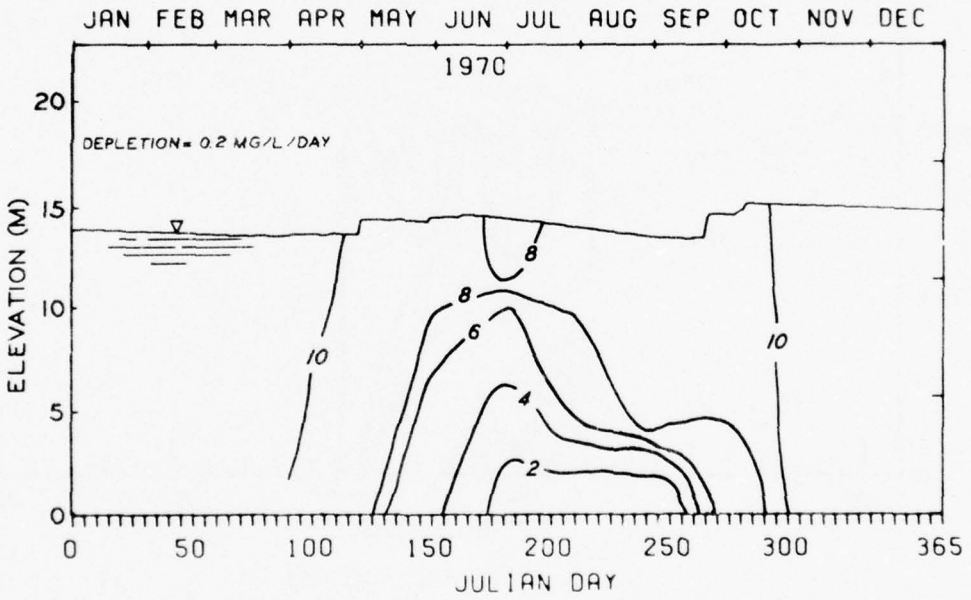
PLATE 10
(Sheet 4 of 5)



COMPUTED DISSOLVED OXYGEN
1970

PLATE 10
(Sheet 5 of 5)





DISSOLVED OXYGEN ISOGRAMS

1970

PLATE 11

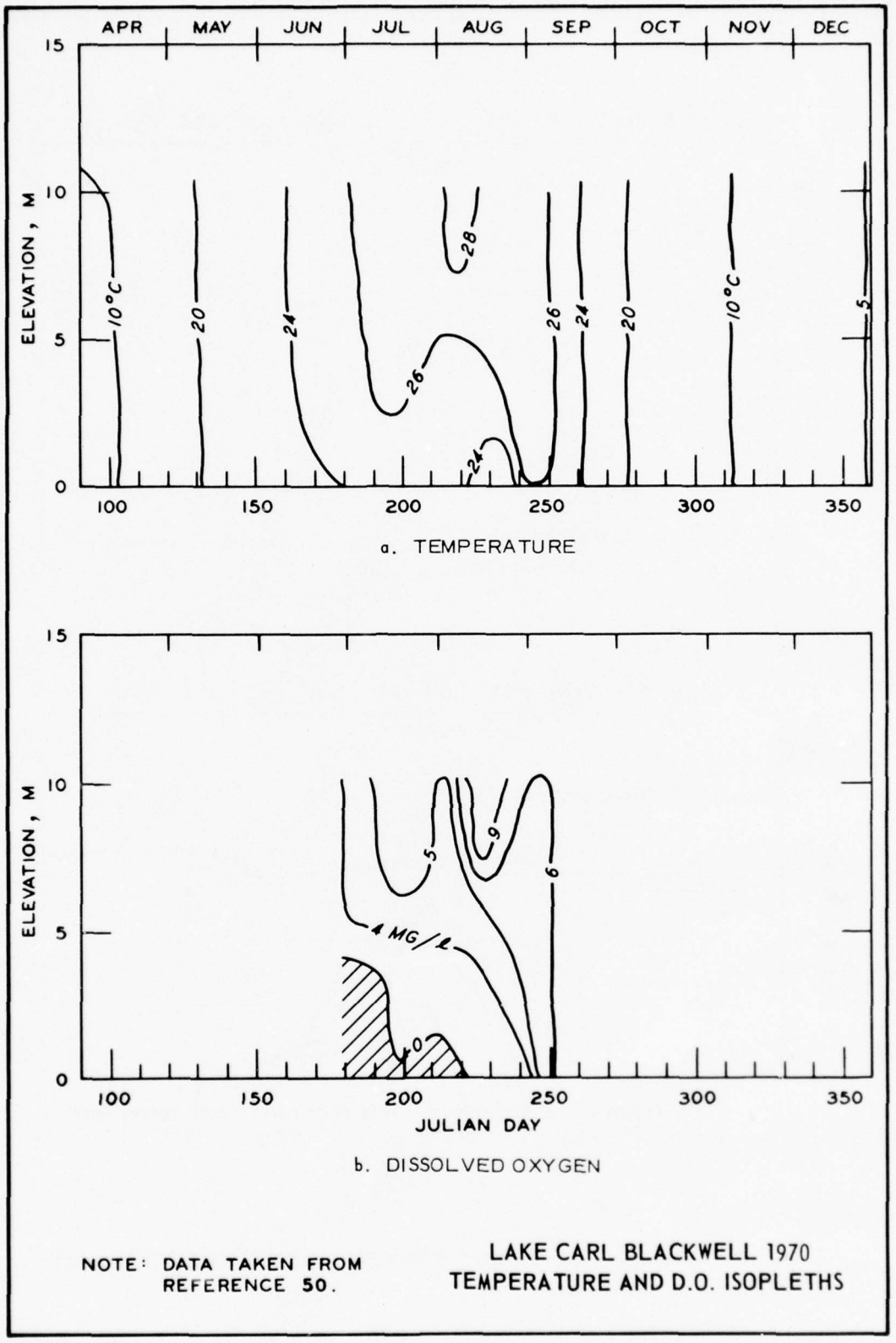
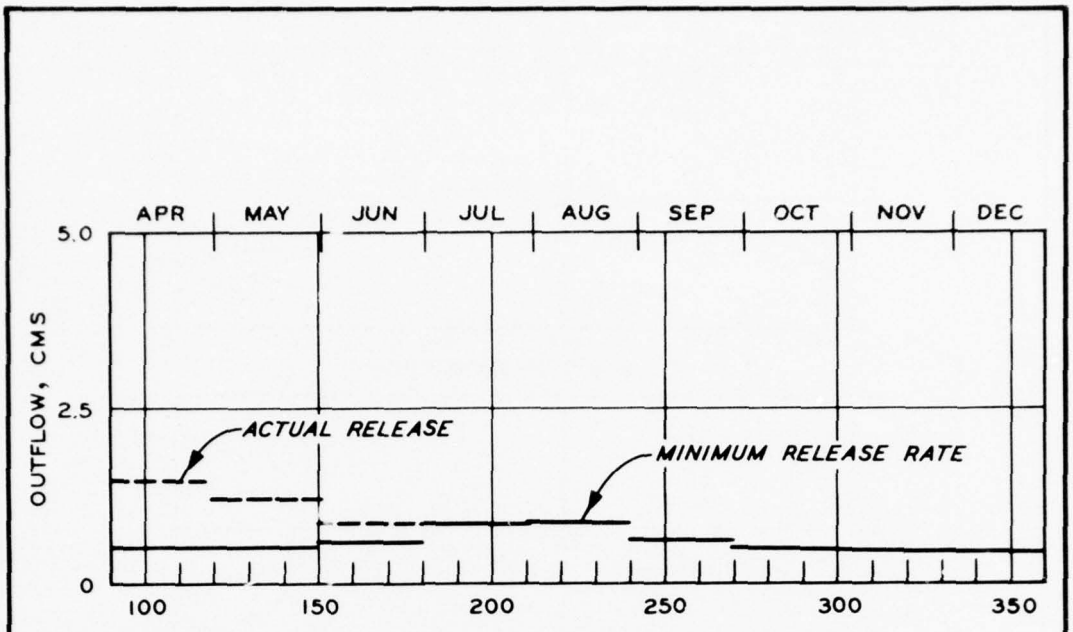
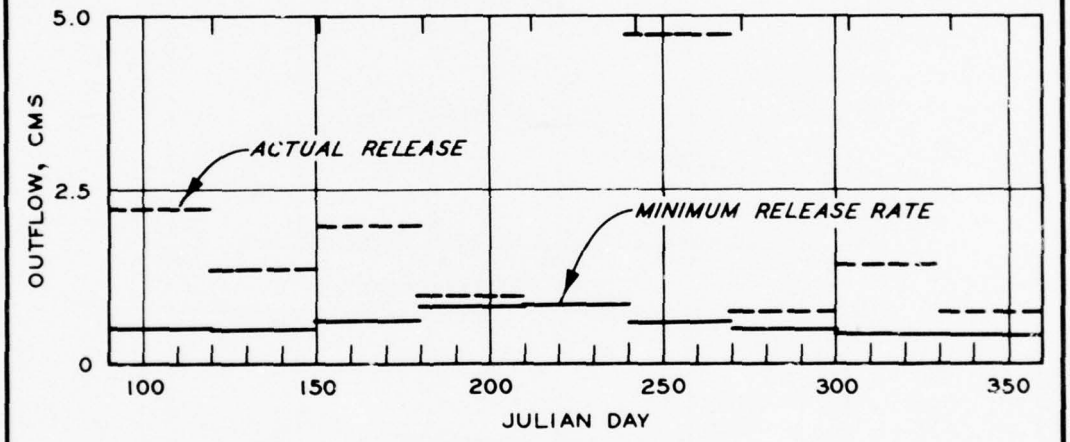

APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
15
10
5
0
ELEVATION, M
10°C
20
24
26
28
24
26
24
20
10°C
5
100
150
200
250
300
350
a. TEMPERATURE
15
10
5
0
ELEVATION, M
5
9
6
1 MG/L
0
100
150
200
250
300
350
JULIAN DAY
b. DISSOLVED OXYGEN
NOTE: DATA TAKEN FROM REFERENCE 50.
LAKE CARL BLACKWELL 1970
TEMPERATURE AND D.O. ISOPLETHS

PLATE 12



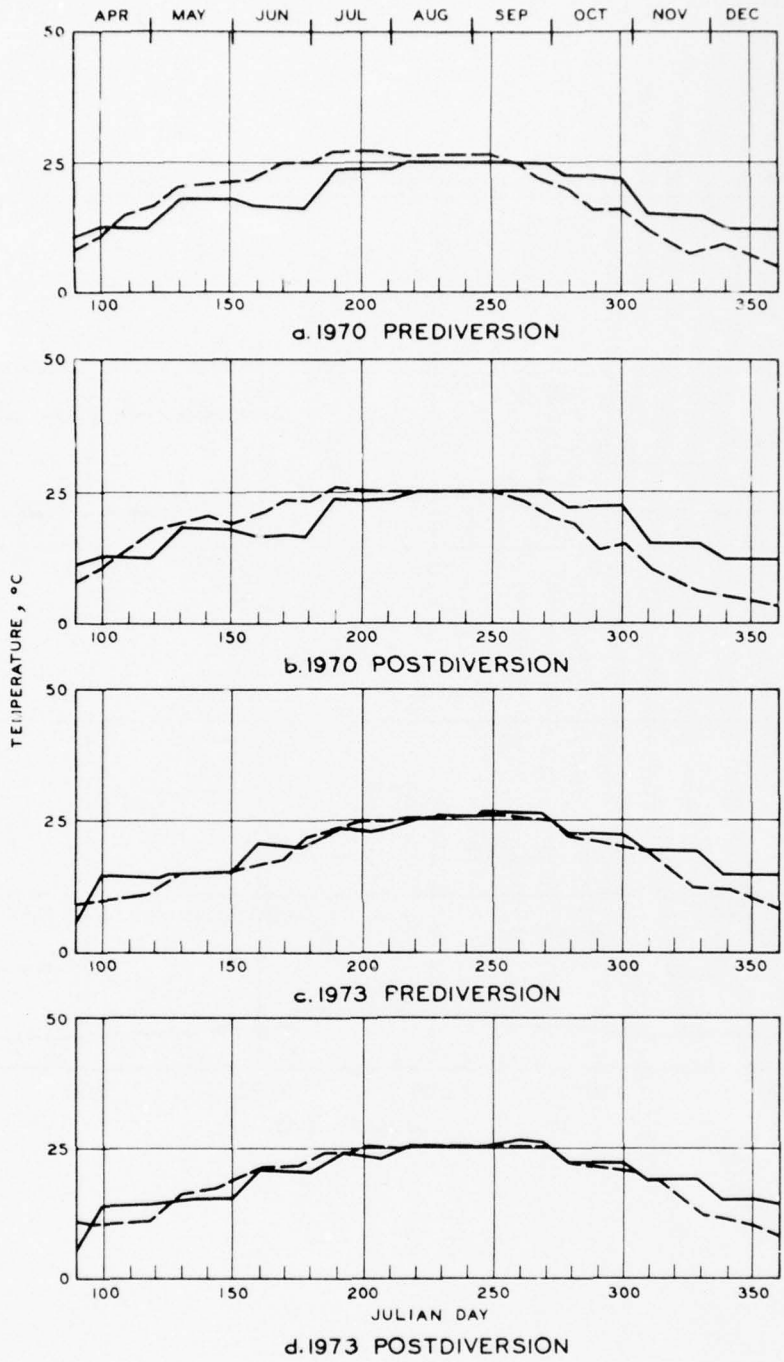
a. 1970



b. 1973

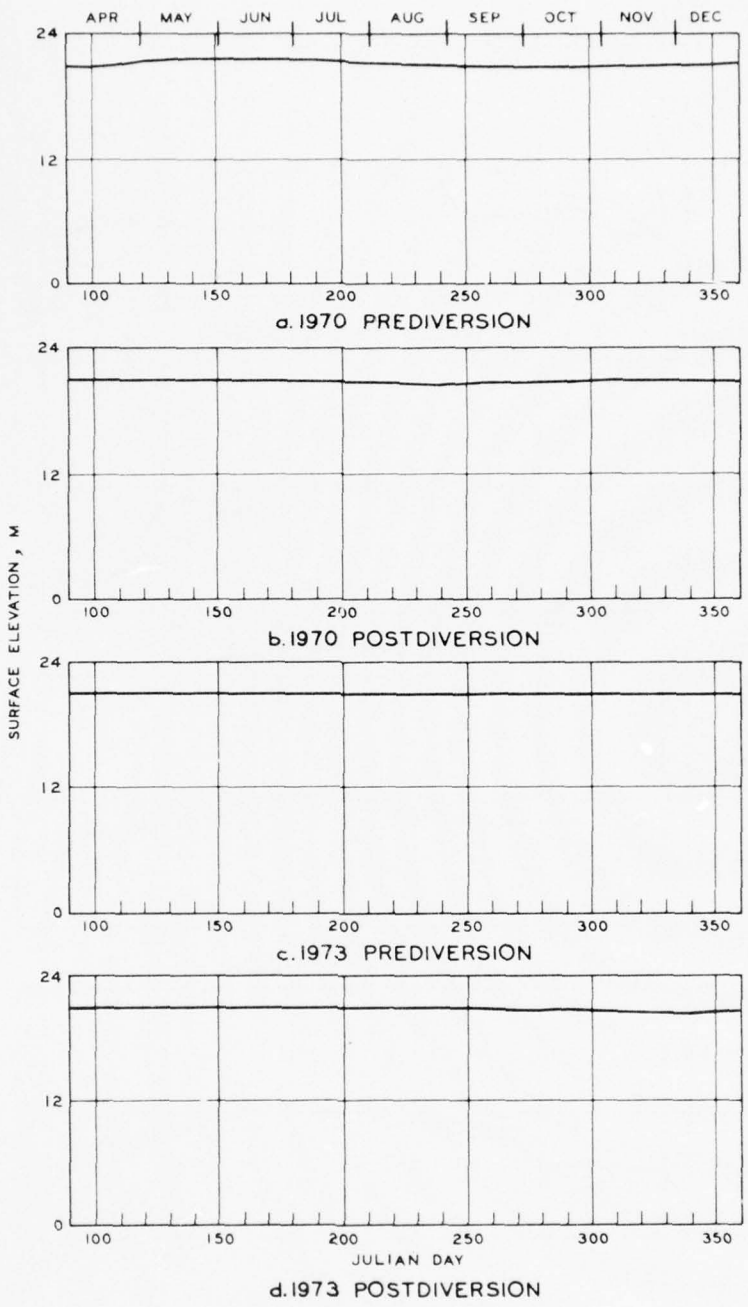
PROJECT RELEASES
POST DIVERSION

PLATE 13



INFLOW AND RELEASE TEMPERATURE

PLATE 14



SURFACE WATER ELEVATION

PLATE 15

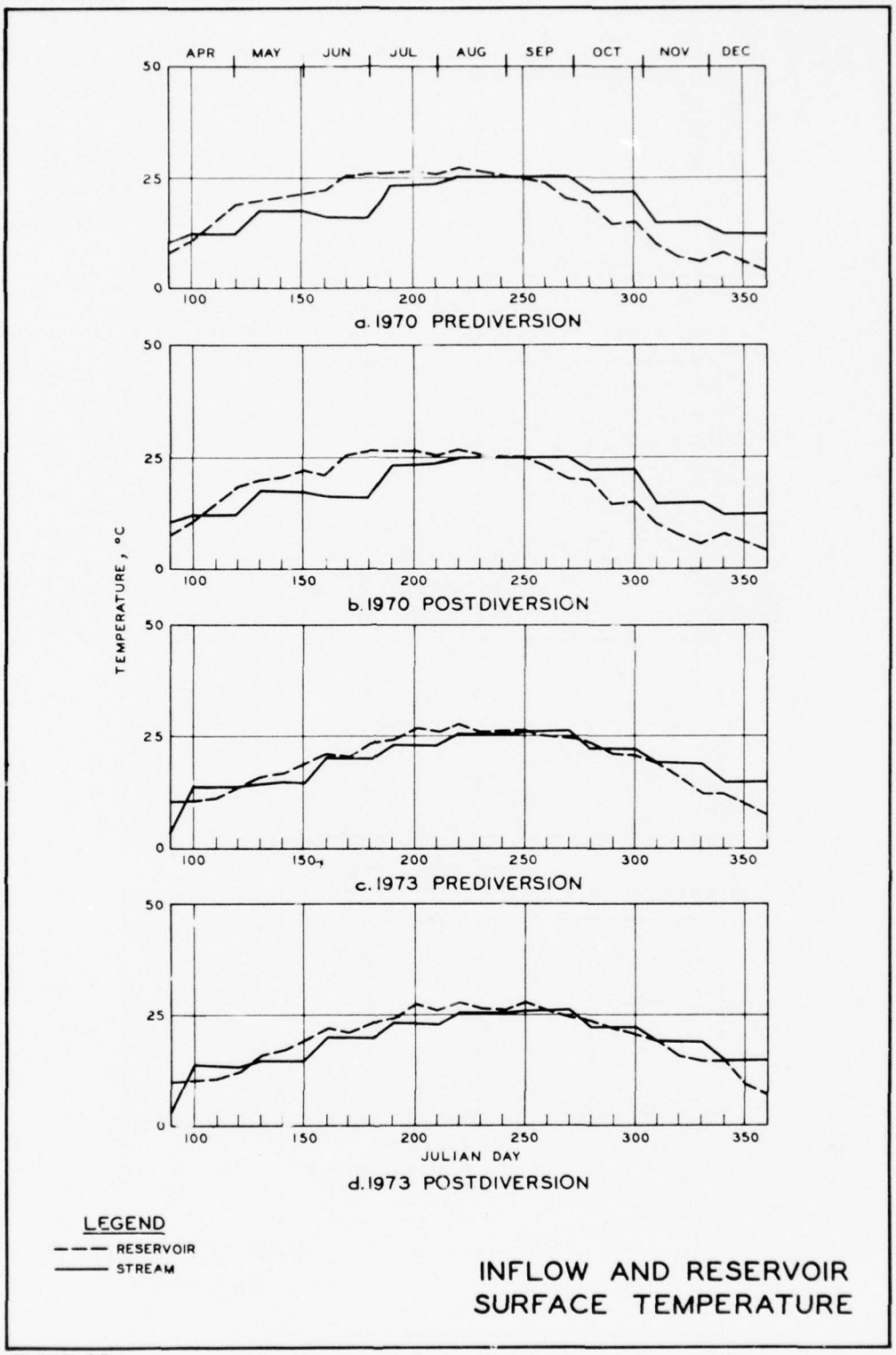
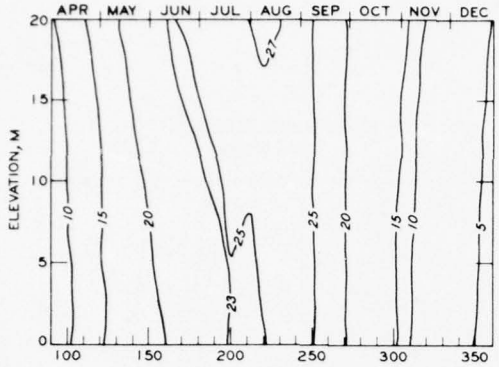
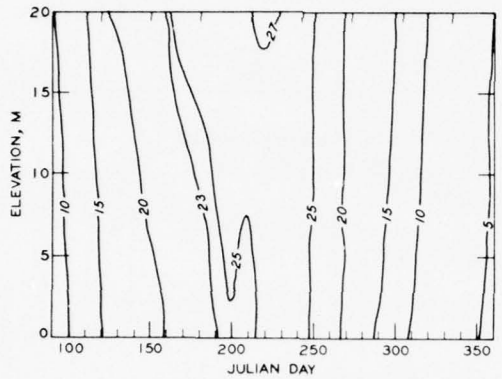

APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
50
25
0
100
150
200
250
300
350
a. 1970 PREDIVERSION
b. 1970 POSTDIVERSION
c. 1973 PREDIVERSION
d. 1973 POSTDIVERSION
TEMPERATURE, °C
JULIAN DAY
LEGEND
RESERVOIR
STREAM
INFLOW AND RESERVOIR
SURFACE TEMPERATURE

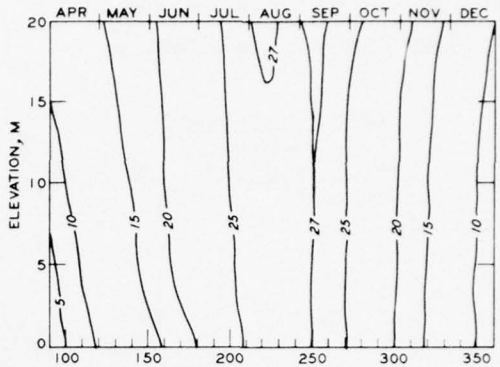
PLATE 16



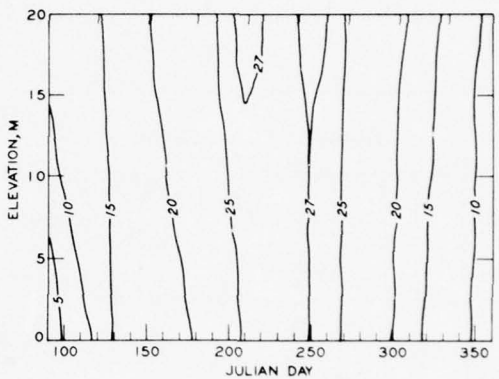
a. 1970 PREDIVERSION



b. 1970 POSTDIVERSION



c. 1973 PREDIVERSION



d. 1973 POSTDIVERSION

LEGEND

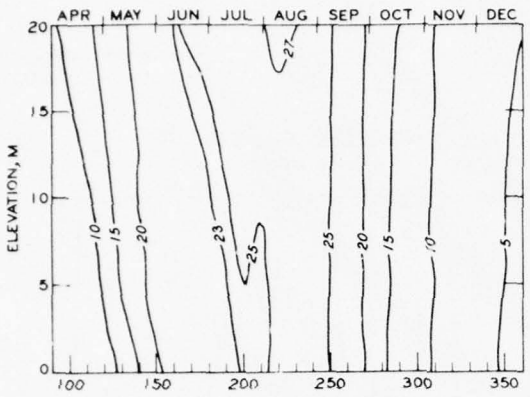
—10— TEMPERATURE, °C

RESERVIOR TEMPERATURE ISOPLETHS

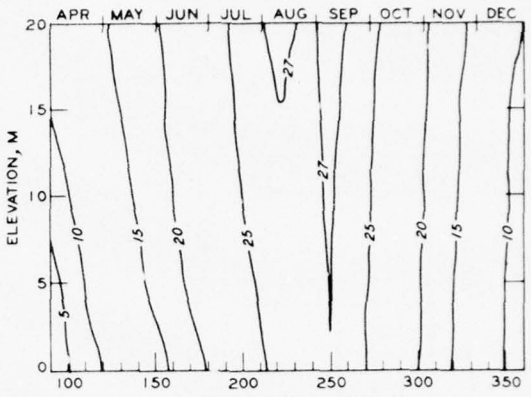
ANNUAL MEAN LIGHT PENETRATION= 2 M
(BASE CASE)

PLATE 17

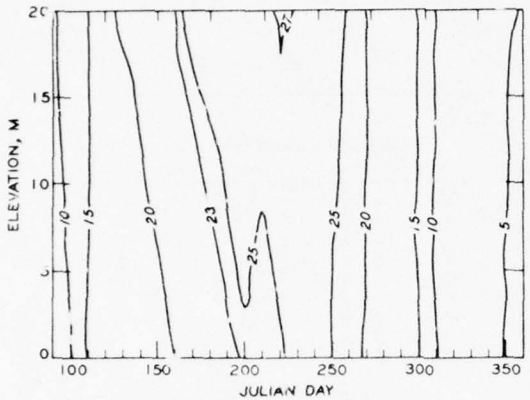
PLATE 18



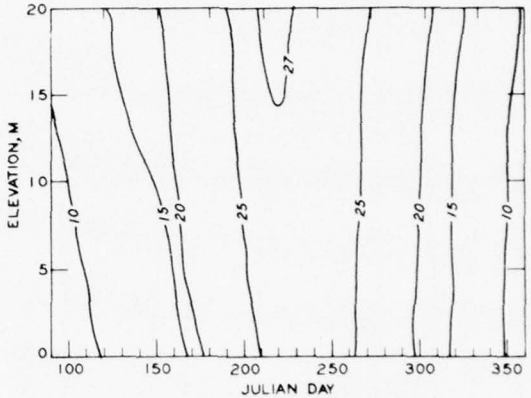
a. 1970 PREDIVERSION



c. 1973 PREDIVERSION



b. 1970 POSTDIVERSION



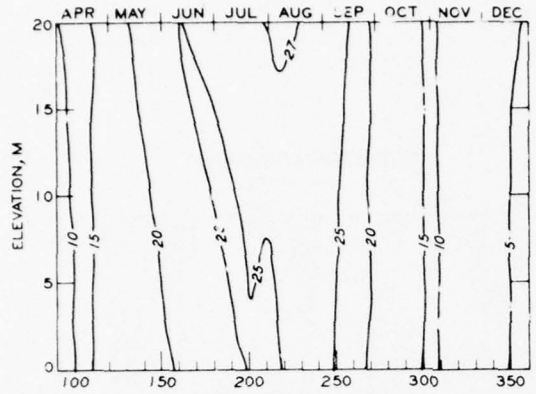
d. 1973 POSTDIVERSION

LEGEND

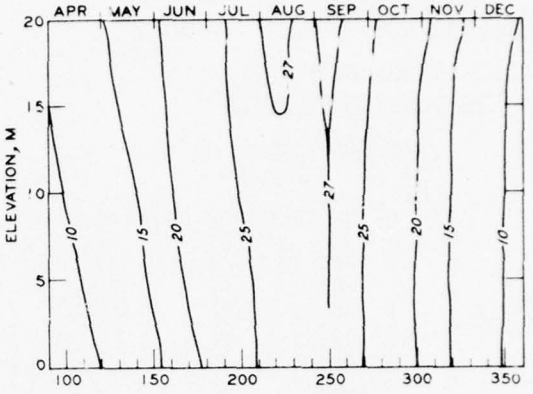
—10— TEMPERATURE, °C

RESERVIOR TEMPERATURE ISOPLETHS

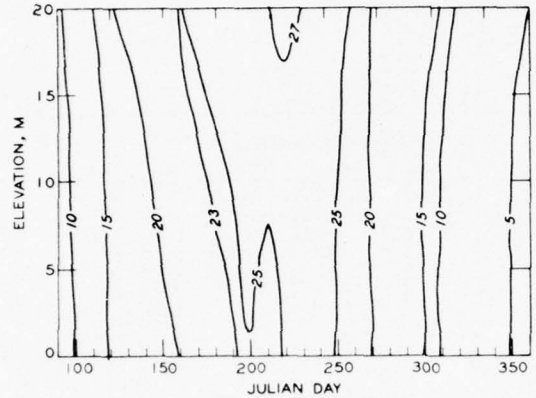
ANNUAL MEAN LIGHT PENETRATION= 1 M



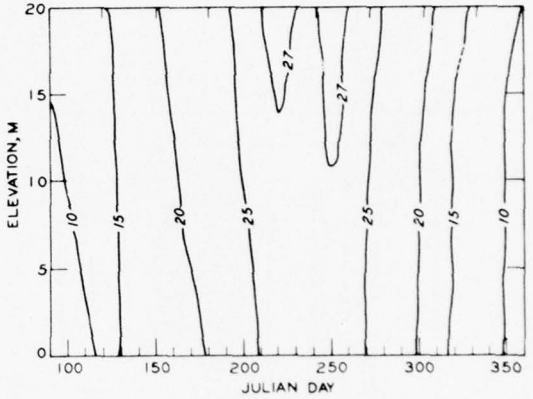
a. 1970 PREDIVERSION



c. 1973 PREDIVERSION



b. 1970 POSTDIVERSION



d. 1973 POSTDIVERSION

LEGEND

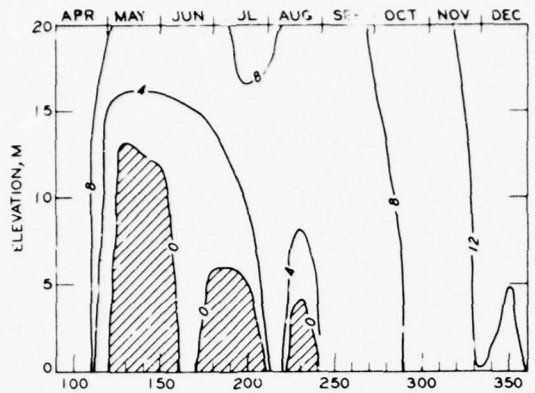
—10— TEMPERATURE, °C

RESERVIOR TEMPERATURE ISOPLETHS

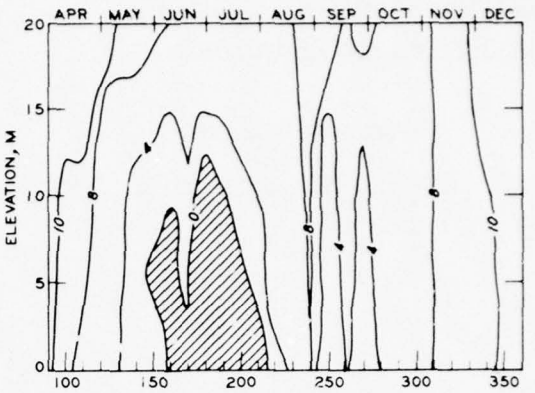
ANNUAL MEAN LIGHT PENETRATION= 3 M

PLATE 19

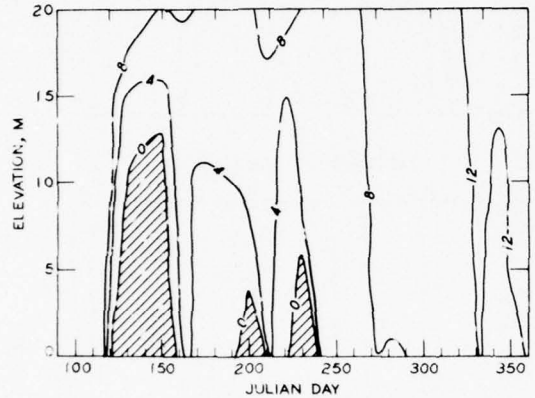
PLATE 20



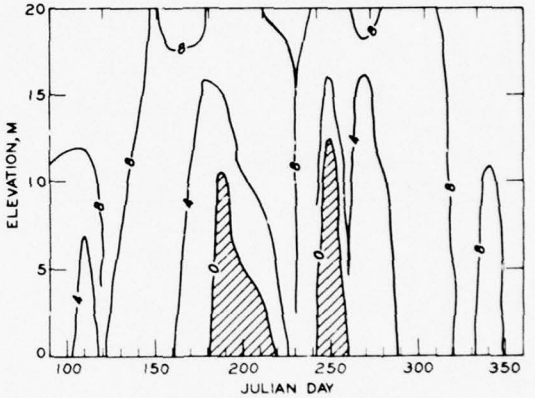
c. 1970 PREDIVERSION



c. 1973 PREDIVERSION



b. 1970 POSTDIVERSION



d. 1973 POSTDIVERSION

LEGEND

—4— D.O., MG/ℓ

OXYGEN DEPLETION

RESERVOIR DISSOLVED OXYGEN ISOPLETHS

ANNUAL MEAN LIGHT PENETRATION= 2 M
(BASE CASE)

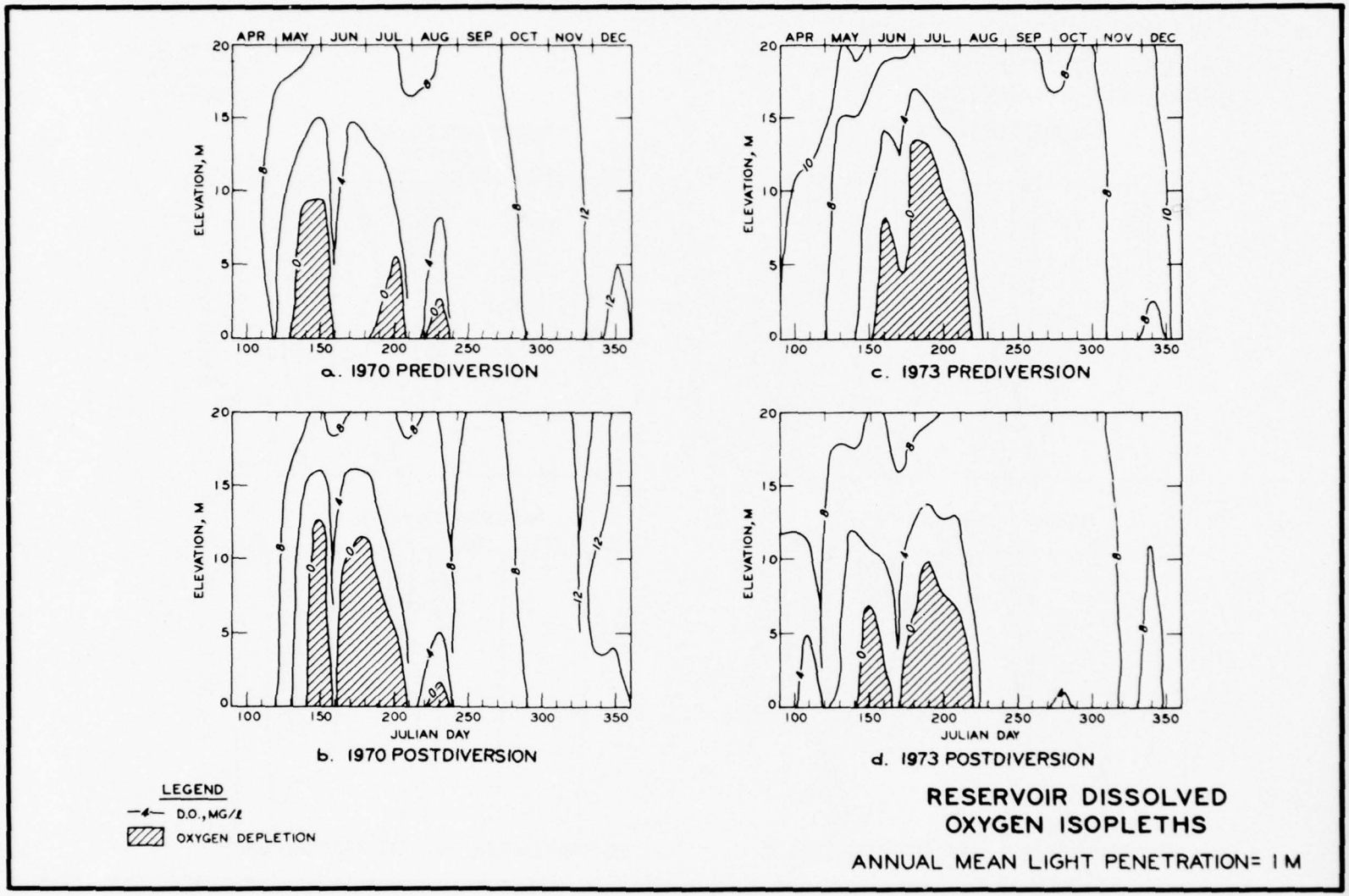
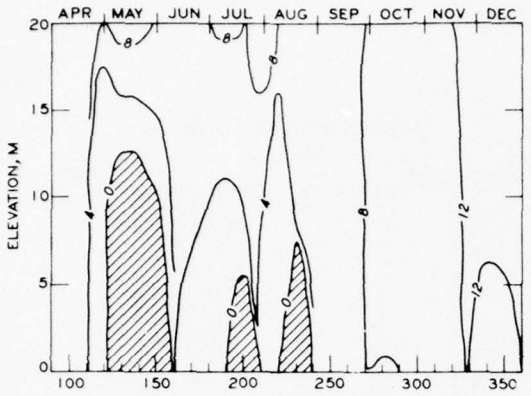
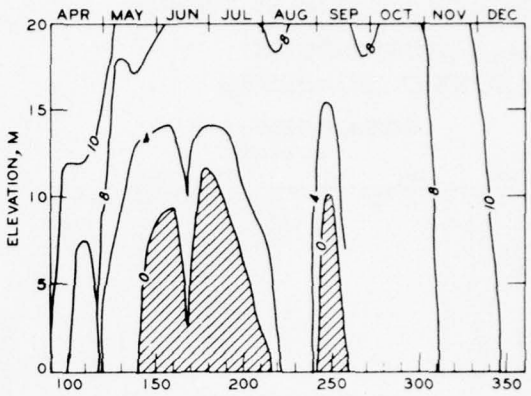

APR MAY JUN JUL AUG SEP OCT NOV DEC
ELEVATION, M
JULIAN DAY
a. 1970 PREDIVERSION
b. 1970 POSTDIVERSION
c. 1973 PREDIVERSION
d. 1973 POSTDIVERSION
LEGEND
D.O., MG/ℓ
OXYGEN DEPLETION
RESERVOIR DISSOLVED OXYGEN ISOPLETHS
ANNUAL MEAN LIGHT PENETRATION= 1 M

PLATE 21

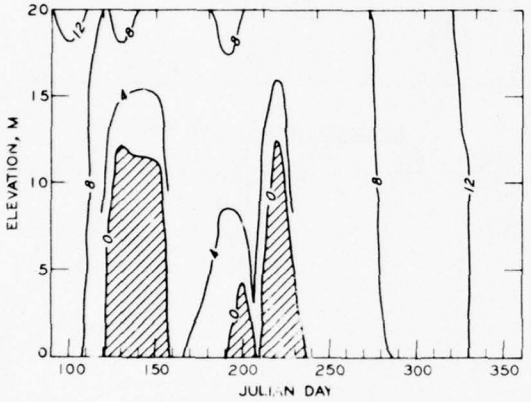
PLATE 22



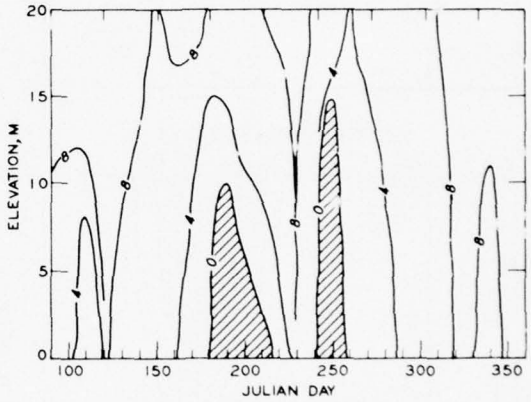
a. 1970 PREDIVERSION



c. 1973 PREDIVERSION



b. 1970 POST DIVERSION



d. 1973 POSTDIVERSION

LEGEND

D.O., MG/ℓ

OXYGEN DEPLETION

RESERVOIR DISSOLVED OXYGEN ISOPLETHS

ANNUAL MEAN LIGHT PENETRATION= 3 M

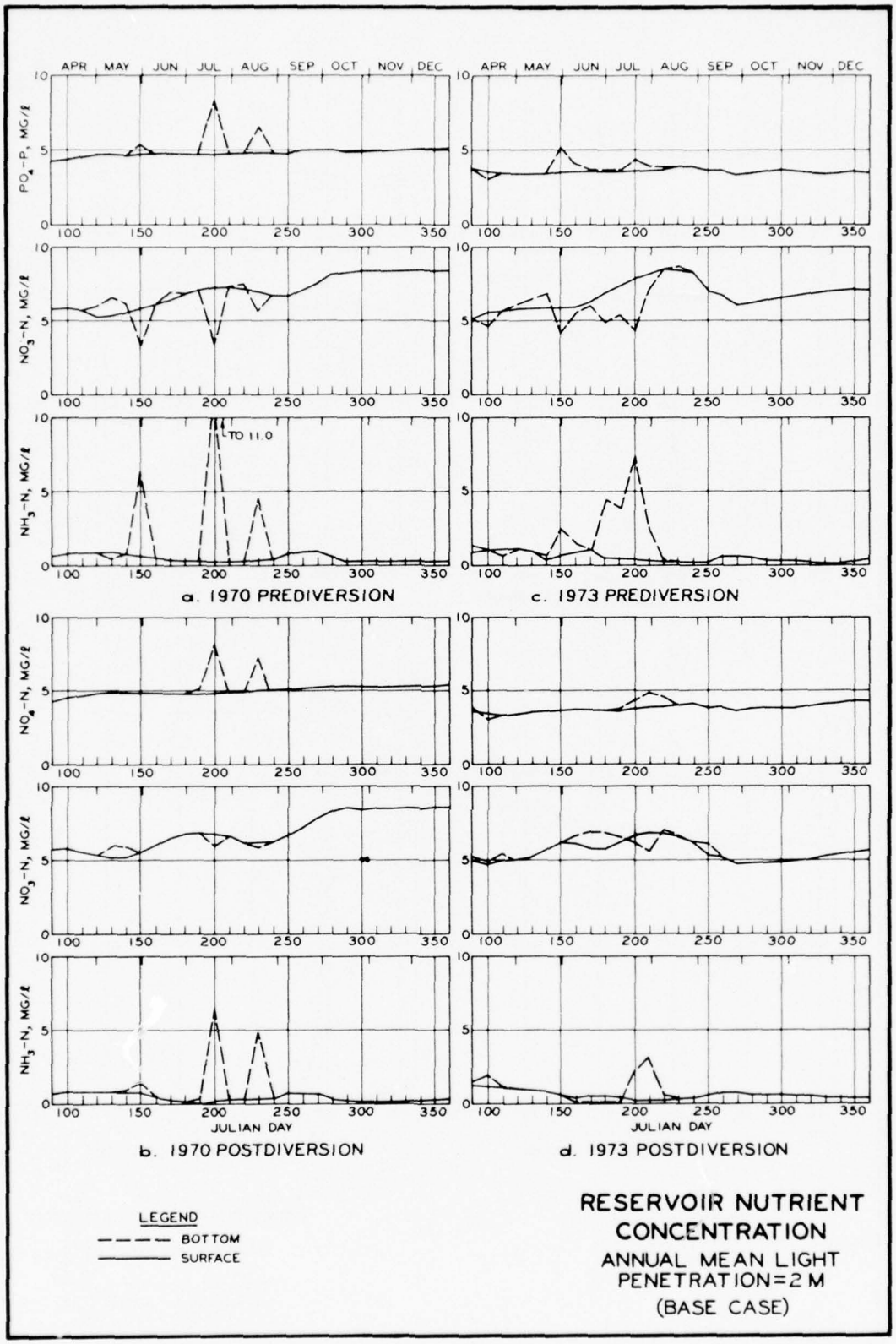

APR MAY JUN JUL AUG SEP OCT NOV DEC
PO_4-P, MG/ℓ
NO_3-N, MG/ℓ
NH_3-N, MG/ℓ
TO 11.0
JULIAN DAY
a. 1970 PREDIVERSION
c. 1973 PREDIVERSION
NO_3-N, MG/ℓ
NH_3-N, MG/ℓ
JULIAN DAY
b. 1970 POSTDIVERSION
d. 1973 POSTDIVERSION
LEGEND
BOTTOM
SURFACE
RESERVOIR NUTRIENT CONCENTRATION
ANNUAL MEAN LIGHT PENETRATION=2 M
(BASE CASE)

PLATE 23

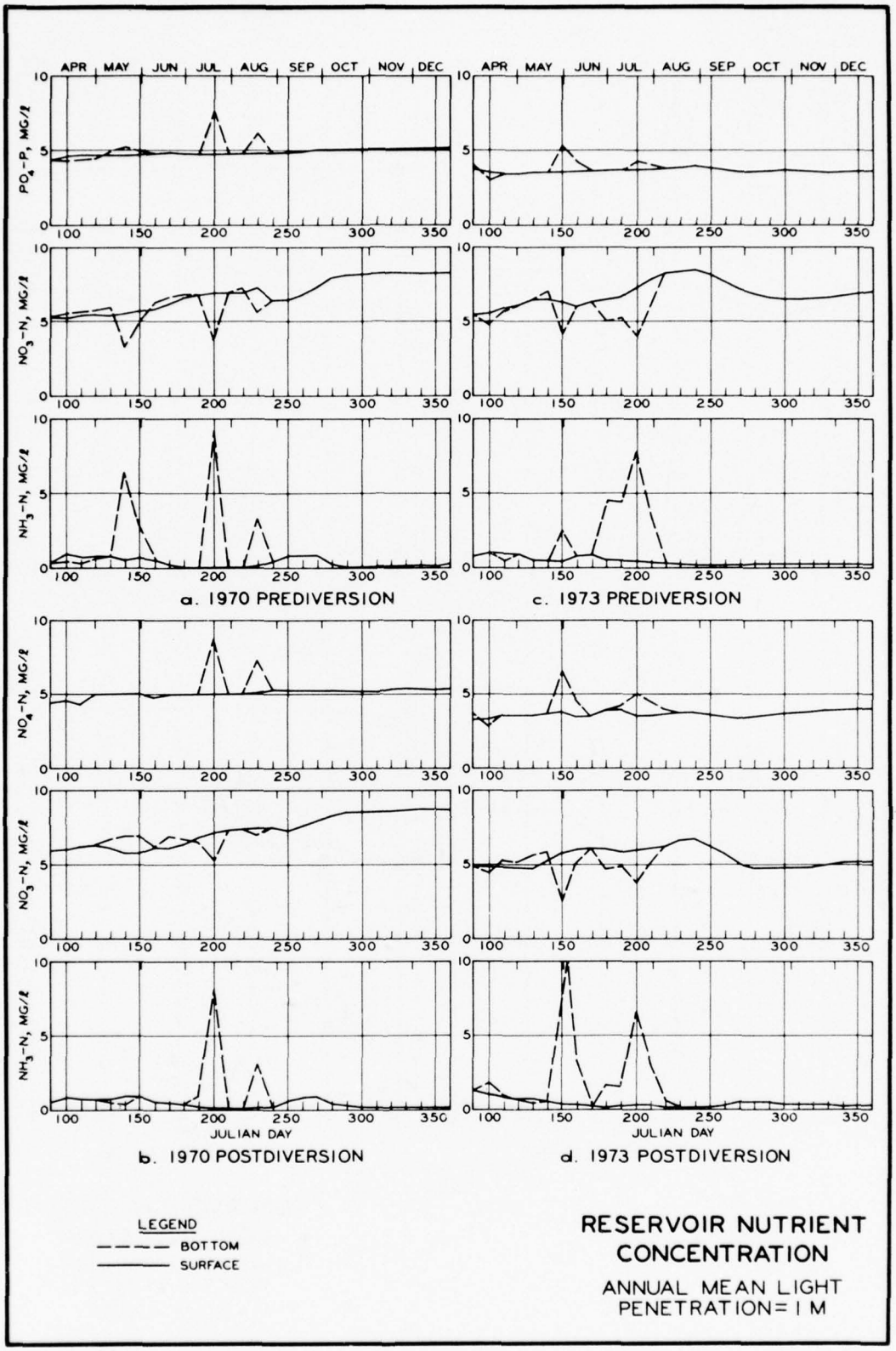

APR MAY JUN JUL AUG SEP OCT NOV DEC
PO4-P, MG/ℓ
NO3-N, MG/ℓ
NH3-N, MG/ℓ
0 5 10
100 150 200 250 300 350
a. 1970 PREDIVERSION
c. 1973 PREDIVERSION
NO4-N, MG/ℓ
NO3-N, MG/ℓ
NH3-N, MG/ℓ
JULIAN DAY
b. 1970 POSTDIVERSION
d. 1973 POSTDIVERSION
LEGEND
BOTTOM
SURFACE
RESERVOIR NUTRIENT CONCENTRATION
ANNUAL MEAN LIGHT PENETRATION = 1 M

PLATE 24

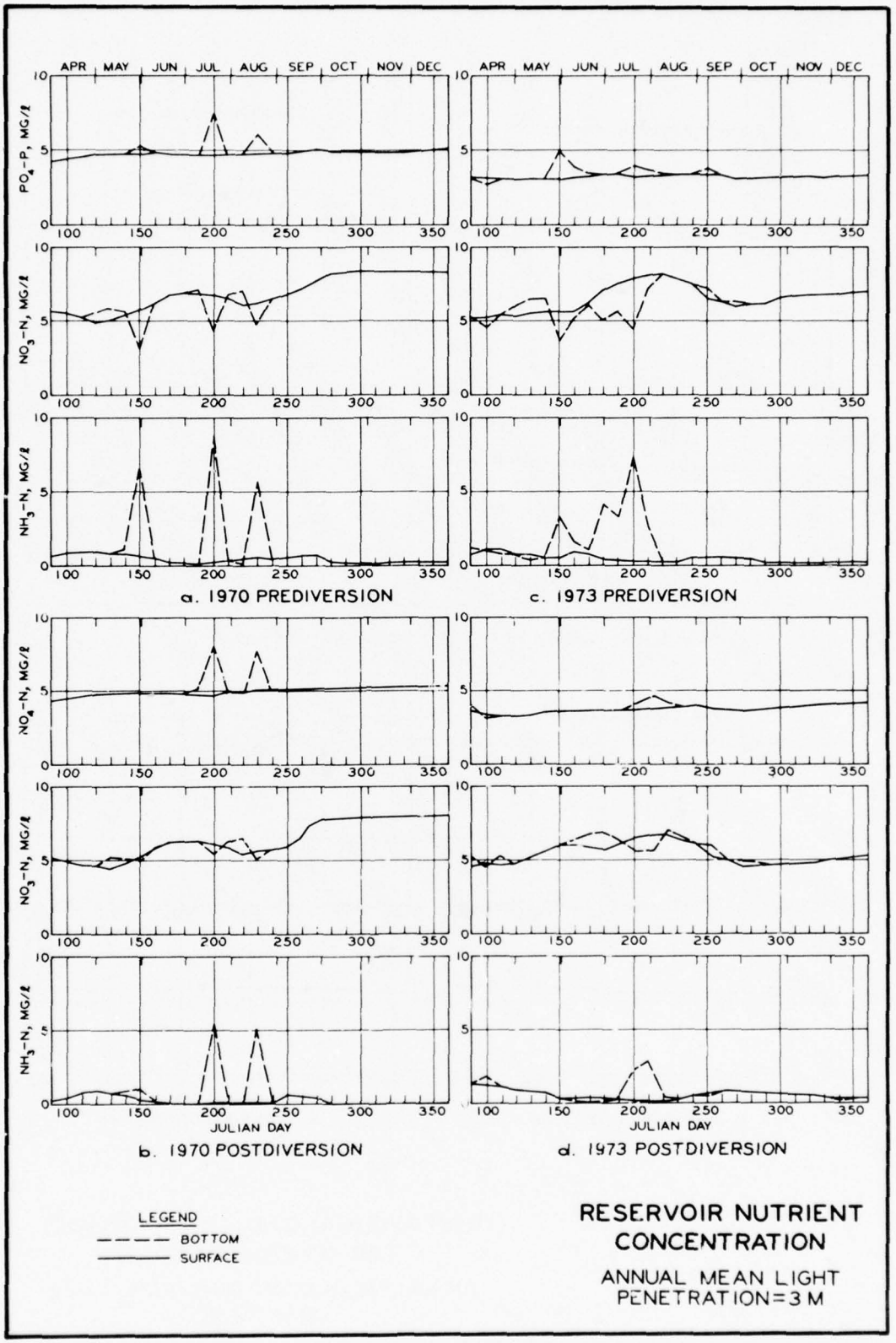
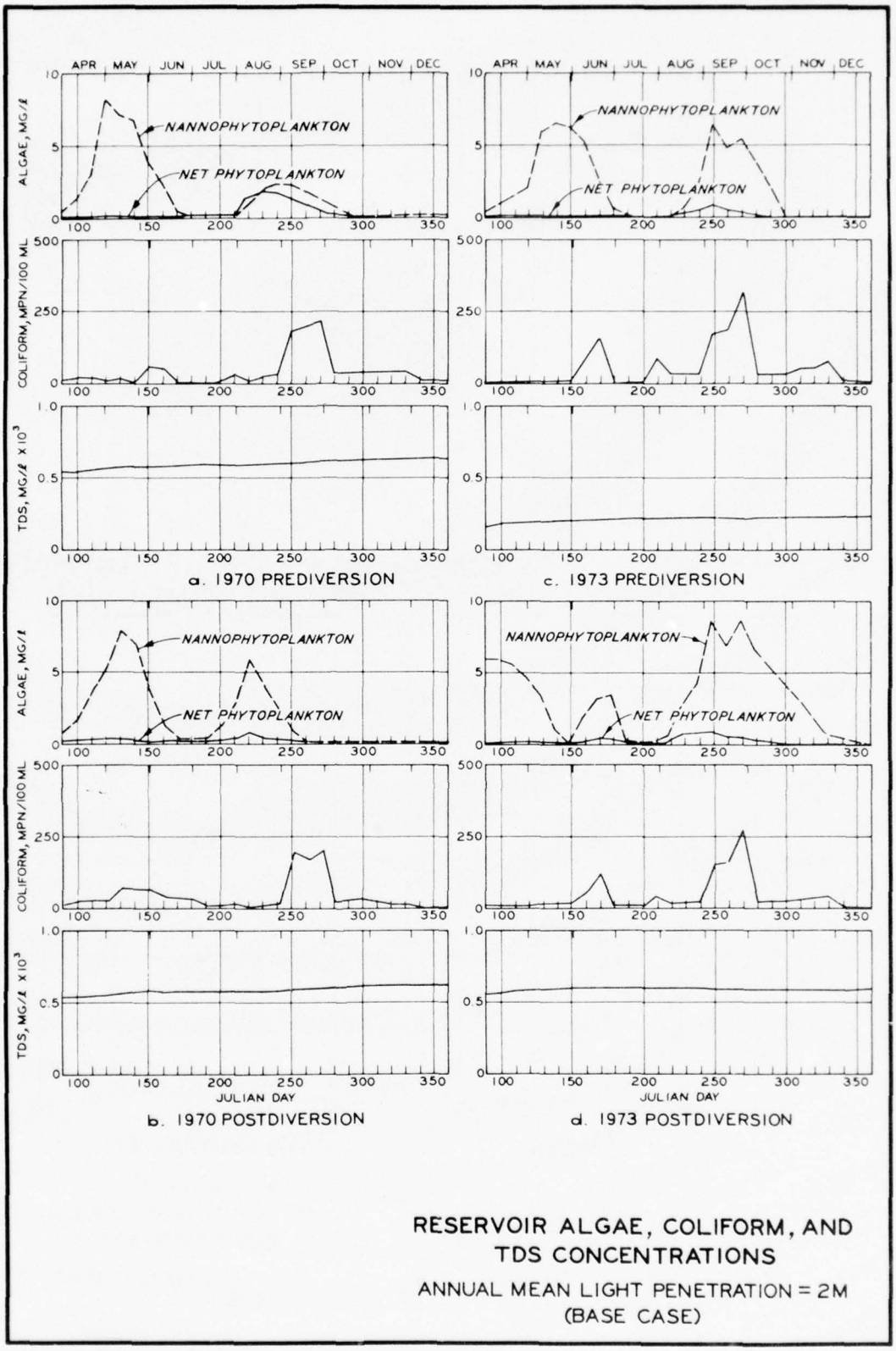

APR MAY JUN JUL AUG SEP OCT NOV DEC
PO_4-P, MG/ℓ
NO_3-N, MG/ℓ
NH_3-N, MG/ℓ
NO_4-N, MG/ℓ
JULIAN DAY
a. 1970 PREDIVERSION
c. 1973 PREDIVERSION
b. 1970 POSTDIVERSION
d. 1973 POSTDIVERSION
LEGEND
BOTTOM
SURFACE
RESERVOIR NUTRIENT CONCENTRATION
ANNUAL MEAN LIGHT PENETRATION=3 M

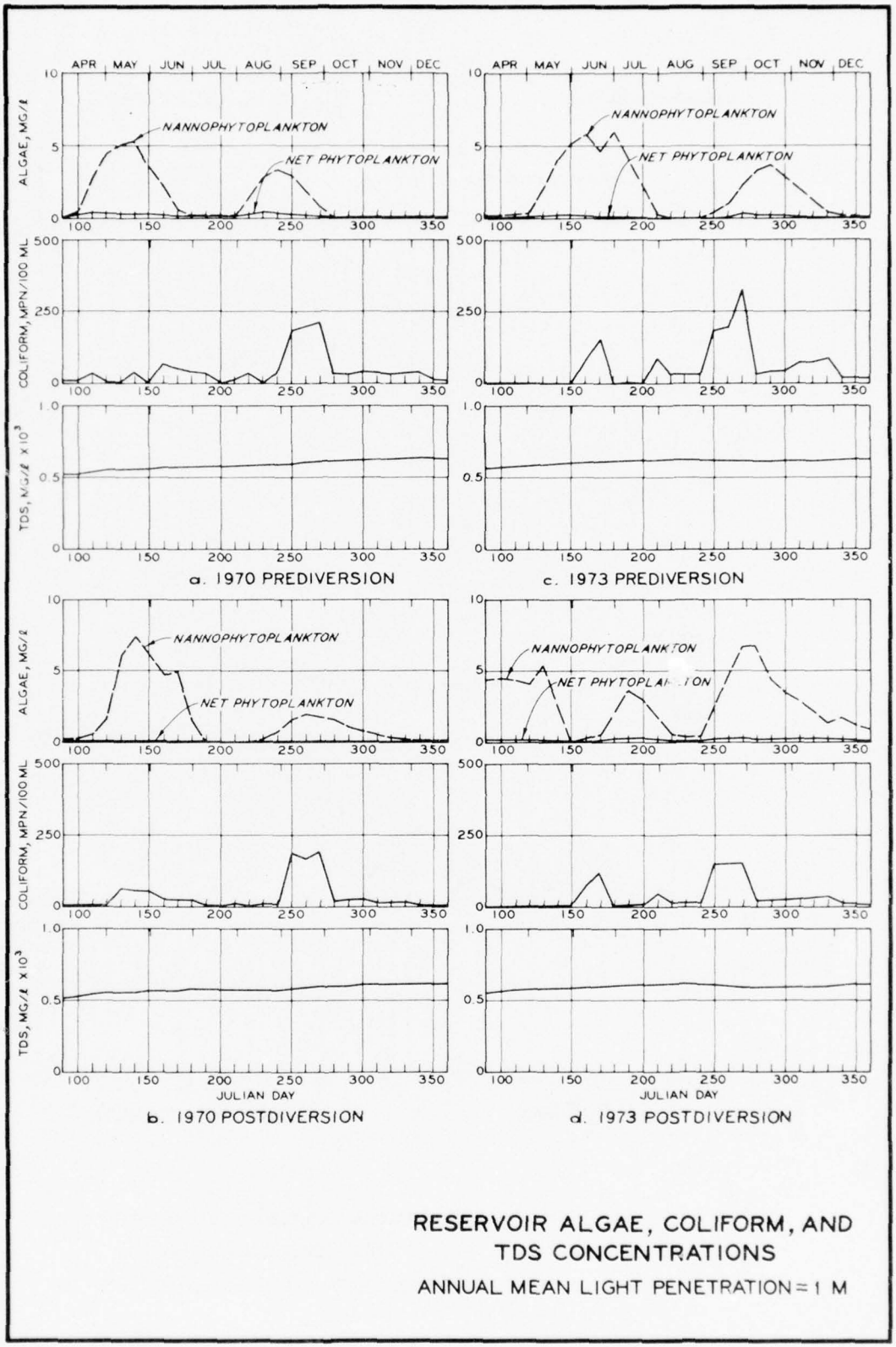
PLATE 25



RESERVOIR ALGAE, COLIFORM, AND TDS CONCENTRATIONS

ANNUAL MEAN LIGHT PENETRATION = 2M (BASE CASE)

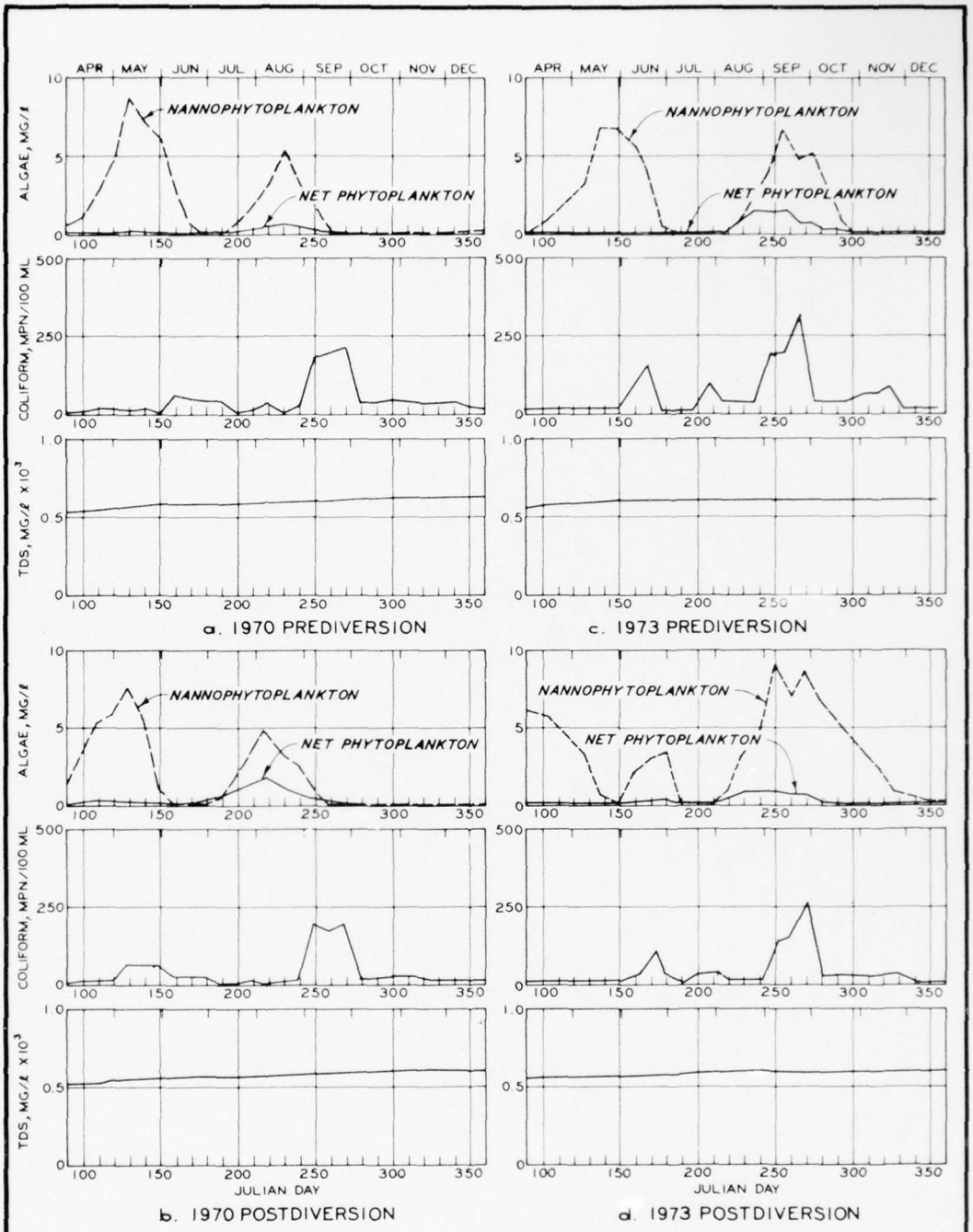
PLATE 26



RESERVOIR ALGAE, COLIFORM, AND TDS CONCENTRATIONS

ANNUAL MEAN LIGHT PENETRATION = 1 M

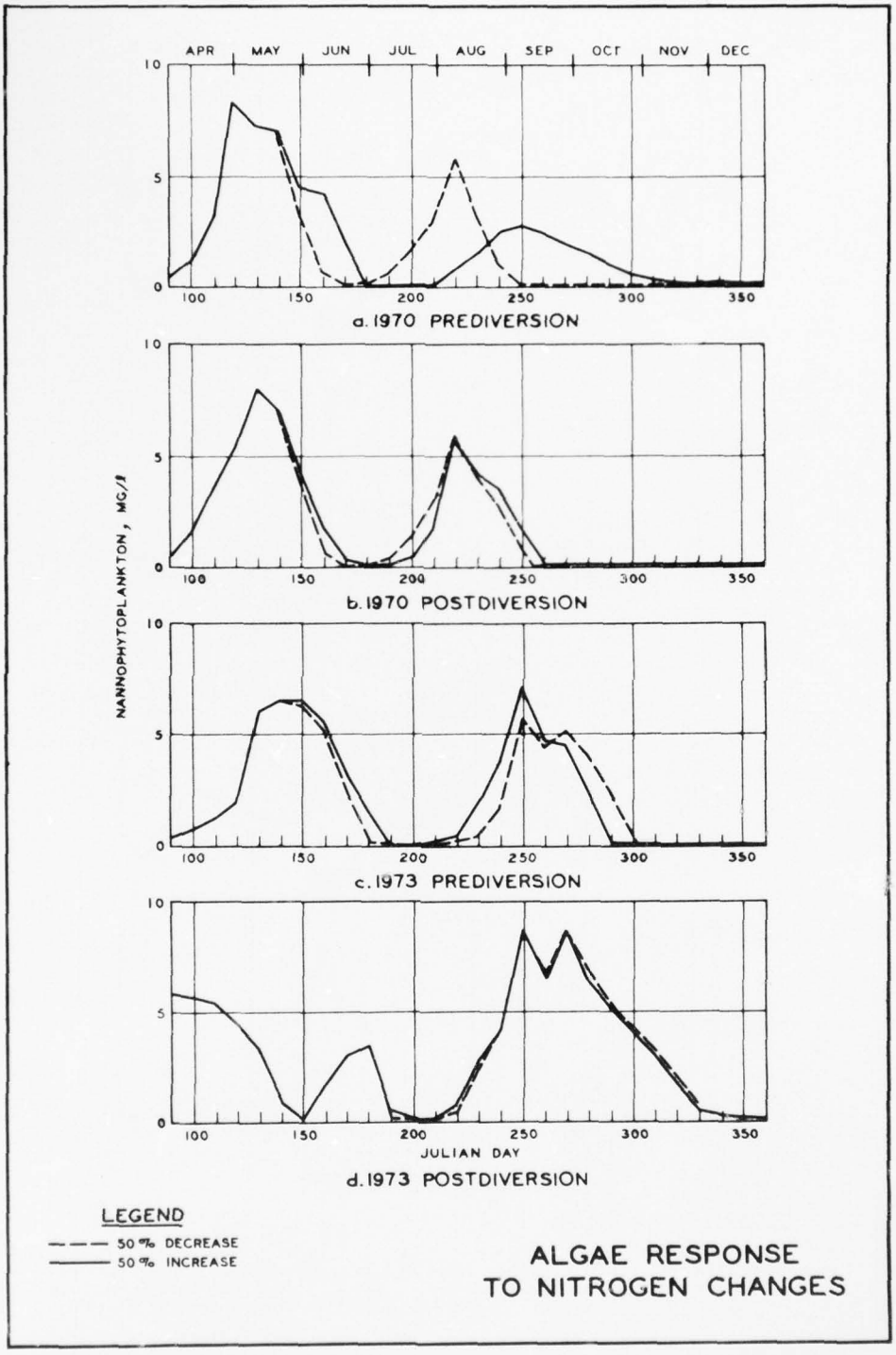
PLATE 27



RESERVOIR ALGAE, COLIFORM, AND TDS CONCENTRATIONS

ANNUAL MEAN LIGHT PENETRATION = 3 M

PLATE 28



ALGAE RESPONSE
TO NITROGEN CHANGES

PLATE 29

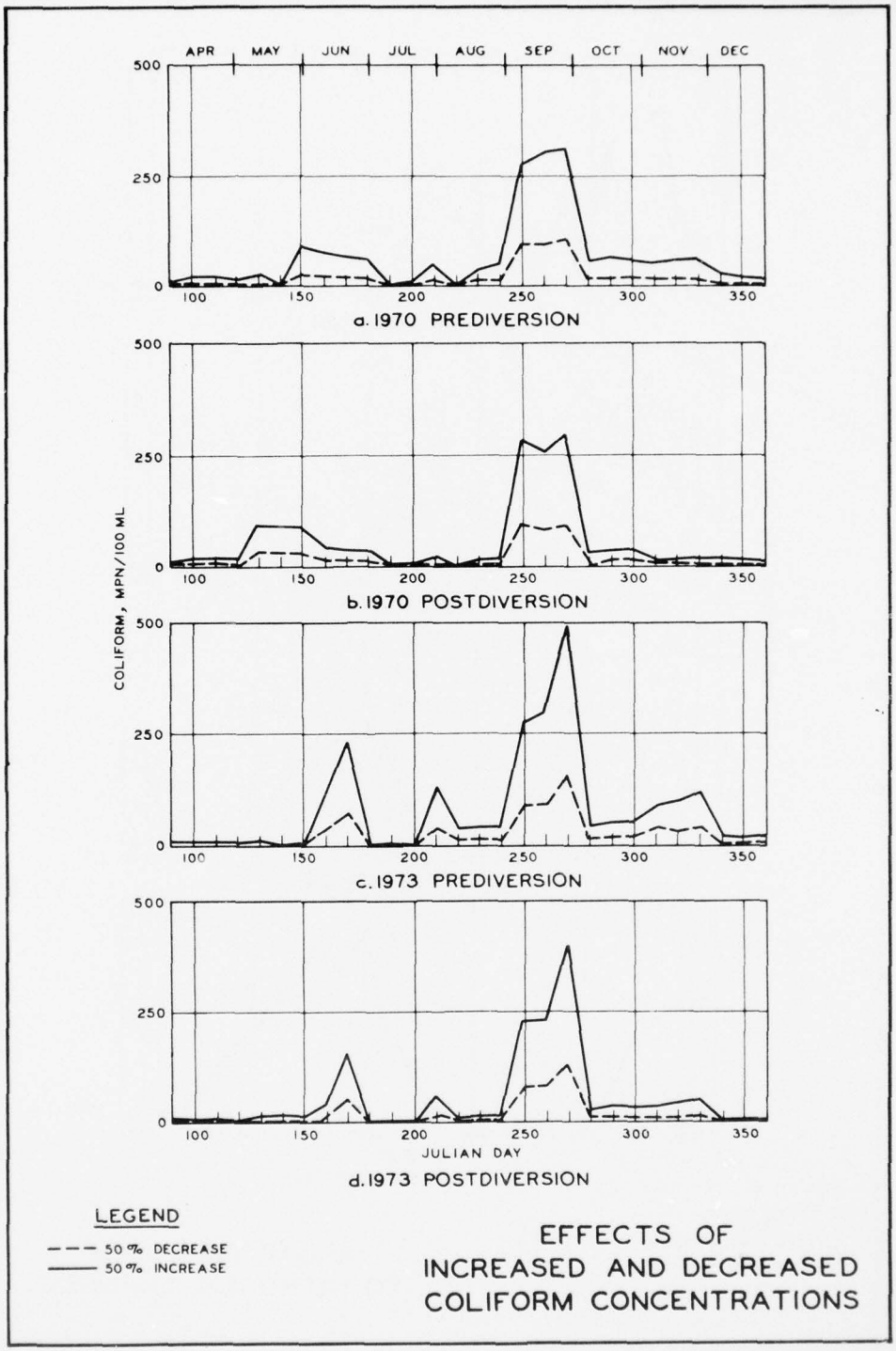
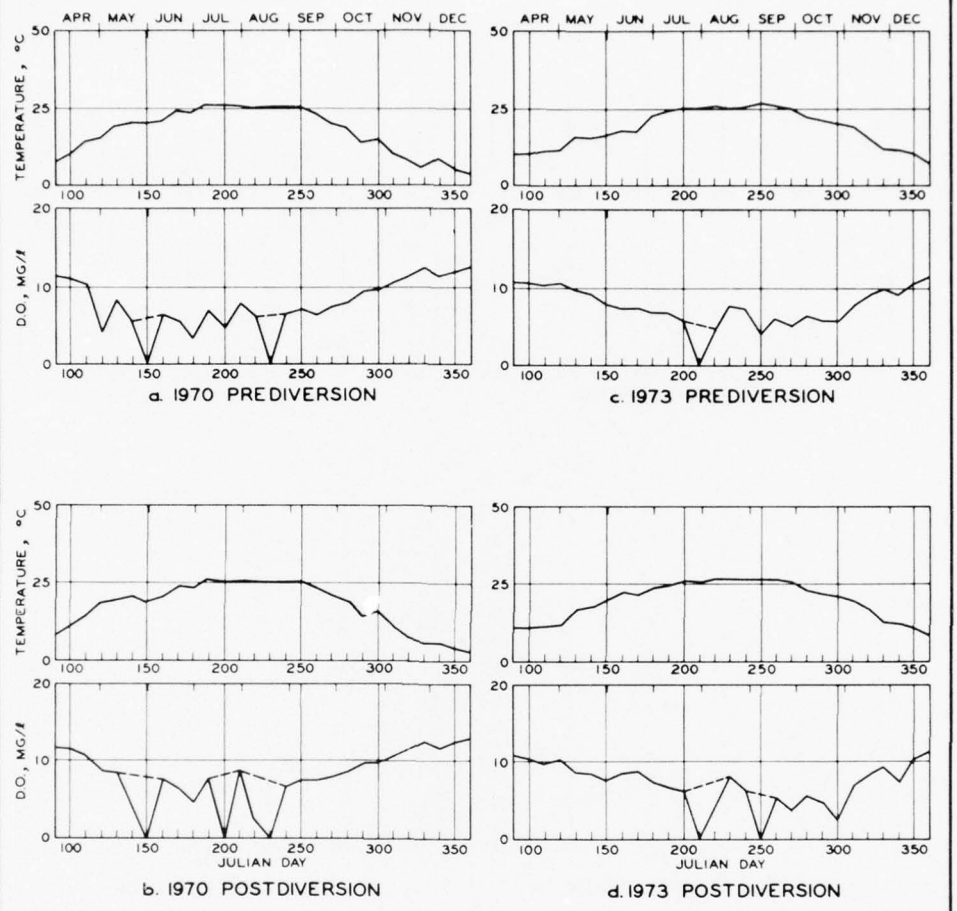

APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
500
250
0
100
150
200
250
300
350
a. 1970 PREDIVERSION
b. 1970 POSTDIVERSION
COLIFORM, MPN/100 ML
c. 1973 PREDIVERSION
JULIAN DAY
d. 1973 POSTDIVERSION
LEGEND
50 % DECREASE
50 % INCREASE
EFFECTS OF
INCREASED AND DECREASED
COLIFORM CONCENTRATIONS

PLATE 30



LEGEND

---- ACTUAL OPERATION

——— SIMULATION

RELEASE TEMPERATURES AND DISSOLVED OXYGEN CONCENTRATIONS IN OUTFLOW

ANNUAL MEAN LIGHT PENETRATION=2M (BASE CASE)

PLATE 31

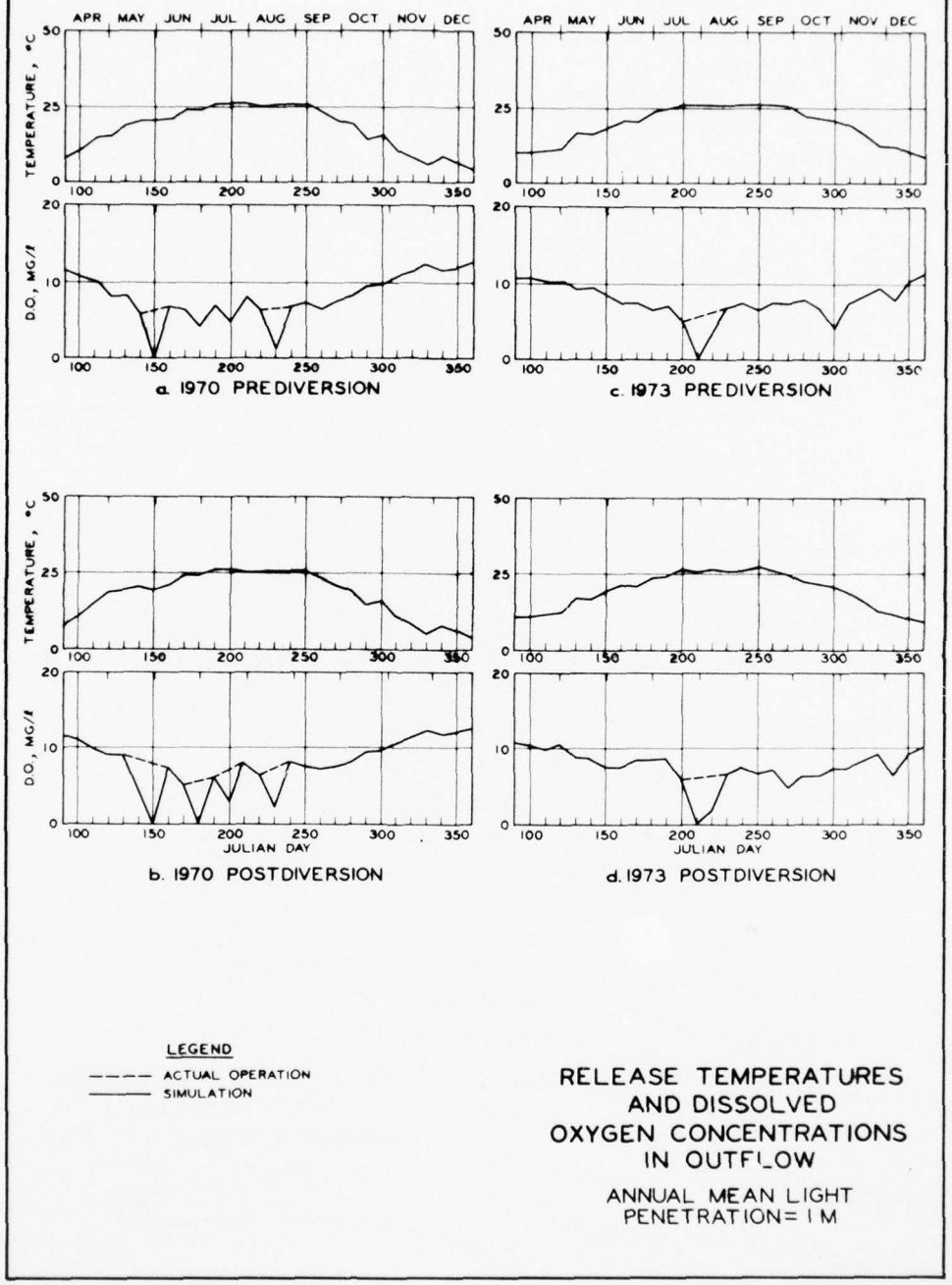
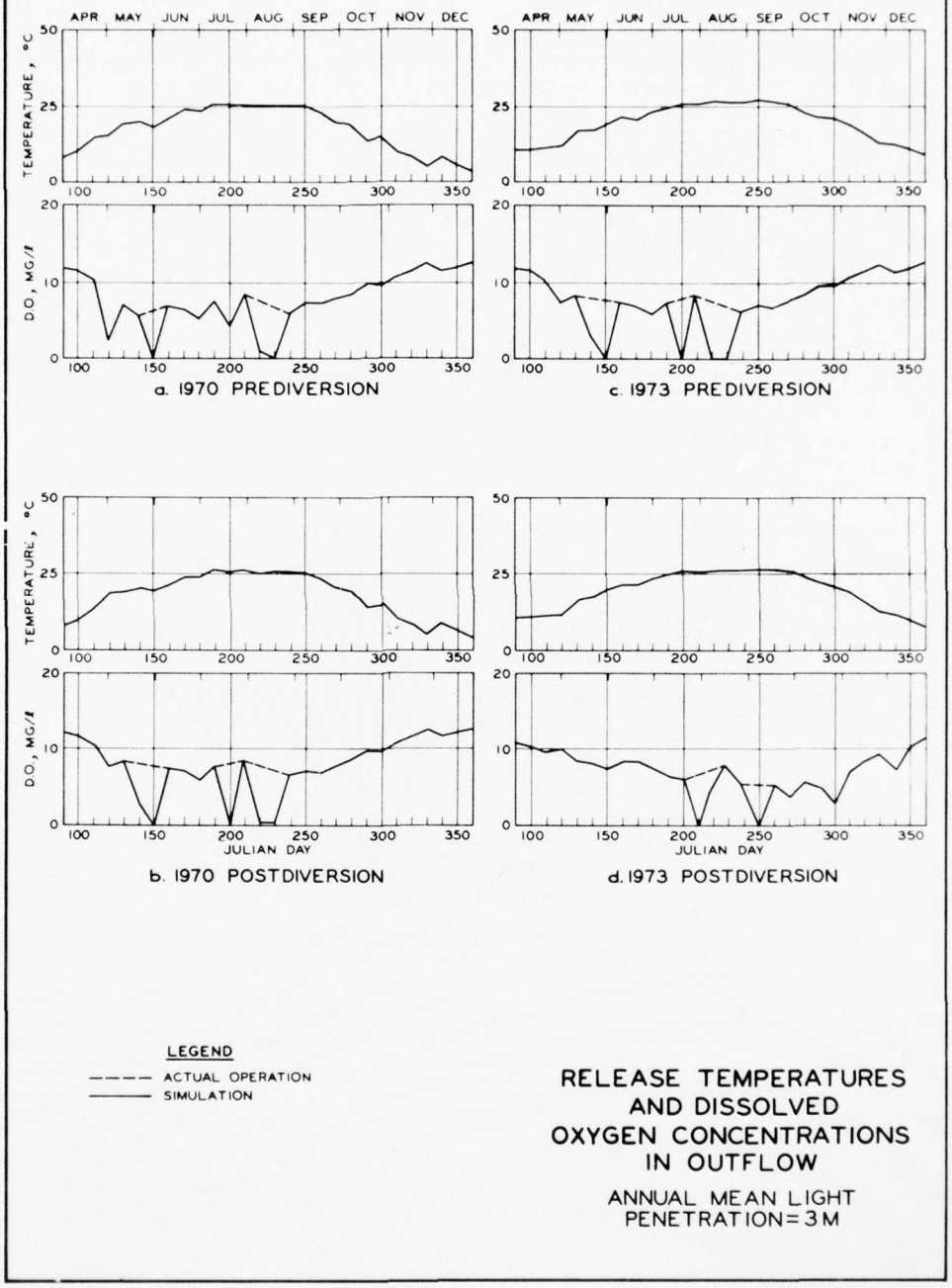

APR MAY JUN JUL AUG SEP OCT NOV DEC
TEMPERATURE, °C
D.O., MG/L
JULIAN DAY
a. 1970 PREDIVERSION
c. 1973 PREDIVERSION
b. 1970 POSTDIVERSION
d. 1973 POSTDIVERSION
LEGEND
ACTUAL OPERATION
SIMULATION
RELEASE TEMPERATURES AND DISSOLVED OXYGEN CONCENTRATIONS IN OUTFLOW
ANNUAL MEAN LIGHT PENETRATION= 1 M

PLATE 32



RELEASE TEMPERATURES AND DISSOLVED OXYGEN CONCENTRATIONS IN OUTFLOW

ANNUAL MEAN LIGHT PENETRATION = 3 M

PLATE 33

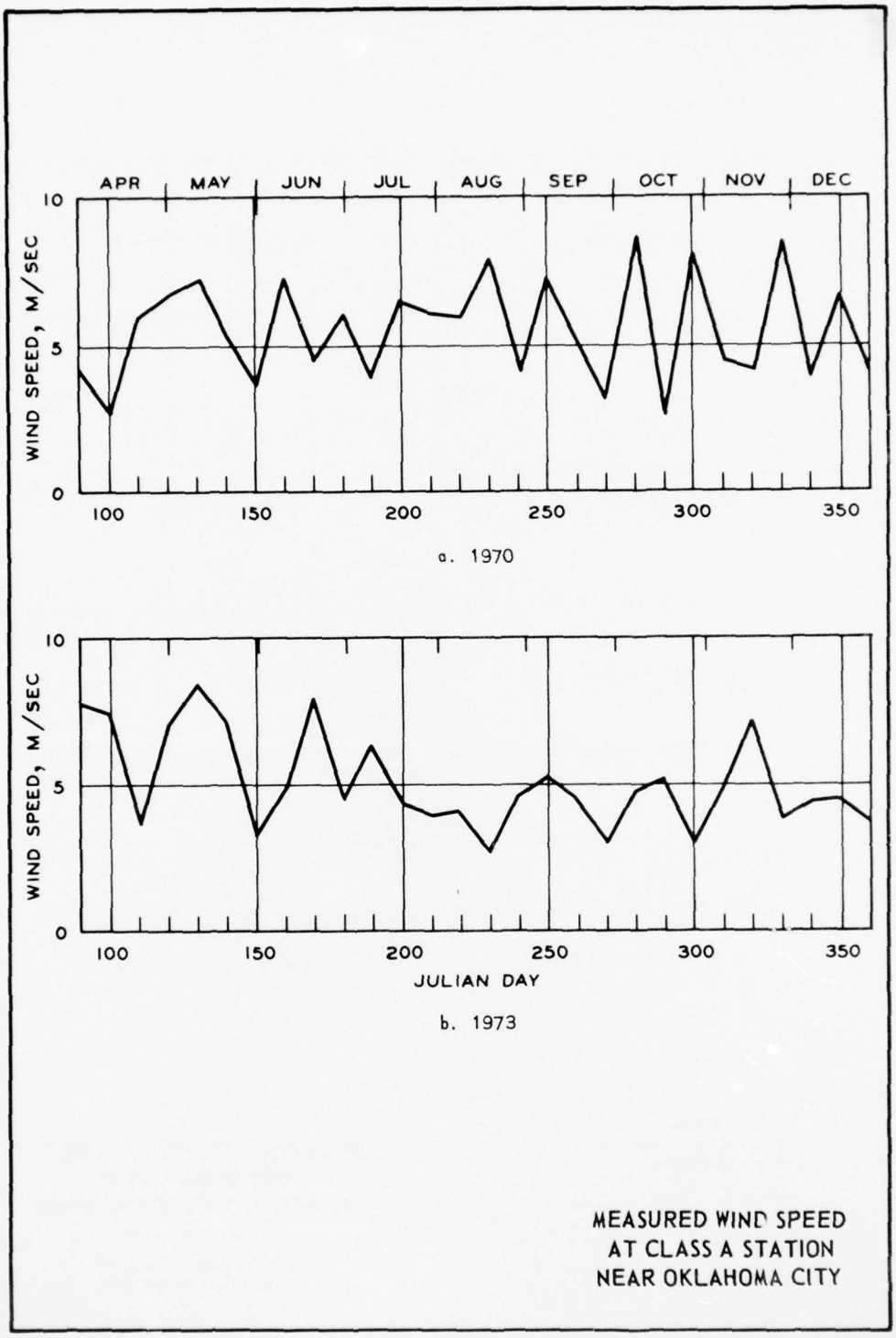

APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
WIND SPEED, M/SEC
10
5
0
100
150
200
250
300
350
a. 1970
WIND SPEED, M/SEC
10
5
0
100
150
200
250
300
350
JULIAN DAY
b. 1973
MEASURED WIND SPEED
AT CLASS A STATION
NEAR OKLAHOMA CITY

PLATE 34

APPENDIX A: COMPILATION OF WATER-QUALITY DATA - LAKES THUNDERBIRD AND EUFAULA SAMPLING PROGRAM

Nutrients

1. Sample values of nitrate, nitrite, ammonia, and total phosphorus are tabulated in Table A1.

Metals

2. Concentrations of metals measured in the water column and sediments are tabulated in Table A2.

Pesticides

3. Concentrations of the pesticides detected in the water column and sediments are tabulated in Table A3. The following pesticides were not detected in any sample:

Aldrin	Lindane
DDT	Toxaphene
Dieldrin	Malathion
Endrin	Methyl parathion
Heptachlor	Parathion
Heptachlor epoxide	Silvex

Other Water-Quality Parameters

4. Constituents not conveniently classified as nutrients, metals, or pesticides are presented in Table A4.

Table A1

Nutrients

Parameter	Location	Site 1, Water Column		Site 2, Water Column		Site 3, Water Column	
		Surface	Bottom	Surface	Bottom	Surface	Bottom
NO_3-N, mg/ℓ	Lake Eufaula	0.36	0.36	0.50	0.52	0.54	0.23
	Lake Thunderbird	0.22	0.25	0.27	0.23	0.27	--
NO_2-N, mg/ℓ	Lake Eufaula	0	0	0	0	0.01	0.01
	Lake Thunderbird	0	0	0	0	0	--
NH_4-N, mg/ℓ	Lake Eufaula	0.05	0.01	0.06	0.02	0.19	0.02
	Lake Thunderbird	0.05	0.08	0.03	0.08	0.07	--
Total-P, mg/ℓ	Lake Eufaula	0.06	0.05	0.07	0.27	0.12	0.16
	Lake Thunderbird	0.03	0.03	0.03	0.06	0.06	--

Note: Dash entry indicates that the parameter was not determined at the indicated elevation.

Table A2

Metals

Parameter	Location	Site 1 Water Column Surface µg/ℓ	Site 1 Water Column Bottom µg/ℓ	Site 1 Sediment µg/g	Site 2 Water Column Surface µg/ℓ	Site 2 Water Column Bottom µg/ℓ	Site 2 Sediment µg/g	Site 3 Water Column Surface µg/ℓ	Site 3 Water Column Bottom µg/ℓ	Site 3 Sediment µg/g
Cadmium, dissolved	Lake Eufaula	1	0	--	0	0	--	0	0	--
	Lake Thunderbird	0	0	--	0	0	--	0	--	--
Cadmium, total	Lake Eufaula	20	<10	2	<10	<10	<1	<10	0	1
	Lake Thunderbird	<10	<10	<1	20	<10	2	<10	--	<1
Chromium, dissolved	Lake Eufaula	0	0	--	0	0	--	0	0	--
	Lake Thunderbird	0	0	--	0	0	--	0	--	--
Chromium, total	Lake Eufaula	0	0	2	0	0	2	0	0	3
	Lake Thunderbird	0	0	3	0	0	7	0	--	3
Copper, dissolved	Lake Eufaula	14	34	--	6	10	--	8	10	--
	Lake Thunderbird	6	6	--	5	7	--	2	--	--
Copper, total	Lake Eufaula	20	50	9	20	10	1	40	50	7
	Lake Thunderbird	<10	10	5	<10	<10	14	<10	--	3
Iron, dissolved	Lake Eufaula	40	70	--	70	70	--	100	140	--
	Lake Thunderbird	30	20	--	20	20	--	20	--	--
Iron, total	Lake Eufaula	1400	1900	120,000	9000	5300	12,000	6700	7900	1,900
	Lake Thunderbird	270	670	43,000	300	440	150,000	820	--	31,000
Lead, dissolved	Lake Eufaula	2	3	--	1	2	--	2	2	--
	Lake Thunderbird	3	1	--	2	1	--	3	--	--
Lead, total	Lake Eufaula	<100	<97	<10	<100	<100	<10	<100	<100	<10
	Lake Thunderbird	<100	<100	<10	<100	<100	10	<100	--	<10
Manganese, dissolved	Lake Eufaula	30	20	--	20	10	--	0	0	--
	Lake Thunderbird	0	20	--	0	0	--	60	--	--
Manganese, total	Lake Eufaula	80	190	840	930	370	130	410	460	1,200
	Lake Thunderbird	20	110	230	40	20	1,100	160	--	640
Zinc, dissolved	Lake Eufaula	30	40	--	0	0	--	30	30	--
	Lake Thunderbird	10	10	--	30	0	--	0	--	--
Zinc, total	Lake Eufaula	40	110	110	100	60	35	120	160	50
	Lake Thunderbird	20	30	10	30	30	160	10	--	10

Note: Dash entry indicates that the parameter was not determined at the indicated elevation.

Table A3

Pesticides

Parameter	Location	Site 1 Water Column Surface µg/ℓ	Site 1 Water Column Bottom µg/ℓ	Site 1 Sediment µg/kg	Site 2 Water Column Surface µg/ℓ	Site 2 Water Column Bottom µg/ℓ	Site 2 Sediment µg/kg	Site 3 Water Column Surface µg/ℓ	Site 3 Water Column Bottom µg/ℓ	Site 3 Sediment µg/kg
DDD unfiltered	Lake Eufaula	0	0	0	0	--	0.7	0	--	0
	Lake Thunderbird	0	0	0	0	0	0	0	--	0
DDE, unfiltered	Lake Eufaula	0	0	0	0	--	0.9	0	--	0
	Lake Thunderbird	0	0	0.5	0	0	0.2	0	--	0
Chlordane, unfiltered	Lake Eufaula	0	0	0	0	--	0	0	--	0
	Lake Thunderbird	0	0	0	0	0	0	0	--	1.7
PCB, unfiltered	Lake Eufaula	0	0	0	0	--	0	0	--	0
	Lake Thunderbird	0	0	0	0	0	0	0	--	2.0
Diazinon, unfiltered	Lake Eufaula	0	0	--	0	--	--	0.01	--	--
	Lake Thunderbird	0	0	--	0	0	--	0	--	--
2,4-D, unfiltered	Lake Eufaula	0	0	--	0.14	--	--	0.01	--	--
	Lake Thunderbird	0	0	--	0	0	--	0	--	--
2,4,5-T	Lake Eufaula	0.01	0.01	--	0.01	--	--	0.01	--	--
	Lake Thunderbird	0	0	--	0	0	--	0	--	--

Note: Dash entry indicates that the parameter was not determined at the indicated elevation.

Table A4

Other Water-Quality Parameters

Parameter	Location	Site 1, Water Column		Site 2, Water Column		Site 3, Water Column	
		Surface	Bottom	Surface	Bottom	Surface	Bottom
Bicarbonate, mg/ℓ	Lake Eufaula	103	125	103	89	138	139
	Lake Thunderbird	206	208	209	209	359	--
Carbonate, mg/ℓ	Lake Eufaula	0	0	0	0	0	0
	Lake Thunderbird	0	0	0	0	0	--
Chlorides, mg/ℓ	Lake Eufaula	70	65	70	60	130	130
	Lake Thunderbird	27	27	27	27	30	--
Conductivity, MMHOS/cm	Lake Eufaula	400	441	400	400	625	630
	Lake Thunderbird	400	420	416	410	650	--
Phenols, µg/ℓ	Lake Eufaula	0	0	1	0	2	3
	Lake Thunderbird	1	2	2	0	2	--
Total dissolved solids, mg/ℓ	Lake Eufaula	245	240	251	234	393	264
	Lake Thunderbird	167	169	202	202	317	--
Sulfate, mg/ℓ	Lake Eufaula	15	14	15	17	28	28
	Lake Thunderbird	8.8	9.0	9.2	9.0	8.6	--
Specific conductance, U-MHO/cm	Lake Eufaula	400	400	400	400	625	630
	Lake Thunderbird	400	420	400	410	650	--
Dissolved oxygen, mg/ℓ	Lake Eufaula	6.4	6.2	6.2	3.4	7.6	7.4
	Lake Thunderbird	8.4	8.0	7.4	6.5	8.6	--
Water temperature, °C	Lake Eufaula	20	19	19	18.5	17.5	17
	Lake Thunderbird	20.5	19.5	20.0	19.0	15.0	--
Fecal coliforms, colonies/100 mℓ	Lake Eufaula	57	64	283	70	9700	9700
	Lake Thunderbird	4*	2*	1*	4*	790*	--
Fecal streptococcus, colonies/100 mℓ	Lake Eufaula	46	34	560	25	950	1250
	Lake Thunderbird	2*	1*	1*	10*	250	--

* Nonideal colony counts.

APPENDIX B: SAMPLE OUTPUT FROM WQRRS SIMULATION FOR THE 1970 POSTDIVERSION CASE

JULY 74 VERSION WRITTEN BY WRB, MODIFIED BY HRC

THIS RUN USES THE DATA CALCULATED BY THE USGS AT THE ARCADIA
SITE WITH CORRECT TEMP PROFILES, I.C.,S AND SECCHI DISK # 2
THIS IS THE POST-DIVERION RUN, 1970 DATA, BASE

SIMULATION DAYS	359
SPECIAL PRINT DAYS	0
OUTPUT FREQ(DAYS)	10
VERT OUTPUT	1
OUTPUT INTERVAL(HRS)	24
NUM OUTLETS	8
WEATHER OBS HRS	24
WEATHER FREQ	1
LOADING FREQ	30
COMPUT INT(/DAY)	1
NUMBER TRIBS	1
ALGAE GROUPS	2
NO.VERT.PROF PTS.	2

NORMAL DZ	1.00
MAXIMUM EL	320.0
EX DEPTH	2.000E 00
EVAP COEFF A	0.
EVAP COEFF B	1.200E-09
MIN STAB	0.
SW EX COEF	0.825E 00

CRITICAL STAB	1.000E-05
LOW GRAD COEF	6.500E-05
CALC INTERCEPT	0.
EXPONENT	0.
REACH LENGTH	6.000E 03
WATER SURFACE	3.110E 02

OUTLET	ELEVATION	EFF WIDTH
1	3.000E 00	6.130E 01
2	9.000E 00	8.990E 02
3	1.100E 01	9.600E 02
4	1.200E 01	1.040E 03
5	1.400E 01	1.070E 03
6	1.500E 01	1.280E 03
7	1.700E 01	1.340E 03
8	1.800E 01	1.480E 03

AD-A039 492 ARMY ENGINEER WATERWAYS EXPERIMENT STATION VICKSBURG MISS F/G 13/2
ARCADIA LAKE WATER-QUALITY EVALUATION.(U)
APR 77 R W HALL, R H PLUMB, K W THORNTON
UNCLASSIFIED WES-TR-Y-77-2 NL

4 OF 4
AD
A039492



END

DATE
FILMED
6-77

REACTION RATE MULTIPLIER PARAMETERS#

	CALIBRATION MAGNITUDES				CALIBRATION TEMPERATURES			
	K1	K2	K3	K4	T1	T2	T3	T4
ALGAE 1	0.10	0.98	0.98	0.10	3.0	23.0	31.0	42.0
ALGAE 2	0.10	0.98	0.98	0.10	4.0	24.0	35.0	45.0
ZOOPLANKTON	0.10	0.98	0.98	0.10	4.0	28.0	32.0	38.0
BENTHIC ANIMALS	0.10	0.98	0.98	0.10	0.	25.0	30.0	38.0
FISH 1	0.10	0.98	0.98	0.10	0.	1E 36	0.	1E 36
FISH 2	0.10	0.98	0.98	0.10	3.0	27.0	30.0	38.0
FISH 3	0.10	0.98	0.98	0.10	0.	22.0	27.0	35.0
BOD	0.10	0.98	0.98	0.10	4.0	30.0	35.0	45.0
NH3-N	0.10	0.98	0.98	0.10	4.0	30.0	35.0	45.0
NO2-N	0.10	0.98	0.98	0.10	4.0	30.0	35.0	45.0
DETRITUS	0.10	0.98	0.98	0.10	4.0	30.0	35.0	45.0

DECAY COEFFICIENTS,PER DAY MAX VALUE

NH3-N	0.140
NO2-N	0.320
DETRITUS	0.001
COLIFORM (AT 20 DEG C)	1.400

Q10 TEMPERATURE COEFFICIENT FOR COLIFORM. 1.040

CHEMICAL COMPOSITIONS OF BIOTA

	C	N	P
ALGAE	0.450	0.080	0.011
ZOOPLANKTON	0.450	0.081	0.012
FISH	0.480	0.080	0.011
BENTHOS	0.470	0.080	0.011
DETRITUS	0.320	0.070	0.009

DIGESTIVE EFFICIENCY OF BIOTA

ZOOPLANKTON	0.630
FISH	0.500
BENTHOS	0.510

MORTALITY RATES,PER DAY MAX VALUE

ZOOPLANKTON	0.500E-02
FISH	0.140E-02
BENTHOS	0.100E-02

RESPIRATION RATES,PER DAY MAX VALUE

PHYTOPLANKTON	0.170E 00

ZOOPLANKTON	0.800E-01
FISH	0.800E-02
BENTHOS	0.100E-01

DETRITUS SETTLING, METER/DAY 0.10000

OTHER PHYTOPLANKTON DATA

SETTLING, METER/DAY	0.15000	0.15000
OXYGENATION FACTOR	1.600	
PREFERANCE	0.700	0.300
SELFSHADING PER MG/L/M	0.250	

MAXIMUM SPECIFIC GROWTH RATE, PER DAY

PHYTOPLANKTON, 2 GROUPS	0.200E 01	0.180E 01	
ZOOPLANKTON	0.250E 00		
FISH, 3 GROUPS	0.	0.400E-01	0.300E-01
BENTHOS	0.400E-01		

HALF-SATURATION CONSTANTS OF ALGAE

	LIGHT	CO2	N	PO4
ALGAE 1	0.003	0.050	0.100	0.030
ALGAE 2	0.005	0.050	0.150	0.060

HALF-SATURATION CONSTANTS FOR ZOO, FISH AND BENTHO

ZOO GRAZE ON ALGAE	1.000
FISH 1 GRAZE ON ZOO	0.100E 36
FISH 2 GRAZE ON ZOO	0.500
FISH GRAZE ON BENTHOS	500.000
BENTHOS GRAZE ON SEDMT	250.000

STOICHIOMETRIC EQUIVALENCE OF CHEMICAL TRANSFORMATION

O2/NH3	3.500
O2/NO2	1.200
O2/DETRITUS	1.200
O2/BIOMASS	1.600

NO	ELEVATION	SEC AREA	CULM VOL	DELTA VOL	DELTA AREA
1	0.	1.000E 00	0.	3.036E 03	6.070E 03
2	1.0	6.070E 03	3.036E 03	1.519E 04	1.823E 04
3	2.0	2.430E 04	1.822E 04	3.745E 04	2.630E 04
4	3.0	5.060E 04	5.567E 04	6.175E 04	2.230E 04
5	4.0	7.290E 04	1.174E 05	9.085E 04	3.610E 04
6	5.0	1.090E 05	2.084E 05	1.150E 05	1.200E 04
7	6.0	1.210E 05	3.234E 05	3.075E 05	3.730E 05
8	7.0	4.940E 05	6.309E 05	7.145E 05	4.410E 05
9	8.0	9.350E 05	1.345E 06	1.288E 06	7.050E 05
10	9.0	1.640E 06	2.633E 06	1.890E 06	5.000E 05
11	10.0	2.140E 06	4.523E 06	2.425E 06	5.700E 05
12	11.0	2.710E 06	6.948E 06	3.145E 06	8.700E 05
13	12.0	3.580E 06	1.009E 07	3.955E 06	7.500E 05
14	13.0	4.330E 06	1.405E 07	4.745E 06	8.300E 05
15	14.0	5.160E 06	1.879E 07	5.610E 06	9.000E 05
16	15.0	6.060E 06	2.440E 07	6.585E 06	1.050E 06
17	16.0	7.110E 06	3.099E 07	7.805E 06	1.390E 06
18	17.0	8.500E 06	3.879E 07	8.665E 06	3.300E 05
19	18.0	8.830E 06	4.746E 07	9.415E 06	1.170E 06
20	19.0	1.000E 07	5.687E 07	1.050E 07	1.000E 06
21	20.0	1.100E 07	6.737E 07	1.150E 07	1.000E 06
22	21.0	1.200E 07	7.887E 07	1.260E 07	1.200E 06
23	22.0	1.320E 07	9.147E 07	1.380E 07	1.200E 06
24	23.0	1.440E 07	1.053E 08	1.525E 07	1.700E 06
25	24.0	1.610E 07	1.205E 08	1.680E 07	1.400E 06
26	25.0	1.750E 07	1.373E 08	1.825E 07	1.500E 06
27	26.0	1.900E 07	1.556E 08	1.980E 07	1.600E 06
28	27.0	2.060E 07	1.754E 08	2.175E 07	2.300E 06
29	28.0	2.290E 07	1.971E 08	2.365E 07	1.500E 06
30	29.0	2.440E 07	2.208E 08	2.565E 07	2.500E 06
31	30.0	2.690E 07	2.464E 08	0.	0.

INITIAL CONDITIONS OF THE ECOSYSTEM

ELEV M	TEMP C	DO MG/L	DOD MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	CCLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L	SEDM G/M2	BENTH MG/M2
20.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	186.
19.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	179.
18.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	173.
17.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	166.
16.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	159.
15.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	152.
14.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	145.
13.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	139.
12.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	132.
11.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	125.
10.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	118.
9.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	111.
8.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	104.
7.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	98.
6.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	91.
5.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	84.
4.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	77.
3.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	70.
2.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	64.
1.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	57.
0.5	3.50	13.00	0.	207.	7.3	8578.	1000.	5000.	200.	4000.	8.0E 00	410.	230.	180.	3.0	500.	0.	50.

FISH BIOMASS, KG/HA DRY WEIGHT

FISH 1 COLD WATER FISH	0.
FISH 2 WARM WATER FISH	20.00
FISH 3 BENTHOS FEEDERS	5.00

SUMMARY OF OUTPUT FOR SIMULATION DAY 90 (MAR) INTERVAL 1 (1 PER DAY)

METEOROLOGIC DATA

SHORTWAVE RAD(KC/M2/S)	1.940E-02
LONGWAVE RAD(KC/M2/S)	7.127E-02
DRY BULB TEMP(DEG C)	8.278
DEW POINT TEMP(DEG C)	7.722
WIND SPEED(M/SEC)	4.187
EVAPORATION RATE(M/M)	0.007
TOTAL EVAPORATION(M)	0.109

GENERAL INFORMATION

WATER ELEV(M)	2.098 01
SURF AREA(M2)	1.208 07
TOTAL VOL(M3)	7.826 07
TOTAL INFLOW(CMS)	7.308-01
TOTAL OUTFLOW(CMS)	5.008-01
RESIDENCE TIME(D)	1.248 03
ALGAL PROD(G/M2/D)	2.368-01

INFLOW AND OUTFLOW QUALITIES

TRIB	TEMP C	DO MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L
1	11.00	10.60	104.	8.3	315.	17000.	156.	210.	12500.	2.6E 03	22.	240.	171.	19.0	1057.
OUT	7.89	11.76	202.	8.7	242.	525.	5759.	242.	4360.	2.8E 00	628.	64.	70.	1.2	524.

OUTFLOW DISTRIBUTION(BOTTOM TO TOP)

GATE	1	2	3	4	5	6	7	8
FLOW CMS	0.	0.	0.	0.	0.	0.	0.	0.50

FISH BIOMASS, KG/HA DRY WEIGHT

FISH 1 COLD WATER FISH	0.
FISH 2 WARM WATER FISH	18.42
FISH 3 BENTHO FEEDERS	4.55

RESERVOIR QUALITIES

ELEV M	TEMP C	DO MG/L	DOD MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L	SEDM G/M2	BENTH MG/M2
20.5	7.99	11.74	0.15	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	20.	159.
19.5	7.99	11.74	0.15	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	20.	156.
18.5	7.99	11.74	0.15	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	21.	153.
17.5	7.99	11.74	0.14	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	21.	149.
16.5	7.99	11.74	0.14	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	21.	147.
15.5	7.99	11.74	0.14	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	21.	143.
14.5	7.99	11.74	0.14	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	21.	139.
13.5	7.99	11.74	0.14	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	21.	136.
12.5	7.99	11.74	0.14	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	21.	132.
11.5	7.99	11.74	0.14	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	21.	127.
10.5	7.99	11.74	0.14	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	21.	123.
9.5	7.99	11.74	0.14	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	21.	118.
8.5	7.99	11.74	0.14	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	21.	113.
7.5	7.99	11.74	0.13	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	20.	108.
6.5	7.99	11.74	0.13	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	20.	102.
5.5	7.99	11.74	0.13	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	19.	95.
4.5	7.99	11.74	0.13	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	19.	90.
3.5	7.99	11.74	0.13	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	19.	84.
2.5	7.99	11.74	0.13	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	18.	77.
1.5	7.99	11.74	0.13	202.	8.7	242.	525.	5766.	241.	4366.	2.8E 00	646.	63.	70.	1.2	524.	17.	71.
0.5	7.98	11.62	0.13	202.	8.7	248.	525.	5771.	241.	4367.	2.8E 00	930.	94.	0.	1.3	524.	9.	63.

SUMMARY OF OUTPUT FOR SIMULATION DAY 120 (APR) INTERVAL 1 (1 PER DAY)

METEOROLOGIC DATA

SHORTWAVE RAD(KC/M2/S)	4.001E-02
LONGWAVE RAD(KC/M2/S)	8.301E-02
DRY BULB TEMP(DEG C)	18.111
DEW POINT TEMP(DEG C)	14.778
WIND SPEED(M/SEC)	6.655
EVAPORATION RATE(M/M)	0.133
TOTAL EVAPORATION(M)	0.162

GENERAL INFORMATION

WATER ELEV(M)	2.128 01
SURF AREA(M2)	1.208 07
TOTAL VOL(M3)	8.098 07
TOTAL INFLOW(CMS)	2.728 00
TOTAL OUTFLOW(CMS)	1.448 00
RESIDENCE TIME(D)	3.448 02
ALGAL PROD(G/M2/D)	3.428 00

INFLOW AND OUTFLOW QUALITIES

TRIB	TEMP C	DO MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L
1	12.30	7.10	112.	7.9	963.	14000.	110.	90.	10580.	4.8E 03	7.	80.	58.	38.0	789.
OUT	18.21	8.46	194.	9.2	56.	877.	5383.	362.	4801.	1.3E 01	5411.	141.	152.	3.4	546.

OUTFLOW DISTRIBUTION(BOTTOM TO TOP)

GATE	1	2	3	4	5	6	7	8
FLOW CMS	1.44	0.	0.	0.	0.	0.	0.	0.

FISH BIOMASS, KG/HA DRY WEIGHT

FISH 1 COLD WATER FISH	0.
FISH 2 WARM WATER FISH	17.12
FISH 3 BENTHO FEEDERS	4.23

RESERVOIR QUALITIES

ELEV M	TEMP C	DO MG/L	DOD MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L	SEDM G/M2	BENTH MG/M2
30.5	18.39	8.21	1.92	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	39.	150.
29.5	18.39	8.21	1.91	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	39.	148.
28.5	18.39	8.21	1.88	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	39.	146.
27.5	18.39	8.21	1.85	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	39.	144.
26.5	18.39	8.21	1.83	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	38.	143.
25.5	18.39	8.21	1.80	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	38.	141.
24.5	18.39	8.21	1.77	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	38.	138.
23.5	18.39	8.21	1.75	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	38.	136.
22.5	18.39	8.21	1.73	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	38.	134.
21.5	18.39	8.21	1.71	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	38.	131.
10.5	18.39	8.21	1.70	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	38.	128.
9.5	18.39	8.21	1.69	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	38.	125.
8.5	18.39	8.21	1.68	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	37.	122.
7.5	18.39	8.21	1.67	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	37.	118.
6.5	18.39	8.21	1.66	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	36.	114.
5.5	18.39	8.21	1.60	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	35.	107.
4.5	18.39	8.21	1.63	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	35.	104.
3.5	18.39	8.21	1.63	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	34.	99.
2.5	18.39	8.21	1.63	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	34.	94.
1.5	18.39	8.21	1.62	194.	9.2	56.	882.	5389.	367.	4817.	1.3E 01	5343.	136.	164.	3.4	547.	31.	88.
0.5	18.18	6.93	1.54	194.	9.2	64.	885.	5452.	367.	4828.	1.4E 01	4317.	113.	0.	3.9	547.	17.	80.

SUMMARY OF OUTPUT FOR SIMULATION DAY 150 (MAY) INTERVAL 1 (1 PER DAY)

METEOROLOGIC DATA

SHORTWAVE RAD(KC/M2/S)	2.633E-02
LONGWAVE RAD(KC/M2/S)	9.778E-02
DRY BULB TEMP(DEG C)	23.500
DEW POINT TEMP(DEG C)	19.333
WIND SPEED(M/SEC)	3.614
EVAPORATION RATE(M/M)	0.083
TOTAL EVAPORATION(M)	0.300

GENERAL INFORMATION

WATER ELEV(M)	2.11E 01
SURF AREA(M2)	1.20E 07
TOTAL VOL(M3)	8.00E 07
TOTAL INFLOW(CMS)	1.53E 00
TOTAL OUTFLOW(CMS)	1.24E 00
RESIDENCE TIME(D)	6.05E 02
ALGAL PROD(G/M2/D)	1.61E 00

INFLOW AND OUTFLOW QUALITIES

TRIB	TEMP C	DO MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L
1	18.00	6.40	120.	7.9	923.	15000.	138.	470.	6430.	4.6E 04	12.	130.	93.	28.0	780.
OUT	19.43	0.	190.	8.8	137.	1375.	5648.	368.	4944.	6.5E 01	382.	7.	0.	3.7	558.

OUTFLOW DISTRIBUTION(BOTTOM TO TOP)

GATE	1	2	3	4	5	6	7	8
FLOW CMS	1.24	0.	0.	0.	0.	0.	0.	0.

FISH BIOMASS, KG/HA DRY WEIGHT

FISH 1 COLD WATER FISH	0.
FISH 2 WARM WATER FISH	20.43
FISH 3 BENTHO FEEDERS	3.41

RESERVOIR QUALITIES

ELEV M	TEMP C	DO MG/L	DOD MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L	SEDM G/M2	BENTH MG/M2
10.5	22.37	7.29	1.62	190.	9.4	33.	755.	5565.	353.	4848.	5.8E 01	3823.	88.	1177.	4.6	558.	70.	60.
9.5	22.35	6.87	1.60	190.	9.4	34.	756.	5574.	353.	4849.	5.8E 01	3741.	87.	1171.	4.6	558.	69.	57.
8.5	22.30	6.16	1.54	190.	9.4	36.	760.	5593.	354.	4853.	5.8E 01	3540.	85.	1154.	4.7	558.	67.	54.
7.5	22.25	5.30	1.48	190.	9.3	37.	766.	5615.	355.	4857.	5.8E 01	3319.	82.	1115.	4.7	558.	66.	49.
6.5	22.19	4.15	1.39	190.	9.3	39.	776.	5646.	356.	4862.	5.8E 01	3025.	39.	1037.	4.7	558.	65.	48.
5.5	22.10	3.32	1.34	190.	9.3	41.	791.	5661.	359.	4867.	5.8E 01	2865.	37.	921.	4.8	558.	64.	39.
4.5	21.99	2.16	1.28	190.	9.3	45.	822.	5680.	363.	4874.	5.8E 01	2665.	34.	721.	4.8	558.	62.	24.
3.5	21.84	0.94	1.20	190.	9.2	51.	877.	5687.	369.	4884.	5.9E 01	2383.	31.	522.	4.8	558.	61.	0.
2.5	21.63	0.	1.10	190.	9.2	58.	985.	5651.	371.	4894.	5.9E 01	2060.	27.	0.	5.0	558.	60.	0.
1.5	21.38	0.	1.06	190.	9.1	66.	1082.	5627.	372.	4904.	6.0E 01	1765.	24.	0.	4.9	558.	59.	0.
0.5	21.08	0.	1.01	190.	9.1	77.	1167.	5610.	374.	4913.	6.0E 01	1473.	20.	0.	4.8	558.	59.	0.
9.5	20.80	0.	0.97	190.	9.0	88.	1235.	5600.	374.	4921.	6.1E 01	1238.	18.	0.	4.7	558.	58.	0.
8.5	20.56	0.	0.94	190.	9.0	99.	1284.	5594.	374.	4927.	6.2E 01	1065.	16.	0.	4.6	558.	57.	0.
7.5	20.36	0.	0.90	190.	9.0	104.	1321.	5592.	374.	4932.	6.2E 01	900.	14.	0.	4.5	558.	56.	0.
6.5	20.20	0.	0.87	190.	8.9	119.	1345.	5593.	373.	4935.	6.3E 01	779.	12.	0.	4.3	558.	55.	0.
5.5	19.96	0.	0.82	190.	8.9	121.	1369.	5601.	372.	4939.	6.3E 01	561.	9.	0.	3.9	558.	51.	0.
4.5	19.76	0.	0.80	190.	8.9	127.	1384.	5611.	370.	4942.	6.4E 01	473.	8.	0.	3.8	558.	51.	0.
3.5	19.56	0.	0.77	190.	8.9	134.	1395.	5623.	369.	4945.	6.4E 01	388.	7.	0.	3.7	558.	50.	0.
2.5	19.41	0.	0.75	190.	8.8	141.	1402.	5634.	368.	4948.	6.5E 01	326.	6.	0.	3.7	558.	49.	0.
1.5	19.29	0.	0.74	190.	8.8	144.	1407.	5644.	366.	4949.	6.5E 01	268.	5.	0.	3.5	558.	45.	0.
0.5	19.23	0.	0.72	190.	8.8	148.	1408.	5650.	366.	4950.	6.5E 01	207.	4.	0.	3.2	558.	24.	0.

SUMMARY OF OUTPUT FOR SIMULATION DAY 180 (JUN) INTERVAL 1 (1 PER DAY)

METEOROLOGIC DATA

SHORTWAVE RAD(KC/M2/S)	5.918E-02
LONGWAVE RAD(KC/M2/S)	9.675E-02
DRY BULB TEMP(DEG C)	27.833
DEW POINT TEMP(DEG C)	21.389
WIND SPEED(M/SEC)	6.038
EVAPORATION RATE(M/M)	0.278
TOTAL EVAPORATION(M)	0.459

GENERAL INFORMATION

WATER ELEV(M)	2.10E 01
SURF AREA(M2)	1.20E 07
TOTAL VOL(M3)	7.89E 07
TOTAL INFLOW(CMS)	1.11E 00
TOTAL OUTFLOW(CMS)	8.27E-01
RESIDENCE TIME(D)	8.23E 02
ALGAL PROD(G/M2/D)	1.06E-01

INFLOW AND OUTFLOW QUALITIES

TRIB	TEMP C	DO MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L
1	16.40	6.20	128.	8.3	356.	6000.	340.	2450.	2710.	3.1E 04	15.	170.	121.	26.0	754.
OUT	24.13	4.75	188.	9.1	60.	140.	6784.	87.	4813.	2.7E 01	16.	6.	285.	5.2	565.

OUTFLOW DISTRIBUTION(BOTTOM TO TOP)

GATE	1	2	3	4	5	6	7	8
FLOW CMS	0.05	0.	0.	0.78	0.	0.	0.	0.

FISH BIOMASS, KG/HA DRY WEIGHT

FISH 1 COLD WATER FISH	0.
FISH 2 WARM WATER FISH	29.77
FISH 3 BENTHO FEEDERS	2.85

RESERVOIR QUALITIES

ELEV M	TEMP C	DO MG/L	DOD MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L	SEDM G/M2	BENTH MG/M2
10.5	26.75	7.63	0.19	188.	9.1	61.	137.	6768.	83.	4807.	2.4E 01	123.	36.	360.	4.5	565.	86.	42.
9.5	26.75	7.63	0.19	188.	9.1	61.	137.	6768.	83.	4807.	2.4E 01	123.	36.	360.	4.5	565.	85.	39.
8.5	26.75	7.63	0.18	188.	9.1	61.	137.	6768.	83.	4807.	2.4E 01	123.	36.	360.	4.5	565.	84.	36.
7.5	26.65	7.21	0.18	188.	9.1	61.	136.	6773.	83.	4808.	2.4E 01	93.	28.	354.	4.6	565.	82.	32.
6.5	26.41	6.85	0.17	188.	9.1	60.	136.	6777.	83.	4808.	2.4E 01	71.	22.	345.	4.7	565.	82.	30.
5.5	26.00	6.41	0.16	188.	9.1	60.	136.	6782.	83.	4809.	2.5E 01	50.	16.	330.	4.8	565.	80.	23.
4.5	25.47	5.92	0.15	189.	9.1	60.	135.	6787.	83.	4810.	2.5E 01	34.	11.	310.	5.0	565.	79.	13.
3.5	24.93	5.40	0.14	188.	9.1	60.	134.	6793.	82.	4810.	2.6E 01	23.	8.	285.	5.1	565.	78.	0.
2.5	24.43	4.89	0.13	188.	9.1	60.	133.	6798.	82.	4811.	2.6E 01	16.	6.	258.	5.2	565.	77.	0.
1.5	23.98	4.40	0.13	188.	9.1	60.	133.	6803.	82.	4811.	2.7E 01	12.	5.	231.	5.3	565.	76.	0.
0.5	23.57	3.92	0.12	188.	9.1	60.	133.	6809.	82.	4812.	2.7E 01	9.	4.	203.	5.4	565.	75.	0.
9.5	23.24	3.51	0.12	188.	9.1	60.	133.	6813.	82.	4813.	2.8E 01	7.	3.	180.	5.5	565.	74.	0.
8.5	22.99	3.18	0.12	188.	9.1	60.	134.	6817.	82.	4813.	2.8E 01	5.	3.	165.	5.5	565.	73.	0.
7.5	22.77	2.89	0.12	188.	9.1	60.	135.	6820.	83.	4814.	2.8E 01	4.	2.	169.	5.4	565.	72.	0.
6.5	22.59	2.64	0.11	188.	9.2	59.	136.	6823.	83.	4814.	2.8E 01	3.	2.	169.	5.2	565.	70.	0.
5.5	22.27	2.32	0.11	188.	9.2	59.	133.	6827.	83.	4814..	2.9E 01	2.	1.	162.	4.6	565.	66.	0.
4.5	22.06	2.08	0.11	188.	9.2	59.	133.	6830.	82.	4815.	2.9E 01	2.	1.	154.	4.7	565.	66.	0.
3.5	21.88	1.88	0.11	188.	9.2	59.	133.	6833.	82.	4815.	2.9E 01	1.	1.	146.	4.8	565.	66.	0.
2.5	21.77	1.73	0.11	188.	9.2	59.	134.	6834.	82.	4815.	2.9E 01	1.	1.	140.	4.8	565.	65.	0.
1.5	21.69	1.60	0.11	188.	9.1	61.	135.	6835.	83.	4816.	2.9E 01	1.	1.	135.	4.6	565.	60.	0.
0.5	21.64	1.53	0.11	188.	9.1	61.	136.	6836.	83.	4816.	2.9E 01	1.	1.	132.	4.2	565.	34.	0.

SUMMARY OF OUTPUT FOR SIMULATION DAY 210 (JUL) INTERVAL 1 (1 PER DAY)

METEOROLOGIC DATA

SHORTWAVE RAD(KC/M2/S)	7.069E-02
LONGWAVE RAD(KC/M2/S)	9.489E-02
DRY BULB TEMP(DEG C)	29.500
DEW POINT TEMP(DEG C)	21.278
WIND SPEED(M/SEC)	6.214
EVAPORATION RATE(M/M)	0.239
TOTAL EVAPORATION(M)	0.702

GENERAL INFORMATION

WATER ELEV(M)	2.06E 01
SURF AREA(M2)	1.20E 07
TOTAL VOL(M3)	7.48E 07
TOTAL INFLOW(CMS)	4.00E-01
TOTAL OUTFLOW(CMS)	8.60E-01
RESIDENCE TIME(D)	1.01E 03
ALGAL PROD(G/M2/D)	1.58E 00

INFLOW AND OUTFLOW QUALITIES

TRIB	TEMP C	DO MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L
1	23.90	5.90	99.	7.9	566.	32000.	70.	1.	19960.	5.0E 04	30.	330.	235.	20.0	914.
OUT	25.26	8.48	187.	9.3	41.	280.	6690.	108.	4956.	1.5E 01	2401.	276.	92.	3.3	569.

OUTFLOW DISTRIBUTION(BOTTOM TO TOP)

GATE	1	2	3	4	5	6	7	8
FLOW CMS	0.	0.	0.	0.	0.	0.	0.	0.86

FISH BIOMASS, KG/HA DRY WEIGHT

FISH 1 COLD WATER FISH	0.
FISH 2 WARM WATER FISH	26.31
FISH 3 BENTHO FEEDERS	2.31

RESERVOIR QUALITIES

ELEV M	TEMP C	DO MG/L	DOD MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L	SEDM G/M2	BENTH MG/M2
10.5	25.49	8.77	1.02	187.	9.3	39.	291.	6641.	113.	4956.	1.5E 01	2798.	307.	97.	3.2	570.	100.	36.
9.5	25.48	8.75	1.02	187.	9.3	39.	291.	6645.	113.	4957.	1.5E 01	2768.	304.	97.	3.2	570.	99.	34.
8.5	25.44	8.56	0.98	187.	9.3	40.	291.	6659.	113.	4959.	1.5E 01	2635.	290.	97.	3.3	570.	98.	31.
7.5	25.38	8.26	0.92	187.	9.3	41.	291.	6679.	113.	4962.	1.5E 01	2442.	272.	96.	3.3	570.	96.	27.
6.5	25.32	7.98	0.87	187.	9.3	43.	290.	6697.	113.	4964.	1.5E 01	2268.	256.	96.	3.3	570.	95.	25.
5.5	25.27	7.71	0.82	187.	9.2	44.	289.	6716.	112.	4966.	1.5E 01	2096.	240.	95.	3.3	570.	94.	18.
4.5	25.22	7.40	0.76	187.	9.2	46.	287.	6737.	112.	4969.	1.5E 01	1907.	221.	93.	3.3	570.	92.	10.
3.5	25.18	7.18	0.72	187.	9.2	47.	284.	6754.	112.	4970.	1.5E 01	1764.	208.	91.	3.4	570.	91.	0.
2.5	25.14	6.96	0.67	187.	9.2	47.	282.	6770.	112.	4972.	1.5E 01	1632.	195.	88.	3.4	570.	90.	0.
1.5	25.10	6.73	0.64	187.	9.2	49.	284.	6786.	112.	4977.	1.6E 01	1501.	181.	83.	3.4	570.	90.	0.
0.5	25.06	6.63	0.61	187.	9.2	50.	275.	6795.	111.	4973.	1.4E 01	1423.	174.	78.	3.4	570.	89.	0.
9.5	25.03	6.46	0.58	187.	9.2	51.	278.	6807.	111.	4978.	1.6E 01	1332.	164.	71.	3.4	570.	88.	0.
8.5	25.01	6.37	0.56	187.	9.2	51.	276.	6814.	111.	4978.	1.5E 01	1283.	158.	62.	3.4	570.	87.	0.
7.5	25.00	6.30	0.55	187.	9.2	51.	276.	6818.	111.	4978.	1.5E 01	1225.	152.	50.	3.4	570.	86.	0.
6.5	24.99	6.24	0.53	187.	9.2	51.	277.	6822.	111.	4980.	1.5E 01	1175.	146.	39.	3.3	570.	85.	0.
5.5	24.96	6.27	0.48	187.	9.2	51.	266.	6828.	110.	4976.	1.4E 01	1028.	127.	30.	3.1	569.	81.	0.
4.5	24.95	6.18	0.49	187.	9.2	53.	275.	6833.	112.	4983.	1.6E 01	1056.	130.	13.	3.3	570.	81.	0.
3.5	24.94	6.18	0.49	187.	9.2	53.	266.	6836.	110.	4978.	1.5E 01	1070.	133.	0.	3.4	570.	81.	0.
2.5	24.92	6.08	0.48	187.	9.2	53.	270.	6841.	110.	4981.	1.5E 01	1053.	131.	0.	3.6	570.	79.	0.
1.5	24.91	6.02	0.47	187.	9.2	53.	271.	6843.	110.	4982.	1.5E 01	1013.	126.	0.	4.3	570.	74.	0.
0.5	24.90	5.99	0.44	187.	9.2	54.	270.	6847.	110.	4982.	1.5E 01	862.	108.	0.	12.4	570.	42.	0.

SUMMARY OF OUTPUT FOR SIMULATION DAY 240 (AUG) INTERVAL 1 (1 PER DAY)

METEOROLOGIC DATA

SHORTWAVE RAD(KC/M2/S)	6.160E-02
LONGWAVE RAD(KC/M2/S)	8.833E-02
DRY BULB TEMP(DEG C)	26.167
DEW POINT TEMP(DEG C)	18.222
WIND SPEED(M/SEC)	4.363
EVAPORATION RATE(M/M)	0.233
TOTAL EVAPORATION(M)	0.954

GENERAL INFORMATION

WATER ELEV(M)	2.03E 01
SURF AREA(M2)	1.10E 07
TOTAL VOL(M3)	7.04E 07
TOTAL INFLOW(CMS)	3.10E-01
TOTAL OUTFLOW(CMS)	8.80E-01
RESIDENCE TIME(D)	9.26E 02
ALGAL PROD(G/M2/D)	1.13E 00

INFLOW AND OUTFLOW QUALITIES

TRIB	TEMP C	DO MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L
1	25.40	5.30	80.	7.8	570.	26000.	360.	1060.	19780.	4.8E 04	31.	350.	247.	21.0	936.
OUT	25.55	6.49	186.	9.3	35.	450.	6377.	205.	5069.	1.3E 01	2578.	212.	511.	3.1	573.

OUTFLOW DISTRIBUTION(BOTTOM TO TOP)

GATE	1	2	3	4	5	6	7	8
FLOW CMS	0.88	0.	0.	0.	0.	0.	0.	0.

FISH BIOMASS, KG/HA DRY WEIGHT

FISH 1 COLD WATER FISH	0.
FISH 2 WARM WATER FISH	22.61
FISH 3 BENTHO FEEDERS	1.84

RESERVOIR QUALITIES

ELEV M	TEMP C	DO MG/L	DOD MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L	SEDM G/M2	BENTH MG/M2
19.5	25.52	6.74	1.16	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	119.	33.
18.5	25.52	6.74	1.16	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	117.	30.
17.5	25.52	6.74	1.13	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	114.	26.
16.5	25.52	6.74	1.11	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	114.	24.
15.5	25.52	6.74	1.08	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	111.	17.
14.5	25.52	6.74	1.05	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	109.	9.
13.5	25.52	6.74	1.03	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	107.	0.
12.5	25.52	6.74	1.01	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	106.	0.
11.5	25.52	6.74	1.01	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	105.	0.
10.5	25.52	6.74	0.99	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	103.	0.
9.5	25.52	6.74	0.99	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	102.	0.
8.5	25.52	6.74	0.98	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	101.	0.
7.5	25.52	6.74	0.97	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	100.	0.
6.5	25.52	6.74	0.96	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	99.	0.
5.5	25.52	6.74	0.90	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	97.	0.
4.5	25.52	6.74	0.89	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	97.	0.
3.5	25.52	6.74	0.85	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	98.	0.
2.5	25.52	6.74	0.83	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	98.	0.
1.5	25.52	6.74	0.82	186.	9.4	29.	442.	6386.	203.	5073.	1.3E 01	2494.	209.	673.	3.0	573.	94.	0.
0.5	25.52	4.32	0.79	186.	9.3	33.	436.	6486.	203.	5087.	1.3E 01	1549.	127.	519.	3.7	573.	59.	0.

SUMMARY OF OUTPUT FOR SIMULATION DAY 270 (SEP) INTERVAL 1 (1 PER DAY)

METEOROLOGIC DATA

SHORTWAVE RAD(KC/M2/S)	1.657E-02
LONGWAVE RAD(KC/M2/S)	8.014E-02
DRY BULB TEMP(DEG C)	13.833
DEW POINT TEMP(DEG C)	10.556
WIND SPEED(M/SEC)	3.349
EVAPORATION RATE(M/M)	0.148
TOTAL EVAPORATION(M)	1.170

GENERAL INFORMATION

WATER ELEV(M)	2.076E 01
SURF AREA(M2)	1.20E 07
TOTAL VOL(M3)	7.60E 07
TOTAL INFLOW(CMS)	3.71E 00
TOTAL OUTFLOW(CMS)	6.10E-01
RESIDENCE TIME(D)	2.37E 02
ALGAL PROD(G/M2/D)	1.93E-02

INFLOW AND OUTFLOW QUALITIES

TRIB	TEMP C	DO MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L
1	25.60	5.80	77.	7.7	680.	20000.	128.	600.	6680.	5.9E 04	31.	360.	252.	21.0	782.
OUT	20.83	7.99	173.	9.3	41.	683.	7793.	310.	5285.	1.9E 02	10.	32.	190.	4.5	598.

OUTFLOW DISTRIBUTION(BOTTOM TO TOP)

GATE	1	2	3	4	5	6	7	8
FLOW CMS	0.	0.	0.	0.	0.	0.	0.	0.61

FISH BIOMASS, KG/HA DRY WEIGHT

FISH 1 COLD WATER FISH	0.
FISH 2 WARM WATER FISH	31.96
FISH 3 BENTHO FEEDERS	1.46

RESERVOIR QUALITIES

ELEV M	TEMP C	DO MG/L	DOD MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L	SEDM G/M2	BENTH MG/M2
10.5	20.00	8.03	0.40	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	131.	41.
9.5	20.00	8.03	0.40	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	131.	38.
8.5	20.00	8.03	0.40	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	129.	35.
7.5	20.00	8.03	0.41	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	126.	30.
6.5	20.00	8.03	0.41	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	126.	28.
5.5	20.00	8.03	0.41	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	123.	20.
4.5	20.00	8.03	0.41	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	121.	11.
3.5	20.00	8.03	0.41	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	119.	0.
2.5	20.00	8.03	0.41	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	118.	0.
1.5	20.00	8.03	0.41	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	117.	0.
0.5	20.00	8.03	0.41	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	115.	0.
9.5	20.00	8.03	0.41	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	114.	0.
8.5	20.00	8.03	0.41	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	114.	0.
7.5	20.00	8.03	0.41	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	112.	0.
6.5	20.00	8.03	0.41	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	111.	0.
5.5	20.00	8.03	0.40	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	111.	0.
4.5	20.00	8.03	0.40	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	111.	0.
3.5	20.00	8.03	0.40	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	112.	0.
2.5	20.00	8.03	0.40	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	113.	0.
1.5	20.00	8.03	0.40	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	109.	0.
0.5	20.00	8.03	0.40	173.	9.3	41.	685.	7843.	309.	5291.	1.9E 02	9.	31.	169.	4.5	599.	70.	0.

SUMMARY OF OUTPUT FOR SIMULATION DAY 300 (OCT) INTERVAL 1 (1 PER DAY)

METEOROLOGIC DATA

SHORTWAVE RAD(KC/M2/S)	1.934E-02
LONGWAVE RAD(KC/M2/S)	6.690E-02
DRY BULB TEMP(DEG C)	7.833
DEW POINT TEMP(DEG C)	6.611
WIND SPEED(M/SEC)	8.197
EVAPORATION RATE(M/M)	0.225
TOTAL EVAPORATION(M)	1.275

GENERAL INFORMATION

WATER ELEV(M)	2.088E 01
SURF AREA(M2)	1.208E 07
TOTAL VOL(M3)	7.675E 07
TOTAL INFLOW(CMS)	1.32E 00
TOTAL OUTFLOW(CMS)	5.40E-01
RESIDENCE TIME(D)	6.73E 02
ALGAL PROD(G/M2/D)	2.18E-02

INFLOW AND OUTFLOW QUALITIES

TRIB	TEMP C	DO MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L
1	22.00	5.30	120.	8.1	463.	1.	1400.	1.	5060.	1.8E 04	5.	60.	42.	25.0	779.
OUT	15.57	9.90	170.	9.2	53.	62.	8502.	41.	5290.	2.6E 01	19.	26.	8.	3.6	606.

OUTFLOW DISTRIBUTION(BOTTOM TO TOP)

GATE	1	2	3	4	5	6	7	8
FLOW CMS	0.	0.	0.	0.	0.	0.	0.	0.54

FISH BIOMASS, KG/HA DRY WEIGHT

FISH 1 COLD WATER FISH	0.
FISH 2 WARM WATER FISH	27.64
FISH 3 BENTHO FEEDERS	1.18

RESERVOIR QUALITIES

ELEV M	TEMP C	DO MG/L	DOD MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L	SEDM G/M2	BENTH MG/M2
20.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	141.	55.
19.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	140.	51.
18.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	138.	47.
17.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	136.	40.
16.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	135.	38.
15.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	133.	27.
14.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	131.	14.
13.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	129.	0.
12.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	128.	0.
11.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	127.	0.
10.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	125.	0.
9.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	124.	0.
8.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	124.	0.
7.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	122.	0.
6.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	121.	0.
5.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	120.	0.
4.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	121.	0.
3.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	122.	0.
2.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	122.	0.
1.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	118.	0.
0.5	15.20	9.95	0.04	170.	9.2	54.	59.	8499.	39.	5290.	2.6E 01	20.	26.	7.	3.6	607.	75.	0.

SUMMARY OF OUTPUT FOR SIMULATION DAY 330 (NOV) INTERVAL 1 (1 PER DAY)

METEOROLOGIC DATA

SHORTWAVE RAD(KC/M2/S)	2.068E-02
LONGWAVE RAD(KC/M2/S)	7.344E-02
DRY BULB TEMP(DEG C)	15.889
DEW POINT TEMP(DEG C)	10.389
WIND SPEED(M/SEC)	8.506
EVAPORATION RATE(M/M)	0.001
TOTAL EVAPORATION(M)	1.379

GENERAL INFORMATION

WATER ELEV(M)	2.078 01
SURF AREA(M2)	1.208 07
TOTAL VOL(M3)	7.548 07
TOTAL INFLOW(CMS)	4.408-01
TOTAL OUTFLOW(CMS)	4.508-01
RESIDENCE TIME(D)	1.948 03
ALGAL PROD(G/M2/D)	1.008-02

INFLOW AND OUTFLOW QUALITIES

TRIB	TEMP C	DO MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L
1	15.00	7.90	112.	8.2	387.	5000.	9058.	150.	9380.	1.7E 04	13.	140.	100.	12.0	788.
OUT	5.68	12.35	169.	9.1	88.	76.	8578.	30.	5348.	1.1E 01	39.	23.	4.	2.4	609.

OUTFLOW DISTRIBUTION(BOTTOM TO TOP)

GATE	1	2	3	4	5	6	7	8
FLOW CMS	0.	0.	0.	0.	0.	0.	0.	0.45

FISH BIOMASS, KG/HA DRY WEIGHT

FISH 1 COLD WATER FISH	0.
FISH 2 WARM WATER FISH	24.45
FISH 3 BENTHO FEEDERS	1.01

RESERVOIR QUALITIES

ELEV M	TEMP C	DO MG/L	DOD MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L	SEDM G/M2	BENTH MG/M2
10.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	148.	69.
9.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	148.	64.
8.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	146.	58.
7.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	144.	50.
6.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	143.	48.
5.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	140.	33.
4.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	139.	18.
3.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	137.	0.
2.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	136.	0.
1.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	134.	0.
10.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	133.	0.
9.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	132.	0.
8.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	131.	0.
7.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	130.	0.
6.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	129.	0.
5.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	128.	0.
4.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	128.	0.
3.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	129.	0.
2.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	129.	0.
1.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	124.	0.
0.5	5.92	12.43	0.01	169.	9.1	92.	77.	8579.	30.	5350.	1.1E 01	39.	22.	4.	2.4	609.	79.	0.

SUMMARY OF OUTPUT FOR SIMULATION DAY 360 (DEC) INTERVAL 1 (1 PER DAY)

METEOROLOGIC DATA

SHORTWAVE RAD(KC/M2/S)	1.612E-02
LONGWAVE RAD(KC/M2/S)	5.423E-02
DRY BULB TEMP(DEG C)	0.167
DEW POINT TEMP(DEG C)	-2.778
WIND SPEED(M/SEC)	4.231
EVAPORATION RATE(M/M)	0.073
TOTAL EVAPORATION(M)	1.431

GENERAL INFORMATION

WATER ELEV(M)	2.06E 01
SURF AREA(M2)	1.20E 07
TOTAL VOL(M3)	7.42E 07
TOTAL INFLOW(CMS)	1.60E-01
TOTAL OUTFLOW(CMS)	4.00E-01
RESIDENCE TIME(D)	2.15E 03
ALGAL PROD(G/M2/D)	7.91E-03

INFLOW AND OUTFLOW QUALITIES

TRIB	TEMP C	DO MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L
1	12.30	8.40	124.	8.1	557.	28000.	2740.	1.	12840.	3.6E 03	26.	290.	102.	22.0	821.
OUT	4.75	12.61	169.	9.0	127.	151.	8601.	50.	5388.	9.3E-01	71.	19.	3.	1.6	610.

OUTFLOW DISTRIBUTION(BOTTOM TO TOP)

GATE	1	2	3	4	5	6	7	8
FLOW CMS	0.	0.	0.	0.	0.	0.	0.	0.40

FISH BIOMASS, KG/HA DRY WEIGHT

FISH 1 COLD WATER FISH	0.
FISH 2 WARM WATER FISH	22.73
FISH 3 BENTHO FEEDERS	0.88

RESERVOIR QUALITIES

ELEV M	TEMP C	DO MG/L	DOD MG/L	ALKA MG/L	PH	CO2C UG/L	NH3N UG/L	NO3N UG/L	NO2N UG/L	PO4P UG/L	COLIFM MPN/100	ALGA1 UG/L	ALGA2 UG/L	ZOO UG/L	DETRI MG/L	TDS MG/L	SEDM G/M2	BENTH MG/M2
30.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	154.	83.
29.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	154.	77.
28.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	152.	70.
27.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	149.	60.
26.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	148.	57.
25.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	146.	40.
24.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	144.	22.
23.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	143.	0.
22.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	141.	0.
21.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	140.	0.
20.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	139.	0.
9.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	138.	0.
8.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	137.	0.
7.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	135.	0.
6.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	134.	0.
5.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	133.	0.
4.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	133.	0.
3.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	134.	0.
2.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	134.	0.
1.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	129.	0.
0.5	4.32	12.72	0.01	169.	9.0	130.	154.	8602.	50.	5390.	9.5E-01	70.	18.	3.	1.6	611.	81.	0.

APPENDIX C: PREDICTED LAKE FISHERY

1. Mr. Robert M. Jenkins, Director, Reservoir Fisheries Research Program, U. S. Fish and Wildlife Service, Fayetteville, Arkansas, provided an estimation of the quantity and quality of the Arcadia Lake Fishery using regression procedures. Definitions of types of reservoirs represented in the subsamples and of environmental variables and compilations of the multiple-regression equations used in estimating fish standing crop and angler harvest and effort are given in Tables C1-C3.

2. The following Arcadia Lake characteristics were assumed for the regression analyses:

Use	Nonhydropower, chemical types 2 and 4*
Area	1278 ha
Mean Depth	6.7 m
Maximum Depth	22 m
Fluctuation	2.4 m
Storage Ratio	1.8
Shore Development	5.2
Dissolved Solids	410 mg/ℓ
Growing Season	220 days
Chemical Type	4

3. Because of initial uncertainty of thermocline and outlet depths, three analyses were conducted for three combinations of thermocline and outlet depths. The three combinations follow:

Depth, m	
Thermocline	Outlet
3.6	3
6.1	9
6.1	15

4. Subsequent thermal simulations using alternative outlet works suggested that the thermocline depth would be 5 to 7 m and that the average outlet depth would be 3 m. However, suggested outlet works were capable of withdrawing water at several depths, so specification

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* Chemical type 2 - most of the dissolved solids are composed of calcium-magnesium, sulfate-chloride.
Chemical type 4 - most of the dissolved solids are composed of sodium-potassium, sulfate-chloride.

of a single outlet depth is difficult. The following discussion of the lake fishery is based on a thermocline depth of 6 m and outlet depth of 9 m.

Predicted Standing Crop

5. Thermocline depth is negatively related and total dissolved solids is positively related to standing crop. Using Equation 3 presented in Table C2, the total standing crop in Arcadia Lake was predicted to increase from 260 kg/ha in the first year of impoundment to 420 kg/ha at age 100.

6. An estimation of total standing crop based on dissolved solids was 290 kg/ha (Equation 4). An estimation based on the same variable but restricted to impoundments of the same chemical type as Arcadia was 200 kg/ha (Equation 10). Of the total standing crop, 74 kg/ha would be clupeids (Equation 19) and 64 kg/ha would be sport fish (Equation 20). The remaining standing crop primarily would be carp, redhorse, carpsucker, and drum.

Predicted Angler Harvest and Effort

7. Sport fish harvest (Equation D from Table C3) would be expected to decrease from 47 kg/ha shortly after impoundment to 13 kg/ha after 100 yr with a mean over the 100-yr period of 18 kg/ha. Results from a study by Jenkins and Morais indicate that average angler harvest equals about 35 percent of the standing crop of sport fishes including carp.[126*] Roughly 60 percent of the top predators, 25 percent of the panfish, and 20 percent of the catfish and carp normally are harvested annually. Estimated black bass harvest decreased from 8 kg/ha following impoundment to 2 kg/ha at age 100 (Equation M). Similarly, sunfish harvest decreased from 14 kg/ha to less than 1 kg/ha (Equation N).

8. The 100-yr mean for angler effort was 82,000 days/yr (Equations E and H), and expected angler days/ha was 64.

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* Raised numerals refer to similarly numbered items in References at end of main text.

Table C1

Definitions of Types of Reservoirs Represented in Subsamples and of Environmental Variables

Term	Definition
All	Total sample, representing all types of reservoirs.
With thermocline	Reservoirs in sample which form a stable thermocline (>1°C in temperature per metre)
Chemical type 1	Most of the dissolved solids in the reservoir water are composed of calcium-magnesium, carbonate-bicarbonate (see Rainwater, F. H., 1962, Hydrologic Invest. Atlas HA-61, Plate 2).
Chemical type 2	Most of the dissolved solids are composed of calcium-magnesium, sulfate-chloride.
Chemical type 3	Most of the dissolved solids are composed of calcium-potassium, carbonate-bicarbonate.
Chemical type 4	Most of the dissolved solids are composed of sodium-potassium, sulfate-chloride.
Hydropower storage	Reservoirs with hydroelectric power generation operation and with storage ratio less than 0.165 (water exchange less than once in 60 days).
Hydropower mainstream	Reservoirs with hydroelectric power generation operation and with storage ratio less than 0.165 (water exchange greater than once in 60 days).
Nonhydropower	Reservoirs in sample which do not have hydroelectric generation function (flood control, irrigation, water supply, recreation reservoirs).
"Selected" reservoirs (Formula E)	Reservoirs less than 70,000 acres, with total dissolved solids less than 600 ppm, and growing season greater than 140 days.
"Mountain" reservoirs	Hydro-hydropower reservoirs in the Great Smoky, Ozark, and Ouachita Mountains.
R^2	Coefficient of determination (portion of total variability explained by formula); Pr = the probability of obtaining an F this large or larger by chance when the hypothesis of no correlation is true; N = the number of reservoirs in sample.
Area	Surface area in acres at average annual pool level when data are available; otherwise, use power, conservation, summer, or operating pool area.
Mean depth	In feet, at listed area.
Outlet depth	Midline depth, in feet, of outlet.
Thermocline depth	In feet of top of thermocline (water temperature change of 1°C/metre or more) on or about 1 August.
Fluctuation	Mean annual vertical fluctuation of reservoir water surface level in feet.
Storage ratio	The ratio of the reservoir water volume in acre-feet (at the surface area listed) to the average annual discharge in acre-feet.
Shore development	The ratio of shoreline length to the circumference of a circle equal in area to that of the reservoir.
Total dissolved solids	Residue on evaporation at 180°C in ppm.
Growing season	Average number of days between first and last frost.
Age of reservoir	In years, following closure of dam.
Standing crop	Estimated crop of fish in pounds per acre as determined by recovery of fishes from coves or open-water areas enclosed by blockoff nets following application of rotenone.
Clupeids	Gizzard shad, threadfin shad, blueback herring, Alabama shad, skipjack herring, alewife.
Sport fish harvest	Estimated harvest of fishes by sport fishermen, in pounds per acre per year.
Commercial fish harvest	Estimated harvest by commercial fishermen or rough fish removal crews, primarily by gill and trammel nets and seines, in pounds per acre per year.

Table C2

Reservoir Fish Standing Crop Estimation Equations

Equation Number	Description and Equation*
1	Estimation of total standing crop - all reservoir types
	log (total standing crop in pounds per acre) = 1.9907 + 0.5690 (log (dissolved solids in ppm/mean depth in feet))- 0.2831 (log (dissolved solids in ppm/mean depth in feet))^2
	R^2 = 0.33 $Pr = < 10^{-8}$ N = 139
2	Estimation of total standing crop - all reservoir types
	log (total standing crop in pounds per acre) = 1.6720 + 0.1776 log (outlet depth) + 0.6925 log (dissolved solids/ mean depth) - 0.2458 (log (dissolved solids/mean depth))^2
	N = 173 R^2 = 0.51 $Pr = < 10^{-8}$
3	Estimation of total standing crop in reservoirs with a stable thermocline
	log (total standing crop) = 1.7589 - 0.405 log (thermocline depth) + 0.432 log (dissolved solids) + 0.106 log (age)
	N = 84 R^2 = 0.35 $Pr = 2.8 \times 10^{-7}$
4	Estimation of total standing crop in reservoirs with a stable thermocline
	log (standing crop) = -0.4016 + 2.2100 log (dissolved solids) - 0.4326 (log (dissolved solids))^2
	N = 99 R^2 = 0.48 $Pr = < 10^{-8}$
5	Estimation of total standing crop in hydropower storage reservoirs
	log (total standing crop) = -0.6126 + 2.3658 log (dissolved solids) - 0.46 (log (dissolved solids))^2
	N = 44 R^2 = 0.74 $Pr = < 10^{-8}$
6	Estimation of total standing crop in hydropower storage reservoirs
	log (total standing crop) = -0.1582 - 0.1446 log (outlet depth) + 2.1301 log (dissolved solids) - 0.3869 (log (dissolved solids))^2
	N = 44 R^2 = 0.77 $Pr = < 10^{-8}$
7	Estimation of total standing crop in hydropower mainstream reservoirs
	log (total standing crop) = -0.6150 + 2.2521 log (dissolved solids) - 0.3762 (log (dissolved solids))^2
	N = 52 R^2 = 0.70 $Pr = < 10^{-8}$
8	Estimation of total standing crop in hydropower mainstream reservoirs
	log (standing crop) = -2.4916 + 0.07354 log (age) + 0.7432 log (growing season) + 2.2673 log (dissolved solids) - 0.3681 (log (dissolved solids))^2
	N = 52 R^2 = 0.73 $Pr = < 10^{-8}$
9	Estimation of total standing crop in nonhydropower reservoirs of chemical types 1 and 3
	log (total standing crop) = 1.2867 + 0.1275 log (age) + 0.1373 log (area) + 0.7027 log (dissolved solids/mean depth) - 0.2459 (log (dissolved solids/mean depth))^2
	N = 47 R^2 = 0.53 $Pr = 2.6 \times 10^{-6}$
10	Estimation of total standing crop in nonhydropower reservoirs of chemical types 2 and 4
	log (total standing crop) = -0.9914 + 2.3317 log (dissolved solids) - 0.417 (log (dissolved solids))^2
	N = 30 R^2 = 0.64 $Pr = 1.8 \times 10^{-6}$
11	Estimation of total standing crop minus clupeids in hydropower storage reservoirs
	Standing crop less clupeids = -108.2 + 70.32 log (growing season) + 55.48 log (storage ratio) + 20.7 log (fluctuation) + 77.61 log (dissolved solids/mean depth) + 24.1 (log (dissolved solids/mean depth))^2
	N = 40 R^2 = 0.58 $Pr = 1.3 \times 10^{-5}$
12	Estimation of total standing crop minus clupeids in hydropower mainstream reservoirs
	Standing crop less clupeids = -352.7 + 308.9 log (dissolved solids) - 32.42 (log (dissolved solids))^2
	N = 49 R^2 = 0.51 $Pr = 1.5 \times 10^{-7}$
13	Estimation of total standing crop minus clupeids in nonhydropower reservoirs of chemical types 1 and 3
	log (standing crop less clupeids) = 0.4299 + 0.1074 log (area) - 0.3491 log (mean depth) + 0.0906 (outlet depth) + 0.2694 log (shore development) + 1.1953 log (dissolved solids) - 0.2617 (log (dissolved solids))^2
	N = 44 R^2 = 0.35 Pr = 0.01

(Continued)

*Equations are based on the U. S. customary system of measures (metric system equations are also available). A table for converting U. S. customary units of measurement to metric (SI) units can be found on page 12. All data transformed to base 10 logarithms. The equations were derived from data on U. S. reservoirs greater than 500 acres in area.

Table C2 (Continued)

Equation Number	Description and Equation*
14	Estimation of standing crop less clupeids, in nonhydropower reservoirs of chemical types 2 and 4
	log (standing crop less clupeids) = 1.7499 + 0.2766 log (outlet depth) - 0.1866 log (shore development) + 0.00984 log (dissolved solids)
	N = 20 R^2 = 0.60 Pr = 0.0018
15	Estimation of standing crop minus clupeids in reservoirs without a stable thermocline
	log (standing crop less clupeids) = -2.0027 - 0.454 log (maximum depth) + 0.391 log (outlet depth) + 0.257 log (dissolved solids) + 1.572 log (growing season)
	N = 51 R^2 = 0.44 Pr = 2.1×10^{-5}
16	Estimation of clupeid standing crop in hydropower storage reservoirs
	log (clupeid standing crop) = -4.12 - 0.2355 log (outlet depth) + 5.7302 log (dissolved solids) - 1.2263 (log (dissolved solids))^2
	N = 40 R^2 = 0.61 Pr = 3.1×10^{-7}
17	Estimation of clupeid standing crop in hydropower mainstream reservoirs
	log (clupeid standing crop) = 0.065 + 0.2114 log (fluctuation) + 1.1572 log (dissolved solids) - 0.1317 (log (dissolved solids))^2
	N = 49 R^2 = 0.48 Pr = 2.4×10^{-6}
18	Estimation of clupeid standing crop in nonhydropower reservoirs of chemical types 1 and 3
	log (clupeid standing crop) = 0.7041 + 2.47 log (dissolved solids/mean depth) - 1.1595 (log (dissolved solids/mean depth))^2
	N = 44 R^2 = 0.40 Pr = 4.5×10^{-5}
19	Estimation of clupeid standing crop in nonhydropower reservoirs of chemical types 2 and 4
	log (clupeid standing crop) = 0.6209 - 0.1875 log (area) + 0.3114 log (fluctuation) + 1.1153 log (dissolved solids) - 0.1962 (log (dissolved solids))^2
	N = 27 R^2 = 0.43 Pr = 0.01
20	Estimation of sport fishing standing crop (trout, pickerels, pike, catfishes, bullheads, white bass, sunfishes, black basses, crappies, sauger, walleye) - all reservoir types
	log (sport fish standing crop in pounds per acre) = 1.7183 + 0.3583 log (dissolved solids/mean depth) - 0.2585 (log (dissolved solids/mean depth))^2
	N = 140 R^2 = 0.16 Pr = 1.3×10^{-5}
21	Estimation of sport fish standing crop in nonhydropower reservoirs of chemical types 2 and 4
	log (sport fish standing crop + 1) = -2.914 - 0.8984 log (maximum depth + 1) + 0.4996 log (outlet depth + 1) + 4.8188 log (dissolved solids + 1) - 1.0206 (log (dissolved solids + 1))^2
	N = 26 R^2 = 0.80 Pr = 7.2×10^{-7}
22	Estimation of black bass standing crop in hydropower storage reservoirs
	log (black bass standing crop) = -4.4928 + 2.1242 log (growing season) + 0.2438 log (storage ratio) + 0.4308 log (fluctuation) + 0.14304 log (dissolved solids) - 0.1646 (log (dissolved solids))^2
	N = 44 R^2 = 0.28 Pr = 0.026
23	Estimation of black bass standing crop in hydropower mainstream reservoirs
	log (black bass standing crop) = -5.6304 + 2.6989 log (growing season) + 0.2575 log (dissolved solids/mean depth) + 0.0408 (log (dissolved solids/mean depth))^2
	N = 50 R^2 = 0.29 Pr = 0.0011
24	Estimation of black bass standing crop in nonhydropower reservoirs of chemical types 1 and 3
	log (black bass standing crop) = -11.2534 + 0.5649 log (mean depth) + 0.1979 log (outlet depth) + 3.6879 log (growing season) + 3.1325 log (dissolved solids) - 0.8234 (log (dissolved solids))^2
	N = 47 R^2 = 0.37 Pr = 0.0016
25	Estimation of largemouth bass standing crop in hydropower storage reservoir
	log (largemouth bass standing crop + 1) = -5.3553 + 2.7328 log (growing season + 1) + 0.3838 log (storage ratio + 1) - 0.4317 log (thermocline depth + 1) + 0.3185 log (fluctuation + 1)
	N = 44 R^2 = 0.34 Pr = 0.0025

(Continued)

Table C2 (Concluded)

Equation Number	Description and Equation*
26	*Estimation of largemouth bass standing crop in hydropower mainstream reservoirs* log (largemouth bass standing crop + 1) = -7.3057 + 0.1267 log (age + 2) + 2.6738 log (growing season + 1) + 2.0599 (storage ratio + 1) + 1.3181 log (dissolved solids + 1) - 0.2117 (log (dissolved solids + 1))^2 N = 50 R^2 = 0.35 Pr = 0.0014
27	Estimation of largemouth bass standing crop in nonhydropower reservoirs of chemical types 1 and 3 log (largemouth bass standing crop + 1) = -7.6987 + 0.2780 log (outlet depth + 1) + 0.2772 log (shore development + 1) + 2.5871 log (growing season + 1) - 0.2025 log (fluctuation + 1) + 2.5711 log (dissolved solids + 1) - 0.6710 (log (dissolved solids))^2 N = 46 R^2 = 0.36 Pr = 0.005
28	Estimation of largemouth bass standing crop in nonhydropower reservoirs of chemical types 2 and 4 log (largemouth bass standing crop + 1) = -0.1787 + 0.3775 log (mean depth + 1) - 0.9500 log (maximum depth + 1) + 0.2669 log (outlet depth + 1) + 1.478 log (dissolved solids + 1) - 0.269 (log (dissolved solids + 1))^2 N = 30 R^2 = 0.36 Pr = 0.04
29	Estimation of spotted bass standing crop in hydropower mainstream reservoirs log (spotted bass standing crop + 1) = -8.9625 + 0.2283 log (area + 1) + 2.7227 log (mean depth + 1) - 4.5815 log (maximum depth + 1) + 0.8527 log (outlet depth + 1) + 5.0895 log (growing season + 1) N = 22 R^2 = 0.56 Pr = 0.015
30	Estimation of smallmouth bass standing crop in hydropower reservoirs log (smallmouth bass standing crop + 1) = -9.9429 + 0.5968 log (age + 2) + 0.3726 log (mean depth + 1) - 0.0038 log (shore development + 1) + 0.06652 log (fluctuation + 1) - 0.1037 log (dissolved solids + 1) N = 36 R^2 = 0.35 Pr = 0.02
31	Estimation of smallmouth bass standing crop in hydropower mainstream reservoirs log (smallmouth bass standing crop + 1) = -5.4005 + 2.2626 log (age + 2) + 0.2889 log (fluctuation + 1) + 2.8760 log (dissolved solids + 1) - 0.9665 (log (dissolved solids + 1))^2 N = 11 R^2 = 0.77 Pr = 0.04

Table C3

Reservoir Angler Report and Harvest Estimation

Equation Number	Description and Equation*
A	Estimation of total annual sport fish harvest - all reservoir types
	log (sport fish harvest) = 0.61 + 1.233 (M) - 0.4774 (log (M))^2 where M = dissolved solids/thermocline depth (or mean depth if a stable thermocline is not formed)
	N = 97 R^2 = 0.28 Pr = 5.7×10^{-7}
B	Estimation of annual harvest of black basses - all reservoir types
	log (black bass harvest) = -22.08 + 16.649 log (growing season) - 2.9868 (log (growing season))^2
	N = 82 R^2 = 0.23 Pr = 6.0×10^{-5}
C	Estimation of annual harvest of sunfishes - all reservoir types
	log (sunfish harvest) = 544.3 - 699.76 log (growing season) + 298.16 (log (growing season))^2 - 42.0805 (log (growing season))^3
	N = 71 R^2 = 0.18 Pr = 0.005
D	Estimation of total annual sport fish harvest - all reservoir types
	log (total sport fish harvest) = -0.8104 - 0.2266 log (area) + 0.2090 log (dissolved solids) + 1.1432 log (growing season) - 0.2713 log (age)
	N = 103 R^2 = 0.22 Pr = 7.2×10^{-5}
E	Estimation of total annual sport fish harvest - selected reservoir types (see definition "j")
	log (total sport fish harvest) = -0.3892 - 0.1519 log (area) + 0.2027 log (dissolved solids) + 0.9796 log (growing season) - 0.3055 log (age)
	N = 46 R^2 = 0.69 Pr = $< 10^{-8}$
F	Estimation of annual sport fish harvest in reservoirs located in the Ozark, Smoky, and Cumberland Mountains
	log (total sport fish harvest) = -1.0329 - 0.2637 log (area) + 0.6297 log (thermocline depth) + 1.3329 log (dissolved solids) - 0.1722 log (age)
	N = 11 R^2 = 0.84 Pr = 0.015
G	Estimation of annual sport fish harvest in terms of fish per acre - all reservoir types
	log (number of sport fish harvested per acre) = 0.2894 - 0.3437 log (area) + 1.2296 log (growing season) - 0.3761 log (age)
	N = 103 R^2 = 0.28 Pr = 9.1×10^{-3}
H	Estimation of annual sport fish harvest rate in terms of pounds harvested per angler-hour of effort - all reservoir types
	log (pounds/angler-hour) = -0.7579 + 0.1187 log (area) - 0.1036 log (storage ratio) - 0.1285 log (age)
	N = 103 R^2 = 13 Pr = 0.004
I	Estimation of total annual harvest based on total angler effort - all reservoir types
	log (total sport fish harvest in pounds/acre) = -0.2613 + 0.8935 log (total annual angler-hours/acre)
	N = 103 R^2 = 77 Pr = $< 10^{-8}$
J	Estimation of total annual angler effort - all reservoir types
	log (annual angler days/acre) = -3.3925 + 0.9473 log (area) - 0.1729 log (area)^2 + 0.2387 log (dissolved solids) + 1.1936 log (growing season)
	N = 103 R^2 = 0.32 Pr = 3×10^{-4}
K	Estimation of annual commercial fish harvest - all reservoir types
	log (commercial harvest) = 6.4819 - 0.492 log (mean depth) - 0.231 log (fluctuation) - 0.204 log (storage ratio) - 2.453 (log growing season) + 0.482 log (age)
	N = 45 R^2 = 0.48 Pr = 9.0×10^{-5}
L	Estimation of total annual sport fish harvest - all reservoir types
	log (total sport fish harvest) = -2.0407 + (10 divided by square root of area) + 0.152 log (shore development) + 0.339 log (dissolved solids) + 1.023 log (growing season) - 0.289 log (age)
	N = 116 R^2 = 0.25 Pr = 8.7×10^{-6}
M	Estimation of black bass harvest - all reservoir types
	log (black bass harvest) = -5.8541 - 0.08691 log (area) + 2.9994 log (growing season) - 0.3336 log (age)
	N = 103 R^2 = 0.29 Pr = 8.0×10^{-6}
N	Estimation of sunfish harvest
	log (sunfish harvest) = -4.2043 - 0.4669 log (area) + 2.957 log (growing season) - 0.6178 log (age)
	N = 103 R^2 = 0.45 Pr = 2×10^{-8}

*Equations are based on the U. S. customary system of measures (metric system equations are also available). A table for converting U. S. customary units of measurement to metric (SI) units can be found on page 12. All data transformed to base 10 logarithms. The equations were derived from data on U. S. reservoirs greater than 500 acres in area.

APPENDIX D: ANALYSES OF PESTICIDES, MERCURY, AND LEAD IN FISH

Introduction

1. On 16 March 1976 fish samples for lead, mercury, and pesticides analyses were received from the Tulsa District. The fishes to be analyzed were obtained by electrofishing in Deep Fork Creek at the Arcadia Reservoir site in Oklahoma. Samples were received on ice, given sample numbers for identification, and immediately deep-frozen in a freezer located in the Analytical Water Quality Laboratory of the Department of Biological Sciences, North Texas State University (NTSU).

2. The analyses were carried out during the period from 16 March to 9 April 1976. Determinations of lead and mercury concentrations in fish tissues and viscera and whole fish-pesticide extractions were completed in the NTSU Analytical Water Quality Laboratory. Pesticide analyses in extracted samples were finished in the Trace Organics Laboratory, Institute of Applied Science, NTSU.

Analytical Procedures

3. After an approved operating procedure was designated, fish samples were thawed and separated for pesticides and metals analyses. Specific identification of each fish was verified using key characters given in Eddy.[127]*

Lead analysis

4. Fish tissue and viscera were ground separately with a Lab Line tissue grinder, and 0.5-1.0 g of each sample was placed in 30 ml of deionized water containing 10 ml of a 4:1 mixture of 30-percent hydrogen peroxide (H_2O_2) and 70-percent nitric acid (HNO_3). The samples were then placed in quartz containers and digested with a 1200 W mercury arc UV lamp according to a modification of the procedure outlined by Strickland and Parsons.[128] Samples were then irradiated until clear,

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NOTE: Appendix D was prepared by Dr. Jack A. Stanford and Sandra A. Zimmerman, Analytical Water Quality Laboratory, Department of Biological Sciences, North Texas State University, Denton, Texas.

* Raised numbers refer to similarly numbered items in References at end of main text.

cooled to room temperature, and brought to a known volume. Lead was measured on the Perkin Elmer atomic absorption spectrophotometer (Model 360) utilizing the Perkin Elmer HGA 2100 graphite furnace attachment. The instrument was operated in the concentration mode and calibrated using various dilutions of certified atomic absorption lead standard. Lead in the sample solutions was read directly from the instrument in the peak mode and peak heights were recorded on a Beckman strip-chart recorder. Replicate 50-µl samples were injected into the furnace and the average value was reported along with standard deviations.

Mercury analysis

5. Fish tissue and viscera were ground separately with a Lab Line tissue grinder, and 0.5 g of each was digested in sulfuric acid-permanganate solution to convert all mercury present to inorganic mercury salts. The digested tissues were then placed in a closed vessel containing a solution which reduced the inorganic mercury salts to elemental mercury, according to a modification of the procedures by Hatch and Ott[129] as outlined by Uthe, Armstrong, and Stainton.[130] Nitrogen gas carried the volatilized mercury from the closed vessel to a 10-cm cell mounted in a Perkin Elmer atomic absorption spectrophotometer (Model 360) where absorption was measured. A calibration curve was prepared using various dilutions of certified atomic absorption mercury standard and a reagent blank. Mercury in the sample solutions was read directly from the calibration curve.

Pesticide analysis

6. Whole fish weighing 20-30 g were ground in a high-speed blender for 2 min. Pesticides were extracted with acetonitrile and transferred into petroleum ether according to the procedures outlined in Volume I of Pesticide Analytical Manual.[131]

7. Chemical residues were separated from coextracted material by Florisil column chromatography. Residues were eluted with methylene chloride in hexane (Eluant 1) for recovery of chlordane, pp' DDT, and lindane, and a mixture of methylene chloride, acetonitrile, and hexane (Eluant 2) was used for recovery of dieldrin and heptachlor epoxide in accordance with the methods described by Mills[132] and Mills et al.[133]

8. Pesticides were measured by sample injections into a Tracor Model 560 gas chromatograph equiped with a Ni 63 electron capture detector. The column used was 1.82 m long by 2 mm wide (inside diameter) and was packed with 1.950V210/1.500V17 on Chromosorb W-HP, 110/120 mesh. The carrier gas (90-percent argon - 10-percent methane) was maintained at a flow rate of 20 ml min^{-1}. Injection port temperature was held at 230°C; column oven and detector temperatures were 200 and 300°C, respectively. Sample aliquots of 5 µl were injected into the instrument and pesticide peaks observed on a strip-chart recorder.

9. Pesticide residues were identified by chromatograph comparison with reagent blanks, a combination pesticide standard (Analab), and a chlordane pesticide standard (EPA). Pesticide concentrations in the fishes were quantified by peak area ratios of a spiked sample to whole-fish samples. The spiked sample consisted of 100 mg of ground catfish (*Ictalurus melas*) tissue and known concentrations of pesticide standards.

Quality control

10. Quality control was monitored during metal analyses by periodic checks with standards, replicate runs whenever possible, and quantification of spiked samples. However, replicates on samples for mercury analyses were not possible due to the small amount of sample remaining after the grinding process as compared to the quantity of sample required for the analysis. Standards were run every fourth sample during this analysis to ensure that instrumental parameters were stabilized. Analytical precision for lead, expressed in standard deviations, is given in Table D1. Analytical efficiency was assured by spiking samples with known concentrations of standards followed by determination of percent recovery. A fish sample containing a 0.004 µg g^{-1} Hg was spiked with 0.100 µg g^{-1} Hg standard; 0.096 µg g^{-1} Hg, or 92 percent was recovered. Two fish viscera samples containing 2.88 µg g^{-1} Pb were spiked with 1.00 and 2.00 µg g^{-1} Pb standards; recoveries were 3.88 µg g^{-1} Pb (100 percent) and 4.81 µg g^{-1} Pb (99 percent), respectively. Thus, the analyses were accurate as well as precise.

11. Pesticides were quality controlled by running replicate standards, reagent blanks, and a spiked sample. The spiked sample was

used to determine concentration levels in fish, thereby masking any matrix interference. Sample replicates could not be run due to lack of sufficient fish tissue.

Results

12. Concentrations of metals and pesticides in the fish samples are given in Tables D1 and D2. (Heptachlor epoxide is reported along with the four pesticides requested by the Corps.)

Summary

13. Lead concentrations in the fish samples ranged from 0.20 to 1.12 $\mu g\ g^{-1}$ in tissue and from 0.94 to 2.87 $\mu g\ g^{-1}$ in viscera. Ranges in mercury concentrations were 0.05 to 0.79 $\mu g\ g^{-1}$ in tissue and from below detection to 0.40 $\mu g\ g^{-1}$ in viscera.

14. Nearly all pesticides in standard solutions were present in varying concentrations in the fish samples. However, only values for heptachlor epoxide were reported (range: 0.33 - 2.70 $\mu g\ g^{-1}$), which were quite high, in addition to lindane (range: B/D* - 0.043 $\mu g\ g^{-1}$), chlordane (range: 0.28 - 11.00 $\mu g\ g^{-1}$), dieldrin (range: 0.003 - 0.039 $\mu g\ g^{-1}$), and pp' DDT (range: 0.003 - 0.039 $\mu g\ g^{-1}$). Analyses of the latter four pesticides were specifically requested by the Corps.

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* B/D = below detection limit.

Table D1

Total Lead and Mercury Concentrations
in Deep Fork Creek Fish*

Sample	Species	Number of Specimens	Weight g	Tissue Content Pb	Tissue Content Hg	Viscera Content Pb	Viscera Content Hg
1	*Lepomis cyanallus*	1	16.70	0.30±0.02	0.79	1.76±0.02	0.24
2	" "	1	15.58	0.30±0.01	0.26	2.08±0.07	0.08
3	" "	1	10.09	0.28±0.01	0.11	2.01±0.02	0.22
4	" "	2	12.74	0.22±0.01	0.30	0.93±0.02	0.40
5	" "	2	9.85	0.20±0.02	0.30	1.18±0.07	B/D**
6	" "	2	8.56	0.29±0.02	0.11	1.03±0.02	0.14
7	" "	3	14.70	0.24±0.05	0.41	1.04±0.04	0.08
8	" "	2	12.86	0.28±0.01	0.31	1.92±0.05	B/D
9	" "	2	11.19	0.20±0.04	0.13	0.94±0.01	B/D
10	" "	3	15.24	1.12±0.01	0.05	2.59±0.06	0.05
11	*Ictalurus melas*	1	200.00	0.42±0.01	0.45	2.87±0.02	0.16

NOTE: Analytical precision for lead is given in ± standard deviations.
* Expressed as µg/g wet weight.
** Below detection limit.

Table D2

Pesticide Concentrations in Deep Fork Creek Fish*

Sample	Species	Number of Specimens	Weight g	Pesticide Concentration				Heptachlor Epoxide
				Lindane	Chlordane	Dieldren	pp'DDT	
1	*Lepomis cyanellus*	2	26.69	0.006	2.80	0.070	0.007	0.740
2	"	1	33.01	0.019	10.00	0.140	0.022	1.200
3	"	1	25.07	0.018	2.30	0.110	0.005	1.500
4	"	1	29.60	0.030	6.00	0.215	0.005	2.700
5	"	1	30.95	0.043	11.00	0.200	0.039	2.400
6	"	2	30.30	0.015	6.30	0.080	0.011	1.100
7	"	1	27.07	0.040	10.30	0.150	0.027	2.400
11	*Lepomis macrochirus*	1	13.62	B/D	0.28	0.003	0.003	0.180
12	*Lepomis humilis*	2	12.53	0.003	2.00	0.040	0.008	0.330

* Expressed in $\mu g\ g^{-1}$ wet weight for whole fish analyses.

In accordance with ER 70-2-3, paragraph 6c(1)(b), dated 15 February 1973, a facsimile catalog card in Library of Congress format is reproduced below.

Hall, Ross W

Arcadia Lake water-quality evaluation, by R. W. Hall, Jr., R. H. Plumb, Jr., K. W. Thornton, R. L. Eley, A. S. Lessem, D. L. Robey, B. Loftis [and] P. E. Saunders. Vicksburg, U. S. Army Engineer Waterways Experiment Station, 1977.

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1. Arcadia Lake, Okla. 2. Mathematical models. 3. Nutrients. 4. Water quality. I. Eley, Rex L., joint author. II. Lessem, Allan S., joint author. III. Loftis, Bruce, joint author. IV. Plumb, Russell H., joint author. V. Robey, Donald L., joint author. VI. Saunders, Peter E., joint author. VII. Thornton, Kent W., joint author. VIII. U. S. Army Engineer District, Tulsa. (Series: U. S. Waterways Experiment Station, Vicksburg, Miss. Technical report Y-77-2)

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