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PROCEEDINGS OF THE DeGRAY LAKE SYMPOSIUM

by

R. H. Kennedy, Joe Nix, Editors

Environmental Laboratory

DEPARTMENT OF THE ARMY
Waterways Experiment Station, Corps of Engineers
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The Environmental and Water Quality Operational Studies (EWQOS) Program, a major water quality research program funded by the Office of the Chief, US Army Corps of Engineers (CE), and conducted by the US Army Engineer Waterways Experiment Station, was designed to assess the relations between design, construction, and operation of Civil Works projects and exhibited water quality characteristics, and to develop and demonstrate technological means for ameliorating water quality problems in a manner compatible with project purposes. An important aspect of the EWQOS Program was the conduct of comprehensive studies of water quality processes in reservoirs. These studies involved both long-term and short-term limnological evaluations of four characteristic CE reservoirs.

DeGray Lake, located in south-central Arkansas, was studied during the period 1967-1982. Research activities ranged from evaluations of geologic and hydrologic processes, to studies of complex biological, chemical, and physical processes influencing water

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quality, to the development of water quality models. A symposium was held to provide a forum for the exchange of information and ideas and an opportunity to synthesize these data in ways that will lead to a greater understanding of DeGray Lake, in particular, and of reservoir limnology, in general. Information gathered as a result of these studies and discussed herein will broaden the understanding upon which sound strategies for managing these valuable resources must be based. This report documents information presented at the symposium.

PREFACE

A symposium, held at DeGray Lodge, DeGray State Park, Arkadelphia, Ark., featured the results of research efforts at DeGray Lake, a US Army Corps of Engineer reservoir located in south-central Arkansas, and Lake Chicot, an oxbow lake located in eastern Arkansas. The DeGray Lake portion of the symposium was sponsored by the US Army Engineer Waterways Experiment Station (WES) as part of Task VIIA of the Environmental and Water Quality Operational Studies (EWQOS) Program. EWQOS was sponsored by the Office, Chief of Engineers (OCE), US Army. The OCE Technical Monitors for EWQOS were Mr. Earl Eiker, Dr. John Bushman, and Mr. James L. Gottesman. Program Manager for EWQOS was Dr. J. L. Mahloch, WES.

The purpose of the EWQOS-sponsored portion of the symposium was to provide the opportunity for individuals who participated in water quality research at DeGray Lake to present and discuss their findings. Research areas ranged from the hydrology and geology of the Caddo River basin and the physical, chemical, and biological characterization of DeGray Lake, to numerical modeling. Together, the results of these efforts provide a detailed description of water and environmental quality in DeGray Lake and the Caddo River basin. This volume is a textual presentation of these results and findings. Papers are reproduced in the form in which they were received from the authors.

This report was edited by Dr. Robert H. Kennedy, Aquatic Processes and Effects Group (APEG), Ecosystem Research and Simulation Division (ERSD), Environmental Laboratory (EL), WES, and Dr. Joe Nix, Department of Chemistry, Ouachita Baptist University, Arkadelphia, Ark. Dr. John Harrison was Chief of EL, Mr. Donald L. Robey was Chief of ERSD, and Dr. Thomas L. Hart was Chief of APEG. Ms. Jessica S. Ruff of the WES Information Products Division was responsible for the preparation of the final manuscript.

COL Allen F. Grum, USA, was the previous Director of WES. COL Dwayne G. Lee, CE, is the present Commander and Director. Dr. Robert W. Whalin is Technical Director.

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INTRODUCTION: DeGRAY LAKE SYMPOSIUM

by

Robert H. Kennedy and Joseph H. Carroll[1/]

Reservoirs, whether constructed for flood control, water supply, or power production, provide valuable environmental and recreational resources of ever-increasing importance. In many areas, particularly those with few natural lakes, reservoirs serve as the primary source for water-based recreation. However, while man's needs for recreation have increased, the quality of these resources has degraded, prompting increased public awareness of environmental issues and continued efforts to ameliorate water quality problems. As with natural lakes, these problems often stem from poor management practices. Since lakes are located in depressions surrounded by relatively large drainage areas, material and nutrient budgets for lakes are dictated by processes which affect both the quantity and quality of material exports from the watershed. Excessive material exports from the watershed, therefore, lead to excessive inputs to the lake or reservoir. These inputs often lead to undesirable water quality conditions, including nuisance algal and macrophyte growth, reduced water clarity, reduced dissolved oxygen concentrations in bottom waters, taste and odor problems, and increased sedimentation.

Much of what we understand about processes influencing water quality and the adverse impacts of excessive material and nutrient inputs is based primarily on extensive studies of natural lake systems. However, a number of substantive differences exist between natural lakes and reservoirs, suggesting the need to carefully reevaluate this understanding as it relates to the development of management strategies for reservoirs. While biological, chemical, and physical processes are the same for both reservoirs and natural lakes, differences of magnitude are

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1/ Research Limnologist and Aquatic Biologist, respectively. Ecosystem Research and Simulation Division, Environmental Laboratory, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

clearly apparent (Thornton et al. 1981). In general, reservoirs are larger, deeper, and located in larger drainage areas than are natural lakes. Unlike natural lakes which often receive water and material inputs from relatively diffuse sources, inputs to reservoirs commonly occur via a single, large tributary entering at a point distant from the point of discharge. These differences lead to higher nutrient and sediment loading rates and shorter residence times for reservoirs. The increased importance of flow and the fact that reservoirs are often long and relatively narrow also lead to the establishment of pronounced longitudinal gradients in water quality. Finally, and perhaps most significantly, reservoirs, which are designed to control and/or modify highly variable flows, exhibit variable water levels and discharge from variable depths in the water column.

The Environmental and Water Quality Operational Studies (EWQOS) Program, a major water quality research program funded by the Office of the Chief, US Army Corps of Engineers (CE), and conducted by the US Army Engineer Waterways Experiment Station, was designed to assess relations between the design, construction, and operation of Civil Works projects and exhibited water quality characteristics, and to develop and demonstrate technological means for ameliorating water quality problems in a manner compatible with project purposes (Keeley et al. 1978). An important aspect of the EWQOS Program was the conduct of comprehensive studies of water quality processes in reservoirs. These studies involved both long- and short-term limnological evaluations of four characteristic CE reservoirs. These included: Redrock Lake, a main stem flood control impoundment located on the Des Moines River, which receives excessive nutrient and sediment loads; West Point Lake, a large, morphologically complex hydropower project located on the Chattahoochee River in westcentral Georgia; Eau Galle Lake, a small flood control project in west-central Wisconsin; and DeGray Lake, a large, deep hydroelectric reservoir on the Caddo River in south-central Arkansas. Of these studies, those at DeGray Lake were the most comprehensive and of the longest duration (1967-1982).

Construction of DeGray Lake began in 1967, and power pool was reached in 1971. The lake was formed by impoundment of the Caddo River near Arkadelphia, Ark., approximately 12.7 km upstream from the confluence of the Caddo and Ouachita Rivers. Flooding of the deeply cut Caddo River valley resulted in the formation of a deep (maximum depth of approximately 60 m near the dam), long (32 km), and narrow lake having a relatively complex shoreline (Table 1). Major features of the lake and dam include numerous coves and islands, two major embayments (Brushy Creek embayment to the south and Big Hill Creek embayment to the north), a deeply cut submerged river channel, and a selective withdrawal outlet structure capable of discharging design flows from any of four gate depths. The Caddo River, which originates in the Ouachita Mountains to the west and north, enters the lake's extreme western end, and riverine flows are frequently observed in the upper one-third of the lake, particularly during high-flow seasons. Although exhibiting severe water quality conditions during early years due to the inundation and rapid decomposition of large stands of timber, the lake presently experiences few water quality problems.

Table 1

Physical Characteristics of DeGray Lake

Characteristic	Value*
Elevation, m NGVD	124.4
Surface area, km^2	54.2
Volume, 10^6 m^3	808
Maximum depth, m	60
Mean depth, m	14.9
Length, km	32
Shoreline length, km	333
Shoreline development ratio	12.8
Drainage area, km^2	1,173
Theoretical hydraulic residence time, years	1.4

* Based on a pool elevation of 124.4 m NGVD.

Research efforts begun prior to impoundment included surveys of water quality conditions along the reach of the Caddo River scheduled for inundation. Following impoundment, studies continued, often utilizing the same sampling stations included during previous studies. The existence of these baseline data satisfied one of the primary criteria used for selecting study sites for the EWQOS studies initiated in 1978 and concluded in 1982. Major sampling stations for these studies included: Station 1, located in a deep, relatively isolated portion of the lake immediately upstream from the dam; Station 4, centered over the old river channel in the more open, main portion of the lake; Station 10, upstream from the two major embayments; and Station 12, located a short distance downstream from the point at which the Caddo River enters the lake (Figure 1).

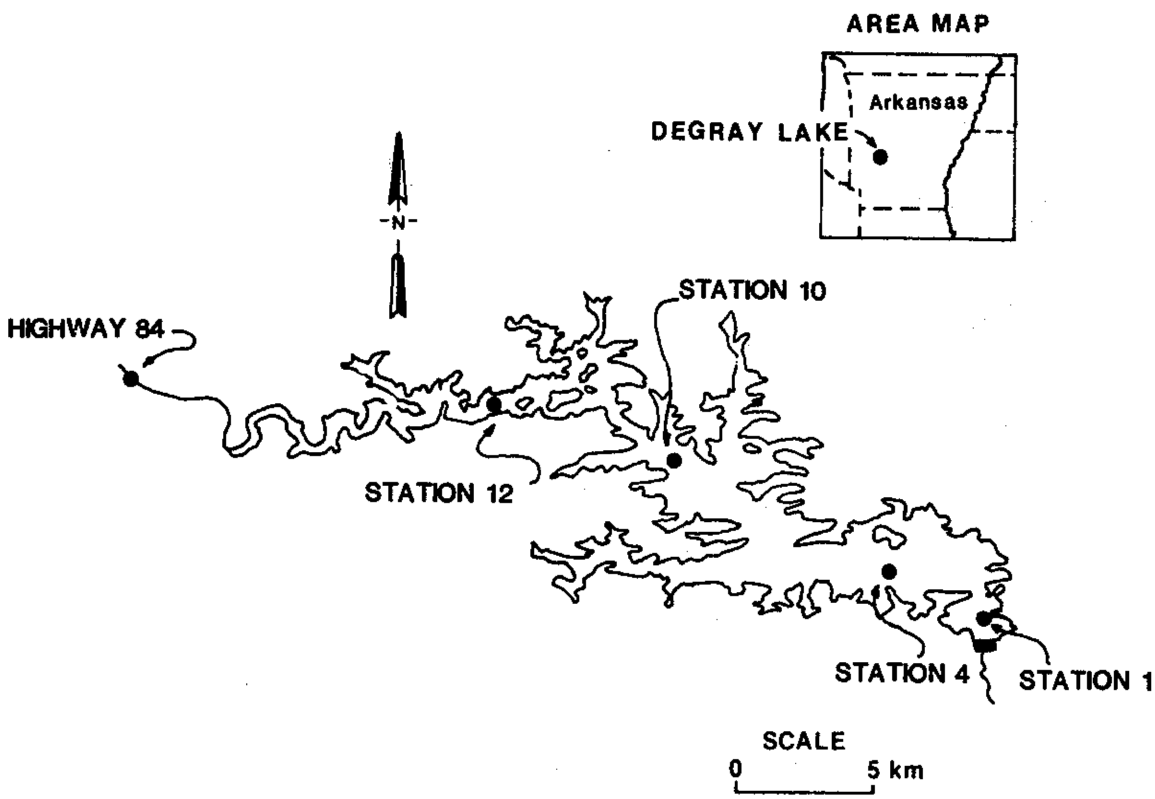

Figure 1. Location of sampling stations for many of the studies conducted at DeGray Lake

Studies conducted during the period 1967 through 1982 involved a number of researchers of varied interests and expertise, representing several different organizations. Research activities ranged from evaluations of geologic and hydrologic conditions and studies of complex biological, chemical, and physical processes influencing water quality, to numerical modeling. The purpose of this symposium is to provide a forum for the exchange of information and ideas, and an opportunity to synthesize these data in ways that will lead to a greater understanding of DeGray Lake, in particular, and of reservoir limnology, in general. Information gathered as a result of these studies and discussed here will broaden the understanding upon which sound strategies for managing these valuable resources must be based.

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GENERAL GEOLOGY AND MINERAL RESOURCES OF THE CADDO RIVER WATERSHED

by

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ABSTRACT

The Caddo River watershed is located in a picturesque area typified by jagged peaks, hogbacks, and broad rather hummocky basins comprising approximately 480 square miles (1240 sq km) in west-central Arkansas. It is located primarily in the Ouachita Mountains with a small portion in the Gulf Coastal Plain.

The exposed sedimentary bedrock in the Ouachita Mountain portions of the area is shale, sandstone, chert, novaculite, limestone, conglomerate, and tuff. These rocks are of deep-water marine origin and exceed 30,000 feet (9100 m) in thickness. They are of Paleozoic age ranging from Early Ordovician (about 490 million years) to Middle Pennsylvanian (about 290 million years). The Ouachita Mountains were formed when these rocks were uplifted by northerly directed compressive forces during late Paleozoic time. This deformation caused extensive thrust faults and complex fold systems. Some rocks were subjected to very low-rank metamorphism and related hydrothermal events as evidenced by locally pervasive shear planes, recrystallization, and numerous milky quartz veins. Following the formation of the Ouachita Mountains there was a long period of erosion and minor arching with thousands of feet of rock being denudated from the area. In the southeast portions of the watershed the Paleozoic strata are overlain with a major unconformity by the slightly tilted mostly shallow-marine Late Cretaceous rocks. Covering the bedrock at places are unconsolidated, mostly flat-lying terrace, alluvial and colluvial deposits composed of clay, sand, gravel and cobbles of Quaternary age.

Mineral resources are quite varied in the watershed with barite, manganese, rock aggregate, slate and quartz crystals comprising most of the past production. Numerous additional mineral resources could have economic potential. Surface and groundwater are very important resources in the watershed.

INTRODUCTION

Location

The Caddo River watershed covers an area of approximately 480 square miles (1240 sq km) in west-central Arkansas which includes portions of Montgomery, Pike, Hot Spring, Clark and Garland Counties. It is located in the central and southern Ouachita Mountains with a small portion at the east end extending into the Gulf Coastal Plain.

Physiography

The Ouachita Mountains consist of several mountain ranges and broad basins that extend from Little Rock, Arkansas westward to Atoka, Oklahoma. The mountains are long narrow ridges, many forming hogbacks, with steep slopes and sharp rather straight and even crests. The mostly mature trellis drainage patterns have primarily developed in the tilted alternating resistant and weak Paleozoic strata. The typically parallel subsequent streams (Caddo and tributaries) flow in fairly deep valleys separated by rather high topography, and exhibit some youthful tranversing V-shaped water gaps. The Fall Line separates the older deformed strata of the Ouachita Mountains and the overlapping Cretaceous marine deposits and other more recent poorly consolidated rocks of the Gulf Coastal Plain. Surfaces of planation are present inland from this ancient boundary indicating that the Ouachita Mountains have been deeply eroded since they were formed, but it is unlikely that they were ever completely covered by these seas. Low rolling hills with undulating narrow valleys typify the topography in the Gulf Coastal Plain. The gently southward dipping Cretaceous strata form a broad homoclinal feature with sluggish consequent trunk streams having a dendritic drainage pattern. Near the Ouachita River junction the Caddo is in an old age stream cycle with negligible downcutting and characterized by a low broad flood plain with meanders and oxbow lakes.

The principal physiographic subdivisions of the Ouachita Mountains in the Caddo River watershed are the Athens Plateau, Trap Mountains, Cossatot Mountains, Caddo Mountains, Crystal Mountains, Mazarn Basin and Caddo Basin (Fig. 1). The total relief is approximately 2000 feet (610 m) ranging from 180 feet (55 m) (sea level) at the junction of the Caddo and Ouachita Rivers in Clark County to over 2200 feet (671 m) in the Caddo Mountains.

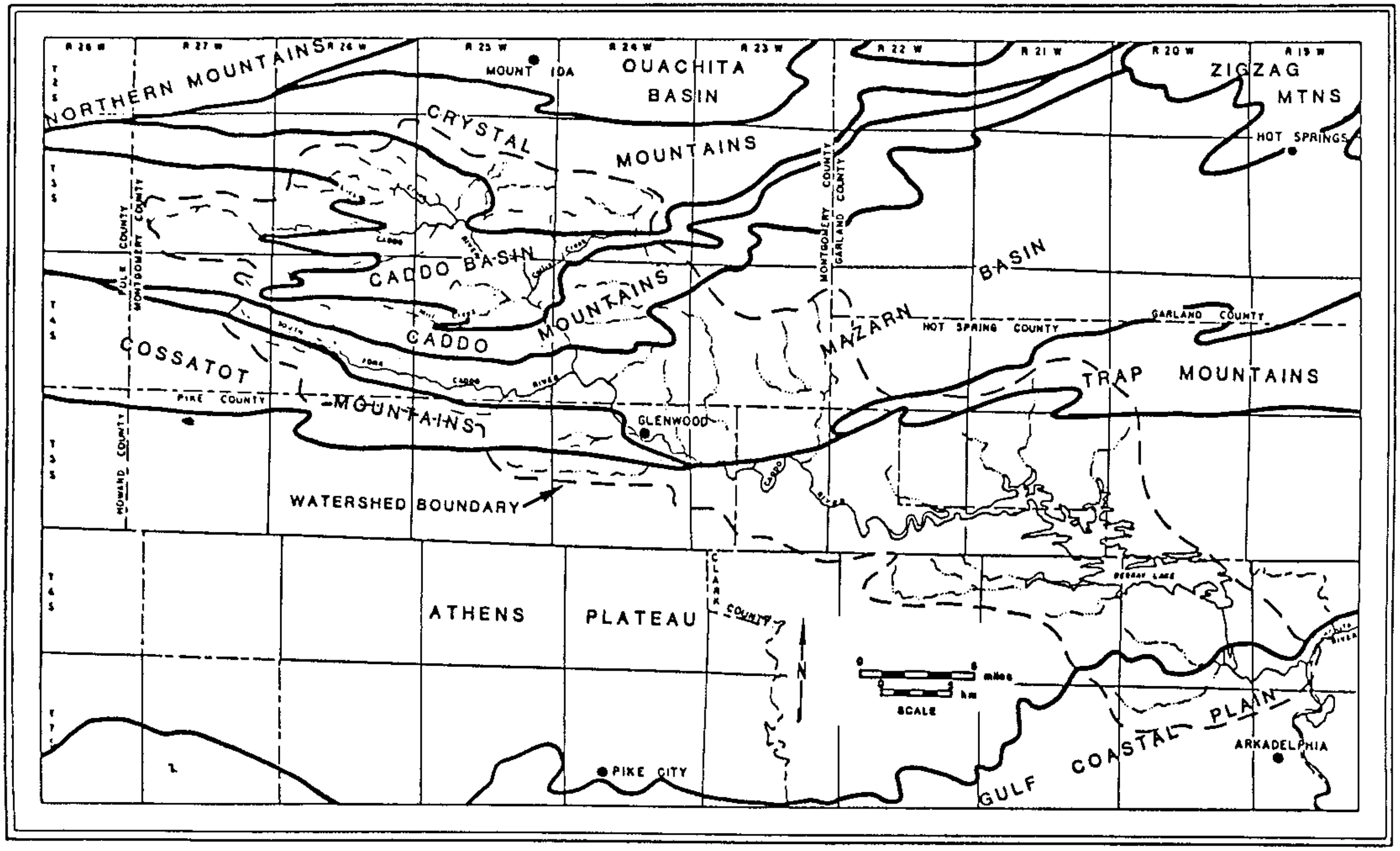

FIGURE 1. PHYSIOGRAPHIC SUBDIVISIONS OF THE CADDO RIVER WATERSHED AND VICINITY

Acknowledgments

We gratefully acknowledge the cooperation of Boyd R. Haley, U. S. Geological Survey; Drew F. Holbrook, consulting geologist; Gary Sick, U. S. Forest Service; Cynthia Shackelford, Arkansas Department of Pollution Control and Ecology; and John D. McFarland III and George W. Colton, Arkansas Geological Commission for their assistance in the preparation of this paper.

GENERAL GEOLOGY

All the rocks in the Caddo River watershed are of sedimentary origin with the exception of a small igneous dike. All formations in the Ouachita Mountains of Arkansas are present in the watershed and range from Early Ordovician to Middle Pennsylvanian age. They were originally deposited as nearly flat layers of mud, sand, gravel, marl, lime, volcanic ash and silica in the marine waters of an ancient deep basin that once occupied the region. With the load and weight of the overlying sediments they were subsequently converted to shale, sandstone, conglomerate, limestone, tuff, chert and novaculite. These rocks were then subjected to intense compressive forces in Late Paleozoic time that transported them towards the north causing them to bend and fold and, in many places, to rupture and fault with ultimately the region being uplifted and forming extensive mountain ranges. This deformation, called the Ouachita orogeny, caused intense pressures and elevated temperatures which slightly metamorphosed these rocks in places, changing some shale to slate and sandstone to quartzite. The Paleozoic rocks exceed 50,000 feet (15,200 m) in thickness in the Ouachita Mountains, but only the lower 30,000 feet (9100 m) are exposed in the watershed. The oldest strata are exposed in the northern portions and the youngest in the southern portions of the watershed.

The uplift produced prominent east-west folds and large thrust faults in the strata. Almost without exception the present land forms are a reflection of the underlying bedrock. The softer less resistant shale, limestone and impure sandstone are more susceptible to erosion and form most of the basins, valley floors, and lower hills. The harder more resistant novaculite, chert, and relatively pure sandstone form the mountains, ridges and peaks.

Subsequent to the Ouachita orogeny the region has been eroded and dissected with minor arching and extensional faulting. Some sizable igneous intrusions, notably in early Late Cretaceous time, occur in adjoining areas at Magnet Cove and Murfreesboro. In Late Cretaceous and possibly Early Tertiary time shallow warm seas lapped upon the southern portions of the area. The gently dipping clay, sand, gravel, marl and chalk of Late Cretaceous age represent the remnants of these deposits.

During Pleistocene and Recent times (Quaternary), the older rocks in the area were further eroded. Terrace, alluvial, and colluvial deposits represent some of the products of these climatically related cycles.

Formation Descriptions

Based on their lithologic character, stratigraphic position, and meager fossil content, 19 formations of the Ordovician, Silurian, Devonian, Mississippian, Pennsylvanian, and Cretaceous Systems (Fig. 2) are grouped into seven units on the generalized geologic map (Fig. 3). It was not feasible to show the Quaternary deposits on this map.

Collier Shale

The Collier Shale is the oldest formation exposed in the Ouachita Mountains of Arkansas. It was named by Purdue (1909) and further defined by Miser and Purdue (1929) for exposures along the headwaters of Collier Creek in the Crystal Mountains. The thickness of the Collier is likely over 1000 feet (365 m) but the base is not exposed. It consists of graphitic to talcose shale with considerable amounts of interbedded, dense to very fine-grained, to sandy, sometimes pellatoidal, or conglomeratic, bluish-gray limestone. There are minor quantities of bluish-black chert, gray calcareous siltstone, fine-grained quartzose sandstone, conglomerate and boulder-bearing breccia intervals. Repetski and Ethington (1977) identified conodont microfossils from limestones in the formation and indicated they were of Early Ordovician age. Previously the Collier had been tentatively assigned to the older Cambrian Period.

Some road aggregates have been obtained from small pits in the Collier for local uses. Limited

Figure 2. – Stratigraphic section of rocks exposed in the Caddo River Watershed

Age	Formation	Maximum Thickness feet (meters)	Description
Quaternary	Alluvial Deposits	20′ (6 m)	clay, silt, sand, gravel, and cobbles
	Terrace Deposits	15′ (4.6 m)	sand, gravel, and some clay
Cretaceous	Nacatoch Sand	200′ (61 m)	sand and some clay and gravel
	Saratoga Chalk	60′ (18 m)	chalk and some marl
	Marlbrook Marl	80′ (24 m)	chalky marl and some sand
	Ozan Formation	125′ (38 m)	marl, sand, or sandy marl
	Brownstown Marl	175′ (53 m)	marl, clay, sand, and some gravel
	Igneous Intrusive		ultramafic dike
Pennsylvanian	Atoka Formation	7500′ (2287 m)	shale, siltstone, and sandstone
	Johns Valley Shale	1500′ (457 m)	shale, sandstone, and some chert
	Jackfork Sandstone	6000′ (1830 m)	sandstone, siltstone, and shale
Mississippian	Stanley Shale	11,000′ (3355 m)	shale, siltstone, sandstone, and some chert and volcanic tuff
Devonian	Arkansas Novaculite	900′ (274 m)	novaculite, chert, shale, and some conglomerate
Silurian	Missouri Mtn. Shale	300′ (91 m)	shale with minor sandstone and conglomerate
	Blaylock Sandstone	1000′ (305 m)	sandstone, siltstone, and shale
Ordovician	Polk Creek Shale	175′ (53 m)	shale and some chert and limestone
	Bigfork Chert	750′ (229 m)	chert, limestone, and some shale and siltstone
	Womble Shale	1900′ (579 m)	shale, limestone, and some sandstone, chert and conglomerate
	Blakely Sandstone	700′ (213 m)	shale, sandstone, and some conglomerate and limestone
	Mazarn Shale	3000′ (915 m)	banded shale with some sandstone and limestone
	Crystal Mtn. Sandstone	850′ (259 m)	sandstone, shale, and some limestone and conglomerate
	Collier Shale	1000′ (305 m)	shale, limestone, and some chert and conglomerate

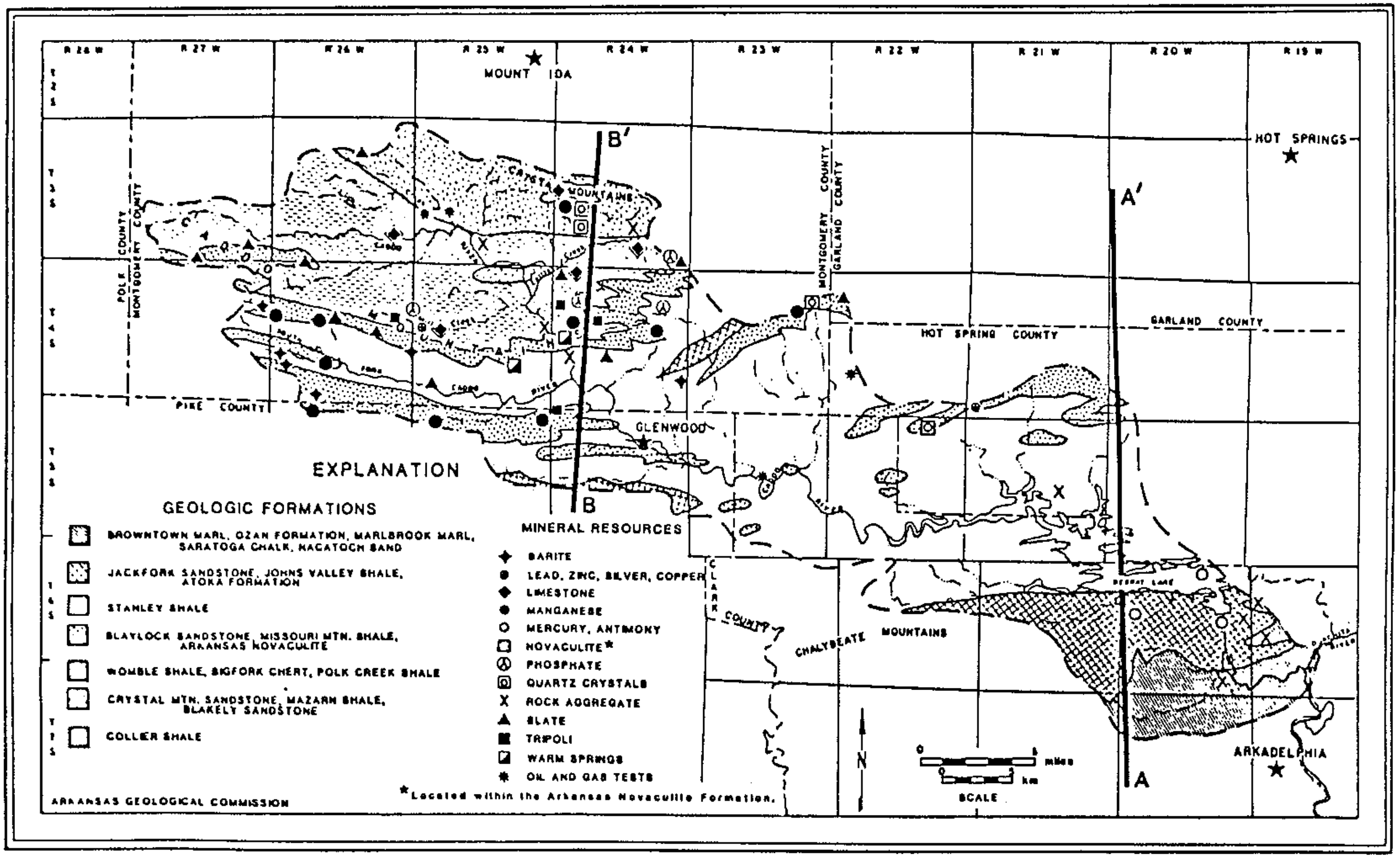

FIGURE 3. GENERALIZED GEOLOGIC AND MINERAL RESOURCES MAP OF THE CADDO RIVER WATERSHED

reserves of limestone for agricultural purposes are also available.

Crystal Mountain Sandstone

The Crystal Mountain Sandstone overlies the Collier Shale and the name was proposed by Purdue (1909) for the massive sandstones containing clear quartz crystals that form the Crystal Mountains.

The Crystal Mountain Sandstone varies from approximately 550 to 850 feet (168 – 259 m) in thickness. The formation is composed of very massive to thin-bedded quartzose, calcareous, light gray to brown, sometimes conglomeratic, medium-grained sandstones. Interbedded black, gray to buff shales are present and are more common in the upper part of the formation. Intervals of thin, dense, very fine-grained to sandy, bluish-gray limestone and calcareous, gray conglomerate and boulder-bearing breccia occur in the lower portions of the formation. The massive sandstone interval is quite resistant and forms the tall ridges and peaks in the Crystal Mountains. Few fossils have been reported from rocks in the formation.

Large reserves of rock aggregate are potentially available from the massive sandstone intervals. Significant amounts of clear quartz crystals are mined from veins dissecting the formation at several sites in and adjacent to the watershed. Some small abandoned manganese mines and prospects also occur in the Crystal Mountain Sandstone.

Mazarn Shale

The Mazarn Shale was named from Mazarn Creek northeast of Norman, Arkansas,in the Crystal Mountains by Miser (1917) and later mapped by Miser and Purdue (1929). The Mazarn has a total thickness of about 2500 to 3000 feet (762 – 915 m) and consists mostly of black shale with some interbedded olive-green shale and silty shale, thinly laminated gray siltstone, brown quartzose sandstone, and dense blue-gray limestone. The alternating black and olive-green shale layers, often with cross-cutting cleavage, give it a banded appearance. Worm burrows and other trace fossils occur in the siltstones, and some conodonts and graptolites are found in the limestones and shales. The Mazarn typically forms fairly broad valleys with some noticeable low ridges. There have been minor quantities of commercial slate and also shale for rural road construction obtained from small pits and quarries in the Mazarn. Some of the limestone intervals could have potential for agricultural purposes.

Blakely Sandstone

The Blakely Sandstone was named from Blakely Mountain north of Hot Springs by Miser (1917) and later mapped by Purdue and Miser (1923). It ranges from 400 to 700 feet (122 – 213 m) in thickness and consists of interbedded thin to fairly massive, fine to medium-grained, sometimes silty or calcareous, quartzose brownish gray sandstones and black to green shales. A gray sandy limestone occurs in places in the upper part of the Blakely and a shale sequence ranging in thickness from 100 to 200 feet (30 – 61 m) is near the middle. Graptolite impressions are present in some of the shales. The Blakely forms tall ridges with small, narrow intervening valleys. Small amounts of quartz crystals have been mined at a few localities near Norman from veins dissecting the sandstones. The massive or thicker packets of sandstones are all suitable for rock aggregate.

Womble Shale

The Womble Shale was named for outcrops near the town of Womble (now called Norman), Arkansas,by Miser (1917). It likely ranges from about 1250 to 1900 feet (386 – 579 m) in thickness and consists mostly of black shale with intervals of dense, bluish-gray limestone and calcareous siltstone. Minor amounts of gray chert, fine-grained quartzose sandstone and conglomerate are also present. Graptolite fossil impressions occur rather commonly in the shale. Repetski and Ethington (1977) describe conodonts in the limestones. Springs issue from joints in the limestone intervals at a number of places. The Womble characteristically forms low, fairly broad valleys with minor east-west trending, rather irregular hills. Recently several companies have tested rocks of the Womble and other formations for base metal deposits. The limestones have been prospected on a small scale for agricultural limestone and decorative black marble. Small amounts of road aggregate have been mined from the Womble.

Bigfork Chert

The Bigfork Chert was named for extensive exposures near Bigfork, Arkansas,immediately west of the Caddo watershed by Purdue (1909). It ranges in thickness from about 550 feet (168 m) in the north to 750 feet (229 m) in the south. It is composed primarily of thin-bedded, highly fractured, gray chert, dense gray limestone, calcareous siltstone and some thin, interbedded black shale. Irregular-shaped "potato" hills are produced by the weathering of the Bigfork. Intense fracturing creates good aquifer conditions in the formation throughout most of the Ouachita Mountains. Some occurrences of the aluminum phosphate mineral wavellite (cats-eye) and variscite have been found in small veins. Because of its finely broken nature, the Bigfork Chert has considerable potential for local supplies of rock aggregate.

Polk Creek Shale

The Polk Creek Shale was named by Purdue (1909) for outcrops along Polk Creek in the Caddo Mountains. The Polk Creek ranges from 110 to 175 feet (34 – 53 m) in thickness. It is a black sooty shale, with some very thin gray chert and a few thin blue-gray limestone intervals. Upper Ordovician graptolite fossils are very common in the formation. It is mostly exposed in narrow strips in valleys but occasionally outcrops on the mountain slopes. There are several old prospects in the sooty shales which likely were unsuccessful ventures for various precious elements.

Blaylock Sandstone

The Blaylock Sandstone was named from Blaylock Mountain on the Little Missouri River by Purdue (1909). It lies between the Missouri Mountain Shale and the Polk Creek Shale and is approximately 1000 feet (305 m) thick in the Cossatot Mountains but it thins dramatically to the north where it is either absent or less than 20 feet (6 m) thick. It consists of alternating thin brownish gray, very fine-grained, silty sandstone and gray shale layers. It typically forms narrow ridges or jagged strips on mountain slopes. A small quantity of sandstone has been used for local building stone. There are a few old misdirected prospects in the formation.

Missouri Mountain Shale

The Missouri Mountain Shale was named by Purdue (1909) for exposures in the Missouri Mountains. The Missouri Mountain Shale lies between the Arkansas Novaculite and the Blaylock Sandstone, or when the Blaylock is absent the Polk Creek Shale. It is typically a red, green or buff shale or slate with minor novaculite and conglomerate layers. It generally is poorly exposed and forms narrow valleys or slopes. The Missouri Mountain is about 50 feet (15 m) thick in the south, reaches a maximum of 300 feet (91 m) in the west-central part, and is between 175 and 200 feet (53 – 61 m) along the east-central portions of the area. It has previously been quarried for ornamental red, green, olive and buff colored slates in the northwestern part of the area.

Arkansas Novaculite

No type locality has been assigned for the highly distinctive Arkansas Novaculite.The exposures along the roadcut adjacent to the Caddo River at Caddo Gap have served as a classic example of the typical development of this formation (Fig. 4). The Arkansas Novaculite consists predominantly of white to light gray novaculite with lesser amounts of gray chert, olive-green to black shale, conglomerate and sandstone. It is about 850 to 900 feet (259 – 274 m) thick in the south, about 650 to 800 feet (198 – 244 m) thick in the central part, and 350 to 400 feet (107 – 122 m) thick along the northeastern part of the watershed. Novaculite is a hard dense rock composed essentially of silica, usually white to light gray in color and resembling unglazed porcelain in general appearance and texture. The formation is divisible into three distinct Divisions throughout most of the area: a Lower Division of massive white novaculite, with minor shale and conglomerate; a Middle Division of dark chert and novaculite interbedded with olive green to black shale and some conglomerate; and an Upper Division of white, often tripolitic and calcareous, white, thin-bedded novaculite. The Arkansas Novaculite is extremely resistant and forms tall, sharp-crested ridges along east-west belts. Novaculite is probably best known as a raw material for making whetstones. There are no active whetstone operations in the area, but suitable materials are undoubtedly present. Tripoli prospects occur in the Upper Division at several localities and abandoned manganese mines

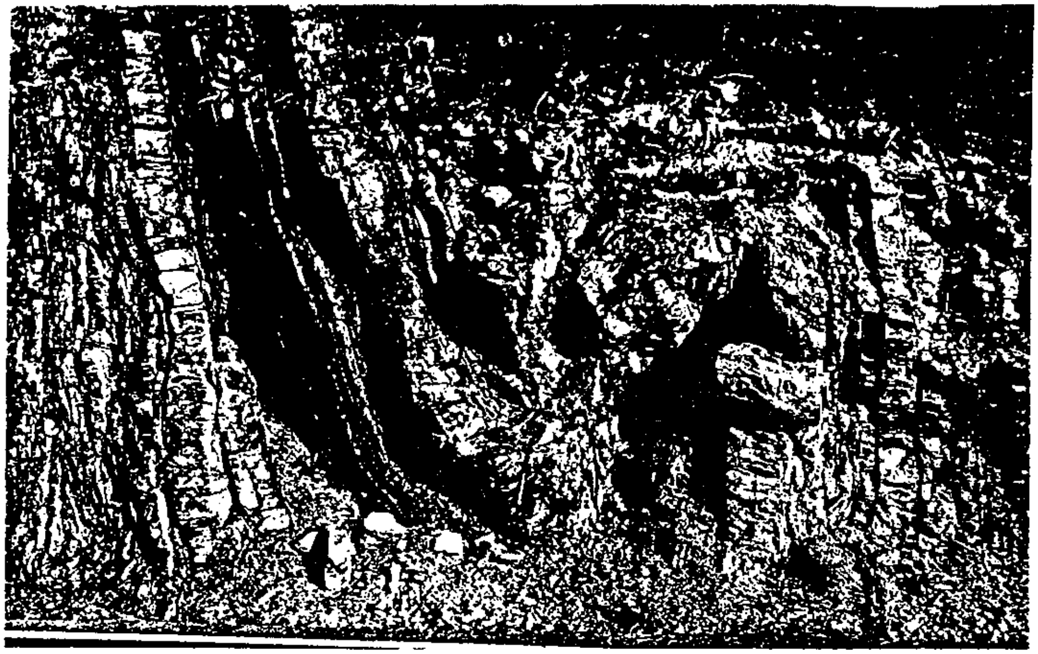

Figure 4. — Complexly folded intervals of dense novaculite and some thin shale in the Lower Division of the Arkansas Novaculite at the north end of the Caddo Gap section on Arkansas Highways 8 and 27.

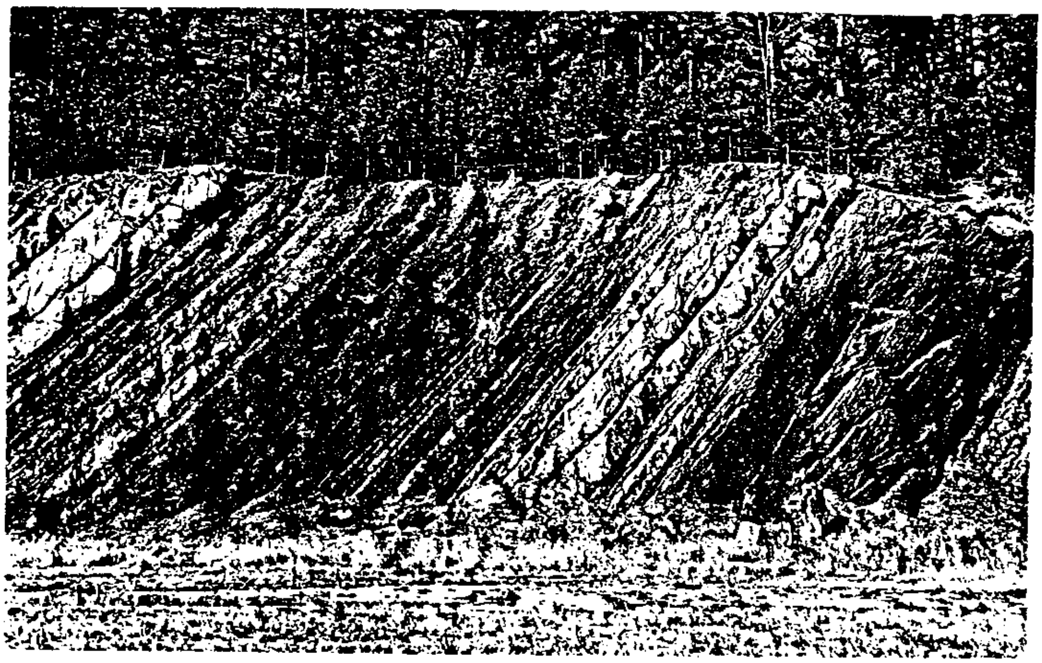

Figure 5. — Steeply dipping interbedded quartzose sandstones (light) and shales (dark) in the upper Jackfork Sandstone near the middle of the De Gray Lake spillway cut.

and prospects are present in both the Lower and (less often) the Upper Divisions. Some quantities of copper, cobalt, nickel, and lithium are often associated with the manganese ore. In recent years the unusual iron phosphate minerals that occur mostly in association with the manganese occurrences have been prospected by mineral collectors. Rock aggregate is readily available from various portions of the formation.

Stanley Shale

The Stanley Shale was named by Taff (1902) for exposures near the town of Stanley (formerly called Standley) in Pushmataha County, Oklahoma. The Stanley Shale has been elevated to a Group in Oklahoma and in some areas in Arkansas, but mainly for the sake of simplicity, these subdivisions are not used in this report.

The Stanley Shale has a maximum thickness of about 11,000 feet (3355 m) and is composed mostly of black to brownish-green shale with lesser quantities of thin to massive, fine-grained, silty gray to brown sandstone. Some thin black siliceous shales and cherts occur in parts of the Stanley and are useful in subdividing the formation. Minor conglomerate and tuff beds are present, mostly in the lower part. Cone-in-cone and other mostly calcareous siltstone concretions typically occur throughout the formation. Some conodonts have been found in the shales and cherts and a prolific invertebrate fauna has been collected from erratic blocks at a few locations in the Stanley (mostly in the northern Ouachita Mountains). Plant fossils are present in some intervals and are useful for age determinations and correlations.

The sandstones decompose upon weathering and form rather low ridges. Thus the Stanley typically forms valleys with a series of low hills. It is the primary bedrock for the Mazarn Basin and much of the Athens Plateau.

Significant barite has been produced from deposits in the lower Stanley in the area. Barite mining is expected to resume at a future date by the Milchem Company from their deposit at Fancy Hill. Slate used for roofing granules is currently mined by Bird and Son, Inc.,from sheared shales in the Stanley near Caddo Gap and unlimited reserves are available. There are several small pits in the shales, siltstones, and sandstones of the Stanley which are used mostly as local sources of rock aggregate. Significant mercury and antimony deposits occur in the Stanely Shale to the south and southwest of the watershed. There is one small antimony prospect known in the watershed on the east end of De Gray Lake.

Jackfork Sandstone

The Jackfork Sandstone was first used by Taff (1902) to designate a sandstone-shale sequence on Jackfork Mountain north of Daisy in Atoka County, Oklahoma. In the Ouachita Mountains of Oklahoma and in some areas in Arkansas, the Jackfork has been elevated to a Group and subdivided into a number of formations, but it will be retained as a formation in this report.

The Jackfork Sandstone of Early Pennsylvanian age has a total thickness of about 6000 feet (1830 m) along the southern part of the Athens Plateau. A classic section of the upper Jackfork Sandstone occurs at the Lake De Gray spillway (Fig. 5). Approximately the lower 2500 feet (762 m) of the Jackfork Sandstone is exposed in the Pigeon Roost Mountain area of the Mazarn Basin.

The Jackfork is composed of thin to massive, light-brown, fine-grained, quartzitic gray sandstone, blue-black to brown siltstone and interbedded gray-black shale. Some of the sandstones contain a few thin conglomeratic layers with pebbles that consist of rounded chert and meta-quartzite. Many of the siltstones contain coalified plant fragments. A few intervebrate fossil fragments and molds occur in the sandstone and conglomerate beds. The massive sandstones are fairly resistant to weathering and typically form ridges with many rock exposures. Little mineraliza-has been noted in the Jackfork in this area. But the possibility exists of discovering deposits such as mercury that occur in veins in the Stanley Shale and Jackfork Sandstone to the southwest near Lake Greeson. Small quantities of mercury have been noted near De Gray Lake. A massive sandstone interval in the upper Jackfork is worked sporadically for commercial aggregate east of De Gray Dam. Several units in the Jackfork have potential for commercial aggregate.

Johns Valley Shale

The Johns Valley Shale was named by Ulrich (1927) for exposures that Taff (1901) had identified as Caney Shale along Johns Valley in Pushmataha County, Oklahoma. The Johns Valley

Shale has received much attention by geologists because of the enormous quantities and, in some cases, vast sizes of erratics derived from Arbuckle and Ozark foreland facies mostly in the frontal Ouachita Mountains of Oklahoma and portions of Arkansas. It is the general consensus of opinion that the erratics were derived by slumping from submarine scarps that flanked the north side of the unstable Ouachita trough.

Walthall (1967) first described the Johns Valley Shale in the southern Athens Plateau area of Arkansas. Stone, Haley and Viele (1973), Haley et al. (1976), Gordon and Stone (1977), and Stone, McFarland and Haley (1981) further defined and also expanded the upper boundary of the formation in this area.

The formation in the watershed is about 1500 feet (457 m) thick and typically consists of gray-black clay shale and rather silty thin to massive brownish-gray sandstone. Some ironstone concretions are dispersed through the shales. Rather chaotic sandstone-shale intervals are present at places in the formation. A few invertebrate fossils occur in a few lenticular siltstone masses that likely were deposited by submarine slumping from a southern source.

Atoka Formation

The Atoka Formation was described and mapped by Taff and Adams (1900) near Atoka, Oklahoma, but a type section was not designated. Reinemund and Danilchik (1957), Stone (1968), and Haley et al. (1976) further defined and established the Atoka Formation in the southern Arkoma Basin and Ouachita Mountains of Arkansas. The Atoka was first differentiated from the Jackfork in the Athens Plateau by Miser and Purdue (1929). Walthall divided the "Atoka Formation" in this area into the Johns Valley Shale and Atoka Formation. During the Arkansas state geologic map project (Haley et al. 1976), the Atoka-Johns Valley boundary was further adjusted.

The Atoka contains about 7500 feet (2287 m) of thin to rather massive, fine to medium-grained subgraywacke (silty) sandstones and interbedded gray-black shales. There are chaotic intervals containing masses of sandstone, siltstone, iron carbonate concretions and possibly some erratics that suggest extensive slurries and slumps derived from submarine scarps generally to the south.

A few conglomerates and calcareous sandstones contain a transported invertebrate mold fauna. The top of the Atoka is not exposed in the area and it is believed that about 20,000 feet (6100 m) was removed by Early Cretaceous erosion. Small quantities of road aggregate have been mined from the Atoka.

Brownstown Marl

The name Brownstown Marl was first applied by Hill (1888) to strata of Late Cretaceous age outcropping in the vicinity of Brownstown, Sevier County, Arkansas. It was subsequently mapped and partially redefined by a number of investigators in southwest Arkansas.

In the eastern part of the Caddo River watershed, near De Gray Dam, the gently southward dipping Brownstown Marl caps hills underlain by the tilted Paleozoic rocks. The Brownstown was deposited in the Gulf Coastal Plain. The break between the older and younger rocks represents the Fall Line –– or a major unconformity with a long period of erosion along the eastern and southern Ouachita fold belt. The Brownstown has a thickness of about 175 feet (53 m) and is composed of marl, clay, sand and gravel. These beds were deposited mostly in a shallow marine environment near an ancient shoreline. Some marine fossils, especially a microfauna, and a few lignitic logs have been noted in the formation. Sand and gravel from the Brownstown was used in construction at De Gray Lake.

Other Upper Cretaceous Formations

The Brownstown Marl is overlain by several rather thin, mostly marine and fossiliferous, younger Upper Cretaceous formations in a small area along the southeastern border of the watershed near Arkadelphia. The following brief description of these formations is mostly from Dane (1929). The Brownstown is overlain unconformably by about 125 feet (38 m) of sandy marl of the Ozan Formation. The Marlbrook Marl unconformably overlies the Ozan and consists of about 80 feet (24 m) of dark blue (fresh) to white (weathered) chalky marl. The Saratoga Chalk which rests unconformably on the Marlbrook contains about 60 feet (18 m) of fairly hard, white chalk with some marl. The Nacatoch Sand unconformably overlies the Saratoga and consists of about 200 feet (61 m) of cross-bedded, yellowish-gray, fine-grained, slightly glauconitic, unconsolidated quartz sand, with minor clay and gravel intervals. About three miles south of the area, the Arkadelphia Marl, which is the youngest Upper Cretaceous Formation in Arkansas, unconformably overlies the Nacatoch.

Terrace Deposits

Minor alluvial terrace deposits of Quaternary (Pleistocene) age occur along the Caddo River watershed. These deposits are thin, probably not exceeding 15 feet (4 – 6 m) in thickness. They consist of sand, gravel and cobbles derived mostly from the more resistant Paleozoic rocks in the area. Near the terminus of the Caddo River with the Ouachita River, the terrace deposits reflect more of the source and transport of the Ouachita River system.

These several levels of terrace deposits were previously more extensive, but have been repeatedly altered by subsequent Pleistocene and Recent events that at times have caused intense erosion and deposition along the watershed.

Alluvium and Colluvium

Clay, silt, sand, gravel, cobbles and, locally, boulders derived from the resistant sedimentary rocks in the area compose the Recent alluvial deposits of the Caddo River and its tributaries. These deposits generally do not exceed 20 feet (6 m) in thickness, except near the terminus of the Caddo River with the Ouachita River where they are somewhat thicker. The colluvial deposits are composed of clay, silt, sand, gravel, cobbles and, locally, boulders that were mostly derived by extensive slope wash from rocks on the surrounding hills and cover most of the valleys and slopes as a thin veneer. A few pimple mounds occur with these deposits. Some of the silt may represent a windblown (loess) component.

Sedimentary History

The following abstract on the sedimentological history of rocks in the Ouachita Mountains, Arkansas, with minor revisions, is from Stone and Haley (1982).

The Paleozoic sedimentary rocks of the Ouachita Mountains in Arkansas range in age from Early Ordovician to Middle Pennsylvanian and have an aggregate thickness in excess of 50,000 feet (15,200 m) —— (about 30,000 feet [9,100 m] crop out in the Caddo River watershed). The stratigraphic sequence including the Early Ordovician age Collier Shale through the Early Mississippian age Hot Springs Sandstone Member at the base of the Stanley Shale is from 7500 (2290 m) to over 12,000 feet (3660 m) thick. The shales, micritic-arenitic limestones, siltstones, sandstones, cherts, novaculites and conglomerates of the sequence are considered proto-Ouachita bathyal platform or trough deposits that represent: 1) indigenous pelagic or hemipelagic deposits; 2) turbidity or bottom current-submarine fan and related deposits combined with episodes of slump and slurry detachments producing the included erratics; and 3) minor Devonian and other intrusives(?). With the exception of Silurian age Blaylock Sandstone which has a probable southeastern source, these rocks were all derived from "northerly" flanking shelf, slope and submarine ridge sources.

Beginning with the Hatton tuff lentil in the lower Stanley Shale of early Chesterian time (Mississippian) and ending in the Middle Pennsylvanian upper portion of the Atoka Formation, over 40,000 feet (12,200 m) of deep-water turbidites——sandstones and shales ——and some cherts combined with submarine slope and platform erratics were deposited in the rapidly subsiding Ouachita trough. The Stanley Shale was derived from a volcanic island arc and other sources to the south and southeast with only a minor source of clay and olistoliths from the north. The Jackfork Sandstone, Johns Valley Shale and Atoka Formation represent coalescing submarine fan accumulations derived from major delta systems to the north, northeast, east, and southeast, and, in part, south with episodes of major slumping, particularly in Johns Valley time, from flanking platform deposits and slope facies to north and northwest. Preconsolidation sediment flow features demonstrate repeated cycles of "southward" slumping in rocks of all ages, except in the extreme southern part of the area, where the Johns Valley Shale and the lower part of the Atoka Formation, have sedimentary structures indicative of northward slumping directions which suggests that they were deposited on the south side of the Ouachita trough as it was apparently being closed by converging structural plates.

After the uplift of the Ouachita Mountains in the late Paleozoic and into Late Cretaceous times, the area, was, for the most part, extensively eroded. The minor deposits that were possibly formed during this long time span were mostly reworked by the partial inundations of the warm Late Cretaceous seas. Shallow marine and alluvial conditions likely prevailed in the area throughout most of early Tertiary time but were subsequently eroded from the immediate area. In the Quaternary (Pleistocene and Recent) there were periods of braided stream alluviation and extensive erosion. Remnants of the terrace deposits occur above the alluvium along the Caddo River watershed.

Quartz Veins

Quartz veins ranging in width from less than an inch to rarely many feet are locally numerous in the Paleozoic rocks. Large veins are especially common along the northern margins in the Crystal Mountains. Most of the veins consist of milky quartz with scattered inclusions of chlorite, adularia, platy calcite, and dickite. These veins contain crystals and clusters of clear to milky quartz that are renowned world wide for their high qualities. The High Point deposit in section 19, T. 3 S., R. 24 W., in the Crystal Mountains is one of the larger occurrences that is sporadically mined for crystals. Numerous other sites occur in the adjoining sections. Locally in the central and southern parts of the area minor quartz veins contain very small complex crystals, some of which are "water-bubble" or "negative" types.

Sulfides are extremely rare in most quartz veins, but a few contain significant quantities of lead, zinc, copper, silver, antimony, mercury and other elements. At times considerable prospecting has been accomplished on these mineralized quartz veins in or adjacent to the area. These fracture-filling veins are considered to be of hydrothermal (epithermal) origin (Miser 1943, 1959, and Engel 1952). Bence (1964) indicated a maximum temperature of about 347°F (175°C) for quartz veins in the Crystal Mountains. Most of these quartz veins are considered of late Paleozoic age having formed during the closing stages of the Ouachita orogeny.

Igneous Rocks

Igneous rocks generally of early Late Cretaceous age occur as nepheline syenite plutons; volcanic and explosion breccia pipes; and numerous small lamprophyric dikes and sills to the east of the watershed at Magnet Cove, Bauxite and Little Rock. Diamond-bearing kimberlite pipes are present to the southwest near Murfreesboro. The only igneous rock known in the area is on the eastern end of Pigeon Roost Mountain in the SW¼ SE¼ section 11, T. 4 S., R. 23 W., where a carbonate-rich lamprophyric igneous breccia has intruded the Arkansas Novaculite and Stanley Shale. Other igneous rocks, especially small dikes and sills probably occur in the watershed, but they have not been noted due to the deep weathering, extensive soil cover and rather thick vegetation.

Some beds of volcanic tuff occur primarily in the lower Stanley Shale and were probably derived from volcanic sources to the south of the Ouachita trough. Volcanic and igneous detritus also occurs in some of the Upper Cretaceous rocks south of the area.

Minor epizonal igneous and metamorphic rock fragments, some as large as boulders, are incorporated in the rocks of the Collier, Crystal Mountain, Blakely and other formations within or adjacent to the area. These are considered to be derived by slumping from mostly Precambrian rock sources that occurred along submarine scarps that flanked the north side of the Ouachita trough.

General Structure

The Paleozoic rocks that crop out in the watershed were involved in the various tectonic stages leading to the development of the Ouachita Mountains, mostly in late Paleozoic times. The intensity of structural deformation in the Paleozoic rocks increases from south to north across the area at the surface (Fig. 6). There are broad folds cut by numerous southward dipping thrust faults in the Pennsylvanian rocks in the southern Athens Plateau. In the central and northern Athens

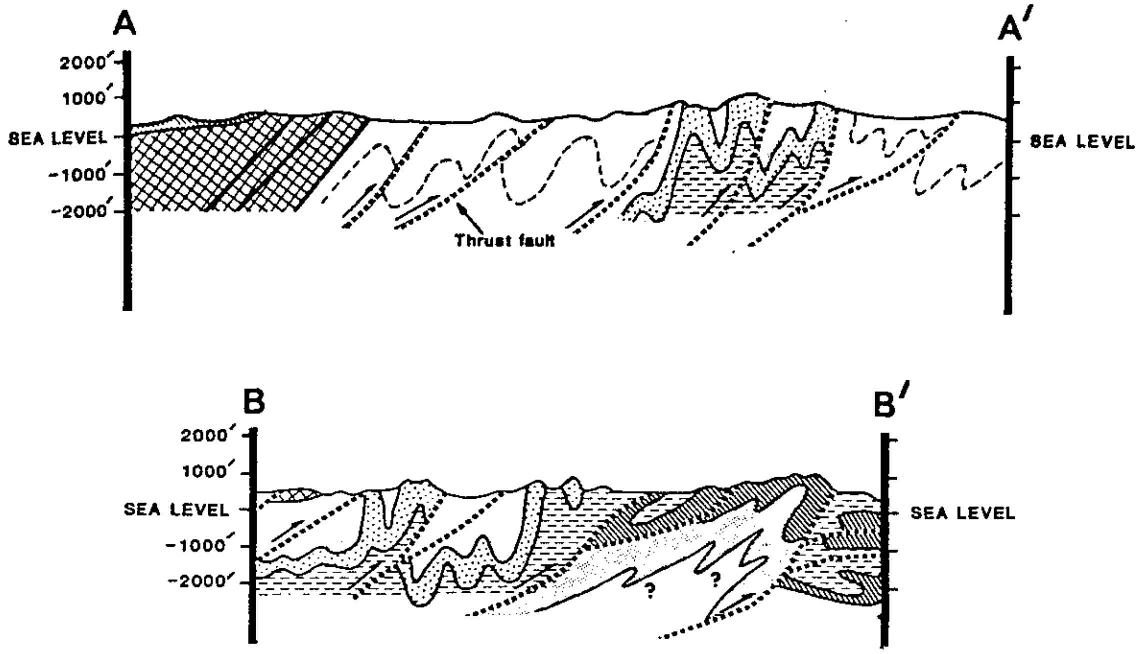

FIGURE 6. GENERALIZED STRUCTURAL CROSS-SECTIONS OF THE ROCKS IN THE CADDO RIVER WATERSHED. (Lines of section are shown on Figure 3)

Plateau within the Mississippian Stanley Shale, there are tight folds broken by both high angle and near bedding plane thrust faults. Minor shearing occurs in some of the shales. The rocks in the Cossatot and Caddo Mountain ranges and the Mazarn Basin are very steep or even overturned, and locally exhibit some shearing. Thrust faults with displacements on individual plates varying from a few feet to many miles disrupt the strata. Small milky quartz veins fill fractures in some of the rocks. Strata in the Caddo Basin and the Crystal Mountains contain exceptionally complex folds (often recombent) with several major low-angle decollements and many smaller thrust faults. These rocks are locally cut by sizable quartz veins and a well-developed rock cleavage occurs in some shales.

Gently southward dipping Late Cretaceous strata cover deformed Paleazoic rocks along the southern border. These and all older rocks are subsequently overlain by nearly flat-lying terrace, alluvial or colluvial deposits of Quaternary age.

Structural History

The following summary on the structural history of the Ouachita Mountains, Arkansas, with some revisions, is from two abstracts by Haley and Stone (1981, 1982).

> The rocks in the Ouachita Mountains may have attained their present structural setting through sequential periods of folding and faulting, with each period of deformation affecting the previous folds and faults to the extent that, in many of the areas, they were backfolded to the point of being overturned southward.
>
> In most previous investigations the deformed Paleozoic rocks of the Ouachita Mountains of Arkansas have been divided into three poorly defined structural parts; the "core area," the "frontal zone" or "frontal belt" to the north, and the "southern Ouachitas" or "southern belt" to the south. Through our recent studies of the surface geology we have divided the area into seven generally east-west trending structural belts. Each belt is a unit having similar structural features and is a northward moving imbricately faulted thrust plate with a major sole fault. From north to south these belts are named Rover, Aly, Nixon, Avilla, Mt. Ida, Hopper and Amity. The Mount Ida, Nixon and Avilla belts include most of the older Paleozoic rocks and are the most intensely deformed.

It is suggested that the simplified sequential phases in the structural development of the Ouachita Mountains are as follows: (A) extensional faults and minor igneous intrusions, (B) major uplift with folding and decollement of the more competent units; (C) thrust faulting; (D) folding with further decollement; (E) thrust faulting and related backfolding, (F) cross faulting and folding with arching; and (G) further arching. It is suggested that Step A took place during the early to middle Paleozoic, Steps B-D during Middle to Late Pennsylvanian, Steps E-F during Late Pennsylvanian through Permian and possibly Triassic, and Step G from Triassic to Recent.

We conclude that: (1) the Ouachita Mountains in Arkansas are allochthonous and formed by northward overriding imbricately faulted thrust plates with major sole faults; (2) some thrust plates possibly involved Precambrian rocks in the subsurface; (3) the structural deformation likely narrowed the initial width of the Ouachita depositional basin by as much as 250 miles (400 km), and (4) a northward(?) dipping fossil Benioff subduction system was present to the south of the Ouachita Mountain outcrop likely south of the Sabine Uplift in northern Louisiana.

ECONOMIC GEOLOGY

Mineral Resources

The principal mineral resources of the Caddo River watershed are barite, manganese, novaculite, quartz crystals, rock aggregate, slate and tripoli. Some of the other mineral occurrences include clay, copper, lead and zinc, limestone and chalk, mercury, phosphate, and iron ore. Significant development of the mineral resources started in the middle 1800's and has grown rather slowly. Through the years there have been many claims, prospects and related ventures for precious and base metals in the area, but they have met with little success. Currently slate is being produced for roofing granules; a barite mine and mill are slated for operation upon the resumption of higher rates and demands. Some quartz crystals are sporadically mined for mineral specimens and other uses and rock aggregates are mined throughout the area.

Potential for future growth is good, because of the demand for the various mineral commodities. The following is a brief discussion of the mineral resources of the watershed including their occurrence, history and potential.

Barite

Barite, a sulfate of barium, is a heavy nonmetallic mineral with a specific gravity of 4.5, usually gray in color, and granular to crystalline. The ore occurs in beds, nodules, or veins in the lower Stanley Shale, in the Middle Division of the Arkansas Novaculite and possibly in the Missouri Mountain Shale.

Barite was first discovered in Arkansas in 1888 in Montgomery County and in 1900 in Hot Spring County just east of Magnet Cove. Production from the Magnet Cove area began in 1940 and from 1944 to 1966 yielded sufficient barite that Arkansas led the nation in its production. From 1944 to 1962 small-scale mining in Montgomery County accounted for about 55,000 short tons (50,000 metric tons) of barite. From 1975 to 1981 significant barite was produced from three deposits in the watershed. Barite mining operations are expected to resume in the immediate future at the Fancy Hill deposit. The principal use of the Arkansas barite has been as a weighting agent for drilling muds in the oil industry.

Barite is found in the central part of the area extending west into the Cossatot-Little Missouri watersheds. In the Caddo watershed two general districts have been recognized and mined: the Fancy Hill District in T. 4 S., R.'s 25-27 W., and the Pigeon Roost Mountain District in T. 4 S., R.'s 23-24 W. The following is a description and the major mines and occurrences in the two districts.

The Fancy Hill District on the southwest border has six distinct deposits that have had extensive investigation with some mining in the past. The ore occurs in the Stanley Shale and the Middle Division of the Arkansas Novaculite. The Gap Mountain deposit is located in the SE¼ of section 19, T. 4 S., R. 25 W. and the NE¼ of section 23, T. 4 S., R. 26 W. The barite occurs in the Stanley Shale in four lenticular deposits. The barite beds are 3 to 30 feet (1 - 9 m) thick and

300 to 1200 feet (91 to 366 m) long. The percent of barite decreases to the west. The Fancy Hill deposit (Henderson) is located in the NE¼ section 29 and the SW¼ section 19, T. 4 S., R. 26 W., on the south flank of the Fancy Hill anticline and associated thrust faults. The barite occurs in the basal Stanley Shale in six lenticular deposits that range from 300 to 1800 feet (91 to 549 m) in length with an ore-bearing zone 30 to 80 feet (9 to 24 m) wide. The barite beds dip very steeply to the south. A mill and some preliminary mining and processing was begun on this deposit by the Milchem Company in 1981-82, but the operation is temporarily closed awaiting an increase in the price and demand for barite. Sulphur Mountain deposit, also known as the McKnight deposit, is located in the northern portions of section 29 and the eastern portions of section 30, T. 4 S., R. 26 W. Mining operations by the Baroid Division of the National Lead Company began on this deposit in 1975 and concluded in 1981 when the cost of extracting and processing the ore exceeded its value. The barite occurs in the lower Stanley Shale. Several thrust faults disrupt the ore body. The strike of the rock is generally to the west-northwest. The Dempsey Cogburn deposit is located on the southeast side of Sulphur Mountain near the center of section 33 and the northern portions of section 32, T. 4 S., R. 26 W. The barite occurs in the lower Stanley Shale. A cross fault forms a boundary at the west end of the mineralized zone. Mining operations by the Baroid Division of the National Lead Company began in 1975 and continued into 1979 when it became infeasible to mine the ore economically. The mineralized zone was about 50 feet (15 m) thick and contained lenses of high grade ore that varied from 15 to 25 feet (5 to 8 m) thick. Interestingly, a rather large orthoconic cephlapod replaced by barite was collected by Craig Brinkley near the top of the barite zone at the east end of the pit and it was identified as *Rayonnoceras vaughianum*, an early Chesterian (Mississippian) form, by MacKenzie Gordon, Jr., of the U. S. Geological Survey. The Boone Springs Creek deposit is located in the SE¼ of section 24, T. 4 S., R. 27 W. Barite occurs as a vein in beds of the Middle Divison of the Arkansas Novaculite on the overturned south flank of the Fancy Hill anticline. The beds containing barite strike N. 45°., and dip 40° NE. The vein material has a maximum thickness of one foot (0.3 m) and a maximum length of 24 feet (7 m). The Polk

Creek Mountain deposit is located in the SW¼ section 12, T. 4 S., R. 27 W. Occurrences of barite have been traced from the east portions of section 12 to the center of section 11, T. 4 S., R. 27 W. Barite occurs as nodules in beds of shale interbedded with novaculite in the Middle Division of the Arkansas Novaculite. Some vein material was located in prospect pits. There has been no recorded production from this locality. In the south-central part of the area in Pike County, an occurrence of barite has been located in the SE¼ NW¼ section 1, T. 5 S., R. 25 W. A lens of quartz and barite one foot (0.3 m) wide and 8 feet (2.4 m) long occurs parallel to the bedding in the Stanley Shale near the contact of the Arkansas Novaculite.

The Pigeon Roost Mountain District is the easternmost occurrence of barite in the area. The barite occurs as three lenticular ore zones in the lower Stanley in section 25, T. 4 S., R. 24 W. and section 30, T. 4 S., R. 23 W. The Southwestern deposit is located in the S½ NE¼ SW¼ section 25, T. 4 S., R. 24 W. The Baroid Division of the National Lead Company began mining this deposit in 1975; and in 1978 when it became economically untenable to further mine the deposit, the operation shut down. The linear open-pit was then reclaimed. Barite occurs in shale and sandstone of the lower Stanley Shale which strikes N. 60° E. and dips 50-66° NW. Thrust faults cut the ore body and its exact stratigraphic placement is unknown. The Central Barite deposit is located in the SE¼ NE¼ section 25, T. 4 S., R. 24 W. Barite consists of two small lenses 100 feet (30 m) in length in the Stanley Shale and appears to be at the nose of a small anticline. The rather shaly barite reaches a maximum thickness of 7 feet. The Northeastern deposit is located in the W½ SW¼ NW¼ section 30, T. 4 S., R. 23 W. The barite occurrence here is similar to the southwestern deposit, but thinner. The mineralized zone occurs along the strike for approximately 500 feet (152 m) with a maximum thickness of the ore zone of 16 feet (5 m) and a total thickness of the ore beds of only 5 feet (1.5 m).

Clay

Clay deposits in weathered shale and slate occur in the watershed but they have not been utilized. There are no estimates on the reserves of clay or an evaluation of their ceramic properties. Some of the abundant shales should provide an adequate supply of quality material for use in making bricks. Clay has been produced in Garland County to the northeast of the area, and from near Mena in Polk County to the northwest of the watershed.

Copper

Traces of copper mineralization are reported to be associated with many of the manganese occurrences in the Arkansas Novaculite. Immediately west of the area in the SE¼ NW¼ section 10, T. 4 S., R. 27 W., at the North Mountain mine significant copper mineralization is associated with the manganese ore. Some copper occurrences are also found in association with wavellite (aluminum phosphate) veins in the Bigfork Chert near the northwest portion of the area. These veins were tested for copper by the Copperlume Corporation in 1972, but commercial concentrations were not found. Some chalcopyrite (copper sulfide) occurs with the lead and zinc veins at the Housely (Point Cedar, Price-Williams) mine north of the community of Point Cedar in the NE¼ NW¼ section 31, T. 4 S., R. 21 W. The copper-bearing minerals found in fractured beds of Arkansas Novaculite are turquoise, malachite, azurite, chrysocolla, chalcopyrite and native copper. There is a slight possibility of copper deposits with economic potential being discovered during exploration in this and nearby portions of the Ouachita Mountains.

Gold

According to Comstock (1888) several prospects and claims were made in various rock formations and veins in or near the Caddo River watershed for gold and other valuable elements in the mid-1880's and earlier. While he indicates that some samples contained rather significant silver values, only a few contained traces of gold. From this period until 1980 there were but a few minor prospects for gold that were soon abandoned. In 1980 another flare of "fever" was revived in much of the area as a result of several newspaper articles describing micron-sized gold mostly in the Arkansas Novaculite. Then several companies, individuals and our organization initiated limited geochemical investigations of these rocks. While our data are very incomplete, we found no significant values of gold.

Iron Ore

Occurrences of limonite and goethite (iron oxides) and pyrite and marcasite (iron sulfides) have been reported in pockets, seams, and veins in the Arkansas Novaculite and occasionally in the Womble Shale, Bigfork Chert, Polk Creek Shale and Stanley Shale. The iron oxides are often associated with the manganese deposits in the novaculite, but they are usually too low in iron, too high in silica, or often too high in phosphorus to be of commercial value. In 1830, 1915 and 1916, a prospect was worked on masses or veins of pyrite and marcasite in the upper Womble Shale on the bank of the Caddo River on the west edge of the community of Caddo Gap. Water from the Caddo River seeped through the rocks filling the shafts and drill holes, thus ending the operation.

Lead, Zinc and Silver

Traces of lead, zinc, and silver are associated with quartz veins, in some Womble limestones and in a few manganese deposits. The most notable occurrence is at the Housely (Point Cedar, Price-Williams) mine located in the NE NW¼ section 31, T. 4 S., R. 21 W., where galena, sphalerite, chalcopyrite, pyrite and other minerals occur in quartz veins mostly in a fault zone in the Arkansas Novaculite and lower Stanley Shale. The deposit was discovered in the 1890's and worked several years through shafts and two major levels, along a linear distance of about 210 feet (64 m). Formerly a mill was located at the site, but it burned in 1973. Significant quantities of silver in galena, sphalerite, and other minerals also have been reported near Silver, Arkansas, about four miles north of the area. A company has recently done some exploratory drilling on sites in upper Womble limestones a few miles west of Caddo Gap. The results of their testing are unknown at present. It is likely that these investigations pertained to zinc anomalies since portions of the Womble are reported to contain some quantities of this element.

At present no economic occurrences of lead, zinc, and silver are known but exploration will likely continue for these and associated elements if their economic value remains rather high.

Limestone

Thin to rather thick beds and intervals of limestone are found in most of the Ordovician rocks. Limestones are especially abundant in the Collier Shale, some portions of the Mazarn Shale and in the upper Womble Shale. Previously, there was sporadic production of limestone from the Collier north of the area. It was used locally for road material and agricultural purposes. There also has been some investigation of limestones near Black Springs and elsewhere for agricultural limestone, lime and as a black marble for use as a decorative building stone. Analysis indicates that the limestone contains 75 to 90 percent $CaCO_3$.

Descriptions of some of the many significant Womble limestone exposures in the area follow:

	Location	Description
1.	NW¼ SW¼ section 26, T. 3 S., R. 24 W.	dense bluish-gray limestone
2.	SW¼ SE¼ section 6, T. 4 S., R. 24 W.	dense bluish-gray limestone, 50 feet thick, contains 86.2% $CaCO_3$.
3.	NE¼ NE¼ section 35, T. 3 S., R. 26 W., and	dark bluish-gray, hard limestone, 15 feet thick, exposed in creek
4.	SE¼ SW¼ section 26, T. 3 S., R. 26 W.	dark bluish-gray, hard limestone, 1 to 8 inches thick in 10-foot-thick zone on south bank of Caddo River.

Limestone is not presently being produced, but there is limited exploration of several deposits by a firm. Further exploration will be necessary to evaluate the apparently significant reserves.

Chalk occurs in the Saratoga Formation, but it is too thin in the watershed for economic uses.

Manganese

Manganese deposits in the watershed have been worked sporadically since 1859. Federal purchase supported the last activity in 1958-59 to the west and north of the area. Both the Upper and Lower Divisions of the Arkansas Novaculite contain manganese deposits which occur as nodules, pockets and short irregular veins varying from a fraction of an inch to rarely four feet in thickness in bedding planes, joint cracks, or as a cement between fragments of novaculite.

The upturned folded novaculite is very resistant to erosion and forms ridges which extend for miles. Most of the ore can be found in highly fractured rock at or near the axes of folds and adjacent to faults. It is not known whether the manganese ore extends to a significant depth in the subsurface, but immediately west of the area on the North Mountain Mine ore was reportedly found in the lower tunnel at 615 feet below the portal. Psilomelane, pyrolusite, and manganite make up the largest part of the ore, but there also is lithiophorite, wad and a few other minerals. The minerals may occur separately, but usually occur together in association with clay and iron oxides.

Table 1 gives the location and a brief description of many of the manganese mines and prospects in the area. Traces of psilomelane in novaculite were also reported at a number of other places including the W½ section 16, T. 3 S., R. 22 W. Some small manganese prospects occur in the northern portions of the watershed in the Crystal Mountain Sandstone at High Peak in the NW¼ NE¼ section 19, T. 3 S., R. 24 W.

Manganese reserves in the area cannot be accurately calculated with the limited amount of exploration that has been done. Improved milling techniques, such as blending of ores to provide a uniform feed, should lower costs and improve the value of the product. The quantity of manganese ore that occurs in any deposit is usually rather small, and would necessitate combined suppliers for milling operations. Higher prices or government support are required for the commercial production of the manganese deposits in the watershed. In times of dire need the reserves could measurably augment the United States supply. Significant by-products, such as copper, might have economic value upon beneficiation of the manganese ore. In future years, these deposits may represent a valuable economic resource.

Mercury and Antimony

The Arkansas mercury (cinnabar) district occurs along a 30-mile (48-km) belt south-southwest of the watershed. It extends eastward from Howard County across Pike County and into Clark County. These deposits occur as epithermal veins in faulted and fractured intervals of the Stanley Shale and Jackfork Sandstone. Since their discovery in 1931 until 1946 about 11,400

Table 1. – Manganese mines and prospects in the Caddo River Watershed

Name	Location	Description
Nelson Manganese Mine	SW¼ SW¼ sec. 13, T. 4 S., R. 24 W.	Psilomelane and a small amount of crystalline manganite are found in veins, pockets, and cavities in novaculite. In 1915, 58 tons were mined, which apparently depleted the deposit. A nearby claim was prospected in 1940.
Jones Valley Mine	NE¼ NE¼ sec. 18, T. 4 S., R. 24 W.	Thin veins of manganite and psilomelane occur in fractured novaculite. Two cuts were made but no production was recorded.
United Minerals Industries Mine	N½ sec's. 28, 29, T. 4 S., R. 26 W.	Manganese oxide cement and novaculite fragment nodules of psilomelane occur in 2- to 8-foot-wide clays. Select samples contained 27.04% manganese.
Plemmons-Woodall Mine	SW¼ SE¼ & NE¼ SW¼ sec. 11, T. 4 S., R. 23 W.	Concentrations of psilomelane, manganite and wad cementing terrace deposits are present. Gravels fill a valley adjacent to a ridge in which veins of psilomelane also fill fractures in weathered novaculite. Most of the gravel has been removed and the veins are insignificant. In 1937 two carloads were shipped and 20 tons were removed in 1940.
Willie Cogburn Claim	E½ sec. 18, T. 4 S., R. 26 W.	Fractures in novaculite contain manganese oxides. There are an estimated several hundred tons of manganese in the deposit.
Monroe-Knold	SE¼ sec. 14, T. 4 S., R. 26 W.	Manganese ore was produced before and during World War I.
Montgomery Manganese Corp.	E½ sec. 16, T. 4 S., R. 26 W.	Manganese ore was produced before and during World War I.
Polk Creek Mountain Prospect	Sec. 13, T. 4 S., R. 27 W. Sec. 18, T. 4 S., R. 26 W.	Manganese ore was produced before and during World War I.
Watkins White Prospect	S½ SE¼ sec. 1, T. 5 S., R. 24 W.	Seams and veins of wad and psilomelane occur with limonite.
Reynolds Mountain Prospect	SW¼ NW¼ sec. 6, T. 5 S., R. 24 W.	Veins of psilomelane, manganite, and brown iron oxide are present in the massive novaculite.
Featherstone Mine	SW¼ NW¼ sec. 6, T. 5 S., R. 24 W.	Psilomelane and limonite occur in fractures and nodular manganese in clay. The mine yielded 40 tons of ore in 1917.
Kettelberger Prospect (formerly Reynolds Mountain Prospect)	NE¼ SE¼, NE¼ SW¼ SW¼ NE¼, SE¼ NE¼ sec. 1, T. 5 S., R. 25 W.	Pyrolusite and manganite occur in fractured novaculite veins. One and one-half carloads of select ore was shipped. There are an estimated 400-500 tons raw ore and an additional 200-250 tons +40% manganese concentrates.

Fagan Mine	Center S½ N½ sec. 9, T. 5 S., R. 25 W.	There are thin veins of sparse manganite with iron oxides.
Bear Mountain Prospect	NE¼ NW¼ sec. 8, T. 5 S., R. 25 W.	Psilomelane and manganite lenses occur along bedding surfaces in the novaculite.
Brushy Mountain Mine	SW corner sec. 5, NW corner, sec. 8, T. 5 S., R. 25 W.	Stalactitic psilomelane and cement or vein occur in novaculite breccia. Four openings were made at the mine in 1916. Twelve percent manganese ore was concentrated to contain 40% manganese. One thousand tons of manganese concentrates were shipped in 1942.
R. M. Cogburn Prospect	NE¼ SW¼ NE¼ sec. 5, T. 5 S., R. 26 W.	Boulders of novaculite having small veins/pockets of psilomelane are present.
North American Manganese Co.	SW¼ SW¼ sec. 5, S½ SE¼ sec. 6, T. 5 S., R. 25 W.	Pockets of psilomelane with inclusions of pyrolusite occur in novaculite. Reserves are estimated at 18,000 tons of 10% manganese ore.

flasks (76 pound — 34 kilogram) were extracted from the ores by retorts or furnaces. Minor mercury mineralization has been noted at two places in the southern part of the Caddo River watershed in faulted sequences of the Jackfork Sandstone.

The antimony district to the southwest of the area encompasses portions of Pike and Sevier Counties in southwestern Arkansas and McCurtain County in southeastern Oklahoma. The most significant mines are near Gilliam in Sevier County. These epithermal deposits occur as stibnite and other minerals in quartz veins associated with fault zones in the Stanley Shale. Since their discovery in 1873 until 1947 approximately 5,390 tons (4890 metric tons) have been produced.

Thin sparse veinlets of stibnite and pyrite are reported by Stroud et al. (1969) filling fractures and bedding planes in folded and faulted sandstones of the Stanley Shale in section 3, T. 6 S., R. 20 W., which is presently within the confines of De Gray Lake. It is reported that an adit exposed a silicified zone approximately 10 feet (3 m) wide that contains metalliferous sulfides. This deposit apparently has little economic importance. Small quantities of antimony minerals also are found in several of the lead, zinc, copper and silver-bearing quartz veins in the area.

Novaculite (Whetstones)

The use of Arkansas Novaculite for whetstones dates back to the early 1800's. Even prior to this, Indians of the area diligently worked the novaculite for projectile points, scrapers and other purposes. Novaculite is a hard, very fine-grained variety of chert or flint rock that is often slightly recrystallized and composed mostly of silica. There are two commercial classes of abrasive grade novaculite: the hard, dense, rather translucent "Arkansas" stone and the softer and more porous "Washita" stone. Most of the whetstone produced from the Arkansas Novaculite has been quarried in the Hot Springs area in Garland County. A small quarry operation (located in section 6, T. 5 S., R. 26 W.) west of the watershed has produced some Washita stone. Future whetstone prospecting and quarrying are anticipated in the novaculite of the area.

There is also some use and future potential for high-quality novaculite in making several other products, including ferrous-silicon which is used in the chemical industry, a filler in fire bricks and interlayered sequences with plastics. Rock aggregate is produced from the novaculite at several sites in the surrounding region and it seems likely that quarries will eventually be situated at advantageous locations in the area.

Phosphate

Widely separated layers, lenses, and nodules of black phosphate-rich rock occur in the basal Stanley Shale. In some places they reach a thickness of 1½ inches (38 mm). The Woodall prospect is located in the NW¼ SW¼ section 11, T. 4 S., R. 23 W. The rock contains from 2.20 to 3.12 percent P_2O_5 but the rock is not of commercial grade. Also some decomposed igneous rock that intrudes the Stanley Shale contains small percentages of P_2O_5.

Some rather unique phosphate minerals have been reported at the following localities:

Dufrenite (rare iron phosphate)	– NW¼ SW¼ section 7, T. 4 S., R. 25 W.
Wavellite and variscite (aluminum phosphate)	– SE¼ NE¼ section 35, T. 3 S., R. 24 W. and
	– NE¼ NW¼ section 8, T. 4 S., R. 24 W.

There are also numerous wavellite and variscite locations in adjoining sections near Bigfork, Arkansas. These minerals may contain some quantities of copper and vanadium. They have been prospected in places, for copper and also for mineral specimens.

Quartz Crystals

The Crystal Mountains comprise portions of one of the principal quartz-producing areas in the world. Clear to milky quartz crystals have been mined here for many years and they have mostly been utilized in the mineral and gem trade, although some quartz has also been acquired for fusing, oscillator and precast concrete aggregate purposes.

Veins in the Crystal Mountain Sandstone are the principal source of quartz crystals in the Caddo River watershed. Large quantities are also obtained from veins in the Blakely Sandstone, but mostly to the northeast in Garland County. Quartz occurs as veins filling joints and bedding planes in the sandstone and crystals occur as pockets or cavities lining or filling them.

For many years large quantities of small to medium size clear quartz crystals and clusters have been produced from the Fisher Mountain and other deposits located in the Crystal Mountain Sandstone along the southern border of sections 3 and 4 and the northern portions of sections 9 and 10, T. 3 S., R. 24 W., about one mile north of the watershed. There are several other adjoining mines that have also produced quartz crystals.

Some of the more significant quartz crystal localities in the area are:

1. The High Peak deposit in the NW¼ NE¼ section 19, T. 3 S., R. 24 W. High-quality clear crystals have been produced at this deposit in recent years from veins in the Crystal Mountain Sandstone. Manganese occurs with the quartz and it has also been prospected.

2. Near the centerline of section 20, T. 3 S., R. 24 W., quartz crystals have been mined from the Crystal Mountain Sandstone.

3. Near the center of the west line of section 21, T. 3 S., R. 24 W., some quartz crystals are obtained from veins in the Crystal Mountain Sandstone.

4. In the NW¼ section 29, T. 3 S., R. 24 W., minor quantities of quartz crystals have been mined from veins in the Blakely Sandstone.

5. Rather unusual, very small, clear to smoky, complex crystal forms, some being water bubble or negative types occur in veins in Stanley sandstones at the Pigeon Roost Deposit in section 11.

T. 4 S., R. 22 W., and at other localities through the central portions of the watershed, including a prospect in the Arkansas Novaculite about 1½ miles (2.4 km) northeast of the community of Point Cedar.

Large quantities of quartz crystals are likely present in the numerous quartz veins in the area. Many additional claims and prospects are expected as a result of the continued demand for clear quartz crystals.

Rock Aggregate

Vast tonnages of rock aggregate and road material are available from the sandstone, novaculite, chert, and sand and gravel deposits in the watershed. Massive sandstones meeting commercial quality specifications occur in the Crystal Mountain, Blakely and Jackfork Formations. Thick sequences of novaculite occur mostly in the Lower Division of the Arkansas Novaculite and intervals of thin chert occur in the Bigfork Chert. All of these rocks have been extensively mined for aggregate, but at present there are only a few commercial operations in the watershed. Shale is very abundant and is used locally for rock aggregate on rural roads and other purposes throughout most of the watershed.

The Murray Quarry in the SE¼ section 13, T. 6 S., R. 20 W., about one mile east of De Gray Dam, has been active for a number of years. Several million tons of quartzose sandstone have been mined and processed from a 90-foot (27-m) thick interval in the upper Jackfork Sandstone.

In 1963 gray sandstone from the Blakely Sandstone was quarried for building stone in the E½ SE¼ section 23, and the W½ SW¼ section 22, T. 3 S., R. 24 W. Hard sandstone beds 6 inches (15.2 cm) to 2 feet (0.6 m) in thickness were quarried from a depth between 4 and 8 feet (1.2 to 2.4 m).

In recent years rather large quantities of rough field stone also have been obtained mostly from surficial deposits. Most of these are sandstones derived from the Crystal Mountain, Blakely, Blaylock, Stanley and Jackfork Formations. There is no reliable estimate of the tonnages of field stone produced.

Sand and gravel deposits occur in the alluvium along the Caddo River and some of the larger streams. The following deposits have been worked along the Caddo River:

1. The E½ section 28, T. 3 S., R. 25 W., contains equal amounts of novaculite and sandstone gravel. Reserves above water level are estimated at 1,503,000 tons (1,363,492 metric tons).
2. NE¼ NE½ section 19, T. 4 S., R. 24 W., has mixed quantities of novaculite and sandstone, gravel and some large boulders. Reserves above water level are estimated at 235,000 tons (213,187 metric tons).
3. The NE NE½ section 13, T. 4 S., R. 25 W., has equal proportions of novaculite and sandstone, gravel and a little shale. Reserves above water level are estimated at 2,916,000 tons (2,645,337 metric tons).
4. The S½ of section 36, T. 6 S., R. 20 W., has mixed chert and sandstone gravel. There is no estimate of the reserves available.

Sand and gravel have also been obtained from deposits in the terraces along the Caddo River. Most of the larger pits are situated near the community of Caddo Valley and the material was used primarily in various projects in the construction of De Gray Lake and related facilities. Most of these deposits are included in the following locations:

1. S½ of section 19 and the N¼ of section 30, T. 6 S., R. 19 W., and the E¼ of sections 22 and 25, T. 6 S., R. 20 W.
2. The E½ of section 35 and the SW¼ of section 26, T. 6 S., R. 20 W.

Sand, gravel, and cobbles also have been mined from lenses in the Brownstown Marl in exposures near De Gray Lake where it has been used for aggregates related to the construction projects at De Gray Lake. Some of these stripping operations are as follows:

1. The S½ of section 12, the N½ of section 13, 6 S., R. 20 W., and the adjoining NW¼ of section 18, T. 6 S., R. 19 W.
2. The S½ of sections 16 and 17, T. 6 S., R. 20 W.
3. The SW¼ of section 22, T. 6 S., R. 21 W.

The Nacatoch Sand represents a potential source of high quality sand. It has been exploited on a limited scale in adjoining areas.

These abundant deposits of sandstone, novaculite, chert, sand and gravel, and shale can provide rock aggregate, road material, and building stone to meet local and many outside demands.

Slate

Slate for building purposes was first mined in T. 3 S., R. 27 W., in 1902. At present it is the foremost mineral product in the area and is being mined and crushed for roofing granules with the dust being used as a filler. Most of the slate mining has been in the Stanley and Missouri Mountain Formations but it also occurs in the Mazarn, Womble and Polk Creek Formations.

The Missouri Mountain slate is buff, red to green, usually soft but with some fairly hard and homogeneous layers. The slate interval varies from less than 50 feet (15 m) to about 300 feet (91 m) and may be duplicated by structure. The unit is widespread and has been extensively prospected.

The basal Stanley Shale has been changed to slate in the closely folded structures, particularly the synclines. The slate is hard, gray to black, and contains some thin sandstone layers. Horizons in the lowermost Stanley were previously known as Fork Mountain slate. The Polk Creek, Mazarn, and Womble Formations contain shale or slaty shale in most places and only portions of these units contain well-developed layers of slate.

Presently slate is being quarried from a large and rather deep pit in the Stanley north of Glenwood in the SE¼ section 21, T. 4 S., R. 23 W., by Bird and Son, Inc. They previously quarried a similar deposit in the NE¼ NE¼ section 31, T. 4 S., R. 25 W., and upon abandonment of the site the dimensions were about 1300 feet (396 m) in length, 200 feet (61 m) in width, and some 150 feet (46 m) in depth. Table 2 is a list of many of the abandoned slate quarries and prospects in the watershed.

Table 2. – Abandoned slate quarries and prospects in the Caddo River Watershed

Location	Formation	Description
NW¼ SW¼, sec. 3, T. 5 S., R. 25 W.	Missouri Mountain Shale	Red slate, area was prospected but reserves were not evaluated.
E½, sec. 10, T. 3 S., R. 23 W.	Missouri Mountain Shale	A prospect at this location reveals a red slate, sonorous with good cleavage.
NW corner sec. 36, T. 3 S., R. 24 W.	Missouri Mountain Shale	A prospect at this location reveals a dark red slate, semi-sonorous, splits easily and has a relatively rough surface; can be quarried in large blocks.
NE¼ SE¼ sec. 9 and NW¼ SW¼ sec. 10, T. 3 S., R. 26 W.	Mazarn Shale or Womble Shale	A prospect uncovered a hard green slate, sonorous, which splits well and has widely spaced joints.
SW¼ SE¼ sec. 32, T. 3 S., R. 26 W.	Missouri Mountain Shale	The area was quarried from 1929-33 for flagstone. Slate is red and green and splits into thin pieces on weathering.
E½ SE¼ sec. 33, T. 3 S., R. 27 W.	Missouri Mountain Shale	Red and green, soft, thin-splitting slate was found in four large openings.
SE¼ NE¼ sec. 18, T. 4 S., R. 26 W.	Missouri Mountain Shale	A red, green, and gray slate was quarried for roofing granules in 1909 and again in 1937.
N½ sec. 35, T. 3 S., R. 27 W.	Missouri Mountain Shale	Red slate was quarried in large blocks at two locations.
S½ NE¼ sec. 4, T. 4 S., R. 27 W.	Stanley Shale	A gray-blue micaceous slate which is sonorous, cleaves, has a rough surface and is weather resistant; was quarried in a pit 100 feet square and 50 feet deep. A blue-black slate was quarried. It has a glassy cleavage surface and crumbles on weathering (neither micaceous nor sonorous).
Sec. 15, T. 4 S., R. 26 W.	Missouri Mountain Shale	A 600-foot exposure of red slate with tight folds and a good cleavage occurs on the south side of Wagner Creek.
SE¼ NW¼ sec. 6, T. 4 S., R. 24 W.	Womble Shale	Inactive quarry in black slate and was used as roofing granules.

Tripoli

Tripoli is normally considered a finely granular, porous, comparatively soft silica of a cryptocrystalline character. It is used principally for an abrasive, polishing agent, and as a filler or additive. The high-silica tripoli occurs in the Upper Division of the Arkansas Novaculite, with smaller deposits in the Lower Division, and in the Bigfork Chert. Activity just west of the area took place sporadically from 1963 to 1972 when four mines produced 32,000 tons (29,030 metric tons) which were processed at Glenwood.

One of the larger known tripoli occurrences in the watershed is located in the SE corner of section 31, T. 4 S., R. 24 W., in the Upper Division of the Arkansas Novaculite. The tripoli at this deposit strikes nearly east-west and dips 18° to the south and some 150 feet (46 m) was penetrated by a drill hole. Indicated and inferred reserves amount to 75,000 tons (68,039 metric tons).

Fodderstack Mountain prospect in the Upper Division of the Arkansas Novaculite is located in the NW¼ section 13, T. 4 S., R. 26 W., and it is 1000 feet (305 m) in length, 28 feet (8.5 m) in width, and at least 20 feet (6 m) in depth.

Another occurrence is located in the SE¼ SW¼ section 12, T. 4 S., R. 26 W., and has a measurable ore of 20,000 tons (18,144 metric tons) and an indicated ore of 100,000 tons (90,718 metric tons). A minor occurrence of tripoli is located in a small cut in the Bigfork Chert at the base of a small hill about one and a half miles southwest of Caddo Gap.

Other known deposits of tripoli are as follows:

1. NE¼ SW¼ section 10, T. 3 S., R. 23 W.
2. NW¼ SE¼ section 12, T. 3 S., R. 23 W.
3. SW¼ SW¼ section 7, T. 4 S., R. 24 W.
4. SE¼ NE¼ section 17, T. 4 S., R. 24 W.
5. SW¼ SW¼ section 13, T. 4 S., R. 25 W.
6. S½ NE¼ section 33, T. 4 S., R. 26 W.

Travel distance to milling sites and lack to railroad facilities have probably hampered mining operations in the area. Future market price could alter this and an increase in activity is possible.

Uranium

Uranium (including thorium, potassium and others) prospecting has on occasion taken place in the watershed. Most of these investigations were concerned with the black carbonaceous shales of the Middle Divison of the Arkansas Novaculite and the Polk Creek Shale. Some limited evaluations were made on anomalies in the Stanley Shale (mostly phosphates), in the igneous breccia at Pigeon Roost Mountain, and in other rock types. There are presently no reported economically important uranium deposits in the area, although several rock types contain radioactive isotopes in excess of normal background.

Oil, Gas and Asphalt

Exploratory drilling in the rocks of the western Ouachita Mountains of Oklahoma and vicinity has resulted in the discovery of a few new oil and gas fields. These mostly occur in highly fractured reservoirs in cherts and novaculites of the Bigfork Chert and Arkansas Novaculite. This combined with a few older oil fields producing from Mississippian and Pennsylvanian sandstones and some small to rather large asphaltite occurrences in the area have kindled interests in the oil and gas potential elsewhere along the Ouachita fold belt.

Some of the surface rocks in the northern part of the Caddo River watershed are slightly metamorphosed, thus their thermal maturity is rather high. The intensity of deformation and recrystallization noticeably decreases in the rocks southward across the area and it is less likely that any generated hydrocarbons were degraded. This entire region is considered to contain allochthonous (transported) sequences of rock and the thermal histories are unknown beneath these postulated thrust faults (decollements). Most of the recent tectonic models for the Ouachita fold belt suggest that less deformed, and in some cases even foreland facies (shallow marine deposits), are at some depth beneath some of the surface rocks. A COCORP deep seismic reflection profile was recently run (Lillie et al. 1983) across the Ouachita Mountains of Arkansas to the west of the watershed and these data afford many insights into the origin of the structural complexities in the area.

In 1967 Max Ensinger drilled two oil and gas tests in Montgomery County. The No. 1 Van Steenwyk was drilled to an apparent total depth of 3627 feet (1107 m) in section 19, T. 3 S., R. 25 W., and the No. 1 Walter Gaston to a total depth of 472 feet (144 m) in section 20, T. 3 S., R. 25 W. Both wells were spudded in the Mazarn Shale and they were abandoned as dry holes.

In the last few years three wells have been drilled for oil and gas in or near the watershed. The Sheraton Oil Corporation drilled two wildcat tests; one, the No. 1 Kyle in section 29, T. 4 S., R. 22 W. (Hot Spring County), was spudded in Stanley Shale and reached a total depth of 4545 feet (1384 m), and another, the No. 1 Bean in section 15, T. 5 S., R. 23 W. (Clark County), was spudded in Stanley Shale and abandoned at 2902 feet (981 m). The Shell Oil Company drilled a well in the Trap Mountains within a few miles of the watershed in section 21, T. 4 S., R. 20 W. (Hot Spring County), that began in the Blaylock Sandstone and bottomed at a total depth of 7868 feet (2405 m).

It is thought that these few wells have failed to evaluate the petroleum potential of the area. A deep test penetrating the various older formations is needed to assist in determining the thermal histories, porosities, permeabilities and other features of the rock at depth. Some asphalt occurrences are present in the Jackfork Sandstone and the Lower Cretaceous rocks to the southwest of the area near Murfreesboro and Dierks, but none have been reported in the watershed.

The Upper Cretaceous rocks that crop out in the area are thought to have been flushed by ground water, so it seems unlikely that they contain oil and gas.

Water Resources

The average rainfall for the watershed is about 50 inches (1270 mm) per year. Ground water is often used for most domestic purposes. The area seems to have sufficient water supplies to meet current and most projected needs. It is suggested that extensive hydrologic studies be performed in an area prior to any major developments that would require large quantities of water. The following is a brief description of the water resources in the area.

Surface Water

The rather high quality and significant volume of water in the Caddo River, including its tributaries, and De Gray Lake is adequate and suitable for most projected needs in the area. For further details on the hydrology, see other reports in the Symposium volume and other sources listed in various references.

Ground Water

Ground water occurs mostly in fractures, joints and separations along the bedding planes of the Paleozoic rocks. In these rocks the highly fractured Bigfork Chert is the best aquifer. The Crystal Mountain Sandstone, the Arkansas Novaculite, and limestone intervals in the Collier Shale, Mazarn Shale, and Womble Shale also may be good aquifers. Quantities usually sufficient for domestic purposes are found in most formations. Highly permeable zones occur in the sands and gravels of the Brownstown, Nacatoch, terrace and alluvial deposits where suitable areas of recharge exist and afford good yields for local uses. Wells that yield 10 gallons (38 liters) of water per minute for a week of continuous pumping are considered high yield in most of the Ouachita Mountains. There are many small springs in the area which issue primarily from the Crystal Mountain Sandstone, Bigfork Chert, Arkansas Novaculite, and limestone intervals in the Collier Shale, Mazarn Shale and Womble Shale.

Ground water supplies are adequate in most areas for household use, but should not be considered for industrial developments or community water supplies. Ground water analyses by the U. S. Geological Survey indicate that it typically is of high quality.

Warm Springs

There are several warm springs in the central Ouachita Mountains of Arkansas. Two such springs are located in the watershed according to Miser and Purdue (1929). The warm springs at Caddo Gap in the Lower Division of the Arkansas Novaculite flow directly into the Caddo River. Recorded temperatures in the early 1900's ranged from $94^{\circ} - 96.8^{\circ}$ F ($34^{\circ} - 36^{\circ}$ C).

Another warm spring is located in the Arkansas Novaculite on the line between sections 23 and 26, T. 4 S., R. 25 W., about two miles southwest of Caddo Gap.

It is suggested that these warm waters represent ground water that has slowly percolated to a significant depth and has been heated by the geothermal gradient, then rapidly ascends to the surface through openings created by extensive fractures and/or fault zones. There is a likelihood that other small warm springs occur in the area. These warm springs afford little prospect as a geothermal resource.

SUMMARY

The information provided in this report has been generalized to give an overall view of the geology and mineral resources. The Caddo River watershed is located mostly in the central and southern Ouachita Mountain Province, but a small portion overlaps into the Gulf Coastal Plain. The rocks include numerous lithologic types: shale, sandstone, chert, novaculite, limestone, tuff, sand, gravel, marl and others. These rocks have been dividied into 19 Formations that range in age from the Early Ordovician to the Recent alluvium and colluvium. The Ouachita Mountains were formed by intense late Paleozoic deformation which caused the uplift and northward transport of the basinal deposits through complex folding and thrusting. Erosion which started with uplift and continues to the present has caused the removal of at least 30,000 vertical feet of sedimentary rock. The warm Cretaceous seas engulfed the southern and eastern portions of the area and deposited marl, chalk and sand that are very gently tilted. Also in Cretaceous times major igneous intrusions took place mostly in nearby areas. Throughout the Quaternary the region has undergone cycles of erosion and deposition related to the various climatic cycles.

Rocks of the watershed are presently the source of slate for roofing granules; sandstone, shale, chert, and other rock for road material; and quartz crystals. Soon there should be further production of barite for drilling mud. The area might have the potential for providing many other needed mineral resources. There is current interest in several other mineral resources. It is likely that the intensity in exploration and mining activity will further increase along with the projected demands for mineral resources.

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HISTORY OF THE CADDO RIVER WATERSHED

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INTRODUCTION

Otto Ernest Rayburn, editor of the newsmagazine Arcadian Life in Caddo Gap, Arkansas, from 1933 to 1941, said of the Arcadian lifestyle around the Caddo River:

> I stood at the Narrows on Caddo River a mile below the village of Caddo Gap, Arkansas, and watched history walk by. In my mind's eye I saw the colorful pageantry of the years passing in review. Solid rock walls hedge the crystal waters of this mountain stream into a churning channel where sleek small-mouth bass play hide and seek among the boulders that have fallen from the walls in ages past. The floor of the stream is lined with brown moss just as it was two or three centuries ago when Indian canoes shot the rapids in Arcadian ecstasy. It is one of the scenic spots of the Wonder State and I like to loiter there and listen to the sublime music of the churning waters. Fly fishermen from far and near pause to tempt the cunning bass with their lure. What an ideal place to stretch one's sentiments and weave colored threads into the tapestry of one's dreams (Rayburn: 1954).

Almost a century earlier on November 14, 1863, in a letter to his wife in Paraclifta, Arkansas, Sevier County, Lieutenant Robert C. Gilliam wrote, "Rumor is that the Federals have come through Caddo Gap" (Hudson: 1958). In 1817, Stephen F. Austin and his speculating brother-in-law, attempting to strike it rich in new frontiers, selected three sites for towns in southwestern Arkansas, one of which was the spot at which the Caddo joins the Ouachita in Clark County (Jones: 1966). On November 30, 1804, William Dunbar and George Hunter passed over the Fourche des Cadaux en route to

Hot Springs during the period called the pioneer naturalist phase in the history of American science. Recommended to Governor Claiborne of Louisiana by Thomas Jefferson, Dunbar and Hunter travelled the Ouachita and Caddo Rivers in order to collect, name and assemble data in the upland region of the territory newly acquired from the Spanish. In 1541, DeSoto's army marched to Tula from Tanico, believed by Colonel William Fordyce to be the Caddo Gap area, and there they encountered the fierce inhabitants known as the Tula Indians (Dickinson: 1980).

From Rayburn's "Arcadian ecstasy" backwards to DeSoto's exploration of the flora and fauna of the region, the Caddo River watershed has been seen, explored and altered by the English, French and Spanish. Prior to DeSoto, however, the inhabitants of the region believed that they had ". . .received the title of the land from the gods" (Lyon: 1952). The name of the Caddo is derived from French "Cadodaquois" of the 18th century and refers to the "Great Chiefs" of the Kadohodacho, The Caddo Indians who may not have come into the valley proper until the late 1700's.

For convenience and clarity, this paper on the Caddo River Watershed has been divided into five sections to correspond with the five distinct lifestyles in the Caddo River region. Those categories are the archaic period, the Caddoan period, the pioneer period, the Civil War and Reconstruction period, and the period of the twentieth century.

ARCHAIC PERIOD

The first settlers of the Caddo region were gatherers and hunters dating from 900 to 1000 BC. In the 1920's S. D. Dickinson and S. P. Dickinson made the first serious study of the archaeology of the Caddo River. Among their findings was a type of chipped, double-bladed axe, typical of the Fourche Maline culture of approximately 100-800 BC. This find was made at Buttermilk Springs and in the area of Collier Creek (Dickinson 1983, Pers. Corresp.). Prior to 1939 (and the DeSoto Commission), the accepted thesis was that the early inhabitants of this region were Caddo of the northern area of the Confederation.

However, V. L. Huddleston dated the site of Brushy Creek, one of the sites excavated in the 1920's, at 1000 BC to the first century AD and identified it as part of the Marksville Culture (Huddleston: 1943). That culture, said Huddleston, was followed by Cole's Creek culture, characterized by its pottery and mound burial technique with the corpse in a flexed position.

CADDOAN (HISTORIC TRIBES)

The year 1400 established a new culture in Arkansas as evidenced by the changes in the type of pottery used. Shell tempered pottery was introduced into the Arkansas region around that time, and from Tom Collier's discoveries in 1925 and Dickinson's excavations in 1927, evidence that such a culture did exist along Collier Creek and Buttermilk Springs was found (Dickinson: 1980). The narratives of the early explorers - Joutel, DeSoto, LaHarpe, and Bossu - who came into Arkansas, provide historical evidence of these Indians after 1400. This epoch is often referred to as the Caddo period, but some evidence exists which could suggest that this group of Indians was not of the ten tribes of the three-group confederation known as the Caddo.

In 1541, DeSoto refers to the Indians in the area of the Caddo Gap region as Tulas. In 1939, Dr. Swanton, from the Smithsonian, along with Colonel John R. Fordyce on the DeSoto Commission, thought these Tula Indians to be Caddo Indians. De La Vega, a Peruvian who reported the DeSoto expedition, wrote that the Tula encounter occurred on a plain or level area between two rivers. Dickinson contended that the level plain at Caddo Gap between the Caddo River and Collier Creek would fit that description (Dickinson: 1980). Huddleston described these Indians as having a ". . .cradle board deformation of the skull with a low, flat forehead" (Huddleston: 1943). Dickinson noted that their long, pointed heads were ". . . deformed in infancy by binding" and that the adults ". . .had their faces tattooed, even extending the decoration over the lips" (Dickinson: 1980).

The Kadahodacho group did inhabit large areas in southwest Arkansas, but there is some question regarding their presence in the Caddo River region as early as the 16th century. There were three major groups of the Caddo in what is now Arkansas, Louisiana and Texas. The Kadahodacho settled in Arkansas; the Natchitoches settled in northwest Louisiana; and the Hasinai inhabited east and central Texas (Lyon: 1952). Joutel observed in his journal in 1687 that in the region of southwestern Arkansas the Indians lived in cottages which were round at the top like beehives. These communal dwellings were some sixty feet in diameter with a fire pit in the center of each hut. After the group or tribe moved out, the houses were burned (Huddleston: 1943). Joutel's route was probably from the Texas Gulf Coast to Fulton, to near Camden, and to Arkansas Post by July 24, 1687. It is possible that the Indians that he saw were in the region south of the Caddo and not on the Caddo proper (Lyon: 1952).

In 1673, Marquette had heard of the Caddo to the West, and in 1691 the Spaniards sought to Christianize the Hasinai in the area west of Arkansas. In 1763 Athanase de Mezieres reported meeting Chief Tinhioven of the upper Caddo (Lyon: 1952).

By the early 1800's, most of the Indians in the area of the Caddo had migrated to Texas, and after a series of treaties (1818, 1820, 1824, 1833, 1835, 1843) the Quapaws were moved through the Caddo River region to the Red River Valley in Louisiana and later to Oklahoma. By 1859, then, there were evidently no Indians living in the area (Herndon: 1922).

PIONEER PERIOD

During the Spanish period, after DeSoto in 1541, the French sent expeditions from Fort Miro (Monroe, Louisiana) into the Ouachita River Valley. Several place names in the Caddo region may be variations of the French influence. Glazy Po and Greasy Cove may be variations of Glaise a Paul, a French hunter of that period (Dickinson, P. Corresp.).

In 1805, the Territory of Louisiana was divided, and the Caddo-Ouachita region was part of the District of New Madrid. In 1806, that lower region of the New Madrid District became the District of Arkansas, and in 1809 there is reference to the white settlers named Barkman on the Caddo, five miles east of what is now Arkadelphia (Butler: 1906).

After 1809 settlers came into the region of the Caddo from Virginia, North Carolina, South Carolina, Georgia, Mississippi, Tennessee, Missouri and Kentucky. In 1785 there were 196 white settlers in all of Arkansas, and in 1820 there were 14,225 (Lacey: 1908).

In 1811, in what became Clark County, John Hemphill operated one of the first industries on the Ouachita, just south of its confluence with the Caddo, known as the Salt Works, a site where the Caddoes had already had a similar operation (WPA: 1941). In addition to a salt works, a brickyard, owned by Bean and O'Baugh, appeared at Blakeley town in 1822; a cotton gin in 1830; and a gristmill in 1832. The first court in the area was held in the county of Clark in 1818 at Jacob Barkman's house, and on June 14, 1819, a courthouse was named at Adam Blakeley's place (Butler: 1906).

What is now Montgomery County, Arkansas, was being settled as early as 1812 and was organized into a county in 1842. The early settlements were on the various streams, including the Caddo River. The settlers were mostly from Kentucky, Tennessee and Missouri. Some of the names of this group were Collier, Whittington, Boles, Boggs, Cunningham, Gaston, Mayberry, Polk and Fisher.

Pike County was created in 1833, with its government seat at Zebulon, now Murfreesboro, at the home of Pascal C. Sorrels. These three counties - Clark, Montgomery, and Pike - comprise the area of the Caddo River.

According to Goodspeed's description of the area, tributaries of the Caddo, such as Brushy Creek, Collier Creek, Williams Creek, Walnut Fort, etc., gave the Caddo the power to drive the heaviest machinery for milling and manufacturing. His description was:

> The water power which can be utilized in running machinery is immense, and when transportation facilities are furnished our almost unlimited forests of valuable timber that cover the wild, uncultivated lands, will be a source of wealth to the lumber men and manufacturers who have the enterprise to avail themselves of so important a factor in the natural resources of Montgomery County (Goodspeed: 1890).

As early as 1890, then, there were visions of using the Caddo for commerce. However, throughout the 19th century, the major sources of income for the settlers in the Caddo River region were the small farm homesteads and the merchant traders. One of those early settler farmers included William Hopper (for whom the community of Hopper is named) who came to Clark County and to Caddo Gap in 1849-51. He owned 1,000 acres in Montgomery County and was a merchant and member of the Farmer's Alliance, New Hope Lodge (AFAAM) and the Christian Church. Alex Thornton was a miller at Black Springs who migrated from Georgia to Texas and then to Arkansas in 1879. Enterprising, he owned a water saw, gristmill, and cotton gin. He was a Baptist.

Martin Collier was one of the oldest settlers in Montgomery County. Coming to Arkansas from Kentucky, Collier settled on land possibly given to him as a grant for having served in the War of 1812. The Federal government set aside 1,162,880 acres in Arkansas to be given as military bounty grants to soldiers who fought in the War of 1812, and by 1830, some 1,037,120 acres were claimed (American State Papers, Public Lands, Vol. 6, p. 911). Adventuresome, yet hospitable, Collier soon became friends of the Indians already living just north of the Gap. His great-great-grandson, Argus Dutton, told how he made friends with the Indians. It seems Martin Collier was somewhat of a musician. He played the fiddle, and the Indians heard the music coming from the Collier homestead and were lured to the "heavenly sound."

Collier's death provides an interesting bit of folklore. One evening, while playing his violin, he saw a light dancing on a ridge. Believing the light was an omen of his death, he put the violin and music away and had his daughter sing to him until he died. Martin Collier's grave, marked only by rocks, is located on a ridge overlooking the creek that bears his name and the homestead that bears his energy and life. Jefferson Collier, one of the descendants of Martin Collier, was born on Collier Creek. Collier told an interesting story of his mother. She knitted a pair of socks out of wool obtained from the head of a buffalo and she sold the socks to William Barkman for a pair of cotton cards and ten dollars. Adam Blakeley owned the first store in what became Clark County in 1809. Butler, Gross, and Rowston were merchants in Black Springs at about the same time.

A period of rapid growth occurred in the region from 1850 to 1860 with the development of county government seats, churches, newspapers and schools. In addition, newspapers such as The Southern Standard started in 1859, and the Pike County Sentinel began in 1855.

The predominant churches in the area were the Methodist, the Baptist, the Christian and the Church of Christ. The Baptist State Convention was organized in Tulip, Dallas County, Arkansas, in 1848 and began to consider a male college, not realized until 1886 in Arkadelphia. The Methodist "Wachita Conference" first met in 1854 in Washington, Hempstead County, Arkansas, and sent Theron Hunt to Montgomery County in 1854-55 as a Methodist Minister (Whaley: 1911).

In 1854 the Methodists had 260 members in the Montgomery County area and 326 white and 62 blacks in Arkadelphia. "White people, and Negroes, masters and slaves, were members of the same congregation, and they worshipped together at the same time and place" (Whaley: 1911).

THE CIVIL WAR AND RECONSTRUCTION

The Civil War brought scores of people through Caddo Gap, including Federal troops on their way from Fort Smith to south Arkansas. Many of the people of the area were displaced because of the war. The family of George T. Smith, ancestor to Alexander Turrentine in Clark County, provides one example. Smith moved to Clark County in 1837, then to Yell County just before the war. After his death, his widow, Zeralda, and family migrated, with considerable hardships, to Paraclifta, Sevier County, Arkansas, by way of Caddo Gap (Turrentine: 1951).

In 1863 Isaac Anderson's youngest son, C. T., went to serve in the Confederate Army in his father's place. He left his home in Fulton and traveled to Washington and to Hot Springs by way of Caddo Gap.

After difficulties with Federal authorities, Albert Pike retreated to the Little Missouri River, west of Caddo Gap, where he is supposed to have written Morals and Dogma and revised the Masonic Ritual. Federal guerillas sought Pike and tracked him to the region of Caddo Gap. While inquiring about his whereabouts in a store, the Federals were averted until Pike was warned of their presence. He fled shortly before the guerillas discovered his cabin and destroyed his library by throwing the books into the Caddo River (Dickinson, 1983 Pers. Corresp.).

During the Civil War, various Confederate manufacturing enterprises were begun in the region. The Arkadelphia Salt Works was leased to the Confederate Army. An ordnance plant, located on the present site of the Merchants and Planters Bank in Arkadelphia, manufactured knapsacks, cartridge boxes, and powder under the leadership of a Captain Pulley. The lead for the material was shipped to Arkadelphia from Joplin, Missouri, and possibly through the Caddo Gap region. The charcoal, saltpeter, and sulphur for that operation came from Pike and Montgomery Counties (McKenzie: 1908). Cloth was manufactured in the area of Murfreesboro, Pike County, Arkansas, in 1862 and later moved to Arkadelphia, in 1885.

It was after the Civil War that the Caddo River region experienced a period of growth in small farm products. The farmers during this period grew fruits, cereal crops, cotton, potatoes, and rice (Goodspeed: 1890 and Butler: 1906). One of the early farms on the Caddo River, up from Barkman's, was that of William Browning McFadden, near the site of the present town of Amity, which was settled in 1842 (Reynolds: 1906). Between the time of the Civil War and 1870 more settlers, such as Alfred and Jane Jones and Isaac Runyan, came to the area.

After the Civil War, however, the area began to change from an agrarian culture to one of mining, railroading and timber industry. In Montgomery County, for example, silver, copper, chloride, quartz and manganese were mined in the late 19th century and early 20th century. In 1893 the railroad from Gurdon to near Caddo Gap was completed on its way to Fort Smith.

Before the Civil War, few lumber mills and no planing mills were located in the region of the Caddo River (McKenzie: 1908). But with the coming of the railroad in the late 19th century came the lumber mills. Located at Glenwood, Amity, and Rosboro, the lumber mills became major sources of revenue for the Caddo region. In Rosboro, Pike County, Arkansas, a mill was started in 1907 as a part of the lumber complex that included mills in Delight, Ozan, Graysonia and Prescott (Pike County Heritage Club, 1978, P. Corresp.). In 1906 the A. L. Clark Lumber Company opened in Glenwood, and the A. Alexander Mill in 1910.

THE TWENTIETH CENTURY

With the coming of the lumber companies and the railroads came the demise of the small farm and the dependence on an industrial economy. Small, single-owned farms began to be replaced by company-owned houses and stores. The single merchant was replaced by the commissary and the company-owned store. However, a new lifestyle began to develop in the early twentieth century that would become the dominant source of economics through the 1920's and 1930's, namely tourism. As early as 1920, many tourists came to the Caddo Gap region from Louisiana and Texas to get away from the cities of Dallas, Houston, Shreveport, Baton Rouge and New Orleans. Hotels, inns, and rooming houses began to develop along the highway and creeks in the region. That interest in the river as a tourist attraction was the seed that led to the development of the Caddo River as a recreational spot with Lake DeGray in the 1960's and early 1970's.

The dream for a lake occurred as early as 1909 with Harvey Couch's dream of a dam across the Caddo. Inspired by railroading, Couch wanted to expand his enterprises from Louisiana to Oklahoma by way of the region of Caddo River in Arkansas. With the coming of the Water Supply Act of 1958, the dam on the Caddo was almost assured. With it came the end of a lifestyle that had existed from the archaic period to the period of the small farmer in the 19th century. It would usher in a new lifestyle which was more transient and less attached than any that had heretofore lived on the Caddo River. The new reservoir was a clear indication that two worlds existed, "the one here and the one there." For those who chose to remain, the decision was absolute. Argus Dutton, who chose to remain, said, "Some of us never had enough sense to leave Collier Creek."

Aware of that impending change, Rayburn, principal of Caddo Gap High School and lover of folklore, said in 1954, of the Caddo, "What an ideal place to stretch one's sentiments and weave colored threads into the tapestry of one's dreams" (Rayburn: 1954). Argus Dutton, descendant of Martin Collier, said in 1983, "We've always had enough good, pure, cold water on Collier Creek. We also have a lot of rattlesnakes, but we've got enough rocks to kill them with" (Dutton: 1983). Maybe the romantic lifestyle still lives in the midst of a real world.

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CADDO RIVER WATERSHED - HYDROMETEOROLOGY[1]
AND RUNOFF WATER QUALITY ANALYSIS

by

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INTRODUCTION

The information described herein briefly summarizes four years of research on the Caddo River Watershed from 1974 thru 1978. An understanding of the watershed's hydrologic and runoff water quality response during storm events was the primary objective of this investigation.

PHYSIOGRAPHY

The Caddo River watershed is located in parts of five counties (Montgomery, Pike, Clark, Garland, and Hot Springs) in south-central Arkansas between latitudes of 33°8' - 34°33' and longitudes 93°2' - 93°57'.

The Caddo River drains the south flank of the Ouachita Mountains in south-central Arkansas and enters the Quachita River just north of Arkadelphia, Arkansas (Figure 1). It flows in a generally southeast direction 125 km. From headwaters at over 400 m msl, the river falls to 165 m msl at Glenwood to 130 m msl at Highway 84 to 54 m msl at its confluence with the Ouachita River (Figure 2) (Ford and Stein 1984). It has a drainage area of 1269 km^2 of which 1173 km^2 are controlled by DeGray Dam; an additional 70 km^2 are controlled by the regulating dam (USAE-VD 1966).

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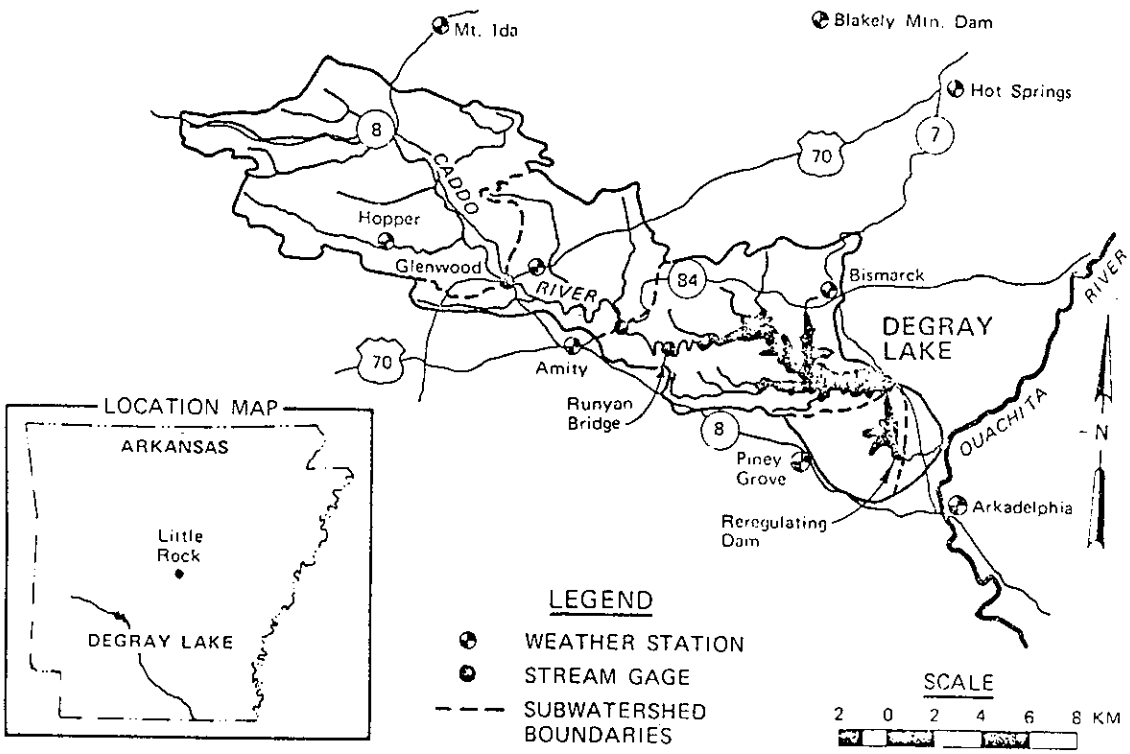

Figure 1. Caddo River Watershed, Arkansas

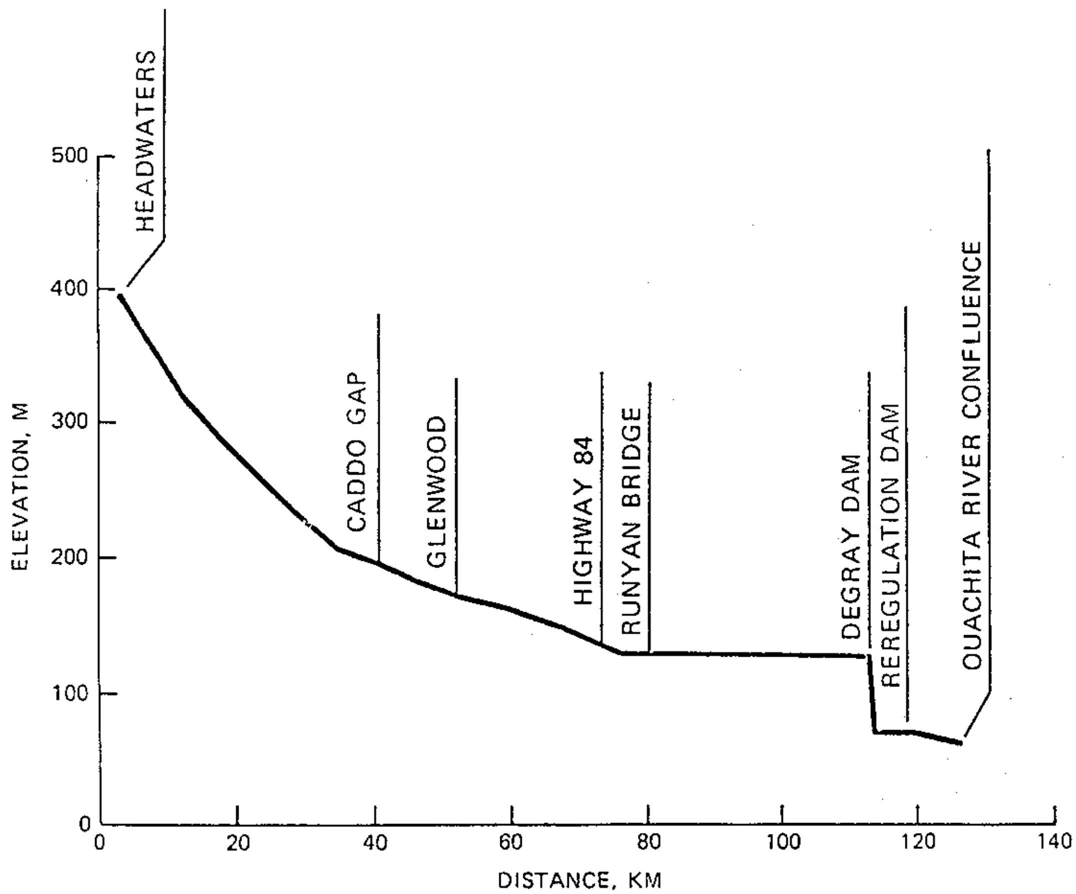

Figure 2. Profile of Caddo River

SOIL ASSOCIATIONS AND LAND USE

To the northwest of the dam, the watershed lies in the Ouachita Mountain section of the Ouachita physiographic province. The watershed lies within two subsections of the Ouachita Mountain section. Above Glenwood, the Novaculite Uplift subsection is characterized by long eastward-trending, even-crested mountains and flat intermontane basins. Ridges in the Caddo River region exceed 600 m msl. Southeast of Glenwood, the Caddo River passes through the Athens-Piedmont Plateau subsection. This area is characterized by non-mountainous east-west ridges rising 75 m above the intervening valleys (Albin 1965). Below the dam, the Caddo River enters the Gulf Coastal Plain, which is characterized by a rolling to hilly terrain over unindurated sedimentary materials.

The predominant soil associations in the Novaculite Uplands region are Pickens and Carnasaw with areas of Pickwick near the streams (Figure 3). These well-drained, acid soils range from shallow on the slopes to moderately deep on the ridge tops and floodplains. Usually brown loam surface soils overlie yellowish silty clay loam or yellow-red silty clay subsoils which grade into clay. Gravel may occur on the slopes and in the valleys. The bedrock is usually steeply inclined fractured shale or sandstone. Outcrops of chert, slate, novaculite and limestone also appear (Perrier et al. 1977).

In the Athens-Piedmont Plateau region, the Sherwood and Carnasaw associations predominate. These well drained, acid soils have dark grayish brown to brown loam or fine sandy clay loam that is gravelly in some areas. These overlie yellowish-red or red sandy clay loam or clay loam subsoil. This region is primarily underlain by shales bounded by beds of sandstone. These formations are thick and generally impermeable.

The Caddo River watershed is mainly rural with only about 1 percent being urban or developed (Figure 4). About 30 percent is classified as agricultural, mostly pastures in the Caddo River floodplain and tributary valleys. Crops include corn, soybeans, and hay. The remainder of the watershed is forested; 50 percent deciduous-mixed and 18 percent coniferous and others. This includes areas which have been clear cut and block planted (Perrier et al. 1977).

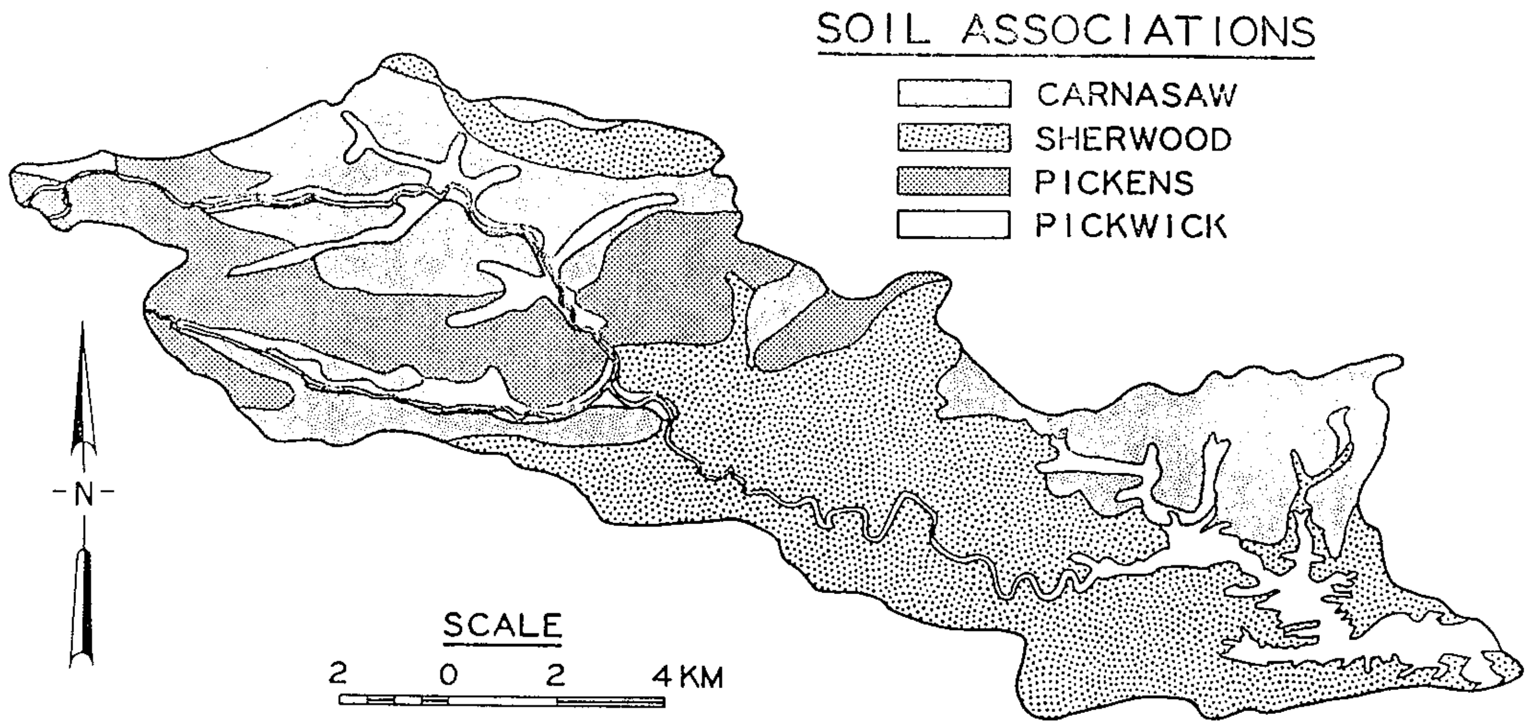

Figure 3. Soil association map of Caddo River watershed

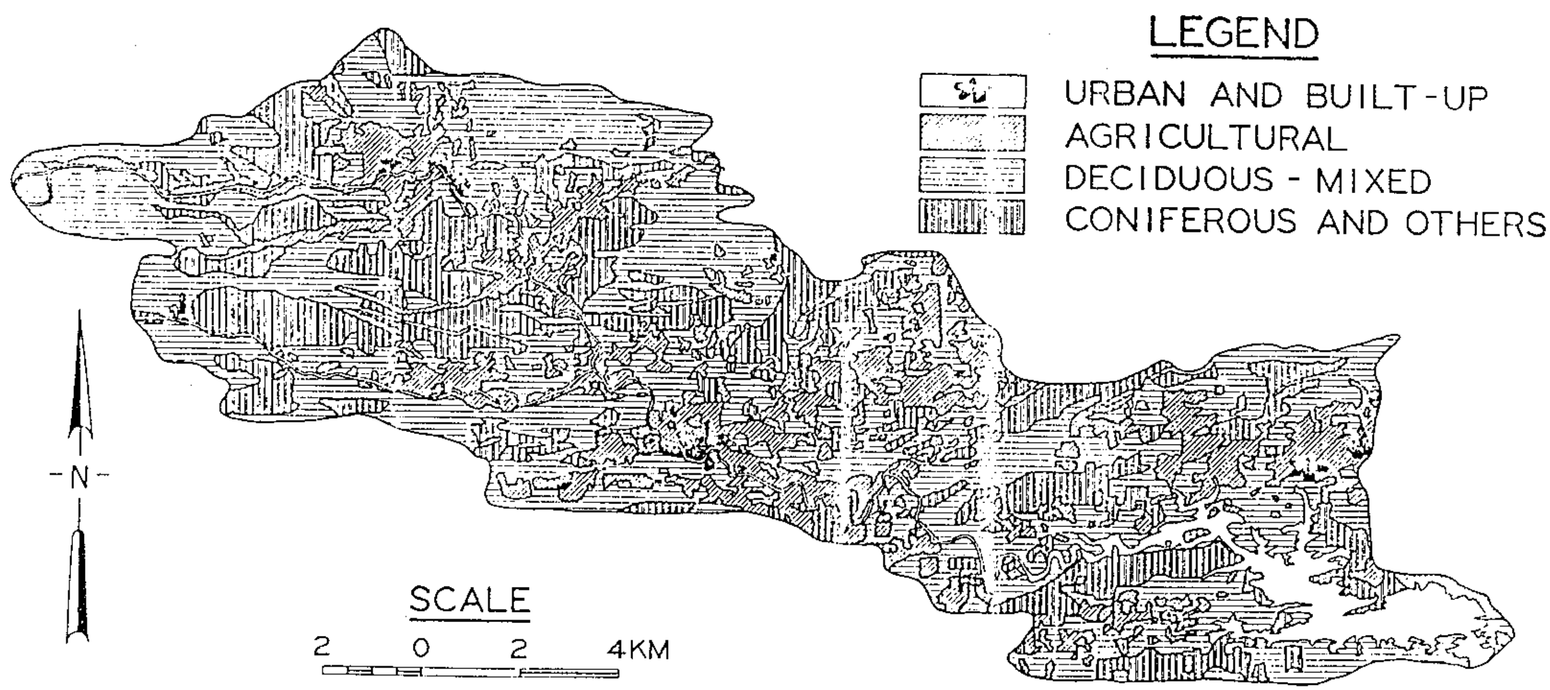

Figure 4. Land use map of Caddo River watershed

CLIMATE

National Weather Service stations do not record temperature within the Caddo River watershed. Daily temperture extremes are recorded at Arkadelphia, about 12 km south of the dam and 60 m below the reservoir surface in elevation. Nearby Malvern and Hot Springs, both about 30 km from the reservoir, and Mount Ida, about 50 km from the reservoir but only 7 km outside the upper watershed, also have recorded temperature extremes. Temperature and other meteorological data are recorded at the Little Rock airport every three hours.

DeGray Lake is exposed to a modified continental climate. Warm, humid air from the Gulf of Mexico causes the summers to be long with temperatures occasionally exceeding 40°C. Winters are generally short and mild but often influenced by polar air masses which have dropped the temperature as low as -20°C (Reinhold 1966, 1969). Mean monthly temperatures at Arkadelphia range from 6°C in January to 28°C in July with a mean annual temperature of 17°C (Ford and Stein 1984). Temperatures in the upper watershed should be comparable to those at Mount Ida, whose monthly averages are 1° to 1.5°C lower than those at Arkadelphia.

The average wind speed is greatest in late winter and early spring and least in mid to late summer. Winds from the southwest to south predominate, especially during the summer months. During the winter a more even distribution prevails. The average barometric pressure is about four millibars higher in the summer than in the winter (Ford and Stein 1984).

On a long-term average, precipitation is fairly well distributed throughout the year with rainfall occurring only 20 percent of the time (Westerdahl 1981). About one-third of the annual precipitation falls from March through May. The spring and winter precipitation is mainly associated with frontal systems from the northwest. During the summer period, precipitation typically occurs as rain from convective thunderstorms and southerly winds, whereas in winter, precipitation frequently occurs as rain from frontal storms with winds from the northwest. Summer rains are mostly scattered showers and thunderstorms which are brief but often intense. During August, September, and October there may be long periods of little or no rain. Precipitation usually increases in the late fall. From January through May frontal storms produce the bulk of precipitation. Some winter precipitation falls as freezing rain, sleet, or snow. Snow occurs from one to four times per year but rarely stays on the ground more than a few days (Reinhold 1966).

HYDROLOGY

While streamflow is intrinsically related to precipitation, many other factors are also involved. Runoff may vary from 15 to 90 percent for individual storms, depending on the season, antecedent soil moisture, and ground-water conditions. Annually about 45 percent of precipitation falling in the Caddo River drainage area enters the Caddo River. During the late spring and summer, the lower watershed contributes the larger portion of streamflow to DeGray Lake while in late fall and winter, the upper watershed contributes more runoff. Hines (1965) indicates that a significant amount of streamflow in the Caddo River is maintained by the discharge of ground water from storage during periods of little precipitation.

The peak discharge frequency curve for the Caddo River at Glenwood shows that the mean annual peak storm flow is about 500 cubic metres per second (cMs). The extreme storm flows during the period of record, 1940-1976, were 2500 cMs in 1968 and 80 cMs in 1941 which correspond to water depths of 10 m and 3 m, respectively.

Water Quality Constituent Loadings

The types of storms may also affect the stream water quality. Washoff constituents in the runoff from different locations are likely to vary with soil types, land use and land use management practices. Also, water from the upper watershed, having been in the river several hours, may experience temperature changes and undergo chemical changes in constituent composition resulting from chemical and biological reactions within the stream channel. The objectives of this study were to statistically analyze and develop functional relationships for water quality constituent loadings during storm event runoff on the Caddo River drainage basin.

Field Sampling and Laboratory Analyses

Precipitation measurements were obtained from a rain gage located at Glenwood, AR. Water stage recorders and appropriate rating curves were used to determine the stream discharge at both the Glenwood and Highway 84 water quality sampling sites.

Water quality sampling for the 13 storm events began in January 1975 and continued through June 1977. Table 1 presents some of the storm characteristics for the specific events sampled. During this period, 479 water samples were collected from the Caddo River. Each sample consisted of a one-liter aliquot of storm water collected from the river bank at a point where the stream appeared to be well mixed. The samples were collected at approximately one-hour intervals with the sampling intervals gradually increasing during the hydrograph recession. An additional one-liter aliquot was collected to determine the sediment content of the runoff. Each sample was stored at 4°C to minimize chemical and microbiological transformation.

Table 2 lists the parameters and analytical method for streamflow water quality characterization used in this study. The pH, temperature, alkalinity, dissolved oxygen, specific conductance, and turbidity were analyzed at the sampling site. The remaining analyses were made on the refrigerated samples within 24 hours in the water chemistry laboratory at Ouachita Baptist University, Arkadelphia, AR.

Statistical Analysis

Statistical methods were used to examine all data using the SPSS computer programs (Nie et al., 1975). In evaluating stream water quality data, it is important to determine the extent of concentration changes as well as correlatable relationships.

Discriminant Analysis

Discriminant analysis was performed on the two data sets obtained at the Glenwood and Highway 84 sampling sites. The mathematical objective of the discriminant analysis was to weight and linearly combine variables to find those variables that discriminate between data sets. Nunnally (1967) suggests that this analysis is employed when the groups are defined *a priori*, and the purpose of the analysis is to distinguish the data sets from one another on the basis of their score profiles.

Factor Analysis

Factor analysis was employed to compare the measured parameters to identify underlying relationships. A central aim of factor analysis is the orderly simplification of a number of interrelated measures (Child 1975).

Table 1

Storm Characteristics by Date, Sampling Gage Site, Peak Stream Flow, Rainfall Amount, Storm Interval, and Rainfall Duration

Storm Date	Sampling Gage Site	Peak Flow cMs	Storm Rainfall cm	Days from Previous Storm	Rainfall Duration Days
31 Jan 75	Glenwood	383.4	10.08	5	2
19 Nov 75	Glenwood	14.2	3.10	9	1
5 Feb 76	Glenwood	5.7	2.74	10	1
3 Mar 76	Glenwood	6.5	0.89	13	2
28 Mar 76	Glenwood	22.2	3.88	3	4
19 Apr 76	Glenwood	12.2	3.66	3	3
12 May 76	Glenwood	23.6	7.45	6	4
15 Jun 76	Glenwood	38.7	8.72	8	4
31 Aug 76	Hwy 84	7.0	1.78	1	4
24 Oct 76	Hwy 84	197.5	5.52	3	2
2 Feb 77	Hwy 84	15.7	2.08	9	1
3 Mar 77	Hwy 84	532.4	10.26	6	2
16 Jun 77	Hwy 84	348.6	11.18	6	6

Table 2

Sample Parameters Measured and Method of Analysis

Parameter	Analytical Determination	Analytical Method
Sodium, Na	Atomic Absorption	Perkin Elmer (1971)
Potassium, K	Spectrometer	Perkin Elmer (1971)
Calcium, Ca	Spectrometer	Lee (1967)
Magnesium, Mg	Spectrometer	Perkin Elmer (1971)
Silica, Si	Heteropoly Blue	Std. Methods (1974)
Nitrate, NO_3	Ultraviolet	Std. Methods (1974)
Ammonia, NH_4	Orion Specific Ion Electrode	Orion (1971)
Total Kjeldahl Nitrogen (TKN)	Digestion-Titration	Std. Methods (1974)
Alkalinity	Methyl Orange Titration	Std. Methods (1974)
Total Organic Carbon (TOC)	Combustion-Electrode	Std. Methods (1974)
Total Phosphorus	Ascorbic Acid Reduction	Std. Methods (1974)
Ortho Phosphate	Ascorbic Acid Reduction	Std. Methods (1974)
Dissolved Oxygen	Electrode Method	Std. Methods (1974)
Total Solids	Glass Filtration	Std. Methods (1974)
Dissolved Solids	Glass Filtration	Std. Methods (1974)
Suspended Solids	Glass Filtration	Std. Methods (1974)
Temperature	Electrometric Thermometer	Std. Methods (1974)
Turbidity	Nephelometric	Std. Methods (1974)
Total Bacteria	Membrane Filters	Std. Methods (1974)
Fecal Coliform	Membrane Filters	Std. Methods (1974)
Total Coliform	Membrane Filters	Std. Methods (1974)
Fecal Streptococci	Membrane Filters	Std. Methods (1974)

Also, it was used to reduce the number of parameters measured so that further sampling efforts could be more efficient.

Regression Analysis

Bivariate regression equations were determined for the 25 water quality parameters versus flow. Also, multiple regression was used to determine predictive loading relations for a particular metal, nutrient, and bacterial group. In addition, information was gained on the variation of the predictive accuracy of the selected multiple regression equation.

RESULTS AND DISCUSSION

The sampling site was changed from Glenwood to Highway 84 bridge prior to the August 31, 1976 storm event. In order to utilize and evaluate all the data that had been sampled, discriminant analysis was used to compare the two data sets to determine if chemical stream loading changed between the two stations. It was assumed that both stations would compare favorably as the soil and land-use characteristics appeared similar to those found elsewhere within the watershed. Due to missing data problems, only 13 of the original 26 parameters were selected for analysis. These 13 parameters produced a very high degree of similarity with a Chi-square = 15.7 and 13 degrees of freedom. To determine a discriminant function only 6 parameters were used, i.e., NO_3-N, PO_4, total phosphorus, Ca, suspended solids, and turbidity. Approximately 36 percent of the two known groups were correctly identified as members of the group to which the parameters actually belong. However, this latter analysis did show the relative contribution of NO_3-N and suspended solids to that function with standardized discriminant coefficients of -0.55 and 0.44, respectively. The contribution of the added watershed area from Highway 84 to the concentration of NO_3-N is less than that contributed from the area above Glenwood; whereas, the increase in suspended solids is greater at Highway 84 than that contributed from the area above Glenwood. The change in concentration of NO_3-N at Highway 84 is lower than at the Glenwood sampling site. This difference is attributable to land use within the region, e.g., more forest lands, less livestock and poultry. In addition, the increase in suspended solids may have been caused by a difference in public versus private forest land management as well as geological development of the Athens Piedmont Plateau and the accompanying differences in soils.

Factor analysis was run comparing nutrients, metals, and bacterial data sets with flow, temperature, pH, specific conductivity, alkalinity, turbidity, total solids, dissolved solids, and suspended solids. It was shown for the three data sets that from rotation to terminal factors, specific conductivity and alkalinity account for similar amounts of variance. Alkalinity was selected for further analysis. Also, total solids, dissolved solids, suspended solids, and turbidity represent similar amounts of variance. Because turbidity could be measured onsite it was also selected for further analysis. In all three factor runs on the data sets, alkalinity was found to be a complex variable and was a measure of more than one theoretical factor.

The range of concentration and the detection limit for the 25 water quality parameters and 13 storm events are shown in Table 3. The low levels of the constituents are indicative of a watershed that has not been greatly damaged by man altering his environment. However, the different ranges of loading indicate the variation that is attributed to the stochastic nature of storm events.

Bivariate regression equations were determined for the 25 water quality parameters versus streamflow in rectangular form as well as semi-log and log-log relations. Table 4 presents the correlation coefficients for the various mathematical expressions. Table 5 presents the "best" form of the bivariate regression equations as determined by the highest correlation coefficient. The number of samples, N, varies because only 17 parameters were measured during the 31 Jan 75 storm and occasional equipment failures account for the remainder of the missing values. As shown in Table 5, 13 of the parameters had correlation coefficients greater than 0.50. The poorest bivariate correlation was the log-log relation of NH_4-N with R = 0.13 and the best correlation was the log-log relation of calcium with R = -0.90. The low correlations of some of the parameters are similar to those discussed by Hobbie and Likens (1973) and Nelson (1970) for concentrations of nonpoint pollutants versus water discharge rates. Extreme-values were a major contribution to low correlations for some of the parameters; but none of the statistics of extremes (Lin 1976) were performed on the data set. Sodium, Na, and potassium, K, load inversely which suggests different origins for these two components. The analysis of ground water in the Interior Highlands of the Ouachita Mountains (Albin 1965, Plebuch and Hines 1969) shows sodium bicarbonate to be a

Table 3

Range of Concentration, Units, and Detection Limit of the Water Quality Parameters in Relation to Each Storm Event

Water Quality Parameters	Units	Detection Limit	31 Jan 75	19 Nov 75	5 Feb 76	3 Mar 76	28 Mar 76	19 Apr 76	12 May 76	15 Jun 76	31 Aug 76	24 Oct 76	2 Feb 77	3 Mar 77	16 Jun 77
NO_3-N	mg/l	0.01	0.07-0.46	0.05-0.34	0.06-0.19	0.04-0.27	0.09-0.31	0.09-0.28	0.12-0.63	0.04-0.43	.04-0.19	0.04-0.62	0.04-0.21	0.23-1.20	0.05-0.62
NH_4-N	mg/l	0.01	0.08-0.11	0.02-0.06	0.07-0.09	0.08-0.19		0.00-0.20	0.00-0.50		0.00-0.04	0.00-0.12	0.00-0.04	0.00-0.04	0.00-0.09
TKN	mg/l	0.02	0.42-1.33	0.17-0.81	0.20-2.60	0.25-0.53	0.30-3.10	0.50-1.90	0.30-2.80		0.22-0.85	0.42-1.70	0.12-0.43	0.47-1.63	0.14-2.54
Total P.	µg/l	5.0	18-214	14-103	13-48	4-33	5-45	7-65	6-79	19-55	11-35	16-76	3-20	11-81	8-529
Diss. P.	µg/l	5.0	1-50	8-39	2-38	6-20	2-39	1-38	14-97	9-59	11-30	9-42	5-16	3-55	8-135
Ortho P.	µg/l	5.0		4-30	2-19	1-10	2-32	4-35	10-44	1-53	10-28	16-49	3-14	8-43	12-96
TOC	mg/l	0.8		0.4-2.1	0.8-2.2		1.0-3.8	0.6-3.2	1.9-13.0	0.7-6.6	3.0-6.5	5.2-21.2	0.8-6.0	0.6-20.1	3.6-18.3
Si	mg/l	0.1		6.0-7.5	5.3-7.1	2.2-4.4	6.6-8.2	6.5-8.0	4.6-9.0	7.2-9.4	6.8-8.4	6.0-9.2	6.3-7.9	3.7-8.5	1.8-6.6
Na	mg/l	0.002	0.9-1.9	0.9-1.2	1.8-2.5	1.7-2.0	1.5-2.3	1.5-1.9	1.3-1.8	1.3-2.0	1.8-2.5	1.2-2.0	1.8-2.5	1.0-1.8	0.8-2.1
Ca	mg/l	0.4	2.0-13.0	13.0-20.0	10.0-15.0	9.4-12.0	9.8-11.1	11.0-15.0	4.1-12.0	9.4-21.0	10.0-18.0	4.6-18.0	8.2-11.2	1.5-7.1	2.8-13.2
Mg	mg/l	0.2	0.6-1.9	1.6-2.2	1.6-1.8	1.3-2.3	1.8-2.0	1.9-2.2	1.2-2.2	1.3-2.3	1.4-1.9	1.3-2.5	1.6-1.8	0.6-1.6	0.7-1.8
K	mg/l	0.4		0.7-2.3	0.6-2.0	0.6-1.1	0.6-1.4	0.7-1.3	0.7-2.6	0.6-3.5	0.9-2.0	0.8-2.3	0.6-1.6	0.6-3.6	0.9-2.6
Total Solids	mg/l	2.0		55-159	31-105	58-85	37-76	34-68	40-572	46-227	54-85	31-245	50-108	35-400	51-817
Diss. Solids	mg/l	2.0		39-62	38-77	33-48	36-57	33-58	34-95	32-83	50-74	14-71	24-39	24-126	35-60
Susp. Solids	mg/l	2.0	10-24.0	6-106	2-47	13-45	1-21	1-22	6-520	4-172	1-34	5-194	11-78	1-294	3-757
Diss. Oxygen	mg/l	0.1		7.2-8.8	6.7-9.2	7.8-9.5	7.8-10.6	6.0-8.5	8.1-9.7	6.2-8.8	7.1-9.0	8.3-11.0	9.5-12.7	7.6-11.4	6.1-10.0
Turbidity	NTU*	0.4	5.3-77.0	0.8-61.0	6.4-52.0	1.7-22.0	2.0-19.0	1.6-20.0	4.7-315	1.4-130	1.7-44	1.8-100	3.1-28.0	0.8-15.5	1.9-170
Alkalinity	mg/l	4.0		39-59	31-44	44-57	29-43	36-45	16-37	22-61	36-63	15-56	24-37	1-24	14-48
pH		0.01		7.2-8.0	6.9-7.5	6.9-7.4	7.0-7.5	7.0-7.3	6.4-7.4	6.9-8.5	7.0-7.5	6.7-7.4	6.8-7.4	5.5-6.9	6.2-7.1
Spec. Cond.	µmhos	7.0		85-129	59-119	58-80	49-74	66-80	41-90	71-148	72-149	35-100	42-73	24-38	20-92
Temperature	°C	0.1		9.4-14.0	4.0-10.6	12.1-19.1	11.3-16.4	16.1-20.0	16.0-21.5	18.9-28.0	19.5-24.8	11.5-15.6	2.8-5.2	8.8-12.8	18.9-24.5
Total Bact.	cells/ml		10-5500	26-11,400	25-6300	94-2300	51-4700	20-14,300	105-31,000	530-75,000	460-17,900	72-19,800	38-3100	220-28,000	56-38,800
Fecal Coli.	cells/ml		0.6-567	0.5-970	1.3-296	1.9-118	1.1-840	3.0-1800	13-2440	3-7800	0.69-1.8	0.49-146	0.3-38	0.19-57	0.08-600
Fecal Strp.	cells/ml			1.4-1010	2.2-840	3.3-370	0.6-330	2.0-530	9-6600	2-8100	1.5-56	1.10-310	0.02-6.8	0.20-73	0.26-5300
Total Coli.	cells/ml			1.2-341	1.3-104	11-180	0.4-180	3.0-260	11.0-1800	16-1100	160-3,600	32-3400	18-960	24-7500	50-11,500

* NTU = Nephelometric Turbidity Units

Table 4

Correlation Coefficients R for Various Forms of the Independent and Dependent Variables

Water Quality Parameter	Rect C/Rect Q*	Log C/Rect Q	Rect C/Log Q	Log C/Log Q
NO_3-N	0.43	0.40	0.64	0.65
NH_4-N	0.01	0.01	0.11	0.13
TKN	0.32	0.36	0.27	0.36
Total P	0.37	0.38	0.43	0.50
Diss. P	0.15	0.12	0.23	0.24
Ortho P	0.42	0.34	0.56	0.50
TOC	0.77	0.61	0.80	0.72
Silica	-0.47	-0.48	-0.38	-0.39
Na	-0.44	-0.46	-0.51	-0.51
Ca	-0.58	-0.71	-0.86	-0.90
Mg	-0.69	-0.76	-0.85	-0.86
K	0.56	0.51	0.56	0.57
Total Solids	0.56	0.61	0.52	0.56
Dissolved Solids	0.23	0.12	0.00	-0.08
Suspended Solids	0.53	0.49	0.62	0.67
Turbidity	0.18	0.20	0.30	0.44
Alkalinity	-0.59	-0.71	-0.85	-0.86
pH	-0.68	-0.70	-0.79	-0.80
Spec. Cond.	-0.49	-0.61	-0.73	-0.81
Temperature	-0.17	-0.10	-0.23	-0.12
Dissolved Oxygen	0.21	0.22	0.33	0.35
Total Bact	0.16	0.24	0.03	0.16
Fecal Coli	0.15	0.27	0.23	0.44
Fecal Strp	0.08	0.26	0.17	0.45
Total Coli	0.22	0.31	0.22	0.36

* C = Concentration, Q = streamflow

Table 5

"Best" Form of the Bivariate Regression Equations Including the Intercept A, Slope B, Correlation Coefficient R, Standard Error of Estimate Se and Number of Samples N

Water Quality Parameters	Form of Variable Conc.	Form of Variable Flow	A	B	R	Se	N
NO_3-N	Log	Log	-1.0388	0.2636	0.65	0.2004	477
NH_4-N	Log	Log	-1.3553	0.0680	0.13	0.3200	297
TKN	Log	Log	-0.3989	0.1394	0.36	0.2320	423
Total P	Log	Log	-1.8046	0.2362	0.50	0.2635	478
Diss P	Log	Log	-1.8588	0.1133	0.24	0.2931	478
Ortho P	Rect	Log	4.9357	12.2493	0.56	11.6013	354
TOC	Rect	Log	-2.0511	6.4448	0.80	3.1604	329
Silica	Log	Rect	0.8421	-0.0006	-0.48	0.1168	355
Na	Rect	Log	1.9177	-0.2627	-0.51	0.2824	479
Ca	Log	Log	1.3650	-0.3536	-0.90	0.1108	479
Mg	Log	Log	0.4108	-0.1861	-0.86	0.0704	479
K	Log	Log	-0.1305	0.1583	0.57	0.1400	351
Total Solids	Log	Rect	1.8381	0.0012	0.61	0.1754	355
Diss Solids	Rect	Rect	49.0799	0.0341	0.23	15.9650	355
Susp Solids	Log	Log	0.7535	0.5839	0.67	0.4184	474
Turbidity	Log	Log	0.7400	0.3078	0.44	0.4053	479
Alkalinity	Log	Log	1.8192	-0.2893	-0.86	0.1054	349
pH	Log	Log	0.8823	-0.0324	-0.80	0.0150	353
Spec. Cond.	Log	Log	2.0974	-0.2539	-0.81	0.1137	319
Temperature	Rect	Log	18.3502	-2.1919	-0.23	5.0009	353
Diss Oxygen	Log	Log	0.8844	0.0349	0.35	0.0598	341
Total Bact	Log	Rect	3.0494	0.0015	0.24	0.6753	432
Fecal Coli	Log	Log	0.1469	0.5619	0.44	0.7666	431
Fecal Strp	Log	Log	0.2478	0.6465	0.45	0.8019	354
Total Coli	Log	Log	2.3126	0.3544	0.36	0.5839	355

primary constituent. The origin of K is not simply geochemical leaching. It is suggested by Babcock and MacDonald (1973) that K may originate from direct input from rain or agricultural and forest management practices.

Tables 6, 7, 8, and 9 present the functional relations of water quality parameters versus flow obtained for the upstage and downstage of the hydrograph. The functional relations established in Table 4 for the various parameters tend to prevail; however, the sensitivity of the slopes B and the correlation coefficients R for the upstage relations are much better than for the downstage. The combined data for the upstage and downstage hydrograph of Table 5 lie somewhere in between. Note that for 8 parameters, i.e., pH, Na, Ca, Mg, DO, alkalinity, specific conductivity, and total bacteria, better correlations are on the downstage of the hydrograph; nonetheless, for the remaining 16 parameters they adhered to the "first flush" phenomenon of the upstage of the hydrograph. The standard error of estimate Se is essentially the same for the upstage, downstage, and combined stage functional expressions for each parameter.

Table 10 shows the multiple regression relations obtained for 11 of the water quality parameters with the dependent variables in the most desirable form. As streamflow, alkalinity, turbidity and stream temperature can be measured onsite for a storm event, the predictions of chemical constituent loadings are easily determined. A comparison of these relations with those of Table 5 shows general improvement with strength of relationship. The correlation coefficient for NO_3-N increased from R = 0.65 for the bivariate regression to R = 0.86, suggesting a much stronger relation with multiple regression. Temperature improves the multiple regression analysis because temperature varies seasonally and the data set was collected throughout the year (see Table 3).

Table 11 shows the accumulative loadings in metric tons for 14 parameters at the sampling location for each of 13 storms. The wide variation in loading is related to antecedent moisture conditions, rainfall amount, rainfall duration, and time of year (Table 1). As an example, suspended solids are related to rainfall amount and duration as demonstrated by the 31 Jan 75 and 3 Mar 77 storms which were both about 10 cm of rainfall covering a 2-day period (suspended solids ≈ 500 MT); whereas, the 16 Jun 77 storm was approximately 11 cm but lasted 6 days (suspended solids ≈ 2000 MT).

Table 6

Correlation Coefficients R for the Various Forms of the Independent and Dependent Variables for the Upstage of the Hydrograph

Water Quality Parameter	Simple Correlations			
	Rect C/Rect Q*	Log C/Rect Q	Rect C/Log Q	Log C/Log Q
NO_3-N	0.54	0.70	0.48	0.64
NH_4-N	0.05	0.17	0.13	0.28
TKN	0.47	0.58	0.46	0.61
TOC	0.78	0.86	0.61	0.74
Total Phos.	0.42	0.55	0.45	0.60
Ortho Phos.	0.52	0.65	0.43	0.53
Diss. Phos.	0.20	0.28	0.20	0.27
Silica	-0.45	-0.43	-0.45	-0.42
Na	-0.40	-0.43	-0.41	-0.45
K	0.63	0.70	0.54	0.68
Ca	-0.66	-0.82	-0.78	-0.87
Ma	-0.76	-0.80	-0.80	-0.81
Temperature	-0.18	-0.27	-0.09	-0.15
pH	-0.60	-0.71	-0.63	-0.73
Spec. Cond.	-0.51	-0.70	-0.63	-0.78
Alkalinity	-0.60	-0.79	-0.70	-0.84
Turbidity	0.19	0.44	0.29	0.63
Total Solids	0.61	0.65	0.68	0.73
Diss. Solids	0.43	0.26	0.32	0.18
Susp. Solids	0.55	0.66	0.55	0.73
Diss. Oxygen	0.17	0.20	0.19	0.22
Total Bact.	0.15	0.08	0.26	0.29
Fecal Coli	0.22	0.33	0.37	0.65
Fecal Strp.	0.10	0.25	0.30	0.57
Total Coli	-0.28	0.34	0.38	0.50

* C = Concentration, Q = streamflow

Table 7

Correlation Coefficients R for the Various Forms of the Independent and Dependent Variables for the Downstage of the Hydrograph

Water Quality Parameter	Rect C/Rect Q*	Log C/Rect Q	Rect C/Log Q	Log C/Log Q
NO_3-N	0.35	0.48	0.34	0.54
NH_4-N	-0.07	-0.00	-0.11	-0.04
TKN	0.18	-0.01	0.25	0.08
Total Phos.	0.30	0.36	0.26	0.38
Diss Phos.	0.08	0.15	-0.12	0.10
Ortho Phos.	0.27	0.44	0.22	0.42
TOC	0.77	0.80	0.63	0.74
Silica	-0.51	-0.46	-0.53	-0.48
Na	-0.49	-0.60	-0.52	-0.58
Ca	-0.60	-0.90	-0.75	-0.93
Mg	-0.71	-0.90	-0.79	-0.91
K	0.55	0.58	0.53	0.56
Diss Solids	-0.00	-0.11	-0.06	-0.15
Susp. Solids	0.52	0.74	0.43	0.65
Total Solids	0.59	0.61	0.57	0.58
Turbidity	0.18	0.28	0.06	0.19
Alkalinity	-0.64	-0.88	-0.78	-0.88
pH	-0.79	-0.84	-0.81	-0.84
Spec Cond.	-0.51	-0.72	-0.62	-0.81
Temperature	-0.15	-0.05	-0.09	0.02
Diss. Oxygen	0.25	0.41	0.25	0.41
Total Bact	0.31	0.15	0.24	0.14
Fecal Coli	0.10	0.16	0.14	0.22
Fecal Strp	0.07	0.22	0.22	0.33
Total Col.	0.15	0.18	0.23	0.29

* C = Concentration, Q = streamflow

Table 8

"Best" Form of the Bivariate Regression Equations Including the Intercept A, Slope B, Correlation Coefficient R, Standard Error of Estimate Se and Number of Samples N for the Upstage of the Hydrograph

Water Quality Parameters	Form of Variable Conc.	Form of Variable Flow	A	B	R	Se	N
NO_3-N	Rect	Log	0.0668	0.1163	0.70	0.08493	216
NH_4-N	Log	Log	-1.4346	0.1237	0.28	0.3070	125
TKN	Log	Log	-0.4930	0.2530	0.61	0.2277	176
TOC	Rect	Log	-1.5871	7.1960	0.86	2.9633	168
Total Phos.	Log	Log	-1.8513	0.3181	0.60	0.3023	216
Ortho Phos.	Rect	Log	3.8708	14.7450	0.65	11.5095	180
Diss. Phos.	Rect	Log	15.5475	6.4991	0.28	15.6013	216
Silica	Log	Rect	0.8223	-0.00055	-0.45	0.1259	180
Na	Log	Log	0.2729	-0.0677	-0.45	0.0948	216
K	Rect	Log	0.5951	0.7483	0.70	0.4759	176
Ca	Log	Log	1.3379	-0.2972	-0.87	0.1208	216
Mg	Log	Lg	0.3836	-0.1533	-0.81	0.0794	216
Temperature	Rect	Log	19.6425	-2.6484	-0.27	6.0448	178
pH	Log	Log	0.8775	-0.0252	-0.73	0.0154	178
Spec Cond.	Log	Log	2.0899	-0.2248	-0.78	0.1157	163
Alkalinity	Log	Log	1.7792	-0.2180	-0.84	0.0858	176
Turbidity	Log	Log	0.6181	0.4871	0.63	0.4232	216
Total Solids	Log	Log	1.6986	0.2620	0.73	0.1637	180
Diss Solids	Rect	Rect	54.1916	0.0637	0.43	15.321	180
Susp. Solids	Log	Log	0.8376	0.5993	0.73	0.3999	215
Diss Oxygen	Log	Log	0.8881	0.02215	0.22	0.0623	178
Total Bact	Log	Log	2.9465	0.3082	0.29	0.7278	216
Fecal Coli	Log	Log	-0.00263	0.8177	0.65	0.6736	210
Fecal Strep	Log	Log	0.1639	0.8818	0.57	0.8354	180
Total Coli	Log	Log	2.3160	0.4982	0.50	0.5743	180

Table 9

"Best" Form of the Bivariate Regression Equations Including the Intercept A, Slope B, Correlation Coefficient R, Standard Error of Estimate Se and Number of Samples N for the Downstage of the Hydrograph

Water Quality Parameters	Form of Variable Conc.	Form of Variable Flow	A	B	R	Se	N
NO_3-N	Log	Log	-0.8815	0.1918	0.54	0.1586	261
NH_4-N	Log	Rect	-1.2087	-0.00054	-0.11	0.3223	173
TKN	Log	Rect	-0.2550	0.00058	0.25	0.2090	247
Total Phos.	Log	Log	-1.7339	0.1642	0.38	0.2117	262
Diss Phos.	Rect	Log	18.5976	3.9386	0.15	13.6081	262
Ortho Phos.	Rect	Log	6.3814	10.0676	0.44	11.4099	174
TOC	Rect	Log	-3.5792	6.6474	0.80	2.8170	161
Silica	Log	Rect	0.8638	-0.00063	-0.53	0.1036	175
Na	Rect	Log	1.9986	-0.3152	-0.60	0.2237	263
Ca	Log	Log	1.4011	-0.3977	-0.93	0.0822	263
Mg	Log	Log	0.4601	-0.2244	-0.91	0.0539	263
K	Rect	Log	0.6079	0.3964	0.58	0.3071	175
Diss Solids	Log	Log	1.6729	-0.0351	-0.15	0.1283	175
Susp. Solids	Rect	Log	38.5132	65.0774	0.74	30.2845	259
Turbidity	Rect	Log	4.2874	9.7628	0.28	17.9030	263
Alkalinity	Log	Log	1.8687	-0.3478	-0.88	0.1028	173
pH	Log	Log	0.8901	-0.400	-0.84	0.0137	175
Spec. Cond.	Log	Log	2.1023	-0.2742	-0.81	0.1064	156
Temperature	Rect	Rect	15.0409	-0.0077	-0.15	5.2997	175
Diss. Oxygen	Rect	Log	7.5998	0.9018	0.41	1.1256	163
Total Bact	Rect	Rect	1594.48	11.2052	0.31	3394.8	215
Fecal Coli	Log	Log	0.3821	0.3089	0.22	0.7960	221
Fecal Strp	Log	Log	0.3424	0.4598	0.33	0.7169	174
Total Col.	Log	Log	2.2339	0.3023	0.29	0.5508	175

Table 10

Multiple Regression Equations for Onsite Measured Parameters Including the Multiple Correlation Coefficients R, Standard Error of Estimate Se, and the Number of Samples N for the Total Hydrograph

Equation	R	Se	N
Ca = 0.2469 (Alk) - 2.2442 (log Q) + 0.0072 (Turbidity) + 3.5436	0.94	1.7464	479
Mg = -0.3711 (log Q) + 0.7333 (log Alk) + 0.92703	0.88	0.2106	479
Na = -0.3713 (log Q) - 0.3548 (log Alk) - 0.2619 (log temp) + 2.8946	0.56	0.2740	479
NO_3 = -0.00839 (Alk) + 0.0411 (log Turbidity) - 0.0320 (log Q) + 0.2128 (log temp) + 0.2759	0.86	0.608	477
TP = 0.6387 (Turbidity) + 45.0753 (log Q) + 1.4209 (Alk) + 22.1252 (log Temp) - 106.9453	0.66	33.1195	478
PO_4 = 9.7308 (log Turbidity) + 10.9756 (log Q) + 0.8865 (Temp) - 20.7632	0.76	9.2561	354
TOC = 6.7884 (log Q) - 0.09313 (log Alk) + 0.1472 (Temp) - 5.6935	0.82	3.0528	329
Si = -0.00738 (Q) - 0.3032 (log Alk) - 0.01233 (Turbidity) + 7.95103	0.52	1.4513	355
FC = 125.1559 (Turbidity) + 305.1986 (Temp) - 176.4974 (Alk) - 934.419 (log Q) + 3286.58	0.58	7509.91	431
Fs = 487.5176 (Turbidity) + 269.082 (Temp) + 1513.1924 (log Q) - 11076.6225	0.52	26867.3	354
TC = 43.6231 (Turbidity) + 847.3761 (log Q) + 36.8695 (Alk) + 38.7894 (Temp) - 2748.0529	0.65	1684.15	355

Table 11

Accumulative Loadings (Metric Tons) for 14 Parameters from the Measurements Obtained for the 33 Storm Events

Date of Storm	Na MT	K MT	Ca MT	Mg MT	Si MT	TOC MT	NO_3 MT	NH_4 MT	TP MT	PO_4 MT	TC 10^{12} Cells	FC 10^{12} Cells	FS 10^{12} Cells	S.S. MT
31 Jan 75	51.80		153.00	37.80			10.30	3.40	2.60			5.54		4794.00
19 Nov 75	1.20	1.40	18.20	2.10	8.10	1.70	0.20	0.04	0.06	0.02	1.60	0.03	0.05	31.00
5 Feb 76	3.30	1.60	20.8	3.00	11.30	2.50	0.18	0.12	0.41	0.01	0.30	0.01	0.10	38.10
3 Mar 76	2.10	0.90	12.20	2.10	4.60		0.21	0.16	0.02	0.01	0.65	0.01	0.01	33.90
28 Mar 76	6.20	3.50	41.90	7.00	29.10	9.30	0.85		0.09	0.06	1.70	0.05	0.02	17.30
19 Arp 76	4.80	2.50	33.50	5.70	21.60	4.60	0.50	0.08	0.09	0.04	1.39	0.09	0.05	26.20
12 May 76	5.90	4.00	31.40	7.20	29.70	16.20	1.19	0.04	0.12	0.07	9.42	0.17	0.29	169.20
15 Jun 76	7.50	6.60	57.40	7.60	38.20	16.80	1.40		0.17	0.14	4.87	0.77	0.68	233.10
31 Aug 76	2.60	1.60	19.60	2.30	10.40	5.30	0.11	0.02	0.03	0.02	1.02	0.01	0.02	7.90
24 Oct 76	25.50	27.00	115.60	25.80	121.60	231.60	6.01	0.74	0.60	0.55	23.75	0.89	1.33	1121.00
2 Feb 77	5.50	3.10	24.90	4.50	18.40	11.90	0.44	0.03	0.03	0.02	0.83	0.01	0.01	118.20
3 Mar 77	70.50	76.80	206.20	49.80	304.60	601.50	17.87	0.22	1.74	1.05	77.03	0.60	0.90	5355.00
16 Jun 77	20.70	26.00	78.70	16.50	59.40	184.50	7.40	0.21	2.71	0.82	43.20	4.01	5.16	2105.00

The type of rainfall/runoff variation associated with two of the storm events, i.e., 19 Nov 75 and 28 Mar 76, are shown in Figures 5 and 6. The 19 Nov 75 storm isohyets were uniformly distributed throughout the watershed yielding simple hydrographs at the five stream gaging sites from the headwaters (Black Springs gage) to the Highway 84 bridge. The 28 Mar 76 storm shows complex hydrographs due to the stochastic nature of the rainfall which occurred over a 4-day period.

The relation of stream discharge, nitrate, and total phosphorus to time is shown in Figure 7 for the 16 Jun 77 storm event. The NO_3-N loaded slowly and peaked after the hydrograph peak and did not return or follow the recession of the stream discharge. The total phosphorus peaked prior to the hydrograph, but tended to follow the downstage of the hydrograph. These shifts in peaks increase the variance when attempting to develop functional relations. The high tail of the NO_3-N reduced downstage correlatable relations with discharge.

Another aspect of the source of variation with the water quality parameters is shown in Figure 8 for Na loading for the 31 Jan 75, 15 Jun 76, 24 Oct 76, and 16 Jun 77 storms in relation to stream discharge. In the first three storms Na loads differently on the upstage than on the downstage of the hydrograph (hysteresis). Also during the 31 Jan 75 storm, Na loaded less on the upstage than on the downstage; however, during the 15 Jun 76 and the 24 Oct 76 storm the process changed directions and Na loaded more on the upstage and less on the downstage. During the 16 Jun 77 storm Na loaded essentially the same for the upstage and the downstage. As noted in Tables 5 and 10, Na did not correlate very highly ($R = -0.51$ and $R = 0.56$, respectively). The source of the variability of Na loading is undoubtedly associated with rainfall/runoff patterns, antecedent moisture conditions, and ground-water elevation, etc.

CONCLUSIONS

The statistical analysis of the 26 water quality loading parameters for the 13 storm events using 479 grab samples showed that functional relations were obtainable with acceptable standard errors of estimate. Discriminant analysis demonstrated that the loading characteristics observed at the Glenwood and Highway 84 bridge were not significantly different; thus, permitting

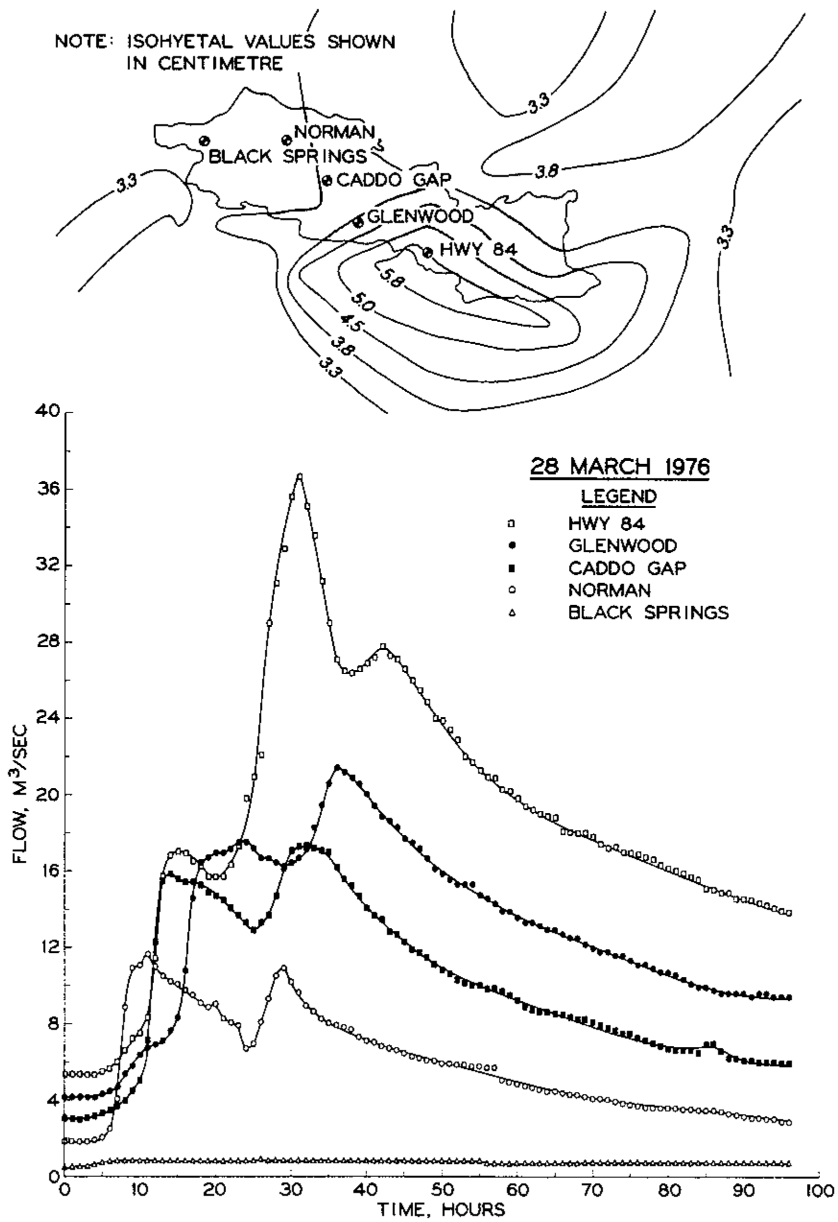

Figure 5. Rainfall isohyets and stream gage hydrographs at five stream gage stations for the 28 Mar 1976 storm

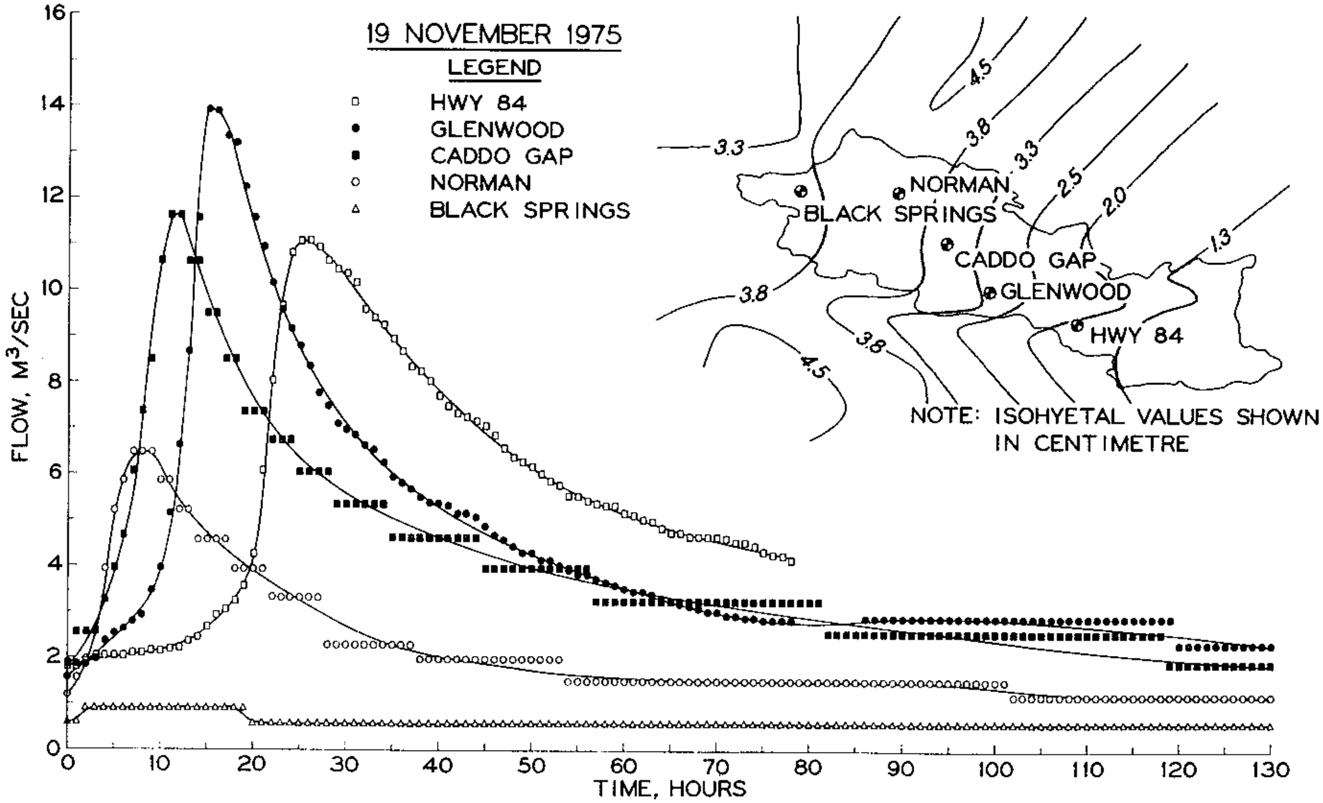

Figure 6. Rainfall isohyets and stream gage hydrographs at five stream gage stations for the 19 Nov 1975 storm

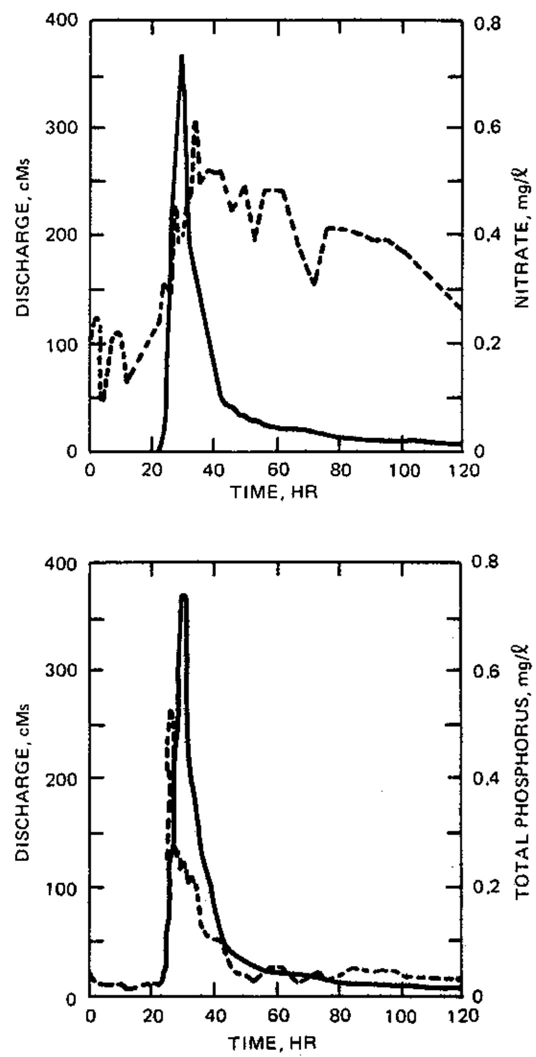

Figure 7. Relationships of stream discharge (solid line) and nitrate and total phosphorus (dashed lines) loading to time for the 16 Jun 77 storm

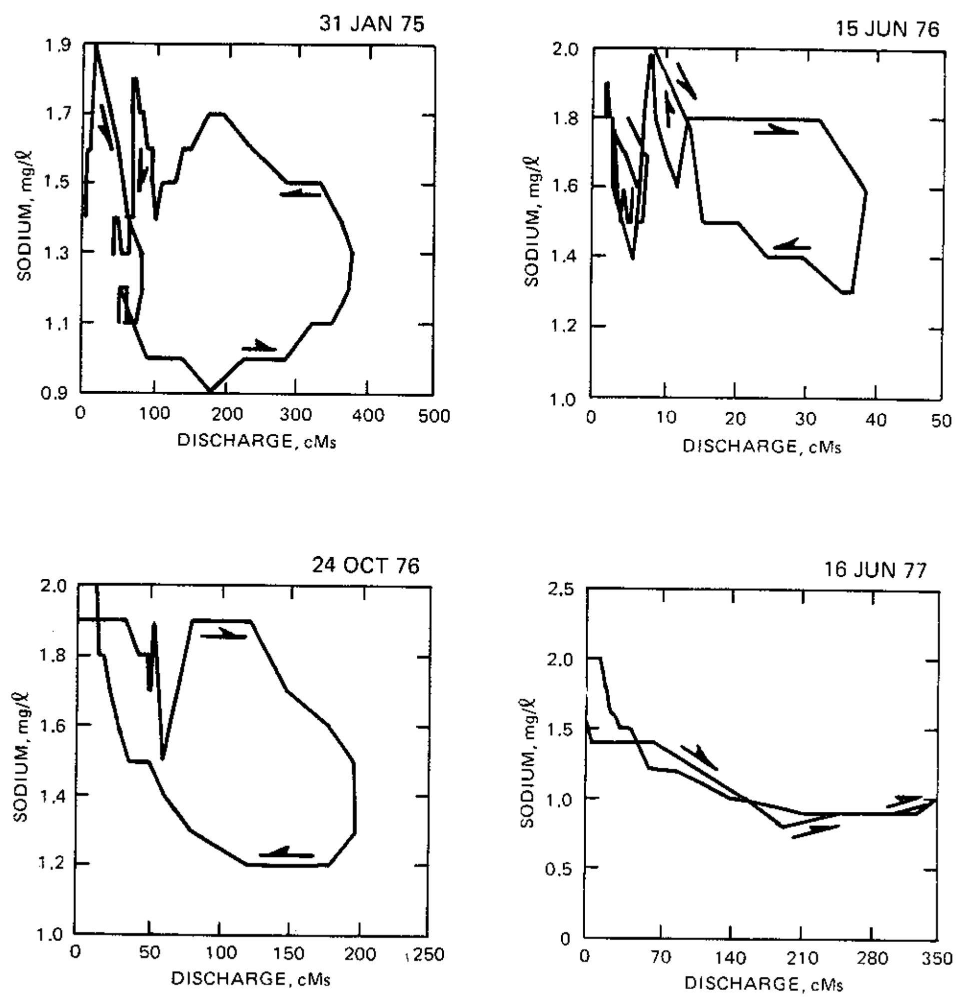

Figure 8. Relationship of sodium loading to stream discharge for the 31 Jan 75, 15 Jun 76, 24 Oct 76, and 16 Jun 77 storms

bivariate and multiple regression analysis on the combined data set. As streamflow, alkalinity, turbidity, and stream temperature can be measured onsite, the prediction of certain chemical loadings can be accurately calculated using multiple regression.

Factor analysis suggested the removal of some parameters from the multiple regression analysis because they measure similar loading responses and estimate similar amounts of variance. Alkalinity and turbidity were chosen for further analysis. Factor analysis indicated that alkalinity was a complex variable and could be associated with more than one theoretical dimension.

All 25 water quality parameters were analyzed using bivariate regression equations of water quality parameters versus flow. The form of the functional relation for use in the "best" bivariate equation was also determined. Although the log-log relation was the most common relation selected, occasionally a rectangular or semi-log expression could be accepted.

The upstage and downstage bivariate relations showed the "first flush" phenomenon as an important indicator for stream loading. In general, the metals (except potassium) loaded on the recession part of the hydrograph suggesting the influence of ground-water loading. The nutrients, fecal coliform, and fecal streptococci were better related to the upstage of the hydrograph where the effects of management of urban development, farms, and forests could be noted early during the storm event.

A great many complex relations added to the measured variance of a specific water quality parameter but the major contributing factors were the stochastic properties of rainfall, nonuniform loading relations (sodium for example), and loading peaks not matching hydrograph peaks.

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significantly individually and as a team toward developing the data base and analyses by which the Caddo River watershed's influence on DeGray Reservoir became better understood.

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DESIGN, CONSTRUCTION, AND OPERATION OF DeGRAY LAKE

by

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HISTORY AND DESCRIPTION OF DeGRAY PROJECT

Project authorization. The DeGray multiple-purpose flood control and power reservoir was included in Senate Document Numbered 117, 81st Congress, first session, as a feature of the comprehensive plan of improvement for the Ouachita River and Tributaries, Arkansas and Louisiana. The projects included in this document were authorized by the River and Harbor Act of 1950 which reads in part as follows:

> Sec. 101. That the following works of improvement of rivers and harbors and other waterways for navigation, flood control, and other purposes are hereby adopted and authorized to be prosecuted under the direction of the Secretary of the Army and Supervision of the Chief of Engineers, in accordance with the plans and subject to the conditions recommended by the Chief of Engineers in the respective reports hereinafter designated: Provided, That the provisions of Section 1 of the River and Harbor Act approved March 2, 1945 (Public, Numbered 14, Seventy-ninth Congress, first session), shall govern with respect to projects authorized in this title; and the procedures therein set forth with respect to plans, proposals, or reports for works of improvement for navigation or flood control and for irrigation and purposes incidental thereto, shall apply as if herein set forth in full:
>
> Ouachita River and tributaries, Arkansas and Louisiana; Senate Document Numbered 117, Eighty-first Congress; and there is hereby authorized to be appropriated the sum of $21,300,000 for the initial and partial accomplishment of the project;....

1/ Hydrologic Technician, Water Control Management Section, Hydraulics Branch, Engineering Division, US Army Engineer District, Vicksburg, Vicksburg, Miss.

2/ Supervisory Hydraulic Engineer, Chief, Water Control Management Section, Hydraulics Branch, Engineering Division, US Army Engineer, District, Vicksburg, Vicksburg, Miss.

The inclusion of municipal and industrial water supply as one of the project purposes was authorized by the "Water Supply Act of 1958," approved 3 July 1958, as amended by the "Federal Water Pollution Control Act Amendment of 1961," approved 20 July 1961.

The approving documents resulted in the "DeGray Project," consisting of a multipurpose reservoir with project purposes of flood control, hydroelectric power generation, recreation, water supply and water quality control.

Description of drainage area. The Caddo River has its source in the mountainous region of Montgomery County, Arkansas, and flows in a southeasterly direction about 84 miles to its junction with the Ouachita River 5.4 river miles above Arkadelphia, Arkansas, at river mile 426.0 above the mouth of Black River.

The watershed of the Caddo River is chiefly mountainous, with elevations ranging from a maximum of 2,200 feet NGVD near the source to about 170 feet NGVD at the mouth. In the upper reaches of the Caddo River above river mile 56, the valley averages about 2,000 feet in width, while the valley downstream from river mile 56 averages a little over 1,000 feet in width, with several reaches about 400 to 700 feet wide. The average stream slope above Norman, Arkansas, is about 40 feet per mile; from Norman to Glenwood, Arkansas, about 10 feet per mile; from Glenwood to the dam site, about 7 feet per mile; and below the damsite, about 4 feet per mile.

Project location. The DeGray Project is located about 7.9 miles above the mouth of the Caddo River and is approximately 5.4 miles north of Arkadelphia, Arkansas. The DeGray Dam controls a drainage area of 453 square miles, which is about 92 percent of the area of the Caddo River watershed.

The regulating dam is located at mile 4.8 on the Caddo River or about 3.1 miles downstream from DeGray Dam and controls about 27 square miles of drainage area between the regulating dam and DeGray Dam.

Pertinent features of the project. The DeGray Project consists of a main dam of earthfill construction with a spillway, intake structure, earth dikes along the reservoir rim, hydroelectric power generating facilities consisting of one conventional unit and one reversible pump turbine unit, a conduit for power and flood control release, penstocks, powerhouse, switchyard, appurtenant structures, and a downstream reregulating dam.

DESIGN OF DeGRAY PROJECT

Dam. The dam consists of a rolled earthfill embankment with a total length of 3,040 feet, of which 1,640 feet are in the main or valley section and 1,400 feet are in a dike section extending over the low right abutment. The crest of the dam is at elevation 453.0 feet NGVD, which provides a maximum dam height of 243 feet above the river channel. This dam height provides 24.5 feet of surcharge pool above the spillway crest of 423 feet NGVD plus a freeboard of 5.5 feet above the surcharge pool. The embankment was constructed of impervious sandy clays and clay gravels from terrace deposits in the vicinity. While these materials are essentially unclassified, excavation from borrow areas and placement in the dam section were controlled as to place the more pervious materials near the downstream slope of the dam. The entire foundation of the dam was stripped to the rock line. A core trench was excavated to firm rock along the centerline of the dam for its entire length. At the fault zone near the left abutment, special treatment was required. A single-line grout was provided under the core trench with additional grouting at the fault zone. Upstream and downstream slopes of the dam were protected by riprap. Pressure relief was provided by rockfill toe drains, filter blankets, and vertical sand drains.

Reservoir storages. Reservoir storage behind DeGray Dam is as follows:

a. Minimum pool. The minimum pool is that portion of the reservoir below elevation 367.0 feet NGVD (the lower limit of power and water supply) and has an area of 6,400 acres and a dead storage capacity of 261,500 acre-feet, equivalent to 10.8 inches of runoff from the drainage area.

b. Power and water supply pool. The power and water supply pool is that portion of the reservoir between elevations 367.0 feet NGVD and 408.0 feet NGVD. The top of the power pool at elevation 408.0 feet NGVD has an area of 13,400 acres and a storage capacity of 393,200 acre-feet, equivalent to 16.3 inches of runoff from the drainage area. Top of the power and water supply pool, elevation 408.0 feet NGVD, was determined in connection with investigations of the type and height of dam. Storage requirements for water supply are fixed, and benefits that would accrue to the project from them are constant. Therefore, power benefits are a controlling factor in determining the capacity in the power and water supply pool. Power benefits for various power pools were made on a comparable basis to determine economic height of dam. The studies were based on an annual load factor of 11 percent. Power storages were based on 21.7-percent drawdown for maximum power pools of 395 and 408 feet, NGVD and amount to 295,000 acre-feet and 393,200 acre-feet, respectively. Storage for the maximum power pool of 424 feet NGVD was limited to 450,000 acre-feet with a drawdown of 17 percent due to difficulty in filling the larger capacities. Based on these factors, a maximum power pool of 408 feet NGVD was determined to be most economical. The power and water supply storage of 393,200 acre-feet of storage can provide a dependable water supply release of 250 million gallons per day (387 cubic feet per second, cfs). The installed hydropower capacity of 68,000 kilowatts includes one conventional unit of 40,000 kilowatts and one reversible turbine with a 28,000-kilowatt motor generator. Hydropower capabilities were based on a 31-month critical low-flow period beginning May 1924 and ending November 1926. Based on the critical low-flow period during the period of record (1923 through 1961), the project can provide a capacity of 62,000 kilowatts at minimum power pool when it is operated at an 11-percent load factor at the same time providing a dependable water supply release of 250 million gallons per day. During the first 25 years of the project life, when the full water supply release will not be required, generation can be increased to provide a load factor of 20 percent during the high power demand months of June, July, and August for 90 percent of the years, and 15 percent during the remaining years. The pumped storage hydroelectric plant at DeGray Dam can produce a prime continuous energy of 6,900 kilowatts. The average annual energy is estimated as 91.1 million kwh and the energy required for pumping as 19.3 million kwh per year.

c. Flood control pool. The flood control pool between elevations 408.0 and 423.0 feet NGVD contains 227,200 acre-feet of storage, which is sufficient to control the dual-peak flood of April 1927, the greatest volume of record in the upper Ouachita Basin. Reservoir inflows for this flood were obtained by drainage area relationship from observed flows of the Ouachita River at Hot Springs, Arkansas. The volume of runoff is equivalent to 11.7 inches from the drainage area. The power and water supply pool was assumed full at the beginning of the flood, and the average daily power release was limited to 2,000 cfs when the flow of the Ouachita River at Arkadelphia exceeded 20,000 cfs. This retention will allow downstream stages to recede, after which reservoir releases will be limited to a maximum of 6,000 cfs for power generation and flood control.

Spillway. The spillway is located in a natural saddle in the rim of the reservoir approximately 4,000 feet east of the damsite. The spillway is a uncontrolled, unlined, broad-crested type with a crest elevation of 423.0 feet NGVD. It consists of an approach channel, a flat control section, and an outlet channel. The control section is 200 feet long and approximately 250 feet wide. It will be located in sandstone sufficiently resistant to scour to afford a satisfactory control. The inlet and outlet channels slope away from the control section at 0.5 percent to provide drainage. The excavated channel section has 2 on 1 side slopes up to the top of rock and 1 on 1.5 side slopes above the top of rock. A 20-foot berm at the top of rock increases stability of the side slopes.

Spillway design storm. In accordance with 1st Indorsement, dated 3 September 1947, to a letter from the Lower Mississippi Valley Division to the Chief of Engineers, dated 18 August 1947, subject: Spillway Design Storms for Red River Basin, design storm values were extrapolated from data in Hydrometeorological Report No. 23, "Generalized Estimates of Maximum Possible Precipitation over the United States East of the 105th Meridian," and were reduced by 5 percent to compensate for differences between basin shape and critical isohyetal patterns. The spillway design flood was routed through the reservoir, with the reservoir level at the beginning of the flood assumed to be at the top of the flood control pool, elevation 423.0 feet NGVD. Routing the spillway design flood inflows through the reservoir resulted in a maximum surcharge pool elevation of 447.5 feet NGVD and a peak outflow of 72,000 cfs which includes 5,000 cfs power release. The maximum pool obtained from this routing is considered to be a safe design for the surcharge pool of DeGray Reservoir. The surcharge design pool at elevation 447.5 feet NGVD has an area of 23,800 acres and a storage capacity of 495,100 acre-feet, equivalent to 20.5 inches of runoff from the drainage area. Routing the spillway design flood with the power and flood control conduits inoperative resulted in a surcharge pool elevation of 448.0 feet NGVD, an increase of only 0.5 foot above the surcharge pool design elevation 447.5 feet NGVD. Pool elevations for various spillway discharges were determined from backwater computation in which Manning's formula was used with "n" values of 0.075 for the small tributary, 0.035 for the outlet and approach channel, and 0.015 for the concrete section. The spillway capacity at a surcharge pool elevation of 447.5 feet NGVD is 67,000 cfs. Velocities in the excavated channel range from about 10 to 21 feet per second for the spillway design discharge of 67,000 cfs.

Freeboard requirements. The freeboard requirements for DeGray Dam and saddle dike were determined in accordance with the method outlined in memorandum, "Conference on Determination of Freeboard Requirements for the McGee Bend Dam, Angeline River, Texas," dated 1 August 1956. Computations of wave height and wave run-up were based upon a computed effective fetch at surcharge-pool elevation 447.5 feet NGVD of 1.1 miles for the main dam and 2.9 miles for the saddle dike. Distances of 2.5 miles for the main dam and 9.3 miles for the saddle dike along the line of maximum fetch were used in computing wind tide. Based on wind velocity duration data in the general vicinity of DeGray Reservoir from data presented in CW-178, Technical Bulletin No. 1, "Concepts for Surface Wind Analysis and Record Velocities," wind velocities of critical duration from 30 to 40 miles per hour (overland) might reasonably be assumed to occur coincident with the maximum reservoir level that would be attained during the spillway flood. A wind velocity of 40 miles per hour overland (52 miles per hour over water) would require 2.5 feet of freeboard for the main dam and 4.0 feet of freeboard for the saddle dike. For the main dam, a freeboard of 3.8 was considered adequate for a wind velocity of 60 miles per hour overland, and for the saddle dike, a freeboard of 5.5 feet would be adequate for a wind velocity of about 60 miles per hour overland. A freeboard of 5.5 feet was adopted, thus fixing the top of the earth dam and the saddle dike at elevation 453.0 feet, NGVD.

Intake structure. Both power and flood control discharges enter the tunnel through a single intake tower. The intake consists of four 21- by 21-foot openings controlled by movable baffle gates and trash racks which can be positioned at three different levels, as requested by the US Fish and Wildlife Service. Water intake centerlines for the three levels are 395.0, 380.0 and 355.5 feet NGVD. Facilities were provided for emergency closure of these gates as required. Regulation of flows for power and flood control are accomplished at the powerhouse and flood control stilling basin, respectively. In addition to the baffle gates and trash racks, raking equipment is provided at each intake opening. A gantry-type hoist, arranged to rotate to all gate slots, is mounted on the operating deck of the tower. This hoist can handle all gate operation, using a lifting beam arranged to pick up or release the gates. The hoist can also be used for removing the gates, trash racks, and raking equipment from the tower. An emergency low-level gated intake is located in the end of the tunnel immediately upstream from the intake tower. Water below elevation 345.0 feet NGVD can be released through this outlet in case of emergency.

Tunnel. A 29-foot-diameter tunnel located through the right abutment of the dam was used for river diversion during construction of the dam. The 29-foot tunnel was modified, during construction of the dam, to carry flows for power and flood control. The 29-foot tunnel is connected to the intake structure with a 90-degree bend at the base of the intake structure. A concrete plug with an emergency gate installed was used to seal off the upstream section of the diversion tunnel in front of the intake. The 29-foot tunnel branches into two penstocks, one 15 feet, 9 inches in diameter for the conventional turbine unit and one 13 feet, 2 inches in diameter for the reversible pump-turbine unit. A steel liner extends from the centerline of the dam to the outlet structure.

Outlet structure. Flood control storage will normally be released by power generation, which will permit the release of about 6,000 cfs at maximum powerpool. However, to serve in making water supply and flood control releases if necessary when the power plant is shut down or to supplement the turbine discharge in drawing the reservoir down in case of emergency, a 6- by 11.5- foot slide gate is located at the end of the power and flood control tunnel. The gate discharge capacity is 6,700 cfs at 423.0 feet NGVD (top of flood control pool), with a maximum design discharge of 6,000 cfs. The outlet gate discharges into the stilling basin used during river diversion through the 29-foot diversion tunnel. The stilling basin was designed for a flow of 37,000 cfs from the 29-foot diversion tunnel and is used for releases not exceeding about 6,000 cfs from the flood control outlet. For a distance of 140 feet from the flood control conduit exit, the floor of the basin has a parabolic shape. The length of the stilling basin floor is 125 feet at elevation 190.0 feet NGVD. The end sill elevation is 197.0 feet NGVD and the two rows of baffle piers are 7 feet high. The elevation of the top of training wall is 235.0 feet NGVD.

Powerhouse. The powerhouse is located on the right abutment adjacent to the flood control stilling basin. It is a windowless, enclosed structure of reinforced concrete. Each generating unit and the erection bay is contained in individual monoliths. Provisions were made, in a separate service bay, for all necessary auxiliary and control equipment associated with the main generating units. As this station is remotely operated from Blakely Mountain Dam, only a minimum of office space and no public facilities were provided in the powerhouse. Measures for the protection of the facilities were given consideration in the detailed design studies. Fallout protection was provided in the powerhouse in accordance with the provisions of Engineer Manual (EM) 1110-2-5000, dated 1 March 1961, and EM-345-461, dated 19 January 1961. Protection was initially to be provided for 30 persons, including families of operating personnel, as the DeGray Reservoir is considered an isolated project. However, the structure lends itself to providing extra shelter space at very little additional cost. Consideration was also given to constructing the powerhouse to provide a degree of blast resistance. A minimum of essential indicating and control instruments was provided in the protected area to permit emergency operation of the power facility. Provisions were also made to allow for later installation of chemical, biological, and particulate filters and their required ventilation machinery, and to permit quick and economic pressurization of the powerhouse.

Turbines. The supply contract under which the turbines were procured placed the responsibility for mechanical design and hydraulic efficiency on the manufacturer and was in accordance with Corps guide specifications. The data presented below are based on criteria and data available in TVE Technical Monograph No. 52 (Rev. Ed.), "Preliminary Selection of Hydraulic Turbines and Powerhouse Dimensions." Since the project must meet its commitments for water supply at all times, it was desirable to install two units. The project consists of one 28,000-kw reversible unit and one 40,000-kw conventional unit. Dependable capacity of the project is 62,000 kw, which was determined by an 11 percent annual load factor for 6,900 kw continuous power. Therefore, it was necessary to install units to develop this capacity at a minimum head of 144 feet. The design head of 171 feet is a compromise between average operating head and average critical period head and will give good operating efficiencies during normal operating pool levels. Generator selection places the critical head at 153 feet where the plant will develop 68,000 kw at full gate. The range of operating heads and the horsepower output required indicated the choice of vertical Francis turbines. The project requires a reversible unit of 28,000-kw capacity to meet pumping requirements from the regulating reservoir. The operating net head varies from a maximum of 201 feet to a minimum of 144 feet. The maximum net head is the reservoir level at top of flood control pool minus the tailwater elevation with the 28,000-kw unit operating, with allowances for the intake and penstock losses and residual velocity in the tailrace. The minimum net head is the reservoir level at bottom of power drawdown minus tailwater elevation for a discharge from both units, each operating at full gate output and with allowances for the conveyance losses. The net head at maximum power pool is 185 feet. The net head fluctuation for power between 185 and 144 feet represents a small range of heads of a Francis unit, the larger being 108 percent and the smaller 84 percent of the design (best efficiency) head of 171 feet. This fluctuation is within the range of good operating efficiencies.

Turbine capability. The capability of the conventional unit is 36,500 kw at minimum head with an equivalent of 50,500 horsepower, assuming a generator efficiency of 97 percent. The output of the turbine at design head of 171 feet and best gate is 55,500 horsepower and 65,500 horsepower at full gate. The turbine output at best efficiency is 85 percent of the maximum output. The capability of the reversible turbine is 25,500 kw at minimum head with an equivalent of 35,400 horsepower, assuming a generator efficiency of 97 percent. The output of the turbine at design head of 171 feet and best gate is 38,700 horsepower and 45,200 horsepower at full gate. The turbine output at best efficiency is 86 percent of the maximum output. Having determined the total capacity of the plant from prime power studies, the amount of reversible capacity was established upon the pumping capability of the unit and the amount of storage available for pumping. The maximum amount of water which can be pumped during a pumping cycle is that storage between the full pool elevation of 221 feet NGVD (1,600 acre-feet). When reduced by the water supply requirement of 460 acre-feet for 14 hours, the maximum pumping volume was about 1,140 acre-feet. This required a pumping rate of about 1,900 cfs for 7 hours and is the approximate capacity of the

28,000-kw reversible unit. Hourly load characteristics indicate that pumping can be done as long as 8 hours during off-peak demands. Since this is the maximum pumping time required, the 28,000-kw reversible unit furnishes sufficient pumping capacity. Pumping time will probably average 6 to 7 hours per pumping cycle. It was estimated that the reversible unit will pump at a rate of 3,200 cfs at a minimum head of 150 feet, and 1,750 cfs at the maximum pumping head of 189 feet. With the pump-back feature, the DeGray Project has an average annual energy capability of 97.8 million kilowatt-hours.

Generators. Two generators were installed. One has a normal rating of 44,444 kilovolt-amperes, 40,000 kilowatts at 0.90 power factor, 13.8-kilovolt, 3-phase, 60-cycle, 60-degree Centigrade temperature rise; and the other a normal rating of 31,111 kilovolt-amperes, 28,000 kilowatts at 0.90 power factor, 13.8-kilovolt, 3-phase, 60-cycle, 60-degree Centigrade temperature rise. The generators are the totally enclosed type arranged for a recirculating air-cooling system with water-cooled surface-air coolers. Each of the generators meets the requirements of Corps Guide Specification CE-2202, "Hydraulic Turbine Driven Alternating Current Generator."

Transformers and switchyard. One three-phase transformer was furnished for each generating unit. The transformers were the oil-immersed type and have a kva rating equal to 115 percent of the kva rating of the generator to which it is connected. The low-voltage transformer winding is rated 13.2 kv and the high-voltage winding has a dual rating of 115 kv for initial operation and 161 kv for future operation. Each transformer meets the requirements of OCE Guide Specification CE-2203, "Power Transformers." The switchyard structure was constructed of galvanized steel and was located on a 470- by 150-foot berm on the downstream dam slope at elevation 264 feet NGVD. Provisions were made for three 161-kv transmission lines terminating in the switchyard; two lines are being used initially and one is for future use. Main and transfer buses were installed on opposite sides of the main structure. Transmission lines were connected to the main bus through oil circuit breakers and to the transfer bus through motor-operated disconnecting switches. The generator power transformers were located on the concrete slab, which serves as the roof of the upstream service area of the powerhouse, and are connected to a common transformer bus through motor-operated disconnecting switches. The transformer bus is connected to the main and transfer buses through oil circuit breakers. Switchyard clearances, insulation levels, and design in general are to be for a nominal 161-kv operating voltage.

Supervisory-control system. The power plant is being operated by a remote-supervisory monitoring and control system installed for controlling the DeGray Power Project from the Blakely Mountain Power Project. The units are fully automatic and suitable for unattended operation. Supervisory control is accomplished by use of a supervisory system which incorporates continuous monitoring of control channels, individual point selection and check back, and priority for alarm or control actions, over telemetered functions. It provides positive control and supervision of generating units, switchyard, and station-service switchgear. It also includes emergency closing of the butterfly valves and emergency tripping of the generator-air-circuit breakers, audible and visual indication of alarms, and selection of telemetered quantities. Telemetering is of the frequency type with continuous recording of plant megawatts and three continuous channels which will handle selected quantities. Available for selection are generator amps, volts, watts, revolutions per minute, gate position, gate limit position, and bus volts. Kilowatt-hour readings, pool and tailwater elevations, and tailwater temperature are telemetered in digital form to an appropriate indicator. Supervisory actions, along with telemetered quantities, are multiplexed using audiotone and telegraphic equipment and transmitted by microwave. Microwave equipment consists of two terminals, each with a standby unit and a reflector located between the two terminals. Three channels are considered satisfactory to accommodate the supervisory-control telemetering and required radio-control links. A fourth channel was incorporated for a microwave-alarm and service channel.

Regulating dam. The regulating dam consists of an earthen embankment, concrete gravity spillway, and sluices. The crest of the dam is at elevation 235.0 feet NGVD, and the crest of the spillway (top of water supply and pumping pool) is at elevation 221.0 feet NGVD. The height of the regulating dam was limited by the topography and because higher pool elevation would cause reductions in power heads at DeGray Dam. The low ground elevation, about 1,700 feet from the left end of the regulating dam, is about 232.0 feet NGVD. No other sites are available downstream for construction to provide additional storage by relocation of the regulating dam. The minimum pool is that portion of the regulating reservoir below elevation 209.0 feet NGVD and has an area of 90 acres and a dead storage capacity of 600 acre-feet. The water supply and pumping pool is that portion of the reservoir between elevations 209.0 feet NGVD and 221.0 feet NGVD (spillway crest elevation). The top of the water supply and pumping pool at elevation 221.0 feet NGVD has an area of 430 acres and a storage capacity of 3,000 acre-feet. The joint-use pool for water supply and pumping with 1,600 acre-feet of regulating storage would normally fluctuate from a minimum pumping elevation of 217.0 feet NGVD to elevation 221.0 feet NGVD, and the average elevation for power generation is 219.0 feet NGVD. The basin with a storage of 1,400 acre-feet between elevation 209.0 feet NGVD and elevation 217.0 feet NGVD, is sufficient to provide the required water supply during the critical weekend period of nongeneration.

Design of the regulating dam. A synthetic 30-minute unit hydrograph for the 27 square miles of drainage area between the DeGray Dam and the regulating dam was used for computing inflow hydrographs from storm rainfall obtained from US Weather Bureau Technical Paper No. 40, "Rainfall Frequency Atlas of the United States," for 1-, 5-, 10-, 25-, 50-, and 100-year frequencies and for a standard project flood estimate. The computed inflows for the various frequency flood hydrographs were routed assuming the reservoir level at the beginning of the flood to be at spillway crest elevation 221.0 feet NGVD, sluices inoperative and, for one condition, with no power release from DeGray Dam and for another condition, with a power release of 6,000 cfs from DeGray Dam. The standard project flood inflow hydrograph was routed under the above assumptions with no power release from DeGray Dam. Rerouting the 50-year frequency flood with 6,000 cfs power release resulted in a pool elevation of 228.5 feet NGVD and a peak outflow of 25,000 cfs; the reservoir at pool elevation 228.5 feet NGVD will have an area of 700 acres and a storage capacity of 4,200 acre-feet. The standard project flood when routed through the reservoir resulted in a pool elevation of 232.0 feet NGVD and a peak outflow of 45,000 cfs; the reservoir at elevation 232.0 feet NGVD will have an area of 830 acres and a storage capacity of 6,800 acre-feet, and when the pool reaches this elevation, flow would bypass the dam at a point about 1,700 feet from the left abutment. Based on these routings, the top of the dam was established at elevation 235.0 feet NGVD, 3 feet above the standard project flood pool elevation of 232.0 feet NGVD and 6.5 feet above the 50-year frequency flood pool elevation, 228.5 feet NGVD. The freeboard provided is considered reasonable for this type of dam.

Reregulating dam structures. The spillway located in the dam is a concrete gravity ogee section with a length of 300 feet and a crest elevation of 221.0 feet NGVD. Four 5- by 9-foot sluices with a total capacity of 4,400 cfs at top of water supply and pumping pool elevation 221.0 feet NGVD (spillway crest) were provided for making water supply releases and to serve in discharging excess flow during peaking periods when power flows would cause the pool to exceed the spillway crest and affect tailwater elevations at DeGray Dam. The sluices discharge into the stilling basin below the spillway. The stilling basin has two rows of baffle piers designed to dissipate the energy from the spillway and sluice flows. The length of the stilling basin floor is 55 feet at elevation 192.0 feet NGVD. The end sill elevation is 195.0 feet NGVD, and the two rows of baffle piers are 5 feet high. The elevation of the top of training walls is 215.0 feet NGVD.

CONSTRUCTION OF DEGRAY PROJECT

Diversion tunnel and intake shaft. The contract for construction of the DeGray tunnel and appurtenances was awarded on 10 January 1964 to a joint venture composed of Winston Brothers Company and Green Construction Company, Minneapolis, Minnesota. Excavation of the upstream intake channel area was begun on 17 February 1964. Original plans were to begin excavation of the tunnel from the upstream end, tunnel approximately 100 feet downstream of the intake shaft, then move the tunneling operation to the downstream end and tunnel upstream until holed through.

Upstream excavation was completed to the stage to allow tunneling to begin on 10 August 1964. On 20 August, with the first three ring beams set and equipment in place to set the fourth, the tunnel caved in, destroying all material in place. After the slide occurred, it was decided to resume tunneling from the downstream end. Excavation to the point necessary for downstream tunneling to begin was completed on 20 October 1964 and the tunnel was driven upstream, reaching station 3+82 on 1 May 1965. On 12 May, excavation was resumed from the upstream end. Station 3+83 was reached on 6 June. This completed tunnel excavation except for invert cleanup. Concrete placement for the intake portal structure began in December 1965 and was completed on 22 March 1965. Placement of backfill concrete to fill overexcavated areas in the tunnel was completed on 13 July 1965.

Placement of tunnel concrete began on 11 September 1965 and was completed on 1 December 1965.

Excavation of the intake shaft was begun on 29 June 1964 and was completed on 19 May 1965. Concrete placement began on 17 November 1965 and was completed by 1 January 1966.

Low-level intake. Construction of the low-level intake began with installation of a safety fence around the work area above the tunnel inlet on 2 May 1969. Chipping of concrete to expose the reinforcing steel was completed on 29 May. Construction of the cofferdam was started on 9 June and river closure was made on 20 June 1969. Placement of concrete in the lower half of the orifice was made on 25 June.

Installation of the lower half of the trashguard was made on 26 June. Because of rain in the upper watershed, water behind the cofferdam rose faster than expected and it was decided to wait until the second cofferdam closure to complete the upper half of the orifice. The upper half of the orifice and trashguard were installed on 12 July. The bulkhead was set against the orifice on 8 August 1969. Drilling of the calyx hole for the gate operating stem was started on 30 June 1969 in the north water passage at the floor of the intake structure. The calyx hole was completed 23 September 1969. The emergency gate was set and tested on 2 February 1970 and placed in final position after lubrication on 19 February 1970.

Intake structure. The contract for construction of the DeGray intake structure and branch tunnels was awarded to Martin K. Eby Construction Co., Inc., of Wichita, Kansas, on 10 June 1966. In early September 1966, excavation of the structure foundation began. In November 1966, concrete work on the structure was started. In May 1967, the intake cylinder gate was set at elevation 345. In November 1967, the semi-gantry crane rail, handrails, hatch covers, and other miscellaneous metals were installed at elevation 453. Work on the guides for baffle gates, trash racks, intake cylinder gate, and spiral stairway was started in August 1967 and completed in late December 1967. Painting of the above items began in September 1967 and continued until April 1968. Installation of the cylinder gate hoist, drive unit, and electrical systems was started in February 1968 and completed in early March 1968. The cylinder gate seals and guides were installed, aligned, and concreted during the same period. Installation of the semi-gantry crane commenced on 22 May 1968 and was completed, including electrical systems, painting, and testing, on 18 July 1968. During the same period the bulkheads, trashracks, and baffle gates were assembled, touch-up painted, and moved into storage. The contract was completed on 18 July 1968 with completion of work on the semi-gantry crane and final cleanup of the intake area.

Dam and dike. The contract for construction of the dam and dike was awarded to Potashnick Construction, Inc., of Cape Girardeau, Missouri, on 4 June 1965. Work was begun on 28 June 1965, and final inspection and acceptance by the Government was made on 28 October 1969.

Construction of the dam was begun by installing a subcofferdam to divert river flow through the diversion tunnel. This was accomplished on 27 May 1966. After the river diversion, the dam foundation was cleaned and a permanent cofferdam, later to become part of the dam, was constructed. On 21 August 1966, with the cofferdam complete to elevation 280.0, heavy rainfall on the upstream watershed caused the pool level to rise rapidly, threatening to overtop the cofferdam. The danger was averted by calling in heavy equipment to raise the elevation of the cofferdam. Work on the dam continued as weather permitted, with no other major problems arising. Miscellaneous work on the dam was finished, and the dam was completed in October 1968.

Construction of the dike was begun by clearing borrow area in June 1965. Clearing of the dike area began on 10 July 1965 and work on the dike proceeded until November 1966, when dike embankment work was halted. During the 1967 season, the dam was given priority and no dike embankment work was done. Work on the dike resumed in June 1968 and proceeded to completion in October 1969 with no major problems encountered.

Powerhouse. The contract for construction of the powerhouse was awarded to Martin K. Eby Construction Co., Inc., of Wichita, Kansas, on 12 July 1968. The job was accepted by the Government on 17 December 1971, and minor items were completed on 8 May 1972.

Excavation for the powerhouse foundation began on 29 July 1968. With excavation 80 percent complete, concrete placement was begun in October 1968. The mass concrete was completed in January 1970 and the powerhouse floors were completed in June 1971. The roof was constructed in September and October 1970 using precast-prestressed Lin Tees.

Layout for the tailrace began on 29 July 1968. Concreting for overexcavation began 18 September 1968. The east wall was completed 4 September 1969. The west wall was completed 6 April 1971. Concrete for the tailrace slab was finished on 17 March 1971, and the trash rack and boat barrier were installed shortly thereafter to complete the tailrace.

Eby Construction Co., Inc., subcontracted the piping and equipment installation work to Martyn Bros. of Dallas, Texas, and the electrical work to C-L Electric Company of Pocatello, Idaho. Installation of the various powerplant systems and equipment was initiated in early 1969.

Installation of the generator sole plates, foundation bolts, and piping up to the units was the responsibility of Eby Construction Co., Inc. The installation of the generators was by Allis-Chalmers Mfg. Co. Assembly of the rotor for Unit #1 was begun on 12 January 1971. In July 1971, just prior to the scheduled mechanical run on Unit #1, readings of the lower seal ring clearances for Unit #1 indicated a range of seal clearance from 0.034 to 0.057 inch, whereas design criteria called for a minimum clearance from 0.050 to 0.060 inch. At a meeting of representatives from OCE, Southwest Division, Omaha District, Vicksburg District, and the DeGray Resident Office, it was decided that the minimum allowable clearance would be 0.050 inch as per design and that sawing of material from the runner without dismantling the unit would be acceptable. Sawing operations were completed on 22 September 1971 and a final check indicated acceptable clearance ranging from 0.051 to 0.071 inch.

The turbine mechanical run for Unit #2 was completed on 19 August 1971. The turbine mechanical run for Unit #1 was completed on 9 October 1971. The mechanical run for the Unit #2 generator was completed on 26 August 1971. Following completion of the Unit #1 mechanical and dry-out run on 9 October 1971, both units were ready for operation. Unit #1 was placed on-line 29 November 1971 and Unit #2 in December 1971.

Reregulating dam. The contract for construction of the DeGray reregulating dam was awarded on 5 August 1969 to Guy H. James Construction Company of Oklahoma City, Oklahoma. Initial preparatory work such as clearing and grubbing and access road construction began on 9 September 1969. Stripping and excavation operations in the diversion channel, first stage cofferdam, and structure areas were started on 22 September 1969. Construction of the first stage cofferdam started on 13 October 1969.

The river diverted through the channel on 30 October 1969, and the first stage cofferdam was completed on 3 December 1969. After completion of the first stage cofferdam, work was concentrated on structure overburden and rock excavation. By mid-May 1970, overburden excavation was 80 percent complete and rock excavation was 75 percent complete.

Concrete operations were started on 18 May 1970. The weir concrete was completed 5 November 1970, with the right retaining wall finished on 20 November 1970 and the stilling basin on 4 November 1970. The remaining miscellaneous sidewalks, building slab, grouted ditches, and launching ramp were completed in the period May-August 1971.

The filter sand, filter gravel, riprap, and embankment operations were subcontracted to McCullough Construction Company of Ashdown, Arkansas. The subcontractor began stripping borrow area on 20 August 1970 and began compacting fill on 25 August 1970. Initial embankment work was on the left bank, east of the diversion channel between stations 10+00 and 14+50. In mid-October 1970, embankment operations were moved to the west bank between station 0+50 and 2+00. Due to weather conditions, the subcontractor worked intermittently during the winter months. On 1 April 1971, degrading of the first stage cofferdams began, with suitable material placed in the dam embankments. On 8 April 1971 the temporary plug was placed in the diversion channel and the river was diverted through the reregulating dam gates. The upstream second stage cofferdam, which formed a portion of the main dam embankment, was completed to grade on 13 April 1971. Embankment operations for the compacted fill on the dam were completed on 31 August 1971.

Riprap operations on the dam slopes and other specified areas were completed in early September 1971. Work proceeded during the 1971 construction season on the operations building, topsoil and sodding, and road work. The base course and surfacing were completed in early November 1971. Final acceptance of the job was made on 12 November 1971.

OPERATION OF DeGRAY PROJECT

Flood control operation. The flood control pool is between elevation 400.0 feet NGVD (top of power pool) and elevation 423.0 feet NGVD (spillway crest). When DeGray's pool elevation exceeds 408.0 feet NGVD, the excess storage will normally be released by power generation as soon as possible. However, there are power restrictions that are strictly adhered to by the project manager. These power restrictions are that any time the river stage of the Ouachita River at Arkadelphia exceeds 17 feet (approximately 20,000 cfs), power generation will be restricted to 816,000 kwh daily, one-half plant capacity, or approximately 2,000 cfs. Generation restriction periods will run from 8 a.m. to 8 a.m. while Arkadelphia is above 17 feet. If generation exceeds the restriction amount at the time the restriction is put in effect, no generation will be made during the remaining 24-hour period ending 8 a.m. No restriction on peaking is required. After flows have receded at Arkadelphia, the release from DeGray will be regulated so as not to exceed a flow of 20,000 cfs at Arkadelphia or a maximum of 6,000 cfs from the dam. When it is necessary to supplement flows through the flood control gate, which will be in case of an emergency or when the power plant is shut down, the project manager will follow automatic flood control restrictions. These automatic flood control restrictions are in effect any time flood control storage is being released through the flood control outlet and they are as follows:

a. Close the flood control gate when the river stage of the Ouachita River at Arkadelphia exceeds 17 feet.

b. Close the flood control gate when 1.00 inch of rain or more occurs at the dam in 24 hours or less.

c. It is the responsibility of the superintendent at the dam to be sure the stage at Arkadelphia is below 17 feet when releasing storage through the flood control gate.

The operations above are automatic. Instructions for following them are furnished to the project manager and the power plant operator at Blakely Mountain Dam who regulates DeGray by a remote-supervisory control system. The Vicksburg District's Water Control Management Section is responsible for the flood control operations at DeGray Dam and will issue special instructions to the project manager when conditions warrant. The method of operation as outlined above gives the optimum control of floods while maintaining sufficient releases to satisfy pumping and water supply requirements. The flood control ability of DeGray was evident during the December 1982 storm which produced 9.71 inches of runoff over the watershed in 48 hours. DeGray's flood control pool stored 223,410 acre-feet of water, reducing the flood crest at Arkadelphia by 2 to 3 feet and preventing $180,000 dollars in damages in the Arkadelphia reach and $190,000 dollars in damages in the Camden reach.

Power operation. The hydropower facilities at DeGray are operated remotely from Blakely Mountain Power Plant. The Southwestern Power Administration is the marketing agent for DeGray's hydropower production. In 1971 the Southwestern Power Administration signed a 20-year contract with Middle South Utilities for the exchange of power from DeGray Project. The contract for sale of power requires generation to meet water supply demands. The joint use of power and water supply pool is that portion of DeGray's pool between 367.0 feet NGVD and 408.0 feet NGVD. The power and water supply pool contains 393,200 acre-feet of storage for water supply and hydropower production.

Power generation and pumpback are scheduled by the customer according to load requirements and are subject to any necessary flood control restrictions. Unless flood control storage is being released, power production is not scheduled by the client utilities. DeGray's power plant remains on-line, but the units remain in reserve standby or in the condensing mode, during off-peak demands. When the power is needed by the customer during peak demand periods, DeGray's power is put on-line in a few seconds, at the request of the client utility, and may only remain on-line for a few minutes or a few hours. This ability to be brought on-line in a few seconds for peaking purposes is what makes the DeGray Project so valuable to the power companies.

Pumpback feature operation. Incorporation of pumpback facilities was possible without extensive modification of the reregulating dam and pool located downstream of the main dam to maintain storage for water quality control and industrial and municipal water supply releases. The pumpback facilities were designed to allow for full utilization of the project's hydropower generation capacity to assist in meeting peak demand load. Client utilities have the capability to constructively use a portion of their excess base power in periods of off-peak demand. Since the project has been in operation, water availability and fuel prices have not required or made justifiable the use of the pumpback facility to offset water discharges used for generating at peak demand periods. Other than in 1971-72 when the pumpback facilities were initially tested and run for a total of about 112 hours, the pumpback feature has been operated as listed below during off-peak periods:

Date	Hours
October 1974	7.6
November 1974	27.13
December 1974	90.77
January 1975	78.5
February 1975	2.0
September 1982	24.37

The reason for this pumpback was the excess energy in the system that would have been lost if not used. Because of economic factors, particularly the cost of fuels used in generating electricity, no other operations have been required or have been deemed economically practical, except for routine checks and sequencing of equipment. The lake levels and downstream water requirements are a factor in the scheduling and amount of use of the pumpback facilities. The Corps serves only as the operator of the pumpback facilities, and the electrical utilities are able to use them when they consider it feasible, subject to previously mentioned factors.

Water supply operation. The Ouachita River Valley Association and Ouachita River Water District (ORWD) conducted parallel investigations with the Department of Health, Education, and Welfare and determined the potential water needs in the tributary area of 250 million gallons per day. The initial release of 152 cfs for pollution control would ultimately be increased by the water district to 387 cfs (250 million gallons per day) for municipal and industrial use. Since December 1971, except when dewatering occurred for maintenance purposes, the minimum average daily release has been maintained from DeGray's reregulating dam. The ORWD was established as the legal local sponsoring agency for water supply from DeGray Lake. In February 1962 ORWD provided the Government an Act of Assurances providing for repayment of water supply as set forth in the Water Supply Act of 1958. Local interests are responsible for that portion of the total project cost assignable to furnishing municipal and/or industrial water supply storage, including interest during construction. This cost is to be repaid within the life of the project, but in no event to exceed 50 years after the project is first used for storage of water for water supply purposes. The Act also provides that no interest be charged on this cost until the supply is first used, but that in no event shall the interest-free period exceed 10 years. In 1980, as the end of the 10-year grace period approached, the Vicksburg District contacted ORWD to request information which would be necessary to develop contractual arrangements to facilitate local interest contributions. ORWD was requested to particularly address (1) development of a contract program with representatives of potential water-using areas; (2) engineering investigations of pipeline distribution facilities, routes, and costs; and (3) plans for funding of appropriate local contributions. The president of ORWD responded in a letter dated 22 July 1980 stating ORWD had been dormant for almost 20 years and that, of the three ORWD directors, one had died and another had moved out of the area.

The Vicksburg District is maintaining communication with ORWD and will continue to encourage them to take action which will enable ORWD to meet their local interest responsibilities as soon as practicable. The

Vicksburg District Real Estate Division has the lead element responsibility for the development and execution of the water supply contract with ORWD. When the ultimate water supply of 387 cfs is needed, releases from the regulating dam will provide an adequate supply.

Temperature control operation. The intake consists of four 21- by 21-foot openings controlled by movable baffle gates and trashracks which can be positioned at three different levels. Water intake centerlines for the three levels are 395.0, 380.0 and 355.5 feet NGVD. The 1,000-square-foot opening below the water surface allows sufficient opening to furnish maximum power flows without excessive velocities through the gate passages. Since the completion of DeGray Project, the intake gates were set at centerline elevation 395.0 feet NGVD until 14 March 1979 when the gates were moved to centerline elevation 355.5 feet NGVD. The intake elevation was changed to coldwater withdrawal (355.5 feet NGVD) to honor a joint request from Arkansas Department of Pollution Control and Ecology, the US Fish and Wildlife Service, the Arkansas Game and Fish Commission, and the Waterways Experiment Station so they could study the effect of hypolimnion releases on the downstream fishery. The study period ran for a 4-year period from March 1979 to March 1983, when the centerline of intake was changed back to elevation 395.0 feet NGVD. The intake will remain at elevation 395.0 feet NGVD until an optimum level can be selected by the parties involved.

Reregulating dam operation. As discussed earlier, the initial minimum water requirement supplied from the reregulating reservoir is 152 cfs and the ultimate requirement is 387 cfs. Up to 400 cfs can be released from the pool during the pumpback cycle without lowering the pool below elevation 217.0 feet NGVD. Releases of 400 cfs over the weekend no-generation period lowers the pool to elevation 209 feet NGVD. The five 5- by 9-foot sluice gates can be operated locally but are normally run by remote control from the Blakely Mountain Dam powerhouse to regulate outflows and minimize fluctuations in the reregulating dam pool and in tailwater elevations. The gates can be positioned at 0.1-foot intervals from zero to full open. Gate settings are determined from reregulating dam pool elevations and the expected power generation and pumpback amounts. Minimum releases during the pumping cycle maintain sufficient storage capacity for pumpback. Immediately following pumpbacks, based on anticipated generation, the gates are adjusted for desired outflows during the generation cycle and required stages in the reregulating pool.

Recreation facilities. The DeGray Project was the first in the Vicksburg District with recreation as a project purpose. The recreation facilities at DeGray have continued to be constructed and updated after the completion of the main dam and pertinent features of the project. The Corps of Engineers operates 17 day-use areas that provide diverse camping and outdoor recreational opportunities. In addition to Corps facilities, resort and commercial developments were made possible through outgrants to state and county agencies and private interests.

The success of DeGray's recreational facilities is evident, with the project averaging over 2 million visitors per year for the past 5 years.

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MATERIAL LOADING TO DeGRAY LAKE, ARKANSAS: PATTERNS AMONG VARIABLES AND SIGNIFICANCE OF STORM EVENT LOADING

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INTRODUCTION

The physical, chemical, biological, and social characteristics of reservoir ecosystems are directly controlled by the input of material from external sources. Nutrients, such as phosphorus, nitrogen, carbon, and silica, control algal and macrophyte growth, which in turn affect the higher levels in the food chain (Wetzel, 1976). The link between the input of nutrients and reservoir productivity has resulted in efforts to relate nutrient loads to the level of lake eutrophication (e.g. Vollenweider, 1968; Rast and Lee, 1978).

The rate of allochthonous sediment input directly controls the rate of depletion of reservoir storage capacity (USDA, 1973). The chemical composition of input particulate matter affects lake sediment characteristics and water clarity which govern the benthic community, internal loading during periods of anoxia, and reservoir water quality (Thornton et al., 1981; Kennedy et al., 1983). Material loading may also cause longitudinal gradients in water quality and sediment characteristics (Kennedy et al., 1982; Gunkel et al., 1984). Since material loads have such a significant impact on reservoir systems, accurate quantification of material loads is essential.

Material loading to a lake can be estimated by four different types of estimators (Wu and Ahlert, 1978): 1) zero-order, 2) direct, 3) statistical, and 4) descriptive. The zero-order method utilizes loading estimates from either the same, or a similar, watershed obtained from a previous study or uses export coefficients as a function of land use (e.g. Reckhow et al., 1980). The zero-order methods are the most simplistic and least expensive. However, care must be taken when transferring data from different areas by evaluating the accuracy and reliability of previous sampling design programs.

The direct method uses routine water quality data and discrete or instantaneous flow data to calculate loads (e.g. Walker, 1981; Verhoff et al., 1980). Direct methods are the most commonly used since routine water quality data is usually all that is available (i.e. no special studies on high flow or storm events).

Statistical methods use correlation and regression analyses to develop stochastic models for predicting concentration and/or load from a set of independent variables (e.g. Omernik, 1977; Jewell et al., 1980). These methods, while often providing good loading estimations, are extremely limited in their applicability to other watersheds, and require sufficient amounts of data (i.e. over the complete range of flow) and computational facilities.

Descriptive methods employ deterministic mathematical models of physical, chemical and hydrological processes (McElroy et al., 1976; Mills et al., 1982; Hydrocomp, 1980). The most common descriptive method employs the Universal Soil Loss Equation (USLE). While the USLE is widely used, and thus facilitates comparisons with other watersheds, it suffers from the same deficiencies as the zero-order method in that the model coefficients usually have to be obtained from the literature. Complex deterministic models are also used; however, these are usually expensive, have high data requirements, and usually have high level of total prediction uncertainty.

While numerous comparisons of material load estimation methods have been done, no one method has been shown to be acceptable for all application variables (Ongley et al., 1977; Smith and Stewart, 1977, Johnson, 1979; Jewell et al., 1980; Westerdahl et al. 1981; Dolan et al., 1981; Whitfield, 1982). Since significant differences exist between baseflow and storm events, the estimation of total annual load by any method is most appropriately done by estimating baseflow and storm load separately (Colston 1974; Stevens and Smith, 1978; Cahill, 1977), and in some cases, by further separating by season or month (Johnson, 1979).

The selection of an estimation method should involve examining the project objective, resources available, uncertainty acceptable, and possible inherent biases (Montgomery and Kennedy, 1986). The use of routine/surveillance data can create bias in annual loading estimations by over-estimating in dilution variables and under-estimation in flow-driven variables (Ongley et al., 1977, Johnson, 1979). This bias results because data from fixed interval sampling frequently contains only a few samples at high flows due to the infrequency of temporal occurrence of storm events. Thus, low (dilution) or high (flow-driven) concentrations that occur in storm events are not well represented in the distribution of water quality variables and cause a bias in the loading estimate. The ideal temporal sampling scheme should be continuous and flow-proportional, but since this is usually not feasible, a discrete sampling program should sample each flow regime with a frequency proportional to the sum of discharges during that flow regime (Stevens and Smith, 1978). Johnson (1979) showed the effect of changing sample size in estimating loads using samples proportional to discharge and duration.

The purpose of this paper is to present a summary of the results obtained in estimating material loads to DeGray Lake, Arkansas. The objectives of the study were to calculate material load estimates for DeGray Lake via the Caddo River, examine patterns in water quality variables during baseflow and storm events, and examine significance of storm event loading.

The Caddo River arises in the Ouachita Mountains of south-central Arkansas and flows southeast for 126 km to its confluence with the Ouachita River (Figure 1). The river is impounded 12.7 km above this confluence to form DeGray Lake, a US Army Corps of Engineers reservoir providing flood control, hydropower, and recreation. The total drainage area of the river is 1269 km^2. In general, the watershed can be divided into two geographic regions: the Interior Highlands, which is part of the Novaculite Uplift (area northwest of Glenwood), and the Athens Piedmont Plateau (below Glenwood) (Perrier, 1977). Land uses in the Caddo River watershed include forest (68 percent), agriculture (30 percent), and urban (2 percent).

The climate of the watershed is generally mild with an average monthly temperature of 17°C (range of 6°C in January to 28°C in July). The average annual precipitation is 134 cm/year at Arkadelphia and 140 cm/year at Glenwood. Monthly precipatation is distributed fairly uniformly throughout the year, although the summer storms tend to be short and intense with long periods between events. Precipatation occurs approximately 20 percent of the time and is rarely distributed evenly over space in the watershed. The amount of precipatation that enters the Caddo River as runoff ranges from 15-90 percent and averages 43 percent (Perrier, 1977). The upper basin (west of Glenwood) contributes a larger portion of steamflow in late fall and winter while the lower basin contributes more in the late spring and summer. During periods of low precipatation groundwater may contribute significantly to streamflow. Snowfall occurs from one to four times per year but will usually melt immediately; hence, there is no major snow melt/runoff event in the spring.

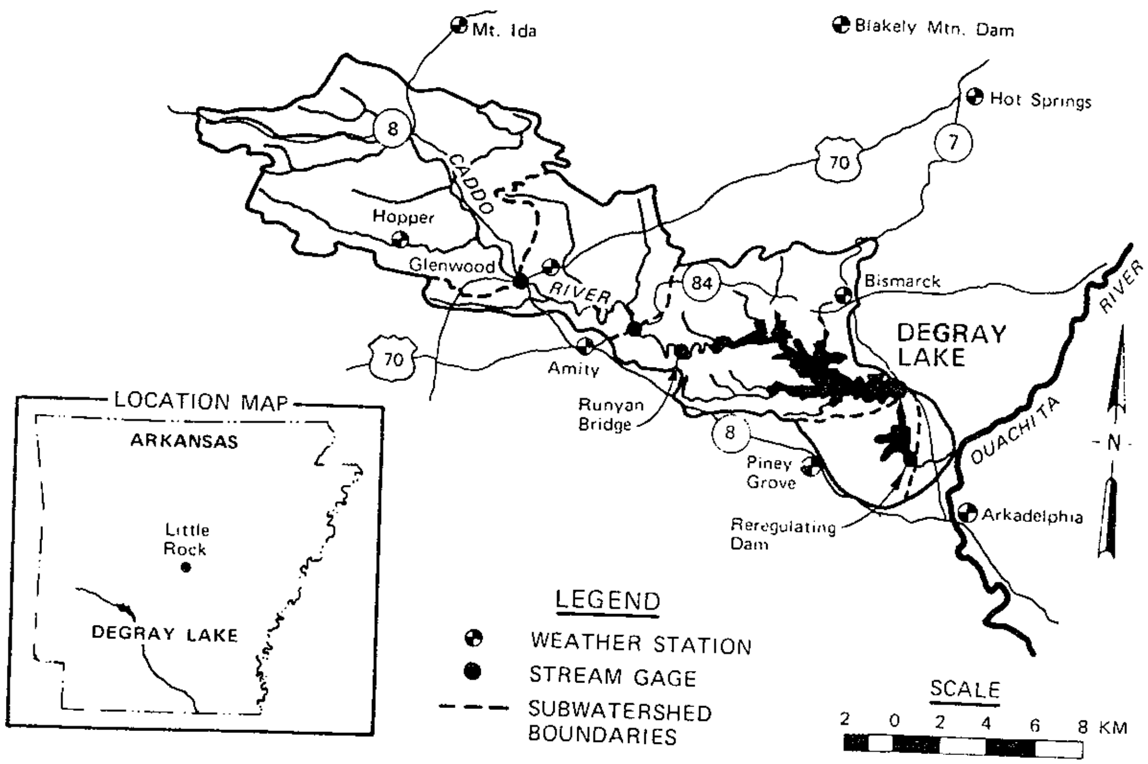


Figure 1. Watershed map of DeGray Lake, Arkansas.

METHODS

Water quality variables were sampled every two weeks from the Hwy 84 bridge on the Caddo River during the period 1 Jan 76 - 31 Dec 80. Samples were collected at midstream approximately 0.5 meter below the surface using a horizontal Van Dorn water sampler. Sample time was 10 a.m. for all samples. Presented in Table 1 are water quality variables and sample preparation, sample storage, and analytical methods used (see Glossary for variable definitions). Samples during storm events, which were defined as major rain events that generated a significant rise in the Caddo River, were collected approximately hourly until the rate of change in flow diminished. A minimum of 10 representative samples were maintained and analyzed. In general, at least four samples each on the rising and falling side of the hydrograph and two at the peak flow were analyzed. Sampling techniques and analytical methods were the same as for the routine samples. Sample collection and analysis were performed by the Water Chemistry Lab, Ouachita Baptist University, Arkansas.

A complete, bi-hourly flow record at the Highway 84 site for the years 1976 - 1980 was established by combining hourly stage measurements from the river gages located at Glenwood and Hwy 84 (Figure 2). For the years 1976, 1977, 1978 and 1980, missing values in the Hwy 84 flow record were calculated by linear interpolation between observed flow values. This was feasible since meteorologic records revealed no rainfall events during the periods of missing flow values. However, during early 1979 missing values occurred during periods of significant rainfall. For these values, a linear regression model of mean daily flows between Glenwood and Hwy 84 using data from the first half of 1979 was developed. This model accounted for travel time between stations and for conditions specific to 1979.

RESULTS

Summary statistics for water quality variables collected during the routine sampling program are presented in Table 2. Water quality relations under baseflow conditions and storm events were evaluated and modeled independently. Baseflow conditions were defined by comparing routinely collected water quality data with the continuous flow record. Samples not collected during or immediately after a storm event were subjectively

Table 1

Analytical Methods

Variable	Sample Preparation	Sample Preparation	Analytical Method	Reporting Units	References
Total Phosphorus	None	H_2SO_4 to pH <2; 4°C	Acid-persulfate digestion; ascorbic acid-molybdate colorimetric reaction	mg P/ℓ	Jeffries et al. 1979
Total Soluble Phosphorus	Filtration (0.45 μ)	H_2SO_4 to pH <2; 4°C	Acid-persulfate digestion; ascorbid acid-molybdate colorimetric reaction	mg P/ℓ	Jeffries et al. 1979
Soluble Reactive Phosphorus	Filtration (0.45 μ)	4°C; analysis within 24 hr	Ascorbic acid-molybdate colorimetric reaction	mg P/ℓ	Skougstad et al. 1979
Particulate Phosphorus		Total Phos. - Total Soluble		mg P/ℓ	
Soluble Unreactive Phosphorus		Total Soluble-Soluble Reactive		mg P/ℓ	
Nitrate Nitrogen	Filtration (0.45 μ)	$HgCl_2$ 40 mg Hg/ℓ)	Brucine-sulfanilic acid	mg N/ℓ	Skougstad et al. 1979
Nitrate/Nitrite Nitrogen	Filtration (0.45 μ)	4°C	Cadmium reduction	mg/ℓ	SMEWW, 1976
Ammonia Nitrogen	None	H_2SO_4 (2ml/ℓ); 4°C	Selective ion electrode	mg/ℓ	SMEWW, 1976
Total Kjeldahl Nitrogen	None	H_2SO_4 (2 ml/ℓ)	Kjeldahl digestion and selective-ion probe	mg N/ℓ	APHA, 1980
Total Solids		4°C	Total residue at 105°C	mg/ℓ	SMEWW, 1976

(Continued)

Table 1 (Concluded)

Variable	Sample Preparation	Sample Preparation	Analytical Method	Reporting Units	References
Dissolved Solids		4°C	Filterable (standard glass fiber filter) residue at 105°C	mg/ℓ	SMEWW, 1976
Suspended Solids		Total - Dissolved		mg/ℓ	
Total Silica					
Dissolved Silica					
Total Magnesium, Calcium, Sodium, Potassium, Manganese, Iron	None	HNO_3 (pH ≤2)	Atomic absorption	mg (X)/ℓ	APHA, 1980
Dissolved Magnesium, Calcium	Filtration (0.45 μ)	HNO_3 (pH ≤2)	Atomic absorption	mg (X)/ℓ	APHA, 1980
Total Organic Carbon	None	H_2SO_4 (pH ≤2); amber glass storage	Persulfate oxidation; infrared analysis	mg C/ℓ	EPA, 1974
Dissolved Organic Carbon	Filtration (pre-combusted glass fiber)	H_2SO_4 (pH ≤2); amber glass storage	Persulfate oxidation; infrared analysis	mg C/ℓ	

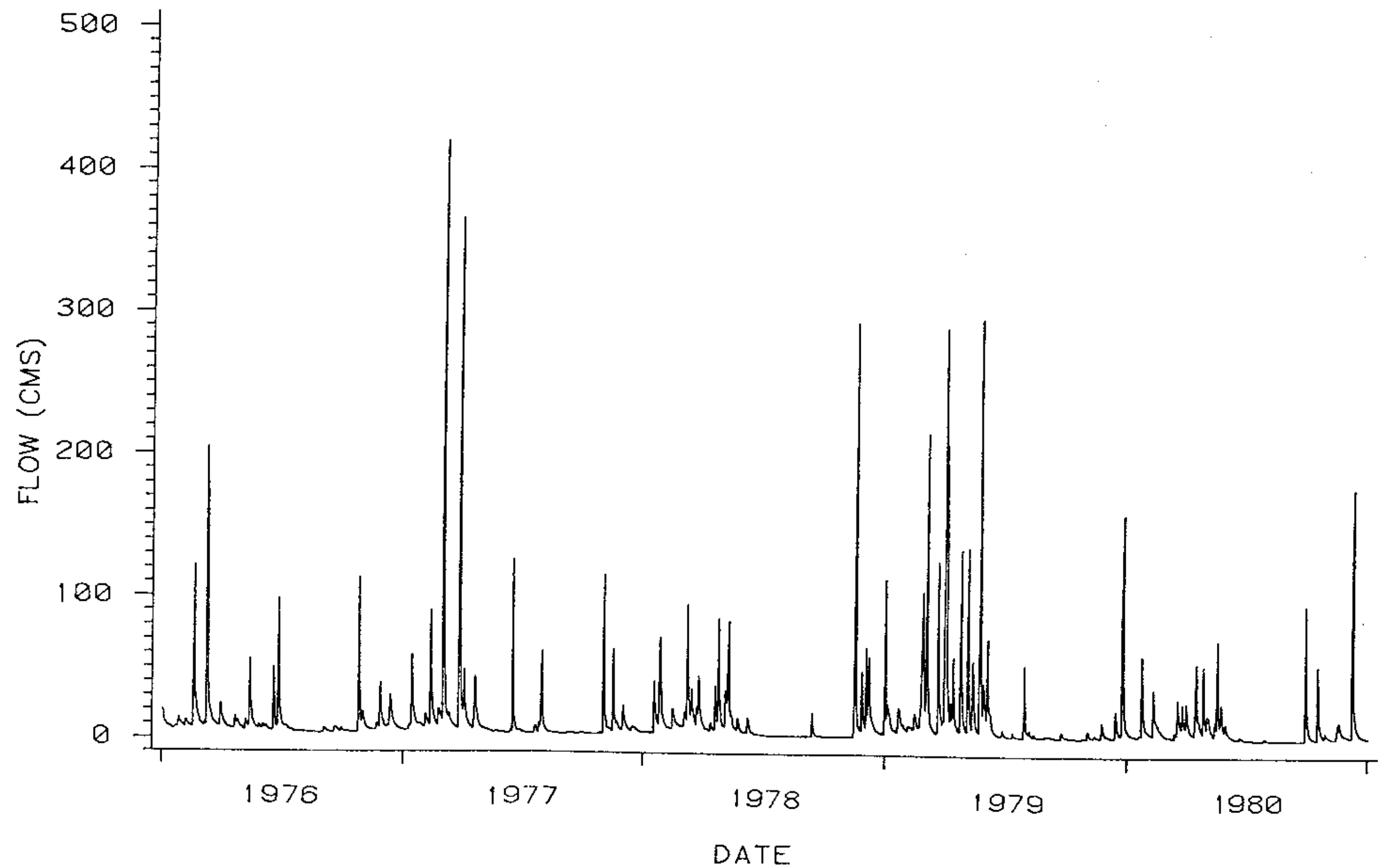


Figure 2. Mean-daily hydrograph for Caddo River at highway 84 bridge.

Table 2

Summary Statistics for Routine Water Quality Variables

Variable	Mean	Standard Deviation	Standard Error of Mean	Coefficient of Variation	Skewness	Kurtosis	Minimum	Maximum	Median	Mode
TP	0.0304	0.0355	0.00252	116.601	6.11	45.48	0.006	0.340	0.0230	0.019
PP	0.0139	0.0314	0.00240	225.219	7.06	57.83	0.000	0.313	0.0070	0.005
SUP	0.0072	0.0063	0.00051	87.482	1.59	3.08	0.000	0.034	0.0050	0.005
SRP	0.0136	0.0096	0.00070	70.655	1.91	6.89	0.000	0.070	0.0120	0.012
NO3N	0.1541	0.1676	0.01720	108.775	3.88	22.95	0.000	1.300	0.1200	0.070
NHYN	0.0319	0.0417	0.00309	130.769	2.12	5.42	0.000	0.220	0.0200	0.000
TKN	0.4312	0.3054	0.02264	70.822	2.02	5.39	0.000	1.970	0.3895	0.400
TS	71.4663	46.8513	3.51165	65.557	7.82	82.06	2.000	584.000	66.0000	69.000
DS	58.4689	19.1386	1.43854	32.733	0.19	0.01	11.000	113.000	58.0000	65.000
SS	17.5423	47.3722	3.97538	270.046	8.62	86.63	0.000	513.000	8.0000	6.000
DSI	6.7301	1.8639	0.13416	27.695	-0.38	-0.24	2.100	11.000	6.8000	8.700
TMG	1.8753	0.4095	0.04158	21.839	0.23	0.27	0.800	3.100	1.9000	1.600
TCA	11.4838	3.8623	0.38818	33.633	0.26	-0.55	3.000	20.000	11.0000	16.000
TNA	2.0915	0.3360	0.02382	16.066	-0.12	1.83	1.000	3.300	2.1000	2.000
TK	0.9297	0.3343	0.02412	35.954	2.05	7.46	0.200	2.700	0.9000	0.800
TMN	0.0489	0.0935	0.00996	191.262	2.06	3.91	0.000	0.400	0.0000	0.000
TFE	0.2953	0.3946	0.04280	133.622	4.01	20.56	0.000	2.800	0.2000	0.200
TOC	3.6305	1.9912	0.15993	54.845	2.31	8.23	0.600	15.100	3.0000	2.400
DOC	3.4000	1.8490	0.15251	54.383	1.65	3.21	0.500	10.700	2.8000	2.800

identified as occurring under baseflow conditions. Only water quality data specifically collected during storm events were used.

Long-term (i.e. annual) changes or trends in water quality may imply changing watershed or hydrologic conditions, which in turn can potentially affect the accuracy of nutrient load estimates. Therefore, time series plots of concentration for the period 1 Jan 76 - 31 Dec 80 were examined for each water quality variable. While such visual tests provide only qualitative results, they do provide a means for determining if the application of statistical tests is warranted. The time series plots exhibited no obvious trends in any of the water quality variables. Since these observations were consistent with the fact that there have been no significant land-use changes in the watershed, no statistical tests were conducted. Similar evaluations of possible seasonal water quality trends were performed by pooling data for all years, but were confounded by the existence of seasonal patterns in flow. However, since seasonal changes in water quality and flow were similar, both sources of variation were addressed by regressing concentration on flow.

Summary statistics for baseflow water quality variables which allowed identification of tendencies, variations, distributions, and possible outliers are presented in Table 3. Typical patterns between concentration and flow are shown in Figures 3 and 4. Mean concentrations in conjunction with flow are usually used to estimate nutrient loads. However, if non-normality is present in water quality variables more robust estimators are warranted. Based on the summary information, stem-leaf diagrams, Box plots, and normal probability plots, the following were chosen as the "best" measure of central tendency for water quality concentration:

Mean - TP, SRP, TKN, DSI, TMG, TCA, TNA, TK, TME, TFE, TS, DS

Median - SUP, PP, NO3N, NH4, TOC, DOC, SS

For SUP, PP, NO3N, NH4N, TOC, and DOC the mean tended to overestimate (skewed right distribution) while for SS the mean tended to underestimate (skewed left distribution), hence the use of the median as central tendency.

Summary statistics for the fractions (percentages of individual constituents) of phosphorus, carbon, and solids during baseflow conditions were also examined (Table 4). This information will be compared with fractions during storm events to examine possible changes in species composition resulting from increased flow. Associations between water quality variables during baseflow were analyzed by examining the correlation matrix and performing a factor

Table 3

Summary Statistics of Baseflow Water Quality Variables

Variable	Mean	Standard Deviation	Standard Error of Mean	Coefficient of Variation	Skewness	Kurtosis	Minimum	Maximum	Median	Mode
FLOW	5.01	2.43	0.220	48.41	1.19	1.69	2.09	15.26	7.34	3.42
TP	0.0221	0.0084	0.00078	38.15	0.40	-0.08	0.006	0.044	0.022	0.017
PP	0.0073	0.0057	0.00057	78.14	1.36	2.58	0.000	0.030	0.006	0.002
DUP	0.0065	0.0054	0.00056	83.58	1.17	1.14	0.000	0.023	0.005	0.005
DRP	0.0117	0.0069	0.00065	59.07	0.77	0.49	0.000	0.033	0.011	0.011
NO_3N	0.1082	0.0872	0.01176	80.64	1.31	2.19	0.000	0.410	0.00	0.070
NO_4N	0.0324	0.0432	0.00418	133.32	2.20	5.75	0.000	0.220	0.220	0.000
TKN	0.3967	0.3107	0.03018	78.32	2.41	7.80	0.000	1.970	0.300	0.400
TS	71.6402	54.2693	5.34731	75.75	8.39	79.57	29.000	584.000	65.000	60.000
DS	60.2308	18.7349	1.83710	31.10	-0.01	-0.06	11.000	104.000	59.500	50.000
SS	16.6154	57.9781	6.56473	348.94	8.36	72.32	0.000	513.000	8.000	6.000
DSI	6.5106	1.9105	0.17972	29.34	-0.35	-0.21	2.100	10.800	6.600	6.800
TMG	1.9849	0.3427	0.04708	17.26	0.93	1.67	1.500	3.100	2.000	1.600
TCH	12.6909	3.2906	0.44371	25.92	0.44	-9.85	7.400	19.400	12.100	8.800
TNH	2.1209	0.3378	0.3150	15.92	-0.26	2.88	1.00	3.300	2.100	2.200
TK	0.8575	0.2199	0.02069	25.64	0.62	4.48	0.200	1.900	0.800	0.800
TMN	0.0589	0.1058	0.01414	179.52	1.85	2.76	0.000	0.400	0.000	0.000
TFE	0.2000	0.1633	0.02202	81.65	2.27	9.60	0.000	1.000	0.200	0.200
TOC	3.2253	1.5577	0.16240	48.29	1.57	2.78	0.600	8.700	2.700	2.400
DOC	3.1092	1.5752	0.16888	50.66	1.49	2.60	0.500	8.600	2.700	2.200

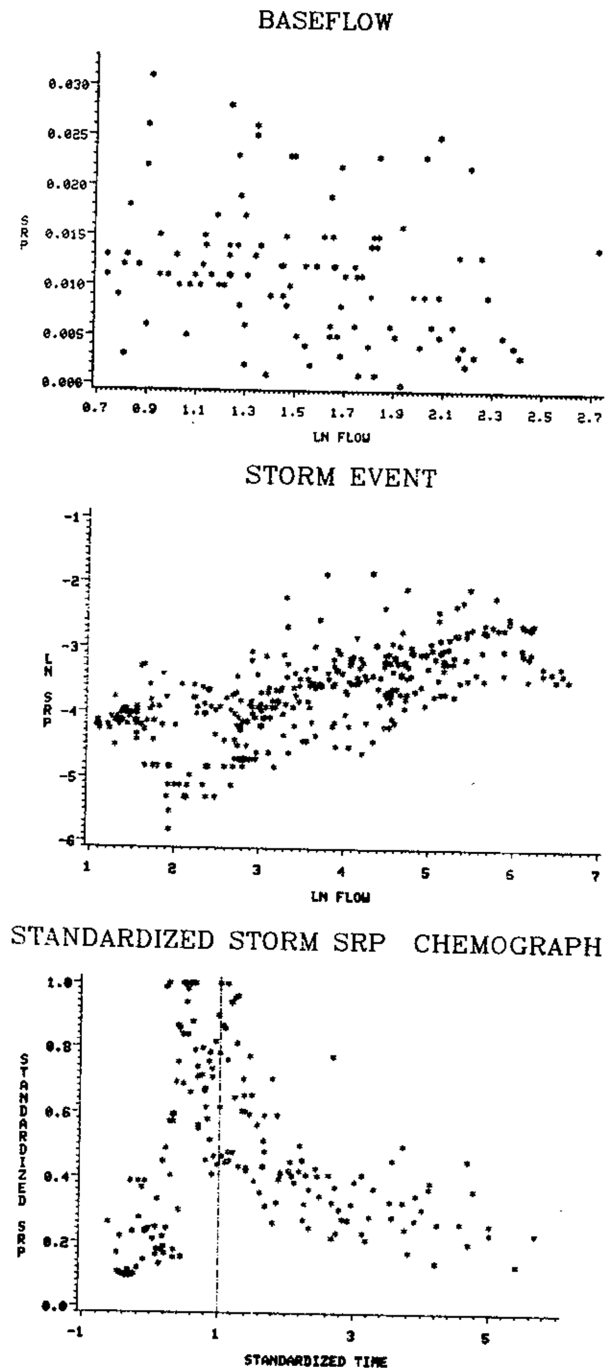


Figure 3. Typical patterns for flow concentrating variables: (a) baseflow concentration versus flow; (b) storm concentration versus flow; and (c) standardized storm chemograph.

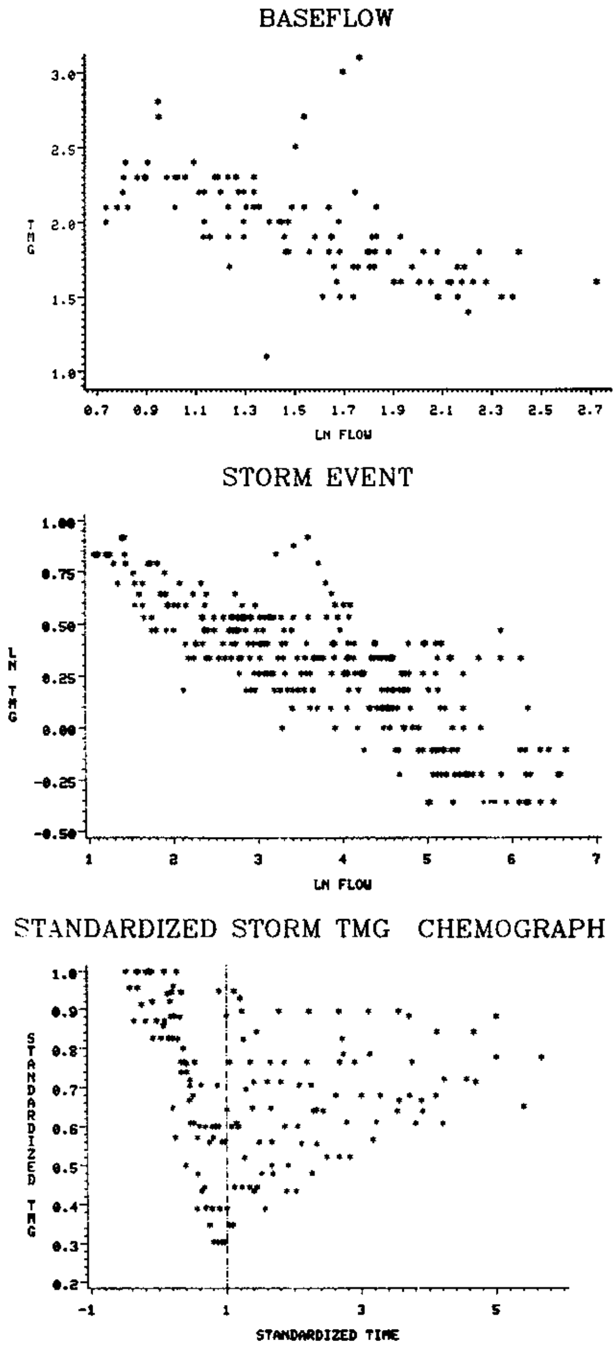


Figure 4. Typical patterns for dilution variables: (a) baseflow concentration versus flow; (b) storm concentration versus flow; and (c) standardized storm chemograph.

Table 4

Summary Statistics of Percent Fractions During Baseflow

Variable	Mean	Standard Deviation	Standard Error of Mean	Coefficient of Variation	Skewness	Kurtosis	Minimum	Maximum	Median	Mode
P-PP	.311	.192	.019	61.68	.38	-.57	0	.812	.307	0
P-SUP	.304	.227	.024	74.49	.69	.09	0	.947	.263	0
P-SRP	.502	.228	.022	45.47	.12	-.59	0	1.0	.50	.33
P-SS	.145	.141	.016	97.61	2.12	5.16	0	.718	.109	0
P-DS	.855	.141	.016	16.50	-2.12	5.16	.282	1	.891	1
P-DOC	.912	.107	.012	11.74	-2.75	8.91	0.4	1.0	.944	1

analysis. The information generated can be used to reduce efforts on similar variables and concentrate on dissimilar variables. The factor analysis utilized an iterated principal axis factoring method, an equamax rotation method, and was limited to a maximum of five factors. While a few possible associations were identified there were no consist patterns between variables.

Based on the exploratory analyses, baseflow water quality concentrations for the entire baseflow period were predicted by either regressing concentration on natural loagarithm transformed flow (if the slope of the model was significant at 0.05 level) or the selected "best" measure of central tendency was used (Table 5).

Water quality data for the 16 storm events were subjectively screened for completeness and appropriateness. Eliminated from further consideration were storms with minimal peak discharges (maximum flow less than 15 cms) or incomplete water quality records on either the rising or falling limb of the hydrograph. Based on this screening, data for 14 storms were retained for analysis (Table 6). Both static storm variables (those in Table 6) and the non-static variables FLOW, TIMESSS (time since start of storm), and AQ (accumulated flow) were used as independent variables in developing statistical water quality models during storm events (see Glossary of Symbols for variables definitions).

Summary statistics for storm water quality variables and fractions of phosphorus, solids, and carbon are presented in Tables 7 and 8, respectively. The correlation matrix and factor analysis (same method as used for baseflow) of storm water quality variables suggested three groups:

Group 1 - TMG, TCA, TNA, TSI

Group 2 - TP, PP, SRP, TKN, SS, TK, TMN, TFE, TOC, DOC, TS, NH4N

Group 3 - SUP, NO3N, DS

Although TS and NH4N fall in group 2, both had some tendencies to associate with groups 1 and 3.

Concentration in storm events is usually most significantly affected by flow and often modeled as a function of flow only. Thus, plots of natural logarithm transformed concentration and flow were examined to suggest potential statistical models for predicting concentration (Figures 3-4). To show the typical pattern of concentration in a storm, concentration and time were standardized around the time of peak flow within a storm (Figures 3-4).

Table 5

Estimators for Water Quality Concentrations During Baseflow

Baseflow Variables with Significant Slope (Ln Flow) < .05

Equation	Prob.	R^2
TP = .02959 - .004989 * Ln Flow	.0039	.0713
PP = .01198 - .0031269 * Ln Flow	.0129	.0614
SRP = .01801 - .004187 * Ln Flow	.0035	.0751
NO_3N = -.00132 + .06759 * Ln Flow	.0102	.1182
TS = 106.484 - 22.9290 * Ln Flow	.0514	.0374
DS = 83.6667 - 15.3600 * Ln Flow	.0001	.1400
TMG = 2.8128 - .4996 * Ln Flow	.0001	.4128
TCA = 22.4879 - 6.0200 * Ln Flow	.0001	.6743
TK = 1.1844 - .2149 * Ln Flow	.0001	.1959

Variables with Non-significant Slopes, Best Measure of Central Tendency

Mean	Median
TKN	SUP
TNA	NH_4N
TMN	SS
TFE	TOC
	DOC

Table 6

Hydrologic Characteristics of 14 Sampled Storms

Storm	Month	Day	Year	DURATION	TVOLUME	MAXQ	MEANQ	TTP	TLS	PQLS	MQL10	MQL25	PQL10	PQL25
7610	10	24	76	137.0	18287946	197.3	48.0	21.0	35.0	7.6	4.1	4.3	4.2	5.7
7702	2	3	77	53.5	2855223	17.1	14.2	21.5	21.0	78.5	9.6	13.9	10.9	78.5
7706	6	17	77	245	19132751	375.7	48.9	9.0	57.9	52.5	4.6	4.4	8.3	8.3
7711	11	1	77	37.5	18167157	471.87	139.0	9.5	93.2	136.3	4.0	3.5	5.2	5.2
7801	1	16	78	170	13257857	63.3	26.0	20.0	46.9	276	4.6	4.7	5.2	5.2
7803	3	6	78	73	16846569	114.2	67.9	17.0	3.7	20.4	12.6	12.5	20.4	26.3
7804	4	17	78	57	5729562	60.7	29.2	9.0	6.7	14.0	8.6	13.0	14.0	64.6
7811	11	15	78	167	44836291	510.7	103.6	17.5	63.1	22.1	2.9	2.8	2.9	2.9
7903	3	1	79	55	41207508	758.7	223.9	14.0	9.7	296.2	32.9	20.7	296.2	296.2
7905	5	3	79	113	23853226	180.9	82.0	14.0	10.2	196.1	31.8	24.2	93.7	196.1
7907	7	27	79	95	8779433	112.8	29.7	11.0	32.6	11.0	3.7	4.2	4.6	11.0
8005	5	12	80	36	2291958	21.4	17.7	12.5	11.5	21.0	10.1	14.0	21.0	104.3
8005.5	5	15	80	36	8107362	103.4	62.8	13.0	3.4	21.4	9.2	13.9	21.4	104.3
8010	10	17	80	87	10118969	162.7	34.7	7.0	18.8	150.0	3.5	11.8	4.6	150.0

Table 7

Summary Statistics of Storm Water Quality Variables

Variable	Mean	Standard Deviation	Standard Error of Mean	Coefficient of Variation	Skewness	Kurtosis	Minimum	Maximum	Median	Mode
FLOW	66.9	99.3	3.50	148.3	3.40	14.09	5.15	758.7	26.7	10.8
TP	0.1094	0.0934	0.0058	85.4	1.35	1.75	0.004	0.529	0.0795	0.034
PP	0.0696	0.0759	0.0048	109.0	1.48	2.06	0	0.394	0.047	0
SUP	0.0119	0.0102	0.0007	86.5	1.19	1.07	0	0.051	0.008	0.008
SRP	0.0342	0.0233	0.0014	68.2	2.00	6.16	0.005	0.157	0.029	0.02
NO_3N	0.288	0.183	0.0111	63.7	0.98	0.62	0.01	0.98	0.22	0.18
NH_4N	0.046	0.069	0.0041	148.4	2.96	13.10	0	0.53	0.02	0
TKN	0.860	0.494	0.0296	57.5	0.85	0.40	0.19	2.5	0.8	1
TS	134.3	114.1	6.92	85.0	2.63	8.59	31	817	94	58
DS	56.1	22.6	1.37	40.2	2.97	15.81	14	216	53	49
SS	81.1	108.4	6.70	133.7	2.88	10.47	0	757	43	12
TSI	5.94	2.17	0.129	36.6	-0.37	0.38	0.5	14.6	6.3	6.3
TMG	1.31	0.35	0.021	26.7	0.20	- 0.07	0.7	2.5	1.3	1.4
TCA	6.09	2.73	0.167	44.7	1.26	2.17	1.6	17	5.5	3.9
TNA	1.67	0.42	0.025	24.9	0.06	- 0.81	0.8	2.7	1.7	1.4
TK	1.52	0.68	0.042	44.5	1.91	5.49	0.6	5.2	1.4	0.9
TMN	0.136	0.092	0.0119	67.3	0.27	- 0.67	0	0.3	0.1	0.1
TFE	1.05	0.69	0.0894	66.1	0.94	0.59	0.2	3.2	0.85	0.5
TOC	9.18	4.89	0.308	53.2	1.22	2.01	0.8	30.5	8.1	7
DOC	7.06	3.64	0.256	51.6	0.56	0.11	0	19.2	6.35	5

Table 8

Summary Statistics of Percent Fractions During Storm Events

Variable	Mean	Standard Deviation	Standard Error of Mean	Coefficient of Variation	Skewness	Kurtosis	Minimum	Maximum	Median	Mode
P-PP	.473	.245	.016	51.8	-.53	-.71	0	.864	.524	0
P-SWP	.160	.172	.012	107.4	1.88	3.97	0	1.0	.102	0
P-SRP	.419	.222	.014	53.0	.94	.28	.083	1.0	.367	1.0
P-SS	.454	.238	.015	52.3	.02	-1.05	0	.938	.451	0
P-DS	.546	.238	.015	43.5	-.02	-1.05	.061	1.0	.599	1.0
P-DOC	.776	.193	.015	24.8	-2.20	6.08	0	1.0	.820	1.0

Standardized concentration is storm concentration divided by maximum storm concentration and standardized time is TIMESSS divided by time of peak flow. The standardized storm concentration plots allow for the visual determination of how concentration increases or decreases with increased flow and where concentration peaks occur in relation to flow.

A stepwise multiple regression technique, using Maximum R2 improvement, was performed to develop statistical models to predict storm water quality. The 11 static and 3 dynamic storm variables were used as potential dependent variables. All independent and dependent variables were transformed to natural logarithms to provide symmetric distributions. An a-priori criterion for all models was that at least one static and dynamic variable must be included. A "best" model (Table 9) was chosen by:

1. Examining the increase of R2 improvement as independent variables were added to fine asymptotic level.
2. Examining the decrease in model MSE as independent variables were added.
3. Significance of parameter and model estimates.
4. Examining changes in Type II SSi, f-ratio, and parameter estimates for possible multicollinearity.

The model variables are listed in order of significance. For each selected model, plots of residuals versus predicted and independent variables not used in the model were examined. All models except NH4N contained dynamic variables. For NH4N, the scatter plots of natural logarithm transformed concentration and flow during storms had no changing relations, thus only a static variable was used in the storm model. The storm models for TMG and TNA had multicollinearity arise when static variables were added. This suggests TMG and TNA can be modeled as a straight function of flow without static hydrologic variables. Summary statistics for the predicted storm water quality concentrations are presented in Table 10. An important note when comparing these statistics with those of the sampled storms is that the predicted set of storms (i.e., all storms during 1976-1980) contains many more storms of lower magnitude (Table 11 and 12). Thus, the water quality probability distributions and summary statistics would be shifted toward those concentrations associated with low magnitude storm events. For example, the mean flow and TP are smaller for predicted (all storms) than observed (14 storms).

Using the continuous (bi-hourly) flow record and estimators for water quality concentrations for baseflow and storm events a bi-hourly concentration

Table 9

Statistical Models for Storm Water Quality Variables

	R^2
LOG TP = -0.5155 + 0.6854*LFLOW - 1.0875*LTTP - 0.1359*LAQ	.734
LOG PP = 0.0506 + 0.9505*LFLOW - 1.2861*LTTP - 0.2616*LAQ	.647
LOG SUP = -3.2533 - 1.509*LTTP - 0.4264*LTLS + 0.6304*LDUR + 0.1858*LFLOW	.374
LOG SRP = -4.6914 + 0.4312*LFLOW - 0.3197*LMQL10	.580
LOG NO_3N = -3.3490 + 0.3772*LTLS + 0.2702*LAQ - 0.0080*TIMESSS - 0.1902*LTVOL	.422
LOG NH_4N = -0.7945 - 0.4581*LTLS - 0.2465*LPQLS	.358
LOG TKN = -0.3271 + 0.4138*LFLOW - 0.0860*LAQ - 0.0996*LPQLS	.471
LOG TS = 4.9865 + 0.4850*LFLOW - 0.1508*LAQ	.672
LOG DS = 4.1666 + 0.1113*LFLOW -0.2441*LTTP	.198
LOS SS = 4.8168 + 0.8599*LFLOW - 0.3032*LAQ	.606
LOG TSI = 2.2793 - 0.1491*LFLOW	.112
LOG TMG = 0.9638 - 0.1799*LFLOW	.560
LOG TCA = 2.8819 - 0.2871*LFLOW	.605
LOG TNA = 0.9011 - 0.1573*LFLOW - 0.0827*LTLS + 0.1804*LTTP	.607
LOG TK = 1.8008 + 0.2405*LFLOW - 0.0967*LAQ - 0.1101*LPQL10 - 0.260*TTP	.567
LOG TMN = -2.1695 + 0.4191*LPQLS + 0.3440*LFLOW - 0.1799*LAQ	.617
LOG TFE = 0.3295 + 0.5866*LFLOW - 0.2484*LAQ + 0.4577*LMQL25	.849
LOG TOC = 1.0228 + 0.3638*LFLOW - 0.1126*LPQLS	.570
LOG DOC = 1.9977 - 0.6419*LPQL10 + 0.2580*LFLOW	.552

Table 10

Summary Statistics of Predicted Storm Concentrations

Variable	Mean	Standard Deviation	Standard Error of Mean	Coefficient of Variation	Skewness	Kurtosis	Minimum	Maximum	Median	Mode
FLOW	37.5706	58.3324	0.835969	155.261	5.9209	45.6416	12.0200	776.520	20.3500	12.9000
TP	0.0632	0.1108	0.001588	175.349	4.1480	24.0831	0.0007	1.574	0.0270	0.3420
PP	0.0323	0.0796	0.001140	246.021	6.1928	60.900	0.0001	1.574	0.0080	
SUP	0.0185	0.0348	0.000500	187.632	3.6371	13.8259	0.0002	0.220	0.0077	
SRP	0.0187	0.0089	0.000128	47.618	3.1052	13.7160	0.0075	0.096	0.0163	
NO_3N	0.3068	0.1262	0.001816	41.153	1.0997	2.3583	0.0342	0.796	0.2913	0.1217
NO_4N	0.0571	0.0405	0.000583	70.924	3.4644	19.6453	0.0142	0.390	0.0460	0.437
TKN	0.4788	0.1932	0.002780	40.359	2.3754	8.5362	0.2269	1.924	2.4273	0.4161
TS	68.2157	33.6947	0.482883	49.394	2.4516	8.3098	30.0623	295.484	57.9670	90.1743
DS	47.0873	13.4269	0.192420	28.515	0.6493	0.5922	22.0086	100.227	46.5481	81.0979
SS	21.0959	23.2811	0.333645	110.359	3.9617	21.776	3.8458	238.309	13.4913	34.7242
TSI	6.0582	0.5924	0.008490	9.779	-1.3578	1.5711	3.6221	6.743	6.2342	
TMG	1.4747	0.1712	0.002453	11.608	-1.3006	1.3506	0.7918	1.676	1.5246	1.6519
TCA	7.1722	1.2587	0.018038	17.550	-1.1174	0.7089	2.6414	8.741	7.5140	8.5653
TNA	2.0973	0.5039	0.007249	24.024	0.6219	0.2286	0.9345	3.776	2.0270	3.0101
TK	0.9968	0.4460	0.006417	44.743	1.2500	2.4092	0.3070	3.959	0.9284	1.0329
TMN	0.1411	0.0950	0.001367	67.364	2.1689	7.1376	0.00374	0.816	0.1150	4.0818
TFE	0.6809	0.5122	0.007369	75.223	3.1900	14.8052	0.1209	4.788	0.5296	1.4064
TOC	5.8231	2.0244	0.029125	34.764	2.5085	9.3445	3.3165	21.541	5.2890	5.001
DOC	2.8346	2.0484	0.029471	72.266	1.8019	6.4826	0.2598	18.360	2.7512	

Table 11

Summary Statistics for Static Storm Variables for all Storms in Continuous Flow Record (n=86)

Variable	Mean	Standard Deviation	Standard Error of Mean	Coefficient of Variation	Skewness	Kurtosis	Minimum	Maximum	Median	Mode
DURATION	111	101	11	91.249	1.64630	3.2044	0.0	486	94	126.0
TVOLUME	15315186	20078784	2165151	131.104	2.49124	7.0362	87212.3	107836992	8941211	87212.3
MAXQ	123	165	18	134.371	2.15146	4.2812	12.1	777	51	14.0
MEANQ	29	18	2	60.044	1.50495	2.2964	12.1	99	24	12.1
TTP	18	34	4	187.283	4.77735	27.9502	0.0	256	10	2.0
TLS	21	25	3	119.993	2.54422	6.5698	0.3	123	12	6.0
MQL10	11	8	1	75.266	2.57584	9.6683	2.5	54	8	11.3
MQL25	13	9	1	68.649	1.55806	3.1974	2.4	51	11	20.4
PQL10	41	72	8	174.428	4.01527	20.3734	2.9	498	15	10.8
PQL25	116	161	17	139.198	2.28625	5.0729	2.9	777	53	115.5
PQLS	118	160	17	135.316	2.28556	5.1453	12.1	777	50	14.0

Table 12

Summary Statistics for Static Variables from Sampled Storms (n=14)

Variable	Mean	Standard Deviation	Standard Error of Mean	Coefficient of Variation	Skewness	Kurtosis	Minimum	Maximum	Median	Mode
DURATION	97	63	17	64.323	1.13	0.79	36	245	80	36
TVOLUME	16676558	12906143	3449312	77.391	1.21	1.00	2291958	44836291	15052209	2291958
MAXQ	225	221	59	98.140	1.35	1.17	17	759	138	17
MEANQ	66	57	15	86.629	1.84	3.69	14	224	48	14
TTP	14	5	1	33.968	0.27	-1.15	7	22	14	9
TLS	30	27	7	91.374	1.15	0.76	3	93	20	3
MQL10	10	10	3	97.322	1.87	2.54	3	33	7	5
MQL25	11	7	2	64.107	0.56	-0.49	3	24	12	14
PQL10	37	78	21	213.670	3.26	11.05	3	296	10	5
PQL25	76	88	24	116.980	1.40	1.66	3	296	45	5
PQLS	75	88	23	116.227	1.55	1.84	8	296	24	8

record was established for the period 1 Jan 76 - 31 Dec 80. A corresponding material load record was established by integrating the area under the material load curve. Summary statistics for predicted daily concentrations and load for all variables are shown in Tables 13 and 14, respectively.

DISCUSSION

Based on the information generated, three groups of water quality variables were identified based on characteristics during baseflow and storm events. The three groups were 1) flow concentrating, 2) flow dilution, and 3) no pattern. The characteristics of each group will be discussed separately and while there were definite patterns between groups, a continuum from one group to another was observed.

The flow concentrating group had a strong positive linear relation between concentration and flow having the same shape curve during storm events (Figure 3). The peak concentration occurred slightly in advanced of peak flow, especially as storm magnitude increased. The mean and standard deviation (std) of observed baseflow samples tended to be less than the mean and std from observed routine, suggesting the significance of storms (i.e., increased concentration and variation with increased flow). Observed baseflow concentrations usually had a negative relation with flow until approximately 12 cms, which probably arose due to dilution. At flows greater than 12 cms, the increased flow caused runoff and resuspension to occur and concentration to increase. The observed storm concentrations had means from 2-6 times larger than baseflow while coefficients of variation (cv) were smaller. Observed storm concentrations were strongly correlated with flow, hence flow was usually the most significant independent variable in the storm model. The mean of predicted storm concentrations were usually less than observed because of the higher percentage of low magnitude storms in the predicted storm set (all storms) in comparison to the set used to develop the storm models (14 storms). This occurred because the higher percentage of small storms caused lower predicted mean flows; therefore the predicted storm concentrations had smaller means and maximums. In comparison, the means and maximums of complete set of predictions (baseflow and storm) were smaller than observed routine conditions. The overall set of predictions also tended to have the same or larger maximum and range. The group consisted of (in order from most applicable to least):

Table 13

Summary Statistics for Predicted Daily Water Quality Concentrations (1976-1980)

Variable	Mean	Standard Deviation	Standard Error of Mean	Coefficient of Variation	Skewness	Kurtosis	Minimum	Maximum	Median	Mode
TP	0.0307	0.0463	0.001085	150.695	7.51	72.74	0.0008	0.728	0.0221	0.0234
PP	0.0126	0.0305	0.000714	242.094	8.79	99.43	0.0001	0.522	0.0072	0.0082
SUP	0.0080	0.0161	0.000377	201.759	8.21	77.43	0.0003	0.198	0.0050	0.0050
SRP	0.0128	0.0050	0.000118	39.167	4.11	27.97	0.0056	0.075	0.0120	0.0128
NO3N	0.1524	0.1024	0.002397	67.203	2.32	6.97	0.0528	0.779	0.1231	0.0818
NHYN	0.0282	0.0226	0.000530	80.326	5.12	41.53	0.0142	2.359	0.0200	0.0200
TKN	0.4148	0.0854	0.001999	20.584	4.80	31.81	0.2346	1.418	0.3967	0.3967
TS	69.0161	16.4512	0.385092	23.837	1.93	11.10	32.5810	215.550	68.8978	78.2843
DS	56.1396	9.8874	0.231447	17.612	-0.65	0.74	22.4687	89.700	56.6805	64.7827
SS	10.9116	10.1771	0.238227	93.268	6.07	49.87	4.4398	148.165	8.0000	8.0000
TSI	6.4100	0.3134	0.007337	4.889	-3.74	15.47	4.1344	6.673	6.5106	6.5108
TMG	1.8822	0.3038	0.007112	16.143	-0.24	-0.64	0.9295	2.410	1.8797	2.1957
TCA	11.4822	3.3740	0.078979	29.385	0.02	-1.11	3.4252	17.671	11.2804	15.0876
TNA	2.1157	0.2243	0.005250	10.601	1.36	11.26	1.0978	3.667	2.1209	2.1209
TK	0.8706	0.2068	0.004841	23.754	2.04	8.96	0.3195	2.370	0.8546	0.9202
TMN	0.0770	0.0517	0.001209	67.052	4.24	24.12	0.0396	0.657	0.0589	0.0589
TFE	0.3061	0.2790	0.006531	91.148	4.69	31.72	0.1411	3.386	0.2000	0.2000
TOC	3.3889	1.5092	0.035328	44.534	2.95	11.14	2.7000	15.607	2.7000	2.7000
DOC	2.7297	0.9137	0.021387	33.472	3.92	40.86	0.2781	15.792	2.7000	2.7000

Table 14

Summary of Statistics for Predicted Daily Material Loads (1976-1980)

Variable	Mean	Standard Deviation	Standard Error of Mean	Coefficient of Variation	Skewness	Kurtosis	Minimum	Maximum	Median	Mode
TP	76168	405452	9491	532.316	11.30	155.26	1003	7441136	10778	6927
PP	42919	276743	6478	644.805	11.75	162.20	123	5004076	3298	2419
SUP	14002	56167	1315	401.140	9.68	119.78	330	896645	2582	1427
SRP	25572	114563	2682	448.008	12.07	183.36	2816	2125526	5689	3793
NO3N	268149	749819	17552	279.628	8.00	88.08	10156	12115757	70730	24178
NHYN	45880	129257	3026	281.728	9.32	139.20	3847	2783098	11272	5910
TKN	673131	2613702	61182	388.290	12.40	207.08	76304	60217665	223558	117220
TS	105124399	426510929	9983866	405.720	12.38	195.72	16951368	9142252474	35728688	23132080
DS	57500988	127073755	2974572	220.994	9.29	115.45	13729237	2245214700	30826080	19142512
SS	41189181	274841573	6433555	667.266	15.15	287.26	1538784	6771193003	4508639	2363903
TSI	6313394	9783503	229014	154.964	6.27	54.94	1252301	132689965	3667368	1923804
TMG	1631884	2195825	51400	134.558	6.14	52.39	463589	29347087	1052527	648795
TCA	8453993	8304427	194392	98.231	5.35	40.01	3399187	102570940	6920418	4458194
TNA	2148530	3581292	83832	166.686	6.50	57.59	407951	45374910	1190410	626700
TK	1166332	3598329	84321	308.517	10.51	144.13	194735	63843344	438940	271893
TMN	170135	816789	19120	480.083	16.41	380.10	11329	23026852	33195	17404
TFE	989392	5923407	138656	598.692	15.09	274.95	38470	126109281	112716	59098
TOC	7132624	29826444	698184	418.169	12.03	195.62	519340	675874164	1521665	797818
DOC	3766951	15273640	357529	405.464	15.77	336.54	384201	409259748	1443711	797818

PP, SS, TP, SRP, TFE, TOC, TKN, TS, NO3N, DS, DOC, TK, and TMN. All variables from PP to TKN were definite members of the group while the other variables had the following discrepancies: TS showed flow dilution until flows exceeded 12 cms; NO3N tended to asymptote out at flows greater than 50 cms; DS had a smaller percentage of storm load; DOC, TMN, and TK had larger variations in concentration at high flows.

In the dilution group, concentration had an inverse relation with flow and peaked simultaneously with flow (Figure 4). Mean observed baseflow concentrations were slightly larger than routine and had a very strong negative relation with flow. The observed mean storm concentration was less than baseflow and had an increase in cv and decrease in minimum and maximum. The storm model was usually a function of only flow with the addition of static variables usually causing multicollinearity. The predicted storm concentrations had a larger mean and smaller std and range than observed storm concentrations because the large percentage of small storms caused less dilution (increased mean), and less change in concentration (smaller std and ranges). The complete set of predictions (baseflow and storm) were very similar to the observed routine data because the biases in baseflow and storm predictions tended to cancel each other. The dilution group consisted of TCA, TMG, TNA, and TSI. Both TCA and TMG were strong members with TNA having larger variation in concentration at low flows and TSI having larger variations at low to medium flows.

Group 3, the no pattern or high variation group, showed very little pattern with flow during baseflow or storms (Figure 5). The lack of pattern may occur because the variables had characteristics of both group 1 and 2 combined or actually exhibit no consistent relationships at all. The observed baseflow concentrations were very similar to observed routine with both sets of data having large cv. The means of observed storm concentrations were slightly higher than those for baseflow. Since no strong influence was exerted by any dynamic independent variable, the peaks of predicted concentrations in storm events were flat. The group consisted of SUP and NH4N.

The contributions of storm event loads to the total monthly and annual loads are typified in Figures 6-7, respectively, with the annual percentages for all variables in Table 15. Before examining the water quality load plots, it is important to examine the amount of water load from baseflow and storm

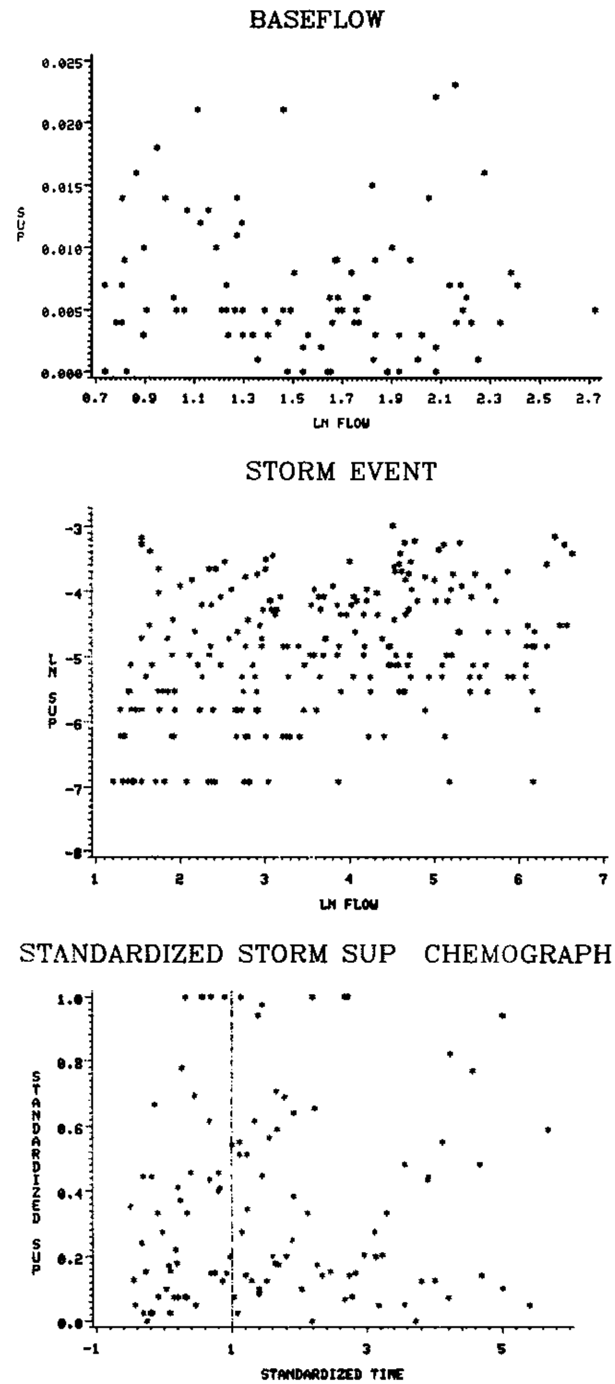


Figure 5. Typical patterns for no pattern variables: (a) baseflow concentration versus flow; (b) storm concentration versus flow; and (c) standardized storm chemograph.

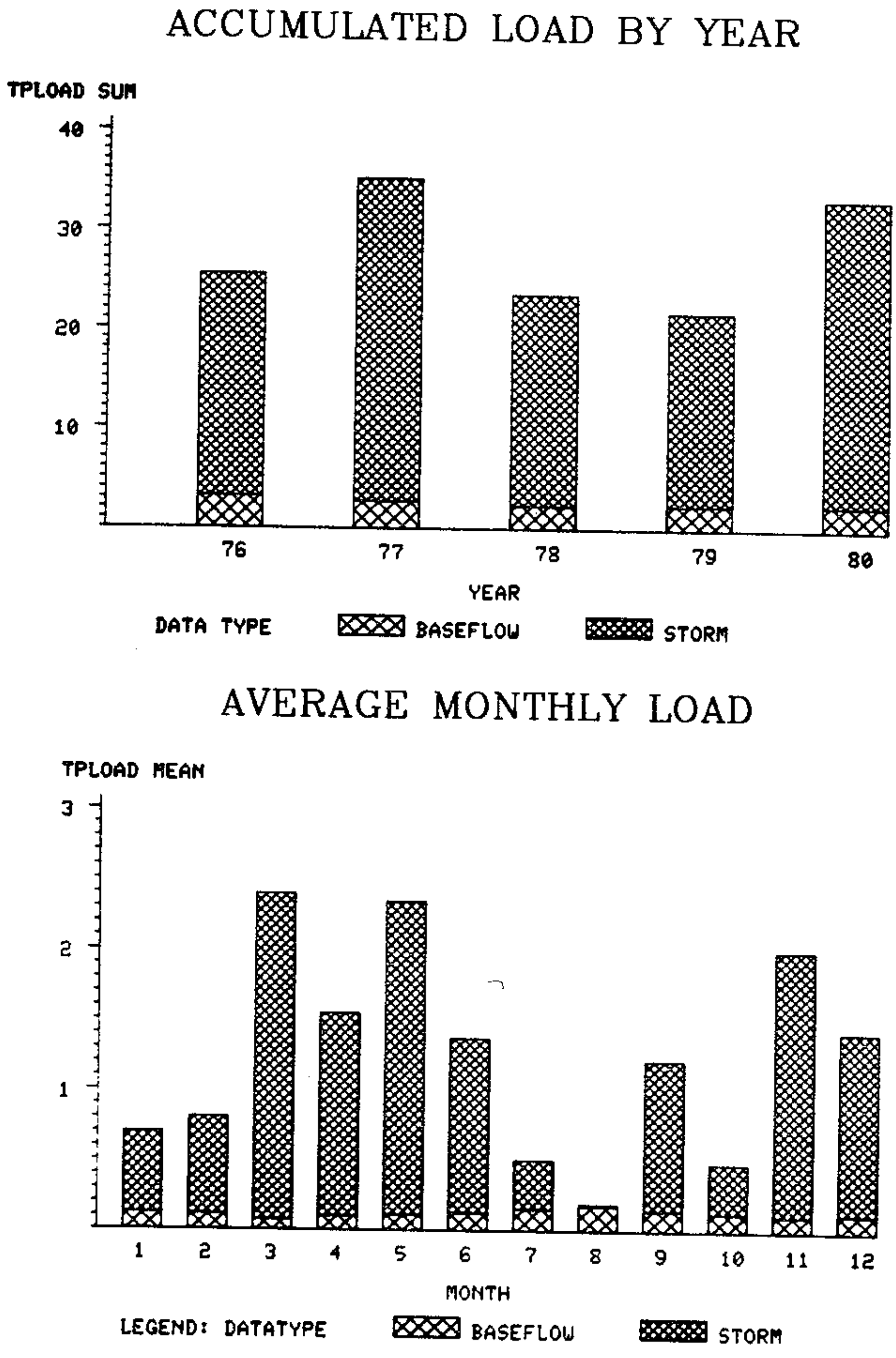
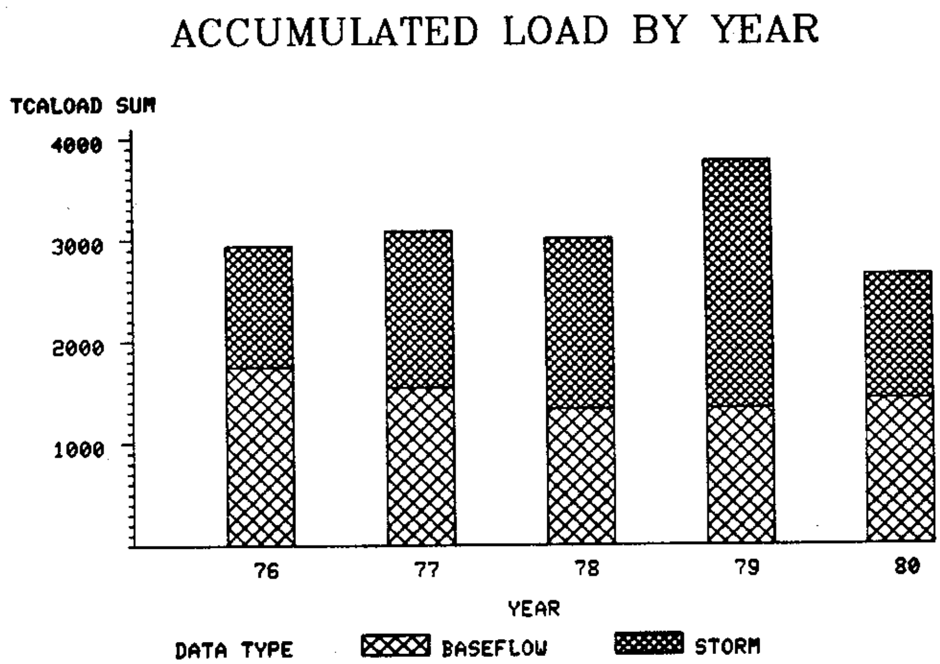


Figure 6. Typical patterns for flow concentrating variables: (a) annual load (separated into baseflow and storm) for 1976-1980; and (b) mean monthly load (separated into baseflow and storm).



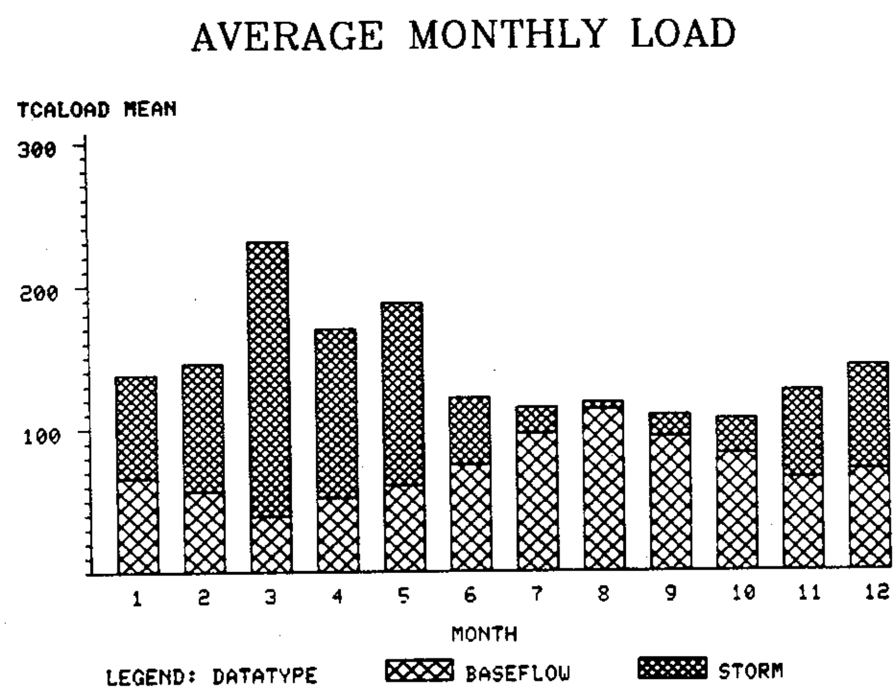


Figure 7. Typical patterns for dilution variables: (a) annual load (separated into baseflow and storm) for 1976-1980; and (b) mean monthly load (separated into baseflow and storm).

Table 15

Percentages and Coefficient of Variation of Total Annual Water Quality Load From Baseflow and Storm Events

VARIABLE	BASEFLOW %	BASEFLOW CV	STORM %	STORM CV	MEAN ANNUAL LOAD (GRAMS)	ANNUAL LOAD CV
FLOW	0.327	28.4	0.673	13.8	4819959388	24.1
TP	0.097	22.6	0.903	2.4	27801150	21.5
PP	0.056	29.0	0.944	1.7	15665417	29.4
SUP	0.130	27.4	0.870	4.1	5110697	29.3
SRP	0.154	36.1	0.846	6.5	9333684	30.5
NO3N	0.164	35.8	0.836	7.0	97874218	27.1
NH4N	0.163	39.9	0.837	7.8	16746252	28.2
TKN	0.220	35.9	0.780	10.1	245692908	31.7
TS	0.234	34.8	0.766	10.7	38370405731	33.2
DS	0.344	20.9	0.656	10.9	20987860603	15.7
SS	0.079	47.0	0.921	4.0	15034050985	45.5
TSI	0.367	25.5	0.633	14.8	2304388669	20.0
TMG	0.415	22.7	0.585	16.2	595637723	17.6
TCA	0.486	18.9	0.514	17.9	3085707310	13.4
TNA	0.363	32.1	0.637	18.3	784213572	29.1
IK	0.240	21.2	0.760	6.7	425711270	13.8
TMN	0.142	45.5	0.858	7.5	62099203	47.8
TFE	0.088	50.8	0.912	4.9	361128026	56.8
TOC	0.146	41.2	0.854	7.1	2603407712	37.8
DOC	0.250	21.8	0.750	7.3	1374937129	12.9

events (Figure 8), for if storm events have no effect on concentration during storms the percentages of water load and water quality load should be very similar. However, if increased flows caused increased concentrations, then storm events should contribute a higher percentage than water load; whereas if dilution of concentration occurred with increased flow, a decrease in storm load percentages should be expected. For concentration variables, storm events dominated the total load (94-98 percent) with medium to large cv's. In contrast, the percentage of storm event contribution to the total load for dilution variables was much smaller (51-64 percent).

A pattern that emerged within all variables was that as the percentage of storm event contribution increased, the cv decreased. This suggests that the variables that are significantly concentrated by flow do so in a consistent manner, while dilution variables have increased variation in storm event percentages in comparison to baseflow. However, this pattern must be viewed in the light that even though a smaller cv existed with variables having large storm loadings, a small variation at the higher percentage actually caused more change in loading than a larger variation at a lower percentage. Another important pattern was that even though dilution variables had high cv's of percentages during storm events, they had lower annual cv's because the lower percentage cv's during baseflow coupled with more baseflow loadings created lower annual variations.

Both baseflow and storm event material loading have seasonal patterns, exemplified in Figures 6-7 and Tables 16 and 17 for all variables, respectively. The total mean monthly loads for concentrating variables occurred on a definite seasonal basis: very high in spring, decreasing in summer and increasing in fall. For dilution variables, the total mean monthly loads were fairly uniform throughout the year. For all variables, the seasonality in storm loading was similar to patterns in water loading with the major peaks in the spring and slightly smaller ones in the fall. In comparison, baseflow material loads peaked in summer. In both storm and baseflow loadings, when the monthly percentage increased, the cv decreased, and may exhibit the same properties of changing percentages and cv as was examined for total annual loads.

The fact that material loads occur in a seasonal manner can have an important impact on the response of the system (lake or reservoir) based on the uneven impulse loading. Thus, even though annual estimates may predict

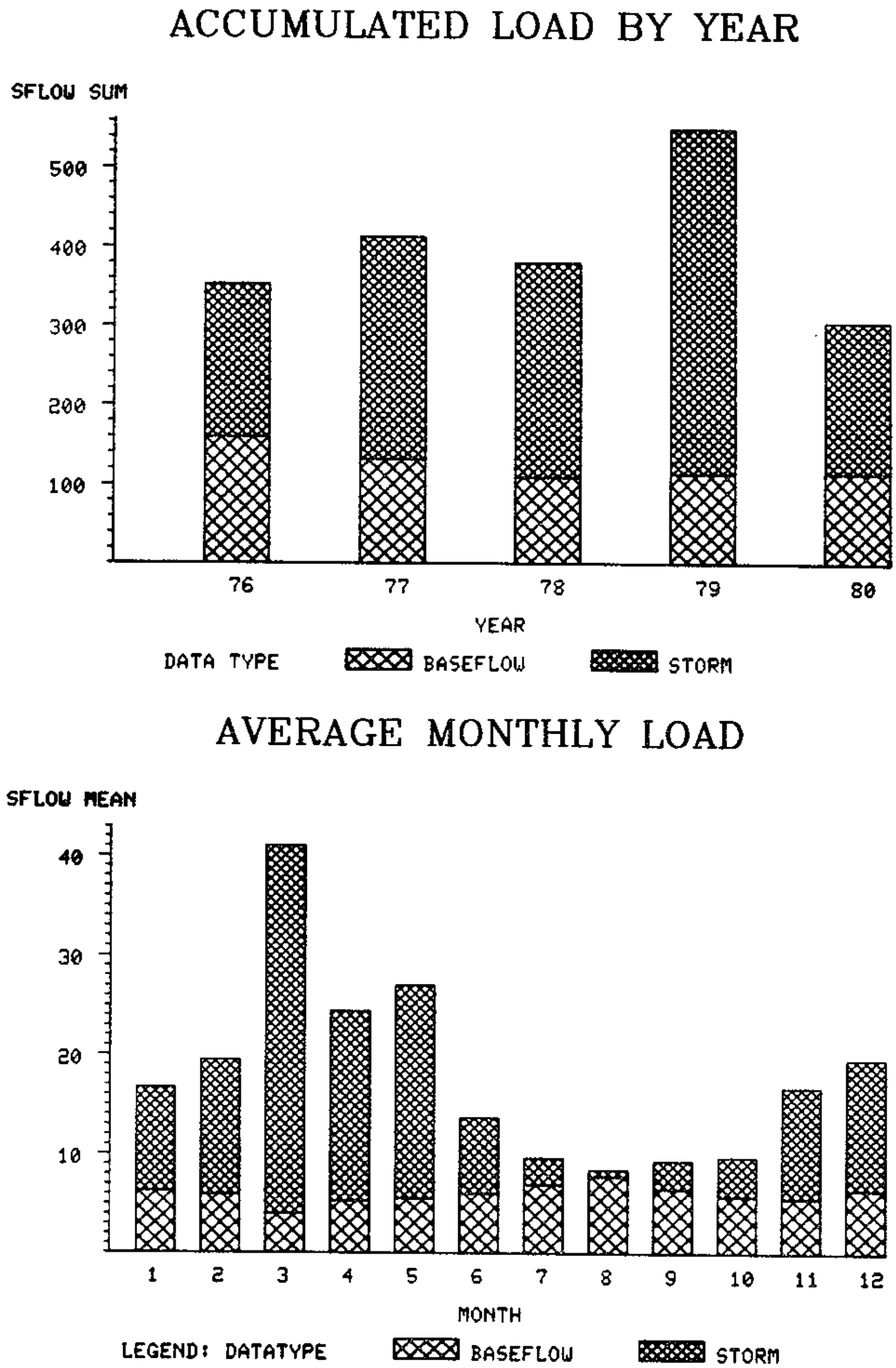


Figure 8. Baseflow and storm: (a) annual water load (1976-1980); and (b) mean monthly water load.

Table 16

Monthly Mean and Coefficient of Variation of Percent Baseflow Load

VARIABLE	JAN S	JAN SCV	FEB S	FEB SCV	MAR S	MAR SCV	APR S	APR SCV	MAY S	MAY SCV	JUN S	JUN SCV	JUL S	JUL SCV	AUG S	AUG SCV	SEP S	SEP SCV	OCT S	OCT SCV	NOV S	NOV SCV	DEC S	DEC SCV
FLOW	.589	33.1	.652	29.5	.874	11.3	.694	34.7	.619	61.1	.437	71.2	.198	140.8	.055	223.6	.204	116.2	.245	128.0	.517	59.3	.599	42.2
TP	.773	15.1	.722	29.9	.915	7.5	.822	24.7	.739	56.7	.659	60.0	.337	137.4	.043	223.6	.364	137.6	.409	105.0	.707	42.0	.795	24.1
PP	.816	13.9	.753	28.5	.929	7.8	.853	20.4	.767	56.1	.689	58.5	.363	137.2	.039	223.6	.380	137.1	.458	98.9	.764	35.1	.838	20.8
SUP	.587	26.8	.708	34.0	.866	10.3	.799	30.1	.756	56.3	.659	58.0	.252	143.0	.052	223.6	.309	141.9	.284	122.8	.610	54.4	.769	24.2
SRP	.711	32.1	.781	23.2	.941	8.0	.786	30.0	.687	57.9	.552	70.4	.276	137.3	.082	223.6	.262	153.9	.336	127.4	.629	53.4	.736	36.1
NO3N	.746	25.1	.760	19.2	.934	5.7	.775	26.0	.662	59.1	.563	70.7	.284	137.3	.076	223.6	.328	140.7	.373	113.8	.645	48.4	.720	32.9
NH4N	.672	30.7	.799	24.1	.935	6.1	.810	26.6	.736	56.4	.598	58.5	.245	159.8	.153	223.6	.213	156.0	.270	126.3	.666	36.7	.780	23.0
TKN	.636	31.6	.706	31.2	.914	9.5	.724	34.4	.658	59.0	.506	69.8	.238	141.3	.072	223.6	.244	156.7	.288	127.4	.580	56.6	.681	41.8
TS	.635	30.8	.701	30.4	.914	10.7	.726	34.7	.651	59.9	.485	72.3	.230	139.0	.054	223.6	.220	162.7	.283	128.8	.562	61.6	.666	44.2
DS	.576	27.6	.632	30.2	.862	11.6	.693	35.2	.616	60.7	.444	71.1	.187	139.2	.041	223.6	.211	159.3	.240	126.9	.513	63.4	.595	41.2
SS	.795	20.8	.816	22.0	.963	5.2	.820	27.1	.727	56.8	.615	64.9	.331	137.2	.095	223.6	.306	144.1	.382	120.4	.693	44.5	.789	31.5
TKN	.571	34.2	.633	29.0	.853	11.9	.680	34.5	.605	61.6	.410	71.3	.184	141.5	.053	223.6	.191	160.5	.229	128.2	.497	58.3	.567	40.9
TMG	.541	36.2	.614	30.3	.836	13.6	.661	36.0	.589	62.5	.374	73.8	.159	142.3	.044	223.6	.160	163.7	.203	130.2	.465	61.1	.531	42.2
TCA	.499	38.8	.586	31.6	.806	16.0	.634	37.7	.564	63.8	.325	77.1	.131	143.7	.034	223.6	.125	164.4	.170	131.3	.420	62.7	.476	43.1
TNA	.570	36.9	.637	31.9	.858	13.1	.678	36.2	.606	62.7	.391	73.7	.180	147.1	.066	223.6	.157	166.3	.212	131.3	.479	57.9	.561	43.9
TK	.652	28.0	.722	23.6	.908	7.9	.755	29.0	.660	58.5	.518	66.4	.246	137.3	.051	223.6	.268	147.7	.309	121.8	.604	52.3	.609	33.0
TMN	.730	31.4	.765	16.7	.946	6.9	.815	24.7	.680	60.2	.557	69.5	.308	137.4	.059	223.6	.251	155.2	.363	119.5	.619	55.3	.711	37.5
TFE	.761	26.2	.848	15.0	.965	5.0	.858	21.6	.732	57.0	.623	63.0	.303	137.7	.088	223.6	.272	149.4	.367	122.0	.676	45.4	.808	25.7
TOC	.713	28.9	.785	24.3	.944	6.5	.789	29.4	.703	57.4	.574	67.6	.274	139.5	.101	223.6	.270	160.0	.325	126.0	.643	48.8	.754	32.3
DOC	.665	28.8	.699	25.7	.895	11.1	.671	31.2	.559	62.0	.432	72.3	.236	137.8	.034	223.6	.283	147.0	.331	124.0	.591	56.1	.642	42.6

Table 17

Monthly Mean and Coefficient of Variation of Percent Storm Load

VARI-	JAN		FEB		MAR		APR		MAY		JUN		JUL		AUG		SEP		OCT		NOV		DEC	
ABLE	B	BCV	B	BCV	B	BCV	B	BCV	B	BCV	B	BCV	B	BCV	B	BCV	B	BCV	B	BCV	B	BCV	B	BCV
FLOW	.411	47.5	.348	55.2	.126	77.8	.306	78.6	.381	99.1	.563	55.3	.802	34.8	.945	13.0	.796	41.2	.755	41.6	.483	63.6	.401	63.0
TP	.227	51.2	.278	77.7	.085	81.6	.178	113.9	.261	160.1	.341	115.8	.663	69.7	.957	9.9	.636	78.6	.591	72.6	.293	101.4	.205	93.5
PP	.184	61.9	.247	87.1	.071	101.5	.147	117.9	.233	184.5	.311	129.8	.637	78.1	.961	9.1	.620	84.0	.542	83.5	.236	113.4	.162	107.9
SUP	.413	38.1	.292	82.3	.134	66.4	.201	119.9	.244	174.6	.341	112.0	.748	48.2	.948	12.3	.691	63.5	.716	48.7	.390	85.2	.231	80.4
SRP	.289	79.0	.219	82.7	.059	126.2	.214	110.2	.313	127.3	.448	86.8	.724	52.3	.918	20.0	.738	54.7	.664	64.4	.371	90.7	.264	100.5
NO3N	.254	73.9	.240	61.1	.066	80.3	.225	89.5	.338	116.1	.437	90.9	.716	54.3	.924	18.3	.672	68.8	.627	67.6	.355	88.1	.280	84.5
NH4N	.328	62.9	.201	95.7	.065	87.9	.190	113.5	.264	157.2	.402	87.2	.755	51.9	.847	40.5	.787	42.3	.730	46.7	.334	73.2	.220	81.7
TKN	.364	55.1	.294	75.0	.086	100.5	.276	90.1	.342	113.7	.494	71.6	.762	44.2	.928	17.3	.756	50.5	.712	51.5	.420	78.2	.319	89.4
TS	.365	53.6	.299	71.3	.086	112.6	.274	92.2	.349	111.9	.515	68.2	.770	41.5	.946	12.8	.780	46.0	.717	50.8	.438	79.0	.334	88.1
DS	.424	37.5	.368	51.9	.138	72.5	.307	79.5	.384	97.6	.556	56.8	.813	32.0	.959	9.6	.789	42.5	.760	40.0	.487	66.9	.405	60.6
SS	.215	76.1	.184	97.3	.037	134.9	.180	123.9	.273	151.3	.385	103.7	.669	67.8	.905	23.4	.694	63.6	.618	74.4	.307	100.4	.211	117.8
TSI	.429	45.5	.367	50.1	.147	69.4	.320	73.2	.395	94.1	.590	49.6	.816	31.8	.947	12.6	.809	37.8	.771	38.0	.503	57.5	.433	53.6
TMG	.459	42.5	.386	48.3	.164	69.6	.339	70.3	.411	89.5	.626	44.1	.841	26.9	.956	10.2	.840	31.2	.797	33.1	.535	53.0	.469	47.9
TCA	.501	38.7	.414	44.8	.194	66.4	.366	65.3	.436	82.7	.675	37.1	.869	21.6	.966	7.9	.875	23.4	.830	26.9	.580	45.3	.524	39.2
TNA	.430	48.9	.363	56.0	.142	79.5	.322	76.2	.394	96.3	.609	47.3	.820	32.2	.934	15.7	.843	31.1	.788	35.2	.521	53.3	.439	56.0
TK	.348	52.4	.278	61.3	.092	78.0	.245	89.4	.340	113.3	.482	71.4	.754	44.8	.949	12.0	.732	54.2	.691	54.6	.396	79.7	.310	73.5
TMN	.270	84.9	.235	54.6	.054	122.2	.185	108.6	.320	127.8	.443	87.2	.692	61.1	.941	13.9	.749	52.1	.637	68.3	.381	89.8	.289	92.2
TFE	.239	83.4	.152	83.7	.035	137.6	.142	130.3	.268	155.9	.377	104.0	.697	59.8	.912	21.6	.728	55.9	.633	70.9	.324	94.9	.192	108.3
TOC	.287	71.6	.215	88.7	.056	109.7	.211	110.0	.297	136.0	.426	90.9	.726	52.6	.899	25.2	.730	55.9	.675	60.6	.357	88.0	.246	99.1
DOC	.335	57.3	.301	59.7	.105	94.9	.329	63.6	.441	78.5	.568	55.1	.764	42.7	.966	7.8	.717	57.9	.669	61.5	.409	81.1	.358	76.4

similar annual loads, by not incorporating the seasonal pattern in material loads, they have limited usefulness. Also, if annual input-output models are used, the fact that seasonal loading exists may have a significant impact on the coefficients used in the model. Another important aspect of spring seasonal loads is the possibility for the large amounts of material loads to be added either directly to the water or sediment and provide the spring starting point for water quality concentrations and an internal nutrient source to become available later in the year (Kennedy et al., 1983).

The change in chemical species composition between baseflow and storm event loadings was most pronounced in variables having dissolved and particulate fractions. The fraction of particulate or suspended material was much larger than dissolved during storm events (eg., PP to SUP and SS to DS). Thus, the variables consisting of a significant amount of both dissolved and suspended species and which have a large percentage of storm loadings may be comprised of many more particulates than reflected by the percentages calculated from using observed routine data and a direct loading estimator. For example, if storm events contribute 90 percent of the total load and 90 percent of storm loads are particulates, then out of 100 units a year, 81 are particulates. The importance of increased percentage of particulates is augmented by presence of seasonal loading, in that the spring load, high in particulates, may be deposited and utilized later during anoxic conditions. However, if no anoxic conditions occur, then the spring peaks high in particulates may overestimate the actual amount of phosphorus available, for the majority, when they would be deposited and lost to the system (i.e., sink). The increase in particulates may be of special importance in riverine or dendritic type reservoirs, where in the headwaters, or arms, flow may diminish enough to allow decomposition of material loads and thus change the species composition (i.e., particulates to dissolved). Hence, when high flows do occur, resuspension and scour could cause a large amount of internal loading.

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GLOSSARY OF SYMBOLS

A. Water Quality Variables

Symbol	Description
DOC	Dissolved organic carbon
DS	Dissolved solids
NH4N	Ammonia nitrogen
NO3N	Nitrate nitrogen
PP	Particulate phosphorus
SRP	Soluble phosphorus
SS	Suspended solids
SUP	Soluble unreactive phosphorus
TCA	Total calcium
TFE	Total iron
TK	Total potassium
TKN	Total Kjeldahl nitrogen
TMG	Total magnesium
TMN	Total manganese
TNA	Total sodium
TOC	Total organic carbon
TP	Total phosphorus
TS	Total solids
P-DOC	DOC/TOC
P-DS	DS/TS
P-PP	PP/TP
P-SRP	SRP/TP
P-SS	SS/TS
P-SUP	SUP/TP

B. Storm Model Independent Variables

Symbol	Description
AQ	Accumulated flow since start of storm (cubic meters)
DURATION	Duration of storm (hours)
FLOW	Flow (cms)
MAXQ	Maximum flow of storm (cms)
MEANQ	Mean flow of storm (cms)
MQL10	Mean flow last 10 days (cms)
MQL25	Mean flow last 25 days (cms)
PQLS	Peak flow last storm (cms)
PQL10	Peak flow last 10 days (cms)
PQL25	Peak flow last 25 days (cms)
TIMESSS	Time since start storm (hours)
TLS	Time since last storm (days)
TTP	Time to peak flow from start of storm (hours)
TVOLUME	Total volume of storm (cubic meters)

THE THERMAL STRUCTURE AND CHARACTERISTICS OF DeGRAY LAKE[1]

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Abstract

Thermal stratification is one of the most important phenomena affecting reservoir water quality. It inhibits the vertical exchange of heat and dissolved and suspended materials and causes chemical and biological reaction rates to vary with depth.

Since the factors which influence stratification vary annually, synoptically, and diurnally, the thermal structure of DeGray Lake can be characterized at these same time scales. Data collected from 1975 to 1980 show that only a few differences in its stratification cycle occur annually. Synoptic events cause deviations or variations in the smoothed thermal cycle seen during an annual overview of the system. Diurnal variations are small and are controlled primarily by the daily cycle of solar radiation.

Temperature is one of the most significant factors impacting reservoir water quality. It directly affects biological growth and chemical reaction rates and indirectly affects vertical exchange through its influence on water density and viscosity: vertical density gradients create buoyancy forces which inhibit mixing while increases in viscosity reduce settling velocities. Since the factors which influence temperature in a reservoir vary at several time scales it is possible to characterize its thermal structure at these same time scales. At DeGray Lake, AR, the thermal structure can be characterized at three time scales: annual, synoptic, and diurnal.

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[1]The data used in this paper were collected by Dr. Joe F. Nix and his staff, Ouachita Baptist University, as part of the Environmental and Water Quality Operational Studies which were sponsored by the Office, Chief of Engineers, US Army Corps of Engineers, Washington, DC. The data analysis and paper preparation were supported by FTN, Ltd.

Thermal Stratification

Thermal stratification is defined as the layering of water due to density differences. The density of reservoir water is influenced by temperature (Fig. 1) and dissolved and suspended solids. The influence of the latter can be considered negligible at DeGray Lake, however, since solids concentrations are usually very low (Ford and Johnson 1983) (it takes approximately 330 mg/l of dissolved solids or 420 mg/l of suspended solids with a specific gravity of 2.65 to equal the effect of a 1°C change at 25°C).

Solar radiation supplies nearly all the energy for heating in the reservoir. When the radiant energy enters the water, it is attenuated with depth by scattering and absorption by the water and suspended material in it. Some of the absorbed energy is lost due to back radiation, conductance (with the atmosphere and sediments), and evaporation (Fig. 2), but the majority is dissipated as heat.

The vertical distribution of the heat within the reservoir depends not only on the attenuation of light but also on mixing due to advection, convection, and wind. These processes are described in detail by Ford (1983).

When stratification is fully developed as a result of the combined effect of these processes, the reservoir can be characterized by several zones (Fig. 3). The warm, turbulent upper stratum of nearly uniform temperature is known as the epilimnion. The cold deep region of the reservoir which remains isolated from the upper portion is termed the hypolimnion. The zone in between these two, the metalimnion, is characterized by a strong temperature gradient: the plane of maximum temperature gradient within the metalimnion is known as the thermocline. An examination of an annual cycle for DeGray Lake illustrates how these features evolve.

Annual Cycle

DeGray Lake is monomictic. Overturn occurs in late January or early February when solar radiation is low and energy from the wind is able to overcome the remaining stratification, mixing the reservoir throughout

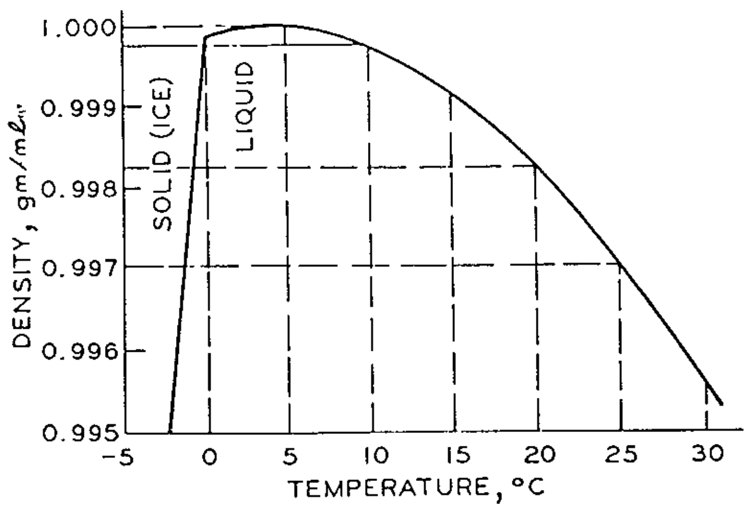

Fig. 1. The relationship between temperature and density.

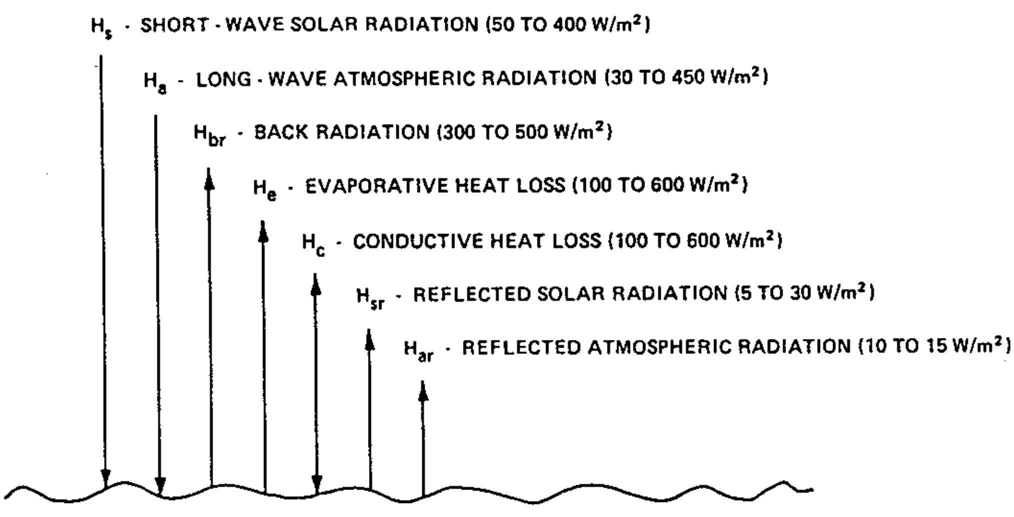

Fig. 2. The heat budget at the reservoir surface.

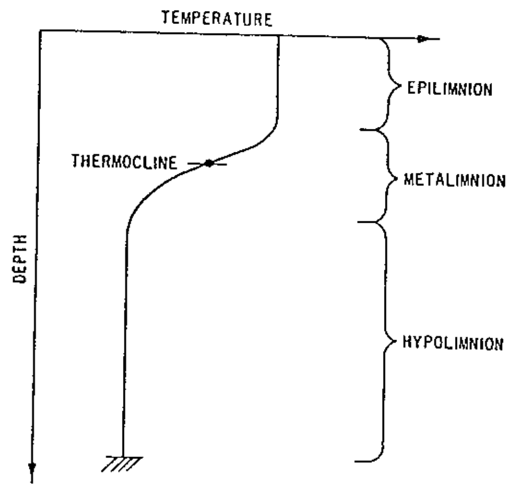

Fig. 3. Terminology for a stratified body of water.

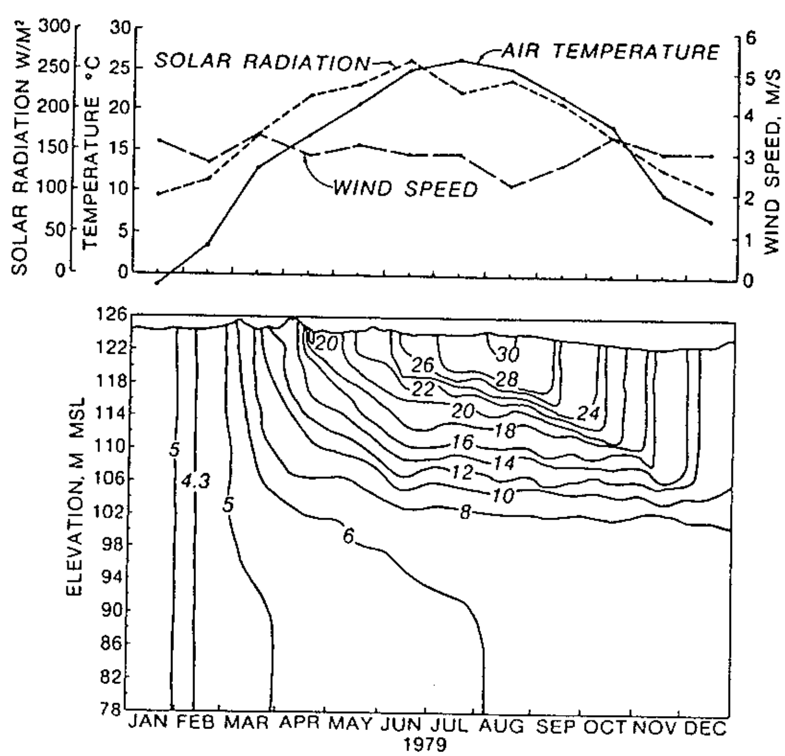

Fig. 4. Stratification cycle for DeGray Lake, 1979 with corresponding meteorological conditions. Isotherms are in ^{o}C.

its depth. In 1979 overturn resulted in a $5^{o}C$ isothermal reservoir (Fig. 4). Stratification began building again in March. Fig. 4 shows that heat was mixed to progressively shallower depths as solar radiation increased and wind speeds remained relatively constant. Near summer solstice the thermocline reached its minimum depth.

The remainder of the year was characterized by a deepening of the thermocline and the initiation of turnover (Fig. 4). As solar radiation began the declining phase of its annual cycle, mixing processes were able to increase the thickness of the epilimnion. Surface temperatures peaked in August (nearly a month later than air temperature due to the large specific heat of water) and then declined steadily as heat losses to the cooler fall and winter air exceeded the gains from solar radiation. These losses caused convective mixing which penetrated to increasingly deeper depths from September to December in Fig. 4.

Similar trends can be seen in the annual cycles from 1975 to 1980 (Figs. 5 - 10). Stratification builds from the bottom each year starting in March (Julian Days (JD) 60 to 90) with a minimum thermocline depth occurring near summer solstice. Surface temperatures peak in July or August (JD 180 to 240) and then decrease continually until overturn. Only a slight increase in hypolimnetic temperatures occurs during the year, however, indicating that little mixing occurs into the hypolimnion during the summer.

Some differences in thermal structure do occur from year to year. As Figs. 5 through 10 show, temperatures in the freely circulating reservoir in January and February vary slightly from 1975 to 1980. Meteorological records (Fig. 11) show that the colder temperatures during January and February of 1977 through 1979 were due to colder than normal winters. The winters of 1974-75, 1975-76, and 1979-80 were near normal, resulting in warmer temperatures during January and February of 1975, 1976, and 1980. These temperatures are important in that they determine the hypolimnetic temperatures for the following year and when overturn eventually occurs.

The slopes of the isotherms in Figs. 5 through 10 also vary from year to year. The steeply sloping isotherms in the metalimnion during 1979 and 1980 (Figs. 9 and 10) are due in part to a change in project operation. In March 1979 the outlet level was lowered to the elevation shown in

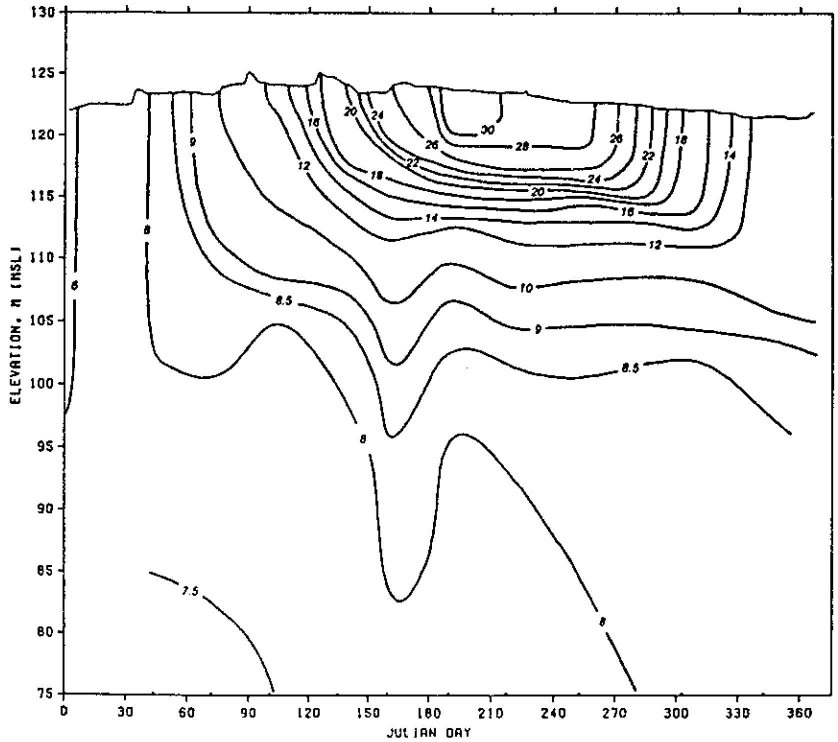

Fig. 5. Stratification cycle for DeGray Lake, 1975.

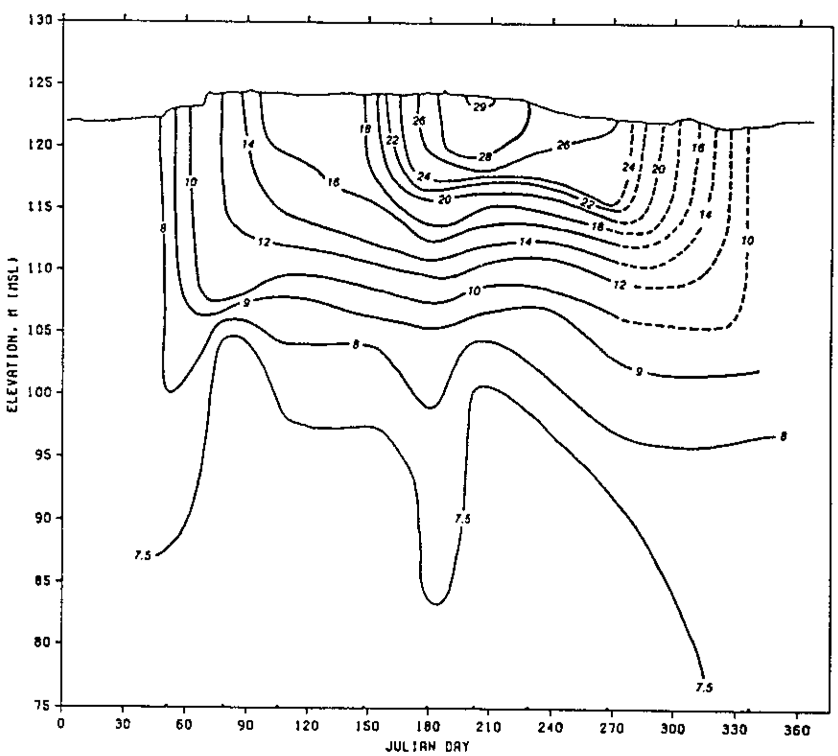

Fig. 6. Stratification cycle for DeGray Lake, 1976.

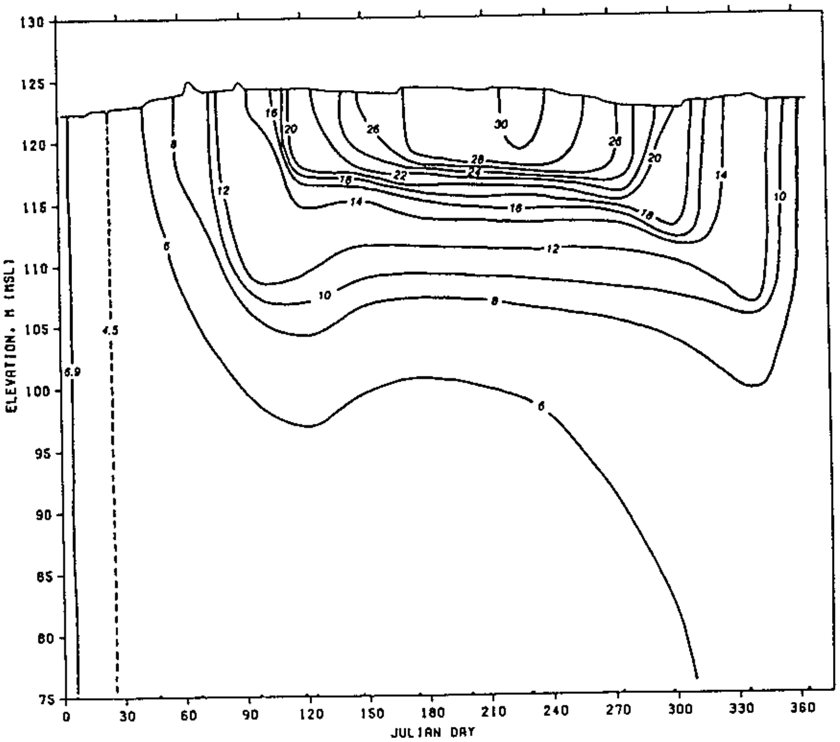

Fig. 7. Stratification cycle for DeGray Lake, 1977.

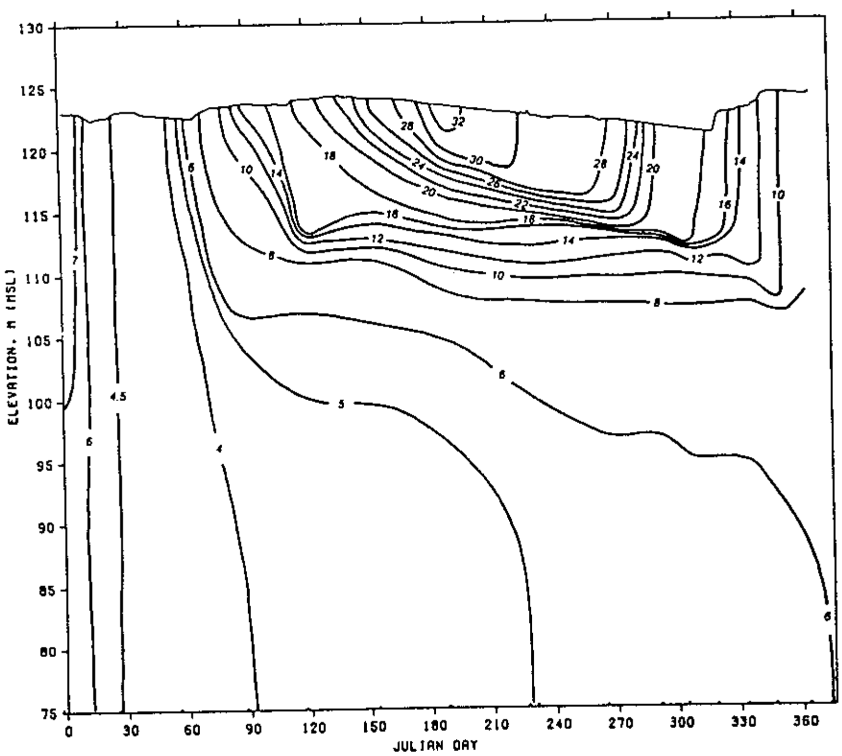

Fig. 8. Stratification cycle for DeGray Lake, 1978.

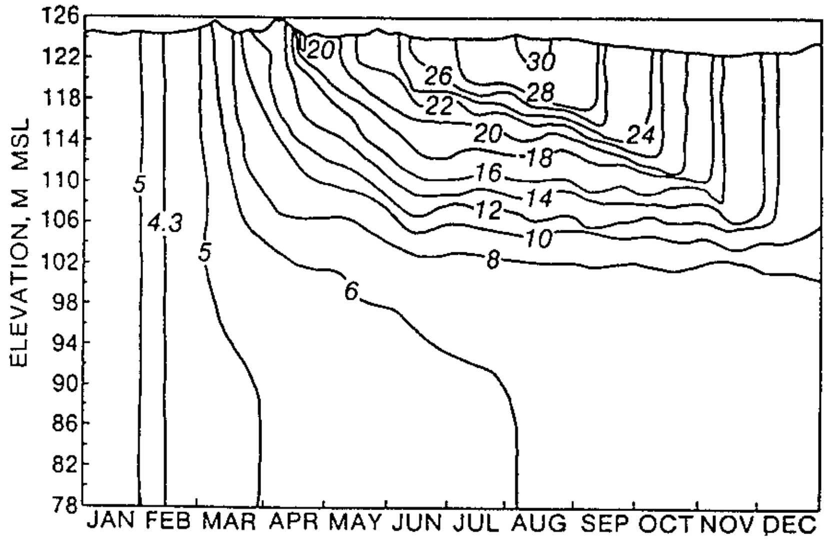

Fig. 9. Stratification cycle for DeGray Lake, 1979.

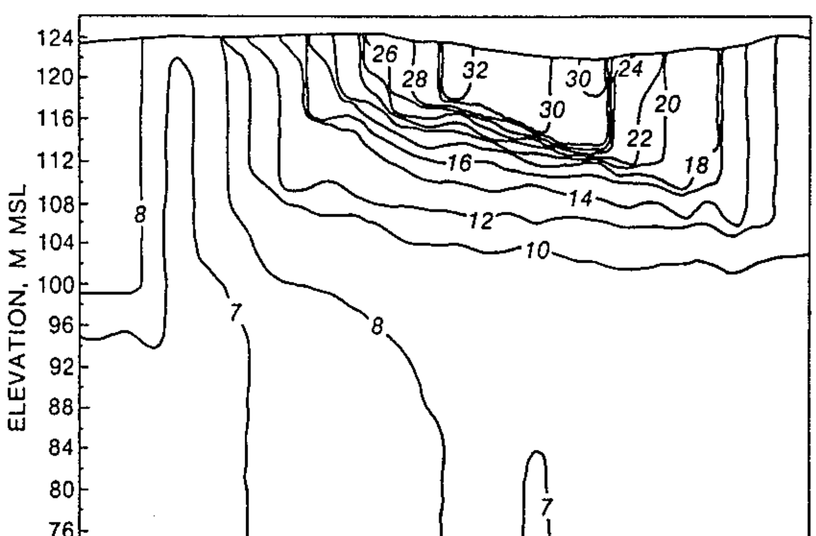

Fig. 10. Stratification cycle for DeGray Lake, 1980.

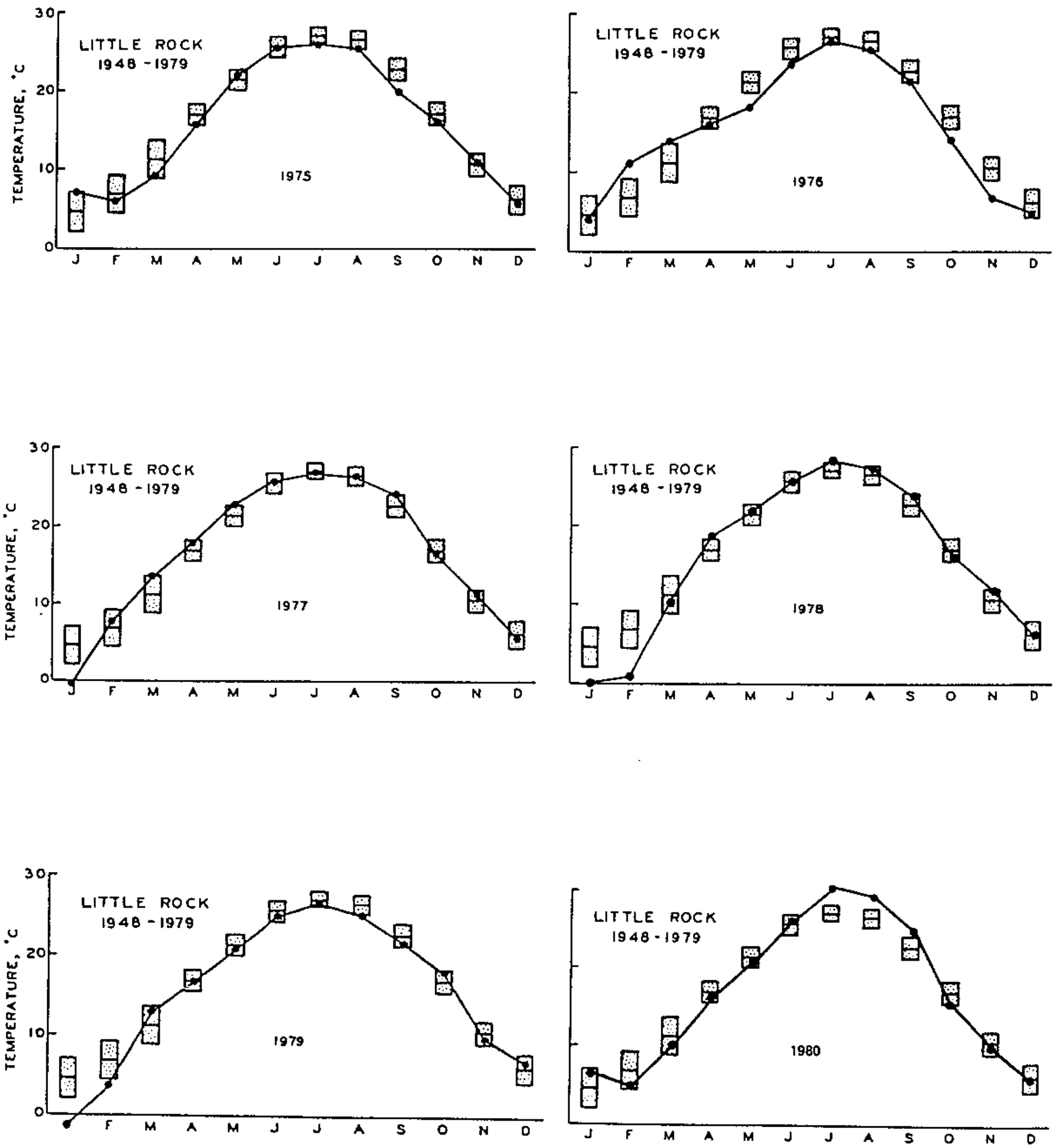

Fig. 11. Mean monthly variation of air temperature compared to mean and standard deviation, 1975-1980.

Fig. 12. The effect was to continually draw down the upper layers of warm water to replace the cooler metalimnetic waters which were being released and, in general, to increase mixing in the metalimnion. Wind speeds during this period were below average indicating that wind mixing probably was not a significant factor. During 1979, however, large storm inflows entered the metalimnion (unlike base flows which entered the epilimnion) and created additional mixing (Fig. 13).

Since inflows and outflows can have a significant impact on the thermal structure of the reservoir, they are important to characterize on an annual basis also. A typical annual distribution of inflow is shown in Fig. 13. Large inflow events occur from March through May with little additional flow occurring until November or December. Outflows occur in response to inflow events and power generation. As a result, outflows are generally larger than inflows during the summer months as evidenced by the falling water surface elevation in Figs. 5 through 10.

Inflow and outflow temperatures follow a general sinusoidal pattern annually. Fig. 14 shows the inflow pattern with measurements made from 1976 to 1979 superimposed upon it. Inflow temperatures generally vary from 4°C in January to 35°C in mid-July, lagging changes in air temperatures by a few days. Outflow temperatures vary less throughout the year, peaking approximately a month later than inflow temperatures. As would be expected, the effect of lowering the outlet level was to decrease outflow temperatures and to delay the peak. Fig. 15 shows these relationships with respect to the variation in surface temperatures.

Synoptic Changes

Variations in the thermal structure of DeGray Lake also occur synoptically or as a result of the passage of weather systems (i.e., take place over several days). An example of such an occurrence is shown in Fig. 16. Data from a thermistor chain and meteorological station from 26 February to 16 March 1980 show the formation and destruction of stratification in response to meteorological events. Cold, windy, and cloudy conditions from 29 February to 1 March produced an isothermal layer 6 m deep. Calm, clear conditions on 2 March allowed stratification to build in the top 6 m only to be destroyed by high winds the following 2 days. Stratification occured again from 5-10 May after 5 days of high solar radiation and calm winds.

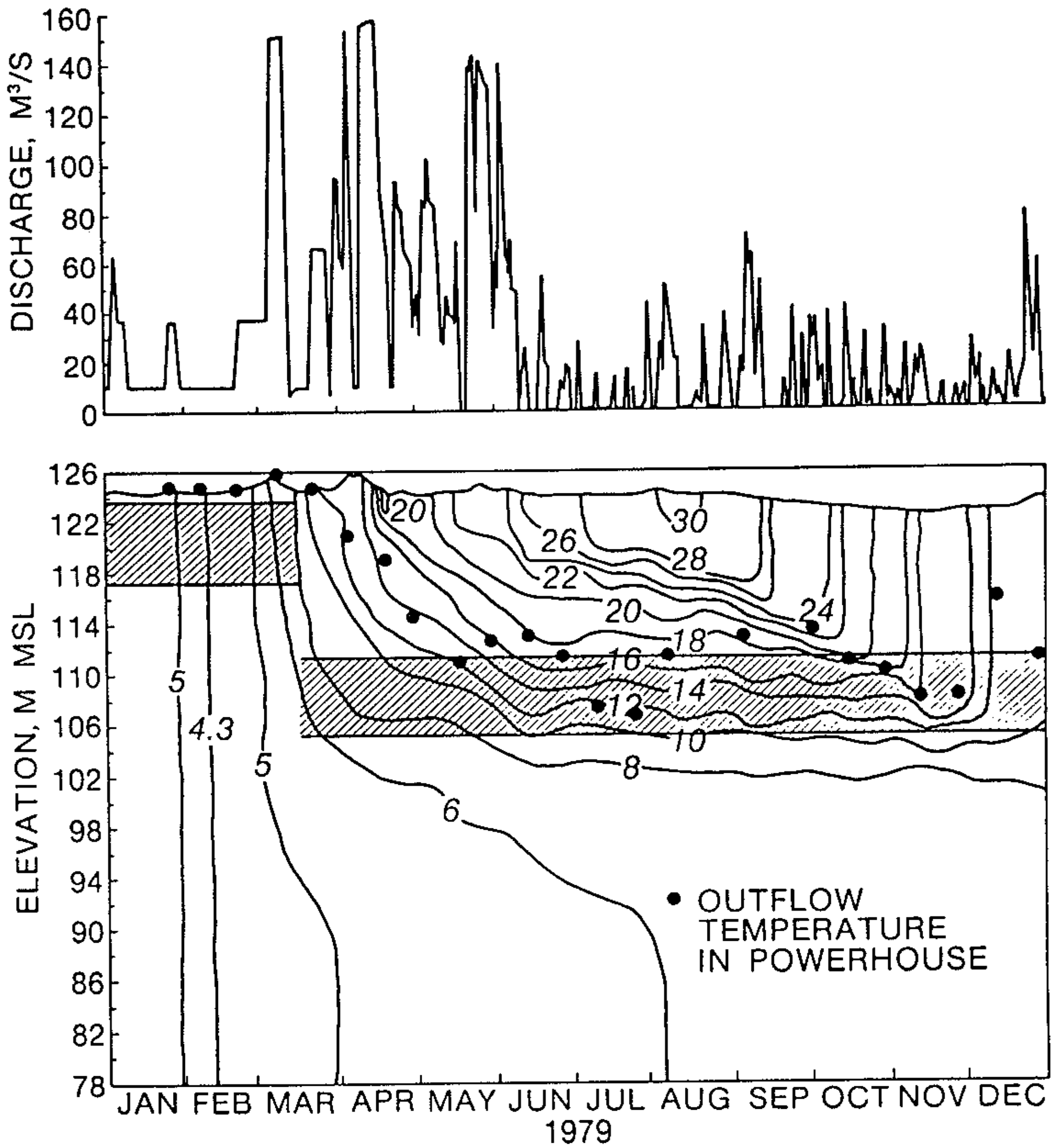

Fig. 12. Stratification cycle for DeGray Lake, 1979 with corresponding outflow rates.

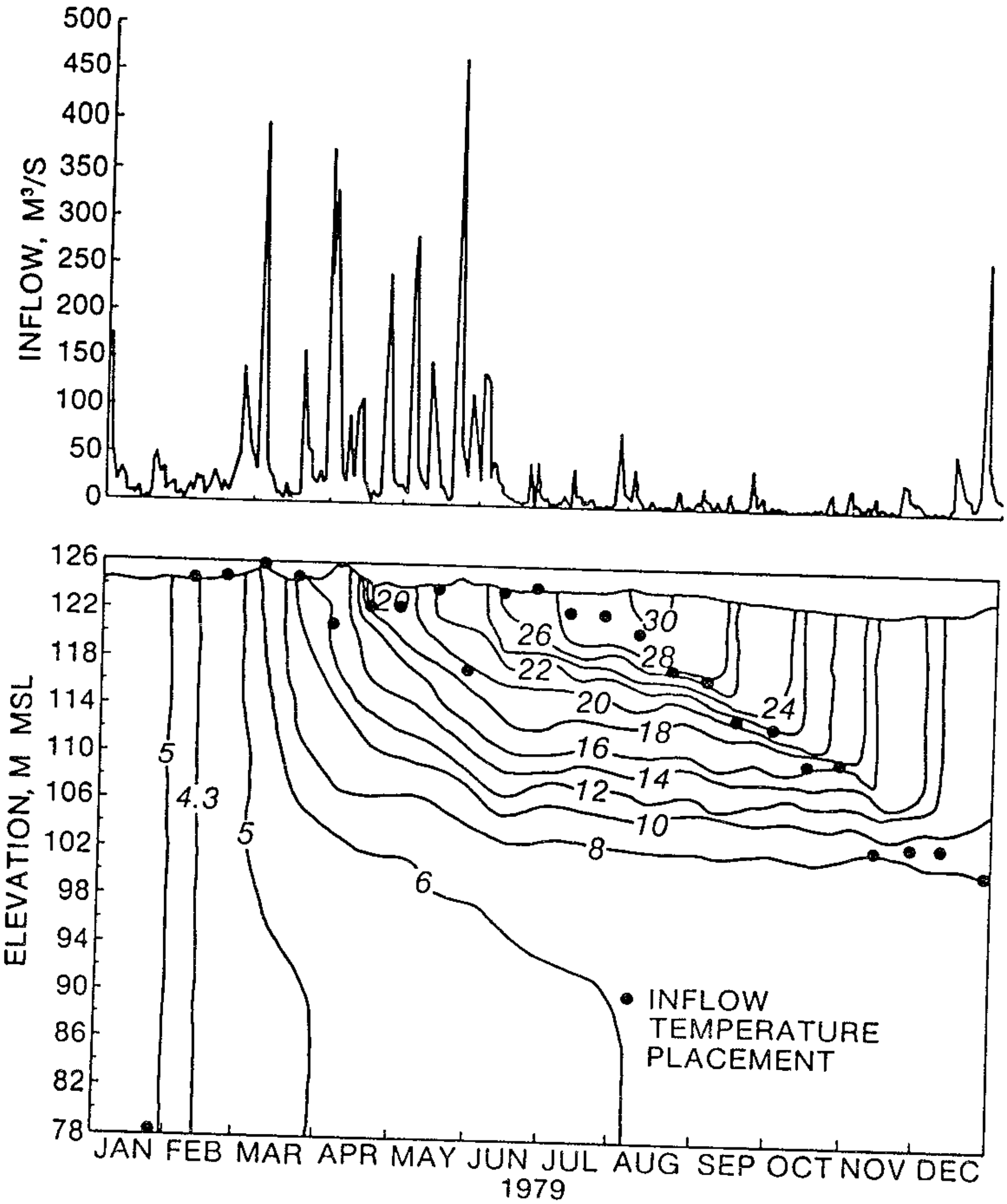

Fig. 13. Stratification cycle for DeGray Lake, 1979 with corresponding inflow rates and placement within the pool.

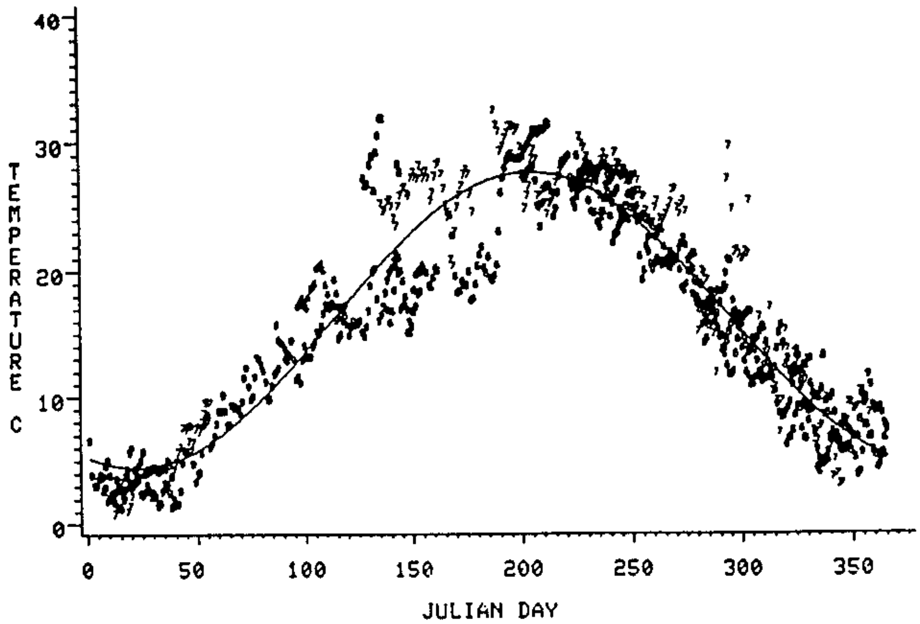

Fig. 14. Sinusoidal variation in Caddo River inflow temperature to DeGray Lake and observed temperatures from 1976–1979. Numbers represent last digit in data year.

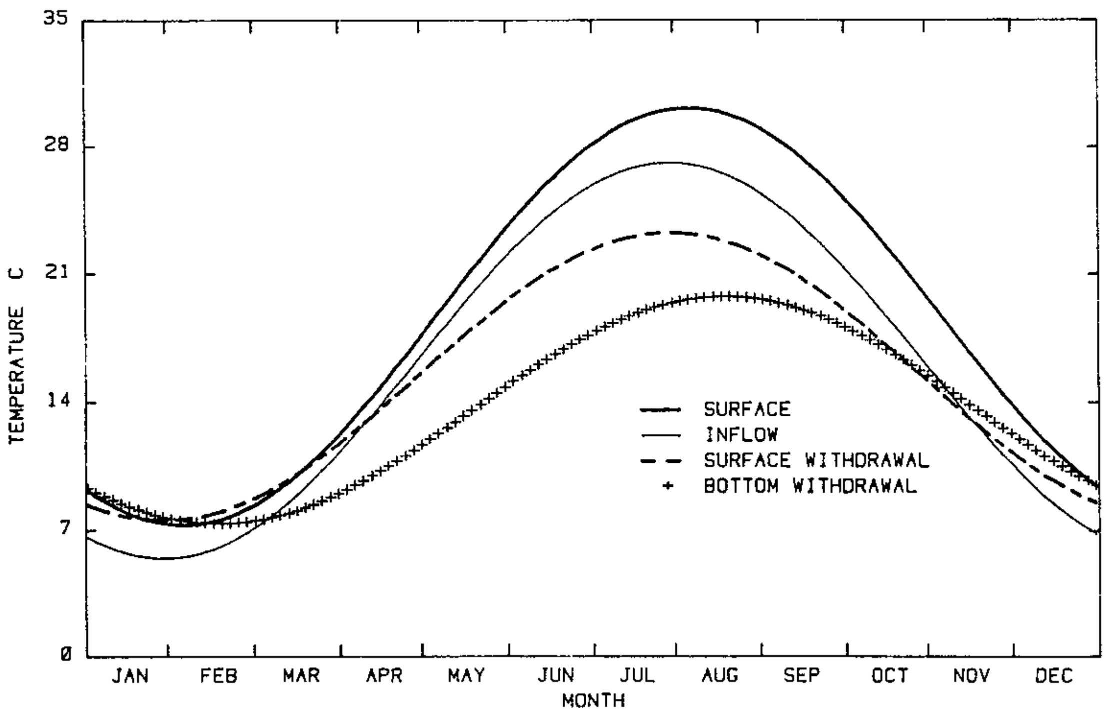

Fig. 15. General relationship between the inflow, outflow, and water surface temperatures at Lake DeGray, 1982–83.

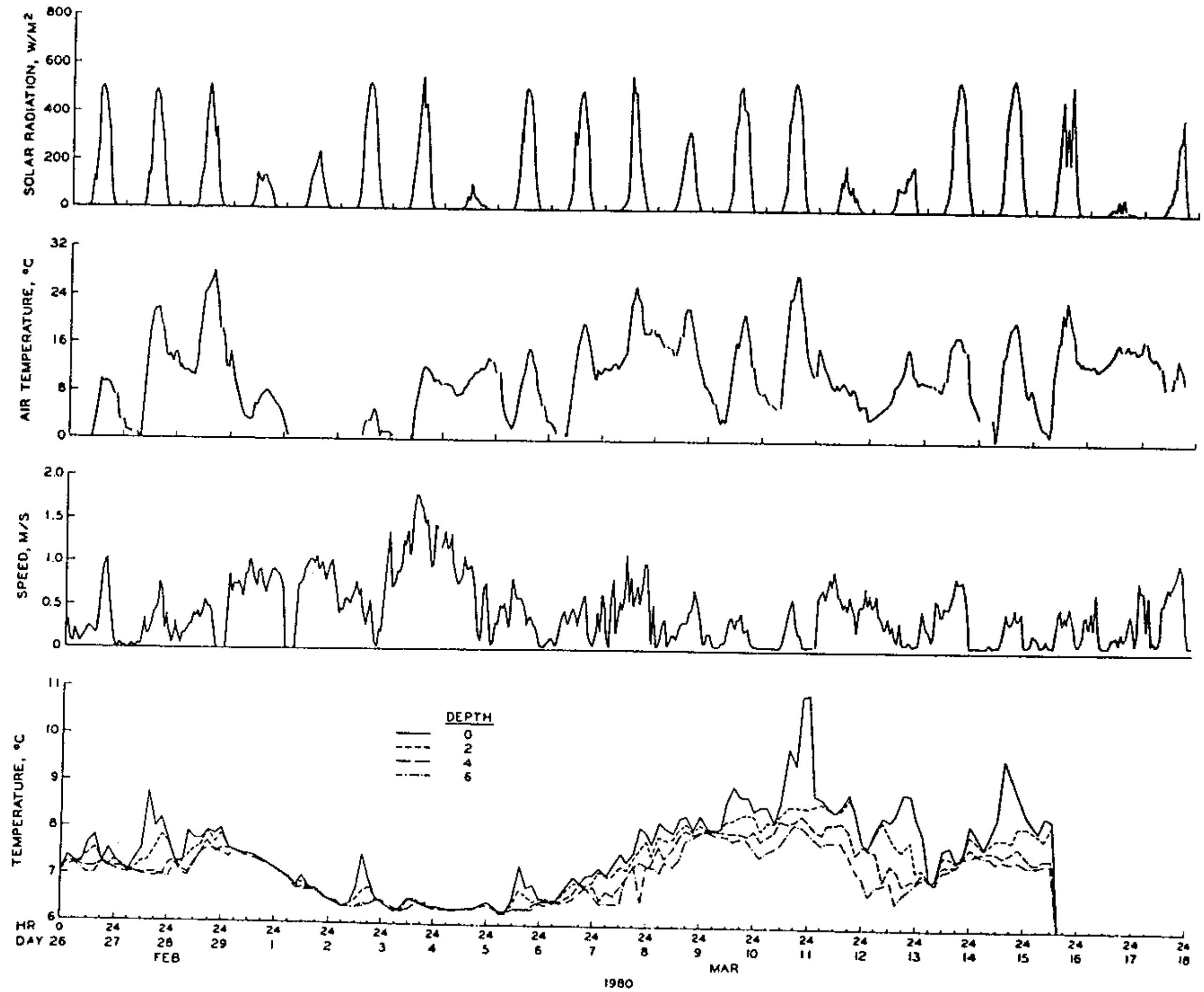

Fig. 16. Response of temperature at the surface, 2, 4, and 6 m to solar radiation, air temperatures, and wind speed, 26 February - 18 March 1980.

Inflow temperatures are also subject to synoptic variations. Fig. 17 shows the effect of a storm event which occurred in late March 1979. Temperatures coinciding with the peak of the hydrograph are nearly 4°C cooler than those preceding and following the storm. As a result, the inflow entered the pool at a lower elevation than base flows during the same period (Fig. 13). Larger temperature variations also occur. An example is shown in Fig. 18 for a storm event which occurred early in May 1979. Seldom do inflow temperatures to DeGray Lake change more than 10°C during synoptic events, however.

Diurnal Variations

Diurnal variations are caused by the well-defined cycle of solar radiation and the somewhat erratic variations in the daily cycle of the wind (winds peaking during the afternoon). In the absence of wind and cloudy conditions, surface layers will heat and stratify during the day; at night they will cool, sink, and mix as the result of radiative losses. A typical July diurnal cycle at DeGray Lake in shown in Fig. 19. At 0800 hrs the top 4 m are isothermal at 28.2°C. After 8 hrs of solar heating, stratification developed throughout the 4 m depth with the surface temperature increasing by over 2.5°C. Diurnal variations of the same magnitude occur in the Caddo River inflow to DeGray Lake. The 2°C variation observed during September 1979 in Fig. 20 caused the inflow to enter the epilimnion from mid-morning to early evening and enter the metalimnion during the remaining hours of the day. Although changes of this nature do not significantly impact the thermal structure of the pool, they must be considered when interpreting the results of a field or modeling study since they do affect where inflowing nutrients and other materials are transported in the reservoir.

Summary

The temporal variations in thermal structure at DeGray Lake can be characterized at three levels: annual, synoptic, and diurnal. On an annual basis, overturn occurs in late January or early February with the onset of stratification in March. The minimum thermocline depth occurs near summer solstice. Surface temperatures peak nearly two months later,

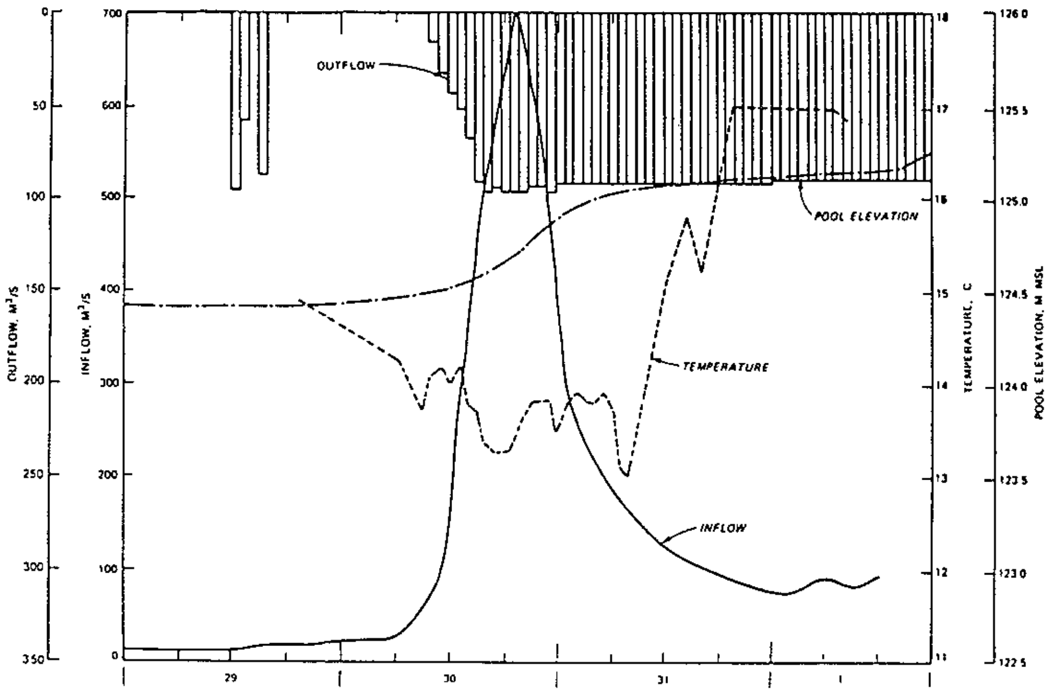

Fig. 17. Inflow rate and temperature, water surface (pool) elevation, and outflow during 29 March - 1 April 1979 storm event at DeGray Lake.

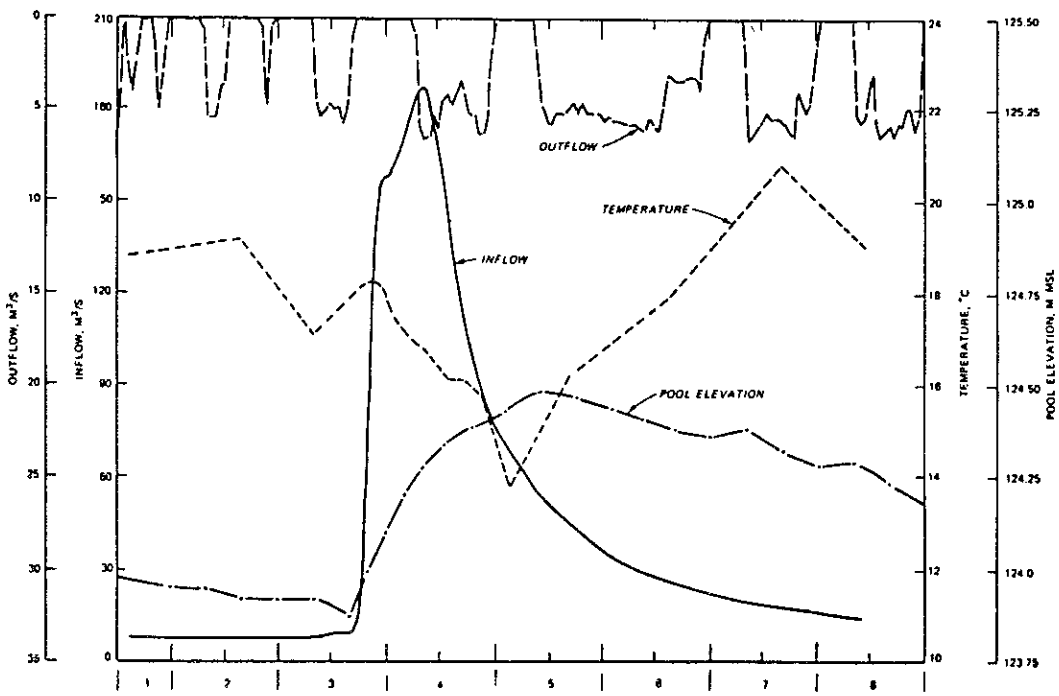

Fig. 18. Inflow rate and temperature, water surface (pool) elevation, and outflow during 1-8 May 1979 storm event at DeGray Lake.

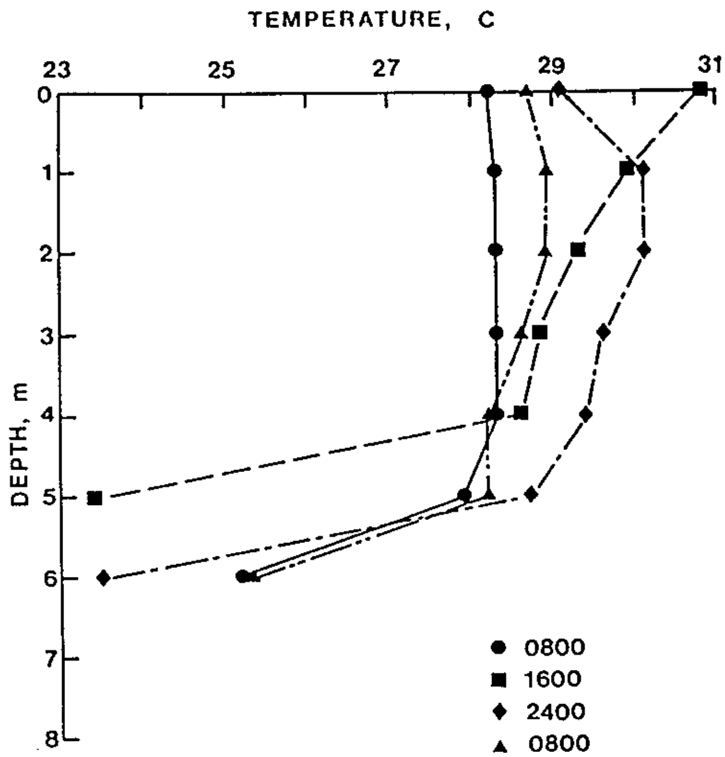

Fig. 19. Typical diurnal temperature variation within the pool during July at DeGray Lake.

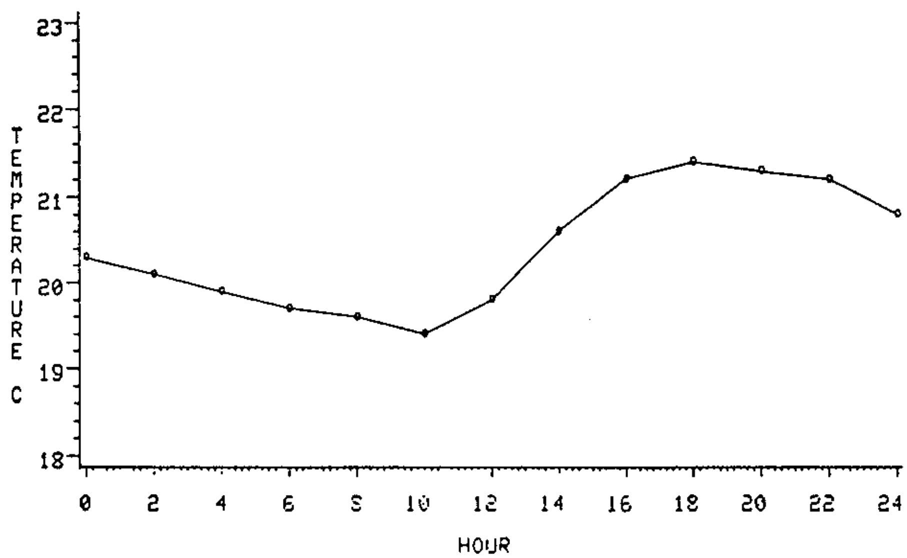

Fig. 20. Diurnal temperature variation in Caddo River inflow to DeGray Lake, 20 September 1979.

however. The hypolimnetic temperature is determined by temperatures during the previous winter and increases little during the year. Inflows and outflows from March through May are large and augment wind mixing in defining the thermal structure of the reservoir.

Synoptic events occur over several days but their cumulative effects determine the thermal structure of the reservoir. Large inflows and outflows occur in March through May as a result of synoptic events and increase mixing within the pool.

Diurnal variations are strongly influenced by solar radiation. During the summer, diurnal variations generally result in a net gain in heat and during the fall, a net loss. Even though diurnal temperature variations within the pool and inflow are small they are, nontheless, important to consider when interpreting field data or modeling results.

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__________ and M. C. JOHNSON. 1983. An assessment of reservoir density currents and inflow processes. Technical Report E-83-7. USAE Waterways Experiment Station, Vicksburg, MS.

Mixing Processes in DeGray Lake, Arkansas[1]

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Abstract

Mixing processes are defined as any mechanism that causes a parcel of water to blend with or be diluted by another parcel of water. Understanding and quantifying these mixing processes are essential to understanding the observed water quality of DeGray Lake since they control the movement and dilution of many water quality constituents. Specific mixing processes include advection, convection, diffusion, dispersion, entrainment, and shear. All of these processes require energy from meteorological forcing, inflows, outflows, and/or project operation, which includes the effects of hydropower operation. The relative magnitude and importance of the individual processes depend on the time and space scales or characterisics of the forcing functions, the degree of thermal stratification, and the physical location in the reservoir. In this paper, each of the mixing mechanisms is described and quantified with respect to season (i.e., spring, summer, fall, winter) and location (i.e., headwater, cove, main body near dam, and epilimnion, metalimnion, and hypolimnion). The relative significance of each process to observed water quality is also discussed.

Mixing is defined as the integrated effect of all processes that cause a parcel of water to blend with or be diluted by another parcel of water. Understanding mixing processes is essential to understanding

[1]The data used in this paper were collected by Dr. Joe F. Nix and his staff, Ouachita Baptist University, as part of the Environmental and Water Quality Operational Studies which was sponsored by the Office, Chief of Engineers, US Army Corps of Engineers, Washington, DC. The data analysis and paper preparation were supported by FTN, Ltd.

reservoir water quality because mixing and transport determine the ultimate fate of all particulate and dissolved constituents in a reservoir. Without mixing, materials would not be transported or diluted. Materials generated at a specific location would remain at that location and would significantly alter many of the horizontal and vertical patterns observed in reservoirs.

Mixing, like other time dependent, multifaceted processes, is difficult and expensive to quantify in the field. In general, velocities are at or near the threshold limit of most current meters. State-of-the-art equipment to measure turbulence is designed for use under laboratory not field conditions. In addition, since these measurements are point measurements, numerous points or instruments are required to spatially define the mixing regime throughout the reservoir. Perhaps the best method to quantify mixing in the field is to use fluorescent dyes. The interpretation of these measurements, however, is also limited to a specific time and location in the reservoir. Because of the difficulties and costs associated with studying mixing in the field, knowledge about reservoir mixing processes is based primarily on results from laboratory experiments, computer model simulations, and the interpretation of temporal and spatial variations in passive water quality constituents such as temperature and solids. This was also true for DeGray Lake. Although both current measurements and dye studies were undertaken in DeGray Lake, many of the observations and conclusions presented in this paper are based on the physical interpretation and integration of laboratory studies with field data (both long-term monitoring and short-term intensive studies) and computer simulation results.

The objective of this paper is to characterize the mixing regime of DeGray Lake and to describe the importance of various mixing processes on the observed water quality. First, a general description of reservoir mixing is presented. Then the reservoir is spatially divided into five compartments based on water quality considerations (i.e., headwaters, epilimnion, metalimnion, hypolimnion, and coves) and the mixing regime within each compartment and between compartments is discussed. Finally the potential impact of pumped storage operations on the DeGray Lake mixing and water quality regimes is discussed.

General Description

As shown in Fig. 1, there are a number of different physical processes that contribute to the mixing regime of DeGray Lake. These processes include advection, convection, diffusion, dispersion, entrainment, and shear.

Advection is transport by an imposed current system. In DeGray Lake, material is advected by inflows, outflows, pumpback currents, and wind drift.

In contrast, convection is vertical transport induced by density instabilities. As surface waters cool, they become more dense than the underlying water, sink, and create convective currents. In hydrodynamics and aerodynamics, the term convection is sometimes used interchangeably with advection, but not in this paper.

Diffusion is the spreading of a constituent by random molecular motions (i.e., molecular diffusion) or by turbulent motion (turbulent diffusion). The flux or transport per unit area by diffusion is described by Fick's law (i.e., flux = diffusion coefficent times a concentration gradient). Differences in mean concentrations or concentration gradients are, therefore, always reduced or attenuated by diffusive processes.

Entrainment is a one-way advective type process which sharpens gradients. A well-mixed, turbulent layer advances into a quiescent layer by entraining the quiescent water.

All of these processes require a source of energy. Water flows downhill (i.e., is advected) because potential energy (PE = mgh) is being converted into kinetic energy (KE = $1/2\ mv^2$). In reservoirs, inflows, outflows, wind, and heat transfer across the air-water interface are sources of KE and/or PE. In addition to mean KE, turbulence or turbulent kinetic energy (TKE) is also generated by these energy sources. Turbulent flows are irregular (random), diffusive (produce mixing), rotational (overturning motions), and dissipative (require a continuous source of energy).

Since the sources of energy for mixing are time-varying, mixing is also time-varying. The mean monthly TKE input rates for the wind and inflow for DeGray Lake are compared in Fig. 2. With the exception of May, the wind contributes more kinetic energy to DeGray Lake than the

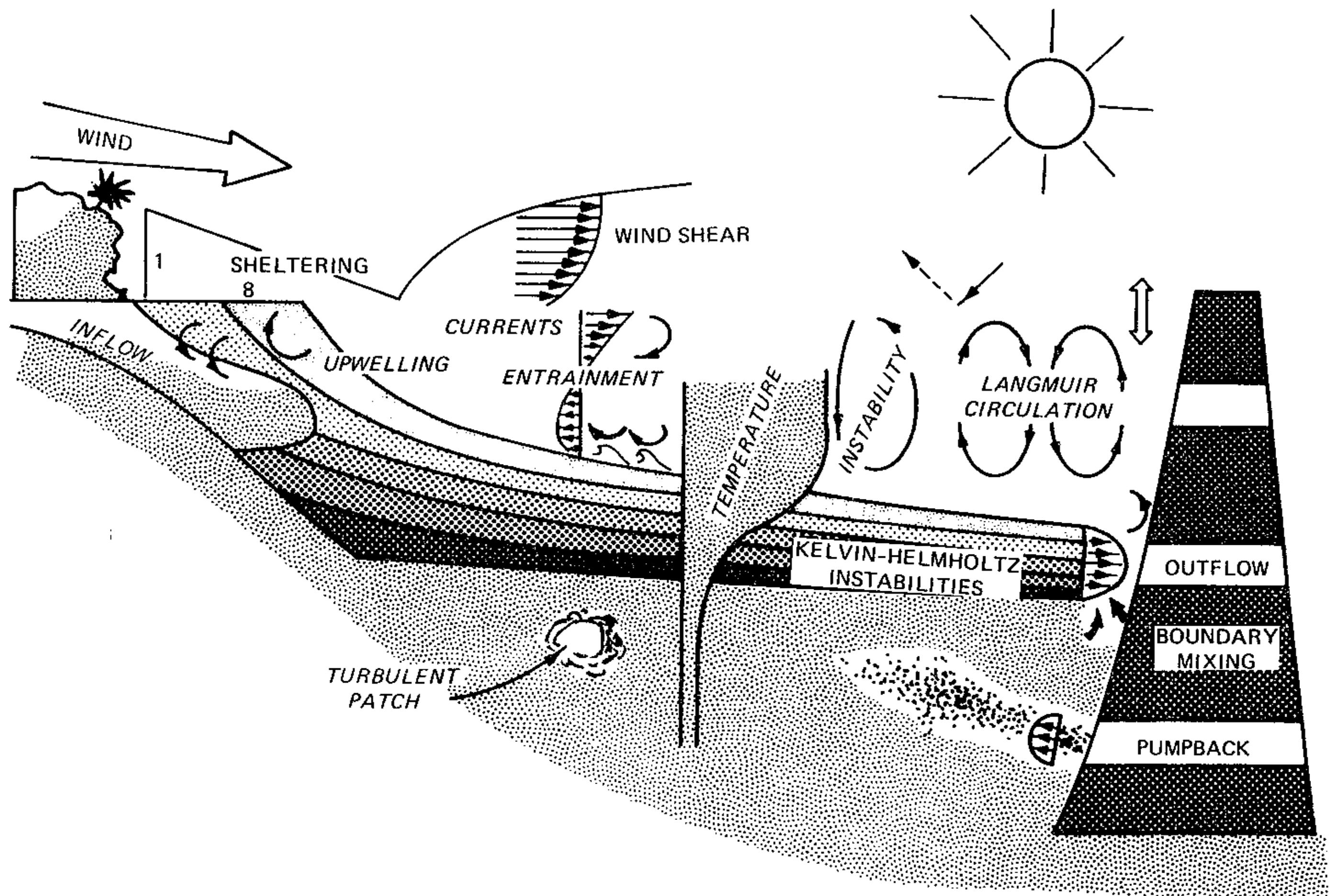


Fig. 1. Pictorial representation of reservoir mixing processes.

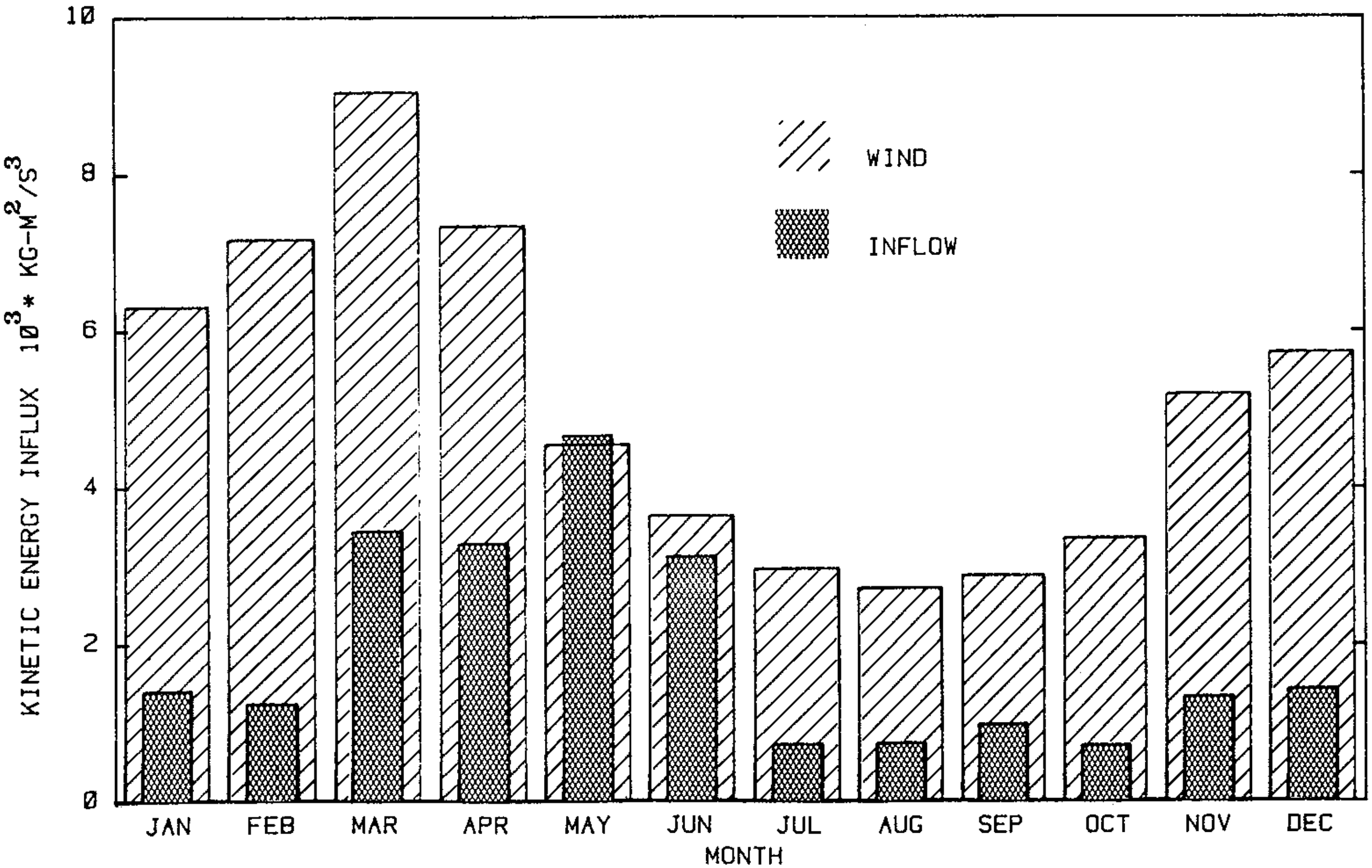


Fig. 2. Comparison of monthly energy inputs from the wind and inflow.

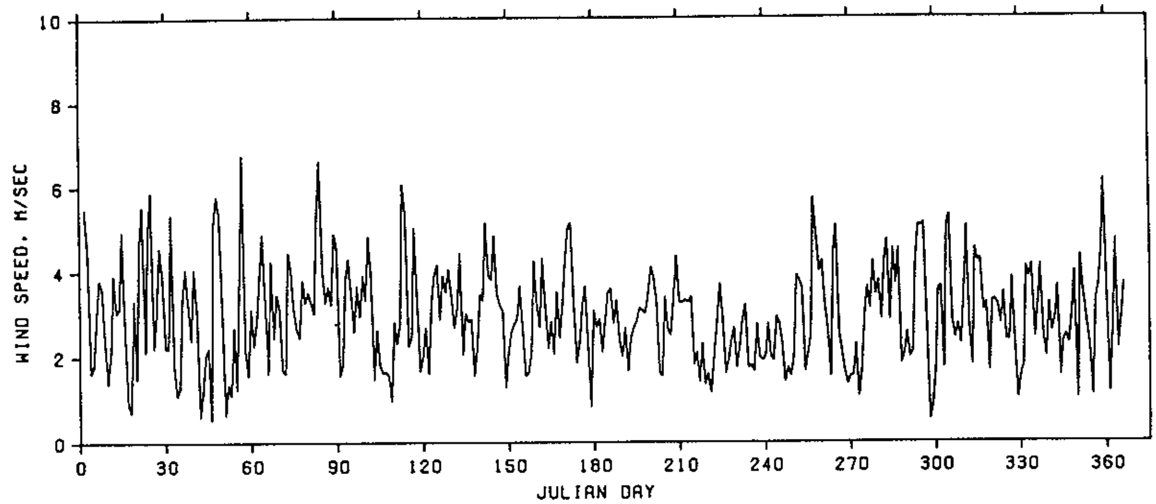
inflows. There is also a definite seasonal trend. The period of maximum TKE input (March - May) coincides with the onset of stratification while the period of minimal TKE input is characterized by strong stratification. The interaction of mixing with the stratification cycle in DeGray Lake is described in detail by Johnson and Ford (1983).

The actual input of TKE to DeGray Lake is more dynamic than indicated in Fig. 2. For example, Fig. 3 illustrates daily variations in wind speed and inflow. Since the TKE input is proportional to the third power of the wind speed and flow rate (See Johnson and Ford, 1983), the daily input of TKE is even more dynamic than the mean monthly values indicated in Fig. 2. For comparative purposes, the TKE input from a 9.8 m/s wind (i.e., 22 mph) on DeGray Lake is equivalent to an inflow rate of 370 m^3/s (~ 13,000 cfs). Because the TKE input rate is proportional to the third power of the wind and flow rates, TKE inputs are not linear and additive. The TKE input from a 6 m/s wind blowing for 4 hours is 36 times greater than the TKE input from a 1 m/s wind blowing for 24 hours. This analogy also applies to outflows at hydropower projects like DeGray Lake because power generation is limited to a few hours and not averaged over 24 hours.

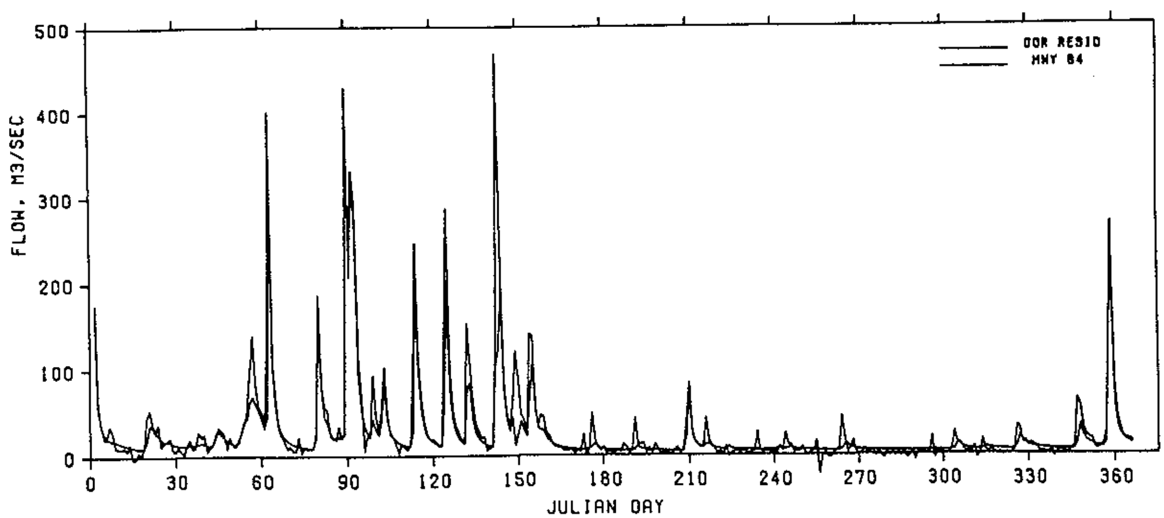
Inflow Mixing (Headwater Area)

DeGray Lake has one major tributary, the Caddo River, which is the major source of water, nutrients, and other quality constituents. When the Caddo River first enters DeGray Lake, backwater effects cause velocities to decrease and depths to increase. If there are no density differences between the Caddo River inflow and DeGray Lake surface waters, the point of maximum velocity remains near the water surface and the inflow moves through the lake as plug flow in an open channel. Concentrations of inflowing constituents decrease due to diffusion and dispersion. If there are density differences between the inflow and reservoir waters, the inflow will push the reservoir water ahead until the buoyancy forces dominate over the advective forces and the inflow either plunges beneath or flows over the reservoir surface waters.

The location of this plunge point (or overflow point) can be used to separate the headwater region from the pelagic region of DeGray Lake. For a mean annual inflow of 18.2 m^3/s and relative density difference



a. Wind



b. Inflow

Fig. 3. Comparison of daily wind and inflows, DeGray Lake, 1979.

($\Delta\rho/\rho$) of 5×10^{-4}, the average location of the plunge point in DeGray Lake is near Station 14 (i.e., 4.5 km into the lake) (Fig. 4). Under low flow conditions, the plunge point will move upstream 3 or 4 km to where the water depths are 1 m or less. During extreme storm events, the plunge point will move downstream into the lake past Station 10. For typical storms, the plunge point is near Station 12.

Upstream of the plunge point, the mixing regime is riverine-like and dominated by advection and dispersion. Velocity gradients in the lateral and vertical directions cause materials located in the center of the stream near the surface to move faster than at the boundaries. Dead zones along the boundaries trap materials and release them at a slower rate causing the inflowing constituent distribution to be skewed. Dye studies in the upstream region, under steady flow conditions, resulted in greater than 10-fold dilutions due solely to dispersion. Similar dilutions can be expected for all water quality constituents even under unsteady storm conditions.

Ford and Johnson (1983) showed that the inflow water and its associated constituents pool at the plunge point. As inflows increase during storm events, this pool of constituents is pushed further into the lake. When the inflows recede, the plunge point retreats back upstream and the constituents that load on the leading edge of the hydrograph remain behind on the water surface. These constituents are then available for biochemical processing in the euphotic zone of the lake. This mixing mechanism may be the major cause of the high constituent concentrations found between Stations 14 and 10 in DeGray Lake (Thornton et al. 1982). Mixing also is associated with the plunge point. Ford and Johnson (1983) estimated mixing coefficients on the order of 12 percent for six storm events on DeGray Lake. The range varied from 4 to 28 percent.

Epilimnion Mixing

The epilimnion of a lake is usually defined as the upper strata of well-mixed, turbulent water. The turbulent or vertically well-mixed epilimnion is maintained by the wind and convective cooling. These processes transport energy directly across the air-water interface. In contrast, the energy input from the Caddo River inflow is concentrated at the upstream end of DeGray Lake (i.e., the headwaters) and the out-

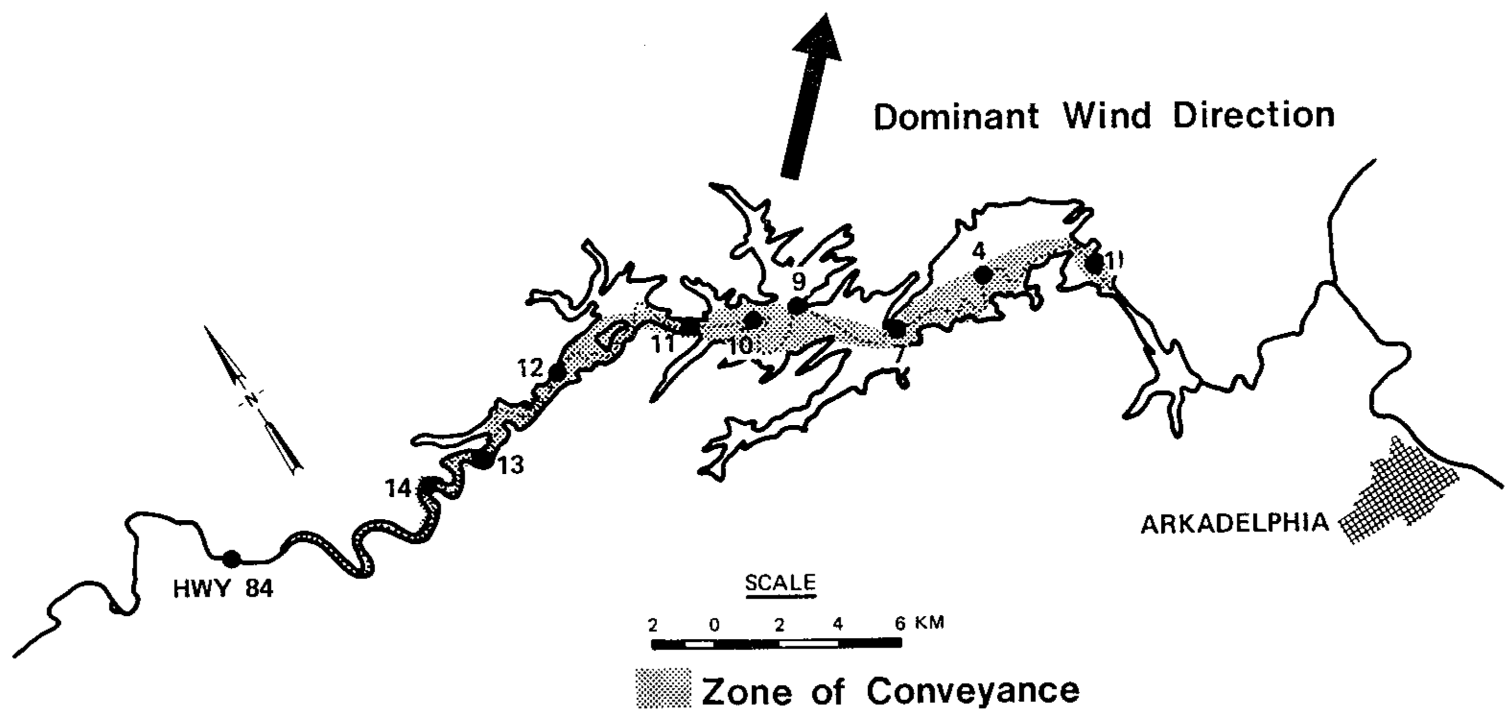


Fig. 4. DeGray Lake, Arkansas.

flow energy input is concentrated near the dam. Because of the dynamic force balance at the plunge point, only a small fraction of the inflow energy usually impacts the epilimnion. During the spring months, however, when inflow temperatures exceed lake temperatures the majority of the inflow energy enters the epilimnion. Energy from the outflows is concentrated in the epilimnion only when the top or surface outlet is used at DeGray Lake (i.e., surface withdrawal). DeGray Lake was operated with surface withdrawal until March 1979 when the withdrawal level was lowered to evaluate the impact of withdrawal depth on both in-lake and downstream water quality and fisheries. The withdrawal level was returned to the surface outlet in March 1983.

When the wind blows across the water surface, the resulting drag force generates surface waves, drift currents, and TKE. The magnitude of the drag force will depend on the wind strength, atmospheric stability, variability of wind speed, length of fetch, and degree of wave development. Since only limited wind speed data is available for DeGray Lake, data from the National Weather Service station at Little Rock, AR, was used. Although differences between the two wind regimes exist, the relative proximity of Little Rock to DeGray Lake and similarities in the geographic locations of the two sites allow for generalities to be made concerning the importance of wind to the mixing regime in DeGray Lake. The representativeness of Little Rock wind speeds for DeGray Lake has also been verified by mathematical simulations (see Johnson and Ford 1983).

The maximum and minimum monthly wind speeds of 4.8 m/s and 3.0 m/s occur in March and June, respectively. As implied in Fig. 2, there is a definite seasonal pattern with higher wind speeds during the winter and spring periods of complete mixing and lower wind speeds occurring during late summer which is characterized by strong stratification. The frequency distribution for mean daily wind speed at Little Rock has a median wind speed of 3.3 m/s and a range from calm to 10 m/s. In general, wind speeds peak in mid-afternoon during all months. The dominant wind direction is southwest (Fig. 4), which is perpendicular to the major axis of DeGray Lake. Since DeGray Lake is also surrounded by high hills and is dendritic in shape, sheltering of the water surface from the wind by the surrounding terrain should be important. The resulting drag force from the wind, therefore, varies temporally and spatially causing both temporal and spatial variations in epilimnetic mixing. Despite these complicating factors, the importance of the wind for mixing in DeGray Lake has been substantiated by mathematical thermal simulations (Johnson and Ford 1983).

Relatively little is known about the partitioning of wind energy into the PE of surface waves, KE of drift currents, and TKE. It is known that the fraction associated with the surface waves is radiated away as potential energy and dissipated along the lake boundaries where surface waves break. If the waves break in open water, turbulence (TKE) is generated that can be used for vertical mixing. The general wave climate thereby acts as an indicator of mixing. Since the drift currents generated by the wind on DeGray Lake are small (i.e., order of 3 to 4 cm/s) and the velocity gradient dissipates quickly with depth, the significance of these currents with respect to mixing is probably minor. Even through the fraction of wind energy available for mixing is small, it is sufficient to maintain a well-mixed epilimnion and entrain metalimnetic water.

Because of the nature of the wind input, two facts are known about epilimnetic mixing. First, the entrainment of metalimnetic water into the epilimnion is not a constant continuous process as implied by smoothed isotherm plots but rather an intermittent process with entrainment coinciding with peaks in wind speed. Therefore, the epilimnion and euphotic zones are intermittently pulsed with nutrients from the metalimnion. Second, wind mixing is not uniform across the water body because of spatial variations in both the wind regime and wind-associated phenomena such as seiching and upwelling. Nutrient pulses, then, would also be expected to exhibit similar spatial variation.

In addition to the wind, convective mixing also acts to maintain a vertically well-mixed epilimnion and entrain metalimnetic waters. Convective mixing is important during the fall months when water surface temperatures are greater than air temperatures. Convective mixing can either be penetrative or nonpenetrative (Fig. 5). In penetrative mixing, a fraction (1 to 4 percent) of the TKE generated by the convective currents is used to entrain metalimnetic water. Because it is difficult to separate the effects of penetrative convection from wind mixing, the relative importance of penetrative convection is unknown at this time.

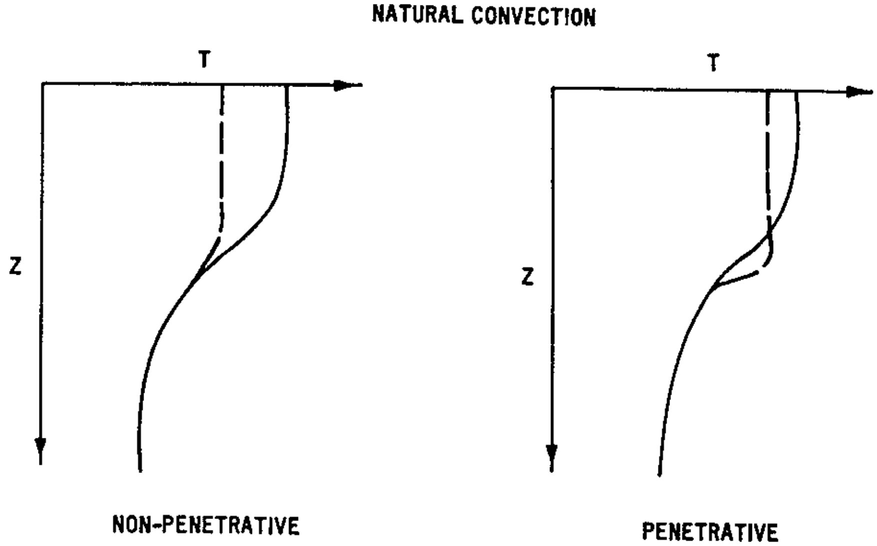


Fig. 5. Comparison of penetrative and non-penetrative convection.

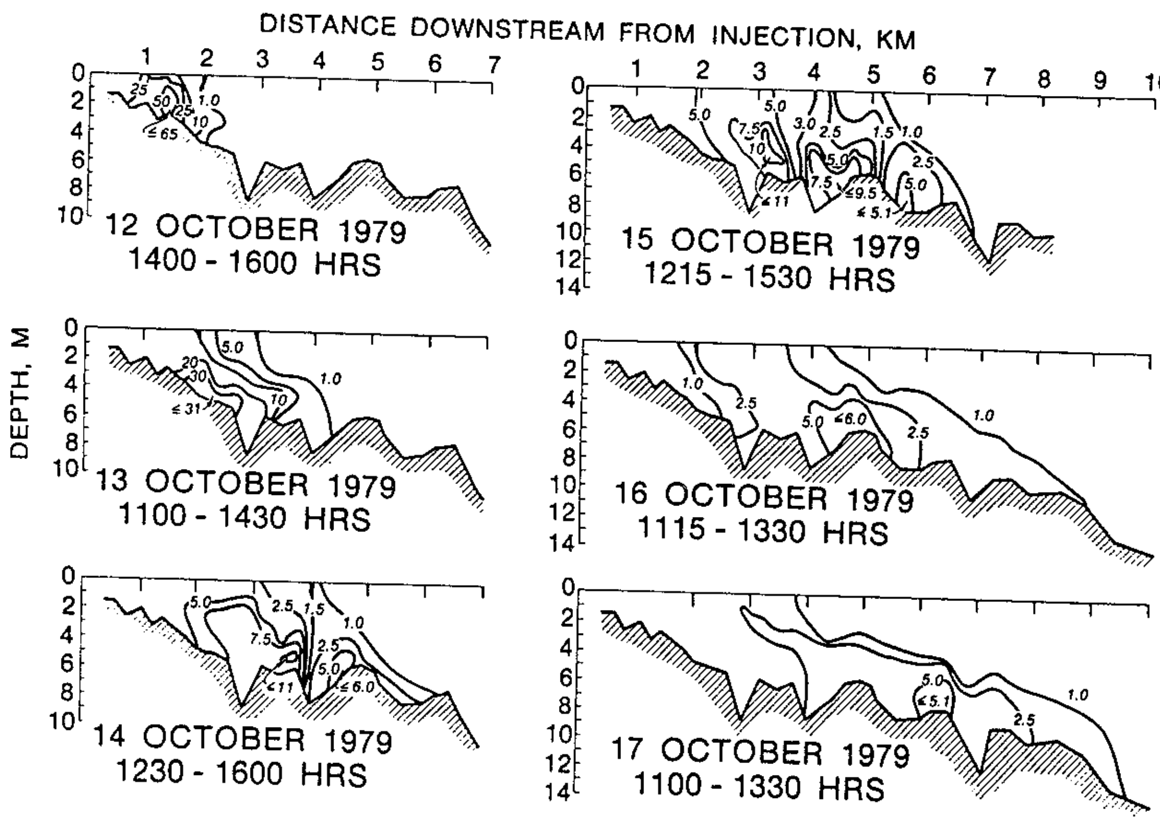


Fig. 6. Effect of convective mixing on dye plume in DeGray Lake.

Convective mixing can, however, alter the concentrations of inflowing constituents by causing more dilution. This is shown in Fig. 6 where the dye cloud was divided into three distinct parts by convective mixing (see Ford and Johnson 1983 for details).

Metalimnetic Mixing

The metalimnion is the transition zone between the epilimnion and hypolimnion. It is usually characterized by a strong density gradient that inhibits mixing. In DeGray Lake, inflows, including storm inflows, enter the metalimnion via interflows, during most of the summer periods. The middle and bottom outlets are also located in the metalimnion. Therefore, much of the advective kinetic energy input is concentrated in the metalimnion during the summer stratification months, especially if the lower two outlets are used. In fact, storm interflows have been observed to short-circuit through the metalimnion of DeGray Lake in seven days. This storm flow and associated metalimnetic mixing had a pronounced influence on the temperature gradient in the metalimnion (Fig. 7).

Kinetic energy from the wind radiates through the metalimnion in the form of internal waves. If these waves should break or become unstable (i.e., Kelvin-Helmholtz instability), a patch of turbulence is generated (Fig. 1). Since the standing timber was topped at elevation 114.3 m (ca 10 m below the water surface), the timber extends into the metalimnion creating obstacles for these internal waves to break against. Once a patch of turbulence is formed, it quickly collapses and spreads horizontally due to the strong density gradients. This type of mixing helps to maintain horizontal uniformity in the metalimnion but does not result in much vertical mixing.

Hypolimnetic Mixing

Although no *in situ* measurements of hypolimnetic mixing were made at DeGray Lake, the constant hypolimnetic temperatures and flatness of the observed isotherms (see Johnson and Ford 1983) indicate hypolimnetic mixing was near molecular levels most of the time. The coefficients used in the thermal modeling studies and the resulting eddy diffusivities from these coefficients also confirm this low level of

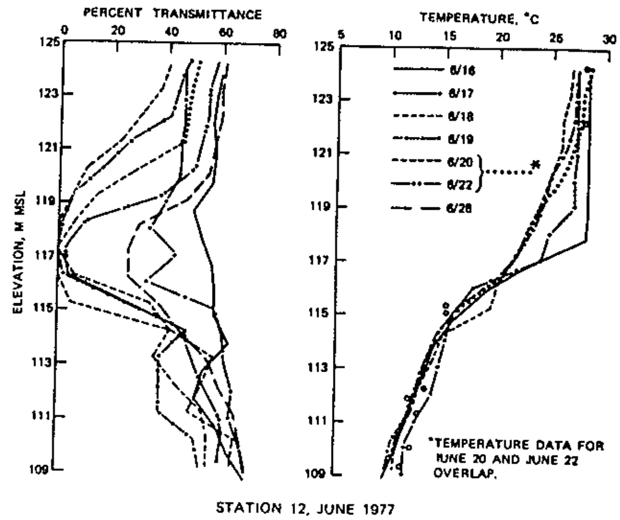


Fig. 7. Changes in metalimnetic temperatures during a storm event.

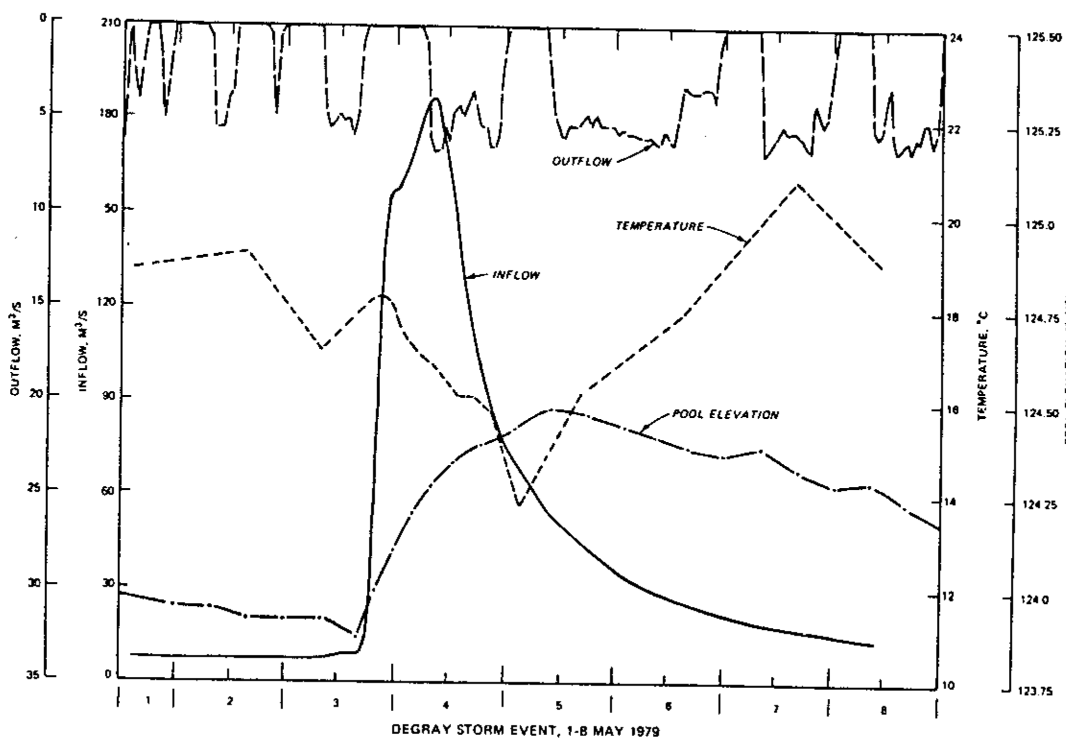


Fig. 8. Pool level changes during a storm event.

mixing. In addition, underwater video pictures in the hypolimnion showed a very fine floc on the surfaces of branches and pine needles that was easily disturbed by any minor movement of water.

This conclusion is not surprising, however, since there is no direct source of TKE to the hypolimnion for mixing. The hypolimnion is protected from TKE generated at the water surface (e.g., by the wind shear) by the density gradient in the metalimnion. Part of this TKE, however, indirectly enters the hypolimnion via seiche motion and internal waves that propagate through areas of density gradients. Mixing occurs when these internal waves break (i.e., Kelvin-Helmholtz instabilites) or when the waves encounter an obstacle boundary and break like surface waves breaking on a beach. Since DeGray Lake was not completely cleared of trees and the trees were topped at an elevation within the metalimnion, the trees provide many obstacles on which waves can break. It is unlikely that the waves will propagate far before encountering an obstacle (i.e., possibly not out of the metalimnion). In general, this this type of mixing is intermittent and patchy. There is insufficient energy available in most lakes to maintain a turbulent hypolimnion (Imberger and Hamblin 1982).

TKE from inflows and outflows is also not directly available to the DeGray Lake hypolimnion. Analysis of the inflow densities, even during storm events, indicates that inflows enter the mid to upper metalimnion and not the hypolimnion. Withdrawals are also limited to the epilimnion and metalimnion since the lowest outlet was in the lower metalimnion. There are no bottom flood gates available to flush the hypolimnion.

Even though the average hypolimnion mixing level in DeGray Lake is near molecular, intermittent periods of intensive mixing will occur. These periods probably will be associated with storm events. During these periods, exhanges of material across the sediment-water interface will be greatly enhanced because of the higher level of turbulence. In contrast, during the quiescent periods, exhanges across the sediment-water interface will be governed by biochemical processes not physical diffusion.

Horizontal Mixing

DeGray Lake, like many other reservoirs, is dendritic in shape with many coves and embayments (Fig. 4). Several of these coves are fed by small or intermittent tributaries. These coves are relatively isolated from the pelagic regions of the lake and may be characterized by different water quality. Dye studies and turbidity tracking during storm events showed that the inflows follow the thalweg of the river and do not mix completely across the lake (Ford and Johnson 1983). The question is how significant is horizontal mixing?

There are three basic horizontal transport mechanisms - varying pool levels, thermal instabilities, and horizontal dispersion. Although DeGray Lake is operated to maintain constant pool levels, seasonal and short-term fluctuations in response to storm events do occur. The actual mechanics of transport will vary with the ambient stratification and the inflow placement or withdrawal depth. Seasonal differences of 1 m or more may result from outflows exceeding inflows during dry months. In general, during the dry months of July through October, water levels fall and there is a net transport out of the coves into the pelagic region of the lake. Both hypolimnetic and epilimnetic water can be transported. Algae or nutrients in coves, therefore, can be transported into the pelagic regions and possibly create water quality problems.

Short-term fluctuation in water levels of 0.5 m or more may result during storm events (Fig. 8). During these events, the high-nutrient flood waters may flow into the coves and be stored. When the flood water is released to return the pool to normal levels, the material processed in the coves during the storage period may move out of the coves into the pelagic area.

During periods of cooling (fall), the shallow water in the coves cools faster than the deeper water (Fig. 9). The resulting temperature differentials create density instabilities and convective currents that can transport material from the coves into the metalimnion of the lake and from the pelagic surface water back into the coves.

The third mechanism, horizontal dispersion, occurs continuously and its magnitude depends on the turbulence and shear generated by the wind, inflows, and outflows. Empirical formulae for horizontal dispersion have been developed by Murthy (1976) and others. In general, horizontal

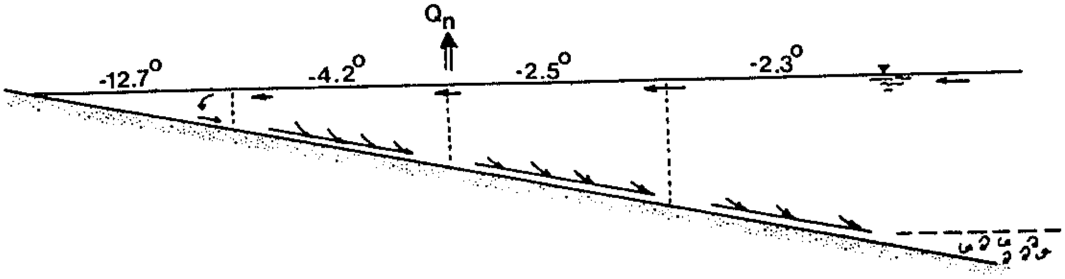


Fig. 9. Thermal currents generated during periods of cooling.

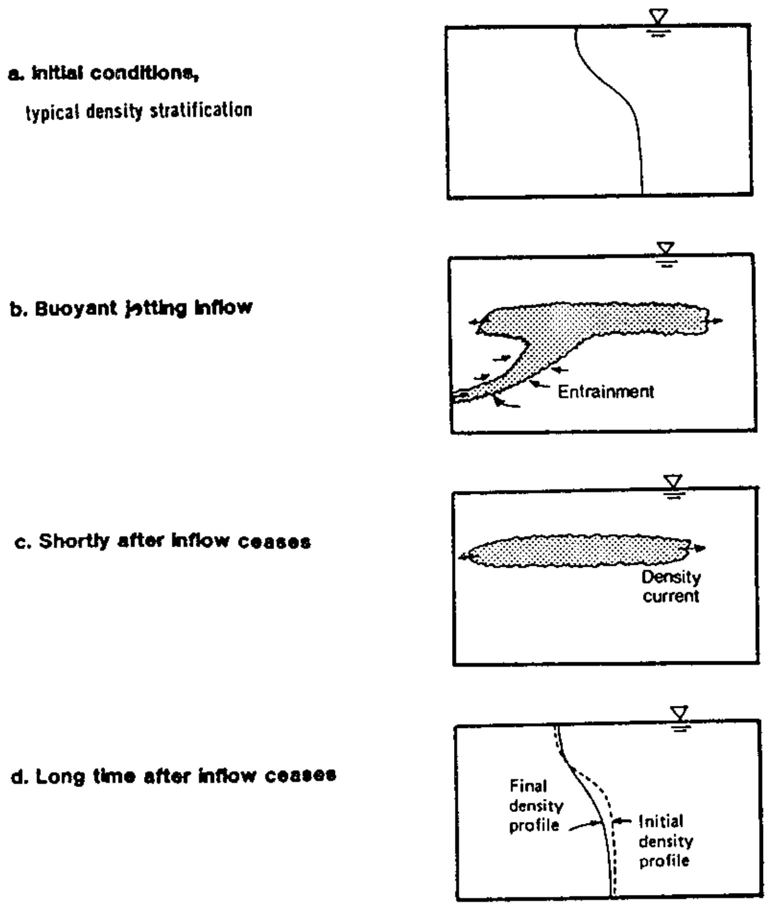


Fig. 10. Schematic of a pumpback jet in a reservoir.

dispersion is an insignificant mixing mechanism. The length scale characteristic of horizontal dispersion in one day is only about 150 m.

Pumped Storage Operations

One operational alternative at DeGray Lake that has not been evaluated but could significantly alter the mixing regime and reservoir water quality is pumping water from the downstream re-regulation pool into the reservoir during periods of low power demand to be used to generate power during periods of high demand. Water is pumped back through a 28,000-kw reversible turbine to any of the three outlet levels. This water can therefore enter the epiliminon or metalimnion depending on gate settings. This capability has been used sparingly at DeGray.

The pumped water can enter the reservoir as a buoyant jet entraining ambient reservoir water (Fig. 10). Although no entrainment measurements were made at DeGray Lake, entrainment rates from other projects are on the order of 100 to 300 percent. The jet will rise until it reaches a level of neutral buoyancy and then spread horizontally as a density current. Depending on the depth that the jet enters the reservoir and amount of entrainment, pumpback operations can significantly alter the thermal and water quality regime of a reservoir.

Conclusions

1. During all months except May, the wind supplied more TKE for mixing DeGray Lake than the Caddo River inflow. The period of maximum TKE input occurs during the spring months of March through May. The period of minimum TKE input occurs during the summer months of July through September.

2. In the headwater region upstream of Station 14, the mixing regime is riverine in nature dominated by advection and dispersion. Inflowing nutrient concentrations can be reduced by a factor of 10 or more due to dispersion.

3. Mixing coefficents at the plunge point averaged 12 percent with a range of 4 to 28 percent. Inflowing constituents also tend to pool at the plunge point. During storm events, the plunge points moves downstream into the reservoir with increasing flow and recedes back upstream

with decreasing flow. Materials pooled around the plunge point or associated with the rising side of the hydrograph may be left behind in the photic zone as the plunge point recedes back upstream.

4. The major source of energy for mixing the epilimnion is the wind. Because of temporal and spatial variations in the wind regime, vertical mixing in the epilimnion will also vary temporally and spatially. Entrainment of metalimnetic water into the epilimnion is an intermittent, sporadic process.

5. Hypolimnetic mixing typically is near molecular levels during most of the year. The major source of energy for turbulent hypolimnetic mixing is also the wind since density underflows do not enter the hypolimnion and the lower outlet in DeGray is located in the metalimnion.

6. Many of the coves and embayments are effectively isolated from the main body of the lake. Inflows including storm events tend to short circuit through the lake following the old river channel and do not mix laterally. The major transport mechanism between the coves and main body appears to result from changes in water levels. There is a net transport out of the coves during the dry summer months as pool levels fall and a net transport into the coves during the fall and winter months as the pool is raised to normal levels. During storm events, material can still be exchanged if pool levels fluctuate but the process is complicated by the ambient stratification, inflow placement, and withdrawal level.

7. If the pumped storage capabilities at DeGray are used extensively, significant changes in the mixing and water quality regime will probably occur.

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DISSOLVED OXYGEN STATUS OF DEGRAY LAKE, ARKANSAS

by

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Introduction

Dissolved oxygen concentrations in lake waters provide a valuable index to water quality conditions and a means for assessing the relative importance of autotrophic and heterotrophic activities. This is especially true for stratified lakes. Since hypolimnetic waters are isolated well below the air-water interface by density differences, these strata contain a finite supply of oxygen during summer months when heterotrophic demands are highest. As oxidizable organic materials are transported to the hypolimnion due to the settling of algae and/or the deposition of organic particulates introduced by tributary streams, demands are placed on oxygen stores. This results in declines in hypolimnetic dissolved oxygen concentrations. When organic inputs are excessive, all or part of the hypolimnion may exhibit anoxic conditions prompting the solubilization of selected sediment constituents and their accumulation in the water column. Additional demands are placed on oxygen stores as a result of chemical oxidation.

Relations between autotrophic activities in the epilimnion and heterotrophic activities in the hypolimnion, as modified by morphological characteristics, have long been used as a means to assess trophic relationships in lakes (see Hutchinson, 1957). These relationships become less well defined in reservoirs, however, due to differences in flow, operation, and construction practices. Density currents, resulting from density differences between river and lake waters, often result in the introduction of oxygenated river water to the reservoir's hypolimnion, thus making oxygen budget analyses difficult. Similar analytical problems occur in reservoirs which discharge water from the hypolimnion. During the years immediately following impoundment, the presence terrestrial organic material (i.e. timber and litter) may also potentially influence dissolved oxygen conditions.

The flooding of the Caddo River during impoundment of DeGray Lake resulted in the inundation of a pine and hardwood forest. Although areas immediately upstream from the dam, near-shore areas, and shallow areas near the inflow of the Caddo River were clear-cut, much of the lake's basin presently contains submerged standing timber. Depending on depth, tree tops may extend to within 10 m of the lake's surface. In addition to representing a significant source of oxidizable organic material, these trees also appear to influence the velocity and lateral dispersion of density currents (Ford, 1986).

A phenomenon commonly observed in newly flooded reservoirs is the occurrence of "trophic upsurge" (Grimard and Jones, 1982). The release of nutrients and organic material from terrestrial detritus and inundated soils and their accumulation in the water column increase nutrient levels and primary productivity. With these increases in organic matter production, hypolimnetic oxygen conditions are further impaired.

Studies conducted at DeGray Lake provided a unique opportunity to document changing conditions during and following impoundment. Since these studies spanned a period of approximately 12 years, the resulting data provide a means for assessing conditions during both the period of "trophic upsurge" and the period of improving conditions which followed (i.e., trophic depression). An understanding of dissolved oxygen dynamics is central to the understanding of other processes influencing water quality conditions in DeGray Lake.

Methods

As discussed by Kennedy and Carroll (1986), four primary sampling stations were established in DeGray Lake for the purpose of routine monitoring.

These included Station 1 and 4 (near-dam stations), Station 10 (mid-lake station), and Station 12 (headwater station). Although variable between years, data were collected at 2- to 4-week intervals using in-situ monitoring techniques. Profiles consisted of measurements at 1-m intervals in surface waters and at intervals ranging from 1 to 10 m at depths below the depth of the summer thermocline (approximately 10 m). Additionally, more intensive studies conducted during 1978-9 resulted in four sets of dissolved oxygen data collected at 1-m depth intervals at approximately 60 stations located throughout the entire lake basin. In-situ measurements of concentration were obtained using a YSI (Yellow Springs Instruments, Yellow Springs, Ohio) dissolved oxygen analyzer or a Hydrolab (Hydrolab Corp., Austin, Texas) dissolved oxygen analyzer.

Periods of anoxia at each station were defined as including all sample dates for which dissolved oxygen concentrations at one or more depths in the hypolimnion were below 0.05 mg/l. Times when dissolved oxygen concentrations were below 0.05 mg/l only in the metalimnion were not considered to be periods of anoxia. The extent to which the hypolimnion exhibited anoxic conditions was gauged by determining the minimum depth at which anoxic conditions were observed during the course of an anoxic period. In cases where anoxic conditions existed in the metalimnion and in the lower strata of the hypolimnion but not at intermediate depths, the depth of the top of the hypolimnetic anoxic zone was recorded.

Area and volumetric hypolimnetic oxygen deficit rates were calculated by comparing oxygen concentrations observed below the depth of the summer thermocline on the last sampling date when isothermal conditions were observed with those observed on the sampling date immediately preceding the establishment of anoxic conditions. Rates were determined by expressing rates of change in hypolimnetic oxygen content (gm/day) as a function of hypolimnetic area (areal deficit rate) or as a function of hypolimnetic volume (volumetric deficit rate). Since the rate at which oxygen is depleted was expected to vary considerably as a function of both depth and time, a second method for calculating volumetric rates was also employed. This involved calculating absolute differences in concentration at each depth during each successive sampling interval. Rates for each depth were expressed as mg/m3 per day. While this method allowed greater insight to the manner in which oxygen concentrations were reduced during each season and throughout the water column, care was taken to consider only those rates occurring at depths within the hypolimnion on each particular sampling date.

Results and Discussion

Pronounced differences in the duration and the timing of the onset of anoxia were apparent between stations and between years (Figure 1-3). Between-year differences were also observed in the minimum depth of anoxia at Station 1. In general, the duration of anoxia was shorter at Station 12, the most upstream station, than at stations further downstream. This observation was a direct consequence of morphologic differences and resultant differences in mixing regime. The shallow depth at this station resulted in shorter periods of thermal stratification than at other stations. While the timing of the onset of thermal stratification was similar for all stations (late April or early May), autumnal turnover began at progressively later dates at downstream stations. Thus, while turnover commonly occurred in late October or early November at Station 12, turnover at Station 1 was generally not complete until January. The consequences of these differences in thermal characteristics were most apparent in the early years (1970-75) when bottom waters at Station 1 remained anoxic for periods ranging from 165 to 225 days. During these same years, periods of anoxia at Station 12 ranged from 90 to 180 days in duration.

A pronounced discontinuity in the severity (i.e., duration and vertical extent) of anoxia was apparent at Station 1 in 1975-76. With the exception of 1973, anoxic conditions prior to 1975 were observed from the bottom to the depth of the thermocline. In subsequent years, anoxic conditions were observed only in lower (depth >42 m) strata of the hypolimnion. While no direct evidence exists to explain this difference, the oxidation of labile organic material (e.g., terrestrial detritus, soil organics and standing vegetation) during the early years could have reduced oxygen demands during the later years. The initial impact and eventual exhaustion of readily oxidizable organic material has been documented in laboratory studies using soil samples from the DeGray Lake basin (Gunnison and Brannon, 1981). This suggestion is further supported by the fact that laboratory simulations of several annual cycles of anoxia/reaeration indicated a similar change in severity in degree of anoxia. Consistent temporal patterns in anoxia at the more upstream

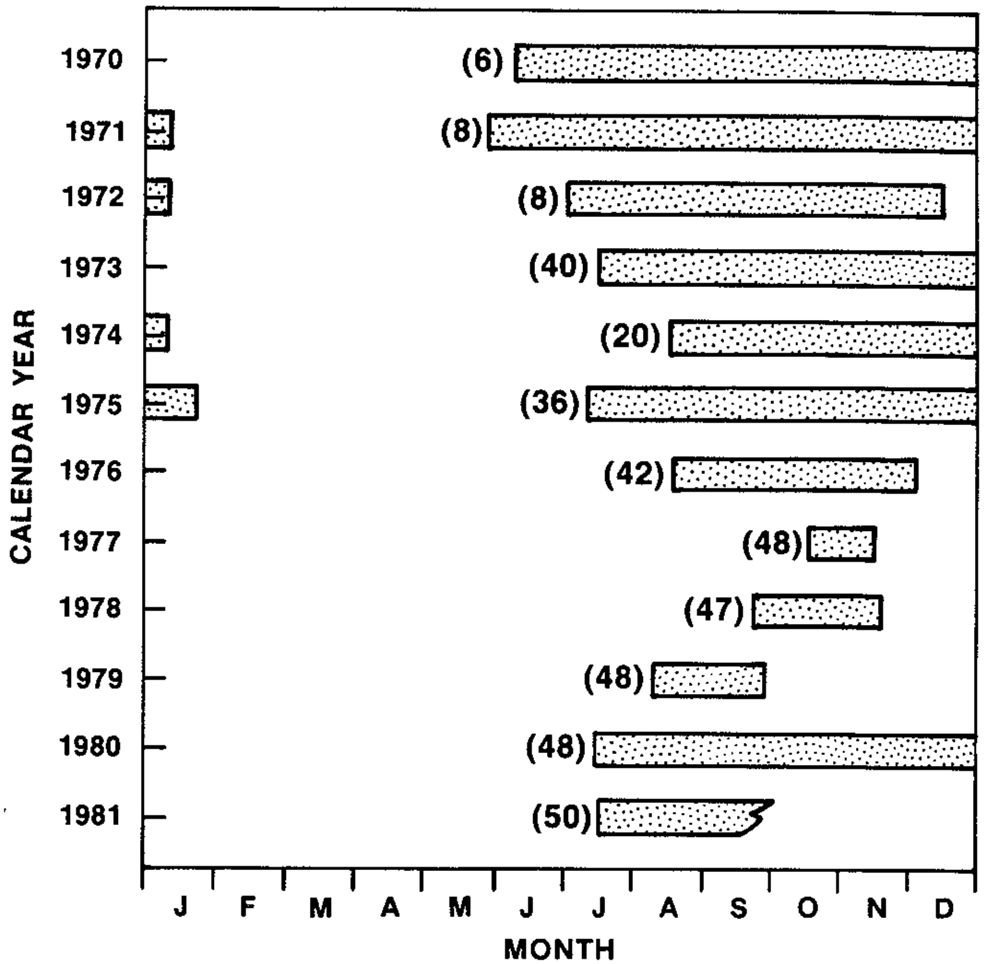

Figure 1. Changes in the duration and extent of hypolimnetic anoxia at Station 1 during the period 1970-1981. Shaded, horizontal bars indicate periods of anoxia. The depth (m) of the maximum upper extent of the anoxic zone is indicated in parentheses.

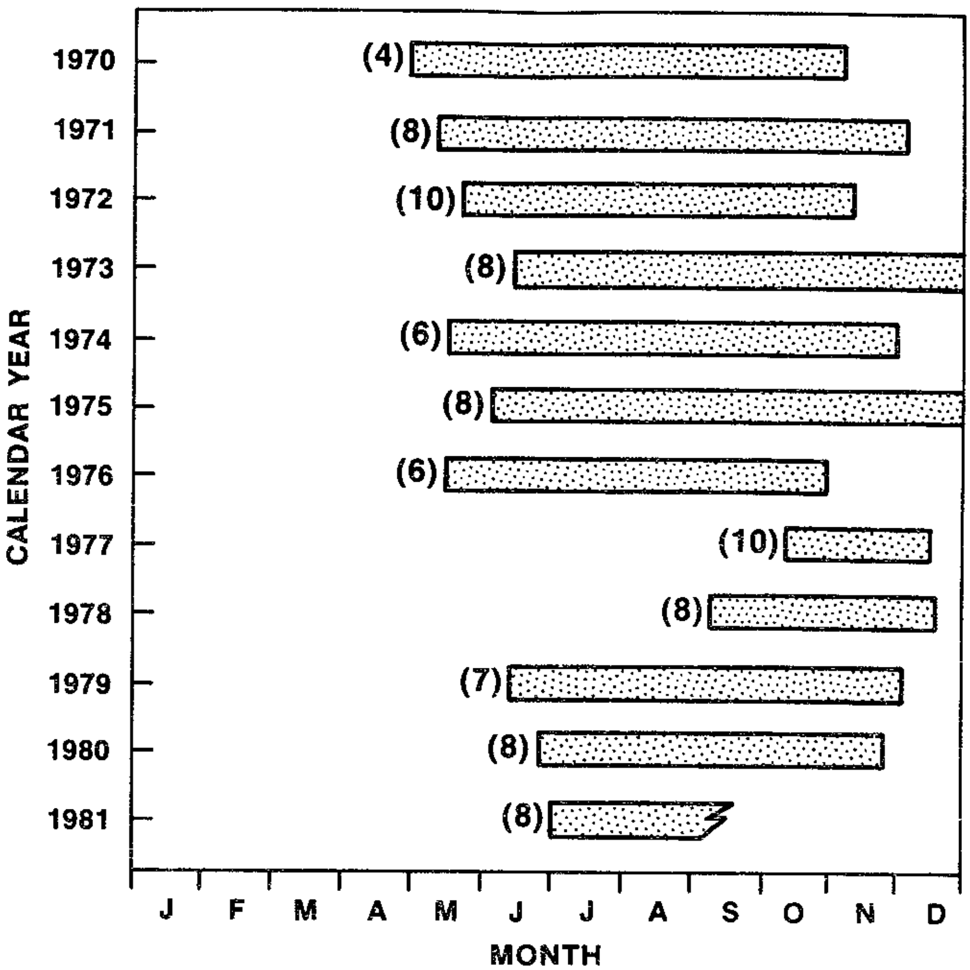

Figure 2. Changes in the duration and extent of hypolimnetic anoxia at Station 10 during the period 1970-1981. Shaded, horizontal bars indicate periods of anoxia. The depth (m) of the maximum upper extent of the anoxic zone is indicated in parentheses.

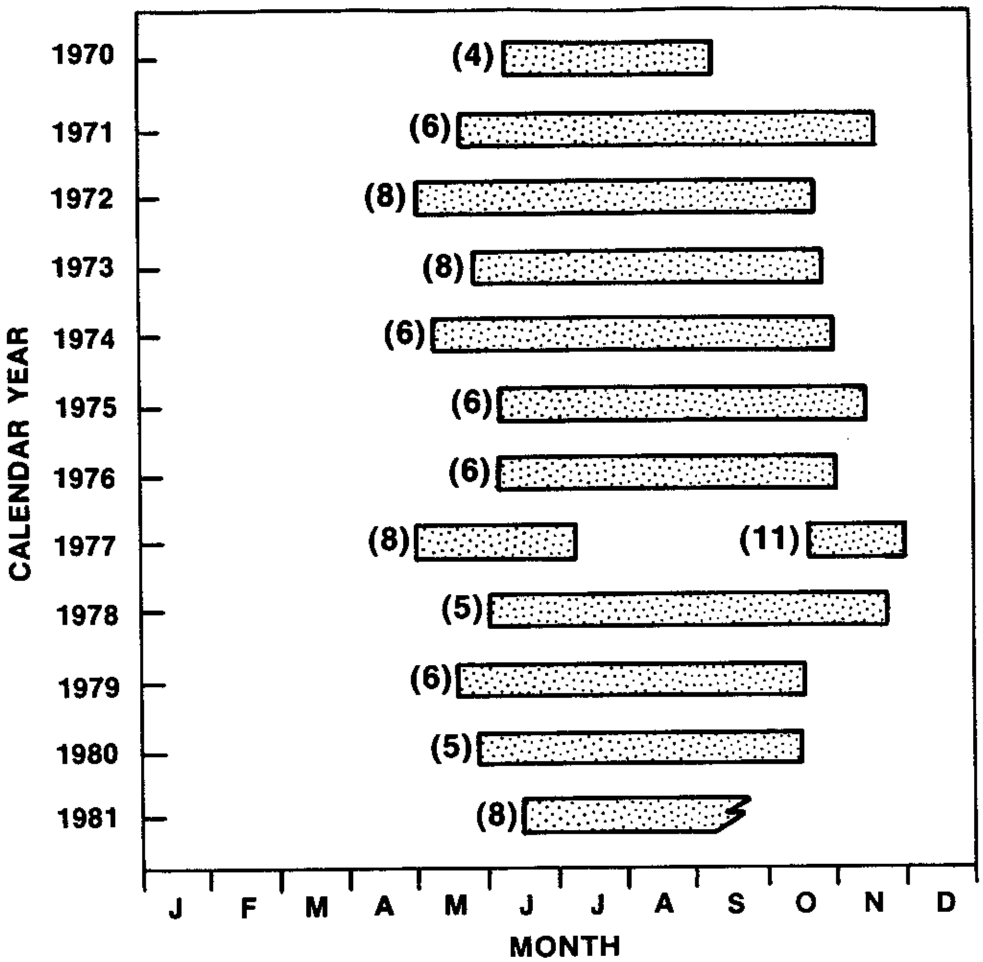

Figure 3. Changes in the duration and extent of hypolimnetic anoxia at Station 12 during the period 1970-1981. Shaded, horizontal bars indicate periods of anoxia. The depth (m) of the maximum upper extent of the anoxic zone is indicated in parentheses.

stations may have been related to the increased importance of algal productivity (Gaugush, pers. comm.) and/or inputs of organic material from the Caddo River (Montgomery, 1986; James and Kennedy, 1986) at these stations.

Temporal differences in oxygen dynamics were further explored by comparing data from 1971-72 and 1980-81, since these data were considered to represent conditions during "early" and "recent" years, respectively. Changes in dissolved oxygen concentration following the onset of thermal stratification during both early and recent years were most rapid in the shallow upstream portion of the lake (Figure 4 and 5). Riverine organic inputs, a shallow and less voluminous hypolimnion, and a large sediment area to hypolimnetic volume ratio in this portion of the lake would lead to a rapid oxygen depletion rate since hypolimnetic dissolved oxygen stores would be relatively small compared to oxygen demands. Similar spatial patterns in oxygen depletion have been observed for other reservoirs. Hannan and Cole (in press) report that anoxia in Canyon Reservoir, a long, deep storage reservoir having a similar morphometry, is restricted to upstream areas and is related to morphometry and flow regime.

Also apparent during both early and recent years was the development of a marked metalimnetic oxygen minimum (Figure 4 and 5). Coincident with rapid declines in oxygen concentration in the upstream portion of the hypolimnion were less rapid declines in oxygen concentrations in the thermocline region. These declines, which were first observed in the upstream third of the lake, progressed downstream throughout the stratified period. The result was the establishment of anoxic conditions in portions of the metalimnion by midsummer. The occurrence of similar patterns of change in metalimnetic oxygen concentrations during both periods and the observation of interflowing density currents in the region of the thermocline (Ford, 1986) suggest the potential importance of autochthonous and allochthonous sources of oxidizable organic matter sources. The settling of algal material (James and Kennedy, 1986), frequently cited as a major factor in the establishment of metalimnetic oxygen minima (Wetzel, 1975), as well as the introduction of oxidizable material from the Caddo River (see Montgomery, 1986) and/or as a result of thermocline erosion and subsequent transport of reduced material downstream (Nix, 1986) may have provided sufficient oxygen demands to deplete oxygen stores in this relatively unmixed portion of the water column. Kim et al. (1983) observed a metalimnetic minimum in Cherokee Reservoir, a deep Tennessee Valley

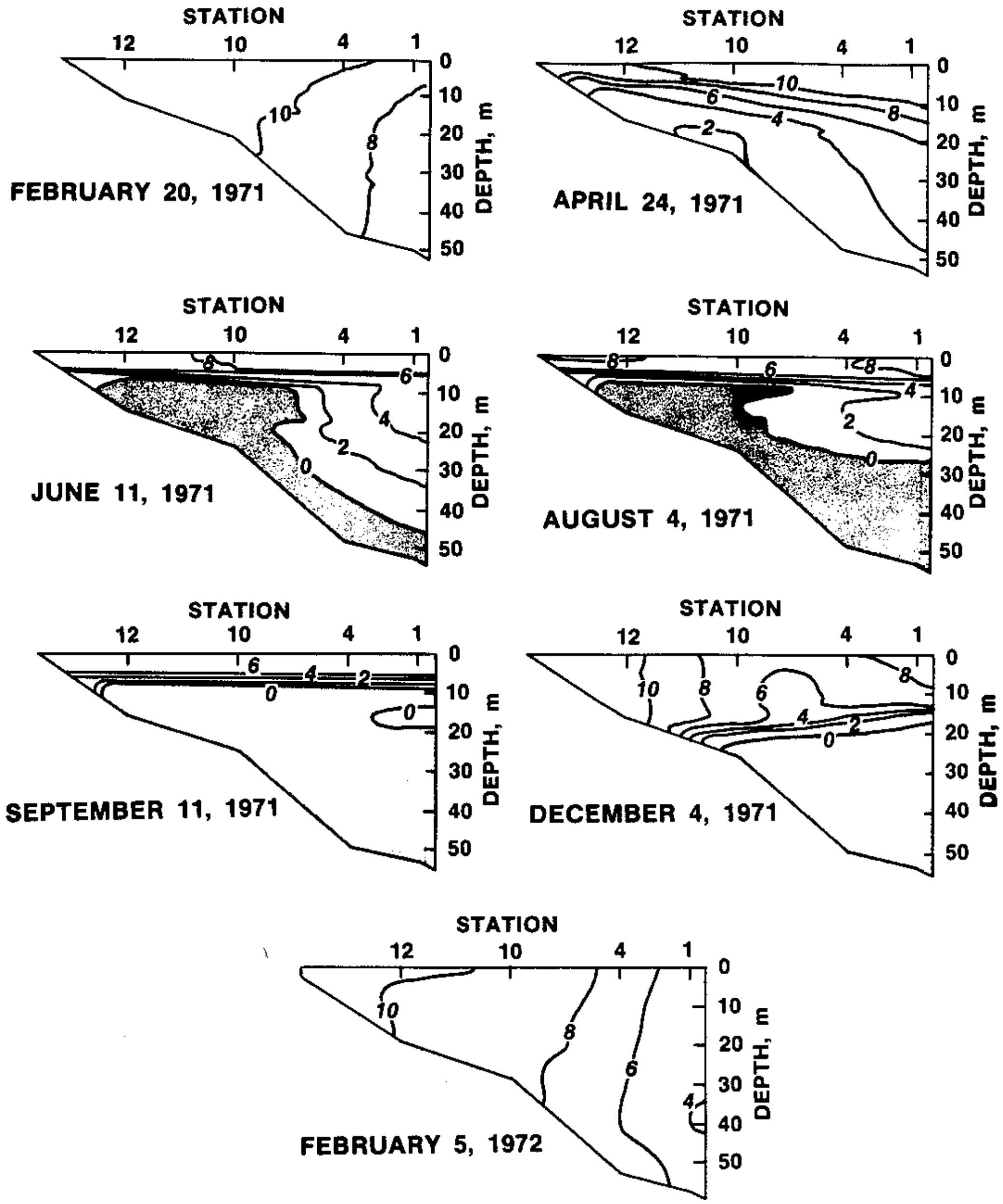

Figure 4. Dissolved oxygen concentrations (mg/l) on selected dates in 1971 and early 1972. Shading indicates anoxic conditions.

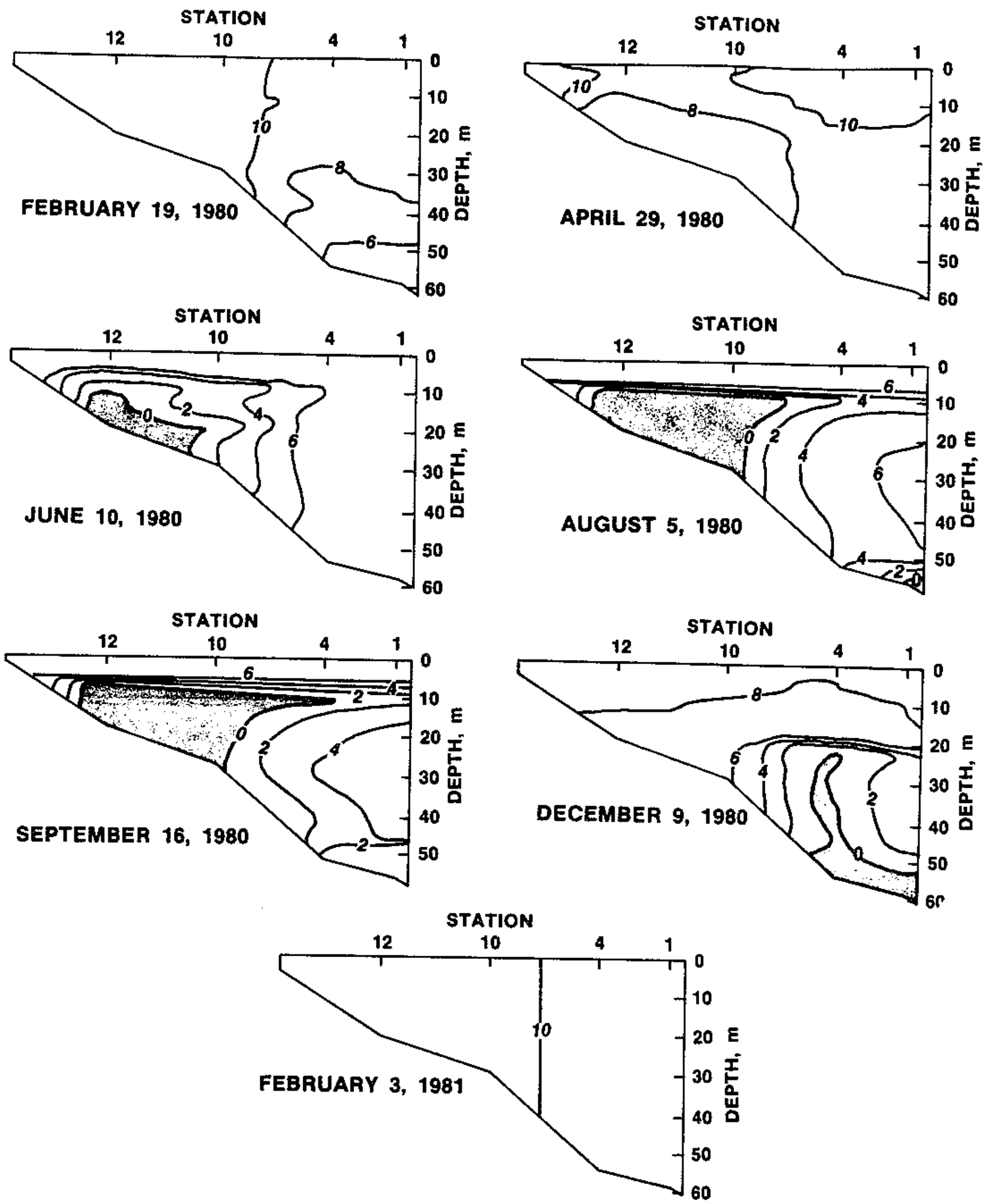

Figure 5. Dissolved oxygen concentrations (mg/l) on selected dates in 1980 and early 1981. Shading indicates anoxic conditions.

Conclusions

Pronounced changes in the oxygen status of DeGray Lake have occurred in the years since impoundment. In years immediately following impoundment, the inundation of soils, vegetation and detritus led to high oxygen demands and the extensive loss of oxygen from the hypolimnion. As readily oxidizable organic materials were depleted, oxygen conditions in the hypolimnion became less severe. During recent years, the oxidation of autochthonous and allochthonous organic materials has played a greater role in the lake's oxygen dynamics. Of particular importance is the deposition of organic materials in the headwater of the lake and the seasonal occurrence of anoxia in the shallow hypolimnion in this region of the lake.

These changes are of significance in understanding other chemical and biological occurrences in the lake. Observations reported here also provide information of general importance in the understanding of events occurring in newly filled reservoirs.

until the entire hypolimnion was involved in anoxia beginning in mid-September.

Patterns in the distribution of rates of decline in oxygen concentration at Station 1 in 1980 differed from those observed in 1971. While both benthic and metalimnetic demands were observed, declines were confined to a shorter period of time. Maximum rates of decline occurred immediately above the sediment/water interface in mid-August, near the thermocline in late June and early July, and throughout the hypolimnion in late May and early June.

Conditions at Station 12 were more severe in later years than in the early years. While anoxic conditions were observed during both years, rates of decline were higher in 1980 than in 1971. With the exception of the near-thermocline area, similar rates of decline in oxygen concentration were observed during both years at Station 10.

These data suggest the influence of two types of oxygen demands and the occurrence of significant temporal trends in their relative importance to the oxygen dynamics of DeGray Lake. Similarities between years (i.e., 1971 and 1980) in the timing, duration and extent of anoxia at Station 12, the station located in an area cleared of vegetation during construction and receiving direct inputs from the Caddo River, would indicate that oxygen demands are exerted primarily by autochthonous production and riverine inputs. This suggestion is supported by recent data indicating the occurrence of high rates of primary productivity (Gaugush, pers. comm.) and chlorophyll maxima (Thornton et al., 1982) in this portion of the lake. Oxygen conditions at Station 1, however, reflect the potential importance of inundated terrigenous material in early years and the relatively greater importance of benthic and metalimnetic oxygen demands during recent years. Similar oxygen conditions across years at Station 10 may have resulted from a progressive change in the relative importance of sources of various oxygen-demanding materials. During early years, terrigenous materials would have represented a significant source of oxygen-demanding organic matter. In recent years, this source would have been supplemented by autochthonous production of organic material in upstream portions of the pool and by riverine inputs, which are frequently observed to influence this reach of the lake (Ford, 1986). Sediment surveys (Gunkel et al., 1984) also indicate the accumulation of sediments in this area of the lake. The presence of these sediments,and the organic material they contain, would lead to the progressive importance of benthic oxygen demands.

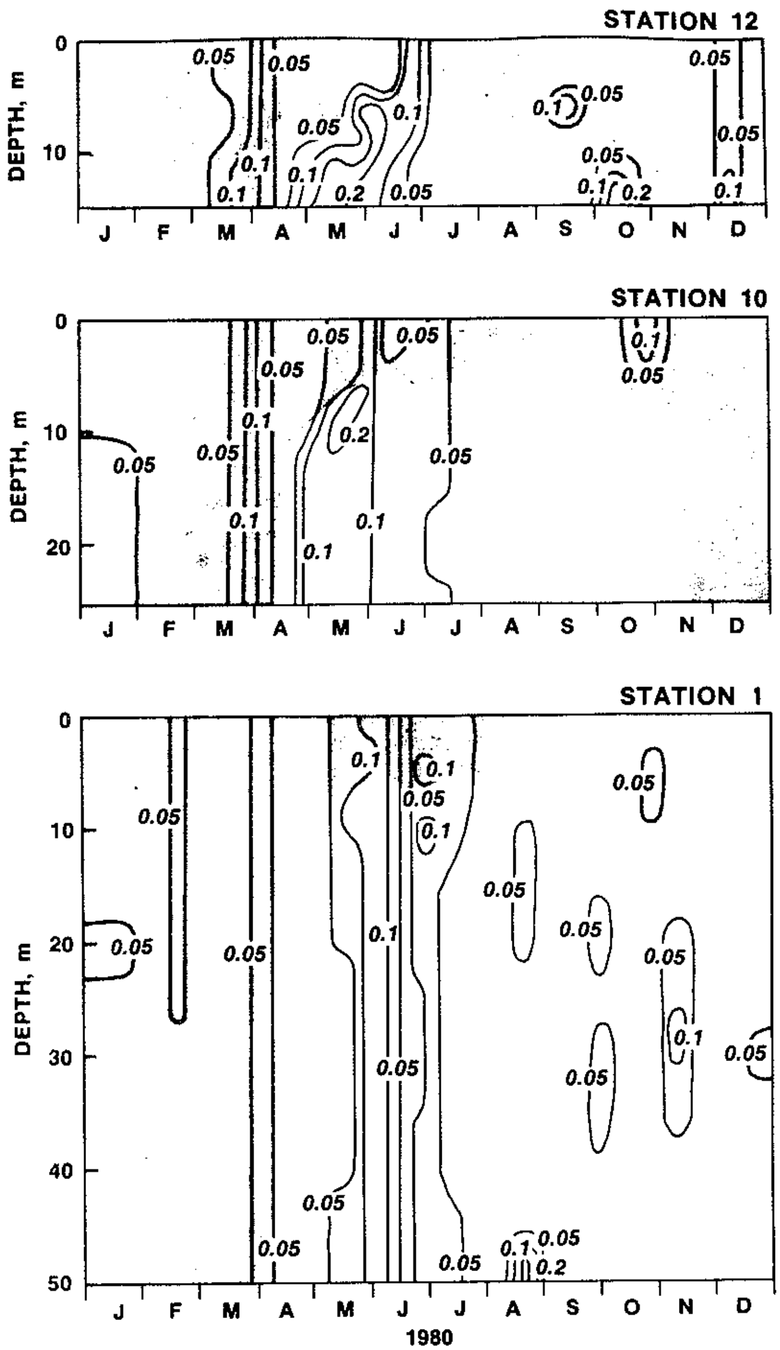

Figure 7. Dissolved oxygen depletion rates (gm/m^3 per day) at Stations 1, 10 and 12 during 1980. Shaded areas represent the upper, well-mixed portion of the water column.

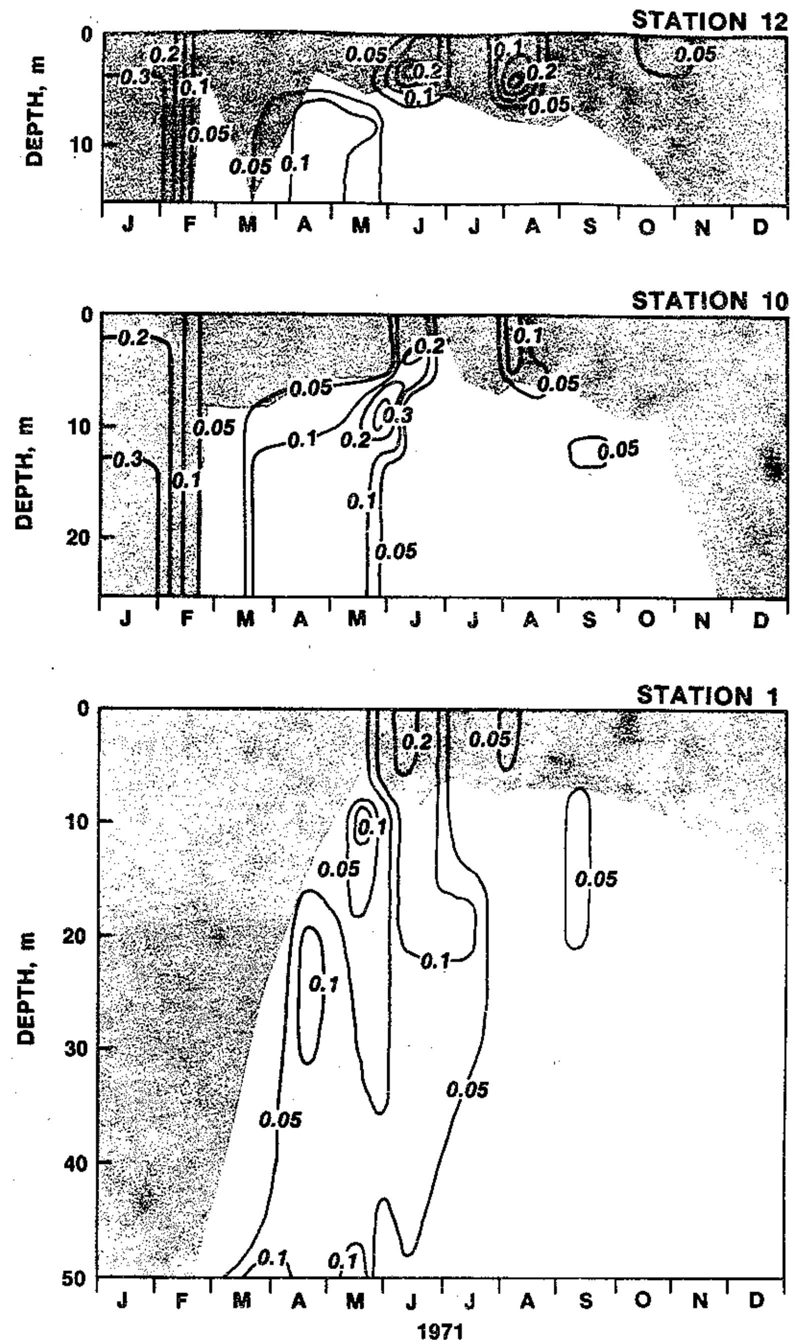

Figure 6. Dissolved oxygen depletion rates (gm/m^3 per day) at Stations 1, 10 and 12 during 1971. Shaded areas represent the upper, well-mixed portion of the water column.

Table 1. Areal and volumetric hypolimnetic oxygen deficit rates.

	Areal Rate[a]			Station	Volumetric Rate[b]		
Year	1	10	12		1	10	12
1970	1090	510	--		93	105	--
1971	830	480	220		62	73	101
1972	550	870	550		36	103	124
1973	510	550	270		33	64	58
1974	560	680	320		36	79	68
1975	800	480	620		52	56	113
1976	490	850	270		32	99	58
1977	560	470	430		36	55	92
1978	340	490	550		22	58	125
1979	890	900	780		57	104	164
1980	690	890	700		45	105	155

a - Areal oxygen deficit rates have units of mg/m^2 per day.

b - Volumetric oxygen deficit rates have units of mg/m^3 per day.

Authority lake. Gordon and Skelton (1977) attributed such observations to reduced algal settling rates in the metalimnion.

During early years, oxygen declines in the metalimnion and in near-bottom strata resulted in complex oxygen profiles in mid-summer and complete hypolimnetic anoxia by late summer. In recent years, hypolimnetic anoxia was restricted to upstream areas and deep areas immediately upstream from the dam, and negative heterograde oxygen curves were routinely observed at mid-pool. However, major portions of the hypolimnion were impacted by deepening of the thermocline in late fall and early winter of 1980 (Figure 5). Reduced material accumulating in the hypolimnion in the vicinity of Stations 12 and 10 was apparently transported to deeper, downstream areas following turnover; this resulted in new demands on oxygen stores in this portion of the hypolimnion. Evidence for the occurrence of downstream transport is provided by data indicating the translocation of nutrients (Kennedy et al. 1983) and selected metals (Nix, 1986; James and Kennedy, 1986).

In apparent contrast to the pronounced trend of improving oxygen conditions discussed above, no clear trends are apparent in the areal or volumetric hypolimnetic oxygen deficit rates (Table 1). Although greatest during 1970, deficit rates at Station 1 during the period 1971-80 were highly variable. Rates at Station 10 and 12, which were also highly variable, were as high or higher in 1979-80 than in early years.

A consideration of rates of change in oxygen concentration throughout the water column during the stratified period in 1971 (Figure 6) and 1980 (Figure 7) provides insight to the manner in which oxygen demands are exerted. While calculation of both areal and volumetric hypolimnetic oxygen deficit rates account for net changes in oxygen content of the entire hypolimnion between the period of isothermal conditions in the spring and the onset of anoxia in bottom waters, depth-specific calculations allow delineation of spatial (i.e., vertical) patterns in oxygen demand. During 1971, decreases in oxygen concentration at Station 1 were apparent throughout the hypolimnion immediately following stratification. Rates of change were greatest near the sediment/water interface and below the developing thermocline, suggesting the occurrence of both a benthic and metalimnetic oxygen demand. Rates continued to be high in bottom waters until late May when anoxic conditions were established in this stratum. A prolonged period of relatively high rates of change in oxygen concentration was observed in upper portions of the hypolimnion

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NUTRIENT DYNAMICS IN DEGRAY LAKE, ARKANSAS

by

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Introduction

Lake and reservoir trophic dynamics are regulated, in part, by the supply and availability of inorganic nutrients required for the growth and reproduction of aquatic plants, particularly algae. While the requirements of algae are diverse, the relative abundances of nitrogen and phosphorus play primary roles in determining algal production. Because of this, nitrogen and phosphorus concentrations are frequently used as indicators of the potential trophic state of freshwaters (Vollenweider, 1969; Carlson, 1977).

The transport of nitrogen and phosphorus from watersheds via streams and rivers is the primary means by which these nutrients are introduced to lakes and reservoirs. However, recycling is often an important process leading to increased availability of these nutrients, particularly in lakes and reservoirs containing large accumulations of sediment and detritus. For newly filled reservoirs, the inundation of vegetation and soils rich in organic materials provides additional internal sources of these nutrients through leaching and solubilization. The potential importance of this latter source of nutrients for new reservoirs was suggested by Grimard and Jones (1982) who described the "trophic upsurge" or increases in nutrient concentrations following the filling of Smallwood and La Grande-2 reservoirs in Canada. The period of trophic upsurge lasted approximately 1-3 years and was followed by a trophic depression period during which nutrient concentrations declined to levels more consistent with the observed external loading rate.

DeGray Lake is a Corps of Engineers hydroelectric impoundment located on the Caddo Rover in south-central Arkansas. The lake was filled in 1969-1970. Inundated during this process were 54 sq km of soils and standing timber. Comprehensive monitoring of selected water quality variables began prior to impoundment and continued until 1982. Described here are changes in the nutrient (phosphorus and nitrogen) status of the lake during the years following impoundment.

Methods

Four primary lake stations (Stations 1, 4, 10, and 12) and a single river station (Hwy 84) were established for monitoring nutrient concentrations. The locations of these stations are described by Kennedy and Carroll (1986). Water samples were initially collected using a nonmetallic sampling bottle and later using a sampling system consisting of a polyethylene hose, a deck-mounted centrifugal pump, and an optional in-line, filter manifold. All samples were transferred to acid-washed bottles and stored in the dark at 4 deg. C. Samples for total nitrogen and phosphorus, and dissolved iron analyses were preserved with acid.

During periods of anoxia, care was taken to avoid aerating samples upon which analyses of soluble forms were to be performed. This required withdrawing samples from the sample bottle using a syringe. The in-line filtration manifold, used during a major portion of the study, provided a means for filtering water samples prior to any possible exposure to air.

Total phosphorus concentrations of unfiltered samples were determined colorimetrically following acid-persulfate digestion (APHA, 1980). Soluble reactive phosphorus concentrations of filtered (0.45-um membrane), undigested samples were determined using the same colorimetric method. Total nitrogen concentrations were determined colorimetrically following acid-persulfate oxidation digestion using methods similar to those described by Raveh and Avnimelech (1979). Dissolved iron concentrations were determined by atomic absorption spectrophotometry.

Results and Discussion

The total nitrogen and total phosphorus content of DeGray Lake during the period 1974-80 is plotted in Figures 1 and 2. Although variable within years, a marked and steady decline in total nitrogen content occurred during this period. Nitrogen content in 1975, five years following impoundment of the lake, ranged from approximately 350 to over 1,150 metric tons. In 1980, the content of nitrogen in the lake was seasonally less variable and ranged from 250 to 600 metric tons.

Total phosphorus content, while declining over the same period, displayed a markedly different trend. Although initially high in 1974 and early 1975, phosphorus content declined sharply in late 1975. While peaks in content continued to occur on a seasonal bases (i.e., peaks in spring and fall), values for seasonal minima were similar (5 to 10 metric tons) for the period 1976 through 1980. During this same period increases in the amplitude of seasonal changes in phosphorus content (i.e., the difference between seasonal maxima and minima) were also observed. As will be discussed below, these latter changes were related to changes in phosphorus and iron relationships, and the cycling of phosphorus between sediment and water.

Trends of change in total nitrogen and phosphorus content were clearly related to impacts of initial impoundment and the changing importance of several processes during subsequent years as the newly filled lake "stabilized." While data for the years immediately following impoundment are lacking, trends observed for the period 1974-80 suggest that internal sources (e.g., the decomposition of inundated organic materials associated with soils and/or terrestrial detritus) were important in maintaining nitrogen concentrations during the early years. In subsequent years, as stores of labile materials were reduced through decomposition, the importance of nitrogen supplies from external sources increased. Changes in the duration and extent of anoxia during this same period (Kennedy and Nix, 1986) support this suggested decline in the quantity of readily oxidizable organic material covering bottom sediments.

While the leaching of soluble phosphorus and phosphorus-containing organic compounds from decomposing organic materials and/or from flooded soils was potentially an important phosphorus source during early years, seasonal patterns of change in phosphorus content suggest the influence of internal

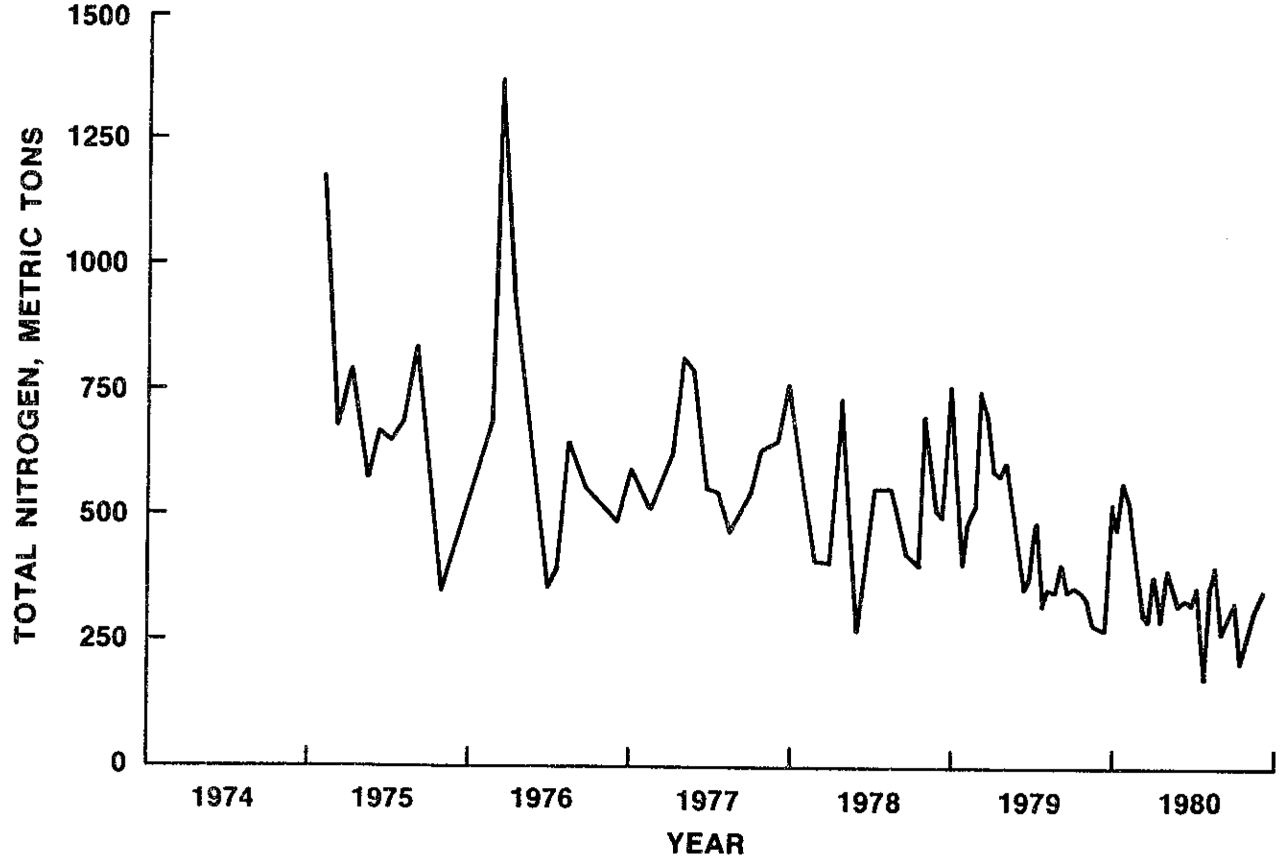

Figure 1. Changes in the total nitrogen content (kg) of DeGray Lake during the period 1974-1980.

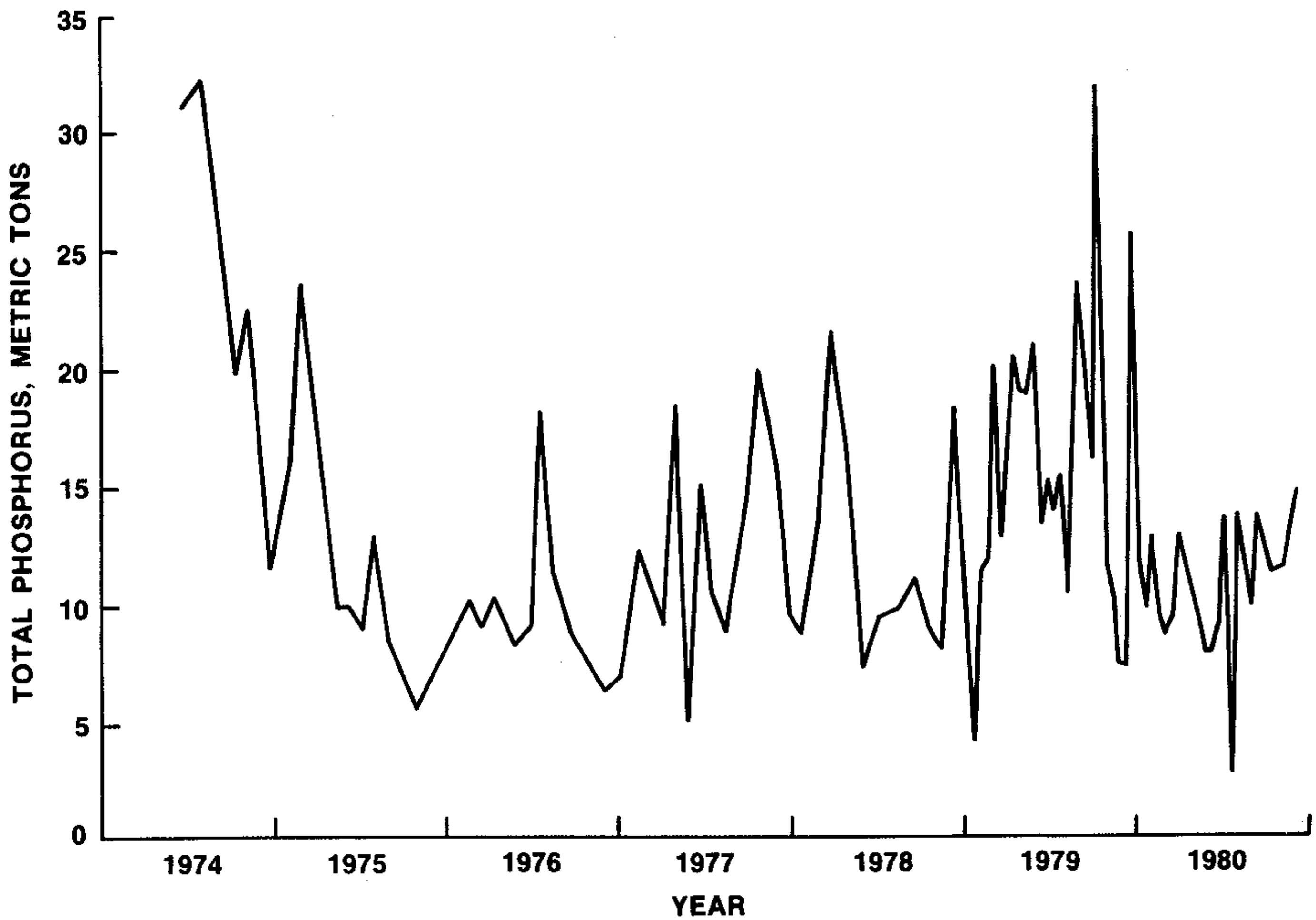

Figure 2. Changes in the total phosphorus content (kg) of DeGray Lake during the period 1974-1980.

recycling. Further, the increased amplitudes of seasonal change in phosphorus content suggest that the importance of internal recycling increased over time.

Depth- and time-related changes in the concentration of soluble reactive phosphorus during 1971 to 1980 are presented in Figures 3 and 4. During 1971 concentrations were relatively high throughout the water column at each station. Also apparent were concentration increases in bottom waters during the summer, stratified months, particularly at Station 12; these increases coincide with the occurrence of anoxia. Concentrations in near-bottom strata ranged from 100 to 150 ugP/1, while those for surface waters ranged from less than 25 ugP/1 to 50 ugP/1.

During 1980, concentrations throughout the water column at the near-dam station (Station 4) and in surface waters at Stations 10 and 12 were much reduced. However, hypolimnetic concentrations at Stations 10 and 12, but not the near-dam station, were greatly elevated during periods of anoxia. With few exceptions, concentrations in surface waters were less than 25 ugP/1. Hypolimnetic concentrations as high as 250 and 350 ugP/1 were observed at Stations 10 and 12, respectively. These increases coincided with periods of anoxia and markedly reduced loading from the Caddo River (see Montgomery, 1986).

Relationships between mean hypolimnetic soluble reactive phophorus concentrations and mean hypolimnetic dissolved iron concentrations for each station during anoxic periods of 1971 and 1980 are plotted in Figure 5. Slopes of regression lines (Table 1) estimate the average soluble reactive phosphorus to dissolved iron ratio (P/Fe) in the hypolimnetic portion of the water column at each station for each year. During 1971, P/Fe values ranged from 0.005 at Station 1 to 0.014 at Station 12. Relationships during 1980 were significant ($p>0.05$) for Stations 10 and 12, but not for Station 1. Phosphorus to iron ratios for the former two stations were 0.026 and 0.041, respectively. Clearly, important changes in the relationship between phosphorus and iron occurred over this period of time. Since few changes in landuse patterns and, therefore, in phosphorus and iron loading from the Caddo River, were documented during this interval of time, it is reasonable to suggest that differences in P/Fe in the water column were attributable to the occurrence of an internal storage and recycling mechanism.

Redox-related release of phosphorus from iron-containing sediments is well documented for freshwaters (see review in Ryding, 1985). As dissolved

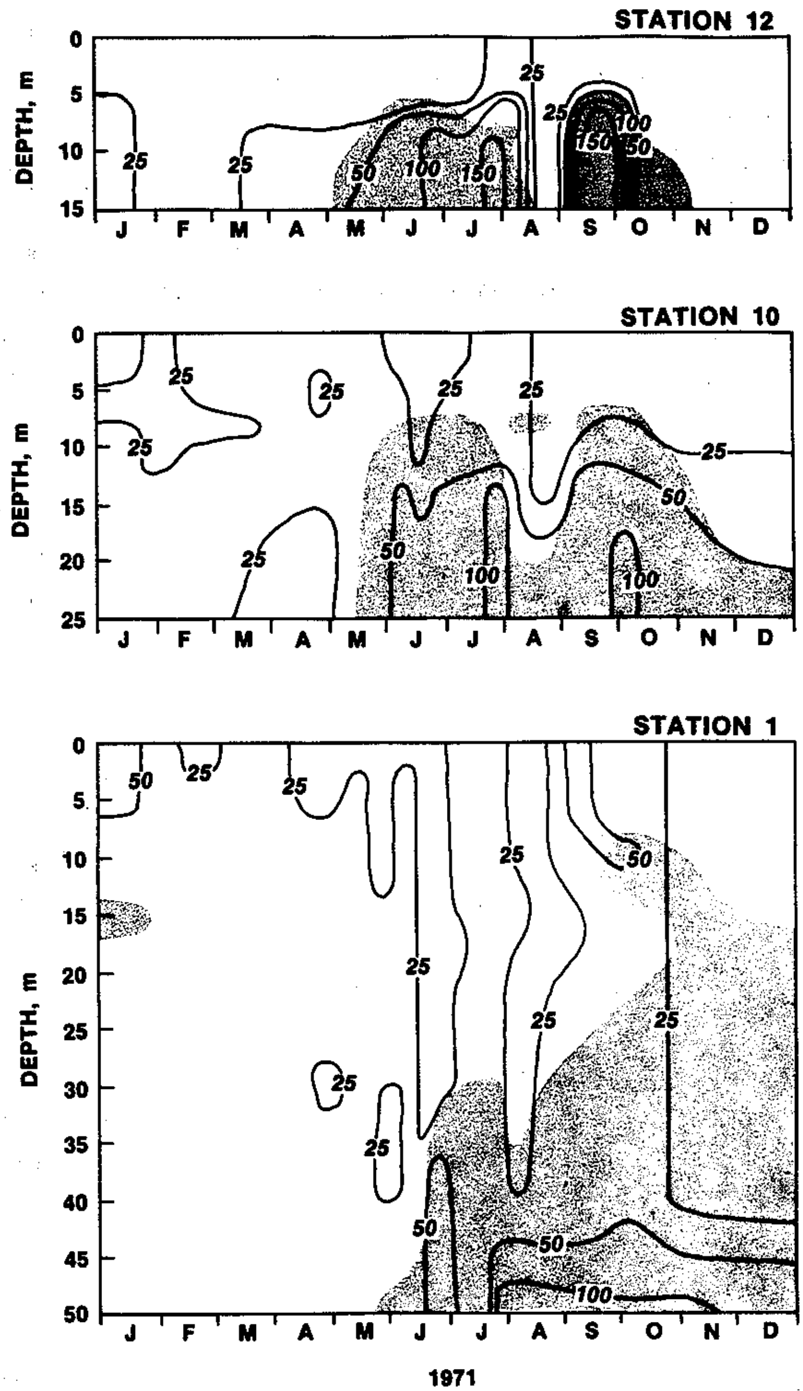

Figure 3. Changes in soluble reactive phosphorus concentrations (ug P/l) at Stations 1, 10, and 12 during 1971. Shading indicates zones of anoxia.

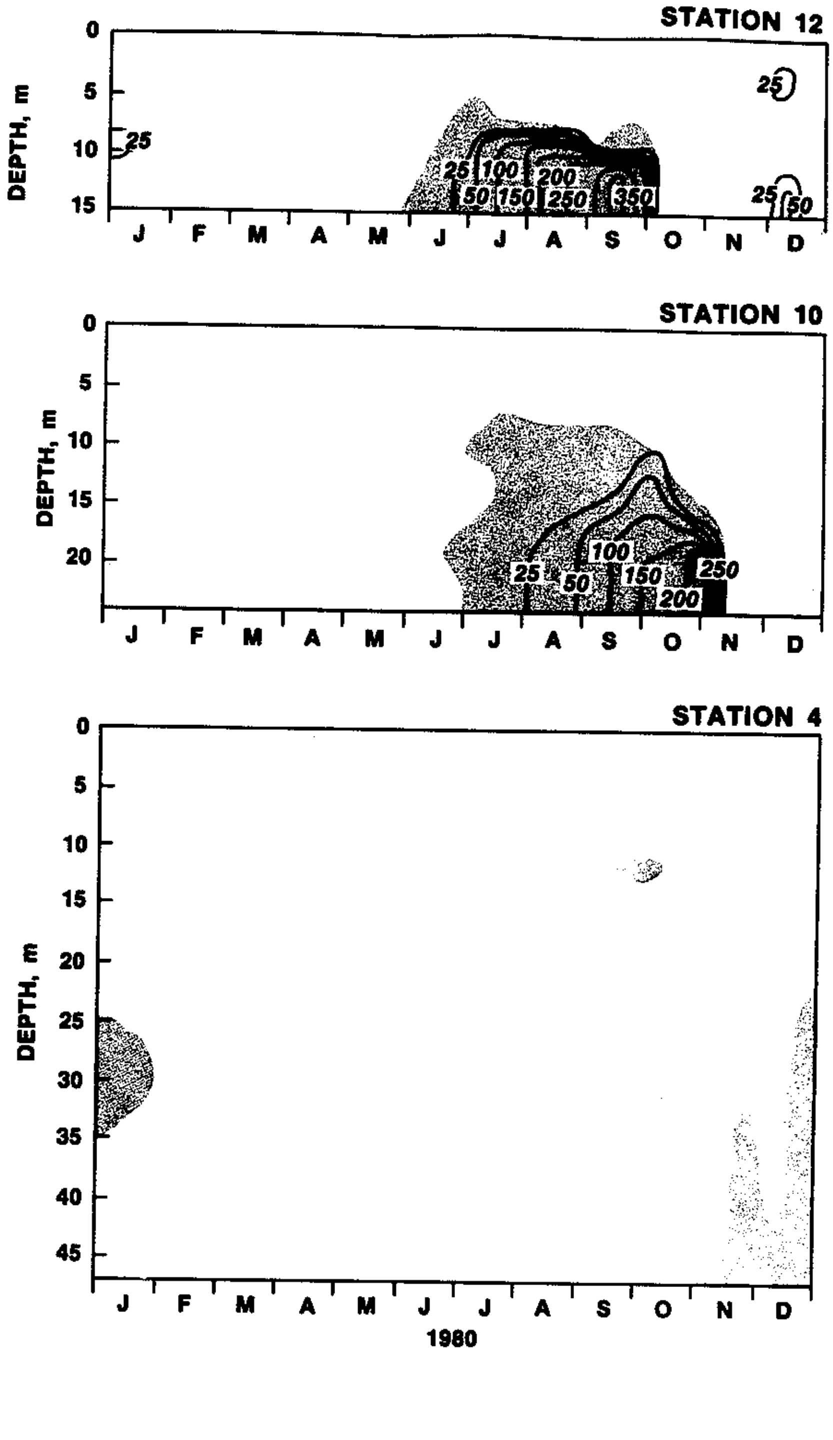

Figure 4. Changes in soluble reactive phosphorus concentrations (ug P/l) at Stations 4, 10, and 12 during 1980. Shading indicates zones of anoxia.

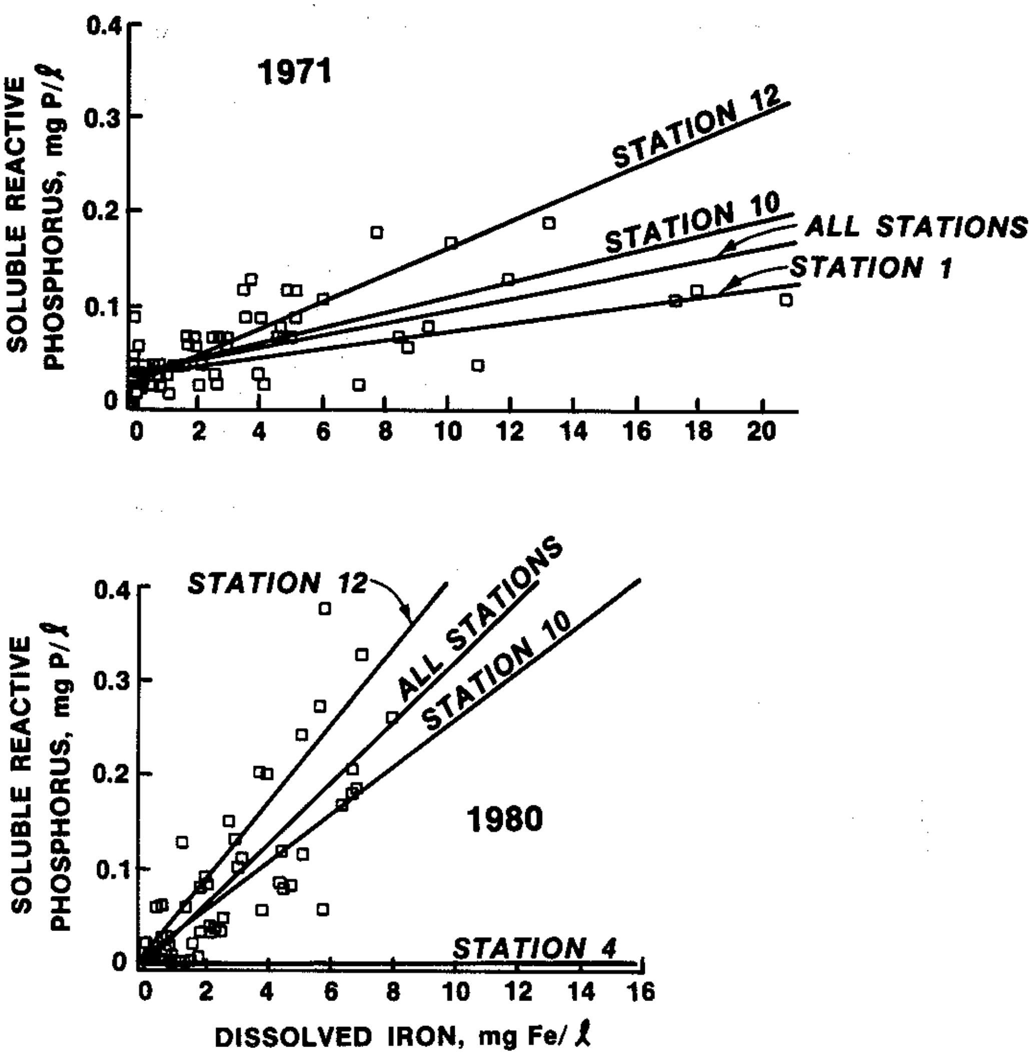

Figure 5. Relationship of mean hypolimnetic soluble reactive phosphorus concentrations to mean hypolimnetic dissolved iron concentrations in 1971 and 1980. Lines represent results of regression analyses for data from Stations 1, 4, 10, and 12, and for pooled data from all stations.

Table 1. Mean hypolimnetic soluble reactive phosphorus to mean hypolimnetic dissolved iron (P/Fe) ratios. Ratios are based on regression analyses of data presented in Figure 5.

Station	Year	
	1971	1980
1	0.005	--
4	--	ns
10	0.008	0.026
12	0.014	0.041
All	0.007	0.032

ns = nonsignificant ($p>0.05$).

oxygen concentrations in bottom waters approach zero and at oxidation-reduction potentials below 200 mV, both phosphorus and iron are solubilized from previously insoluble iron/phosphorus complexes at sediment surfaces and the concentrations of these elements increase in the overlying water column. When redox conditions change, as following mixing and reaeration, iron and phosphorus are again removed from the water column as a coprecipitate.

Kennedy et al. (1986) report the seasonal deposition of influent particulates in the headwater area of DeGray. The long-term effect of this deposition has been a greater accumulation of sediments in this area of the lake (Gunkel et al., 1984). Also observed here were higher concentrations of both iron and phosphorus when uplake and downlake sediments were compared.

These spatial patterns in the quantity and quality of sediments have been influenced by seasonal loadings from the watershed, flow regime and seasonal changes in oxygen conditions. Materials transported from the watershed, particularly during the spring high-flow period, settle and accumulate in the shallow, upstream area of the lake. Following the establishment of stratified conditions, the oxidation of organic material in this shallow portion of the hypolimnion results in anoxia and the release of iron and phosphorus. Advective transport is minimal during the summer, low-flow period and the concentrations of iron and phosphorus increase in the upstream portion of the hypolimnion. During fall mixing, reaeration of the hypolimnion results in the redeposition of iron and phosphorus complexes.

In the years since impoundment of DeGray Lake, the seasonal occurrence of these events has led to the observed accumulation of iron and phosphorus in the lake's headwater area and an increased interaction between these two elements. The increase in the amplitude of seasonal change in phosphorus content of the lake is a direct consequence of these occurences.

Conclusions

Important changes in the relative influences of various processes on the dynamics of nitrogen and phosphorus have occurred during the period following impoundment of DeGray Lake. Nitrogen inputs from the watershed have remained relatively constant, while the importance of nitrogen sources within the lake has diminished over time coincident with declines in the quantity of oxidizable organic material inundated during initial filling. Phosphorus concentrations, while also initially influenced by the inundation of terrestrial material, are now regulated to a greater extent by interactions with iron and by seasonal changes in oxygen status. Events of greatest impact on phosphorus dynamics are those occurring in the headwater area of the lake. It is here that external phosphorus loads are initially stored in sediments and later recycled during periods of anoxia. Since this coincides with the period of minimal phosphorus loading from the Caddo River, recycling from bottom sediments plays a potentially important role in determining the trophic status of this lake.

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DISTRIBUTION OF IRON AND MANGANESE IN DeGRAY RESERVOIR, ARKANSAS

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Introduction

The spatial and temporal distribution of iron and manganese in seasonally anoxic lakes has been studied by numerous investigators. A basis for the relationship between redox potential of the aquatic environment and the chemistry of these elements was described early by Mortimer (1941). Stumm and Morgan (1970) summarized the aquatic chemistry of iron and manganese transformations in various natural water systems, including the seasonally anoxic lake. Recent sediment trap studies have provided new information on the flux of these elements in seasonally anoxic lakes (Davison et al.1982).

Assessing the distribution of iron and manganese species has become a standard limnological practice in studying seasonally anoxic lakes. Consequently, considerable data exists on the distribution of iron and manganese and provides a basis for understanding the factors influencing iron and manganese cycling in various types of lakes. However, the reliability of analytical techniques, sample handling procedures, and sample preservation techniques have prevented a detailed comparison of data from many of these studies (Stumm and Lee, 1960; Collienne,1983). Even with these limitations, the generalized chemistry of iron and manganese in limnetic environments has been well developed (Hutchinson, 1957; Golterman, 1975; Wetzel, 1975).

Cycling of these elements in a stratified lake generally begins with the development of an oxygen depressed zone at the mud-water interface in the hypolimnion. Reduced species of iron and manganese are formed, which are stable in the low redox environment of the anaerobic bottom muds. Since these reduced species are soluble, they diffuse into the overlying water column. The thickness of this anoxic zone usually increases throughout the period of stratification and in some cases, reaches the top of the hypolimnion before fall turnover. As hypolimnetic water becomes oxygenated by mixing, the reduced species of iron and manganese are oxidized to insoluble compounds and settle from the water column.

This rather simplified description of iron and manganese cycling in a seasonally anoxic lake can be complicated by numerous factors. For example, manganese can be reduced from its insoluble (+4) state to the soluble (+2) state at significantly higher redox potentials than the corresponding reduction of iron from its insoluble (+3) state to its soluble (+2) state. As pointed out by Stumm and Morgan (1970) and Davison et al. (1982), this difference in chemistry may result in different patterns of cycling of these two elements in seasonally anoxic lakes. It has also been suggested that the source of reduced iron in the anoxic hypolimnion is diffusion from bottom muds, while the source of reduced manganese may be *in situ* reduction as oxidized manganese particles settle through the anoxic zone. It seems likely, however, that reduced manganese could originate from both bottom muds and *in situ* reduction.

Different redox chemistry for iron and manganese could result in different patterns of cycling due to varying oxidation and precipitation rates. Gordon

(1983) has pointed out that the mechanisms for iron and manganese oxidation are quite different with iron following essentially first order kinetics while manganese follows a somewhat different path probably involving autocatalysis. The differences in the oxidation mechanism could result in manganese remaining in suspension significantly longer than iron following mixing of the water column.

Damming of a river can, in some cases, result in a body of water which has significantly different properties from most lakes. Although not completely separable, a reservoir usually has characteristic sedimentation patterns and density currents that are affected by such factors as flow regime and outlet depth. These factors may significantly impact the cycling and distribution of iron and manganese in seasonally anoxic reservoirs. Gunnison and Brannon (1981) have addressed some of these factors and have provided a detailed sequence of redox reactions resulting in the production of soluble iron and manganese species in reservoir hypolimnia.

DeGray Reservoir, which was formed by the damming of the Caddo River in Southwest Arkansas, has been the site of field studies since impoundment began in 1969. Numerous studies have provided iron and manganese, as well as other water quality data on this reservoir at periodic intervals from 1969 through 1981. Data collected over this 12-year period indicate the spatial and temporal distribution of iron and manganese can be explained in terms of established concepts of redox chemistry when appropriately coupled with characteristic processes of deep water reservoirs.

Methods

Filling of DeGray Reservoir was begun in 1969 with the conservation pool being reached in 1971. The general characteristics of DeGray are summarized in Table 1.

Sampling stations were located directly over the old river channel at points ranging from the dam to the upstream portion of the impoundment (Figure 1).

From 1971 through June 1978, samples were taken at various depths using a van Dorn type sampler. From July 1978 through the 1982 samples were taken by pumping from the desired depth. Comparison studies conducted in 1978 indicated that there was no significant difference in samples taken by the two different methods. Immediately after collection of each sample, an aliquot was acidified and reserved for analysis of total iron and total manganese. A second aliquot was immediately filtered through a 0.45-micron filter, acidified with nitric acid and reserved for analysis of soluble iron and manganese. Iron and manganese were determined using standard atomic absorption spectroscopy techniques (direct aspiration, flame method). A detection limit of 0.1 mg/L was obtained for both analyses.

Temperature, dissolved oxygen, and pH were determined *in situ* using a Hydrolab or a Martek water quality analysis system. Prior to 1978, temperature and dissolved oxygen were determined using a Yellow Springs Dissolved Oxygen Analyzer.

Results and Discussion

Since the pattern of cycling of iron and manganese in a reservoir is closely

TABLE 1

Normal Pool Elevation	124.4 M msl
Volume	8.07×10^8 M^3
Surface Area*	54.2 k M^2
Shoreline Length*	333 kM
Maximum Depth*	60 M
Average Depth*	14.9 M
Drainage Area	1,173 kM^2

*at elevation 124.4 M

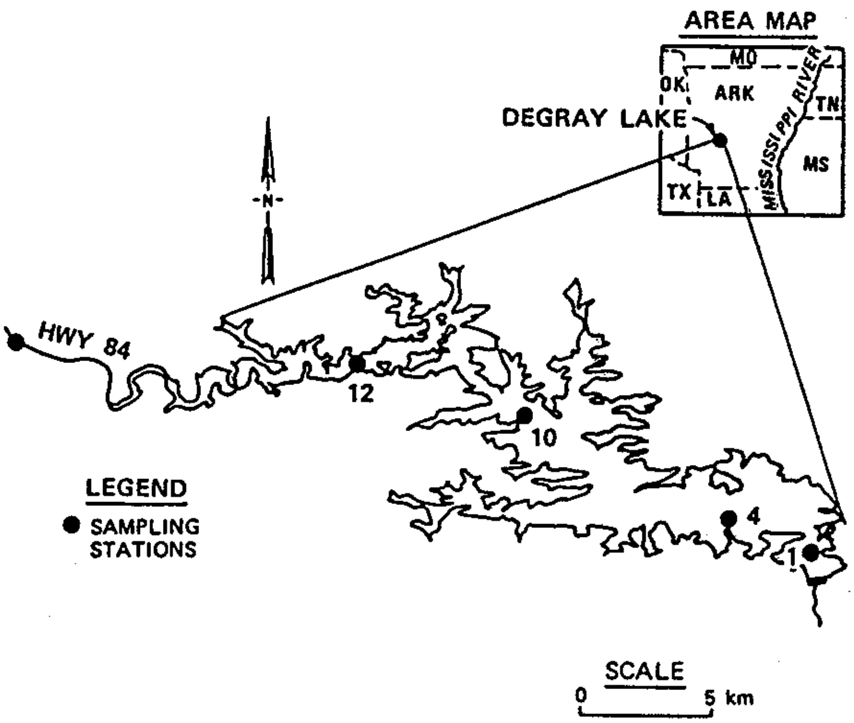

Figure 1. Sampling location in DeGray Lake, Arkansas

related to its dissolved oxygen regime, a brief review of the spatial and temporal distribution of dissolved oxygen in DeGray Reservoir is presented.

As expected, hypolimnetic dissolved oxygen depletion occurred in a greater hypolimnetic volume during the early years following impoundment. Figure 2 presents the dissolved oxygen profiles observed in Station 1 during mid-summer from 1970 through 1981. Severe hypolimnetic dissolved oxygen depletion was observed at Station 1 during the early years of impoundment but moderated by 1973 and 1974. Subsequently, the general trend has been to observe higher hypolimnetic dissolved oxygen concentrations.

Although the exact pattern of oxygen depletion varied, a metalimnetic dissolved oxygen minimum was observed each year. The relationship of this metalimnetic minimum to the transport of iron and manganese will be discussed later.

The data presented in Figure 2 are representative only of the downstream portion of the reservoir. Patterns observed at upstream stations were significantly different. In Figure 3, dissolved oxygen data observed at Stations 10 and 12 are presented for 1976. It is clear that following the onset of stratification, dissolved oxygen depletion is first observed in the upstream hypolimnion and that it gradually progresses in a downstream direction. Although the distance to which anaerobic hypolimnetic conditions extend into the reservoir varied each year, the pattern of early oxygen depletion in the upstream hypolimnion was observed from 1970 through 1981. The metalimnetic dissolved oxygen minimum is also observed at the upstream stations early in the year but the pattern is confounded as the entire hypolimnion becomes anaerobic.

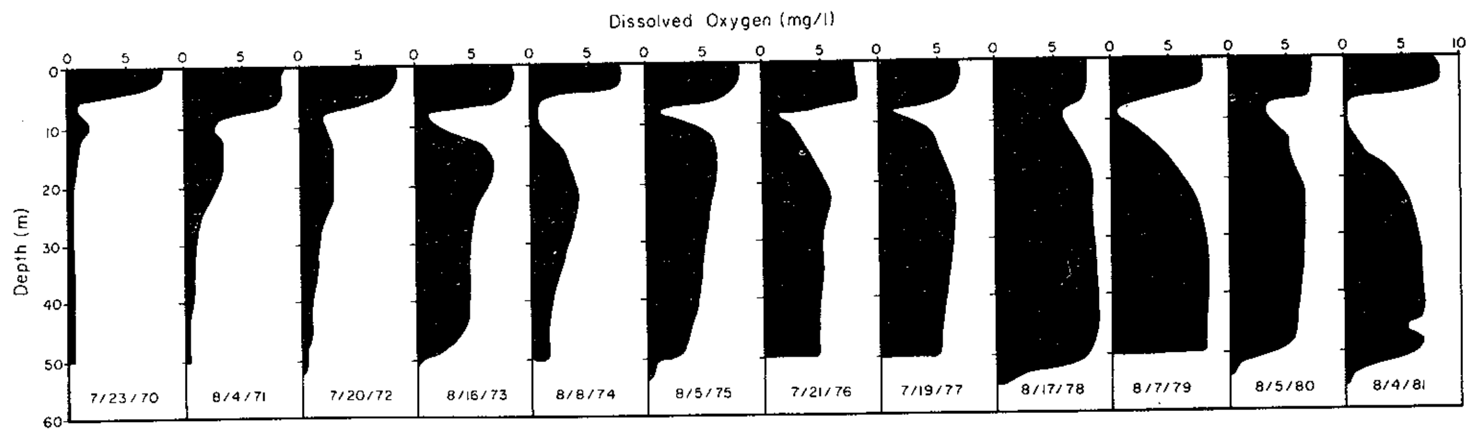

Figure 2. Late summer Dissolved Oxygen at Station 1 from 1970-1981.

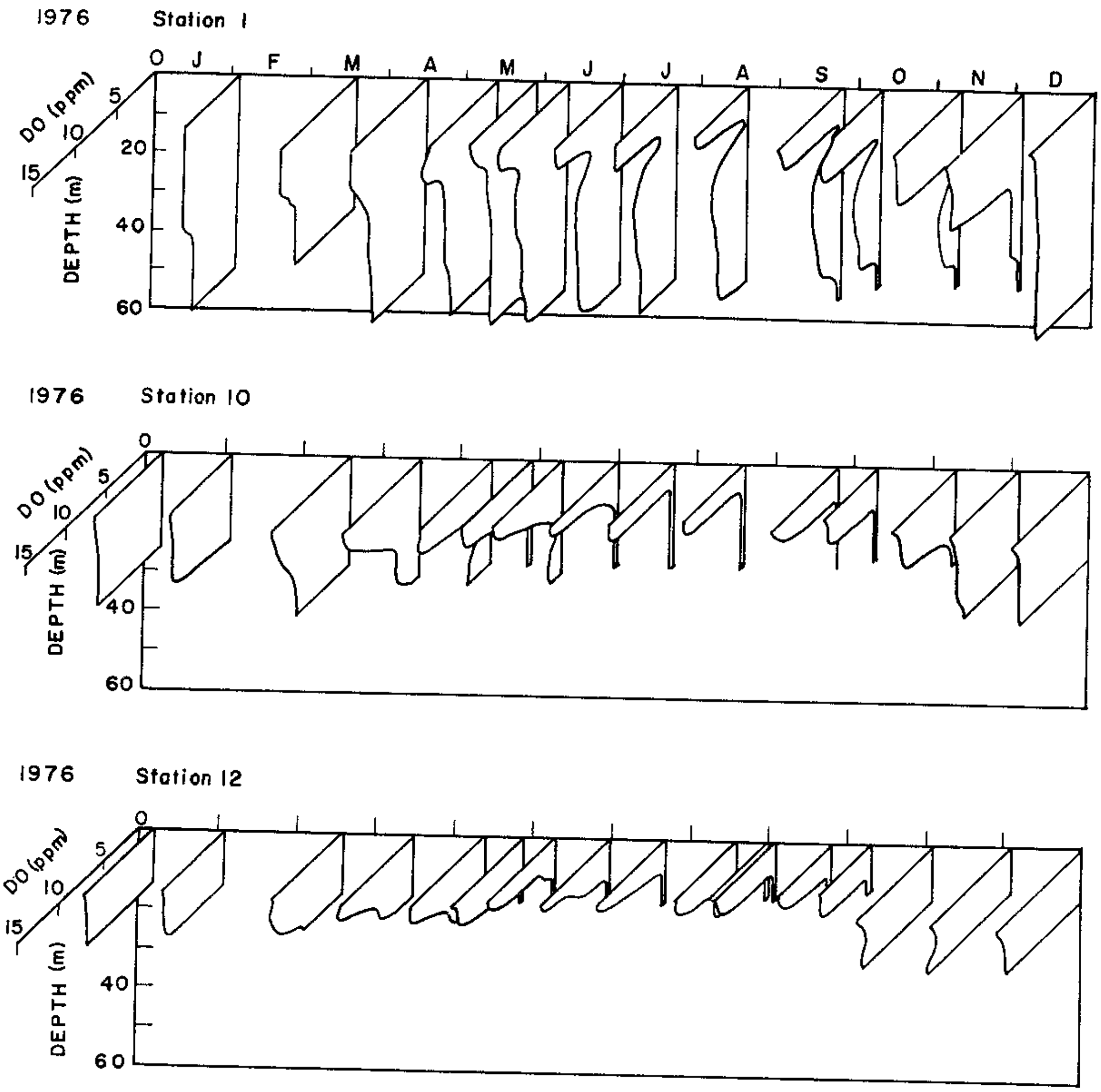

Figure 3. Dissolved Oxygen at Stations 1, 10, and 12 for 1976.

Soluble manganese concentrations observed in DeGray during 1972-1973 and 1979-1980 are summarized in Figures 4 and 5, respectively. Prior to 1973, soluble manganese concentrations were somewhat higher than those shown in Figure 4. From 1973 to 1979 the distribution of soluble manganese was similar to that shown in Figure 5 with a slight trend toward lower concentrations in successive years. Soluble iron concentrations for 1979 and 1980 are shown in the same format in Figure 6. The dotted lines in Figures 4, 5, and 6 indicate the 0.5 mg/L dissolved oxygen isopleth.

The appearance of elevated concentrations of soluble iron and manganese coincides with the initial development of low dissolved oxygen concentrations in the upstream hypolimnion of the reservoir. Elevated concentrations of iron and manganese gradually progress downstream in the hypolimnion of the reservoir, ultimately reaching the portion of the reservoir represented by Stations 1 and 4. The observation that elevated soluble iron concentrations lag behind elevated soluble manganese concentrations is consistent with the redox chemistry of these two elements.

The fact that manganese reduction and, thus, solubilization occur at higher redox potentials than iron reduction also explains why elevated concentrations of manganese were observed in the downstream section of the reservoir in 1979 and 1980 while there was no significant accumulation of iron observed in this same region. Although dissolved oxygen concentrations were below 0.5 mg/L in the downstream section of the reservoir, apparently, the redox potential was not reduced enough to permit significant iron reduction. The accumulation of soluble iron in the downstream section of the reservoir was observed in 1971

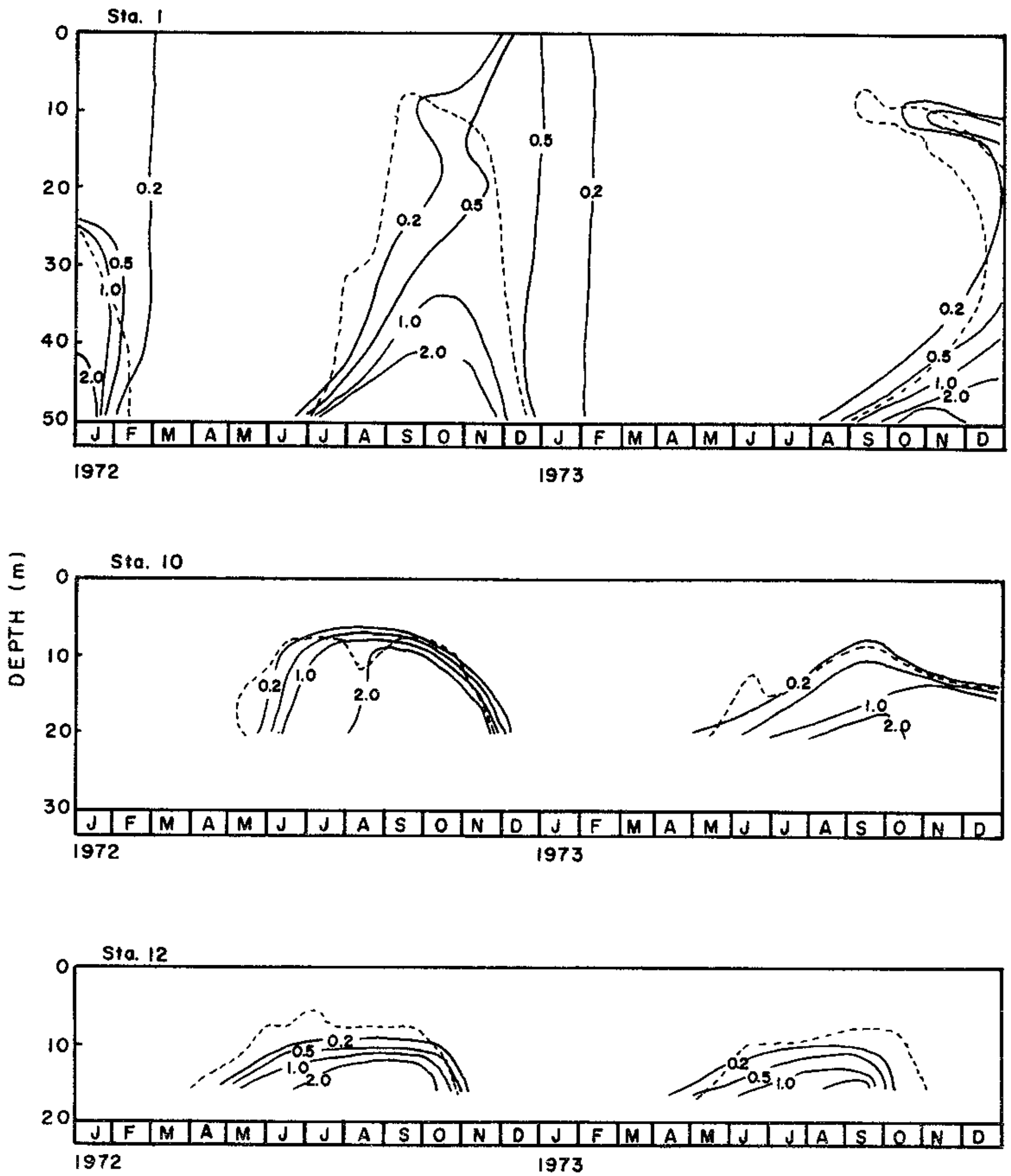

Figure 4. Soluble manganese (mg/L) at Station 1, 10, and 12 for 1972 and 1973.

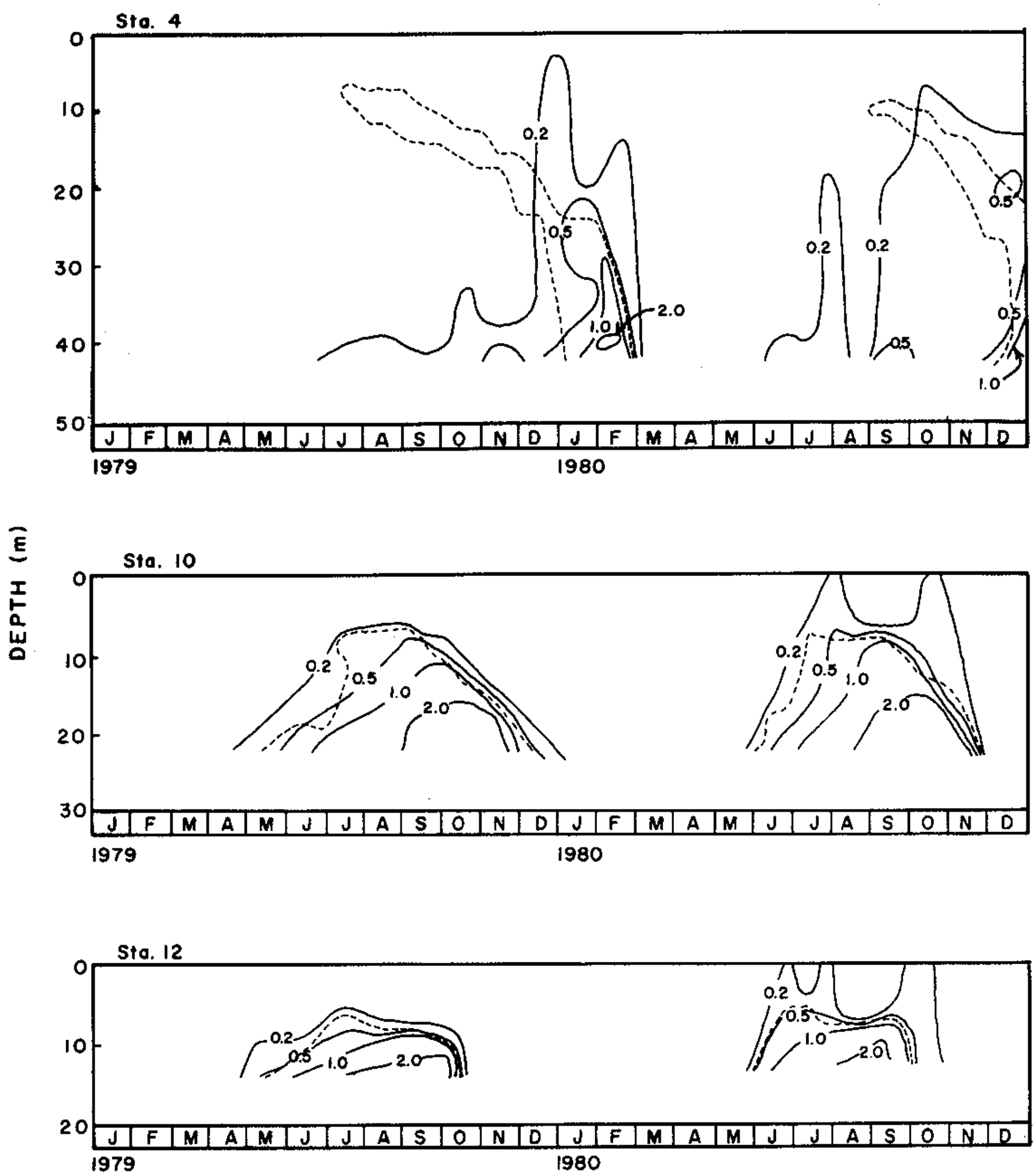

Figure 5. Soluble manganese (mg/L) at Station 1, 10, and 12 for 1979 and 1980.

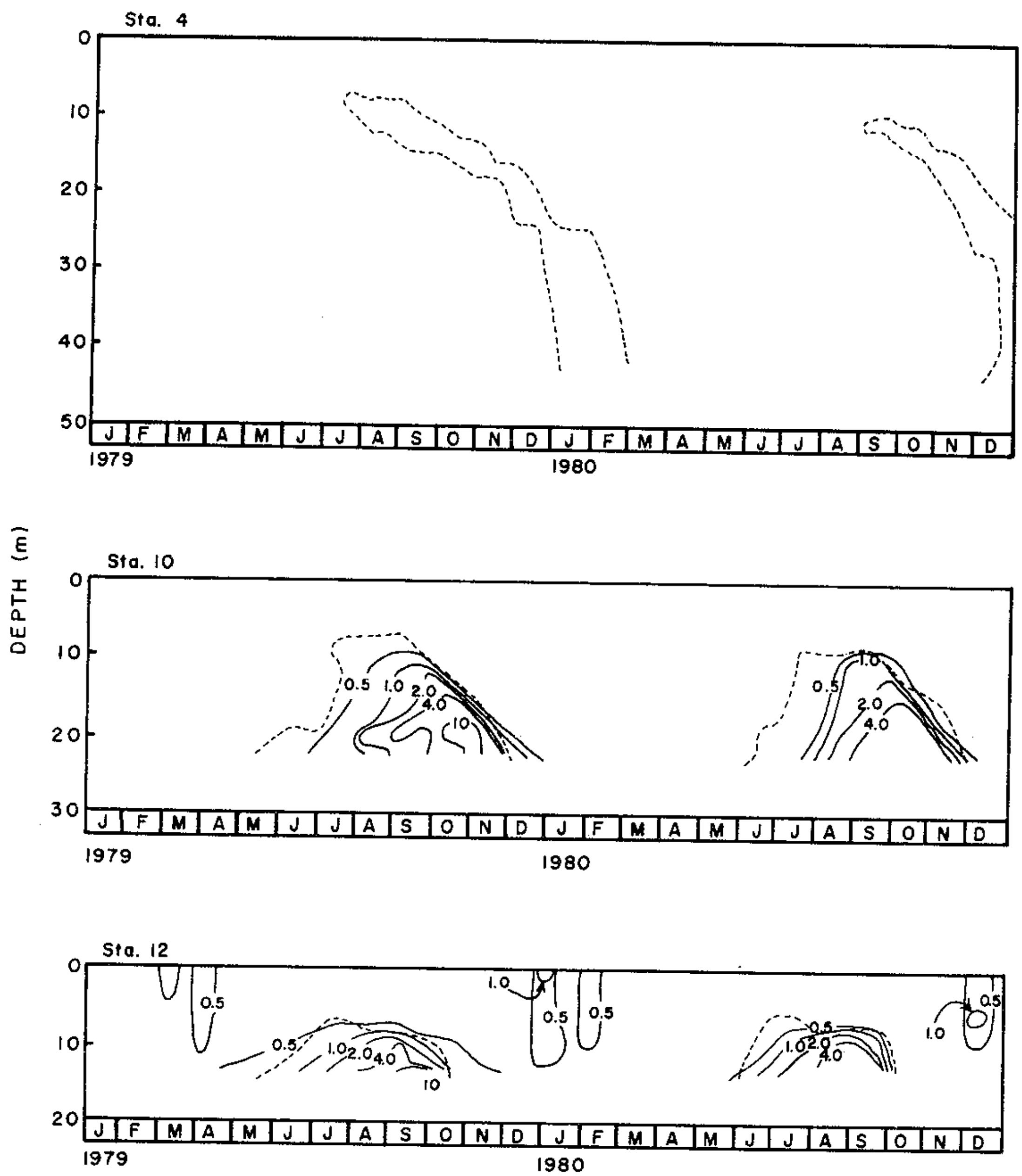

Figure 6. Soluble iron (mg/L) at Station 1, 10, and 12 for 1979 and 1980.

and 1972 but was not observed again through 1981.

Elevated concentrations of both iron and manganese persisted until oxygen was introduced by mixing. The dissolved oxygen and temperature profiles indicated the mixed layer depth was relatively uniform from the upstream portion of the reservoir to the downstream section. With thermocline erosion in fall and early winter, oxygen is introduced into the shallower upstream areas earlier than at downstream locations. Consequently, oxidation, precipitation, and subsequent settling of iron and manganese from the water column occur fairly early at the upstream station while significant concentrations of iron and manganese remain at more downstream locations. For example, elevated concentrations of iron and manganese were removed from the water column at Station 12 by mid-October 1979 while significant concentrations still existed at Station 10 (see Figures 5 and 6). In fact, accumulation of elevated concentrations of iron and manganese did not begin at Station 4 (downstream section of the reservoir) until almost complete mixing had occurred at Station 10 (mid-stream section of the reservoir). The general pattern for the distribution of iron and manganese in DeGray Reservoir is conceptualized in Figure 7.

Davison et al. (1982) has suggested that the origin of dissolved manganese in the water column of a stratified lake may be in situ dissolution of settling particulate manganese. The data in Figures 4, 5, and 6 show that increases in the concentration of soluble iron and manganese coincided with the depth at which dissolved oxygen first reached 0.5 mg/L. This strongly suggests that solubilization and migration from bottom sediment is the primary source of both iron and manganese. Of particular note is the manganese data taken at Station 4

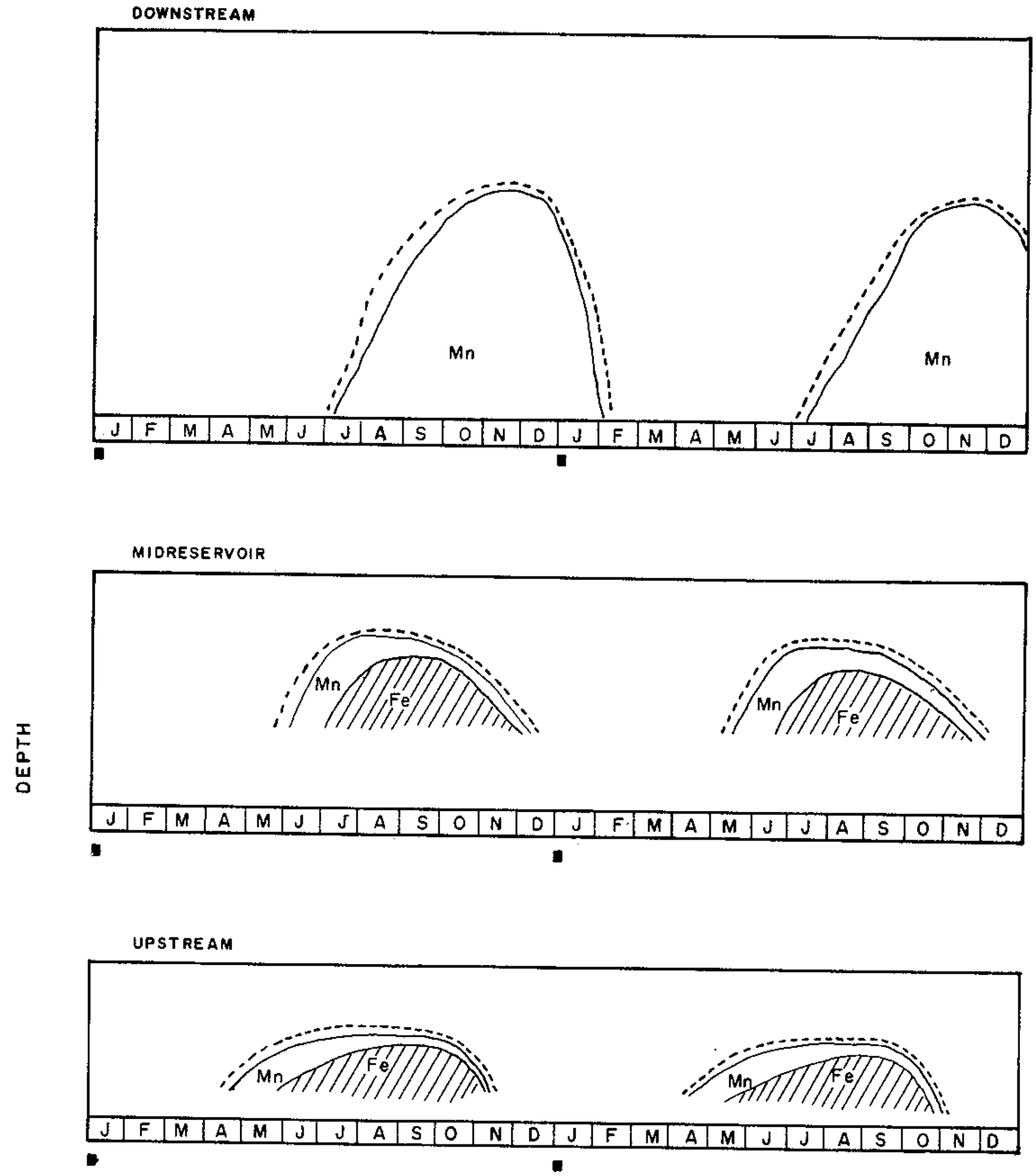

Figure 7. Conceptualized distribution of elevated concentration of iron and and manganese in three regions of a reservoir. Dotted line indicates 0.5 mg/L dissolved oxygen isopleth.

during the stratified period in late 1979 and early 1980 (see Figure 5). Although low dissolved oxygen conditions (less than 0.5 mg/L) existed throughout the late summer and fall of 1979, an increase in the manganese concentration was not observed until the low dissolved oxygen zone (as defined by the 0.5 mg/L isopleth) developed near the bottom of the reservoir. Manganese concentrations, then, increased until mixing occurred at Station 4 in late February. It thus seems likely that most of the manganese which migrates throughout the anoxic hypolimnion originates in the bottom sediment rather than from in situ reduction of settling particulate manganese.

As shown in Figure 6, slightly elevated concentrations of iron were observed in the mixed layer at Station 12 (upstream section of the reservoir) during 1979 and 1980. Concurrent increases in manganese were not observed during these periods. An examination of flow records of the Caddo River upstream from DeGray as well as other water quality information collected at these stations clearly show that these periods of elevated iron concentrations coincide with the introduction of storm water into the upstream section of the reservoir. Although iron in the fraction passing through a 0.45-micron filter is operationally defined as "soluble," the elevated iron concentrations observed during these periods probably reflect colloidal iron being transported during periods of high flow. Organically chelated iron entering the reservoir during storm periods cannot be ruled out; however, these anomalous concentrations of iron quickly dissipate after the storm period suggesting sedimentation as a removal mechanism.

Nix (1981) showed that the concentration of soluble manganese in the upstream metalimnetic region of DeGray was often elevated during the late period of stratification. Figure 8 shows dissolved oxygen concentrations (dotted lines) and soluble manganese concentrations (solid line) observed at Station 4 (downstream section of reservoir) from September 1979 through February 1980. The metalimnetic dissolved oxygen minimum was well developed by early September and persisted until deepening of the mixed layer intruded into the metalimnion in December. Slightly elevated concentrations of soluble manganese were observed at depths coinciding with the metalimnetic dissolved oxygen minimum as long as it existed. After disappearance of the metalimnetic dissolved oxygen minimum, the layer slightly higher in dissolved manganese was observed just under the lower extent of the mixed layer.

Although in situ reduction of settling particulate manganese in the low oxygen metalimnion (Davison et al. 1982) cannot be eliminated as a source of this dissolved manganese, these observations are also consistent with a model of advective transport as summarized in Figure 9. As shown in Figures 4 and 5, elevated concentrations of manganese first appear in the upstream hypolimnion of DeGray. As the anaerobic zone progresses in a downstream direction, soluble manganese increases in the deeper portion of the hypolimnion. At mid-reservoir (Station 10), low dissolved oxygen concentrations in the deeper portion of the hypolimnion allow the accumulation of soluble manganese with a classical distribution showing an increase in concentration toward the bottom. Above the layer containing elevated manganese concentrations is a portion of the hypolimnion which contains moderately high dissolved oxygen concentrations even in late fall. This prevents the further upward migration of reduced manganese. But soluble manganese again appears in the low dissolved oxygen metalimnion. The

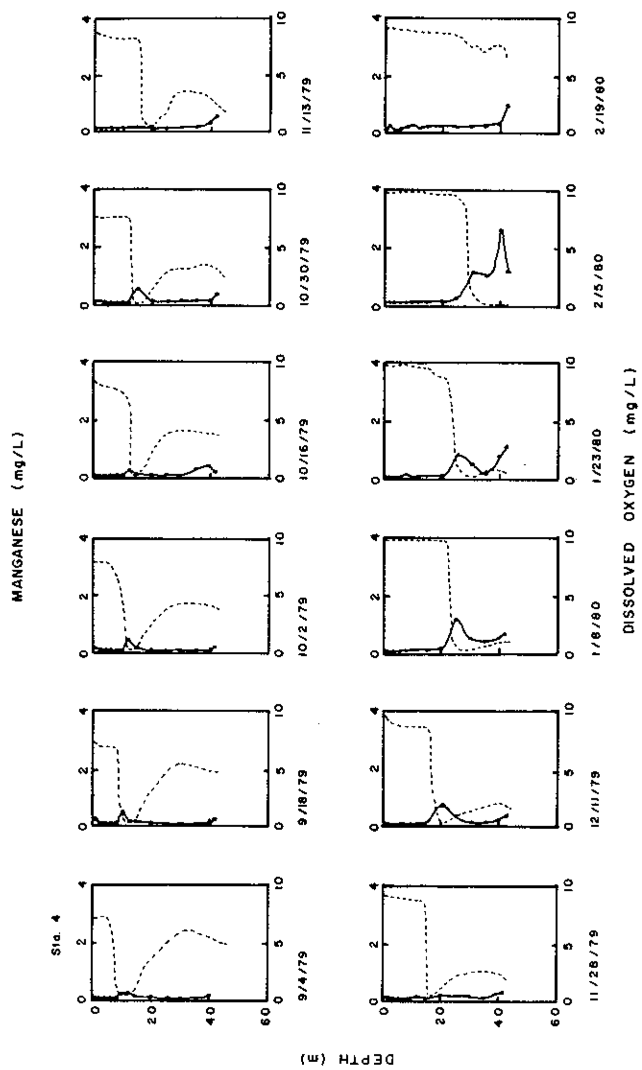

Figure 8. Dissolved oxygen (dotted line) and soluble manganese (solid line) at Station 4 (downstream section of reservoir) from September 1979 through February 1980.

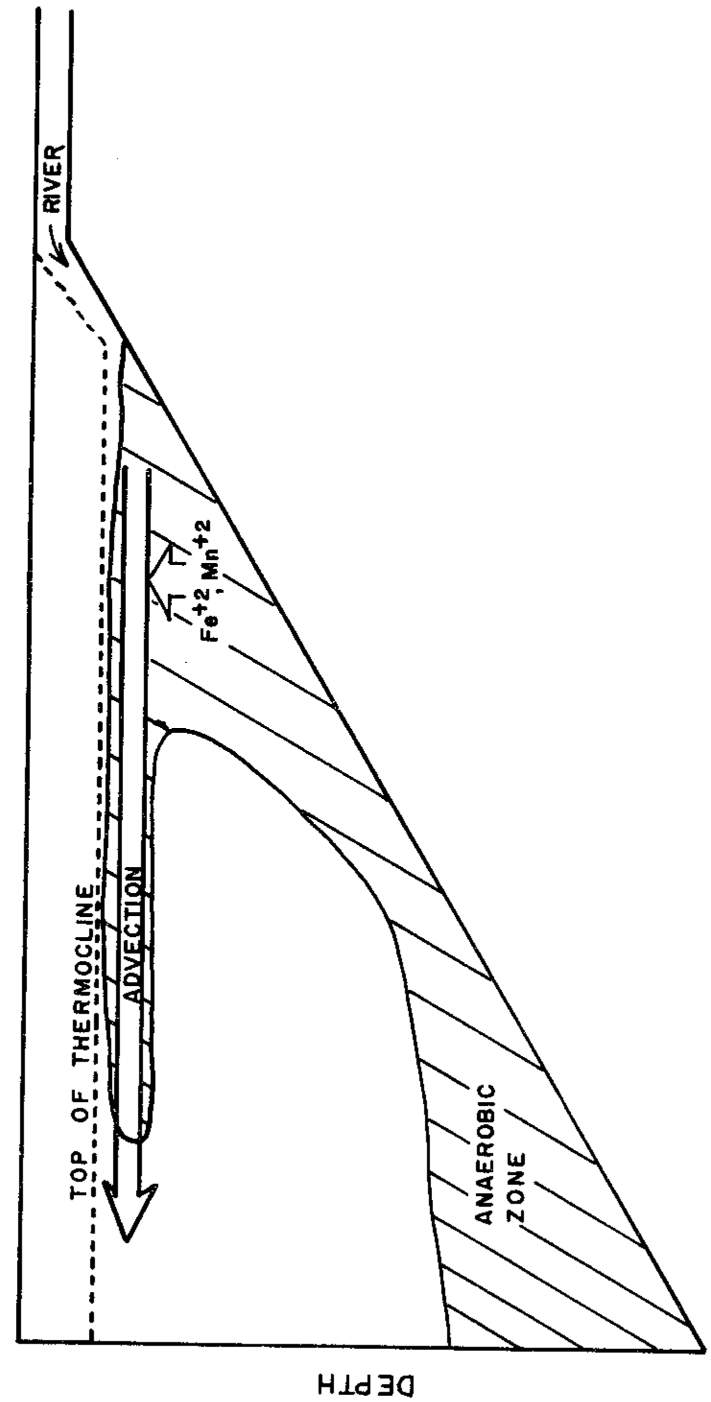

Figure 9. Conceptualized transport of upstream hypolimnetic water into the downstream metalimnion.

separation of the deep layer containing elevated manganese concentrations from the low dissolved oxygen metalimnion by a layer containing moderately high dissolved oxygen concentration is even greater in the downstream section of the reservoir due to increased water depth. Even with this additional separation, small increases in soluble manganese were observed in the metalimnion at Station 4 (see Figure 8).

Advective transport within restricted layers of Lake Powell were reported by Johnson and Merritt (1979). High turbidities in the metalimnion of DeGray have been observed following storms which occur during the stratified period (unpublished data). Nix (1981) suggested that even during periods of low flow, density currents persist near the top of the metalimnion in DeGray. This work also showed that the elevation at which increased metalimnetic concentrations of manganese were observed coincided with the depth at which the temperature of the reservoir was the same as the inflowing Caddo River.

Ford (this volume) has demonstrated the presence of density currents in DeGray using dye tracer techniques.

Assuming that advective transport is occurring in a restricted layer of DeGray near the top of the metalimnion, even during the low flow summer and fall period, it seems likely that the movement of this density current would intersect that portion of the upstream hypolimnion containing low dissolved and possibly reduced iron and manganese and entrain these reduced species into the density current. Subsequent downstream movement of this water throughout the remaining period of stratification would result in the downstream transport of these reduced species in the metalimnetic region. The data in Figure 8 suggest that during the late portion of 1979 and early 1980, advective transport resulted in elevated concentrations of manganese even in the downstream section of the reservoir (Station 4).

The advective transport mechanism suggested in Figure 9 would result in the preferential transport of manganese over iron. As shown in Figures 4, 5, and 6, and summarized in Figure 7 the upward extent of soluble manganese is greater than soluble iron during the stratified period. This would allow the density current to "strip" soluble manganese from the upstream hypolimnion and transport it downstream without the concurrent transport of iron. The result of such a process over an extended period of time would be for the sediment in the downstream section of the reservoir to become enriched in manganese relative to iron. Gunkel et al. (1984) observed that the sediment collected in the downstream section of DeGray Reservoir had higher manganese concentrations as compared to sediment taken from the upstream section of the reservoir. The transportation of significant quantities of iron by this mechanism would only occur during years where stratification persisted over a longer period than is usual.

Summary

Although the cycling of iron and manganese in DeGray can generally be explained by the same processes that occur in most seasonally anoxic lakes, it is necessary to impose processes which seem to be characteristic of deep reservoir systems to adequately explain the distribution of these metals in DeGray Reservoir.

The solubilization of iron and manganese clearly begins in the upstream section of the hypolimnion and progresses into the downstream hypolimnion throughout the period of stratification. As expected, manganese generally appears first and extends further into the reservoir than iron.

The observation that elevated concentrations of these metals do not generally begin to appear in the water column until the low dissolved oxygen zone develops near the bottom strongly suggests that the origin of both iron and manganese is bottom sediments.

Advective transport by a density current near the top of the metalimnion results in the entrainment of upstream hypolimnetic water and ultimate transport of reduced species into the downstream metalimnion of the reservoir. This process can result in the preferential transport of manganese over iron through the reservoir system. The presence of elevated concentrations of reduced species in the metalimnetic regions of reservoirs should be given consideration in the design of the intake structures to be used by municipalities and industries.

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LONG-TERM PHYTOPLANKTON SUCCESSION IN DeGRAY RESERVOIR (1975-1980)

by

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ABSTRACT

Annual and long-term phytoplankton succession studies on DeGray Reservoir included a regular collection program in each of the reservoir's major compartments from 1975 through 1980. Samples were collected in association with various chemical, physical, and biological studies. The data indicate distinct, but independent, temporal distribution patterns for the major taxa. Critical period analysis shows that the successional dynamics for each lake compartment were relatively distinct from adjacent compartments during the early phases but tended toward greater homogeneity. The structure of the phytoplankton assemblage as defined by the number of species per sample is described temporally for each compartment. A distinct trend from a diverse assemblage to a major single taxon dominance was observable for the summer maximum, fall transition, and to a lesser extent during the winter. The data indicate a reduction in qualitative variability with an increased abundance throughout the 5-year sampling period.

INTRODUCTION

Long-term studies of annual phytoplankton succession on larger freshwater lakes have been limited to certain European lakes, such as Lake Windemere (Macan 1970) and others in Denmark, Sweden, Germany, etc., and in the United States-Canadian great lakes. A recent study by Hopkins and Lea (1983) reviewed the phytoplankton composition and biomass during a 10-year study in Lake Erie. Also several multiyear studies have been conducted on Lake Michigan by the Great Lakes Research

Center. Similar long-term analyses have been conducted for selected reservoirs in the Tennessee Valley Authority system: Nickajack and Chickamauga Reservoirs by Placke and Poppe (1980); Guntersville Reservoir (Poppe, Placke, and Wright 1982); and other reservoirs by additional researchers within the TVA system. In the immediate region, a smaller reservoir, Lake Fayetteville, has been studied by Meyer (1971, 1973, 1974), Rice (1978), and Poppe (1978) for seasonal succession, macro- and micro-nutrients, and long-term forecasting associated with the phytoplankton. Multiyear studies of phytoplankton succession on large reservoirs in the adjacent area are limited to Beaver Lake by Meyer (1973 and 1974). A review of the literature suggests that the number of reservoirs with extended, multiyear analyses of annual standing crops of phytoplankton, seasonal succession, and long-term trend analysis is limited.

In this paper a review of the long-term trends in total phytoplankton standing crop abundance, changes in taxonomic composition, and a detailed analysis of the annual succession in the three principal compartments of DeGray Lake are described. The annual successional events subdivide the phytoplankton into the primary taxonomic components which collectively comprise greater than 95 percent of the standing crop. In addition, the changes in the seasonal abundance of the phytoplankton are presented to estimate or forecast the anticipated future trends.

RESEARCH SITE DESCRIPTION

DeGray Lake, a multipurpose reservoir with pump-back capability, is located in the Ouachita Mountains near Arkadelphia, Arkansas. The 12-year-old reservoir has a maximum surface area of 6,883 ha with a shoreline of 362 km and a drainage basin of 117,377 ha. It is ca. 30 km in length with a maximum depth of 60 m and a mean depth of 14.9 m.

The reservoir contains three major compartments (Figure 1). The compartments include several sampling sites; however, only four were included as long-term stations. The near-dam compartment extends from the dam through station 7 with station 4 as the representative site.

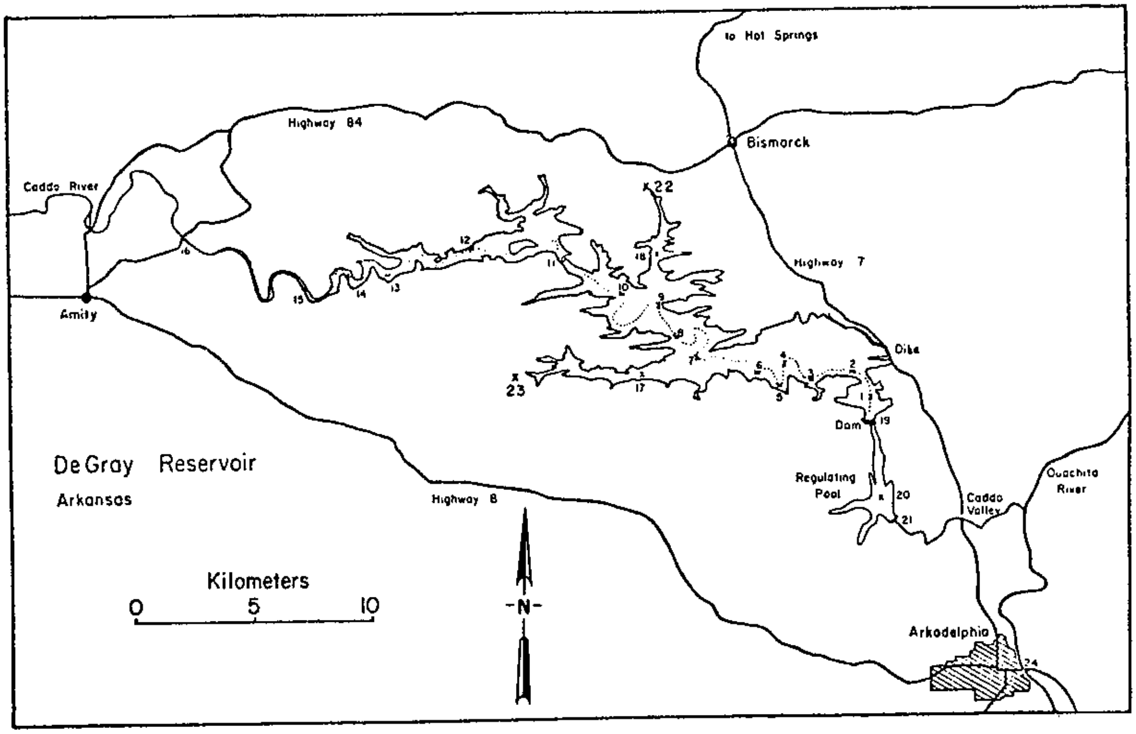

Figure 1. Sampling station previously established on DeGray and watershed. Stations 4, 10, and 12 are the representative stations for the "dam," "midlake," and "river" compartments, respectively

The middle compartment begins at station 7 and is terminated by the constriction at station 11. Station 10 is used as the standard section for this compartment. The riverine or upper compartment extends uplake until station 13 or 14. This latter compartment is represented by station 12.

MATERIALS AND METHODS

Aliquots of the samples collected for the various chemical, physical and biological studies were preserved with M_3 fixative (Meyer 1971). Ten milliliters of the preserved samples was settled in counting chambers and enumerated with the aid of a Wild M-40 phase-contrast inverted microscope.

RESULTS AND DISCUSSION

The results and discussion of the research are presented in three phases. Phase I reviews the several principal classes of algae which comprise 95 percent of the standing crop from June 1974 through November 1976. Phase II summarizes multiyear trends based upon selected seasonal events within each of the compartments. The data upon which Phase I and II are based are summarized in Table 1. These data include the summer, fall, and winter seasonal phases of the annual cycle for each of the lake compartments. The first value is the number of cells per

Table 1

Summary of the Mean Number of Cells per Liter × 10^3 and Taxa by Compartment

DATE		COMPARTMENT		
Month	Year	Dam	Midlake	River
August	1975	181(12)	289(12)	224(19)
October	1975	301(11)	283(10)	330(14)
December	1975	243(14)	309(10)	706(17)
August	1976	184(11)	352(18)	466(21)
October	1976	115(11)	56(10)	339(23)
February	1977	11(2)	297(7)	58(5)
July	1977	509(12)	362(9)	986(13)
October	1977	80(8)	57(10)	345(23)
December	1977	63(8)	273(10)	303(15)
August	1978	431(15)	524(12)	983(17)
October	1978	394(18)	457(14)	490(13)
February	1979	43(6)	112(10)	96(7)
August	1979	1180(21)	456(16)	281(14)
October	1979	372(21)	456(16)	281(14)
February	1980	63(9)	108(13)	60(10)

liter × 10^3 (Phase I) while the values enclosed within parentheses are the mean number of cells per liter (Phase II). Phase III further reduces the data to overall reservoir trends during each of the major seasons for the time span 1975-1980.

The data for Phase I are summarized by compartment and depth for the several principal classes of algae which represent the primary components of the standing crop of phytoplankters. These classes include the Chrysophyceae, Cryptophyceae, Bacillariophyceae, Chlorophyceae, and Cyanophyceae. Finally, the total abundance of the phytoplankton is summarized for each compartment.

Chrysophyceae: Initially these algae were of minor importance with temporary subsurface appearance in the river and midlake compartments with a later reduced occurrence near the dam. Maximum concentration of the combined chrysophytes was ca. 2 million cells per liter. These taxa plus reduced numbers of cryptomonads and diatoms are the primary contributors to the winter flora.

Cryptophyceae: The brown flagellates are present throughout the annual cycle. These organisms have become increasingly important as the reservoir ages and have extended their presence throughout the water column and into each compartment. Only during the autumn are their numbers reduced. The maximum concentration was ca. 1.2 million cells per liter.

Bacillariophyceae: The diatoms were early significant contributors to the standing crop in the dam compartment. Collectively, these plankters express a bimodal distribution with summer and winter maxima. *Fragilaria* is the dominant diatom in the summer while *Melosira* and *Asterionella* attain their maxima during the winter. These taxa later expand into the midlake and riverine compartments. The maximum concentration is present in the upper epilimnion of all compartments. Populations of less than 1 million cells per liter were encountered.

Chlorophyceae: The diverse populations of coccoid and capsoid green algae attained their major abundance from spring through late summer with reduced concentrations in early autumn. A resurgence may develop in midautumn to late autumn before declining to the winter minimum. These populations tended to be nearly evenly distributed throughout the epilimnion in all compartments. The maximum concentration of the assemblage of green algae was less than 1 million cells per liter.

Cyanophyceae: The blue-green algae were sporatic in occurrence during the early phases of the research but increased in importance. The blue-greens were predominant in the midsummer to late summer and early autumn in each of the compartments, with the greatest initial concentration near the dam. The maximum abundance of this assemblage was greater than 9 million cells per liter.

The combined or total abundance of phytoplankton follows independent seasonal cycles within each of the three compartments. Within the near-dam compartment there is a definitive unimodal distribution. The maximum is reached during the summer and the minimum in the winter, with spring and autumn intermediate populations. The maximum may approach 20 million cells per liter while the minimum is slightly less than 500 thousand cells per liter--a seasonal difference factor of ca. 40. The midlake compartment lacks a well-defined seasonality. Also there is greater vertical mixing and less stratification of species in this compartment. The midlake waters typically contain phytoplankton populations of ca. 5 million cells per liter; however, populations of twice this concentration may occur in the upper meter for brief periods in late August or early September. The riverine compartment is mixed except during low flow. In this compartment the late summer maximum of ca. 15 million cells per liter develops subsurface. This population initially expands toward the surface and then migrates to great depth prior to and in association with thermal destratification.

The second phase of the analysis is based upon the selection of critical periods in the annual succession for the expression of multiyear trends. The critical periods selected include the maximum during late summer, the onset of the autumnal decline, and the winter minimum. The vernal phase was not included since it is a highly variable period with an erratic initiation to the expansive growth phase leading to the summer maximum.

The data for this phase are expressed in two formats. One mode of expression is to examine the mean number of cells per liter. The second measure is an estimate of the diversity by comparing the variation in number of species present. The statistical analysis of diversity is presented in a parallel study by Dr. Wayne L. Poppe (included in this volume).

The mean number of cells per liter (cpl) data summarize the abundance of phytoplankters in the upper 10 m of the water column. As prior research has shown, this column represents the zone which contains the major portion of the total standing crop throughout the year. The long-term dynamics of each of the three compartments are presented separately from summer 1975 through winter 1980.

Dam: The multiyear seasonal variation in the mean number of plankters per liter is shown in Figure 2. This compartment is noted for its major differences between the summer maximum and winter minimum. The initial mean concentration was slightly greater than 300,000 cpl. In succeeding years the summer concentration increased so that by the end of the study the observed maxima were nearly 1.2 million cpl. In contrast, the winter minima remained at approximately the same concentration through time, ca. 90,000. The constancy of the winter population densities suggests that the limiting factors may be physical, i.e., light and temperature, rather than chemical.

Midlake: The seasonal dynamics of the phytoplankton for the midlake compartment are shown in Figure 3. The seasonal cycling of phytoplankton in this compartment has the typical sine wave-like pattern seen in the dam compartment. The cyclic components of these two populations, however, are not synchronized and have different amplitudes. The midlake compartment has less variation, with maxima progressing from ca. 300,000 cpl to ca. 500,000 cpl through time. The minimum also increases over time. The 1976 fall minimum was ca. 70,000 cpl while the later minima were nearly double this value. The initial minimal concentration occurs during the fall but later shifts to the winter. The winter minimum in the midlake compartment is greater than that in the

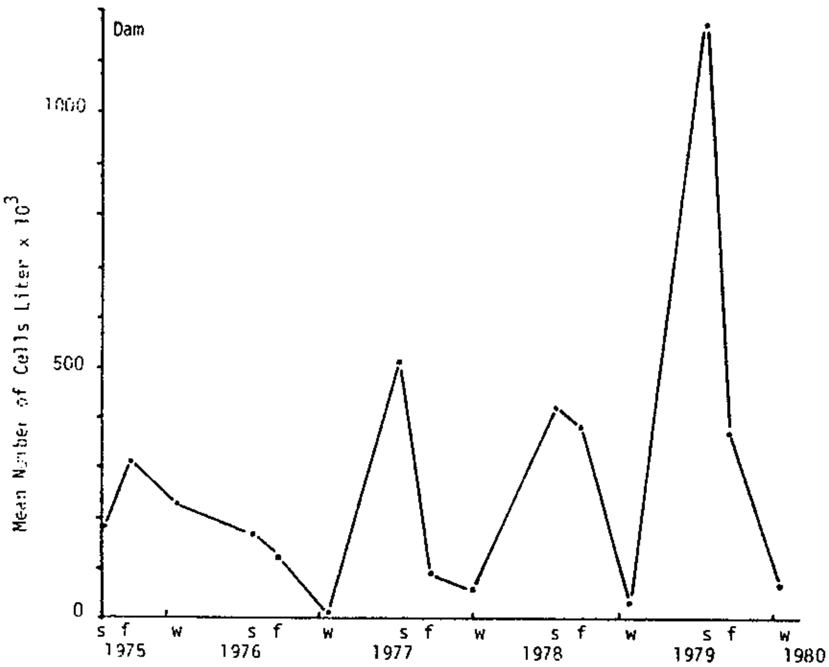

Figure 2. Long-term trends in the mean number of cells per liter within the dam compartment

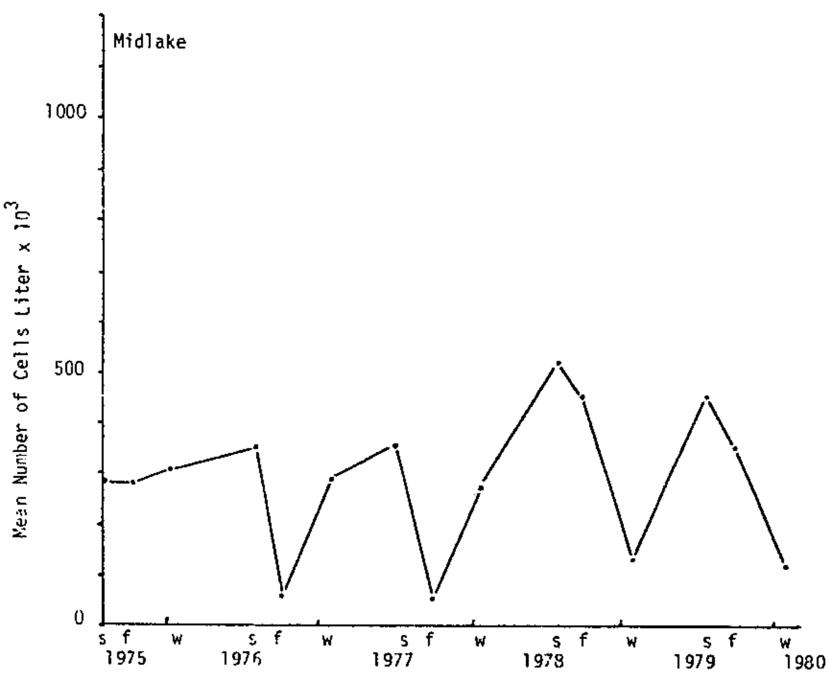

Figure 3. Long-term trends in the mean number of cells per liter within the midlake compartment

dam compartment by a factor of two. It should be noted that the maxima and minima remained relatively stable within a moderate dynamic range.

River: The river or uplake compartment is strongly influenced by storm events since the major portion of the watershed drains into the portion of the lake which has the minimum volume. Nutrient loading, volume dilution, and reservoir height, as well as a combination of these factors, have the greatest impact on the standing crop in the upper compartment.

In the river-influenced compartment the population concentrations are highly variable with summer maxima and winter minima (Figure 4). The exception to this general pattern was the large concentration in the winter of 1976 produced by transitory populations of cryptomonads and diatoms. The population concentrations are more highly variable in this compartment than in either of the other compartments. The maxima and minima are less predictable than either the dam or midlake populations. Initially, the summer maxima increased from ca. 700,000 cpl to greater than 1 million cpl; however, in 1979, the concentration decreased by a factor of 2. A similar shift is noted in the winter concentrations. The abundances in 1977, 1979, and 1980 are nearly equal at ca. 100,000 cpl, but the fall and winter standing crops of 1977 and 1978 contain ca. 400,000 cpl. This compartment has greater diversity in the number of taxa present and changes in the number of taxa. Thus, this region of the reservoir is the most highly dynamic, with great variation in the abundance, variety of species present, and changes in taxa through time. A certain portion of this dynamism is associated with the transport of organisms from the river and backwaters into the reservoir by storm events. During the most stable period (the summer), diversity in space and time remains high.

Associated with the change in abundance of organisms through time is the succession of taxa during the annual cycle and during the aging of the reservoir. The data reported in the figures given below were derived from the previous samples. The diversity of organisms present within each compartment over time is expressed as mean number of species in the 10-m water column.

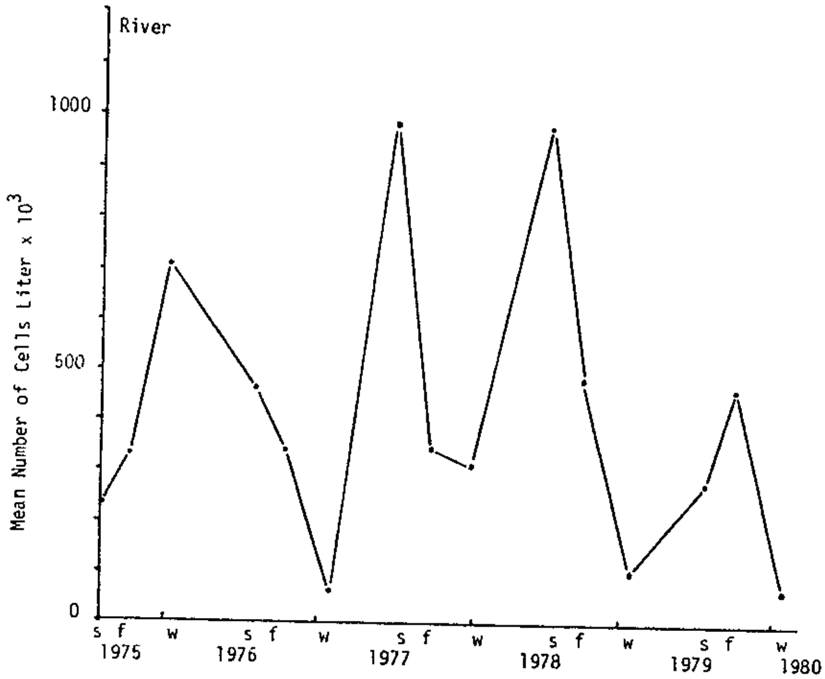

Figure 4. Long-term trends in the mean number of cells per liter within the river compartment

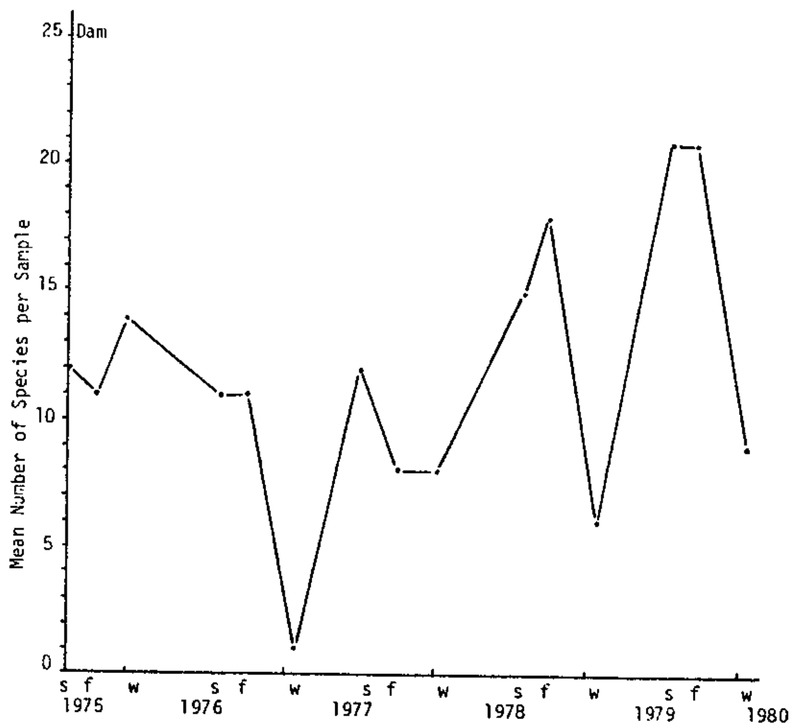

Figure 5. Long-term trends in the mean number of species per liter within the dam compartment

Dam: Within the dam compartment the diversity was seasonally dynamic with increasing diversity (Figure 5). Initially the diversity was high, with greater than 11 species per sample throughout the year. By 1976 a regular cyclic pattern developed, with winter minima and summer-fall maxima. The minimum was initially low, with a single dominant diatom in winter 1977 and increasing diversity in the later years to about 10 as a typical number of species per sample. The summer samples showed a parallel increase in diversity through time. The initial values were 11 to 12, followed by rapid increases to 22 in 1979. The increase in diversity is apparently associated with the introduction and establishment of new species during the maturation of the reservoir. It should be noted that the maximum number of species tended to occur in the fall during the transition period.

Midlake: The central compartment parallels the dam compartment but with dampened amplitudes (Figure 6). The initial number of species is approximately 11 in 1975 and early 1976. There is a temporary maximum in the summer of 1976 associated with the transport of uplake species into the midlake compartment by stormwaters. In general, there is a tendency for the maximum to occur during the transition between summer and winter floras followed by a moderate winter decline. There is a tendency of increasing diversity with greater amplitude between the maximum and minimum periods. Also, the minimum number of species is greater in this compartment than the dam compartment.

River: The uppermost compartment is the most dynamic with the greatest variability and diversity (Figure 7). Summer-fall values tend to be greater than 20 species per sample and increase to greater than 25. As with the previous compartment, the greatest diversity develops in the fall in the later samples. The maximum number of species is greater in this compartment (26) than the former compartments; however, the winter minimum is intermediate between the dam and midlake sectors. Both the maxima and minima values indicate a general trend of increasing diversity over time.

The third phase summarizes the data by further reduction into a composite of the predominant phytoplankton taxa, by major time period,

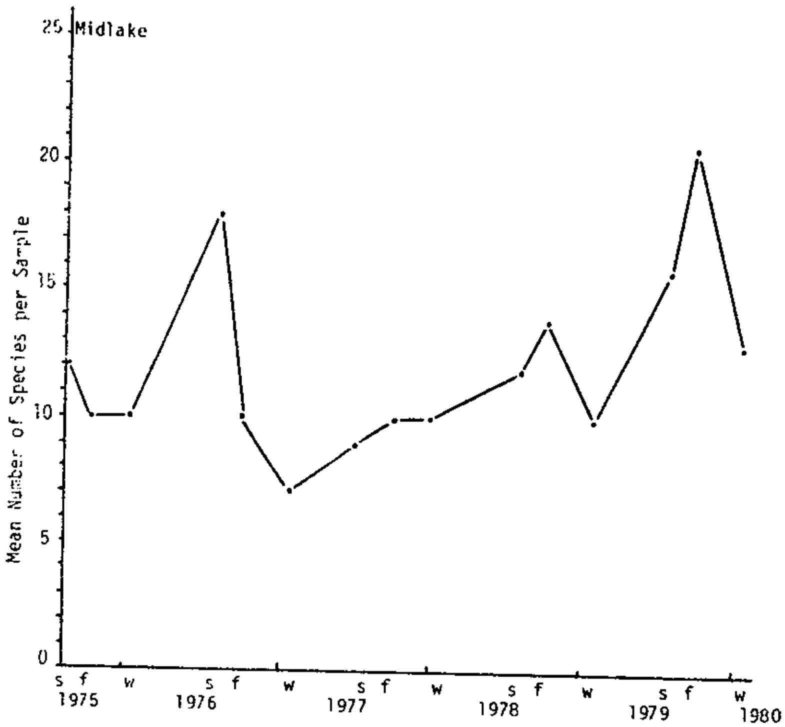

Figure 6. Long-term trends in the mean number of species per liter within the midlake compartment

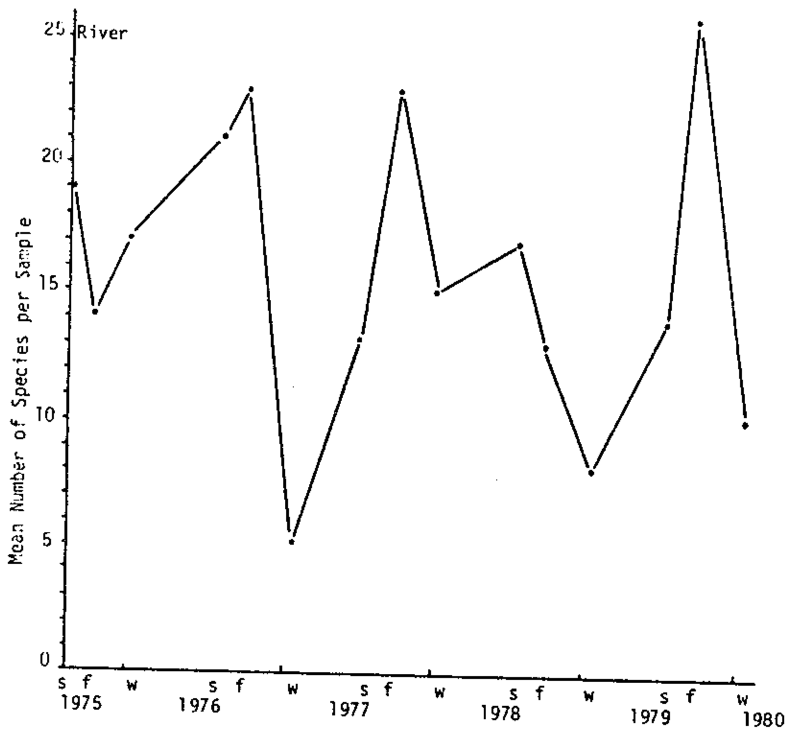

Figure 7. Long-term trends in the mean number of species per liter within the river compartment

within each of the three major compartments. Table 2 consolidates the data by combining the taxa which constitute 95 percent or greater of the standing crop into an overview of the reservoir dominant assemblage for each of the seasons.

Table 2

Summary of the Dominant Phytoplankton Taxa by Season Through Time

Date	Dominant Phytoplankton Taxa
Summer	
1975	Greens, cryptomonads, diatoms, and blue-greens
1976	Greens, blue-greens, and cryptomonads
1977	Blue-greens and greens
1978	Blue-greens
1979	Blue-greens
Fall	
1975	Cryptomonads, greens, and diatoms
1976	Greens, diatoms, blue-greens, and cryptomonads
1977	Greens, blue-greens, diatoms, and cryptomonads
1978	Blue-greens
1979	Blue-greens
Winter	
1976	Greens, cryptomonads, and diatoms
1977	Diatoms
1978	Diatoms and cryptomonads
1979	Diatoms
1980	Diatoms and greens

During the first summer (1975), the green algae were the major component in the plankton in the dam and river compartments. Of secondary importance in all compartments were the flagellated cryptomonads, with diatoms important in the near-dam compartment. The blue-green algae were of tertiary and primary importance in the dam and midlake compartment, respectively. In the summer of 1976 the green plankters were primary in importance at the dam and midlake stations and of secondary importance in the upper reservoir stations. The blue-green algae were inverse in their distribution, i.e., of primary importance in the river compartment and of secondary importance in the remaining sectors of the reservoir. Cryptomonads were present but in small numbers near the dam and as tertiary contributors in the other regions of the reservoir. By 1977 the diversity had been further reduced so that the plankton standing crop was dominated by blue-green taxa with secondary contributions of diatoms near the dam and greens near the river entrance. The trend toward reduction in diversity continued through years 1978 and 1979. The blue-green phytoplankters dominated at all stations throughout the reservoir with only minor contributions of green algae at all sites and a few diatoms near the dam during 1978. These additional taxa were unimportant in 1979.

The initial fall (1975) populations were dominated by cryptomonads, coccoid greens, and diatoms. In 1976 the greens continued to be of primary importance at the two terminal compartments with blue-greens of greater importance in the middle sector. The diatoms remained as subdominants in the larger compartments while cryptomonads are abundant in the river compartment. The midlake water contained a more diverse population with not only bluegreens and diatoms, but also coccoid greens and cryptomonads. Bluegreen algae were of minor importance in the river compartment. In fall 1977, cryptomonads were of secondary importance in all compartments with blue-greens predominant near the dam and coccoid greens in the remainder of the reservoir. Blue-greens were tertiarily important in the two upper segments. Dinoflagellates and diatoms were minor contributors in the dam and river compartments, respectively. By

1978 and 1979 the blue-green algae were clearly the predominant forms with an admixture of taxa associated with these populations.

Winter standing crops were typically composed of fewer taxa and cells. The composition present in 1976 was an admixture of green algae, cryptomonads, and diatoms throughout the reservoir. The cryptomonads and diatoms were present in nearly equal numbers at all stations, but the greens were of primary importance at the near-dam station and diminished progressively in concentration and importance uplake. During winter 1977, the diatoms constituted greater than 95 percent of the taxa present at all stations. These taxa were major contributors to winter populations in 1978 along with the cryptomonads. The cryptomonads had their greatest concentration near the dam and decreased in numbers further uplake. Thus, two interrelated gradients were present: diatoms increasing and cryptomonads decreasing in importance with increasing distance from the dam. In winter 1980, the cryptomonad contribution was insignificant and the diatoms were predominant, with a limited presence of coccoid green algae.

SUMMARY

In general, the trend during the summer and fall seasons was a gradual reduction in the diversity of major taxa present. In addition, there was a tendency toward predominance by the blue-green algae. The blue-green algae were, therefore, increasing in importance in space and in time. These organisms expanded from their early dominance in the summer-midlake position to the role of predominant plankter in all compartments during the entire summer and fall seasons. Similarly, the winter assemblage tended to be diatom-dominated with minor annual variation in the relative frequency of other contributors.

Associated with the trend toward reduction in major taxa has been an increase in abundance and number of species. The abundance data suggest that the reservoir is tending toward increasing fertility, and thus supporting a greater number of cells per liter. Concomitant with the increase in abundance is an increase in number of species present.

These species, however, are from a reduced breadth of algal classes. These data suggest that species are not replaced on a one-to-one basis, but rather that more than one species may replace the lost taxon and that the replacement taxon will be from the major algal class in the direction of succession.

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USE OF EMPIRICAL PHYTOPLANKTON INDICES FOR PRETROPHIC STATE ANALYSIS

by

Wayne L. Poppe

ABSTRACT

This paper is intended to examine and qualify one facet of pre-eutrophication analysis: phytoplankton indices. Sorenson's quotient and Pielou's percent similarity index are used to describe similarities and differences within and among stations, both temporally and spatially. The results of these analysis indicate that very few similarities exist within DeGray Reservoir, which implies that these analyses alone are of little use in describing the phytoplankton quality of the reservoir. These analyses do, however, merit use when coupled with analyses (presented in papers of these proceedings) and suggest that a modification of plankton sampling will be required to obtain "average" conditions for modeling purposes.

INTRODUCTION

The word "eutrophic" was first used in the literature by Weber in 1907 to designate nutrient-rich layers within bogs. In a limnological sense, eutrophic means "well fed," indicating an abundance of life-supporting nutrients. The word eutrophication is now commonly used to describe the response of water bodies to nutrient (including organic) enrichment.

Eutrophication is a natural aging process by which lakes become more biologically productive (until eventually aquatic plants cover the entire surface of the lake, and the basin fills in with detritus and sediments). Rivers and reservoirs with short hydraulic retention times do not age in the same sense as lakes, but added nutrients may increase their productivity above desirable levels. "Cultural eutrophication" results when anthropogenic activities (e.g., release of wastes of municipalities and industries into a body of water or runoff from heavily fertilized farmland) accelerate the natural aging process. As enrichment proceeds, the composition and abundance of the biotic assemblage change. Massive growths of algae, called "blooms," occur periodically and under extreme conditions may turn the water a pea-green color. The most typical offenders are the blue-green algae, primitive bacteria-like organisms that may produce noxious floating scums or release toxins which kill fish and other animals. Massive growths also tend to deplete the life-supporting oxygen in the water as they decompose. Less dramatic growths of these and other algal organisms may produce taste and odor problems in municipal water supplies, increase water treatment costs by causing filter clogging, and impart less palatable taste to fish caught in the area. Fish yield often increases under moderatively productive or eutrophic conditions, but as eutrophication proceeds sport fish populations are reduced and replaced by rough fishes, and aquatic weeds and algae become more abundant in shallow areas--potentially interfering with boating, fishing, swimming, and aesthetics of the water. It is not certain that the biotic changes occurring during cultural eutrophication parallel the changes that would occur during normal lake and reservoir maturation. The very rapid changes during cultural eutrophication may give rise to such poor water quality that the designation "polluted" is warranted.

It is generally accepted that cultural eutrophication results from the addition of excessive amounts of nitrogen and/or phosphorus to lake and reservoir systems. The possible causal roles of complex organic compounds, vitamins, and minerals (which may also be abundant in wastewater) have been defined but have been demonstrated to be of importance in only a few cases.

Management of accelerated eutrophication has focused on nutrient removal from municipal and industrial wastes; diversion of waste to alternate water bodies; management of nonpoint sources; or symptomatic relief through oxygenation of water, mechanical removal of excessive aquatic weeds, and the application of herbicides to control algal and macrophyte growth. Though there is some question whether the process of eutrophication is ultimately reversible, marked improvements in water quality have occasionally been noted when remedial measures such as sewage diversion, nutrient removal, or nutrient inactivation were applied. The rate of recovery may be relatively rapid (on the order of a few years), but may also occur very slowly as nutrients continue to be released from the basin sediments for years after excessive nutrient enrichment has ceased (Wisconsin Department of Natural Resources 1974).

Several techniques are available to assess eutrophication and the relative trophic classification and quality of reservoirs. These include: (1) comparison of established trophic indicators with other reservoir systems, (2) multivariate statistical analysis of water quality indicators, (3) measures of biological productivity or response to nutrient enrichment, and (4) remote sensing imagery analysis. Although each technique merits use in eutrophication analysis, a single analysis is often inadequate to evaluate alternative management practices. A better evaluation is obtained by combining the results of each of the available analytical techniques.

Comparative data, for example, can provide a relative reservoir trophic classification, but the evaluation may be biased if the variability between or within reservoirs is limited. Nutrient ratios and empirical nutrient load-reservoir response models, using inflow, outflow, and in-reservoir nutrient concentrations, can be useful supplements to these data. The value of such techniques in eutrophication analysis is limited, however, in cases where the nutrients are unavailable for biological productivity. Many reservoirs in southern states have been classified as eutrophic by empirical nutrient load-reservoir response models (e.g., models developed by Vollenweider, Kirchner, and Dillon); yet, the biological productivity expected from these loadings is not observed. This phenomenon can be explained by examining the nutrient species present. A large percentage of the nutrients within these reservoirs, as well as those entering and leaving the reservoirs, are biologically unavailable. Therefore, biological response indicators present a more sensitive analysis than measurements of nutrient concentrations. Algal assay results and measurements of chlorophyll, standing crop, and productivity may actually be more reliable indicators of trophic status than observed nutrient concentrations.

Descriptive statistical analysis may provide additional insight to classification problems. In the case where data do not correlate with predictor variables, methods such as empirical indices or principal components analysis may extract information from the data set that would otherwise be overlooked.

In the case where a data base for eutrophication analyses is either limited or nonexistent, remote sensing techniques may provide an alternative method of relative trophic classification. LANDSAT imagery is now being used by several State and Federal agencies to complement field and laboratory trophic status measurements and to monitor reservoirs that are not being monitored manually.

This paper presents the use of accepted plankton indices to describe similarities among and within stations. Additionally, the use of these analyses as related to eutrophication analyses is presented.

METHODS AND MATERIALS

Similarity of algal communities within and among reservoir sections was determined using a two-step approach. Sorenson's Quotient of Similarity, SQS (McCain 1975) was first calculated to determine similarities based solely on presence/absence of genera (qualitative characteristics). Next, a percentage similarity (PS) index (Pielou 1975) was calculated to determine similarities based on both qualitative and quantitative characteristics. In both cases, values of 60 to 70 percent or greater were assumed to show similarity.

SQS was calculated as follows:

$$SQS = 2s/(x + y) \cdot 100$$

where

x = number of taxa at station x

y = number of taxa at station y
s = number of taxa in common between
stations x and y

Percentage similarity index was calculated as follows:

$$PS = 200\sum_{i=1}^{s}\min\ (P_{iX},\ P_{iY})$$

where P_{iX} and P_{iY} are the quantities of species i at stations X and Y as proportions of the quantities of all s species at the two stations combined.

Both coefficients were calculated because they are additive and should be used in combination to provide the greatest information. If comparisons between two locations provided low SQS and PS values, the communities were considered different. If SQS was high but PS low, communities were composed of similar genera but differed either in absolute cell density or in relative abundance of genera present. When SQS was low and PS high (a rare occurrence), communities were still considered similar because the low SQS probably related to random occurrence of rare genera which affects SQS much more than PS. If both coefficients were high, communities were similar in generic composition, relative abundance of genera present, and absolute cell number.

RESULTS

Tabulated Sorenson's quotient and Pielou's percent similarity values are presented in Tables 1 and 2, respectively.

Spring season (March-May) Station 1 and 4 SQS values varied considerably with some similarities observed between the 0- to 3-m and 3- to 5-m depths. Corresponding PS values were only significant in May 1976 with 0-, 3-, and 5- to 10-m depths showing similarities, and in 1977, 3- to 5-m depths. Summer SQS values averaged higher than those during the spring season with the exception of August 1976. The remainder of upper-epilimnetic SQS values were generally in the 60 to

70 range, indicative of similar genera present among depths. PS values generally corresponded favorably with SQS values with fairly strong (values of 80 or above) similarities observed. Autumn season (October) SQS and PS values generally remained significant at the 0- to 3-m and 3- to 5-m depths, but were reduced from the summer values.

Station 10 SQS spring values were generally significant at the 0- to 5-m depths, with only two values above 60 observed below those depths. PS values were, with two exceptions, significant only in May 1976 and then only at the 0- to 5-m depths. Station 10 summer SQS and PS values were quite variable, with August 1976 and 1978 showing significant values in the 0- to 5-m depths. The remainder of PS and SQS values were generally lower than 50, indicative of little interaction among depths. Only one value above 70 was observed in the PS or SQS values during the autumn season. Values in the low 60's were observed at the 0- to 3-m and 3- to 5-m depths without any strong observations below those depths.

Station 12 SQS spring values were generally above 60 at the 0- to 5-m depths, with values at a significant level observed between the 3- to 10-m and 5- to 10-m depths in May 1977. The remainder of the values were low without any significant levels observed between upper and lower epilimnetic depths. The only significant PS values observed during the spring were in May 1976 where all depth combinations between 0 and 10 m showed values above 75. SQS and PS summer values were generally low at station 12 with only a few values above 60 observed. As with summer values, SQS and PS autumn values were quite variable with only a few values observed above 60.

SQS station comparisons (all depths combined) were, with a few exceptions, above 50 with values in the 60's being common. PS values, on the other hand, were generally lower than 50 with only a few values observed in the 60's.

DISCUSSION

Some generalities and intuitive conclusions can be drawn from these data even though a considerable amount of variability exists. Reservoir segmentation undoubtedly plays a key role in the stability of algal communities from one depth to another, and between stations. This is reflected in the data where both SQS and PS values within stations were highest at stations 1, 4, and 10, with values at station 12 being highly variable and generally not significant. The observed stability (although weak) of plankton communities at and below station 10 is most probably due to the nature of the water column itself. Above station 10, the water column is turbulent; hence, algal communities do not develop in a strictly stratified sense, but rather are constantly being turned over. This generality is opposed to areas below station 10 where long epilimnetic residence times allow for upper-epilimnetic development and subsequent slow settling. Furthermore, significant values were observed between the 0- to 3-m and 3- to 5-m depths with a few significant values observed below these depths. This is most likely due to light and flow limitations at lower depths, with upper-epilimnetic depths showing similarities because of similar physical conditions.

Another factor of paramount concern that influences the effectiveness of evaluating "average" plankton abundance and composition (and hence, trophic state) is the sampling methodology. Discrete depth samples are certainly of value but are not effective in describing average epilimnetic or hypolimnetic conditions, particularly when the sampling depths are separated by 1 to 5 m. Microlayers of plankton communities are probably more often than not missed by discrete sampling. This certainly will affect indices used to describe plankton dynamics and may lead to erroneous management decisions based on available modeling routines. It has been shown that a 1 to 3 order of magnitude over-underestimation of standing crop can be made by using discrete sampling depths (Poppe 1978). Conversely, discrete sampling does allow for development of settling rates, water column comparisons, thermal impacts, etc. For these reasons, development of an appropriate sampling regime must be implemented at project initiation and must reflect the needs of the project and not necessarily traditional sampling strategy.

Such is the case for trophic state analysis of reservoirs in general. Bimonthly or quarterly sampling of plankton at a single depth (or in a few cases, two to three depths) leaves much to be desired when trying to develop trophic state indices, or for that matter reservoir management decisions. All too often, discrete plankton sampling has resulted in poor management decisions, particularly for water supply reservoirs. Water supply reservoirs and multipurpose reservoirs require special attention to evaluate all intended water quality uses. With this in mind, specialized sampling strategies are required now and in the future to obtain the best product for the effort spent.

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Table 1. Sorenson's Quotient
Stations 1 and 4, Season/Depth Comparison

[1]May 1976

	0	3	5	10	20	40
0	100					
3	63.2	100				
5	47.1	47.1	100			
10	70.6	63.2	57.1	100		
20	70.6	70.6	50.0	57.2	100	
40	62.5	62.5	54.6	61.5	66.7	100

[4]May 1979

	0	1	3	5	8	10
0	100					
1	47.1	100				
3	68.6	54.1	100			
5	75.9	58.1	75.0	100		
8	71.4	46.7	58.1	72.0	100	
10	53.9	42.9	48.3	60.9	72.7	100

[1]July 1977

	0	3	5	10	20	40
0	100					
3	72.0	100				
5	72.0	69.2	100			
10	50.0	57.1	57.1	100		
20	37.5	47.1	47.1	20.0	100	
40	40.0	37.5	37.5	36.4	40.0	100

[1]May 1977

	0	3	5	10	20	40
0	100					
3	72.0	100				
5	66.7	63.6	100			
10	50.0	47.1	61.5	100		
20	26.7	25.0	50.0	57.1	100	
40	15.4	14.3	20.0	40.0	0	100

[1]August 1975

	0	3	5	10	20	40
0	100					
3	64.0	100				
5	58.3	66.7	100			
10	61.5	34.8	63.6	100		
20	44.4	58.3	60.9	56.0	100	
40	33.3	13.3	42.9	37.5	35.3	100

[4]August 1978

	0	2	6	10	20	40
0	100					
2	74.4	100				
6	58.1	60.0	100			
10	40.0	41.0	44.4	100		
20	18.2	25.8	42.1	44.4	100	
40	0	6.9	0	12.5	25.0	100

[1]March 1978

	0	3	5	15	20	40
0	100					
3	35.3	100				
5	50.0	76.2	100			
15	66.7	50.0	63.2	100		
20	40.0	40.0	42.9	46.2	100	
40	28.6	16.7	18.2	20.0	40.0	100

[1]August 1976

	0	3	5	10	15	40
0	100					
3	46.2	100				
5	53.3	62.5	100			
10	30.8	66.7	62.5	100		
15	9.5	0	18.2	0	100	
40	0	0	0	0	0	100

[4]August 1979

	0	1	3	5	8	10
0	100					
1	79.2	100				
3	66.7	66.7	100			
5	69.6	65.0	70.3	100		
8	55.8	48.7	47.1	62.9	100	
10	43.2	38.7	42.9	48.3	46.2	100

(Continued)

(Sheet 1 of 8)

Table 1 (Continued)
Stations 1 and 4, Season/Depth Comparison

[1]October 1975

	0	3	5	10	20	40
0	100					
3	66.7	100				
5	66.7	78.3	100			
10	47.6	40.0	43.5	100		
20	37.5	26.7	22.2	53.3	100	
40	15.2	16.7	13.3	16.7	28.6	100

[4]October 1978

	0	1	3	5	8	10
0	100					
1	45.8	100				
3	45.0	50.0	100			
5	46.5	41.9	51.4	100		
8	43.2	43.2	55.2	56.3	100	
10	34.3	40.0	51.9	40.0	58.3	100

[1]October 1976

	0	3	5	10	20	40
0	100					
3	66.7	100				
5	66.7	81.2	100			
10	46.7	42.9	48.0	100		
20	30.8	33.3	38.1	33.3	100	
40	30.8	33.3	38.1	50.0	50.0	100

[4]October 1979

	0	1	3	5	8	10
0	100					
1	68.4	100				
3	66.7	59.5	100			
5	71.4	75.0	68.3	100		
8	63.6	57.1	55.8	65.2	100	
10	65.3	51.1	66.7	62.8	64.2	100

(Continued)

(Sheet 2 of 8)

Table 1 (Continued)
Station 10, Season/Depth Comparison

May 1976

	0	3	5	10	15	20
0	100					
3	63.2	100				
5	66.7	63.6	100			
10	52.6	70.0	63.6	100		
15	53.3	62.5	55.6	75.0	100	
20	42.9	40.0	47.1	40.0	54.6	100

May 1979

	0	1	4	5	8	10
0	100					
1	78.3	100				
4	60.0	61.1	100			
5	69.8	71.8	60.6	100		
8	42.4	41.4	34.8	46.2	100	
10	57.1	58.1	56.0	57.1	55.6	100

July 1977

	0	3	5	10	15	20
0	100					
3	23.5	100				
5	50.0	64.0	100			
10	36.4	40.0	42.1	100		
15	33.3	0	14.3	40.0	100	
20	57.1	12.5	26.7	22.2	40.0	100

May 1977

	0	3	5	10	15	20
0	100					
3	68.6	100				
5	66.7	70.0	100			
10	53.9	44.4	45.2	100		
15	52.2	41.7	42.9	66.7	100	
20	36.4	26.1	29.6	57.1	54.6	100

August 1975

	0	3	5	10	15	20
0	100					
3	66.7	100				
5	48.3	64.0	100			
10	50.0	60.0	56.0	100		
15	31.6	26.7	40.0	40.0	100	
20	12.5	0	23.5	33.3	28.6	100

August 1978

	0	2	4	10	15	20
0	100					
2	82.4	100				
4	24.0	28.6	100			
10	44.4	60.9	28.6	100		
15	9.5	11.8	25.0	0	100	
20	0	0	0	0	0	100

March 1978

	0	3	5	10	20	25
0	100					
3	-	100				
5	-	77.8	100			
10	-	0	0	100		
20	-	0	0	0	100	
25	-	0	0	0	100	100

August 1976

	0	3	5	10	15	20
0	100					
3	69.8	100				
5	63.2	53.7	100			
10	22.2	20.0	32.0	100		
15	30.8	27.6	41.7	0	100	
20	8.7	7.7	9.5	0	22.2	100

August 1979

	0	1	3	5	8	10
0	100					
1	42.4	100				
3	41.4	61.1	100			
5	42.9	51.4	51.6	100		
8	50.0	45.2	44.4	46.2	100	
10	28.6	45.7	58.1	53.3	53.9	100

(Continued)

(Sheet 3 of 8)

Table 1 (Continued)
Station 10, Season/Depth Comparison

October 1975

	0	3	5	10	15	20
0	100					
3	58.3	100				
5	60.0	50.0	100			
10	57.1	47.6	58.8	100		
15	38.1	28.6	35.3	44.4	100	
20	13.3	0	18.2	16.7	33.3	100

October 1978

	0	1	3	5	8	10
0	100					
1	59.8	100				
3	68.8	47.4	100			
5	47.4	59.1	43.8	100		
8	8.7	6.9	11.8	17.4	100	
10	8.7	13.8	11.8	26.1	25.0	100

October 1976

	0	3	5	10	15	20
0	100					
3	69.2					
5	64.0	64.0	100			
10	40.0	28.6	30.3	100		
15	14.3	7.1	7.7	36.4	100	
20	44.4	33.3	29.4	31.6	16.7	100

October 1979

	0	1	3	5	8	10
0	100					
1	61.2	100				
3	49.0	60.9	100			
5	57.7	57.1	61.2	100		
8	44.4	52.4	57.1	53.3	100	
10	44.0	55.3	68.8	52.0	51.2	100

(Continued)

Table 1 (Continued)
Station 12, Season/Depth Comparison

May 1976

	0	3	5	10	12
0	100				
3	60.6	100			
5	70.6	69.0	100		
10	63.2	72.7	70.6	100	
15	38.5	57.1	45.5	38.5	100

May 1979

	0	3	5	8	10
0	100				
3	61.5	100			
5	51.0	54.2	100		
8	27.3	34.2	44.4	100	
10	28.6	25.6	40.0	55.6	100

October 1978

	0	3	5	10	12
0	100				
3	35.2	100			
5	36.4	34.8	100		
10	27.6	52.6	33.3	100	
12	8.7	0	0	0	100

May 1977

	0	3	5	10	15
0	100				
3	61.1	100			
5	65.0	68.8	100		
10	55.6	85.7	68.8	100	
15	23.1	44.4	27.3	44.4	100

October 1975

	0	3	5	10	13
0	100				
3	62.5	100			
5	58.1	64.0	100		
10	62.1	60.9	63.6	100	
13	16.0	31.6	22.2	25.0	100

October 1979

	0	3	5	8	10
0	100				
3	52.9	100			
5	57.1	59.0	100		
8	50.9	45.3	58.3	100	
10	50.8	45.6	61.5	59.1	100

March 1978

	0	3	5	10	15
0	100				
3	0	100			
5	0	0	100		
10	0	0	80.0	100	
15	0	0	80.0	0	100

October 1976

	0	3	5	10	12
0	100				
3	73.5	100			
5	64.5	67.9	100		
10	39.2	48.9	41.0	100	
12	30.4	35.0	35.3	60.9	100

August 1975

	0	3	5	10
0	100			
3	71.4	100		
5	57.1	60.0	100	
10	46.2	50.0	35.7	100

(Continued)

(Sheet 5 of 8)

Table 1 (Continued)
Station 12, Season/Depth Comparison

August 1976

	0	3	5	10	15
0	100				
3	66.7	100			
5	57.1	68.2	100		
10	43.8	41.2	58.8	100	
15	51.6	42.4	48.5	43.5	100

August 1978

	0	2	4	6	12
0	100				
2	64.5	100			
4	43.2	58.8	100		
6	38.7	50.0	29.4	100	
12	26.1	30.0	23.1	20.0	100

July 1977

	0	3	5	10	15
0	100				
3	48.0	100			
5	54.6	38.5	100		
10	51.9	30.0	50.0	100	
15	10.5	0	20.0	14.3	100

August 1979

	0	3	5	8	10
0	100				
3	59.3	100			
5	57.1	64.5	100		
8	21.1	18.9	17.4	100	
10	11.8	20.0	28.6	50.0	100

(Continued)

(Sheet 6 of 8)

Table 1 (Continued)
Station Comparisons, Depths Combined

August 1975

	1	10	12
1	100		
10	60.6	100	
12	57.1	70.6	100

September 1975

	1	10	12
1	100		
10	71.0	100	
12	73.3	71.0	100

October 1975

	1	10	12
1	100		
10	51.6	100	
12	55.6	61.5	100

December 1975

	1	10	12
1	100		
10	80.0	100	
12	61.9	60.0	100

May 1976

	1	10	12
1	100		
10	69.0	100	
12	51.3	55.6	100

June 1976

	1	10	12
1	100		
10	54.6	100	
12	67.9	55.0	100

August 1976

	1	10	12
1	100		
10	70.8	100	
12	54.2	67.9	100

September 1976

	1	10	12
1	100		
10	42.9	100	
12	42.9	59.1	100

October 1976

	1	10	12
1	100		
10	67.6	100	
12	65.1	62.9	100

November 1976

	1	10	12
1	100		
10	16.7	100	
12	27.3	38.5	100

January 1977

	1	10	12
1	100		
10	18.2	100	
12	13.3	81.8	100

February 1977

	1	10	12
1	100		
10	44.8	100	
12	51.4	52.5	100

April 1977

	1	10	12
1	100		
10	37.5	100	
12	30.0	61.5	100

May 1977

	1	10	12
1	100		
10	46.2	100	
12	53.7	62.5	100

June 1977

	1	10	12
1	100		
10	50.8	100	
12	50.0	60.3	100

July 1977

	1	10	12
1	100		
10	64.5	100	
12	74.3	58.8	100

December 1977

	1	10	12
1	100		
10	52.8	100	
12	50.0	68.5	100

March 1978

	1	10	12
1	100		
10	56.0	100	
12	0	0	100

September 1978

	4	10	12
4	100		
10	69.2	100	
12	53.7	60.0	100

October 1978

	4	10	12
4	100		
10	64.1	100	
12	56.8	61.3	100

December 1978

	4	10	12
4	100		
10	63.4	100	
12	50.0	57.1	100

March 1979

	4	10	12
4	100		
10	64.3	100	
12	53.9	75.0	100

June 1979

	4	10	12
4	100		
10	75.8	100	
12	64.6	78.8	100

July 1979

	4	10	12
4	100		
10	76.5	100	
12	60.8	67.6	100

(Continued)

(Sheet 7 of 8)

Table 1 (Concluded)
Station Comparisons, Depths Combined

August 1979

	4	10	12
4	100		
10	64.6	100	
12	53.8	54.8	100

October 1979

	4	10	12
4	100		
10	62.5	100	
12	51.1	65.4	100

November 1979

	4	10	12
4	100		
10	71.8	100	
12	62.5	62.8	100

December 1979

	4	10	12
4	100		
10	58.5	100	
12	52.9	51.3	100

January 1980

	4	10	12
4	100		
10	62.2	100	
12	53.9	69.1	100

February 1980

	4	10	12
4	100		
10	60.5	100	
12	46.8	57.7	100

March 1980

	4	10	12
4	100		
10	57.1	100	
12	61.1	61.5	100

(Sheet 8 of 8)

Table 2. Pielou Percent Similarity Matrix
Stations 1 and 4, Season/Depth Comparison

[1]May 1976

	0	3	5	10	20	50
0	100					
3	84.3	100				
5	81.2	82.9	100			
10	65.7	68.6	59.3	100		
20	31.5	38.2	28.2	49.9	100	
50	16.4	15.9	9.9	22.9	48.9	100

[1]August 1975

	0	3	5	10	20	50
0	100					
3	45.8	100				
5	58.4	58.5	100			
10	22.7	15.9	26.1	100		
20	26.7	27.4	35.6	50.7	100	
50	40.0	11.0	25.7	21.4	18.6	100

[4]August 1978

	0	2	6	10	20	40
0	100					
2	73.2	100				
6	83.1	72.3	100			
10	8.5	4.1	5.1	100		
20	5.8	4.6	8.2	18.2	100	
40	0	0.2	0	2.2	3.9	100

[1]May 1977

	0	3	5	10	20	40
0	100					
3	58.9	100				
5	47.6	78.9	100			
10	16.5	11.8	11.3	100		
20	16.3	9.4	11.2	46.2	100	
40	3.0	2.5	2.0	28.6	0	100

[1]August 1976

	0	3	5	10	15	40
0	100					
3	60.0	100				
5	60.3	68.7	100			
10	16.9	28.7	38.9	100		
15	2.4	0	3.7	0	100	
40	0	0	0	0	0	100

[4]August 1979

	0	2	4	6	8	10
0	100					
2	51.2	100				
4	29.7	10.0	100			
6	38.5	20.8	78.7	100		
8	50.4	20.9	23.2	30.6	100	
10	18.5	4.9	4.5	5.1	5.6	100

[1]March 1978

	0	3	5	15	20	40
0	100					
3	34.6	100				
5	45.5	40.0	100			
15	51.3	31.1	48.7	100		
20	29.4	30.0	25.0	44.4	100	
40	16.0	12.9	8.7	22.2	30.8	100

[1]July 1977

	0	3	5	10	20	40
0	100					
3	84.8	100				
5	54.1	56.9	100			
10	27.5	32.2	16.7	100		
20	15.9	17.0	8.1	35.9	100	
40	5.6	5.4	2.5	12.5	30.8	100

[1]October 1975

	0	3	5	10	20	50
0	100					
3	67.2	100				
5	58.4	85.0	100			
10	36.9	63.3	64.8	100		
20	14.9	10.6	9.9	26.2	100	
50	10.9	9.1	8.5	13.0	53.3	100

1 = Station 1
4 = Station 4

(Continued)

(Sheet 1 of 7)

Table 2 (Continued)
Stations 1 and 4, Season/Depth Comparison

[4]May 1979

	0	2	4	6	8	10
0	100					
2	49.4	100				
4	52.4	59.3	100			
6	19.2	42.7	39.9	100		
8	10.1	11.8	16.9	31.3	100	
10	8.3	16.5	11.5	39.4	19.3	100

[1]July 1978

	0	3	5	10	20	40
0	100					
3	69.7	100				
5	18.6	25.9	100			
10	35.1	39.9	18.1	100		
20	5.1	6.4	4.3	8.5	100	
40	0	0	1.5	0	22.2	100

[1]October 1976

	0	3	5	10	20	50
0	100					
3	58.6	100				
5	44.7	70.6	100			
10	18.7	21.1	27.3	100		
20	13.9	18.5	24.4	30.8	100	
50	5.8	7.8	10.5	40.0	28.6	100

[4]October 1978

	0	2	4	6	8	10
0	100					
2	79.7	100				
4	49.4	61.4	100			
6	56.6	69.1	84.8	100		
8	66.7	70.9	68.4	75.4	100	
10	9.6	8.9	15.2	13.0	17.6	100

[4]October 1979

	0	2	4	6	8	10
0	100					
2	73.6	100				
4	25.8	27.9	100			
6	19.6	21.6	35.9	100		
8	26.2	35.4	51.5	41.4	100	
10	60.4	67.1	36.7	35.0	38.6	100

(Continued)

(Sheet 2 of 7)

Table 2 (Continued)
Station 10, Season/Depth Comparison

May 1976

	0	3	5	10	15	20
0	100					
3	85.8	100				
5	81.4	79.7	100			
10	23.3	25.6	30.3	100		
15	18.5	18.6	24.4	44.2	100	
20	11.8	12.3	17.7	26.6	56.3	100

May 1979

	0	4	5	6	8	10
0	100					
4	19.2	100				
5	42.9	33.1	100			
6	16.3	26.0	26.4	100		
8	6.9	9.0	16.0	36.0	100	
10	7.5	34.1	22.4	36.2	30.7	100

July 1977

	0	3	5	10	15	20
0	100					
3	13.8	100				
5	12.6	71.0	100			
10	29.6	7.1	5.5	100		
15	11.1	0	1.0	15.4	100	
20	52.2	4.4	5.6	44.4	22.2	100

May 1977

	0	3	5	10	15	20
0	100					
3	52.9	100				
5	52.7	71.8	100			
10	67.1	34.2	35.0	100		
15	27.2	11.4	12.5	39.4	100	
20	26.6	11.3	13.5	37.9	66.7	100

August 1975

	0	3	5	10	15	20
0	100					
3	72.0	100				
5	55.2	71.4	100			
10	40.3	34.9	33.8	100		
15	10.0	5.0	8.0	14.6	100	
20	4.9	0	11.1	28.6	30.8	100

August 1978

	0	2	6	10	15	20
0	100					
2	89.1	100				
6	67.7	68.4	100			
10	3.4	4.2	3.5	100		
15	2.3	2.7	2.6	0	100	
20	0	0	0	0	0	100

March 1978

	3	5	10	20	25
3	100				
5	61.5	100			
10	0	0	100		
20	0	0	0	100	
25	0	0	0	50.0	100

August 1976

	0	3	5	10	15	20
0	100					
3	71.3	100				
5	63.0	68.5	100			
10	12.8	12.4	12.2	100		
15	17.6	19.6	18.3	30.7	100	
20	20.1	22.3	19.2	0	36.0	100

August 1979

	0	2	4	6	8	10
0	100					
2	43.5	100				
4	44.6	60.4	100			
6	34.3	54.5	73.8	100		
8	42.8	42.6	44.6	40.8	100	
10	34.0	26.5	26.1	21.2	46.3	100

(Continued)

(Sheet 3 of 7)

Table 2 (Continued)
Station 10, Season/Depth Comparison

October 1975

	0	3	5	10	15	20
0	100					
3	73.1	100				
5	52.3	45.9	100			
10	55.7	52.2	72.3	100		
15	8.8	8.4	12.5	16.2	100	
20	2.0	0	3.9	3.3	25.8	100

October 1978

	0	2	4	6	8	10
0	100					
2	51.5	100				
4	68.8	64.5	100			
6	37.6	61.4	51.0	100		
8	1.0	1.0	1.1	1.1	100	
10	0.2	0.4	1.0	0.6	46.7	100

October 1976

	0	3	5	10	15	20
0	100					
3	51.8	100				
5	43.3	40.0	100			
10	29.0	14.9	19.2	100		
15	6.8	3.5	4.8	28.6	100	
20	28.2	20.3	18.5	30.8	12.5	100

October 1979

	0	2	4	6	8	10
0	100					
2	61.3	100				
4	23.0	21.7	100			
6	19.8	20.3	31.8	100		
8	45.1	31.3	20.5	19.9	100	
10	25.0	22.8	20.3	9.6	41.6	100

(Continued)

Table 2 (Continued)
Station 12, Season/Depth Comparison

May 1976

	0	3	5	10	15
0	100				
3	76.0	100			
5	77.8	83.9	100		
10	84.1	83.5	77.7	100	
15	8.6	11.9	9.7	7.7	100

May 1977

	0	3	5	10	15
0	100				
3	35.27	100			
5	37.7	75.34	100		
10	23.4	18.4	15.9	100	
15	12.2	5.5	5.9	26.3	100

March 1978

	0	3	5	10	15
0	100				
3	0	100			
5	0	0	100		
10	0	0	40.0	100	
15	0	0	26.7	71.4	100

May 1979

	0	3	5	8	10
0	100				
3	25.5	100			
5	5.4	5.3	100		
8	0.5	0.7	18.5	100	
10	0.6	0.3	7.0	14.7	100

August 1976

	0	3	5	10	15
0	100				
3	55.8	100			
5	57.2	50.5	100		
10	26.9	19.7	26.4	100	
15	24.3	14.5	19.3	26.7	100

July 1977

	0	3	5	10	15
0	100				
3	7.6	100			
5	69.0	4.2	100		
10	11.3	19.7	8.5	100	
15	11.0	0	6.9	40.7	100

August 1978

	0	2	6	8	12
0	100				
2	53.8	100			
6	6.5	11.0	100		
8	4.1	21.3	25.7	100	
12	3.2	13.2	7.7	44.1	100

August 1979

	0	3	5	8	10
0	100				
3	38.8	100			
5	30.3	52.6	100		
8	10.1	3.8	8.0	100	
10	0.6	8.8	23.8	19.9	100

October 1975

	0	3	5	10	13
0	100				
3	81.9	100			
5	51.9	58.6	100		
10	57.3	62.1	74.3	100	
13	3.1	5.4	6.1	6.2	100

October 1976

	0	3	5	10	12
0	100				
3	58.8	100			
5	34.1	39.2	100		
10	14.0	23.7	21.9	100	
12	8.4	13.5	19.8	53.1	100

October 1978

	0	3	5	10	12
0	100				
3	34.5	100			
5	67.1	48.0	100		
10	10.4	14.8	9.7	100	
12	1.2	0	0	0	100

October 1979

	0	3	5	8	10
0	100				
3	18.7	100			
5	19.1	55.5	100		
8	16.4	29.6	37.5	100	
10	22.2	47.3	48.5	48.2	100

(Continued)

(Sheet 5 of 7)

Table 2 (Continued)
Stations 1, 4, 10, and 12 Depths Combined - Station Comparison

August 1975

	1	10	12
1	100		
10	73.1	100	
12	33.1	42.3	100

September 1975

	1	10	12
1	100		
10	51.3	100	
12	38.3	32.8	100

October 1975

	1	10	12
1	100		
10	59.6	100	
12	25.9	24.6	100

December 1975

	1	10	12
1	100		
10	46.0	100	
12	43.1	53.0	100

May 1976

	1	10	12
1	100		
10	67.0	100	
12	68.3	76.2	100

June 1976

	1	10	12
1	100		
10	29.8	100	
12	34.1	51.3	100

August 1976

	1	10	12
1	100		
10	58.1	100	
12	34.7	56.0	100

September 1976

	1	10	12
1	100		
10	28.7	100	
12	18.3	39.1	100

October 1976

	1	10	12
1	100		
10	42.7	100	
12	41.0	33.7	100

November 1976

	1	10	12
1	100		
10	10.5	100	
12	14.4	29.8	100

January 1977

	1	10	12
1	100		
10	3.1	100	
12	2.2	80.3	100

February 1977

	1	10	12
1	100		
10	10.2	100	
12	20.6	36.5	100

April 1977

	1	10	12
1	100		
10	21.4	100	
12	23.1	45.5	100

May 1977

	1	10	12
1	100		
10	17.8	100	
12	35.0	77.7	100

June 1977

	1	10	12
1	100		
10	38.6	100	
12	33.7	51.4	100

July 1977

	1	10	12
1	100		
10	29.1	100	
12	19.0	49.0	100

December 1977

	1	10	12
1	100		
10	37.1	100	
12	32.8	57.5	100

March 1978

	1	10	12
1	100		
10	38.6	100	
12	0	0	100

September 1978

	4	10	12
4	100		
10	43.1	100	
12	35.6	60.1	100

October 1978

	4	10	12
4	100		
10	53.5	100	
12	51.5	49.3	100

December 1978

	4	10	12
4	100		
10	47.9	100	
12	38.6	25.5	100

March 1979

	4	10	12
4	100		
10	51.5	100	
12	42.8	60.3	100

June 1979

	4	10	12
4	100		
10	27.0	100	
12	13.7	57.8	100

July 1979

	4	10	12
4	100		
10	50.8	100	
12	43.7	63.2	100

(Continued)

(Sheet 6 of 7)

Table 2 (Concluded)
Stations 1, 4, 10, and 12 Depths Combined - Station Comparison

August 1979

	4	10	12
4	100		
10	43.4	100	
12	20.6	49.1	100

January 1980

	4	10	12
4	100		
10	52.2	100	
12	21.7	32.2	100

October 1979

	4	10	12
4	100		
10	57.6	100	
12	21.2	34.1	100

February 1980

	4	10	12
4	100		
10	48.6	100	
12	31.8	26.9	100

November 1979

	4	10	12
4	100		
10	38.7	100	
12	26.6	40.0	100

March 1980

	4	10	12
4	100		
10	40.2	100	
12	28.6	40.1	100

December 1979

	4	10	12
4	100		
10	24.8	100	
12	14.2	52.2	100

(Sheet 7 of 7)

SIZE DISTRIBUTION OF PLANKTONIC AUTOTROPHY AND MICROHETEROTROPHY IN DeGRAY RESERVOIR, ARKANSAS

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ABSTRACT

Naturally occurring assemblages of phytoplankton and bacterioplankton were radiolabelled with sodium ^{14}C-bicarbonate and sodium ^{3}H-acetate and size fractionated to determine the size structure of planktonic autotrophy and microheterotrophy in DeGray Reservoir, an oligotrophic impoundment of the Caddo River in south-central Arkansas. Size distributions of autotrophy and microheterotrophy were remarkably uniform seasonally, vertically within the water column, and along the longitudinal axis of the reservoir despite significant changes in environmental conditions. Planktonic autotrophy was dominated by small algal cells with usually >50% of the photosynthetic carbon uptake accounted for by organisms <8.0 µm. Microheterotrophic activity in the 0.2- to 1.0-µm size fraction, presumably associated with free-living bacterioplankton not attached to suspended particles, usually accounted for >75% of the planktonic microheterotrophy. Longitudinal patterns in autotrophic and microheterotrophic activities associated with >3-µm and >1-µm size fractions, respectively, suggest an uplake to downlake shift from riverine to lacustrine environmental influences within the reservoir.

INTRODUCTION

The plankton ecologist's perception of the environment that he investigates is shaped to a large extent by the methods used for collecting and examining plankton samples. For years, our views of plankton community structure, metabolism, and trophic interactions were restricted to the organisms retained by a 64-µm pore size plankton net; i.e., the "net plankton." Smaller organisms that passed through the net (<64 µm, the "nanoplankton") were unnoticed until it was realized that they were responsible for much of the biomass and most of the metabolic activity occurring in the planktonic environment (e.g., Rodhe et al. 1958, Holmes 1958, Yentsch and Ryther 1959). Recent investigations of the size distributions of planktonic biomass and metabolic activities, using epifluorescence and scanning electron microscopy, radioisotopic labelling, differential filtration methods, and autoradiography, have focused attention on progressively smaller organisms. Because particle size is a primary determinant of the food resources available to consumers and the efficiency of energy transfer through foodwebs (Ryther 1969; Gliwicz 1969; Parsons and Lebrasseur 1970; Kerr 1974; Sheldon et al. 1972, 1977), the size distributions of planktonic autotrophy (algal photosynthesis) and microheterotrophy (bacterial productivity) are of considerable ecological interest.

The contribution of nanoplankton (<64 µm) to phytoplankton production and biomass is now well documented (e.g., Rodhe 1958, Ryther and Yentsch 1958, Gilmartin 1964, Malone 1971, Kalff 1972, Kalff and Knoechel 1978), and recent studies have demonstrated the importance of microplankton (<8 µm) to algal community productivity in nutrient-poor planktonic environments (Paerl 1977, Paerl and Mackenzie 1977, Ross and Duthie 1981, Munawar and Munawar 1975, Li et al. 1983, Platt et al. 1983). The predominance of small algae in oligotrophic environments is usually attributed to high cell surface to volume ratios and a resulting enhanced ability to grow at low nutrient concentrations (Dugdale 1967, Eppley et al. 1969, Caperon and Meyer 1972, Parsons and Takahashi 1973, Friebele et al. 1978). The relative importance of the larger algae generally increases in more productive systems where nutrient availability is higher (e.g., mesotrophic and eutrophic lakes, estuaries, upwelling and coastal marine environments) relative to oligotrophic lakes and the open ocean (Kalff and Knoechel 1978, Malone 1980, Watson and Kalff 1981, Schlesinger et al. 1981). Net plankton can make major contributions to phytoplankton productivity by virtue of large biomass accumulations (Kalff and Knoechel 1978). However, the biomass-specific productivity of large algae is usually low (Stull et al. 1973, Kalff and Knoechel 1978), and zooplankton grazing occurs primarily on small (<30 µm) cells that are most efficiently filtered, ingested, and assimilated (Burns 1968, Gliwicz 1969, Porter 1977).

Phytoplankton photosynthesis is the primary means of organic matter production in most planktonic systems; however, bacterial production may also be an important trophic resource for planktonic consumers (Pomeroy 1974, Sieburth 1976, Sieburth et al. 1978, Peterson et al. 1978, Porter et al. 1979, Ducklow 1983). Bacterial uptake of algal excretion products returns an otherwise unharvestable portion of the primary production to the grazer food chain (Paerl 1974, 1978; Cole 1982). In aquatic systems that receive considerable organic matter loading from their watersheds (e.g., reservoirs and riverine lakes), the microheterotrophic conversion of biologically available allochthonous dissolved organic matter (DOM) to bacterial biomass may significantly supplement ecosystem productivity (Kuznetsov 1968, Sorokin 1972) if the bacterial production is efficiently harvested by planktonic consumers. Bacteria associated with detrital particles or aggregates appear to be more efficiently harvested by macrozooplankton than are free-living bacterioplankton by virtue of their greater effective particle size (Peterson et al. 1978, Hobbie and Wright 1979, Kimmel 1983). Small (<30 µm) ciliates and heterotrophic microflagellates may provide a trophic link between the free-living bacterioplankton and macrozooplankton (Sieburth et al. 1978, Porter et al. 1979, Pace and Orcutt 1981, Beaver and Crisman 1982); however, the energetic cost of additional trophic transfers may diminish the significance of this linkage to the foodweb.

There is no general agreement in the literature on the relative importance of free-living and attached bacteria in planktonic environments. Numerous investigators have observed bacterial colonization of suspended particles and microbial-detrital aggregates (Seki 1972; Paerl 1973, 1975; Bent and Goulder 1981), but others have reported most bacterioplankton to be free-living (Wiebe and Pomeroy 1972, Hobbie and Rublee 1975, Ferguson and Rublee 1976). Measurements made in coastal and open ocean systems indicate that generally 80% or more of the bacterial biomass and activity is due to free-living rather than attached bacteria (Azam and Hodson 1977, Wiebe and Pomeroy 1972, Ducklow and Kirchman 1983), and similar results have been obtained for a variety of natural lakes (Paerl 1980) and reservoirs (Kimmel 1983). However, attached bacteria have been reported to dominate microheterotrophic activity in planktonic systems having high concentrations of suspended particles; e.g., near-shore waters in large lakes and in coastal regions (Paerl 1977, 1980), and turbid rivers (Jannasch 1956) and estuaries (Hanson and Wiebe 1977, Bent and Goulder 1981). Paerl and Goldman (1972) concluded that suspended particles transported by turbid stream inflow to ultraoligotrophic Lake Tahoe stimulated planktonic microheterotrophy by serving as both a surface for microbial attachment and an enriched microenvironment for bacterial growth.

Jannasch and Pritchard (1972) reemphasized earlier suggestions (Waksman and Carey 1935a,b; Zobell and Anderson 1936) that adsorption of dissolved inorganic and organic nutrients increases concentration gradients at particle surfaces and thereby promotes microbial attachment to suspended particles in oligotrophic environments. Fluvial inputs of suspended particles to reservoirs provide a greater number of particles and a greater surface area for bacterial attachment and growth than occur in most oceanic and lacustrine environments. Whether a similar enhancement of microbial activity in association with suspended particle surfaces occurs in higher-nutrient environments, such as particle-rich reservoirs, remains uncertain (Goldman and Kimmel 1978). However, Marzolf and co-workers (Marzolf 1980, Marzolf and Arruda 1981, Arruda et al. 1983) have demonstrated that DOM adsorption and bacterial growth associated with suspended clay particles can be of major importance to reservoir zooplankton when significant phytoplankton production is prevented by abiogenic turbidity.

Kimmel (1983) surveyed several reservoirs of differing trophic status and reported that microalgae (<8.0 µm) and free-living bacteria (<1.0 µm) were primarily responsible for planktonic autotrophy and microheterotrophy, respectively, in the impoundments examined. However, his sampling was limited both spatially and temporally, and did not include an oligotrophic system. Here we report the results of a more thorough sampling of DeGray Reservoir, an oligotrophic impoundment of the Caddo River in south-central Arkansas. Previous water quality studies of DeGray Reservoir have shown it to possess marked longitudinal gradients in nutrient concentrations, water clarity, algal biomass, and phytoplankton productivity (Thornton et al. 1982; Kennedy et al. 1982; J. Nix, unpublished data). This spatial heterogeneity provided us the opportunity to examine within a single system the responses of naturally occurring phytoplankton-bacterioplankton assemblages to gradients of environmental factors hypothesized to control the size distributions of planktonic autotrophy and microheterotrophy.

METHODS

DeGray Reservoir was sampled on three occasions (31 August - 2 September 1982, 1-4 February 1983, and 21-23 June 1983), representative of late-summer, mid-winter, and early-summer environmental conditions, respectively. During each sampling trip, we obtained near-surface (1 to 2 m) samples from stations located along the longitudinal axis of the reservoir and a vertical series of samples at selected stations. Water samples were collected with a submersible pump connected to a weighted opaque hose, pumped into 10-L plastic cubitainers, and then subsampled for various measurements and experiments. Water temperature, dissolved oxygen, pH, and conductance were measured with either a Hydrolab or a Martek monitoring system. Incident solar radiation was monitored with a calibrated mechanical pyrheliograph, and photosynthetically active radiation (PAR, 400-700 nm) was measured in situ with a Li-Cor quantum meter equipped with a spherical submersible sensor. In vivo chlorophyll fluorescence (IVF) was also determined in situ (Lorenzen 1966). Dissolved nutrients, chlorophyll a concentrations, and phytoplankton productivity were estimated by standard automated methods (Stainton et al. 1974), methanol extraction (Marker et al. 1980), and ^{14}C uptake (Goldman 1963, Vollenweider 1971), respectively.

Size distributions of planktonic autotrophy and microheterotrophy were determined by isotopic labelling and differential filtration of natural phytoplankton-bacterioplankton assemblages (Kimmel 1983). Subsamples in 130-mL light and dark bottles were inoculated with 0.5 mL $NaH^{14}CO_3$ solution (56.5 mCi/mmol specific activity, 5.5 µCi/mL) and 0.1 mL sodium ^{3}H-acetate (10 Ci/mmol, 25.0 µCi/mL; 0.5 µg/L acetate enrichment over ambient concentration) to label autotrophs and microheterotrophs, respectively, and incubated in situ for 4-5 h. Vertical series samples were incubated at the depths from which they were taken. Longitudinal series samples were obtained from the mixed layer (from 1-2 m) at stations along the reservoir longitudinal axis (Fig. 1) and incubated at a single station at the depth of photosynthetically saturating light (150-300 µE m^{-2} sec^{-1}), usually at 2-3 m. In September and February, samples were double-labelled (inoculated with both ^{14}C-bicarbonate and ^{3}H-acetate); however, because of problems in detecting adequate ^{3}H activity in February samples, all June samples were inoculated separately. Selected samples were poisoned with 1 mL saturated $HgCl_2$ solution, inoculated, and incubated to provide a correction for radioisotope adsorption to particles and filters.

Immediately after incubation, 15-mL aliquots were gently vacuum filtered (<100 torr) in parallel through 47-mm Nucleopore polycarbonate filters of 0.2, 1.0, 3.0, and 8.0 µm pore diameter. Filters and retained particles were rinsed three times with deionized water and placed in plastic minivials. Six milliliters of Aquasol fluor were added to each minivial, and all samples were radioassayed using a Packard 4640 liquid scintillation spectrometer. Automatic external standardization, calibrated with quenched series of ^{14}C and ^{3}H

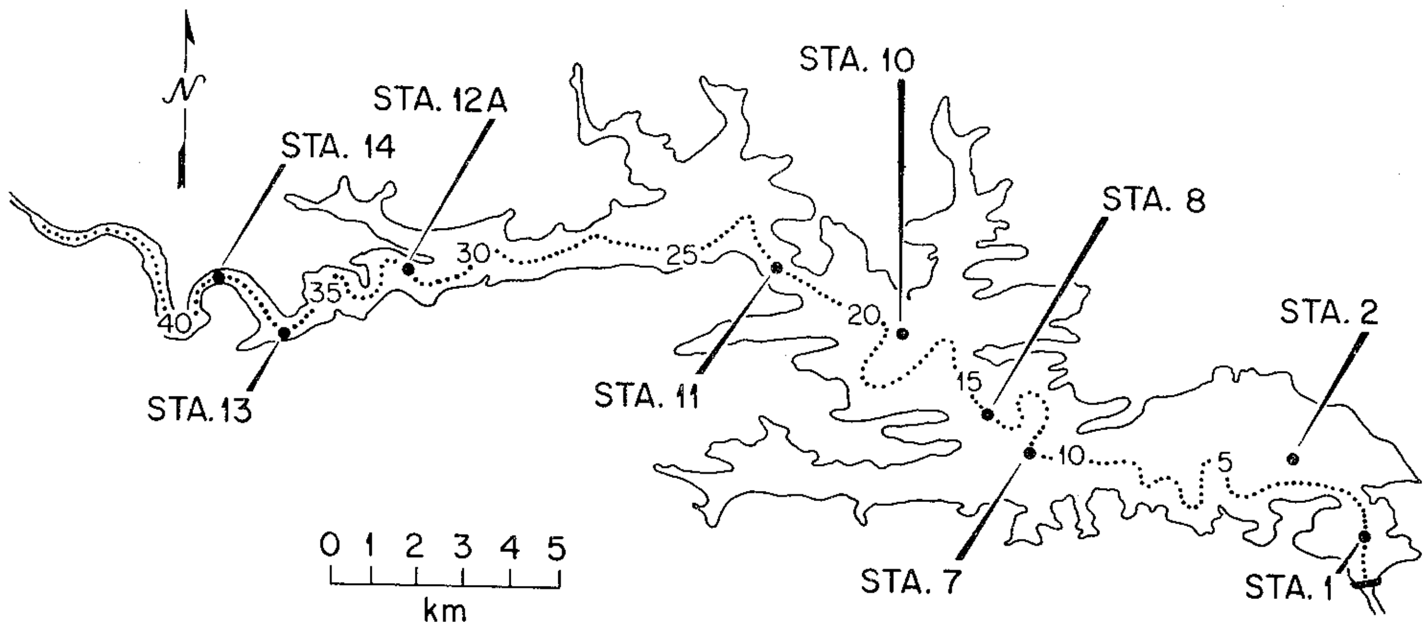

Fig. 1. Map of DeGray Reservoir, an impoundment of the Caddo River located in south-central Arkansas. The submerged river channel (thalweg) and the distance above the dam (in kilometers) are indicated by the dotted line and interspersed numerals, respectively. Sampling stations located along the longitudinal axis of the reservoir are also shown.

standards, was used to correct for sample quenching. Planktonic autotrophy was estimated as the difference between light and dark bottle ^{14}C uptake and microheterotrophy as ^{3}H uptake in the dark. All samples were corrected for radioisotope adsorption. Size distributions of autotrophic and microheterotrophic activities were expressed as percentages of the activity retained by the 0.2-µm filter.

RESULTS AND DISCUSSION

Vertical Patterns

Light availability restricted significant planktonic autotrophy to the upper 5 to 6 m of the water column in DeGray Reservoir (Fig. 2). Phytoplankton productivity at the depth of maximum photosynthesis was highest in February and lowest in September (Fig. 3a); however, integral primary production did not vary greatly on a seasonal basis (19, 23, and 25 mg C m^{-2} h^{-1}) in September, February, and June, respectively) due to a progressive increase in euphotic zone depth and, probably, in algal nutrient deficiency from winter to late summer.

Vertical changes in the size distribution of autotrophy were not statistically demonstrable due to the low number of samples; however, within the euphotic zone (>1% surface light), the relative importance of the >3-µm size fraction appeared to decline with increasing depth (Fig. 3b). This apparent decrease with depth in the activity of "larger" cells (i.e., 3- to 8-µm and >8 µm) was more gradual in September and June than in February, suggesting a direct relationship to light intensity. However, at lower light levels (<1% surface light), the fraction of total autotrophy associated with >3-µm particles increased (at 5 m in February and at 6 m in June). In June, all autotrophy at 6 m was associated with >3-µm organisms, with 71% of the total in the 3- to 8-µm fraction and 29% >8 µm. Cells >8 µm appeared to be somewhat more important during the summer months than in mid-winter, but in almost all cases, <50% of the total autotrophic activity occurred in the >8-µm size fraction.

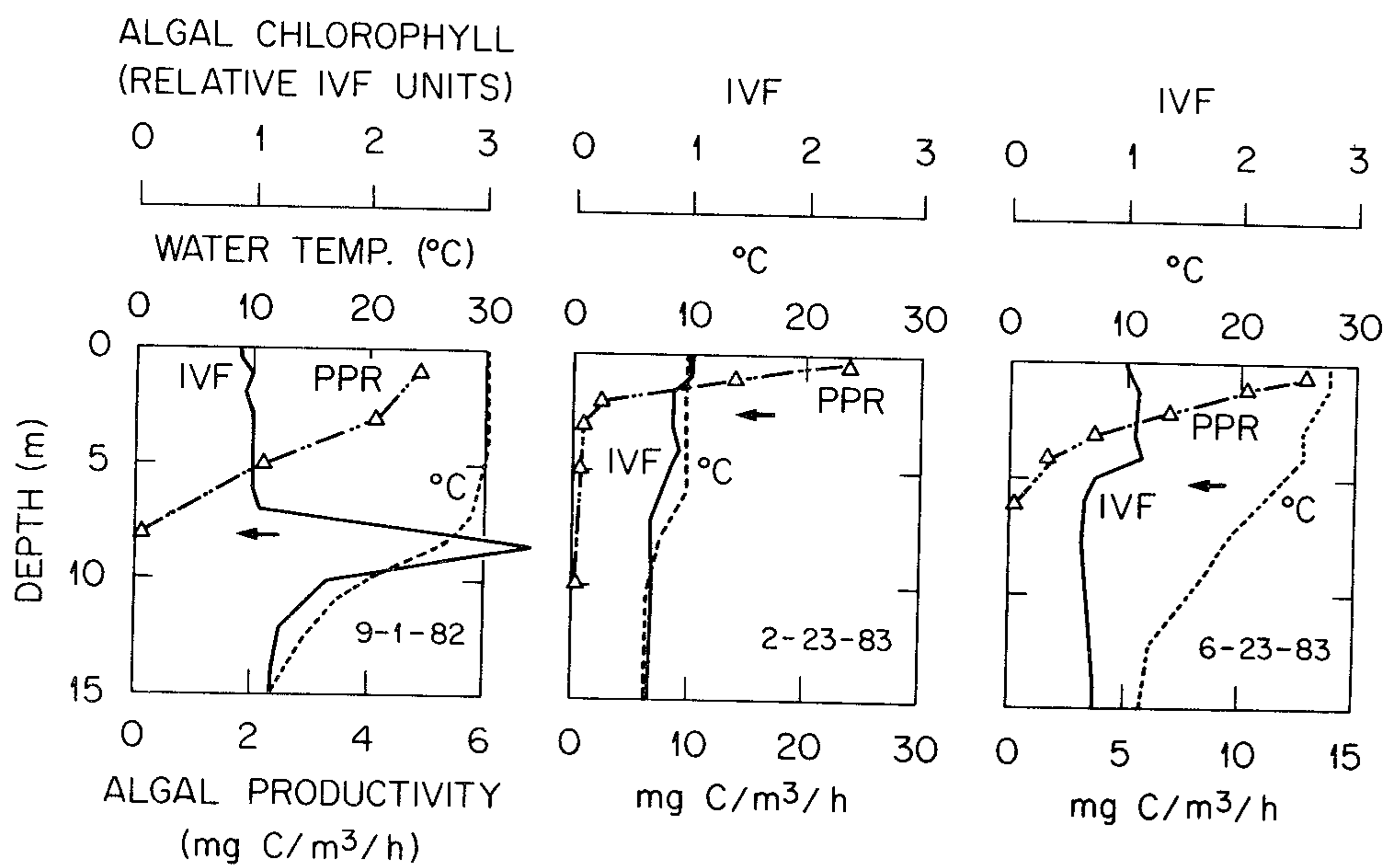

Fig. 2. Representative vertical profiles of water temperature, algal biomass [as indicated by in vivo chlorophyll fluorescence (IVF) measurements], and phytoplankton productivity (PPR) for September 1982, February 1983, and June 1983 sampling periods. Arrows indicate the depth of the euphotic layer (=1% surface light penetration). Note that the phytoplankton productivity scale differs for each date.

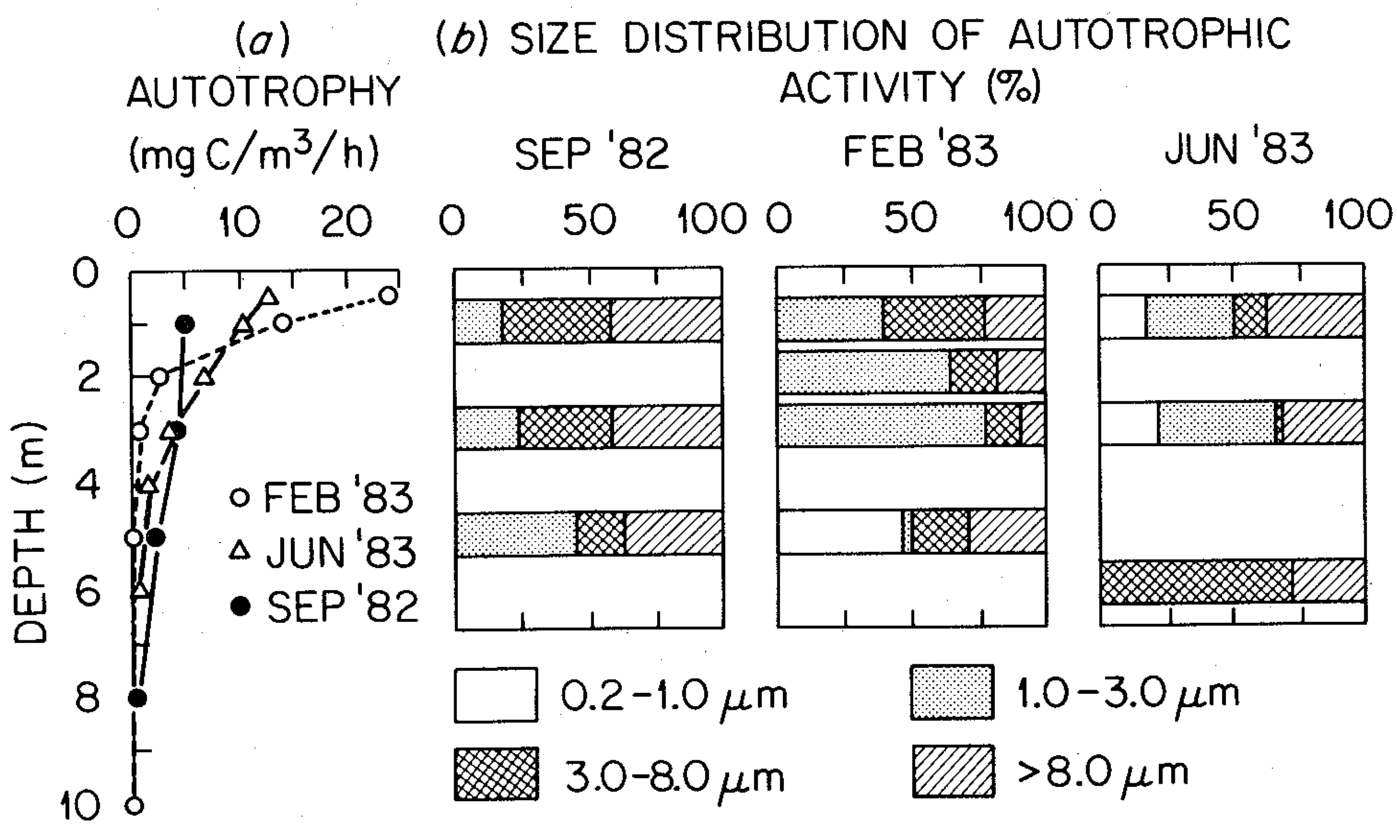

Fig. 3. (a) Vertical distributions of total planktonic autotrophy for 1 September 1982, 23 February 1983, and 23 June 1983 at lacustrine stations in DeGray Reservoir. (b) Changes in the size distribution of autotrophic activity with water column depth.

These results are of interest in regard to the influences of nutrient and light availability on size-dependent phytoplankton growth. Much research has indicated that small algal cells (e.g., nanoplankton) have a competitive advantage over larger cells with respect to nutrient uptake and cell growth in nutrient-poor environments. However, the models of Laws (1975) and Shuter (1979) assume that both growth and respiration rates are inversely related to cell size and, therefore, predict that large cells should grow faster than small cells at low light intensity. In DeGray Reservoir, most of the autotrophic activity occurs in smaller size fractions than those specificially considered in algal growth models. However, the observed vertical patterns in the size distribution of autotrophy appear to support the hypothesis that the competitive advantage of small cells in nutrient-poor environments is reduced at low light intensities (Schlesinger et al. 1981).

Comparisons of microheterotrophic activity are problematic because of the numerous organic substrates potentially available for microbial uptake in natural waters and our lack of knowledge of the identities, concentrations, and relative availabilities of these substrates. We used ^{3}H-acetate as an analog of low molecular weight, dissolved organic compounds that should be readily available for bacterial uptake. Levels of planktonic microheterotrophy, as indicated by ^{3}H-acetate uptake, were significantly higher (ANOVA, $F_{[2,13]} = 29.4$, $P < 0.01$) in September 1982 than in February and June 1983 (Fig. 4a). Average turnover times of the ^{3}H-labelled acetate pool for the depths sampled were 9.6, 16.9, and 20.3 h for September, February, and June vertical profiles, respectively. Microheterotrophic activity did not vary significantly with depth ($r = -0.40$, 6 df, NS) even in September when the vertical structure of water temperature, algal biomass, and phytoplankton productivity was pronounced (Fig. 2).

We were unable to determine the size distribution of microheterotrophy in February due to a misjudgement of the proportion of ^{14}C and ^{3}H activities added to the samples. As a result of our error, ^{3}H activity in the larger size fractions was undetectable relative to the ^{14}C activity present. However, in September and

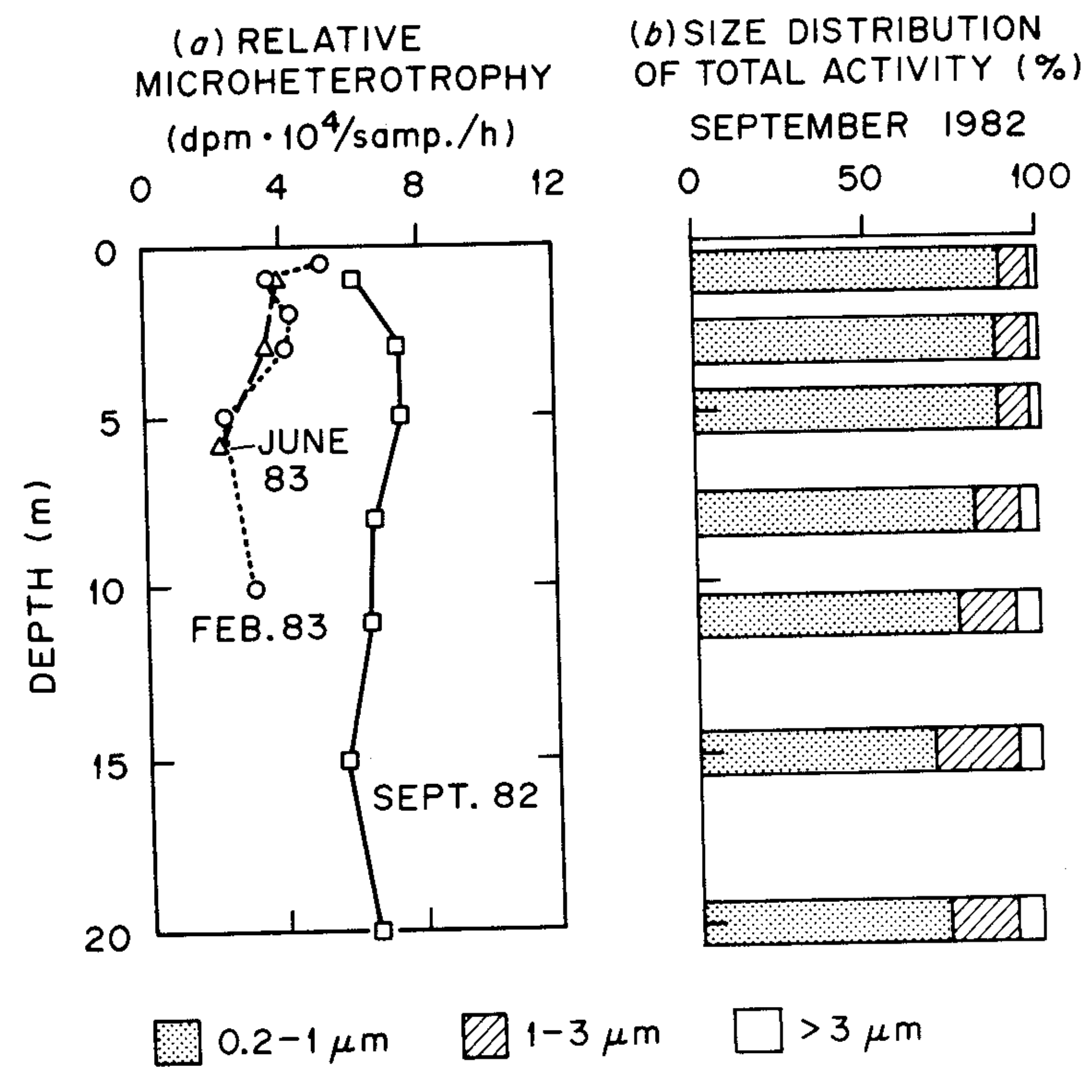

Fig. 4. (a) Vertical distributions of planktonic microheterotrophy (as indicated by ^{3}H-acetate uptake) for 1 September 1982, 23 February 1983, and 23 June 1983 at lacustrine stations in DeGray Reservoir. (b) Changes in the size distribution of microheterotrophic activity with water column depth for 1 September 1982.

June samples, microheterotrophy was dominated (usually >75%) by the <1.0-µm size fraction, indicating uptake by free-living bacterioplankton. The September 1982 vertical profile (Fig. 4b) shows that the importance of attached bacteria (>1 µm) was low ($\bar{X}$ = 9.8%, range = 7.5-11.7% of the total microheterotrophic activity) in the mixed layer (0 to 7 m), but increased significantly (ANOVA, $F_{[1,5]} = 37.6$, $P < 0.01$) in the metalimnion and hypolimnion ($\bar{X}$ = 24.1%, range = 18.6-26.4%). This shift toward larger particle sizes with depth was likely a result of decreased availability of labile DOM supplied by algal excretion and increased concentrations of algal-derived particulate organic detritus. However, even in metalimnetic and hypolimnetic samples, microheterotrophic activity associated with particles >3.0 µm was usually <10% ($\bar{X}$ = 9.1%, range = 6.2-10.8%) of the total.

Longitudinal Patterns

Longitudinal changes in the magnitudes of near-surface planktonic autotrophy and microheterotrophy were most apparent in September 1982, although uplake to downlake reductions in both were observed on all three sampling dates (Fig. 5). Phytoplankton productivity was high in the upper portion of the impoundment (stations 14 and 13) in September 1982, decreased rapidly toward midlake (stations 12A and 11), and then declined to <10% of the uplake level in the lower portion of the reservoir (stations 10, 7, and 2). Microheterotrophy showed a similar, but less marked, longitudinal pattern. Except for a two-fold reduction between stations 14 and 13 in September, microheterotrophic activity was relatively constant along the longitudinal axis of the reservoir, decreasing only slightly from uplake to downlake stations.

Longitudinal trends in the size distributions of planktonic autotrophy and microheterotrophy were less apparent than changes in the magnitudes of the total autotrophic and microheterotrophic activities within DeGray Reservoir (Fig. 6, Table 1). As in vertical profiles, much (X = 54%, range = 32-77%) of the total autotrophic activity was associated with the 1.0- to 8.0-µm size fraction; however, autotrophy in the >8-µm size fraction was quite important and comprised most of the remaining activity (X = 40%, range = 23-65%). Usually, there was

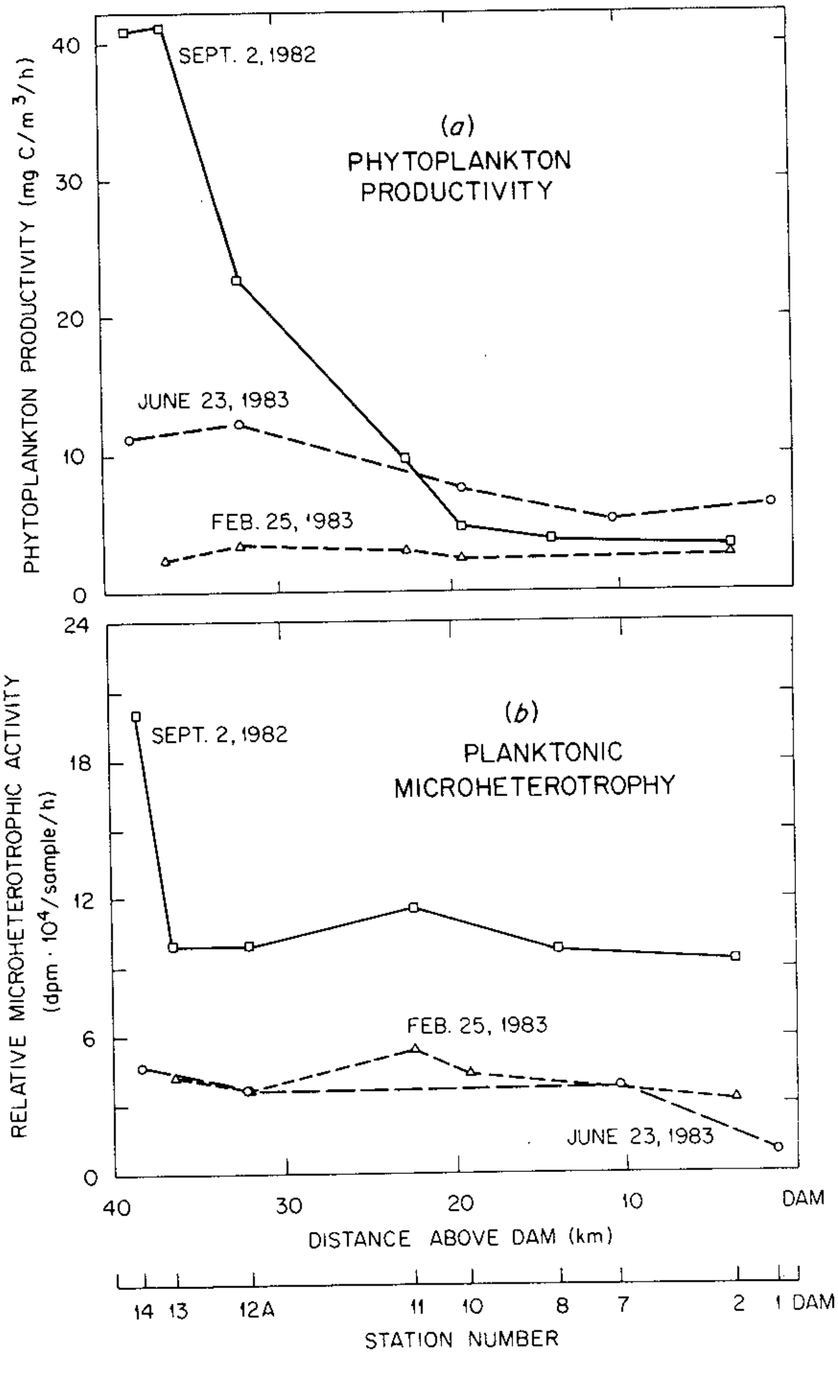

Fig. 5. Longitudinal patterns in the magnitudes of (a) phytoplankton productivity (autotrophy) and (b) planktonic microheterotrophy in DeGray Reservoir.

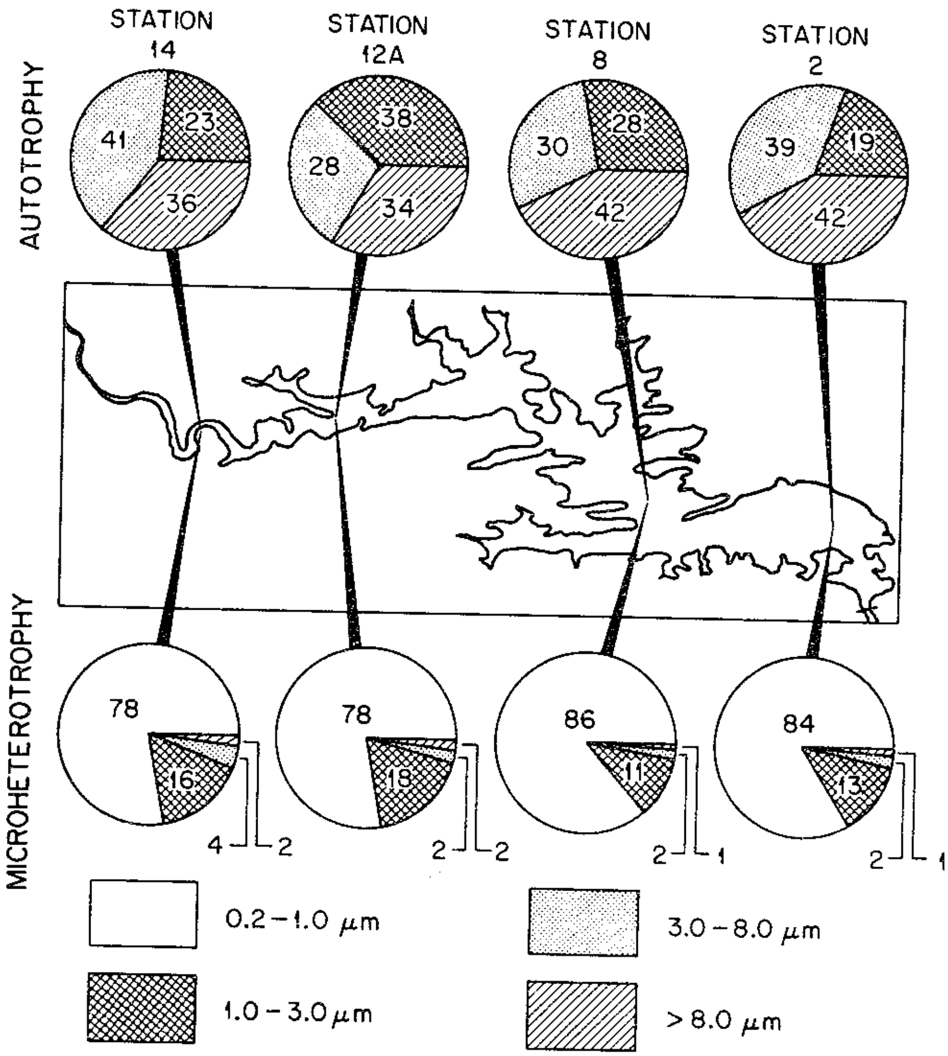

Fig. 6. Size distributions of planktonic autotrophy and microheterotrophy in near-surface (1 to 2 m) samples from selected stations along the longitudinal axis of DeGray Reservoir, 2 September 1982. Results expressed as percentages of the total activity.

Table 1. Size distributions of planktonic autotrophy and microheterotrophy in near-surface samples along the longitudinal axis of DeGray Reservoir. Station numbers increase with increasing distance from the dam; see Fig. 1 for station locations. Total autotrophic and microheterotrophic activities are defined as ^{14}C and ^{3}H activities, respectively, in particulate matter retained by a 0.2-µm pore diameter Nucleopore filter.

Date and Station	% Autotrophic Activity Retained				% Microheterotrophic Activity Retained			
	0.2-1.0	1.0-3.0	3.0-8.0	>8.0 µm	0.2-1.0	1.0-3.0	3.0-8.0	>8.0 µm
Sept. 2, 1982								
Sta. 14	0	23.1	41.4	35.7	78.2	15.8	4.5	1.5
13	0	32.9	43.8	23.3	75.8	16.0	6.2	2.0
12A	0	37.8	27.7	34.5	78.4	17.6	2.1	1.9
11	0	45.1	27.2	27.7	88.7	9.7	1.0	1.0
8	0	28.1	30.3	41.5	86.8	10.7	1.8	1.0
2	0	19.0	38.7	42.3	83.9	12.9	2.5	1.0
Feb. 25, 1983								
Sta. 13	0	36.8	22.6	40.6	--	--	--	--
12A	0	36.2	25.8	37.9	--	--	--	--
11	11.6	26.2	28.4	33.8	--	--	--	--
10	15.3	18.5	20.4	45.7	--	--	--	--
2	0	54.8	17.7	27.5	--	--	--	--
June 23, 1983								
Sta. 14	0	19.5	27.1	53.3	81.1	14.7	1.9	2.3
12A	0	38.8	23.7	37.5	75.9	20.3	2.2	1.5
7	18.0	32.4	12.2	37.4	83.6	13.5	1.4	1.4
1	2.9	20.6	11.6	64.8	72.5	21.8	2.4	3.3

little if any autotrophic activity detected in the <1.0-μm fraction (Table 1). The size distribution of planktonic microheterotrophy was even more uniform. Generally, >75% ($\bar{X}$ = 80%, range = 72-89%) of the total microheterotrophic activity was associated with the <1.0-μm size fraction, indicative of free-living rather than attached bacteria. Of the microheterotrophy apparently due to bacteria associated with suspended particles, most was in the 1.0- to 3.0-μm size fraction, with usually <5% ($\bar{X}$ = 4.3%, range = 2.0-6.2%) of the total activity associated with particles >3 μm.

If autotrophy >3.0 μm and microheterotrophy >1.0 μm (those size fractions likely to be most available to planktonic macroconsumers) are examined, longitudinal patterns are discernible that suggest changes in controlling mechanisms along the longitudinal axis of the reservoir (Fig. 7). In September 1982, autotrophy >3.0 μm decreased gradually from station 14 to 11, then increased again further downlake. In June 1983, the decrease in the relative importance of >3-μm autotrophy extended further downlake to station 7 before increasing again at station 1. In contrast, autotrophy >3 μm remained at a lower, but relatively constant, level from uplake to midlake and then declined downlake in February 1983. This different pattern likely resulted from the extension of riverine conditions throughout the reservoir during the 1982-83 winter following a record flood in December (J. Nix, personal communication).

Settling and/or grazing losses of larger cells, reduced cell size in response to decreasing nutrient availability downlake, or a size-dependent growth response to a shift from a relatively fluctuating advective nutrient supply in the most riverine portion of the impoundment to a lower, but more constant, level of available nutrients supplied by internal recycling further downlake could account for the June and September declines in the relative importance of >3-μm autotrophy in the upper portion of DeGray Reservoir. The experimental results of Turpin and Harrison (1979) suggest that the growth of larger cells may be favored by the temporal patchiness of a limiting nutrient. The subsequent increases in autotrophy associated with the >3-μm size fraction downlake are more difficult to explain. However,

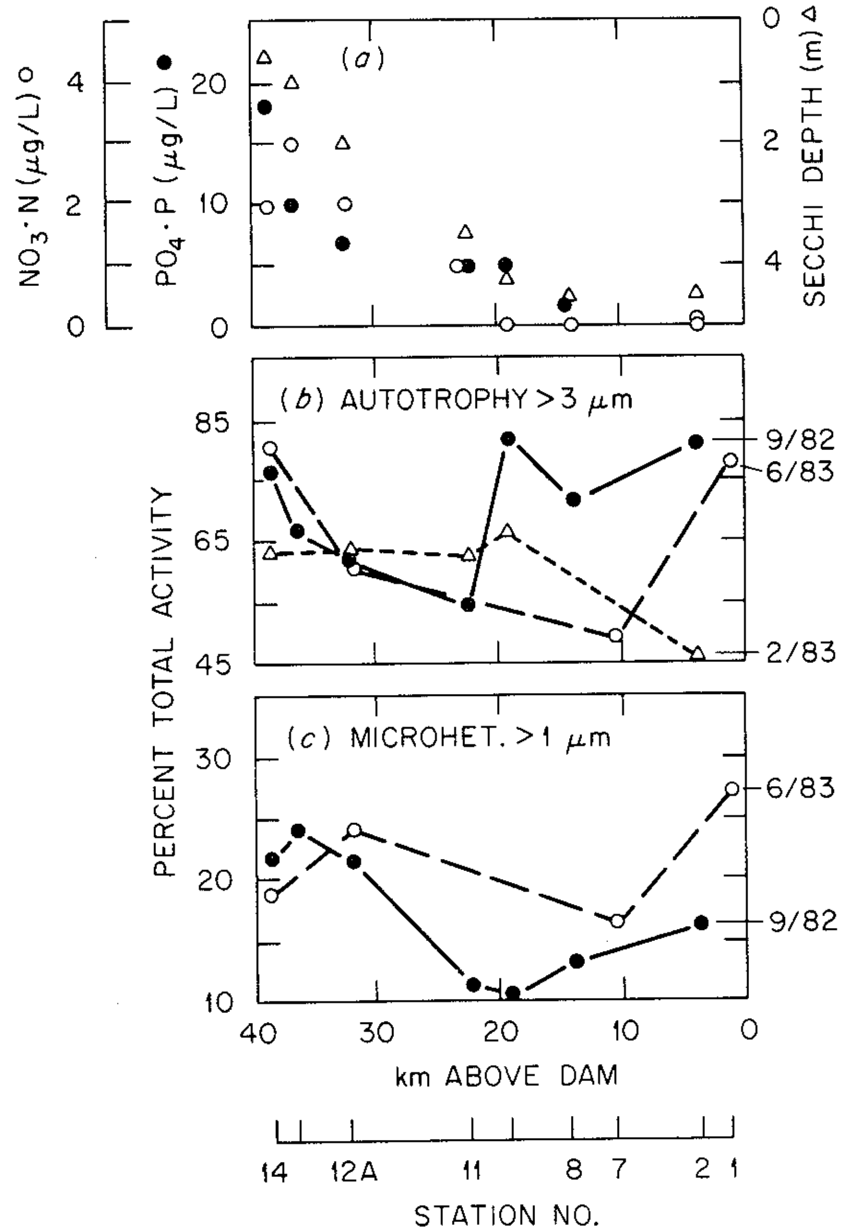

Fig. 7. Longitudinal patterns in dissolved nutrient concentrations, water transparency (as reflected by Secchi disc readings), and in the size fractions of planktonic autotrophy and microheterotrophy (>3 µm and >1 µm, respectively) most available to macrozooplankton. Nutrient and water transparency data are for the September 1982 sampling period.

an uplake to downlake shift from diatoms and blue-greens to dominance by green algae is indicated by preliminary examination of phytoplankton analyses (Kimmel, unpublished data). Therefore, it appears likely that we may be viewing the result of both size-dependent and species-specific algal growth responses to a changing combination of environmental controls along the reservoir longitudinal axis.

We observed similar longitudinal patterns in microheterotrophy associated with >1.0-μm particles (Fig. 7), also suggesting the operation of different environmental controls in the upper and lower portions of DeGray Reservoir. Bacterial association with suspended particles and detrital aggregates in nutrient-poor pelagic environments is believed to be mediated by adsorption of dissolved organic compounds and inorganic nutrients to particle surfaces, thus creating an enriched microenvironment for bacterial growth (e.g., Seki 1972; Jannasch and Pritchard 1972; Paerl and Goldman 1972, Paerl 1973, 1978). However, size distribution measurements made in less oligotrophic environments have suggested that the availability of suspended particles for bacterial attachment may be of greater influence than nutrient availability or trophic state (Paerl 1980, Bent and Goulder 1981, Kimmel 1983). Our measurements of the size distribution of planktonic microheterotrophy in DeGray Reservoir indicate that both mechanisms operate simultaneously within a broad range of environmental conditions, as exemplified by the superimposed gradients of nutrient availability and suspended particle concentrations along the reservoir longitudinal axis. In September 1982 and June 1983, attached bacteria (>1 μm) accounted for 19-24% of the total microheterotrophic activity in the relatively nutrient-rich and particle-rich upper portion of the reservoir, but decreased in importance toward midlake as suspended particle levels (as reflected by increasing Secchi depth) decreased (Fig. 7). Microheterotrophy in the >1.0-μm size fraction increased again in the lower portion of the reservoir as nutrient levels declined and, presumably, as enriched microenvironments at particle surfaces became more important for bacterial growth. Therefore, our data are consistent with a shift from control of the size distribution of planktonic microheterotrophy by suspended particle availability uplake to control by nutrient availability downlake. Similar patterns have been observed along longitudinal transects of riverine estuaries (A. V. Palumbo, unpublished data). These results should help resolve numerous, apparently contradictory, observations regarding the relative importance of free-living versus attached bacteria in various planktonic environments.

SUMMARY AND CONCLUSIONS

The size distributions of planktonic autotrophy and microheterotrophy in oligotrophic DeGray Reservoir were remarkably uniform, both spatially and temporally (Figs. 3, 4, 6; Table 1). As previously observed in oceanic and coastal systems (Azam and Hodson 1977) and in unproductive freshwater lakes (Paerl 1977, Ross and Duthie 1981, Munawar and Munawar 1975), planktonic autotrophy was dominated by small algae with usually >50% of the total carbon uptake accounted for by the <8-µm size fraction. Similarly, free-living bacterioplankton (<1.0 µm) were responsible for 75-90% of the planktonic microheterotrophy.

Longitudinal changes in planktonic autotrophy and microheterotrophy associated with particles >3 µm and particles >1 µm, respectively, suggest that the environmental controls on the size distributions of algal and bacterial activities may shift along the longitudinal axis of the reservoir (Fig. 7). Potential explanations for the observed longitudinal patterns are necessarily speculative and, inevitably, produce more questions than answers. However, these results are significant from at least three viewpoints:

(1) These data (and those presented in the following companion paper in this volume) demonstrate that the spatial heterogeneity characteristic of reservoir ecosystems can be used to an advantage by limnologists as an experimental tool. Superimposed gradients of flow velocity, suspended particle levels, nutrient concentrations, and light availability along reservoir longitudinal axes provide research opportunities for relating the responses of biotic communities to a wide range of environmental conditions within a single aquatic system.

(2) The observed longitudinal trends in both the magnitudes and the size distributions of planktonic autotrophy and microheterotrophy are interpretable from a conceptual view of reservoirs as "river-lake hybrids" or transitional environments. Reservoirs appear to combine numerous features of river and lake environments and, to at least some degree, a shift from more riverine to more lacustrine conditions occurs within individual impoundments (Thornton et al. 1982; Kimmel et al., in press; Kimmel and Groeger, in press). The river-lake hybrid analogy has been used in a qualitative sense by numerous authors over the years, but is now receiving more serious attention as a potentially useful framework for explaining the spatial and temporal heterogeneity, diversity, and ecological structure of reservoirs as a class of aquatic ecosystems (see papers in Thornton, in press).

(3) Although the occurrence of longitudinal gradients in physical and chemical factors and in water quality within reservoirs is now becoming relatively well documented (Thornton et al. 1982; Kennedy et al. 1982; Kennedy, this volume), biological and ecological responses to such environmental gradients are not well known.

Size distributions of planktonic autotrophy and microheterotrophy in DeGray Reservoir correspond well to values previously reported for several more productive impoundments (Kimmel 1983). Together, these data show that, over a broad range of environmental conditions, the predominant fractions of planktonic autotrophy and microheterotrophy are associated with <8-μm algae and <1-μm bacteria, respectively. Furthermore, these results support the view that pelagic ecosystem metabolism is dominated by very small organisms (Pomeroy 1974, Sieburth et al. 1978, Williams 1981, Ducklow 1983) and demonstrate that such a view applies not only to unproductive open ocean, coastal, and lacustrine environments, but also extends to other more productive lakes and reservoirs.

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SPATIAL AND SEASONAL PATTERNS OF PHOTOSYNTHETIC CARBON METABOLISM IN DeGRAY LAKE PHYTOPLANKTON

Alan W. Groeger and Bruce L. Kimmel

ABSTRACT

Flow of recently assimilated carbon into intracellular, macromolecular pools was used to assess the physiological state of the phytoplankton community in DeGray Lake during September 1982 and February and June 1983. Patterns from both vertical profiles and along the longitudinal axis of the reservoir were examined. Phytoplankton responded to increasing light intensities by sequestering more carbon into polysaccharides. Within the lacustrine zone of the reservoir, which usually extended uplake from the dam at least 25 to 30 km along the river channel, the phytoplankton were in a similar physiological state and, during June and September, showed carbon allocation patterns characteristic of nutrient deficiency. High percentages of carbon flowing into lipid production and the results of other nutrient deficiency assays suggest that phytoplankton productivity was nitrogen limited in the lacustrine zone of DeGray Lake in June 1983.

INTRODUCTION

The biochemical composition of algal cells in culture varies with changes in growth phase, light regime, and nutrient depletion (Myers 1946, Cook 1963, Fogg 1975). Protein may constitute more than 50% of the dry weight of algae during exponential growth (Parsons et al. 1961, Healy 1975). As growth rate declines in relation to a depleted nutrient supply, polysaccharide and lipid make up an increasingly larger fraction of the biomass (Healy 1975). While the biochemical composition of natural phytoplankton assemblages has been used with some success as a physiological indicator (Barlow 1982), this approach lacks sensitivity and particulate detritus can be a serious interference.

In this study, we employed an extension of the standard ^{14}C-productivity measurement to trace the flow of photosynthetically reduced carbon into intracellular, macromolecular pools (Fig. 1) in DeGray Lake phytoplankton. This method provides estimates of both the rate of phytoplankton photosynthesis and the rate of flow of photosynthate into different biosynthetic pathways. Characterization of the relative allocation of photosynthetically reduced carbon in relation to light, temperature, and nutrient conditions *in situ* may improve our understanding of how these factors affect algal growth under natural conditions. To reiterate a point which may help to avoid confusion later: we measured the rates at which these molecular fractions were being produced and not the biochemical composition of the cells.

METHODS

DeGray Lake (Fig. 2) was sampled on three occasions: August 31-September 2, 1982, February 1-4, 1983, and June 21-23, 1983. Water samples were collected with a submersible pump and hose and pumped into 10-L plastic cubitainers. For most incubations, water was siphoned from the cubitainer into 130-mL, clear, glass-stoppered bottles. Aliquots of 300 to 500 μL of $^{14}C\text{-}HCO_3$ (47.43-51.63 $\mu Ci \cdot mL^{-1}$, specific activity = 56.5 $\mu Ci \cdot mmol^{-1}$) were added to each bottle, and the

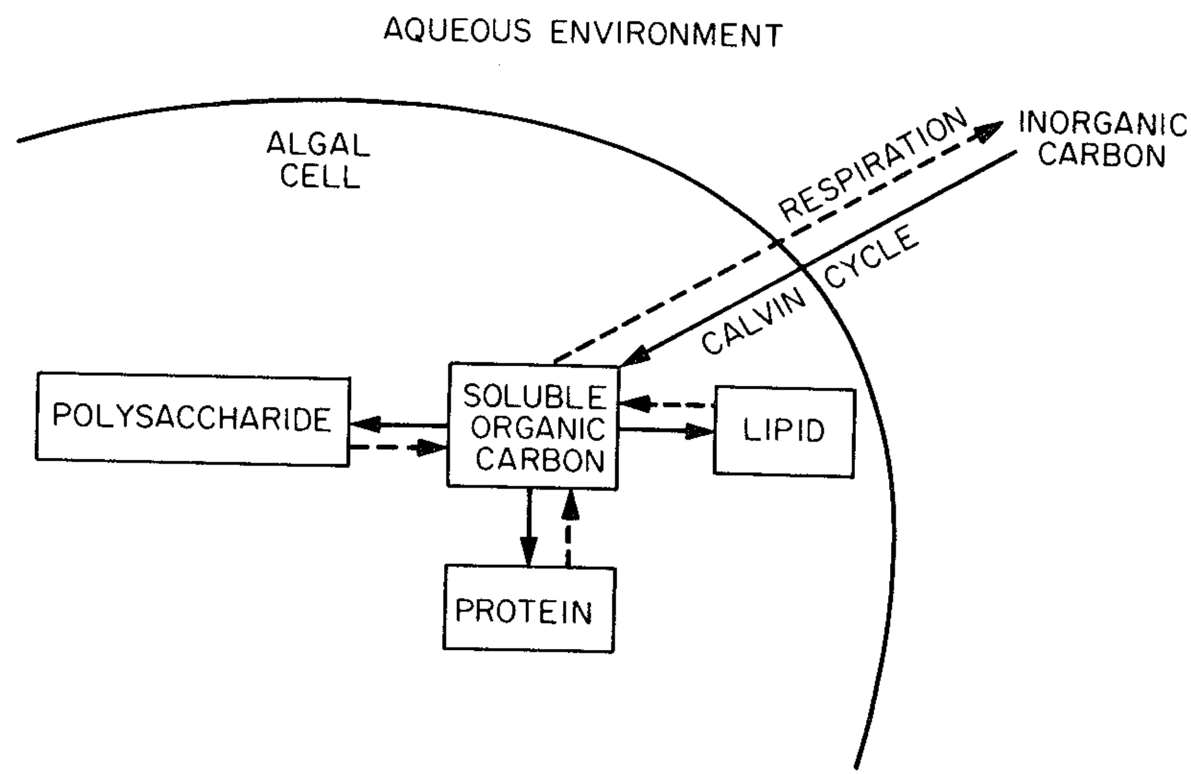

Fig. 1. Photosynthetic and respiratory flow of carbon between the major biochemical pools within an algal cell.

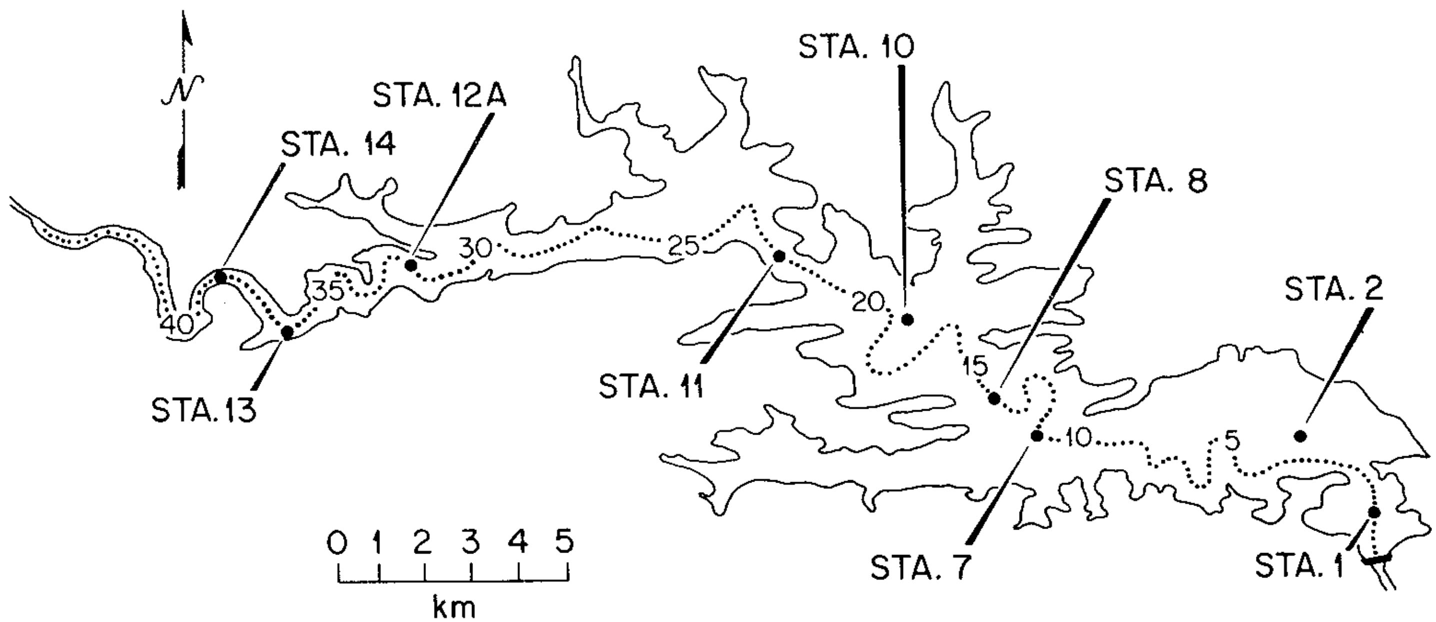

Fig. 2. Map of DeGray Lake, southcentral Arkansas, with the sampling sites designated.

bottles were hung at depth in triplicate or quadruplicate. In February, single 1120-mL bottles were used at each depth, and 4.5 mL of ^{14}C-HCO_3 (51.63 $\mu Ci \cdot mL^{-1}$) were added. Samples collected from stations located along the reservoir longitudinal axis were treated as above, but incubated at one station at a depth where light intensity approximated photosynthetically saturating levels (150-300 $\mu E \cdot m^{-2} \cdot s^{-1}$). In June, nutrient-enriched samples were also incubated in parallel with longitudinal samples. Triplicate bottles were injected with 1-mL additions of K_2HPO_4 and NH_4Cl solutions, increasing nutrient concentrations within the bottles 10 $\mu g\ P \cdot L^{-1}$ or 100 $\mu g\ N \cdot L^{-1}$. After the incubation period (3-5 h), whole-bottle volumes or subsamples were filtered through glass-fiber filters (Whatman 934-AH) under low vacuum pressure (<140 torr), and the filters were then placed in 150- by 20-mm test tubes and frozen.

The procedure for fractionating cellular constituents was modified from Li et al.(1980). After being returned to the laboratory, the filters were wetted with 0.5 mL of water, refrozen, and the cellular fractionation carried out within 2 weeks. To initiate the fractionation, test tubes containing the filters were taken from the freezer, and 0.7 mL water, 3.0 mL methanol, and 1.5 mL chloroform were added, and then the tubes were vigorously agitated on a vortex mixer for 1 min. The filters were allowed to extract for at least 2 h. The contents of the tubes were then filtered through a second glass-fiber filter (Gelman A/E, 25 mm dia) and rinsed consecutively with 1.5 mL chloroform and 1.5 mL water, with the filtrate and rinse collected in a centrifuge tube. This mixture was then shaken for 1 min on a vortex mixer and centrifuged at 1200 x g for 10 min. The lower chloroform layer was separated from the upper water-methanol layer with a Pasteur pipette, placed in a 20-mL glass scintillation vial, and dried over a N_2-gas stream. The residue was redissolved in 10 mL Aquasol II scintillation fluor. A 1-mL aliquot of the water-methanol extract was placed in a 7-mL glass scintillation vial, and 5 mL Aquasol II was added and mixed. The two remaining filters were placed in a test tube with 4 mL of 5% trichloroacetic acid (TCA), and the tube was placed in a 95°C water bath for 1 h. The TCA solution and filters were then poured onto a third glass-fiber filter (Gelman A/E, 25 mm dia.), filtered, and rinsed with an additional 4 mL of 5% TCA. A 2-mL aliquot of the TCA solution was placed in a 7-mL glass scintillation vial, and 4 mL of Aquasol II fluor was added and mixed. The three filters were placed in separate scintillation vials and fluor was added. Samples were radioassayed on a Packard 4640 or 4640C scintillation counter with external standards, and counting efficiency ranged from 90% for the chloroform-soluble residue to 82% for the hot TCA-soluble fraction. We refer to the chloroform-soluble fraction as lipid, the methanol-soluble fraction as soluble metabolites, the hot TCA-soluble fraction as polysaccharide (this fraction also includes nucleic acids), and the hot TCA-insoluble residue on the filters as protein (this fraction may also contain structural polysaccharides).

Three additional procedures were carried out in June to assess the nutrient status of the phytoplankton along the longitudinal axis of the reservoir. Phosphorus turnover times were determined by measuring $^{32}P\text{-}PO_4$ uptake rates (Lean 1973). Ammonia-enhanced dark carbon uptake was used as an indicator of nitrogen demand. Triplicate control and ammonia-spiked bottles (enriched by 100 $\mu g\ N{\cdot}L^{-1}$) were injected with 0.5 mL $^{14}C\text{-}HCO_3$ (5.46 $\mu Ci{\cdot}mL^{-1}$) and kept in the dark for 4 h (Yentsch et al. 1977). We also used ^{14}C-methyl ammonia, an ammonia analog, as a second assay of algal nitrogen demand. Five hundred, 300, 100, 50, and 25 μL of ^{14}C-methyl ammonia (2.29 $\mu Ci{\cdot}mL^{-1}$, specific activity = 25 $\mu Ci{\cdot}mmol^{-1}$) were added to replicated subsamples from each station and incubated for 30 min (Vincent 1981). All samples were filtered and radioassayed as described above. Chlorophyll a was extracted in methanol and measured spectrophotometrically (Marker et al. 1980).

RESULTS

Vertical Patterns of Photosynthetic Carbon Allocation

A carbon allocation experiment was carried out at two depths in conjunction with a vertical measurement of primary productivity at Station 10 on September 1 (Fig. 3). The water was clear (extinction coefficient = 0.51 for the epilimnion), and the 1% light level was

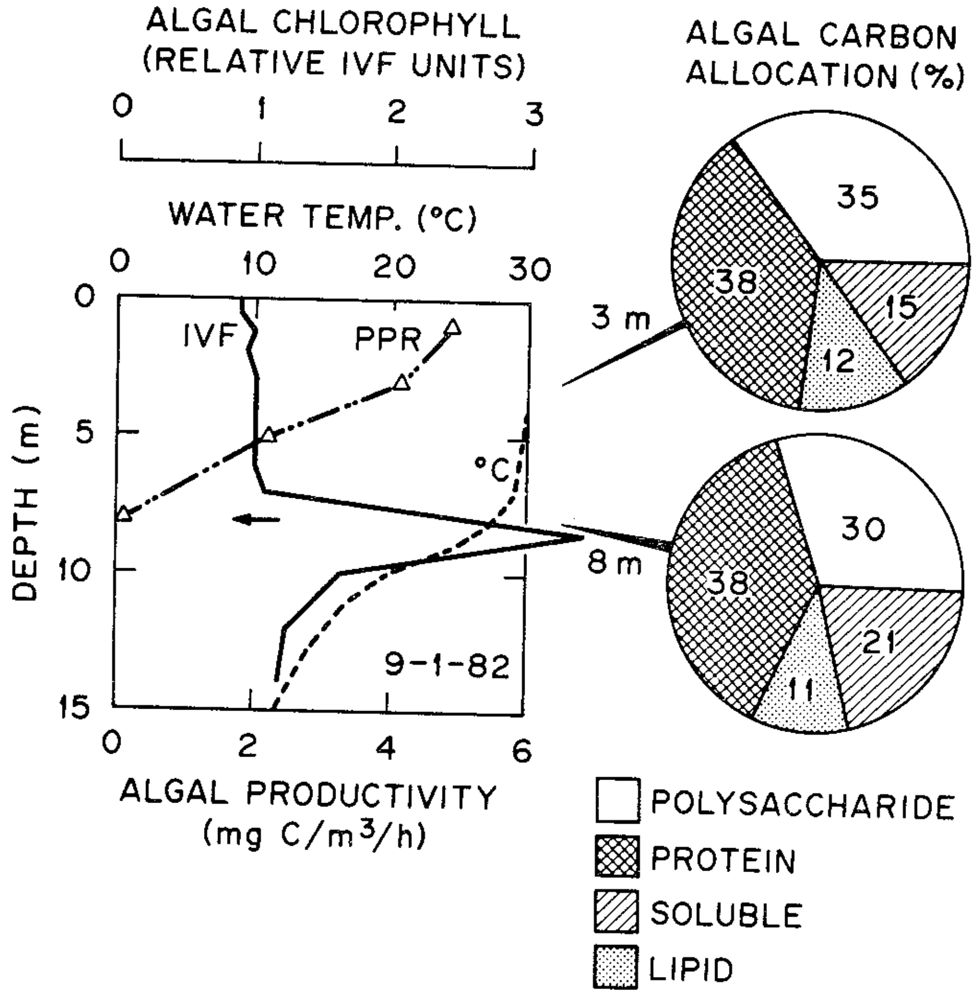

Fig. 3. Vertical profiles of phytoplankton primary productivity (PPR) in vivo chlorophyll fluorescence (IVF), temperature, and photosynthetic carbon flow into intracellular metabolic pools at Station 10, September 1, 1982 in DeGray Lake. Solid arrow indicates the depth of the euphotic layer (=1% surface light intensity).

slightly below 8 m. One sample was taken from 3 m to represent the mixed layer, and a second sample was obtained at 8 m where the level of *in vivo* fluorescence indicated a large chlorophyll peak in the upper metalimnion. Phytoplankton productivity decreased greatly between the two depths; however, carbon allocation patterns were very similar (Fig. 3). Protein constituted the largest macromolecular fraction at both depths (38%), and polysaccharide declined in relative importance with depth from 35 to 30%. The lipid and soluble metabolite pools together accounted for about 30% of the total fixed carbon.

A second vertical pattern of carbon allocation was determined at Station 10 on February 23. A large turbidity plume had moved through the reservoir in December and the water was still relatively cloudy (vertical extinction coefficient = 1.67, 1% light level = 2.5 m). Because the upper portion of the water column was well mixed, a single water sample was collected from 0.5 m and incubated at 1, 2, and 3 m. The proportion of carbon flowing into polysaccharide showed a distinct vertical pattern, making up 44% of the fixed carbon at 1 m and decreasing to 29% at 3 m (Fig. 4). The carbon flowing into protein increased with depth (from 21% at 1 m to 39% at 3 m), and the lipid fraction was greater than the protein at 1 m. Soluble metabolites accounted for <15% of the fixed carbon at all depths.

Vertical profiles were also carried out at Station 7 on June 22 and Station 1 on June 23 (Fig. 5). Vertical patterns were similar at both stations, with protein being the largest fraction at all depths (35-50%). Polysaccharide was the second most important fraction (22-33%), and lipids and soluble metabolites accounted for 20 and 10%, respectively. Vertical extinction coefficients were 0.84 and 0.74 at Stations 7 and 1, respectively, and the 1% light level was at about 6 m for both stations.

Longitudinal Patterns of Photosynthetic Carbon Allocation

Primary productivity and chlorophyll concentrations along the longitudinal axis of DeGray Lake are shown in Fig. 6 and Table 1, respectively. Patterns for longitudinal phytoplankton carbon allocation for September 1982 and February and June 1983 are shown in Fig. 7.

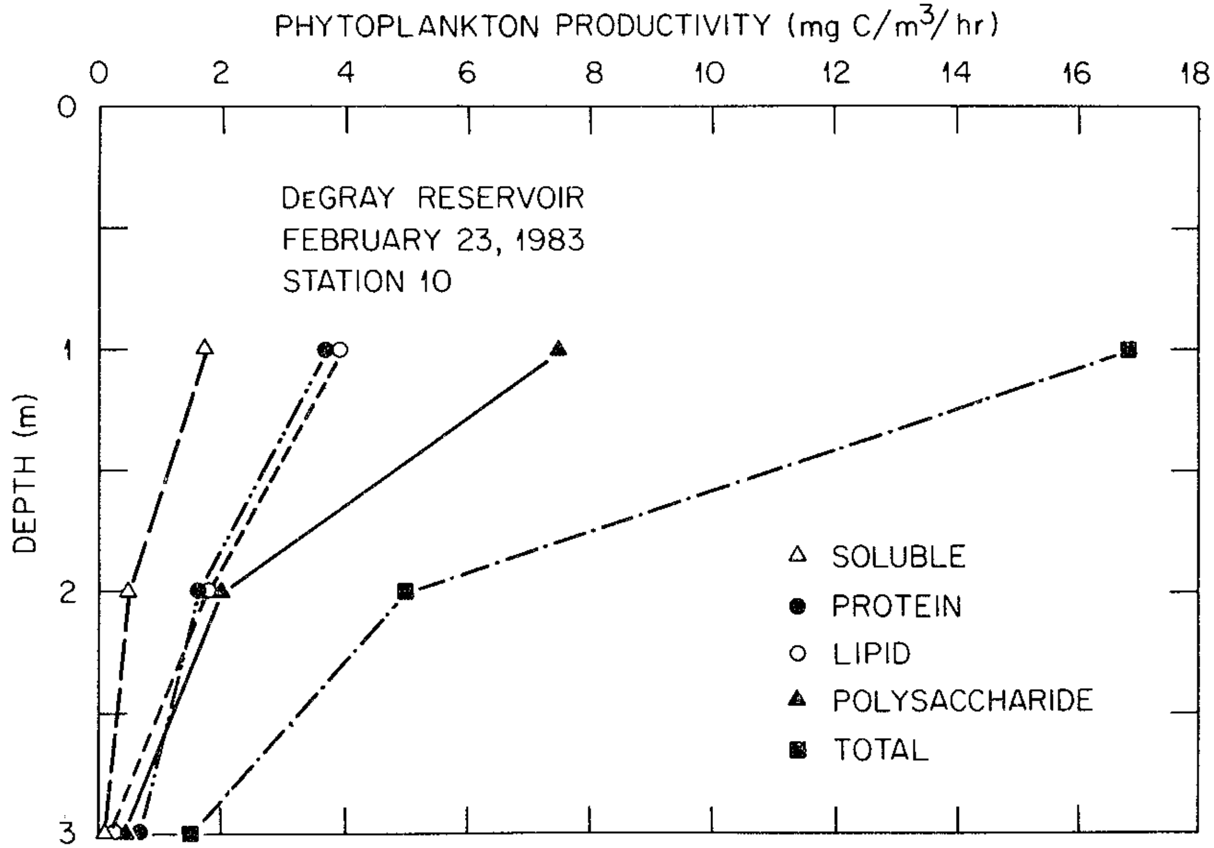

Fig. 4. Vertical profiles of phytoplankton productivity (total carbon uptake) and photosynthetic carbon allocation at Station 10, February 23, 1983 in DeGray Lake.

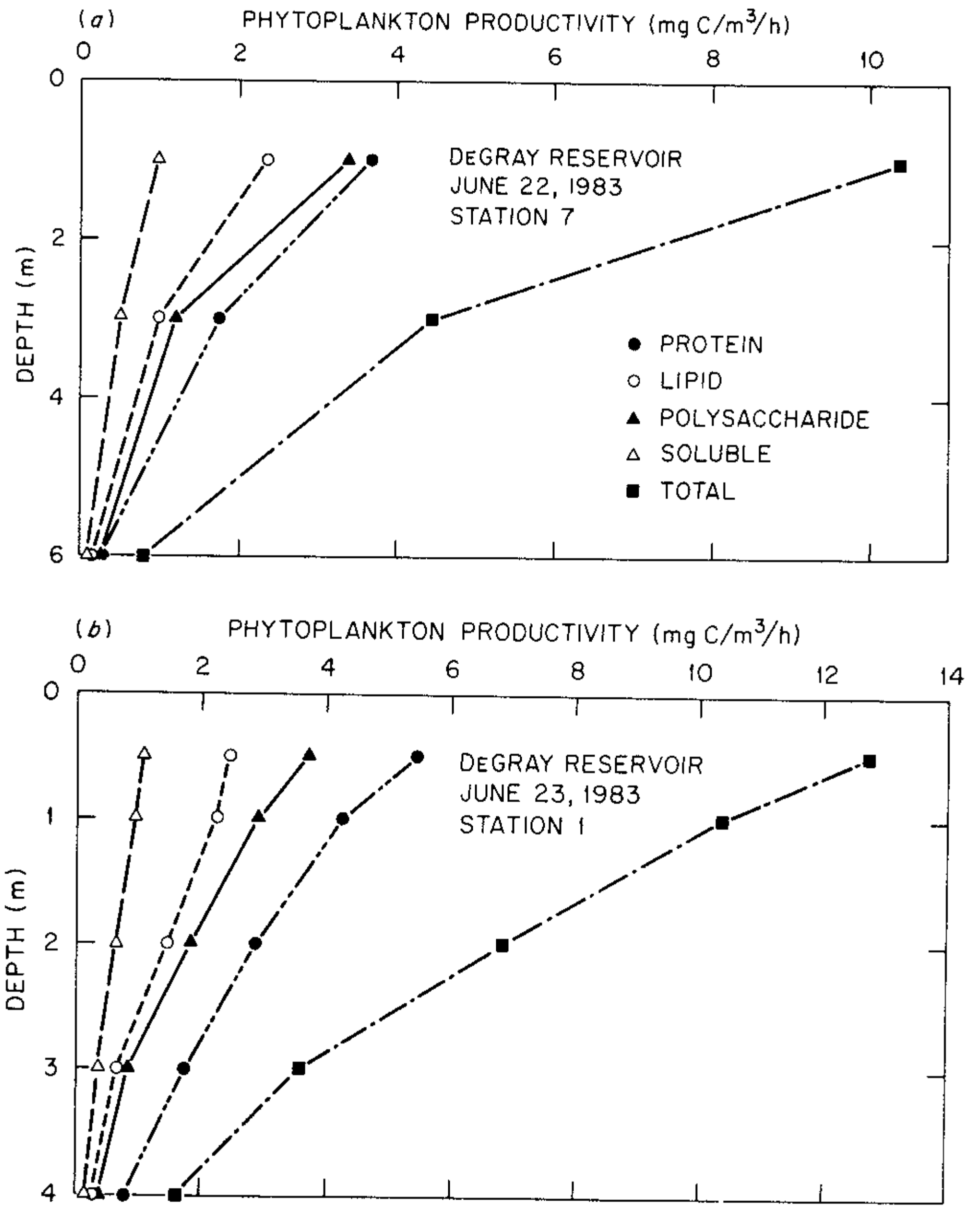

Fig. 5. Vertical profiles of phytoplankton productivity (total carbon uptake) and photosynthetic carbon allocation at (a) Station 7 and (b) Station 1 in DeGray Lake on June 22-23, 1983.

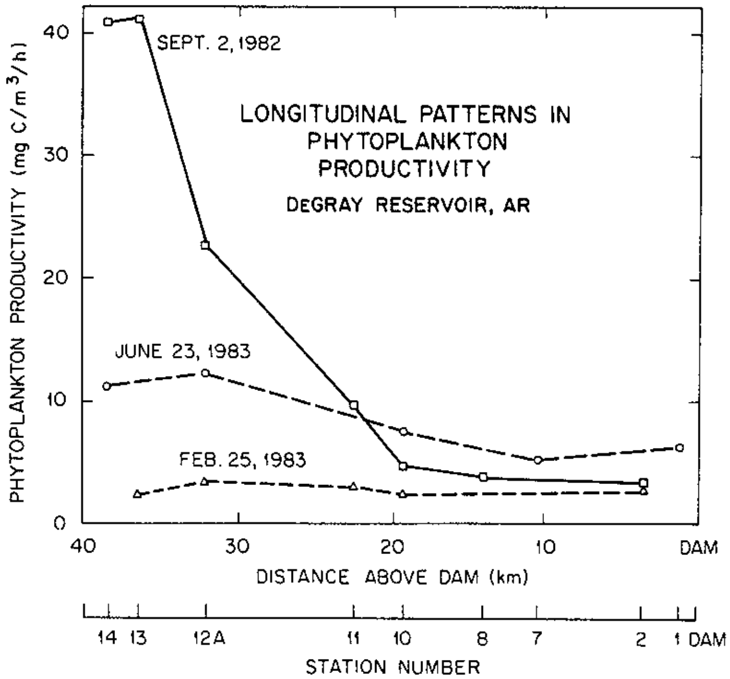

Fig. 6. Seasonal patterns in the longitudinal distribution of phytoplankton productivity (expressed on a volumetric basis) in DeGray Lake.

Table 1. Values of chlorophyll a (mg m^{-3}), methyl ammonia uptake (percent of 100 nmol taken up per hour), ammonia-enhanced dark carbon uptake (percent stimulation of carbon uptake over control), and PO_4 turnover time (h) for stations located along the longitudinal axis of DeGray Lake.

		Stations							
		14	13	12A	11	10	7	2	1
Sept. 82	Chlorophyll	-	-	5.0	-	1.9	-	1.2	-
Feb. 83	"	<1.0	<1.0	1.0	1.3	-	1.3	-	
June 83	"	10.5	-	9.3	-	6.2	5.4	-	5.4
	Methyl ammmonia uptake	1.64	-	1.92	-	1.92	3.23	-	5.41
	Dark C enhancement	-25	-	105	-	74	120	-	47
	PO_4 turnover time	>2	-	>2	-	>2	>2	-	>2

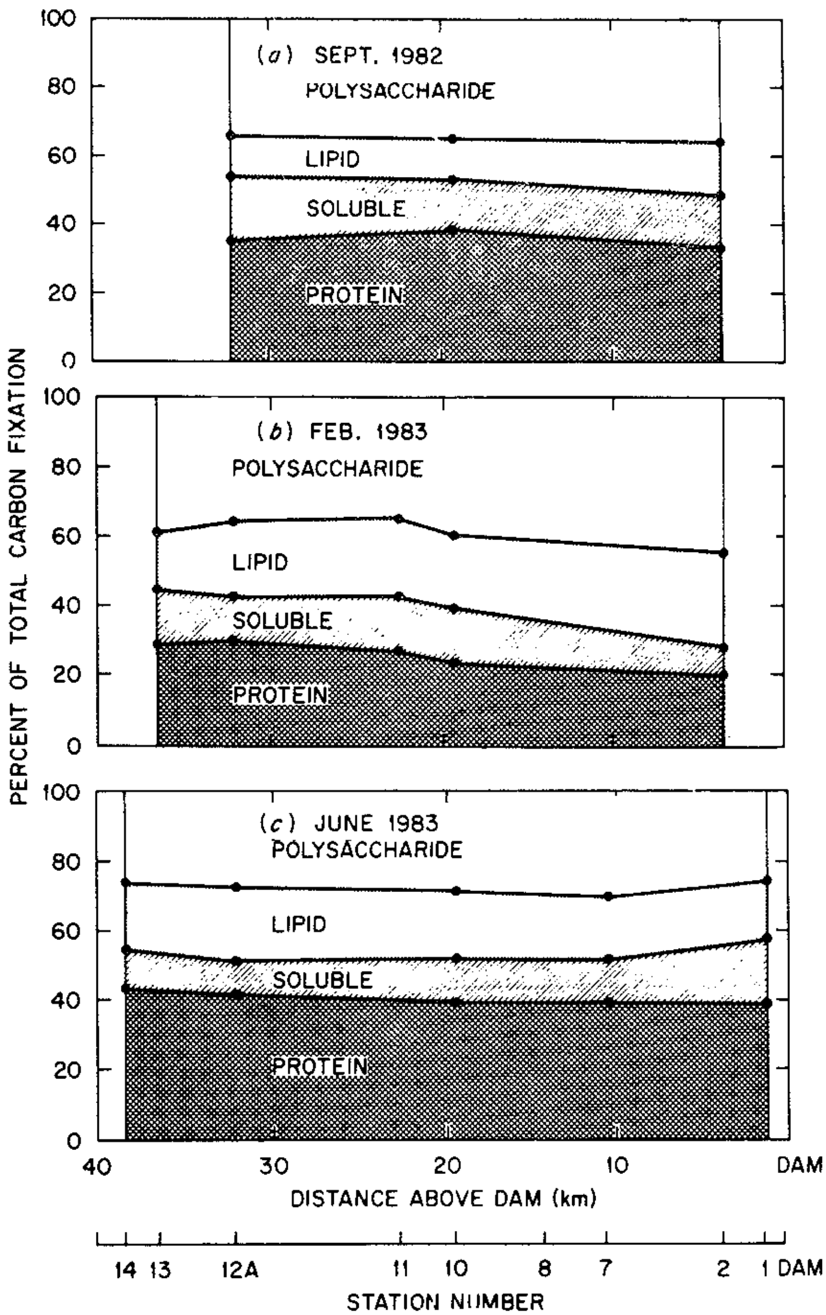

Fig. 7. Longitudinal patterns of photosynthetic carbon allocation into intracellular metabolic pools of DeGray Lake phytoplankton: (a) September 1982, (b) February 1983, and (c) June 1983.

Longitudinal trends for each sampling period were similar to those described above for vertical profiles. In September, the protein and polysaccharide were both in the mid-to-high 30% range, with protein being greater than polysaccharide at Stations 12A and 10. Lipids and soluble metabolites ranged from 11 to 19%. In February, from 35 to 45% of the photosynthetically fixed carbon was flowing into polysaccharide, 20 to 29% into protein, 17 to 28% into lipid, and 8 to 16% into soluble metabolites. Relative polysaccharide and lipid production was greatest, and protein correspondingly least, downlake at Station 2. In June, protein again was the largest fraction, making up 38 to 48% of the fixed carbon; polysaccharide constituted 25 to 31%, lipid 17 to 21%, and soluble metabolites 10 to 19%.

Longitudinal Patterns of Nutrient Status and Response to Nutrient Enrichment

During the June sampling, we attempted to assess longitudinal patterns in phytoplankton nutrient deficiency in DeGray Lake. No significant ^{32}P uptake was measured at any of the stations over the incubation and, therefore, estimated PO_4 turnover times exceeded the 2-h incubation period. Methyl ammonia uptake increased from uplake to downlake stations in a stepwise fashion (Table 1). Ammonia enhancement of dark carbon uptake was low at Station 14 and much higher at downlake stations (Table 1). Responses to nutrient (PO_4 and NH_4) additions also showed distinct patterns (Fig. 8). Carbon fixation in phosphorus-enriched samples was depressed relative to controls at all stations, while carbon fixation in the ammonia-enriched samples was about equal at Stations 14, 7, and 1, but slightly less than controls at 12A and 10. From relative changes in the metabolic pools (Fig. 8), phosphorus enrichment appears to have had little effect on altering the pathways of recently reduced carbon, but acted only to reduce the rate of photosynthesis. Ammonia enrichment, conversely, altered patterns of carbon flow relative to the control and had a similar effect at each station. Both the absolute rate and the proportion of carbon flow into the lipid pool were reduced at each station. At Stations 14, 7, and 1, where total carbon fixation in control and ammonia-enriched samples was

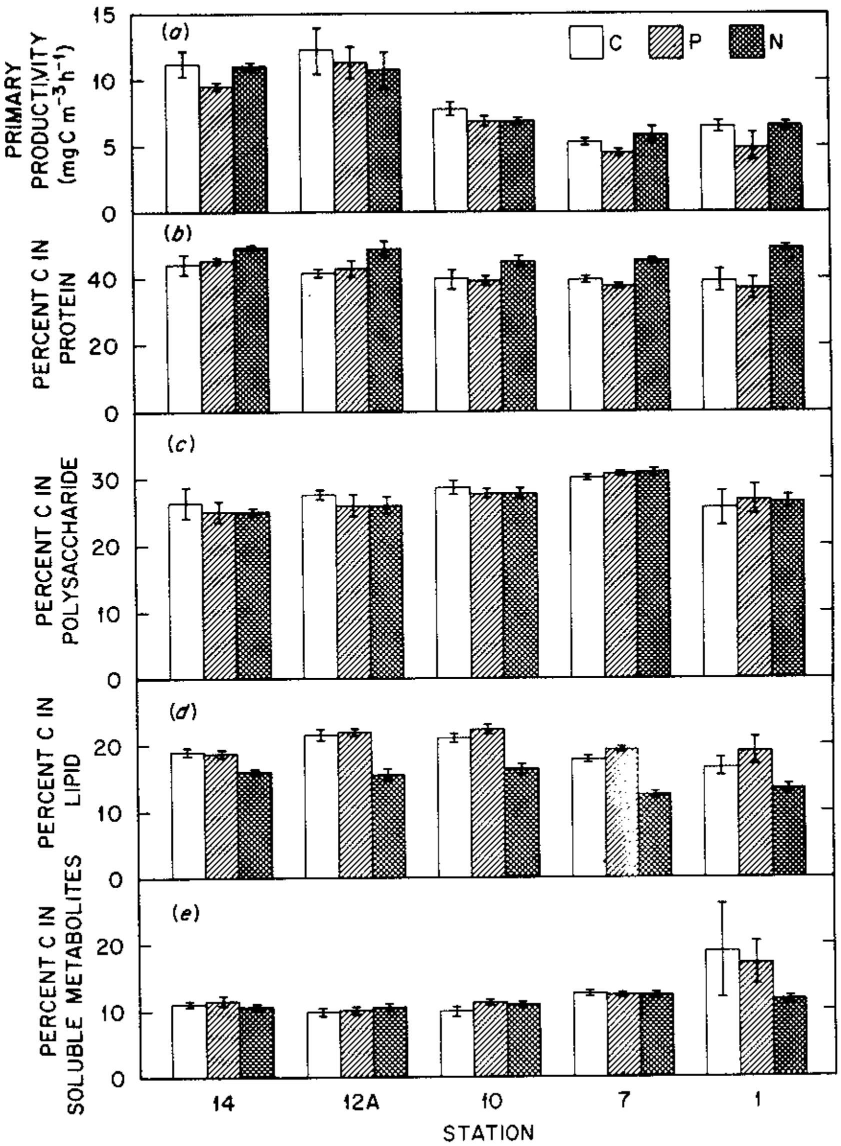

Fig. 8. Longitudinal patterns in photosynthetic carbon allocation responses to experimental phosphorus and nitrogen enrichments in DeGray Lake, June 23, 1983. The upper panel (a) shows total carbon fixation rates for stations located along the reservoir longitudinal axis; C = control (unenriched) samples, P = phosphorus-enriched samples, N = nitrogen-enriched samples. Lower panels indicate the fraction of the total carbon uptake allocated to intracellular pools of protein (b), polysaccharide (c), lipid (d), and soluble metabolites (e).

about the same, the decrease in lipid synthesis was approximately balanced by increased protein synthesis. Rates of polysaccharide and soluble metabolite accumulation were relatively insensitive to ammonia enrichment.

DISCUSSION

Vertical profiles of carbon allocation patterns for DeGray Lake phytoplankton consistently indicated an increase in the proportion of the recently fixed organic carbon flowing into the polysaccharide pool with increasing light intensity (Figs. 3, 4, and 5). This observation is consistent with a larger set of data from four Tennessee reservoirs (Groeger and Kimmel, in preparation) and with the results of similar measurements by others in a variety of marine and freshwater environments (Morris et al. 1974, Gibson 1975, Konopka and Schnur 1980, Li et al. 1980). Photosynthesis rates exceeding rates of protein synthesis (and/or rates of nitrogen uptake) may result in temporary storage of reduced carbon as polysaccharide and, consequently, polysaccharide accumulation at high light intensities. Additionally, at high light levels, photosynthetic carbon reduction greatly in excess of the organic carbon, or reducing power, needs of algal cells may help protect the cellular light-capturing (photochemical) machinery by dissipating energy. Myers (1946) observed that maximum algal growth rates are achieved at light levels less than those needed to saturate photosynthesis, suggesting that the large amounts of polysaccharide produced at high light intensities are not critical for growth processes.

A second relationship emerging from our data on short-term carbon allocation patterns in phytoplankton from DeGray and other reservoirs is that under high (but not inhibiting) light intensities, the polysaccharide to protein ratio increases with an improved physiological state of the phytoplankton assemblage (Groeger and Kimmel, in preparation). Determinations of the biochemical composition of cultured algae (Cook 1963, Fogg 1975) and naturally occurring phytoplankton assemblages (Barlow 1982) suggest an opposite trend: the relative proportion of protein in algal biomass tends to increase (and the polysaccharide:protein ratio decreases) during favorable growth conditions. However, as the physiological state of the DeGray Lake phytoplankton declined, probably due to nutrient deficiencies, polysaccharide:protein synthesis rates neared or dropped below 1. Assimilation numbers (mg C fixed•mg Chl $a^{-1} \cdot h^{-1}$) at saturating light intensities in September 1982 and June 1983 were <2 (except at Station 12A during September), low values for warm water temperatures (Curl and Small 1965; Groeger and Kimmel, in preparation) indicative of phytoplankton in poor physiological condition. Correspondingly, these low assimilation numbers were accompanied by low polysaccharide:protein ratios (1.0 and 0.68 in September and June, respectively) for recently fixed carbon. In contrast, February 1983 values for polysaccharide:protein ratios (1.5) and assimilation numbers (>2.5 at a much lower water temperature) indicated a healthier algal community.

Although longitudinal gradients in phytoplankton productivity were apparent in September 1982 and June 1983 (Fig. 6), carbon allocation patterns indicated that DeGray Lake phytoplankton assemblages were in a similar physiological condition throughout most of the impoundment (Fig. 7). This physiological uniformity in the DeGray Lake phytoplankton was somewhat unexpected, since we have observed marked longitudinal changes in algal physiological status (as indicated by nutrient deficiency assays, photochemical capacity, assimilation number, and carbon allocation patterns) in other reservoirs during the growing season (Groeger, unpublished data; Kimmel, unpublished data). However, DeGray Lake is more dominated by lacustrine conditions during the growing season than impoundments having higher rates of river inflow and shorter water residence times. In September 1982 and June 1983, the lacustrine zone in DeGray Lake extended approximately 30-40 km upstream from the dam and constituted more than 80% of the total reservoir surface area (Fig. 2).

Results of nutrient deficiency assays conducted in June 1983 (Table 1) suggest that mixed-layer phytoplankton in the DeGray Lake lacustrine zone (i.e., downlake of Station 13) were nitrogen limited.

Phosphorus turnover times (>2 h at all stations) greatly exceeded values reported for phosphorus-limited lakes (Lean and Nalewajko 1979) and indicated that, relative to its rate of supply, phosphorus was not in high demand by the DeGray Lake phytoplankton. Dark carbon uptake responses to ammonia enrichment exceeded 100% over unenriched controls at two lacustrine stations (12A and 7), the level generally considered to indicate nitrogen limitation (Yentsch et al. 1977). Values for methyl ammonia uptake increased from uplake to downlake and were relatively high compared to those recorded in Normandy Lake, a nitrogen-limited reservoir in southcentral Tennessee. Carbon flow into lipids was also elevated on this date compared to values for the larger data set (Table 2). In Normandy Lake we have observed enhanced carbon flow into lipid production, increasing up to a maximum of >35% late in the growing season when nitrogen deficiency is most severe (Groeger and Kimmel, in preparation).

When considered together, these data indicate nitrogen-limited phytoplankton growth in the lacustrine zone of DeGray Lake during June 1983. This state of nitrogen deficiency may have been induced by a large turbidity plume associated with a December storm (J. Nix, personal communication), which reached all the way to the dam as an overflow and was likely concurrent with a large phosphorus input [95% of the phosphorus entering DeGray Lake is in particulate form (Montgomery and Kennedy, this volume)]. Total phosphorus levels in DeGray Lake are normally highest in the spring (Kennedy, this volume), which suggests that development of phytoplankton nitrogen deficiency in the lacustrine zone could be common in spring or early summer, even though annual nutrient loading values are characteristic of a phosphorus-limited system.

In algae under laboratory conditions, increasing the intracellular ammonia concentration stimulates carbon flow into the tricarboxylic acid cycle and thereby enhances amino acid and protein production (Kanazawa et al. 1970). Results of the experimental ammonia-N enrichment that we conducted in June indicate that the introduction of reduced inorganic nitrogen stimulated protein production in natural phytoplankton assemblages at the cost of lipid production (Fig. 8).

Table 2. Relative proportions of photosynthetic carbon production (expressed as a percent of the total production) in intracellular biochemical pools. The general data set and P_{max} subset (values determined for samples at or near saturating light intensities) consist of results from a number of aquatic systems (Groeger and Kimmel, unpublished data). Standard deviations for the means are in parentheses.

	n	Protein	Polysaccharide	Lipid	Soluble metabolites
General data set	309	32.8 (8.8)	38.5 (9.3)	14.7 (6.1)	14.0 (5.2)
P_{max} subset	98	28.7 (7.0)	41.8 (9.2)	16.3 (6.6)	13.4 (5.8)
DeGray data					
Sept 82	3	35 (1.5)	35 (2.5)	13 (1.3)	17 (2.1)
Feb. 83	5	26 (1.9)	39 (1.2)	22 (0.6)	14 (1.7)
June 83	5	41 (1.5)	28 (1.1)	19 (0.6)	13 (2.1)

This suggests that increased intracellular ammonia directly or indirectly acts to switch carbon from fatty acid synthesis into the tricarboxylic acid cycle, and, conversely, cellular depletion of nitrogen increases lipid synthesis at the cost of protein. Results from Normandy Lake show that ammonia additions have no effect on the rate of total carbon fixation or on lipid production in phytoplankton communities that are not nitrogen limited. Therefore, our preliminary results suggest that the measurement of carbon allocation patterns may be a particularly sensitive tool for detecting nitrogen deficiency in phytoplankton and for determining the degree of physiological stress imposed by that deficiency.

Depression of photosynthetic rates upon phosphorus enrichment of phytoplankton samples has been observed by others (Healy 1979, Lean and Pick 1981). Lean and Pick (1981) attributed this depression to the shunting of photosynthetically generated ATP from carbon reduction into active transport of phosphorus. If this were the major process inhibiting carbon fixation, carbon allocation patterns, which are very sensitive to the relative sizes of the various phosphorylated-adenylate pools, should be markedly altered. However, the results of our phosphorus-enrichment experiments in DeGray Lake indicate that this is not the case. A better physiological explanation may be that phosphorus additions greatly increase the phosphate concentration within the cytoplasm and then, through a triosephosphate-phosphate translocator, the phosphate is moved into the chloroplast at the same time Calvin cycle intermediates are released from the chloroplast into the cytoplasm. This interrupts the capability of the Calvin cycle to regenerate and recycle its intermediate molecules and thus temporarily depresses carbon fixation. This mechanism has been well documented in both isolated chloroplasts and whole cells (Heber 1974, Heldt et al. 1977) and is present in at least some algae (Klein et al. 1983).

SUMMARY

In addition to yielding insights into the metabolic and physiological responses of phytoplankton assemblages to environmental conditions (e.g., light and nutrient availability), our determinations of photosynthetic carbon allocation patterns suggest the following conclusions in regard to the phytoplankton community of DeGray Lake:

1. Within the mixed layer of the lacustrine zone, DeGray Lake phytoplankton exhibited a relatively uniform physiological condition (as indicated by polysaccharide:protein production ratios) along the longitudinal axis of the impoundment. Judging from our experience with other reservoirs, we suggest that this uniformity resulted from the extension of lacustrine conditions throughout most of the reservoir during the growing season.
2. DeGray Lake phytoplankton assemblages were in poor physiological condition during the growing season (as indicated by our September 1982 and June 1983 data), probably as a result of nutrient deficiency.
3. Results of nutrient deficiency assays and measurements of photosynthetic carbon allocation suggest that DeGray Lake phytoplankton assemblages were nitrogen limited in June 1983. Whether this nitrogen limitation represents an unusual event is unknown; however, the reservoir is generally thought to be phosphorus limited. Presumably, a similar suite of measurements later in the growing season would have indicated phosphorus, rather than nitrogen, limitation. However, the occurrence of relatively high total phosphorus levels associated with winter and spring inflows to the reservoir (Montgomery and Kennedy, this volume) suggests that nitrogen-limited phytoplankton growth may occur commonly early in the growing season.

ACKNOWLEDGMENTS

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SPATIAL AND TEMPORAL DISTRIBUTION OF ZOOPLANKTON IN DEGRAY RESERVOIR, ARKANSAS [1]

by

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Abstract

Vertical, stratified zooplankton samples were taken at monthly intervals from six selected stations in DeGray Reservoir during 1976-81. Temperature, specific conductance, dissolved oxygen concentration, and Secchi disk transparency also were monitored during each sampling period. Qualitative

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and quantitative analyses of zooplankton samples included identification, enumeration, and dry weight determinations. Data were standardized and analyzed using one-way analyses of variance and Duncan's Multiple Range Test. Total zooplankton concentrations were greatest at up-lake stations, but minimal in the uppermost riverine location. Concentrations of entomostracans at up-lake stations did not differ significantly from those at down-lake stations only during summer months. Dry weight values were more uniform among stations than were concentration values, except for contrastingly low values at the uppermost riverine station. Zooplankton concentrations and dry weight values peaked in spring. Concentrations of entomostracans corresponded to this unimodal pattern to account for the spring peak in total dry weights, but rotifers displayed a trimodal abundance pattern. The greatest zooplankton concentrations, as well as the highest dry weight values, were recorded in 1980. Effects on zooplankton assemblages of changing to a hypolimnial from an epilimnial release mode were not clearly apparent.

As the proportion of our fresh-water resources contained in reservoirs increases, so does our dependence on reservoirs for recreation, water supplies, power generation, and flood control. Our knowledge of large hydropower reservoir ecosystems is less complete than for other kinds of lake ecosystems, due primarily to their relatively short history and unique characteristics by way of comparison to natural lakes (oxbows are the only natural lakes occurring in Arkansas). Long-term studies provide data, which when carefully interpreted, help to elucidate reservoir responses to various (and varying) physical, chemical, and biological conditions.

During the six-year period considered in this paper, 2,350 zooplankton samples were taken. Thus, time and space limitations forced us to treat only broad taxonomic categories, unless exceptional patterns were recognizable at more specific levels. Our statistical approach was designed to identify trends in vertical and horizontal distributions of zooplankton, as well as annual and apparent long-term cycles. Because standardization was necessary for comparative purposes, the data presented are unitless; e.g., we found significantly greater concentrations of zooplankton near the lake surface than near the bottom, but quantitative differences (e.g., organisms/liter) are not readily discernible from the transformed data. Accordingly, the purpose of this presentation is to provide a qualitative description of spatial and temporal patterns of zooplankton in DeGray Reservoir, Arkansas, for the six-year period 1976-1981. Possible interactions among zooplankters, phytoplankters, fishes, water levels, etc., are not considered.

Materials and methods

Field and laboratory procedures--Zooplankton samples were taken at least once each month, April 1976 through July 1981, from six fixed locations on DeGray Lake by U.S. Fish and Wildlife personnel. The locations sampled were stations designated 1, 4, 7, 10, 12, and 14, station 1 being nearest to the dam and station 14 being near the inflowing Caddo River (Fig. 1). Vertical, stratified samples of zooplankton were taken using a specially constructed Birge closing net (0.5-m-diameter mouth aperture; no. 20 Nitex mesh) at 5-0 m (interval 1), 10-5 m (interval 2), 15-10 m (interval 3), 20-15 m (interval 4), and 2 m off the bottom to 20 m (interval 5). Samples were preserved in 3% formalin in the field. In the laboratory, samples were diluted or concentrated to specific volumes--typically to 100 ml in 3% formalin, although volumes of concentrates varied from 25-250 ml depending on apparent relative zooplankton densities. All metazoan zooplankters in each of two 1-ml Sedgewick-Rafter subsamples taken from each sample were identified and enumerated. Larvae of *Chaoborus* sp. were counted directly from each sample concentrate. Total zooplankton dry weights were determined from each sample concentrate using the sucrose density-gradient procedure described by Schmitz and McCraw (1981). Physical and chemical data, including temperature, specific conductance, dissolved oxygen

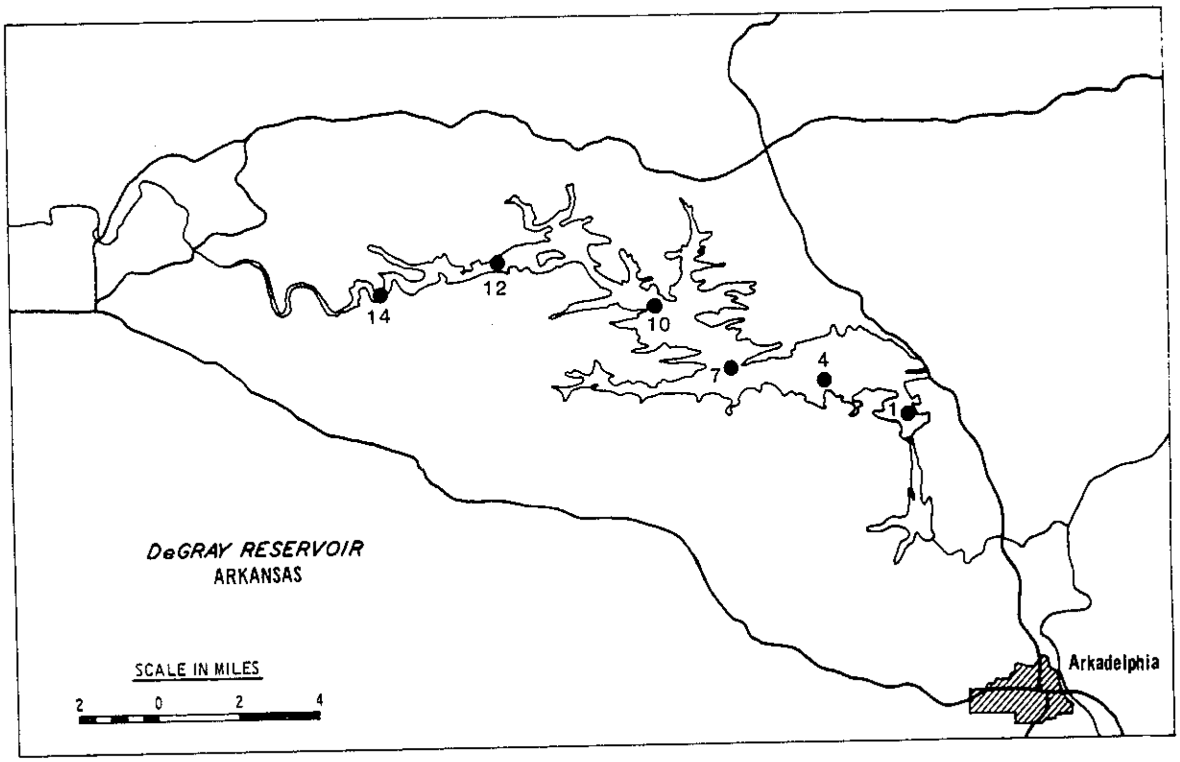


FIG. 1. Map of DeGray Reservoir, Arkansas, showing locations of zooplankton sampling stations.

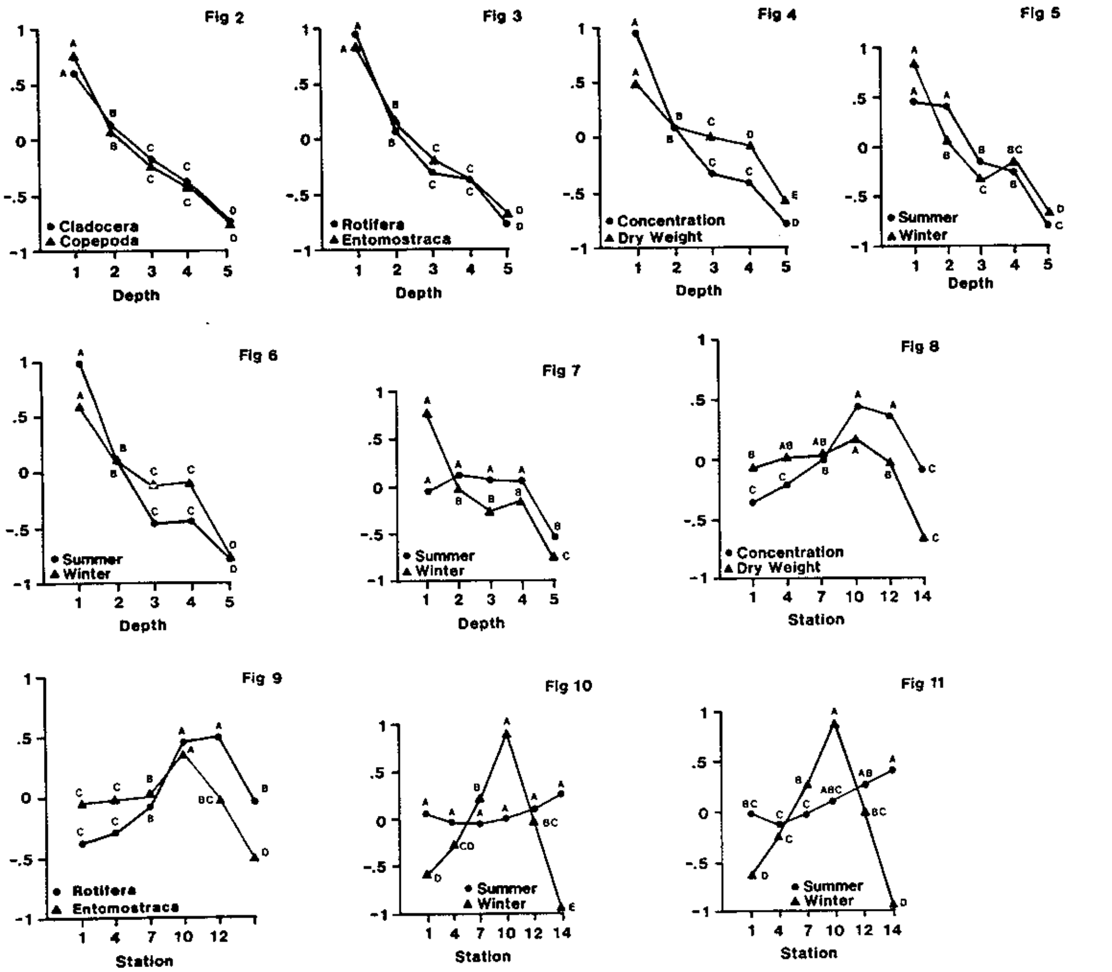
concentration, and Secchi disk transparency, were collected in the field by U.S. Fish and Wildlife Service personnel.

Statistical procedures--Data in three of the four dimensions sampled (i.e., vertical, horizontal, month, and year) were alternately standardized to a mean of 0 and a standard deviation of 1 so that the effects of a single dimension could be isolated. Analyses were by the SAS (Barr et al. 1979) with one-way analyses of variance and Duncan's Multiple Range Test. The level of significance for all tests was 0.05. Data for 1976 and 1981 were not included in analyses when the effects of year was tested because samples for all months of these years were not available. For seasonal definitions, the months December, January, and February were considered to be winter and summer consisted of July, August, and September.

Results

Vertical distribution--Zooplankton concentrations and dry weights decreased with increasing depth. Designated variables for purposes of this study included concentrations of Copepoda, Cladocera, Entomostraca (Copepoda and Cladocera considered together), Rotifera, and total zooplankton, as well as zooplankton dry weights. All variables displayed greater values in depth interval 1 (see Figs. 2, 3, 4), and when depth intervals were ranked in order of decreasing values, the results were 1 (5-0 m), 2 (10-5 m), 3 (15-10 m), 4 (20-15 m), and 5 (2 m off bottom-20 m) in the case of each variable. For mean zooplankton dry weights, Duncan's test revealed that greater values could be found in any given depth interval than for any deeper interval; thus, mean dry weights were significantly different among all depth intervals sampled. *Chaoborus* sp. and *Daphnia schødleri* were the only zooplankters exhibiting abundance peaks in deeper water. *Chaoborus* sp. concentrations were significantly greater within depth intervals 2 and 3 than within interval 1. Significantly higher concentrations of *D. schødleri* were observed in depth intervals 2, 3, and 4, with lower concentrations occurring in intervals 1 and 5.

For several taxa, vertical distributions varied seasonally.



FIGS. 2-11. Graphic analyses of data on concentrations and total dry weights of zooplankton in DeGray Reservoir, Arkansas, subjected to Duncan's Multiple Range Test. Data standardized to a mean of 0 and a deviation of 1. Means represented by the same letters in alignment are not significantly different (α=0.05). Fig. 2. Vertical distribution of Cladocera and Copepoda. Fig. 3. Vertical distribution of Rotifera and Entomostraca. Fig. 4. Vertical distribution of total zooplankton concentrations and total dry weights. Fig. 5. Vertical distribution of Entomostraca in summer and winter. Fig. 6. Vertical distribution of Rotifera in summer and winter. Fig. 7. Vertical distribution of zooplankton total dry weights in summer and winter. Fig. 8. Horizontal distribution of total zooplankton concentrations and total dry weights. Fig. 9. Horizontal distribution of Rotifera and Entomostraca. Fig. 10. Horizontal distribution of Entomostraca in summer and winter. Fig. 11. Horizontal distribution of Copepoda in summer and winter.

Especially contrasting were summer and winter patterns, not surprising when one considers thermally stratified vs. unstratified conditions. For example, winter concentrations of Daphnia galeata were greatest in depth interval 1 (intervals ranked 1, 2, 3, 4, and 5), but summer concentrations were significantly higher in intervals 4, 3, and 2, with lowest concentrations occurring in intervals 1 and 5. D. schødleri also exhibited highest winter and lowest summer concentrations within depth interval 1. Entomostracans were more evenly distributed throughout the first 10-m depth in the summer than in winter; i.e., concentrations within depth interval 2 did not differ significantly from those in interval 1 during summer (Fig. 5). Duncan's test showed that the vertical distribution of rotifers was virtually the same during summer and winter (Fig. 6). The same pattern emerged when all months of the year were combined. In fact, vertical distribution patterns displayed by the rotifers emerged when total zooplankton concentration was considered.

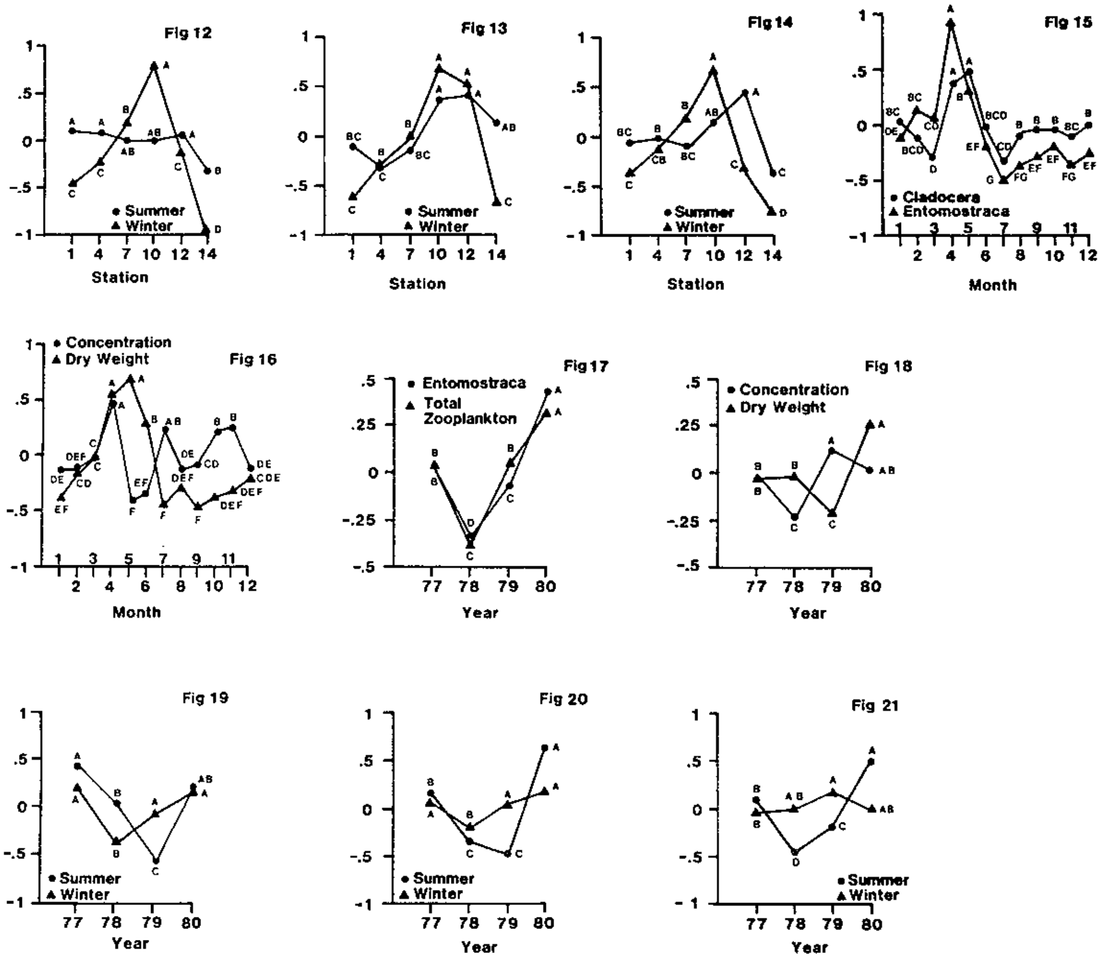
Summer and winter contrasts between vertical distributions of zooplankton dry weights were marked. Winter mean dry weights were concentrated significantly greatest within depth interval 1, while significant differences were virtually nonexistent during summer among depth intervals 1, 2, 3, and 4 (Fig. 7).

Although the reservoir was not thermally stratified during the winter months, temperatures differed significantly among depth intervals. Temperatures within interval 1 were significantly higher, and those within three of the four others differed significantly from one another. Intervals could be ranked by decreasing mean temperatures the same for winter as for summer (1, 2, 3, 4, and 5).

Vertical distribution patterns of zooplankton abundance and zooplankton dry weights at station 1 near the dam were similar to the patterns observed when data from all stations were considered.

Horizontal distribution--A general parallel pattern of horizontal distribution was apparent in both total zooplankton concentrations and total dry weights. Lowest values occurred at the uppermost riverine station(14); peak values characterized stations 12 and 10, decreasing progressively downlake (Fig. 8). Maximal mean dry weights were apparent at station 10 with gradually decreasing, but not significantly different, values downlake. However, the mean dry weight of total zooplankton at the riverine station 14 was significantly lower than realized for any other location (Fig. 8). This same basic pattern was reflected in concentrations of entomostracans with low values at station 14, significantly greater values at station 10, and a gradual decrease toward the dam (Fig. 9). Cladoceran concentrations were not significantly different among stations 1, 4, 7, and 10, although a pattern of decreasing values downlake from station 10 could be discerned; station 12 yielded a significantly lower mean concentration than down-lake stations, but they were significantly higher than those at station 14. Rotifer concentrations decreased downlake from a peak at station 12. The mean concentration of rotifers at station 14 was lower than those at stations 10 and 12 but higher than those of stations 1, 4, and 7 (Fig. 9).

Horizontal distribution patterns of entomostracans were markedly different in summer and winter. Winter was characterized by a strong unimodal peak at station 10 with significantly decreasing concentrations uplake and downlake (Fig. 10), whereas mean concentrations were not significantly different between any two stations during summer (Fig. 10). Copepod concentrations peaked at station 14 during summer; values declined downlake with the exception of a slightly greater value at station 1 over station 2 (Fig. 11). Winter concentrations of copepods paralleled entomostracans with a strong unimodal peak at station 10 and significantly lesser values at stations 1 and 14 (Fig. 11). With the exception of station 14, summer concentrations of cladocerans were relatively uniform, while the winter pattern corresponded closely to that of the copepods, having a maximal mean concentration at station 10 and minimal means at stations 1 and 14 (Fig. 12). Summer and winter patterns of rotifer distribution were similar with maximum concentrations at stations 10 and 12; however, summer concentrations at stations 1 and 14 increased relative to other stations (Fig. 13).



FIGS. 12-21. Graphic analyses of data on concentrations and total dry weights of zooplankton in DeGray Reservoir, Arkansas, subjected to Duncan's Multiple Range Test. Data standardized to a mean of 0 and a deviation of 1. Means represented by the same letters in alignment are not significantly different (α=0.05). Fig. 12. Horizontal distribution of Cladocera in summer and winter. Fig. 13. Horizontal distribution of Rotifera in summer and winter. Fig. 14. Horizontal distribution of zooplankton total dry weights in summer and winter. Fig. 15. Seasonal distribution of Cladocera and Entomostraca. Fig. 16. Seasonal distribution of rotiferan concentrations and zooplankton total dry weights. Fig. 17. Annual mean concentrations of Entomostraca and total zooplankton. Fig. 18. Annual distribution of rotiferan concentrations and zooplankton total dry weights. Fig. 19. Summer and winter distributions of Entomostraca by year. Fig. 20. Summer and winter distributions of total zooplankton concentrations. Fig. 21. Summer and winter distributions of Rotifera by year.

The mean dry weight pattern for total zooplankton in winter corresponded to the unimodal distribution of entomostracans showing a maximal value at station 10 (Fig. 14). Mean dry weights peaked at station 12 in summer, declining significantly in both up- and down-lake directions (Fig. 14).

Annual cycles and multi-year patterns--The maximal mean concentration of entomostracans occurred in April. Lesser, but significantly high, concentrations occurred in February and May (Fig. 15). Both April and May were months of significant cladoceran maxima (Fig. 15). Noteworthy were contrasting maximal concentrations of *Bosmina* in January and minimal populations in July. Rotifers displayed a trimodal pattern with a spring peak in April, a summer peak in July, and a fall peak during October and November. Minimal mean concentrations of rotifers occurred in May and June--between the spring and summer peaks; maximal and minimal concentrations occurred in consecutive months, April and May (Fig. 16). Mean concentrations of *Chaoborus* sp. were greatest in June and July; there were no significant differences among late fall, winter, or early spring months. Mean dry weights of zooplankton were significantly greater in April, May, and June than in other months; minimal values occurred in July and September (Fig. 16).

Mean concentrations of zooplankton (Fig. 17), Entomostraca in particular (Fig. 17), mean dry weights of zooplankton (Fig. 18), and water temperatures all were significantly greater in 1980 than in the other years examined. Rotifers dominated the plankton in 1979, a year characterized by lowest mean dry weights, which, intuitively, might well be expected (Fig. 18). Higher mean concentrations of rotifers, not significantly different from those of 1979, also were seen for 1980. Noteworthy were significantly lower mean concentrations of all major groups of zooplankton in 1978, a year characterized by mean water temperatures lower than in the other years studied. Of course, the data for certain specific taxa occasionally contrasted to the general pattern; e.g., *Daphnia schødleri* exhibited its greatest mean concentrations in 1978. Ranking patterns of annual mean water temperatures corresponded to those of annual mean concentrations of zooplankton. Conversely, mean dry weights of zooplankton did not correspond with mean water temperatures; the greatest mean dry weight values were recorded for 1980, but these values were followed by those of 1978 (Fig. 18).

On a year-to-year basis, less variability is indicated for winter than for summer. Significant differences in mean concentrations of Entomostraca and total zooplankton were noted among three summers, while mean values for only one winter (1978) were significantly different from the others examined (Figs. 19, 20). Considering rotifer concentrations, means in each of the four summers differed significantly, but winters were notably consistent (Fig. 21). Minimum summer and winter mean temperatures occurred in 1978. It is noteworthy that summer temperatures within depth interval 1 did not vary significantly from year to year.

Discussion

With the exception of contrastingly low values in the extreme up-lake riverine segment of DeGray Reservoir, zooplankton concentrations decreased both with distance downlake and depth. Highest concentrations of zooplankton probably occur near the surface in most lakes (Patalas 1969, Andronikova 1972, Short 1977, Starzykowa 1972, Mittelholzer 1973, Miracle 1977, Pace and Orcutt 1981, Novotny and Hoyt 1982), especially during periods of thermal stratification or stagnation. Applegate and Mullan (1967) reported that concentrations and weights of Entomostraca became progressively lower from the upper end of Beaver Reservoir to the dam; concentrations of rotifers were highest uplake, but the uppermost riverine segment was not sampled. Rada (1970) noted relatively lower standing crops of zooplankton in the upstream portion of Big Bend Reservoir, South Dakota.

Horizontal distribution patterns of zooplankton are less likely to follow a consistent pattern in reservoirs with more than one major inflow or that are fed by another reservoir. Certainly, water exchange rates of reservoirs, or even parts of individual reservoirs, influence zooplankton assemblages. According to Brook and Woodward (1956), approximately 18 days are needed for zooplankton populations to become established. Relatively smaller mean standing crops of zooplankton are associated with shorter water exchange times. In fact, Johnson (1964) reported that exchange times of less than 15 days exerted a more marked effect than longer exchange times. Moreover, Starzykowa (1972) found fewer species of zooplankton in reservoirs with shorter exchange rates.

Apparently, the upper segment of DeGray Reservoir provides an environment most favorable to establishment of qualitatively and quantitatively rich zooplankton assemblages, after water movement resulting in more rapid exchange is no longer significant (Nix et al. 1974). Maximal concentrations of rotifers occurred farther uplake (station 12) than did those of the Entomostraca (station 10), perhaps because of shorter life cycles and development on the part of the former. During the summer, when, presumably, water exchange rates are slower and reproductive rates of most zooplankters are relatively rapid, at least some taxa (e.g., Copepoda) are able to exploit the more riverine segment to a much greater degree than during the winter. On the other hand, decreasing concentrations of zooplankton in a down-lake direction cannot be explained on the basis of our available data. However, horizontal gradients of nutrients, wherein concentrations decline as water moves through the reservoir (see other papers in this volume), strongly suggest that declining concentrations of zooplankton in a down-lake direction are related to similar gradients of nutrients and phytoplankton abundance.

Seasonal differences in vertical distribution patterns of zooplankton dry weights may have been influenced by increased concentrations of large plankters, such as Daphnia galeata and D. schødleri, moving into deeper waters during the summer. However, any conclusive statements must await acquisition of dry weight data at the specific level, but there is little doubt that larger entomostracans contribute significantly to total dry weight, even though their relative concentrations are low. While Dumont et al. (1975) determined and catalogued dry weights for many species of fresh-water zooplankters occurring in northern Europe, a number of which are cosmopolitan and also occur in DeGray Reservoir, their data are not applicable for our purposes. Members of southern temperate zooplankton populations often are larger than their northern temperate or subarctic counterparts. Accordingly, one might reasonably presume that the metabolic machinery of many southern temperate zooplankters is geared more toward protein synthesis--and thus growth to larger size--rather than energy storage in the form of fat droplets.

Miracle (1977) found marked vertical stratification of rotifers during periods of stagnation, but more homogeneous distribution during mixis. Miracle further found more variance in time than space, and more variance in vertical distribution than horizontal. Apparently, vertical patterns of rotifer concentrations were not affected significantly by thermal stratification. At least some rotifer species in DeGray Reservoir may migrate vertically during thermal stratification, but the net effect on total rotifer concentrations is not discernible. Moreover, variable vertical patterns would tend to mask year-to-year variations when all years studied are considered together. By way of contrast to the entomostracans, variations between summer and winter in vertical and horizontal distributions of rotifers were almost negligible.

Spring peaks in zooplankton concentrations have been reported by numerous investigators; e.g., Pennak (1946), Reed and Olive (1956), Applegate and Mullan (1967), Rada (1970), Peterka (1970), Andronikova (1972), Bernardi and Soldavini (1976), Comita (1972), Damico (1973), V'yushkova (1974), Leighton (1980), and Novotny and Hoyt (1982). While comparative data are inconsistent with reference to peak intensity and species composition of maximal zooplankton assemblages, entomostracans are more consistent than rotifers by being typically unimodal, displaying their maximal concentrations in spring or early summer. Entomostraca in DeGray Reservoir exhibited a large spring peak and a smaller fall peak, a "classical" annual cycle. Pennak (1946) emphasized the uniqueness of each lake and noted that the annual cycle described by him probably is biased toward larger and deeper lakes.

Novotny and Hoyt (1982) reported a distinct unimodal rotifer peak in Barren River Lake, Kentucky, during January and February, a pattern which runs contrary to the generally expected minimal concentrations in winter. In DeGray Reservoir, rotifers were distinctly trimodal. In spite of summer and fall rotifer peaks, dry weight values only exceeded the mean during April, May, and June to clearly underscore the significant contribution of entomostracans to total zooplankton biomass during the spring maximum.

Rankings of mean annual zooplankton concentrations and mean annual water temperatures correspond, the mean values for each being significantly higher for 1980 and significantly lower for 1978. Andronikova (1972) found a similar correspondence between biomass and heat content in a Russian lake. In DeGray Reservoir, the warmest year (1980) saw the highest mean dry weights of zooplankton, but the second highest values were calculated for the coldest year (1978). This is noteworthy because the minimal mean concentrations of all major groups of zooplankton were recorded for 1978. However, Daphnia schødleri achieved its maximal concentrations during the summer of 1978, and almost certainly this species was a significant contributor to total dry weights for that year. Thus, a shift in size classes--and to some extent, species composition--produced a zooplankton assemblage of greater biomass during 1978, in spite of minimal mean concentrations recorded for that year.

Any effects on zooplankton assemblages resulting from conversion to hypolimnial from epilimnial water release through DeGray dam were not clearly apparent. Martin and Stroud (1973) found a two-year study inadequate to distinguish effects of outlet depth from other factors. Quite probably, other variables such as temperature, rainfall, lake levels, and release volumes combine to prohibit isolating the effects of release mode. However, if the apparent positive relation between annual heat content and concentrations of most zooplankters holds true, then one might expect epilimnial releases to reduce heat content and zooplankton concentrations and hypolimnial releases to increase them, as long as water exchange rates are not limiting to zooplankton population development.

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BACTERIAL INDICATORS OF WATER QUALITY FROM SOUTHWEST ARKANSAS RESERVOIRS: DEGRAY LAKE[1]

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Abstract

Indicator bacteria were monitored in DeGray Reservoir, Arkansas, and its major tributaries for several years. Storms were primarily responsible for loading bacteria into the reservoir. Stratified distribution patterns developed according to thermal density gradients existing within the lake with peak densities appearing most often in the epilimnion and thermocline region. Maximum densities of bacteria were associated with storm plumes. Total coliforms decreased, and fecal coliforms and streptococci increased, during the study. These changes were significant in the upper compartment of the reservoir. Taxa isolated as total coliforms were predominantly members of the Enterobacteriaceae. FC/FS ratios indicated fecal contamination of non-human origin.

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Introduction

Groups of bacteria characterized as total coliforms (TC), fecal coliforms (FC) and fecal streptococci (FS) are used to assess the levels of fecal contamination in natural waters. Their presence in relatively high numbers (EPA, 1972) suggests inferior quality water and in a more applied sense, the possible presence of human pathogens. Most natural waters carry some indicator bacteria. Their numbers, species composition and distribution depend on numerous environmental variables.

Indicator bacteria were monitored in DeGray Reservoir for several years primarily to gain a better understanding of a reservoir ecosystem. The data presented here display some of the patterns of variation observed in these groups of bacteria.

I thank Dr. Charles W. Leming for his assistance in data presentation.

Materials and Methods

Lake samples were collected from three stations (1, 10 and 12) representing the three major compartments of the reservoir (figure 1). A riverine station at highway 84 bridge on the Caddo River was used for storm studies. Samples from the river station and two smaller tributaries, Valley Creek near station 12 and Big Hill Creek near station 10, were used for determination of FC/FS ratios.

The reservoir stations were sampled biweekly at 2-m. intervals down to 12 m., and at 5-m. intervals from 15-m. depth to near bottom (Van Dorn sampler). All other samples were from the surface.

Samples collected in sterile Whirl-Pak bags were refrigerated until the onset of analysis. Incubation was begun within 24 hrs. after sample acquisition. Enumeration was according to standard methods (APHA, 1976) of membrane filtration (Millipore HA 045). Both typical and atypical colonies were counted for total coliforms. Taxa were characterized by the API 20E system (Analytab Products, Inc.). Culture media were m-Endo (Millipore), m-FC broth and m-Enterococcus agar (Difco). Quality control was performed according to EPA guidelines (Bordner and Winter, 1978), and as recommended by Analytab Products, Inc.

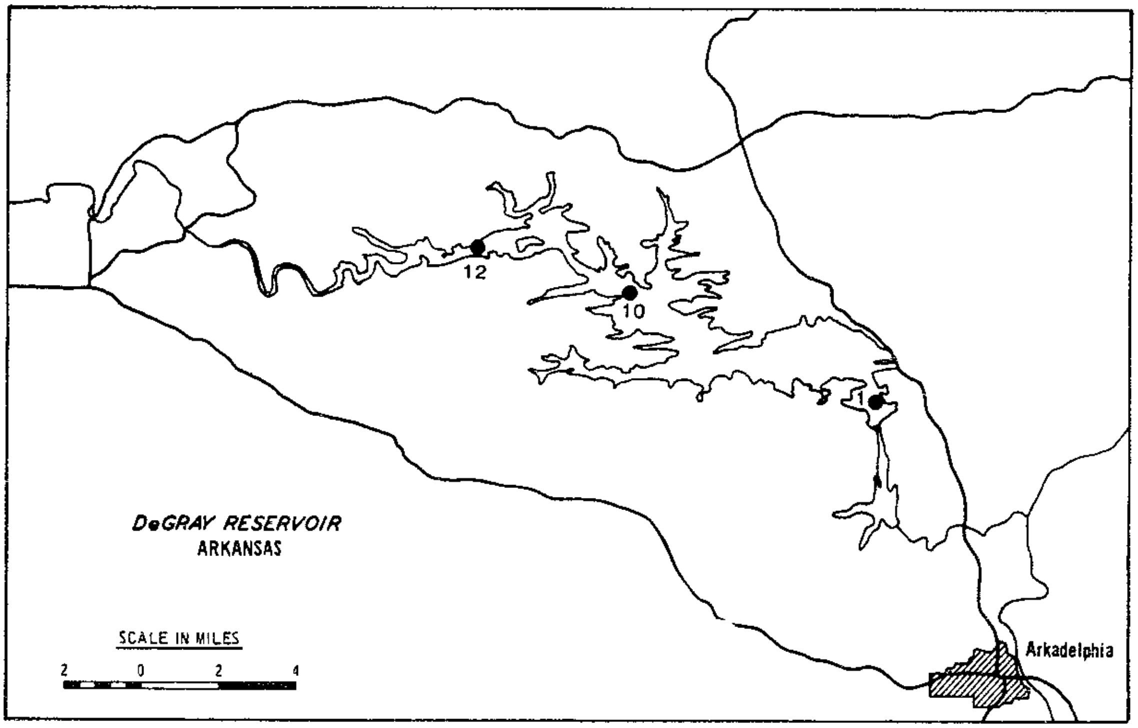

Figure 1. Map of DeGray Reservoir with study sites indicated.

Results and Discussion

The annual means for total coliforms indicated a decrease during the study which was significant at station 12 (figure 2). The opposite trend was seen for fecal coliforms and streptococci. Analysis of FC data produced significant correlation coefficients at both stations 10 and 12 (figure 3). FS increased significantly at station 12 (figure 4). These increases probably related to several factors such as greater sedimentation in the upper reservoir and the loading of bacteria by storm runoff. Several studies have demonstrated the sediments serve as reservoirs for indicator bacteria (Gerba and McLeod, 1976; Hendricks, 1972) which can be resuspended by the scouring action of storm plumes, bubble formation (Weber et al., 1983) and man's activities such as dredging (Grimes, 1975). Studies have also shown that some indicator bacteria may actually proliferate in sediments (Hendricks, 1972). Previous investigations have indicated that TC can fluctuate independently of FC or FS (Lin et al., 1974; Sayler, 1975).

The monthly distributions of FC over a 5-year period are presented in figure 5. Vertical lines represent the standard deviation and give an indication of range in vertical distributions. Each mean represents approximately 15 samples at station 12, 25 at station 10 and 50 at station 1. Apparent gaps in the data resulted from means of less than 0.5 rounded off to zero. The study was terminated in September 1980.

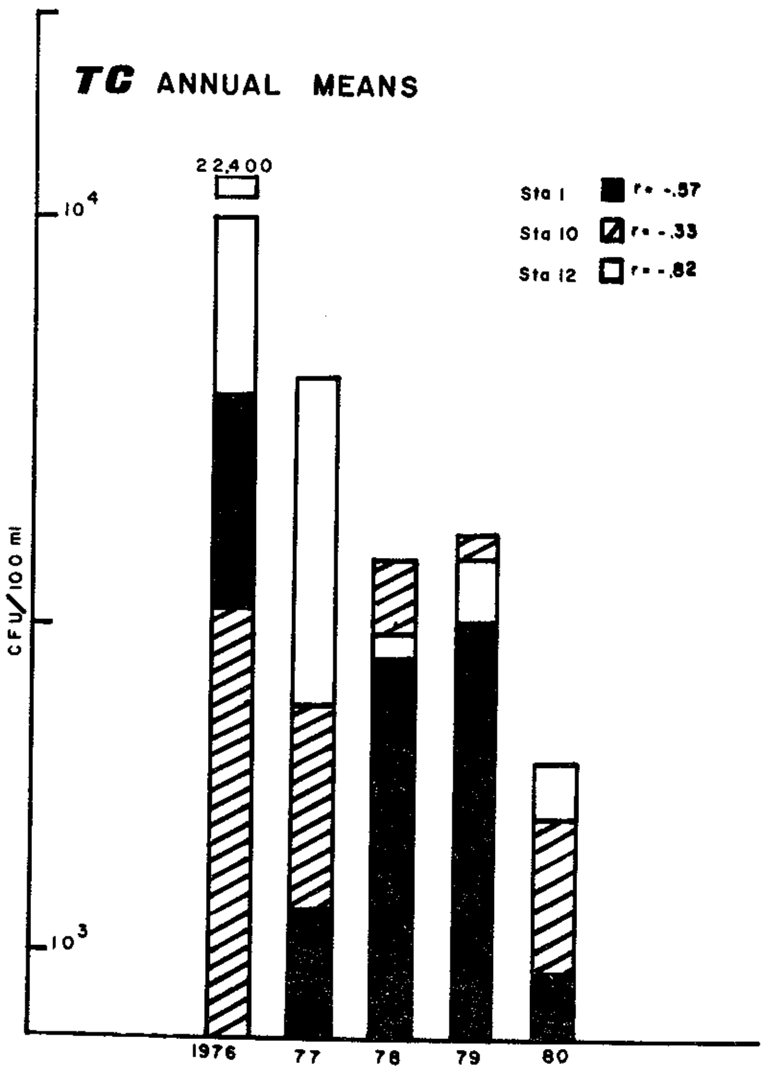

Figure 2. Annual means for total coliforms at stations 1, 10 and 12. Correlation coefficients (r) determined by linear regression.

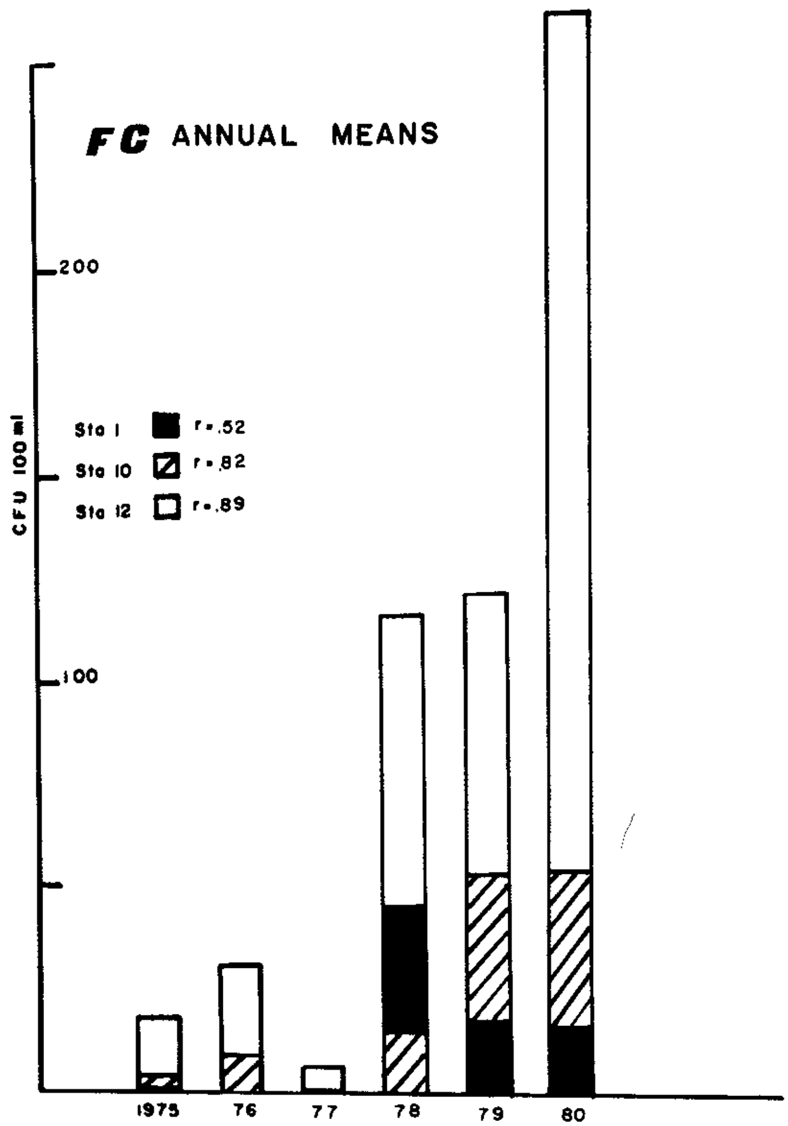

Figure 3. Annual means for fecal coliforms at stations 1, 10 and 12. Correlation coefficients (r) determined by linear regression.

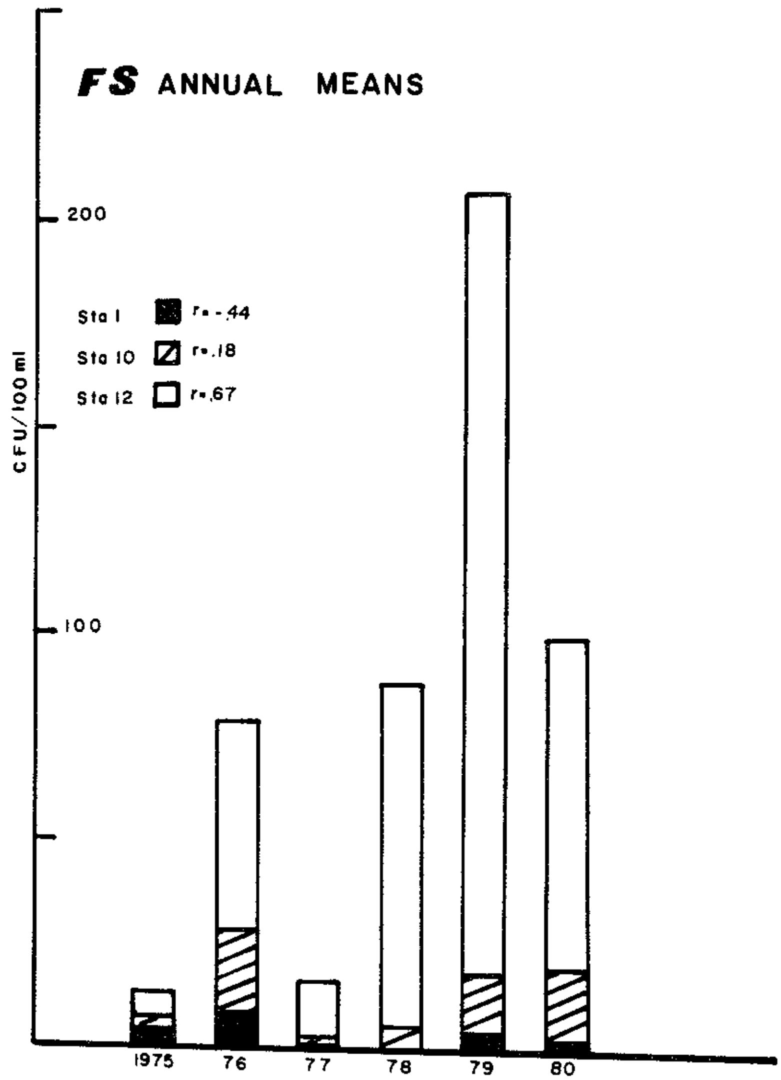

Figure 4. Annual means for fecal streptococci at stations 1, 10 and 12. Correlation coefficients (r) determined by linear regression.

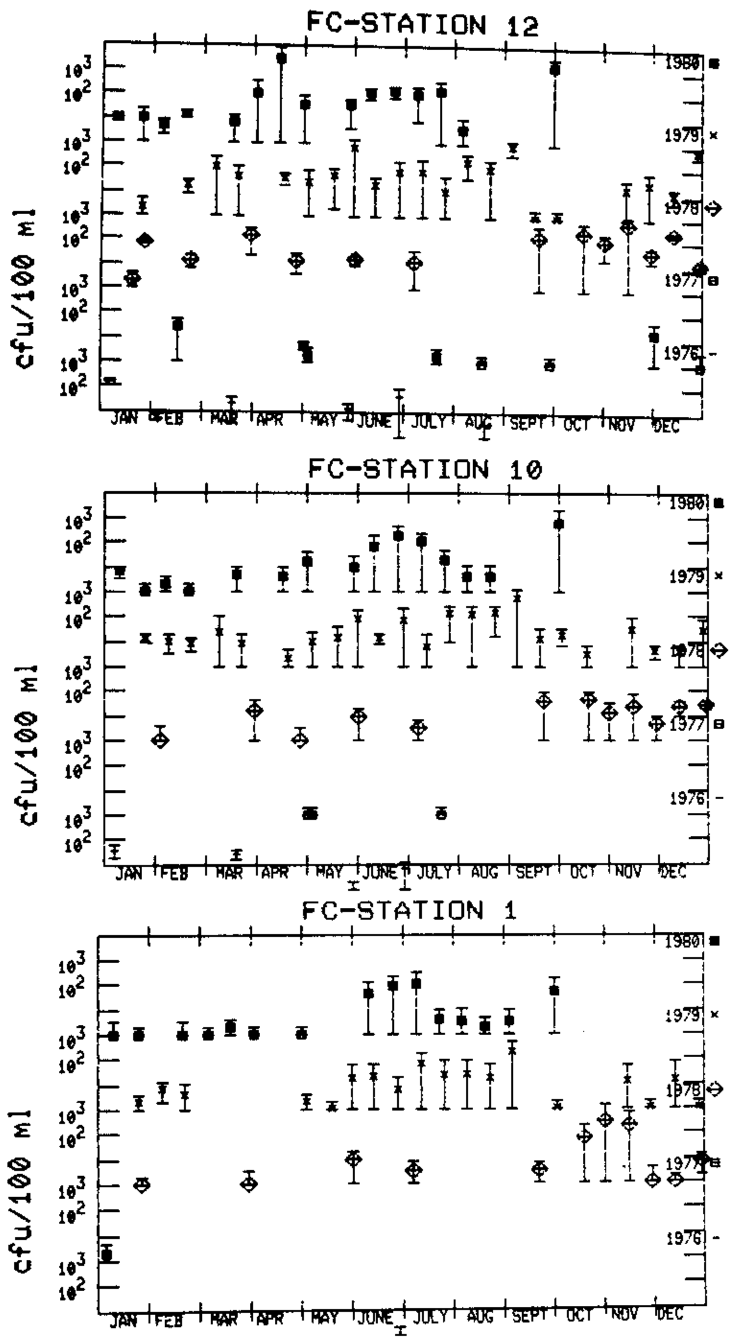

Figure 5. Monthly distributions of fecal coliforms. Vertical lines represent standard deviations.

These data again revealed a progressive increase in FC throughout the study. The only apparent seasonal trend seemed to be an increase in both numbers and ranges leading into the summer months of thermal stratification, followed by a decline during late summer and early fall. Most individual months with high means, such as September 1980, could be correlated with periods of heavy rainfall.

Similar data for FS is presented in figure 6. Again the numbers and ranges appeared to increase during the study. Summer increases were not as evident as those for FC, although a late spring to early summer increase was suggested.

Two factors appeared to have the greatest influence on distribution of indicator bacteria; surges from storm runoff and thermally-induced density gradients within the reservoir. Thornton et al. (1980) showed that storm surges could enter the reservoir as either overflows, innerflows or underflows, as dictated by stormwater density relative to established gradients within the reservoir. After seeking a depth of corresponding density, the storm plume would proceed some distance through the reservoir until dispersed. Bacterial distribution responded accordingly, often increasing by several orders of magnitude with passage of the plume and corresponding in vertical distribution to the location of the storm waters.

Data in figure 7 show the flushing effect of runoff on

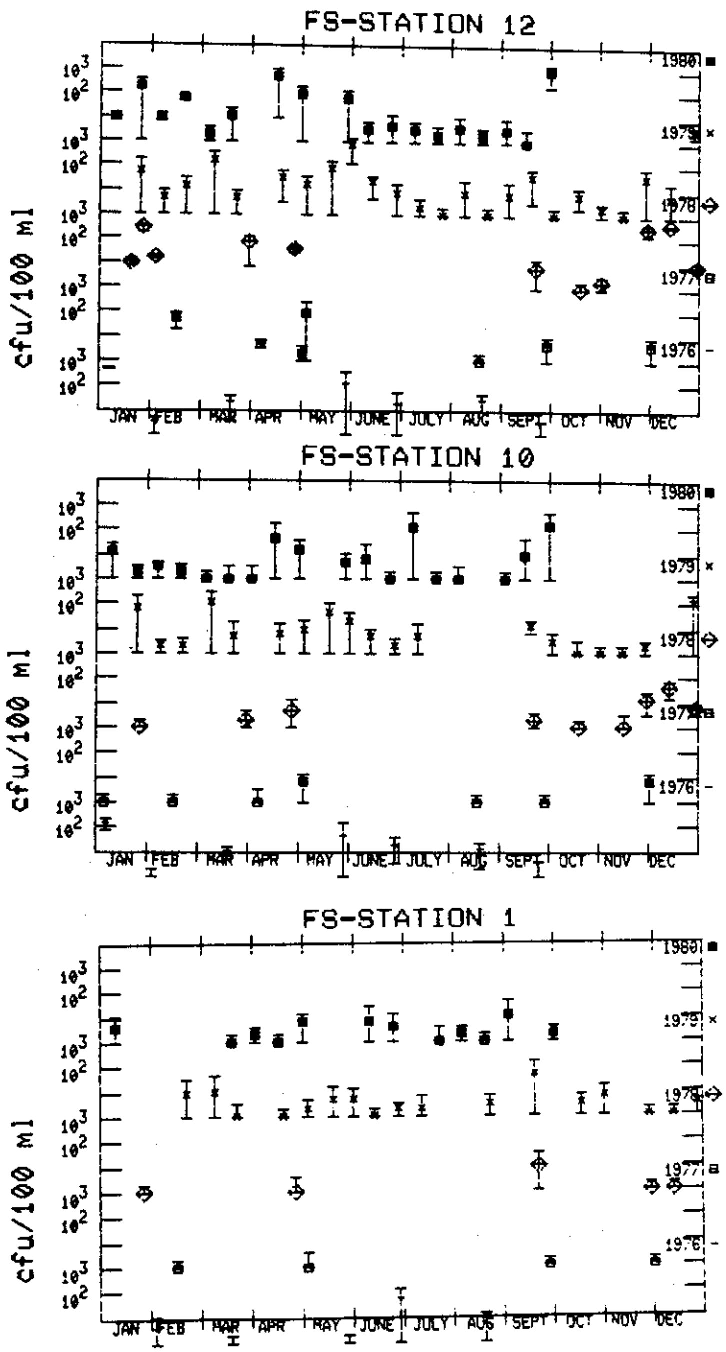

Figure 6. Monthly distributions of fecal streptococci. Vertical lines represent standard deviations.

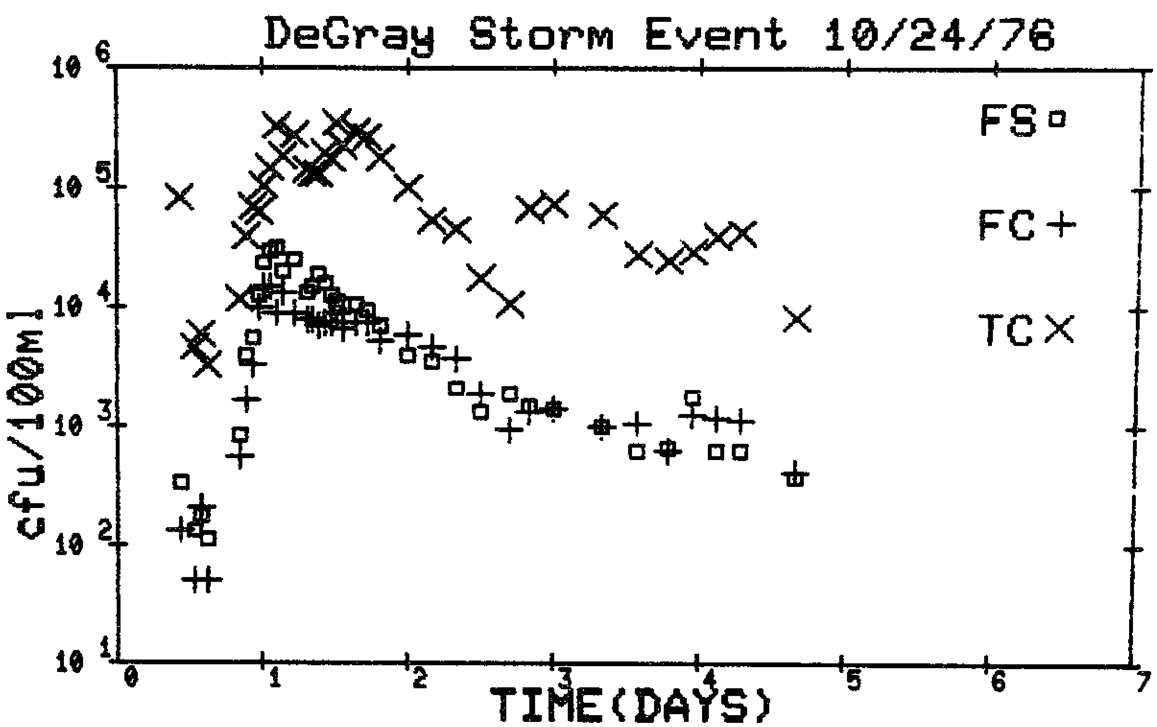

Figure 7. Pattern of bacterial loading during a storm from samples collected at the Caddo River station on October 24, 1976.

bacteria from the watershed. Loading of bacteria was very rapid, reaching densities of 10^5-10^6 cfu/100 ml. during the first few hours after rainfall began. Several days were usually required for the restoration of baseline levels following the storm.

Bacterial numbers tended to peak slightly before the hydrograph, a phenomenon termed "first flush" (Thornton et al., 1980; Perrier et al., 1977; Davis et al., 1977; Jodie, 1975). The subsequent decline generally conformed to the decending arm of the hydrograph.

More than 30 storms were investigated during 1976-80. All seemed to have similar characteristics dependent upon the nature of the storm (Perrier et al., 1977). Storms having more than one period of heavy rainfall would result in a reflective pattern of bacterial loading such as the bimodal distribution in figure 8 for two successive periods of rainfall. Heavy rainfall over several days tended to produce less definite patterns although high levels of bacteria were maintained throughout the storm (figure 9).

Vertical distribution of bacteria within the reservoir generally responded to density gradients during thermal stratification, and probably to more discrete gradients throughout the year. Greater numbers were usually present in the epilimnion although occasional highs were observed in the hypolimnion. Most often these peak densities could be traced to preceding storm flows.

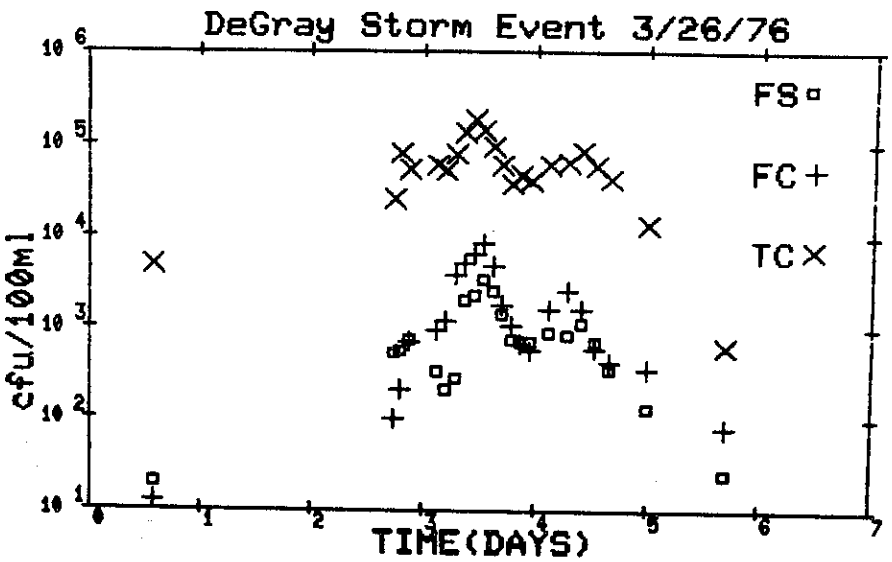

Figure 8. Pattern of bacterial loading during a storm from samples collected at the Caddo River station on March 26, 1976.

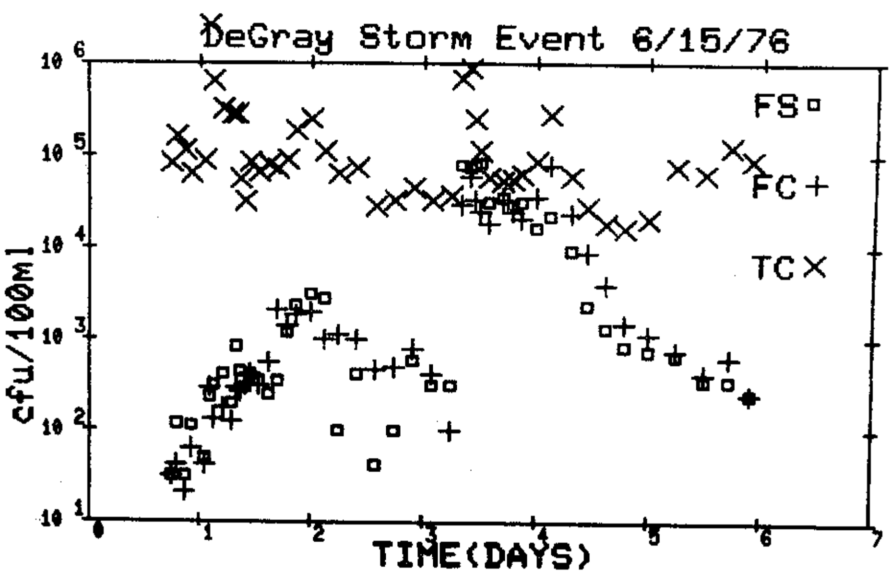

Figure 9. Pattern of bacterial loading during a storm from samples collected at the Caddo River station on June 15, 1976.

TC data are presented for January-September 1980, as examples of vertical distribution patterns commonly observed. Temperatures in ^{o}C. are indicated at various depths. In winter, narrow strata of bacteria were present and no detectable coliforms at some depths (figures 10 and 11). Concurrent with increasing spring temperatures, the strata began to widen, usually associated with storm plumes (figures 12 and 13). During late spring and early summer, the strata again narrowed with the onset of thermal stratification and low rainfall (figure 14). By June (figure 15), TC greatly increased at depths corresponding to the thermocline and into the hypolimnion. During July (figure 16), a ten-fold increase occurred and distribution was rather uniform below the thermocline. These data suggested growth of some bacteria and subsequent sedimentation from the epilimnion. In August (figure 17), a more stratified profile again appeared with highest numbers in the epilimnion and upper thermocline. The maximum bacterial density was reached during the first week of September (figure 18). A decrease followed during mid-month with a more uniform profile (figure 19), again suggesting a "bloom" and subsequent "rain" of bacteria from the epilimnion. TC decreased and became more stratified during the last week of September following approximately eight inches of rainfall which cooled the lake's surface 6^{o} C. and depressed the thermocline (figure 20).

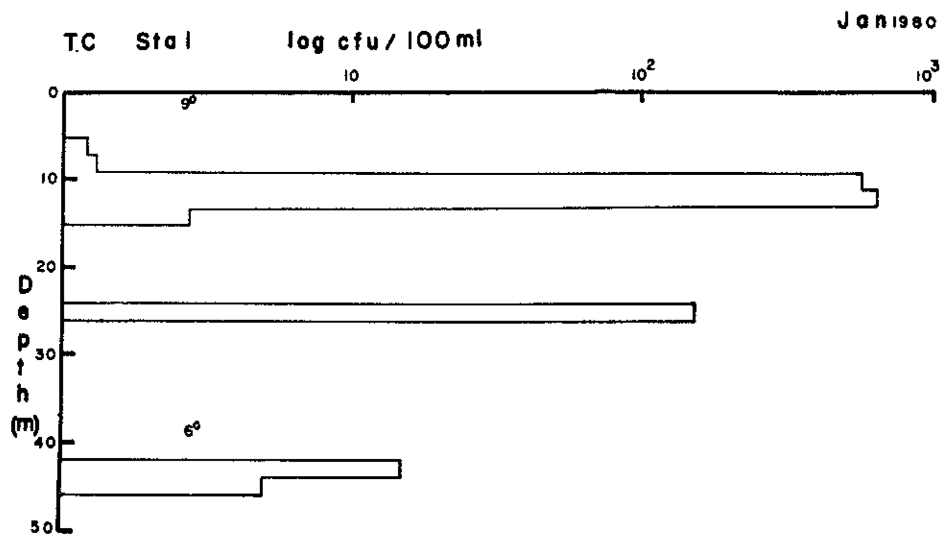

Figure 10. Distribution of total coliforms at station 1 during January 1980.

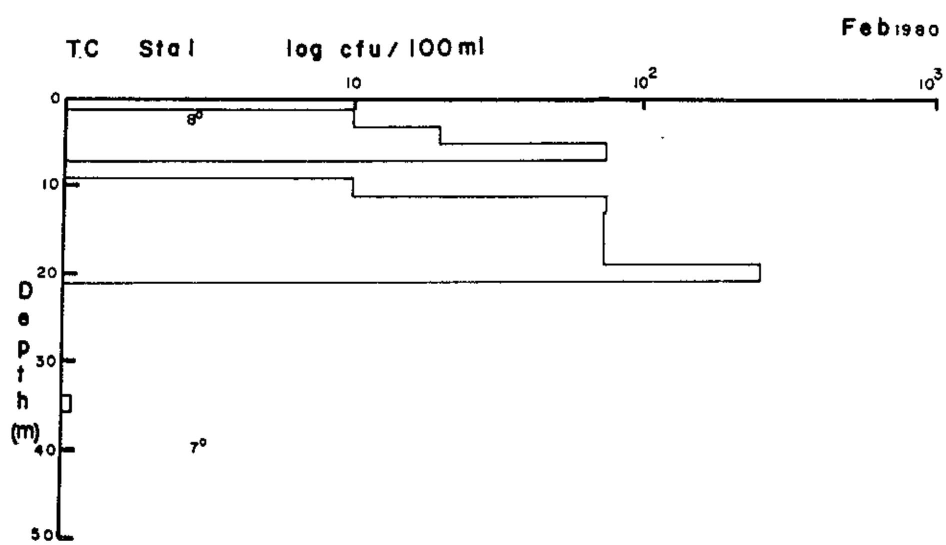

Figure 11. Distribution of total coliforms at station 1 during February 1980.

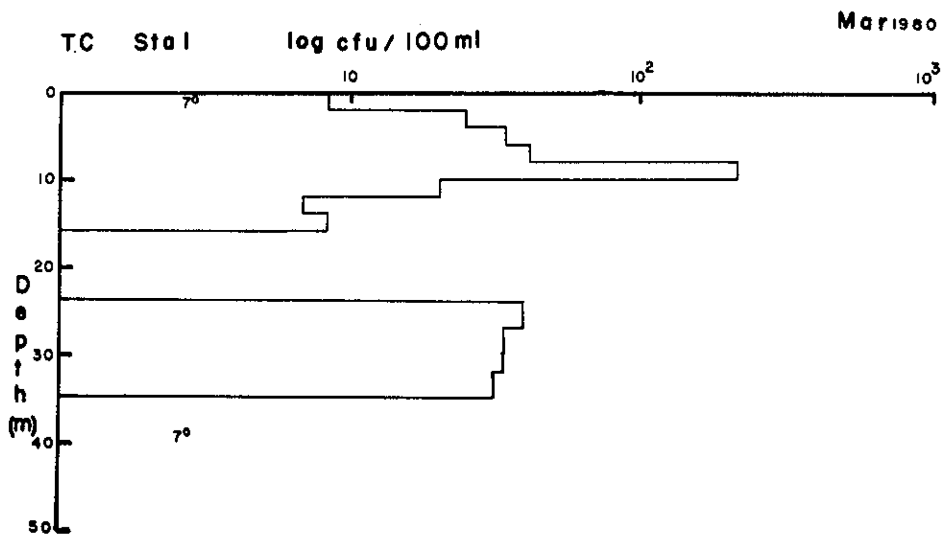

Figure 12. Distribution of total coliforms at station 1 during March 1980.

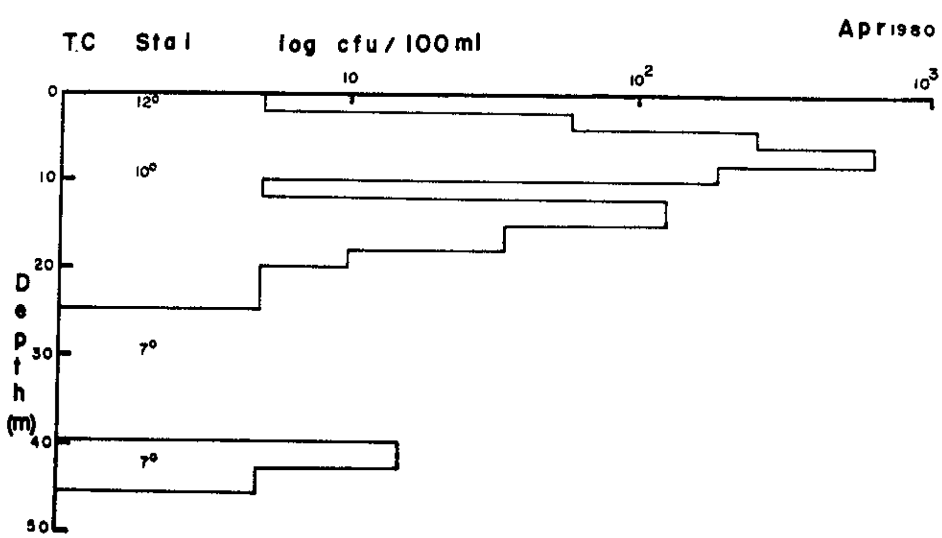

Figure 13. Distribution of total coliforms at station 1 during April 1980.

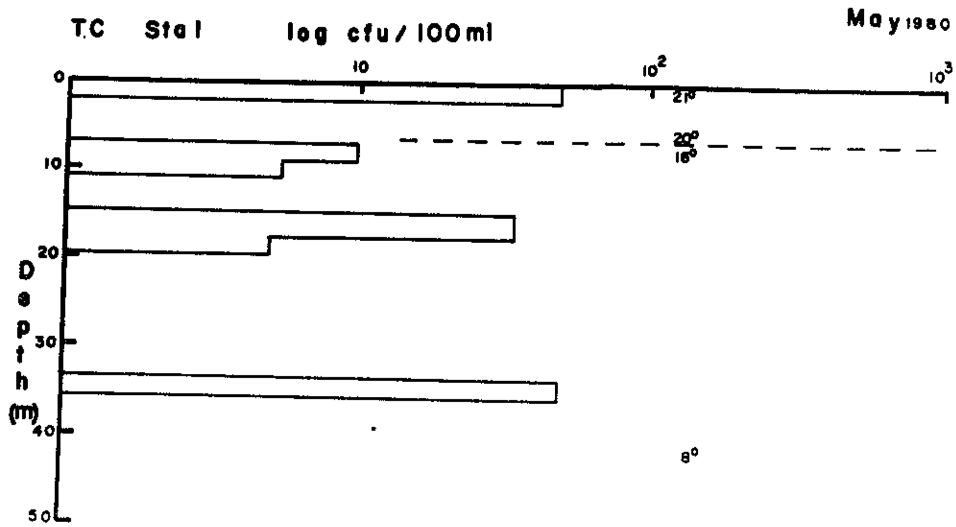

Figure 14. Distribution of total coliforms at station 1 during May 1980.

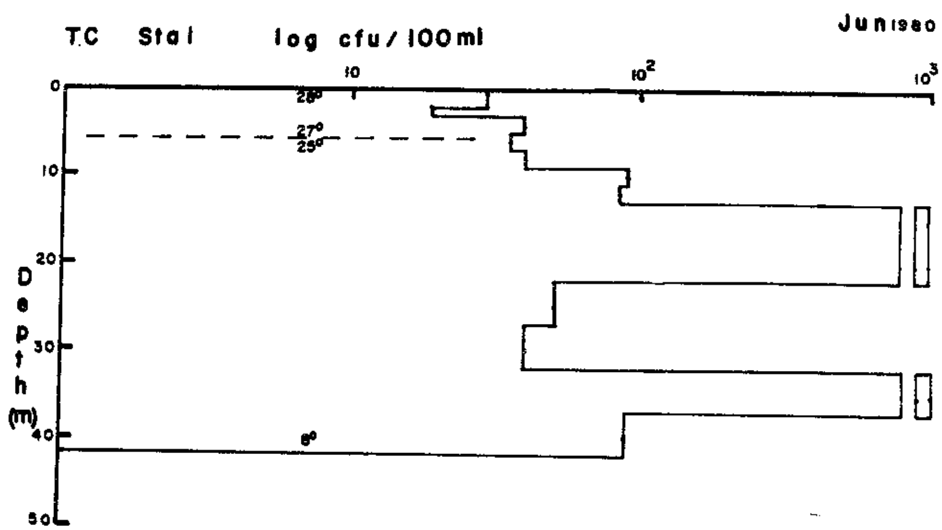

Figure 15. Distribution of total coliforms at station 1 during June 1980.

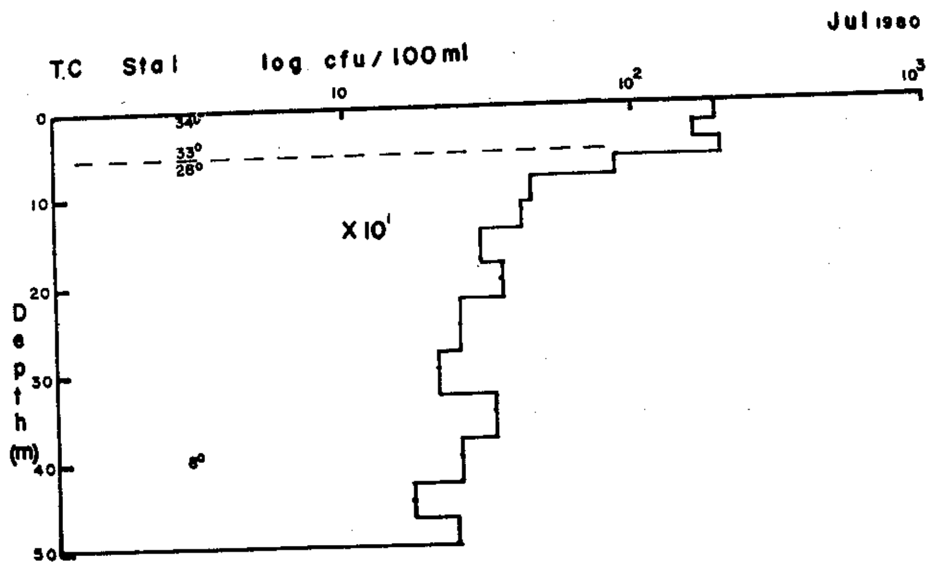

Figure 16. Distribution of total coliforms at station 1 during July 1980.

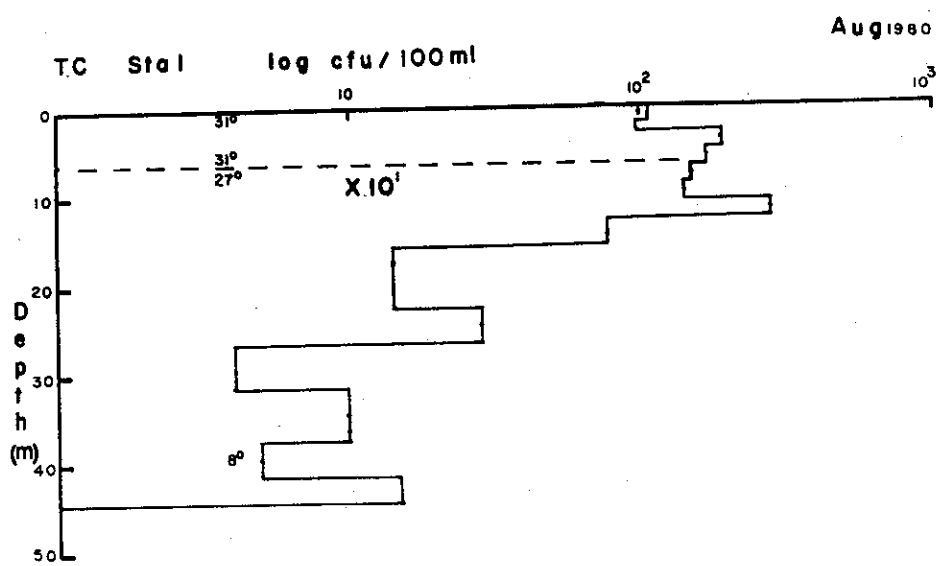

Figure 17. Distribution of total coliforms at station 1 during August 1980.

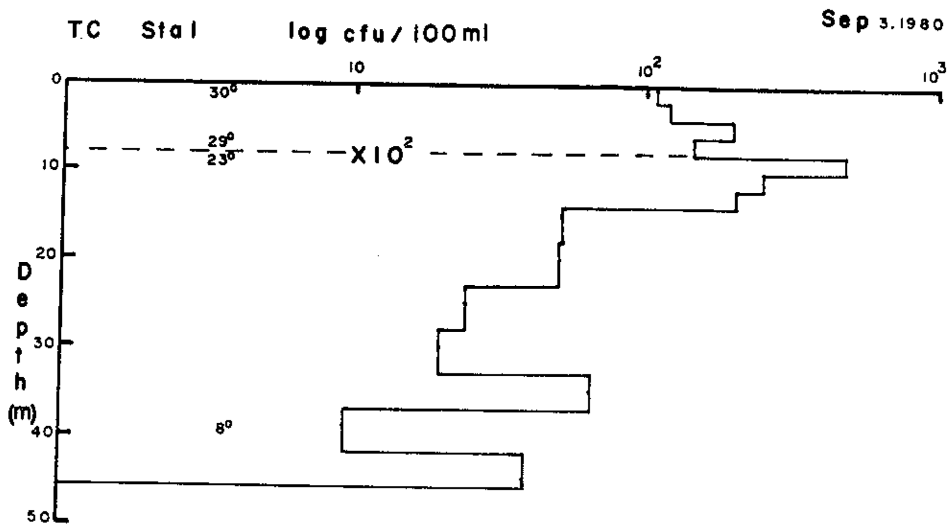

Figure 18. Distribution of total coliforms at station 1 on September 3, 1980.

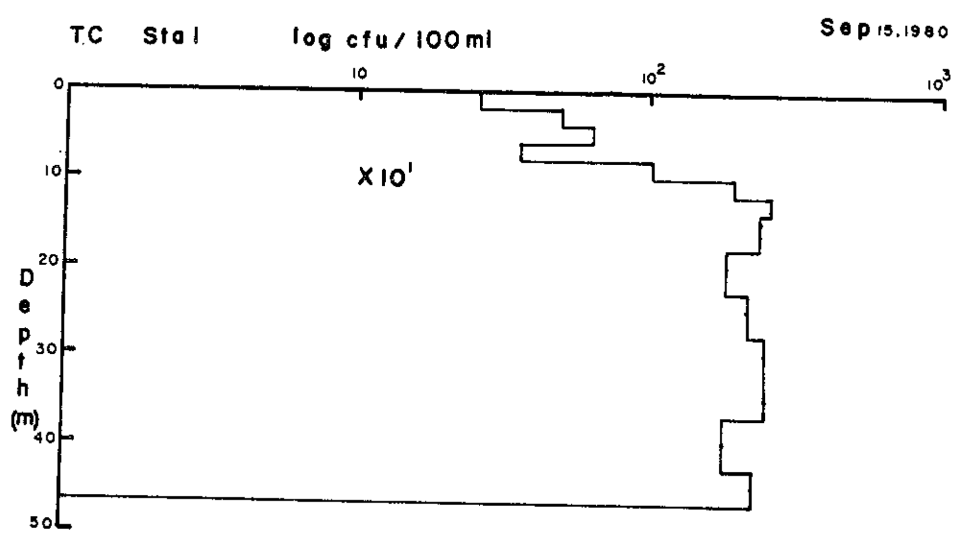

Figure 19. Distribution of total coliforms at station 1 on September 15, 1980.

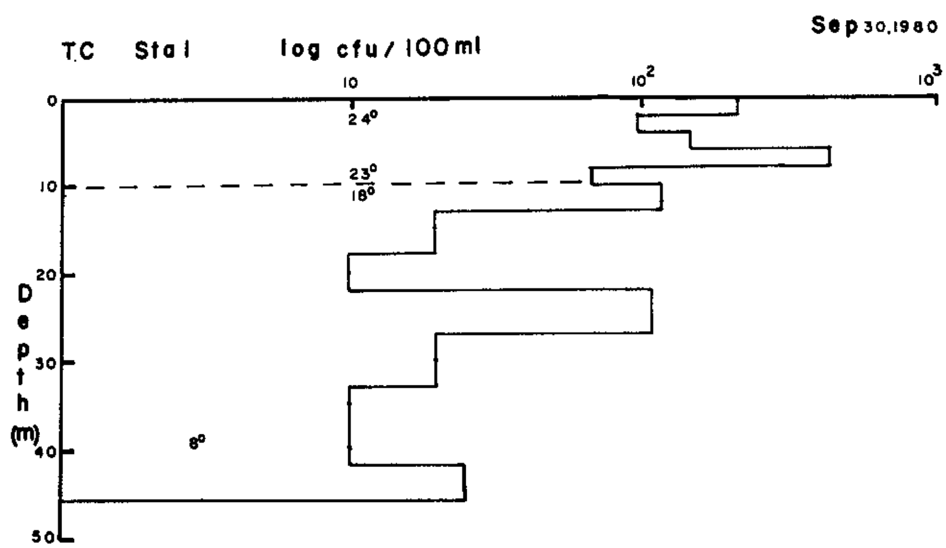

Figure 20. Distribution of total coliforms at station 1 on September 30, 1980.

Since TC quality is more poorly defined than either FC or FS, a study was undertaken during January-May 1980 to determine taxa being isolated as total coliforms. An attempt was made to select the dominant types according to colony form and coloration from enumeration plates. Data in table 1 show taxa and their proportions relative to the total number of isolates. The *Escherichia-Enterobacter-Klebsiella-Citrobacter* complex generally constitutes the coliform group (APHA, 1976). About 65% of the isolates belonged to this complex and an additional 12% belonged to the same family, Enterobacteriaceae. Other taxa making up the remaining 12% are not usually considered coliforms. A comparison of isolates from routine sampling of the reservoir with those from the river station during a storm indicated little difference in bacterial quality of storm surges relative to those taxa in the reservoir prior to the storm (table 1).

The origin of indicator bacteria was apparently form non-point sources on the watershed. Studies of the bacterial burden carried by the river failed to show significant contributions by the small centers of human populations above the reservoir (unpublished data), and no significant correlation with recreational activity was found (Johnson, 1980).

Data in table 2 show the bacterial load carried by three tributaries during an approximate one-year period. The Caddo River had the lowest numbers but since it accounts

Table 1. Bacterial quality from the reservoir sampling stations (January-May 1980) compared to the river station during a storm study on May 12, 1980 (Johnson, 1980).

Taxa	Reservoir Stations Isolates	%	River Station Isolates	%
Escherichia coli	241	33	45	28
Enterobacter spp.	118	16	32	20
Klebsiella spp.	86	12	19	12
Hafnia alveii	35	5	0	-
Citrobacter spp.	28	4	6	4
Serratia spp.	30	4	4	3
Providencia spp.	16	2	11	7
Proteus spp.	9	1	3	2
Edwardsiella tarda	1	-	0	-
Salmonella sp.	1	-	0	-
Enterobacteriaceae	565	77	120	76
Aeromonas hydrophila	48	7	11	7
Pseudomonas spp.	36	5	4	3
Acinetobacter spp.	16	2	8	5
Plesiomonas shigelloides	14	2	13	8
Others	55	8	2	1
Non-enterobacteriaceae	169	23	38	24
Total isolates	733	100	158	100

Table 2. Indicator bacteria carried by three tributaries of DeGray Reservoir during 1979-80.

Location	TC	FC cfu/100ml	FS	No. paired data	FC/FS % of samples <0.7	FC/FS % of samples >4.0
Caddo River	$2,375 \pm 2,943$	98 ± 226	118 ± 108	19	79	5
Big Hill Creek	$12,741 \pm 19,270$	267 ± 504	320 ± 328	25	52	8
Valley Creek	$6,898 \pm 16,646$	134 ± 255	161 ± 232	28	64	7

for more than 70% of the flow into the reservoir, the river was considered the primary source of indicator bacteria from the watershed.

FC/FS ratios provide an index of pollution which suggests the origin of fecal contamination (Geldreich and Kenner, 1969). Ratios greater than 4.0 indicate human sources and less than 0.7 other warm-blooded animals. The value of such data has been questioned by many studies, primarily because of variable die-off rates and the failure of standard methods to recover injured cells (Bissonnette et al., 1975). However, organisms in transit toward the reservoir should show the least effects of die-off and their ratios may be of some value. Most paired data suggested non-human contamination (table 2). About 28% were of intermediate range, and only 7% indicated human contamination. Most of the latter group were from high-flow periods following storms. Therefore, non-point sources such as cattle grazing on the watershed were the probable sources of fecal contamination.

Total coliforms decreased, and fecal coliforms and streptococci increased, during the study. Rainfall runoff from the watershed was primarily responsible for loading bacteria into the lake with peak densities as high as 10^6/100 ml. reached after a storm. These bacteria were distributed according to density gradients within the reservoir where most probably died off. Some data suggested TC type organisms may have grown during the summer months.

Maximum densities in the water column were associated with storm plumes or summer months. Bacteria were usually distributed in a stratified pattern even when the reservoir was not thermally stratified. Taxa isolated as coliforms were primarily enterobacteriaceous. FC/FS ratios suggested non-human origins of indicator bacteria from non-point sources on the watershed.

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POPULATION DYNAMICS OF THE FISHES IN DeGRAY LAKE, ARKANSAS, DURING EPILIMNIAL AND HYPOLIMNIAL RELEASE

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Abstract

Three major methods were used in assessing the fish populations of DeGray Lake, Arkansas, during epilimnial release (1974-78) and hypolimnial release (1979-82): fish samples were collected annually with rotenone in three coves. Largemouth bass population estimates were made in three coves in April each year and mid-water trawls were used to assess young-of-the-year populations and production of shad. Biomass in each cove generally declined during epilimnial release but remained relatively stable during hypolimnial discharge. Annual changes in total standing crop were largely influenced by fluctuations in shad (Dorosoma spp.) populations. A strong longitudinal gradient developed with age; the cove representing the upper lake region had the greatest total biomass each year except 1974 and showed the least year-to-year variation and smallest decline over the 9 years. Lakewide mean estimates of largemouth bass declined each year during epilimnial release, fluctuating at lower levels during hypolimnial discharge. Lakewide production of largemouth bass averaged 26.6 kg/ha during epilimnial release and 18.2 kg/ha during hypolimnial release. Shad, sunfishes (Lepomis spp.) and crappies (Pomoxis spp.) made up 98% of larval fish collected by mid-water trawling. Densities (n/m^3) of larval sunfish were higher during epilimnial release than during hypolimnial discharge. Crappies and shad were higher during hypolimnial release. Estimates of shad production during hypolimnial release were 67% greater than during epilimnial discharge. Relationships between the environmental factors (age, water level fluctuation, outlet depth) and standing crops are discussed.

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Introduction

The study of reservoir outlet design, especially selective withdrawal depth, received considerable attention during the 1960's and early 1970's. A number of modeling studies were conducted, many of which were combined with laboratory and field data (Kittrell 1959; Imberger and Fisher 1970; Bohan and Grace 1973; Roesner et al. 1974; Moore 1976; Keeley et al. 1978). Biological productivity of reservoirs, which has been studied extensively, was reviewed by Wright (1967). However, biological productivity in relation to water release patterns has received less attention. Jenkins and Morais (1971) reported that outlet depth was directly related to the number and size of fish harvested and that reservoir area, dissolved solids, growing season, and age of impoundment were the most important environmental variables tested. During 1968-71, the impact of epilimnial and hypolimnial release on fish populations and limnology was studied at Nolin and Barren River reservoirs in southwestern Kentucky by the Sport Fishing Institute (Martin and Stroud 1973). The study showed that a 2-year experiment was insufficient to separate effects of climatic events from possible influence of the discharge regime.

In 1974 the National Reservoir Research Program began research on DeGray Lake. Principal objectives were to compare effects of epilimnial and hypolimnial discharge on the biota in the reservoir. We made quantitative measurements of biomass of reservoir fishes and determined production and distribution of important prey (shad, *Dorosoma* spp.) and predator (black bass, *Micropterus* spp.) fishes. The DeGray study was designed to evaluate these effects during 4 years of epilimnial release and 4 years of hypolimnial release.

Study Area - DeGray Lake, located on the Caddo River in west-central Arkansas, was impounded in 1969. At normal pool elevation of 124.4 m above sea level, the reservoir has an area of 5,427 ha, and maximum and mean depths of 57 m and 15 m. The reservoir extends in a west to northwest direction for about 32 km and has a shoreline length of 333 km. For the assessment of spatial variation of selected biological and physical characteristics, the lake was divided into three sampling sections representing the upper, middle, and lower regions. The upper section (1,238 ha) consists mainly of narrow river channel bordered by small coves containing standing timber. The middle section (2,549 ha) includes open water areas and many large coves; two major tributaries flow into it. The lower region (1,661 ha) has large open water areas bordered by steep rocky shorelines.

The multi-level intake structure at DeGray Lake is four-sided tower with openings in each face. This structure allows water to be selectively withdrawn from one of three 6.4-m^2 openings on each side, the midpoints of which are at elevations 120.4, 115.8 and 108.2 m above mean sea level (msl). All water releases after impoundment and through 1978 were made from the epilimnial (120.4 m msl) outlet. During the 4-year period of 1979-82 all water releases were made through the hypolimnial (108.2 m msl) outlet. A mean annual discharge of 703.38 x 10^6 m^3 provides a storage ratio of 1.15.

Methods

We used 3 major methods in assessing the fish populations of DeGray Lake. These included sampling cove communities with rotenone in August each year, largemouth bass population estimates in April of each year, and mid-water trawling for young-of-the-year (YOY) fish.

August rotenone sampling - Rotenone was applied to 3 coves annually to assess changes in standing crops of fishes in DeGray Lake. The 3 coves represent the upper, middle, and lower regions of the lake. Each cove, encompassing 1 to 2 ha, was blocked off with 25-mm-mesh net (stretch measure) to prevent fish from escaping. Scuba divers made sure the net reached bottom. A mixture of approximately 10 parts water and 1 part rotenone (5% emulsifiable) was applied by pressure spraying equipment to the surface of shallow areas and to deeper areas through a perforated pipe (3 m long) attached to 4 m of garden hose.

When fish began to surface they were collected with dip nets and a "scoop boat" similar to that described by Baker (1962). Fish were sorted by species, measured, counted, and weighed. Scale samples were taken from representative samples of selected species. Fish were collected for 2 days; weights of fish collected the second day were calculated from length-weight relationships based on fish collected the first day. Fish were tabulated according to species and by 25-mm (1-in) groups. Biomass (kg/ha) was computed for each cove. Analyses were made by species, major categories of related species, sample location, and discharge regime.

Spring largemouth bass populations - A boom-type, boat-mounted electroshocker with variable pulsed, direct current was used each spring (1975-82) to make shoreline population estimates of largemouth bass. Estimates were made in late April when water temperatures were 16 to 18°C. All sampling was done at night.

Population estimates were based on the mark and recapture method described by Bryant and Houser (1971). Estimates were made after four consecutive nights of electrofishing; a complete circuit of the cove was made each night. All bass were counted, weighed, and measured (total length) and scale samples were taken for age determination. The caudal fin was clipped from bass 250 mm long or less, and larger fish were tagged with Floy dart tags. Population estimates were expresssed as fish per hectare. Because coves differed greatly in the relation of shoreline length to surface area, a bias is introduced when numbers or weight of bass are compared on a surface area basis. Populations from coves encompassing large surface in relation to shoreline length were underestimated when compared on an areal basis. To compensate for this bias, we derived adjustment factors in two steps. First, we determined ratios of shoreline length to surface area (m/ha) for each sample cove. These ratios were 326.5, 84.9, and 278.4 m/ha for the uplake, midlake and downlake coves to the uplake cove, which had the highest shoreline to surface ratio, by dividing the shoreline to surface area ratio of each cove into that of the uplake cove. Population estimates (number of bass per hectare) were then multiplied by the appropriate adjustment factors: 1.00 for the uplake cove, 3.84 for the midlake cove, and 1.17 for the downlake cove.

Age and growth estimates were based on empirical data from 10,028 bass collected during spring population estimates. Since the fish were collected before the beginning of the growing season, each fish was credited with an annulus at the outer edge of the scale.

Ivlev (1966) defined production as the total elaboration of fish tissue during time interval ΔT, including that formed by individuals that do not survive to the end of ΔT. We used the method of Chapman (1968) to calculate production of largemouth bass. Numbers of bass per hectare, together with the average weight of fish in each age class, were used to calculate annual production for each section of the lake. Data collected in August cove sampling with rotenone were used to estimate production of young-of-the-year fish. Such samples were taken in each section of the lake in 1975-82.

Midwater trawling - Populations of YOY fish were sampled with midwater trawls every 2 weeks from May through August each year (because of low catches, sampling was stopped after July in 3 of 9 years). Twenty oblique hauls to a depth of 7 m were completed on each sampling date. Sampling sites were selected at random from a gridded map of the lake. Four hauls were made in compartment I (uplake area) and eight each in compartments II and III. All sampling was done at night because YOY fish have been shown to be more evenly distributed and nearer the surface at night than during the day (Netsch et al. 1971).

All trawling was done from an aluminum boat (8.5 m long, 3.2 m beam) powered by a diesel engine. Trawls were released and retrieved by two hydraulic winches. During 1975 and 1976 a 1.88-m^2 frame trawl described by Houser (1972) was used for sampling; from 1977 through 1982 a 2-m^2 Tucker trawl was used. Series of tests were conducted to compare the relative efficiency of the two trawls and a mean efficiency factor for each taxon was determined and used to adjust the catches of the frame trawl (Dewey and Moen 1983a).

Fishes were preserved in 10% formalin and later identified and counted. We identified specimens to the lowest possible taxon, using keys developed by May and Gasaway (1967) and Hogue et al. (1976). Because we were unable to identify shad shorter than 20 mm, the data for that length range were pooled for analyses. Shad longer than 20 mm were enumerated by species, and those longer than 25 mm were considered juveniles. Sunfishes and crappies were not identified to species; specimens longer than 20 mm were considered juveniles. Catch data were reported as number/m^3 and as biomass for each taxon.

Catches of shad were used to calculate production at 2-week intervals beginning in mid-May when abundance was greatest; methods followed those of Chapman (1968).

Results

August rotenone sampling - More than forty species of fish were collected during nine years (1974-82) of cove rotenone sampling (Table 1). Annual total biomass expressed as a lakewide mean varied from 409 kg/ha in 1974 to 139 kg/ha in 1977 and averaged 205 kg/ha. The standing crop in individual coves ranged from 588 kg/ha for the mid-lake cove (compartment II) in 1974 to 88 kg/ha in the down-lake cove (compartment III) in 1980. Biomass in each of the 3 coves generally declined during epilimnial release years but remained relatively stable during hypolimnial discharge. The cove representing the upper region of the lake (compartment I) had the greatest total biomass each year except 1974 and showed the least year-to-year variation and the smallest decline over the 9-years of sampling. There was a marked decrease in the standing crop from up-lake to down-lake coves during the last six years of sampling (Table 2). Although a general decline in biomass occurred in each cove during epilimnial release, the difference in their means for each discharge regime was not significant ($P \gtrless 0.05$). We found a significant correlation between biomass and age of the reservoir only for the down-lake cove ($r = -0.83$, $P \gtrless .01$). Shad, sunfish, and black bass accounted for 96% of the mean biomass during epilimnial release and 77% during hypolimnial discharge (Table 3). Annual changes in total standing crop were largely influenced by fluctuations in shad populations. Variation in both total standing crop and standing crop less shad was greater during epilimnial release than during hypolimnial discharge. (Fig. 1).

Shad - Gizzard shad made up 16 to 70% (average 36%) of the total annual biomass, decreasing sharply from 287 kg/ha in 1974 to 27 kg/ha in 1978 during epilimnial discharge, fluctuating from 28 to 53 kg/ha during 4 years of hypolimnial release. Most of the population biomass was composed of adults; YOY made up 1 percent or less of the weight of gizzard shad in 6 of the 9 years,

Table 1. Species of fish in DeGray Lake rotenone samples, 1974-1982.

Common name	Scientific name
Longnose gar	*Lepisosteus osseus*
Gizzard shad	*Dorosoma cepedianum*
Treadfin shad[1]	*Dorosoma petenense*
Grass pickerel	*Esox americanus vermiculatus*
Chain pickerel	*Esox niger*
Common carp	*Cyprinus carpio*
Shiners[2]	*Notropis* spp.
Bluntnose minnow	*Pimephales notatos*
Northern hogsucker	*Hypentelium nigricans*
Spotted sucker	*Minytrema melanops*
River redhorse	*Moxostoma carinatum*
Black redhorse	*Moxostoma duquesnei*
Golden redhorse	*Moxostoma erythrurum*
Black bullhead	*Ictalurus melas*
Yellow bullhead	*Ictalurus natalis*
Channel catfish	*Ictalurus punctatus*
Madtoms	*Noturus* spp.
Flathead catfish	*Pylodictis olivaris*
Pirate perch	*Aphredoderus sayanus*
Topminnows	*Fundulus* spp.
Mosquitofish	*Gambusia affinis*
Brook silverside	*Labidesthes sicculus*
Inland silverside	*Menidia beryllina*
White bass[1]	*Morone chrysops*
Hybrid white x striped bass[1]	*Morone* sp.
Green sunfish	*Lepomis cyanellus*
Warmouth	*Lepomis gulosus*
Bluegill	*Lepomis macrochirus*

(Continued)

Table 1 (Concl.)

Common name	Scientific name
Longear sunfish	*Lepomis megalotis*
Redear sunfish	*Lepomis microlophus*
Smallmouth bass	*Micropterus dolomieui*
Spotted bass	*Micropterus punctulatus*
Largemouth bass	*Micropterus salmoides*
White Crappie	*Pomoxis annularis*
Black Crappie	*Pomoxis nigromaculatus*
Logperch	*Percina caprodes*
Darters	*Etheostoma* spp.

[1] Introduced species

[2] Includes goldenshiner, *Notemigonus crysoleucas*; bigeye shiner, *Notropis boops*; and steelcolored shiner, *Notropis whippler*.

Table 2. Estimated total standing crop (kg/ha) based on cove rotenone samples in August during epilimnial and hypolimnial discharge DeGray Lake, 1974-1982.

DISCHARGE REGIME	YEAR	LOCATION			LAKEWIDE Mean
		Up-lake	Mid-lake	Down-lake	
Epilimnial	1974	404	588	237	409
	1975	328	187	224	246
	1976	271	130	145	186
	1977	160	148	109	139
	1978	213	170	106	163
Mean		275	245	164	229
Hypolimnial	1979	241	184	106	177
	1980	245	119	88	151
	1981	220	178	129	176
	1982	305	136	103	181
Mean		252	154	107	171

Table 3. Average standing crops (kg/hectare) of major species of fish collected in cove rotenone samples during epilimnial and hypolimnial discharge regimes, DeGray Lake, 1974-1982.

Species	Discharge Regime			
	Epilimnial		Hypolimnial	
	Mean kg/ha	Percent	Mean kg/ha	Percent
Gizzard shad	102.9	44.6	40.9	24.1
Bluegill	33.1	14.4	29.7	17.5
Longear	24.5	10.6	18.1	10.6
Largemouth bass	16.9	7.3	11.9	7.0
Threadfin shad	8.8	3.8	16.0	9.4
Golden redhorse	8.4	3.6	20.9	12.3
Warmouth bass	6.3	2.7	2.7	1.6
Redear sunfish	5.5	2.4	4.5	2.6
Green sunfish	4.2	1.8	3.6	2.1
Black crappie	3.9	1.7	3.0	1.8
Channel catfish	3.5	1.5	1.4	0.8
Logperch	2.5	1.1	2.7	1.6
Spotted bass	0.8	0.4	3.0	1.8
All others	9.2	4.0	11.5	6.8
TOTAL	230.5	100.0	169.9	100.0

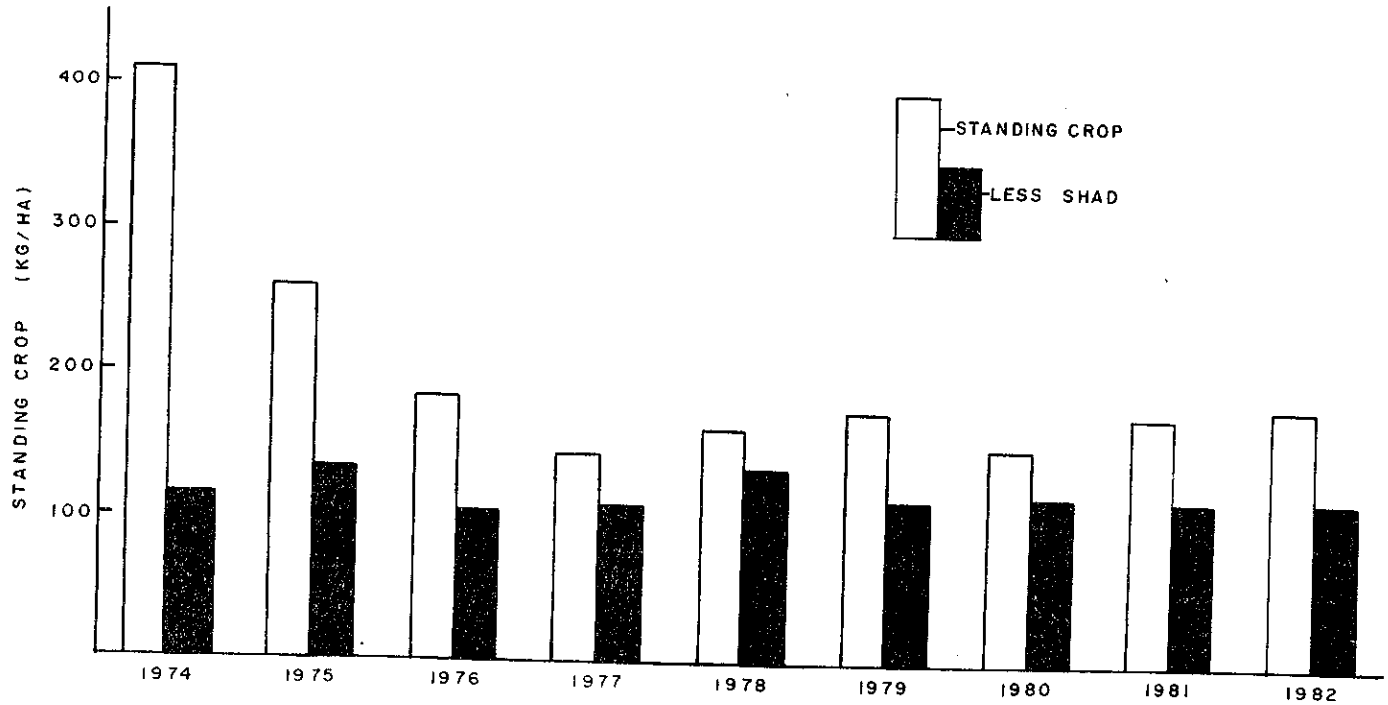

Fig. 1. Changes in total standing crop of fishes (open bass) and total standing crop less shad (solid bars), based on cove rotenone samples taken from 3 coves in DeGray Lake, Arkansas, 1974-82.

reaching a maximum of 11.5% in 1977 at a time when the spawning population was the lowest recorded during the study period (152 fish/ha).

Threadfin shad populations varied from less than 0.1 kg/ha in 1978 to 34.4 kg/ha in 1982, and made up from 0.02% to 19.5% of the total annual standing crop. A low population (0.5 kg/ha) was also recorded in 1977. Winter mortality of adults was considered the prime cause of subsequent poor reproduction (Dewey and Moen 1982a). Biomass of threadfin shad ranged from 7.7 to 17.6 kg/ha in the remaining 6 years and exceeded that of gizzard shad only in 1982 when threadfin shad made up 55% of the total weight (64 kg/ha) of shad.

Sunfishes - Five species of sunfish (bluegill, longear, redear, green, and warmouth) made up 32% of total fish biomass during epilimnial release and 34% during hypolimnial discharge even though their mean biomass decreased 19% (from 73 to 59 kg/ha) (Table 3). Bluegills were consistently the most common sunfish in the rotenone samples during both discharge regimes, making up nearly 50% of the total sunfish biomass. Standing crops of bluegills were 44.5 and 47.5 kg/ha in 1974 and 1975 respectively but only 24.3 kg/ha in 1976. Biomass of bluegills fluctuated less than 5 kg/ha between any two adjacent years from 1976 through 1982 (range 24.3 to 34.6 kg/ha, mean 28.4). The large standing crops of bluegills in 1974 and 1975 was composed mostly of fish 140-213 mm long, remnants of a large population produced in the early years of impoundment (Moen et al. 1980).

Longear sunfish made up 10.6% of total biomass of fish during both discharge periods (Table 3). They made up 33% of the sunfish biomass during epilimnial release (range 18.8 to 29.5 kg/ha, mean 24.5). During hypolimnial release longear made up 31% of the sunfish crop (range 16.5 to 22.4 kg/ha, mean 18.7). On an annual basis the longear standing crop exceeded that of bluegills only in 1976 when the bluegill biomass was at its lowest point (24.3 kg/ha) and longear (25.0 kg/ha) had their second best year.

The combined weight of the 3 remaining species of sunfish (warmouth, redear and green) averaged about 7% of the total fish biomass and 21% of the sunfish population during each discharge regime. Biomass of each of these species fluctuated from year to year and no distinct trends were recorded. Warmouth biomass varied from 1.9 to 6.5 kg/ha, redear from 2.8 to 6.8 kg/ha, and green-sunfish from 2.0 to 8.9 kg/ha.

The cove representing the middle region of the lake (compartment II) produced the highest standing crop of sunfishes during both discharge regimes (Table 4) and that representing the lower region produced the lowest. The mean biomass of sunfishes recorded for each region of the lake was less during hypolimnial release than for the previous period of epilimnial release (11%, 14%, and 38% for compartments I, II, and III respectively and 20% less for lakewide means).

Black basses - The mean biomass of the three species of black bass was 17.9 kg/ha during epilimnial release and 15.5 kg/ha during hypolimnial discharge and made up slightly less than 10% of the total biomass during each period. Although biomass was greatest in the mid-lake cove during both types of discharge, mean standing crop increased in the upper portion of the lake during hypolimnial release and decreased in the middle and lower regions (Tables 3 - 4).

Largemouth bass made up 70 to 90% of the total black bass weight each year with mean annual standing crops that varied from 9.7 to 23.9 kg/ha. In individual coves the biomass of largemouth bass ranged from 5.5 to 46.0 kg/ha and two-fold or greater fluctuations from one year to the next were common for all 3 coves during both types of release.

Standing crops of largemouth bass were not greatly influenced by the biomass of YOY or age I fish unless the weight of adults was exceptionally low. The biomass of either of these two categories seldom exceeded 20% of the total weight of largemouth bass; YOY averaged 2.2 kg/ha (range 0.6-8.2); biomass of age I fish averaged 2.0 kg/ha (range 0.4-6.3). Reproduction measured by weight of YOY in August was 2.9 kg/ha during epilimnial release and 1.8 kg/ha during hypolimnial discharge, a 38% decrease. The down-lake cove had slightly better production of YOY largemouth bass than the other two, but ranked third in biomass of older fish. Reproduction in any one year was

Table 4. Average biomass (kg/ha) of major groups of fishes during epilimnial (E) and hypolimnial (H) release regimes in DeGray Lake, Arkansas, 1974-1982.

GROUP	SAMPLE LOCATION							
	Up-lake		Mid-lake		Down-lake		Lakewide Means	
	E	H	E	H	E	H	E	H
Shad	147.9	84.1	115.7	38.9	71.6	47.7	111.7	63.5
Sunfishes	71.6	63.9	84.7	72.5	63.1	39.0	73.1	58.5
Black basses	12.4	15.8	27.0	19.1	14.0	10.5	17.8	15.1
Suckers	28.0	61.6	0.3	1.0	0.1	0.1	9.5	14.5
Catfishes	2.9	4.9	3.9	5.8	9.6	1.7	5.6	4.3
Crappies	5.1	6.1	7.5	5.3	0.7	3.0	4.4	4.8
Sportfish	94.2	89.7	120.7	97.3	86.5	54.1	100.5	80.4

not consistent among the 3 coves except for the peak year of 1977 when YOY were abundant in all 3 coves. The best year for bass reproduction during hypolimnial release and second highest in 9 years was 1982 (3.0 kg/ha compared to 5.4 kg/ha in 1977). Although high reproduction was occasionally followed by an increase in yearlings the following year, yearling bass crops were not significantly correlated with populations of YOY the previous August. Positive, but weak correlations were noted for the upper and mid-lake coves ("r" = .12 and .47 respectively), but fairly strong negative correlation ("r" = - .602) was found in the down-lake cove.

Spotted bass made up only 5% of the weight of black bass during epilimnial release but increased each year, especially in the upper two-thirds of the lake and made up 20% of the total black bass biomass during hypolimnial discharge. YOY as a portion of the standing crop of spotted bass in the rotenone samples was significantly greater than for largemouth bass; YOY spotted bass made up 20% or more of the spotted bass weight in 22 of the 27 samples and exceeded 50% in 3 of the 9 samples conducted in the up-lake cove.

Smallmouth bass were scarce during both discharge regimes, making up less than 2% of the black bass weight during epilimnial release and about 3% during hypolimnial discharge. Smallmouth did not appear in our samples until 1976. They were the most abundant in the down-lake cove where 18 of the 23 fish were recorded. Arkansas Game & Fish biologists who conducted rotenone samples on DeGray Lake 1968-1973 reported that smallmouth bass were abundant only in 1968, the first year of impoundment. In their September 1968 sample, smallmouth bass biomass was recorded as 10 kg/ha, nearly 3 times greater than largemouth bass in that sample (Arkansas Game and Fish, unpublished data).

<u>Suckers</u> - Most of the biomass of suckers was concentrated in the upper region of the lake (Table 4). Golden redhorse made up 80 to 99% of the total biomass of suckers each year in the up-lake cove, increasing from 7 kg/ha in 1974 to 72 kg/ha in 1982. During both release periods only shad and sunfish had greater biomasses. The spotted sucker was the most common sucker taken in the lower two coves.

Catfishes - The biomass of catfishes was highly variable from year to year and among the 3 coves in any one year. Among the 4 species in the catfish category (channel catfish, flathead catfish, yellow bullhead, and black bullhead) only the channel and flathead catfish contributed significantly to the annual mean weight of catfishes, contributing 53 and 43% respectively. Although yellow bullheads contributed less than 4% of annual mean weight, they were noted in all but 4 of the 27 samples. Black bullheads were identified in only 4 of the 27 samples. Fluctuation in biomass of flathead catfish was usually the result of the presence or absence of one or two large fish. The decrease in mean biomass of catfishes in the down-lake cove during hypolimnial release (Table 4) was due to the absence of channel catfish following high biomass in 1974-77 (7.5 - 15.0 kg/ha).

Crappies - Crappies made up 1.9% of the lakewide mean biomass during epilimnial release and 2.8% during hypolimnial discharge even though their biomass for each period was similar (Table 4). Black crappies were dominant each year except 1980 when white crappie mean biomass (1.4 kg/ha) was about 30% greater than for black crappie. Annual mean biomass of black crappie varied from 0.9 to 8.4 kg/ha; lakewide mean biomass declined from 3.9 kg/ha during epilimnial release to 2.7 kg/ha during hypolimnial discharge. Black crappies did not display a definite trend in the lakewide populations; however, they were consistently more abundant in the mid-lake cove than in either of the other two coves. Annual biomass of white crappie varied from less than 0.1 to 5.1 kg/ha; mean biomass increased from 0.4 kg/ha during epilimnial release to 1.7 kg/ha during hypolimnial discharge. White crappie populations were low 1974-76, increased significantly 1977-80 and then decreased the last two years. White crappie biomass was consistently higher in the up-lake cove than the other two. Except for 1979, white crappie populations were usually less than 0.1 kg/ha in the mid-lake and down-lake coves.

Miscellaneous fish - Among the 20 or more species or groups that we have placed in the category of miscellaneous fish, only logperch, minnows, and silversides individually or as a group contributed 0.1 kg/ha or more to the mean biomass each year. Their combined mean biomass (5.2 kg/ha) accounted for

2.3% of the mean biomass of all fish during the study period.

Madtoms, darters, and topminnows appeared in most samples but contributed relatively little to the biomass as individual groups. Only two carp were sampled (one in each release period). No carp were reported in the 1968-73 rotenone sampling by Arkansas Game and Fish biologists. Apparently, few carp were impounded at the time of gate closure.

White bass and the hybrid of white bass x striped bass were taken occasionally, but because they prefer pelagic water are seldom sampled in coves. White bass were introduced by the Arkansas Game and Fish Commission in 1969 and the hybrid white x striped bass in 1975. Electrofishing surveys on the spawning grounds in the upper end of the lake each spring 1975-81 have indicated that there were no reproductive failures for white bass during this monitoring period.

Sportfish - The sportfish category includes black basses, sunfishes, crappies, catfishes, white bass, and white bass x striped bass hybrids (14 species). These are the species most pursued by fisherman. Although there was a 3-fold variation in biomass of sport fish among the 27 samples, fluctuation was not that large within a single cove. Biomass in the up-lake cove fluctuated the least. The mid-lake cove had the best population of sport fish in 7 of the 9 years of sampling. Standing crops were lower in each cove during hypolimnial release, especially in the lower region of the lake where mean biomass for the period decreased 37% from mean weight during epilimnial discharge (Table 5).

Spring largemouth bass population dynamics - Because largemouth bass were considered the most abundant predators in DeGray Lake and the most important sportfish, they were chosen for intensive study. Electrofishing was used in three coves each spring to determine annual trends in abundance, production, growth, survival, mortality, and spatial distribution in the lake. Dewey and Moen (1982b) reported on these facets of largemouth population dynamics as they applied to the period of epilimnial release (1975-1979). Although hypolimnial

Table 5. Standing crop of sport fish (kg/ha) in sampling coves, DeGray Lake, 1974-1982.

Release Regime	Year	LOCATION Up-lake	Mid-lake	Down-lake	Lakewide Mean
Epilimnial	1974	94.3	119.5	92.9	102.2
	1975	99.3	133.5	113.7	115.5
	1976	84.2	91.5	67.4	81.0
	1977	90.0	111.0	79.2	93.4
	1978	103.3	147.9	79.5	110.2
Mean		94.2	120.7	86.5	100.5
Hypolimnial	1979	83.5	119.8	54.6	86.0
	1980	75.1	69.5	57.4	67.3
	1981	93.7	102.9	53.9	83.5
	1982	106.3	97.1	50.4	84.6
Mean		89.7	97.3	54.1	80.4

discharge commenced in 1979 the spring bass estimates were not influenced by the change, thus only three spring estimates were available during hypolimnial release.

During epilimnial discharge mean population estimates ranged from 86 to 163 bass per hectare (mean 119); the populations were dominated by bass of ages I and II (73% to 96%). Populations were highest in the mid-lake cove each year except 1977, agreeing with August cove samples which also indicated that the standing crop of largemouth bass was highest in mid-lake each year. During hypolimnial release lakewide population estimates were 84, 77, and 41 bass per hectare for the up-lake, mid-lake and down-lake coves respectively (mean 68). Populations were again dominated by bass ages I and II (75% to 97%). Population estimates decreased each year during the 8 years except 1979 and 1982 when increases from the previous year of 30% were recorded in lakewide means and from 3% to 75% in individual coves. The decrease between epilimnial and hypolimnial means for individual coves was 25%, 25%, and 53% for the up-lake, mid-lake, and down-lake coves, respectively (Table 6).

Lakewide production of largemouth bass averaged 26.6 kg/ha during eplimnial release and 18.2 kg/ha during hypolimnial discharge, a 32% reduction that followed the trend of population estimates (Table 7). Variation among individual coves was high but generally declined in each cove each year. Production was highest in the mid-lake cove each year, where estimates remained relatively stable after declining substantially in 1976-77. Production was also estimated for 12 year classes at various ages (Table 8). Although production of a single year class varied between coves for a given growth year, production in all three coves usually showed the same trend, especially during the first three years of growth. Production was greatest during the first or second year of life. Among the seven year classes for which data are available for the first two years of life, production was greatest the first year for the 1976, 1977, 1979, and 1980 year classes; production was greatest in the second year for the 1974, 1975, and 1978 year classes. Substantial growth occurred during the third year, but because total mortality rates were high, most of the production contributed by a single year class was made during the first two growing seasons.

Table 6. Population estimates (number of age I+ largemouth bass per hectare each April) for coves in DeGray Lake (95% C.I. in parentheses), and lakewide mean estimates, 1975-1982.

Release Regime	Year	Cove Location			Lakewide Mean
		Up-lake	Mid-lake	Down-lake	
Epilimnial	1975	110 (86-147)	202 (161-302)	140 (114-183)	163
	1976	142 (116-182)	175 (146-227)	63 (45-98)	134
	1977	144 (105-200)	84 (53-102)	75 (56-113)	95
	1978	75 (50-119)	103 (78-140)	66 (52-156)	86
	1979	101 (62-193)	137 (102-192)	91 (62-153)	115
Hypolimnial	1980	112 (88-266)	106 (79-152)	46 (34-64)	89
	1981	69 (49-89)	53 (26-78)	28 (13-74)	49
	1982	71 (45-91)	72 (36-41)	49 (32-63)	65

Table 7. Annual production (kg/ha) of largemouth bass estimated from spring electrofishing collections by cove, DeGray Lake, 1975-82.

Release Regime	Years of Capture	Sample Site: Up-lake	Sample Site: Mid-lake	Sample Site: Down-lake	Lakewide Mean
Eplimnial	1975-76[a]	8.9	42.6	21.6	30.9
	1976-77	21.2	31.9	30.2	29.1
	1977-78	20.1	31.5	25.7	27.2
	1978-79	14.3	26.3	11.5	19.1
Mean		18.6	33.1	22.2	26.6
Hypolimnial	1979-80	20.5	25.6	15.4	21.3
	1980-81	13.4	18.8	10.1	14.9
	1981-82	12.5	23.8	15.0	18.5
Mean		15.5	22.7	13.5	18.2

[a]Collections were made in April of these two years and production was determined for the year between sampling periods.

Table 8. Estimated production of largemouth bass (kg/ha) in DeGray Lake by year class, 1975-82.

Year Class and Year of Collection	Age Groups	LOCATION			Lakewide Mean	Total Production by Year Class
		Up-lake	Mid-lake	Down-lake		
1970						1.0
1975-76	V-VI	0.5	1.9	--	1.0	
1971						1.0
1975-76	IV-V	0.4	1.1	0.1	0.6	
1978-79	VI-VIII	--	0.7	0.1	0.4	
1972						4.9
1975-76	III-IV	1.4	4.2	1.4	2.7	
1976-77	IV-V	0.3	1.3	1.3	1.1	
1977-78	V-VI	--	1.5	1.0	1.0	
1978-79	VI-VII	--	0.1	0.2	0.1	
1973						4.6
1975-76	II-III	1.6	3.8	3.6	3.2	
1976-77	III-IV	1.2	1.1	0.7	1.0	
1977-78	IV-V	--	--	0.8	0.2	
1978-79	V-VI	--	0.4	0.2	0.2	
1974						30.3
1974-75	0-I	3.3	17.7	5.4	8.9	
1975-76	I-II	7.8	18.4	10.9	13.7	
1976-77	I-III	3.0	6.5	4.6	5.1	

(Continued)

(Sheet 1 of 3)

Table 8 (Con't)

1977-78	III-IV	1.5	2.3	2.0	2.0	
1978-79	IV-V	--	0.7	0.8	0.6	
1975						25.9
1975-76	0-I	7.7	11.5	5.6	8.4	
1976-77	I-II	12.8	13.4	7.5	11.5	
1977-78	II-III	3.8	5.7	4.1	4.8	
1978-79	III-IV	1.5	1.1	1.1	1.2	
1976						15.9
1976-77	0-I	3.9	6.9	13.4	8.2	
1977-78	I-II	6.8	7.3	4.4	6.3	
1978-79	II-III	2.0	0.6	0.5	0.9	
1979-80	III-IV	0.3	0.4	0.8	0.5	
1977						27.7
1977-78	0-I	8.0	23.0	9.7	15.6	
1978-79	I-II	6.3	9.9	15.0	7.6	
1979-80	II-III	2.2	5.3	4.2	3.9	
1980-81	III-IV	--	0.4	0.8	0.6	
1978						21.1
1978-79	0-I	4.5	12.8	3.6	8.1	
1979-80	I-II	10.0	11.5	6.5	9.3	
1980-81	II-III	3.1	4.0	1.6	2.9	
1981-82	III-IV	0.3	0.8	1.2	0.8	

(Continued)

(Sheet 2 of 3)

Table 8 (Concluded)

1979						13.8
1979-80	0-I	7.8	7.8	3.9	6.5	
1980-81	I-II	5.9	6.4	2.2	4.8	
1981-82	II-III	2.9	2.5	2.0	2.5	
1980						9.1
1980-81	0-I	4.4	6.5	5.9	5.6	
1981-82	I-II	5.8	3.2	1.6	3.5	
1981						9.4
1981-82	0-I	3.6	15.9	8.8	9.4	

(Sheet 3 of 3)

Annual survival rates estimated from ratios of the total number of bass per hectare (less age I bass) to the total number in the preceding year were lowest in the upper region of the lake and highest in the down-lake cove. Abnormally high survival was noted in 3 of the 21 determinations because of an artifact in sampling where the number of age II bass collected exceeded the number of age I bass collected the previous year (Table 9).

Instantaneous mortality rates (Z) were relatively high, corresponding to low survival rates. Total mortality has highest (mean, 0.76; range 0.21-0.86) from age I to II. Mortality from age II to III ranged from 0.45 to 0.91 (mean, 0.71), and from age III to IV, 0.31 to 0.92 (mean, 0.63). Differences between epilimnial and hypolimnial survival and mortality rates were not significant.

Although bass populations declined during the study they continued to grow at about the same rate. The difference between epilimnial and hypolimnial mean empirical length at each annulus varied but was not significant (Table 10). Mean relative weights (W_r: Wege and Anderson 1978) for each age group during the two release regimes indicate little difference between epilimnial and hypolimnial relative weight for age groups I-IV. However, relative weights of bass of age groups V-VIII taken from the mid-lake and down-lake areas during hypolimnial release were consistently less than those for fish of the same age groups during epilimnial release (Table 11).

Pelagic fishes - YOY fish collected by mid-water trawls included gizzard shad, threadfin shad, crappies, sunfishes, brook silversides, Mississippi silversides, logperch, black basses, white bass, catfishes, and minnows (mostly *Notropis* spp.). Shad, sunfishes, and crappies made up more than 98% of the fish collected.

Shad reached peak abundance during the last week of May or first week of June each year. Lakewide peak catches of shad varied from 1/m^3 in 1975 to 12.3/m^3 in 1979. Peak catches in the upper region of the lake ranged from 2.3 to 8.5/m^3 and were highest in this region in 5 of the 8 years of sampling. The lower region of the lake usually produced the lowest catches; however, this area of the lake produced the highest recorded catch, 8 hauls on May 17, 1979 (averaged 19.7 shad per cubic meter). Because of net avoidance, down-lake water

Table 9. Annual survival(S) and instantaneous mortality rates (Z) for largemouth bass in DeGray Lake, 1975-82.

Years of Capture	LOCATION					
	Up-lake		Mid-lake		Down-lake	
	S	Z	S	Z	S	Z
1975-76	0.22	1.53	0.45	0.79	0.26	1.35
1976-77	0.39	0.94	0.24	1.41	0.81[a]	0.20
1977-78	0.15	1.90	0.28	1.26	0.35	1.04
1978-79	0.33	1.11	1.01[a]	0.04	0.83[a]	0.19
1979-80	0.30	1.21	0.37	0.99	0.32	1.14
1980-81	0.11	2.23	0.28	1.26	0.30	1.19
1981-82	0.26	2.98	0.28	1.26	0.53	1.34

[a]Number of age II bass collected exceeded the number of age I bass collected in the previous years.

Table 10. Mean empirical lengths (mm) of 7,710 largemouth bass (sexes combined) collected in April 1975-79 during epilimnial (E) release and 2,318 largemouth bass collected in April 1980-82 during hypolimnial (H) release, DeGray Lake, Arkansas.

Release Regime and Year classes	Mean Length at each annulus								
	1	2	3	4	5	6	7	8	9
E 1978-1966	160	284	351	400	442	483	530	551	547
H 1981-1974	179	278	345	390	446	467	527	564	

Table 11. Mean relative weight of largemouth bass of each age group collected from three regions of Lake DeGray each spring during epilimnial (E) release (1975-79) and hypolimnial (H) release (1980-82).

Age Group	LOCATION					
	Up-lake		Mid-lake		Down-lake	
	E	H	E	H	E	H
I	80	79	79	79	76	79
II	83	82	82	82	80	81
III	81	86	83	84	81	82
IV	90	89	87	86	85	84
V	85	97	98	87	92	88
VI	91	88	101	83	106	93
VII	96	103	107	85	112	96
VIII	97		107	98	120	87
IX			101	99		

clarity, and (in 1977 and 78) the winterkill of threadfin shad, catches usually decreased sharply in July, especially in the mid-lake and down-lake compartments.

The pelagic distribution and seasonal abundance of larval and juvenile shad permitted reasonable estimates of production from the midwater trawl samples. Lakewide production estimates ranged from 10.4 kg/ha to 60.7 kg/ha over the 8 years of sampling; production during hypolimnial release was 67% higher than during epilimnial release. Production was also greater in each of the three regions during hypolimnial release and highest in the upper region in all years except 1975&1979 (Table 12). No significant correlation was found between bass production and shad production. If shad production were the key to bass production, we believe that bass production (Table 6) should have been higher in the upper section of the lake during both release regimes and generally higher during hypolimnial discharge.

Crappies were most abundant in mid-May in the midwater trawl collections; however, discharge samples indicated that peak abundance occurred in early May for most years, before trawling began (Dewey and Moen 1982c). Lakewide peak catches of crappies varied from $0.21/m^3$ to $2.46/m^3$ averaging $0.55/m^3$ during epilimnial discharge and $1.31/m^3$ during hypolimnial release. The mid-lake area of the lake produced the highest catches of crappies in 7 of the 8 years of sampling. Mean densities were usually lowest in the lower section of the lake.

Larval sunfishes were usually most abundant during June each year. Lakewide peak catches varied from 0.48 to $6.52/m^3$, averaging $2.45/m^3$ during epilimnial discharge and $1.35/m^3$ during hypolimnial release. Higher mean abundance during epilimnial release was largely due to an exceptionally successful hatch of sunfishes in 1977. Mean densities of YOY sunfishes were similar throughout the lake each year except 1977 when they were consistently highest in the upper area. A mean density of $10.27/m^3$ in the upper section of the lake in June was the highest in 8 years of sampling.

Among the less abundant taxa, silversides were most numerous, followed by minnows, and logperch.

Table 12. Estimated production (kg/ha) of young-of-the year shad in DeGray Lake, 1975-1982.

Release Regime	Year of Collection	LOCATION			Lakewide Mean
		Up-lake	Mid-lake	Down-lake	
Epilimnial	1975	12.9	17.3	14.7	15.5
	1976	80.6	42.3	36.6	49.2
	1977	14.9	8.4	10.1	10.4
	1978	34.7	13.1	14.7	18.5
Mean		35.8	20.3	19.0	23.4
Hypolimnial	1979	63.3	71.8	41.8	60.7
	1980	67.0	35.2	21.5	38.2
	1981	76.9	38.6	22.1	42.3
	1982	18.8	15.3	21.0	15.1
Mean		56.5	40.2	26.6	39.1

[1]weighted.

Summary and Discussion

Annual rotenone sampling indicated a 26% decline in mean biomass of fishes from the epilimnial to hypolimnial discharge regime (Table 3). However, when we looked at groups of related species and some of the important individual species we found considerable variation among coves and years, and between release regimes. A general decline was noted in total biomass during epilimnial release followed by relative stability during hypolimnial release (Table 2). Less change was shown in the up-lake cove than in the other two coves. For each cove during hypolimnial release, there were individual years when the total standing crop exceeded the total crop for one or more years during epilimnial release (Table 2).

Mean biomass of shad was 43%, 11% and 33% less in the up-lake, mid-lake, and down-lake coves during hypolimnial release than during epilimnial discharge. Mean biomass of sunfishes was also less for each cove during hypolimnial release. Black basses increased in the upper cove and decreased in each of the lower two coves. Suckers increased two-fold in the upper cove and three-fold in the mid-lake cove, with no change in the down-lake area. Catfishes increased in the upper two coves and decreased sharply in the lower region. Crappie increased in the upper and lower cove and decreased in the mid-lake cove (Table 4). The mean standing crop of sport fish in each cove during hypolimnial release was less than that for epilimnial release, and the decline was highest in the down-lake cove and least in the up-lake area. However, variation was high between years in both release periods as noted for other categories (Table 5).

Spring population estimates and production estimates of largemouth bass were considerably lower during hypolimnial release (Table 6 & 7), although growth (length) survival and instantaneous mortality rates appeared to show little difference.

Larval and juvenile shad and crappie populations were higher and sunfishes lower during hypolimnial release. Larval shad production was 67% higher during hypolimnial release.

In the search for cause-and-effect relationships for the changes observed we can point to numerous combinations of physical, chemical, and biological factors that may have influenced these changes. Over 15 years ago, Jenkins (1968) stated that "readily-measured keys to potential fish production have been sought by fishery biologists for over 30 years." Jenkins (1968) also reviewed the more important indices developed by other workers and explored by multiple regression analyses the effects of nine environmental factors (reservoir area, mean depth, total dissolved solids, storage ratio, shore development, age, water level fluctuation, outlet depth, and growing season) on standing crops reported from 127 U.S. reservoirs. None of the regressions provided an explanation of more than 37 percent of the total variability in standing crop, although the correlations were highly significant. Among the nine factors only four (total dissolved solids, age of the reservoir, water level fluctuation, and outlet depth) may have changed enough during our study to have influenced the standing crop. Ryder's (1965) morphoedaphic index (MEI, dissolved solids/mean depth) was tested by Jenkins (1970) in relation to the standing crops of 140 reservoirs. The MEI explained 33 to 62% of the variability of standing crops, depending on the type of reservoir. More recent investigations have shown that chlorophyll concentration (Oglesby 1977) and total phosphorus (Hanson & Leggett 1982) were better predictors of fish production than either TDS or MEI. Prepas (1983) concluded that TDS does not seem to be a useful parameter in predicting lake productivity of large natural lakes used alone or in the MEI.

No important differences were recorded in TDS values for DeGray Lake between epilimnial and hypolimnial release. However, there was a trend toward lower TDS from up-lake to down-lake stations during both release periods. Although data concerning chlorophyll concentrations in DeGray Lake are relatively scarce, these data do show significantly higher concentrations in upstream samples and as expected phosphorus concentrations also followed this trend (TDS, chlorophyll and phosphorus from unpublished data collected by Ouachita Baptist University under contract with the Environmental and Water Quality Operational Studies Program, Waterways Experiment Station, Vicksburg, Mississippi).

We found a significant negative correlation between standing crop and age of the reservoir only for the down-lake area of the lake. However, aging of the reservoir is considered important.

Stable or rising water levels during the spawning season have been associated with increased survival of largemouth bass (Jackson, 1958; Patriarche and Campbell 1958; Bross 1967; Aggus and Elliott 1975). Large year classes of several species of fish in the large reservoirs on the Missouri River have been associated with a rise and continuance of high spring water level (Gasaway 1970; Moen 1974; Beckman and Elrod 1971). However, Rainwater and Houser (1975) were unable to correlate total biomass and numbers per hectare with water levels in Beaver Lake, Arkansas. Dewey and Moen (1982b) studied the seasonal biomass of largemouth bass in DeGray Lake and concluded that the mortality of YOY bass in late summer of each year was critical in determining year class strength. Although DeGray water levels were subject to sharp increases during the study (maximum of 6 m in June 1974) the rise during the spawning and nursery period of April-July (Fig. 2) was usually of short duration and less than 1 meter (a one-meter increase inundates about 300 hectares). The largest numbers of YOY largemouth bass found in the August rotenone samples in DeGray Lake (1977, 1978 and 1982) were produced when spring and early summer water levels were stable or decreasing.

Changing the outlet from epilimnial to hypolimnial withdrawal was intended to be the most important change in environmental effects. The premise that a hypolimnial release stores heat and discharges nutrients and that an epilimnial release has the opposite effect is widely accepted (Murphy 1962; Johnson and Berst 1965; Wright 1967; Dorris 1969; Martin and Arneson 1978). Based on this premise we would expect a decline in production and standing crops of fish during hypolimnial release. Standing crops in DeGray Lake were relatively stable, production and numbers of largemouth bass declined, and YOY shad production increased during hypolimnial release. The increase in shad production was not unexpected. The change in outlet depth affected the entrainment of YOY shad more than the other taxa. The entrainment of shad was more than 10 times greater during epilimnial release than during hypolimnial discharges (Dewey and Moen, 1983b). Regression analysis indicated that fish standing crop was positively related to outlet depth in hydropower-storage reservoirs (Jenkins 1970).

With few exceptions, reservoirs have relatively high production immediately following impoundment and then decline in productivity to some lower level. This

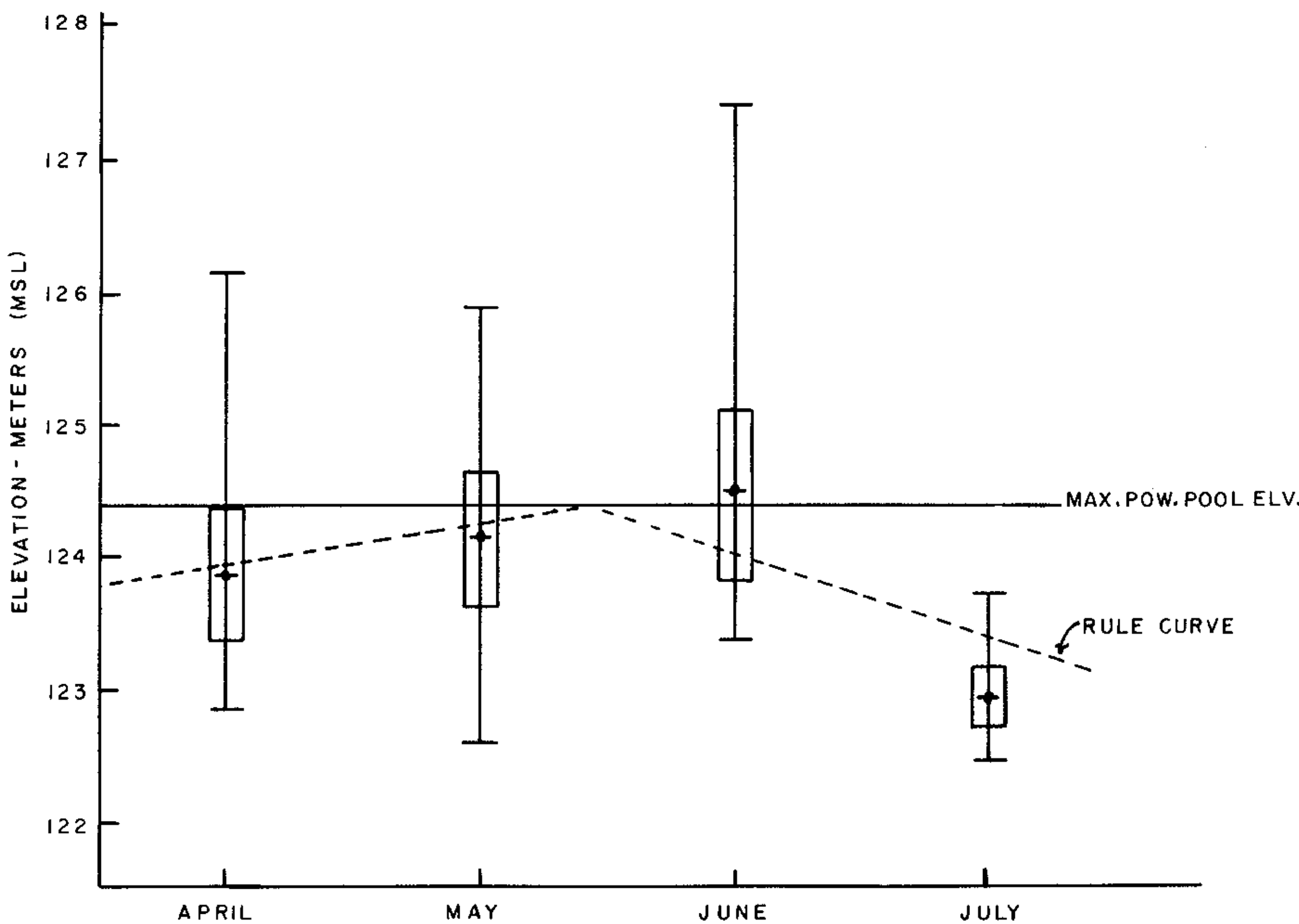

Fig. 2. Fluctuations in water surface elevations, DeGray Lake, Arkansas, 1974-82. Brackets indicate maximum for each month and bars include one standard deviation either side of mean.

lower level of productivity, expressed in terms of the average standing crop of fishes over the long term, is often referred to as "carrying capacity." Krumholz (1948) suggests that carrying capacity should indicate the upper limit of weight of fish that can be supported by a body of water over an extended period of time. More recently, Jenkins (1974) proposed that carrying capacity be defined as the minimal standing crop at the most critical period in early spring. However, most standing crop estimates for reservoirs are made in the fall (August or September) near the time when peak biomass normally occurs. Regardless of the preference in time of observation we believe that the standing crop estimates we are reporting for DeGray Lake, based on estimates that were obtained using standardized methods, represent carrying capacity of the representative coves in early August, especially for the period 1975-82. In using this 8-year period, we assume that the estimates for 1974 represent a relatively high period of productivity following impoundment and that a period of more normal or steady state of productivity was attained after 1974. This also allows for a more reasonable comparison between the estimates for epilimnial versus hypolimnial discharge (Table 13). Ploskey and Jenkins (1982) studied the fish and fish-food interactions of DeGray (1974-80) and found a decline in surplus fish production and that the ratio of available prey to fish predators decreased as a result of a decrease in biomass of prey fish, stating that these trends reflect decreasing secondary production commonly associated with reservoir aging after impoundment.

We have concluded that the observed changes in standing crops of fishes and other facets of their population dynamics were not significantly related to the change from an epilimnial to hypolimnial release, but were more closely related to alterations of the environment that were brought about by aging of the reservoir.

Table 13. Average standing crop of fish (kg/ha) and (in parentheses) range, based on cove rotenone samples in August during 4 years of epilimnial release (1979-82) and the 8-year mean (95% of confidence limits in parentheses), DeGray Lake, Arkansas.

Sample Location	DISCHARGE REGIME		Eight-year Mean
	Epilimnial	Hypolimnial	(95% C L)
Up-lake	243 (160-328)	253 (220-305)	247 (202-292)
Mid-lake	158 (130-187)	154 (119-184)	157 (135-179)
Down-lake	146 (106-224)	107 (88 -129)	126 (90-162)
Lake-wide	184 (139-246)	171 (151-181)	177 (249-209)

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ENTRAINMENT OF LARVAL FISH FROM DeGRAY LAKE, ARKANSAS, DURING EPILIMNIAL AND HYPOLIMNIAL DISCHARGES

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Abstract

Vertical distribution and density of reservoir populations of fish larvae vulnerable to entrainment were related to estimated entrainment during water releases from the epilimnion and hypolimnion. Relations between larval entrainment and outlet depth, time of day, season and discharge volumes from 1976 to 1982 are described. Linear regression models were developed to predict larval loss during each type of discharge from densities of larvae immediately above the dam. The slopes of the regression equations indicated a higher degree of entrainment during epilimnial than during hypolimnial releases for all taxa. Percentages of the larvae immediately above the dam that were entrained at different densities (no./m^3) were also calculated for each taxon. These data do not indicate a response to the depth of the outlet but a passive entrainment during both epilimnial and hypolimnial discharges. The change in the outlet depth affected the entrainment of shad (Dorosoma) more than that of other taxa. Most of the entrainment percentages for shad were more than 10 times greater during epilimnial than during hypolimnial discharges. Percentages of the total reservoir larval population entrained were calculated and these indicated little impact due to entrainment during either type of discharge.

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There is growing recognition that mortality of aquatic biota from entrainment or impingement at power plant intakes adversely affects estuarine, riverine, lacustrine, and coastal ecosystems. The period of vulnerability of larval fishes to entrainment is influenced by various biological characteristics of the larvae such as motility; physiological and behavioral responses to factors such as temperature, salinity, oxygen concentration, and currents; vertical and horizontal distribution in the vicinity of the power plant intake; and growth rate (Boreman 1977). The magnitude of total larval fish entrainment also depends on physical factors such as the depth of the outlet and the amount of water discharged seasonally. Entrainment of larval fish during power generation has been well documented (Grimes 1975; Synder 1975; Boreman 1977; Mathur et al. 1977; Walburg 1971; Kelso and Milburn 1979; Kelso and Leslie 1979; Logan and Masnik 1979; Hatch 1980; Prince and Mengel 1980). However, no comparison of larval fish entrainment during epilimnial and hypolimnial releases from the same reservoir has been reported.

In 1974 the U. S. Fish and Wildlife Service, U. S. Army Corps of Engineers Waterways Experiment Station, Arkansas Game and Fish Commission, and State and private universities began intensive studies on DeGray Lake and its tailwater to evaluate effects of altered outlet depth on biota and water quality of the reservoir and tailwater. One phase of this research involved measuring the loss of larval fish through DeGray Dam during power generation. Moen and Dewey (1978) studied losses of larval fish in 1976 and 1977 during epilimnial release.

This study describes the relations between larval fish entrainment and outlet level, time of day, season, and discharge volume in 1976-82, and compares the seasonal entrainment of shads (Dorosoma), crappies (Pomoxis), and sunfishes (Lepomis) during epilimnial and hypolimnial discharges.

STUDY AREA

DeGray Lake, in the Ouachita Mountains of west-central Arkansas, was impounded in 1969. At normal pool elevation (124.4 m above mean sea level) the area of the reservoir is 5,428 hectares, and maximum and mean depths are 57 and 15 m. The reservoir extends in a west-northwest direction for about 34 km and has a shoreline 207 km long. A multi-level release structure allows water to be selectively withdrawn from one of three 6.4-m-square openings (on a square intake tower), the midpoints of which are at elevations of 120.4, 115.8, and 108.2 m above mean sea level (Middleton 1967). Within a typical generation period, the intake plume extends from the surface to a depth of 15 m during epilimnial release and from 4 to 35 m depth during hypolimnial release (Fig. 1). Therefore, during hypolimnial release the intake plume extends into the epilimnion which extends to depths between 6 and 7 m during summer stratification. Water was withdrawn from the upper outlet through 1978 and from the lower opening in 1979-82.

Discharge of water through DeGray Dam is for "peaking" power, maintenance of downstream flows, and flood control. Discharge rate ranged from 35 to 155 m^3/second during the study. At maximum discharge rates, water current velocities 20 m in front of the intake tower were 0.51 m/second during epilimnial release and 0.025 m/second during hypolimnial release; at the sampling site below the dam, current velocity was 1.2 m/second.

METHODS

Larvae were collected 40 m downstream from the discharge with a stationary net of 0.790-u mesh 3 m long and 1 m in diameter (Moen and Dewey, 1978). The net was equipped with a flowmeter and a collection bucket. Samples were taken near the middle of the water column at depths of 1.0 to 1.5 m. A 2-h discharge on a predetermined schedule of power generation rate and time on each sampling date was arranged through the U. S. Army Corps of Engineers and Arkansas Power and Light Company. After allowing at least a 20-minute flushing period to clear the sampling area, we fished the net on each sampling day for 5-minute periods separated by 15-minute intervals (six samples) in 1976, and for 10 minutes separated by 20-minute invervals (four samples) from 1977 to 1981. Samples were collected weekly from April to July 1976-81. Sampling was conducted in August in 1976-78, but was discontinued after July in 1979-81 because few larvae were collected by late July. Diel sampling was done in May each year, 1977-81.

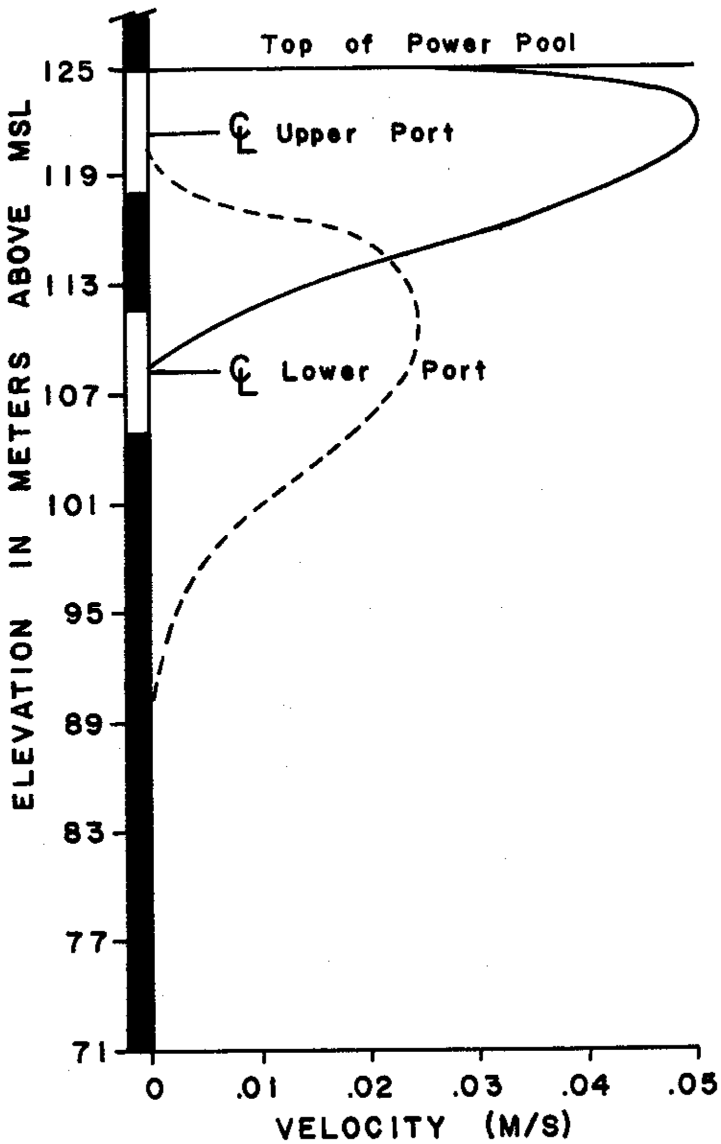

Fig. 1. Typical intake plume during epilimnial (solid line) and hypolimnial (dashed line) release, DeGray Lake (msl = sea level).

Diel entrainment samples were collected at 30-minute intervals from 0900 to 1500 hours during the day and from 2100 through 2400 hours at night. Samples were also collected during a long-term power generation period in June 1981.

The volume of water discharged each week was calculated from daily discharge records furnished by the U. S. Army Corps of Engineers. The midpoint of each weekly summation was the weekly sampling date. The mean number of larvae per cubic meter of water filtered on the sampling date multiplied by the total volume of water discharged during the corresponding week provided the estimated larval fish loss.

Midwater trawling immediately above the dam was used to measure the abundance of larval fish vulnerable to entrainment (Dewey and Moen, 1982). Semi-monthly comparisons of midwater trawl catches and catches of larvae in the discharge were made from May to mid-August each year except 1980, when weekly comparisons were made. In some years, trawling and discharge sampling were discontinued after July due to avoidance of the trawl and intake structure as young-of-the-year fishes grew larger. All trawl collections were made at night, and discharge samples were collected during the day. Discharge samples were taken the day before trawl collections 1976-1978, and on the same day as trawl collections in 1980 and 1981. In 1979, 2 days elapsed between discharge and trawl samples.

In June 1980, April and May 1981, and April, May, and June 1982, larval fish were sampled at discrete depths to determine vertical distribution of larval fish in the area of the intake structure. In 1980, stratified hauls were made at the surface and at depths of 5, 7, 10, 12 and 15 m. In 1981 and 1982, replicate stratified hauls were made at the surface and at depths of 3, 5, 7, 10, 12, and 15 m. For each haul, the trawl was lowered to the appropriate depth, opened, fished for 3 minutes, and closed before retrieval. Stratified sampling was done during daytime in 1980 and 1981, and during both afternoon (1300 to 1600 hours) and night (2100 to 2400 hours) in 1982. The afternoon sampling was conducted to represent daytime vertical distribution, since most of the power generation at DeGray occurs in the afternoon. Entrainment samples were collected in 1982 during the same time period that sampling was being conducted to determine the vertical distribution. Six samples were collected at 20-minute intervals during each sampling period.

Lakewide population estimates were calculated from midwater trawl collections made at 2-week intervals in the upper, middle, and lower sections of the reservoir. Epilimnial water to a depth of 7 m was sampled by making oblique hauls. Mean densities (no./m^3) for each section of the reservoir were multiplied by the volume of the epilimnion in that section to obtain a population estimate. Weekly estimates of entrainment were compared with the lakewide population estimate to determine the percentage of the reservoir population entrained.

In decreasing order of abundance, collections of larvae included shads, sunfishes, crappies, logperch (*Percina caprodes*), brook silversides (*Labidesthes sicculus*), shiners (*Notropis*), darters (*Etheostoma*), channel catfish (*Ictalurus punctatus*), flathead catfish (*Pylodictis olivaris*), centrarchid basses (*Micropterus*), and white bass (*Morone chrysops*). However, only the three most abundant taxa collected (shads, sunfishes, crappies) are included in this analysis.

RESULTS

Relation Between Outlet Depth and Larval Fish Entrainment

Seasonal paterns of larval shad entrainment were similar during epilimnial and hypolimnial releases (Table 1). Losses of shad and crappies peaked in May each year except 1981. During hypolimnial release, few shad were collected in the discharge after June. The loss of crappies declined markedly after May during epilimnial release, but continued to be significant through July during hypolimnial release. Entrainment of sunfishes was highest in June each year except 1980. Few sunfishes were entrained before late May because sunfishes spawn later than shad and crappies. The decline of entrained sunfishes after June was much more pronounced during hypolimnial than during epilimnial release.

To further delineate the effects of outlet depth on larval fish entrainment, we compared densities of larvae (no./m^3) in discharge samples with densities of larvae in the reservoir near the intake structure for most sampling dates. The densities of larvae in discharge samples, expressed as percentages of the densities above the dam (Tables 2 and 3), were significantly higher ($P < 0.01$)

Table 1. Monthly estimated loss (millions) of larval shads, crappies, and sunfishes and (in parentheses) millions of cubic meters of water discharged (in parentheses) from DeGray Lake, 1976-81. (Negative sign indicates none collected; plus sign indicates no sampling because of paucity of larvae in previous collecting period.)

Month and Taxon	Epilimnial release			Hypolimnial release		
	1976	1977	1978	1979	1980	1981
April						
Shad	2.1	10.4	22.1	10.3	.3	2.9
Crappie	3.8	8.4	4.3	33.7	4.1	9.2
	(34.8)	(81.6)	(38.1)	(230.8)	(24.1)	(47.4)
May						
Shad	32.0	25.8	108.3	338.7	1.3	10.4
Crappie	4.7	15.0	13.0	33.7	4.1	9.2
Sunfish	1.2	7.2	1.0	1.1	.5	3.3
	(32.6)	(27.1)	(66.4)	(192.2)	(89.0)	(43.2)
June						
Shad	16.6	1.1	.8	2.6	.4	18.3
Crappie	.4	1.8	1.6	6.3	2.0	22.4
Sunfish	10.4	24.8	8.0	2.7	2.0	12.2
	(60.3)	(27.2)	(32.4)	(73.0)	(39.8)	(194.0)
July						
Shad	2.7	.2	.1	+	+	.2
Crappie	-	-	.1	.1	.1	.1
Sunfish	6.0	17.2	6.0	.1	2.2	.7
	(40.0)	(18.0)	(28.0)	(13.0)	(35.0)	(68.2)
August						
Shad	.1	.2	1.0	+	+	+
Crappie	-	-	.1	+	+	+
Sunfish	2.5	3.0	.5	+	+	+
	(53.0)	(23.0)	(23.0)	(32.1)	(18.3)	(34.1)

Table 2. Larval fish entrained during daytime epilimnial release as percent of larval density (no./m^3) by taxon, in the reservoir near the dam.

Date of sampling	Taxon: Shad	Crappie	Sunfish	Total larvae
1976				
May 20	49.4	47.2	89.5	50.0
Jun 3	34.5	54.7	100.0	64.7
Jun 17	9.1	50.0	42.1	21.9
Jul 1	19.8	25.0	100.0	54.9
Jul 15	14.7	0	98.9	100.0
Jul 29	8.3	0	100.0	49.7
Aug 19	1.2	1/	1/	1.2
1977				
May 10	45.8	24.9	1/	44.6
May 26	23.3	100.0	98.8	61.6
Jun 9	21.8	53.2	100.0	100.0
Jun 23	53.8	2/	66.4	67.7
Jul 7	13.6	1/	61.1	60.7
Jul 21	77.4	1/	54.1	54.5
1978				
May 9	100.0	21.5	1/	74.5
May 30	67.5	88.0	81.0	65.4
Jun 8	9.1	100.0	15.9	16.5
Jun 22	13.6	47.9	100.0	89.1
Jul 6	5.5	40.0	77.5	68.1

1/ No larvae collected near the intake or the discharge.

2/ Larvae collected in the discharge but not near the intake.

Table 3. Larval fish entrained during daytime hypolimnial release, as percent of larval density (no./m^3) by taxon, in the reservoir near the dam.

Date of sampling	Taxon: Shad	Taxon: Crappie	Taxon: Sunfish	Total larvae
1979				
May 20	2.0	9.8	1/	2.2
May 30	3.3	24.9	10.1	5.6
Jun 13	2.9	32.5	30.7	8.6
Jun 28	0.6	6.3	32.1	3.2
Jul 11	2.1	71.4	51.6	22.9
Jul 25	3/	14.8	2/	0.9
1980				
May 6	3/	1.4	1/	0.5
May 15	0.1	1.8	3/	0.5
May 19	0.1	2.6	3/	0.5
May 27	2.3	17.8	3.3	5.5
Jun 3	0.8	18.6	2.3	3.8
Jun 10	1.2	76.9	18.8	16.3
Jun 17	3.0	100.0	37.3	17.6
Jun 24	2.4	2/	3/	3.2
Jul 2	0.7	2/	20.8	9.1
Jul 8	0.6	66.7	58.8	2.6
Jul 15	0.8	2/	100.0	8.4
Jul 22	3.1	2/	2/	7.3
Jul 29	3/	1/	2/	15.2
Aug 14	3/	1/	1/	3/
1981				
May 7	1.8	6.8	32.0	5.3
May 21	11.9	18.9	48.8	15.4
Jun 4	11.5	28.5	23.4	16.6
Jun 18	0.4	82.1	7.5	12.5
Jul 2	1.9	16.0	52.5	6.4
Jul 16	3/	2/	2/	2.0
Jul 30	3/	1/	2.8	0.4
Aug 31	3/	1/	1/	0

1/ No larvae collected in discharge or in vicinity of intake.

2/ Larvae collected in discharge but not in vicinity of intake.

3/ Larvae collected in vicinity of intake but not in discharge.

during epilimnial than during hypolimnial release. Percentages of larval shad entrained were especially high in May and June during epilimnial release. Crappies appeared to be more vulnerable to entrainment after June during hypolimnial release. Sunfish were vulnerable to entrainment from May through July during both discharge regimes.

Density of larvae discharged through the dam was compared with density of larvae in the reservoir for each sampling date. Regression analyses used to describe these relations for the principal taxa during the two discharge regimes (Table 4) indicated a positive relation between the densities of larvae above the dam and densities of larvae in water discharged during both types of releases. Correlation coefficients ($\underline{r}$) were highly significant ($\underline{P} < 0.01$) except for crappies and sunfish during hypolimnial releases -- indicating that a significant amount of the variation in the number of larvae in discharge samples is explained by densities of reservoir larval fish population near the intake structure.

During both type releases, larval entrainment was influenced by the volume of water discharged during the period in which larvae were vulnerable to entrainment. Positive correlations between monthly water discharge volumes and monthly estimates of entrainment during both release regimes were found for all taxa except sunfishes during epilimnial release. Significant correlations ($\underline{P} < 0.01$) were noted for shad during epilimnial and hypolimnial release ($\underline{r}$ = 0.52 and 0.84) and for crappies during hypolimnial release ($\underline{r}$ = 0.63). Total discharge and greatest monthly variation in water volume released from April to August were both highest during hypolimnial release (Table 1).

Relation Between Time of Day and Larval Fish Entrainment

Diel sampling was used to compare densities of larvae entrained during power generation in the morning and afternoon, and at night under both discharge regimes. A Wilcoxon two-sample non-parametric test (Sokal and Rohlf, 1969) was used to compare entrainment densities between these diel periods. The results of many comparisons were inconclusive (Table 5). The trend for shad was most definitive during epilimnial release when morning and daytime losses were compared with night entrainment densities. Entrainment densities were

Table 4. Linear regression equations designed to predict densities (no./m^3) of larval fish in reservoir discharge (Y) from densities of larvae above dam (X).

Release and taxon	Regression equation	Correlation coefficient (r)
Epilimnial		
Shad	$Y = -0.149 + 0.624\ X$	.91*
Crappie	$Y = 0.030 + 0.224\ X$	.81*
Sunfish	$Y = 0.221 + 0.446\ X$	.64*
Total larvae	$Y = 0.036 + 0.571\ X$	.87*
Hypolimnial		
Shad	$Y = 0.009 + 0.021\ X$	.82*
Crappie	$Y = 0.026 + 0.074\ X$	.52
Sunfish	$Y = 0.014 + 0.066\ X$	.52
Total larvae	$Y = 0.070 + 0.023\ X$	.63*

* Statistically significant ($\underline{P} < 0.01$).

Table 5. Summary of Wilcoxon two-sample tests comparing entrainment densities during morning (M), afternoon (A), day (D), and night (N). The number next to the capital letter refers to the number of tests at which entrainment densities were significantly higher (P < .05) during that period (nsd indicates that no significant differences were shown).

Taxon and test comparison	Discharge regime: Epilimnial	Discharge regime: Hypolimnial
Shad		
M vs A	1M, 2A, 1nsd	1N, 2nsd
M vs N	4N	1M, 2nsd
A vs N	3A, 1nsd	1A, 1N, 2nsd
D vs N	3N, 1nsd	2D
Crappies		
N vs A	4nsd	1M, 1A, 2nsd
M vs N	1N, 1nsd	1M, 1N, 2nsd
A vs N	1N, 1nsd	5A, 3nsd
D vs N	2nsd	2D, 1N, 1nsd
Sunfish		
M vs A	1nsd	1A, 1nsd
M vs N		2nsd
A vs N		2N, 1A, 1nsd
D vs N		1N, 1nsd
Total		
M vs A	1M, 1nsd	1M, 2A, 1nsd
M vs N	1M, 1nsd	4N
A vs N	1A, 1nsd	2N, 1A, 4nsd
D vs N	2D	2N, 2nsd

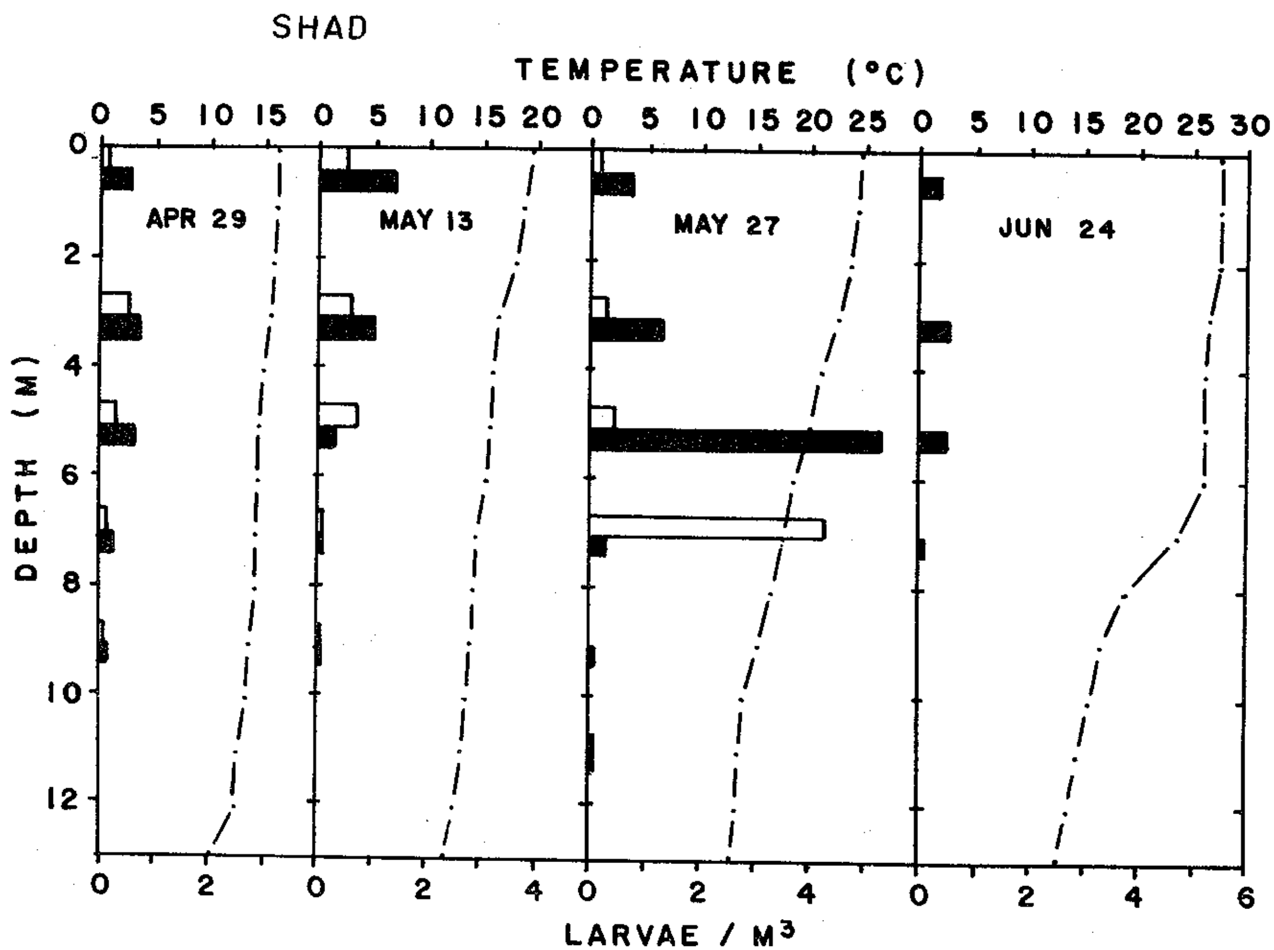

Fig. 2. Diel vertical distribution of larval shad in April, May and June 1982. Open bars represent daytime densities and solid bars nighttime densities. Broken lines are temperature profiles (scale at top).

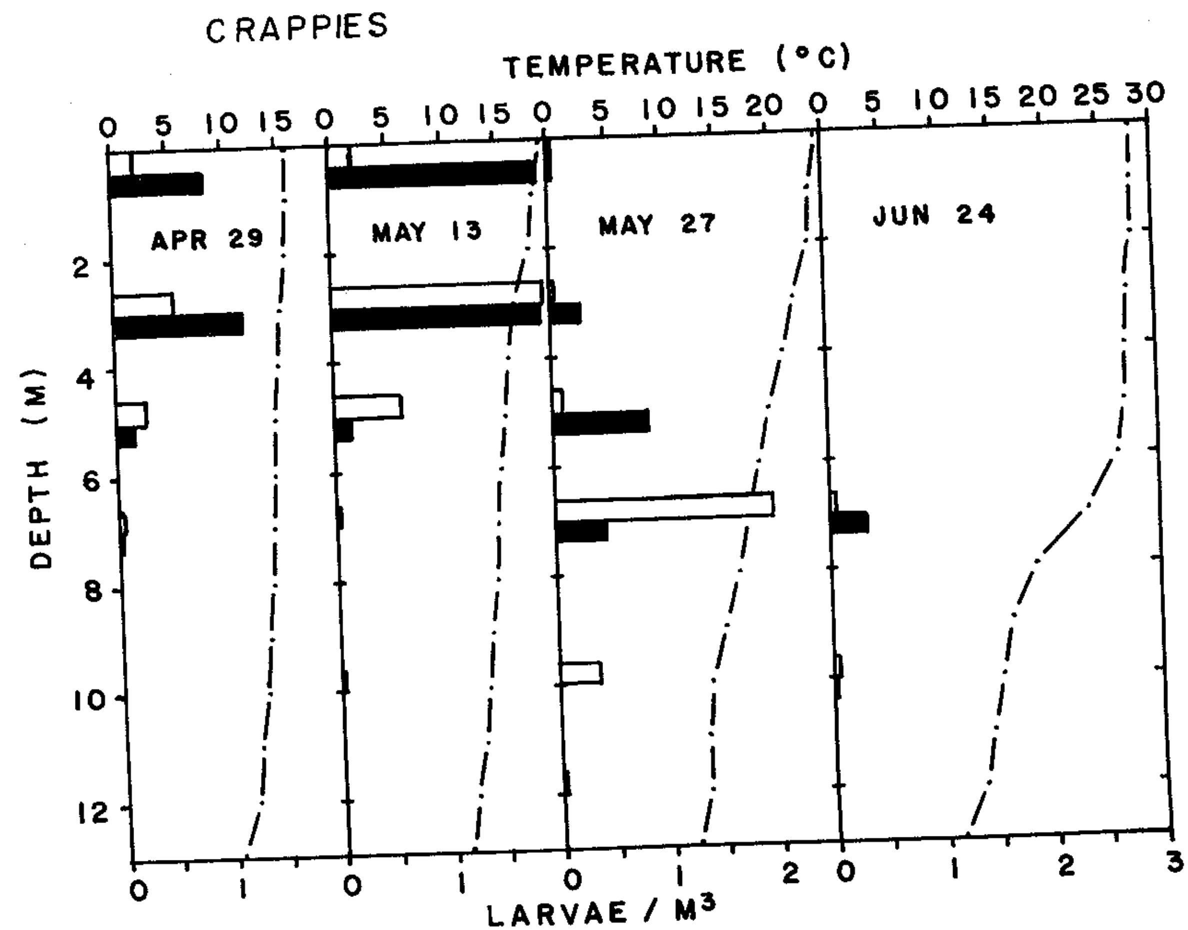

Fig. 3. Diel vertical distribution of larval crappies in April, May and June 1982. Open bars represent daytime densities and solid bars nighttime densities. Broken lines are temperature profiles (scale at top).

significantly higher ($\underline{P} < 0.01$) at night in seven of eight comparisons. Using the same time period comparisons during hypolimnial release, we found densities to be significantly higher during the day in three of five comparisons. Results were generally inconsistent for crappies. In five of eight tests comparing afternoon and night power generation, densities of crappies entrained were significantly higher ($\underline{P} < 0.01$) in the afternoon during hypolimnial release. No definitive trends were indicated for sunfish during either release regime, because most diel tests were conducted in early May, before sunfish fry dispersal. Entrainment densities of all taxa combined were higher during the day during epilimnial release in 4 of 6 comparisons and higher at night in 8 of 15 comparisons during hypolimnial release.

Vertical Distribution of Larvae Vulnerable to Entrainment

Daytime sampling in late April 1981 and 1982 indicated that most shad were found between the surface and a depth of 7 m. Daytime densities in mid-May were highest at 7 m in 1981 and at 5 m in 1982. Samples in late May and June indicated maximum daytime densities at 7 m in all three years. During night sampling, shad were concentrated mainly between the surface and 5 m from April through June (Fig. 2).

In 1981, daytime densities of crappies were highest at 7 m in April and May and at 10 m in June. In 1982, however, densities were highest at 3 m in April and mid-May and at 7 m in late May, and at 7 m in June (Fig. 3).

Sunfish were not collected until May in both years. Daytime densities were high at the surface and at 3 m in May 1981 and 1982. Sunfish were concentrated at 7 and 10 m in June 1980 and at 7 m in June 1982. Nighttime densities were high at the surface and at 3 m during late May and June (Fig. 4).

Impact of Entrainment on Reservoir Larval Fish Populations

Estimated annual loss of larval fishes ranged from 14.7 million in 1980 to 400.9 million in 1979; both extremes occurred during hypolimnial release (Table 6). Losses were greatest in May 1979 when the estimated number of

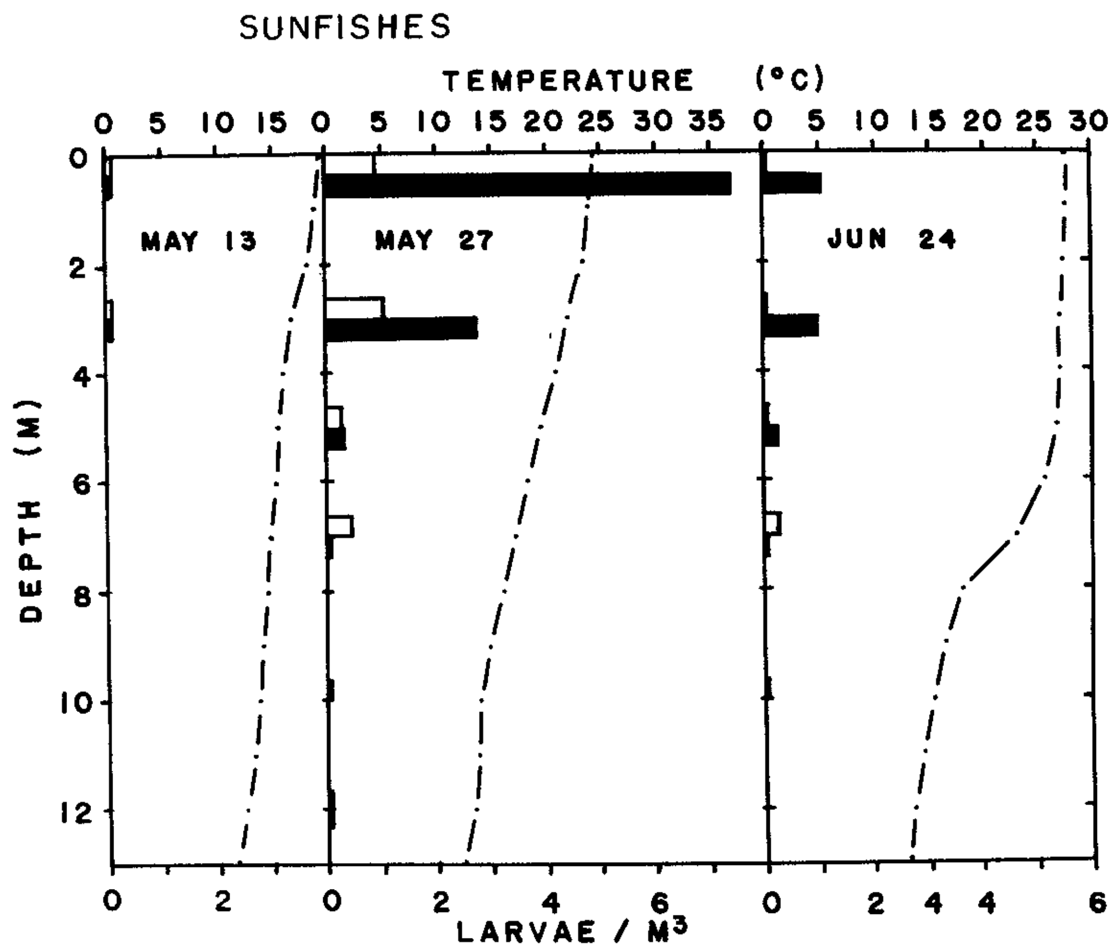

Fig. 4. Diel vertical distribution of larval sunfishes in May and June 1982. Open bars represent daytime densities and solid bars nighttime densities. Broken lines temperature profiles (scale at top).

Table 6. Estimated loss in numbers (millions) and (in parentheses) weight (kilograms) of larval fish from DeGray Lake, 1976-81.

	Release and year					
	Epilimnial			Hypolimnial		
Taxon	1976	1977	1978	1979	1980	1981
Shad	53.5 (593)	37.8 (354)	132.2 (1325)	351.6 (585)	2.1 (6)	31.8 (76)
Crappie	8.8 (57)	25.2 (261)	19.0 (201)	42.5 (665)	7.5 (335)	35.6 (122)
Sunfish	20.1 (58)	52.2 (373)	15.0 (94)	3.8 (20)	4.2 (5)	16.1 (27)
Others	1.6 (13)	3.7 (18)	2.0 (24)	3.0 (32)	1.0 (4)	1.0 (1)
Total	84.1 (721)	119.0 (1005)	168.1 (1644)	401.0 (1302)	14.8 (351)	84.1 (226)

shad that passed through the dam reached about 339 million (Table 1). Fewer sunfish but more shad and crappie were discharged during hypolimnial than during epilimnial release. Estimated annual loss of larval fish biomass ranged from 226 kg in 1981 to 1,644 kg in 1978. Although 35% more larvae were entrained during the 3 years of hypolimnial release, estimated biomass loss was much higher during epilimnial release (3.370 kg compared with 1,878 kg). During hypolimnial discharge in early May 1979, when entrainment was highest, the mean weight per fish (mostly shad) was extremely low.

Although these annual losses appear to be relatively high in some years, the proportion of the total reservoir population entrained was low during both release regimes. Proportions of the standing crop of larval fish that were lost through the discharge (May through July) were estimated by comparing weekly estimates of entrainment with corresponding reservoir population estimates. During epilimnial release, weekly percentages of the reservoir population of each taxon entrained ranged from 0.01 to 8.6% for shad, 0 to 3.3% for crappies and 0 to 2.6% for sunfishes. Mean entrainment percentages were highest for shad (1.2%) and crappies (1.1%) in 1978 and for sunfishes (1.0%) in 1976.

During hypolimnial discharge, weekly percentages of the population entrained were much lower for shad (0-1.1%) but higher for crappies, ranging to 10.6%. Weekly entrainment percentages for sunfishes were similar during both release regimes; the greatest weekly loss (4.5%) and the highest mean loss (1.6%) were in 1981. The mean entrainment percentage for crappies (2.5%) was highest in 1979 (Table 7).

DISCUSSION

Many factors may influence entrainment of larval fishes, including vertical distribution of taxa in the vicinity of the intake structure, the depth at which water is released, differences in population density, and the volume of water discharged. Studies of the vertical distribution of pelagic clupeid and centrarchid larvae have shown that most are in the upper 5 m of the water column (Netsch et al., 1971; Storck et al., 1978; Lewis and Siler 1980). Because the outlet for hypolimnetic release extended from depths of 12 to 18.4 m, we expected at the outset of this study that fewer larvae would be

Table 7. Means of weekly percentages of reservoir standing crops of larval fish discharged, May through July, DeGray Lake, 1976-81.

Discharge and year	Taxon: Shad	Crappie	Sunfish	Total
Epilimnial				
1976	0.7	0.7	1.0[1/]	1.0[1/]
1977	0.9	0.4	0.3	0.4
1978	1.2	1.1	0.4	1.0
Hypolimnial				
1979	0.1	2.5	0.3	0.2
1980	0.03	0.6	0.2	0.1
1981	0.2	1.7	1.6	0.5

[1/] Does not include July.

entrained during hypolimnial release.

We believe that diel changes in vertical distribution were related to differences in larval entrainment. However, these diel differences in entrainment were not always statistically significant. Daytime vertical distribution studies at DeGray Lake showed that larval shad were mainly in the upper 7 m of the water column from April through June. Therefore, they would appear to be more vulnerable to epilimnial release at that time. In May 1979, however, when reservoir populations were the highest noted, large numbers of shad were discharged during hypolimnial release (Table 1). Before stratification, and when populations in the reservoir were extremely high (as in May 1979), densities of shad possibly increased at depths greater than 7 m during daytime. Shad were more vulnerable to epilimnial release and less vulnerable to hypolimnial release at night (Table 5) since they have been found in our study and by other investigators (e.g. Netsch et al. 1971) to migrate upward at night. Crappies at DeGray Lake were relatively more abundant than the other taxa during daytime at depths of 7 m and greater. This distribution would make them more vulnerable to hypolimnial discharge during daytime power generation. We demonstrated some upward migration at night which would change the degree of vulnerability to entrainment. Sunfishes were concentrated primarily between the surface and 3 m during nighttime sampling, which would make them more vulnerable to night generation during epilimnial release. During daytime, sunfishes were found in both the epilimnion and metalimnion from late May through June and therefore were vulnerable to both releases. Higher reservoir populations of sunfishes during the three years of epilimnial release (Multi-Outlet Reservoir Studies, unpublished data) partly explain significantly higher densities of sunfish entrained during epilimnial discharge. Since most of our diel sampling was conducted in May, we were unable to make many seasonal comparisons.

We conclude that differences in diel entrainment vary with the season, and that thermal stratification and other abiotic factors, such as water transparency, influence the vertical distribution of larval fish, alter their vulnerability during either discharge regime. Daytime concentrations of shad and crappies in May were deeper in 1981 than in 1982. Secchi disc transparencies in May were 2.0-2.5 m greater in 1981 than in 1982.

Using linear regression models, we described a direct relation between larval fish abundance above the dam and larval fish entrainment. Similar models could be developed by fishery managers to estimate loss of these taxa through reservoir discharge if they knew the abundance levels of larval fish in the reservoir. The fishery manager could also sample larval fish in the discharge, using little sampling effort, and predict reservoir larval fish populations in the vicinity of the intake. The regressions indicate that a large portion of the variation in entrainment was due to the abundance of larval fish in the vicinity of the intake structure. These data do not indicate any response to the depth of the outlet, but do indicate passive entrainment during both release regimes. The positive correlation of larval entrainment with water volume discharged, at both release depths, also indicates a passive entrainment of larvae.

Apparently that change in the outlet depth affected the entrainment of shad more than that of the other taxa. Most of the entrainment percentages were over 10 times greater during epilimnial than during hypolimnial release. Even though extremely high numbers of shad were entrained in 1979 during hypolimnial release, significantly more shad would have been lost during an epilimnial release in that period. We estimated about 12 million shad were entrained during the last week of May 1979. However, using the predictive regression equation, we estimated that 218 million shad would have been entrained in this same period during an epilimnial release.

We did not measure the total impact of entrainment because we did not begin monitoring the reservoir larval fish population in most years until mid-May. However, our reservoir sampling did include most of the period in which larval fish were vulnerable to entrainment. Jensen et al. (1982), in determining the impact of 15 power plants on the fish stocks of Lake Michigan, found that although large numbers of alewives (Alosa pseudoharengus), rainbow smelt (Osmerus mordax), and yellow perch (Perca flavescens) were killed annually by entrainment or impingement, the proportions of the population affected were relatively low. However, these investigators stated that the loss of fish biomass was not negligible and that entrainment and impingement impacts need to be considered in the design of new intake facilities. We also noted little impact on the total reservoir larval population during either discharge regime, even though rates of entrainment were greater during epilimnial release. The loss of prey production, especially during years of low populations and high reservoir discharges during epilimnial release, could be significant.

The slopes of the regression equations predicting larval fish loss from reservoir larval fish densities indicate a higher degree of entrainment during epilimnial release for all taxa. The significantly higher percentages of entrainment for shad, sunfishes, and total fish larvae during epilimnial release augment these relations. Thus, entrainment, especially of shad, would have been much greater from 1979 through 1981 if the release had been epilimnial. Since our data indicate passive larval fish loss by reservoir release, the seasonal vertical distribution of larvae vulnerable to release and the outlet depth must determine the percentage of larvae entrained. Some fishery managers may favor high levels of entrainment. Hudson and Lorenzen (1980) related the beneficial effects of entrainment of fish, plankton, insects, and nutrients on the tailwater or the next downstream reservoir. The presence of a multi-outlet release design and a knowledge of diel and seasonal patterns of vertical distribution of the different taxa of larvae allow for selectivity in regulating the degree of entrainment.

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PATTERNS OF SEDIMENTATION AT DEGRAY LAKE, ARKANSAS

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ABSTRACT

Seasonal, longitudinal, and vertical patterns in deposition, measured with sedimentation traps, were identified at DeGray Lake, a large U.S. Army Corps of Engineers impoundment. During winter and spring, high river loads associated with storm events caused pronounced longitudinal gradients in water quality and deposition. The headwater region acted as a nutrient sink during these periods, experiencing the greatest deposition rates. During the summer stratified months, river nutrient loads were minimal and material deposition decreased substantially. Marked longitudinal gradients in C-14 productivity in summer were apparently related to the release of nutrients stored in headwater sediments during spring. Deepening of the thermocline and the occurrence of algal blooms led to deposition increases in the headwater region in late summer.

In general, the deposition of material loads in the headwater region effectively reduced particulate loading to down-reservoir locations. The storage of these materials in the headwater sediments had implications for detrital dynamics and nutrient cycles along the lake's longitudinal axis.

INTRODUCTION

Reservoirs are dominated by river inputs, advective flow regimes, and regulated outflows (Carmack et al., 1979; Johnson and Merritt, 1979; Ford and Johnson, 1981). These hydrodynamic characteristics and the transport and sedimentation of influent materials provide a physical setting fostering the establishment of physical, chemical, and biological gradients from headwaters to dam (Baxter, 1977; Gloss et al., 1980; Kennedy et al., 1982; Thornton et al., 1982). While such water quality gradients have been documented in reservoirs, there is a scarcity of information regarding similar patterns in sedimentation (Hakanson, 1976; Pharo and Carmack, 1979). However, it is reasonable to believe that sedimentation is also a longitudinal process, directly and indirectly affected by river inflows. As river water enters a reservoir, changes in channel morphometry are accompanied by a reduction in horizontal transport (Ford and Johnson, 1981; Kennedy et al., 1983b), which would result in the deposition of material loads primarily in the upper reaches of a reservoir. The deposition of material to this region would, therefore, have a bearing on particulate nutrient incomes to down-reservoir locations.

The existence of longitudinal water quality gradients led Thornton et al. (1981) to postulate a conceptual model dividing reservoirs into three metabolically distinct zones on the basis of hydrodynamics, morphometry, and sedimentation; namely, the riverine, transition, and lacustrine zones. These zones, which extend from the headwaters to the dam respectively, often exhibit marked differences in turbidity, nutrient concentrations, algal productivity and the relative importance of allochthonous organic matter. The input of river-borne organic material subsidizes detrital food webs in the riverine zone and the metabolic state here may approach heterotrophy. As advective velocities diminish and particles settle, reservoir metabolism may shift to autochthonous production, similar to that of a lacustrine system. The displacement of these materials to the benthos has not been studied in reservoirs, although this kind of information has implications for detrital dynamics (Wetzel et al., 1972).

Reservoir ecosystems must be conceived as a continuum of gradients which are driven by river inputs (Thornton et al., 1981; Kennedy et al. 1982b). The role of sediment transport and sedimentation act to both govern these gradients and displace organic carbon and other nutrients to the sediment for further decomposition and recycling. This study, sponsored by the U.S. Army Corps of Engineers as a part of the Environmental and Water Quality Operational Studies (EWQOS), was conducted to examine the influence of river inputs on seasonal, longitudinal, and vertical patterns in sedimentation in DeGray Lake, Arkansas. Research was directed toward the hypothesis that longitudinal gradients in sedimentation occur from the headwater to the dam region, and that the importance of allochthonous inputs to the sediment are greater in the upper reaches of the reservoir, diminishing toward the dam region.

STUDY SITE DESCRIPTION

DeGray Lake, located on the Caddo River in south-central Arkansas, was impounded in 1969 as a multipurpose reservoir for hydropower, flood control, and recreational use (Figure 1). The lake is long (32 km), dendritic (shoreline development ratio of 13), has mean and maximum depths of 9 and 60 m, respectively, and thermally stratifies in summer. The lake receives a majority of its material and water incomes from the Caddo River and has an average theoretical hydraulic residence time of 1.2 years.

The Caddo River drains a large (1162 km^2), predominantly forested watershed extending from the river's headwaters in the Ouachita Mountains to its confluence with the Ouachita River near Arkadelphia, Arkansas. Landuses in the watershed include timber production, undisturbed forest, and pasturage; urban and residential development is minimal. River discharges average 10 m^3/sec but short-term, storm-related increases occur in winter and spring. Discharges during these periods frequently exceed 200 m^3/sec.

DeGray Lake presently experiences only minor water quality-related problems. Nutrient and chlorophyll concentrations, although relatively low in recent years, exhibit marked longitudinal gradients from headwaters to dam (Thornton et al., 1982). Hypolimnetic anoxia is restricted to shallow, upstream areas and deep areas immediately upstream from the dam. These conditions contrast markedly with oxygen conditions immediately following impoundment, when inundated timber and other terrestrial organic material exerted a significant demand on oxygen supplies. During this period (ca. 1970-1974) the entire hypolimnion exhibited anoxia.

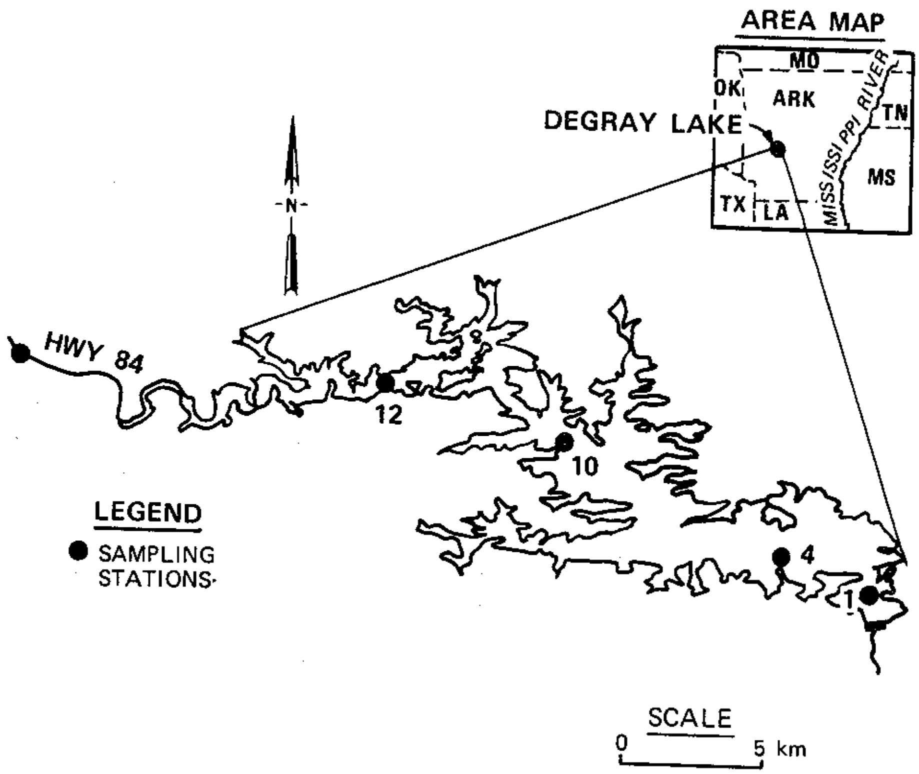

Figure 1. Location of sampling stations in DeGray Lake, Arkansas.

METHODS

Four sampling locations were chosen to characterize seasonal, longitudinal, and vertical gradients in water quality. These included three lake stations and one station on the Caddo River immediately upstream from the lake (Figure 1). Lake stations were located 4.9 km (Station 4), 12.4 km (Station 10) and 19.3 km (Station 12) upstream from the dam and were assumed to be characteristic of the near-dam, mid-lake and headwater region of the lake, respectively.

Sediment traps were similar to those described by Kimmel et al. (1977) and Rowe and Gardner (1979) (Figure 2) and were designed to reduce biases in the measurement of settling rates. Since measurement biases are potentially related to trap shape (Hargrave and Burns, 1979) and aspect ratio (Lau, 1979), a cylindrical trap with an aspect ratio of 3.0 (39-cm height/13-cm diameter) was chosen for this study. Main features of the trap devises included an aluminum carousel frame holding up to four cylindrical traps, a messenger-actuated closing mechanism, and a supporting buoy system. The closing mechanism consisted of a butterfly valve within each cylinder which was held open by the release mechanism during deployment. Upon retrieval, the release mechanism was activated by messenger, thereby causing the valve to close. With valves closed, the trap-containing carousels were raised to the surface for sampling.

Triplicate sediment traps were deployed at depths of 5 and 15 m at each sampling station. Triplicate traps were also suspended at 45 m (2 m above sediments) in the dam region (Station 4). Prior to deployment, traps were filled with lake water underlain with 1 liter of 5% salt solution and mercuric chloride (750 µg/ℓ) to reduce turbulent resuspension and decomposition, respectively. Deployment periods ranged from 20 to 48 days.

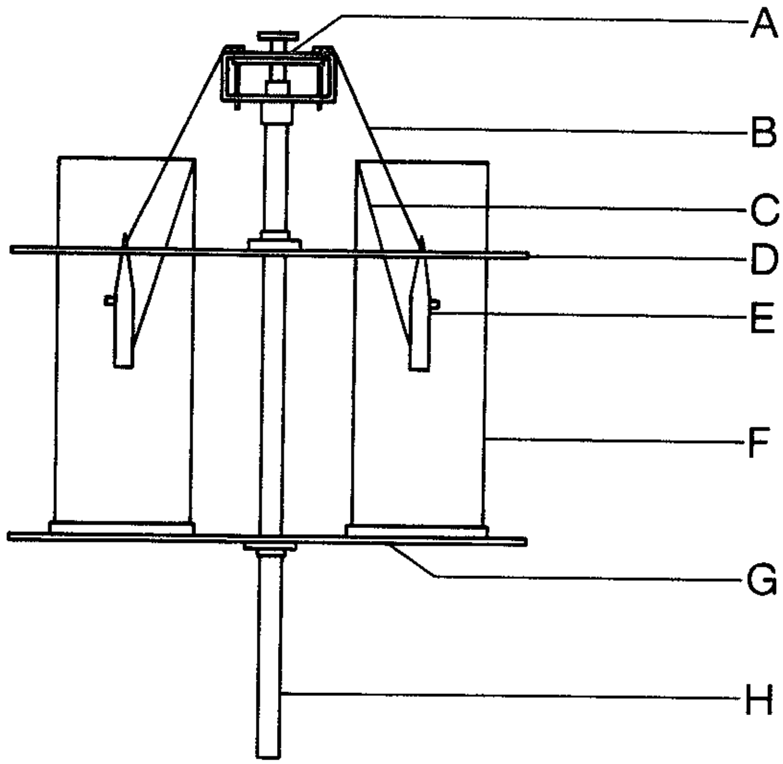

Figure 2. A sediment trap carousel containing cylindrical traps.
a) Ekman release mechanism, b and c) surgical tube springs.
d) butterfly valve, e) sedimentation trap cylinder,
f) aluminum, carousel-holding frame, g) suspension cable,
and h) supporting center shaft.

Upon retrieval, water and collected material were removed from each cylinder and stored in acid-washed plastic containers. Total sample volume, including water, sediment and material scraped from cylinder walls ranged from 3 to 4 liters. Each cylinder was then cleaned and redeployed.

Samples were kept dark and returned to the laboratory where they were refrigerated at 4°C until analyzed. Sample analysis was begun within three days of collection. Subsamples for analysis were obtained following thorough mixing. Particulate dry weight, organic carbon, Kjeldahl nitrogen, phosphorus, iron, manganese, and pigment concentrations were determined using standard analytical methods as described below. These concentrations were converted to sedimentation rates by the equation,

$$R = [C / A \cdot D]\ V$$

where:

R = sedimentation rate ($mg \cdot m^{-2} \cdot day^{-1}$)

C = concentration ($mg \cdot m^{-1}$)

A = area of sediment trap mouth (m^2)

D = deployment period (days)

V = total sample volume (ℓ)

Also calculated were average sedimentation rates for each region of the lake (i.e., headwater, mid-lake, and near-dam). These rates were calculated as the area-weighted average of rates observed at each deployment depth.

Total and dissolved organic carbon; total Kjeldahl nitrogen; total, soluble, and soluble reactive phosphorus; and turbidity were determined biweekly for the Caddo River, and for Station 12 (headwater), Station 10 (middle), and

Station 4 (dam region). Water samples were collected at each lake station at 2-m intervals from the surface to 15 m, then at 5-m intervals to the lake bottom. River discharges were recorded approximately 10 km above the lake at Highway 84 using a continuous-recording stage recorder. The determination of material loading is reported in Montgomery and Kennedy (1984). Carbon-14 productivity and chlorophyll a were measured at the surface and at one-meter intervals down to 7-m. Incubation periods were centered on solar noon and ranged from 2 to 3.5 hours (APHA, 1980). Depth-integrated hourly carbon uptake rates were converted to daily rates according to Stephens and Gillespie (1976).

Chemical analyses of the settling material and water samples were carried out using the standard methods (APHA, 1980). Particulate dry weight was determined on precombusted glass fiber filters. Total and dissolved organic carbon was determined by infrared analysis following an acid purge to remove inorganic carbon and persulfate wet chemical oxidation. Kjeldahl nitrogen was determined with an ammonium probe after an acid-mercuric sulfate digestion. Phosphorus concentrations were determined colorimetrically following acid-persulfate digestion. Total iron was determined by atomic absorption spectroscopy after nitric acid digestion. Algal pigments, which were extracted by maceration in 90% acetone, were calculated using a trichromatic equation.

RESULTS AND DISCUSSION

Caddo River discharge and suspended solids loading displayed marked seasonal variations which were associated with the occurrence of storm events (Figure 3). Spring freshets reached peaks of 104 $m^3 \cdot sec^{-1}$ and 99 $m^3 \cdot sec^{-1}$ in late April and mid-May, respectively. Peaks in loading corresponded with peak discharges during this period with suspended solid inputs reaching a maximum of 350 $kg \cdot 10^6 \cdot day^{-1}$ in May. Flow decreased substantially in summer, averaging only 5 $m^3 \cdot sec^{-1}$ from June through late-September. Suspended solids loading was also minimal during this period, contributing less than 5 percent of the total annual load. The greatest discharge event occurred during a single storm in early December, when suspended solids loading reached 1826 $kg \cdot 10^6 \cdot day^{-1}$. In general, storm events accounted for the majority (95 percent) of the annual suspended solids loading.

During high flow events, river water would have entered the reservoir as an overflow, underflow, or interflow, depending on water density differences between river and reservoir, and vertical density differences within the lake (Carmack et al. 1979; Ford and Johnson, 1981). Based on the assumption that temperature is the principal determinant of water density, estimates of the vertical placement of river inflows were determined by comparing river and lake water-column temperatures from the three sampling locations (Figure 4). It was assumed that river water entered the lake within strata for which temperature approximated (±1°C) that of the inflowing river water. From January until March, the lake was nearly isothermal (Figure 5) and river temperature was generally lower than that of the lake, suggesting the occurrence of weak underflows or complete mixing with the entire water column (Figure 4). During the onset of thermal stratification (April), river water potentially entered

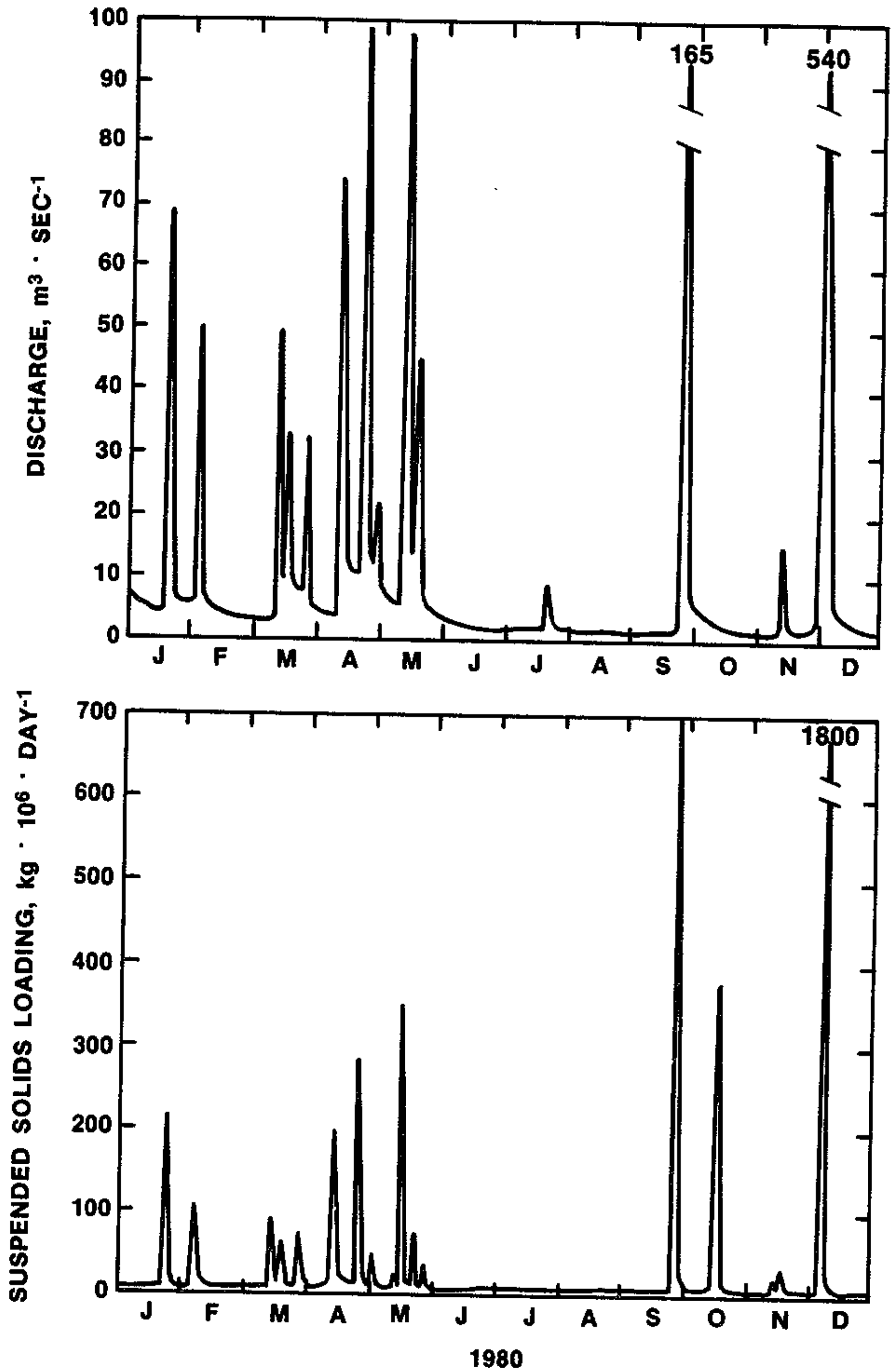

Figure 3. Daily changes in a) discharge ($m^3 \cdot sec^{-1}$), and b) suspended solids loading ($kgs \cdot 10^6 \cdot day^{-1}$) for the Caddo River in 1980.

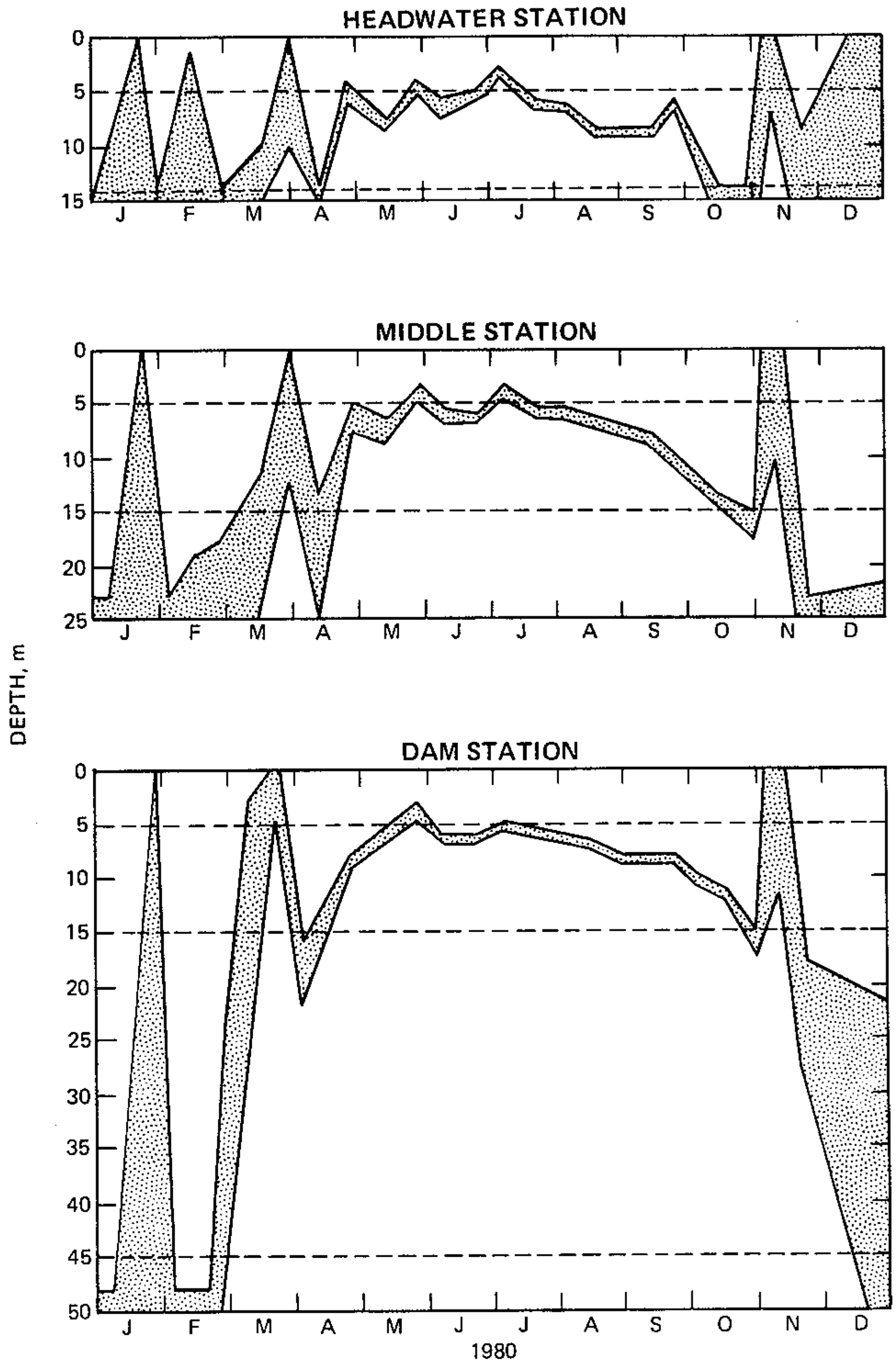

Figure 4. Approximate zones of the vertical placement of river inflows at the three stations (shaded areas) in 1980. Dashed lines indicate depths of trap deployment.

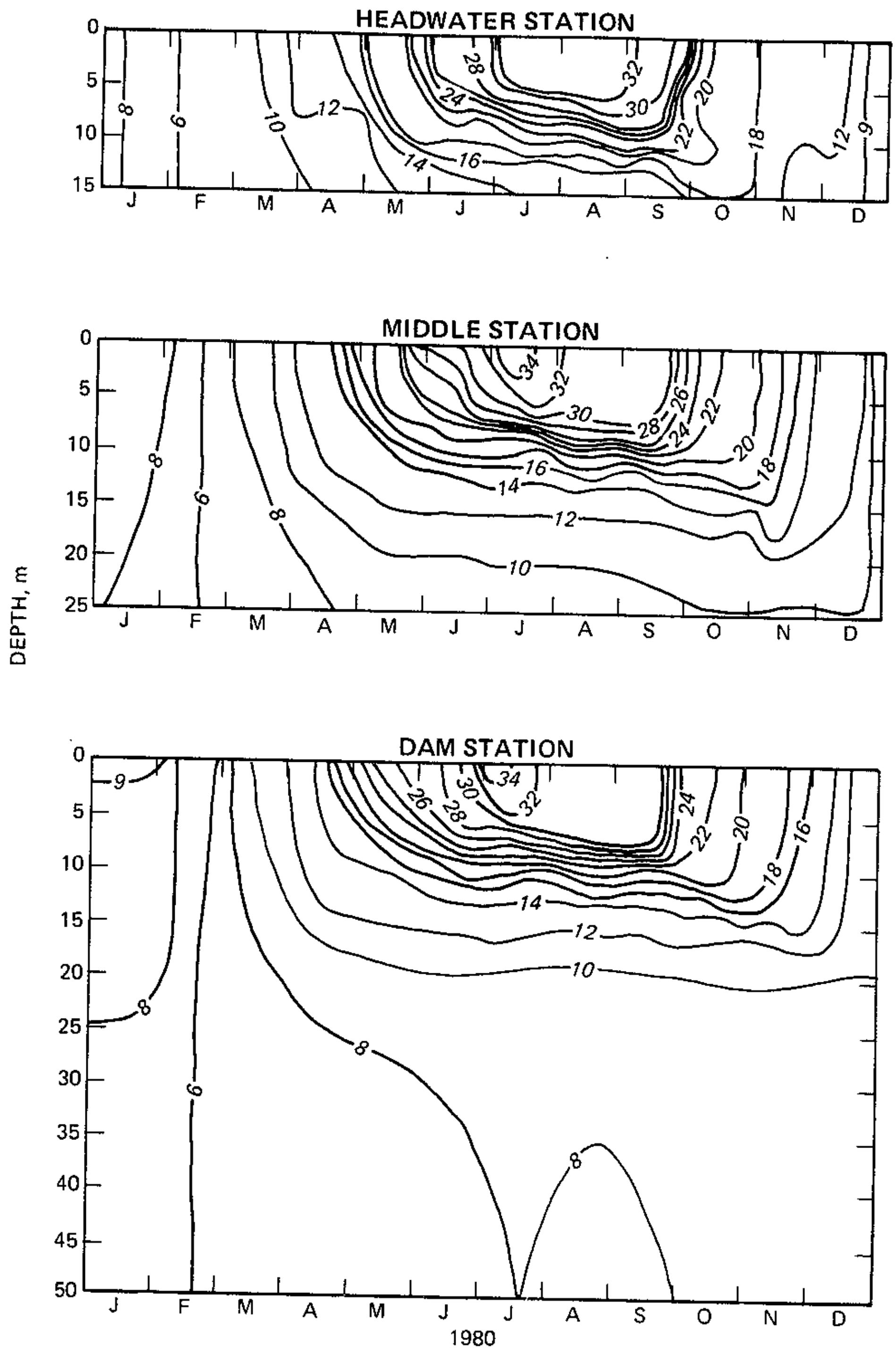

Figure 5. Seasonal changes in water temperature (°C) at the three sampling locations in DeGray Lake during 1980.

the headwater station as a near-bottom interflow. During periods of low river discharge from June until September, thermal stratification was evident (Figure 5) and river water would have entered the reservoir at the approximate depth of the thermocline. Convective mixing and progressive thermocline deepening would have caused the depth of interflow to increase from 5 m in early May to 10 m by late September. After autumnal circulation in the headwater station, river flow potentially mixed with the entire water column.

The suggested occurrence of density currents was confirmed by patterns in the distribution of turbidity in the upper reaches of the reservoir (Figure 6). Following the mid-April freshet (Figure 6a), when the reservoir exhibited weak thermal stratification, turbid river water was confined to depths near the thermocline, suggestive of an interflow. Turbidity exceeded 20 NTU in the headwater region, but decreased down-reservoir indicating that river inputs primarily affected the upper stretches of the reservoir. During a period of low flow (September), turbid river water was again confined as an interflow, but values were lower, suggesting that the Caddo River had minimal impact on downstream portions of the reservoir during this period.

Seasonal and longitudinal variations in reservoir turbidity followed seasonal changes in Caddo River loading (Figure 7). During high-flow events (January until May), turbidity was high at the headwater station and decreased toward the dam. During periods of low flow and during summer stratification, turbidity was greatly diminished and between-station differences were minimal. Two distinct peaks in turbidity in the headwater region coincided with late-September and December storm events. However, turbidity remained low at the middle and dam stations during this period.

Seasonal, longitudinal, and vertical patterns in seston deposition reflected changes in suspended solids loading and turbidity (Figure 8).

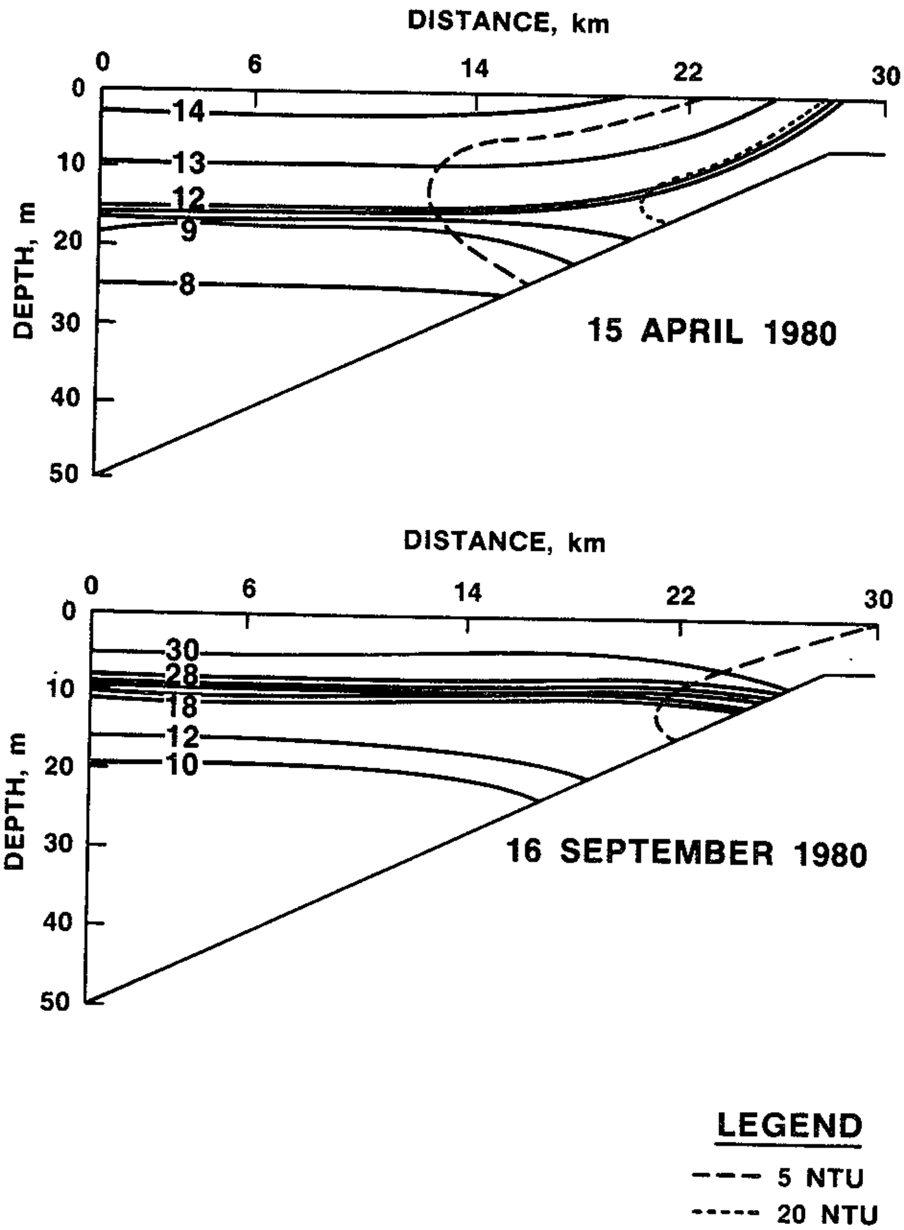

Figure 6. The longitudinal distribution of turbidity (NTU) in DeGray Lake a) following a mid-April freshet, and b) during low Caddo River flow (<5 $m^3 \cdot sec^{-1}$) in September.

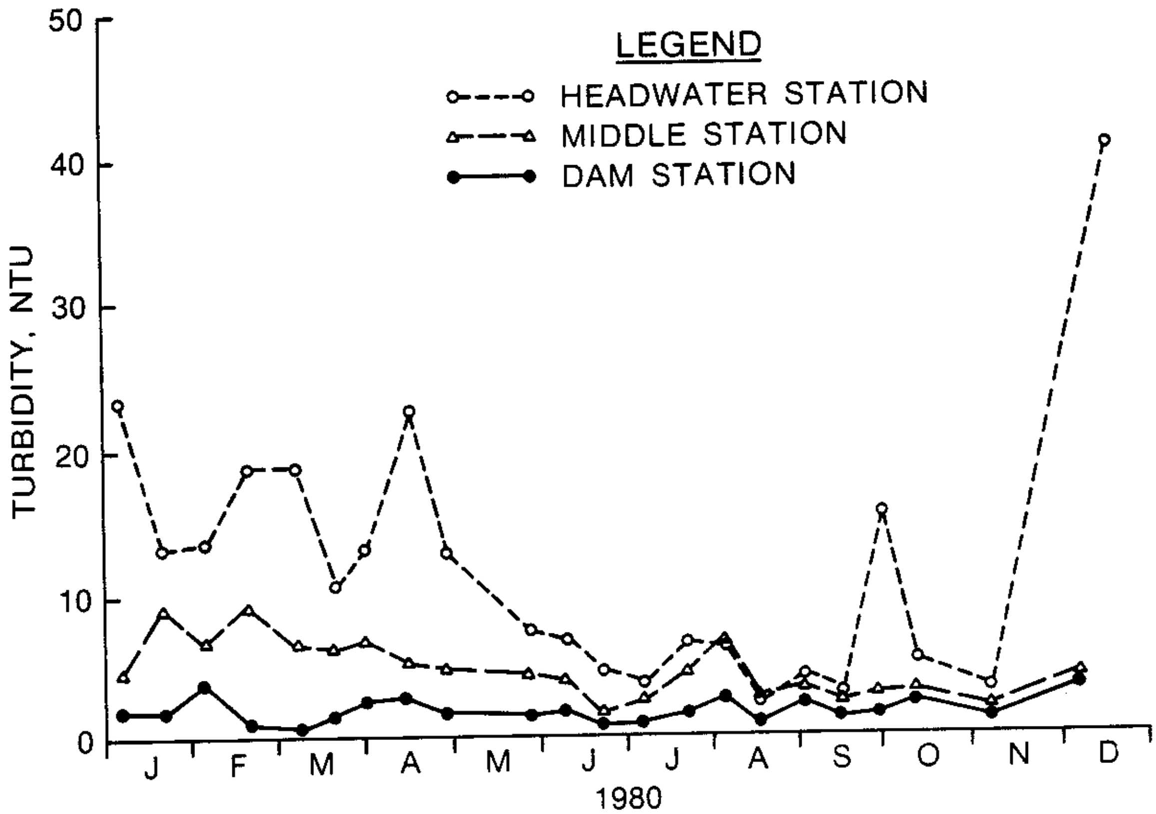

Figure 7. Seasonal changes in mean turbidity (NTU) at the headwater, middle, and dam stations in DeGray Lake in 1980.

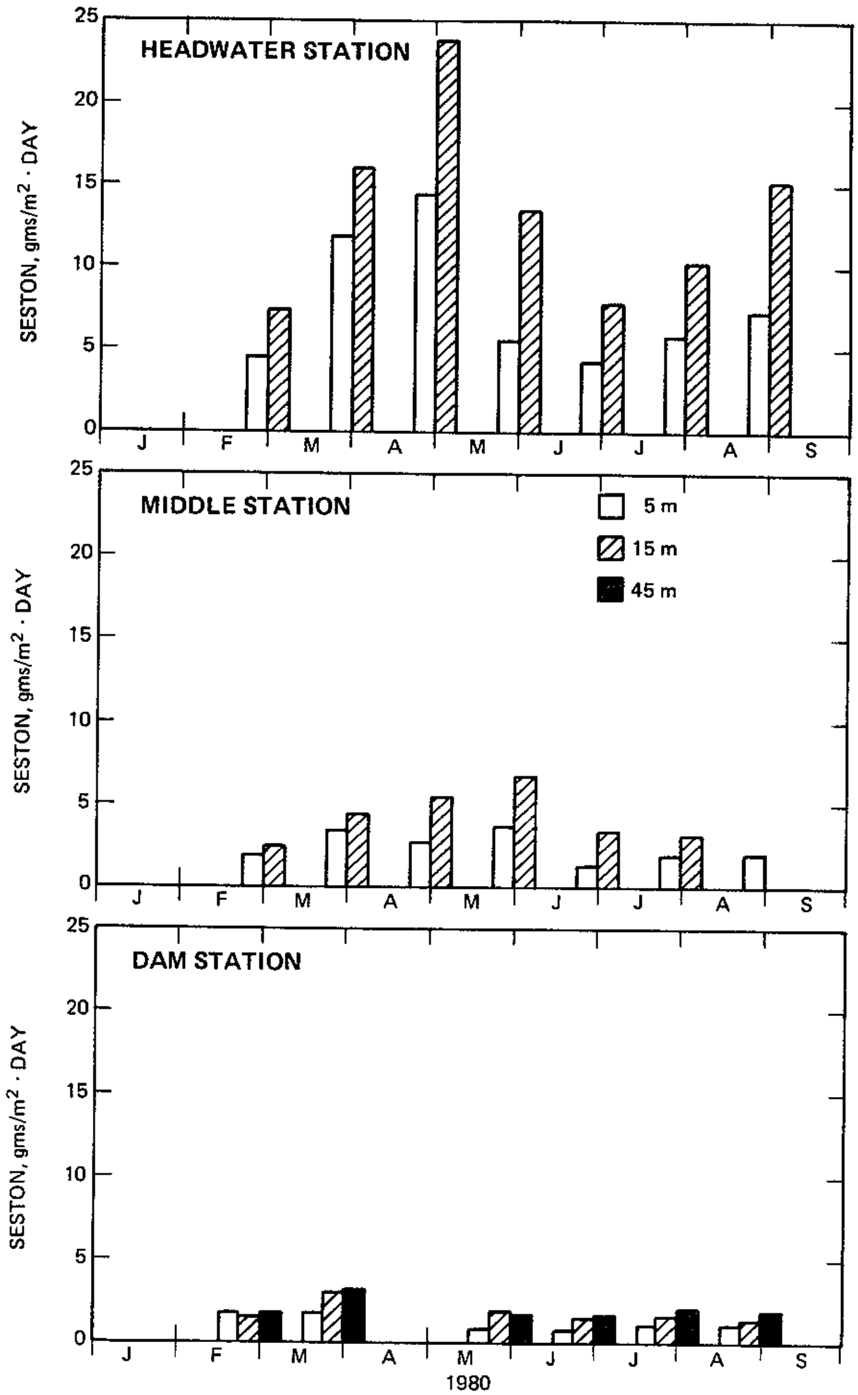

Figure 8. Seston deposition rates (gm • m^{-2} • day^{-1}) at the 5-m, 15-m, and 45-m depths at the headwater, middle, and dam stations.

In addition, between-station differences in deposition patterns indicated that the majority of the influent material settled in the upper reaches of the reservoir. At the headwater station, seston deposition steadily increased at both 5-and 15-m depths from February through early-June, in response to peaks in suspended solids loading. Highest deposition rates corresponded with three major freshets in April and May. From February through March, trap rates were similar at 5-and 15-m depths. Since isothermal conditions existed, river loads may have mixed with the entire water column at the headwater station, accounting for high deposition at the 5-m depth. During April and May freshets, after the onset of thermal stratification, trap rates were highest at the 15-m depth. This observation was consistent with the occurrence of interflows which would have introduced river-borne loads at depths below the 5-m trap. Suspended solids loading had less impact on seston deposition at the two downstream stations; rates here were markedly lower than those observed at the headwater station. However, differences between rates at the 5-and 15-m trap depths also reflected the occurrence of an interflow at the middle station during freshets in April and May.

During summer stratification, seston deposition rates decreased substantially at the headwater and middle stations, coincident with a marked reduction in suspended solids loading. Deposition rates again increased during August and September, particularly at the 15-m trap at the headwater station. This increase, however, was not accounted for by an increase in river loading. As will be discussed later, particle resuspension and deposition to the deeper areas may have caused this increase.

Depositional patterns for iron in the three regions of the reservoir (Figure 9) were similar to those for seston deposition. Pronounced longitudinal gradients in depositional rates coincided with periods of elevated river

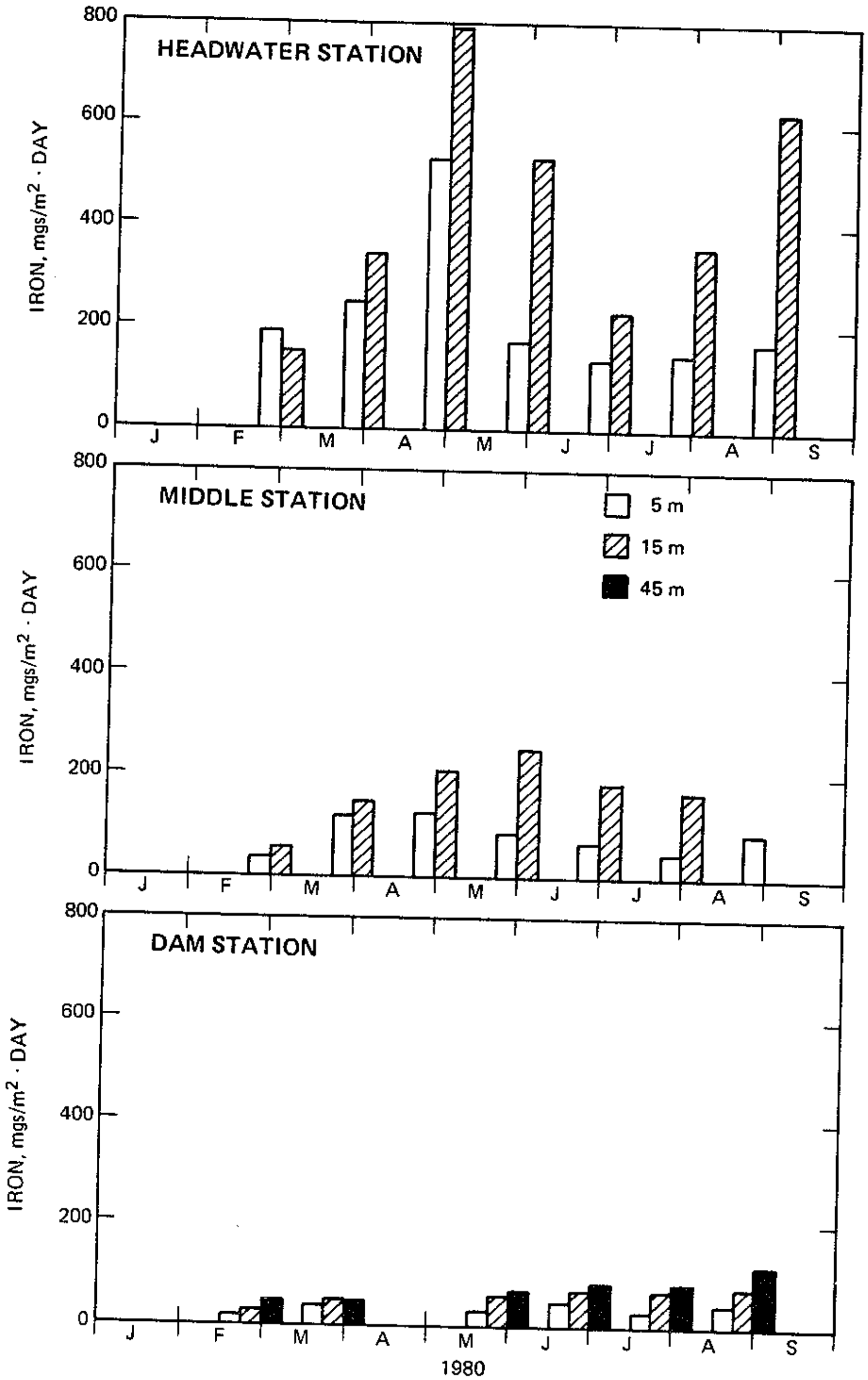

Figure 9. Iron deposition rates (mg · m^{-2} · day^{-1}) at the 5-m, 15-m, and 45-m depths at headwater, middle and dam stations.

discharge and rates were highest at the headwater station. Markedly lower rates at down-reservoir stations indicated that most of the river-borne iron settled in the upper reaches of the reservoir. Depth-related differences were apparent in the headwater and middle stations during April and May freshets. River water entering the reservoir at depths below the 5-m trap would account for increased deposition in the 15-m trap. Following a decline in iron deposition in July, depositional rates at the 15-m depth increased significantly at the headwater station in August and September. Deposition rates at the down-reservoir stations were lower and relatively constant. As discussed below, differences in deposition between the 5- and 15-m traps at the headwater station during late summer were related to processes affecting the distribution and abundance of iron in the water column.

Patterns of total organic carbon loading, C-14 productivity and chlorophyll a reflected seasonal differences in the importance of allochthonous and autochthonous sources of organic material in the reservoir (Figure 10). As was observed for suspended solids, storm events caused major peaks in total organic carbon loading from January until mid-May, then loading decreased in summer. Although C-14 productivity data were not taken during the spring freshets, chlorophyll a concentrations were generally low at each station during this period, suggesting relatively low rates of productivity. Chlorophyll a concentrations were highest during the summer stratified period, coincident with peaks in C-14 productivity. Longitudinal gradients in C-14 productivity during this period were also pronounced. The headwater station exhibited a productivity maximum of 600 $mg \cdot m^{-2} \cdot day^{-1}$ during early thermal stratification in June and a later peak in late July. Rates were lower down-reservoir and peaks were less distinct. Considering the seasonal patterns in both organic carbon loading and algal productivity, it is suggested that

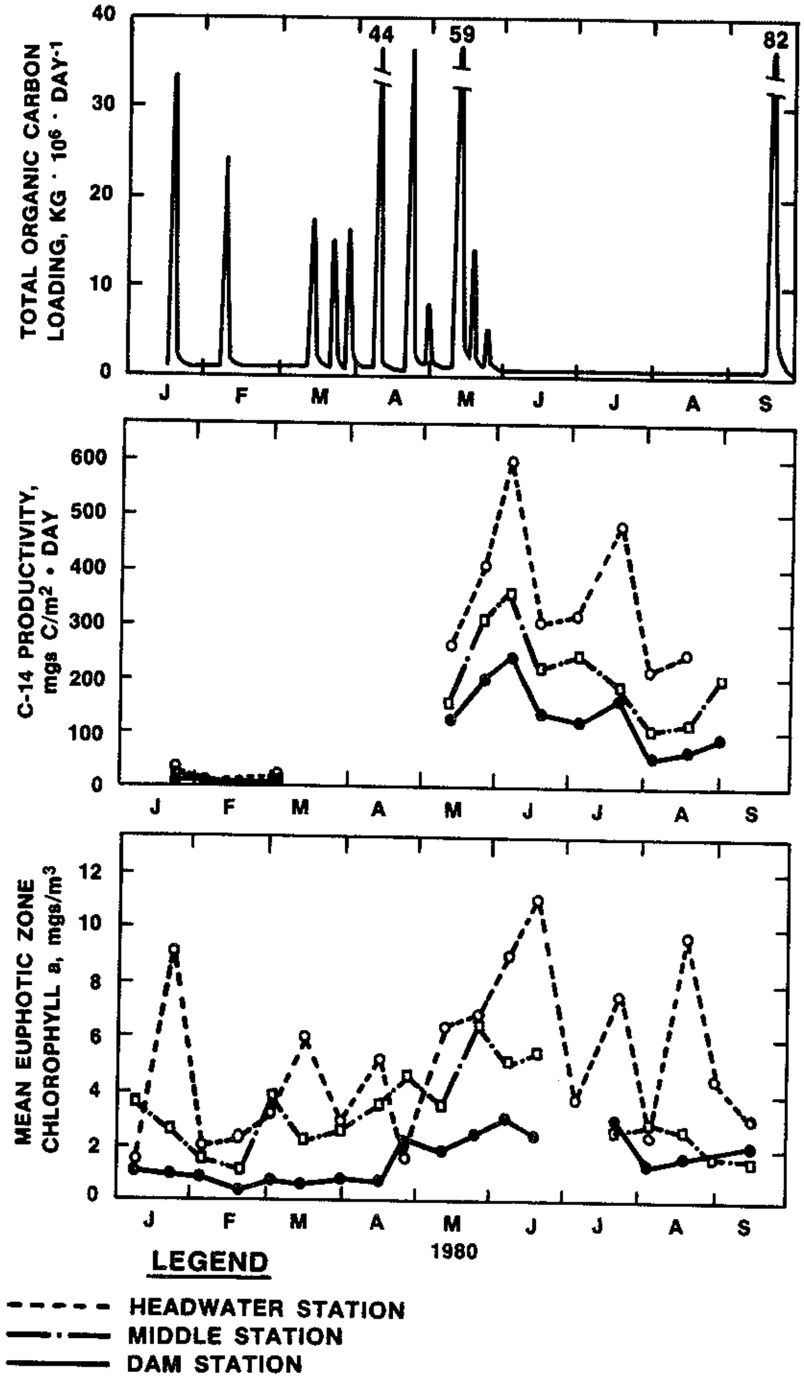

Figure 10. Seasonal changes in a) daily total organic carbon loading (kg C · 10^6 · day^{-1}) for the Caddo River, b) C-14 primary productivity (mg · C · m^{-2} · day^{-1}) at the three sampling stations in DeGray Lake, and c) mean eutrophic zone chlorophyll a concentrations (mgs · m^{-3}) at the three sampling locations in DeGray Lake.

allochthonous sources of organic carbon were more significant at the headwater station during spring freshets while autochthonous sources were more significant during the summer and fall phytoplankton blooms.

Seasonal and longitudinal patterns of organic carbon and chlorophyll a deposition generally reflected differences in the importance of organic material sources (Figures 11 and 12). Organic carbon deposition displayed spring maxima in the headwater region which corresponded to peaks in total organic carbon loading. Trap rates were lower down-reservoir indicating that the extent of deposition was limited to the upper reaches of the reservoir. Chlorophyll a trap rates were lower during this period and did not exhibit a spring maximum. During periods when autochthonous sources were of greatest importance, the headwater station exhibited late summer maxima in both chlorophyll a and organic carbon trap rates, which corresponded to July and August peaks in chlorophyll a and C-14 productivity. Chlorophyll a deposition increases during this period were greatest at the 15-m depth and reflected the vertical distribution of chlorophyll a in the water column, since distinct chlorophyll a peaks were apparent near the thermocline at the 8-m depth of the headwater station (Figure 13). However, organic carbon trap rates were similar at both depths indicating that a substantial amount of carbon was settling from the epilimnion. Lower deposition rates in the middle and dam region stations corresponded to decreased rates of productivity down-reservoir. However, the bimodal peaks in productivity were not reflected by sedimentation increases in down-reservoir stations.

Patterns of nitrogen and phosphorus deposition were generally similar to that of organic carbon (Figures 14 and 15). Considerable nitrogen and phosphorus sedimentation occurred at the headwater station during spring freshets. A second seasonal maximum occurred in late August and September. Since river

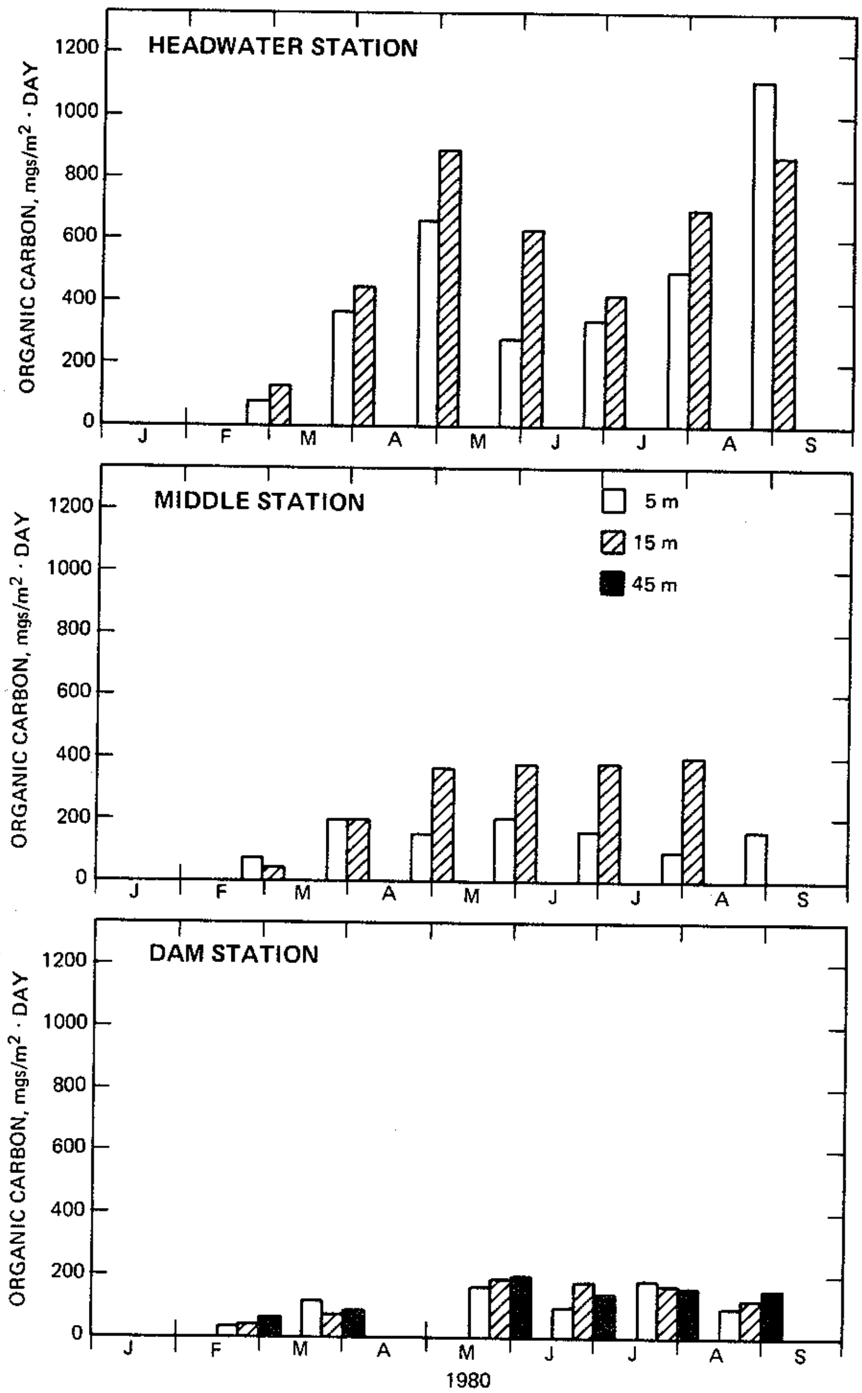

Figure 11. Organic carbon deposition rates (mg $\cdot$ m^{-2} $\cdot$ day^{-1}) at the 5-m, 15-m, and 45-m depths at the headwater, middle, and dam station.

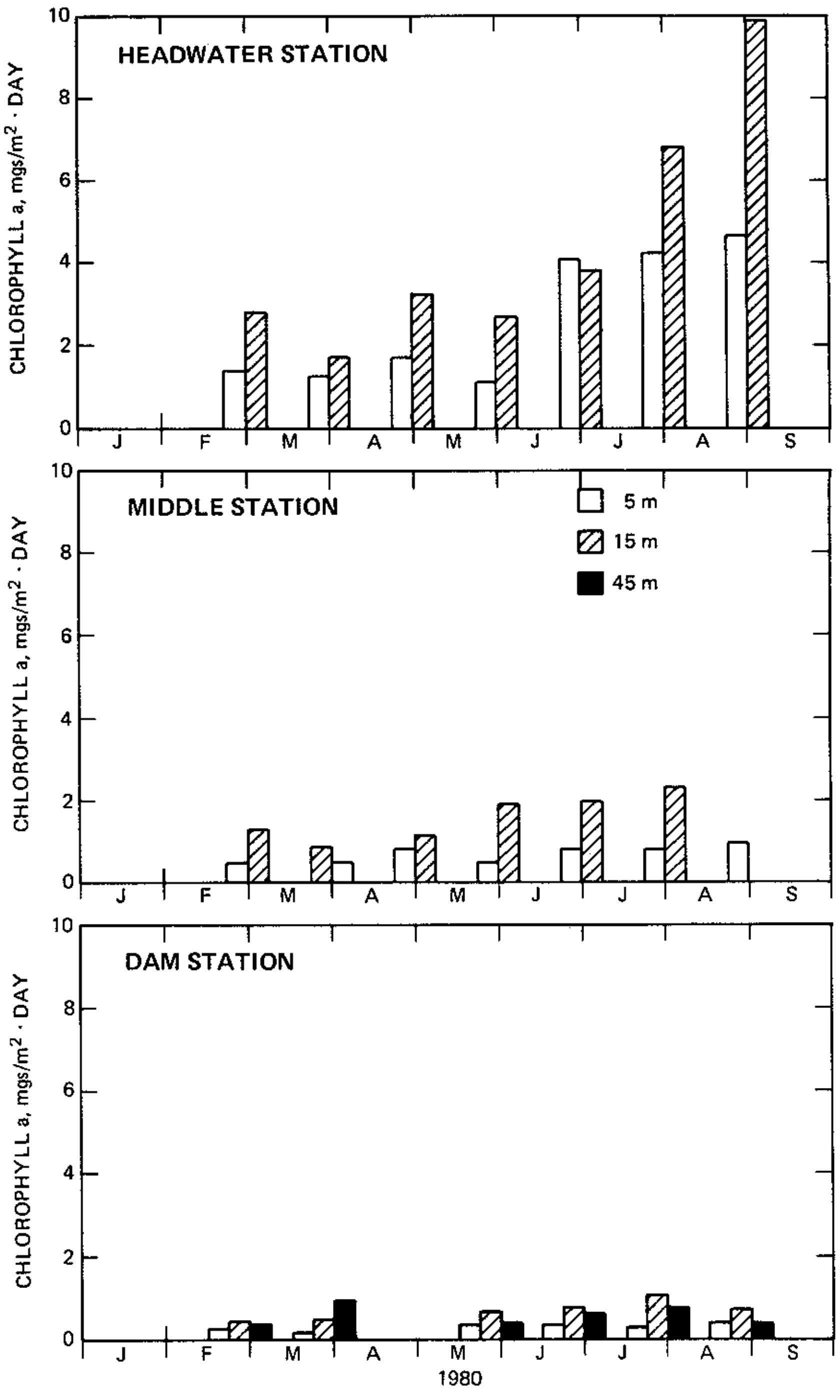

Figure 12. Chlorophyll a deposition rates (mg · m^{-2} · day^{-1}) at the 5-m, 15-m, and 45-m depths at the headwater, middle, and dam stations.

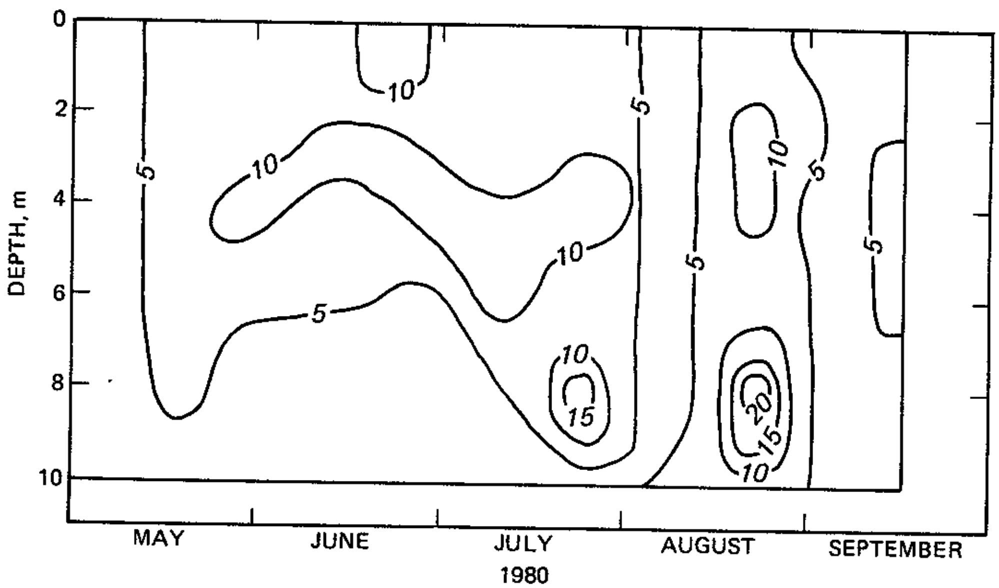

Figure 13. Seasonal and vertical changes in chlorophyll a (mg • m^{-3}) at the headwater station in DeGray Lake.

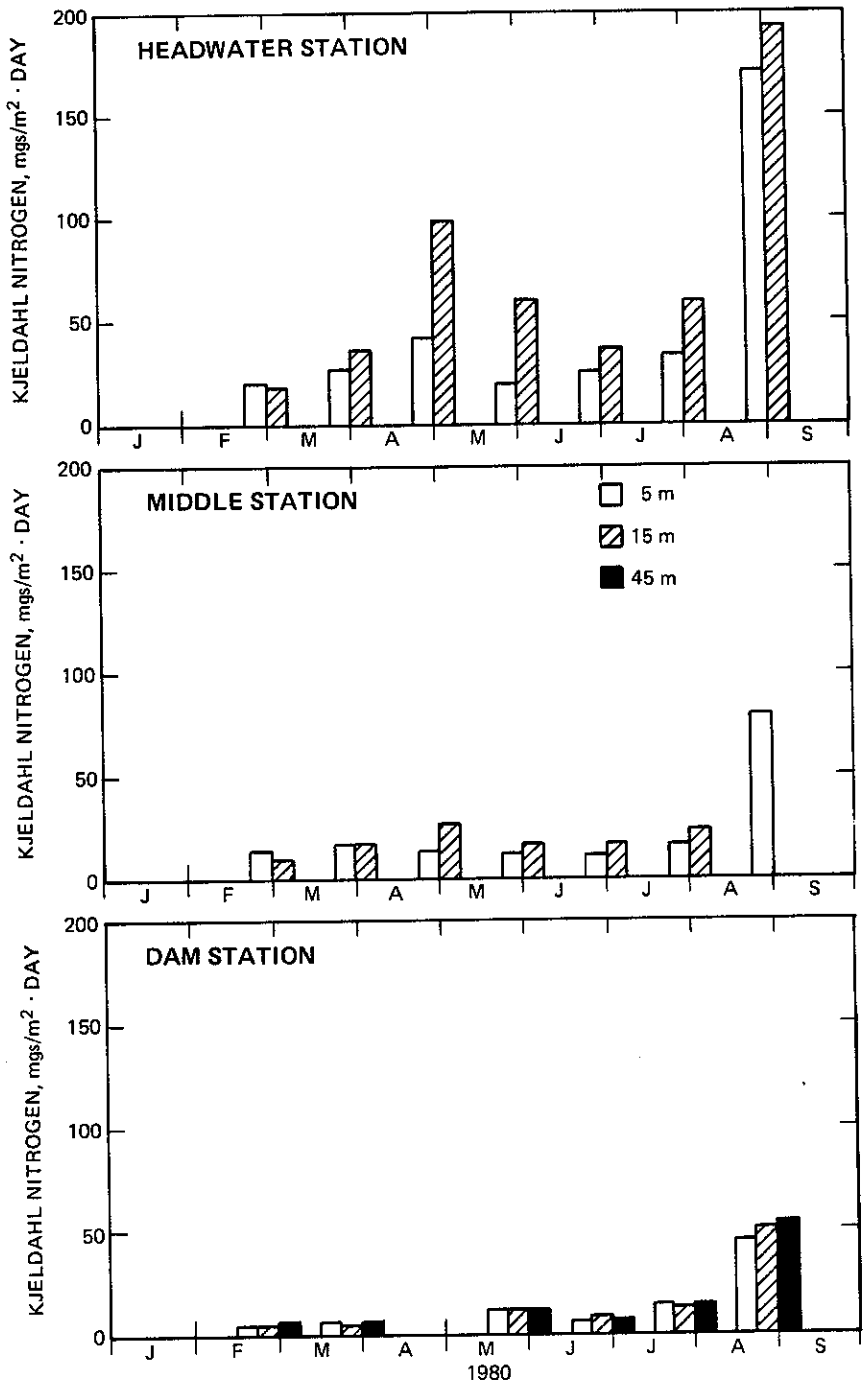

Figure 14. Kjeldahl nitrogen deposition rates (mg · m^{-2} · day^{-1}) at the 5-m, 15-m, and 45-m depths at the headwater, middle, and dam stations.

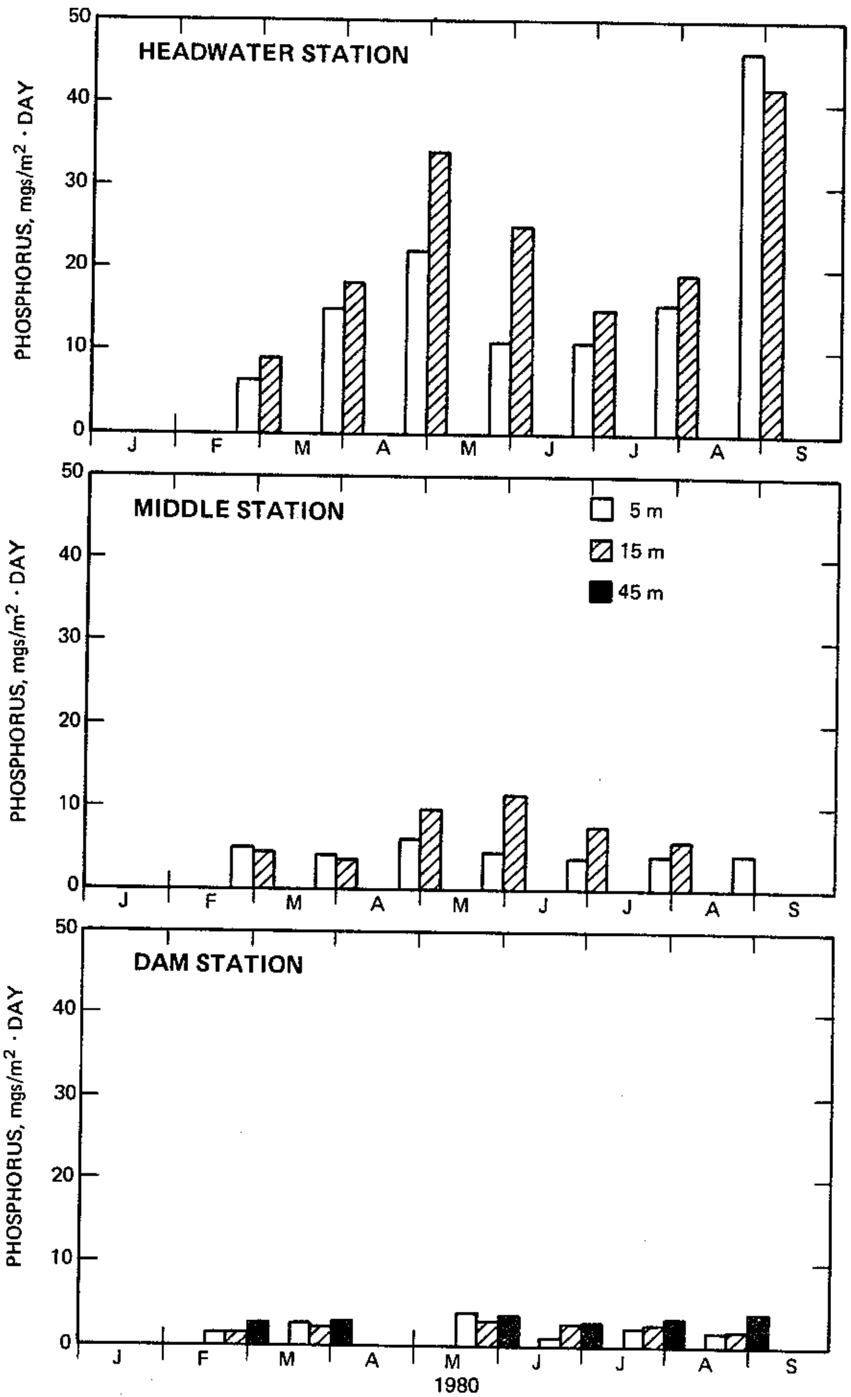

Figure 15. Phosphorus deposition rates (mg · m^{-2} · day^{-1}) at the 5-m, 15-m, and 45-m depths at the headwater, middle, and dam stations.

discharge was minimal during this period, sources for these nutrients must have come from 1) mixing processes associated with thermal destratification, and 2) settling phytoplankton. Seasonal deposition of phosphorus down-reservoir was lower; however, a noticeable peak in nitrogen sedimentation occurred at each station in August-September.

Relationships between the water quality of the Caddo River and DeGray Lake and the quality of settling material were more clearly shown by a comparison of mean annual particulate organic carbon:particulate phosphorus (C:P) and particulate Kjeldahl nitrogen:particulate phosphorus (N:P) ratios from the headwater, middle, and dam stations (Tables 1 and 2). The mean annual C:P ratio of the Caddo River water was comparable to mean annual ratios from the headwater station water column, settling material, and permanent sediment. The C:P ratios for settling material were low compared to the theoretical ratio of 40:1 for phytoplankton (Redfield et al., 1963), suggesting that much of the settling phosphorus was associated with inorganic materials such as clays or hydrous ferric oxides. Mean annual C:P ratios increased down-reservoir, suggesting that the relative contribution of carbon increased due to 1) maximal deposition of phosphorus influent primarily at the headwater station, and 2) increased importance of organic carbon production as the major source of carbon.

Particulate Kjeldahl nitrogen was not measured in the water column, but similar comparisons were evident between the Caddo River and headwater station with mean annual total N:P ratios. Higher total N:P ratios in the water columns of the middle and dam stations again suggested that river inputs primarily affected the headwater station. Mean particulate N:P ratios from the settling material and permanent sediment were low, suggesting that phosphorus was preferentially removed from the water column relative to nitrogen.

Table 1. A comparison of mean particulate organic carbon: particulate phosphorus ratios (gm=gm) from the Caddo River, reservoir water column, settling material, and permanent sediment.

	CADDO	HEADWATER	MIDDLE	DAM
WATER COLUMN	33:1	27:1	49:1	77:1
SETTLING MATERIAL		25:1	40:1	62:1
PERMANENT SEDIMENT*	23:1	17:1	41:1	74:1

* Values from Gunkel et al. (1983)

Table 2. A comparison of Kjeldahl nitrogen: phosphorus ratios (gm:gm) from the Caddo River, reservoir water column, settling material, and permanent sediment.

	CADDO	HEADWATER	MIDDLE	DAM
WATER COLUMN*	16:1	16:1	27:1	37:1
SETTLING MATERIAL**		3:1	5:1	8:1
PERMANENT SEDIMENT†	3:1	2:1	3:1	5:1

* Values represent total Kjeldahl nitrogen:total phosphorus ratios.

** Values represent particulate Kjeldahl nitrogen:particulate phosphorus ratios.

† Values from Gunkel et al. (1983).

In general, annual deposition of material was highest at the headwater station with pronounced longitudinal gradients toward the dam (Table 3). At the headwater station, higher deposition reflected greater sources of allochthonous and autochthonous material, and possible additional sources from resuspension. Down-reservoir, spring loading had less impact and C-14 productivity was lower, consistent with lower rates of sedimentation. The existence of longitudinal gradients in the deposition of organic material provided implications for detrital processing, to which patterns of hypolimnetic oxygen depletion in the three regions might be related (Figure 16). At the headwater station, the small-volume hypolimnion, rich in recently deposited organic material, was conducive to rapid deoxygenation by May. Larger hypolimnetic volumes and diminished deposition down-reservoir delayed anoxia until July at the middle station, and much later, November, at the dam station. Associated with anoxia at the headwater station were increases in soluble iron and phosphorus in the hypolimnion (Kennedy et al., 1983a), suggesting that the headwater sediment acted as a nutrient source in the summer.

Table 3. Annual sedimentation, normalized by area, from the headwater, middle, and dam stations (mgs $\cdot$ m^{-2} $\cdot$ study period^{-1})

	HEADWATER	MIDDLE	DAM
DRY WEIGHT	8550	2980	1520
CARBON	456	202	123
NITROGEN	45	22	15
PHOSPHORUS	18	5	2
IRON	269	103	50
CHLOROPHYLL a	2.93	0.92	0.23

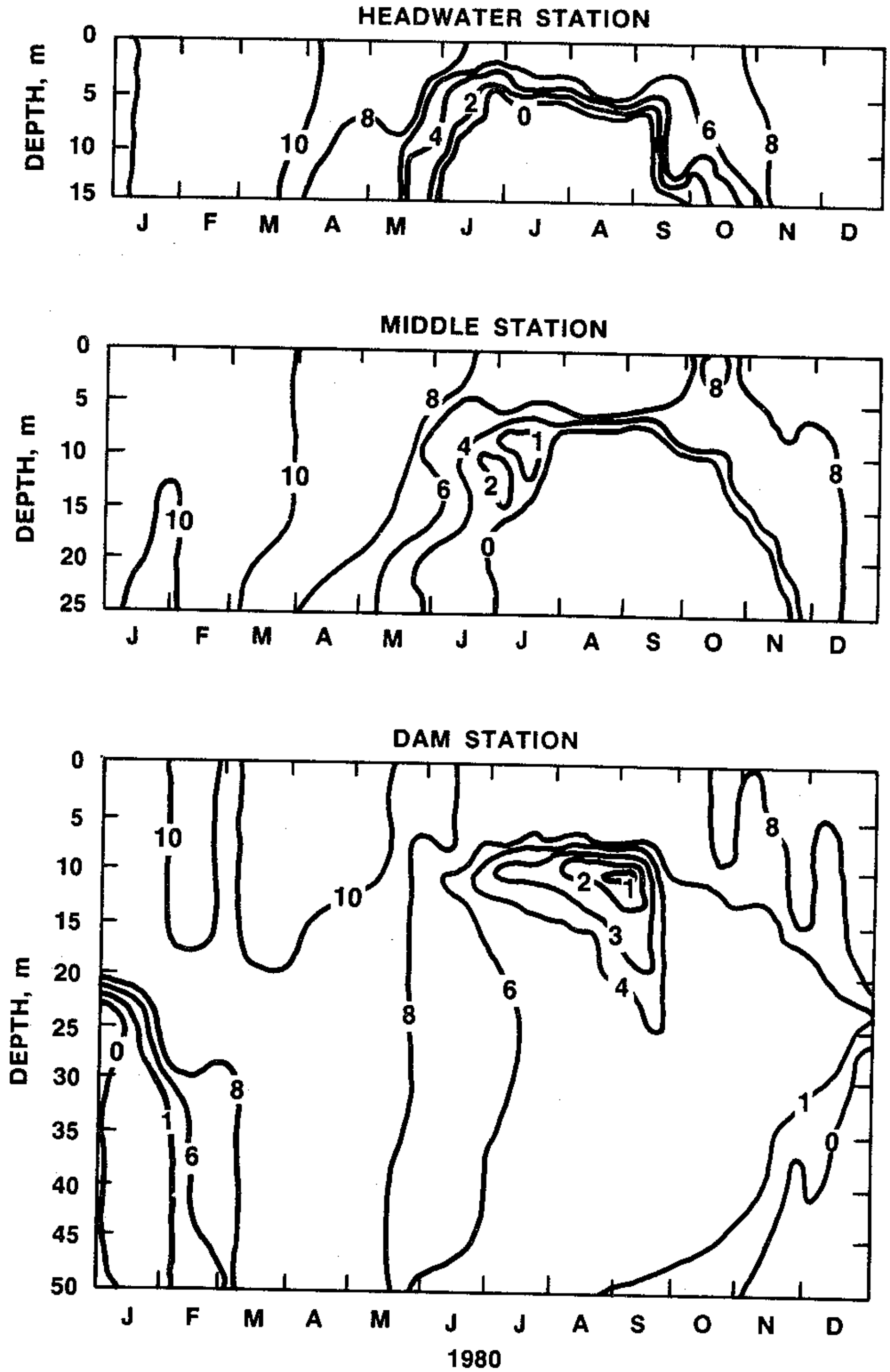

Figure 16. Seasonal and vertical changes in dissolved oxygen (mg · ℓ^{-1}) at the headwater, middle, and dam stations in DeGray Lake.

GENERAL DISCUSSION AND CONCLUSIONS

An understanding of seasonal, longitudinal and vertical patterns in material sedimentation provided insight to interactions between material loadings and reservoir water quality characteristics (Figure 17). In late winter and early spring material loadings from the Caddo River were high and longitudinal water quality gradients were pronounced (Figure 17a). River water proceeded through the reservoir as a weak underflow during periods of isothermy, and as a well-defined interflow following the onset of thermal stratification. Material fluxes to the sediment were greatest at the headwater station and distinct longitudinal gradients in deposition were apparent. During isothermal conditions, river loads probably mixed vertically in the headwater region resulting in similar deposition rates at both 5-and 15-m depths. During the stratified period in early spring, deposition at the headwater station was highest at the 15-m depth, below the area of interflow. At the middle station, smaller deposition increases at the 15-m depth, below the area of interflow, suggested some horizontal movement down-reservoir. Deposition was lowest in the dam region and not affected by river loading. Chlorophyll a concentrations were low during this period indicating that major sources for deposition came from river loading.

High inflow nutrient concentrations and sedimentation, in turn, appeared to stimulate longitudinal gradients in C-14 productivity and chlorophyll a concentrations in early summer (Figure 17b). As nutrient-rich river water moved through the reservoir in early spring, uptake by algae and removal by sedimentation progressively decreased nutrient concentrations down-reservoir, thereby limiting productivity near the dam. Deposition decreased markedly in early summer in the absence of riverine influences. Soon after thermal

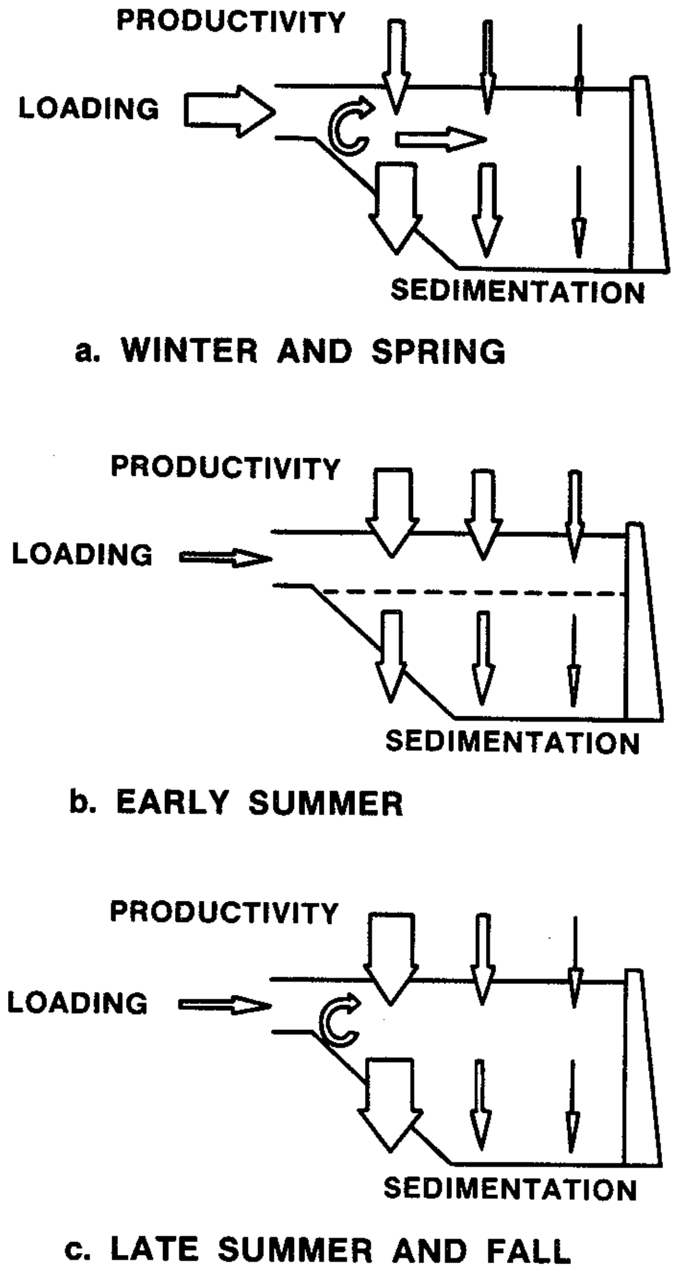

Figure 17. Generalized diagram of major material fluxes in DeGray Lake. The relative magnitudes of flux are indicated by arrow thickness.

stratification, hypolimnetic anoxia occurred at the headwater station with accompanying solubilization of iron, phosphorus, and other constituents (Kennedy et al., 1983a). Thus, major material flux would be from the sediment to the water at the headwater station and diminished flux at the other stations.

In late summer, progressive thermal destratification in the headwater region caused chemical precipitation of iron from the hypolimnion and potential redistribution of particles from the shallow areas to the deeper portion of the region (Figure 17c). Kennedy et al. (1983a) reported soluble iron accumulation in the anoxic hypolimnion of the headwater station during the summer stratified period. Sediment trap data indicated that iron ultimately returned to the headwater sediment during thermal destratification and reoxygenation. Iron hydroxide precipitation probably did not occur in the oxygenated epilimnion and entrainment of hypolimnetic iron would be prevented by its rapid oxidation reaction. The lack of iron deposition fluctuations at the 5-m depth was consistent with this. Iron precipitation would be confined to the hypolimnion during reoxygenation, which was supported by trap rate increases at the 15-m depth in late summer. Anoxic conditions were less evident at the middle and dam stations, and iron precipitation was minimal.

At the headwater station, seston deposition also exhibited depth-related increases by late summer. However, a smaller but distinct increase occurred at the 5-m depth, which was unlike iron. A large percent (87%) of the headwater region surface area is less than 5 m in depth, suggesting that wind activity and convective mixing could resuspend particles from shallow areas which then settle from the epilimnion. Greater deposition at the 15-m trap of the headwater station indicated potential resuspension and redistribution of material within the hypolimnion as well. River loading was minimal during this period, supporting the contention that seasonal maxima of iron and seston at the headwater station resulted from chemical precipitation and sediment redistribution phenomena associated with thermal destratification. Thermal stratification persisted in the middle and dam regions, and deposition maxima did not occur.

Carbon, nitrogen, and phosphorus displayed a different pattern of deposition in the headwater station during late summer. High trap rates at the 5-m depth indicated a considerable nutrient flux from the epilimnion, which was associated with a pronounced phytoplankton bloom and thermal destratification. Thus, major fluxes to the sediment in late summer were associated with thermal events and periods of algal growth. The magnitude of flux was greatest in the headwater region and diminished at the downstream stations.

In general, riverine influences were the cause of longitudinal gradients in deposition and water quality. For instance, high suspended solids loading caused increased turbidity and seston deposition in the upper reaches of the reservoir. As advective velocities decreased, the removal of suspended material by sedimentation acted to diminish particulate loads to down-reservoir locations, thereby sustaining longitudinal gradients in turbidity. Kennedy et al. (1983a) documented similar total phosphorus gradients in DeGray Lake (1980) during high river loads, which reflected longitudinal deposition patterns.

Gradients in material deposition had important ecological significance for nutrient cycling and reservoir metabolism. From deposition data, it was evident that the headwater region acted as a nutrient sink for phosphorus. The annual mean C:P ratios at the headwater station were low (25:1) compared to a more typical ratio of 40:1 for algae (Redfield et al., 1963), suggesting that a portion of the settling phosphorus load was inorganic, adsorbed to clays and ferric hydroxides. The freshly deposited phosphorus became resolubilized during anoxia at the headwater station, accumulating in the hypolimnion by mid-summer (Kennedy et al., 1983a). Kennedy et al. (1983a) showed that eiplimnetic total phosphorus concentrations increased at the headwater station in late summer (1980). Since river loading was low, their data suggested the potential upward flux of hypolimnetic phosphorus through seiche activity and convective mixing. Coincident with elevated concentrations in the epilimnion was an algal bloom which potentially utilized the recycled phosphorus for growth. Phosphorus deposition provided a link between spring phosphorus loading and internal recycling in the headwater region. The storage of riverborne phosphorus in the headwater sediment, therefore, potentially regulated phosphorus dynamics down-reservoir where storm events had little effect.

The sediment trap data also supported the heuristic model of Thornton et al. (1981), which divided reservoirs into metabolic zones on the basis of water quality gradients and the importance of organic carbon sources. Loading and deposition of allochthonous organic matter may subsidize detrital food webs in the upper reaches of a reservoir causing the metabolic state to approach heterotrophy. Down-reservoir, where riverine influences are less pronounced, autochthonous productivity may become the more significant source of organic carbon. Longitudinal gradients in organic carbon deposition provided inferences for a general scheme of major fluxes of both sources to the sediment (Figure 18). Near the river mouth, allochthonous carbon loading predominates during storm events. As advective velocities decrease down-reservoir, riverine influences become less pronounced (Figure 18a). Deposition of allochthonous carbon is probably not maximal at the river mouth because turbulent forces often cause particles to remain suspended and move further down-reservoir (Kennedy et al., 1982). Maximal allochthonous

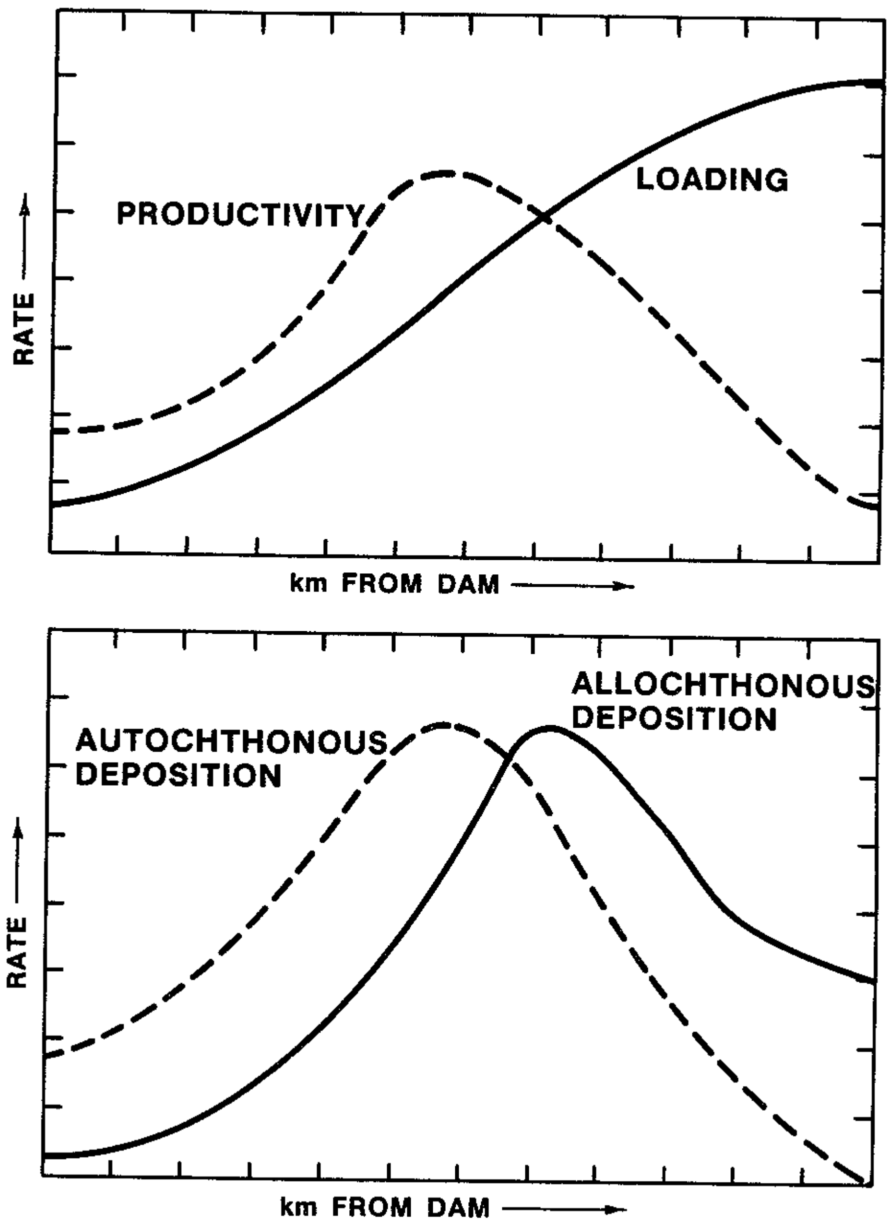

Figure 18. Generalized diagram of major organic carbon fluxes in DeGray Lake.

deposition would occur when advective velocities decrease near the area of the plunge point. Ford and Johnson (1981) have shown that due to the length of DeGray Lake, advective velocities frequently stall between the headwater and middle station, which coincided with maximal deposition rates observed in the headwater station during spring freshets in 1980. Allochthonous carbon deposition becomes markedly diminished down-reservoir, which was supported by low trap rates in the middle and dam regions during spring loading.

Autochthonous productivity and deposition are generally low near the river mouth because turbid water often limits algal growth by the amount of available light (Gloss et al., 1980; Kennedy et al., 1982). Light transmittance then increases down-reservoir as suspended materials settle from the eutrophic zone, stimulating productivity where nutrient conditions are most favorable. Autochthonous deposition is greatest in the zone of high productivity, again located near the headwater region of DeGray Lake. Nutrient depletion and lower productivity toward the dam are accompanied by lower deposition rates.

Gradients in the timing of hypolimnetic anoxia would correspond to gradients in organic carbon deposition and changes in channel morphometry. Near the river mouth, dissolved oxygen demands would be offset by turbulent mixing and reintroduction of oxygen. Changes in channel morphometry down-reservoir are accompanied by thermal stratification and the existence of a hypolimnion. In the shallow headwater region, hypolimnetic oxygen depletion is extensive where freshly deposited organic material is utilized by detrital communities. Down-reservoir, increasing hypolimnetic volume and lower deposition rates would increase the time required for anoxic conditions to occur. In DeGray Lake, anoxic conditions first developed at the headwater station in late May, spreading to the dam station by November (Figure 16).

These conceptual models of material flux may also be applicable to other reservoirs and lakes receiving river loads. Gradients in water quality and sedimentation have potential ecological significance for ordering events in time and along a lake's axis (Kennedy et al., 1983b). The accumulation of organic carbon and other nutrient loads in the headwater sediments may regulate detrital dynamics and nutrient cycles at the down-stream stations. Studies on longitudinal patterns of benthic oxygen demand in large reservoirs might provide insight into the processing of particulate organic loads and recycling of nutrients to which depositional patterns are likely related.

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ANAEROBIC CHEMICAL RATE COEFFICIENTS OF DeGRAY LAKE*

by

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Introduction

DeGray Lake annually experiences oxygen depletion in bottom waters and subsequent accumulation of anaerobic products within its hypolimnion during the stratification period. These anaerobic release processes from sediment to anoxic waters can severely downgrade water quality. Therefore, to adequately model the water chemistry of DeGray Lake, the accumulation and fate of anaerobic products must be considered. One means to accomplish this is to apply an anaerobic subroutine based on the conceptual model of Gunnison and Brannon (1981) which has been incorporated into a comprehensive, one-dimensional reservoir water quality model (CE-QUAL-R1). This subroutine, however, has little utility if appropriate, site-specific anaerobic rate coefficients for DeGray Lake are unavailable. Although some anaerobic release rates have been reported for other aquatic environments (Gordon 1976; Sonzogni et al. 1977), these rate coefficients are of limited use because of the limited numbers of compounds for which rates exist and the doubtful applicability of these rates to DeGray Lake.

This report presents a compilation of anaerobic release rates for DeGray Lake sediments generated using DeGray Lake sediment in large-scale laboratory reactor units (Gunnison et al. 1980). Correlations between anaerobic release rates and sediment characteristics are also discussed.

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* This work was supported by the US Army Corps of Engineers Environmental and Water Quality Operational Studies Program.

Materials and Methods

Sediment samples for determination of anaerobic release rates were collected from station 12 in DeGray Lake (Figure 1). Sediment samples were taken with an Ekman dredge in early summer 1979 under a water depth of approximately 12 meters. The hypolimnion over this sediment was anaerobic during the sampling period. Sediment was placed into 208-ℓ steel drums with polyethylene liners, sealed with airtight lids, and transported to the Waterways Experiment Station (WES). Upon arrival at the WES, the sediment was stored at 20°C until the laboratory incubation studies were conducted within a month.

The sediment sample was mechanically mixed, then placed in large-scale reactor units (Figure 2) to a depth of 15 to 20 cm for experimental analysis. The construction and operation of the reactor units were described in detail elsewhere (Gunnison et al. 1980). The study was conducted in an environmental chamber at a constant temperature (e.g. 20°C) throughout the study.

Prior to initiation of an experiment, 210ℓ of distilled water was added to each reactor unit, and the sediment-water contents of each unit were permitted to equilibrate for a least 30 days with constant aeration and mixing. After equilibration, an initial water sample was taken to provide baseline data under aerobic conditions. Then aeration was discontinued, and the reaction columns were sealed off from the atmosphere. The circulation pump that was used for mixing achieved a complete turnover of reaction column water once every 2 min; this allowed samples to be representative of the entire water column.

Reaction columns were run steadily for at least 100 days and sampled for various physical and chemical parameters (except dissoved oxygen (DO)) at 0, 10, 15, 25, 50, 75, and 100 days. The DO content was measured daily from the initiation of the experiment to the point at which it was no longer detectable or for a period of 30 days, whichever occurred first. In the latter case, DO was subsequently measured at 10-day intervals.

Chemical Analysis

Total Kjeldahl nitrogen and total phosphorus were converted to ammonium and inorganic phosphate, respectively, by digesting water and sediment samples following the procedure of Ballinger (1979). Various forms of inorganic

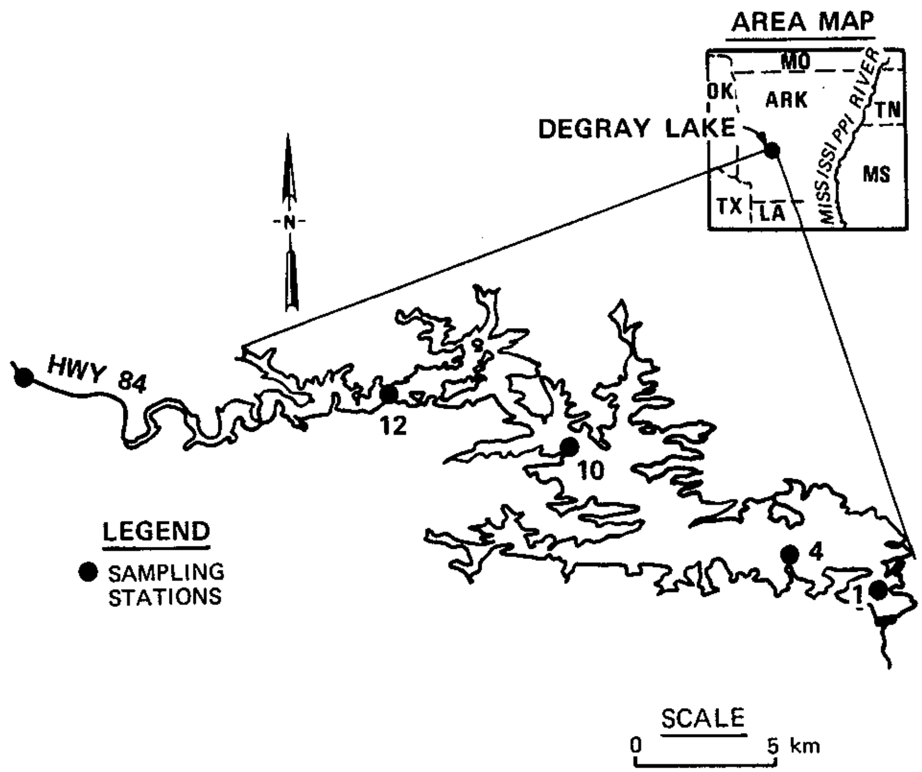

Figure 1. Sampling location in DeGray Lake, Arkansas

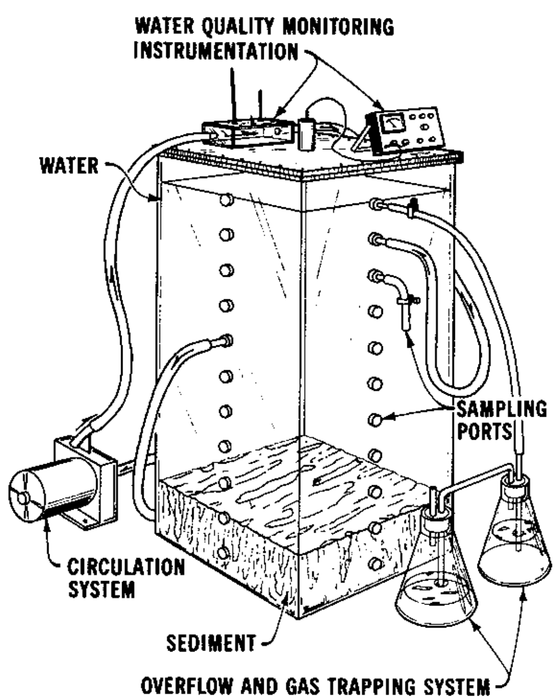

Figure 2. Design of large reactor units

nitrogen and phosphate were determined with a Technicon Autoanalyzer II in accordance with procedures recommended by the U. S. Environmental Protection Agency (Ballinger 1979).

Total carbon and inorganic carbon, including soluble and particulate forms, were determined with a Beckman carbon analyzer (Model 915A) equipped with an infra-red detector. Iron and manganese concentrations were determined with an Atomic Absorption Spectrophotometer. Total carbon in the sediment was determined by the microcombustion procedure of Konrad et al. (1970) or directly converted to gaseous forms in a induction furnance (Leco Model 521-100) and measured by Leco carbon determinator (Model WR-12) equipped with thermistor detector.

Water and sediment pH were measured by the glass electrode. A specific ion meter (Orion Model 404) was used for all measurements. Specific conductivity (corrected at 25°C) was estimated by a mhometer (Lab-Line Lectro Model MC-1), and total alkalinity was determined by the potentiometric titration method (APHA 1980). Water temperature and DO were estimated by using an oxygen meter (Yellow Springs Instruments Model 54) equipped with an oxygen/temperature probe.

Calculation of Rate Coefficients

Anaerobic rate coefficients were determined by performing linear regression analyses of mass release per unit area ($mg \bullet m^{-2}$) versus time. Anaerobic rate coefficients are in the form of fluxes ($mg \bullet m^{-2} \bullet day^{-1}$).

RESULTS AND DISCUSSIONS

Properties of Sediment

The sediment sample obtained from station 12 contained a high proportion of clay and silt with a carbon content of about 2 percent. The very fine texture of the sediment indicated the sediment was probably deposited from the Caddo River watershed, since the sediment properties varied widely from original soil before impoundment. The properties of the sediment used in the chamber study are listed in Table 1. DeGray Lake interstitial water contained 46.4 and 6.4 μg/ml of soluble Fe and Mn, respectively.

Table 1
The Properties and Particle Size Distribution of DeGray Sediment (Station 12) Used for Reactor Unit Study

Particle Size Distribution			Sediment pH	Total Carbon	Total Kjeldahl Nitrogen	Total Phosphorus	Total Iron	Total Manganese
Clay	Silt	Sand						
%				%		µg/g		
30.0	61.0	9.0	7.0	2.05	1.60	80.9	2,365	401

During the summer stratification period, dissolved oxygen in the bottom waters disappeared and remained absent in the hypolimnion from June through late December. At this time high concentrations of anaerobic products such as ammonium, hydrogen sulfide, reduced iron and manganese were released from sediment and accumulated in the anoxic hypolimnion.

Chemical Accumulation in the Anoxic Overlying Water

Many factors affect the levels of reduced substances that accumulate in anoxic waters. Some of the most important factors are duration of stratification, rate of dissolved oxygen depletion, intensity of reduction and the concentration of the component of interest in the sediment.

The longer a reservoir remains stratified, the more time there is for oxygen to become depleted and subsequently for reduced substances to accumulate in the bottom waters. If there is minimal oxygen demand, relatively long periods of stratification may not result in adverse water quality impacts. The major factor affecting the rate of oxygen depletion is the amount of biologically available organic matter. Addition of a carbon source to sediment reactor units greatly increased the rate of dissolved oxygen depletion, resulting in complete depletion of dissolved oxygen within 15 days for DeGray sediment. Much longer periods of time were required for depletion of dissolved oxygen when no allochthonous carbon source was available (e.g., 50 days for DeGray) (Figure 3). The rapid depletion of dissolved oxygen in carbon-amended DeGray reactor units resulted in iron and manganese release earlier in the stratification period. Release does not occur indefinitely,

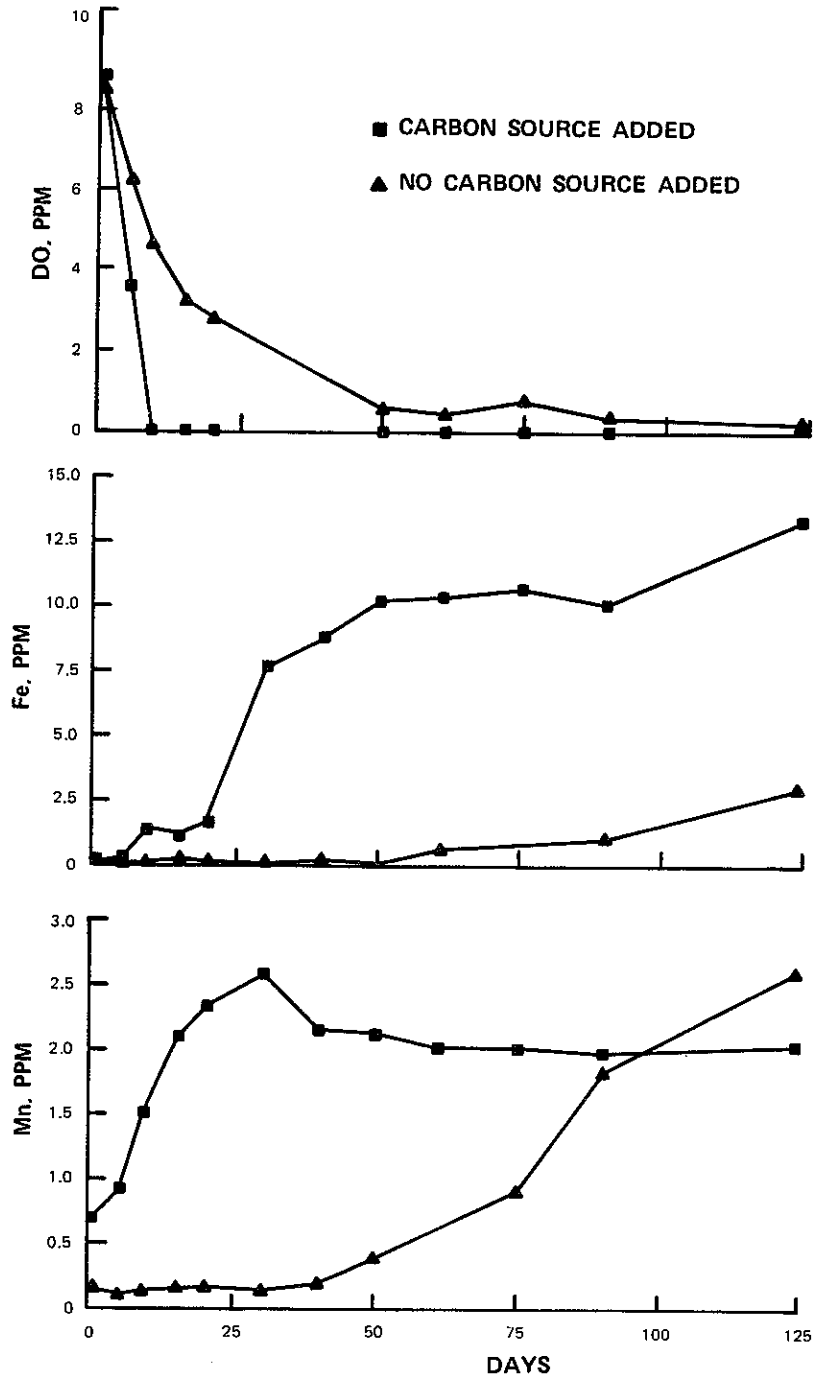

Figure 3. Effect of external carbon source (cellulose) on dissolved oxygen depletion and metal release in DeGray sediment-water system

however, but generally reaches an asymptotic limit. An influx of organic matter may or may not result in a higher final concentration manganese, or other chemicals of interest.

Maximum concentrations of reduced substances that were attained in the anaerobic water columns overlying DeGray and other CE reservoir sediments in our laboratory studies are presented in Table 2. The waters overlying DeGray sediment have high metal and N concentrations and less carbon than do waters overlying most of the other CE reservoirs. The concentrations of reduced chemicals accumulating in the bottom waters can have a considerable effect on the water quality.

Table 2

Maximum Concentrations (mg/ℓ) of Reduced Substances Observed in Reactor Unit Water During Approximately 100 Days of Stratification

	Reservoir	
Parameter	DeGray	Average of 9 CE reservoirs
Fe	0.78	2.71 ± 2.73
Mn	0.58	1.72 ± 1.62
NH_4^+-N	1.14	0.71 ± 0.51
Ortho-P	0.01	0.09 ± 0.11
Total P	0.56	0.30 ± 0.76
TKN	2.63	1.33 ± 0.75
TOC	2.60	4.34 ± 3.01

Anaerobic Rate Coefficients

Anaerobic rate coefficients for DeGray Lake and for other CE reservoirs are presented in Table 3. The DeGray system has a higher release rate of N and lower rate of C than do from other CE reservoir systems under anaerobic conditions. These coefficients are in the form of fluxes of reduced materials from sediments ($mg \bullet m^{-2} \bullet day^{-1}$). A negative flux indicates that during the transition from aerobic to anaerobic conditions or during anaerobic conditions, the mass of that constituent in the overlying water decreased. A positive flux indicates that the constituent of interest was released to the water from the sediment. As can be seen, a wide variety and range of sediment-specific rate coefficients for use in water quality numerical models were obtained.

Table 3

Anaerobic Rate Coefficients ($mg \bullet m^{-2} \bullet day^{-1}$) Derived from Reaction Chamber Studies in Laboratory at 20°C

Parameters	DeGray Lake	Other CE Reservoirs*
DO	-166	-149.7 ± 67.8
NH_4^+-N	8.4	7.5 ± 5.9
NO_3^--N	-18.7	-11.5 ± 8.9
TKN	11.1	16.7 ± 17.4
ORP	0	2.1 ± 3.2
TP	4.1	3.8 ± 3.7
SO_4^{-2}	0	-175 ± 37
Fe^{+2}	14.6	78.2 ± 68.8
Mn^{+2}	9.7	46.7 ± 45.9

* Value = Mean of a reservoir ± one standard deviation

Anaerobic release rates presented in Table 3 for iron, manganese, and orthophosphate-P are lower than the limited data available from the literature. However, release rates of total phosphorus and ammonium-N are in the same range as literature value (Gunnison and Brannon, 1981). With the exception of iron and manganese, anaerobic rate coefficients of DeGray derived from reaction chambers are similar to the averages of other CE reservoirs. Release rate coefficients of iron and manganese were quite low in DeGray compared to those measured from other CE reservoirs in laboratory studies.

Selection of Appropriate Anaerobic Rate Coefficients

Selection of appropriate anaerobic rate coefficients may be approached as follows: (a) use of direct laboratory or field measurements from the reservoir concerned, (b) use of release rates from reservoirs near the project of interest, (c) use of release rates from reservoirs with similar chemical and physical properties to the project of interest, or (d) for some parameters, measurement of sediment interstitial water concentrations and sediment porosity.

Direct measurements using procedures in this report are one of the more accurate but difficult methods of arriving at release rate coefficients. Modelers may also have at their disposal actual field measurements of parameters measured over time during anaerobic conditions. If such field data exist, information concerning the volume of anaerobic water, concentration changes over time, and the surface area of sediment exposed to anaerobic water will allow calculations of fluxes ($mg \bullet m^{-2} \bullet day^{-1}$) to be made.

If a reservoir is in the vicinity of a project for which rate coefficients exist, there is a strong possibility that rate coefficients will be similar. Prior to acting on such an assumption, modelers should ensure that general atmospheric parameters, water circulation and stratification in the reservoirs being compared are similar and that sediment accumulation and general sediment characteristics are also comparable.

Statistical analysis of the relationship between rate coefficients and physical and chemical characteristics of sediment was conducted. Results showed that there was no relationship between rate coefficients and any sediment characteristics listed in Table 1. This was not surprising in view of the complexity of the physical and chemical processes influencing release of materials from sediments (Berner, 1971; Klump and Martens, 1981; Lerman, 1979). However, if such data exist or can be obtained for an existing project of interest, similarities in content of organic matter and reducible substances such as iron and manganese should allow similar release rates to be used as a first approximation. Comparisons should be limited to existing stabilized projects since soil organic matter in preimpoundment areas differs substantially from that of sediments in existing projects (Gunnison et al. 1983).

In theory, fluxes of anaerobic constituents from sediment to overlying waters should be predictable. The fluxes of dissolved constituents across the sediment-water interface are driven by advective transport of the species in pore water solution and attached to sediment particles as well as by molecular diffusion (Lerman, 1979). However, in our reactor units there was no sedimentation, and fluxes of Fe, Mn, orthophosphate-P, and NH_4^+-N were measured when the sediment column and overlying water were anaerobic. In the absence of oxygen, there should, therefore, be little chance for precipitation reactions to reduce the flux of these dissolved species. Under these conditions advective transport and precipitation will be minimal and can be ignored, and the appropriate flux equation reduces to (Berner, 1971):

$$J_s = -\phi_o D_s \left(\frac{\partial c}{\partial z}\right)_{pw} \qquad (1)$$

where

J_s = flux (mg • cm_s^{-2} • sec^{-1}, where s denotes bulk wet sediment)

ϕ_o = porosity at the interface (cm_{pw}^3 • cm_s^{-3}, where pw denotes pore water)

D_s = bulk sediment molecular diffusion coefficient (cm_s^2 • sec^{-1})

$\left(\frac{\partial c}{\partial c}\right)_{pw}$ = pore water concentration gradient at the sediment-water interface (mg • cm_{pw}^{-3} • cm_s^{-1})

Flux of Fe computed using equation (1) for DeGray sediment is presented in Table 3. Comparison of the flux with measured flux showed that observed flux exceeded the predicted by equation (1). A similar relationship between measured and calculated fluxes of other chemicals was reported elsewhere (Brannon et al. 1984). This indicates that processes at the sediment-water interface in addition to simple molecular diffusion was influencing releases, a phenomenon also noted by other workers (Eaton, 1979; Klump, 1980; Klump and Martens, 1981).

Rate data presented in this paper were obtained at 20°C. However, if hypolimnetic waters of a project are substantially lower than 20°C (i.e., 10°C for DeGray Lake hypolimnion), then the rates reported herein will have to be reduced to adequately represent release rates at the lower temperature. In another study (Gunnison et al., 1983), we have examined the effect of temperature on anaerobic release rates and oxygen depletion. Apparently, lowering temperatures approximately 10°C results in halving of anaerobic release rates and oxygen depletion rates. This agreed well with the concept of Q_{10}, that for a 10°C rise in temperature, a doubling in reaction rates occurs (Hoar, 1966). Halving reaction rates obtained at 20°C for reservoir waters at 10°C, if release rates would otherwise be similar, therefore appears to be a valid means of obtaining a rate coefficient estimate.

This laboratory investigation has provided chemical processes rate data that can serve as a basis for an anaerobic subroutine of the numerical reservoir water quality model CE-QUAL-R1. Incorporation of these rate constants into CE-QUAL-R1 will allow evaluation of the impact of anaerobic conditions on the DeGray Lake ecosystem. Results of this study also provided information on the maximum levels of nutrients and metals that would be expected to accumulate in the anoxic hypolimnion of DeGray Lake under stratification.

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Heterogeneities and Water Quality - Understanding DeGray Lake Water Quality Through Sampling[1]

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Abstract

Reservoir water quality exhibits temporal and spatial variability that must be considered in designing sampling or monitoring programs. Spatial variability in longitudinal, lateral, and vertical dimensions can be incorporated by considering the plunge zone, littoral area, and stratification regime, respectively. Sampling schemes should incorporate temporal variability by considering important limnological periods. Temporal variability over daily, synoptic, and seasonal time scales can be incorporated by using variable interval sampling. Quality assurance and quality control in the field are absolutely essential in any sampling program and are stressed. Since the purpose of the monitoring program is to reach conclusions from the water quality data, data analysis must be an integral part of the program.

Our understanding and knowledge of reservoir water quality; our use of mathematical water quality models; and our formulation of appropriate water quality management strategies are all predicated on data. Field data can be the strongest

[1]The data used in this paper were collected by Dr. Joe F. Nix and his staff, Ouachita Baptist University, as part of the Environmental and Water Quality Operational Studies which were sponsored by the Office, Chief of Engineers, U.S. Army Corps of Engineers, Washington, D.C. The data analysis and paper preparation were supported by FTN, Ltd. Without Dr. Nix and his staff, the DeGray studies, our insight, and knowledge of reservoir limnology would not have been achieved. We acknowledge them with gratitude.

evidence to support or refute water quality conclusions. However, if the data is not representative of field conditions, it can be the strongest deterrant to proper understanding of reservoir water quality processes and the formulation of appropriate management strategies. Since significant spatial and temporal variability may exist in reservoir water quality, the sampling program must characterize these dynamic water quality patterns. Adequate characterization is required to determine if water quality objectives are being achieved within authorized project purposes.

One objective of sampling program design is to minimize the uncertainty associated with our conclusions about reservoir water quality based on collecting 'representative' field data. To place this problem in perspective, let us consider the size of the reservoir to be studied and the sample population used to characterize reservoir water quality. DeGray Lake contains 8 x 10^8 or roughly one billion cubic meters of water. Multiple field crews sampling from early morning until late evening, could, perhaps, collect 1000 one-liter water quality samples. This represents one cubic meter of water. This single cubic meter of water, then, must describe the water quality conditions occurring in one billion cubic meters of water in DeGray Lake. This is possible if the water quality samples are collected from representative reservoir locations at representative times.

This paper emphasizes approaches for determining appropriate sampling locations and times to incorporate longitudinal, lateral, vertical, and temporal variability in reservoir water quality monitoring programs. Initial guidance on the optimization of precision, required number of samples, and funding constraints was provided by Thornton et al. (1982).

Objectives

Before the first water sample is collected, the objectives or purpose of the monitoring program must be clearly defined. This obvious first step, in many instances, does not receive adequate attention. Objectives that may be clear and lucid in an individual's mind become vague and uncertain when formally translated in a document or explained to someone else. Although water quality monitoring programs generally represent the most cost-effective approach for evaluating reservoir water quality, monitoring programs are still expensive. It is extremely important, therefore, that program objectives, purposes, and the intended use of the data are clearly understood prior to implementing the monitoring program.

Identifying and defining monitoring program objectives also are important since the sampling design must reflect these objectives. Specific sampling designs will reflect specific study objectives. Estimating 'average' water quality conditions can require a very different design than estimating maximum and minimum constituent concentrations. Unfortunately, sampling design generally is the most neglected phase of the monitoring program and may represent a major source of bias or error in the monitoring program. The sampling design should incorporate considerations of length scales (spatial variability), time scales (temporal variability), and the interaction between the time and length scales.

Spatial Patterns

Incorporating water quality heterogeneities in the monitoring program requires consideration of longitudinal, lateral, and vertical gradients. Water quality patterns in DeGray Lake, although dynamic, reflect three general longitudinal zones or areas - a headwater or riverine zone, a transition zone, and a lacustrine or lake-like area (Fig. 1) (Thornton et al. 1981, Kennedy et al. 1982). Characteristics of these three areas are listed in Table 1. These three zones are dynamic and reflect the hydrologic variability in DeGray Lake and the Caddo River but these zones also retain their general characteristics through time.

Longitudinal gradients - The three longitudinal zones, although time-varying, may be identified by describing the extent of the transition zone. The area upstream of the transition zone

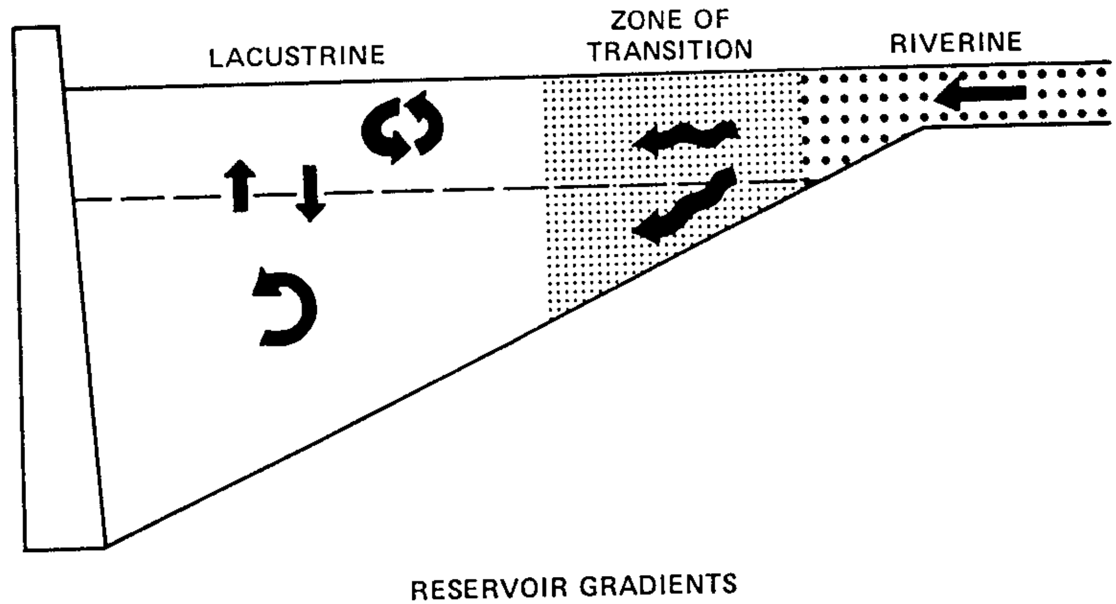

Fig. 1. Three longitudinal zones in a reservoir - riverine, transition, and lacustrine.

Table 1. Characteristics of the riverine, transition, and lacustrine zone (After Thornton et al. 1981, Kimmel 1983).

Riverine Zone	Transition Zone	Lacustrine Zone
1. Narrow, river channel.	1. Broader, deeper basin.	1. Deep, lake-like basin.
2. Advectively dominated regime.	2. Advective > Buoyant regime.	2. Buoyant dominated regime.
3. Well-mixed.	3. Intermittent - seasonal stratification.	3. Strong, seasonal stratification.
4. High turbidity, SS.	4. Plunge point	4. Inorganic particulate sedimentation low.
5. High nutrient concentration.	5. Increased sedimentation, light penetration.	5. High light penetration.
6. Light limited primary productivity.	6. Relatively high nutrient concentration - Advective entrainment.	6. Nutrient limited primary productivity.
7. Allochthonous > Autochthonous organic matter.	7. Primary productivity/m^3 high.	7. Autochthonous > Allochthonous organic matter
8. "Eutrophic."	8. Allochthonous - autochthonous organic sources.	8. "Oligotrophic."
	9. "Mesotrophic."	

reflects riverine characteristics while the area downstream represents the lacustrine zone. The transition zone in DeGray Lake and many other reservoirs is defined by the location of the plunge point. Ford and Johnson (1981, 1983) reviewed inflow mixing processes and formulations for predicting the plunge point depth. One general formulation for the plunge point was:

$$d = \left(\frac{1}{F_o}\right)^{1/3} \left(Q^2/(W^2 \cdot g \cdot \frac{\Delta\rho}{\rho})\right)^{1/3}$$

where

d = hydraulic depth at the plunge point

F_o = densimetric Froude number

Q = inflow rate

W = width of the zone of conveyance

g = gravitational acceleration

$\Delta\rho$ = density difference between the lake surface and inflow waters.

Using representative lower and upper flow rates and the corresponding density differences at these times, the general upstream and downstream boundaries of the transition zone can be determined, respectively (Fig. 2). The actual plunge point location may occur further upstream or downstream as a function of lower or higher flows (e.g., storm flow). Depending on the specific study objectives, a lower or higher range of flow rates may be used to characterize the transition zone. Locating a sampling station upstream of the upper boundary of the transition zone, within the transition zone, and downstream of the lower boundary can characterize these longitudinal patterns in reservoir water quality.

Another useful approach, if sufficient water quality data is available, is to perform a variance component analysis of variance. A variance component analysis partitions the total variance into its constituent parts to determine which component is contributing the greatest percentage of the variability to the

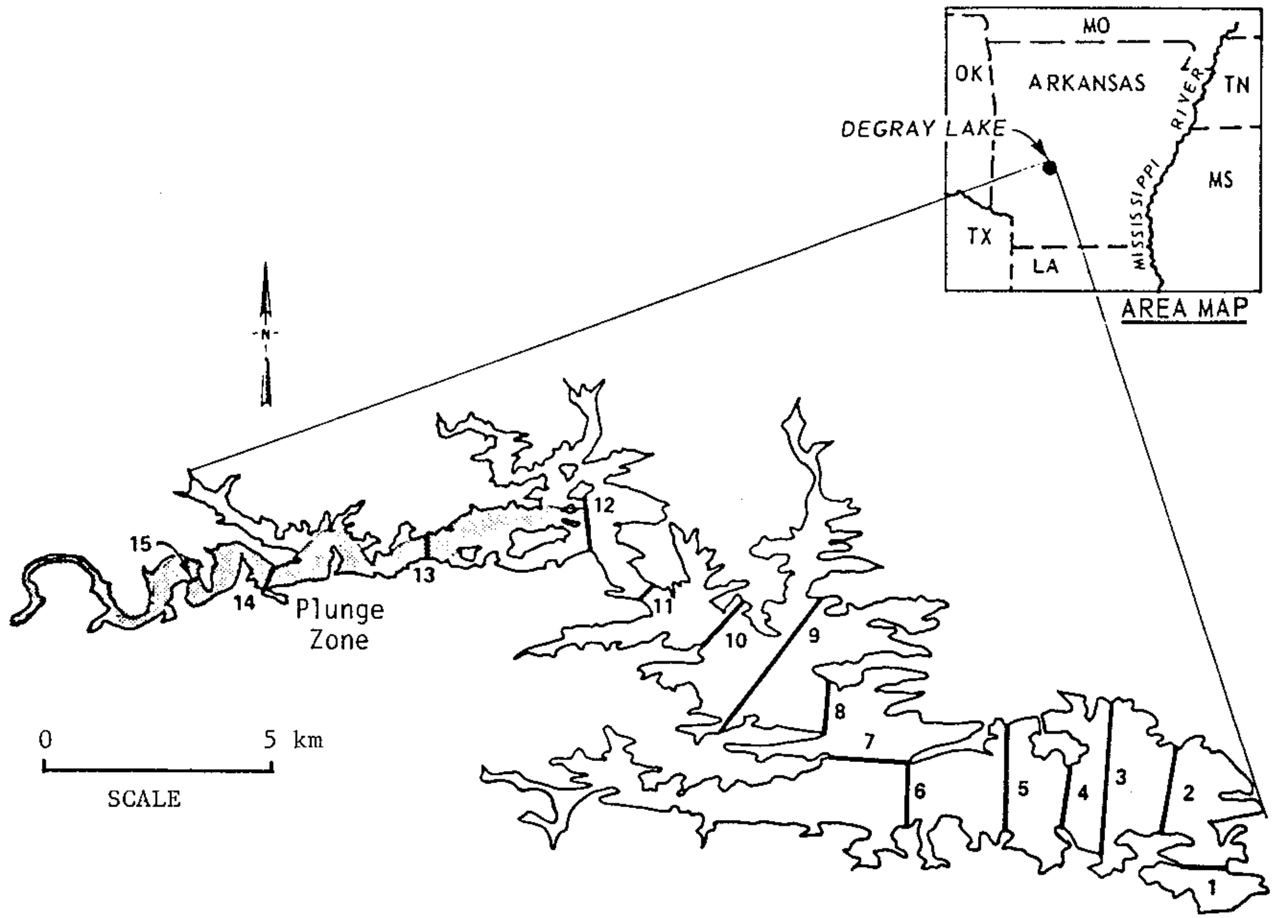

Fig. 2. Extent of the plunge zone in DeGray and, therefore, the general chracterization of the transition zone. The riverine zone is generally upstream and the lacustrine zone is generally downstream from the transition zone.

data. Sampling intensity can be shifted from components with low variability to components with high variability for a more cost effective sampling approach. Thornton et al. (1982) determined 40-90 percent of the water quality variability in DeGray Lake throughout the year was associated with longitudinal patterns. The remainder of the variability was associated with vertical gradients. A simple statistical model was developed to describe longitudinal patterns occurring in DeGray Lake and determine appropriate sampling locations (Thornton et al. 1982).

Lateral gradients - Lateral variability also may influence conclusions drawn from a water quality sampling program. Jirka and Harleman (1979) used an aspect ratio (i.e., ratio of the reservoir length/width) to classify cooling ponds according to longitudinal and lateral patterns. An aspect ratio greater than 4 indicated longitudinal dispersion dominated and lateral variability was minimal. An aspect ratio less than 4 indicated lateral circulation patterns were important and could begin to influence water quality patterns. An aspect ratio less than 4 could require a sampling station in the littoral area and over the old thalweg or river channel.

Another order of magnitude approach to assess the potential for lateral variability is to compare the width of the littoral zone to the width of the reservoir (Fig. 3). The depth (Z_p) corresponding with two times the Secchi depth or the 1 percent light penetration can be computed and topographic maps or sediment transect survey cross sections used to determine where this depth impinges on the bottom. The width from this point to the shore (W_p) can be compared with the total reservoir width (W_t). If this ratio is significant, lateral variability may be important and a sampling station in the littoral area may be required.

The development of volume or mean/maximum depth (i.e., Z/Zm) may indicate potential lateral variability. Most lakes have values ranging between 0.20 and 0.5. Ratios greater than 0.5 indicate steep-sloped systems with little littoral area whereas ratios much less than 0.20 are typical of lakes with a deep hole

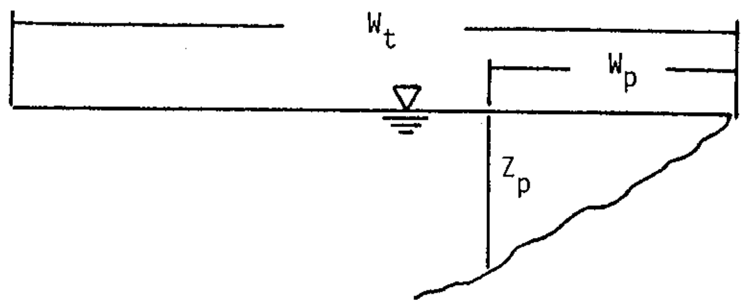

Fig. 3. Comparison of potential width of the littoral area (W_p) to total reservoir width (W_t) using twice the Secchi depth (Z_p) to define boundary of the littoral area.

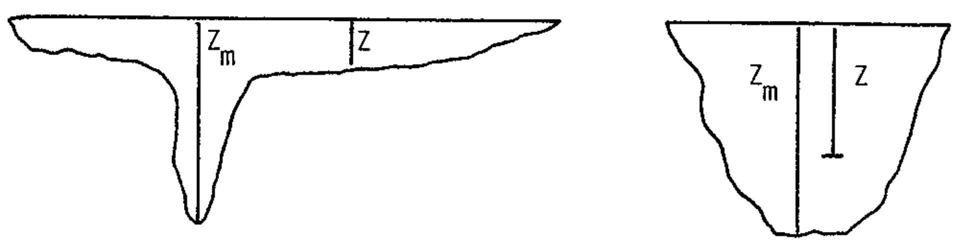

Fig. 4. Development of volume (Z/Z_m) as an estimate of potential lateral variability.

but potentially extensive littoral areas (Fig. 4). Additional information is required to determine if lateral variability is important in reservoirs with ratios from 0.2 - 0.5.

Specific conductance and transmissometer readings can be used as surrogates for dissolved and suspended constituents. Large variations in specific conductance or transmissometer readings across a transect may indicate significant lateral variability in dissolved or particulate constituents and require lateral sampling stations. Periodic measurements across selected transects represent a relatively quick and inexpensive approach for determining lateral variability.

Vertical gradients - Significant vertical variability has been recognized in most water quality sampling and monitoring programs. Thermal stratification can result in three clearly defined vertical zones in the reservoir - epilimnion, metalimnion, and hypolimnion (Fig. 5). In situ temperature, DO, specific conductance and, perhaps, pH measurements can be taken at 1-2 m intervals to describe the vertical patterns of these constituents within the water column. Once the thermal and DO regime is determined, water quality samples generally are collected at representative depths to characterize water quality conditions throughout the water column. One approach is to collect samples near the water surface, bottom of the epilimnion or mixed layer - top, middle, and bottom of the metalimnion - and top, middle, and bottom of the hypolimnion. For systems that exhibit a classical thermal stratification pattern, this sampling approach may be appropriate. However, the onset of stratification and erosion of the metalimnion implies the sampling depths may vary on each sampling date. In addition, determining appropriate sampling depths can be difficult in reservoirs that do not exhibit a classical stratification pattern. A fixed sampling depth approach makes field sampling and comparisons of constituent concentrations among depths easier. Depths can be selected to characterize the stratification pattern during the major portion of the growing season.

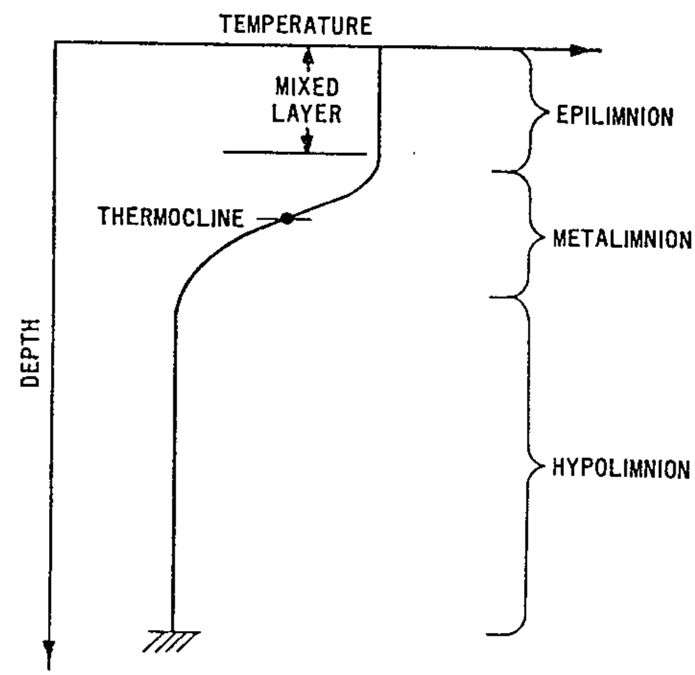

Fig. 5. Vertical zonation in lakes and reservoirs based on the thermal regime.

The distribution of physical and chemical constituents throughout the water column may be different than the distribution of biological organisms. Biological organisms can regulate their buoyancy, and to some degree, control their distribution throughout the water column. Discrete samples at a specific depth may not be representative of biotic concentrations throughout the mixed layer due to microstratification (e.g., concentration of organisms in a band a few cm thick). Thornton et al. (1980) compared mean total coliform densities in the mixed layer for discrete depth samples versus integrated samples. On some dates, the mean densities for the integrated samples were two orders of magnitude larger than the mean densities for discrete depth samples (Table 2). Integrated samples over the mixed layer may provide a better estimate of 'average' conditions in the epilimnion for biological constituents such as chlorophyll concentrations or plankton densities.

Temporal Patterns

The sampling date and sampling time during the day are as important in characterizing reservoir water quality as the sampling location. Reservoir water quality monitoring programs typically sample the tributary inflow and reservoir stations on some fixed interval, usually every month. This fixed interval sampling, however, may not correspond with the important limnological periods influencing reservoir water quality. Fixed interval sampling every 30 days on the Caddo River during 1977 could have missed sampling all of the storm flow that occurred in the Caddo River (Fig. 6). Montgomery (1983) has indicated the importance of sampling storm flow and the significant storm event loads on reservoir nutrient budgets. While storm events are important external nutrient loading sources, the development of hypo- or metalimnetic anoxia may be an important internal nutrient loading source (Kennedy 1983). This internal source becomes important during low flow periods. Important limnological periods are linked to the extremes in water quality variables (e.g., storm event loading, high flows, low DO). One

Table 2. Comparison of discrete sampling depth mean with value from integrated samples for total coliforms (after Thornton et al. 1980).

STATION 10

DATA 1977	MEAN FOR DEPTHS 0, 3, 5 METERS TOTAL COLIFORMS CELLS/100 ML	INTEGRATED SAMPLE OF UPPER 5 METERS TOTAL COLIFORMS CELLS/100 ML
5-27	237	5000
5-31	467	700
6-2	167	58000
6-6	333	10700
6-9	633	26000
6-12	600	-
6-15	367	52000
6-16	4233	2500
6-18	933	3000
6-21	1000	36000
6-23	5600	19800

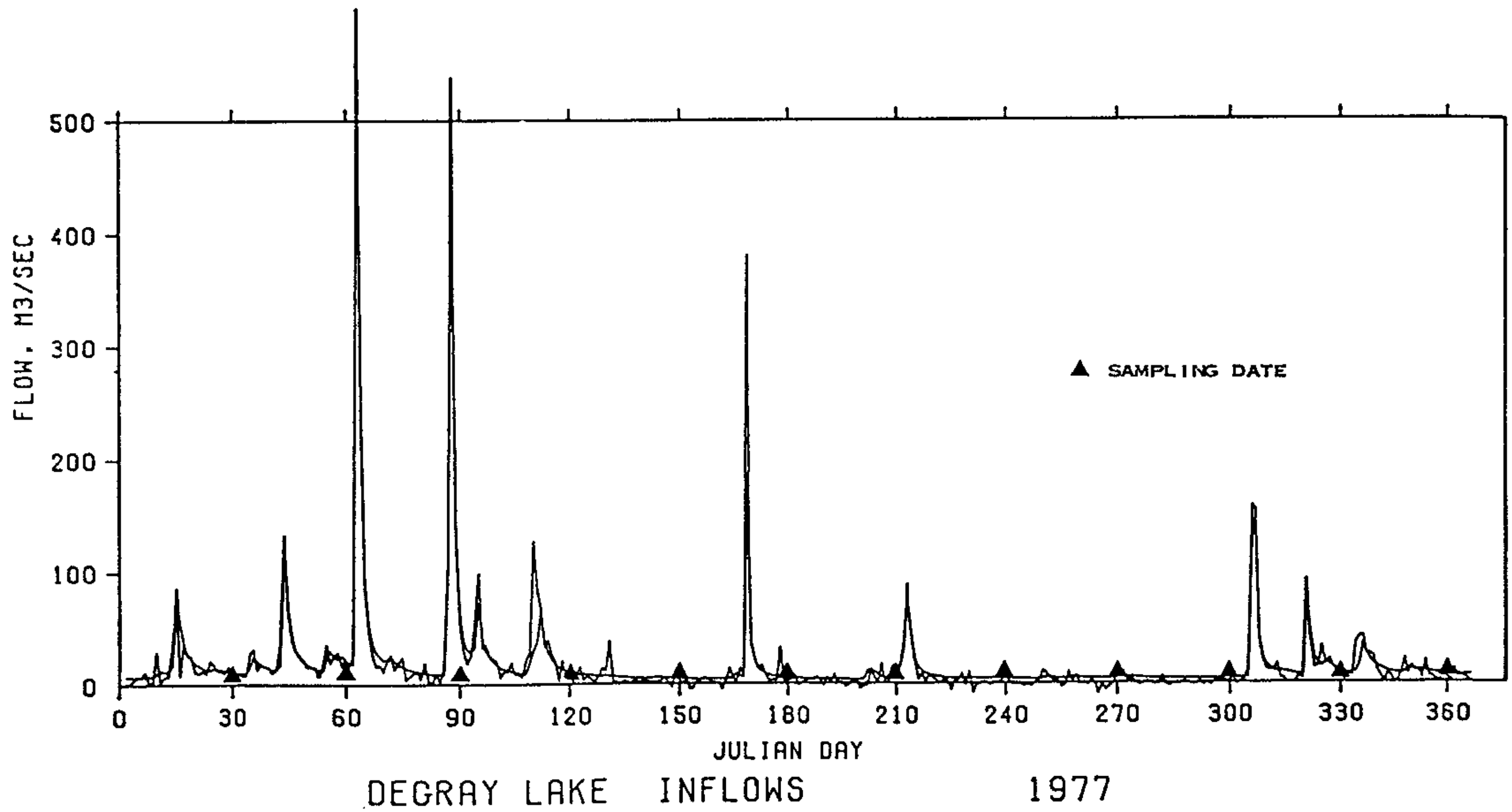

Fig. 6. Large flow events in Caddo River and fixed interval sampling approach. During 1977, a 30-day sampling approach could have missed sampling all storm events.

approach for incorporating these important limnological periods in the monitoring program is to use a variable interval sampling approach.

Variable interval sampling occurs with respect to the important periods influencing reservoir water quality such as periods of elevated flow, strong stratification, or high internal loading (Table 3). No additional sampling dates are required beyond the number required for a fixed interval sampling approach (e.g., monthly interval) but more information is obtained about reservoir water quality throughout the year than with a fixed sampling interval. The intervals selected should be based on the study objectives and important periods influencing the system. In northern reservoirs, the period immediately preceding ice out is often characterized by low DO and may be a critical period for study. In southern reservoirs that remain well-mixed throughout the winter, a single sampling date during this well-mixed period following fall turnover may be adequate to characterize water quality during this season.

Synoptic time scales (Johnson 1983) also can be incorporated by intensifying the sampling interval during periods when synoptic weather patterns may be important. Kennedy (1983) discusses the entrainment of nutrients from anoxic zones and their potential impact on productivity in the epilimnion. Intensifying sampling during the peak of the growing season (e.g., July or August) can indicate the influence of synoptic patterns on water quality constituents such as nutrient, chlorophyll, or bacterial concentrations.

Daily or diel and diurnal patterns also influence reservoir water quality. During the early morning hours before the surface waters in the reservoir warm, temperatures are relatively uniform throughout the epilimnion (Fig. 7a). As solar insolation warms the surface waters, however, secondary stratification patterns can be established in the epilimnion (Fig. 7a). During the night, convective mixing may again eliminate these secondary stratification patterns resulting in a relatively uniform temperature in the epilimnion. These secondary patterns can

Table 3. Incorporation of the important limnological periods in the sampling program using variable interval sampling.

DATE	EVENT
MID - MARCH	ISOTHERMAL
MID- APRIL LATE APRIL MID - MAY	ELEVATED FLOW ONSET STRAT.
EARLY - MID JUNE MID - LATE JUNE	INCREASED BIOL. ACTIVITY INCREASED PUBLIC USE
MID - JULY	STRONG STRAT. LOW RUNOFF
EARLY AUGUST MID - AUGUST LATE AUGUST	ANOXIC CONDITIONS PLANKTON BLOOMS LOW FLOW
EARLY - MID OCTOBER EARLY - MID NOVEMBER	THERMOCLINE EROSION
MID - LATE DECEMBER	ISOTHERMAL

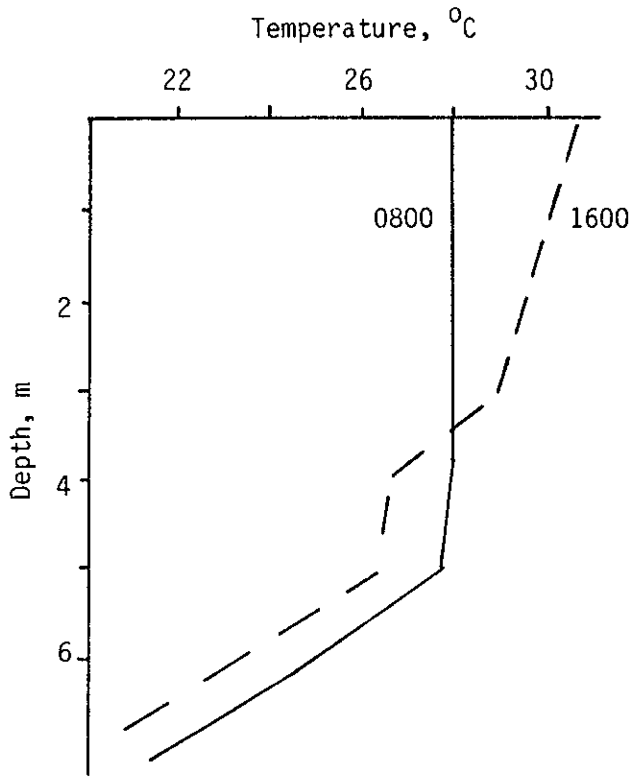

Fig. 7a. Influence of nocturnal convective cooling (0800 hr) and diurnal surface warming (1600 hr) on mixed layer temperatures.

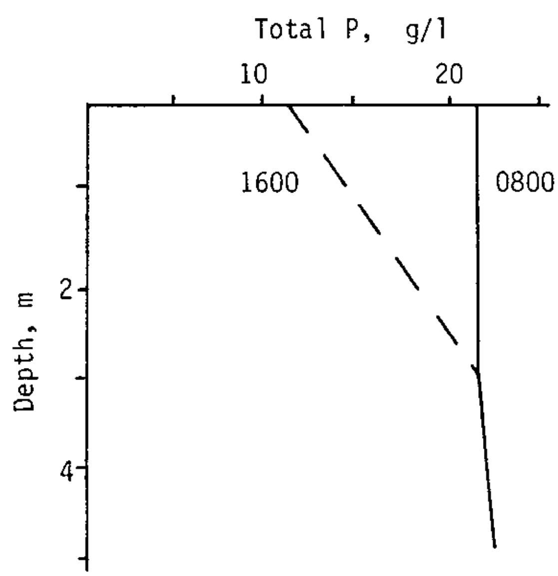

Fig. 7b. Influence of mixing (0800) and microstratification (1600) on nutrient concentrations in the photic zone.

influence other water quality constituents such as nutrients and plankton. During the early morning hours, total phosphorus concentrations may be uniformly distributed throughout the mixed layer (Fig. 7b). However, during the afternoon hours, phosphorus concentrations may be reduced (or increased) due to nutrient uptake (or release) by plankton in this stratified surface layer (Fig. 7b). Microstratification of plankton (Poppe, 1983) or bacteria in response to secondary thermal stratification patterns can influence the collection of representative samples and, ultimately, the conclusions that would be drawn from this data. Sample collection during the early morning period may be more representative of 'average' conditions in the epilimnion or mixed layer while samples collected during the afternoon may represent extreme concentrations (maxima or minima) for various constituents. The time of sampling should be noted for all water quality samples since it may markedly influence the interpretation of this data.

Quality Assurance/Quality Control

Quality assurance (QA) is the orderly application of practices and procedures during sampling and analysis to improve precision (APHA 1980). This involves collecting replicate samples, splitting samples, and other procedures to determine sampling variability. Quality control (QC) is the application of practices and procedures during sampling and analysis to ensure the accuracy of results (USGS 1977). Spiking field samples with known constituent concentrations and participating in EPA's analytical QC programs are important in obtaining representative water quality data. QA and QC begin prior to field sampling and must be considered throughout the field sampling program. Laboratory QC and QA program cannot compensate or remedy poor field sampling procedures.

Interpretation

It is difficult to refute sound conclusions drawn from a well-designed and implemented field data collection program. This assumes, however, the data is collated, graphically displayed, and at a minimum, examined for apparent patterns. Collection of data simply to satisfy a regulatory requirement or edict does not satisfy the intent of these regulations. It also represents a considerable expenditure of funds that could be put to more beneficial use. In 1977, the CE spent $25 million on water quality data collection programs. Proper interpretation of this data and use in formulating environmentally sound management strategies can represent one of the most cost-effective approaches in achieving reservoir water quality objectives. At a minimum, inflow versus outflow data and inpool constituent profiles should be plotted and simple summary statistics (e.g., mean, median, standard error, etc.) should be calculated from the data. If these procedures are not applied to the data and the data used by the engineer and scientist, then the most important source of water quality information on the reservoir is lost and speculation supersedes sound engineering judgement.

Interdisciplinary Approach

The measured water quality data represents the reservoir response to the interactions of the physical, chemical, and biological components of the system in conjuction with water control. Assessing reservoir water quality, then, is a multifaceted problem and requires the knowledge from many disciplines. Since this knowledge rarely resides in any single individual or discipline, an interdisciplinary approach is required to analyze and solve water quality problems. An interdisciplinary approach implies multiple disciplines (e.g., engineering, physics, chemistry, biology, economics, etc.) cooperating and interacting to address and solve water quality problems and not simply the assemblage of multiple disciplines working independently on the problem. CE District and Division offices are uniquely suited to use an interdisciplinary approach on water quality problems because the required multidisciplinary expertise exists in all CE field offices. The three major elements in CE field offices - Planning, Engineering, and

Construction/Operations - all have responsibility for various aspects of reservoir water quality. Problems identified and resolved during initial planning phases can minimize problems during later design phases and subsequently during reservoir operation and management. This interdisciplinary expertise must be used in assessing and managing reservoir water quality.

Summary

Our knowledge, understanding, conclusions, and management strategies for reservoir water quality are predicated on data. Greater attention, therefore, must be placed on ensuring this data is representative of water quality conditions and processes occurring within the reservoir.

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Thermal Modeling of DeGray Lake[1]

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Abstract

A one-dimensional reservoir thermal model (CE-THERM-R1) was developed to simulate the effects of reservoir operation on both the in-lake and downstream temperature regimes. During model development, emphasis was placed on formulating algorithms describing inflow mixing, selective withdrawal, and internal mixing which accurately simulate the dynamics of mixing in reservoirs. An integral energy approach was used to simulate mixed layer dynamics.

The model was applied to two reservoirs of similar morphometry yet different operating strategies. The model was calibrated on one data set from DeGray Lake, AR, then verified using four other independent data sets from DeGray Lake and Lake Greeson, AR. The model was able to reproduce observed phenomena such as the onset of stratification, surface temperatures, depth of the mixed layer, density gradient in the metalimnion, and hypolimnetic temperature.

Increasing environmental awareness has led to the use of reservoir water quality models for assessing the effects of reservoir operation on both in-lake and downstream water quality. A key element in any water quality modeling study is the accuracy of the temperature predictions. Temperature dynamics not only establish stratification patterns, but

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also influence temperature dependent physical, biological, and chemical processes. Thermal stratification is one of the most significant phenomena impacting reservoir water quality. If a water quality model cannot accurately simulate the dynamics of the thermal cycle (e.g., onset of stratification, mixed layer depth, and turnover periods), then the simulations of the other water quality parameters and processes are questionable.

In most water quality modeling applications, predictive models are calibrated with data from one year and then applied to evaluate various management strategies. Little effort has gone into verifying the results from these predictions. In some cases, the model must be recalibrated if it is to be used for another year or another lake. Ideally, once a model has been calibrated on one lake for one year, it should be capable of simulating three conditions: the same lake but a year characterized by different hydrometeorological conditions, e.g., wet versus dry year; the same lake but with a different operation scheme, e.g., bottom versus surface withdrawal; and a different lake that is similar in location and morphometry.

The objective of this study was to verify the model described herein, CE-THERM-R1, by calibrating on one data set and verifying against four independent data sets characterized by different hydrometeorologic conditions and operating strategies. Predictions were evaluated with emphasis placed on the model's ability to realistically reproduce observed phenomena (e.g., onset of stratification, mixed layer depths, density gradient in metalimnion, etc.) rather than matching individual data points.

CE-THERM-R1 is a submodel of CE-QUAL-R1, a Corps of Engineers (CE) one-dimensional reservoir water quality model (Environmental Laboratory 1982). CE-THERM-R1 simulates temperature, dissolved solids, and suspended solids and can be used for thermal analysis in pre- and post-impoundment studies of reservoirs where vertical gradients of temperature and solids may be important. In model development, emphasis was placed on formulating algorithms describing inflow mixing, selective withdrawal, and internal mixing which accurately simulate the dynamics of mixing in reservoirs. Specific details concerning the model can be found in Environmental Laboratory (1982).

As previously stated, the model is one-dimensional. The reservoir is assumed to be made up of a series of horizontal layers of variable thickness. The layers expand and contract to accommodate inflows and outflows. The use of a variable layer scheme has the advantages of improved conservation of mass and reduced numerical dispersion.

The heat budget includes terms for shortwave radiation, longwave radiation, back radiation, evaporative heat loss, conduction, and convection. Each term is computed from meteorological data routinely collected at National Weather Service (NWS) stations.

The model currently has the capability to simulate inflow from two tributaries. Placement of the inflow is determined by comparing the inflow density with the density of each layer. Temperature, as well as suspended and total dissolved solids are considered in calculation of the inflow density. The inflow zone is centered on the layer in the lake with a density closest to the inflow density. If the inflow density is smaller or greater than the density of any layer, the inflow is placed in the surface or bottom layer of the reservoir, respectively. The thickness of the inflow zone is dependent on the inflow rate and the density gradient in the region. Once the inflow zone is established, the inflow is volumetrically distributed among the layers in the zone.

Withdrawals from the reservoir can be made through as many as eight selective withdrawal ports and/or flood gates. The outflow can be specified for each individual port or, if desired, the model will determine outflows and port operations needed to meet a downstream temperature objective. Withdrawal zone computations are based on the experimental work of Bohan and Grace (1973).

The model includes both entrainment and diffusion as mixing processes. Entrainment is a one-way advective process that sharpens gradients. It is the process by which energy supplied by wind shear and convection deepens the upper mixed layer. Vertical diffusion results from the combined effects of inflows, outflows, wind generated currents, turbulence, waves, etc. Diffusion is a two-way dispersive process by which gradients are always reduced. The diffusion coefficient is assumed to be proportional to the rate of dissipation of the energy supplied by the wind, inflows, and outflows.

Mixing Processes

Entrainment - Entrainment is simulated in the model using an integral energy approach which assumes the reservoir is composed of two layers, an upper layer that is well mixed and a lower stable layer of varying density. Turbulent kinetic energy (TKE) supplied at the water surface by wind shear and convective cooling is used to overcome the density gradient in the lower layer and entrain water into the upper layer (or mixed layer). The TKE produced by wind shear (TKEW) is:

$$\mathrm{TKEW} = \int_{A_s} \mathrm{SHELCF} * W_* * \tau * \Delta t * dA \tag{1}$$

in which A_s = surface area; SHELCF = empirical sheltering coefficient; W_* = shear velocity in water; τ = shear stress at air-water interface; and Δt = computational interval. This equation assumes that a certain fraction of TKEW is available for mixing. Since wind does not act on the entire surface of many lakes, the TKEW must be modified by a site-dependent sheltering coefficient (SHELCF) equal to the ratio of the water surface area exposed to the wind to the total surface area.

The TKE available from convective cooling (TKEC) is:

$$\mathrm{TKEC} = -\mathrm{PEFRAC} * Q_n * A_s * \mathrm{ZMIX} * g * \alpha * \Delta t / C_p \tag{2}$$

in which PEFRAC = empirical calibration coefficient; Q_n = net heat flux across the air-water interface; ZMIX = depth of the mixed layer; g = acceleration due to gravity; α = coefficient of thermal expansion for water; and C_p = specific heat of water. TKEC is zero when Q_n is positive. The total TKE potentially available for entrainment is:

$$\mathrm{TKE} = \mathrm{TKEW} + \mathrm{TKEC} \tag{3}$$

Because mixing processes are dissipative and not efficient, the total TKE must be modified to determine the actual TKE available for entrainment. Based on a parameterization of the TKE balance, Bloss and Harleman (1980) formulated an entrainment function based on the Richardson number (Ri):

$$f(Ri) = 0.057 * Ri * \left[(29.46 - \sqrt{Ri}) \; / \; (14.20 + Ri) \right] \tag{4}$$

in which $Ri = g\Delta\rho \; / \; (\rho_w W_*^2)$; $\Delta\rho$ = density difference across the interface; and ρ_w = density of water. The total energy available for mixing is therefore:

$$TKE_a = TKE * f(Ri) \tag{5}$$

After the energy available for entrainment is computed, it is compared to the energy required to entrain the underlying layer (i.e., energy required to lift the underlying layer up to the center of mass of the mixed layer):

$$W_L = \Delta\rho * \Delta V * g * (ZMIX - Z_g) \tag{6}$$

in which $\Delta\rho$ = density difference between the mixed layer and the layer underlying it; ΔV = incremental volume to be entrained; and Z_g = depth of the center of mass of the mixed layer. If the TKE_a is larger than W_L, entrainment occurs and ZMIX increases. The energy values are then recalculated and entrainment continues until TKE_a is no longer larger than W_L.

Diffusion - Diffusion due to wind-generated currents, inflows and outflows is assumed to be proportional to the rate of energy dissipation. Since the TKE generated by the wind (Eq. 1) mixes the surface layer directly through entrainment and mixes the metalimnion and hypolimnion indirectly through seiche motion and breaking of internal waves, this TKEW is assumed to be dissipated throughout the entire reservoir. The rate of energy dissipation per unit volume is:

$$DISW = TKEW/(\rho_w * V * \Delta t) \tag{7}$$

in which V = reservoir volume.

In contrast, diffusion resulting from inflows and outflows is restricted to those layers where flow occurs. The TKE generated by advection (AKE) is:

$$AKE(I) = 1/2 * \rho_w * q(I) * \Delta t \quad \left[q(I) \ / \ (B(I) * Z(I))\right]^2 \tag{8}$$

in which q(I) = flow rate in layer I; B(I) = width of layer I; and Z(I) = thickness of layer I. Since this energy is assumed to be restricted to layer I, the rate of dissipation per unit volume is:

$$DISF(I) = AKE(I)/(\rho_w * \Delta V(I) * \Delta t) \tag{9}$$

in which ΔV(I) = volume of layer I. Since mixing is inhibited by stable density gradients, a buoyancy or stability parameter, N^2, defined by:

$$N^2 = (g/\rho_w) * (\partial\rho/\partial Z) \tag{10}$$

in which $\partial\rho/\partial Z$ = the density gradient, is used to modify the diffusion coefficient. The equation for the diffusion coefficient for interface I is:

$$DC(I) = \left[CDIFW * DISW + CDIFF * (DISF(I) + DISF\ (I-1))/2\right] / (1 + N^2)^r \tag{11}$$

where CDIFW and CDIFF are the calibration coefficients for wind and flow, respectively, and r is the exponent for the stability term.

Study Reservoirs

DeGray Lake and Lake Greeson are CE projects located in the Ouachita Mountains in southwestern Arkansas (Fig. 1). DeGray Lake is a multipurpose (hydropower with pump storage, flood control, and recreation) reservoir on the Caddo River approximately 100 km southwest of Little Rock, AR. Lake Greeson is also a multipurpose project (flood control hydropower, fish and wildlife habitat enhancement, and recreation) on the Little Missouri River about 144 km southwest of Little Rock. While both reservoirs are fairly deep (DeGray - 57 m, Greeson - 51 m), DeGray is a larger reservoir, having a surface area almost twice that of Greeson. Table 1 compares the general characteristics of the two reservoirs.

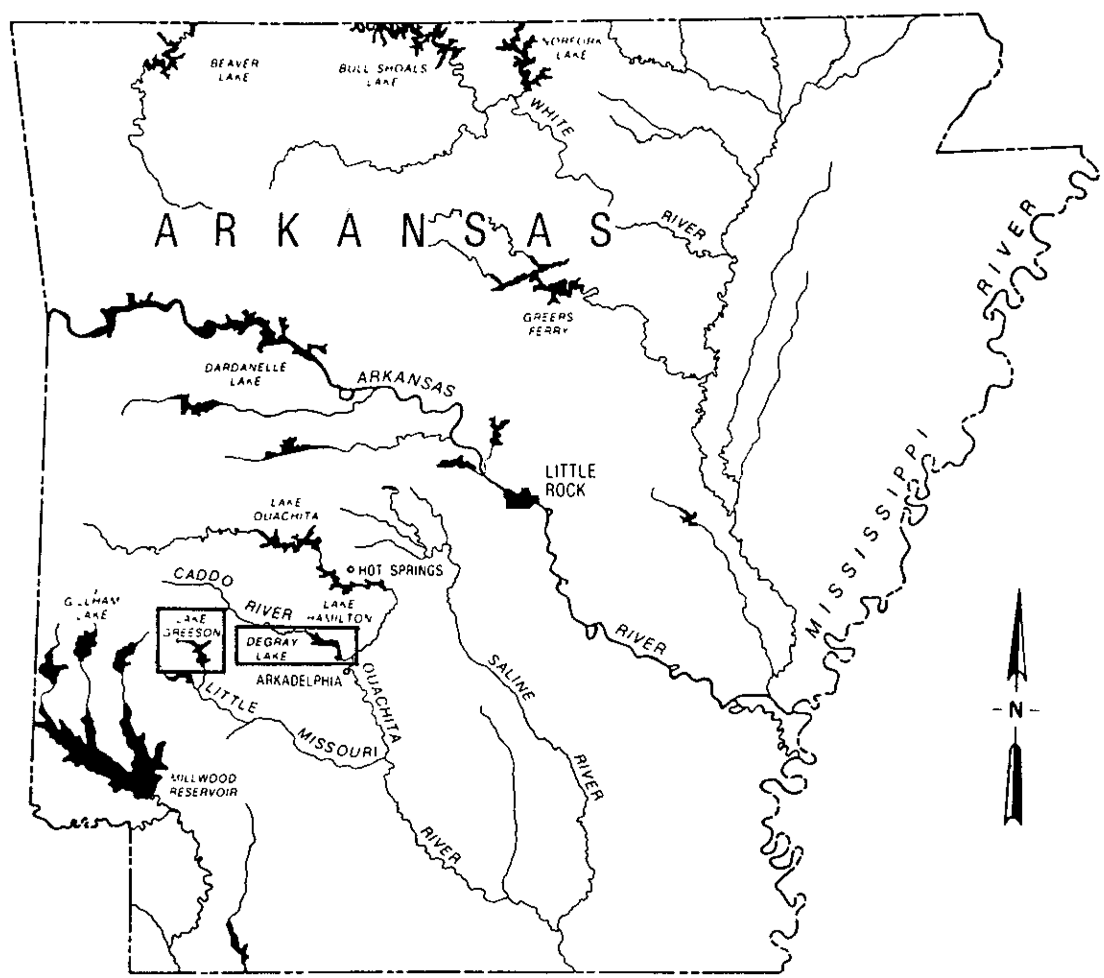


Fig. 1. Location map.

Table 1. Study reservoirs characteristics.

	DeGray	Greeson
Drainage basin, in square kilometers	1173	614
Effective length, in kilometers	32	16
Maximum depth, in meters	57	51
Power pool		
Elevation, meters above mean sea level	124.5	164.4
Volume, in 10^8 cubic meters	7.91	3.45
Surface area, in 10^7 square meters	5.34	2.91
Outlet structure		
Number of ports	3	2
Regulated discharge, in cubic meters per second	170	168

Both reservoirs stratify strongly during part of the year. Although the stratification dynamics vary with reservoir and year, a typical stratification cycle for DeGray Lake is shown in Fig. 2. In DeGray Lake, turnover usually occurs in late January or early February, and the reservoir remains isothermal for approximately one month. Stratification typically begins in March. Stratification starts near the bottom of the lake (Ford and Stefan 1980) with minimum thermocline depth occurring in late June at the time of maximum solar radiation. Thereafter, thermocline depths increase until the surface temperature approaches the hypolimnion temperature and turnover occurs. Hypolimnion temperatures remain relatively constant (6-8°C) throughout the stratification cycle and from year to year. The slope of the isotherms in the upper hypolimnion indicate the amount of mixing that is taking place. Flat isotherms indicate little mixing.

Model Applications

The model was calibrated on DeGray Lake for 1976, then applied to DeGray Lake for 1975 and 1979, and to Lake Greeson for 1973 and 1975. DeGray Lake was operated with surface withdrawal until March 1979 when

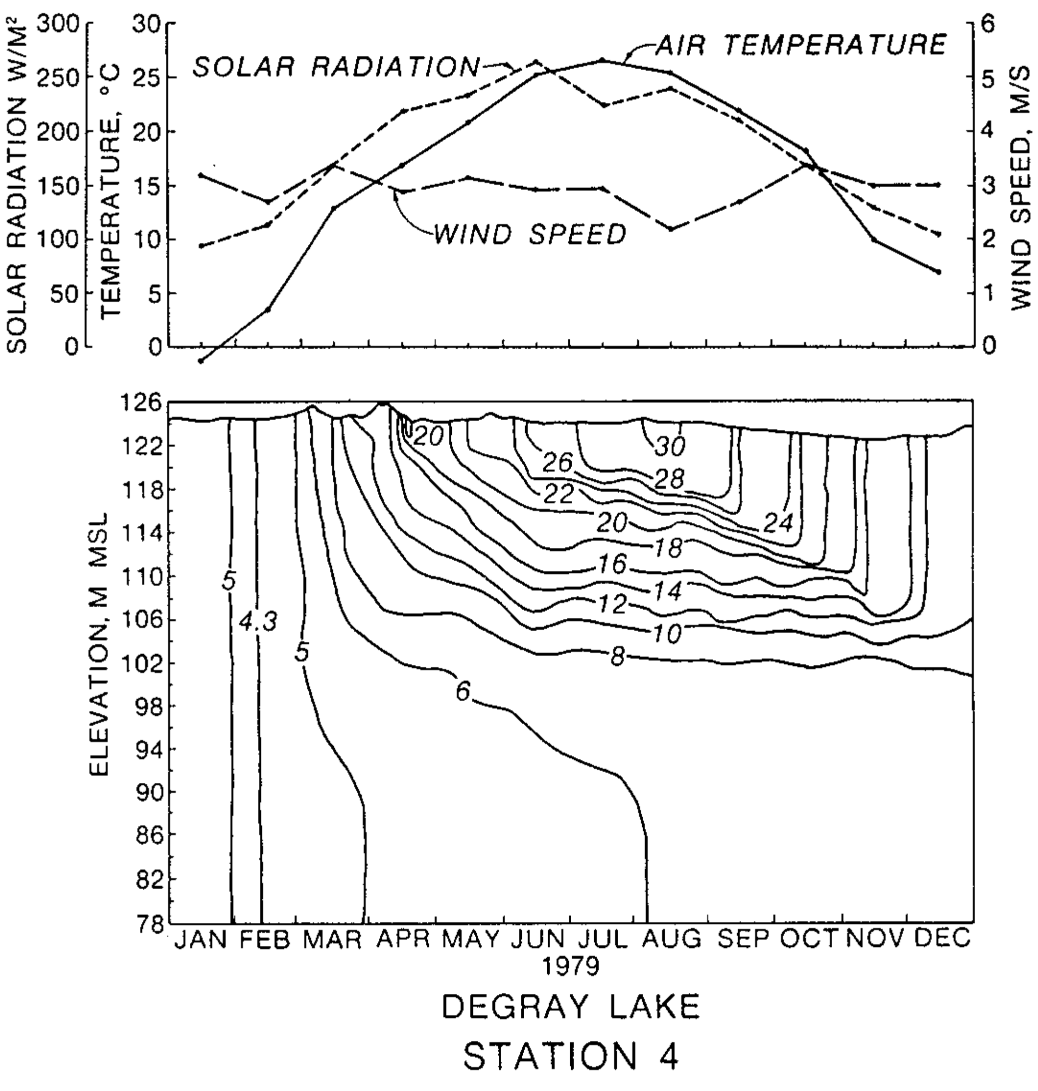


Fig. 2. Typical stratification cycle for DeGray Lake.

the withdrawal depth was changed to bottom withdrawal. Lake Greeson has only bottom withdrawal capability. Predictions were evaluated with respect to the onset of stratification, mixed layer depths, surface temperatures, hypolimnion temperatures, metalimnetic gradient, spring overturn, short-term mixing events, and sharpening of the thermocline gradient. The results from these simulations of DeGray Lake and Lake Greeson will be discussed to illustrate the model's ability to simulate changes in hydrometeorological conditions, changes in withdrawal level, and its ability to simulate a similar but different lake. Since the impact of solids on density is negligible in these two systems, only temperature results will be shown here.

Data Requirements - Morphological, meteorological, and operational data are required by CE-THERM-R1. The morphological data include the physical characteristics of the basin, area-capacity curves, and outlet structure characteristics. The basin characteristics include the reservoir length, width, fetch, etc. This information was obtained from bathymetric maps of each reservoir. The area-capacity curves and outlet structure details were obtained from the design memoranda for each reservoir.

Required meteorological data included dry bulb temperature, dew point temperature, wind speed and direction, cloud cover, barometric pressure, and precipitation. This data was obtained for both lakes from the NWS station at Little Rock, AR. The three-hour observations were averaged to yield mean daily values. Initial comparisons with meterological data collected at DeGray Lake indicated that the data from Little Rock were representative, although temperatures tended to be slightly cooler at Little Rock (Ford and Ford 1983). This may not be true for Lake Greeson, however, since it is located farther from Little Rock.

Operational data included inflows, inflow density, and outflows. Outflows and port operations for both reservoirs were obtained from the CE daily operation records. For DeGray Lake, inflows were obtained from the gage at Highway 84 and modified to account for the ungaged portion of the watershed (Ford and Ford 1983). Inflow temperatures were recorded hourly at Highway 84 and averaged over a daily period. These inflow temperatures were used to calculate the density of the inflow. For Lake

Greeson, inflows were obtained from the daily operation records (i.e., back-calculated from the outflows and pool elevation) and inflow temperatures were generated using a regression technique with air temperature (Stafford 1978).

Model Calibration - The model was calibrated on DeGray Lake 1976 by adjusting five mixing coefficients to match simulated temperature profiles with measured profiles. This year was selected because it represented a 'normal' year. The mean annual precipitation and wind speed were near the long-term mean while the mean annual air temperature was 1.1°C cooler than the long-term average. The 1976 mean monthly variations compared with the long-term mean and standard deviation are in Fig. 3. An isotherm plot for DeGray 1976 is shown in Fig. 4.

Simulations were started on Julian Day 54 (23 Feb) with a measured temperature profile. Initially, the model was calibrated by adjusting the sheltering coefficient (SHELCF) to match mixed layer depths during periods of wind mixing and adjusting the convective mixing coefficient (PEFRAC) to match mixed layer depths during periods of cooling. During these initial simulations, the diffusion coefficients in the metalimnion and hypolimnion were set to molecular values (i.e., CDIFW and CDIFF = 0). After SHELCF and PEFRAC were determined, the diffusion coefficients were adjusted to match hypolimnetic and metalimnetic temperatures. CDIFW (Eq. 11) was adjusted to account for wind mixing events and CDIFF (Eq. 11) to account for periods of large inflows and outflows. The stability parameter, r, (Eq. 11) was used to achieve the correct gradient in the metalimnion. The calibrated coefficients were SHELCF = 1.0, PEFRAC = 0.0, CDIFW = 0.0001, CDIFF = 0.010, and r = 0.5.

The calibration results are shown in Fig. 5. The onset of stratification (Day 78, 18 Mar) was simulated correctly although the predicted mixed layer depth was too deep. Surface temperatures were within the measurement error and diurnal variations ($\pm$ 1°C). Mixed layer depth predictions were in excellent agreement with observed mixed layer depths except on Days 78 and 230 (18 Mar and 17 Aug); on these two days the predicted layer depth was too deep. This may be attributable to wind setup and seiche motion. The metalimnetic gradient and hypolimnetic temperature predictions also matched the actual conditions, except for a small discrepancy at approximately Elev. 40 m. This resulted from too

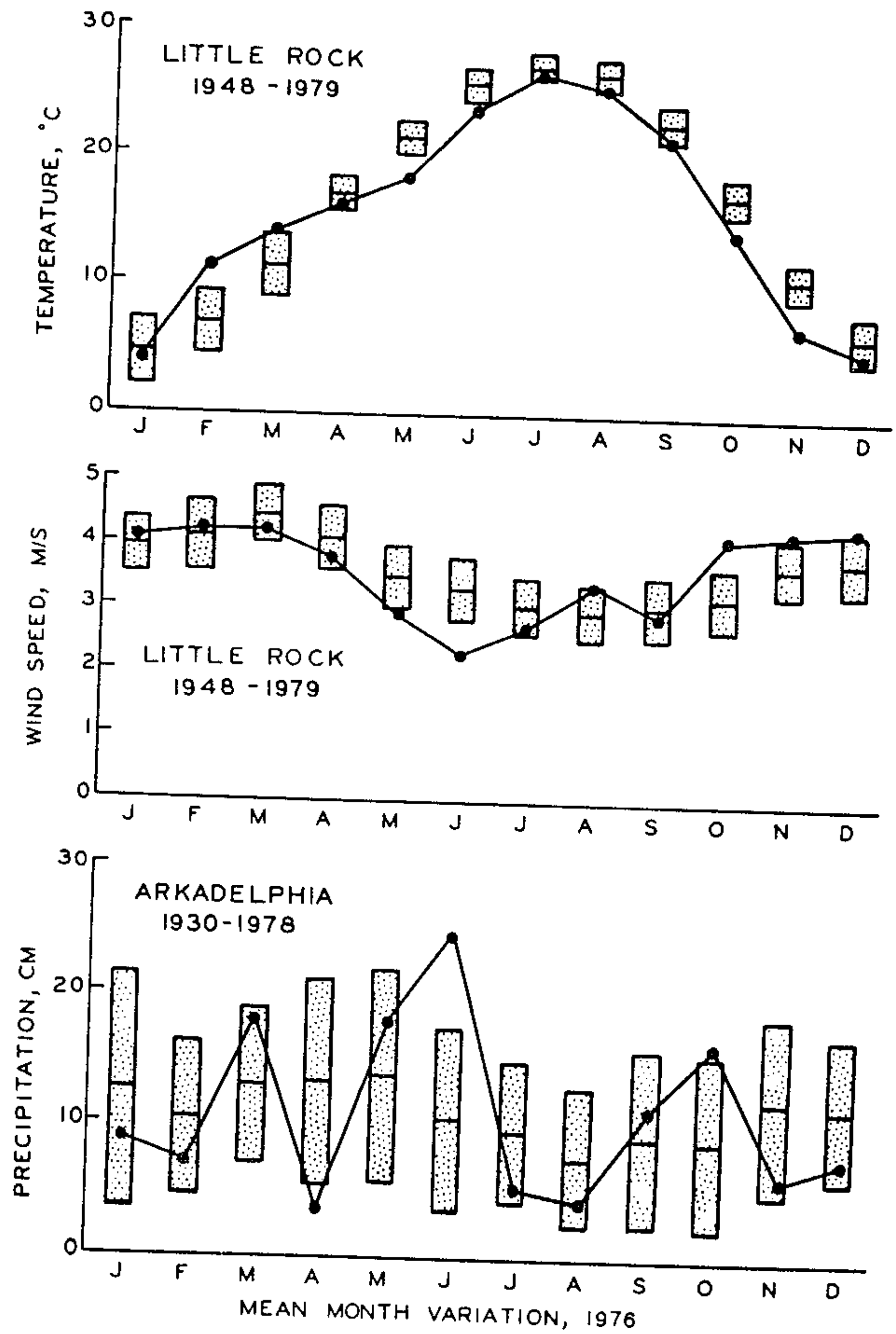
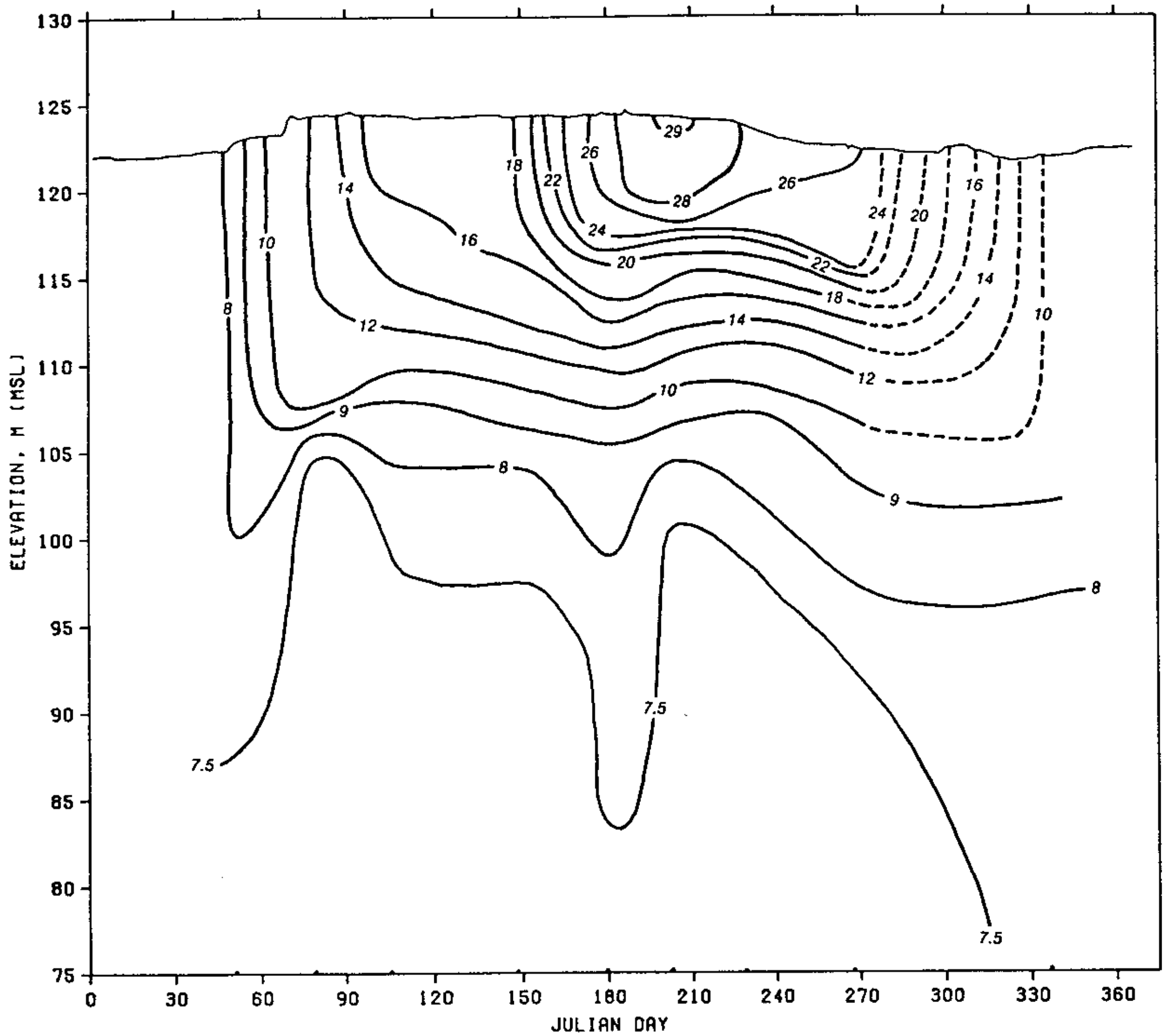


Fig. 3. Mean monthly variation of temperature, wind speed, and precipitation compared to period of record mean and standard deviation, 1976.



DE GRAY LAKE 1976

Fig. 4. Isotherms for DeGray Lake, 1976.

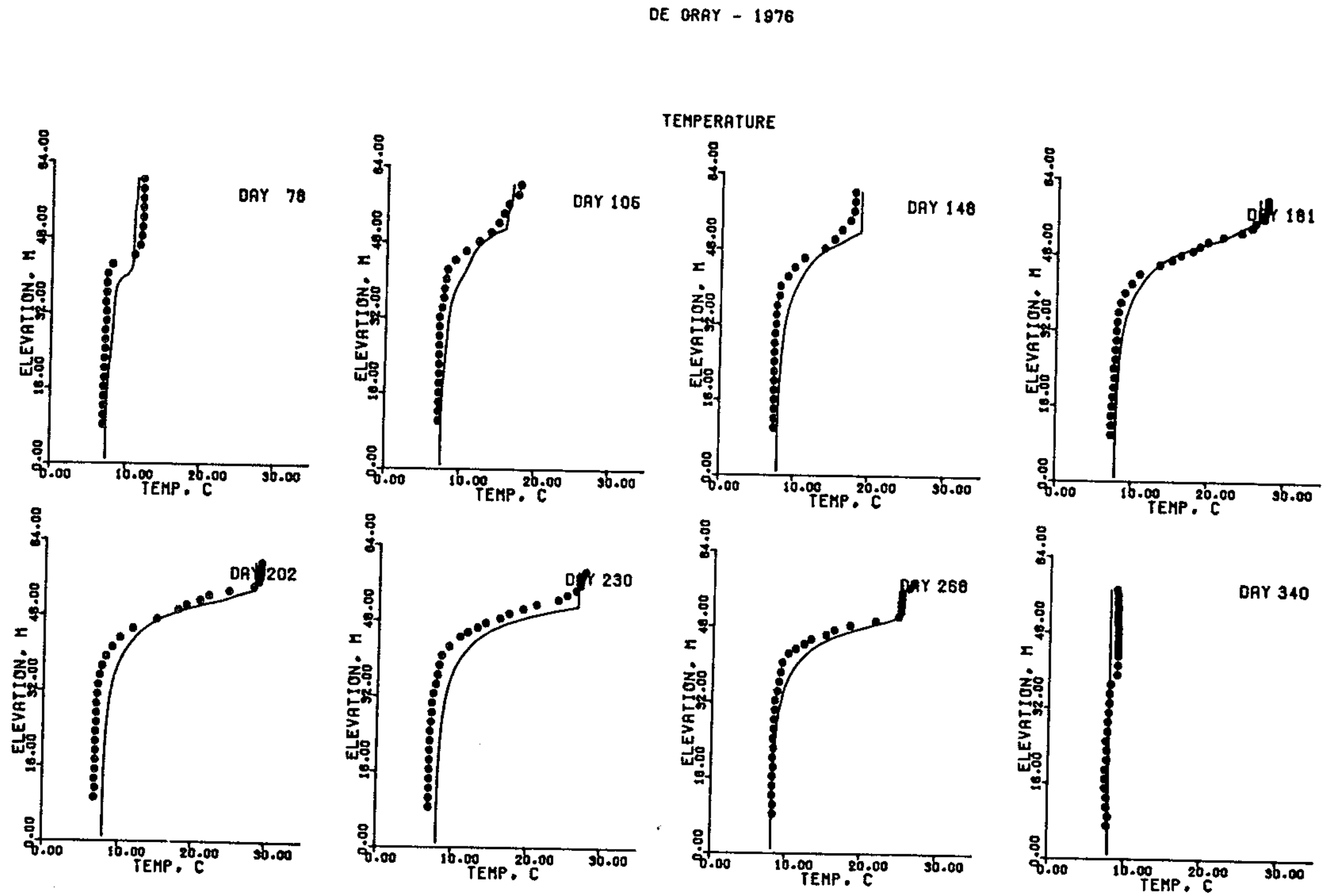


Fig. 5. Calibration results for DeGray Lake, 1976 (• actual data; —— simulated data).

much mixing in the spring and remained throughout the simulation. When the simulations were started later in the year with the measured profile on Day 78 (18 Mar), this discrepancy did not occur and results were improved.

DeGray 1975 - DeGray 1975 was selected to verify the model's ability to simulate changes in hydrometeorologic conditions; 1975 was warmer, had greater runoff, and less wind than 1976 (Fig. 6). In 1975, the lake did not heat up as quickly as in 1976 and hypolimnetic temperatures were slightly warmer (Fig. 7). To simulate 1975, only the meteorological, operational, and update data were changed to reflect conditions during 1975. None of the calibrated coefficients used in the 1976 simulation were changed.

As in the 1976 simulation, the model predictions were good (Fig. 8). The onset of stratification, surface temperatures, hypolimnetic temperatures, metalimnetic gradient, and mixed layer depths were all predicted. Surface temperature predictions on Day 247 (4 Sep) were not as good as in 1976. The mixed layer depth was also slightly off after Day 247 (4 Sep).

DeGray 1979 - In March of 1979, the release level of DeGray was dropped 12 m to evaluate the effects of release level on the water quality and fisheries in the lake and downstream. 1979 was also cooler, had significantly more runoff, and had slower wind speeds than either 1976 or 1975 (Fig. 9). 1979, therefore, was an excellent opportunity to test the model. Again, only meteorological, operational, and update data were changed to model DeGray Lake during 1979. Isotherms for DeGray 1979 are shown in Fig. 10. Model predictions for 1979 (Fig. 11) were similar to 1975 and 1976 in accuracy. Because of the change in withdrawal depth, a deeper mixed layer and weaker metalimnetic density gradient were observed and predicted. Discrepancies in model predictions occurred in surface temperature on Days 261 and 303 (18 Sep and 30 Oct) and metalimnetic gradient on Days 219 and 261 (7 Aug and 18 Sep). Errors in the surface temperatures may be due to differences in climatic conditions over the lake compared to Little Rock and/or the time of day the actual data was taken. Since both DeGray and Greeson are located in the Ouachita Mountains and Little Rock is not, differences are expected. With the change in release level, more mixing occurred in the metalimnion. This was predicted, as evidenced by the change in slope of the

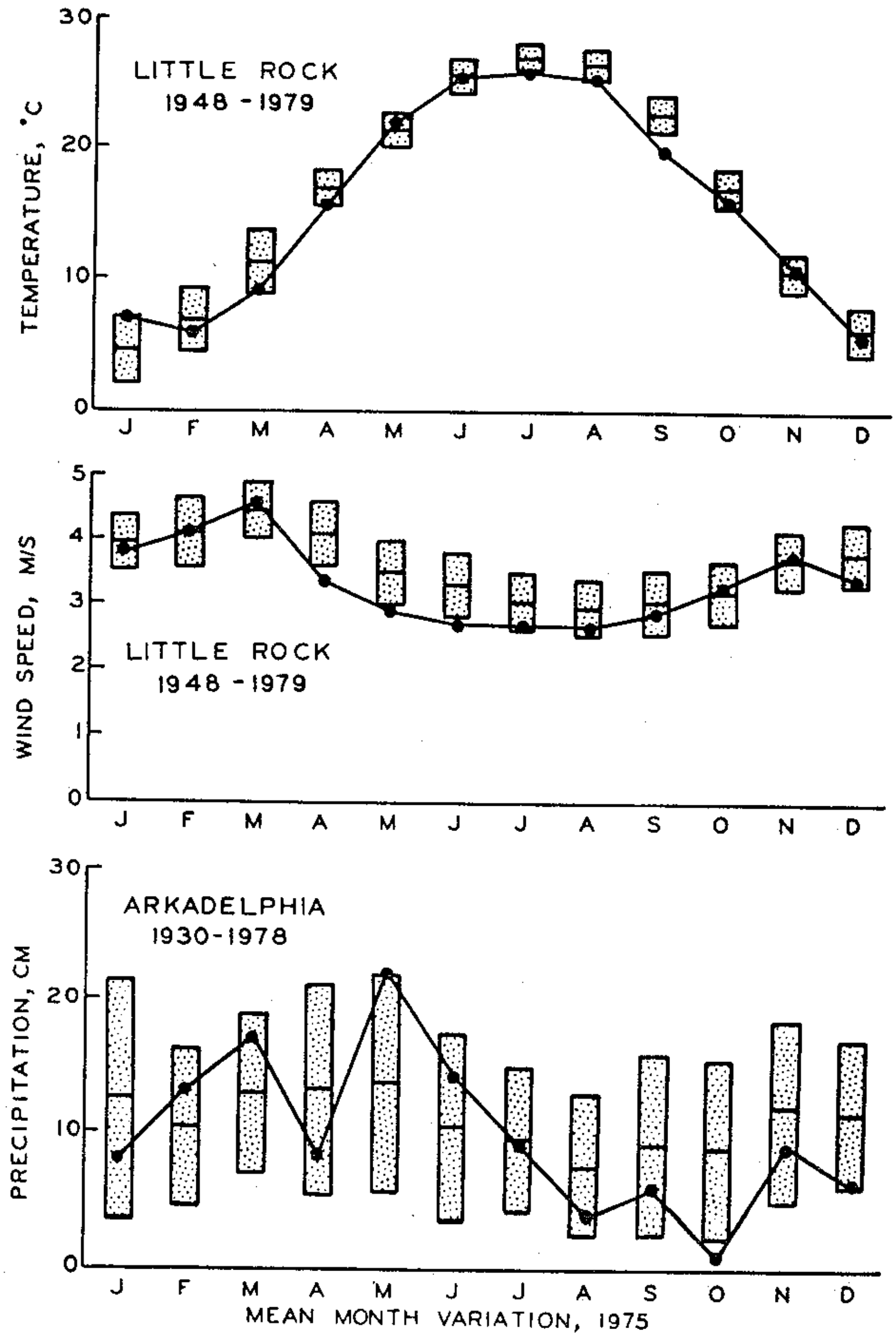
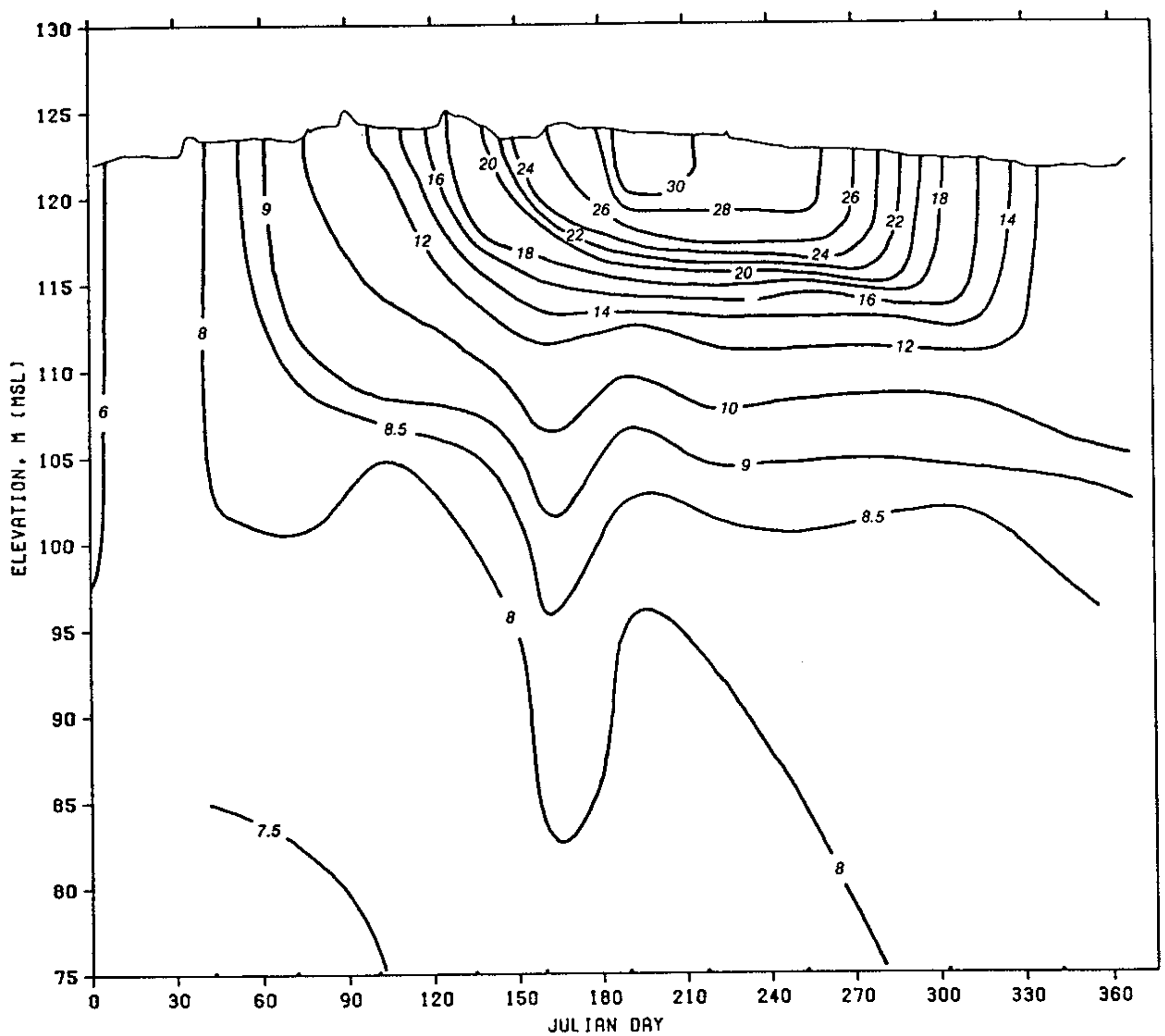


Fig. 6. Mean monthly variation of temperature, wind speed, and precipitation compared to period of record mean and standard deviation, 1975.



DE GRAY LAKE 1975

Fig. 7. Isotherms for DeGray Lake, 1975.

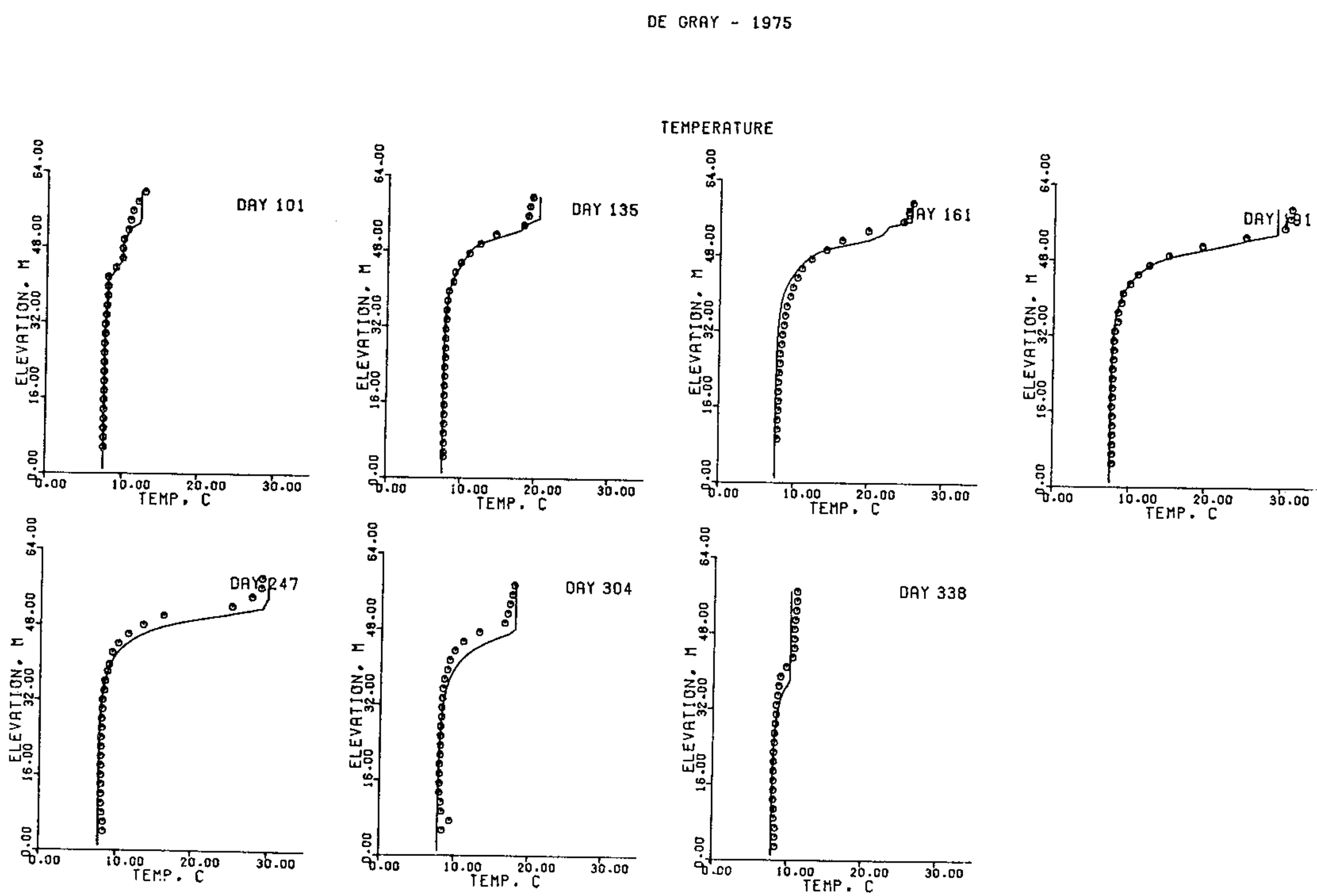


Fig. 8. Simulation results for DeGray Lake, 1975 (• actual data; —— simulated data).

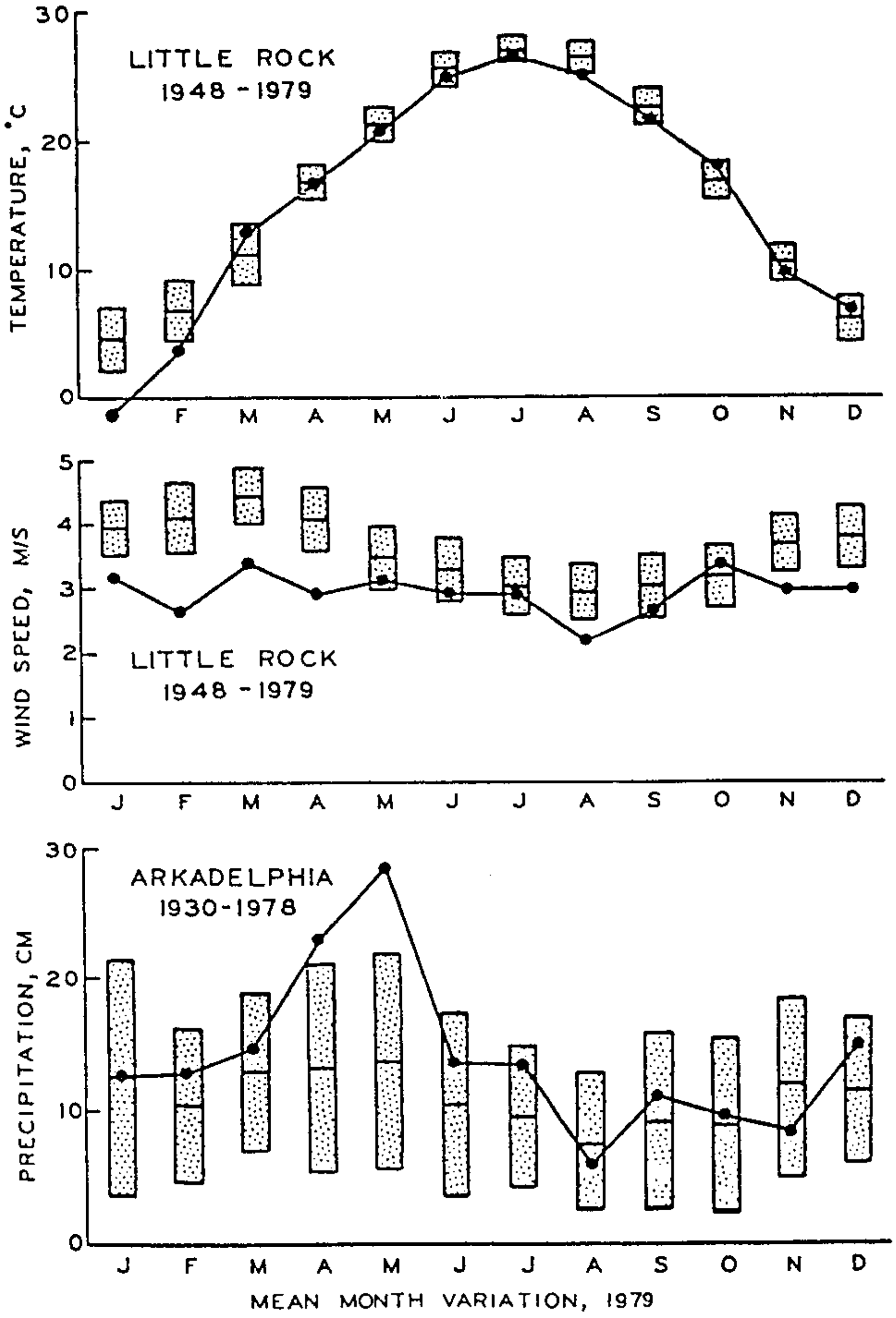


Fig. 9. Mean monthly variation of temperature, wind speed, and precipitation compared to period of record mean and standard deviation, 1979.

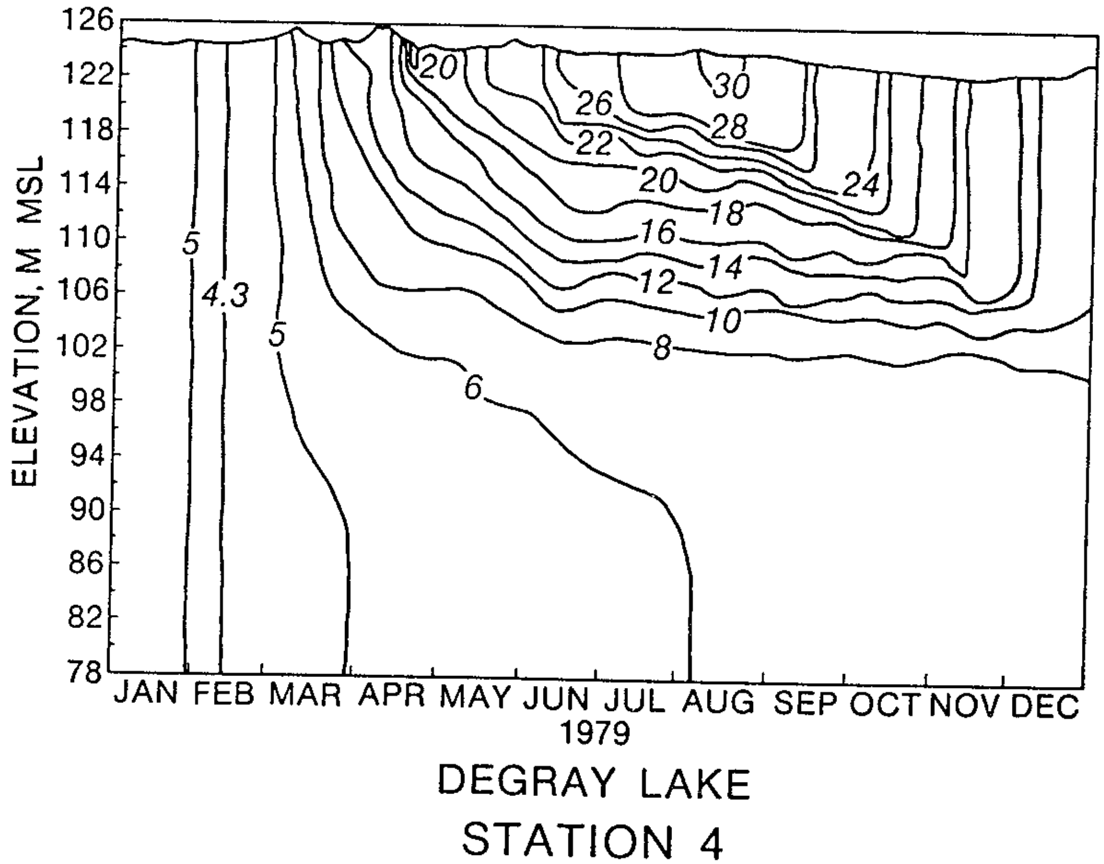


Fig. 10. Isotherms for DeGray Lake, 1979.

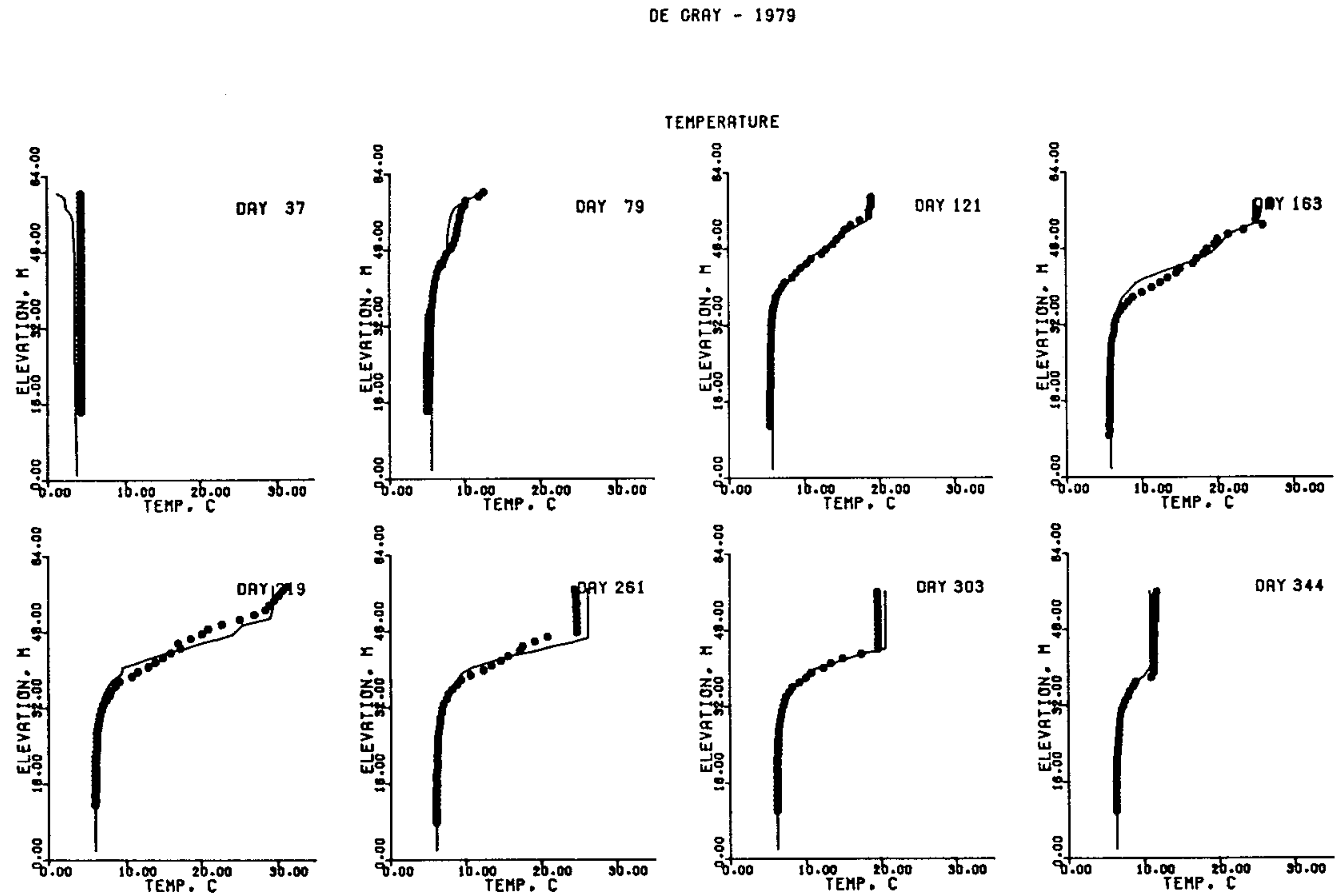


Fig. 11. Simulation results for DeGray Lake, 1979 (• actual data; ——— simulated data).

metalimnetic temperature gradient.

Greeson 1975 - Greeson 1975 was selected to compare with DeGray 1975 to evaluate the model's ability to simulate a different lake that is similar in morphometry and location. Site-specific characteristics such as area-capacity curve, outlet structure details, inflows and outflows, and extinction coefficient were the only data changed to model Lake Greeson. The mixing coefficients and meteorologic forcing functions were not changed from those values used for DeGray.

As shown in Figs. 7 and 12, the thermal structure for DeGray and Greeson are different. Model simulations for Lake Greeson for 1975 are shown in Fig. 13. On several days (e.g., Day 209, 28 Jul), the simulations were not as accurate as the DeGray simulations, but they were acceptable. The onset of stratification was simulated correctly, including the double thermocline on Day 125 (5 May). The greatest discrepancy occurred on Day 209 (28 Jul). After careful analysis of the data, it was not clear what mixing process could have caused the thermocline to rise as observed. The model was unable to simulate this phenomenon. In general, the most consistent problem with these simulations was matching the surface temperature on Days 209, 251, and 286 (28 Jul, 8 Sep, and 13 Oct). This can, again, probably be attributed to using meteorological data from Little Rock.

Greeson 1973 - Lake Greeson for 1973 was selected to evaluate changes in hydrometeorology and withdrawal depth. In Arkansas, 1973 was a very wet year. The differences between the thermal structure in 1973 and 1975 (Figs. 12 and 14) clearly indicate the impact of increased flow on mixing; the metalimnetic gradient was not as sharp as in 1975.

Predicted temperature profiles for Lake Greeson in 1973 are shown in Fig. 15. Considering the model was calibrated on another lake for another year and that 1973 was unusually wet, the simulation results are excellent. With a few discrepancies, all major features were predicted accurately. Discrepancies occurred in the hypolimnetic temperature on Day 95 (5 Apr), metalimnetic gradient on Day 207 (26 Jul), mixed layer depth on Days 95 and 334 (5 Apr and 30 Nov), and surface temperatures on Days 225, 292, and 334 (12 Sep, 19 Oct, and 30 Nov).

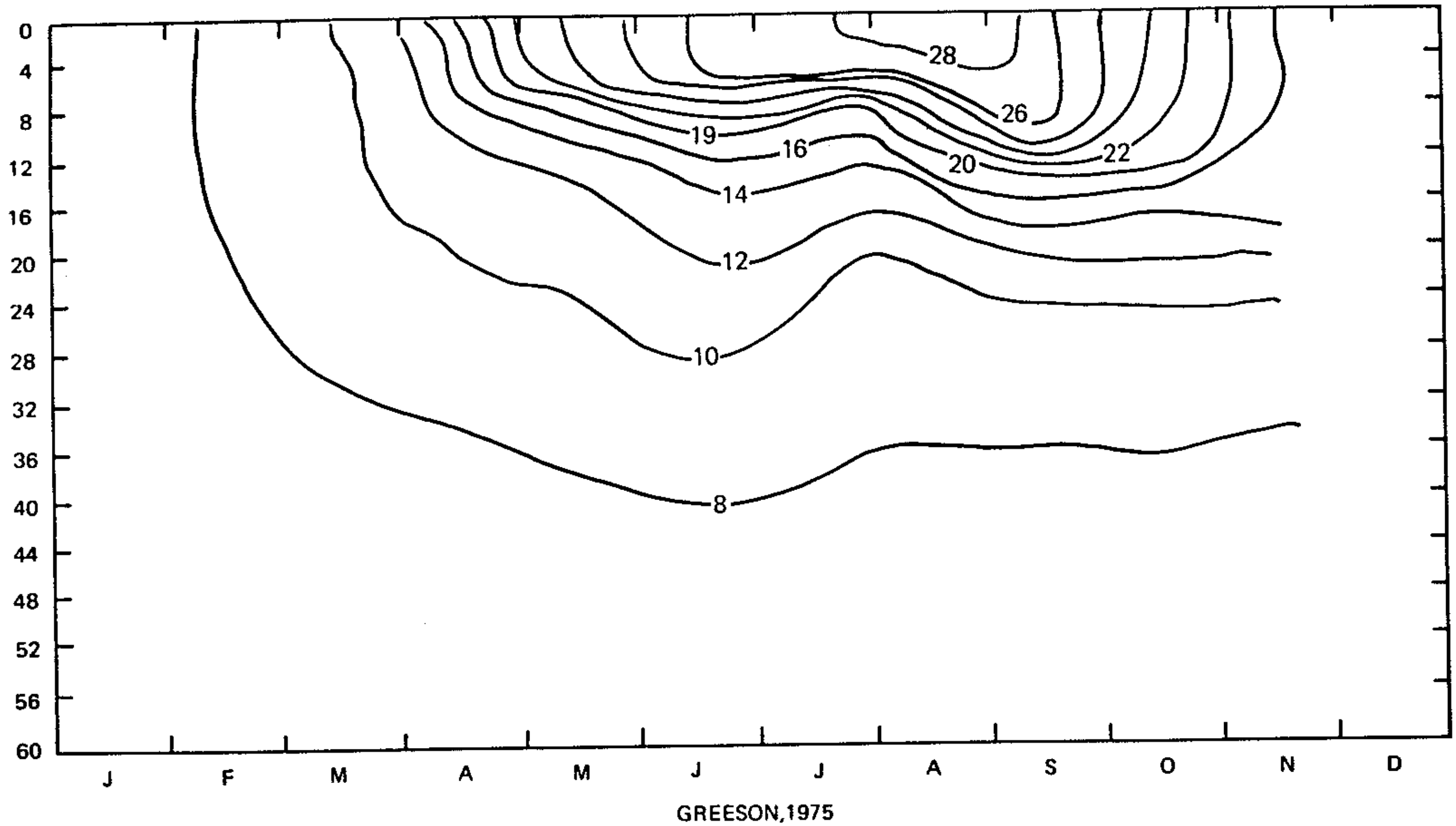


Fig. 12. Isotherms for Lake Greeson, 1975.

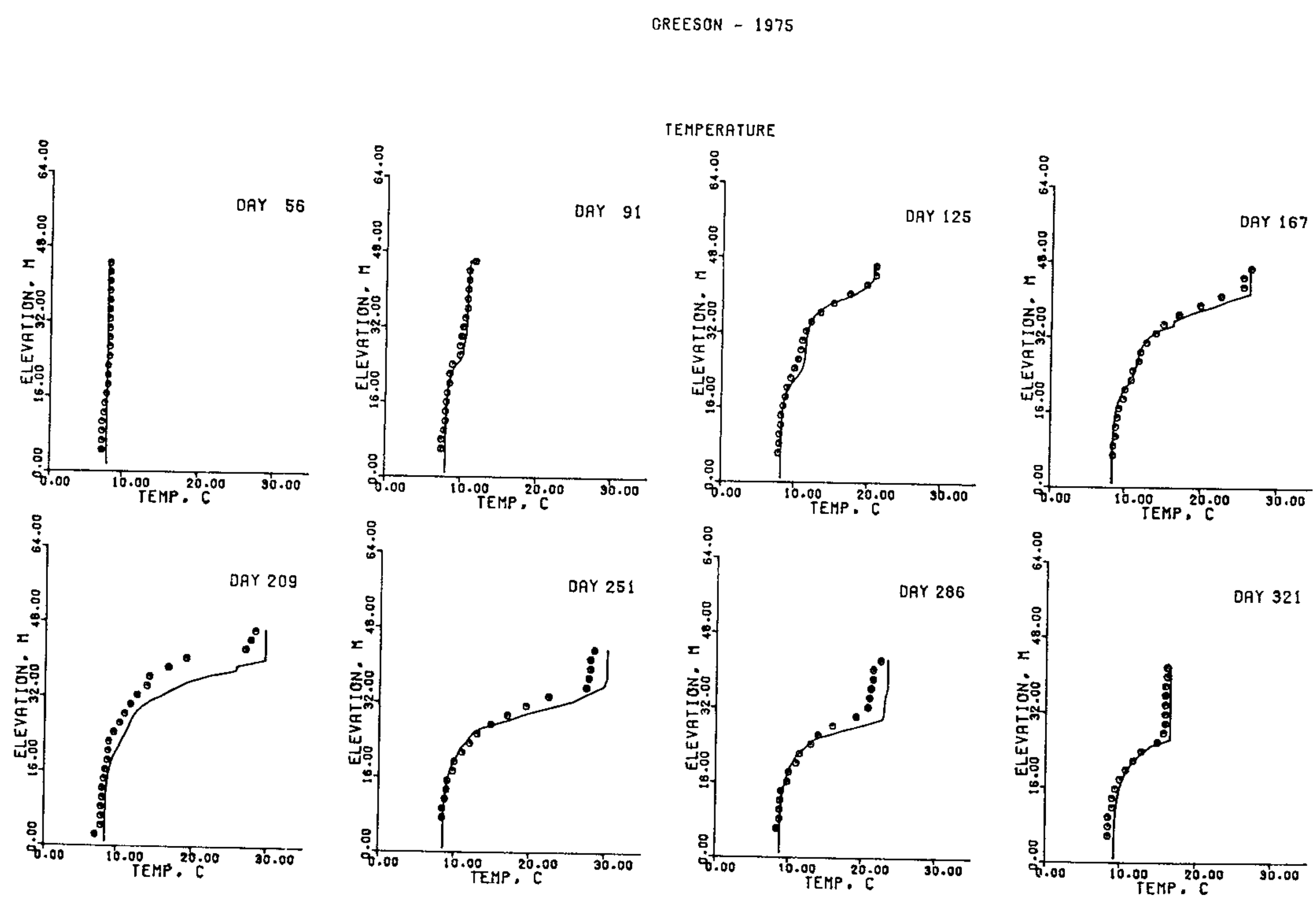


Fig. 13. Simulation results for Lake Greeson, 1975 (• actual data; ——— simulated data).

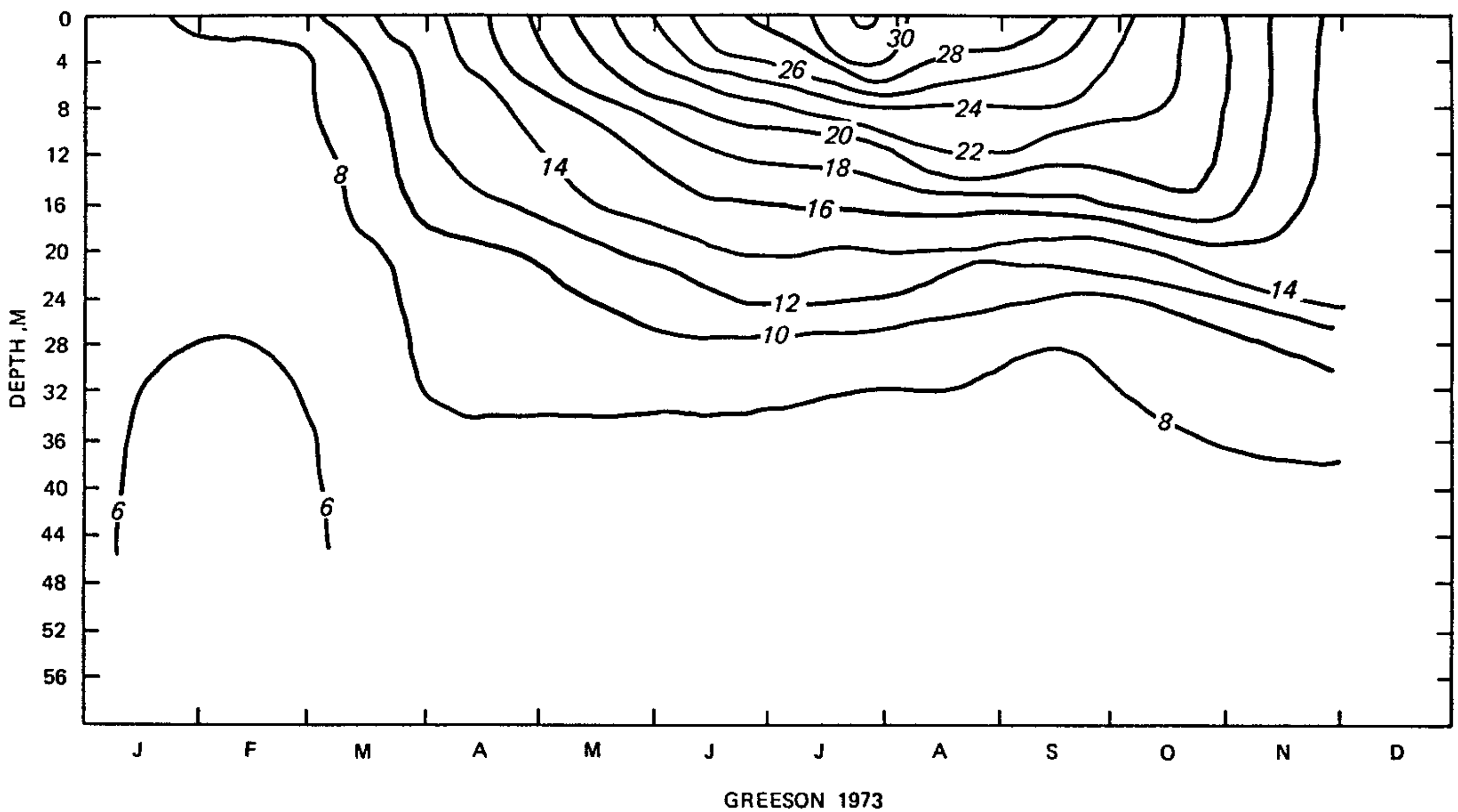


Fig. 14. Isotherms for Lake Greeson, 1973.

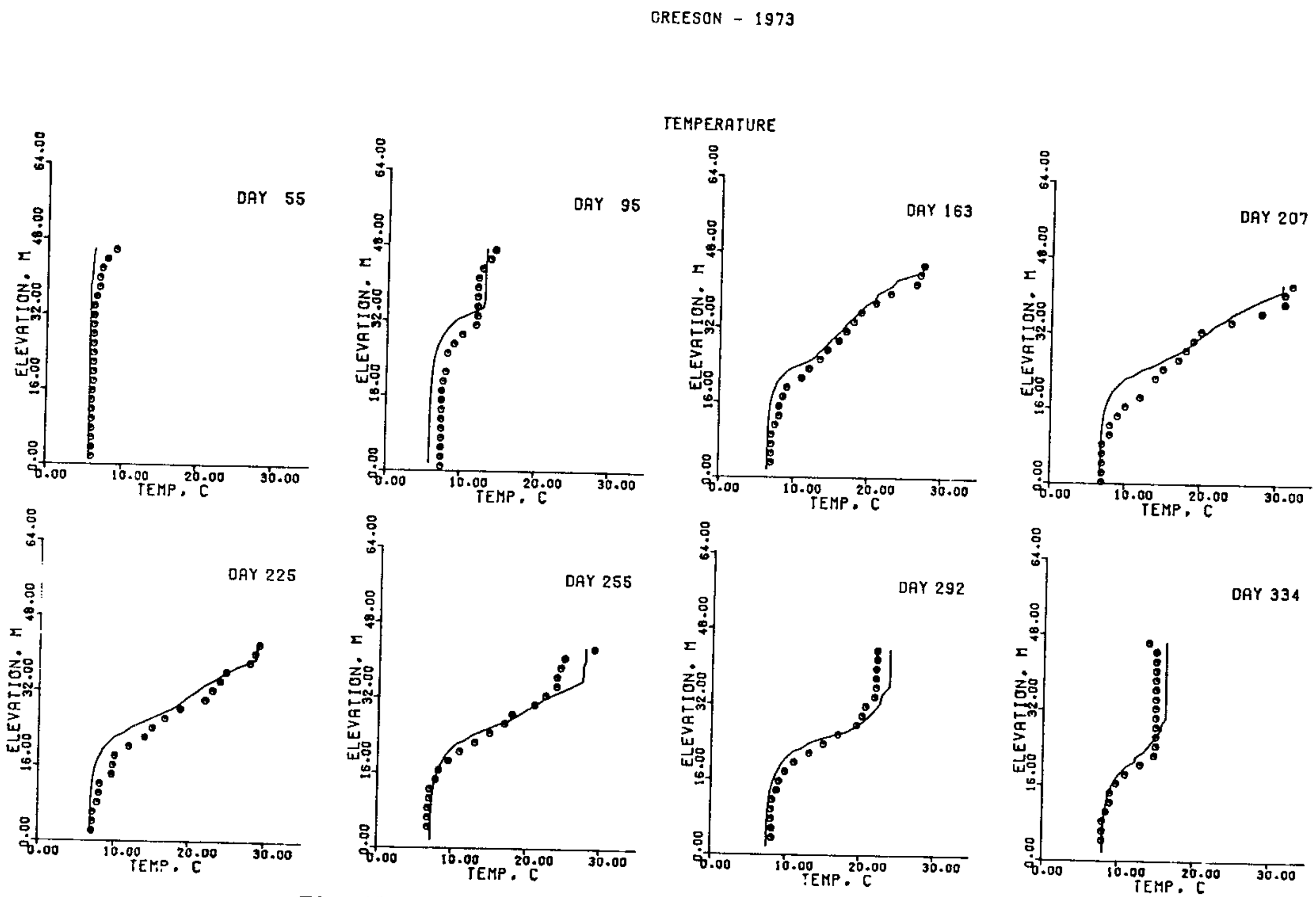


Fig. 15. Simulation results for Lake Greeson, 1973 (• actual data; —— simulated data).

Summary

A one-dimensional thermal reservoir model, CE-THERM-R1, was described and verified. Model algorithms were formulated to realistically simulate mixing dynamics in reservoirs. Model verfication was achieved by calibrating on one data set and verifying against four independent data sets which encompass changes in hydrometeorological conditions, operating strategies, and different reservoirs. During verification, emphasis was placed on the model's ability to simulate observed temperature dynamics. In all situations, the onset of stratification, overturn, mixed layer depths, surface temperatures, metalimnetic density gradients and hypolimnetic temperatures were accurately predicted.

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Notation

The following symbols are used in this paper:

- A_s = surface area
- AKE(I) = turbulent kinetic energy produced by advection in layer I
- B(I) = width of layer I
- C_p = specific heat of water
- CDIFF = calibration coefficient for flow
- CDIFW = calibration coefficient for wind
- DC(I) = diffusion coefficient across interface I
- DISF(I) = rate of dissipation of the energy supplied by the flow in layer I
- DISW = rate of dissipation of the energy supplied by the wind
- dA = incremental area
- g = acceleration due to gravity
- N^2 = buoyancy or stability parameter
- PEFRAC = calibration coefficient for convective cooling
- Q_n = net heat flux across the air-water interface
- q(I) = flow rate in layer I
- Ri = Richardson number
- r = exponent on stability term
- SHELCF = site-dependent sheltering coefficient
- TKE = turbulent kinetic energy
- TKEC = turbulent kinetic energy produced by convective cooling
- TKEW = turbulent kinetic energy produced by wind shear
- Δt = time step
- V = reservoir volume
- ΔV = incremental volume to be entrained
- $\Delta V(I)$ = volume in layer I
- W_* = shear velocity in water
- W_L = amount of energy required to entrain a layer
- Z(I) = thickness of layer I
- Z_g = depth of center of mass of the mixed layer
- ZMIX = depth of the mixed layer

α	= coefficient of thermal expansion for water
ρ_w	= density of water
$\Delta\rho$	= density difference between two layers
$\partial\rho/\partial Z$	= vertical density gradient
τ	= shear stress at the air-water interface

Two-dimensional modeling of storm event inflow processes at DeGray Lake, Arkansas[1]

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Abstract

Storm flows contribute large amounts of nutrients, bacteria, and suspended sediment to a reservoir. Often these flows enter the pool as density currents which are difficult to analyze because of their unsteady and three-dimensional nature.

The two-dimensional reservoir hydrodynamic model LARM (Laterally Averaged Reservoir Model) was used to analyze inflow processes following a storm event at DeGray Lake, Arkansas. Examined were the dilution, mixing, travel times, and placement of four water quality constituent plumes. Since each constituent loaded at different times during the inflow hydrograph, differences in transport were observed. LARM was able to correctly predict inflow placement, travel times, and dilutions of each constituent plume.

Storm events and associated nonpoint source loadings of suspended solids, nutrients, and bacteria can have major impacts on reservoir water quality. Recent studies have shown that significantly more material enters a lake during storm events than during base flow conditions (Perrier et al. 1977 and Westerdahl et al. 1981). During storms, certain constituents characteristically load on the rising side of the hydrograph (e.g., phosphorus, coliform bacteria, and turbidity) while others load on

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[1] This study was supported by the U.S. Army Corps of Engineers, Office, Chief of Engineers, Environmental Impact Research Program, and Environmental and Water Quality Operational Studies. The field data were collected by Dr. Joe F. Nix and his staff at Ouachita Baptist University, Arkadelphia, AR, under contract with the U.S. Army Engineer Waterways Experiment Station.

the falling limb (e.g., nitrate and many metals) (Perrier et al. 1977 and Westerdahl et al. 1981). The ultimate fate of a constituent will depend not only on how it loads on the hydrograph but also on how it moves through the reservoir.

During the past eight years, storm event loadings to DeGray Lake, Arkansas, have been measured under varying hydrometeorological conditions and have also been followed through the reservoir with turbidity measurements (Ford et al. 1980). Even though the average theoretical residence time of DeGray Lake is 1.4 years, turbid storm flows have been observed to move through the entire reservoir as density currents in less than eight days. In other instances, storm flows do not pass completely through the reservoir but stall and collapse. Analyzing data from these storms is difficult because of their unsteady and three-dimensional characteristics (Ford and Johnson 1983).

The objective of this investigation was to analyze the movement of a storm flow and its associated load through DeGray Lake using the two-dimensional reservoir hydrodynamic model, LARM (Laterally Averaged Reservoir Model).

Background

DeGray Lake is a U.S. Army Corps of Engineers multipurpose reservoir located on the Caddo River in south-central Arkansas (Fig. 1). The source of the Caddo River is in the Ouachita Mountains: the 1193-km^2 basin controlled by DeGray Dam is mainly forested with some noncultivated agriculture. The 7.91×10^8 m^3 lake is 32 km long, has a surface area of 53.4 km^2, a maximum depth of 57 m, and an average theoretical hydraulic residence time of approximately 1.4 years. Specific project purposes include hydropower with pumped storage capabilities, flood control, and recreation. The outlet structure is capable of multilevel discharge of both base and flood flows.

In 1972, DeGray Lake and the Caddo River drainage basin were selected as field prototypes to investigate impacts of reservoir operation and management on reservoir water quality, and to collect data necessary to verify water quality models. Since then, the Waterways Experiment Station, in cooperation with the U.S. Fish and Wildlife Service and Ouachita Baptist University, has been routinely collecting

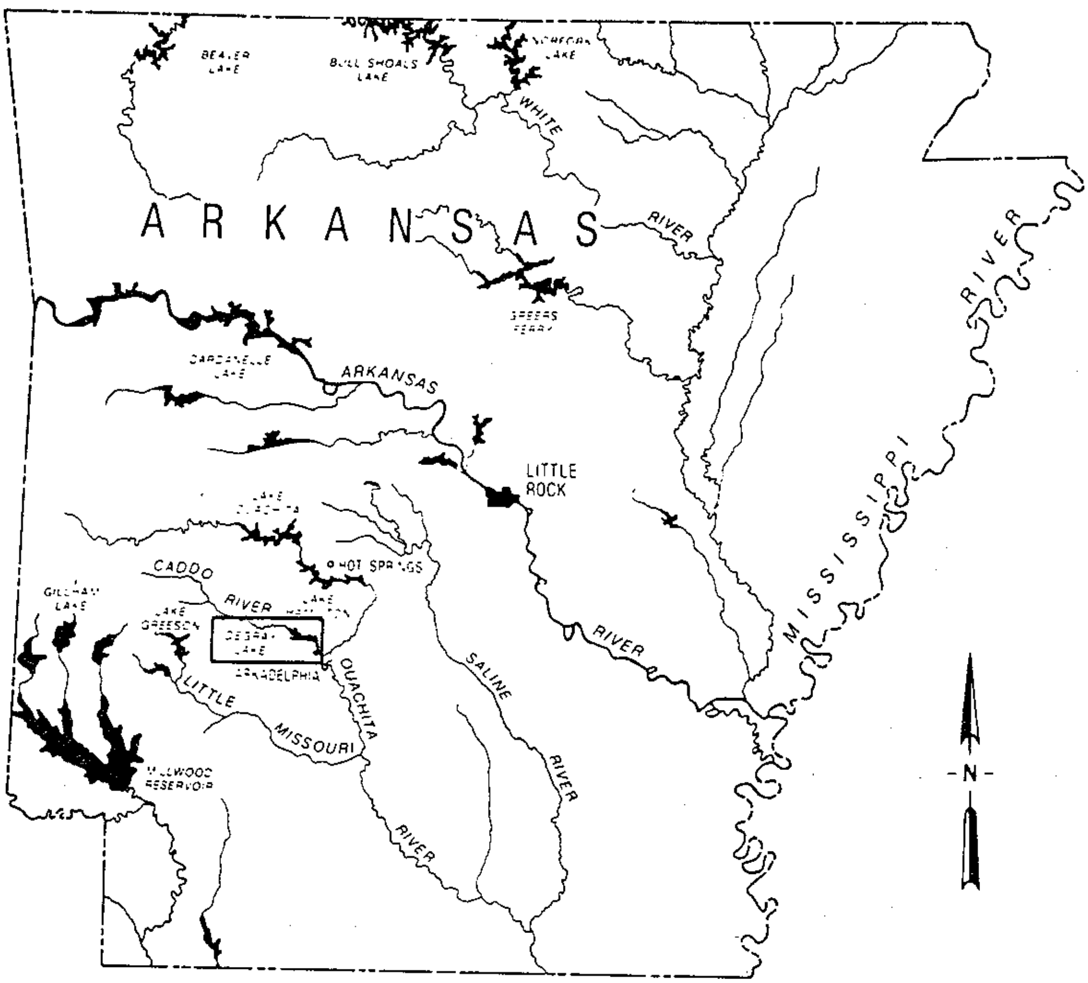

Fig. 1. Location of DeGray Lake.

water quality data in the inflow, outflow, and at Stations 1, 10, and 12 within the pool (Fig. 2). In addition, a number of specific studies have been undertaken to quantify storm loadings to the reservoir (Perrier et al. 1977, Thornton et al. 1980, and Westerdahl et al. 1981), to determine how storm inflows pass through the reservoir (Ford et al. 1980, Ford and Johnson 1980, and Thornton et al. 1980), and to quantify temporal and spatial variations in reservoir water quality (Thornton et al. 1982).

On 25 May 1977, a thirty-day intensive study was undertaken to investigate reservoir dynamics on a daily time scale. Temperature, dissolved oxygen, specific conductance, pH, chlorophyll a, and coliform bacteria were measured daily at Stations 1, 10, and 12. A major storm occurred near the end of the study.

On 13 June approximately 2.5 cm of rain fell on the portion of the basin immediately surrounding the lake. The Caddo River flow at the Highway 84 gaging station increased from 0.5 m^3/s on 12 June to a peak flow of 6.9 m^3/s at 2000 hours on 13 June. Little additional rain fell until 16 June when approximately 7.7 cm of rain fell over the watershed. The Caddo River stage at Highway 84 rose 3.7 m as a result. Flows increased from approximately 5 m^3/s at 0400 hours on 17 June to a peak of 370 m^3/s at 1200 hours, while stream temperatures dropped from from 24°C to 19.6°C at the peak of the hydrograph (Fig. 3).

The reservoir pool elevation rose 0.48 m following the storm. Water surface temperatures dropped 1.4°C at Station 12. Baseline sampling in the pool at approximately 0900 on 16 June revealed turbidity increases in the headwater (Fig. 4). This increase was probably due to the runoff from the small storm of 13 June since runoff from the large storm of 16 June did not affect the Caddo River flow until 17 June as evidenced by the hydrograph in Fig. 3. By 18 June, an interflow was well established. The flow plunged near Station 13 and became an interflow at a depth where the temperature was near 20°C. Since this temperature is similar to the stream temperatures recorded during the peak flow, and since the contribution of solids to the density difference is negligible, there appeared to be little mixing or dilution of the density current.

On 20 June, the flow in the Caddo dropped to less than 10 m^3/s

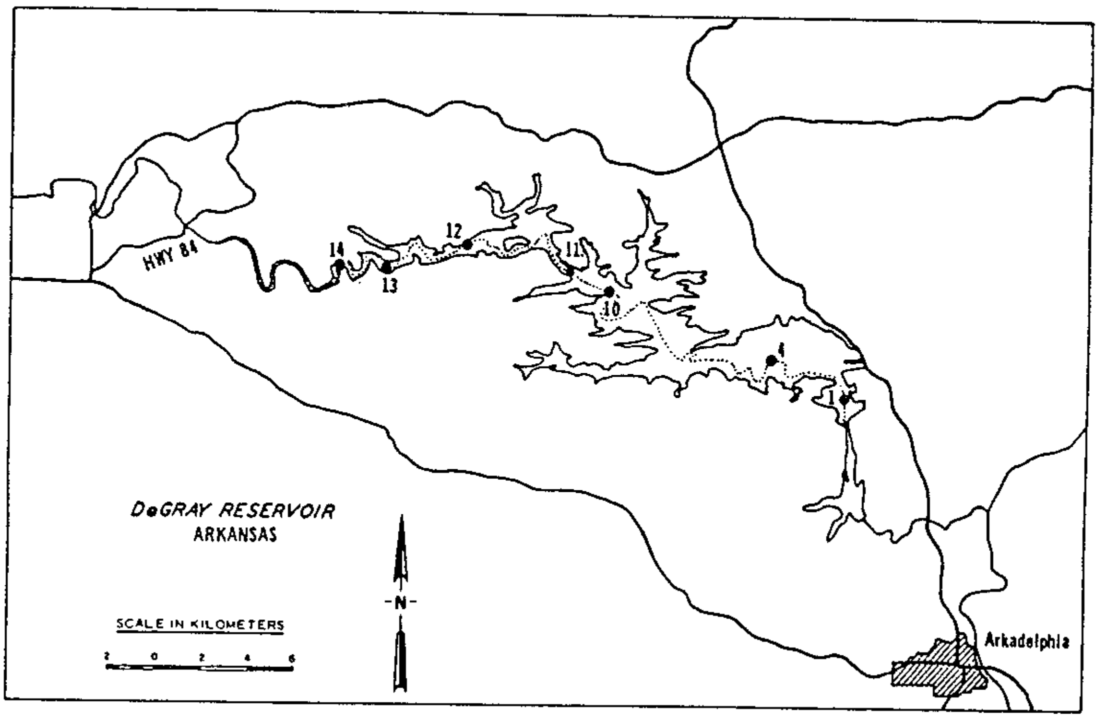

Fig. 2. DeGray Lake sampling stations.

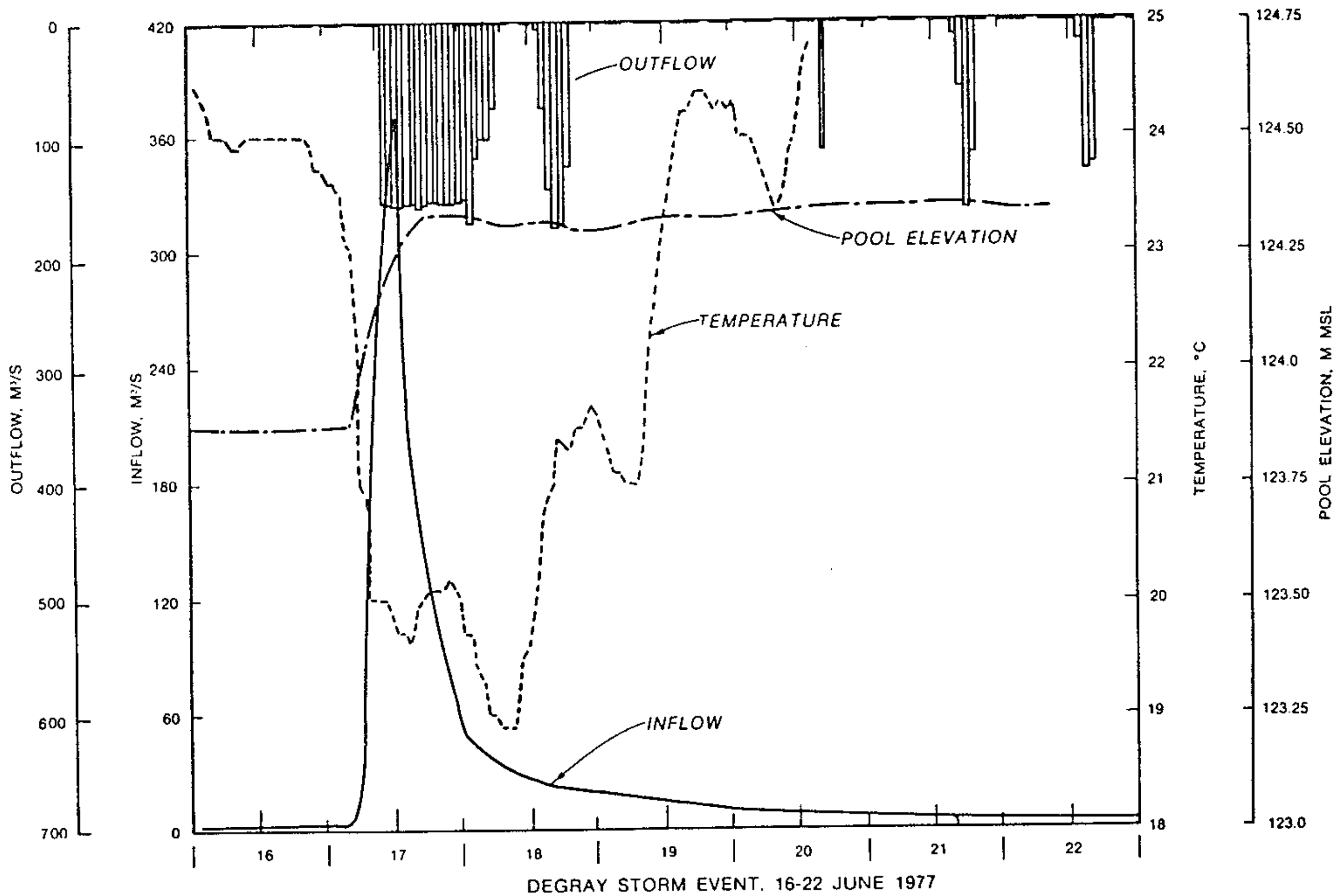

Fig. 3. Time history of inflow, outflow, pool elevation, and inflow temperature during the DeGray storm event, 16-22 June 1977.

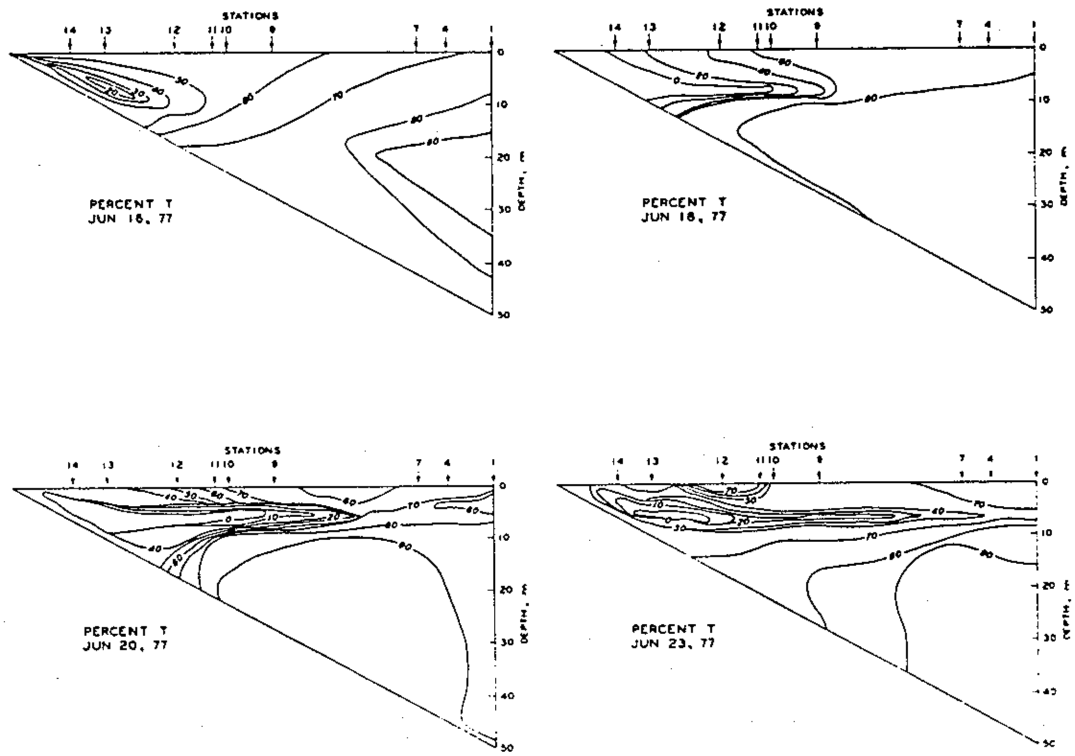

Fig. 4. Observed turbidity in DeGray Lake (measured as percent transmittance of light).

and its temperature approached that recorded in the river before the storm. Few project releases were made during 18-22 June (Fig. 3), and as a result, the interflow stalled with its center of mass between Stations 9 and 10.

The data collected during the above described event are used herein to evaluate LARM'S ability to simulate reservoir dynamics.

LARM

LARM was systematically developed over a period of years for the Corps of Engineers to assist in the analysis of water quality problems in rivers and reservoirs where buoyancy forces are important and lateral homogeneity can be assumed. The governing equations are laterally and layer averaged two-dimensional equations employing the Boussinesq and hydrostatic assumptions. Horizontal eddy coefficients are assumed constant while the vertical coefficients are dependent on the Richardson number. LARM generates time-varying velocity, temperature, and water quality constituent fields, and water surface elevations on a longitudinal and vertical grid. LARM also expands or contracts the finite difference grid in both the longitudinal and vertical directions in response to changes in reservoir elevation.

Since the model was originally intended for simulations over long time periods, a finite difference technique was selected that would not restrict time steps by free surface gravity wave criteria. Water surface elevations are first computed implicitly; then, velocity components are computed explicitly. Finally, temperature and other water quality constituent concentrations are computed implicitly.

Throughout its development, LARM has been thoroughly tested against analytical solutions and field applications (Edinger and Buchak 1975, 1979, 1982, 1983 and MacAurthur 1979). Previous applications (Edinger and Buchak 1975, 1979, 1982, 1983; Gordon 1979; MacAurthur 1979) have shown excellent predictions under varying reservoir operating conditions and velocity fields. In a recent review of numerical reservoir hydrodynamic models, Johnson (1981) concluded that LARM offers the greatest promise of all models evaluated and recommended it for further development.

DeGray Application

Calibration - LARM requires detailed reservoir morphometric data and time-varying hydrometeorological data as input. The geometry of the lake is approximated by cells of equal length and thickness, but of variable width. Cell sizes were determined from topographic maps and reservoir sediment range cross sections. Previous studies (Ford and Johnson 1980) showed that the conveyance path follows the old river channel in DeGray Lake. The reservoir was divided into 32 cells of equal length (994 m) along this path. A cell thickness of 2 m was selected for vertical resolution. Cell widths were determined by GEDA (USAE Hydrologic Engineering Center 1976). Given cross-section data defining reservoir morphometry, GEDA interpolates between the given cross sections and produces tables of various hydraulic parameters (including top width) for transects evenly spaced along the longitudinal axis of the reservoir. The input cross sections were adjusted to calibrate the GEDA-calculated reservoir surface area and volume.

Time-varying hydrometeorological data required for LARM simulations include inflows, outflows, dew point and equilibrium temperatures, heat exchange coefficients, solar radiation, and wind speed. Reservoir inflow records were obtained at bi-hourly intervals from the gage at Highway 84 (Fig. 2) and were modified following Ford and Ford (1983) to account for the ungaged areas. Values for reservoir outflows were obtained from daily operational records. Equilibrium temperatures and solar radiation were computed on an hourly basis and a daily average heat exchange coefficient was calculated using meteorological data from the National Weather Service Station at Little Rock, AR, located approximately 100 km from the lake.

LARM simulations were started under near-isothermal conditions on 16 February with a measured temperature profile and a zero velocity field. It took approximately 5 days for the velocity field to develop. The model was calibrated by adjusting the vertical eddy diffusivity to match measured temperature profiles. Since the volume of flow through any given cell during a time step must be less than the cell's volume, the large storm flow dictated the use of a six-minute time step.

Velocity - Predicted velocity fields for 16, 18, 20, and 23 June are shown in Fig. 5. On 16 June, prior to the peak of the inflow hydro-

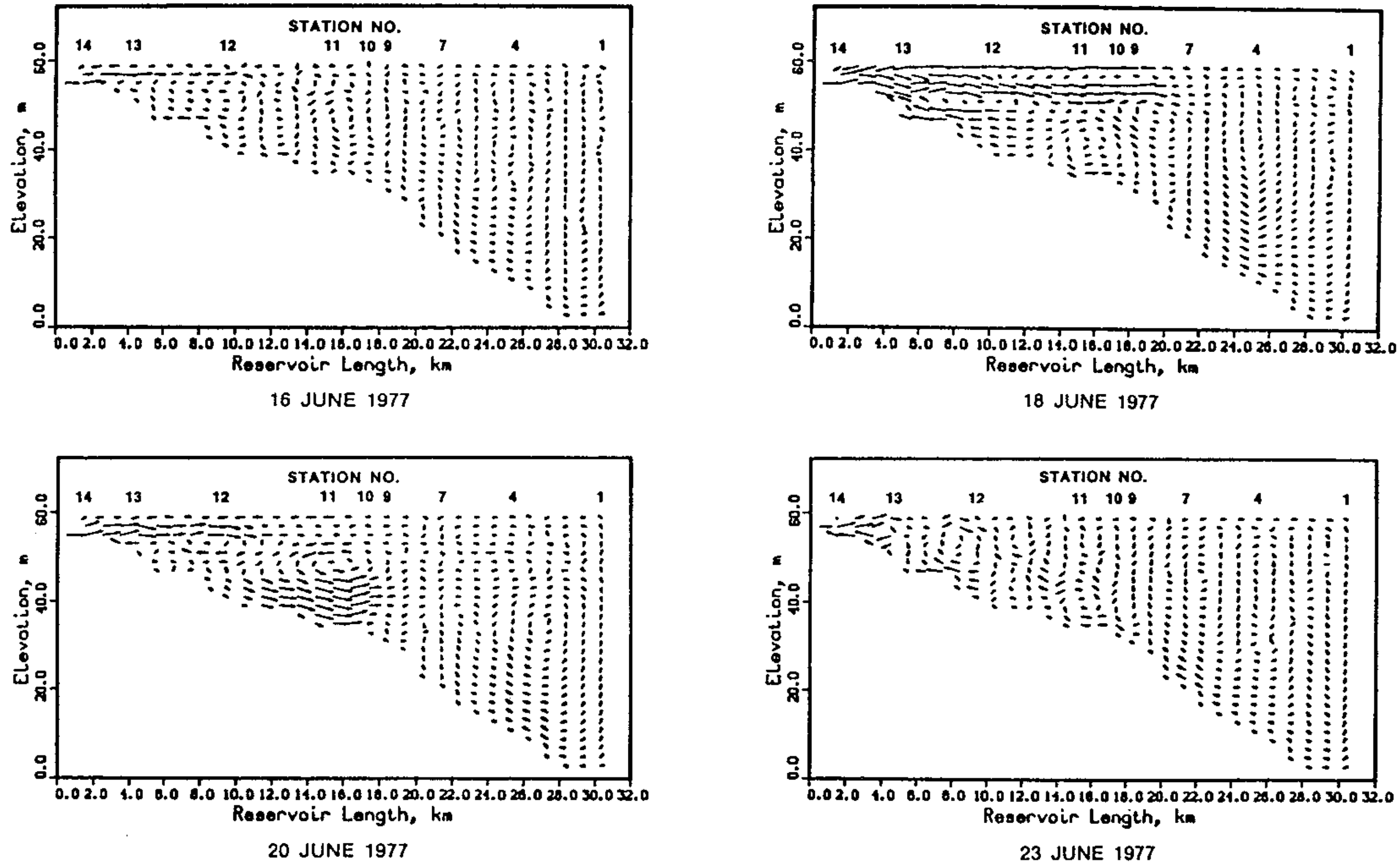

Fig. 5. Velocity vectors in DeGray Lake predicted by LARM.

graph (Fig. 3), the large velocities evident from Station 14 to Station 12 in the second layer may be attributed to the small storm which which occurred three days earlier. The observed turbidity between Stations 14 and 12 for 16 June (Fig. 4) must also have resulted from this small storm. By 18 June the increased flow resulting from the large storm event subsided at the Highway 84 gage site and began to enter the pool at a depth of approximately 6 m. A well-defined interflow was predicted with large velocities intruding to Station 9 and with a strong upstream return current. By 20 June large velocities were predicted only upstream of Stations 12 at the 6-m depth (third layer). The observed 0 percent transmittance isopleth on 20 June is near Station 9 where the interflow stalled (Ford and Johnson 1980). The large velocities predicted in the hypolimnion near Station 11 (in the constriction in the reservoir) are probably due to unrealistically narrow cells created by GEDA.

On 23 June a well-defined current was predicted from the headwaters to Station 13. Several large eddies were also predicted. The bottom return current upstream of Station 12 may be responsible for the pocket of turbid water observed between Stations 12 and 13 in Fig. 4.

Turbidity - Inflow turbidity was measured at the Highway 84 gage site only during the period of the 16 June storm hydrograph. Turbidity increased during the rising limb of the hydrograph and reached a maximum of 165 NTU's four hours prior to the peak of the hydrograph (Fig. 6). Since turbidity data collected in the pool were measured as percent transmittance, it was impossible to compare inflow and pool data directly. The objectives of the turbidity simulations were, therefore, to help explain the percent transmittance data in the pool and to determine how the inflow moved through the reservoir. Turbidity was simulated as a conservative substance with zero background turbidity in the pool.

Results for the turbidity simulations are shown in Figs. 7a, 7b, 8a, and 8b. The observed data (Fig. 4) showed turbidity in the headwaters on 16 June, but the simulation results showed no increase until 18 June (Fig. 7a). It was suspected that the observed turbidity on 16 June was probably due to the earlier small storm event. Since no inflow turbidity data existed prior to 16 June, a synthesized spike of turbidity was injected on the rising side of the small storm hydrograph in a second simulation. The results of this simulation (Fig. 8a and b) gave more support to the small storm hypothesis. Changes in temperature profiles and percent transmittance at Stations 12 (Fig. 9) and 10 (Fig. 10) reflected the influx of cool, turbid Caddo River water. Surface temperatures at Station 12, and to a lesser extent, Station 10, decreased coincident with pronounced increases in turbidity at the depth of the thermocline.

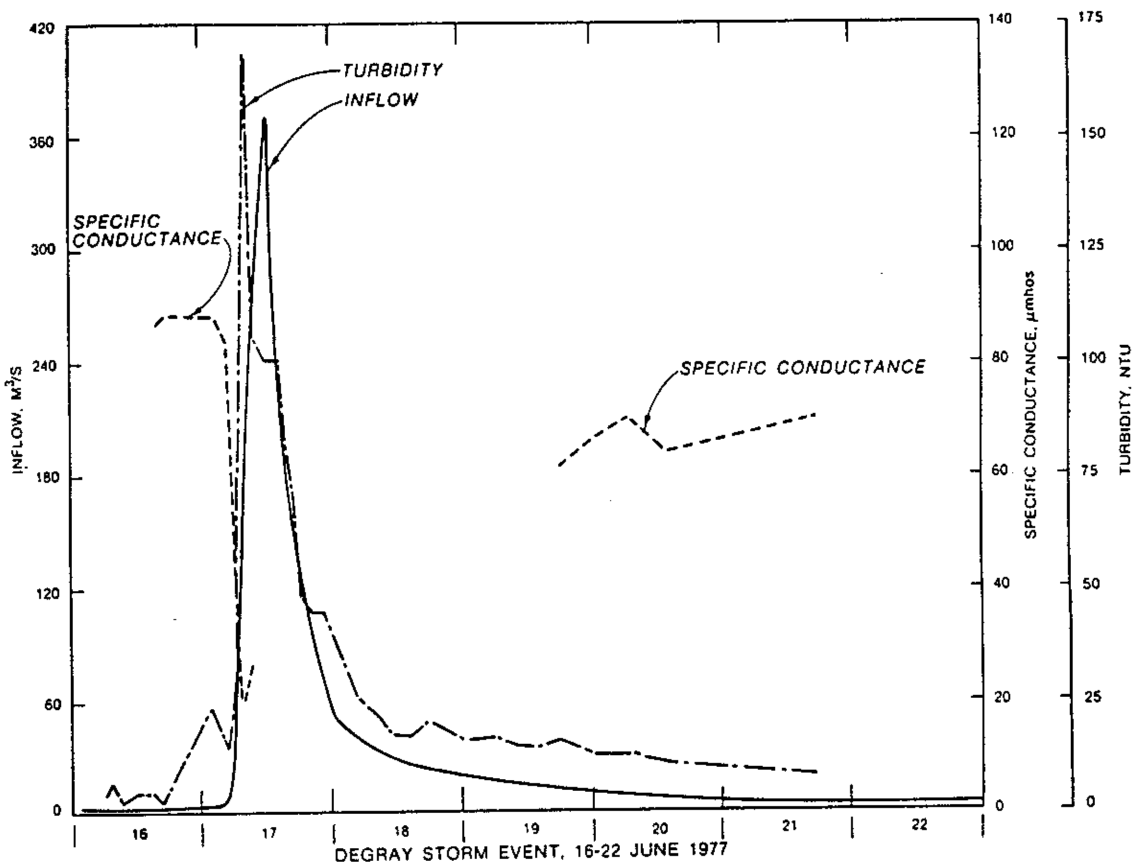

Fig. 6. Time history of inflow and turbidity concentrations during the DeGray storm event, 16-22 June 1977.

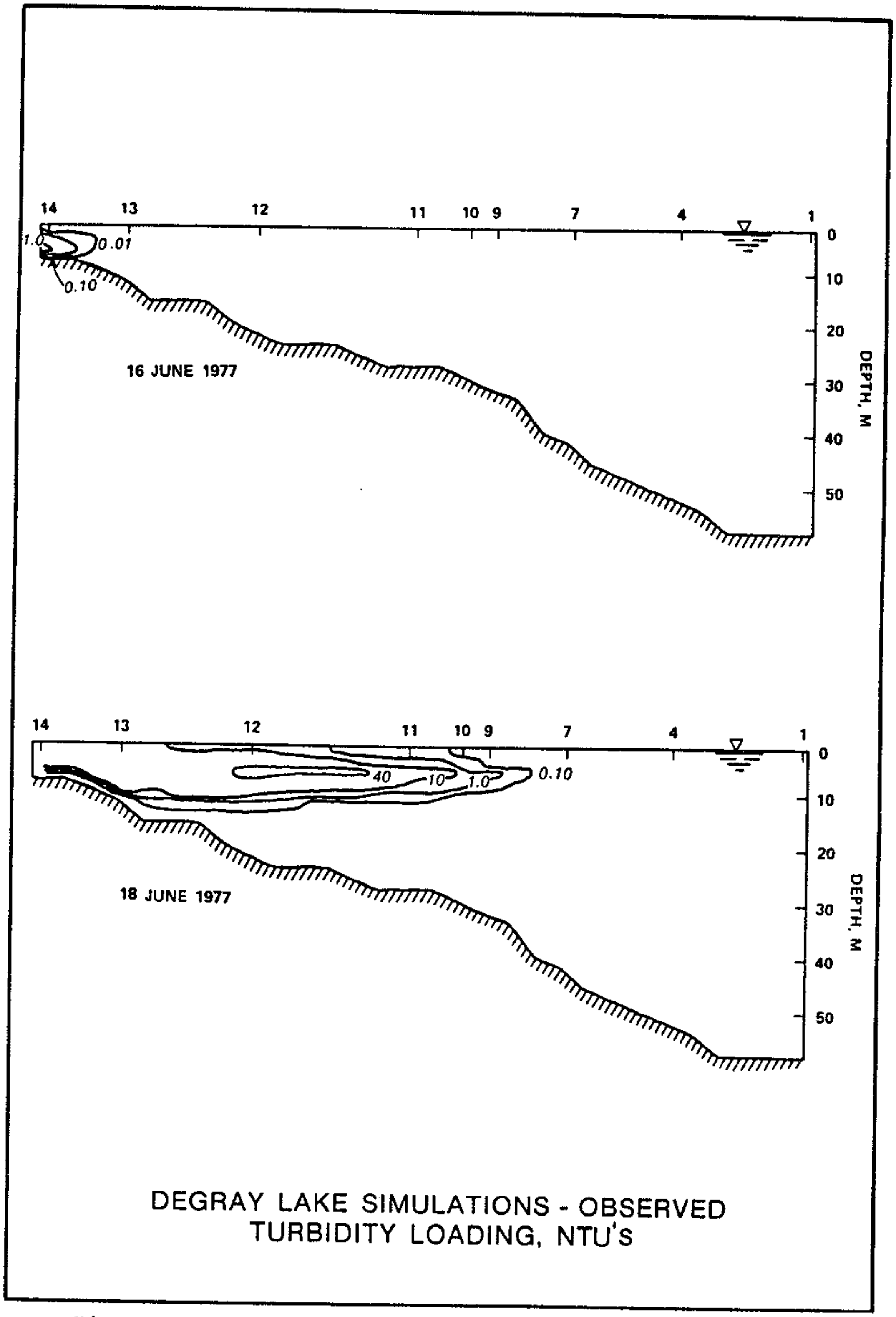

Fig. 7a. Observed turbidity loading during the DeGray storm event, 16-22 June 1977.

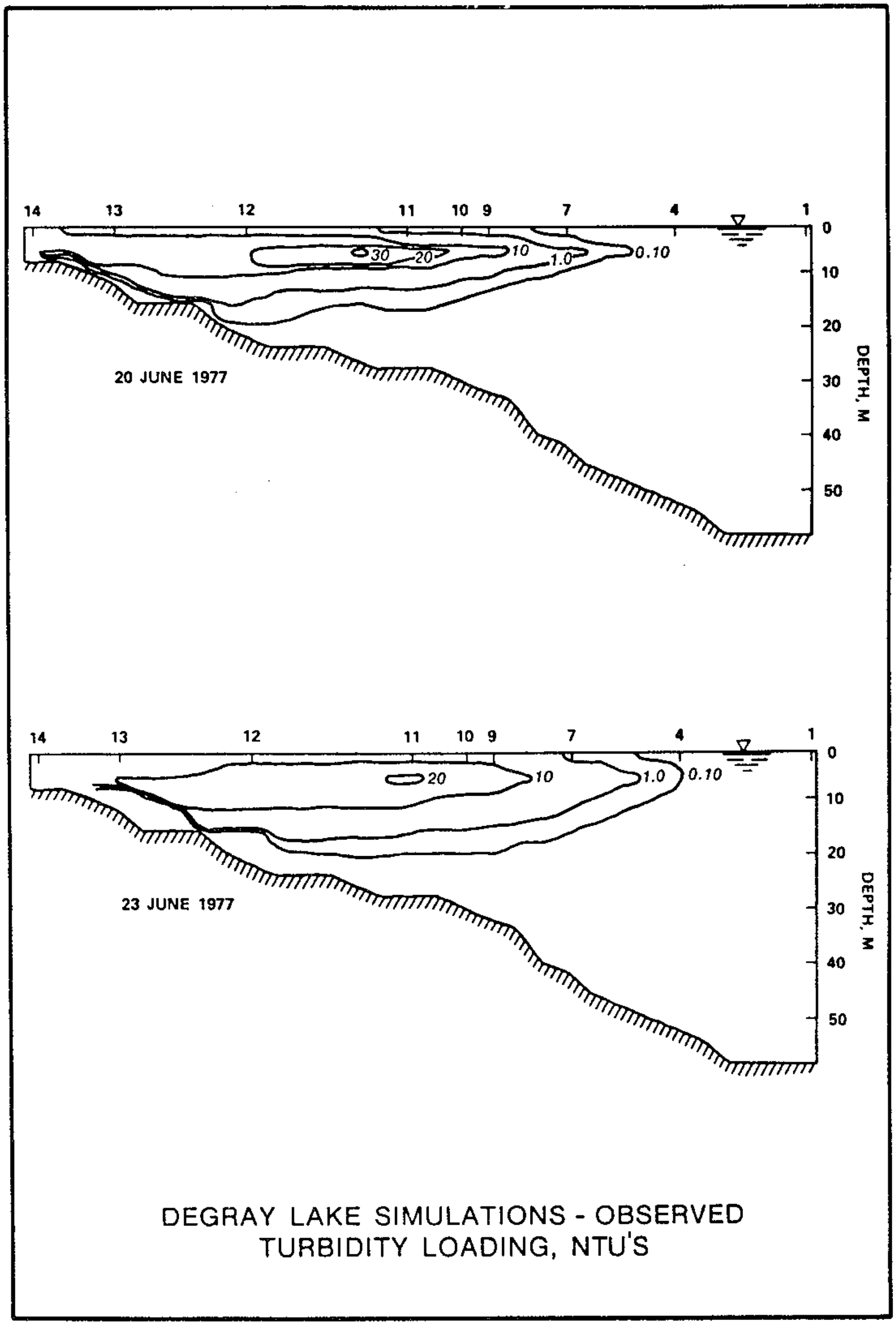

Fig. 7b. Observed turbidity loading during the DeGray storm event, 16-22 June 1977.

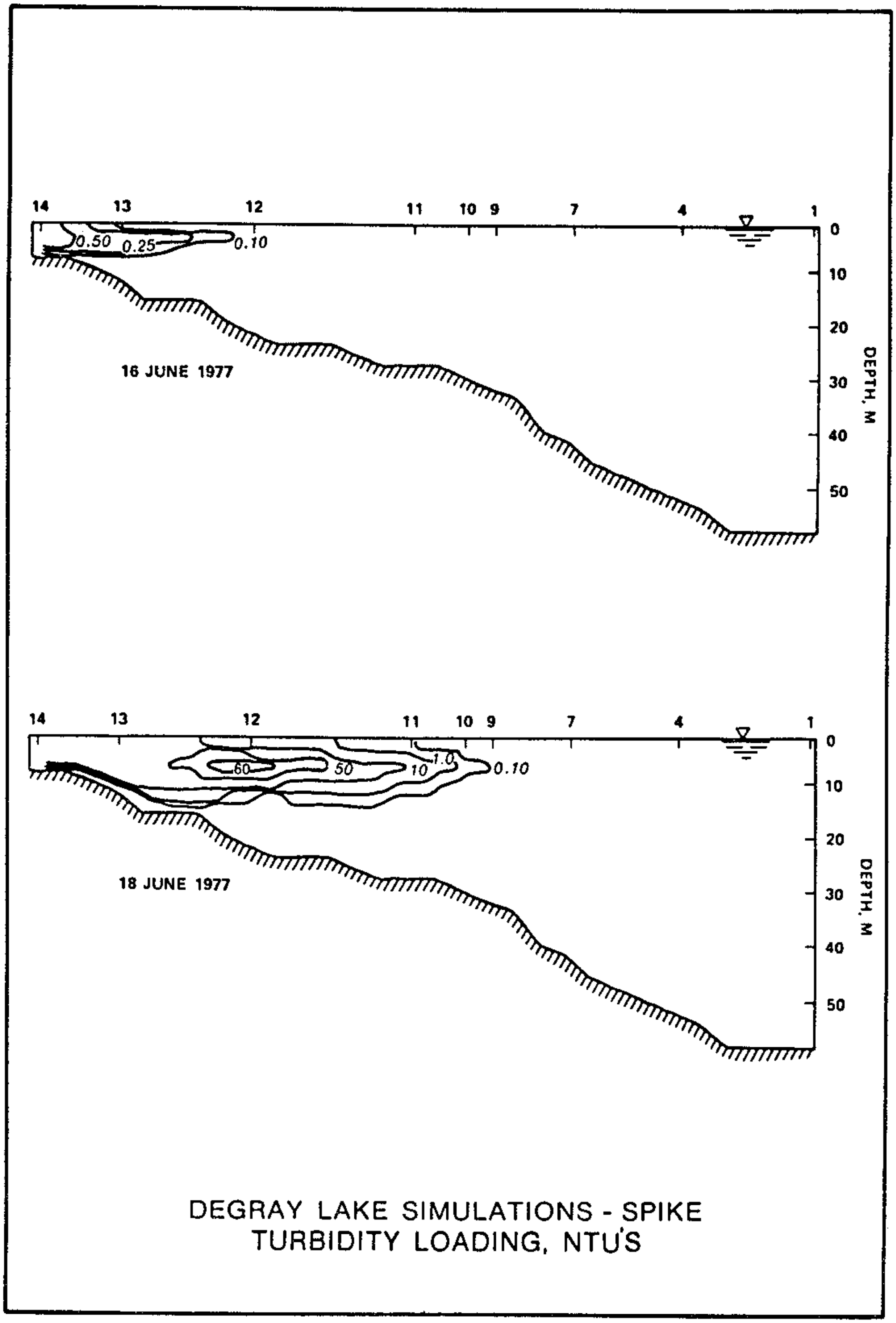

Fig. 8a. Synthesized spike turbidity loading during the DeGray storm event, 16-22 June 1977.

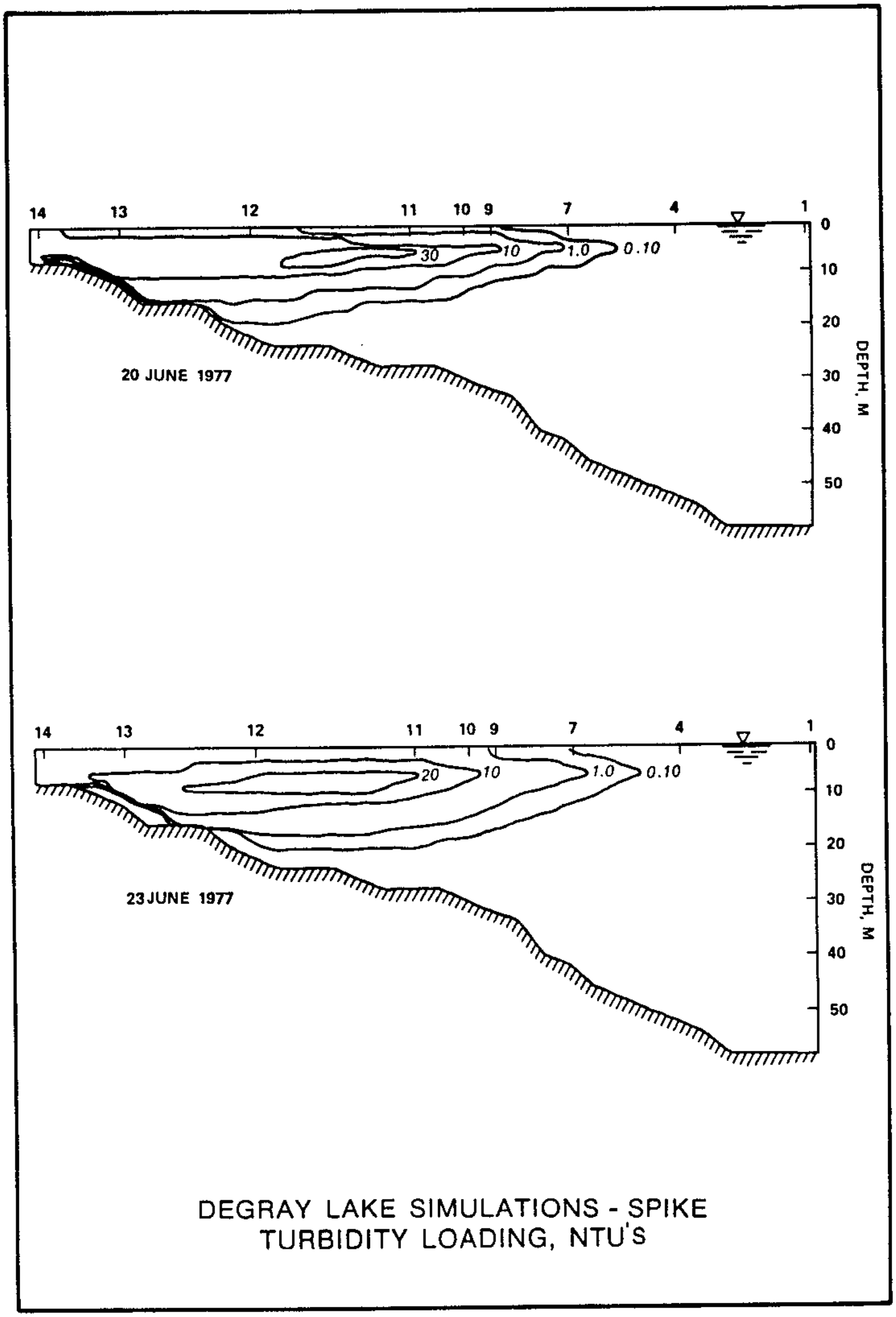

Fig. 8b. Synthesized spike turbidity loading during the DeGray storm event, 16-22 June 1977.

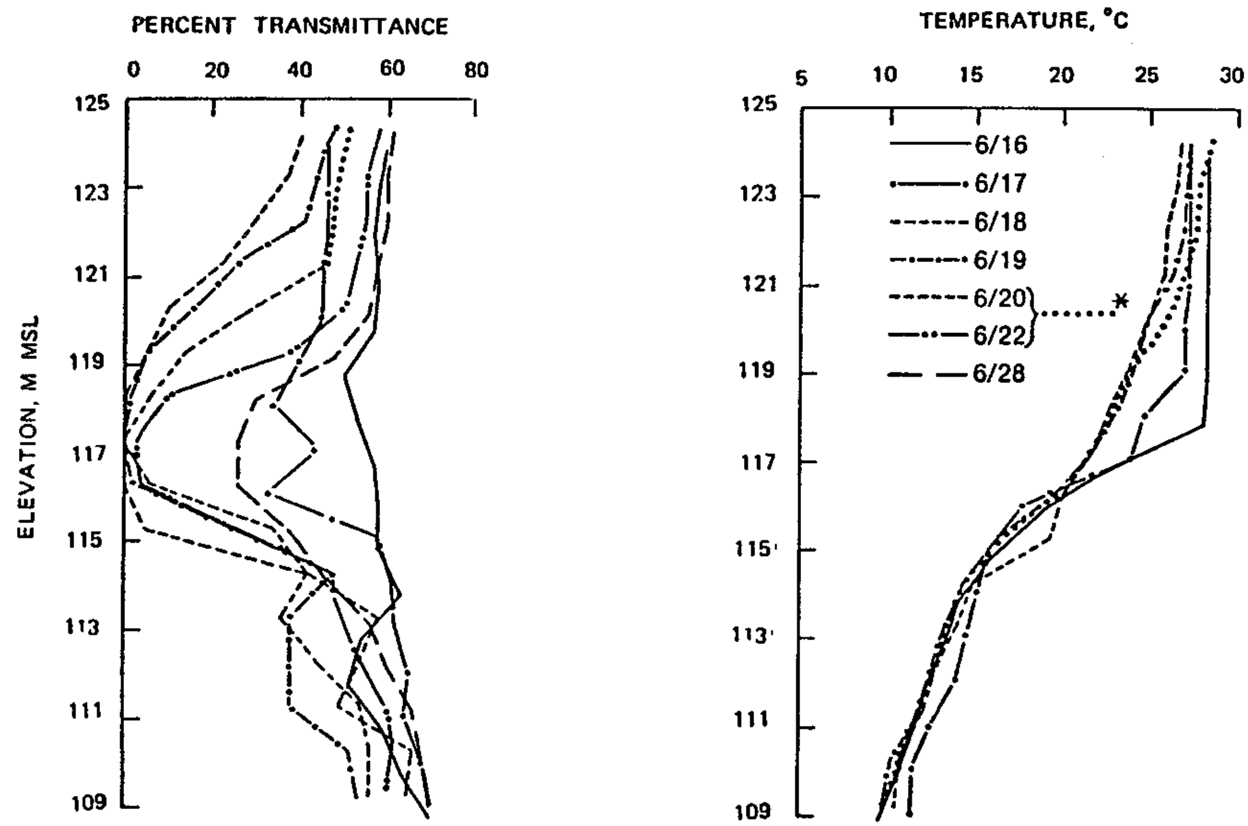

Fig. 9. Observed turbidity and temperature profiles at Station 12, June 1977.

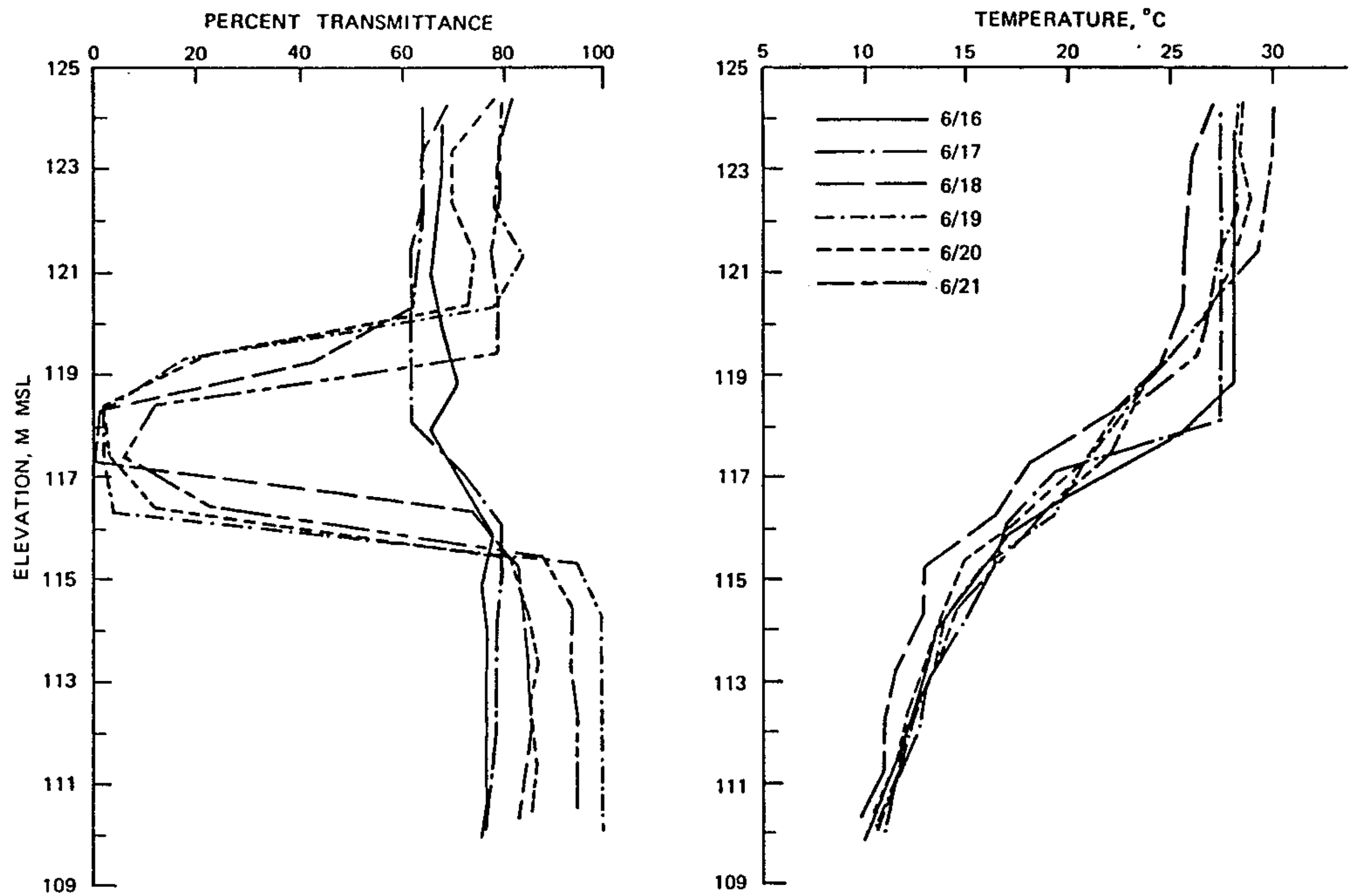

Fig. 10. Observed turbidity and temperature profiles at Station 10, June 1977.

Fecal Coliforms - Fecal coliforms were simulated both as a conservative substance and with first-order decay to determine the relative importance of dilution and die-off with respect to the total observed decrease of coliforms in the pool. In contrast to turbidity, fecal coliforms increased with the flow and attained a maximum of 60,000 col/100 ml coincident with the hydrograph peak (Fig. 11). In the fecal coliform simulations, background concentrations were assumed to be zero since measurements prior to the storm showed no colonies within the lake. Die-off rates were obtained from Thornton et al. 1980.

Results from the fecal coliform simulations can be used to distinguish between the effects of the mechanisms that reduced fecal coliform concentrations in the pool. When fecal coliforms were modeled as a conservative substance (Figs. 12a, 12b), a peak of 20,700 col/100 ml was predicted at Station 12. Thus, dilution alone accounted for a threefold reduction of the peak inflow concentration. When a laboratory-derived decay rate of 1.6/day was used, die-off was added as a mechanism of population decline and the peak dropped to 5000 col/100 ml (Fig. 13a). Results for the simulation using a decay rate of 3.2/day derived from field observations show the effects of flocculation and settling as well as mixing and die-off (Figs. 14a, 14b). In this case the peak concentration was reduced to 1000 col/100 ml in agreement with the observed value of 560 col/100 ml. All three simulations correctly predicted that the leading edge of the coliform plume would reach Station 10 and stall with its center of mass between Stations 12 and 11 at a 5-6 m depth.

Total Phosphorus - Phosphorus concentrations in the Caddo River inflow peaked at 530 µg/l four hours prior to the hydrograph peak (Fig. 15). No information was available on the amount of phosphorus that entered during the small storm on 13 June. If the 2.73 metric tons of phosphorus which entered during the large storm were instantaneously dispersed throughout the lake, it would have been sufficient to raise phosphorus concentrations by 3.5 µg/l. If the phosphorus load was retained above Station 10, then the concentrations above Station 10 would have increased by 20 µg/l. Since the measured phosphorus concen-

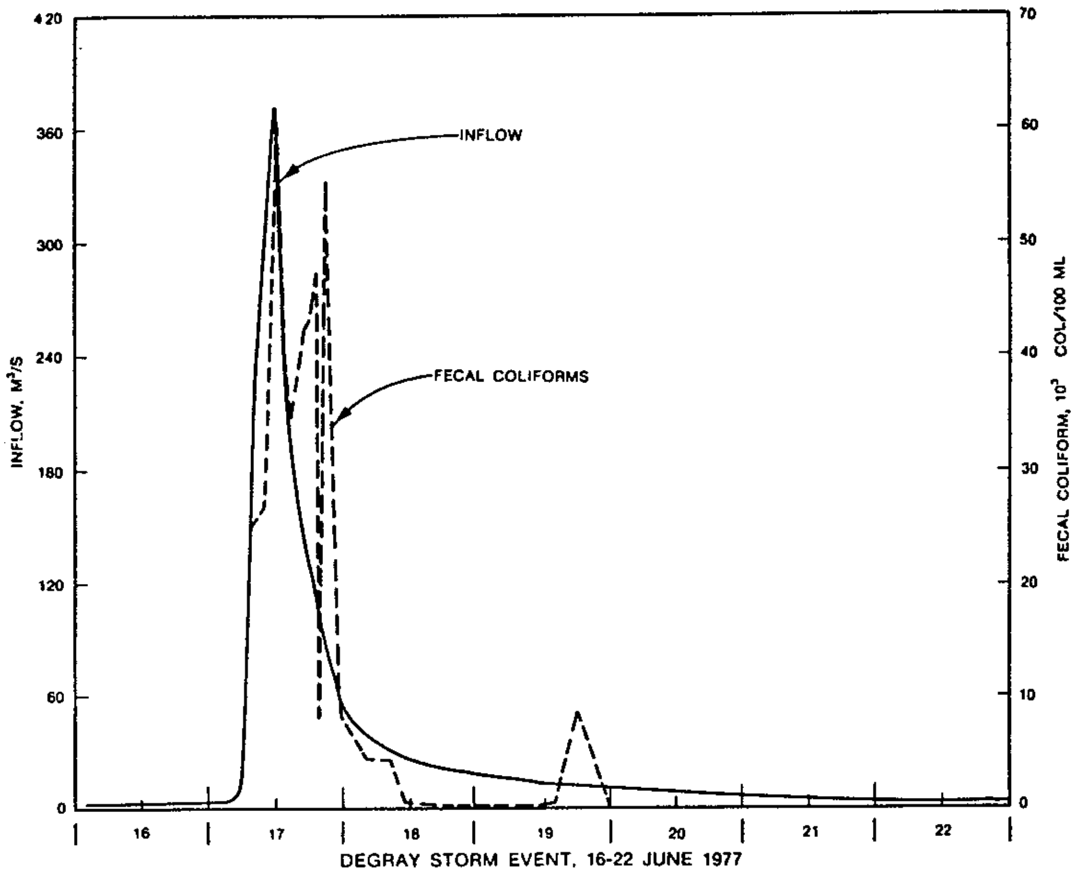

Fig. 11. Time history of inflow and fecal coliform concentrations during the DeGray storm event, 16-22 June 1977.

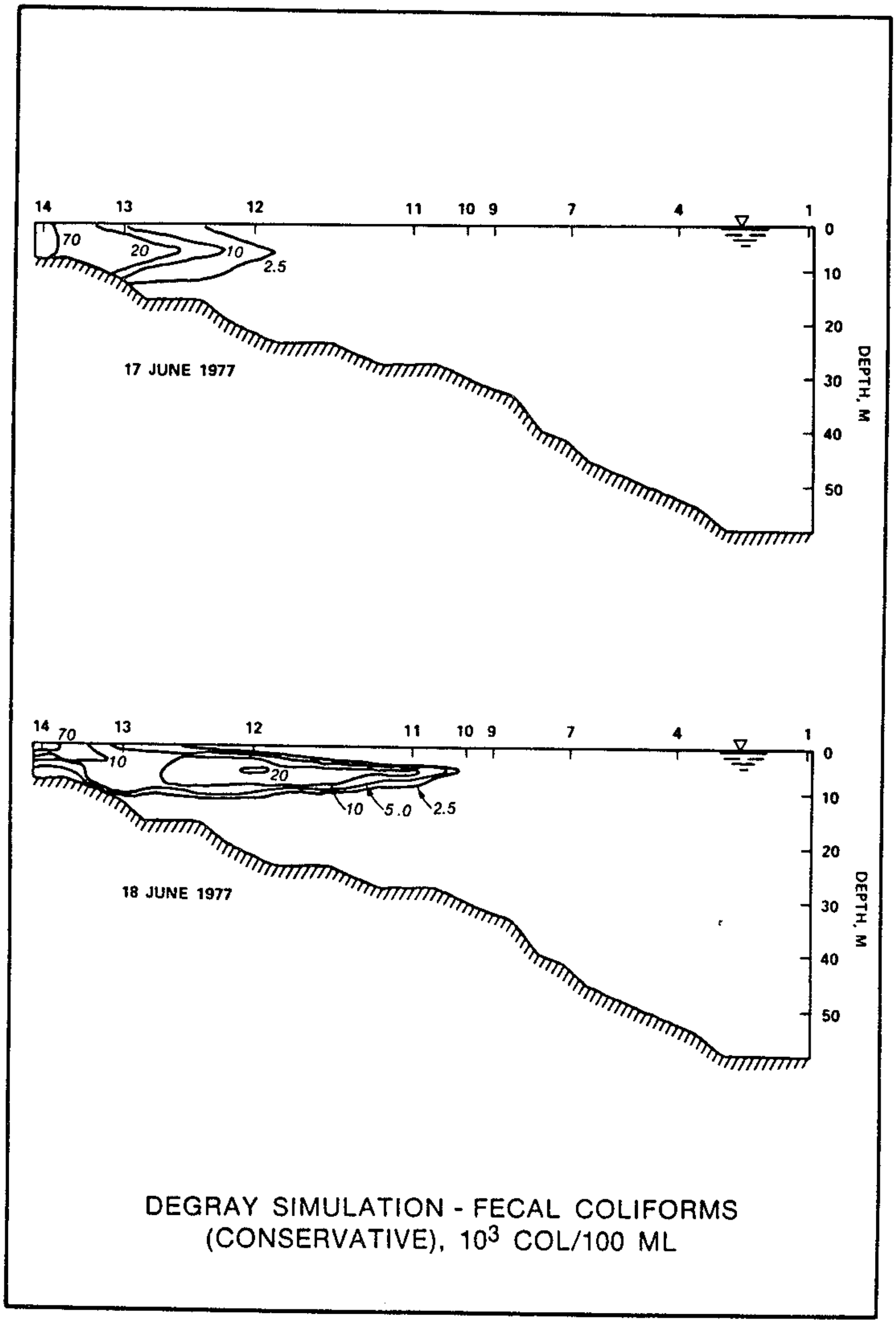

Fig. 12a. Simulated fecal coliform loading (as a conservative) during the DeGray storm event, 16-22 June 1977.

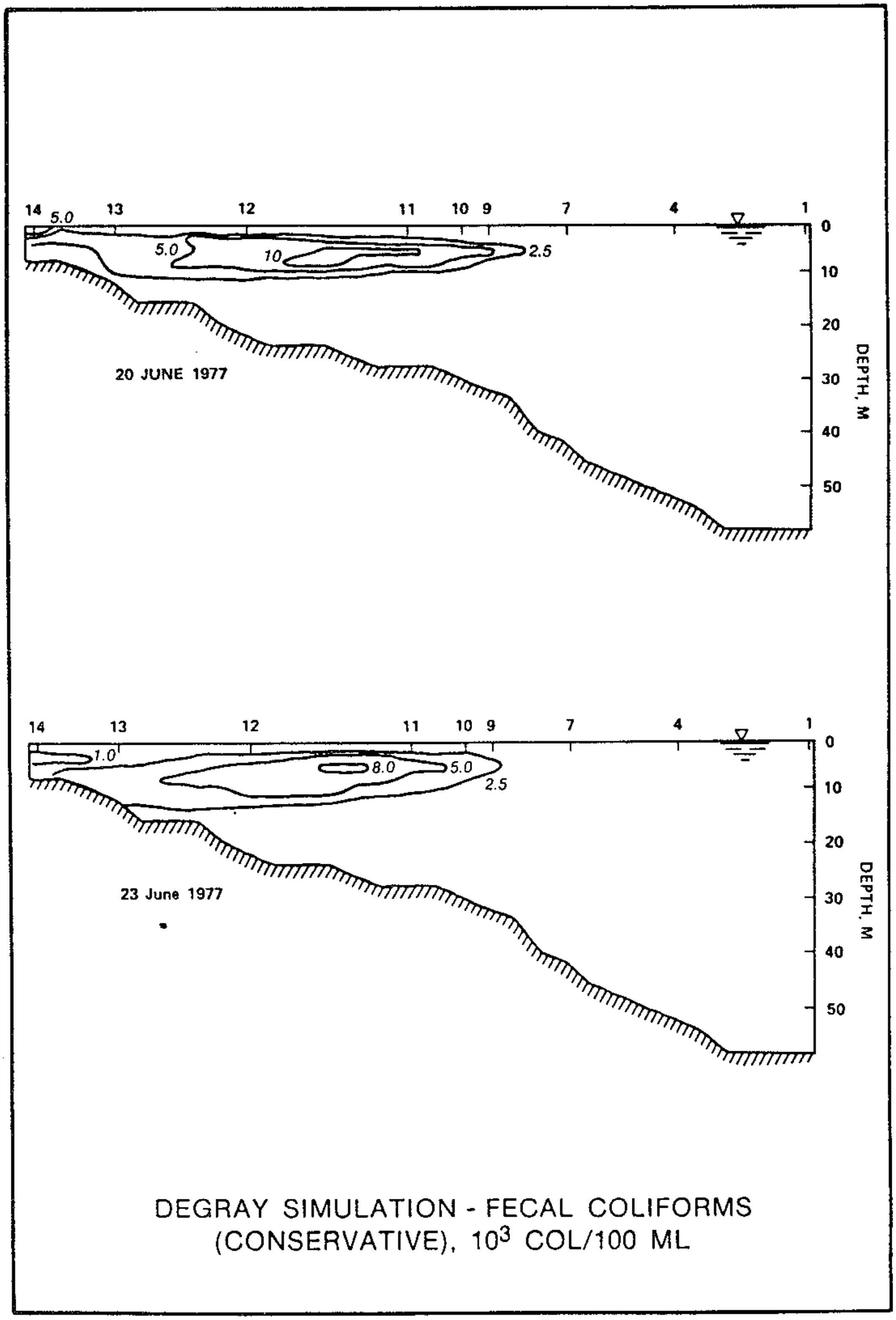

Fig. 12b. Simulated fecal coliform loading (as a conservative) during the DeGray storm event, 16-22 June 1977.

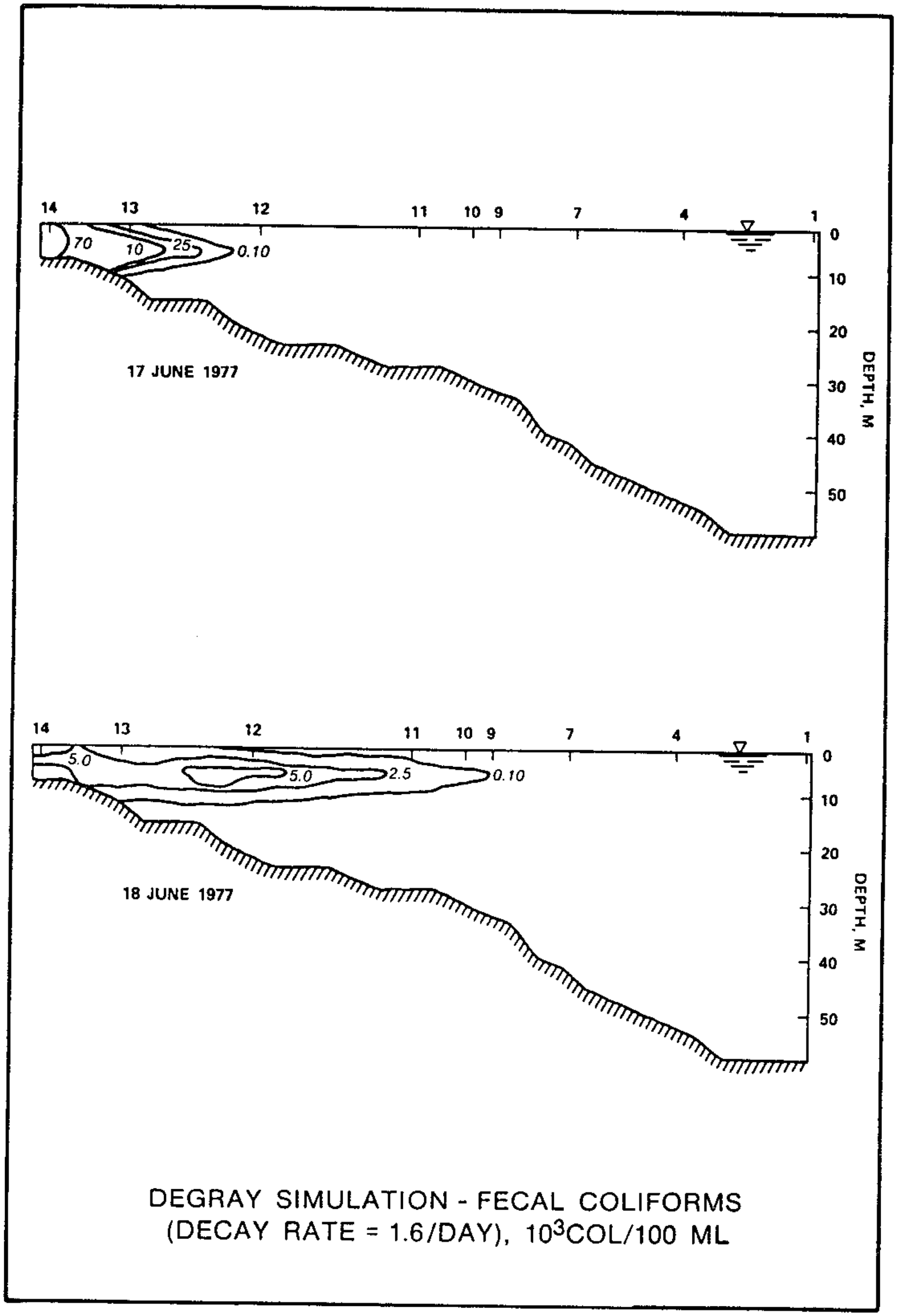

Fig. 13a. Simulated fecal coliform loading (with decay rate = 1.6/day) during the DeGray storm event, 16-22 June 1977.

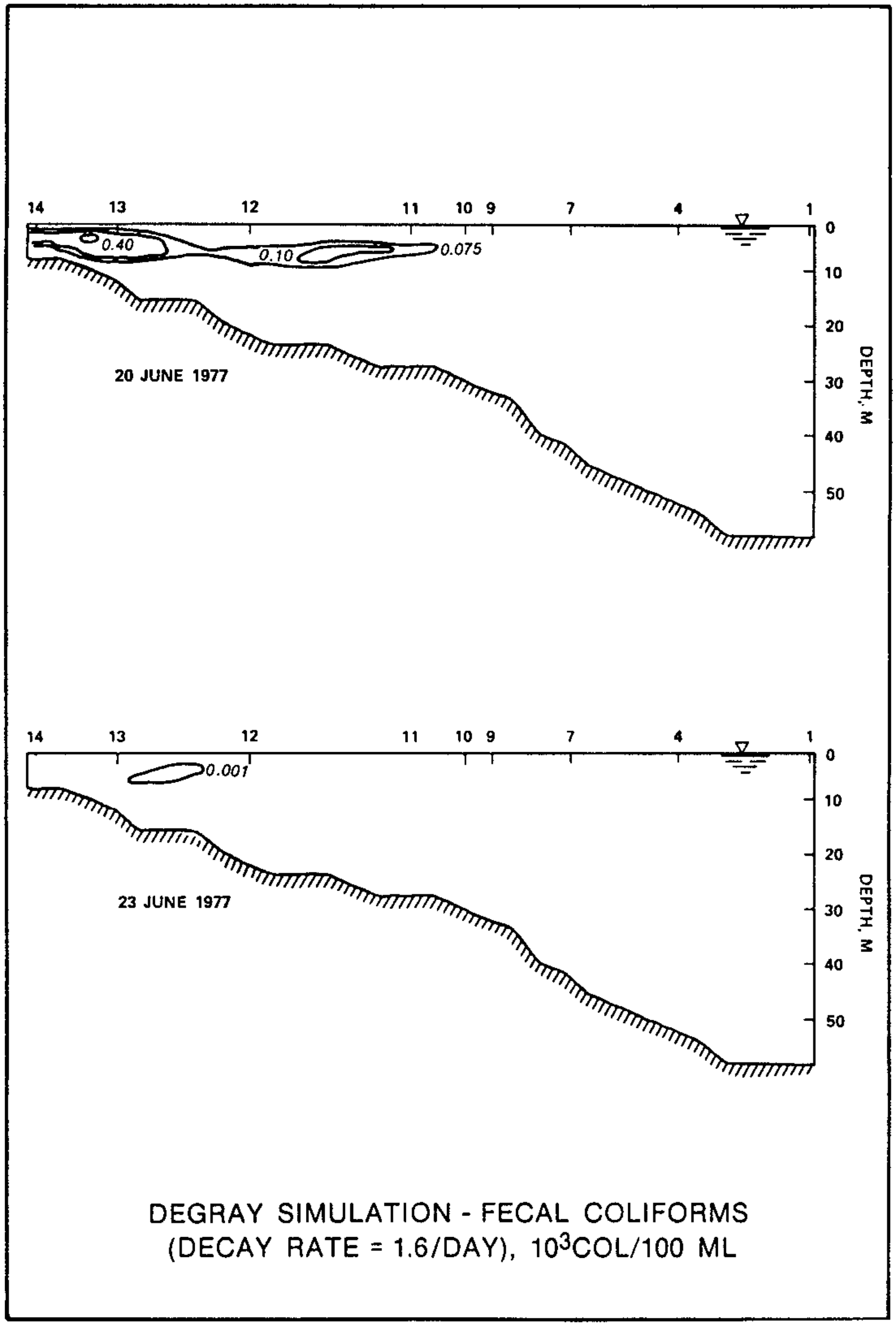

Fig. 13b. Simulated fecal coliform loading (with decay rate = 1.6/day) during the DeGray storm event, 16-22 June 1977.

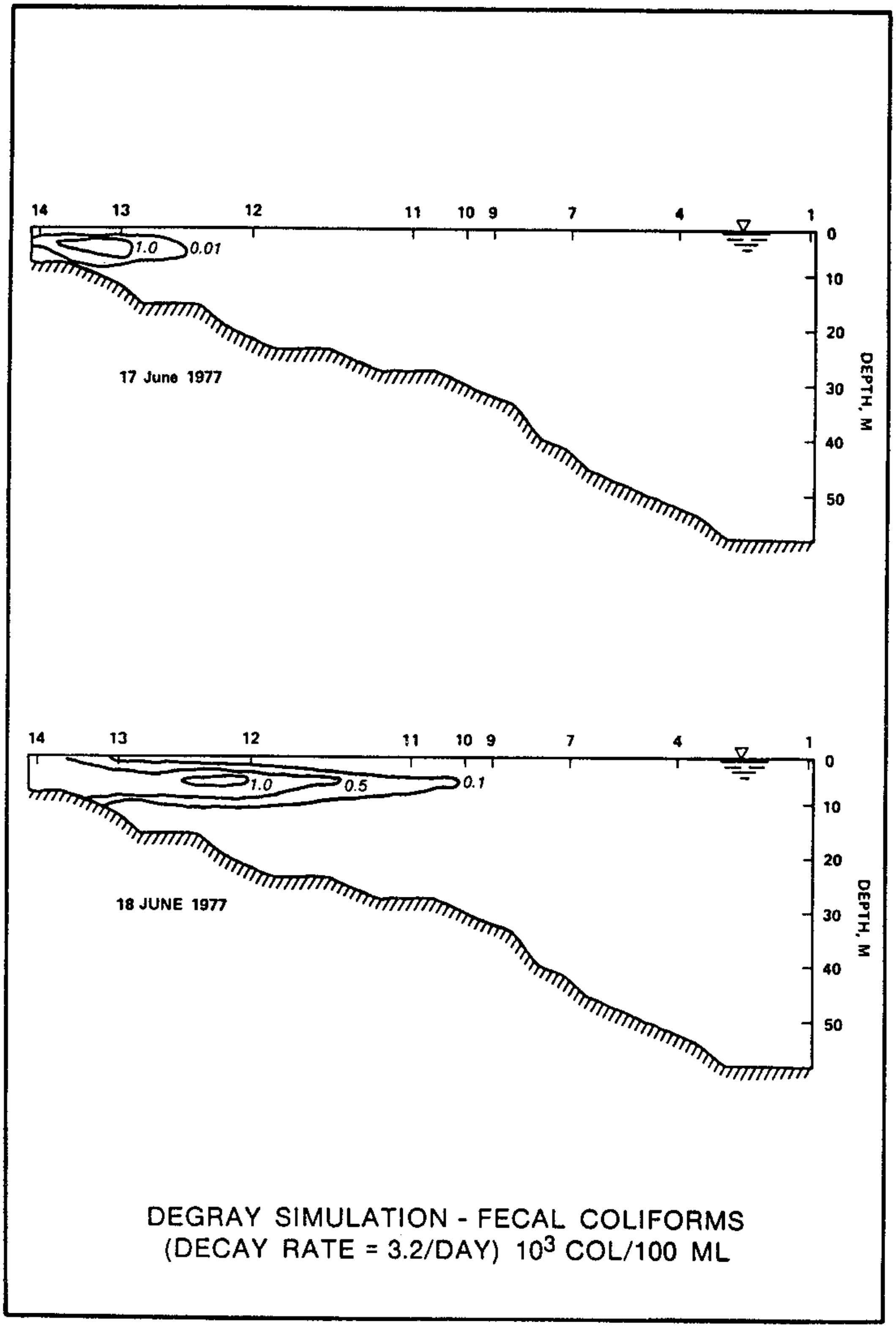

Fig. 14a. Simulated fecal coliform loading (with decay rate = 3.2/day) during the DeGray storm event, 16-22 June 1977.

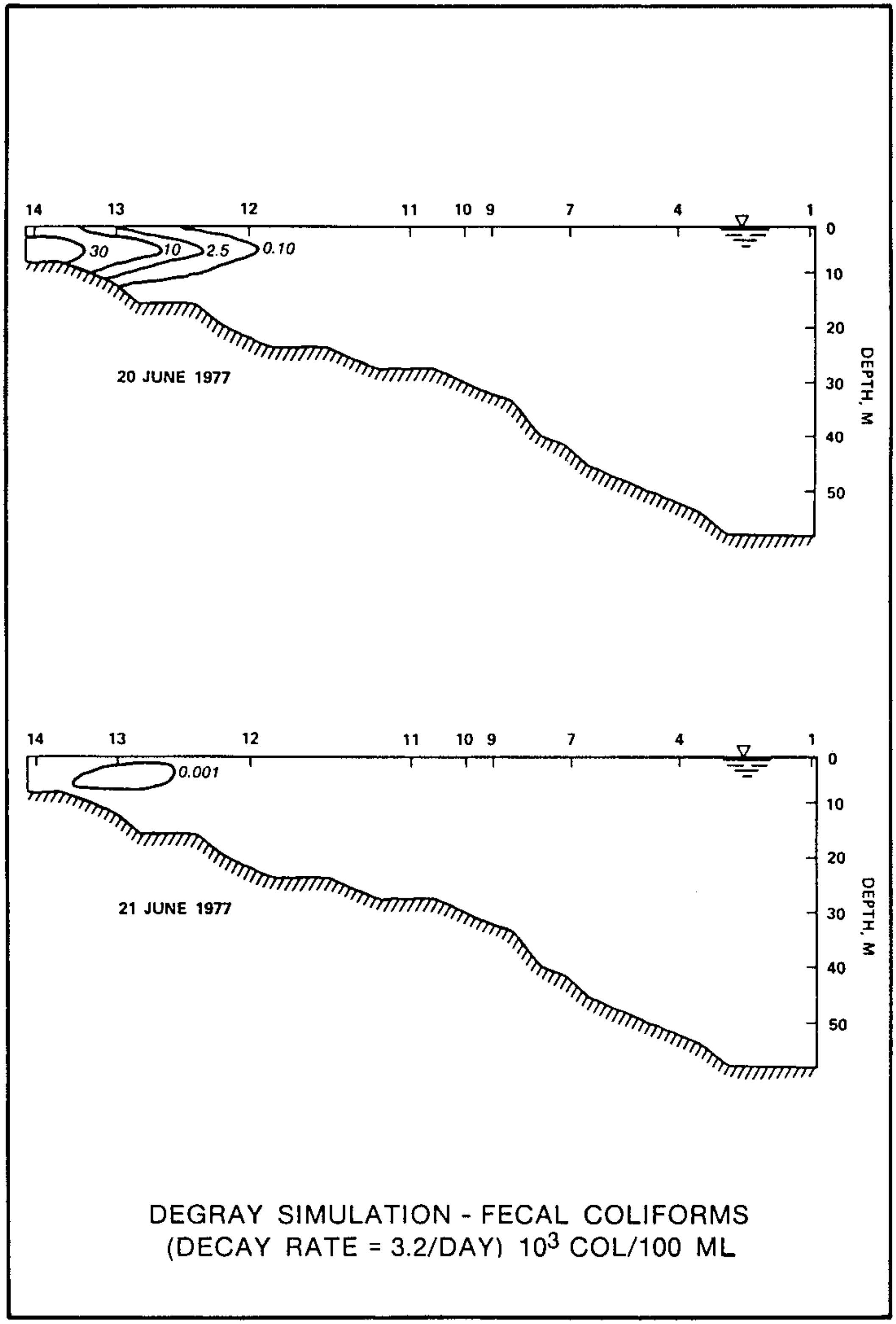

Fig. 14b. Simulated fecal coliform loading (with decay rate = 3.2/day) during the DeGray storm event, 16-22 June 1977.

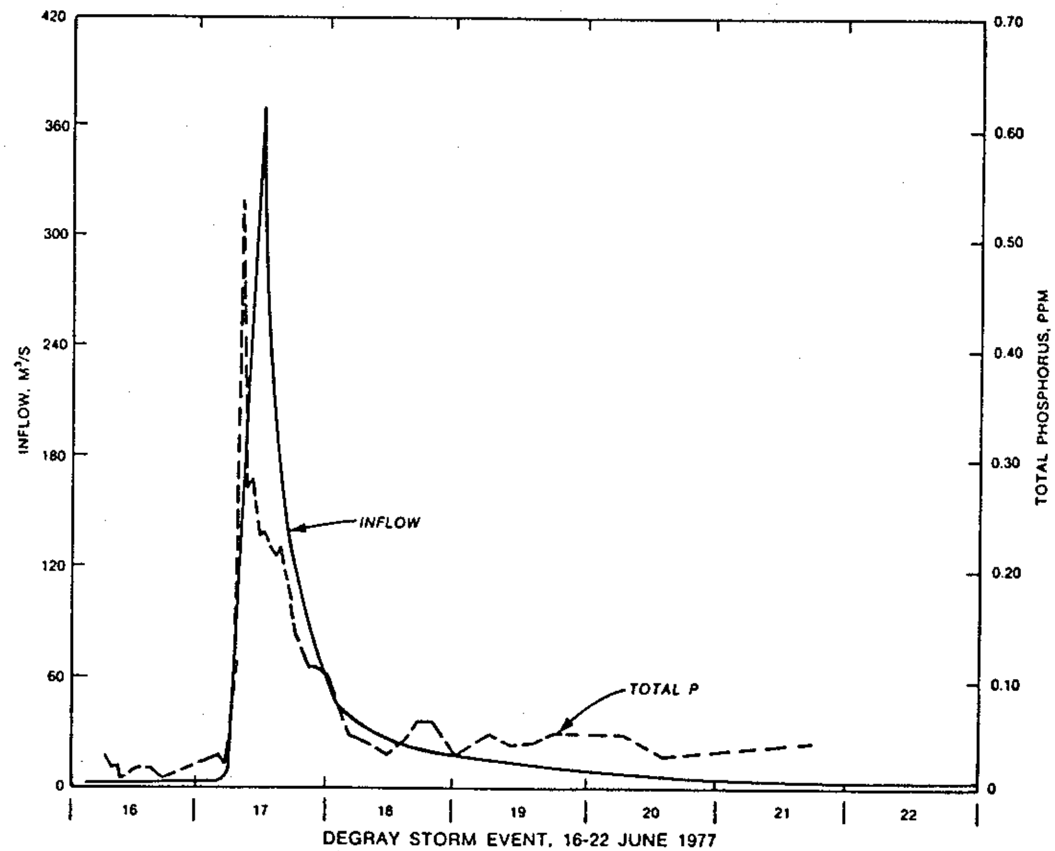

Fig. 15. Time history of inflow and total phosphorus concentrations during the DeGray storm event, 16-22 June 1977.

trations increased by less than 20 µg/l at Stations 12 and 10 during the study period, phosphorus was modeled as a conservative substance to estimate the importance of dilution and mixing in reducing the in-pool phosphorus concentrations.

Observed phosphorus concentrations at Station 12 varied from 10-20 µg/l in the surface waters to a maximum of 70 µg/l below the thermocline. The increases in the hypolimnion were possibly due to releases from anaerobic sediments. On 18 June concentrations increased by approximately 15 µg/l in the zone of inflow at Station 12. On 20 and 23 June concentrations decreased at Station 12 throughout the entire water column, especially within the hypolimnion.

At Station 10, field data from 16 to 23 June showed near-uniform concentration profiles with an increase of 10 µg/l occurring on 18 June throughout the depth. No other changes occurred in the Station 10 profiles during the remainder of the study.

When phosphorus was modeled as a conservative substance (Figs. 16a, 16b) predicted concentrations were 100 µg/l in the inflow at Station 12 and 10 µg/l at the leading edge near Station 10 on 18 June. Transport and mixing, therefore, were responsible for diluting inflow phosphorus concentrations by a factor of five. The remaining differences between the observed and predicted concentrations may be attributed to adsorption of phosphorus to suspended sediments from the turbid inflow. During the same period that phosphorus was observed to decrease at Station 12, turbidity increased in the hypolimnion, possibly due to settling of suspended particles from the turbidity plume (Figs. 4 and 10). It is possible, therefore, that settling particles acted as adsorption media and purged the water column of phosphorus.

Nitrate - In contrast to total phosphorus concentrations which peaked during the rising limb of the hydrograph, nitrate concentrations reached a maximum of 0.62 mg/l six hours after the hydrograph peak (Fig. 17). A total of 6.82 metric tons of nitrate-nitrogen entered the reservoir as a result of the large storm. Concentrations should have increased by 0.25 mg/l and 0.05 mg/l, if this mass were instantaneously dispersed in a volume of water equivalent to that existing above Stations 12 and 10, respectively. Field data at Stations 10 and 12 showed no increase in nitrate between 18 and 20 June, and a decrease

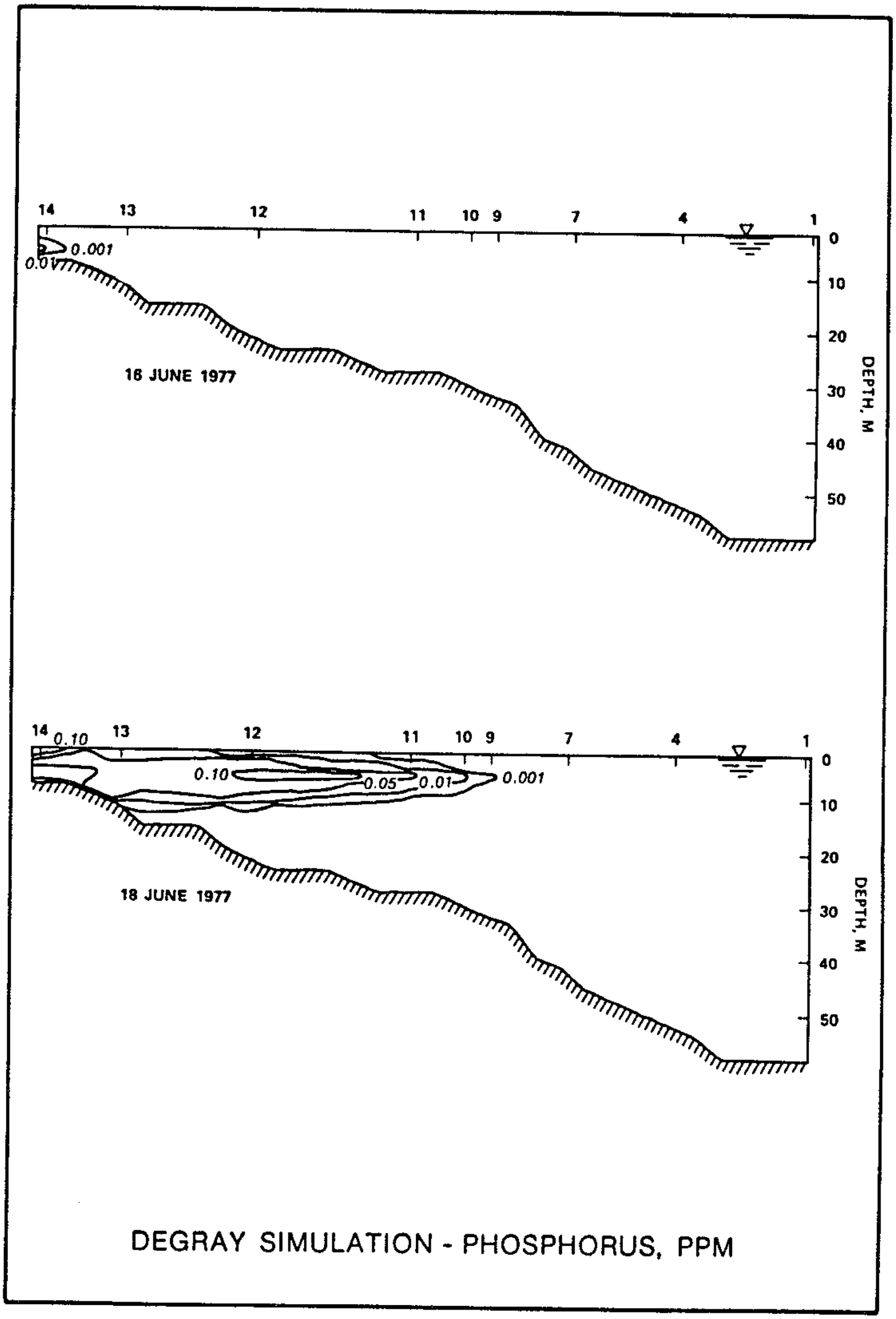

Fig. 16a. Simulated phosphorus loading during the DeGray storm event, 16-22 June 1977.

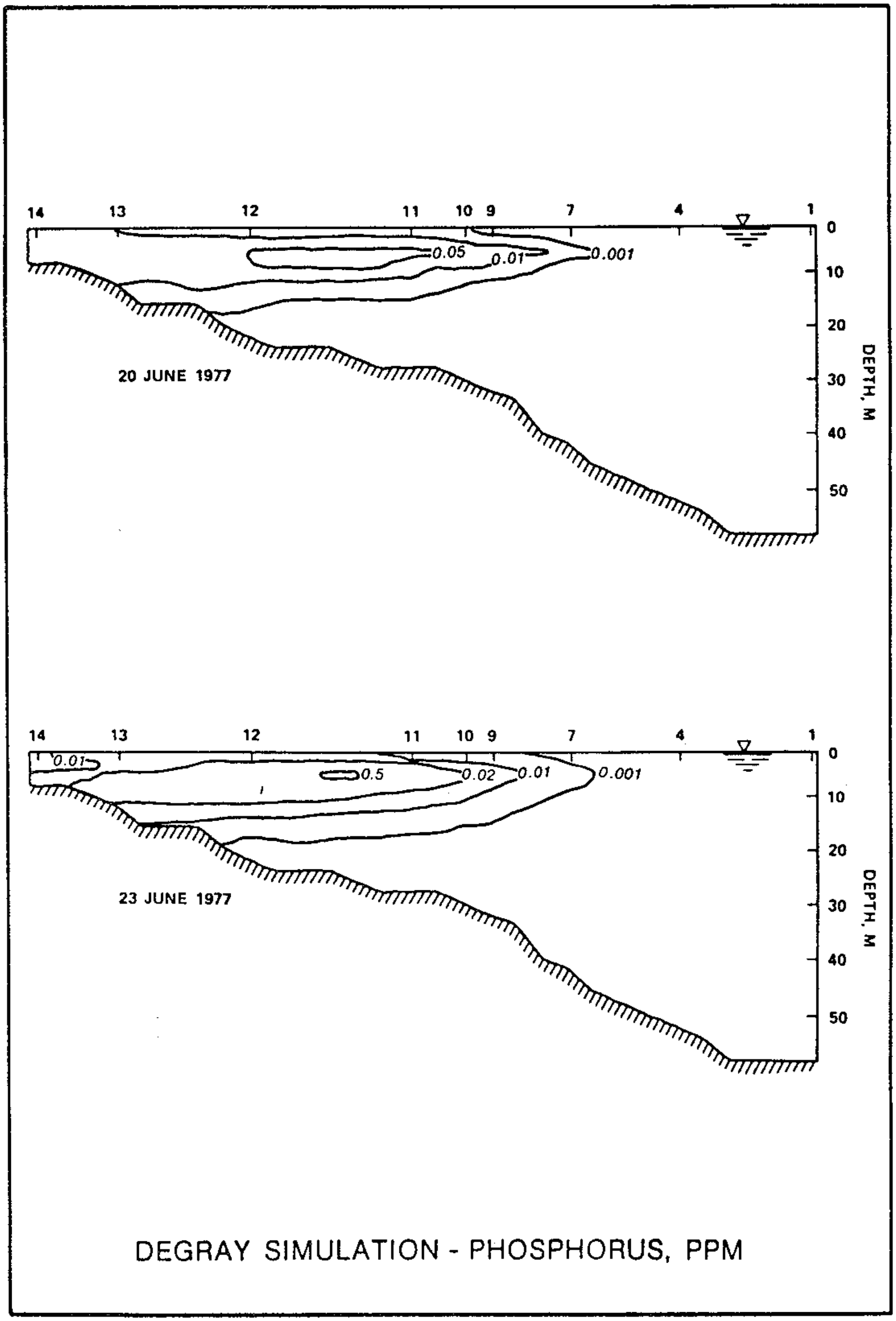

Fig. 16b. Simulated phosphorus loading during the DeGray storm event, 16-22 June 1977.

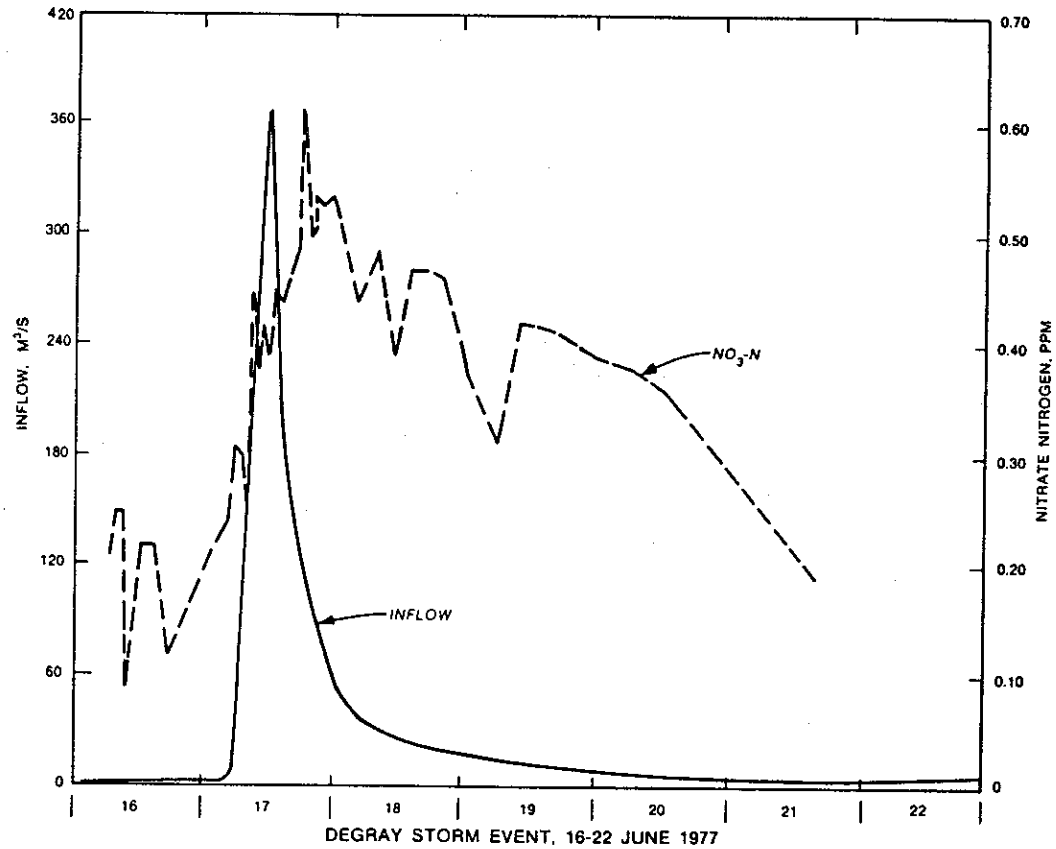

Fig. 17. Time history of inflow and nitrate-nitrogen concentration during the DeGray storm event, 16-22 June 1977.

of approximately .01 mg/1 between 20 and 23 June.

Nitrate was also simulated as a conservative constituent to determine the importance of mixing on in-pool nitrate concentrations. Peak concentrations of 0.30 mg/1 and 0.10 mg/1 were predicted at Stations 12 and 10, respectively. Since nitrate concentrations peaked on the falling side of the hydrograph, the nitrate plume did not progress into the pool as far as the phosphorus plume (Figs. 16a, 16b, 18a, 18b). By 20 June, the center of mass of the nitrate plume was predicted between Stations 13 and 12, with the peak concentration at Station 12 reduced to 0.20 mg/1. Between 20 and 23 June concentrations were predicted to decrease to 0.15 mg/1 at Station 12 and 0.5 mg/1 at Station 10.

The low nitrate concentrations observed in the pool may have resulted from the entrainment of ferrous iron and manganous manganese from the anoxic zone that existed in the headwaters of the pool during this period. Nitrate could have served as an oxidizing agent, donating its electrons to cause the precipitation of ferric iron and manganic manganese while the nitrogen became available for the formation of ammonium. Substantial increases in ammonium were observed in and below the inflow region at Station 12 during the period 18-28 June. No increases in ammonium occurred at Station 10. The plume may not have progressed far enough to affect nitrate concentrations at Station 10 since predictions for 18-23 June show the leading edge of the nitrate plume just at Station 10.

Summary and Conclusions

Storm flows load significantly more suspended solids, nutrients, and bacteria than base flows. The impact of storm loadings on reservoir water quality is still largely unknown. Field measurements quantifying these impacts are rare since they are expensive and difficult to implement; often the data acquired are difficult to interpret. LARM proved to be a valuable tool for evaluating these data.

In this study LARM correctly predicted inflow placement, travel times, dilutions, and coliform die-off. LARM also showed differences in the transport of turbidity, fecal coliforms, nitrogen, and phosphorus through the reservoir. These differences resulted from the unique

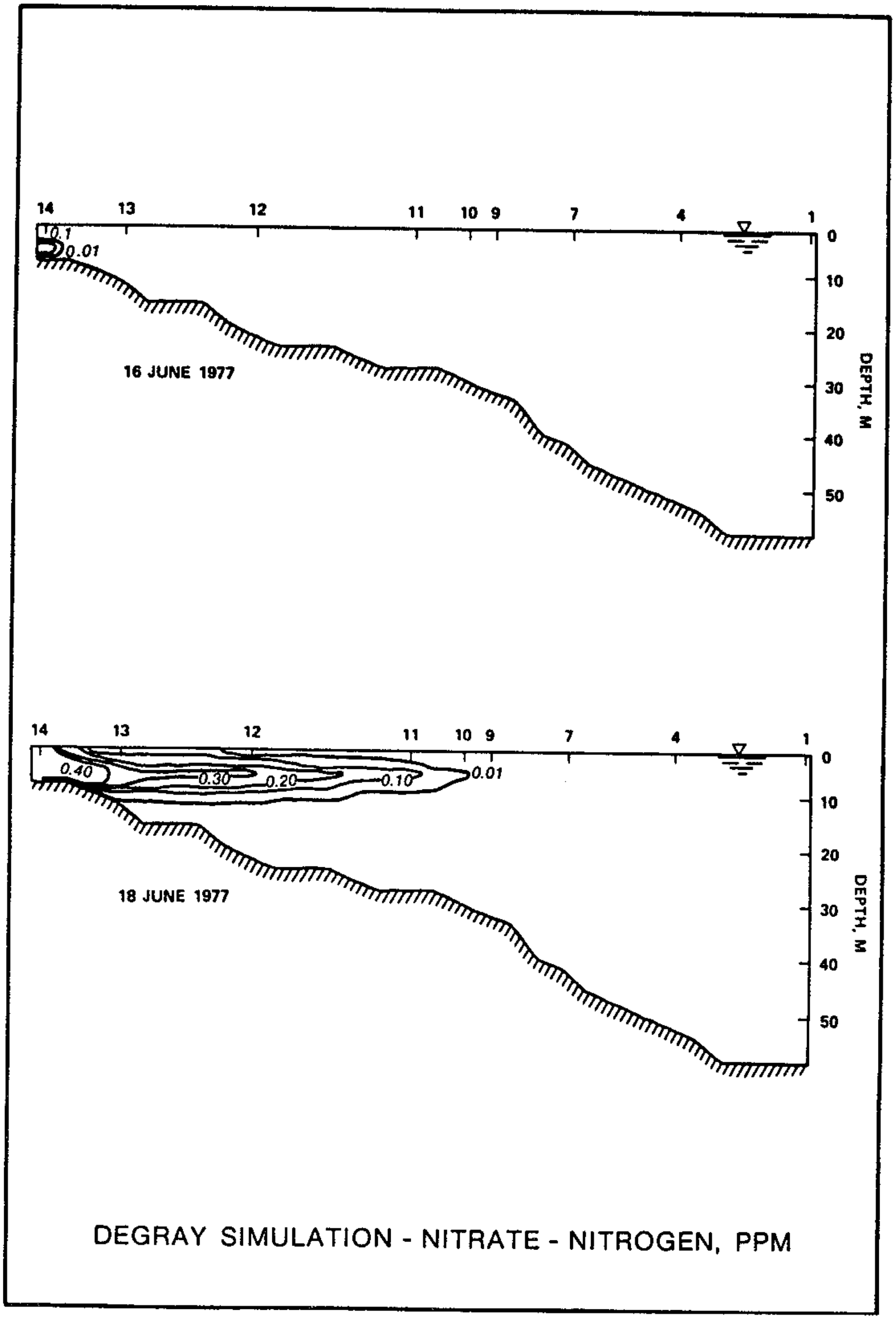

Fig. 18a. Simulated nitrate-nitrogen loading during the DeGray storm event, 16-22 June 1977.

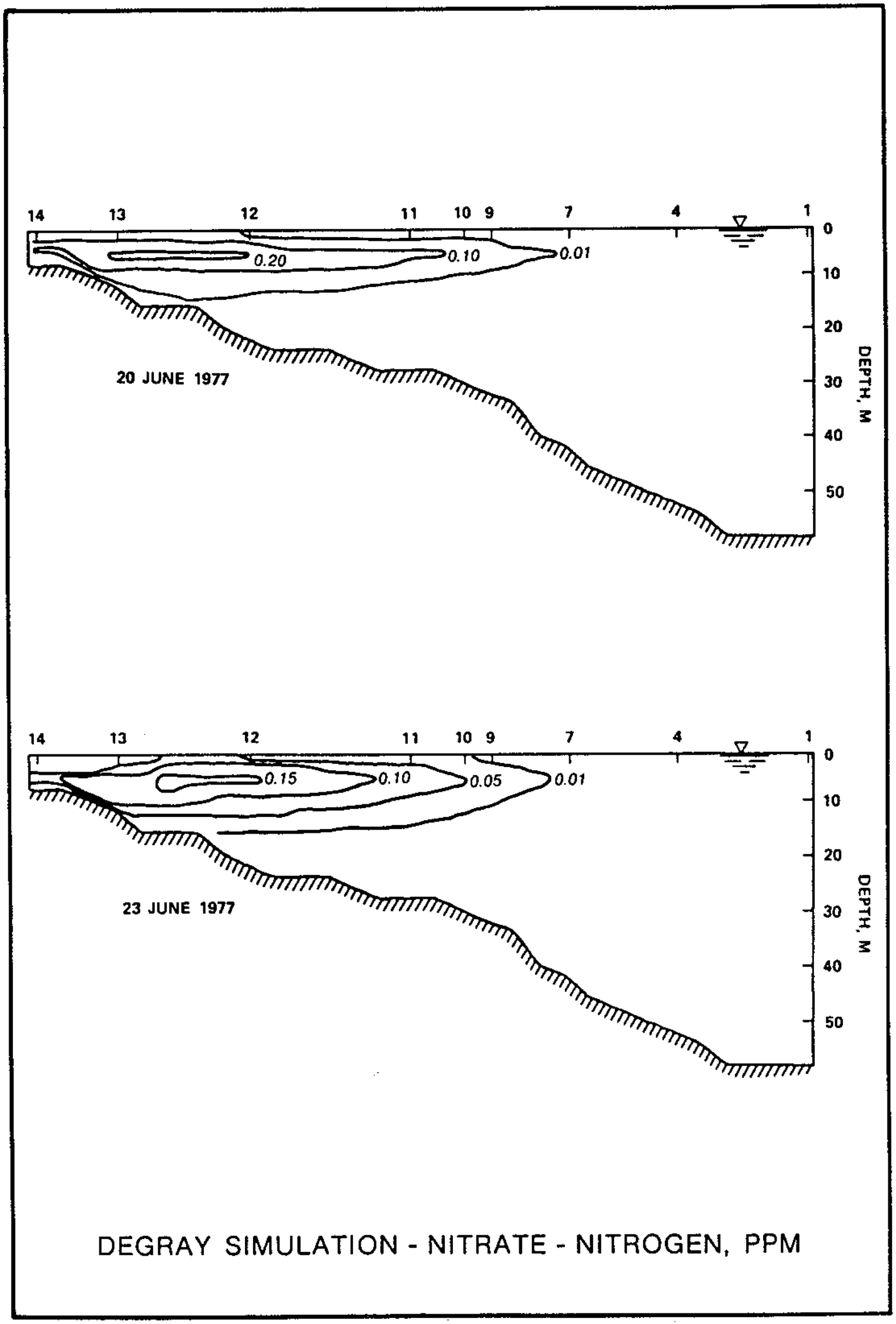

Fig. 18b. Simulated nitrate-nitrogen loading during the DeGray storm event, 16-22 June 1977.

manner in which each constituent loaded on the hydrograph. Because inflow concentrations of turbidity, fecal coliforms, and total phosphorus peaked on the rising side of the hydrograph, their plumes were were predicted to progress farther into the pool than the nitrate plume, which peaked on the falling side of the hydrograph. The simulations showed that mixing and dilution significantly reduced inflow concentrations of all the constituents modeled, but not to the levels observed in the pool. Other physical and biochemical processes appear to be just as important in reducing concentrations in the pool. More processes and interactions must be included in the model, therefore, to determine the total impact of storm flows on reservoir water quality.

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MODELING ANAEROBIC PROCESSES IN RESERVOIRS - DeGRAY LAKE

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Introduction

The ability to predict anoxia in reservoirs has long been a part of reservoir ecosystem models used by the Corps of Engineers (CE). However, anoxia is often associated with other unwelcome water quality effects including release from sediments of phosphorus and reduced forms of nitrogen, manganese, iron, and sulfur. Phosphorus and nitrogen serve to stimulate algal growth which may have contributed, through respiration and organic decay (oxygen-consuming processes), to the development of anoxia in the first place. Iron and manganese cause staining problems in water supplies and require added expense for their removal. Sulfur, in the form of hydrogen sulfide gas, is noxious smelling, toxic, and corrosive. Thus, the ability to predict the onset and extent of potential problems associated with anoxia or low dissolved oxygen (D.O.) conditions would be a valuable asset for reservoir planners and managers.

To this end, the Corps of Engineers, through its Environmental and Water Quality Operational Studies (EWQOS) Program, initiated the task of modifying its one-dimensional, numerical model of reservoir water quality, CE-QUAL-R1 (Environmental Laboratory 1982), to include so-called "anaerobic subroutines." These subroutines were formulated to predict the effects of anoxic conditions on the release from sediments of the nutrients phosphorus and nitrogen; the reduced metals iron and manganese; and reduced sulfur, or sulfide. The algorithms also consider oxidation and reduction reaction pathways associated with the presence of these elements in the water column.

Primary sources of information needed for the development of the new water quality subroutines were the field studies data generated at DeGray Lake. The intensive and extensive long-term studies made it possible to study the temporal and spatial patterns of the system's chemical constituents and to conceptualize how best to model the geomicrobial processes of interest in a manner compatible with the existing one-dimensional water quality model.

The relevant data initially available for modeling processes affected by anoxia came from one year of biweekly sampling for oxygen, orthophosphate, nitrate, ammonia, dissolved and total manganese, dissolved and total iron, sulfate and total sulfide. Several sampling stations along the length of the reservoir provided a longitudinal perspective of the biogeochemical processes. Many sampling depths ranging from the reservoir's surface to its sediments were incorporated in the study, presenting as complete a picture as may exist for any system.

The initial part of the modeling task required collation of data and generation of graphs for analysis. Once graphs were completed, I began the chore of poring over the graphs to elucidate temporal and spatial relationships among anoxia and anaerobic processes in the DeGray Lake system. Two most interesting and perplexing observations concerned the timing of the appearance of reduced substances in the water column. The first seeming oddity was the coexistence of significant concentrations of oxygen and dissolved manganese, iron, and sulfide. If we are to believe traditional redox chemistry, this is quite anomalous behavior. The second "surprise" involved the "simultaneous" appearance of dissolved manganese, iron, and sulfide in the water column. Again, the basic tenets of limnology and redox chemistry would lead one to expect them to appear sequentially through time in order of the standard reduction sequence.

Reconciling these apparent paradoxes, if not a difficult task, required, at least, a complete re-evaluation of how one conceptualizes the biogeochemical processes as they develop in reservoirs. Personally, it might actually have been a first attempt at perceiving a reservoir in a three-dimensional framework since one usually sees data presented on two-dimensional axes representing the body of water as concentration versus depth curves. The final result of this process had to be reducible to a reasonable, one-dimensional representation of the anaerobic biogeochemical dynamics of the reservoir.

In the end, resolving the perceived paradoxes seems rather simple. The solution consists of allowing simultaneous release from the sediments of nutrients, metals, and sulfide when the D.O. concentration falls to 0.5 mg/l, a value given the variable name OXYLIM in the model. This value also initiates reduction processes in the water column. Thus, reduced materials and phosphate can appear simultaneously and concurrently with oxygen. Differences in concentration through time are achieved by controlling individual rates of release. Similarly, reduction rates in the water column are controlled individually.

Another simplification needed to circumvent lack of data on chemical dynamics within the sediments leading to releases is the use of zero-order kinetics (concentration independent) to describe the rates of release from sediments; once the oxygen concentration reaches the 0.5-mg/l level, sediment releases start at fixed rates on an areal basis until the source is depleted or the oxygen concentration rises above 0.5 mg/l. When the D.O. concentration in the water column exceeds 0.5 mg/l, oxidation processes predominate, and sediment releases cease. The following discussion of the structure of the "anaerobic" subroutines should clarify how they operate within the context of the overall water quality model.

Modeling Anaerobic Processes with CE-QUAL-R1

Figures 1-4 describe the framework within which the "anaerobic" subroutines function with respect to considerations of general water quality, hydraulics, and hydrodynamics. The overall organization of CE-QUAL-R1 appears in Figure 1. The anaerobic variables are grouped to the left of the figure, with the exception of the compartments for nitrogen and phosphorus in sediments. Unlike most of the water quality variables, the anaerobic ones are not highly coupled to variables modeling organisms. With the exceptions of phosphorus and nitrogen in sediments, the only linkages common to the rest of the model are to oxygen.

Figure 2 isolates the anaerobic variables and also includes other nitrogen and phosphorus compartments. The variables are divided among those associated with sediments and those occurring in the water column. Processes affecting each variable within a layer are divided between those which take place under oxidizing (D.O. > OXYLIM) or reducing (D.O. $\leq$ OXYLIM) conditions.

For purposes of thermal, hydrodynamic, and hydraulic modeling, CE-QUAL-R1 divides a reservoir into a series of horizontal layers (Figure 3). This layer structure is carried over into water quality modeling. All water quality variables pictured in Figures 1 and 2 are calculated for each layer of the reservoir which the model simulates.

In addition to the interactions and reactions among the water quality variables in each layer, exchanges of materials between layers due to mixing processes are also considered. Figure 4 depicts the various ways in which a compartment may lose or acquire material. Inflow, outflow, diffusion, and settling are physical processes which directly deplete or add to the mass of a compartment.

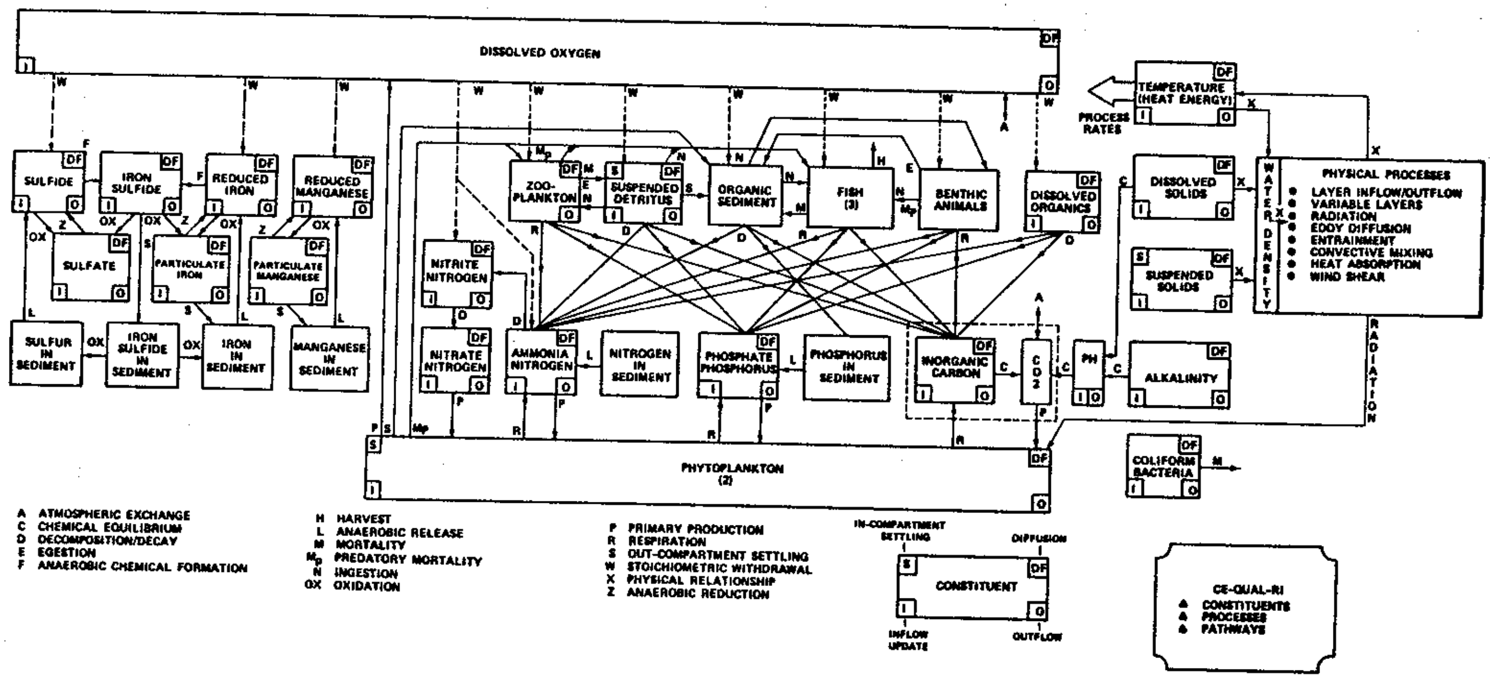


Figure 1. Overall organization of CE-QUAL-R1.

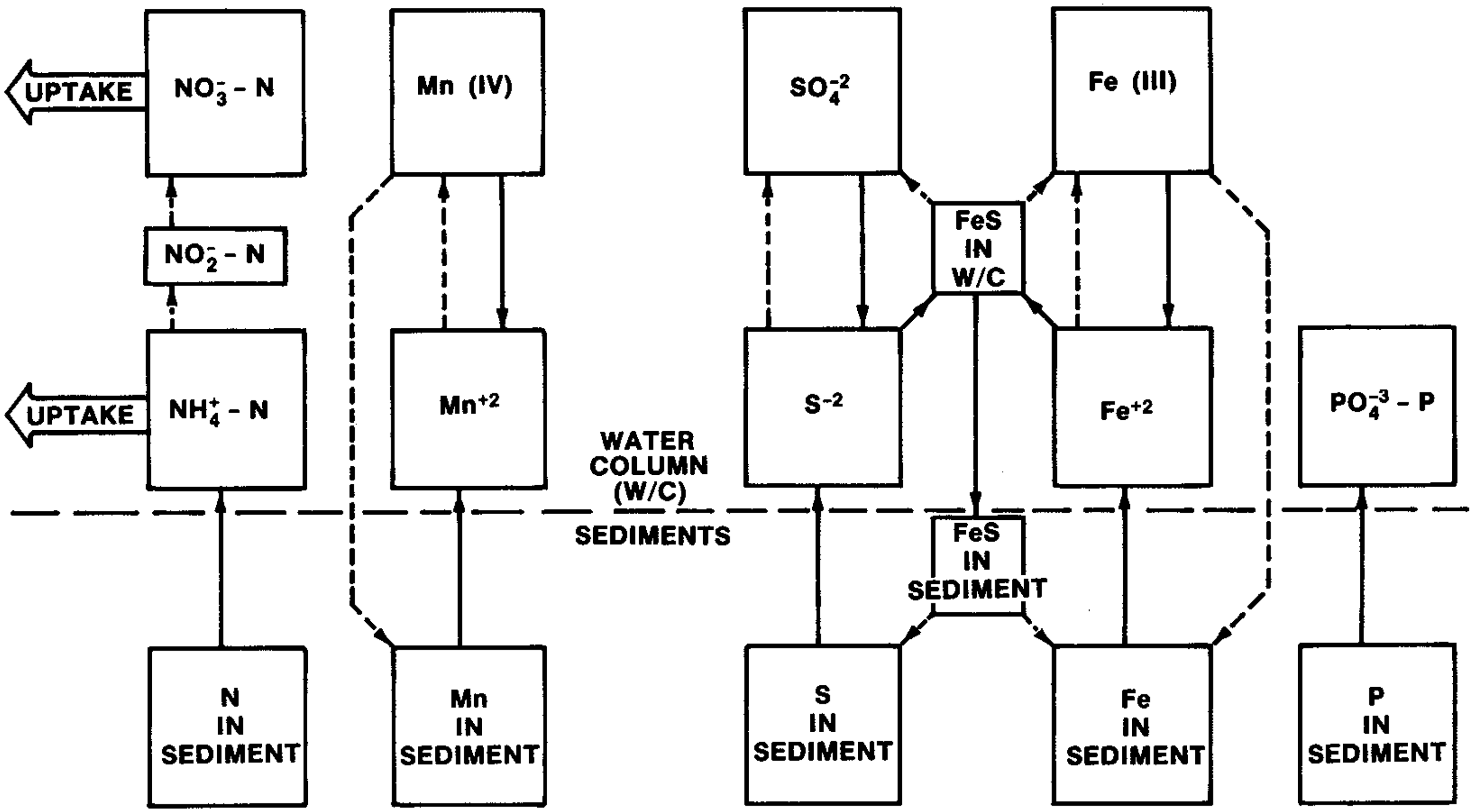


Figure 2. Simplified chemistry associated with anaerobic variables.

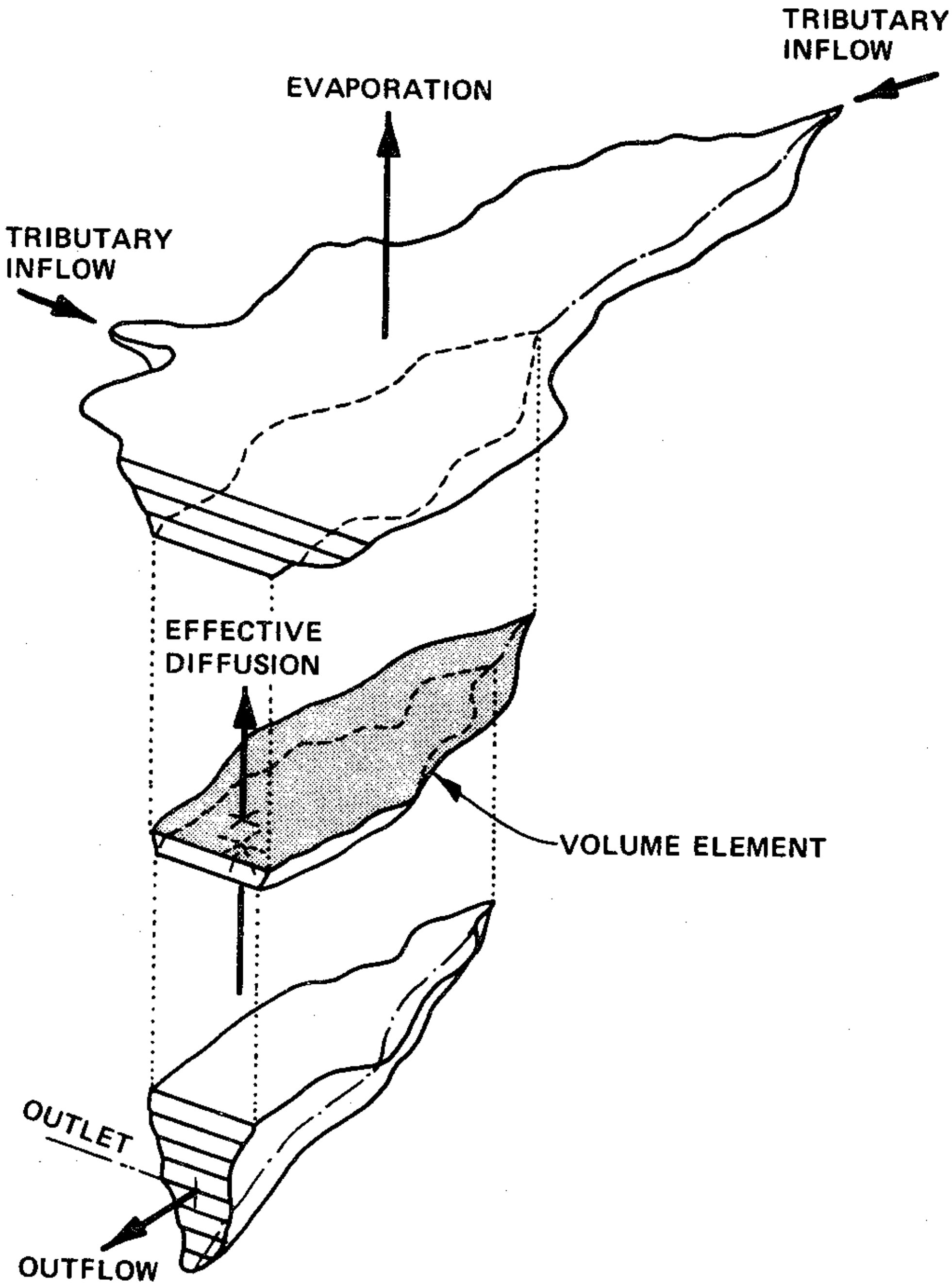


Figure 3. Geometric representation of a stratified reservoir and mass transport mechanisms.

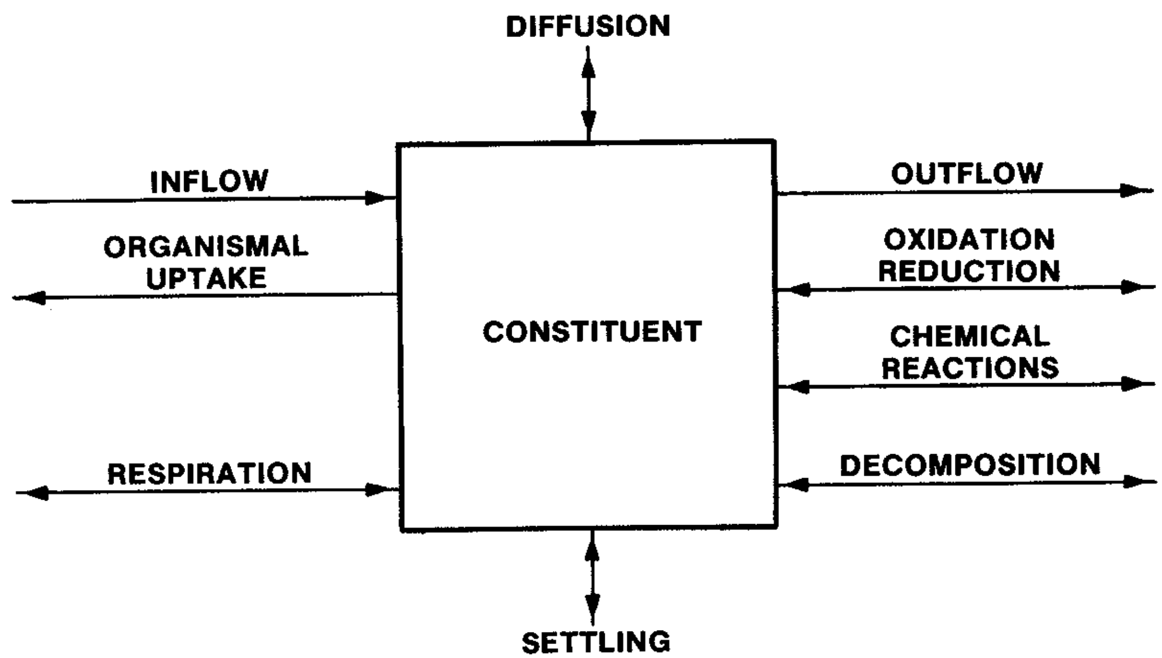


Figure 4. Potential compartment fluxes.

Organisms may directly remove compartment constituents through absorption of nutrients or ingestion. Respiration and decomposition can deplete organic components of compartments by rendering them into their primary inorganic chemical constituents; these processes also release their products to nutrient compartments. Implicit geomicrobial mediation effects changes in redox state and chemical speciation as do chemical reactions.

In order to achieve reasonably accurate simulations of conditions associated with anoxia in reservoirs, it is first necessary to predict temporally and spatially the onset of anoxia. Figure 5 depicts a nominal simulation of the development of anoxia in the metalimnion of DeGray Lake in 1979. The simulations yield profiles which match the timing and depth of the anoxic metalimnion. Hypolimnetic oxygen depletion occurs at a slightly too rapid rate.

Improvement of fit to field observations may be achieved by modifying select coefficients. The following example is chosen to demonstrate model sensitivity and to suggest possible ways to improve fit. Figures 6a and 6b demonstrate the effect of doubling rates of decay for dissolved organic matter and sediment--processes to which the model is extremely sensitive. It can be seen that doubling the DOM decay rate has a major effect on the oxygen concentration throughout the water column, while doubling the sediment decay rate has a minimal influence on D.O. in the epilimnion and metalimnion, but affects the hypolimnion somewhat more. Such data would suggest that improvement in the hypolimnetic oxygen depletion rate may be achieved through judiciously decreasing the sediment decay rate.

Figure 7 presents a series of profiles describing the distribution of manganese, iron, and sulfide in the water column after their release from sediments under anoxic conditions. As references, oxygen profiles are shown from the same dates to allow consideration of some of the difficulties one faces when developing a model from field data which occasionally do not follow

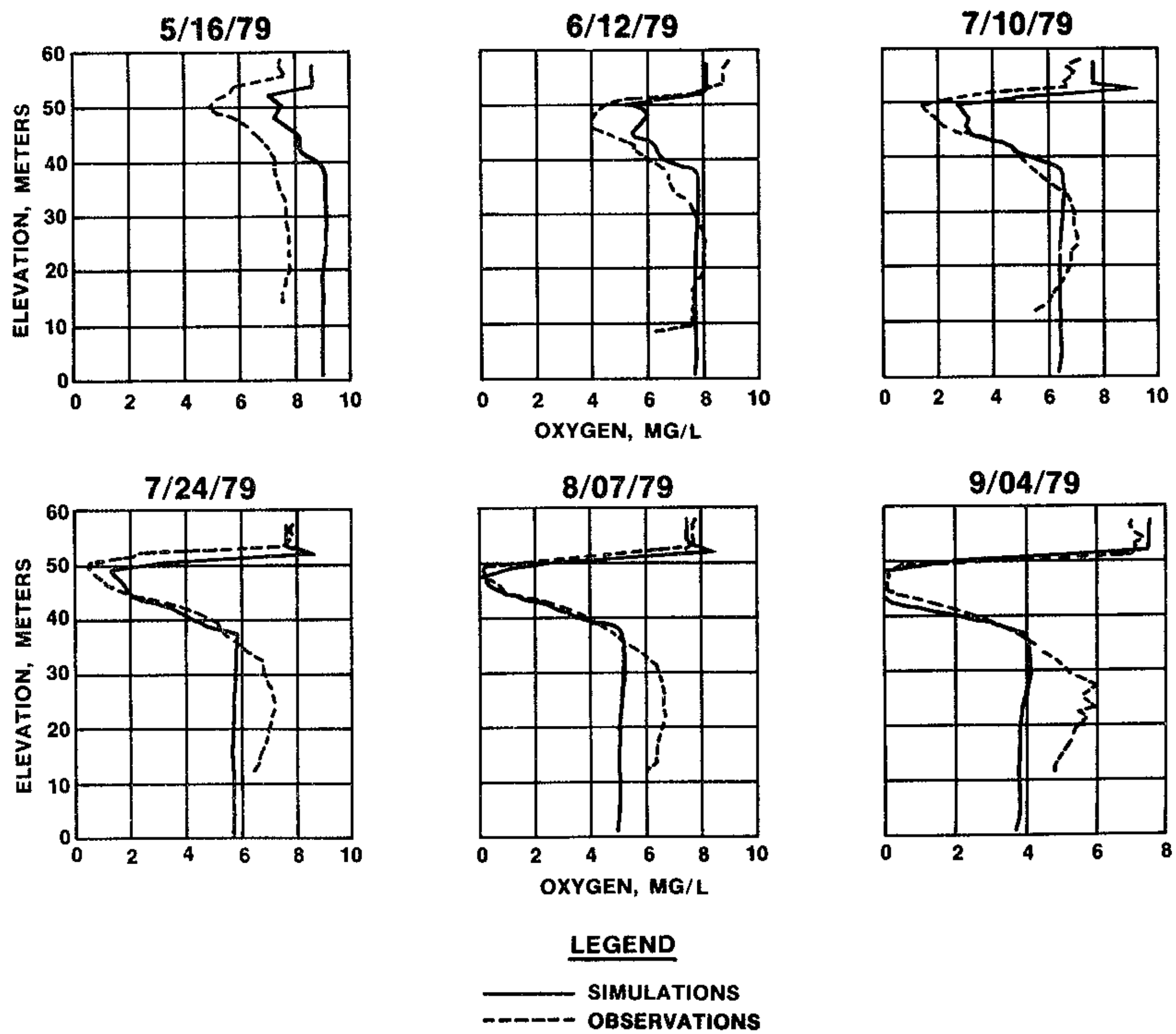


Figure 5. Development of anoxia in DeGray Lake in 1979.

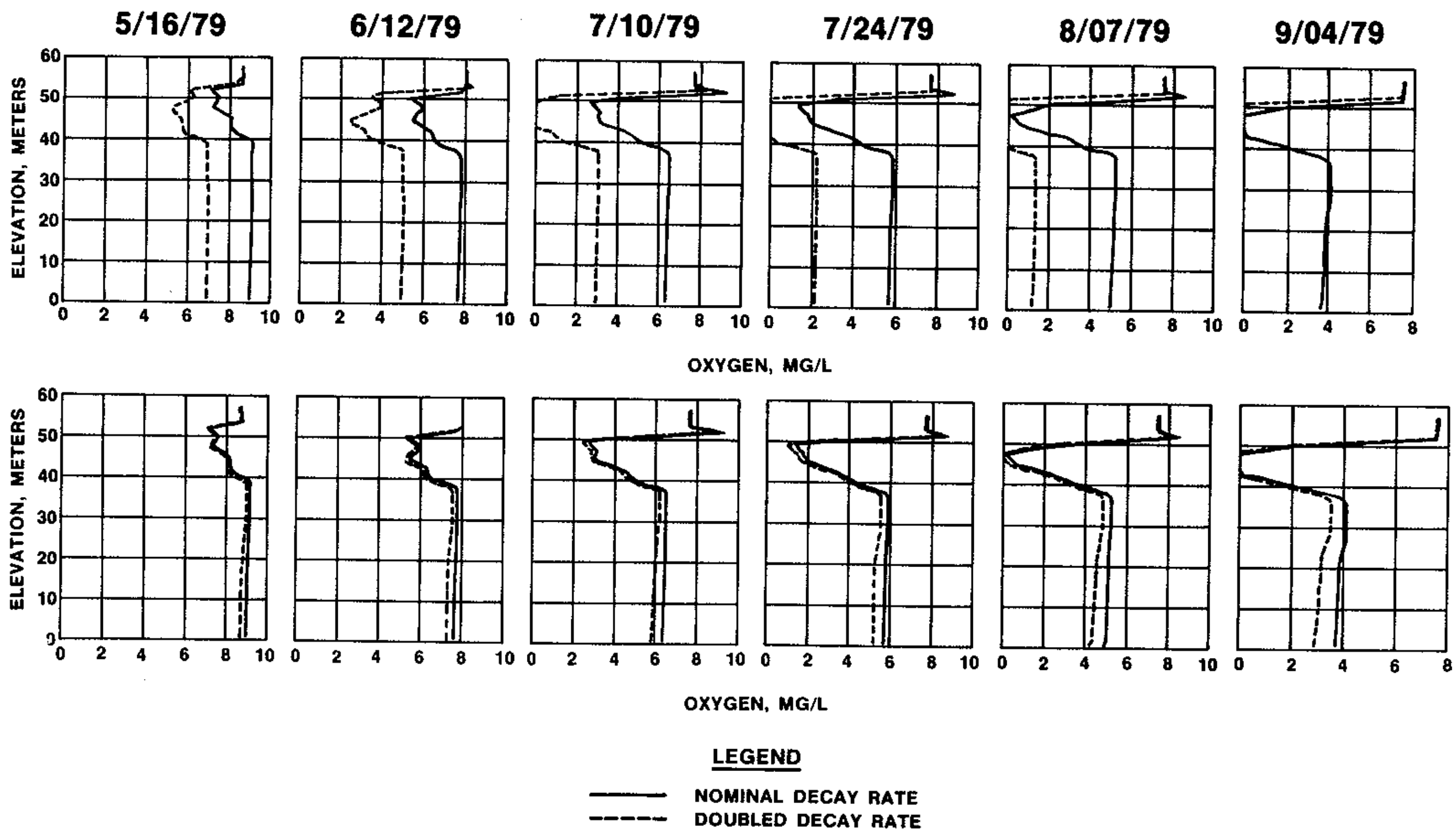


Figure 6. (a) Effect of doubling dissolved organic matter decay rate on dissolved oxygen concentration (top row). (b) Effect of doubling sediment decay rate on dissolved oxygen concentration (bottom row).

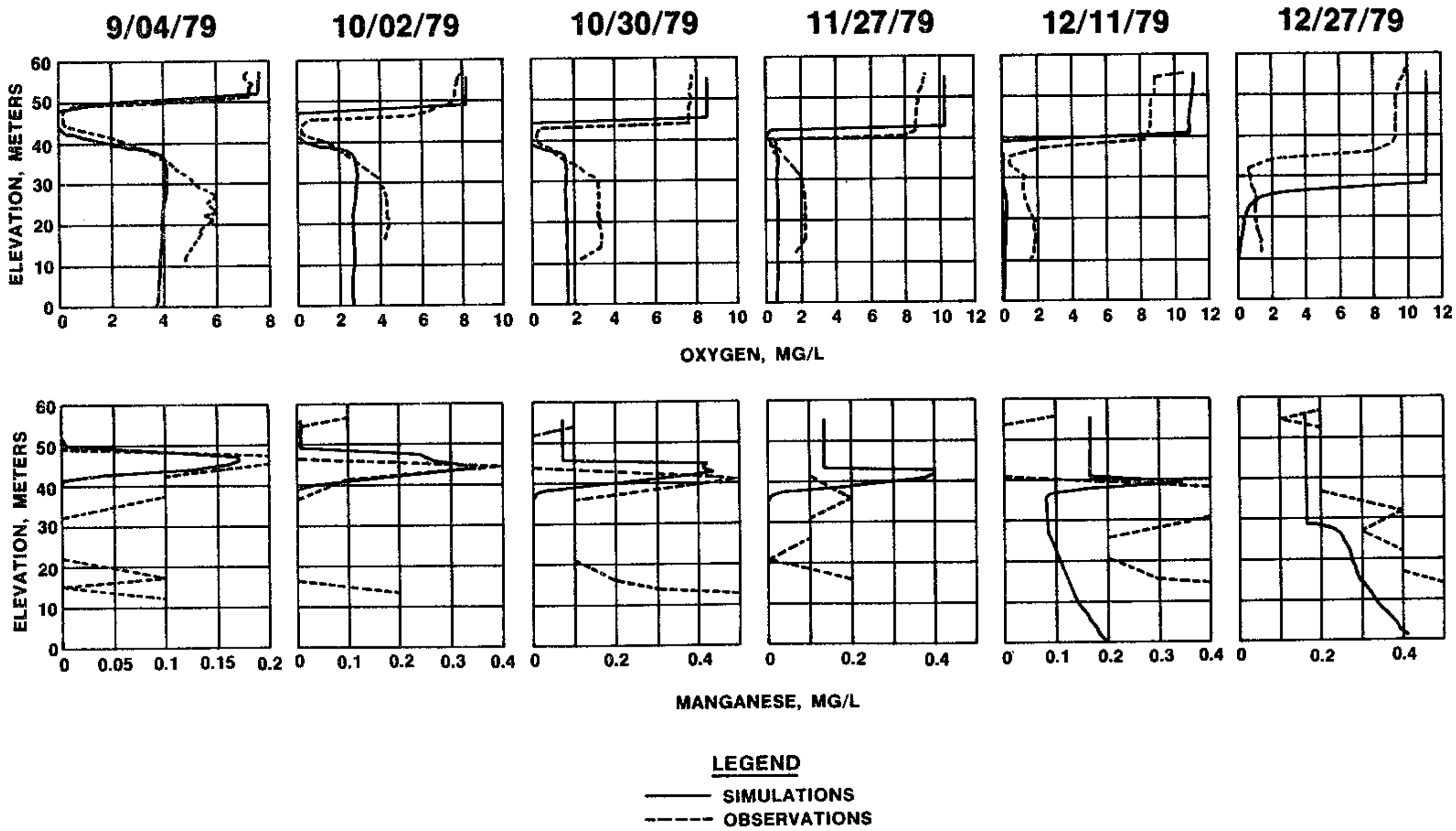
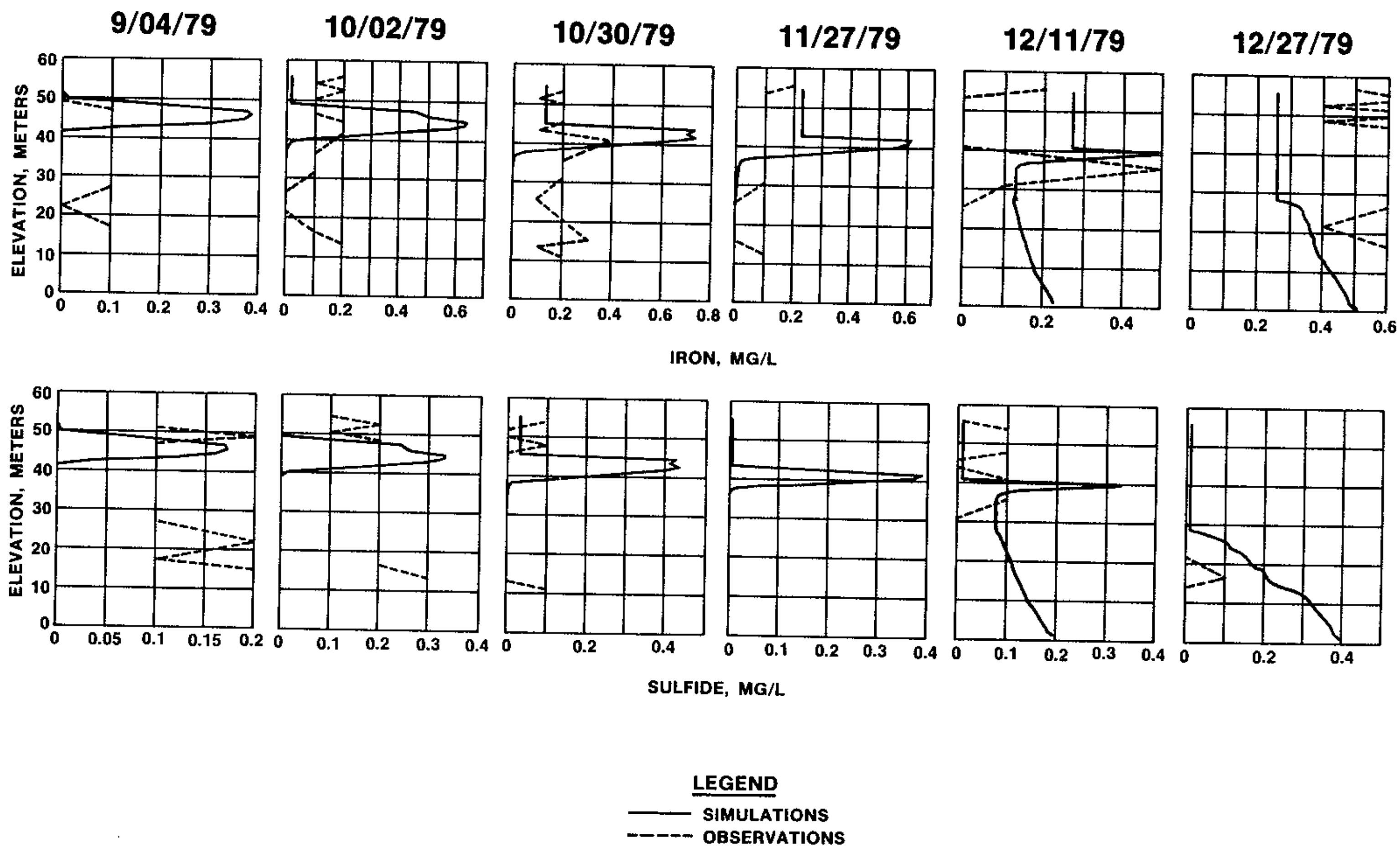


Figure 7. Comparison of (a) Oxygen and manganese concentration profiles when anoxic conditions were present in DeGray Lake in 1979. (Continued).



b. Iron and Sulfide

Figure 7. (Concluded)

expected trends. For the most part, the nominal simulation of D.O. matches field observations fairly well. Some deviation can be seen in the epi- and hypolimnetic regions. The simulations at the surface predict more oxygen than observed and the deepest regions show less. Further model calibration can improve fit, but it is not the intention of this paper to follow that detailed procedure.

As might be expected, the simulations for manganese, iron, and sulfide exhibit very similar profiles. The reasons for this are quite clear. Control over anaerobic release of all these materials is mediated through the D.O. concentration. Distribution in the water column is also influenced by hydrodynamics which affect each element identically. The differences in the three series of simulations are largely due to differences in oxidation rates--coefficients which can be altered during the calibration process.

As stated previously, sediment release of dissolved (reduced) materials occurs simultaneously with D.O. depletion of the OXYLIM value. Release rates are calibrated to correspond to concentrations observed in the metalimnion where D.O. is first exhausted. The major discrepancy in the model vis-à-vis observed conditions is found in the simulations of "anaerobic" materials in the hypolimnion. Field data indicate that dissolved manganese and iron and sulfide are present in the hypolimnetic waters even when D.O. concentrations exceed 3-4 mg/l. It is not until December when the simulated hypolimnetic D.O. concentrations decrease below the OXYLIM value that manganese, iron, and sulfide appear according to the model. An important fact may mitigate this shortcoming of the model: the model simulates a location in DeGray Lake which does not actually correspond to the sampling site. The sampling site is approximately 13 meters shallower than the simulated reservoir depth. Thus, a depth of 42 meters at the sampling site may represent the bottom of the reservoir where sediment-water interactions are important, while the corresponding depth in the simulations is still 10-15 meters above the bottom in the water column.

Another discrepancy may not be so easily resolved. While D.O. steadily decreases in both field observations and simulations, it is not the case that the concentrations of dissolved or reduced materials constantly increase. A certain amount of inconsistency or variability is inevitable. But the systematic decreases in manganese, iron, and sulfide observed in the 27 November samples are not simply reconciled. These unexpected differences may be ascribed to several causes. It is conceivable that samples were mishandled but this is highly unlikely as the effect is so broad. It is more probable that hydrologic effects not apparent from examination of a small subset of the data are responsible. In fact, a temporary hiatus in the depletion of D.O. appeared to have occurred in the month of November. During this period, the otherwise steady progression of events stalled, and the concentrations of anaerobically generated materials decreased. The exact biogeochemical and hydrological events that transpired can only be the subject of conjecture. In any event, the simulations do manage to continue to mirror the general details of anaerobic conditions. Their deviations from "expected" or actual behavior demonstrate the difficulty in producing truly accurate simulations in a stochastically varying environment.

Summary and Conclusions

In this paper, I have attempted to present a background discussion for the development of subroutines describing biogeochemical events associated with anoxia in reservoirs. The problems in conceptualization of a new modeling approach for inclusion in an existing one-dimensional model were stressed. These difficulties arose from attempts to reconcile traditional methods of studying anaerobic processes, such as isolated chambers in laboratory situations, with more poorly defined environmental conditions where hydrodynamics and naturally varying conditions may generate unsteady conditions.

Finally, modeling results are presented. Simulations demonstrating critical aspects of modeling relevant to the prediction of the onset of anoxia are described. Coefficients used are somewhat dated; i.e., the coefficients used in the simulations presented here originated from an earlier calibration study, but, since that time, corrections and improvements have been incorporated into the thermal portions of the model. In spite of these changes, the water quality simulations are still acceptable and future recalibration should improve them.

The additions made to CE-QUAL-R1 and described here should provide reservoir planners and managers with increased flexibility in considering dam operation alternatives with respect to water quality problems.

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MONITORING COLDWATER DISCHARGES FROM DeGRAY RESERVOIR USING THERMAL INFRARED IMAGERY

by

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ABSTRACT

The aquatic environment downstream from large reservoirs is controlled to a large extent by the reservoir releases. If the reservoir has a hypolimnial release, the discharge may be relatively cold. Reservoir release should be addressed as a whole, rather than a few one-dimensional in situ points, which may not adequately relate thermal patterns. Thermal infrared (IR) imagery (aircraft platform) was utilized to observe the relative temperature differentials. This technique revealed thermal mixing patterns (across and downstream) of DeGray Reservoir to a point several miles below the dam.

INTRODUCTION

The life associated with the aquatic environment in any location has its species composition and activity regulated by water temperature. Essentially, all of these organisms are "cold-blooded" or poikilothermous; that is, the temperature of the water regulates their metabolism and ability to survive and reproduce effectively. Industrial uses for process water and for cooling are regulated by water temperature. Therefore, temperature is an important physical parameter which to some extent regulates may of the beneficial uses of water (Coutant 1967).

A major source of thermal variation is hypolimnetic releases from reservoirs. The three Vicksburg District, Corps of Engineers, reservoirs in Arkansas--Lake Ouachita, Lake Greeson, and DeGray Lake--all contribute hypolimnetic releases to the Ouachita Basin.

Lake Ouachita resulted from impoundment of the Ouachita River behind Blakely Mountain Dam, constructed by the Corps of Engineers in 1952, and is the uppermost of a series of three reservoirs on the Ouachita River. Lake Hamilton and Lake Catherine are located immediately downstream of Lake Ouachita. The two lower reservoirs were constructed by Arkansas Power and Light Company.

The penstock at Blakely Mountain Dam provides for a single-level outlet 84.0 feet below the normal pool elevation of Lake Ouachita. The lake is extremely deep (193 feet at maximum power pool) and stratifies with the penstock in the hypolimnion. The resulting discharge from the lake is considerably colder than temperatures normally found in streams in central Arkansas. Variable discharge rates, due to power generation at Blakely Mountain Dam, result in extreme water temperature fluctuations (Pers. Commun., 1979, C. K. Baxter, US Fish and Wildlife Service, Vicksburg, Miss.).

Lake Greeson resulted from impoundment of the Little Missouri River behind Narrows Dam, constructed by the Corps of Engineers in 1950. The penstock provides for a single-level outlet 68.0 feet below the maximum power pool of 548 feet NGVD. Lake Greeson is a deep (158 feet at maximum power pool), coldwater-release lake that exhibits strong stratification throughout 9 to 10 months of the year (Stafford 1978).

DeGray Reservoir was created by a dam built on the Caddo River in 1969. DeGray Dam allows for epilimnial, intermediate, or hypolimnial discharge. The maximum power pool elevation is 408 feet NGVD. The multioutlet intake structure allows water to be selectively withdrawn from one of the three 21- by 21-foot openings, the midpoints of which are at elevations 395.0, 380.0, 355.5 NGVD, respectively. All releases during data collection were made from the lower (hypolimnetic) inlet.

Releases from the upstream reservoirs in the Ouachita system can result in turbulent flows in the Ouachita River, ranging from 500 to 65,000 cfs with a maximum of 170,000 cfs on record at Arkadelphia, Arkansas. Much of the release comes in the form of hydropower generation. This type of release is usually highly variable in volume (20 to 6,000 cfs at DeGray) and in time (5 minutes to 24 hours), which results in extreme water temperature variations. During late summer and fall, the water temperature of the Ouachita River immediately below Lake Ouachita is, at times, as much as 20° C below normal.

APPROACH

During reservoir releases, in situ temperature measurements can give only one-dimensional data. However, due to the turbulent mixing patterns, a two-dimensional measurement could give a much better understanding of associated thermal patterns. To map two-dimensional patterns, an instaneous data collection over the entire study area would be needed. Remote sensing would be the only possible means to carry out the necessary data collection. Two types of thermal remote sensing data are available: aircraft and thematic mapper (TM) aboard LANDSAT 4. TM data were not utilized for several reasons:

1. Problems aboard LANDSAT 4 have limited data acquisition.
2. Ground resolution is 120 by 120 meters, which may be too coarse for the intended purpose.
3. The timing of reservoir releases with overpasses further limits available data.

It was therefore determined that aerial photography was the best means of obtaining the required data.

To meet data needs, aerial thermal infrared imagery was collected. The imagery was flown by Tennessee Valley Authority, Mapping Services Branch. A gyrostabilized Daedalus scanner was used to measure thermal energy in the 8- to 14-μm wavelength band and record the results on 5-inch-wide photographic film. Flight altitude was approximately 5,000 feet for a minimal scale of approximately 1:20,000. The scanner instantaneous field of view was approximately 2.5 meters on the ground. The original film negatives give reference blackbody densities which are related to apparent temperature. The film images were digitized, by an analog to digital process. This transformation (film to digital) is, in principle, based on a light source behind a transparent photograph. The variations on optical density in the photograph result in variations of light intensity at a photodetector (closed-circuit video camera), which result in an appropriate voltage change in the output. This output signal is then sampled at appropriate time intervals during scanning, and each sample voltage is converted into its appropriate digital value. This digital value is then stored on a computer-compatible medium (Department of the Army 1979). These digital data were calibrated by using a 5 by 5 matrix to correspond with ground truth data. Display of data was on an RGB color monitor as a visual aid in the digital analysis. Figure 1 is a computer reproduction of digital data that were stored on floppy discs, with arbitrary graytone values. Figure 2 is an enhanced (density slice) portion of Figure 1.

DISCUSSION

This type of aerial photography can provide a useful two-dimensional pattern of surface temperature. In the investigation of the effect of thermal discharge on the downstream environment, it becomes important to recognize "cold spots" and warm areas. The hydrodynamics of streamflow also become obvious in the bendways. The graytones in Figure 2 indicate darkest areas are related to surface water temperatures below 21.6° C and the lightest above 26.4° C, with a linear correlation between these values.

The influence of the Caddo and Ouachita Rivers shows a relatively common mixing pattern. However, the colder water (darker graytone) does appear to submerge below the warmer (lighter graytone) surface water. Bendways downstream of the confluence show a turnover or surfacing of the denser water. Appearance of "cold spots" other than bendway would indicate shallow areas. These areas would indicate minimal energy transfer since graytones appear consistent. Analysis of additional photographs indicates that hypolimnetic hydropower generation from DeGray (or in combination with Blakely Mountain) has thermal effects for several miles downstream. These thermal variations in stream cross section (bank to bank) are not sensitive to the human eye, unless the information is modified as in Figures 1 and 2. The establishment of

Fig. 1. Computer reproduction of digital data

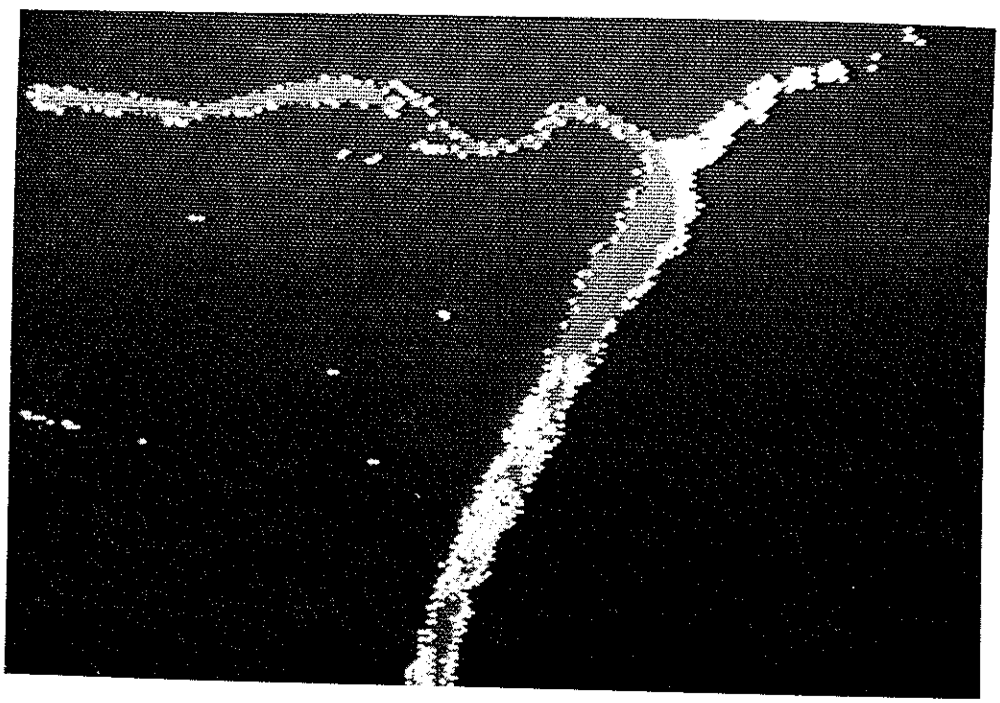

Fig. 2. Enhanced (density slice) of Caddo and Ouachita Rivers, Arkansas

thermal flow patterns can be related to other physical conditions, such as:

1. Percent saturation of dissolved oxygen.
2. Reaction rates.
3. Suspended solids.
4. Hydraulic mixing patterns.

This cross-sectional information would be desirable in the location of monitoring and sampling sites. Variations in data collection could be related to these thermal patterns.

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RESERVOIR LIMNOLOGY: AN OVERVIEW

by

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Abstract

A comparative analysis of structural and functional characteristics of reservoir and natural lake ecosystems indicated strong similarities in all functional, metabolic properties. The irregular and erratic changes in physical and certain chemical properties of reservoirs leads to a number of aborted biotic growth and development responses that result in reduced diversity, less specialization, and increased ecosystem instability in reservoirs. Based on existing knowledge, numerous reservoir management techniques could be modified to increase the biotic stability, as well as the longevity, of reservoirs.

Introduction

A basic premise of reservoir limnologists is that these ecosystems are different from natural lakes. In certain characteristics reservoirs do have properties that are fundamentally different from those of most natural lake ecosystems. One can ask first how do reservoirs differ from natural lakes. A more important question, however, is what properties of the two types of ecosystems are functionally the same.

These questions are neither trite nor simple; they are, however, important. The overriding objective in scientific inquiry into the operation of the natural world is to seek unity. We seek the functional commonality that exists amongst our incomplete understanding from an array of informational segments.

In the 1920s and 1930s, a few perceptive limnologists attempted to understand aspects of commonality among lakes and general mechanisms that led to observed differences in biota and their productivity. Acceptance of their understanding was long delayed by the publication of hundreds of reports of often small structural differences in lake properties. These differences led to an excessive classification, the lake typology, which simultaneously resulted in a greater understanding of specific lakes but impeded appreciation of functional similarities in aquatic ecosystems.

Lake ecosystems are divisible into their structural and functional characteristics. Clearly, the physical and certain chemical properties of reservoirs are more chaotic and irregular than exist in natural lakes. It is of course important that these specific structural differences be understood in sufficient detail to permit effective management and utilization of reservoirs. Understanding of biotic and metabolic responses to these more dynamic properties of artificial lakes, however, focuses largely upon time, i.e. is the time required for population and community responses to develop adequately before structural properties are again altered beyond physiological tolerance and reproductive limits. Within those chaotic structural alterations, the biota function similarly in both reservoirs and natural lakes; usually the differences are only a matter of extent of process intensities and rates at which they occur.

Comparison of Properties between Reservoir and Natural Lake Ecosystems

The major physical, chemical, and biotic characteristics of reservoirs have been summarized a number of times (e.g. Margalef, 1975; Ryder, 1978; lengthy review of Thornton, et al., 1984). As information on the properties of reservoir ecosystems accumulates, it is clear that diverse characteristics prevail from marked regional variations in climate, morphology, geology, and hydrodynamics which make generalizations difficult. Because of the greater stability that is usually found in the structural and functional properties of natural lakes, it is instructive to attempt a sequential comparison of these ecosystems with the more irregular characteristics of reservoirs (Table 1).

Reservoirs are constructed for human exploitation primarily in regions where natural lakes are sparse or unsuitable (e.g. saline). Effects of this distribution are numerous. Climate commonly tends to be more extreme in range, water temperatures are somewhat higher, growing seasons are longer, and precipitation inputs are closely balanced to, or less than, evaporative losses in reservoirs than in lakes (Wetzel, 1984). The drainage basins of reservoirs are nearly always much larger in relation to the water surface area than is the case among natural lakes (Table 1). The morphometry of reservoir basins is usually dendritic, narrow, and elongated

Table 1. Comparative characteristics and properties between reservoir and natural lake ecosystems (from Wetzel, 1984).

Properties	Reservoirs	Natural Lakes
Geographical distribution	Predominantly southern (N. Hemisphere) in non-glaciated regions	Predominantly northern in glaciated regions
Climate	Precipitation often low and evaporation high or greater than precipitation	Precipitation commonly exceeds evaporative losses
Drainage basins	Usually narrow, elongated lake basin in base of drainage basin; area large in comparison to lake area (ca. 100:1 to 300:1)	Circular, lake basin usually central; area usually small in comparison to lake area (ca. 10:1).
Shoreline development	Great; astatic	Relatively low; stable
Water level fluctuations	Large, irregular	Small, stable
Thermal stratification	Variable, irregular; often too shallow to stratify in riverine and transitional zones; often can temporarily stratify in lacustrine zones	Natural regime; often dimictic or monomictic
Inflow	Most runoff to lake via river tributaries (high stream orders); penetration into stratified strata complex (over-, inter-, underflows) and often flow is directed along old riverbed valley	Runnoff to lake via small tributaries (low stream orders) and diffuse sources; penetration into stratified waters small and dispersive
Outflow (withdrawal)	Highly irregular with water use; withdrawal from surface layers or from hypolimnion	Relatively stable; surface water
Flushing rates	Short, variable (days to several weeks); increase with surface withdrawal, disruption of stratification with hypolimnetic withdrawal	Long, relatively constant (1 to many years)
Sediment loading	High with large drainage basin area; flood plains large; deltas large, channelized; gradation rapid	Low to very low; deltas small, broad; gradation slow
Deposition of sediments	High in riverine zone, decreasing exponentially down-reservoir; greatest in old riverbed valley; highly variable rates seasonally	Low, limited dispersal; relatively constant rates seasonally
Suspended sediment in water	High, variable; high percentage clay and silt particles; turbidity high	Low to very low; turbidity low
Allochthonous particulate organic matter (POM)	Moderate, especially fine POM during spates and inundation of floodplains	Low to very low
Water temperatures	Somewhat higher (generally more southern climate)	Generally lower (concentrated in more northern climatic regions)
Dissolved oxygen	Somewhat lower solubilities (higher temperatures); great horizontal variability with inflow, withdrawal, and POM loading patterns; metalimnetic oxygen minima more common than maxima	Somewhat higher solubilities (lower temperatures); small horizontal variability; metalimnetic oxygen maxima more common than minima
Light extinction	Horizontal gradients (in kilometers) predominate; light extinction irregular and often very high, particularly in riverine and transitional zones from abiogenic particulate matter; euphotic zone commonly increases in lacustrine zones	Vertical light gradients (in meters) predominate; variable but relatively low extinction from dissolved organic compounds and biogenic particulate matter.
External nutrient loadings	General higher than to natural lakes (larger drainage basin, more human activity, greater water level fluctuations); variable, often unpredictable	Variable but relatively predictable; loadings often moderated by biogeochemical influences of wetland/littoral interface zones
Nutrient dynamics	Horizontal gradients predominate; dependent upon sedimentation rates, residence times, and flow regimes; concentrations in water decrease with distance from headwaters; irregular internal loading	Vertical gradients dominate; often low internal loading, particularly in lakes without severe culturally induced eutrophication

(Continued)

Table 1 (Continued)

Properties	Reservoirs	Natural Lakes
Dissolved organic matter (DOM)	Allochthonous and benthic sources predominate; irregular, often high; refractory DOM predominates	Allochthonous and littoral/wetland sources predominate; relatively constant, often high; refractory DOM predominates
Littoral zone/wetland	Irregular and limited by severe water level fluctuations	Dominates primary production in most lakes; important to regulation of nutrient and dissolved and particulate organic matter loadings
Phytoplankton	Marked horizontal gradients; volumetric primary productivity (or P_{max}) decreases from headwaters to dam; areal primary productivity relatively constant horizontally; light and inorganic nutrient limitations predominate	Vertical and seasonal gradients predominate; small horizontal gradients; light and inorganic nutrient limitations predominate
Bacterial heterotrophy	Pelagic, particle-associated and benthic bacterial heterotrophy predominates in riverine zones	Benthic and littoral/wetland bacterial heterotrophy predominates in most lakes
Zooplankton	Maximal development common in transition zone; horizonal patchiness high; particulate detritus (including adsorbed DOM) variably augments phytoplankton as food source	Vertical and seasonal gradients predominate; horizontal patchiness moderate; phytoplankton is a predominant food source
Benthic fauna	Low diversity with minimal and irregular littoral zone; productivity low to moderate; initially high with inundated terrestrial vegetation	Moderate to high diversity; productivity moderate to high
Fish	Predominantly warm-water species composition; differences often related to initial stocking; spawning success variable (low with low water levels), egg mortality increases with siltation, larval success reduced with less refugia; productivity initially (5-20 y) high, then decreasing. Occasional two-story fishery (warm- and cold-water species) successful, particularly in mountainous reservoirs.	Warm- and cold-water species composition; spawning success good, egg mortality lower, larval success good; productivity moderate
Biotic community relationships	Diversity low; niche specialization broad; growth selection rapid (r); immigration-extinction processes rapid; net production high soon after inundation, then decreasing	Diversity high; niche specialization moderately narrow; growth selection variable, relatively homeostatic (K); immigration-extinction processes slow; production low to moderate, relatively constant
Ecosystem succession rate	Similar to lakes but greatly accelerated; greatly stressed by human manipulations of basin and drainage basin	Similar to reservoirs but greatly protracted

because they are commonly formed in river valleys and at the base of the drainage basin. Runoff waters to reservoirs occur mainly from large-order streams, which provide high energy for erosion, large sediment load carrying capacities, and extensive penetration of dissolved and particulate loads into the recipient lake basin. The inflows to reservoirs are primarily channelized and are often not intercepted by energy-dispersive and biologically active wetlands and littoral interface regions. As a result runoff inputs are large and more directly coupled to precipitation events, and extend much further into the lake basin than is the case in most natural lakes. These properties result in large, irregularly pulsed nutrient and sediment loading to reservoir basins.

Water level fluctuations of reservoirs are extreme and irregular as a result of these inflow characteristics, land use practices in the large drainage basins that are not conducive to water retention, channelization of primary inflows, and large, irregular water withdrawals. Establishment of productive, stabilizing wetland and littoral flora is commonly eliminated by alternate inundation and exposure of areas of littoral sediments. Erosion and resuspension of sediments of floodplains augment high sediment loadings from the drainage basin. Nutrient release from sediments can be increased by the alternation between aerobic and anaerobic conditions induced by irregular flooding. The extensive physical particulate and metabolic nutrient sieving capacities of wetlands and littoral communities that function so effectively in most natural lake ecosystems (cf. Wetzel, 1979, 1983) are commonly reduced or eliminated in many reservoirs.

A reservoir can be viewed as a very dynamic lake, in which a significant portion of its volume possesses characteristics of and functions biologically like a river (Wetzel, 1984). The riverine section of a reservoir often operates similarly to large, turbid rivers, in which turbulence, sediment instability, high turbidity, reduced light, and other characteristics preclude extensive photosynthesis in spite of high nutrient availability. The resulting limited photic zone reduces areal productivity, even though phytoplanktonic primary productivity per unit water volume in surface strata can be high. This reduction in primary productivity is only partially compensated for by turbulent, intermittent recirculation of algae into the small photic zone, analogous to the situation in large rivers (e.g. Wetzel, 1975; Minshall, 1978; Bott, 1983). Areal primary productivity increases with greater light penetration and depth of the photic zone as the turbidity lessens in the downstream progression through the transitional to the lacustrine regions of reservoirs. Nutrient limitations can then occur to varying degrees as nutrient losses exceed renewal rates. Photosynthetic limitation by light is a dominant control of reservoir productivity, as is the case in many productive natural lake and river ecosystems.

Release of nutrients from sediments, 'internal loading,' can be higher in reservoirs than in natural lakes where it is commonly low in relation to external sources. Irregular inflow and withdrawal hydrodynamics in reservoirs can disrupt thermal stratification and oxygenation patterns that suppress nutrient releases from sediments and redistribution in physically more stable natural lakes.

Dissolved organic matter (DOM) received from allochthonous sources is usually quantitatively similar in both reservoirs and natural lakes. Large sedimentation rates of inorganic particles in reservoirs can potentially remove significant amounts of DOM from the water, although sorption sites for DOM on many inorganic particles are likely saturated before they reach the reservoir. In contrast, particulate organic matter (POM) loading from allochthonous sources are usually a small part of the organic matter budgets in natural lakes (Wetzel, 1983). Fine POM of allochthonous sources, however, can be large in reservoirs, particularly during spates and floodplain inundation at high water level periods. It appears that DOM adsorbed to inorganic particulates and fine POM can supplement living particles as a food source for microconsumers (e.g. Arruda, et al., 1983). However, these detrital sources augment but do not supplant photosynthetic food sources of zooplankton.

No evidence exists that algal heterotrophy of DOM is any more significant in light-limited reservoirs than in natural lakes. Algal heterotrophy is energetically inefficient and these plants are poorly adapted to compete enzymatically with bacteria for dissolved organic substrates (Wetzel, 1983).

Evidence indicates that bacterial heterotrophic productivity is similar to or only slightly greater in reservoirs than in natural lakes of comparable productivity (Wetzel, 1983). As in most natural lakes, much of the bacterial metabolism in reservoirs is benthic in sediments rather than pelagial. However, because of the limited wetland and littoral development and irregularly high loading of fine POM, pelagial bacterial heterotrophy can be greater in reservoirs than is the case in many lakes.

Fish biology and productivity are highly variable in dynamic, constantly changing reservoir ecosystems (Wetzel, 1984). The high fish productivity commonly observed soon after reservoir formation can be related to higher benthic fauna productivity associated with greater habitat variability and refugia among inundated terrestrial vegetation.

Many reservoirs are not completely cleared of forest and shrub vegetation prior to inundation, particularly in the riverine areas. Although few quantitative data exist on the importance of dead, standing vegetative substrata for 'epiphytes' and benthic fauna, many qualitative estimates indicate these substrata and associated fauna can be major food sources for fish (K. W. Thornton, pers. comm.). As these standing substrata decay and attendent biota decline, fish must shift to alternate predominantly pelagial food sources. The high turbidity can decrease visual predation on pelagic zooplanktonic food sources. Fluctuating water levels, high siltation, and heavy predation often result in high egg and larval fish mortality in littoral areas.

The environmental conditions of reservoir ecosystems tend toward large, rapid, and erratic fluctuations (Table 1). Often insufficient time exists for complete population growth and reproductive expansion to occur before a succeeding major disturbance occurs. These instabilities result in biota that tend to be few in species and well adapted with broad physiological tolerances (low diversity, less specialization, rapid growth). As in all restrictive, stressed environments, the productivity of the adapted organisms can be high, as high or greater than in more homeostatic lakes.

Implications for Reservoir Management

Despite divergence of physical and resulting biotic structural characteristics from certain general properties of natural lake ecosystems, reservoirs are lake ecosystems in every way of process operation. Process regulating parameters in reservoirs can be irregular and often extreme; these chaotic fluctuations restrict organisms to those with broad ranges of physiological tolerances and wide behavioral adaptations. The basic processes of individual, community, and ecosystem metabolism are the same in reservoirs and natural lakes. Much of the information obtained from research on natural lakes can be applied successfully to the more dynamic reservoirs.

Time is a critical factor in nearly all response functions. In order for most of the processes (e.g. stratification, sedimentation, population growth, competitive exclusion, etc.), as we conventionally know them largely from natural lake ecosystems, to develop and be effective, adequate time is required under relatively undisturbed conditions. As natural or manipulated conditions become more irregular and variable, responses are increasingly incomplete before they are altered or destroyed. The result is an increasingly chaotic succession of responses, reductions in interdependability, and less biotic stability. Although certain physical factors (e.g. precipitation rates) are beyond the control of reservoir management, many factors (e.g. runoff rates, stratification-withdrawal relationships) can be effectively regulated and thereby enhance stability. Although certain reservoir objectives (e.g. flood control, hydroelectric power) are often primary in the construction and operation of reservoirs, other important functions of multi-purpose reservoir ecosystems can be enhanced by operational procedures that minimize these chaotic responses in physical and chemical properties. Any manipulative means to minimize the magnitude of change and to lengthen the time between disturbances will lead to greater biotic diversity and system stability.

Acknowledgments

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