Georgia-Pacific Corporation
Crossett, Arkansas

DEVELOPMENT OF A WATER QUALITY MODEL OF THE OUACHITA RIVER

VOLUME I: WATER QUALITY ANALYSES

Prepared by:

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July 27, 1992

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July 27, 1992

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GPCO0130

Subject: Ouachita River Dissolved Oxygen Modeling Report

Dear Mr. Champion:

This report summarizes the results of HydroQual's efforts over the previous three years, with the cooperation of you and members of your staff, to refine the previously developed dissolved oxygen model of the Ouachita River. It is our belief that a well calibrated model of the Ouachita River is now available for use in performing waste load allocations. At the same time, these studies have also led to an improved understanding of the factors controlling water quality in the river, and the significance of natural background loads during both high stage and low stage conditions. These results should ultimately prove useful in any efforts undertaken to obtain a seasonal or flow dependent permit limit and to evaluate the possibility of a site specific dissolved oxygen criteria for the Ouachita River.

We would like to acknowledge the fine effort put forth by you and your staff, as well as the effort of Mr. Benjamin Wu of HydroQual, which has resulted in the successful completion of this effort. We have been pleased to have had this opportunity to work with you and look forward working with you again in the future.

Should you have any questions concerning the results contained herein, please contact us at your convenience.

Very truly yours,

HYDROQUAL, INC.

Paul R. Paquin

Paul R. Paquin

Very truly yours,

HYDROQUAL, INC.

Thomas J. Mulligan PRP

Thomas J. Mulligan

TJM:caf

ACKNOWLEDGEMENT

Many individuals contributed to the successful completion of the studies described in this report. Mr. Traylor Champion of the Georgia Pacific Mill in Crossett is gratefully acknowledged for his continuing assistance in responding to our numerous questions over the course of this study. Mr. Michael Barry has also contributed in this regard, as well as having been closely involved with the data collection efforts put forth by mill personnel for a number of years. Many of the analyses presented herein would not have been possible without these data. The efforts of a number of HydroQual personnel are also acknowledged. This study has been conducted under the general technical direction of Mr. Thomas Mulligan. Mr. Paul Paquin was responsible for routine supervision of the analyses performed. Mr. Benjamin Wu performed most of the data analyses and model simulations, and had primary responsibility for design of field and laboratory studies. He also participated in some of the field surveys along with Mr. Wilfred Dunne. Mr. Joseph Strelkoff also conducted data analysis tasks. Ms. Carol Ford provided the word processing expertise required for completion of this report. The cooperation and assistance of each of these individuals is greatly appreciated.

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SECTION 1

SUMMARY, CONCLUSIONS AND RECOMMENDATIONS

Georgia Pacific Corporation operates a 1,500 ton per day pulp and paper mill, chemical plant and plywood mill in Crossett, Arkansas. The mill discharges its biologically treated process wastewater to the Ouachita River, via Coffee Creek, about one mile north of the Arkansas-Louisiana State line. The river then flows through a sparsely populated, forested watershed, with no other significant point source loads entering the river for approximately 30 miles. Bayou Bartholomew joins with the Ouachita River near the downstream end of this river reach, while further downstream additional point sources are also discharged to the river.

The state of Arkansas, in cooperation with the state of Louisiana and the United States Environmental Protection Agency (USEPA), is currently updating the NPDES permit for the Georgia Pacific discharge. As part of this process, a previously developed water quality model of the Ouachita River has been refined for use in performing an assessment of the impact of the mill's discharge and other background loads on the dissolved oxygen resources of the river.

in response!

This report contains a review of the data and modeling analyses forming the basis for development of a calibrated model of dissolved oxygen in the Ouachita River, between the Saline River in Arkansas, and Sterlington, Louisiana. The modeling analyses were initially performed between 1978 and 1980 (Hydroscience, 1979 and HydroQual, 1981) and then updated, in light of more recent data, during the last two years. This section presents a summary of the data analyses performed and model calibration and projection results. Conclusions and recommendations are also included. A detailed discussion of these results is contained in subsequent sections of Volume I of this report. Volume II includes tabular summaries of data used and other related material.

1.1 OVERVIEW OF SEASONAL DISSOLVED OXYGEN LEVELS IN THE OUACHITA RIVER

Dissolved oxygen data for the Ouachita River are available from as far back as 1956 (Velz, 1962) and have been collected on a routine basis during many of the years since that time. During the 1970s mill personnel conducted routine water quality surveys on the Ouachita River at stations located approximately every 5 miles between State Highway 82 in Arkansas, almost 15 miles upstream of the discharge, and Sterlington, Louisiana, about 30 miles downstream of the discharge. These surveys included measurements of temperature, dissolved oxygen and color. Prior to 1978, the surveys were usually performed once per week during periods of the year when the river was within its banks. For several years during the late 1970's, however, data were collected during both low and high stage conditions. These data, reviewed below, represent the most complete and nearly continuous set of data available for monitoring seasonal changes in dissolved oxygen in the Ouachita River during periods of high and low flow conditions. As such, they will be used to illustrate several consistent patterns in water quality which have been evident during both high stage and low stage periods throughout the period of record.

River stage and spatial profiles of dissolved oxygen during representative periods of time during 1979 are shown on Figure 1-1. The daily variation in river stage (left axis) and water surface elevation (right axis, for a datum of 44.1 feet) are presented on the upper graph of Figure 1-1. As indicated, the river is at bank full conditions at a stage of 19 feet and overflows its banks when the river stage rises above this level. During 1979, this condition occurred early in January. At a stage of slightly more than 30 feet, the flood plain is inundated with water for several miles on both sides of the river, beginning a distance of 25 or more miles upstream of Coffee Creek (see Figure 6-3). River flow during this time is generally not reported, but is expected to be on the order of 25,000 to 50,000 cfs, versus 7Q10 conditions of less than 1,000 cfs.

Four time intervals, labeled "a" through "d" on the upper chronological plot of river stage, have been selected to illustrate the dissolved oxygen profile of the river under different temperature and flow conditions. The averages of the weekly monitoring

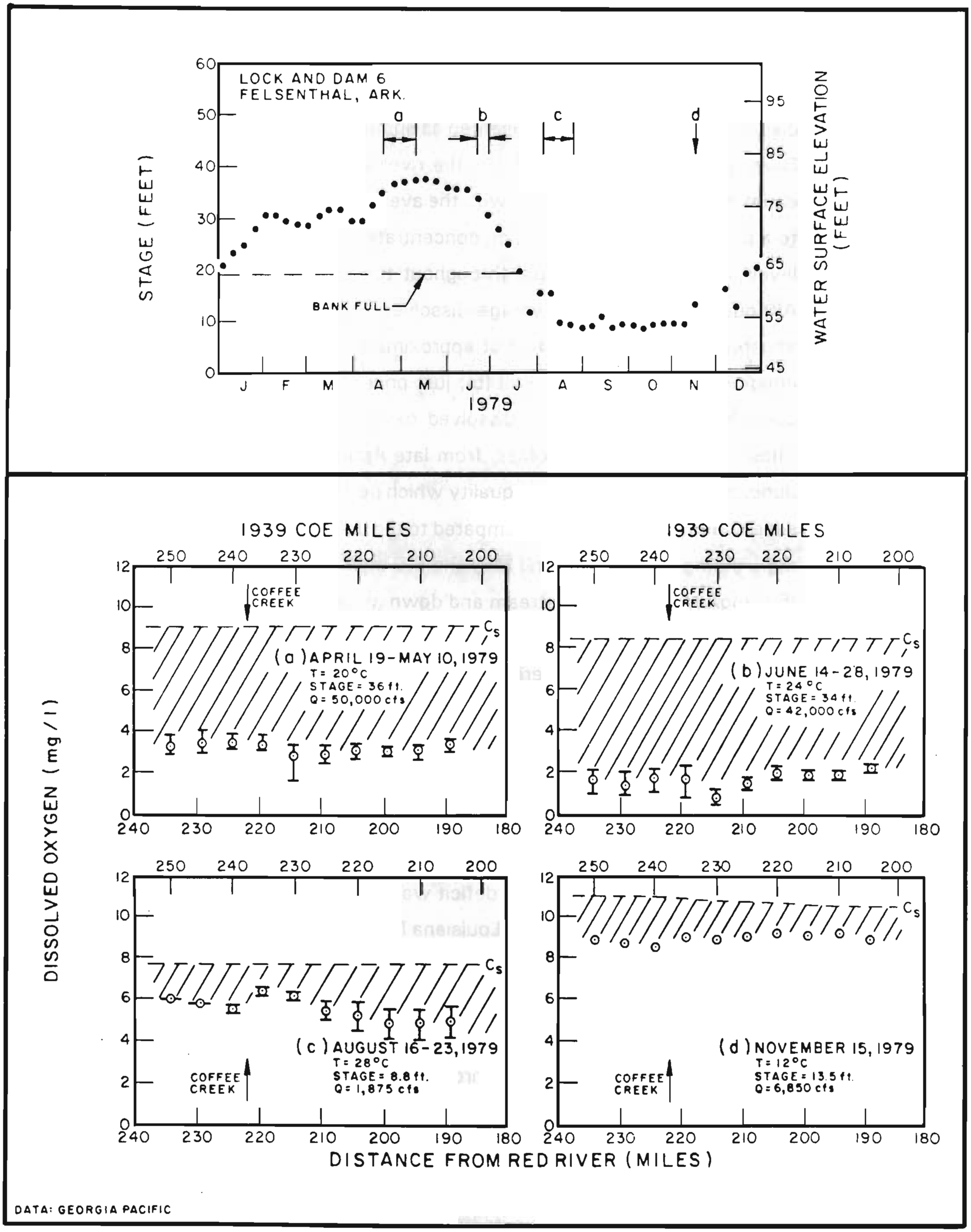


FIGURE 1-1. SPATIAL PROFILES OF QUACHITA RIVER DISSOLVED OXYGEN LEVELS, ROUTINE SURVEY DATA, SELECTED PERIODS DURING 1979

dissolved oxygen data are presented as spatial plots for each of these periods on the four lower panels. During period (a), the river was near its maximum 1979 stage and an estimated flow of 50,000 cfs, with the average water temperature of 20°C corresponding to a dissolved oxygen saturation concentration of 9 mg/L. Background dissolved oxygen levels averaged 3 to 4 mg/L throughout the 12 mile reach upstream of Coffee Creek. Although slightly lower average dissolved oxygen levels did occur downstream, it is apparent that the large deficit of approximately 6 mg/L was dominated by the upstream conditions. Over time interval (b), just prior to when the flood waters receded, similar conditions occurred. Here, dissolved oxygen levels were generally less than 2 mg/L. These first two spatial profiles, from late April to early May and from the latter half of June, are typical of the water quality which persisted in the river throughout the spawning season during 1979. When compared to the dissolved oxygen standards in Arkansas (6.5 mg/L during March, April and May and 5.0 mg/L during the rest of the year) and Louisiana (5.0 mg/L), applicable upstream and downstream of the state line at milepoint (MP) 221, respectively, the spatial profiles for these periods indicate that significantly impaired dissolved oxygen levels were prevalent in the Ouachita River.

In contrast to the high stage profiles (a) and (b), time interval (c) took place shortly after the river was back within its banks. Although the water temperature of 28°C was higher and river flow lower (1,875 cfs), average dissolved oxygen concentrations of 5 to 6 mg/L represented a marked improvement relative to the preceding high stage period. The average dissolved oxygen deficit was about 1.8 mg/L upstream of Lock and Dam 6, and 2.6 mg/L in the vicinity of Louisiana MP 195. Finally, spatial profile (d) illustrates the dissolved oxygen profile for another low stage condition, at a flow of 6,850 cfs and a temperature of 12°C, as observed on November 15, 1979. Here, the spatial profile was again quite uniform, with dissolved oxygen concentrations of about 9 mg/L and dissolved oxygen deficits of 1 to 2 mg/L throughout the study area.

The preceding review of the routine survey data illustrates several important points which generally apply to periods of high and low stage conditions in the Ouachita River. First, when the river is within its banks, background deficits in the vicinity of MP 234 are typically 2 mg/L as a result of nonpoint source impacts. Second, when the river is flooded, as occurs during most years, background deficits as high as 6 to 7 mg/L are observed a considerable distance upstream of Georgia Pacific's discharge, and this deficit propagates throughout the study area. This high background deficit is generally observed after periods of sustained flood conditions, and usually dissipates as the flood water recedes to the main channel. The dissolved oxygen profile during flooded conditions can be as low as 1 to 2 mg/L for extended periods of time, lasting as long as several months, and the dissolved oxygen standards of 5 and/or 6.5 mg/L will not be achieved.

More complete discussions of the processes considered to be controlling dissolved oxygen levels during low stage and high stage conditions are presented in Sections 5 and 6 of this report, respectively.

1.2 SUMMARY OF MODEL CALIBRATION RESULTS

A principal objective of this study was to determine the significance of the impact of Georgia Pacific's discharge on dissolved oxygen levels of the Ouachita River relative to that of natural background and upstream loads. To achieve this goal, data were collected to refine and verify a previously developed water quality model. The model was then used to project the impact of Georgia Pacific's load on river dissolved oxygen levels during critical river stage and flow conditions.

The model was calibrated using data sets representing a significant range of flow, temperature and loading conditions, including low flow periods approaching the 7Q10 river flow. The model calibration results for dissolved oxygen during these five low stage periods are shown on Figure 1-2. As shown, the model results (the solid line profiles) are in very good agreement with average data over this wide range of conditions (about 800 to 3,000 cfs and 20 to 30°C).

A unique component of the model which has been developed is the inclusion of a uniformly distributed $CBOD_u$ load which has been confirmed through measurements of

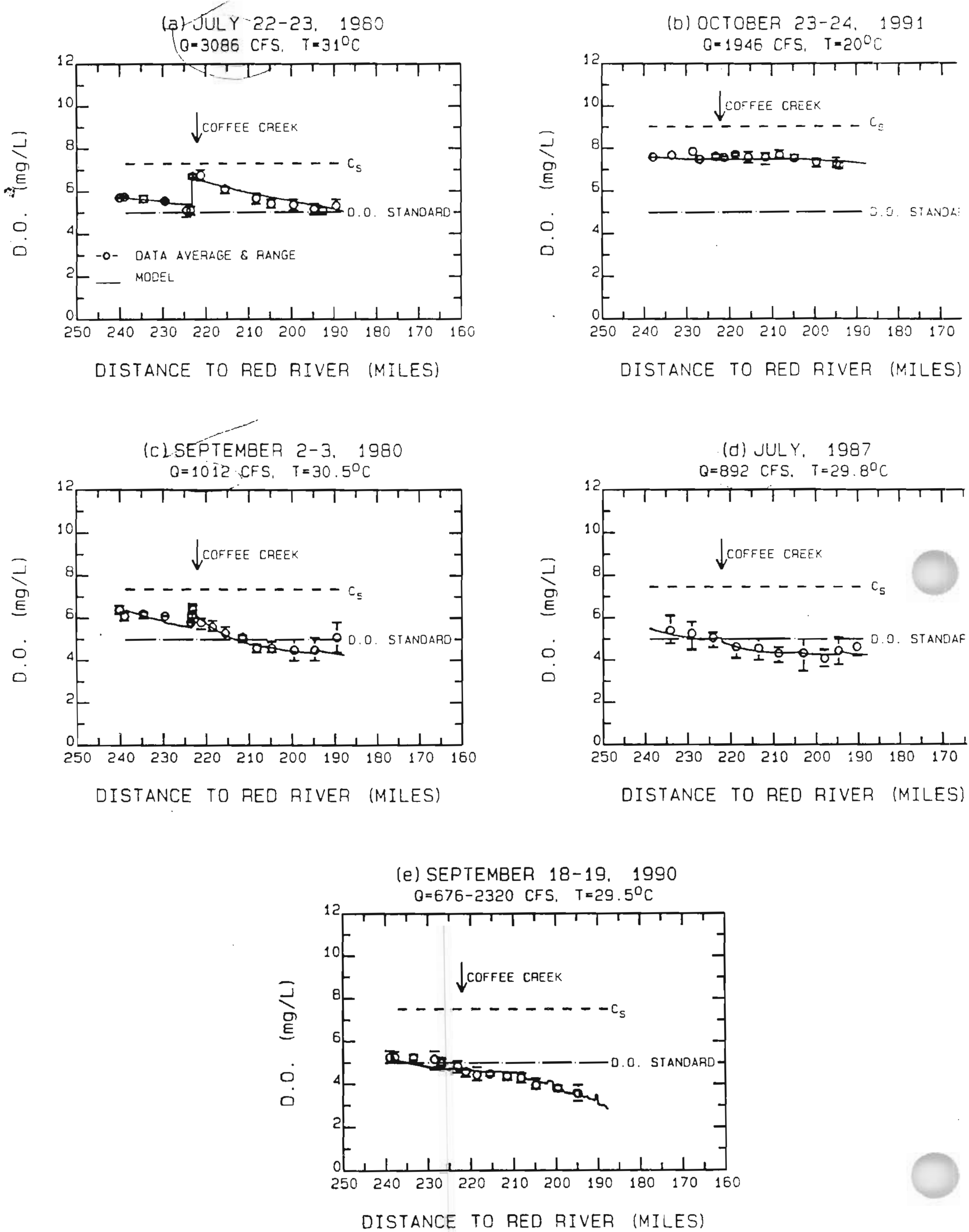


FIGURE 1-2. SUMMARY OF MODEL CALIBRATION RESULTS FOR DISSOLVED OXYGEN FOR FIVE LOW STAGE DATA SETS

fluxes from sediment core samples taken from the Ouachita River (Section 5.1.4). It appears that this load varies over the course of the year and that it is related to the seasonal flooding which routinely takes place in the Ouachita River. These features are incorporated in the model calibration and model projection analyses as well.

As described in Section 6, the model input parameters assigned for low stage conditions are assigned for the high stage calibration analyses as well. However, a difference between high and low stage conditions is that the main channel water quality is significantly impacted by the relatively depleted dissolved oxygen levels of flood plain waters (Figure 6-4). This oxygen depletion is attributed to the combined effects of the SOD of the flooded forest soils, oxidation of dissolved organic carbon released from the decomposition of soil organic matter, and the low reaeration rate in these relatively quiescent, wind sheltered waters (Section 6.3.2; Figure 6-10). Exchange of these deoxygenated waters with water in the main channel is considered to be a significant factor in the resulting instream dissolved oxygen levels. As a result, the high stage model simulations incorporate a simplified representation of the interaction between the flooded forest and the main channel of the river, via a simple exchange process. As shown on Figure 1-3, use of this approach provides a reasonable basis for accounting for the observed instream profiles during high flow - high stage periods.

During low stage periods approaching 7Q10 conditions, the model analyses described herein indicate that the impact of the Georgia Pacific permit load is slightly more than 1 mg/L (Section 5.4 and Figure 5-32). The calculated deficit due to the effluent discharge is negligible during the more critical high stage periods, primarily as a result of the high degree of available dilution, and to a lesser degree, lower instream temperatures and the relatively short travel time in the study area. For example, for the June 1980 high stage survey, with the available dilution of about 400 to 1, the ultimate CBOD concentration of 75 mg/L produces an instream deficit of less than 0.1 mg/L. Hence the deficit resulting from the load is at most only a small fraction of the 6 mg/L total deficit typically observed during high stage conditions.

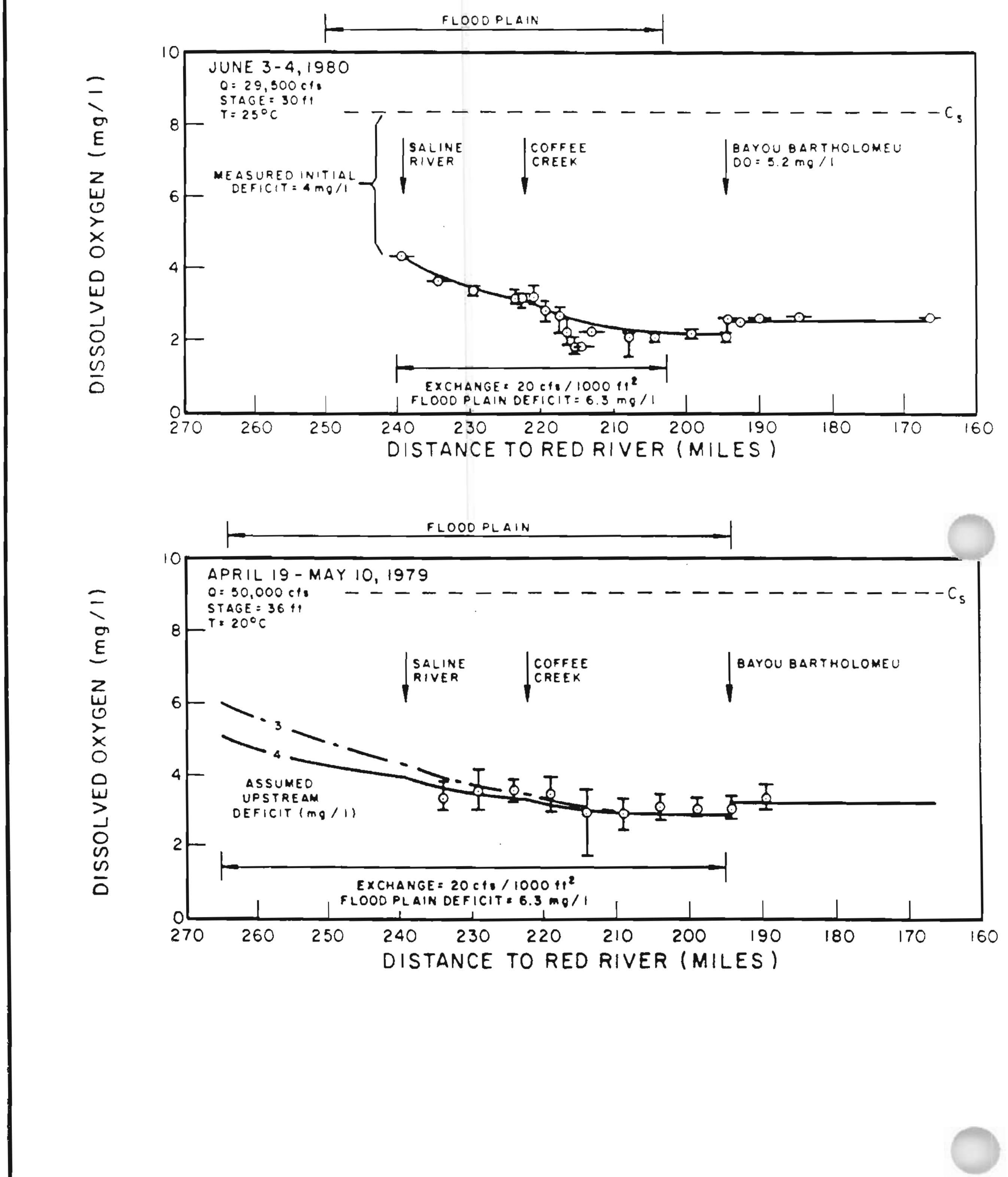


FIGURE 1-3. SUMMARY OF MODEL CALIBRATION RESULTS FOR DISSOLVED OXYGEN FOR TWO HIGH STAGE DATA SETS

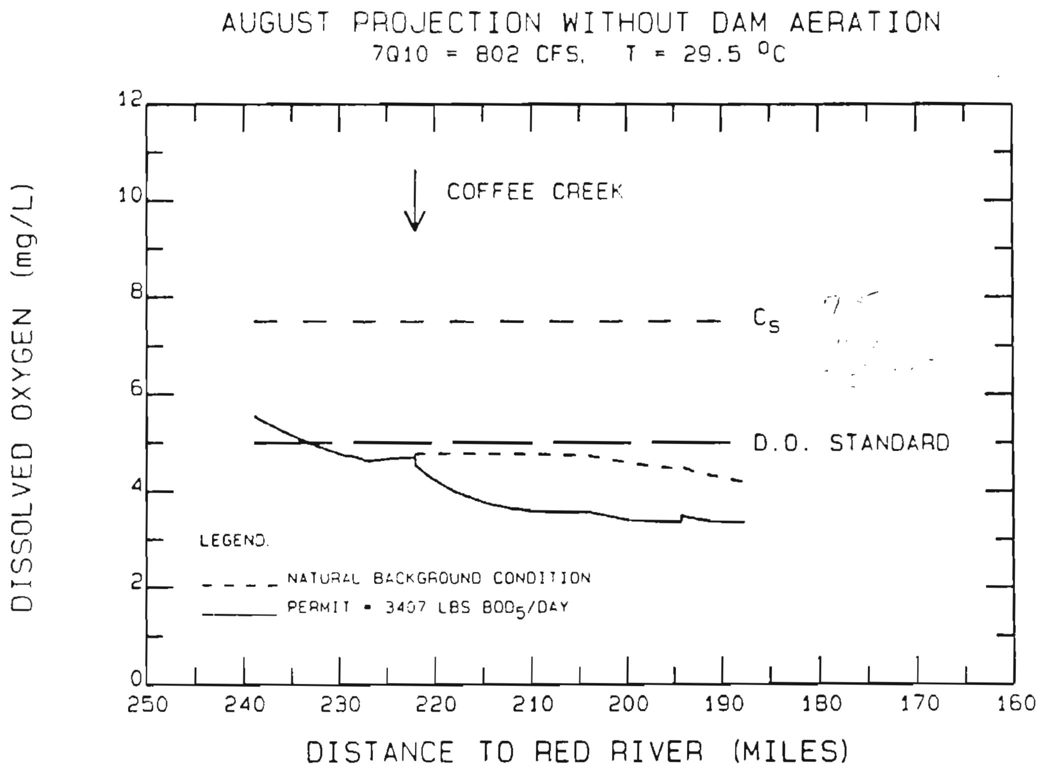
1.3 SUMMARY OF DISSOLVED OXYGEN MODEL PROJECTIONS

Projections of dissolved oxygen levels in the Ouachita River for selected critical monthly flow and temperature conditions have been completed. The projections include: (1) an evaluation of natural background conditions, in the absence of any impact from the mill and (2) projections of the overall dissolved oxygen response when the impact of the mill is included at the current BOD_5 permit limit. Results for natural background conditions are included to place in perspective the expected water quality, both with and without the impact of the mill discharge, with water quality standards in the Ouachita River. Model projections were performed for both low stage and high stage conditions.

1.3.1 Low Stage Conditions

Model projections have been completed on a monthly basis for the typical low stage, high temperature months of July through September (Section 7.2.1). These are the months for which the current permit is dependent on river flow. Figure 1-4 presents results for August, the month having the lowest monthly 7Q10 (802 cfs) of any month during the year and a high temperature month as well. During this month the projected dissolved oxygen concentration due to natural background conditions, in the absence of dam aeration (upper panel) and with no discharge from the mill is generally between 4.5 and 4.8 mg/L throughout the study area, with a minimum of 4.1 mg/L (upper dashed profile). This is below the current dissolved oxygen standard of 5 mg/L and as a result, no assimilative capacity is available for additional point source loads. With the mill discharge at the permit limit included in the projection, the minimum dissolved oxygen decreases to about 3.3 mg/L, with the maximum deficit attributed to the mill discharge (1.2 mg/L) occurring in the vicinity of MP 200.

With aeration at the dam (lower panel) conditions are generally improved in the river, with the dissolved oxygen standard achieved between the dam and the Arkansas-Louisiana state line (MP 221), both for natural background conditions and with the mill discharge included. Dissolved oxygen levels are also generally improved downstream of



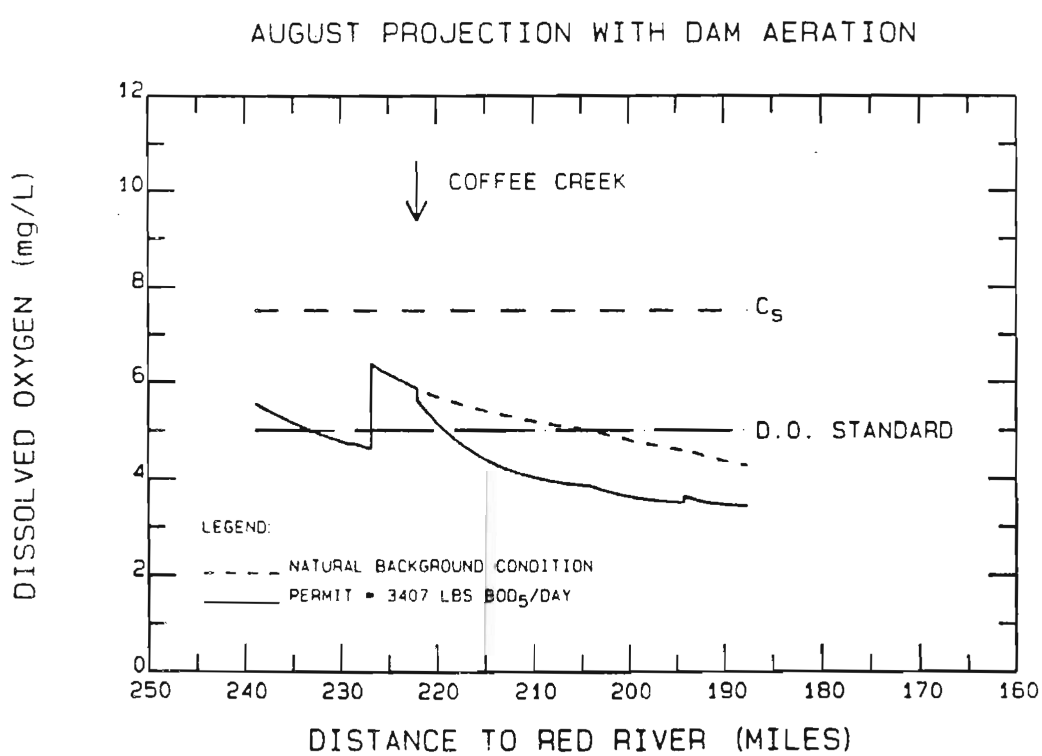


FIGURE 1-4. LOW FLOW DISSOLVED OXYGEN PROJECTION FOR JULY

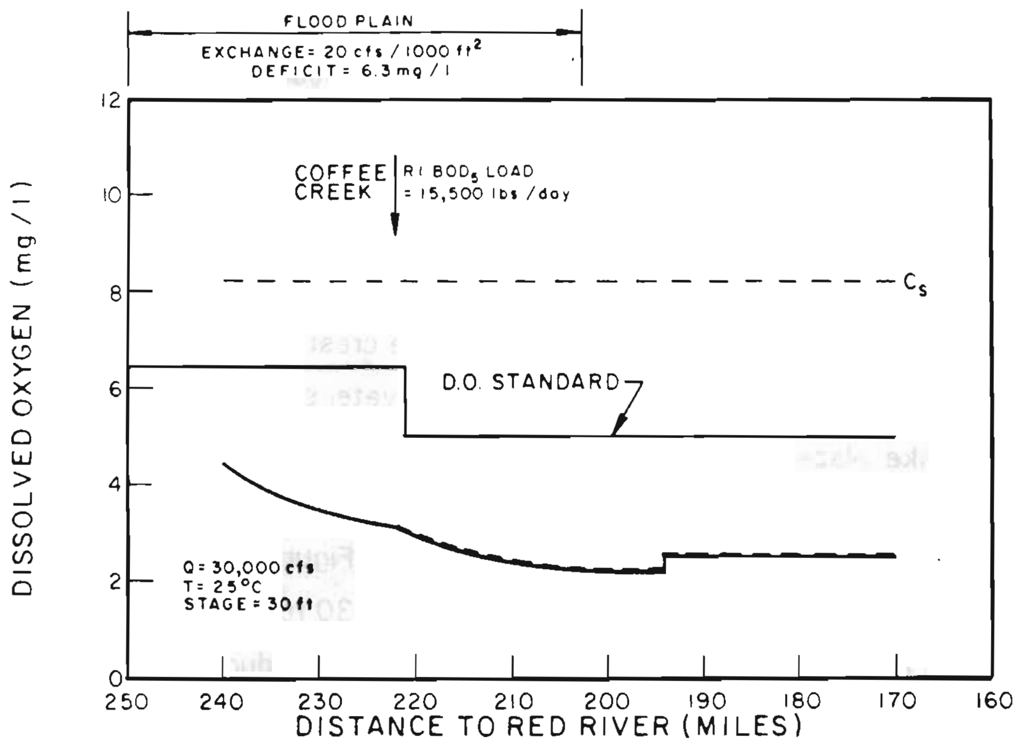


FIGURE 1-5. REPRESENTATIVE HIGH STAGE DISSOLVED OXYGEN PROJECTION

the state line, with dissolved oxygen levels above 5 mg/L as far downstream as MP 204. Further downstream, however, even under natural background conditions alone (no mill discharge) dissolved oxygen levels are still projected to be less than 5 mg/L and minimum dissolved oxygen levels at the downstream end of the study area are essentially the same as for model projections without dam aeration.

Section 7 provides a more complete description of model projection results for low stage conditions.

1.3.2 High Stage Conditions

During high stage conditions the crest of the dam at Felsenthal is submerged (the upper and lower pools are at the same water surface elevation) and dam aeration does not take place. Also, most of Coffee Creek and all of Mossy Lake are flooded, so the discharge from the mill's lagoon system to the Ouachita River flood waters is at the test point designated R1, rather than at TP3. Figure 1-5 presents a projection for a river flow of 30,000 cfs, T = 25°C, a river stage of 30 feet (water surface elevation of 74 feet) and a flood plain deficit of 6.3 mg/L (as observed during the 1980 studies). The upstream dissolved oxygen decreases from the upstream boundary concentration of about 4 mg/L to about 2 mg/L near MP 210, and then is relatively constant in the downstream direction as spatial steady state is approached. This projection is essentially the same both with and without the mill discharge at R1 (solid and dashed line profiles, respectively), since the deficit associated with the effluent load is less than 0.1 mg/L and dissolved oxygen levels are dominated by natural background conditions.

The projected high stage dissolved oxygen is significantly less than the dissolved oxygen standard of 6.5 mg/L in Arkansas and 5 mg/L in Louisiana throughout the study area. As demonstrated by data to be presented herein (see e.g., Figures 6-1 and 6-2), the dissolved oxygen profile will often be lower than that shown on Figure 1-5. This will occur because of the increased magnitude of background loads associated with flooding conditions which persist into warmer temperature periods, as well as with times when floodwaters are higher than assumed for this simulation. Given the dystrophic nature of these waters during high stage conditions, a change in the dissolved oxygen standard should be considered.

1.3.3 Impact Analysis: Coffee Creek to Columbia Lock and Dam

The analysis presented herein has focused on the reach of the Ouachita River upstream of Bayou Bartholomew. The data reviewed were used to refine the previously developed model in this upstream area and to obtain a better understanding of the relative impacts of the Georgia Pacific discharge and background loads on dissolved oxygen levels in the river. Although the analyses completed since 1990 have not been directed at verifying or refining the calibration of the model between Bayou Bartholomew and Columbia Lock and Dam, the impact of the Georgia Pacific load on this reach can be estimated. The analysis is performed by including only the load from Coffee Creek in the model and eliminating all other sources and sinks of dissolved oxygen, with the exception of reaeration. This assessment does not attempt to project the absolute dissolved oxygen concentration that will result with all of the other point and non-point source loads entering this downstream reach of the river, but rather, it evaluates the contribution to the total deficit that is attributed to the load from the mill at Crossett.

The estimated impact of the Georgia Pacific discharge on dissolved oxygen levels of the Ouachita River, between Coffee Creek and Columbia Lock and Dam, at the August 7Q10 drought flow and a BOD_5 load of 3,407 lbs/day, is shown on the upper panel of Figure 1-6. Here, all other sources and sinks of dissolved oxygen deficit, including upstream sources, SOD and downstream point sources, have been set equal to zero. The deficit due to the mill, is a maximum of slightly less than 1.2 mg/L in the vicinity of MP 208, about 13 miles upstream of Bayou Bartholomew. This deficit is primarily a result of carbonaceous BOD oxidation, although a minor component of about 0.1 mg/L is due to the dissolved oxygen deficit of the effluent at the mouth of Coffee Creek. Further downstream, near the City of Monroe (MP 158), approximately 20 days travel time from Coffee Creek, the deficit due to the mill is estimated to be reduced to only 0.15 mg/L.

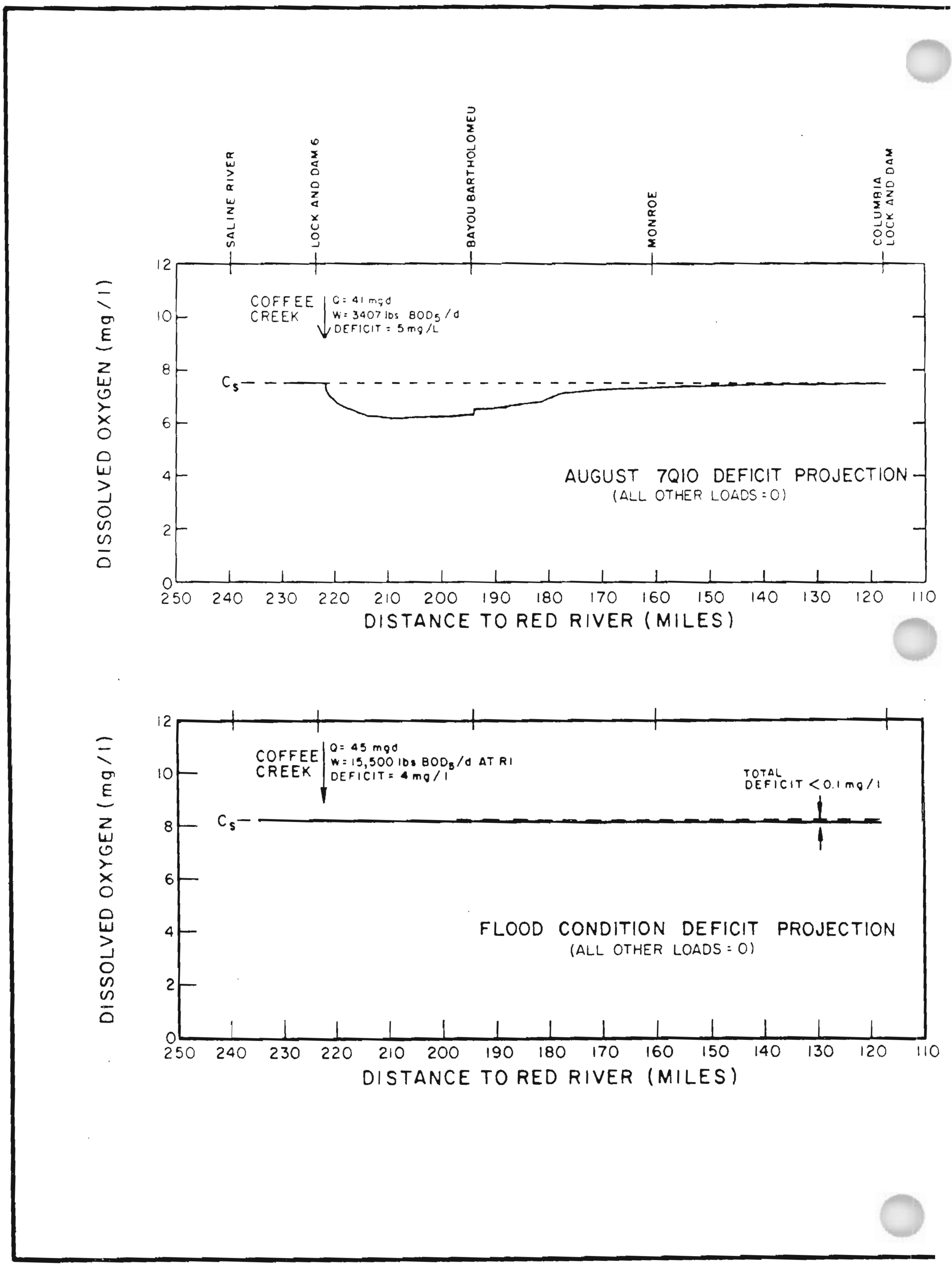


FIGURE 1-6. IMPACT OF GEORGIA PACIFIC DISCHARGE ON DO IN QUACHITA RIVER
(COFFEE CREEK TO COLUMBIA LOCK AND DAM)

Downstream of this location, the projection results indicate that the mill has a negligible impact on dissolved oxygen levels in the Ouachita River. It should be noted that these projections are based on steady state conditions in the river. Given the projected travel time of 100 days or more in the study area considered here, this assumption is clearly not appropriate. However, the results should indicate the order magnitude of the projected far downstream impacts.

The projected impact of Georgia Pacific's discharge during the flooded conditions used in these projections is illustrated on the lower graph of Figure 1-6. Due to the very high degree of dilution of 400 to 1 and the relatively short time of travel, on the order of several days, the impact of the load is negligible throughout the reach of the river between Coffee Creek and Columbia Lock and Dam. The low dissolved oxygen levels projected under flood conditions are almost entirely a result of non-point source background loads.

1.4 DISCUSSION OF RESULTS

As described in this report, and briefly reviewed above, a well calibrated water quality model of the Ouachita River has been developed. Considerable effort has been put forth to measure and/or independently evaluate the requisite model input parameters used in the analyses presented herein. Further, the demonstrated ability of the model to reproduce simulated dissolved oxygen levels over wide ranges of river stage, flow and temperature, including critical conditions with regard to dissolved oxygen in the river, adds support to the validity of the calibrated model. It follows that the current model is suitable for use in projecting instream dissolved oxygen levels at critical flows and temperatures in the Ouachita River.

The model has been used to project dissolved oxygen levels in the Ouachita River, for both natural background conditions and with the impact of the Georgia Pacific mill discharge at Crossett included, under both low stage and high stage conditions. It is shown that during low stage, low flow and high temperature periods the dissolved oxygen standard in the river will not be achieved as a result of naturally occurring background loads alone. With the impact of the mill included at monthly 7Q10 conditions, the dissolved oxygen level in the river will be reduced as much as an additional 1.2 mg/L at the current permit limit. As discussed above and in Section 7, minimum dissolved oxygen levels at the monthly 7Q10 are projected to be as low as 3.0 mg/L during July and 3.3 mg/L during August with the mill impact included.

During high stage periods, naturally occurring background oxygen demand dominates water quality in the Ouachita River. Upstream dissolved oxygen levels during these periods of very high flow can be as low as 1 to 2 mg/L and these low dissolved oxygen levels persist throughout the system, to as far downstream as Monroe, and presumably beyond. The impact of the mill discharge during these extreme periods is estimated to be less than 0.1 mg/L, in comparison to an overall dissolved oxygen deficit in the river of 6 mg/L or higher.

In summary, extensive work has been done to characterize background non-point source loads to the Ouachita River for both high and low stage conditions. The magnitude, duration and frequency of the mill's impact during these two periods is significantly different. During high stage periods, dissolved oxygen levels can be expected to be as low as 1 to 2 mg/L, for extended periods of time, during most years. The low dissolved oxygen levels begin 10 to 15 miles or more upstream of Coffee Creek, in an area which cannot possibly be impacted by the discharge from the mill. During these high stage high flow periods, the mill's impact is estimated to be on the order of 0.1 mg/L, a small percentage of the overall dissolved oxygen deficit in the river. The observed low dissolved oxygen levels are due to the naturally dystrophic conditions in this watershed.

A low stage minimum dissolved oxygen level of about 3.0 mg/L will be associated with the once in 10 years, 7 day average low flow condition occurring during July, but will generally not persist once a significant increase in river flow occurs. During these limited periods of time, natural background conditions alone result in projected dissolved oxygen levels in the river of about 3.8 mg/L, with the impact of the mill discharge, at the curr[ent] flow dependent permit limit and monthly 7Q10 flows, further reducing the minimum dissolved oxygen to 3.0 mg/L. The maximum impact due to the mill's discharge is 1.2 mg/L under these conditions.

1.5 CONCLUSIONS AND RECOMMENDATIONS

The following conclusions and recommendations are made based on the results described in this report:

1. A well calibrated water quality model of the Ouachita River has been developed. In consideration of the demonstrated ability of the model to reproduce simulated dissolved oxygen levels over wide ranges of river stage, flow and temperature, the current model is suitable for use in projecting instream dissolved oxygen levels at critical river flow and temperature conditions.

2. High Stage Periods - Natural Background nonpoint source loads produce dissolved oxygen levels in the river as low as 1 to 2 mg/L for extended periods of time lasting one month or longer. The impact of the Georgia Pacific discharge on the river dissolved oxygen during these periods is less than 0.1 mg/L.

3. Low Stage Periods - During the critical monthly 7Q10 flow and high temperature periods nonpoint source background loads alone result in minimum dissolved oxygen levels in the Ouachita River of about 4 mg/L. The maximum impact of the Georgia Pacific mill discharge on river dissolved oxygen levels during the monthly 7Q10 and high temperature periods is about 1.2 mg/L.

4. Dam aeration at Felsenthal will improve dissolved oxygen in much of the study area during low flow - high temperature periods once suitable flow release procedures are implemented. However, little improvement in

dissolved oxygen will occur at the critical location downstream of the discharge. Dam aeration will have no effect during high stage periods.

5. The observed low dissolved oxygen levels in the Ouachita River are due to the naturally dystrophic conditions in this watershed during high stage periods. Given the dystrophic nature of these waters, and the inability of the current dissolved oxygen standards to be met as a result of natural background nonpoint source impacts alone, without any point source loads discharged to the system, a change in the dissolved oxygen standard should be considered.

SECTION 2

INTRODUCTION

Georgia Pacific Corporation operates a 1,500 ton per day pulp and paper mill, chemical plant and plywood mill in Crossett, Arkansas. The mill, which obtains about 75 percent of its raw water supply from the Saline River and 25 percent from groundwater, discharges its biologically treated process wastewater to the Ouachita River. The effluent enters the river via Coffee Creek, about one mile north of the Arkansas-Louisiana State line, and almost 30 miles upstream of the next significant point source load entering the river, near Bayou Bartholomew. Downstream of Bayou Bartholomew, a number of industrial and municipal loads enter the Ouachita, including discharges from Olinkraft, IMC and the City of Monroe.

The state of Arkansas, in cooperation with the state of Louisiana and the United States Environmental Protection Agency (USEPA), is currently updating the NPDES permit for the Georgia Pacific discharge. A water quality model of the Ouachita River originally developed in 1978 by Hydroscience Incorporated (Dallas, Texas) for the State of Louisiana, as part of the "Ouachita River Basin Water Quality Management Plan," (Hydroscience, 1979), and subsequently refined by HydroQual Incorporated (1981) as part of a more detailed evaluation of the mill's impact, was used to set the existing permit limit. At that time, the model had been applied to three low flow intensive water quality surveys and two high stage survey data sets. That model has been further refined for use in updating the mill's permit, via the collection of additional field data during 1990 and 1991. This report summarizes the results of both the earlier and more recent modeling efforts.

2.1 GENERAL PURPOSE

The principal objective of this study was to determine the impact of Georgia Pacific's discharge on the dissolved oxygen (DO) content of the river, relative to the impact of the natural background and upstream loading conditions. To achieve this goal, data were collected for use in verifying and refining the previously developed model of the

Ouachita River. The model was then used to project the impact of Georgia Pacific's load on the dissolved oxygen content of the river during critical river stage and flow conditions.

2.2 DESCRIPTION OF STUDY AREA

The Ouachita River milepoint (MP) system conventionally used, and which will be followed herein, is referenced with respect to distance from the Red River. The headwaters of the Ouachita River originate in the Ouachita Mountains of central Arkansas, near the Oklahoma border. The river flows in a southeast direction, past Camden, Arkansas (MP 328.9; using post-1984 MP designations, as discussed below) and Smackover Creek (MP 298.9) and enters Louisiana at MP 221, about one mile downstream of Coffee Creek. The Ouachita River has a drainage area of 10,835 square miles at the state line of Arkansas and Louisiana and a total drainage area of 18,864 square miles at the point where the Tensas joins the Ouachita to form the Black River. The confluence of the Black River and the Red River is located approximately 221 river miles downstream o the Arkansas state line.

Coffee Creek (MP 222) enters the Ouachita River about 5 miles downstream of Lock and Dam 6 (Figure 2-1). As part of the Ouachita and Black Rivers Navigation Project, the U. S. Army Corps of Engineers completed reconstruction of Felsenthal Lock and Dam 6 in September of 1984. The new Lock and Dam 6 is located at MP 226.9, at the old abandoned Missouri Pacific railroad, along a man-made bypass of a meander in the pre-1984 river channel. The new location is 3.7 miles upstream of the old Lock and Dam 6 (MP 223.2), which has been removed. The original river channel was abandoned and closed with an overflow closure dam; therefore, the current post-1984 river channel is approximately 1.1 miles shorter than the pre-1984 river channel. With the new dam in place, the water surface elevation of the upper pool during low stage conditions has been increased about three feet. At the same time, the current channel upstream of the old dam location and downstream of the new dam is much shallower than before, as it is now a part of the lower pool. Pre and post 1984 modeling analyses to be presented refl these changes in channel geometry.

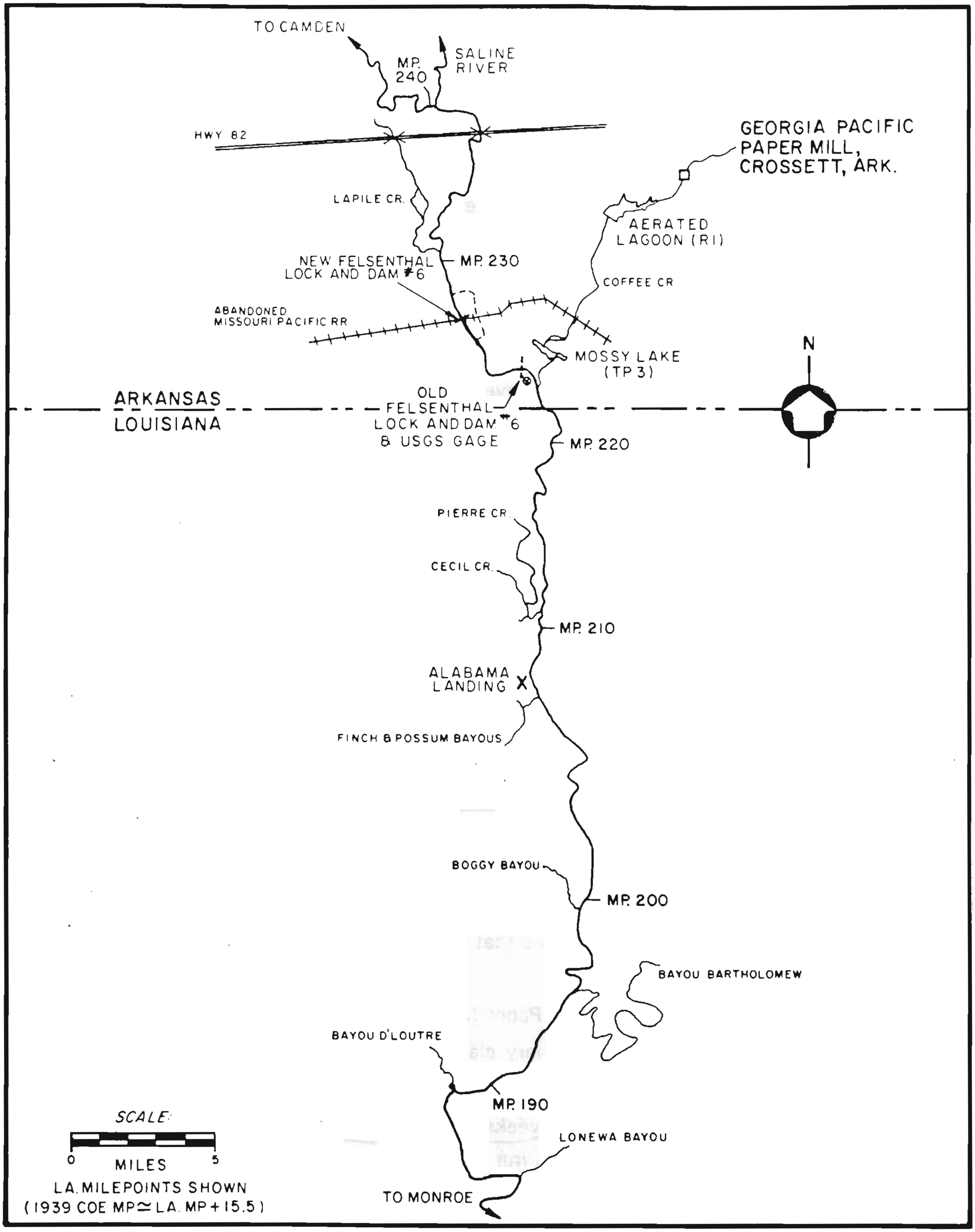


FIGURE 2-1. OUACHITA RIVER STUDY AREA

Unless indicated otherwise, MP designations referred to herein which are upstream of the closure dam, at the upstream end of the bypassed meander, refer to distance along the new channel. Note that model results and spatial profiles of data collected prior to September 1984 will correspond to the old MP designations. (Table 4-1, presented later in this report, includes a comparison of the various MP systems that have frequently been used.) As a general rule, subtract 1.1 miles from pre-1984 MP designations upstream of the new Lock and Dam 6 to convert to post-1984 MP designations.

The general area of interest in this study is the 121.4 mile reach of the Ouachita River between the Saline River (MP 238.4), 17 miles upstream of the Georgia Pacific discharge, and Columbia Lock and Dam (MP 117), located 45 miles downstream of the City of Monroe. Particular emphasis was placed on analyzing the portion of the river located upstream of Bayou Bartholomew (MP 194) where Georgia Pacific is the only significant point source load entering the river. This reach of the Ouachita River is illustrated on Figure 2-1. The 18-mile river reach located between the Saline River and Coffee Creek was included in order to obtain a better understanding of the impact of the upstream non-point source loads. This region is sparsely populated and does not directly receive any point source loads. The cities of Camden and Smackover Creek (not shown), are located 90 and 60 miles upstream of the Saline River, respectively. The impact on the study area of discharges from ~~these cities~~ is expected to be negligible, and while not explicitly included in the modeling analysis, residual effects would be reflected in the upstream boundary conditions that have been measured.

The Georgia Pacific Paper Mill is located in Crossett, Arkansas. The process wastewater undergoes primary clarification followed by extended aeration. The 625 million gallon aerated lagoon, which also treats the domestic wastewater from Crossett, provides on the order of 2 weeks detention time at a wastewater flow rate of 45 mgd. The effluent from the lagoon (RI) flows via Coffee Creek to Mossy Lake where additional treatment is obtained (about 10 days detention time), after which it discharges to the Ouachita River at MP 222, about 5 miles downstream of the new Lock and Dam 6 (1.2

miles downstream of the old lock and dam) and 0.9 miles upstream of the state line. The entire Coffee Creek watershed is located on land owned by Georgia Pacific and historically has been considered part of the mill's treatment system. So are many other streams in Ark.

The United States Geological Survey (USGS) maintains a continuous recording gage near old Lock 6, providing daily estimates of river flow throughout most of the year. A number of relatively small tributaries enter the river between the dam and Bayou Bartholomew, but the intervening drainage area over this distance represents an increase of less than 4 percent relative to the 10,850 square miles at old Lock 6. Hence, the river flow is approximately constant throughout this reach of the river. Bayou Bartholomew (MP 194) does account for a significant increase in flow to the Ouachita River and further downstream additional point source loads also enter the river, making the system increasingly complex to analyze.

2.3 REVIEW OF WATER QUALITY CRITERIA AND STANDARDS FOR DISSOLVED OXYGEN

Prior to reviewing some historical data from the study area it is pertinent to first briefly review the relevant criteria and standards for dissolved oxygen. This review will cover the state water quality standards in Arkansas and Louisiana as well as the national water quality criteria for dissolved oxygen.

Arkansas - Pursuant to the Arkansas water quality standards (BNA, 1992a) the primary season ($T < 22^{o}C$) dissolved oxygen standard for the Ouachita River in Arkansas is 6.0 mg/L and at temperatures less than $10^{o}C$, it is 6.5 mg/L. During the months of March, April and May, approximately the spawning season for most fish species, the standard is also 6.5 mg/L. The critical season ($T > 22^{o}C$) standard is 5 mg/L, but allowance is made during this period for a 1 mg/L diurnal depression below the critical standard for no more than 8 hours within a 24 hour period. The state currently requires mill personnel to monitor the receiving water dissolved oxygen at the five foot depth in the water column, once per month during the months of July, August and September.

Louisiana - As will be shown, due to the location of Coffee Creek just upstream of the state line, most of the impact of the discharge from the Crossett paper mill occurs in Louisiana. The dissolved oxygen criteria for the Ouachita River between the Arkansas-Louisiana state line and Columbia Lock and Dam is 5 mg/L, although naturally occurring variations below this criterion are allowed for "short periods" of time (e.g., as a result of the reduction in photosynthetic activity and oxygen production by plants during hours of darkness). The state of Louisiana specifies that instream monitoring is to be based on dissolved oxygen measurements at the 1 meter depth (approximately 3 feet) below the water surface.

The Louisiana regulations (BNA, 1992b) also note that exceptions to the state standards are sometimes allowed as a result of natural water quality or physical limitations of the water body. This determination is made on a case by case basis through a use attainability analysis. Of particular relevance here are exceptions for the category of "naturally dystrophic waters," as described in the Louisiana Water Quality Standards (Louisiana Administrative Code, Title 33, 1991). Para-phrasing from this document:

> Such waterbodies are described as being stained with organic material and are low in dissolved oxygen as a result of natural conditions. They must be primarily affected by natural sources of oxygen demanding material or naturally occurring cycles of oxygen depletion. They typically include or are surrounded by wetlands, including lowland forests, swamps and marshes. Though seasonally deficient in dissolved oxygen, they may fully support fish and wildlife propagation and other water uses. Low dissolved oxygen concentrations (< 5 mg/L) typically occur seasonally during warmer months of the year in naturally dystrophic waterbodies.

It will be evident from the results to be presented that in many respects the preceding characteristics apply to the reach of the Ouachita River within the study area investigated in this report.

The state administrative authority and the EPA must approve and designate a waterbody for the naturally dystrophic classification. Waterbodies are considered on a case by case basis by conducting a use attainability analysis. This type of analysis is conducted to gather data to document the characteristics of a naturally dystrophic waterbody. A minimum dissolved oxygen criterion of 3 mg/L for the period of May 1 through October 31 is generally applied to naturally dystrophic waterbodies, although other dissolved oxygen criteria and seasonal periods may apply if supported by the use attainability analysis. To date, these studies have not been conducted for the Ouachita River.

National Ambient Water Quality Criteria for Dissolved Oxygen - The national water quality criteria document for dissolved oxygen defines the ambient dissolved oxygen concentration necessary for the protection of aquatic life (USEPA, 1986a; USEPA 1986b). The criteria document indicates that "where natural conditions alone create dissolved oxygen concentrations less than 110 percent of the applicable criteria means or minima or both, the minimum acceptable concentration is 90 percent of the natural concentration." Given the significance of natural background conditions which will be shown to exist in the Ouachita River, this latter point is particularly germane to the situation under consideration herein.

2.4 OVERVIEW OF HISTORICAL WATER QUALITY CONDITIONS

The Ouachita River is a hydrologically unique river system which regularly experiences extremes of both very low flow and extremely high flow/high stage flood conditions. During most of the year the river is within its banks and flow is regulated by a series of locks and dams. Of particular interest here are the dams at Columbia and Felsenthal. The Corp of Engineers is obligated by existing regulations to maintain prescribed water surface levels (pool depths) in order to maintain navigable waterways. As a result, during low flow periods, release of water at these dams is controlled in order to maintain the required pool elevations. The presence of these dams and the associated hydraulic controls have several important ramifications on water quality. First, restricting flow past the dam necessarily reduces flow to the downstream reach, thereby exacerbating what may already be critically low flow conditions. This problem is compounded by the fact that the dam at Columbia creates an impoundment of water which has a very low hydraulic gradient, and hence diminished reaeration capacity. Both of these factors have a detrimental effect on the assimilative capacity of the river.

At the other extreme, the Ouachita River regularly experiences periods when both the river flow and stage rise markedly and water inundates a 5 mile wide flood plain for more than 60 miles upstream of Alabama Landing (MP 208). This flood plain is comprised almost entirely of forest lands. Historical water quality data, discussed in detail in Section 6 of this report and described briefly below, indicate that the dissolved oxygen level in the river becomes severely depressed when this condition occurs. This is somewhat surprising, since most river systems experience a greater assimilative capacity and improved water quality with increased flow.

Figure 2-2 presents temporal plots of river stage and dissolved oxygen deficit (DO saturation - DO concentration) over a 2 year period, from October 1978 through September 1980, to illustrate this phenomenon. The upper panel shows river stage over this time period, with bank full conditions corresponding to a river stage of 63.1 feet (19 feet + 44.1 ft datum). The two lower panels show that dissolved oxygen deficits over this time period were very similar at a location about 13 miles upstream of Coffee Creek (middle panel) and at a location about 30 miles downstream of Coffee Creek (lower panel). This is especially so near the end of the high stage period, when the river flow was on the order of 30,000 cfs or more, and dissolved oxygen deficits in the river were greater than 6 mg/L (DO approaching 1 mg/L). The rapid decrease in dissolved oxygen deficit (i.e., the increase in dissolved oxygen) as the river stage falls to less than bank full elevation, during both of the years shown, is a further indication that the severely impaired dissolved oxygen levels are primarily controlled by the high water levels which inundate the adjacent flood plain.

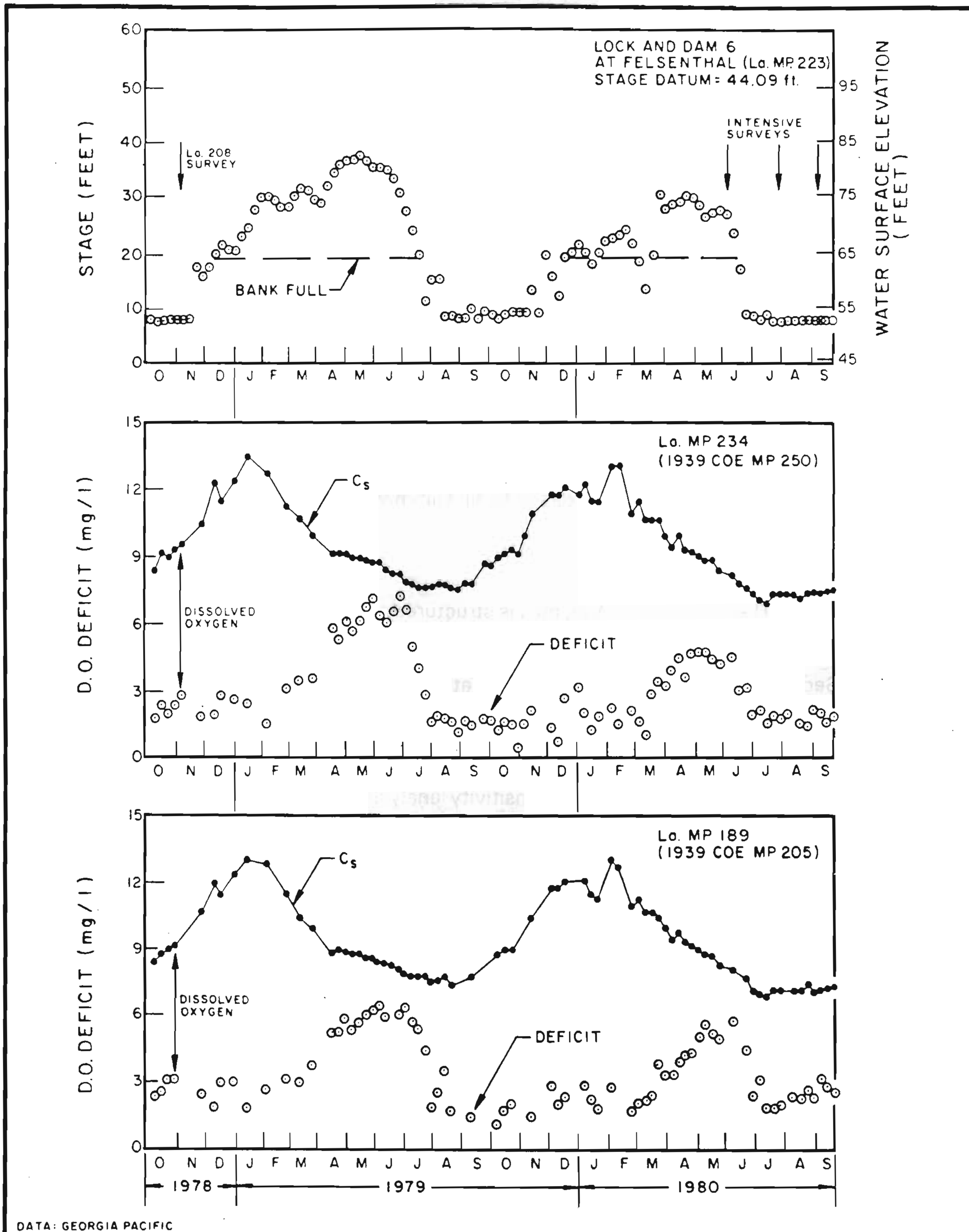


FIGURE 2-2. CHRONOLOGY OF OUACHITA RIVER STAGE AND DO DEFICIT, ROUTINE SURVEY DATA, 10/1/78 - 9/30/80

The aforementioned high stage and low stage hydrologic conditions represent opposite extremes which are regularly observed in the Ouachita River Basin. The cause-effect relationships between these flow conditions and the resulting dissolved oxygen levels will be evaluated in order to assess the relative impacts of the Georgia Pacific discharge and background loads on water quality of the Ouachita River.

2.5 STRUCTURE OF REPORT

This report is comprised of two volumes. This volume, Volume I, contains a description of the data and model analyses that have been performed. A separate Appendix, Volume II, provides tabular summaries of the data that have been used and other related information.

The remainder of Volume I is structured as follows. Section 3 describes the method of analysis and model framework used to model water quality in the Ouachita River. Section 4 then presents model calibration results for conservative tracers, constituents used to establish the spatial distribution of flow within the study area. Section 5 follows with BOD-DO calibration results for low stage conditions, including the evaluation of model inputs, calibration results, a sensitivity analysis of important model parameters, and a discussion of the components of dissolved oxygen deficit in the Ouachita River. Section 6 includes a review of data obtained during high stage conditions and the high stage model calibration analysis. Model calibration results are then followed by the model projections in Section 7. The report concludes with Section 8, a list of references cited.

SECTION 3

METHOD OF ANALYSIS

When an organic waste is discharged to a receiving water, dissolved oxygen is utilized by bacteria during stabilization of the waste material. As the dissolved oxygen concentration decreases to less than the saturation value, an imbalance is created. In order to restore river dissolved oxygen to its equilibrium state, atmospheric oxygen is transferred into solution across the air/water interface of the river. The rate at which the oxygen is utilized is assumed proportional to the concentration of biologically degradable organic material as well as chemically oxidizable substances. The rate coefficient is a function of temperature. The rate at which dissolved oxygen is replaced is proportional to the dissolved oxygen deficit, with its coefficient also a function of both temperature, and more importantly, the rate of turbulent renewal of the air/water interface. The combined effects of these reactions, biological oxidation and atmospheric reaeration, in conjunction with the spatial translation produced by the freshwater flow, produces a characteristic longitudinal distribution of dissolved oxygen which decreases to some minimum value and then recovers to saturation. The dissolved oxygen concentration may also be affected spatially by sediment oxygen demand, and temporally by the respiration and photosynthetic activity of aquatic plants. These sources and sinks of dissolved oxygen are illustrated schematically on Figure 3-1.

A material balance may be made which includes all factors affecting the longitudinal distribution of any substance. The following general expression is obtained:

$$\frac{\partial c}{\partial t} = -\frac{Q}{A}\frac{\partial c}{\partial x} \pm \Sigma S \qquad 3\text{-}1$$

in which c is the concentration of the substance, t represents time, Q is the freshwater discharge, A denotes the river cross-sectional area, x is the longitudinal distance and S represents the various sources and sinks of the material in the system. Equation 3-1 states that the time rate of change of concentration of a substance at a particular river

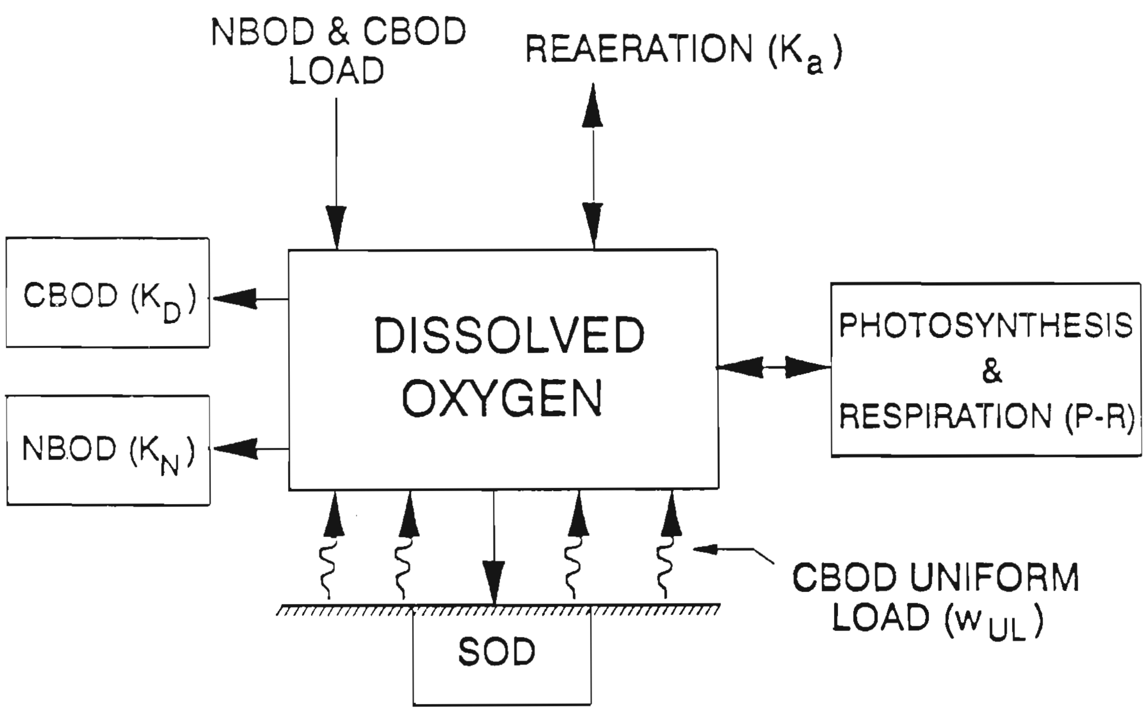


FIGURE 3-1. MODEL FRAMEWORK: SOURCES AND SINKS OF OXYGEN INCLUDED IN DISSOLVED OXYGEN MODEL OF THE OUACHITA RIVER

location is proportional to the longitudinal gradient, plus the sources and sinks of the material. The first term on the right side of Equation 3-1 represents the net balance of the material at a location due to the freshwater discharge and the second term represents the net accumulation or reduction associated with the sources and sinks of the material.

3.1 CARBONACEOUS BIOCHEMICAL OXYGEN DEMAND (CBOD)

Re-expressing Equation 3-1 for carbonaceous BOD under steady-state conditions, where there is no change in concentration with time, yields:

$$0 = -u \frac{dL}{dx} - K_r L \qquad 3\text{-}2$$

in which U is the freshwater velocity, L the BOD concentration, and K_r the stream BOD removal rate coefficient. The first term on the right side of Equation 3-2 represents the advective transport of BOD by the freshwater flow and the second term indicates first-order biological oxidation, a sink of BOD. Moreover, the coefficient, K_r, reflects all factors contributing to the removal of BOD, such as river settling, in addition to oxidation. The solution to Equation 3-2 for the boundary conditions that an initial BOD concentration, $L = L_0$, exists at $x = 0$, and that the BOD concentration approaches zero at large distances from the origin, is given by:

$$L = L_0 e^{-\frac{K_r x}{U}} \qquad 3\text{-}3$$

3.2 NITROGENOUS BIOCHEMICAL OXYGEN DEMAND (NBOD)

The expression for the distribution of nitrogenous BOD in the river is analogous to the expression for carbonaceous BOD:

$$N = N_o e^{-\frac{K_n x}{U}} \qquad 3\text{-}4$$

in which N is the nitrogenous BOD concentration at any distance, x, N_o is the nitrogenous BOD concentration at x = 0, and K_n is the nitrogenous BOD oxidation rate coefficient.

3.3 DISSOLVED OXYGEN

The dissolved oxygen distribution may be formulated in a similar manner. Expressing dissolved oxygen in terms of the dissolved oxygen deficit, D:

$$D = C_S - C \qquad 3\text{-}5$$

where C_s is the dissolved oxygen saturation concentration and C is the dissolved oxygen concentration, there results:

$$O = -U\frac{dD}{dx} - K_a D + K_d L + K_n N + S + R - P_a \qquad 3\text{-}6$$

in which K_a is the atmospheric reaeration rate coefficient, K_d is the river deoxygenation rate coefficient and K_n is the nitrogenous oxidation rate coefficient. If carbonaceous BOD is only removed by direct oxidation, the deoxygenation rate coefficient, K_d, is equal to the BOD removal rate coefficient, K_r. The terms on the right side of Equation 3-6 represent, respectively: the downstream transport of oxygen deficit with the freshwater flow; atmospheric reaeration, a source of dissolved oxygen and hence a sink of deficit; biological oxidation (CBOD and NBOD), a source of deficit; SOD; and algal respiraton (R) and average photosynthesis (P_a). The carbonaceous BOD concentration, L, and nitrogenous BOD, N, in Equation 3-6 may be replaced by the functional forms of Equations 3-3 and 3-4 and the resulting expression integrated with the previously indicated boundary conditions, to obtain:

$$D = \frac{K_d L_o}{K_a - K_r}\left[e^{-\frac{K_r x}{U}} - e^{-\frac{K_a x}{U}}\right] + \frac{K_n N_o}{K_a - K_n}\left[e^{-\frac{K_n X}{U}} - e^{-\frac{K_a x}{U}}\right] + \left(\frac{S}{K_a} + R - \frac{P_a}{K_a}\right)\left[1 - e^{-\frac{K_a x}{u}}\right] + D_o e^{-\frac{K_a x}{U}} \qquad 3\text{-}7$$

Here, D_o represents the initial dissolved oxygen deficit at $x = 0$. The initial BOD concentrations, L_o and N_o in Equation 3-7, must be expressed in terms of ultimate oxygen demand.

The dissolved oxygen concentration may be determined from the calculated deficit values in accordance with the following:

$$C = C_S - D \qquad 3\text{-}8$$

in which C is the dissolved oxygen concentration at any location and C_S the dissolved oxygen saturation value, a function of water temperature in freshwater streams. Flow and concentration discontinuities in the system due to wasteload or tributary effects must be included in the water quality mass balances. At each location of a waste discharge or tributary, a material balance must be calculated which incorporates the effect of BOD, dissolved oxygen or flow added to the system. In this manner, the parameters L_o, N_o and D_o in Equations 3-3, 3-4 and 3-7 are reinitialized at every discontinuity and a new origin, $x = 0$, is established to comply with the boundary conditions.

3.4 REAERATION COEFFICIENT

The atmospheric reaeration coefficient for isotropic turbulence conditions as defined by O'Connor and Dobbins (1956), K_a, is a function of the stream hydraulic characteristics such that:

$$K_a = \frac{13U^{1/2}}{H^{3/2}} \qquad 3\text{-}9$$

in which U is the average river velocity in fps, and H represents the mean river depth in feet. This expression tends to underestimate the rate of reaeration in deep slow moving bodies of water, and as a result, the minimum rate of reaeration has been limited to the value determined from the surface mass transfer coefficient, K_L. K_L is related to the reaeration rate coefficient, K_a, by the following expression:

$$K_a = K_L \frac{A}{V} = \frac{K_L}{H} \qquad 3\text{-}10$$

where:

K_a = reaeration rate coefficient (1/day)
K_L = surface mass transfer coefficient (ft/day)
A = surface area of river reach (ft^2)
V = volume of river reach (ft^3)
H = average depth (ft)

In general, for velocities less than 0.5 ft/sec and wind speeds less than 8 miles per hour, a minimum value of K_L = 2 ft/day is used as an approximate lower limit. This relationship is used in the model for the deep, slowly moving portions of the Ouachita River. For implementation purposes, when the value of K_a calculated with Equation 3-9 is less than the value obtained with Equation 3-10, the latter relationship is employed.

3.5 TEMPERATURE EFFECTS

All reaction rate coefficients previously indicated may be related to temperature as follows:

$$K_T = K_{20}\theta^{T-20} \qquad (3\text{-}11)$$

Here, K_T is the value of the coefficient at temperature, T, K_{20} is the 20°C value and θ is a constant which characterizes the temperature dependence of the reaction rate coefficient. A value of θ = 1.024 is used for K_a and θ = 1.047 for K_r, K_d and K_n. Sediment oxygen demand is adjusted for temperature using θ = 1.08.

SECTION 4

WATER QUALITY DATA AND MODEL CALIBRATION FOR LOW STAGE CONSERVATIVE TRACER ANALYSIS

The equations defining the spatial distribution of BOD and dissolved oxygen, as presented in the previous section, "Method of Analysis," have been applied to the analysis of the Ouachita River intensive survey data. The general approach, which follows that used in earlier model calibration efforts (HydroQual, 1981), consists of: (1) evaluation of system geometry, time of travel relationships and flow distributions to be used in the model, and (2) evaluation of model parameters needed for calibration of the BOD-DO model to field data. The first of these steps, which makes use of conservative tracer data to insure that representative flow distributions are used, is reviewed in this section. The flow distributions determined in this way are subsequently used in the BOD-DO modeling analyses described in Section 5. Application of the water quality model of the Ouachita River to high stage flooded conditions, which involves a somewhat different approach, is then presented in Section 6.

4.1 PHYSICAL DESCRIPTION OF MODEL

4.1.1 <u>Hydrology</u>

The distribution of Ouachita River drainage area as a function of distance from the Red River is illustrated on the upper panel of Figure 4-1. The tributaries shown represent those having a drainage area of greater than 1 percent of the Ouachita River drainage area at the confluence with the main stream (Yanchosek and Hines, 1979; Sloss, 1971). Upstream of the Saline River the Ouachita has a drainage area of approximately 7,200 square miles. The areal contribution of the Saline River increases this by about 50 percent, resulting in a cumulative drainage area at Lock 6 of approximately 10,850 square miles. Between Lock 6 and Columbia Lock and Dam, the 3 largest tributaries in terms of drainage area are Bayou Bartholomew (1,665 square miles), Bayou de Loutre (430 square miles) and Bayou D'Arbonne (1,904 square miles). The cumulative drainage area at Columbia Lock and Dam, including the contribution of these rivers and of a number of smaller tributaries,

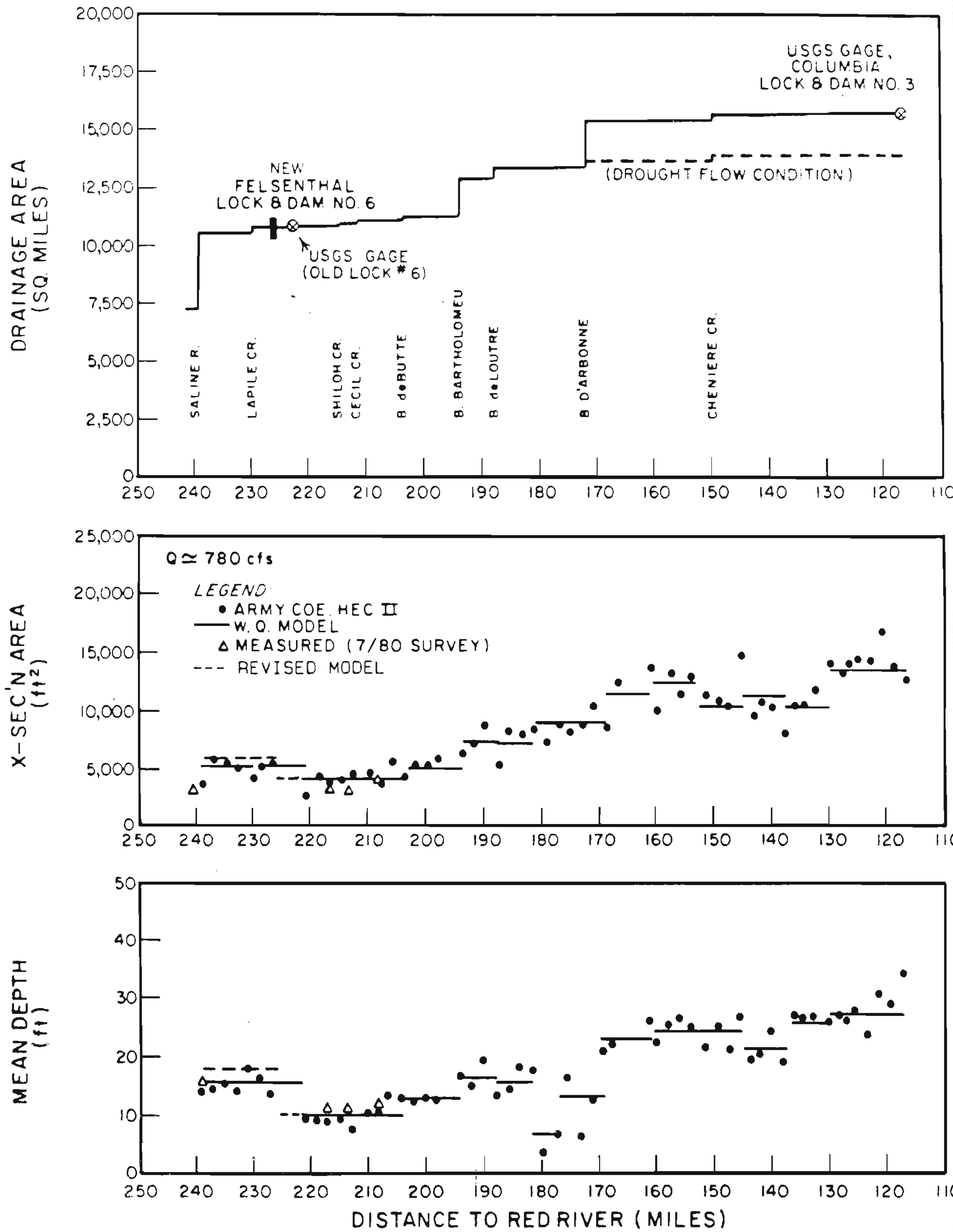


FIGURE 4-1. OUACHITA RIVER HYDROLOGY AND CHANNEL GEOMETRY

is 15,630 square miles. During drought flow conditions, Bayou D'Arbonne contributes only 297 square miles of drainage area to the Ouachita River (the difference drains into Lake D'Arbonne) resulting in a drainage area at Columbia of 14,023 square miles. This represents approximately a 30 percent increase in drainage area relative to the drainage area at Lock 6.

The flow distribution used in the model during low flow conditions is based on the drainage areas shown on Figure 4-1. The difference in flow between Lock 6 and Columbia Lock and Dam that is not attributable to known tributary flows and wastewater flows is distributed to the remaining tributaries in proportion to their drainage areas. This procedure provides a reasonable first approximation, but is not without its deficiencies. Due to the fact that the Ouachita River flow is regulated at the dams, there are times when the flow at Columbia is less than or equal to the flow at Felsenthal. While this inconsistency can be partially attributed to the inherent limitations on the accuracy of reported flows, it also reflects periods of time when the pool level above the dam at Columbia is being raised. This situation occurred at the time of the September 1980 survey, and as a result, tributary flows downstream of Lock 6 were assigned on the basis of an estimated runoff coefficient of 0.05 cfs/mi^2 for that survey.

4.1.2 Channel Geometry

The channel geometry and time of travel relationships used in the Ouachita River model are based on the HEC II flood plain model developed by the US Army Corps of Engineers. This model utilizes detailed measurements of channel cross-sections obtained during an 1890 survey of the study area. The two lower panels on Figure 4-1 compare the cross-sectional areas and depths determined from HEC II model inputs, at a low flow condition of 780 cfs throughout the study area (filled circle plot symbols), to the geometry used in the model (the solid lines). Note that in the river reach upstream of MP 223.2, the revised cross-sectional areas and depths resulting from the new Felsenthal Lock and Dam 6, which maintains the upper pool level 3.4 feet higher than the old pool level, are also indicated (dashed lines). The revised geometry is used in model simulations corresponding to the post-1984 time period.

As a preliminary check on the reliability of the HEC II cross-section information, a limited number of channel cross-sections were measured at the time of the July 1980 survey. These results are also shown on the two lower panels of Figure 4-1 (open triangles). Although somewhat lower cross-sectional areas and greater depths were observed in some instances, the results are comparable relative to the considerable variation in channel geometry which exists between Lock 6 and Columbia Lock and Dam.

Mathematical expressions which relate time of travel and river geometry to flow have been used to incorporate the HEC II results into the model of the Ouachita River. The procedure used is illustrated on Figure 4-2. The average river velocity determined by the computed HEC II time of travel (open circle plot symbols) has been plotted versus flow for selected reaches in the study area. As shown, there is an excellent linear relationship between log velocity and log flow within the range of 780 to 13,000 cfs. This linear relationship holds equally well for most of the study area, although between the Saline River and MP 194 it is only valid up to a flow of 5,460 cfs. The following functional form is used to describe this flow-velocity relationship:

$$U = C_u \, Q^{P_u} \tag{4-1}$$

where:

U = average velocity within the reach, feet per second

C_u = velocity coefficient

Q = average flow within the reach, cfs

P_u = velocity exponent

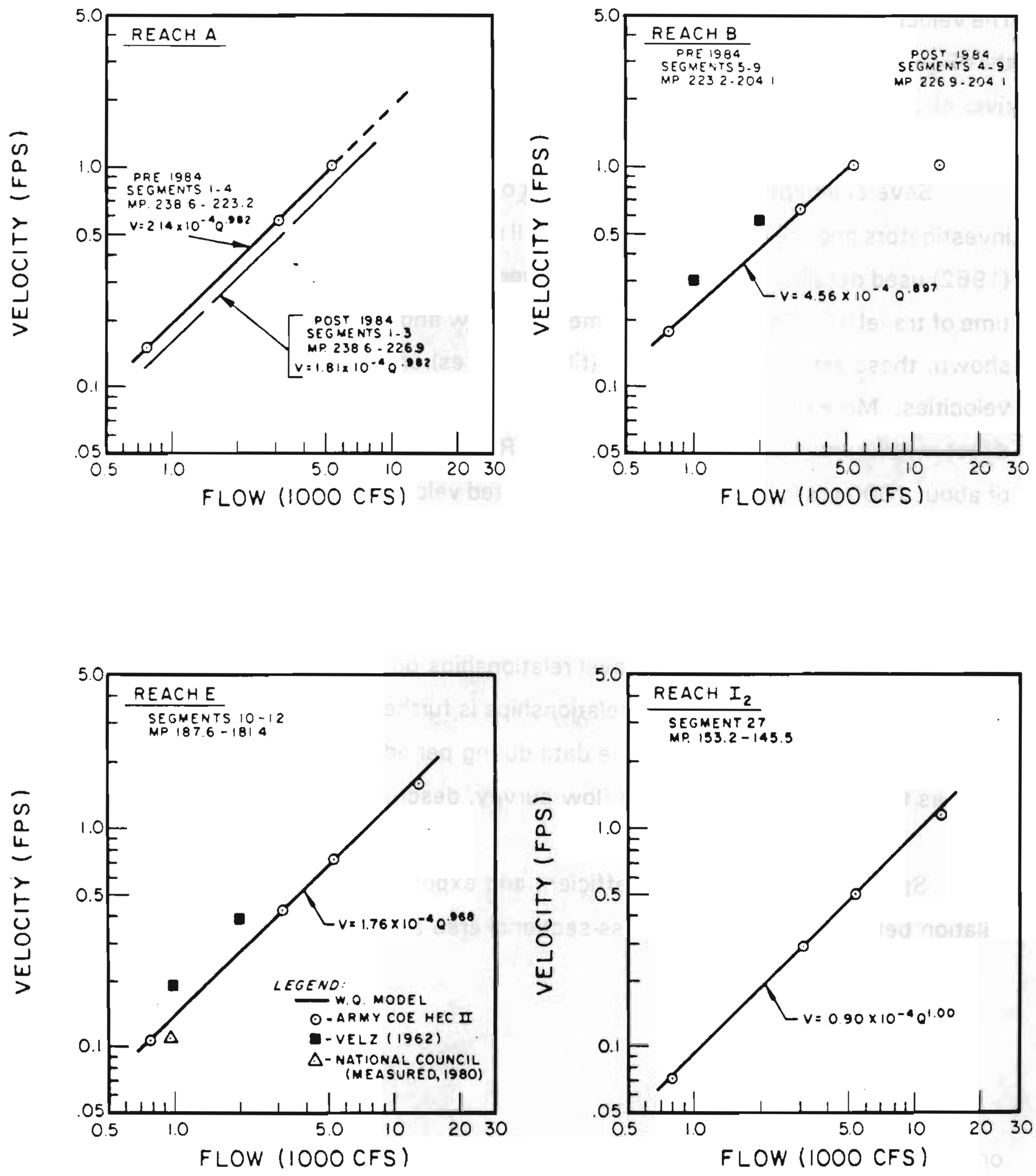


FIGURE 4-2. SUMMARY OF AVAILABLE ESTIMATES AND MEASUREMENTS OF TIME OF TRAVEL VELOCITY

The velocity coefficients and exponents for each reach are summarized in Table 4-1. (Note that Figure 4-2 and Table 4-1 include geometry relationships for both pre and post 1984 river channels.)

Several independent measurements of river velocity have been reported by other investigators and are compared to the HEC II results on Figure 4-2. In one early study Velz (1962) used detailed measurements of channel geometry to develop indirect estimates of time of travel based on channel volume, river flow and volume displaced per unit time. As shown, these estimates of velocity (filled squares) are somewhat higher than the HEC II velocities. More recently, the National Council conducted a dye tracer study to obtain a direct measurement of time of travel within Reach E, near Sterlington, Louisiana, at a flow of about 1000 cfs (NCASI, 1982). The reported velocity of 0.11 fps is slightly lower than, but in reasonable agreement with, the velocity estimated with the HEC II model of 0.14 fps. In view of the aforementioned results, the relative scarcity of supplementary field data, and the comprehensive nature of the HEC II data base, it was concluded that it was reasonable to base the time of travel relationships on the HEC II model. The validity of using HEC II based time of travel relationships is further supported by results of modeling analyses of conservative substance data during periods of time variable flow conditions, such as the September 1990 low flow survey, described later in this section.

Specifying the velocity coefficient and exponent for a reach implies the following relation between the average cross-sectional area and flow:

$$A = \frac{1}{C_u} Q^{(1 - P_u)} \qquad (4\text{-}2)$$

or

TABLE 4-1. OUACHITA RIVER MODEL GEOMETRY
AREA, VELOCITY AND DEPTH COEFFICIENTS AND EXPONENTS

Pre 1984

Time of Travel Reach	Upstream La. MP	Model Segments 1980	Model Segments 1978	Area Coefficient	Area Exponent	Velocity Coefficient	Velocity Exponent	Depth Coefficient	Depth Exponent
A	238.6	1-4		4,673	0.018	2.14×10^{-4}	.982	14.30	.013
B	223.2	5-9	1-5	2,192	0.103	4.56×10^{-4}	.897	7.17	.050
C	204.1	10	6	3,636	0.054	2.75×10^{-4}	.946	12.00	.018
D	194.3	11-13	7-9	4,950	0.070	2.02×10^{-4}	.930	15.03	.011
E	187.6	14-16	10-12	5,682	0.032	1.76×10^{-4}	.968	16.24	.006
F	181.4	17	13	8,696	0.0	1.15×10^{-4}	1.0	6.90	0.0
G	177.1	18-20	14-16	8,696	0.0	1.15×10^{-4}	1.0	13.6	0.0
H	168.9	21-24	17-20	11,630	0.0	0.86×10^{-4}	1.0	23.9	0.0
I	161.2	25-27	21-23	11,900	0.0	0.84×10^{-4}	1.0	24.7	0.0
(I_1)	161.2	25-26	21,22	12,500	0.0	0.80×10^{-4}	1.0	24.7	0.0
(I_2)	153.2	27	23	11,110	0.0	0.90×10^{-4}	1.0	24.7	0.0
J	145.5	28	24	11,500	0.0	0.87×10^{-4}	1.0	22.2	0.0
K	138.2	29	25	10,000	0.0	1.00×10^{-4}	1.0	26.0	0.0
L	130.2	30	26	13,330	0.0	0.75×10^{-4}	1.0	27.7	0.0

Revised Upstream Geometry Post 1984

Time of Travel Reach	Upstream La. MP	Model Segments 1990-1991	Area Coefficient	Area Exponent	Velocity Coefficient	Velocity Exponent	Depth Coefficient	Depth Exponent
A	238.6	1-3	5,520	0.018	1.81×10^{-4}	.982	17.20	.011
B	226.9	4-9	2,192	0.103	4.56×10^{-4}	.897	7.17	0.050

Note: Area, velocity and depth relationships are consistent with HEC II model results at flows less than 5,460 cfs in Reaches A through C and at flows less than 13,000 cfs in Reaches D through L.
Old Felsenthal Lock and Dam 6 located at MP 223.2
New Felsenthal Lock and Dam 6 located at MP 226.9

$$A = C_a\, Q^{P_a} \tag{4-3}$$

where:

- A = average cross-sectional area within the reach, square feet
- C_a = area coefficient (= $1/C_u$)
- P_a = area exponent (= $1 - P_u$)

A procedure analogous to that described for velocity was used to establish relationships between reach average depth and river flow as well. The cross sectional area and depth coefficients and exponents for each reach are also summarized in Table 4-1. An exponent of 0.0 is indicative of a reach in which the cross-sectional area is insensitive to changes in flow over the prescribed range of flows from which the exponent was determined.

It is important to note that the channel geometry is relatively insensitive to a seven-fold increase in the river flow to 5,460 cfs. This is because the Ouachita River is comprised of a series of impoundments rather than free-flowing reaches. Based on HEC II results at a flow of 5,460 cfs, the increases in cross-sectional area and depth in the study area are generally less than 10 percent, with the greatest increases noted in the upper reaches of the Ouachita River, in Louisiana.

4.1.3 Model Segmentation

The water quality model of the Ouachita River developed for the Louisiana 208 Study (Hydroscience, 1978) was comprised of 26 constant parameter model segments between the old Lock and Dam No. 6 (MP 223.2) and Columbia Lock and Dam. The original model was extended approximately 20 miles upstream of the old Lock and Dam 6 to the Saline River, as illustrated on Figure 4-3, as part of the 1980 modeling effort. The model was generally segmented at places where significant tributaries or waste loads enter the system, or where there is a significant change in channel geometry, as shown

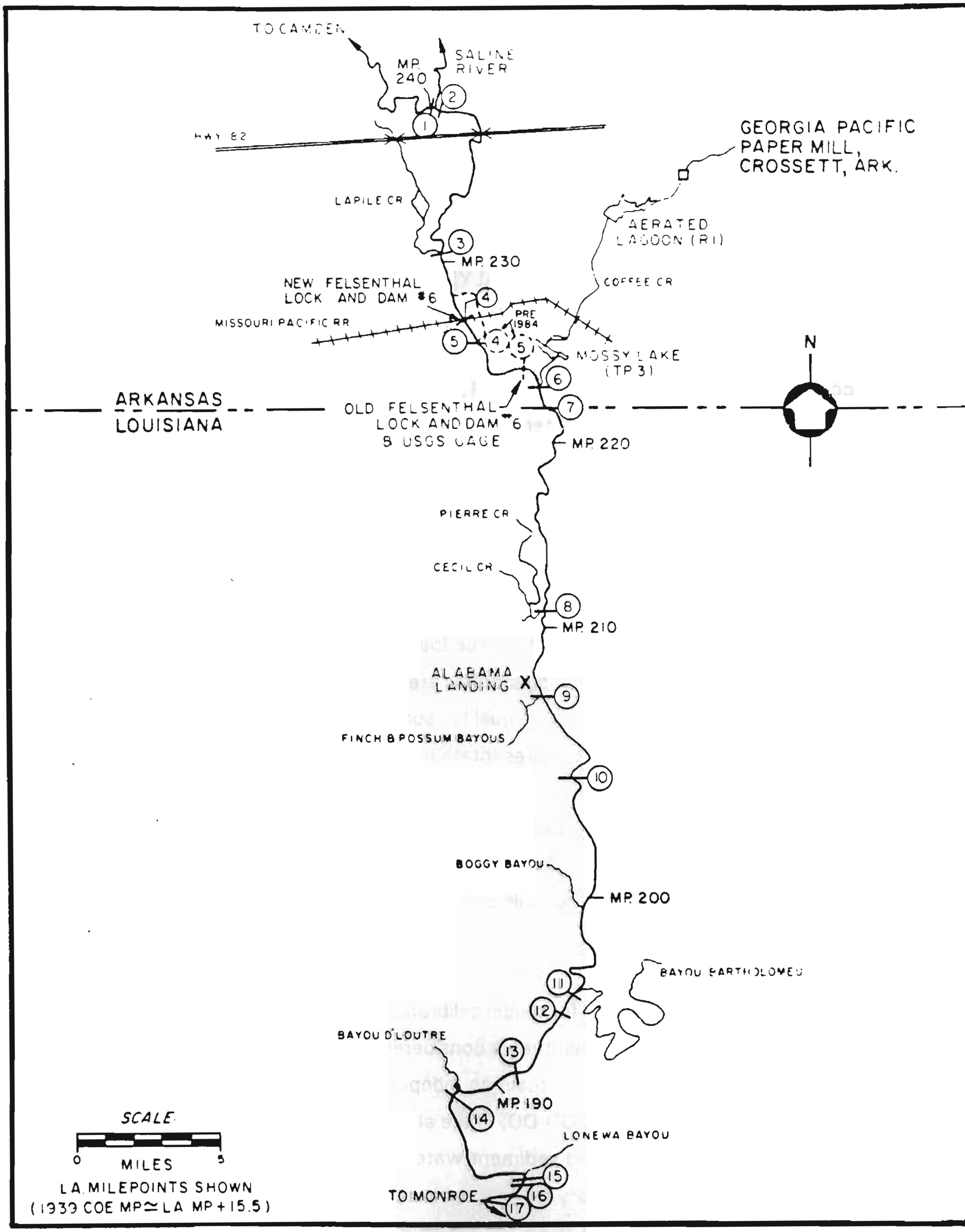


FIGURE 4-3. OUACHITA RIVER MODEL SEGMENTATION, SALINE RIVER TO LONEWA BAYOU

previously on Figure 4-1. The current modeling effort focuses on the river reach extending from just upstream of the Saline River, at approximately MP 238.9, to a location about 5 miles downstream of Bayou Bartholomew, at about MP 190.

4.2 CONSERVATIVE TRACER ANALYSIS

A rather extensive data base has been established, including intensive surveys conducted during 1980, 1990 and 1991, for the purposes of monitoring water quality and for use in development of a water quality model of the Ouachita River. The water quality stations sampled during these intensive surveys are summarized in Table 4-2 and their locations are indicated on Figure 4-4 (filled triangles). Since the new Felsenthal Lock and Dam went into operation in September 1984, some sampling locations were added to reflect the new dam location and associated changes in channel geometry. The first nine stations listed in Table 4-2 were established for the purpose of defining background water quality associated with non-point source loads and residual effects of distant upstream point source loads. The remaining stations are generally consistent with stations sampled as part of earlier 1978 water quality surveys, with several minor modifications incorporated to ensure that representative samples were collected in the vicinity of tributaries. Although there are some exceptions, most stations were sampled during each survey. It is noted that discrepancies in MP locations of as much as 0.5 miles may occur in some cases, depending on the organization conducting the survey and the manner in which MP's were recorded, but this should not significantly affect the results presented herein.

As stated previously, for model calibration purposes the water quality model is first applied to the analysis of constituents considered to be conservative, such as chloride and conductivity. This is done to obtain an independent check of the flow distribution used in the model calibration for BOD-DO, in the absence of other complicating factors such as decay rates, algal effects and sediment-water column interactions. Since a steady state model is used, it is necessary to assign constant model inputs when conducting these analyses. Ideally, data sets used for analysis will also represent conditions which are

TABLE 4-2. DESCRIPTION OF OUACHITA RIVER SAMPLING STATIONS

Station Designation	River Milepoint Louisiana Pre 1984	Post 1984	COE	Description
B1	240.0	238.9	255.5	Ouachita River, 0.6 mi. upstream of Saline River
BT1	239.4*	238.3*	254.9*	Saline River, 1.2 mi. from mouth
B2	238.9	237.8	254.4	Ouachita River, 0.5 mi. downstream of Saline River
B3	234.5	233.4	250.0	Ouachita River, 2.1 mi. downstream of Highway 82
BT2	230.0*	228.9*	245.5*	Lapile Creek, 0.5 mi. from mouth
B4	229.5	228.4	245.0	Ouachita River, 0.5 mi. downstream of Lapile Creek
B5	-	226.9		Ouachita River, 0.1 mi. upstream of new lock 6
B5A	-	226.7		Ouachita River 0.1 mi. downstream of new lock 6
BT3	226.0*	226.0*	241.5*	Grand Marais, 1.0 mi. south of railroad trestle
1	223.3	223.3	239.0	Ouachita River, 0.1 mi. upstream of old lock 6
2	223.1	223.1	238.5	Ouachita River, 0.1 mi. downstream of old lock 6
R1	-	-	-	Effluent from Georgia Pacific's aerated lagoon
TP3	222.0*	222.0*	238.0*	Effluent from Mossy Lake (Coffee Creek)
3	221.1	221.1	236.6	Ouachita River at Arkansas-Louisiana state line
3A	218.5	218.5	234.0	Ouachita River near Little Mallard Lake
4	215.5	215.5	231.0	Ouachita River, .25 mi. upstream of Shiloh Creek
4A	211.5	211.5	227.0	Ouachita River, .25 mi. upstream of Cecil Creek
T1	211.2*	211.2*	226.7*	Cecil Creek, 0.5 mi. from mouth
5	208.2	208.2	223.7	Ouachita River at Alabama Landing, .25 mi. upstream of Finch (Possum) Bayou
6	204.7	204.7	220.2	Ouachita River, at bend in river
7	199.5	199.5	215.0	Ouachita River, .25 mi upstream of Boggy Bayou
8	194.7	194.7	210.2	Ouachita River, .25 mi. upstream of Bayou Bartholomew
9	-	194.4	209.9	Ouachita River .1 mi. downstream of Bayou Bartholomew

* Ouachita River Milepoint at confluence with tributary
Note: Routine monitoring stations are very five miles, generally between ACOE MP 255 and MP 205.

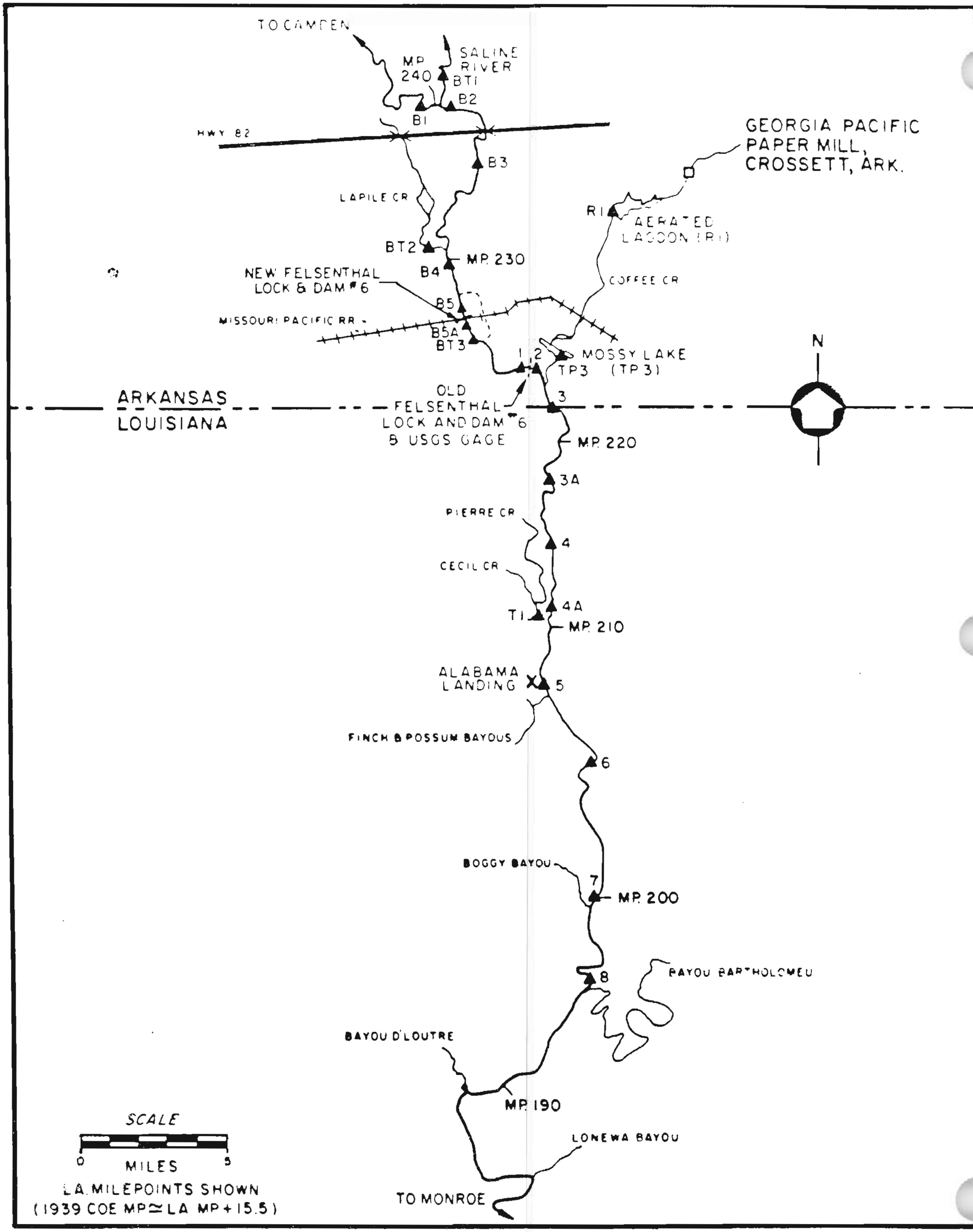


FIGURE 4-4. LOCATION OF SAMPLING STATIONS

reasonably steady in time, and if this is so, the approach is straight forward. As will be seen, however, this ideal situation is frequently not realized and the model must therefore be applied with care if a meaningful evaluation is to be performed.

Results of conservative tracer analyses will first be presented for several 1980 data sets. More recent analyses of intensive survey data sets from 1990 and 1991 will then be reviewed.

4.2.1 1980 Conservative Tracer Analyses

The water quality data and model results for three 1980 surveys are presented on Figure 4-5. The two upper graphs present results for the June high stage survey, when the river flow was estimated to be on the order of 30,000 cfs. Both the data and model indicate a very flat profile, with only a slight increase in the level of either constituent downstream of Coffee Creek. The high waste effluent chloride and conductivity levels are diluted by approximately 400 to 1 and thus have a negligible impact on the river at this high flow. This is in marked contrast to the instream response for the low flow, low stage surveys shown on the middle and lower panels of Figure 4-5, as discussed below.

During the July survey (middle panels), river flow at the old Lock 6 was 3086 cfs (3 day average) and the resulting instream concentration downstream of Coffee Creek shows a discernible increase relative to upstream levels. The calculated chloride profile generally brackets the measured chloride concentrations downstream of the discharge when the range in variation in upstream concentration and flow are considered. The calculated conductivity profile is in even better agreement with observed data, as the upstream conductivity levels were generally more consistent. Note that for this survey, the river flow decreased to 1120 cfs on the morning of July 22. Hence, the afternoon samples at the state line sample station were significantly elevated relative to the morning samples and this difference is readily accounted for by the flow transient. Further downstream stations would not have been expected to be influenced by this factor due to the increased travel time from the discharge point to those locations.

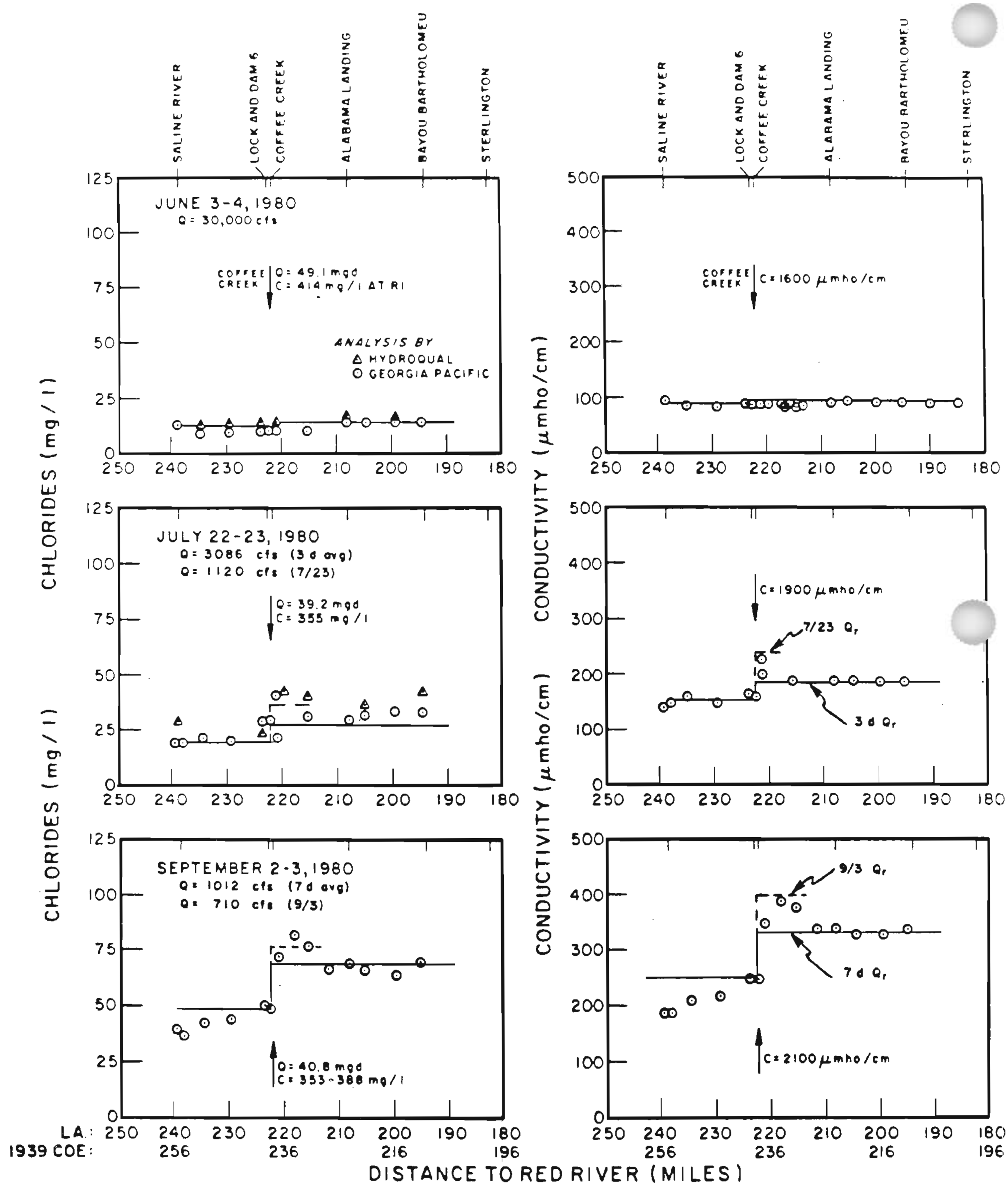


FIGURE 4-5. MODEL CALIBRATION, CONSERVATIVE SUBSTANCES (1980 SURVEYS)

During the relatively low flow survey of September 1980 (7 day average Q = 1012 cfs and 1 day average Q = 710 cfs at Lock 6), a much greater increase in both chloride and conductivity levels was observed at the point where Coffee Creek enters the Ouachita River (lower panels on Figure 4-5). The uncharacteristic gradually increasing levels of both constituents in the reach upstream of Lock 6 during this survey could not be calculated by the model unless a distributed load of a conservative substance was assumed. Alternatively, the gradual increase may also reflect a transient upstream boundary condition. The range of downstream model results shown, obtained using the measured concentrations at Lock 6 as the upstream boundary condition, in conjunction with the range of one day and seven day average flows, is in reasonably good agreement with the observed data, generally bracketing the range of instream levels immediately downstream of the discharge. Use of the observed concentrations at Lock 6 was considered reasonable in that it provides a means of checking the important mass balance at Coffee Creek. The fact that the model is slightly high in the downstream reach may reflect variation in the upstream boundary or effluent concentrations during the week prior to the survey.

4.2.2 September 1990 Conservative Tracer Analysis

The next low stage intensive survey was conducted during September 1990. Figure 4-6 shows a detailed time history of upstream river flow for this month. As indicated by the arrows, the survey was conducted on September 18 and 19, shortly after a low flow period, at the start of a rise in the hydrograph. The river flow averaged about 1200 cfs for the 7 day period prior to the survey, with this averaging period about equal to the travel time from Coffee Creek to the downstream end of the study area, near Bayou Bartholomew, at this flow rate. The daily flow prior to the survey was variable, however, ranging from a low of 676 cfs on September 15, 3 days prior to the survey, to 2,320 on September 17, the day before the survey.

The three fold variation in flow during the week prior to the September 1990 survey necessarily complicated the modeling analysis, given that a steady state model was being

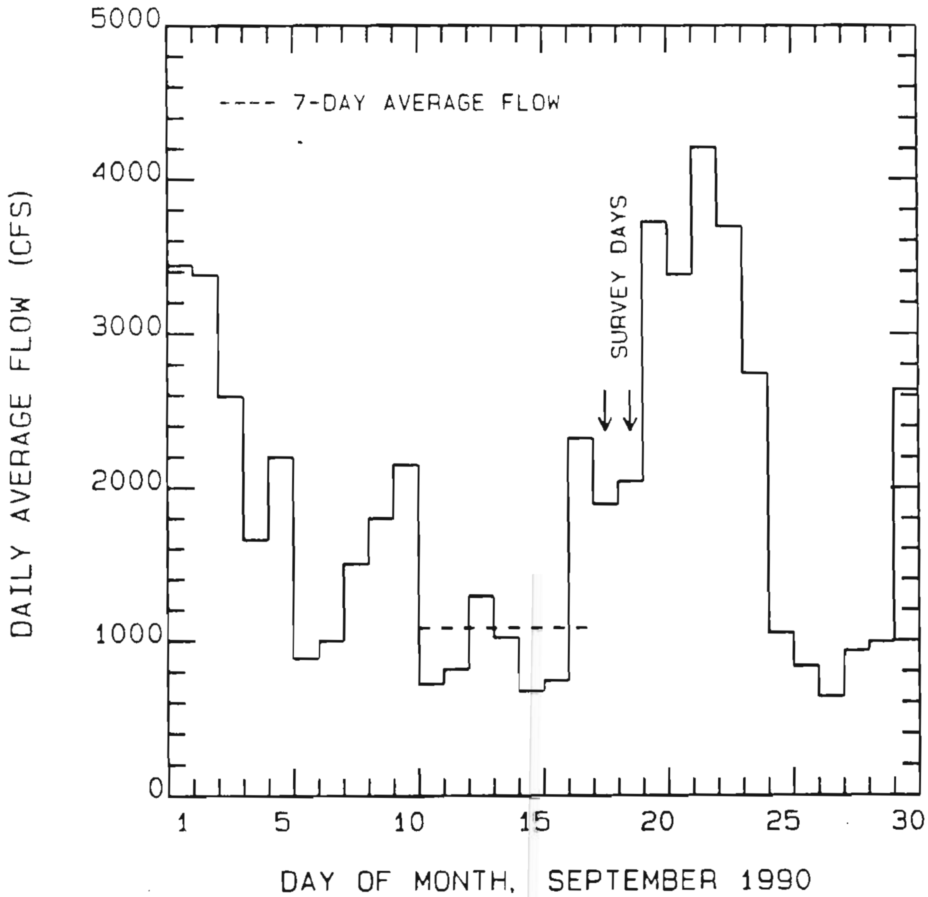


FIGURE 4-6. DAILY OUACHITA RIVER FLOW VERSUS TIME, SEPTEMBER 1990

employed. The approach followed is illustrated on Figure 4-7, using conductivity data. Data from upstream of Coffee Creek are shown for both days on both panels. The upper and lower panels show downstream data collected on September 18 and 19, respectively, as it was determined that it would be easier to evaluate the data downstream of the mass balance location on a daily basis, rather than averaging over the two day sampling period. Consider the upper panel first, for data collected on September 18, 1990. The solid line profile represents the model calculation for a conservative substance, with each break in the solid line corresponding to one day of travel time in the downstream direction. Travel tisme is estimated from the velocity-river flow relationships presented previously (Figure 4-2), and is shown on a scale directly above the downstream concentration profile for reference purposes. Within the first reach downstream of the discharge (MP 222 to MP 215.4), the flow rate as the river water flowed by Coffee Creek was estimated as the 24 hour average flow rate for the day of sampling, and the mass balance calculation is performed using this flow. Since on that day the river flow was relatively high, at 1,890 cfs, the effluent concentration is significantly diluted and concentrations in the river are relatively low. A similar result is obtained for the next reach downstream of the discharge (MP 215.4 to MP 207.4), corresponding to the water which passed by the discharge the day before sampling took place, when the river flow was 2,320 cfs. These first two reaches are in marked contrast to the next two reaches (MP 207.4 to 202.0), which reflect dilution associated with the two low flow days of September 16 and 15 (742 and 676 cfs, respectively), when relatively little dilution was available and resulting instream concentrations are therefore greatest. In both of these extremes, the model reproduces instream concentration variation quite well. This is also the case for the next two reaches between MP 202.0 and 194.4, when the river flow was about 1020 and 1290 cfs as it moved by the discharge (5 and 6 days earlier), and between MP 194.4 and 190.6, corresponding to low river flows of 723 to 817 cfs, about one week before the first sample date.

A similar analysis to that described above for September 18 was also performed for September 19, as shown on the lower panel of Figure 4-7. Note that the peak concentrations in the downstream reach have shifted a distance corresponding to one day

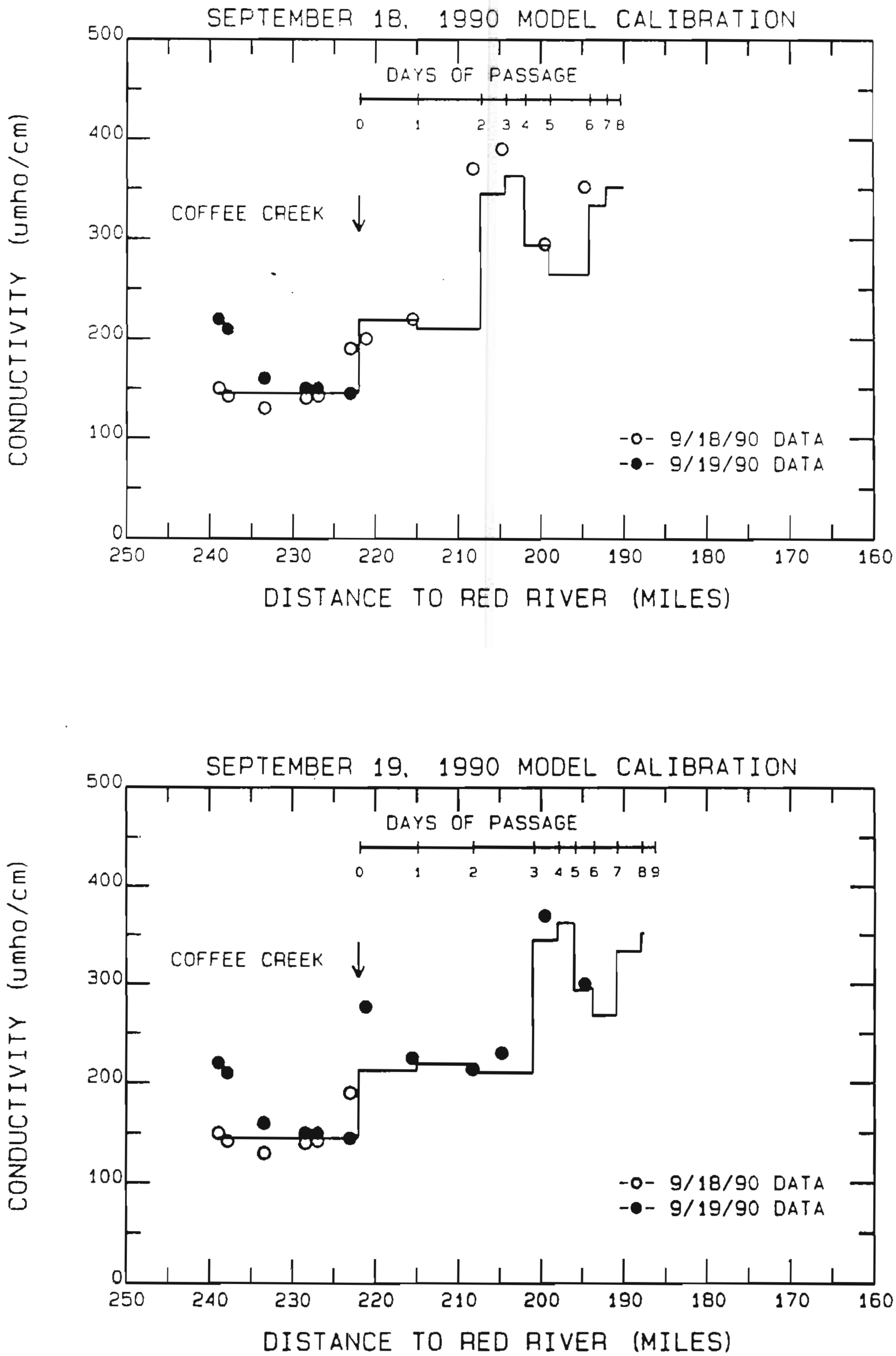


FIGURE 4-7. MODEL CALIBRATION RESULTS FOR CONDUCTIVITY, SEPTEMBER 18 AND 19, 1990

of travel time in the downstream direction, and both model and data are again in generally good agreement. In this regard, the time variable flow data and resulting time varying instream concentration profile provide what is effectively a time of travel study, with the analysis demonstrating fairly good agreement of the model with field observations. Furthermore, when repeated using both chloride and TDS data, comparable results are achieved (Figure 4-8).

4.2.3 October 1991 Conservative Tracer Analysis

The final data set analyzed as a conservative tracer is from the October 23 and 24, 1991 intensive survey. The daily flows for the week prior to this survey were relatively constant in time, averaging 1,946 cfs. Figure 4-9 compares results of the conservative tracer data to the model for this data set. All three tracers are relatively constant in the upstream region, increase as a result of the discharge from Coffee Creek, and then remain approximately constant further downstream, although the TDS data exhibit more variability than the other two tracers downstream of the discharge. One sample, collected near the state line (MP 220), is approximately twice as high in concentration of the other downstream data for each constituent, for some unexplained reason. Model results, indicated by the solid line profile, are in excellent agreement with the conductivity data (middle panel) and reasonable agreement with the TDS (bottom panel), but are consistently high in the downstream region in comparison to the chloride data, (top panels). It is expected that the accuracy of the laboratory analysis for chloride (colorometric determination) may be affected by the color of the river water and mill effluent. Conductivity and TDS are therefore believed to provide better measurements of conservative tracers. Flows assigned in the conservative tracer analysis shown on Figure 4-9 are therefore used in dissolved oxygen model simulations presented in Section 5.

4.2.4 Discussion of Conservative Tracer Analyses

The modeling analysis of conservative substances generally confirms the validity of the relatively low Saline River flows typically reported by the Arkansas USGS. Although

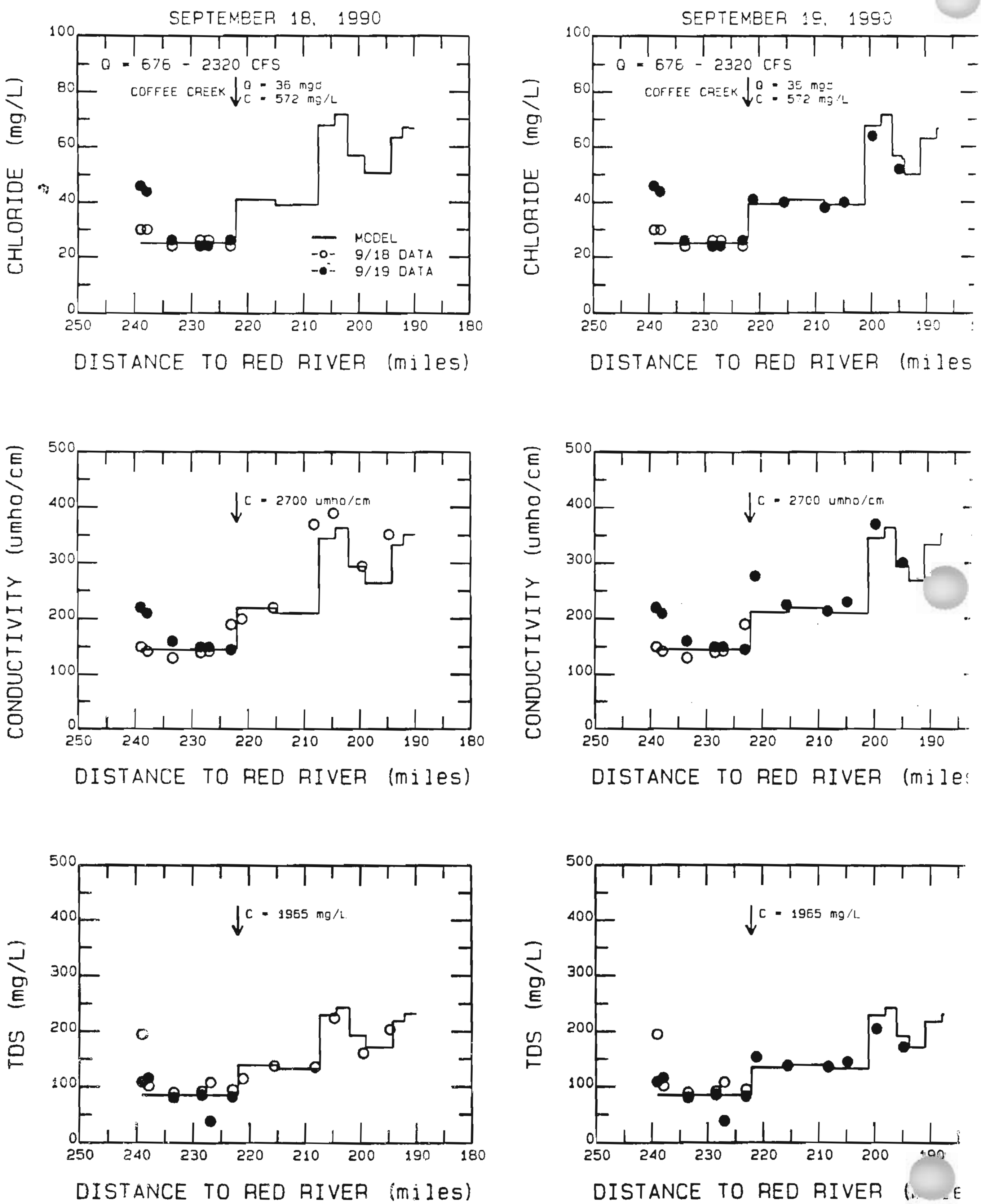


FIGURE 4-8. MODEL CALIBRATION, CONSERVATIVE SUBSTANCES
SEPTEMBER 18 AND 19, 1990

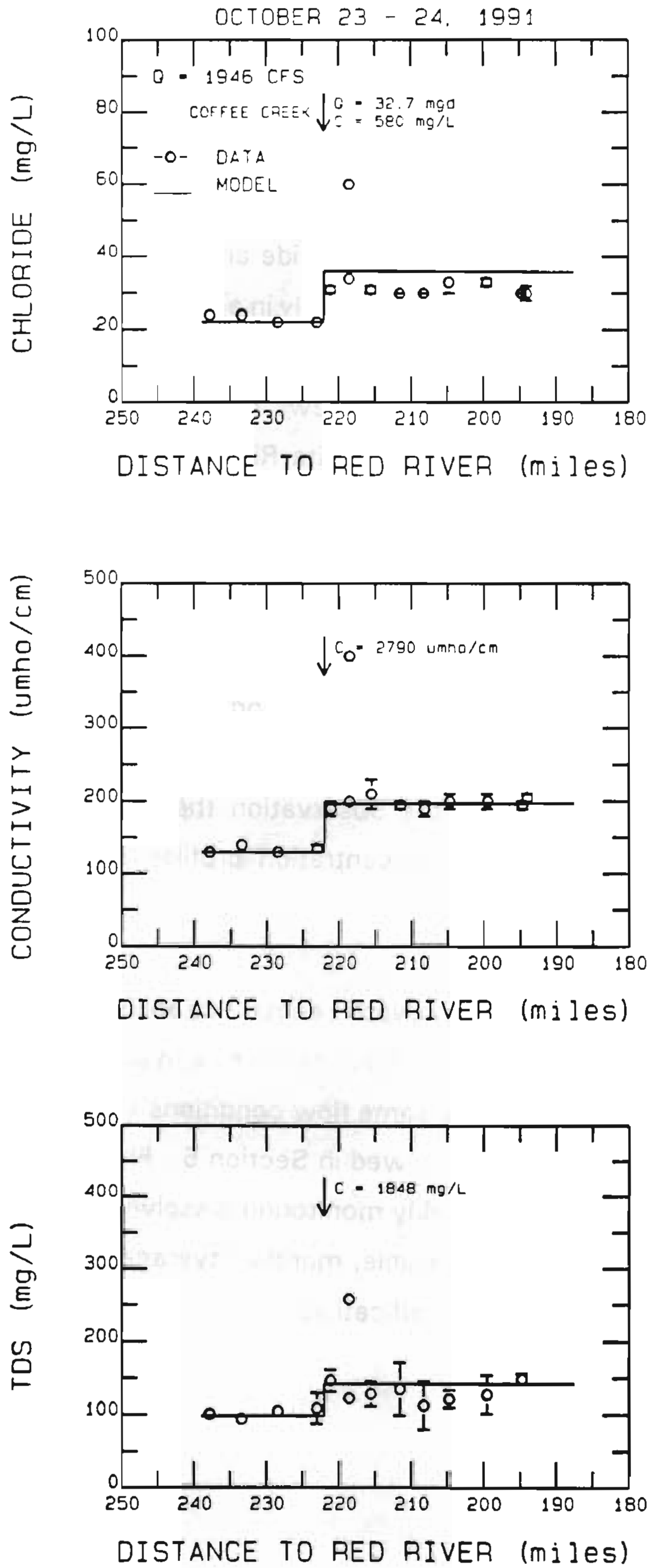


FIGURE 4-9. MODEL CALIBRATION, CONSERVATIVE SUBSTANCES, OCTOBER 23 AND 24, 1991

the Saline River has a drainage area at its mouth of approximately 50 percent of that of the Ouachita River, reported flows were considerably less than would be expected if the flow was proportional to drainage area. Measured chloride and conductivity levels in the Saline River were generally quite different than the levels measured in the Ouachita River. The fact that Ouachita River chloride and conductivity levels upstream and downstream of the Saline River were consistently in agreement, therefore, could only occur if the Saline River flow was relatively low. All other tributaries within the survey area, with the exception of Bayou Bartholomew, have drainage areas which are several orders of magnitude less than the Ouachita River. As such, very high concentrations would be required in conjunction with the relatively low flows that have been estimated in order for them to have a significant impact on the Ouachita River.

It also worthwhile to point out that although variations in flow at the time of the 1980 and 1990 low flow surveys complicated interpretation of the conservative tracer data, it also provided a demonstration that the flow information was reasonably goc This is supported by the observation that reported variations in flow were generally reflected in instream concentration profiles and in mass balance calculations at the discharge.

Overall, the conservative substance analyses indicate that appropriate upstream and effluent flows and concentrations have been specified. The analyses presented above are important in that these same flow conditions will be used in the nitrification and BOD-DO model results to be reviewed in Section 5. Flows will also be assigned for analysis of a July 1987 routine weekly monitoring dissolved oxygen data set, but since conservative tracer data are not available, monthly average upstream and effluent flows will be used without independent verification.

SECTION 5.

WATER QUALITY DATA AND MODEL CALIBRATION
FOR LOW STAGE BOD-DO ANALYSIS

Development of a model of dissolved oxygen for the Ouachita River requires specification of a number of model input parameters which must be properly evaluated if the model is to be successfully used as a tool for predicting water quality for alternative projection scenarios. These input parameters are generally evaluated independently of model simulations and then tested, and in some cases adjusted, based on comparisons of computed water quality responses to field data. This latter stage of the analysis is often referred to as the model calibration/verification phase of the model development effort, with the distinction between "calibration" and "verification" becoming somewhat blurred as feedback of information learned from analysis of each new verification data set is incorporated in analyses of earlier preliminary calibration data sets.

This section first reviews available information used to quantify the model input parameters for the dissolved oxygen model of the Ouachita River (Section 5.1). The results of model calibration/verification analyses will then be presented (Section 5.2) for the five low stage - low flow data sets analyzed. This review is followed by a discussion of the sensitivity of the model to selected model inputs (Section 5.3) and an evaluation of the contribution of the various sinks of dissolved oxygen to the dissolved oxygen deficit (Section 5.4). As a result of the distinctly different factors currently understood to be controlling water quality during high stage conditions, high stage modeling analyses are presented separately, in Section 6.

5.1 EVALUATION OF SOURCES AND SINKS OF DISSOLVED OXYGEN

The mathematical model used for the analyses described herein is program RIVER. RIVER is a relatively simple one dimensional, steady state, analytical solution model incorporating the following sources and sinks of oxygen: first order removal of CBOD and

NBOD, SOD, net photosynthesis and reaeration (Figure 5-1). CBOD and NBOD can be input to the model as boundary conditions, point sources and uniformly distributed loads. Each of these model components requires specification of one or more model inputs. This section summarizes the methods used to evaluate these model input parameters.

5.1.1 Reaeration

Reaeration is estimated using the O'Connor Equation (Section 3.4, Equation 3-9). This predictive equation provides results which are consistent with the limited field measurements reported for the Ouachita River, as summarized in Table 5-1 (after NCASI, 1982). A lower bound to the reaeration rate coefficient is set by a surface transfer coefficient of 2 ft/day, applicable under low wind conditions and typically assigned in deeper, low velocity reaches (Figure 5-2 and O'Connor, 1983).

TABLE 5-1. COMPARISON OF MEASURED AND PREDICTED REAERATION RATE COEFFICIENTS

		Reaeration Rate (1/d)	
Flow (cfs)	River Milepoint[a]	Measured[b]	O'Connor Equation
1,200	192 (2 mi)	0.02 ± 1000%	0.02
850	312 (18 mi)	0.17 ± 18%	0.16

[a]Length of river reach shown in parentheses.
[b]NCASI, 1982.

Reaeration at Lock and Dam 6 was significant during the 1980 surveys, with an average dam reaeration rate coefficient corresponding to 0.1/foot of free fall (range = 0.05 to 0.15/foot) for about a 12 foot elevation difference between the upper and lower pools. This reaeration generally resulted in an improvement in dissolved oxygen when river water flowed over the dam. Since 1984, when the new dam was placed in operation and the upper pool elevation was raised 3.4 feet, water has not been released over the top of the dam and this improvement in dissolved oxygen has not been realized. Rec

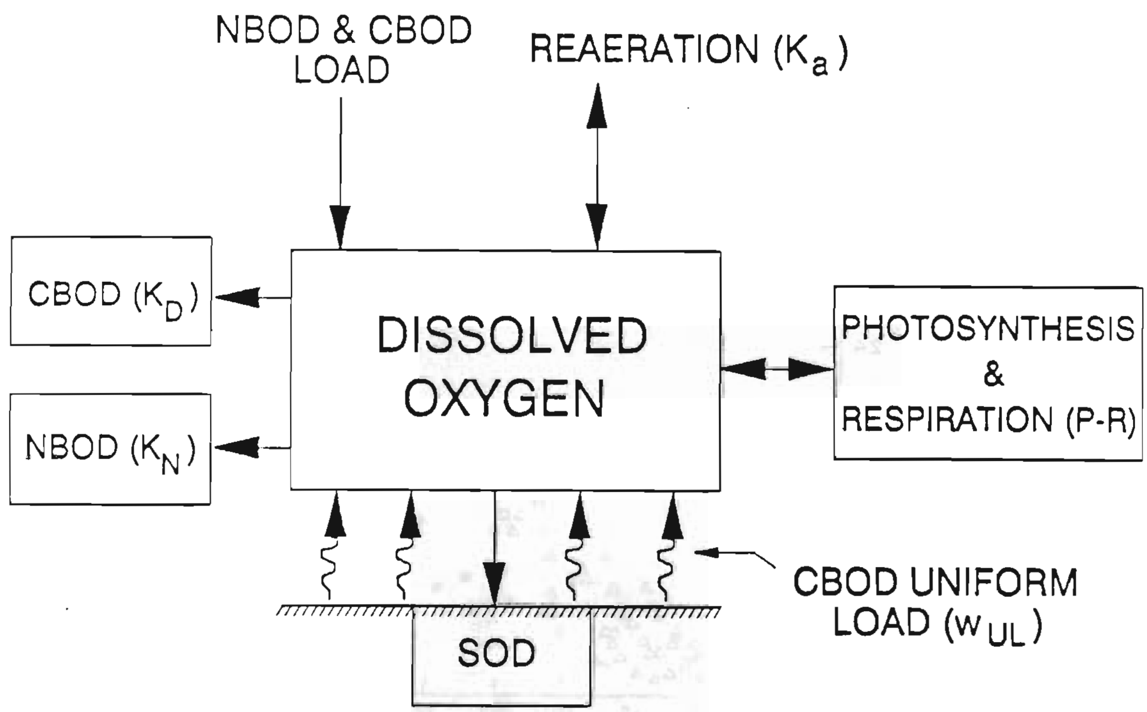


FIGURE 5-1. MODEL FRAMEWORK: SOURCES AND SINKS OF OXYGEN INCLUDED IN DISSOLVED OXYGEN MODEL OF THE OUACHITA RIVER

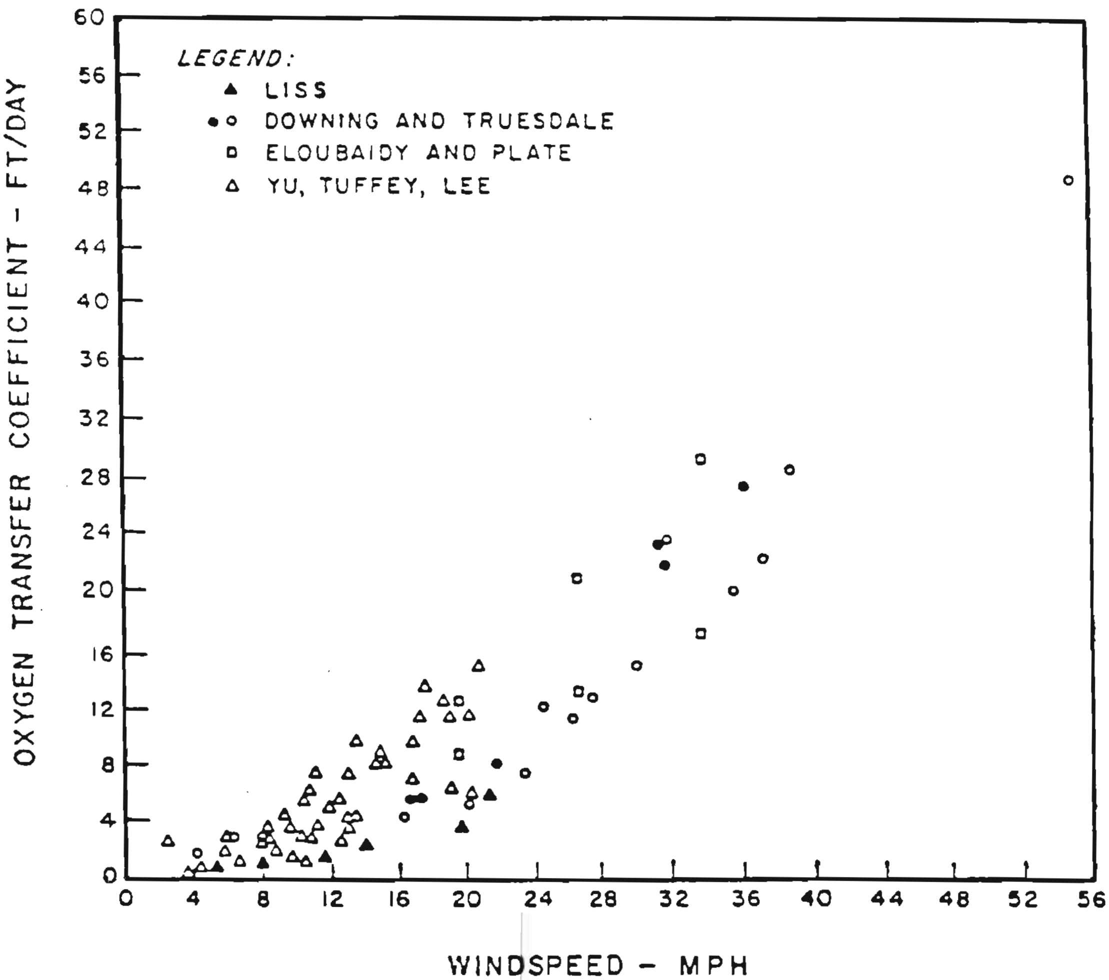


FIGURE 5-2. OXYGEN TRANSFER COEFFICIENT AS A FUNCTION OF WIND SPEED

structural modifications made during 1991 are expected to result in reaeration at the dam again, but the structural changes have not been approved as of this time and data documenting expected enhancements in dam reaeration have not been made. This effect is considered with regard to model projections for the Ouachita River (Section 7).

5.1.2 Algal Photosynthesis and Respiration

Phytoplankton affect the dissolved oxygen level in a natural water body through photosynthesis and respiration. Photosynthesis is a photochemical reaction in which carbon dioxide and water, in the presence of chlorophyll, are converted to organic carbon and oxygen. Because sunlight is the source of energy for this reaction, photosynthesis occurs during the daytime. The organic carbon produced by photosynthesis is constantly being oxidized to provide energy for metabolic functions. This oxygen consuming reaction is called respiration and occurs during both day and night. Since program RIVER represents algal effects on a steady state basis, the program inputs must be converted to 24 hour depth averaged values of gross photosynthesis (P_a) and respiration (R). The program uses the net rate ($P_n = P_a - R$), which may be either a source or sink of oxygen, to compute the spatial distribution of dissolved oxygen in the river.

The net production of oxygen by photosynthesis varies during the year as a function of a number of variables, including sunlight, temperature and available nutrients. The discussion which follows describes some of the analyses performed and illustrates the survey to survey differences that have been observed.

5.1.2.1 June 3 and 4, 1980

During the June 3 and 4, 1980 high stage survey (Appendix A1), the first intensive survey conducted during the initial modeling effort completed by Georgia Pacific, the effect of photosynthesis was not found to be significant. Morning and afternoon measurements of dissolved oxygen remained unchanged and total chlorophyll 'a' levels of 2 to 4 μg/L were also indicative of low phytoplankton levels.

5.1.2.2 July 22 - 23, 1980

Downstream sampling was performed on a overcast day and afternoon dissolved oxygen levels at some stations were slightly lower than morning levels. This is in contrast to upstream samples, collected on a clear day, which showed slightly elevated dissolved oxygen levels in the afternoon relative to the morning.

5.1.2.3 September 3 - 5, 1980

As a result of the increased level of photosynthetic activity observed at the downstream end of the study area during this intensive survey (Appendix A3), supplemental in situ diurnal dissolved oxygen monitoring studies were performed. These measurements of the diurnal variation of dissolved oxygen consisted of monitoring the dissolved oxygen concentration with depth in the water column over the daylight hours (Appendix E1). Results of this study formed the basis for the analysis of the effect of algal photosynthesis and respiration originally incorporated in the Ouachita River model. (More recent light and dark bottle study results will be presented subsequently.)

The dissolved oxygen data collected at the time of the September 3 water quality survey and on September 5, when the diurnal measurements were made, are presented on Figure 5-3. The diurnals were measured at Station 2, downstream of Lock 6 and upstream of Coffee Creek, as well as at Stations 4 and 6, about 10 and 20 miles downstream of Coffee Creek, respectively. Solar radiation data recorded on September 3 and 5, 1980 at Northeast Louisiana University, in Monroe, Louisiana, is shown in the upper graph of this figure. On September 5, somewhat erratic solar radiation was recorded due to scattered cloud cover, and the solar radiation decreased somewhat faster with the onset of early evening thunderstorms. Throughout most of the day, however, radiation levels were comparable on the two days. At Station 2, the variation of the dissolved oxygen concentration during the day was less than 1 mg/L, based on averages over depth. Within the water column, as shown on the right side, early morning measurements indicated vertical concentration gradients were minimal. At 3:40

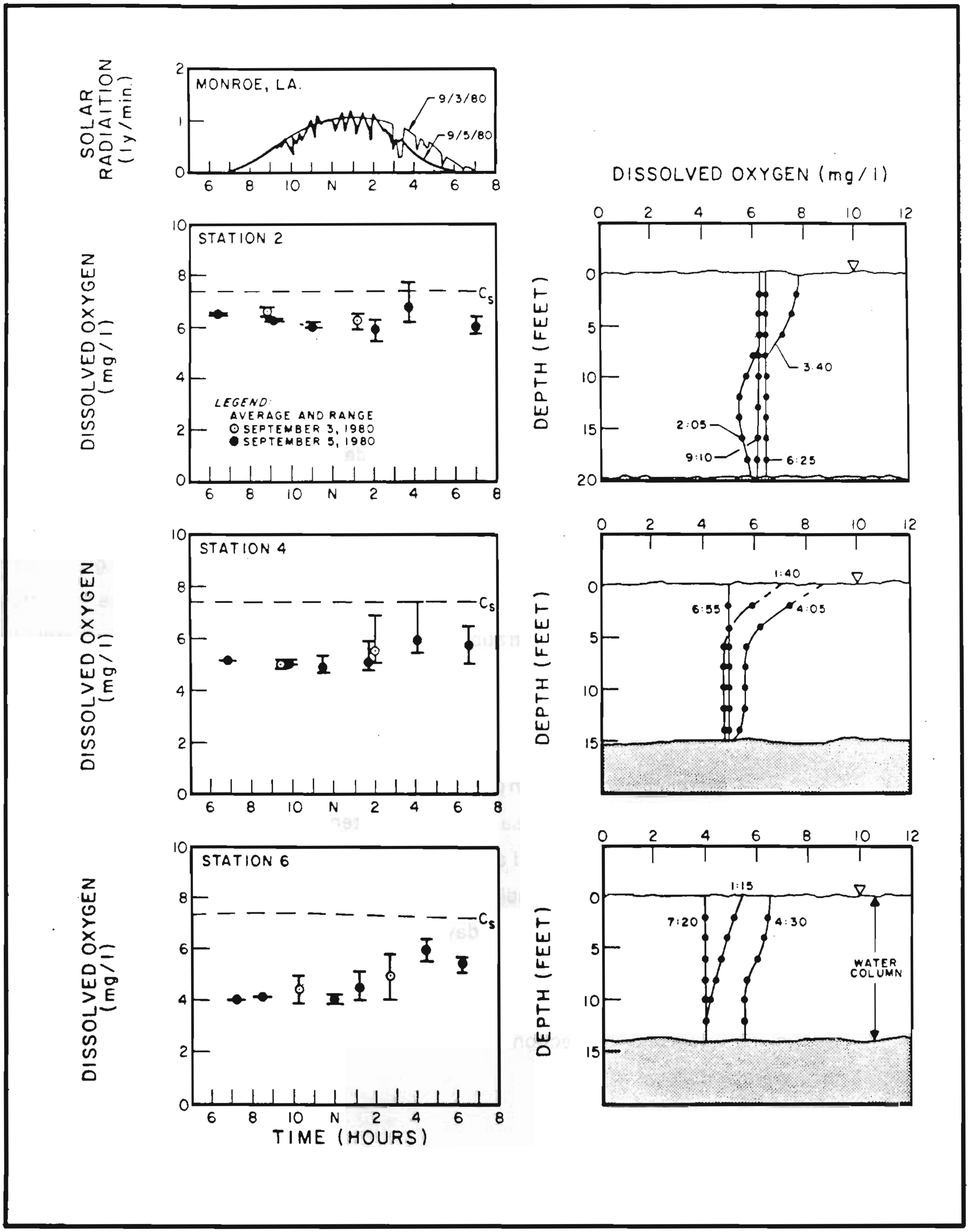


FIGURE 5-3: DIURNAL DISSOLVED OXYGEN STUDY,
SEPTEMBER 3 AND 5, 1980

dissolved oxygen levels were highest near the surface due to the increased rate of photosynthesis in the surface waters, or euphotic zone, where sunlight penetration is significant. The light extinction coefficient of approximately 0.8/ft reduced the light intensity to 1 percent of the surface intensity within 6 ft of the water surface.

The change in dissolved oxygen concentration at Station 4 was more pronounced than at Station 2, with an increase of slightly more than 1 mg/L occurring over the course of the day. Vertical stratification was also more clearly evidenced here, and by late afternoon the dissolved oxygen concentration was beginning to increase throughout the water column. This increase at the greater depths is attributed to the combined effects of oxygen production in the euphotic zone and vertical mixing within the water column. Station 6 data showed the greatest diurnal variation of dissolved oxygen concentration, ranging from 4 mg/L to 6 mg/L over the day. Once again, early morning vertical profiles within the water column reflect a uniform concentration from surface to bottom. The dissolved oxygen concentration increased steadily during the afternoon, with concentrations at the surface increasing first and the increase at deeper levels lagging slightly in time.

Morning and afternoon dissolved oxygen data (open data symbols) measured during the intensive water quality sampling on September 3 are also included with the diurnal data of Figure 5-3. The good agreement of these data with the diurnal variation observed on September 5th is an indication that the measured diurnals are representative of conditions that occurred on the day of the intensive water quality survey.

The diurnal dissolved oxygen measurements indicate an increasing diurnal variation in the downstream direction. Chlorophyll 'a' measurements for the September survey were as follows:

Station	Total Chlorophyll 'a' (μg/L)	Phaeophytin Corrected Chlorophyll 'a' (μg/L)
B2	21.9	3.7
1	6.5	3.1
4	6.2	nil
8	7.9	3.1

With the exception of the relatively high total chlorophyll 'a' concentration observed at station B2, these data suggest that a uniform algal population existed within the study area when these measurements were made. This is more apparent if one considers the phaeophytin corrected chlorophyll 'a' concentrations (study area average = 3.3 μg/L), which are a better indication of the active phytoplankton in the water column. (These latter results were in good general agreement with the pre-1980 STORET data, which averaged 3.1 μg/L at Felsenthal, for 17 samples).

The smaller diurnal variation observed at Station 2 in comparison to the further downstream stations may in part reflect the dampening effect which dam reaeration has on upstream diurnal activity. Additionally, with the low reaeration rate in the Ouachita River, a distance of 30 to 40 miles is required for the dissolved oxygen variation due to photosynthesis to approach its spatial steady state and to achieve a fully developed diurnal range.

The rates of photosynthesis and respiration were estimated for the September survey based the "delta method" (Chapra and Di Toro, 1991). This procedure, which provides an estimate of P_m, the maximum depth averaged rate of photosynthesis over the day, is dependent only on estimates of the reaeration coefficient and the diurnal range, or "delta" dissolved oxygen concentration. The procedure is independent of estimates of other sources and sinks of oxygen such as carbonaceous BOD, ammonia nitrification and SOD, and it is also not very sensitive to the estimate of K_a for systems like the Ouachita River, where K_a is low (<2/day). Algal respiration was estimated as 10 percent of the photosynthesis rate under ideal light conditions, and as a result, the net rate of photosynthesis (averaged over the depth of the water column and over night and day) was effectively zero (0.06 mg/L/day).

It is interesting to note that the estimated rate of gross photosynthesis was relatively high, considering the low measured chlorophyll 'a' concentrations, but consistent with the significant diurnal variation measured during the September 1980 survey. The level of photosynthesis required to produce the observed diurnal variation of dissolved oxygen could possibly be attributed to the contribution of rooted aquatic plants and floating algae, neither of which would be reflected in the chlorophyll 'a' measurements. Since sunlight penetration in the river was limited to the near surface layer, respiration was sufficient to offset the primary production and the net effect of primary productivity on the average daily dissolved oxygen level in the river during the September 1980 survey was not significant.

5.1.2.4 July 7, 1990

Results during this high stage survey (Appendix A4) were similar to the June 1980 survey. Dissolved oxygen levels measured five feet below the surface showed little evidence of diurnal activity at the selected stations which were checked. Chlorophyll 'a' levels were less than 2 μg/L.

5.1.2.5 September 20 and October 1 and 2, 1990

More recent data continue to indicate that algal productivity is not a major factor in the Ouachita River, upstream of Bayou Bartholomew. During the September 18 and 19, 1990 intensive water quality survey (Appendix A5), morning and afternoon dissolved oxygen measurements, averaged over depth in the water column, were not markedly different. Light and dark bottle studies were performed shortly after this survey to further evaluate this factor. The data are summarized in Appendix E2. Figure 5-4 presents typical results for these studies for a sample from Station 4. The upper left panel shows dissolved oxygen versus time in a light bottle containing river water and submerged i

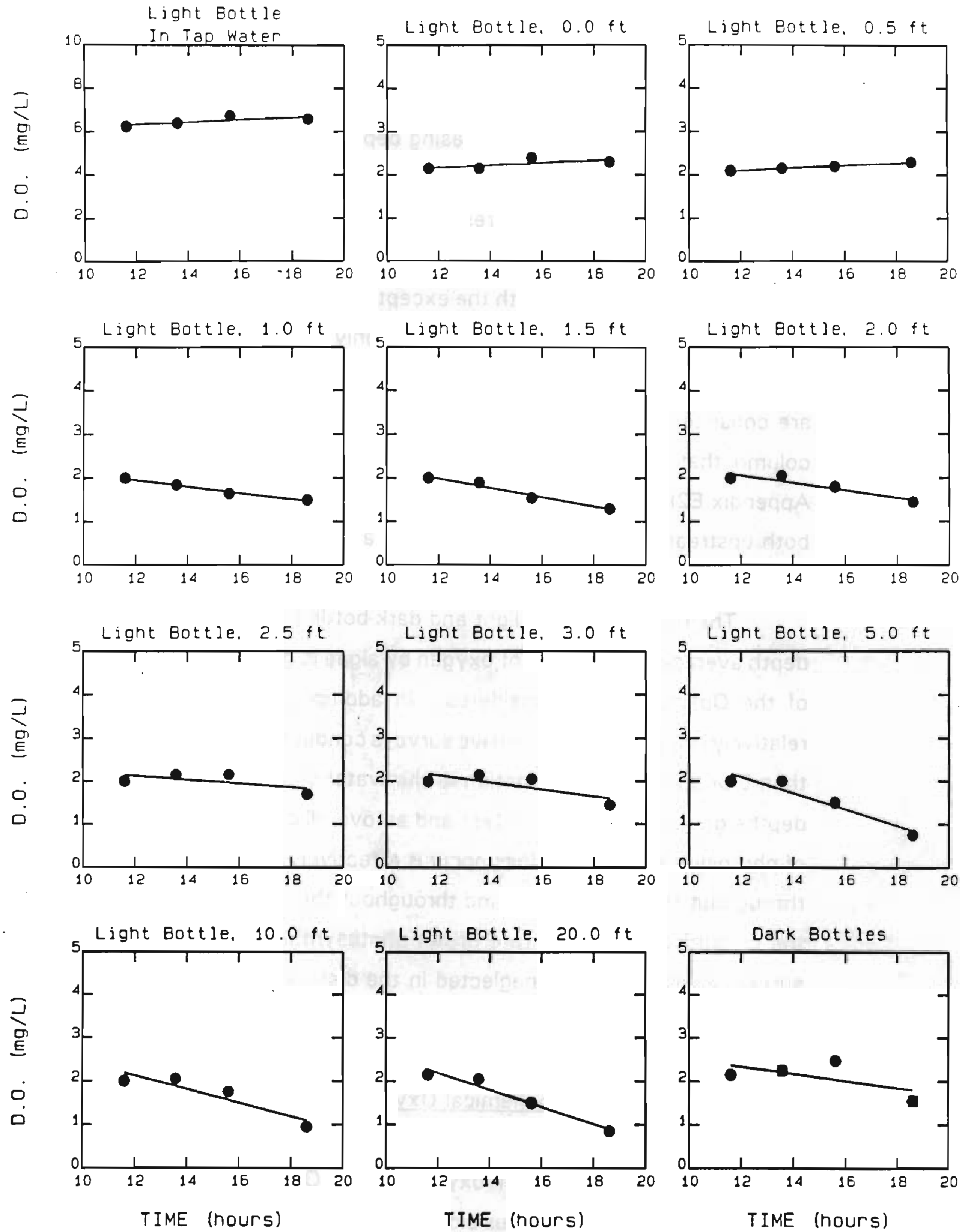


FIGURE 5-4. LIGHT AND DARK BOTTLE STUDY RESULTS, STATION 4, OCTOBER 1, 1990

shallow tray of tap water, as a means of measuring the productivity immediately at the water surface, where light intensity is greatest. Dissolved oxygen data over time, for bottles kept in the water at increasing depths beneath the water surface (at the surface and at 0.5, 1, 1.5, 2, 2.5, 3, 5, 10 and 20 feet) are then shown on the panels going from left to right and top to bottom, respectively. Finally, dark bottle results, in which the bottle was kept out of the sunlight for purposes of measuring community respiration, are shown on the lower right panel. With the exception of the bottle submerged in tap water and the two bottles at 0.0 and 0.5 feet, where only slight increases in dissolved oxygen were measured, net oxygen production was not generally observed at any depth. These results are consistent with the rapid extinction of the incident solar radiation within the water column that was measured (about 80 percent per foot beneath the water surface; Appendix E2). The rather high rate of light extinction in the water column was observed both upstream and downstream of Lock and Dam 6.

The results of these light and dark bottle studies provide further evidence that th depth averaged production of oxygen by algae is generally not significant within the reach of the Ouachita River considered. In addition, chlorophyll 'a' levels continued to be relatively low during the intensive surveys conducted during 1990 and 1991 (typically less than 5 μg/L) and light extinction in the water column has been consistently high (secchi depths generally less than 5 feet and an overall depth of 20 feet or more). The low level of photosynthesis which does occur is effectively offset by algal respiration, which occurs throughout the 24 hour day and throughout the entire depth of the water column. With the exception of the low rate of net photosynthesis estimated for the September 1980 survey, algal effects are neglected in the dissolved oxygen model of the Ouachita River that has been developed.

5.1.3 Carbonaceous Biochemical Oxygen Demand (CBOD)

An important sink of oxygen in the Ouachita River is oxidation of carbon, as measured in the carbonaceous biochemical oxygen demand (CBOD) test. The model of t Ouachita River considers three main sources of CBOD to the river system: (1) the load from the Georgia Pacific pulp and paper mill at Crossett, (2) the upstream boundary load, and (3) a distributed source of CBOD which is attributed to the release of dissolved organic carbon (DOC) from anaerobic decomposition of particulate organic material in the bottom sediments. Since the nearest major upstream point source load to the river is more than 50 miles upstream the residual effects of upstream point sources are not significant. The upstream boundary condition to the study area is believed to represent the cumulative effects of distributed sources in the distant upstream region, north of the Saline River, that are similar in nature to those described below, for the current study area (from the Saline River to Bayou Bartholomew).

5.1.3.1 BOD Time Series Results

CBOD inputs for the calibration analyses to be presented were assigned based on measured upstream, tributary and effluent concentrations. For permitting purposes, practical considerations necessitate use of the five day total BOD concentration. This is a measure of the amount of oxygen consumed in a BOD bottle over the first five days of incubation of the sample at 20°C. For modeling purposes, it is necessary to use the overall cumulative oxygen demand of the sample after all of the degradable organic carbon in the sample has been oxidized (the "ultimate" CBOD), a process which typically requires a much longer incubation period of 60 days or more. BOD time series were generally performed to evaluate the ultimate carbonaceous biochemical oxygen demand ($CBOD_u$). In these tests, the measured total BOD is corrected for nitrification in the BOD bottle by subtracting the nitrogenous oxygen demand, which is estimated based on stoichiometric considerations (4.57 times the change in nitrate nitrogen in the BOD bottle). Figure 5-5 shows representative BOD time series data for river samples. For paper mill effluents the ratio of $f = CBOD_u/BOD_5$ as a function of effluent BOD_5 has also been characterized (Figure 5-6). This relationship can be used in the absence of $CBOD_u$ data, such as in model projections for waste load allocations based on BOD_5. BOD time series data from the 1980, 1990 and 1991 sampling efforts are included in Appendices A1 - A7 (intensive surveys) and Appendix C2 (upstream and effluent monitoring programs).

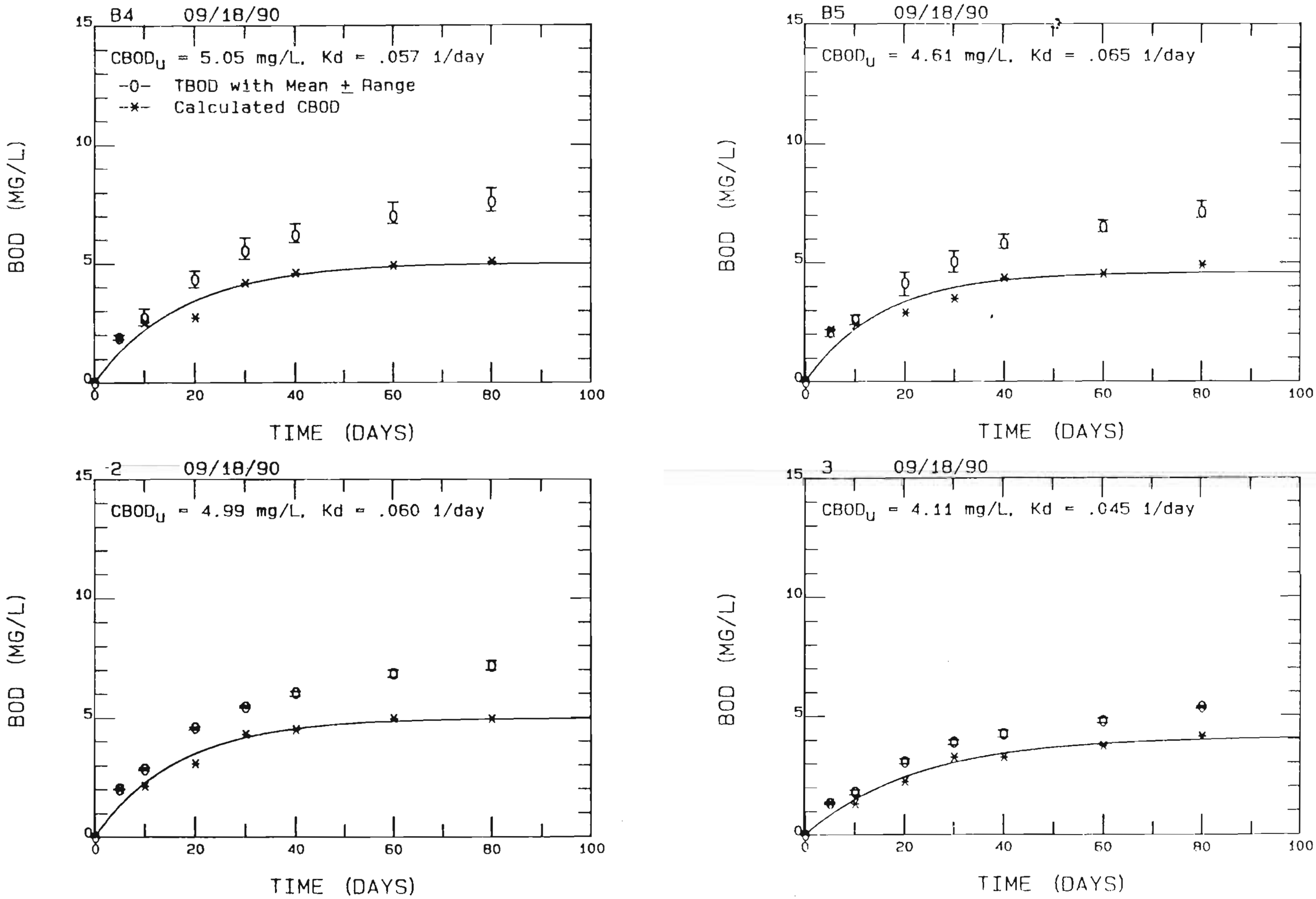


FIGURE 5-5. REPRESENTATIVE BOD TIME SERIES RESULTS FOR RIVER SAMPLES

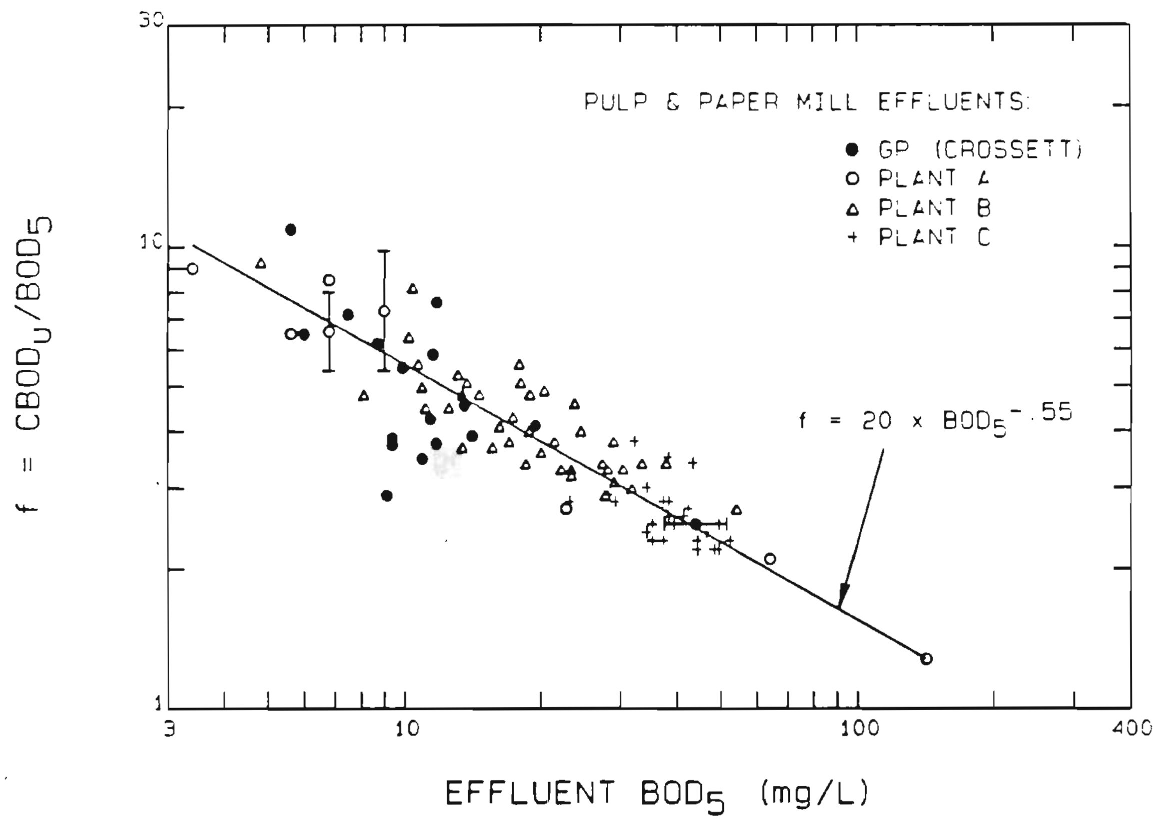


FIGURE 5-6. RELATIONSHIP OF $f = CBOD_U/BOD_5$ TO EFFLUENT BOD_5 FOR PAPER MILL EFFLUENTS

5.1.3.2 CBOD Oxidation Rate

An important motivation for conducting the experimental studies completed during 1991 was to obtain an improved definition of the magnitude of the background and point source CBOD oxidation rate coefficients used in the water quality model of the Ouachita River. Prior to 1991 the CBOD oxidation rate in the river had been estimated in two ways. First, the BOD bottle rate estimated from BOD time series analyses, typically 0.05/day at 20°C, was assigned to the uniform background load. Next, as part of the model calibration analysis, the oxidation rate of the effluent was set at 0.1/day at 20°C based on comparison of the model results with instream $CBOD_u$ data downstream of the discharge, as well as from consideration of the corresponding comparison of model and data for dissolved oxygen (i.e., from the agreement of model and data with regard to the overall dissolved oxygen balance in the river).

In addition to collection and analysis of an additional intensive survey data se several other approaches were used during the 1991 studies to better define the CBOD oxidation rates. These approaches included measurement of total organic carbon (TOC) data in conjunction with CBOD time series data (at $t = 0$ and at $t = 60$ days to estimate the biodegradable TOC by difference), and Warburg respirometer studies. The results of these special studies are discussed below, prior to a review of the model calibration, where comparisons of model results to instream CBOD spatial profiles will be considered as a further method of evaluation.

Several steps were taken during the October 23 and 24, 1991 intensive survey to obtain a better definition of the oxidation rate of effluent $CBOD_u$. First, TOC was measured at each sample station where samples for CBOD time series measurements were collected. It was intended that the test precision of TOC relative to BOD would be improved and hence the results more readily interpretable. Additional stations were also added in the reach immediately downstream of the discharge to provide a better statistical basis for evaluating the oxidation rate from the fit of the model to the instream dat These TOC data are shown with trend lines indicating the general shape of the profiles o

Figure 5-7. "Total" TOC, the initial TOC of the sample at the start of the BOD time series test, is shown on the upper left panel. The upper right panel shows residual TOC levels in the BOD bottle at the end of the BOD time series test (at t = 60 days), when essentially all of the degradable TOC has been oxidized. (This is consistent with the observation that the oxygen uptake in the BOD bottle is nearly complete at that time). This residual, nondegradable TOC is highly refractory and can be approximated as a conservative substance. The similarity in form of the spatial profile of these measurements to the conservative tracer profiles presented previously (Figure 4-9) supports this interpretation. It also follows that since this residual TOC fraction does not increase significantly in the downstream direction any TOC release from the sediment is likely to consist of a degradable organic carbon.

Subtraction of the residual TOC from the "total" TOC yields the results shown on the bottom left panel of Figure 5-7, the spatial profile of degradable TOC. The profile is remarkably similar in form to the $CBOD_u$ profile (lower right panel), but unfortunately, the data do not provide a significantly better defined gradient in the river reach downstream of Coffee Creek (about MP 222). Hence, while this approach confirms the structure of the $CBOD_u$ data, it is of limited use in defining the TOC or $CBOD_u$ oxidation rate coefficient.

The TOC data just reviewed can be used to evaluate the oxygen utilization requirements of the degradable TOC. This information will be needed in assessing the TOC flux data reviewed in the next section. First, note that the $CBOD_u$ spatial profile, though similar in shape, is about a factor of four greater in magnitude than the corresponding degradable TOC profile. The upper panel of Figure 5-8 presents a cross-plot of the $CBOD_u$ concentration (the oxygen utilization requirement of the carbon utilized as substrate in the BOD time series test) versus "delta" TOC (the biodegradable component of the TOC), with lines having slopes of 2.67 and 4.0 grams oxygen/gram carbon drawn through the data. The line corresponding to the lower number would be associated with oxidation of carbon in the form of glucose and is generally below most of the data. The line corresponding to 4.0 grams oxygen/gram carbon more nearly passes through the body of the data, but in either case significant variability exists in the data about the line. A probability plot of the

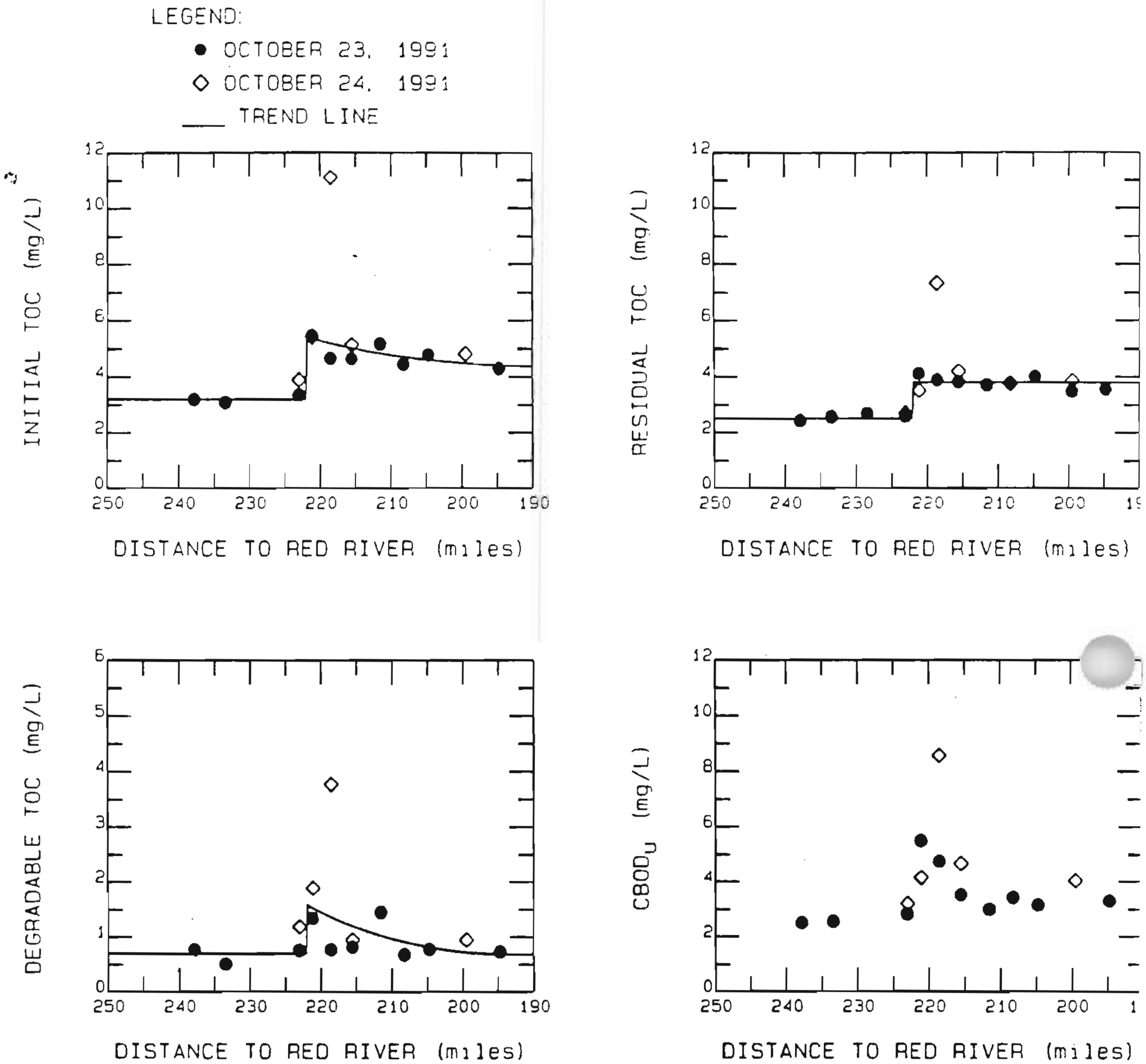
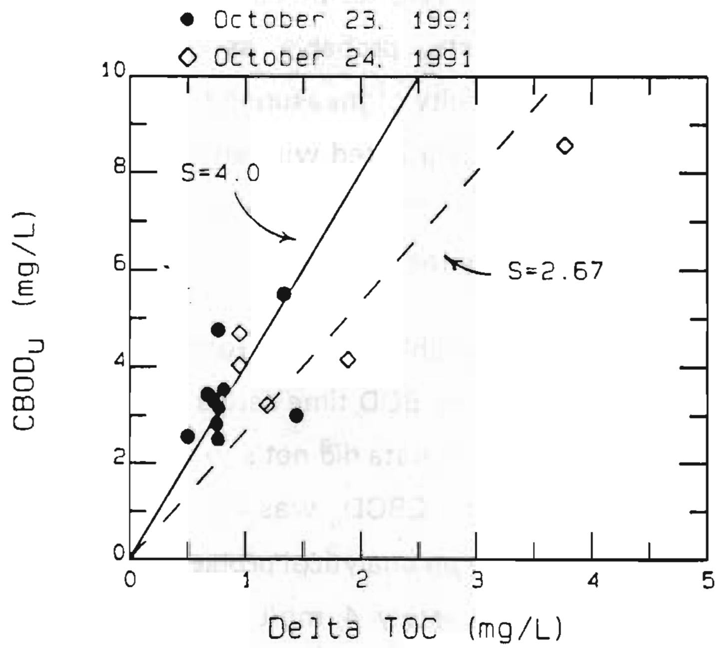


FIGURE 5-7. SPATIAL PROFILES OF INITIAL TOC, RESIDUAL TOC, DEGRADABLE TOC AND CBOD$_u$ DATA, OCTOBER 23 AND 24, 1991



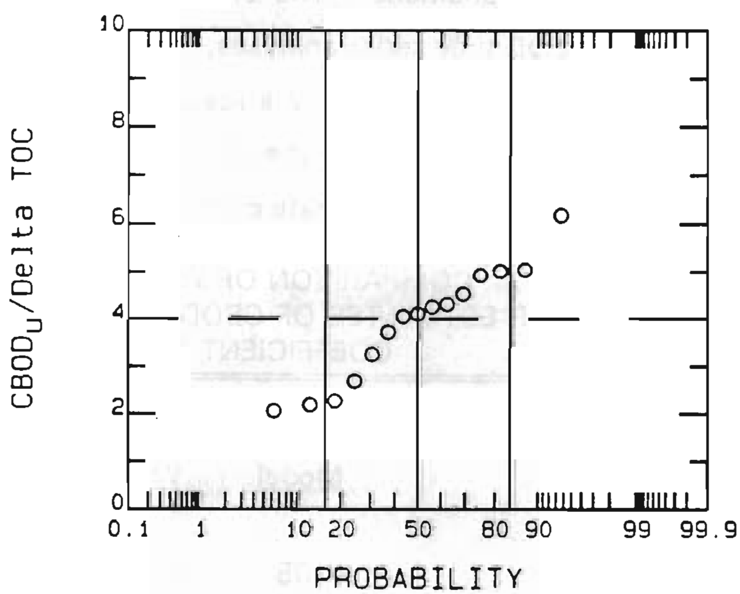


FIGURE 5-8. OXYGEN UTILIZATION REQUIREMENTS OF DEGRADABLE TOC IN WATER COLUMN

ratio of $CBOD_u$ to degradable TOC for these data is shown on the lower panel. Although considerable variability exists, probably as a result of taking differences of low concentrations and the difficulty of measuring such low level TOC's and BOD's, the data are approximately normally distributed with an average oxygen utilization requirement of 4 grams oxygen/gram of carbon utilized in the BOD bottle. This ratio is indicative of organic carbon in a relatively reduced state in comparison to glucose.

Similar results are available for data from July 31 and August 1, 1991. The TOC data at the start and end of the BOD time series for July 31 indicate a degradable TOC of about 1.0 mg/L. The August 1 data did not show any change in the t = 0 and t = 60 day TOC results, even though the $CBOD_u$ was about 3.5 mg/L (corresponding to a TOC of about 0.8 mg/L), suggesting an analytical problem. Using the July 31 TOC results and an average $CBOD_u$ of approximately 4 mg/L, the oxygen utilization requirement of the degradable TOC is again about 4 grams oxygen/gram carbon.

Warburg respirometer tests were also used to quantify oxidation rates in samples incubated under mixed conditions. The oxygen uptake curves were analyzed using procedures similar to BOD time series analyses, but unexplained lags in the oxygen uptake rate at the start of some tests, possibly a result of an imperfect seal, complicated the interpretation of the results. Hence, the analysis could not be considered definitive. However, as shown in Table 5-2, the rate coefficients determined on the basis of these

TABLE 5-2. COMPARISON OF WARBURG TEST RESULTS TO OTHER ESTIMATES OF CBOD DEOXYGENATION RATE COEFFICIENT (1/DAY)

	Model	Warburg	BOD Time Series
NPS: Upstream	.05	0.02-0.06	0.03 to 0.10
Downstream	.05		
GP Effluent	-	0.04-0.16	
GP Effluent in River	.10	0.08-0.11	

studies were generally consistent with rate coefficients estimated from both the BOD time series tests and from analysis of instream spatial profiles (to be reviewed in Section 5.2).

5.1.4 Sediment-Water Column Interactions

The second key objective of studies completed during 1991 was to obtain an improved definition of the magnitude of sediment-water column interactions, including both sediment oxygen demand (SOD) and the distributed $CBOD_u$ load that has been used in the water quality model of the Ouachita River. SOD has previously been shown to be an important component of the oxygen balance in the Ouachita River and hence additional data were warranted. The uniform load inferred from the water column CBOD data in previous analyses was also believed to be a significant factor in the oxygen balance of the river, but until the studies described herein were completed, it had not been directly measured. Assignment of a uniform load was originally motivated by the observation that upstream and far downstream (more than about 10 to 15 miles downstream of the discharge) CBOD concentration profiles in the Ouachita River were typically constant in space, other than variation associated with what appeared to be random sampling and measurement error (see calibration Figures 5-19 to 5-21, to be presented). This is a characteristic instream response for a uniformly distributed input of a non-conservative substance.

Previously, the approach has been to calculate the distributed load (W_{ul}) from the analytical expression relating it to the resulting spatial steady state concentration (L_0) of the degradable material of interest:

$$W_{ul} = A\ K_d\ L_0 \qquad (5\text{-}1)$$

Here, K_d is the oxidation rate of the material, A is the channel cross sectional area and L_0 is the average instream concentration ($CBOD_u$, in this case) at spatial steady state. For purposes of calculating W_{ul}, the oxidation rate coefficient of the background load was taken as a typical bottle rate for upstream BOD time series samples (0.05/day at 20°C), adjusted to temperature T using the following expression:

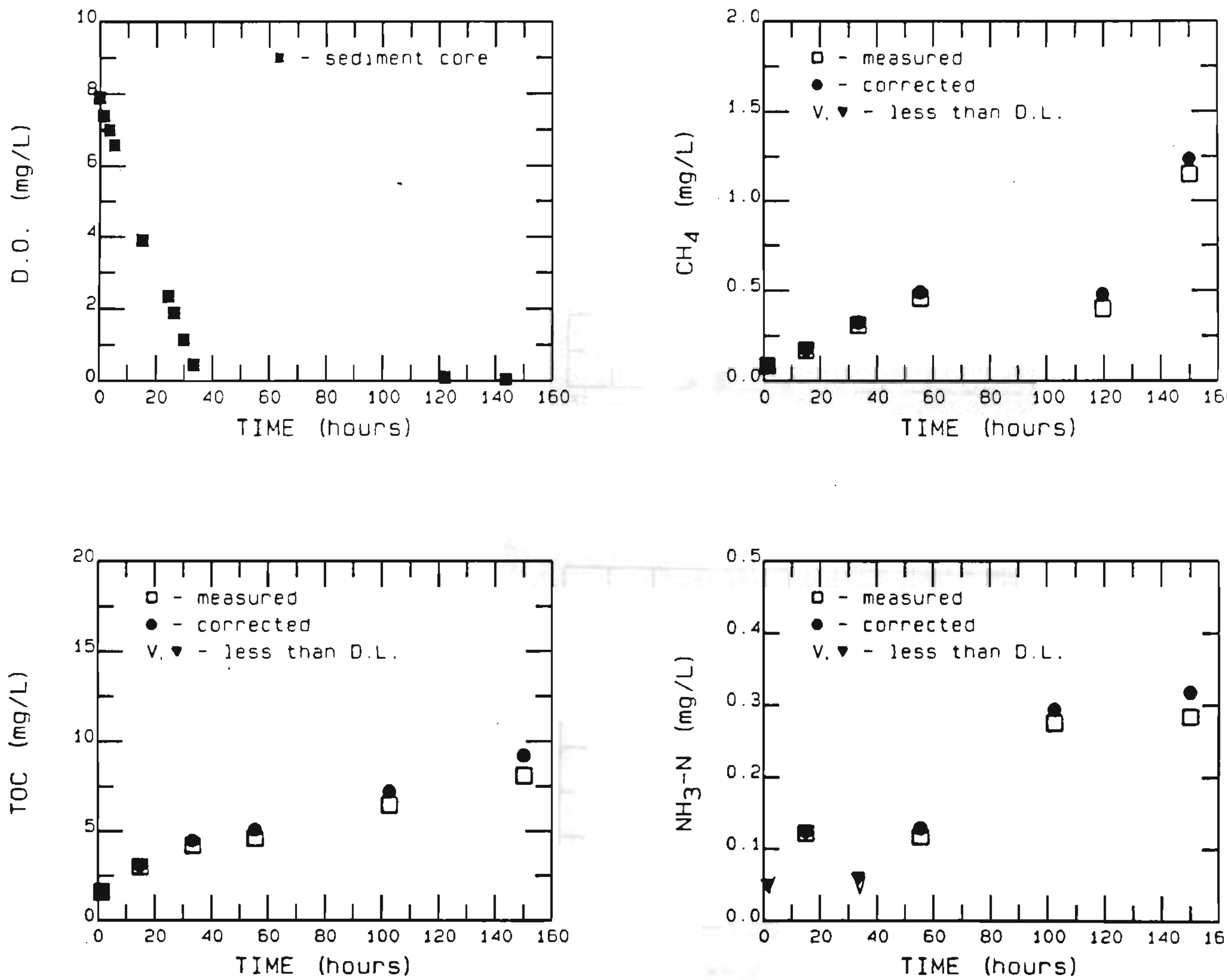
$$K_d(T) = K_d(20)\ \theta^{(T-20)}, \quad \theta = 1.047 \tag{5-2}$$

Since the distributed load had never been directly measured, it was decided to conduct studies to verify the presence of the sediment release and to confirm the order of magnitude of the distributed loads that have been used in the model. At the same time, these studies provided an opportunity to obtain additional measurements of SOD in the Ouachita River.

5.1.4.1 Description of Sediment Flux Studies

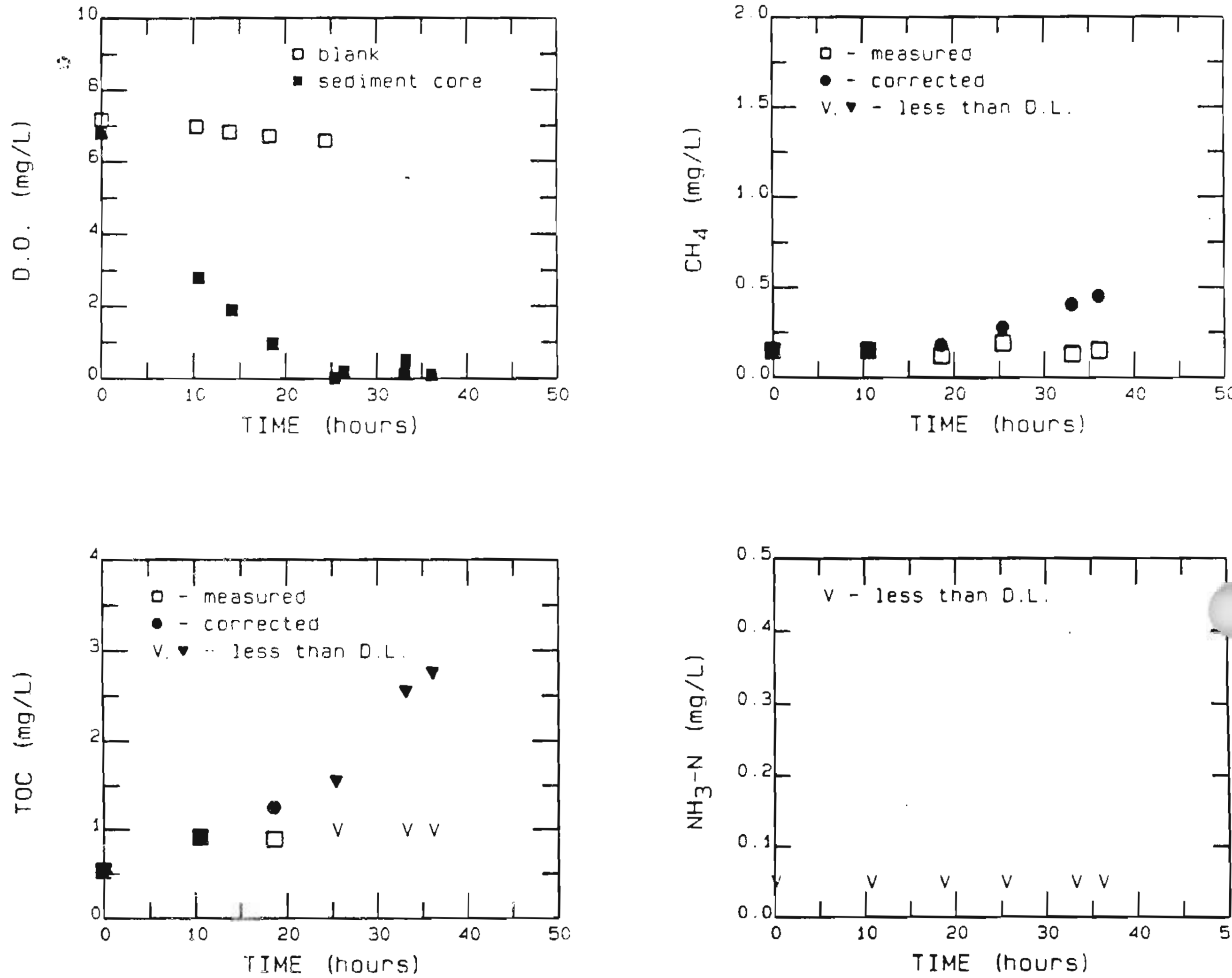
Three sets of Ouachita River core samples were collected during July and September of 1991 and used to obtain measurements of fluxes of a variety of sediment decomposition end products. A detailed description of the methods of sample collection and testing is presented in Appendix D2, along with the test data. Briefly, four inch diameter sediment cores with about four to six inches of overlying water were collected by a diver and returned to the laboratory for testing. The cores were connected to a flow through system and the overlying water displaced with aerated, distilled water for three or more volume replacements. The system was then sealed and operated as a closed loop recirculation system in order to maintain complete mix conditions in the overlying water. The supernatant was then sampled over time to monitor changes in concentration of various dissolved constituents during sequential aerobic and anaerobic periods. Care was taken not to disturb the sediment core itself during these procedures.

Typical results for the July and September sampling efforts, shown on Figures 5-9 and 5-10, generally include time series measurements of dissolved oxygen (for measuri



DATE: 7/22/91
MILEPOINT: 210
OVERLYING WATER DEPTH: 15.5 cm
SEDIMENT DEPTH: 14.5 cm

FIGURE 5-9. SEDIMENT FLUX STUDY RESULTS FOR MP 210
JULY 22, 1991



DATE: 9/23/91
MILEPOINT: 238.0
OVERLYING WATER DEPTH: 9.5 cm
SEDIMENT DEPTH: 20 cm

FIGURE 5-10. SEDIMENT FLUX STUDY RESULTS FOR MP 238
SEPTEMBER 23, 1991

SOD), methane, TOC and ammonia nitrogen. Other variables measured at different frequencies between the July and September data sets included temperature, organic nitrogen, nitrite + nitrate nitrogen, H_2S and sulfate. The gas ebullition rate was also monitored but not found to be significant. For dissolved methane, ammonia and TOC, the data are adjusted for the mass of material removed over time due to sampling and replacement with deionized water, as indicated by the open and closed plot symbols. This correction was not overly significant for the July samples, and hence a similar sampling approach was used when the September samples were processed. Because of the lower release rates observed in the September testing (note TOC scale change between Figures 5-9 and 5-10) the problem was accentuated. Fortunately, the required interpretation of these test results emphasizes the initial release rate during the aerobic period of incubation, when this correction is of lesser significance. Also, during this aerobic portion of the test, the TOC flux occurs in addition to the direct oxidation (SOD) occurring at the sediment-water interface. A detailed discussion of the results of the SOD and TOC flux studies follows.

5.1.4.2 Sediment Oxygen Demand Results

SOD results for the three sets of core samples are summarized in Table 5-3. The SOD's were evaluated from the initial rate of decrease of dissolved oxygen during the sediment flux tests to account for the effect of overlying water dissolved oxygen concentration on SOD. During this portion of the test the overlying water dissolved oxygen is generally in the range of 4 to 7 mg/L, conditions comparable to instream dissolved oxygen levels during low flow periods when the river is within its banks. The effect of overlying water dissolved oxygen is apparent in many of the samples analyzed, as the rate of dissolved oxygen decrease in the overlying water tends to be reduced as the dissolved oxygen is reduced (e.g., Figure 5-10).

Figure 5-11 compares the spatial profile of 1991 laboratory core SOD data to 1980 SOD results by HydroQual (laboratory results using surficial sediments collected with an Ekmann dredge) and 1980 in situ results (NCASI, 1982). Test conditions with regard to

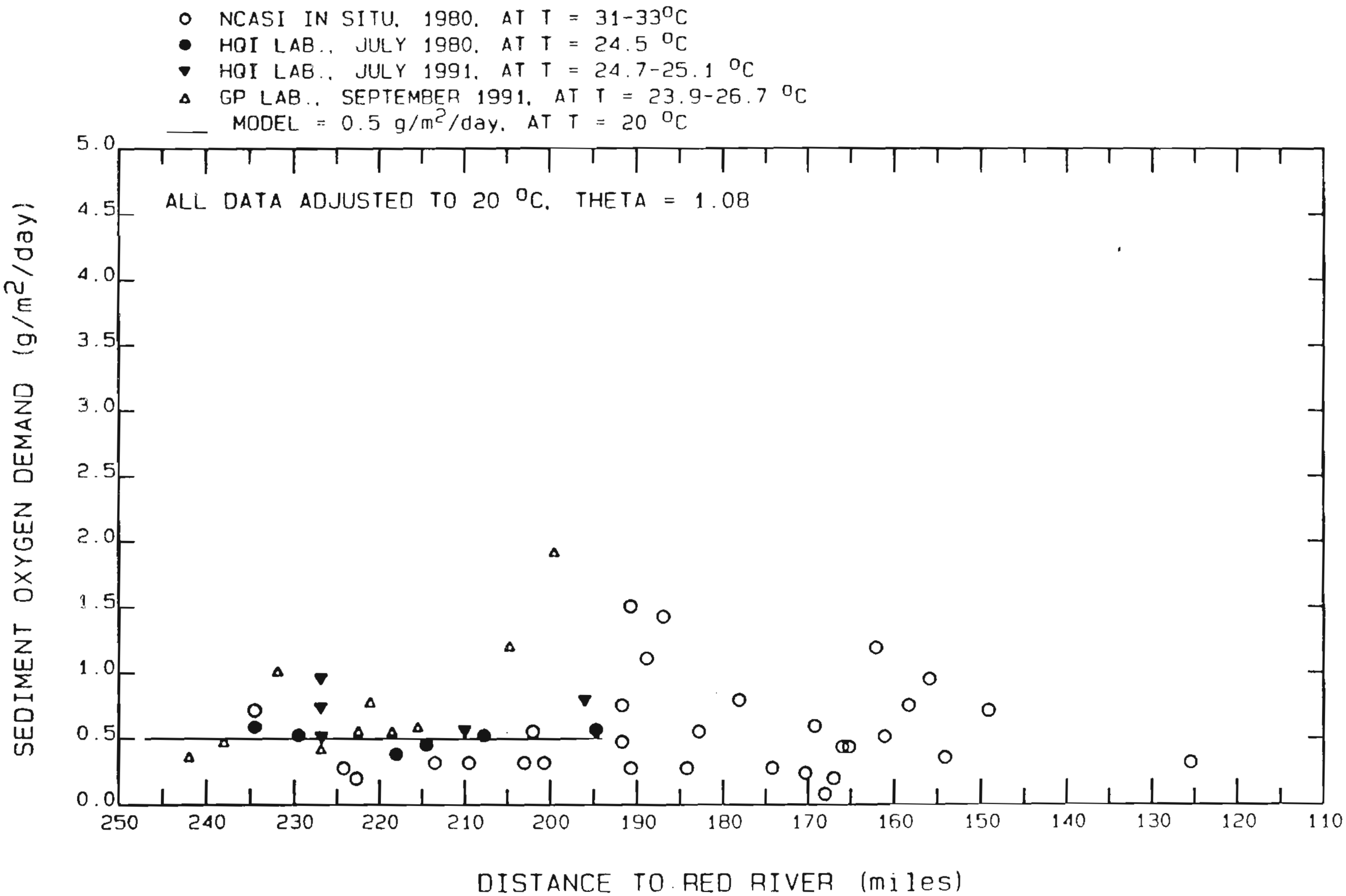


FIGURE 5-11. SPATIAL PROFILE OF AVAILABLE OUACHITA RIVER SOD DATA

temperature are indicated above the graph and data have been adjusted to 20°C, using $\theta = 1.08$, for comparison purposes. As shown, the NCASI results (open plot symbols) tend to be lowest in the region of overlap of the data sets, but downstream of the present study area, south of Bayou Bartholomew, SOD's reported by NCASI are similar to levels reported in the upstream reach by HydroQual. Each of the SOD data sets exhibit considerable variability in space, a result typically observed for SOD data, due to the spatial non-homogeneity of sediment deposits and the difficulty of making the measurements themselves. Generally, however, the results are in good agreement, given the significant differences of measurement techniques employed.

TABLE 5-3. SUMMARY OF 1991 SOD TEST RESULTS

Date	MP of Sample Location	Average Water Temperature (°C)	Sediment Oxygen Demand ($g/m^2/d$)
July 22, 1991	226.9 (West Side)	24.7	1.05
	226.9 (Mid-Channel)	25.1	1.42
	226.9 (East Side)	24.7	0.73
	210.0	24.9	0.82
	196.0	24.7	1.14
September 23, 1991	242.0	26.5	0.60
	238.0	26.7	0.80
	231.9	26.5	1.68
	226.9	26.4	0.70
	222.5	25.5	0.86
September 25, 1991	221.1	24.7	1.12
	218.5	24.7	0.80
	215.5	24.5	0.84
	203.7	24.4	1.70
	199.5	23.9	2.59

The initial evaluation of a spatially average SOD of 0.5 $gm/m^2/day$ at 20°C was based on the 1980 SOD results obtained by HydroQual (filled circles). Given the variability generally evident in the SOD measurements, the additional data collected in 1991 were judged to be consistent with the earlier data, particularly in the reach of the study area between Felsenthal Dam (MP 226.9) and Bayou Bartholomew (MP 194.5). The probability distribution of SOD data from this river reach is shown on Figure 5-12; the average SOD is 0.59 grams/m^2/day (N = 21) with a median SOD of 0.51 grams/m^2/day. Excluding the single unusually high measured value of 2 grams/m^2/day reduces the average SOD to 0.51 grams/m^2/day as well. In view of the preceding results, SOD in the model is set at 0.5 grams/square meter/day at 20°C and adjusted to other temperature conditions via:

$$SOD(T) = SOD(20)\ \theta^{(T-20)} \qquad (5\text{-}3)$$

with θ = 1.08.

5.1.4.3 TOC Flux Results

TOC was used as the primary measure of $CBOD_u$ released by the sediment, rather than relying directly on $CBOD_u$, due to the significantly lower sample volume requirements and relative ease of measurement. As noted previously, and as illustrated by comparison of the TOC concentration versus time plots on Figures 5-9 and 5-10, measured fluxes tended to be lower when the September testing was performed. (Note the change in scales for TOC on Figures 5-9 and 5-10). This same pattern was generally exhibited by the remainder of the core samples analyzed during the two sampling periods.

It is appropriate to first discuss the manner in which the data were used to estimate the sediment release rates. The TOC fluxes were determined from the slopes of the concentration time curves for each core. The July samples showed a consistent pattern of decreasing release rate over time, reflecting either stabilization of the labile sediment organic matter undergoing decomposition, or perhaps, reflecting a decrease in the concentration gradient between the sediment pore water and the overlying water as the overlying water concentration increased. With regard to the latter explanation, pore water

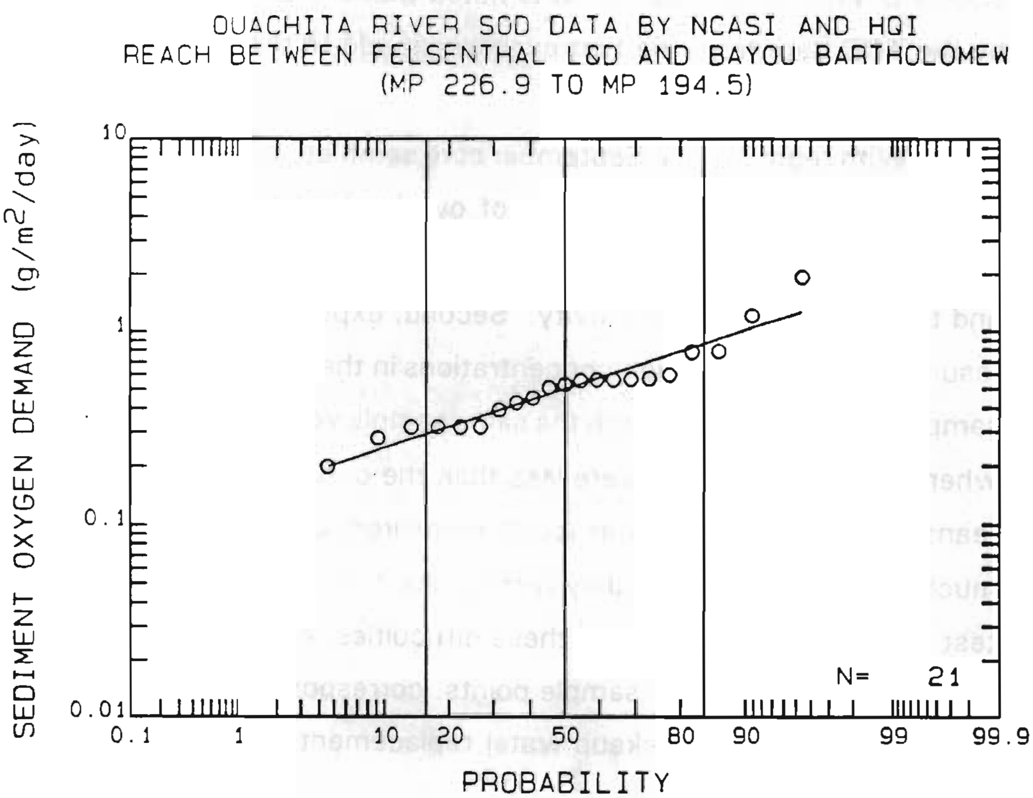


FIGURE 5-12. PROBABILITY DISTRIBUTION OF SOD MEASUREMENTS BETWEEN FELSENTHAL DAM AND BAYOU BARTHOLOMEW

TOC results from a single core (MP 199.5), sampled with a pressurized head space displacement squeezer (Jahnke, 1988), indicated a maximum pore water TOC of about 23 mg/L at a depth of 10 cm below the sediment water interface. Hence concentrations of 10 mg/L or more in the overlying water could affect the TOC flux to the overlying water. Given these considerations, the estimate of the release rate of TOC from the sediment was based on the initial slope for the July data set, when overlying water concentrations were generally in the range of 2 to 5 mg/L. This approach also precluded the need to rely on data corrected for makeup water replacement over the course of the test, a secondary source of error in the results. It is noted that the consistent time series trends indicated by the TOC data from the first experiment add to the credibility of the analytical results.

With regard to the September core samples, greater difficulty was encountered, for several reasons. First, the depth of overlying water in some of the core samples was deeper than desired (16 to 22 cm) in comparison to the July samples (about 11 to 16 cm) and this reduced test sensitivity. Second, expecting results comparable to the July test results, with relatively high concentrations in the overlying water, the overall frequency of sampling was increased, with the same sample volumes collected. This created difficulties when some TOC results were less than the original 1 mg/L detection limit and had to be reanalyzed. Given the lower levels measured, concentration differences over time were much smaller than for the July test results and sample to sample differences bordered on test precision. As a result of these difficulties, estimates of fluxes were typically based on the first three or four sample points, corresponding to the aerobic portion of the test, when corrections for makeup water replacement were generally not significant.

Table 5-4 presents a summary of the measured TOC fluxes estimated as described above. The spatial profile of these sediment TOC fluxes is shown on Figure 5-13. The results correspond to temperatures at laboratory test conditions of about 25 $\pm 2^{o}$C. As shown, the results are somewhat variable, both for a given sampling period, and between periods. The average TOC flux for July was 0.27 grams C/m^2/day and during September it was about 50 percent of this, at 0.16 grams C/m^2/day. Although use of data sets from only two points in time is not considered sufficient to define a longer term seasonal trend,

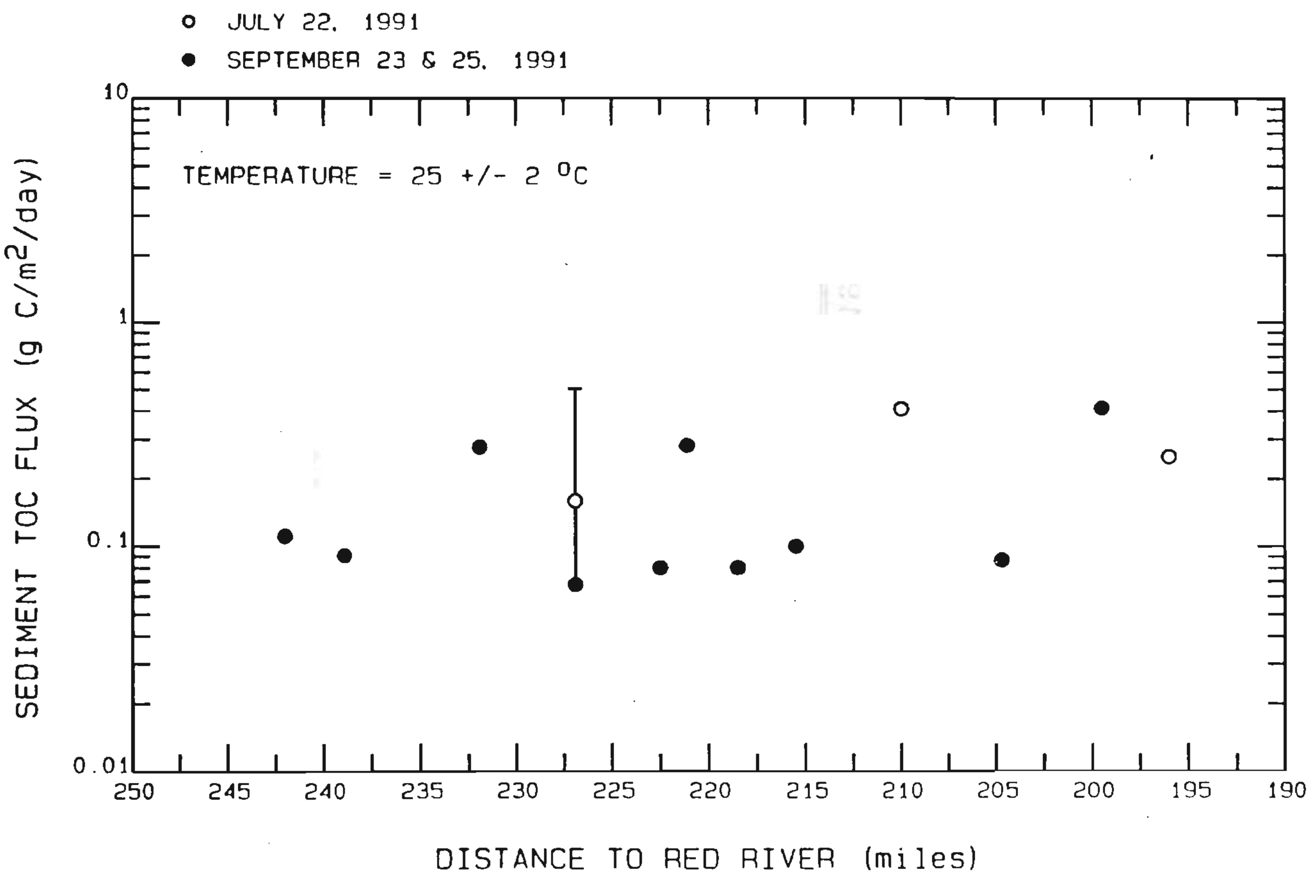


FIGURE 5-13. SPATIAL PROFILE OF OUACHITA RIVER TOC FLUX MEASUREMENTS

this decrease in TOC fluxes is consistent with the observation, discussed subsequently, that upstream TOC and CBOD concentrations generally decrease with increasing time after flooding (Figure 5-15). From this observation, it can be inferred that the uniformly distributed load, i.e., the sediment release rate of TOC, probably decreases over time as well.

It should be noted that the ammonia fluxes measured during these experiments were generally quite low. Prior studies of this type have confirmed (SWRPC, 1987) that sediments decompose in accordance with a carbon to nitrogen ratio of about 5.5 (Redfield, 1963). As a check on the test results, therefore, the total carbon and nitrogen fluxes were

TABLE 5-4. SUMMARY OF 1991 TOC FLUX MEASUREMENTS

Date	MP of Sample Location	Average Water Temperature (°C)	Sediment TOC Flux (g C/m²/d)
July 22, 1991	226.9(West Side)	24.7	0.067
	226.9(Mid-Channel)	25.1	0.509
	226.9(East Side)	24.7	0.120
	210.0	24.9	0.412
	196.0	24.7	0.251
		July Average:	0.272
September 23, 1991	242.0	25.8	0.111
	238.0	26.5	0.091
	231.9	25.9	0.277
	226.9	25.4	0.068
	222.5	24.3	0.081
September 25, 1991	221.1	24.4	0.282
	218.5	25.0	0.081
	215.5	24.3	0.100
	204.7	24.2	0.087
	199.5	23.9	0.414
		September Average:	0.160

computed for comparison to the Redfield ratio. (This check could only be performed for the July test results because of the sampling and analytical difficulties discussed above for the September tests.) For the July core samples the carbon flux includes the organic carbon flux from the sediment and carbon associated with dissolved and gaseous methane (CH_4-C, times 2, to account for 1 mole of CO_2 produced for each mole of methane produced). The nitrogen flux primarily consists of dissolved organic and ammonia nitrogen fluxes and N_2 gas. For these tests, gas production was not significant and dissolved CH_4-C fluxes, as observed during the anaerobic period, were generally less than 10 percent of the TOC flux. The organic nitrogen was found to be the primary component of the nitrogen flux, and since it was only measured at the end of the anaerobic period, it was necessary to base the computation on the cumulative release from the sediment over the duration of the test. Also, the correction for makeup water replacement was neglected, since the effect of makeup water was generally to dilute the overlying water, and would affect both carbon and nitrogen in a similar way.

The results of the Redfield ratio comparison for the July sediment core flux experiment are shown on Figure 5-14, along with test results from similar types of decomposition studies performed using Milwaukee River, Milwaukee Harbor and near shore Lake Michigan sediments. As shown, the results for the Ouachita River sediments (filled circle plot symbol) generally fall along the line corresponding to the Redfield ratio of 5.5. The fluxes measured as part of this study are consistently at the low end of the test results shown, reflecting the relatively stable nature of sediments in the Ouachita River, in comparison to the Milwaukee Harbor sediments (samples 1, 2, a-e and A-E), which are heavily impacted by combined sewer overflow (cso) solids. The fluxes from the Ouachita River sediments are comparable in magnitude to the fluxes measured at an upstream background station in the Milwaukee River (samples f and F) and with a near shore Lake Michigan sediment (samples g and G). The good agreement of these results with the Redfield ratio indicates that the relative magnitudes of the measured fluxes is reasonable and adds additional support to the validity of the test procedures.

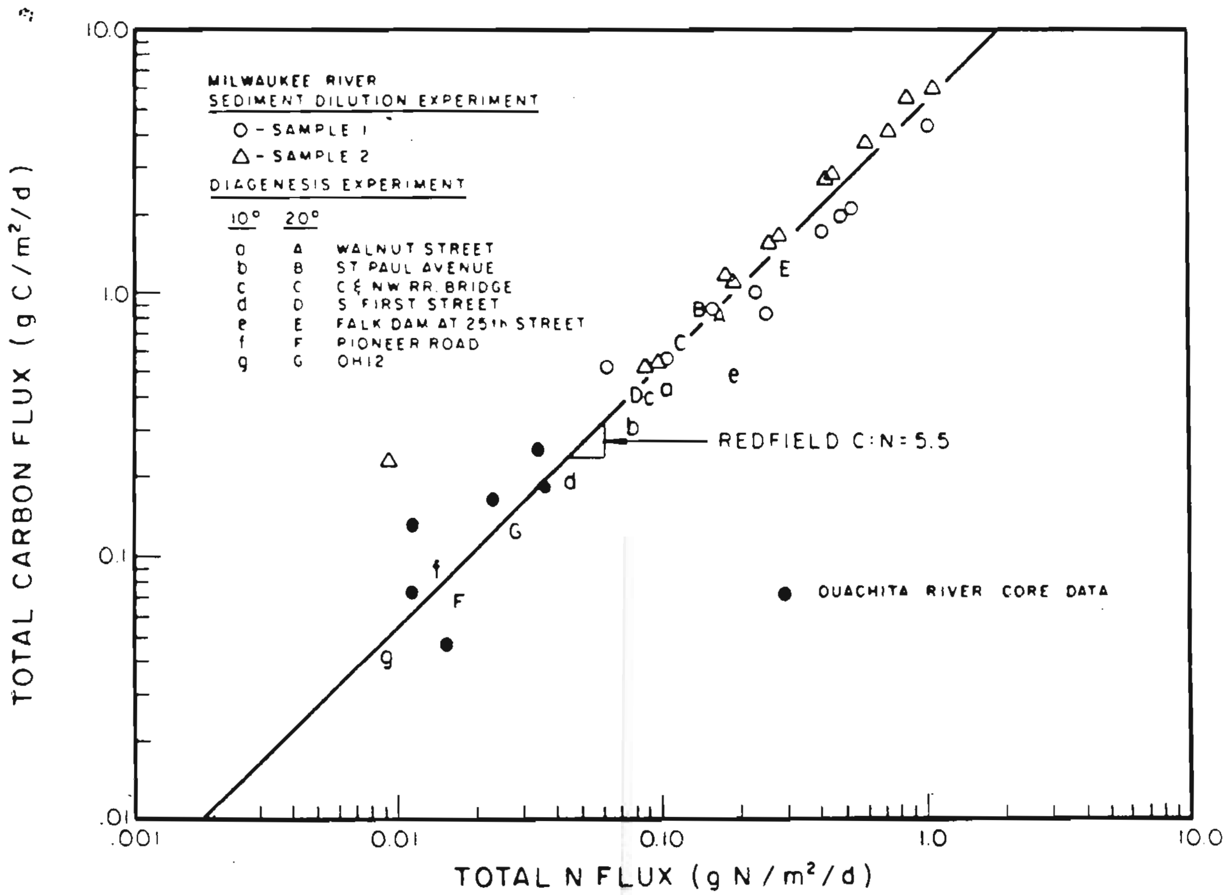
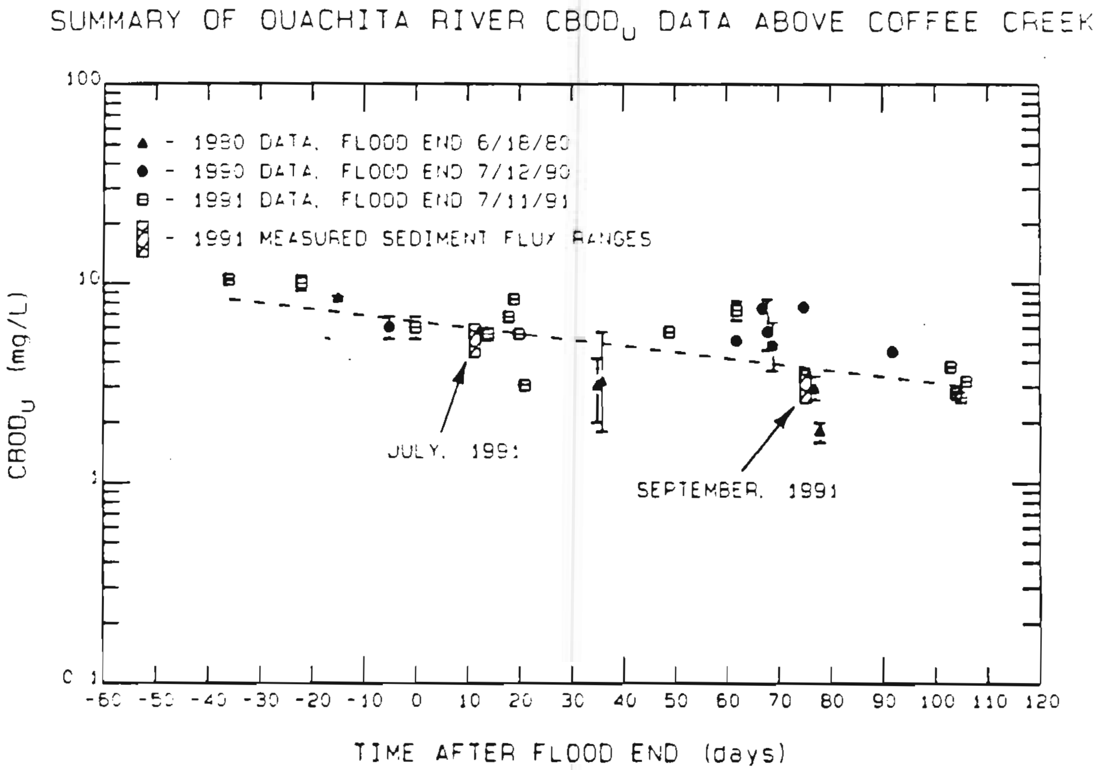


FIGURE 5-14. COMPARISON OF CARBON AND NITROGEN FLUXES FOR JULY SEDIMENT CORES TO MILWAUKEE HARBOR SEDIMENT FLUXES AND THE REDFIELD C:N RATIO

The primary objective in conducting the sediment flux tests was to measure the aerobic fluxes from the sediment, as water column dissolved oxygen levels in the Ouachita River are generally in the range of 4 to 7 mg/L during low stage periods in the summer. It is considered that this primary objective was achieved. In addition, the anaerobic fluxes were measured to gain an improved understanding of the sediment processes taking place, including during high stage conditions, when dissolved oxygen levels are much lower for extended periods of time. Prior studies have indicated that when the dissolved oxygen in the overlying water reaches zero, the SOD can no longer be exerted, and dissolved constituents (e.g., methane and ammonia) previously oxidized at the sediment-water interface begin to diffuse to the overlying water at an increased rate. Generally, for the July tests, the methane flux measured during the anaerobic phase, although increasing somewhat relative to the aerobic period, was not high enough to account for the SOD. At the same time, the ammonia fluxes were much lower than expected as well. A labile fraction of the pore water TOC might also have been expected to contribute to the SOD, but the absence of an increase in the TOC flux during the anaerobic period indicates that this was not the case. These issues, though not adequately resolved to date, concern areas of modeling which are very much research related. It is hoped that over time, as more is learned about sediment decomposition processes, that a more complete understanding will be achieved.

5.1.4.4 Relationship of TOC Fluxes to Distributed CBOD Load

As indicated previously, it has been observed that the long term seasonal trend in the Ouachita River is for the upstream $CBOD_u$ concentration to decrease with time after flooding. The data in support of this observation are shown on the upper panel of Figure 5-15. In addition, the lower panel shows that the available TOC data follow a similar and more consistent trend. If the water column TOC and CBOD originate from a distributed release from the sediment, it can be inferred from these data that the distributed load must be decreasing over time as well. Although the exact manner in which this takes place is not well known, the following scenario is hypothesized. Recession of the flood plain waters would be expected to result in delivery of a particulate organic load (e.g., forest



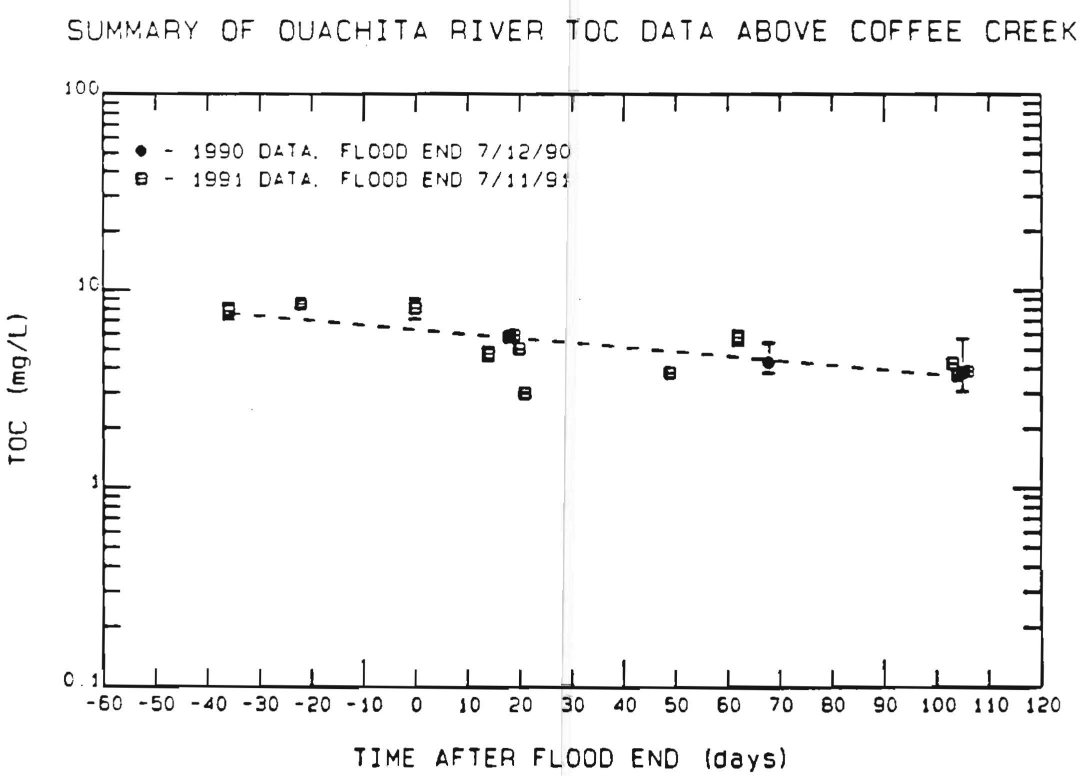


FIGURE 5-15. VARIATION OF UPSTREAM CBOD$_u$ AND TOC CONCENTRATIONS WITH TIME AFTER FLOODING

litter) to the main channel, and via settling, to the bottom sediments. (This phenomenon has been reported to occur in a number of studies where the influence of seasonal flooding of large rivers has been reported; see, e.g., Cuffney, 1984 or Hedges et al., 1986.) As this particulate organic material decomposes in the bottom sediments, soluble organic material is released to the pore water and then diffuses to the overlying water. Hence, the release is expected to be highest at about the time the river stage decreases to bank full level, when most of the particulate load enters the river, and decrease thereafter. This scenario is consistent with the decreasing trend observed over time in the Ouachita River.

Further support for the preceding hypothesis is given by the empirical fit of the $CBOD_u$ data with time, which indicates a rate of decrease of 0.7 percent/day. This rate of decrease is consistent with stabilization rates of 0.2 to 0.4 percent/day reported for forest litter (Cuffney, 1984) and of 0.5 to 5 percent/day (10°C and 25°C, respectively) that is estimated from decomposition rate data for aquatic macrophyte litter incubated under anaerobic conditions (Godshalk and Wetzel, 1976). The TOC data shown on the lower panel also conform to a similar rate of decrease, although in this case the apparent rate of decrease over time appears lower due to the non-degradable fraction of TOC present in the water column. (Approximately one-third of the water column TOC was found to be degradable.)

The remaining step in this evaluation of sediment release rates is to compare the magnitudes of the measured TOC fluxes to the uniform load of CBOD used in the model. An example calculation showing the basis for this comparison is presented in Table 5-5 for reference purposes. Several points should be emphasized concerning the calculation. First, step 1 involves two key assumptions. One is that the oxygen utilization requirement of the TOC flux is 4 grams oxygen/gram carbon, as indicated by the change in TOC levels in the CBOD time series tests (Section 5.1.3.2). Also, it is assumed that essentially all of the TOC flux from the sediment is biodegradable. This latter assumption is consistent with the observation that the non-degradable water column TOC profile does not increase in the downstream direction, in either the upstream or downstream reach (Figure 5-7). Additional support for this assumption is provided by the decomposition studies of Godshalk and Wetzel (1976). These investigators conducted aerobic and anaerobic decomposition studies of macrophytic plant material which indicated that 90 percent or more of the anaerobically produced DOC released by decomposing macrophytes is oxidized in the water column under aerobic conditions. Assuming similar conditions apply for the Ouachita River sediments, i.e., that the sediment flux of TOC to the water column is essentially biodegradable under aerobic conditions, the TOC flux can be converted to a $CBOD_u$ release rate. To the degree that the material released by the sediments is not biodegradable, the calculations shown in Table 5-5 should be reduced by a similar amount.

TABLE 5-5. EXAMPLE CALCULATION TO CONVERT TOC FLUX TO A UNIFORM $CBOD_u$ LOAD AND WATER COLUMN $CBOD_u$ CONCENTRATION

1. Convert September 1992 TOC flux (measured at $T = 25°C$) to oxygen equivalents (i.e., units of $CBOD_u$)

$$CBOD_u \text{ Flux} = (0.16 \text{ g C/m}^2\text{/day}) \times (4 \text{ grams } O_2\text{/g C})$$
$$= 0.64 \text{ g } O_2\text{/m}^2\text{/day at } T = 25°C$$

2. Adjust flux to 30°C for comparison to high temperature survey data sets

$$CBOD_u \text{ Flux } (T=30) = CBOD_u \text{ Flux } (T=25) \times 1.120^{(30-25)}$$
$$= 0.64 \times 1.762$$
$$= 1.13 \text{ g } O_2\text{/m}^2\text{/day}$$

3. Convert areal flux to lbs/mile/day

$$W_{ul} = 1.13 \text{ g } O_2\text{/m}^2\text{/day} \times \text{Width (ft)}/.925$$
$$= 1.13 \times <336 \text{ to } 493 \text{ ft}> / .925$$
$$= 410 \text{ to } 602 \text{ lbs } CBOD_u\text{/mile/day}$$

Note: 0.925 factor converts to consistent units.

4. Convert to expected resulting $CBOD_u$ concentration (L_0) (using Equations 5-1 and 5-2)

$$L_0 = W_{ul}/[.3293 \text{ A (ft}^2) \text{ } K_d \text{ (1/d)}]; \; K_d = 0.05 \times 1.047^{(30-20)} = 0.079/d$$
$$= <410 \text{ to } 602>/[.3293 \times 6300 \times 0.079]$$
$$= 2.50 \text{ to } 3.67 \text{ mg/L}$$

The corresponding values for the July core data are as follows:

1. TOC Flux ($T = 25$) = 0.272 g C/m^2/day = 1.09 g O_2/m^2/day
2. $CBOD_u$ Flux ($T=30$) = 1.92 g O_2/m^2/day
3. W_{ul} = 697 to 1,023 lbs/mi/day
4. L_0 = 4.24 to 6.25 mg/L

Another point to take note of concerning the calculations of Table 5-5 is that the fluxes are adjusted from 25°C to 30°C to provide a more direct comparison with most of the survey data sets, which represent comparable instream conditions. The temperature correction calculation uses θ = 1.120 for sediment decomposition processes. This value is based on analyses of in situ sediment decomposition studies (θ = 1.123 with a standard error of 0.023) performed by HydroQual as part of a modeling study of Milwaukee Harbor (HydroQual, 1984; SWRPC, 1985) and decomposition rate studies of calcium magnesium acetate in soils (theta = 1.117 with a standard error of 0.016; Connolly et al., 1990, HydroQual, 1990 and McFarland and O'Reilly, 1992). The results of the latter study are shown on Figure 5-16. These studies were performed using a recently developed roadway deicer for which concern has been raised about potential impacts on dissolved oxygen when snowmelt runoff enters a receiving water. Of particular relevance here is that the organic constituent for which θ was evaluated was acetic acid, a likely end product of particulate material undergoing anaerobic decomposition.

As shown in Table 5-5, the calculation based on the measured TOC fluxes corresponds to $CBOD_u$ concentrations of about 4.2 to 6.2 mg/L in July and 2.5 to 3.7 mg/L in September. These concentration ranges are compared to the measured upstream concentration data on Figure 5-15 and are seen to be in reasonably good agreement. Based on the calculations in Table 5-5, the measured fluxes (adjusted to 30°C correspond to uniform loads of 697 to 1,023 lbs/mile/day on July 22 and 410 to 602 lbs/mile/day on September 23-25, 1990). As will be shown, these distributed loads are comparable to the uniform loads of 354, 422 and 812 lbs/mile/day used in the three high temperature surveys (29.5 to 31.0°C) analyzed with the model.

5.2 DISSOLVED OXYGEN CALIBRATION/VERIFICATION

The preceding section described the evaluation of model input parameters that was performed independently from the model. This section reviews the nitrification and BOD-DO model calibration results. These analyses are performed to provide a means of

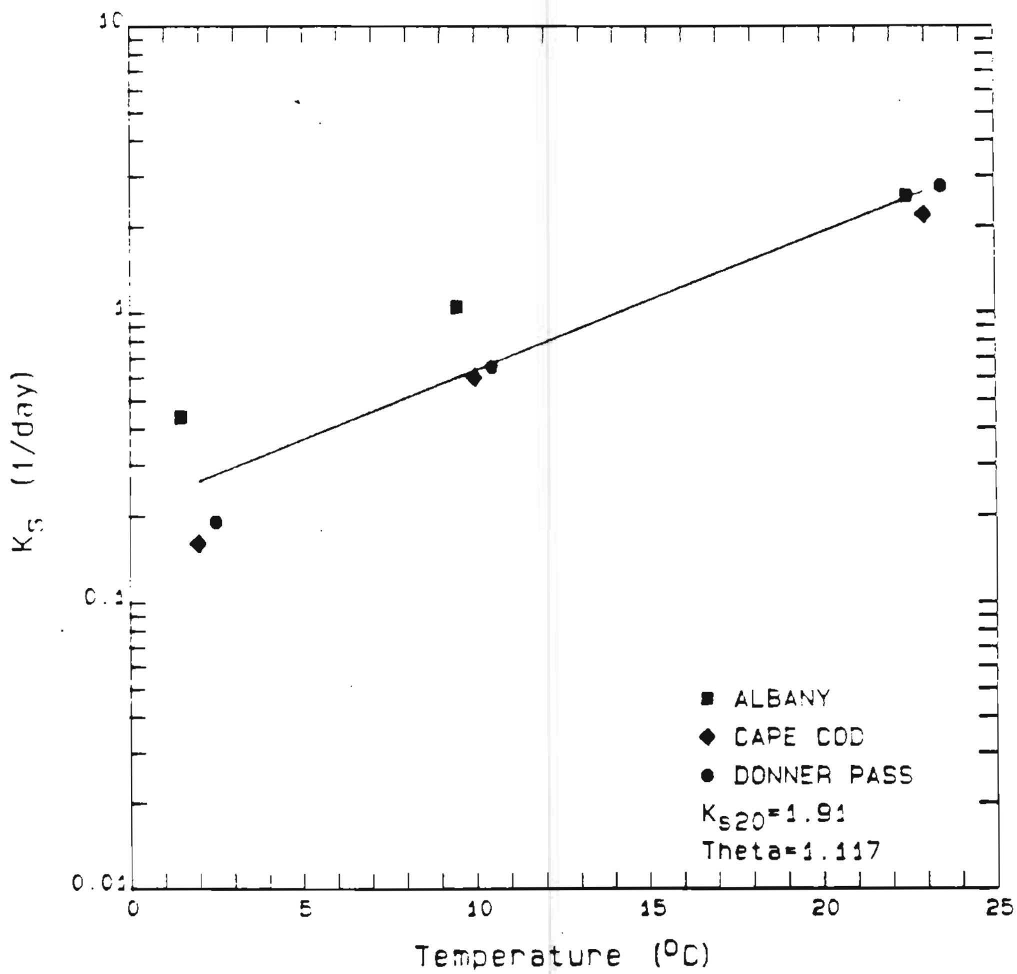


FIGURE 5-16. DECOMPOSITION RATE VERSUS TEMPERATURE OF THE ROADWAY DEICER CALCIUM MAGNESIUM ACETATE (CMA) IN SOIL

evaluating additional model input parameters and to further test the validity of the parameters previously evaluated for use.

5.2.1 Nitrification Analysis

Nitrification in the Ouachita River was evaluated on the basis of modeling analyses of two intensive survey data sets. These were the only two data sets reviewed which appear to have adequate data for evaluation of the instream nitrification rate. The modeling analyses are summarized on Figures 5-17 and 5-18. For the July 1980 survey shown on Figure 5-17, the TKN data (upper left panel) indicate a decrease in concentration in the downstream direction from Georgia Pacific's discharge at milepoint 222, relative to the concentrations computed if nitrogen were a conservative substance (the solid line). Note however, that the corresponding calculated nitrate-nitrogen increase associated with a nitrification rate of 0.1/day (the dashed line) is inconsistent with the observed nitrate data, while the profile corresponding to the zero nitrification rate, the solid line, is much more consistent with the observed nitrate profile. This inconsistency between model and data for TKN can be explained in several ways. First, with the exception of the concentration at the furthest downstream location the spatial decrease downstream of the discharge during the July 1980 survey is generally small, and hence the decrease may possibly be associated with analytical error. Alternatively, the variation of downstream concentrations might also be due to variation of stream flow and effluent nitrogen loading, of which there was limited data over time.

When the analysis is repeated on the basis of ammonia nitrogen, it is seen that the ammonia nitrogen data appear conservative, that is, the data indicate that the nitrification rate is zero, and when a non-zero nitrification rate is used the calculated decrease in the ammonia profile in the downstream direction is inconsistent with the data. Similarly, the nitrate data are also best represented by a zero nitrification rate.

The preceding comments generally apply to the September 1990 survey data set as well (Figure 5-18). As described in detail in Section 4, this survey was conducted

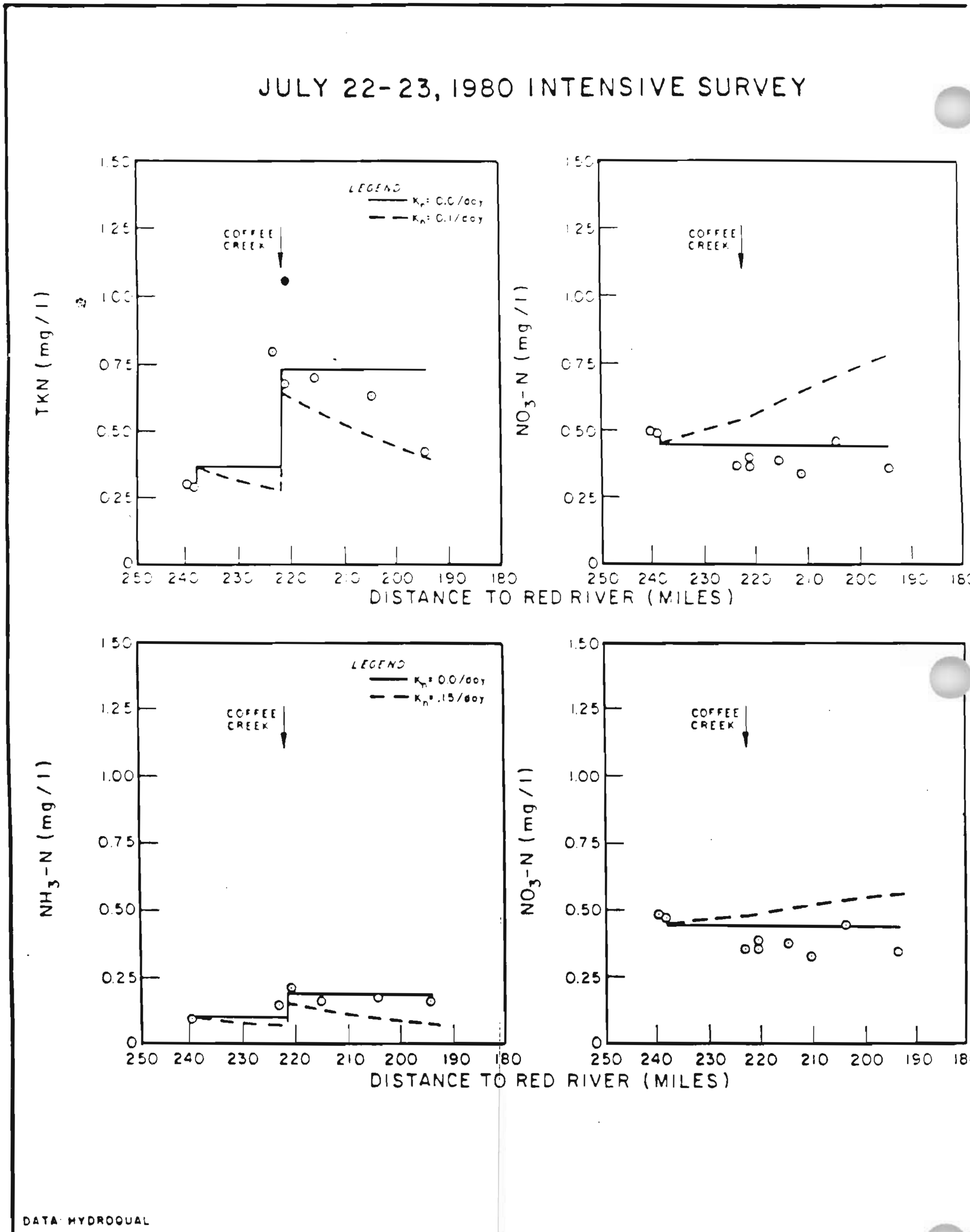


FIGURE 5-17. NITRIFICATION RATE ANALYSIS: JULY 1980 SURVEY

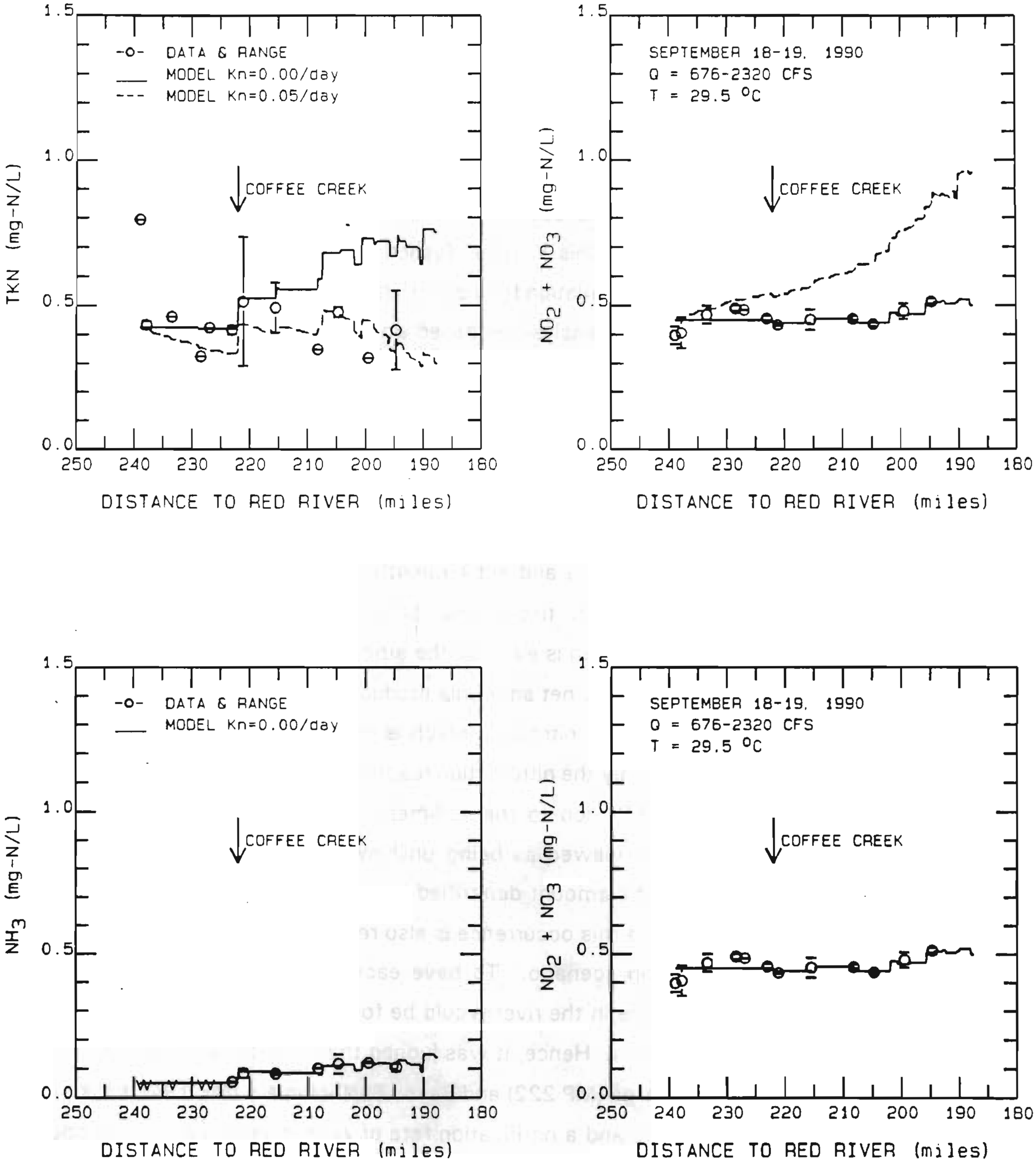


FIGURE 5-18. NITRIFICATION RATE ANALYSIS: SEPTEMBER 1990 SURVEY

during variable river flow conditions and as a result, the calculated profile for a conservative substance, for example the solid line for TKN, increases in the downstream direction. This occurs because water at the downstream end of the reach being evaluated flowed by the discharge during a relatively low flow period, when the available dilution was minimum. Assignment of a nitrification rate of 0.05 per day provides a better fit of the TKN data than a zero rate coefficient, but again, the nitrate-nitrogen profile which results from assignment of this K_n (the dashed line) is inconsistent with the data, in comparison to the model calculation for a conservative substance (i.e., $K_n = 0.0$, the solid line). As before, when the analysis is based on ammonia nitrogen being converted to nitrate-nitrogen, the calculation performed with a zero nitrification rate provides excellent fits of both ammonia and nitrate-nitrogen.

The preceding scenarios which assume nitrification takes place require that several concurrent reactions are occuring in order to be acceptable. First, only the analyses based on TKN converted to ammonia and subsequently to nitrate provide any indication that nitrification is occurring, but, this interpretation requires that the organic nitrogen converted to ammonia nitrogen is equal to the amount of ammonia nitrogen converted to nitrate nitrogen, such that the net ammonia production is zero. Also, the analyses result in a net production of nitrate nitrogen, which is not observed. The possibility that the nitrate-nitrogen produced by the nitrification reaction in the water column is removed from the water column via diffusion to the sediment, where it undergoes denitrification is feasible. However, it is viewed as being unlikely that the amount of nitrate-nitrogen produced would equal the amount denitrified in the sediment, such that there is no net production of nitrate, and this occurrence is also required to explain the observed nitrate data under a nitrification scenario. To have each of these situations occurring while nitrification is taking place in the river would be fortuitous, especially over the course of two independent data sets. Hence, it was judged that nitrification in the reach between Georgia-Pacific's discharge (MP 222) and Bayou Bartholomew (MP 194) is taking place at a negligible rate, if at all, and a nitrification rate of zero is assigned in the model.

The analyses just described provide only indirect evidence that nitrification is not occurring in the further downstream reaches of the Ouachita River. It is possible that nitrification is occurring downstream of Sterlington, but additional field studies and/or analyses of other data sets from the lower river should be completed prior to making this assumption.

Additional sensitivity analyses are described in Section 5.3.4 to assess the effect of nitrification on the dissolved oxygen balance in the Ouachita River.

5.2.2 CBOD - Dissolved Oxygen Analysis

The results discussed previously were incorporated in the analysis of CBOD and dissolved oxygen. Geometry and time of travel relationships based on the HEC II model were utilized with the flow distributions determined from the analysis of conservative substances (Section 4). Table 5-6 summarizes values of model parameters assigned in the model calibration. These parameter values were generally assigned based on the analyses described previously in this section.

TABLE 5-6. SUMMARY OF MODEL PARAMETERS

CBOD Oxidation Rate (K_d), Background	0.05/day
Effluent	0.10/day
NBOD oxidation Rate (K_n)	0.0/day
SOD (T = 20°C)	0.5 g/m^2/day
Uniform CBOD Load	(a)
Reaeration Rate	(b)
Correction for Temperature Effects:	
$K_T = K_{20}\theta^{T-20}$; θ =	
Reaeration	1.024
SOD	1.080
BOD Oxidation	1.047
Anaerobic Decomposition	1.120

[a]Variable (See Table 5-7)
[b]O'Connor Equation $K_a = 13\ u^{1/2})/(H^{3/2})$
Minimum $K_a = 2/H$ (Figure 5-2)

Flow, temperature and uniform loads assigned for the five data sets used in the model calibration for low stage conditions are summarized in Table 5-7. The order shown follows the order of presentation, which is generally in order of decreasing river flow. As shown, the survey flows vary from a high flow survey of about 3,000 cfs to a low flow survey of about 800 cfs, and finally the variable flow survey with flows ranging from 676 cfs to 2,320 cfs. (The annual 7Q10 is 780 cfs.) Each calibration data set uses a different uniform load. This approach is consistent with the convention of using survey specific measured upstream CBOD concentrations when modeling an intensive survey data set. When model projections are made, as described in Section 7, it will be necessary to specify the upstream CBOD concentration and in this instance, it will also be necessary to specify a uniform load. The general approach to be followed will be discussed in Section 7.

Model calibration results for the four intensive surveys and one set of time averaged routine monitoring weekly survey data (once per week for five weeks) for low stage conditions are presented on Figures 5-19 through 5-24 and will be discussed in this section. As noted previously, the oxidation rate of the background load is 0.05/day at 20°C, while the oxidation rate of the mill load is set at 0.10/day at 20°C. The latter rate was set to achieve what was considered to be the best overall fit of the spatial profiles of $CBOD_u$ in the river (upper panel of Figures 5-19 through 5-21 and Figure 5-24), with consideration also given to results of the dissolved oxygen balance as well (shown on the lower panels). The overall BOD removal rate (K_r) was set equal to the BOD deoxygenation rate (K_d), and hence no settling of BOD is incorporated in the model. This is consistent with the fact that the effluent passes through a lagoon having a 10 day detention time under average effluent flow conditions and then through Mossy Lake which has a 20 day detention time. It is also a conservative approach, since removal of BOD from the water column by settling would reduce the oxygen demand associated with oxidation of BOD in the water column.

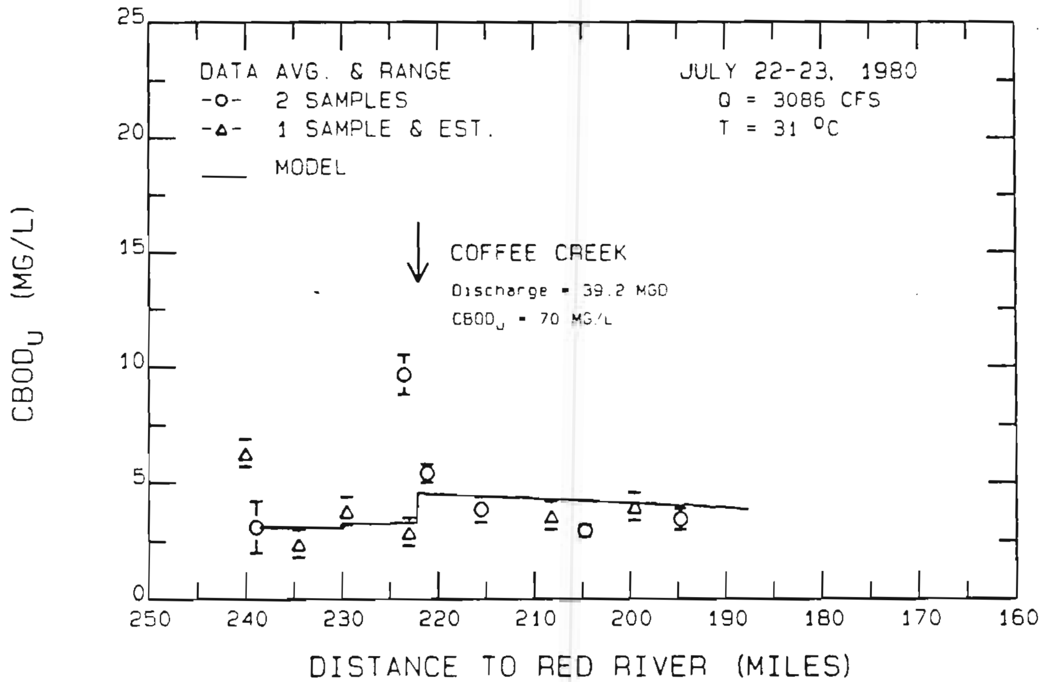
TABLE 5-7. SUMMARY OF FLOW, TEMPERATURE AND UNIFORM $CBOD_u$ LOADS FOR MODEL CALIBRATION DATA SETS

Survey Dates	Upstream Flow (cfs)	Temperature (°C)	Upstream $CBOD_u$ (L_0) (mg/L)	Uniform $CBOD_u$ Load[a] (lb/mi-d)	Equivalent Sediment Flux[b] (g $CBOD_u$ m^2/d)
07/22/80-07/23/80	3,086	31.0	3.0	422	0.78-1.15
10/23/91-10/24/91	1,946	20.0	2.5/2.83[c]	294	0.55-0.80
09/02/80-09/03/80	1,012	30.5	2.5	354	0.66-0.97
7/87	892	29.8	2.8[d]	384[e]	0.72-1.05[e]
09/18/90-09/19/90	676 -2,320	29.5	4.9	812[f]	1.51-2.21[f]

[a]$W_{ul} = .3292\ A\ K_r L_o$; $K_r = 0.05 \times 1.047^{(T-20)}$
[b]higher value upstream (A varies spatially)
[c]boundary condition/average; used average for W_{ul}
[d]no $CBOD_u$ data, used average of 1980 surveys
[e]estimated at old A = 5,312 ft^2 (455 lb/mi/d at new A = 6,300 ft^2)
c [f]used K = 0.08 (≃ 0.077; roundoff)

5.2.2.1 July 22 and 23, 1980 Intensive Survey
(Q = 3086 cfs and T = 31°C)

This was the first low flow survey used to calibrate the BOD-DO model when analyses were first performed during 1980. Initial attempts at calibrating the model for carbonaceous BOD were based on a single BOD removal rate coefficient throughout the study area. Due to the relatively uniform spatial profile of CBOD upstream of Coffee Creek, it was not possible to obtain good agreement between model and data using a single deoxygenation rate coefficient. The uniform spatial distribution of CBOD shown on Figure 5-19, both in the upstream and far downstream regions, suggested that the CBOD removal rate was very low, and/or that a uniform load was present. Thus, a minimum



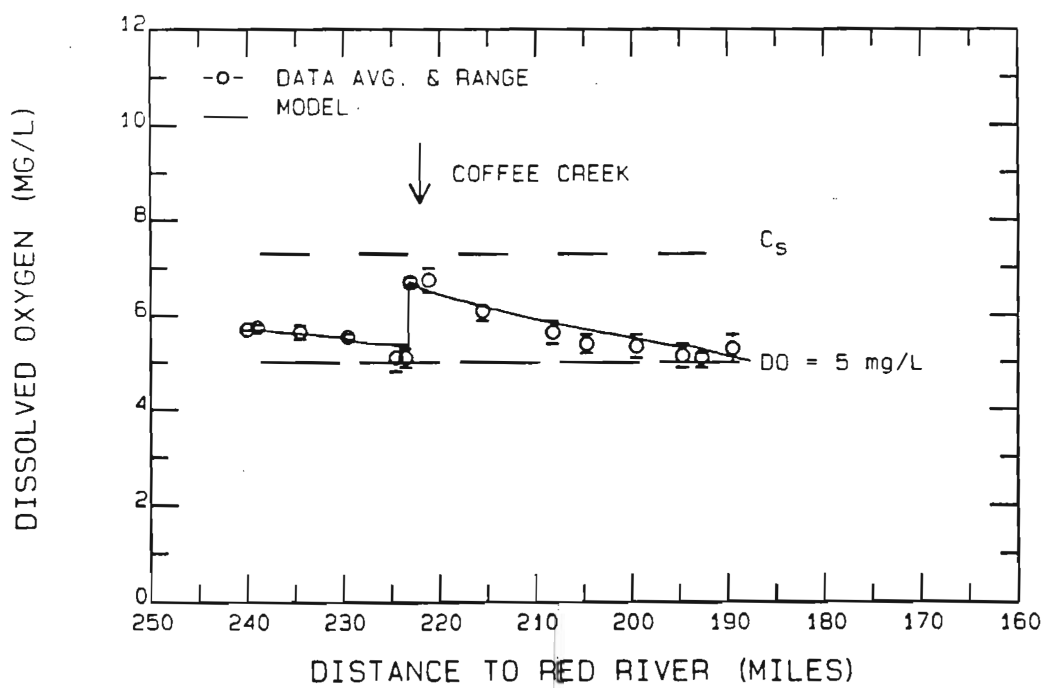


FIGURE 5-19. BOD-DO MODEL CALIBRATION RESULTS: JULY 22-23, 1980

value of $K_r = K_d = .05$/day (T = 20°C), based on a typical BOD bottle rate, was assigned in conjunction with the uniformly distributed CBOD load (Equations 5-1 and 5-2) which would result in a constant background CBOD profile of 3 mg/L. The rate of oxidation of the load from Georgia Pacific was considered independently, based on this and the other surveys to be discussed, resulting in an estimated CBOD removal rate coefficient of K_r = 0.1/day at 20°C. The resulting CBOD model profile is in reasonable agreement with the observed data. The BOD profile for the July 1980 survey was typical of the profiles observed in the other intensive surveys to be reviewed.

The model results are compared to the July survey dissolved oxygen data on the lower panel of Figure 5-19. The reaeration rate coefficient used reflects the reduction in river flow which occurred at the time of the survey. The effect of using a low deoxygenation coefficient for the background CBOD upstream of Lock and Dam 6 is that the observed data and calculated model profile show only a slight decrease of about 0.5 mg/L over a distance of 15 miles. It is significant to note that the dissolved oxygen level is at about 5 mg/L upstream of the dam, even under this relatively high flow condition, and hence the data alone indicate that natural background conditions will at times result in dissolved oxygen levels at or below the 5 mg/L standard.

The increase in dissolved oxygen near MP 223 is due to reaeration at the old Lock and Dam 6 and is evident during the two 1980 low stage surveys only. More recent surveys reflect operation of the new Lock and Dam, with release of water at the base of the dam, rather than over the crest. Operation is expected to return to release over the top of the dam in the near future, pending final inspection and approval of recently completed structural modifications to the dam.

Downstream of Coffee Creek there is a steady decrease and then leveling off of the dissolved oxygen profile and model and data are in reasonably good agreement.

5.2.2.2 October 23-24, 1991 Intensive Survey
(Q = 1946 cfs and T = 20°C)

This survey was conducted late in the year because relatively high flow conditions persisted throughout most of the summer. $CBOD_u$ concentrations, shown on the upper panel of Figure 5-20, were similar to the July 1980 survey, with upstream concentrations of about 2.5 mg/L and relatively flat concentration profiles in both the upstream and downstream reaches. The dashed profile in the reach immediately downstream of the discharge indicates the effect of the reduced flow on the day of the survey, which results in an elevated profile for a short distance downstream. The solid profile is based on the five day average flow, consistent with the travel time in the downstream study area. Agreement between model and the $CBOD_u$ data is considered good. Dissolved oxygen data and model results, shown on the lower panel, are high for this survey (above 7 mg/L) in comparison to the other data sets to be reviewed, reflecting the relatively low water temperature of only 20°C. The agreement between model and the dissolved oxygen data is also good. Note that for this post 1984 data set, there is no increase in dissolved oxygen levels at the location of the new Lock and Dam at Felsenthal (near MP 227).

5.2.2.3 September 2 and 3, 1980 Intensive Survey
(Q = 1012 cfs and T = 30.5°C)

This was the second low stage intensive survey conducted during 1980. As shown on the upper panel of Figure 5-21, BOD profiles are again relatively uniform in the upstream and further downstream river reaches, and as a result of the relatively low flow condition in comparison to the two previous surveys, a more significant increase is observed in $CBOD_u$ concentration at the discharge from Coffee Creek. The model profile reproduces the measured profile fairly well at this low flow condition, where the travel time in the downstream reach is about seven days.

Dissolved oxygen model results, shown on the lower panel of Figure 5-21, are also in excellent agreement with the data in both the upstream and downstream reach. Note

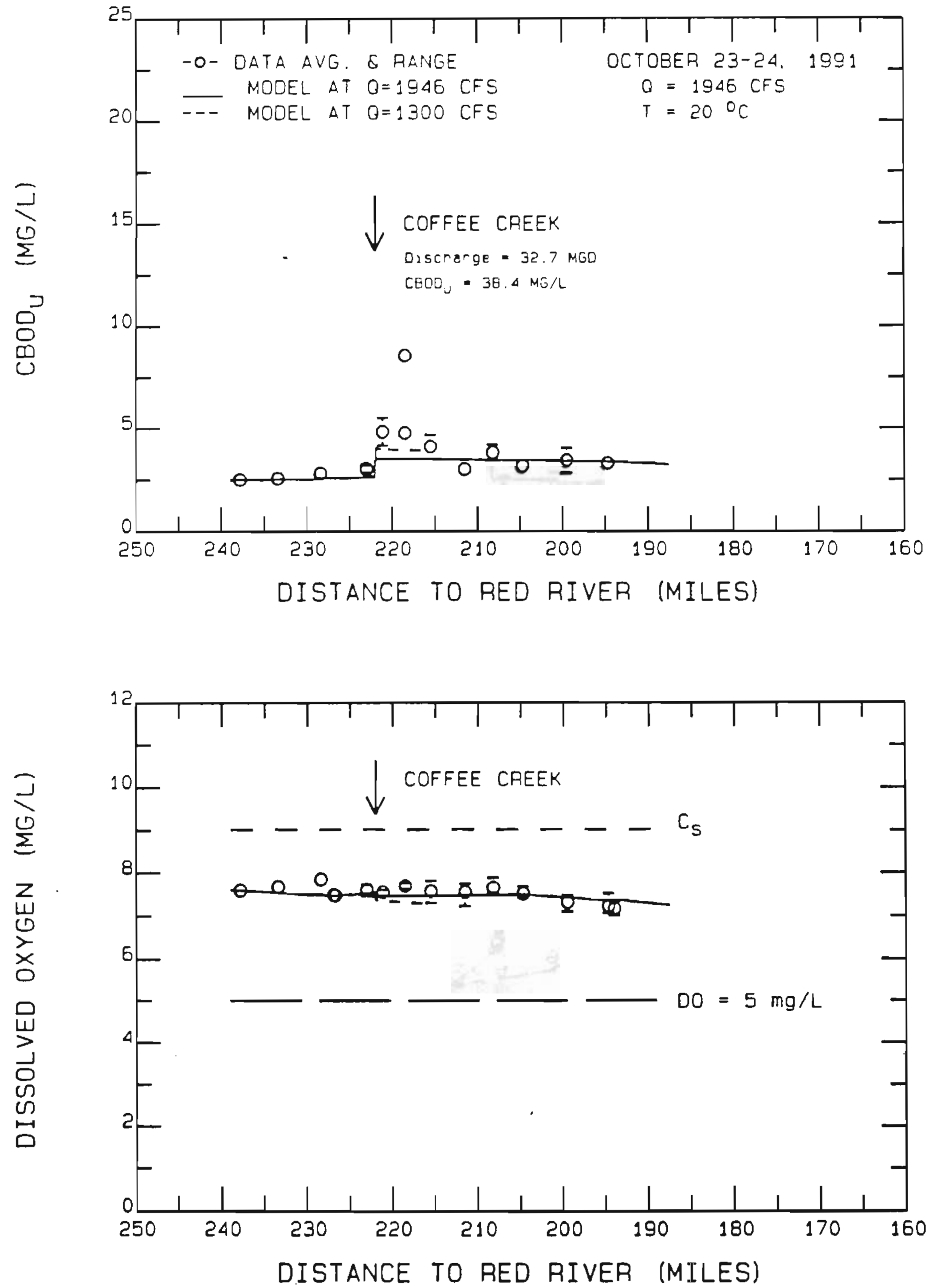


FIGURE 5-20. BOD-DO MODEL CALIBRATION RESULTS: OCTOBER 23-24, 1991

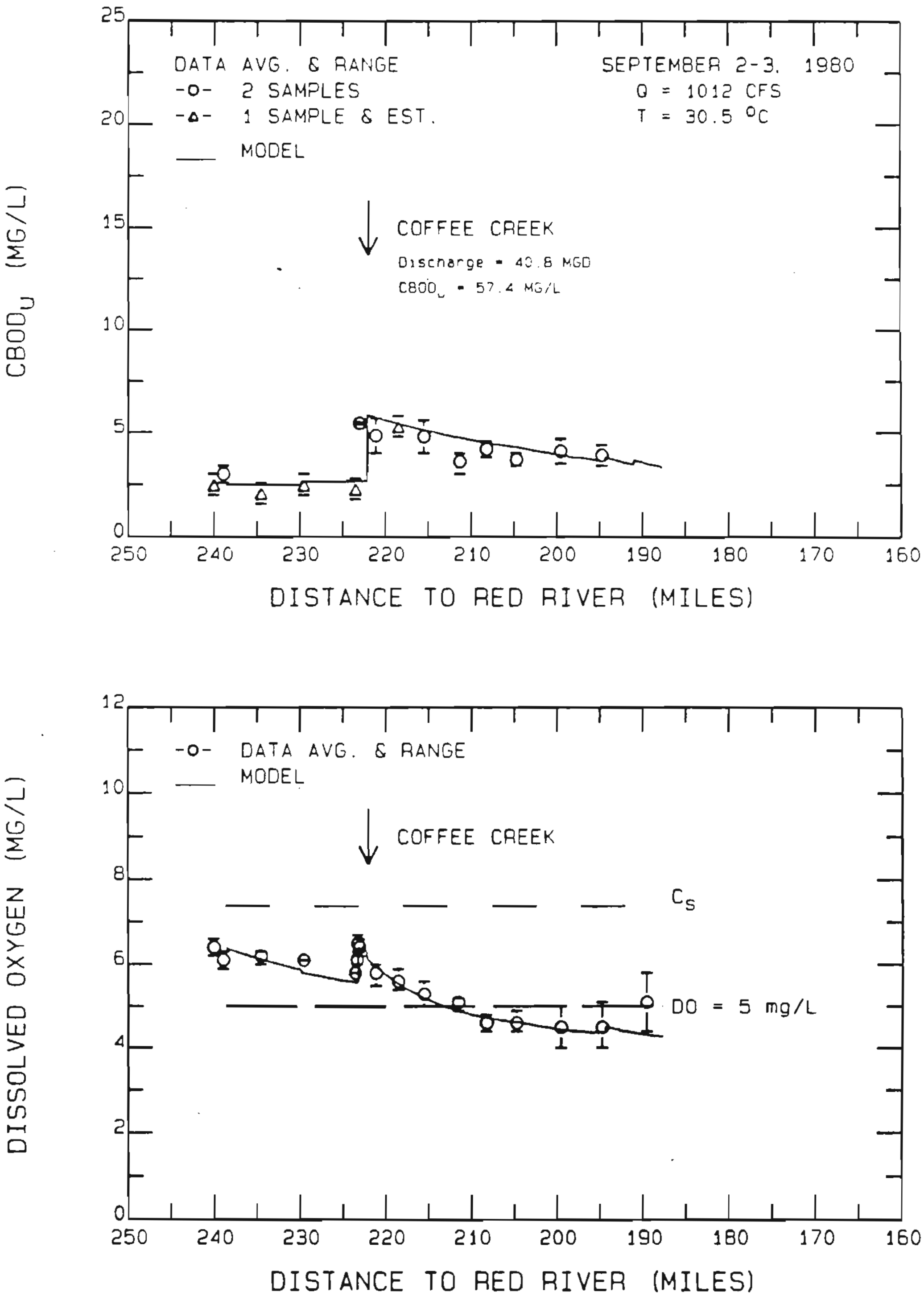


FIGURE 5-21. BOD-DO MODEL CALIBRATION RESULTS: SEPTEMBER 2-3, 1980

again that the increase in dissolved oxygen upstream of the discharge is associated with reaeration at the old Lock and Dam 6. In the downstream reach, instream dissolved oxygen levels decrease to about 4 mg/L at this relatively low flow and high temperature survey condition. With the exception of the most downstream sample location, the model fits the data quite well.

5.2.2.4 <u>July 1987 Routine Monitoring Data</u>
(Q = 892 cfs and T = 29.8°C)

This data set is based on five sets of routine weekly monitoring dissolved oxygen data reported by Georgia Pacific for the month of July 1987. The data were measured at the five foot depth in the water column during sample runs started at about 8:00 in the morning and usually finished by early in the afternoon. Conditions during 1987 during the low stage period are summarized on the temporal plot shown on Figure 5-22, with filled plot symbols during July corresponding to data from the period analyzed with the model. The upper left panel shows river flows on days that routine surveys were conducted (USGS provisional data). Based on these data the average river flow for the month of July was 892 cfs, close to the monthly 7Q10 for July of 894 cfs, and the temperature was 29.8°C. As shown on the upper right panel, the average effluent BOD_5 load for this survey was 3,405 lbs/day and using a ratio of $CBOD_u/BOD_5 = 4$, the $CBOD_u$ load was estimated at 13,620 lbs/day. Since instream BOD data were not available from the routine monitoring surveys, the upstream $CBOD_u$ concentration and the uniform load used with this data set was based on the average of the two intensive surveys completed prior to the time this analysis was performed. The flow and loading conditions during July 1987 were taken to be representative of pseudo-steady state conditions for modeling purposes.

The 4 lower panels of Figure 5-22 show temporal plots of dissolved oxygen (5 foot depth) for the spring and summer of 1987. The 2 middle panels present data from stations upstream of the discharge (MP 234 and MP 224) of Coffee Creek and indicate that the background dissolved oxygen was about 5 mg/L during the period simulated. The lower panels show that data in the downstream reach was often in the range of 4 to 5

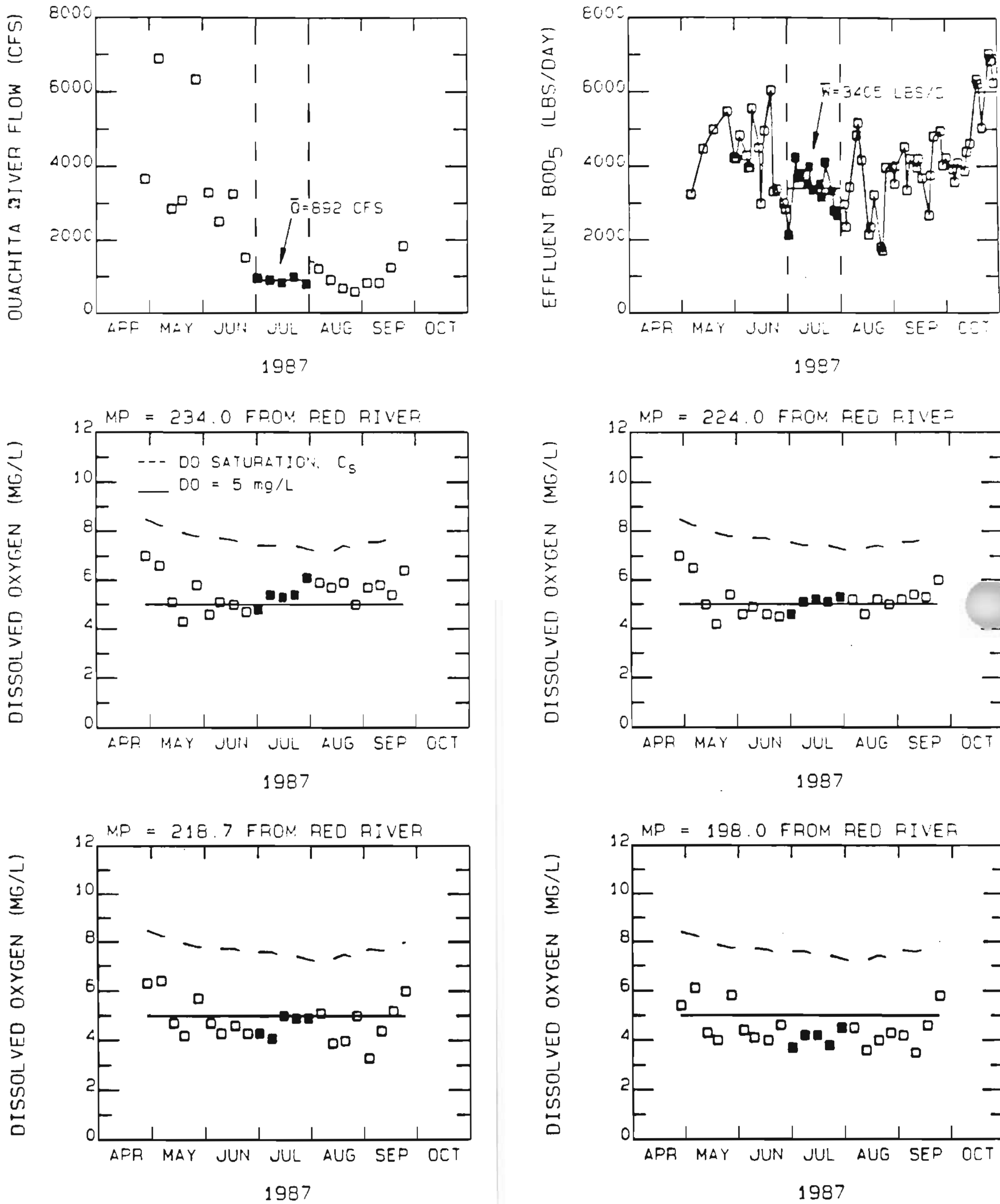


FIGURE 5-22. EFFLUENT LOAD, RIVER FLOW AND DO DURING JULY 1987 SURVEY PERIOD

mg/L during these months.

Model results are compared to the time averaged dissolved oxygen data for the 5 July 1987 surveys on Figure 5-23. Here, the range reflects the variation over the five different surveys, rather than over depth or time of day. The model results, based on the monthly average flow, temperature and loading conditions are seen to be in good agreement with the average dissolved oxygen levels for the month. This is all the more significant in that the flow and temperature conditions are close to what would be considered critical low flow and high temperature conditions for the month of July.

5.2.2.5 September 18 and 19, 1990 Intensive Survey
(Q = 676 - 2320 cfs and T = 29.5°C)

The final calibration data set to be reviewed is for the September 1990 survey which was preceded by a period of variable flows. The flow distributions determined for use as part of the conservative tracer analysis (Figures 4-6 through 4-8) were used as the basis of this analysis. Flows preceding this survey included daily average flows that were the lowest of any survey analyzed (676 cfs) as well as several days when the flow was about a factor of three higher than this low flow condition (over 2,000 cfs). As a result of this significant variation in flow, the steady state model was run in a pseudo-time variable mode. That is, water passing the discharge on each individual day prior to the survey was considered independently over time as it moved downstream, and travel time for each parcel of water reflected the cumulative travel time associated with the sequence of variable daily flows that moved by the discharge. To use this method, a series of 17 steady state simulations was required, rather than the single steady state simulation used to evaluate the other data sets previously reviewed.

The upper panel of Figure 5-24 compares the model results with the $CBOD_u$ data for this survey. As shown, good agreement is obtained between model and data. Note again how the upstream and downstream profiles are characteristically flat. It is interesting to note, however, that in contrast to previously reviewed $CBOD_u$ profiles, the furthest

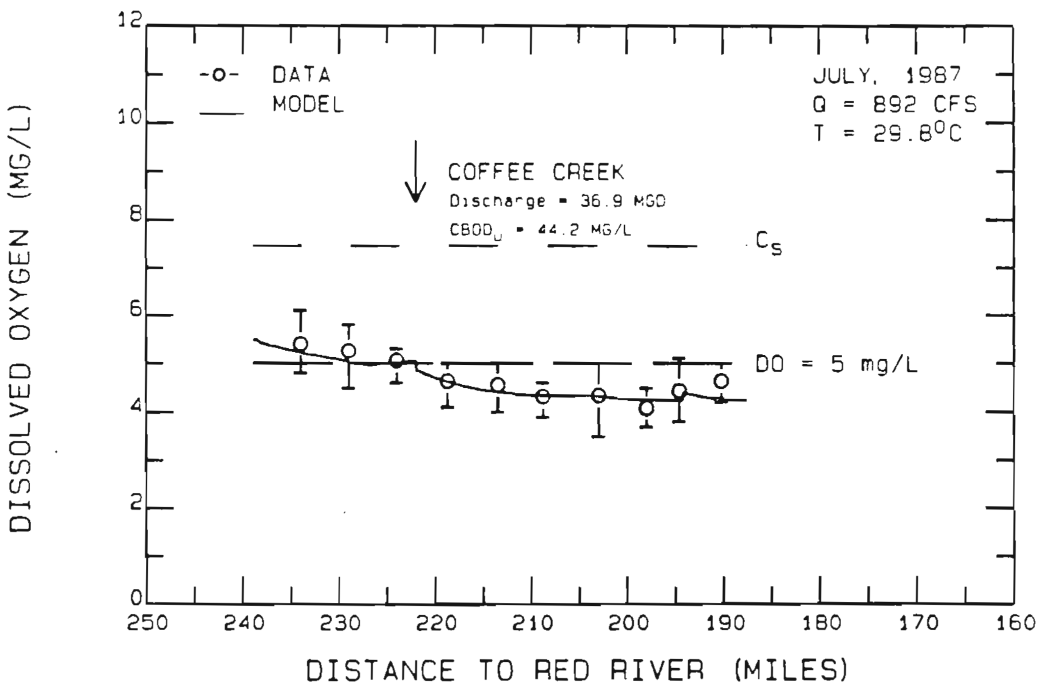


FIGURE 5-23. BOD-DO MODEL CALIBRATION RESULTS: JULY 1987

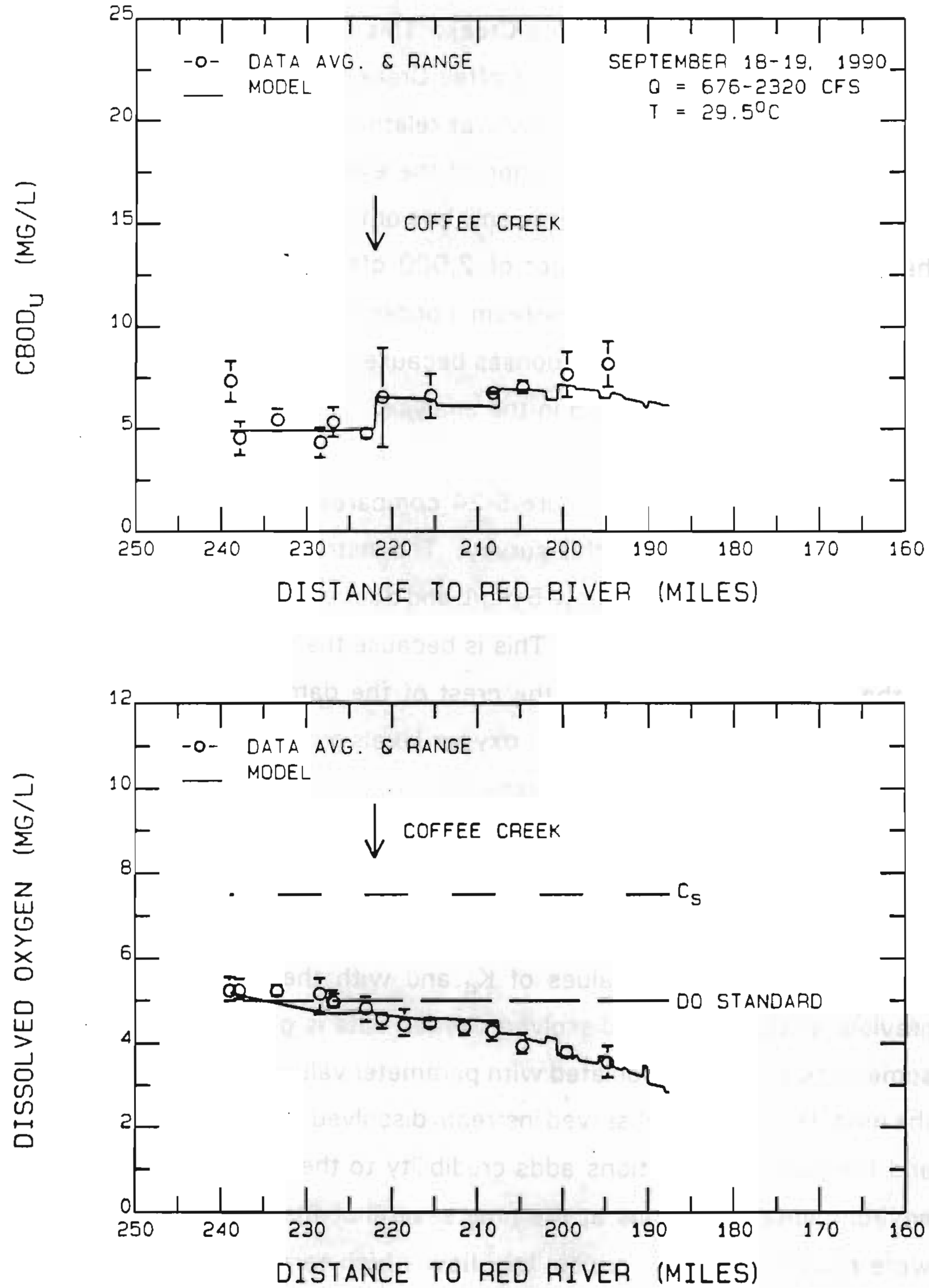


FIGURE 5-24. BOD-DO MODEL CALIBRATION RESULTS: SEPTEMBER 18-19, 1990

downstream $CBOD_u$ concentrations are not reduced with respect to concentrations in the reach just downstream of Coffee Creek. This occurs because water at the downstream end of the study area flowed by Coffee Creek about 7 days prior to the date of sample collection, on a day when the flow was relatively low (about 800 cfs) and relatively little river water was available for dilution of the effluent. The water just downstream of the discharge moved by the discharge only one or two days before sampling took place, when the river flow was on the order of 2,000 cfs, and hence much better dilution of the effluent was obtained and instream concentrations were relatively low. The model reproduces these instream responses because variations in flow and differences in travel time are properly represented in the analysis.

The lower panel of Figure 5-24 compares the time variable model output to the dissolved oxygen data for this survey. The instream concentration just upstream of the discharge is at or slightly below 5 mg/L and does not increase as a result of dam reaeration at Lock and Dam 6 (MP 226). This is because the water was being released at the base of the dam, rather than over the crest of the dam, at the time of this survey. In the downstream reach, the dissolved oxygen levels gradually decrease to about 3.5 mg/L, and model and data are in good agreement.

5.2.2.6 Discussion of BOD-DO Model Calibration Results

At the assigned values of K_d and with the other model parameters discussed previously, the fit of the dissolved oxygen data is generally considered good. Although some uncertainty is associated with parameter values assigned in the model, the fact that the model reproduces observed instream dissolved oxygen levels over a wide range of flow and temperature conditions adds credibility to the overall analysis. It is significant that environmental conditions at the time several of the calibration data sets were collected were representative of critical low flow - high temperature conditions and hence error associated with extrapolation to more extreme conditions is minimized.

It is noted that the model calibration results are based on comparisons of model output to the average and range of morning and afternoon dissolved oxygen readings, averaged over depth, in order to approximate daily average conditions and to characterize the variation in dissolved oxygen over the day. Dissolved oxygen readings with depth in the water column were typically near the surface, at mid-depth and about three feet above the bottom, although depths varied somewhat by survey. Near surface readings were generally at a 1 foot depth during 1980, 0.5 and 5 feet during 1990, and 3 and 5 feet during 1991. A comparison of depth average dissolved oxygen levels to the dissolved oxygen at the five foot and three foot depths, the depths specified for instream monitoring purposes by Arkansas and Louisiana respectively, indicated that a bias did not generally exist between the depth averaged concentrations and readings at these regulatory depths. These results are shown on Figure 5-25 in terms of spatial plots of the residual dissolved oxygen (depth average dissolved oxygen minus dissolved oxygen at five feet in Arkansas and dissolved oxygen at three feet in Louisiana). The July 31 and August 1, 1991 data (upper panel) indicate a bias of less than 0.1 mg/L throughout the study area. Data from October 23 and 24, 1991 (lower panel) show a bias of less than about -0.2 mg/L at upstream stations (depth average dissolved oxygen less than dissolved oxygen at five foot depth) and 0.2 to 0.3 mg/L at downstream stations. With regard to the October data, since the model is calibrated against depth averaged data, use of the model to predict the dissolved oxygen at the three foot depth would be conservative. Further since State regulations allow for natural occurring variations of dissolved oxygen from the standard due to diurnal algal effects it is most appropriate to compare the steady state calculated dissolved oxygen concentration with the daily depth average dissolved oxygen.

The BOD-DO model calibration was based primarily on analysis of the five low flow-low stage data sets reviewed in this section. Kinetic coefficients established from analysis of these data sets, in conjunction with analyses described earlier in this section, were also used to evaluate the impact of the Georgia Pacific discharge during flooded, high stage conditions, as described in Section 6.

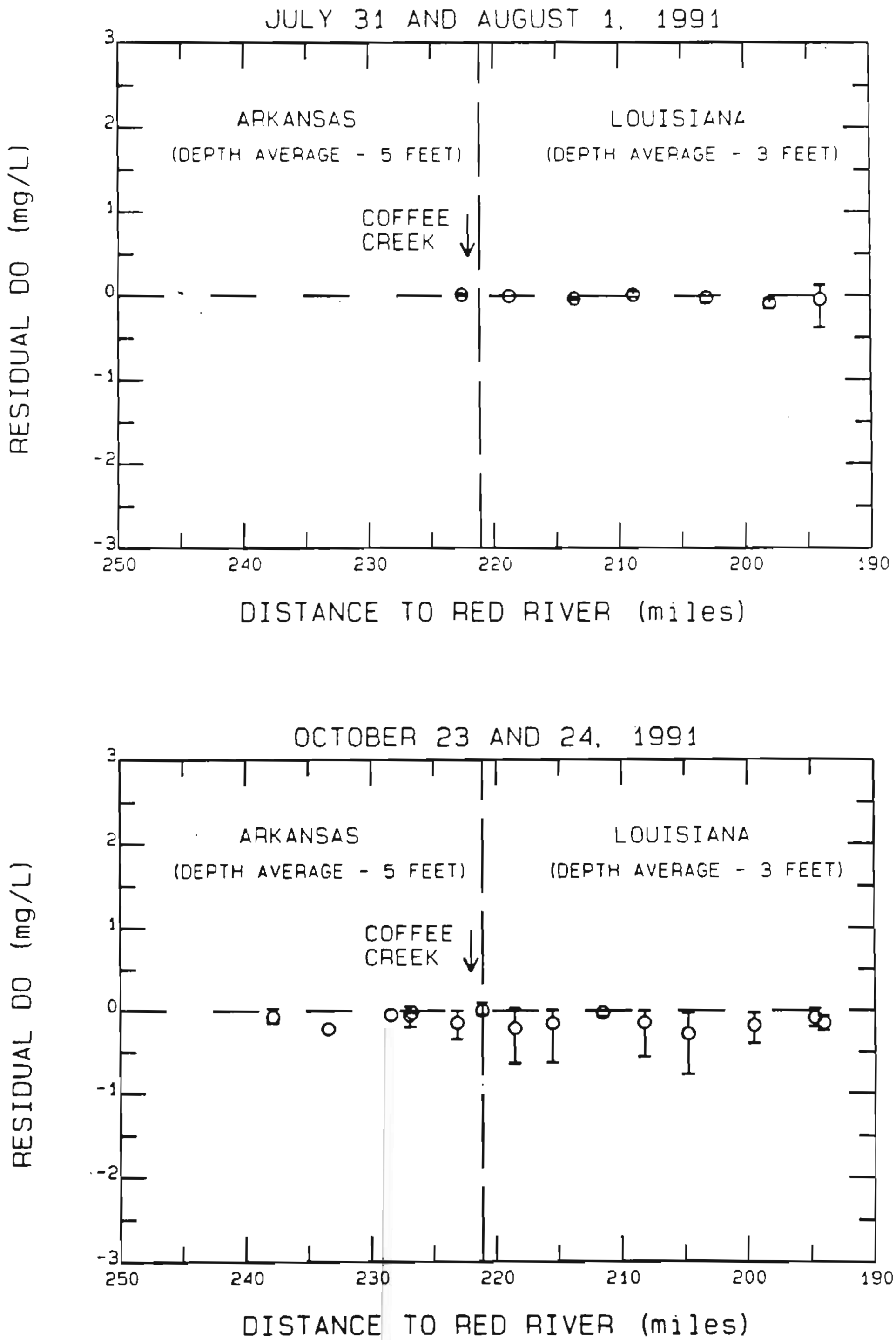


FIGURE 5-25. BIAS IN DO READINGS AT 3 FOOT AND 5 FOOT DEPTHS RELATIVE TO DEPTH AVERAGE DO READINGS (1991 DATA)

5.3 MODEL SENSITIVITY ANALYSIS

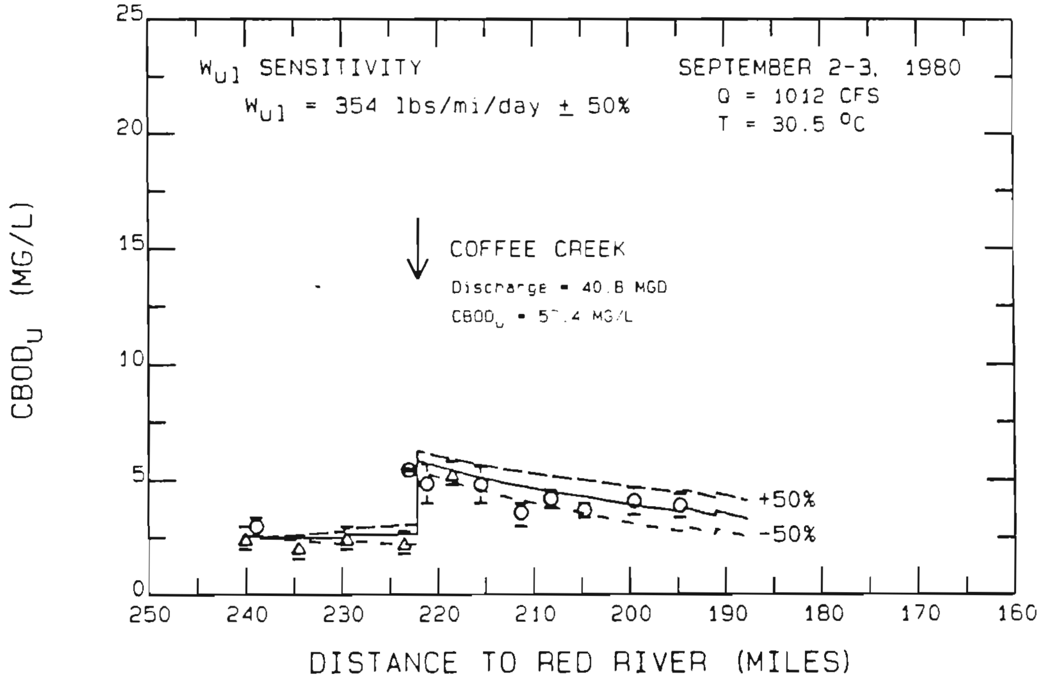
Development of a water quality model requires estimates of a number of model parameters which have a direct bearing on the validity of the results. It is not possible to determine a precise value for each model input incorporated in the model framework and as a result it is informative to perform a sensitivity analysis. As utilized herein, the sensitivity analysis addresses the following question: How sensitive are the results to perturbations in the assigned values of the key model parameters? The analysis is intended to provide insight into the reliability of the water quality model. The September 1980 BOD/DO verification was used as the base case for most of the sensitivity analyses performed, since this was a relatively low flow and high temperature survey condition.

5.3.1 Uniform CBOD Load

A uniformly distributed background $CBOD_u$ load of about 0.8 grams/m^2/day (354 lbs/mile/day) was calculated for the September 1980 survey to reproduce the uniform profile observed upstream of Coffee Creek. The BOD profile, shown on the upper panel of Figure 5-26, is somewhat sensitive to a ±50 percent change in this parameter, with a difference of almost 1 mg/L in some areas (the base case is denoted by the solid line profile). The dissolved oxygen profile (lower panel of Figure 5-26) is only slightly sensitive to this parameter, however, indicating that a precise determination of W_{ul} is not critical. For other surveys when the uniform load is as high as 1.5 to 2.2 grams/m^2/day (about 800 lbs/mile/day during September 1990), the results would be approximately twice as sensitive to the uniform load.

5.3.2 Load BOD De-oxygenation Rate Coefficient

Figure 5-27 presents the model sensitivity to perturbations of the BOD removal rate coefficient for the effluent load. The September 1980 base case model results using $K_r = K_d = 0.1$/day are indicated by the solid line on the spatial plots of BOD and dissolved oxygen shown on Figure 5-27. K_r and K_d were varied by 50 percent (0.05/day to



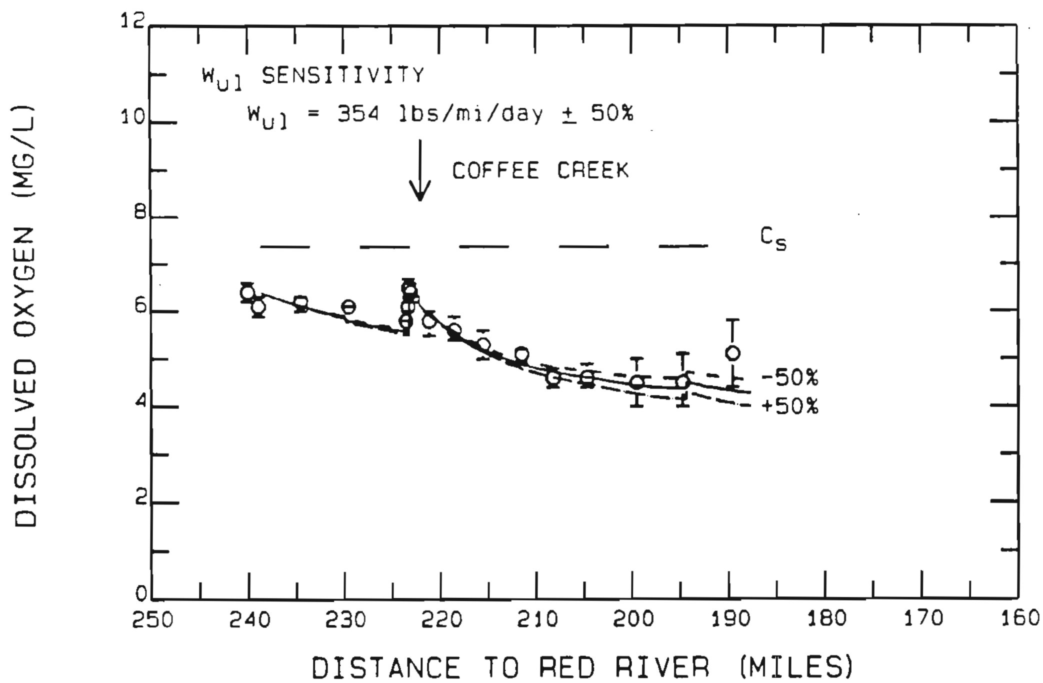


FIGURE 5-26. SENSITIVITY ANALYSIS: SEPTEMBER 2-3, 1980 SURVEY UNIFORM $CBOD_u$ LOAD

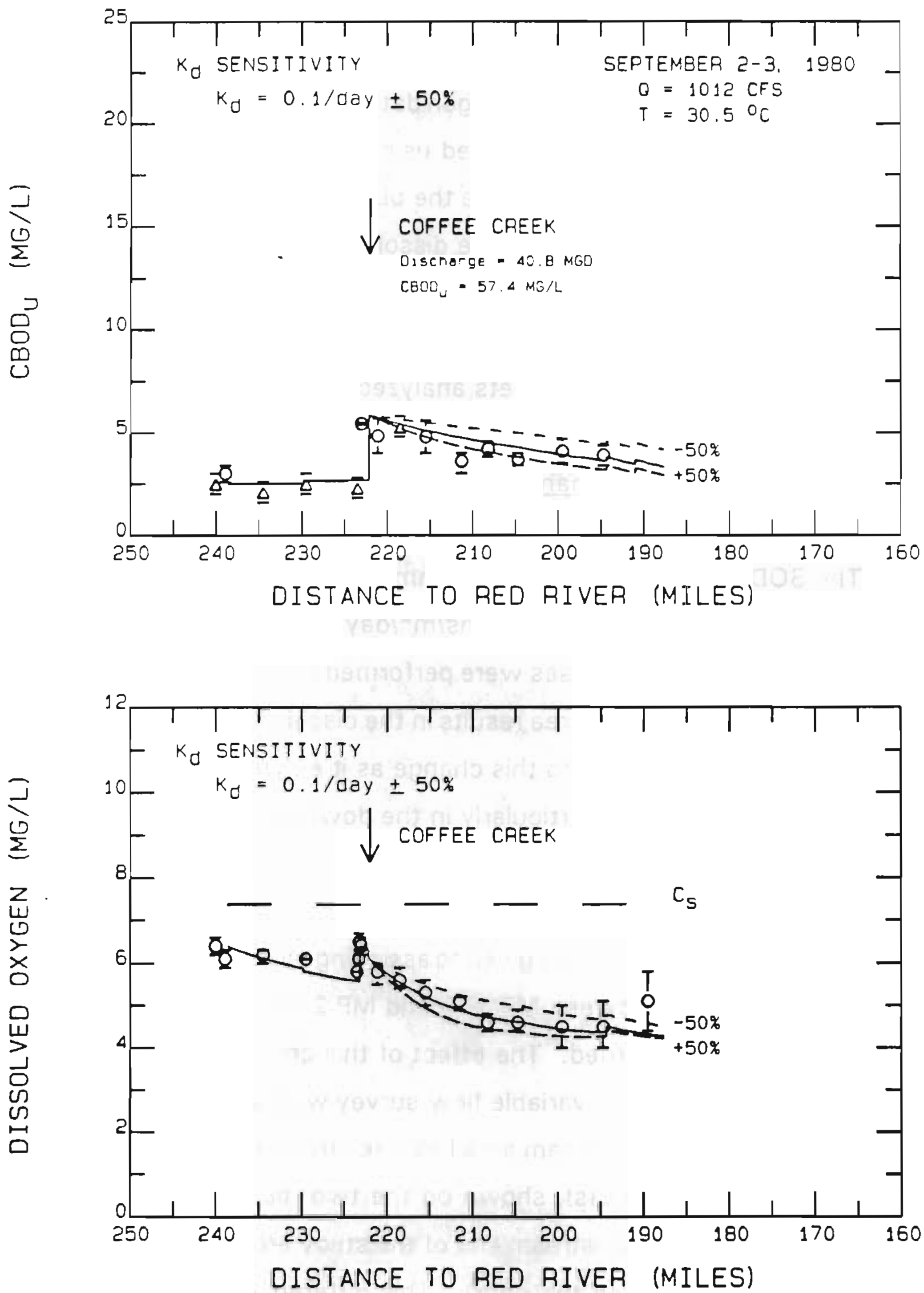


FIGURE 5-27. SENSITIVITY ANALYSIS: SEPTEMBER 2-3, 1980 SURVEY
BOD DE-OXYGENATION RATE COEFFICIENT, Kd

0.15/day) and the corresponding change in the computed profiles is shown by the dashed lines. Using K_r = 0.15/day, the fit of the BOD data is comparable (the lower dashed profile), but the fit of the dissolved oxygen data is not quite as good (lower dashed profile). A poorer fit of the BOD data is obtained using K_r = 0.05/day and the dissolved oxygen profile increases about 0.4 mg/L above the observed averages, resulting in a poorer fit of the dissolved oxygen data as well. The dissolved oxygen model profile is in best overall agreement with the data based on a value of K_r = 0.1/day. It should also be noted that use of $K_r = K_d$ = 0.1 generally results in the best overall fit of the BOD and dissolved oxygen data from the other data sets analyzed as well.

5.3.3 Sediment Oxygen Demand

The SOD data were previously summarized on Figure 5-11. On the basis of these data a spatial average SOD of 0.5 grams/m^2/day was assigned throughout the study area. Two types of sensitivity analyses were performed for SOD. First, a ± 50 percent change in SOD throughout the study area results in the dissolved oxygen profiles shown on Figure 5-28. The model is sensitive to this change as it either underestimates or overestimates the dissolved oxygen data, particularly in the downstream reach, when these changes in SOD are made.

Further consideration was given to assigning an increased SOD of 0.8 grams/m^2/day (at 20°C) in the reach between MP 190 and MP 204, where several relatively high SOD measurements were recorded. The effect of this change is shown for all 4 steady state surveys on Figure 5-29 (the variable flow survey was not evaluated). An increase to 0.8 grams/m^2/day in the downstream area has a relatively minor effect during the higher flow surveys (3086 and 1946 cfs), shown on the two panels on the left, and an impact of about 0.7 mg/L at the downstream end of the study area during the more critical low flow surveys (the two panels on the right). This difference is considered significant for the lower flow surveys and leads to an inconsistency with the dissolved oxygen data which would need to be offset in some manner if the higher SOD were employed. In view of the excellent agreement of the model with observed spatial profiles of dissolved oxygen, over

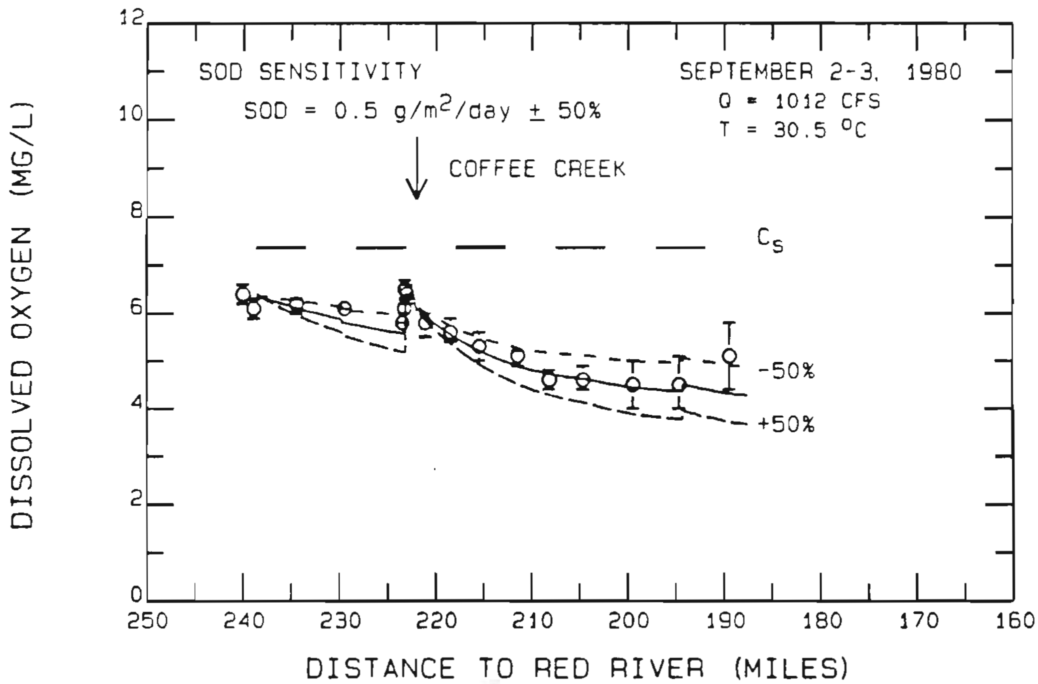


FIGURE 5-28. SENSITIVITY ANALYSIS: FOUR LOW STAGE SURVEYS
SEDIMENT OXYGEN DEMAND - THROUGHOUT STUDY AREA

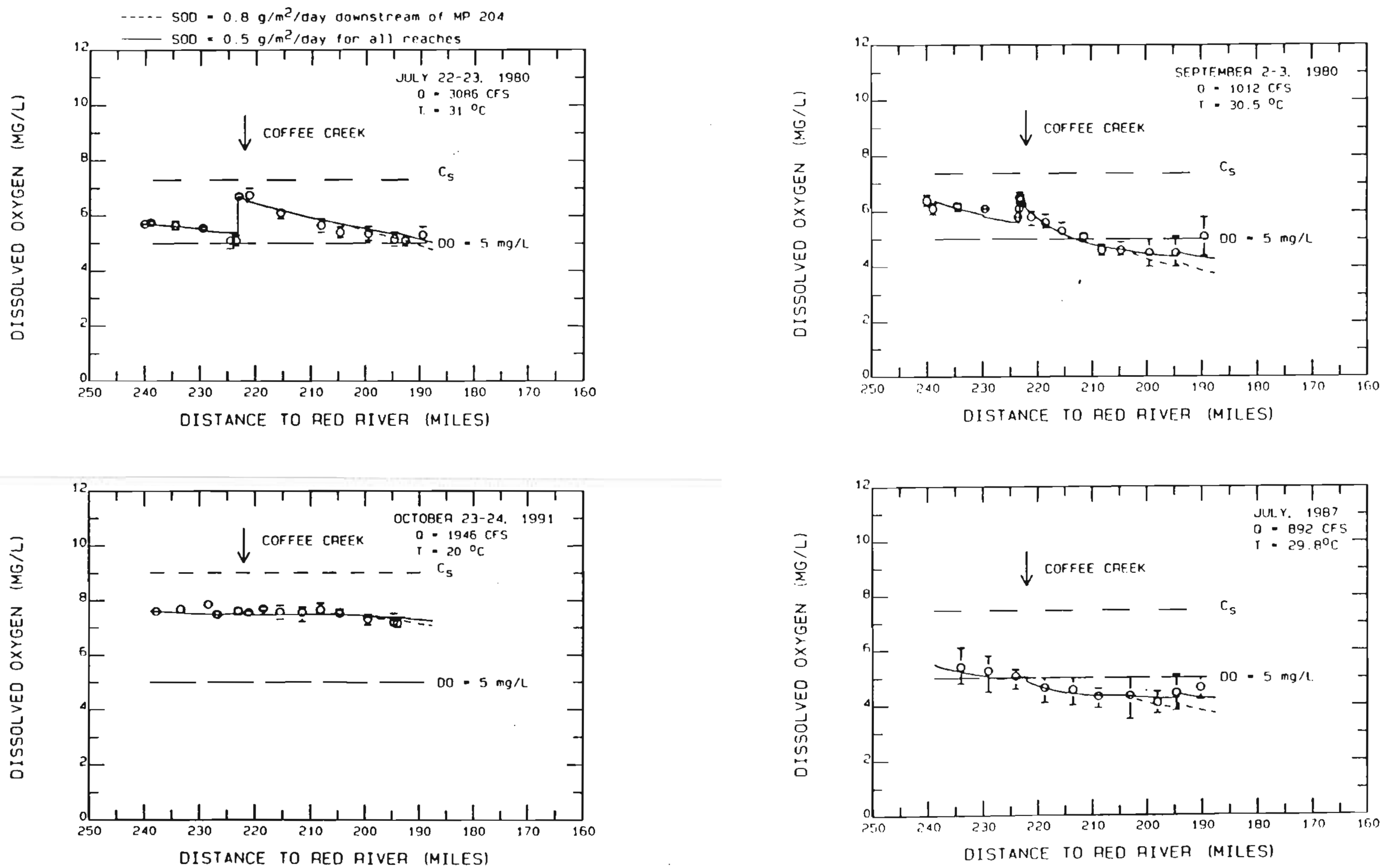


FIGURE 5-29. SENSITIVITY ANALYSIS: SEPTEMBER 2-3, 1980 SURVEY
SEDIMENT OXYGEN DEMAND - DOWNSTREAM OF MP 204

a wide range of conditions, when the independently evaluated model coefficients are used (solid line profiles of Figure 5-29, and Figure 5-24), as well as the known variability in SOD measurements, an SOD of 0.5 grams/m^2/day is used.

5.3.4 Nitrification Rate Coefficient

Model sensitivity analyses for nitrification were in part presented previously in Section 5.2.1, for several surveys where TKN (organic + ammonia nitrogen), ammonia and nitrate nitrogen data were available (Figures 5-17 and 5-18). The result of these analyses was that nitrification rates of 0.10/day (July 1980) and 0.05/day (September 1990) applied to TKN resulted in a nitrate nitrogen increase in the downstream region which clearly exceeded the instream nitrate data. A zero nitrification rate applied to ammonia nitrogen resulted in the best overall fit of the ammonia nitrogen data and also reproduced downstream nitrate profiles (assumed conservative) quite well.

The September 1980 survey was not amenable to the same type of modeling analysis as just described for the July 1980 and September 1990 surveys because of either analytical problems with the nitrogen data, loading variability, or some other unexplained or poorly understood phenomenon. The instream TKN data vary erratically, but generally increase downstream of Coffee Creek and the nitrate data increase slightly and then oscillate between about 1.0 and 0.1 mg/L in the downstream reach. As a result, the nitrogen data for this survey are disregarded and attention is focused on the possible impact of nitrification on the September 1980 dissolved oxygen profile. A rate coefficient of 0.1/day is applied to TKN (effluent TKN = organic + ammonia = 4.5 mg N/L) and to ammonia nitrogen (effluent ammonia = 1.7 mg N/L). The upper panel of Figure 5-30 shows that the impact on dissolved oxygen is significant when nitrification is applied to TKN, with the calculated profile considerably below the observed dissolved oxygen data by 1 mg/L or more. In contrast, when nitrification is applied to ammonia nitrogen only, a more reasonable scenario, the impact is negligible (lower panel of Figure 5-30).

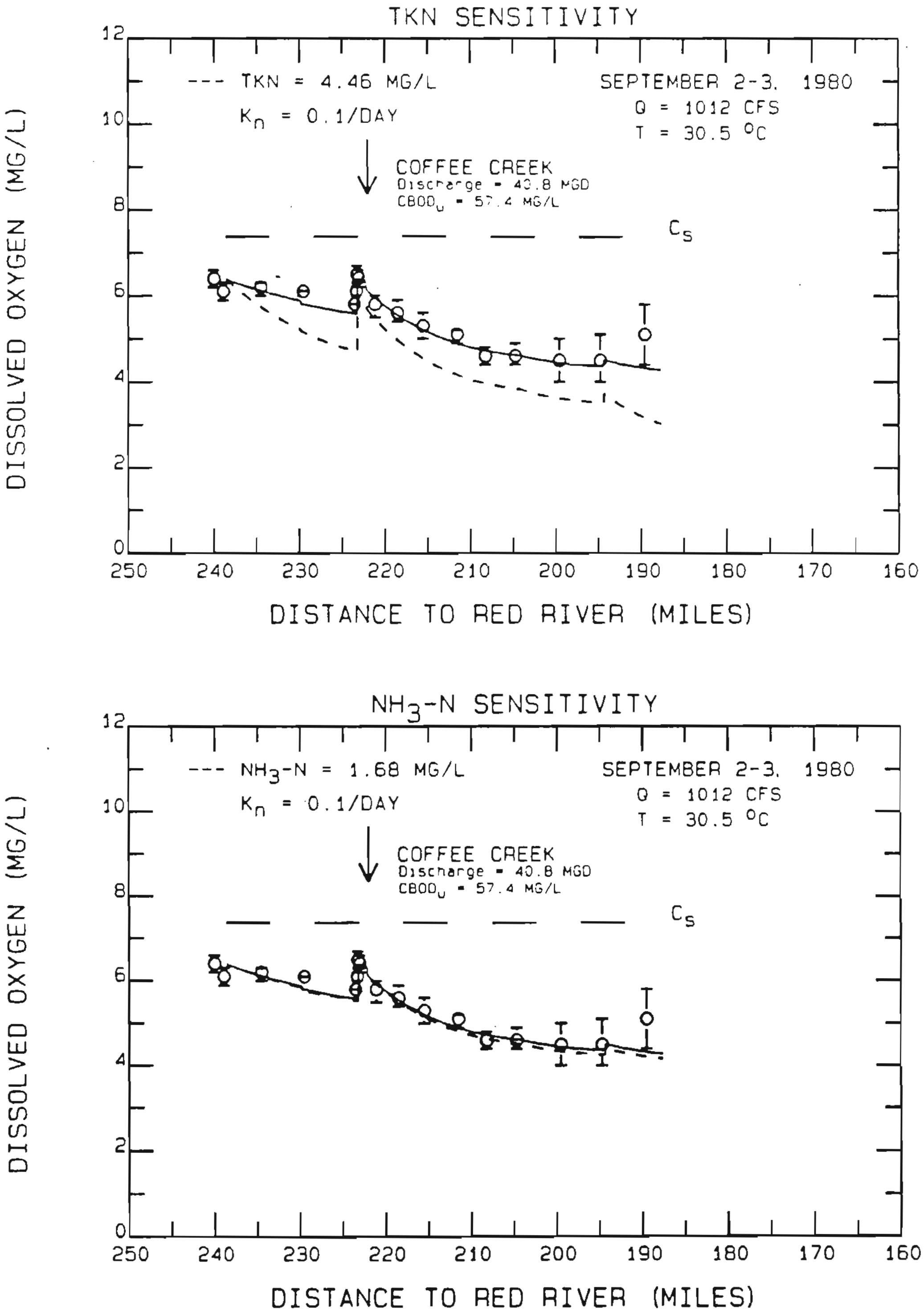


FIGURE 5-30. SENSITIVITY ANALYSIS: SEPTEMBER 2-3, 1980 SURVEY NITRIFICATION RATE COEFFICIENT

The main reason the impact is so much greater when TKN is assumed to nitrify is that the upstream boundary for TKN was 1 mg/L and for ammonia nitrogen it was only about 0.1 mg/L. These boundary concentrations are consistent with the 1991 upstream monitoring data (Appendix C2) where TKN levels were typically in the range of 0.5 to 2 mg/L and ammonia nitrogen levels were consistently about 0.1 mg/L or less.

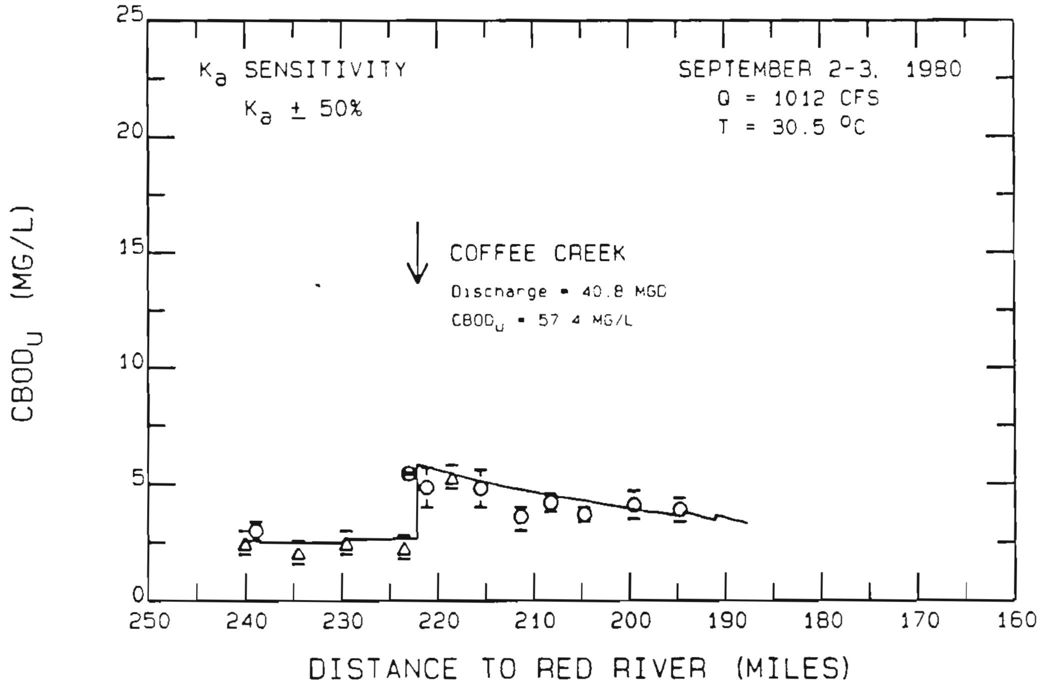
5.3.5 Reaeration Rate Coefficient

Model sensitivity to the reaeration rate coefficient (K_a) is presented on Figure 5-31. The BOD profile, which is not affected by reaeration, is shown on the upper panel and dissolved oxygen on the lower panel. As shown, the calculated dissolved oxygen profile is sensitive to a ± 50 percent change in the reaeration rate coefficient. Departure from the observed averages is slight in the background reach, upstream of Lock 6, where a 50 percent increase in K_a still provides a reasonable fit of the data. In the downstream reach, however, which was considered independently (the dissolved oxygen was reset to the observed data downstream of the dam), a 50 percent increase in K_a results in a profile which is significantly higher than the observed data and as much as almost 1.0 mg/L higher than the base case results. A 50 percent reduction in K_a results in calculated concentrations that are about 1.5 mg/L lower than the base case results at the downstream end of the study area.

5.3.6 Discussion of Sensitivity Analysis Results

Based on the sensitivity analyses presented above, the dissolved oxygen concentration in the river, within the range of coefficients studied, is relatively insensitive to the CBOD oxidation rate of the Georgia Pacific waste load and to the uniform background CBOD loading.

The dissolved oxygen results are somewhat more sensitive to the sediment oxygen demand, particularly at lower flow conditions, but it is considered that a reasonable estimate of the SOD in the river has been used, based on the available field and laboratory



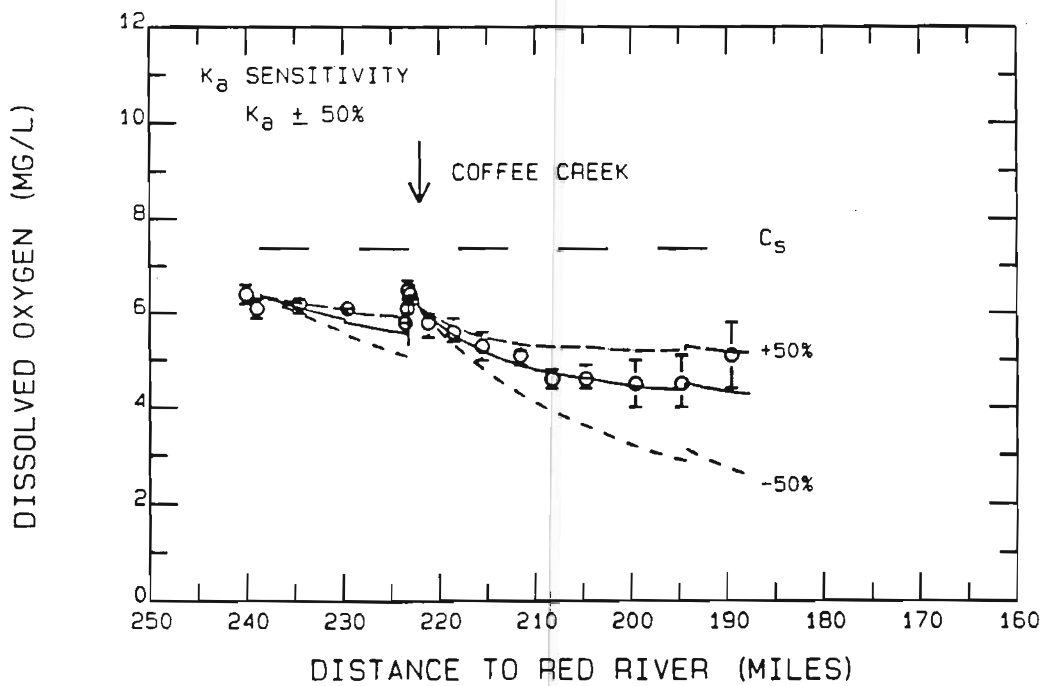


FIGURE 5-31. SENSITIVITY ANALYSIS: SEPTEMBER 2-3, 1980 SURVEY REAERATION RATE COEFFICIENT

measurements.

It is concluded from the nitrification analyses, both in Section 5.2.1 and in Section 5.3.4, that nitrification is not occurring in the reach of the Ouachita River between Coffee Creek and Bayou Bartholomew. This conclusion is based on both the absence of an accumulation of nitrate nitrogen in the downstream direction, and also on the substantial impact on the dissolved oxygen levels in the downstream region which would need to be offset in some other manner in order to maintain the overall oxygen balance in the river.

The calculated dissolved oxygen profile is also sensitive to the reaeration rate coefficient. The model uses the O'Connor formulation for predicting reaeration, and as described previously (Section 5.1.1), this formulation is consistent with the limited field data that exists for reaeration in the Ouachita River. In the absence of additional reaeration rate data, which is difficult to obtain for a slow moving water body such as the Ouachita River, it is judged that this reaeration rate formulation should continue to be used.

5.4 COMPONENTS OF DEFICIT

Estimates of the relative contributions of the various sources and sinks of dissolved oxygen to the total deficit in the river can be readily determined using the mathematical model of the Ouachita River that has been developed. Figure 5-32 compares the components of dissolved oxygen deficit for the low flow model calibration to the data set of July 1987 (Q = 892 cfs and T = 29.8°C) and for the high flow model calibration to the intensive survey data set of July 1980 (Q = 3086 cfs and T = 31°C). During the low flow survey shown on the left, the maximum dissolved oxygen deficit associated with the effluent (upper panels) increases to about 1 mg/L in the vicinity of MP 200, and then decreases downstream of that location. During the higher flow survey, shown on the right, the maximum deficit due to the load is generally less than 0.5 mg/L.

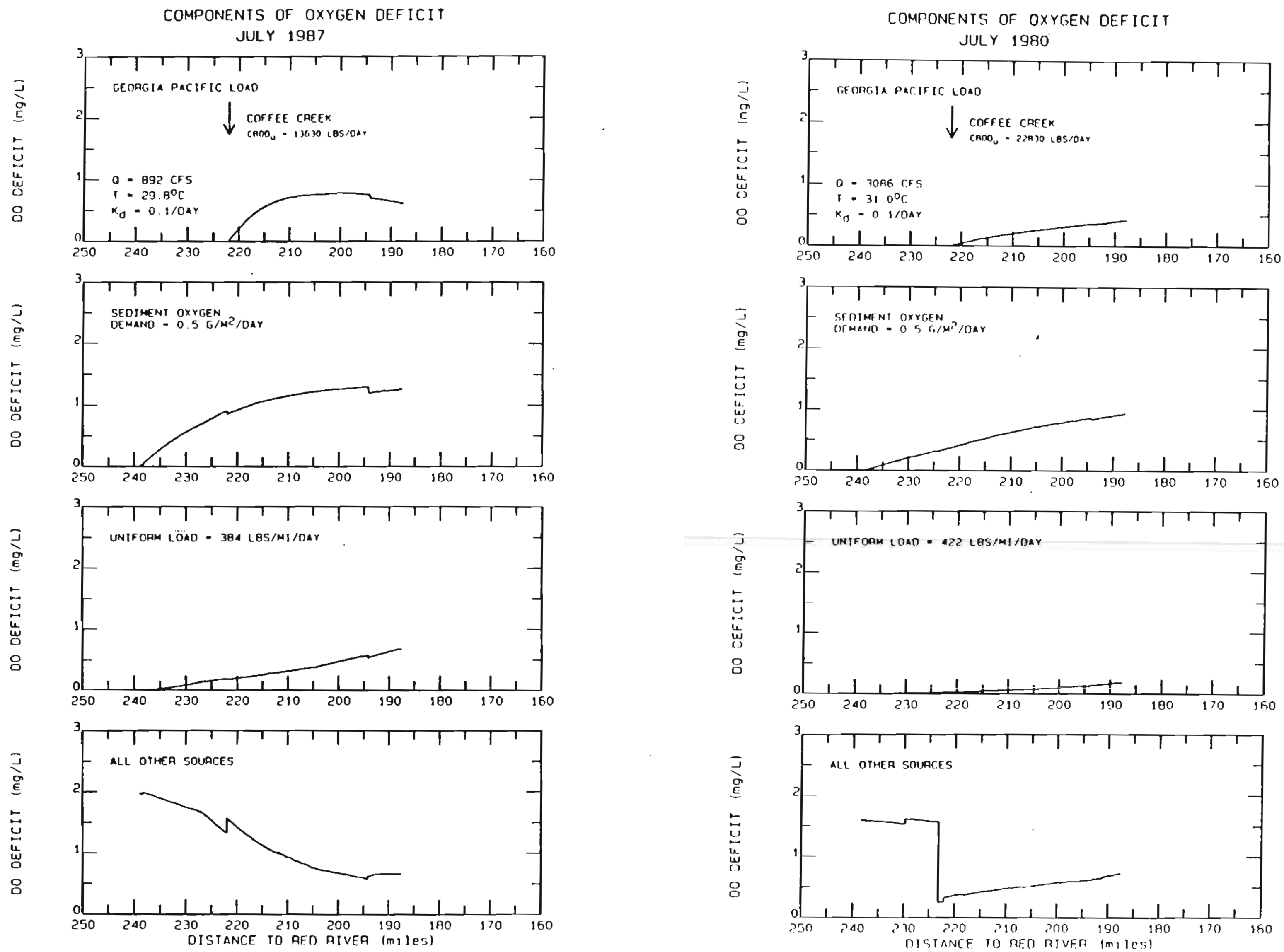


FIGURE 5-32. COMPONENTS OF DISSOLVED OXYGEN DEFICIT

The deficit contribution associated with the load during July 1987 shows that the model extends far enough downstream during relatively low flow conditions, including 7Q10 conditions, that the location of maximum deficit is reached within the study area. All of the effluent CBOD has not yet been oxidized at the downstream end of the study area (about MP 190) under these conditions, however. In comparison, during the higher flow conditions of the July 1980 analysis, the river flow was about 3,000 cfs and the travel time was relatively short. Thus, the critical location does occur downstream of the study area. At this higher river flow the impact of the mill discharge is relatively limited and dissolved oxygen levels in the study area were greater than 5 mg/L. The reach modeled was selected on the basis of initial estimates that the critical location with respect to the Crossett mill's impact would occur within the study area during low flow periods and also, to avoid analysis of the increasingly complex downstream reach where a significant number of additional point sources discharge to the river.

SOD is also a significant source of deficit in the river (second panel), at about 1.3 mg/L during the low flow survey and 1.0 mg/L during the high flow survey, in the downstream region. Although the SOD of 0.5 $g/m^2/day$ at 20°C is not considered high, it has a significant impact due to both the elevated temperature conditions of these model simulations and the relatively low reaeration capacity of the Ouachita River. Deficit associated with the uniform load is less important but still significant (third panel), increasing in the downstream direction to about 0.7 mg/L during the low flow survey and to only about 0.2 mg/L during the high flow survey. During September 1990, when the background load was estimated to be about a factor of two higher than it was for the July 1987 and July 1990 surveys, the impact of the uniform load was more than 1 mg/L.

The deficit from all other sources, including upstream CBOD and deficit boundary conditions and tributary deficit and CBOD inputs, is shown on the bottom panels of Figure 5-32. Of these, the initial deficit near MP 240 is the most significant component of deficit during both surveys, at about 1.5 and 2.0 mg/L. This deficit can be attributed to SOD and distributed loads in the region upstream of the present study area. As such, it can alternatively be viewed as an additional contribution to the deficit associated with the SOD and uniform load shown on the second and third panels of this figure. In this regard, the deficits associated with SOD and W_{ul} do not actually start at zero at the upstream end of the model, as shown.

SECTION 6

WATER QUALITY DATA AND MODEL CALIBRATION ANALYSIS FOR HIGH STAGE CONDITIONS

A rather extensive data base, dating back to the 1950's (see e.g., Velz, 1962), has been developed for the purpose of monitoring water quality of the Ouachita River. These data were collected over both time and space and hence both seasonal and spatial trends have been well documented. Of particular interest is the marked impact of high stage conditions on water quality, and particularly dissolved oxygen levels in the Ouachita River. Since the principal objective of this study is to evaluate the impact of the Georgia Pacific discharge on dissolved oxygen, the data review presented in this section will focus on this water quality variable and on constituents having a direct effect on dissolved oxygen levels in the river. Emphasis will be placed on a review and analysis of high stage data obtained during the late 1970's and early 1980's, as more recent efforts, as described in Section 5 of this report, have focused on the analysis of low flow-low stage conditions. Since conditions have not changed significantly over time with regard to water quality during high stage periods these results are applicable to present day conditions. However, a recent 1990 high stage intensive survey data set will also be discussed.

This section begins with a review of data pertaining to high stage conditions (Section 6.1). This initial discussion is followed by a description of modeling analyses completed to investigate the factors controlling dissolved oxygen levels in the Ouachita River during these extreme periods (Section 6.2). A discussion of model sensitivity and components of deficit under high stage conditions is also presented (Section 6.3).

6.1 REVIEW OF DATA

Three general sources of data are reviewed to characterize water quality during high stage conditions. The first source of data is the routine water quality survey data (Appendix B) collected by Georgia Pacific personnel from the mill at Crossett, Arkansas for purposes of compliance monitoring. The second source of data consists of approximately 9 weeks of flood plain monitoring data obtained during 1980 (Appendix C1) for purposes of documenting conditions on the flood plain and within the main channel during periods of high stage conditions. Finally, intensive survey data (Appendices A1 and A4) were collected by Georgia Pacific personnel, in some instances in cooperation with HydroQual, Inc., to provide data to be used to extend the model developed for low stage conditions to the somewhat unique conditions occurring during high stage periods. Each of these data sets provides a different view of conditions during high stage periods and can be used to obtain an improved understanding of the interaction of flood plain waters with the main channel of the Ouachita River.

6.1.1 Routine Water Quality Monitoring Data

Georgia Pacific Corporation has been conducting routine water quality surveys on the Ouachita River since about 1970. These surveys were usually conducted between State Highway 82 in Arkansas and Sterlington, Louisiana (La MP 234.5- 189.5, or 1939 COE MP 250-205). The data include measurements of temperature, dissolved oxygen and color at stations located approximately every five miles along the Ouachita River. Prior to 1978, the surveys were usually performed once per week during the period of the year when the river was within its banks. For several years [illegible] the late 1970's, however, data were collected during both low and high stage conditions. These data, reviewed below, represent the most complete and nearly continuous set of data available for monitoring seasonal changes in dissolved oxygen in the Ouachita River during periods of high and low flow conditions.

Since as long ago as the mid-1950s it has been consistently observed that impaired dissolved oxygen levels are associated with flooded river conditions (see, Velz, 1962). In order to gain a better understanding of this relationship, the dissolved oxygen deficit and Ouachita River stage from the 1978-1979 and 1979-1980 water years have been plotted chronologically, as shown on Figure 6-1. (These data were previously presented on Figure 2-1.) The old Lock 6 stage is presented on the upper graph, rather than flow, because river flow is not reported when the river is out of its banks. Since zero stage corresponds to a mean water elevation (MWE) of 44.09 feet above mean sea level, the water surface

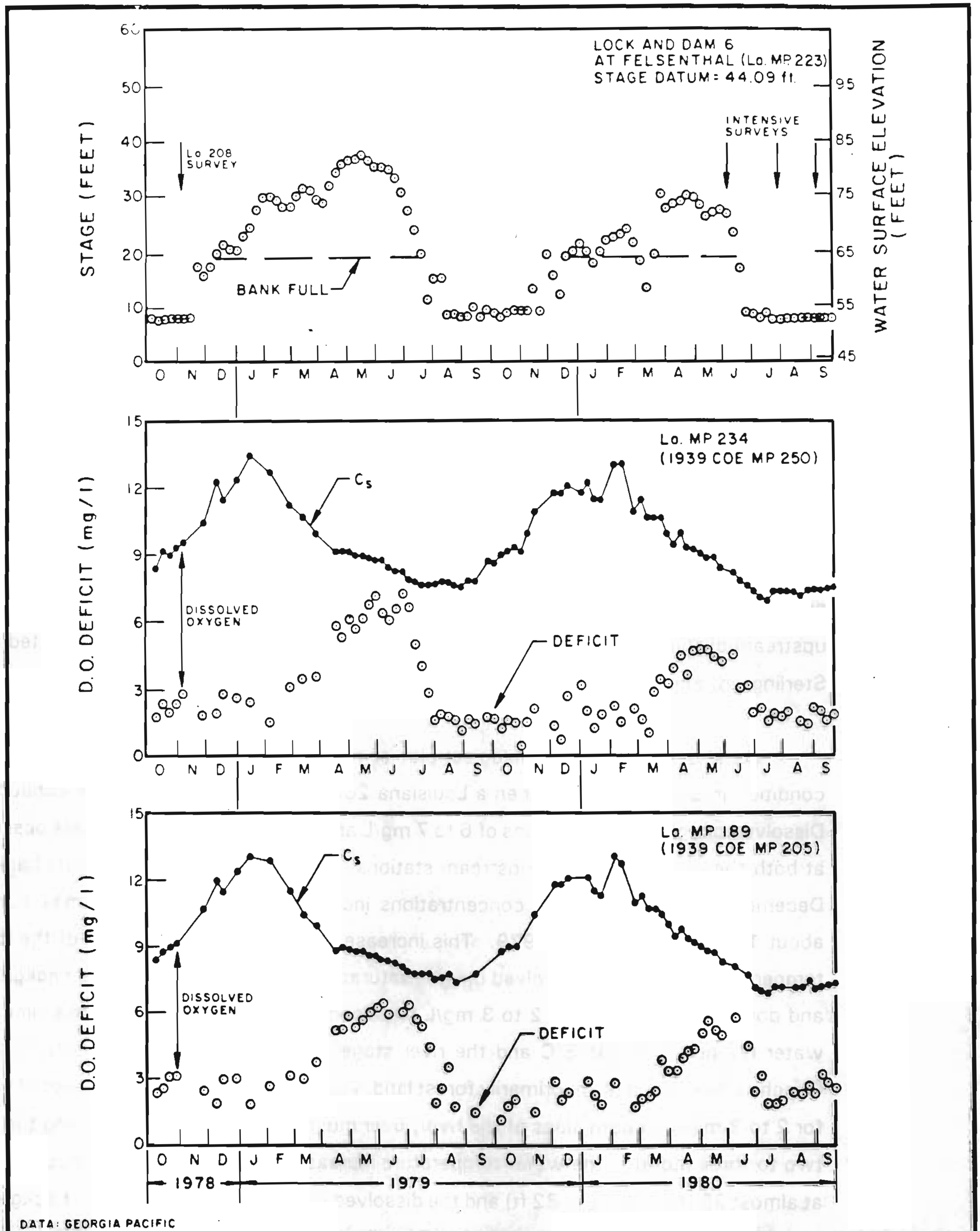


FIGURE 6-1. CHRONOLOGY OF OUACHITA RIVER STAGE AND DO DEFICIT, ROUTINE SURVEY DATA, 10/1/78 - 9/30/80

elevation may be obtained directly by adding the stage to this datum. Thus, the water surface elevation corresponding to the reported river stage is shown on the right axis of the upper graph. The lower pool stage, downstream of Lock 6, is approximately 8.0 feet (MWE = 52.09 ft) during low flow conditions of 1000 to 2000 cfs. The river is out of its banks, or "bank full," at a stage of about 19 feet (MWE = 63.09 ft) which corresponds to a flow of approximately 13,000 cfs.

The two lower graphs of Figure 6-1 show the dissolved oxygen saturation concentration and dissolved oxygen deficit at the upstream (middle panel) and downstream (lower panel) river reaches over which the routine surveys were performed. Dissolved oxygen deficit is the difference between the dissolved oxygen saturation concentration, the maximum that could exist in the river at any given temperature (in the absence of net photosynthesis) and the observed river dissolved oxygen concentration. The middle graph presents data collected at a background station, near Highway 82, more than 12 miles upstream of the Georgia Pacific discharge. The lower graph presents data collected near Sterlington, approximately 33 miles downstream of the Georgia Pacific discharge.

As shown on the chronological plot of river stage, the river was at a low stage condition in October 1978, when a Louisiana 208 water quality survey was conducted. Dissolved oxygen concentrations of 6 to 7 mg/L and deficits of 2 to 3 mg/L were observed at both the upstream and downstream stations. After the river overflowed its banks in December, dissolved oxygen concentrations increased steadily toward a maximum of about 11 mg/L in February 1979. This increase was primarily a reflection of the lower temperatures and higher dissolved oxygen saturation concentration, since the background and downstream deficits of 2 to 3 mg/L remained relatively constant. At this time, the water temperature was 3°C and the river stage was 31 feet (MWE = 75 feet). The Ouachita River flood plain, primarily forest land, was inundated with 10 to 15 feet of water for 2 to 3 miles on both sides of the river, over most of the survey area. During the next two to three months, the water temperature increased steadily. The river stage peaked at almost 38 feet (MWE = 82 ft) and the dissolved oxygen deficit, at both the background and downstream stations, increased to 7 mg/L. With the accompanying decrease in the saturation concentration, minimum dissolved oxygen concentrations of 1.0 and 1.6 mg/L were reported at the background and downstream stations respectively.

It was not until the middle of June that the flood waters began to recede. At that time, dissolved oxygen levels in the river had been as low as 1 to 2 mg/L throughout much of the river, for much of the spawning season, with deficits of 6 to 7 mg/L having been sustained for a period of 12 weeks. Hence, it is apparent that the depressed dissolved oxygen levels cannot be attributed to the effects of the receding flood waters. To the contrary, as the flood waters receded, the deficits responded immediately by decreasing to 2 mg/L, as observed during the period of time preceding the 1978 through 1979 flooding. The river was within its banks by mid-July, and shortly thereafter the dissolved oxygen concentration recovered from a minimum of 1 mg/L at low temperature and high flow conditions to about 5 to 6 mg/L, even though the flow was much lower and the water temperature had increased to 27°C.

It should be noted that although the 1978-1979 flood represented a relatively extreme level of flooding, similar and even more extreme flood conditions have been experienced in recent years. With regard to 1978-79, the water surface elevation approached 82 feet above mean sea level, and the onset of flooding began in the vicinity of MP 265 to 270, 30 to 35 miles upstream of the first routine survey sampling station. More recently, during 1990, the maximum water surface elevation was closer to 90 feet, approximately equal to the highest level on record, and flood waters did not recede until mid-July.

Inspection of Figure 6-1 for the 1979-80 water year shows a very similar if not quite as dramatic pattern of events occurred as the river flooded and receded. During this water year, the water surface elevation rose to about 76 feet and the limits of the flooding extended only as far upstream as MP 255, 15 miles upstream of the Saline River. A review of data collected from 1970 through 1977 suggests that similar conditions occurred whenever the river flooded. Although surveys were not usually performed when the river was flooded during these earlier years, observed deficits during the first two to three weeks after the flood waters receded consistently showed a decreasing trend (i.e., increasing dissolved oxygen).

The spatial profile of dissolved oxygen during selected periods of time during 1979 are shown on Figure 6-2. Four time intervals, labeled "a" through "d" on the upper chronological plot of river stage, have been selected to illustrate the dissolved oxygen profile of the river under different river stage, temperature and flow conditions. The averages of the weekly monitoring dissolved oxygen data are presented as spatial plots for each of these periods on the four lower panels. During period (a), the river was near its maximum 1979 stage at an estimated flow of 50,000 cfs and the average water temperature of 20°C corresponds to a dissolved oxygen saturation concentration of 9 mg/L. Background dissolved oxygen levels averaged 3 to 4 mg/L throughout the 12 mile reach upstream of Coffee Creek. Although slightly lower average dissolved oxygen levels did occur downstream, it is apparent that the rather large deficit of approximately 6 mg/L was dominated by the upstream conditions. Over time interval (b), just prior to when the flood waters receded, similar conditions occurred. Here, dissolved oxygen levels were generally less than 2 mg/L. These first two spatial profiles, from late April to early May and from the latter half of June, are typical of the water quality which persisted in the river throughout the spawning season during this year.

Time interval (c) took place shortly after the river was back within its banks. Although the water temperature of 28°C was higher (dissolved oxygen saturation concentration = 7.8 mg/L) and river flow lower (1,875 cfs), average dissolved oxygen concentrations of 5 to 6 mg/L represented a marked improvement relative to the preceding time interval. The average dissolved oxygen deficit was about 1.8 mg/L upstream of Lock and Dam 6 and 2.6 mg/L in the vicinity of Louisiana MP 195. Finally, spatial profile (d) illustrates the dissolved oxygen profile at a flow of 6850 cfs and a temperature of 12°C, as observed on November 15, 1979. Here, the spatial profile was again uniform, with dissolved oxygen concentrations of about 9 mg/L and deficits of 1 to 2 mg/L throughout the study area.

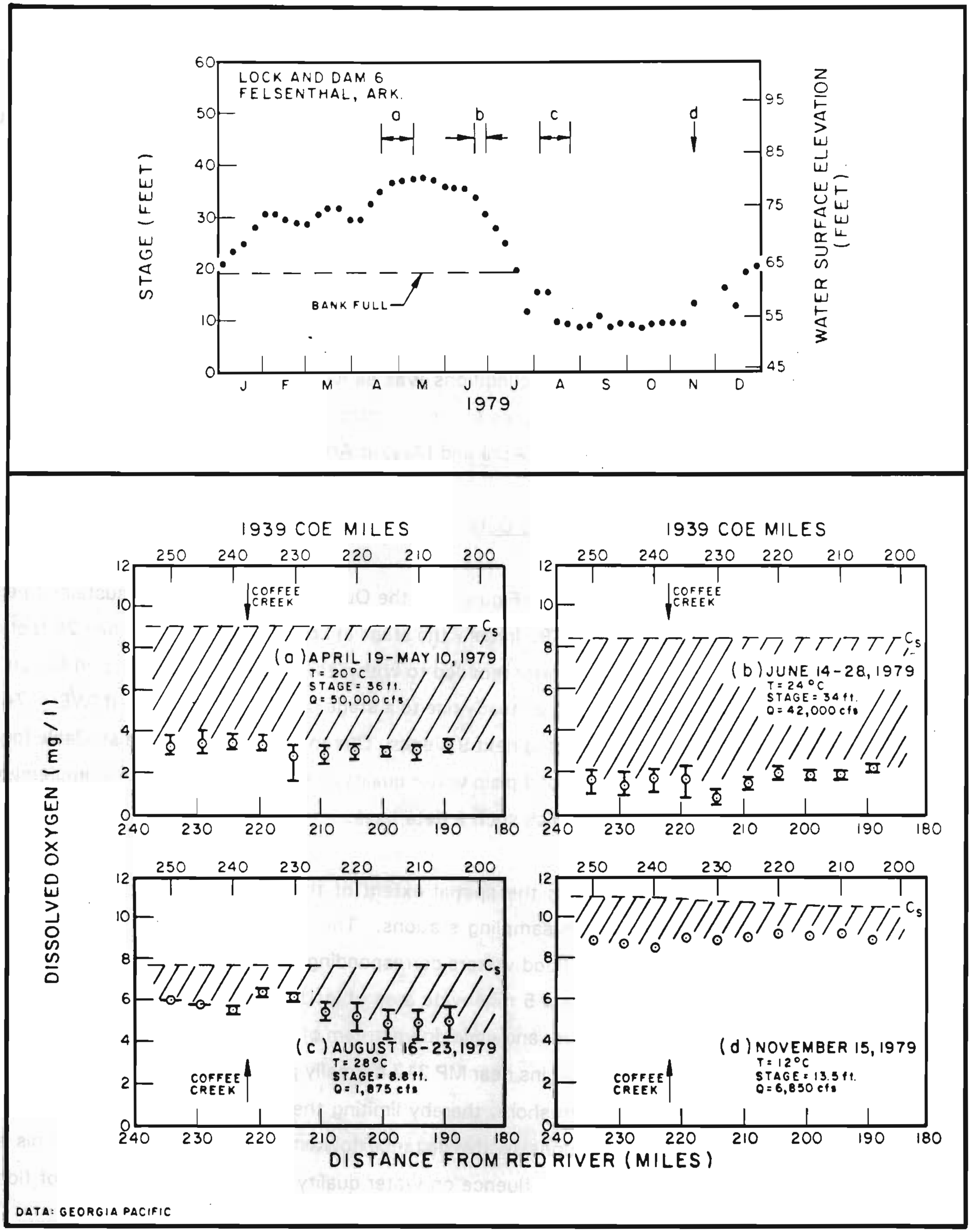


FIGURE 6-2. SPATIAL PROFILES OF OUACHITA RIVER DISSOLVED OXYGEN LEVELS, ROUTINE SURVEY DATA, SELECTED PERIODS DURING 1979

The preceding review of the routine survey data illustrates several important points. First, during the period of time when the river was within its banks, the background deficit in the vicinity of MP 234 was typically 2 mg/L. Second, when the river was flooded, a background deficit as high as 6 to 7 mg/L was observed a considerable distance upstream of Georgia Pacific's discharge, and this deficit propagated throughout the study area. The high background deficit was generally observed after a period of sustained flood conditions and usually dissipated as the flood water receded to the main channel. The dissolved oxygen profile during flooded conditions was as low as 1 to 2 mg/L, and for extended periods of time, lasting as long as several months, the dissolved oxygen standard of 5 mg/L (6.5 mg/L during March, April and May, in Arkansas) was not achieved.

6.1.2 1980 Flood Plain Survey Data

As shown previously on Figure 6-1, the Ouachita River entered a sustained period of flooding in December 1979. Initially the stage at Lock 6 remained less than 25 feet and on several occasions, the water receded to within the river banks. Finally, on March 11, 1980, the water level began a steady rise to a stage of more than 30 feet (MWE = 74.09 feet), where it remained for the next 9 weeks. Due to the paucity of data available for the purpose of characterizing flood plain water quality, a sampling program was implemented on April 22, 1980 to establish such a data base.

Figure 6-3 illustrates the spatial extent of the flood plain and the approximate location of the flood plain sampling stations. The 75 foot contour line represents the approximate limits of the flood waters corresponding to a 30 foot stage. As shown, the flooded forest land covers a 5 mile wide area of land which begins about 15 river miles upstream of the Saline River and ends downstream of Alabama Landing, in the vicinity of MP 210. A levee which begins near MP 217 normally prevents the river from flooding the bean fields on the eastern shore, thereby limiting the eastern flood plain to a relatively narrow strip of land for a considerable distance downstream from this location. This levee is also believed to have an influence on water quality because during periods of flooded

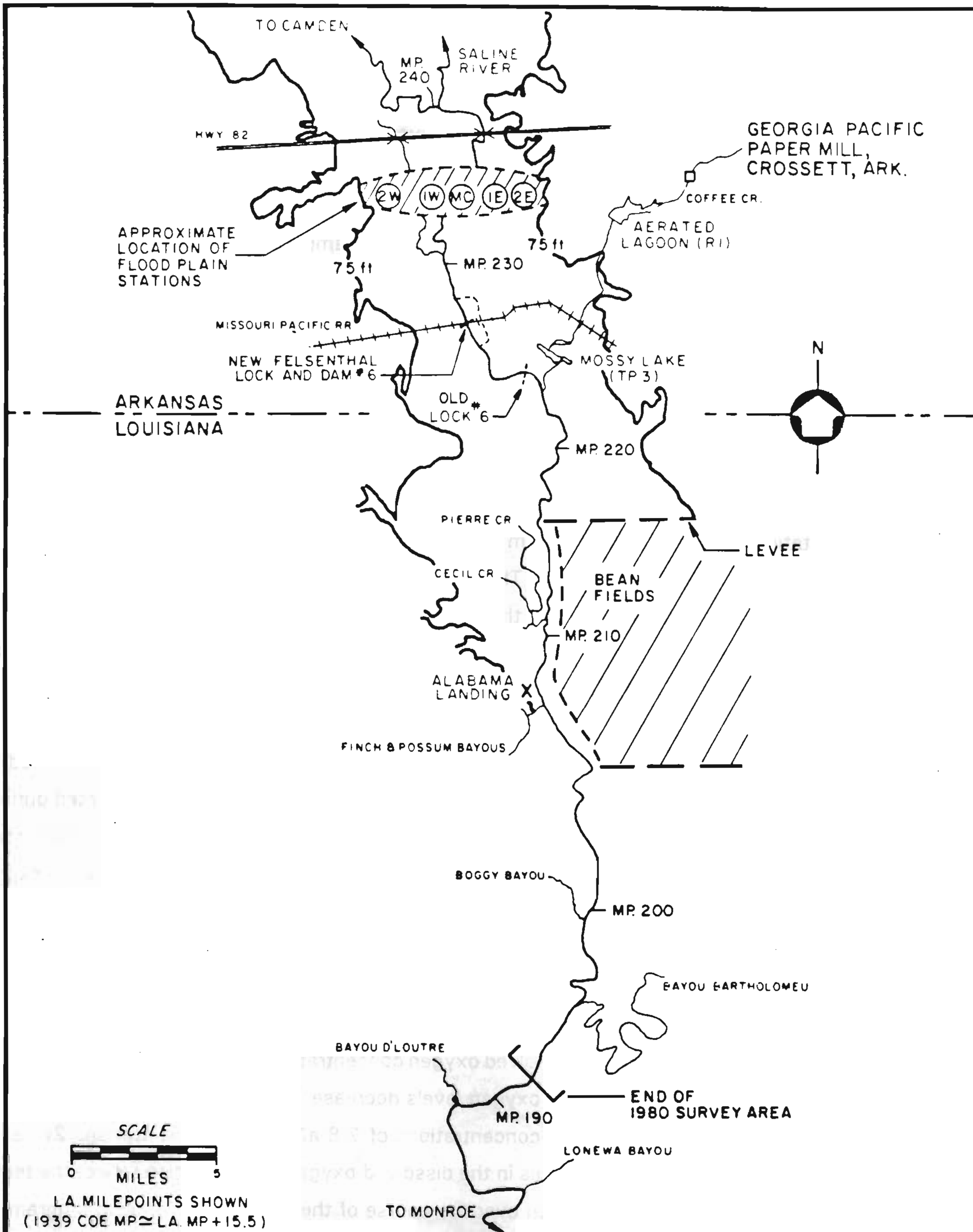


FIGURE 6-3. LOCATION OF FLOOD PLAIN SAMPLING STATIONS

conditions it controls the location where water from the flood plain is diverted back to the main channel.

As shown on Figure 6-3 the flood plain sampling stations were located along an east-west transect which crossed the main channel of the Ouachita River about 10 to 12 river miles upstream of Coffee Creek. Stations were located approximately one and two miles away from the main channel, on both the east (Stations 1E and 2E) and west (Stations 1W and 2W) sides of the river. These stations, as well as a main channel station (MC) located near MP 234 were usually sampled once per week from April 22, 1980, 6 weeks after the river was last within its banks, until the water receded from the flood plain in the latter part of June. Temperature and disso[illegible]easured at each station, and surface and bottom composite samples were analyzed by Georgia Pacific for pH, BOD_5, COD and color. The ultimate carbonaceous BOD was measured on split samples that were shipped to the HydroQual laboratory, General Testing Corporation, in Hackensack, New Jersey. The CBOD was determined from long term BOD's incubated both with and without nitrification suppression with TCMP.

Spatial plots of the BOD_5 and dissolved oxygen profiles along the flood plain transect are presented on Figure 6-4. The average and range of data collected during the eight week period of the flood plain sampling program is shown for each station. Observed BOD_5 levels of 1 to 3 mg/L were representative of natural occurring background concentrations and tended to be somewhat higher with increasing distance from the main channel. Station 2W, located on the western side of the flood plain and furthest from Georgia Pacific, had the highest average BOD_5 concentration of almost 2.5 mg/L. The dissolved oxygen profile, shown on the lower graph of Figure 6-4, had the opposite shape, with the highest average dissolved oxygen concentration of 4.5 mg/L occurring at the main channel station. Dissolved oxygen levels decrease in the direction of the fringes of the flood plain, having average concentrations of 2.8 and 3.5 mg/L at stations 2W and 2E, respectively. The wide ranges in the dissolved oxygen concentration reflect the temporal decrease in dissolved oxygen over the course of the flood plain sampling program. One

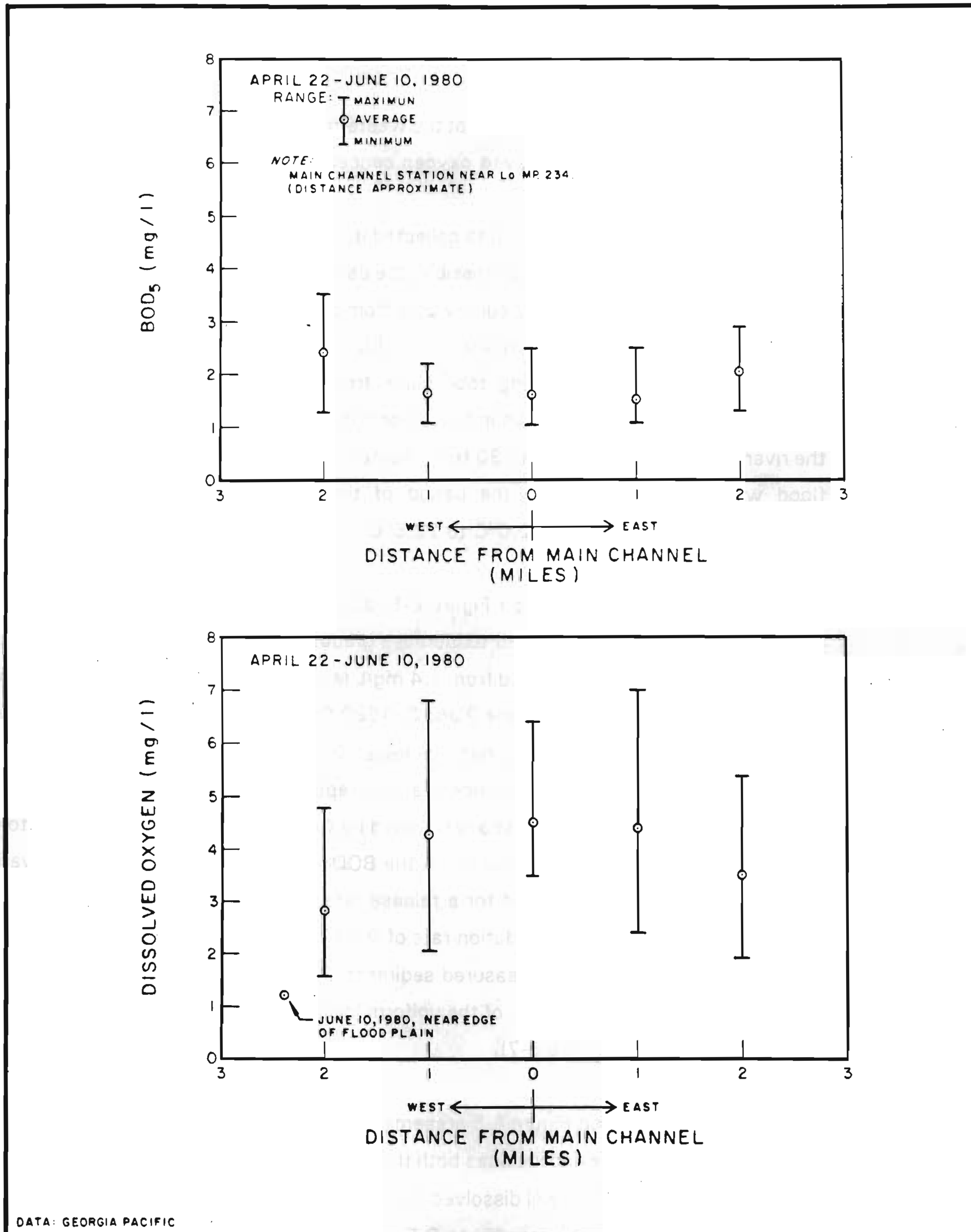


FIGURE 6-4. SPATIAL PROFILES OF BOD_5 AND DISSOLVED OXYGEN ALONG EAST - WEST FLOOD PLAIN TRANSECT

additional measurement of 1.2 mg/L, at the western edge of the flood plain, represents the minimum depth averaged dissolved oxygen concentration observed.

The temporal variation of data collected during the flood plain sampling program is summarized on Figure 6-5. When possible, the data are supplemented with routine survey data and intensive water quality survey data from the Ouachita River. The horizontal axis shows the duration of flooding, referenced to March 11, 1980, when the river overflowed its banks. Flood plain sampling took place from 6 to 13 weeks after the river was experiencing flood conditions, as indicated on the graph of river stage. During this time, the river stage was usually 28 to 30 feet. Sampling was necessarily terminated when the flood waters receded. Over the period of time shown on the graphs, the water temperature increased from 12.0°C to 23.5°C.

The BOD_5 data shown on Figure 6-5, although quite variable relative to the low concentrations measured, tended to increase gradually throughout most of the sampling period. Concentrations increased from 1.4 mg/L (average of all stations) in the sixth week to 2.1 mg/L at the time of the June 2 and 3, 1980 Ouachita River survey. Thirteen weeks after the initial flooding of the river, a lower BOD_5 concentration of 1.3 mg/L was measured. The average BOD_5 concentrations reported by General Testing are in close agreement with BOD_5 concentrations reported by Georgia Pacific. It is of interest to note that the curves shown superimposed on the BOD_5 data correspond to the time variable buildup that would be expected for a release rate from the flooded soils of 1.4 to 2.0 grams $CBOD_u/m^2$/day at an oxidation rate of 0.05/day at 20°C and f = 4. These release rates are comparable to the measured sediment TOC fluxes when converted to $CBOD_u$ (Table 5-4) and within the range of the uniform loads used in the Ouachita River model for low stage conditions (Table 5-7).

The final graph on Figure 6-5 presents the change in the average dissolved oxygen concentration with time and includes both the flood plain data and routine river survey data at MP 234. The main channel dissolved oxygen concentration was 9.5 mg/L at the onset of flooding, but decreased steadily to 3.5 mg/L (open circles). The average flood plain

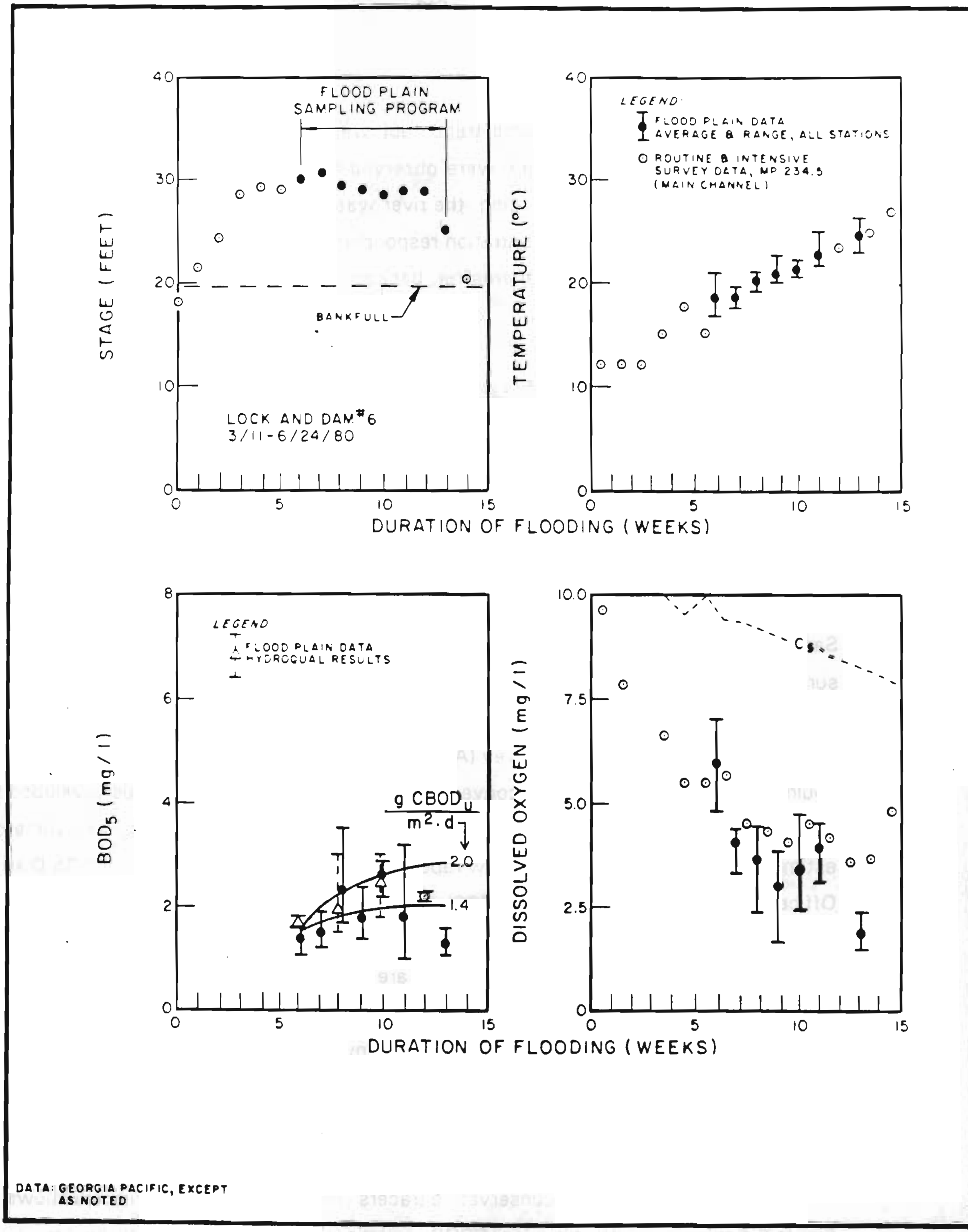


FIGURE 6-5. TEMPORAL VARIATION OF FLOOD PLAIN SURVEY DATA, MARCH 11 THROUGH JUNE 24, 1980

concentrations followed the same trend, but were consistently lower (filled circles). Average deficits of about 5 mg/L were observed during this period of time. Fourteen weeks after the initiation of flooding, the river was back within its banks, and the main channel dissolved oxygen concentration responded by increasing to 4.8 mg/L in slightly more than one week. Shortly thereafter, background deficits were once again about 2 mg/L in the vicinity of MP 234.

6.1.3 Intensive Water Quality Survey Data

On June 3, 1980 and July 7, 1990, periods of high stage conditions, intensive water quality surveys were conducted on the Ouachita River between the Saline River and Bayou Bartholomew. The surveys were performed for the purpose of obtaining data to calibrate the mathematical model of the Ouachita River for high stage periods and to define the impact of the Georgia Pacific discharge relative to background loads at those times. Sampling and analytical requirements were similar to the schedule for the low stage surveys, as described previously.

During the June 1980 survey (Appendix A1), the river was flooded and river flow could not be estimated using the conventional stage-discharge relationships developed for low flow conditions. The flow at the Arkansas-Louisiana state line was therefore estimated to be 77 percent of the flow reported at Columbia Lock and Dam (USGS District Office, Baton Rouge, Louisiana, 1980). Thus, during the June survey the flow was approximately 30,000 cfs. Since Mossy Lake was covered by floodwater, the treatment system flow and loads for the June survey are based on the discharge from the aerated lagoon at test point R1. The 10 day average effluent flow as well as the flow during the survey were approximately 50 mgd, and the flow preceding the survey was relatively constant in time. The $CBOD_u$ load at R1 was 31,900 lbs/day (76.5 mg/L) at the time of the June survey.

Spatial profiles of the conservative tracers chloride and conductivity are shown on Figure 6-6, along with the nitrogen, $CBOD_u$ and dissolved oxygen data, for the June 1980

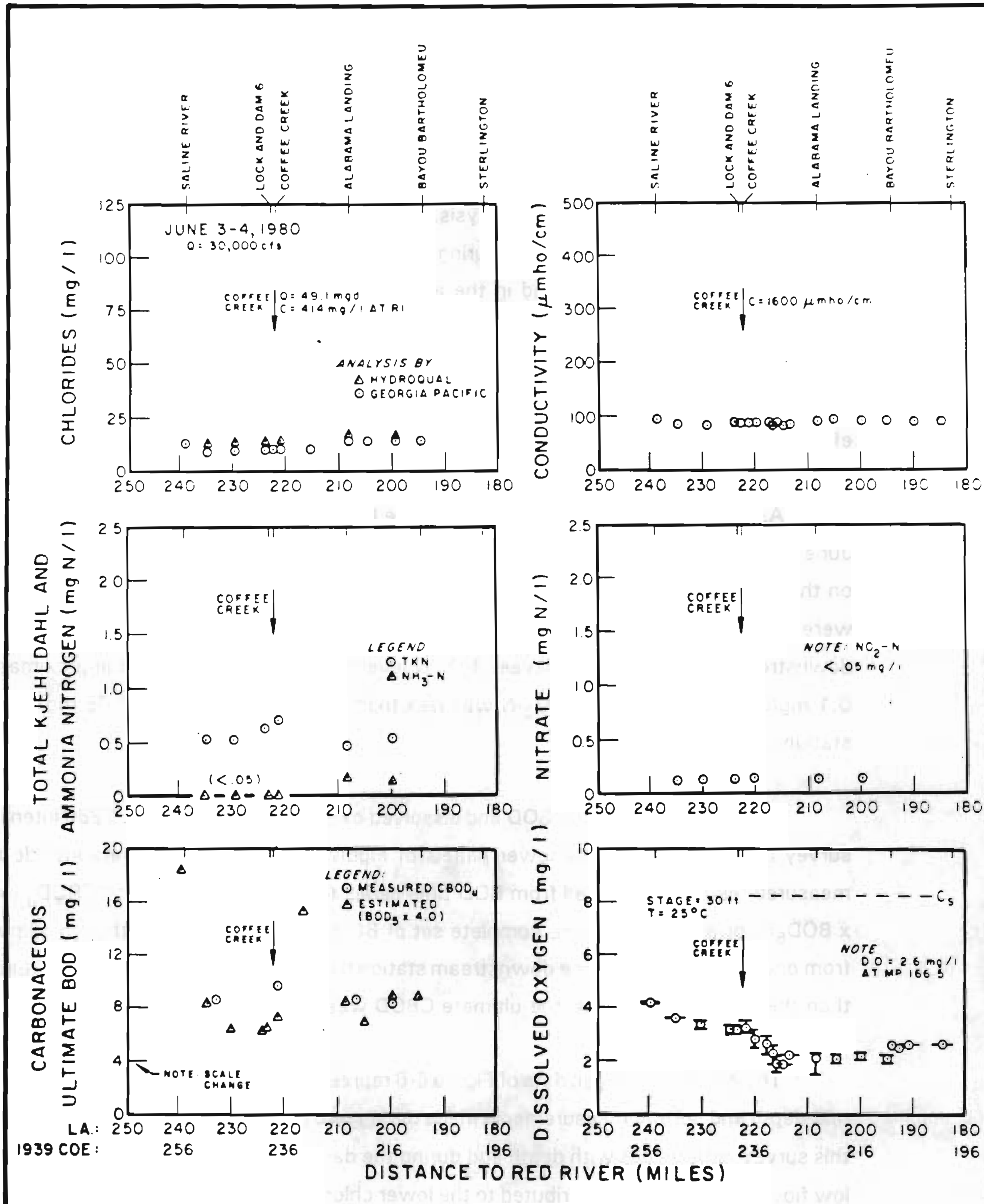


FIGURE 6-6. SPATIAL PROFILES OF HIGH STAGE WATER QUALITY SURVEY DATA, JUNE 3-4, 1980

survey. The tracer data were previously compared to model results for this high stage survey on Figure 4-5. These conservative substances were used to verify the flow distribution used in the modeling analysis. The exact location where the Georgia Pacific discharge enters the main channel during flooded conditions is not known and thus the location of Coffee Creek is used in the absence of more precise information. As was shown, both constituents had very uniform profiles from the Saline River to Sterlington. The calculated chloride and conductivity profiles were generally in agreement with this due to the extremely high river flow, which provided approximately 400 to 1 dilution of the mill effluent.

As shown on the middle panels of Figure 6-6 the nitrogen data collected during the June water quality survey also had relatively uniform spatial profiles. TKN levels, shown on the left, were 0.5 to 0.7 mg/L throughout the study area while NH_3-N concentrations were below the detection limit upstream of th [illegible] y 0.1 to 0.2 mg/L downstream of the bean field levee. NO_3-N levels were also quite low, at approximately 0.1 mg/L at all stations and NO_2-N was less than the detection limit of 0.05 mg/L at all stations.

Ultimate carbonaceous BOD and dissolved oxygen data for the June 1980 intensive survey are illustrated on the lower panels of Figure 6-6. The $CBOD_u$ data include the measured results determined from BOD time series tests and the estimated $CBOD_u$ (= 4 x BOD_5), for a separate, more complete set of BOD_5 measurements. Although samples from one background and one downstream station had considerably higher concentrations than the remaining stations, the ultimate CBOD was typically 6 to 9 mg/L.

The dissolved oxygen data of Figure 6-6 represent the average and range of surface, mid-depth and bottom measurements made during two passes over the study area. During this survey, differences with depth and during the day were small in comparison to typical low flow surveys. This is attributed to the lower chlorophyll-a levels, increased depth, and higher level of turbulent mixing which characteristically occur during high stage periods. The dissolved oxygen concentration was generally less than 4 mg/L, decreasing in the downstream direction from a maximum of 4.3 mg/L at the Saline River to a minimum of about 2 mg/L downstream of MP 215. This gradient is inconsistent with the 1979 profiles presented previously, on Figure 6-2. The difference can be attributed to the difference in the river stage, as discussed below.

There was no reaeration at Lock and Dam 6 during this and other high stage surveys because the upper and lower pools are at the same level and the crest of the dam is actually beneath the water surface (12 feet under water for the June 1980 survey). Downstream of MP 215 the dissolved oxygen profile is almost constant, with a step increase to 2.6 mg/L at the confluence with Bayou Bartholomew. Although not shown on this graph, a dissolved oxygen concentration of 2.6 mg/L was also measured at MP 166.5, at the railroad bridge in Monroe.

Similar water quality data were reported for the July 1990 high stage intensive survey. These data are included in Appendix A4. BOD and dissolved oxygen data for this high stage survey will be described with the discussion of the model, presented subsequently.

6.2 WATER QUALITY MODELING ANALYSIS FOR HIGH STAGE CONDITIONS

A number of simplifying assumptions are required in order to apply the water quality model of the Ouachita River, developed under low stage conditions, to flooded conditions. The key difference is that it is postulated that dissolved oxygen profiles observed during flooded river conditions are a result of an exchange of river water with flood plain water having a relatively low dissolved oxygen concentration. The model was originally applied to the June 1980 intensive survey and to routine survey data from 1978 in order to test this hypothesis. More recently, it was used to analyze the July 1990 intensive water quality survey data set.

6.2.1 Representation of Channel Geometry and Hydrology for High Stage Conditions

To apply the water quality model to high stage conditions it is necessary to define the geometry of the main channel. Average depths for each reach were estimated by adding the difference in river stage between low flow and high flow conditions to the average depth at the low flow condition. It was also assumed that most of the river flow was restricted to the area between the banks of the main channel, even when the water surface was 10 to 20 feet higher than the top of the river bank. The rationale for this latter assumption is that the flood plain is generally forested land and the river flow tends to follow the path of least resistance, i.e., the open pathway formed by trees along the banks of the main channel. Finally, the ratio of the cr[illegible] areas during flooded and low flow conditions was taken to be the same as the corresponding ratio of the average depths.

Using the preceding channel geometry as a first approximation and assigning the BOD removal rate coefficients determined from the low flow analysis, it was not possible to obtain agreement between the calculated model profile and the observed dissolved oxygen data. For example, for the June 1980 data set (lower right panel of Figure 6-6), the calculated dissolved oxygen profile remains almost constant, equal to the initial condition of 4.3 mg/L, while the observed data decrease to about 2 mg/L and then remain constant throughout the study area.

The discrepancy between these preliminary model results and the observed data at high flow conditions was postulated to be a result of an exchange of river water with the relatively low dissolved oxygen water of the flood plain. This exchange occurs across the interfacial area between the top of the river bank and the water surface when the river is at flood stage. A schematic diagram showing the elevation of the top of the bank of the main channel relative to the water surface elevations during low flow and flooded conditions is presented on Figure 6-7 (pre-1984 conditions are shown). During low stage periods the water surface elevation is below the top of the river bank and the wate surface elevation of the lower pool (downstream of Lock 6) is about 9 feet less than that

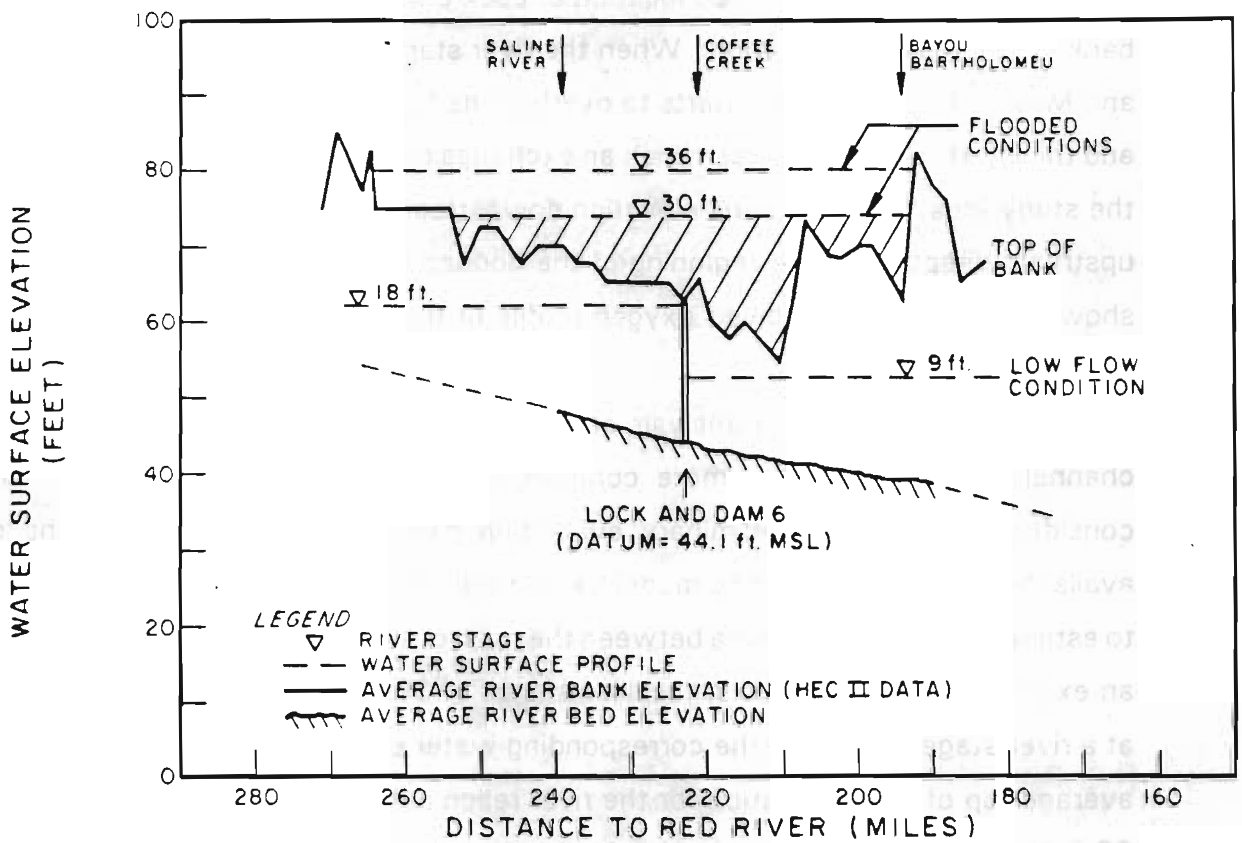


FIGURE 6-7. COMPARISON OF RIVER BANK ELEVATION AND WATER SURFACE PROFILES

of the upper pool (about 12.5 feet lower since 1984). At a river stage of 30 feet, as occurred at the time of the June 1980 survey, the water surface elevation is considerably higher than the top of the river bank throughout most of the study area. The shaded area of Figure 6-7 represents the average area available for exchange of water between the river and the flood plain, on each side of the main channel. Note that this area increases significantly between the 1980 location of Lock 6 and MP 210, where the top of the river bank is at its lowest elevation. When the river stage is 36 feet, as occurred during April and May of 1979, the river starts to overflow its banks 10 to 15 miles further upstream, and the interfacial area across which an exchange of water occurs is increased throughout the study area. The low bank elevation downstream of Lock 6 and the movement in the upstream direction of the beginning of the flooded region as stage increases will both be shown to affect the dissolved oxygen profile of the river.

Although a rigorous analysis of the interaction between the flood plain and main channel requires use of a more complex hydrodynamic model and would involve a considerable effort, a preliminary evaluation has been performed with the currently available data and the existing model framework. Table 6-1 summarizes the method used to estimate the interfacial area between the main channel and the flood plain, across which an exchange of water occurs, for river stages of 30, 36 and 23.8 feet. As an example, at a river stage of 30 feet the corresponding water surface elevation is 74.1 feet and the average top of bank elevation for the river reach between the Saline River and Lock 6 is 66.5 feet. Thus, the average depth of water over the bank of the river is the water surface elevation minus the bank elevation, or 7.6 feet. It is across this interface, on both sides of the main channel, that water from the river and flood plain are exchanged. Given this depth, the interfacial area of 80,300 ft^2/mile between the main channel and flood plain is directly determined. Similarly, the interfacial area of 153,100 ft^2/mile was estimated for the river reach between Lock 6 and MP 210. The interfacial area almost doubles in this second reach due to the lower elevation of the top of the river bank.

The transport mechanism of an exchange of water between the main channel and flood plain, in terms of cfs per unit area, was incorporated into the analytical solution for the dissolved oxygen deficit used in program river. Although it is recognized that the methodology employed represents a significant simplification of the mode of transport actually taking place, it will be shown that it provides a reasonable basis for explaining the manner in which flood plain water quality affects dissolved oxygen levels in the main channel.

TABLE 6-1. SUMMARY OF AREA AVAILABLE FOR EXCHANGE BETWEEN MAIN CHANNEL AND FLOOD PLAIN

River Stage[a] (ft)	Water Surface Elevation (ft)	River Reach (MP-MP)	Average Bank Elevation (ft)	Average Depth at Bank (ft)	Area for Exchange (1,000 sq ft/mi)
30	74.1	239-223	66.5	7.6	80.3
		223-210	59.6	14.5	153.1
36	80.1	264-239	73.1	7.0	73.9
		239-223	66.5	13.6	143.6
		223-210	59.6	20.5	216.5
23.8	67.9	239-227	65.9	2.0	21.1
		227-216	61.1	6.8	71.8
		216-208	61.1	6.8	71.8

[a]Datum = 44.09 ft

6.2.2 Model Application to High Stage Survey Data

6.2.2.1 June 3-4, 1980 Intensive Survey

Since the requisite data were not available to obtain an independent estimate of the rate of exchange between the flood plain and main channel, it was necessary to assign a rate of exchange on the basis of the fit of the dissolved oxygen profile. An average flood plain deficit of 6.3 mg/L, as measured about one week after the June 1980 intensive survey, was used to define the flood plain water quality. The calculated dissolved oxygen profile is compared to the observed data for the June 1980 survey on the upper graph of

Figure 6-8. Here, a rate of exchange of 20 cfs/1000 ft^2 has been used to fit the data. The model results are in excellent agreement with the data.

The relatively sharp rate of decrease of the dissolved oxygen concentration in the vicinity of MP 216 is partially accounted for by the increased interfacial area available for flood plain exchange, downstream of the old Lock and Dam 6 location. Although not incorporated in the model result shown, this decrease could also be associated with the influence of the bean field levee, which may be diverting low dissolved oxygen flood plain water back to the main channel in this same area. This possibility has yet to be confirmed.

Further downstream, by about MP 208, spatial steady state is approached and the profile levels off. The increase of 0.6 mg/L at MP 194 is due to Bayou Bartholomew, which had a dissolved oxygen concentration of 5.2 mg/L. The resulting deficit propagates downstream at least as far as MP 167 at Monroe, due to the short travel time and low reaeration rate associated with the increased main channel depths that occur during flooded conditions.

6.2.2.2 April 19 to May 10, 1979 Routine Monitoring Data

The routine survey data of April 19 to May 10, 1979, presented previously on Figure 6-2, was originally used to verify the approach used for modeling the flooded river conditions. During this period of time the river stage was 36 feet (MWE = 76.09 feet) and the region of flooding began approximately 15 miles further upstream than in June 1980. The dissolved oxygen profile was relatively constant throughout the study area at this higher stage. Assigning the same initial dissolved oxygen deficit of 4.0 mg/L at the beginning of the flooded region (MP 264), the same flood plain deficit of 6.3 mg/L, and a rate of exchange of 20 cfs/1000 ft^2 yields the calculated profile shown on the lower graph of Figure 6-8. The results are in good agreement with the routine survey data and indicate that spatial steady state is approached further upstream. This explains why the dissolved oxygen profile downstream of the Saline River tends to be flatter as the river stage

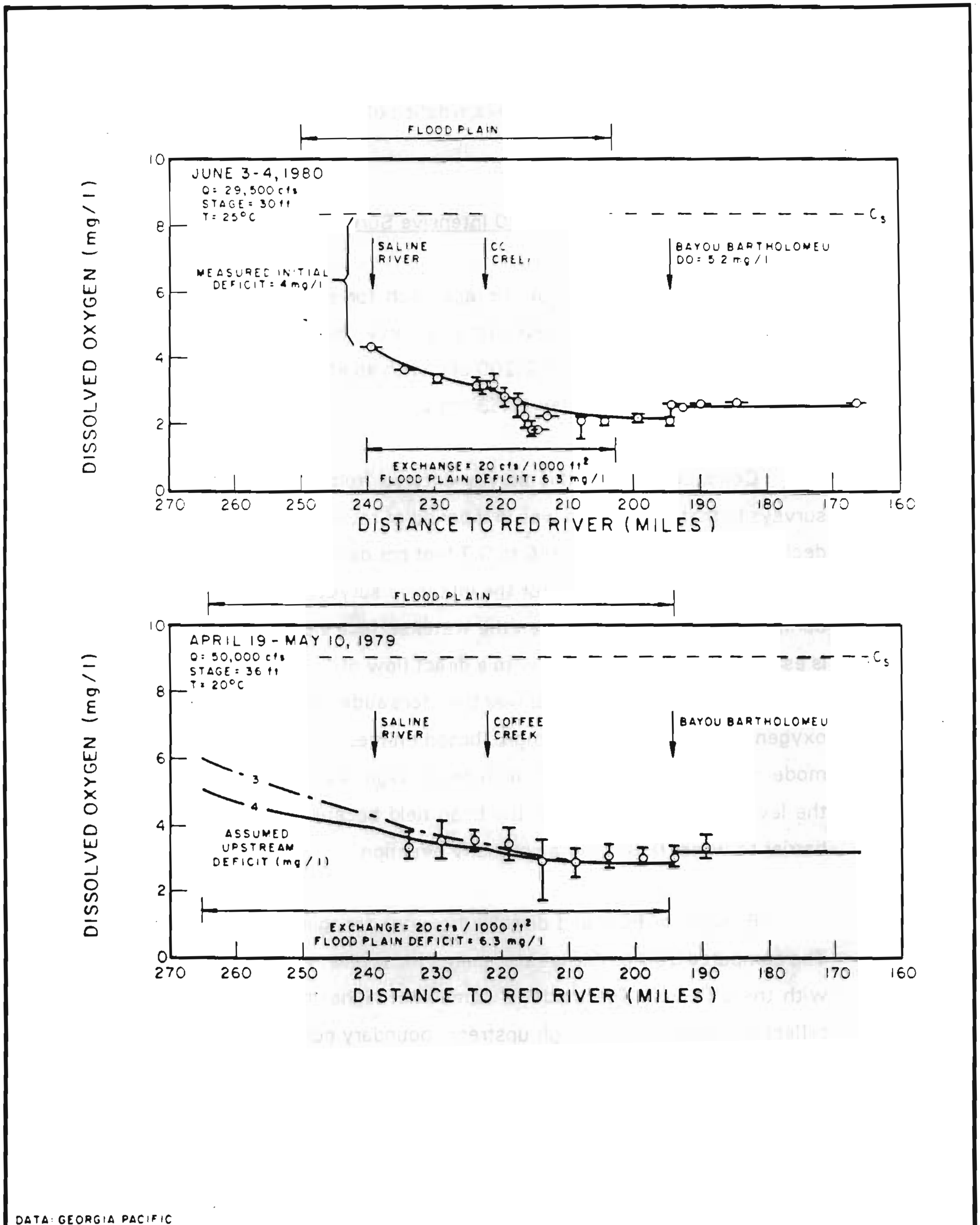


FIGURE 6-8. HIGH STAGE DO MODEL CALIBRATION RESULTS, JUNE 3-4, 1980 AND APRIL 19 - MAY 10, 1979

increases. A lower assumed upstream deficit of 3 mg/L results in similarly good agreement with the routine survey data.

6.2.2.3 July 7, 1990 Intensive Survey

A more recent test of the approach for simulating high stage conditions was performed using the July 1990 intensive survey data set. At the time of this survey the river flow was estimated at 22,200 cfs, with an effluent flow of 45.2 mgd and a $CBOD_u$ load at R1 of 29,050 lbs/day (77.3 mg/L).

Conditions during this survey differed from the previously discussed high stage surveys in that the system was in a period of receding flood waters, with the water level declining at a rate of about 0.6 to 0.7 feet per day. On July 1 the stage was at 71.4 feet and on the day of sampling for the intensive survey, July 7, it was at 67.9 feet. Unlike during high stage periods when the water surface elevation is constant, this fall in stage is estimated to be associated with a direct flow of flood plain water to the river averaging about 900 cfs/mile. This water was therefore added to the river with an assigned dissolved oxygen deficit of 5.7 to 6.7 mg/L, based on mea [illegible] ts, as part of the model simulation. A relatively high percentage (40 percent) was added in the vicinity of the levee at the north end of the bean field because this structure presents a physical barrier to water flowing in a southerly direction.

Results for BOD and dissolved oxygen for this analysis are shown on Figure 6-9. The computed BOD profile is essentially a flat profile, with no significant impact associated with the effluent. The model is somewhat higher than the data, but this may simply reflect use of a relatively high upstream boundary condition. At the downstream sample points the model is about equal to observed concentrations. Within the $CBOD_u$ range of 5 to 7 mg/L much of the variability in the data may be related to test precision.

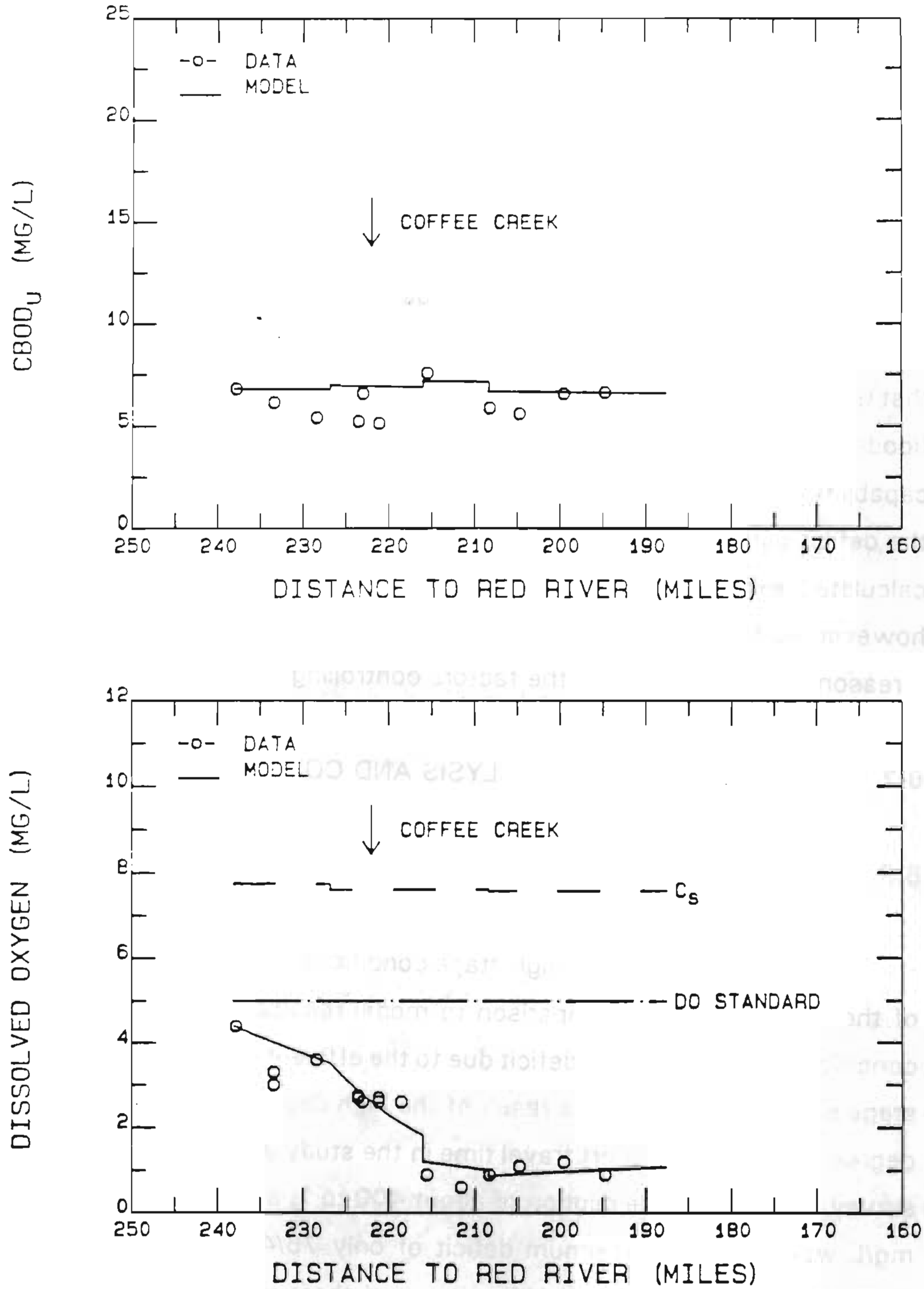


FIGURE 6-9. HIGH STAGE DO MODEL CALIBRATION RESULTS,
JULY 7, 1990 WATER QUALITY SURVEY

With regard to the dissolved oxygen profile, shown on the lower panel, the model profile is very similar to the data. The decrease near MP 216, at the bean field levee, is well represented, and the downstream profile of about 1 mg/L is also reproduced.

The modified approach used for this survey, in which a flow from the flood plain to the river is used in addition to an exchange of water between the flood plain and the river, is justified in light of the falling stage that was observed. It is acknowledged, however, that use of a hydrodynamic model to define the detailed transport of water throughout the flooded region would be more mechanistically realistic. Additionally, the predictive capability of the model would be enhanced if the flood plain itself were modeled such that the deficit currently assigned to the flood plain water exchanged with the river could be calculated explicitly as part of the simulation. These refinements are not essential, however, as the present representation of main channel-flood plain interactions provides a reasonable explanation of the factors controlling current conditions.

6.3 MODEL SENSITIVITY ANALYSIS AND COMPONENTS OF DEFICIT

6.3.1 Impact of the Georgia Pacific Load

The model results for high stage conditions tend to be relatively insensitive to most of the model inputs in comparison to model results discussed previously for low stage conditions. The calculated deficit due to the effluent is generally negligible during the high stage surveys, primarily as a result of the high degree of available dilution and to a lesser degree, the relatively short travel time in the study area. For example, for the June 1980 survey, at the available dilution of about 400 to 1, an ultimate CBOD concentration of 75 mg/L would have a maximum deficit of only 75/400 = 0.19 mg/L if it was oxidized instantaneously as it entered the river and there was no reaeration. Of course, at an oxidation rate of only 10 percent/day (K_d = 0.1/day) and a travel time on the order of one day, only a fraction of this deficit could be exerted within the study area during high stage periods such as this. Hence the load deficit is at most only a small fraction of the 6 mg/L total deficit typically observed during high stage conditions.

6.3.2 Components of Flood Plain Deficit

The factor having the most important influence on the simulated main channel profile is the dissolved oxygen concentration assigned to the flood plain water, which dominates water quality in the river. Since the flood plain has not been explicitly modeled, however, it is not possible to perform a detailed evaluation of the factors contributing to this deficit. However, a simplified model framework can be used to represent flood plain conditions and to evaluate the components of deficit within this region. Specifically, the causes of the high dissolved oxygen deficits/low dissolved oxygen concentrations typically observed during periods of flooding will be assessed. A simplified model of the sources (surface transfer) and sinks (CBOD oxidation and SOD) of dissolved oxygen will be used to obtain insight into the phenomena which occur. The controlling differential equation, expressed in terms of dissolved oxygen deficit, (dissolved oxygen saturation minus dissolved oxygen concentration), is as follows:

$$\frac{dD}{dt} = K_d L_o + \frac{S_B}{H} - \frac{K_L}{H} D \tag{6-1}$$

Here, D is the dissolved oxygen deficit, L_o the ultimate carbonaceous BOD, K_d the first order BOD oxidation rate coefficient, S_B the benthal or sediment oxygen demand (SOD), K_L the surface transfer coefficient of oxygen between the atmosphere and water column and H the average depth of water on the flood plain. Net inflow or outflow, to or from the flood plain, are neglected, a reasonable approximation during periods of constant water surface elevation.

The steady state solution of Equation 6-1 for dissolved oxygen deficit is given by:

$$D = \frac{1}{K_L/H} (K_d L_o + \frac{S_B}{H}) \qquad (6\text{-}2)$$

From this expression, it is clear that the deficit will be inversely related to the surface transfer coefficient of oxygen, K_L. Figure 5-2 presented previously, illustrates how this coefficient varies as a function of wind speed. At moderately low wind speeds of less than 10 mph, the data indicate a range of transfer coefficients of about 1 to 3 feet/day can be expected. A value of 2 feet/day is typically used for open channels. Within the protected conditions of the flood plain, where wind induced turbulence is minimized by the forest canopy, a value as low as 1 foot/day may be reasonable.

The solution for dissolved oxygen deficit (Equation 6-2) is shown as a function of SOD, for a range of ultimate CBOD concentrations of 0 to 20 mg/L, a CBOD oxidation rate of 0.05/day, surface transfer coefficients of 1 and 2 feet/day, a depth of 10 feet and a temperature of 20°C on Figure 6-10. The deficit is shown to be quite sensitive to SOD, with deficits of about 2 to 3 mg/L associated with the range of SOD of 1 to 2 grams/m^2/day and a surface transfer coefficient of 2 feet/day. The impact of CBOD oxidation is relatively small, at less than 0.5 mg/L at an ultimate CBOD concentration of less than 20 mg/L. These deficits are increased by a factor of two, to 4 to 6 mg/L, at the lower surface transfer coefficient of 1 foot/day. Thus, even in the absence of CBOD oxidation, SOD alone can reasonably be expected to result in dissolved oxygen deficits of 6 mg/L, or corresponding dissolved oxygen concentrations as low as 2 mg/L. These expected deficit concentrations would be still higher at increased temperatures of 25 to 30°C, as occurs in late spring and early summer periods of flooding.

It is of interest to note that the steady state concentration of $CBOD_u$ for the conditions described above, and the areal fluxes of 1.4 to 2.0 grams $CBOD_u$/m^2/day shown on the lower left panel of Figure 6-5, is about 10 to 12 mg/L. Given the preceding results, however, it is evident that oxidation of CBOD is relatively unimportant in comparison to SOD with regard to flood plain deficit. In light of this, it is relevant to

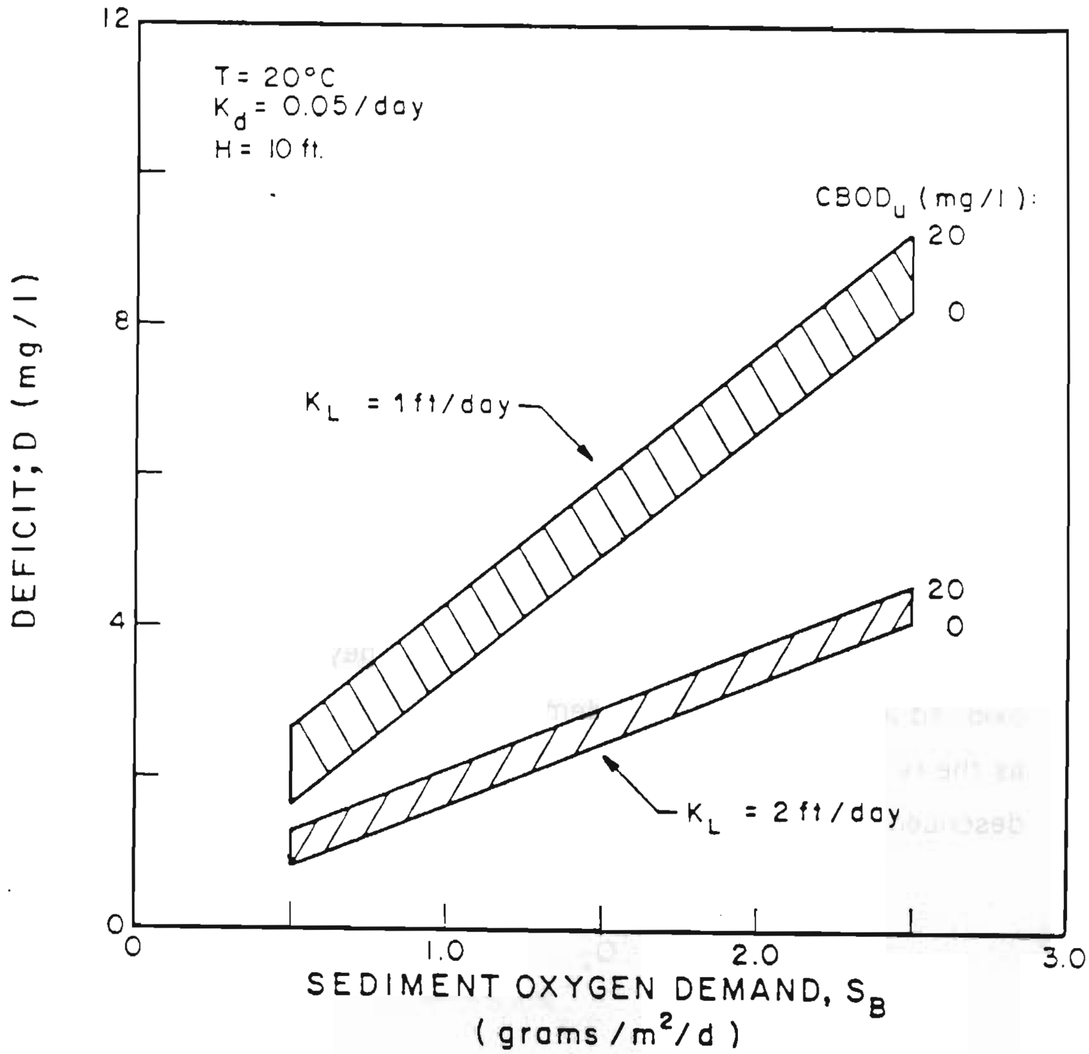


FIGURE 6-10. RELATIONSHIP OF DISSOLVED OXYGEN DEFICIT ON FLOOD PLAIN TO SOD, CBOD OXIDATION AND OXYGEN SURFACE TRANSFER COEFFICIENT

consider the magnitude of the sources of oxygen demanding material that could give rise to SOD on the flood plain.

6.3.3 Natural Background Loads and Flood Plain SOD

Preliminary investigations have been performed to evaluate the cause-effect relationship between sources of particulate organic carbon in the study area and background oxygen demand. Previously completed studies of Milwaukee Harbor, Flushing Bay and the Potomac River have shown that the magnitude of sediment oxygen demand and the release of unoxidized decomposition end products from sediments can be related to the flux of particulate organic matter to the sediment-water interface (HydroQual, 1987a; HydroQual, 1987b; SWRPC, 1987). Briefly, the particulate organic matter (POM), here represented as CH_2O, undergoes anaerobic decomposition in the sediments to produce reduced end products. These reduced end products diffuse through the pore water to the sediment-water interface where they are released to the water column or are oxidized as sediment oxygen demand. A typical reaction in which methane is produced as the reduced end product of POC (represented here as CH_2O) decomposition may be described as follows:

$$2CH_2O \rightarrow CH_2 + CO_2 \tag{6-3}$$

The dissolved methane (CH_4) in the interstitial water diffuses to the sediment-water interface, where it is oxidized by micro-aerophilic bacteria that are present throughout the environment. The oxidation reaction is as follows:

$$CH_4 + 2O_2 \rightarrow CO_2 + 2H_2O \tag{6-4}$$

The stoichiometry of these reactions indicates that 2.67 grams oxygen/gram POC or 5.33 grams oxygen/gram CH_4-C is required to satisfy the oxygen demand. The reaction is further complicated when the methane concentration exceeds saturation and gas bubbles are formed, and when overlying water dissolved oxygen levels are low enough to inhibit complete oxidation at the sediment-water interface, resulting in the release of unoxidized dissolved methane to the water column. (Alternatively, if the anaerobic reaction does not proceed to completion, where methane is formed, DOC is released from the sediment by diffusion.)

An associated production of ammonia also occurs as part of the decomposition reaction, and this ammonia diffuses to the sediment water interface where a portion of it may also be oxidized. Other pathways can be followed as well, such as the production and oxidation of hydrogen sulfide, but the end result is the same, i.e., the potential oxygen demand generated by the sediments is stoichiometrically related to the flux of reactive particulate organic material delivered to the sediment-water interface.

For the Ouachita River flood plain, an estimate of the flux of particulate material in the form of forest litter has been made on the basis of literature values (Cromack, 1973; Bray and Gorham, 1964; Carlise et al., 1966; Sykes and Bunce, 1970). Table 6-2 summarizes mean litter production measurements for a variety of forest types. Since these data represent mean values, site specific conditions are often higher than the averages reported. The data show that litter production tends to increase in warmer regions, where productivity is greatest, as would be expected. The range of mean litter production measurements in warm temperate and tropical forests is 1.15 to 2.90 grams/m^2/day.

The potential oxygen demand generated by the decomposition of forest litter on the flood plain can be estimated from the mean litter production rates presented in Table 6-2. Assuming a stoichiometric oxygen requirement of the forest litter of 2.67 grams oxygen/gram particulate carbon reacted and that the forest litter is approximately 50 percent carbon (Cromack, 1973), a range of 1.15 to 2.90 grams solids/m^2/d (warm temperate and tropical forests) is equivalent to a corresponding range of oxygen demand of 1.55 to 3.87 grams/m^2/day. This range compares well with oxygen respiration rates for forest litter from flood plain soils of 1.4 to 7.5 grams/m^2/day (Ogeechee River) and 2.0

TABLE 6-2. SUMMARY OF FOREST LITTER MEASUREMENTS FOR A VARIETY OF FOREST TYPES

Forest Type	N	Mean Litter Production: Solids ($g/m^2/d$)	Mean Litter Production: Carbon[a] ($g/m^2/d$)	Oxygen Equivalent[b] (g/m^2d)	Solids Load[c] (lbs/day)	Reference
Warm Temperate	1	1.15	.58	1.55	16,000	Cromack (10)
Tropical	18	2.90	1.45	3.87	41,000	Bray & Gorham (11)
Warm Temperate	39	1.40	0.70	1.87	20,000	Bray & Gorham (11)
Cool Temperate		.93	0.46	1.23	13,000	Bray & Gorham (11)
Cold Temperate	6	.27	.13	.36	3,800	Bray & Gorham (11)
N. American Warm Temperate Deciduous	11	1.30	.65	1.74	19,000	Bray & Gorham (11)
Cool Temperate Deciduous		1.[illegible]0	.50	1.34	14,000	Bray & Gorham (11)
Oak Dominated British Forest		1.[illegible]	.55	1.47	15,000	Carlisle, Brown & White (12)
Meathop Wood		[illegible]	.70	1.87	20,000	Skykes & Bunce (13)

[a] $C = .5 \times TSS$
[b] $O_2 = 2.67 \times$ Carbon
[c] Area = 6.4 km^2

to 5.1 gram^2/m^2/day (Black Creek) as reported by Cuffney (1984). It is easily large enough in magnitude to account for the SOD required to generate deficits of 6 mg/L, as discussed with regard to Figure 6-10, even if only 50 percent of the organic carbon is reactive. Also, since these estimates are annual averages, and most of the forest litter is expected to decompose as conditions become warmer during the spring, the likely demand during periods of spring flooding is probably higher still. Thus, naturally occurring forest litter can account for a significant natural background oxygen demand in the flood plain waters and this can have a significant impact on the Ouachita River during periods when significant interaction between the river and flood plain occur.

SECTION 7

MODEL PROJECTIONS

Projections of dissolved oxygen levels in the Ouachita River for selected critical monthly flow and temperature conditions are described in this section. A description of the basis for selection of conditions used in model projections is first reviewed. This is followed by the model projections which include: (1) an evaluation of natural background conditions, in the absence of any impact from the mill and (2) projections of the overall dissolved oxygen response when the impact of the mill is included at the current BOD_5 permit limit.

7.1 SPECIFICATION OF MODEL PROJECTION CONDITIONS

Critical monthly model projection conditions for flow, temperature, upstream and uniformly-distributed BOD and dissolved oxygen deficit are presented in Table 7-1 and shown on Figure 7-1. The monthly 7Q10 projection flows used for the Ouachita River at the Arkansas-Louisiana state line were determined from long-term records (1970-1990) at the USGS gage at Camden (Appendix F; Advent, 1992). These flows, ranging from 802 cfs in August to 5,269 cfs in February, were estimated by extrapolating drainage area water yields at Camden to the Arkansas-Louisiana state line at 7Q10 conditions. The flows generated in this manner appear reasonable based on a comparison with the results of a statistical analysis of flow data at Felsenthal from 1984 to the present (the years during which the new Felsenthal dam has been in operation). These recent Felsenthal data are not adequate for use in performing a statistically rigorous low-flow analysis because of the relatively short period of record since the new dam operations, which influence flow at this location, commenced.

The mean monthly temperatures presented on Figure 7-1 (middle panel) are for a 28-year period of record (1950 through 1977) at the USGS station at Felsenthal. The monthly mean temperatures range from 9.6°C in December to 29.5°C in July and August. Analysis of the temperature data during the more critical warm weather months indicated that flow and temperature were uncorrelated, and hence the average temperature is used.

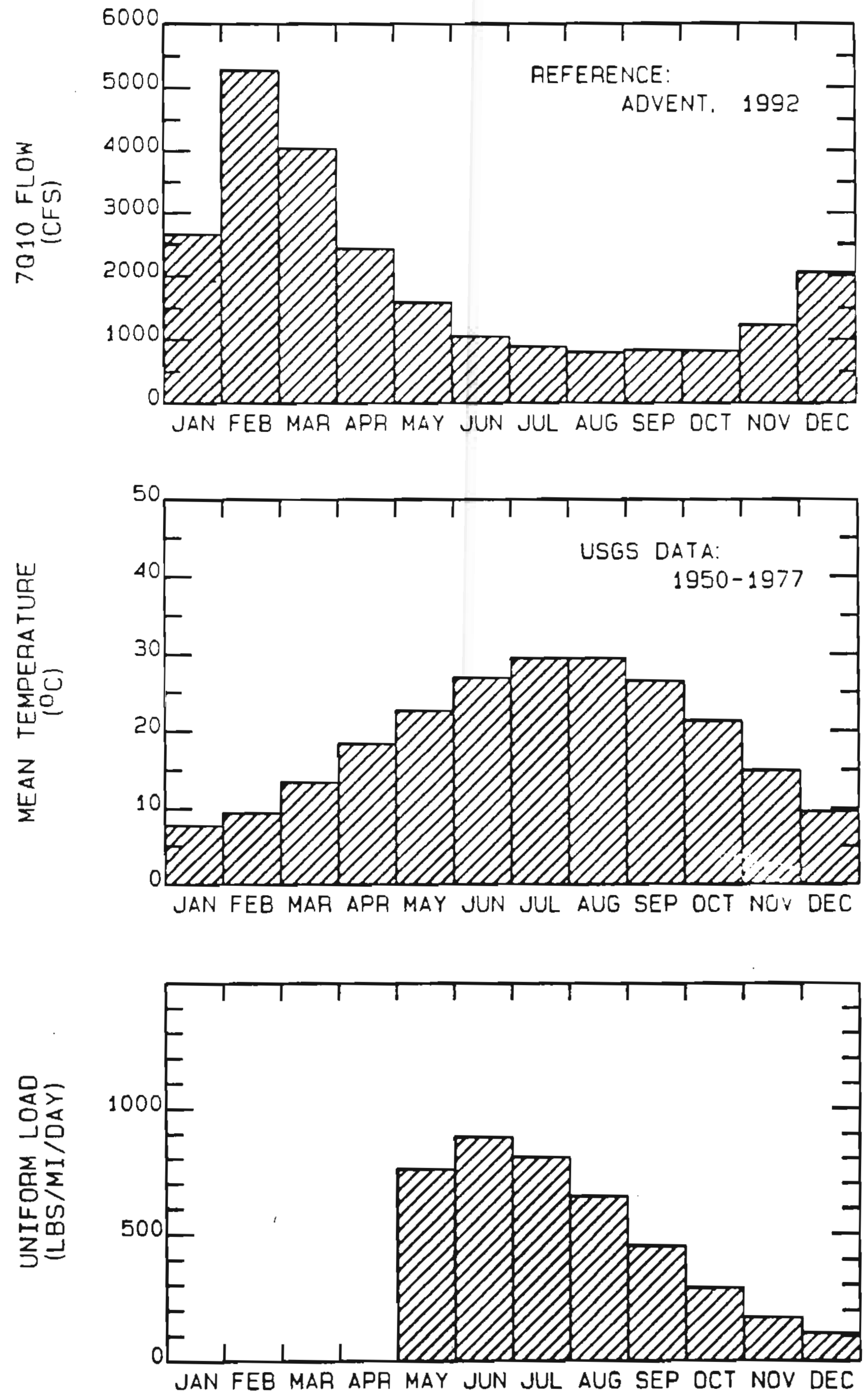


FIGURE 7-1. MONTHLY AVERAGE CONDITIONS USED IN QUANCHITA RIVER DISSOLVED OXYGEN MODEL PROJECTIONS

TABLE 7-1. SUMMARY OF MONTHLY AVERAGE CONDITIONS USED IN OUACHITA RIVER DISSOLVED OXYGEN MODEL PROJECTIONS

			Ultimate CBOD		
Month	(a) Monthly 7Q10 cfs	(b) Water Temperature, °C	(c) Upstream Boundary mg/L	(d) Uniform Load lbs/mi/d	(e) Upstream D.O. Deficit mg/L
May	1,591	22.7	6.44	757	2.5
June	1,049	26.9	6.22	886	2.4
July	894	29.5	5.02	806	2.3
August	802	29.5	4.03	647	2.0
September	829	26.5	3.24	453	2.0
October	822	21.3	2.61	288	1.8
November	1,231	14.9	2.10	172	1.7
December	2,059	9.6	1.70	109	0.8

(a)Advent, 1992 (Appendix F)
(b)1950 through 1977 average at Felsenthal
(c)Figure 7-2, Equation 7-1
(d)Equation 7-2
(e)Average of data at Stations B1, B2 and B3 (1970 thorugh 1981) and 1987 through 1991)

Upstream ultimate CBOD concentrations are set based on the relationship with time from the end of flooding, as shown on Figure 7-2. This relationship can be expressed as:

$$L_o = 6.44\ e^{-.0071t} \tag{7-1}$$

where Lo is the upstream $CBOD_u$ concentration (mg/L) and t is time after the end of flooding (days). The median date at the end of flooding for the period of 1958 through 1991 is June 10,

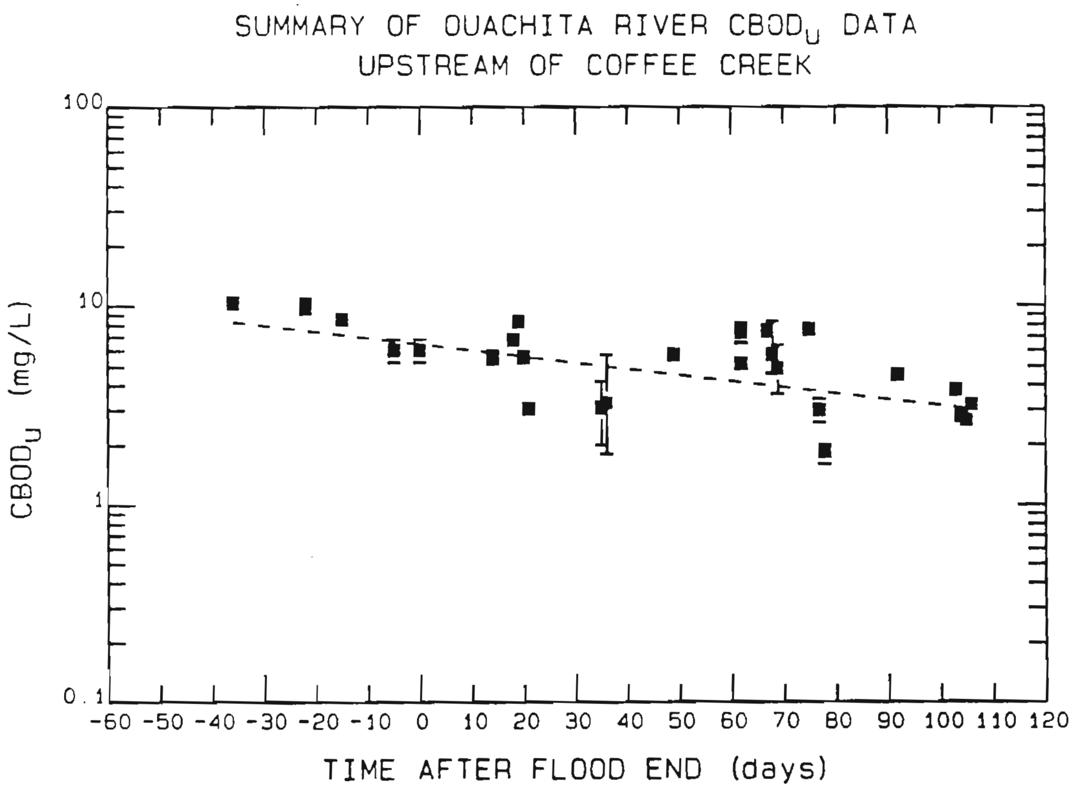


FIGURE 7-2. VARIATION OF UPSTREAM CBODu CONCENTRATION WITH TIME AFTER FLOODING

or Julian Day 161. The $CBOD_u$ values presented in Table 7-1 decrease from approximately 6.4 mg/L in May to 1.7 mg/L in December.

The uniformly distributed $CBOD_u$ load (W_{ul}, lbs/mile/day) is estimated using the following relationship:

$$W_{ul} = 0.33\ A\ K\ L_o \tag{7-2}$$

where A is the channel cross sectional area (6300 ft^2), Lo is the upstream $CBOD_u$ concentration (mg/L) and K is the CBOD oxidation rate coefficient (1/day), evaluated at the monthly average temperature. As shown on the lower panel of Figure 7-1, monthly uniform $CBOD_u$ loads range from 886 lbs/mi/d during June to 453 lbs/mi/d in September, the more critical months, to as low as 109 lbs/mi/d in December.

The mean monthly upstream dissolved oxygen deficits are established from a statistical analysis of upstream temperature and dissolved oxygen data (routine monitoring stations and intensive survey stations B1, B2 and B3; i.e., generally at stations located between MP 230 and MP 240) for the period from 1970 to 1981 and 1987 to 1991. As shown in Table 7-1, these values range from less than 1 mg/L in February to about 2.5 mg/L in May. These deficits are assigned to both the upstream model boundary and to tributary inputs.

Model projections presented for natural background conditions are performed with the mill BOD_5 load set to zero, but the effluent flow at an average condition. This is a reasonable approximation as the effluent flow is typically only a small percentage (less than 5 percent) of the total river flow. Further, in the absence of the mill withdrawal from the Saline River, additional flow would be available at the upstream boundary of the model and would offset any flow reduction at Coffee Creek.

Table 7-2 summarizes the current mill permit limits for BOD_5, where for the months of July, August and September the permit limit shown corresponds to the "daily maximum limitation" at 7Q10 (the monthly 7Q10 river flow is also indicated). The daily maximum BOD_5 mass discharge as defined for purposes of permit compliance, is calculated by "multiplying the

TABLE 7-2. RESPRESENTATIVE EFFLUENT PERMIT LOADS FOR MODEL PROJECTIONS

			(c) Permit BOD_5			
Month	(a) State Line 7Q10 cfs	(b) Flow mgd	Concn mg/L	Load lbs/d	(d) f	$CBOD_u$ mg/L
May	1,591	41	23.4	[illegible]	3.5	82.6
June	1,049	41	23[illegible]			82.6
July	894	41	10.8			58.3
August	802	41	10.0	3,407	[illegible]	56.2
September	829	41	10.2	3,490	5.6	56.9
October	822	41	23.4	12,000	3.5	82.6
November	1,231	41	23.4	12,000	3.5	82.6
December	2,059	41	23.4	12,000	3.5	82.6

(a)Extrapolaiton from 1970 through 1990 Camden flow record
(b) Expected average effluent flow rate
(c)Permit Load based on 1986 permit report
(d)Figure 5-6: $f = 20\ BOD_5^{-.55}$

average BOD_5 concentration from the previous seven days by the daily (effluent) flow and by 8.34." The current daily maximum BOD_5 permit limit actually increases linearly with river flow to 12,000 lbs/day at 3,520 cfs during these warmer months. An effluent flow of 41 mgd (63.4 cfs) is assigned for all projections at the permit limits. This is the expected average effluent flow rate during future years. The corresponding BOD_5, f and $CBOD_u$ values are also shown, with the BOD_5 and $CBOD_u$ concentrations ranging from 10.0 and 56.2 mg/L in August to 23.4 and 82.6 mg/L in December, respectively. An effluent dissolved oxygen deficit of 5 mg/L is also assigned, as discussed above.

The oxidation rate coefficients (K_d, at 20°C) as determined in these studies are 0.1/day for effluent CBOD and 0.05/day for background CBOD. These rate coefficients are adjusted to the appropriate water temperature using:

$$K_d(T) = K_d(20) \times \theta^{(T - 20)} \quad (7\text{-}3)$$

where θ = 1.047. This approach is consistent with the method used in the model calibration analysis.

Instream aeration is estimated as described previously in Sections 3 and 5. For purposes of quantifying the effects of dam aeration in some of the projections which follow, the following relationship (Hydroscience, 1968) is used:

$$D_b = D_a \left(1 - 0.037 H_d * 1.024^{(T-20)}\right) \quad (7\text{-}4)$$

Here, D_b and D_a are the deficit below and above the dam respectively and H_d is the height of the dam in feet. This formula was developed for dams up to 15 feet in height and should be verified using actual data from Lock 6 once release over the crest of the dam goes into operation.

Since nitrification was not judged to be a significant component in the calibration analyses described in Section 5, nitrification is not included in the projections to be presented. Net photosynthesis was not observed in the field studies and therefore it is not included in the projections either.

7.2 MODEL PROJECTION RESULTS

Model projection results will next be presented for both low stage and high stage conditions. Results for natural background conditions are included to place in perspective the expected water quality, both with and without the impact of the mill discharge, with water quality standards in the Ouachita River.

7.2.1 Low Stage Conditions

Model projections have been completed on a monthly basis for the typically low stage, high temperature months of July through September. These are the months for which the current permit is dependent on river flow. Figures 7-3 through 7-5 present results at the monthly 7Q10 flows for these three months, respectively. The upper panels of these figures show results without aeration at the dam at Lock 6 and the lower panels show the improvement when aeration is included. During the month of July (Figure 7-3), the projected dissolved oxygen concentrati[illegible] due to natural background conditions, in the absence of dam aeration (upper panel) is generally about 4.0 to 4.5 mg/L throughout the study area, with a minimum of 3.8 mg/L (upper dashed profile). With the mill discharge at the permit limit in [illegible] projection, the minimum dissolved oxygen decreases to about 3 mg/L, with the maximum deficit attributed to the mill BOD_5 load (1.1 mg/L) occurring in the vicinity of MP 200.

The lower panel of Figure 7-3 shows the corresponding profiles when aeration at the dam is included in the projection. Conditions are generally improved in the river, with the dissolved oxygen standard achieved between the dam and the Arkansas - LOuisiana state line (MP 221), both for natural background conditions and with the mill discharge included. Downstream of the state line dissolved oxygen levels are also generally improved. However, even under natural background conditions alone the profile is projected to be less than 5 mg/L downstream of MP 210, and minimum dissolved oxygen levels at the downstream end of the study area are essentially the same as the model projections without dam aeration.

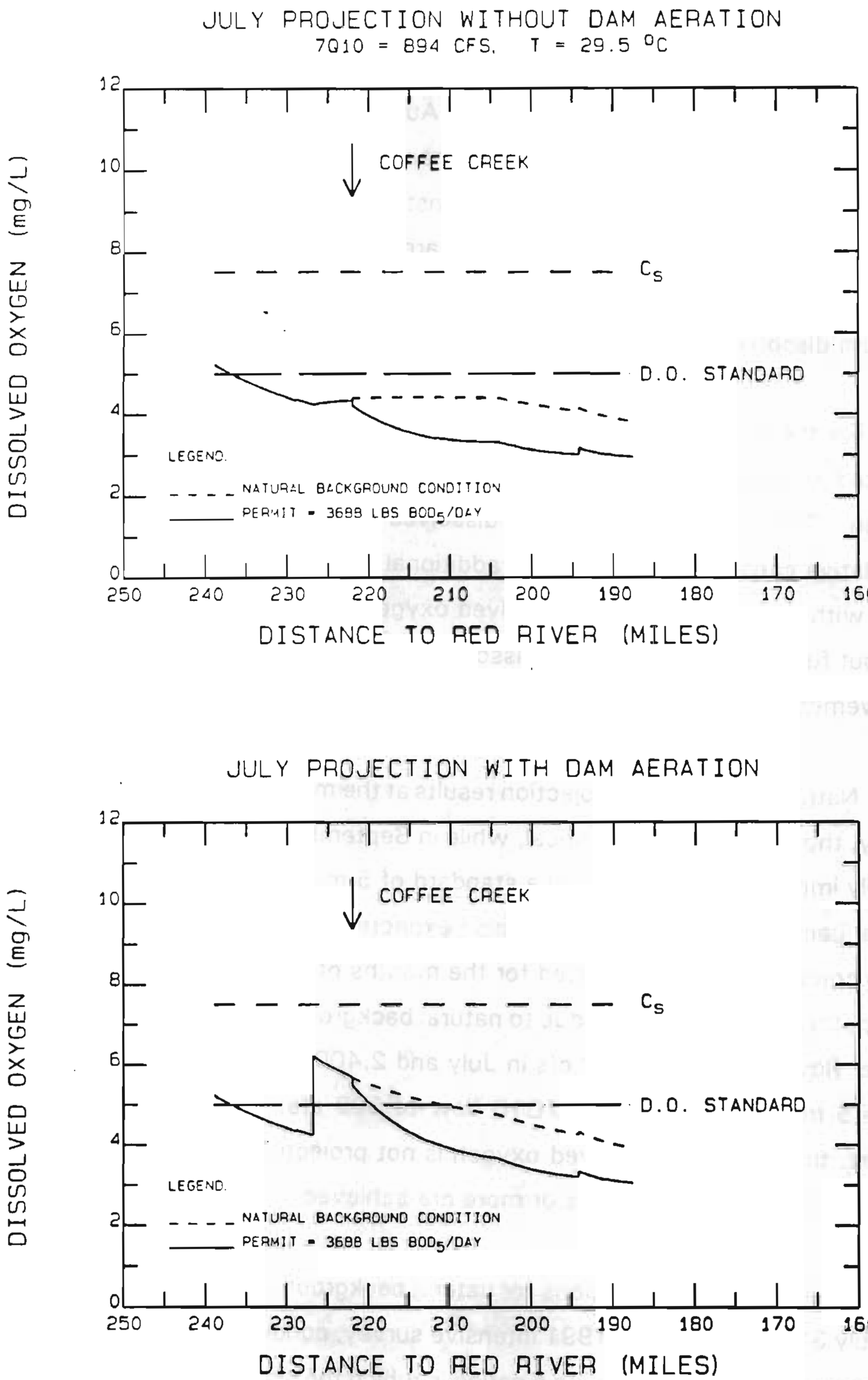


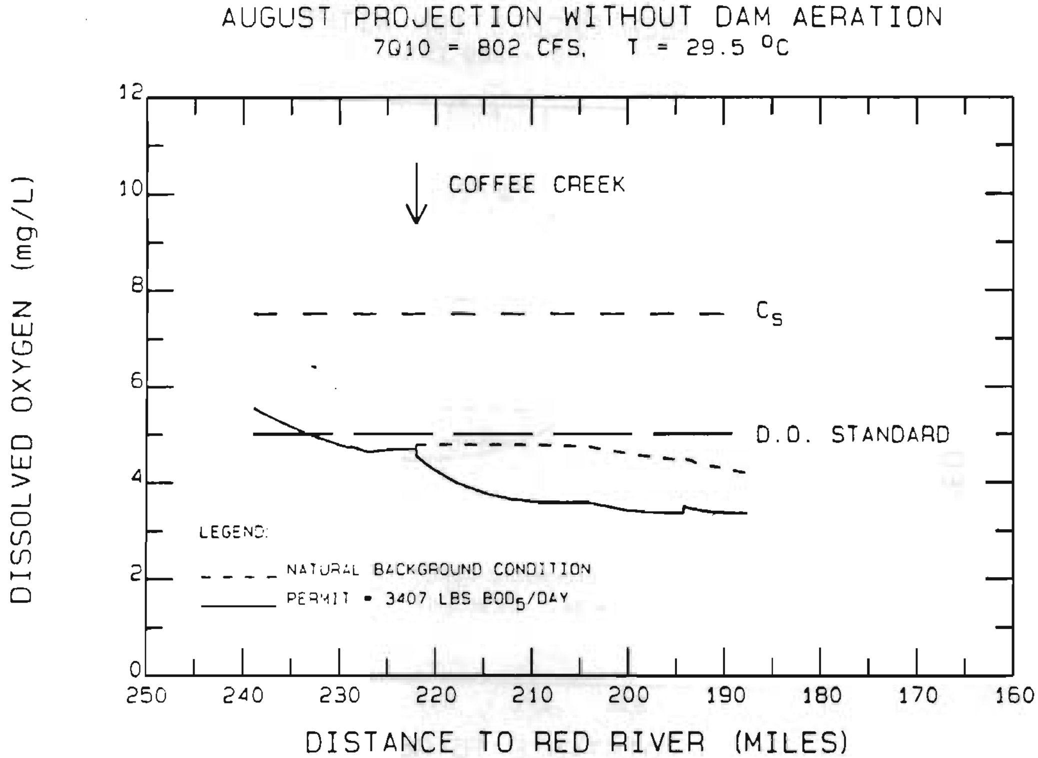
FIGURE 7-3. LOW FLOW DISSOLVED OXYGEN PROJECTION FOR JULY

Results analogous to July are shown for the August and September 7Q10 conditions on Figures 7-4 and 7-5. The model results in August (Figure 7-4) are very similar to those of July. Natural background conditions are less than the standard for most of the study area in the absence of dam aeration, as well as downstream of MP 204 when dam aeration is included. During September (Figure 7-5), conditions are improved and the dissolved oxygen standard of 5 mg/L is achieved under natural background conditions. At the current permit limit, however, the minimum dissolved oxygen concentration in the study area is 4.3 mg/L.

For the July and August projections described above, natural background conditions are projected to be less than 5 mg/L throughout most of the study area, in the absence of dam aeration. This is below the current dissolved oxygen standard of 5 mg/L and as a result, no assimilative capacity is available for additional point source loads. Conditions are significantly better with dam aeration, with dissolved oxygen levels above 5 mg/L between the dam and MP 210, but further downstream the dissolved oxygen is again still less than 5 mg/L, and negligible improvement is projected toward the downstream end of the study area.

Natural background projection results at the monthly 7Q10 for June and August are similar to July, though slightly less critical, while in September and October, conditions are projected to steadily improve and be above the standard of 5 mg/L, even without dam aeration. Results for natural background projections are also expected to be improved as river flow increases above 7Q10 conditions, as summarized for the months of July through September on Figure 7-6. As shown, the dissolved oxygen due to natural background conditions is projected to exceed 5 mg/L at river flows of about 3,500 cfs in July and 2,400 cfs in August, while September results are above 5 mg/L at the monthly 7Q10 flow of 829 cfs. Hence, during the months of July and August, the minimum dissolved oxygen is not projected to be above the 5 mg/L standard until river flows of about 2,500 cfs or more are achieved.

The preceding projections for natural background conditions are consistent with results of the July 31 and August 1, 1991 intensive survey, conducted about three weeks after the end of high stage conditions, following a period in which the discharge from the mill was stored in Mossy Lake for approximately a two week period. The river flow preceding this survey averaged 2,



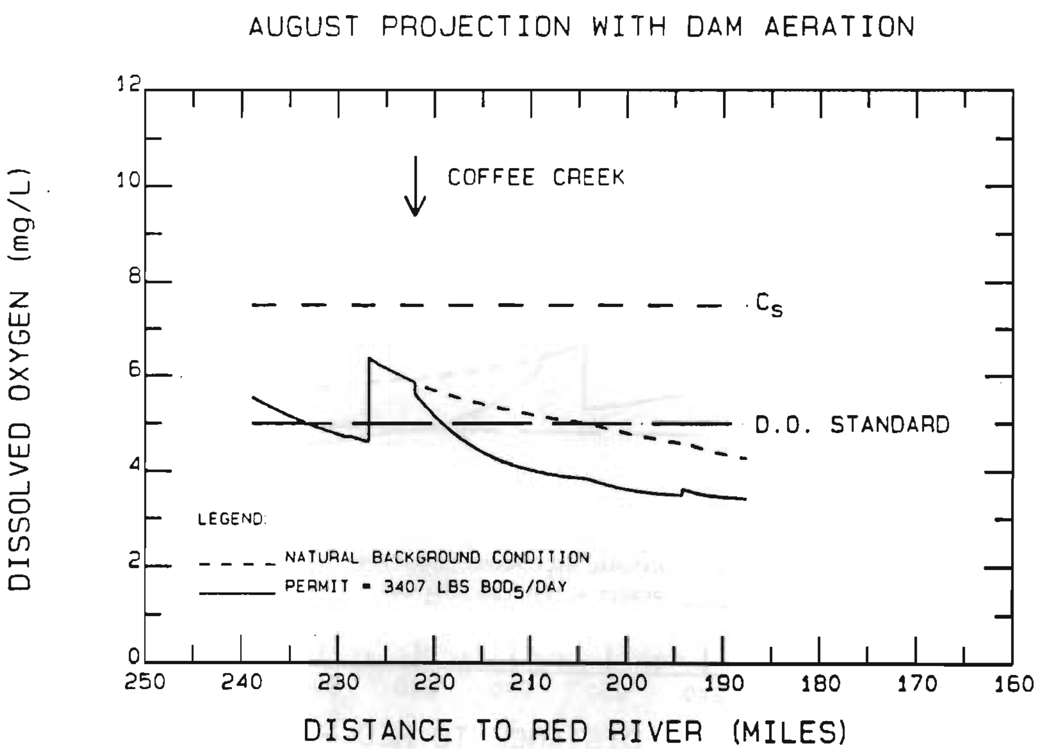
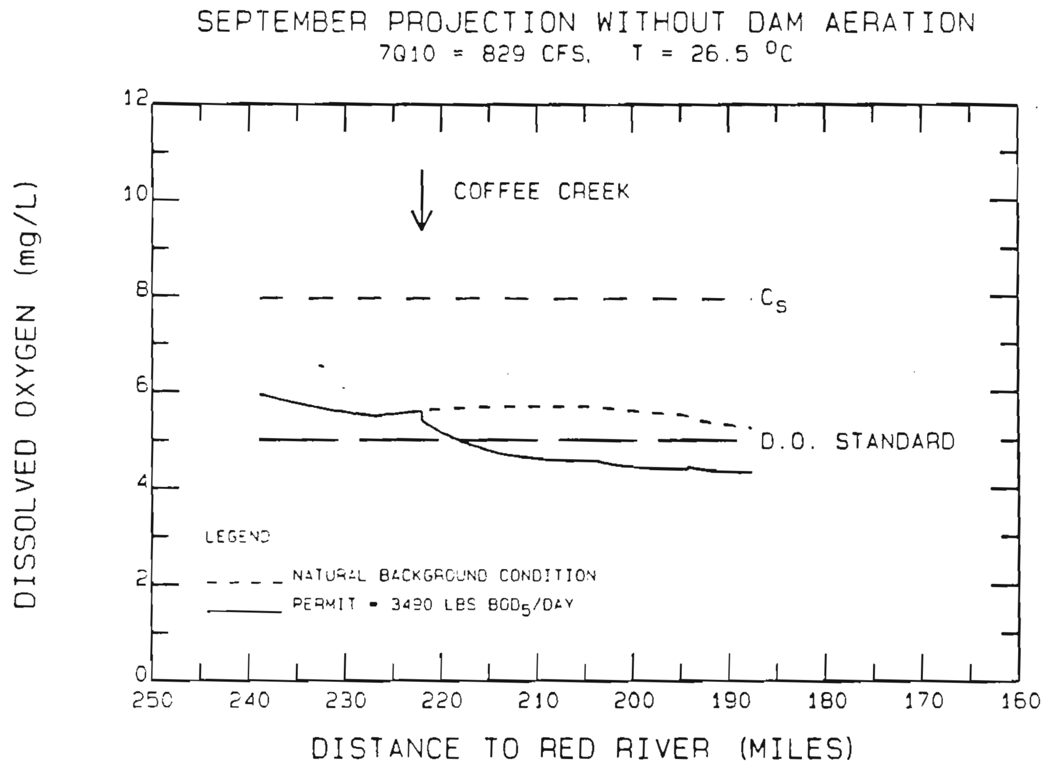


FIGURE 7-4. LOW FLOW DISSOLVED OXYGEN PROJECTION FOR AUGUST



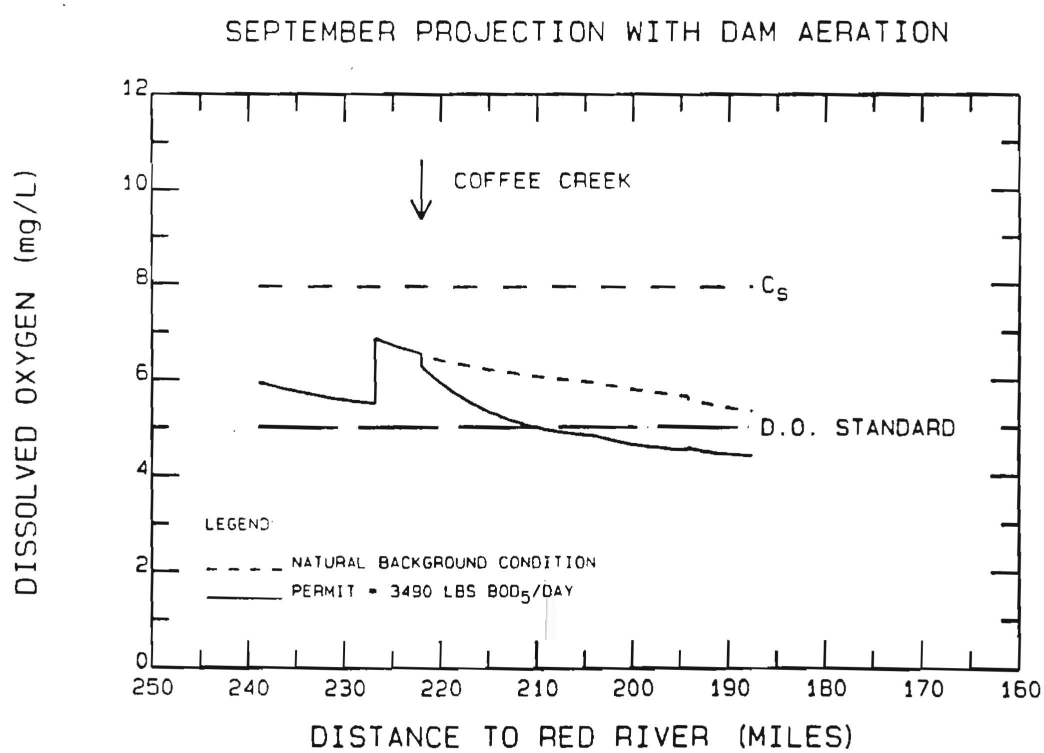


FIGURE 7-5. LOW FLOW DISSOLVED OXYGEN PROJECTION FOR SEPTEMBER

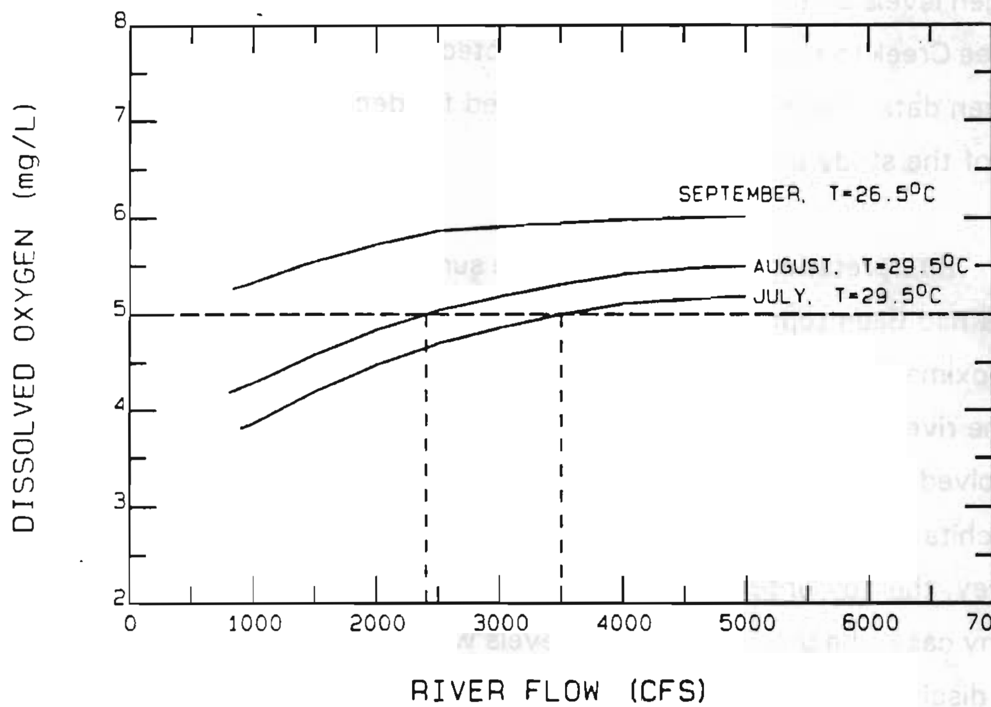


FIGURE 7-6. VARIATION OF MINIMUM RIVER DO WITH RIVER FLOW UNDER NATURAL BACKGROUND CONDITIONS

cfs, corresponding to a travel time of about 4 days in the study area downstream of the discharge. Figure 7-7 shows the conservative tracer profiles and model results for this survey, with the uniform spatial profiles confirming the absence of any discharge from the mill. The $CBOD_u$ profile on the upper panel of Figure 7-8 confirms this observation, and generally fits the observed data as well, except possibly at the downstream end of the study area. Dissolved oxygen levels on the lower panel are slightly below 5 mg/L between the station just upstream of Coffee Creek to about MP 210, as predicted for natural background conditions, but the dissolved oxygen data then exhibit an unaccounted for decrease to less than 4 mg/L by the downstream end of the study area.

Interpretation of the no discharge survey data is complicated by the fact that the bean field dikes had been topped by the record high water stage during 1991 (water surface elevation of approximately 90 feet) and at the time of the survey, water was being pumped from the fields to the river in the vicinity of MP 210. It appears that these operations have impacted the river dissolved oxygen downstream of MP 210. In addition, as happens rather frequently in t Ouachita River, the river flow increased by approximately a factor of two during the course of the survey, thereby upsetting what would otherwise have been relatively steady state conditions. In any case, the dissolved oxygen levels were observed to be less than 5 mg/L in the absence of any discharge, as predicted for zero discharge conditions.

When natural background conditions prevent meeting the dissolved oxygen objective of 5 mg/L, as in the projection results described above, consideration should be given to allowing the minimum acceptable concentration in the river to be 90 percent of the natural concentration, pursuant to the national criteria document for dissolved oxygen (USEPA, 1986a and 1986b).

7.2.2 High Stage Conditions

During high stage conditions the crest of the dam at Felsenthal is submerged (the upper and lower pools are at the same water surface elevation) and dam aeration does not take place. Also, most of Coffee Creek and all of Mossy Lake are flooded, so the discharge from the m treatment system to the Ouachita River flood waters is at the test point designated R1, rather

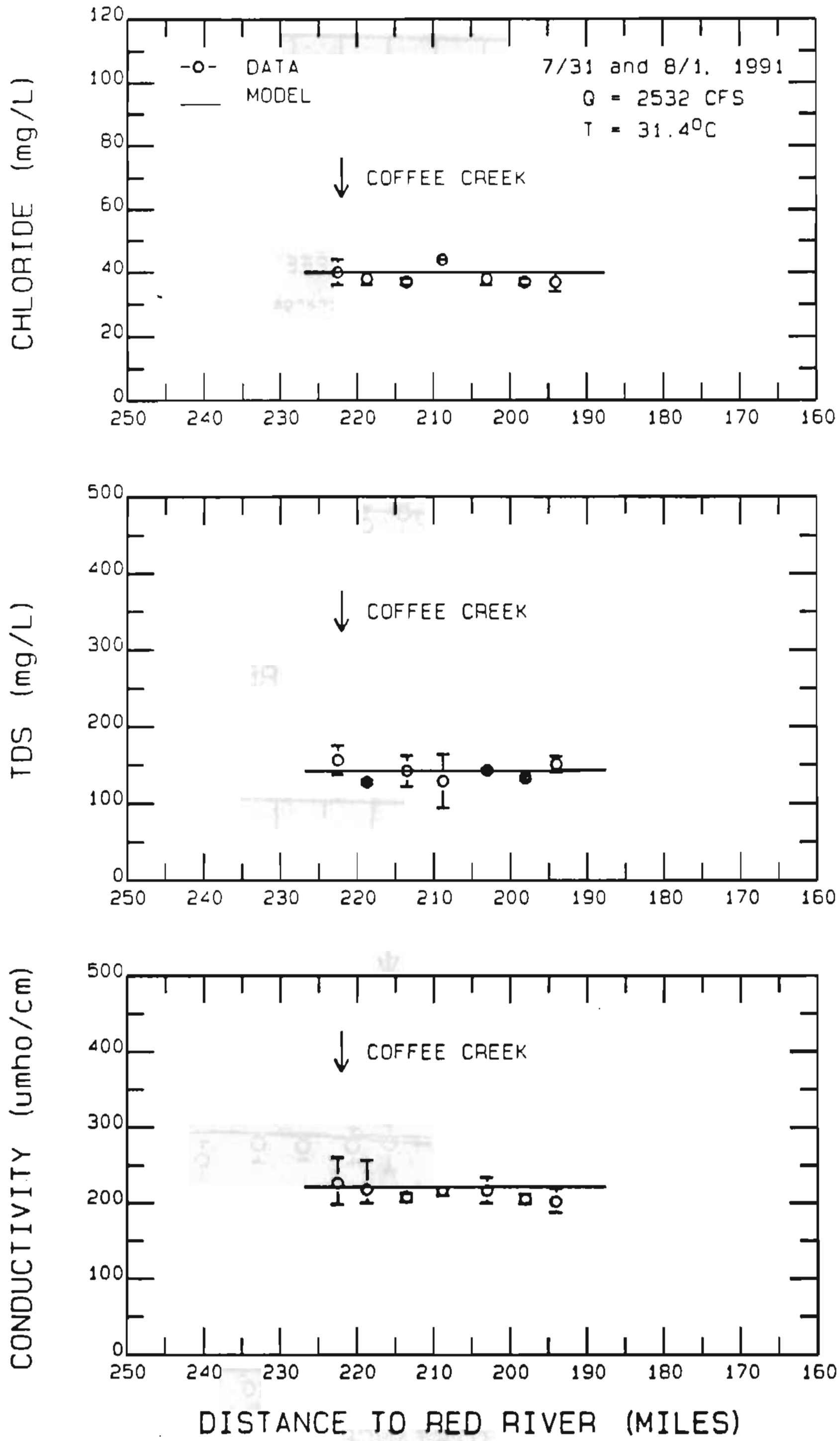


FIGURE 7-7. CONSERVATIVE TRACER MODEL AND DATA FOR ZERO DISCHARGE SURVEY

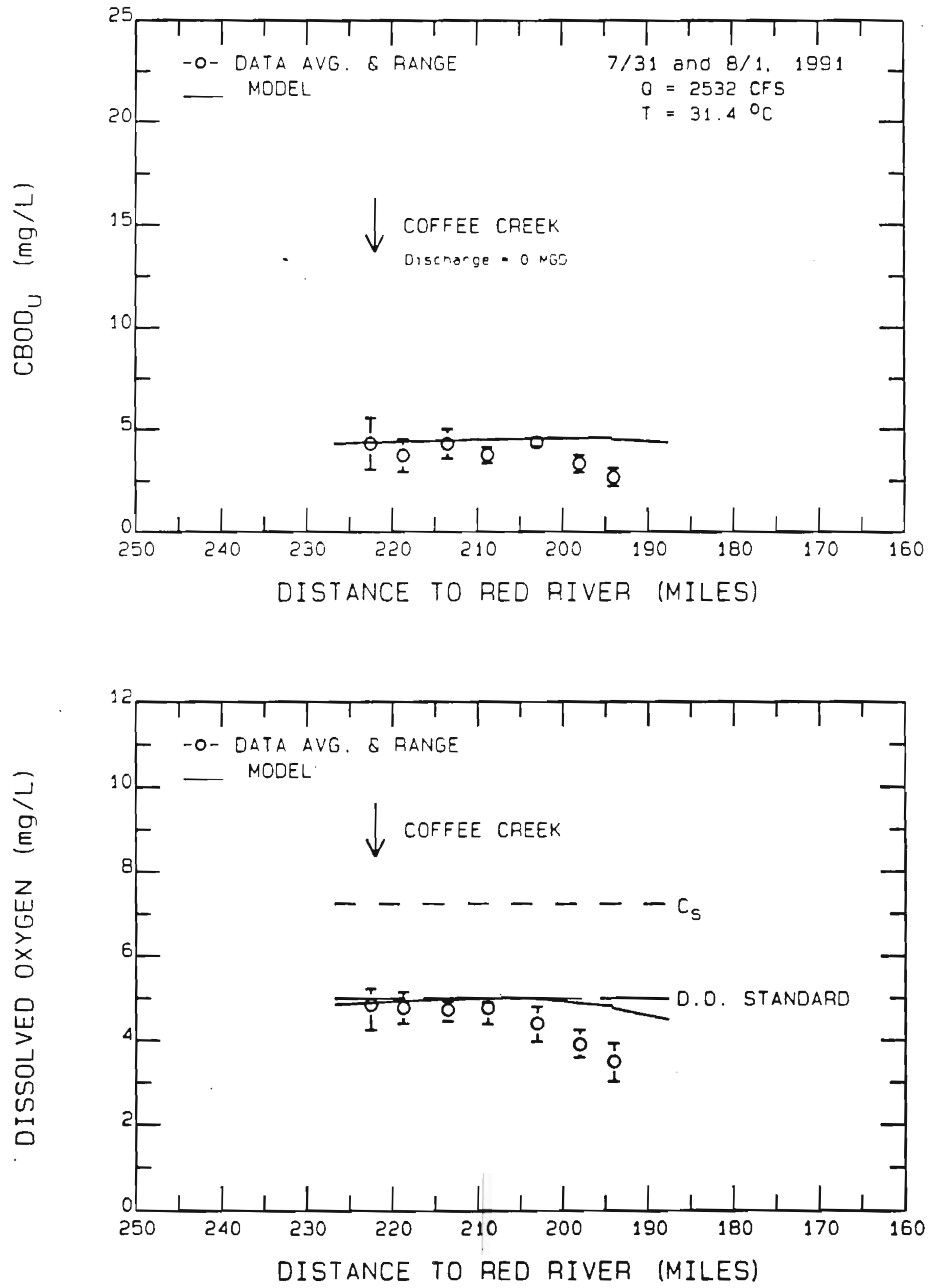


FIGURE 7-8. BOD-DO MODEL AND DATA FOR ZERO DISCHARGE SURVEY

than at TP3. Figure 7-9 presents a projection for a river flow of 30,000 cfs, T = 25°C, a river stage of 30 feet (water surface elevation of 74 feet) and a flood plain deficit of 6.3 mg/L (as observed during the 1980 studies). The upstream dissolved oxygen decreases from the upstream boundary concentration of about 4 mg/L to about 2 mg/L near MP 210, and then is relatively constant in the downstream direction as spatial steady state is approached. This projection run is essentially the same both with and without the mill discharge at R1, since the deficit associated with the effluent load is calculated to be less than 0.1 mg/L and dissolved oxygen levels are dominated by natural background conditions.

The projected high stage dissolved oxygen is significantly less than the dissolved oxygen standard of 6.5 mg/L in Arkansas and 5 mg/L in Louisiana throughout the study area. As demonstrated by some of the data presented previously (see e.g., Figure 6-1 and 6-2), the dissolved oxygen profile will tend to be lower than that shown on Figure 7-9 during relatively extended periods of flooding which persist into warmer temperature periods, as well as at times when floodwaters are higher than assumed for this simulation. Given the dystrophic nature of these waters during high stage conditions, a change in the dissolved oxygen standard should be considered.

7.2.3 Impact Analysis: Coffee Creek to Columbia Lock and Dam

The analysis presented herein has focused on the reach of the Ouachita River upstream of Bayou Bartholomew. The data reviewed were used to refine the previously developed model in this upstream area and to obtain a better understanding of the relative impacts of the Georgia Pacific discharge and background loads on dissolved oxygen levels in the river. Although the analyses completed si[illegible] 1990 have not been directed at verifying or refining the calibration of the model between Bayou Bar[illegible] [illegible]olumbia Lock and Dam, the impact of the Georgia Pacific load on this [illegible] [illegible]nated. The analysis is performed by considering only the load from Coffee Creek in the model and eliminating all other sources and sinks of dissolved oxyg[illegible] with the exception of reaeration. This assessment does not attempt to project the absolute dissolved oxygen concentration that will result with all of the point and non-point source loads entering this downstream reach of the river, but rather, it evaluates the contribution to the

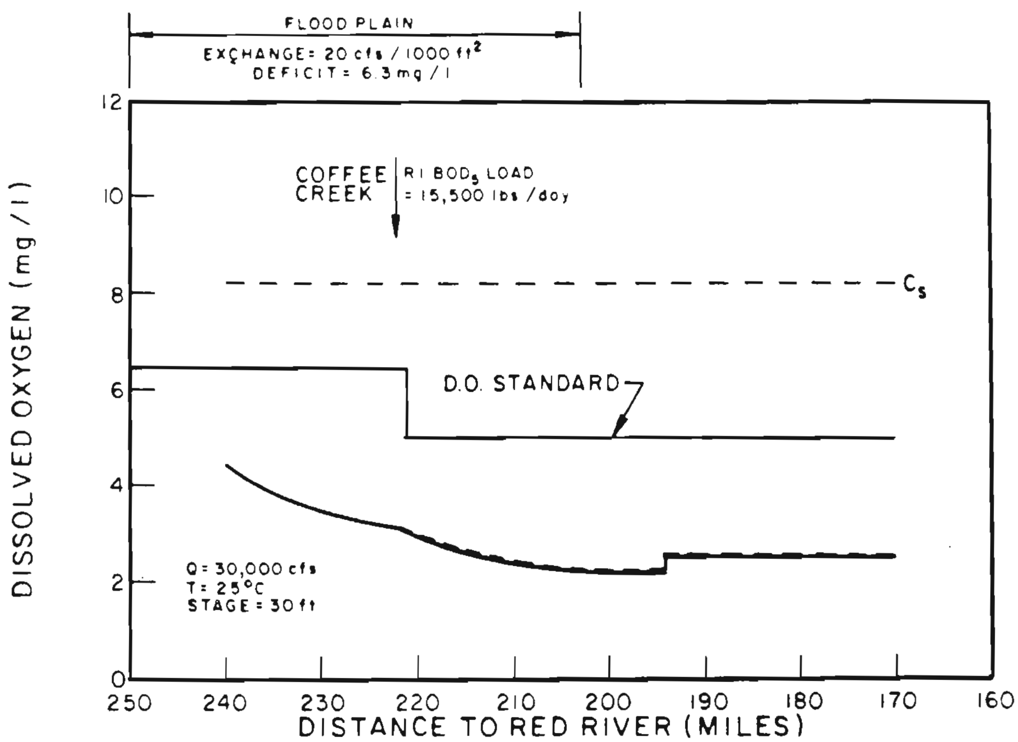


FIGURE 7-9. REPRESENTATIVE HIGH STAGE DISSOLVED OXYGEN PROJECTION

total deficit that is attributed to the load from the mill at Crossett.

The estimated impact of the Georgia Pacific discharge on dissolved oxygen levels of the Ouachita River, between Coffee Creek and Columbia Lock and Dam, at the August 7Q10 drought flow and a BOD_5 load of 3,407 lbs/day, is shown on the upper panel of Figure 7-10. Here, all other sources and sinks of dissolved oxygen deficit, including upstream sources, SOD and downstream point sources, have been set at zero. The deficit due to the mill, the difference between the dissolved oxygen saturation concentration and the model profile, has a maximum of 1.2 mg/L in the vicinity of MP 208. This deficit is primarily a result of carbonaceous BOD oxidation, although a minor component of about 0.1 mg/L is due to the dissoloved oxygen deficit of the effluent at the mouth of Coffee Creek. Further downstream of Bayou Bartholomew, near the City of Monroe (MP 158), the deficit due to the mill is estimated to be on the order of 0.15 mg/L. Downstream of this location, which is approximately 20 days travel time from Coffee Creek, the projection results indicate that the mill has a negligible impact on dissolved oxygen levels in the Ouachita River. It should be noted that these projections are based on steady state conditions in the river. Given the projected travel time of 100 days or more in the study area considered here, this assumption is clearly not appropriate. However, the results should indicate the order magnitude of the projected far downstream impacts.

The projected impact of Georgia Pacific's discharge during the flooded conditions used in these projections is illustrated on the lower graph of Figure 7-10. Due to the very high degree of dilution of 400 to 1 and the relatively short time of travel, on the order of several days, the impact of the load is negligible throughout the reach of the river between Coffee Creek and Columbia Lock and Dam. The low dissolved oxygen levels projected under flood conditions are almost entirely a result of non-point source background loads.

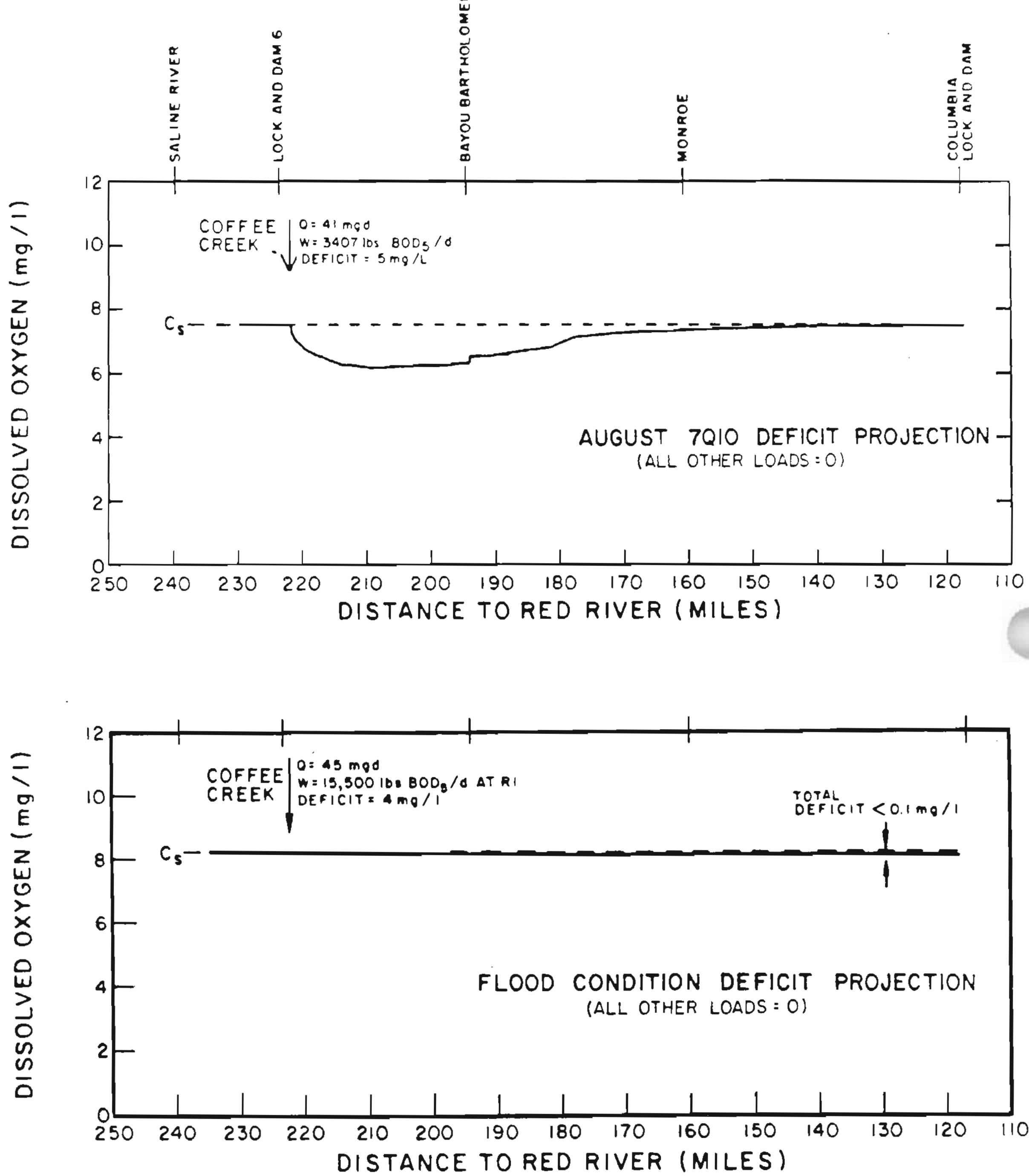


FIGURE 7-10. IMPACT OF GEORGIA PACIFIC DISCHARGE ON DO IN QUACHITA RIVER (COFFEE CREEK AND COLUMBIA LOCK AND DAM) DURING LOW STAGE AND HIGH STAGE CONDITIONS

SECTION 8

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