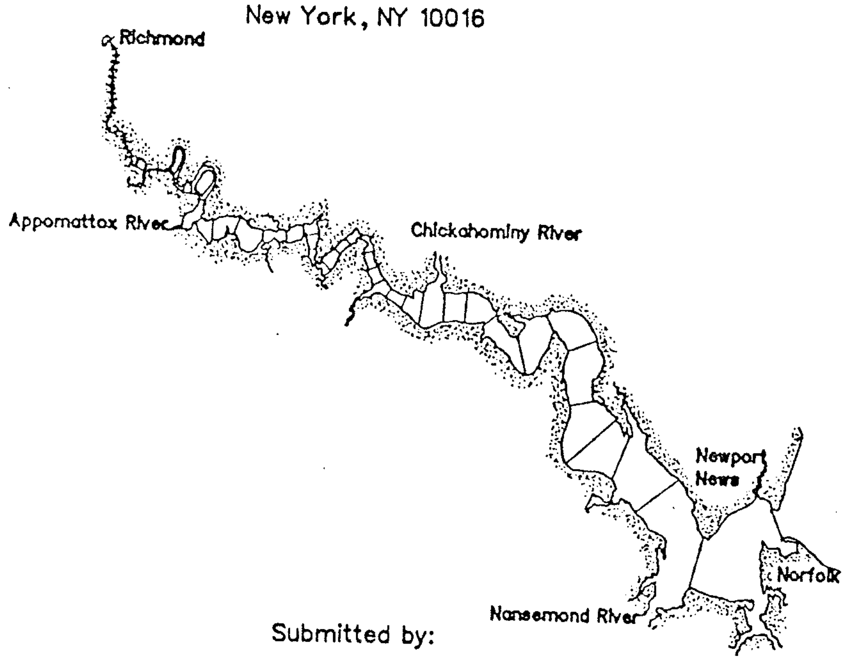
A Final Report

Eutrophication Modeling of James Estuary, Virginia

Submitted to:

The Soap and Detergent Association
475 Park Avenue South at 32nd Street
New York, NY 10016



Submitted by:

Wu-Seng Lung, PhD, PE
Principal Investigator

December 1987

SCHOOL OF ENGINEERING AND APPLIED SCIENCE

DEPARTMENT OF CIVIL ENGINEERING

UNIVERSITY OF VIRGINIA
CHARLOTTESVILLE, VIRGINIA 22901

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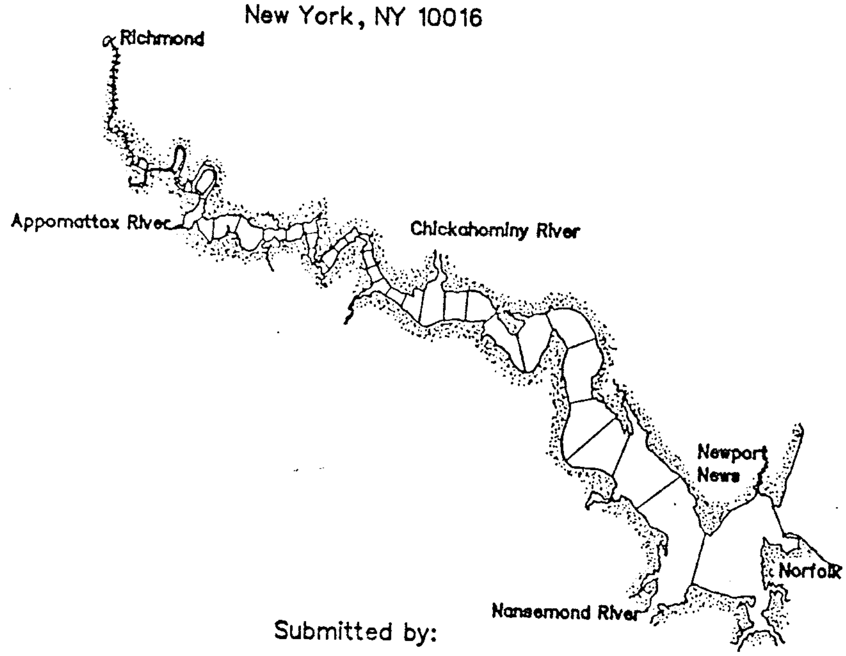

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Nile P. Testerman assisted the tracer labeling analysis of phosphorus for this study.

EXECUTIVE SUMMARY

A modeling study has been conducted to trace the fate and transport of nutrients (nitrogen and phosphorus) in the lower James River Estuary. The study consists of the following key technical tasks:

- Review of the results from an earlier modeling study of the upper estuary
- Expansion of the upper estuary model to include the lower estuary
- Analysis of available data in the lower estuary
- Calibration of the expanded model
- Model projections to quantify the fate and transport of nutrients

The following paragraphs highlight these tasks.

Model calibration results from a previous study on the upper estuary (Lung, 1986b) were reviewed for two separate data sets in July and September 1983. The review can be summarized as follows. Nutrients were not the limiting factors under the 1983 summer condition. Rather, turbidity in the water column offered considerable light attenuation. In fact, the turbid estuarine water provided over 80% reduction in algal growth rates from the optimum growth level.

The James River model (JMSRV), which was used in the last modeling study to assess the impact of phosphorus control on the water quality of the upper estuary, was expanded (from a 50-segment configuration to a 62-segment configuration) to include the lower estuary. Hydraulic geometry data of the lower estuary was derived from physical data and incorporated into the model.

Receiving water data was examined to select a complete data set for model calibration. A water quality monitoring study under the 208 program for the Hampton Roads area provided an intensive water quality survey in July 1976. At the same time, an intensive water quality survey was conducted for the upper estuary from Richmond to the Chickahominy River by the State Water Control Board (SWCB) and the Richmond-Crater 208 consortium in July 1976. Thus, a receiving water quality data set complete for the entire estuary from Richmond to Newport News was assembled for model calibration use. Point source discharge records at municipal and industrial facilities were obtained to independently quantify BOD and nutrient loads to the model. Municipal wastewater treatment facilities in the lower estuary were comprised of several plants operated by the Hampton Roads Sanitation District (HRSD) and their data were obtained from HRSD. Industrial loads were derived from SWCB records.

The expanded model was first applied to calibrate the July 1976 data set. Subsequently, the calibrated model was tested using the July and September 1983 data sets to provide additional confidence in the modeling framework. Results from the model calibration and validation analyses indicated that the seasonal conditions in the James River Estuary were reproduced by the model. While the July 1976 and July 1983 data sets had almost identical freshwater and temperature conditions in the estuary, peak chlorophyll a levels in these two surveys were quite different. Such a difference was explained by the model, suggesting a higher light extinction coefficient in the estuary in July 1976 was the cause. As a result, lower chlorophyll a levels were generated in July 1976 than July 1983.

Model projections were then conducted to quantify the relative importance of various nutrient sources in the upper estuary. Three major nutrient sources were examined: POTWs, industries, and upstream input. [A fourth one, nutrient releases from the sediment, was found to have a much smaller impact on the water quality than the three major sources.] Among the major sources evaluated, POTWs in the upper estuaries were found to play a dominate role in contributing to nutrient concentrations in the lower estuary.

The modeling analysis also indicated that while phosphorus controls in the upper estuary would increase inorganic nitrogen concentrations entering the lower estuary, their impact on the algal growth potential in the lower estuary would be insignificant. Algal growth in the lower estuary is highly suppressed due to considerable turbidity levels in the water column. Phosphorus control would reduce to lower concentrations, the relatively limited orthophosphate concentrations available for algal growth in the lower estuary.

A modeling study currently underway is designed to extend the analysis further downstream into the Chesapeake Bay such that the nutrient transport from the James River Basin can be accurately quantified within the context of this modeling framework. In addition, the question as to what role the nutrients from the James River basin play in contributing to eutrophication in Chesapeake Bay can be addressed in the current effort.

1. Introduction and Purpose

The James River basin (Figure 1) potentially contributes a significant amount of the phosphorus entering the Chesapeake Bay, ranging from 24% to 36% depending on the hydrologic conditions (Lung, 1986a). Such a high phosphorus input is due to the fact that none of the publicly owned treatment works (POTWs) in the basin currently practice phosphorus removal. In addition, there is no other form of nutrient control existing in the James River basin. [A phosphate detergent ban has been passed in Virginia and will become effective on January 1, 1988.] At the present time, approximately 15% to 30% of the total phosphorus loads to the Bay, again depending on the hydrologic condition, are from the POTWs in the James River basin on an annual basis. More importantly, POTWs account for about 55% to 75% of the total phosphorus loads from the James River basin with its majority coming from sources below the fall line (Lung, 1986a).

Results from a recent modeling study of point source phosphorus control in the James River basin indicate that while present nutrient levels in the <u>upper</u> James River Estuary are adequate to support algal growth, a reduction of nutrient inputs by removing phosphorus at POTWs would lead to a phosphorus limiting condition thereby lowering the phytoplankton biomass levels (Lung, 1985, 1986b). Under the 7-day 10-year low flow conditions, phosphorus removal would reduce present peak chlorophyll *a* level by 50% if an effluent total phosphorus limit of 2 mg/l is applied. Further reduction in the peak chlorophyll *a* level may be achieved with effluent limits of 1 mg/l, 0.5 mg/l, and 0.2 mg/l.

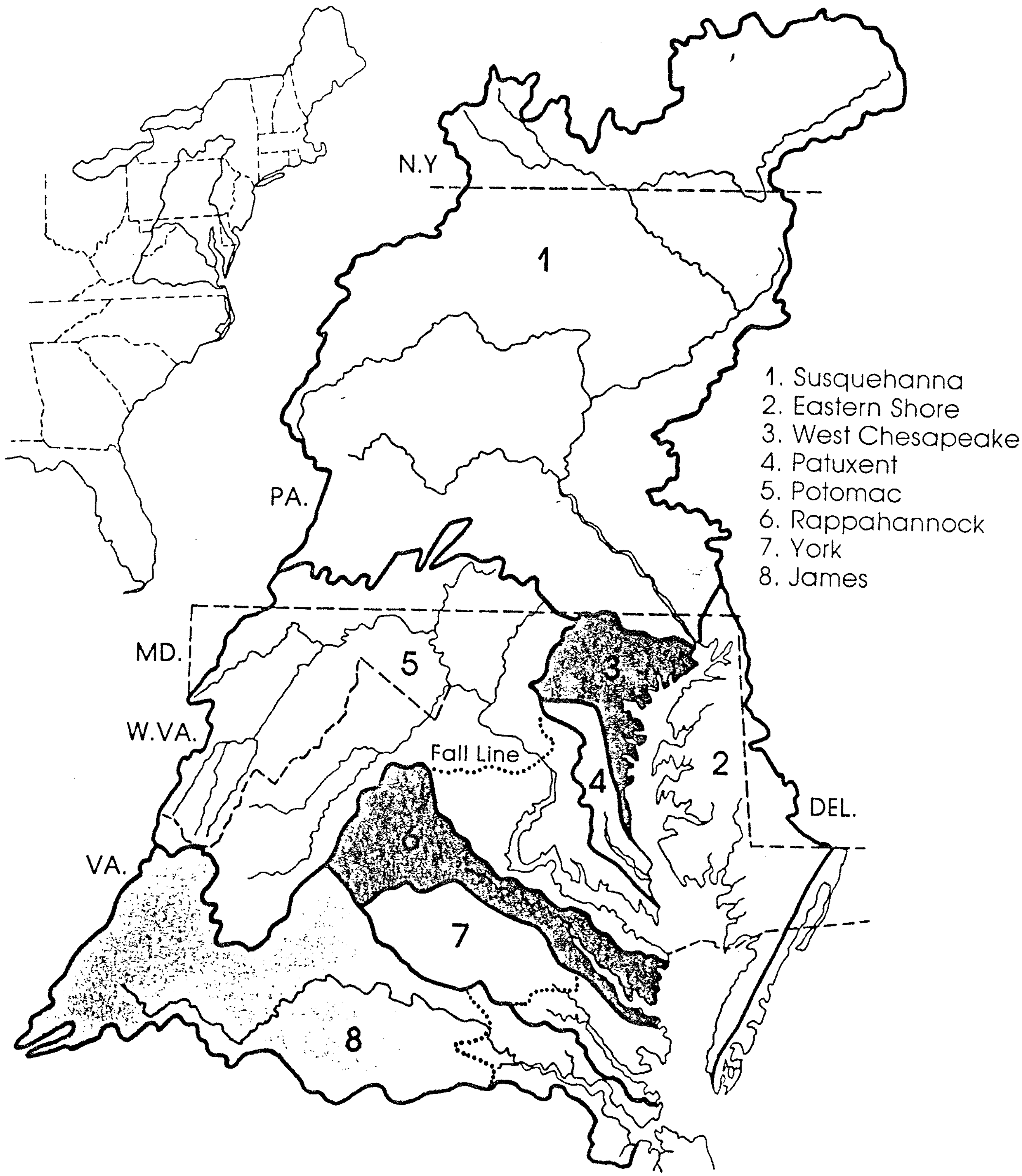


Figure 1. The Chesapeake Bay and James River Basin

[The modeling results also indicated that phosphate detergent bans would only slightly reduce the chlorophyll a levels.]

Under the phosphorus removal scenarios the inorganic nitrogen (NH_3, NO_2^-, and NO_3^-) concentrations in the estuary would increase in the downstream direction because they would not be utilized by the reduced algal biomass. This result raises an interesting equation: would phosphorus removal cause a nitrogen increase and result in greater production in the lower estuary and the Chesapeake Bay which are believed to be nitrogen limited?

The purpose of this study is to address the above question by expanding the <u>upper</u> James Estuary study to the <u>lower</u> estuary. In addition, recent data (Cerco, 1985) on sediment nutrient release fluxes and oxygen demand rates were incorporated into the expanded model to better understand the fate and transport of nutrients in the James River Estuary. That is, how much phosphorus originating from the upper estuary will enter the lower estuary under present conditions? Further, would nitrogen increases caused by phosphorus controls affect the algal growth potential in the lower estuary?

2. Results from Previous Modeling Study

Model calibration results from the previous study on the upper estuary (Lung, 1986b) are summarized in Figure 2 for two separate data sets in July and September 1983. In general, the increase in ammonia nitrogen below Richmond was due to the ammonia discharge from point sources such as the Richmond wastewater treatment plant and other POTWs and industrial facilities. A small portion of such an increase was probably due to the release of ammonia from the sediment in this section of the river (Cerco, 1985; HydroQual, Inc., 1986). However, the increase in ammonia does not sustain beyond river mile 90 because of

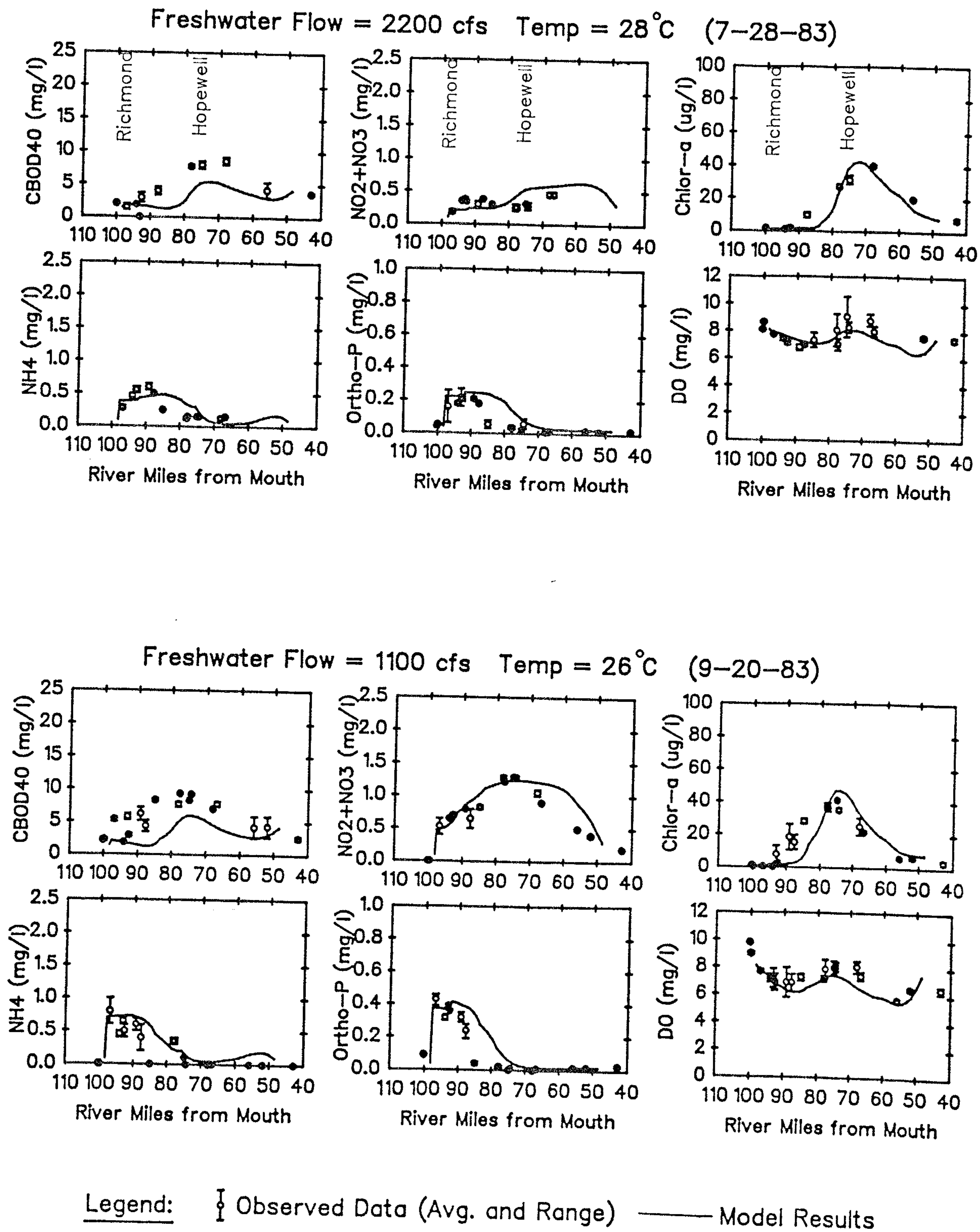


Figure 2. Model Calibration – Upper James Estuary 1983

phytoplankton uptake and nitrification. Note that the increase in phytoplankton chlorophyll a concentration started at this river reach. The orthophosphate profile in Figure 2 closely resembles the ammonia profile along the upper estuary. Again, the sharp increase in orthophosphate concentration was due to wastewater discharges from point sources and subsequent decrease in concentration was due to algal uptake. The lowest level of orthophosphate is about 0.01 mg/l of P, which is much higher than the Michaelis-Menton constant (0.001 mg/l) limiting the algal growth in the model.

Additional insights into the phytoplankton growth-nutrient dynamics may be obtained by examining the factors affecting the algal growth rate. Figure 3 presents the average depth of the water column, light extinction coefficient, effects of light and nutrients on algal growth, and algal growth rates in the upper James River Estuary under the September 20, 1983 condition (with freshwater flow at Richmond = 1,100 cfs and water temperature = 26°C). It should be pointed out that average channel depths below Richmond increase for the first 20 miles and then decrease, reaching some shallow sections near the Hopewell area. Because of the shallowness of the water column near Hopewell (less than 10 ft. deep), the algal growth rate reaches a local maximum (about 0.5/day) resulting in the peak of chlorophyll a levels shown at river mile 76 in Figure 2. The light extinction coefficient increases progressively from 1.3/m in the downstream direction, reaching a maximum of 3/m below Hopewell. Firstly, it is this high degree of light extinction that causes the algal biomass (in chlorophyll a) to decline following the biomass peak. [Phytoplankton settling is another cause of the decline of the biomass.] Secondly, light extinction levels along the

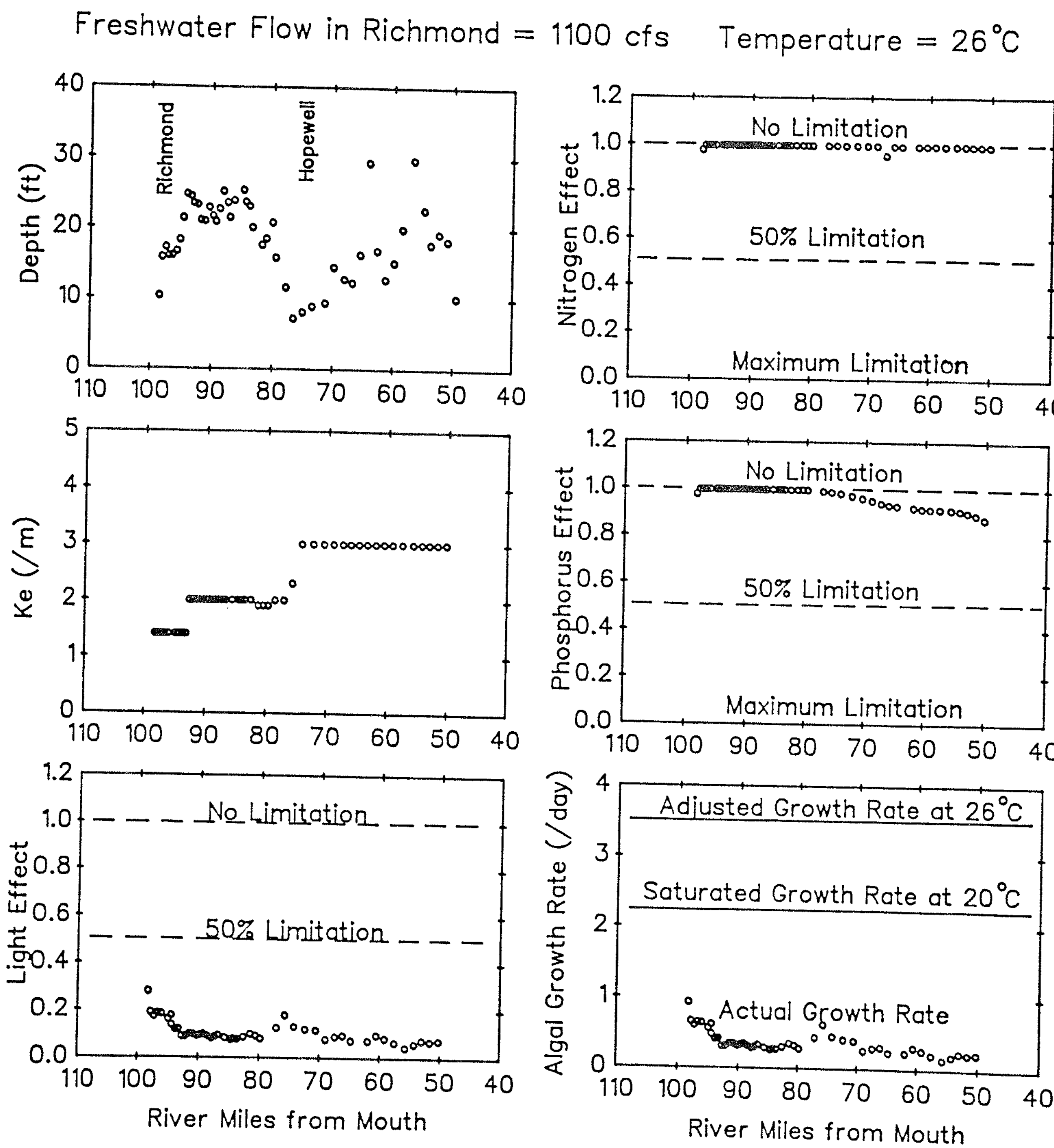


Figure 3.
Factors Affecting the Algal Growth Rate

entire study area were relatively high. As a result, light extinction is the principal factor limiting algal growth in the upper James Estuary. That is, the turbid water reduces algal growth rates by over 80% from the optimum growth rate. On the other hand, nutrient (nitrogen and phosphorus) limitations on algal growth were relatively small compared with light limitation. Finally, actual growth rates along the upper estuary have a profile very similar to that of the light effect, further implying the importance of light attenuation in the water column. The growth rates were much lower than the optimum levels at 20°C (2.2/day) and at 26°C (3.5/day).

The above discussions indicate that light attenuation, rather than nutrient levels is the key factor controlling the algal growth in the upper James River Estuary at the present time.

3. Approach and Methods

To address questions related to the fate and transport of nutrients in the lower James Estuary, a model for the entire estuarine system is needed. In this study, the James River model (JMSRV) used in the study of the upper estuary (Lung, 1986b) was used. The first task was to expand the upper estuary model into the lower estuary. The expanded model was then calibrated with available data to provide credibility. Subsequently, the calibrated model was employed to evaluate the impact of various nutrient control alternatives and to determine the fate of nutrients originating from the upper basin in the estuary. The following sections describe the technical tasks for this study.

3.1 Model Modifications

The James River model (JMSRV), which was used in the last study (Lung, 1986b) to assess the impact of phosphorus control on the water quality of the upper James River Estuary, was expanded (from a 50-segment configuration to a 62-segment configuration) to include the lower estuary. Basically, 12 segments were added to the upper estuary model. Hydraulic geometry of the lower estuary was derived from physical data and incorporated into the model. Table 1 lists the hydraulic geometry for the entire estuarine system under tidally averaged conditions. The upstream model boundary remained the same as in the last study (Lung, 198b), i.e., near Richmond, while the downstream model boundary has been extended to the mouth of the river. Figure 4 shows the segmentation for the James River Estuary used in this study. Model kinetic interrelationships between model system variables in each segment are shown in Figure 5. No changes in model kinetics were made for this study. A complete description of the model kinetics formulation has been presented elsewhere (Hydroscience, Inc., 1980; Lung, 1985).

3.2 Receiving Water Data

Water quality data needed to calibrate the expanded model for the entire estuary is quite limited. A water quality monitoring study under the 208 program for the Hampton Roads area conducted by the Virginia Institute of Marine Sciences (VIMS) included an intensive water quality survey in July 1976 and two slack water quality surveys in August 1976 for the lower estuary from the Chickahominy River to the mouth (Kuo, 1986). In addition, an intensive water quality survey was conducted for the upper estuary from Richmond to the Chickahominy River by the State Water Control Board in cooperation with VIMS and by the Richmond-Crater 208 consortium in July 1976. Thus, a receiving water quality data set for the entire estuary from Richmond to Newport News can be assembled

Table 1.

REACH NO.	MILES FROM MOUTH	TOTAL AREA SQUARE FEET	REACH VOLUME MILL CU FT	REACH DEPTH FEET	TIDAL VELOC FEET / SEC
1	98.50	4500.00	11.40	10.30	0.00
2	98.10	6272.00	21.80	15.80	0.11
3	97.50	7470.00	20.80	17.20	0.14
4	97.00	8322.00	25.40	16.00	0.17
5	96.40	7711.00	27.90	16.10	0.23
6	95.70	7384.00	20.60	16.70	0.32
7	95.20	8220.00	24.20	18.30	0.32
8	94.70	10098.00	33.60	21.40	0.31
9	94.10	11125.00	46.30	24.80	0.37
10	93.50	18133.00	42.40	24.50	0.27
11	93.00	14012.00	43.30	23.50	0.36
12	92.40	13345.00	43.50	23.30	0.39
13	91.80	14103.00	45.10	21.10	0.42
14	91.20	14361.00	51.30	21.00	0.45
15	90.50	13383.00	46.10	22.90	0.52
16	89.90	15732.00	38.90	21.70	0.46
17	89.40	13710.00	43.90	20.90	0.54
18	88.80	14031.00	49.10	22.70	0.54
19	88.20	16957.00	65.10	25.20	0.45
20	87.50	18252.00	35.40	23.60	0.43
21	87.10	15282.00	56.70	21.50	0.52
22	86.40	15413.00	128.00	23.90	0.61
23	84.90	16957.00	35.80	25.40	0.74
24	84.50	16928.00	54.20	23.70	0.91
25	83.90	17287.00	48.20	23.10	0.97
26	83.40	19199.00	175.00	20.10	0.97
27	81.80	22207.00	75.10	17.60	1.08
28	81.10	18450.00	105.00	18.50	1.33
29	80.10	21249.00	63.40	20.80	1.16
30	79.50	18792.00	216.00	15.80	1.33
31	77.80	29336.00	231.00	11.60	1.24
32	76.50	38066.00	331.00	7.20	1.62
33	75.00	45442.00	417.00	8.10	1.59
34	73.40	53309.00	625.00	8.90	1.59
35	71.30	59344.00	426.00	9.40	1.59
36	69.90	55868.00	543.00	14.50	1.84
37	68.10	58361.00	444.00	12.80	1.96
38	66.80	70909.00	530.00	12.30	1.71
39	65.50	83457.00	575.00	16.30	1.54
40	64.10	72000.00	488.00	29.40	1.82
41	62.70	60000.00	499.00	16.90	2.22
42	61.30	74990.00	499.00	12.80	1.80
43	59.90	60000.00	440.00	15.20	2.32
44	58.50	59148.00	557.00	20.00	2.43
45	56.70	58000.00	736.00	29.80	2.52
46	54.90	96824.00	553.00	22.70	1.53
47	53.80	93444.00	697.00	17.80	1.62
48	52.40	95201.00	682.00	19.30	1.63
49	51.00	89198.00	788.00	18.30	1.84
50	49.50	109727.00	1205.00	10.20	1.64
51	47.80	158694.00	1209.00	20.10	1.45
52	46.30	147929.00	1089.00	12.10	1.40
53	44.90	146765.00	1900.00	14.60	1.39
54	42.30	129638.00	2140.00	17.30	1.82
55	39.40	149165.00	2549.00	14.60	1.55
56	36.60	195684.00	3280.00	12.00	1.22
57	33.20	169355.00	3360.00	12.60	1.51
58	29.70	194434.00	3349.00	16.20	1.46
59	26.40	189468.00	3939.00	14.80	1.64
60	23.40	307514.00	8589.00	12.40	1.03
61	18.10	306683.00	8480.00	14.30	1.17
62	13.00	2200000.00	1510080.00	10.00	1.50

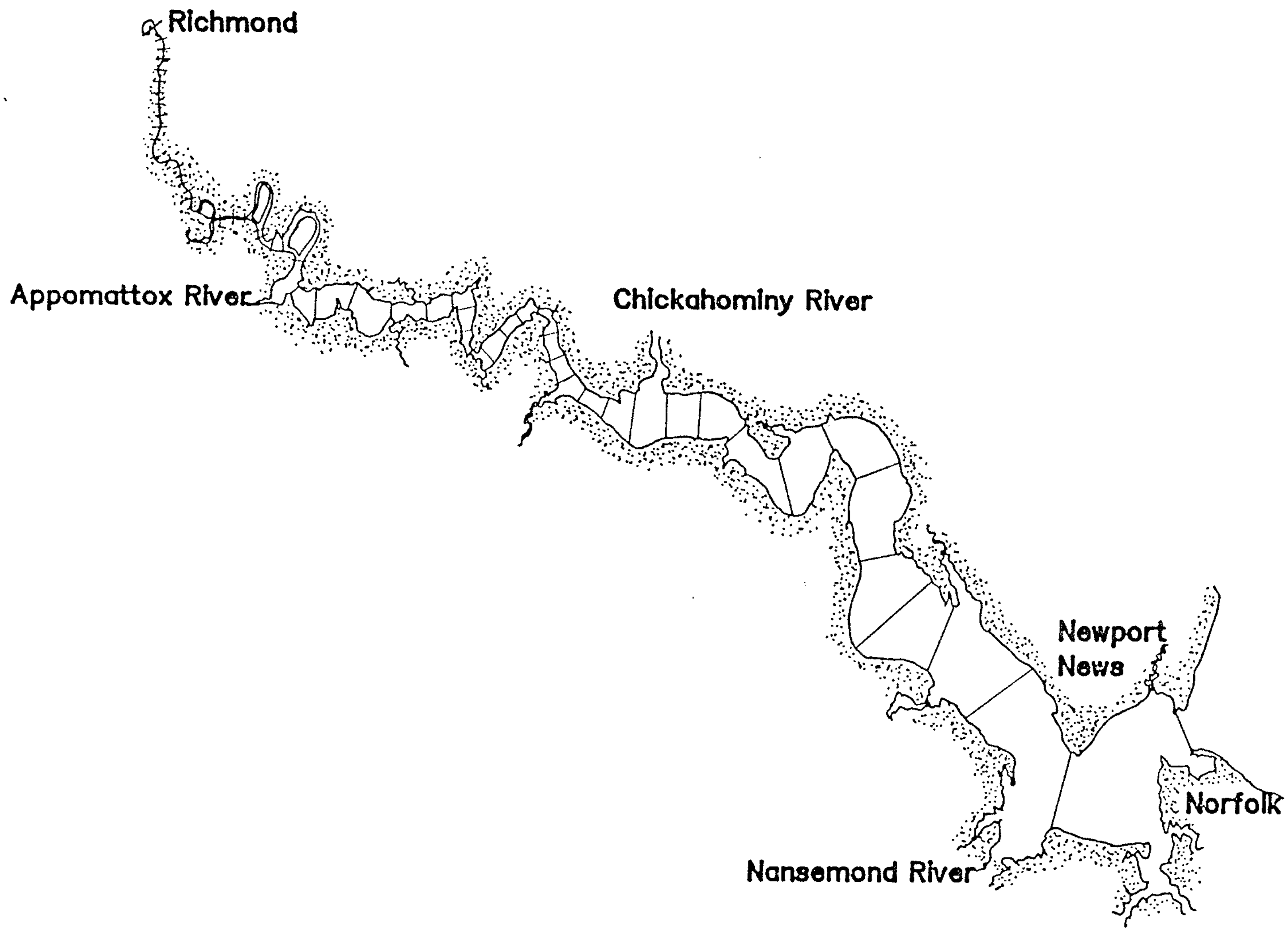


Figure 4. Model Segmentation

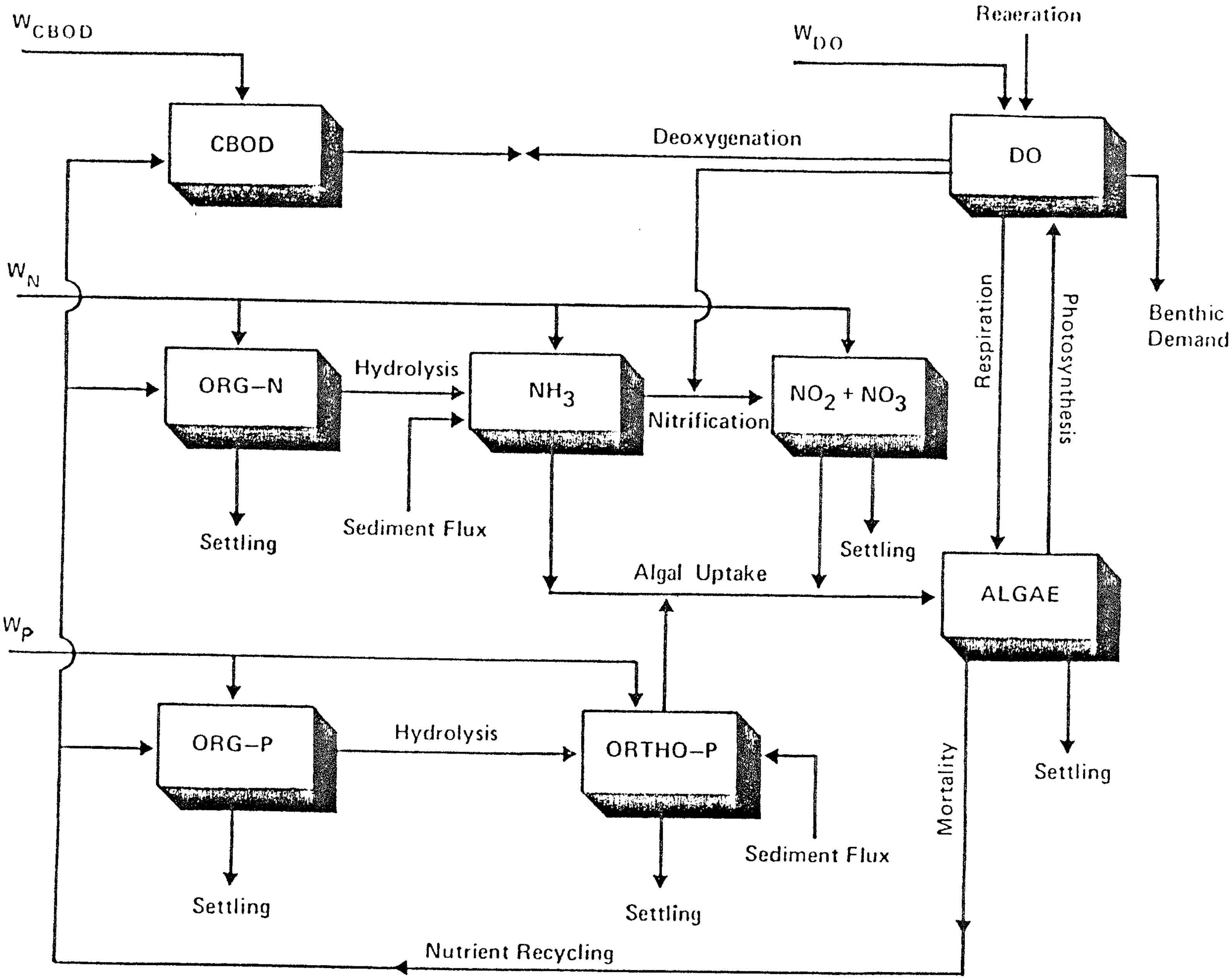


Figure 5. Model Kinetic Interrelationships

from these water quality monitoring studies. Sampling locations for these studies are shown in Figure 6. In general, the water quality constituents analyzed from samples included total Kjeldahl, ammonia, nitrite and nitrate nitrogen; orthophosphate and total phosphorus; chlorophyll a; $CBOD_5$ and $CBOD_{30}$; dissolved oxygen; and several physical parameters such as temperature, pH, conductivity, and salinity. These historic data were obtained from VIMS for this study (Anderson, 1986). Subsequent analyses of the data reduced them to a form suitable for model calibration in this study. A summary of the July 1976 data for the entire estuarine system is presented in Figure 7. Maximum, minimum, and average values over a tidal cycle are shown for key water quality constituents: ultimate CBOD, organic nitrogen, ammonia, nitrite + nitrate, organic phosphorus, orthophoshate, chlorophyll a, and dissolved oxygen. Figure 7 shows that the trends of water quality in the upper estuary (from mile 100 to mile 40) in July 1976 are very much similar to that observed in July 1983 (see Lung, 1985) except the peak chlorophyll a level was much lower in 1986. Subsequent modeling analysis examined probable causes for such a difference in peak chlorophyll a levels. In the lower estuary from Williamsburg to Newport News, no particular trend existed in July 1976. Although there are a few wastewater treatment plants (e.g., Williamsburg, Boat Harbor, James River, etc.) discharging their effluents into the lower James River, their impacts are relatively insignificant as the flow and river channel width are much larger in the lower estuary than the upper estuary.

The slack water data collected in the lower estuary in August 1976 were not used in this study because no synoptic data was collected in the upper estuary.

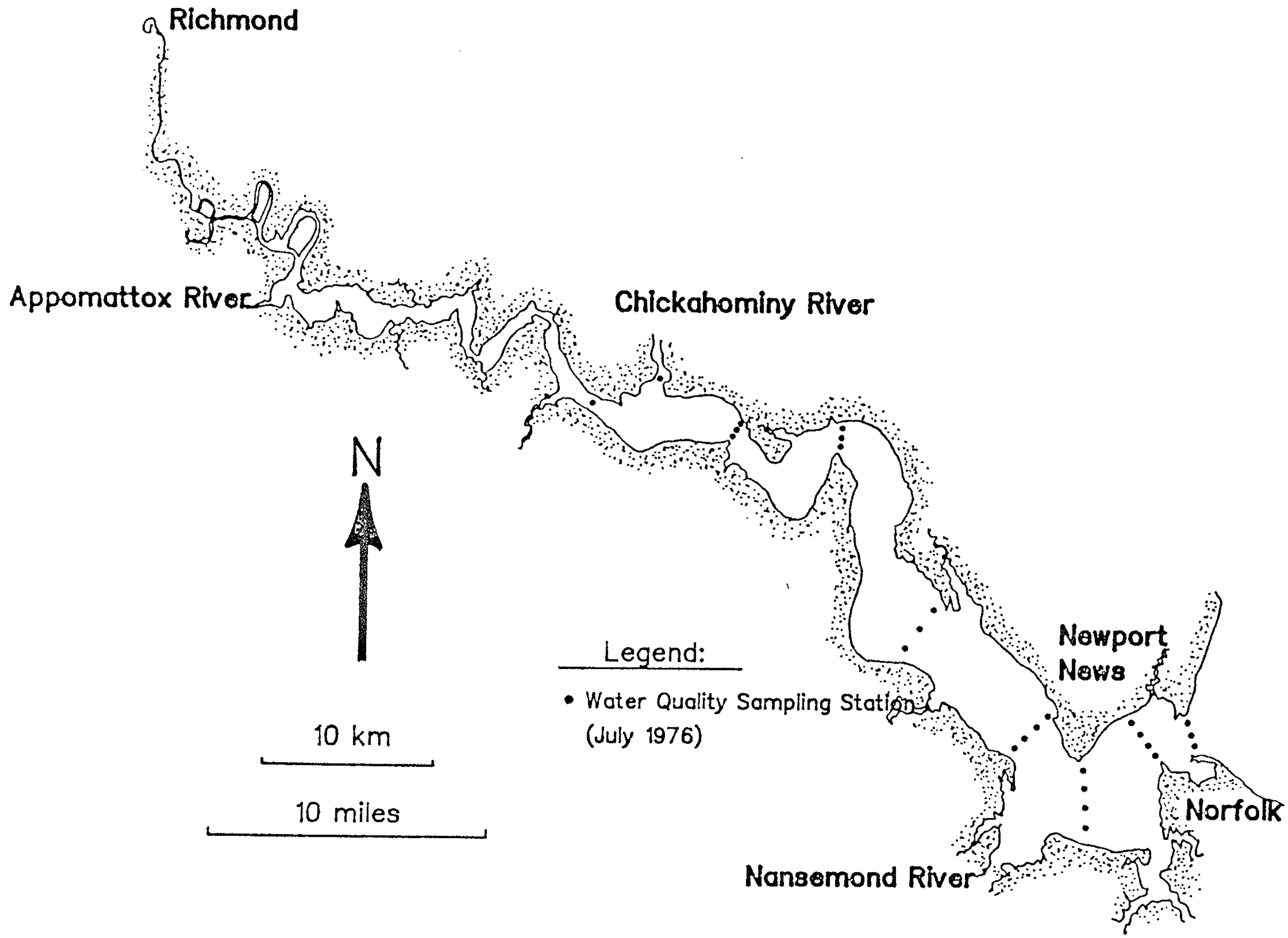
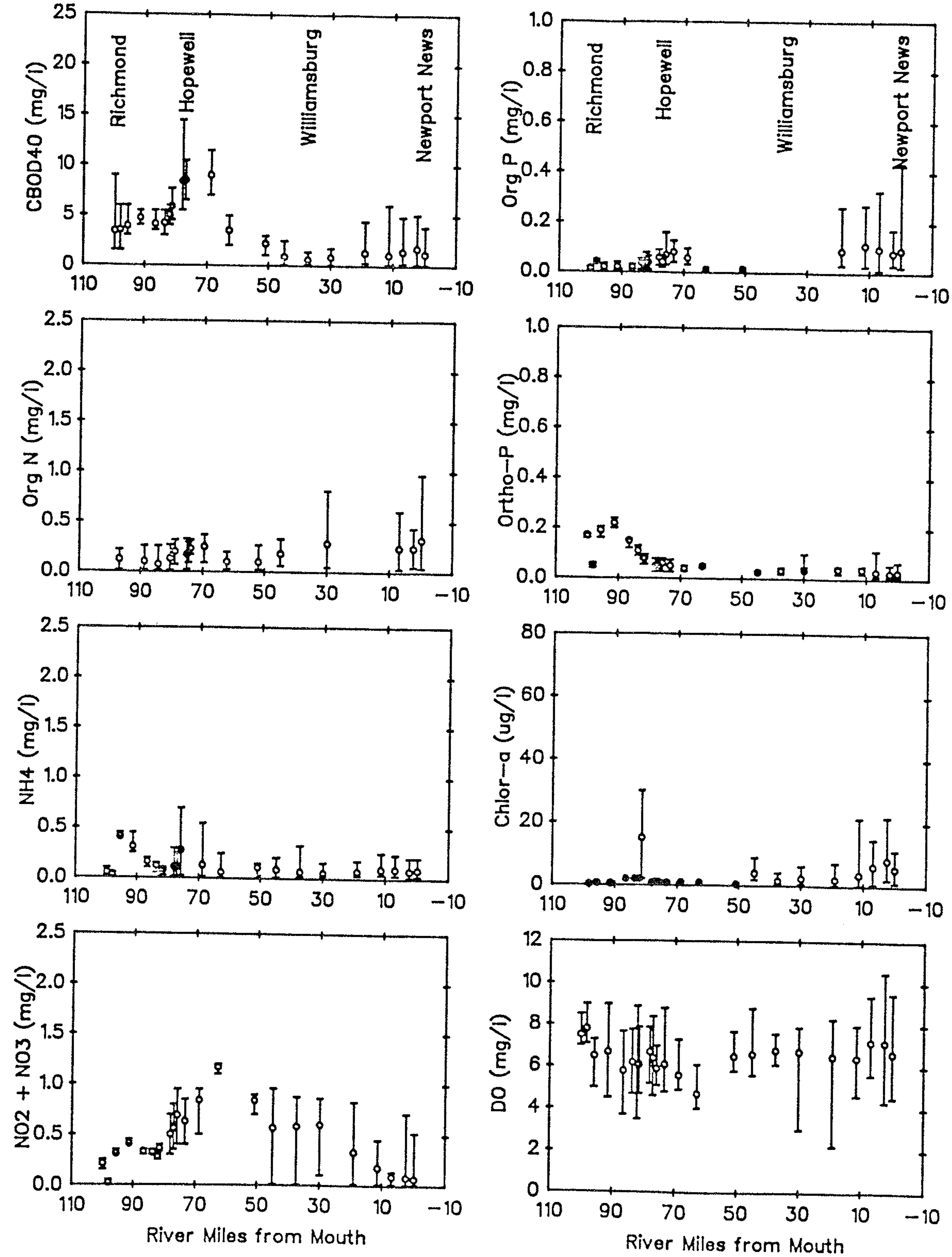


Figure 6. Water Quality Sampling Stations — July 1976



Legend: Observed Data (Avg. and Range)

Figure 7. Water Quality Conditions of the James Estuary July 1976

3.3 Point Source Loadings

Point source discharge records at municipal and industrial facilities were obtained from various sources. BOD and nutrient loads in the upper estuary for the July 1976 survey have been compiled by Hydroscience, Inc. (1980) and were readily available. Similar data for the summer 1983 surveys of the upper estuary have been compiled by Lung (1985) for modeling use from recent surveys (Grizzard and Weand, 1984). Loading rates from the treatment plants in the lower estuary were obtained from the Hampton Roads Sanitation District (Lawrence, 1987). Industrial discharges were obtained from SWCB records. The point source loading rates are summarized in Tables 2, 3, and 4 for the July 1976, July 1983, and September 1983 surveys, respectively.

3.4 Freshwater Flows

The river flow data, particularly those in the James River and Kanawha Canal near Richmond, the Appomattox and Chickahominy Rivers were obtained from the U.S. Geological Survey's surface water records. Figure 8 presents the hydrograph at these gaging stations during the summer of 1976. Climatological data from Byrd Airport for the two weeks prior to the July 1976 survey show rainfall occurred on July 15, 22, 24, 28 and 29 yielding 0.68, 0.17, 0.57, 0.07 and 0.19 inches, respectively. It is understood that these rainfall events were localized and did not produce significant runoff or sewer overflows (Hydroscience, Inc., 1980). The survey period, therefore, provided suitable hydrological and climatological conditions for a steady state modeling analysis.

3.5 Procedure of Analysis

The expanded model was used to first analyze the July 1976 data. Model coefficient values developed in the previous modeling study (Lung,

Table 2. Major Wastewater Loadings (lbs/day) for July 1976 Condition

Discharger	$CBOD_u$	Org.N	NH_3	NO_2+NO_3	Total P	Org.P	Ortho-P
Richmond	4014	803	4347	1596	2229	557	1672
DuPont	2766	108	38	422	64	16	48
Falling Creek	615	99	773	20	337	68	269
Am. Tobacco	57	56	4	185	39	5	34
Philip Morris	73	25	2	54	98	49	49
Allied-Chester	4664	151	64	185	41	18	23
Allied-Hopewell	28437	871	11885	2903	126	74	52
Hopewell	3476	82	494	2	162	39	123
Hopewell Industries	99192	515	438	117	115	49	66
Williamsburg		466	1515	6	627	287	340
James River		48	844	62			223
Boat Harbor		1415	2846	16	1444	756	688
Army Base		837	2123	10	960	454	506
Lamberts Point		1927	3683	12	1740	865	875

Table 3. Major Wastewater Loadings (lbs/day) for July 28, 1983 Condition

Discharger	$CBOD_{40}$	Org.N	NH_3	NO_2+NO_3	Total P	Org.P	Ortho-P
Richmond	5642	1282	3216	1379	2314	144	2170
DuPont	427	217	0	63	12	6	6
Falling Creek	1067	398	328	311	461	109	351
Proctors Creek	312	312	45	36	156	64	91
Reynolds Metals	3	0	0	0	0	0	0
American Tobacco	16	60	14	3	40	17	22
ICI	17	8	0	4	1	1	0
Philip Morris	485	26	8	351	140	52	88
Allied-Chester	3859	46	3	35	0	0	0
Allied-Hopewell	16502	1163	1055	1514	60	47	13
Hopewell	10347	5046	6989	429	322	119	203
Williamsburg	306	176	54	8	160	40	120
James River	198	189	905	308	775	1	774
Boat Harbor	924	188	2818	13	792	2	790
Nansemond	710	102	519	169	279	10	269
Army Base	98	51	2017	10	450	0	450
Lamberts Point	18400	921	3201	17	385	253	132

Table 4. Major Wastewater Loadings (lbs/day) for September 20, 1983 Condition

Discharger	$CBOD_{40}$	Org.N	NH_3	NO_2+NO_3	Total P	Org.P	Ortho-P
Richmond	4512	4927	3916	2332	2328	144	2184
DuPont	202	230	38	9	5	2	2
Falling Creek	714	336	116	745	502	111	390
Proctors Creek	2602	208	103	33	179	25	154
Reynolds Metals	1	3	0	2	2	2	0
American Tobacco	60	27	1	31	6	0	6
ICI	31	8	0	4	1	1	0
Philip Morris	368	27	6	267	106	39	66
Allied-Chester	2480	42	3	61	9	6	3
Allied-Hopewell	12680	3363	2069	2349	80	66	13
Hopewell	8929	7048	5904	326	347	205	142
Williamsburg	229	15	196	65	162	125	37
James River	436	221	878	25	534	187	347
Boat Harbor	410	340	2719	13	867	167	700
Nansemond	770	178	938	34	362	12	350
Army Base	413	393	2063	11	569	263	306
Lamberts Point	21893	520	3087	17	418	332	86

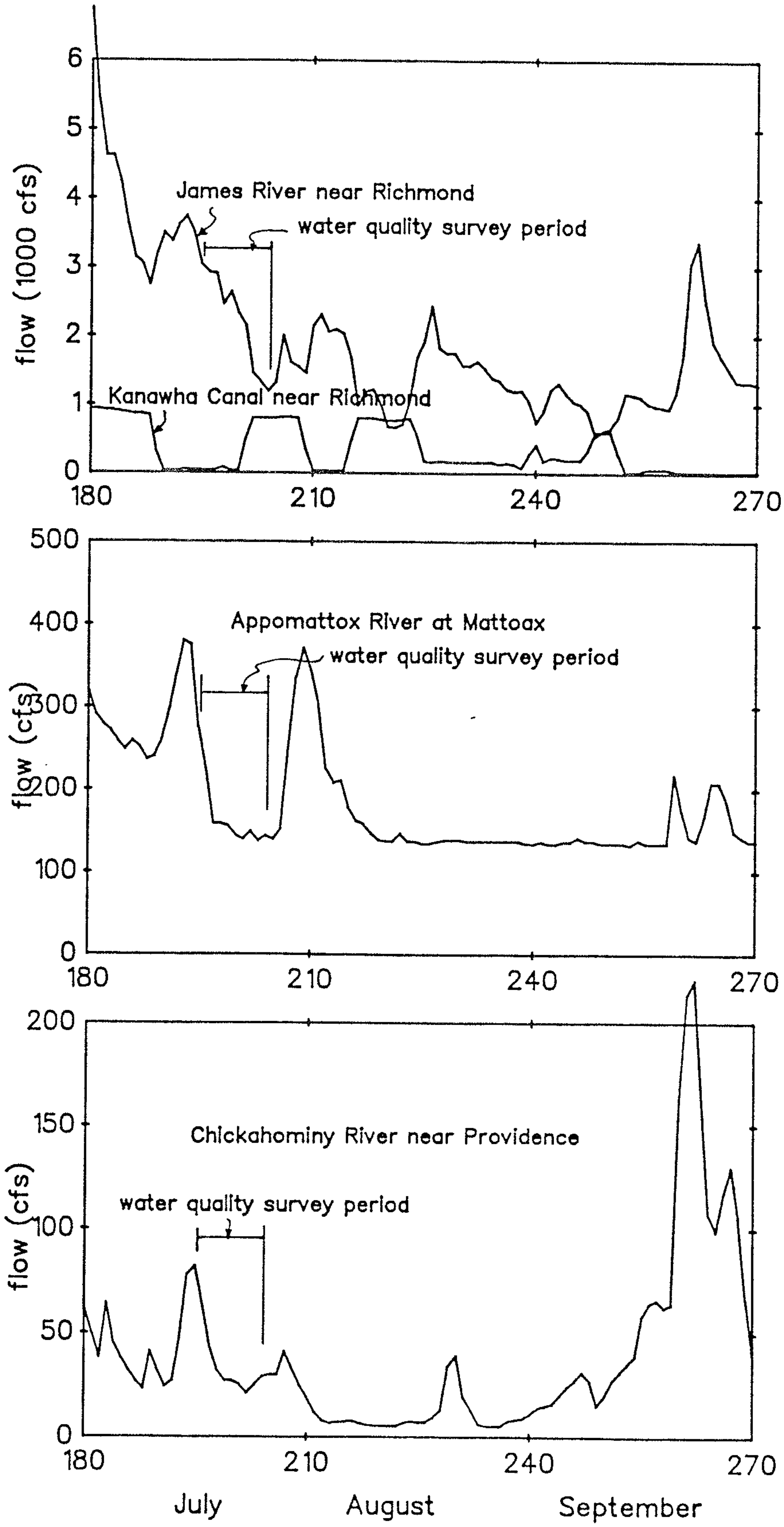


Figure 8. Hydrographs of Summer 1976

1986b) were adopted. The expanded model was further tested using the data from July and September 1983 although no data in the lower estuary was available during these two surveys. In addition, information from a recent study of BOD wasteload allocations by HydroQual, Inc. (1986) was reviewed and incorporated, as appropriate, for this study. Further, sediment nutrient fluxes and oxygen demand rates measured by Cerco (1985) were also analyzed and incorporated. Incorporating the additional information resulted in small changes in model coefficients. Subsequently, the updated set of model coefficients was retested using the July 1976 data to substantiate the model validity.

The final step of analysis was using the calibrated model to quantify the contribution of various nutrient sources in the upper estuary to the water quality conditions in the lower estuary. The emphasis of the analyses was quantifying the fate and transport of nutrients in the James River Estuary under present conditions.

4. Model Calibration and Sensitivity Analyses

Calibration and validation of the expanded JMSRV model of the entire James River Estuary was performed using three data sets (July 1976, July 1983 and September 1983). Normally, the transport (i.e., physical displacement of constituents by advection and dispersion) is first calibrated in modeling analyses. Transport calibration and validation involves the determination of appropriate physical characteristics such as channel dimensions and dispersion coefficients such as channel dimensions and dispersion coefficients so that mass movement can be accurately defined over a range of flow conditions. The next step is to calibrate the kinetic model components (i.e., alterations through chemical or biological reactions) that affect changes in water quality constituents in space and time. Calibration and validation of kinetic coefficients involve the determination of the major biochemical influences existing in the system and the quantification of the rates at which they proceed.

The calibration process uses one set of observed data to define the various model coefficients that lead to calculated water quality profiles representative of the prototype system. The validation process uses a second set of observed data, substantially different from the first, to ensure that the coefficients selected during calibration are indeed capable of representing water quality behavior over a range of environmental conditions. Once a common set of model coefficients capable of providing calculated profiles representative of estuarine water quality over a range of flow, temperature or loading conditions is determined the model is judged validated and capable of projecting receiving water quality in response to hypothetical flows or loadings. The environmental conditions associated with these three data sets chosen for model calibration and validation are summarized as follows:

Date/Period	Freshwater Flow (cfs) at Richmond	Water Temperature (°C)
July 1976	2,300	28
July 1983	2,200	28
September 1983	1,100	26

4.1 Calibration of Mass Transport

Parameters defining system geometry, including segment volumes, cross-sectional areas and depths, shown in Table 1 were applied in the present study. The value of the tidal dispersion coefficient was determined by reproducing the results of dye test simulations in the U.S. Army Corps of Engineers physical model of the James River Estuary, using the mathematical model (Hydroscience, Inc., 1980). As such, the July 1976 mass transport pattern calibrated by Hydroscience, Inc. was used in this study. Mass transport patterns for July and September 1983 have been calibrated by Lung (1986b) and were used in the present study.

4.2 Model Results for July 1976

The freshwater flow near Richmond (including the James River and Kanawha Canal) used for this analysis was derived from the hydrographs shown in Figure 8 and was equal to 2300 cfs. BOD and nutrient loading rates listed in Table 2 were incorporated into the model. Available light intensity and water temperature were about 450 Langley/day and 28°C, respectively (Hydroscience, Inc., 1980). Model loads from point sources are presented in Table 2. Model calibration results are summarized in Figure 9 for ultimate CBOD, organic nitrogen, NH_4^+, NO_2^- + NO_3^-, organic phosphorus, orthophosphate, chlorophyll a, and dissolved oxygen. Calculated water quality profiles are in general agreement with observed conditions in the prototype system. Phytoplankton biomass levels were modest in the system. In regard to demands on the dissolved oxygen resources of the river, the model provides an accurate representation of prototype conditions.

4.3 Model Results for July and September 1983

The recent report by HydroQual, Inc. (1986) suggested some slight changes in several model coefficients such as saturated algal growth rate and oxygen to chlorophyll a ratio in biomass. These changes were incorporated in the model as part of the recalibration using the July and September 1983 data. In addition, adjustment on sediment oxygen demand rates was made according to the information from Cerco (1985). Cerco's measurements on nutrient releases from the sediment indicated

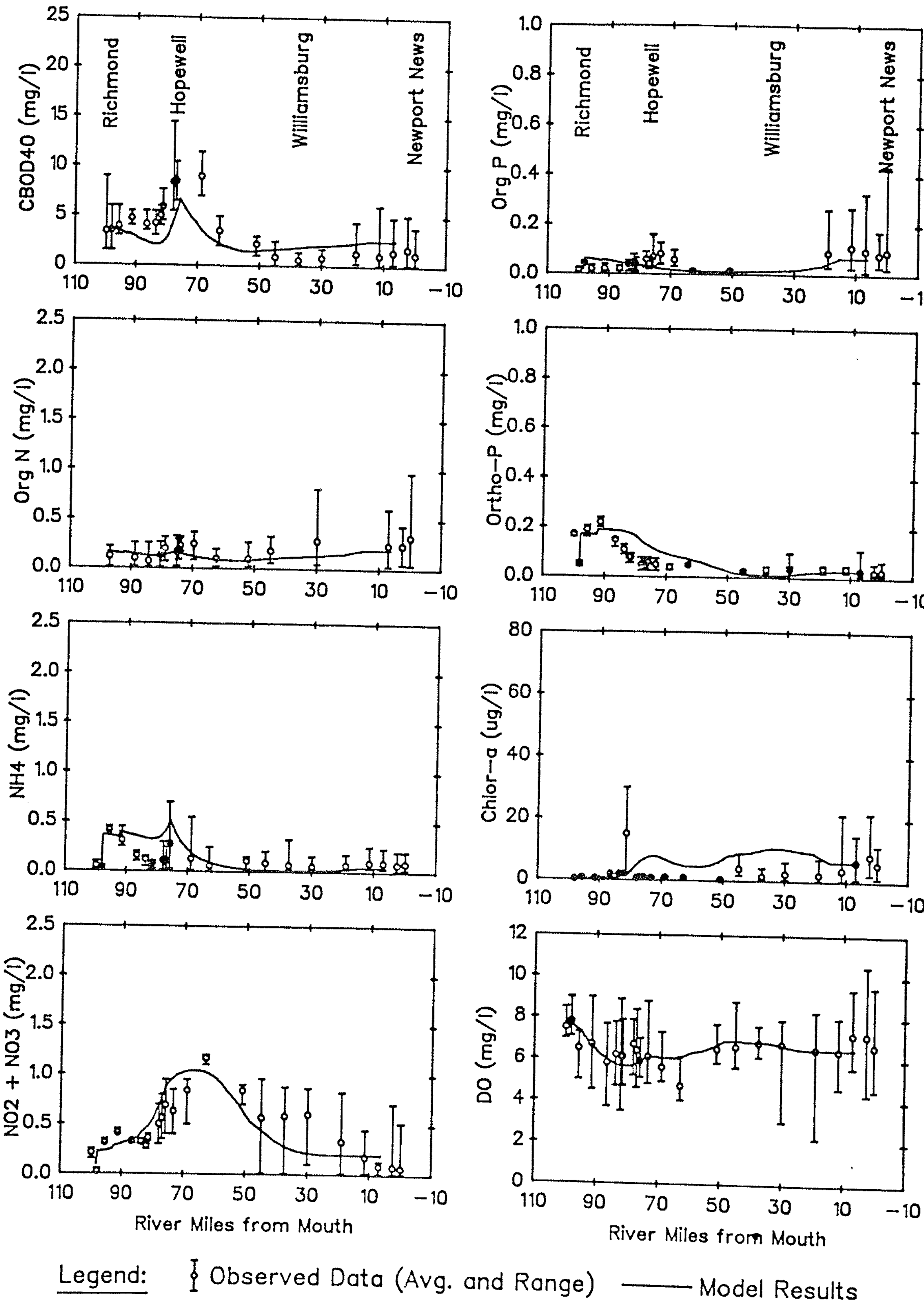


Figure 9. Model Calibration (July 1976 Data)

only modest fluxes for ammonia and orthophosphate (HydroQual, Inc., 1986). Thus, small adjustments were made in the model for this study to accommodate this information. The updated model produced results matching the observed data in the July and September 1983 surveys (Figures 10 and 11, respectively). In general, the results for the upper estuary are very similar to the previous model results as shown in Figure 2. It is therefore concluded that the model is adequately validated for projection purposes.

4.4 Sensitivity Analyses

It should be pointed out that while the freshwater flow rates were almost the same between the July 1983 and July 1976 surveys, algae achieved a much higher peak biomass level in 1983 (over 40 μg/l chlorophyll *a*) than in 1976 (below 10 μg/l chlorophyll *a*). A close examination of the model results as well as the available data indicated that the light extinction coefficient in the water column near the Hopewell area was much higher (up to 4.6/m) in July 1976 than that in July 1983. Such a result further substantiates the finding reported in the preceding study that light is a dominating factor in controlling the algal growth potential in the upper James River Estuary. As expected, nutrient supplies were sufficient in both the July 1976 and July 1983 surveys and were not limiting the algal growth.

A sensitivity run of the model was made to better demonstrate the effect of light on the algal growth by incorporating the light extinction coefficients from the July 1983 condition into the July 1976 model run. Figure 12 presents the results of such an analysis. With lower light extinction coefficients, the algal biomass would reach a peak of 44 μg/l near the Hopewell area, a response very similar to that

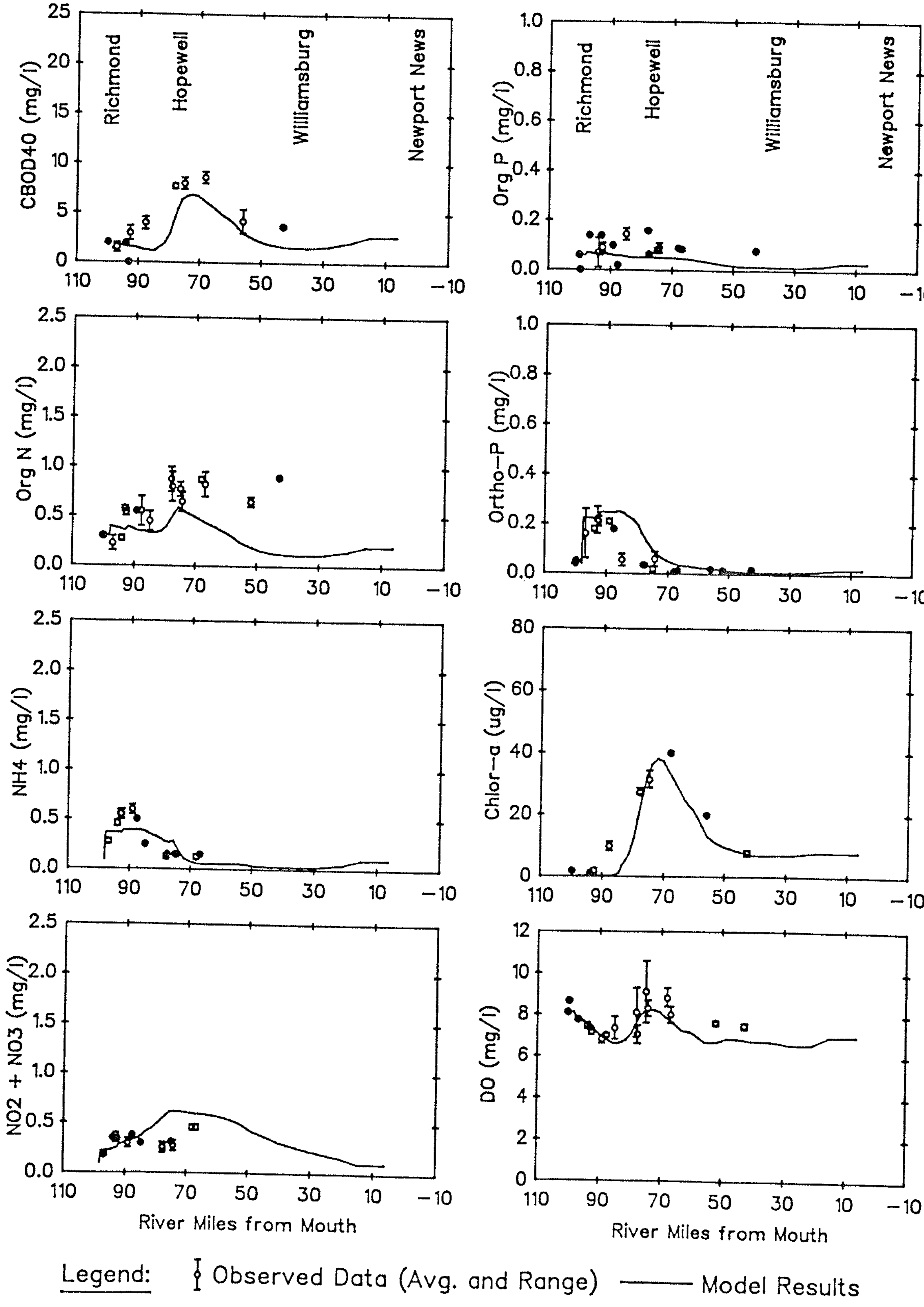


Figure 10. Model Calibration (July 28 1983 Data)

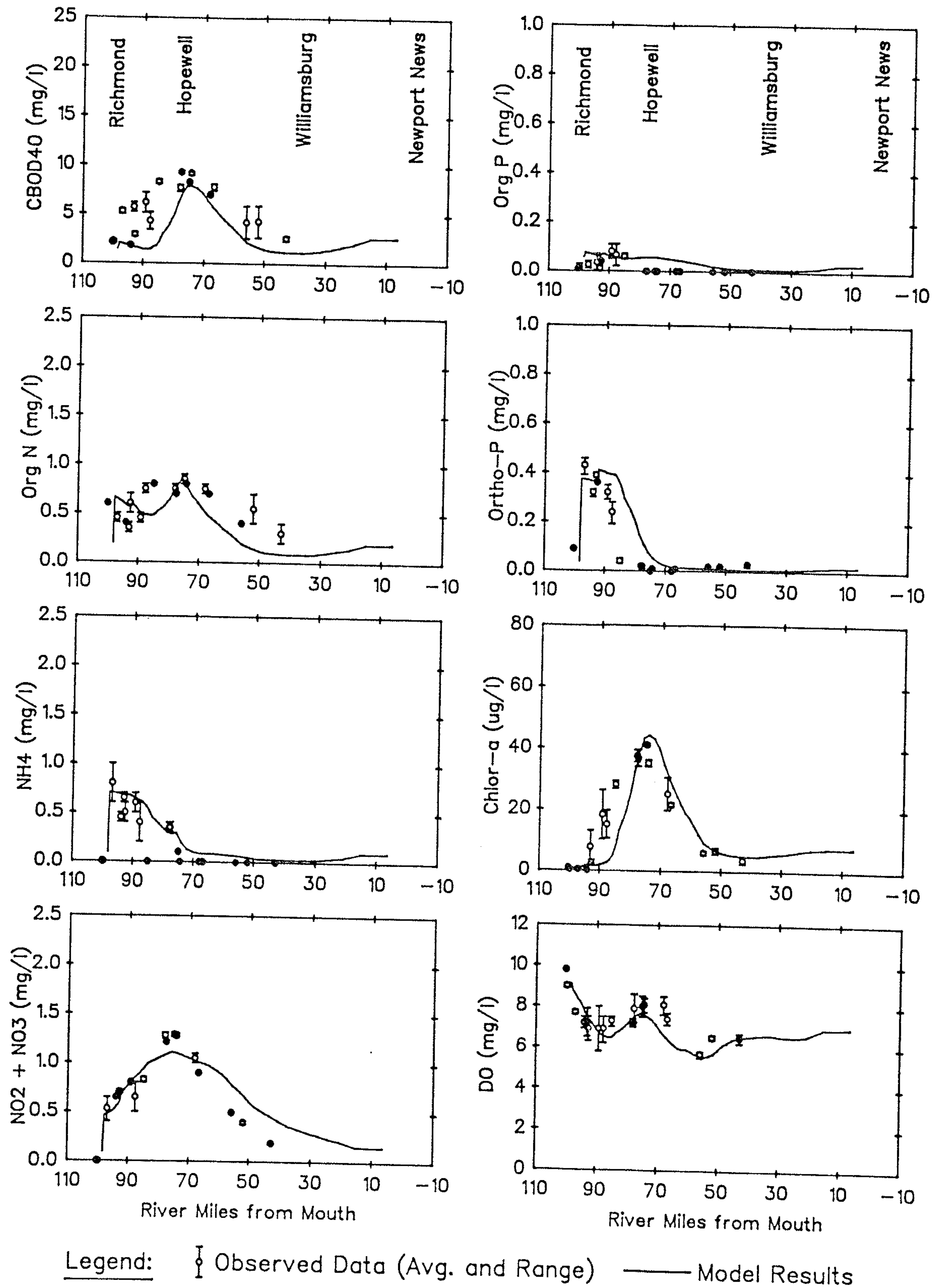
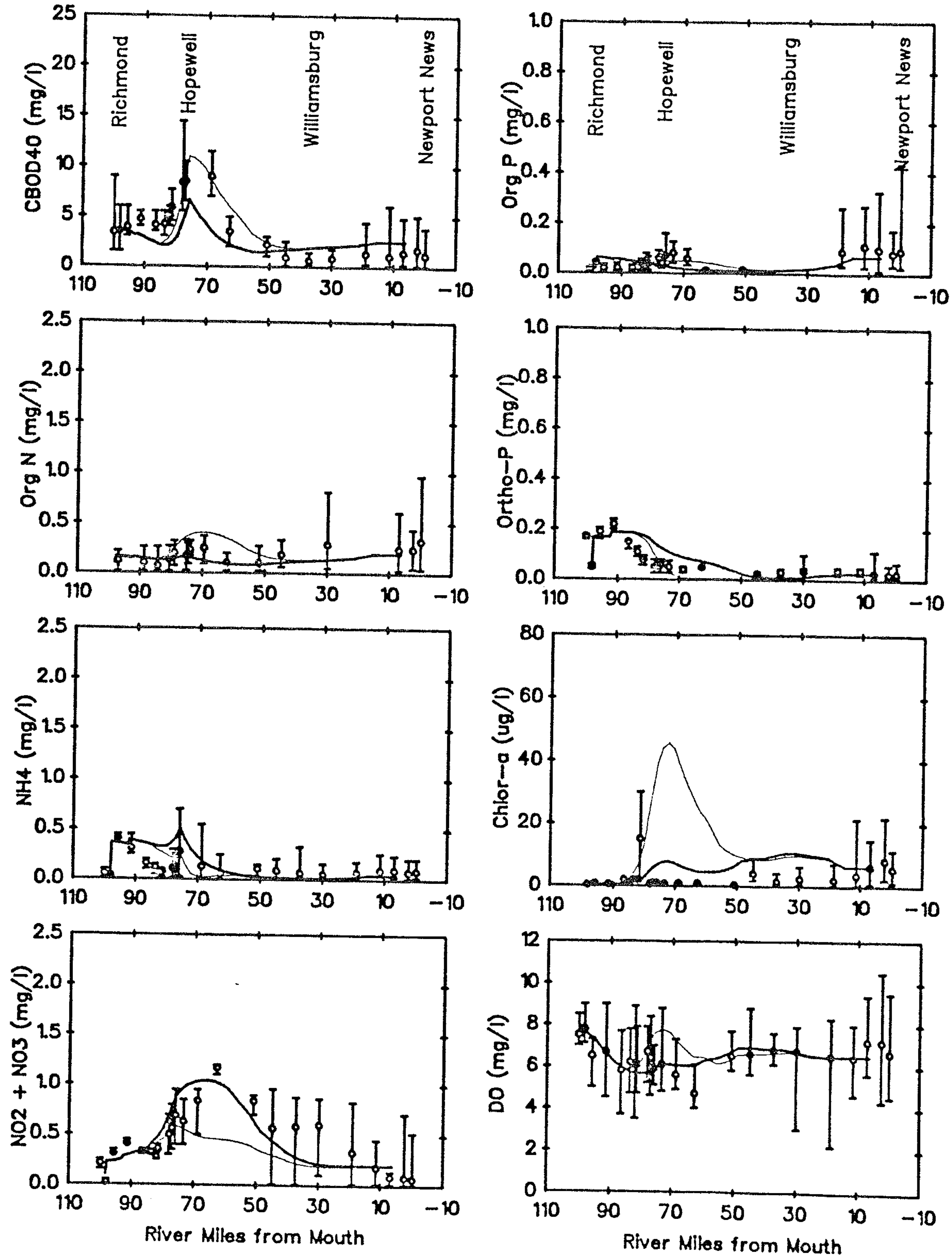


Figure 11. Model Calibration (September 20 1983 Data)



Legend: Observed Data (Avg. and Range) Model Results

Figure 12. Model Sensitivity Analysis (July 1976 Data)

of the July 1983 condition (see Figure 10). In fact, the calculated values of other water quality constituents are almost identical to those associated with the July 1983 condition.

The expanded James Estuary model has now been calibrated using 3 sets of water quality data (July 1976, July and September 1983) with satisfactory results. Table 5 presents the model coefficients from the calibration.

5. Model Projections

5.1 Contribution of Nutrient Sources in the Upper Estuary to the Lower Estuary

The main goal of this study is to quantify the fate and transport of nutrients originated from the upper estuary. The validated model was used to quantify the effect of major nutrient sources in the upper estuary on the nutrient concentrations entering the lower estuary. The September 1983 condition was chosen as the basis for such an analysis. Basically, three major nutrient sources were considered: nutrients (total nitrogen and total phosphorus) from POTWs, industrial facilities, and upstream boundary input. Other sources such as lateral input from the watershed and sediment release have been found insignificant compared with other sources.

Individual model runs (under the September 1983 condition) were conducted by removing these nutrient sources one at a time. Nutrient concentrations (organic nitrogen, ammonia nitrogen, nitrite/nitrate nitrogen, organic phosphorus, and orthophosphate) at the upstream boundary of the lower estuary (near the Chickahominy River) calculated by the model were recorded from the model results. These concentrations were then compared with the concentrations associated with the base run (September 1983 model calibration results). Figure 13 summarizes the

Table 5. James River Model Parameters
July 1976 Calibration

Kinetics Coefficients (Base e @ 20°C)

Oxygen Transfer	ft/day	3.00	
Deoxygenation	1/day	0.10	
Nitrification	1/day	0.05	(Segments 1-30)
		0.15	(Segments 31-62)
Hydrolysis - N	1/day	0.05	(Segments 1-30)
		0.10	(Segments 31-62)
- P	1/day	0.05	(Segments 1-30)
		0.10	(Segments 31-62)
Setting - N	ft/day	0.25	
- P	ft/day	0.75	
- Chl 'a'	ft/day	0.75	
Growth	1/day	2.00	
Respiration	1/day	0.10	
Death	1/day	0.10	
Extinc. Coef.	1/meter	1.4	(Segments 1-10)
		2.2	(Segments 11-21)
		2.5	(Segments 22-26)
		1.9	(Segments 27-29)
		2.3	(Segments 30-31)
		3.9	(Segments 32-34)
		4.6	(Segments 35-42)
		3.0	(Segments 43-62)
Hours of Daylight	hrs	14.5	
Benthic Demand	gm/m^2-day	0.5	(Segments 1-62)

Stoichiometry & Constants

C/CHL Ratio	mg/μg	0.035
N/CHL Ratio	mg/μg	0.007
P/CHL Ratio	mg/μg	0.001
O_2/C Ratio	mg/μg	2.67
Half. Sat.		
Conc. - N	mg/l	0.005
- P	mg/l	0.001
Sat. Light	langleys/day	300.
Avail. Light	langleys/day	450.

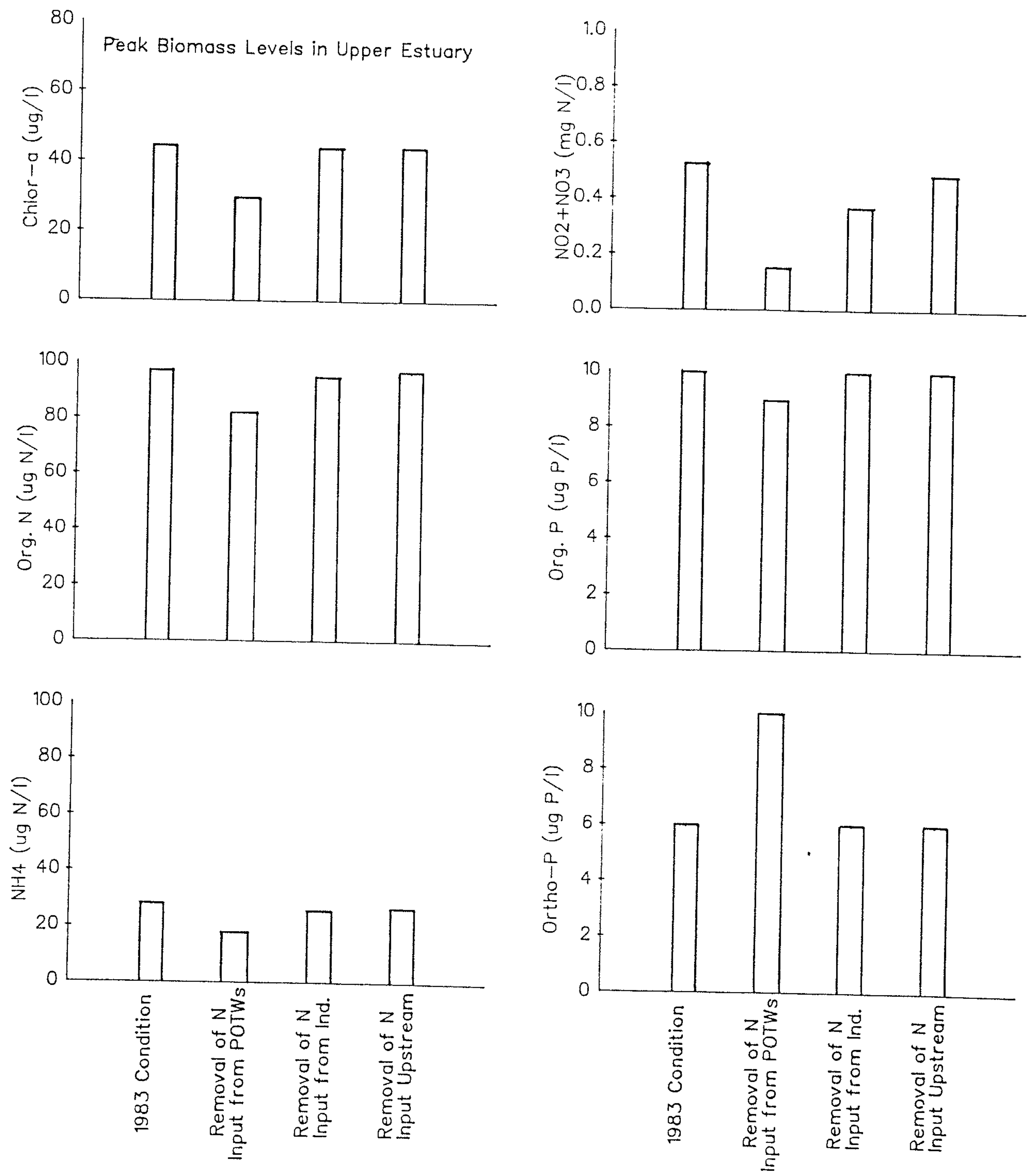


Figure 13. Contributions of Nitrogen Sources to the Water Quality in the Lower Estuary

comparisons for the nitrogen sources. Also shown in Figure 13 are peak chlorophyll a levels in the upper estuary associated with the simulation scenarios. The results indicate that eliminating nitrogen from POTWs would lower the peak chlorophyll a level in the upper estuary from about 44 μg/l to 30 μ /l. As expected, concentrations of nitrogen components entering the lower estuary would be lower than the concentrations under the so called "base" condition. While the orthophosphate concentration would increase over the level associated with the base condition, the organic phosphorus concentration would decrease due to reduced phytoplankton biomass. The results also imply that out of the 97 μg/l of organic nitrogen entering the lower estuary, about 15 μg/l is from the POTW discharges. Similarly, about 10 μg/l of ammonia and 0.375 mg/l of nitrite/nitrate are from the POTWs. The other two nitrogen sources were found to contribute a very small portion of the nutrients entering the lower estuary. That is, their impacts on the lower estuary were insignificant.

Results describing the impacts of phosphorus sources are summarized in Figure 14. Again, POTWs are the major sources of phosphorus input in terms of contributing to nutrient transport in the lower estuary. The peak chlorophyll a level in the upper estuary could reach only about 12 μg/l if the POTW phosphorus loads were eliminated. Under this scenario, the nitrite/nitrate level entering the lower estuary would increase (to 0.72 mg/l) over the level in September 1983. Comparing Figures 13 and 14 show that impacts POTW loads are more significant for phosphorus than nitrogen. Such a finding is consistent with the fact that during summer low flow months, phosphorus loads, while mostly associated with point sources, exert a greater influence on the water quality than nitrogen

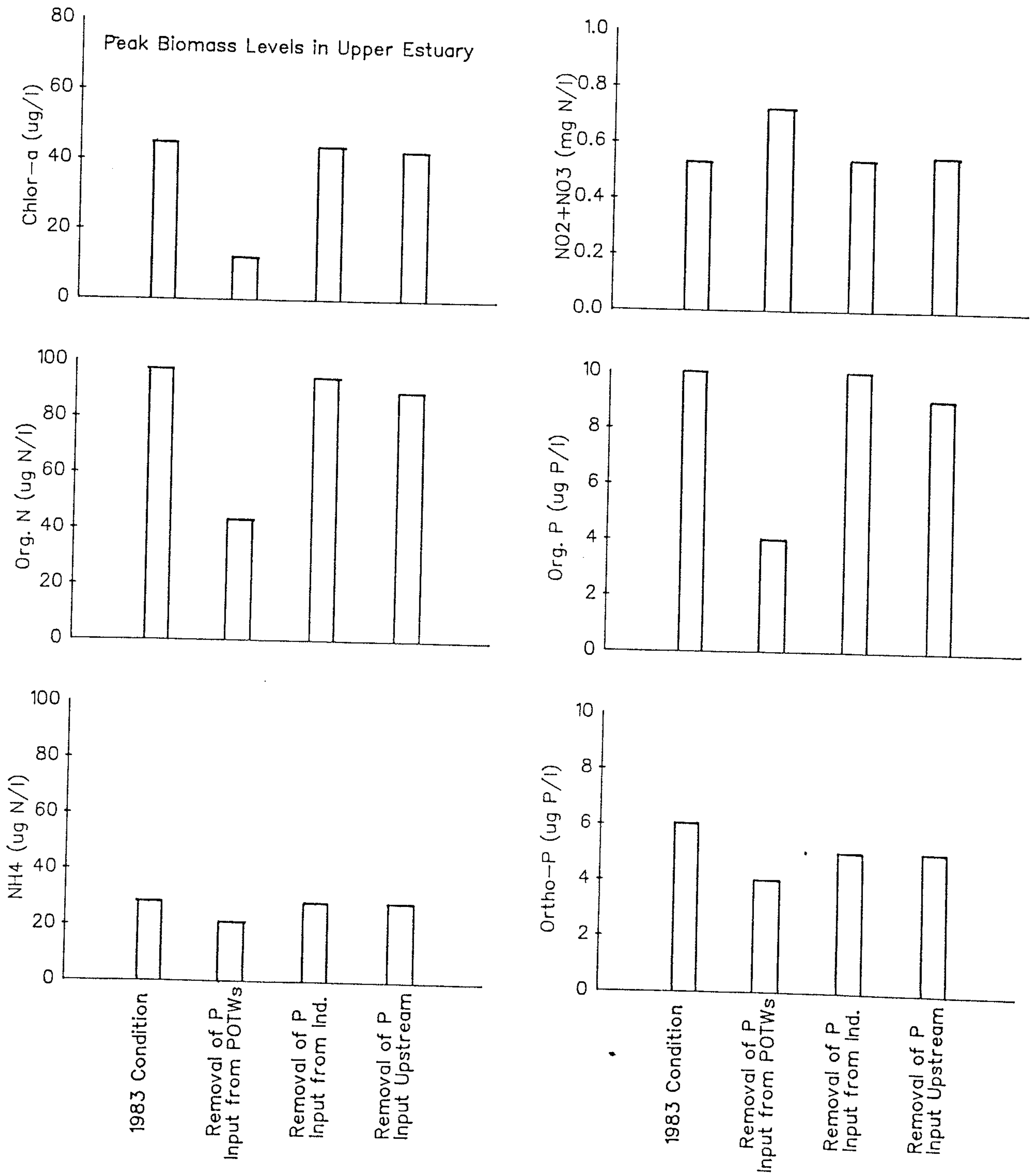


Figure 14. Contributions of Phosphorus Sources to the Water Quality in the Lower Estuary

loads which come primarily from nonpoint sources. [The summer months of 1983 were observed with very low river flows compared with historic records.]

Results from the model calibration run and the run with POTW nitrogen loads eliminated are shown in Figure 15. [Results from other runs with industrial and sediment release loads eliminated are not plotted because they were found not as important as the POTW loads.] In the upper estuary, nitrogen concentrations would be much lower than those associated with the calibration run if POTW loads were removed. Phytoplankton biomass would be slightly lower because of the low biomass levels. Reduced ammonia concentrations in the upper estuary would also mean higher dissolved oxygen due to retarded nitrification. Similar results from the analysis of removing POTW phosphorus loads (see Figure 16) indicate much lower algal biomass in the upper estuary associated with reduced CBOD organic nitrogen and organic phosphorus levels. Orthophosphate concentrations would also become much lower in the upper estuary. On the other hand, inorganic nitrogen components would increase in the upper estuary and, as shown earlier, such an increase would transport additional inorganic nitrogen into the lower estuary (Lung, 1986b). Dissolved oxygen concentrations would be lowered because of increased nitrification and reduced algal photosynthesis in the system.

5.2 Effects of Nutrient Control on the Water Quality of the Lower Estuary

The calibrated model was then used to address another question: What is the effect of nutrient control in the upper estuary on the water quality of the lower estuary? This question was raised in an earlier

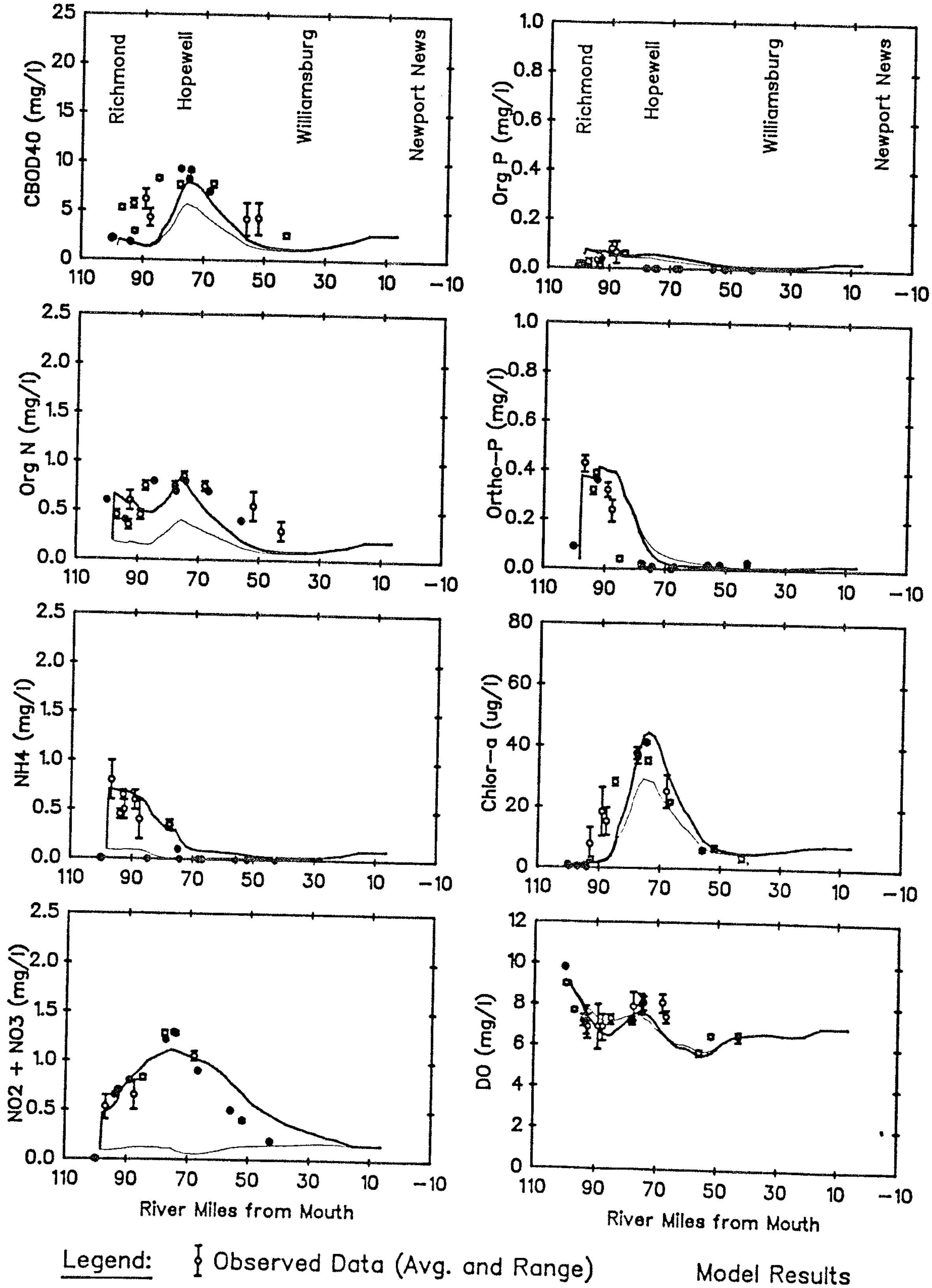
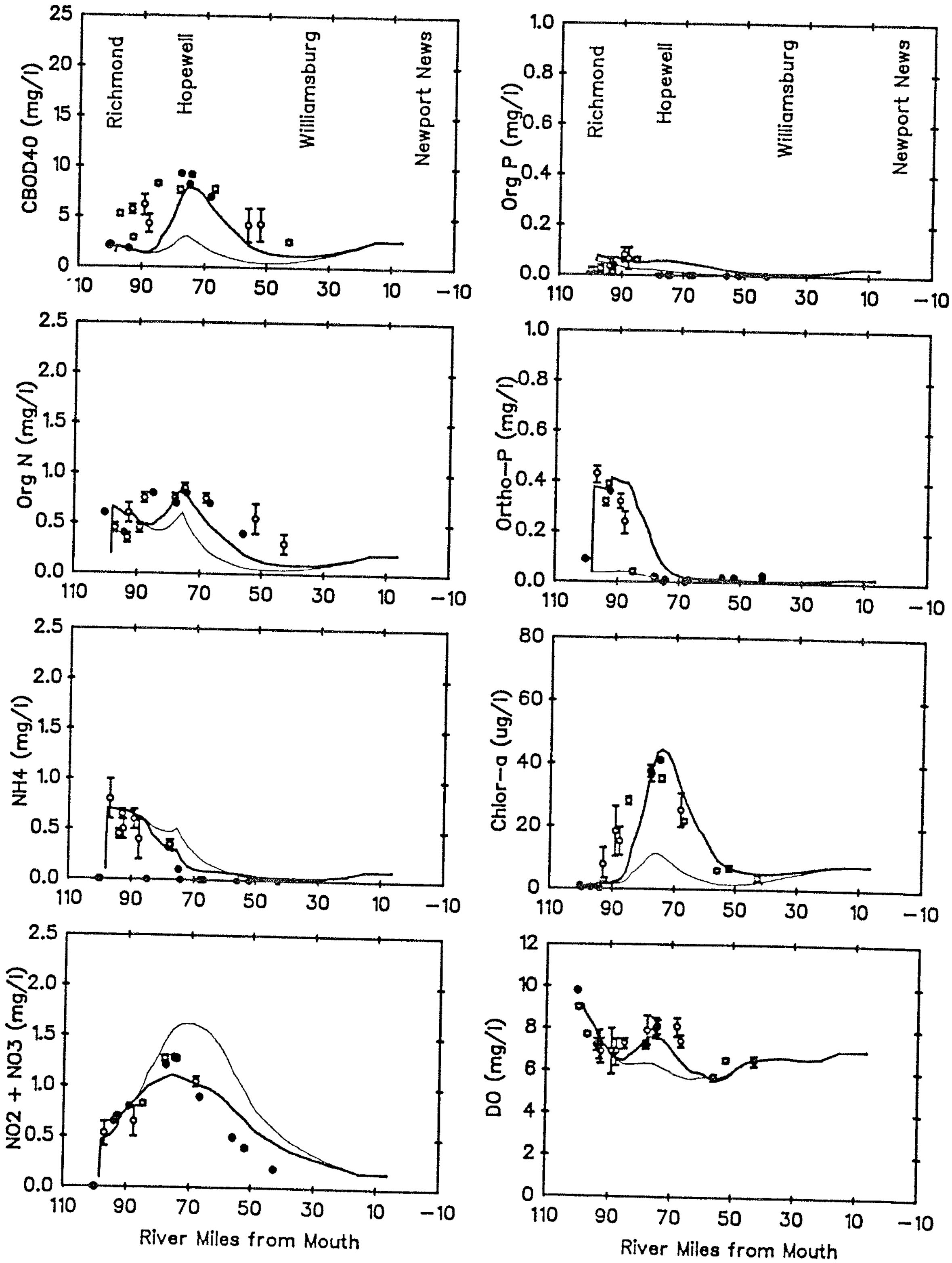


Figure 15. Effects of POTW Nitrogen Loads on the Water Quality



Legend: Observed Data (Avg. and Range) Model Results

Figure 16. Effects of POTW Phosphorus Loads on the Water Quality

study (Lung, 1986b) in light of the model results that inorganic nitrogen levels would increase in the water column under various phosphorus control alternatives. Would the increase in inorganic nitrogen increase the potential of algal growth in the lower estuary? To address this question, a number of phosphorus control scenarios developed in the earlier study (Lung, 1986b) were evaluated in this study:

- phosphate detergent ban (25% reduction in total phosphorus loads at POTWs)
- phosphate detergent ban (35% reduction in total phosphorus loads at POTWs)
- phosphorus removal at POTWs (effluent total phosphorus limit at 2 mg/l)
- phosphorus removal at POTWs (effluent total phosphorus limit at 1 mg/l)
- phosphorus removal at POTWs (effluent total phosphorus limit at 0.5 mg/l)
- phosphorus removal at POTWs (effluent total phosphorus limit at 0.2 mg/l)

The model projection runs were conducted at the 7-day 10-year low flow condition. The 7-day 10-year low flow in Richmond is 680 cfs (Engineering Science Co., 1974). A water temperature of 28°C was assumed in the analysis. All other model parameters and coefficients were kept the same as those used in the model calibration analysis.

The results of model projections are summarized and presented in Figure 17. Only the orthophosphate, chlorophyll a, and inorganic nitrogen concentrations are shown for each simulation scenario. It should be pointed out that the system responses in the upper estuary are identical to those presented in the earlier study (Lung, 1986b). That is, nitrogen concentrations would increase in the upper estuary due to reduced

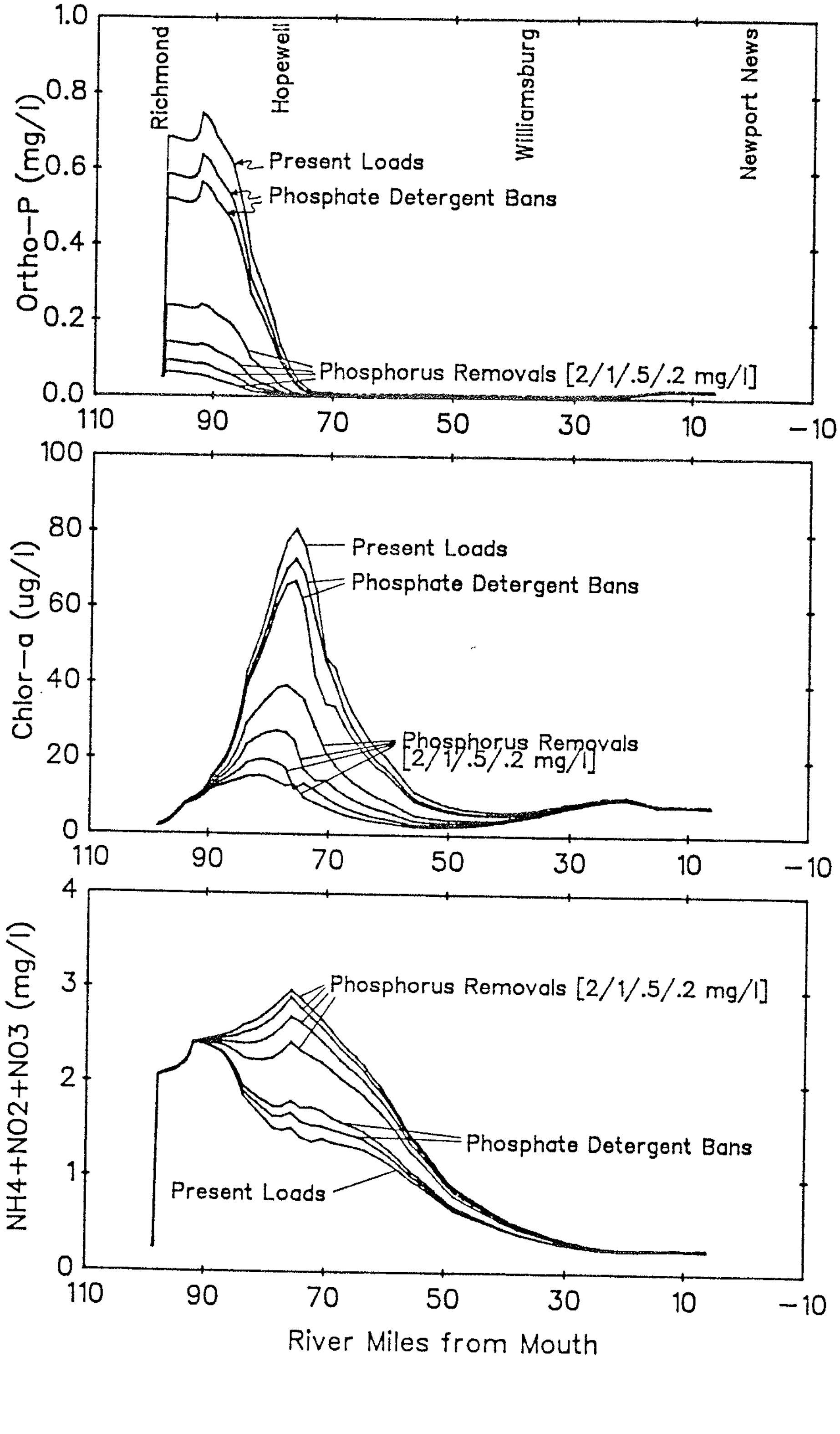


Figure 17. Projected Nutrient and Chlorophyll a Concentrations

algal uptake associated with phosphorus control, particularly the phosphorus removal alternatives. However, such an increase in inorganic nitrogen would have practically no impact on the algal biomass levels when compared with the base levels in the lower estuary. Further examination of the model results indicated that while nitrogen concentrations would increase, reduced phosphorus (orthophosphate) levels would result in serious phosphorus limitation in the lower estuary. In addition, high turbidity levels in the lower estuary (Neilson and Ferry, 1978) would significantly suppress the algal growth rate. As a result, while phosphorus controls in the upper estuary would provide additional nitrogen input to the lower estuary, they would not affect the algal growth potential in the lower estuary.

6. Fate and Transport of Phosphorus in the James River Estuary

One of the key questions often asked is to where nutrients from wastewater discharges would be transported. For example, phosphorus from wastewaters in the upper estuary could be incorporated into the biomass of phytoplankton in the water column, deposited into the sediments, or transported to the lower estuary. Perhaps a more meaningful question is: how much phosphorus in the algal biomass at a certain location in the upper estuary is from a particular source? In BOD/DO modeling analyses, a similar question is: how much dissolved oxygen deficit at a given location is from the point sources, the sediments, or other sources? Usually, a component analysis is performed to quantify the contribution of individual sources to the dissolved oxygen deficit. A similar component analysis is not appropriate for eutrophication modeling analysis simply because of the nonlinear relationship of the phytoplankton growth-nutrient dynamics in the model. That is, results from a component analysis would not be adequate to quantify the percent composition in algal biomass in terms of various sources of phosphorus.

In limnological studies, $^{32}PO_4$ is added as a tracer to determine the fate of phosphorus in the system by measuring the amount of ^{32}P in various components of the system. Such a concept of using ^{32}P as a tracer can be adopted to our study in a mathematical fashion. That is, a source or sources of phosphorus can be numerically labeled and added to the James River Estuary. The James River Estuary can then be used to quantify the amount of such labeled phosphorus in different components of the water column: organic phosphorus, orthophosphate, and algal biomass. Basically, a ^{32}P tracer analysis will be conducted using the model.

Thus, the James River Estuary model was modified to perform a so called "numerical tagging" analysis. First, 3 components were added as system variables to the model: labeled organic phosphorus, labeled orthophosphate, and labeled phosphorus in the algal biomass. That is, parallel calculations of labeled and unlabeled phosphorus were incorporated into the model. Special care was needed to treat the nonlinear relationship between algal growth rate and phosphorus concentrations. Kinetic interrelationships between these labeled system variables are the same as those unlabeled. In general, algal growth rates were calculated based on the total concentration of labeled and unlabeled orthophosphate. However, when either labeled or unlabeled orthophosphate is exhausted, algal growth and associated phosphorus would be shifted to the other component to avoid generating negative orthophosphate concentrations by the model.

The modified model (with 3 additional system variables) has been thoroughly tested for conservation of mass as well as numerical accuracy. For example, when a single source of phosphorus is labeled, the modified model would still generate the same total organic phosphorus, orthophosphate, and algal biomass levels as the original model calculated. Next, 4 categories of phosphorus input to the James River Estuary were labeled one at a time, the total organic phosphorus, orthophosphate, and algal biomass were added up and found equal to the total concentrations of these system variables calculated by the original model.

Results from the above described analysis are summarized and presented in Figure 18 using the calibration data set of September 1983. It is seen that POTWs in the upper estuary contributed about 75% of the total algal biomass (as chlorophyll a) in the water column. Upstream (nonpoint) and downstream boundary conditions provided another 15% of the algal biomass. Industrial wastewaters played a very small role in contributing the algal biomass in the James River Estuary. The Appomattox River which receives wastewater discharges from the City of Petersburg contributed an insignificant amount of phosphorus to the algal biomass in the mainstream of the estuary.

It should be stressed that while results from the numerical tagging analysis did not affect the conclusion of this modeling study (i.e., POTWs are the most significant phosphorus source contributing to the eutrophication of the upper James River Estuary), the significance of POTW phosphorus sources can now be accurately quantified. Particularly, the relative significance of various phosphorus sources for the upper James River Estuary can be determined as to the contribution to the algal biomass in the system. Therefore, the overwhelming importance of the POTW phosphorus input in the upper James River Estuary is once again confirmed.

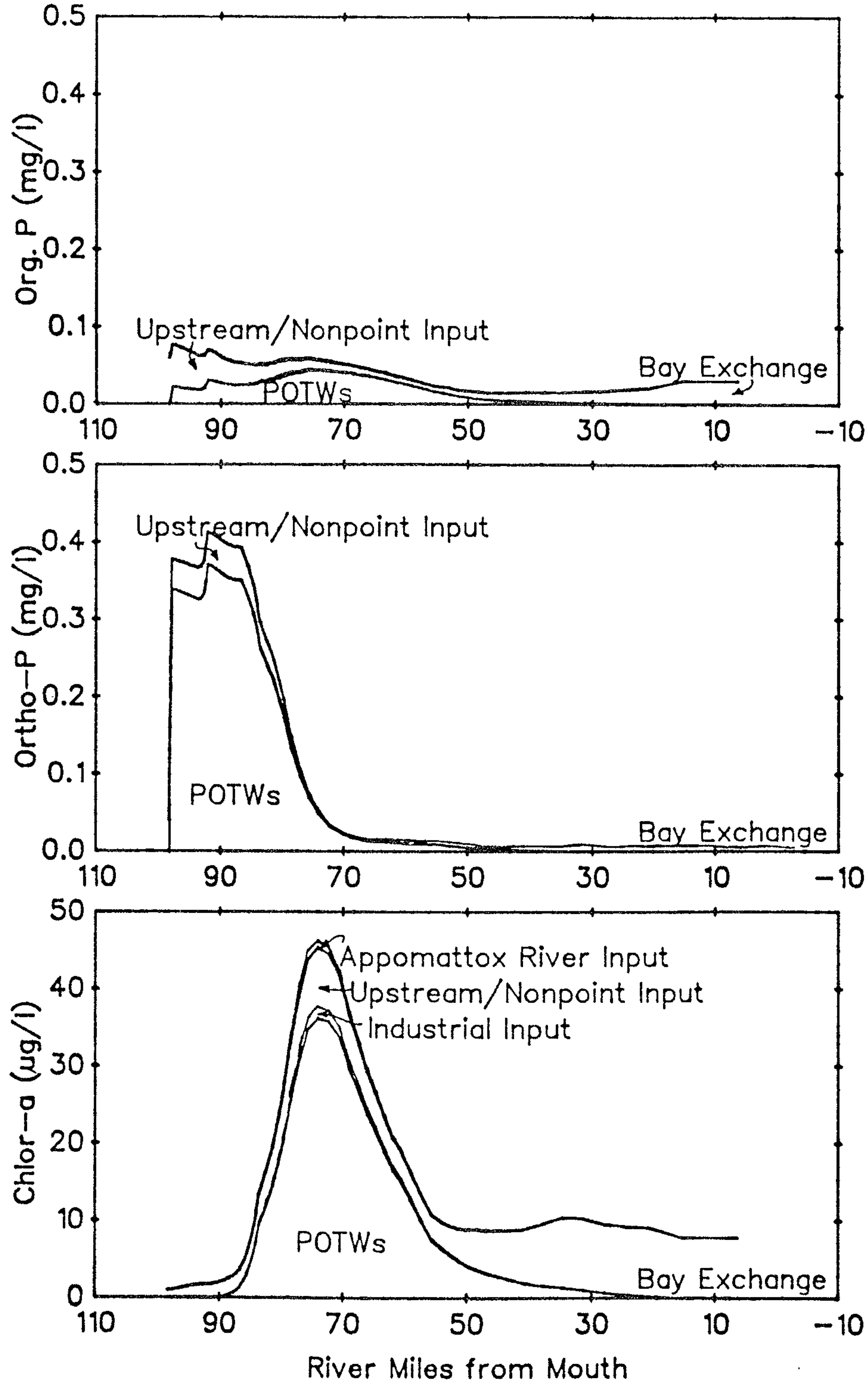


Figure 18. Contribution of Phosphorus Sources (September 1983 Data)

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