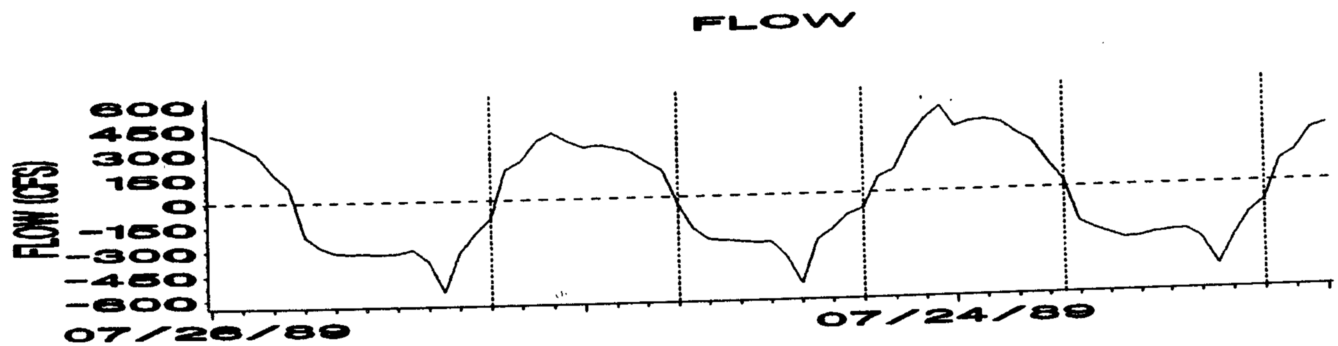
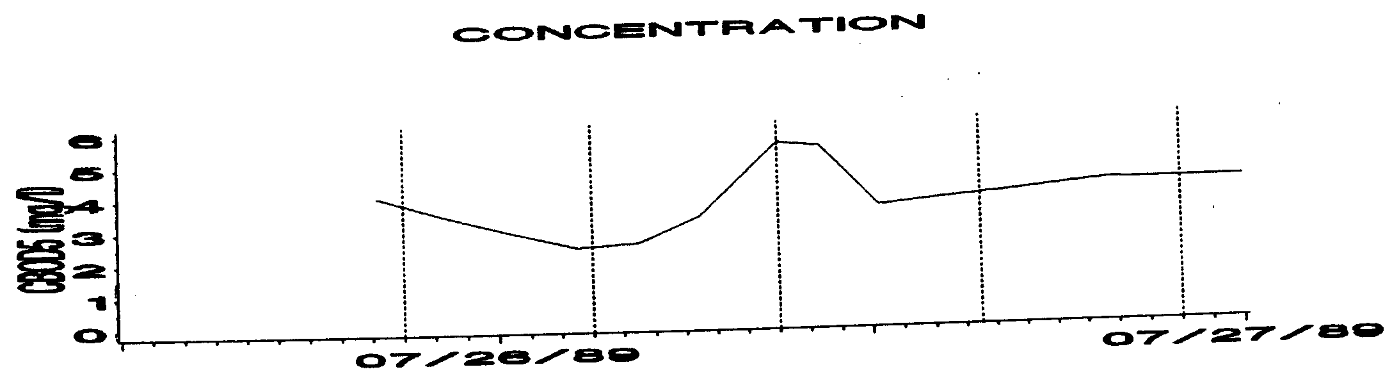
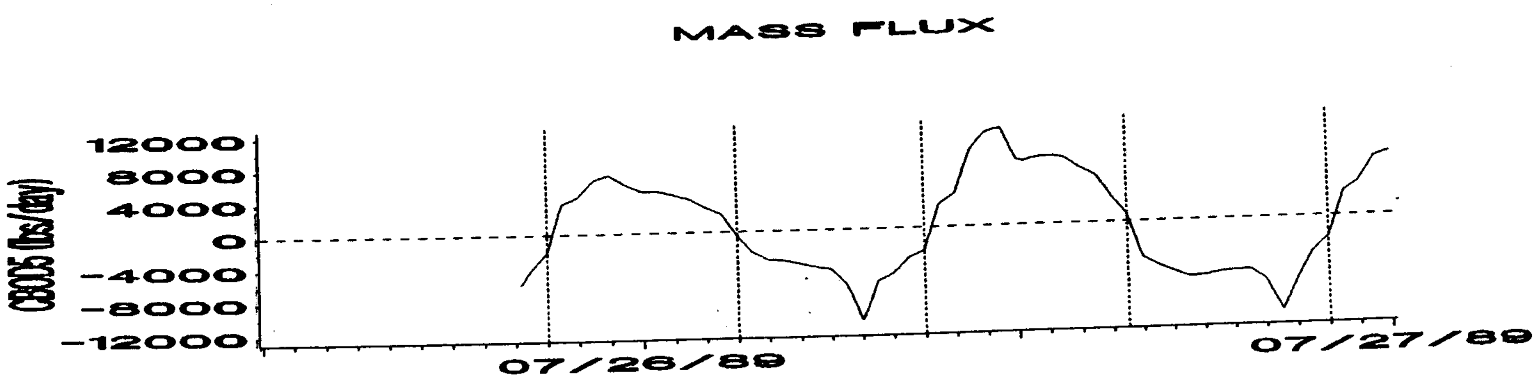
Appendix A-2-2
Section 7: Figures for Station S15

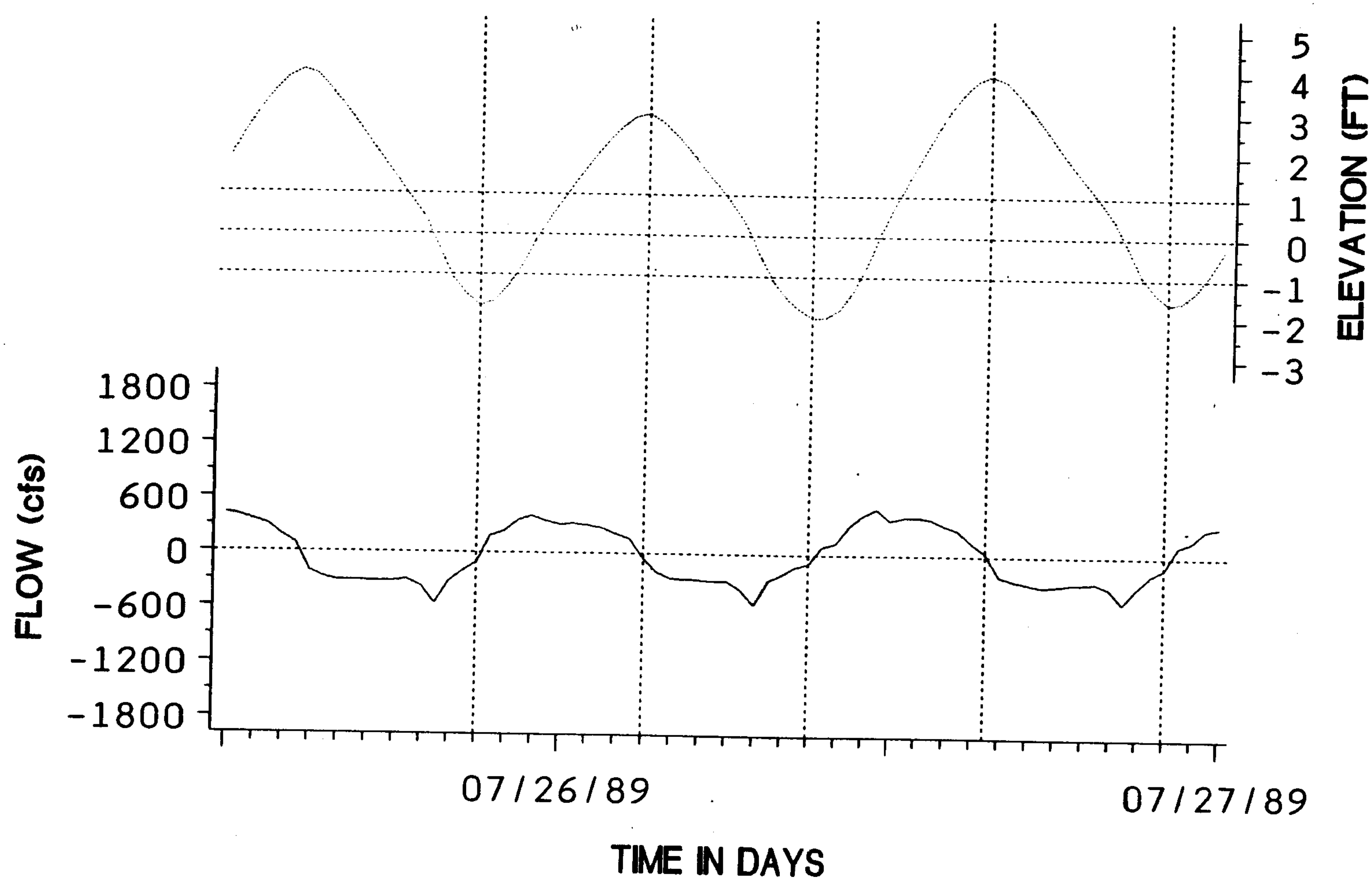
CBOD5 CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



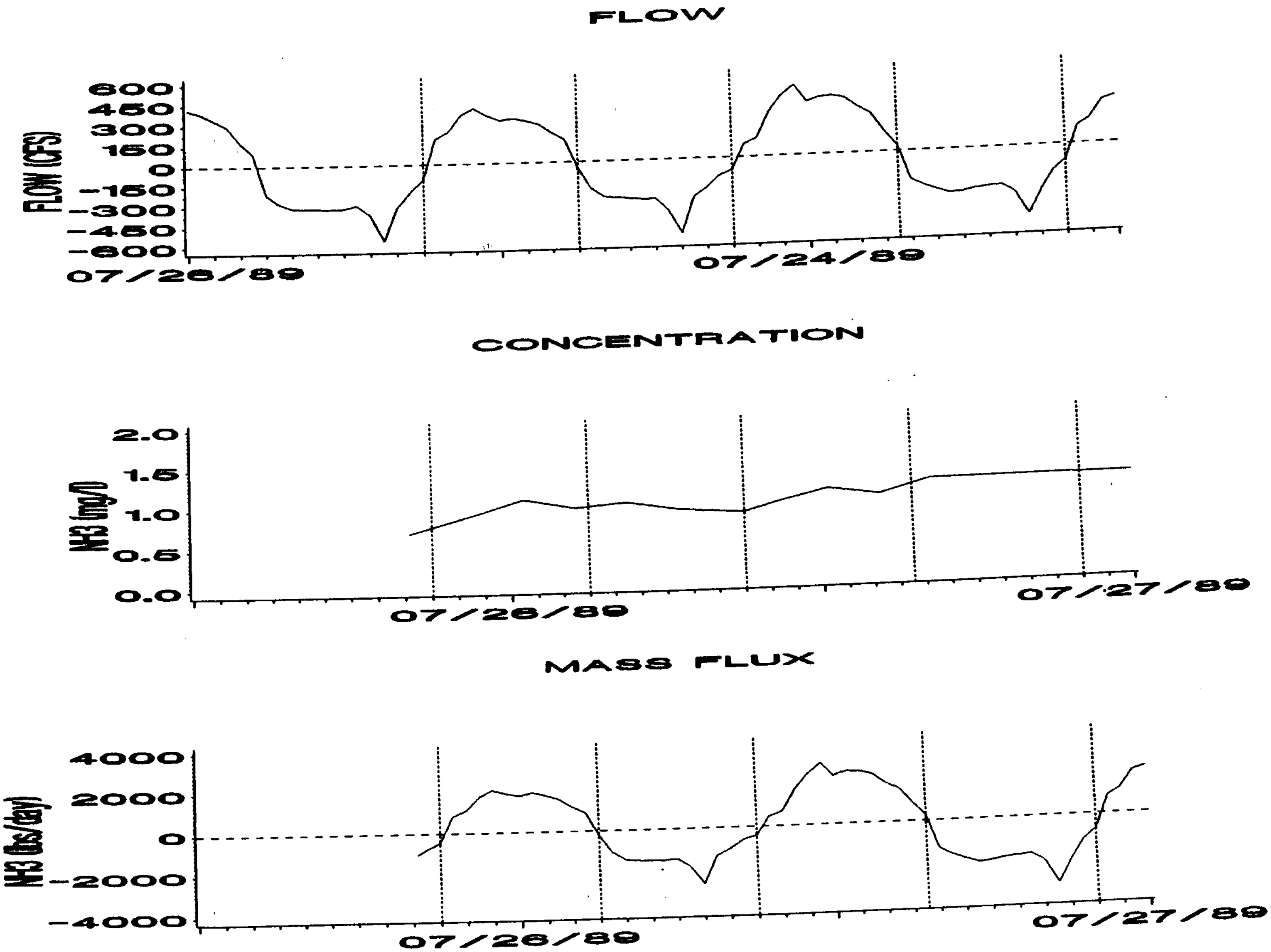




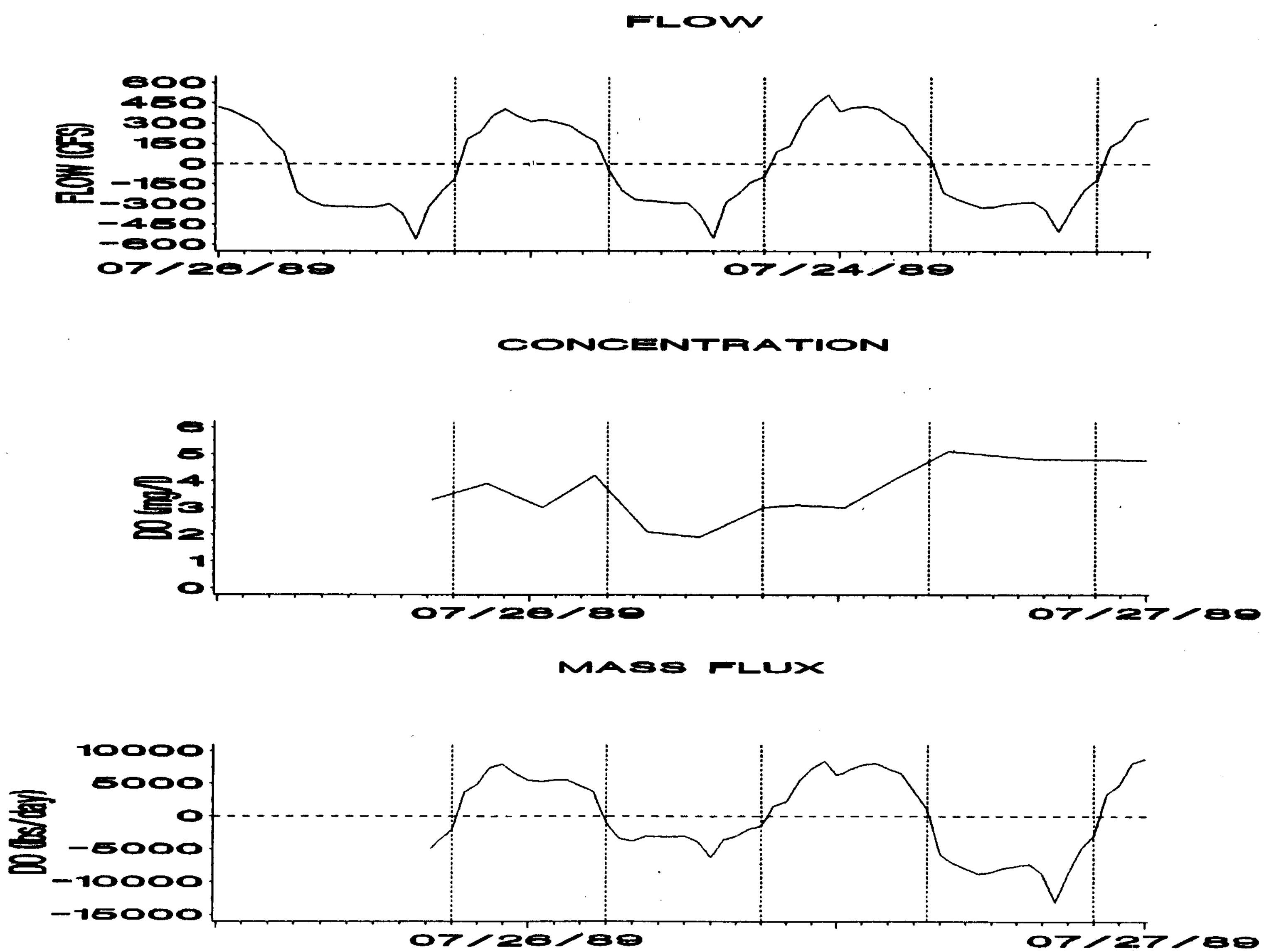
BERRYS CREEK UPSTREAM (S15) - JULY 1989 - FLOW VERSUS TIDE
FLOW CALCULATED FROM AREA*VEL,VELOCITY MEASURED BY OSI



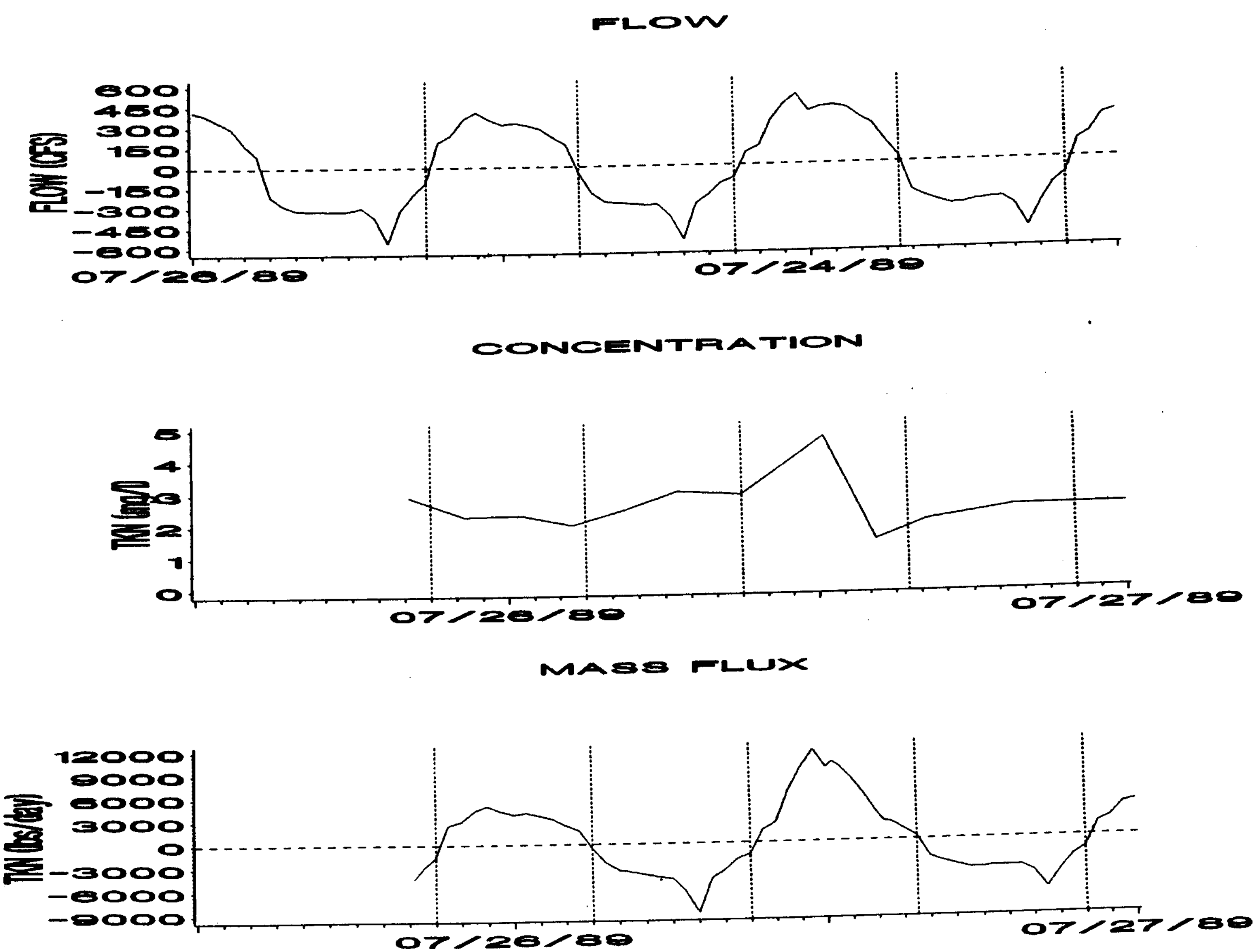
NH3 CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



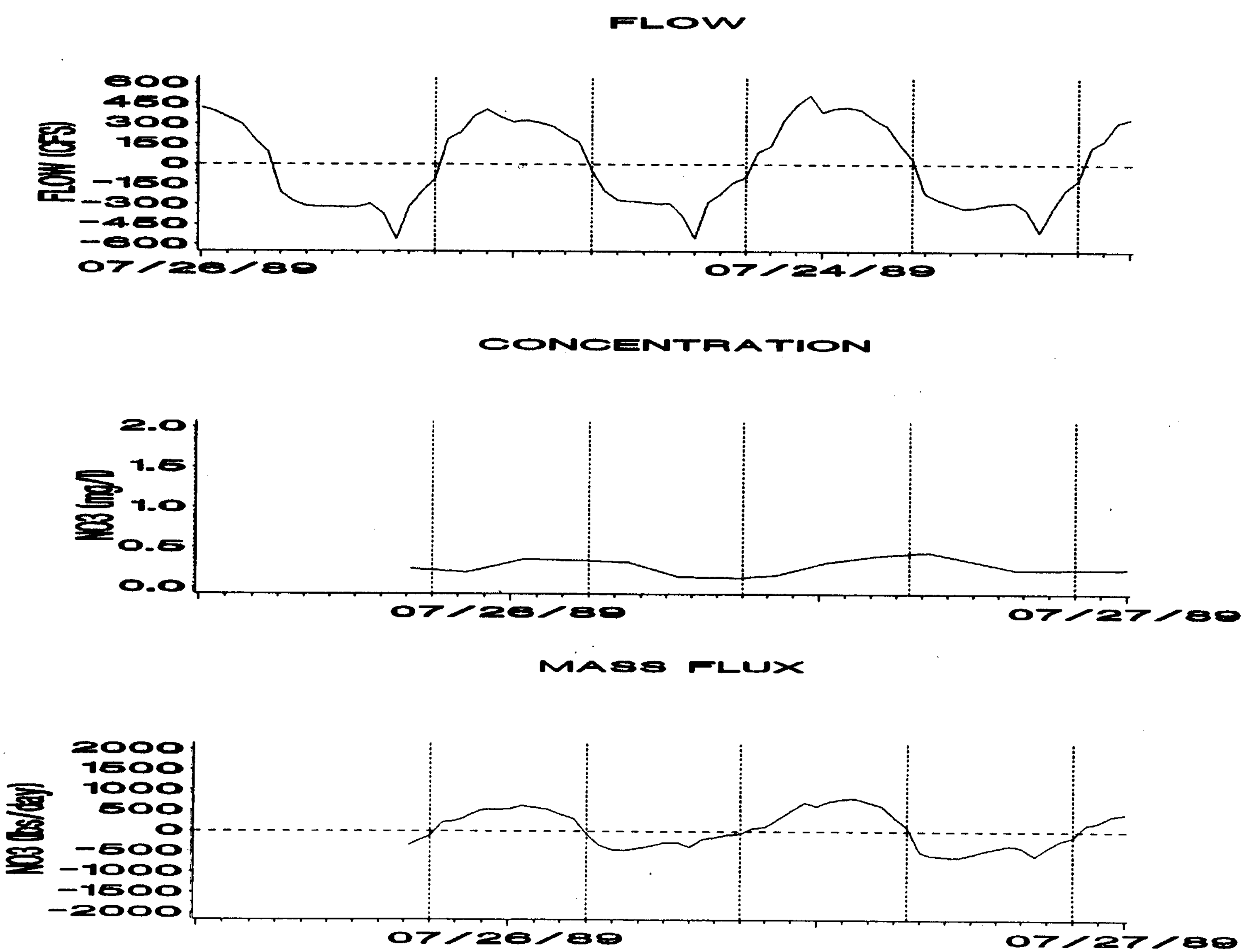
DO CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



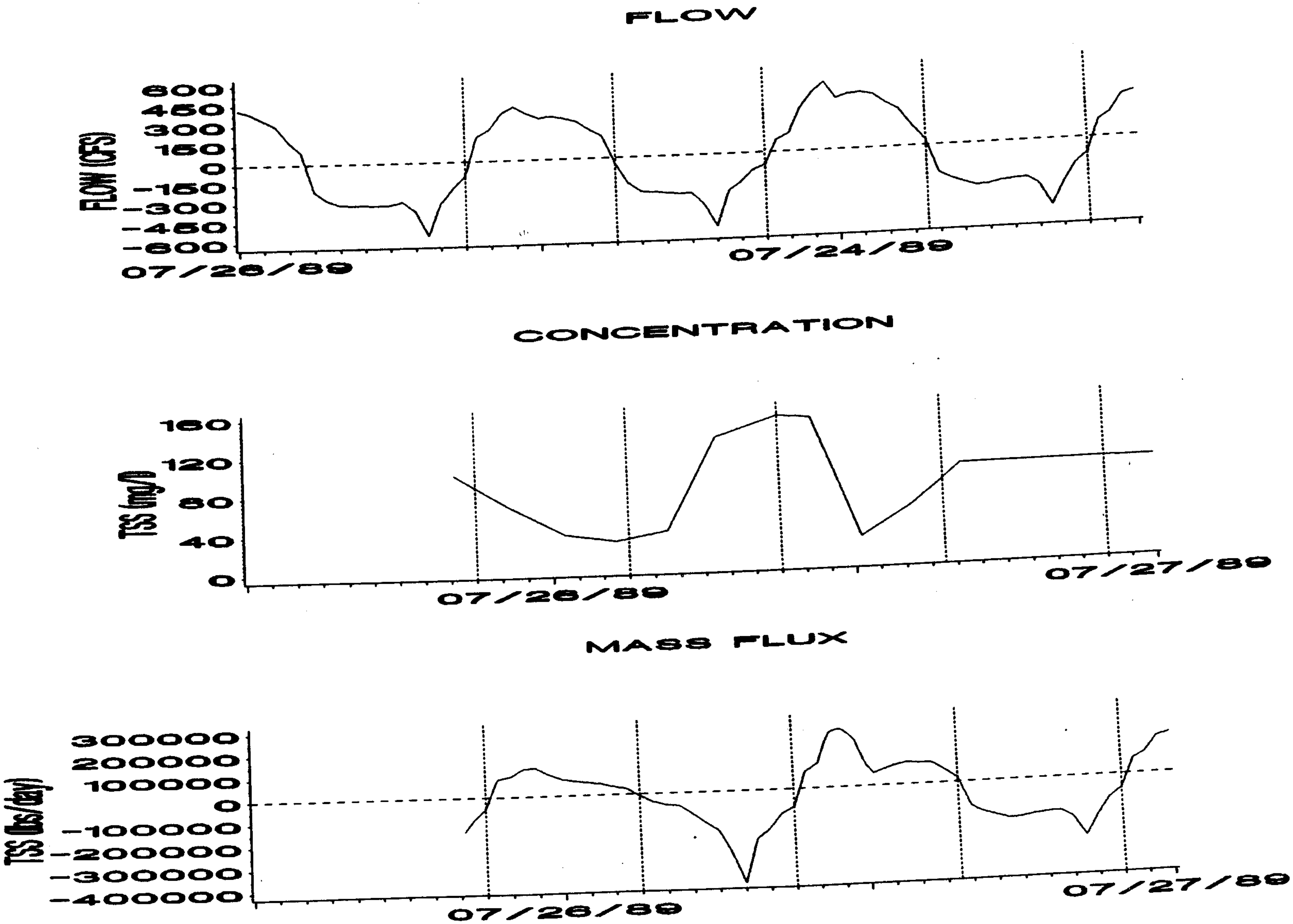
TKN CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



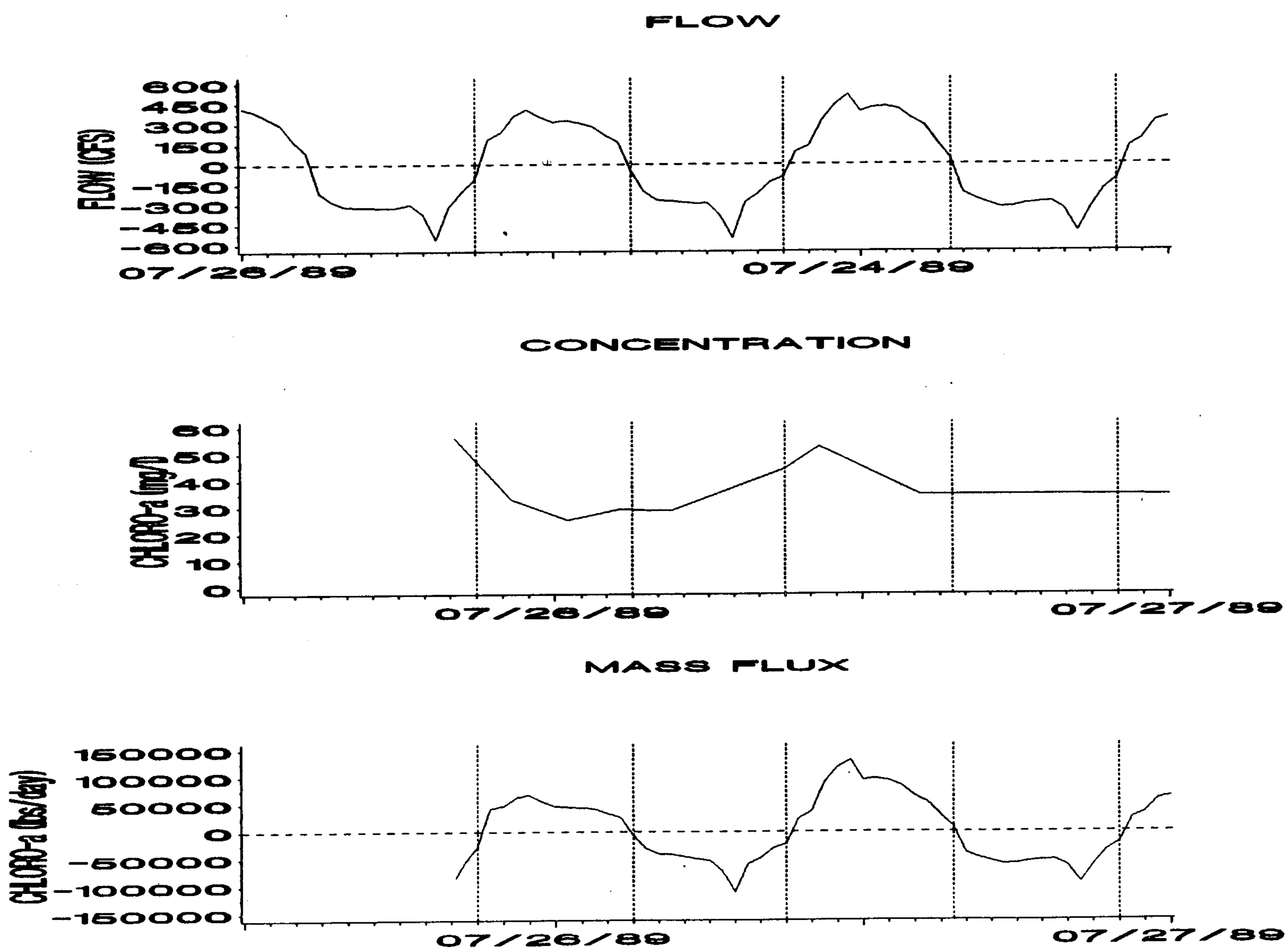
NITRATE CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



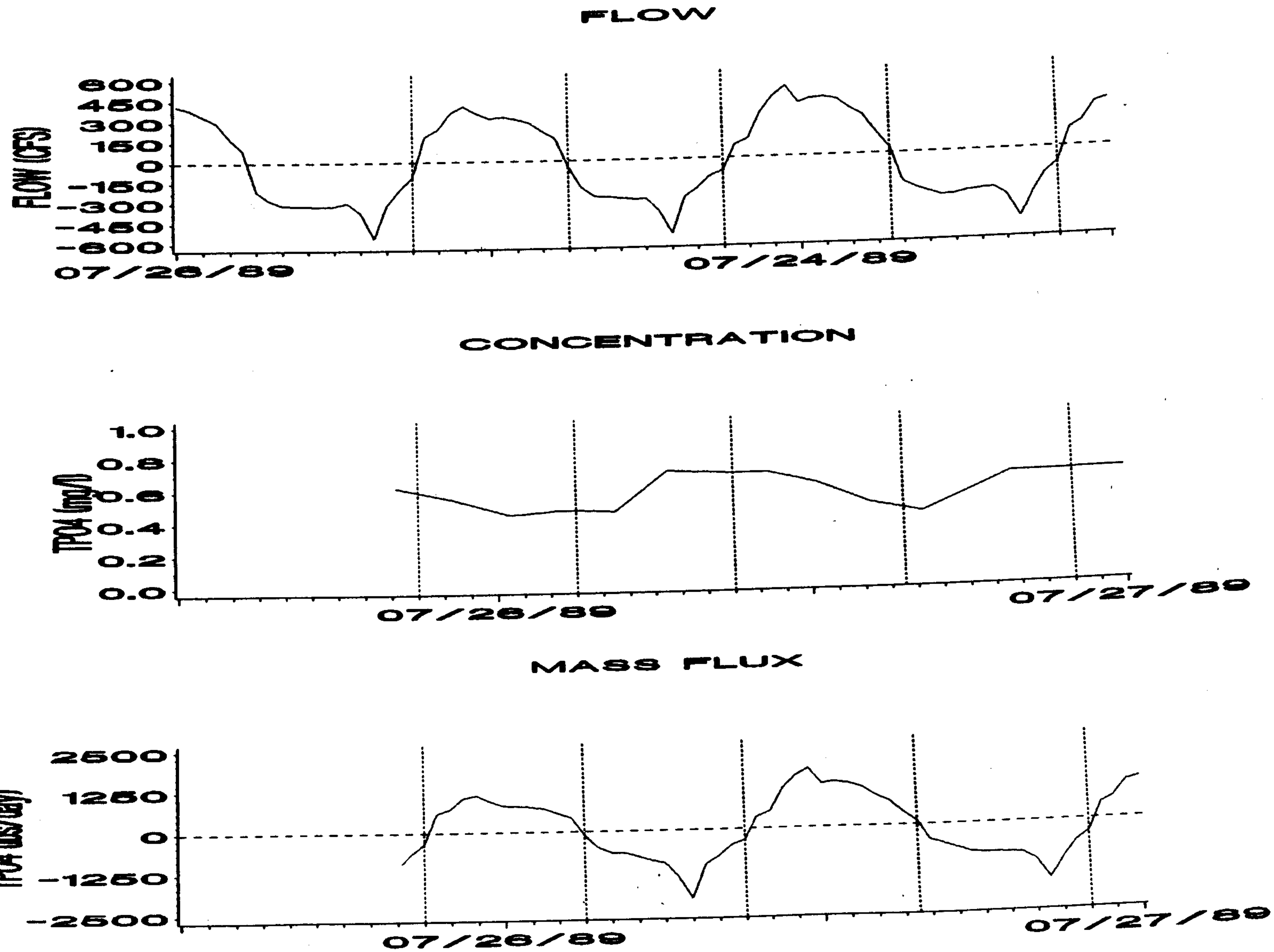
TSS CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



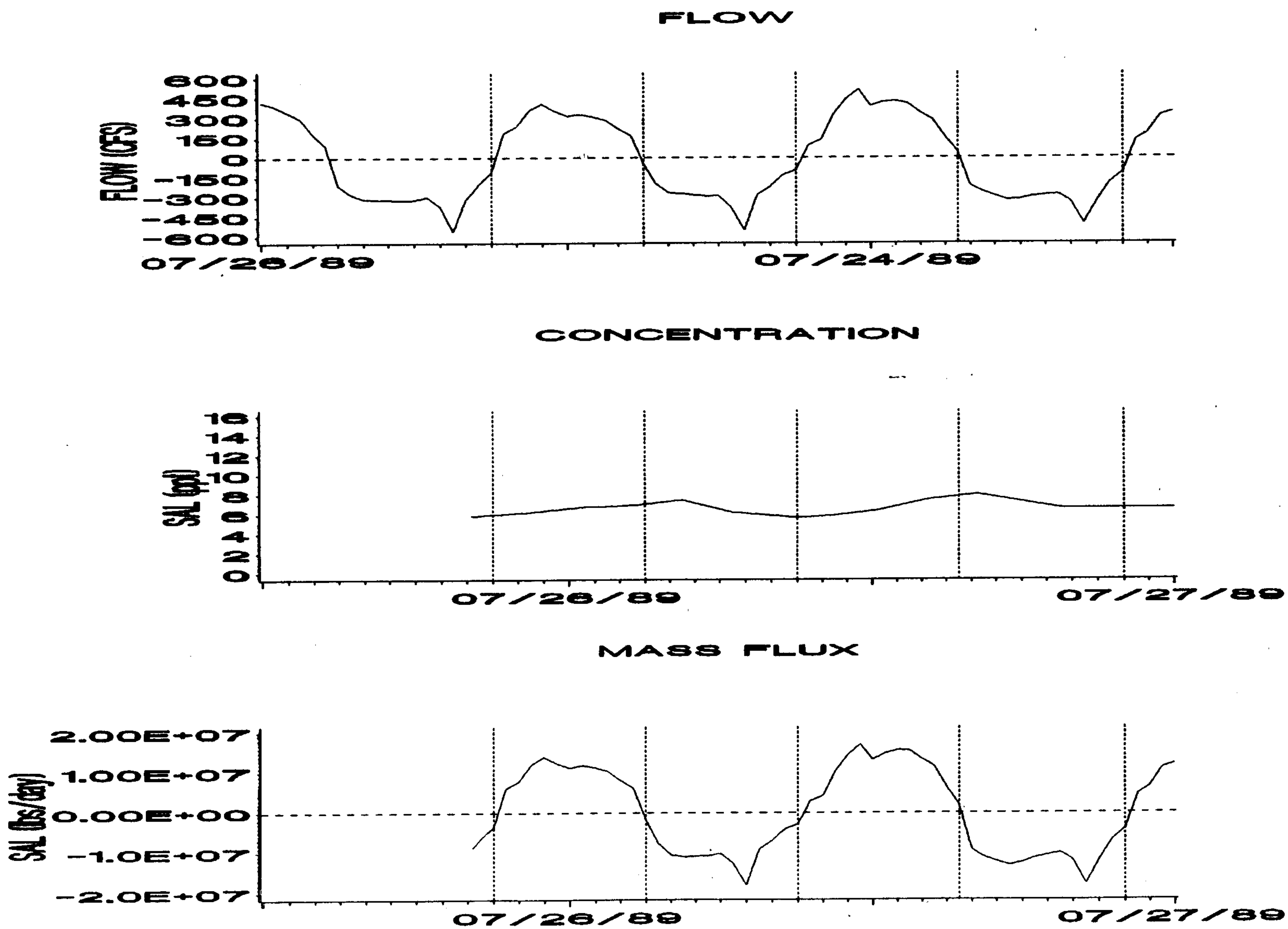
CHOLRO-a CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



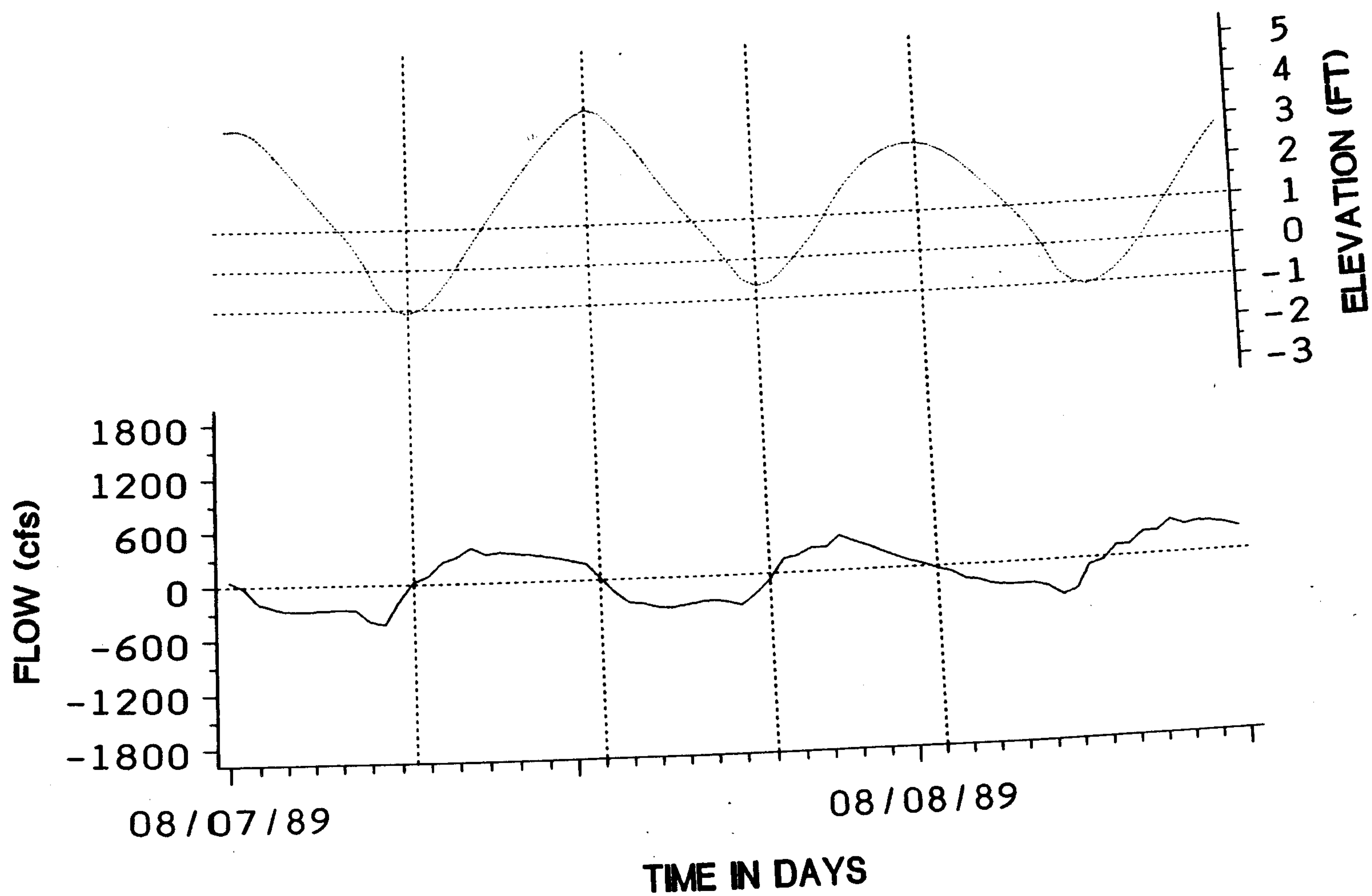
TOTAL PHOSPHATE CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



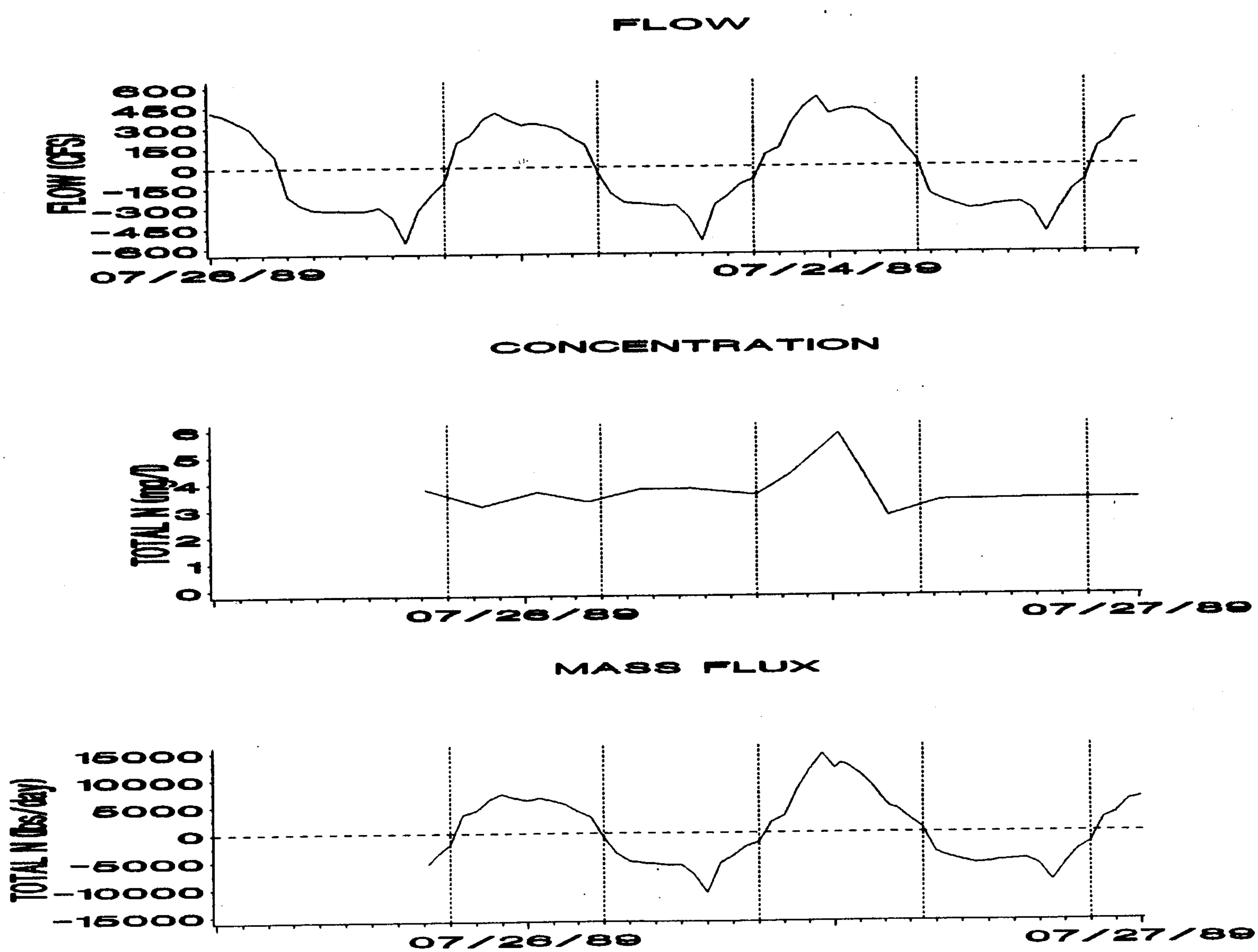
SALINITY CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



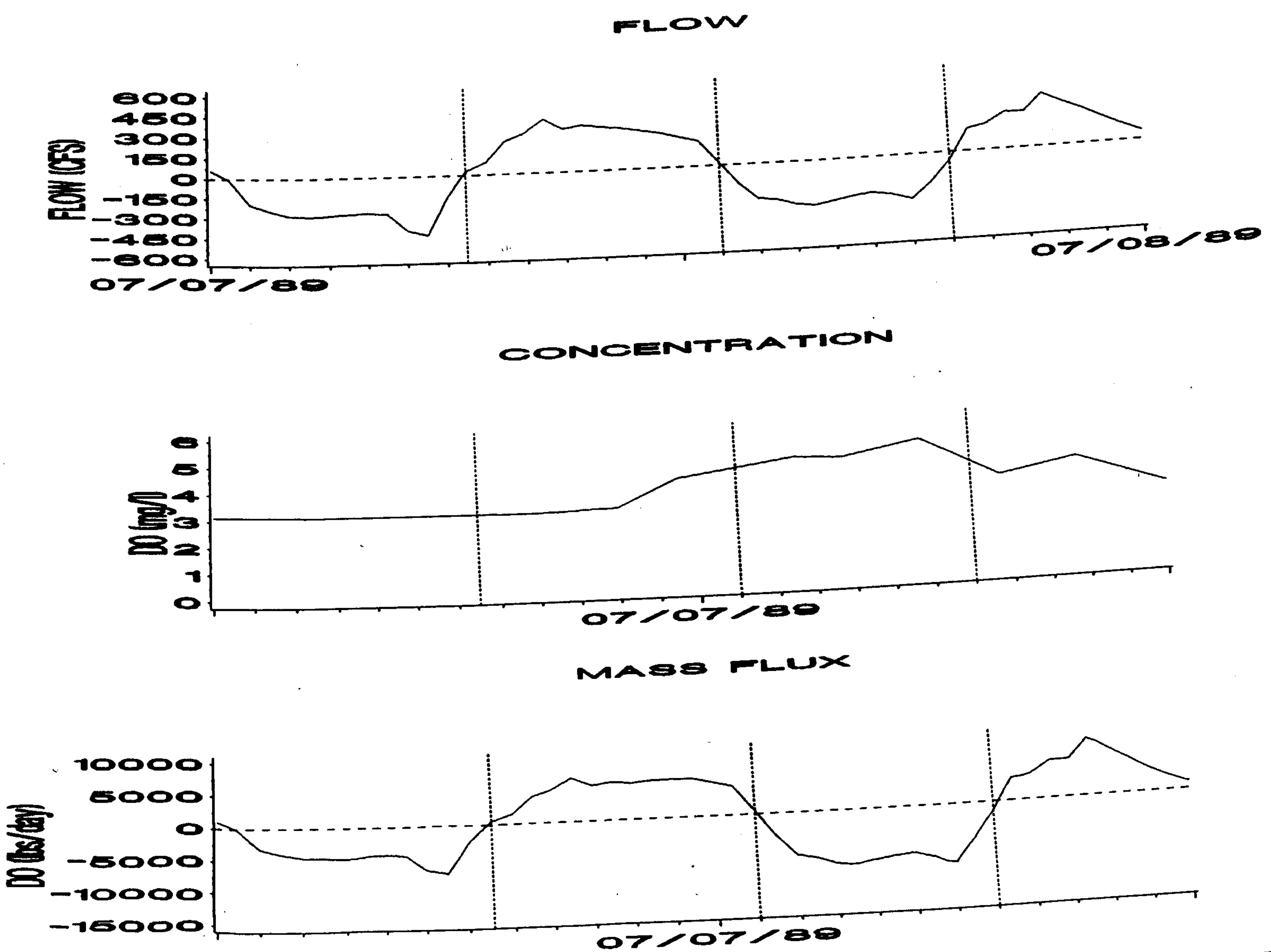
BERRYS CREEK UPSTREAM (S15) - AUGUST 1989 - FLOW VERSUS TIDE
FLOW CALCULATED FROM AREA*VEL, VELOCITY MEASURED BY OSI



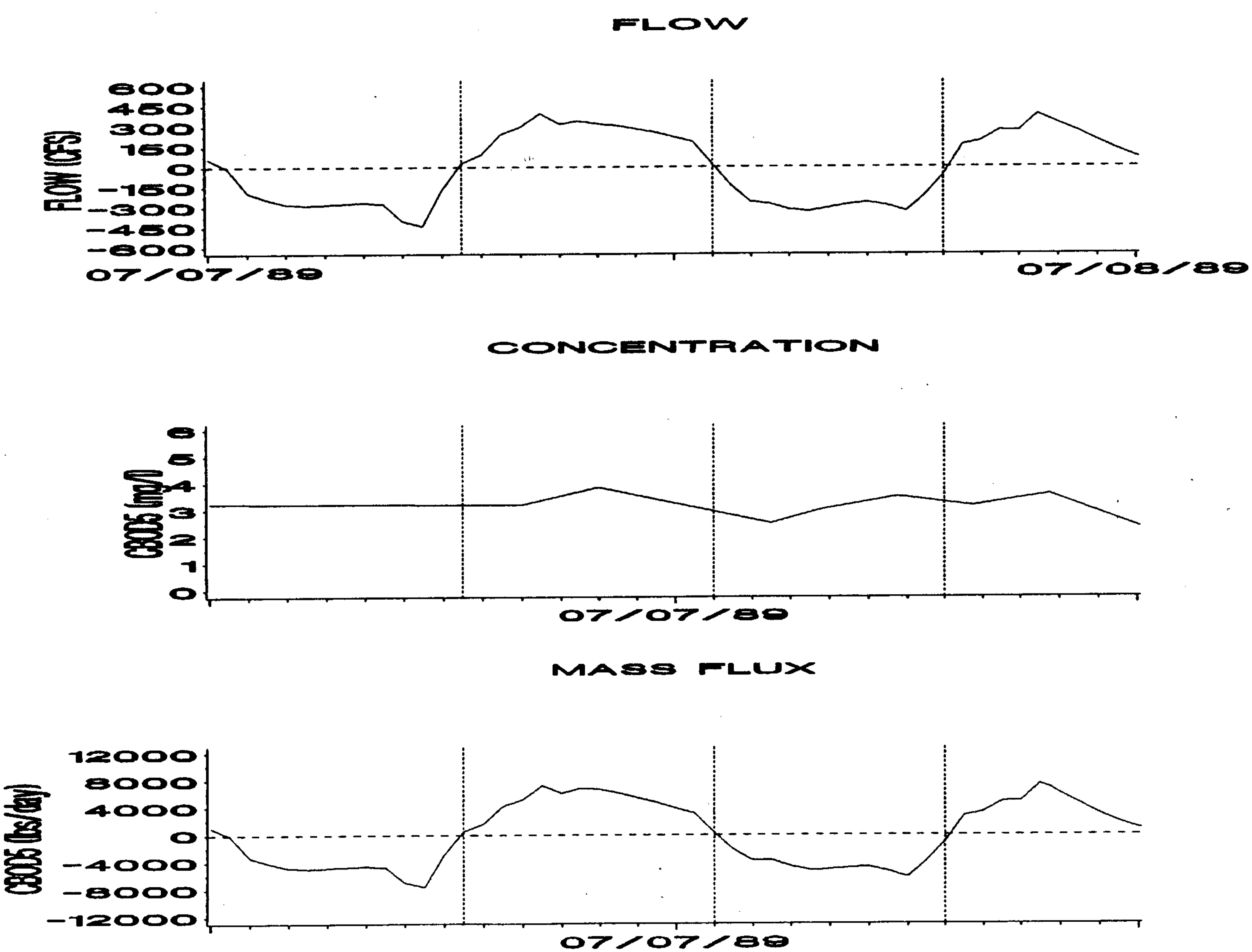
TOTAL NITROGEN CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



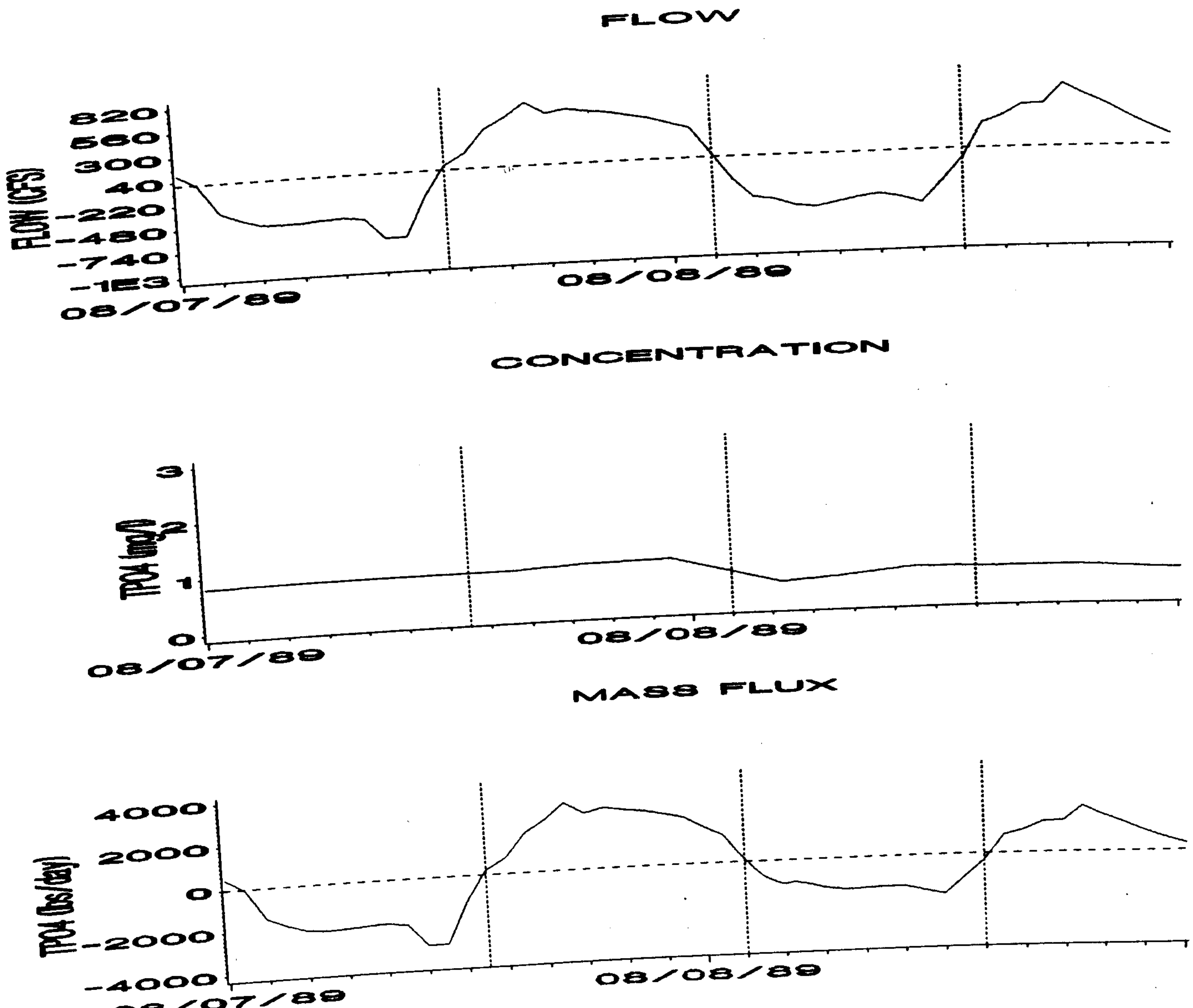
DO CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S15 - BERRYS CREEK UPSTREAM



CBOD5 CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S15 - BERRYS CREEK UPSTREAM

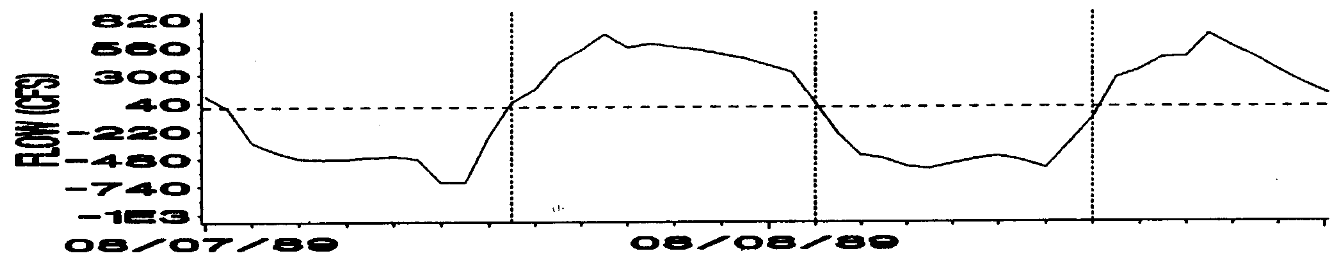


TOTAL PHOSPHATE CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S14 - BERRYS CREEK AT NJ TURNPIKE

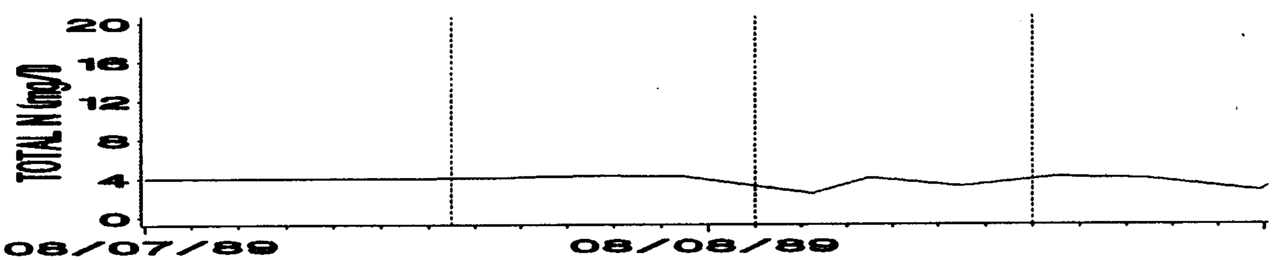


TOTAL NITROGEN CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S14 - BERRYS CREEK AT NJ TURNPIKE

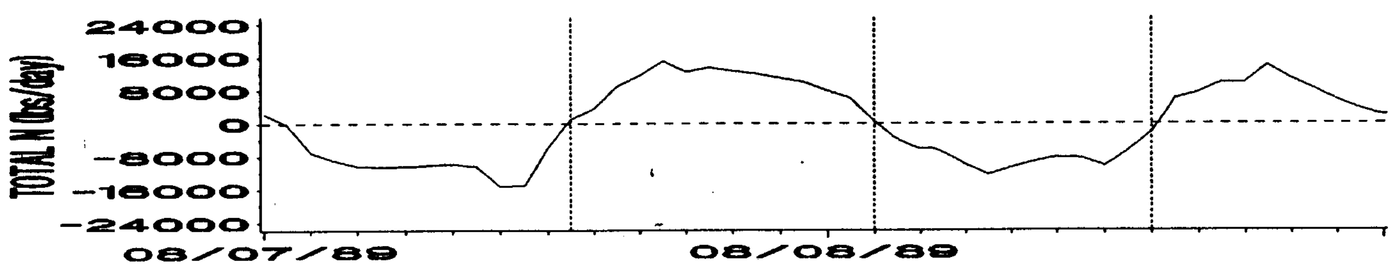
FLOW



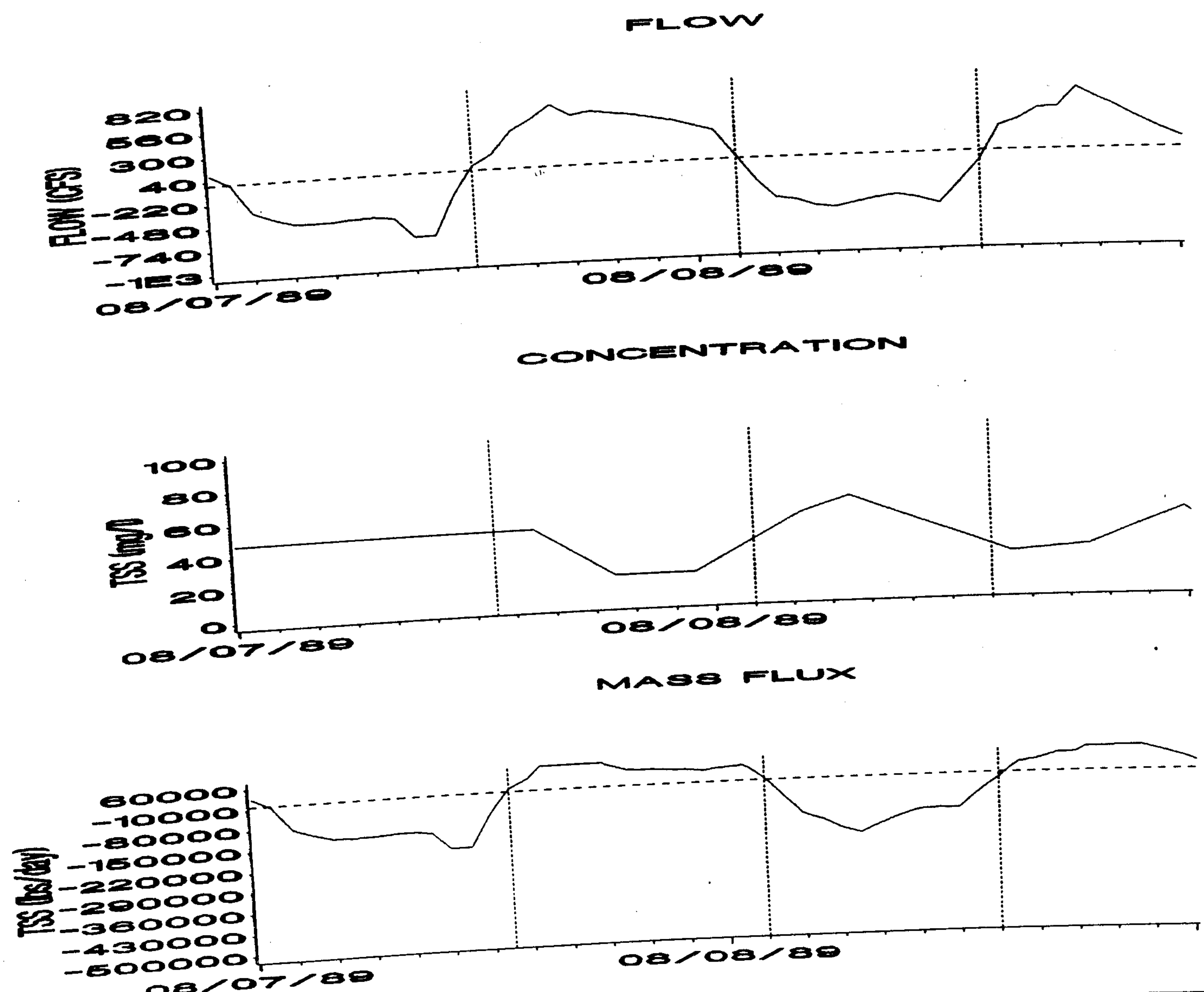
CONCENTRATION



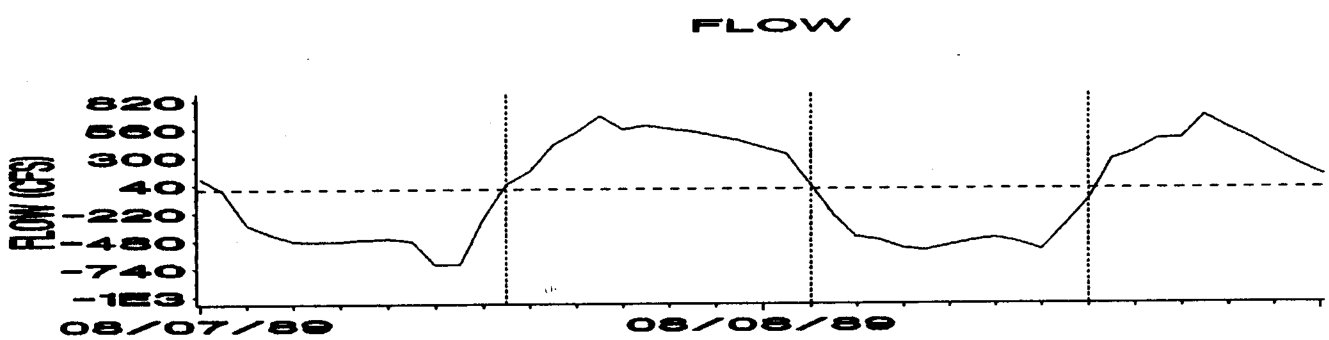
MASS FLUX

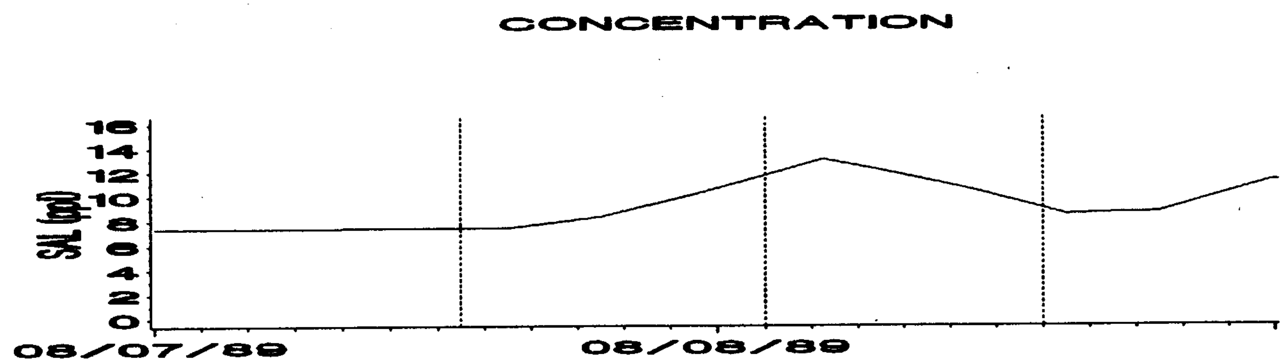


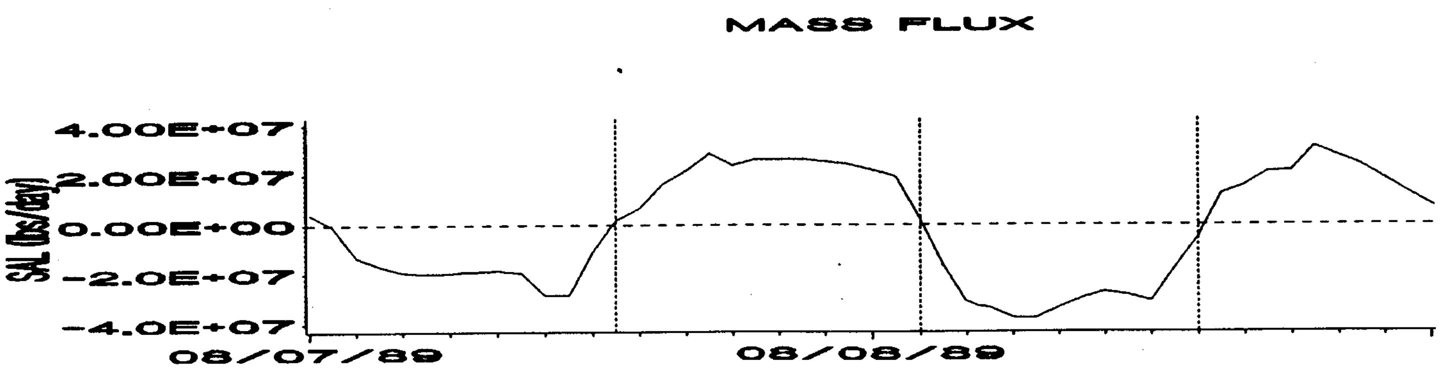
TSS CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S14 - BERRYS CREEK AT NJ TURNPIKE



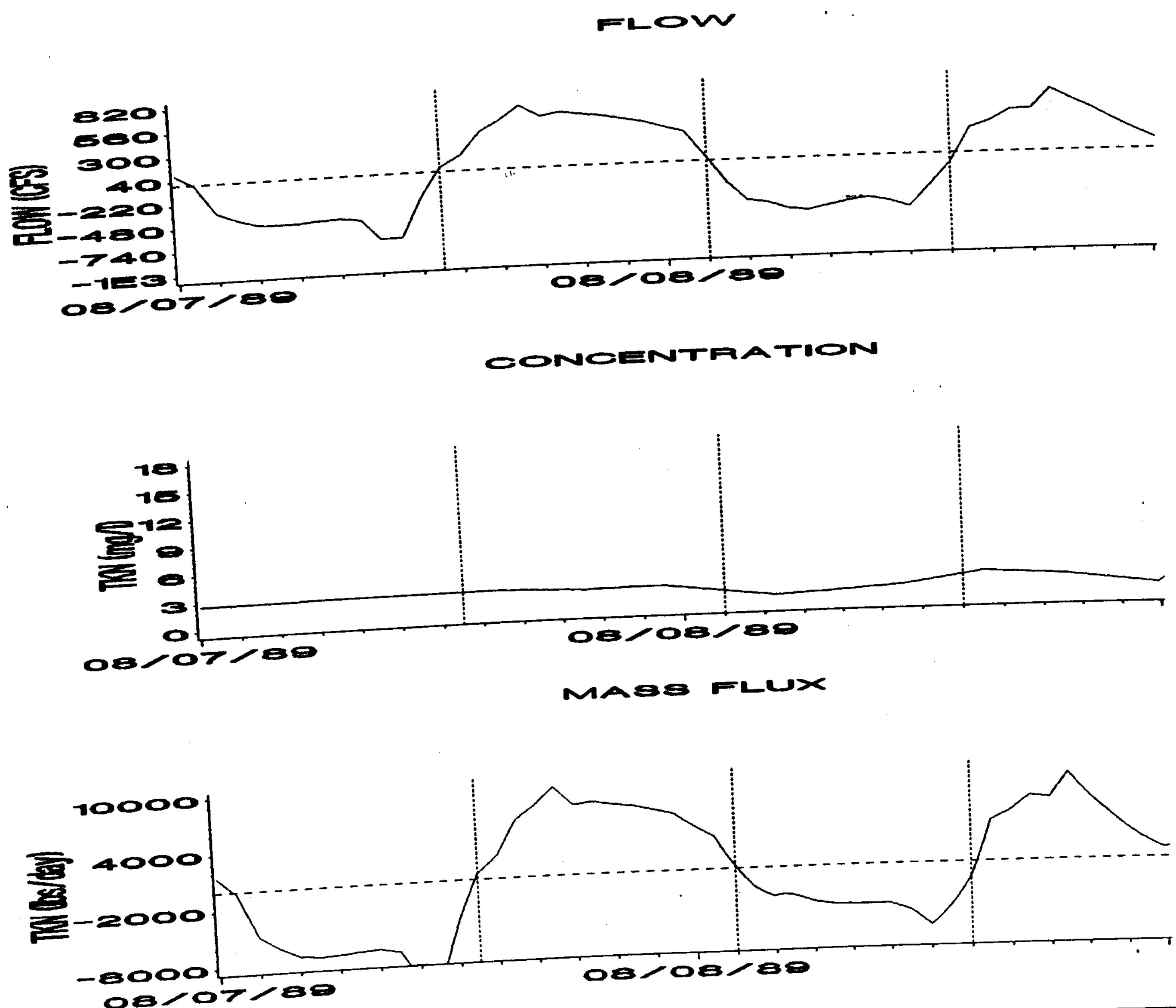
SALINITY CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S14 - BERRYS CREEK AT NJ TURNPIKE

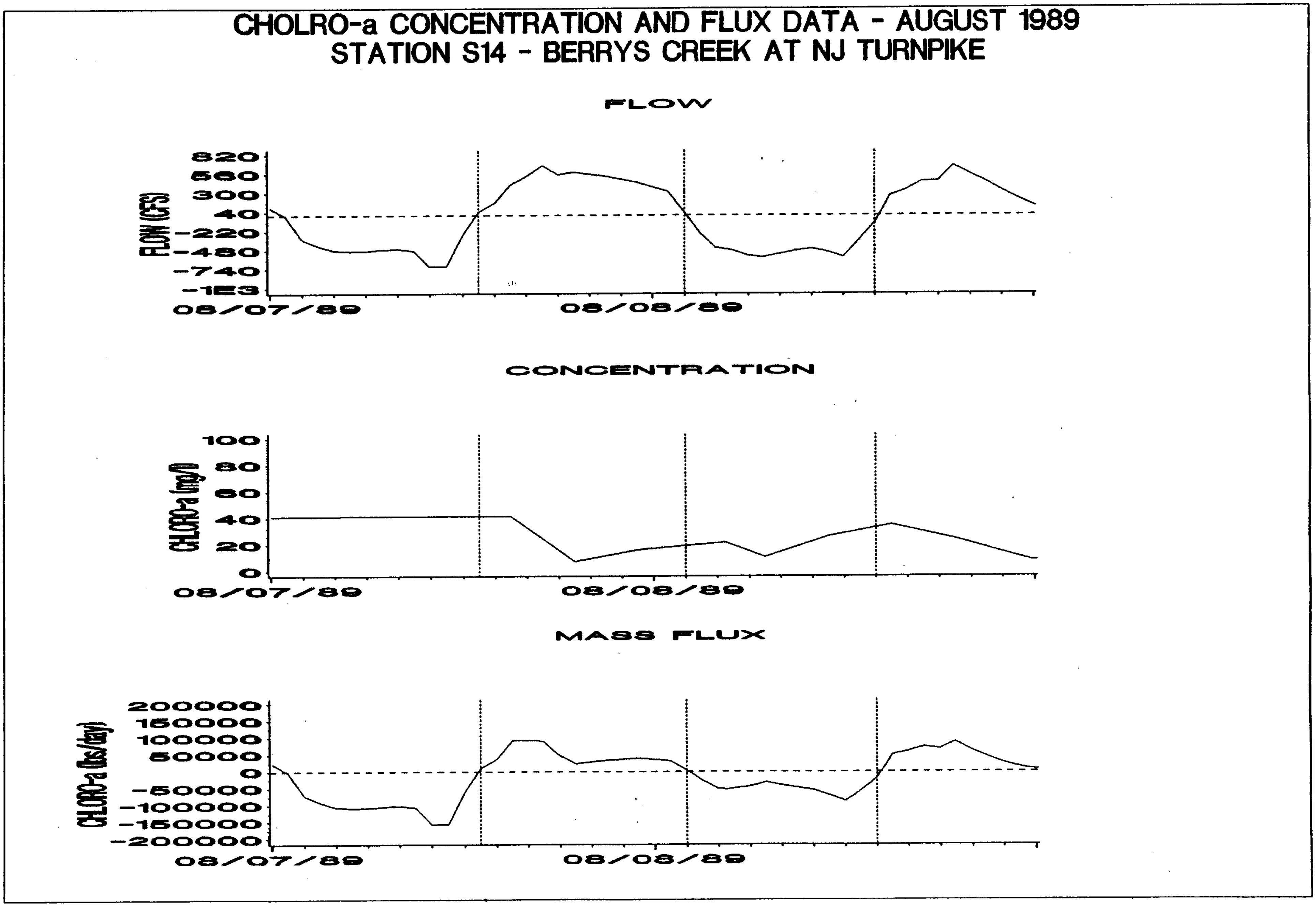






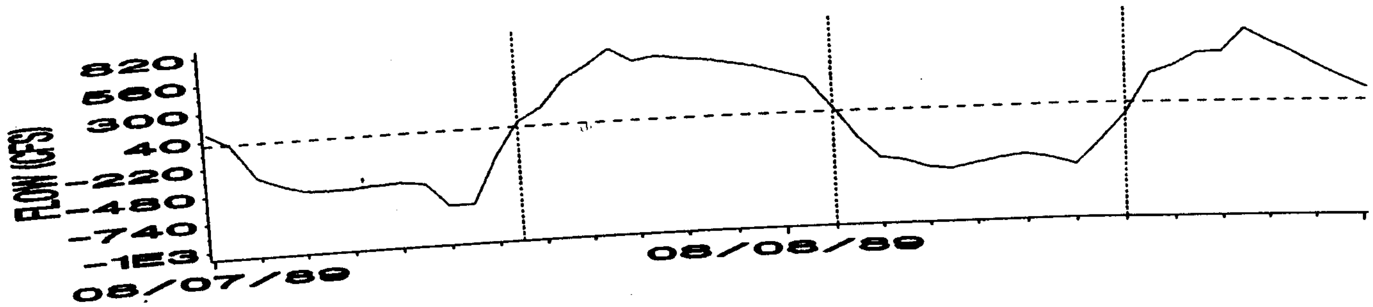
TKN CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S14 - BERRYS CREEK AT NJ TURNPIKE



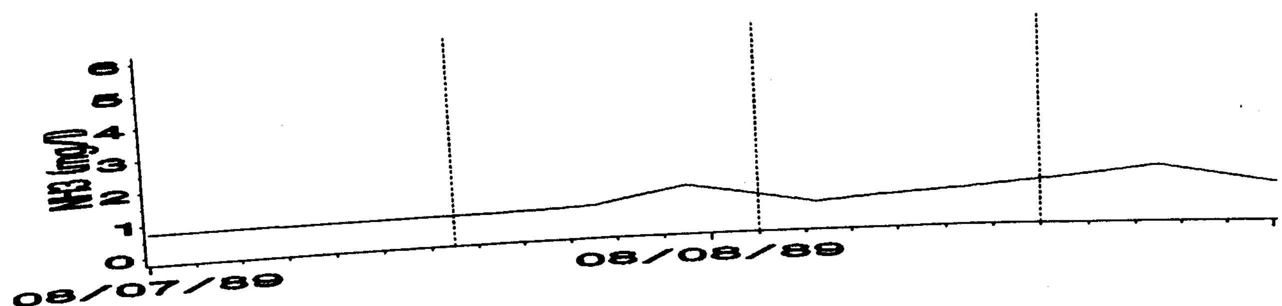

CHOLRO-a CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S14 - BERRYS CREEK AT NJ TURNPIKE
FLOW
FLOW (CFS)
820
560
300
40
-220
-480
-740
-1E3
08/07/89
08/08/89
CONCENTRATION
CHLORO-a (mg/l)
100
80
60
40
20
0
08/07/89
08/08/89
MASS FLUX
CHLORO-a (lbs/day)
200000
150000
100000
50000
0
-50000
-100000
-150000
-200000
08/07/89
08/08/89

NH3 CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S14 - BERRYS CREEK AT NJ TURNPIKE

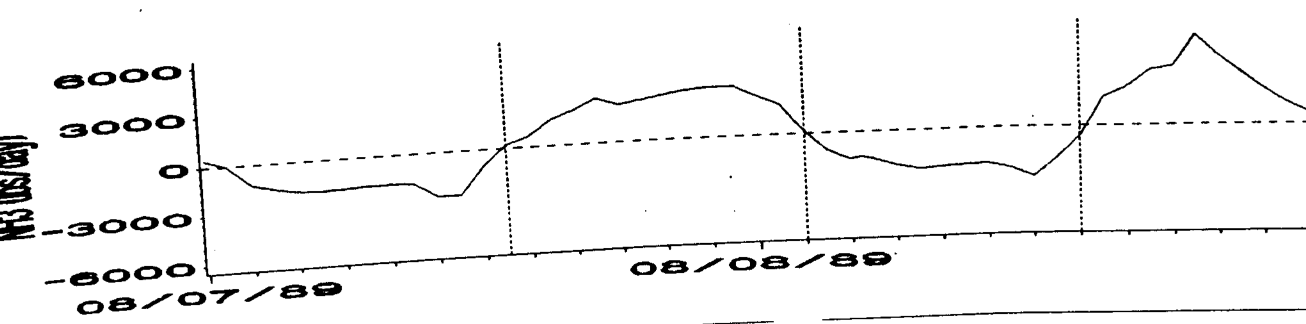
FLOW



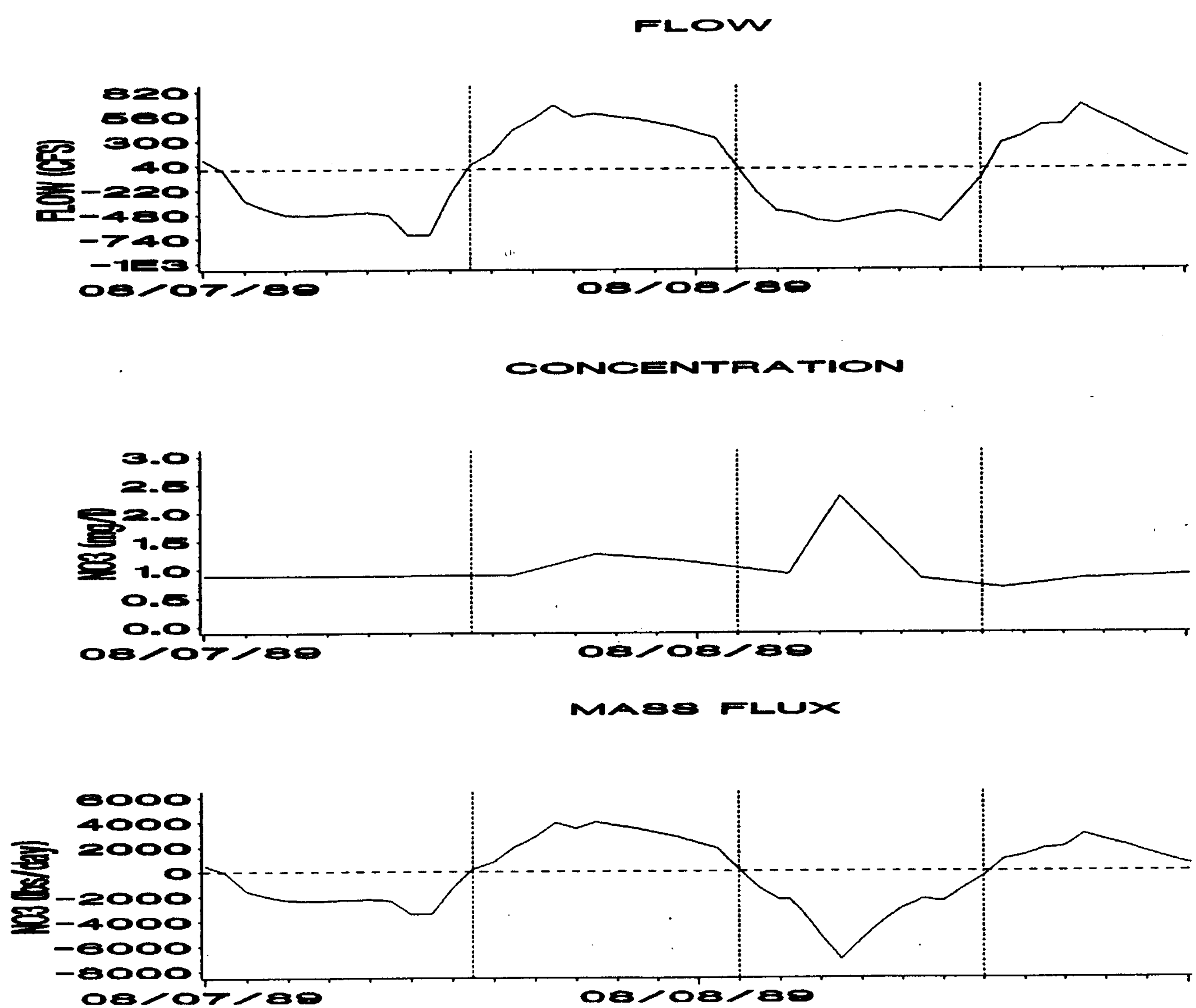
CONCENTRATION



MASS FLUX

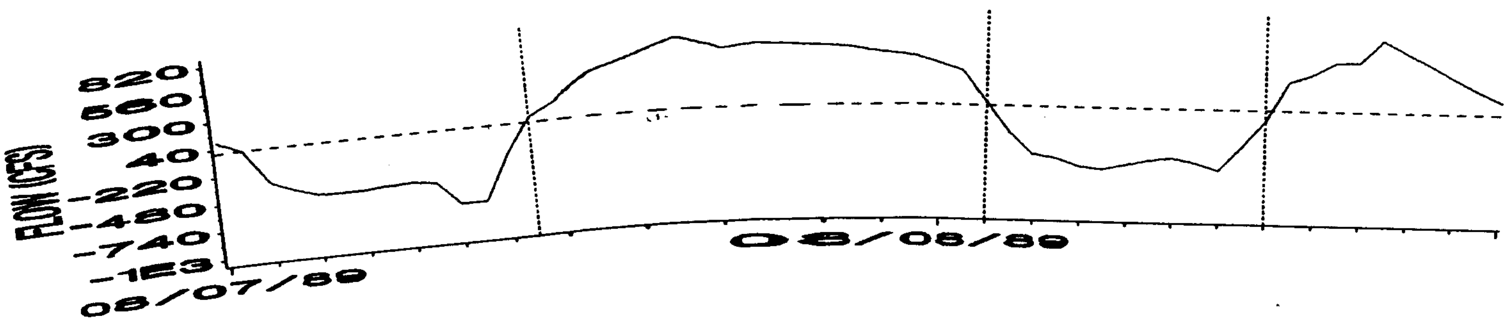


NITRATE CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S14 - BERRYS CREEK AT NJ TURNPIKE



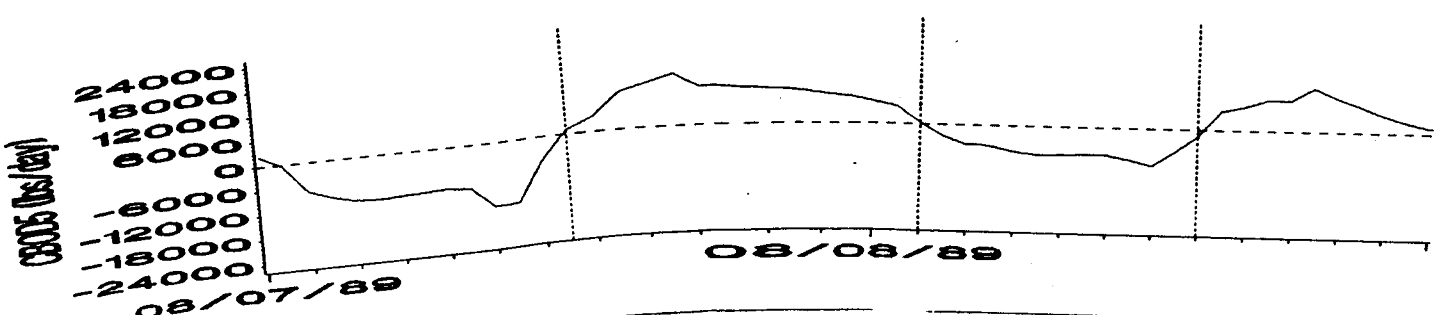
CBOD5 CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S14 - BERRYS CREEK AT NJ TURNPIKE

FLOW



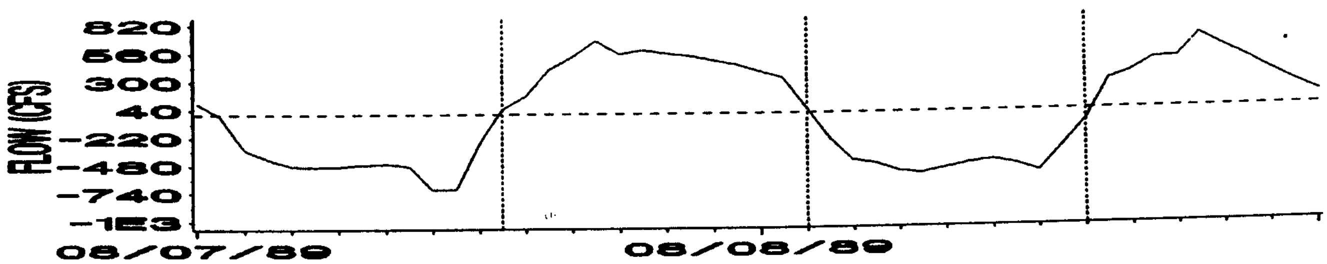
CONCENTRATION

MASS FLUX

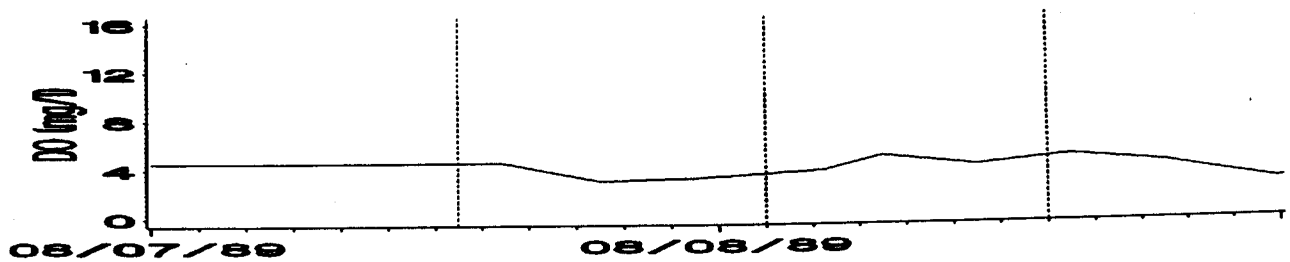


DO CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S14 - BERRYS CREEK AT NJ TURNPIKE

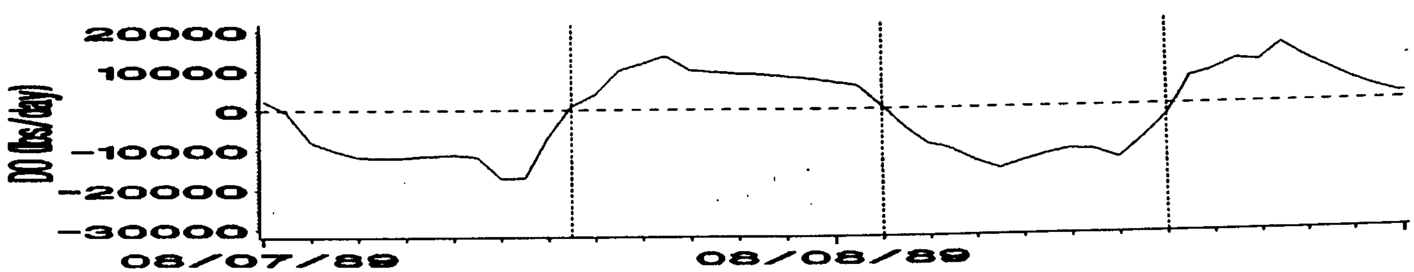
FLOW



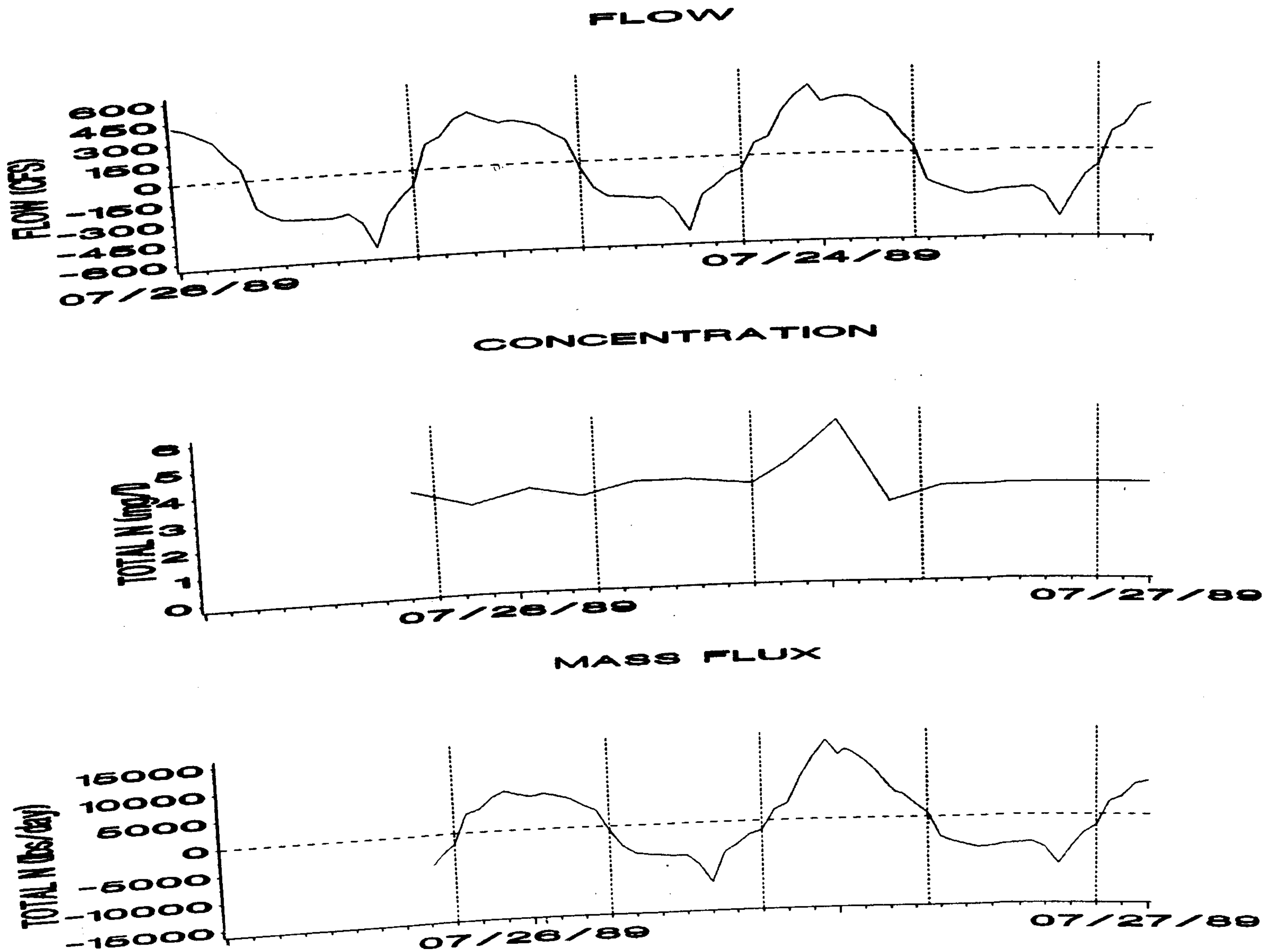
CONCENTRATION



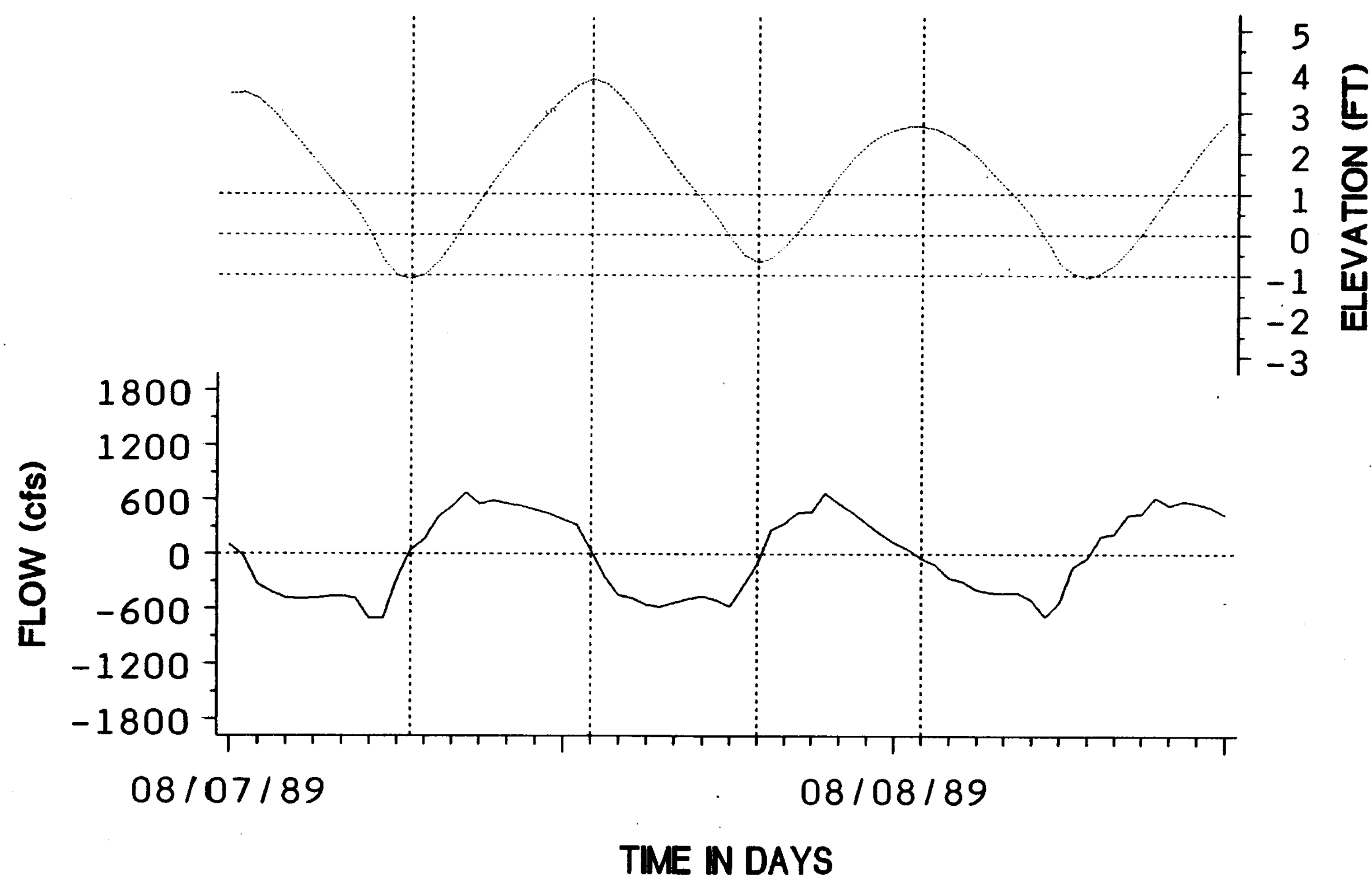
MASS FLUX

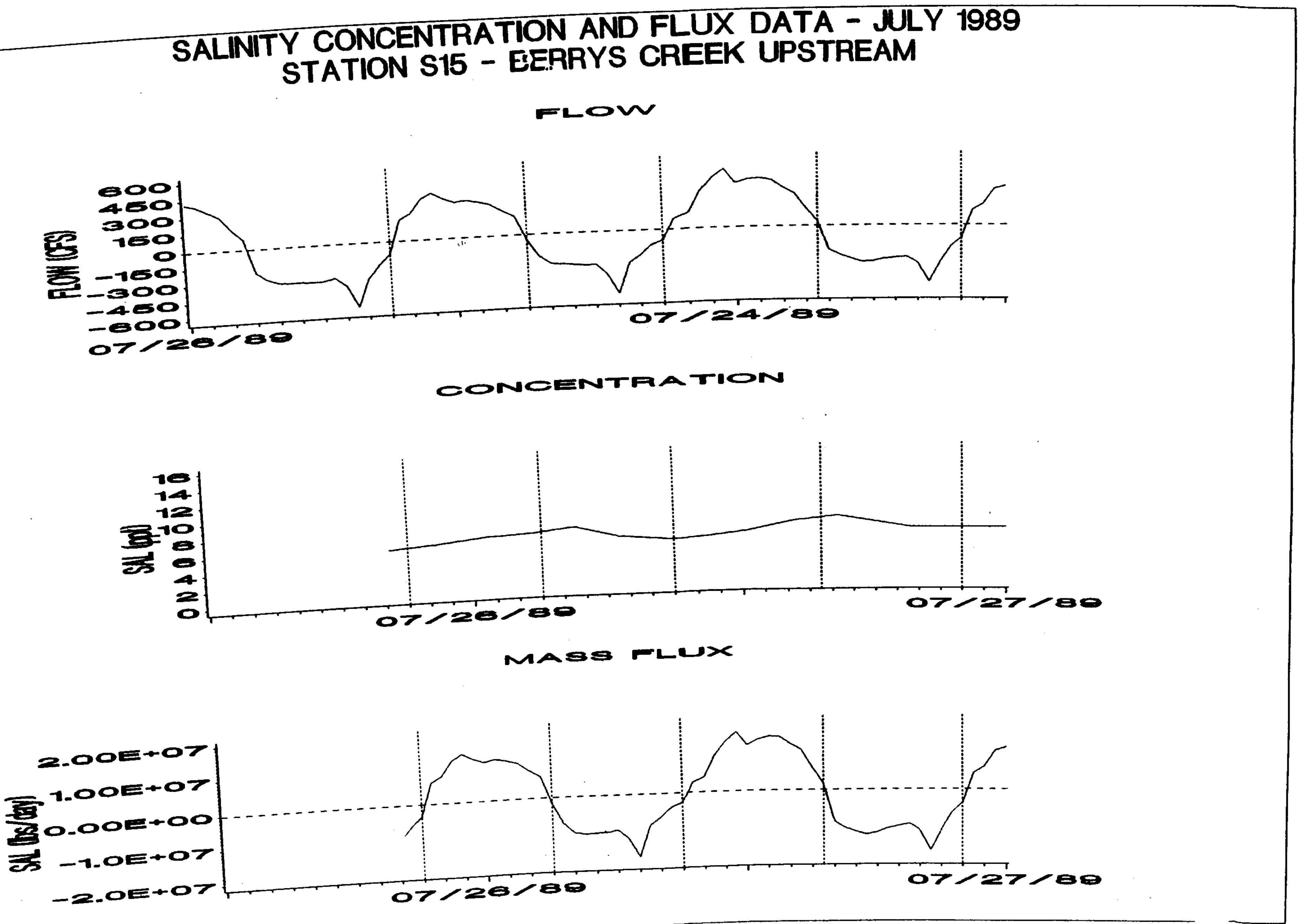


TOTAL NITROGEN CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



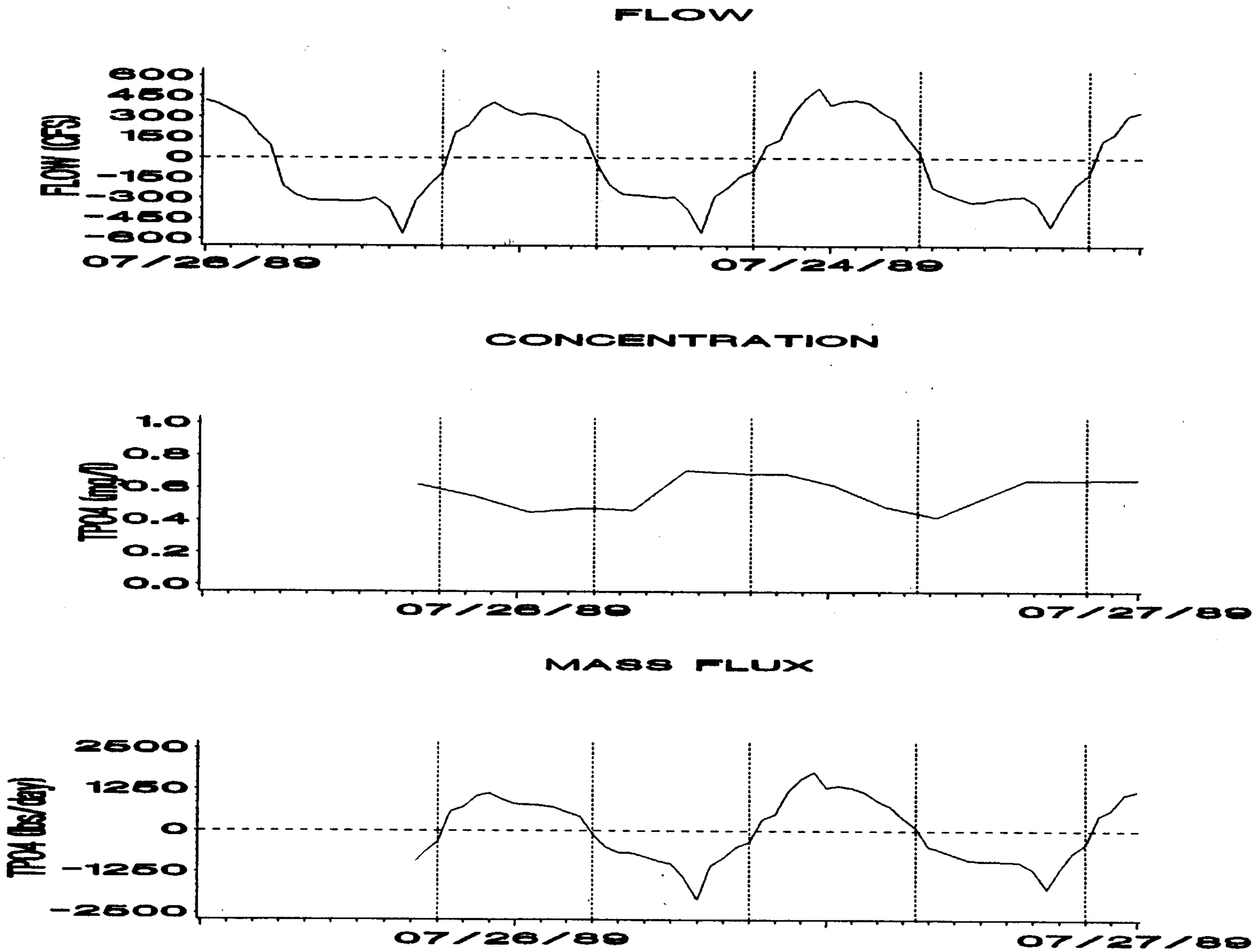
BERRYS CREEK AT TURNPIKE (S14) - AUGUST 1989 - FLOW VERSUS TIDE
FLOW CALCULATED FROM AREA*VEL, VELOCITY MEASURED BY OSI



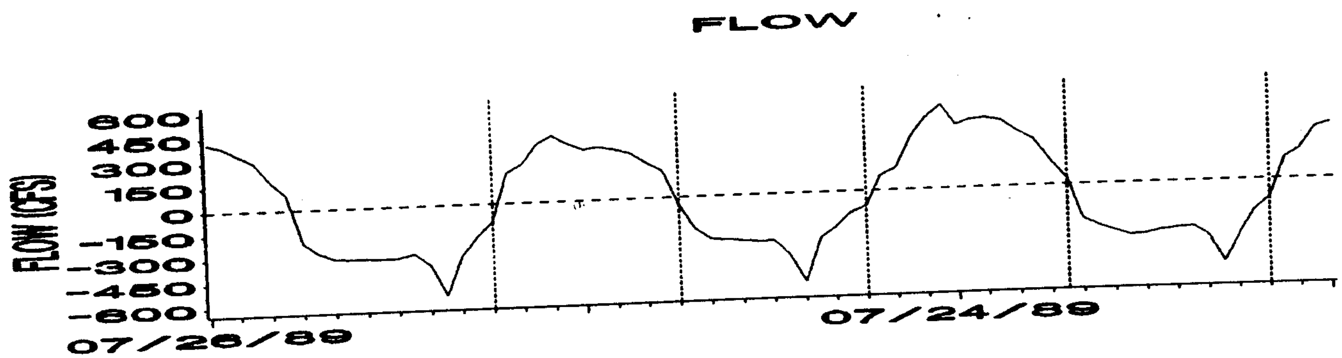


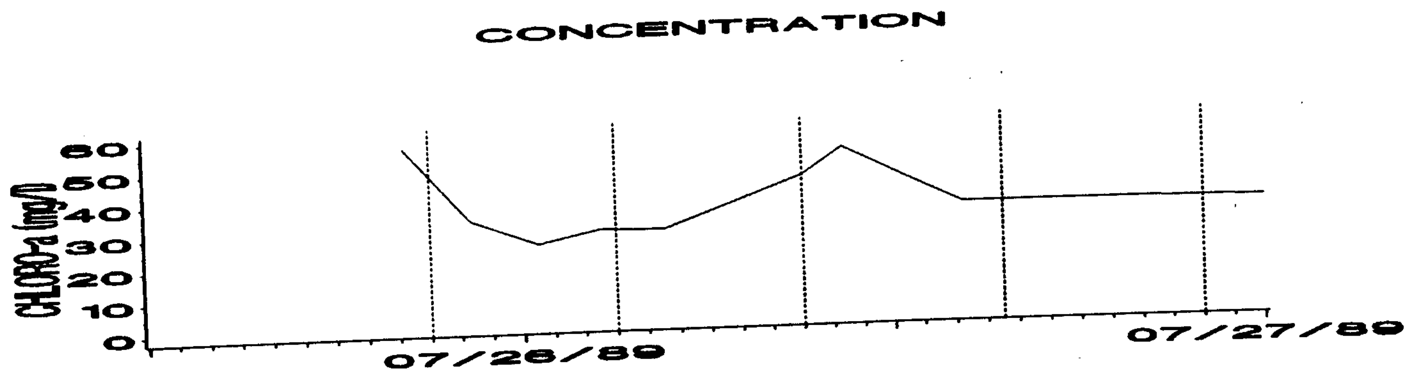
SALINITY CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM
FLOW
FLOW (CFS)
600
450
300
150
0
-150
-300
-450
-600
07/26/89
07/24/89
CONCENTRATION
SAL (ppt)
16
14
12
10
8
6
4
2
0
07/26/89
07/27/89
MASS FLUX
SAL (lbs/day)
2.00E+07
1.00E+07
0.00E+00
-1.0E+07
-2.0E+07
07/26/89
07/27/89

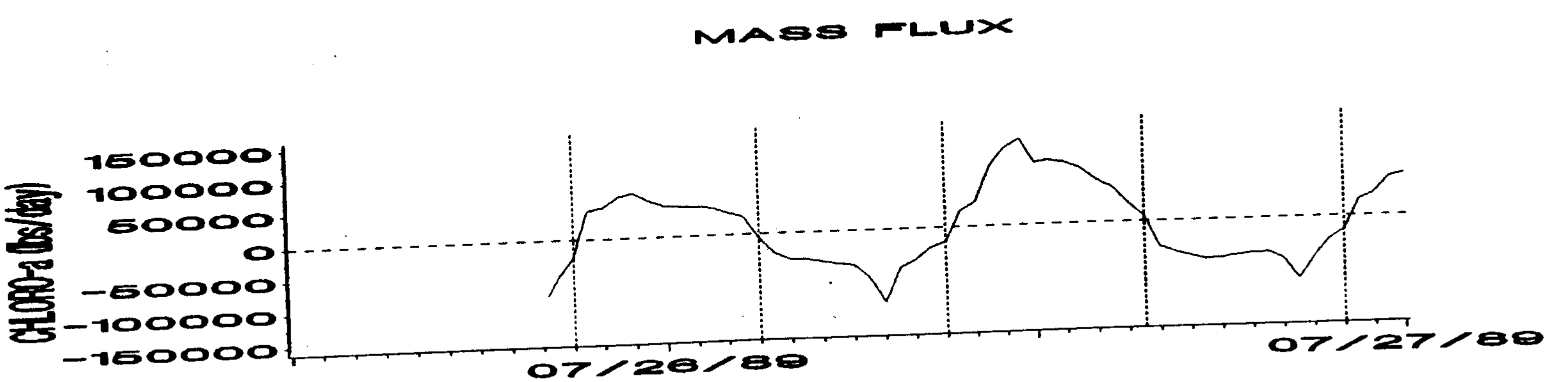
TOTAL PHOSPHATE CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



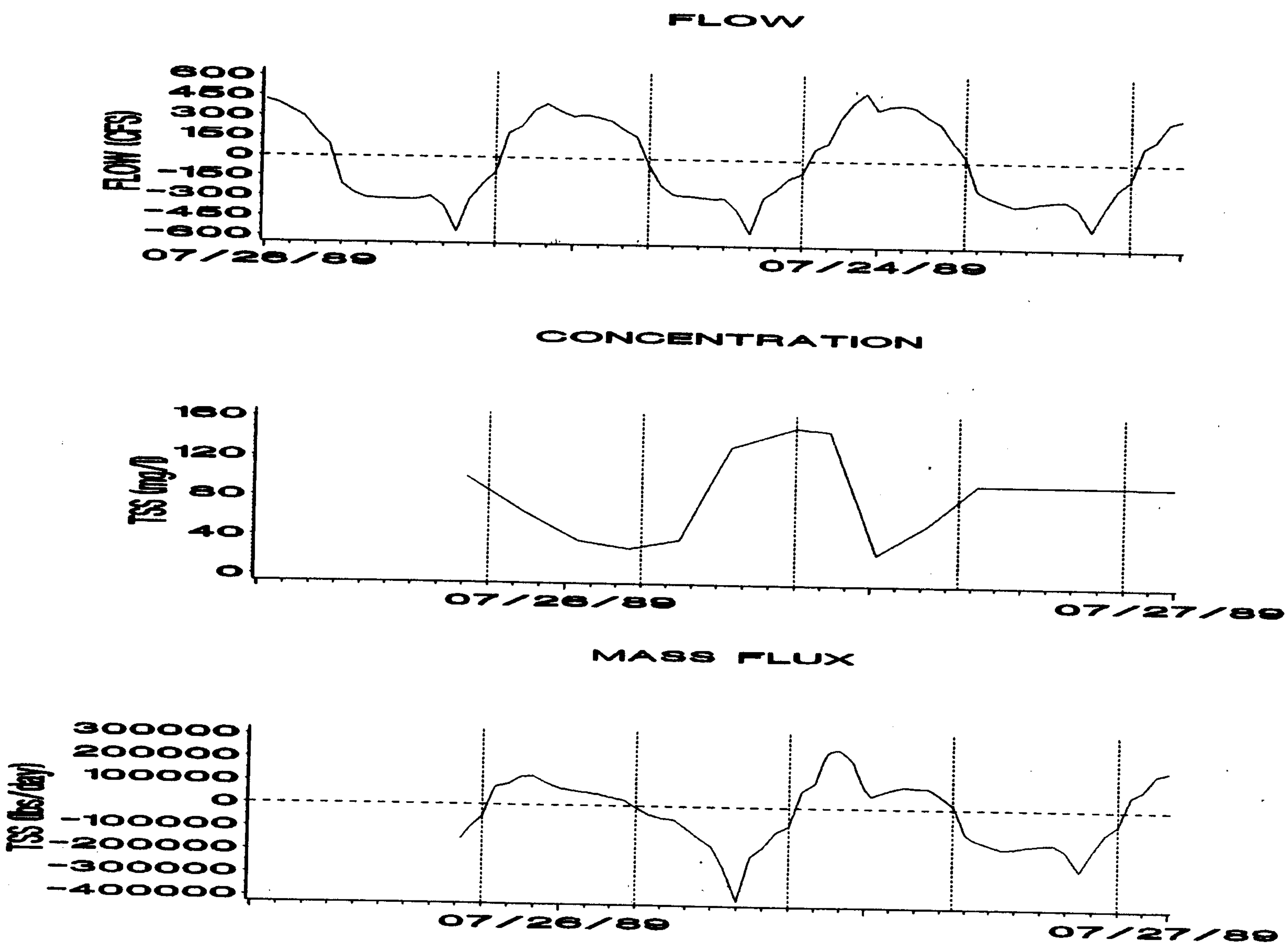
CHOLRO-a CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



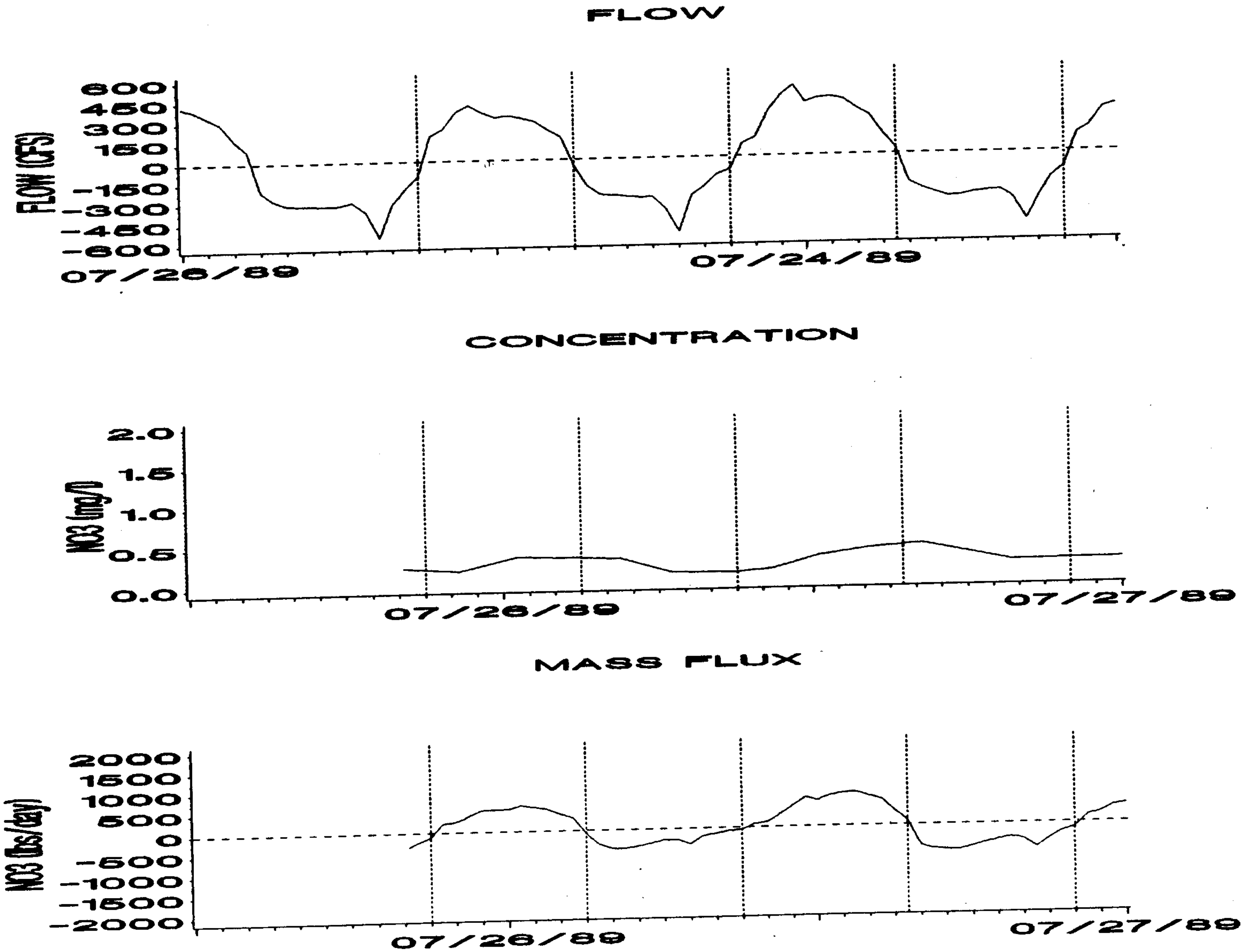




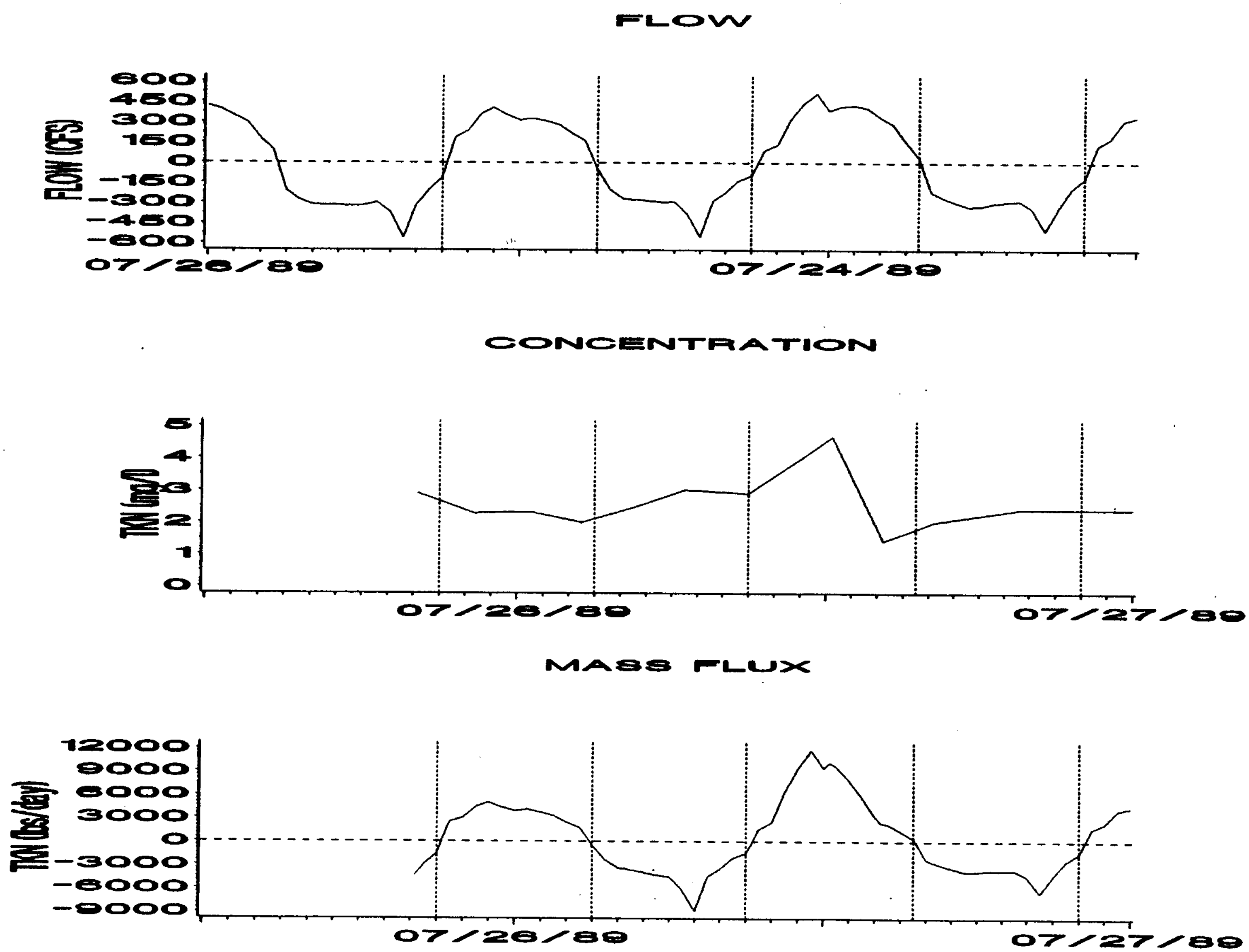
TSS CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



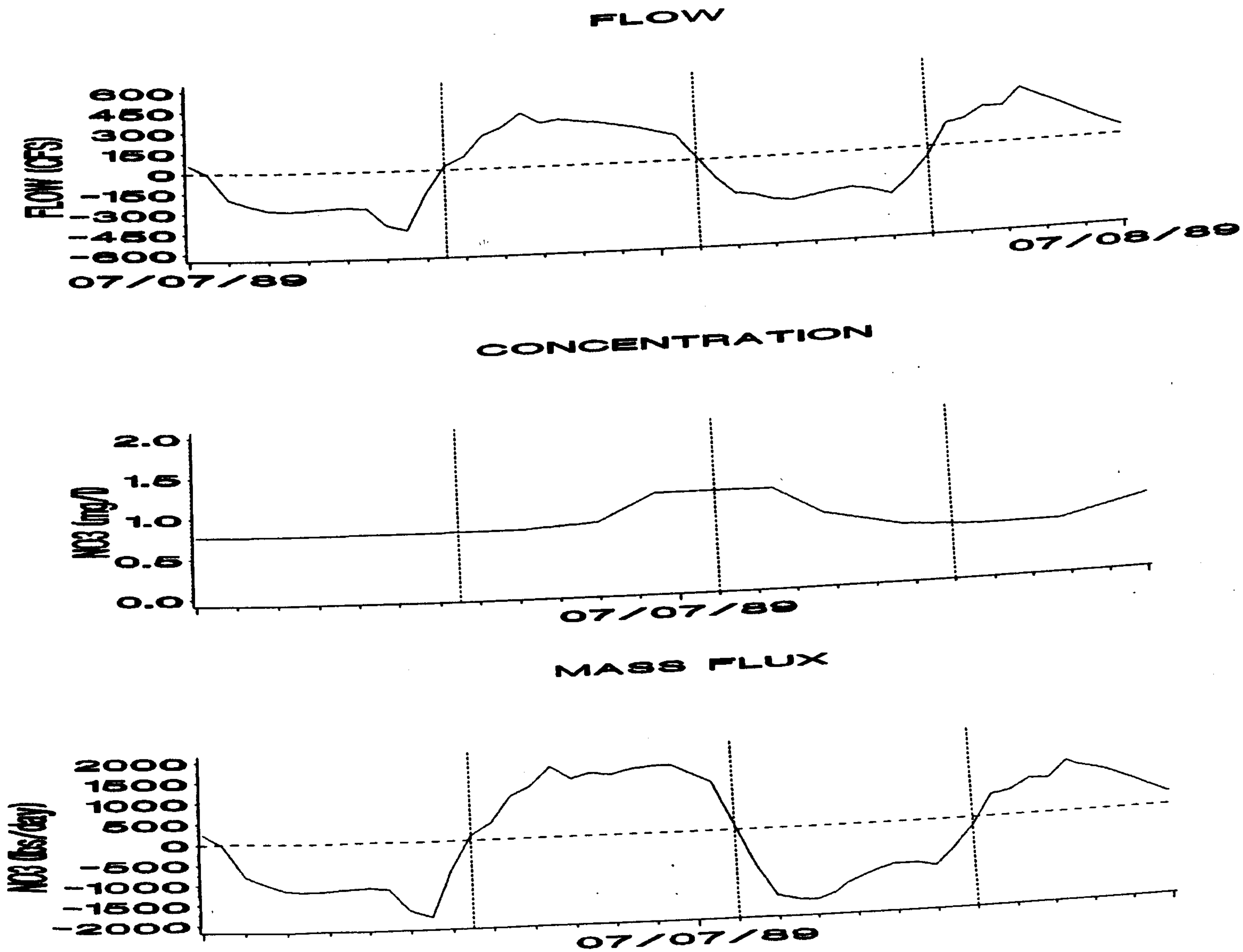
NITRATE CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



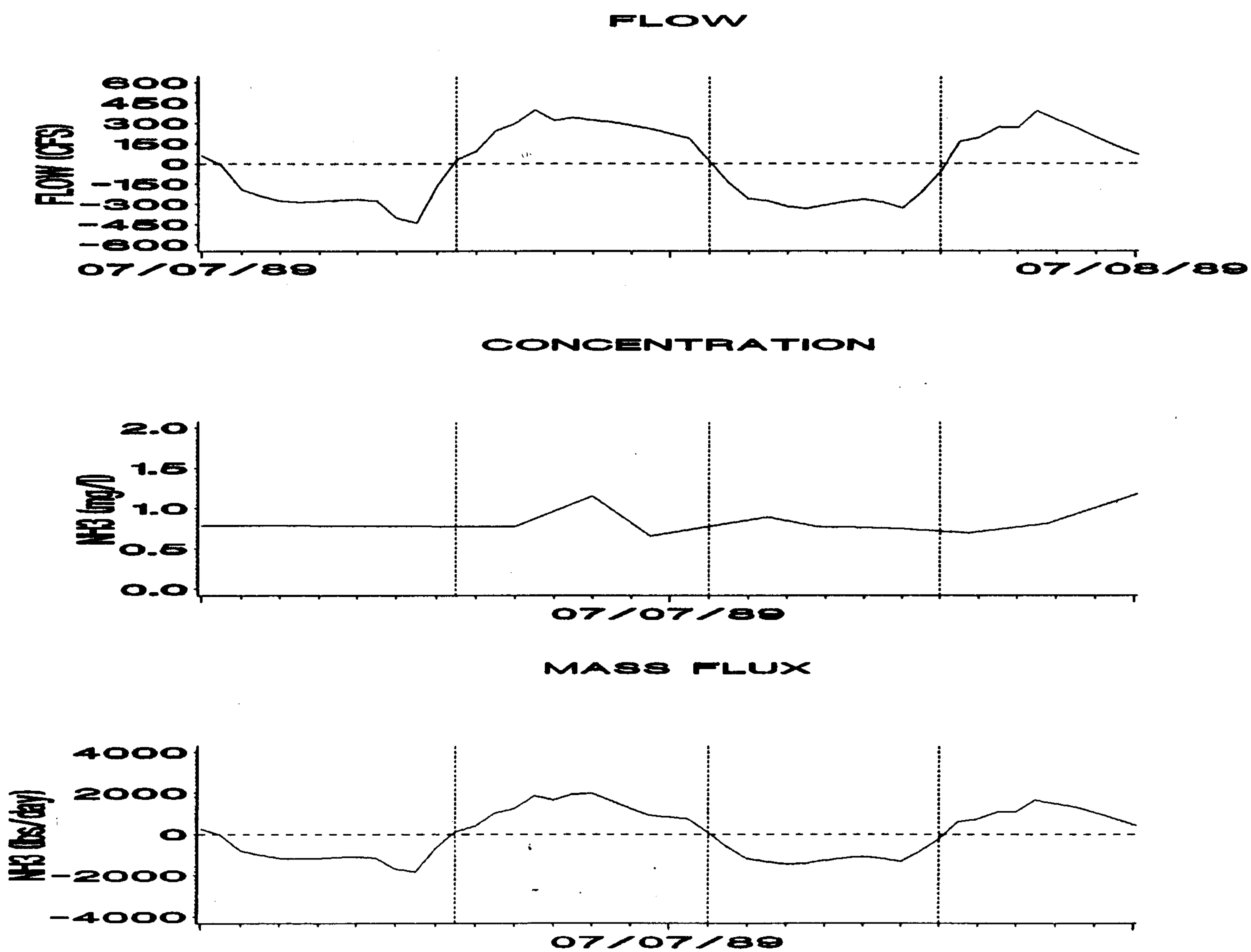
TKN CONCENTRATION AND FLUX DATA - JULY 1989
STATION S15 - BERRYS CREEK UPSTREAM



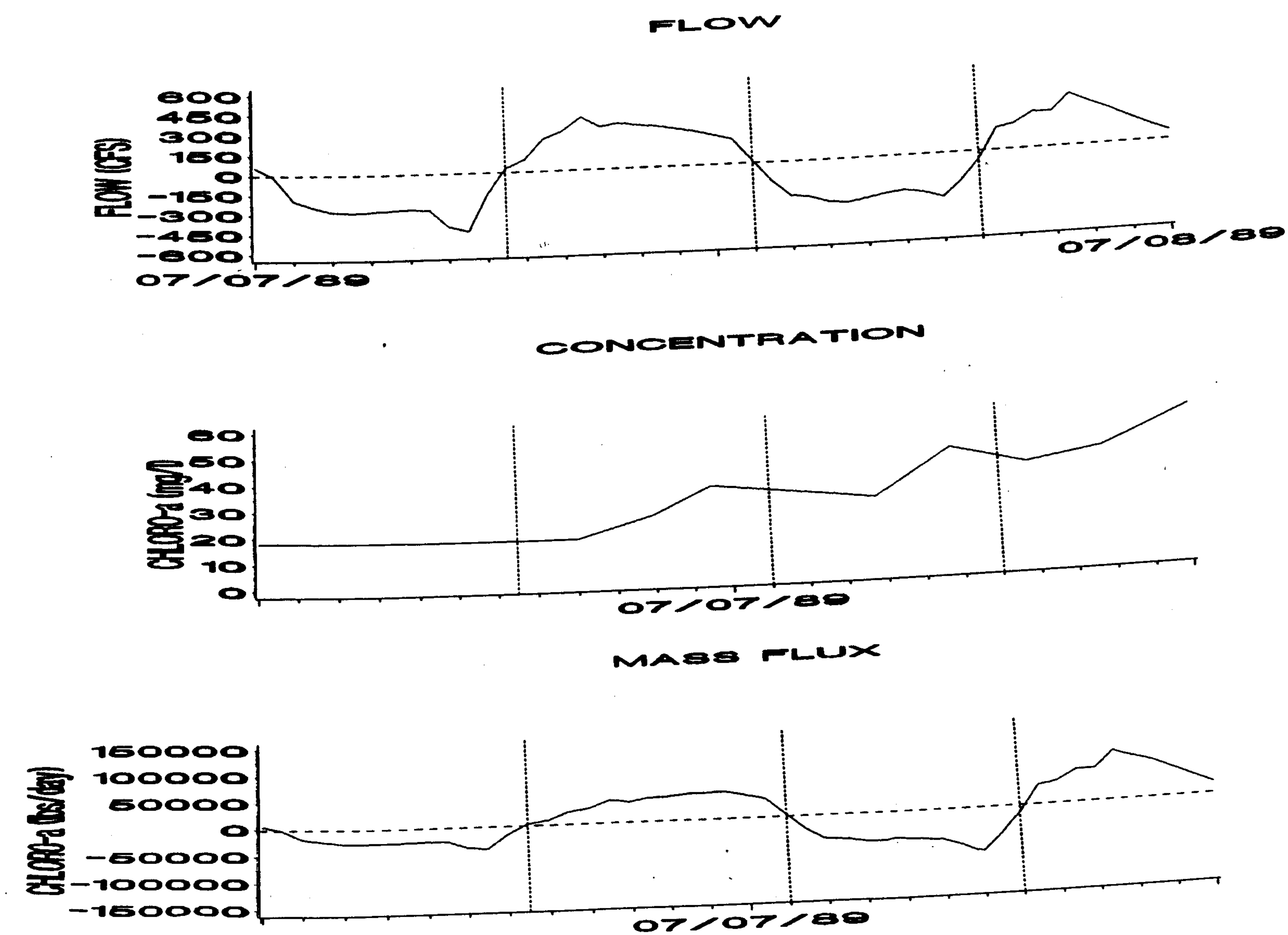
NITRATE CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S15 - BERRYS CREEK UPSTREAM



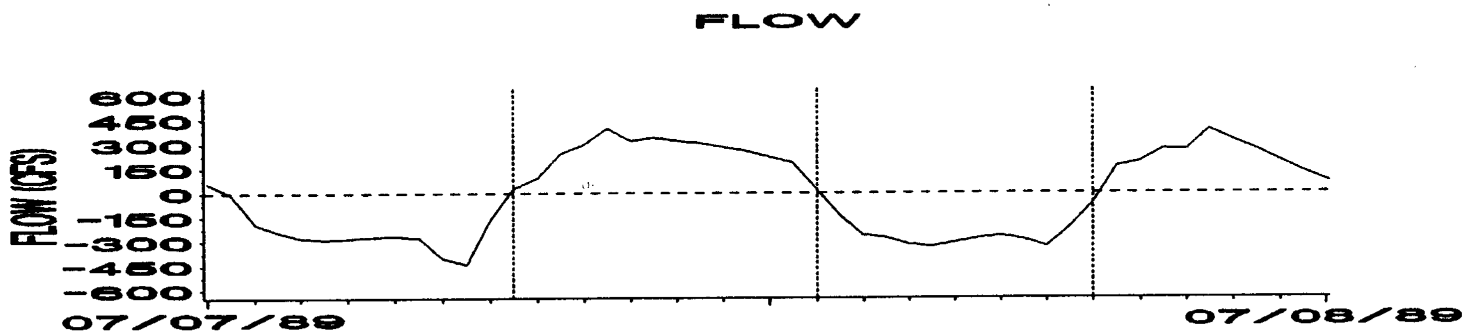
NH3 CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S15 - BERRYS CREEK UPSTREAM

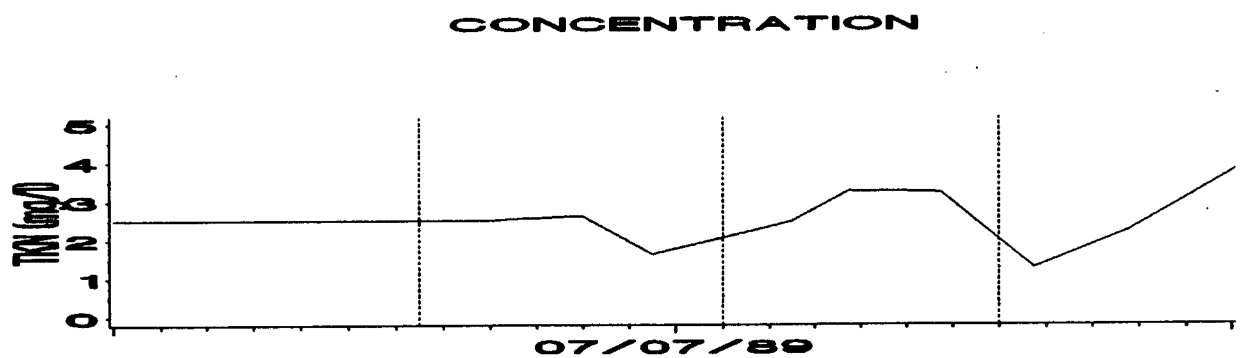


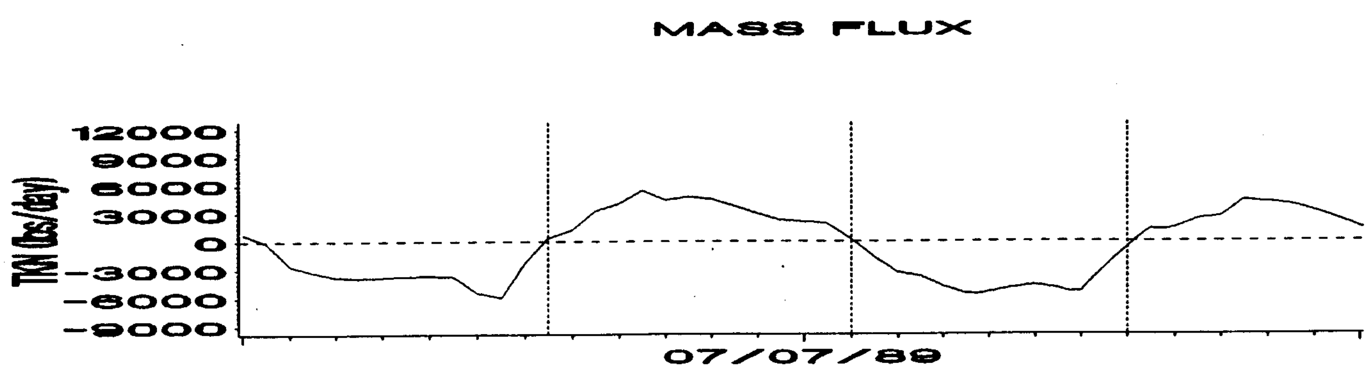
CHOLRO-a CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S15 - BERRYS CREEK UPSTREAM



TKN CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S15 - BERRYS CREEK UPSTREAM

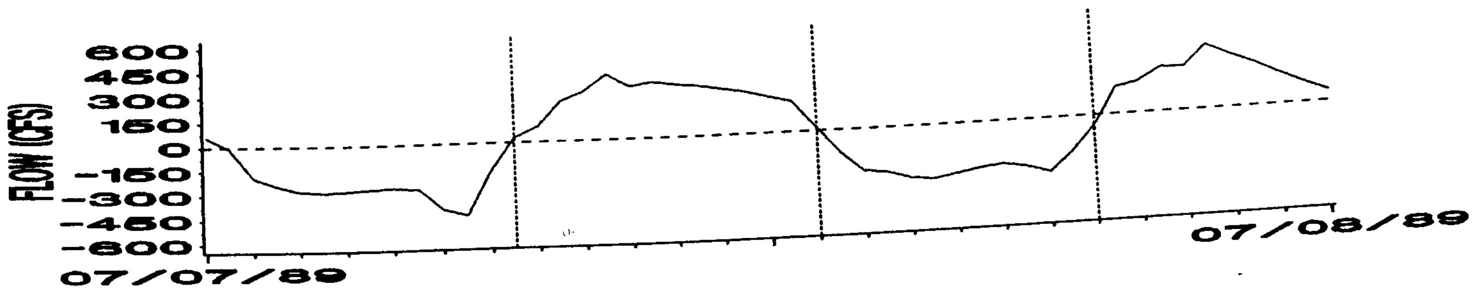




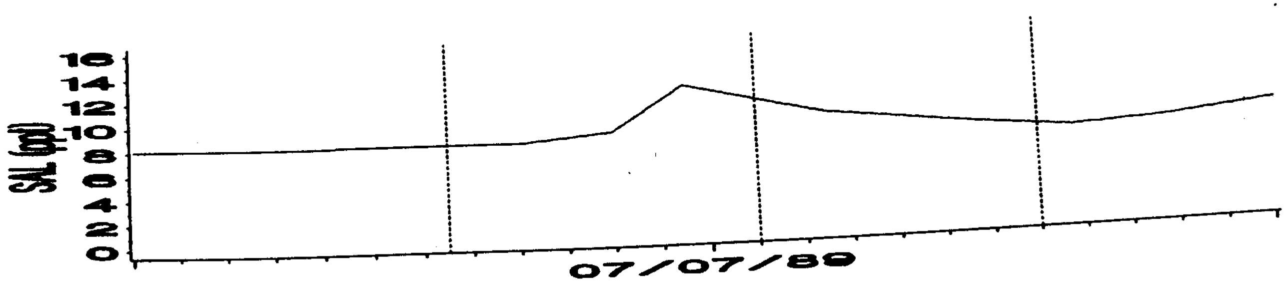


SALINITY CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S15 - BERRYS CREEK UPSTREAM

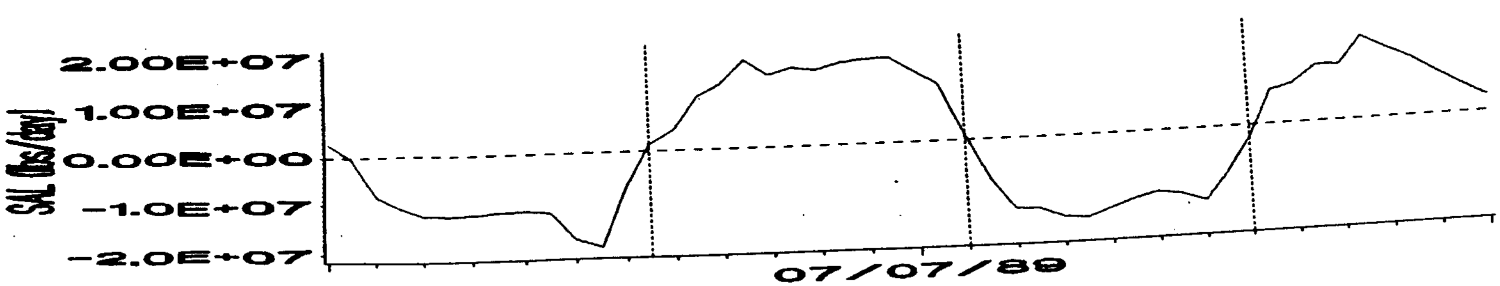
FLOW



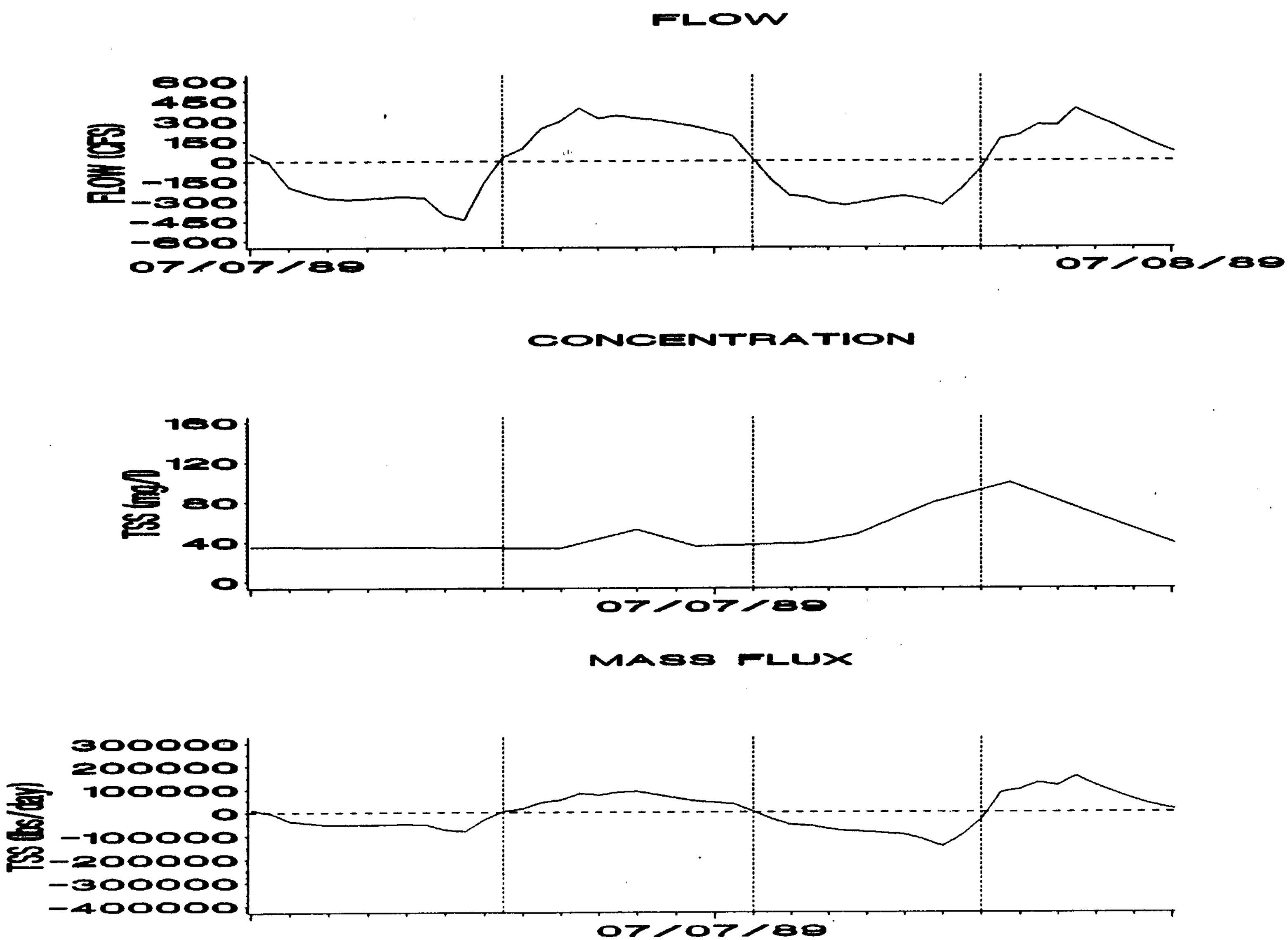
CONCENTRATION



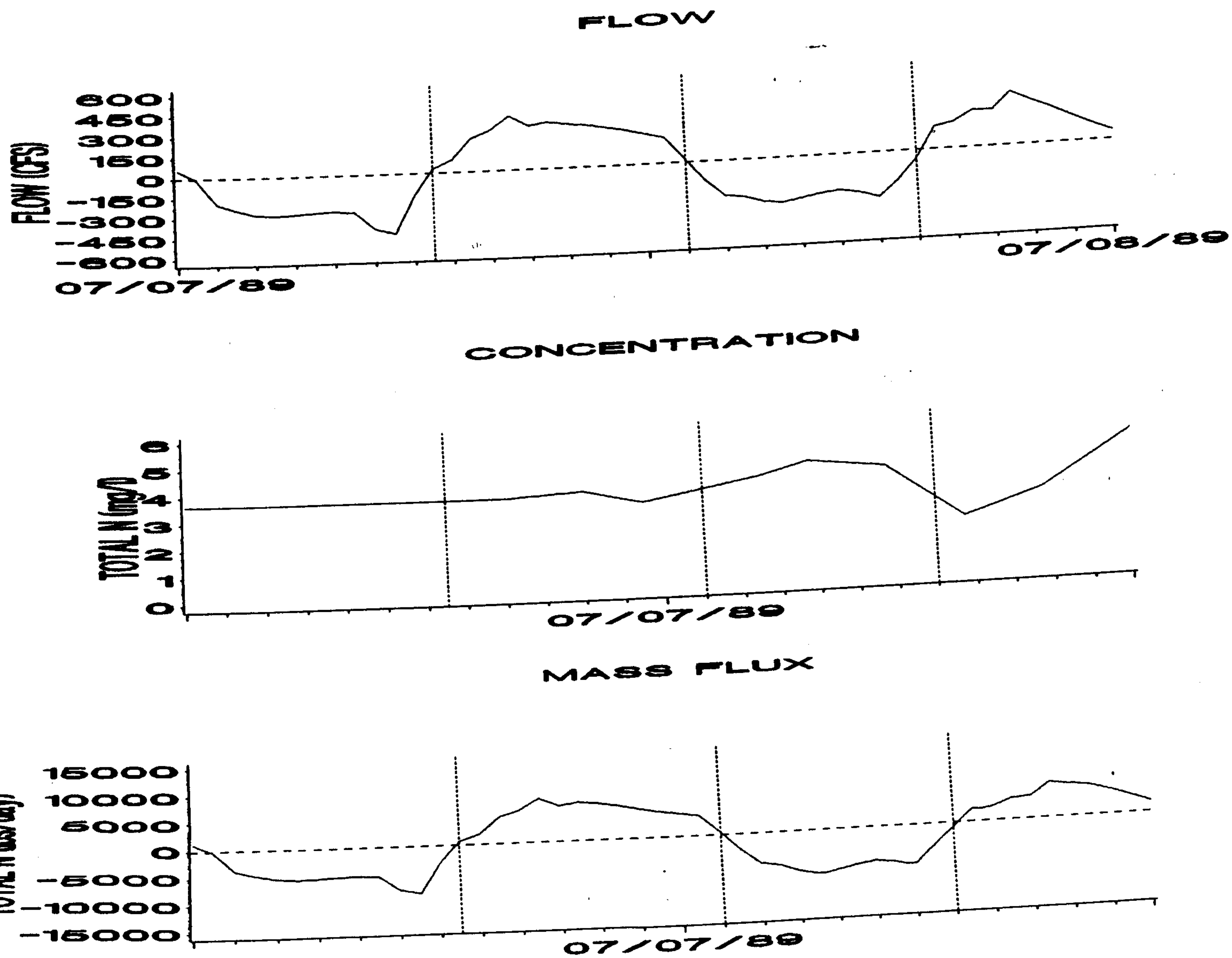
MASS FLUX



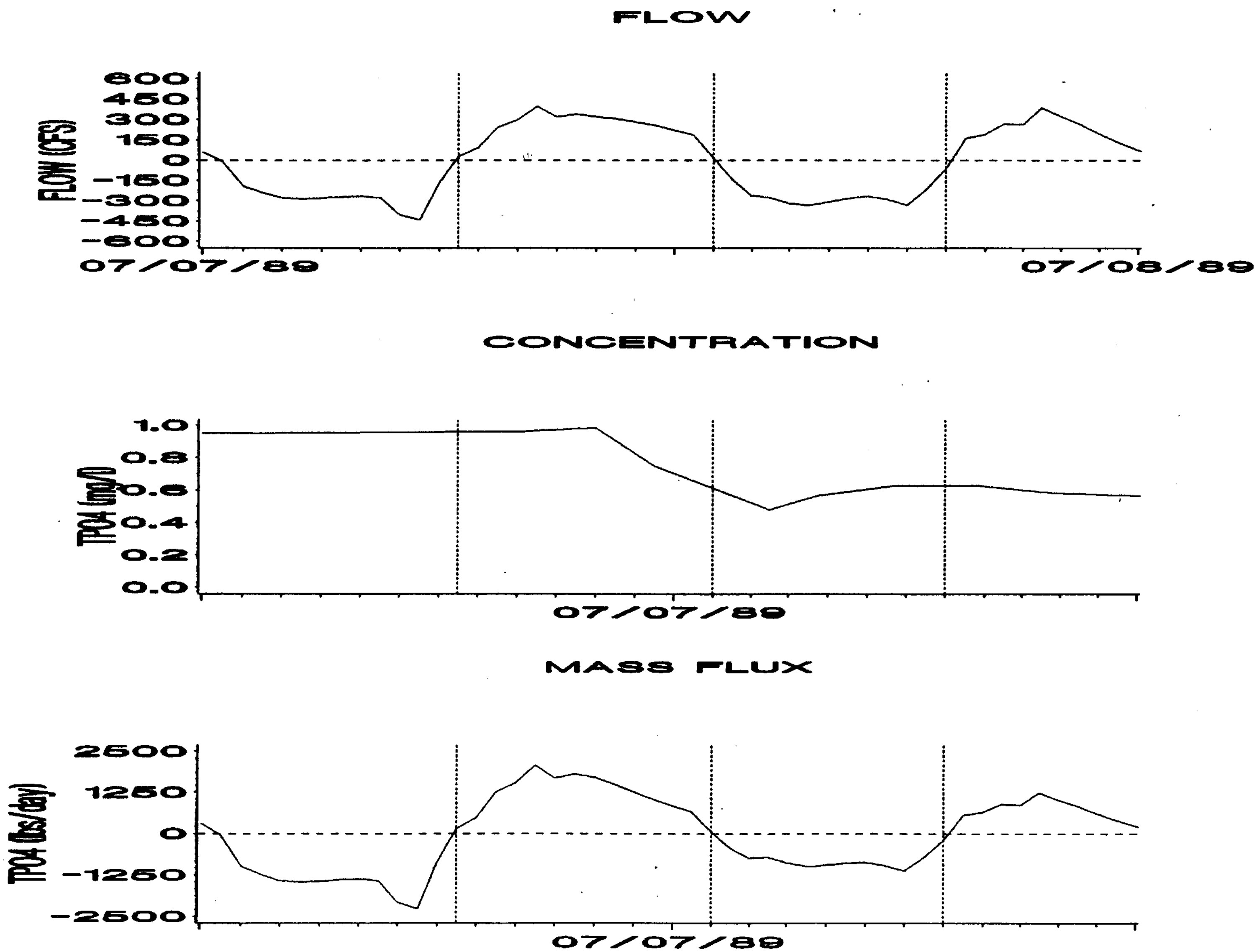
TSS CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S15 - BERRYS CREEK UPSTREAM



TOTAL NITROGEN CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S15 - BERRYS CREEK UPSTREAM



TOTAL PHOSPHATE CONCENTRATION AND FLUX DATA - AUGUST 1989
STATION S15 - BERRYS CREEK UPSTREAM



APPENDIX A-2-3

Lower Hackensack River Nutrient Study

SUMMARY

The Hackensack River estuary receives nutrient inputs from multiple sources. Because much of the River's fresh water is impounded for drinking water supply, fresh water input to the estuary is very low. Therefore, the residence time for nutrients in the estuary is controlled primarily by tidal flushing and by a combination of biological and geochemical processes. The data reviewed for this report focus on the BCUA sewage treatment plant as a source of nutrients, particularly ammonium nitrogen. Input to the estuary from the treatment plant is sufficient to maintain substantially elevated ammonium concentrations both upstream and downstream from the plant. About 10% of the ammonium is nitrified to nitrate in the water column downstream from the plant. Up to 26% of the BCUA ammonium nitrogen load may ultimately be denitrified in the estuary's sediments and returned to the atmosphere as nitrogen gas. The ammonium flux from the sediments along the River axis was ordinarily 5% to 20% of the loading from the treatment plant, with occasional surges to about 50%. Nitrate flux from the sediments was typically less than 10% of the nitrogen contributed by the plant. The total nitrogen and total phosphorus data for April 1988 do not show strong potential limitation of algal biomass by either nitrogen or phosphorus in the reaches near the treatment plant. The estuary is subjected to vigorous tidal mixing which keeps fine-grained sediment in suspension, thereby extinguishing light rapidly with water depth. Therefore, the scarcity of light for photosynthesis

LOWER HACKENSACK RIVER NUTRIENT STUDY

Nutrient Distribution Patterns,

Sediment Oxygen Demand, and Denitrification

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for

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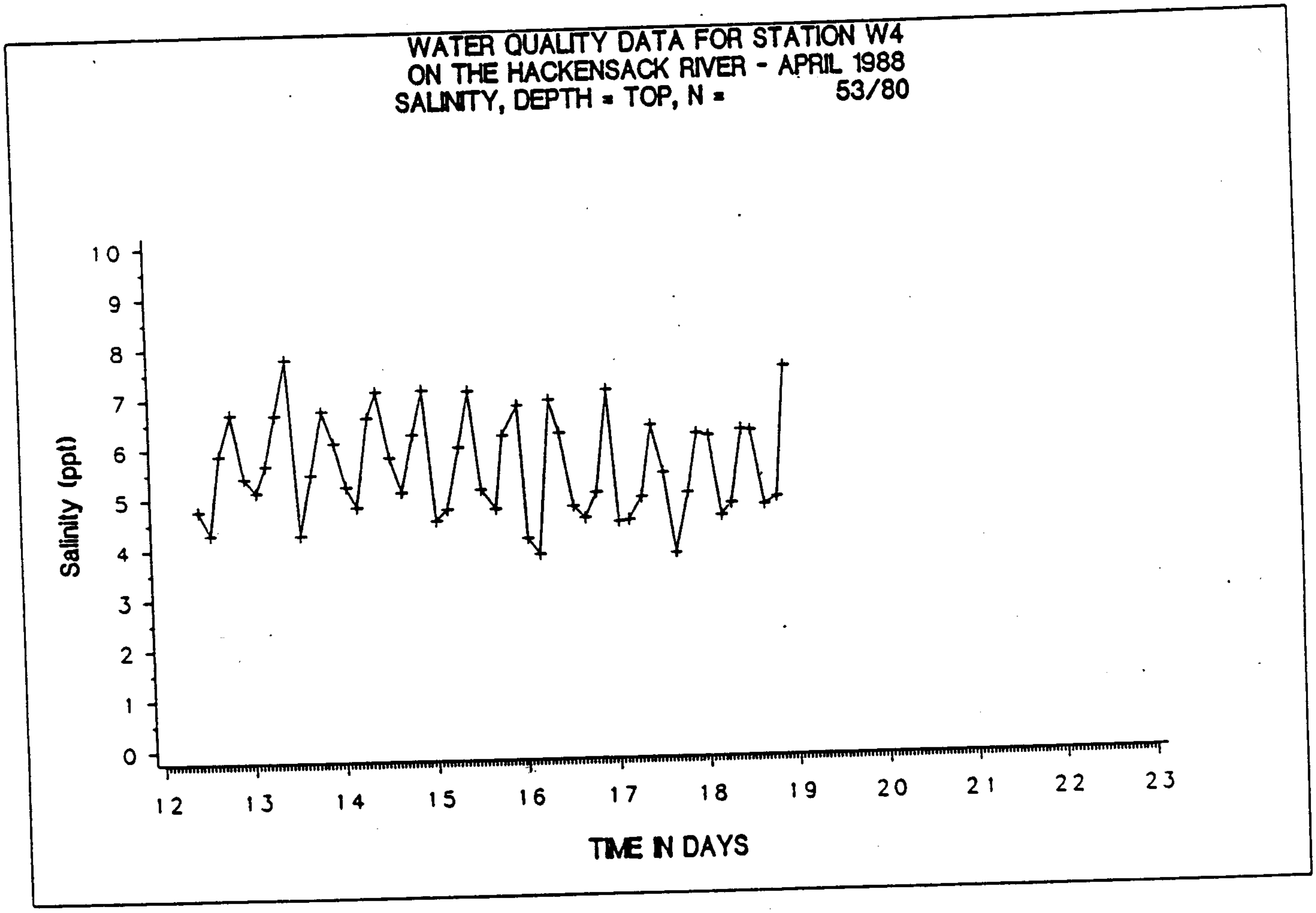
below the surface could be a significant factor controlling algal biomass as measured by chlorophyll **a** concentrations. Vigorous tidal mixing also has two other effects. First, with the fine-grained organic detritus held in suspension, much of the oxygen demand of the system is exerted in the water column, sustained by oxygen that is continuously mixed downward from the surface. Second, when the organic detritus does settle, it exerts only a modest sediment oxygen demand. The exception to this generalization can be found at Station W-6, well upstream from the treatment plant, where most of the organic material in the sediments is derived from recently living phytoplankton. In August the SOD at W-6 was 10 times the SOD near the plant, but the CBOD above the sediments was more nearly the same.

Figure 2



WATER QUALITY DATA FOR STATION W4
ON THE HACKENSACK RIVER - APRIL 1988
SALINITY, DEPTH = TOP, N = 53/80
Salinity (ppt)
0
1
2
3
4
5
6
7
8
9
10
12
13
14
15
16
17
18
19
20
21
22
23
TIME IN DAYS

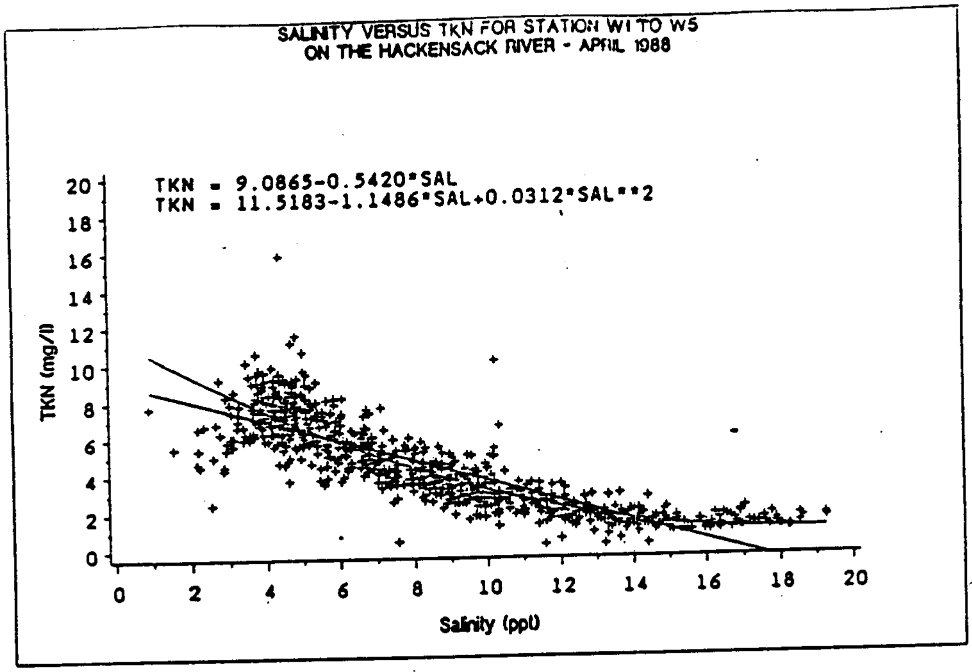
NUTRIENT PATTERNS

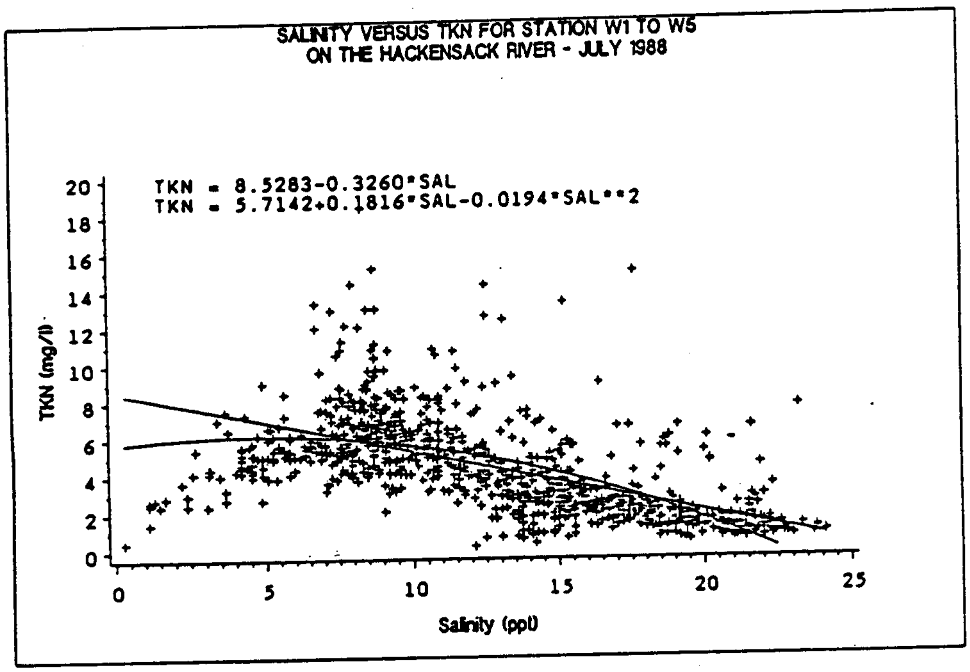
Background

The nutrient data for the main axis of the River were collected by General Testing Corporation and are presented here as graphs of the nutrient versus salinity. Salinity was chosen for the independent variable rather than station location or river mile for two reasons. First, the tidal excursion in the river is large and the salinity gradient is compressed with respect to the distance between stations, so there are be significant fluctuations at each station and overlap between adjacent stations. For example, the surface salinity at station W-4 ranges between about 4.5% and 7.5% over a tidal cycle (Figure 2). Second, this presentation facilitates comparing changes in nutrient concentration by the conservative process of mixing with the integrated sum of the non-conservative processes such as plankton uptake and settling. Each graph in Figures 3 through 6 is fitted with two equations - a linear best fit and a simple quadratic fit. In most cases, neither one is satisfactory, reaffirming the requirement for a mathematical model able to deal with multiple non-linear relationships.

Results

The concentrations of nutrients ammonium (NH3) and total Kjeldahl nitrogen (TKN) peak near the location of the BCUA outfall. The concentration of each parameter decreases in a non-linear fashion downstream from the outfall. Clearly,


SALINITY VERSUS TKN FOR STATION W1 TO W5
ON THE HACKENSACK RIVER - APRIL 1988
TKN = 9.0865-0.5420*SAL
TKN = 11.5183-1.1486*SAL+0.0312*SAL**2
TKN (mg/l)
Salinity (ppt)


SALINITY VERSUS TKN FOR STATION W1 TO W5
ON THE HACKENSACK RIVER - JULY 1988
TKN = 8.5283-0.3260*SAL
TKN = 5.7142+0.1816*SAL-0.0194*SAL**2
TKN (mg/l)
Salinity (ppt)

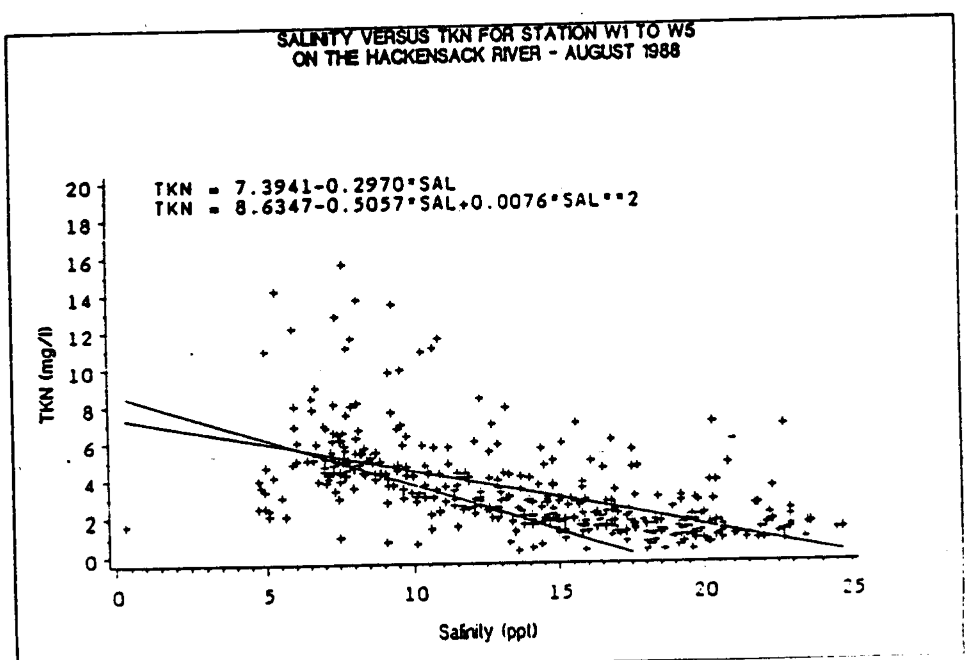
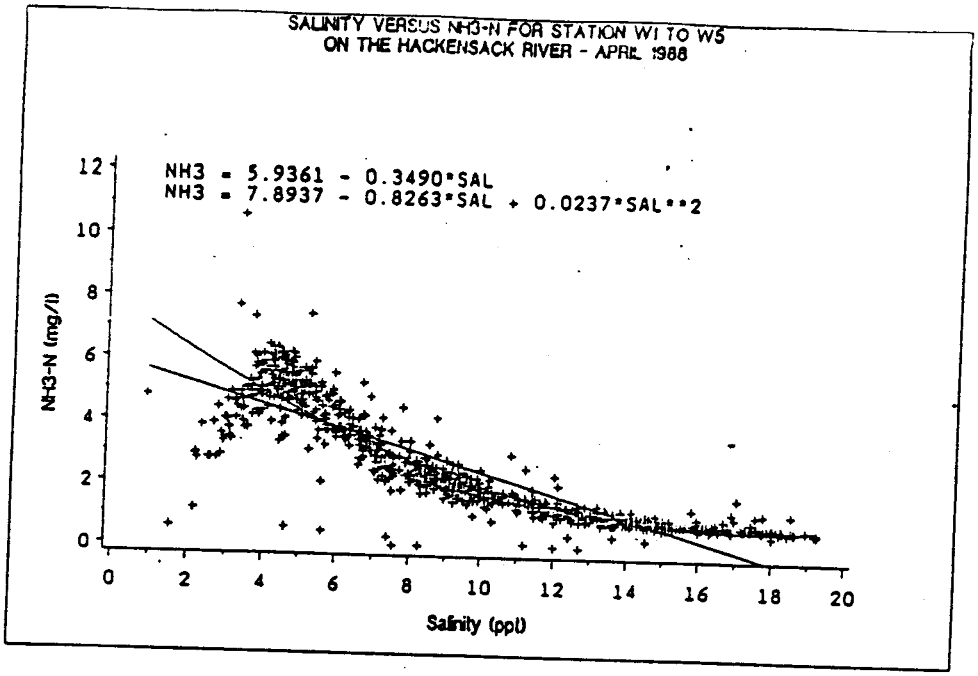
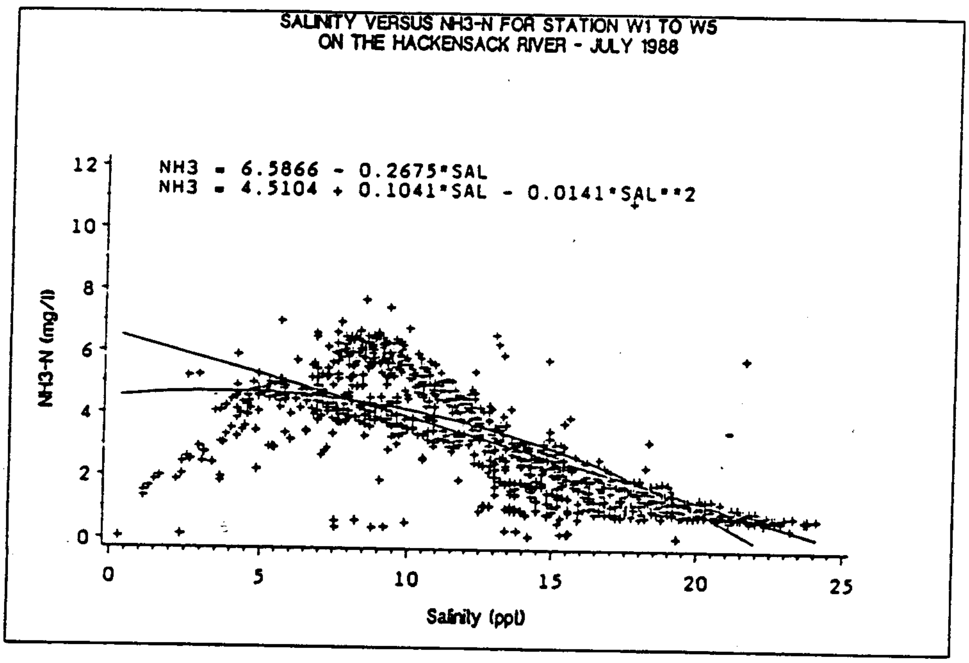

SALINITY VERSUS TKN FOR STATION W1 TO W5
ON THE HACKENSACK RIVER - AUGUST 1988
TKN = 7.3941-0.2970*SAL
TKN = 8.6347-0.5057*SAL+0.0076*SAL**2
TKN (mg/l)
Salinity (ppt)

Figure 4


SALINITY VERSUS NH3-N FOR STATION W1 TO W5
ON THE HACKENSACK RIVER - APRIL 1988
NH3 = 5.9361 - 0.3490*SAL
NH3 = 7.8937 - 0.8263*SAL + 0.0237*SAL**2
NH3-N (mg/l)
Salinity (ppt)


SALINITY VERSUS NH3-N FOR STATION W1 TO W5
ON THE HACKENSACK RIVER - JULY 1988
NH3 = 6.5866 - 0.2675*SAL
NH3 = 4.5104 + 0.1041*SAL - 0.0141*SAL**2
NH3-N (mg/l)
Salinity (ppt)

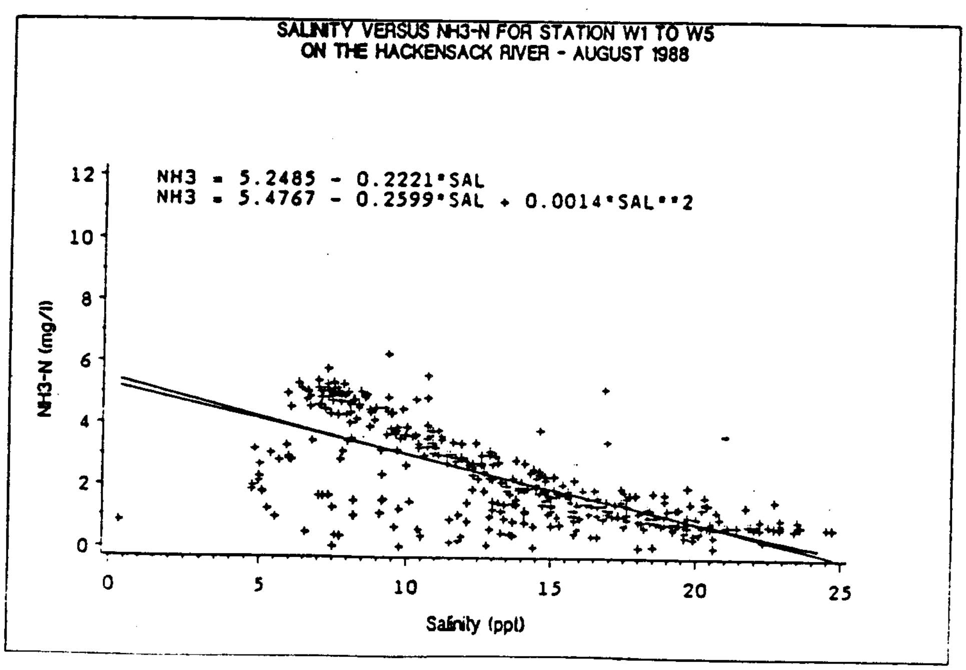
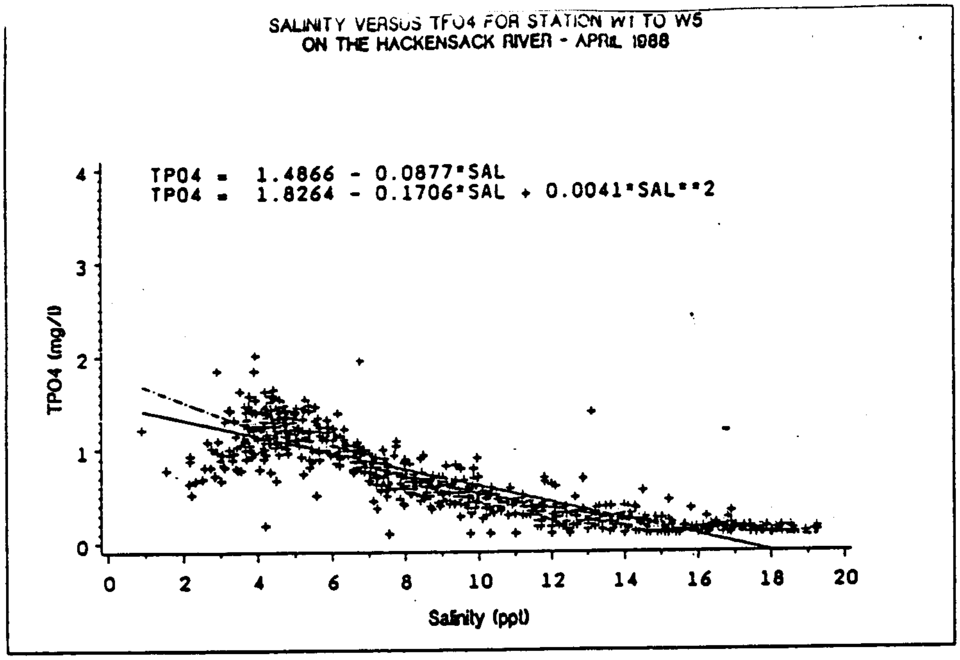
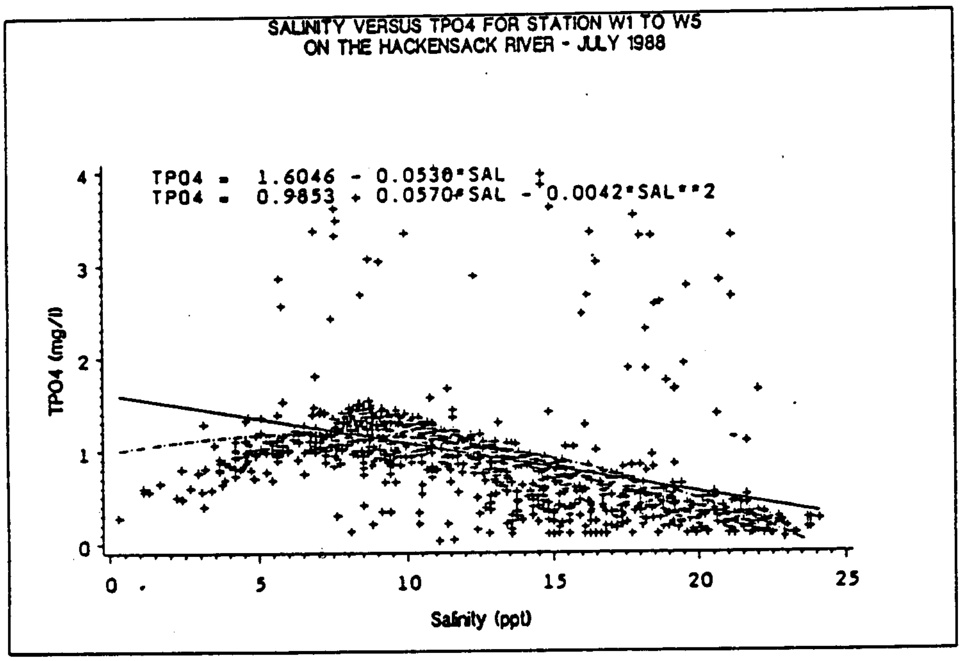

SALINITY VERSUS NH3-N FOR STATION W1 TO W5
ON THE HACKENSACK RIVER - AUGUST 1988
NH3 = 5.2485 - 0.2221*SAL
NH3 = 5.4767 - 0.2599*SAL + 0.0014*SAL**2
NH3-N (mg/l)
Salinity (ppt)

Figure 3


SALINITY VERSUS TPO4 FOR STATION W1 TO W5
ON THE HACKENSACK RIVER - APRIL 1988
TPO4 = 1.4866 - 0.0877*SAL
TPO4 = 1.8264 - 0.1706*SAL + 0.0041*SAL**2
TPO4 (mg/l)
Salinity (ppt)


SALINITY VERSUS TPO4 FOR STATION W1 TO W5
ON THE HACKENSACK RIVER - JULY 1988
TPO4 = 1.6046 - 0.0538*SAL
TPO4 = 0.9853 + 0.0570*SAL - 0.0042*SAL**2
TPO4 (mg/l)
Salinity (ppt)

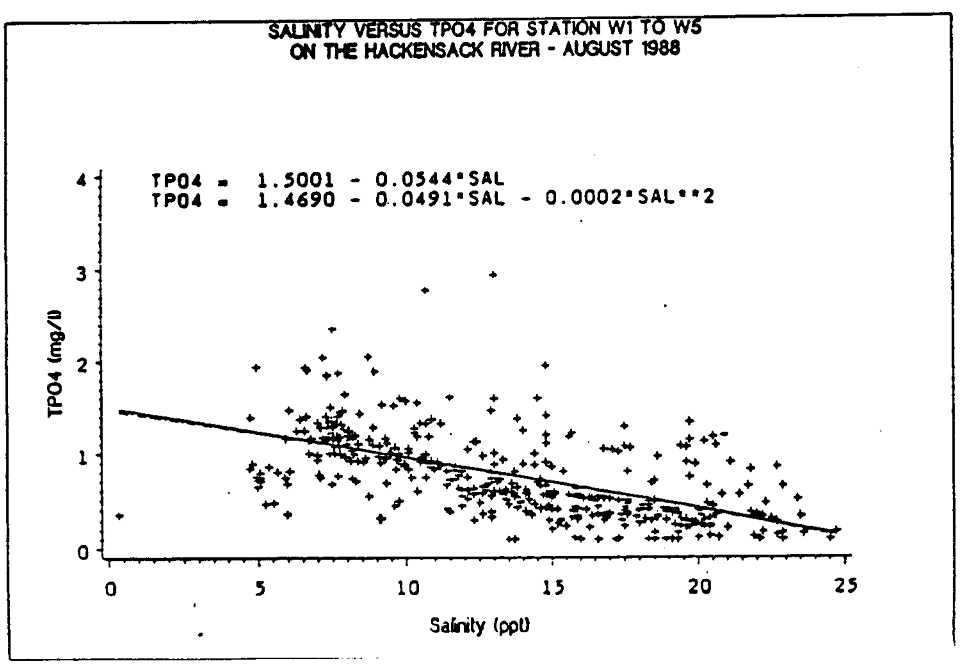

SALINITY VERSUS TPO4 FOR STATION W1 TO W5
ON THE HACKENSACK RIVER - AUGUST 1988
TPO4 = 1.5001 - 0.0544*SAL
TPO4 = 1.4690 - 0.0491*SAL - 0.0002*SAL**2
TPO4 (mg/l)
Salinity (ppt)

Figure 5

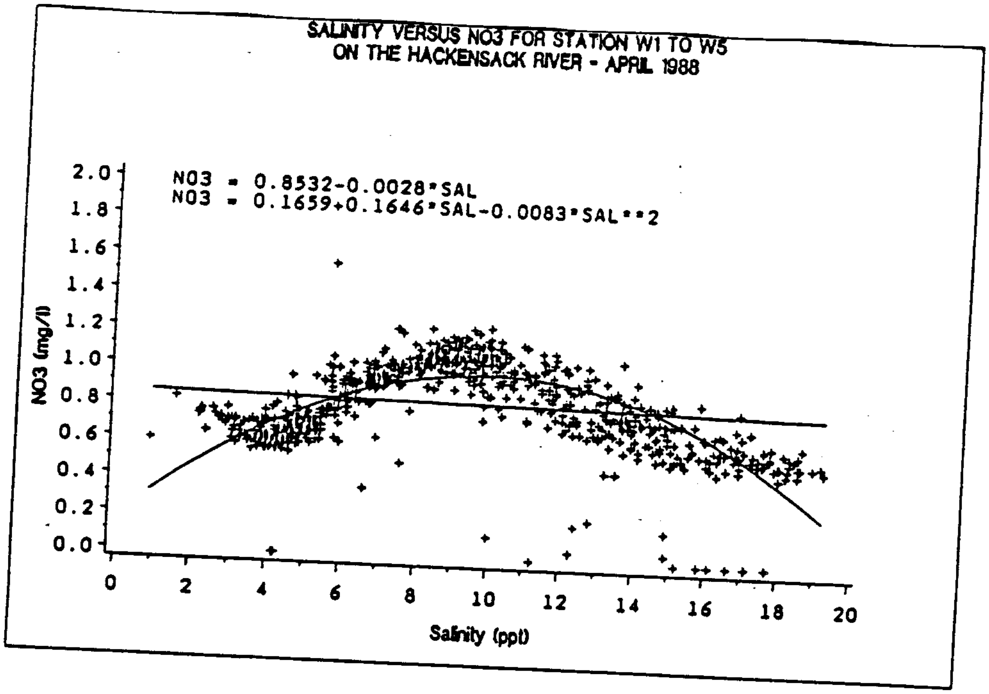
there is dilution but because the decrease is not linear, other processes must be occurring. In April, the ammonium concentration decreases from about 5 mgl^{-1} at salinity 4.5‰ at the outfall to about 2 mgl^{-1} at salinity 9‰. Dilution alone would reduce the ammonium concentration to 2.5 mgl^{-1}, so about 0.5 mgl^{-1} ammonium is lost by other processes. Comparing the soluble ammonium with TKN, it appears that the particulate nitrogen concentration is about 2 to 4 mgl^{-1} near the outfall and about 0.5 to 1 mgl^{-1} near the mouth of the estuary. The pattern is similar for July and August, except the salinity is higher in the upstream reaches.

Total phosphorus (TPO4) reaches peak concentrations of about 1.5 mgl^{-1} at the outfall and decrease to about 0.2 mgl^{-1} near the mouth. The pattern of the TPO4 data plotted against salinity is non-linear, so there is a small amount of particulate material settling from the water column. However, the average total phosphorus decrease from the peak of 1.3 mgl^{-1} at salinity 4.5‰ to 0.6 mgl^{-1} at salinity 9‰ suggests that dilution is responsible for a significant portion of the concentration reduction in April.

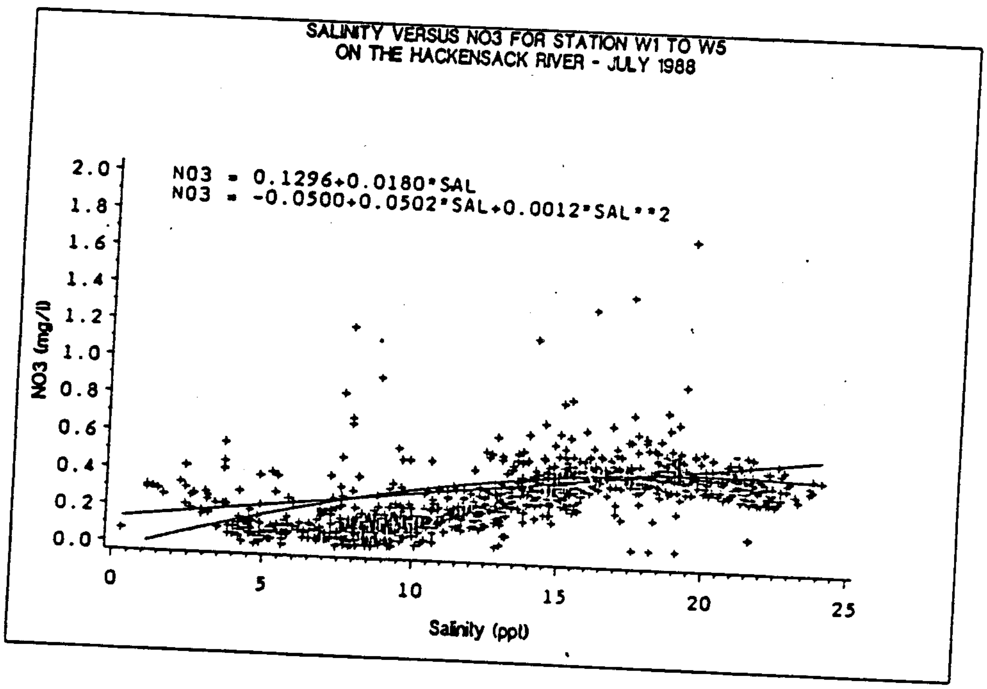
Nitrate (NO3) shows a distribution pattern quite different from the other nutrients. Nitrate concentrations seem to be slightly higher upstream of the outfall and decrease to about 0.6 mgl^{-1} at the outfall, then increase again downstream to a peak of about 1.1 mgl^{-1} at salinity 9 to 10‰, followed by a decrease back to about 0.6 mgl^{-1}. This pattern suggests nitrification of ammonium to nitrate in the water column followed by dilution downstream of the peak. The nitrate increase closely matches the quantity of ammonium decrease, mentioned above, which is not accounted for by dilution. Because both the nitrate concentration and the salinity change by about 40% downstream from the peak, the dominant process again appears to be dilution. There appear to be minimal biological processes operating on nitrate in this region during April. Given the high ammonium concentrations, one would not predict significant nitrate uptake by the phytoplankton. A different explanation is required for the nitrate minimum which appears in the water column data near the outfall in April and July. This is discussed later in the denitrification section.

A significant temperature range exists in the vicinity of the outfall caused by the heated water discharged from the PSE&G power generating plant across the River. Figure 7 shows the temperature patterns at W-4, downstream from both discharges, which have a typical range of 11°C to 17°C with excursions to 10°C and 18°C. Large short-term fluctuations such as these are bound to influence benthic biological processes, and could also affect water column processes, in ways not predicted from the generalization that biological rates double when temperature is increased by 10°C.

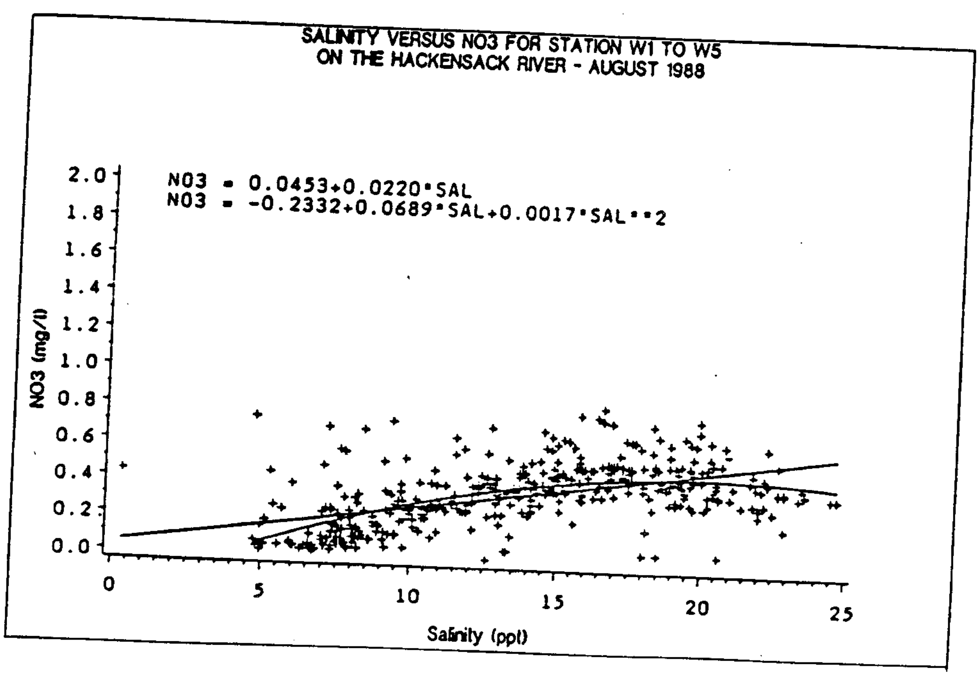
Potential for nutrient limitation is an important issue. Because soluble ammonium, nitrate, and phosphate were always measurable in the water downstream from the outfall, neither nitrogen nor phosphorus were limiting phytoplankton abundance.



SALINITY VERSUS NO3 FOR STATION W1 TO W5
ON THE HACKENSACK RIVER - APRIL 1988
NO3 = 0.8532-0.0028*SAL
NO3 = 0.1659+0.1646*SAL-0.0083*SAL**2
NO3 (mg/l)
Salinity (ppt)



SALINITY VERSUS NO3 FOR STATION W1 TO W5
ON THE HACKENSACK RIVER - JULY 1988
NO3 = 0.1296+0.0180*SAL
NO3 = -0.0500+0.0502*SAL+0.0012*SAL**2
NO3 (mg/l)
Salinity (ppt)



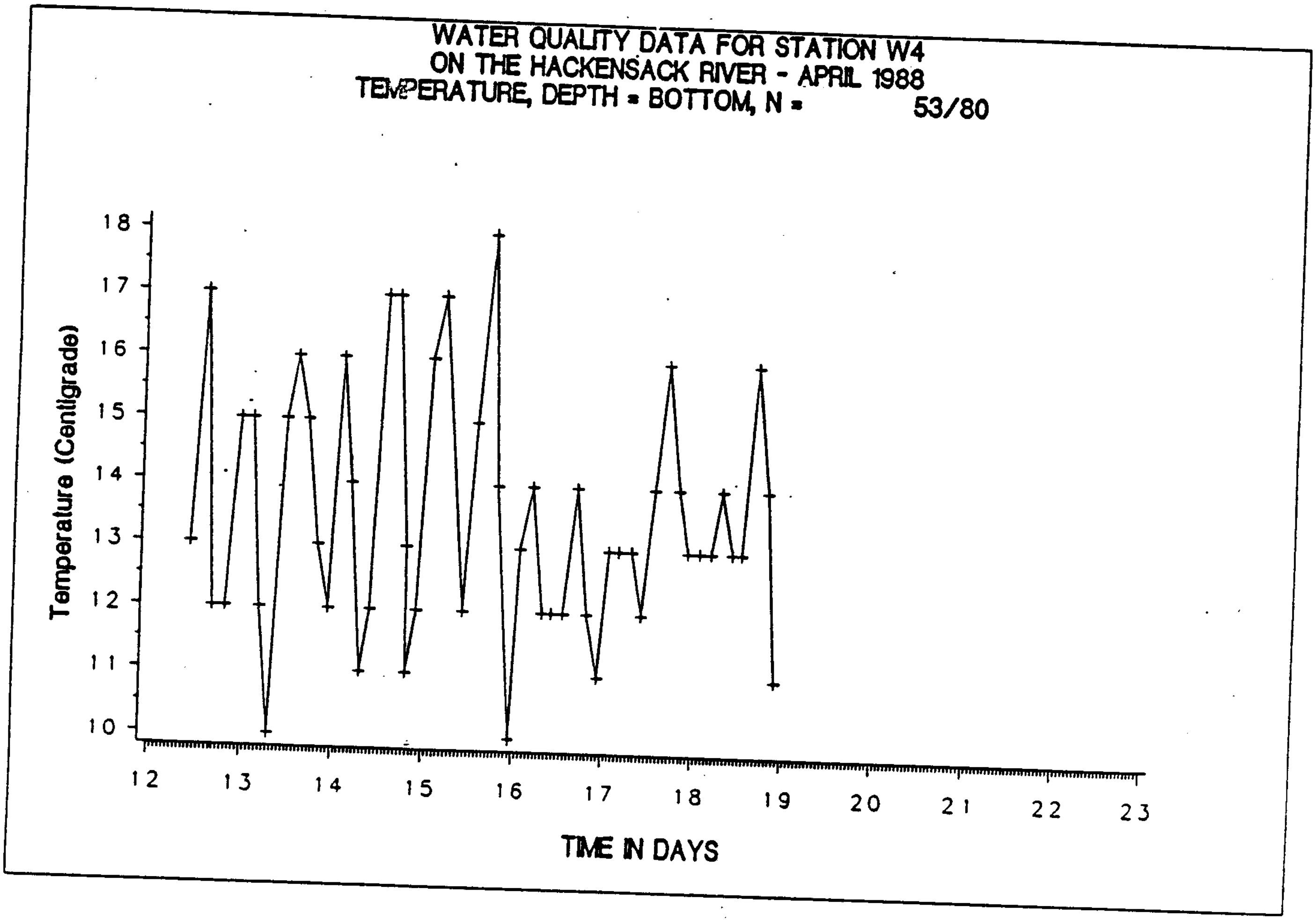
SALINITY VERSUS NO3 FOR STATION W1 TO W5
ON THE HACKENSACK RIVER - AUGUST 1988
NO3 = 0.0453+0.0220*SAL
NO3 = -0.2332+0.0689*SAL+0.0017*SAL**2
NO3 (mg/l)
Salinity (ppt)

Figure 6

Nevertheless, the potential for one of them to become limiting can be assessed by comparing ratios of total nitrogen, expressed as TKN in this case, and total phosphorus to the classical values for well nourished phytoplankton. Table 1 shows the ratios of the mean TKN to total phosphorus concentrations for each station in April. Ratios of the parameters would indicate the potential for nitrogen availability to limit phytoplankton biomass if the N:P is less than about 7.2 by milligrams or 16.0 by atoms, and the potential for phosphorus limitation when ratios are greater than these values.

Station W-1 at the River mouth has N:P very close to the expected normal value, which does not suggest the potential for either nutrient to be limiting. Without including nitrate in the ratio, the other River axis stations, suggest a slight potential for nitrogen to limit phytoplankton biomass. If nitrate had been included, the nutrient concentrations would have been closer to the expected ratio for well nourished phytoplankton. However, because soluble nutrient concentrations never approach zero, the potential for either nitrogen or phosphorus to become limiting in the River was not realized in April. In contrast, the ratio at station W-9 in Overpeck Creek shows sufficiency of both nutrients with respect to typical phytoplankton requirements, and W-10 in Berry's Creek the ratio shows a strong potential for phosphorus limitation.

Figure 7


WATER QUALITY DATA FOR STATION W4
ON THE HACKENSACK RIVER - APRIL 1988
TEMPERATURE, DEPTH = BOTTOM, N = 53/80
Temperature (Centigrade)
18
17
16
15
14
13
12
11
10
12
13
14
15
16
17
18
19
20
21
22
23
TIME IN DAYS

Discussion

The picture that emerges is one of a nutrient enriched system containing significant concentrations of particulate organic nitrogen. The biological and nutrient dynamics of the Hackensack are reflected in the distribution patterns of nutrient concentration. Although the distribution patterns for the parameters discussed above were non-linear with respect to salinity, dilution of nutrient from the BCUA outfall by sea water is nevertheless a major process reducing concentrations. About 10% of the soluble ammonium in the outfall area appears to be nitrified in the water column downstream. The TKN and TPO4 plots suggest some particle settling. The strong tidal action must inhibit settling of most of the fine grained material. The visual opacity of the water column and the high soluble nutrient concentrations suggest light availability could limit phytoplankton growth. On the other hand, a highly enriched and strongly mixed system could produce much higher chlorophyll concentrations than were observed downstream from the treatment plant. It is not yet possible to rule out the notion that organic suspended solids in the River inhibit phytoplankton growth by some biological or biochemical interaction which was not measured.

Table 1

HACKENSACK RIVER - APRIL 1988 SAMPLING

RATIOS OF TKN : TP

STATION	TKN:TP by mg	TKN:TP by atoms
W-1	6.9	15.2
W-2	6.5	14.4
W-3	5.9	13.1
W-4	5.9	13.0
W-5	6.2	13.8
W-6	5.6	12.5
W-7	6.0	13.3
W-8	6.3	13.8
W-9	7.5	16.5
W-10	10.3	22.8
Redfield ratio =	7.2	16.0

(1) $SOD = ax + 0.2\ |v|$

where x = SOD (g O_2 $m^{-2}d^{-1}$) in cores at $v = 0$.

a = factor between 1.5 to 4.0 to adjust for core SOD underestimating SOD in open water.

v = current velocity up to 20 cm s^{-1}. At velocities greater than 20, let $v = 20$.

Different values for the open water factor (**a**) will have to be tried to give the best calibration of the model to the Hackensack data.

Method

Sediment oxygen demand (SOD) was measured in the water overlying intact cores incubated in the laboratory. The overlying water was not stirred continuously during incubation, but it was stirred during the procedure for measuring dissolved oxygen by probe. The procedures for obtaining the cores and transporting them to the shore laboratory were tested on June 27, 1988, at Station W-5 near Overpeck Creek. A plastic core liner was fit inside the metal corer tube with a core catcher and a cutter inserted in the lower end. After several attempts yielding poor cores, two of which contained wood plugs, a segment of telephone pole was recovered. Due to the high proportion of the bottom covered by wooden debris and the unconsolidated nature of the sediment, it was necessary to modify the original method and change sampling locations. Cores were subsequently taken by carefully pushing the core liner, without a core catcher, into

SEDIMENT OXYGEN DEMAND

Background

The rationale for carefully obtaining, transporting, and sampling intact cores relates to using the data for modeling. Two alternatives for incubating the cores were considered. Throughout the course of incubation the overlying water could have been stirred or not stirred. Stirring has the advantage of maintaining a more uniform concentration of oxygen throughout the overlying water. Its disadvantages are potential to mix the surface sediment into the water, difficulty of estimating and reproducing water flow rate over the sediment surface of all cores, uncertainty in extrapolating from stirred cores to the River, and added complexity of the incubation. Not stirring has the advantage of experimental simplicity. A consequence of not stirring is that an oxygen gradient may develop in the overlying water which could lead to suppression of SOD at low oxygen concentrations. Therefore, not stirring during incubation should approximate the minimum value for SOD that would occur in the River. The minimum value could then be corrected upward empirically. Boynton et al (1981) described a relationship between SOD and water flow over the sediments contained by **in situ** chambers. Their findings can be generalized to the following equation which should allow for varying SOD in the Hackensack River over the tidal cycle based upon the absolute value of the tidal current velocity:

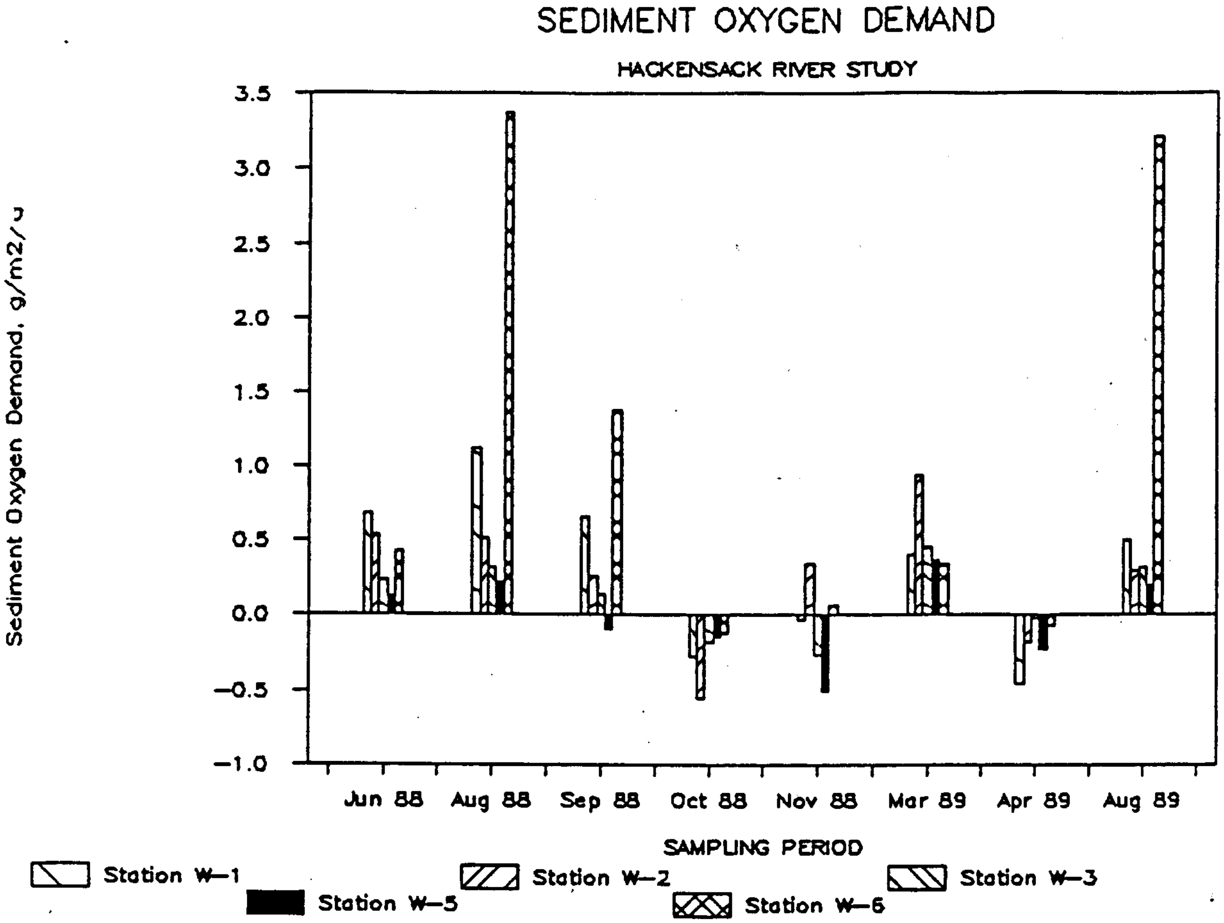


Figure 8

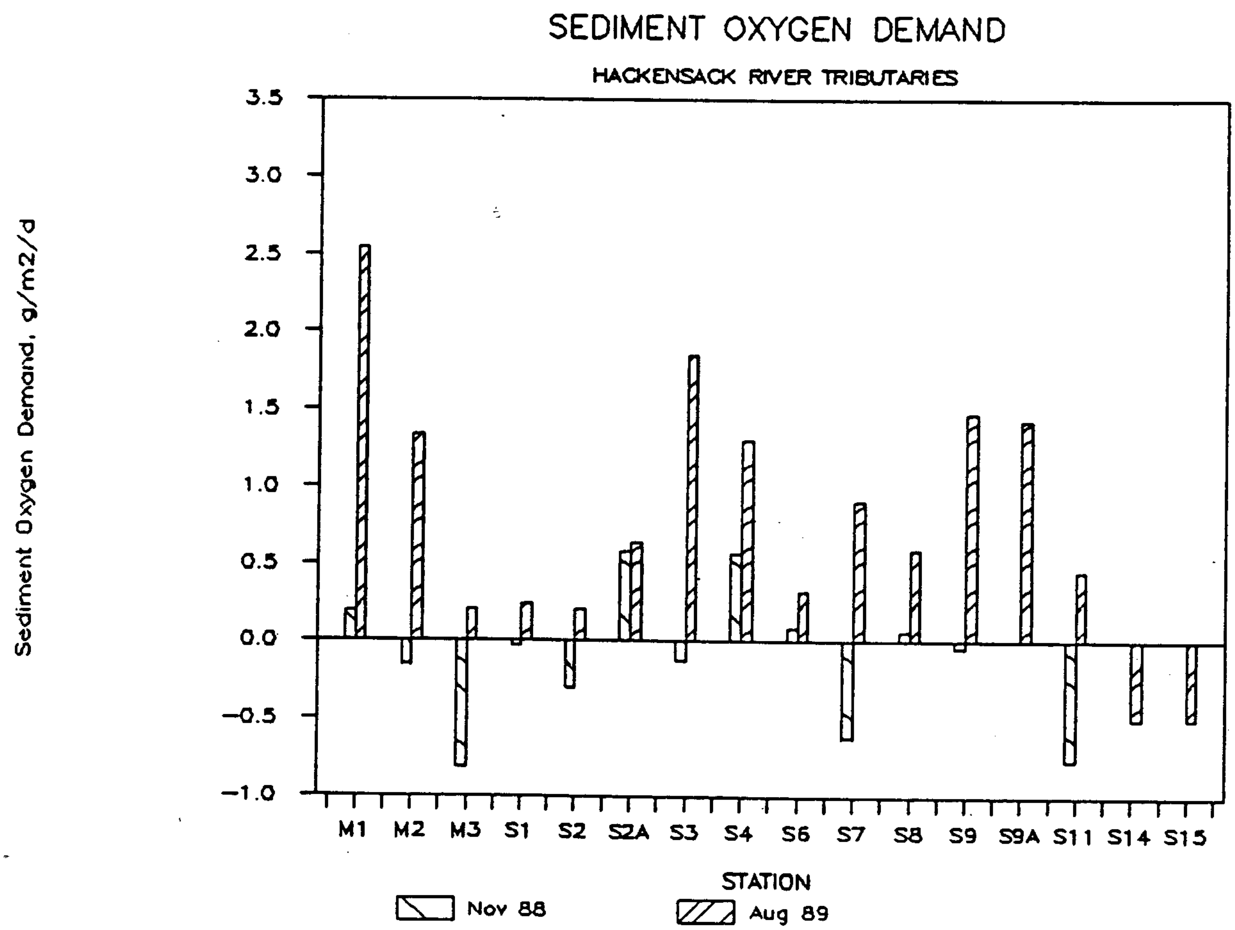


Figure 9

the sediment by hand. The plastic liner was then cut to a convenient length to fit the insulated carrying case. Using this method suggested by the field crew, satisfactory cores were obtained and transported to the shore laboratory within two hours.

The sampling locations were moved from the channel into shallow water in order to better control the coring process. This should be satisfactory for our purposes because more will be learned from cores taken carefully in the shallows than from poor quality cores from deeper water. It was suggested to General Testing that the coring could be moved inshore at the other stations, but the intertidal zone must be avoided. Sediments were sampled from the River axis in June, August, September, October, and November 1988, and March, April and August 1989. In addition, three marsh stations and 10 tributary stations were sampled as part of an intensive tributary sampling program in November 1988.

Results

The rates of sediment oxygen demand are summarized in Figure 8 for the main axis of the River and in Figure 9 for the tributaries. During June, August, and September, SOD values in the region from W-5 to W-3 were less than those at station W-6, upstream from the treatment plant, and at stations W-1 and W-2, downstream near the River mouth. In both August and September, the water at station W-6 contained oxygen concentrations above saturation, probably due to photosynthetic activity of the phytoplankton. Nevertheless,

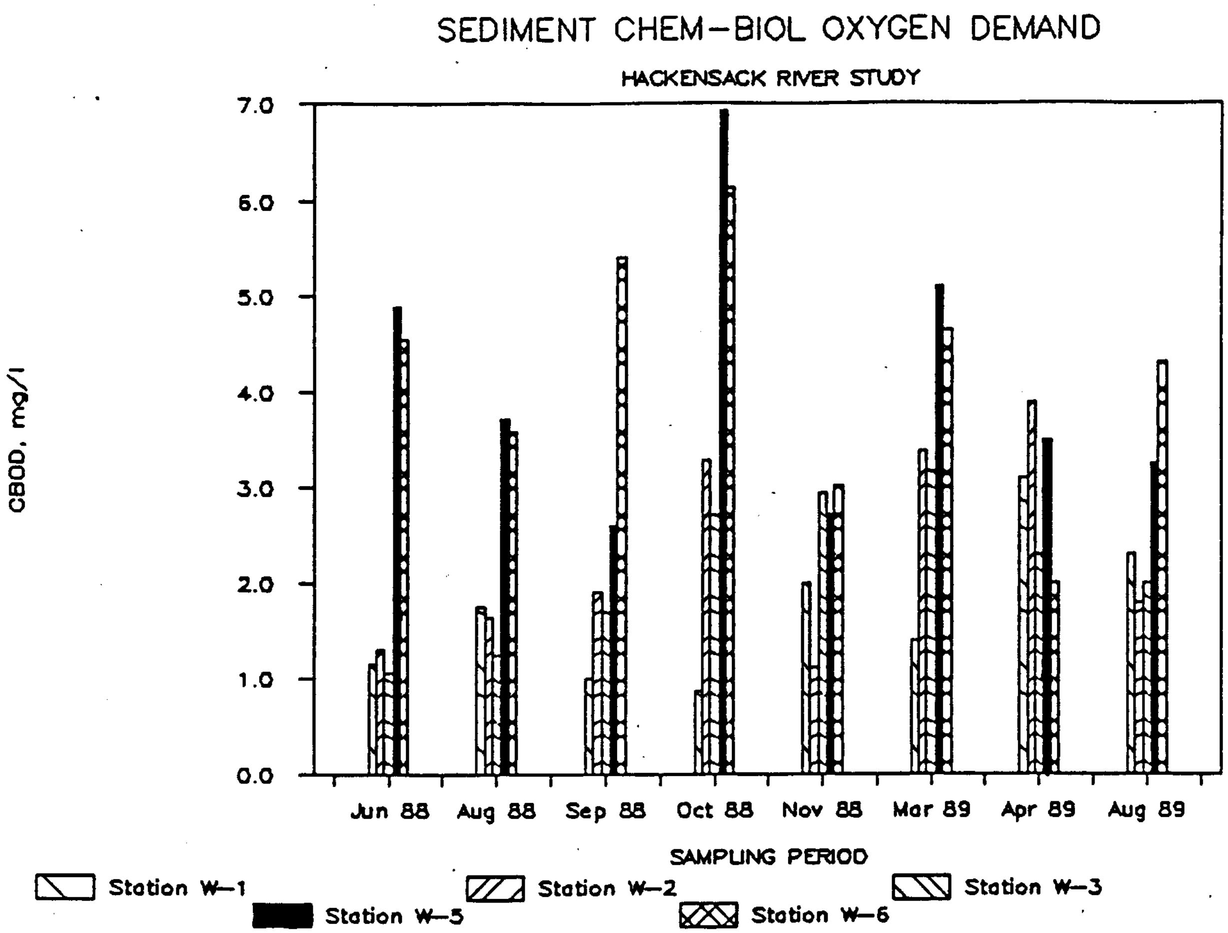


Figure 10

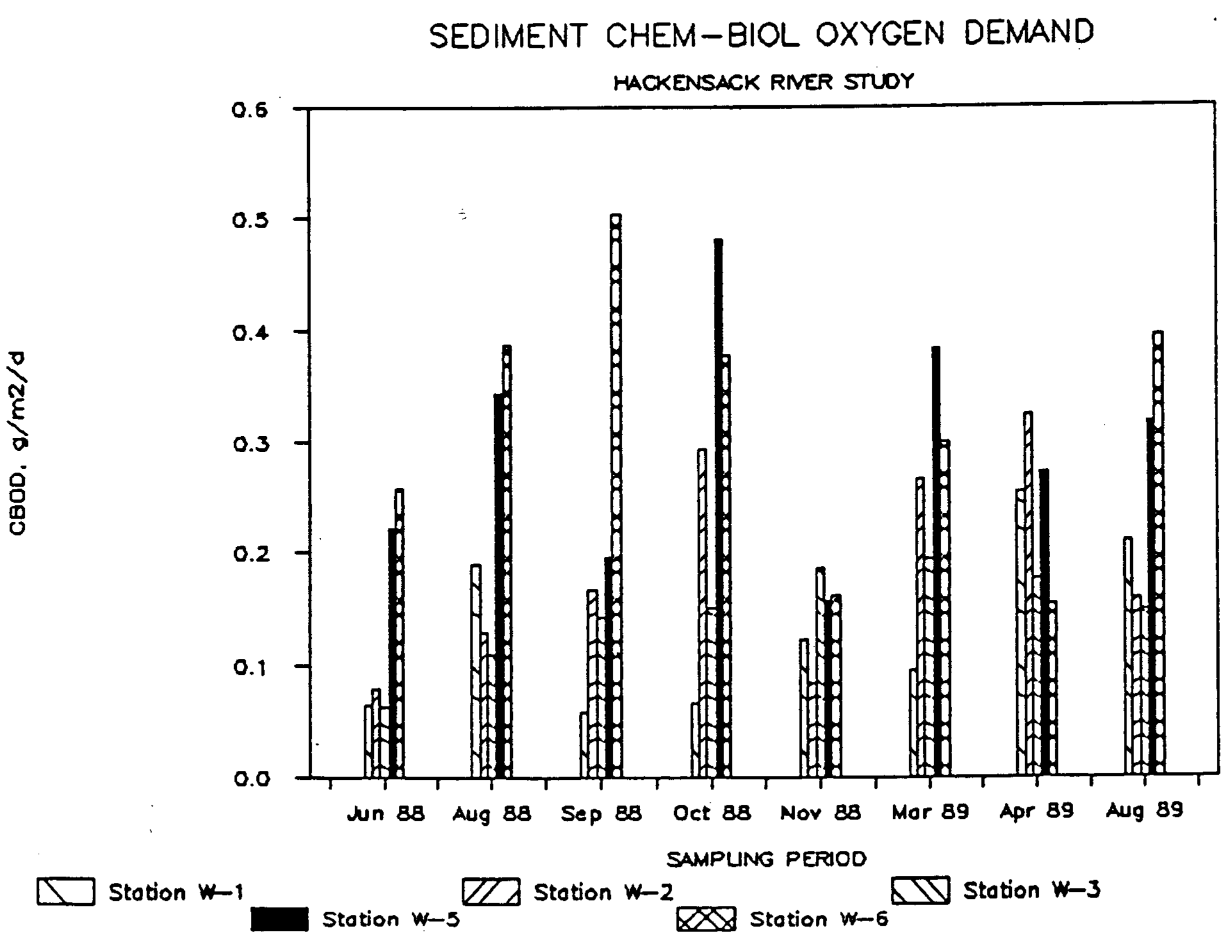
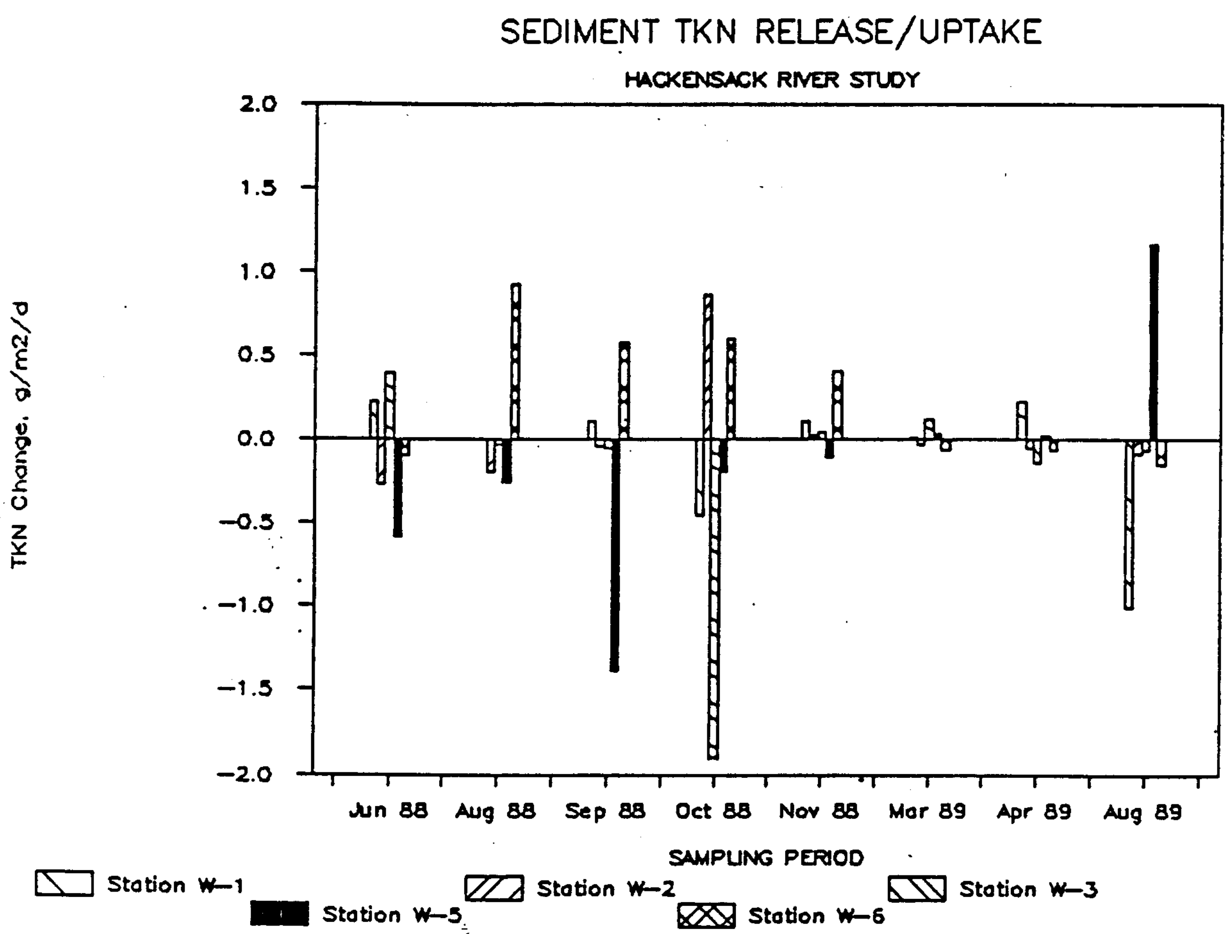


Figure 11

the loss of oxygen by equilibration with the atmosphere was small compared to consumption by CBOD and SOD during core incubations. The lower values for SOD in the middle reaches of the River are consistent with the virtually undetectable loss of TKN and TPO4 from the water column immediately downstream from the treatment plant during April; tidal action appears to inhibit settling. In September, a negative SOD value was calculated for stations W-5. This pattern was widespread in October, November and April. The high CBOD5 measurements in the water overlying the cores during these incubations caused this result. Figure 10 presents CBOD5 as a concentration in mgl^{-1} and Figure 11 expresses the CBOD5 in $gm^{-2}d^{-1}$. October CBOD5's ranged up to 6.90 mgl^{-1} and November values were up to 3.20 mgl^{-1}. The CBOD5's were both high enough and variable enough to cause the SOD values to appear very small, or negative, when the total oxygen change in the core samples was corrected for CBOD5 in order to calculate the sediment contribution to oxygen demand.

Figure 12



SEDIMENT TKN RELEASE/UPTAKE
HACKENSACK RIVER STUDY
TKN Change, g/m2/d
2.0
1.5
1.0
0.5
0.0
−0.5
−1.0
−1.5
−2.0
Jun 88
Aug 88
Sep 88
Oct 88
Nov 88
Mar 89
Apr 89
Aug 89
SAMPLING PERIOD
Station W—1
Station W—2
Station W—3
Station W—5
Station W—6

SEDIMENT - WATER COLUMN EXCHANGES

Background

The exchanges of constituents between the sediments and the water column often have significant effects on nutrient concentrations in estuaries. Release of soluble nutrient ions or organic material contributes to eutrophication. Settling of particulate nutrients and organic material removes these constituents from the water column, but usually contributes to SOD. Because the Hackensack River model calculates nutrient concentrations which result from both input to and losses from the water, it is necessary to estimate the sediments role as a source or sink.

Method

The exchanges of nutrients between the sediments and overlying water were estimated from the changes of each constituent's concentration during the core incubation experiments. Water was sampled from cores at the beginning and end of incubation, filtered for soluble nutrients or not filtered for total nutrients, and frozen until analyses were performed. A sample of near-bottom water, collected when the sediment core was taken, was similarly analyzed.

Results

The direction of constituent fluxes varied from station to station and from one sampling period to the next. The TKN (Figure 12) increased in 17 incubations and decreased in 22.

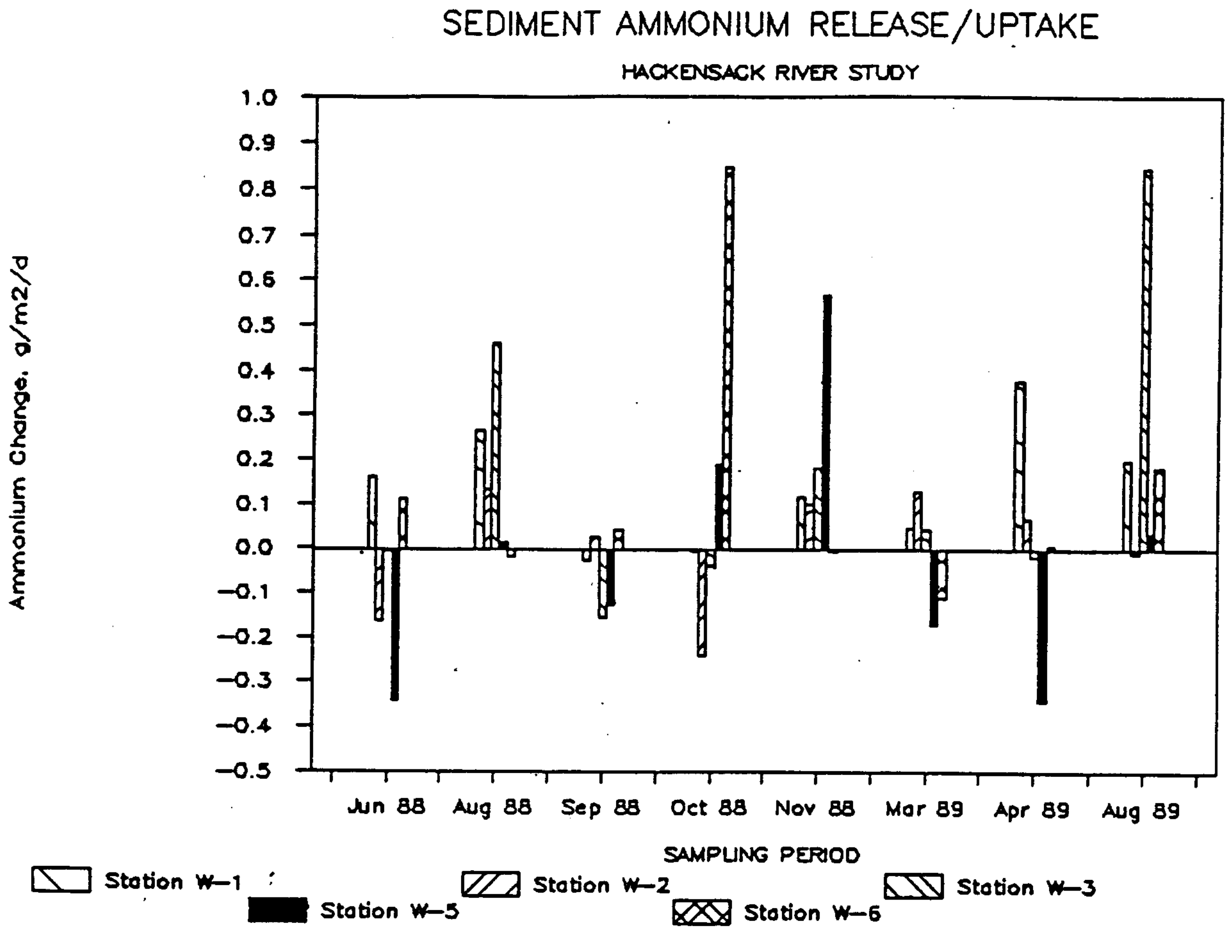


Figure 13

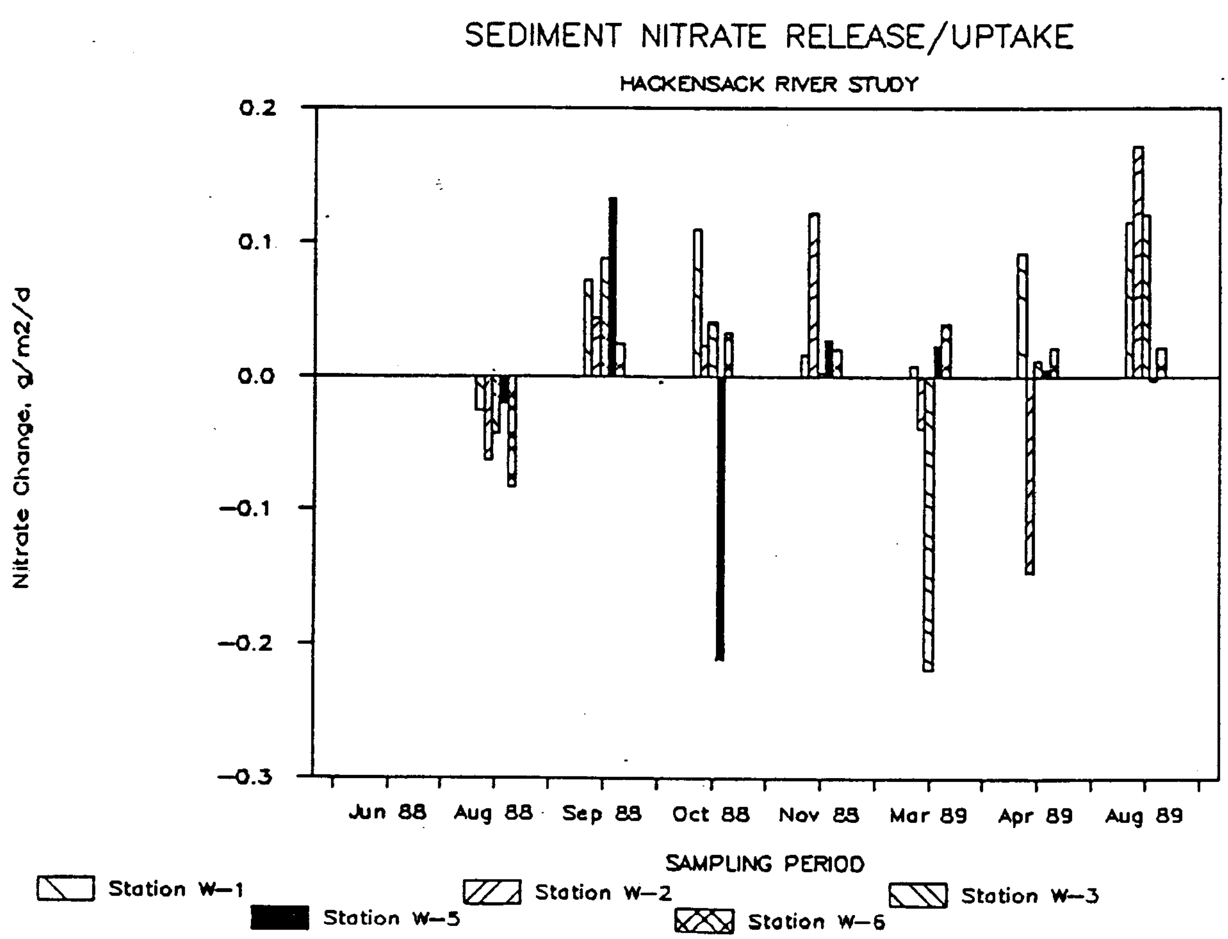


Figure 14

The range of values was approximately -1.9 to-1.2 g N $m^{-2}d^{-1}$, with the smallest changes occurring during the colder months. Ammonium (Figure 13) increased in 23 incubations and decreased in 12. The range of ammonium flux values was -0.3 to 0.8 g N $m^{-2}d^{-1}$. Nitrate (Figure 14) was released from cores in 24 incubations and was taken up in 9. Dissolved orthophosphate (Figure 15) was released in 17 incubations and taken up in 16. In every case examined, cores taken at station W-5 exhibited orthophosphorus flux into the sediments. Flux rates ranged from -0.10 to 0.08 g P $m^{-2}d^{-1}$. Total phosphorus (Figure 16) was released in 21 incubations and was taken up, or settled out of the water, in 16. Rates were generally between -0.05 and 1.3 g P $m^{-2}d^{-1}$, with the exception of one core from station W-5 in August 1989 which lost about 0.3 g P $m^{-2}d^{-1}$ from the water.

Due to the wide variability of the flux data from the Hackensack River cores, another factor has been computed which facilitates comparison of sediment-water column in the Hackensack with those in the Chesapeake Bay. The parameter SOD/DO is shown in Table 2 for each River station. The use of this parameter is discussed below. The nutrient flux data for the two tributary surveys is summarized in Figures 17 and 18. In general, the patterns and their variability in the tributaries are as great as those for the River.

Discussion

The sediment sampling program, conducted by scientists from the University of Maryland, in the Chesapeake Bay has

Table 2. Sediment Oxygen Demand divided by Dissolved Oxygen concentration in cores from the Hackensack River.

Station	Date	SOD/DO, m/d
W-1	Jun 88	.18
	Aug 88	.32
	Sep 88	.23
	Oct 88	.27
	Nov 88	.23
	Mar 89	.25
	Apr 89	.30
	Aug 89	.30
W-2	Jun 88	.20
	Aug 88	.28
	Sep 88	.30
	Oct 88	.33
	Nov 88	.30
	Mar 89	.29
	Apr 89	.30
	Aug 89	.29
W-3	Jun 88	.20
	Aug 88	.29
	Sep 88	.29
	Oct 88	.27
	Nov 88	.26
	Mar 89	.23
	Apr 89	.29
	Aug 89	.25
W-5	Jun 88	.15
	Aug 88	.29
	Sep 88	.28
	Oct 88	.25
	Nov 88	.25
	Mar 89	.28
	Apr 89	.28
	Aug 89	.32
W-6	Jun 88	.19
	Aug 88	.34
	Sep 88	.32
	Oct 88	.29
	Nov 88	.22
	Mar 89	.24
	Apr 89	.27
	Aug 89	.30

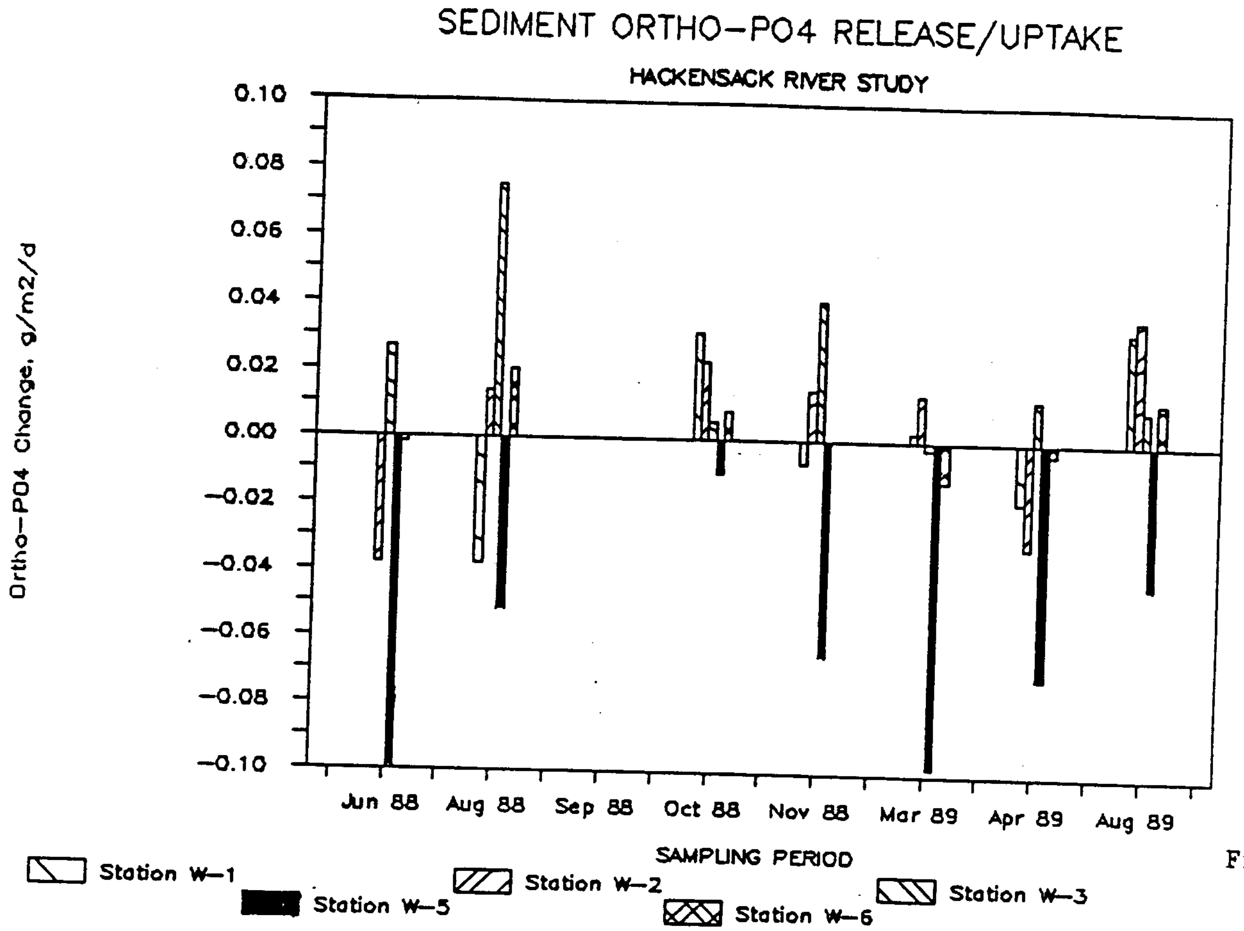


Figure 15

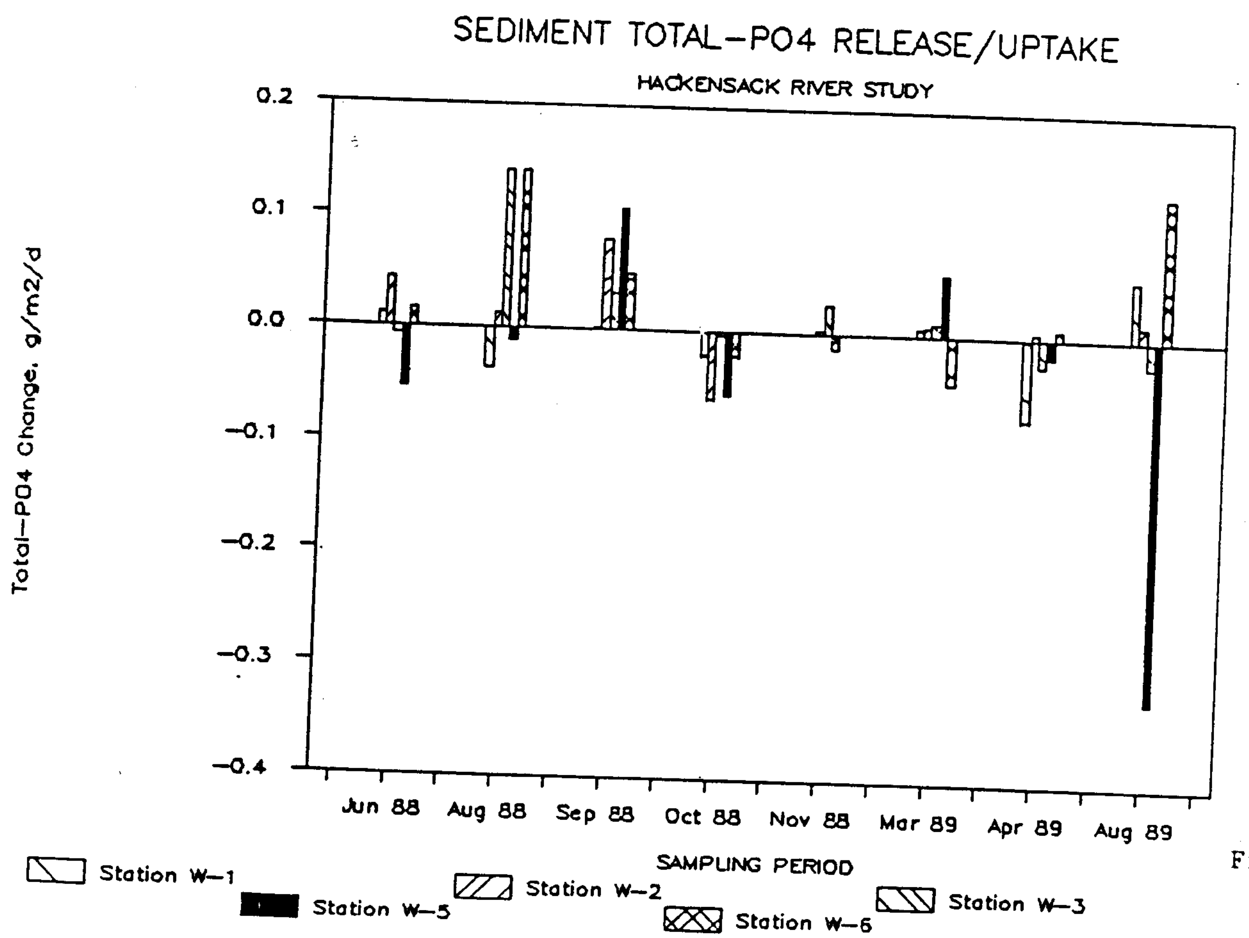


Figure 16

yielded a large set of reliably collected data which has been applied to the task of producing a sediment model. The sediment model is built upon both empirical and theoretical bases (DiToro, 1989). An interesting and useful finding is that the mass transfer of nutrients out of the sediment can be related to the oxygen mass transfer coefficient which is defined as the SOD divided by the corresponding overlying water DO concentration. There is significant variability of these parameters, even in the larger Chesapeake data set, as shown in Figure 19. The SOD/DO for the Hackensack cores ranged from 0.165 to 0.333 (the units convert to md^{-1}). Referring to the Chesapeake data in Figure 19, the ammonium fluxes out of the sediment at those values of SOD/DO range from 0.010 to 0.110 $g\ N\ m^{-2}d^{-1}$. As indicated above, the range of ammonium fluxes in the Hackensack cores was much greater than this. Nevertheless, the average of the 21 positive (out of the sediments) ammonium flux measurements for the Hackensack is 0.217 $g\ N\ m^{-2}d^{-1}$, and the average of 16 negative (into the sediments) measurements is -0.111 $g\ N\ m^{-2}d^{-1}$.

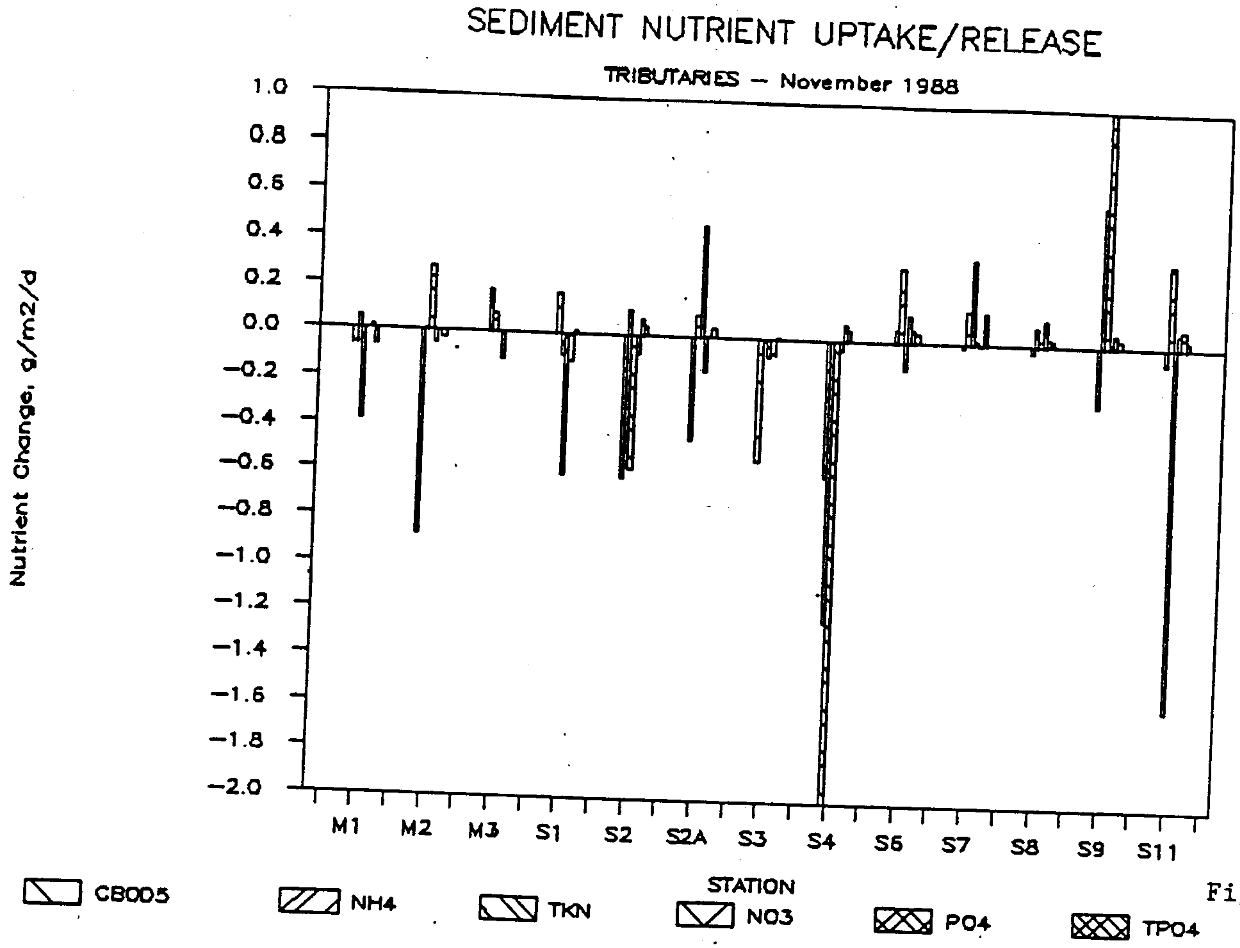


Figure 1?

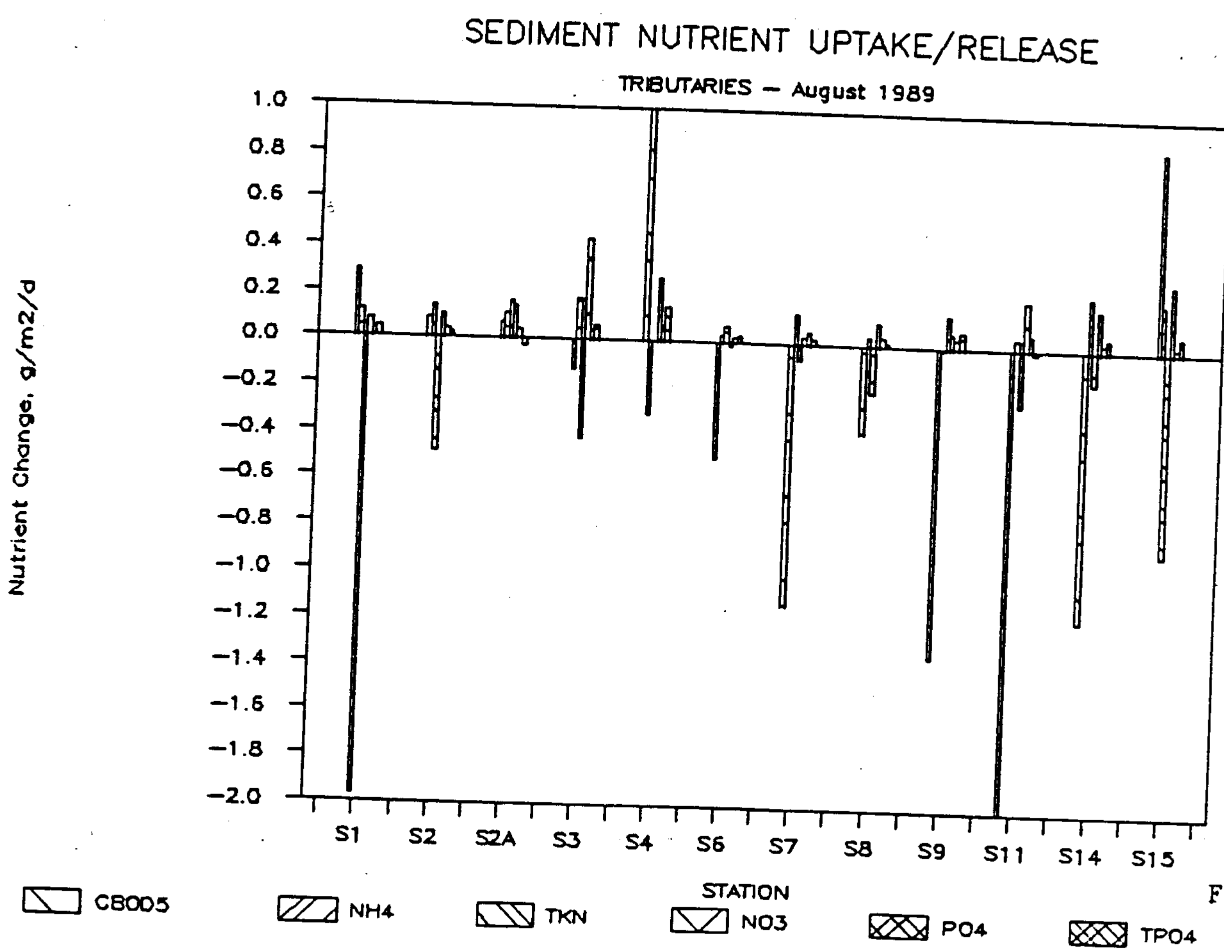


Figure 18

DENITRIFICATION MEASUREMENTS

Background

Denitrification has been shown to be a significant sink for nitrogen in several coastal systems (Seitzinger, 1988). In Narragansett Bay it removes about 50% of the dissolved inorganic nitrogen entering from the rivers and from sewage discharges (Seitzinger et al, 1984). An amount equal to about 35% of the organic nitrogen in the sediments is returned to the environment as nitrogen gas. Denitrification rates in both the Providence River (Seitzinger et al, 1984) and Ochlockonee Bay (Seitzinger, 1987), an unpolluted system in Florida, vary over an annual cycle.

Denitrification rates have been correlated with other, more frequently measured, parameters. A relation between ammonium flux from sediments and nitrogen gas flux was observed by Kemp et al (1982)

(2) $$N_2 = 0.19\ NH_4^+ - 4.2$$

units are mg N $m^{-2}d^{-1}$

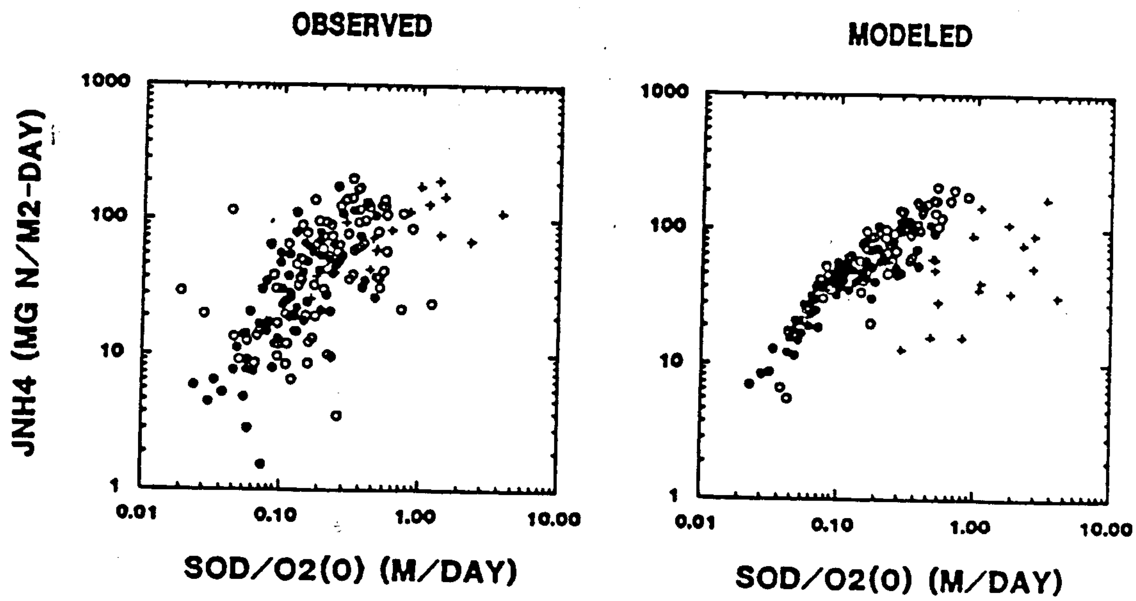
One estimate for denitrification was based on nitrate concentration in the overlying water where the N_2 flux of 2.3 mg N $m^{-2}d^{-1}$ was observed at a nitrate concentration of 0.28 mg N l^{-1} (Kemp et al, 1982). Dry creek beds in Great Sippewisset Marsh (Kaplan et al, 1979) showed a relation between denitrification rate and surface temperature

(3) $$N_2 = 4.36(^\circ C)$$ units as in (2)

Figure 19

CHESAPEAKE BAY

SEDIMENT MODEL COMPARISON - AMMONIA vs SOD/O2(0)



FCLX8P9 - SDDCHS3

and downward toward the anaerobic layer. Denitrification certainly occurs as the nitrate reaches the anaerobic layer. However, it has been postulated that there are also anaerobic microzones within the aerobic layer in which denitrification occurs. Observations from **in situ** dome experiments indicate that very little nitrate escapes from the sediments into the overlying water, and that nitrate is often taken up by sediments rather than released, under aerobic conditions.

Method

Sediment samples were collected from Sawmill Creek marsh and from the Hackensack River just upstream from station W-4 in December 1988 and September 1989. The sediments were incubated in the laboratory in special chambers, designed by Seitzinger, closely following her methods (Seitzinger et al, 1980). Dr. Seitzinger was consulted about the chamber fabrication, the experimental procedure, and the results obtained. However, she is not specifically responsible for any interpretations of the data.

Results

The results of the two experiments are presented in Tables 3 and 4, and in Figures 20 and 21. For the first experiment, sediment samples were squeezed under aerobic conditions to obtain interstitial water for nitrogen analyses. The results shown in Table 5 confirm the presence of ammonium and nitrate, necessary substrates for nitrification and denitrification processes. In both denitrification

Mixed slurries of intertidal sediment from Delaware Inlet, New Zealand yielded denitrification rates 50 to 500 times the **in situ** rates (Kaspar, 1983). Measurements made on mixed slurries indicate potential for the measured process to occur, but can be unrealistically high compared to the natural system.

Denitrification is a common process in soils and in freshwater and marine sediments. The denitrification reaction is bacterially mediated under obligatory anaerobic conditions. In sediments it is most often linked to the nitrification process in which ammonium is oxidized to nitrate, with small quantities of nitrite and nitrous oxide also produced, under aerobic conditions. Nitrification has been shown to be the principal source of nitrite and nitrate for denitrification in coastal sediments (Jenkins and Kemp, 1984). In some environments, nitrate appears to diffuse into the sediments, presumably in response to a gradient driven by nitrate consumption for denitrification, but the literature does not portray this as a universal or even as a major contributor to denitrification. Nitrification has also been shown to contribute up to 30% of the SOD in some environments.

An appropriate conceptual model is a two-layered sediment with ammonium diffusing either upward from the anaerobic deep sediment layer or downward from the overlying water into the aerobic sediment surface layer. In the aerobic layer the ammonium is oxidized to nitrate which then diffuses along its concentration gradient both upward toward the overlying water

Table 4

Hackensack River Denitrification Study

September 18 to October 18, 1989

Date	Day	Nitrogen Gas Released umol/m2/hr N1	N2
24 Degrees C			
9-20	2	409	148
9-22	4	241	-40
9-24	6	406	30
9-26	8	1408	262
9-28	10	1362	464
9-30	12	234	140
10-2	14	61	-14
10-4	16	91	21
10-6	18	184	157
10-8	20		
10-10	22	225	214
10-12	24	22	-44
10-14	26	-22	-3
10-16	28	192	71
10-18	30	28	-32
	Mean	346	98

Table 3

Hackensack River Denitrification Study

January 5 to February 3, 1989

Date	Day	Nitrogen Gas Released umol/m2/hr N1	N2
18 Degrees C			
1-5	1	20	468
1-6	2	1069	83
1-7	3	167	106
1-8	4	143	68
1-9	5		106
1-10	6	20	23
1-11	7	180	91
1-12	8	67	174
1-13	9	-93	287
1-14	10		
1-15	11	123	498
1-16	12	93	830
1-17	13	97	728
1-18	14	-67	75
1-19	15	67	140
1-20	16	40	136
1-21	17	73	162
1-22	18	-7	264
	Mean	125	249
26 Degrees C			
1-23	19	27	143
1-24	20	110	162
1-25	21		
1-26	22	-123	313
1-27	23	60	242
1-28	24	53	181
1-29	25	73	400
1-30	26	70	226
1-31	27	20	121
2-1	28	50	177
2-2	29	40	166
2-3	30	97	181
	Mean	43	210

Table 5

HACKENSACK RIVER STUDY

SEDIMENT INTERSTITIAL WATER NITROGEN

WINTER DENITRIFICATION EXPERIMENT

mg/l

	Station N-1 Marsh	Station N-2 River
NH4	5.20	4.33
NO3	0.40	0.96
NO2	0.17	0.16

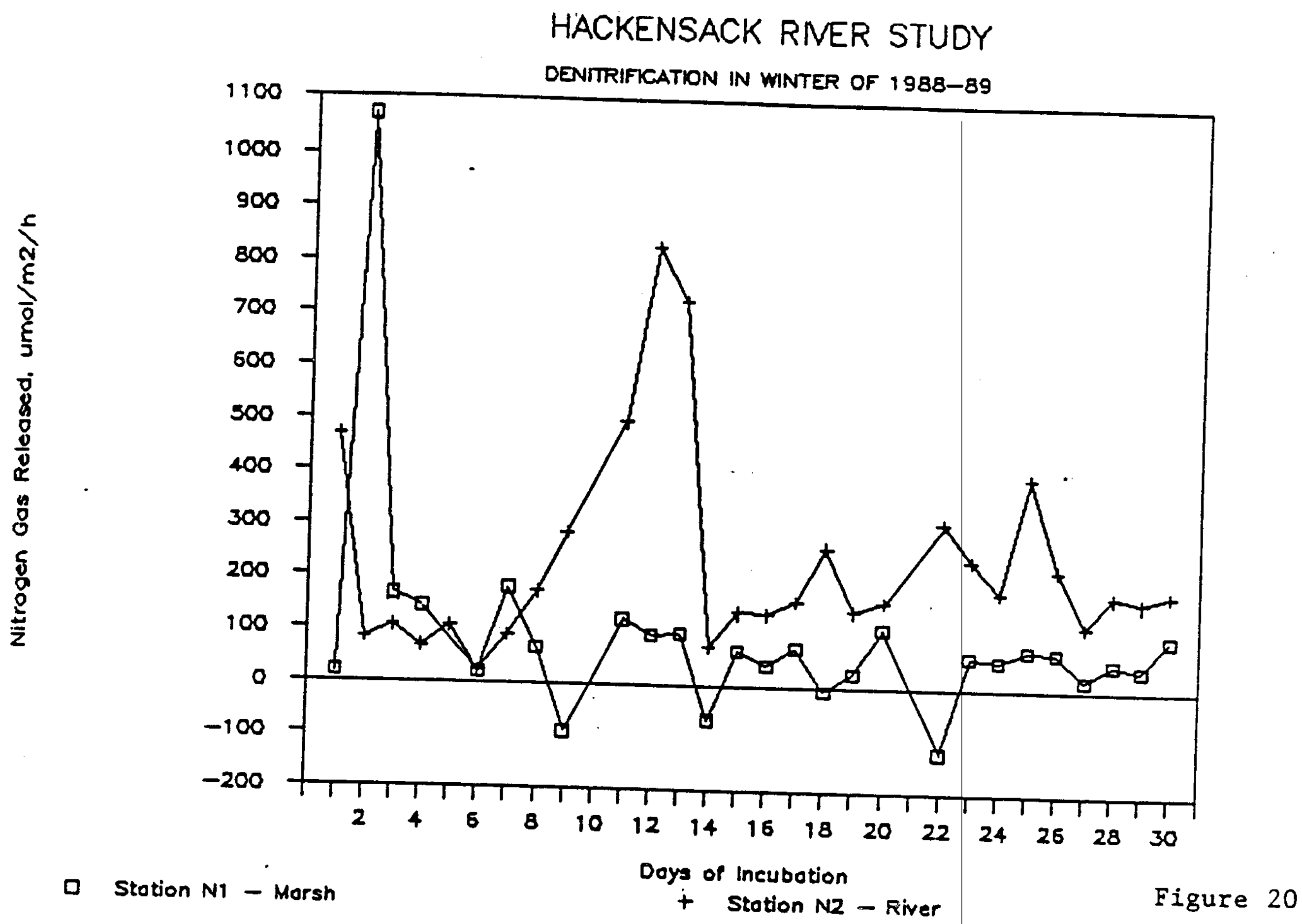


Figure 20

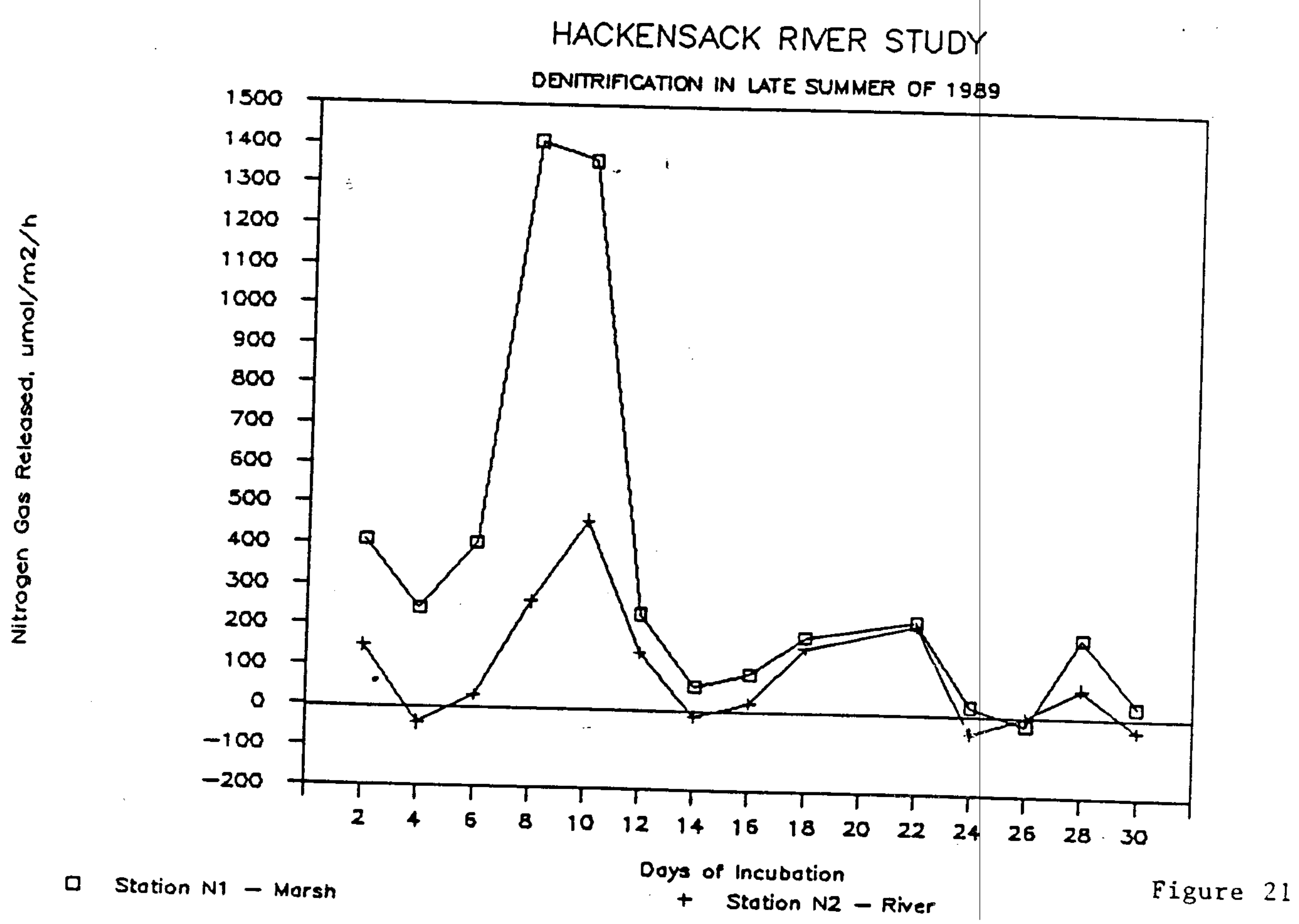


Figure 21

experiments, nitrogen gas evolution into the space above the sediments was quite variable with time. In the 'winter' experiment, large releases were observed in both cores during the first 3 days of measurements. This can be attributed to handling of the sediments during sampling, which probably disturbed the steady state interstitial oxygen, nitrate and ammonium gradients. Bubble formation in the sediments and release into the overlying space was also observed, along with the appearance of methane in the gas chromatographic analyses. Incubation temperature was increased from 18 C to 26 C after Day 19, but there is no obvious response of the system to temperature change apparent from Figure 7. Following the initial peak in the marsh core, denitrification rates were about 100 umol N $m^{-2}h^{-1}$ at 18 C and about 50 umol N $m^{-2}h^{-1}$ at 26 C.

The 'late summer' experiment, conducted with sediments collected in September 1989 and incubated at 24 C, did show some higher denitrification rates than the previous experiment at 26 C. The mean values, including the peaks between Days 8 - 10, were about 350 umol N $m^{-2}h^{-1}$in the marsh core and about 100 umol N $m^{-2}h^{-1}$ in the river core. The variations in nitrogen gas release were fewer than in the previous experiment and were synchronized in both cores. Excluding the large peaks, both sets of sediments evolved nitrogen at an average rate of about 100 umol N $m^{-2}h^{-1}$.

With all measurements included, the mean denitrification rates for each experiment (Tables 3 and 4) range from 98 to

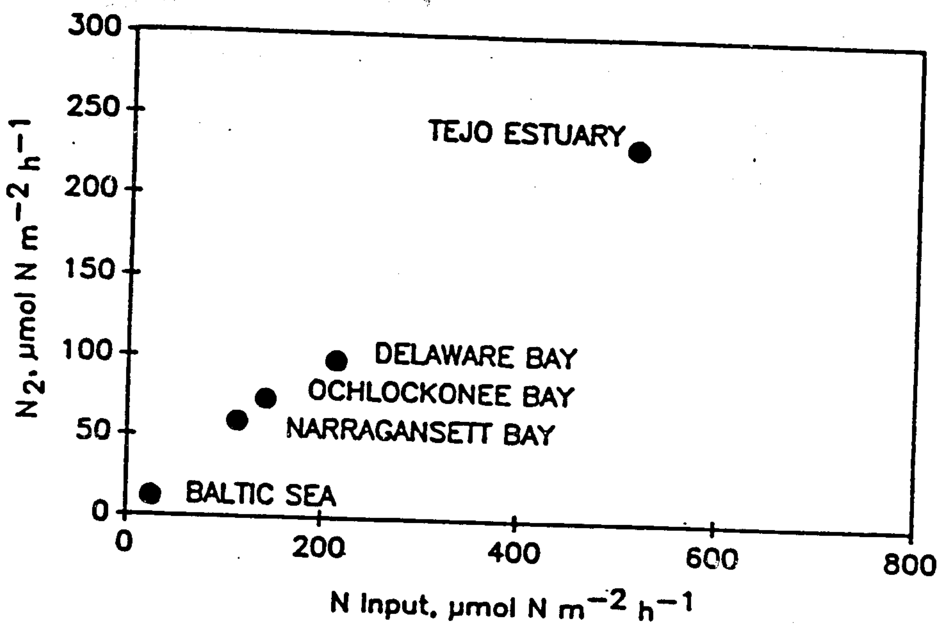


Fig. 2. Denitrification rates vs. external N inputs rates in estuaries.

Figure 22. Taken from Seitzinger (1988).

nitrogen inputs to five estuaries with nitrogen removal by denitrification. Her calculations indicate that an amount equivalent to approximately one-half of nitrogen inputs is removed by denitrification. If the denitrification rate in the Hackensack River were consistently near the upper bound of 728 umol N $m^{-2}h^{-1}$ (Table 3) throughout the River, an amount of nitrogen equivalent to 26% of the BCUA nitrogen load would be removed by denitrification. However, at a denitrification rate of 200 umol N $m^{-2}h^{-1}$, only about 7% of the load would be removed.

Information in the literature indicates that the typical source of nitrogen for denitrification in the sediments is ammonium which has been released from organic matter and nitrified. However, in the Hackensack River adjacent to the BCUA outfall the potential exists for ammonium to diffuse into the sediments from the overlying water, and by this route enter the nitrification-denitrification processes. The interstitial water analysis from one sediment core taken near Station W-4 yielded an ammonium concentration of 4.33 mgl $^{-1}$. The overlying water ammonium concentration in the vicinity of the outfall frequently exceeded this value during the surveys taken in April, July, and August 1988. In the same interstitial water sample, the nitrate concentration was 1.12 mgl^{-1}, which is approximately twice the overlying water concentration measured in April and several times the concentration in July and August. The region where ammonium peaks in the water also seems to show a minimum nitrate

249 umol N $m^{-2}h^{-1}$ in the River and from 43 to 346 umol N $m^{-2}h^{-1}$ in the marsh.

Discussion

The peak nitrogen releases from cores clearly show the potential of the denitrification process to respond quickly to the changes which modulate the availability of oxygen and ammonium for nitrification and which, in turn, produces the nitrate for denitrification. These peaks also reveal the potential for denitrification to produce large quantities of nitrogen gas from soluble ionic nitrogen species in the sediments. Seitzinger (1988) observed peak denitrification rates of 1067 umol N $m^{-2}h^{-1}$ in the Tejo Estuary and 888 umol N $m^{-2}h^{-1}$ in enriched MERL mesocosms. These values are comparable to the peaks reaching 1408 umol N $m^{-2}h^{-1}$ in the Hackensack experiments. Even if the 'typical' rate of denitrification in the Hackensack River were taken to be 200 umol N $m^{-2}h^{-1}$, the River lies within the ranges reported for eutrophic lakes and several coastal ecosystems, such as Delaware Bay, Ochlockonee Bay, Narragansett Bay, Kysing Fjord, and the Tama estuary (Seitzinger, 1988).

The Hackensack River may be compared to other estuaries in terms of loading rate per unit area. The nitrogen loading rate from the BCUA treatment plant is calculated to be 470 pounds per hour which, if distributed over the axis of the River, yields an areal loading of approximately 2800 umol N $m^{-2}h^{-1}$. This rate is off the scale of Seitzinger's (1988) Figure 2, reproduced here as Figure 22, which compares

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concentration. One explanation for these observations could be diffusion of ammonium into the sediments, nitrification to nitrate, and then immediate denitrification such that the nitrate does not escape in great quantity. The nitrate minimum in the overlying water could result from nitrate diffusing downward into a zone of nitrate depletion in the upper sediment layer where denitrification could be proceeding at a rapid rate.