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Neuse River Estuary Modeling and Monitoring Project Stage 1: An Examination of Long Term Nutrient Data in the Neuse River Watershed

Craig A. Stow and Mark E. Borsuk
Duke University
Nicholas School of the Environment
Box 90328
Durham, NC 27708

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Abstract

We examined nutrient loading patterns in the Neuse River, and estuarine nutrient concentrations in the Neuse River estuary from the 1970s through 1998. Total phosphorus loads have declined at all river stations considered, primarily due to a 1988 phosphate detergent ban. This phosphorus load decrease was quickly mirrored by an estuarine phosphorus concentration decrease. Nitrogen loading patterns in the river varied among stations. The furthest upstream station exhibited nitrogen decreases after the 1983 dam completion and impoundment of Falls Lake. At Clayton, the first station downstream of the Research Triangle, the total nitrogen load has increased steadily since the mid-1980's, primarily as a steady nitrate concentration increase. However, this increase is not reflected at the downstream Kinston station, where no clear nitrogen trends are discernable. Complete river profile plots of the ratio of median concentrations for the first and second half of the time period indicate that both nitrate increases and phosphorus decreases are damped moving downstream. This damping may be, in part, a result of the diminishing contribution of point-sources downstream. However, the completeness of the nitrate damping and the lack of a coincident total kjeldahl nitrogen increase indicate that compensating decreases must be occurring downstream of the Clayton station, or that nitrate increases may be mitigated by instream denitrification. The absence of clear nitrogen loading trends at Kinston is reflected in an absence of pronounced nitrogen concentration trends in the estuary. The lack of clear downstream nutrient increases suggests that current water quality impairment in the lower river and estuary may result from chronic nutrient overload rather than recent changes in the watershed.

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Summary and Conclusions

Despite a widely held perception that nutrients in the Neuse River system are rapidly increasing, our analysis of available data from the 1970s through 1998 indicates little support for this supposition. Estimated riverine phosphorus loads decreased in the late 1980s and this load decrease is reflected in lower phosphorus concentrations in the estuary. Nitrogen increases from the Research Triangle Area appear to have been partially offset by the impoundment of Falls Lake, which formed a large retention basin, and by other watershed processes, possibly including instream denitrification losses. We found no clear nitrogen loading trends at Kinston, nor did we find pronounced nitrogen concentration trends at the estuarine stations. We speculate that recent eutrophication problems may arise for several not necessarily exclusive reasons: 1) a changing N:P ratio that has shifted the suite of organisms in the estuary, 2) a chronic nutrient overload that began before the commencement of regular ambient monitoring, and 3) changing perceptions and expectations with a rapidly increasing coastal population.

Recommendations

The absence of pronounced recent nutrient load or concentration increases in the Neuse River system suggests that the drivers of contemporary estuarine eutrophication symptoms are not as straightforward as generally believed. We suggest that at least some of the symptoms of eutrophication in the estuary may result from a chronic nutrient overload rather than recent or short-term events. If this is true, then recovery of the ecosystem may also be a long-term process and it will be necessary to set expectations for recovery to an appropriate time-scale. The terms "algal bloom", "anoxia", "fishkill", and "*Pfiesteria*" have been used almost synonymously to describe eutrophication problems in the estuary. While they may all be eutrophication symptoms, they are distinct responses that may be operating on different time-scales. It may be appropriate to consider them separately when evaluating the likely response of the estuary to pending management actions.

Introduction

The Neuse River estuary in North Carolina has received considerable attention in recent years due to characteristic symptoms of excessive eutrophication: algal blooms, low dissolved oxygen, fish kills and outbreaks of toxic microorganisms. These problems have generally been attributed to increasing watershed nutrient loads (Burkholder et al. 1995; Paerl et al. 1995; Pelley 1998), and management options, including riparian buffers and source reductions, are currently being considered to control nutrient inputs. Increased nutrient loads are a typical consequence of the kinds of changes that have occurred in this watershed over the past several decades. The upper portion of the basin includes much of North Carolina's Research Triangle (defined by the cities of Raleigh, Durham, and Chapel Hill), an area that has experienced economic prosperity and rapid population growth since the 1970s. Population expansion and development are also occurring in lower portions of the basin with an increasing coastal population and a growing commercial hog-farming industry.

The Neuse River originates north of Durham, NC, flows approximately 320 km through the central piedmont to the coastal plain and comprises an area of 16,108 km^2 (Figure 1). The major land uses in the basin are agriculture (35%) and forestry (34%) with the remainder consisting primarily of urban areas, wetlands, scrub, and open water. In 1983 the US Army Corps of Engineers completed a dam that impounded the upper 35 km of the river to create Falls Lake, a multipurpose recreation and drinking water reservoir (NC Department of Environment, Health and Natural Resources 1993).

Concern about nutrient loads in the watershed began in the early 1980s and led to a phosphate detergent ban that became effective in 1988 (NC Department of Environment, Health and Natural Resources 1991). Since the ban, most attention has focused on the role of nitrogen as an algal stimulant and the state legislature recently mandated a 30% nitrogen load reduction. The establishment of Total Maximum Daily Loads (TMDLs) to further regulate nutrient discharges is pending.

Despite recent and imminent management actions to curb nutrient loads, there is a general perception that nutrients in the river are rapidly increasing in close concert with population and development. We examined the available long-term ambient monitoring data in both the river and the estuary to estimate riverine nutrient loading, and estuarine concentration responses. The results establish an historical context to help evaluate management options and also provide insights on the origin and behavior of nutrients in the watershed. This information will be useful to calibrate water quality models for establishment of TMDLs, particularly in the quantification of model prediction uncertainty. To successfully predict river response to future management actions it is first necessary to understand how the river has behaved in the past.

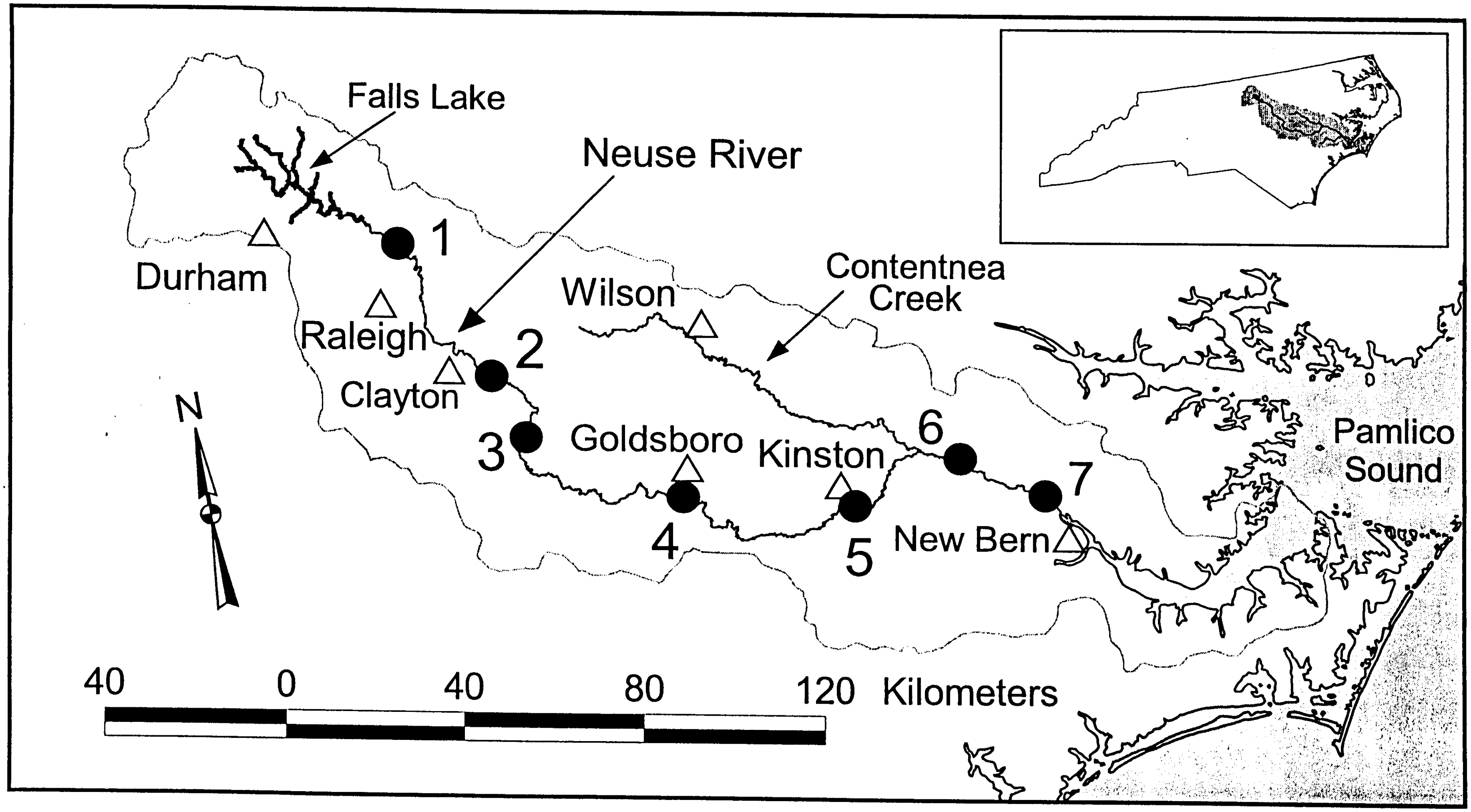


Figure 1- Neuse River Watershed and location of the river monitoring stations

Methods

Riverine Loading

We estimated nutrient loads using nutrient concentration data provided by the North Carolina Division of Water Quality (DWQ) and flow data provided by the United States Geological Survey (USGS). The DWQ maintains six ambient monitoring stations on the Neuse River mainstem (Figure 1). Periodic nutrient sampling began at some of these stations in the late 1960s, and regular monitoring, on approximately a monthly basis, began in the late 1970s. In this analysis, we use data from 1979 to 1998, the period of most consistent monthly monitoring. River concentration data are from surface grab samples following standard collection and analytical protocol (NC Department of Environment, Health and Natural Resources 1996). The USGS has recorded daily average flow at five of these stations, however the period of overlap between concentration and flow data was only sufficient to use three of the six stations for our estimates. The three stations considered are station 1, located just below the Falls Lake dam, station 2 at Clayton, NC, and station 5 at Kinston, NC (Figure 1). The Falls Lake station captures treated sewage and runoff from the city of Durham, which enter tributaries to Falls Lake. The Clayton station is located below Research Triangle Park and reflects both urban runoff and treated sewage discharges from the cities of Raleigh and Cary and from several smaller municipalities. The Kinston station is the furthest downstream location at which flow data are available.

To estimate nutrient loads at the Clayton and Kinston stations we used available concentration data and daily flow data to develop flow:concentration regression models for total kjeldahl nitrogen (TKN), nitrate (nitrate + nitrite), total nitrogen (nitrate + TKN), and total phosphorus. We estimated the models as:

$$\log(C) = \beta_{year}(\text{year}) + \beta_{flow}(\log \text{flow}) + \varepsilon, \qquad (1)$$

where C is nutrient concentration, β_{year} and β_{flow} are model slope and intercept parameters, respectively, year is a categorical variable, and ε is the model disturbance term. This model formulation thus allowed the intercepts to differ by year, but maintained a constant slope among years. This approach permitted the flow:concentration relationships to change with time and provided a measure of flow-adjusted concentration over time. To convert concentration estimates from the log to the natural metric we considered the three estimators discussed by Cohn et al. (1989) and chose the quasi-maximum likelihood estimator (QMLE). Because our sample sizes were large, and our model mean squared errors were small, all three estimators yielded similar results. We picked the QMLE estimator because it is consistent with point estimates that would be obtained using a Bayesian approach with a non-informative prior distribution. Daily concentration prediction variance in the natural metric was estimated as:

$$\sigma^2_c = \exp[2\log(C) + \sigma_\mu^2]\left(\exp[\sigma_\mu^2]\left(1 - \frac{2\sigma_\varepsilon^2}{m}\right)^{-m/2} - \left(1 - \frac{\sigma_\varepsilon^2}{m}\right)^{-m}\right)$$

where log C is the predicted concentration, σ_μ^2 is the prediction variance of the mean (log metric) σ_ε^2 is the variance of the error term, and m is the error degrees of freedom (Cohn et al 1989). We estimated daily loads as: measured flow times predicted concentration, and we estimated daily load variance as: σ^2_c times flow-squared. We then estimated annual total load and annual load variance by summing the estimated daily loads and variances, respectively.

The Falls Lake station did not exhibit pronounced flow:concentration relationships for the nutrients modeled. After considering several possible options we modeled concentration at this station as:

$$\log(C) = \beta_{year}(\text{year}) + \beta_d(\text{d})\beta_{month}(\text{month}) + \varepsilon. \qquad (2)$$

where year is a categorical predictor, d is a categorical variable differentiating the periods before and after completion of the Falls Lake dam, and month is a categorical predictor. In this model there is an intercept term, β_{year}, allowing yearly changes, a seasonal term, β_{month}, allowing seasonal concentration changes, and an interactive term, β_d, allowing the seasonal relationship to be different before and after dam completion. We used estimated concentrations from this model and daily flow values to estimate annual nutrient loads and variances as described for the Clayton and Kinston stations.

Estuarine Concentrations

We obtained nutrient concentration data from the DWQ for ten estuarine stations (Figure 2). Periodic sampling at three of these stations began as early as 1975. By 1979, approximately monthly sampling had begun at seven of the stations. The remaining three stations were not sampled on a monthly basis until 1989. These nine stations span the gradient from freshwater to saline conditions.

To investigate estuarine nutrient concentration changes with time, we used a nonparametric, locally weighted regression technique, referred to as *loess*. Nonparametric, in this context, should not be confused with familiar nonparametric order statistics. Nonparametric means that the model is not a concise algebraic formula summarized by one or more parameters. Nonparametric methods are advantageous for time series applications because time series data often display complex patterns that defy description by simple parametric functions (Cleveland, 1993). Loess modeling permits easy visualization of qualitative changes in nutrient concentrations over time. The loess method is particularly useful in identifying short term changes that may not be detected using traditional methods.

The loess method is implemented by calculating a weighted least squares fit for each time point using a neighborhood weight function (Statistical Sciences, 1995). For a given point, t_o, the k nearest neighbors of t_o constitute a neighborhood, $N(t_o)$. The number of neighbors, k, is usually specified as a proportion of the total number of points, called the *span*. Weights are then assigned to each point, t_i, in the neighborhood using the tri-cube weight function:

$$W\left(\frac{|t_o - t_i|}{\Delta(t_o)}\right)$$

where:

$$W(u) = \begin{cases} (1-u^3)^3, & \text{for } 0 < u < 1 \\ 0 & \text{otherwise} \end{cases}$$

and $\Delta(t_o)$ equals the largest distance between t_o and another point in the neighborhood, $\max_{N(to)}$

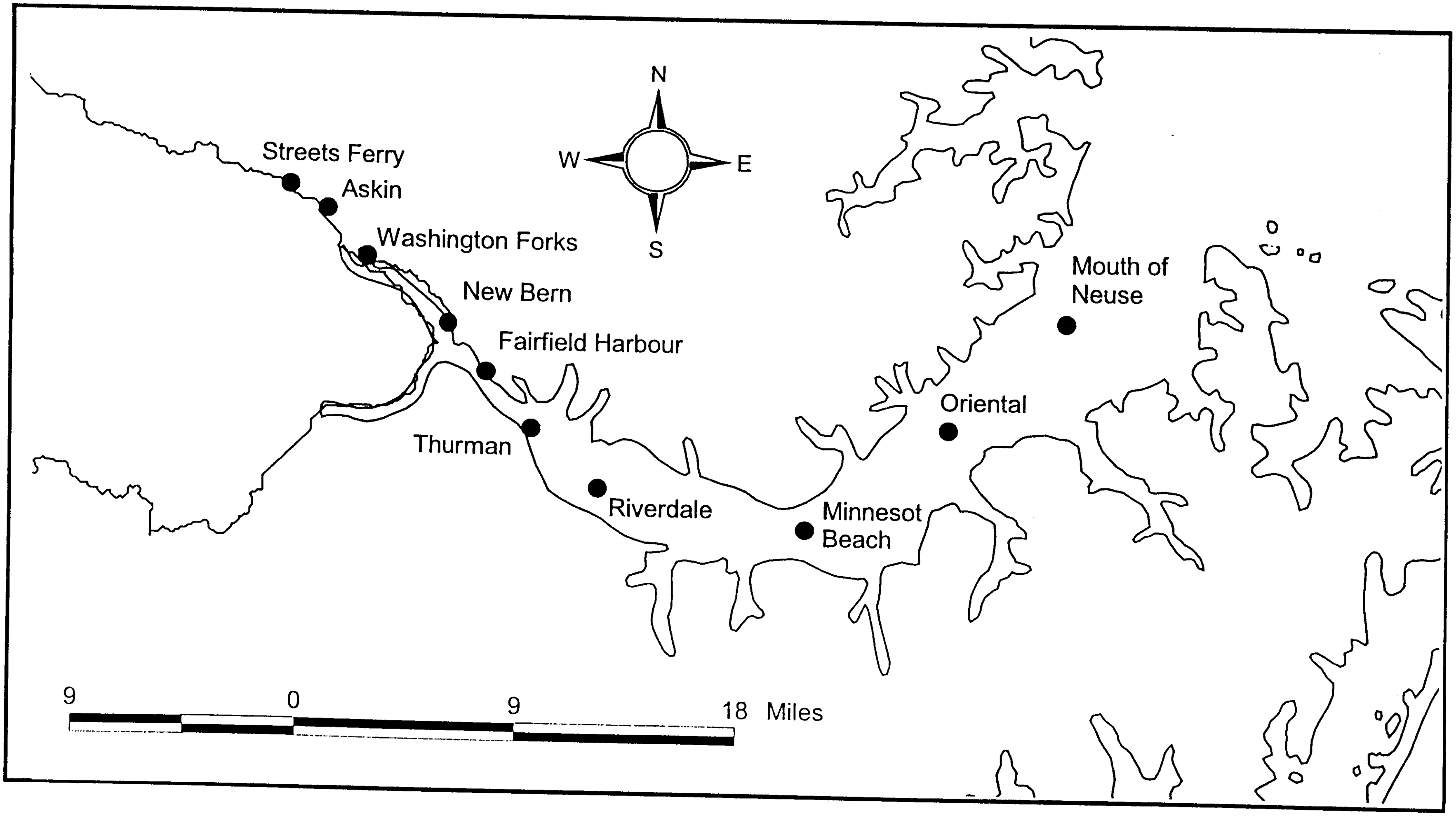


Figure 2 – Estuary map and location of estuary sample sites.

$|t_o-t_i|$. This weight function has a maximum at t_o, decreases in either direction, and becomes zero at the edges of the neighborhood. This function, therefore, assigns the greatest influence to those observations whose t_i lie closest to t_o.

The weighted least squares fit of y (the concentration, in this case) on the neighborhood $N(t_o)$ is then calculated using the weight function and used as the fitted value of y at that point. This process is repeated for each time point and the results are displayed graphically by connecting the fitted values by line segments. The least squares fit may be chosen to be either linear or quadratic, depending on the relative curvature of the data. This decision, as well as the choice of span width, is based on a combination of judgment and trial and error. The objective is to produce a fit line as smooth as possible that still tracks the underlying pattern in the data (Cleveland 1993). In this study, a quadratic fit was chosen for all ten stations, and a span of 1/3 was used for all stations except Fairfield Harbour, Riverdale, and Minnesot Beach. For these three stations a span of 1/4 was used due to the shorter period of record.

To prevent the seasonal cycle in nutrient concentrations from obscuring the longer term patterns, we deseasonalized the concentration data before initiating the loess procedure. This was done by performing a regression on the log-transformed data for each station using month as a classification variable. The residuals of this regression were retained and added to the overall mean for each station and this total was re-exponentiated to yield deseasonalized concentrations.

Results

Riverine Loading

Annual flows varied considerably at each station from 1979-98 (Figure 3). Though no trends are apparent, the completion of the Falls Lake dam in 1983 removed extreme high and low daily flows at both the Falls Lake and Clayton stations (not shown). Mean flows at the Falls Lake, Clayton, and Kinston stations since the dam completion (1984-98) have been 86%, 96%, and 98%, respectively, of the mean flows before dam completion (1979-83), suggesting higher evaporation rates resulting from the reservoir. Mean flows at the Falls Lake, Clayton and Kinston stations since the phosphate detergent ban (1988-98) have been 91%, 103%, 106%, respectively, of the flows prior to the ban (1979-87). From 1979-98 the Falls Lake station flow has averaged 63% and 25% of the Clayton and Kinston flows, respectively, and the Clayton station flow has averaged 40% of the Kinston station flow, consistent with the relative sizes of each station's subwatershed.

Summary statistics indicate that the models used to estimate annual loads predict behavior reasonably well for most nutrient fractions (Table 1). Total phosphorus models generally yielded the highest R^2 values because phosphorus concentration is strongly related to flow and has changed over time. Conversely, TKN models produced the lowest R^2 values. TKN generally exhibited the least overall variability and the weakest flow relationship. Available concentration data provided good coverage of the observed flow ranges at both the Clayton and Kinston stations, indicating minimal extrapolation was necessary to predict unobserved concentrations. At Clayton the observed flow range was 12.1-17.6 (log $m^3 \cdot day^{-1}$) and concentration data were available for a range of 12.4-16.8. The flow range at Kinston was 13.1-18.0 (log $m^3 \cdot day^{-1}$) and concentration data covered the range 13.4-17.4.

Annual total phosphorus loads have declined from 1979-98 at all three stations (Figure 4). The decrease at the Falls Lake station resulted principally from the completion of the dam and formation of Falls Lake in 1983, effectively creating a large retention basin for phosphorus to settle from the river water. The phosphate detergent ban that became effective in 1988 may also have contributed to the Falls Lake station decline, as the City of Durham treated sewage effluents enter tributaries of Falls Lake. Declines at the Clayton and Kinston stations are primarily attributable to the phosphate detergent ban and increases in wastewater treatment plant efficiency that occurred over the period of record (NC Department of Environment, Health and Natural Resources 1991). At Clayton the average annual total phosphorus load was 334 metric tons from 1979-87 and 135 metric tons from 1988-98, a decline of approximately 60% since the phosphate ban. Comparable values at Kinston are 450 and 237 metric tons before and after the ban, respectively, representing a 47% phosphorus decrease. These decreases can be attributed to decreased phosphorus concentrations (Figure 5) rather than a substantial difference in river flow since the ban (Figure 3). Flow from 1988-98 has been 103% and 106%, of flow from 1979-87 at Clayton and Kinston, respectively.

Nitrogen loading patterns over time differed among the three stations, with the Falls Lake station exhibiting nitrogen decreases, the Clayton station exhibiting apparent increases in nitrate and total nitrogen, and the Kinston station displaying no clear patterns (Figure 6). Nitrogen decreases

Figure 3 – Total yearly flow at Falls Lake Dam (bottom line), Clayton (middle line), and Kinston (top line) stations.

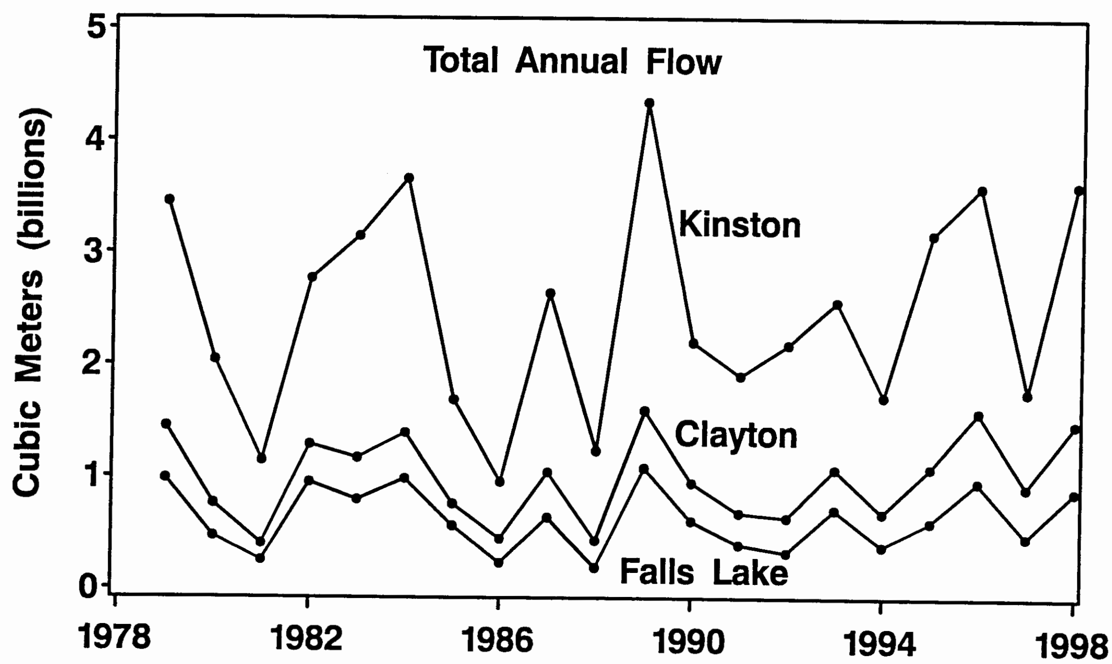


Figure 4 –Estimated yearly total phosphorus loads - Mean ± 2 standard errors.

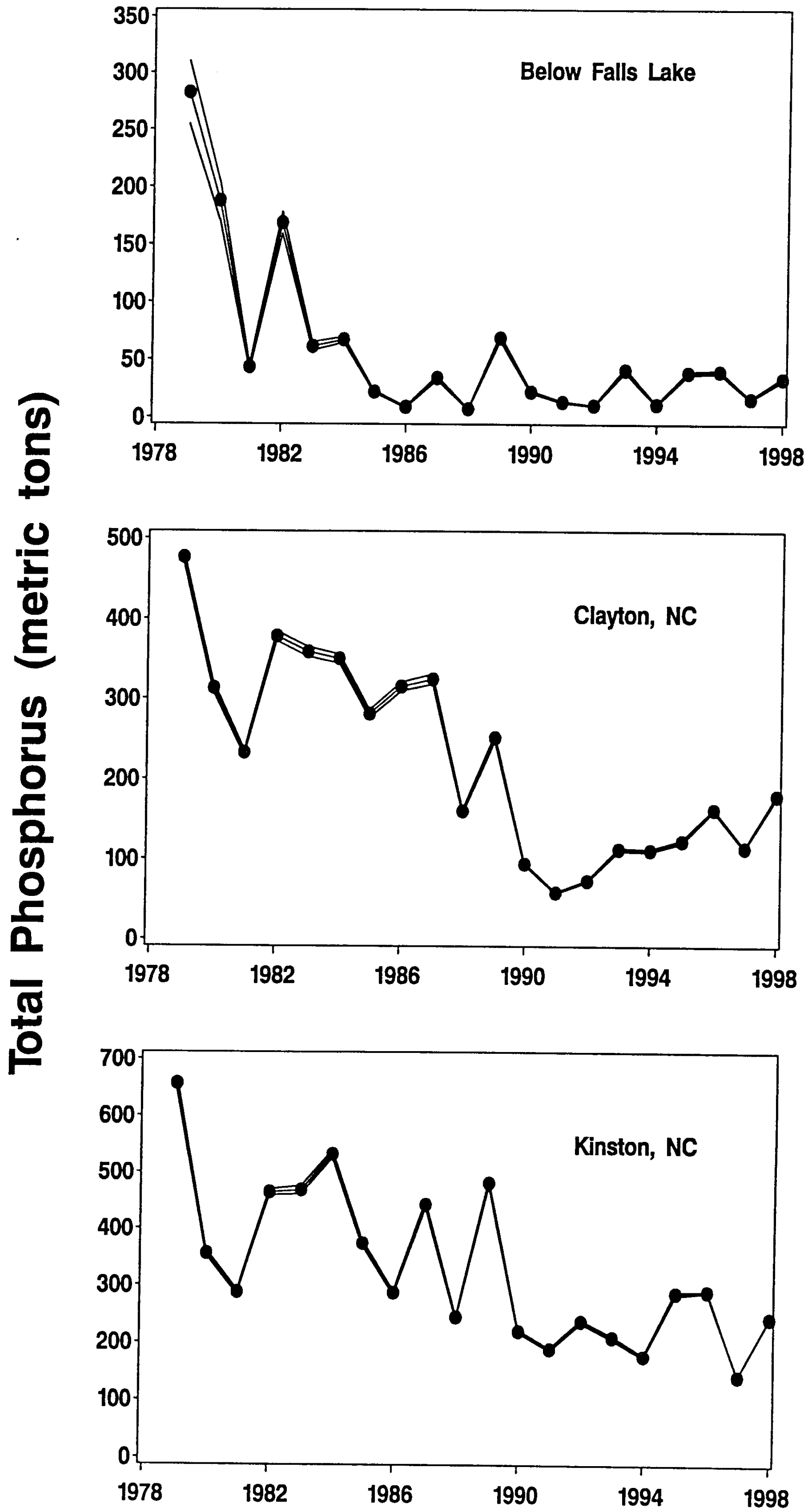


Figure 5 - Yearly flow-adjusted total phosphorus concentration ± one standard error. Concentrations are presented and the median daily flow, from 1979-1998, for both stations.

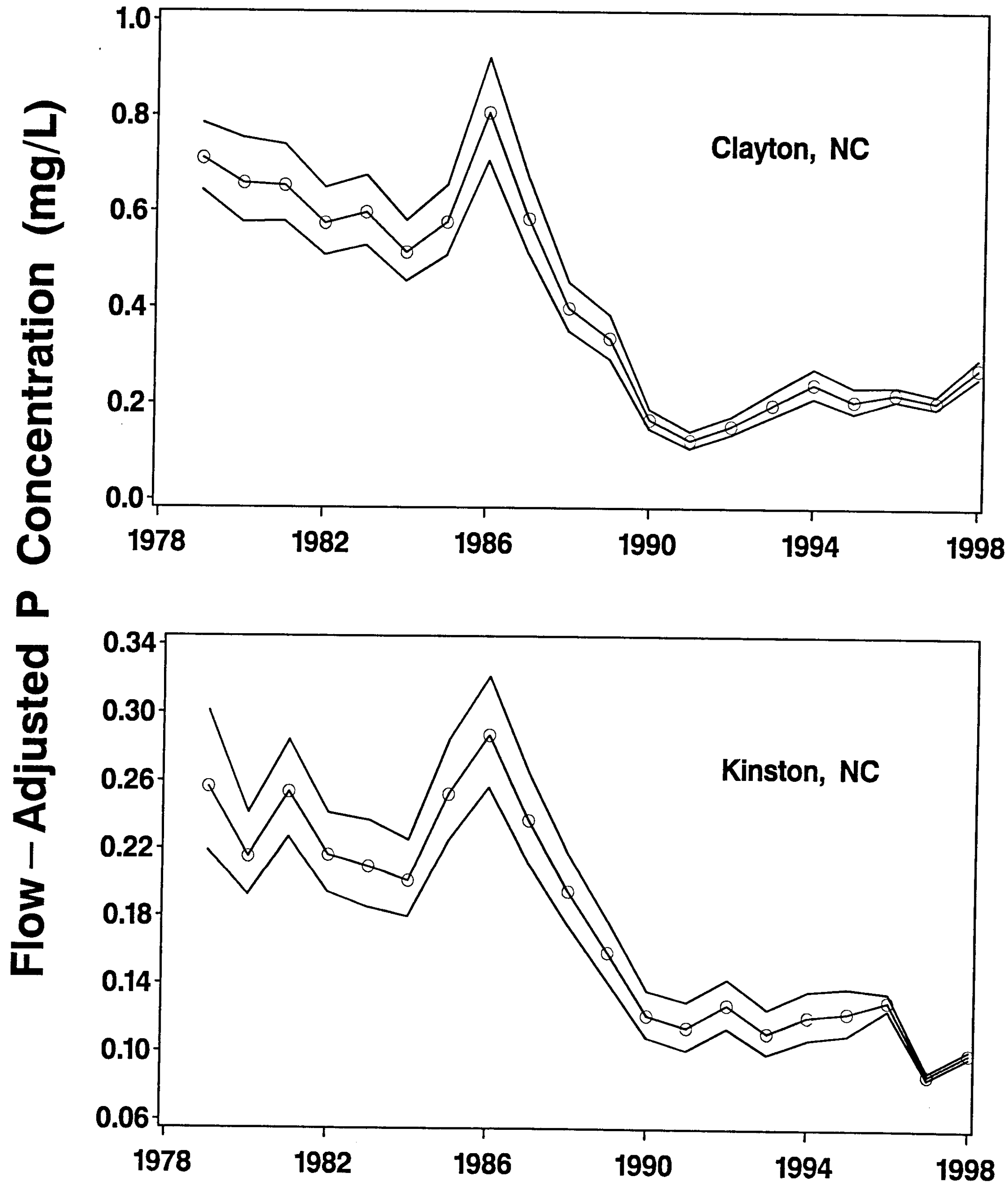


Figure 6 - Estimated yearly total nitrogen (circle), nitrate (triangle), and TKN (square) loads ± two standard errors.

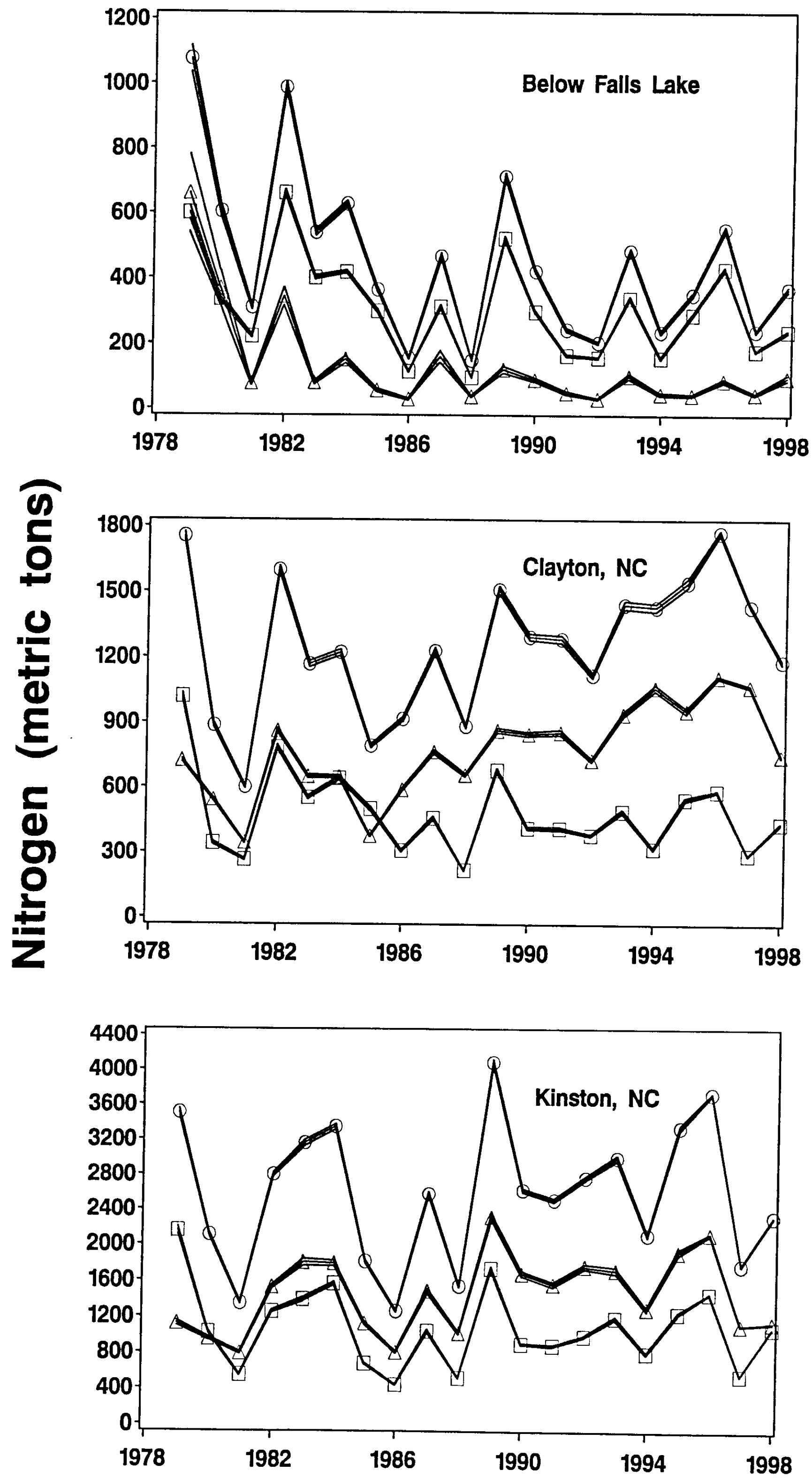


Table 1 – Summary of Models Used to Estimate Annual Loads

Station		Total Phosphorus	Total Kjeldahl Nitrogen	Nitrate	Total Nitrogen	
Falls Lake	n	210	211	209	209	
	mse	0.288	0.097	1.47	0.103	
	F	1,570	4,984	488	4,248	
	R^2	0.76	0.36	0.43	0.49	
Clayton	n	286	286	285	285	
	mse	0.184	0.103	0.098	0.084	
	F	5,342	8,172	6,648	6,847	
	R^2	0.77	0.41	0.82	0.70	
Kinston	n	618	619	619	619	
	mse	0.069	0.079	0.183	0.055	
	F	35,641	23,552	8,421	24,703	
	R^2	0.74	0.34	0.40	0.49	

at the Falls Lake station, like phosphorus decreases, resulted from the 1983 dam completion that formed Falls Lake. The lake provides a substantially increased hydraulic residence time that permits enhanced physical and biological nutrient removal. Total Kjeldahl nitrogen has been the dominant nitrogen form at the Falls Lake station, indicating that most nitrogen leaves Falls Lake as living or decaying organic matter. Since 1983, TKN has constituted approximately 75% of the nitrogen load at this station and nitrate has constituted the remainder.

The total nitrogen load at Clayton has increased steadily since the mid-1980s, resulting from a consistent nitrate increase over that period (Figure 6). This increasing pattern may actually have begun earlier but been partially offset by the upstream nitrogen load reduction associated with the 1983 dam completion. Prior to 1986, TKN and nitrate occurred in approximately equal proportions, but since 1986 both the nitrate load and relative proportion have increased. The TKN load has remained steady since the mid 1980s and may have declined slightly since the early 1980s. Flow-adjusted nitrogen concentrations at Clayton depict these patterns more clearly (Figure 7). From 1979-98 flow-adjusted TKN concentration decreased approximately 1.1% per year, while flow-adjusted nitrate increased approximately 5.6% annually. Despite the steady total nitrogen load increase since the mid-1980s, loads in the 1990s are comparable to peak loads in 1979 and 1982 (Figure 6).

The steady nitrate and total nitrogen load increases at Clayton are not clearly reflected downstream at Kinston (Figure 6). The total nitrogen load and flow-adjusted concentration (Figure 7) appear somewhat elevated at Kinston from approximately 1990-1995, but the trends are not consistent like the trends at Clayton. Nitrate and TKN loads at Kinston are close over the entire period, with neither demonstrating clear patterns (Figure 6). Flow-adjusted concentrations indicate the TKN concentration has generally decreased over the entire period, but nitrate concentration has varied (Figure 7). The flow-adjusted total nitrogen concentration has also varied with no overall trend.

Estuarine Concentrations

Loess plots reveal that total phosphorus concentrations at all ten stations have declined over the period of record (Figure 8). This decline is not linear, nor monotonic, however, and the use of the loess method allows for easy identification of short term patterns in concentration changes. Concentration increases in approximately 1981 and 1986, for example, are evident in the "bumps" in the fitted curves for the four upstream stations. The sharpest decreases at these stations are seen for the years 1986-1992, with a slight leveling out in concentrations in the past five years. The timing of the highs and lows is consistent with the patterns observed in flow-adjusted concentrations at the upstream river stations (Figure 5). Overall phosphorus concentrations, as well as the influence of river patterns, decrease down-estuary, with the lowest concentrations and most linear trend located at the station near the mouth of the estuary.

Total nitrogen concentrations in the estuary show a less marked pattern with time than phosphorus (Figure 9). However, a similar bump in 1981 is evident, perhaps due to low flow conditions that year (Figure 3). Since 1985, little change is evident, with perhaps a slight decrease in concentrations in the past three years. Total nitrogen concentrations, like phosphorus, also decrease moving downstream with the lowest levels near the mouth. Nitrate

Figure 7 - Yearly flow-adjusted total nitrogen (circle), nitrate (triangle), and TKN (square) concentrations ± one standard error. Concentrations are presented and the median daily flow, from 1979-1998, for both stations.

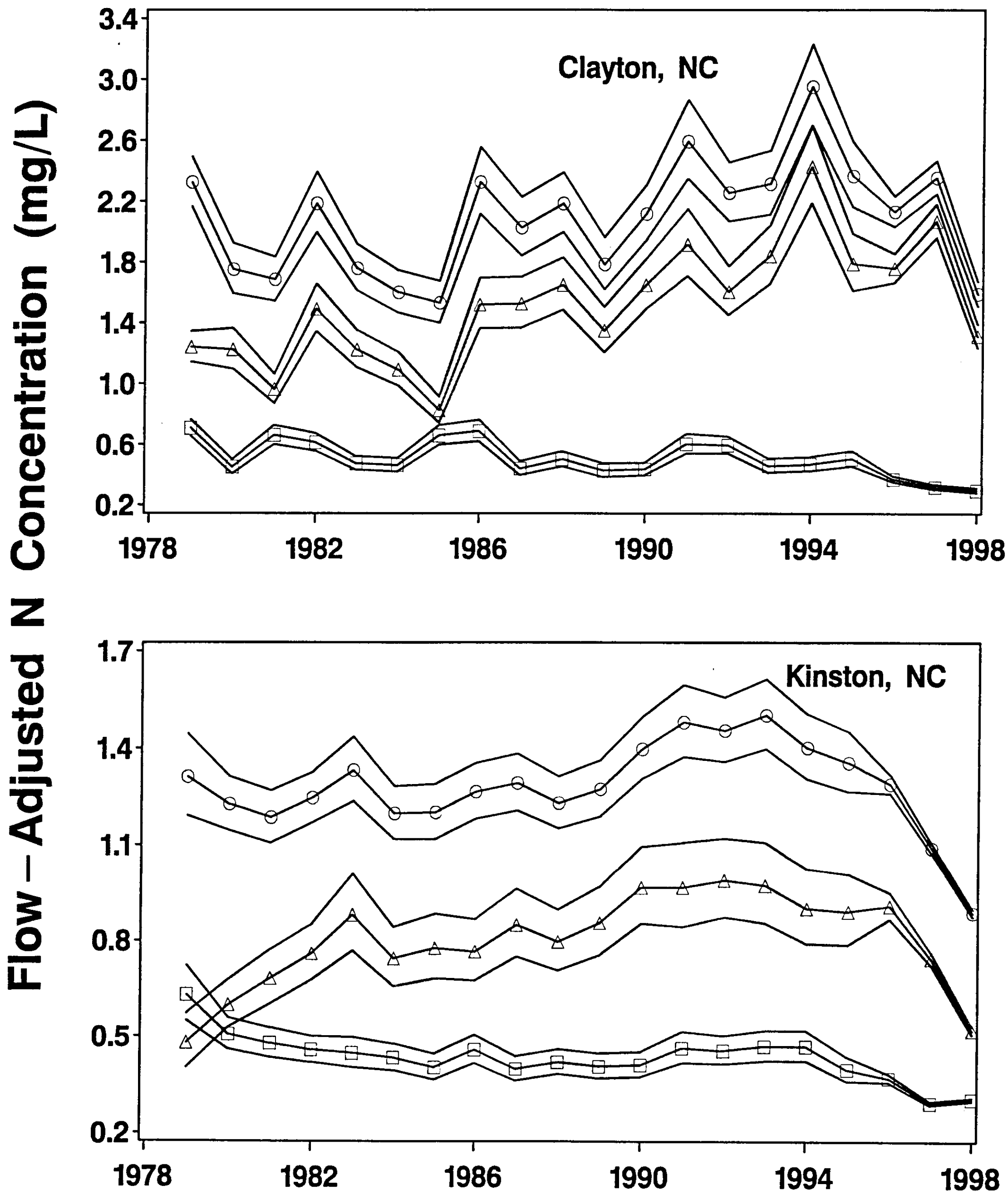


Figure 8 - Loess plots of deseasonalized total phosphorus concentrations in the estuary.

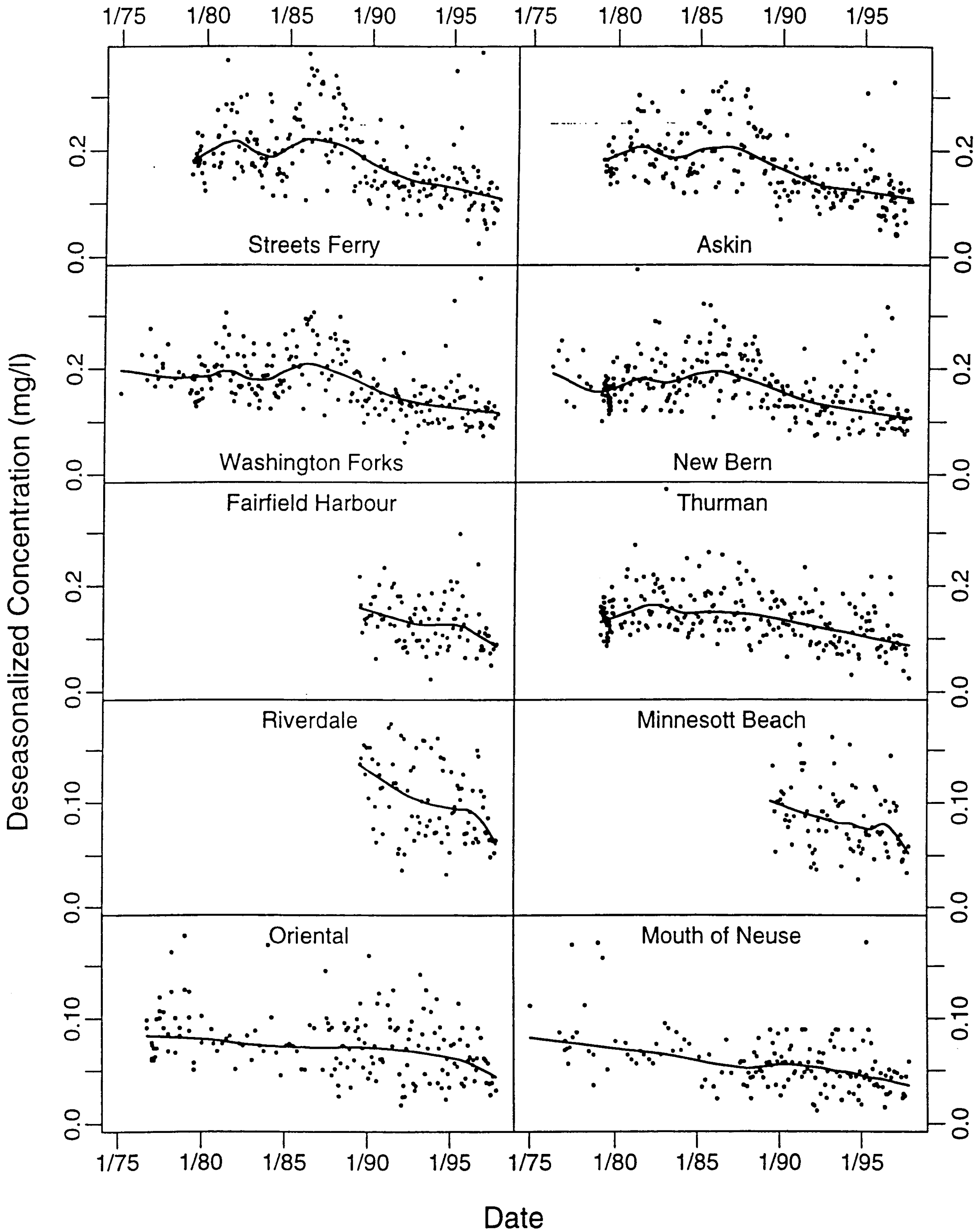


Figure 9 - Loess plots of deseasonalized total nitrogen concentrations in the estuary.

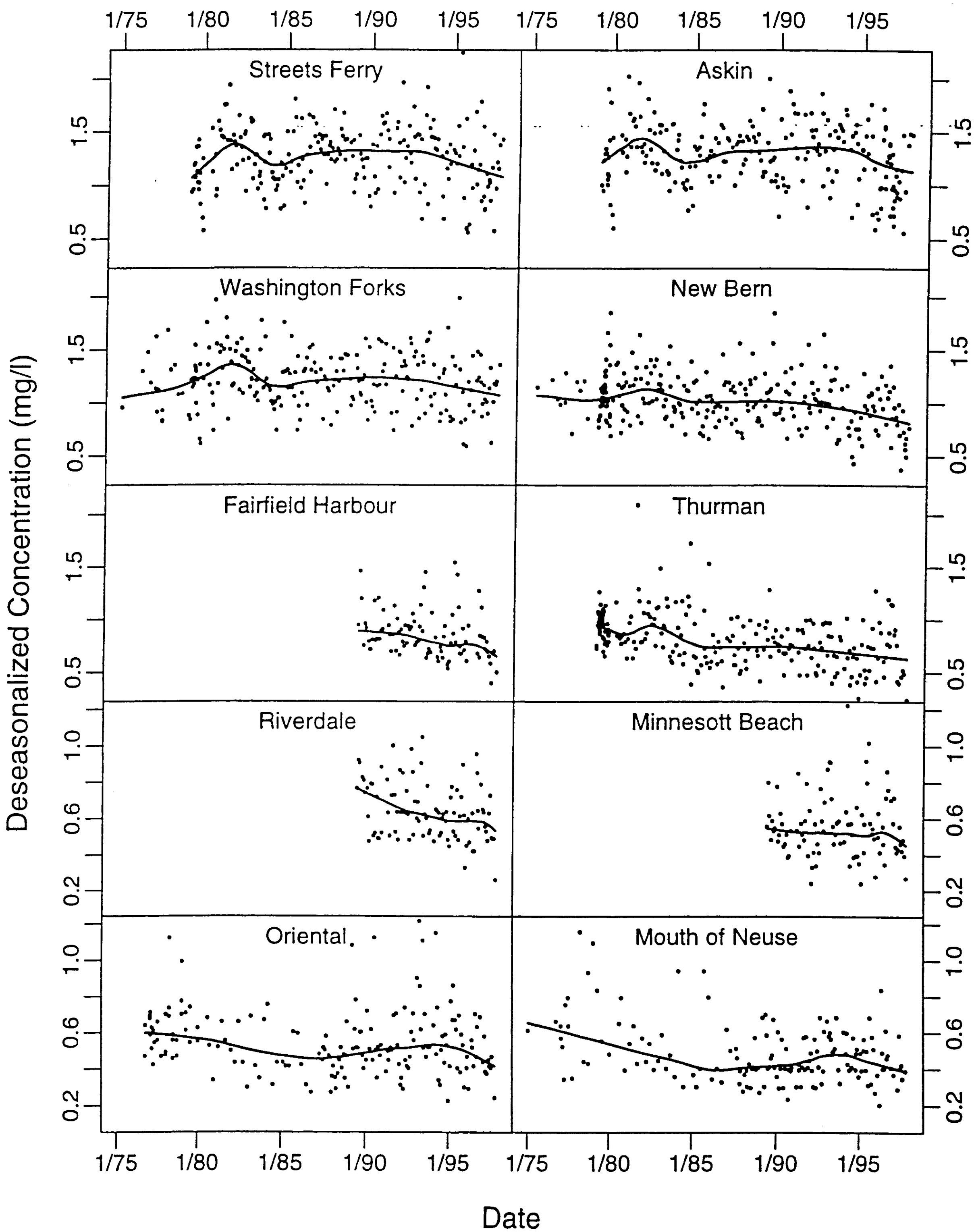


Figure 10 - Loess plots of deseasonalized nitrate concentrations in the estuary.

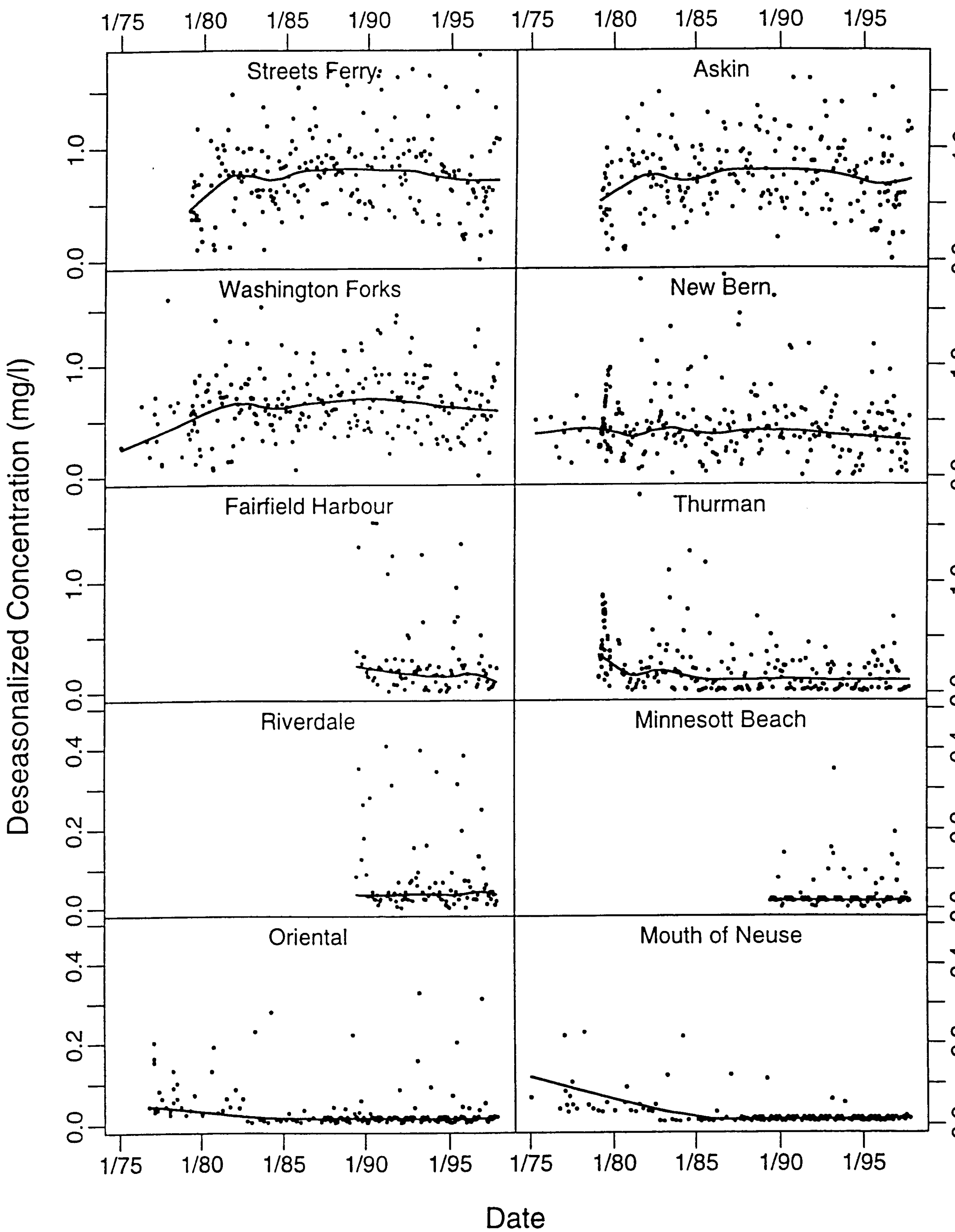
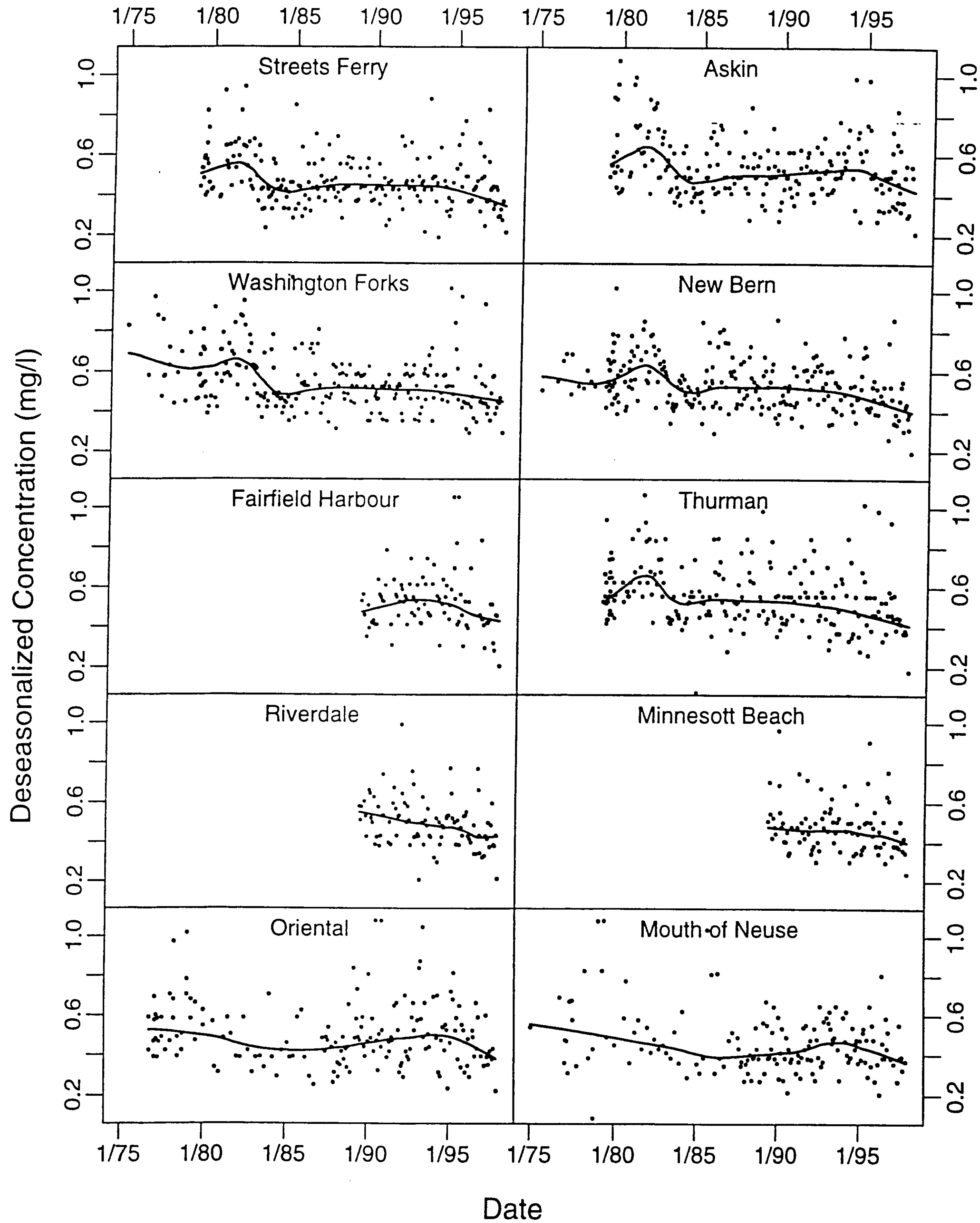


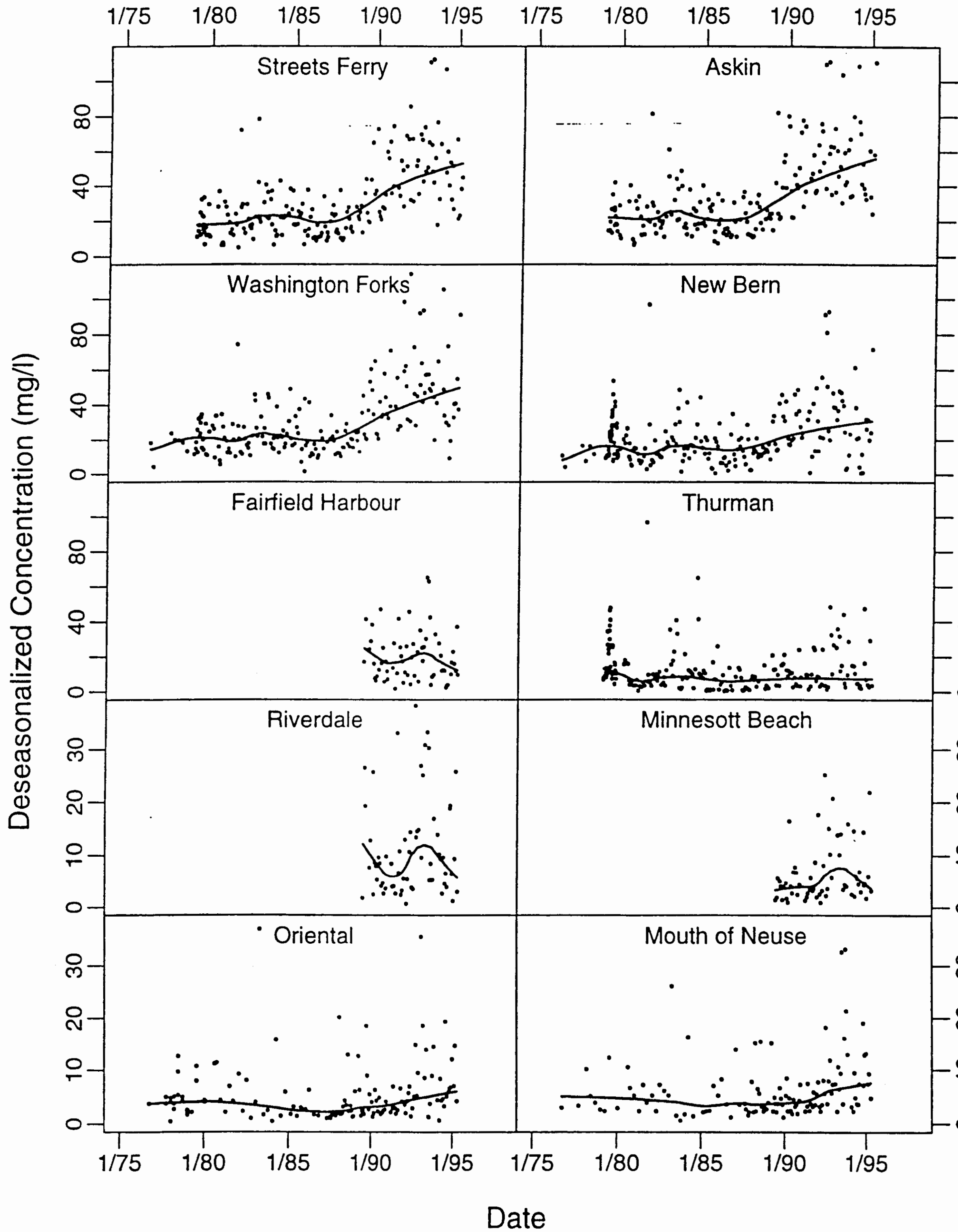
Figure 11 - Loess plots of deseasonalized total kjeldahl concentrations in the estuary.



and TKN plots (Figures 10-11) similarly do not show any pronounced patterns over time, although they do reveal that the 1981 bump is due primarily to the TKN fraction. Concentrations at each station also reveal that while nitrate concentration is reduced in the transition from freshwater to saline conditions, the TKN fraction, which includes organic nitrogen, remains at a consistent level throughout the estuary.

The molar ratio of dissolved inorganic nitrogen to dissolved inorganic phosphorus is often used as an indicator of phytoplankton limitation in aquatic systems. In the Neuse estuary, this ratio has changed substantially over the period of record. Loess plots of the four upstream stations (Figure 12) indicate a pronounced increase beginning in 1987. After a steady pre-1987 fitted value of approximately 20, the ratio has increased to a fitted value as high as 60 in late 1994. The continuation of this pattern cannot be evaluated after 1994 as the measurement of orthophosphate was halted by the DWQ in 1995. Increases in the N:P ratio at the downstream stations are generally less pronounced or obscured by the short sampling periods.

Figure 12 - Loess plots of deseasonalized dissolved inorganic N:P ratios in the estuary.



Discussion

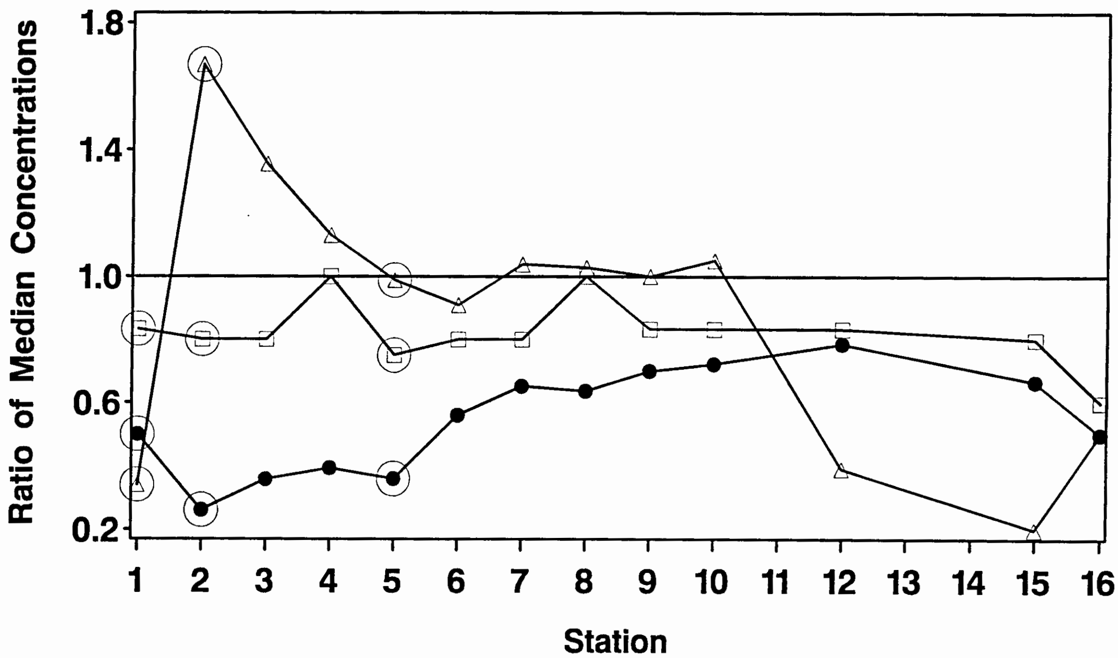
Phosphorus loads have decreased at all river stations examined, largely because of point source controls that included the 1988 phosphate detergent ban (Figure 4). Phosphorus load decreases in the river are reflected in estuarine total phosphorus concentrations (Figure 8), indicating a rapid response to upstream load decreases.

In contrast, nitrogen loading patterns vary by station and the nitrogen fraction considered. Flow-adjusted concentrations (Figure 7) reveal that the steady total nitrogen load increase at Clayton from approximately 1985 to 1998 (Figure 6) resulted entirely from an increasing nitrate fraction, while the TKN fraction has remained steady or decreased slightly. The total nitrogen load at Kinston appears to have also increased from 1986 through 1996, however a comparison with the flow pattern at Kinston over that period (Figure 3) indicates that this period of increasing total nitrogen is largely due to flow variability rather than a systematic increase in either the nitrate or TKN fractions. This is confirmed by the lack of clear trends in flow-adjusted concentrations (Figure 7). Nitrogen concentrations in the estuary exhibit no pronounced trends (Figures 9-11), consistent with the absence of clear loading trends at the Kinston station.

The absence of consistent, overlapping flow data at three of the six river monitoring stations prohibits a quantitative assessment of yearly nutrient differences along the entire river because the effect of flow variability on nutrient concentrations cannot be accommodated. To provide a more aggregated view of nutrient differences with time at all six river stations and the ten estuarine stations we calculated the median phosphorus, nitrate, and TKN concentrations for the periods from 1988-1998, and 1979-1987 and plotted the ratio of the respective median concentrations for each station from upstream to downstream (Figure 13). The median is a measure of central tendency resistant to extreme observations. Considering the ratio of medians over two extended time periods mitigates short-term flow-induced variability. For phosphorus this time breakdown represents the periods before and after the phosphate detergent ban. For nitrogen this breakdown is more arbitrary, dividing the period from 1979-1998 approximately in half.

Total phosphorus median concentration ratios exhibit patterns consistent with observed loading and flow-adjusted concentration patterns at the Falls Lake, Clayton, and Kinston stations (Figure 13). Total phosphorus median ratios of less than one at these three stations reflect the phosphorus decreases observed in the latter half of the time period considered. The decrease appears largest at Clayton (station 2), which is strongly influenced by treated sewage effluents from the Raleigh-Durham area. Median ratios indicate that phosphorus has decreased at all stations, but moving downstream from Clayton (station 2) to Streets Ferry (station 7) the magnitude of the phosphorus decrease appears to diminish. This is probably because the phosphorus decrease resulted largely from point-source controls, and point-sources constitute a smaller proportion of the overall phosphorus load further downstream. This is supported by the observation that hydrology is the principal driver at Kinston, as evidenced by the similarity between the patterns of flow (Figure 3) and nutrient load (Figures 4 and 6). Once in the estuary the median ratios change only slightly.

Figure 13 - Median concentration from 1988-1998 divided by median concentration from 1979-1987, for all sixteen (river plus estuary) monitoring stations. Circles indicate the Falls Lake (station 1), Clayton (station 2), and Kinston (station 5) stations.



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