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UNIVERSITY OF MARYLAND CENTER for ENVIRONMENTAL SCIENCE

CHESAPEAKE BAY
WATER QUALITY MONITORING PROGRAM
ECOSYSTEM PROCESSES COMPONENT (EPC)

LEVEL ONE REPORT #17 (INTERPRETIVE)

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MARYLAND CHESAPEAKE BAY WATER QUALITY MONITORING PROGRAM

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LEVEL ONE REPORT NO. 17

INTERPRETIVE REPORT

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PREPARED FOR:

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1. INTRODUCTION

During the past decade much has been learned about the effects of both natural and anthropogenic nutrient inputs (*e.g.*, carbon, nitrogen, phosphorus, silica) on such important estuarine features as phytoplankton production, algal biomass, seagrass abundance and distribution and oxygen conditions in deep waters (Nixon, 1981, 1988; Boynton *et al*., 1982; Kemp *et al.*, 1983; D'Elia *et al.*, 1983; Garber *et al.*, 1989; Malone, 1992; and Kemp and Boynton, 1992). While our understanding is not complete, important pathways regulating these processes have been identified and related to water quality issues. Of particular importance here, it has been determined that (1) algal primary production and biomass levels in many estuaries (including Chesapeake Bay) are responsive to nutrient loading rates, (2) high rates of algal production and algal blooms are sustained through summer and fall periods by benthic recycling of essential nutrients (3) deposition of organic matter from surface to deep waters links these processes of production and consumption, and (4) submerged aquatic vegetation (SAV) communities are responsive to water quality conditions, especially light availability.

1.1 Conceptual Model of Estuarine Nutrient and Water Quality Processes in Chesapeake Bay

Nutrients and organic matter enter the bay from a variety of sources, including sewage treatment plant effluents, fluvial inputs, local non-point drainage and direct rainfall on bay waters. Dissolved nutrients are rapidly incorporated into particulate matter via biological, chemical and physical mechanisms. A portion of this newly produced organic matter sinks to the bottom, decomposes and thereby contributes to the development of hypoxic or anoxic conditions and loss of habitat for important infaunal, shellfish and demersal fish communities. The regenerative and large short-term nutrient storage capacities of estuarine sediments ensure a large return flux of nutrients from sediments to the water column that can sustain continued high rates of phytoplanktonic growth and biomass accumulation. Continued growth and accumulation supports high rates of deposition of organics to deep waters, creating and sustaining hypoxic and anoxic conditions typically associated with eutrophication of estuarine systems. To a considerable extent, it is the magnitude of these processes that determines water quality conditions in many zones of the bay. Ultimately, these processes are driven by inputs of organic matter and nutrients from both natural and anthropogenic sources. If water quality management programs are instituted and loadings of organic and nutrients decrease, changes in the magnitude of the processes monitored in this program are expected and will serve as a guide in determining the effectiveness of strategies aimed at improving bay water quality and habitat conditions. The schematic diagram in Figure 1-1. summarizes this conceptual eutrophication model where increased nitrogen (N) and phosphorus (P) loads result in a water quality degradation trajectory and reduced N and P loads lead to a restoration trajectory.

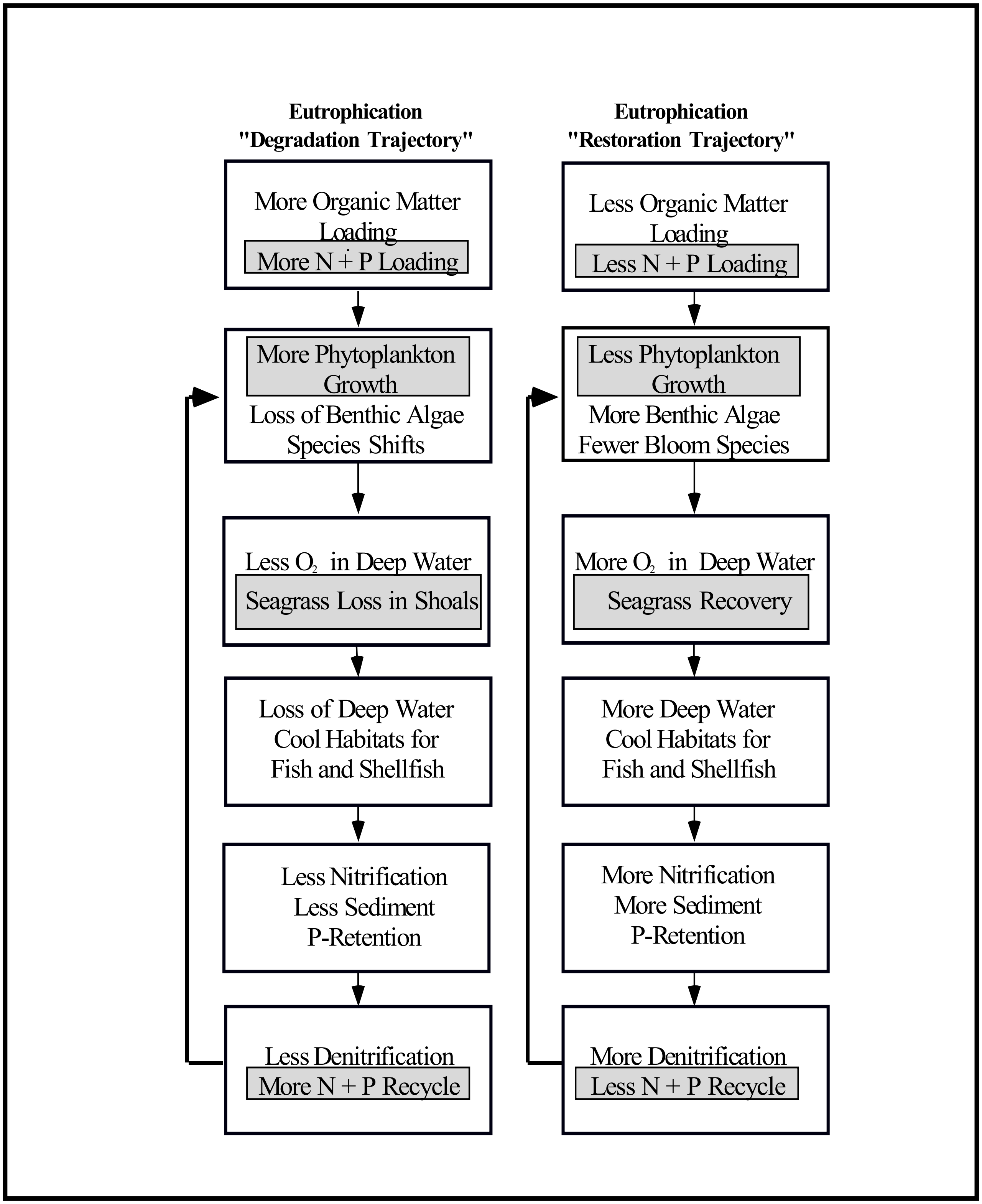

Figure 1-1. A simplified schematic diagram indicating degradation and restoration trajectories of an estuarine ecosystem. Lightly shaded boxes in the diagram indicate past and present components of the EPC program in the Patuxent River and Tangier Sound. (Adapted from Kemp, *pers. comm.*, HPEL)

Within the context of this model a monitoring study of sediment processes and SAV habitat conditions has been developed. The Ecosystem Processes Component (EPC) has been gathering information since 1985. Initial program components included monitoring of Sediment-Water Oxygen and Nutrient Exchanges (SONE; 1985-1997) at multiple locations (8-10) in the bay and tributaries and monitoring of the vertical flux of sediments and organic particulates at one location in the mainstem bay (VFX; 1985-1992). More recently the SONE program was modified to a more spatially intensive effort focused on the Patuxent River (MINI-SONE program; 1996-1999). In 1992, 1995-1997 a small program was instituted at one location in the Patuxent River to monitor, at high measurement frequencies, dissolved oxygen conditions. Finally, extensive SAV habitat evaluations were initiated in the Patuxent River (1997-1999) and were expanded to Tangier Sound during 1999. In all of these monitoring activities the working hypothesis is if nutrient and organic matter loadings decrease, the cycle of high organic deposition rates to sediments, sediment oxygen demand, release of sediment nutrients, continued high algal production, and high water column turbidity will also decrease. As a result, the potential for SAV recolonization will increase and the status of deep water habitats will improve.

1.2 Objectives of the Water Quality Monitoring Program

The EPC of the Maryland Chesapeake Bay Water Quality Monitoring Program conducted monitoring of sediment-water oxygen and nutrient exchanges (MINI-SONE), investigated techniques to increase the spatial coverage of sediment-water oxygen and nutrient exchange measurements based on sediment chlorophyll-a and other bottom water quality distributions, and evaluated habitat conditions relative to SAV reintroduction. The Patuxent River estuary, where EPC efforts are concentrated, is an area of particular interest because substantial reductions in nutrient loading rates have been achieved in this system. During 1999, SAV habitat conditions were investigated and a new technique for mapping surface water quality conditions was employed in Tangier Sound.

The EPC has been modified since its inception in 1984 while maintaining the overall objectives that are consistent with those of other Monitoring Program Components:

1. Characterize the present status of the Patuxent estuary (including spatial and seasonal variation) relative to sediment-water nutrient exchanges and sediment oxygen consumption rates.

2. Determine the long-term trends that develop in sediment-water nutrient exchanges and sediment oxygen consumption rates in response to pollution control programs in the Patuxent estuary.

3. Continue to develop statistical techniques for estimating the performance of estuarine sediments. These will allow greater spatial resolution than is possible with current approaches and allow “whole-estuary” estimates of sediment processes to be directly compared with nutrient loading rates from terrestrial and atmospheric sources.

4. Evaluate near-shore water quality conditions relative to SAV habitat across a range of spatial and temporal scales in the Patuxent River estuary and Tangier Sound. This program element also includes an investigation of the potential for light attenuation due to epiphytic fouling of SAV leaves.

5. Integrate the information collected in this program with other elements of the monitoring program to gain a better understanding of the processes affecting Chesapeake Bay water quality and its impact on living resources.

1.3 Status of the EPC of the Maryland Chesapeake Bay Water Quality Monitoring Program

The Chesapeake Bay Water Quality Monitoring Program was initiated to provide guidelines for restoration, protection and future use of the mainstem estuary and its tributaries and to provide evaluations of implemented management actions directed towards alleviating some critical pollution problems. A description of the complete monitoring program is provided in Magnien *et al.* (1987) and the Chesapeake Bay program web page (*http://www.chesapeakebay.net* and *http://www.dnr.state.md.us/bay/monitoring/eco/index.html*). In addition to the EPC program portion, the monitoring program also has components that measure:

1. Freshwater, nutrient and other pollutant input rates,
2. chemical and physical properties of the water column,
3. toxicant levels in sediments and organisms,
4. phytoplankton and zooplankton community characteristics (abundances, biomass and primary production rates) and
5. benthic community characteristics (abundances and biomass).

The first phase of the study was undertaken over a period of four years (1984 through 1987) and had as its goal the characterization of the existing state of the bay, including spatial and seasonal variation, which were keys in the identification of problem areas. During this phase of the program the EPC measured sediment-water oxygen and nutrient exchange rates and determined the rates at which organic and inorganic particulate materials reached deep waters and the sediment surface. Sediment-water exchanges and depositional processes are major features of estuarine nutrient cycles and play an important role in determining water quality and habitat conditions. The results of EPC monitoring have been summarized in a series of interpretive reports (Boynton *et al.*, 1985, 1986, 1987, 1988, 1989, 1990, 1991, 1992, 1993, 1994, 1995, 1996, 1997, 1998 and 1999). The results of this characterization effort have largely confirmed the importance of deposition and sediment processes in determining water quality and habitat conditions.

The second phase of the monitoring effort, completed during 1988 through 1990, identified interrelationships and trends in key processes monitored during the initial phase of the program.

The EPC was able to identify trends in sediment-water exchanges and deposition rates. Important factors regulating these processes have also been identified and related to water quality conditions (Kemp and Boynton, 1992; Boynton *et al.*, 1991).

In 1991 the program entered its third phase. During this phase the long-term 40% nutrient reduction strategy for the bay was reevaluated. In this phase of the process, the monitoring program was used to assess the appropriateness of targeted nutrient load reductions as well as provide indications of water quality patterns that will result from such management actions. The preliminary reevaluation report (Progress Report of the Baywide Nutrient Reduction Reevaluation, 1992) included the following conclusions: nonpoint sources of nutrients contributed approximately 77% of the nitrogen and 66% of the phosphorus entering the bay; agricultural sources were dominant followed by forest and urban sources; the "controllable" fraction of nutrient loads is about 47% for nitrogen and 70% for phosphorus; point source reductions are ahead of schedule and diffuse source reductions are close to projected reductions; further efforts are needed to reduce diffuse sources; significant reductions in phosphorus concentrations and slight increases in nitrogen concentrations have been observed in some areas of the bay; areas of low dissolved oxygen have been quantified and living resource water quality goals established; simulation model projections indicate significant reductions in low dissolved oxygen conditions associated with a 40% reduction of controllable nutrient loads.

During the latter part of 1997 the Chesapeake Bay Program entered another phase of re-evaluation. Since the last evaluation, programs have collected and analyzed additional information, nutrient reduction strategies have been implemented and, in some areas, habitat improvements have been accomplished. The overall goal of the 1997 re-evaluation was the assessment of the progress of the program and the implementation of necessary modifications to the difficult process of restoring water quality, habitats and living resources in Chesapeake Bay. During this portion of the program, EPC has been further modified to include intensive examination of SAV habitat conditions in several regions of the Chesapeake Bay in addition to retaining long-term monitoring of sediment processes in the Patuxent estuary.

At the present time the Chesapeake Bay Program is developing a revised agreement titled Chesapeake 2000 wherein the participants "*commit ourselves to nurture and sustain a Chesapeake Bay Watershed Partnership*" in order to achieve goals related to bay water quality, habitats and living resources and sound land uses. More specifically, this Agreement focuses on: 1) living resource protection and restoration; 2) vital habitat protection and restoration; 3) water quality restoration and protection; 4) sound land use and; 5) individual responsibility and community engagement. The current EPC of the Maryland Chesapeake Bay Water Quality Monitoring program has activities that are exactly aligned with goals two and three.

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2. SEDIMENT-WATER OXYGEN AND NUTRIENT EXCHANGES: (MINI-SONE and High Resolution Mapping)

R.M. Stankelis, W.R. Boynton, J.M. Frank and F.M. Rohland

2.1 Introduction and Background

More than a decade of study has shown that nutrient regeneration and release by sediments in many estuaries can be a significant internal source of nutrients to the water column (*e.g.* Boynton *et al*., 1995; Boynton *et al*., 1998). Moreover, this nutrient release or exchange has significant potential to negatively affect water quality and living resources. However, the utilization and regeneration of nutrients within an estuary is governed by processes that are both spatially and temporally variable. Therefore, in order to evaluate an estuary's response to changes (especially reductions) in external nutrient loading, it is important to collect data on appropriate spatial and temporal scales. While previous studies have shown that the highest nutrient releases by sediments occur during the summer months (Boynton *et al.*, 1988), less work has been done to evaluate the importance of spatial variability. Traditionally, sediment-water oxygen and nutrient exchange (SONE) measurements were made at a few fixed-location stations thereby providing only an indication of spatial gradients and variability. Yet, observed spatial variability does not remain fixed in time, and it is uncertain whether sediment-water exchanges (fluxes) at these fixed stations respond to changes in external nutrient loading in a linear or non-linear fashion.

The EPC has adopted new techniques that increase the spatial resolution of measurements in the Patuxent River. In 1996, six additional sediment-water exchange stations were added to four long-term stations to provide a better assessment of the range of conditions found within the Patuxent River estuary. In order to be cost effective, sediment-water exchanges at these new stations were measured with an abbreviated technique called MINI-SONE, in which a single sediment core was monitored instead of the traditional SONE technique, in which three replicate cores and a blank core were monitored and net fluxes calculated. Previous studies had shown that variation among replicate cores from a single location was insignificant compared to variation among sites. Therefore, it was believed additional stations would provide a more accurate assessment of sediment-water exchanges across the estuary as a whole, and thus more easily evaluate the river's response to nutrient management strategies.

In 1998 and 1999, traditional SONE measurements (with replication) were not made at the four long-term monitoring stations (BUVA [Buena Vista], MRPT [Marsh Point], BRIS [Broomes Island], and STLC [St. Leonard Creek]) on the Patuxent River. Instead, these stations were measured with the abbreviated MINI-SONE technique. These data were then merged with previous data sets for the calculation of status and trends at the four long-term monitoring stations. The other six MINI-SONE stations were monitored as usual in 1999.

While the addition of MINI-SONE measurements to the monitoring program significantly increased the ability to examine the full range of sediment-water fluxes on the Patuxent River, the development of statistical regression models has provided a cost effective way to more fully examine the spatial variability across an estuary. Based upon easily measured water and sediment parameters, these models can provide estimates of sediment-water fluxes at high spatial resolutions. These data can be compared to the long-term fixed monitoring stations and be used to evaluate whole estuary responses to changes in nutrient loading. With this in mind, high-resolution mapping of surficial sediment chlorophyll-*a* bottom water nutrient concentrations, and certain bottom water physical parameters was conducted at 37 stations (including the 10 MINI-SONE stations) across the mesohaline portion of the Patuxent River in 1999. Surficial sediment chlorophyll-*a* maps have been produced since 1996.

2.2 Station Locations

2.2.1 MINI-SONE and Long-term Patuxent River Station Locations

Ten MINI-SONE stations were chosen to represent the fullest possible range of sediment-water flux conditions found on the Patuxent River (Figure 2-1). Four of these stations, St. Leonard Creek (STLC), Broomes Island (BRIS), Marsh Point (MRPT) and Buena Vista (BUV) were previously monitored using the full suite of measurements referred to as Sediment-water Oxygen and Nutrient Exchanges (SONE) and are now referred to as the long-term monitoring stations. The remaining six MINI-SONE stations were PX07, PX15, PX21, PX23, PX25 and PX33.

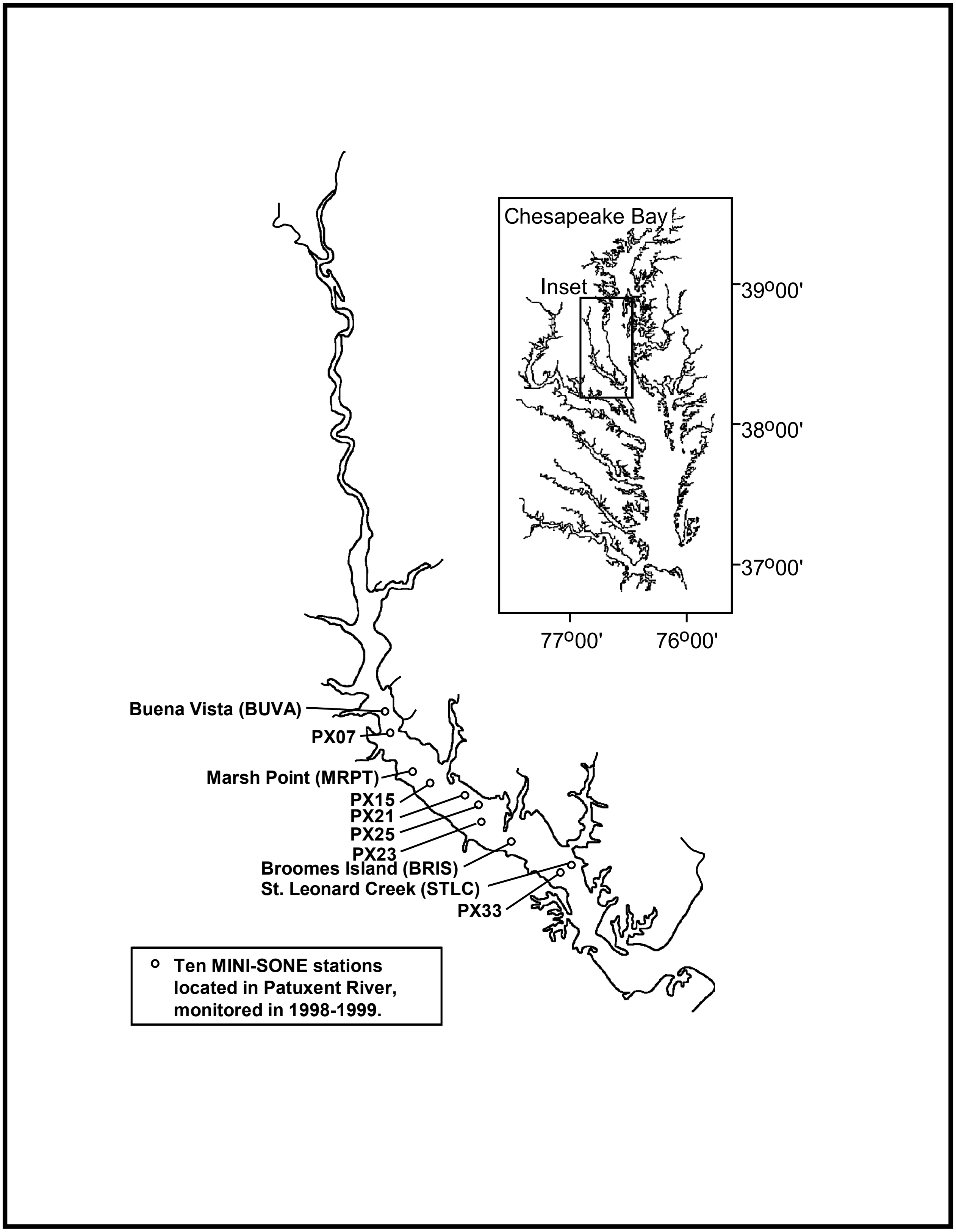

Figure 2-1. Location of ten MINI-SONE Stations sampled in the Patuxent River, Chesapeake Bay. *Relative locations of stations are shown in the Patuxent River and do not reflect exact geographic locations. Exact latitude and longitude are given in Rohland et al. (1999).*

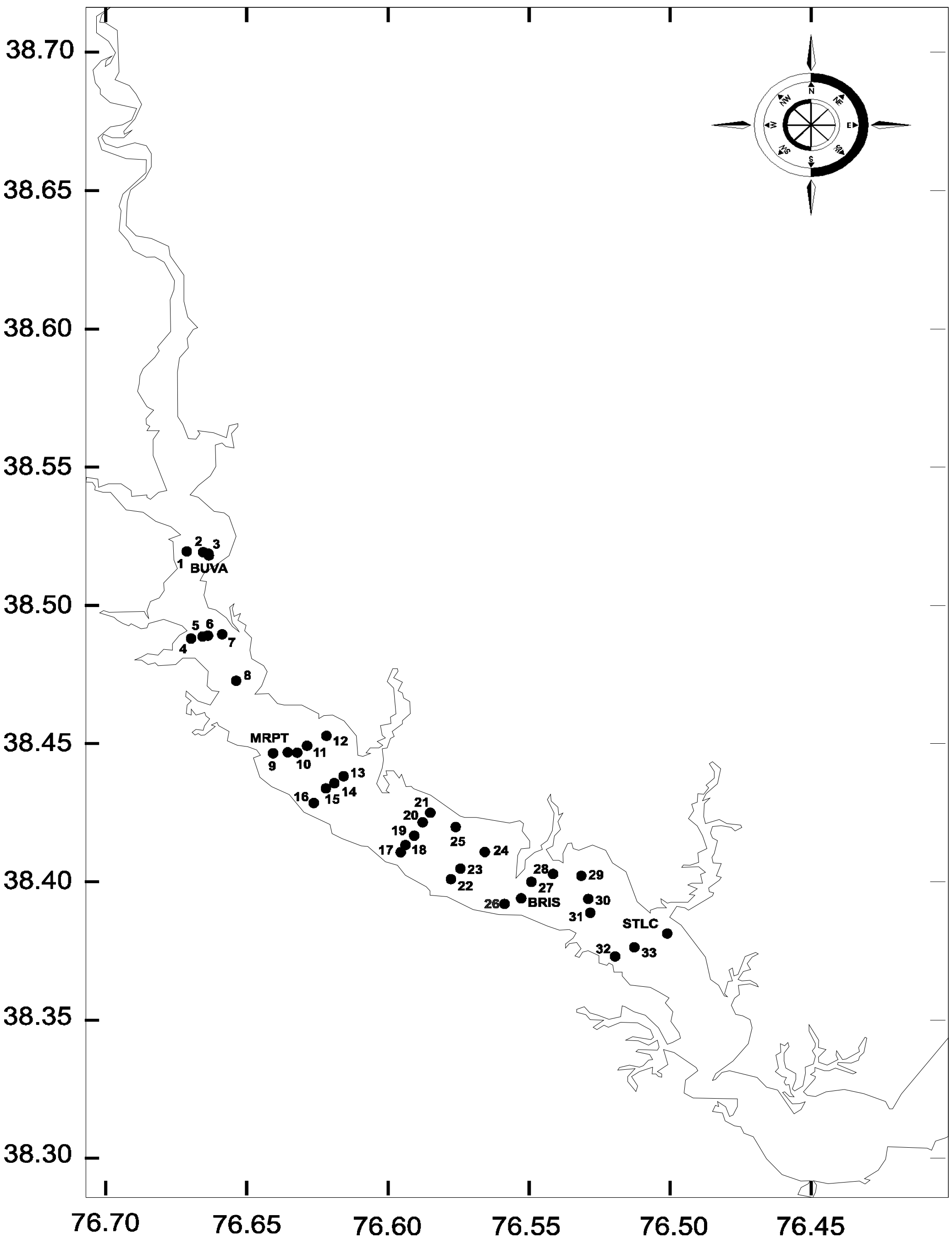

Figure 2-2. Location of 37 stations in the Patuxent River used in the High Resolution Sediment and Bottom Water Mapping.
Latitude and longitude are in decimal degrees. Relative locations of stations are shown in the Patuxent River and do not reflect exact geographic locations. Exact latitude and longitude are given in Rohland et al. (1999).

2.2.2 High Resolution Sediment and Bottom Water Mapping Locations

Thirty seven (37) stations were sampled in the Patuxent River, representing a salinity and depth range between the most upriver station BUVA (Buena Vista) and the most down river station STLC (St. Leonard Creek, Figure 2-2).

2.3 Sampling Frequency

2.3.1 Sampling Frequency for MINI-SONE

The sampling frequency for MINI-SONE is based on the seasonal patterns of sediment-water exchanges observed in previous studies conducted in the Chesapeake Bay region (Kemp and Boynton, 1980, 1981; Boynton *et al.*, 1982; and Boynton and Kemp, 1985). These studies indicated four distinct periods during an annual cycle (Boynton *et al.*, 1998). Previous studies also indicate that short-term temporal (day-month) variation in these exchanges is small; however, considerable differences in the magnitude and characteristics of fluxes appear among distinctively different estuarine zones (*i.e.*, tidal fresh *vs.* mesohaline regions). In light of these results, the monitoring design adopted for MINI-SONE studies involved four monthly measurements made between June and September, 1999.

2.3.2 Sampling Frequency for High Resolution Sediment and Bottom Water Mapping

The high resolution mapping stations were sampled monthly during May through September 1999. These months were chosen as optimum times to document the spatial distribution and magnitude of labile organic material deposited to sediments during and following spring and summer algal bloom. Bottom water nutrient concentrations, dissolved oxygen, and temperature, along with sediment redox potential (Eh) were also measured at these stations.

2.4 Field Methods for MINI-SONE and High Resolution Sediment and Bottom Water Mapping

A detailed description of all field and laboratory methods can be found in the Ecosystems Processes Component Quality Assurance Plan-FY2000 (Rohland *et al*., 1999).

2.5 Results

Since estuarine ecosystems are both temporally and spatially variable, an accurate evaluation of ecosystem status and response to nutrient loading require data on the appropriate spatial and temporal scales. Since 1996, the EPC has increased the spatial coverage of monitoring of the Patuxent River to include 10 MINI-SONE stations, and 37 high resolution sediment and bottom water mapping stations. This increased spatial coverage allows the calculation of estuary-wide estimates of certain parameters and comparison of differences in these estimates using various subsets of the data. The assumption is that integrated estuary-wide estimates are more accurate for evaluating aspects of estuarine health because these estimates take into account spatial variability. In this way, it is possible to evaluate how well the four long-term monitoring stations (BUVA, MRPT, BRIS and STLC) actually reflect the estuary as a whole. As the data base increases over the years relationships and ecosystem linkages will become apparent.

2.5.1 River Flow

In the Patuxent River and in many other estuaries, river flow is often a good estimate of point source nutrient loading. For this reason, river flow is an important external forcing function, influencing both temperature and salinity patterns, circulation as well as nutrient loading rates. However, not only is the magnitude of river flow important to understanding patterns, but also the timing of flow events that affect nutrient uptake and utilization. Therefore, an examination of inter-annual and monthly flow patterns helps explain variation in estuarine processes such as sediment-water exchanges. Annual average Patuxent river flow was 280 cfs in 1999, 437 cfs in 1998, 412 cfs in 1997 and 704 in 1996; all but 1999 were higher than the long-term average of 378 cfs (Figure 2-3). While average annual river flow during 1996-1998 was higher than the long-term (21-year) average (and 1999 a drought year with a September hurricane), the patterns of monthly average river flow differed significantly among the years.

During 1999 peak late-winter and spring flows were hardly equal to the long-term annual average, indicative of a substantial drought. If the September hurricanes did not occur, flow for 1999 would have been among the lowest on record. In 1998 and 1997 peak monthly river flow occurred in March (1131 and 875 cfs respectively), while in 1996, peak flow occurred in December (1357 cfs, Figure 2-3). Because many estuarine processes respond to nutrient loading on time scales of weeks to months, the timing of flow events can be an important consideration. For example, Patuxent river flow was higher during the spring of 1998 compared to 1997 or 1996. This resulted in significantly higher sediment-water fluxes at many locations within the river in 1998 compared to 1997, yet average annual flow was not very different between the years (437 cfs and 412 cfs respectively). In addition, differences in flow also affect the spatial variation found in the river. High flow conditions tend to transport important processes, such as the chlorophyll-*a* maximum, down river compared to lower flow years (Boynton and Kemp, 2000). This may also affect the deposition of labile material to the sediment surface, which in turn affects sediment-water exchanges.

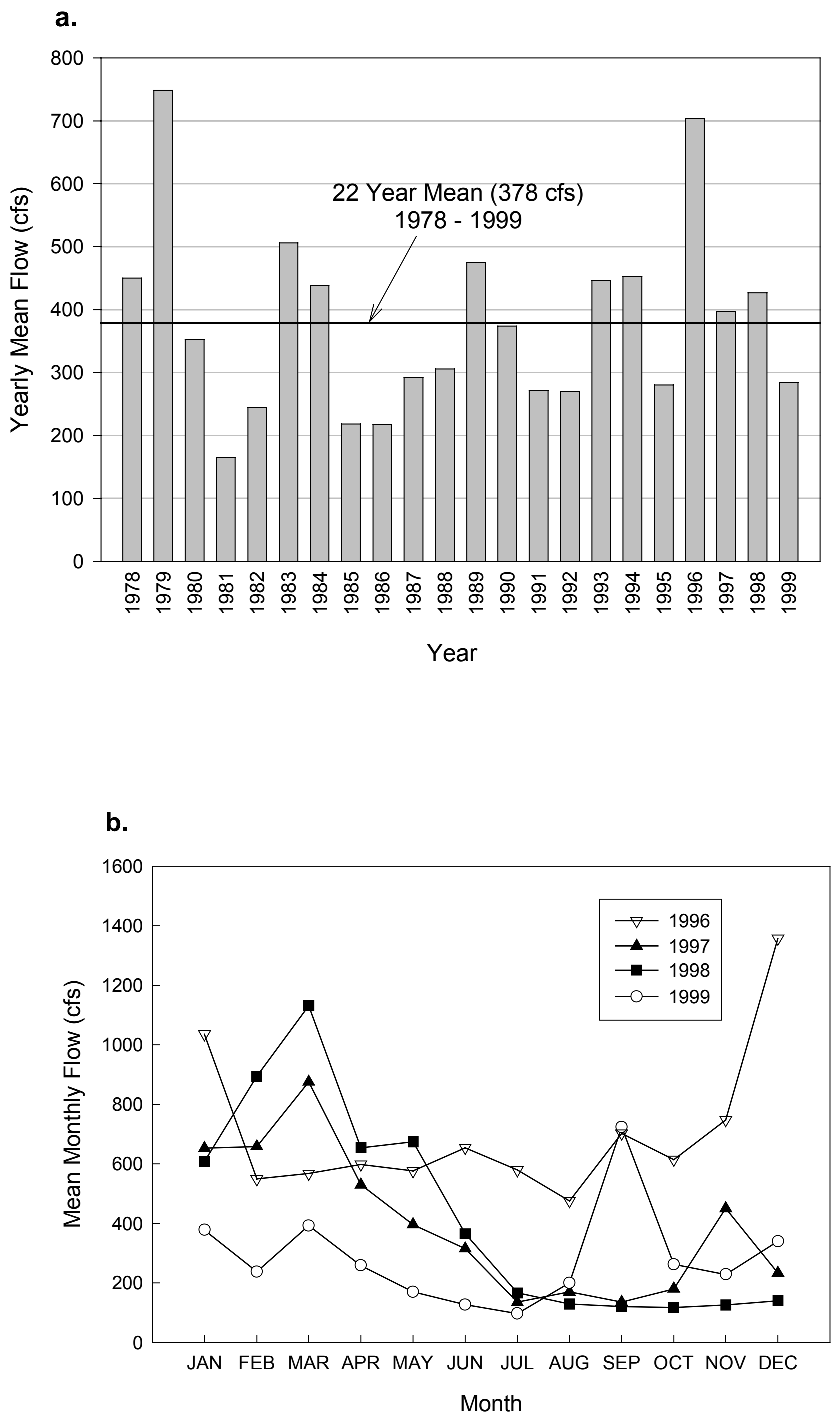

Figure 2-3. (a) Patuxent River average annual river flow for the period 1978 through 1999, (b) Patuxent River average monthly river flow from 1996 through 1999.

2.5.2 Surficial Sediment Chlorophyll-*a*

A number of studies have shown that sediment-water nutrient and oxygen exchanges (fluxes) are responsive to the amount of labile material deposited to the sediment surface (*e.g.* Boynton *et al*., 1992; Garber *et al*., 1989). The use of surficial sediment chlorophyll-*a* as an index of labile organic material has proven useful for predicting certain sediment-water exchanges (*e.g.* Cowan and Boynton, 1996; Cowan *et al*., 1996; Boynton *et al*., 1998, 1999). In fact, an analysis of Patuxent River MINI-SONE flux data from 1996 through 1999 has shown that surficial sediment chlorophyll-*a* concentrations measured approximately one month prior to sediment-water fluxes are highly correlated with several sediment-water exchanges (See Section 2.5.5.3). In addition, significant monthly variation in the standing stock of chlorophyll-*a* on the sediment surface indicates that deposition of organic matter is a continuing process, and that increases in standing stock can occur throughout the summer when deposition exceeds decomposition (Boynton *et al*., 1998). For example, in 1997 the highest mean sediment chlorophyll-*a* concentrations were found in September (Figure 2-4.a).

2.5.2.1 Temporal Variation

The amount of organic material deposited to the sediment surface often varies significantly both seasonally and annually. The pattern of monthly variation in sediment chlorophyll-*a* concentrations observed in 1999 followed typical estuarine depositional models with maximum concentrations highest in the spring and lowest in the summer (Figure 2-4.a). However, the difference between spring and summer concentrations was not as great as in past years. For example, the maximum sediment total chlorophyll-*a* concentration in May 1999 was 146.1 mg m^{-2}, while the maximum concentration in May 1998 was 386.6 mg m^{-2}. Minimum chlorophyll-*a* concentrations were similar for both years. This variation is likely due in large part to differences in winter-spring river flow between these years. A plot of May sediment chlorophyll-*a* concentrations versus winter-spring river flow shows a striking positive relationship that explains 86% of the annual variation assuming a linear relationship (Figure 2-4.b).

2.5.2.2 Spatial Variation

In addition to seasonal and yearly variation in sediment chlorophyll-*a* concentrations in the Patuxent River, spatial variation can also be an important feature. A visual representation of this variation was created by constructing contour maps using *Surfer* contouring software, and data from the 37 mapping stations. As in past years, data were interpolated to a uniform grid of 0.002 degrees longitude and latitude, and maps were plotted with the same contouring intervals. Because chlorophyll-*a* concentrations were much lower in 1999 compared to previous years, these maps do not show the subtle variation in sediment chlorophyll-*a* concentrations that were actually found within the river. However, they do provide a useful comparison to maps from past years.

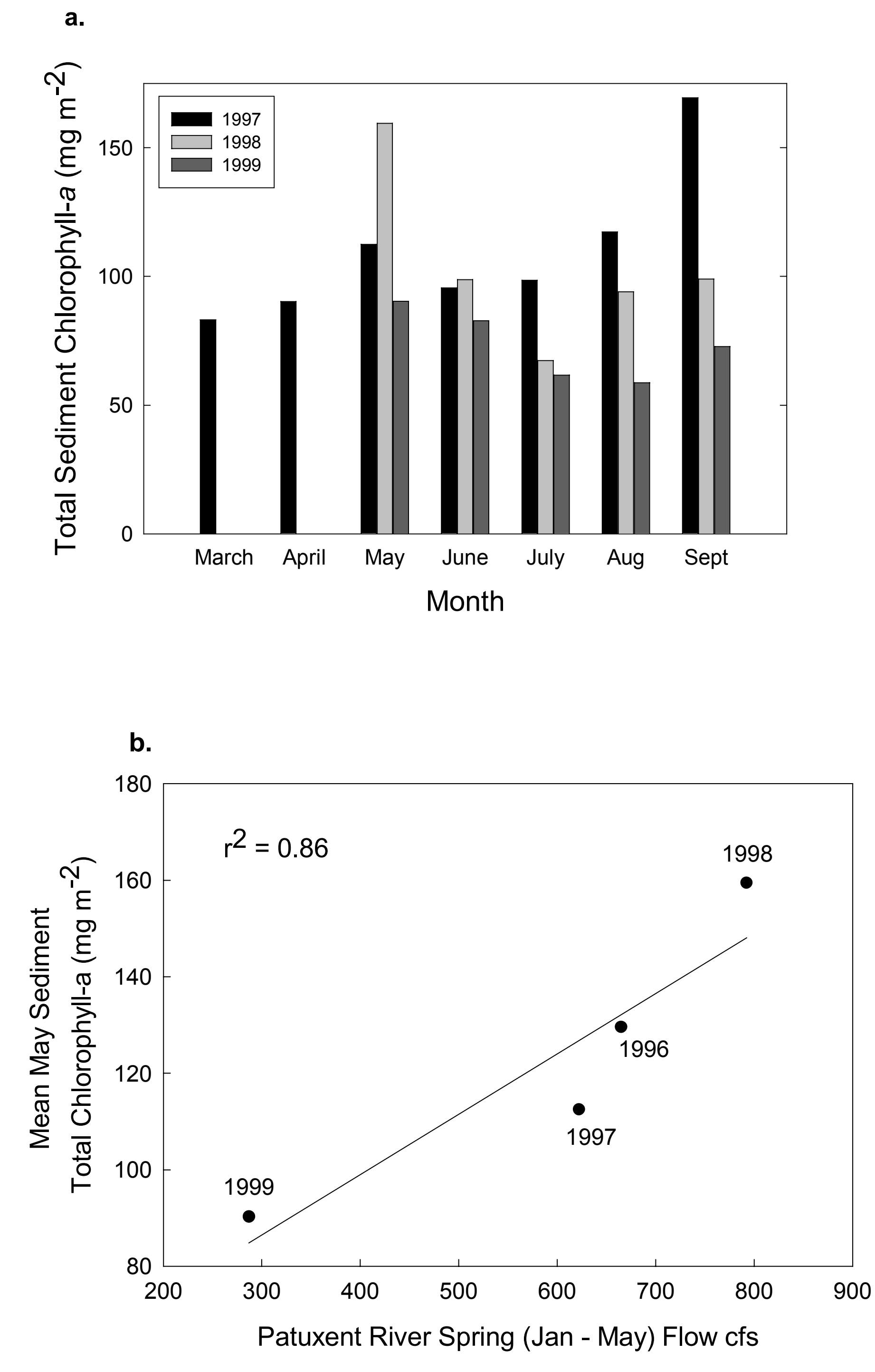

Figure 2-4. a. Mean, monthly total sediment chlorophyll-*a* concentration (collected to 1 cm depth) from 37 mapping stations on the Patuxent River during 1997 through 1999, and b. May total sediment chlorophyll-*a* concentrations vs. Patuxent River spring flow.

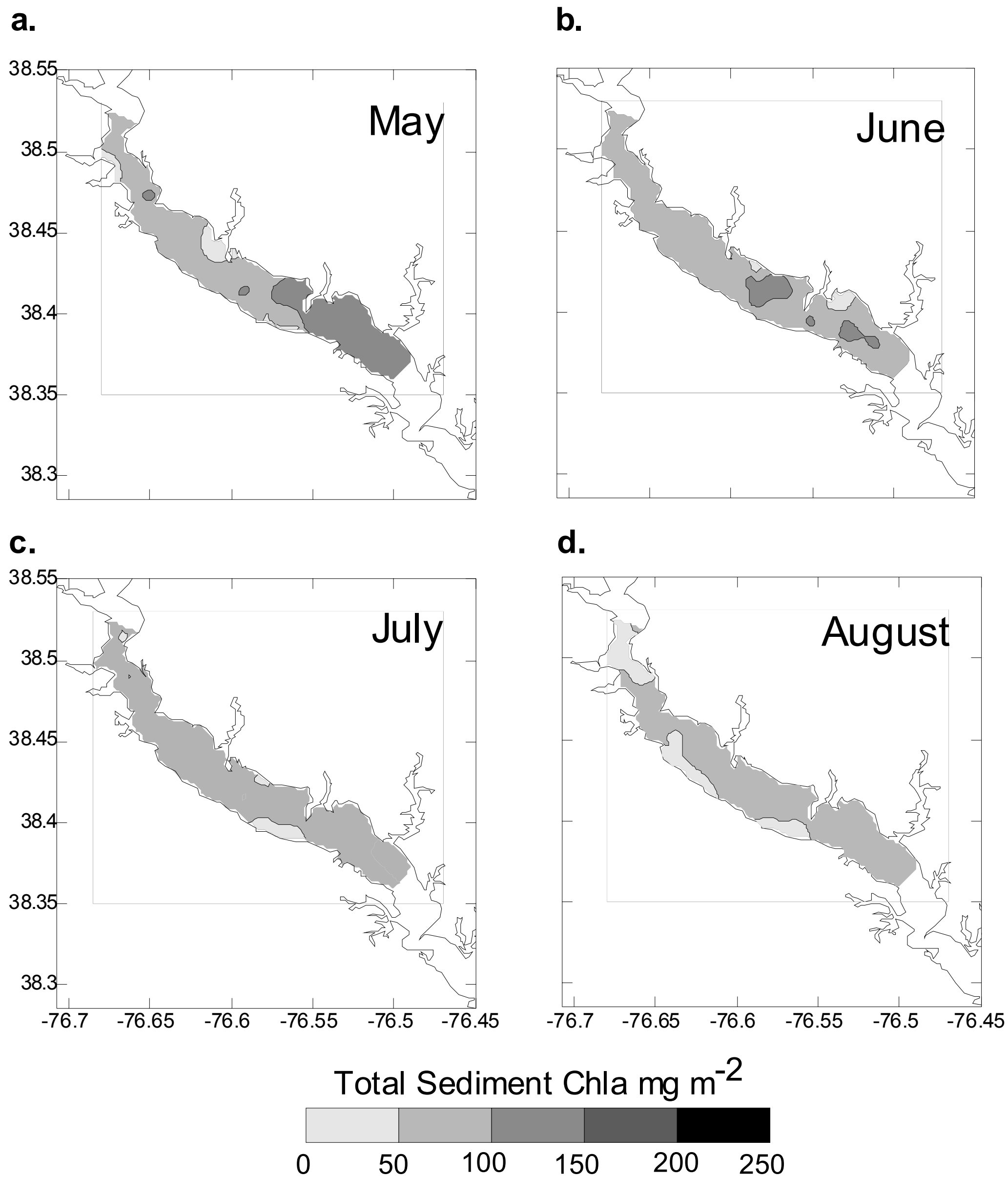

Figure 2-5. Sediment total chlorophyll-*a* concentrations contoured within the study area based on samples collected from 37 mapping stations during (a) May, (b) June, (c) July, and (d) August.

2.5.3 MINI-SONE Sediment-Water Oxygen and Nutrient Fluxes: 1999 Patuxent River Study

Monthly average sediment-water fluxes derived from the complete sediment-water oxygen and nutrient exchanges (SONE) data set are summarized using box and whisker plots (Figures 2-6.1 through 2-6.4) for four flux variables: sediment oxygen consumption (SOC), ammonium (NH_4^+), nitrite plus nitrate (NO_2^- + NO_3^-), and phosphate (PO_4^-). Data collected at four stations in the Patuxent River were used to construct these plots. Two stations, Buena Vista (BUVA) and St Leonard Creek (STLC) were sampled during a period of thirteen calendar years (1985 through 1997) while the remaining two stations, Marsh Point (MRPT) and Broomes Island (BRIS), were sampled during a shorter period of nine years (1989 through 1997). The order of the four stations in these figures reflects their spatial position in the Patuxent River from the turbidity maximum zone (Buena Vista [BUVA]) to the middle regions of the estuary (Marsh Point [MRPT] and Broomes Island [BRIS]) to the estuary mouth (St. Leonard Creek [STLC]). Superimposed on these graphs are the single MINI-SONE flux measurements made at these four stations during 1999.

Construction of the box and whisker plot, a derivation of the original Tukey (1977) box graph, follows the method used in the SAS procedure (SAS, 1988; PROC UNIVARIATE PLOT). The bottom and top edges of the box are located at the sample 25th and 75th percentiles. The center horizontal line is drawn at the sample median and the central plus sign (+) is at the sample mean. The central vertical lines, "whiskers", extend from the box as far as the data extends or to a distance of at most 1.5 interquartile ranges, where an interquartile range is the distance between the 25th and the 75th sample percentiles. Any value more extreme than this is marked with a zero (0) if it is within three interquartile ranges of the box, or with an asterisk (*) if it is still more extreme. The width of each box is proportional to the total number of samples collected at each station and used in the analysis.

2.5.3.1 Sediment Oxygen Consumption (SOC)

The magnitude of 1999 SOC observations was generally similar to those observed in previous years. However, there were several important differences between 1999 SOC data and earlier years. Specifically, at stations where bottom water dissolved oxygen concentrations tend to be depressed during summer months, SOC rates are also generally depressed, as expected due to the influence of low dissolved oxygen concentrations (< 2.0 mg l^{-1}) on SOC rates. However, 1999 was a drought year, especially prior to the September hurricanes, and in these years of very low river flow, dissolved oxygen concentrations in deep waters tend to be more elevated than usual. Relatively elevated summer bottom water dissolved oxygen conditions result from a complex interaction between water column stratification (less in years of low flow thereby allowing for more atmospheric reaeration of bottom waters via mixing) and more limited amounts of organic matter reaching deep waters and sediments (because of reduced nutrient delivery from diffuse sources and hence lower rates of algal biomass accumulation). For example, SOC rates were elevated in June and July at MRPT and at BRIS in July and August. Both stations also exhibited higher than normal dissolved oxygen concentrations in bottom waters (> 1.0 mg l^{-1}). In addition, SOC rates were not markedly higher at BUVA than at other Patuxent River stations during 1999

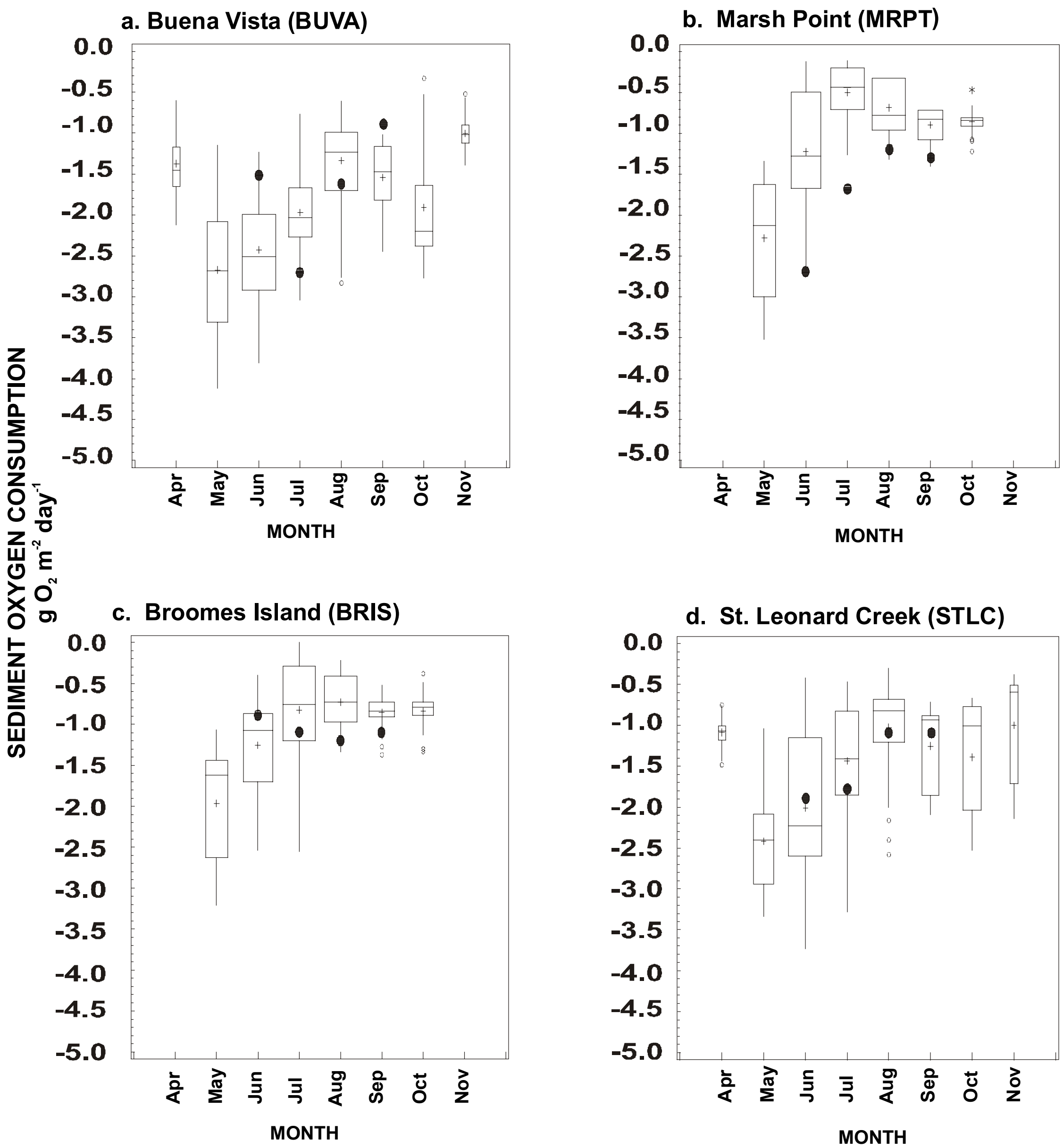

Figure 2-6.1. Box and whisker plots for sediment oxygen consumption (SOC) rates for April to November at four SONE stations located in the Patuxent River.
(a) Buena Vista [BUVA] (b) Marsh Point [MRPT] (c) Broomes Island [BRIS] and (d) St. Leonard Creek [STLC].

The complete SONE flux data set was used to produce the graph. Monthly values at Broomes Island (BRIS) and Marsh Point (MRPT) are based on data from 1989 through 1997. September values for all stations only include six years of data (1991 through 1997). The bold solid dots indicate a single flux measured during the MINI-SONE study 1999. Negative values indicate fluxes from water to sediment. Occasionally hypoxic stations are Broomes Island (BRIS) and Marsh Point (MRPT). Hypoxia is defined here as less than 1.0 mg l^{-1} dissolved oxygen in bottom waters.

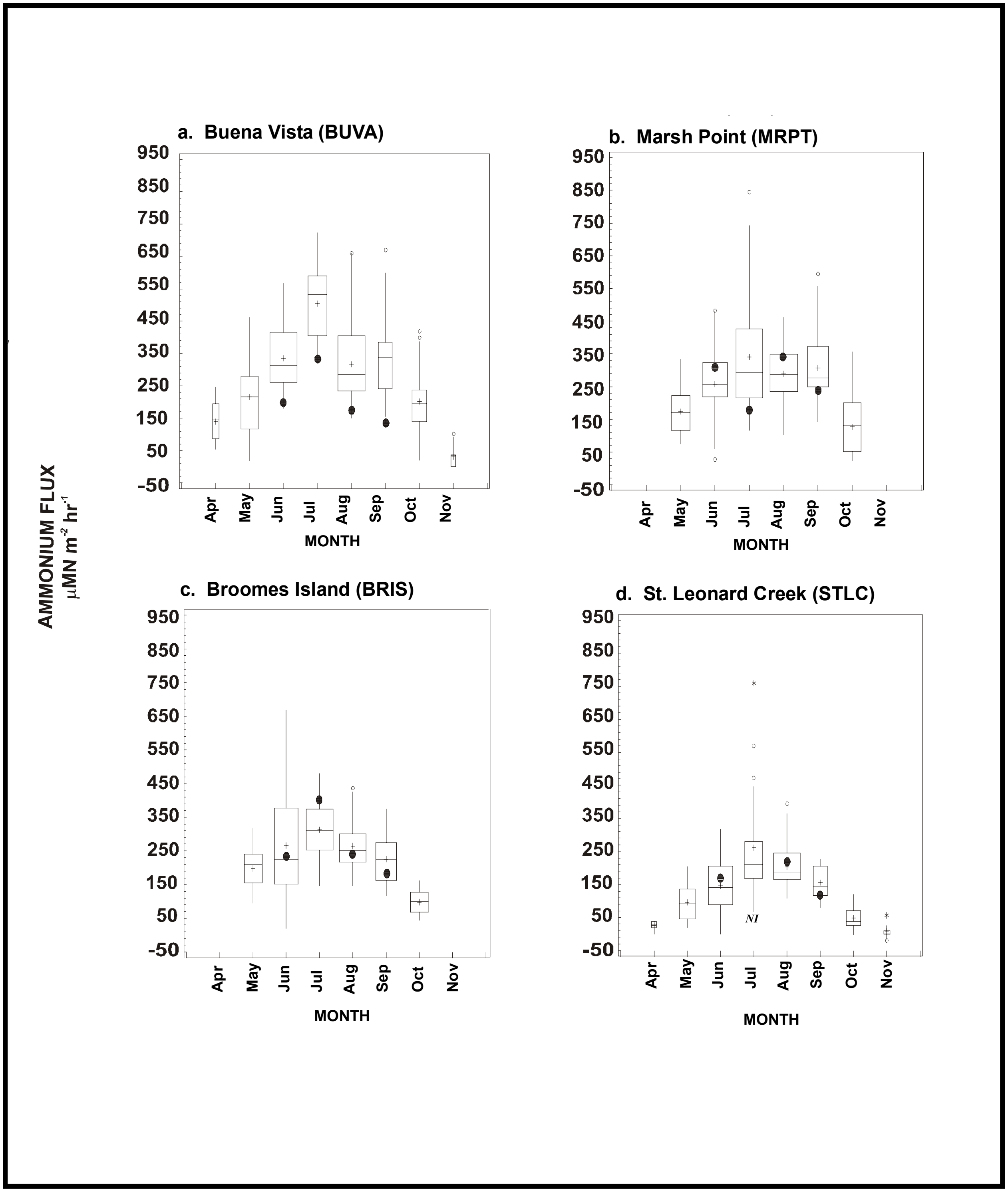

Figure 2-6.2. Box and whisker plots for ammonium (NH_4^+) flux rates for April to November at four SONE stations located in the Patuxent River.
(a) Buena Vista [BUVA] (b) Marsh Point [MRPT] (c) Broomes Island [BRIS] and (d) St. Leonard Creek [STLC].

The complete SONE flux data set was used to produce the graph. Monthly values at Broomes Island (BRIS) and Marsh Point (MRPT) are based on data from 1989 through 1997. September values for all stations only include six years data (1991 through 1997). The bold solid dots indicate a single flux measured during the MINI-SONE study 1999. Negative values indicate fluxes from water to sediment. Occasionally hypoxic stations are Broomes Island (BRIS) and Marsh Point (MRPT). Hypoxia is defined here as less than 1.0 mg l^{-1} dissolved oxygen in bottom waters.

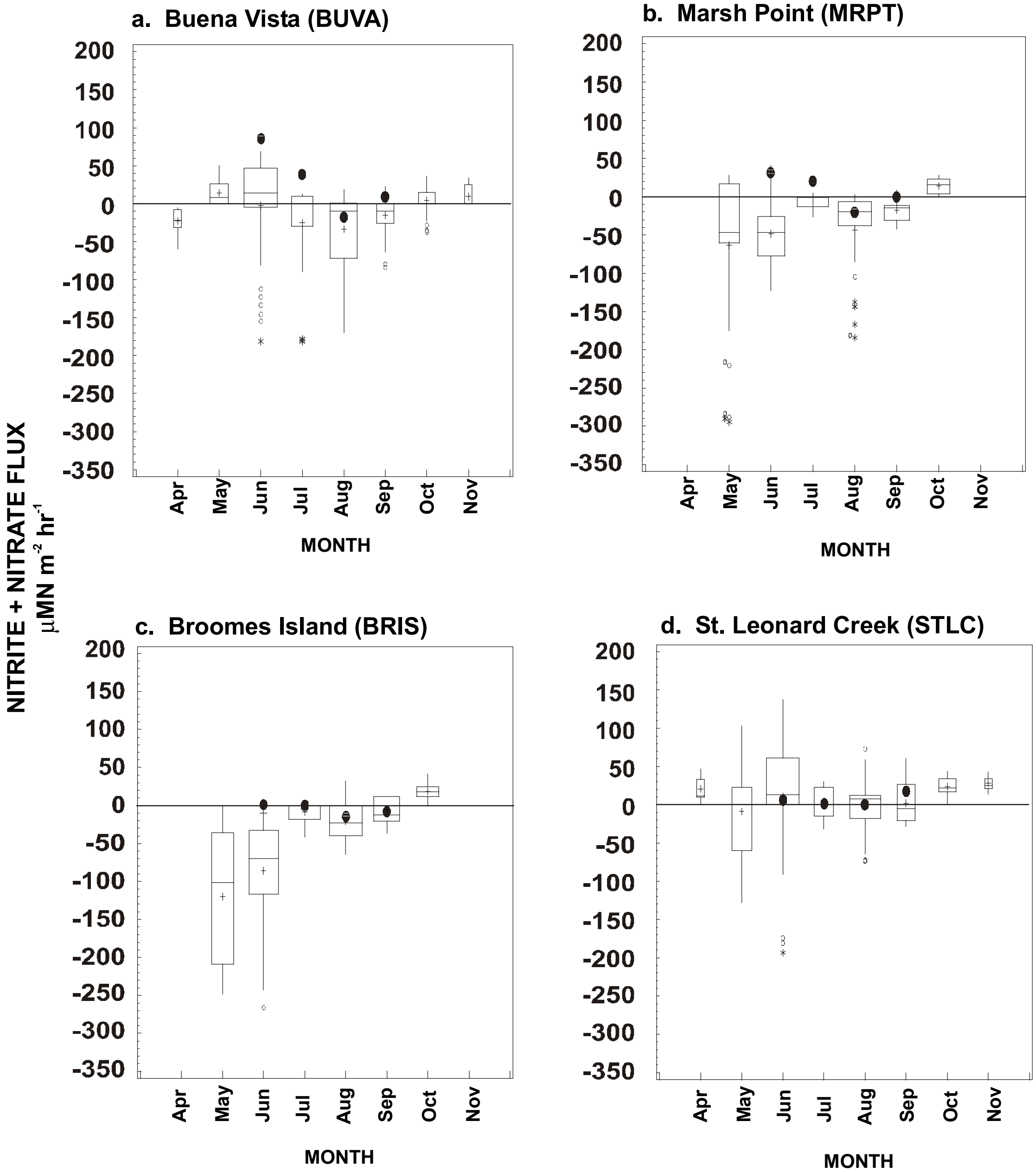

Figure 2-6.3. Box and whisker plots for nitrite plus nitrate (NO_2^- + NO_3^-) flux rates for April to November at four SONE stations located in the Patuxent River.
(a) Buena Vista [BUVA] (b) Marsh Point [MRPT] (c) Broomes Island [BRIS] and (d) St. Leonard Creek [STLC].

The complete SONE flux data set was used to produce the graph. Monthly values at Broomes Island (BRIS) and Marsh Point (MRPT) are based on data from 1989 through 1997. September values for all stations only include six years data, (1991 through 1997). The bold solid dots indicate a single flux measured during the MINI-SONE study 1999. Negative values indicate fluxes from water to sediment. Occasionally hypoxic stations are Broomes Island (BRIS) and Marsh Point (MRPT). Hypoxia is defined here as less than 1.0 mg l^{-1} dissolved oxygen in bottom waters.

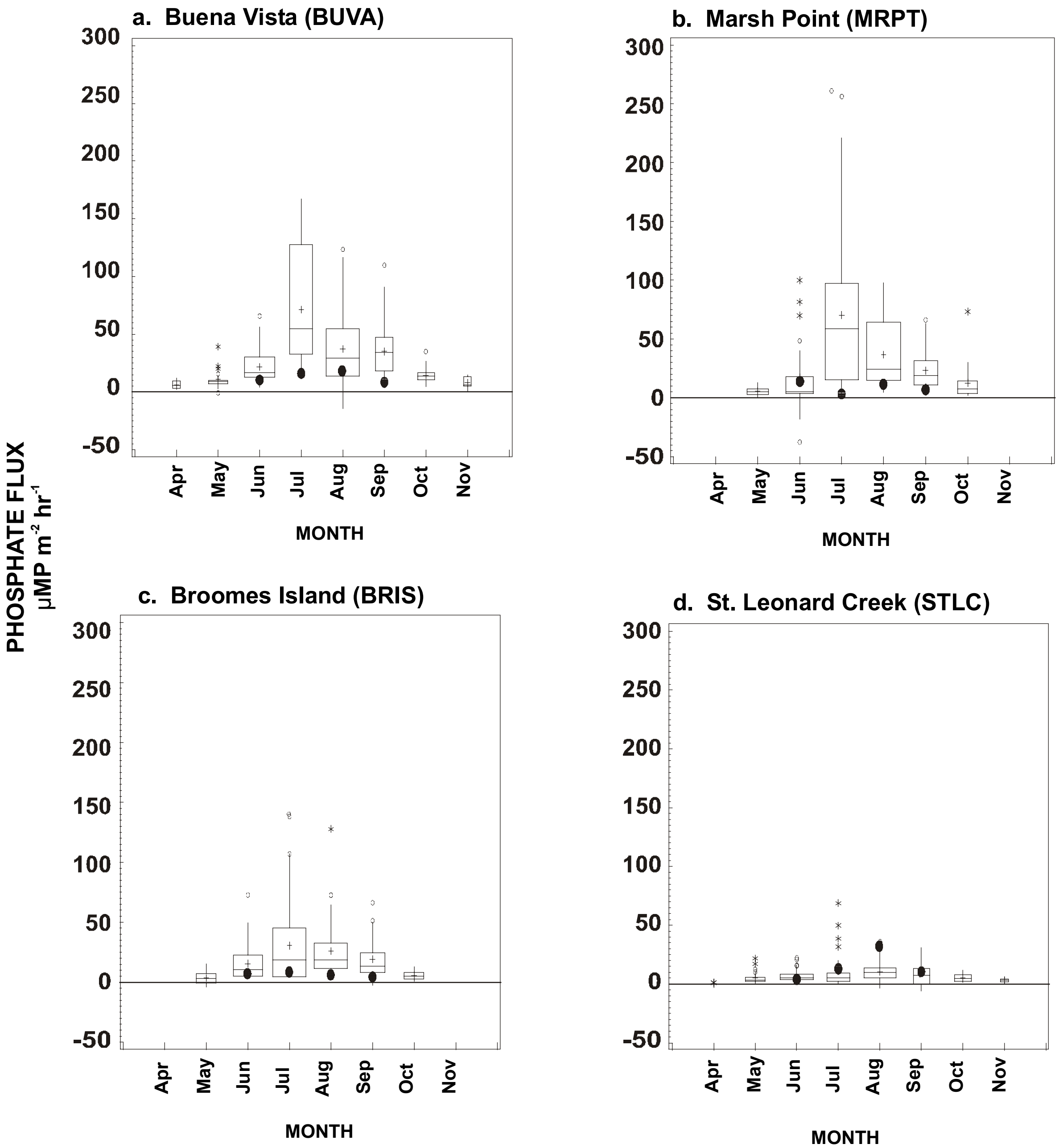

Figure 2-6.4. Box and whisker plots for phosphorus (PO_4^{-3} or DIP) flux rates for April to November at four SONE stations located in the Patuxent River.
(a) Buena Vista [BUVA] (b) Marsh Point [MRPT] (c) Broomes Island [BRIS] and (d) St. Leonard Creek [STLC].

The complete SONE flux data set was used to plot the graph. Monthly values at Broomes Island (BRIS) and Marsh Point (MRPT) are based on data from 1989 through 1997. September values for all stations only include six years data (1991 through 1997). The bold solid dots indicate a single flux measured during the MINI-SONE study 1999. Negative values indicate fluxes from water to sediment. Occasionally hypoxic stations are Broomes Island (BRIS) and Marsh Point (MRPT). Hypoxia is defined here as less than 1.0 mg l^{-1} dissolved oxygen in bottom waters.

rates at this station are usually high) despite the fact that this site has well oxygenated bottom waters (the water column at this site is well mixed) that promote enhanced SOC rates and a well developed benthic macroinvertebrate community (which also contributes to SOC). The particularly low rates observed during June and September, 1999 may well be related to organic matter limitation due to low nutrient loads and subsequent low rates of algal biomass delievery to sediments.

In general, sediment oxygen consumption (SOC) rates at these stations in the Patuxent River tend to be higher (more negative) in low flow years which, in turn, tend to be years in which bottom water dissolved oxygen concentrations remain elevated. The chain of coupled processes leading to this result probably includes the following. In low flow years diffuse source nutrient loads are reduced because of lower run off. This leads to smaller algal blooms and reduced deposition of organic matter to bottom sediments. In addition, lower river flow leads to less developed vertical stratification and hence the opportunity for more mixing of oxygen-rich surface waters with deeper waters. The combination of reaeration from surface waters and modest supplies of organic matter to support respiratory processes results in dissolved oxygen concentrations being maintained at levels > 1 mg l^{-1} (at which SOC can continue to occur).

2.5.3.2 Ammonium (NH_4^+) Fluxes

Ammonium fluxes recorded in 1999 generally followed temporal trends exhibited in previous years. Fluxes tended to peak in July (early summer) and decline during the latter portion of the summer. The general magnitude of ammonium fluxes during 1999 tended to be at or below the long-term mean at all Patuxent stations. Of the 16 fluxes measured at these sites during 1999 (*i.e.*, four sites x four monthly measurements), only one was well above the long-term mean value. Fluxes were particularly small at BUVA and MRPT, the two stations most proximal to the fall line nutrient sources. Ammonium fluxes were closer to average at the more down river stations (BRIS and STLC). Decreased ammonium fluxes suggests a decrease in the organic matter supply rate to sediments which probably reflects lower than normal nutrient loading rates during the very low flow winter and spring of 1999. In addition, DO concentrations in deep waters were unusually elevated in July. High bottom water DO promotes lower NH_4^+ fluxes in part because nitrification is promoted under higher DO conditions.

2.5.3.3 Nitrite + Nitrate (NO_2^- + NO_3^-) Fluxes

In general, nitrate fluxes do not constitute a large fraction of the nitrogen exchange between estuarine sediments and bottom waters. On occasion, large fluxes from water to sediments or from sediments to water do occur. However, no large fluxes from sediment to water or water to sediments were observed during 1999. Fluxes during June and July at BUVA and MRPT were more positive than usual. However, even small nitrate fluxes from sediments to overlying waters provide a useful indication of sediment conditions. Specifically, production and release of nitrate from sediments is a strong indication that sediment nitrification is occurring. This process requires at least low levels of dissolved oxygen and is hence an indication that surface sediments have been in contact with oxygenated waters. During 1997 (a low flow year), ten of 16 nitrite plus nitrate flux measurements were more positive than the long-term average, indicating good sediment quality conditions. During 1998 (a wet spring) only 5 of 16 flux measurements were indicative of sediment nitrification. To provide addition contrast, during 1996 (an exceptionally high flow year) the overwhelming pattern was nitrite plus nitrate flux ($NO_2^- + NO_3^-$) from water to sediments which was to be expected during a wet year when water column nitrate concentrations were high. During 1995, a very low flow year, stations in the Patuxent River exhibited relatively high rates of sediment nitrate release or much lower rates of nitrogen uptake. In fact, at the St. Leonard Creek (STLC) station sediments released nitrate through the entire monitoring period, a pattern never before observed. During 1999 (another very dry year) nitrite plus nitrate fluxes were predominately positive (12 of 16 fluxes were from sediments to water). These are the types of nitrate fluxes to be expected under reduced nutrient load conditions (this was the case in 1995 and 1999) both because these conditions favor improved dissolved oxygen conditions in deep waters and sediments and lower concentrations of nitrate in overlying waters. The direction and magnitude of nitrite plus nitrate fluxes between sediments and overlying waters appears to serve quite well as an indicator of sediment quality.

2.5.3.4 Dissolved Inorganic Phosphorus (PO_4^{-3} or DIP) Fluxes

The spatial and temporal patterns of phosphorus flux in the Patuxent River in 1999 are consistent with the conceptual model of factors controlling these fluxes. With one minor exception, phosphate releases from sediments were at or well below median fluxes at all stations during the summer of 1999. At both BUVA and MRPT fluxes were very much reduced and were at record low levels at MRPT in July. During 1999 very low phosphate fluxes were observed at stations having modest to high dissolved oxygen concentrations in bottom waters, emphasizing the strong control dissolved oxygen concentrations have on phosphorus releases from sediments. When bottom water dissolved oxygen concentrations are even somewhat elevated (>1.5 mg l^{-1}) phosphorus is bound by iron oxides at the sediment surface and is not released to overlying waters.

It may be premature to conclude that reduced phosphorus inputs from point and diffuse sources is the cause of the pattern observed in the Patuxent River but the pattern observed during 1999 (and 1995, another low flow year) is consistent with this line of reasoning.

2.5.4 Sediment-water Exchanges (MINI-SONE stations)

In 1999, as predicted, sediment-water fluxes varied considerably among all MINI-SONE stations. To a great extent this result is a direct consequence of the selection and placement of MINI-SONE stations within this portion of the estuary. In fact, these stations were selected to measure the widest possible range of conditions present in the lower Patuxent River. However, despite this variation, several patterns have emerged that are worth noting.

Overall the mean ammonium flux in 1999 (230.3 μM N m^{-2} hr^{-1}) was lower than 1998 (297.8 μM N m^{-2} hr^{-1}) although not statistically significant (paired t-test, P = 0.57, Figure 2-7.b). At eight of ten stations, ammonium flux was lower in 1999 compared to 1998 and was likely due to differences in river flow and nutrient loading to the system between years. Not only does river flow affect the magnitude of these fluxes, but it also affects variation among stations. For example, in 1998 (which was a high flow year compared to 1999) the standard error for ammonium flux among stations was 63.6 µM N m^{-2} hr^{-1} compared to 42.1 µM N m^{-2} hr^{-1} in 1999. Even with reduced variation in ammonium flux among stations, more than a ten-fold difference (469 µM N m^{-2} hr^{-1} at PX33 and 41.9 µM N m^{-2} hr^{-1} at PX25) was still observed among stations during 1999. While the relative ranking of ammonium flux among stations was not completely consistent between 1998 and 1999, those stations with very low ammonium fluxes in 1998 were also low in 1999.

Sediment oxygen consumption (SOC) also varied considerably among stations in 1999. The maximum mean SOC value of -2.03 mg O_2 m^{-2} day^{-1} was found at station PX07 while the minimum value -0.77 mg O_2 m^{-2} day^{-1} was found at station PX15 (Figure 2-7.a). In general, the approximate ranking of SOC rates among stations was similar in 1999 compared to 1998. For example, those stations with high SOC rates in 1998 were also generally high in 1999, and those stations with low SOC rates in 1998 were low in 1999. While overall, the magnitude of SOC rates was larger in 1999 (- 1.388 mg O_2 m^{-2} day^{-1}) compared to 1998 (-1.293 mg O_2 m^{-2} day^{-1}) there was no significant difference between years (paired t-test, $P = 0.57$).

Combined nitrite plus nitrate (NO_2^- + NO_3^-) mean flux among MINI-SONE stations varied considerably in 1999 with some stations releasing nitrite plus nitrate to the water column, while other stations taking it up. Summer mean values ranged from a maximum positive flux of 37.68 µM N m^{-2} hr^{-1} out of the sediment at station PX07 to a minimum of –9.30 µM N m^{-2} hr^{-1} into the sediment at PX15 (Figure 2-7.c). Although the magnitude of sediment-water flux at these stations differed greatly between years, the direction of sediment-water flux at most stations remained the same. Taking all stations into consideration, mean nitrite plus nitrate flux was more positive (out of the sediment) in 1999 (+ 9.109 µM N m^{-2} hr^{-1}) compared to 1998 (-8.028 µM N m^{-2} hr^{-1}; paired t-test, $P < 0.05$). This result was not surprising considering lower than normal river flow and higher than average bottom water dissolved oxygen conditions were found in 1999.

Mean phosphate (PO_4^{-3}) flux among stations was significantly lower in 1999 compared to 1998 ($P < 0.05$) and was among the lowest on record. This was likely the result of lower than average river flow and loading to the estuary. The maximum mean phosphate (PO_4^{-3}) flux was 36.55 µM P m^{-2} hr^{-1} at station PX33, while the minimum mean flux was 0.72 µM P m^{-2} hr^{-1} at station PX23 (Figure 2-7.d). However, the ranking of phosphate flux among stations was not consistent between 1998 and 1999. For example, phosphate flux at station PX23 was the highest 1998, yet was the lowest value found in 1999 (Figure 2-7.d).

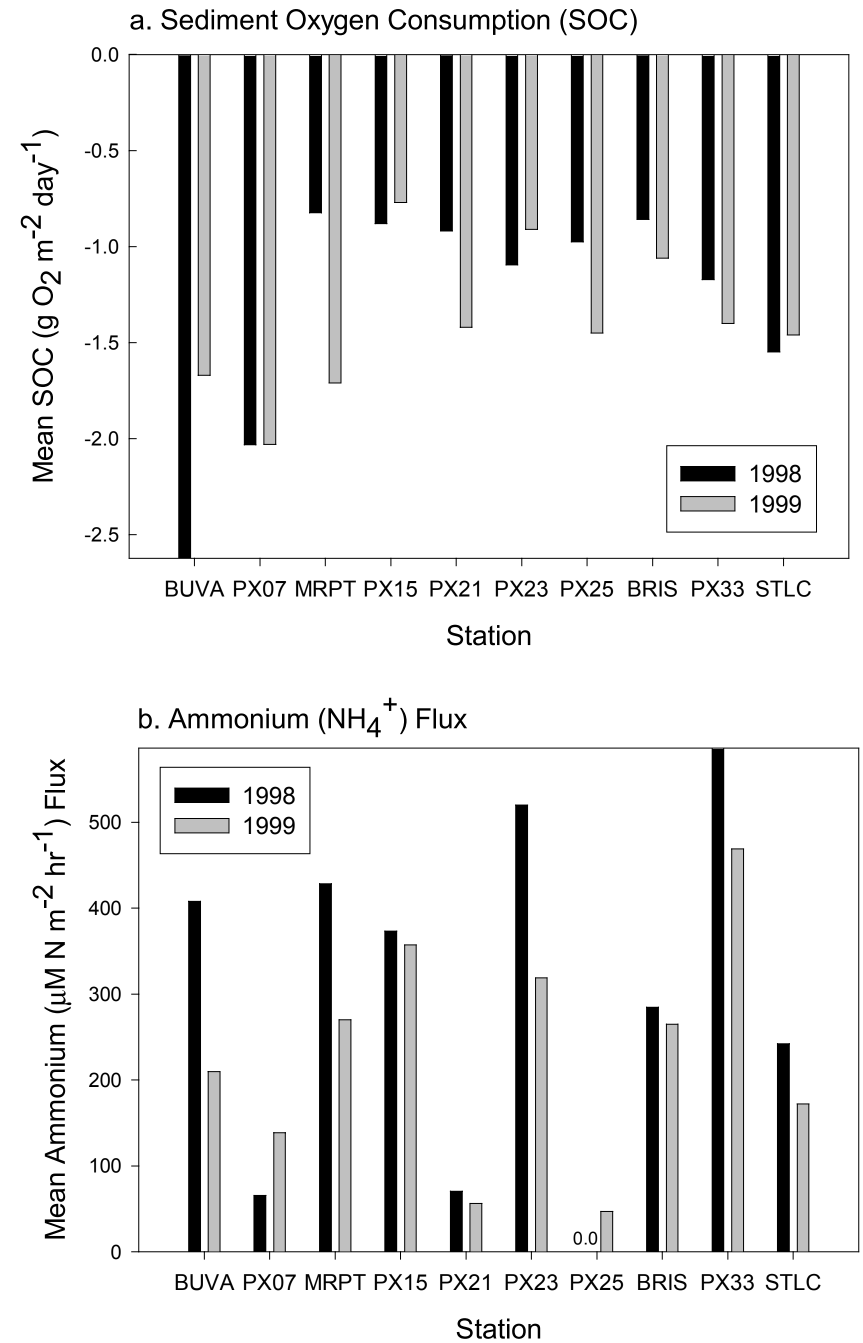

Figure 2-7. Comparison of Patuxent River MINI-SONE mean flux values calculated from monthly measurements from June through September 1998 and 1999 for:

a. sediment oxygen consumption (SOC), and

b. ammonium (NH_4^+) flux.

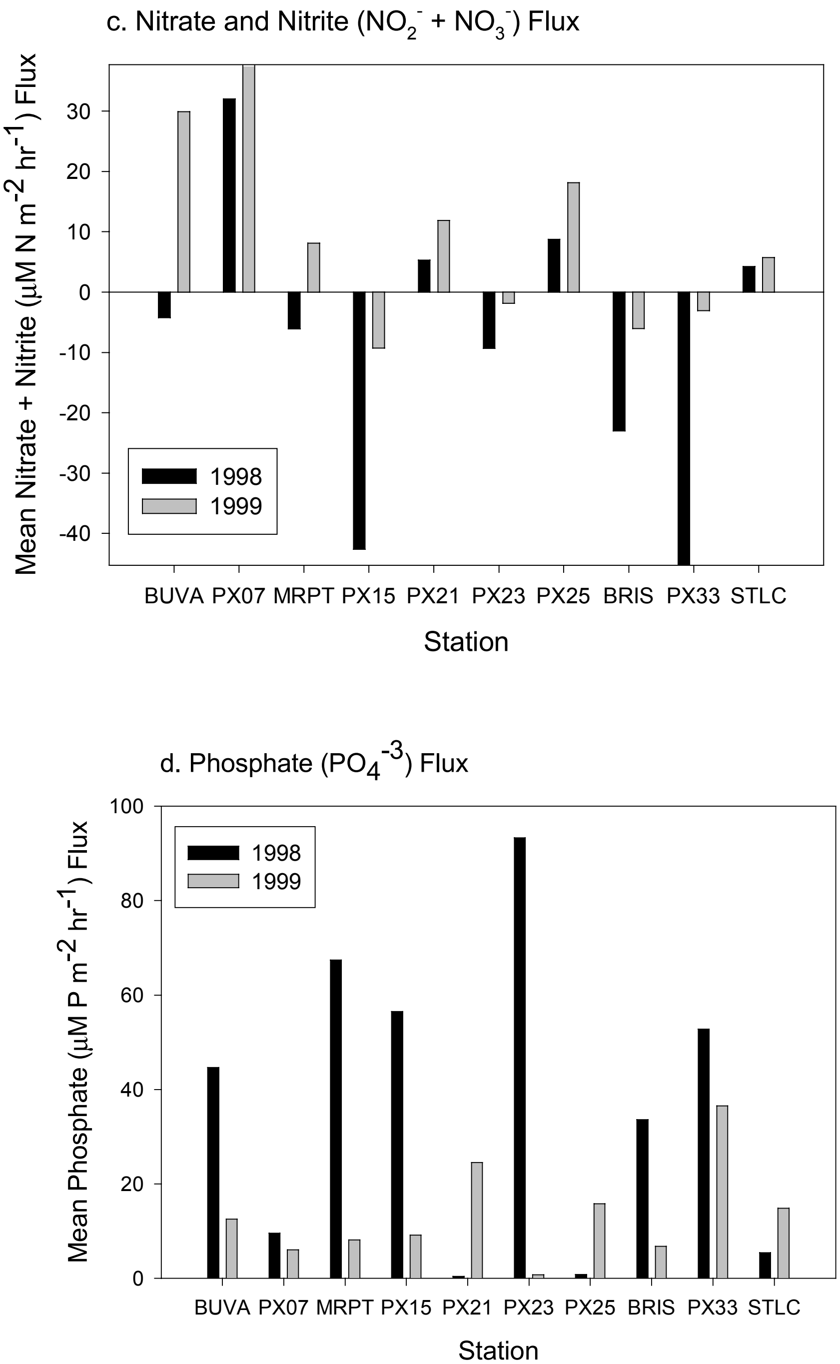

Figure 2-7. Comparison of Patuxent River MINI-SONE mean flux values calculated from monthly measurements from June through September 1998 and 1999 for:

c. nitrite plus nitrate (NO_2^- + NO_3^-), and

d. phosphate (PO_4^{-3}) flux.

2.5.5 Multivariate Analysis of MINI-SONE Data

2.5.5.1 Background

It is well known that the regeneration and transformation of nutrients in estuarine sediments is linked chemically and physiologically to a suite of bottom-water and sediment parameters. Previous studies have shown that factors such as water temperature, salinity and bottom water dissolved oxygen concentrations correlate well with certain sediment-water fluxes, (*e.g.* Boynton *et al*., 1980; Cowan and Boynton, 1996; Cowan *et al.*, 1996; Boynton *et al*., 1998). For example, bottom water nutrient concentrations influence diffusion gradients and sediment-water exchanges (*e.g.* Boynton and Kemp, 1985; Sundby *et al*., 1992). In addition, certain sediment properties such as sediment chlorophyll-*a* concentrations and sediment oxidation-reduction potential (Eh) have been well correlated with several sediment-water fluxes (*e.g.* Boynton *et al.*, 1998). In particular, sediment chlorophyll-*a* concentration is a good indicator of labile organic material available for recycling or regeneration and often explains much of the variation observed in sediment-water ammonium fluxes (Boynton *et al*., 1998). The goal of this analysis was to further develop statistically significant regression relationships between a few easily measured sediment and water quality parameters, and certain sediment-water exchanges (fluxes).

2.5.5.2 Data Sources and Analysis Methods

The statistical models presented here were constructed from sediment-water flux, water quality and surficial sediment data collected from six MINI-SONE, and four long-term SONE stations on the Patuxent River from 1996 through 1999. In 1998 and 1999, sediment-water flux measurements at the four long-term monitoring stations were collected with the abbreviated MINI-SONE technique. Previously, data from these long-term stations were collected using the traditional SONE technique in which three replicate cores were measured to estimate sediment-water exchanges, instead of a single core used in the MINI-SONE technique. Despite this small difference, data from all four years was treated similarly in this analysis. Sediment-water flux and water quality data were collected monthly from June through September of each year, while surficial sediment chlorophyll-*a* was collected from May through September. The techniques used to measure sediment-water flux, and sediment chlorophyll-*a* are outlined in Rohland *et al.* (1999). Data were analyzed as summer season means (4 month average) for each station. In general, the relationships developed for summer seasonal means were stronger and had higher predictive power than those for individual station observations (Boynton *et al*., 1998). For that reason, in 1999, regression equations were only developed using the four-month summer season mean values. Data were examined in two ways. First, data from 1999 was pooled with previous data and multivariate regressions were run with the best-fit models from previous years to evaluate their overall generality and fit. Second, for each sediment-water flux, a stepwise regression procedure (SAS statistical software version 6.12) was used to sort through various combinations of variables to help evaluate the most parsimonious model using a combination of water quality and sediment parameters from all the available data.

2.5.5.3 Results and Conclusions

The model fitting procedure used in 1999 was essentially the same as that used in previous years, in which the most parsimonious model was selected for each sediment-water flux. However, these models were again based upon relevant biogeochemical properties, not just for the best statistical fit. While sediment-water fluxes are in general responsive to the physical and chemical environment in predictable ways, subtle differences caused by inter-annual variability in the timing and magnitude of events has led to relationships that vary slightly from year to year. For this reason, regression models using multi-year data are more applicable in the long-term, but often have lower predictive power than models derived from data collected in a single season. Despite this limitation, several patterns have emerged and most regression relationships utilize the same physical, chemical and biological parameters compared to previous years. Only minor changes in the regression coefficients were needed to improve the applicability of these models to incorporate inter-annual differences in nutrient loading rates, temperature and salinity conditions.

The regression model for ammonium (NH_4^+) flux is a good example of the strength and robustness of these relationships. The chosen model once again uses sediment chlorophyll-*a* lagged one month (TCHL1M), and sediment redox potential (Eh) at 1 centimeter depth (SEDEHM1) as the predictor variables. These parameters were also selected as the most important predictor variables when the 1996 and 1997 data sets were analyzed separately (Boynton *et.al.*, 1998). With four full years of data (1996 – 1999) only minor changes in the parameter coefficients were required to maintain a highly predictive model that explains 79% of the variation in sediment-water ammonium flux ($r^2 = 0.79$, Figure 2-8). This result is particularly impressive because overall sediment-water ammonium flux was higher in 1999 than what would be predicted based upon nutrient loading rates. A plot of yearly mean ammonium flux versus winter-spring river flow shows that despite very low river flow in 1999, ammonium flux did not follow the pattern observed in the previous three years (Figure 2-9). The strength of this regression relationship is important because ammonium flux is the largest internal source of nitrogen to the water column during the summer months and is the preferred form of nitrogen for biological utilization (*e.g.* Boynton *et al.*, 1998; Valiela, 1998). The use of these relationships may prove very useful for providing first order estimates of sediment-water ammonium fluxes when actual measurements are not possible.

For sediment-water phosphate (PO_4^{-3}) flux, as with ammonium flux, only minor changes in the regression coefficients were required when data from 1999 was included in the full data set. As in last year, sediment chlorophyll-*a* concentration lagged one month (TCHL1M) and bottom water phosphate concentration (BWPO4) were selected as the most appropriate predictor variables. In 1999, sediment-water phosphate flux was extremely low across all stations, yet the selected regression model was still able to explain 74% of the observed variation in summer mean flux ($r^2 = 0.74$ Figure 2-10) using data from all available years. Since this model fitting procedure was begun, bottom water phosphate concentrations have remained the most important predictor variable in all years. This result illustrates how bottom water nutrient concentrations can influence diffusion gradients and sediment-water fluxes.

As with several of the other measured sediment-water fluxes, sediment oxygen consumption (SOC) also is well related to the amount of labile organic material on the sediment surface and the current sediment redox (Eh) potential. Unlike some of the other flux measurements, SOC appears to be quite sensitive to way redox potential is measured. In 1998 a small procedural change in the way sediment Eh was measured was initiated, however it resulted in very large changes in the relationship between SOC and Eh. This significantly altered the relationship between these variables and decreased the power of regression models when multi-year data was used. In 1999, this procedural change was removed and measurements were continued with the same method used previously. For this reason, the regression models generated for SOC will not include data from 1998. When data from 1996, 1997, and 1999 are merged into a single data set, the preferred regression model contains the same input parameters used in the 1996, and 1997 model. They include bottom water temperature (TEMP), sediment chlorophyll-*a* concentration lagged one month (TCHL1M), and sediment redox potential at 1 cm below the surface (SEDEHM1). With this model 76% of the variation in SOC is explained ($r^2 = 0.76$). As with other sediment-water regression models, estimates of SOC can be made to increase the spatial resolution within an estuary to obtain better estimates of whole system response to changes in nutrient loading.

For nitrite plus nitrate ($NO_2^- + NO_3^-$) sediment-water flux, a combined data set with all four years substantially degraded the predictive power of the relationship. With all four years of data merged into a single data set, only 53 % of the variation in sediment-water flux was explained using the selected model from previous years. While a more predictive regression equation can be found using all of the available data and step-wise selection procedure, the selected parameters are not consistent from year to year. It appears that inter-annual variation in the input variables substantially alters the relationship between bottom water and sediment conditions and sediment-water flux. In fact, inter-annual differences in sediment-water fluxes at certain stations were substantial, while at other stations the values were comparable. The reason for this lack of agreement between regression equations generated from different years is unknown at this time. Perhaps the linear models that work so well for ammonium and phosphate fluxes are not adequate to explain nitrite and nitrate flux and more sophisticated non-linear models are required. Despite this lack of generality, the data indicate that within a specific year, sediment-water fluxes can be well estimated when calibrated with actual flux measurements. For example, using 1999 data alone a regression equation was constructed that explained 89% of the variation in summer mean nitrite plus nitrate flux ($r^2 = 0.89$). Using actual data collected within a specific year or season, estimates of nitrite plus nitrate can be made at many more locations than could be possible with direct measurements to increase the spatial resolution within an estuary.

All of the regression equations discussed above use sediment chlorophyll-*a* concentration as an important input variable and further illustrate the importance of the deposition of labile organic material to the sediment surface to sediment-water exchanges. Because sediment chlorophyll-*a* concentrations are spatially variable, we can assume that sediment-water fluxes are spatially variable as well. With direct sediment - water exchanges only being measured at a few fixed locations, this variability may not be taken into account. The use of statistically generated regression equations to estimate sediment - water exchanges at locations where direct flux

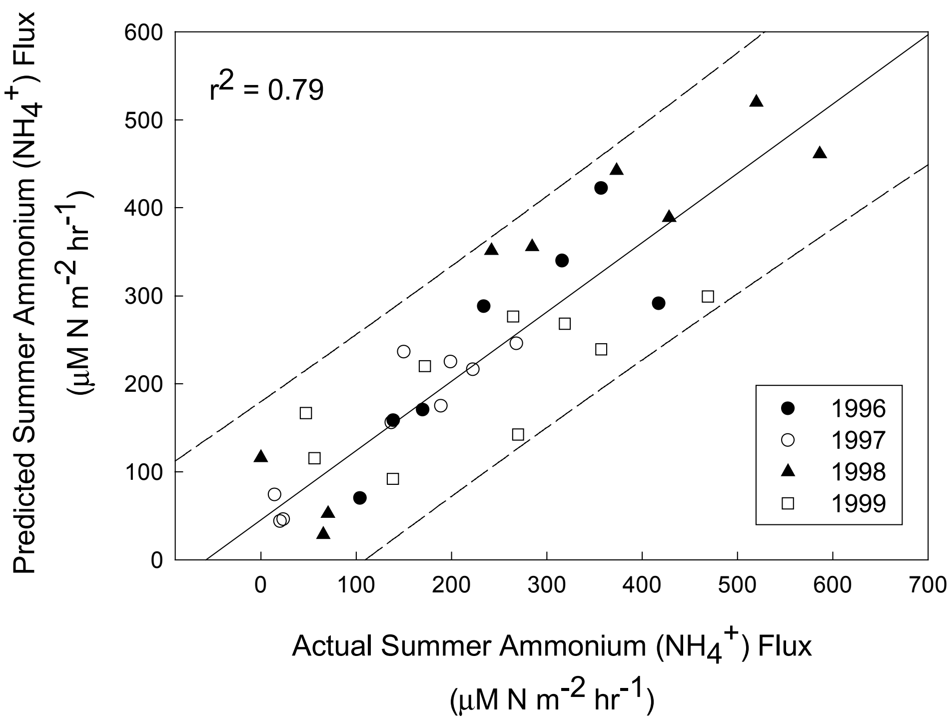

Figure 2-8. Predicted, sediment-water ammonium flux (NH4+) at MINI-SONE stations versus observed ammonium flux at these stations. Regression equation developed from four-month summer means collected from 1996 through 1999 on the Patuxent River. Data from the Buena Vista (BUVA) station was not included in the analysis.

The regression equation is:

Predicted ammonium flux = 1.819(TCHL1M) – 0.640 (SEDEHM1) + 156.09

Where TCHL1M = Mean total sediment chlorophyll-*a* (from May – August) to 1cm Depth,

SEDEHM1 = mean sediment Eh at 1 cm depth (from June – September).

Dashed lines indicate 95% confidence intervals for individual observations.

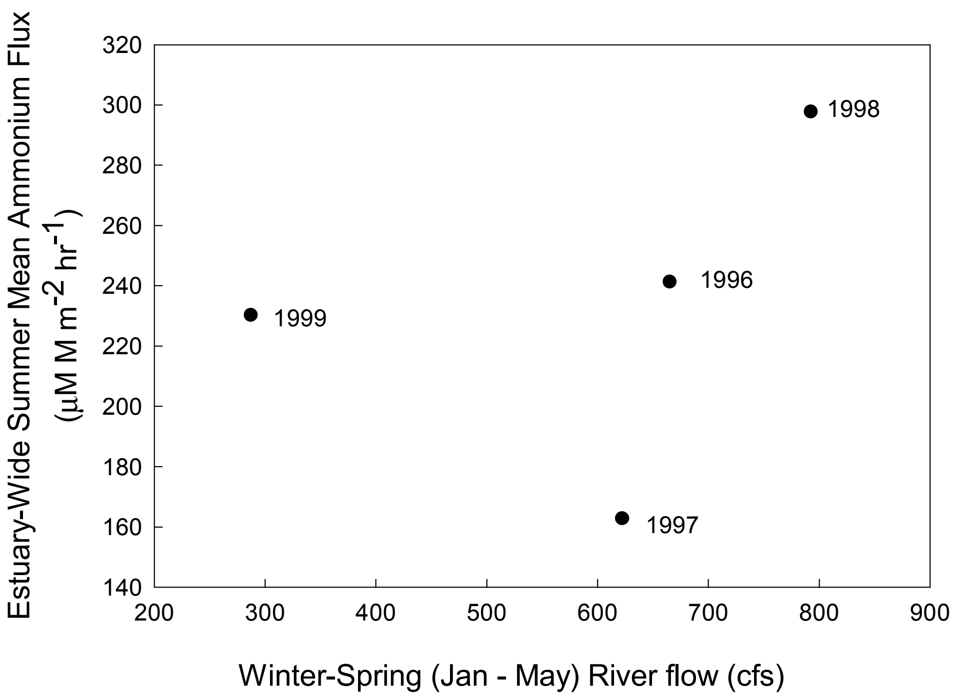

Figure 2-9. Yearly mean ammonium sediment-water flux versus winter-spring river flow on the Patuxent River.

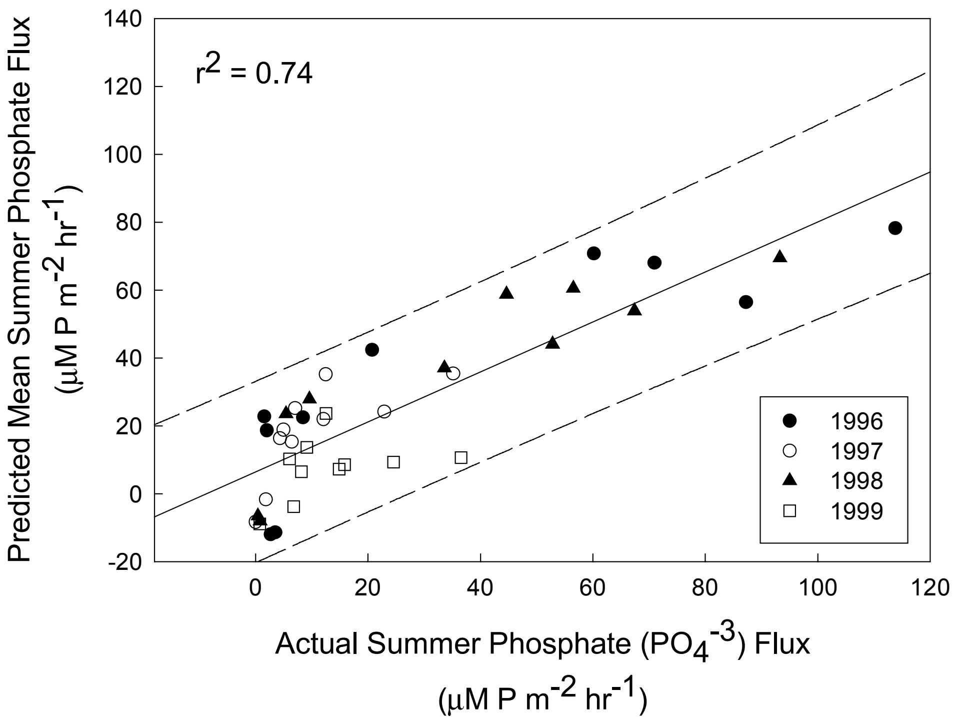

Figure 2-10. Predicted, sediment-water phosphate flux (PO_4^{-3}) at MINI-SONE stations versus observed phosphate flux at these stations. Regression equation developed from four-month summer means collected from 1996 through 1999 on the Patuxent River.

The regression equation is:

Predicted phosphate flux = 32.904(BWPO4) + 0.313(TCHL1M) – 33.883

Where BWPO4 = mean bottom water phosphate concentration

TCHL1M = mean total sediment chlorophyll-*a* (from May – August) to 1cm depth.

Dashed lines indicate 95% confidence intervals for individual observations

measurements are not possible may help include this spatial variation in estuary wide responses to changes in nutrient loading.

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3. SEDIMENT-WATER FLUX STATUS AND TRENDS:

1999 PATUXENT RIVER STUDY

W.R. Boynton and F.M. Rohland

The development of management actions to implement the 40% nutrient load reduction strategy has been a major thrust of the Chesapeake Bay Program during its third phase beginning in 1991. Prior to this, the Chesapeake Bay Water Quality Monitoring Program developed a database of information related to water quality conditions throughout the bay system. These data were used to describe conditions (status) in the bay system and identify areas of poor water quality. The Ecosystem Processes Component (EPC) Program has been a part of this effort since 1984.

A part of the Ecosystem Processes Component (EPC) Program was also designed to examine the sediment-water flux data in order to identify long-term trends in sediment-water nutrient and oxygen exchanges. In previous Interpretive Reports (Boynton *et al.*, 1993, 1994) results of statistical testing for trends were presented and discussed. As an addition to this, a time series of important environmental variables (river flow, bottom water dissolved oxygen concentrations and key sediment-water fluxes) were presented in graphical format in Interpretive Report #12 (Boynton *et al.*, 1995). These figures included monthly average data covering the first ten years of the monitoring program (1985 - 1994) collected from six sediment oxygen and nutrient exchanges (SONE) stations. The purpose of these analyzes was to explore the data to determine temporal trends and to provide a basis for relating important environmental conditions to the characteristics of sediment fluxes.

More recently (1998) a standardized protocol was developed by the Monitoring Program to examine data for status and trend characteristics. This protocol is described and used in the following sections to characterize the current status of sediment-water exchange processes at four Patuxent River stations and to evaluate the Patuxent River data set for interannual trends.

3.1 Sediment-Water Quality Status in the Patuxent River

A standardized protocol has been developed for scaling data in order to summarize the status of each parameter (Perry, *pers. comm.*). The status of each station is determined by comparison to a benchmark data set comprise of all flux data for the years 1985-1990 collected by the SONE program. The SONE program has no counterpart in the Virginia section of the bay so the data from Maryland are the only data used in the benchmark data set.

Each station is rated as poor, fair, or good relative to the benchmark data. These ratings were obtained as follows.

1. For each parameter in the benchmark data set, a transformation is chosen that yields a distribution that is symmetric and reasonably well approximated by the logistic cumulative distribution function (CDF). For the flux parameters, a signed square root transformation was used for all parameters except SOC for which a signed fourth root transformation was used.

2. A logistic CDF based on the mean and variance of each parameter of the benchmark data set is used to perform a probability integral transform on all data in the most recent 3-year period. This results in data in the interval (0,1) which follows a uniform distribution.

3. The 3 year mean of this 0-1 data is computed as an indicator of status in the current three year period. The median of n observations taken from a uniform distribution follows a Beta distribution (a symmetric, two parameter distribution) with parameters (m,m) where m = (n+1)/2.

4. Based on the Beta density, the distribution of 3-year medians from the benchmark data is divided into thirds. If the median of the current three year period is in the upper third (where upper is chosen as the end of the

distribution that is ecologically desirable) then the status rating is good, a median in the middle third is rated fair, and a median in the lower third is rated poor.

3.1.1 Notes on the Benchmark

The development of the benchmark for each of the five variables of the EPC-SONE program is different from that used in other portions of the monitoring program. It is most important to note that the stations were not segregated on the basis of salinity zones. As a result of this, every flux measurement made at all four Patuxent River stations was used to develop the benchmark for each parameter. This benchmark is a relative scale, and "good" fluxes can not necessarily be considered to indicate a recovered system. In other portions of the monitoring program separate benchmarks were developed for tidal fresh, oligohaline, mesohaline and polyhaline areas of the bay using only station data collected within those regions. The EPC-SONE program has three of the four stations monitored classified as mesohaline while the fourth station (Buena Vista [BUVA] in the Patuxent River) can only be classified as oligohaline a small fraction of the time; on an annual average basis this station (Buena Vista [BUVA]) would also be classified as mesohaline. Therefore, a single benchmark is constructed for each of the five variables; in effect, the variable benchmark is synonymous with the mesohaline benchmark.

3.1.1.2 Notes on the Current Status for the Patuxent River

A median value for the years 1997, 1998 and 1999 was calculated. The use of the last three years of data provides an "indicator" value of the status of the parameter relative to all other years during which measurements were taken. The median value of the last three years of data has the effect of reducing the influence of extreme climatic conditions (*i.e.* very wet or very dry years) since such extremes do not usually occur several years in succession. Since river flow and nutrient loading rates are important variables which either directly or indirectly influence sediment-water exchanges, it is important to note that 1997 was an average year, 1998 was very wet during winter and spring and 1999 was an extremely dry year until September when several hurricanes passed the area.

3.1.2 Evaluation of the Current Status for the Patuxent River

i. Sediment Oxygen Consumption (SOC)

The current status (median of 1997, 1998 and 1999 data) of sediment oxygen consumption (SOC) fluxes at the four SONE stations in the Patuxent River is indicated in Figure 3-1.a. It seems appropriate to judge higher values of SOC as good in the context of this evaluation for several reasons despite the fact that high SOC rates indicate that sediments are using dissolved oxygen. The main reason for adopting this approach is that SOC rates are responsive to DO

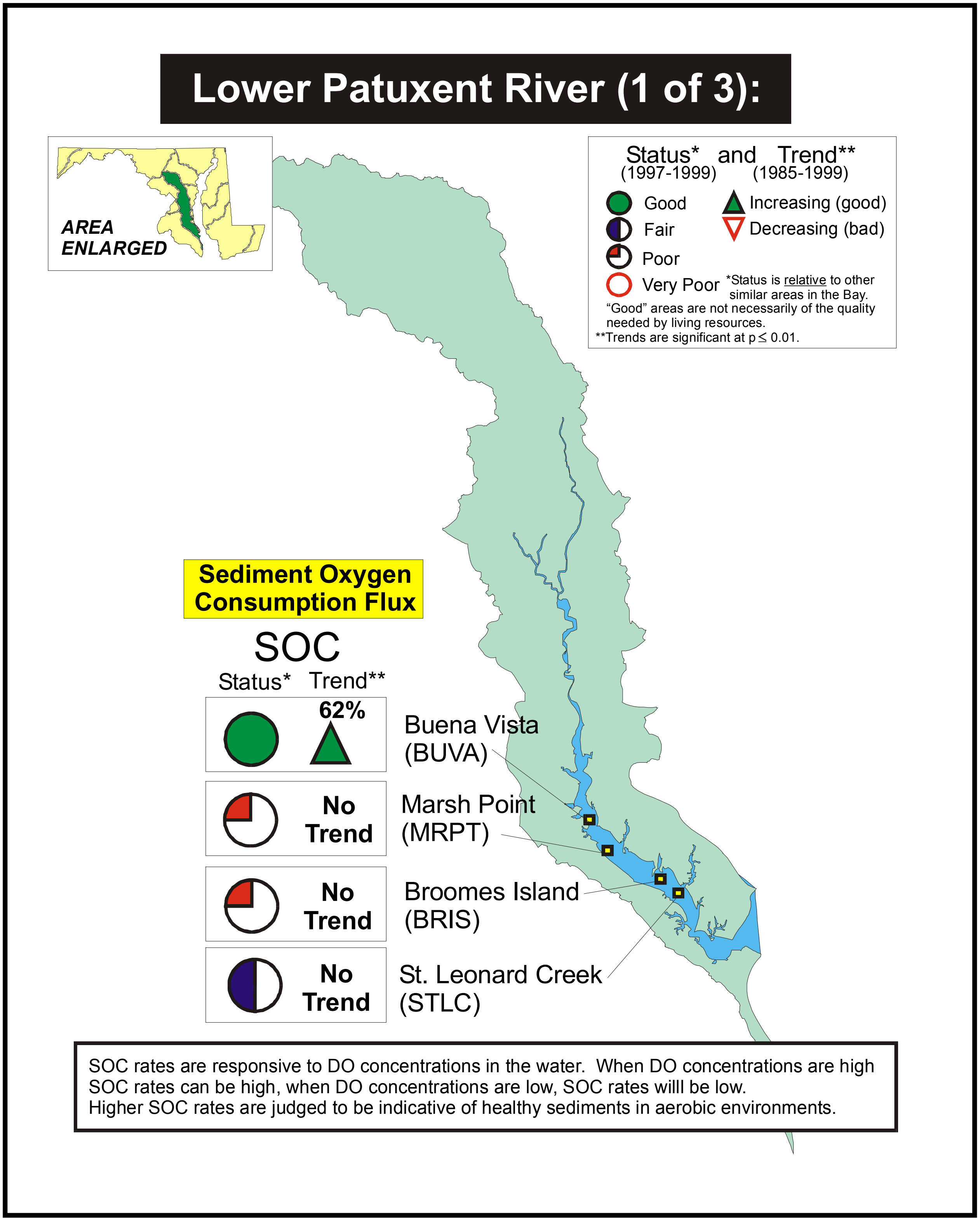

Figure 3-1.a. Map showing status and trends at four stations in the Lower Patuxent River for sediment oxygen consumption (SOC) fluxes (observed data).

Observed data indicates that no river flow adjustments were applied to the raw data concentrations in the water. When dissolved oxygen concentrations in the water are high, SOC rates can be high. Since restoration of increased dissolved oxygen in bottom waters is a goal of the management program we have adopted the position of treating higher SOC rates as indicative of healthy sediments in aerobic environments. Among the four SONE stations in the Patuxent river, two had SOC rates in the poor range, one was fair and one was in the good range. The pattern of SOC fluxes in the Patuxent River provides substantiation that the benchmark is appropriate. SOC fluxes progress from good down-river to poor at the head of the deep water channel at station Marsh Point (MRPT). This pattern would be expected based on proximity to nutrient sources and dissolved oxygen conditions. The station most upriver (and closest to nutrient sources) has a status of good (Buena Vista [BUVA]). This largely results because the water column is well mixed at this station and the propensity for low water column dissolved oxygen (DO) conditions are much reduced at this site.

ii. Ammonium (NH_4^+)

The current status (median of 1997, 1998 and 1999 data) of ammonium fluxes at the four SONE stations in the Patuxent River is indicated in Figure 3-1.b. In the case of ammonium fluxes it appears appropriate to judge high values as poor because of the well-established direct relationship between ammonium availability and excessive phytoplankton biomass accumulation. Among the four SONE stations in the Patuxent River two had ammonium fluxes in the fair range, and two were in the poor range. It should be noted that high river flow years have a particularly strong influence on ammonium fluxes (fluxes increase). One of the three years considered was a high flow year (1998) and ammonium fluxes were poor at upriver sites proximal to nutrient sources. The two down river sites were in the fair category (St. Leonard Creek [STLC] and Broomes Island [BRIS]). These downriver sites may be expected to move towards the good category when river flows return to more normal levels.

iii. Nitrite (NO_2^-)

The current status (median of 1997, 1998 and 1999 data) of nitrite flux at the four SONE stations in the Patuxent River is indicated in Figure 3-1.c. In the case of nitrite fluxes it appears appropriate to judge high values (positive values) as good because of the well-established linkage between nitrite evolution from sediments and oxidized sediment conditions. Among the SONE stations, three had nitrite fluxes in the good range and one was in the poor range. Stations are expected to change from poor to fair or fair to good when dissolved oxygen (DO) conditions in bottom water improve, even if only enough to allow some nitrification activity to occur.

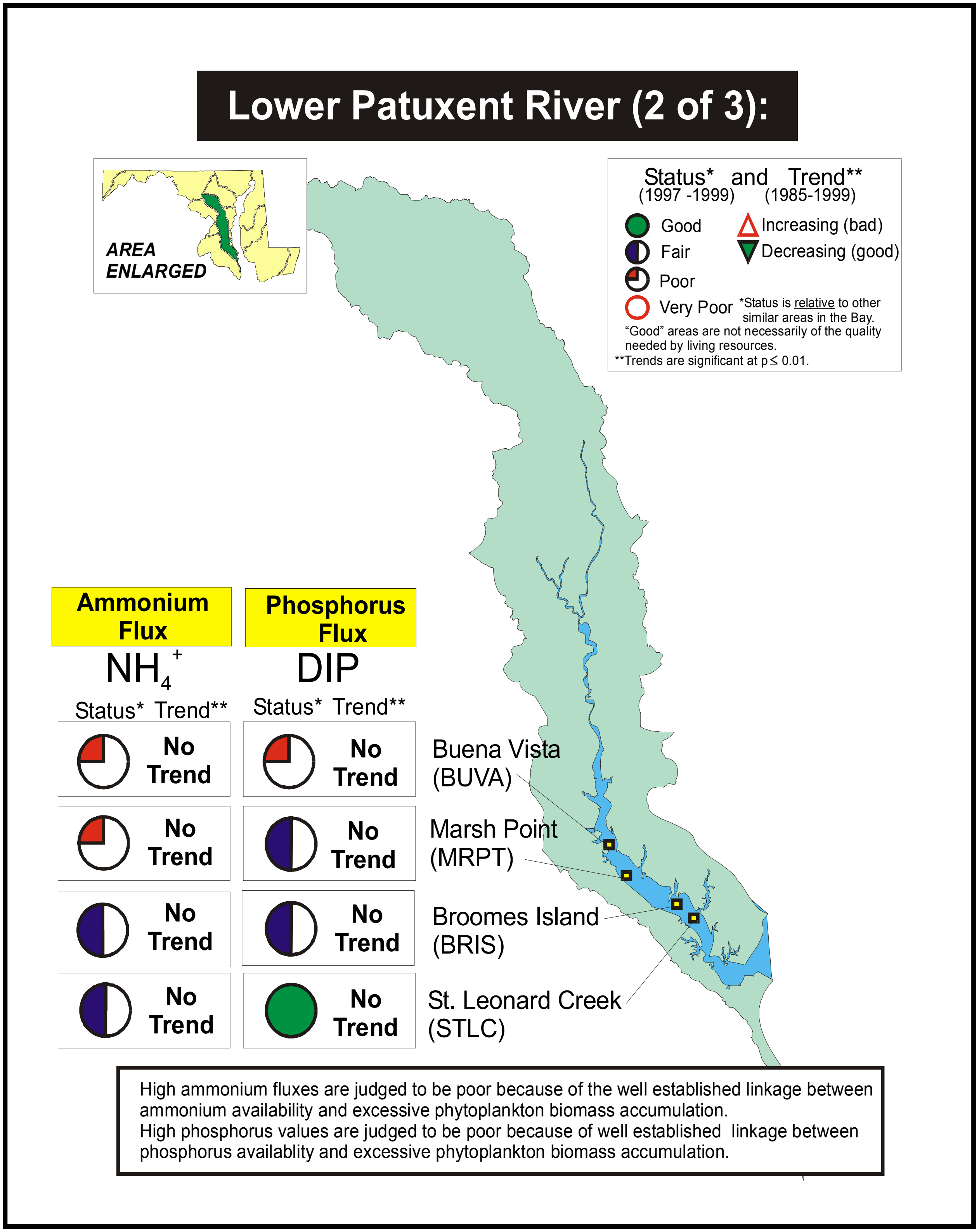

Figure 3-1.b. Map showing status and trends at four stations in the Lower Patuxent River for ammonium (NH_4^+) and phosphorus (PO_4^{-3}) fluxes (observed data).

Observed data indicates that no river flow adjustments were applied to the raw data.

vi. Nitrite plus Nitrate ($NO_2^- + NO_3^-$)

The current status (median of 1996, 1997 and 1998 data) of nitrite plus nitrate fluxes at the four SONE stations in the Patuxent River is indicated in Figure 3-1.c. In the case of nitrite plus nitrate fluxes it appears appropriate to judge high values (positive values) as good because of the well established linkage between nitrite plus nitrate evolution from sediments via complete nitrification and oxidized sediment conditions. Among the four SONE stations in the Patuxent River, two were judged to be good, St. Leonard Creek (STLC) and Buena Vista (BUVA), while Broomes Island (BRIS) and Marsh Point (MRPT) were found to be poor.

v. Dissolved Inorganic Phosphorus (PO_4^{-3} or DIP)

The current status (median of 1997, 1998 and 1999 data) of dissolved inorganic phosphorus fluxes at the four SONE stations in the Patuxent River is indicated in Figure 3-1.b. In the case of phosphorus fluxes it appears appropriate to judge high values as poor because of the well-established linkage between phosphorus availability and excessive phytoplankton biomass accumulation. Among the four SONE stations in the Patuxent River, two stations had phosphorus fluxes in the fair range, Marsh Point (MRPT) and Broomes Island (BRIS). The station farthest upstream, Buena Vista (BUVA), was in the poor range while St. Leonard Creek (STLC), the station furthest down stream, was in the good range. It should be noted that high river flow years have a particularly strong influence on phosphorus fluxes (fluxes increase) and one of the three years considered, 1998, was an exceptionally high flow year.

3.2 Sediment-Water Oxygen and Nutrient Exchanges (SONE) Trends: 1999 Patuxent River Study

A standardized protocol was strongly recommended by the Monitoring Program for determining interannual trends of each parameter (Eskin *et al.*, 1993). This approach used the non-parametric seasonal Kendall test. In results presented here, sediment oxygen and nutrient (SONE) flux data were NOT corrected for river flow, as is the case for testing other variables for trends within the monitoring program. This correction was not attempted because the temporal and spatial linkages between flow and sediment responses have not been clearly established.

3.2.1 Current Testing (Seasonal Kendall Test) for Seasonal Trends: 1985 - 1999 Data from the Patuxent River

Trend analysis is one method which can be use to assess the changes within the Bay system and the effectiveness of program design to restore optimum conditions in the Bay as well as prevent further deterioration of present conditions. The Seasonal Kendall test is recommended by the

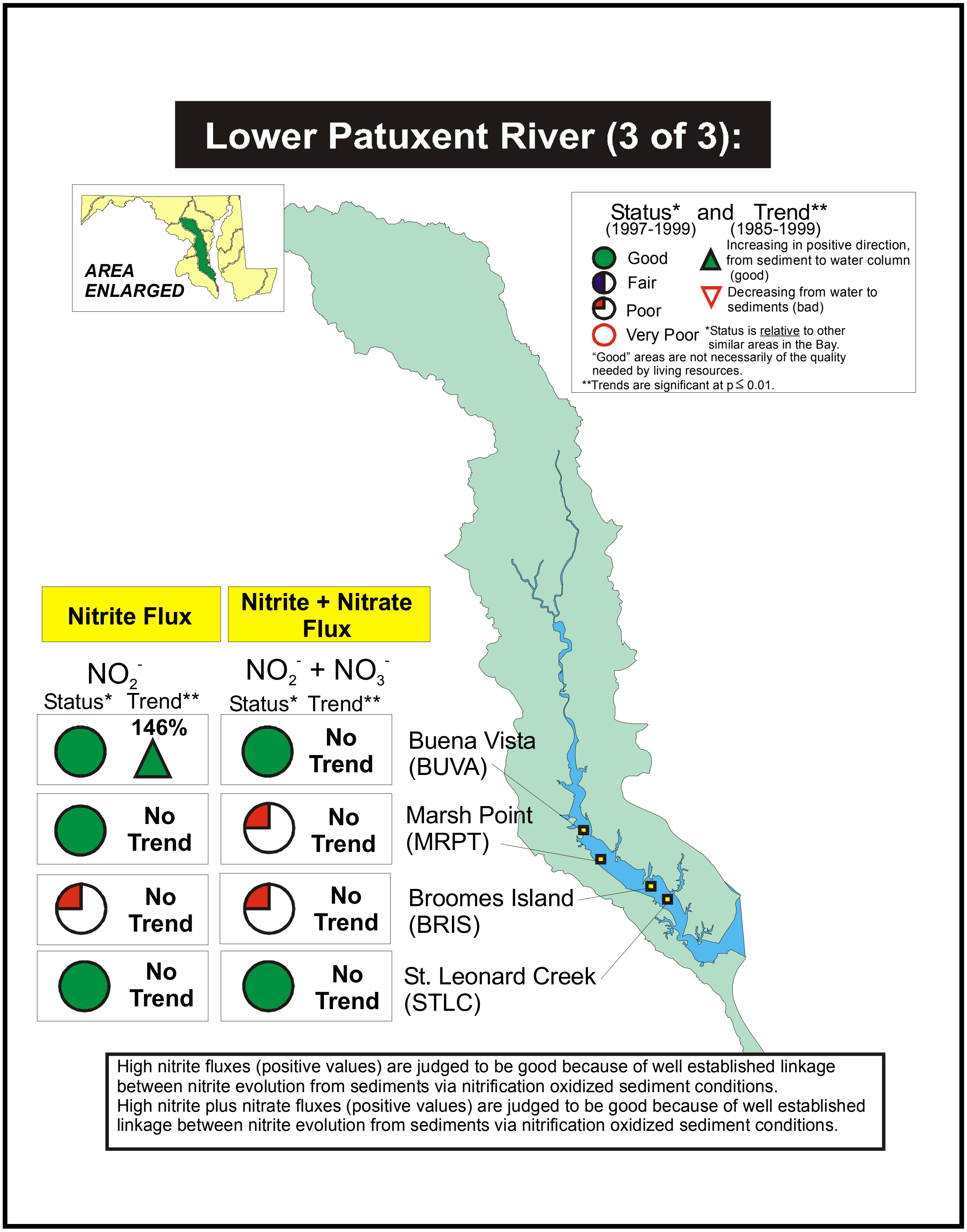

Figure 3-1c. Map showing status and trends at four stations in the Lower Patuxent River for nitrite (NO_2^-) and nitrite plus nitrate ($NO_2^- + NO_3^-$) fluxes (observed data).

Observed data indicates that no river flow adjustments were applied to the raw data.

Monitoring Program as the preferred statistical procedure for trend assessments. The seasonal Kendall test is non-parametric and is a generalization of the Mann-Kendall test. It is applied to data sets exhibiting seasonality. The test does not assume a specific parametric form. Details of the statistical method are given in Gilbert (1987).

3.2.2 Flux Data Set for Four Patuxent River

Flux data were collected over a period of fifteen years (1985 - 1999) during seven months (April through November) at 4 stations in the Patuxent River (Buena Vista [BUVA], Broomes Island [BRIS], Marsh Point [MRPT] and St. Leonard Creek [STLC]). Flux data typically exhibit strong seasonality that may increase the variance of the data. In order to characterize the data initially, manual QA/QC checks were completed. Extreme outliers were examined and in certain cases these data were discarded. Monthly variation and distribution of flux data are presented using box and whisker plots (Section 2.2.3.1). It has been recommended that for water quality data the median (rather than the mean) be used to determine the center point of the data set, particularly since it is well known that environmental quality data are usually positively skewed (Helsel, 1990). Separate analyzes were performed for each sediment oxygen and nutrient exchange (SONE) variable. A probability level of 0.01 was used to assess the significance of the results using observed data (data not "corrected" for river flow effects).

3.2.3 Results of Kendall Tests for Detection of Inter-Annual Trends for the Patuxent River

Three graphics (Figures 3-1.a., 3-1.b. and 3-1.c.) summarize results of the five flux variables, sediment oxygen consumption (SOC), ammonium (NH_4^+), inorganic phosphorus, nitrite (NO_2^-) and nitrite plus nitrate ($NO_2^- + NO_3^-$), measured at four sites (Buena Vista [BUVA], Broomes Island [BRIS], Marsh Point [MRPT] and St. Leonard Creek [STLC]) in the Patuxent River estuary. An overview of the significance of trends is summarized in Table 3-1. Annual values for observed data are presented in Table 3-2.

Testing for trends at the annual time scale resulted in few statistically significant results ($p < 0.01$). In the Patuxent River estuary sediment oxygen consumption (SOC) fluxes indicated a significant increasing trend at the upper estuary station at Buena Vista (BUVA). It is important to note that increasing values (increasingly negative) of sediment oxygen consumption (SOC) indicate that dissolved oxygen flux from water to sediments has increased during the study period and in this context is considered to be an improving trend in sediment quality. A marginally significant increasing trend (at probability level $p < 0.05$) was indicated for both ammonium (NH_4^+) and nitrite (NO_2^-) in the Patuxent River estuary for several stations (Buena Vista [BUVA)] and Marsh Point [MRPT]) and, in the case of these nutrients, is considered to be a degrading trend for ammonium and an improving trend for nitrite.

There were no significant annual trends for dissolved inorganic phosphorus or nitrite plus nitrate fluxes in the Patuxent River estuary. During the last 14 years both wet and dry years have been recorded (relatively high and low diffuse source loading years, respectively) which tend to

Table 3-1. A condensed summary of significant trends (observed data) detected for sediment-water exchange data using seasonal Kendall Test statistic.

More details can be found in Table 3-2 and Table 3-3.

Observed data indicates that no river flow adjustments were applied to the raw data.

*Significance: * p = 0.05, ** p = 0.01; *** p = 0.001*

NOTE: Upward pointing arrows indicate that the trend was judged as improving;
Downward pointing arrows indicate that the trend was judged as degrading.

Station	**Month**								**ANNUAL**
	APR	**MAY**	**JUN**	**JUL**	**AUG**	**SEP**	**OCT**	**NOV**	
a. Sediment Oxygen Consumption (SOC; g O_2 m^{-2} day^{-1} yr^{-1})									
BUVA				* ↑	* ↑				*** ↑
BRIS					* ↑				
b. Ammonium (NH_4^+; μM N m^{-2} hr^{-1} yr^{-1})									
MRPT		* ↓							
STLC					* ↑				
c. Nitrite (NO_2^-; μM N m^{-2} hr^{-1} yr^{-1})									
BUVA		* ↑							* ↑
d. Nitrite plus Nitrate (NO_2^- + NO_3^-; μM N m^{-2} hr^{-1} yr^{-1}) No significant trends									
e. Dissolved Phosphorus (PO_4^{-3}; μM Pm^{-2} hr^{-1} yr^{-1})									
MRPT			* ↑						

Table 3-2. Table of Seasonal Kendall Test Statistics (observed data) at four SONE stations for four seasonal and an annual variable.

Observed data indicates that no river flow adjustments were applied to the raw data.

*Significance: ** p = 0.01; *** p = 0.001*

a. Annual Trends

STATION	SOC	NH_4^+	NO_2^-	$NO_2^- + NO_3^-$	PO_4^{-3}
St. Leonard Creek (STLC)					
Sign	-49	46	21	6	14
p value	0.15	0.18	0.43	0.88	0.71
Slope	-0.025	1.685	0.334	0.042	0.068
Marsh Point (MRPT)					
Sign	-38	23	35	45	17
p value	0.13	0.37	0.14	0.07	0.51
Slope	-0.040	8.450	0.514	1.713	0.747
Broomes Island (BRIS)					
Sign	-47	-11	-14	41	-29
p value	0.06	0.68	0.59	0.09	0.254
Slope	-0.043	-3.400	-0.077	1.150	-0.364
Buena Vista (BUVA)					
Sign	-94	32	55	3	-26
p value	0.007***	0.37	0.04*	0.95	0.45
Slope	-0.069	3.493	0.860	0.000	-0.369

produce high and low sediment fluxes. Since high/low load years have occurred without pattern, trends are difficult to detect unless they are very large and persist for several years.

3.2.4 Results of Seasonal Kendall Tests for Detection of Monthly Trends for the Patuxent River

The results from the monthly Seasonal Kendall tests are presented as a table using observed rather than flow corrected data (Table 3-3). The Seasonal Kendall Test Statistic value indicates the direction of slope ("+" indicate a positive or increasing slope while "-" indicates a negative or decreasing slope). Different probability levels for significance are indicated in Table 3-3. The n value indicates the number of observations used in the analysis**.**

i. Sediment Oxygen Consumption (SOC)

Three significant negative yet improving trends were indicated for sediment oxygen consumption (SOC) fluxes at $p < 0.05$ at Buena Vista (BUVA) for July and August and at Broomes Island (BRIS; Table 3-3.a).

ii. Ammonium (NH_4^+)

A significant trend was indicated for ammonium (NH_4^+) fluxes at $p < 0.05$ in May at Marsh Point (MRPT; degrading trend) and at St. Leonard Creek (STLC) in August (degrading trend; Table 3-3.b).

iii. Nitrite (NO_2^-)

A positive (improving) significant trend was indicated for nitrite (NO_2^-) fluxes at $p < 0.05$ in the Patuxent River at Buena Vista (BUVA) in May (Table 3-3.c).

iv. Nitrite plus Nitrate ($NO_2^- + NO_3^-$)

No significant trends were observed for nitrite plus nitrate fluxes (Table 3-3.d).

v. Dissolved Inorganic Phosphorus (PO_4^{-3} or DIP)

A positive (improving) significant trend was found for phosphorus (PO_4^{-3}) fluxes at $p < 0.05$ at Marsh Point (MRPT) in June (Table 3-3.e).

Table 3-3. Table of Monthly Seasonal Kendall Test Statistics (observed data) at four SONE stations for five SONE variables.
Observed data indicates that no river flow adjustments were applied to the raw data.
"." or blank cells in the table indicate that no data was collected or the data was insufficient to perform the analysis.
*Significance: * p = 0.05; ** p = 0.01; *** p = 0.001*

a. Sediment Oxygen Consumption (SOC; g O_2 m^{-2} day^{-1} yr^{-1})

STATION	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV
PATUXENT RIVER: Buena Vista (BUVA): 1985 - 1999								
Sign	3	-10	-9	-29	-43	6	-9	-3
p value	.	0.28	0.69	0.03*	0.04*	0.61	0.24	.
N	3	8	15	11	15	9	7	3
Marsh Point (MRPT): 1989 - 1999								
Sign		-3	4	-17	-18	-1	-3	
p value		0.72	0.86	0.21	0.18	1.00	1.00	
n		6	10	10	11	9	6	
Broomes Island (BRIS): 1989 - 1999								
Sign		5	5	-7	-27	-12	-11	
p value		0.47	0.73	0.64	0.04*	0.26	0.06	
n		6	10	11	11	9	6	
St. Leonards Creek (STLC): 1985 - 1999								
Sign	3	-10	19	-18	-22	-13	-5	-3
p value	.	0.28	0.37	0.18	0.25	0.26	0.56	.
n	3	8	15	11	14	9	7	3

b. Ammonium (NH_4^+; μM N m^{-2} hr^{-1} yr^{-1})

STATION	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV
PATUXENT RIVER: Buena Vista (BUVA): 1985 - 1999								
Sign	-3	10	-7	17	33	-16	-3	1
p value	.	0.28	0.77	0.21	0.11	0.12	0.77	.
n	3	8	15	11	15	9	7	3
Marsh Point (MRPT): 1989 - 1999								
Sign		13	-7	3	21	-16	9	
p value		0.02*	0.60	0.87	0.12	0.12	0.14	
n		6	10	11	11	9	6	
Broomes Island (BRIS): 1989 - 1999								
Sign		-3	-3	9	-17	2	1	
p value		0.72	0.86	0.53	0.21	0.92	1.00	
n		6	10	11	11	9	6	
St. Leonards Creek (STLC): 1985 - 1999								
Sign	1	-4	8	1	41	-6	5	0
p value	.	0.72	0.73	1.00	0.05*	0.61	0.56	.
n	3	8	15	10	15	9	7	3

Table 3-3. Table of Monthly Seasonal Kendall Test Statistics (Observed data) at four SONE stations for five SONE variables (Continued)
Observed data indicates that no river flow adjustments were applied to the raw data.
"." or blank cells in the table indicate that no data was collected or the data was insufficient to perform the analysis.
*Significance: * p = 0.05; ** p = 0.01; *** p = 0.001*

c. Nitrite (NO_2^-; $\mu M\ N\ m^{-2}\ hr^{-1}\ yr^{-1}$)

STATION	**APR**	**MAY**	**JUN**	**JUL**	**AUG**	**SEP**	**OCT**	**NOV**
PATUXENT RIVER: Buena Vista (BUVA): 1985 - 1999								
Sign	0	13	-5	31	6	4	6	0
p value	.	0.02*	0.75	0.02	0.73	0.76	0.23	.
n	1	6	11	11	12	9	5	1
Marsh Point (MRPT): 1989 - 1999								
Sign		3	-1	13	7	2	11	
p value		0.72	1.00	0.29	0.64	0.92	0.06	
n		6	10	10	11	9	6	
Broomes Island (BRIS): 1989 - 1999								
Sign		-3	-4	-16	-1	8	4	
p value		0.72	0.86	0.24	0.88	0.45	1.00	
n		6	10	11	11	9	6	
St. Leonards Creek (STLC): 1985 - 1999								
Sign	0	1	-21	14	15	9	3	0
p value	.	1.00	0.12	0.29	0.34	0.48	0.72	.
n	1	6	11	10	12	9	6	1

d. Nitrite plus Nitrate ($NO_2^- + NO_3^-$; $\mu M\ N\ m^{-2}\ hr^{-1}\ yr^{-1}$)

STATION	**APR**	**MAY**	**JUN**	**JUL**	**AUG**	**SEP**	**OCT**	**NOV**
PATUXENT RIVER: Buena Vista (BUVA): 1985 - 1999								
Sign	-3	-10	12	17	-7	2	-8	0
p value	.	0.28	0.58	0.21	0.74	0.92	0.38	.
n	3	8	15	11	14	8	7	3
Marsh Point (MRPT): 1989 - 1999								
Sign		-5	17	13	9	8	3	
p value		0.47	0.16	0.34	0.53	0.48	0.72	
n		6	10	11	11	9	6	
Broomes Island (BRIS): 1989 - 1999								
Sign		-3	19	10	6	8	1	
p value		0.72	0.11	0.41	0.70	0.48	1.00	
n		6	10	11	11	9	6	
St. Leonards Creek (STLC): 1985 - 1998								
Sign	-3	2	-16	16	-17	18	7	-1
p value	.	0.90	0.46	0.22	0.43	0.08	0.38	.
n	3	8	15	10	15	9	7	3

Table 3-3. Table of Monthly Seasonal Kendall Test Statistics (Observed data) at four SONE stations for five SONE variables (Continued).

Observed data indicates that no river flow adjustments were applied to the raw data.

"." or blank cells in the table indicate that no data was collected or the data was insufficient to perform the analysis.

*Significance: * p = 0.05; ** p = 0.01; *** p = 0.001*

e. Dissolved Phosphorus (PO_4^{-3}; μM Pm^{-2} hr^{-1} yr^{-1})

STATION	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV
PATUXENT RIVER: Buena Vista (BUVA): 1985 - 1999								
Sign	-3	2	-19	-5	15	-6	-9	1
p value	.	0.90	0.32	0.76	0.55	0.61	0.24	.
n	3	8	14	11	13	9	7	3
Marsh Point (MRPT): 1989 - 1999								
Sign		1	27	-3	-11	-8	11	
p value		1.00	0.02*	0.88	0.44	0.48	0.06	
n		6	10	11	11	9	6	
Broomes Island (BRIS): 1989 - 1999								
Sign		3	-7	-3	-21	-4	3	
p value		0.72	0.60	0.88	0.12	0.76	1.00	
n		6	10	11	11	9	6	
St. Leonards Creek (STLC): 1985 - 1999								
Sign	-2	4	7	13	2	-12	1	1
p value	.	0.72	0.77	0.35	0.96	0.26	1.00	.
n	3	8	15	11	15	9	7	3

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4. SUBMERGED AQUATIC VEGETATION (SAV) HABITAT EVALUATION

R.M. Stankelis, W.R. Boynton and J.M. Frank

4.1 Introduction

Declines in submerged aquatic vegetation (SAV) populations during the last half of the twentieth century have been well documented in a variety of shallow coastal estuaries worldwide (Den Hartog and Polderman, 1975; Kemp *et al*., 1983; Orth and Moore, 1983, 1984). A variety of studies have suggested that increased anthropogenic inputs of dissolved nutrients and particulate matter have been primarily responsible for degraded water quality conditions and reduced light available to rooted macrophyte populations (*e.g.*, Sand-Jensen, 1977; Cambridge *et al.*, 1986; Kemp *et al*., 1983; Twilley *et al*., 1985; Silberstein *et al.*, 1986). While light availability is generally agreed to be the most critical resource limiting the extent and distribution of SAV populations, an understanding of the conditions that are necessary and sufficient to provide adequate light has been elusive. A number of studies have demonstrated that epiphytes can substantially reduce the amount of available light reaching the leaf surface (*e.g.*, Burt *et al*., 1995; Boynton *et al.*, 1999a). However, epiphyte loads can be modified to a great extent by a variety of factors such as epiphyte grazer density (*e.g.* Neckles *et al*., 1993 ; Williams and Ruckelshaus, 1993), light availability (Boynton *et al.*, 1999a), nutrient availability (Kemp *et al*., 1983; Burt *et al*., 1995), wave action (*e.g.* Koch and Beer, 1996) and leaf turnover rates.

As a result of this apparent complexity and the difficulties associated with determining mechanisms and causal factors, field monitoring of water quality remains an important tool for understanding why SAV thrives, survives or declines at specific locations. In Chesapeake Bay, field monitoring is particularly important because of the large range of conditions found within the Bay and tributary rivers. In some Chesapeake Bay tributaries, modest reductions in nutrient loading have been achieved in recent years resulting in improved water quality conditions (Boynton *et al*., 1995). However, many of these tributaries, including the Patuxent River, that were historically populated with SAV beds, have not shown significant recovery of SAV. In other areas, such as Tangier Sound, SAV acreage has declined significantly in recent years despite a general increase in the previous decade.

Therefore, in 1997 the EPC began an ambitious and diversified study of near-shore water quality conditions important to SAV growth and survival. Information gathered during the first two years of investigation was used to refine and modify the study during 1999. These changes included the addition of selected locations in Tangier Sound (Janes Island State Park) and a modification of overall sampling frequency. As in past years, the SAV habitat evaluation was composed of two discrete but complimentary study elements: near-shore water quality monitoring and epiphyte growth monitoring.

4.1.1 Near-shore Water Quality Evaluation

The primary goal of the near-shore water quality evaluation was to measure a suite of water quality parameters directly in the shallow near-shore habitat to assess compliance with established SAV habitat requirements. In 1992, the Chesapeake Bay Program established a set of habitat criteria for five water quality parameters thought to be most important for SAV growth and survival (Batuik *et al*., 1992). These include water column dissolved inorganic nitrogen (DIN), dissolved inorganic phosphorus (DIP), water column light attenuation (Kd), water column total suspended solids (TSS), and water column chlorophyll-*a* (Tchl*a*). While a second, more comprehensive revision and evaluation of these habitat requirements is currently in review, reference in this report will be made to the habitat criteria specified in the 1992 synthesis. At present, the vast majority of routine water-quality monitoring is done at river channel locations, often distant (kilometer scale) from actual SAV habitats. These data, although very useful, may not reflect near-shore conditions due to a variety of localized conditions such as resuspension of sediments, point source discharges, or existing algal communities. Therefore, data for this study collected directly in near-shore SAV habitats will provide more exact information about water quality conditions in these locations. The secondary goal of this study was to provide corresponding water quality data to be used in the evaluation of the epiphyte growth study, where water quality affects light attenuation secondarily through the stimulation of epiphytic growth on SAV leaves.

4.1.2 Epiphyte Growth Study

The epiphyte growth study was designed with two goals in mind. The first was to evaluate epiphyte accumulation rates using artificial substrates at locations where SAV currently does not exist. The second was to compare epiphyte accumulation rates to water quality data at various locations along the axis of the Patuxent River and at selected paired sites in Tangier Sound. This comparison will provide field data for calibration of models predicting epiphyte biomass based upon simple, water quality data. In recent years there has been an increased interest in measuring the amount of light that actually reaches the leaf surface to help refine SAV habitat light requirements (Batuik *et al*., 2000). This study addresses this issue in a variety of ways.

In 1998, a comparison of epiphyte fouling rates on live SAV and Mylar strips was conducted. This compared epiphytic growth rates on transplanted live SAV to artificial substrates and helped calibrate and interpret results obtained using artificial substrates. The results of this study suggested that Mylar strips could be used as an acceptable surrogate for live plants in order to estimate light attenuation from epiphytic fouling (Boynton *et al*. 1999a). While a number of comparisons of epiphyte accumulation rates have been made between live SAV blades and artificial substrates (seagrass mimics) with conflicting results (Lin *et al*., 1996; Pinckney and Micheli, 1998), such comparisons are made more difficult because of differences in technique, geographic region, length of exposure, and SAV species. Despite potential limitations, artificial substrates can be used effectively to compare the effects of different water quality conditions on epiphyte accumulation rates and light attenuation when live plants are not available (Burt *et al*., 1995, Boynton *et al.*, 1999a). In addition, artificial substrates can be standardized between sites, and provide a quick assessment of epiphyte growth potential at SAV restoration sites.

4.2 Methods

4.2.1 Near-shore Water Quality Evaluation

4.2.1.1 SAV Water Quality Station Locations

In 1999, six mesohaline stations on the Patuxent River and two paired stations near Janes Island State Park on the eastern shore of Tangier Sound were monitored. The Patuxent River stations were monitored in 1998 and were selected to reflect a variety of nutrient, salinity and wave exposure regimes (Figure 4-1). The two paired stations near Janes Island were selected to provide comparison of vegetated and nonvegetated sites. A detailed explanation of sampling methods is given in Rohland *et al*. (1999).

4.2.1.2 SAV Water Quality Sampling Frequency

Sampling was conducted aproximately bi-weekly from April through May 1999 with some weekly sampling from June through October 1999. A total of 20 SAV sampling cruises were completed (Table 4-1). The exact sampling schedule is also given in Boynton *et al*. (1999b).

Table 4-1. SAV Water Quality Sampling Dates. A total of 20 cruises were completed.

STATIONS	SAMPLING DATES 1999						
	April	May	June	July	August	September	October
SVBA	26	10	3	16	5	2	2
SV02		20	9	22	10	9	
SV5A		28	17	29	17	17	
SV06			23			24	
SV07			29				
SV09							

4.2.2 Epiphyte Growth Survey

4.2.2.1 Epiphyte Sampling Locations and Frequency of Sampling

The epiphyte growth survey was completed concurrently with the SAV water quality element at all six Patuxent River and two Janes Island paired sites (Tables 4-1 and 4-2). The exact sampling schedule is also given in Boynton *et al.* (1999b).

Table 4-2. Jane's Island Sampling Dates. A total of 8 cruises were completed.
G = Grass; S = Unvegetated (Sand)

STATIONS	SAMPLING DATES 1999			
	June	July	September	October
JI1G	16	6	1	1
JI1S	22		8	
JI2G	29		23	
JI2S				
JI2P				

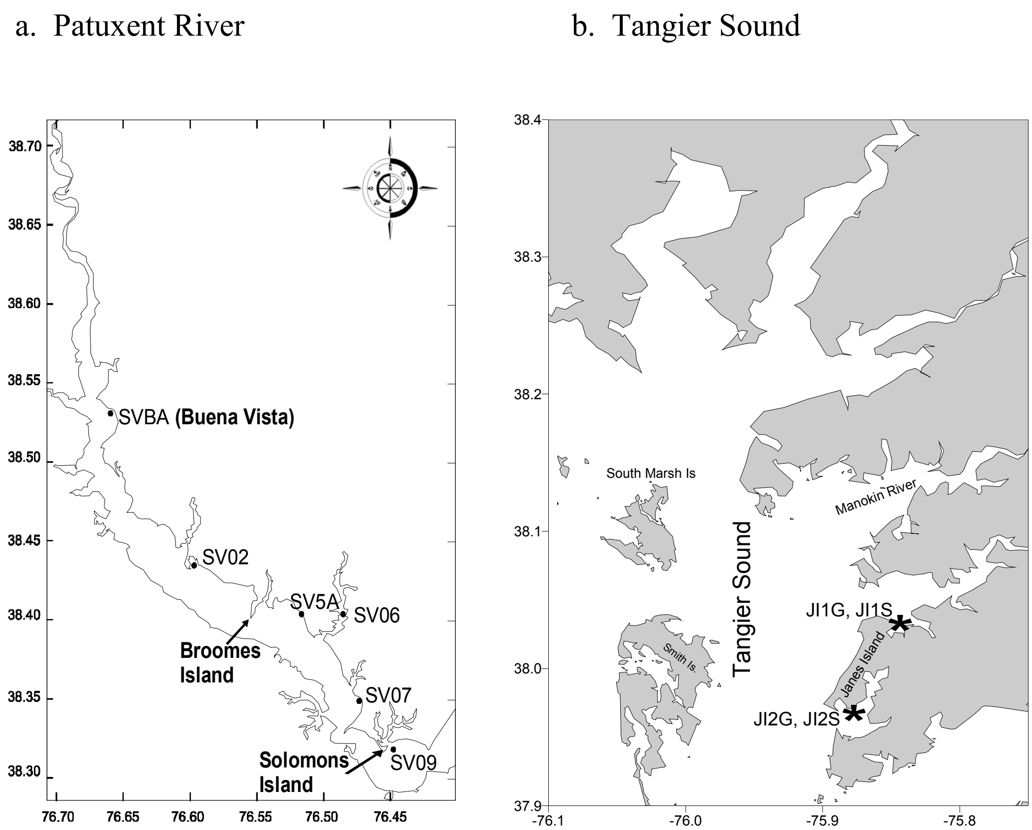


Figure 4-1. Maps of 1999 Submerged Aquatic Vegetation (SAV) monitoring stations in a. Patuxent River, and b. Tangier Sound.

Latitude and longitude are in decimal degrees Relative locations of stations are shown in the Patuxent River and Tangier Sound and do not reflect exact geographic locations. Exact latitude and longitude are given in Rohland et al. (1999).

4.3. Results

4.3.1 Results of Near-shore Water Quality Evaluation

4.3.1.1 Physical Parameters

A summary of physical parameters measured at all Patuxent River SAV monitoring sites is found in Table 4-3. Janes Island data is not listed because sampling was only conducted for a few weeks in the late spring and early fall and would have limited comparative value. The full data set is available in Boynton *et al*. (1999b).

4.3.1.2 Dissolved Nitrogen Concentrations (DIN)

Water column dissolved inorganic nitrogen (DIN) concentrations varied modestly at most sampling stations from April through October, possibly because of very low rainfall in the spring of 1999. Peak DIN concentrations (14.2 and 21.2 µM N) were found in April at the two most down-river stations (SV07 and SV09 respectively), and in September (28.1 µM N) at the most up-river station (SVBA) after several storm events (Figure 4-2, Table 4-3). However, throughout most of the SAV growing season, DIN concentrations were quite low and for the most part remained below the maximum DIN concentration (10.2 µM N) established for Tier II SAV habitat requirements (Batuik *et al*., 1992).

Despite differences in river flow and assumed nutrient loading rates between 1998 and 1999, no significant difference ($P > 0.05$) in mean DIN concentration was found at any site along the river. While mean DIN concentrations were lower at the two most down-river sites in 1999 compared to 1998, no statistical difference was found.

4.3.1.3 Dissolved Phosphorus Concentrations (DIP)

Water column dissolved inorganic phosphorus (DIP) concentrations varied seasonally at all stations (Figure 4-3, Table 4-3). For most stations, DIP concentrations increased marginally throughout the summer. However, the northernmost mesohaline station (SVBA) at Buena Vista increased dramatically over this period reaching a maximum of 2.18 µM P on 19 August, 1999. SONE monitoring in this region of the river indicates large sediment releases of DIP during summer seasons as a possible source of this phosphorus which is likely the result of anoxic conditions in deeper portions of the river.

On a seasonal basis, the highest mean DIP concentration was found at station SVBA (1.39 µM P). However, this concentration was significantly lower than in 1998. The lowest mean DIP concentration was found at station SV06 (0.15 µM P). Mean DIP concentrations at all stations were lower in 1999 compared to 1998, probably a result of lower than average river flow and subsequent nutrient loading. Seasonal mean values at all but station SVBA were below the maximum DIN concentration (0.32 µM N) established for Tier II SAV habitat requirements (Batuik *et al*., 1992).

Table 4-3. Table of the maximum, minimum and median values of most water column nutrient and physical water quality data recorded at the six mesohaline SAV sites visited in the Patuxent River from late April through October 1999.

		SVBA	SV02	SV5A	SV06	SV07	SV09
Temp (C^o)	Max	28.27	28.30	28.13	28.94	28.38	28.37
	Min	16.92	16.45	15.69	16.13	15.58	15.06
	Median	24.64	24.06	24.23	24.41	24.58	23.71
Salinity (ppt)	Max	14.67	17.07	17.61	17.32	18.10	18.57
	Min	0.80	11.95	12.05	4.01	11.80	12.15
	Median	12.22	14.29	14.61	14.35	15.21	15.68
Dissolved Oxygen (mg l^{-1})	Max	10.36	9.37	10.36	10.56	15.12	11.33
	Min	3.60	5.30	5.11	4.15	3.72	5.46
	Median	5.63	7.14	7.15	8.23	8.16	8.05
Total Suspended Solids (mg l-1)	Max	41.2	35.0	54.7	30.4	77.00	49.20
	Min	14.8	7.90	8.10	7.70	6.70	6.40
	Median	22.60	19.20	15.00	12.00	11.80	10.80
Total Chlorophyll-a (µg l-1)	Max	23.09	29.49	24.57	20.40	77.43	24.17
	Min	6.89	5.44	3.89	6.49	3.85	3.34
	Median	14.18	10.59	9.95	12.31	8.29	10.59
Dissolved Inorganic Nitrogen (µmol N)	Max	28.1	8.05	14.62	7.77	14.2	21.2
	Min	0.86	0.28	0.43	0.14	0.29	0.70
	Median	6.23	0.79	1.13	0.85	1.75	2.19
Dissolved Inorganic Phosphorus (µmol P)	Max	2.18	1.50	0.76	0.31	0.48	0.53
	Min	0.13	0.08	0.08	0.06	0.06	0.08
	Median	1.59	0.12	0.15	0.13	0.17	0.13
Light Attenuation Coefficient [Kd] (m^{-1})	Max	4.08	2.81	2.74	2.41	4.13	1.76
	Min	1.38	0.74	0.69	0.67	0.56	0.39
	Median	2.30	1.80	1.41	1.33	1.01	0.90

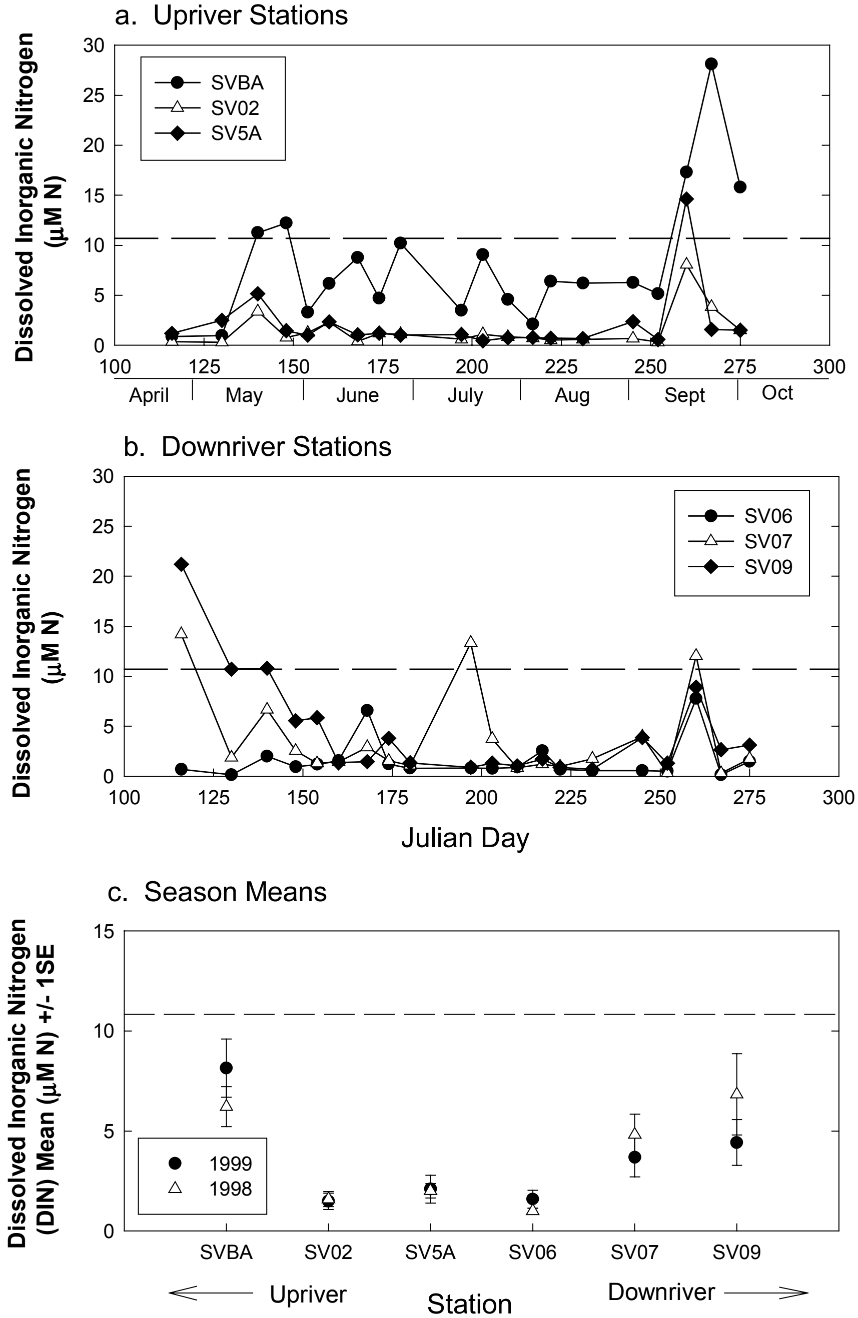


Figure 4-2. Dissolved inorganic nitrogen (DIN) concentrations for 1999 at a. upriver stations, b. downriver stations, and c. seasonal means for the Patuxent River (April through October 1998 and 1999).

Dashed lines represent minimum Tier II mesohaline SAV habitat requirement as specified in the Chesapeake Bay Submerged Aquatic Vegetation Requirements and Restoration targets: A technical synthesis (Batuik et al.,1992).

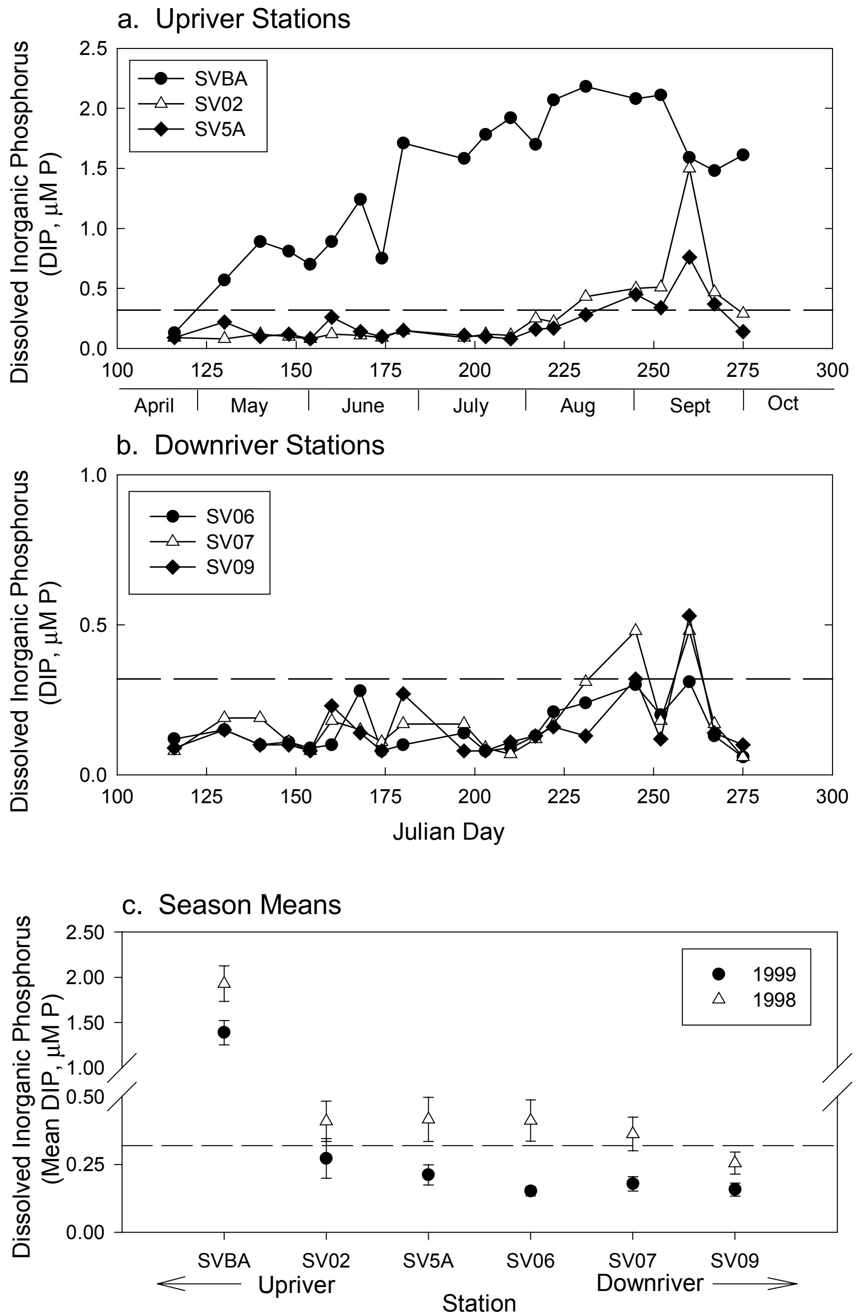


Figure 4-3. Dissolved inorganic phosphorus (DIP) concentrations for 1999 at a. upriver stations, b. downriver stations, and c. seasonal means for the Patuxent River (April through October 1998 and 1999).

Dashed lines represent minimum Tier II mesohaline SAV habitat requirement as specified in the Chesapeake Bay Submerged Aquatic Vegetation Requirements and Restoration targets: A technical synthesis (Batuik et al.,1992).

4.3.1.4 Water Column Light Attenuation

Water column light attenuation (Kd) did not appear to show any strong seasonal trends during the time period sampled (April – October). However, a strong spatial gradient was again found along the axis of the river (Figure 4-4, Table 4-3). Station SVBA (Buena Vista) had the highest mean light attenuation at (2.44 m^{-1}), while station SV09 (CBL) had the lowest at (1.02 m^{-1}). All but the two most up-river stations (SVBA and SV02) had mean light attenuation coefficients (Kd) below the recommended maximum Kd of 1.5 m^{-1} for Tier II restoration targets in mesohaline waters (Batuik *et al*., 1992). Water clarity along the axis of the river was significantly better ($P < 0.05$) in 1999 compared to 1998, indicating a mean improvement in water column Kd of 0.27 m^{-1}. This difference was likely the result of lower than average rainfall (and presumed nutrient loading) during the spring and summer of 1999.

4.3.1.5 Water Column Total Suspended Solids

Water column total suspended solids (TSS) did not show any distinct seasonal trends (Figure 4-5, Table 4-3). However, aperiodic peaks in total suspended solids were likely the result of isolated weather events resuspending bottom sediments. On a seasonal basis, there was a strong spatial gradient along the axis of the river with a maximum mean value of 24.78 mg l^{-1} at station SVBA and a minimum of 13.92 mg l^{-1} at station SV06 (Figure 4-5). This was not surprising since water column light attenuation is positively correlated with total suspended solids ($r^2 = 0.48$). Mean values in 1999 were not significantly different from 1998 ($P > 0.05$). On a seasonal basis, only stations SV06 (St. Leonard Creek) and SV09 (CBL) had mean values below the maximum TSS limit of 15 mg l^{-1} for mesohaline tier II habitat requirements (Batuik *et al*., 1992).

4.3.1.6 Water Column Chlorophyll-*a*

Median water column chlorophyll-*a* values at all stations was significantly lower ($P < 0.01$) in 1999 compared to 1998 (Figure 4-6, Table 4-3). All stations had mean chlorophyll-a concentrations below the maximum limit of 15 µg l^{-1} for mesohaline tier II habitat requirements (Batuik *et al*., 1992). However, there was no significant temporal trend detected during the sampling period. On a spatial basis, there was no significant down-river trend with a maximum seasonal mean value of 14.16 µg l^{-1} at station SVBA and a minimum seasonal mean of 11.55 µg l^{-1} at station SV06 (Figure 4-6).

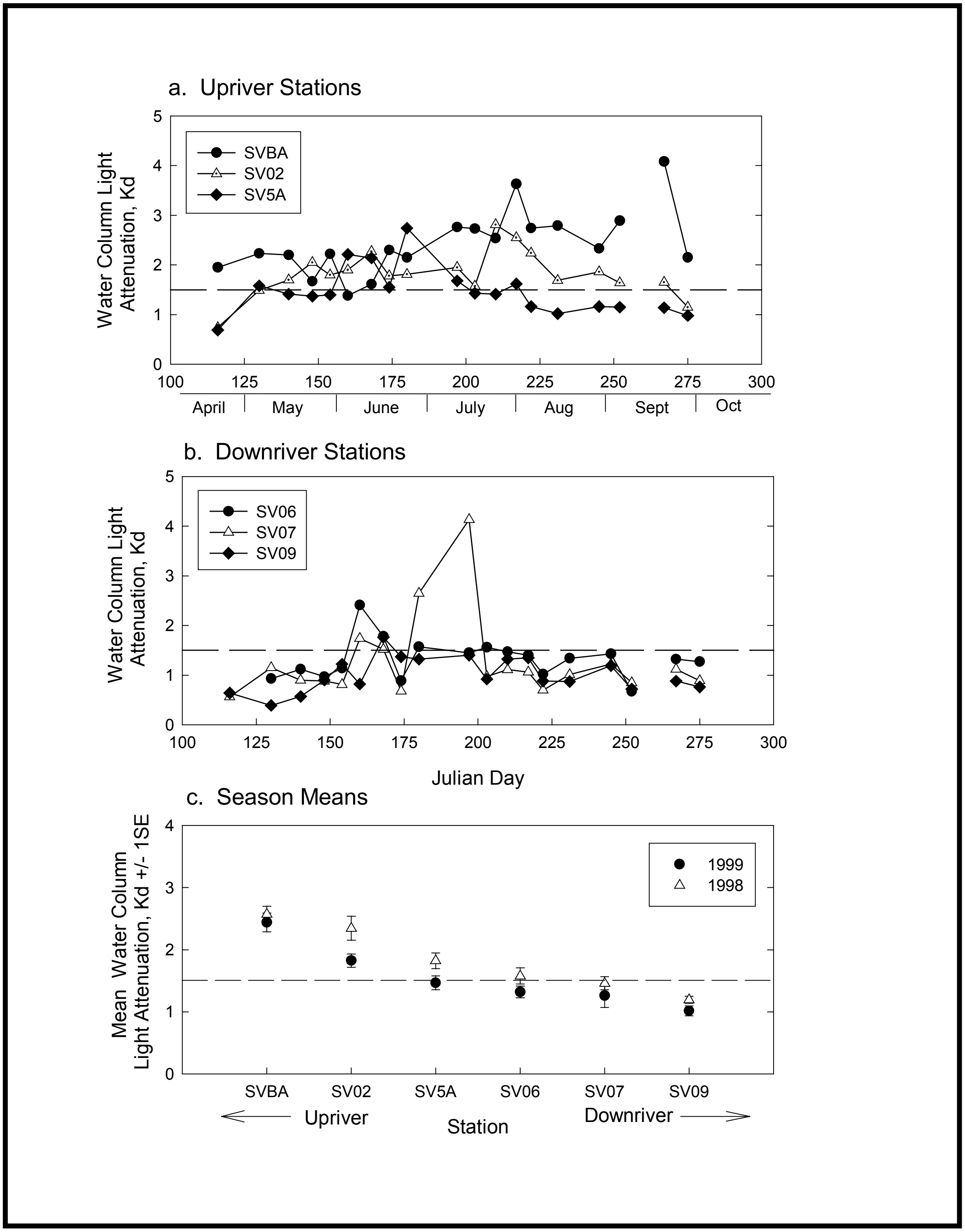


Figure 4-4. Water column light attenuation (Kd) for 1999 at a. upriver stations, b. downriver stations, and c. seasonal means for the Patuxent River (April through October 1998 and 1999).
Dashed lines represent the upper limit for Tier II mesohaline SAV habitat requirements as specified in the Chesapeake Bay SAV Technical Synthesis (Batuik et al.,1992).

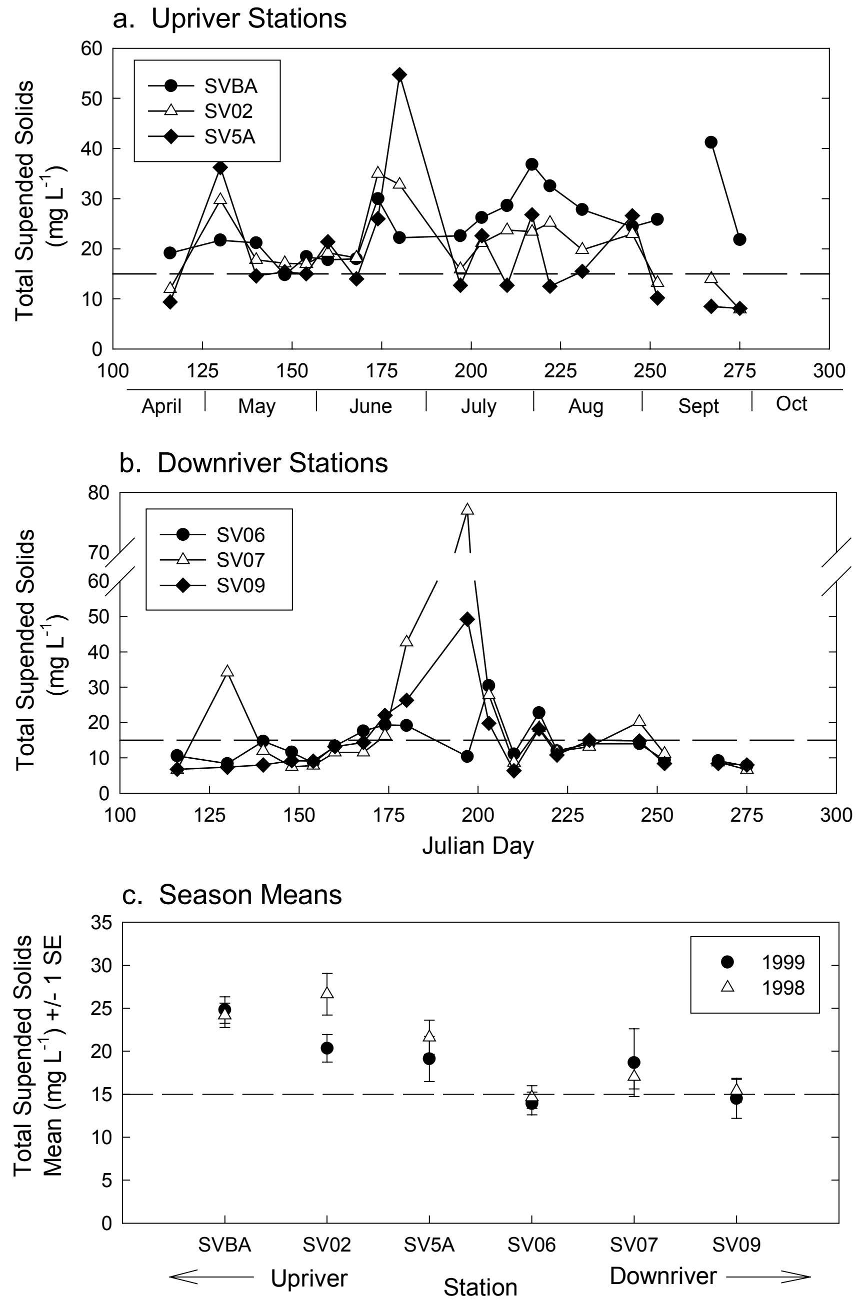


Figure 4-5. Water column total suspended solids (TSS) concentrations for 1999 at a. upriver stations, b. downriver stations, and c. seasonal means for the Patuxent River (April through October 1998and 1999).

Dashed lines represent minimum Tier II mesohaline SAV habitat requirement as specified in the Chesapeake Bay Submerged Aquatic Vegetation Requirements and Restoration targets: A technical synthesis (Batuik et al.,1992).

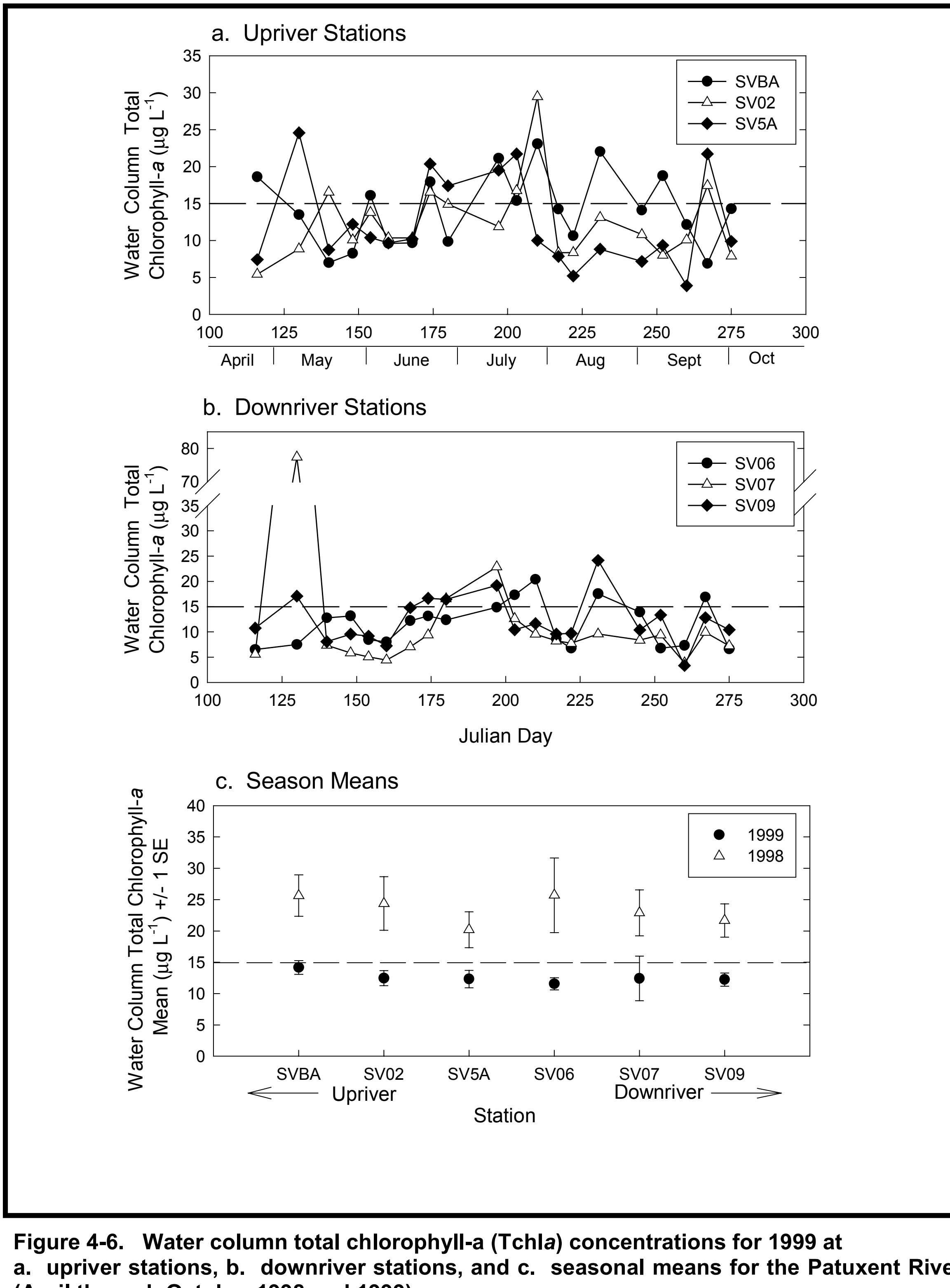


Figure 4-6. Water column total chlorophyll-a (Tchl*a*) concentrations for 1999 at a. upriver stations, b. downriver stations, and c. seasonal means for the Patuxent River (April through October 1998 and 1999).

Dashed lines represent minimum Tier II mesohaline SAV habitat requirement as specified in the Chesapeake Bay Submerged Aquatic Vegetation Requirements and Restoration targets: A technical synthesis (Batuik et al.,1992).

4.3.2 Results of Epiphyte Growth Study

4.3.2.1 Epiphyte Light Attenuation

Epiphyte accumulation rates were not measured on live plants in 1999. All epiphyte work conducted on the Patuxent River and near Janes Island was done with Mylar strips. The *in-situ* exposure period was also limited to 6-8 days.

Several important spatial and temporal patterns were observed in epiphyte light attenuation. Significant week to week variation was seen at most stations and was likely the result of weekly variation in weather and water conditions (Figure 4-7a, b). By May 1999, light attenuation at several up-river stations reached values between 80 - 100% of light exposure. However, fouling rates were extremely variable throughout the sampling season, with some sites experiencing very low fouling rates in July and August, which in past years was characterized by uniformly high fouling rates.

Spatially, epiphyte light attenuation did not show a substantial gradient down the axis of the river as in 1998 (Figure 4-7c).

4.3.2.2 Epiphyte Chlorophyll-*a* and Dry Weight Accumulation

Both epiphyte chlorophyll-*a* and dry weight exhibited a few significant aperiodic spikes in accumulation rates over the entire sampling period (Figures 4-8 and 4-9). This is likely the result of short-term variation in water and weather conditions and is also reflected in high weekly variation in epiphyte light attenuation. For this reason, seasonal medians are plotted instead of seasonal means for these parameters. Since epiphyte light attenuation is a result of both organic and inorganic material deposited to the Mylar surface it was not surprising that the temporal and spatial patterns for epiphyte chlorophyll-*a* and dry weight help explain the epiphyte light attenuation patterns.

The greatest median values for epiphyte chlorophyll-*a* accumulation rate (0.240 $\mu g\ cm^{-2}\ day^{-1}$) and dry weight accumulation (0.200 $mg\ cm^{-2}\ day^{-1}$) were found at SV09 (CBL). This was also the case in 1998 in which the highest fouling rates were also found at station SV09. The minimum median chlorophyll-*a* accumulation rate (0.035 $\mu g\ cm^{-2}\ day^{-1}$) and the minimum dry weight accumulation rate (0.040 $mg\ cm^{-2}\ day^{-1}$) were found at station SV06, along with the lowest mean epiphyte light attenuation.

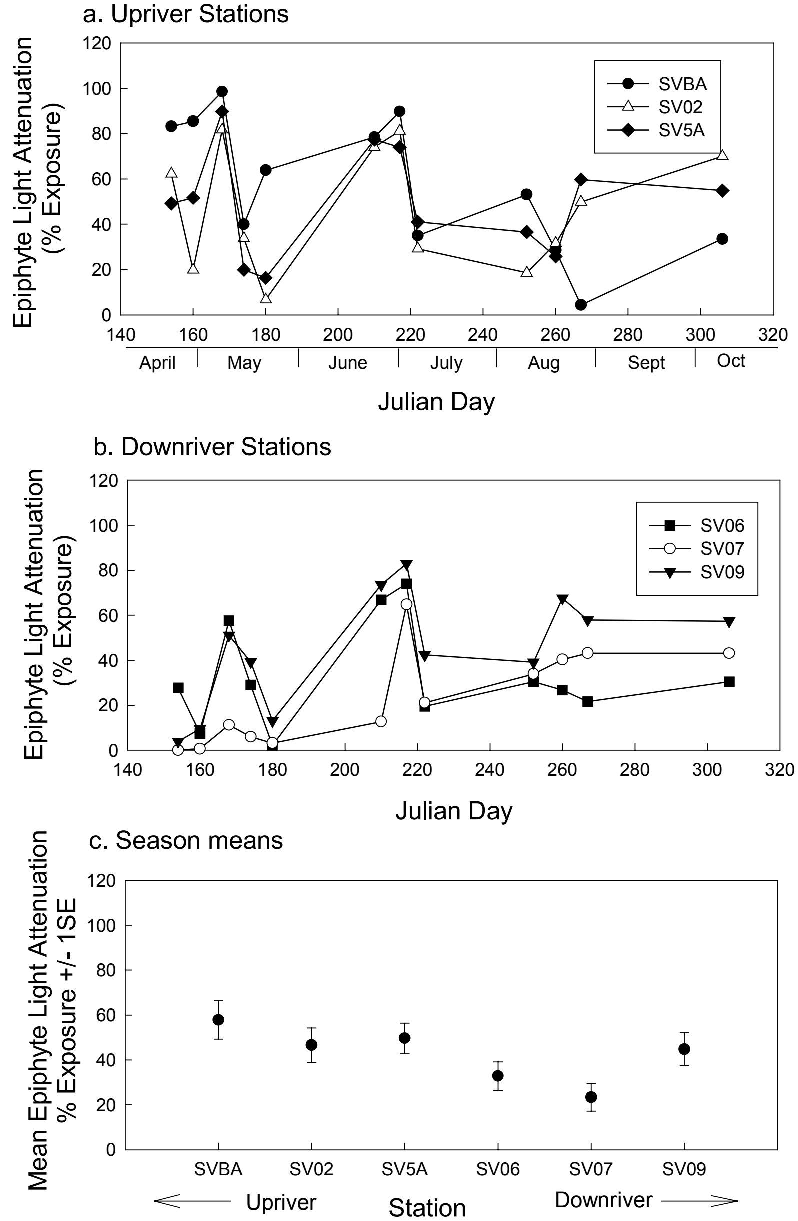


Figure 4-7. Epiphyte light attenuation through Mylar® strips deployed along the Patuxent River for *in-situ* exposures of 6-8 days from June through October 1999: a. upriver stations, b. downriver stations, and c. seasonal means.

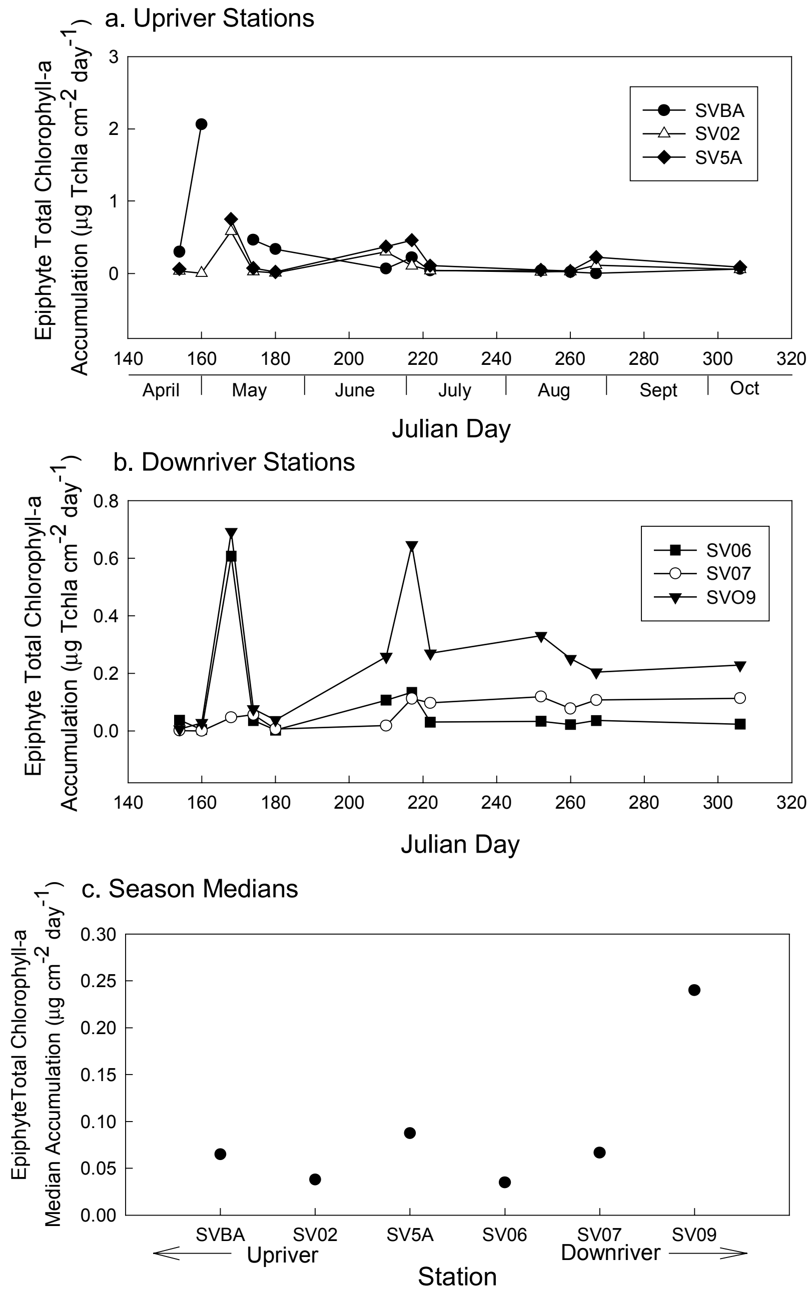


Figure 4-8. Epiphyte total chlorophyll-*a* accumulation on Mylar® strips deployed along the Patuxent River for *in-situ* exposures of 6-8 days from June through October 1999: a. upriver stations, b. downriver stations, and c. seasonal medians.

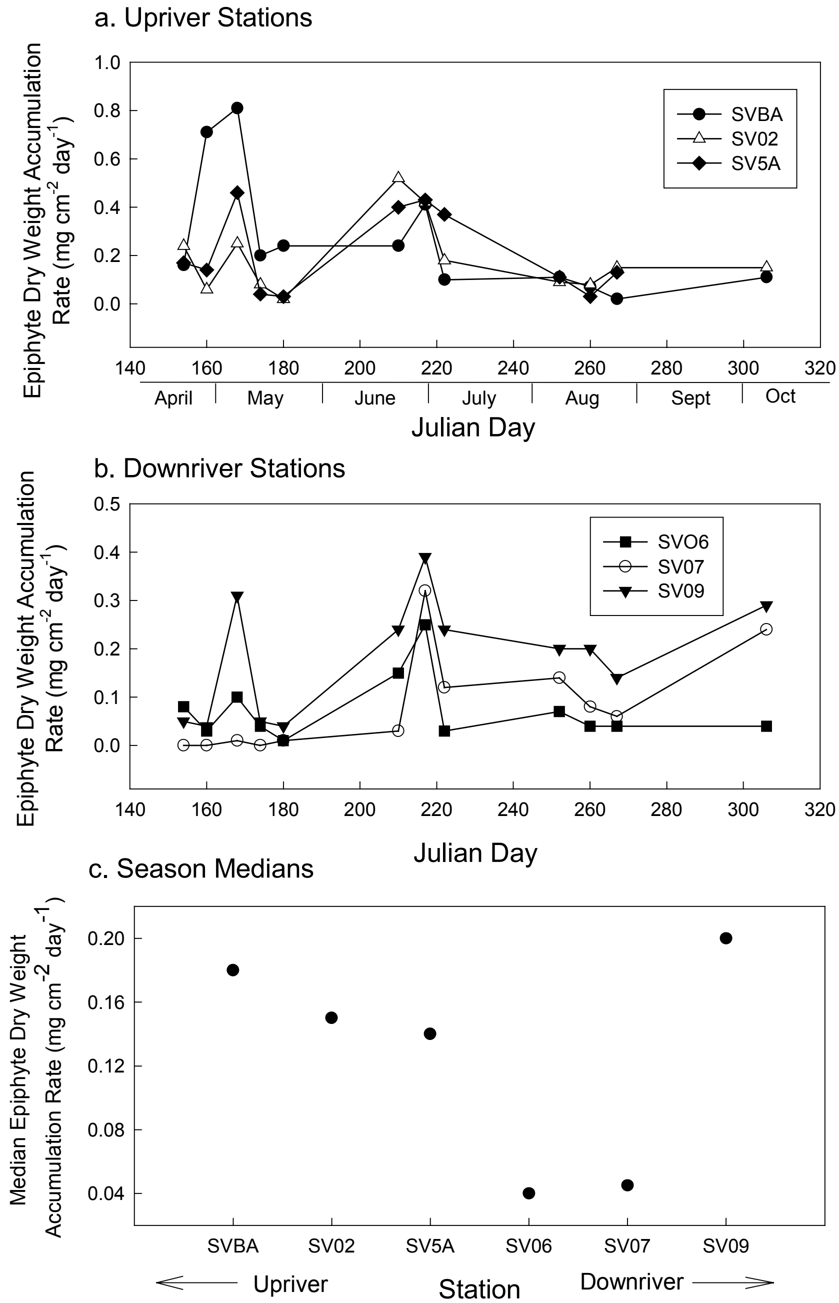


Figure 4-9. Epiphyte dry weight accumulation on Mylar® strips deployed along the Patuxent River for *in-situ* exposures of 6-8 days from June through October 1999: a. upriver stations, b. down-river stations, and c. seasonal medians.

4.3.2.3 Correlations and Relationships

With three years of data, strong relationships continue to be found between epiphyte light attenuation and epiphyte dry weight as well as between epiphyte light attenuation and epiphyte chlorophyll-*a* (Figure 4-10). These relationships provide good estimates of potential light attenuation to SAV leaves when epiphyte chlorophyll-*a* or dry weight can be measured. In addition, the data show that regardless of Mylar® strip deployment method, (1997 top down vs. 1998 bottom up; see Boynton *et al*., 1998) the relationships between epiphyte light attenuation and epiphyte biomass are similar. These relationships also show that relatively small amounts of epiphytes can attenuate a very large percentage of the available light. For example, only 2 mg cm^{-2} of epiphyte dry weight can attenuate almost 80% of available light. During most of the summer season, this amount of fouling can accumulate in as little as one week of *in-situ* exposure.

While epiphyte standing stock or accumulated biomass may be affected by external factors such as grazer densities (*e.g.* Neckels *et al.*, 1993) or hydrodynamic shear (*e.g.* Strand and Weisner, 1996), in the absence of these factors epiphyte growth rates are ultimately regulated by dissolved nutrient availability, temperature, and light flux. However, at any given point in time, only one of these parameters can actually limit or constrain the growth of epiphytes. This data set provides a unique opportunity to examine epiphyte chlorophyll-a accumulation rates along the Patuxent River and compare them to light and nutrient availability. Since deployment intervals were standardized across the entire season, and temperatures remained fairly constant, summer mean values can be used to compare chlorophyll-a accumulation rates against light and nutrient availability.

A plot of median epiphyte chlorophyll-a accumulation rates versus PLW (percent surface light through the water column) seems to indicate that factors controlling epiphyte growth differed significantly between 1998 and 1999. In 1998 epiphyte accumulation rates (as measured by chlorophyll-*a*) appeared to be light limited because of the strong relationship to available light ($r^2 = 0.97$). However, this relationship was not present in 1999. Since dissolved nutrient concentrations were slightly lower in 1999 compared to 1998 (Figure 4-10) and water column light attenuation (Kd) was significantly lower in 1999 compared to 1998 (Figure 4-10) variation in both these factors contributes to uncertainty in explaining differences in these patterns.

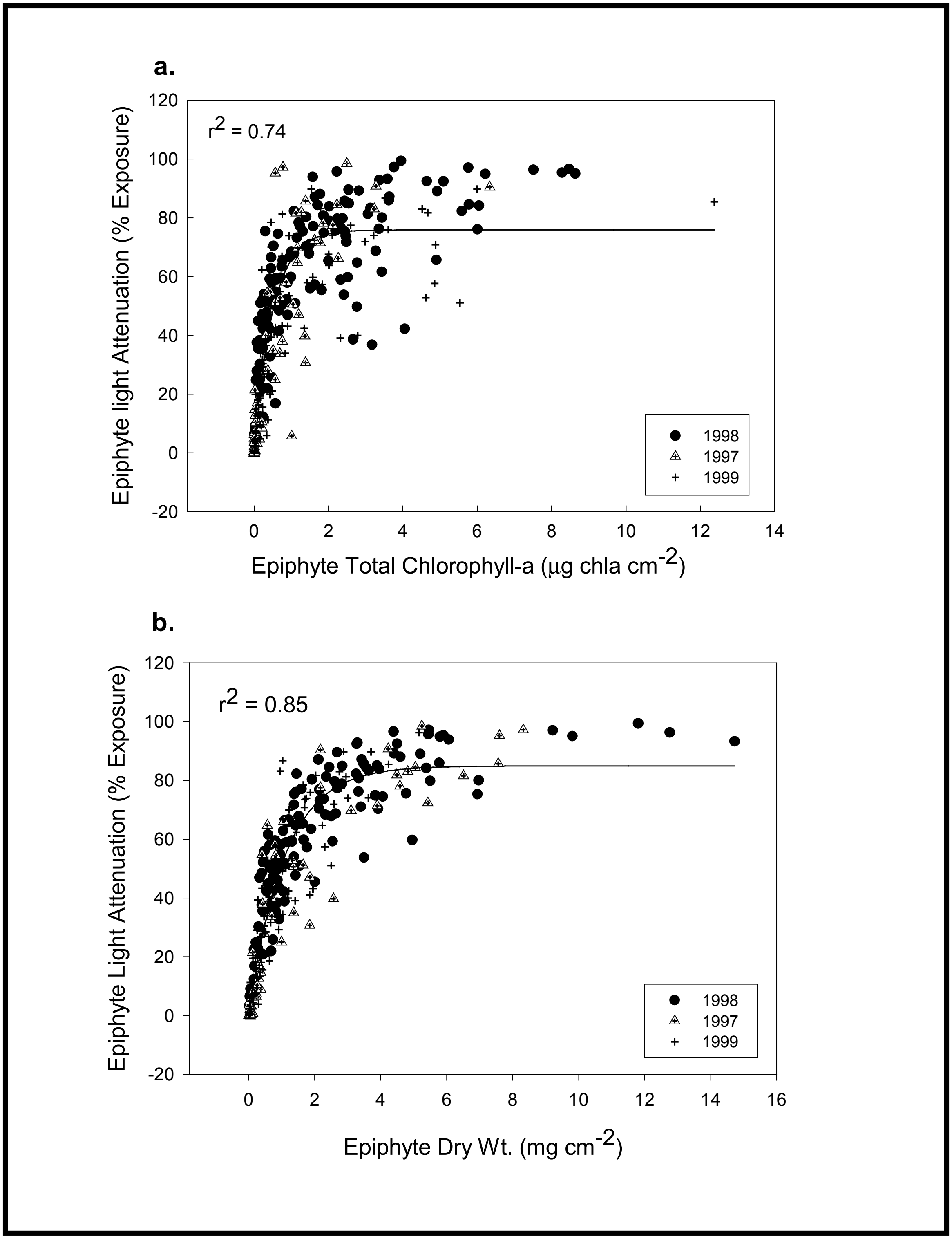


Figure 4-10. Epiphyte light attenuation from Mylar® strips deployed along the Patuxent River 1997, 1998, and 1999 versus:

a. epiphyte chlorophyll-a , where LA = 77.36*(1-exp(-2.082*Epchla), and

b. epiphyte dry weight , where LA = 84.638*(1-exp(-0.963*Epdw).

LA= epiphyte light attenuation, Epchla = epiphyte total chlorophyll-a cm^{-2}, and Epdw = epiphyte dry weight cm^{-2}.

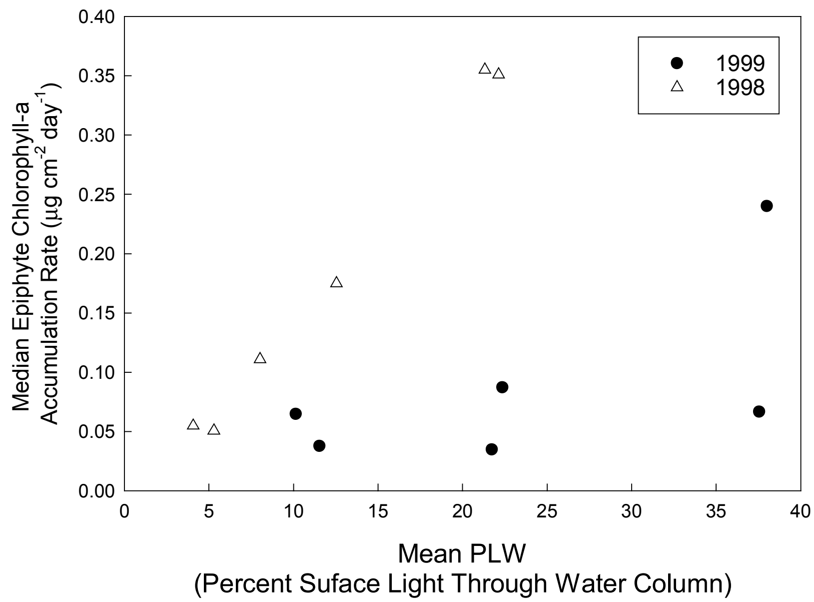


Figure 4-11. Estimated seasonal mean PLW (percent surface light through the water column) versus median epiphyte chlorophyll-a accumulation rates for Patuxent River 1998 and 1999.

4.3.3 Comparisons between Janes Island (Tangier Sound) and Patuxent River SAV Habitats

A comparison of SAV water quality data and epiphyte accumulation rates between Janes Island and the Patuxent River is particularly useful because healthy SAV beds are found at Janes Island but are absent from the Patuxent River sites. Although sampling at the two paired Janes Island sites only occurred during the late spring and early fall of 1999 (Table 4-2), a comparison to Patuxent River study sites during these same intervals remains useful because these intervals include portions of the active growing season for eelgrass *Zostera marina.*

4.3.3.1 Water Quality Parameters

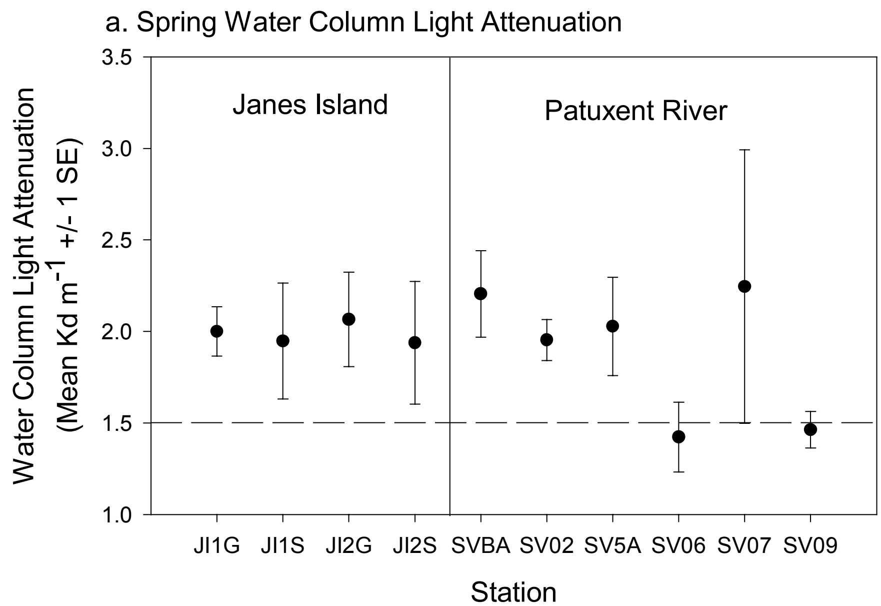
In the late spring and early summer sampling period (June 16 - July 16, 1999), mean water column light attenuation (Kd) at all the Janes Island sites and all but 2 of the Patuxent River sites was above the maximum mesohaline Kd limit of 1.5 m^{-1} for mesohaline tier II SAV habitat requirements (Batuik *et al*., 1992, Figure 4-12a). In contrast, during the late spring and early fall sampling period (September 1 – October 2, 1999) mean water column light attenuation at all the Janes Island sites was higher than 4 of the most down-river Patuxent River stations (Figure 4-12b). Based upon these limited observations, light attenuation at the Janes Island sites, where SAV is present, is comparable or even worse than several sites in the lower Patuxent River, which do not have SAV beds.

Mean water column dissolved inorganic nitrogen (DIN) concentrations at the Janes Island sites were also very similar to Patuxent River sites during the late spring and early summer sampling period (Figure 4-13). Only the most up-river Patuxent River site (SVBA) had a mean DIN concentration significantly higher than all other locations. During the late spring and early fall sampling period however, DIN concentrations at 3 of the 4 Janes Island sites was somewhat lower (although not statistically significant) than most of the Patuxent River sites. Once again, the mean DIN concentration at SVBA was significantly higher than all other sites.

Mean water column dissolved inorganic phosphorus (DIP) concentrations during the late spring and early summer sampling period were very similar to those found at all but the most up-river Patuxent River sites (Figure 4-14a). Only SVBA had a mean DIP concentration significantly higher than all other sites which is likely the result of large phosphorus releases from deeper sections of the river during anoxic events near this station. In contrast, during the late spring amd early fall sampling interval, DIP concentrations at all Janes Island sites were lower than any Patuxent River site. This appears to be the one instance where water quality conditions differed measurably between Janes Island and lower Patuxent River locations.

4.3.3.2 Epiphyte Accumulation Rates

Epiphyte sampling was limited to a few weeks in each season (some sites and seasons only have two rate measurements) so caution should be exercised when making comparisons between locations and season (for that reason, no error bars were placed around the estimates of mean fouling rates). Based upon limited sampling in the late spring and early summer and late spring and early fall sampling periods, epiphyte fouling rates (as measured by chlorophyll-a) do not appear to be significantly different at Janes Island compared to Patuxent River locations in the spring. While strictly suggestive, it also appears that epiphyte fouling rates at the 2 unvegetated (sand) stations at Janes Island were slightly higher than at the 2 vegetated sites. It is unknown at this time the reason for this difference. However during the late summer and early fall sampling interval, the two down-river Patuxent sites (SV07 and SV09) appear to have epiphyte fouling rates higher than those found at Janes Island. No statistical tests were performed on these data.



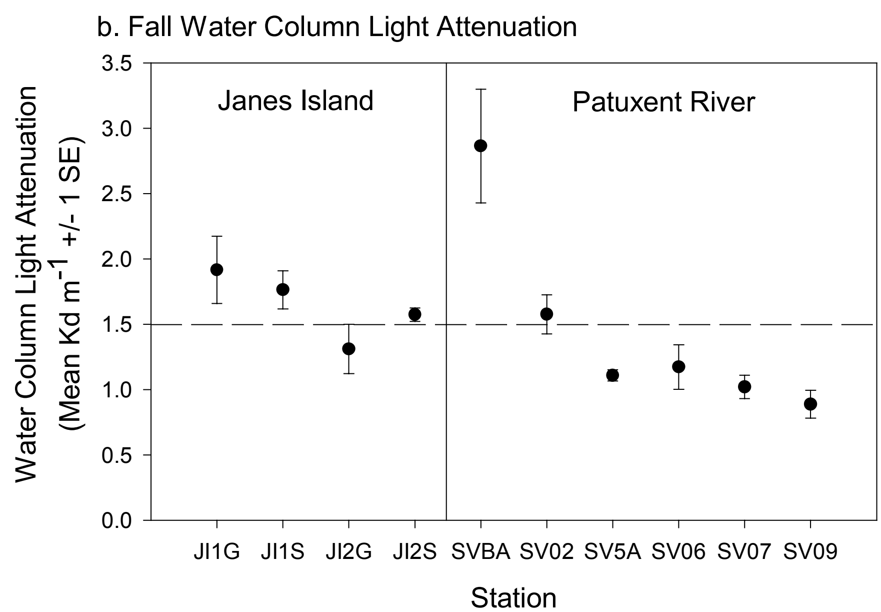


Figure 4-12. Comparison of water column light attenuation (Kd) at Janes Island sites to Patuxent River sites during
a. June 16 – July 16, 1999, and b. September 1 – October 2, 1999.
Dashed lines represent minimum Tier II mesohaline SAV habitat requirement as specified in the Chesapeake Bay Submerged Aquatic Vegetation Requirements and Restoration targets: A technical synthesis (Batuik et al., 1992).

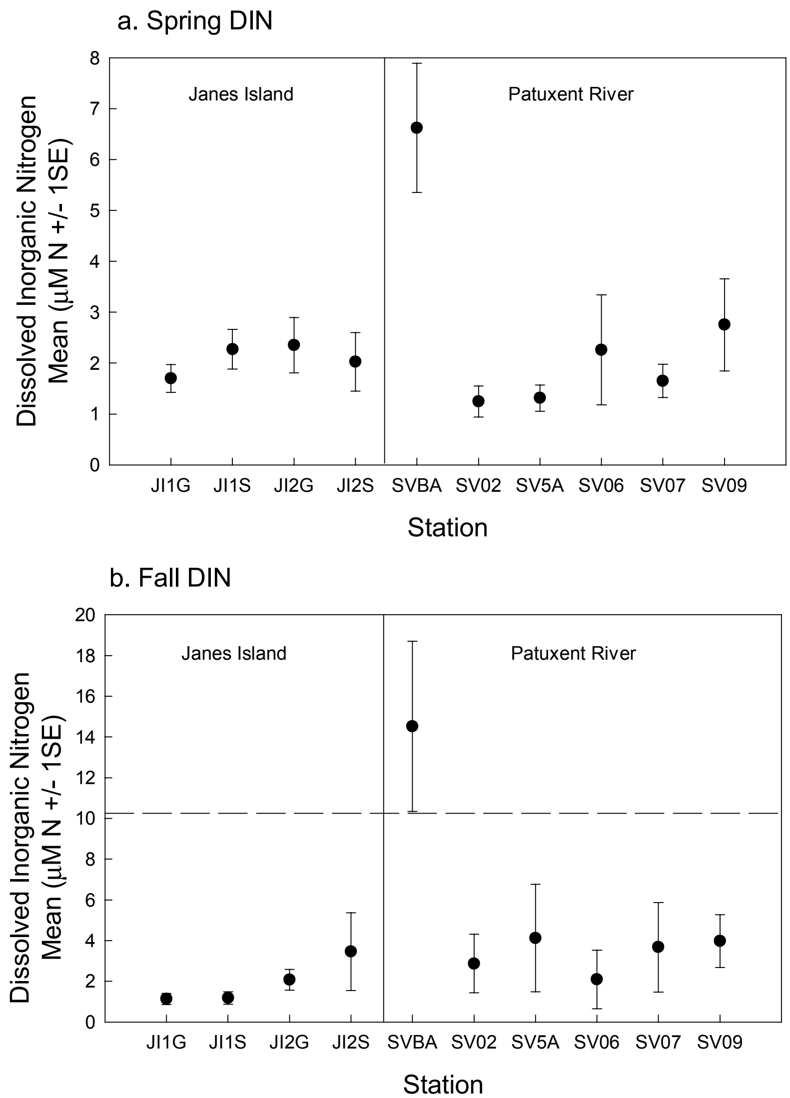


Figure 4-13. Comparison of water column DIN concentrations at Janes Island and Patuxent River
a. spring 1999, and b. fall 1999.
Dashed lines represent minimum Tier II mesohaline SAV habitat requirement as specified in the Chesapeake Bay Submerged Aquatic Vegetation Requirements and Restoration targets: A technical synthesis (Batuik et al.,1992).

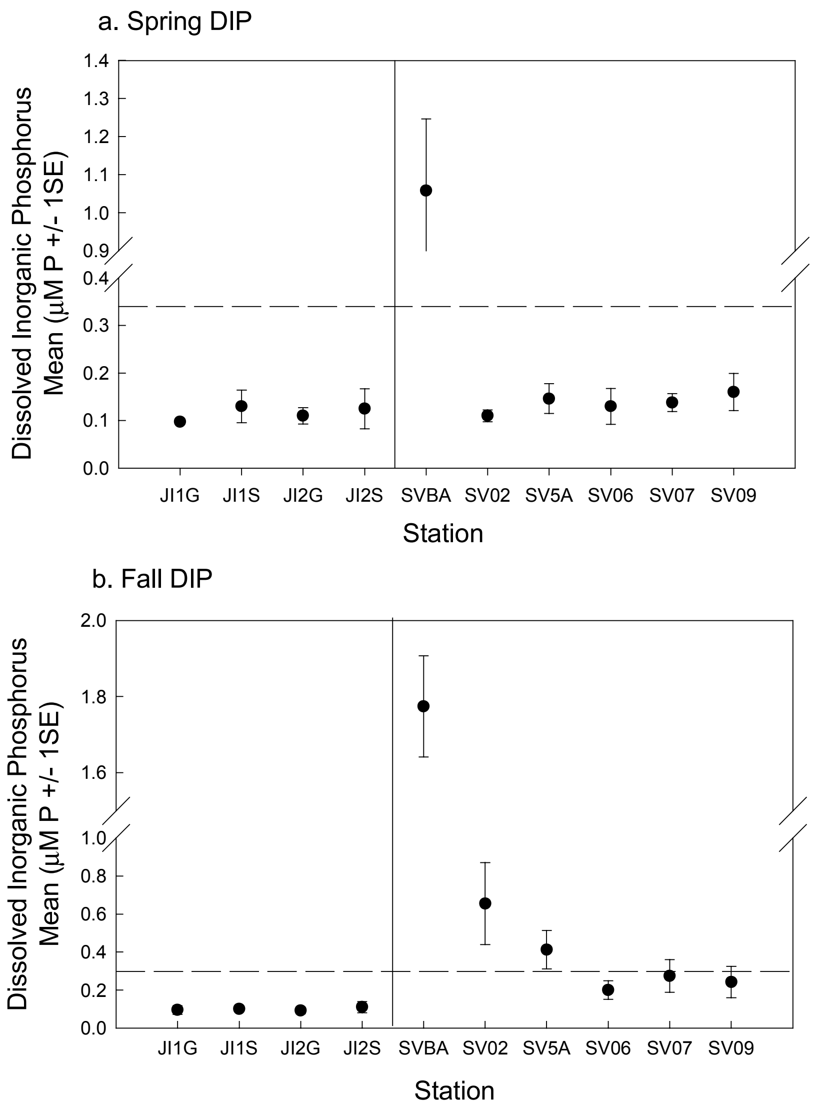


Figure 4-14. Comparison of water column dissolved inorganic phosphorus at Janes Island sites to Patuxent River sites during
a. June 16 – July 16,1999, and b. September 1 – October 2, 1999.
Dashed lines represent minimum Tier II mesohaline SAV habitat requirement as specified in the Chesapeake Bay Submerged Aquatic Vegetation Requirements and Restoration targets: A technical synthesis (Batuik et al., 1992).

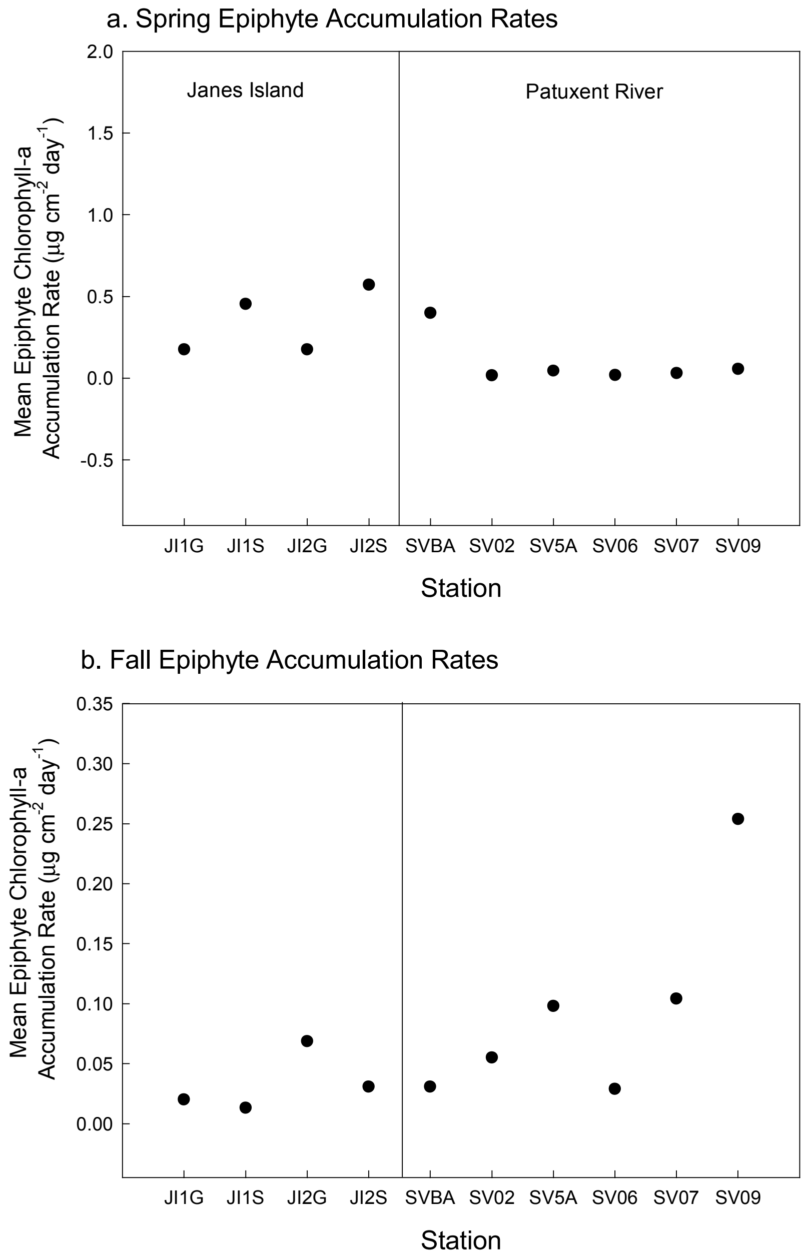


Figure 4-15. Comparison of Epiphyte chlorophyll-a accumulation rates at Janes Island sites to Patuxent River sites during
a. June 16 – July 16, 1999, and b. September 1 – October 2, 1999.

4.4 Discussion and Conclusions

4.4.1 Near-shore Water Quality Evaluation

In the Patuxent River, near-shore water quality during the 1999 SAV growing season was improved relative to 1997 and 1998 for virtually every parameter measured and every station sampled. For example, water column light attenuation was improved at every station with four of the most down-river stations meeting or exceeding mesohaline tier II habitat requirements compared to only three stations in 1998 (Batuik *et al.*, 1992). Dissolved nutrient concentrations were also lower in 1999 compared to recent years. Seasonal means at every station fell well below the DIN mesohaline tier II habitat limits and all stations except SVBA (the most up-river station) fell within DIP mesohaline tier II habitat limits (Batuik *et al.*, 1992). While water column total suspended solids (TSS) was lower at most stations in 1999 compared to recent years, water column chlorophyll-a concentrations were dramatically lower in 1999 at every station sampled. These changes are most likely the result of a lower than average spring river flow and subsequent reductions in nutrient loading.

Since water quality sampling at Janes Island only occurred in later spring and early fall, only a limited comparison to Patuxent River water quality conditions is possible. However, no patterns were found that immediately suggest reasons why healthy SAV beds remain near Janes Island but are absent from much of the Patuxent River.

For example, water column light attenuation at the Janes Island sites (both vegetated and unvegetated) was not significantly lower than many locations sampled on the Patuxent River (Figure 4-12). In fact, water clarity at several Patuxent River sites was superior to that found at Janes Island, even though Patuxent River sites (especially SV09) do not support naturally occurring SAV beds. While two Patuxent River sites (SV06 and SV07) have supported measurable beds *of Zannichellia palustris* during spring sampling periods, these beds usually senesce by the beginning of June and probably do not modify the local environment to any great degree. Based upon this limited data set, it appears that in 1999 dissolved nutrient concentrations (DIN and DIP) at Janes Island sites were comparable to Patuxent River sites in the fall. Since near-shore water quality conditions can vary significantly over short time scales as a result of local wind and weather conditions, a comparison limited to a few weeks of observations may not be adequate to make a true comparison between regions. For this reason it is not surprising that observed water quality conditions at Janes Island and Patuxent River do not immediately explain differences in SAV coverage between these two locations.

4.4.2 Epiphyte Growth Study

Although a number of studies have contributed to our understanding of the complex interactions between epiphyte growth, nutrient dynamics and SAV survival, a full understanding of these processes is not yet complete. However, several important patterns and observations have emerged from this work that may provide some insight into SAV epiphyte growth and light attenuation.

The relationships between epiphyte biomass (expressed as either chlorophyll-a or dry weight) versus percent light attenuation appear to be relatively robust and may be used to obtain first order estimates of light attenuation from epiphyte biomass (Figure 4-10). Despite differences in the deployment method of Mylar strips between 1997 and 1998/1999 (see Boynton et al., 1998), a fixed mass of epiphyte material expressed as either dry weight or chlorophyll-a biomass will attenuate the same amount of light regardless of the way in which the material was exposed to fouling. These relationships also compare favorably to those found by Burt *et al*. (1995) for artificial substrates exposed for 80 days off the western coast of Australia. In both studies, an epiphyte chlorophyll-a biomass as small as 1 μg cm^{-2} predicts approximately a 60% reduction in the available light.

While it is important to know how much light may be attenuated by a fixed mass of epiphytic material, an equally important measure is the rate of epiphyte accumulation. In general, fouling rates on the Patuxent River appear to be quite high compared to most other studies reported in the literature. However, the typical deployment intervals from other studies ranged from 60 – 100 days (Horner, 1987; Burt *et al*., 1995; Pinckney and Micheli, 1998) so exact comparisons are difficult. Even with these relatively short *in-situ* exposure times, mean light attenuation by the epiphyte layer was substantial and ranged from 23% at station SV07 to 58% at station SVBA. In addition, these values were somewhat lower than fouling rates in 1998 where the maximum mean light attenuation was 75% at SV5A. In 1997 and 1998 Mylar strips exposed for longer in-situ intervals accumulated significantly more epiphyte biomass. Thus, it seems likely that SAV blade tissue older than 6-8 days would likely be fouled to an even greater degree and be subjected to a corresponding higher light attenuation.

Because epiphyte sampling at the Janes Island locations was restricted by a short sampling window, lost epiphyte collectors and storms, only the most cursory comparison to fouling rates along the Patuxent River is possible. With those considerations, it appears that in 1999 epiphyte fouling rates (as measured by epiphyte chlorophyll-a) in the spring were comparable if somewhat higher than locations on the Patuxent River (Figure 4-15a). However, this sampling was done during a period when fouling rates begin to ramp up as temperatures increase. It is possible that sampling at Janes Island even a week later than Patuxent River sampling would be enough to generate these observed differences. The same rational could be used to explain differences during the fall sampling. Epiphyte fouling rates at Janes Island during the early fall also appear to be comparable to many locations on the Patuxent River (Figure 4-15b). Only Station SV09 (most down-river station) had fouling rates higher that all others. Station SV09 is also the only down-river Patuxent station not to support substantial natural beds of SAV during at least part of the season (very small patches of *Ruppia maritima* were found at SV09 in 1999). It is possible that extremely high fouling rates during the summer and fall at this station reduce light availability to the leaf surface to levels below what SAV can tolerate on an annual basis. This hypothesis seems possible considering that water quality conditions alone (Kd, and dissolved nutrients) at SV09 are well below mesohaline tier II habitat criteria.

Because a full suite of water quality parameters is collected as part of this study it allows us some insight into conditions that may be necessary or sufficient to support abundant epiphyte growth. In 1998, a strong relationship between epiphyte chlorophyll-a accumulation and light availability suggested light, rather than nutrients, limited epiphyte growth along the Patuxent River. This was particularly relevant because mean concentrations of dissolved nutrients already fell below maximum habitat criteria established for SAV restoration (Batuik *et al*., 1992). However, in 1999, water quality conditions along the mesohaline portion of the river were improved compared to 1998 and epiphyte chlorophyll-a accumulation rates appear to be controlled by other factors rather than simple light availability. A further analysis and possible experimentation will be required to separate the effects of light and nutrients on epiphyte fouling in this portion of the Patuxent River. Further monitoring of the Patuxent River and other Chesapeake Bay regions may help answer some of these questions. Lastly, this data should be used to help calibrate the models that the Chesapeake Bay Program is currently developing to predict epiphyte light attenuation.

4.4.3 Observations regarding SAV transplant success on the lower Patuxent River

In recent years there have been a variety of small and moderate SAV transplant efforts in the lower Patuxent River with different degrees of success. Perhaps the most encouraging has been the persistence of a small test patch (less than 0.25 m^2) of *Zostera marina* transplanted to the sand flat in front of CBL (SV09) in the summer of 1998 as part of a short-term experiment. Despite high summer fouling epiphyte fouling rates, and the loss of all above ground biomass, this patch has survived through the spring of 2000. While this patch has not grown or spread to any great extent, its survival is a good sign that conditions may be adequate at CBL for longer term survival. This initial success was followed by another small test patch planted in the fall of 1999 with the purpose of evaluating longer term survival. This second test patch did very well during the fall growing season and was in fact quite healthy and beginning to spread in January 2000, when a pair of mute swans or diving ducks found both patches and removed virtually all above ground biomass. Fortunately, both patches were able to recover during the spring growing season despite poor water clarity compared to recent years. Finally in the spring of 2000, a larger test planting was done at CBL with approximately 1500 *Z. marina* shoots in three large plots. One plot has been caged with storm fencing to eliminate disturbance from birds or rays. Thus far, the transplants seem to be tolerating the conditions well. A formal evaluation of the transplant will be conducted in the fall.

In addition to this work at CBL the Alliance for the Chesapeake has also been transplanting a variety of species to an area just north of Point Patience. Several small test patches of *Z. marina* and sago pondweed *Potomogeton pectinatus* planted in the fall of 1999 have survived to the spring of 2000. In April of 2000, at this same location the Alliance for the Chesapeake planted approximately 5000 shoots in a mixture of *Z.*

marina, R. maritima, and *P. pectinatus.* At this point, the success of this effort cannot be determined.

Further monitoring of these transplants will provide valuable insight into possible long-term SAV survival in the lower Patuxent River. However, other species that cannot rely on stored carbohydrate reserves may be more susceptible to declining water quality conditions and epiphyte loading during summer months. For example, a healthy bed of *Potomogeton pectinatus* was observed at station SV07 during the spring and early summer of 1997 indicating adequate water quality conditions during this time. However, by August of 1997 the entire bed had died. Although the exact cause of this die-off is unknown, high epiphytic fouling rates were observed during this time and may have contributed to the total light attenuation at this site. Unfortunately, this bed of *P. pectinatus* did not return in 1998.

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5. HIGH RESOLUTION MAPPING OF WATER QUALITY IN TANGIER SOUND

J.D. Hagy and W.R. Boynton

5.1 Introduction

The Tangier Sound region of Chesapeake Bay is surrounded to the west by the complex of sparsely populated islands that includes Tangier Island to the south and Smith and Bloodsworth Islands to the north. To the east, the sound is flanked by a series of small estuarine rivers that drain largely rural, agricultural watersheds. Other than the deep central channel of the Sound itself, shallow water depths (<3 m) prevail throughout much of the region. These areas have historically supported extensive beds of submersed aquatic vegetation (SAV), which was one likely reason for the once thriving fishery for soft crabs (intermolt individuals of the blue crab *Callinectes sapidus*) in the region.

In the 15 years since regular aerial surveys began, the total area of SAV in Chesapeake Bay and its tributaries has increased (EPA, 1999). However, the trend in SAV abundance in Tangier Sound has been downward since the early 1990s (EPA, 1999). This has prompted increased interest in understanding the possible causes of this change in Tangier Sound. Bay-wide, declines in SAV abundance from historical levels have been attributed in part to nutrient enrichment, which reduced the light available to SAV by promoting growth of phytoplankton in the water column and epiphytic algae growing directly on the leaves of the SAV. In addition, suspended solids provide an additional source of water column light attenuation.

The relevant water quality parameters needed to assess suitability of water quality in the Tangier Sound region for SAV have been collected since 1985/86 by MD DNR at 7 stations in the study area (Figure 5-1). This study utilized a high-resolution surface water quality mapping system (DATAFLOW IV; similar to system described in Madden and Day, 1992) to evaluate the nature of spatial variability in water quality in the Tangier Sound region. The study was designed as a pilot study to identify (1) what features of the overall water quality distribution might be captured by the regular monitoring stations, (2) what features might be overlooked and (3) how high resolution mapping might be used to better inform for water quality management efforts in Tangier Sound.

5.2 Sampling Design, Methods and Calibration

The design of the DATAFLOW IV surface water quality mapping instrument is described in detail in Rohland *et al.* (1999). This includes specifications of the water quality sensor systems and the field and laboratory procedures for calibration. Only specific features of the study design and calibration that bear directly on the results are presented in this report.

5.2.1 Sampling Track

An approximately 400 nautical mile track throughout the Tangier Sound region was transited twice in 1999, once during July 24-25 and once during October 12-13 (Figure 5-7). This route was designed to provide reasonable spatial coverage of all regions of the study area. In addition, the route emphasized onshore to offshore transects so as to characterize surface water quality in waters with total depths from ranging from the shallowest nearshore regions to the deepest channels. Approximately 50 inshore-offshore transects were transited during each sampling period. The vessel draft was approximately 0.6 m. Sampling was frequently conducted in waters of less than 1.5 m total depth and somewhat less often in waters less than 1.0 m deep. The exact vessel track is plotted on each water quality map so that the reader can evaluate independently which features of water quality illustrated on the maps are justified by the data and which may be artifacts of the interpolation and contouring process that was used.

5.2.2 Stop Stations and Field Calibrations

Sixteen stop stations were occupied during each of the two study periods (Figure 5-1). As described by Rohland *et al.* (1999), the data collected at these stations were required to provide field calibrations for the transmissometer and fluorometer in the DATAFLOW IV system. The data collected included direct measurements of down-welling PAR (photosynthetically active radiation) at 0.1, 0.5 and 1.0 m depth and secchi depth. We also noted the wave height and approximate wind speed and direction. Water samples were collected for later determinations of total and active chlorophyll-*a*, total suspended solids, and dissolved oxygen (Winkler titrations).

<u>Chlorophyll-*a*</u>

The *in vivo* fluorescence values observed using DATAFLOW explained most of the variation in chlorophyll-*a* concentrations during both the summer (r^2=0.93; Figure 5-2) and fall cruises (r^2=0.85; Figure 5-2). While the parameters of the regression functions were different in the two cruises (Table 5-1), this is not unexpected. Several factors are known to influence *in vivo* fluorescence besides chlorophyll-*a* concentration. Examples include past light exposure and current nutrient status of the phytoplankton (Kiefer 1973a, 1973b).

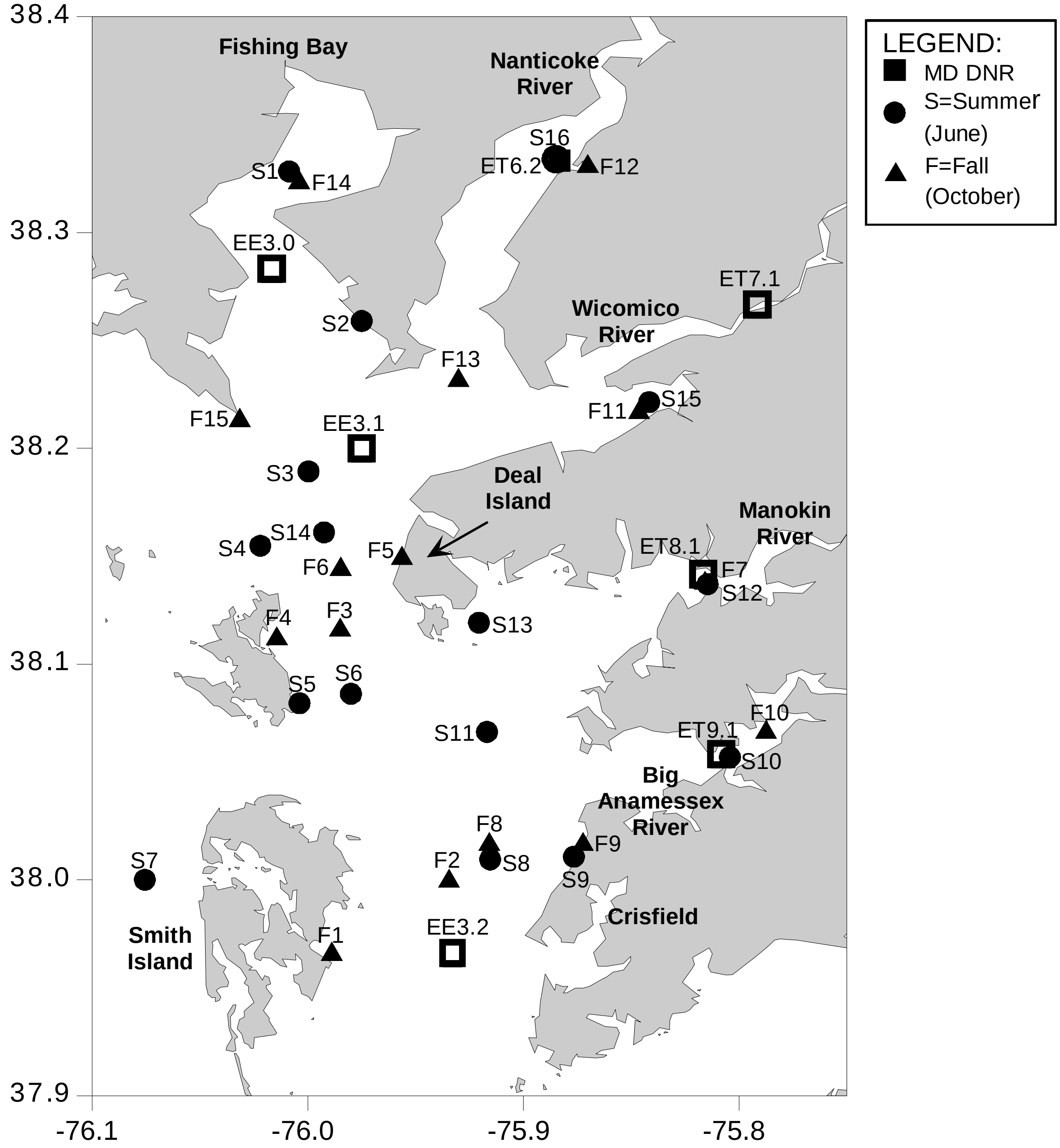

Figure 5-1. The location of Chesapeake Bay water quality monitoring program (CBMP) stations and the stop stations on both the June (S) and October (F) cruises to Tangier Sound.

Relative locations of stations are shown in Tangier Sound and do not reflect exact geographic locations.

Table 5-1. Parameters of linear regressions (Model II) relating total and active chlorophyll-*a* concentrations measured via laboratory analysis to *in vivo* fluorescence values obtained using the DATAFLOW instrument. All regressions are of the form $Y = a + b(V)$ where *V* is the *in vivo* fluorescence.

	June 24-25		October 12-13	
	Slope (β)	Intercept (α)	Slope (β)	Intercept (α)
Total Chl-*a*	18.38	-0.07	32.11	-6.18
Active Chl-*a*	15.87	-0.09	28.94	-6.00

Calibrations did not appear to vary regionally within the study area with the exception of stations 3, 5,6, and 14 during the June cruise. These stations had a chlorophyll-*a* concentration 2-3 μg l^{-1} higher than at other stations with similar fluorescence values. It is not clear what sets these stations apart from the others. The are all located in open water on the western side of Tangier Sound, rather than in a tributary. However, station 4, which had chlorophyll-*a* values on the regression line, was also located on the western side of Tangier Sound. Each of these stations was occupied in the afternoon of June 24; however, station 4 was also occupied on the afternoon of June 24. Station 4 and other stations occupied on the following afternoon all had chlorophyll-*a* concentrations in agreement with the regression line. With the uncertainties in mind, the chlorophyll-*a* concentrations used for generating June maps were calculated using the regression line fitted to all observations other than stations 3, 5, 6 and 14. For the October cruise, all observations were included in the regression.

Water Clarity and Total Suspended Solids

The DATAFLOW transmissometer measures transmittance of light through a 10 cm column of water and returns a voltage between approximately 0 V(blocked path reading) and 4.85 V, the value obtained in air. Deionized water returns only a slightly lower reading than that obtained in air. In Tangier Sound, transmittance varied from less than 0.07 V to approximately 3.53 V. The inverse of secchi depth was linearly related to transmissometer voltage in both June (r^2=0.95) and October (r^2=0.93; Figure 5-3, Table 5-2), allowing highly confident predictions of estimated secchi depth to be made using transmissometer data. Light attenuation coefficient (Kd) was also highly correlated with transmissometer (June: r^2=0.80, October: r^2=0.94; Figure 5-5, Table 5-2), allowing good predictions of light attenuation coefficient from transmissometer readings. Finally, total suspended solids (TSS) concentrations were also strongly correlated with transmissometer voltage (June: r^2=0.93, October: r^2=0.89; Figure 5-5, Table 5-2). For TSS, however, there were several clear outliers where TSS was higher than predicted by the regression based on the remaining data. There may be a clear explanation for the October outliers. Two of three (stations 1 and 4) observations were in very shallow water (1 m) during a time of very intense wave-driven resuspension (0.5m breaking waves). Large particles could be resuspended in these energy regimes, but may not contribute to light attenuation as much as an equivalent mass of smaller particles. Station 2 in October, which also had higher than expected TSS was in deeper waters, but was also occupied during a time of high wind and wave energy.

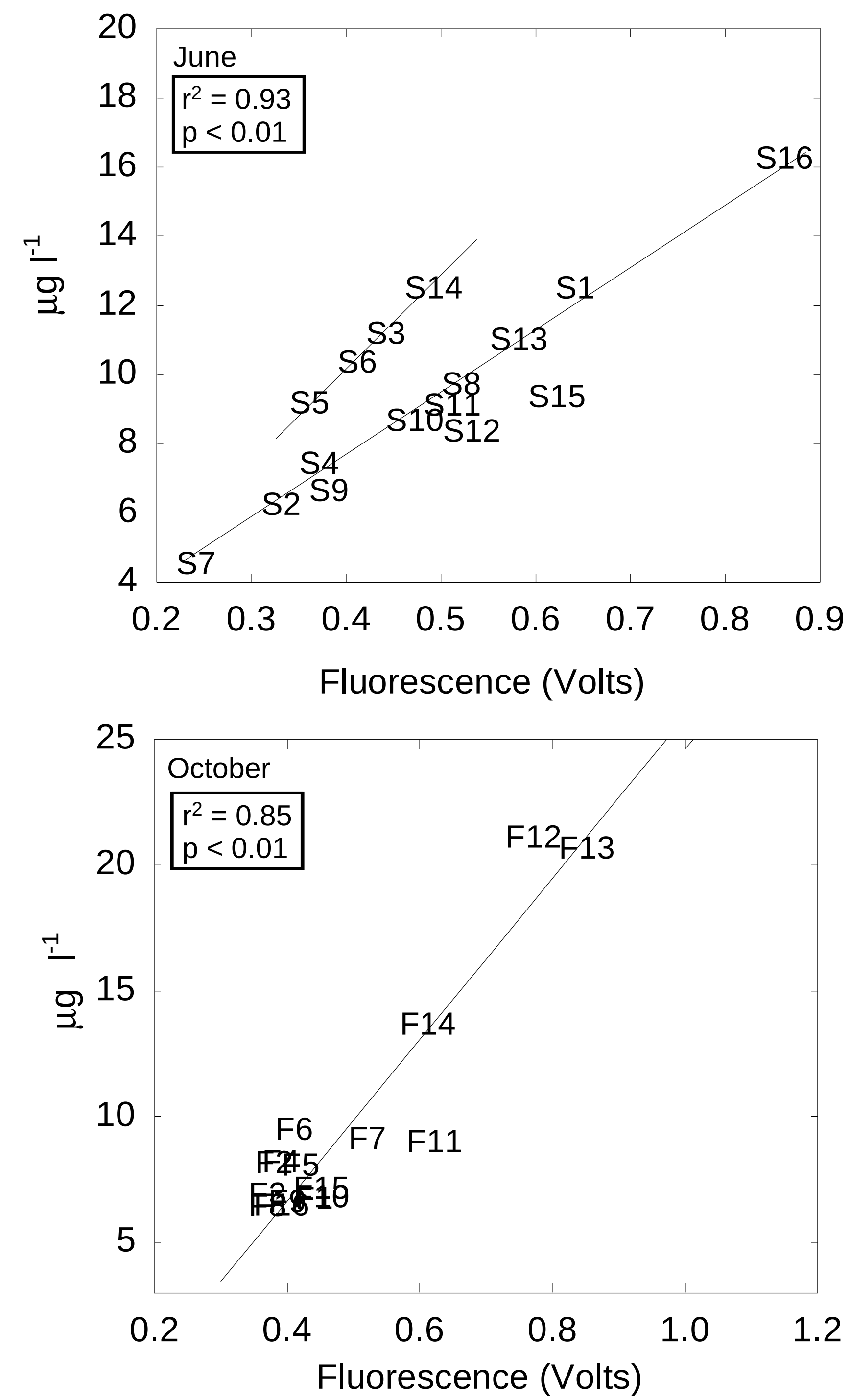

Figure 5-2. Field calibrations relating fluorescence (volts) to total chlorophyll-*a* in June (upper) and October (lower). June stations 3, 6 and 15 were excluded from the regression. Observation labels correspond to station numbers in Figure 5-1.

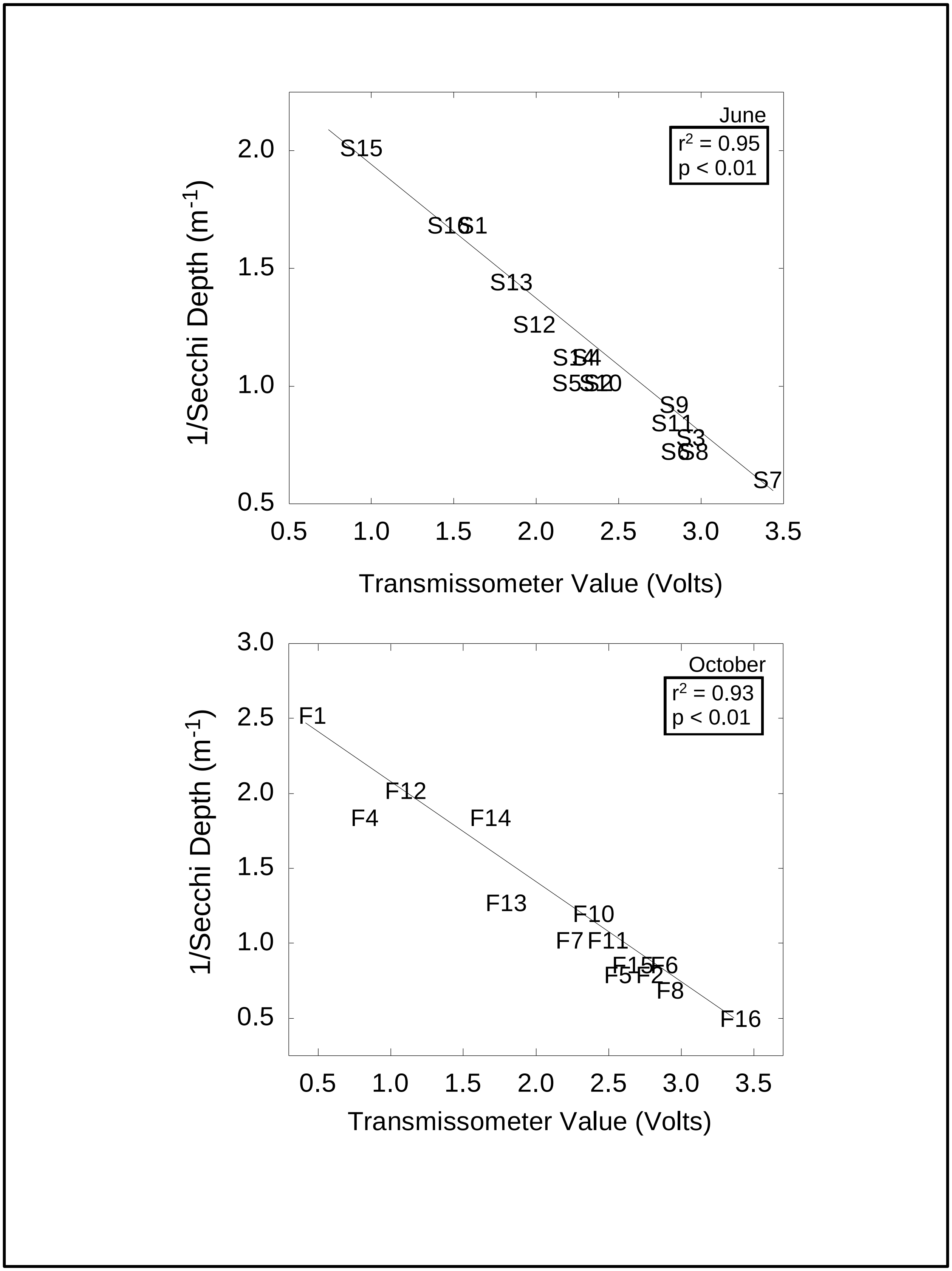

Figure 5-3. Field calibrations of inverse secchi depth vs. transmissiometer value in June (upper) and October (lower). Observation labels correspond to station locations in Figure 5-1.

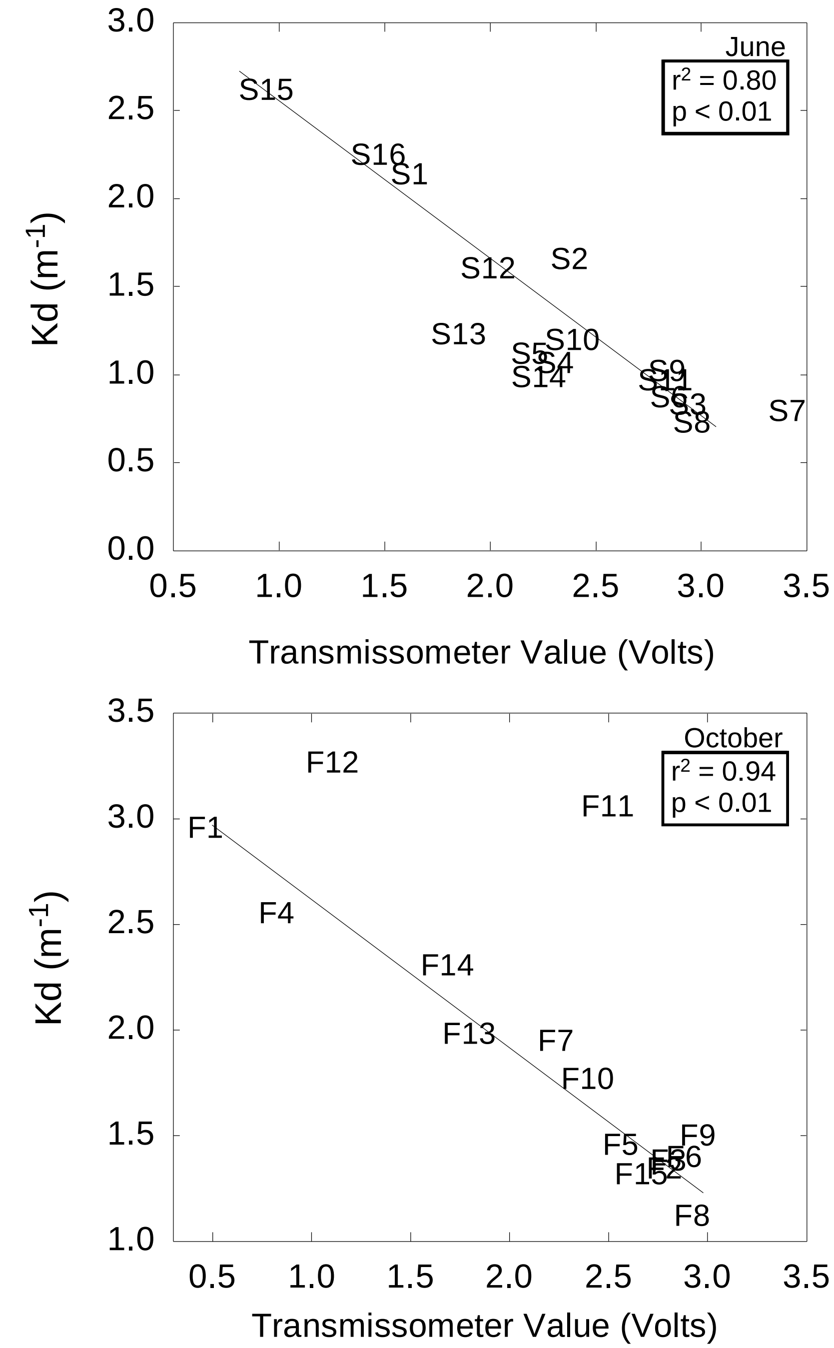

Figure 5-4. Field calibrations of light attenuation coefficient vs. transmissiometer value in June (upper) and October (lower). Stations 12 and 11 were excluded as outliers from October calibrations. Observation labels correspond to station locations in Figure 5-1.

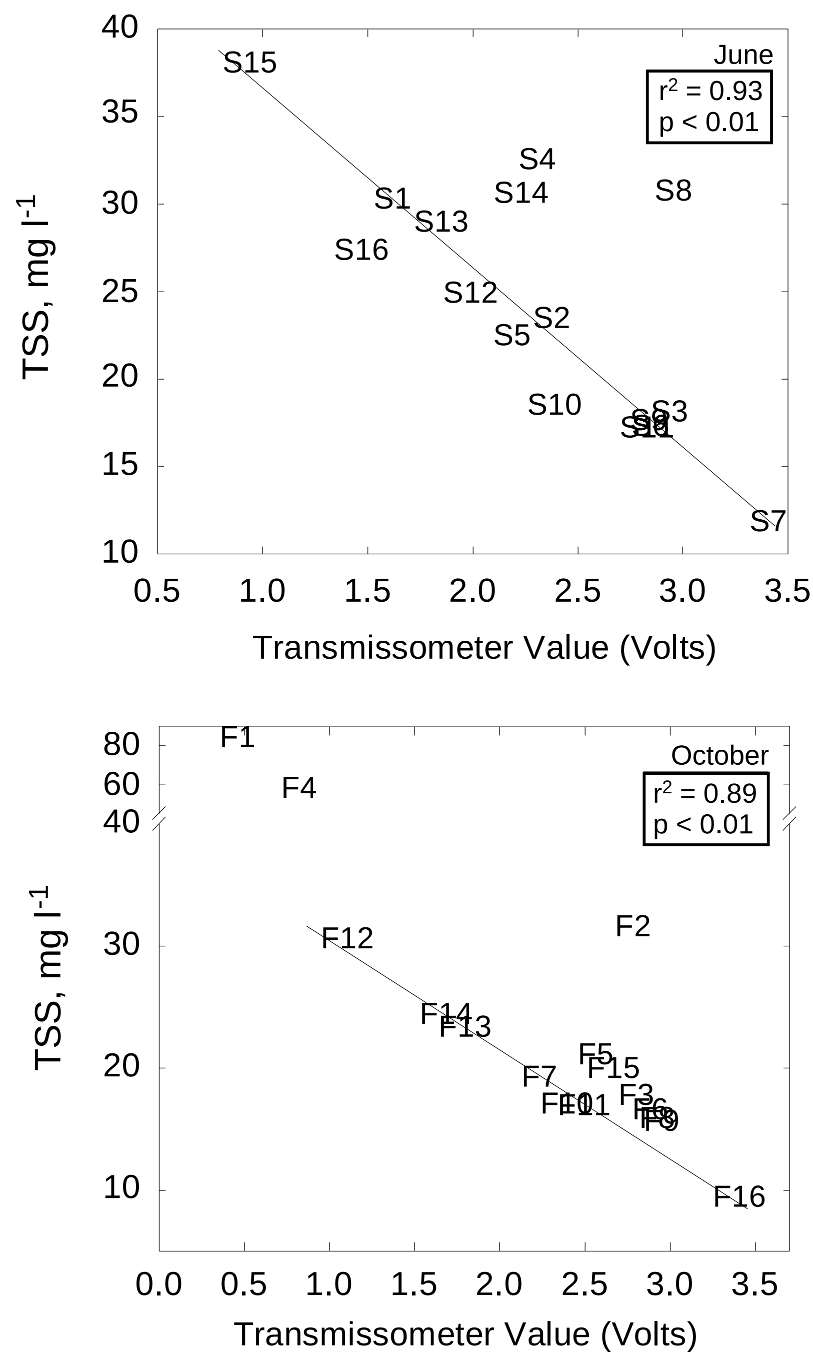

Figure 5-5. Field calibrations relating transmissometer values (volts) to total suspended solids in June (upper) and October (lower). Observations 4, 8, and 14 were excluded as outliers from the June calibration. Observations 1, 2 and 4 were excluded as outliers from the October calibration. Observation labels correspond to station locations in Figure 5-1.

Table 5-2. Parameters of linear regressions relating inverse secchi depth (1/SD), light attenuation coefficient (Kd), and total suspended solids (TSS).

	June 24-25		**October 12-13**	
	Slope (β)	Intercept (α)	Slope (β)	Intercept (α)
1/SD (m^{-1})	0.60	2.50	0.66	2.64
Kd, m^{-1}	-0.79	3.13	-0.66	3.22
TSS (mg l^{-1})	-9.81	44.95	-7.79	37.92

5.2.3 Interpolation and Contouring Procedures

All maps were produced using Surfer software (Golden Software) and the interpolation and contouring procedures included therein. Interpolations were generated using a limited quadrant search. Because of the presence of parallel tracklines of closely spaced data, a quadrant search was used to ensure that not all observations were obtained from the nearest track line. A limited search radius was used that was the smallest possible without creating gaps in the interpolated field. The good coverage of data ensured that this radius was small enough to prevent interpolation across most or all points of land. The edge of the interpolated area in each map reveals the size of the search radius, since these edges are exactly 1 radius beyond the nearest plotted observation. Interpolation using the nearest observations identified by the search procedure was performed using the default kriging procedures available in Surfer.

5.3 Results and Discussion

5.3.1 Major Features of Water Quality Distributions

The water quality distributions revealed several strong patterns, including regional differences, inshore-offshore differences, and strong wind and wave effects (Figures 5-7, 5-8, 5-9, 5-10, 5-11 and 5-12). In both July and October, elevated chlorophyll-*a* concentrations were observed in both the Nanticoke and Wicomico Rivers, with the highest concentrations in the Nanticoke River (Figures 5-7, 5-8). Light attenuation, along with all correlated properties (Kd, Secchi Depth, TSS) was also higher in these two rivers (Figures 5-9, 5-10, 5-11 and 5-12). Light attenuation throughout most of these tributaries reduced light to less than 20% of incident at 1 m depth, indicating that on these dates, these areas may not be suitable for SAV survival.

In general, turbidity in nearshore areas was higher than in deeper areas. For example, during June, turbidity along the eastern shore of the mid-Bay islands clearly increased on each of 12 inshore-offshore transects (Figure 5-11). Also during June, turbidity increased on 4 inshore-offshore transects along the shore of Deal Island (Figure 5-11). No corresponding inshore-offshore gradient in chlorophyll-*a* (Figure 5-7) was observed, however, indicating suspended solids were most likely attenuating light. On June 24, when the western portion of the study area was sampled, winds were 5-10 kt from the southeast, generally along-shore, while seas were calm. This suggests that strong wind-driven resuspension was not responsible for patterns along Smith and Bloodsworth Island. In contrast, a steady 10 kt southwest wind built substantial waves

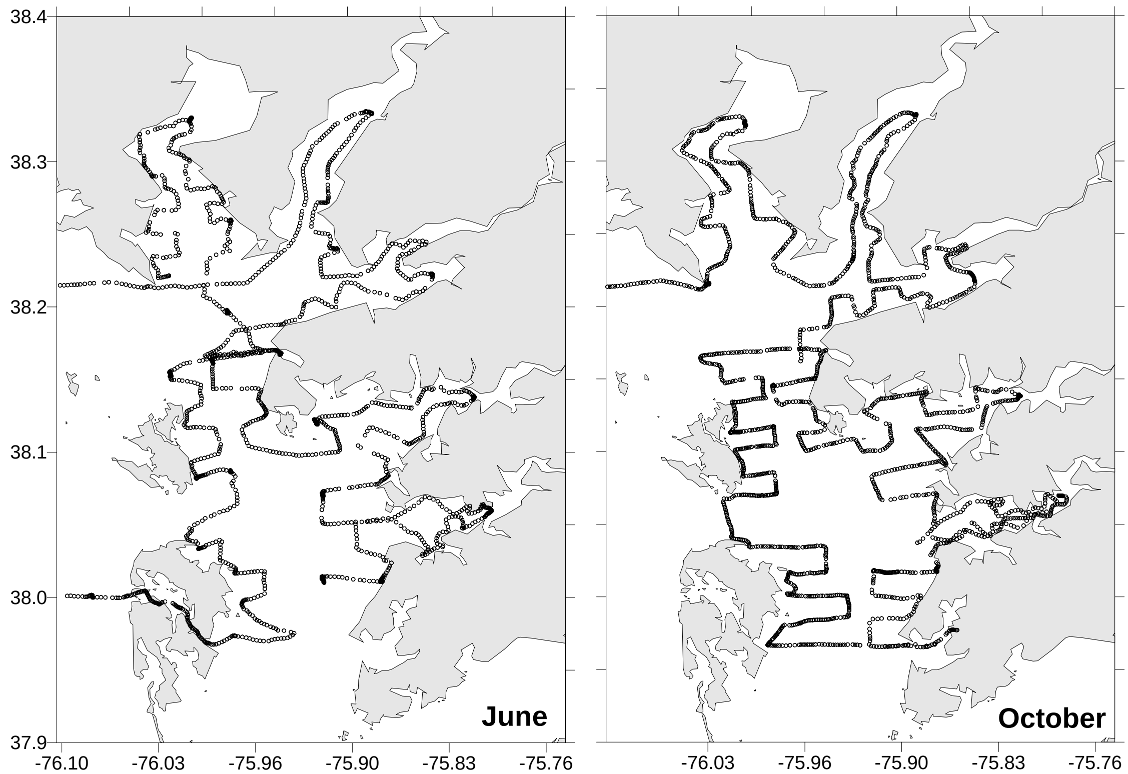

Figure 5-6. Cruise track lines transited by R/V Pices:

a. June 24 and 25

b. October 12 and 13

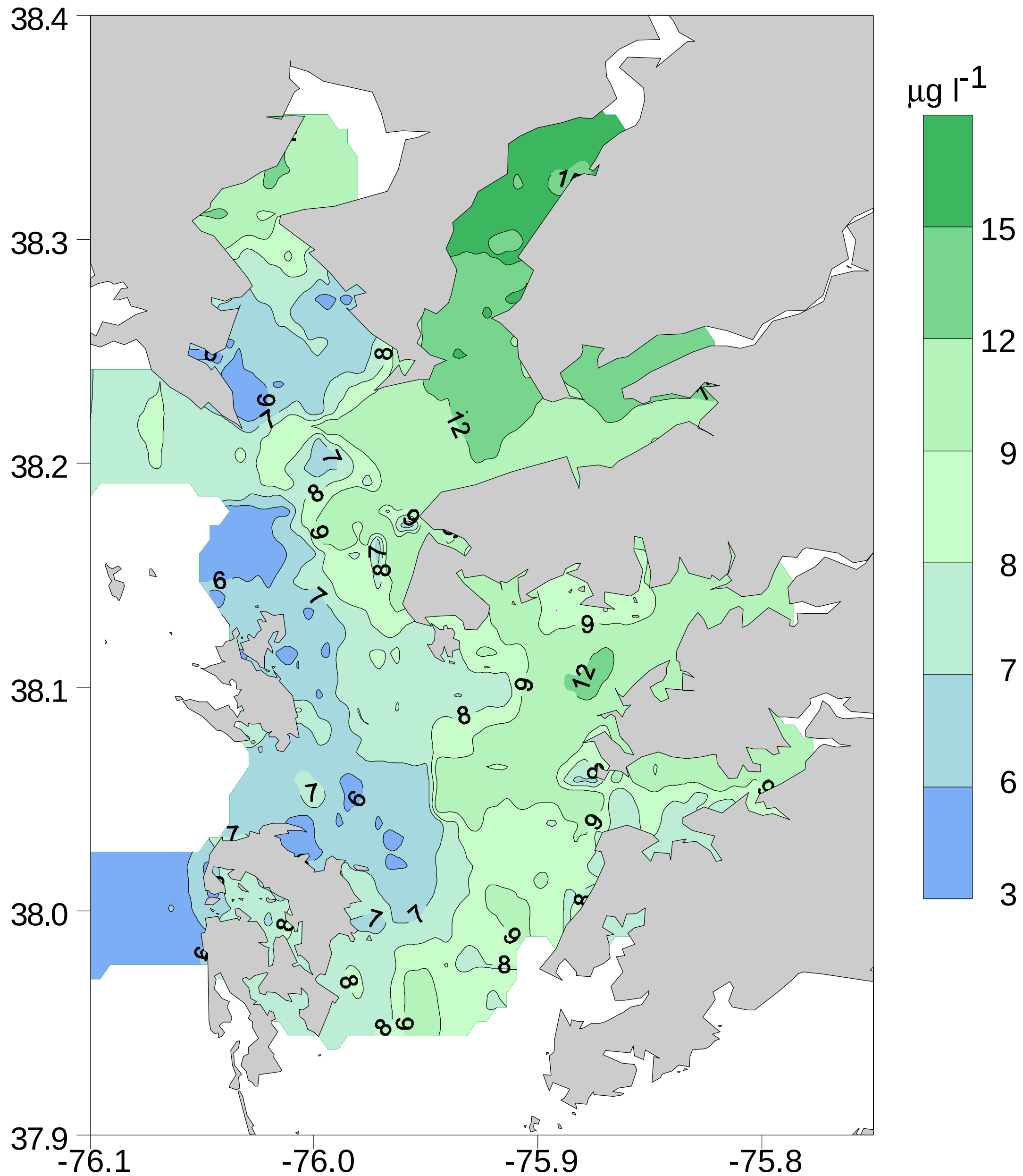

Figure 5-7. Interpolated map of total chlorophyll-*a* concentrations in Tangier Sound region on June 24-25, 1999, along the track-line transited by R/V Pices.

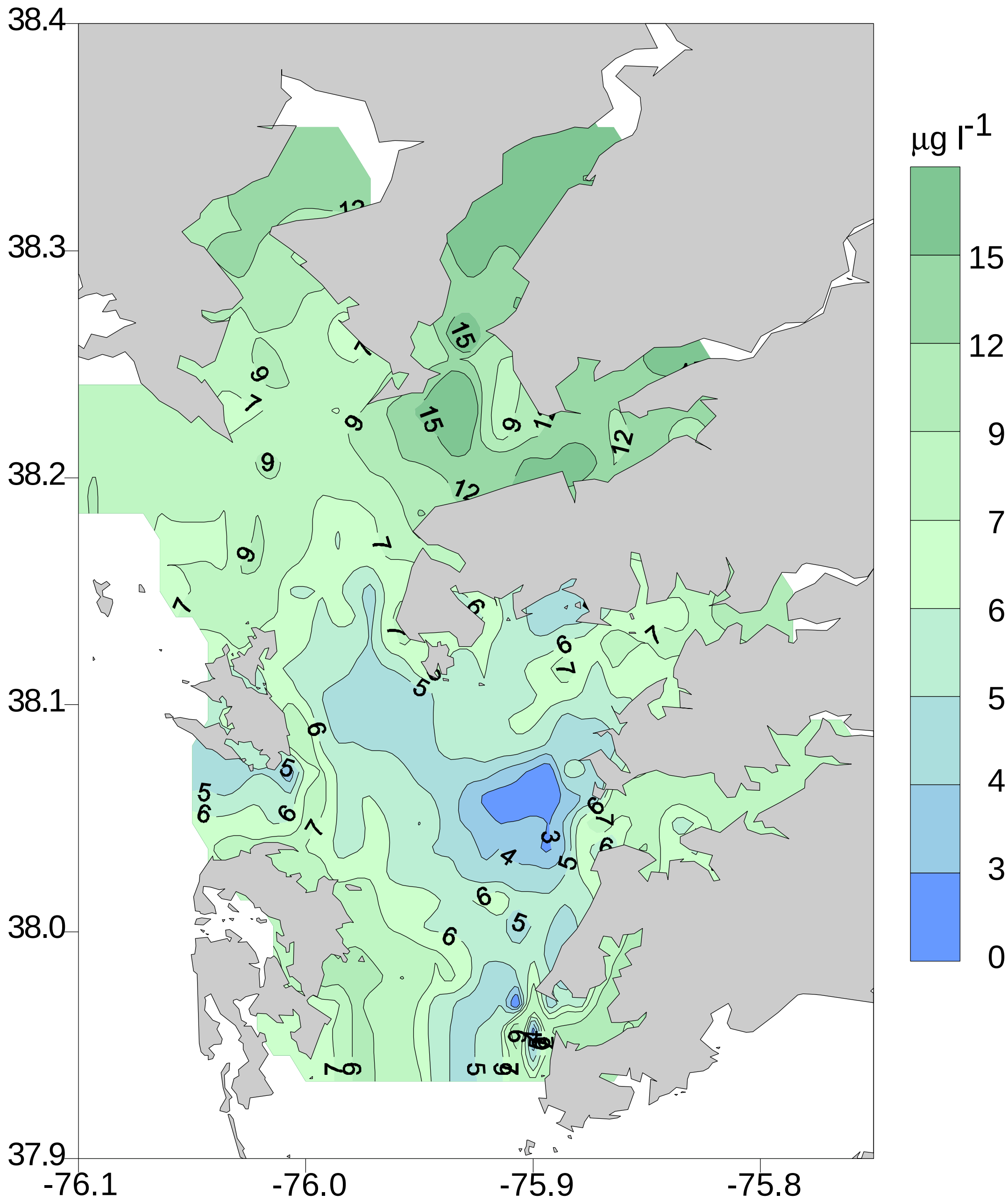

Figure 5-8. Interpolated map of total chlorophyll-*a* concentrations in Tangier Sound region on October 12-13, 1999, along the track-line transited by R/V Pices.

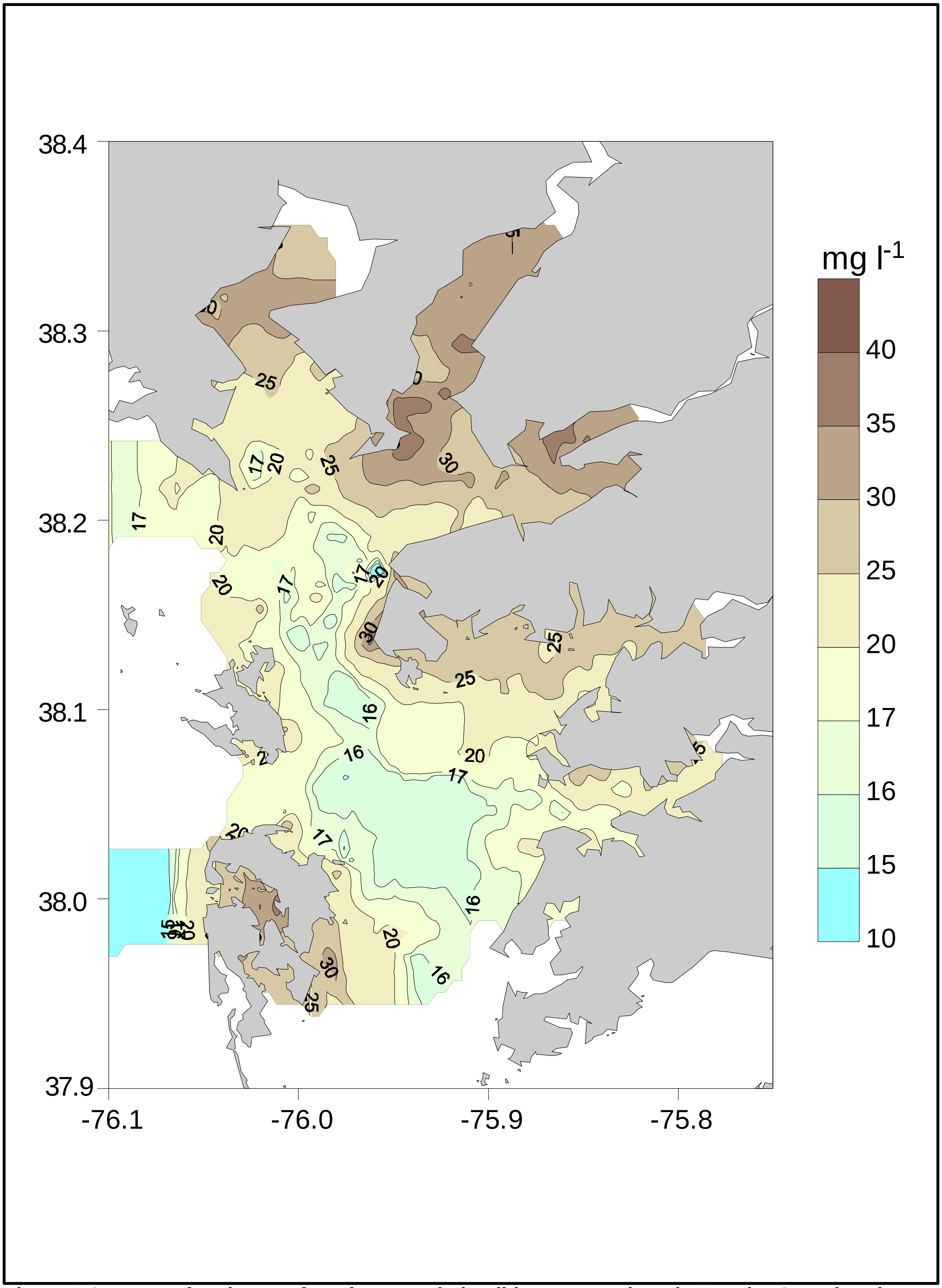

Figure 5-9. Interpolated map of total suspended solid concentrations in Tangier Sound region on June 24-25, 1999, along the track-line transited by R/V Pices.

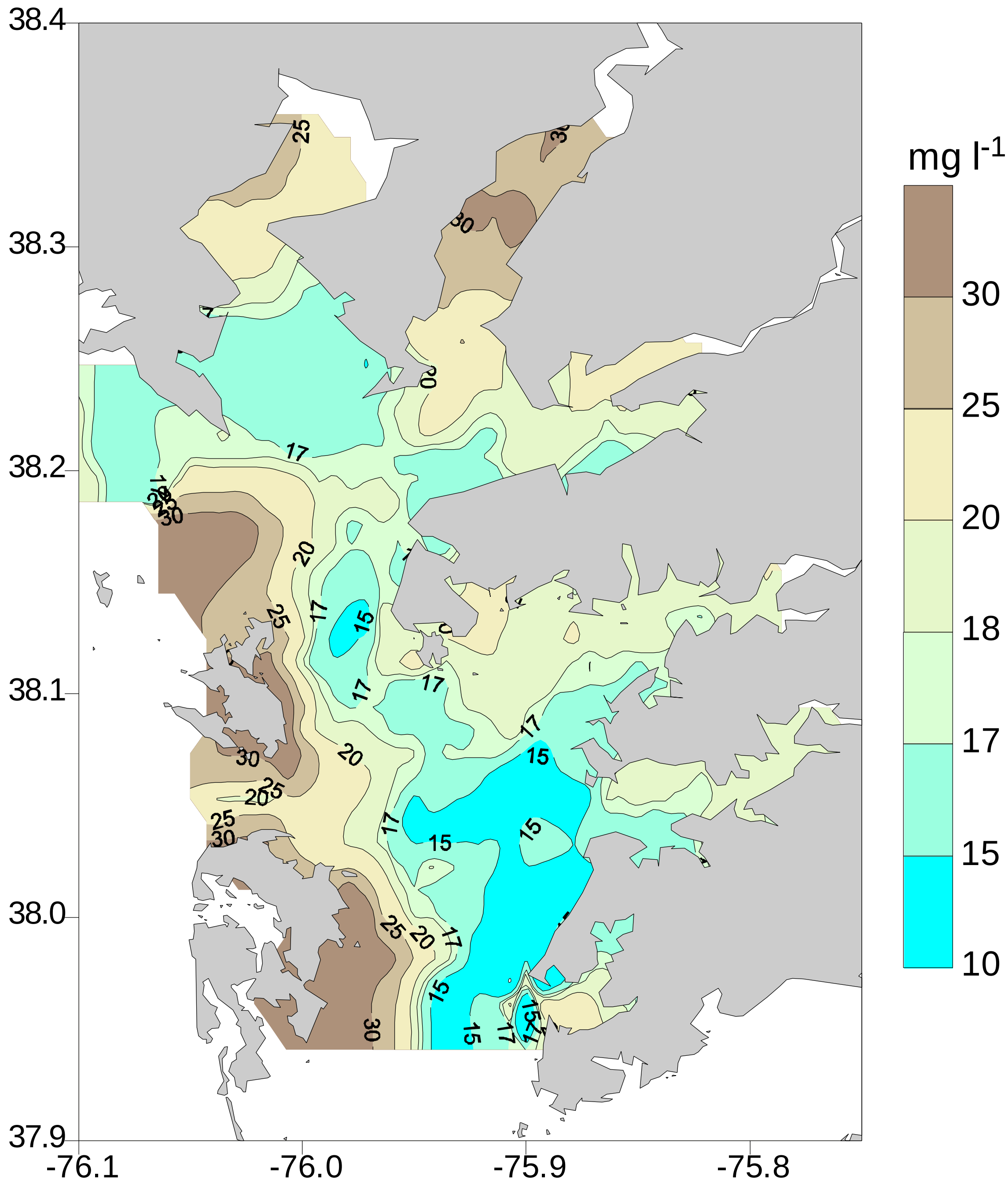

Figure 5-10. Interpolated map of total suspended solid concentrations in Tangier Sound region on October 12-13, 1999, along the track-line transited by R/V Pices.

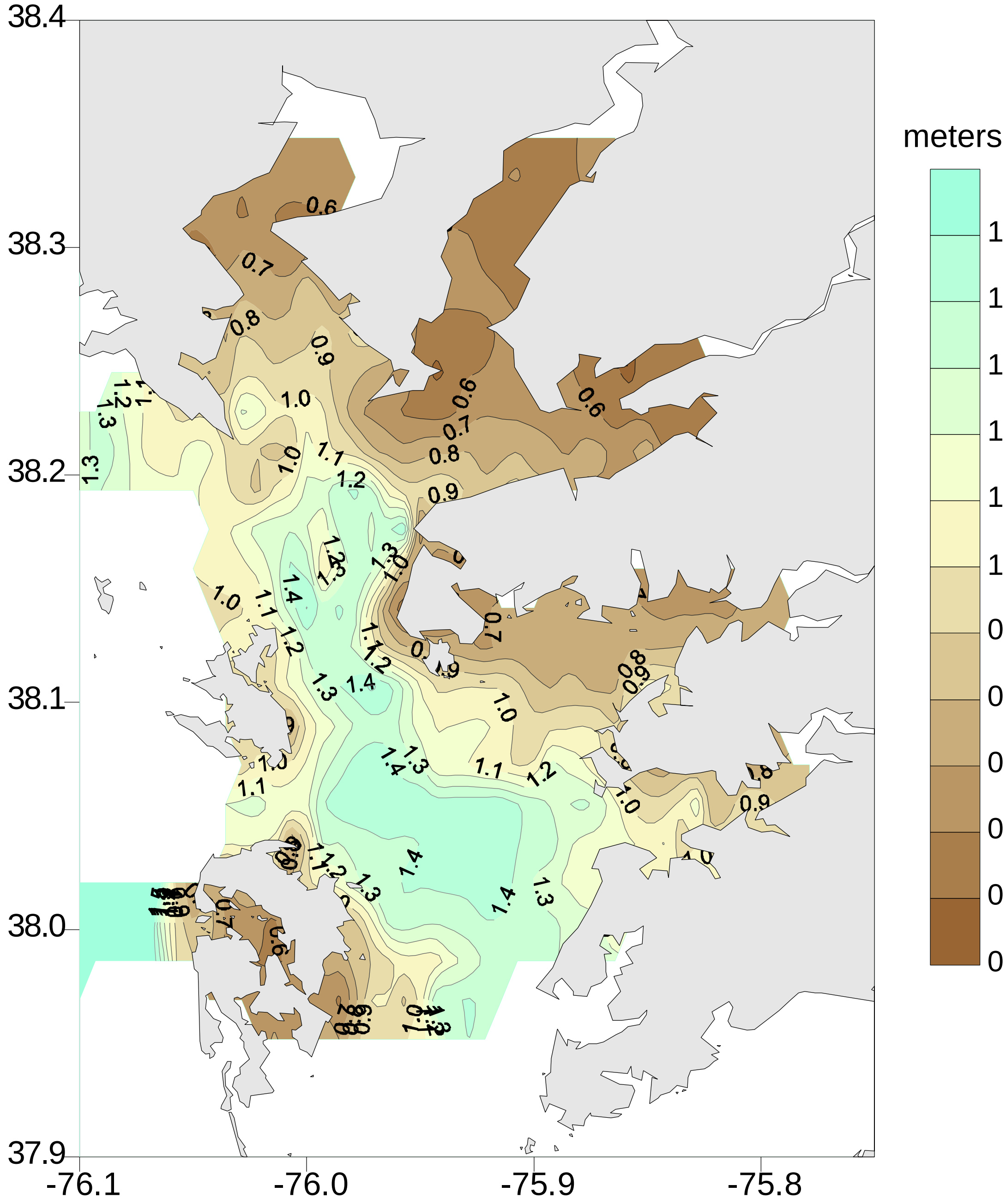

Figure 5-11. Interpolated map of estimated secchi depth in Tangier Sound region on June 24-25, 1999, along the track-line transited by R/V Pices.

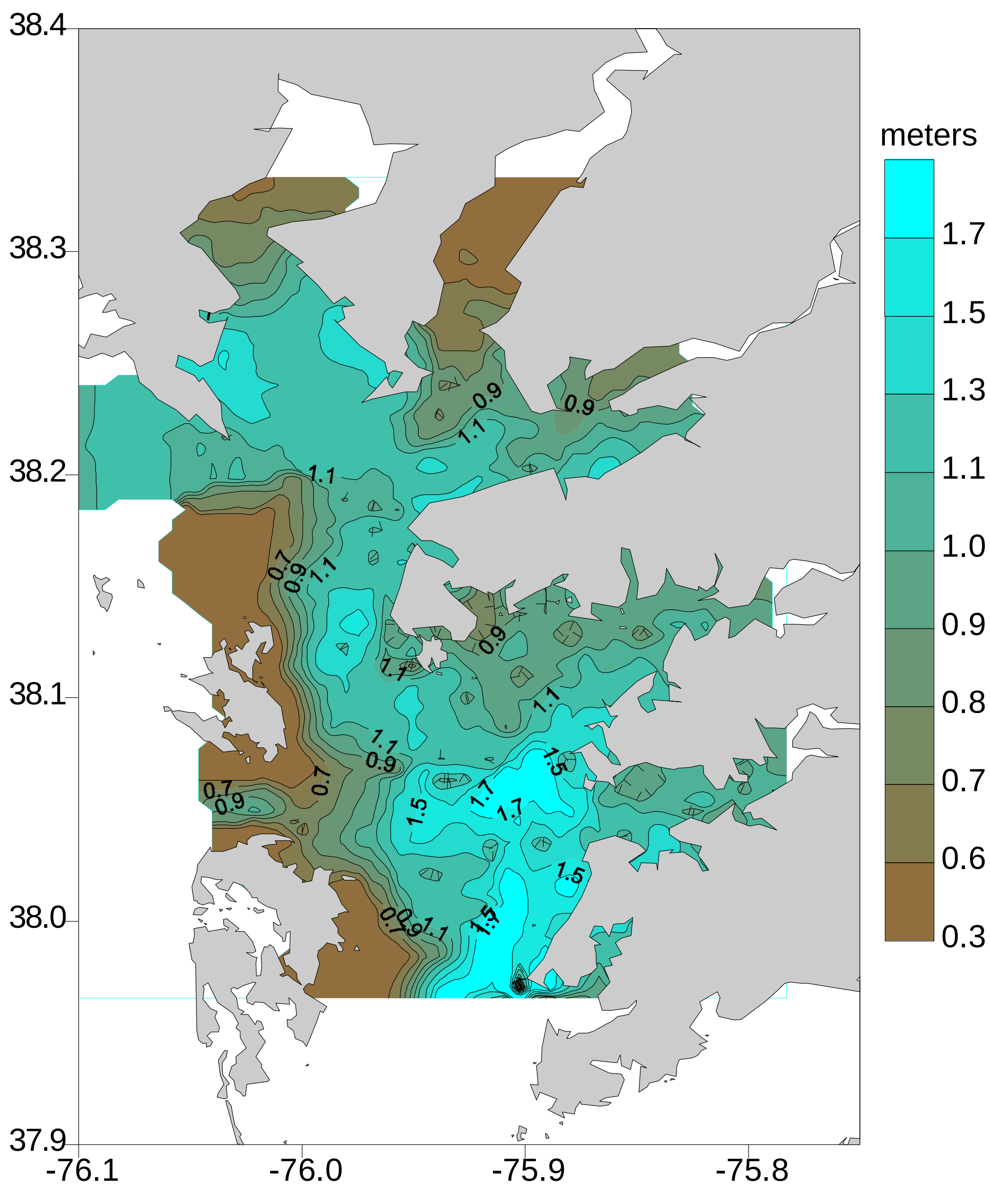

Figure 5-12. Interpolated map of estimated secchi depth in Tangier Sound region on October 12-13, 1999, along the track-line transited by R/V Pices.

in Tangier Sound on June 25. This likely resuspended sediments to generate the stronger inshore-offshore turbidity gradients observed along Deal Island (Figure 5-11).

Conditions in October were particularly revealing with respect to the effects of wave energy and sediment resuspension. A strong 15 kt northeast to east-northeast wind generated large waves in Tangier Sound before noon on October 12. Upon reaching the western shore of Tangier Sound, these waves were sufficient to break on the shoals extending outward from the islands. Station 1 was located in approximately 1 m depth (Figure 5-1), inside the zone where waves were breaking. Secchi depth at this station was 0.4 m (or less), while the TSS concentration was 82 mg l^{-1}. Similar conditions were observed throughout the island shoreline, extending northward beyond station 4, where similar conditions were found (Figure 5-10; TSS=57 mg l^{-1}). By early afternoon, however, winds diminished to less than 5 knots. Relatively homogeneous conditions were observed throughout the remaining southern portion of Tangier Sound (Figure 5-10). On October 13, however, still in calm wind conditions, elevated suspended solids concentrations were observed in the Nanticoke River, indicating that regional differences persisted despite wind conditions (Figure 5-10).

5.3.2 Adequacy of Fixed Monitoring Program Station Locations

Clearly, two days of mapping are not sufficient to characterize the full range of water quality distributions that might be observed. Thus, these maps alone really cannot be used to properly evaluate how well a long-term record at several locations may approximate average conditions overall. It is clear, however, that the data at these fixed stations can never represent the strong inshore-offshore turbidity gradients that were apparently created by wind blowing from either the south (as in June) or the north (as in October). Depending upon the frequency of such events, there may be significant differences in the suitability of habitat as determined by offshore water quality monitoring stations and the conditions experienced inshore. In this context, it would be of interest to find out (1) the frequency, direction, and intensity of wind events and (2) the time required for turbidity to decline to a baseline following cessation of wind-forcing.

Patterns in chlorophyll-*a* were not nearly as strong and were dominated on these dates by overall differences between major regions. In this respect, the array of fixed stations may convey a view of these inter-basin differences similar to that provided by high-resolution mapping. As a counterpoint, the mapping data do convey the seaward extent of elevated chlorophyll-*a* concentrations. In the case of the Nanticoke, especially in October, concentrations remained elevated throughout the river and did not decrease strongly toward the river mouth (Figures 5-7 and 5-8).

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6. MANAGEMENT SUMMARY

Based on a review of previous Ecosystem Processes Component (EPC) Reports (Boynton *et al*., 1989, 1990, 1991, 1992, 1993, 1994, 1995, 1996, 1997 and 1998), and the analyses presented in this report, the following observations are provided that have relevance to water quality management in the Patuxent River estuary and Tangier Sound.

Nutrient loading rate estimates (fall line load of TN and TP; above and below fall line point source loads of TN and TP) for the Patuxent River were reviewed for the period 1984-1998. A summary of that review is again included here because changes in these loads are of central interest in the Bay Program. Fall line loads of TP (which include above fall line point source inputs) have decreased dramatically between 1984 and 1995 (4-5 fold); recent loads would have been even lower except for relatively high inputs associated with flood events (*e.g.* May 1989, March 1993 and March 1994 and much of 1996). Because of the severe drought during 1999 it is expected that TP loads during 1999 will prove to be among the lowest on record. Fall line TN loads have also decreased over this period but not nearly as much as TP loads; similar increased loads of TN were associated with flood events. The regression of TN load versus time is significant ($p < 0.01$) for both the full period of time and the post 1989 period with annual load decreases of about 230 kg day^{-1} $year^{-1}$. We expect that TN loads will also prove to be much reduced during 1999, again because of the effects of the drought in reducing diffuse source run off of TN. This inspection of loads will be extended to include 1999 data when they become available. There is strong evidence that substantial nutrient load reductions have occurred in recent years.

Given the distinctive spring pulse in flow observed during 1998, diffuse source loads were probably moderate, at least compared to those observed during 1996 and some of the other high flow years observed during the late 1980s (1989) and 1990s (1993, 1994, 1996). It is also important to note that while loads increased in 1993 and 1994 (years of strong river flow) the increases were small, barely larger than loads associated with recent dry year loads, and much smaller than loads associated with wet years during the late 1980's.

Dissolved oxygen conditions in the Patuxent River were examined using monthly data collected at the four long-term sediment-water exchange (SONE) stations. In general dissolved oxygen conditions in deep water at these sites was quite good. For example, dissolved oxygen remained above 2.7 mg l^{-1} at station MRPT which in the past has often exhibited low dissolved oxygen values through much of the summer. Similarly, at station BRIS dissolved oxygen remained above 1.5 mg l^{-1}. The relatively brief period during which hypoxic conditions were present was considered to be a sign of improving water quality conditions. DO conditions in 1999 were similar to those observed during 1995 which also was a low flow (and low nutrient load) year.

Sediment oxygen consumption rates (SOC) were similar to the long-term average at the most up-river and down-river stations (BUVA and STLC, respectively). However, rates were larger at the two mid-river sites during most of the 1999 sampling period. These enhanced values very probably resulted because dissolved oxygen concentrations in deep water were higher during 1999 than in most previous years. SOC rates become limited (reduced in magnitude) when bottom waters are depleted in dissolved oxygen. A significant positive trend in SOC rates was detected at the most up-river station (BUVA) during 1999; no other significant trends were detected. The status of SOC was good at the most up-river site (BUVA), poor at the two mid-river sites (MRPT and BRIS) and fair at the most down-river site (STLC).

Ammonium (NH_4^+) fluxes at long-term monitoring stations (BUVA, MRPT, BRIS and STLC) in the Patuxent River were generally smaller than or equal to the long-term average and much lower than fluxes measured during 1998 which was a year characterized by a large spring input of freshwater and nutrients. The relatively low fluxes observed at these stations during 1999 is very probably a response to reduced nutrient loads associated with 1999 drought conditions. The large reductions in ammonium flux between 1996 and 1998 (years of high nutrient load and very high river flow) and 1999 parallels patterns of spring flow and nutrient loading. This "same year" response by sediments to loading conditions indicates that while sediments are the largest storage of nutrients in these systems, the portion of the stored material that is biologically active is not large enough to influence fluxes in subsequent years. In short, this is evidence for relatively limited nutrient memory and the potential for rapid (year rather than decade scale) responses to management actions. There were no trends in ammonium fluxes detected at SONE sites in the Patuxent River. Ammonium fluxes at the two up-river sites (BUVA and MRPT) were judged to be in the poor range and fluxes at the two down-river sites (BRIS and STLC) were in the fair range.

Positive ***sediment nitrate and nitrite fluxes*** (fluxes directed from sediments to the water column) are a definite sign of sediment nitrification activity that is a microbial process converting ammonium to nitrite and then nitrate and one that requires that oxygen be present. Positive nitrate fluxes are a sign of good sediment quality. Positive fluxes were observed during 1999 at all stations for most of the sampling period. There was one significant temporal trend detected for nitrite flux at the most up-river station (BUVA) wherein this flux has been increasing (a positive trend). Nitrite flux status was good at three sites (BUVA, MRPT and STLC) and poor and one site (BRIS). There were no significant trends for nitrate fluxes; nitrate flux status was good for up and down-river sites and poor at the two mid-river sites. We continue to believe that the presence of positive nitrate flux is a good tool for monitoring the general biogeochemical health of sediments.

During 1999, ***inorganic phosphate fluxes*** *(PO_4^- or DIP)* were near or below the long-term average at all sites except during August when above average values were observed at one site (STLC). At three of the sites (BUVA, MRPT and BRIS) phosphorus fluxes were far below average rates in July and August. Experimental studies involving phosphorus (PO4-) flux and dissolved oxygen (DO) conditions indicated a tight negative relationship between flux and DO status. When dissolved oxygen conditions improve, phosphorus flux decreases. In addition, these experimental studies indicated that the time needed for estuarine sediments to respond to decreased phosphorus loads is probably quite short (weeks to months) despite large storages of particulate nutrients in sediments (Jasinski, 1995). It appears that sediment phosphorus fluxes have responded to reduced inputs of phosphorus and that sediments do not contain active phosphorus reserves that can sustain high sediment releases much beyond the annual time scale. There were no significant temporal trends in phosphorus fluxes at any station. Phosphorus flux status was poor at the up-river site (BUVA), fair at the two mid-river sites (MRPT and BRIS) and good at the down-river site (STLC).

Results of ***sediment chlorophyll mapping*** in the Patuxent River during 1999 indicated some month to month variability in the mass of deposited chlorophyll from May through September. Sediment chlorophyll-a levels were highest following deposition of the spring phytoplankton in May and then were lower for the rest of the summer. During most of the monitoring period chlorophyll mass tended to be highest in deeper areas (possibly because of particulate material focusing) and in the saltier portion of the mesohaline reach. It is in this reach that water column monitoring data indicate that spring and summer algal blooms occur with regularity. Thus, there is an emerging understanding linking production and deposition of labile organic matter in this system. Finally, sediment chlorophyll-a mass in sediments during 1999 were the lowest on record (1996-1999) and this is consistent with a drought year wherein nutrient loads were also low and probably limited the production of phytoplankton. A very significant relationship was observed between winter-spring river flow (which delivers a large portion of the terrestrial nutrient load) and sediment chlorophyll-a mass. We would expect deposition of chlorophyll-a to continue to decrease as nutrient loads decrease in the future.

For the forth year, a ***MINI-SONE*** set of measurements was completed at six stations in the Patuxent River (in addition to measurements made at the four long-term sites in the Patuxent River). MINI-SONE measurements are a simplification of SONE measurements (*e.g.* one sediment core per station) and have been added to the EPC program as a means of increasing the spatial extent of sediment process measurements and to assist in the development of predictive statistical models of sediment-water exchanges. MINI-SONE flux measurements made in 1999 were almost uniformly smaller in magnitude than those observed during 1998. This is consistent with current understanding of the influence of nutrient loading on sediment-water exchanges (river flows and associated nutrient loads were higher in 1998 than in 1999). Using sediment chlorophyll mass as one of several key variables (others include sediment Eh and bottom water oxygen and nutrient concentrations), statistically significant ***regression models*** (linear single and multiple variable models) were developed for important fluxes. Results continued to indicate that this approach has great merit. We recommend that MINI-SONE and spatially intensive mapping of sediment chlorophyll-a and bottom water quality conditions be adopted for both traditional monitoring purposes and for verification of statistical sediment-water flux modeling. It is clear that spatial variability in these processes is an important feature and that this variability is understandable in the context of environmental conditions. Finally, spatially intensive measurements allows reasonable estimates to be made at larger scales such as the full estuary which are of direct interest to management.

During 1999 an ambitious and broad ***evaluation of littoral zone habitats*** was continued in the lower 35 km of the Patuxent River estuary (mesohaline zone) concerning the suitability of this region for ***SAV*** growth and possible reintroduction. The stimulus for this program was the observation that substantial nutrient load reductions recently achieved in this system have led to improving water quality conditions with little or no resurgence in SAV. The goal of this program element was to accurately measure and characterize many of the complex and interacting parameters necessary for SAV growth and survival in this shallow water habitat. As part of the baseline assessment, a full suite of water quality parameters was measured along the salinity gradient of the estuary from April through October 1999. Results of near shore water quality sampling indicate substantial temporal and spatial variation along the longitudinal axis of much of the Patuxent River. In general, water quality conditions were much better during the spring months of April, May and June, but deteriorated rapidly through the summer months. Although down river locations appear to have slightly better water quality conditions compared to up-river locations, overall most locations experienced water quality conditions, for at least brief periods of time, that do not meet estimated minimum habitat requirements for SAV growth and survival. In addition, estimates of epiphytic light attenuation during summer months suggested that after 5-7 days of exposure, epiphytic growth can potentially remove up to 80% of the available light reaching the leaves of SAV. Despite water quality conditions that overall were near or exceeded estimated limits for SAV growth and survival, certain early spring species of SAV were able to exist, and at some locations thrive, on the Patuxent River. These species complete their life cycle before water quality conditions become detrimental. Epiphytic fouling appears to be directly and very strongly correlated with light availability and, because of this, fouling rates were greatest in the clearer portions of the estuary. This finding suggests that nutrient concentrations (or probably more accurately the rate at which nutrients are delivered to SAV epiphytes) is still too high to be limiting to the growth of epiphyte communities. The 1999 monitoring program continued to provided baseline information about these near shore habitats; we recommend that additional monitoring be conducted to evaluate inter-annual variability since the success and growth of SAV requires consistent water quality conditions from year to year. Finally, since light availability is a critical requirement for SAV survival, we recommend a more diversified study of SAV epiphyte light attenuation and development of a simple and useful monitoring tool for SAV habitat evaluation.

A series of ***high resolution maps*** of water quality properties was initiated in Tangier Sound in support of efforts to understand the status of SAV habitat conditions in this area of the bay. The monitoring is conducted using a system called DataFlow wherein water quality variables such as turbidity, temperature, salinity, dissolved oxygen, and chlorophyll-a are co-registered with DGIS position, time of measurement and water depth. The system is used from small boats operating at >20 knots allowing for large spatial coverage in small amounts of time and operations in very shallow waters (<1 m).

Two sets of measurements have been completed in Tangier Sound (June and October, 1999) and each survey collected approximately 8,000 observations of each variable. The spatial maps of variables clearly indicated regions that meet SAV light and chlorophyll-a criteria as well as regions that do not. River plumes were clearly evident as were areas of very turbid waters caused by short-term wind events. The mapping exercise will continue in 2000 in Tangier Sound and new maps will be made in spring, summer and fall.

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