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

LEVEL ONE REPORT #19 (INTERPRETIVE)

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LEVEL ONE REPORT NO. 19

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1. INTRODUCTION

Almost two decades ago an historic agreement led to the establishment of the Chesapeake Bay Partnership whose mandate was to protect and restore the Chesapeake Bay ecosystem. The year 2000 saw the signing of *Chesapeake 2000* a document that incorporated very specific goals addressing submerged aquatic vegetation (SAV) restoration and protection and the improvement and maintenance of water quality in Chesapeake Bay and tributaries rivers.

The first phase of the Chesapeake Bay Program was undertaken during 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 to 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 bay sediments. 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, 1999, 2000 and 2001). The results of this characterization effort have confirmed the importance of deposition and sediment processes in determining water quality and habitat conditions. Furthermore, it is now clear that these processes are responsive to changes in nutrient loading rates.

The second phase of the program 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 was about 47% for nitrogen and 70% for phosphorus; point source reductions were ahead of schedule and diffuse source reductions were close to projected reductions; further efforts were 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 indicated 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.

Chesapeake 2000 involves the commitment of the participants *"to achieve and maintain the water quality necessary to support aquatic living resources of the Bay and its tributaries and to protect human health."* 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) stewardship and community engagement. The current EPC program, has activities that are aligned with the habitat and water quality goals described in this agreement.

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),
the Chesapeake Bay program web page http://www.chesapeakebay.net/monprgms.html
and DNR web page 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).

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

During the past two decades much has been learned about the effects of both natural and

anthropogenic nutrient inputs (*e.g.*, 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.

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 matter 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.

Within the context of this model a monitoring study of sediment processes and SAV habitat conditions has been developed. The 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), were expanded to Tangier Sound during 1999 and

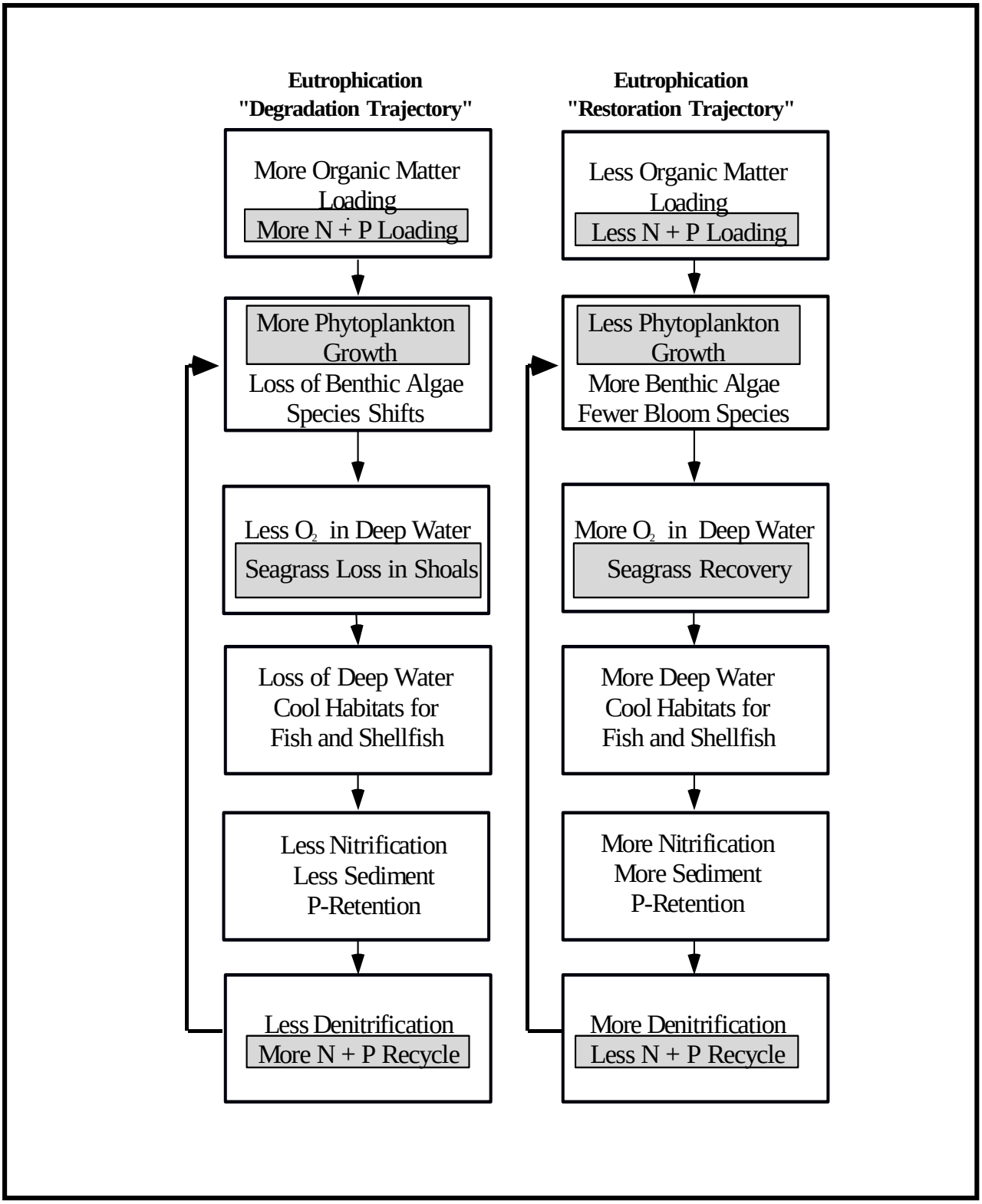

Figure 1-1. A simplified schematic diagram indicating degradation and restoration trajectories of an estuarine ecosystem. Lightly shaded boxes in the diagram indicate past and present components of the EPC program in the Patuxent River and Tangier Sound. (Adapted from Kemp, *pers. comm.*, HPEL)

further expanded in 2000 to also include the Magothy River. 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), and evaluated habitat conditions relative to SAV reintroduction. The Patuxent and Magothy River estuaries and Tangier Sound, where EPC efforts were concentrated during the years 2000 and 2001, are areas of particular interest because substantial reductions in nutrient loading rates have been achieved in one system (Patuxent) and SAC community status is of high concern in the others. Measurement of near-shore habitat conditions in the Severn River were added to the 2001 EPC activities.

The EPC has undergone program modification since its inception in 1984 but its overall objectives are consistent with those of other Monitoring Program Components:

1. Characterize the present status of the Patuxent River 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 River estuary.

3. Evaluate near-shore water quality conditions relative to SAV habitat across a range of spatial and temporal scales. Near-shore mapping and measurement of water quality conditions was conducted in Tangier Sound and the Magothy and Severn Rivers. Epiphyte accumulation rates and associated water quality conditions were measured at six sites in Tangier Sound and one location in the Patuxent River.

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

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2. SEDIMENT-WATER OXYGEN AND NUTRIENT EXCHANGES: MINI-SONE

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

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2.1 Introduction and Background

More than a decade of monitoring 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, sediment nutrient releases have significant potential to negatively affect water quality and living resources. The EPC program monitors sediment flux monthly during summer periods. Previous studies have shown that the highest nutrient releases by sediments occur during the summer months (Boynton *et al.*, 1988). Sediment-water oxygen and nutrient exchange (SONE) measurements were made at four fixed-location stations of the Patuxent River estuary.

Beginning in 1996, the EPC adopted a new technique that increased the spatial resolution of SONE-type measurements. For several years additional sediment-water exchange stations were added to the normal sampling regime to provide better assessments of the range of sediment-water exchanges found within the Patuxent River estuary,especially as a function of water depth. In order to be cost effective, sediment-water exchanges 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. Previous studies had shown that variation among replicate cores from a single location was small compared to variation among sites. Therefore, additional stations, distributed along depth gradients, would provide a more accurate assessment of sediment-water exchanges in the estuary

as a whole, and thus be more useful for evaluating whole ecosystem responses to nutrient management strategies.

This more intensive "mapping" of sediment-water exchanges was conducted during 1996-1999 using the MINI-SONE approach. During 2000 and 2001, the mapping approach was discontinued but sediment-water exchanges were monitored 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 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.

2.2 Station Locations for MINI-SONE Long-term Patuxent River Station Locations

Four stations, St. Leonard Creek (STLC), Broomes Island (BRIS), Marsh Point (MRPT) and Buena Vista (BUVA) were previously monitored using the full suite of measurements referred to as SONE. These sites are now referred to as the long-term monitoring stations and are monitored using an abbreviated MINI-SONE approach. Station locations sampled during 2001 are shown in Figure 2-1 (See also Table 2-1) as are nearby water quality monitoring stations.

2.3 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). Previous studies also indicated 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 and shallow *vs.* deep areas). In light of these results, the monitoring design adopted for MINI-SONE studies involved four monthly measurements at four stations in June, July, August and September 2001. Sampling dates for these cruises together with alpha-numeric cruise identification codes can be found in Table 2-2.

2.4 Field Methods for MINI-SONE

2.4.1. Water Column Profiles

At each MINI-SONE station, vertical water column profiles of temperature, salinity and dissolved oxygen are measured at 2 meter intervals from the surface to the bottom. Turbidity of surface waters is measured using a Secchi disc.

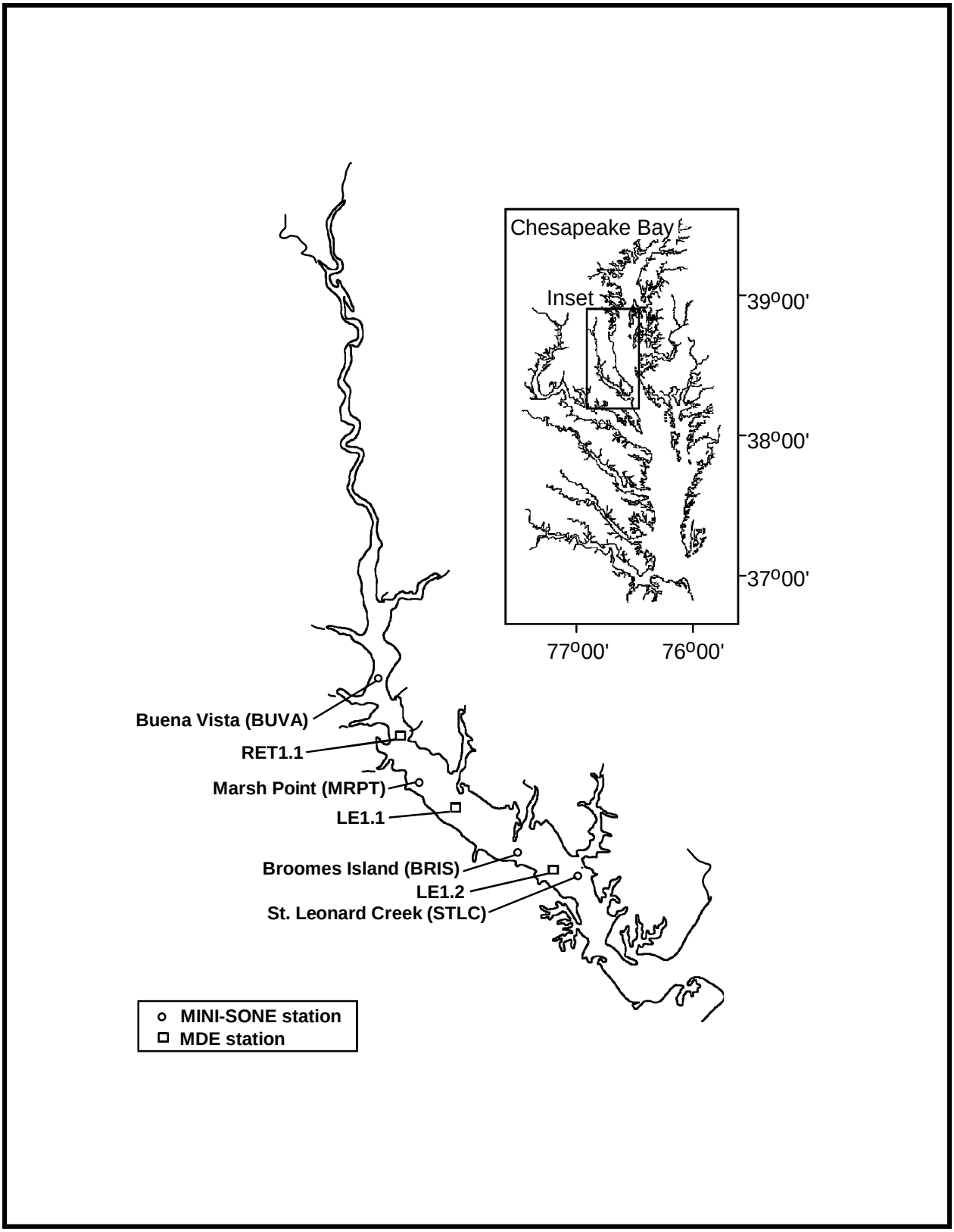

Figure 2-1. Location of four MINI-SONE Stations sampled in the Patuxent River, MD.
Location of stations shown here do not reflect exact geographic locations (See Table 2-1).

Table 2-1. MINI-SONE Station Code, Grid Location and Nearest MDE Station

STATION CODE	LATITUDE (DGPS) NAD 83	LONGITUDE (DGPS) NAD 83	STATION DEPTH (m)	CHESAPEAKE BAY STATION	BAY SEGMENT
Patuxent River					
BUVA	38° 31.050’	76° 39.783’	5.5	RET1.1	RET1
MRPT	38° 26.767’	76° 37.900’	6.9	LE1.1	LE1
BRIS	38° 23.600’	76° 33.067’	15.3	LE1.2	LE1
SLTC	38° 22.817’	76° 30.067’	6.6	LE1.2	LE1

Table 2-2. MINI-SONE Cruise Identifier

CRUISE	DATE	BEGIN DATE	END DATE	RESEARCH VESSEL
MINI-SONE 21	JUN 2001	JUN 14	JUN 14	Orion
MINI-SONE 22	JUL 2001	JUL 20	JUL 20	Orion
MINI-SONE 23	AUG 2001	AUG 16	AUG 16	Orion
MINI-SONE 24	SEP 2001	SEP 12	SEP 12	Aquarius

2.4.2 Water Column Nutrients

Near-bottom (approximately 1/2 meter above the bottom) water samples are collected using a high volume submersible pump system. Samples are filtered, where appropriate, using 0.7 µm GF/F filter pads, and immediately frozen. Samples are analyzed by Nutrient Analytical Services Laboratory (NASL) for the following dissolved nutrients: ammonium (NH_4^+), nitrite (NO_2^-), nitrite plus nitrate ($NO_2^- + NO_3^-$) and dissolved inorganic phosphorus corrected for salinity (DIP or PO_4^{-3}).

2.4.3 Sediment Profiles

At each MINI-SONE station an intact sediment core is used to measure the redox potential (Eh) of the sediment porewater. Sediment redox (mV) is measured at the sediment surface, one and 2 centimeters below the surface and every 2 centimeters thereafter to 10 cm depth. Additionally, surficial sediments are sampled for total and active sediment chlorophyll-*a* to a depth of 1 cm. Particulate carbon (PC), particulate nitrogen (PN), particulate phosphorus (PP), are sampled to a depth of 1 cm.

2.4.4 Sediment Flux Measurements

The protocols used in MINI-SONE flux estimates are an abbreviated set of measurements of the standard SONE techniques. MINI-SONE stations use a single sediment core with no blank.

Intact sediment cores constitute a benthic microcosm where changes in oxygen, nutrient and other compound concentrations are determined.

A single intact sediment core is collected at each station using a modified Bouma box corer. These cores are then transferred to a Plexiglass cylinder (15 cm diameter x 30 cm length) and inspected for disturbances from large macrofauna or cracks in the sediment surface. If the sample is satisfactory, the core is fitted with an O-ring sealed top containing various sampling ports, and a gasket sealed bottom (Figure 2-2). The core is then placed in a darkened, temperature controlled holding tank where overlying water in the core is slowly replaced by fresh bottom water to ensure that water quality conditions in the core closely approximate *in situ* conditions.

During the period in which the flux measurements are taken, the cores are placed in a darkened temperature controlled bath to maintain ambient temperature conditions. The overlying water in a core is gently circulated with no induction of sediment resuspension via stirring devices attached to oxygen probes. Oxygen concentrations are recorded and overlying water samples (35 ml) are extracted from each core every 60 minutes during the incubation period. Standard SONE stations are incubated for 4 hours and a total of 5 measurements are taken, while MINI-SONE stations are incubated for 3 hours with a total of 4 measurements taken. As a water sample is extracted from a core, an equal amount of ambient bottom water is added to replace the lost volume. Water samples are filtered and immediately frozen for later analysis for ammonium (NH_4^+), nitrite (NO_2^-), nitrite plus nitrate ($NO_2^- + NO_3^-$) and dissolved inorganic phosphorous (DIP or PO_4^{-3}). Oxygen and nutrient fluxes are estimated by calculating the mean rate of change in concentration over the incubation period and converting the volumetric rate to a flux using the volume:area ratio of each core.

2.4.5. Chemical Analyses used in MINI-SONE Element

Methods for the determination of dissolved and particulate nutrients are as follows: ammonium (NH_4^+), nitrite (NO_2^-), nitrite plus nitrate ($NO_2^- + NO_3^-$), and dissolved inorganic phosphorus (DIP or PO_4^-) are measured using the automated method of EPA (1979); particulate carbon (PC) and particulate nitrogen (PN) samples are analyzed using an Elemental Analyzer; particulate phosphorus (PP) concentration is obtained by acid digestion of muffled-dry samples (Aspila *et al.*, 1976); methods of Strickland and Parsons (1972) and Parsons *et al.* (1984) are followed for chlorophyll-*a* analysis.

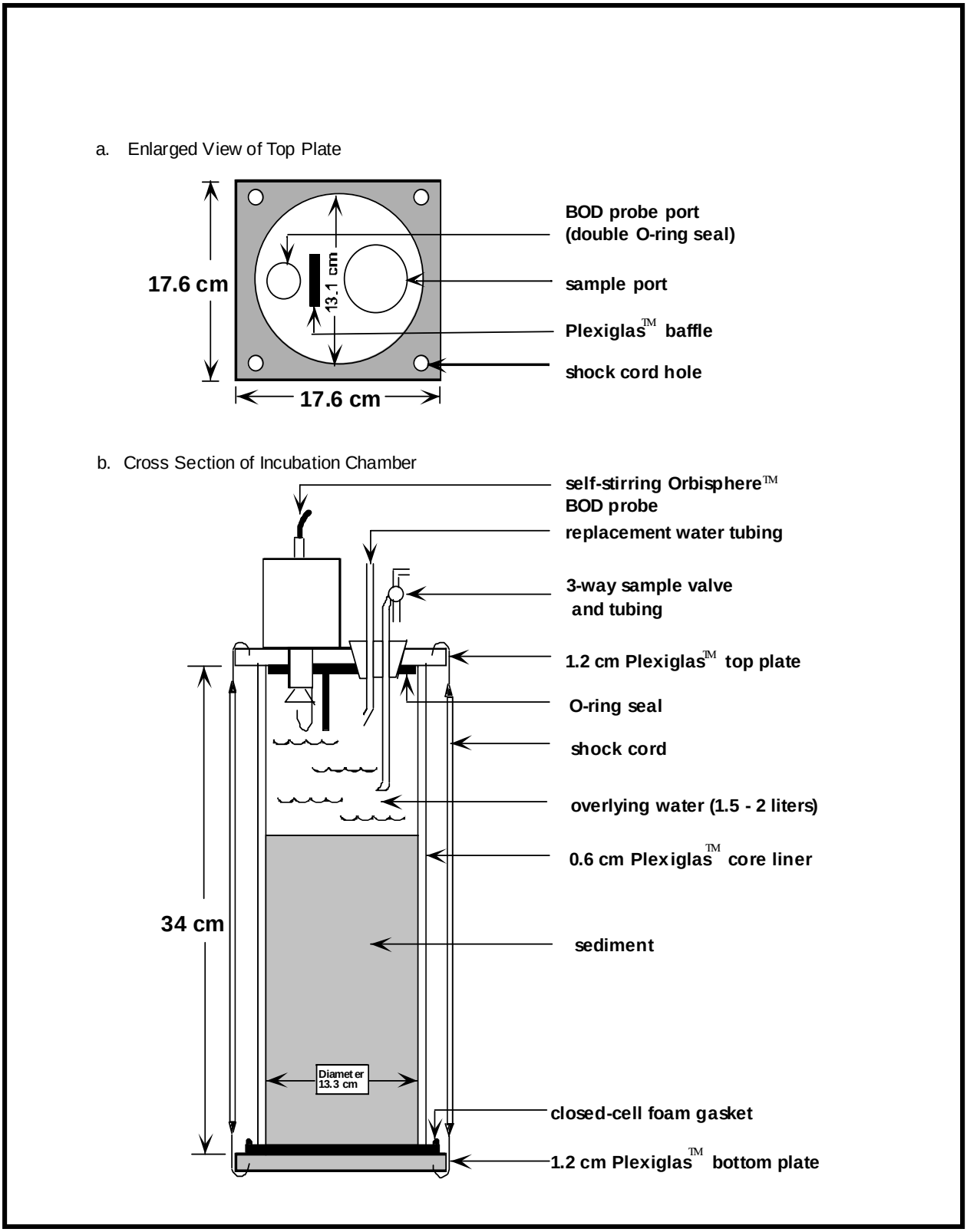

Figure 2-2. Schematic Diagram of the Incubation Chamber
a. Enlarged View of Top Plate.
b. Cross Section of Incubation Chamber

2.5. River Flow

In the Patuxent River, and in other coastal plain estuaries, river flow is often a good indicator of several important external forcing functions that influence estuarine conditions. River flow influences temperature and salinity patterns, circulation and nutrient loading rates. Not only is the magnitude of river flow important, but also the timing of flow events that can affect such processes as nutrient uptake and subsequent deposition of phytodetritus. 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 304 cfs in 2001, 315 cfs in 2000, 285 cfs in 1999, 437 cfs in 1998, 412 cfs in 1997 and 704 in 1996; riverflow values for the last three years were below the twenty-four year average of 372 cfs (Figure 2-3.a.). The patterns of monthly average river flow also differed significantly during recent years.

There were two peaks in river flow during the first half of 2001, one in March (506 cfs) and one in June (601 cfs), while during the second half of 2001 river flows were uniformly low (Figure 2-3b). In 1999 there were two peaks, one in March (392 cfs) and one in September (723 CFS), while in 2000 the peak flow occurred in April. Many estuarine processes respond to nutrient loading on time scales of weeks to months so the timing of flow events can be an important consideration. 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.

2.6 MINI-SONE Sediment-Water Oxygen and Nutrient Fluxes:

2001 Patuxent River Study

Monthly average sediment-water fluxes derived from the complete sediment-water oxygen and nutrient exchanges (SONE) data set (1985 - 1997) are summarized using box and whisker plots (Figures 2-4.1 through 2-4.4) 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 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 MINI-SONE flux measurements made at these four stations during 2001.

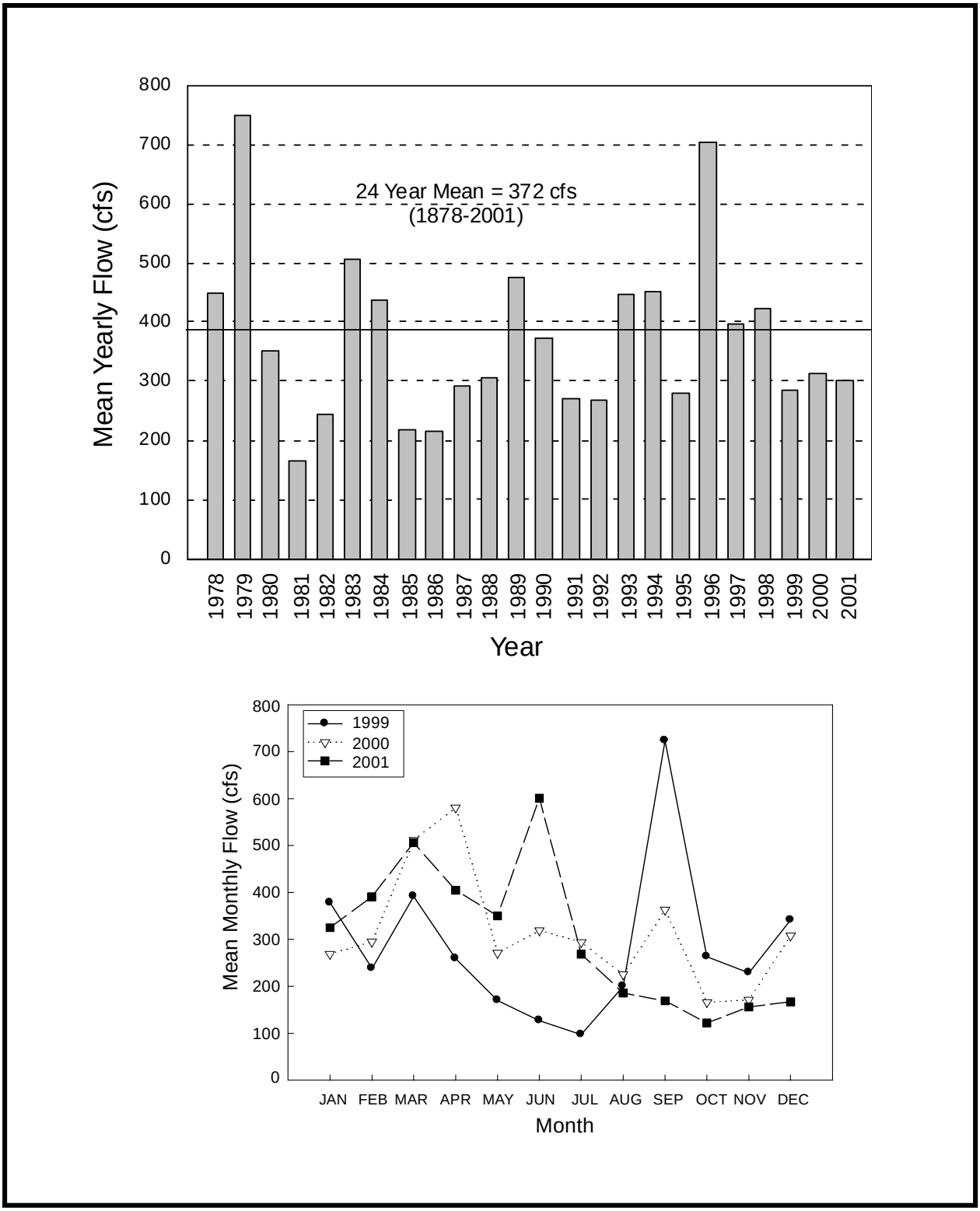

Figure 2-3. (a) Patuxent River average annual river flow for the period 1978 through 2001 (calendar year), at USGS station, 01594440 Patuxent River near Bowie, MD.

(b) Patuxent River average monthly river flow from 1999 through 2001 (calendar year), at USGS station, 01594440 Patuxent River near Bowie, MD.

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. In Figure 2-4 the complete SONE flux data set was used to produce the box and whisker plots. The bold solid dots indicate a single flux measured during the MINI-SONE study 2001.

2.6.1 Sediment Oxygen Consumption (SOC)

Lower than normal dissolved oxygen concentrations in bottom waters (> 1.2 mg l^{-1}) were observed at all stations during June and also at the deeper stations (BRIS and MRPT) in August. The magnitude of 2001 SOC observations at all four stations were noticeably lower than average during June. At these stations bottom water dissolved oxygen concentrations were quite depressed during this month, as were SOC rates due to the influence of low dissolved oxygen concentrations (< 2.0 mg l^{-1}) on SOC rates. Both 2001 and 2000 were intermediate flow years. In dry years, with low river flow, dissolved oxygen concentrations in deep waters tend to be more elevated than usual. 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 and subsequent deposition).

2.6.2 Ammonium (NH_4^+) Fluxes

Ammonium fluxes recorded in 2001 as in 2000 were higher than normal releases in July and August at the two up-river stations (BUVA and MRPT). Fluxes reached peak values in July at the two down-river sites (BRIS and STLC). Ammonium fluxes were generally similar to long-term mean values during June and September.

The ammonium flux pattern observed during 2001 was unusual in several respects. First, as noted above, maximum values occurred later (August) than usual (July) at the two up-river stations. This suggests a significant but delayed delivery of organic matter to sediments at these sites. The normal pattern of peak fluxes occurring in July was observed at the down-river sites. We have interpreted this pattern as being the result of remineralization of spring bloom organic matter. Decreased fluxes in August and September reflected the decreased supply of labile organic matter to estuarine sediments. The second unusual aspect of ammonium flux during

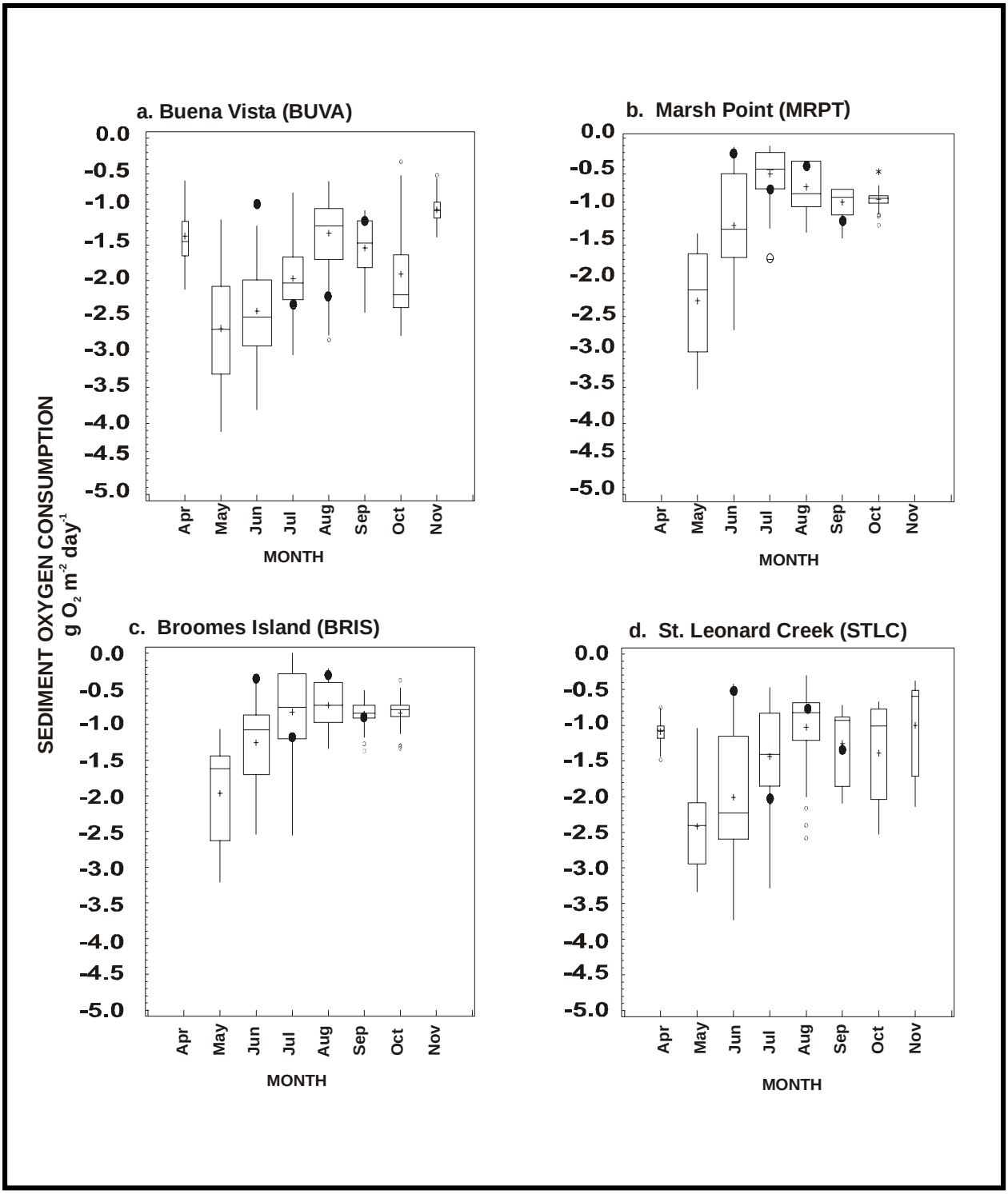

Figure 2-4.1. Box and whisker plots for sediment oxygen consumption (SOC) rates for April to November at four SONE stations located in the Patuxent River.
(a) Buena Vista [BUVA] (b) Marsh Point [MRPT] (c) Broomes Island [BRIS] and (d) St. Leonard Creek [STLC].

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 2001. 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.

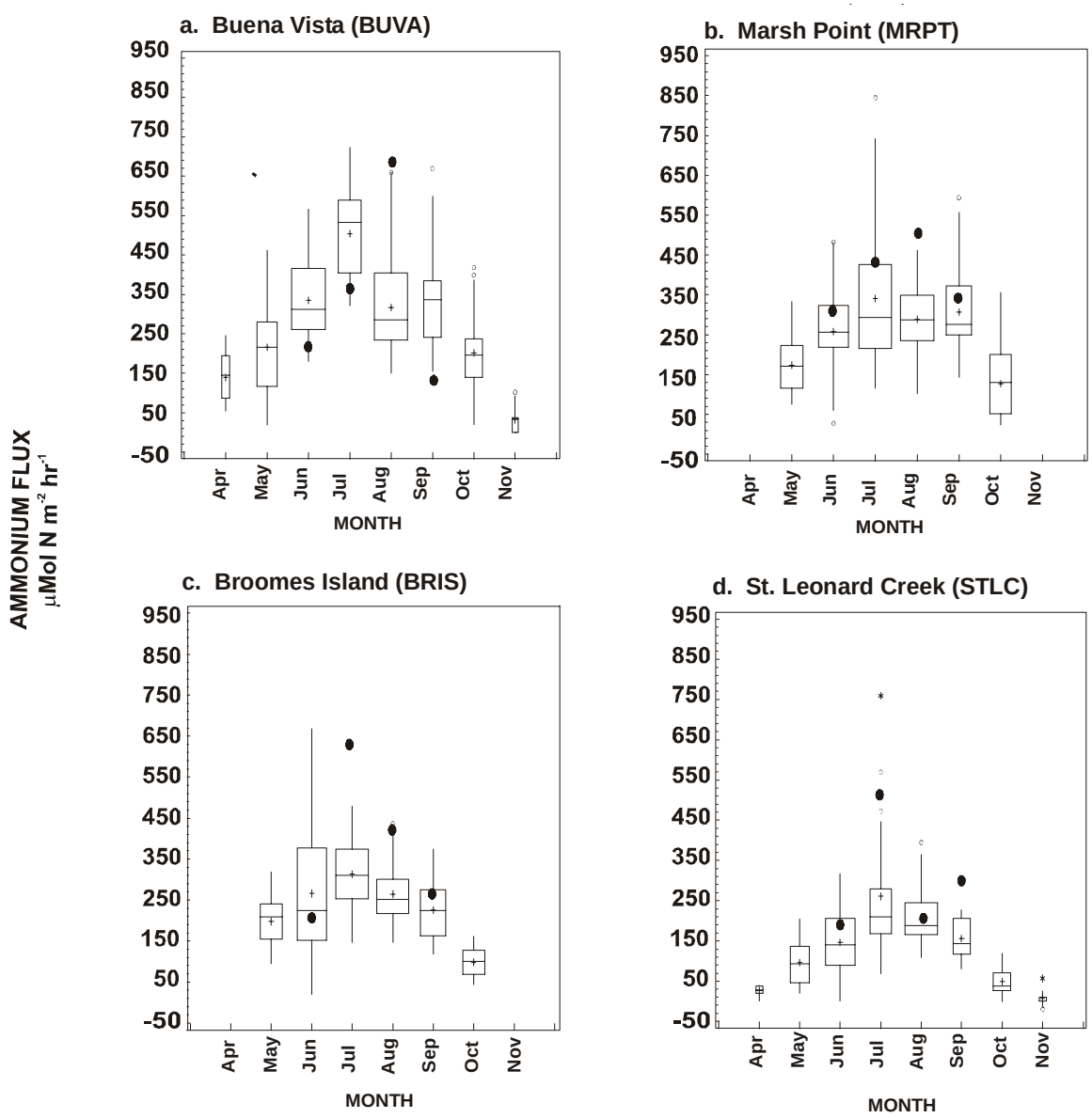

Figure 2-4.2. Box and whisker plots for ammonium (NH_4^+) flux rates for April to November at four SONE stations located in the Patuxent River.
(a) Buena Vista [BUVA] (b) Marsh Point [MRPT] (c) Broomes Island [BRIS] and (d) St. Leonard Creek [STLC].

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 2001. 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.
NI indicates that the data were not interpretable.

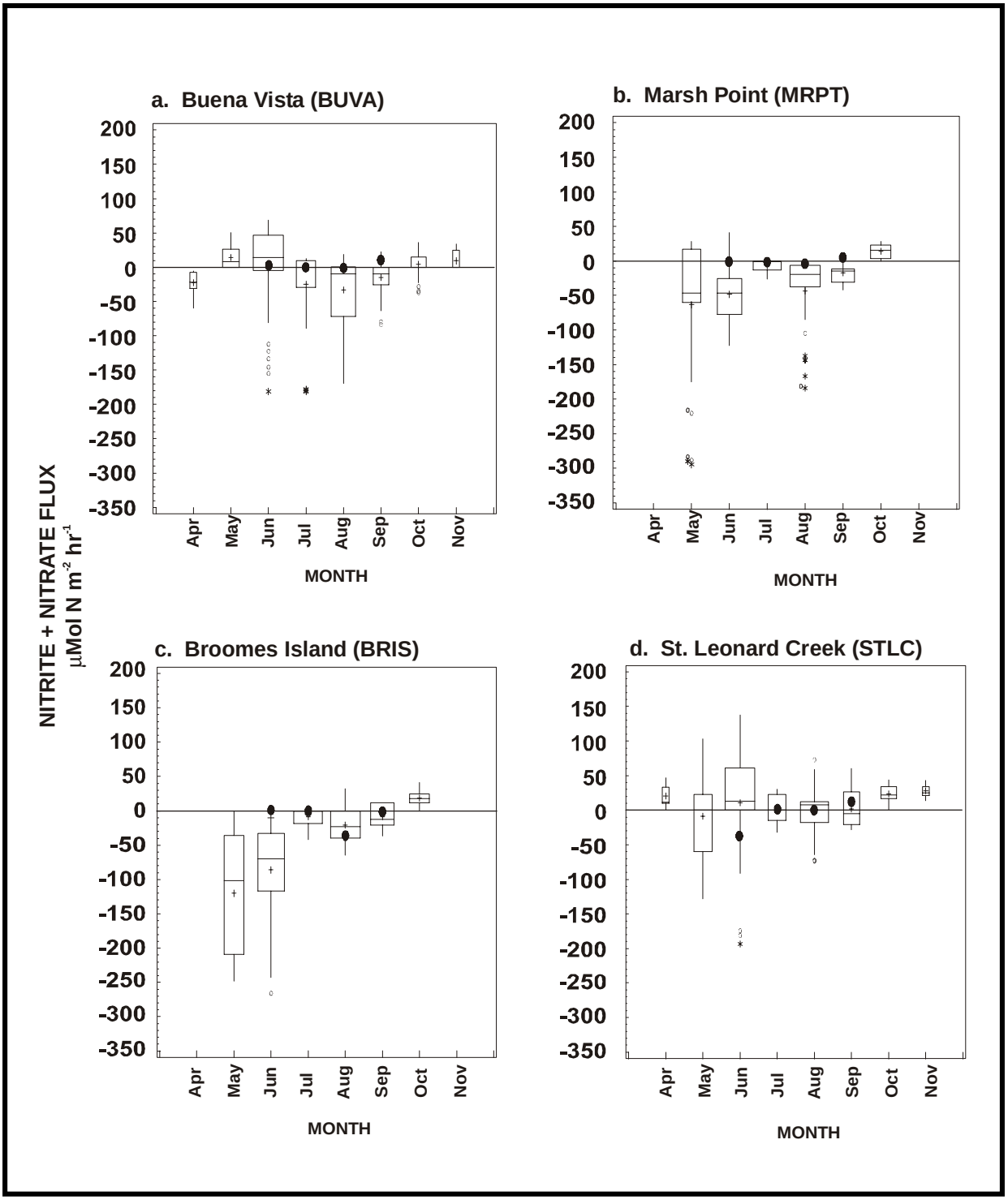

Figure 2-4.3. Box and whisker plots for nitrite plus nitrate (NO_2^- + NO_3^-) flux rates for April to November at four SONE stations located in the Patuxent River.
(a) Buena Vista [BUVA] (b) Marsh Point [MRPT] (c) Broomes Island [BRIS] and (d) St. Leonard Creek [STLC].

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 2001. 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.

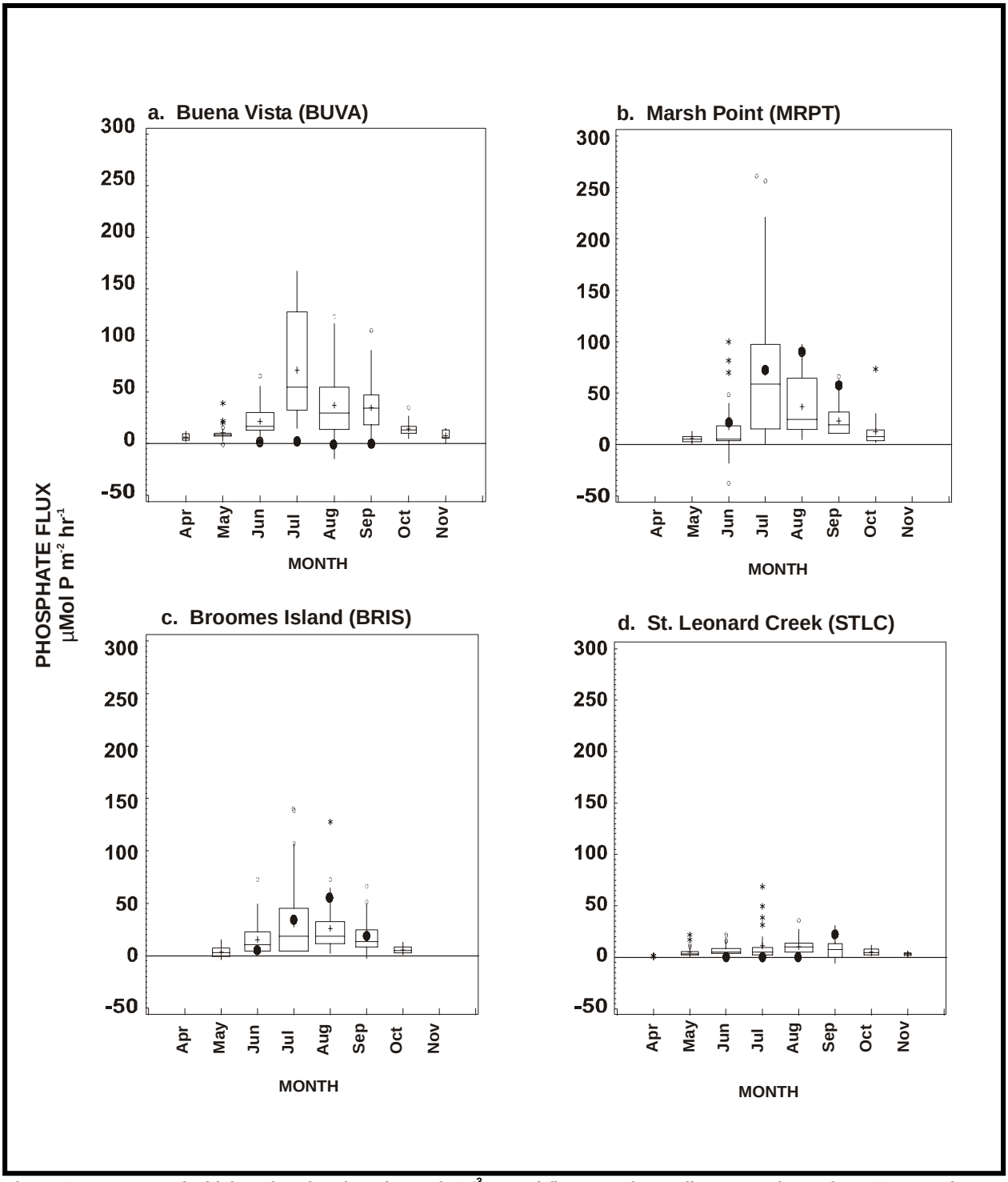

Figure 2-4.4. Box and whisker plots for phosphorus (PO_4^{-3} or DIP) flux rates for April to November at four SONE stations located in the Patuxent River.
(a) Buena Vista [BUVA] (b) Marsh Point [MRPT] (c) Broomes Island [BRIS] and (d) St. Leonard Creek [STLC].

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 2001. 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.
NI indicates that the data were not interpretable.

2001 concerns the general magnitude of the flux. In 12 of 16 observations, NH_4^+ flux was greater than the long-term mean and in 7 of 16 cases it was greater than the 75th percentile value. We would have predicted lower values for 2001 based on river flow as a surrogate for nutrient loading rates. River flow during 2001 was relatively low and we would also have predicted a small spring bloom, lower deposition rates of spring bloom organic matter to sediments and lower sediment-water fluxes. During the spring (March - May) of 2001, Mikita (2002, *unpublished*) maintained a buoy adjacent to the BRIS site. This buoy measured flourescence in surface and bottom waters at 15 minute intervals for approximately 100 days. A very large spring diatom bloom (chlorophyll-*a* ~ 50 - 70 µg l^{-1} in both surface and bottom water) was observed. While the larger bloom was not consistent with low river flow, the deposition of this bloom is consistent with larger than expected NH_4^+ fluxes.

2.6.3 Nitrite plus Nitrate ($NO_2^- + NO_3^-$) Fluxes

In general, nitrite plus nitrate ($NO_2^- + NO_3^-$) fluxes do not constitute a large fraction of the nitrogen exchange between estuarine sediments and bottom waters during summer periods. On occasion, large fluxes from water to sediments do occur but these mainly occur during early drping when NO_3^- concentrations in the water are high. Most fluxes during 2001, were small or near zero.

Even small nitrite + nitrate ($NO_2^- + NO_3^-$) fluxes from sediments to overlying waters provide a useful indication of sediment conditions. Specifically, production and release of nitrite plus 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 2001 most nitrite plus nitrate fluxes were very small and close to the long-term average. During 1998 (a wet spring) only 5 of 16 flux measurements were indicative of sediment nitrification. To provide additional 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 nitrite plus nitrate ($NO_2^- + NO_3^-$) concentrations were high. During 1995, a very low flow year, stations in the Patuxent River exhibited relatively high rates of sediment nitrate release. In fact, at the St. Leonard Creek (STLC) station sediments released nitrite plus nitrate through the entire monitoring period, a pattern never before observed. During 1999 (another very dry year) nitrite plus nitrate ($NO_2^- + NO_3^-$) fluxes were predominately positive (12 of 16 fluxes were from sediments to water). These are the types of nitrite plus nitrate ($NO_2^- + NO_3^-$) fluxes to be expected under reduced nutrient load conditions (as 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 nitrite plus nitrate ($NO_2^- + NO_3^-$) in overlying waters. The direction and magnitude of nitrite plus nitrate ($NO_2^- + NO_3^-$) fluxes between sediments and overlying waters appears to serve quite well as an indicator of sediment quality.

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

The spatial and temporal patterns of phosphorus flux in the Patuxent River in 2001 are consistent with the conceptual model of factors controlling these fluxes. At both MRPT and BRIS, stations subject to hypoxic bottom waters, fluxes were elevated while fluxes at BUVA were at record low levels. During 1999, and again in 2001, 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 not released to overlying waters.

2.7 Comparisons Among Sediment-Water Exchanges during 1999-2001

Average summer sediment oxygen consumption (SOC) in 2001 (0.63 – 1.65 g O_2 m^{-2} day^{-1}) was similar to 2000 (0.60 – 1.61 g O_2 m^{-2} day^{-1}). There were slight decreases at two of the four stations (*i.e.* BUVA and BRIS) between 1999 and 2000 although the decrease was small and probably not environmentally important, however the change was large at one station, MRPT (Figure 2-5.a.). Fluxes in SOC rates during 2000 were low (0.6 - 09 g O_2 m^{-2} day^{-1}) compared to 1999 (1.7 g O_2 m^{-2} day^{-1}) the large difference in SOC was probably caused by differences in bottom water DO conditions among these years. In 1999, DO was elevated during the summer period probably in response to a severe drought. As we have pointed out in a previous report (Boynton *et al.*, 1998), SOC rates are suppressed by low oxygen levels (2000) and enhanced at high oxygen levels (1999). In general, the approximate ranking of SOC rates among stations during 1999 - 2001 was similar to the long term pattern. For example, those stations with higher SOC rates were also those stations having high bottom water DO conditions (*i.e.*, BUVA and STLC). Those stations with low SOC rates had lower DO conditions. No significant difference found between these three in the series of paired t-tests.

Mean ammonium fluxes in 2001 (302 - 388 µM N m^{-2} hr^{-1}) were lower than in 2000 (313 – 514 µM N m^{-2} hr^{-1}). The year 2000 and 2001 NH_4^+ fluxes (paired t-test, $p = 0.02$ and $p = 0.001$ respectively) were significantly higher than in 1999 (Figure 2-5.b.). At all stations, ammonium flux was greater in 2000 and 2001 than in the drought year of 1999 and was likely due to differences in the size of the phytoplankton bloom between years.

Nitrite plus nitrate (NO_2^- + NO_3^-) flux among MINI-SONE in 2001 (3.7 - -9.2 µM N m^{-2} hr^{-1}; Figure 2-5c) was higher at most stations than in 2000 indicating minimal uptake of nitrogen by the sediments. Taking all stations into consideration, mean nitrite plus nitrate flux was more negative (into the sediment) in 2000 (+ 9.109 µM N m^{-2} hr^{-1}) compared to 1999 (-8.028 µM N m^{-2} hr^{-1}; paired t-test, $P < 0.05$) and 2001. Three of the fluxes were positive (from sediment to water) during the 1999 drought year. This pattern is thought to have resulted because of higher DO concentrations in deep waters typically associated with low flow, drought years.

Mean phosphate (PO_4^{-3}) fluxes among stations in 2001 (2 – 60 µM P m^{-2} hr^{-1}) were lower than in 2000 (22 – 105 µM P m^{-2} hr^{-1}), and similar to values observed during 1999 (6 – 39 µM P m^{-2} hr^{-1};

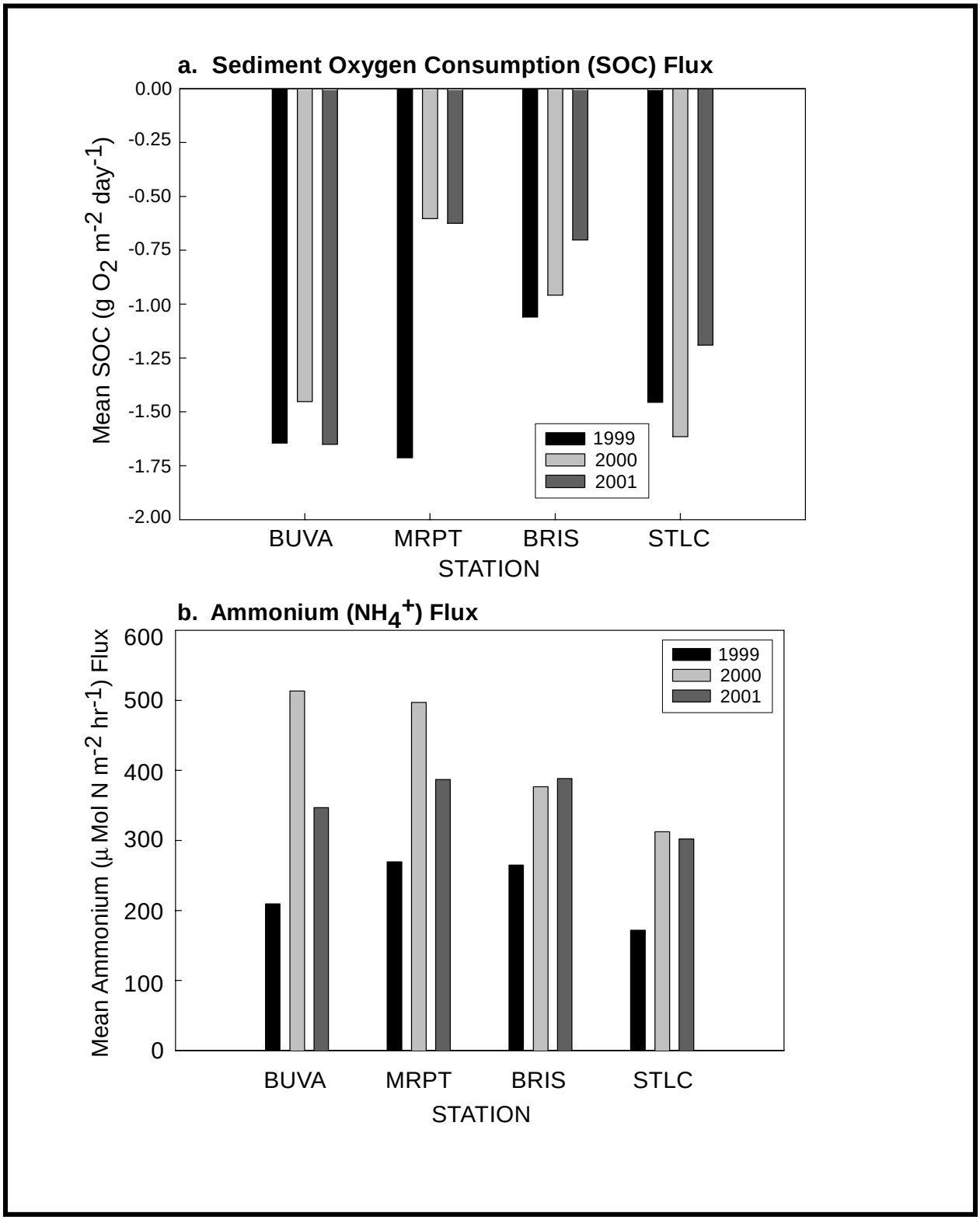

Figure 2-5. Comparison of Patuxent River MINI-SONE mean flux values calculated from monthly measurements from June through September 1999 - 2001 for:

a. sediment oxygen consumption (SOC), and

b. ammonium (NH_4^+) flux.

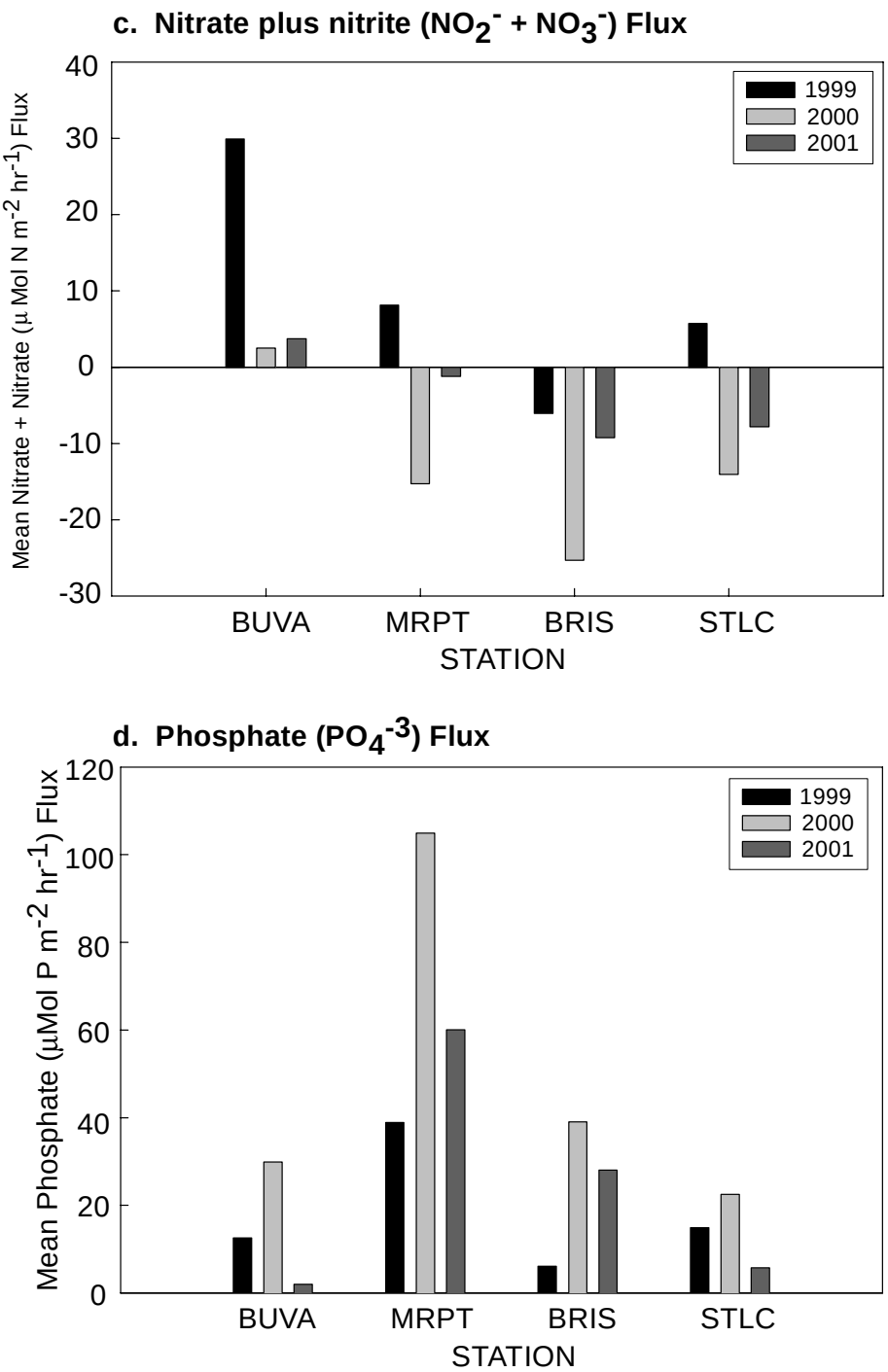

Figure 2-5. Comparison of Patuxent River MINI-SONE mean flux values calculated from monthly measurements from June through September 1999 - 2001 for:
c. nitrite plus nitrate (NO_2^- + NO_3^-), and
d. phosphate (PO_4^{-3}) flux.

Figure 2-5.d). Dissolved oxygen concentrations at the sediment-water interface probably played a role in regulating PO_4^{3-} fluxes. For example, the maximum mean phosphate (PO_4^{-3}) flux was 105 µM P m^{-2} hr^{-1} in 2000 at Marsh Point (MRPT), which was also the station having low DO conditions (<0.80 mg l^{-1}) during July through September, 2000.

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3. SEDIMENT-WATER FLUX STATUS AND TRENDS:

2001 PATUXENT RIVER STUDY

W.R. Boynton and F.M. Rohland

3. SEDIMENT-WATER FLUX STATUS AND TRENDS: 29

2001 PATUXENT RIVER STUDY 29

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With the signing of *Chesapeake 2000* a commitment was made to continue efforts to achieve and maintain the 40 percent nutrient reduction goal agreed to in 1987, as well as some additional goals, which will be adopted for the tributaries south of the Potomac River. The major goal is " to achieve and maintain the water quality necessary to support the aquatic living resources of the Bay and its tributaries and to protect human health." A part of the Ecosystem Processes Component (EPC) Program was also designed to collect the sediment-water flux data and to examine these data in order to identify long-term trends. 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 analyses 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.

In 1998, a standardized protocol was developed by the Monitoring Program to examine data for status and trend characteristics. This protocol is described below 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 comprised 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 median 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.

   The Beta distribution is a two parameter distribution whose density function is defined by the mathematical expression (Patel *et al.*, 1976):

$$f(x;a,b) = \frac{x^{a-1}\ (1-x)^{b-1}}{B(a,b)} \qquad 0 < x < 1,\ a > 0,\ b > 0$$

   The function B(a,b) is a beta function which is defined in terms of the gamma function as follows:

$$B(a,b) = \frac{\Gamma(a)\quad \Gamma(b)}{\Gamma(a+b)}$$

If the argument of the gamma function is a positive integer greater than 1, then the gamma function is define as a factorial:

$$\Gamma(a) = (a-1)!$$

which is the definition needed for this application. On other parts of its domain the gamma function is defined by a definite integral (Abramowitz and Stegun, 1972)

If the two parameters a and b are equal, then the beta distribution is symmetric.

The beta distribution arises as the sampling distribution for the median of a sample taken from a uniform distribution (Roussas, 1973). If n observations are taken from a uniform distribution, the median of these n observations will follow a beta distribution with both the a parameter and the b parameter equal to *(n+1)/2*. It is logical that the distribution of the median would be symmetric because the original uniform distribution is symmetric. If for simplicity we define *m = (n+1)/2,* then the median of the uniform data is said to follow a B(m,m) distribution. The mathematical expression is

$$B(x;m,m) = \frac{x^{m-1}(1-x)^{m-1}}{B(m.m)}$$

In Chesapeake Bay Program status calculations, the data are transformed to the uniform distribution using the probability integral transform for the log-logistic distribution. The observed median of the transformed data is taken as an indicator of status. The beta density is used to define the probability of observing a similar median from the benchmark population. If the observed median is in the upper 33% of medians from the benchmark population, status is rated as good. If the observed is in the middle 33% status is rated as fair. An observed median in the lower 33% rates as 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 cannot 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.

3.1.2 Notes on the Current Status for the Patuxent River

A median value for the years 1999, 2000 and 2001 was calculated. The use of the last three years of data provides an "indicator" value of the status of the parameter relative to measurements taken in the benchmark period. 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 1999 was an extremely dry year until September when several hurricanes passed the area, 2000 exhibited a modest spring peak and low flows through the summer and fall while 2001 was very similar to 2001 with modest peaks in April and June and low riverflow values during the second half of the year.

3.1.3 Evaluation of the Current Status for the Patuxent River

i. Sediment Oxygen Consumption (SOC)

The current status (median of 1999, 2000 and 2001 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 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 that were poor and two in the good range. Over the last six years the pattern of SOC flux in the Patuxent River has provided substantiation that the benchmark is appropriate. The six- year record indicates that SOC fluxes progress from good down-river to fair 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. The two stations at the head and the mouth of the river (Buena Vista [BUVA] and St. Leonard Creek [STLC]) have had a consistent pattern where the status has been good over the six years while the two mid stations (Marsh Point [MRPT] and Broomes Island [BRIS]) show status changes from fair to poor over the six year period.

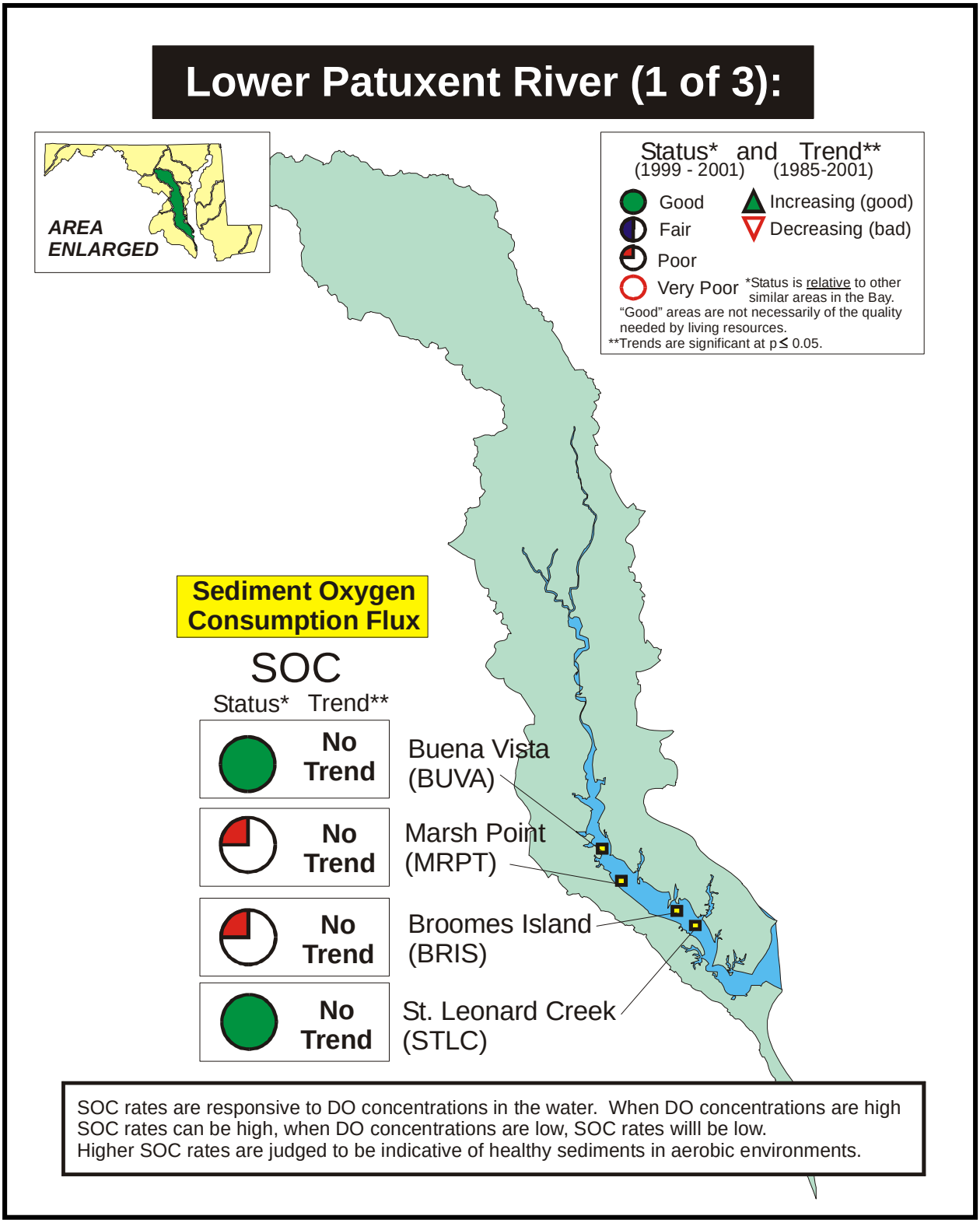

Figure 3-1.a. Map showing status and trends at four stations in the Lower Patuxent River for sediment oxygen consumption (SOC) fluxes (observed data).

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

ii. Ammonium (NH_4^+)

The current status (median of 1999, 2000 and 2001 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. All four SONE stations in the Patuxent River were in the poor range during 2001. It should be noted that high river flow years have a strong influence on ammonium fluxes (fluxes increase). However, all three years in this analysis exhibited modest to low flows. In contrast to river flow and associated nutrient loads, spring chlorophyll-*a* concentrations in the vicinity of BRIS were very high in 2000 and 2001. When this material sank to the bottom it provided ample labile organic material to support high NH_4^+ fluxes.

iii. Nitrite (NO_2^-)

The current status (median of 1999, 2000 and 2001 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 fair 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. The poor status at Broomes Island (BRIS) in 1999 changed to good in 2000 and to fair in 2001. The six-year pattern shows an improvement of the status at all four stations.

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

The current status (median of 1999, 2000 and 2001 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, one was judged to be good, Buena Vista (BUVA). The other three stations, Broomes Island (BRIS), Marsh Point (MRPT) and St. Leonard Creek (STLC), were fair. The six year pattern shows some improvement of the status at all four stations.

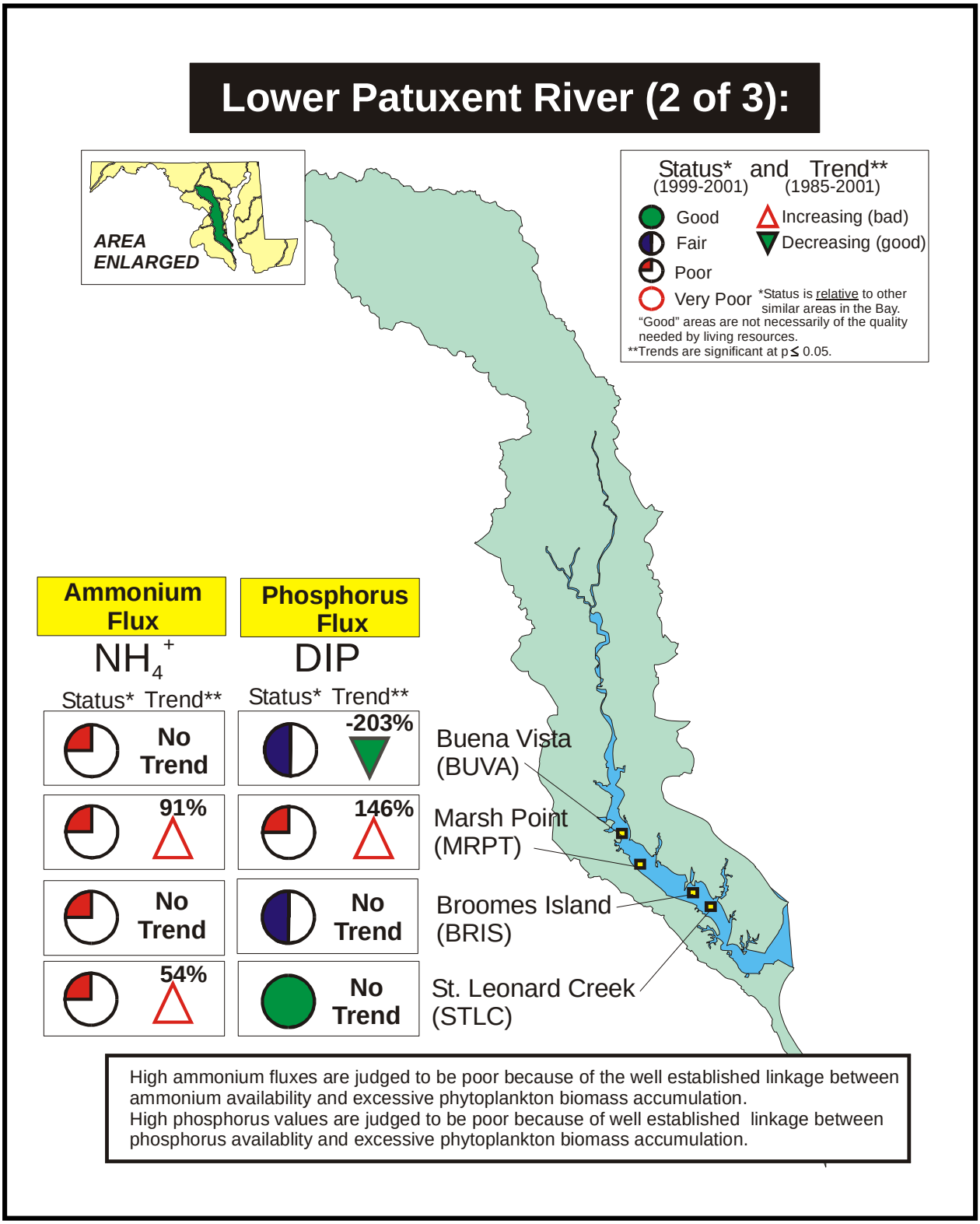

Figure 3-1.b. Map showing status and trends at four stations in the Lower Patuxent River for ammonium (NH_4^+) and phosphorus (PO_4^{-3}) fluxes (observed data).

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

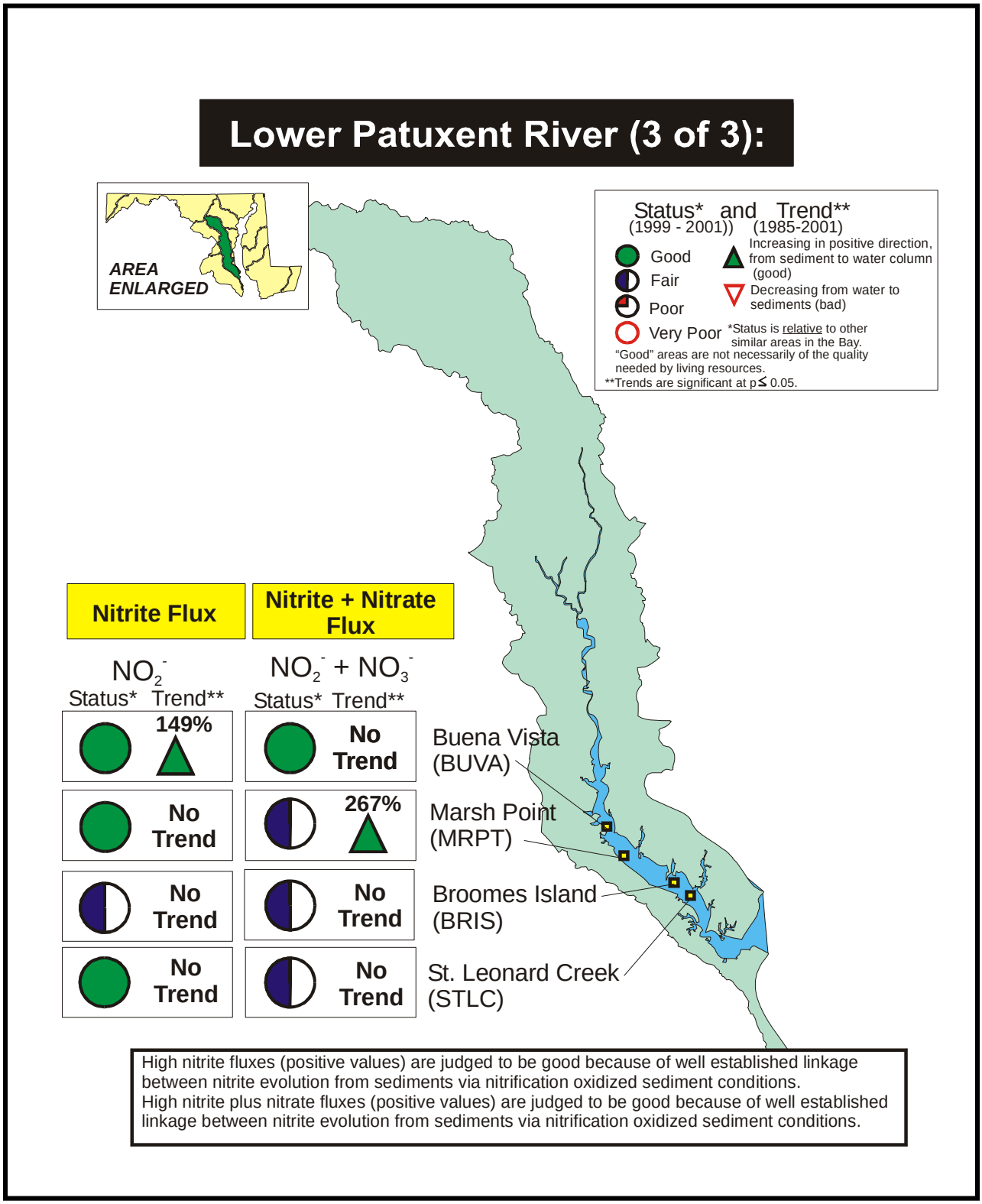

Figure 3-1.c. Map showing status and trends at four stations in the Lower Patuxent River for nitrite (NO_2^-) and nitrite plus nitrate (NO_2^- + NO_3^-) fluxes (observed data).

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

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

The current status (median of 1999, 2000 and 2001 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, Buena Vista (BUVA) and Broomes Island (BRIS). Marsh Point (MRPT) was in the poor category while St. Leonard Creek (STLC), the station farthest downstream, which went from good to fair in 2000, reverted back to good in 2001. It should be noted that high river flow years have a particularly strong influence on phosphorus fluxes (fluxes increase) and all three years considered in this evaluation had low to modest flows.

3.2 Sediment-Water Oxygen and Nutrient Exchanges (SONE) Trends: 2001 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 adjusted for river flow, as is the case for testing other variables for trends within the monitoring program. This adjustment 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 - 2001 Data from the Patuxent River

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

Flux data were collected over a period of seventeen years (1985 - 2001) 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, including sediment oxygen consumption (SOC), ammonium (NH_4^+), inorganic phosphorus, nitrite (NO_2^-) and nitrite plus nitrate ($NO_2^- + NO_3^-$) fluxes, 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 six statistically significant results ($p < 0.05$). In the Patuxent River estuary no significant trends were found for sediment oxygen consumption (SOC) fluxes. A significant increasing trend (at probability level $p < 0.01$) was indicated for ammonium (NH_4^+) at St. Leonard Creek (STLC) and a lesser trend ($p < 0.05$) at Marsh Point (MRPT). A significant increasing trend ($p < 0.05$) for nitrite (NO_2^-) was found at Buena Vista (BUVA) and a positive trend ($p < 0.05$) at Broomes Island (BRIS) for nitrate plus nitrite ($NO_2^- + NO^{3-}$). Significant annual trends ($p < 0.05$) for dissolved inorganic phosphorus (DIP) were found at Buena Vista (BUVA) and Marsh Point (MRPT). During the last seventeen years both wet and dry years have been recorded (relatively high and low diffuse source loading years, respectively) which tend to produce high and low sediment fluxes. Since high/low load years have occurred without pattern, trends are difficult to detect unless they are large and persist for several years.

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					* ↓				
STLC						* ↓			
b. Ammonium (NH_4^+; μM N m^{-2} hr^{-1} yr^{-1})									
BUVA					** ↑				
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})									
BRIS		* ↑							
MRPT ↑									*
e. Dissolved Phosphorus (PO_4^{-3}; μM Pm^{-2} hr^{-1} yr^{-1})									
BUVA									* ↓
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 = 0.05, ** 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	-48	118	34	-33	-2
p value	0.24	0.004**	0.30	0.43	0.98
Slope	-0.020	3.489	0.289	-0.467	0.000
Marsh Point (MRPT)					
Sign	-27	75	45	64	73
p value	0.40	0.02*	0.13	0.04*	0.02*
Slope	-0.009	13.063	0.321	1.697	1.833
Broomes Island (BRIS)					
Sign	-37	27	-12	53	-5
p value	0.25	0.40	0.72	0.08	0.90
Slope	-0.024	3.533	0.000	0.245	-0.030
Buena Vista (BUVA)					
Sign	-74	36	68	19	-79
p value	0.08	0.39	0.04*	0.65	0.05*
Slope	-0.041	2.888	0.748	0.171	-1.063

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)

A significant negative (improving) trend continues for sediment oxygen consumption (SOC) at Buena Vista (BUVA) for August and a significant negative trend at St. Leonard Creek (STLC; $p < 0.05$) for September (Table 3-3.a).

ii. Ammonium (NH_4^+)

A significant trend was indicated for ammonium (NH_4^+) fluxes at $p < 0.01$ in August at Buena Vista (BUVA; degrading trend). The trends in May and August at Marsh Point (MRPT; degrading trend) and at St. Leonard Creek (STLC) in August (degrading trend; Table 3-3.b) weakened still further ($p < 0.03$).

iii. Nitrite (NO_2^-)

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

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

A positive (improving) significant trend was indicated for nitrite plus nitrate fluxes ($NO_2^- + NO_3^-$) fluxes ($p < 0.05$) at Broomes Island in June (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}) flux ($p < 0.02$) 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 - 2001								
Sign	3	-10	22	-26	-58	7	-9	-3
p value	.	0.28	0.39	0.13	0.02*	0.64	0.24	.
N	3	8	17	13	17	11	7	3
Marsh Point (MRPT): 1989 - 2001								
Sign		-3	15	-27	-7	-2	-3	
p value		0.72	0.34	0.11	0.71	0.94	1.00	
n		6	12	13	13	11	6	
Broomes Island (BRIS): 1989 - 2001								
Sign		5	22	-16	-22	-15	-11	
p value		0.47	0.15	0.36	0.22	0.27	0.06	
n		6	12	13	13	11	6	
St. Leonards Creek (STLC): 1985 - 2001								
Sign	3	-10	44	-31	-20	-26	-5	-3
p value	.	0.28	0.08	0.07	0.39	0.05*	0.56	.
n	3	8	17	13	16	11	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 - 2001								
Sign	-3	10	-18	10	62	-23	-3	1
p value	.	0.28	0.44	0.58	0.01**	0.09	0.77	.
n	3	8	16	13	17	11	7	3
Marsh Point (MRPT): 1989 - 2001								
Sign		13	8	12	40	-7	9	
p value		0.02*	0.63	0.50	0.02*	0.64	0.14	
n		6	12	13	13	11	6	
Broomes Island (BRIS): 1989 - 2001								
Sign		-3	0	30	-14	13	1	
p value		0.72	1.00	0.08	0.43	0.35	1.00	
n		6	12	13	13	11	6	
St. Leonards Creek (STLC): 1985 - 2001								
Sign	1	-4	29	20	54	13	5	0
p value	.	0.72	0.25	0.19	0.03*	0.35	0.56	.
n	3	8	17	12	17	11	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^-; μM N m^{-2} hr^{-1} yr^{-1})

STATION	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV
PATUXENT RIVER: Buena Vista (BUVA): 1985 - 2001								
Sign	0	13	-11	36	11	13	6	0
p value	.	0.02*	0.52	0.03*	0.58	0.35	0.23	.
n	1	6	13	13	14	11	5	1
Marsh Point (MRPT): 1989 - 2001								
Sign		3	-3	15	14	5	11	
p value		0.72	0.88	0.28	0.43	0.76	0.06	
n		6	12	12	13	11	6	
Broomes Island (BRIS): 1989 - 2001								
Sign		-3	-10	-8	-3	8	4	
p value		0.72	0.52	0.66	0.90	0.58	1.00	
n		6	12	13	13	11	6	
St. Leonards Creek (STLC): 1985 - 2001								
Sign	0	1	-19	16	23	10	3	0
p value	.	1.00	0.27	0.29	0.22	0.48	0.72	.
n	1	6	13	12	14	11	6	1

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

STATION	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV
PATUXENT RIVER: Buena Vista (BUVA): 1985 - 2001								
Sign	-3	-10	1	21	3	15	-8	0
p value	.	0.28	1.00	0.22	0.93	0.28	0.38	.
n	3	8	17	13	16	11	7	3
Marsh Point (MRPT): 1989 - 2001								
Sign		-5	26	15	16	9	3	
p value		0.47	0.09	0.37	0.36	0.53	0.72	
n		6	12	13	13	11	6	
Broomes Island (BRIS): 1989 - 2001								
Sign		-3	33	14	5	3	1	
p value		0.72	0.03*	0.33	0.81	0.88	1.00	
n		6	12	13	13	11	6	
St. Leonards Creek (STLC): 1985 - 2001								
Sign	-3	2	-34	12	-27	11	7	-1
p value	.	0.90	0.17	0.44	0.28	0.44	0.38	.
n	3	8	17	12	17	11	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 - 2001								
Sign	-3	2	-33	-22	4	-19	-9	1
p value	.	0.90	0.11	0.20	0.90	0.16	0.24	.
n	3	8	15	13	17	11	7	3
Marsh Point (MRPT): 1989 - 2001								
Sign		1	36	8	8	9	11	
p value		1.00	0.02*	0.67	1.00	0.53	0.06	
n		6	12	13	13	11	6	
Broomes Island (BRIS): 1989 - 2001								
Sign		3	-4	6	-22	9	3	
p value		0.72	0.84	0.76	0.20	0.53	1.00	
n		6	12	13	13	11	6	
St. Leonards Creek (STLC): 1985 - 2001								
Sign	-2	4	-9	13	-2	-8	1	1
p value	.	0.72	0.74	0.46	0.97	0.58	1.00	.
n	3	8	17	13	17	11	7	3

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4. SUBMERGED AQUATIC VEGETATION (SAV) HABITAT EVALUATION

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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 (Kemp *et al*., 1983; Orth and Moore, 1983; Cambridge and McComb, 1984; Orth and Moore, 1984). In response to these changes, 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 availability to rooted macrophyte populations (Sand-

Jensen, 1977; Cambridge and McComb, 1984; Kemp *et al*., 1983; Twilley *et al*., 1985; Silberstein, 1986). While light availability is generally agreed to be the most critical resource limiting the extent and distribution of SAV populations, an understanding of what conditions are necessary and sufficient to provide adequate light has proven to be most elusive. For example, a number of studies have demonstrated that epiphytes can substantially reduce the amount of available light reaching the leaf surface (*e.g.*, Horner, 1987; Burt *et al*., 1995; Stankelis *et al.*, 1999). 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 (Stankelis *et al.*, 1999), nutrient availability (Kemp *et al*., 1983; Burt *et al*., 1995), wave action (*e.g.* Koch, 1996) and leaf turnover rates. Because of this inherent complexity and the difficulties of determining mechanisms and causal factors, field monitoring of water quality remains an important tool for documenting conditions at specific locations where SAV thrives, survives or declines. In Chesapeake Bay, field monitoring is particularly important because of the large range of conditions found within the Bay and it's tributaries.

In 2001, measurements of water quality and epiphyte fouling rates were made concurrently at several near-shore monitoring locations distributed within several mesohaline areas of Chesapeake Bay. These concurrent measurements made at a variety of locations that both support SAV and those that do not, allow for a level of comparative ecology not possible before and help shed light on the importance of epiphytic fouling for SAV growth and survival in Chesapeake Bay. Measurements at sites located in Tangier Sound and the mouth of the Patuxent River were supported by the EPC component of the Maryland Chesapeake Bay Water Quality Monitoring Program. Monitoring at other sites located in the lower Potomac River were supported by the new Woodrow Wilson Bridge SAV mitigation project. All measurements however, were made to conform to standards established by the EPC. The work reported here has been divided into two complementary components, the Near-shore Water Quality Evaluation and the Epiphyte Growth Study.

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 (USEPA, 2000). These five water quality parameters thought to be most important for SAV growth and survival 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*). Although DATAFLOW high-resolution sampling is being developed for the evaluation of shallow-water habitats, the vast majority of routine water-quality monitoring is still done at river channel locations often distant from actual SAV habitats. Consequently, these off-shore data may not reflect near-shore conditions due to a variety of localized conditions such as: resuspension of sediments, point source discharges, or existing macro algal communities. Therefore, data for this study, collected directly in near-shore SAV habitats, will provide more exact information about water quality conditions at 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.

4.1.2 Epiphyte Growth Study

The epiphyte growth study was designed to compare epiphyte accumulation rates to water quality data at selected sites in Tangier Sound as well as the lower Potomac River and a long-term site at the mouth of the Patuxent River. This comparison will provide field data for calibration of models predicting epiphyte biomass based upon, water quality data. In 1998, a comparison of epiphyte fouling rates on live SAV and Mylar® strips was conducted to compare epiphytic growth rates on transplanted live SAV to the artificial substrates to help calibrate and interpret results obtained using artificial substrates. The results of that study suggested that Mylar® strips could be used as an acceptable surrogate for live plants in order to estimate light attenuation from epiphytic fouling (Stankelis *et al*. 1999). Despite potential limitations, artificial substrates can be used effectively to compare the effects of differing water quality conditions on epiphyte accumulation rates and light attenuation when live plants are not available (e.g., Burt *et al*., 1995; Pinckney and Micheli, 1998; Stankelis *et al.*, 1999). In addition, artificial substrates can be standardized between sites, and provide a quick assessment of epiphyte growth potential at SAV restoration sites.

4.2 Station Locations and Sampling Dates

4.2.1 Near-shore Water Quality Evaluation

4.2.1.1 Water Quality Station Locations

In 2001, six stations were monitored in Tangier Sound as well as a single station in the lower Patuxent River estuary. The six stations in Tangier Sound were selected to provide a variety of water quality and wave exposure conditions (Figure 4-1.a, Table 4-1). The Patuxent River station (SV09), located on the sand flat adjacent to CBL, has been monitored since 1997 (Figure 4-1.b, Table 4-1). Additional sites located in the lower Potomac are shown in Figure 4-1.c and Table 4-1.

4.2.1.2 Water Quality Sampling Frequency

Sampling was conducted in three seasonal time blocks (spring, summer and fall). Four weekly samples were collected during each seasonal block for a total of 12 SAV sampling cruises (Table 4-2).

4.2.2 Epiphyte Growth Survey

4.2.2.1 Station Locations and Sampling Frequency

The epiphyte growth survey was completed concurrently with the near-shore water quality evaluation (Table 4-1, Table 4-2).

Table 4-1 Station codes, grid location, and nearest DNR station.

Geographic Location	Station Code	Latitude	Longitude	Nearest DNR Station	Bay Segment
Janes Island North	JI1G	38º 01.620	75º 50.509	ET9.1	BIGMH
Janes Island South	JI2G	37º 58.249	75º 52.609	EE3.2	TANMH
Manokin River	MRGC	38º 08.835	75º 50.349	ET8.1	MANMH
Smith Island Big Thoroughfare	SIBT	37º 58.147	75º 59.553	EE3.2	TANMH
Smith Island Back Cove	SIBC	38º 01.262	76º 00.133	EE3.2	TANMH
South Marsh Is. South Point (CBL)	SMSP	38º 04.571	76º 01.653	EE3.2	TANMH
Patuxent River	SV09	38º 19.016	76º 27.119	LE1.4	PAXMH
Potomac River Piney Point	PRPP	38º 08.307	76º 30.265	LE2.2	POTMH
Potomac River Judith Sound	PRJS	38º 00.355	76º 28.082	LE2.2	POTMH
Potomac River Calvert Bay	PRBD	38º 06.026	76º 23.626	LE2.2	POTMH
Potomac River Sage Pond	PRSP	38º 07.413	76º 25.795	LE2.2	POTMH

Table 4-2. Sampling dates for water quality measurements and epiphyte rate measurements in 2001.

Region	Stations	Water Quality Measurements	Epiphyte Rate Measurements
Tangier Sound	JI1G, JI2G, MRGC, SMSP, SIBT, SIBC	5/15, 5/23, 5/29, 6/5 7/9, 7/17, 7/23, 7/31 9/17, 9/25, 10/1, 10/9	5/23, 5/29, 6/5 7/17, 7/23, 7/31 9/25, 10/1, 10/9
Lower Potomac	PRSP PRPP, PRJS, PRBD,	5/18, 5/25, 6/1, 6/9 7/13, 7/20, 7/27, 8/3 9/14, 9/21, 9/28, 10/5	5/25, 6/1, 6/9 7/20, 7/27, 8/3 9/21, 9/28, 10/5
(CBL) Patuxent River	SV09	5/24, 5/31 7/9, 7/16, 7/23, 7/30 10/02, 10/10, 10/19	7/16, 7/23, 7/30 10/02, 10/10, 10/19

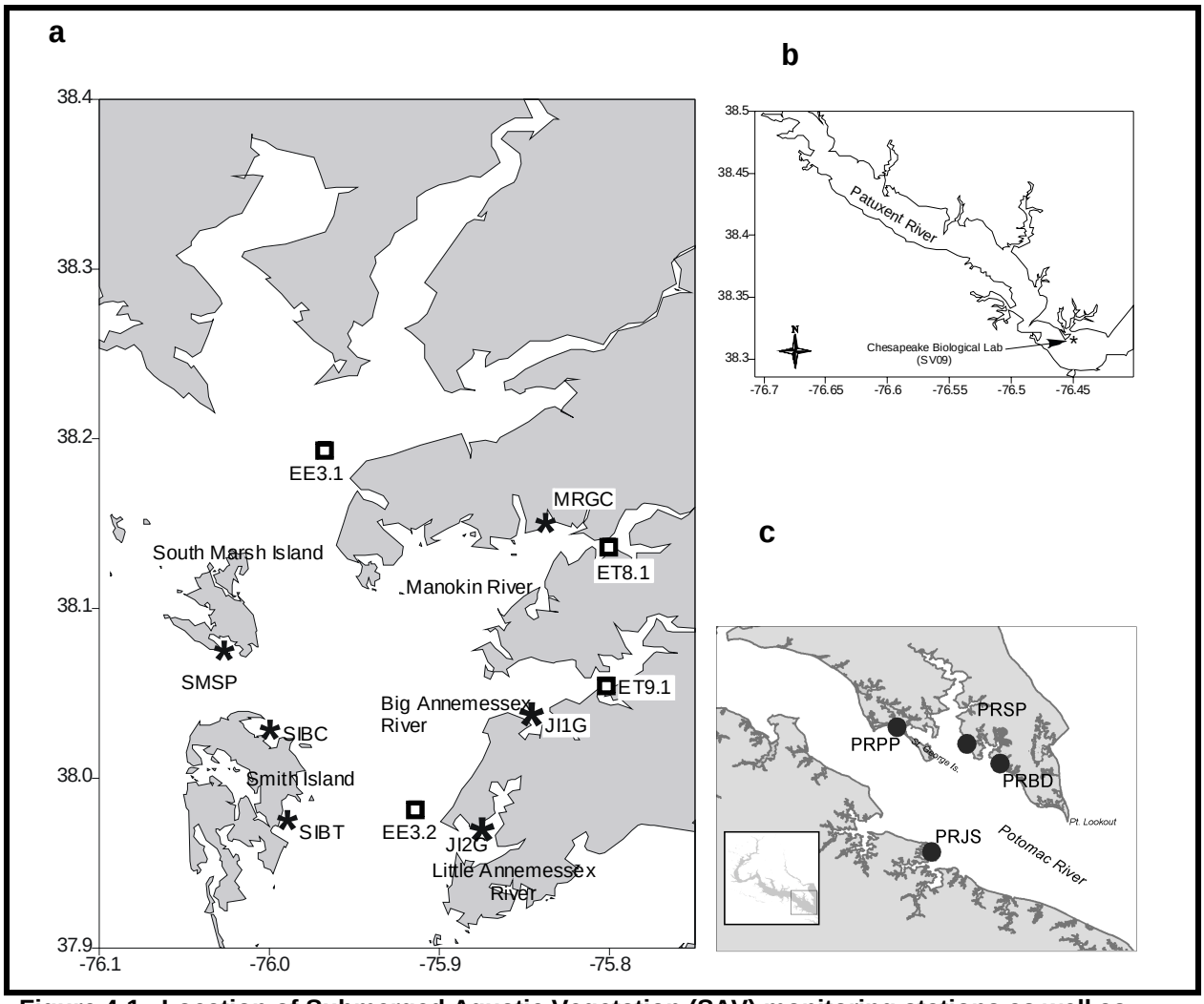

Figure 4-1. Location of Submerged Aquatic Vegetation (SAV) monitoring stations as well as nearest DNR monitoring sites in (a) Tangier Sound, (b) Patuxent River and (c) lower Potomac River, in 2001. *Latitude and longitude are in decimal degrees.*

4.3 Field Methods

4.3.1 Physical Parameters

Temperature, salinity, conductivity, and dissolved oxygen measurements were made with a Yellow Springs International (YSI) 600R, YSI 6920 or YSI 6600 multi-parameter water quality monitor suspended at 0.5 meters below the water surface. Water column turbidity was estimated with a secchi disk where possible, while water column light flux in the photosynthetically active frequency range (PAR) was measured with a Li-Cor LI-192SA underwater quantum sensor. When possible, measurements were collected at three discrete water depths in order to calculate water column light attenuation (Kd). Weather and sea-state conditions such as air temperature, percent cloud cover, approximate wind speed and direction, total water depth, and wave height, were also recorded.

4.3.2 Water Column Nutrients, Chlorophyll-*a* and Suspended Solids

Whole water samples were collected at approximately 0.5 meters below the water surface by using a hand held bilge pump or the outflow from the DATAFLOW intake. A portion was immediately filtered with a 25 mm, 0.7 µm (GF/F) glass fiber filter. Both the filtered portion and the remaining whole water samples were placed in coolers for transport back to the laboratory for further processing. The filtered portion was analyzed by the Nutrient Analytical Services Laboratory (NASL) for ammonium (NH_4^+), nitrate (NO_2^-), nitrite plus nitrate ($NO_2^- + NO_3^-$) and phosphate (PO_4^{-3}). Whole water portions were filtered in the laboratory using 47 mm, 0.7 µm (GF/F) glass fiber filters and were transferred to NASL for analysis of the following parameters: total suspended solids (TSS), total volatile solids (TVS), and total and active chlorophyll-*a* concentrations, where total chlorophyll-*a* includes chlorophyll-*a* plus breakdown products.

4.3.3 Epiphyte Growth Measurement Method

In order to assess the light attenuation potential of epiphytic growth on the leaves of submerged aquatic vegetation (SAV) artificial substrata, thin strips of Mylar® polyester plastic, were deployed at each sampling location for a period of 6 to 8 days. The use of transparent Mylar® plastic provided a means to estimate light attenuation due to epiphytic growth and sediment accumulation, as well as to quantify the organic and inorganic components of the fouling.

4.3.3.1 Description of Epiphyte Collector Arrays

Each collector array (Figure 4-2) consisted of a square PVC frame with a vertical PVC shaft in the center of the square. To this shaft was attached a line with a small surface float that allows

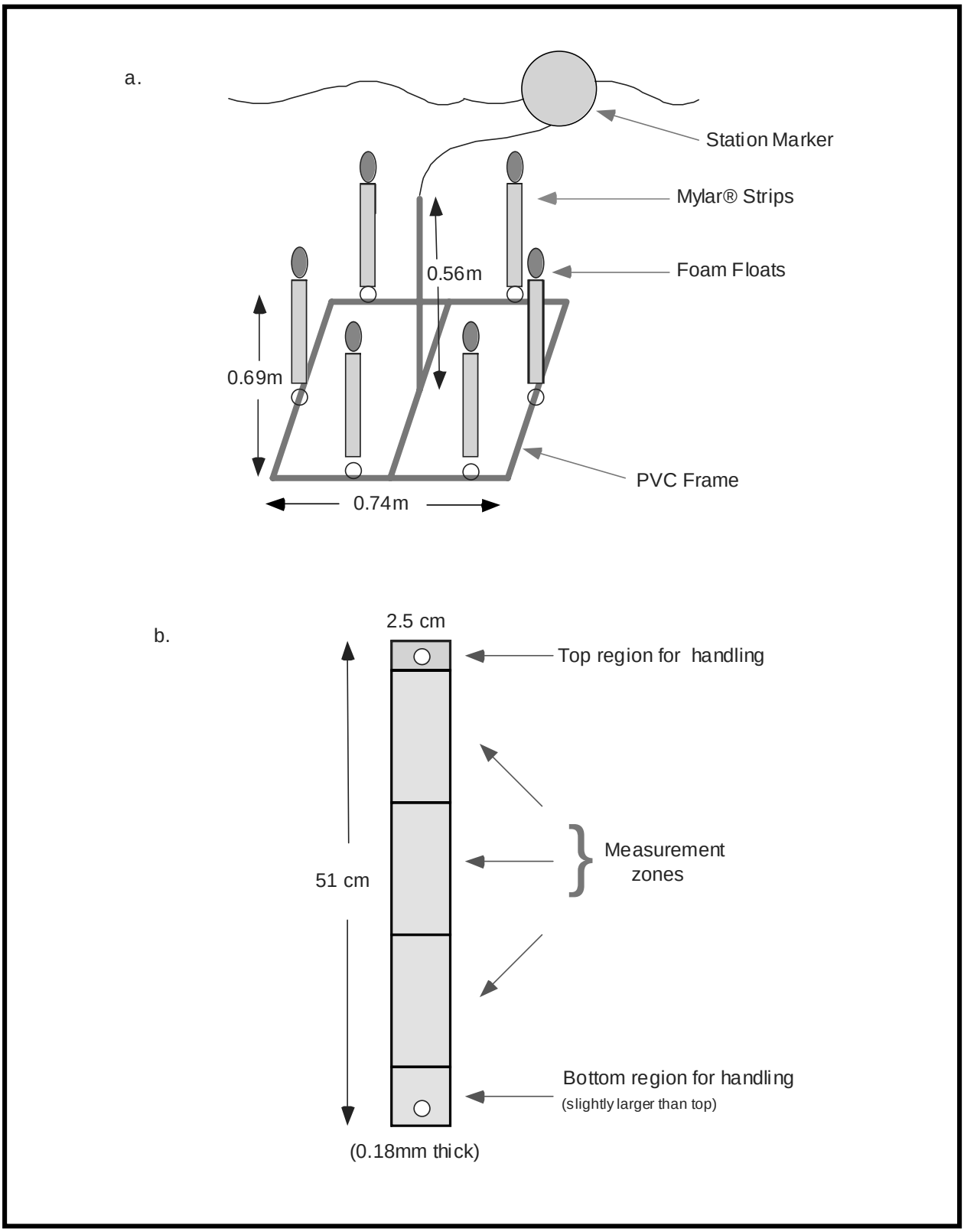

Figure 4-2. Diagram of SAV Epiphyte Collector Array.

a. Epiphyte Collector Array

b. Mylar® strips

for easy location of the collector. Each collector array held up to six strips per deployment. Mylar® strips (2.5 cm wide x 51 cm long and 0.7 mil thick) were attached to the frame so that the top was allowed to move freely in the water column. Small foam floats (~3.5 x 3.3 cm) were attached to the top of the strip to help maintain a vertical position in the water column at all times.

4.3.3.2 Sampling the Epiphyte Collector Arrays

On each sampling date, two Mylar® strips were collected, one to be analyzed for chlorophyll-*a* mass, and another for total dry mass/inorganic dry mass. While suspended in the water, Mylar® strips were gently removed from the array and cut with scissors to remove the middle 1/3 marked section (64.5 cm^2, Figure 4-2). This section was once again cut in half and placed in a 60 ml plastic centrifuge tube for transport back to the laboratory. The tube was then placed in a cooler for transport back to the laboratory. The samples were immediately frozen upon arrival at the laboratory prior to further processing.

4.3.3.3 Processing Organic/Inorganic Epiphyte Material

The Mylar® strip sections collected for dry mass/inorganic mass analysis were scraped of all material and rinsed with distilled water. Scraped material and rinse water were diluted to a fixed volume (300 - 500 ml). The solution was mixed as thoroughly as possible on a stir plate until homogenized. A small aliquot (10 to 50 ml) was then extracted with a glass pipette and filtered through a 47 mm, 0.7 µm (GF/F) glass fiber filter. Once filtered, the pads were immediately frozen and delivered to NASL for analysis.

4.3.4 Chemical Analysis Methodology

Methods for the determination of dissolved nutrients were as follows: ammonium (NH_4^+), nitrite (NO_2^-), nitrite plus nitrate ($NO_2^- + NO_3^-$), and dissolved inorganic phosphorus (DIP or PO_4^-) were measured using the automated method of EPA (1979). Methods of Strickland and Parsons (1972) and Parsons *et al.* (1984) were followed for chlorophyll-*a* analysis. Total suspended solids (TSS) and total volatile solids (TVS) were measured with a gravimetric method.

4.3.5 Estimating Epiphyte Light Attenuation

Estimates of epiphyte light attenuation were calculated using measurements of epiphyte dry mass and existing relationships between dry mass and light attenuation (Figure 4-3.a, 4-3.b). These relationships were developed using direct measurements of epiphyte light attenuation and dry

mass accumulated on Mylar® strips deployed at a number of locations from 1997 to 1999 (Boynton *et al*. 1998; Stankelis *et al*., 1999; Stankelis *et al*., 2000). These estimates along with corresponding measurements of water column light attenuation (Kd) allow us to calculate the percent of surface light reaching the depth of the SAV blade through the water column (PLW) and the percent surface light reaching the blade of SAV through the epiphyte layer at the leaf surface (PLL). Calculations of these metrics defined by the Chesapeake Bay Program (USEPA, 2000) are shown below in Table 4-3.

Table 4-3. Calculation of % Surface Light Reaching Leaf Surface (PLL)

$PLW = (I_Z/I_0)*100 = 100*[e^{-kd*Z}]$	Where:	Iz = Light flux (PAR) at depth
$PLL = [e^{-kd*Z}][1-LA/100]$		I_0 = Light flux (PAR) at surface
		LA = Epiphyte light attenuation
		Z = Observation depth (m)

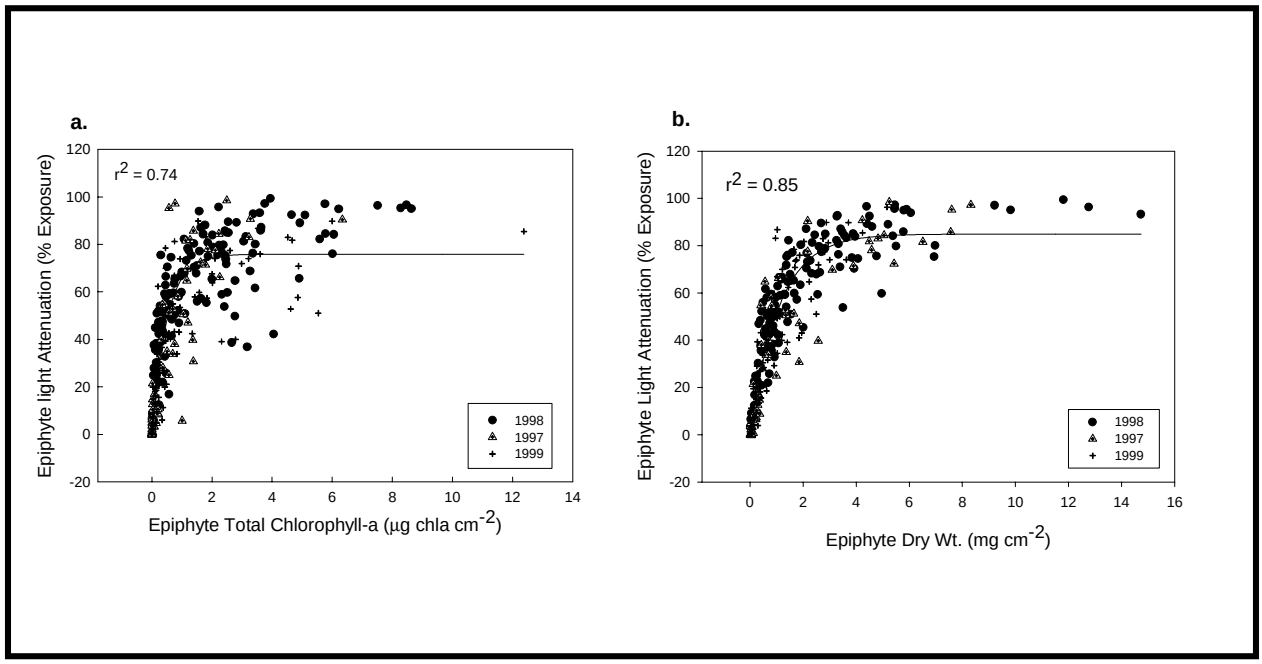

Figure 4-3. (a) Epiphyte light attenuation vs. epiphyte chlorophyll-*a*, where light attenuation = $77.36*(1-e^{-2.082 * Epi\ Chla})$ and (b) epiphyte light attenuation vs. epiphyte dry mass where light attenuation = $84.634*(1-e^{-0.963 * Epi\ drywt})$.

4.4 Results

4.4.1 Results of Near-shore Water Quality Evaluation

Results presented in this section include data collected as part of the Ecosystems Processes Component (EPC) as well as similar information collected as part of the new Woodrow Wilson Bridge mitigation project in the lower Potomac River. The stations monitored in the lower Potomac share many of the same characteristics of sites located in Tangier Sound and the Patuxent River and are thus particularly well suited for the comparative type of presentation that follows in this section. Both sets of data were collected within the same time frame and with similar techniques.

4.4.1.1 Physical Parameters

The full data set is available in Ecosystems Processes Component, Level One Report # 19, Data Report (Boynton *et al*., 2002).

4.4.1.2 Dissolved Nitrogen Concentrations (DIN)

There was a strong seasonal pattern in dissolved inorganic nitrogen (DIN) concentrations at all stations. Higher concentrations were found in the spring followed by progressively lower concentrations during summer and fall (Fig.4-4). At all stations, DIN concentrations remained below the 10.7 μmol l^{-1} N mesohaline habitat limit established by the USEPA (2000). The maximum DIN concentration (10.57 μmol l^{-1} N) was recorded at station SV09 on May 24, 2001 while the minimum concentration recorded was 0.16 μmol l^{-1} N at station SMSP on October 9, 2001. While there was variation among sites within each region, there were no significant differences between Tangier Sound and the lower Potomac in each season (Mann-Whitney rank test, $p > 0.05$).

4.4.1.3 Dissolved Phosphorus Concentrations (DIP)

Overall, DIP concentrations exhibited a modest increase from the spring to the summer, and remained stable through the fall sampling (Figure 4-5). During the spring season, dissolved inorganic phosphorus (DIP) concentrations were uniformly low (Figure 4-5.a) and well below the 0.32 μmol l^{-1} P mesohaline habitat limit established by the USEPA (2000). No significant difference was found between sites in Tangier Sound and the lower Potomac, (t-test, $p > 0.05$) during the spring. Insufficient data were available from SV09 in the spring to be included in a statistical analysis. During the summer season, mean values and variance among sites increased,

however no statistical difference was found between regions (Mann-Whitney rank test, $p > 0.05$). In the fall season, only station SV09 at CBL had values higher than the other stations

4.4.1.4 Water Column Light attenuation

In all seasons, water column light attenuation coefficient (Kd) values varied substantially among stations within each region (Figure 4-6). The ranking of stations remained fairly consistent across all seasons. Some stations always exceeded the 1.5 m^{-1} mesohaline habitat Kd limit established by the USEPA (2000), while others remained below that limit. Mean Kd values (all stations) did not vary much with season (spring 1.43 m^{-1}, summer 1.53 m^{-1}, fall 1.48 m^{-1}), variation among stations increased from the spring through the fall. During the fall, individual station means ranged from a minimum of 0.63 m^{-1} (station SV09), to a maximum of 2.34 m^{-1} (station PRJS). No significant differences in Kd values were found between regions in any season (Mann-Whitney rank test, $p > 0.05$).

4.4.1.5 Water Column Total Suspended Solids

The temporal and spatial patterns of total suspended solids (TSS) concentrations were more complicated than many of the other parameters measured (Figure 4-7). The relative ranking of stations within a season did not remain the same over all seasons. While TSS concentrations at some stations were consistently high (PRSP), and others consistently low (SV09), several stations, such as PRBD and SMSP, experienced large seasonal shifts in mean values. This temporal variability was likely the result of short-term re-suspension of sediments at these stations. Despite this variation, overall TSS concentrations were lowest in the spring (21.9 mg l^{-1}) and highest in the fall (28.0 mg l^{-1}). In all seasons, there was no significant difference in TSS concentrations between Tangier Sound and the lower Potomac (Mann-Whitney rank test, $p > 0.05$). Only stations SV09 and PRPP had mean TSS concentrations below the 15 mg l^{-1} mesohaline habitat limit (USEPA, 2000).

4.4.1.6 Water Column Chlorophyll-*a*

Variation in water column chlorophyll-a concentration among stations was highest in the spring (Figure 4-8.a) when the lowest mean concentration (4.22 $\mu g\ l^{-1}$) was found at station SIBC and the highest (14.55 $\mu g\ l^{-1}$) at station PRJS. Both variation among stations and mean values within region were lowest the fall. During the fall, the lowest mean concentration (5.08 $\mu g\ l^{-1}$) was found at station SV09, while the highest (11.46 $\mu g\ l^{-1}$) was found at station PRJS. No significant difference was found between regions in any season (Mann-Whitney rank test $p > 0.05$). In

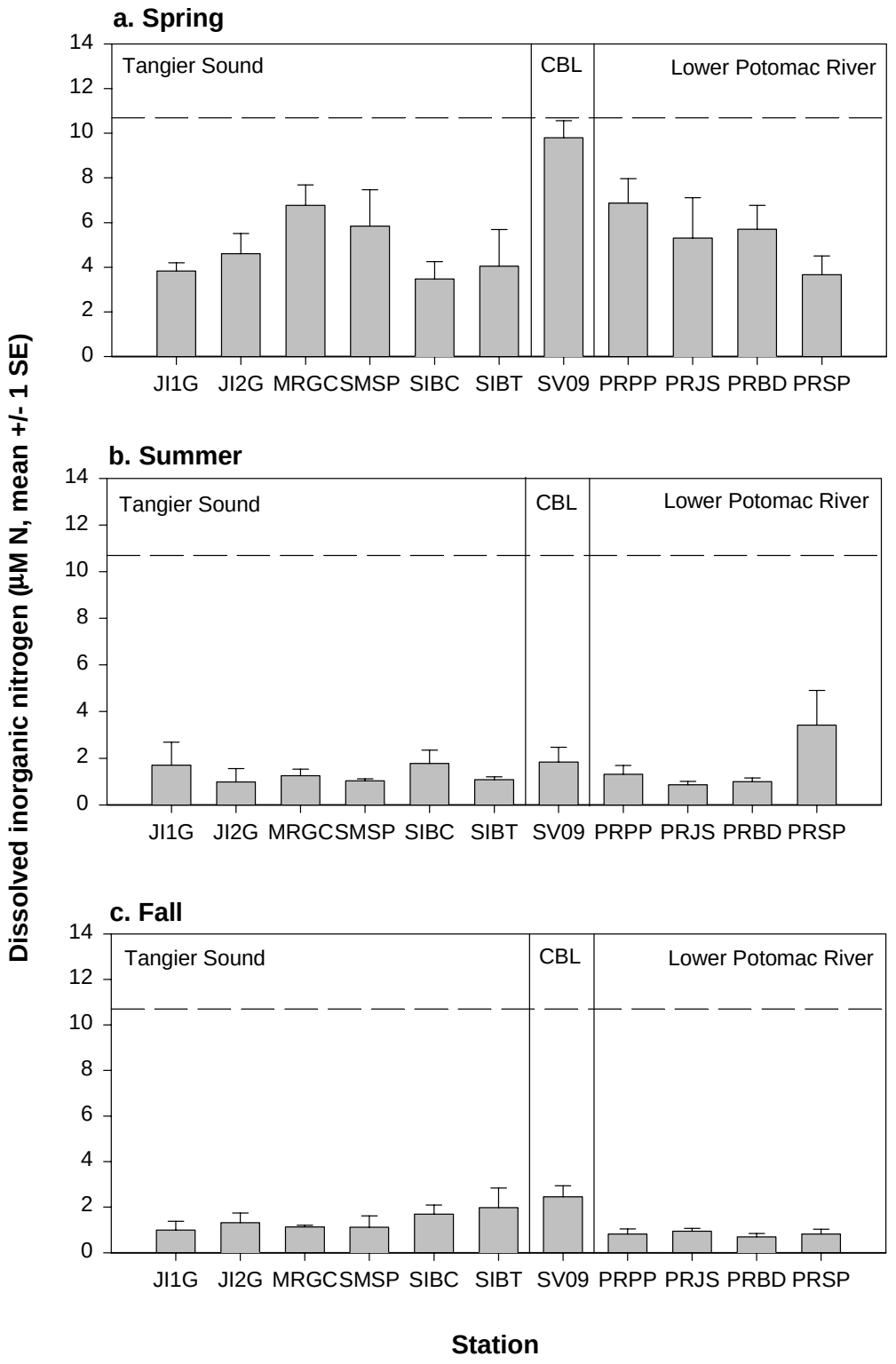

Figure 4-4. Mean (+/- 1SE) dissolved inorganic nitrogen (DIN) concentrations for (a) spring, (b) summer and (c) fall in Tangier Sound, Patuxent River (CBL), and the Lower Potomac River, 2001. Dashed lines represent minimum Tier II mesohaline SAV habitat requirement (USEPA, 2000).

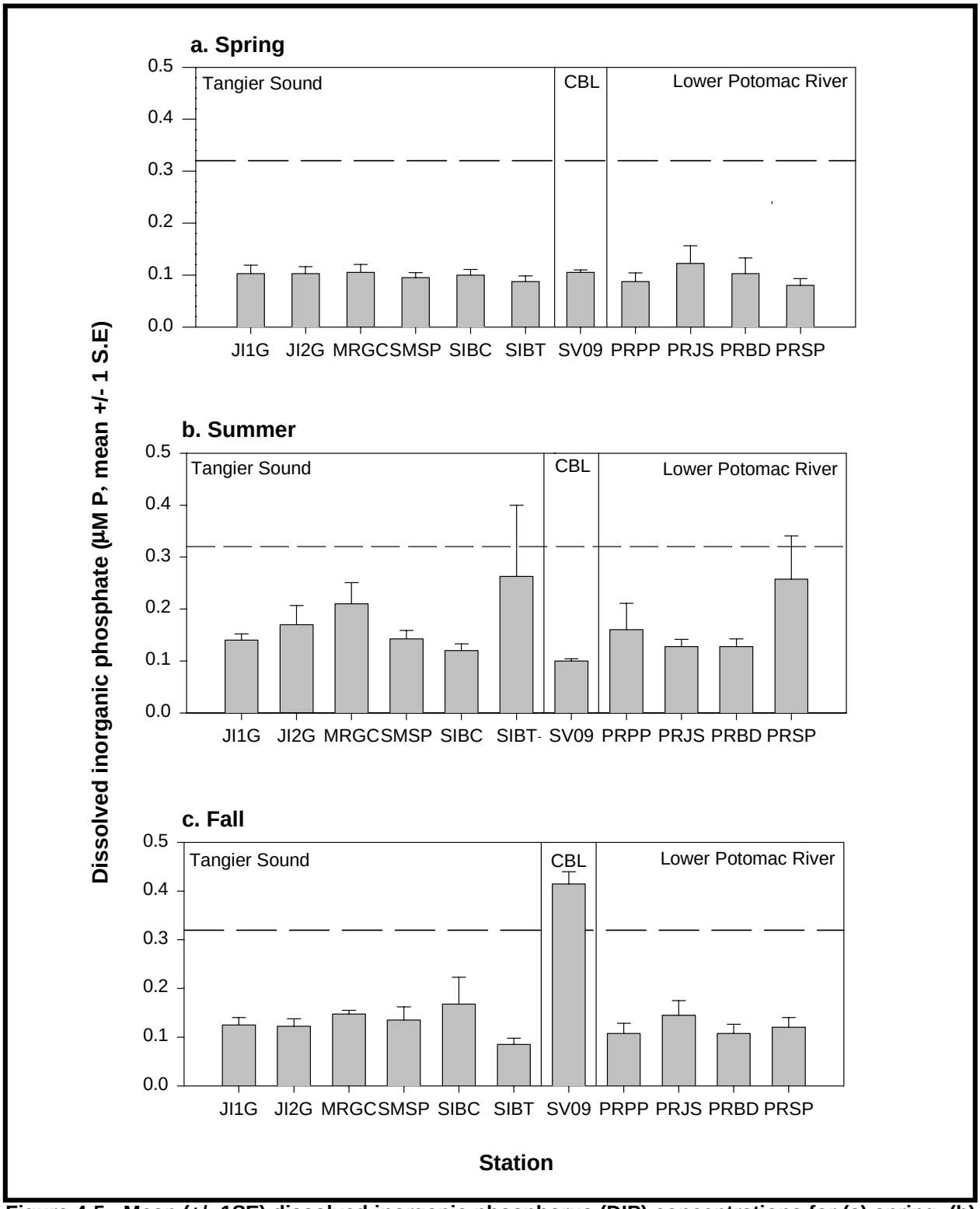

Figure 4-5. Mean (+/- 1SE) dissolved inorganic phosphorus (DIP) concentrations for (a) spring, (b) summer, and (c) fall for Tangier Sound, the Patuxent River (CBL), and the Lower Potomac River, 2001. Dashed line represents upper limit Tier II mesohaline SAV habitat requirement (USEPA, 2000).

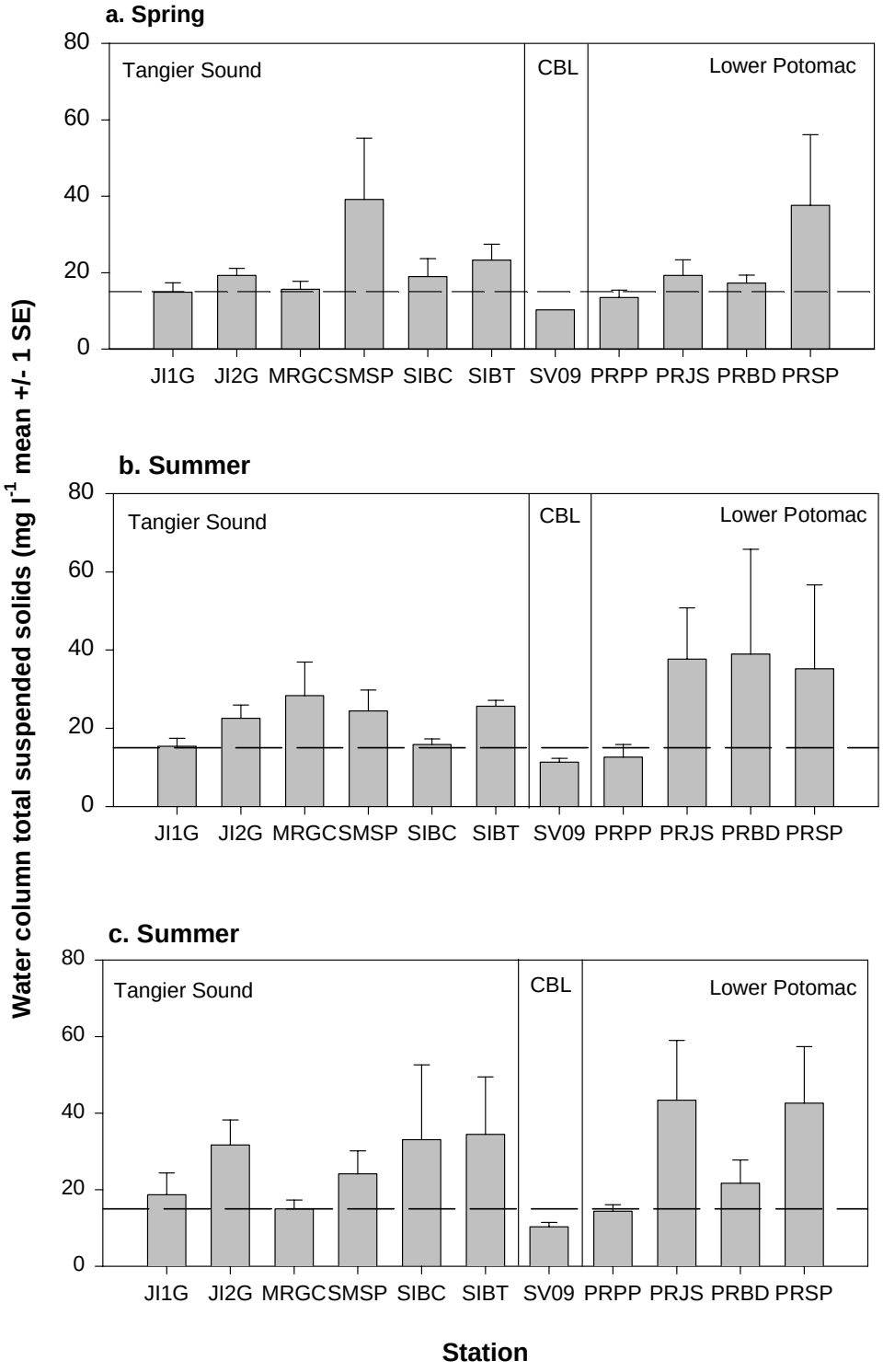

Figure 4-6. Mean (+/- 1SE) water column light attenuation coefficient (Kd) for (a) spring, (b) summer, and (c) fall for Tangier Sound, the Patuxent River (CBL), and the lower Potomac River, 2001. Dashed line represents the upper limit Tier II mesohaline SAV habitat requirement (USEPA, 2000).

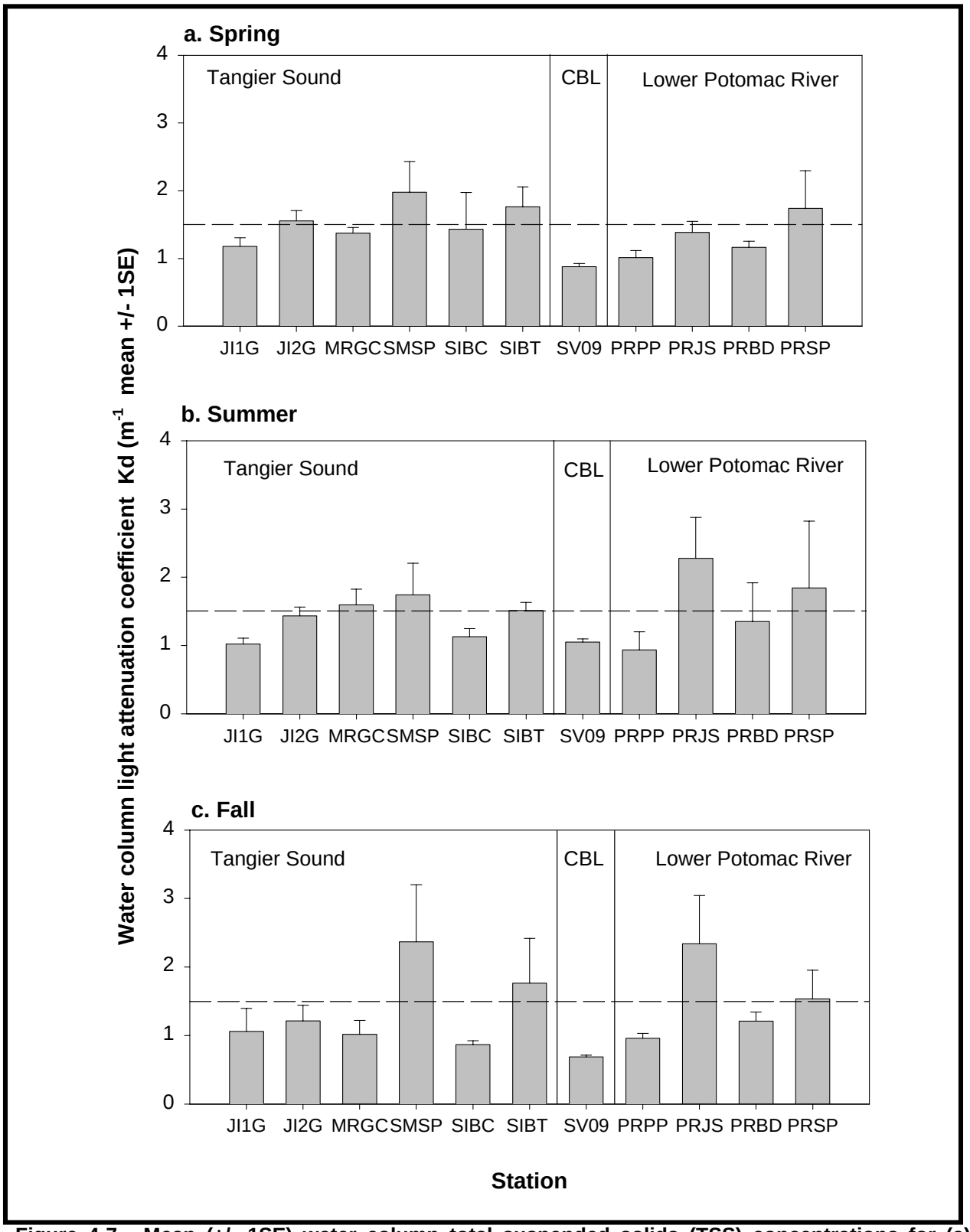

Figure 4-7. Mean (+/- 1SE) water column total suspended solids (TSS) concentrations for (a) spring, (b) summer, and (c) fall for Tangier Sound, the Patuxent River (CBL) and the lower Potomac River, 2001. Dashed line represents the upper limit Tier II mesohaline SAV habitat requirement (USEPA, 2000).

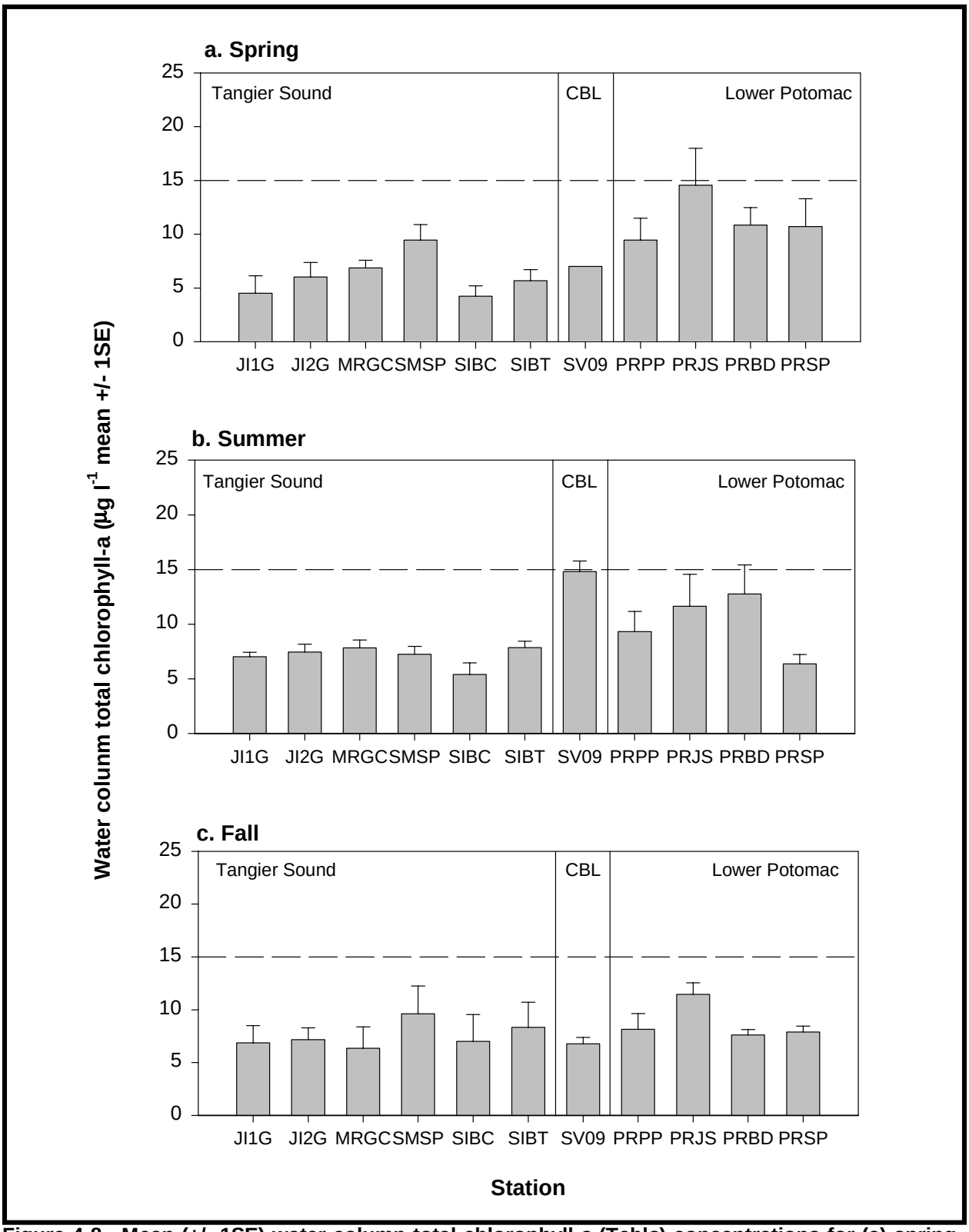

Figure 4-8. Mean (+/- 1SE) water column total chlorophyll-*a* (Tchl*a*) concentrations for (a) spring, (b) summer, and (c) fall for Tangier Sound, the Patuxent River (CBL) and the lower Potomac River, 2001. Dashed line represents the upper limit Tier II mesohaline SAV habitat requirement (USEPA, 2000).

addition, mean concentrations at all stations remained below the mesohaline habitat limit of 15 $\mu g\ l^{-1}$ (USEPA, 2000) throughout the year.

4.4.2 Results of Epiphyte Growth Study

4.4.2.1 Epiphyte Dry Mass

During the spring season epiphyte dry mass accumulation rates at nine of ten stations were uniformly low compared to rates observed during either the summer or fall (Figure 4-9.a). The one exception to this pattern was an unusually high mean fouling rate at station MRGC (2.47 mg cm^{-2} $week^{-1}$). The second highest fouling rate (0.39 mg cm^{-2} $week^{-1}$) found at station PRSP, was dramatically lower and similar to rates found at the other stations. During the summer season, however, there was considerable variation in fouling rates among all sites in both Tangier Sound and the lower Potomac (Figure 4-9.b). In contrast, epiphyte accumulation at MRGC was among the lowest observed. During the fall season, the magnitude of variation in fouling rates among stations was less than during the summer. Once again, the relative ranking of fouling rates among stations was very different than the previous seasons. For example, fouling at station PRSP was among the lowest observed instead of the highest as in previous seasons. No significant difference was found in dry mass fouling rates between regions in the spring or summer seasons (Mann-Whitney rank test, $p > 0.05$). During the fall sampling, epiphyte accumulation rates in Tangier Sound were significantly higher than the lower Potomac (Mann-Whitney rank test, $p < 0.05$).

4.4.2.2 Epiphyte Chlorophyll-*a*

The temporal and spatial patterns observed in epiphyte chlorophyll-*a* accumulation rates were similar to those for epiphyte dry mass. During the spring season, the highest fouling rate was found at station MRGC (Figure 4-10.a). Variation among sites increased during the summer season with the highest epiphyte chlorophyll-*a* fouling rate measured at station SV09 (1.95 μg cm^{-2} $week^{-1}$). Overall, epiphyte chlorophyll-*a* accumulation rates during the fall season (0.60 μg cm^{-2} $week^{-1}$) declined slightly compared to the summer (0.65 μg cm^{-2} $week^{-1}$), but remained fairly high. Fouling at station SV09 remained very high and was consistent with measurements collected in previous years.

4.4.2.3 Epiphyte Light Attenuation (PLW and PLL)

As with epiphyte accumulation rates, the percent surface light reaching the leaf surface (PLL) among the sites measured was temporally and spatially variable (Figure 4-11). During the

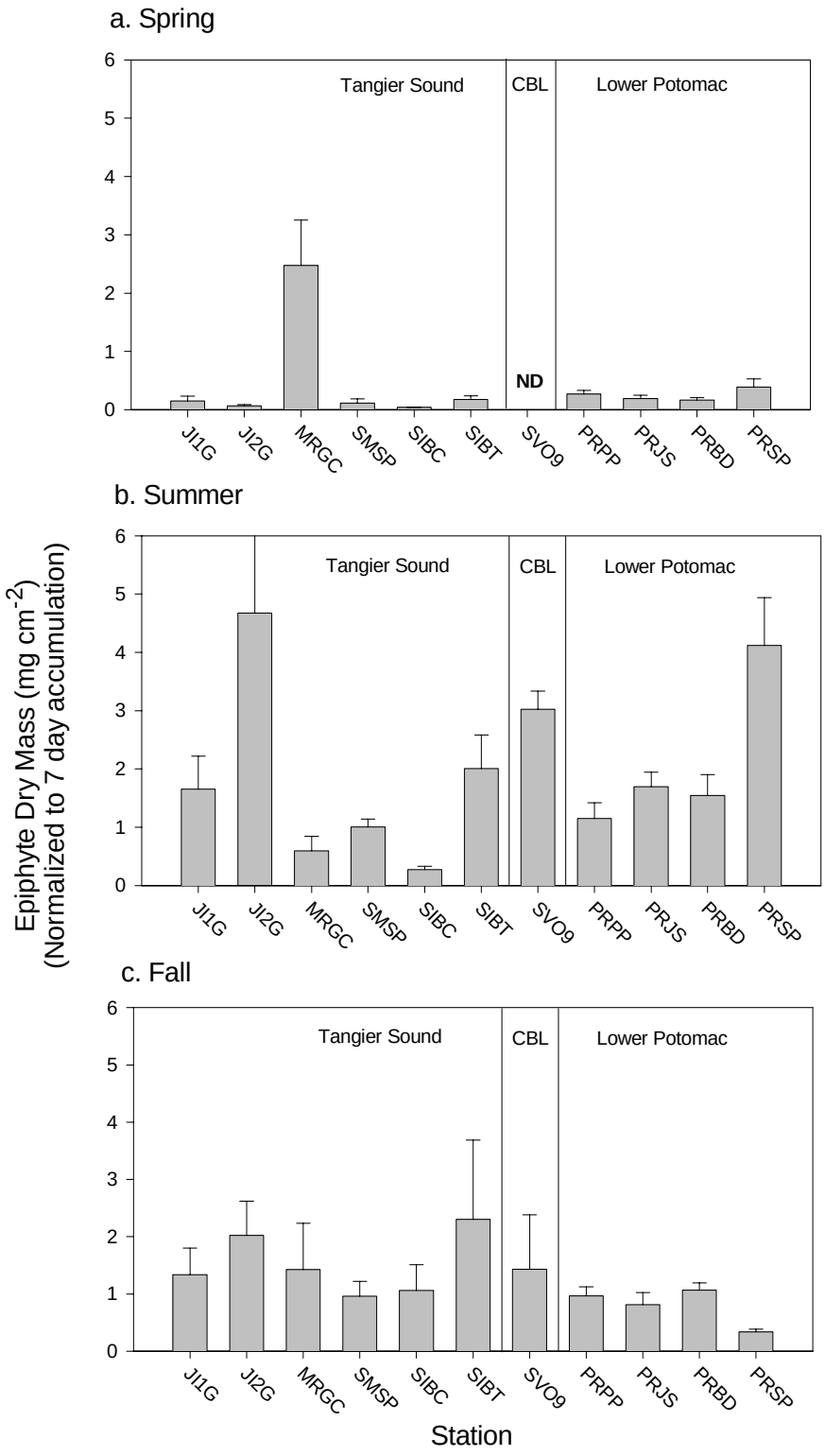

Figure 4-9. Mean (+/- 1SE) epiphyte dry mass accumulation on Mylar® strips deployed for exposures of 6-8 days in a) spring, b) summer and c) fall along the Patuxent River and Tangier Sound, 2000.

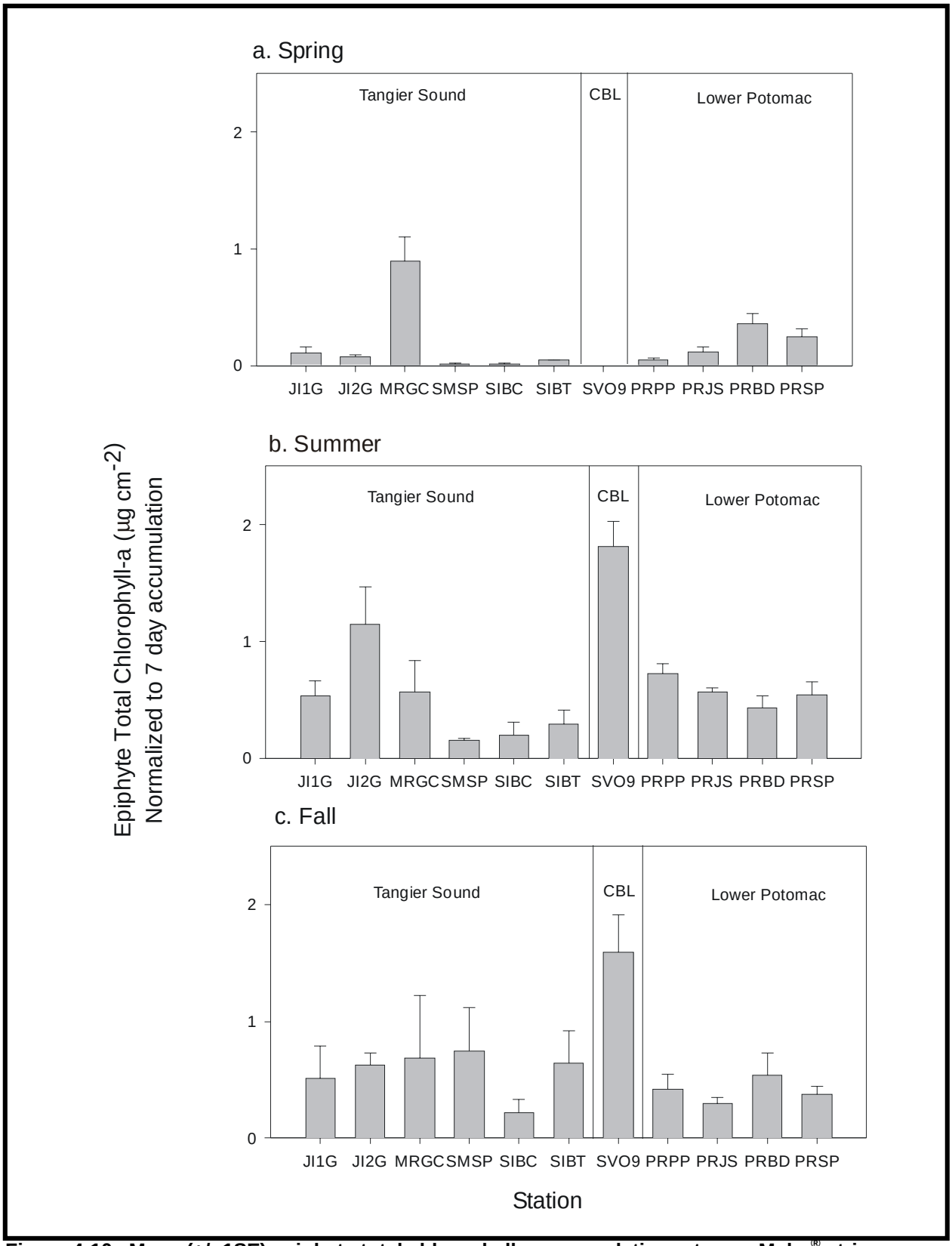

Figure 4-10. Mean (+/- 1SE) epiphyte total chlorophyll-*a* accumulation rates on Mylar® strips deployed for *in-situ* exposures of 6-8 days in a) spring, b) summer and c) fall in Tangier Sound, the Patuxent River, and the lower Potomac River, 2001.

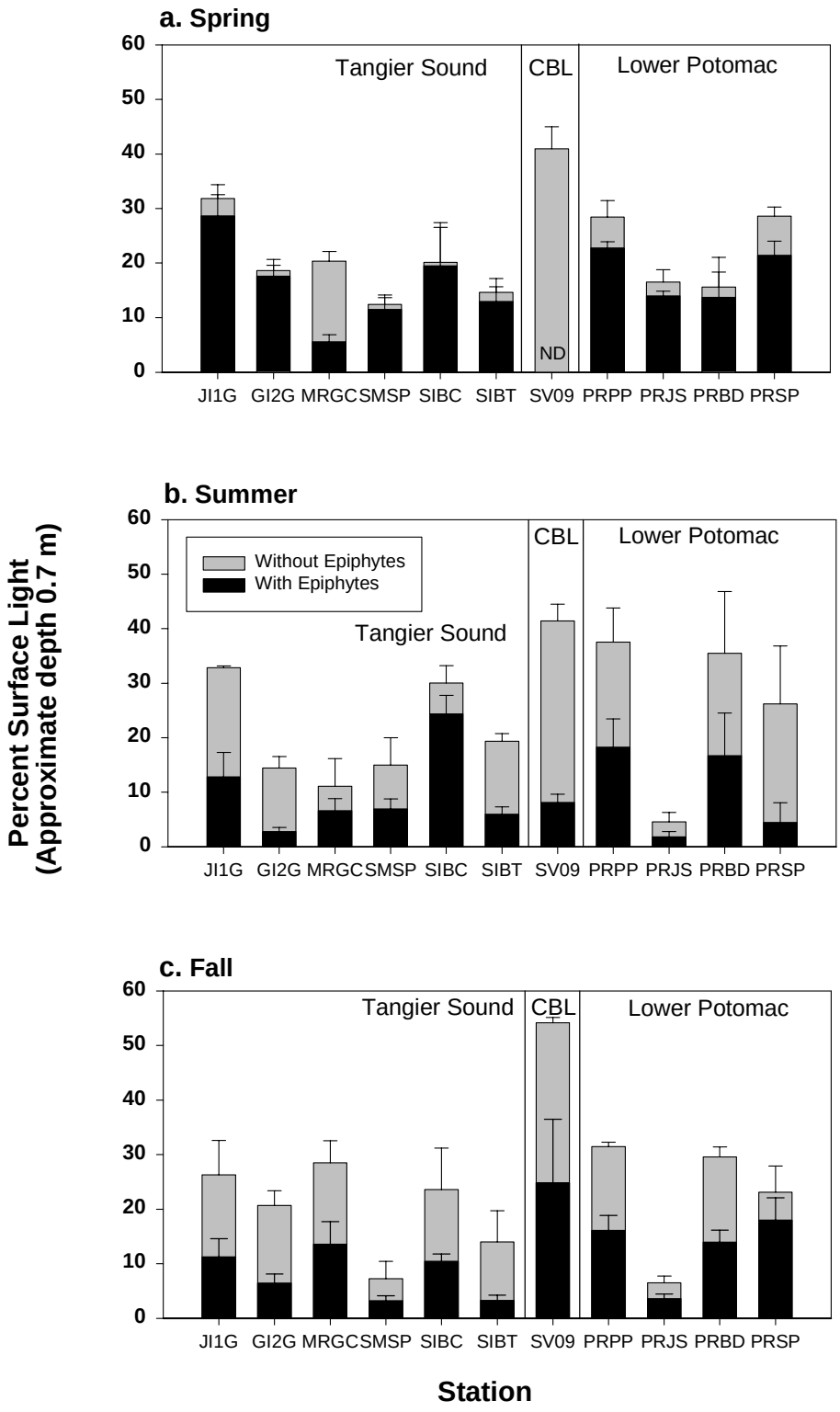

Figure 4-11. Mean light available through the water column (PLW) at approximately 0.7m depth and light at the leaf surface (PLL) calculated by dry mass accumulation on Mylar® strips in a) spring, b) summer and c) fall in Tangier Sound, the Patuxent River, and the lower Potomac River, 2001.

spring, light attenuation by epiphytes was a minor component of total light attenuation to the leaf surface at 9 of 10 stations more than 10% of surface light reached the leaf surface based upon a week-long accumulation of epiphytic material. During that same period, some stations had light at the leaf surface (PLL) greater than 20% of surface irradiance. During the summer and fall, both epiphtye fouling rates, and water column light availability through the water (PLW), were highly variable among stations in all regions. Percent light at the leaf surface (PLL) ranged from a minimum of 2 % at station PRJS up to a maximum of 24% at station SIBC.

4.4 Discussion and Conclusions

Of the five water quality parameters (DIN, DIP, Kd, TSS, Chl-*a*) considered most important for the health and survival of SAV, there were no differences, on a regional basis, between Tangier Sound and the lower Potomac River sites. However, there were differences among each of the parameters, to the extent they exceeded or remained below the mesohaline SAV habitat limits established by the USEPA (2000). With few exceptions, concentrations of both dissolved inorganic phosphorus (DIP) and dissolved inorganic nitrogen (DIN) were below the mesohaline habitat limits established by the USEPA. The other parameters (TSS, Chl-*a*, and Kd), because of their correlation with each other, displayed very similar temporal and spatial patterns. On a regional basis, their overall means were very close to the mesohaline habitat limits. However, as expected, there were important and measurable differences among individual stations within each region for each of the parameters. Values at some stations never meet the habitat limit, while others always exceeded it. Another important difference among sites was the presence or absence of healthy SAV. For example, five of the eleven stations were located within healthy SAV beds. The other stations were completely barren or had very sparse or patchy SAV. These site differences illustrate the variation in habitat conditions that was observed for locations in close proximity to one another.

Data collected as part of the epiphyte growth study allowed a more detailed examination of light availability to SAV and the presence or absence of SAV. By sorting light availability to the leaf surface (PLL) by the presence or absence of SAV and plotting those values against water column light availability (PLW, Figure 4-12) we can see that below a PLW of 12%, healthy populations of SAV were not found. This is consistent some with some literature values that suggest a minimum of 11% surface light is needed for eelgrass survival (Short *et al.*, 1995). However, at higher PLW light levels, no difference was found between PLL and the presence or absence of SAV (ANCOVA, $P > 0.05$). This result indicates that measurements of epiphyte fouling rates alone are not good indicators of SAV survival at specific locations. There may be several reasons for the lack of correlation between the presence of SAV and acceptable water clarity (at light levels above 12%). First, those stations without SAV but with adequate water quality conditions may in many cases be propagule limited, and therefore would sustain SAV if propagules were present. This is likely the case for stations SV09 and PRPP, where eelgrass transplant test plots have shown high promise for success. In both cases, native populations of eelgrass are many kilometers away, thus limiting natural recruitment to the area. In other cases, such as station PRBD, sediment instability may also result in lack of recruitment for certain

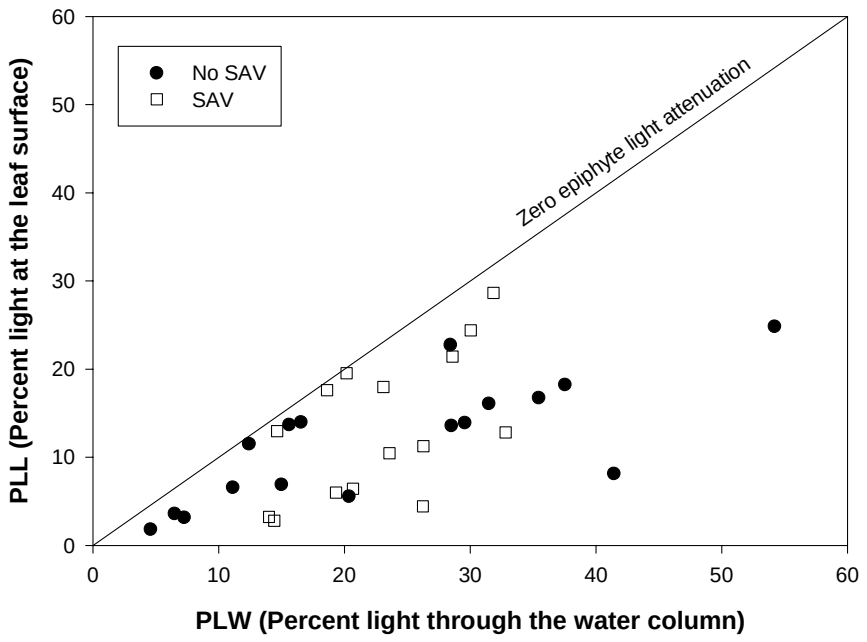

Figure 4-12. Comparison of PLL vs. PLW between stations located in Tangier Sound, the lower Potomac, and the Patuxent River, with healthy SAV populations and those without in 2001. Diagonal line represents the 1:1 or zero epiphyte attenuation limit.

species such as widgeon grass, which is found in nearby areas. At the other end of the spectrum, SAV was found at locations such as station SIBT where estimates of light to the leaf surface (PLL) were among the lowest recorded (6% summer, 3% fall). There are several potential reasons for this apparent departure from the literature values (11% - 37%; USEPA, 2000). First, these estimates of PLL are based upon a week-long exposure to epiphytic fouling. However, each SAV shoot continually produces new leaf tissue clear of heavy fouling and able to take full advantage of the light through the water column. Therefore, the rate at which new leaves are produced, along with the epiphyte fouling rate will determine how much light the entire SAV shoot will receive. Second, these estimates of PLL represent the most extreme set of values observed over the course of a year. However, species such as eelgrass respond favorably and accumulate resources earlier in the season when water quality and temperature is more favorable to growth (Moore *et al.*, 1997). In addition, other studies (Stankelis *et al.*, 1999) have shown that epiphyte accumulation rates are very minimal during the early spring (March – April) and do not contribute significantly to light attenuation. If that scenario is valid, then even high epiphytic accumulation rates during the late spring, summer and fall, do not alone determine whether SAV can survive at a particular location. The data collected by the EPC is currently being analyzed to better understand the dynamics of epiphyte fouling and to use this understanding to help make more informed management decisions regarding nutrient loading and SAV survival.

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5. HIGH RESOLUTION MAPPING OF SURFACE WATERS

B.W. Bean, R.M. Stankelis, J.M. Lawrence and W.R. Boynton

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5.1 Introduction

An evaluation of surface water patterns in water quality parameters was made using the DATAFLOW V mapping system in the Magothy and Severn Rivers, and Tangier Sound in 2001. The DATAFLOW mapping system, when deployed from a small research vessel, allowed for the estimation of several water quality parameters with high spatial resolution in both shallow (approx. 1.0 m) and deeper waters. Shallow littoral zones serve as important habitats for many aquatic organisms and SAV communities. However, traditional water quality monitoring has been conducted almost exclusively in deeper channel waters, and conditions in these areas do not adequately represent shallow zones. Thus, it is important to be able to collect water quality data in both habitats to determine the extent of any gradients in water quality parameters. The goal for 2001 was to perform a systematic monitoring program in three areas of interest, including Tangier Sound, and the Magothy and Severn Rivers. At each site a DATAFLOW mapping cruise was conducted, and traditional water column samples were collected at a series of fixed calibration stations. The DATAFLOW cruise track covers as much area as possible, in both shallow and deeper waters. In addition, light data and dissolved nutrients were measured at calibration stations in order to better define temporal patterns throughout the estuary. The data that were collected substantially improves characterization of water quality conditions in the near shore habitats as well as system-wide water quality.

5.2 Methods, Locations and Sampling Frequency

5.2.1 DATAFLOW V

DATAFLOW V is a compact, self-contained surface water quality mapping system, suitable for use in a small boat operating at speeds of up to 20 knots. A schematic of this system is shown in Figure 5-1. Surface water (0.6m deep) is collected through a pipe ("ram") deployed from the transom of the vessel. Assisted by small bilge pumps, water is then passed through a bubble trapping device to ensure that no air bubbles are conveyed to the sensors. After being debubbled, water passes through a flow meter and finally to an array of water quality sensors which record the water quality variables, time, and geographic position. The total system volume is approximately 3.8 liters.

The heart of the system is a YSI 6600 data sonde and 650 high memory logging display. The YSI sonde is configured to measure dissolved oxygen, temperature, conductivity, salinity, turbidity, and fluorescence. GPS position data is transmitted by a Garmin e-Trex model unit to the 650 logging unit through a NMEA 0183 version 2.0 data format. Positioning errors are typically less than 15 m. Depth data are collected with an auxiliary Garmin 168 global positioning system with a built-in depth sounder. The Garmin 168 GPS transmits NMEA data to a Westcor RDT 3200 portable computer using Procomm plus communication software. The data is transmitted in a NMEA 0183 version 2.3 data format. Data files are merged by time stamp at a later date. Although the flow rate does not affect any of the sensor readings, decreased flow is an indication of either a partial blockage or an interruption of water flow to the instrument and affects the water turnover rate of the system. An inline flow meter wired to a low-flow alarm alerts the operator of potential problems as they occur. Currently the low-flow alarm is set to 4.5 l min^{-1}. Twin "Rule Pro Series" bilge pumps provide approximately 8-12 liters/min. of flow to the system. The system can operate on a single pump.

5.2.2 Sampling locations and frequency

Dataflow cruises were performed on a bi-weekly basis on both the Magothy and Severn Rivers, for a total of fourteen cruises during 2001. Six cruises were completed in Tangier Sound, two each in the spring, summer, and fall (Table 5-1). Cruise tracks were chosen to provide a reasonable coverage of each water body while sampling both near-shore and offshore waters. A sample cruise track is shown for each region in Figure 5-2. In the Magothy and Severn Rivers, a total of 8 calibration stations were made during each cruise. In Tangier Sound, a total of 18 calibration stations were made per cruise. The selection of calibration station locations in each region was made to sample the greatest possible range of water quality conditions found during each cruise and to sample a broad spatial area. Every effort was made to maintain the same location of calibration stations between cruises. The location of several calibration stations were also chosen to correspond to Chesapeake Bay Program water quality monitoring stations within each region, and these stations were sampled during each cruise. The coordinates for those stations are listed in Table 5-2.

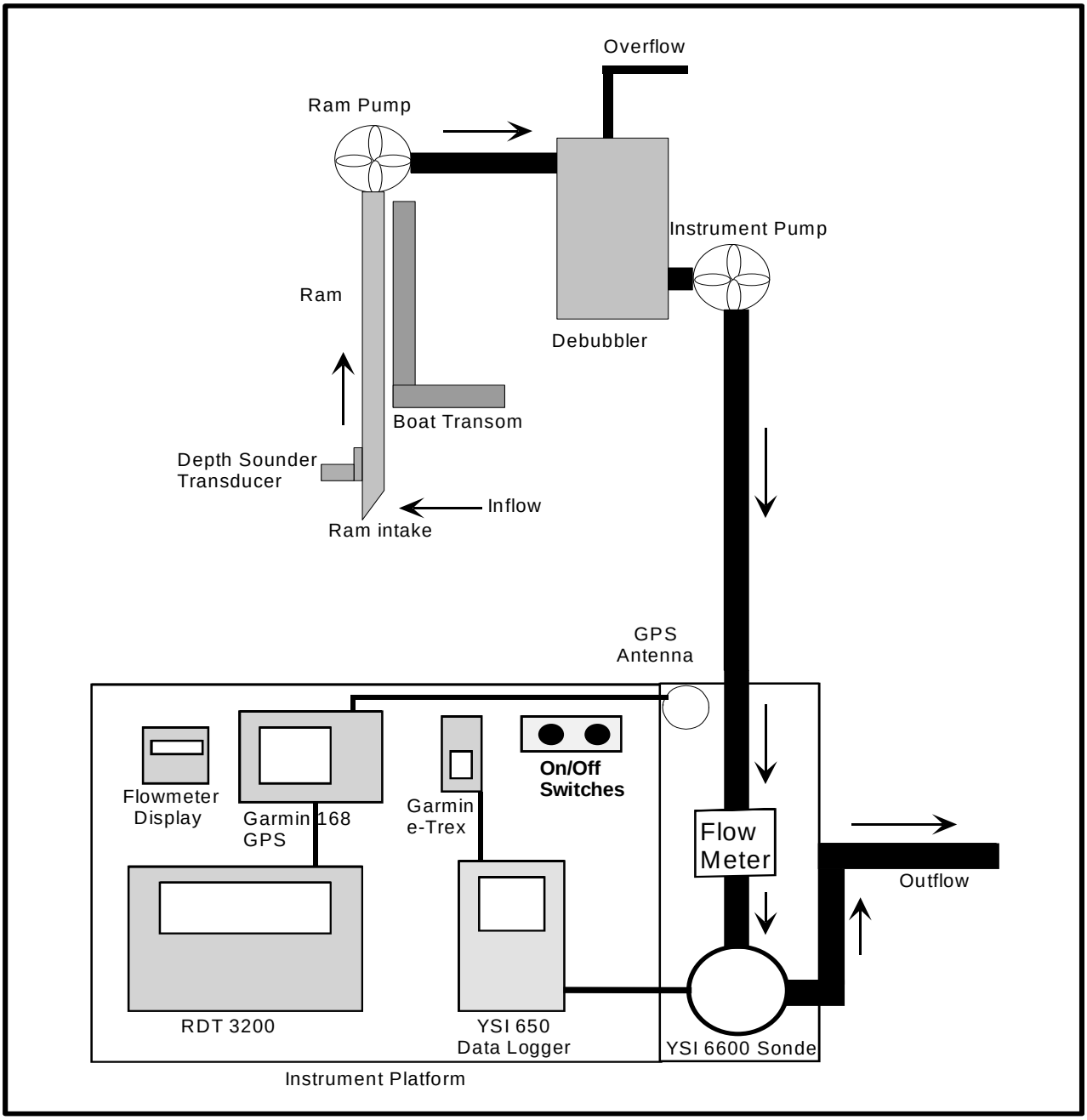

Figure 5-1. Schematic diagram of DATAFLOW V illustrating the path of water through the instrument.

Seawater is picked up behind the transom of the research vessel through the "ram." A centrifugal pump mounted on the ram ("ram pump") pulls up the seawater and pushes it into the debubbler. The debubbler fills and overflows. Air and excess water are pushed out of the debubbler through the overflow hose. A centrifugal pump ("instrument pump") mounted at the bottom of the debubbler draws water out of the pump and pushes it to the sensors. The water first runs through a flow meter. The flow meter is wired to a horn that sounds if the flow rate falls below 4.5 l min^{-1}. If flow is interrupted during sampling, the horn sounds informing operators that a problem exists. The water exits the flow meter and enters the YSI flow-through chamber. The water runs across the sensor probes and exits the flow-through chamber before being discharged overboard. The displays for the YSI 650 data logger, Garmin 168 GPS, Garmin e-Trex GPS, flow meter display, and RDT 3200 are located on the instrument platform.

Table 5-1. DATAFLOW cruise dates in 2001. Tangier Sound cruises typically required three days of mapping, thus dates listed below represent the initial day of each cruise. Magothy River and Severn River cruises were completed in a single day.

Region	Spring	Summer	Fall
Tangier Sound	5/22, 6/4	7/16, 7/30	9/24, 10/8
Magothy River	4/18, 5/3, 5/16, 5/31, 6/13, 6/27	7/11, 7/25, 8/8, 8/22	9/5, 9/18, 10/2, 10/18
Severn River	4/19, 5/4, 5/21, 5/31, 6/14, 6/28	7/12, 7/26, 8/9, 8/24	9/6, 9/19, 10/3, 10/19

Table 5-2. Location of DATAFLOW calibration stations (stations coincident with Chesapeake Bay Program water quality monitoring stations noted with *).

Region	Station	Latitude (deg mins)	Longitude (deg mins)
Tangier Sound	EE3.0*	38° 16.870’	76° 00.855’
	EE3.1*	38° 11.730’	75° 58.427’
	EE3.2*	37° 58.825’	75° 55.395’
	ET6.2*	38° 20.513’	75° 53.300’
	ET8.1*	38° 08.234’	75° 48.208’
	ET9.1*	38° 03.380’	75° 48.289’
	MRGC	38° 08.830’	75° 50.344’
	JI1G	38° 01.642’	75° 50.506’
	JI2G	37° 58.194’	75° 52.493’
	SIBT	37° 58.127’	75° 59.551’
	SIBC	38° 02.025’	76° 00.650’
	SMSP	38° 04.321’	76° 01.172’
	TM03	38° 05.071’	75° 53.874’
	TM04	38° 06.694’	76° 00.867’
	TM06	38° 07.015’	76° 04.030’
	TM08	38° 20.013’	76° 00.228’
	TM10	38° 11.980’	75° 53.001’
	TM11	38° 14.932’	75° 50.427’

Region	Station	Latitude (deg mins)	Longitude (deg mins)
Magothy River	MG01	39°03.482’	76°26.105’
	MG02	39°03.189’	76°26.934’
	MG03	39°04.037’	76°28.661’
	MG04* (WT6.1)	39°04.588’	76°30.211’
	MG05	39°05.194’	76°31.495’
	MG06	39°05.189’	76°28.870’
	MG07	39°05.321’	76°26.048’
	MG08	39°04.683’	76°27.349’

Region	Station	Latitude (deg mins)	Longitude (deg mins)
Severn River	SR01	38°58.088’	76°27.215’
	SR02	39°00.162’	76°29.433’
	SR03* (WT7.1)	39°00.438’	76°30.334’
	SR04	39°00.438’	76°32.148’
	SR05	39°02.295’	76°32.995’
	SR06	39°03.777’	76°34.211’
	SR07	39°02.253’	76°34.151’
	SR08	39°01.232’	76°31.593’

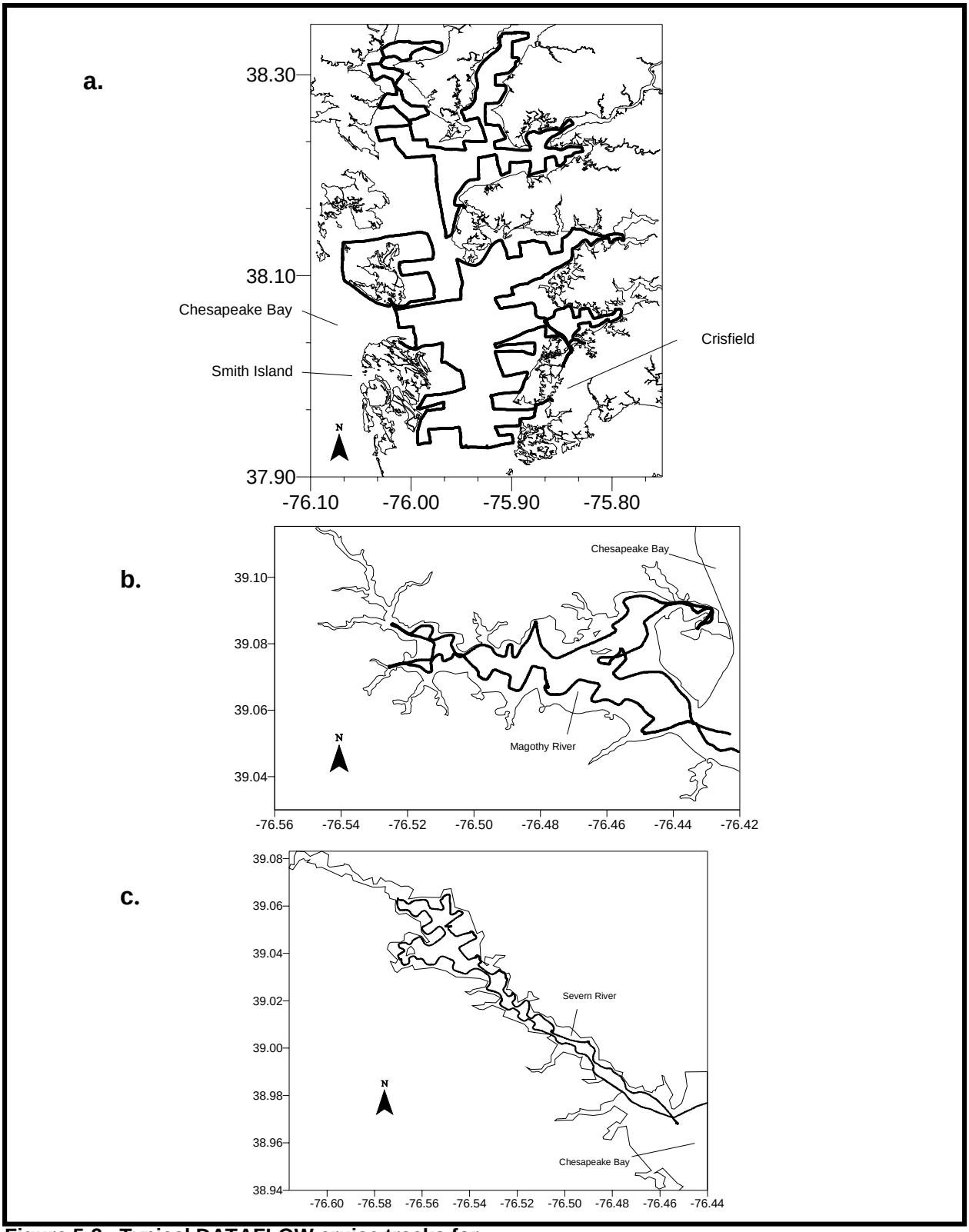

Figure 5-2. Typical DATAFLOW cruise tracks for
a) Tangier Sound, September 24, 2001
b) Severn River, August 9, 2001, and
c) Magothy River, August 22, 2001.

5.2.3 Calibration Stations

At each calibration station, a series of measurements and water samples were taken. LICOR (water column light availability) measurements and Secchi depths were measured, and were later regressed against turbidity data gathered by the YSI Sonde. Whole water samples were taken, from which were determined both total and active chlorophyll-a values, as well as Total Suspended Solids (TSS) and Total Volatile Solids (TVS). These chlorophyll-a values were later compared to chlorophyll-a values obtained from the YSI data Sonde. In addition, sets of water samples were taken to determine concentrations of dissolved nutrients. Dissolved inorganic nitrogen (DIN; summation of NH_4^+, NO_2^-, NO_3^-) and dissolved inorganic phosphorus (DIP) were measured at each site. A detailed explanation of all field and laboratory procedures is given in Rohland et al. (2000).

5.2.4 **Contour Maps**

Contour maps were created to visualize the spatial patterns of water quality. "Surfer" contouring software (Golden Software) was used to create the contour maps presented in this report. Interpolation using the nearest observations was performed using the default Kriging procedures available in the software. Contour maps created from this data can be created with a number of different interpolation methods. The maps provided simply illustrate one method used to view the patterns contained within the data set. Other interpolation methods may generate slightly different results.

5.3 Results

5.3.1 Dissolved Nutrient Data

Table 5-3 contains maximum, minimum, and median values of DIN and DIP at each station throughout the season from the Magothy and Severn rivers. DIN ranged from 0.14 μmol to 69.3 μmol, and DIP ranged from 0.02 μmol to 1.66 μmol. Both rivers had similar median DIN values, which were between 1.0 - 2.0 μmol, and median DIP values between 0.10 - 0.20 μmol. The grand median value for DIN from the Magothy was 2.12, while the grand median DIN from the Severn was 1.52. A rank sum test was performed, and results indicated no statistical difference between sites ($p = 0.382$). For DIP, the grand median value from the Magothy was 0.149, and 0.150 from the Severn. An ANOVA test was applied, and results indicated no statistical difference between the two rivers ($p = 0.943$). However, while the Magothy River had similar dissolved nutrient levels throughout the river, the Severn River had higher levels of both DIN and DIP at the mouth of the river.

Table 5-3. Dissolved Nutrient Concentrations from the Magothy and Severn Rivers, April-October 2001.

Magothy River		MG01	MG02	MG03	MG04	MG05	MG06	MG07	MG08
Dissolved	Max	66.3	60.7	58.2	57.6	65.5	61.2	66.9	69.3
Inorganic Nitrogen	Min	0.54	0.77	0.72	0.98	0.73	0.31	0.54	0.81
(μmol N)	Median	2.44	1.98	1.43	2.25	7.52	1.37	2.60	1.15
Dissolved	Max	1.19	0.45	0.87	0.72	1.37	1.01	0.87	0.59
Inorganic Phosphorus	Min	0.02	0.03	0.02	0.02	0.03	0.02	0.04	0.05
(μmol P)	Median	0.21	0.16	0.13	0.11	0.16	0.19	0.10	0.13

Severn River		SR01	SR02	SR03	SR04	SR05	SR06	SR07	SR08
Dissolved	Max	62.7	39.8	35.3	31.5	30.6	29.5	29.3	33.4
Inorganic Nitrogen	Min	0.14	0.64	0.51	0.51	0.52	0.79	0.73	0.61
(μmol N)	Median	1.82	1.45	1.58	1.25	2.65	2.07	1.39	0.99
Dissolved	Max	1.66	0.65	0.51	0.28	0.66	0.77	0.58	0.44
Inorganic Phosphorus	Min	0.08	0.05	0.04	0.04	0.04	0.03	0.05	0.04
(μmol P)	Median	0.19	0.20	0.14	0.12	0.13	0.15	0.12	0.15

5.3.2 Calibration Data

Chlorophyll-*a* regressions were completed in which data collected from the YSI data sonde was compared to chlorophyll-*a* values determined by Nutrient Analytical Services Lab (NASL) from the water sample collected at the calibration stations. On the Severn River, the regressions were fairly strong, though there was some variance between cruises (r^2 values between 0.69 and 0.99). Because of the variance between cruises, an attempt was made to better define the relationship between the two values. The results for all of the cruises were combined in analysis, resulting in an r^2 of 0.82. When all data were used, the predictability of the results decreases somewhat, when compared to predictability based on data collected from a single cruise (an r^2 of 0.82 versus an r^2 of 0.99). However, the confidence in the results increases, as the full data set contains over 100 observations, versus only eight for a single cruise. The regressions for Tangier Sound were strong as well, and display a significant positive relationship between the YSI values and those calculated from the filtered water sample.

Regression analyses were also performed to examine turbidity relationships, in this case NTU (the turbidity unit of measure for the YSI probe) versus Kd (m^{-1}). These turbidity regressions were generally not as strong than those observed for chlorophyll-*a*. It can be somewhat problematic to compare turbidity with different units of measure. For the Severn River, r^2 values ranged from 0.38 to 0.96, with the regression for the entire season resulting in an r^2 of 0.582. The regression was a bit stronger for Tangier sound, the entire season having an r^2 of 0.83.

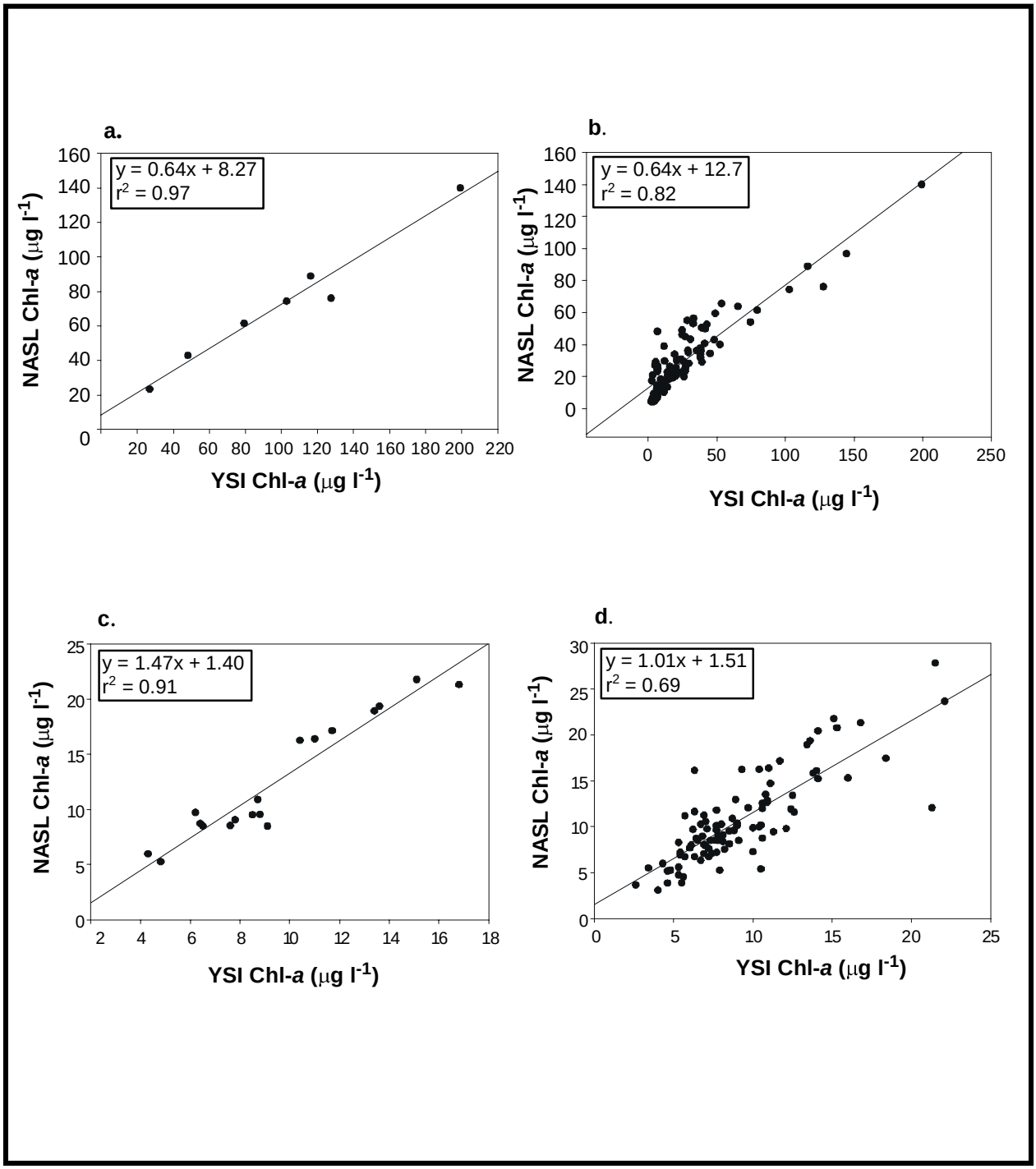

Figure 5-3. Chlorophyll-*a* calibration curves for
a. Severn River, May 21, 2001
b. Severn River 2001 (14 cruises)
c. Tangier Sound July 30, 2001 and
d. Tangier Sound 2001 (6 cruises).

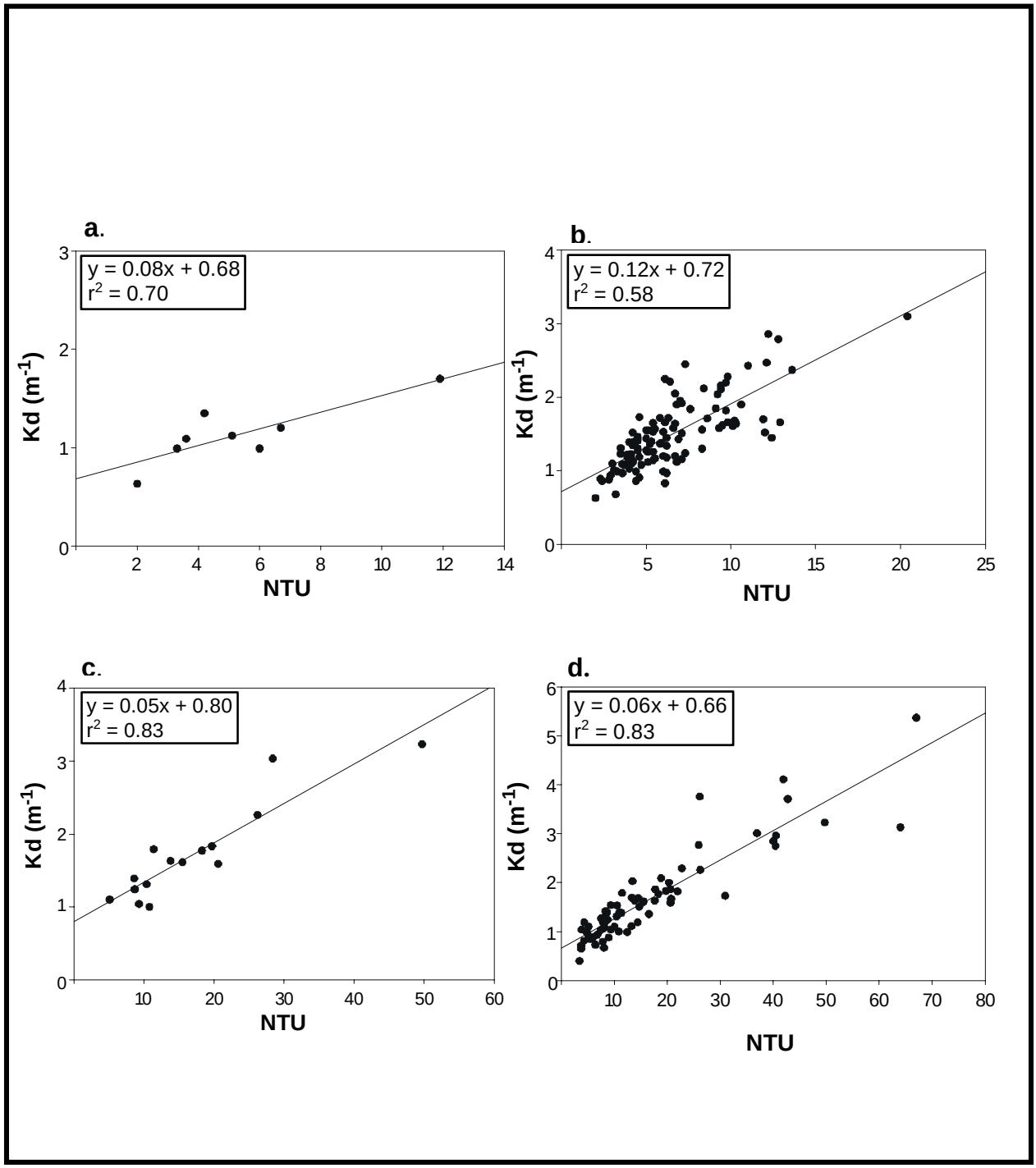

Figure 5-4. Turbidity calibration curves for

a. Severn River, April 19, 2001
b. Severn River 2001 (14 cruises)
c. Tangier Sound May 22, 2001 and
d. Tangier Sound 2001 (6 cruises).

5.4 Discussion

5.4.1 Temporal variations in spatial pattern

Significant differences in spatial patterns were observed between consecutive cruises in 2001. During three consecutive cruises on the Magothy river, 13 June, 27 June, and 11 July, 2001, different patterns of chlorophyll-a distribution were observed (Figure 5-5). On the first date, chlorophyll-a levels appear low across the river, with two localized pockets of chlorophyll-a, one to the north of the river, one close to the mouth (east). Two weeks later, chlorophyll-a concentrations had increased significantly at the mouth of the river, suggesting an effect from the mainstem Chesapeake Bay. Further into the estuary, chlorophyll-a concentrations remained low. During the third cruise, a more complex pattern had emerged, with fragmented pockets of chlorophyll-a distributed throughout the estuary. Spatial patterns continued to vary significantly between cruises throughout the year.

5.4.2 Differences in near-shore and off-shore waters

One potential future analysis is to examine the differences between near-shore shallow water SAV habitats (<2 meter depths) and off-shore waters (>3 meters). Observations for the Magothy river cruise on 27 June, 2001 were sorted by depth. Any suspect data points were excluded from the analysis. Mean, median and standard error were determined for shallow water areas and deep water areas (Table 5-4). Mean and median chlorophyll values were observed to be higher in deep water regions than in shallow water. Both mean and median turbidity values were similar in both shallow and deep water. Further analysis will be needed to determine if this pattern holds true for other systems at other times of the year.

Table 5-4. Comparisons of shallow (<2 meter depth) and deep (>3 meters) for a) chlorophyll-*a* (µg/L) and b) turbidity (NTU). Magothy river cruise, 27 June, 2001.

a.

Chlorophyll-a	**Observations**	**Mean**	**Median**	**Std. Error**
Shallow	420	36.2	11.7	2.56
Deep	1647	45.0	18.7	1.19

b.

Turbidity	**Observations**	**Mean**	**Median**	**Std. Error**
Shallow	420	7.52	6.55	0.12
Deep	1647	7.66	6.40	0.07

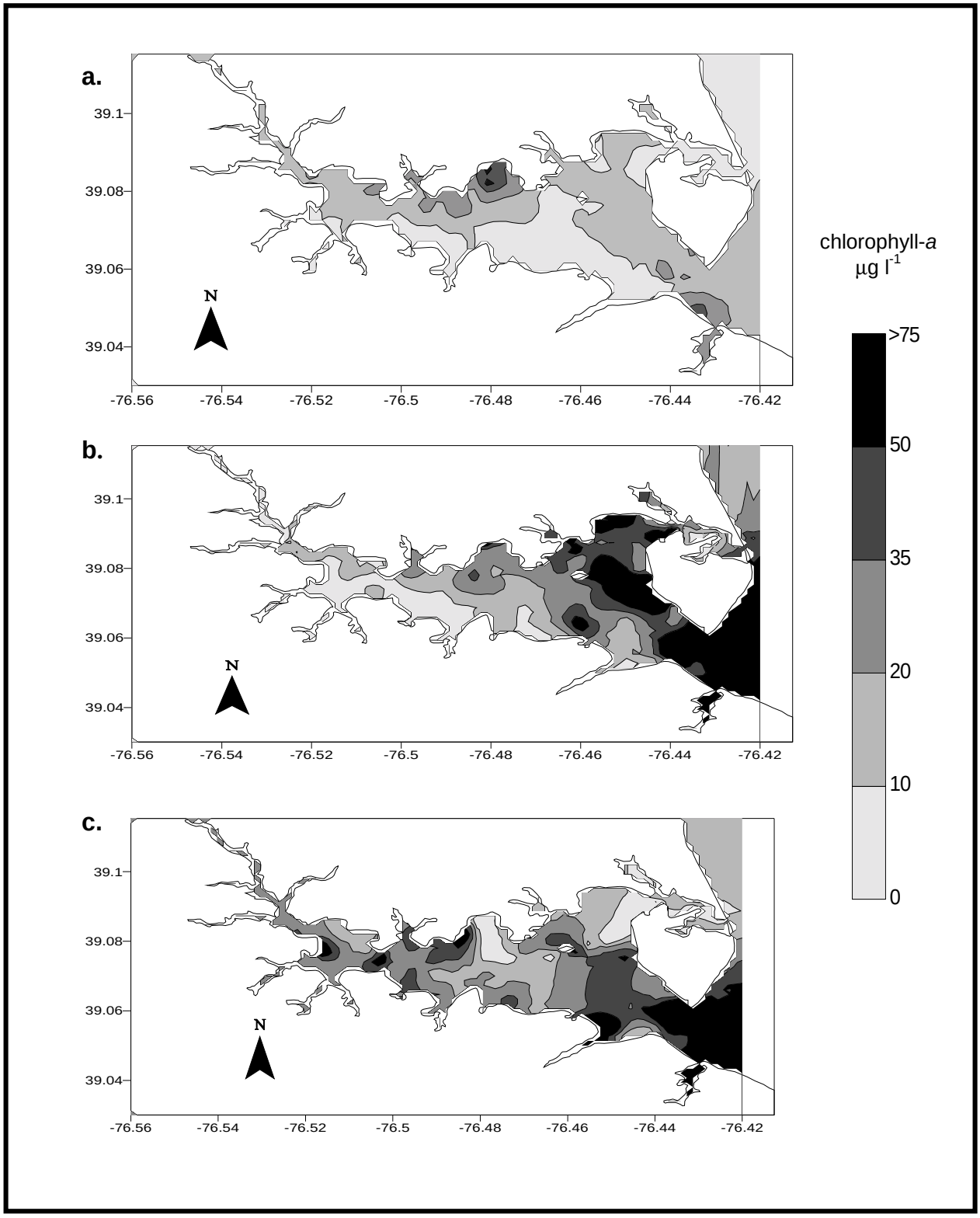

Figure 5-5. Interpolated maps generated from chlorophyll data for three consecutive cruises on the Magothy River: a. 13 June, 2001, b. 27 June, 2001 and c. 11 July, 2001.

5.4.3 Sources of error with interpolated maps

The creation of contour maps of water quality parameters based on DATAFLOW measurements allows for the visualization of spatial patterns. However, this process will incur a series of errors from various sources that should be recognized while interpreting these maps. One area of error involves the sensors used by DATAFLOW. The first source of error is incurred by the sensors themselves. The magnitude of this error will depend on the particular parameter being measured. One such error is associated with the dissolved oxygen probe. If tiny air bubbles cling to the probe membrane, or there is insufficient flow across the membrane, inaccurate readings can occur. Errors can also occur because individual sensors have different response times. As water is moving through the flow chamber, some sensors are able to take readings at shorter intervals. For example, the temperature/conductivity probe takes readings almost instantaneously, while the dissolved oxygen probe has response times of closer to one second. Another issue of sensor lag times concerns water residence time, and the effects of boat speed. If the boat is moving at a faster speed than the water is able to move through the system, then the DATAFLOW instruments will be recording data for a location somewhere behind the boat's position.

A second set of errors arises from the interpolation process that creates a regular spatial grid from the actual data collected along the vessel cruise track. This error depends on many factors, including the spacing and density of actual observations and the method of interpolation. Error is likely to increase as the distance between actual observations increases. The error can be compounded when there are many observations from a single location, namely when the research vessel is anchored at a calibration station. Certain interpolation test results can be distorted when multiple identical observations at the same location are used. Finally, the creation of contour maps of parameters such as turbidity and chlorophyll-a are actually estimated from regression relationships calculated from data collected at the calibration stations will incur further error in the contour maps. In other words, sensor data for turbidity and chlorophyll-a are converted to values directly comparable to those used by the Chesapeake Bay Program. While this procedure ensures compatibility between DATAFLOW and conventional monitoring, some error in the contour maps will result. This error will depend on the data collected during each cruise and may vary considerably. A detailed and thorough analysis of these errors has not yet been performed. As such, interpretation of these contour maps should be made with caution. Despite these limitations, the general patterns observed should remain valid.

5.4.4 Influence of Number of Data Points

Two mapping parameters were altered in an attempt to further understand ways that interpolated maps can be manipulated. The first of these parameters was the number of observations in the data set. A typical DATAFLOW cruise results in approximately 3,000 observations. It is possible that 3,000 observations are necessary to accurately interpolate a contour map. However, it is also possible that a smaller number would result in a similar interpolated result. Chlorophyll-*a* contour maps were generated from a typical DATAFLOW cruise, first using all of the observations, then using only the data from the eight calibrations stations on that cruise, and finally from the single DNR long-term monitoring station. From these maps, an area weighted mean chlorophyll value for the river for that date was calculated. While the maps themselves look somewhat different (Figure 5-6), statistically there is no significant difference between the

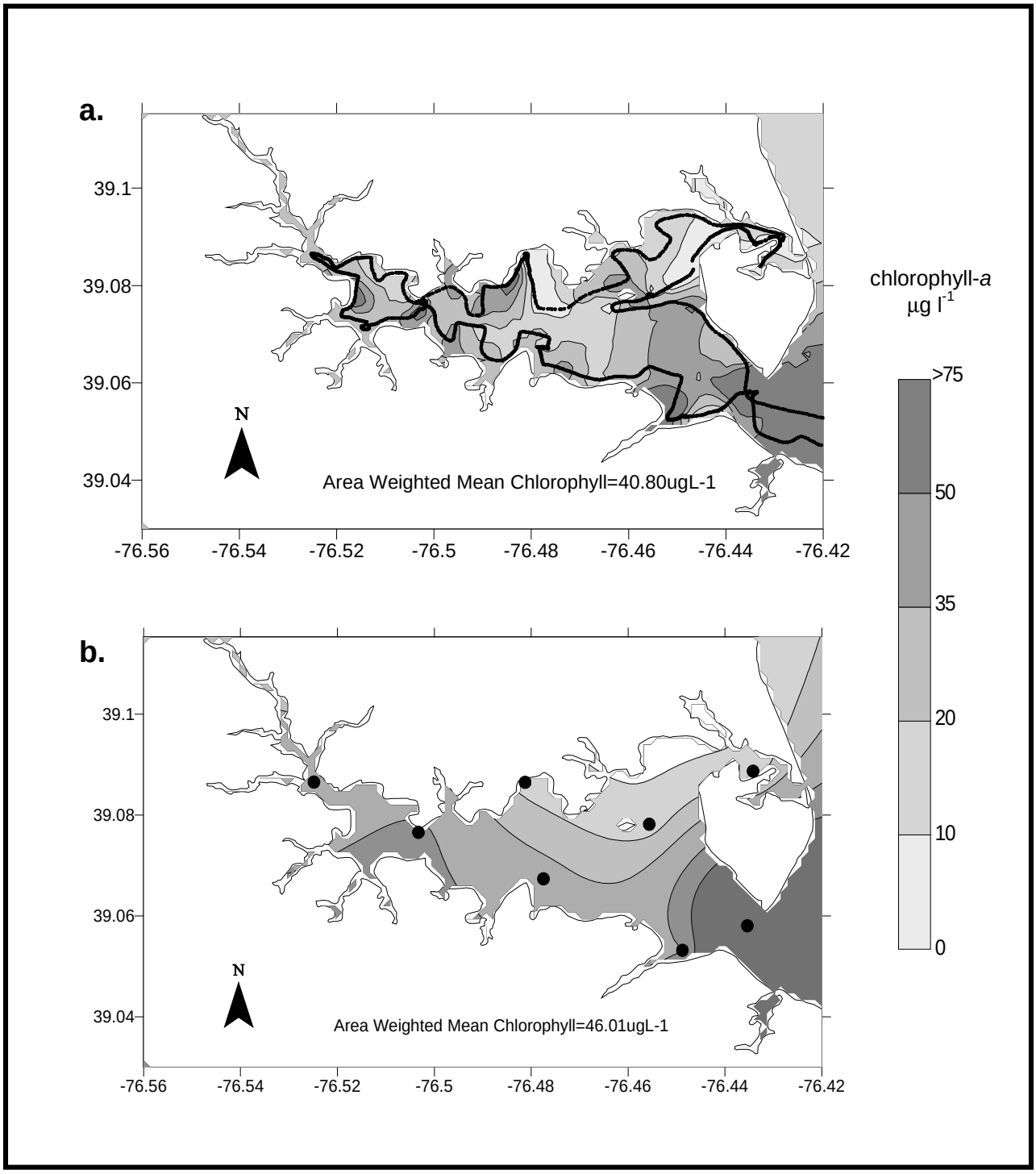

Figure 5-6. Examples of different interpolation results based on different number of observations. Data are from a Magothy River cruise conducted on 11 July, 2001.

a. Map generated from 2,836 observations from DATAFLOW cruise (map includes cruise track).

b. Map generated from data collected at 8 calibration stations (dots indicate location of calibration stations).

mean chlorophyll-*a* values generated from the maps (Figure 5-7). This suggests that the 3,000 data points generated from a DATAFLOW cruise may be more than are necessary to estimate the levels of chlorophyll in that system for that day. It is interesting to note, however, that the different values generated by the same method for turbidity were statistically different (Figure 5-8).

5.4.5 Influence of Cell Size

The second parameter that was altered was the spatial size of the cell used in the interpolation scheme. When creating a contour map in the Surfer application, cell size can be altered. Thus, each contour map is made up of a collection of cells of some uniform size. Cell size basically determines the resolution of the contour map; the smaller the cell, the more defined the patterns can be. As the cell size increases, the patterns become less defined (Figure 5-8). In the example provided in Table 5-4, the area weighted mean chlorophyll value increases slowly as cell size increases. However, at very large cell sizes, the estimated concentration increases sharply. It has not yet been determined what the ideal cell size is, that is to say, what cell size will best depict the conditions monitored.

Table 5-5. Examples of different interpolation results based on different cell size (based on July 11, 2001 Magothy River cruise).
All maps in this report were generated using the cell size marked with an asterisk (*).

Cell Size (km^2)	Area Weighted Mean Chlorophyll-*a* ($\mu g\ l^{-1}$)
0.004586	35.92
0.018582*	40.80
0.077508	49.19
0.327041	70.67

5.2 Future Directions

While the DATAFLOW system has been a valuable tool for monitoring water quality in the Chesapeake Bay and its tributaries, there are still several key issues yet to be resolved regarding its application. First, when using DATAFLOW to examine water quality parameters in SAV habitats, it is crucial to know the depth of the system at the point of sampling. The current depth transponder used with the DATAFLOW system displays reliable depth soundings in deep waters; however, in water less than two meters, the device is not as reliable. When cruising at high speeds in shallow areas, the current depth meter often fails to give an accurate reading. In addition, there are no nautical charts that can be trusted to accurately represent the depths in the shallow areas of SAV habitats. How can water quality data generated from a DATAFLOW cruise be reliably compared to the bathymetry data for the system in question? It may be

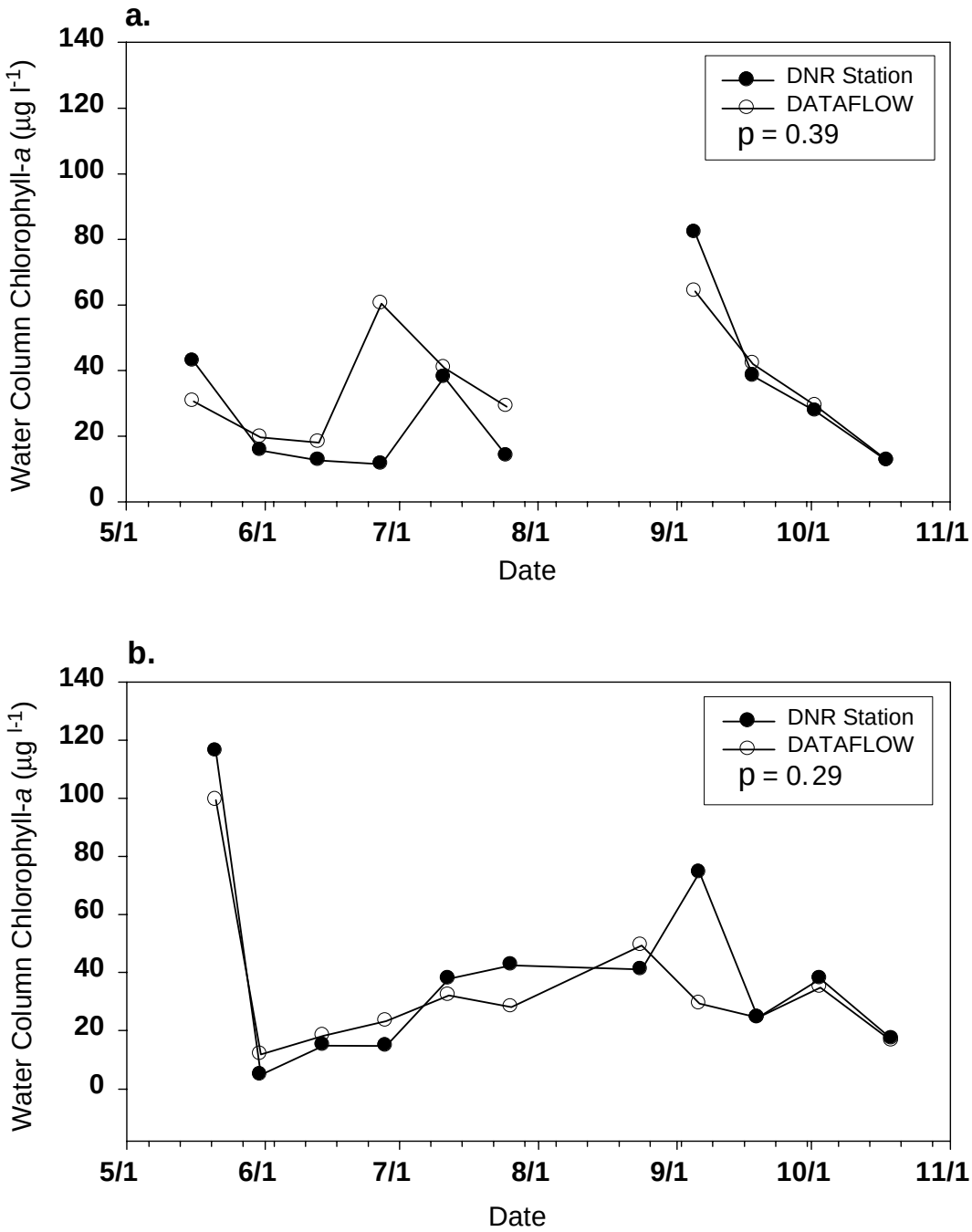

Figure 5-7. Area weighted mean chlorophyll-*a* values generated from interpolated data using single DNR calibration station versus DATAFLOW data for the:

a. Magothy River, and
b. Sever
c. n River in 2001.

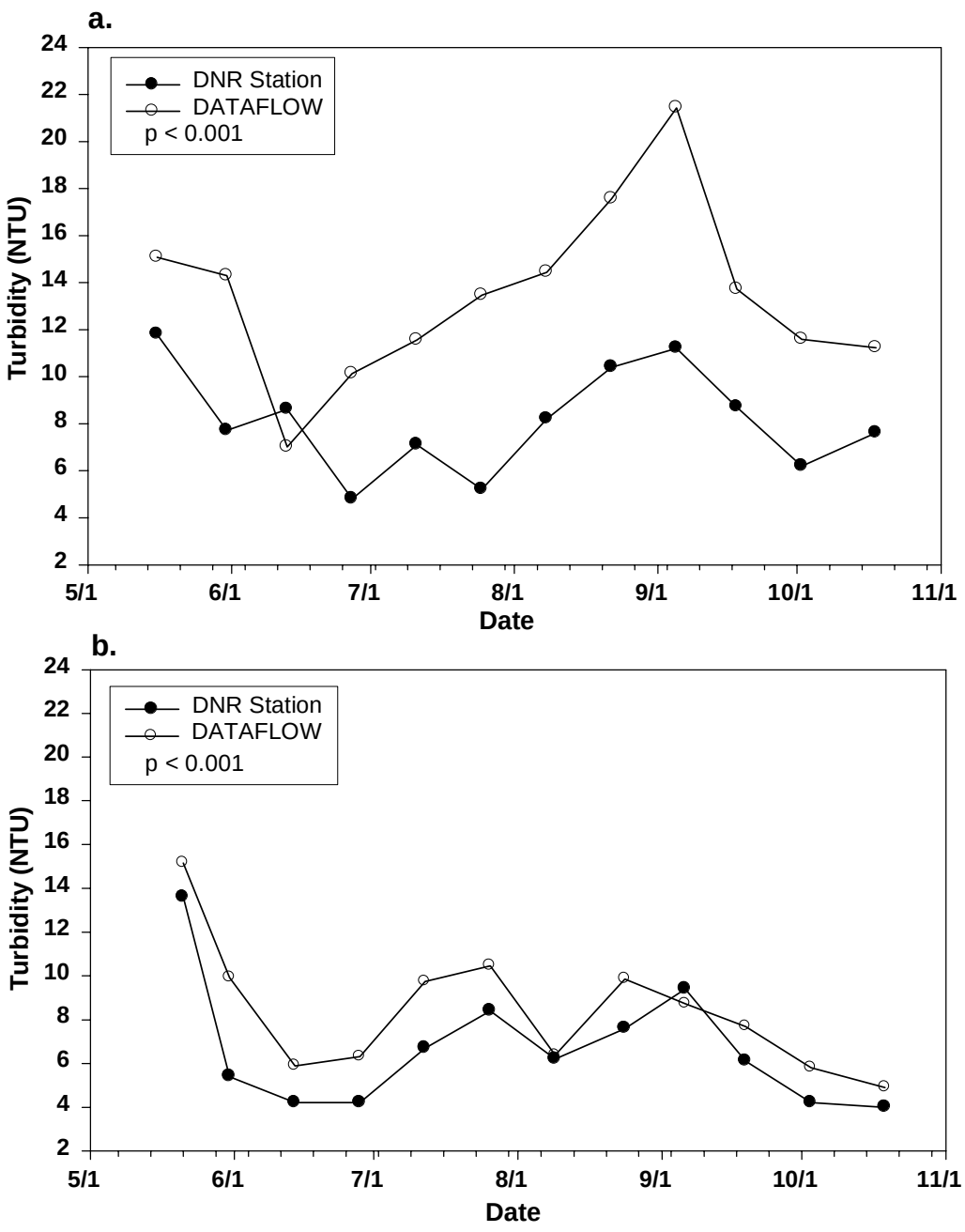

Figure 5-8. Area weighted mean turbidity values generated from interpolated data using single DNR calibration station versus DATAFLOW data for the

a. Magothy River, and

b. Severn River in 2001.

possible to obtain an existing bathymetric chart, and then overlay the contour map generated from the DATAFLOW data. This would allow the interpreter to analyze the data over varying depths. In the case that no reliable bathymetric data set can be found, it may be necessary to complete a "depth cruise" in the beginning of the season. This cruise would involve very slowly, deliberately covering as much of the tributary as possible, and using the resulting depth data to interpolate a bathymetric data set that can be used in conjunction with DATAFLOW observations.

Different cruise tracks can be utilized to answer different questions. For example, a square wave patterned cruise track over an entire estuary can be used to examine gradients between shallow water and deeper water. This is the type of cruise track that was performed in 2001. However, in order to focus on water quality in SAV habitat areas (<2m water depth), a square wave pattern would not be the most efficient method. In this case, a cruise track that follows the two-meter contour of the system would be more appropriate. It is then important to tailor the type of cruise track to the interest of the investigation. Future work will necessitate further defining what type of cruise track is most appropriate for a certain study.

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6. SUBMERGED AQUATIC VEGETATION IN THE MESOHALINE REGION OF THE PATUXENT ESTUARY: PAST, PRESENT AND FUTURE STATUS

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Abstract

The loss of submerged aquatic vegetation (SAV) from the Patuxent estuary during the latter part of the 20th century was explored using diverse data sets that included: historic SAV coverage and distribution data, SAV ground truth observations, water clarity and nutrient loading data, and epiphyte light attenuation measurements. Analysis of aerial photography from 1952 showed that SAV was abundant and widely distributed along the entire mesohaline region of the estuary; however, by the late 1960s, rapid declines in SAV took place following large increases in nutrient loading to the estuary. An examination of water clarity and epiphyte data suggest that the processes that led to the loss of SAV varied in strength along the axis of the estuary. In the upper mesohaline region, secchi depths were consistently less than established mesohaline SAV habitat requirements at 1m water depth, suggesting that water clarity was responsible for SAV decline. In the lower mesohaline region, where water clarity was consistently above SAV requirements, high epiphyte fouling rates significantly reduced light available to SAV. Experimental results show that epiphyte fouling had the capacity to reduce available light to SAV blades from 30% to 7% of surface light within a week, and likely contributed to the local decline and near total loss of SAV during the late 1960s and early 1970s. As a result, the

prognosis for near-term SAV recovery within the mesohaline portion of the estuary seems unlikely given existing water quality conditions.

6.1 Introduction

The Patuxent estuary, like many other temperate estuaries, has experienced dramatic declines in the coverage of submerged aquatic vegetation (SAV) during the last half of the 20th century (*e.g.*, Den Hartog and Polderman, 1975; Orth and Moore, 1983, 1984; Cambridge and McComb, 1984). Numerous studies suggest that reductions in light available to SAV, as a result of increased nutrient loading to these systems, was the primary cause of these declines (*e.g.* Wetzel and Hough 1973; Philips *et al*., 1978; Kemp *et al*., 1983; Dennison and Albert, 1985; Twilley *et al*., 1985). However, light available to the SAV blade is attenuated not only by the water column, but also by epiphytes and their associated communities that colonize SAV blades (*e.g.* Borum, 1985; Burt *et al*., 1995; Twilley *et al*., 1985). Recent efforts to better assess habitat quality for SAV in Chesapeake Bay have included the effects of epiphyte light attenuation in establishing accurate habitat requirements (Batuik *et al.*, 2000). In the Patuxent estuary, a significant but widely scattered amount of information is available documenting the changes in nutrient loading, water clarity, and SAV coverage that took place during the latter half of the 20th century. However, recent studies focusing on epiphyte light attenuation (Stankelis *et al.*, 1999) provide an important linkage between historical SAV coverage and water quality data to Chesapeake Bay SAV habitat criteria (Batuik *et al.*, 2000). In this paper we compare these diverse data sets to gain a better understanding of the processes that led to the decline of SAV, validate current SAV water quality criteria, and provide a prognosis for future recovery.

6.2 Land Use History and Estuarine Characteristics

A variety of historical observations and scientific studies indicate that SAV was present and abundant along most portions of the Patuxent estuary until late in the 20th century. For example, studies using pollen dated sediment cores found that SAV was continuously present from approximately 1200 AD to the early 1970s at several locations along the estuary (Brush and Davis, 1984; Brush and Hilgartner, 2000). In addition, historical documents such as "The Old Plantation" (Hungerford, 1859) describe the Patuxent during the mid-1830s as one of the clearest rivers flowing into the Chesapeake. Passages such as, "So transparent are its waters that far out from the shore you may see, in the openings of the sea-weed forest [SAV], on its bottom the flashing sides of the finny tribes as they glide over the pearly sands", give an indication of water clarity during that time. In 1850, approximately 86% of the watershed was agricultural pastureland and 14% was forested. By the 1970s, forested lands had risen to 57% of total land area while agricultural use had decreased. In the last 30 years land use has changed even further with urban and residential areas increasing, and forest and agricultural land decreasing (Costanza *et al.*, 1995).

While land use within the watershed (2400 km^2) has changed substantially over the last 150 years, human population growth did not increase rapidly until the second half of the 20th century. For example, in 1900 the total population of the watershed was 28,000, yet had only increased to

37,000 by 1940. Between 1940 and 1970 population increased more than six fold, to 246,000. The present population is in excess of 600,000 and land use is the most urban oriented of all major Chesapeake tributaries (Maryland Office of State Planning, 2000). Sewage treatment discharge also increased in association with population growth. Between 1940 and 1980, nutrient loading rates increased, with the greatest changes occurring between the late 1960s and mid-1980s. Sewage treatment plant discharges increased from 11,000 m^3 d^{-1} in 1963, to 136,000 m^3 d^{-1} in 1980, and finally to 190,000 m^3 d^{-1} in the late 1990s (Domotor *et al.*, 1989; USEPA, 2002).

Mean annual freshwater input to the estuary at the fall line is 10.6 m^{-3} sec^{-1}, which is the sixth largest source of freshwater entering Chesapeake Bay. The estuarine portion of the Patuxent is approximately 65 km in length and has littoral zone habitat sufficient to support SAV in all three salinity zones. However, the amount of SAV habitat (water depths < 1m) is not evenly distributed among these regions. For example, there are 20.9 km^2 capable of supporting SAV in the mesohaline zone, 5.8 km^2 in the oligohaline zone and only 2.0 km^2 of SAV habitat in the tidal fresh zone. Because the mesohaline zone contains the largest area capable of supporting SAV, we focused on changes that occurred within that region of the estuary. This region of the estuary extends from the mouth of the estuary at Drum Point, 35 km upriver to Chalk Point (Figure 6-1). During the summer months, median water residence time in the mesohaline portion of the estuary is approximately 35 days (Hagy *et al.*, 2000).

6.3 Methods and Analysis

Data from a number of different sources were required in this analysis and included: aerial photographs of SAV distribution and coverage, SAV ground truth observations, fall line nutrient loading data, and water quality data collected from numerous studies. However, differences among study locations, seasons in which data were collected, and methods of data collection, restricted, in some cases how information could be compared and analyzed. Results of recent studies of light attenuation by epiphytic accumulation on SAV were used to help interpret changes in SAV distribution and provide a link between increases in nutrient loading and SAV decline due to light limitation. These data were also compared to current light based SAV habitat requirements (Batuik *et al.*, 2000).

6.3.1 Interpretation of Aerial Photography

Estimates of SAV distribution in the mesohaline region of the Patuxent estuary, prior to the decline of SAV populations, were made by photographic interpretation of historic Soil Conservation District aerial photographs taken in 1938, 1952 and 1964 (National Archives and Record Service). Temporal changes in SAV coverage were assessed by comparing these data to similar estimates of SAV coverage collected annually since 1984 by the Virginia Institute of Marine Sciences (VIMS, 1999). Photographs from 1952 cover the entire mesohaline portion of the estuary, and are directly comparable to data collected in recent years. However, in 1938 photographs were only available for the region between Chalk Point and Broomes Island. In order to assess changes in the distribution and coverage of SAV between 1938 and 1952, only

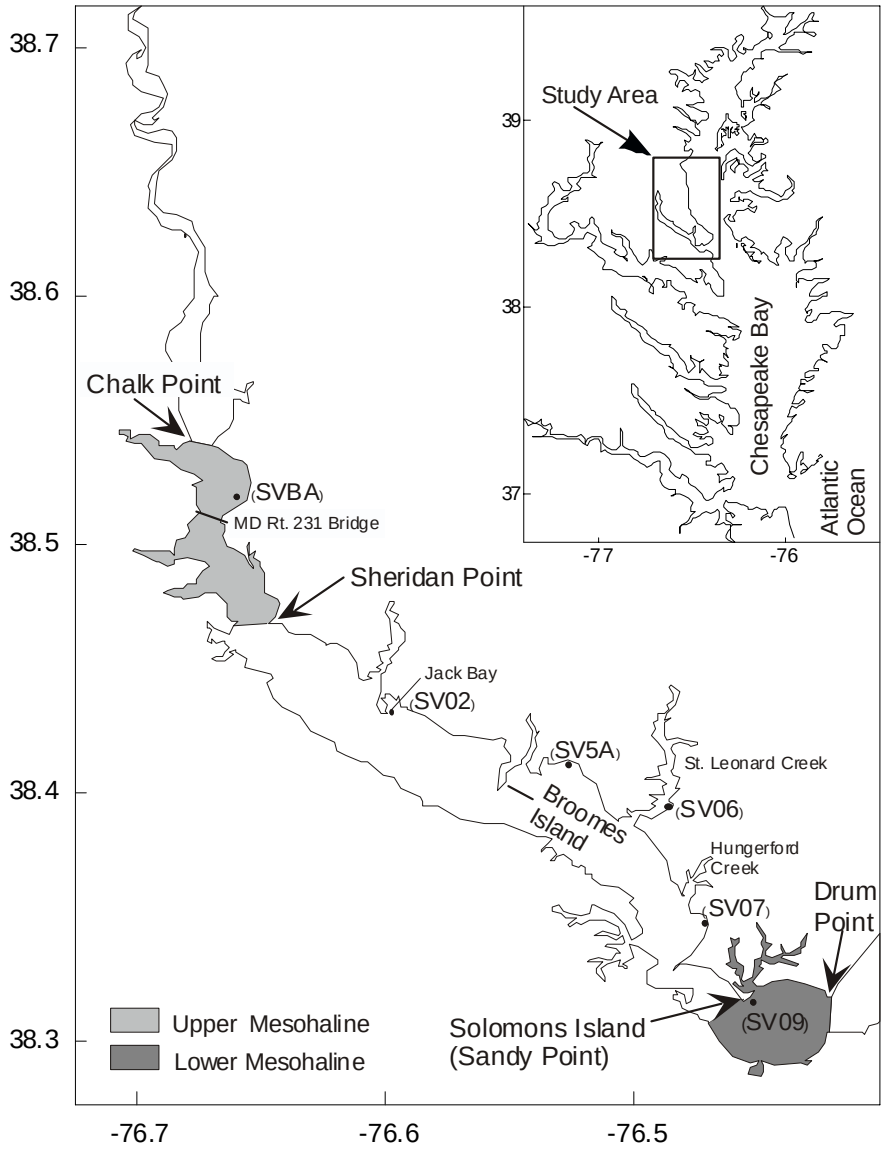

Figure 6-1. Mesohaline region of the Patuxent River estuary. Shaded areas represent locations where water quality data were collected. Epiphyte study locations are also shown.

those areas in common were compared. Limited photographs of sufficient quality were also available for the Broomes Island area as well as Solomons Island for 1964 and provided additional information concerning the temporal pattern of SAV decline.

For 1952, 24x24 inch black and white photographs were digitally scanned at 150 dpi, and georeferenced to orthophoto quarter quadrangle maps. For 1938 and 1962 9x9 positives were scanned at 300 dpi and geo-referenced to State Highway Administration and stream layer rasters accurate to quad scale (40'). After geo-referencing, a composite image was formed from the most desirable sections of each photo. SAV beds were traced directly upon the scanned images as a vector layer, at an on-screen scale of 12,000:1, with the original positives used to help identify difficult to determine areas. On-screen scales for all photographs were kept at 12,000:1, to provide an on-screen visual image of similar quality to the 24,000:1 photos directly interpreted by VIMS in the current SAV monitoring program. This decision had the effect of generating similar minimum bed sizes and final vectors directly comparable between 1938, 1952, 1964 and those interpreted by VIMS in the more recent surveys (VIMS, 1999).

6.3.2 Water Quality and Nutrient Loading

Secchi depth was chosen as an indicator of water transparency because it was the most commonly collected parameter in the studies reviewed. Although current SAV habitat requirements in Chesapeake Bay are based upon light available at the SAV leaf surface (Batuik *et al.*, 2000), water clarity (secchi depth) provides a secondary diagnostic tool indicating light penetration to a fixed depth. For comparative purposes we used a water depth of 1m to convert the minimum water column light requirement of 22% surface light for SAV in the mesohaline zone (Batuik, *et al.*, 2000) into a minimum secchi depth of 1m.

Since estuaries are temporally and spatially quite heterogeneous, differences in the season and location at which data were collected reduced the amount of data that could be compared. While the SAV growing season typically includes the period April - October, (Batuik *et al.*, 2000) data for this comparison were limited to the period between 15 June and 15 September of each year because the majority of data were available for that period. Secchi data were summarized and pooled from two regions within the mesohaline portion of the estuary (Figure 6-1). In the upper mesohaline zone, data were collected from sites located between Chalk Point and Sheridan Point. In the lower mesohaline zone, data from sites located between Sandy Point and Drum Point were used.

Estimates of nutrient loading at the fall line for the period 1960 - 1977 were reconstructed using river flow, rainfall, and nutrient concentration data (Hagy *et al.*, 1998). Nutrient loading data from 1978 to the present were collected as part of the USGS Fall Line Monitoring Program.

6.3.3 Epiphyte Light Attenuation

Estimates of epiphyte light attenuation were made by exposing artificial substrata to natural fouling at six near-shore locations along the mesohaline portion of the Patuxent estuary. These

stations were distributed from just above the MD Route 231 bridge (SVBA), to Solomons Island (SV09) near the estuary mouth (Figure 6-1). Artificial substrata in the form of thin strips of Mylar® polyester plastic (2.5 cm x 51 cm x 0.7 mil) were deployed at approximately 1.0 m water depth (mean water depth), on a weekly basis from June through October, 1998. Strips were exposed to fouling for periods of 6-8 days during each deployment. Small foam floats (~3.5 x 3.3 cm) were attached to one end of each strip to maintain an upright position in the water column, yet allow the strip to move with water currents. The other end was fastened to a weighted PVC frame placed on the sediment surface. Previous studies using this technique have shown that Mylar® strips can be an adequate surrogate for live grass blades for the estimation of epiphyte accumulation during short-term (one week) in-situ deployments (Stankelis *et al.*, 1999). Estimates of light attenuation due to epiphytic fouling were accomplished by measuring the difference between light flux transmitted through fouled strips and clean unfouled strips. Light flux measurements were made with the light attenuation measurement apparatus or LAMA (Figure 6-2). The LAMA consisted of a standard 60 watt light source with a light diffuser screen, a water bath, and a Li-cor model 192 SA quantum sensor. This configuration was similar to that used by Burt *et al.*, (1995). All light flux measurements were made in 0.2 µm filtered seawater. The Li-192SA quantum photo sensor measures photosynthetically active radiation (PAR) in the 400-700 nm range. The LAMA was also configured such that light flux reaching the sensor through a blank (clean) strip was in the range of 90-105 µmol m^{-2} sec^{-1}. Epiphtye material from both sides of these strips were removed and analyzed for total chlorophyll-a mass, and dry mass per unit area. Estimates of water column light attenuation were also made concurrently with the epiphyte collections at each location and were used in the calculation of percent light through the water column (PLW), where k_d (m^{-1}) is the water column light attenuation coefficient, and Z (m) is the mean tidal depth (MTL).

$$(1) \qquad PLW = 100exp[(k_d)(Z)]$$

In order to include the contribution of epiphtye light attenuation we used the PLL statistic

$$(2) \qquad PLL = PLW[1-LA/100]$$

following the method outlined by Batuik *et al.*, (2000), where LA (% light exposure) is the measured epiphyte light attenuation. In this way we were able to compare our estimates to current SAV habitat requirements.

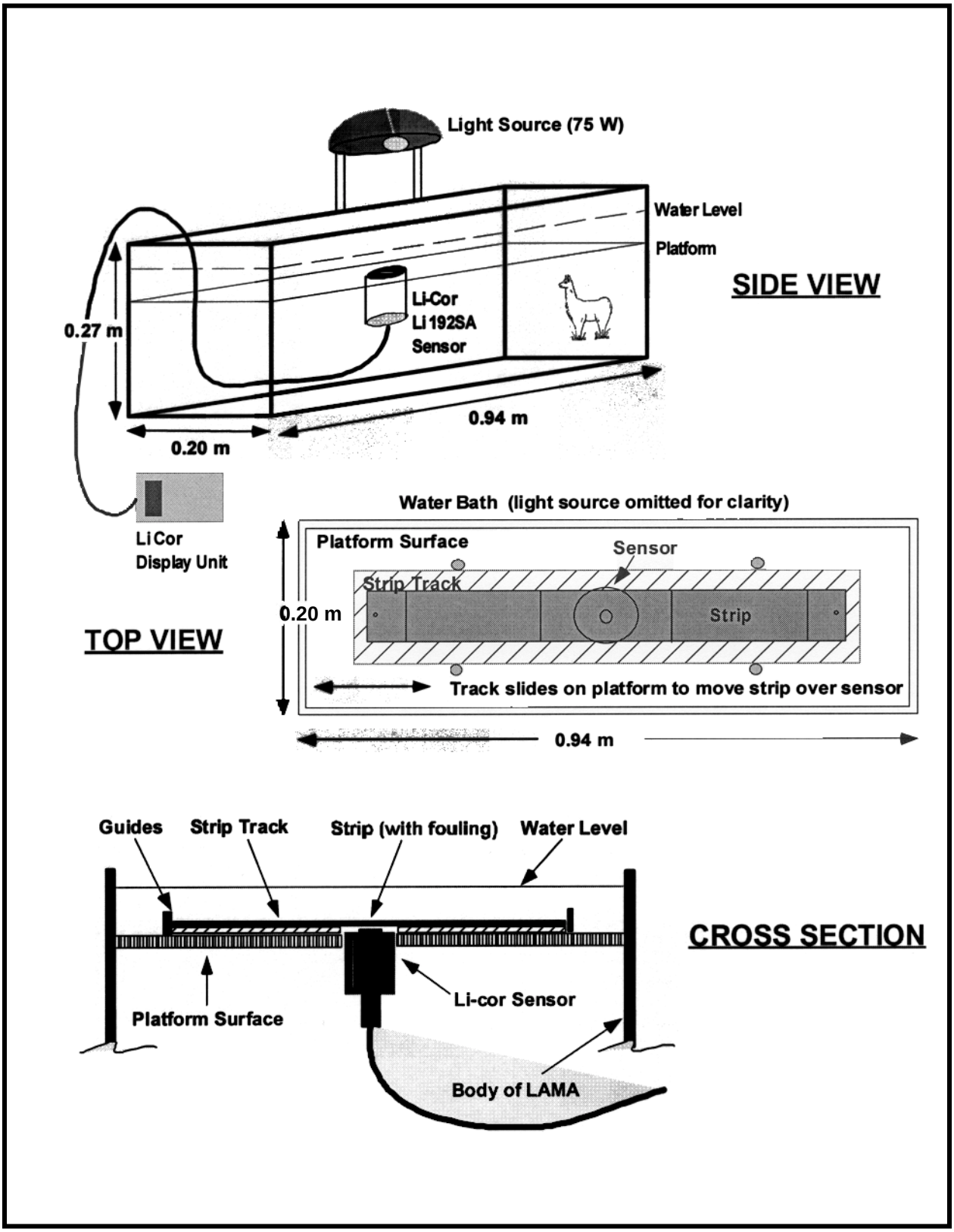

Figure 6-2. Diagrammatic sketch of the light attenuation measurement apparatus (LAMA).

6.4 Results

6.4.1 SAV Coverage and Distribution in the Patuxent River estuary

Interpretation of the 1938 aerial photographs identified 715 ha of SAV within the area located between Chalk Point and Broomes Island. In comparison, 397 ha were identified in 1952 within the same region of the estuary. Based on detailed evaluation of photos, it appears that differences in total SAV area were due to loss of SAV from the deeper portions of littoral areas. Despite a decline in total coverage between 1938 and 1952, SAV was still widely distributed along the entire mesohaline region of the Patuxent estuary in 1952 (Figure 6-3). While such a change may be consistent with declining water quality, these differences could also be attributed to natural fluctuations in SAV distribution, differences observed between seasons, as well as differences in water quality at the time the photos were taken. For example, the 1938 photos were taken on 24 April, while the 1952 photos were taken on 26 June. *Zannichellia palustris*, one of the more abundant species in the lower Patuxent, begins to die back in late May, potentially resulting in a lower SAV coverage for photos taken after this time.

A comparison of the SAV coverage of the Broomes Island area among years 1938, 1952, and 1964 indicate SAV coverage in 1964 (133 ha) intermediate between 1938 (217 ha) and 1952 (86 ha; Figure 6-4). A similar comparison of SAV coverage of the Solomons area among the same three years indicates virtually no change in distribution or coverage of SAV (Figure 6-4). Recent aerial surveys show that even during a minor resurgence in SAV during the mid-1980s, total SAV coverage was a small fraction of that observed a few decades earlier (Figure 6-5). Since 1990 only small ephemeral beds have been observed in the mesohaline portion of the estuary. A visual comparison of a low altitude photo of the Solomons area taken in 1938 to a photo taken in 1999 further illustrates the changes in SAV abundance that have taken place during the 20th century (Figure 6-6).

In addition to photographic evidence of SAV coverage and distribution in the upper mesohaline portion of the estuary, a series of SAV ground truth observations were made during the 1960s and 1970s. In 1964, Anderson (1969) found large beds of *Ruppia maritima*, *Potamogeton perfoliatus*, and *Najas flexilis* on both shores of the Patuxent upstream and downstream of the then non-operational Chalk Point power generating station. The following year, after initiation of power-plant operations, Anderson *et. al.*, (1968) found the *R. maritima* population very much reduced and replaced by *P. perfoliatus*. From 1964 to 1968, the Academy of Natural Sciences of Philadelphia (ANSP) reported healthy beds of *P. perfoliatus* at Sheridan Point and *Myriophyllum sp.*, *Elodea nutallii*, and *Fustuca eliatior* present at lower densities nearby (ANSP, 1964 – 1968). In 1969 SAV density at Sheridan Point was much reduced compared to previous years, and by 1970, the area was devoid of SAV (ANSP, 1970 - 1971). Since that time SAV has not been observed in that region of the river. Aerial photography from the late 1960s is of insufficient quality to compare these observations to other regions of the river.

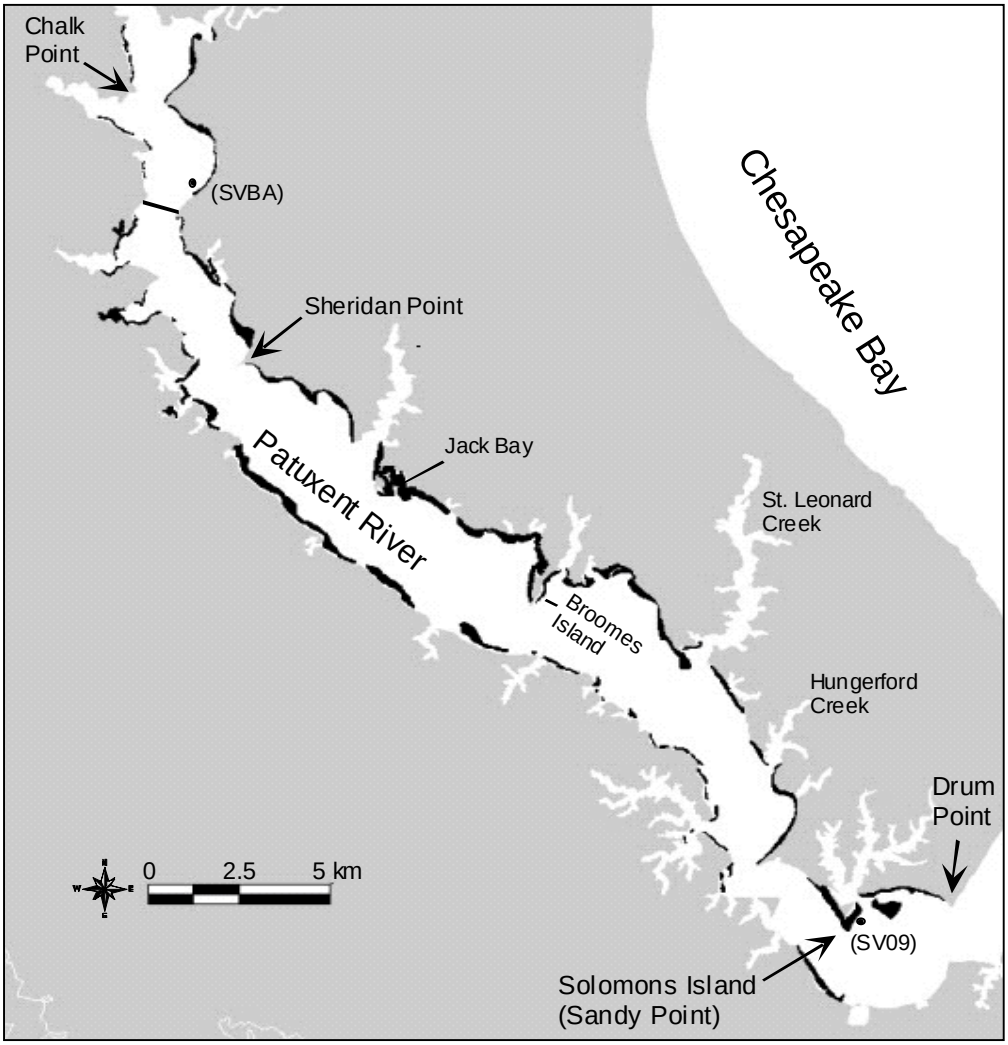

Figure 6-3. Estimated distribution of SAV in the mesohaline reach of the Patuxent River in 1952. Area upriver of Broomes Island was photographed on June 26. Area below Broomes Island was photographed on October 22.

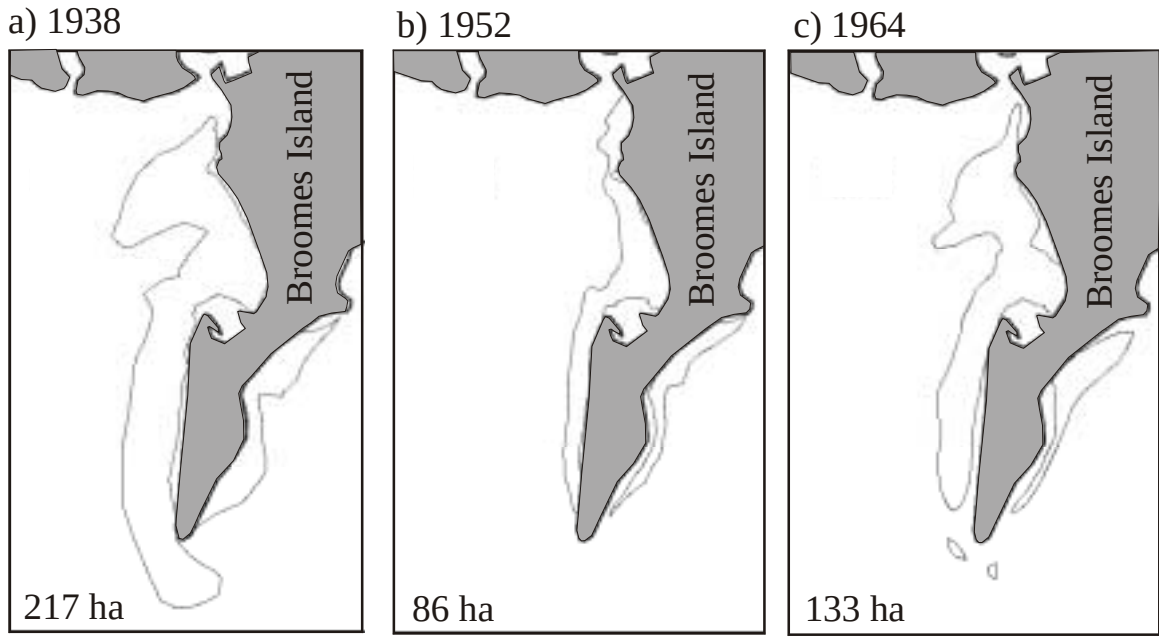

Figure 6-4a – 6-4c. Distribution of SAV around Broomes Island in a) 1938, b) 1952, c) 1964.

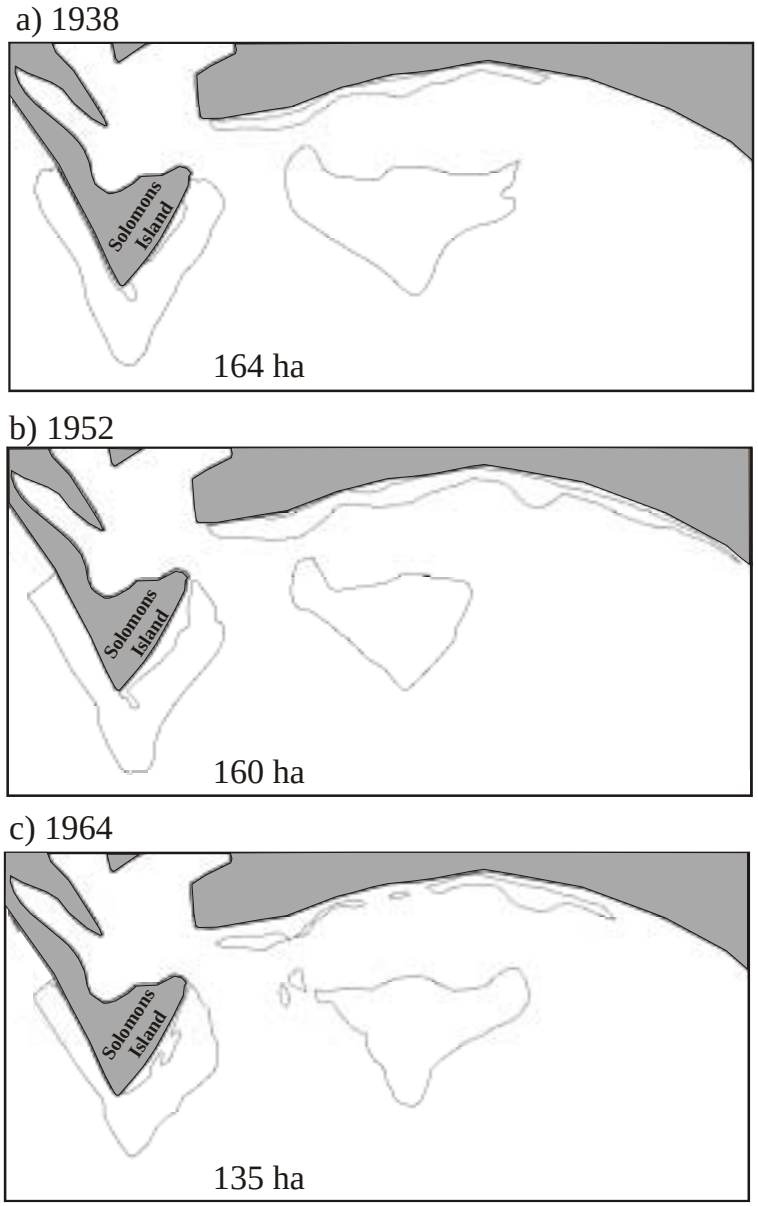

Figure 6-4.d – 6-4.f. Distribution of SAV around Solomons Island in d) 1938, e) 1952, f) 1964.

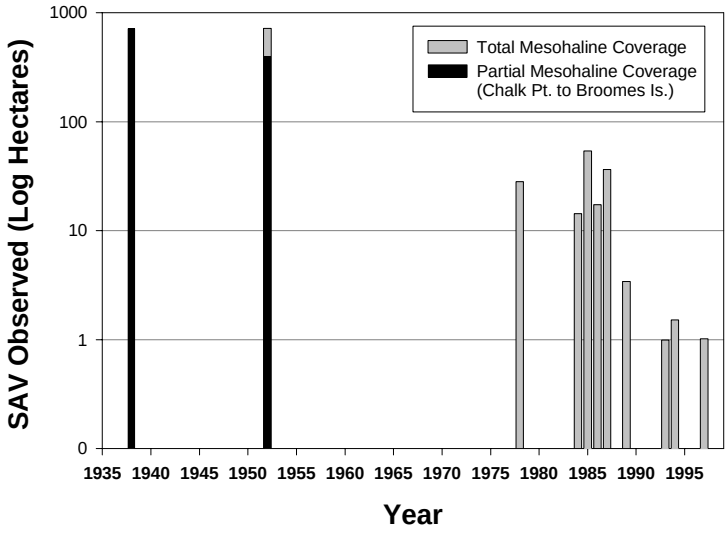

Figure 6-5. Estimated SAV coverage in the mesohaline portion of the Patuxent River estuary. Area estimated in 1938 represents only a portion of the total SAV habitat, due to lack of interpretable photographs. Data from 1952 and 1975 to 1997 include the whole mesohaline portion of the estuary (VIMS, 1999).

a.

b.

Figure 6-6. Aerial view of SAV beds around Solomons Island in (a) 1938, and (b) 1999.

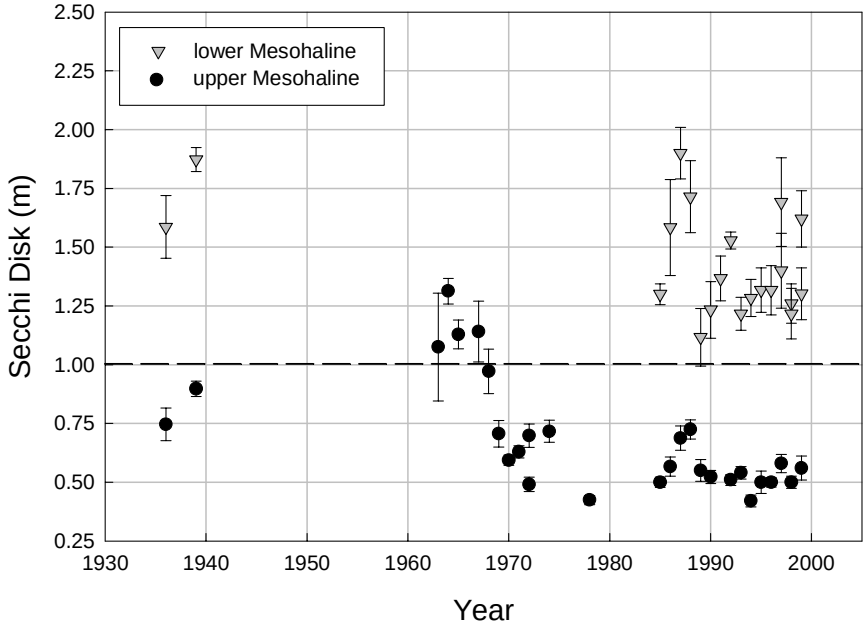

Figure 6-7. Mean (+/- 1SE) secchi depth data from two regions within the mesohaline portion of the Patuxent River estuary. Data were collected from 15 June through 15 September of each year. Dashed line represents mesohaline secondary SAV habitat limit Batuik *et al.*, (2000) using the conversion k_d = 1.45/secchi depth for a depth of 1m.

6.4.2 Water Transparency and Nutrient Loading

In the upper mesohaline region (Chalk Point to Sheridan Point), secchi depth data were available for three distinct time periods. The earliest measurements were recorded in 1936 and 1939 (Newcomb and Brust, 1940; Nash, 1947) during a period when SAV was still abundant in many areas. During this time, mean summer season secchi depth was greater than 0.75 m (Figure 6-7). While this value was below the 1 m secchi depth regarded as necessary for SAV at the 1m depth contour (Batuik *et al.*, 2000), SAV was abundant at that time (Figure 6-5). Although no field data were found through the 1940s and 1950s, aerial photographs from 1952 show that SAV was still present along many reaches of the estuary suggesting that water quality had not yet deteriorated to where SAV could not survive. The next series of secchi depth data were available from 1964 through 1974, (ANSP, 1965 – 1975). During this period, mean secchi depth dropped sharply from a maximum of 1.3 m in 1964 to a minimum of 0.5 m in 1972 (Figure 6-7). During this time, large changes in nutrient loading rates were taking place within the estuary. Largely because of increases in sewage discharges, both total nitrogen (TN) and total phosphorus (TP) loads at the fall line more than doubled between 1960 and 1975 (Figure 6-8). From 1985 through 1998, mean secchi depths (USEPA, 2002), while variable, have remained depressed compared to conditions found during the late 1930s, and well below the 1m secchi depth needed for healthy SAV at the 1 m depth contour (Batuik *et al.*, 2000).

In the lower mesohaline region (Sandy Point to Drum Point) the earliest known secchi depth measurements were made in 1937 and 1939 (Newcome and Brust, 1940; Nash, unpublished). These data indicate that summer (June 15 – Sept 15) secchi depth was much greater in the lower river (mean = 1.9 m) compared to the upper mesohaline zone (mean = 0.8 m, Figure 6-7). Unfortunately, no water transparency data were located from the 1960s and early 1970s. Data collected since 1985 show high inter-annual variability in water clarity conditions (USEPA, 1999). For example, in 1987, the median summer secchi depth was 1.9 m while two years later the median summer secchi depth had decreased to 1.1 m. Despite this variation, water clarity has consistently exceeded that estimated as needed for SAV growth to depths of 1 m (Batuik, *et al.*, 2000).

6.4.3 Epiphyte studies

Mean epiphyte fouling rates increased linearly from upper to lower mesohaline locations and differed by a factor of six within the whole mesohaline portion of the estuary. The lowest mean fouling rate (0.60 µg chla cm^{-2} $week^{-1}$) was found just south of Chalk Point at the most turbid site (SVBA), while the highest mean fouling rate (3.36 µg chla cm^{-2} $week^{-1}$) was found at the least turbid site, Sandy Point (SV09). Estimates of percent surface light reaching the leaf surface (PLL) were calculated using epiphyte dry mass after a week of accumulation (Figure 6-9). This relationship shows that relatively small increases in epiphyte material can translate into large increases in light attenuation. In order to examine the relationship between light available through the water column as well as through an epiphyte layer, a plot of mean PLW and PLL versus location along the estuary was constructed (Figure 6-10). As water clarity improved in the down-estuary direction, progressively higher epiphyte biomass attenuated a higher fraction of

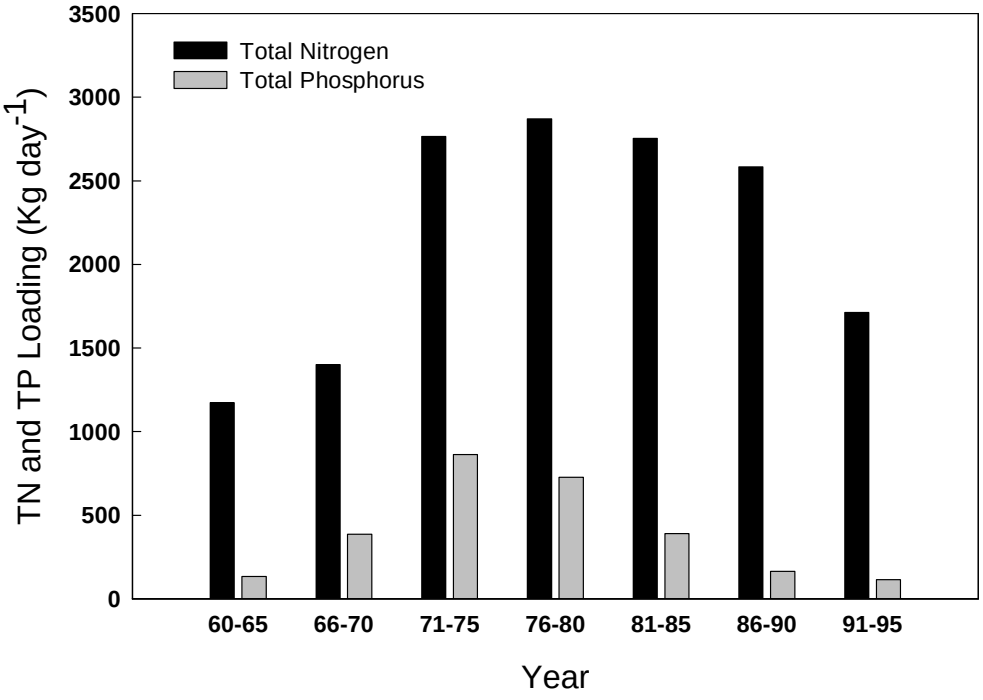

Figure 6-8. Estimated Patuxent River annual total nitrogen (TN) and total phosphorus (TP) loading rates measured at the fall line from 1960 to 1995. Data were averaged by 5-year intervals. Adapted from Hagy *et al.* (1998).

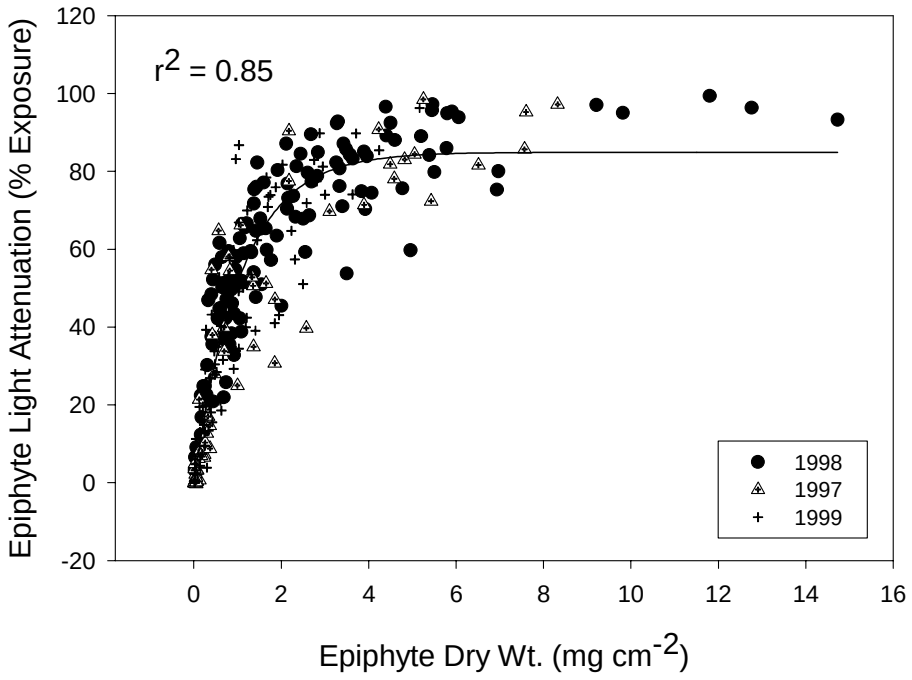

Figure 6-9. Epiphyte light attenuation vs. epiphyte dry mass where light attenuation = 84.634{1- exp[(-0.963)(Epi drywt)]}.

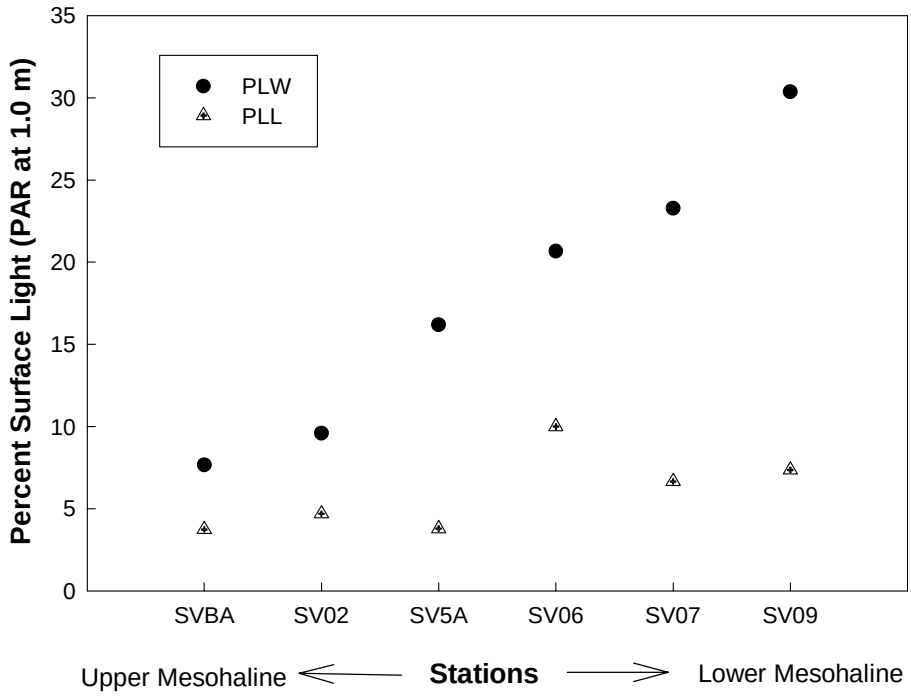

Figure 6-10. Average SAV growing season (Apr – Oct) seasonal estimates of light availability at one meter depth along the Patuxent River estuary during 1998. PLW is percent light reaching 1.0 m depth, and PLL is percent light at the leaf surface and includes the contribution of epiphyte light attenuation. PLW = 100 exp[(-k_d)(Z)] and PLL = PLW[1-LA/100] where LA = epiphyte light attenuation.

the light available to SAV, thereby reducing the benefits of clearer water to the SAV blade. High epiphyte fouling rates at Sandy Point (SV09) reduced PLL to 7% of surface irradiance, down from a potential PLW of 30% after a week of exposure. In contrast, at the most turbid site (SVBA), epiphytes had a relatively smaller impact on the light available to SAV, reducing PLL to 4% from a potential PLW of 8%.

6.5 Discussion

Data show that SAV was present and widely distributed along the whole mesohaline portion of the estuary through the early 1960s. While historic photos provided a baseline for the amount of SAV present in the mesohaline estuary prior to eutrophication, insufficient data were available to document the detailed temporal sequence of the decline of SAV coverage. Differences in water clarity (which affects delineation of SAV beds) and the time of year of the photography made this analysis impossible. However, photographic interpretation of the SAV coverage near Broomes and Solomons Islands show that SAV coverage in 1964 was similar to that found much earlier in the century. By the late 1960s and early 1970s, TN and TP loading at the fall line had increased dramatically, stimulating changes to the estuarine ecosystem. For example, community primary production and respiration in the upper mesohaline increased by factors of 3.7 and 1.8 respectively between 1964 and 1992 (Sweeney, 1995). It appears that by the mid-1970s SAV populations became too sparse to be detected by aerial photography in the mesohaline portion of the estuary. Available data indicate that increased nutrient loading to the Patuxent River estuary was primarily responsible for the SAV decline. However, we suggest that mechanisms responsible for this decline, and the strength of the response to eutrophication differed by location within the mesohaline estuary.

In the upper mesohaline region, both water clarity and SAV coverage declined during the late 1960s. By 1969, mean summer secchi depth in that region had decreased to 0.7m, coincident with the first recorded decline in SAV. In 1970, mean summer secchi depth had declined to 0.6m and no SAV was observed in previously vegetated areas. Since that time, mean secchi depth has remained less than 0.6m, which is much less than needed for healthy SAV growth at a depth of 1m. SAV has not been observed in that area of the estuary since. Recent studies focusing on light attenuation by epiphytes further suggest that epiphytes contribute relatively little to light attenuation compared to the water column attenuation at these upper mesohaline locations. For example, at the study site located near Chalk Point (SVBA), summer season fouling only reduced light to the leaf surface an additional 4% of surface radiation, further indicating that water clarity was the primary factor responsible for the local extinction of SAV in this area.

In contrast, SAV in the lower mesohaline region were exposed to a different set of water quality conditions. While aerial photographs showed that SAV was still present and abundant through the early 1960s, by the late 1960's and early 1970s, SAV around Solomons Island was observed in some years but not in others (K. Wood, personal communication). This suggests that SAV communities, though stressed, were able to maintain minimal recruitment and growth during this time. However, the impact of Hurricane Agnes in June of 1972 resulted in major losses of SAV in all regions of the Bay (Orth and Moore, 1983). The extreme conditions brought about by this

storm were likely a significant contributor to the final local extinction of SAV in this region. Since no data or written documentation of water clarity were available from the late 1960s and early 1970s, we cannot conclude that declining water clarity alone was responsible for the loss of SAV. In fact, secchi depth measurements collected since the mid-1980s indicate that water clarity should be sufficient to support SAV to the one-meter depth contour (Batuik *et al.*, 2000). However, persistent SAV populations have not become re-established. During the last few decades, several SAV species have been observed in small patches within this region of the estuary (Moore, 2000). However, these populations have rarely persisted for more than a single season, and in many cases only a few months. For example, in 1997 a bed of *Stukenia pectinata* was found near Hungerford Creek (Figure 6-1), but did not survive beyond summer. Similarly, *R. maritima* was observed, in small patches (1-4 m^2) along shoreline areas near the mouth of the Patuxent in the summer of 1999. These isolated patches also did not persist into the next season. An exception to this generalization has been the frequent appearance of the early spring annual *Zannichellia palustris*, which has been found in many of the smaller tributaries, and along the lower 25 km of the main estuary (personal observation). However, this species germinates in late winter (generally February/March) and completes its life cycle by mid-June.

Several mechanisms may explain why these small patches of SAV have not persisted, including waterfowl grazing or disturbance by cownose rays. However, it appears that light attenuation, due to epiphytic fouling of SAV leaves is a significant stress to SAV throughout the lower mesohaline and is the probable mechanism contributing to the loss of SAV from this area. During the summer, epiphyte accumulation in this area (e.g., SV09) can reduce the amount of light reaching SAV blades from approximately 30% of surface irradiance to less than 7% within a week, which is far below the 15% estimated as needed for continued SAV survival. While an exact determination of light availability to a whole plant would depend on many variables such as leaf age, water depth, and hydrodynamics around the blade, these data show that epiphtye accumulation can have a large impact on light availability to SAV.

6.6 Conclusions and Prognosis for Recovery

The qualitative responses of SAV to eutrophication have been examined in a number of field, mesocosm, and laboratory studies. However, the number of direct and indirect processes that influence the growth, survival and distribution of SAV make quantitative, *in-situ* predictions much more elusive. Even less well known are the conditions necessary to restore SAV to large areas that have suffered complete losses of SAV populations due to eutrophication (Duarte; 1995, 2000).

It appears likely that a variety of factors are limiting the resurgence of SAV in the Patuxent estuary. In the upper mesohaline region, water transparency remains far below what has been estimated as the minimum habitat requirements for SAV (Batuik *et al.*, 2000). Consistent with this, SAV has not been observed in this region since the late 1960s, and the prospects for recovery seem remote, given the persistently poor water clarity conditions in this area. Management actions have reduced nutrient loads to the head of the estuary (Figure 6-8) but there has not been a concomitant reduction in turbidity in the upper mesohaline littoral zones.

In the lower mesohaline region, it appears that several factors may be restricting SAV from becoming re-established including epiphyte fouling of SAV, physical disturbances, waterfowl grazing and lack of a proximal seed source for some species (*e.g.*, Zostera marina). It seems very likely that epiphyte-induced light limitation is the primary factor suppressing recovery of SAV, in the same fashion that this mechanism probably caused the massive decline of SAV several decades earlier. Multiple, small (1-16 m^2) SAV transplants in the lower mesohaline region all suffered severe epiphytic fouling and transplants have not persisted beyond a year or two. Small, naturally occurring beds of other species (*e.g.*, *S. pectinata*) have been observed in recent years but these also rapidly fouled and none have persisted more than a single season. Small, newly established SAV beds may be particularly sensitive to grazing pressure from non-native mute swans (*Cygnus olor*) and physical disturbance by cownose rays (*Rhinoptera bonasus*) than larger, more established, populations. The results of transplant experiments we have conducted in the lower mesohaline Patuxent, using both *Ruppia maritima* and *Z. marina*, suggest that protection of new transplants from grazing and physical disturbance may be necessary for initial SAV survival, but these plants still must contend with epiphyte shading. The near-term prognosis for SAV recovery to the lower mesohaline estuary seems poor, unless further improvements can be made in water quality that can limit epiphytic fouling rates and effective means developed to protect developing SAV beds from excessive grazing and disturbance.

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7. A WATERSHED PERSPECTIVE ON NUTRIENT ENRICHMENT, SCIENCE AND POLICY IN THE PATUXENT RIVER, MARYLAND: 1960 - 2000

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"How the land is used is a basic factor in the ecological health of the Chesapeake Bay."
Year 2020 Panel Report, 1988

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Abstract

The Patuxent River, Maryland, is a nutrient-enriched tributary of the Chesapeake Bay. Nutrient inputs from sewage outfalls and non-point sources have grown substantially during the last four decades, and chlorophyll-a levels have increased markedly, with concomitant reductions in water quality and dissolved oxygen concentrations. The Patuxent has gained national attention because it was one of the first river basins in the U.S. for which basin-wide nutrient control standards were developed. These included a reduction in non-point source inputs and a limit on both nitrogen (N) and phosphorus (P) loadings in sewage discharges intended to return the River to 1950s conditions. Full implementation of point source controls occurred by 1994, but population growth and land-use changes continue to increase total nutrient loadings to the River. The present paper provides the perspectives of scientists who participated in studies of the Patuxent River and its estuary over the last three decades, and who interacted with policy-makers as decisions were made to develop a dual nutrient control strategy. Although nutrient control measures have not yet resulted in dramatic increases in water quality, we believe that without

them, more extensive declines in water quality would have occurred. Future reductions will have to come from more effective non-point source controls since future point source loadings will be difficult to further reduce with present technology. Furthermore, changing land use will present a challenge to policy-makers faced with sprawling population growth and accelerated deforestation.

7.1 Introduction

Judged in terms of its size and flow, and in comparison with other tributaries of the Chesapeake Bay, the Patuxent River is quite unremarkable. The Patuxent ranks only seventh in freshwater flow to the Bay proper and is dwarfed in comparison with its nearest neighbor to the south, the Potomac, which itself is considerably smaller than the northernmost and largest of all of the Chesapeake's tributaries, the Susquehanna. In earlier times, there was an appreciable yield of fish and shellfish from the Patuxent River, although its harvest still represented but a small portion of the yield of the full Chesapeake estuarine system (Maryland Department of Natural Resources, Tidewater Administration 1989). Yet, in many ways, the Patuxent looms larger than its better-known counterparts. First, it exemplifies a river and estuary for which there has been a clear public recognition that excessive nutrient enrichment from upstream sources has had substantial effects on living resources in its estuarine reaches. Second, the Patuxent has been a site of particularly active research programs and for which there is one of the longest environmental data records for dissolved oxygen (DO), turbidity, sea grasses, and nutrient concentrations. There are two research institutions on the shores of the Patuxent that have been conducting studies on the river and estuary for decades (Chesapeake Biological Laboratory; CBL and Academy of Natural Science Estuarine Research Center; ANSERC), and there is also a substantial historical record of data that precedes the time of nutrient enrichment and extends through the period of greatest change. Third, the Patuxent has garnered considerable legislative and regulatory attention because of its location adjacent to the nation's Capital; that in turn has affected the role of the Patuxent debate in altering public policy. Finally, the entire basin is within one state, which simplifies, but does not eliminate jurisdictional complications.

During the late 1970s and early 1980s — a time when environmental changes in the Chesapeake gained national attention — the Patuxent was deemed by many as the State of Maryland's most prominent environmental concern. Water clarity decreased, once abundant sea grasses all but disappeared, and commercial fisheries declined. Consequently, the Patuxent became a symbol of a perceived decline in environmental quality and an important driver of public policy for nutrient management in the Bay. After federal and state investments totaling millions to upgrade sewage treatment facilities in the 1980s and 1990s were completed, there were still questions about the policies and practices adopted for the Patuxent: how successful were they in achieving their intended goals?

The following paper reviews four decades of change, 1960 - 2000, in the environmental quality and science-based policy for this river and estuary. Although the full 1960 - 2000 time-frame will be discussed, most of the focus will be on the late 1970s and early 1980s, when fundamental science played a large role in policy decisions. Here the turbulent but important years are documented during which environmental awareness, science and policy were all debated and

formulated together. By focusing on several key policy-related events of the early 1980s, we chronicle fascinating concomitant developments in science and public policy that have enabled better responsiveness to environmental stresses in the region.

While Malone *et al.* (1993) documented key events in public policy that pertained to the Chesapeake Bay, the work reported here amplifies those events that are relevant to the Patuxent. Additional information on public policy and the Chesapeake Bay include Powers (1986), and D'Elia and Sanders (1987).

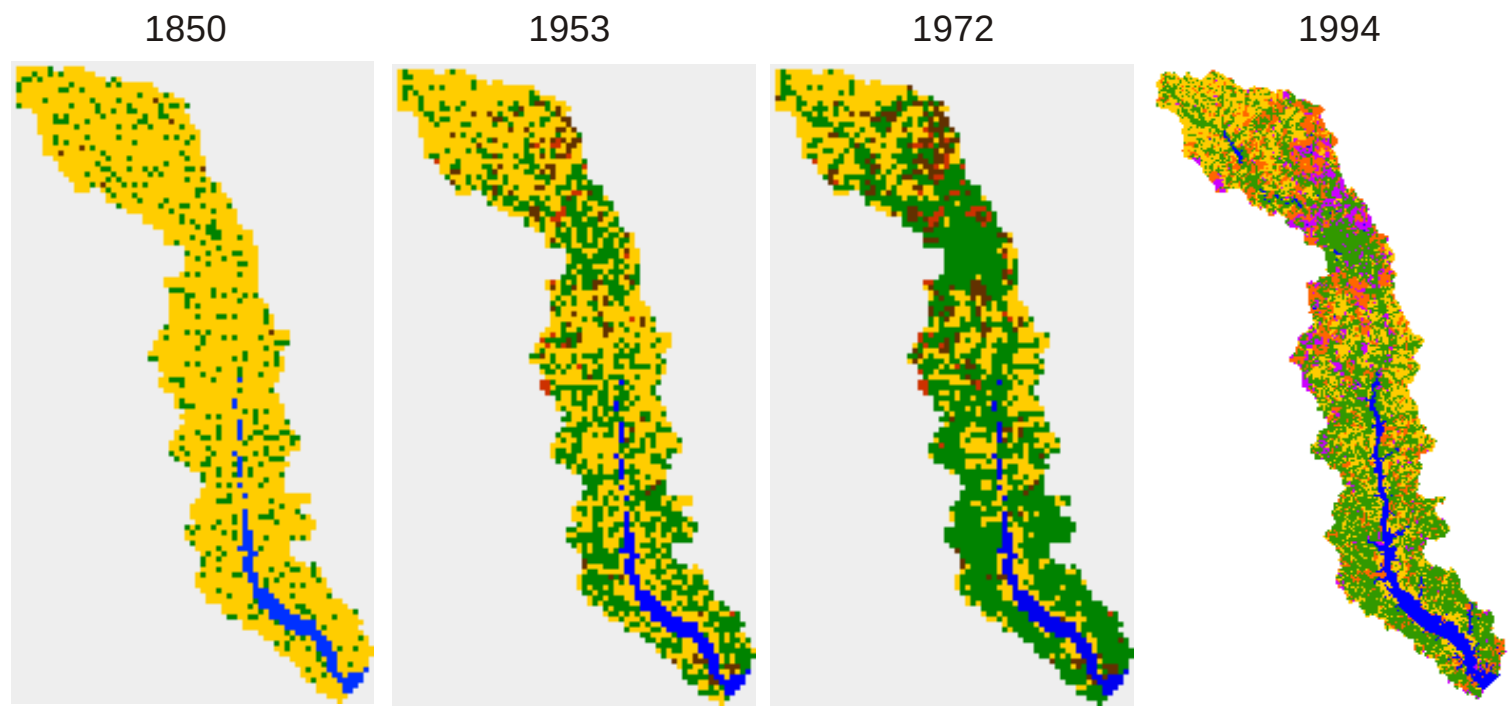
7.2 Demography and Land Use

Demography and land use are without question the driving factors of water quality in the Chesapeake and its tributaries (Year 2020 Panel Report, 1988), particularly because of their influence on nutrient and sediment loading rates. Striking changes in land use have occurred in the Patuxent Basin over the last 150 years (Figure 7-1). In colonial times, as was characteristic of much of the mid-Atlantic and northeastern U.S., the Patuxent watershed underwent widespread deforestation for agricultural purposes. By the middle of the 19th century, over 85% of the forest had been cut down, and the majority of the land was dedicated to agricultural uses (Figure 7-1). Historical records and paleoecological studies indicate very high rates of sedimentation beginning in the early 19th century (Kahn and Brush, 1994).

By the middle of the 20th century, with the demographic shifts from rural areas to industrialized cities, much of the agricultural use had declined. Considerable reforestation occurred during the first half of the 20th century. Development of suburban and rural areas between Baltimore, MD and Washington, DC. was rapid and extensive during the post World War II period. The rural electrification of the 1930s; the rise of the nation's capital as a seat of power and wealth, and as a commercial and cultural center; the development of improved federal and state highway systems; the development of disposable wealth and other factors; all contributed to a greater accessibility to agricultural or undeveloped areas. Together these factors all set the stage for one of the fastest rates of growth that any area of the U.S. has experienced (Culliton *et al.*, 1990). Before the 1960s, when the basin was predominantly rural and agricultural, basin population was less than 25,000. By the early 1960s, population growth began to accelerate at a rate that continues unabated to the present (Figure 7-2).

The population growth of the 1960s required more public infrastructure, and there was considerable focus on developing sewage treatment plant facilities (STPs) in the more populated parts of the basin (Figure 7-3). The links between the two are clear and obvious. Figure 7-2 also shows the rapid growth of sewage treatment effluent that accompanied population growth. In addition to adding new treatment capacity for new housing, existing housing was also being gradually converted from septic service to public sewerage.

During the 1950s, only about 60% of the land was used for agricultural purposes with most of the remainder being forested. However, residential and urban uses increased from negligible (0.3%) in the 1850s to approximately 6% by 1953. The growth of residential and urban centers notwithstanding, by the end of the 1960s, the Patuxent watershed forest achieved nearly 60% coverage (Figure 7-1) because of decreasing agricultural uses.



	1850	1953	1972	1994
Forest	14.3	37.3	57.0	51.6
Residential	0.3	4.6	8.0	15.5
Urban	0.0	1.2	2.3	4.3
Agro	85.5	57.0	32.8	28.5

Figure 7-1. A time-series of land use maps for the Patuxent River basin (1850, 1953, 1972, 1994).
Maps were adapted from Voinov (2001). Values are percents of total area.

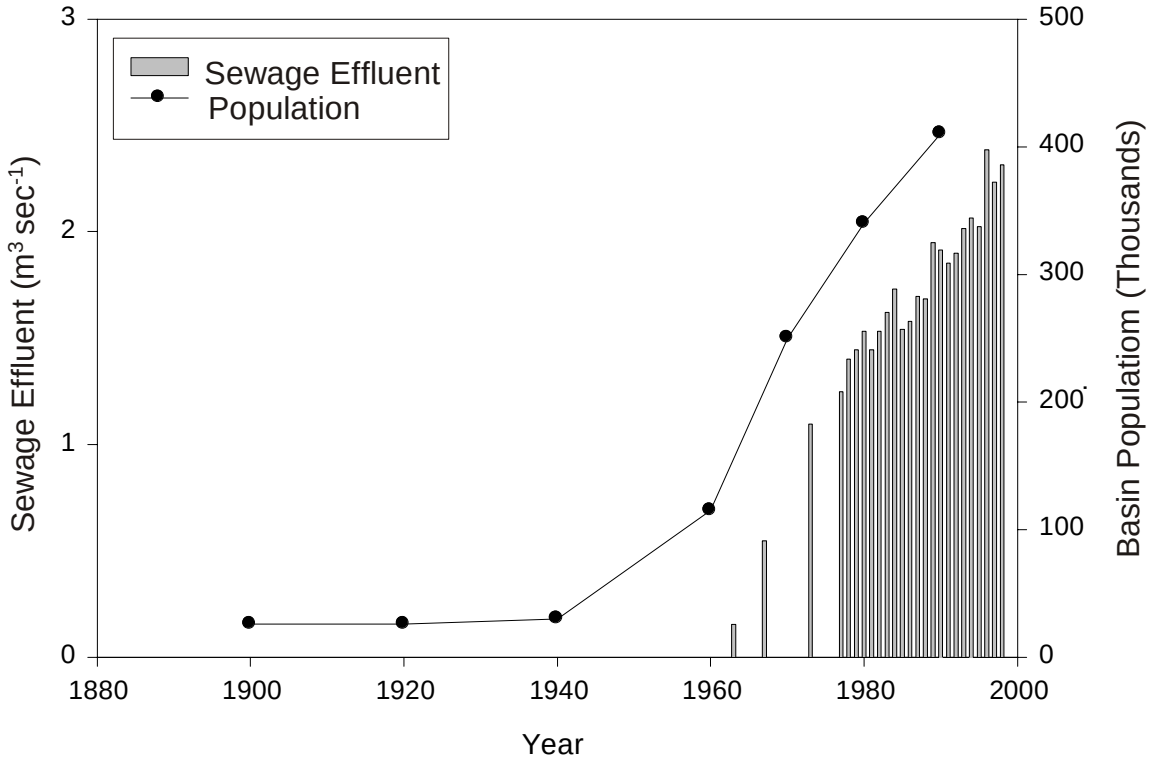


Figure 7-2. Population and sewage effluent trends in the Patuxent River basin (1900 - 1999).
Data are from Maryland Office of Planning (2001).

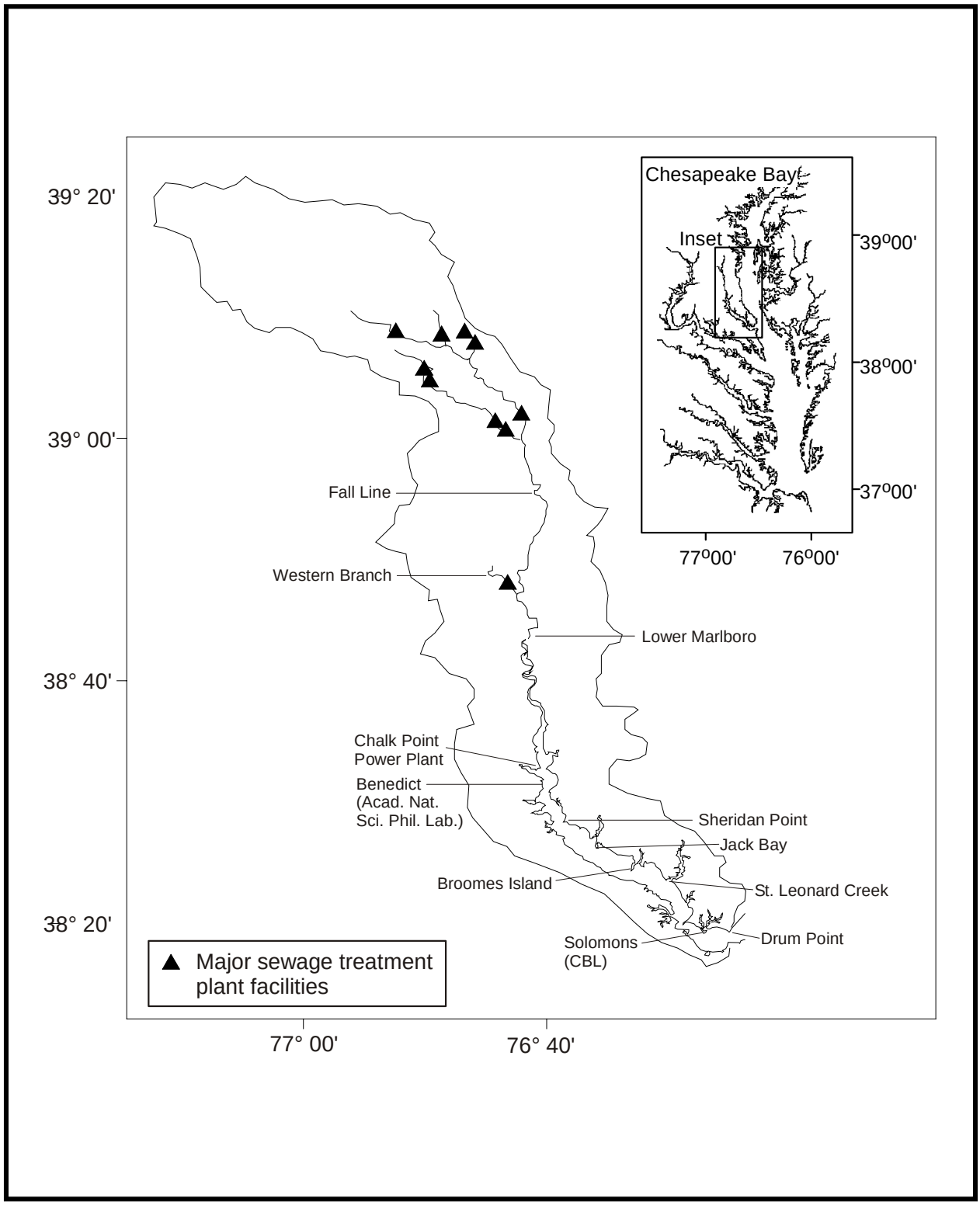


Figure 7-3. A simple map of the Patuxent River drainage basin, river and estuary.

Sites mentioned in the text are labeled on this diagram. Major STP locations are also indicated.

Since non-point sources (NPS) of nutrients are higher in clear-cut forested and agricultural settings, land use undoubtedly had an effect on nutrient inputs to the river prior to the 20th century, in comparison with the environmental conditions prior to European colonization. However, given that fertilizer applications were not substantial until the latter half of the 20th century, agricultural nutrient inputs from runoff prior to 1900 are not believed to be large in comparison with present day practices, and from the work of Brush (1984a, 1984b) it appears that river and estuarine areas did not exhibit nutrient-enriched characteristics.

Rapid rates of population growth and changing land use have continued into the 1990s. The first half of the 20th century was characterized by a relatively constant population below 25,000, followed by abrupt changes that began in the 1960s yielding a current basin population of about 536,000 inhabitants. This represents a twentyfold increase in population in just four decades. Although agricultural use of land continued to decline, forestation peaked in the early 1980s and has been declining since. By 1994, only 50% of the land area was forested, while there was a striking increase in residential and urban uses. In just two decades the Patuxent basin has become almost 20 percent residential and urban. The counties between, and to the south of Washington, DC and Baltimore, MD were becoming bedroom communities for those cities. These changes all affected nutrient loading to the Patuxent River and estuary.

The so-called "2020 Report" (Year 2020 Panel Report, 1988) to the Chesapeake Executive Council was a landmark assessment of the demographic problems facing the Bay and its watershed, and underscored the importance of land use changes. Any statements applied broadly to the entire Chesapeake watershed when this report was published applied all the more to the Patuxent Basin which was experiencing even faster population growth and land use change. According to the report, "The entire Chesapeake basin population grew almost 50% between 1950 and 1980 [while] the amount of land used for residential and commercial purposes increased 180%." Given that the Bay's watershed constituted one of the fastest growing areas of the nation during 1950 - 1980, the Patuxent's problems were further amplified. According to the report, "undeveloped land has been converted to developed land at a rate that exceeds the rate of population growth." The report was clear in its diagnosis of the cause of the Bay's maladies in 1988, "More than any single development factor, we were concerned with low density sprawl." It was also blunt in its prescription for a cure, "States must take the lead to establish and implement policies and programs that result in compact and efficient growth patterns."

7.3 The Role of Science and Public Awareness

Prior to the 1960s, Bay watermen and others felt that the Chesapeake and its tributaries offered limitless bounty in fish and shellfish harvests. There was little public recognition of over-harvesting or water quality changes although there were squabbles over whose right it was to partake of estuarine production. Indeed, even as late as the 1970s, natural resource management officials were still reinforcing the perception that all was well (*e.g.*, Chesapeake Research Consortium, 1977).

Despite a lack of public awareness during the early 1960s of limits to Bay production or assimilative capacity for increasing nutrient loads, scientific interest in the biology and ecology

of key Bay species fortunately resulted in the development of several modest research laboratories. These facilities have played important roles in providing the information that has raised public concerns about the Bay's condition. The first of these facilities, CBL, located near the mouth of the Patuxent River, was established in the mid-1920s by Dr. Reginald V. Truitt, a faculty member of the University of Maryland. CBL scientists undertook many of the earliest published studies on estuarine nutrient and oxygen dynamics (*e.g.*, Newcombe and Brust, 1940; Nash 1947) that now provide important baseline information about the Bay prior to the 1960s when larger, publicly funded, monitoring and research programs began to be established. Given CBL's location at the mouth of the Patuxent River, there was particular emphasis on the study of the Patuxent itself, some of which was published but most of which remained as unpublished data in laboratory and field notebooks of researchers.

Fortuitously, many of the original notebooks were preserved in boxes in the library attic of CBL. In the early 1970s, CBL Professor Donald R. Heinle recognized the value of these unpublished historical data in establishing a scientifically documented case for environmental change in the Chesapeake system. Heinle began to summarize this information. He found that turbidity levels had increased markedly during the period 1940 - 1970. Concomitant with that change were alarming decreases in sea grass distribution and abundance, decreases in deep water oxygen concentrations and an increase in nutrient levels, all of which Heinle deemed to be signs of nutrient enrichment. With this information in hand, he approached Dr. L. Eugene Cronin, former CBL Director, and then Director of the Chesapeake Research Consortium in Annapolis, MD. Cronin, himself a longtime student of the Bay, had his own concerns about declining water quality in the Bay. Cronin, in turn, briefed U.S. Senator Charles McC. Mathias, who arranged for federal appropriations to be made to undertake the first studies under the aegis of the USEPA Chesapeake Bay Program. This funding enabled Heinle *et al.* (1980) to conduct a major review of historical data for the entire Bay system that established a sound scientific case that water quality had declined prior to, and during the 1970s.

In the early 1960s, ANSERC, which had considerable interest in the effects of power plant siting and operation on estuaries, also established a laboratory on the Patuxent River estuary at Benedict, MD. Through its studies of the Chalk Point Power Plant at Benedict, Maryland (Figure 7-3), as well as studies of living resources in the river, this laboratory also contributed to an unusually rich historical record. This information proved to be enormously important in determining the extent of change that occurred in the post-1960 era. ANSERC moved to a new facility further down the estuary in the 1990s and staff continues to examine estuarine questions.

7.4 1930s to 1980s: Change and Recognition

Our understanding of the changes in water quality in the river and estuary from the mid-1930s to the mid-1960s is based largely on less than complete evidence. Nonetheless, it is generally agreed that significant changes in the trophic structure and water quality of the Patuxent began to change markedly during this period. By far the best information is from the synthesis study by Heinle *et al.* (1980). Of all the data available to these authors, none were as reliable as a time series of Secchi depth. Although Secchi depth becomes less precise in very turbid conditions, particularly below penetration depths less than 1 m, or when particle sizes change with time in the water, Secchi measurements enable discrimination of turbidity levels due to phytoplankton concentrations in waters such as in the lower Patuxent estuary. Thus, most of the lower 30 km of

the river could be readily evaluated for differences in measurements taken from the 1930s through the late 1970s.

The Heinle *et al.* (1980) Secchi depth data are displayed along with more recent obsevations, as a function of salinity, in Figure 7-4. The authors concluded that the lower Patuxent estuary was substantially more turbid in the 1970s that it had been 40 years before. There appear to be two possible explanations for this pattern. The first is that loadings of inert particulates had either increased, or particle size had decreased substantially and this resulted in increased turbidity. Such loading changes presumably could have been from diffuse source increases due to land-use changes, particularly in the upper portion of the basin. The other alternative, which seemed far more plausible to Heinle *et al.* (1980), was that phytoplankton concentrations in the river had increased substantially over that time. Two lines of evidence support such a hypothesis: (1) increases in nutrient concentrations lead to increases in phytoplankton production and biomass, (2) phytoplankton production increased secondary production or, more likely, sank to deeper waters and decayed and, because of microbial decomposition, oxygen concentrations were depleted. Indeed, phosphate concentrations increased in both the riverine and estuarine portions of the Patuxent from the late-1930s to the mid-1970s (Figure 7-5). Furthermore, DO concentrations at depth were substantially lower in the 1970s than recorded four decades earlier (Figure 7-6).

While evidence accumulated that the trophic status of the Patuxent was changing rapidly by the 1970s, it was also becoming clear that living resources were being affected (Heinle *et al.*, 1980). In the Patuxent, probably the best overall proxies for the state of living resources are sea grass densities (discussed elsewhere in this volume) and oyster harvest, the latter being something that is well known from Department of Natural Resources (DNR) records. In addition, since the 1960s there have been studies conducted in the Patuxent and elsewhere to determine the condition of oysters in the estuary (*e.g.*, Roosenburg, 1969; Riedel *et al.*, 1995). By the late 1970s, virtually everyone recognized the critical condition of the oyster fishery in the estuary. Not only had harvests decreased dramatically, but there was also evidence that oyster diseases, particularly “Dermo,” were increasing in the oysters left unharvested. The end of the 1970s saw considerable uncertainty about the proximate causes of the changes that were now being scientifically documented. The State of Maryland and the USEPA at times seemed oblivious that any change was occurring (Chesapeake Research Consortium, 1977). However, by the end of the 1970s, after a successful lawsuit by the three Southern Maryland counties against the more northern counties, the State and USEPA, acknowledged that change had occurred. A great debate emerged at this time throughout the Bay whether commercial over-harvesting, shellfish disease, increased sewage loading and declining water quality, or a combination of these were responsible for the irrefutable bad news about the status of the oyster fishery. There was much finger pointing, but little action on any front.

With the advent of powerful computers and simulation software, numerical modeling of water bodies gained considerable credibility and attention in the early 1970s. The newness of this space-age technology, as well as the apparent unambiguity and decisiveness of the results produced by models, led many policy makers to accept the results without question. By present standards, the computers used and the two-dimensional, steady-state models developed were unsophisticated relative to the task (HydroQual, 1981). Even more significant though, the

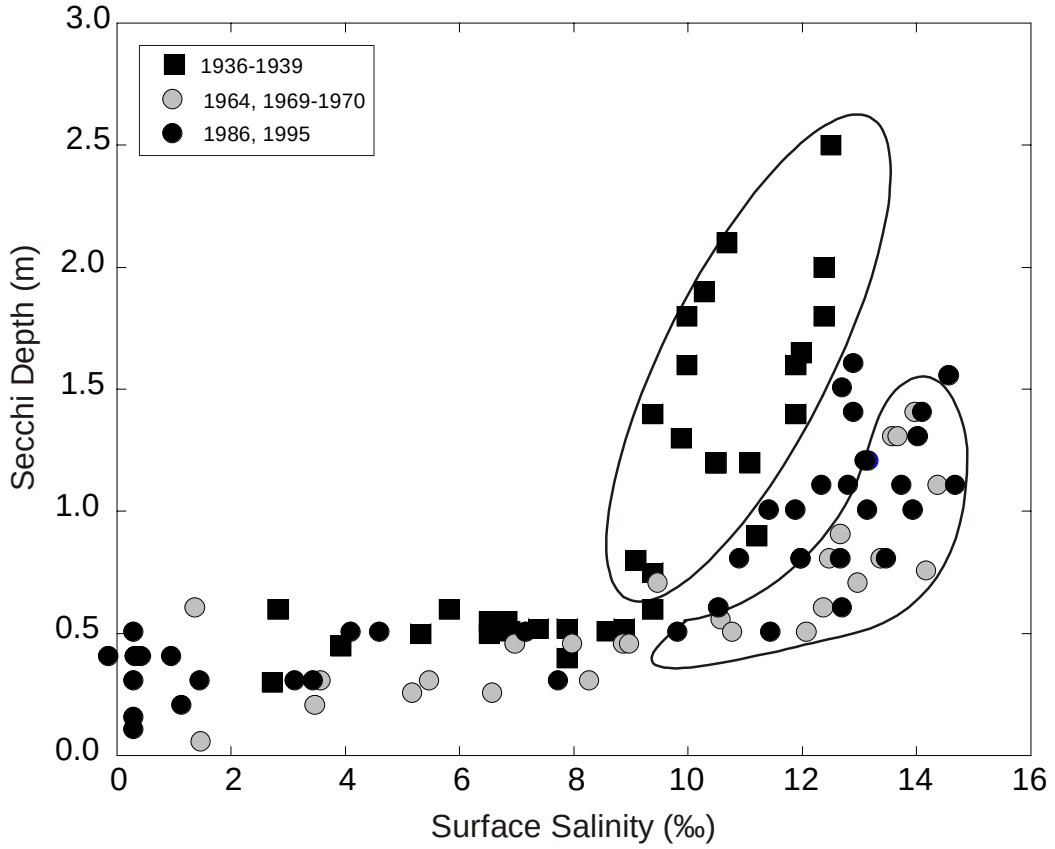


Figure 7-4. Summer (July) Secchi depth measurements from data collected between 1936 and 1995 displayed as a function of salinity for the oligohaline and mesohaline regions of the Patuxent River estuary.

Encircled data indicate Secchi depths in the early and more recent periods in the mesohaline portion of the estuary. This figure was adapted from Heinle et al. (1980). Data from the 1985 - 1995 period are from the Maryland Department of Natural Resources (2001).

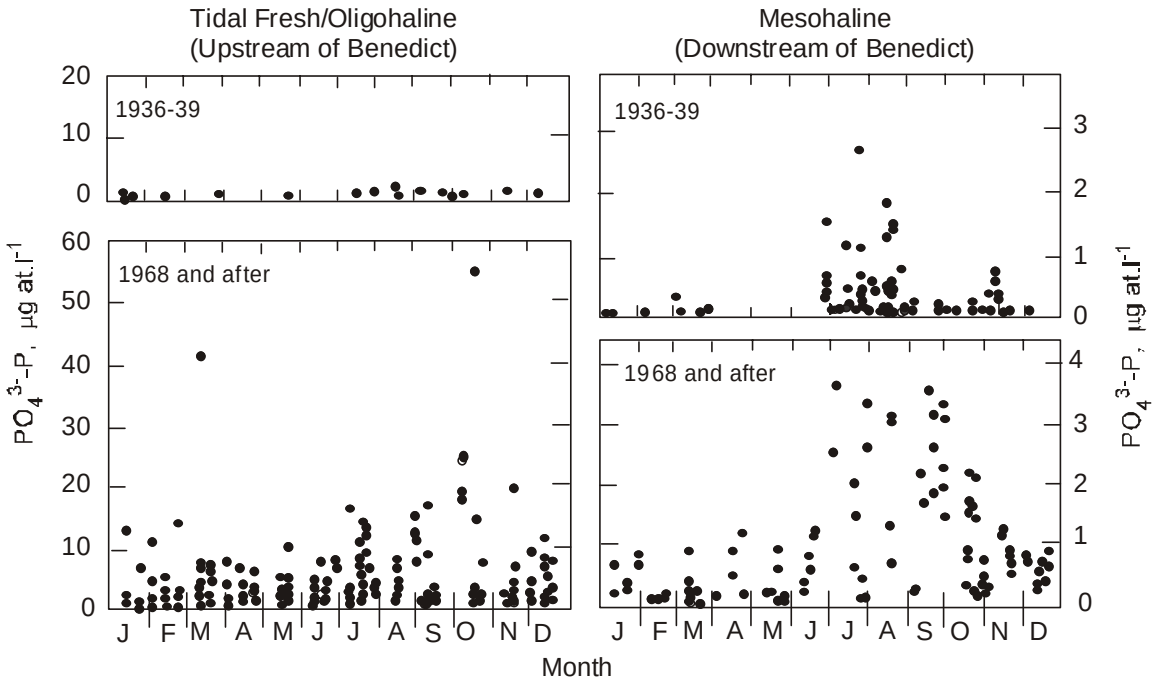


Figure 7-5. Dissolved inorganic phosphorus concentrations from two time periods (1936 - 1939; and after 1968) and two regions (tidal fresh/oligohaline areas upstream of Benedict, MD and mesohaline area downstream of Benedict, MD) of the Patuxent River estuary.

Note different scales of left and right panels. This figure was redrafted from Heinle et al. (1980).

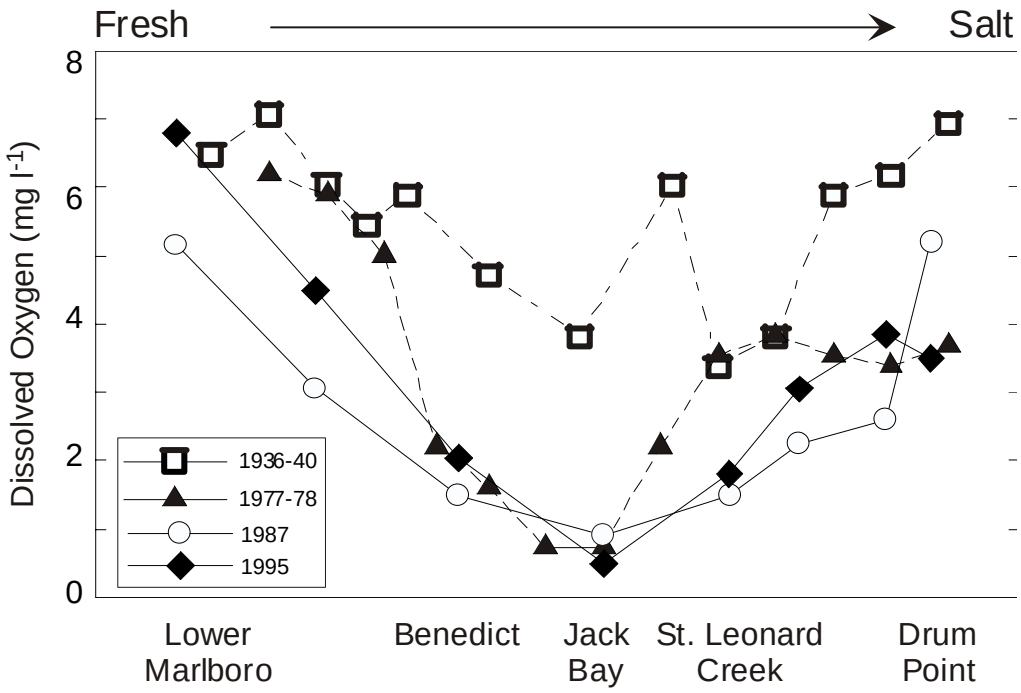


Figure 7-6. Summer (July) dissolved oxygen concentrations in bottom waters of the Patuxent River estuary from tidal fresh regions (Lower Marlboro) to the junction of the estuary with Chesapeake Bay (Drum Point).

The distance between the upper and lower stations is about 50 km. Data from 1936 - 1940 and 1977 - 1978 are from Heinle et al. (1980); data from 1987 and 1995 are from Maryland Department of Natural Resources (2001). Heinle et al. (1980) found about 77 summer season bottom DO measurements from the 1936 - 1940 period. Additionally , salinity values for the 1936 - 1940 period were within the range of more contemporary values indicating that the DO measurements observed during 1936 - 1940 were not the result of extreme hydrologic events. Locations are shown in Figure 7-3.

conceptual understanding behind the models was incomplete or flawed. For example, in the 1970s some of the most debated causes of declining water quality related to the role of the benthos, the relative roles of point and non-point nutrient sources to rivers and estuaries, the relative importance of N and P as limiting nutrients in rivers and estuaries, and the influence of trophic structure on the expression of nutrient enrichment. None of these conceptual, scientific issues were close to resolution in the 1970s, and they were not incorporated into the computer models of the day. The result was that the computer models were seriously flawed and incapable of providing adequate projections of water quality under different management alternatives.

Although funding was available to study these conceptual issues, it was not sufficient and was rarely supplied by the agencies responsible for making key management decisions. "Creative use" of funding was necessary to enable fundamental studies of estuarine processes: for example, one of the authors (WRB) initiated studies of benthic oxygen demand and nutrient regeneration in estuaries using funding directed at power plant impacts because no other agency, federal or state, had interest in supporting these studies. The other authors used funding from county government to pursue issues related to nutrient limitation.

Environmental awareness grew in the 1960s and with it developed a national interest in passing landmark federal legislation in the early 1970s that would have broad implications in the study, understanding and policy development for the Patuxent Basin. Undoubtedly the most relevant of this legislation was the 1972 Federal Water Pollution Control Act (PL 92-500), later referred to as the Clean Water Act (CWA). Several provisions of this legislation had particular relevance, namely §208, which mandated the formation of basin-wide wastewater treatment management, and §316, which dealt with power plant impacts. Although the original legislation was overly ambitious and not easily implemented, in our view, these provisions did result in increased scrutiny of the Patuxent's degrading water quality and led ultimately to research that identified nutrient enrichment as the leading cause for concern.

As chair of the Technical Advisory Committee advising the State of Maryland in the development of the §208 Basin Plan in the late 1970s, Dr. Donald Heinle had a unique opportunity to work with Maryland DNR officials as they determined the course of action they would take for wastewater management in the basin (the 1972 legislation did not provide for non-point source management). He was aware of the work of Ryther and Dunstan (1972) and others that suggested that the role of N as a limiting nutrient deserved more attention; this was one of the key breakthroughs in understanding coastal and estuarine eutrophication. Yet the State of Maryland's plan called only for P control. Dr. Heinle's suggestion that N removal be considered through advanced wastewater treatment processes met with considerable resistance from both DNR and USEPA, which were at the time committed to P removal nationwide.

In 1976, the three counties in Southern Maryland were increasingly aware of the rapidly increasing sewage load to the river, declining water quality, loss of sea grasses, and decreasing oyster and fish harvests. Ultimately, the southern counties sued the upriver counties, the State of Maryland and the USEPA to prevent further development and wastewater discharges. Taken together, the controversy over rapid increase in sewage discharges and concern over the State's failure to provide for nitrogen removal and the lawsuit led to very acrimonious relationships

among the regulatory agencies, citizen advocacy organizations, the Southern Maryland counties and the scientific community.

7.5 1980s: Diagnosis and Prescription

Monitoring data from the 1980s continued to show severe declines in water quality and living resources in some areas and continued poor conditions in others. For the Patuxent, there was the grim picture of a disappearing oyster industry and the near-elimination of sea grasses as the waters were becoming increasingly turbid and unattractive (Stankelis *et al.* this volume, Chapter 6).

Ryther and Dunstan's (1972) seminal paper led other scientists to investigate the possibility that N was an important limiting nutrient in other coastal areas. While Heinle was the first to propose this for the Patuxent River, there was no funding to support studies to examine the issue in more detail; he and his colleagues had to use indirect evidence to support their contention (*i.e.*, examination of N:P ratios and nutrient concentrations). In contrast, the computer models of the 1970s had led the State and USEPA to believe that P was the single limiting nutrient and if controlled in sewage effluent, would lead to substantial improvements in water quality. Indeed, the models were constructed to make this inevitable under most scenarios. Sanitary engineers emphasized that if a nutrient element, either N or P, were to be removed via an advanced wastewater treatment (AWT) process, it should be P, because it was much less expensive and easier to do. Academic scientists were skeptical that the role of N could be so easily dismissed — it appeared that the underlying factor driving policy towards P removal was lower cost.

In 1980, Maryland Governor Harry Hughes recognized the controversial role of the DNR in Patuxent decision making and he determined that authority to deal with water quality issues should rest instead in the newly created Office of Environmental Programs in the Maryland Department of Health and Mental Hygiene (DHMH). Governor Hughes hired a young lawyer, William E. Eichbaum, as assistant secretary to head this newly formed department, which was staffed primarily with State of Maryland sanitary engineers familiar with Patuxent water quality issues and with biases towards the P removal position adopted by DNR. The reorganization was a pivotal event and the selection of Eichbaum had a profound effect on the future of sewage treatment decision-making for this river and estuary and ultimately, we think, on national policy. One of Eichbaum's first acts was to tour the Patuxent Basin and meet with public officials, scientists, and interested citizens to gain a clear perspective of the problem. Eichbaum made a concerted effort to understand the complex scientific issues involved, and he quickly realized that there would be endless litigation unless he could somehow bring the parties together — including several estuarine scientists — to develop a compromise. Accordingly, he brought 30 key players together for three intensive days (December 2-4, 1981) of closed discussion and negotiation in a professionally mediated format. Two of the authors (CFD and WRB) of this paper attended the "Charette," and, as it turned out, played useful roles in the outcome of the negotiations, particularly as they related to the issue of N loading rates.

The Charette would turn out to be a remarkable event in the environmental history of the Patuxent, and indeed the Chesapeake itself, for several reasons. First, it marked the first time

that a consensus was reached on nutrient inputs for a tributary based on desired water quality outcomes and defined nutrient inputs. This implied that a clear linkage exists between the two and presaged the Total Maximum Daily Load (TMDL) approach now mandated throughout the Chesapeake and the US. Second, it was the first recognition by public officials from state and county agencies that N was indeed a key problem to be resolved. Third, it involved the acceptance by the State of Maryland of its own increased responsibility to meet the needs of AWT without federal cost sharing.

During the first day of the Charette, there were presentations by technical people. One of the key understandings Eichbaum had with all Charette attendees was that both “sides” needed to recognize and state clearly their positions and expectations for successful negotiations. Accordingly, officials from Howard County, who had instituted state-of-the-art chemical P removal, insisted that they had done their part already and that they should not be expected to do more. Officials of the three southern counties, represented by the Tri-County Council for Southern Maryland, took the position, given the scientific perspectives held by estuarine scientists involved, that N removal was required to make substantial improvement in the lower, estuarine part of the river. State Senator C. Bernard Fowler, the leader of the Southern Maryland delegation, was particularly influential in that he was able to recount his personal experiences of the striking decline of water quality in the Patuxent. Wading waist deep in the river at Broome’s Island during the 1950s, a time during which large harvests of fish and shellfish were common in this section of the river, Fowler could see his feet. "That's no longer the case," he said. He started an annual event in which citizens wade into the water to see if there is any change in water clarity (Horton 1993). Agreement was eventually reached on a key goal for the Patuxent of returning water quality conditions to those common during the 1950s.

The scientists present emphasized the value of establishing a clear definition of what would be achieved under conditions where nutrient loads were returned to the values of the 1950s. They recommended the designation of the Broome’s Island/Sheridan Point areas as key areas to be expected to respond favorably to lower nutrient loadings. This section of the river had been, by the end of the 1970s, the location where chlorophyll-a values exceeded 50 $\mu g\ l^{-1}$ and where oxygen concentrations in sub-pycnoclinal waters were very hypoxic for four or more months each year. Figure 7-6. shows the historical changes in dissolved oxygen in the river at selected locations. Sheridan Point is very near Jack Bay, and the figure clearly indicates the profound changes in DO concentration observed in this area.

Another clear outcome of Charette discussions was a consensus to focus on DO and chlorophyll-a concentrations as proxies for water quality and ecosystem health. While these parameters are commonly used as proxies today, public officials and interested citizens, who at the time were new to the scientific concepts of nutrient enrichment, did not know what indicators were best for measuring improvements in the Patuxent. Recognition that chlorophyll-a and DO concentrations were good choices for this was an important development in the history of the much larger Chesapeake Bay Program water quality monitoring effort.

Although it was not common practice at the time, the Charette recommended that the State of Maryland establish a loading standard for the Patuxent, similar to the TMDL approach used today. Effluent standards at that time were strictly concentration based. The decision was made

to establish a loading limit that approximated that of the 1950s, which reflected the environmental quality targeted for the entire river. The original nutrient load estimates suggested that the 1950s N and P loads were approximately 40% of those in 1980, a percentage very similar to that adapted Bay-wide in 1987. While these estimates were weak, in some instances, they were adopted as a goal.

Ironically Heinle left the State before the Charette, but he had played a significant role prior to the Charette at raising awareness about the probable controlling role of N as a limiting nutrient in the estuary, particularly the need to control N inputs in sewage effluents. However, no studies had been conducted locally to verify such a role for N. Moreover, federal policy at the time was very much opposed to undertaking either N removal or a dual nutrient control strategy. Accordingly, any statewide action to remove N from effluent would put federal cost sharing at risk. State and county authorities would have to bear the costs, which for N were substantial, because the primary option available was a chemically based (methanol) denitrification process that was very expensive in terms of implementation, operation, and management.

It is hardly surprising that neither the State nor the upstream counties were in favor of N removal, given the daunting cost considerations they were facing. Nonetheless the Southern Maryland delegation remained staunch in its position that N removal must be implemented. The negotiations appeared to be in serious jeopardy, almost to the end of the Charette, when Washington Suburban Sanitary Commission (WSSC) head General McGeary, made a major concession that soon after appeared to have cost him his job. In short, he approved the conversion of the Western Branch STP to advanced wastewater treatment to remove N to the 5 mg l^{-1} level (pre-treatment concentrations ~ 20 mg l^{-1}).

Unfortunately, although the accord reached at the Patuxent Charette appeared to have represented a resolution including N control of STP effluents, both the State of Maryland and USEPA were reluctant to accept this agreement. They resisted further attempts by citizen activists to gain wider acceptance of N controls and continued to criticize the hypothesis that N was the growth-limiting nutrient element in the estuary. This controversy precipitated an interesting 1983 legal challenge by Calvert County citizen activist, William Johnston, that resulted in an administrative law hearing on the N issue in the Patuxent. Johnston called 8 scientists as expert witnesses, including the authors of the present paper. His contention was that removing N would result in reduction in chlorophyll-a levels and concomitant improvement of bottom-water DO concentrations. Instead the State argued that removing N would simply promote N-fixation, which would nullify any improvement. Moreover, the State contended that it would be possible to remove P to such a low level that it would become limiting, in effect ignoring the contribution of sediment-derived P which was known to be important (Boynton *et al.*, 1980).

Although USEPA seemed unwilling to fund experimentation on N enrichment effects — and by doing so open the possibility that N removal would be substantiated — we were able to obtain funding to undertake such studies. The authors of the present paper proposed a series of mesocosm experiments to examine factors affecting phytoplankton growth and species composition in the river (D'Elia *et al.*, 1986; Sanders *et al.*, 1987). This experimental system had originally been developed to examine toxicant effects, but it was easily adapted to nutrient-

enrichment experiments. Over objections of his staff, Eichbaum agreed to fund part of the microcosm work, which was also funded by FMC and Procter and Gamble, which had vested interests in the N versus P controversy because of their production of phosphate-based detergents and fertilizers, and by Maryland Sea Grant development funds and Calvert County, MD.

Industry funding can be controversial so a precondition of accepting the funding was that there could be no special demands placed on the investigators, who were free to publish results in the open literature (*e.g.*, Sanders *et al.* 1987), and that OEP would review experimental protocols. A special feature of this effort was that we were able to bring together a spectrum of funding sources (local, state, federal and industry), in fact, the very groups who were on the opposite sides of legal and legislative issues. This was indeed a remarkable example of science being brought to bear on a contentious legal and public policy issue.

The mesocosm experiments developed cogent evidence that N had a very strong role in promoting phytoplankton growth during the spring, summer and fall seasons and responses to P-enrichment were weak and restricted to the winter (Figure 7-7). Despite this strong scientific evidence to support the case for N removal, agency resistance to implementing it remained strong. Two breakthrough events occurred that caused reconsideration of both the federal and state opposition to N removal. First, the evolving sewage treatment technology developed a cost-effective alternative for N removal. The new Biological Nutrient Removal (BNR) process, championed by VPI scientist and engineer Clifford Randall, took advantage of the organic component of primary sewage liquor and the high B.O.D. in the secondary process to remove N through denitrification. In addition to its cost effectiveness from the operation and maintenance standpoint, this process required relatively little upgrading of existing facilities. Second, as Malone *et al.* (1993) have noted, another substantial breakthrough in the acceptance of the N hypothesis occurred in 1984 when the USEPA Chesapeake Bay Program Scientific and Technical Advisory Committee (STAC) strongly endorsed N control. Taken together, these two actions caused a rapid change of both the USEPA (U.S. EPA 1986) and State position on N control of sewage.

7.6 1990s: Action, Implementation and Results

By the 1990s extensive monitoring and research data were available to allow for construction of comprehensive N and P budgets for the Patuxent, including a forty-year record of changes in nutrient inputs from point and non-point sources at the fall line (Figure 7-8).

Available information concerning nutrient additions to the Patuxent River estuary, including those from point and diffuse sources and from atmospheric deposition of N and P compounds directly to surface waters of the estuary, are summarized in Table 1. The geographic location of sources according to position either above or below the Patuxent fall line are also indicated. Finally, two evaluations of point source loads, the first averaged for a five-year period (1985 - 1990) prior to the institution of BNR at STPs and for a five-year period (1993 - 1998) after BNR was established. Several interesting points emerged. First, N and P loading to the Patuxent estuary averaged about 16.0 and 1.1 g m^{-2} yr^{-1}, respectively. Relative to other estuarine systems

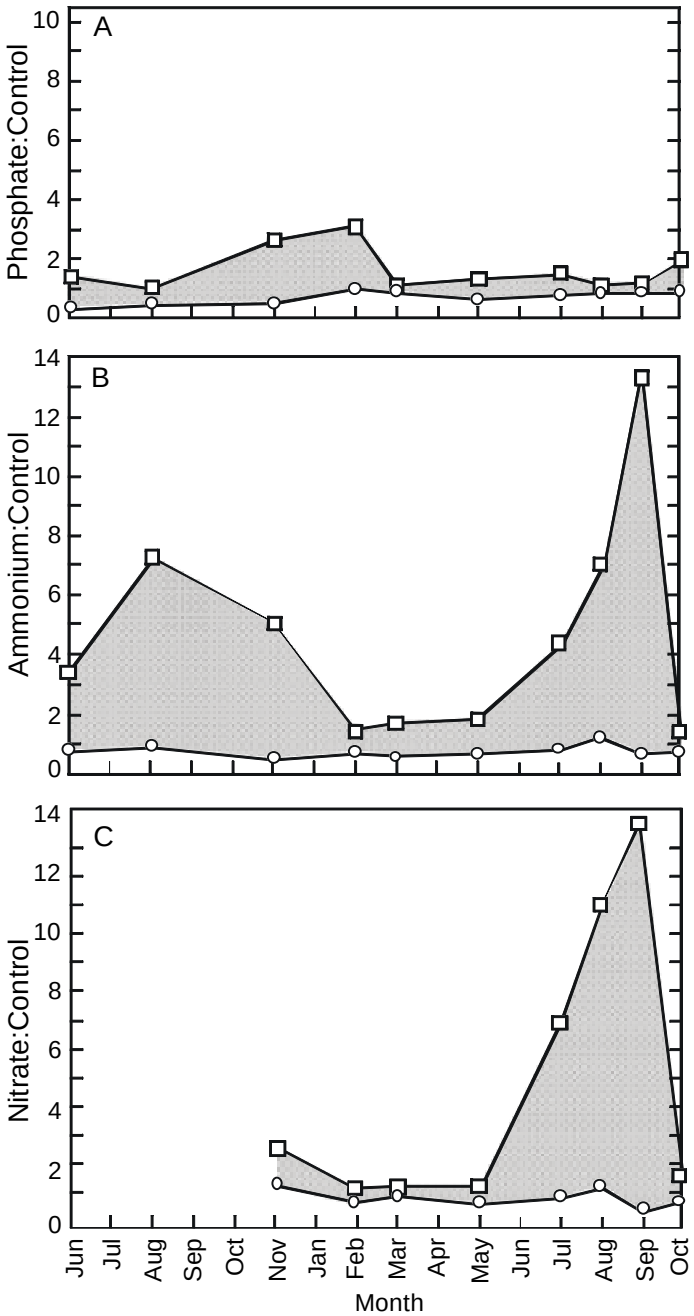


Figure 7-7. Range of ratios of Relative Fluorescence Units (RFU — a proxy for chlorophyll-*a* concentration) of triplicate nutrient-enriched experimental mesocosms (1 m^3) to mean of triplicate unenriched control mesocosms during each experiment from June, 1983 through October, 1984.

(A) phosphate-enriched to control; (B) ammonium-enriched to control; (C) nitrate-enriched to control (nitrate enrichments were not done prior to November 1983). The greater the darkened portions of each panel, the greater the nutrient enrichment potential for a given nutrient. This figure was adapted from D'Elia et al. (1986).

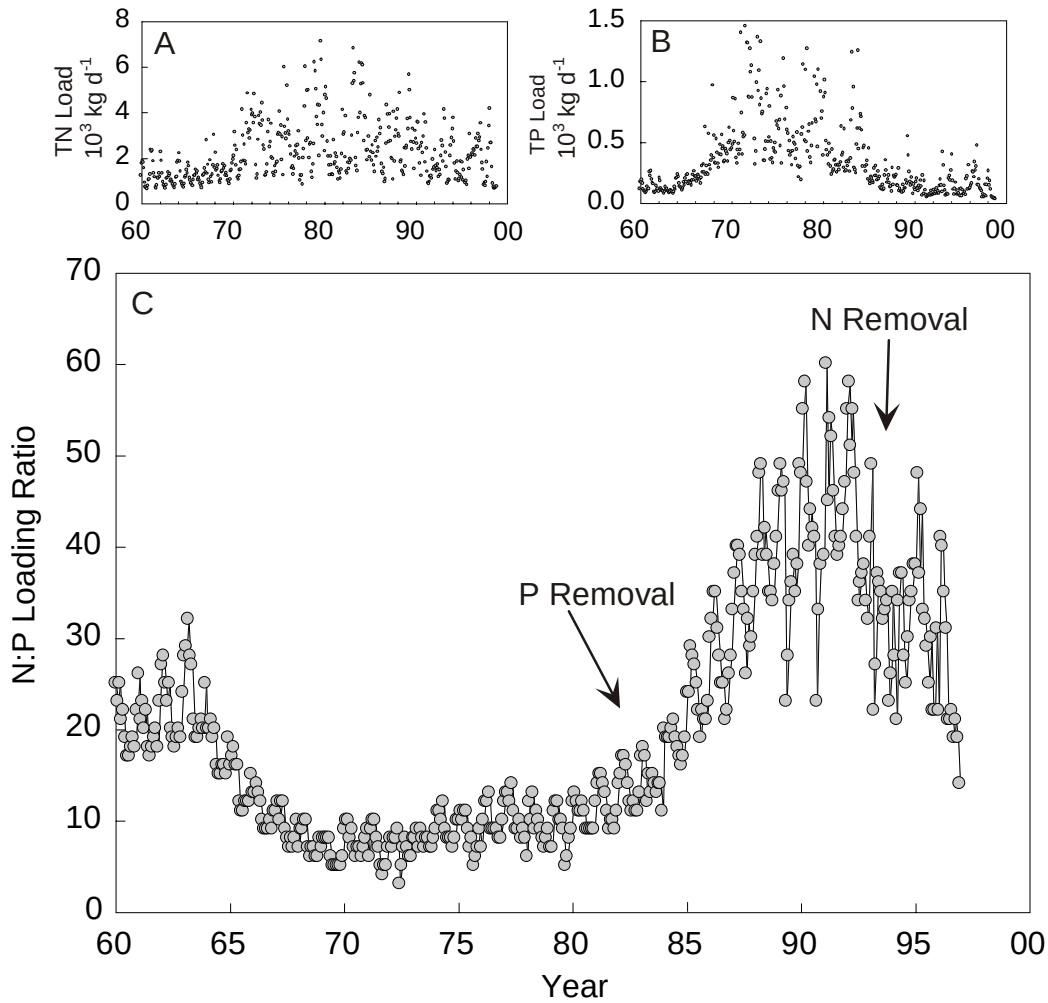


Figure 7-8. A time series of average monthly TN (A) and TP (B) loads at the fall line of the Patuxent River estuary from 1960 to 1999 and monthly average TN:TP ratios (molar basis) (C) for nutrient inputs at the fall line of the Patuxent River estuary from 1960 to 1999.

Loads from 1960 to 1977 were estimated by Hagy et al. (1998) and loads from 1978 to 1999 are from Langland et al. (2001).

in the USA and other areas of the world, these are moderate input rates. Systems such as Baltimore, Boston and Tokyo Harbors, and the Potomac River estuary are far more heavily loaded with N and P, while many coastal lagoons and large enclosed seas, such as the Baltic, are less loaded than the Patuxent (Boynton *et al*., 1995). So, the Patuxent is not an extreme case, with respect to loading rates, though there have been, and continue to be, clear signs of the negative effects of nutrient enrichment with regard to chlorophyll-a and DO. Secondly, the Patuxent has been viewed by environmental management agencies as a "point-source dominated" ecosystem. As indicated in Table 7-1, point sources played an important role prior to BNR, representing about 38 % of the TN load, but even during this period, point sources of TN did not dominate N loads. During the post-BNR period, point sources contributed about 20% of the load. Large reductions have been made in point source inputs but future significant reductions in the Patuxent, and in most Chesapeake Bay tributaries, will need to focus on the more difficult to control diffuse and atmospheric sources of N (Boynton *et al.*, 1995).

Fall line N and P load data also show the striking effects of land use, demographic change and sewage control strategies (Hagy *et al.* 1998; Figure 7-8.b). Clearly reflected is the effect of P removal from sewage in the early 1980s followed by the P ban in detergents in the mid-1980s (Figure 7-8.b). Trends in TN loading were not as dramatic and occurred later because N removal from sewage was not fully implemented at the larger treatment plants until 1993 (Figure 7-8.a).

The nutrient budget above the fall line is dramatically affected by sewage inputs, which are a major part of the allochthonous nutrient inputs. This conclusion, which is illustrated by the N:P loading ratios derived from the data shown in Figures 7-8.a and 7-8.b, also demonstrates the profound effect that the sewage treatment processes had on the nature of the residual loads. As the input of untreated, P-rich sewage increased in the 1960s, N:P ratios dropped substantially, well below the Redfield ratios of 16:1 (Figure 7-8.c). These ratios can be misleading, since effective N:P delivery rates of allochthonous nutrients to plankton in the water column depend on seasonal differences in benthic exchanges and other processes (D'Elia *et al.* 1986). There was a dramatic reduction (~4 times) in TP loads following modifications of STPs to tertiary levels and the implementation of standards for low-P detergents. The N:P ratios increased to achieve a maximum of approximately 45:1, indicating an N-rich nutrient source. TN loads were also reduced in an even more dramatic fashion on a seasonal basis (~6x) following adoption of BNR technologies. As N decreased, so did N:P ratios and by the mid-1990s, they were approximately at the Redfield ratio, similar to the ratio observed some 40 years earlier.

However, inasmuch as N is currently removed only during the warmer portions of the year (N-removal is temperature dependent), cool season loads are tending upward in response to increasing flows from treatment plants (Figures 7-9.a, and 7-9.c). Increasing discharges will erode progress made in controlling these point source inputs. If flows continue to increase, average annual TN loads in just a few more years will be about the same as they were in 1985, prior to initiation of major efforts to control point source discharges.

Although the effects of both point source P and N removal clearly affect the nutrient loading rates and the N:P ratio, unequivocal evidence of recovery in terms of improved water clarity, elimination of algal blooms, large decreases in hypoxic water volumes or general recovery of sea grasses has yet to be recorded, especially in the mesohaline portions of the estuary.

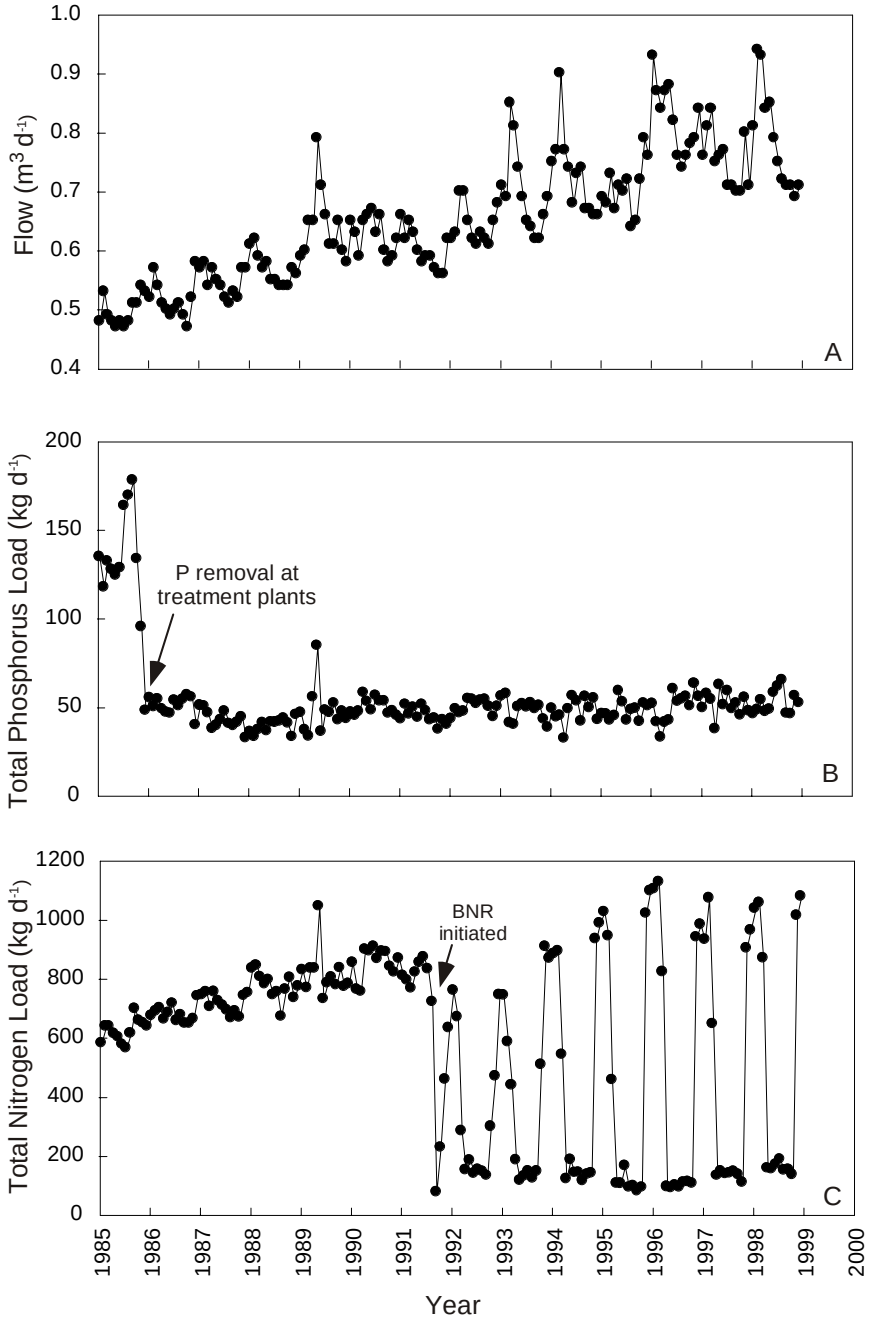


Figure 7-9. Point source flow and loading from below the fall line: Flow (A), TP (B) and TN (C).

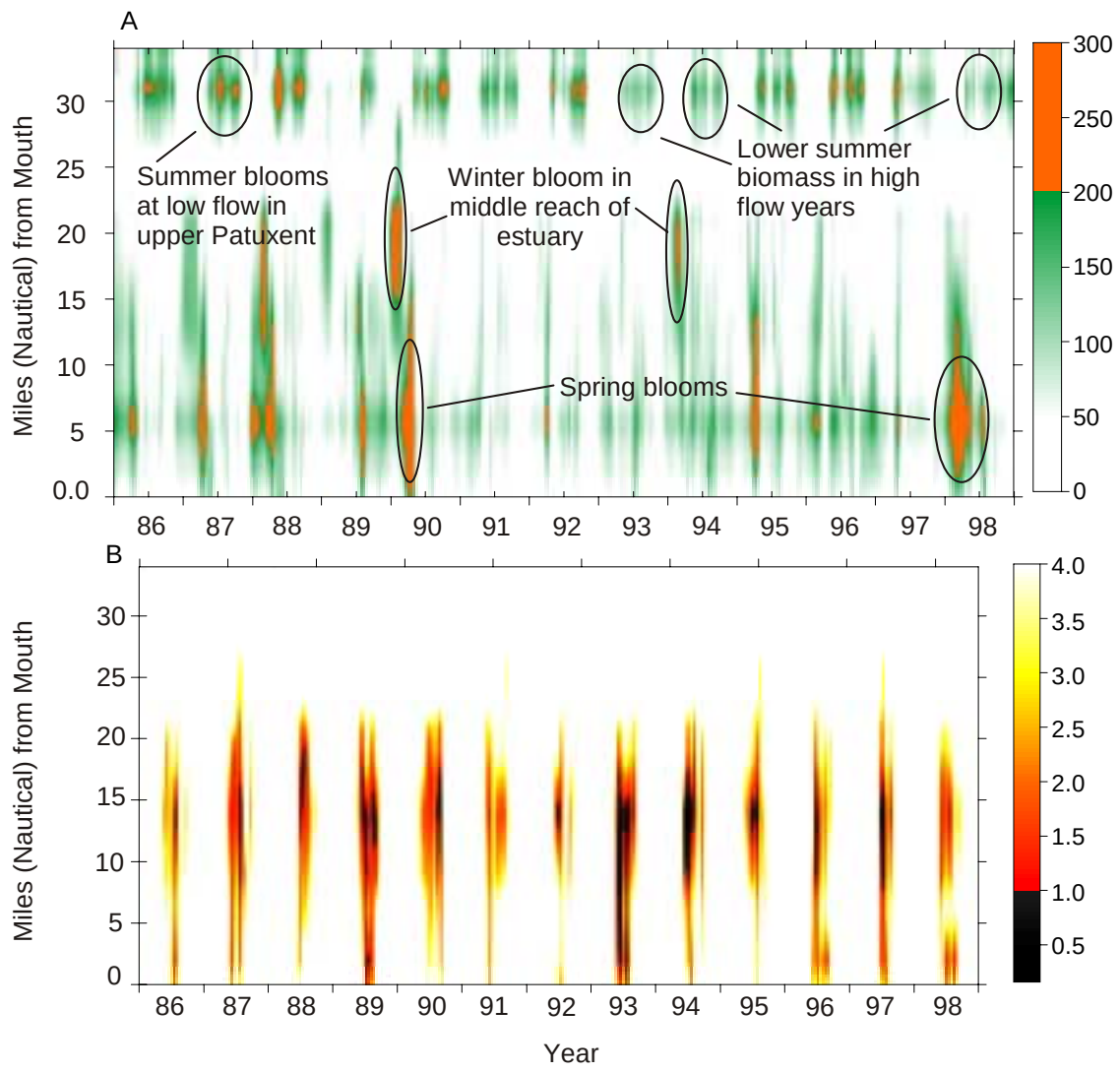


Figure 7-10. A time-space plot of water column integrated chlorophyll-*a* (A) and dissolved oxygen conditions in deep waters (B) of the Patuxent River estuary from 1986 through 1998.

Table 7-1. A summary of mean daily total N and P inputs to the Patuxent River estuary. Data and methodological details are provided in Boynton *et al.* (1995). Post-BNR point source data are from Wiedeman and Cosgrove (1998). Areal nutrient loads referred to in the text were computed using a total estuarine surface area of 137 x 10^6 m^2 (Cronin and Pritchard 1975).

	Total N **(Kg d^{-1})**	***Total P*** **(Kg d^{-1})**
Point Sources		
Pre-BNR above fall line	1577	124
Pre-BNR below fall line	744	60
Post-BNR above fall line	744	57
Post-BNR below fall line	454	50
Diffuse Sources		
Above fall line	984	43
Below fall line	2220	148
Direct Atmospheric Deposition	603	24
Pre-BNR Total Load (Kg· d^{-1})	6128	399
Areal Load (g N or P m^{-2} yr^{-1})	16	1.1

Concentrations of chlorophyll-*a* and DO in deep waters are two variables often used as indicators of nutrient enrichment in aquatic ecosystems. Beginning in 1986, the Chesapeake Bay Water Quality Monitoring Program began systematic measurements (bi-monthly measurement frequency at 10 stations along the axis of the estuary) and a reliable record of water quality during this period is now available. Chlorophyll-a and DO data have been summarized in time-space plots (Figure 7-10). Both regular patterns and interannual variability are evident in both plots. For example, spring blooms occur every year in the lower estuary but are much larger in high river flow years than in drought years. Similarly, summer accumulations of chlorophyll-*a* occur in the upper reaches of the estuary, especially during dry summers when water residence times are longer than normal. Deep waters of the estuary have DO concentrations in excess of 4 mg l^{-1} during the cool periods of the year but hypoxia develops every year during the summer period. However, hypoxic intensity, duration and areal extent vary strongly among years with hypoxic characteristics being most intense during wet years when vertical stratification becomes greatest. We have emphasized in the above discussion the importance of nutrient additions originating from the land and atmosphere. These are not the only sources. Application of a box model developed for the Patuxent by Hagy *et al.* (2000) indicated that on an annual basis N and P was transported from the estuary to the Bay. However, during summer, inorganic N and P are imported from the Bay and contribute to poor water quality conditions during that portion of the year. It appears that nutrient input reductions from point and diffuse sources have not yet been large enough to modify, in any large fashion, the patterns of water quality that have been well documented via the monitoring program. It appears that larger reductions will be needed, especially in the diffuse source category.

7.7 Future Prospects

Although nutrient control measures have not yet resulted in dramatic increases in water quality or restoration of living resources, unrelenting attention to reducing N and P inputs will be required to help offset the effects that would otherwise be experienced with population increases. Moreover, because water quality is better in low-load years (low flow years) than in high load years (high flow years (Figure 7-10) additional nutrient decreases should help to improve water quality conditions. In this sense, significantly better non-point source (NPS) controls will be a key issue, as further reductions in point source loadings will not be as large or as cost-effectively achieved as in earlier periods.

Irrespective of the progress that has been made in mitigating the effects of population growth and concomitant land use changes, large NPS inputs still remain the greatest threats to the Patuxent and the Chesapeake Bay environmental quality. However, serious NPS reduction programs are relatively new and there is a great deal of room for improvement and expansion of these efforts. Additionally, there has been increased interest and excitement in restoration of estuarine habitats known to exert positive effects on water quality (*e.g.*, marshes, submerged aquatic vegetation (SAVs) and oyster reefs). In our opinion, sustainable limits will be reached once all reasonable attempts made to further reduce non-point and point source inputs no longer result in salubrious changes to water quality. This will present policy-makers with a considerable dilemma, because the main alternative available will be to prevent further net immigration to the area (and associated development) something that contravenes the fundamental freedoms Americans now have to choose where to live and how to use private lands.

7.8 Conclusions

Early in this paper the following question was posed, "*Have the policies and practices adopted for the Patuxent been successful in achieving their intended goals?*" While there are signs of recovery on the horizon, it is still premature to state so unequivocally. It could take another five or even ten years to see if the policies and practices adopted in the mid-1980s will ultimately achieve the stated goal of returning the Patuxent's water quality to that of four decades ago. In our view, the public and public officials must learn to accept that rapid solutions to environmental problems are rarely available. It can take as long or longer to solve an environmental problem as it took to cause it, and even then, given scientific uncertainties and other environmental changes that may be occurring (*e.g.*, sea level rise), solutions may be elusive There are, though, some unequivocal conclusions that can now be made. One of the most important "*lessons*" of the Patuxent was the prominent role that science and scientifically driven monitoring programs have had in affecting policy-making. The scientists who worked on and near the Patuxent played a large role in formulating the close connection that developed between state-of-the-art science and evolving public policy. During the critical period of time from the late 1970s through the mid 1980s many scientists had regular and intensive contact with elected and appointed State of Maryland and federal officials, non-government organizations, citizen groups, the press, and even the courts. Importantly, these scientists were generally able to focus their advocacy on science and the application thereof to public policy rather than on given policy actions. This emphasis on science kept their credibility high with politicians and the public, but

did not compromise the credibility of their research findings. Key individuals in the press (*e.g.*, Tom Horton of the Baltimore Sun) did much to help interpret the science for the lay public, and abetted the efforts of scientists to get relevant research into the everyday dialogues of those public officials making key decisions for the Patuxent.

The Patuxent case study also shows how important the availability of high quality monitoring data and long records have proven to be in diagnosing problems, understanding the causes and developing effective remediation strategies (Heinle *et al*,. 1980; D'Elia *et al.*, 1992). No less important has been the availability of adequate funding for scientific research. In this case, it largely resulted from the recognition of key officials, among them Assistant Secretary of OEP, DHMH William Eichbaum, State Senator C. Bernard Fowler and U.S. Senator Charles Mathias, that scientific understanding was incomplete and that research could add a clearer understanding to the public dialogue. Their willingness to commit themselves to provide special funding for research on the Patuxent and Chesapeake was essential in making available the resources needed. However, it would be far better to have available stable and long-term sources of funds to support applied peer-reviewed environmental research on basins such as the Patuxent.

Another lesson of the Patuxent is that research on pollution abatement technology, such as BNR, can really pay off. Research into improving public works technology is supported at a fraction of what is done for industrial research and development efforts. This is very short-sighted.

Finally, while we recognize the need to keep management options as straightforward and cost-effective as possible, the rigid approach to implementing AWT protocols of the late 1970s and early 1980s needs to be avoided in the future. Fear of the costs of implementing N removal on the Patuxent and loss of federal matching funds led policy-makers at the time to conclude, despite substantial scientific evidence to the contrary, that a less expensive P removal strategy could be forced to work. Flexibility in defining standards and prescribing solutions is key to effective management of nutrient enrichment problems. Cost is always an issue, and ultimately the public must determine if the costs to undertake a scientifically sound remediation program is practical. If it is deemed not to be, other alternatives that are scientifically supportable must be sought, or the technology must be improved to make it more cost-effective.

While it is premature to conclude that the N-removal strategy for the Patuxent has been successful in reversing the damage of four decades of excessive nutrient enrichment, it does appear that for the present, further degradation has abated. The next decade should provide the answer to the initial question unless population growth simply overwhelms the system. As the Year 2020 Panel Report (1988) so correctly stated, "*How the land is used is a basic factor in the ecological health of the Chesapeake Bay*." That, in large measure, is a function of demography.

Acknowledgments

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8. 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, 1998, 1999, 2000, and 2001), 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 for the Patuxent River were again reviewed for the period 1985-2000 as a portion of a synthesis effort supported, in part, by the UMD CES IAN Program. 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, and 1998). Because of the severe drought during 1999, TP loads during 1999 were 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} $year^{-1}$. TN loads were also reduced during 1999, again because of the effects of the drought in reducing diffuse source run off of TN. There is strong evidence that substantial nutrient load reductions at the fall line have occurred in recent years. However, it also appears that in the years following (post-1993) the installation of BNR capabilities at the large sewage treatment plants in the Patuxent (all but one of which are located above the fall line) diffuse source loading of TN below the fall line has increased, partly because the late 1990's were wetter than the earlier years, and partly because the middle and lower portions of the Patuxent basin have been rapidly developing. Preliminary estimates of annual nitrogen loading to the full Patuxent system appear to not have changed between the pre (1985-1990) and post-BNR years (1993-2000).

Dissolved oxygen conditions in the Patuxent River were examined using monthly data collected at the four long-term sediment-water exchange (MINI-SONE) stations. In general dissolved oxygen conditions in deep water at the deeper sites (MRPT and BRIS) were poor to fair in 2001. For example, dissolved oxygen remained below 1.0 mg l^{-1} at station MRPT during June - August. During the drought year of 1999 DO never decreased below 2.7 mg l^{-1} at this site indicating the importance of flow and nutrient loads on DO conditions. It is worth noting that there was a very substantial spring bloom during 2001. Bottom water DO conditions at all of our sites were depressed in June, 2001, even at sites normally not exposed to hypoxic conditions (BUVA and STLC). Bottom water DO levels had increased to above 2.0 mg l^{-1} by mid-September.

Sediment–Water Oxygen and Nutrient exchanges measured during 2001 contrasted with those measured during 1999 (a drought year) and the contrast is consistent with the

conceptual model of how sediment-water exchanges are regulated in estuarine systems. For example, SOC rates were larger at the two mid-river sites during most of the 1999 sampling period compared to 2001. These enhanced values very probably resulted because dissolved oxygen concentrations in deep water were higher during 1999 than in most previous years or during 2001. SOC rates become limited (reduced in magnitude) when bottom waters are depleted in dissolved oxygen. No significant trends were detected for SOC . The status of SOC was good at the most up-river site (BUVA), poor at the two mid-river sites (MRPT and BRIS) and good at the most down-river site (STLC).

Ammonium (NH_4^+) fluxes were also larger, and at several sites much larger, during 2001 than during the 1999 drought and most other years. The relatively low fluxes observed during 1999 were very probably a response to reduced nutrient loads associated with drought conditions. The large reductions in ammonium flux between adjacent years of high (1998) and low (1999) nutrient load is also instructive. This annual-scale 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 two interannual-scale trends in ammonium fluxes detected at SONE sites in the Patuxent River (MRPT and STLC) and both were increasing (judged to be a degrading trend). Ammonium fluxes at all sites were judged to be in the poor range.

Positive sediment nitrate and nitrite fluxes (fluxes directed from sediments to the water column) are a definite sign of sediment nitrification activity, 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. However, during 2001 fewer positive sediment nitrate and nitrite fluxes were observed, consistent with generally lower DO conditions and higher river flows. We continue to believe that the presence of positive nitrate flux is a good tool for monitoring the general biogeochemical health of sediments. There were two interannaul-scale trends found for nitrite and nitrite plus nitrate, the former at BUVA and the latter at MRPT. Both were judged to be positive trends.

During 2001, inorganic phosphate fluxes (PO_4^{3-} or DIP) were similar to the long-term averages and considerably higher than those observed during the 1999, a drought year. During the drought year DIP fluxes were near or below the long-term average at all sites. At three of the sites (BUVA, MRPT and BRIS) phosphorus fluxes were far below average rates in July and August of 1999. Experimental studies involving phosphorus 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 two significant temporal trends in phosphorus fluxes. At BUVA the trend was towards reduced fluxes (a positive trend) and at MRPT (a deeper and generally hypoxic station) the trend was towards larger fluxes (a degrading trend). Phosphorus flux status was fair at the up-river site (BUVA), poor and fair at the two mid-river sites (MRPT and BRIS, respectively) and good at the down-river site (STLC).

During 2001 a comparison ***of littoral zone habitats*** was made for several locations and regions within the mesohaline portion of the Bay focusing on the parameters important to submerged aquatic vegetation (SAV). The goal of this investigation was to accurately measure and characterize many of the complex and interacting parameters necessary for SAV growth and survival in these shallow water habitats. This included measurement of the five water quality parameters (DIN, DIP, Kd, TSS, Chl-*a*) determined most important to the growth and survival of SAV, and to compare measured values to the habitat limits specified by the USEPA (USEPA, 2000). In addition comparisons of epiphyte fouling rates were made between regions and as well as between locations with healthy SAV populations and those without. On a regional basis, there were no differences in water quality between Tangier Sound and the lower Potomac River. Dissolved nutrient concentrations (DIN, DIP) were in general below the SAV mesohaline habitat limits at all stations. However, on a regional basis, the other parameters (Kd, TSS, Chl-*a*) were very close to the established SAV habitat limits. Within each region, significant differences were found among stations in many of the parameters measured with some stations consistently meeting the SAV habitat criteria while others consistently did not. Epiphyte fouling rates were also quite variable among stations within each region. Because of this variation, no difference was found between Tangier Sound and the lower Potomac River. Finally, no difference in epiphyte fouling rate was found between stations with healthy SAV populations and those without. This result suggests that high epiphyte fouling rates alone cannot be used as an indicator for the restoration of SAV.

High spatial resolution water quality data was collected in Tangier Sound and the Magothy and Severn Rivers in 2001 using the DATAFLOW V mapping system. The goal of this effort was to identify the spatial and temporal scales of water quality variability in these systems and to further develop this method of data collection for enhanced near-shore and tributary monitoring. The information collected on fourteen cruises on the Magothy and Severn Rivers as well as six cruises in Tangier Sound provided the data necessary to explore and develop the most appropriate ways of using and validating this data. While this evaluation process is not yet complete, several important results have been found. The spatial patterns found on both rivers were very dynamic. Large changes in both the concentration and distribution of turbidity and chlorophyll-*a* were found between successive bi-weekly cruises, suggesting that sampling at longer intervals would not adequately capture the variation in these systems. Calibration of DATAFLOW sensor output to laboratory-based analysis of water samples during a single cruise can provide the best estimate of water quality (r^2 up to 0.97 for Chl-*a*) given that a wide range of values are encountered during each cruise. However, when

water quality conditions are relatively homogenous during a single cruise, calibration can still be accomplished with a small loss of accuracy by using relatively robust relationships derived when multiple cruises are combined. As a way to explore methods of data interpretation, chlorophyll-*a* and turbidity values measured at the single DNR monitoring station (fourteen cruises) were compared to estimates derived from an area-weighted interpolation of DATAFLOW data. For both the Magothy and the Severn rivers, there were no significant differences between estimates obtained from the single DNR monitoring station on each river and DATAFLOW estimates for water column chlorophyll-*a*. However, river-wide estimates for turbidity were significantly higher using DATAFLOW estimates compared to the single DNR station on both rivers. Finally, the cell size for data interpolation was altered to demonstrate that as cell size increases, estimates for river-wide chlorophyll-*a* concentrations also increase. These results will provide some information to help guide the standardization of DATAFLOW data processing.

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