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ECOSYSTEM PROCESSES COMPONENT (EPC)

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MARYLAND CHESAPEAKE BAY WATER QUALITY MONITORING PROGRAM

ECOSYSTEMS PROCESSES COMPONENT (EPC)

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INTERPRETIVE REPORT
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Table of Contents

	Page No.
List of **Figures**	**ii**
List of **Tables**	**iv**
Executive Summary	**E-1**
Introduction and Objectives	**1-1**
Dissolved Oxygen Criteria Assessment	**2-1**
Community Metabolism as an Indicator of Water Quality Impairment and Restoration	**3-1**
Spatial Analysis of Water Quality Conditions	**4-1**
Mattawoman Creek: A Case Study	**5-1**
Modeling ConMon Data for Dissolved Oxygen Criteria Assessment	**6-1**
Upper Patuxent River Estuary: Macrophyte Establishment Case Study	**7-1**

Table of Contents

List of Figures

		Page No.
1-1	A simplified schematic diagram indicating degradation and restoration trajectories of an estuarine ecosystem.	1-4
2-1	Percent failure (full data set) versus maximum duration of DO failure (hours) for all ConMon stations (N = 25) analyzed.	2-11
3-1	Screen shot showing the required input format needed for Metabolism.xls for calculation of metabolism variables.	3-5
3-2	A map showing the location of ConMon sites used in all analyses in this chapter.	3-7
3-3	Bar graphs of mean weekly rates of gross primary production (Pg*) and community respiration (Rn).	3-13
3-4	Bar graphs of monthly rates of gross primary production (Pg*) at Baltimore Harbor (MCH).	3-14
3-5	Annual discharge in cubic feet per second (cfs) for the Susquehanna River (USGS station 01578310) and the Patapsco River (USGS station 01586000).	3-14
4-1	Case study watersheds used in analysis.	4-3
4-2	Examples of cross-channel salinity gradients observed in interpolated maps.	4-11
4-3	Surface salinity profiles of average spring, summer and fall salinity for Gunpowder and Middle Rivers.	4-12
4-4	Surface salinity profiles of average spring, summer and fall salinity for South and West/Rhode Rivers.	4-12
4-5	Salinity zones of estuaries.	4-13
4-6	Average annual chl-*a* concentrations by estuary.	4-15
4-7	Standard deviations of average annual chl-a concentrations by estuary.	4-16
4-8	Frequency of elevated chl-*a* for the Gunpowder River.	4-18
4-9	Frequency of elevated chl-*a* for the Middle River.	4-19
4-10	Frequency of elevated chl-*a* for the South River.	4-20
4-11	Frequency of elevated chl-*a* for the West/Rhode River.	4-21
4-12	Example of chl-*a* kriged data for West/Rhode river showing cruise track.	4-22
4-13	Distribution of SAV in year of peak extent over maps of frequency of elevated chl-*a*.	4-23
4-14	Scatterplots of stressors and drivers.	4-25
4-15	Scatterplots of stressors and drivers.	4-26
4-16	Scatterplots of stressors and drivers.	4-27
4-17	Scatterplots of stressors and drivers.	4-28
4-18	Septic distribution within and outside the 1,000 ft. buffer.	4-29
4-19	Interannual percentage change in chl-*a*, TN and TP for subestuaries.	4-31
4-20	% Change in annual chl-*a* in subestuary compared to % change in annual chl-*a* in Bay mainstem.	4-31
4-21	Elasticity of chl-*a* with respect to TN within sub-estuaries.	4-32
4-22	Elasticity of chl-*a* with respect to TP within sub-estuaries.	4-32
4-23	Cross-source Elasticity of chl-*a* with respect to Susquehanna TN.	4-33
4-24	Cross-source Elasticity of chl-*a* with respect to Susquehanna TP.	4-33
4-25	Local share statistic of chl-*a* response in sub-estuary compared to chl-*a* change in Bay mainstem.	4-34
4-26	Comparison of elasticity metrics.	4-34
5-1	A map of Mattawoman Creek drainage basin boundary, stream network and major sampling locations referred to in the text	5-3
5-2	Simplified maps showing major land uses in the Mattawoman Creek watershed for the time periods 1973, 2002 and 2010.	5-4
5-3	Monthly (small dots) and annual average (large dots) surface water pH values collected from 2 monitoring sites on Mattawoman Creek (MAT0016 and MAT0078) from 1986-2011.	5-8

Table of Contents

List of Figures

		Page No.
5-4	Monthly (small dots) and annual average (large dots) surface water $NO_2 + NO_3$ concentration collected from 2 monitoring sites on Mattawoman Creek (MAT0016 and MAT0078) from 1986-2011.	5-9
5-5	Monthly (small dots) and annual average (large dots) surface water orthophosphate (PO_4) concentration collected from 2 monitoring sites on Mattawoman Creek (MAT0016 and MAT0078) from 1986-2011.	5-10
5-6	Monthly (small dots) and annual average (large dots) surface water Total Nitrogen (TN) concentration collected from 2 monitoring sites on Mattawoman Creek (MAT0016 and MAT0078) from 1986-2011.	5-11
5-7	Monthly (small dots) and annual average (large dots) surface water Total Phosphorus (TP) concentration collected from 2 monitoring sites on Mattawoman Creek (MAT0016 and MAT0078) from 1986-2011.	5-12
5-8	Monthly (small dots) and annual average (large dots) of Secchi Disk depth (m) collected from a monitoring site on Mattawoman Creek (MAT0016) from 1986-2011.	5-13
5-9	Monthly (small dots) and annual average (large dots) surface water chlorophyll-*a* concentration collected from 2 monitoring sites on Mattawoman Creek (MAT0016 and MAT0078) from 1986-2011.	5-14
5-10	Monthly average values for surface water turbidity (NTUs) based on ConMon data collections at two Mattawoman Creek sites (XEA3687 from 2004-2010 and XEB5404 from 2009-2010). In general, months included for each year were April-October.	5-15
5-11	Monthly average values for surface water dissolved oxygen concentration based on ConMon data collections at two Mattawoman Creek sites (XEA3687 from 2004-2010 and XEB5404 from 2009-2010).	5-16
5-12	Monthly average values for surface water chlorophyll-*a* concentration based on ConMon data collections at two Mattawoman Creek sites (XEA3687 from 2004-2010 and XEB5404 from 2009-2010).	5-18
5-13	Mean monthly (Apr-Oct) estimates of community gross primary production (g O_2 m^{-3} day^{-1}) and community respiration (g O_2 m^{-3} $night^{-1}$) for the period 2004-2010.	5-19
5-14	A summary of annual SAV community coverage (ha), with SAV density classes, for Mattawoman Creek for the period 1971-2010.	5-21
5-15	Annual maps of SAV spatial coverage (and SAV density estimates) for Mattawoman Creek from 1994-2010.	5-22
5-16	Annual summary of SAV coverage (ha), water clarity (indexed with Secchi Disk measurements) and algal biomass (indexed with chlorophyll-*a* concentration) for the period 1986-2010 in Mattawoman Creek.	5-23
5-17	Scatter plots of average annual chlorophyll-*a* concentration versus SAV coverage (a) and summer average Secchi Disk depth versus SAV coverage (b) for Mattawoman Creek.	5-24
5-18	Plots of average monthly TN (a) and TP (b) loads (kg N or P per day) and freshwater discharge rate (m^3 sec^{-1}).	5-26
5-19	Plots of annual point and diffuse TN loads to Mattawoman Creek.	5-28
5-20	Scatter plot of annual TP versus TN loads for a variety of estuarine and coastal marine ecosystem (small gray circles; see Boynton *et al*. 1995 for references for these sites).	5-30
5-21	A schematic diagram of a nutrient budget (TN) model developed for Mattawoman Creek.	5-32
5-22	A scatter plot of TN load versus chlorophyll-*a* concentration developed for Mattawoman creek and other shallow Chesapeake Bay tributaries.	5-34
5-23	A scatter plot of chlorophyll-*a* versus Secchi Disk depth developed for Mattawoman Creek and two other shallow Chesapeake Bay systems.	5-36
5-24	A scatter plot of chlorophyll-*a* versus hypoxic period (DO $<$ 3.2 mg L^{-1}) developed for Mattawoman Creek and one other shallow Chesapeake Bay systems.	5-37

Table of Contents

List of Figures

		Page No.
5-25	A semi-quantitative plot of changes in algal biomass (indexed as chlorophyll-*a*), water clarity (indexed as Secchi Disk depth) and SAV coverage as a function of TN loading rate.	5-39
6-1	Continuous monitoring Conmon station locations (●) in the Maryland Coastal Bays.	6-2
6-2	Multiple linear regression model results for: (a) 2005, (b) 2006, and (c) 2007.	6-4
7-1	Map showing regional location (a) and spatial details (b) of the Patuxent basin and Patuxent River estuary.	7-3
7-2	A time-series plot of freshwater flow (cfs) of the Patuxent River measured at the USGS gage located at Bowie, MD.	7-8
7-3	A time-series plot of total nitrogen (TN) load to the Patuxent River measured at the USGS gage located at Bowie, MD.	7-9
7-4	A time-series plot of total phosphorus (TP) load to the Patuxent River measured at the USGS gage located at Bowie, MD.	7-10
7-5	An incomplete time-series plot of nitrate plus nitrite load to the Patuxent River measured at the USGS gage located at Bowie, MD.	7-10
7-6	An incomplete time-series plot of dissolved inorganic phosphorus (PO_4) load to the Patuxent River measured at the USGS gage located at Bowie, MD.	7-11
7-7	A time-series plot of discharge flow, TN and TP loads from the Western Branch WWTP.	7-12
7-8	Time-series plots of SAV coverage (ha) in the OH (a) and TF (b) zones of the Patuxent River estuary from 1978-2008.	7-13
7-9	Time-series plot of TSS (a) and Secchi Disk depth (b) at station TF 1.3 in the Patuxent River estuary (1985-2007).	7-15
7-10	Time-series plot of NO_{23} (a) and PO_4 (b) at station TF 1.3 in the Patuxent River estuary (1985 – 2007).	7-16
7-11	Time-series plot of chlorophyll-*a* concentration at station TF 1.5 in the Patuxent River estuary (1985-2007).	7-17
7-12	Time-series plots of a selection of water quality variables taken from both the historical record and the more recent monitoring program data base (1985-2007).	7-20
7-13	Time-series plots of a selection of water quality variables taken from both the historical record and the more recent monitoring program data base (1985-2007).	7-21
7-14	A time-series plot of surficial sediment particulate phosphorus (PP) concentration collected at the Buena Vista station (rkm 40) between 1978 and 2002.	7-22
7-15	Time-series plots of net sediment-water exchange rates collected at Buena Vista (rkm 40) in the Patuxent River estuary.	7-24
7-16	Vertical bar chart of annual (April-October) gross primary production (Pg*) and community respiration (Rn) for the period 1964-2005.	7-26
7-17	Multiple scatter plots of observed SAV coverage versus modeled SAV coverage for the combined TF and OH zones of the Patuxent River estuary.	7-30
7-18	Scatter plot of Western Branch WWTP NO_{23} load versus annual SAV coverage (ha) in the combined TF and OH portion of the estuary.	7-31
7-19	Scatter plot of NO_{23} concentration at Station TF 1.3 versus annual SAV coverage (ha) in the combined TF and OH portion of the estuary.	7-32
7-20	Scatter plots of observed versus modeled SAV coverage in the TF (a) and OH (b) zones of the estuary for the post-recovery period (1994-2007).	7-34

List of Tables

		Page No.
2-1	Chesapeake Bay Dissolved oxygen criteria (reproduced from USEPA 2003, Table 1).	2-2
2-2	DO criteria assessments used for this study.	2-3

Table of Contents

2-3	**Example of ConMon data files generated for dissolved oxygen criteria analysis and metabolism calculations based on modern ConMon data sets.**	**2-4**
2-4	**List of ConMon stations used in DO % attainment and low DO duration analyses.**	**2-5**
2-5	**Dissolved oxygen criteria attainment analysis for select ConMon stations.**	**2-6**
3-1	**List of ConMon stations used in Metabolism calculations.**	**3-6**
3-2	**A summary of community metabolism rates (Pg* and Rn) for summer periods (June-August) for a selection of Chesapeake Bay ConMon sites.**	**3-8**
4-1	**DATAFLOW© sampling years by estuary.**	**4-4**
4-2	**Land use of case study watersheds.**	**4-4**
4-3	**Data sources used in analysis.**	**4-6**
5-1	**A summary of land use/land cover in Mattawoman Creek watershed for three time periods (1973, 2002 and 2010).**	**5-4**
5-2	**A copy of the Maryland DNR watershed profile for Mattawoman Creek containing watershed, wetland and indicator data.**	**5-5**
5-3	**A listing of SAV species common and less common in the shallow waters of Mattawoman Creek.**	**5-7**
5-4	**A summary of DO criteria assessment results developed for Mattawoman Creek. Data were from ConMon sites XEA3687 (2004-2010) and XEB5404 (2009-2010).**	**5-17**
5-5	**Examples of differing rates of community gross primary production based on ConMon data collected at these sites for one or more years.**	**5-20**
5-6	**Multiple estimates of annual average TN and TP loads to Mattawoman Creek.**	**5-27**
6-1	**Corrrelation coefficients (*r*) between 5-day mean of daily DO range and water column properties including PAR, temperature, salinity, turbidity and chlorophyll-a.**	**6-3**
6-2	**The coefficient of determination (r_2) between observations and linear regression models using different numbers of independent variables during 2005, 2006, and 2007.**	**6-5**

Executive Summary 2011

Dissolved Oxygen Criteria Assessment

- Data from **25 ConMon stations** were analyzed for both **percent failure** and **duration of failure** events for DO criteria. Both instantaneous (< 3.2 mg L^{-1}) and 30-day mean (< 5 mg L^{-1}) criteria were analyzed.

- **Percent failure ranged from no failures (0%) to failing almost half the time (44%). Duration of low DO events** (failing instantaneous or 30-day mean criteria) **ranged from 15 minutes to 137 hours (almost 6 days).**

- At all sites and years, applying the 30-day criteria resulted in either no change or an increase in both percent failure and maximum duration of failure compared to the instantaneous criteria. These % failure changes ranged up to 26%. **It is an important finding that the 30-day criterion is protective of the instantaneous criteria.**

Community Metabolism as an Indicator of Water Quality Impairment and Restoration

- Community metabolism rates were computed for 15 sites. **Metabolism is a fundamental rate process known to be sensitive to nutrient loading rates and because of these two characteristics is a useful monitoring index of Bay condition.**

- **Clear and large differences in metabolic rates (primary production and community respiration) were found between very enriched, moderately enriched and less enriched sites in the Bay.** Small tributaries with restricted flushing appeared to have the highest rates of metabolism and near-shore metabolism appeared to be higher than larger, deeper off-shore sites.

- The possibility of including a semi-realtime display of metabolism data for key locations is suggested as an addition to the Eyes on the Bay web site.

Spatial Analyses of Water Quality:DATAFLOW©

- **Salinity patterns observed using Dataflow© data reveal that mainstem Bay water is, generally speaking, having more influence on conditions on the right side of W shore tributaries looking up estuary, whereas watershed conditions are likely to be having more influence on the left-hand shores, particularly near the mouths of these tributaries.** The effect is less pronounced in the more southern tributaries that we evaluated, the South and the West/Rhode. Given the consistency of these patterns across the 8 estuaries studied this year and last, it is possible that these patterns are caused by Coriolis-induced tidal rectification, although other forces cannot be ruled out without further work.

- **Spatial analysis in the new estuaries showed finding consistent with previous years that areas with elevated chlorophyll-*a* were not randomly distributed in the estuaries.**

However, unlike 3 of the previous 4 estuaries, these estuaries demonstrated a consistent pattern of highest average chl-*a* in the upper reaches, often localized near the mouths of small tributaries. **These findings suggest that local watershed conditions are more likely to be driving persistently elevated chl-*a* but ephemeral blooms could have multiple drivers.**

- A simple comparison of potential drivers of chl-*a* and SAV responses among the eight estuaries examined so far was not substantially improved by the addition of the additional four estuaries this year. **The use of a more spatially detailed dataset of septic locations may have been important for identifying a significant positive correlation between septic density and persistently elevated chl-*a*.** These results suggest the need for multi-variate models that can control for multiple factors and these models are being explored in a companion project.

- New indicators were tested to evaluate whether they could be used to measure the relative influence of sub-estuary vs. Bay mainstem drivers on local conditions. We compared elasticities that use interannual percentage changes in water quality conditions in order to isolate the effects of drivers on outcomes while holding the physical variables of the estuary constant. Results suggested that changes in the total mass of chl-*a* measured over a year (created using time and area-weighted chl-*a* concentration) were most correlated with changes in the chl-*a* in the Bay mainstem. **Of the potential nutrient drivers, chl-*a* was more responsive to within-estuary TN and TP changes and less responsive to Susquehanna TN and TP changes.** However, the sensitivity of response varied by estuary and year. A limitation on the use of these indicators is that the direction of change in the driver is inconsistent with the change in the response of chl-*a*. It is more common to have chl-*a* change in the opposite direction of the change in nutrients, while a change in chl-*a* in the mainstem changes in the same direction as chl-*a* in a subestuary, just over half the time.

Mattawoman Creek: A Case Study

- **The Mattawoman Creek analysis in this report is the fourth in a series of case studies examining estuarine water quality and habitat responses to strong management actions**. The goal of these analyses is to better inform management concerning expected system responses (magnitude of response, causes of responses and timing of responses) and thereby assist in furthering Chesapeake Bay restoration activities.

- **In the case of Mattawoman Creek a strong management action (N and P load reduction associated with WWTP operations) initiated during 1990 and completed by 1995 resulted in strong declines in algal chlorophyll-*a*, increases in water clarity and a very strong rebound in SAV coverage.** Chlorophyll-*a* began declining after a 3-year lag, water clarity began increasing after a 5-year lag and SAV began responding immediately but reached high levels of coverage 6-7 years after WWTP modifications.

- **We also conducted comparative analyses (e.g., using data from other systems similar to Mattawoman Creek) and found very similar relationships between nutrient loads and system responses** in terms of water clarity, SAV coverage and algal chlorophyll-*a* concentration. These findings tend to generalize system responses to nutrient load reductions.

- Data from the Chesapeake Bay Water Quality model was used to estimate the net exchange of N between the Potomac River and Mattawoman Creek and we found that **the Potomac generally serves as a dissolved inorganic nitrogen source to Mattawoman Creek**. However, the creek exports little total nitrogen (TN) and that indicates the creek is serving as a strong nitrogen sink, both for locally derived N as well as N imported from the Potomac River. Water and habitat quality in Mattawoman Creek should increase further if water quality is improved in the Potomac River.

Modeling ConMon Data for Dissolved Oxygen Criteria Assessment

- This effort addresses the issues of time and space in DO monitoring by developing a simple statistical model of day-scale DO patterns based on ConMon data.

- Initial results showed **5-day mean of daily DO range was significantly correlated with temperature, salinity and chlorophyll-*a*, but the model exhibits multicollinearity and suggests that application of different numbers of independent variables should be examined for better relationships.**

Upper Patuxent River Estuary: Macrophyte Re-Establishment Case Study

- In conjunction with other sources of funding (US EPA Bay Program and Maryland Sea Grant) we developed a case study of the upper tidal portion of the Patuxent River estuary, a nutrient enriched system located on the Chesapeake Bay western shore. **The analysis was geared toward providing management with guidance relative to the timing (lag times) and magnitude of estuarine responses to significant management actions** (reductions in WWTP loadings of N and P).

- **We found that point source reductions of P had little effect on SAV coverage in the tidal fresh and oligohaline regions of the estuary. However, reductions in N from local WWTP had a large and rapid effect on SAV coverage. Within two years SAV coverage had increased from near-zero to a point where much of the shallow water habitat contained dense SAV communities.**

- **We also found that local versus more distant WWTP load reductions were most important in supporting SAV resurgence. The local WWTP is currently being upgraded again and it would be prudent to continue monitoring this estuarine segment to document other water quality and habitat changes associated with those activities.**

- Following SAV restoration there was considerable year-to-year variation in SAV coverage and this appears to be mainly related to weather patterns; SAV coverage decline when summer conditions were very dry and water temperature very high. Both of these factors have negative effects on freshwater SAV communities, especially the stress associated with spikes in salinity.

Chapter 2

Dissolved Oxygen Criteria Assessment

E.M. Bailey, C.L.S. Hodgkins, M.A.C. Ceballos and W.R. Boynton

2-1 INTRODUCTION 1
2-2 CHESAPEAKE BAY DO CRITERIA 2
2-3 METHODS, DATA SOURCES AND DATA MANIPULATIONS 3
2-3.1 DATA SOURCES, QA/QC AND FILE MANAGEMENT 3
2-3.2 DO CRITERIA ASSESSMENT AND LOW DO DURATION ESTIMATION 4
2-4 RESULTS AND DISCUSSION 6
2-4.1 TESTING CON MON SITES FOR DO CRITERIA ASSESSMENT AND LOW DO DURATION 6
2-5 REFERENCES 12

2-1 Introduction

For the last 25 years, water quality monitoring in Chesapeake Bay and tributary rivers was largely based on monthly or bi-monthly sampling at fixed stations located over the deeper (channel) portions of these systems (i.e. long-term monitoring program). Such a design had many benefits, especially those related to developing seasonal to inter-annual scale indices of water quality status and trends. However, as in virtually all environmental science activities, any measurement sampling scheme is not adequate for addressing all questions. About a decade ago, a new program was initiated, first on a pilot-scale basis, to add measurements of water quality for shallow near-shore habitats. Concern for SAV habitat quality was a prime consideration in developing this program.

The ConMon program (so named to indicate the near-CONtinuous MONitoring feature of this activity) used in-situ sensor systems (YSI© Sondes) programmed to take measurements of a suite of water quality variables every 15 minutes. Included in the water quality suite was water temperature, salinity, pH, DO, turbidity and chlorophyll-a. In most instances ConMon sites are active from April – October (the SAV growing season and the period when low DO concentrations are most frequently encountered) and in most cases sites remained active for three years. In a few cases, sites have remained active for up to 10 years, thus serving as long-term or sentinel sites. To place this sampling intensity in perspective, at a typical long-term monitoring site, about 16 measurements of water quality variables were collected per year. In contrast, at a ConMon site about 20,500 measurements per year are obtained, an intensity of measurement about three orders of magnitude higher than traditional monitoring and an intensity of measurement needed to resolve diel-scale DO dynamics.

There have been over 95 sites in the Maryland Bay and Maryland Coastal Bays where ConMon data have been collected. The program is continuing although at fewer sites than in the recent past. The considerable spatial extent (encompassing sites with water quality varying from quite good to very poor) of these data sets allows for comparative analyses wherein it is likely that relationships between near-shore water quality and management actions can be found.

There are several prime uses of ConMon data sets. First, they have been used as a guide in selecting and monitoring SAV habitat restoration sites. Second, these data have "opened our eyes" to a new scale of hypoxia, namely diel-scale hypoxia wherein DO concentrations can reach critically low levels at night (and especially in the immediate post-dawn hours). Third, these data can be used to make estimates of community production and respiration, both of which are fundamental ecosystem features known to be related to nutrient loading rates. Fourth, these data can be used in DO criteria assessments for shallow open water sites (USEPA 2007).

It is the second and fourth ConMon uses that are the focus of this chapter and we approach this issue in three ways. First, we provide examples of DO criteria % non-attainment for several sites in the Bay system adding to the analyses presented in our previous report (Boynton *et al.* 2011). Second, we examine ConMon data at selected sites to estimate the DURATION of low DO events and relate these to DO criteria attainment or non-attainment. Finally, we examine ConMon data from a variety of sites with a selection of recent data for sites in more open water "exposed" locations to compare patterns of % non-attainment and duration of low DO events.

2-2 Chesapeake Bay DO Criteria

Starting in 2003 (and in subsequent updates) the U.S. Environmental Protection Agency (EPA) established dissolved oxygen (DO) criteria for the Chesapeake Bay and its tidal tributaries. EPA defined habitats based on designated uses and tailored DO criteria to account for different spatial and temporal conditions. Extensive reviews were done to relate DO criteria concentrations to living resources. Numeric criteria were developed for monthly, weekly, daily and instantaneous DO concentrations (Table 2-1).

Table 2-1. Chesapeake Bay Dissolved oxygen criteria (reproduced from USEPA 2003, Table 1).

Designated Use	Criteria Concentration/Duration	Protection Provided	Temporal Application
Migratory fish spawning and nursery use	7-day mean ≥ 6 mg liter^{-1} (tidal habitats with 0-0.5 ppt salinity)	Survival/growth of larval/juvenile tidal-fresh resident fish; protective of threatened/endangered species.	February 1 - May 31
	Instantaneous minimum ≥ 5 mg liter^{-1}	Survival and growth of larval/juvenile migratory fish; protective of threatened/endangered species.	
	Open-water fish and shellfish designated use criteria apply		June 1 - January 31
Shallow-water bay grass use	Open-water fish and shellfish designated use criteria apply		Year-round
Open-water fish and shellfish use	30-day mean ≥ 5.5 mg liter^{-1} (tidal habitats with 0-0.5 ppt salinity)	Growth of tidal-fresh juvenile and adult fish; protective of threatened/endangered species.	Year-round
	30-day mean ≥ 5 mg liter^{-1} (tidal habitats with >0.5 ppt salinity)	Growth of larval, juvenile and adult fish and shellfish; protective of threatened/endangered species.	
	7-day mean ≥ 4 mg liter^{-1}	Survival of open-water fish larvae.	
	Instantaneous minimum ≥ 3.2 mg liter^{-1}	Survival of threatened/endangered sturgeon species.[1]	
Deep-water seasonal fish and shellfish use	30-day mean ≥ 3 mg liter^{-1}	Survival and recruitment of bay anchovy eggs and larvae.	June 1 - September 30
	1-day mean ≥ 2.3 mg liter^{-1}	Survival of open-water juvenile and adult fish.	
	Instantaneous minimum ≥ 1.7 mg liter^{-1}	Survival of bay anchovy eggs and larvae.	
	Open-water fish and shellfish designated-use criteria apply		October 1 - May 31
Deep-channel seasonal refuge use	Instantaneous minimum ≥ 1 mg liter^{-1}	Survival of bottom-dwelling worms and clams.	June 1 - September 30
	Open-water fish and shellfish designated use criteria apply		October 1 - May 31

[1] At temperatures considered stressful to shortnose sturgeon (>29°C), dissolved oxygen concentrations above an instantaneous minimum of 4.3 mg liter^{-1} will protect survival of this listed sturgeon species.

Based on these USEPA dissolved oxygen criteria we examined % failure, total duration of failure, day-scale mean concentration and diel variability of dissolved oxygen measured at select ConMon locations. After consultation with Maryland Department of Natural Resources staff and Criteria Assessment Protocol Workgroup (CAP) input, we applied criteria that best suited the ConMon station location and temporal data set (Table 2-2).

Table 2-2. DO criteria assessments used for this study.

Criteria Type	CAP Protocol Description	Modification	Criteria (mg L^{-1})
Instantaneous	Evaluate on each hour	Evaluate using all available data (every 15 minutes)	≥ 3.2
1-day Mean	Average for each 24 hour period	Did not use (only applies in areas below pycnocline in summer)	≥ 2.3
7-day Mean	Begin on day 1 of calendar month, evaluate first 4 weeks, ignore trailing days	Divide all available days for calendar month into 4 equal size bins, use 4 "weekly" averages	≥ 4.0
30-day Mean	Begin on day 1 of calendar month, ignore trailing days	Use all available data for calendar month	≥ 5.0

2-3 Methods, Data Sources and Data Manipulations

2-3.1 Data Sources, QA/QC and File Management

Continuous monitoring data was obtained from Maryland Department of Natural Resources Tidewater Ecosystems Assessment division (B. Cole) in electronic (.txt file) format (dnr_cmon_sonde_2001-08). This file contained all the collected ConMon data from 2001 to 2008. A SAS® (www.sas.com) program was written to remove any data with failing error codes (as detailed in the MDDNR QAPP: Michael *et al.* 2009) and missing data (entire row removed). The SAS® program also allowed selection of data by station and year.

The SAS program named "CleanBen3" (Bailey, Wainger, Perry and Hall pers. comm. 2010) used to import, clean and select ConMon data:

```
/* assign the path to the location of permanent data files */
libname conmon 'C:\Documents and Settings\boynton\My Documents\My SAS
Files\9.2\ConMon';
run;
  data conmon.cleanBen3;
  set SASUSER.Ben;
%let validCodes = 'NULL','NUL','null','nul','Null','Nul';
if WTEMP_A in (&validCodes) and SPCOND_A in (&validCodes) and SALINITY_A in
(&validCodes) and DO_SAT_A in (&validCodes)and DO_A in (&validCodes);
keep SAMPLE_DATE SampleTime STATION SONDE Layer TOTAL_DEPTH TOTAL_DEPTH_A
BATT BATT_A WTEMP WTEMP_A SPCOND SPCOND_A SALINITY SALINITY_A DO_SAT DO_SAT_A
```

```
DO DO_A PH PH_A TURB_NTU TURB_NTU_A FLUOR FLUOR_A TCHL_PRE_CAL TCHL_PRE_CAL_A
CHLA CHLA_A COMMENTS;
run;
/* creates the parameter Year so we can choose data by year */
data metabdata_year;
set conmon.cleanBen3;
Year = substr(SAMPLE_DATE, 1, 4);
run;
/*    final step to out put data by year and station */
data metabdataxfb;
set metabdata_year;
where STATION = 'XFB2184';
run;
```

All 2009 and 2010 data was selected from separate files cleaned by Ben Cole (dnr_Cmon_2009_2010.mdb). Any data with invalid codes was removed prior to delivery to our group. The data was selected using this SAS® program (CleanBenNew2):

```
/* assign the path to the location of permanent data files */
libname conmon 'C:\Documents and Settings\boynton\My Documents\My SAS
Files\9.2\ConMonFY2012';
run;
  data metabdataxfb2009;
  set conmon.Ben2009; /* select conmon.Ben2010 for 2010 data*/
  where STATION = 'XFB2184';
  run;
```

Data files generated in SAS® were exported to Microsoft© Excel (.xls) and organized into files by station and year. Data files included the parameters: sample date, time, station (code) water temperature (ºC), water temperature error code, salinity, salinity error code, dissolved oxygen saturation (%), dissolved oxygen saturation error code, dissolved oxygen (mg L^{-1}), dissolved oxygen error code and year. An example of one of these files is shown below (Table 2-3). Files were given filenames to identify the type of data (Metabdata) the station (first three letters of the station code) and year (e.g., Metabdataxfb2004clean).

Table 2-3. Example of ConMon data files generated for dissolved oxygen criteria analysis and metabolism calculations based on modern ConMon data sets.

SAMPLE_DATE	SampleTime	STATION	WTEMP	WTEMP_A	SALINITY	SALINITY_A	DO_SAT	DO_SAT_A	DO	DO_A	Year
2004-06-07	1/0/1900	XFB0231	24.84	NULL	0.11	NULL	120.1	NULL	9.95	NULL	2004
2004-06-07	1/0/1900	XFB0231	24.7	NULL	0.11	NULL	119.7	NULL	9.94	NULL	2004
2004-06-07	1/0/1900	XFB0231	24.57	NULL	0.11	NULL	118.9	NULL	9.9	NULL	2004
2004-06-07	1/0/1900	XFB0231	24.35	NULL	0.11	NULL	120.5	NULL	10.08	NULL	2004
2004-06-07	1/0/1900	XFB0231	24.33	NULL	0.11	NULL	120.6	NULL	10.09	NULL	2004
2004-06-07	1/0/1900	XFB0231	24.35	NULL	0.11	NULL	120.7	NULL	10.09	NULL	2004
2004-06-07	1/0/1900	XFB0231	24.39	NULL	0.11	NULL	119.8	NULL	10.01	NULL	2004
2004-06-07	1/0/1900	XFB0231	24.45	NULL	0.11	NULL	118.7	NULL	9.9	NULL	2004
2004-06-07	1/0/1900	XFB0231	24.43	NULL	0.11	NULL	117	NULL	9.77	NULL	2004

2-3.2 DO Criteria Assessment and Low DO Duration Estimation

Data from 25 ConMon stations (Table 2-4) were QA/QC'd through the SAS program and organized into files by station and year.

Table 2-4. List of ConMon stations used in DO % attainment and low DO duration analyses. Stations in red are located in Maryland's Coastal Bays, stations in blue are open/exposed stations and stations in green are located in tributaries.

Mainstem Exposed Station Analysis			
Tributary	Station	Code	Year
Honga River	Muddy Hook Cove	XCG5495	2010
Honga River	House Point	XCG9168	2010
Eastern Bay	Kent Point	XGF0681	2006
Chesapeake Bay	Downs Park	XHF6841	2010
Chesapeake Bay	Fort Howard	XIF1735	2010
Sassafras River	Betterton Beach	XJH2362	2010
Chesapeake Bay	Susquehanna Flats	XKH0375	2010

Other Select Stations			
Tributary	Station	Code	Year
Corsica River	Sycamore Point	XHH3851	2005 2006 2007 2008
Patapsco River	Fort McHenry	XIE5748	2005 2006 2007 2008
Sassafras River	Betterton Beach	XJH2362	2006 2007 2008
Chesapeake Bay	Sandy Point South	XHF0460	2004 2005 2006 2007 2008

Maryland Coastal Bays		
Station	Code	Year
Bishopville Prong	XDM4486	2003 2004 2005 2006 2007 2008
Turville Creek	TUV0021	2003 2004 2005
Public Landing	XBM8828	2005 2006 2007 2008

Magothy River		
Station	Code	Year
Stonington	XHF3719	2001 2002 2003
Whitehurst	CTT0001	2002 2003

Patuxent River		
Station	Code	Year
Jug Bay	PXT0455	2003 2004 2005 2006 2007 2008
Benedict	XED0694	2003 2004 2005
Pin Oak	XDE4587	2003 2004 2005 2006 2007
CBL	XCF9029	2003 2004 2005

Potomac River		
Station	Code	Year
Piscataway Creek	XFB2184	2004 2005 2006 2007 2008
Indian Head	XEB5404	2009 2010
Mattawoman Creek	XEA3687	2004 2005 2006 2007 2008 2009 2010
Fenwick	XFB0231	2004 2005 2006 2007 2008
St. George's Island	XBF7904	2006 2007 2008

For each station and year the total hours the sonde measured DO was calculated. Two criteria were used (3.2 mg L^{-1} (instantaneous) and 5.0 mg L^{-1} (30-day mean)) and for each the total hours (for the entire year's data set) the sonde measured DO concentration below the criteria was calculated and a % failure determined. In addition, the total duration (continuous subsequent readings) of low DO (below the criteria) was also calculated and a maximum single duration below the criteria was determined (hours).

Please see http://mddnr.chesapeakebay.net/eyesonthebay/index.cfm for station locations, site information and additional quality assurance/sampling procedures.

2-4 Results and Discussion

2-4.1 Testing Con Mon sites for DO Criteria Assessment and Low DO Duration

High frequency DO data was analyzed from 25 ConMon stations (Table 2-5) to obtain the total hours below criteria (instantaneous and 30-day mean), the % failure and the maximum duration of a below-criteria event (Table 2-6). We calculated the duration of below-criteria events to investigate not only how often a station was exposed to low DO, but also how long the low DO persisted. We focused on shallow tributary stations in the Maryland portion of the Chesapeake Bay and a few stations located in the Maryland coastal bays. Data from these select stations encompassed the years 2001 to 2010 and generally most data sets included data from March to December.

Table 2-5. Dissolved oxygen criteria attainment analysis for select ConMon stations. Criteria shown in blue denotes the instantaneous criteria (3.2 mg L^{-1}) and purple denotes 30-day mean criteria (5.0 mg L^{-1}).

Mainstem Exposed Stations

Location	Station	Year	Date Range	Total Days of Measurement	Criteria < 3.2 mg L^{-1}			Criteria < 5.0 mg L^{-1}		
					Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)	Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)
Honga River – Muddy Hook Cove	XCG5495	2010	4/6 to 11/1	172	5	0.1	3	83	2	10
Honga River – House Point	XCG9168	2010	4/6 to 11/1	157	0	0	0	12	0.3	3
Eastern Bay – Kent Point	XGF0681	2006	3/23 to 10/31	153	11	0.3	7	42	1	13
Chesapeake Bay – Downs Park	XHF6841	2010	3/25 to 11/3	179	83	2	10	384	9	31
Chesapeake Bay – Fort Howard	XIF1735	2010	3/25 to 11/3	174	23	1	4	223	5	8
Sassafras River – Betterton Beach	XJH2362	2010	4/6 to 11/1	157	0	0	0	8	0.2	2
Chesapeake Bay – Susquehanna Flats	XKH0375	2010	4/22 to 10/21	191	0	0	0	4	0.1	2

Maryland Coastal Bays

Location	Station	Year	Date Range	Total Hours	Criteria < 3.2 mg L^{-1}			Criteria < 5.0 mg L^{-1}		
					Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)	Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)
Bishopville Prong	XDM4486	2003	4/16 to 12/22	5839	1761	30	60	2589	44	121
	XDM4486	2004	3/11 to 12/21	6806	1012	15	38	1943	29	75
	XDM4486	2005	3/2 to 12/20	6929	1121	16	34	1989	29	68
	XDM4486	2006	3/15 to 12/12	4591	484	11	16	1191	26	24
	XDM4486	2007	3/15 to 12/17	4156	496	12	18	1100	26	21
	XDM4486	2008	3/19 to 12/10	5961	491	8	15	1286	22	35
Turville Creek	TUV0021	2003	3/26 to 12/22	6342	397	6	14	1273	20	35
	TUV0021	2004	3/11 to 12/21	6802	384	6	17	1392	20	37
	TUV0021	2005	3/2 to 12/20	6214	299	5	13	1057	17	21
Public Landing	XBM8828	2005	3/2 to 12/20	6303	20	0	4	330	5	35
	XBM8828	2006	3/15 to 12/20	4768	68	1	10	733	15	41
	XBM8828	2007	3/15 to 12/12	5492	14	0	4	384	7	17
	XMB8828	2008	3/19 to 12/10	5688	22	0	8	424	7	18

Magothy River

Location	Station	Year	Date Range	Total Hours	Criteria < 3.2 mg L^{-1}			Criteria < 5.0 mg L^{-1}		
					Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)	Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)
Stonington	XHF3719	2001	5/1 to 12/31	5862	26	0.4	7	310	5	37
	XHF3719	2002	1/1 to 11/26	7102	40	1	7	288	4	23
	XHF3719	2003	3/27 to 11/7	5147	28	1	7	234	5	26
Whitehurst	CTT0001	2002	5/1 to 11/18	4383	98	2	9	817	19	47
	CTT0001	2003	3/28 to 11/7	5083	36	1	6	362	7	22

Patuxent River

Location	Station	Year	Date Range	Total Hours	Criteria < 3.2 mg L^{-1}			Criteria < 5.0 mg L^{-1}		
					Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)	Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)
Jug Bay	PXT0455	2003	4/4 to 12/31	6130	0	0	0	153	3	20
	PXT0455	2004	1/1 to 12/31	8000	5	0	2	590	7	35
	PXT0455	2005	1/1 to 12/4	8678	35	0	11	586	7	67
	PXT0455	2006	1/1 to 12/31	8464	2	0	2	498	6	43
	PXT0455	2007	1/1 to 12/31	7717	0	0	0	297	4	18
	PXT0455	2008	1/1 to 12/31	8779	13	0	4	584	7	22
Benedict	XED0694	2003	6/17 to 11/10	3288	206	6	8	1062	32	44
	XED0694	2004	4/9 to 10/29	4870	122	3	6	1051	22	37
	XED0694	2005	4/19 to 10/31	4434	413	9	9	1440	32	25
Pin Oak	XDE4587	2003	6/26 to 11/10	2804	43	2	9	292	10	43
	XDE4587	2004	3/3 to 11/29	6382	20	0	8	142	2	43
	XDE4587	2005	4/6 to 10/29	4077	69	2	15	306	8	30
	XDE4587	2006	6/26 to 11/10	3335	24	1	11	110	3	13
	XDE4587	2007	3/22 to 10/31	4058	31	1	7	245	6	17
CBL	XCF9029	2003	6/20 to 11/20	3142	27	1	7	288	9	44
	XCF9029	2004	3/1 to 12/29	6474	27	0	15	135	2	65
	XCF9029	2005	4/6 to 10/31	4182	39	1	16	288	7	39

Potomac River

Location	Station	Year	Date Range	Total Hours	Criteria < 3.2 mg L^{-1}			Criteria < 5.0 mg L^{-1}		
					Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)	Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)
Piscataway Creek	XFB2184	2004	4/21 to 11/1	3870	285	7	14	803	21	27
	XFB2184	2005	4/13 to 10/24	4656	268	6	16	740	16	91
	XFB2184	2006	3/21 to 10/31	5374	29	1	3	294	5	23
	XFB2184	2007	3/21 to 10/31	5007	23	0.5	5	151	3	11
	XFB2184	2008	3/31 to 11/3	4556	19	0.4	7	133	3	13
Indian Head	XEB5404	2009	8/28 to 11/18	1969	3	0.1	3	158	8	10
	XEB5404	2010	4/1 to 10/27	5016	21	0.4	4	477	10	19
Mattawoman Creek	XEA3687	2004	4/21 to 11/1	4489	11	0.2	2	29	1	5
	XEA3687	2005	3/31 to 10/24	4616	86	2	7	455	10	23
	XEA3687	2006	3/21 to 10/31	4699	33	1	14	152	3	42
	XEA3687	2007	3/21 to 10/31	5038	10	0.2	3	210	4	13
	XEA3687	2008	3/26 to 10/28	4990	2	0.04	1	111	2	6
	XEA3687	2009	4/13 to 11/18	5233	178	3	15	667	13	21
	XEA3687	2010	4/1 to 10/27	4665	726	16	25	1240	27	56
Fenwick	XFB0231	2004	4/21 to 10/27	4358	0	0	0	44	1	9
	XFB0231	2005	3/31 to 10/21	4763	3	0	2	143	3	11
	XFB0231	2006	4/5 to 10/31	3519	5	0	3	83	2	16
	XFB0231	2007	3/21 to 10/31	4219	0	0	0	36	1	8
	XFB0231	2008	3/26 to 10/21	4975	0	0	0	13	0	7
St. Georges Creek	XBF7904	2006	4/25 to 10/31	4536	59	1	11	600	13	36
	XBF7904	2007	4/3 to 10/30	4536	69	2	4	623	14	38
	XBF7904	2008	3/27 to 10/21	4885	223	5	15	1001	20	22

Other Select Stations

Location	Station	Year	Date Range	Total Hours	Criteria < 3.2 mg L^{-1}			Criteria < 5.0 mg L^{-1}		
					Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)	Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)
Sycamore Point	XHH3851	2005	4/1 to 12/31	6092	590	10	35	1530	25	61
	XHH3851	2006	1/5 to 12/31	6815	527	8	37	1258	18	108
	XHH3851	2007	1/1 to 12/20	6663	804	12	33	1356	20	90
	XHH3851	2008	1/1 to 12/31	8204	503	6	15	1212	15	30
Ft. McHenry	XIE5748	2004	3/20 to 10/26	4943	337	7	14	1087	22	47
	XIE5748	2005	3/23 to 7/13	2686	184	7	14	389	14	102
	XIE5748	2006	4/11 to 11/6	4427	356	8	19	1179	27	63
	XIE5748	2007	4/3 to 11/14	5162	712	14	52	1891	37	112
	XIE5748	2008	4/4 to 11/29	5617	986	18	23	2114	38	75
Betterton Beach	XJH2362	2006	4/1 to 10/26	4530	0	0	0	0	0	0
	XJH2362	2007	4/1 to 11/1	4931	0	0	0	2	0.04	1
	XJH2362	2008	4/1 to 10/29	4971	0	0	0	3	0.1	2
Sandy Point South	XHF0460	2004	5/6 to 11/18	4312	6	0.1	2	339	8	137
	XHF0460	2005	7/11 to 10/27	1937	0	0	0	60	3	6
	XHF0460	2006	3/24 to 10/19	4200	0	0	0	13	0.3	4
	XHF0460	2007	4/10 to 10/22	3128	4	0.1	1	86	3	11
	XHF0460	2008	3/24 to 10/22	4321	0	0	0	39	1	6

The total hours (per year) of DO collection at each station ranged from ~ 2000 to 8800 hours. DO criteria failure ranged from no failures (0%) to failing almost 1/2 of the time (44%). Duration of low DO events (failing instantaneous or 30-day mean criteria) ranged from 15 minutes to 137 hours (almost 6 days).

At all sites and years applying the 30-day criteria (5.0 mg L^{-1}) resulted in either no change or an increase in both % failure and maximum duration of failure compared to the instantaneous criteria (3.2 mg L^{-1}). These percent failure changes ranged from 0 to 26%.

Two newly analyzed sites in the Corsica River (Sycamore Point) and Chesapeake Bay (Sandy Point South) had failure rates similar to previously reported enriched sites (Ft. McHenry, Patapsco River and Bishopville Prong, MD Coastal Bays). Sycamore Point DO % failure ranged from 6 – 12% (instantaneous) and 15 – 25% (30-day) overall with maximum duration of low DO events ranging from 15 – 37 hours (< 3.2 mg L^{-1}) and 30 – 108 hours (< 5 mg L^{-1}). At Sandy Point South the maximum single duration of failure for the 30-day criteria (< 5 mg L^{-1}) reached 137 hours (the longest for all stations analyzed thus far) in 2004. Subsequent years showed this station with much lower DO criteria failures.

Most sites showed higher durations of failure as the percent total failure increased (Figure 2-1).

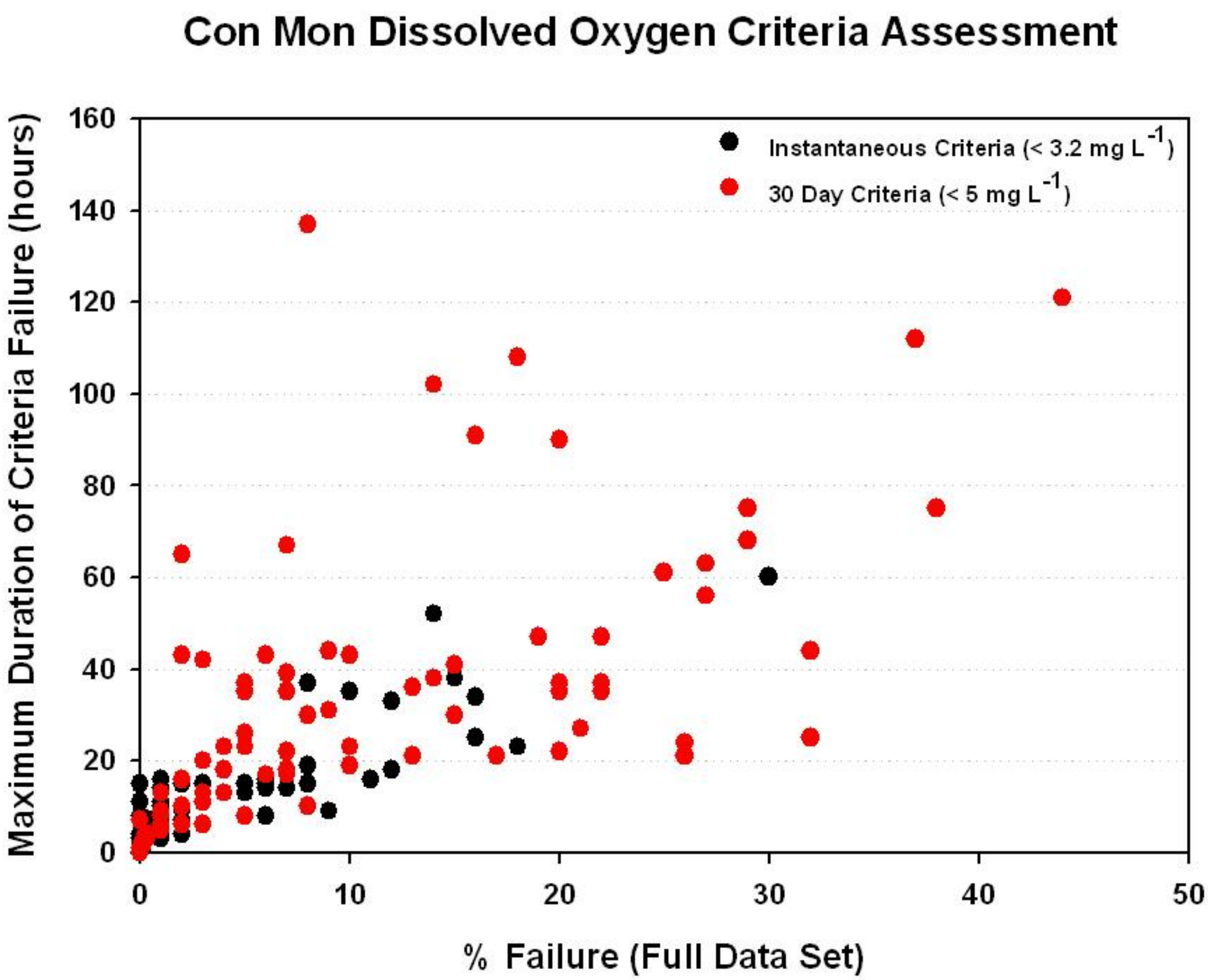

Figure 2-1. Percent failure (full data set) versus maximum duration of DO failure (hours) for all ConMon stations (N = 25) analyzed. Black dots represent data using instantaneous criteria (< 3.2 mg L^{-1}) and red dots represent 30 day criteria (< 5 mg L^{-1}).

Some stations experienced high duration of failures in years when the overall percent failure was fairly low (< 10%). We found no significant pattern or relationship using the stations we have analyzed thus far. We would like to expand this analysis to look at seasonal patterns in order to correct the total percent failure for total hours (season length) of sampling and to develop distributions of duration of failures.

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Chapter 3

Community Metabolism as an Indicator of Water Quality Impairment and Restoration

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3-1	INTRODUCTION AND OBJECTIVES	1
3-2	METHODS	2
3-2.1	BASIC CONCEPT FOR COMPUTING COMMUNITY PRODUCTION AND RESPIRATION	2
3-2.2	DESCRIPTION AND OPERATION OF METABOLISM MACRO	3
3-3	DATA SOURCES AND LOCATION	5
3-4	RESULTS	7
3-4.1	COMPARISONS OF METABOLISM IN MULTIPLE MARYLAND TRIBUTARY RIVERS	7
3-4.2	WEEKLY RATES OF PRODUCTION AND RESPIRATION	11
3-4.3	SENTINEL STATION TIME-SERIES	13
3-5	DISCUSSION	15
3-5.1	NEAR-SHORE VERSUS OFF-SHORE RATES OF PRIMARY PRODUCTION	15
3-5.2	DETECTION OF SIGNIFICANT CHANGES IN PRODUCTION	16
3-5.3	MULTI-YEAR TRENDS IN PRODUCTION	17
3-5.4	RELATIONSHIPS BETWEEN PRODUCTION AND NUTRIENT LOADING RATES	18
3-5.5	WEB USE OF CONMON BASED METABOLISM MEASUREMENTS	18
3-6	REFERENCES	19

3-1 Introduction and Objectives

Community production and respiration have repeatedly been shown to be responsive to nutrient enrichment in lakes (e.g., Vollenweider 1976) and many estuaries (e.g., Boynton et al 1982; Boynton and Kemp 2007). In the case of many Chesapeake Bay areas, nutrient enrichment was cited as one of the reasons for listing waterways as being impaired and in need of restoration. In many instances measurements of fundamental ecosystem processes such as primary production and respiration are too expensive or simply too difficult to undertake. However, the State of Maryland DNR established multiple water quality monitors making measurements of water quality variables needed to make these estimates. In this chapter we report on the methods and results of community production and respiration computations for multiple sites in Maryland tributary rivers.

System metabolism (i.e. community production and respiration; basically the production and utilization of organic matter) has gained broad application in estuarine areas. Perhaps the best single example of this was reported by Caffrey (2004) who assembled high frequency DO, temperature and salinity data from 42 sites located within 22 National Estuarine Research Reserves between 1995 and 2000. Caffrey computed the same metabolism estimates developed here and found the following: 1) highest production and respiration rates occurred in the SE USA during summer periods; 2) temperature and *nutrient concentrations were the most important factors explaining variation in rates* within sites; 3) freshwater sites were more heterotrophic than more saline sites; 4) *nutrient loading rates explained a large fraction of the variance* among sites and; 5) *metabolic rates from small, shallow, near-shore sites were generally much larger than in adjacent, but larger, deeper off-shore sites*.

The fact that nutrient loading rates and concentrations were strong predictors of rates is especially relevant to restoration efforts being made in Chesapeake Bay tributaries and the fact that near-shore rates were larger than off-shore rates is very relevant to issues related to DO criteria assessments. Additionally, Danish investigators have been using this technique in a variety of shallow Danish systems and they have started to use four different approaches for estimating the metabolic parameters of interest (Gazeau *et al*. 2005), including the open water DO approach. Their evaluations suggest that all techniques produce similar estimates of production or respiration. This convergence of estimates suggests a robust set of variables and that is consistent with the needs of a monitoring program.

This effort represents a continuing activity by the EPC of the Maryland Biomonitoring Program. This activity is consistent with the process-based approaches we have recommended for many years and this effort is another such example. The final algorithm we have adopted to compute metabolism was developed by David Jasinski, formerly with the Chesapeake Bay Program. The new algorithm is more efficient and has the capability of changing some parameters in the computation (e.g., air-water DO diffusion coefficient, time step in the computation). Because the ConMon system at each sampling site is in place for about 200 days per year (potentially every day from April through October) a large number of rate measurements (~200) of system production (related to nutrient conditions) and system respiration (related to hypoxia) can be made and examined. Such a large number of observations at a large number of sites is likely unprecedented in estuarine monitoring programs.

Specific objectives of this effort include the following: 1) a summary of mean rates of community P and R for multiple sites in Maryland tributary rivers; 2) qualitatively relating these rates to nutrient loading rates; and 3) continue work on a format for translating these data to a web page for use by Bay managers and the general public

3-2 Methods

3-2.1 Basic Concept for Computing Community Production and Respiration

The basic concept and method for computing community production and respiration was developed by Odum and Hoskin (1958) and, with numerous modifications, has been used since for estimating these rate processes in streams, rivers, lakes, estuaries and the open ocean. The technique is based on following the oxygen concentration in a body of water for a 24 hour period. During hours of daylight, oxygen concentration increases in the water due to the release of O_2 as a by-product of photosynthesis. During hours of darkness, O_2 concentration declines due to O_2 consumption by both primary producers and all other heterotrophs. The rate processes (gross photosynthesis, Pg*; nighttime respiration, Rn) are estimated by computing the rate of change in O_2 concentrations during day and night periods. This rate of change is then corrected for O_2 diffusion across the air-water interface and the result is an estimate of Pg* and Rn. ConMon data are exactly the type of data needed for these computations in that all the needed variables are measured (dissolved oxygen, temperature and salinity), the measurement frequency is high (15 minute intervals) and the measurement period is for 9 or more months. It is very rare when a rate process can be estimated with such temporal intensity.

3-2.2 Description and Operation of Metabolism Macro

Based on earlier work by Burger and Hagy (1998) for calculating community metabolism from near-continuous monitoring data, an automated Excel spreadsheet (Metabolism.xls) was developed by Mr. David Jasinski. The worksheet was automated using Microsoft's Visual Basic for Applications (VBA) programming language. Briefly, the steps the spreadsheet undertakes are as follows:

1. An excel file, containing the continuous monitoring data configured by the user in a requisite format (Figure 3-1) is read into the spreadsheet.

2. Dates and times are reformatted into a continuous time variable or serial number.

3. Sunrise and Sunset times for each date are calculated based on the latitude and longitude of the station.

4. Rows are inserted into the dataset to create an observation at sunrise and sunset on each day.

5. Each observation in the dataset is assigned a daypart – Sunrise, Day, Sunset, or Night

6. Each observation is assigned to a "Metabolic Day". Each metabolic day begins at sunrise on the current day and continues to the observation immediately before sunrise on the following day.

7. For sunrise/sunset observations created in Step 4, values for water temperature, salinity, dissolved oxygen and dissolved oxygen saturation are calculated by taking the mean of the observations immediately before and after sunrise and sunset.

8. The change in DO, time, air/sea exchange and oxygen flux is calculated between each consecutive observation.

9. The minimum and maximum DO values are calculated between sunrise and sunset on each day and these values are labeled "metabolic dawn" and "metabolic dusk".

10. Sums of the changes in DO, time, air/sea exchange and DO flux (step 8) are calculated for each metabolic day for the periods between sunrise and metabolic dawn, metabolic dawn and metabolic dusk, metabolic dusk and sunset, and sunset and the following sunrise.

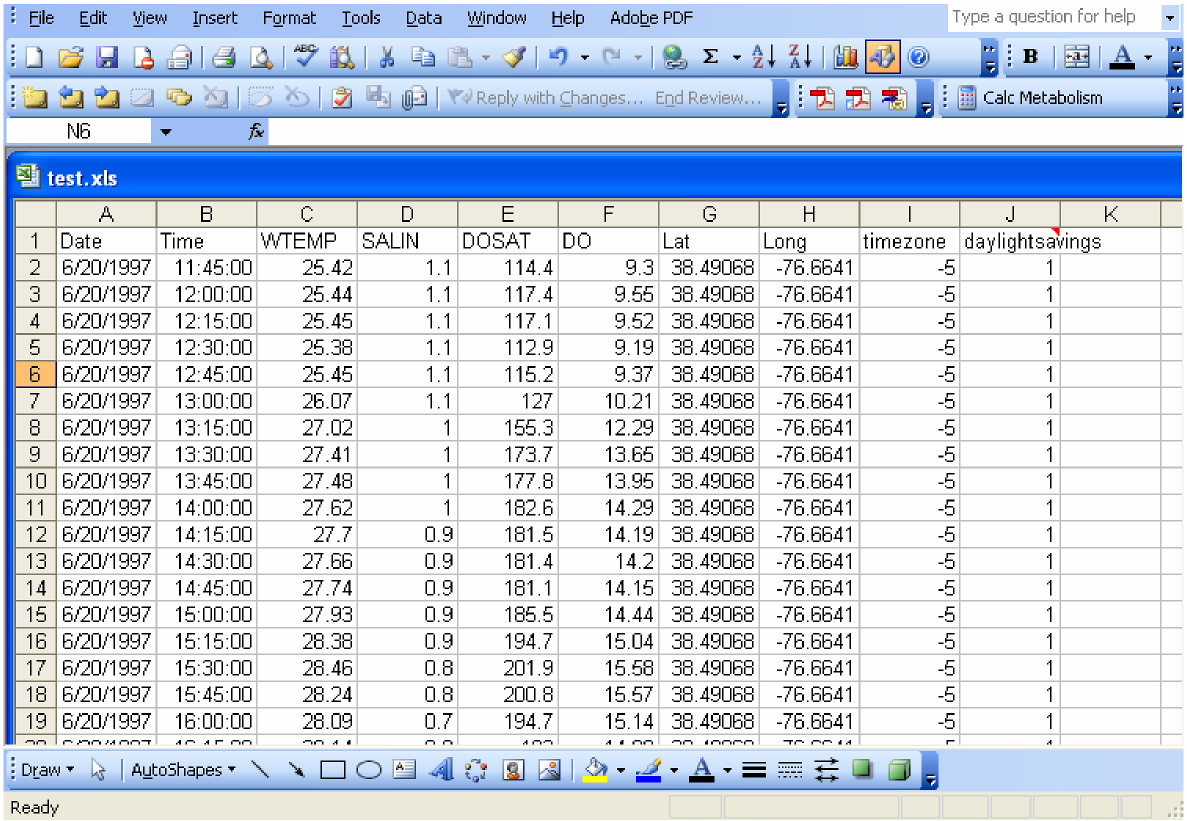
11. From these sums, 6 metabolic variables are calculated and these include: rn, rnhourly, pa, pa_star, pg, pg_star.

These variables are defined as follows:
rn = Nighttime (sunset to following sunrise) summed rates of DO flux corrected for air/water diffusion.
rnhourly = rn divided by the number of nighttime hours

pa = The sum (both positive and negative) of oxygen flux (corrected for air-water diffusion) for the dawn, day and dusk periods.
pa_star = summed oxygen flux (corrected for air-water diffusion) for the day period
pg = pa + daytime respiration. Daytime respiration = rnhourly * (number of hours of daytime+dawntime+dusktime).
pg_star = pa_star + daytime respiration as defined above.

Air-water diffusion of oxygen is considered in these computations and the diffusion correction is based on the difference between observed DO percent saturation and 100% saturation multiplied by a constant diffusion coefficient. For these computations a diffusion coefficient of 0.5 g O_2 m^{-2} hr^{-1} was selected as generally representative of conditions frequently encountered in estuarine tributary situations (Caffrey 2004).

One of the primary assumptions of this method is that temporal changes in DO measured by the continuous monitors are due solely to metabolism (i.e. oxygen production from photosynthesis and oxygen loss from respiration) occurring at the station and not due to advection of water masses with different oxygen conditions moving past the instrument. Because Chesapeake Bay is a tidal system, this may not always be the case. Depending on the hydrodynamics of a given station, this assumption may be more or less realistic and may also be variable from date to date. One way of censoring dates where DO is affected by advection is to preview the data graphically prior to metabolism calculations and determine if there is a relationship between salinity and DO. Large changes in salinity suggest moving water masses and therefore, advection. These dates could then be flagged and reviewed before metabolism variables are calculated.



	A	B	C	D	E	F	G	H	I	J	K
1	Date	Time	WTEMP	SALIN	DOSAT	DO	Lat	Long	timezone	daylightsavings	
2	6/20/1997	11:45:00	25.42	1.1	114.4	9.3	38.49068	-76.6641	-5	1	
3	6/20/1997	12:00:00	25.44	1.1	117.4	9.55	38.49068	-76.6641	-5	1	
4	6/20/1997	12:15:00	25.45	1.1	117.1	9.52	38.49068	-76.6641	-5	1	
5	6/20/1997	12:30:00	25.38	1.1	112.9	9.19	38.49068	-76.6641	-5	1	
6	6/20/1997	12:45:00	25.45	1.1	115.2	9.37	38.49068	-76.6641	-5	1	
7	6/20/1997	13:00:00	26.07	1.1	127	10.21	38.49068	-76.6641	-5	1	
8	6/20/1997	13:15:00	27.02	1	155.3	12.29	38.49068	-76.6641	-5	1	
9	6/20/1997	13:30:00	27.41	1	173.7	13.65	38.49068	-76.6641	-5	1	
10	6/20/1997	13:45:00	27.48	1	177.8	13.95	38.49068	-76.6641	-5	1	
11	6/20/1997	14:00:00	27.62	1	182.6	14.29	38.49068	-76.6641	-5	1	
12	6/20/1997	14:15:00	27.7	0.9	181.5	14.19	38.49068	-76.6641	-5	1	
13	6/20/1997	14:30:00	27.66	0.9	181.4	14.2	38.49068	-76.6641	-5	1	
14	6/20/1997	14:45:00	27.74	0.9	181.1	14.15	38.49068	-76.6641	-5	1	
15	6/20/1997	15:00:00	27.93	0.9	185.5	14.44	38.49068	-76.6641	-5	1	
16	6/20/1997	15:15:00	28.38	0.9	194.7	15.04	38.49068	-76.6641	-5	1	
17	6/20/1997	15:30:00	28.46	0.8	201.9	15.58	38.49068	-76.6641	-5	1	
18	6/20/1997	15:45:00	28.24	0.8	200.8	15.57	38.49068	-76.6641	-5	1	
19	6/20/1997	16:00:00	28.09	0.7	194.7	15.14	38.49068	-76.6641	-5	1	

Figure 3-1. Screen shot showing the required input format needed for Metabolism.xls for calculation of metabolism variables.

Another way of dealing with advection is to incorporate in the code a method of detecting changes in DO associated with changes in salinity. It might then be possible to apply a site specific correction factor to remove the advection affect on DO. These possibilities could be investigated further in the future. At the present time we examine data from each site graphically and if there are erratic patterns in dissolved oxygen or salinity we do not attempt calculations for that site. In addition, the algorithm indicates when a site has unusual dissolved oxygen patterns (e.g., increases in dissolved oxygen during hours of darkness) and these computations are excluded.

3-3 Data Sources and Location

All data in this section were obtained from the Maryland DNR (MDNDR) ConMon program (Cole 2011) and are available from the following website (http://mddnr.chesapeakebay.net/eyesonthebay/index.cfm).

Table 3-1. List of ConMon stations used in Metabolism calculations.

Station	Code	Year
Patapsco River		
Station	Code	Year
Baltimore Harbor (MCH)	XIE5748	2009
Baltimore Harbor (MCH)	XIE5748	2010
Corsica River		
Station	Code	Year
Sycamore Point (COR)	XHH3851	2009
Sycamore Point (COR)	XHH3851	2010
Possum Point Surface (PPT)	XHH4931	2008
The Sill Surface (SIL)	XHH4916	2008
Magothy River		
Station	Code	Year
Whitehurst (WHI)	CTT0001	2008
Elk River		
Station	Code	Year
Locust Point Marina(LOC)	XKI3890	2008
Hollywood Beach (HOL)	XKI0256	2008
Bohemia River		
Station	Code	Year
Long Point (BOH)	XJI8369	2008
South River		
Station	Code	Year
Harness Creek Upstream (HCU)	ZDM0002	2008
Harness Creek Downstream (HCD)	ZDM0001	2008
Potomac River		
Station	Code	Year
Piscataway (PIS)	XFB2184	2004
Piscataway (PIS)	XFB2184	2005
Piscataway (PIS)	XFB2184	2006
Piscataway (PIS)	XFB2184	2007
Piscataway (PIS)	XFB2184	2008
Mattawoman (MAT)	XEA3687	2004
Mattawoman (MAT)	XEA3687	2005
Mattawoman (MAT)	XEA3687	2006
Mattawoman (MAT)	XEA3687	2007
Mattawoman (MAT)	XEA3687	2008
Mattawoman (MAT)	XEA3687	2009
Mattawoman (MAT)	XEA3687	2010
Indian Head (IND)	XEB5404	2009
Indian Head (IND)	XEB5404	2010
Bretton Bay (BBY)	XCD5599	2008
St. Mary’s River		
Station	Code	Year
St. Mary’s College (SMC)	XCF1440	2008

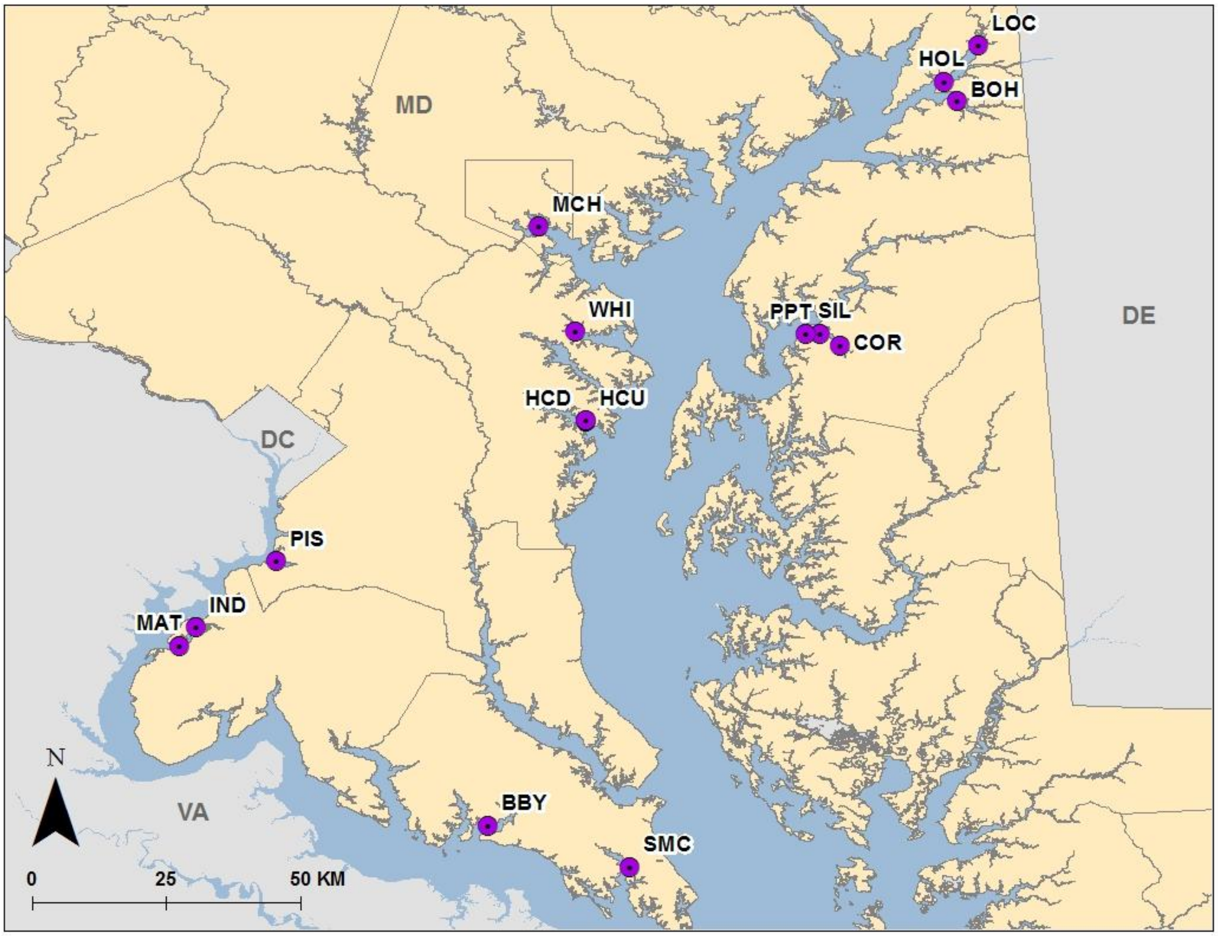


Figure 3-2. A map showing the location of ConMon sites used in all analyses in this chapter.

3-4 Results

3-4.1 Comparisons of Metabolism in Multiple Maryland Tributary Rivers

We begin this discussion by presenting summer rates (June-August) of community metabolism at multiple sites in tributaries of the Maryland portion of Chesapeake Bay. We chose to focus on summer rates because it is now clear that these rates are at maximum levels in these generally enriched systems during those months and because this is the period of the year during which DO criteria failures are most common. Data are organized in Table 3-2 by the following categories: a) Potomac River tributaries; b) Western Shore tributaries; c) Upper Bay tributaries; and d) Corsica River sites.

At the Potomac River tributary sites rates of Pg* generally decreased from upstream to downstream locations. Average summer rates ranged, across all sites, from about 3.6 to 17 g O_2 m^{-3} day^{-1}. Pg* to Rn ratios (organic matter produced during the day to organic matter consumed during the night) ranged from 1.4 to 4.5. Summer respiration ranged from 1.3 to 4.7 g O_2 m^{-3} $night^{-1}$.

Table 3-2. A summary of community metabolism rates (Pg* and Rn) for summer periods (June-August) for a selection of Chesapeake Bay ConMon sites. Data were organized by tributary or portion of Chesapeake Bay and this is indicated at the top of each table. * Note that results for Indian Head in 2009 contain data only for August.

Summer Metabolism (June-August)

Potomac River Tributaries						
Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Piscataway	2004	-4.2	2.3	12.7	5.1	3.0
Piscataway	2005	-3.7	2.1	12.9	4.5	3.5
Piscataway	2006	-3.9	1.8	13.8	4.4	3.5
Piscataway	2007	-4.7	2.0	16.9	5.1	3.6
Piscataway	2008	-4.0	2.3	14.9	5.7	3.7
Indian Head	2009*	-2.0	0.4	3.6	0.1	1.9
Indian Head	2010	-2.0	1.3	5.1	1.8	2.6
Mattawoman	2004	-1.3	0.7	7.5	1.9	1.4
Mattawoman	2005	-1.6	1.3	6.3	2.4	3.9
Mattawoman	2006	-2.8	1.4	10.1	3.3	3.6
Mattawoman	2007	-2.2	1.2	9.8	3.5	4.5
Mattawoman	2008	-2.3	1.2	8.5	2.5	3.7
Mattawoman	2009	-2.2	1.8	5.9	3.3	2.7
Mattawoman	2010	-2.5	1.3	8.1	2.2	3.2
Bretton Bay	2008	-3.6	1.7	9.5	3.2	2.6
St. Mary's College	2008	-3.5	2.2	8.4	4.2	2.4

Summer Metabolism (June-August)

Western Shore Tributaries						
Patapsco River						
Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Baltimore Harbor	2009	-4.6	1.9	12.2	4.7	2.6
Baltimore Harbor	2010	-4.9	2.2	13.1	5.5	2.7
Magothy River						
Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Whitehurst	2002	-2.7	1.3	7.0	2.1	2.6
Whitehurst	2003	-2.1	1.3	7.0	2.2	3.4
South River						
Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Harness Creek Upstream	2008	-4.7	1.5	12.6	3.0	2.7
Harness Creek Downstream	2008	-4.6	1.5	13.2	3.5	2.9

Summer Metabolism (June-August)

Upper Bay Tributaries						
Elk River						
Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Locust Point Marina	2008	-2.2	1.1	8.7	2.1	3.9
Hollywood Beach	2008	-1.4	0.6	4.1	1.7	2.8
Bohemia River						
Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Long Point	2008	-2.1	0.8	5.9	1.7	2.8

Summer Metabolism (June-August)

Corsica River						
Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Sycamore Point	2009	-4.5	2.2	15.4	5.3	3.5
Sycamore Point	2010	-4.5	2.2	15.6	5.8	3.5
Possum Point Surface	2008	-3.3	1.1	10.1	2.5	3.1
The Sill Surface	2008	-2.6	1.2	8.0	3.4	3.1

Rates of Pg* in the Western Shore tributaries ranged from 7 to 13.2 g O_2 m^{-3} day^{-1}. The Magothy River station exhibited the smallest rates of Pg* and rates remained the same from 2002 to 2003. Rates of Pg* were similar in both the Patapsco and South Rivers. Pg* increased slightly in Baltimore Harbor from 2009 to 2010, but were close to Pg* rates calculated in 2007 and 2008 (Boynton *et al.*, 2011). Respiration ranged from 2.1 to 4.9 g O_2 m^{-3} $night^{-1}$ and again was similar in the Patapsco and South Rivers. Pg* to Rn ratios were generally similar in the Western Shore tributaries, ranging from 2.6 to 3.4.

The Upper Bay tributaries exhibited moderate rates of Pg*, ranging from 4.1 to 8.7 g O_2 m^{-3} day^{-1}. Community metabolism rates in these tributaries were calculated for 2008 only. Hollywood Beach in the Elk River had the lowest rates of Pg*, while further upstream in the Elk River, at

Locust Point Marina, rates were roughly two times higher. Nighttime respiration ranged from 1.4 to 2.2 g O_2 m^{-3} $night^{-1}$. The Pg* to Rn ratio in the Bohemia River was 2.8 and in the Elk River the Pg* to Rn ratio ranged from 2.8 to 3.9.

Like the Potomac River, rates of community metabolism in the Corsica River decreased from upstream to downstream locations. However the gradient in the Corsica was much more severe. At The Sill, downstream in the Corsica, the 2008 average summer rate of Pg* was 8.0 g O_2 m^{-3} day^{-1}. Upstream at Sycamore Point, rates of Pg* are nearly double at 15.4 and 15.6 g O_2 m^{-3} day^{-1} for 2009 and 2010 respectively. Pg* to Rn ratios throughout the river are about the same, ranging from 3.1 to 3.5. Respiration ranges from 2.6 at The Sill to 4.5 at Sycamore Point.

3-4.2 Weekly rates of Production and Respiration

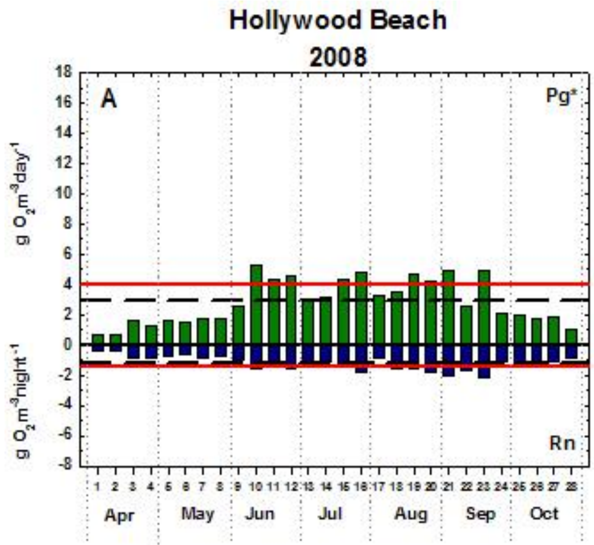
Weekly metabolism rates were calculated by averaging the following days for each month: 1-7 (week 1); 8-14 (week 2); 15-21 (week 3); 22-end of month (week 4). We calculated weekly means when 3 or more days of data were available for a given ‘week’. Weeks with less than 3 days were not included. Week 4 calculations could have as many as nine values included in the average, depending on the number of days in that month.

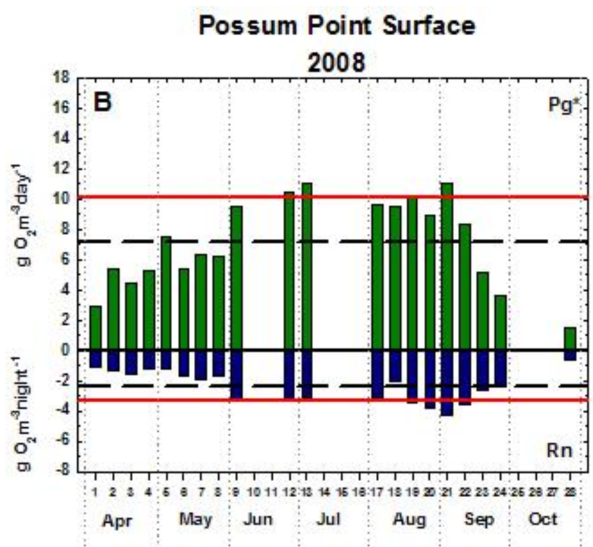
Here we present a selection of ConMon stations ranging from low to high rates of weekly community metabolism (Figure 3-3). Low rates of community metabolism were found at Hollywood Beach in the Elk River, where summer mean Pg* and Rn are 4.1 and 1.4 and annual Pg* and Rn are 2.9 and 1.2. Metabolism gradually increases through April and May, with peak production occurring in the second week of June through mid-September. In late-September and October metabolism rates decrease to about half of what the summer high rates were.

Community metabolism rates at Possum Point in the Corsica River were moderate. Summer Pg* and Rn were 10.1 and 3.3. Annual mean rates of Pg* and Rn were 7.1 and 2.4. Metabolism rates at this station appear to rise faster in April and May and peak in June into early September. Though much of the October data was flagged, it does appear that metabolism rates decline relatively fast through September and October.

Community metabolism rates at the downstream Harness Creek station were high. Primary production rates ranged from about 3 to 17 g O_2 m^{-3} day^{-1}, with a summer average of 13.1 and an annual average of 10.6. Respiration rates ranged from around 1.0 to 6.5 g O_2 m^{-3} $night^{-1}$, with a summer average of 4.6 and an annual average of 3.9. June through August rates were about two to three times as high as April rates; highest rates occurred in July.

All three sites exhibited a similar temporal trend with rates increasing in April and May, reaching their peak in June through September and then decreasing in mid- to late- September and October.


Hollywood Beach
2008
A
Pg*
Rn
g O2m-3day-1
g O2m-3night-1
Apr
May
Jun
Jul
Aug
Sep
Oct


Possum Point Surface
2008
B
Pg*
Rn
g O2m-3day-1
g O2m-3night-1
Apr
May
Jun
Jul
Aug
Sep
Oct

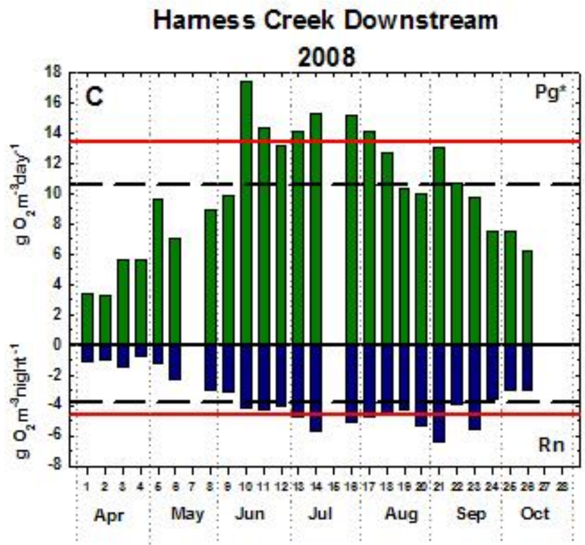


Figure 3-3. Bar graphs of mean weekly rates of gross primary production (Pg*) and community respiration (Rn) showing: (A) low rates at Hollywood Beach in the Elk River; (B) moderate rates at Possum Point surface in the Corsica River; (C) high rates at Harness Creek downstream in the South River. The red lines represent summer (June-August) mean rates of Pg* and Rn. The dashed lines represent annual (April-October) mean rates of Pg* and Rn.

3-4.3 Sentinel Station Time-Series

We have calculated metabolism for a seven year period at the Baltimore Harbor station in the Patapsco River (Figure 3-4). We generally observed similar temporal trends previously seen at ConMon stations, however there are some interesting observations to be made with this longer time series. During 2004-2009 we saw a substantial difference in Pg* rates from spring to summer and summer to early fall, where spring and fall rates ranged from about 3 to 8 g O_2 m^{-3} day^{-1} and summer rates range from 8 to 16. In 2010, spring and fall rates are higher at around 8 g O_2 m^{-3} day^{-1}. 2010 also has the highest summer and annual mean rates of Pg*. There is a slight increase in Pg* during the period of record. The largest increase occurred in 2007 and 2008, also the time when flow was the lowest (Figure 3-5).

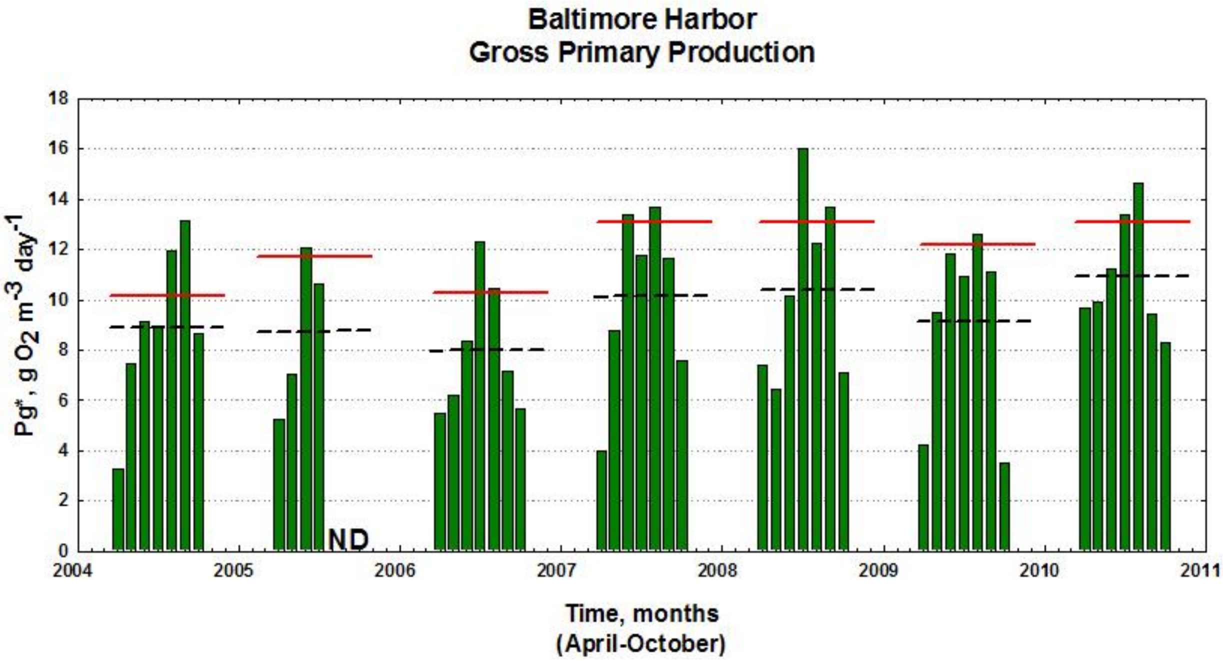


Figure 3-4. Bar graphs of monthly rates of gross primary production (Pg*) at Baltimore Harbor (MCH). The red lines represent summer (June-August) mean rates of Pg*. The dashed lines represent annual (April-October) mean rates of Pg*. ND is an indication of no data for a period of time.

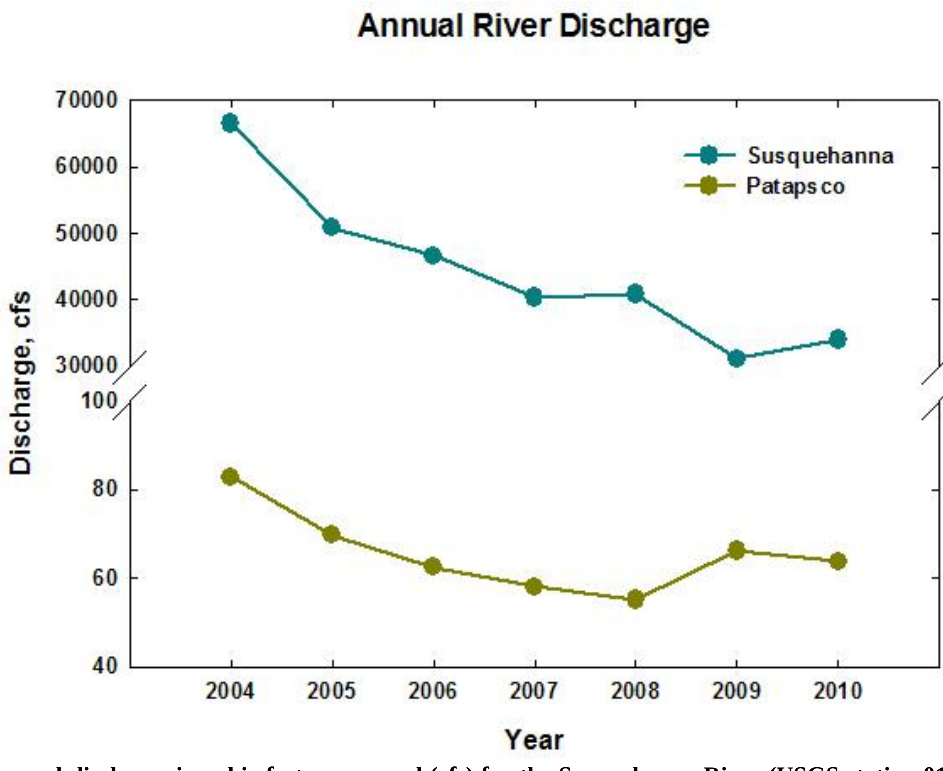


Figure 3-5. Annual discharge in cubic feet per second (cfs) for the Susquehanna River (USGS station 01578310) and the Patapsco River (USGS station 01586000). Data are from the USGS surface-water annual statistics (http://waterdata.usgs.gov/md/nwis/sw).

3-5 Discussion

3-5.1 Near-shore versus Off-shore Rates of Primary Production

We begin this discussion of Bay metabolic rates by comparing rates measured at ConMon sites (shallow, near-shore locations) with similar rates measured at various locations along the mainstem of the Bay. The basic question addressed here concerns the potential for substantially different rates of production between these two habitats. We believe the general understanding is that volumetric rates offshore are smaller than the same rates in shallow areas (e.g., Caffrey 2004). The reason for this might be related to the fact that shallow areas can have a suite of primary producers (SAV, benthic microalgae, macroalgae and phytoplankton) while deeper area primary production is based solely on phytoplankton. In addition, shallow areas are proximal to land-based nutrient supplies and sediment re-cycling of nutrients is in very close proximity to the suite of primary producers in shallow areas but often separated by a pycnocline in deeper waters.

We have selected several data sources for these comparisons and those include the work of Boynton *et al.* (1982), Scardi and Harding (1999), Smith (1992) and Boynton and Kemp (2000). All these investigators measured primary production rates in the mainstem Bay for multi-year periods. In these cases, as in many instances where different data sets are compared, there is an issue concerning the methods of measurement. The ConMon measurements are based on an oxygen approach (Odum and Hoskin 1958) as were those of Smith (1992). Scardi and Harding (1999), and data presented in Boynton *et al.* (1982), and Boynton and Kemp (2000) used a C-14 approach. To make useful comparisons we have converted the oxygen-based estimates of production to carbon units using a conversion factor of 1 unit of oxygen-based production equals 0.3 units of carbon. This conversion used a photosynthetic quotient of 1.25 (ratio of CO_2 reduced to O_2 liberated in photosynthesis) and the normal stochiometric equivalents associated with carbon fixation.

Scardi and Harding (1999) reported primary production rates from the mainstem Bay ranging from close to zero to about 3.0 g C m^{-2} day^{-1}. Boynton and Kemp (2000) reported annual average rates ranging from about 1 to 2.5 g C m^{-2} day $^{-1}$ in the mesohaline region of the Bay. Smith (1992) reported peak summer rates for the north, middle and south Bay of 1.2, 2.6 and 2.8 g C m^{-2} day $^{-1}$. Boynton *et al.* (1982) reported similar rates for the mesohaline Bay except for the year of tropical storm Agnes (June, 1972) and the following year when peak summer rates ranged between 6 and 8 g C m^{-2} day $^{-1}$. These latter rates are very large compared to others in the record and attest to the impact of this record storm event. The general impressions we get from review of these data are 1) peak production rates occurred during summer and 2) under normal circumstances rates were lower in the upper Bay and higher in the mid and lower Bay, often in the range of 2.0 – 3.0 g C m^{-2} day $^{-1}$. The ConMon data from which we computed production rates spanned the range of rates observed in open waters of the Bay. For example, in Baltimore harbor summer rates were consistently higher than summer rates in the Bay and averaged about 4 g C m^{-3} day^{-1}. Likewise, in Piscataway Creek (tributary of the upper tidal Potomac), rates were even higher (~4.5 g C m^{-3} day^{-1}) as they were in the upper portion of the Corsica and South Rivers. All of these "high rate" locations have several things in common. First, they are all exposed to large nutrient loads. While we do not yet have quantitative measures of load magnitude for all of these sites, the information we do have indicates high loading rates for all these systems (e.g., Bailey et al 2008). Second, all sites with especially high rates are in tributary locations or are tributaries of tributaries. In short, they tend to be smaller systems and tend to have some restriction between the site and more open and larger water bodies. The factors causing these higher rates very likely involves an excessive nutrient supply but also restricted circulation (reduced flushing) that allows time for efficient use of nutrients by primary producers. There were also a few sites having summer production rates comparable to or somewhat lower than those routinely measured in open Bay waters. These included sites where management actions have reduced the nutrient load (e.g., Indian Head) and sites that were more proximal to open waters (e.g., Elk River). However, the clear message resulting from these computations is that production rates in these shallow waters are comparable to or are higher than those typically measured off-shore. These enhanced shallow water rates occur even though the nominal water column at ConMon sites used in these computations was one meter at all sites; many stations in the mainstem bay had much larger euphotic zones (~2-4 m) in which production was generated. Thus, the nearshore sites also had more concentrated production rates. These results are consistent with those reported by Caffrey (2004) for a much larger selection of measurements made at NERS sites.

It is also useful to “look backwards” and compare rates of Pg* collected in the Cory Historical Data Set we reported on earlier (Boynton *et al.*, 2011). These data were collected in the mesohaline region of the Patuxent River estuary between 1964 and 1969, a period of time prior to large increases in nutrient loading rates in this system. During the years 1964-1966 average rates of Pg* (converted to carbon equivalents) were about 0.9 g C m^{-3} $day^{-1}$and these increased to about 1.2 g C m^{-3} $day^{-1}$by 1968. All of these “pre-eutrophication” rates measured in the Patuxent are considerably lower than current rates and lower or much lower than those reported above for moderately and very enriched systems. One of the real values of the Cory data set is that it provides a rare view into Bay characteristics during the last stages of the pre-eutrophication period and, in a sense, gives us a target to aim for in restoration efforts.

3-5.2 Detection of significant changes in production

Several years ago we conducted an analysis to determine the minimum significant difference (MSD) needed for metabolism parameters (Pg* and Rn) to be considered statistically different. This is an important issue as ultimately we want to determine if Bay metabolic rates are declining in response to management actions. This work was lead by Elgin Perry, a research statistician very familiar with Chesapeake Bay issues and data (Bailey *et al.*, 2008). This exercise was problematic because the MSD is sample size dependant (e.g., not every 24 hr record of data is useful in metabolism computations) and there is an issue of auto-correlation for Pg* (e.g., large values tend to be followed by large values and small values tend to be followed by small values). Statistical advice suggested that if metabolic differences exceeded 2.5 standard errors the difference was significant. In most months where Pg* and Rn were computed at least 20 days (often 25 days during summer months) of results were available and the needed differences for significance were between 2.1 and 2.4 g O_2 m^{-3} $day^{-1}$for Pg* and between 0.7 and 0.8 g O_2 m^{-3} $night^{-1}$ for Rn.

These results clearly indicate that we can readily distinguish between heavily enriched, moderately enriched and less enriched sites in the Bay system. For example, the mean Pg* values for the Mattawoman Creek ConMon site were about 8.2 g O_2 m^{-3} day^{-1} (during a 9 year period) and were 12.0 g O_2 m^{-3} $day^{-1}$for the Fort McHenry ConMon site in Baltimore Harbor (during a 8 year period). These rates are clearly and significantly different. Similar results were found when comparing other sites with varying degrees of impact.

We also want to detect differences at a single site where managements actions have taken place or weather conditions were such that metabolism could be expected to be elevated or suppressed. The answer again appears to be yes. In the case of Mattawoman Creek the range in Pg* values during a nine year period was from 6.3 to 10.1 g O_2 m^{-3} day^{-1}. There were in this time series several years that were statistically higher or lower than other years. It appears that this tool is sensitive enough to distinguish between sites with varying degrees of nutrient impact and to discern either improving or deteriorating water quality conditions at a single site.

3-5.3 Multi-year trends in production

The ConMon Program has established several sites where monitoring during the April-October period was been continued for more than three years which is the normal rotation time for a ConMon site. These areas where monitoring has continued are referred to as sentinel sites and the concept here is to observe both longer term variability in a variety of water quality parameters and have a record at sites either undergoing or expected to undergo strong management actions. We have examined metabolism data (Pg*) for two such sites, one being in Mattawoman Creek (7 year record; 2004-2010) and the other at Ft McHenry in Baltimore Harbor (7 year record; 2004-2010).

In the case of the Fort McHenry site, rates of Pg* were generally high (summer rates > 10 gO_2 m^{-3} day^{-1}, indicative of a nutrient enriched system. Peak rates were always observed during June-September in all years and occurred mainly during the July-August period. Summer rates of Pg* during 2004 and 2006 were significantly lower than rates measured during 2007 – 2010. During the full monitoring period (April – October) rates of Pg* exhibited the same pattern but differences were smaller among years. Developing a local (Baltimore Harbor) nutrient input budget was beyond the scope of this effort but we are working with Bay Program staff to obtain model-based estimates of nutrient loads to this system to examine metabolism data for responses to nutrient load changes. In place of those data we did examine annual river discharge from the Patapsco and the Susquehanna Rivers as a surrogate for nutrient loads (Figure 3-5). During the period of record, flows from both rivers declined suggesting a reduction in nutrient loads to this system and this is opposite the response that might be expected from a variable such as Pg*. However, other factors such as changes in point source discharges and harbor water residence time also come into play and may be responsible for the increased rates of Pg* It is also quite possible that using river flow is not an adequate substitute for estimating nutrient loading rates.

We also examined the time-series of Pg* based on Mattawoman Creek data. In this case rates of Pg* both increased and decreased during the period of record (2004-2010). Rates were lower than at the Fort McHenry site consistent with lower nutrient loading in the Mattawoman system (see Chapter 5 in this report). We also examined summer Pg* data for relationships with external dissolved inorganic nitrogen (DIN) loads. We found a weak but positive relationship to DIN loads using DIN load and Pg* data for the same year. However, a much stronger relationship emerged (hyperbolic model; $r^2 = 0.72$) when an average lag of two years was introduced into the analysis (i.e., Pg* rates this year related to DIN loads this year and the previous year). The idea that these systems retain some nutrient memory has been receiving increasing support in recent years and this may be another example of that feature. Lags and thresholds are system features we need to keep in mind when examining water quality and habitat data from responses to either weather conditions or management actions.

3-5.4 Relationships between production and nutrient loading rates

The estuarine literature has a growing number of examples relating nutrient loads to system performance, often expressed as algal biomass (chlorophyll-a) or some rate measurement (e.g., Pg* or some related variable). Several of these studies have been examined in Boynton and Kemp (2008). To produce such an analysis for Chesapeake Bay and tributary rivers requires system-specific nutrient loading rates including both point and diffuse sources. During the past year we have worked with Bay Program staff to obtain freshwater flow, nitrogen, phosphorus and sediment loads (month and annual basis) for a variety of Maryland and Virginia systems. This process was not as simple as first expected and required considerable effort on the part of Bay Program staff. We recently received large portions of these data are have begun the process of arranging these files for use in this analysis. We have indicated that completion of this analysis is one of several primary goals of the FY 2013 EPC contract. We should mention that in addition to nutrient loading rate relationship to algal biomass and production rates we will also be considering other key variables that have been found to influence rates and biomass in estuarine system (e.g., estuarine flushing time, water clarity, depth and other morphometric features).

3-5.5 Web use of ConMon based metabolism measurements

For the past few years the idea of placing metabolism computations on the Eyes on the Bay website has been discussed with MD-DNR and CBL staff. For a variety of reasons we have not been able to have this happen. Perhaps with our increased ability to produce metabolism computations and increases in web expertise at DNR we can move forward on this potential product. There are several attractive features of doing this and these include the following: 1) metabolism is a fundamental process in all estuarine ecosystems, knowing what these rates are is similar in importance to farmers knowing how well crops are growing; 2) there is ample evidence that these rates respond, sometimes in complicated ways, to changes in nutrient loads. It would be very instructive to feature on the web site places that are undergoing strong management actions and be able to literally watch the system respond. At very least, people would be able to watch the rates increase and decrease as the seasons progress; 3) we have available data from a variety of sites, including one site with data from the 1960s representing conditions before serious nutrient effects occurrence. The chance to show people comparisons between sites and with earlier data is very compelling; and 4) the web site would be dynamic because there are both small and large changes in metabolism on a daily, weekly and seasonal time scales…that should add interest to the site. We will continue to meet with DNR staff to see if this project can move forward given time and fiscal constraints.

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Chapter 4

Spatial Analysis of Water Quality Conditions and Drivers

L.A. Wainger and A.R. Bayard

4-1	GOALS	1
4-2	INTRODUCTION	1
4-3	STUDY AREA	3
4-4	METHODS	5
4-4.1	DATA SOURCES	5
4-4.2	SPATIAL DATA ANALYSIS TECHNIQUES	7
4-4.2.1	KRIGING TECHNIQUES	7
4-4.3	SPATIO-TEMPORAL SUMMARIES	8
4-4.4	INTEGRATED SPATIAL ASSESSMENT	8
4-4.5	ELASTICITIES AND REGIONAL COMPARATIVE STATISTICS	8
4-5	RESULTS AND DISCUSSION	9
4-5.1	INTRA-ANNUAL VARIABILITY	9
4-5.1.1	SALINITY SPATIAL PATTERNS	9
4-5.1.2	CHL-*A* SPATIAL PATTERNS IN SPACE AND TIME	14
4-5.1.3	RELATIONSHIP OF CHL-*A* PATTERNS TO SAV	22
4-5.2	INTEGRATED ECOSYSTEM ASSESSMENT	23
4-5.3	INTERANNUAL VARIABILITY	29
4-5.3.1	PERCENTAGE CHANGE IN DRIVERS AND RESPONSE VARIABLES	29
4-5.3.2	ELASTICITIES CALCULATED WITHIN A SUB-ESTUARY	32
4-5.3.3	CROSS-SOURCE (LOCAL VS. REGIONAL) ELASTICITY	32
4-5.3.4	LOCAL SHARE STATISTIC	33
4-6	CONCLUSIONS AND MANAGEMENT IMPLICATIONS	35
4-6.1	FUTURE DIRECTIONS	35
4-7	REFERENCES	37

4-1 Goals

This analysis explored seasonal and annual spatial variability of water quality conditions in four shallow sub-estuaries of the Chesapeake Bay to promote our goal of understanding the potential responsiveness of sub-estuaries to management actions. The four sub-estuaries we evaluated this year were combined with data from four other estuaries evaluated last year to improve our ability to detect relationships between drivers and water quality outcomes measured as elevated chlorophyll-*a* or SAV abundance. The spatial data were used to evaluate whether estuaries have distinct water quality zones with localized differences in water quality characteristics that persist through time. We also compared interannual changes in sub-estuaries to mainstem conditions to understand the strength of connections between the local and regional waterbodies.

4-2 Introduction

Shallow water conditions affect the habitat of a wide variety of organisms and influence multiple benefits that people derive from coastal systems. Therefore, understanding the controls on shallow water conditions provides information necessary to improve the benefits produced in this zone of maximum human-estuary interaction. While the drivers of seasonal patterns of water quality conditions are fairly well understood, the spatial patterns of water quality conditions have not been as thoroughly explored, particularly at the scale of small sub-estuaries or zones within a given estuary. Temporal or seasonal changes in net primary production and dissolved oxygen are thought to be largely controlled by temperature and nitrogen (N) supply during the season of peak productivity in most estuarine systems. However, phosphorus (P) and the N:P ratio can also be controlling factors during some time periods and in some estuaries.

The same variables that explain temporal variation can explain spatial variability, but estuarine physical factors appear to play an equally important, or more important, role in the response of sub-estuaries or estuarine zones to nutrient inputs. Factors such as water flushing time, tidal range, the ratio of watershed area to estuarine area, circulation patterns, and other physically-based conditions are all recognized as important variables determining water quality for a given load of nutrients. Therefore, teasing apart the relative importance of internal estuarine variables from landward and seaward drivers of conditions is necessary for understanding sources of variability in water quality conditions.

In the Chesapeake Bay, shallow sub-estuaries can be strongly affected by drivers within both the local watershed and the Bay mainstem. Tidal circulation and relative intensity of loadings can result in either net import or net export of nutrients from any given estuary (Boynton *et al*. 2008, Shen and Wang 2007). As a result, it is important to examine conditions in both regions when seeking to understand the influence of many interacting stressors.

The work described in this report builds upon prior year's work evaluating spatial patterns of water quality variability in shallow sub-estuaries of the Chesapeake Bay, primarily on the Western Shore of the Bay. Our analyses use DATAFLOW© monitoring data to examine how potential drivers of water quality vary by location and which estuarine characteristics appear to control the system response to nutrient and sediment inputs. Because of its high spatial resolution, DATAFLOW© reveals patterns and patchiness of conditions in estuaries that would not otherwise be captured. In particular, the data provide useful information on the spatial extent of adverse conditions to better understand their significance for living resources (e.g., whether patchiness provides refugia).

We use two primary response variables to characterize the shallow estuarine systems in this study: chlorophylla (chl-*a*) and extent of submerged aquatic vegetation (SAV) relative to its historic distribution. Chl-*a* measures the concentration of phytoplankton and can be used to identify algal bloom events and their effects on water quality. The amount or concentration of phytoplankton (or algae in general) is considered an important water quality indicator since it directly indicates water clarity and light penetration and, when elevated, suggests potential risk of the presence of harmful algae and depleted oxygen. Phytoplankton growth, which is stimulated by nutrient influx from urban and agricultural sources, contributes to seasonal hypoxia and anoxia in deep and shallow waters of the Bay. Such water quality conditions can have widespread effects on the aquatic ecosystem by altering food webs.

In all analyses, the goal was to summarize data over space and time (a season, year or multi-year period) in the most ecologically relevant ways. For example, rather than using mean chl-*a* from a few scattered stations, we used the spatial detail of DATAFLOW© to estimate the area of the estuary that had elevated chl-*a* for a given month and year and used the repeated sampling to evaluate the frequency with which the elevated chl-*a* occurred. We considered the area over which chl-*a* was elevated using a threshold of 15 µg/l to define "elevated" because this has been suggested to be a limiting factor on SAV habitat quality (Batiuk *et al.* 2000). Also, we evaluated whether the chl-*a* was elevated at least 20% of the time (equivalent to 2+ cruises) to capture persistent conditions. The spatial data thus allow us to consider a more complete picture of habitat conditions.

4-3 Study Area

Four case study subestuaries were added to a growing database of water quality and watershed variables that can be used to describe areas of the Chesapeake Bay. The four estuaries added this year include the: Gunpowder, Middle, South, and West/Rhode (Figure 4-1). DATAFLOW© cruise data used in the analyses ranged from years 2003-2006 (Table 4-1). As with the previously chosen subestuaries, which included the: Bush, Corsica, Magothy, and Severn (EPC Report #28, Boynton *et al.* 2011), these additional four were chosen for analysis because their relatively small size and shallowness was expected to make them more responsive to watershed inputs. The Middle, South, and West/Rhode rivers fall within the Mesohaline zone (5-18 ppt), and the Gunpowder river falls within the Oligohaline zone (0.5-5 ppt) in the upper half of the Bay.

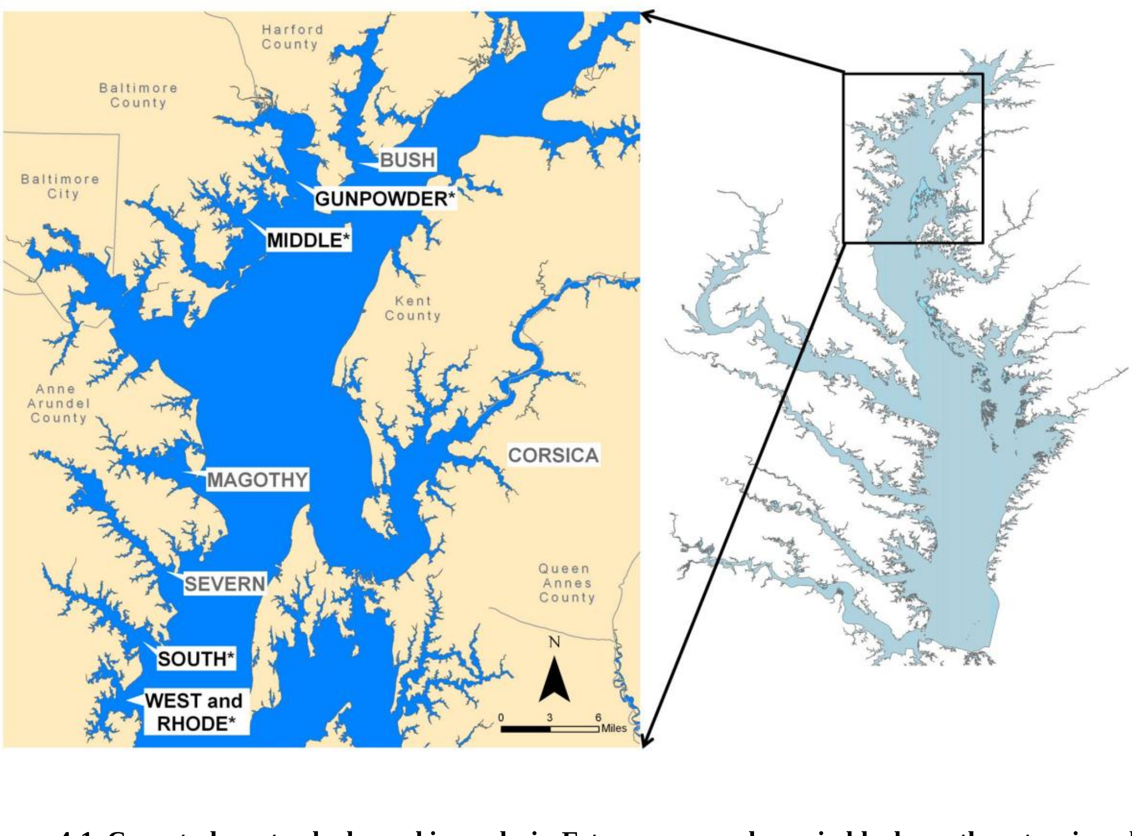


Figure 4-1. Case study watersheds used in analysis. Estuary names shown in black are the estuaries added in 2012.

The watersheds have a mix of suburban, agricultural and natural land uses/covers (based on data from USGS 2006, Table 4-2). The West/Rhode represents the highest agricultural watershed (25%), followed by the Gunpowder with 22% agricultural cover. Forest dominates the Gunpowder, South and West/Rhode watersheds with 40%, 49% and 40% cover respectively, while 65% of the Middle watershed is developed with only 2% agricultural cover. Note that the database used to calculate land covers is known to greatly underestimate low density residential land use, so some of the area represented as forest is likely to be interspersed with residential land uses.

Table 4-1. DATAFLOW© sampling years by estuary.

Sub-estuary	DATAFLOW© Cruise Years
Bush	2003-2005
Gunpowder	2003-2005
Middle	2003-2005
Corsica	2003-2005
Magothy	2001-2003
Severn	2001-2003
South	2004-2006
West and Rhode	2004-2006

Table 4-2. Land use of case study watersheds
(*Note: totals do not always equal 100% due to rounding factors.)

Aggregated Classes	Gunpowder (%)	Middle (%)	South (%)	West/Rhode (%)
Agriculture	**22**	**2**	**10**	**25**
crop	7	1	7	18
pasture	14	1	3	6
Developed	**31**	**65**	**28**	**12**
Low Intensity	10	21	13	5
Medium Intensity	5	20	4	1
High Intensity	2	8	2	0
Developed Open Space	13	15	8	5
Other Open Space	1	0	0	0
Grassland / Herbaceous	1	0	1	0
Forest	**40**	**15**	**49**	**40**
Deciduous	30	12	39	33
Evergreen	1	0	0	0
Mixed	5	2	7	4
Shrub Scrub	2	1	3	3
Wetland	**7**	**18**	**13**	**23**
Woody	5	17	3	22
Emergent	2	1	0	1
*TOTALS**	*100*	*100*	*100*	*100*

4-4 Methods

We evaluate water quality conditions using field data from DATAFLOW©, a variety of supporting GIS data, and modeled outputs from the Chesapeake Bay Watershed Model (CBWM). DATAFLOW© data are interpolated to provide a more even and comprehensive analysis of the estuary since it estimates data in unsampled areas and reduces the bias due to the non-random sampling pattern. Kriging also introduces some error into the data, however, we evaluate results in terms of whether they may be biased and discuss any issues found.

4-4.1 Data Sources

Data used in this analysis and their sources are summarized in Table 4-3. For information on field data collection techniques, please refer to Maryland Department of Natural Resources Chesapeake Bay Shallow Water Quality Monitoring Program (http://mddnr.chesapeakebay.net/eyesonthebay/documents/SWM_QAPP_2011_2012_FINALDraft1.pdf).

Table 4-3. Data sources used in analysis.

VARIABLES	SOURCE
Estuary volume (Calculated in GIS using file: NHD Area)	Simley, J.D., Carswell Jr., W.J., 2009. The National Map—Hydrography: U.S. Geological Survey Fact Sheet 2009-3054, 4 p. ; http://viewer.nationalmap.gov/viewer/
Impervious surface	MRLC NLCD Percent Developed Imperviousness, 2006. http://seamless.usgs.gov/
Landcover	USGS Chesapeake Bay Land Cover Data (CBLCD) Series, 2006.
Mean depth (Calculated in GIS using file: M130_37076C5_BIG1.dem)	NOAA , U.S. Estuarine Bathymetric Datasets, VA/MD (M130) Bathymetric Digital Elevation Model (30 meter resolution); http://estuarinebathymetry.noaa.gov/
Nitrogen, Phosphorous, and Flow delivered from the Susquehanna River	USGS National Water Information System; http://waterdata.usgs.gov/nwis
Nitrogen, Phosphorous and Flow delivered from watershed	US EPA Chesapeake Bay Program Watershed Model v. 5.3.2; Gary Shenk and Guido Yactayo Pers. Comm.
SAV Extent	Orth, R.J., D. J. Wilcox, J. R. Whiting, L. S. Nagey, A. L. Owens, and A. K. Keene, 2010. Distribution of Submerged Aquatic Vegetation in Chesapeake and Coastal Bays. VIMS Special Scientific Report Number 152. Final report to EPA. Grant No. CB97377401-0, http://www.vims.edu/bio/sav/
SAV Potential extent	Potential SAV habitat was calculated as the spatial merge of all mapped SAV from years 1971-2009 (Orth, *et al.* 2010).*
Septics within 1,000 ft. buffer	Tetra Tech, 2011. Chesapeake Bay TMDL Phase 1 Watershed Implementation Plan: Decentralized Wastewater Management Gap Closer Research and Analysis, March 2011. GIS data and report prepared for: Maryland Department of the Environment, 1800 Washington Blvd., Baltimore, MD 21230
Watershed Surface Area: Estuary surface Area ratio	Chesapeake Bay Program, Land Segments, 2010. U.S. EPA, 2010. Chesapeake Bay Phase 5 Community Watershed Model In preparation EPA XXX-X-XX-010 Chesapeake Bay Program Office, Annapolis, Maryland. December 2010 [http://www.chesapeakebay.net/about/programs/modeling/53/]; And USGS National Hydrography Dataset, 2009. [http://nhd.usgs.gov/
Chesapeake Bay Mainstem monthly average chl-*a* concentrations	Chesapeake Bay Program Office (Liza Hernandez, pers. comm. May, 2012)

*We evaluated whether a more recent range of SAV distribution (e.g., 1990-present) would be a superior representation of potential habitat because we hypothesized that SAV distribution might have become more depth-restricted in its range. However, in comparing the data for available years, we found that the area of SAV occurrence and average depth bay-wide has remained relatively constant from 1991 to 2009. Our finding is consistent with other work showing that habitat range expanded dramatically between the 1970s and the1980s and remains constant since about 1991 (Orth, *et al*. 2010). As a result, our potential SAV habitat map captures current conditions well in aggregate although some localized changes in areas that support SAV has occurred within some estuaries.

4-4.2 Spatial Data Analysis Techniques

A variety of GIS techniques available within ArcMap 9.3.1 were used to evaluate multiple types of spatial data on the watershed and estuary and statistically summarize conditions in space and time. Conditions within a given map pixel were made using the cell statistics tool, available within Spatial Analyst. Mean and standard deviations within a cell through time were generated from kriged output for seasonal and annual time periods to create maps of summary statistics.

To create our two primary response variables, we considered the spatial extent and temporal duration of elevated chl-*a* and SAV. Elevated chl-*a* was measured as the area of the estuary that exceeded 15 ug/L for more than one cruise date (or >20% of samples). The area of elevated chl-*a* was then divided by the total number of cruises for that year, which provided the frequency of times that each cell exceeded 15 µg/L. These values were summarized for an average exceedence per year per estuary. This value was divided by the potential area of SAV habitat in that estuary. Cross-sections of kriged salinity output were made to explore the relationship between changes in depth and water quality when gradients were present using the 3D Analyst Interpolate Line and Create Profile Graph tools.

In addition to the elevated chl-*a* variable, we created an alternative response variable that represented the area-weighted and time-weighted mass of chl-*a* measured throughout the year. The kriged output of chl-*a* concentration was converted to an annual total mass of chl-*a* by multiplying each monthly observation by 30.5 days per cruise, summing over the area of the estuary, and assuming a 1 m depth of measurement to provide a total volume measured. The variable was divided by area of the estuary to create the "chl-*a* concentration" variable that represents an area and time weighted annual total concentration.

4-4.2.1 Kriging Techniques

Kriging (ESRI 2001) was used to create continuous maps of water quality variables from samples taken with DATAFLOW©. Using the geostatistical toolbox available within ArcMap (ESRI 2010), patterns of spatial covariance in the data were used to fit a statistical model to each cruise that described how the data varied in space and to establish weights on observations that minimized estimation variance. As in most types of interpolation, the closest observations are given the largest weight when estimating un-sampled points, unless the user specifies otherwise.

In the Gunpowder estuary, kriging methods were adapted to handle gradients in water quality conditions that affected interpolation results. Rather than basing observation weights only on proximity, we used a quadrant approach to develop the weights used in the model. In brief, the quadrant approach ensures that points that are given the most weight are drawn from multiple compass directions when estimating unsampled locations. The software (ESRI 2010) allows the orientation of quadrants (or octants) to be varied and we selected standard quadrants of NE, SE, SW and SE for our purposes. The quadrant approach was helpful for producing a more realistic interpolation of datapoints without substantially increasing the computational burden.

4-4.3 Spatio-Temporal Summaries

Spatio-temporal summaries were used to characterize the percentage of each estuarine area that contained persistent elevated chl-*a* annually or over the multi-year period of DATAFLOW© collection. We summarized conditions using kriged data by estuary using a threshold of 15 µg/l of chl-*a*[1] in order to identify areas with potential water quality limitations on SAV habitat quality (Batiuk *et al.* 2000). In addition, we evaluated whether the chl-*a* was elevated in at least 20% of the sample, which translates to >1 sample for all estuaries. The percent of time that each sample represents will vary by the number of samples available in a given year. The temporal detail that we were able to represent varied from semi-monthly to monthly in our DATAFLOW© datasets.

4-4.4 Integrated Spatial Assessment

Scatter diagrams between driver and response variables are used to evaluate which variables are likely to be most useful for explaining conditions. We combined data from last year's analysis in order to create a data set for eight case study estuaries, for 3 years each (n = 24). The drivers include potential stressors or explanatory variables and include estuarine water quality, watershed descriptors, weather drivers, and estuary physical configurations. On the y axis, two types of system responses are represented: 1. % estuary having elevated chl-*a* ($\geq$ 15 µg L^{-1}) more than 20% of the time and 2. % of historical SAV beds containing SAV. Best fit regression lines are shown for reference and are shown as solid lines if the Spearman rank correlations were significant ($p < 0.05$) and dashed lines otherwise. Significance tests of the regression models would not be accurate given the temporal autocorrelation of the data.

4-4.5 Elasticities and Regional Comparative Statistics

To compare responsiveness of sub-estuaries while controlling for interannual variability, we calculated elasticities of water quality outcomes relative to drivers of water quality. Elasticities were calculated as the ratio of the interannual % change in a water quality condition (chl-*a*) to the % change in a driver of water quality (TN or TP). For example:

$$E_{chla|TN} = ABS\left[\frac{\left(\frac{chla_{t2} - chla_{t1}}{chla_{t1}}\right)}{\left(\frac{TN_{t2} - TN_{t1}}{TN_{t1}}\right)}\right] \qquad \text{(Eqn 1)}$$

Where chl-*a* is measured as the mass of chl-*a* (kg) when summed over all cells in the estuary and all cruises; TN is the annual loading to the subestuary as calculated by CBP; and *t1* and *t2* refer to the first and second years being compared. Similar to the common economic index *price elasticity of demand*, this statistic explains how much chl-*a* changed in a given 2-year period, relative to the change in nutrient loadings over that same time period. ABS is the absolute value of the term which is used to focus interpretation in terms of responsiveness of chl-*a* to the nutrient rather than including the positive or negative sign that indicates trends between years. Raw % change values are also provided to show whether the direction of change between nutrients and chl-*a* is the same or opposite.

We also examined the "cross-source" elasticity by comparing changes in chl-*a* within a sub-estuary to changes in Susquehanna nutrient inflows (TN and TP), as measured at the Conowingo gaging station. This cross-source elasticity allows us to compare whether chl-*a* changes in the estuary are more responsive to changes in the mainstem rather than the local watershed. The Susquehanna River loads are a reasonable proxy for the magnitude of nutrients entering the Maryland portion of the Bay in a given year since the Susquehanna contributes over 50 percent of the total streamflow according to numerous sources and "62 percent of the total nitrogen load, and 34 percent of the total phosphorus load from the nontidal part of the Chesapeake Bay Basin" (Belval and Sprague 1999). Phosphorus releases during storms are increasing as the sediment level behind the Conowingo dam increases (STAC 2000).

The final elasticity indicator we examined was the "local share" indicator that can be used to compare and distinguish responses between sub-estuaries and the mainstem. This metric is similar to the cross-source elasticity, but uses the response variable for the local estuary and the mainstem to evaluate whether a local response is different than would be expected if responses were due only to regional drivers. The equation is:

$$LS_{i|B} = ABS\left[\frac{\left(\frac{chla_{it2} - chla_{it1}}{chla_{it1}}\right)}{\left(\frac{chla_{Bt2} - chla_{Bt1}}{chla_{Bt1}}\right)}\right] \qquad \text{(Eqn 2)}$$

Where *i* is the sub-estuary and *B* is the Bay. A value > 1 indicates a stronger local response than would be predicted from regional drivers and a value < 1 a weaker local response, or less of a change in chl-*a* in the sub-estuary relative to the change over the same period in the mainstem. A positive value indicates that the variable increased between years. The ratio is valuable as an indicator because it removes the effect of estuary size and other local factors because it examines the interannual changes only which effectively holds constant all physical parameters of the estuary.

4-5 Results and Discussion

4-5.1 Intra-annual Variability

4-5.1.1 Salinity Spatial Patterns

Salinity is often used as a tracer to understand the relative effect of landward and seaward forcings, however, a companion project that we are conducting has suggested that relatively fresher water can enter Western shore tributaries from the Bay after storms. DATAFLOW© analyses in four new sub-estuaries provided findings consistent with our findings showing frequent and seasonally consistent cross-channel salinity gradients near the mouths of small sub-estuaries on the W Shore. Specifically, we found that during the spring, salinity is usually lower on the right side of the estuary (looking up estuary) and in the summer this pattern reverses and salinity is higher on the right side of the estuary (Figures 4-2 – 4-4). The effect is less apparent in the southernmost estuaries that we evaluated – the South and West/Rhode.

During the spring, seasonally high freshwater flows are originating from all tributaries. Given the relatively high percentage of inflow to the mainstem from the Susquehanna, the mainstem flows on the W shore can be periodically very fresh due to mainstem Coriolis effects that push Susquehanna flow to the W side of the mainstem. Salinity patterns observed in the sub-estuaries are consistent with a circulation pattern created by counterclockwise flow in the mainstem pushing seasonally fresh Bay water into the mouths of W shore tributaries, causing lower salinity on the right side. When mainstem Bay water salinity increases in the summer, the pattern remains consistent with Bay water entering the right side of the estuary.

Whether or not these cross-channel patterns are directly caused by Coriolis-type forces, remains a question, since these small sub-estuaries may be too narrow or shallow to experience these forces. Further, wind and groundwater may also contribute to cross-channel gradients. However, given the seasonal consistency of these patterns and the fact that the salinity gradient is often correlated with the relative salinity of the mainstem (salinity on the right side of the estuary is positively correlated with mainstem salinity), it is possible that these patterns are caused by *Coriolis-induced tidal rectification* (Huijts et al. 2009, Friedrichs pers comm.), in which transverse currents effectively push the riverine outflow to the left when looking upstream in subestuaries both large and small. Alternatively, if such patterns were caused by groundwater inflow or density-driven flow, the salinity gradient would not be expected to reverse seasonally (Figures 4-3 – 4-4). Further, the consistency across estuaries regardless of sampling dates and years suggests that Coriolis-induced forces may be more likely than wind as a driver.

As a result of the observed salinity patterns, the DATAFLOW© data reveal that mainstem Bay water is, generally speaking, having more influence on conditions on the right side of these tributaries looking up estuary, whereas watershed conditions are likely to be having more influence on the lefthand shores, particularly near the mouths of these W shore tributaries. The salinity and other water quality patterns are highly dynamic in shallow waters and our analysis does not rule out wind or other factors being important in explaining observations on a given date.

When salinity patterns are averaged over all cruise dates in the 3-year sampling period for the four estuaries added this year, the dominant pattern is a down-estuary salinity gradient (Figure 4-5) that does not show the ephemeral cross-estuary gradients. The exception is the Middle River which shows an inverse salinity gradient down estuary. In other words, the salinity is higher in the upper reaches and lower at the mouth. The river is generally fresh, therefore this average gradient could have been produced by a few extreme events of fresh water entering the mouth during a time when salinity was relatively high (e.g., due to summertime levels of evapotranspiration). Overall, these divisions of the estuary based on salinity are the most consistent and persistent zonation that we find in the estuary.

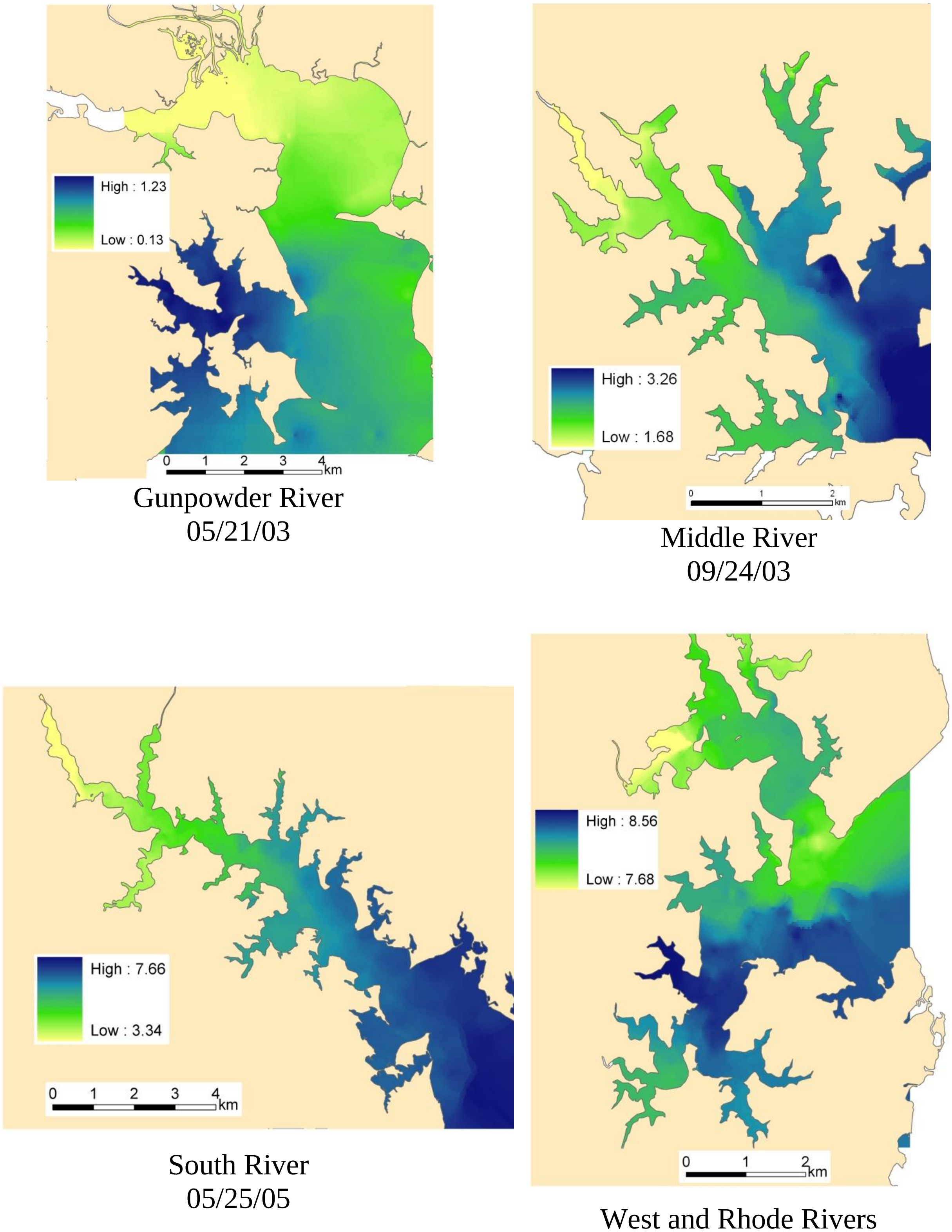


Figure 4-2. Examples of cross-channel salinity gradients observed in interpolated maps.

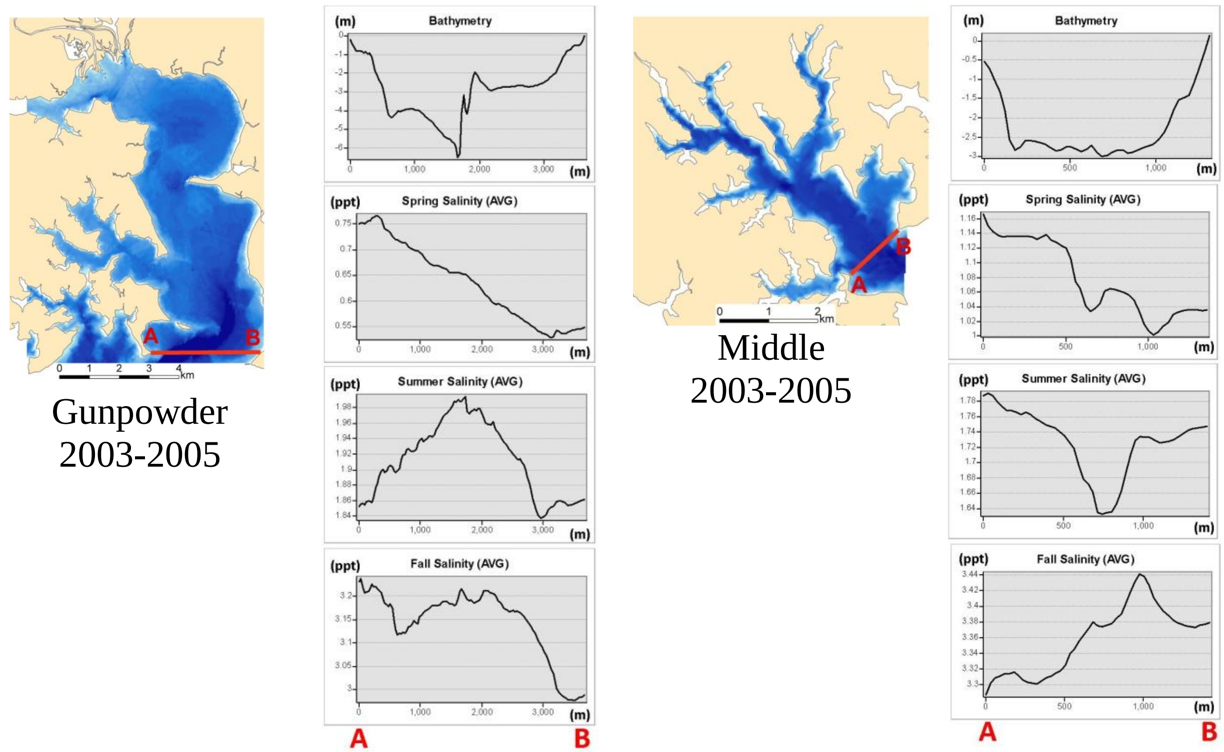


Figure 4-3. Surface salinity profiles of average spring, summer and fall salinity for Gunpowder and Middle Rivers.

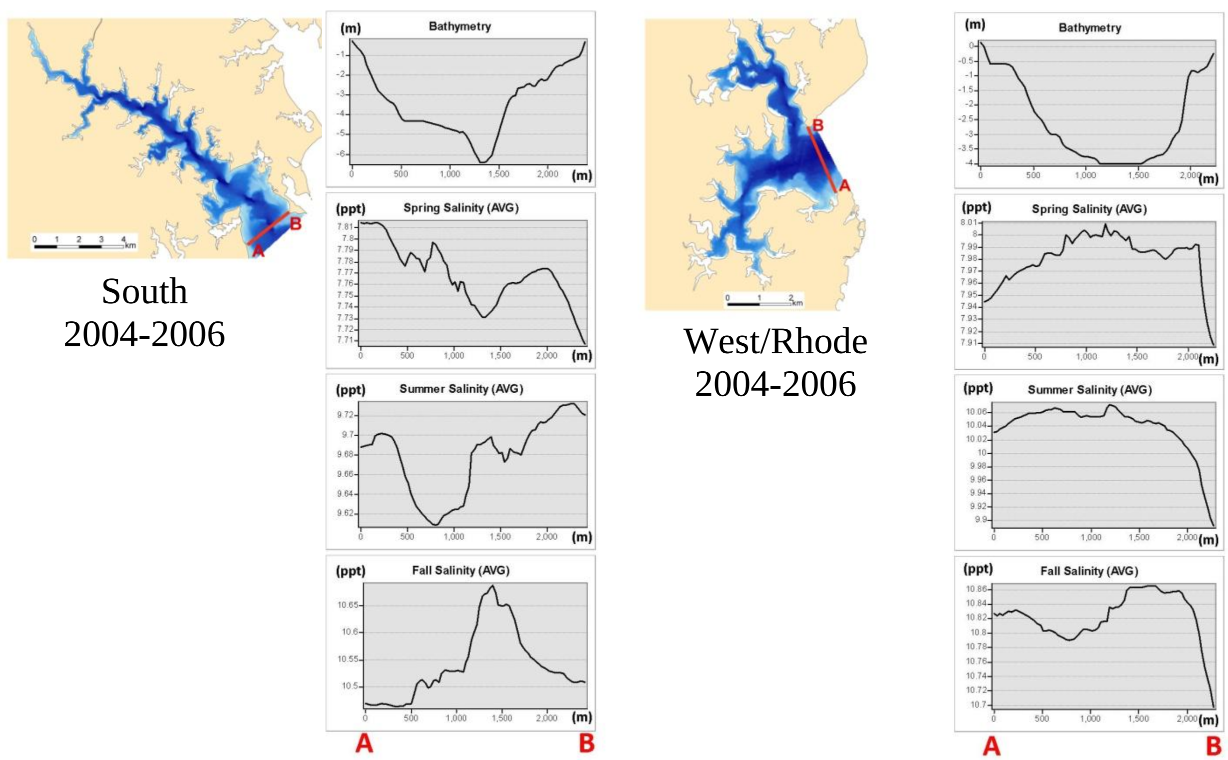


Figure 4-4. Surface salinity profiles of average spring, summer and fall salinity for South and West/Rhode Rivers.

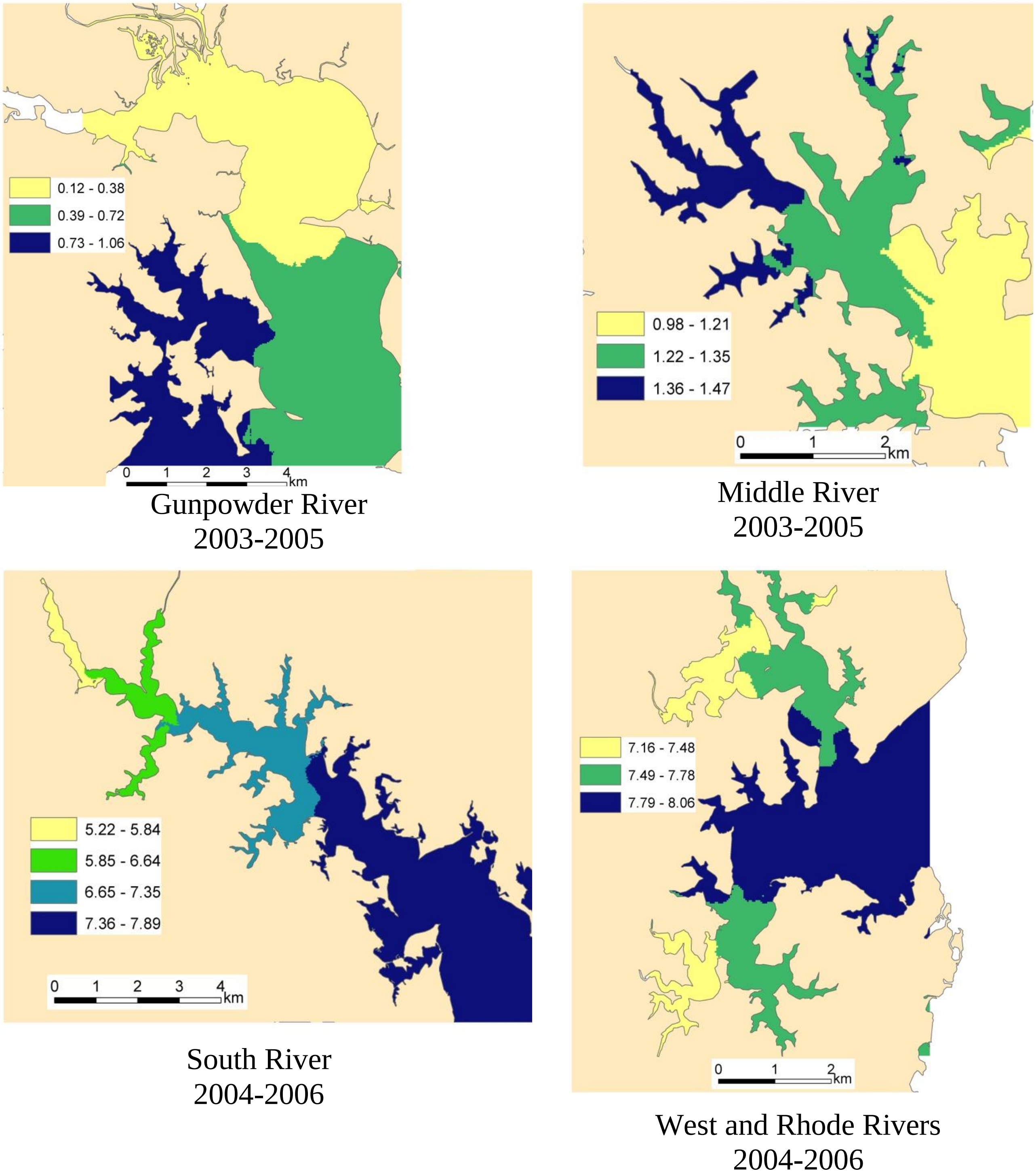


Figure 4-5. Salinity zones of estuaries.
Zones are based on annual average salinity by pixel and divisions are created using Jenks natural breaks. This technique for dividing up areas is similar to cluster analysis because this classification groups similar values within classes and maximizes differences between classes.

4-5.1.2 Chl-*a* Spatial Patterns in Space and Time

The kriged results from individual DATAFLOW© cruises show that elevated chl-*a* can be widespread in the estuary on a given date but such patterns do not persist throughout the sampling season. Rather, areas of the highest chl-*a* concentrations and the most persistently elevated chl-*a* tend to be localized. The highest average chl-*a* tends to occur in the upper reaches, or fresher areas, of these estuaries with the exception of the Gunpowder which also shows moderately elevated chl-*a* near the mouth and on the right-hand side looking up estuary near the mouth (Figure 4-6). The areas with the greatest variability in chl-*a* over time tends to be localized to a few areas that are not necessarily the same as the areas with the highest average chl-*a* (Figure 4-7). The Gunpowder and the Middle rivers show areas with substantial deviations near their mouths suggesting that Bay nutrient inputs or changes in flushing time within this region may sometimes drive more ephemeral phytoplankton growth.

The locations of elevated chl-*a* are more consistent in these four estuaries than in the ones evaluated last year (Bush, Corsica, Magothy, Severn) because these estuaries show highest levels consistently in the upper watershed. This is in contrast to the Bush, Magothy and Severn which showed average chl-*a* was highest in the middle or lower parts of the estuary. The Corsica results are consistent with findings in the new estuaries of highest levels in the upper reaches. The average chl-*a* reaches the highest magnitude in the South among the 8 estuaries evaluated so far although the West/Rhode, Magothy and Severn show comparable average levels that are not much below the peak for the South River.

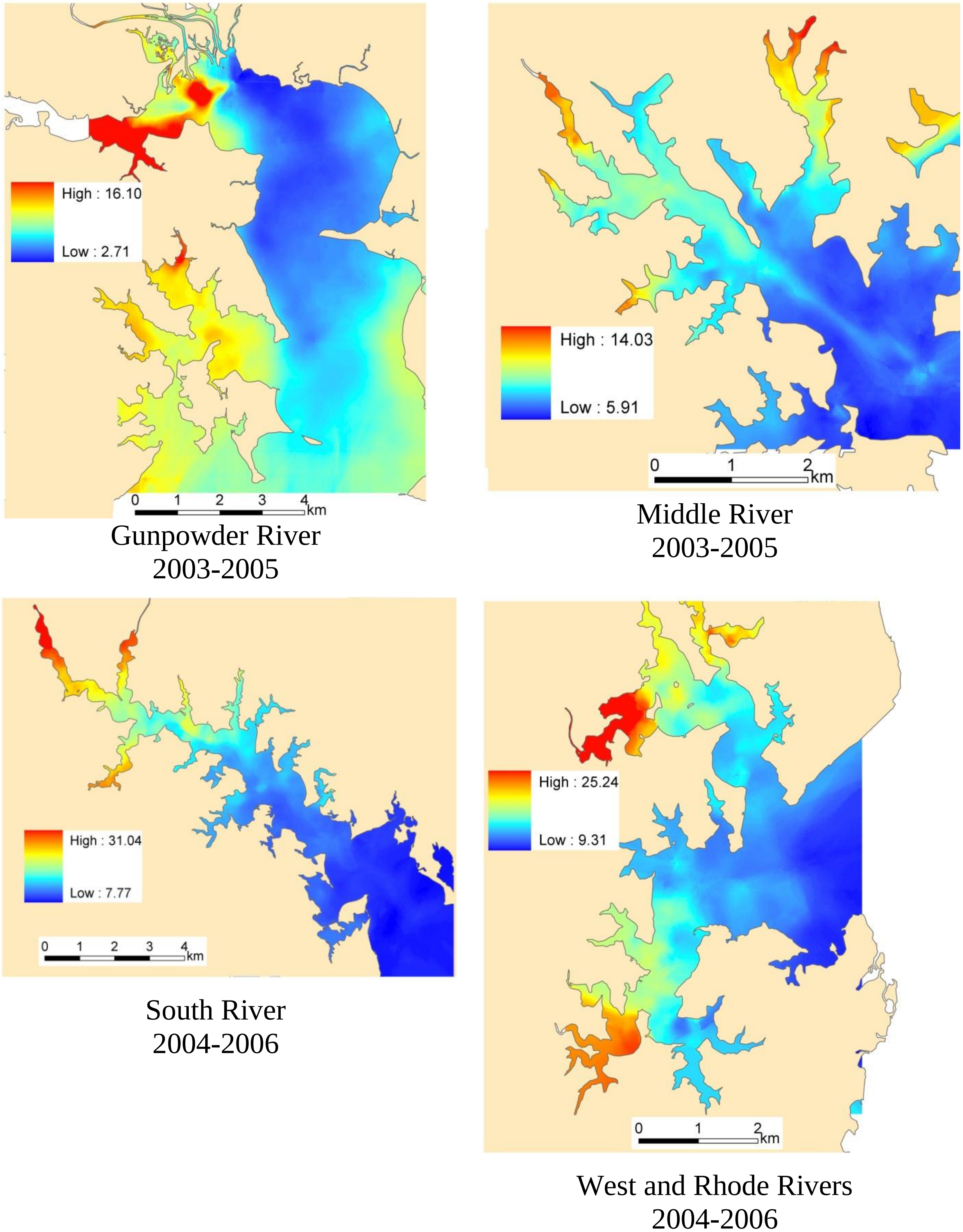
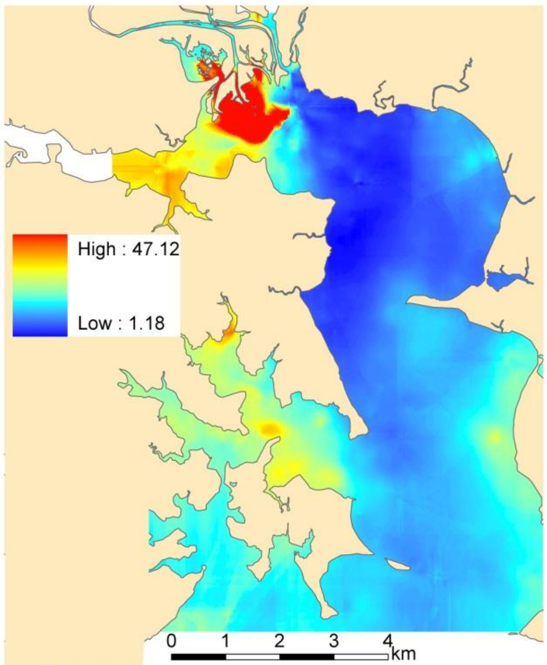
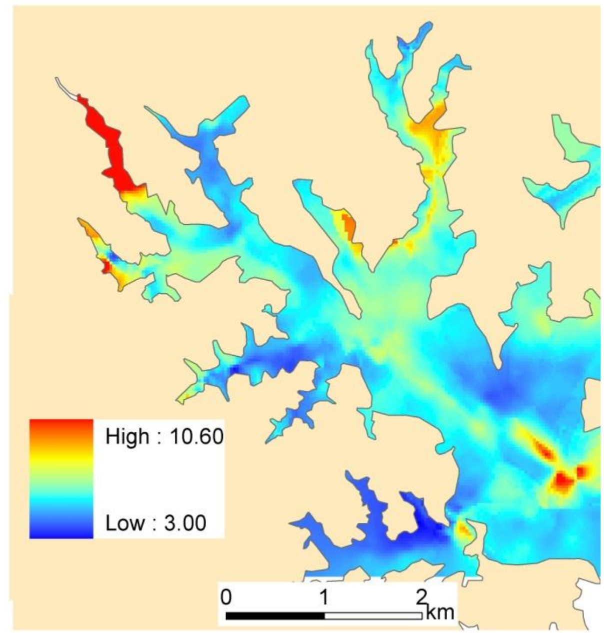


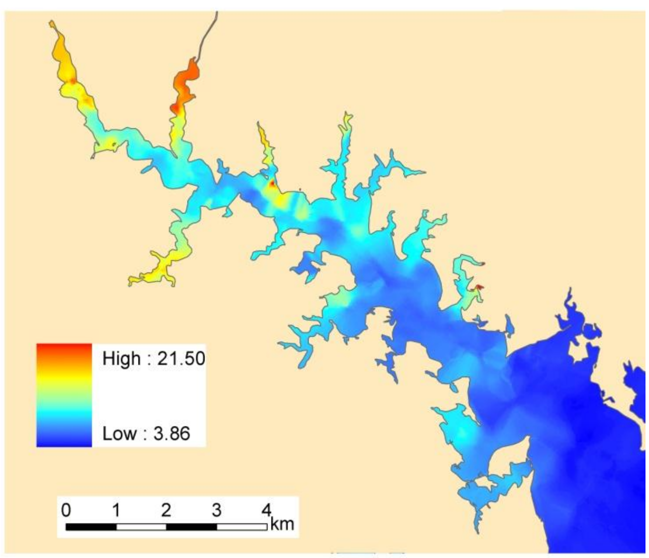
Figure 4-6. Average annual chl-*a* concentrations by estuary.



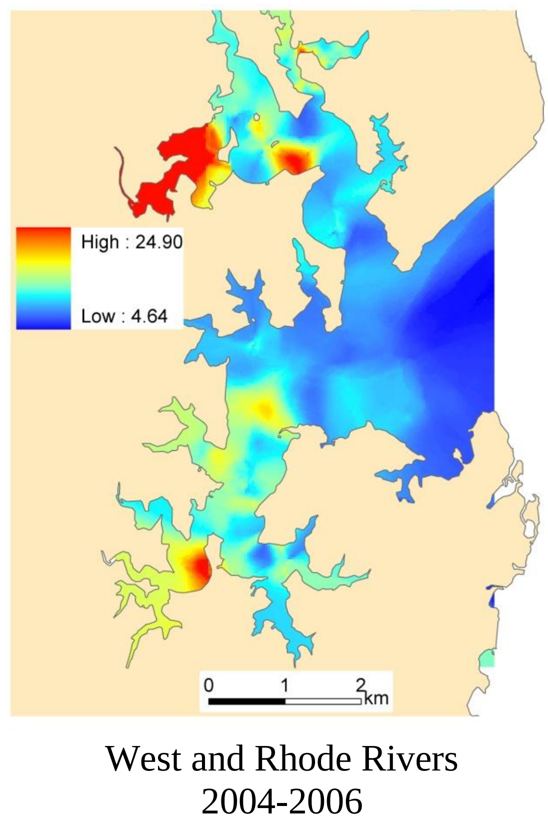
Gunpowder River
2003-2005



Middle River
2003-2005



South River
2004-2006



West and Rhode Rivers
2004-2006

Figure 4-7. Standard deviations of average annual chl-a concentrations by estuary.

A major goal of analyzing the spatial data is to understand whether spatial patterns in water quality conditions are consistent year to year. If this is true, then it might be useful to divide the estuary into zones with distinct driver and response conditions. A major tool in this analysis are the "frequency" maps which show the frequency with which elevated chl-*a* (defined as > 15 ug/l) occurs in a given area in a given year. By comparing these results across years, we can identify areas with persistent conditions both intrannually and interannually.

What the latest data show is consistent with previous findings in that rough patterns do emerge but boundaries of patches shift over time. For example, in the South River, we see a general pattern of consistently elevated chl-*a* in the upper reaches but the lower boundary of the patch that is most frequently elevated chl-*a* (shown in red or orange) is only consistent in 2 of the 3 years (Figures 4-8 – 4-11). Further, the frequency is lower in one of the years. Some of this variability is due to the fact that DATAFLOW© cruises are snapshots of conditions per month and therefore do not fully represent the frequency of conditions. Therefore, small changes in frequency from year to year may not be important.

Other results show some consistency of spatial and temporal patterns. The Gunpowder shows limited areas of consistently elevated chl-*a* except in the upper reaches (Figure 4-8). The Middle River also showed only small localized patches of consistently elevated chl-*a* during the relatively wet years of 2004-2005 but showed consistently higher chl-*a* throughout the river during 2006, which was considered an average year in terms of total rainfall (Figure 4-9). The West and Rhode estuary is reasonably consistent in showing elevated chl-*a* in the NW and NE portions of the upper Rhode River and the SW portion of the West River (Figure 4-11). The bay at the confluence of these rivers tends to show lower frequency of elevated chl-*a* but the response or pattern is not completely consistent year-to-year.

Note that the striking N-S line between patches with no elevated chl-*a* and some elevated chl-*a* in 2004 in the West/Rhode estuary does not appear to be an artifact of the kriging since data points spanned this line along its length in multiple cruises (Figure 4-12). Similarly, elevated chl-*a* at the edges of the spatial data (e.g. for the South River, Figure 4-7) are not based on extrapolating data beyond the sampled points.

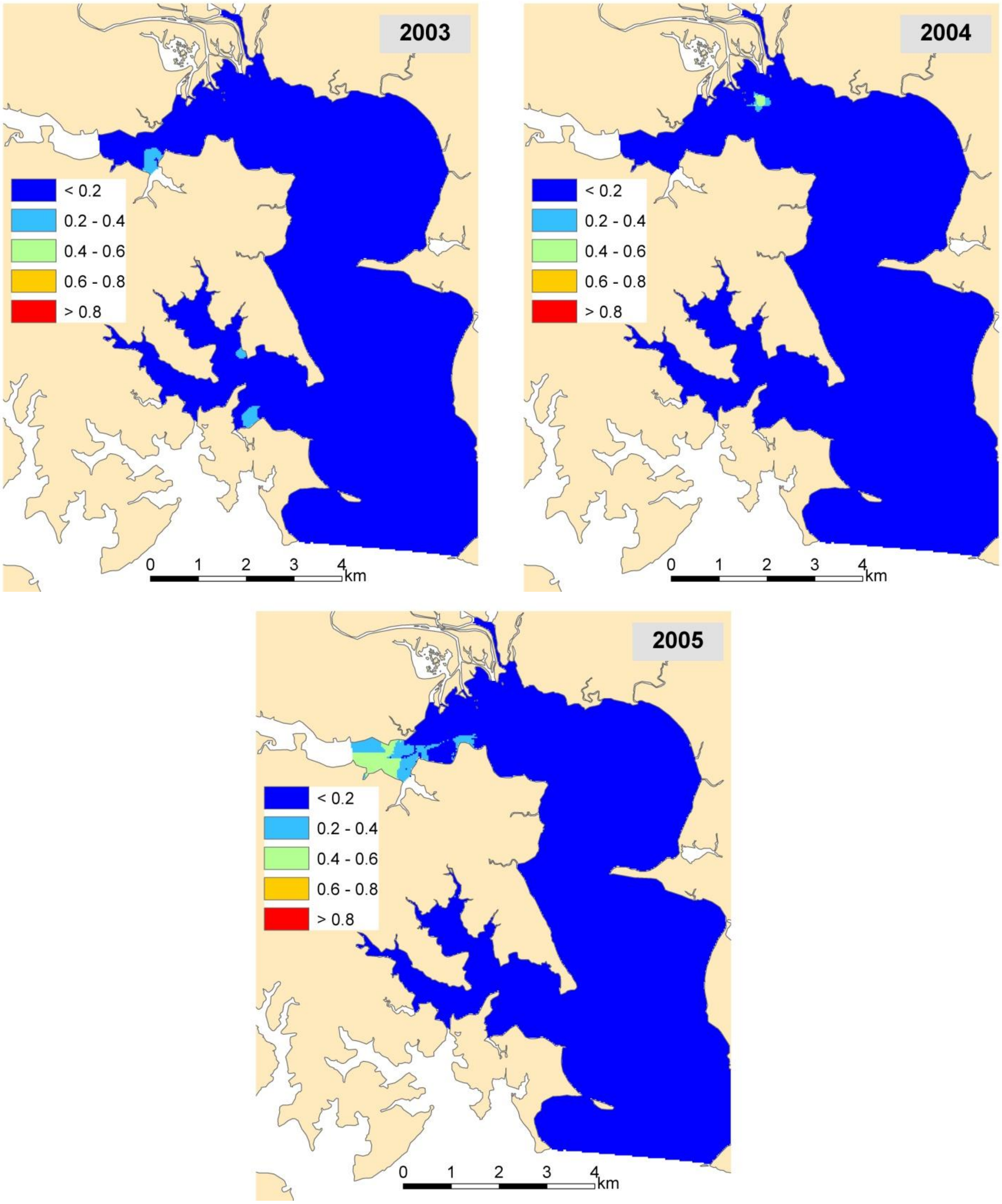


Figure 4-8. Frequency of elevated chl-*a* for the Gunpowder River

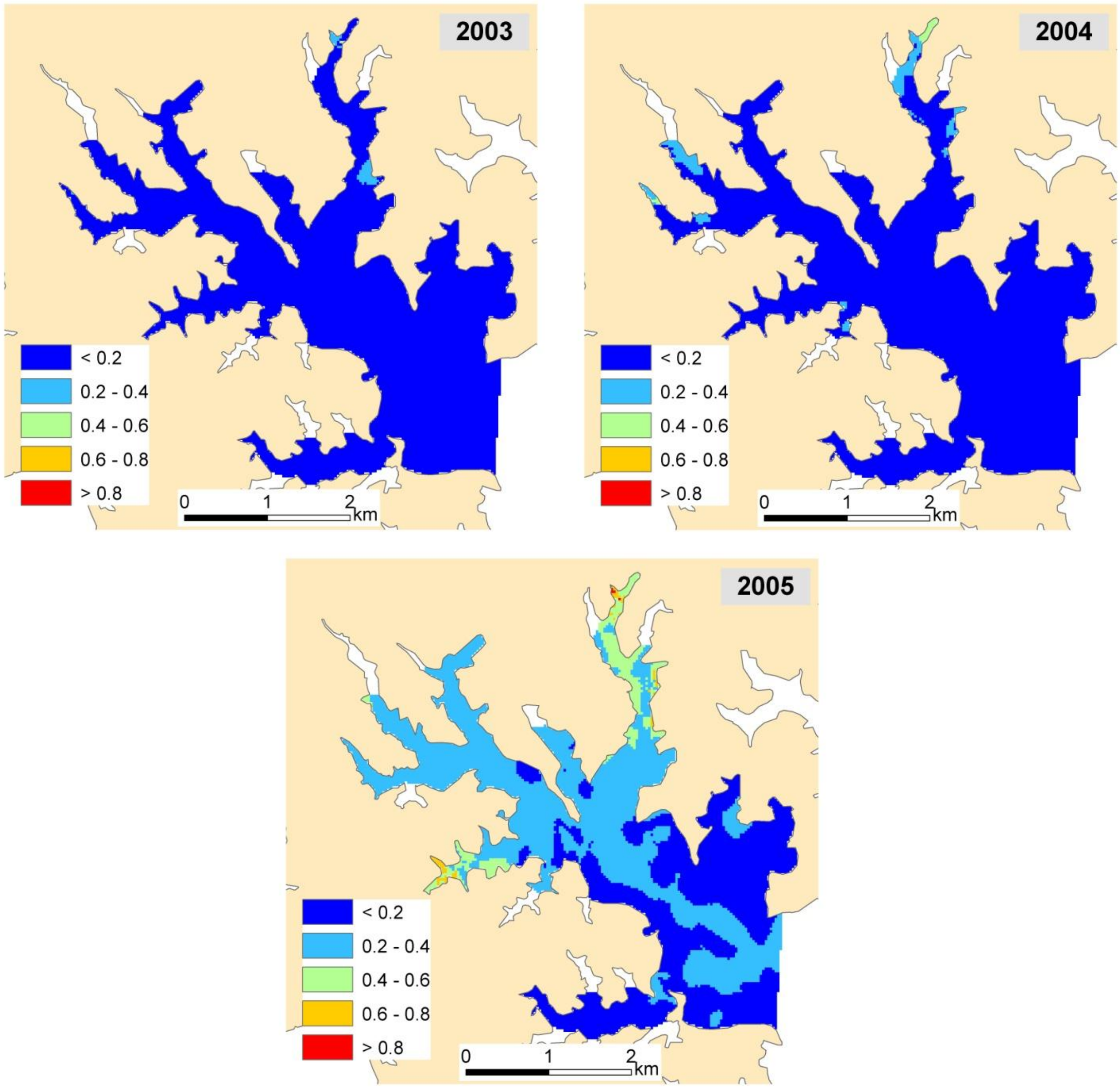


Figure 4-9. Frequency of elevated chl-*a* for the Middle River

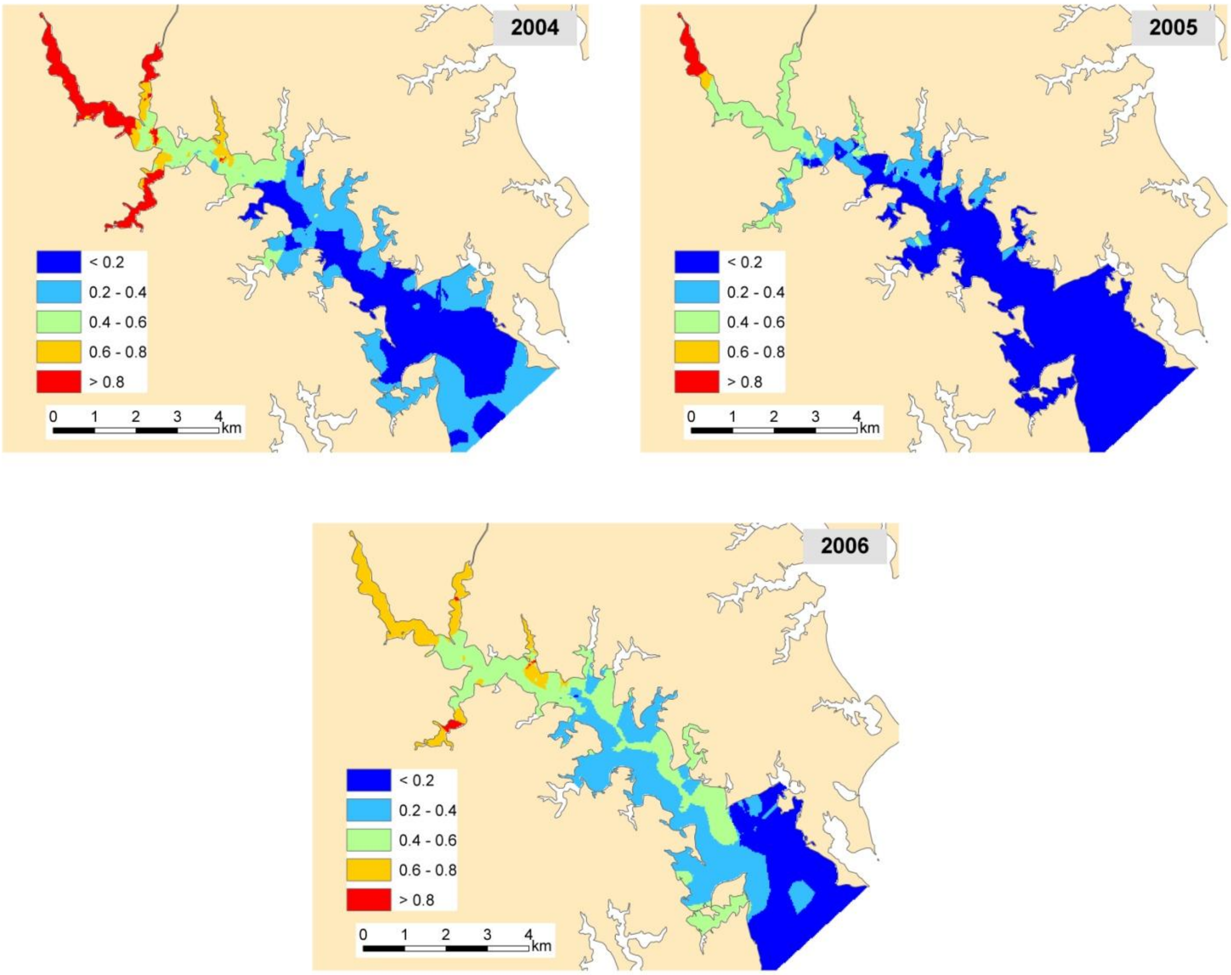


Figure 4-10. Frequency of elevated chl-*a* for the South River.

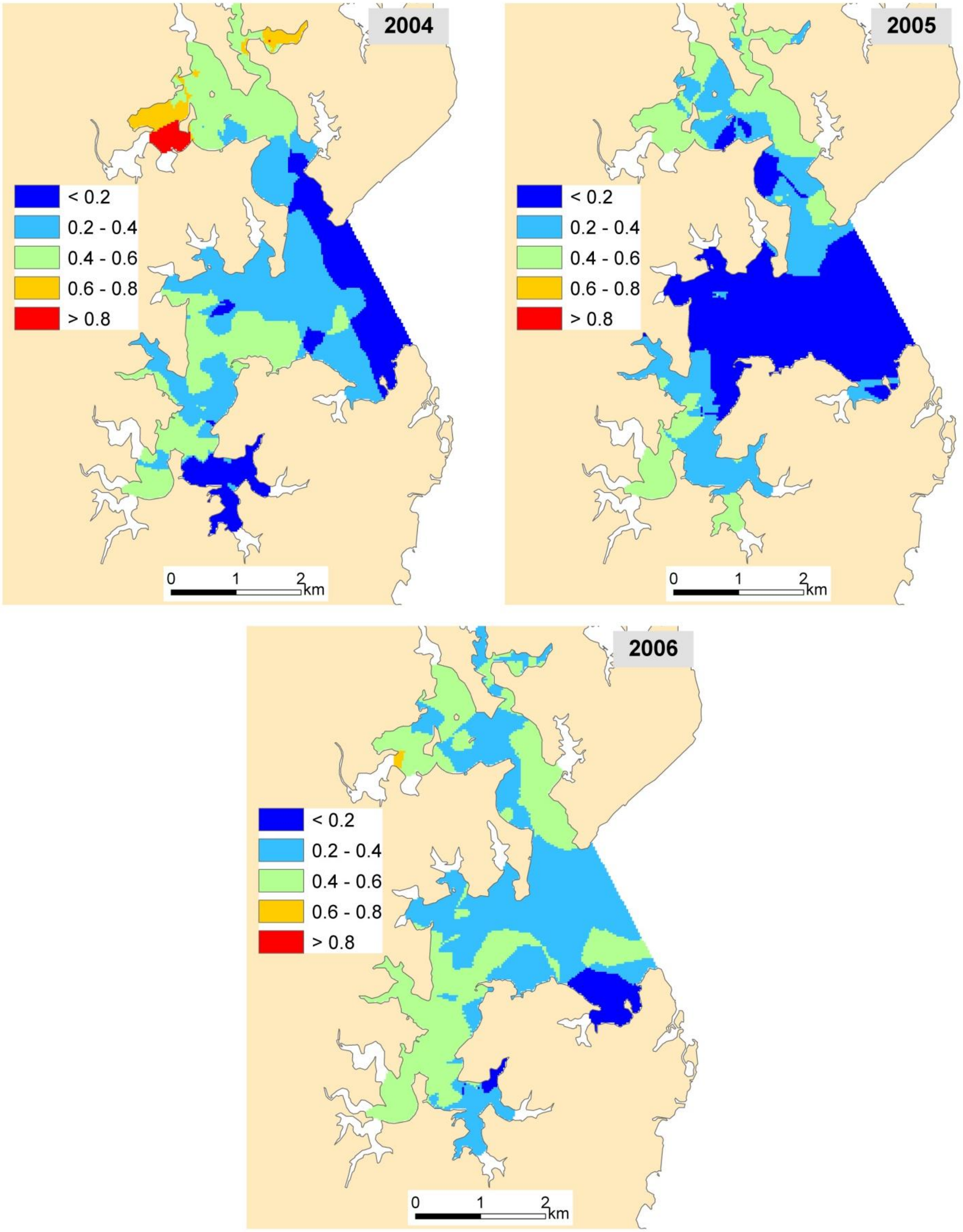


Figure 4-11. Frequency of elevated chl-*a* for the West/Rhode River.

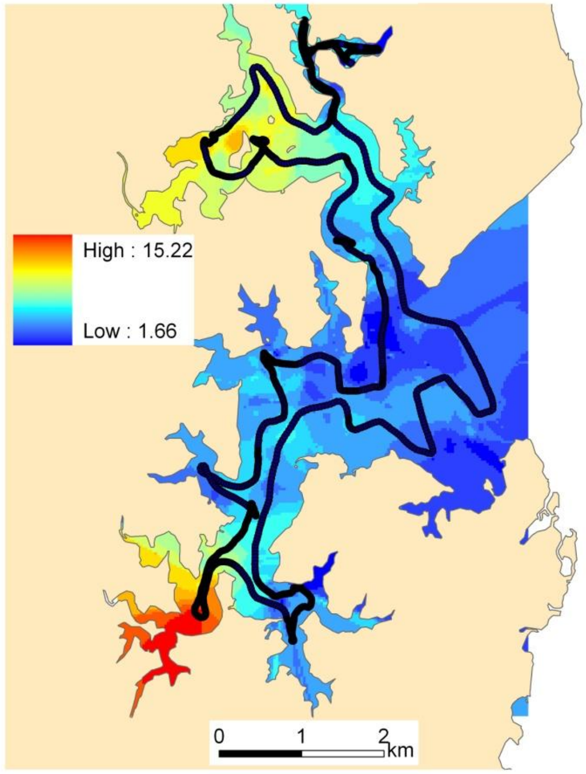


Figure 4-12. Example of chl-*a* kriged data for West/Rhode river showing cruise track.
Data from 04/28/2004 show that the straight line between different frequencies of elevated chl-a that occur in the SE part of the West/Rhode estuary do not appear to be an artifact of the data distribution.

4-5.1.3 Relationship of Chl-*a* Patterns to SAV

Because our previous work showed limited correlation between annual average conditions of chl-*a* as measured with DATAFLOW© and SAV occurrence, we do not show spatially-averaged water quality statistics for SAV bed locations. However, maps of SAV distribution do reveal that SAV appears most commonly in areas with low frequency of elevated chl-*a*. Data for the years with maximum SAV coverage in three of the sub-estuaries, show this pattern (Figure 4-13). The fourth sub-estuary we examined, the West/Rhode River had no SAV mapped by the VIMS project in DATAFLOW© analysis years of 2004-2006. For the Gunpowder and South rivers, 2004 had the maximum peak SAV. Peak SAV was greatest in 2005 for the Middle river. SAV extent was variable for the Gunpowder over the analysis period of 2003-2005 and similarly variable for the South for 2004-2006. For example, there was no SAV in the South in 2006. However, area of SAV was similar in the Middle across all three years (2003-2005).

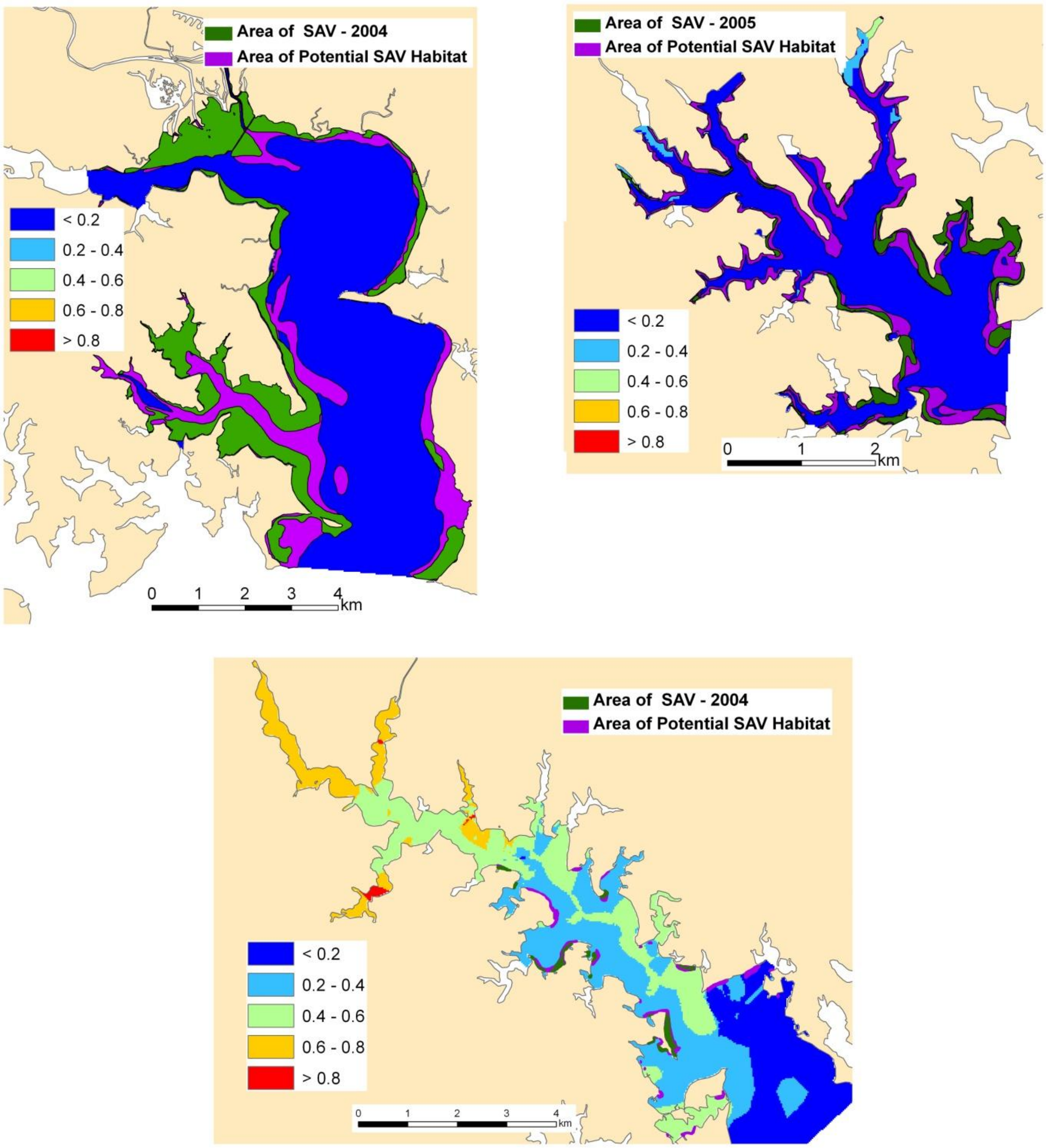


Figure 4-13. Distribution of SAV in year of peak extent over maps of frequency of elevated chl-*a*. Areas with SAV generally show low frequency of elevated chl-*a*. However, in the Gunpowder, beds in the northern part of the estuary are occurring in the small area with persistent elevated chl-*a* (apparent in Figure 4-8).

4-5.2 Integrated Ecosystem Assessment

An integrated ecosystem assessment (IEA) is a tool to support ecosystem-based management of coastal ecosystems by testing links and feedbacks between stressors and key ecosystem responses (Levin *et al.* 2009). We added data points to the limited IEA we conducted last year to test whether relationships were more likely to take on expected relationships with additional data points. As with last year, we have found some expected and unexpected relationships between stressor and response variables (Figures 4-14 – 4-17) but do not find the added data substantially alter the results compared to last year. Work in our companion project will evaluate whether multi-variate statistical analyses or non-linear relationships can improve the ability to use these types of variables to explain variability among estuaries.

Figures 4-14 – 4-17 show scatterplots representing the relationship between one driver and one system response variable for our eight case study estuaries over 3 years of study. The drivers include a wide range of potential stressors or explanatory variables of water quality outcomes including estuarine water quality, watershed descriptors, weather drivers, and estuary physical configurations. On the y axis, two types of system responses are represented: 1. % estuary having elevated chl-*a* ($\geq$ 15 µg L^{-1}) more than 20% of the time and 2. % of historical SAV beds containing SAV. The two response variables (chl-*a* and SAV) are not independent and we expect the two response variables to have opposite responses to any given stressor or driver. The Spearman Rank correlation coefficient between these two variables is negative (ρ = -0.31) but not significant at the 0.05 level. Chl-*a* concentration is a direct stressor on SAV growth, but more generally, we expect chl-*a* concentration to increase with stress (e.g., water quality decline), while we expect SAV to decrease in response to stress.

Many different variable relationships are tested to give an integrated assessment of stressors and responses. The analysis suggests that some variables are well-correlated with SAV abundance and intensity of elevated chl-*a* (denoted with * in figures). However, the weakness of some relationships and the unexpected directions of correlations between outcomes and variables that would generally be considered to be obvious drivers of condition (denoted with an X above figures), reflects both the complexity of estuarine systems and the difficulty of identifying individual explanatory factors in the face of complex interactions of system variables.

Some of the variables with significant correlations (based on Spearman Rank correlations) were: average depth (0.76 with chl-*a*), summer salinity (0.66 with elevated chl-*a* and -0.51 with % SAV), septic density (0.49 with chl-*a*), discharge-to-estuary volume ratio (-0.44 with chl-*a*), average August Temperature (0.41), and total nitrogen per unit discharge from non-point sources (NPS) (-0.41 with SAV). Total nitrogen per unit discharge from point sources (PS) or the total of PS and NPS were non-significant with chl-*a* and SAV but showed correlation in the expected direction (positive for chl-*a* and negative for SAV). This was in contrast to significant correlations that had signs opposite from expected. Those correlations included annual Susquehanna loads of TN and TP, which were positively and significantly correlated with chl-*a* in the subestuaries.

We found that only a few relationships changed the direction of correlation when we added the four new estuaries. Septics within 1,000 ft. buffer changed its correlation to positive, the expected direction, with chl-*a* but August temperature switched to an unexpected positive relationship with SAV (Figures 4-14 and 4-15). A possible explanation for the better performance of the variable septics near the watershed was that an updated database of septic counts was used this year that improved the ability to identify septics in the buffer (Tetra Tech, 2011 and Figure 4-18).

% Area of Estuary with Elevated Chl-*a*

% Area of Estuary with Peak SAV

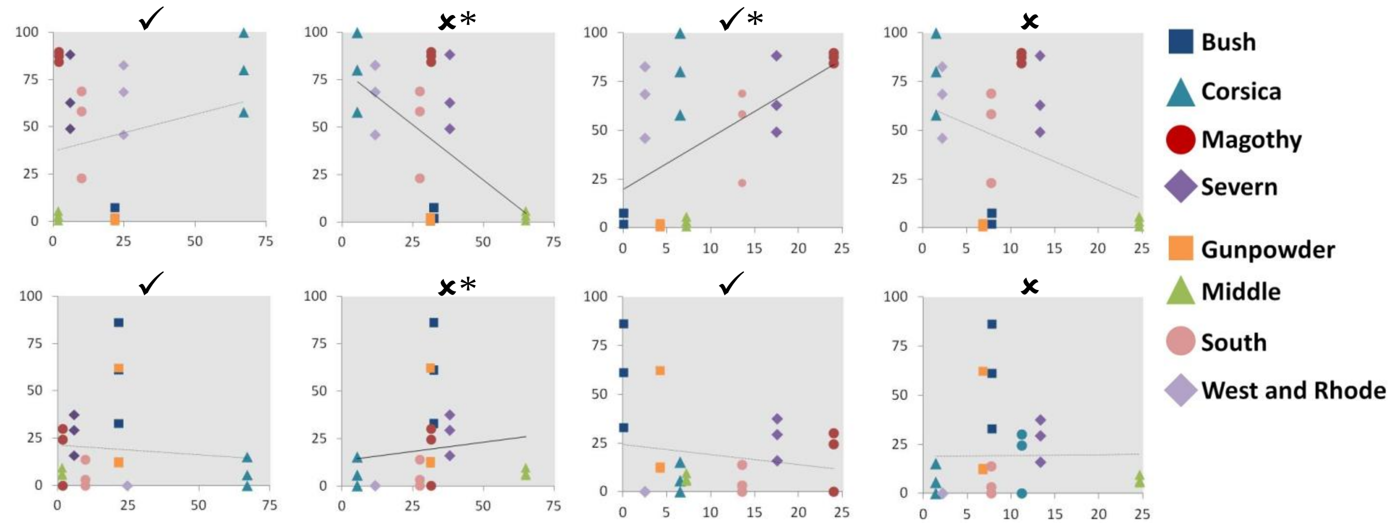


Figure. 4-14 **Agriculture (%)** **Development (%)** **Septics within 1,000 ft. buffer** **Impervious Surface**

% Area of Estuary with Elevated Chl-*a*

% Area of Estuary with Peak SAV

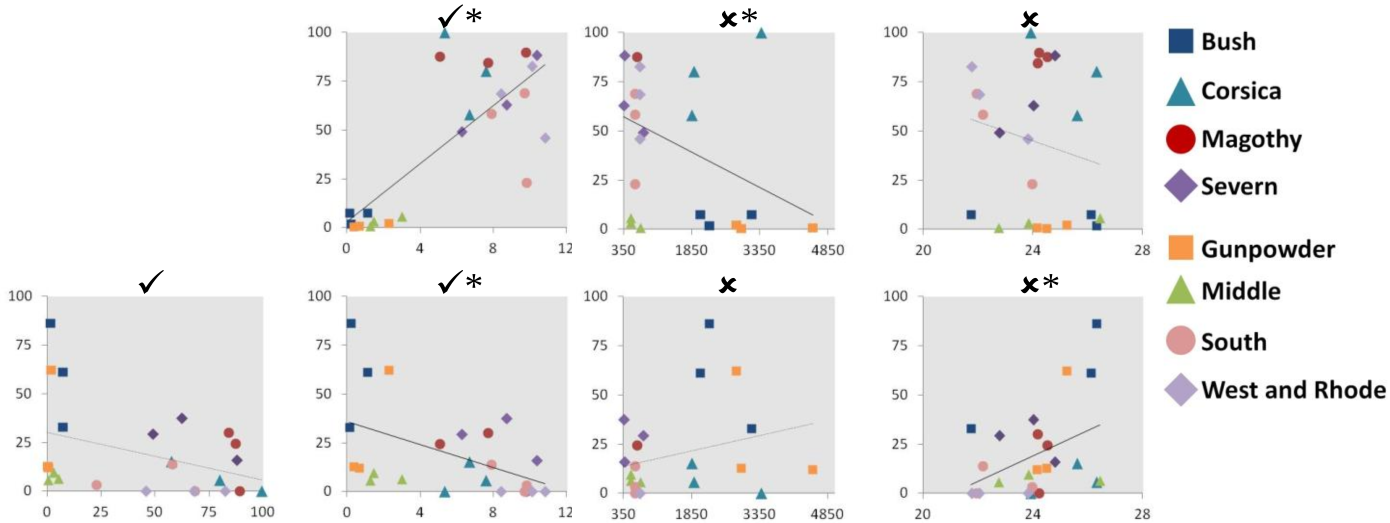


Figure. 4-15 **% Area of Estuary with Elevated Chl-*a*** **Median Summer Salinity** **Q/V** **August Temperature (°C)**

% Area of Estuary with Elevated Chl-*a*

% Area of Estuary with Peak SAV

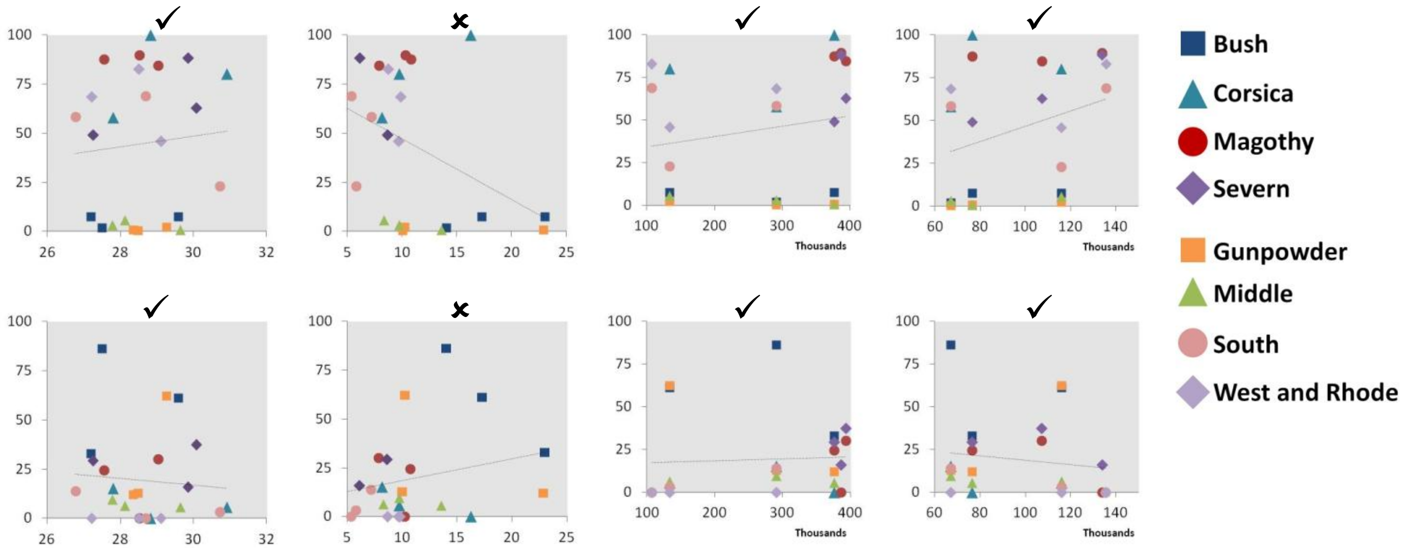


Figure. 4-16 | Maximum Temperature (°C) | Turbidity (NTU) | Spring Susquehanna Flow (m^3) | Summer Susquehanna Flow (m^3)

% Area of Estuary with Elevated Chl-*a*

% Area of Estuary with Peak SAV

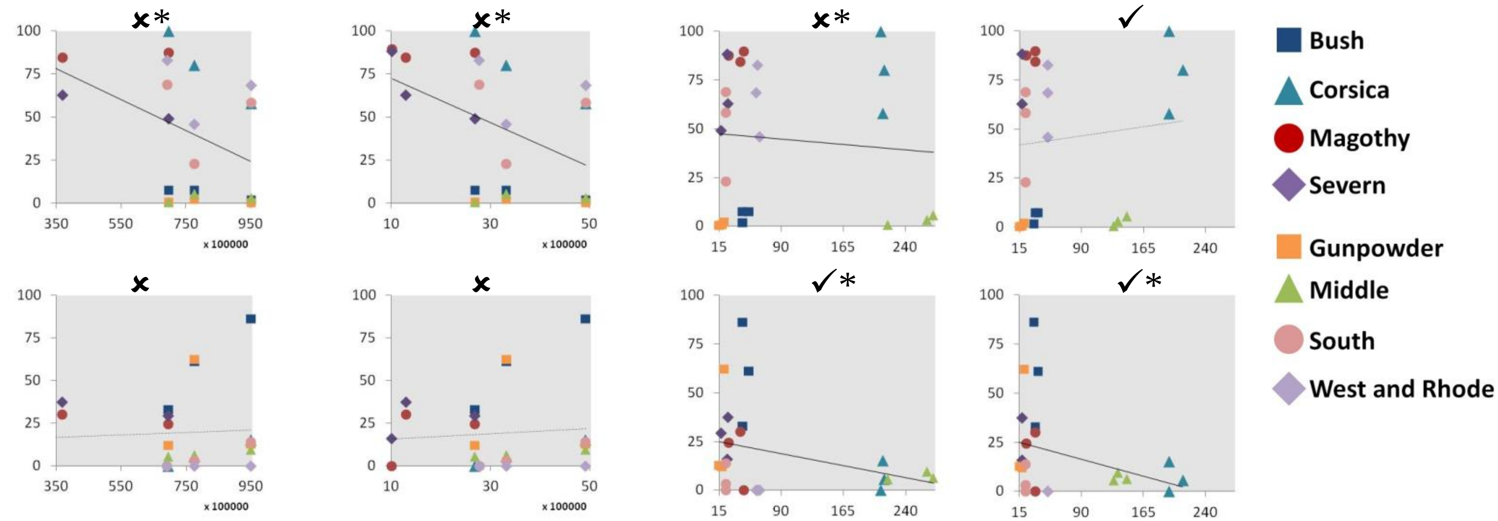


Figure. 4-17 | **Annual Susquehanna TN (kg/year)** | **Annual Susquehanna TP (kg/year)** | **TN Discharge** | **TN NPS Discharge**

Figures. 4-14 – 4-17. Scatterplots of stressors and drivers.
Best fit regression lines are shown for reference and are shown as solid lines if the Spearman rank correlations were significant ($\rho < 0.05$) and dashed lines otherwise. A check mark (✓) indicates an expected response and an X (×) indicates an unexpected response.

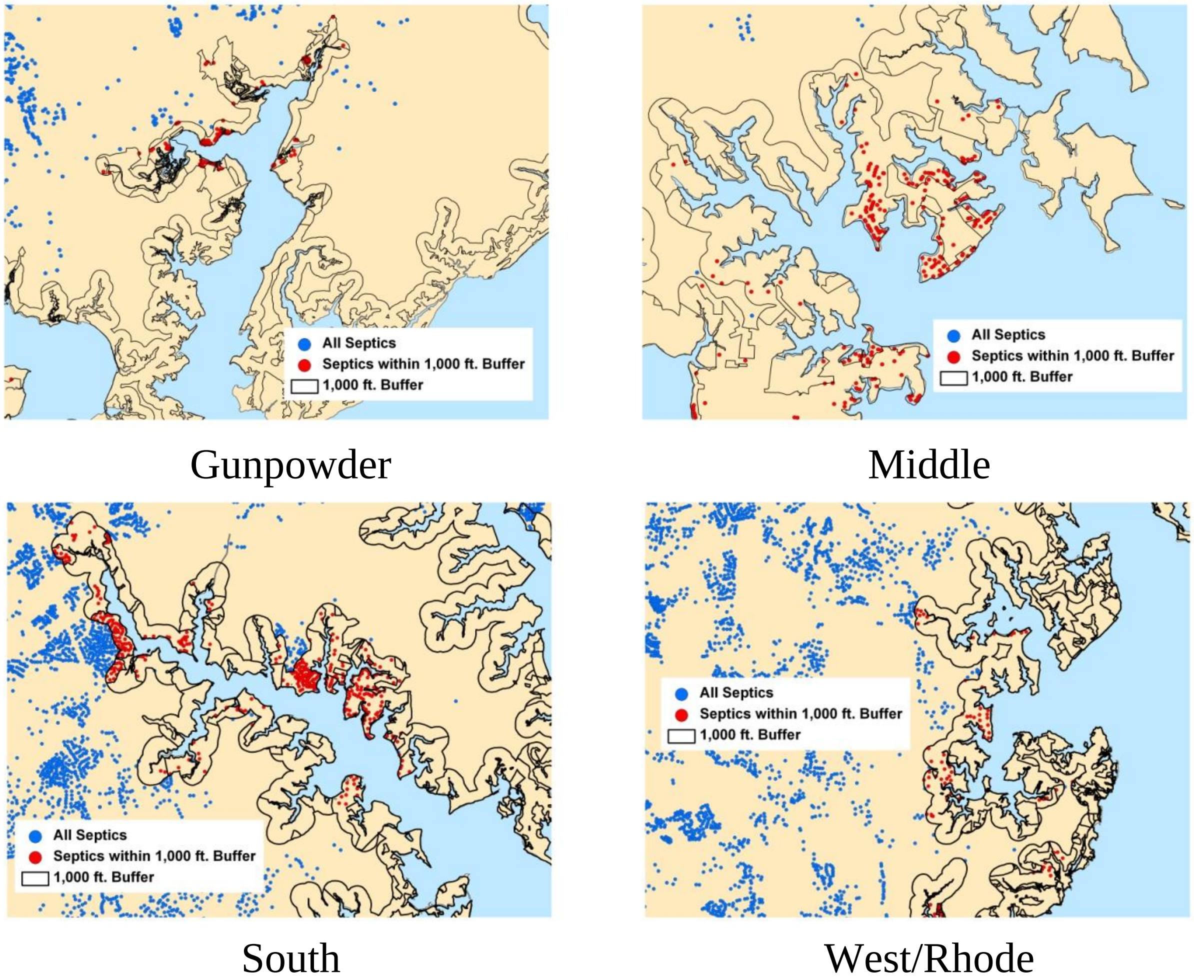


Figure 4-18. Septic distribution within and outside the 1,000 ft. buffer.

4-5.3 Interannual Variability

4-5.3.1 Percentage Change in Drivers and Response Variables

We compared percentage changes in water quality conditions between years in order to test whether such indicators would be useful for understanding interannual drivers. The value of looking at interannual percentage changes is that the analysis holds the physical variables of the estuary constant in order to isolate the effects of drivers (see Methods section for further explanation). A comparison of the % change in nutrients and % change in chl-*a* between years showed these relationships are inconsistent through time and also inconsistent between estuaries. The interannual percentage change in chl-*a* does not show a consistent relationship to the percentage change in nitrogen and phosphorus loadings (Figure 4-19) nor to the % change in chl-*a* in the Bay mainstem (Figure 4-20). In other words, an increase or a decrease in nutrients within an estuary is insufficient for explaining whether chl-*a* will increase or decrease between any pair of years. The data show that it is more common to have chl-*a* change in the opposite direction of the nutrient change. A change in chl-*a* in the mainstem is consistent with a change in chl-*a* in a subestuary, just over half the time.

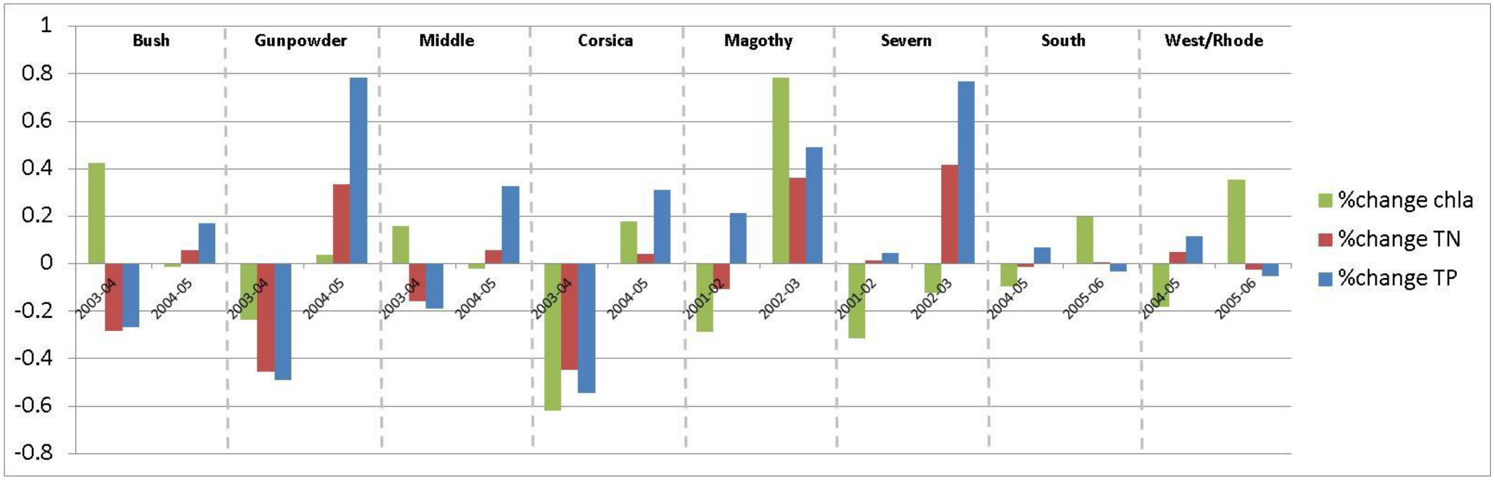


Figure 4-19. Interannual percentage change in chl-*a*, TN and TP for subestuaries.
Estuaries are ordered from left to right by their position (North to South) in the Bay. Note that the direction and magnitude of change in nutrients between years (red and blue bars) is not a consistent predictor of the chl-a response (green bars) of an estuary over the same period.

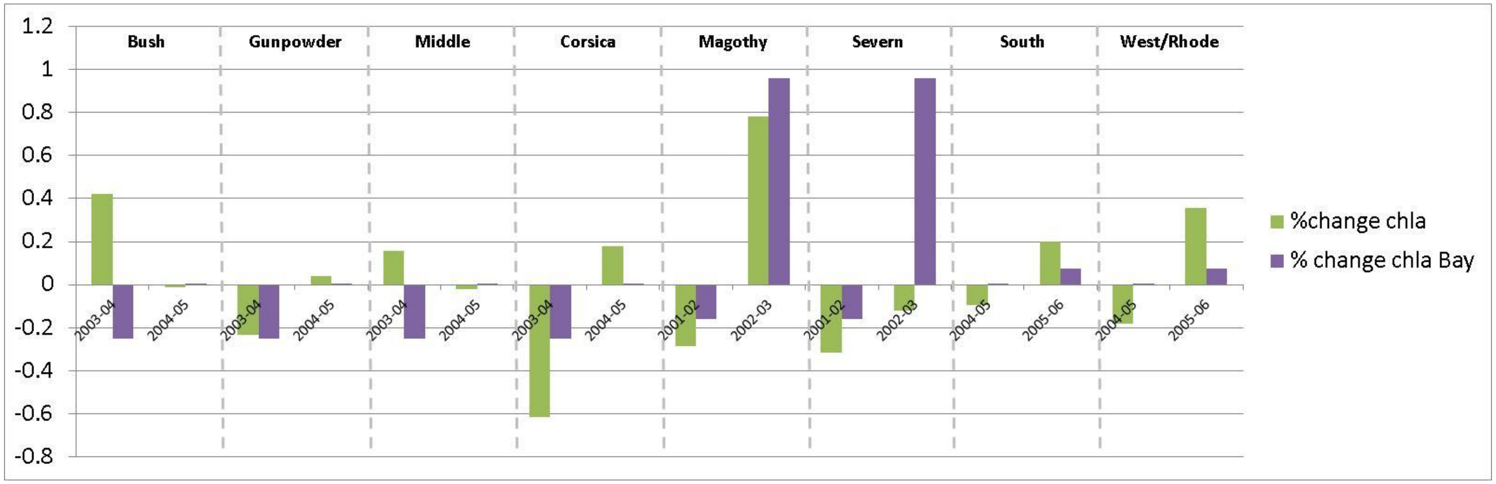
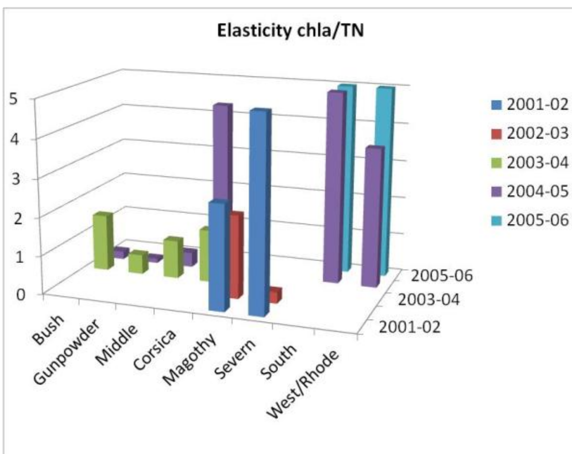


Figure 4-20. % Change in annual chl-*a* in subestuary compared to % change in annual chl-*a* in Bay mainstem.
Note that the interannual change in chl□a in a sub-estuary (green bars) does not consistently have the same sign (+/-) as the corresponding change in the Bay (purple bars), suggesting that sub-estuary water quality conditions are sometimes de-coupled from Bay mainstem water quality conditions.

4-5.3.2 Elasticities Calculated Within a Sub-estuary

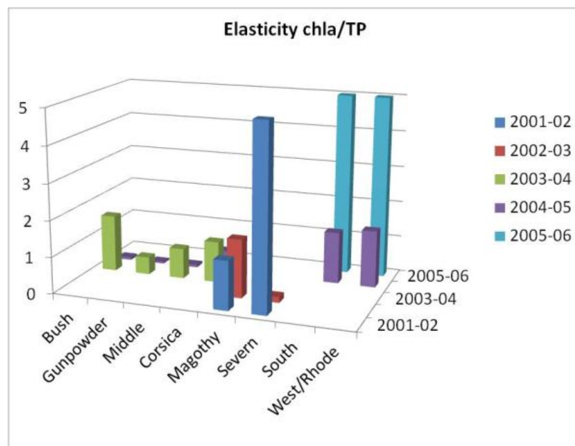
The basic elasticities calculated use data within a given subestuary to show the % change in chl-*a* relative to the % change in TN or the % change in TP (Figures 4-21 – 4-22). Magnitudes near 1 suggest that chl-*a* is responding proportionally to changes in the nutrient. A value well above 1 indicates extra sensitivity to a change in that nutrient and a value below 1 indicates relative insensitivity to a change in the nutrient. For the elasticity with respect to nitrogen, the values are close to one for many estuaries for changes from 2003-2004, both of which were “wet years” based on average annual flow into the Bay. Values were less than one (or unresponsive) between 2004 and 2005 for the estuaries in the northern portion of the Bay (Bush, Gunpowder, Middle). Changes in the Severn in 2002-03 were also unresponsive to TN. Values for other estuaries were well above 1. For phosphorus, the patterns are largely the same with a few notable exceptions. The Corsica between years 2003-04 was largely unresponsive to TP (elasticity = 0.57) in contrast to the high responsiveness to TN in that period (elasticity = 4.48). Another notable difference was decreased sensitivity of chl-*a* to TP in the South and West/Rhode relative to TN.



	Bush	Gun-powder	Middle	Corsica	Magothy	Severn	South	West/ Rhode
2001-02					2.719	25.725		
2002-03					2.163	0.295		
2003-04	1.493	0.516	1.005	1.383				
2004-05	0.209	0.113	0.388	4.483			7.670	3.644
2005-06							31.583	14.887

Figure 4-21. Elasticity of chl-*a* with respect to TN within sub-estuaries.

Note that the axis does not show the full range of data to allow better visualization over the most important range of values between 0-2. Total magnitude is less important than whether the value is above or below 1

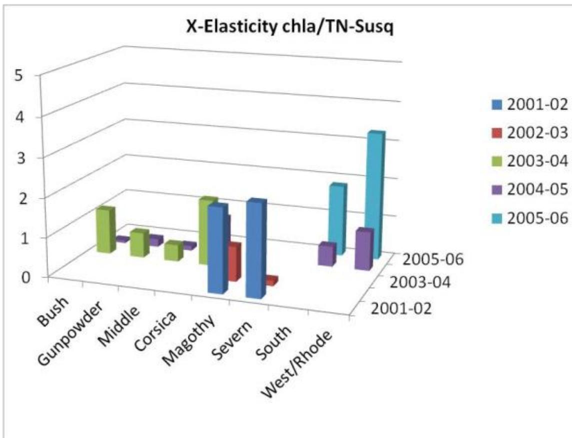


	Bush	Gun-powder	Middle	Corsica	Magothy	Severn	South	West/ Rhode
2001-02					1.338	6.722		
2002-03					1.603	0.160		
2003-04	1.570	0.481	0.834	1.133				
2004-05	0.068	0.048	0.068	0.572			1.421	1.568
2005-06							6.323	6.811

Figure 4-22. Elasticity of chl-*a* with respect to TP within sub-estuaries.

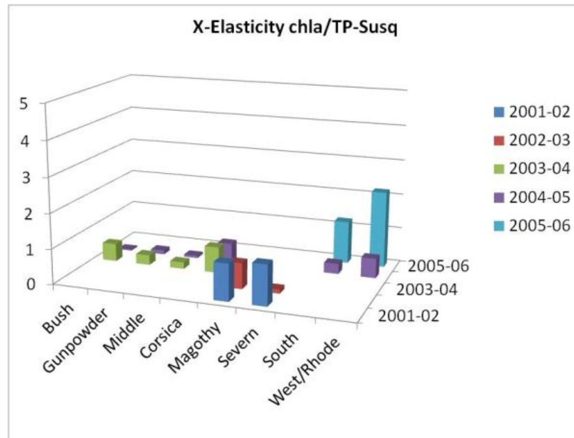
4-5.3.3 Cross-source (Local vs. Regional) Elasticity

The second type of elasticity that we calculated evaluated the % change in chl-*a* relative to the % change in the nutrients coming from the Susquehanna River (Figures 4-23 – 4-24). This elasticity thus tests responsiveness of within-sub-estuary water quality outcome with respect to Bay mainstem or regional loads. About 2/3 of cross-source elasticities for TN and 3/4 of elasticities for TP are less than one suggesting that the majority of responses are insensitive to Susquehanna loadings of these nutrients. The values for estuaries in the upper Bay appear more consistently close to one which is what we might expect for these estuaries close to the Susquehanna. But since we do not have measurements for the same years across estuaries, we cannot rule out that this effect is due to interannual differences rather than spatial differences.



	Bush	Gun-powder	Middle	Corsica	Magothy	Severn	South	West/ Rhode
2001-02					2.127	2.325		
2002-03					0.884	0.139		
2003-04	1.157	0.644	0.430	1.691				
2004-05	0.064	0.205	0.121	0.968			0.520	0.996
2005-06							1.832	3.280

Figure 4-23. Cross-source Elasticity of chl-*a* with respect to Susquehanna TN.



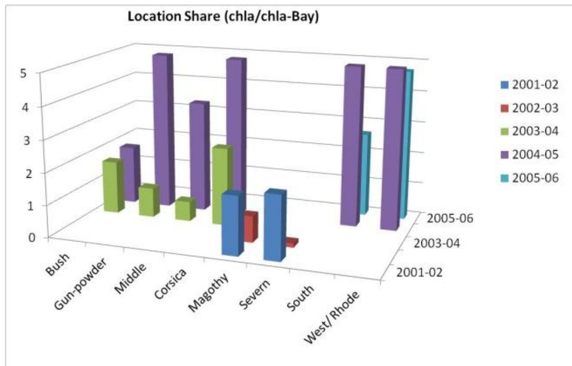
	Bush	Gun-powder	Middle	Corsica	Magothy	Severn	South	West/ Rhode
2001-02					1.049	1.147		
2002-03					0.733	0.115		
2003-04	0.509	0.283	0.189	0.745				
2004-05	0.036	0.116	0.068	0.545			0.293	0.561
2005-06							1.220	2.183

Figure 4-24. Cross-source Elasticity of chl-*a* with respect to Susquehanna TP

4-5.3.4 Local Share Statistic

The local share statistic is used to suggest how much of a local change in chl-*a* might be due to regional forces by comparing the % change in the annual chl-*a* mass (area-weighted and time-weighted) in the sub-estuary to the annual mass in the Bay mainstem (Eqn 2). This statistic is greater than 1 in all but 4 cases (estuary-year combinations) shown in Figure 4-25, suggesting that the chl-*a* in the sub-estuary is reflecting chl-*a* conditions in the mainstem. Although just under half the time, the percentage changes are in opposite directions, or in other words, negatively correlated (Figure 4-20).

A comparison of all the elasticity metrics show that interannual changes in the total mass of chl-*a* in a sub-estuary are most correlated with changes in chl-*a* in the mainstem rather than changes in nutrients (Figure 4-26). Among the potential nutrient drivers, chl-*a* changes were most responsive to within-estuary TN and TP and largely unresponsive to Susquehanna TN and TP. Evidence for this finding is that the median elasticity score calculated across all estuaries and years was highest for Bay chl-*a*, greater than one for within-estuary nutrients, and less than one for cross-source (Susquehanna) nutrients. However, a major limitation on the use of these indicators is that the direction of change in the driver is inconsistent with the change in the response of chl-*a*, a result that is masked by this metric. By examining the raw percentage changes, we see that it is more common to have chl-*a* change in the opposite direction of the change in nutrients and a change in chl-*a* in the mainstem is consistent with a change in chl-*a* in a subestuary, just over half the time.



	Bush	Gun-powder	Middle	Corsica	Magothy	Severn	South	West/ Rhode
2001-02					1.807	1.975		
2002-03					0.816	0.128		
2003-04	1.678	0.933	0.623	2.452				
2004-05	1.858	5.969	3.518	28.122			15.091	28.945
2005-06							2.640	4.725

Figure 4-25. Local share statistic of chl-*a* response in sub-estuary compared to chl-*a* change in Bay mainstem.

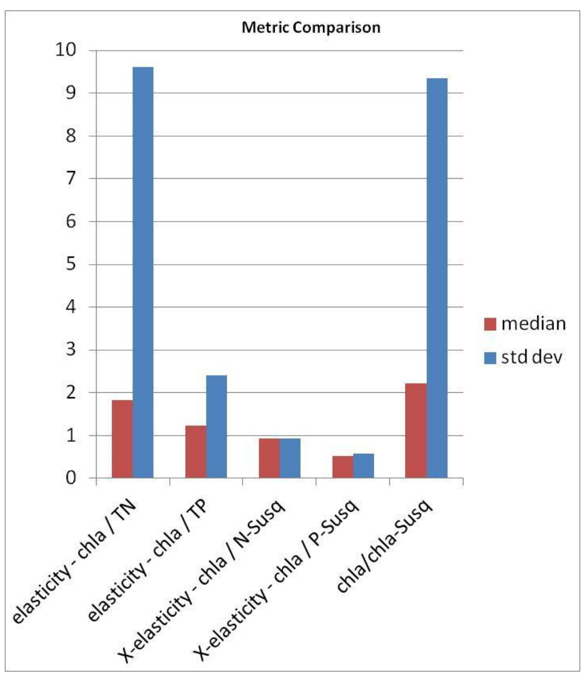


Figure 4-26. Comparison of elasticity metrics.

4-6 Conclusions and Management Implications

The four estuaries examined demonstrated a consistent pattern of highest average chl-*a* in the upper reaches, often localized to the mouths of small tributaries. Because average chl-*a* was relatively low near the mouths of these tributaries, it suggests that nutrient sources in the watershed may be most important for driving these conditions and because nutrients are not being taken up as frequently by phytoplankton in these estuaries, that they are more likely to be net exporters of nutrients compared to the Magothy and Severn estuaries.

Salinity patterns observed using DATAFLOW© data reveal that mainstem Bay water is, generally speaking, having more influence on conditions on the right side of W shore tributaries looking up estuary, whereas watershed conditions are likely to be having more influence on the left-hand shores, particularly near the mouths of these tributaries. The effect is less pronounced in the more southern tributaries that we evaluated, such as the South and the West/Rhode. Given the consistency of these patterns across the 8 estuaries, it is possible that these patterns are caused by Coriolis-induced tidal rectification, although other forces cannot be ruled out without further work. This result suggests that areas with persistent blooms near sub-estuary mouths and on the right side looking up estuary may be recalcitrant to changes in local watershed management efforts.

A simple comparison of potential drivers of chl-*a* and SAV responses among the eight estuaries examined so far was not substantially improved by the addition of the additional four estuaries this year. The strongest correlation we found was the same as last year. Namely, at the whole-estuary level, persistently elevated chlorophyll-*a* was negatively correlated with SAV abundance, as expected. These results suggest the need for multi-variate models that can control for multiple factors and these models are being explored in a companion project.

Because annual chl-*a* mass does not change in the same direction as nutrients or mainstem chl-*a* from year to year, it is difficult to compare which elasticity indices are most reflective of sub-estuary interannual changes. Nonetheless, the fact that interannual changes are not consistent is an interesting finding in and of itself. Whether or not the changes are positively or negatively correlated does not seem related to the magnitude of change in nutrients nor does it remain consistent within any estuary or year (Figures 4-19 and 4-20). As a result, we can conclude that nutrient loadings within a given year, by themselves, are not enough to explain the spatial and temporal persistence of phytoplankton blooms, even when we hold many physical variables of the estuary constant by considering only the interannual changes in response. This suggests that responses in sub-estuaries to management efforts that reduce nutrient inputs are likely to show a highly variable response year to year. As a result, it may take years of monitoring data to detect trends in chl-*a* responses.

4-6.1 Future Directions

We have explored a variety of indicators that might be used to evaluate responsiveness of water quality conditions to management actions by location and have learned which indicators are more or less valuable for distinguishing responses. However, much work remains to be done to integrate spatial and temporal data sets to improve these indicators. Further, we have largely focused on either very fine spatial detail or summarized conditions for an entire sub-estuary over the year. The work of the larger group suggests that temporal divisions of drivers and responses and considering lags between them may improve indicator performance. Therefore, in future work, we hope to integrate different research efforts to develop estuarine responsiveness indicators through the use of statistical modeling that will serve to identify which subareas of the Bay are likely to be most and least responsive to management actions.

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Chapter 5

Mattawoman Creek: A Case Study

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5-1	INTRODUCTION	1
5-2	DESCRIPTION OF MATTAWOMAN CREEK WATERSHED AND ESTUARY	2
5-3	PREVIOUS STUDIES IN THIS SMALL ESTUARY	5
5-4	MATTAWOMAN CREEK AS A CASE STUDY SITE	6
5-5	APPROACH OF THIS EVALUATION	6
5-6	MATTAWOMAN CREEK WATER QUALITY AND HABITAT ANALYSIS	6
5-6.1	LONG-TERM WATER QUALITY MONITORING DATA	6
5-6.2	HIGH FREQUENCY CONMON DATA	14
5-7	COMMUNITY METABOLISM IN MATTAWOMAN CREEK	19
5-8	SAV IN MATTAWOMAN CREEK	21
5-9	NUTRIENT BUDGET FOR MATTAWOMAN CREEK	25
5-9.1	CURRENT AND HISTORICAL NUTRIENT SOURCES	25
5-9.2	INTERNAL NUTRIENT SINKS	28
5-9.3	EXCHANGES WITH POTOMAC RIVER	29
5-9.4	NUTRIENT SYNTHESIS	30
5-10	ECOSYSTEM LINKAGES: NUTRIENT CAUSE-EFFECT CHAIN FOR MATTAWOMAN CREEK	33
5-10.1	NUTRIENT LOADS AND ALGAL BIOMASS	33
5-10.2	ALGAL BIOMASS AND WATER CLARITY	35
5-10.3	ALGAL BIOMASS AND HYPOXIA	36
5-11	FUTURE EXPECTATIONS	38
5-12	REFERENCES	40

5-1 Introduction

During the past few years there has been a growing interest in the science and water quality management communities for analyses directed toward understanding ecosystem responses to management actions. This type of analysis has been a tradition in fisheries management but such analyses have not been as common in the water quality management community. Recent case studies of water quality management and ecosystem responses in the Chesapeake Bay region include the analysis completed for the Corsica River (Boynton *et al.* 2009), the Back and Upper Patuxent Rivers (Boynton *et al.* 2011) and two analyses relating nutrient conditions to sea grass (SAV) community status and trends (Orth *et al.* 2010; Ruhl and Rubicki 2010). The major point in all of these examinations concerns the restoration trajectory of these systems following strong management actions. In simple terms, these studies investigate what happens when nutrient loads are reduced. Do these systems follow a restoration trajectory similar to the degradation trajectory or do they exhibit different restoration pathways, with temporal response lags or other complicating features? A useful and brief discussion of this topic, with several examples, was developed by Kemp and Goldman (2008).

The analysis provided here focused on Mattawoman Creek, a small oligohaline/tidal freshwater tributary of the upper Potomac River estuary. This site was selected for analysis for several reasons. For many years (1970s-mid-1990s; possibly earlier than the 1970s) this system was quite eutrophic with large algal blooms and an absence of Submerged Aquatic Vegetation (SAV). A major reduction of point source nutrient loads was achieved during the mid-1990s and the system now supports regions of good water quality, extensive SAV beds and fish spawning areas. It has been the focus of study and interest by state, federal and volunteer organizations interested in preserving and improving the habitat quality in the face of growing development and regional planning. A recent report by the Interagency Mattawoman Ecosystem Management Task Force (MDDNR, MDP, MDE, MD State Highway Administration, USDOT, USFWS, USEPA, ICPRB & UMD Anthropology Department 2012) details the case for protection of the watershed resources of Mattawoman Creek. The report describes the estuarine portion of the creek as "what a restored Chesapeake Bay would look like" and highlights the biodiversity, exceptionally large forest tracks and productive fisheries.

A portion of our analysis is devoted to documenting and creating a cause-effect chain of ecosystem responses in Mattawoman Creek. The Mattawoman watershed is currently characterized as having substantial land areas in forested and other low impact land uses. The impervious surface coverage in the watershed is relatively low but is approaching levels associated with water quality, habitat and living resource degradation (>15%). There are potential plans for major highway construction crossing this watershed and associated development plans, all of which would tend to change stream flows and loads of nitrogen (N), phosphorus (P) and sediments to this system. There is considerable concern that water quality, habitats (in particular SAV communities) and the very substantial recreational largemouth bass fishery would be compromised if N, P and sediment loads increased. A portion of this analysis uses comparative ecological methods to suggest loading rates associated with water quality and habitat maintenance and degradation.

5-2 Description of Mattawoman Creek Watershed and Estuary

The Mattawoman Creek basin encompasses 245 km^2 of land, 7.4 km^2 of open tidal waters and 2.5 km^2 of wetlands; intertidal area is very small (Fig. 5-1). The basin to estuarine surface area ratio is about 33, a very high value. For comparison, the Back River system has a ratio of about 5 and the full Chesapeake Bay system has a ratio of about 14. The significance of this ratio is that it provides a qualitative index of the relative influence of adjacent land on receiving waters….the larger the ratio the more land drains into a specified estuarine area. Some refer to this as the dilution ratio and use it as an index (admittedly a simple index) of estuarine susceptibility to pollution effects. The high ratio for Mattawoman Creek indicates an elevated potential for pollution effects. The shallow nature of this system further exacerbates this issue because there is simply not much tidal water to dilute the effects of land-derived nutrients, sediments or other pollutants.

The dominant 2010 land use in the Mattawoman basin was forested lands (54%); agricultural land uses accounted for 9.3% of watershed land uses (Table 5-1). Urban, suburban and other developed land uses occupied 35% of the basin land area. Between 1973 and 2010 urban lands increased by about a factor of three (12% to 35% of basin area) and agricultural and forested lands have both decreased. Changes in barren land and wetlands have been very small (Fig. 5-2). We obtained from Maryland Department of Natural Resources (MD-DNR) estimates of impervious surface in the Mattawoman watershed for the period 1950 to 2010. In 1950 less than 2% of the watershed had impervious surfaces. From 1950 to the mid-1980s there was a linear increase in impervious area and impervious cover reached about 5% by the mid-1980s. There was an increased rate of impervious surface cover between the mid-1980s and 2010. Impervious cover was just over 10% in 2010. As a rule of thumb, small basins with impervious cover greater than 10% often have impaired waterways (MD-DNR, pers. comm.).

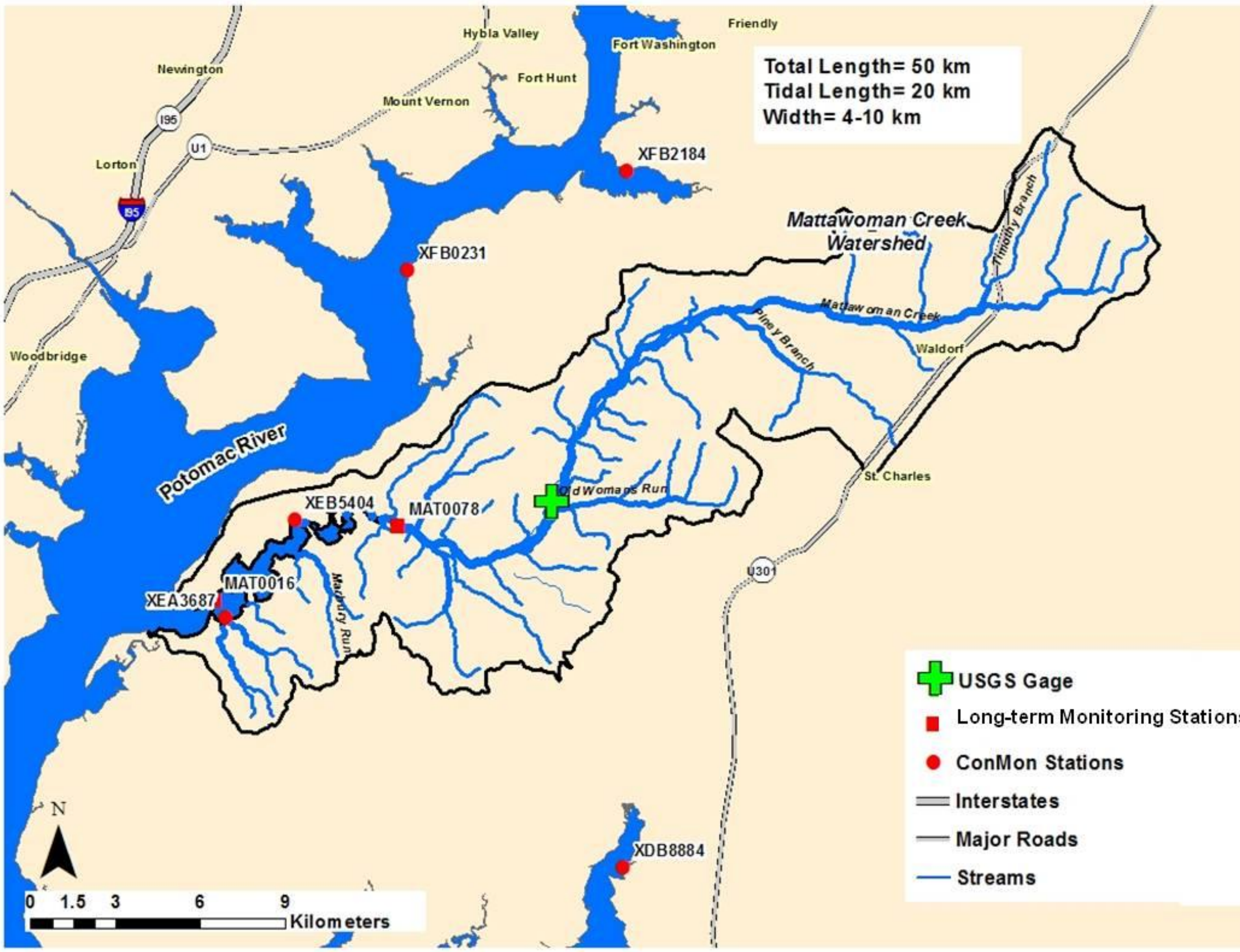

Figure 5-1. A map of Mattawoman Creek drainage basin boundary, stream network and major sampling locations referred to in the text

Mattawoman Creek is typical in size and volume of many of the small tributaries of the Potomac River estuary (Cronin and Pritchard 1975). Mattawoman Creek is about 50 km in total length; the lower 20 km are tidal (Fig. 5-1). The upper portion of the tidal estuary is narrow and meandering (25-100 m wide) and moderately turbid. The lower portion of the creek system is much wider (1-3 km), deeper (mean depth ~ 1.5 m), clearer, and vertically well-mixed most of the time. The surface area and volume of the tidal estuary is 7.4 x 10^6 x 10^6 m^2 and 10 x 10^6 m^3, respectively. SAV are a very prominent feature of this system covering about 3.5 km^2 of bottom during 2010 (~47% of creek surface area).

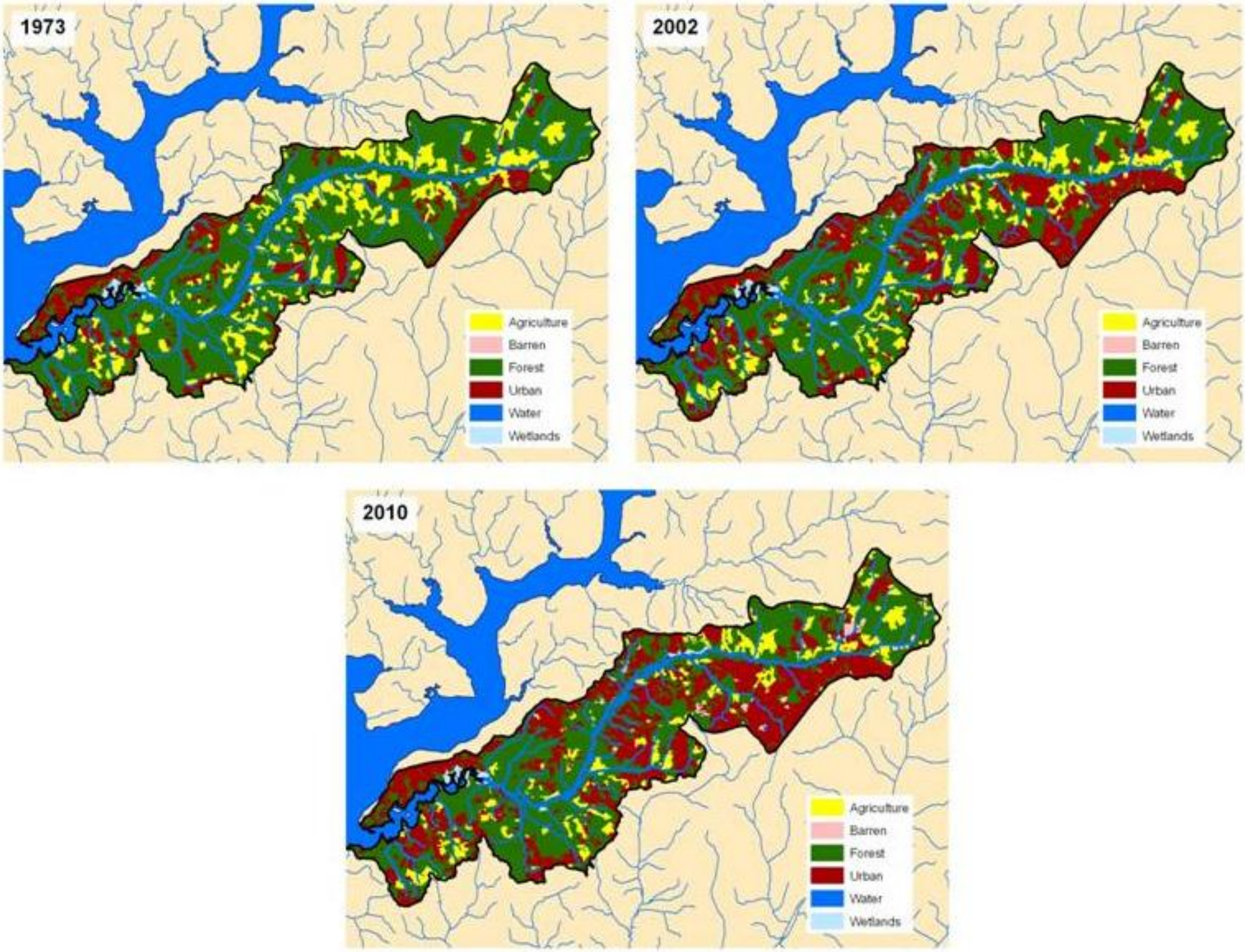

Figure 5-2. Simplified maps showing major land uses in the Mattawoman Creek watershed for the time periods 1973, 2002 and 2010. Created by A. Bayard from www.mdp.state.md.us and http://www.mdp.state.md.us/OurWork/landuse.shtml.

Table 5-1. A summary of land use/land cover in Mattawoman Creek watershed for three time periods (1973, 2002 and 2010). Created by A. Bayard from www.mdp.state.md.us and http://www.mdp.state.md.us/OurWork/landuse.shtml.

Land Use/ Land Cover Type	1973 (HA)	2002 (HA)	2010 (HA)	1973 (%)	2002 (%)	2010 (%)
Agriculture	3,951	2,901	2,280	16.16	11.87	9.32
Barren	0	48	243	0	0.20	0.99
Forest	17,193	14,477	13,142	70.37	59.21	53.75
Urban	3,053	6,672	8,447	12.49	27.29	34.55
Water	69	88	87	0.28	0.36	0.35
Wetlands	184	263	252	0.75	1.08	1.03
TOTAL AREA	**24,450**	**24,451**	**24,451**	**100**	**100**	**100**

Table 5-2. A simple set of watershed and estuarine characteristics for the Mattawoman Creek system.

Metrics	Watershed Characteristics
Land area, km^2	245
open tidal waters, km^2	7.4
wetland area, km^2	2.5
basin area:estuarine area	33

Metrics	Estuarine Characteristics
Creek length (tidal and non-tidal), km	50
Tidal creek length, km	20
Tidal creek mean depth, m	1.5
Tidal creek surface area, m^2	7,400,000
Tidal creek volume, m^3	10,000,000
SAV coverage (% of bottom area)	47

5-3 Previous Studies in this Small Estuary

When we initiated this analysis we were surprised to learn there has been a diversity of water quality and habitat measurements made in this relatively small tributary system during a considerable period of time. For example, some water quality variables (e.g., chlorophyll-*a*) date from the early 1970s. SAV coverage estimates also date back to the early 1970s. Other water quality variables (e.g., some nutrient concentrations) were also occasionally measured along the axis of the system. The strong impression we get from review of these early data is of a nutrient-enriched system with substantial algal blooms and no SAV communities. In 1984-85 the State of Maryland partnered with the US EPA Chesapeake Bay Program to initiate a long-term Biomonitoring Program and two water quality stations were established in the tidal portion of Mattawoman Creek (Fig. 5-1). In addition, this estuary also had a USGS River Input station located above the head of tide and from these data freshwater flow and nutrient and sediment loads could be estimated (http://va.water.usgs.gov/chesbay/RIMP/dataretrieval.html). Unfortunately, this site was active only from 2005-2011. Given plans for substantial land use changes in this basin it would be prudent to re-activate this station. Beginning in 2004 a ConMon site was established at the Smallwood State Park marina and this site has been active since then providing high frequency (15 minute interval) measurements from April – October of each year. An additional ConMon site was established at Indian Head (upper portion of the tidal waters of the creek) during 2009 and this site also remains active. All of these data are available from the Chesapeake Bay Program web site (http://www.chesapeakebay.net/) or linked sites. We also obtained land use model data for the Mattawoman watershed for 1985, 2002 and 2010 and these data were useful in estimating N, P and sediment loads for several time periods not available from the USGS record. Finally, we obtained data from the Chesapeake Bay Program water quality model and these data were used to estimate nutrient exchanges between the Potomac River estuary and Mattawoman Creek (G. Shenk, pers. comm.). Other data needed for this analysis (e.g., denitrification and nutrient burial rates, SAV N content, atmospheric N deposition rates, pre-colonial land use nutrient yield rates) were obtained from the literature, with an emphasis on Chesapeake Bay coastal plain information where possible.

5-4 Mattawoman Creek as a Case Study Site

As with all the sites considered for analysis, Mattawoman Creek was selected because there have been strong management actions achieved in this system. In the case, the major management actions were related to a very substantial diversion of WWTP discharges. Point source TN loads declined from about 332 kg N day^{-1} before management actions to about 26 kg N day^{-1} following actions. Similar severe reductions of point source TP loads were evident after management actions. There were only minor reductions in non-point TN and TP loads (Shenk, pers. comm.). Many issues concerning nutrient loads and ecological consequences have been addressed by D'Elia *et al.* (2003), Fisher *et al.* (2006), Boynton *et al.* (2008), Testa *et al.* (2008), and Testa and Kemp (2008). The CBP Biomonitoring Program's SAV aerial surveys indicated a dramatic response by the SAV community following nutrient load reductions in this system. This response indicated Mattawoman Creek would be a useful system to examine for ecological restoration dynamics. As indicated earlier, there is also concern in this basin related to possible large land use changes and the effects these might have on water quality and estuarine habitats and living resources.

5-5 Approach of this Evaluation

The focus of this case study analysis concerns water quality (chlorophyll-*a*, dissolved oxygen and water clarity) and habitat conditions (SAV communities) in Mattawoman Creek. We examine how these features have responded to past management actions and how the creek may respond to future land use alterations. Specifically, the analysis begins with an examination of time series water quality and habitat condition data, largely from 1985 to the present but also including some data from earlier times (1970s). We then summarize information available (from multiple sources) concerning nutrient loading rates from the surrounding basin, the atmosphere and the adjacent Potomac River and compare these with other estuarine systems. Using this local information and literature sources a nutrient budget is developed which places nutrient sources and sinks into perspective, an exercise useful for future nutrient management decisions. Finally, we attempt to develop a "cause-effect" chain relating nutrient loads to algal biomass, hypoxia, water clarity and SAV community status. This last analysis uses a comparative approach wherein data from other small, shallow, low salinity systems are combined in order to develop robust relationships among variables (Kemp and Boynton 2012). Ultimately, this analysis strives to provide guidance concerning the likely responses of this ecosystem to modifications in basin land use that would increase nutrient and sediments loads.

5-6 Mattawoman Creek Water Quality and Habitat Analysis

5-6.1 Long-term Water Quality Monitoring Data

We begin this analysis by examining time series data (1986-2011) collected at Long-term water quality sites (downstream site=MAT0016; upstream site=MAT0078) in Mattawoman Creek (Fig. 5-1).

Water temperature ranged from near-zero to 33 °C during the period of record. Maximum temperatures were generally higher at the downstream site and minimum temperatures were often lower at the upstream site. Annual average temperatures were 2-3 °C cooler at the upstream site. There were no obvious long-term trends in water temperature. We note that water temperatures in excess of 30 °C are considered extreme in the Chesapeake Bay system and can be harmful to some plant and animal species.

Mattawoman Creek is largely a tidal freshwater system. The highest salinity recorded was about 2.7 at the downstream site during 1987. Measureable salinity was rarely recorded at the upstream monitoring site. Higher and persistent salt was observed during the drought years of 1999-2002. During high flow (wetter) years no measureable salinity was recorded during any month of the year. The SAV community in Mattawoman Creek is composed of both invasive and native freshwater species (Table 5-3). During drought years, if higher salinity water intrudes into the creek SAV species may experience some physiological stress due to increased salinity. If this is the case, salinity stress may contribute to the observed inter-annual variability of SAV coverage.

Patterns of surface water pH in the creek were quite distinctive (Fig. 5-3). Monthly and annual average pH values differ by 0.5 to 2.5 pH units between upstream (lower pH) and downstream (higher pH) sites. It is likely the higher pH values at the downstream site were caused by the

Table 5-3. A listing of SAV species common and less common in the shallow waters of Mattawoman Creek. Note that two of the three common species are not native species. Table references: http://web.vims.edu/bio/sav/index.html , L. Karrh (MD-DNR) and N. Rybikki (USGS).

Mattwoman SAV Species	
Most Common	**Also Observed**
Ceratophyllum demersum (coontail) *Hydrilla verticillata* (hydrilla) *Myriophyllum spicatum* (Eurasian watermilfoil)	*Najas guadalupensis* (southern naiad) *Najas minor* (spiny naiad) *Najas flexilis* (northern naiad) *Heteranthera dubia* (water stargrass) *Potamogeton pusillus* (slender pondweed) *Vallisneria americana* (wild celery)

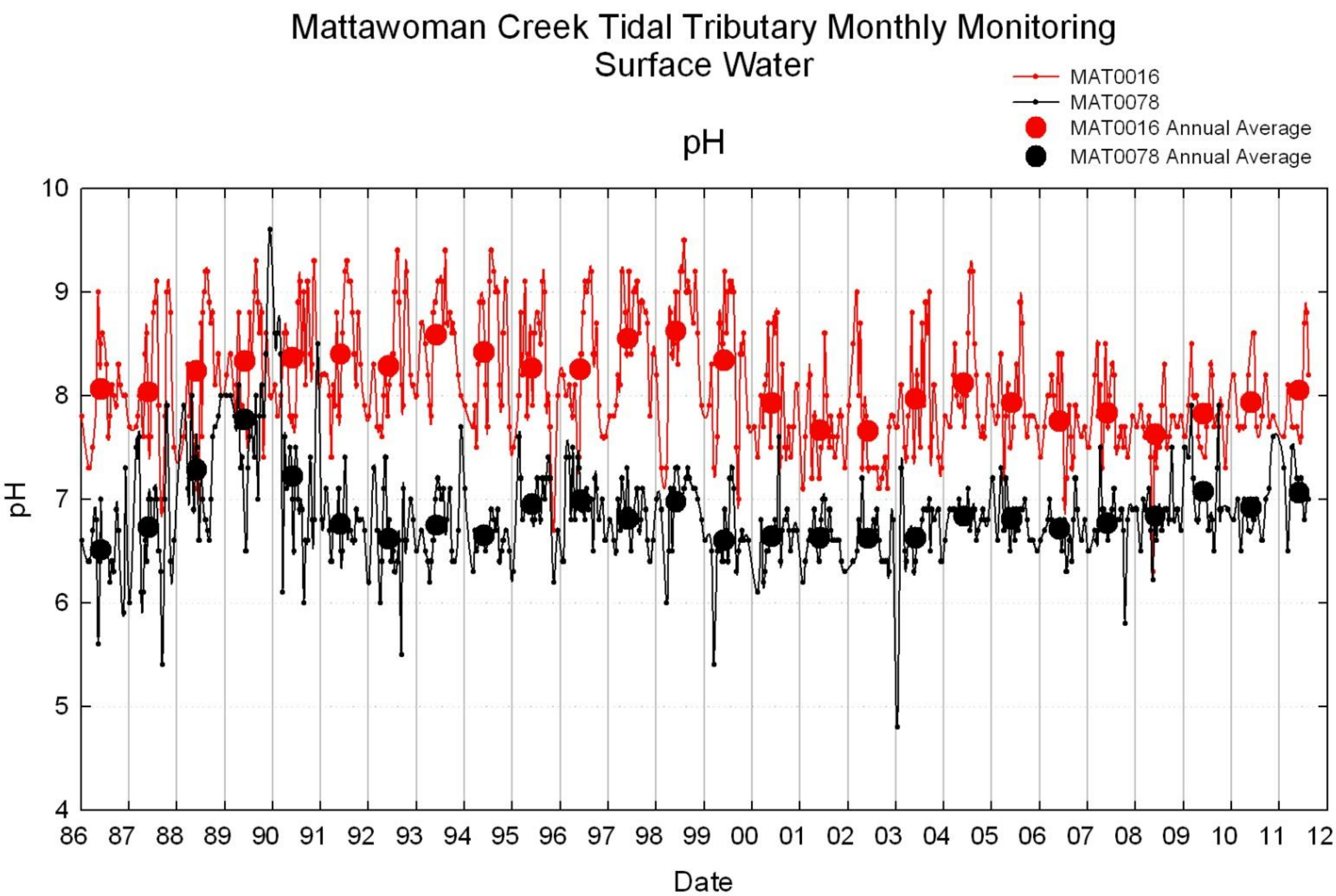

Figure 5-3. Monthly (small dots) and annual average (large dots) surface water pH values collected from 2 monitoring sites on Mattawoman Creek (MAT0016 and MAT0078) from 1986-2011. See Figure 5-1 for station location. Data from http://www.chesapeakebay.net/data.

large algal stocks typical of this area prior to WWTP discharge modifications. The difference in pH values decreased after algal stocks decreased in size and persistence. The fact that pH was highest during summer periods is consistent with the pattern in algal biomass. From 1986 – 2000 warm season pH values exceeded 9.0 either a few or many times during each summer period. The significance of this is that P sorbed to iron-rich sediment particles becomes soluble at pH of 9.0 or greater and thus becomes available to support additional algal growth. In essence, algal growth causes increases pH (phytoplankton use CO_2 which shifts pH up), increased pH liberates P from sediments and this P serves to enhance algal growth…an autocatalytic cycle which accelerates eutrophication (Kemp *et al.* 2005).

Nitrate plus nitrite (NO_{23}) and phosphate (PO_4) are essential plant nutrients, the excessive supply of which is often a root cause of estuarine eutrophication. Time series of these nutrients are provided in Figures 5-4 and 5-5. Concentrations of NO_{23} ranged from 0.003 to about 3 mg L^{-1} and were uniformly higher at the downstream site throughout the period of record. This sharply contrasts with most estuarine sites wherein nutrient concentrations decrease with distance from the riverine (upstream) sources. In the case of Mattawoman Creek the higher NO_{23} concentrations at the downstream site likely reflect proximity to the Potomac River which is

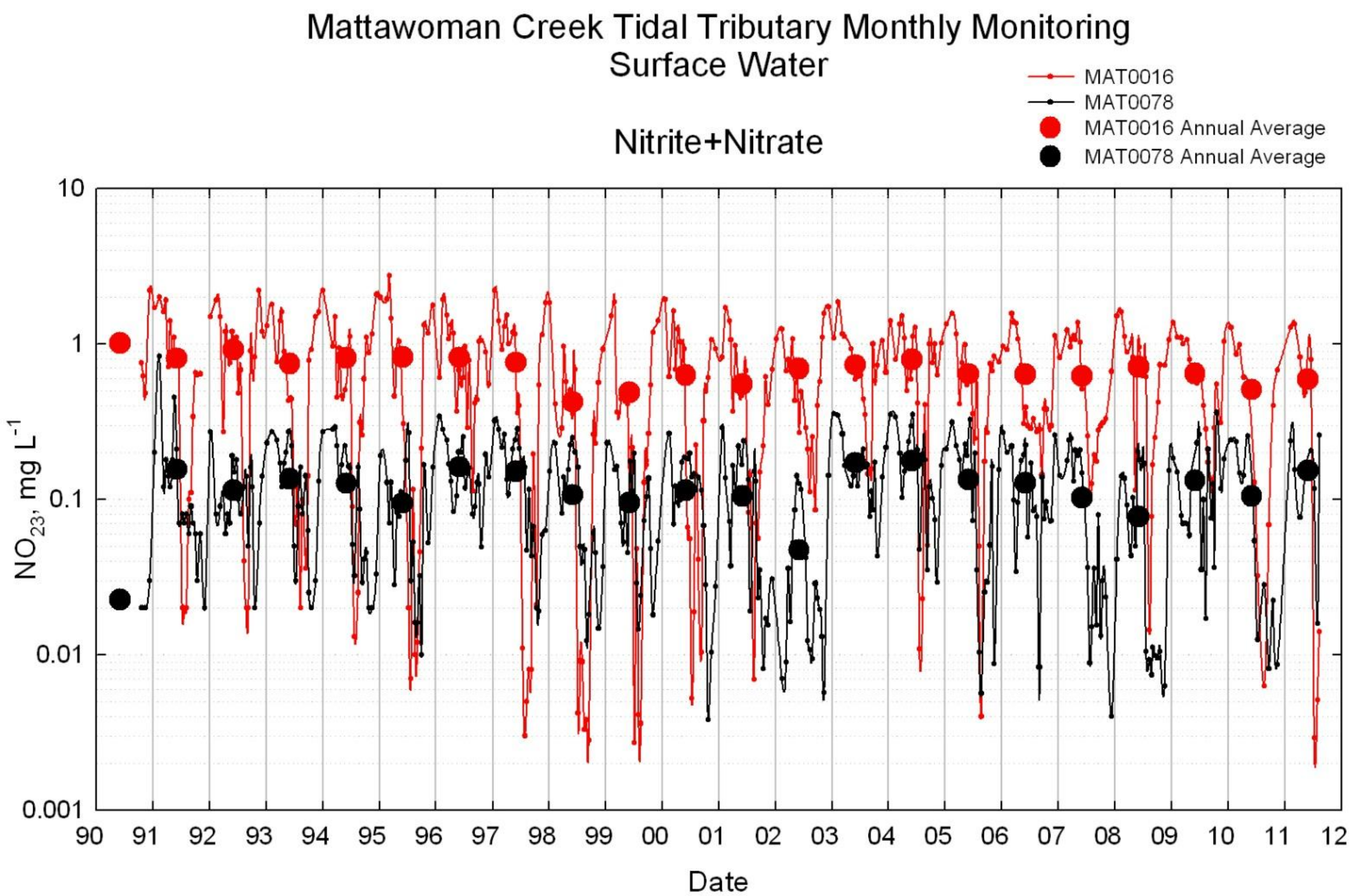

Figure 5-4. Monthly (small dots) and annual average (large dots) surface water NO_2 + NO_3 concentration collected from 2 monitoring sites on Mattawoman Creek (MAT0016 and MAT0078) from 1986-2011. See Figure 5-1 for station location. Data from http://www.chesapeakebay.net/data.

characterized by elevated N concentrations for much of the year. Highest NO_{23} concentrations occurred during winter spring, coincident with periods of high Potomac and local river flow. Concentrations were at times 2 orders of magnitude lower during the warm periods of the year coincident with rapid SAV and phytoplankton growth and with suspected periods of the year when denitrification rates are highest. During summer periods NO_{23} concentrations were frequently below N half-saturation (k_s) values for estuarine phytoplankton (< 0.035 mg L^{-1}). Although somewhat difficult to see, NO_{23} concentrations (Fig. 5-4) at the downstream site have decreased slowly over time, possibly as a result of Potomac River WWTP upgrades and diversion of WWTP discharges away from Mattawoman Creek. No obvious trends in NO_{23} concentration were evident at the upstream site.

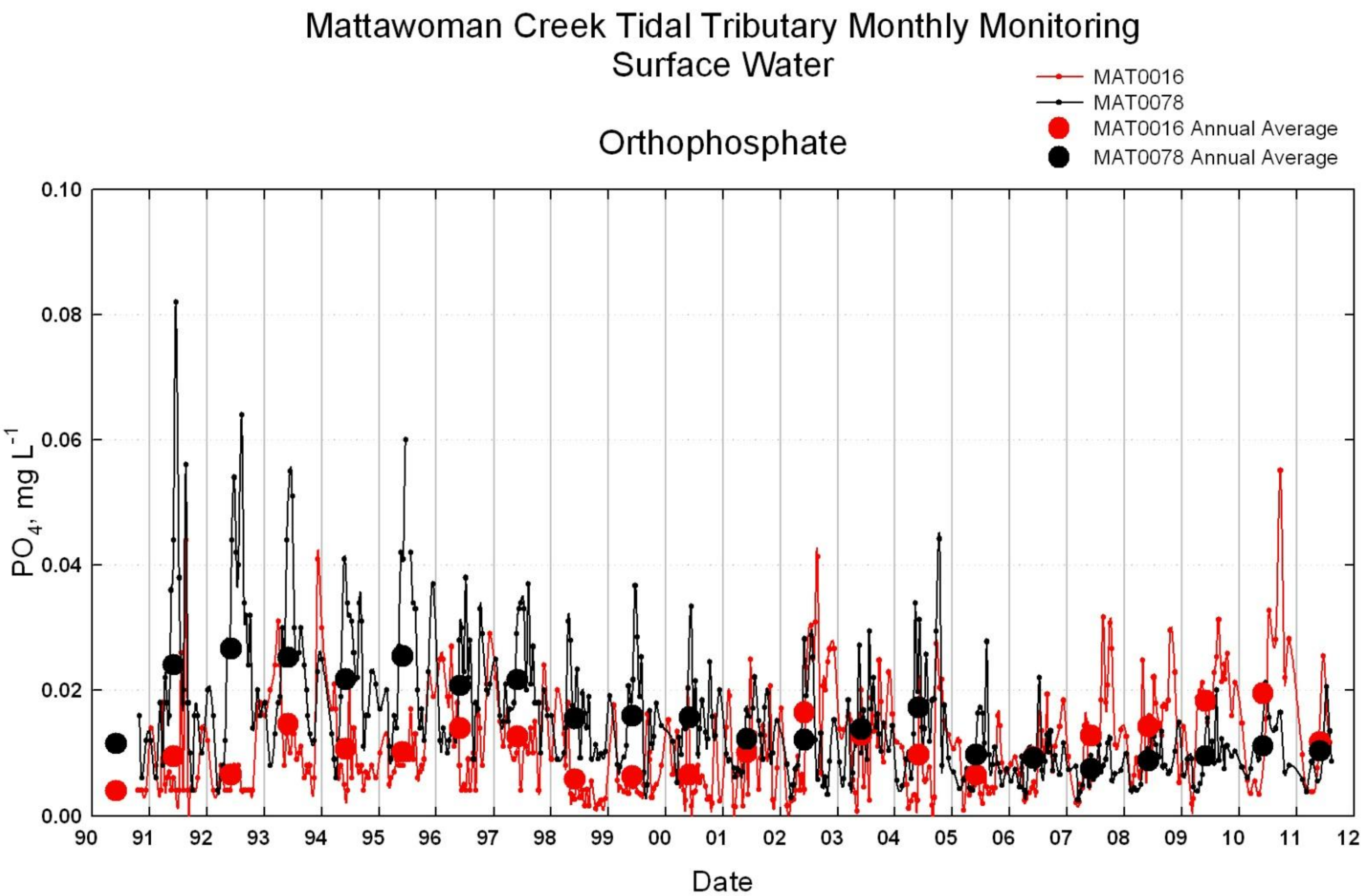

Figure 5-5. Monthly (small dots) and annual average (large dots) surface water orthophosphate (PO_4) concentration collected from 2 monitoring sites on Mattawoman Creek (MAT0016 and MAT0078) from 1986-2011. See Figure 1 for station location. Data from http://www.chesapeakebay.net/data.

The time-series of PO_4 concentrations in Mattawoman Creek indicate a complex pattern (Fig. 5-5). Concentrations ranged from 0.005 to 0.08 mg L^{-1} at the upstream site and from about 0.002 to 0.06 mg L^{-1} at the downstream site. These are typical values for a low salinity estuarine ecosystem (Boynton and Kemp 2008). In this case, PO_4 concentrations were higher at the upstream site, as expected, during the early portion of the record (1991-2004) and then declined to levels lower than those at the downstream site. Since 2005 PO_4 concentrations at the downstream site have been increasing. The reasons for these patterns are not clear at this time.

Concentrations of total nitrogen (TN) and total phosphorus (TP) are shown for the period 1991-2011 in Figures 5-6 and 5-7. During the period of record TN concentration ranged from about 0.6 to 3.0 mg L^{-1} at the downstream station and from 0.25 to 2.0 mg L^{-1} at the upstream site. TN concentrations were generally highest at the downstream site during the cold seasons and highest at the upstream site during warm seasons. There was no strong temporal change in TN concentration at the upstream site but there was a clear decline in concentration at the downstream site, likely related to modifications in WWTP discharge. TP concentrations were generally similar between upstream and downstream sites for the period of record and ranged from the level of detection (~0.01 mg L^{-1}) to about 0.3 mg L^{-1}. Highest TP values consistently

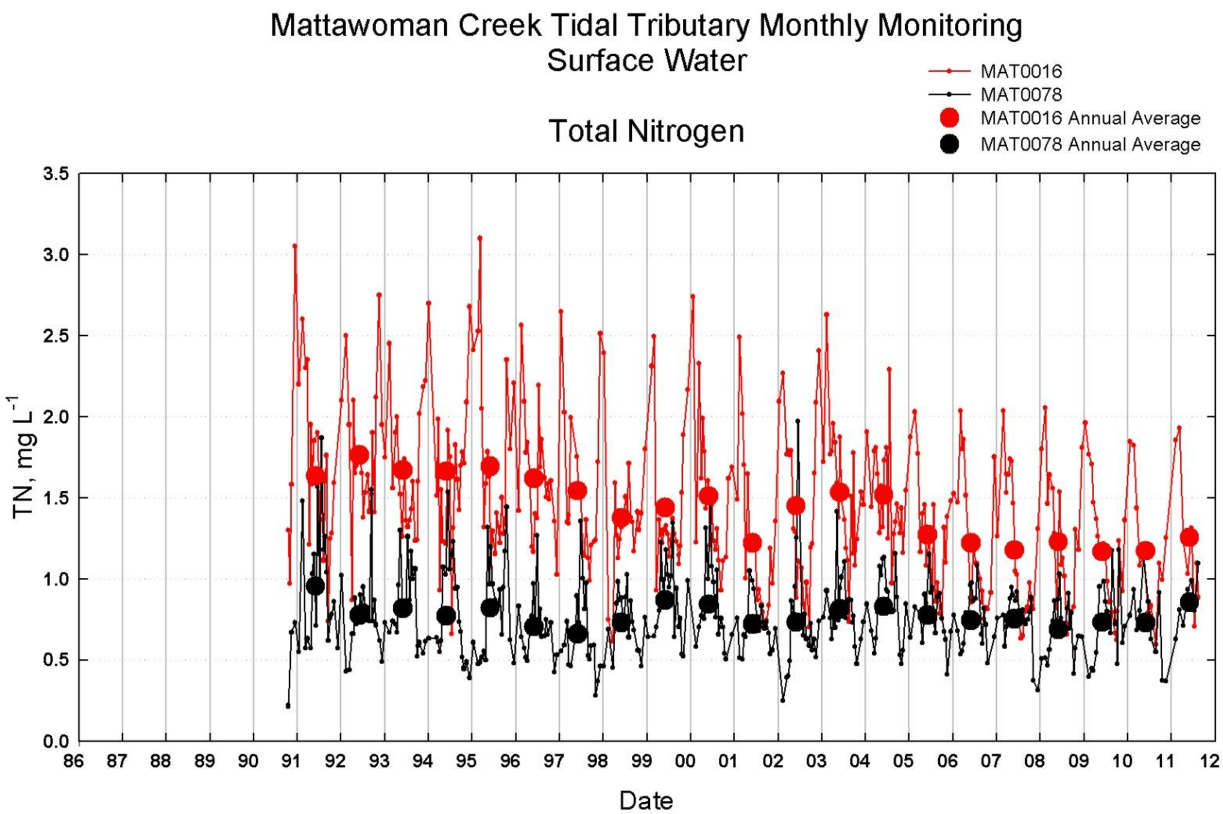

Figure 5-6. Monthly (small dots) and annual average (large dots) surface water Total Nitrogen (TN) concentration collected from 2 monitoring sites on Mattawoman Creek (MAT0016 and MAT0078) from 1986-2011. See Figure 5-1 for station location. Data from http://www.chesapeakebay.net/data.

occurred during the warmer portions of the year, a pattern frequently observed in shallow estuarine environments (Boynton and Kemp 2008), caused by active sediment releases of P at a time of the year when autotroph growth is limited by N (Fig. 5-7).

Secchi Disk data are only available for the downstream station for the period 1986-2011 (Fig. 5-8). Secchi Disk values ranged from about 0.2 to 2.7 m. There was a clear trend in water clarity with values of about 0.5 m early in the record and then increasing sharply after 2004 to an annual average of about 1.1 m during 2009. Secchi Disk values also dropped sharply during the first portion of 2011 for reasons that are not clear at this time. Water clarity is a key issue regulating SAV community health. In the adjacent Potomac, Ruhl and Rybicki (2010) reported strong correlations between water clarity and SAV community density, coverage and species composition. At their sites Secchi Disk values in excess of 0.65 m were associated with bed expansion, increased plant density and a return of native species. The Secchi Disk measurements reported here were made at sites along the main channel of Mattawoman Creek rather than in SAV beds. It may be that these values underestimate water clarity in the SAV beds as shown by Gruber and Kemp (2010) based on detailed water clarity and other measurements inside and outside SAV beds in the mesohaline Chesapeake Bay. Conversely, the Secchi Disk

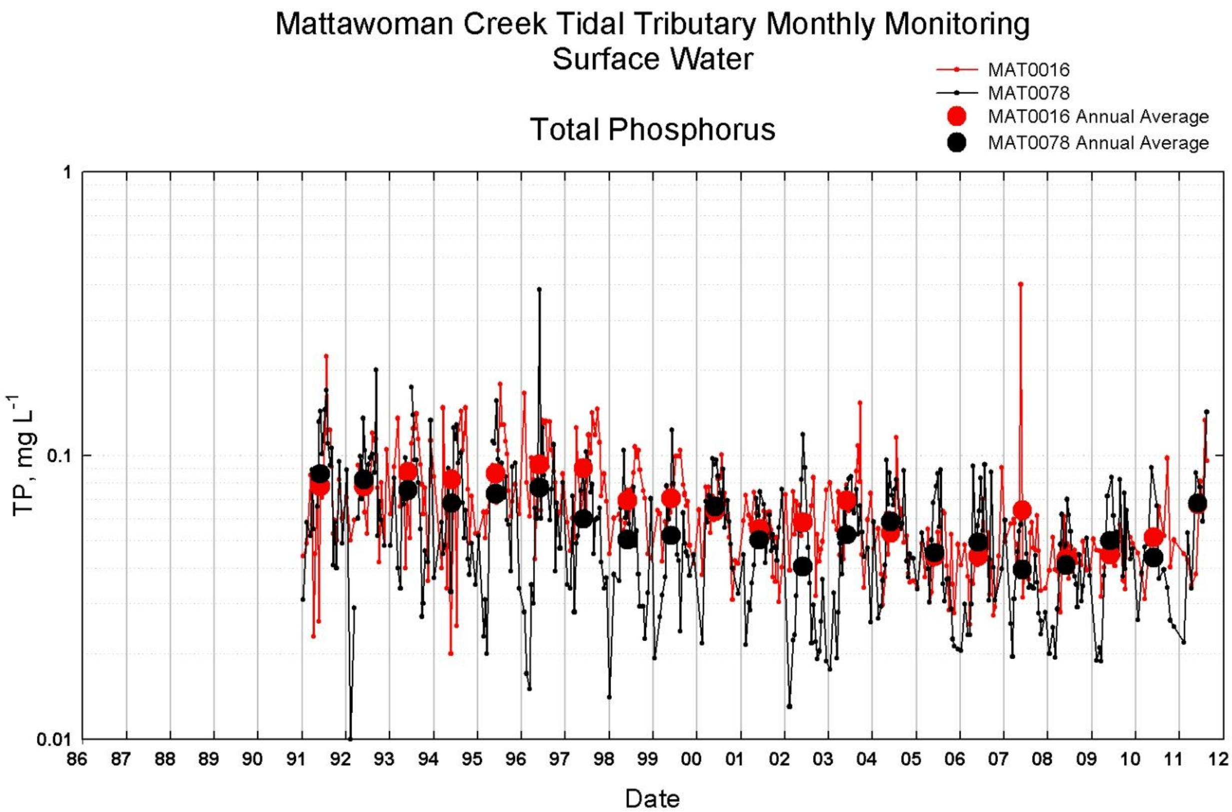

Figure 5-7. Monthly (small dots) and annual average (large dots) surface water Total Phosphorus (TP) concentration collected from 2 monitoring sites on Mattawoman Creek (MAT0016 and MAT0078) from 1986-2011. See Figure 5-1 for station location. Data from http://www.chesapeakebay.net/data.

measurements in the channel might also be higher than normal because SAV beds line much of the shoreline and tend to suppress sediment resuspension and efficiently trap sediments (Ward *et al.* 1984).

Chlorophyll-*a* concentrations varied between 0.3 and 110 µg L^{-1} at the downstream site and from 0.15 to 30 µg L^{-1} at the upstream site (Fig. 5-9). Typical values at the downstream site were higher, at times an order of magnitude higher, than at the upstream site. It is likely that a combination of limited light and shorter water residence time both contributed to lower algal biomass at the upstream site. There did not appear to be any long-term trend in chlorophyll-*a* concentration at the upstream site but there was a good deal of inter-annual variability ranging from 1.5 to 8 µg L^{-1}. However, there were several distinctive temporal trends at the downstream site. Chlorophyll-*a* concentrations were generally high (annual average concentration 20-40 µg

L^{-1}) from 1986-1998. Concentrations then steadily declined through 2009 (to about 5 µg L^{-1}). Since then there has been a rapid increase in concentration to an annual average of 18 µg L^{-1}

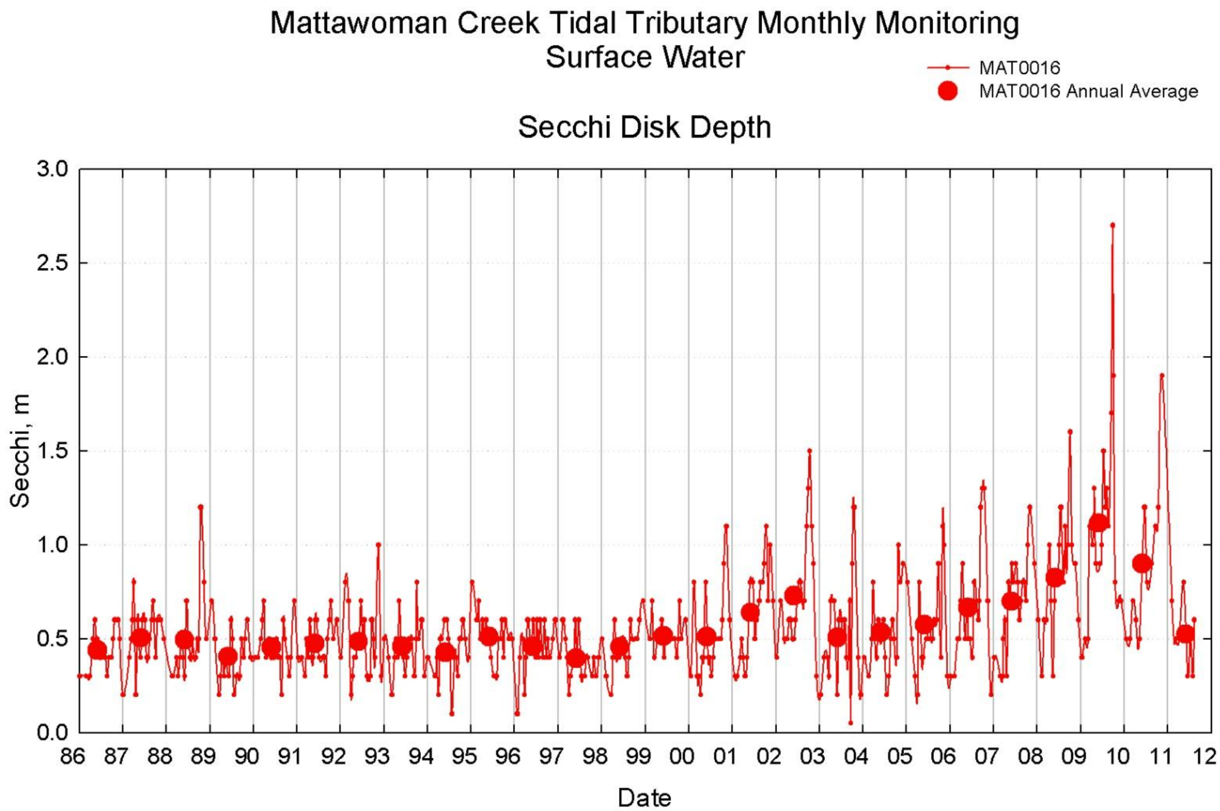

Figure 5-8. Monthly (small dots) and annual average (large dots) of Secchi Disk depth (m) collected from a monitoring site on Mattawoman Creek (MAT0016) from 1986-2011. See Figure 5-1 for station location. Data from http://www.chesapeakebay.net/data.

during the first portion of 2011. The decline seemingly could be attributed to changes in WWTP operations; however, the cause of the recent increase is not clear at this time.

The general picture of water quality that emerges from these data indicates an increase in water quality associated with changes in WWTP operations. Water column pH, NO_{23}, PO_4, TN, TP and algal biomass all declined and water clarity and SAV community metrics both increased.

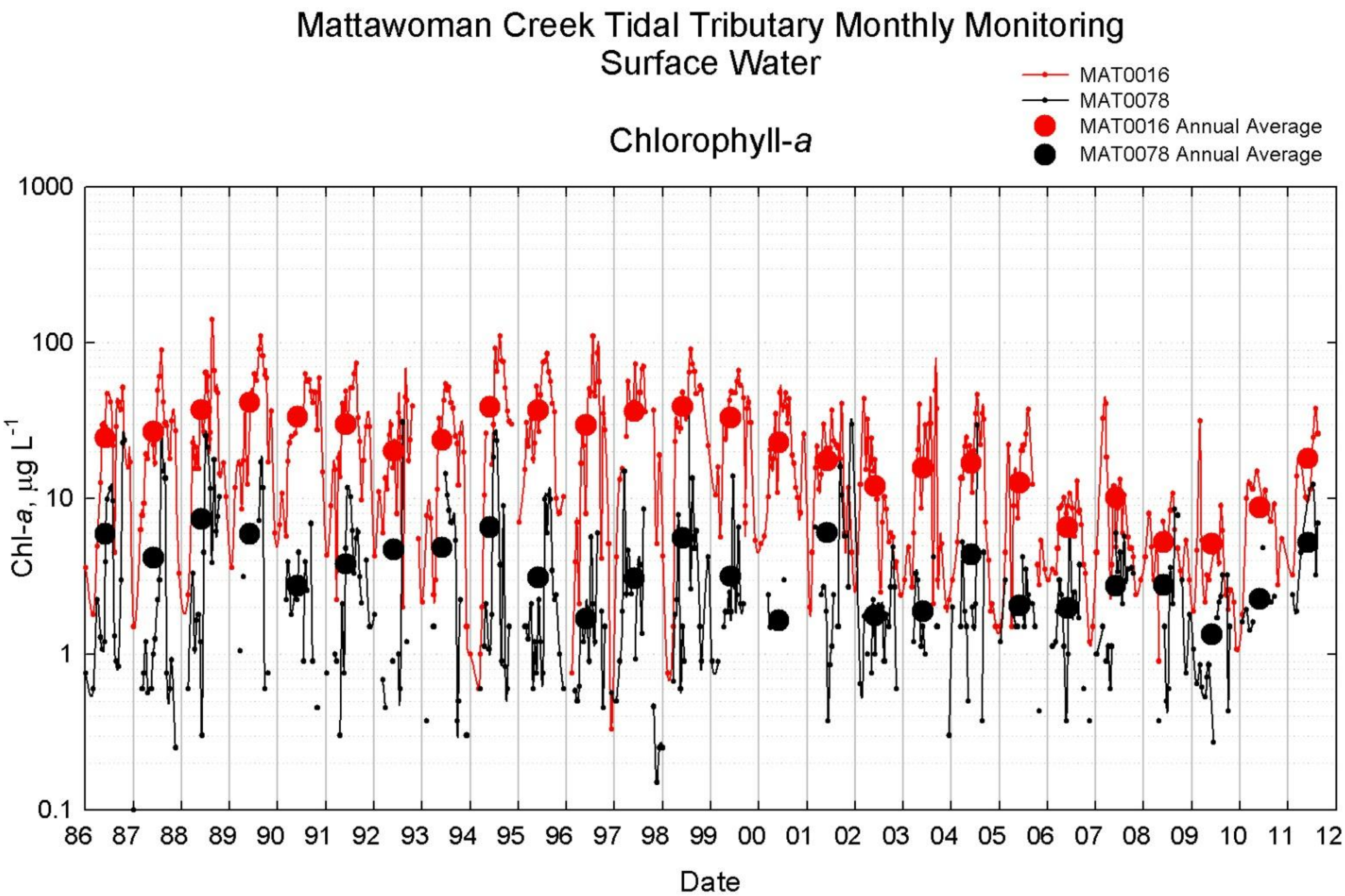

Figure 5-9. Monthly (small dots) and annual average (large dots) surface water chlorophyll-*a* concentration collected from 2 monitoring sites on Mattawoman Creek (MAT0016 and MAT0078) from 1986-2011. See Figure 5-1 for station location. Data from http://www.chesapeakebay.net/data.

5-6.2 High Frequency ConMon Data

In addition to the traditional monthly (or bi-monthly) water quality sampling of Mattawoman Creek, two high frequency monitoring sites have also been established (Fig. 5-1) and these provide water quality at 15 minute intervals from April-October. One site (XEA3687) is located in the downstream portion of the creek at Smallwood State Park and the other site (XEB5404) is located upstream at Indian Head. The first site has been operational since 2004 while the Indian Head site has only been operational since the last portion of 2009. Data collected at these sites include temperature, salinity, pH, water clarity (as NTUs), dissolved oxygen and chlorophyll-*a*. We have averaged these data to monthly values; the high frequency measurements (not averaged) were used to estimate community metabolism and these results will be presented later. ConMon values for temperature, salinity and pH were similar (but of much shorter duration) to those obtained from the monthly monitoring work and will not be repeated here; turbidity, DO and chlorophyll-*a* data are presented below.

With one exception, high frequency turbidity data (NTUs) exhibited a regular pattern with higher values during winter-spring (>20 NTUs) and much lower values during summer and fall (>5 NTUs; Fig. 5-10). There was also a general trend between 2004 and 2010 of increasing water clarity at the downriver site (the record is very short at the upstream site). The clearest water occurred during the SAV growing season in all years; the presence of extensive SAV beds may well have contributed to this excellent water clarity as described by Gruber and Kemp (2010) for other SAV beds in Chesapeake Bay.

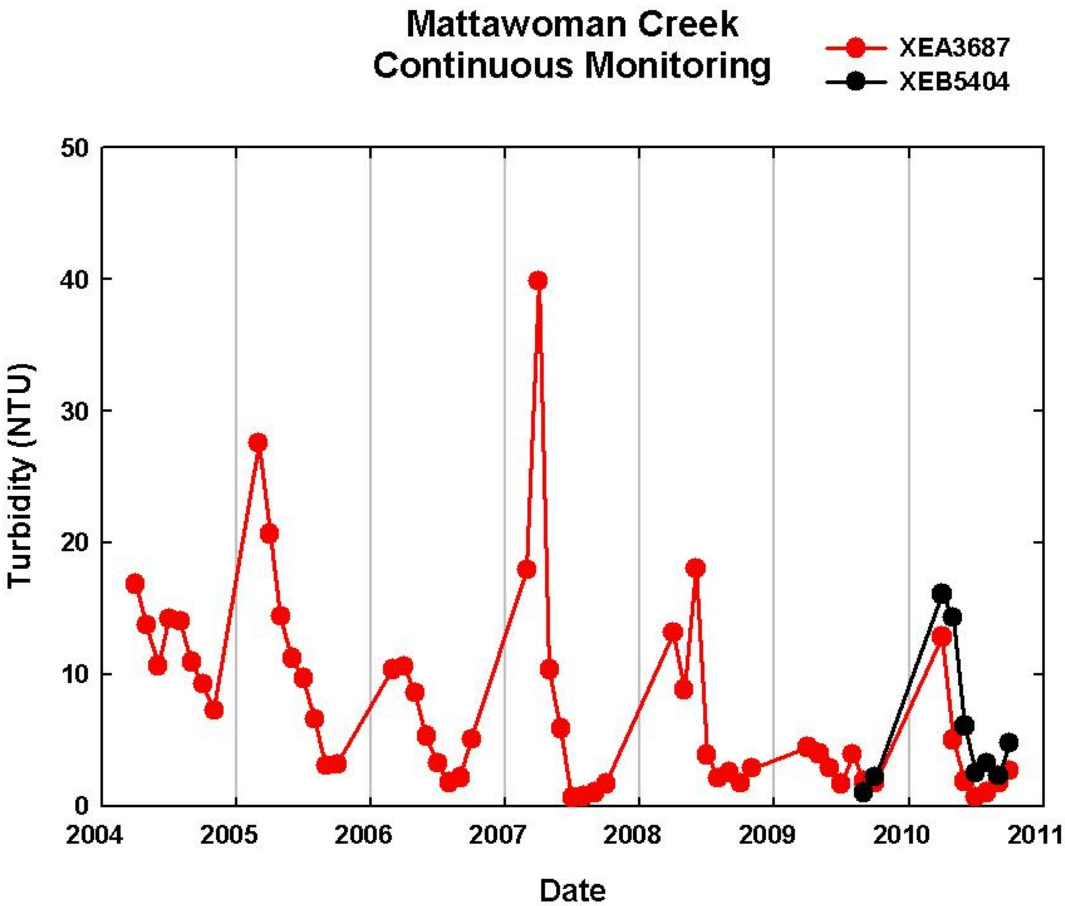

Figure 5-10. Monthly average values for surface water turbidity (NTUs) based on ConMon data collections at two Mattawoman Creek sites (XEA3687 from 2004-2010 and XEB5404 from 2009-2010). In general, months included for each year were April-October. See Figure 5-1 for station location. Data from http://www.chesapeakebay.net/data.

Dissolved oxygen concentrations also exhibited regular patterns with highest values during winter (due, in part, to greater oxygen solubility in cold water) and lower values during the summer (Fig 5-11). However, there was also a trend evident in this short time-series wherein both cold and warm season DO concentrations were decreasing. During 2006, average maximum (winter) and minimum (summer) DO concentration was about 12.5 and 7 mg L^{-1}, respectively. By 2010 those concentrations had decreased to 10.5 and 3.5 mg L^{-1}, respectively. The reasons for this trend are not completely clear at this time. However, field staff from MD-DNR report that during the period 2004-2008 SAV grew right up to the dock where the ConMon meter was attached…in short, during those years the sonde was in the SAV bed. During 2009 and 2010 the SAV bed had moved away from the dock but was still very thick immediately adjacent to the dock. It may be that actively growing SAV generally elevated summer DO concentrations in earlier years. During 2009-2010 the SAV community may have impeded water circulation adjacent to the pier and, in effect, created a settling pond wherein respiratory activity (oxygen consumption) dominated over photosynthetic (oxygen producing) activity. This trend is of concern because during several months of 2010 30-day DO criteria (5 mg L^{-1}) were violated.

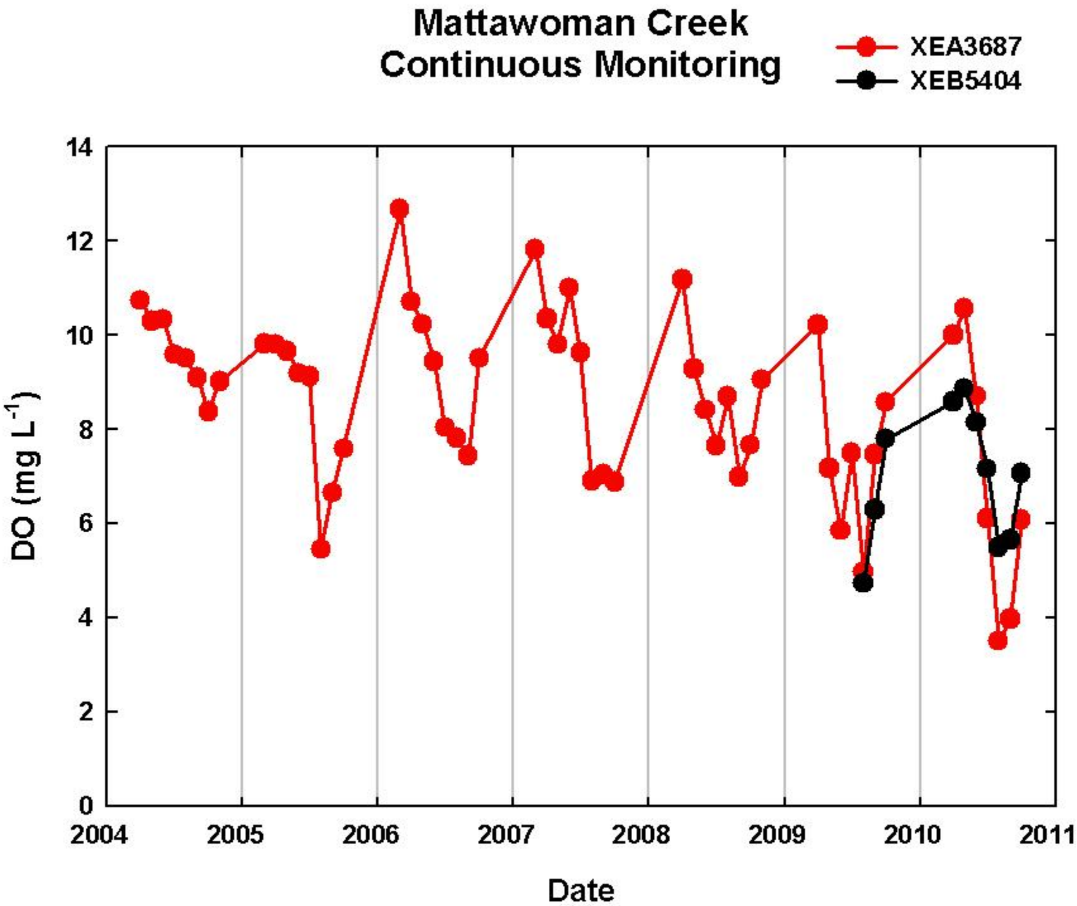

Figure 5-11. Monthly average values for surface water dissolved oxygen concentration based on ConMon data collections at two Mattawoman Creek sites (XEA3687 from 2004-2010 and XEB5404 from 2009-2010). In general, months included for each year were April-October. See Figure 1 for station location. Data from http://www.chesapeakebay.net/data.

We have also utilized high frequency ConMon data from both sites to compute summer season (Jun-Aug) compliance with DO criteria (Table 5-4). Descriptions of data sources and computation methods are provided in another chapter of this report (Chapter 2). In brief, percent failure of the instantaneous DO criteria (3.2 mg L^{-1}) at site XEA3687 was very low (< 3%) during summers of 2004-2008. At this site the 30-day DO mean criteria (5 mg L^{-1}) was also not regularly violated during this period except during 2005 when the failure rate was 14%. However, during 2009, and especially 2010, both DO criteria failure rates markedly increased (8 and 21% for instantaneous and 26 and 37% for 30-day mean criteria). In this case the 30-day mean criterion was more protective of DO conditions than the instantaneous criteria. It should also be noted that the continuous duration of DO below criteria levels tracked DO percent failure rates. The ConMon site at Indian Head (XEB5404) has only been active during 2009-2010 and hence not much can be concluded from these data. During both years at the Indian Head site the percent failure of the 30-day DO criteria was protective of the instantaneous criteria and during both years 30-day criteria failures during summer were in excess of 10%.

Table 5-4. A summary of DO criteria assessment results developed for Mattawoman Creek. Data were from ConMon sites XEA3687 (2004-2010) and XEB5404 (2009-2010). Criteria assessments (pass fail percentages and hours below DO criteria concentration) were developed for all months when ConMon data are collected (Apr-Oct) most DO failures occurred during the summer period (Jun-Aug). Data from www.eyesonthebay.net.

Location / Station ID	Year	Month	Total Measurement		Instantaneous Criteria < 3.2 mg L^{-1}			30-day Criteria < 5.0 mg L^{-1}		
			Hours	Days	Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)	Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)
Mattawoman Creek XEA3687	2004	June	720	30	0	0	0	0	0	0
		July	744	31	3	0	1	11	1	3
		August	744	31	8	1	2	16	2	5
		Summer Total	2208	92	11	0.5	2	27	1	5
	2005	June	704	29	0	0	0	0	0	0
		July	744	31	0	0	0	3	0	3
		August	744	31	65	9	16	296	40	21
		Summer Total	2192	91	65	3	16	299	14	21
	2006	June	720	30	26	4	14	70	10	42
		July	744	31	2	0.2	1	37	5	8
		August	538	22	1	0.1	1	14	3	4
		Summer Total	2002	83	29	1	14	120	6	42
	2007	June	720	30	0	0	0	0	0	0
		July	743	31	0	0	0	3	0.4	1
		August	744	31	10	1	3	110	15	9
		Summer Total	2207	92	10	0.4	3	113	5	9
	2008	June	720	30	0	0	0	26	4	4
		July	744	31	0	0	0	13	2	2
		August	673	28	1	0.1	1	49	7	6
		Summer Total	2137	89	1	0.1	1	88	4	6
	2009	June	720	30	24	3	6	205	28	19
		July	744	31	6	1	2	58	8	8
		August	744	31	139	19	15	323	43	21
		Summer Total	2208	92	170	8	15	585	26	21
	2010	June	720	30	0	0	0	7	1	4
		July	728	30	111	15	9	267	36	35
		August	744	31	356	48	31	543	75	56
		Summer Total	2192	91	467	21	31	817	37	56

Location / Station ID	Year	Month	Total Measurement		Instantaneous Criteria < 3.2 mg L^{-1}			30-day Criteria < 5.0 mg L^{-1}		
			Hours	Days	Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)	Hours Below Criteria	% Failure	Maximum Single Duration Below Criteria (Hours)
Indian Head XEB5404	2009	June	*	*	*	*	*	*	*	*
		July	*	*	*	*	*	*	*	*
		August	88	4	3	3	3	57	65	10
		Summer Total	88	4	3	3	3	57	65	10
	2010	June	720	30	1	0.1	0.3	2	0.2	1
		July	744	31	0	0	0	42	6	6
		August	744	31	17	2	3	241	32	12
*No Data		Summer Total	2208	92	18	1	3	258	13	12

Monthly average chlorophyll-*a* concentrations were also computed using ConMon data (Fig. 5-12). The record for the downstream ConMon site (XEA3687) is relatively short (2004-2010) but there seem to be some trends developing. Chlorophyll-*a* concentrations ranged from about 2 to 18 µg L^{-1} during the 7 year period, substantially lower than values reported for the late 1970s-1990s. Highest concentrations consistently occurred during winter-spring and lowest values during summer-fall. In addition, peak values declined from about 18 µg L^{-1} during 2004 to about 13 µg L^{-1} (or less) after 2007 and the number of months exhibiting low concentrations also increased. These high frequency data indicate a decline in chlorophyll-*a* during this relatively short period of time. Limited data from the upstream ConMon site (XEB5404) exhibited the same pattern as the downstream site during portions of 2009-2010.

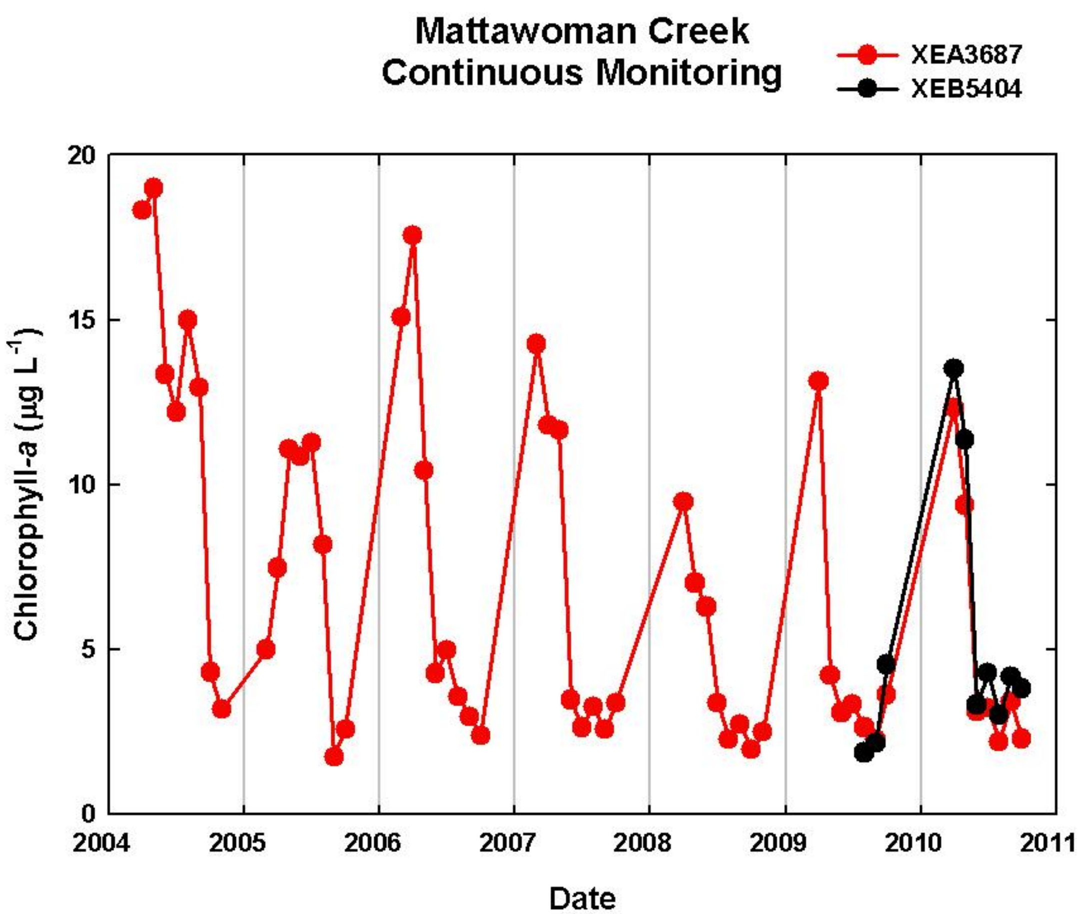

Figure 5-12. Monthly average values for surface water chlorophyll-*a* concentration based on ConMon data collections at two Mattawoman Creek sites (XEA3687 from 2004-2010 and XEB5404 from 2009-2010). In general, months included for each year were April – October. See Figure 5-1 for station location. Data from www.eyesonthebay.net.

5-7 Community Metabolism in Mattawoman Creek

ConMon data are ideal for computing estimates of community production (photosynthesis) and respiration which are basic properties of all ecosystems. We have adapted the Odum and Hoskin (1958) approach to computing community metabolism. Details of this method and the

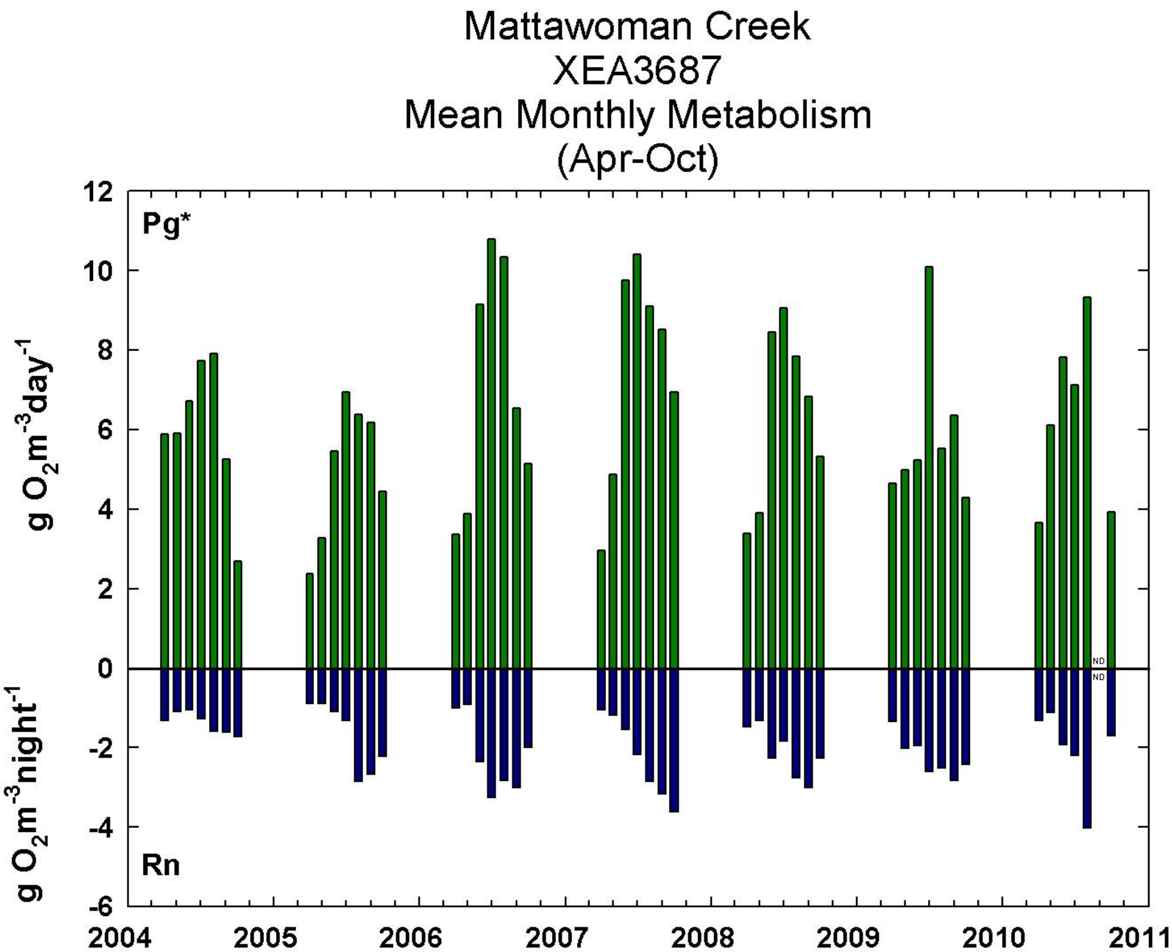

Figure 5-13. Mean monthly (Apr-Oct) estimates of community gross primary production (g O_2 m^{-3} day^{-1}) and community respiration (g O_2 m^{-3} night $^{-1}$) for the period 2004-2010. These estimates were generated following the technique of Odum and Hoskin (1958). Data used in these computations were from the ConMon site XEA3687 in Mattawoman Creek. See Figure 5-1 for station location. Data used in these computations are available at the Eyes on the Bay web site (www.eyesonthebay.net).

computational scheme we used are provided elsewhere in this report. In brief community production is inferred from the daytime increase in DO concentration and community respiration from the nighttime decline in DO concentration. Both rates are corrected for oxygen diffusion between the water and atmosphere which, in turn, is governed by water temperature and salinity effects on dissolved oxygen saturation in water.

Rates of community gross photosynthesis (Pg*) ranged from about 2 to 11 g O_2 m^{-3} day^{-1} while community respiration rates ranged from 1 to 4 g O_2 m^{-3} $night^{-1}$ (Fig. 5-13). In general rates were lower during spring (Apr-May) and fall (Sep-Oct) and highest during Jun-Aug, particularly during July. Summer average rates were low during 2004-2005, increased considerably during 2006 and then declined through 2010. In isolation these values do not mean a great deal. To place these in perspective we have summarized metabolism rates for a variety of Chesapeake Bay systems ranging from very nutrient enriched to less enriched (Table 5-5). In general, these rates increase in proportion to nutrient loading rates (Caffrey 2004). What is evident is that rates in Mattawoman Creek tend to be low compared with rates measured in moderately (e.g., Patuxent) and heavily enriched (e.g., upper Potomac and Corsica Rivers) ecosystems. This result is consistent with the fact that SAV are abundant in Mattawoman Creek and such communities are not usually associated with heavily enriched systems. A recent examination of SAV community status relative to nutrient loads for a large number of small estuaries in southern New England clearly indicated that SAV were associated with low nutrient loading rates (Lattimer and Rego 2010) and SAV resurgence in portions of Chesapeake Bay were associated with nutrient load reductions (Orth *et al.* 2010). In another portion of this report we attempt to more quantitatively relate nutrient loading rates to community production rates using Mattawoman data and data from a variety of Chesapeake Bay systems.

Table 5-5. Examples of differing rates of community gross primary production based on ConMon data collected at these sites for one or more years. Summer = June-August. See text for description of calculation method. Data from www.eyesonthebay.net.

Nutrient Enrichment Status	System or Location	Summmer Average Gross Primary Production g O_2 m^{-2} day^{-1}
Very Enriched	Bishopville (MD Cstl Bays)	**17.0**
	Turville Ck (MD Cstl Bays)	**13.0**
	Piscataway Ck (Upper Potomac)	**16.0**
	Fenwick (Upper Potomac)	**15.3**
	Upper Corsica River	**12.3**
	Back River	**14.3**
Moderately Enriched	St. Georges Ck (Lower Potomac)	**7.3**
	Stonington (Magothy)	**7.5**
	Public Ldg (MD Cstl bays)	**7.3**
	Mattawoman Ck (Upper Potomac)	**8.1**
Less Enriched	Betterton Beach (Sassafras)	**4.8**
	Rehobeth (Pocomoke)	**3.5**
	Piney Pt (Lower Potomac)	**5.0**

5-8 SAV in Mattawoman Creek

The resurgence of SAV in Mattawoman Creek is one of a limited number of success stories in the Chesapeake Bay region. It appears that substantial nutrient reductions from point sources (e.g., WWTPs) within Mattawoman Creek (and possibly the mainstem Potomac as well) initiated a cascade of events leading to water quality conditions supportive of SAV growth.

The spatially aggregated pattern of SAV occurrence in Mattawoman Creek from 1971-2010 is shown in Figure 5-14. From 1977 (and very likely before 1975) SAV were absent from the creek system. Beginning in 1989 SAV reappeared in a limited fashion and covered a small percentage of creek bottom (~5%) through 1997. After 1997 there was a very rapid increase in SAV coverage and these emerging beds were quite dense. Beginning in 2002 SAV beds covered about 40-50% of the surface area of the creek and have become an important component of this tributary system.

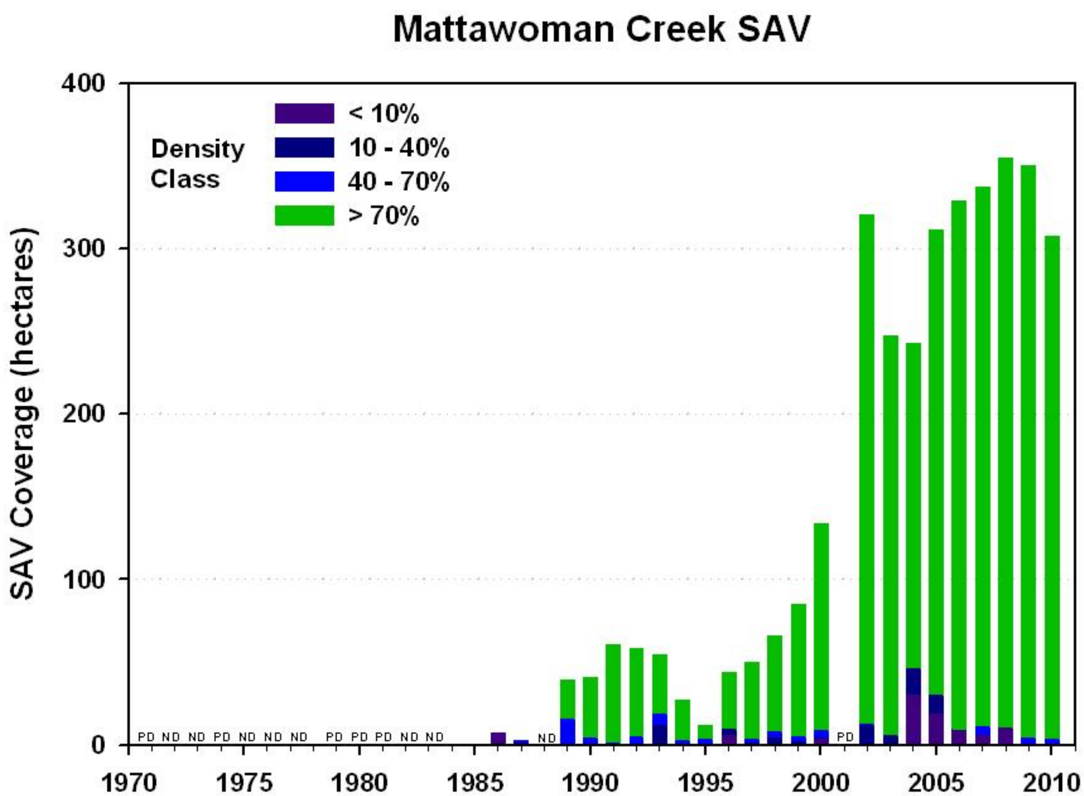

Figure 5-14. A summary of annual SAV community coverage (ha), with SAV density classes, for Mattawoman Creek for the period 1971-2010. ND = no data and PD = not fully mapped. Data were obtained from http://www.vims.edu/bio/sav.

The spatial pattern of SAV community recovery was also distinctive (Fig. 5-15). Beginning in 1996 SAV appeared in the upper portions of the creek and began to extend downstream through 2000. There was no aerial evaluation of SAV in Mattawoman Creek during 2001 but by 2002 SAV had spread along both the north and south shores of the creek all the way to the creek mouth. In more recent years (2005-2010) SAV appear to have extended to deeper water, again along both shores of the creek; during 2008-2010 the deeper channels of the creek appeared to be the only areas not colonized by SAV. This pattern of re-invasion in the upstream areas of the creek is similar to the pattern observed in other shallow, low salinity systems including Gunston Cove (a VA tributary of the Potomac) and the upper Patuxent River. It may be that these areas are re-colonized first because they are proximal to seed and other vegetative propagules surviving in small streams.

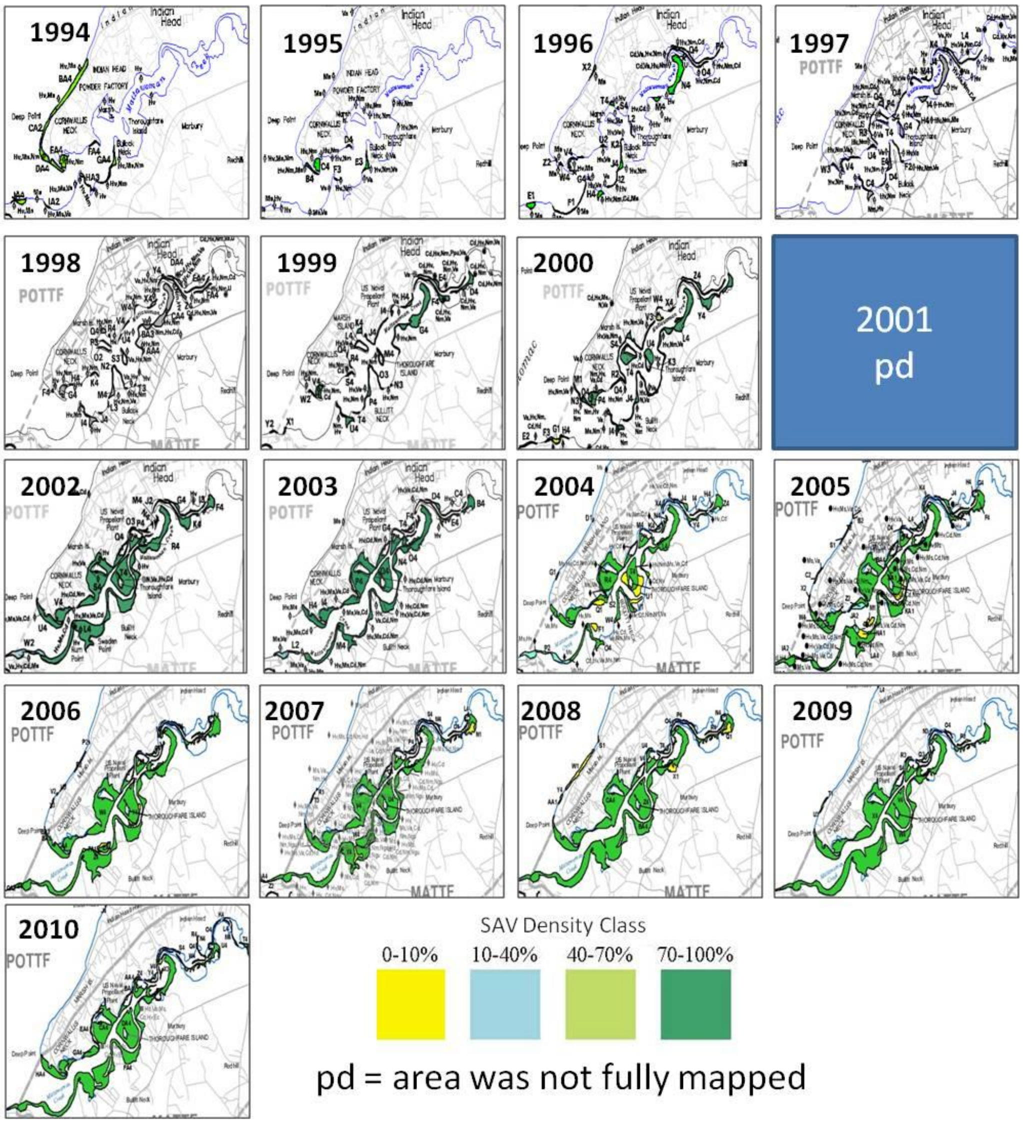

Figure 5-15. Annual maps of SAV spatial coverage (and SAV density estimates) for Mattawoman Creek from 1994-2010. Data were obtained from http://www.vims.edu/bio/sav.

The emerging understanding of SAV re-invasion seems to be related to a chain of cause-effect events and these appear to have occurred in Mattawoman Creek (Fig. 5-16). In general it seems like re-invasion follows nutrient input reductions. In some cases P seems to be the key element (Gunston Cove; C. Jones, pers. comm.) and re-invasion is preceded by a considerable lag period likely caused by the effects of excess P slowly purging from estuarine sediments. In other cases there appears to be minimal lag and N seems to be the key element (e.g., upper Patuxent; see

Chapter 7 in this report). Associated with nutrient input reduction there is a reduction in algal biomass (indexed as chlorophyll-*a*) and a corresponding increase in water clarity. In Mattawoman Creek there was a nutrient load reduction and this was followed by declines in chlorophyll-*a* concentration and increases in water clarity and then increases in SAV coverage.

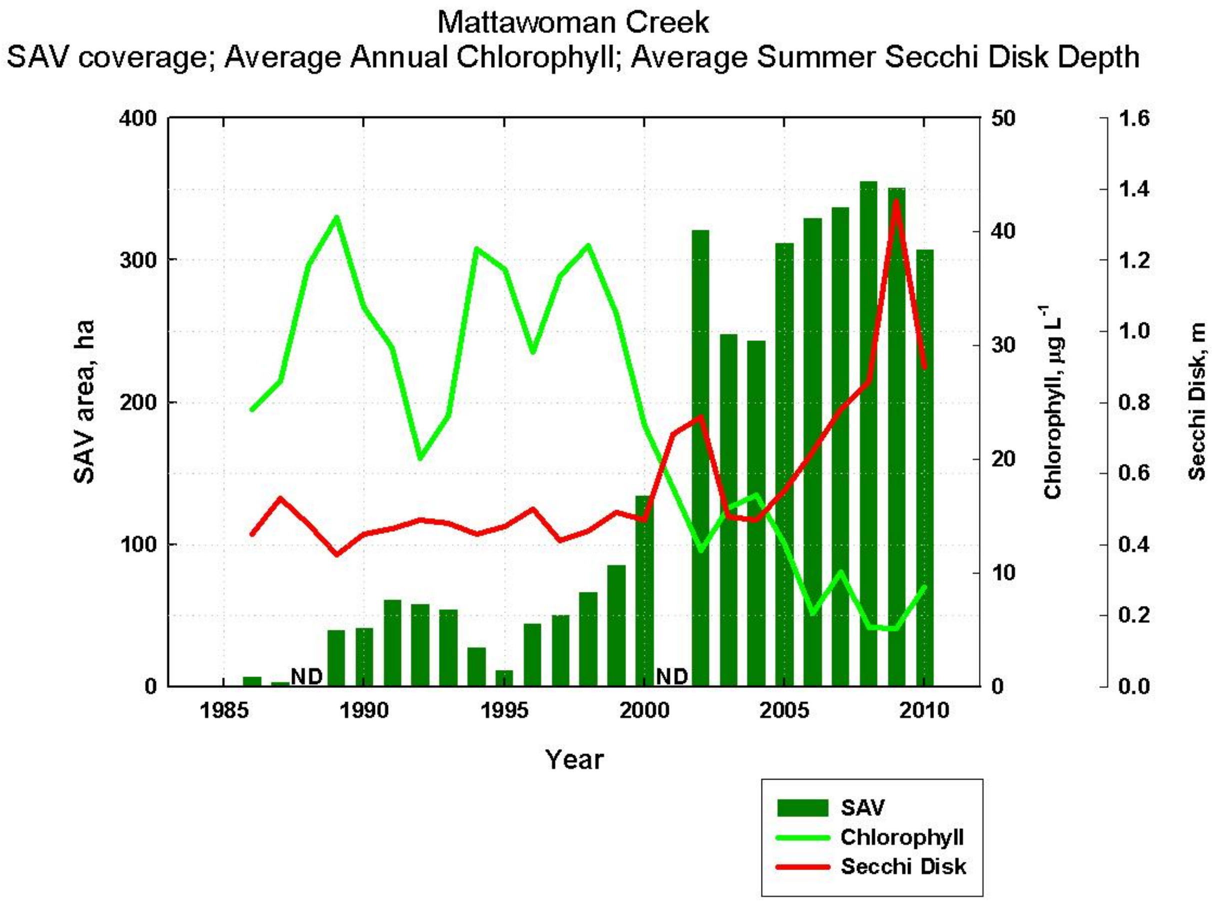

Figure 5-16. Annual summary of SAV coverage (ha), water clarity (indexed with Secchi Disk measurements) and algal biomass (indexed with chlorophyll-*a* concentration) for the period 1986-2010 in Mattawoman Creek. Note the large change in SAV coverage and water clarity associated with the large decline in algal biomass. All data sources have been previously described.

We have examined the Mattawoman data set for possible threshold responses relative to SAV re-invasion (Figs. 5-17a and b). The clearest of these appears to be related to water column chlorophyll-*a* concentration. When annual average chlorophyll-*a* concentration was in excess of about 18 µg L^{-1} SAV coverage was either close to zero or minimal. In contrast, when chlorophyll-*a* concentrations dropped below 18 µg L^{-1} SAV coverage expanded very quickly; below this chlorophyll-*a* "threshold value" some other factor or factors apparently regulate inter-annual variability in SAV coverage. There is also some indication of threshold behavior related to water clarity as indexed by Secchi Disk depth. In this case there was a very sharp increase in SAV coverage when Secchi Disk depths exceeded about 0.5 m (Fig. 5-17b). Ruhl and Rybicki (2010) reported a similar response in the adjacent tidal freshwater Potomac River although the "critical" Secchi Disk depth was slightly higher at 0.65 m. There now appear to be a number of cases in the Chesapeake system (in both small and large low salinity portions of the system) where nutrient load reductions have been followed by SAV re-invasion and rapid expansion. It still remains quantitatively uncertain what factors regulate lag times (when they occur) and under what conditions N or P load reductions might be the key element initiating the re-invasion process.

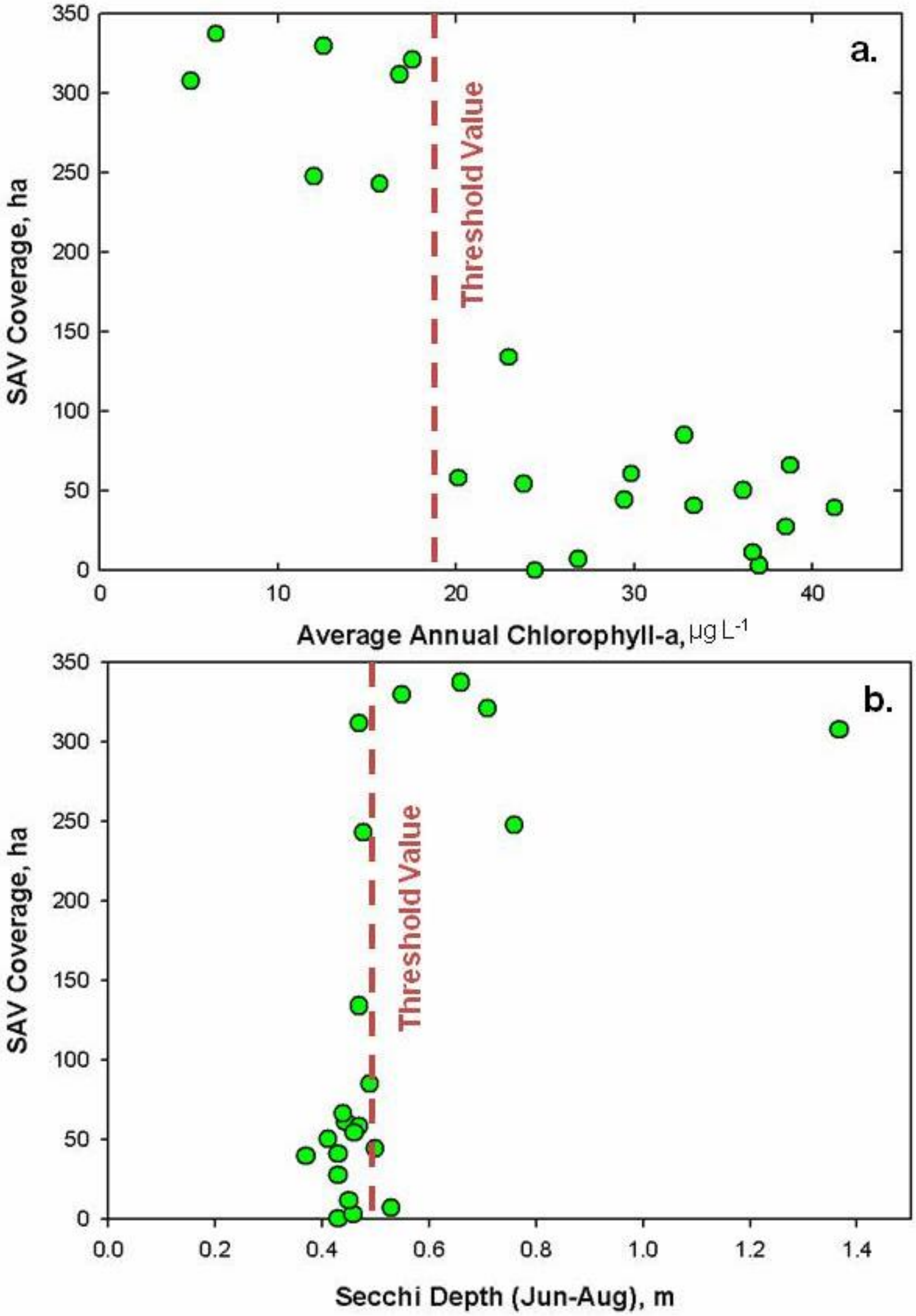

Figure 5-17. Scatter plots of average annual chlorophyll-*a* concentration versus SAV coverage (a) and summer average Secchi Disk depth versus SAV coverage (b) for Mattawoman Creek. Both plots seem to indicate a large change in SAV coverage associated with chlorophyll-*a* threshold of about 18 µg L^{-1} and Secchi Disk depth of about 0.5 m. All data sources have been previously noted.

5-9 Nutrient Budget for Mattawoman Creek

In order to develop a nutrient budget for Mattawoman Creek a relatively large and diverse data set needed to be assembled. Key components included nutrient inputs from all sources, including atmospheric deposition of N, internal nutrient losses and nutrient exchanges with the Potomac River. In the following sections each of these data sets are presented and explained and finally a mass balance N budget is developed. The key attraction of a mass balance is that it is a quantitative framework against which we can test our understanding of system-scale nutrient dynamics in an ecosystem. In other words, if the budget balances (within reason) we have reason to believe we have included and properly evaluated all of the important processes. However, if the budget does not balance then we know we have made an important error or neglected some critical process. Finally, reasonably balanced budgets allow us to separate large from small processes and this is an important step in choosing effective management actions.

5-9.1 Current and Historical Nutrient Sources

The USGS maintained a water quality and quantity gauge in the Mattawoman Watershed from 2005-2011 (no data collected during 2009). This gauge station monitored water, nutrient and sediment discharges from 59% of the basin land area. Unfortunately, activity at this site was discontinued after 2011 because of a lack of funding. Flow and N and P load patterns varied seasonally as well as inter-annually (Fig. 5-18). For example, during 4 of the 6 years of record, flow and loads were highest during winter-spring and much lower during summer and fall, a pattern typical of other tributaries of the Bay and of the main rivers entering the Bay. However, during 2006 and 2011 fall tropical storms passed through the area and flow and loads exhibited large but temporary increases (Table 5-6). On an inter-annual basis, N loads varied by almost a factor of two (180-343 Kg N day^{-1}) and P by just over a factor of two (23.5 to 49.7 Kg P day^{-1}).

Diffuse source loads were also estimated based on the Chesapeake Bay Program land use model (Fig. 5-19 and Table 5-6) and those estimates were very similar to those derived from the USGS stream monitoring data (G. Shenk, pers. comm.). A single year (2000) of loading data were provided by Maryland Department of Environment’s TMDL work on Mattawoman Creek and these values were intermediate between the 1985 values from the Bay model and more recent estimate that were made after WWTP modifications were completed. Based on the Bay Program model it does not appear that diffuse loads alone (not including point sources) have changed much between 1985 and the present time.

The major change in nutrient input to Mattawoman Creek is related to point source reductions (Fig. 5-19). During 1990 point source loads were about 950 kg N day^{-1} and were a much larger source than diffuse loads. Point source loads declined very sharply to about 50 Kg N day^{-1} by 1995 and then decreased again beginning in 2000. Point source loads have been very low since then and now represent a small fraction of total nutrient load to the system.

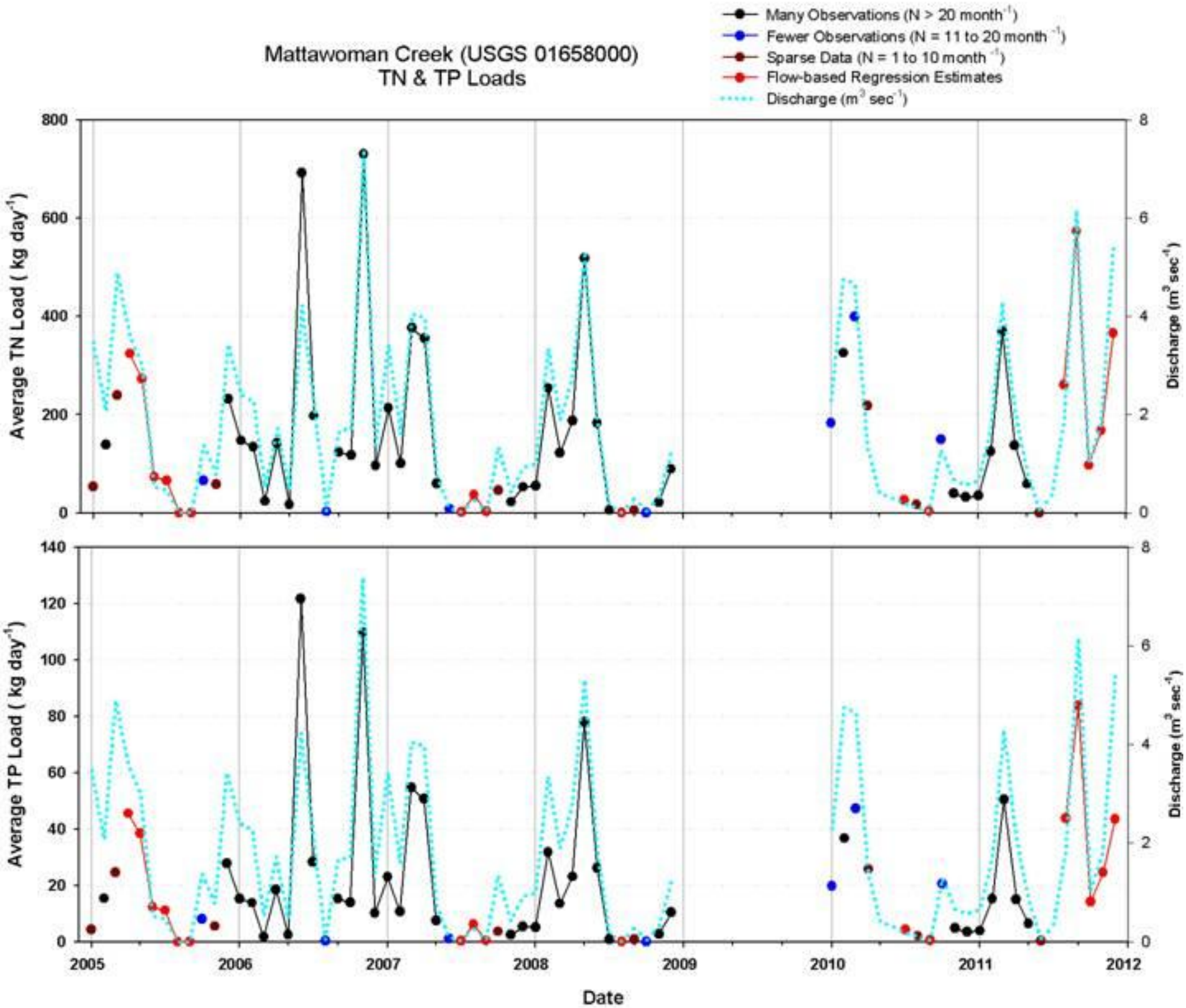

Figure 5-18. Plots of average monthly TN (a) and TP (b) loads (kg N or P per day) and freshwater discharge rate (m^3 sec^{-1}). Data were from the USGS stream gauge on Mattawoman Creek (USGS 016558000). This gauge was actively maintained from 2005-2011 and then discontinued. There were also gaps in the load record and these were estimated based on flow – load relationship developed with this data set. The amount of data available for monthly load estimates is also indicated in the figure. Data from http://waterdata.usgs.gov/nwis.

Table 5-6. Multiple estimates of annual average TN and TP loads to Mattawoman Creek. Both point and diffuse loads are included. Direct atmospheric deposition to surface water of the creek are included in the USGS estimates. Import of TN and TP from the Potomac is not included. USGS data for the gauged portion of the watershed were scaled up using a simple linear ratio of gauged to non-gauged areas to represent the full basin area.

Data Source / Condition	Year	Annual TN Load (Kg N day $^{-1}$)	Annual TP Load (Kg P day $^{-1}$)	Reference
Pristine forested basin	Pre-European settlement	100	2.7	Boynton *et al.* 1995
MDE TMDL Computations	2000	424	33.3	MDE TMDL 2004
CBP landscape	1985	591	39.2	CBP Landscape Model;
model estimates	2002	276	33.0	G. Shenk, pers. comm.
	2010	243	26.1	
USGA River Input Monitoring	2005	216	27.3	USGS River Input monitoring data
	2006	343	49.7	
	2007	180	23.5	
	2008	204	27.2	
	2009	no data	no data	
	2010	204	24.1	
	2011	310	42.6	

Direct deposition of N to the surface waters of Mattawoman Creek represents another nutrient source. We used atmospheric deposition data from Boynton *et al.* (2008) that included all forms of N in both wet and dry deposition (0.81 mg N L^{-1} as an annual average concentration). Given that precipitation averages about 1 m year $^{-1}$ direct atmospheric deposition contributes about 6000 kg N year^{-1} or about 16 kg N day^{-1} to the creek system. This estimate indicates that direct N deposition is a small component of the N budget for this system. However, rain (and dry deposition) falls on the full basin and all this rain contains N compounds. Regional assessments of nitrogen additions and losses from landscapes have become more common and some have focused on estimating the portion of N export from landscapes resulting from atmospheric deposition of N (Howarth *et al.* 1996). In the case of the Chesapeake Bay basin Fisher and Oppenheimer (1991) and more recently Castro *et al.* (2003) estimated that about 25% and 22%, respectively, of atmospheric N deposition to the landscape is exported to estuarine waters. No direct measurements are available for the Mattawoman basin. However, if the most recent estimate of 22% is applied to the Mattawoman basin, about 120 kg N day^{-1} would reach estuarine waters as a component of diffuse source loading, or about 49% of the total diffuse source load. In this larger view, atmospheric deposition is a very important part of the N input signature for this system. If this rough estimate proves to be generally correct, emphasis on decreasing atmospheric deposition of N is an important management objective.

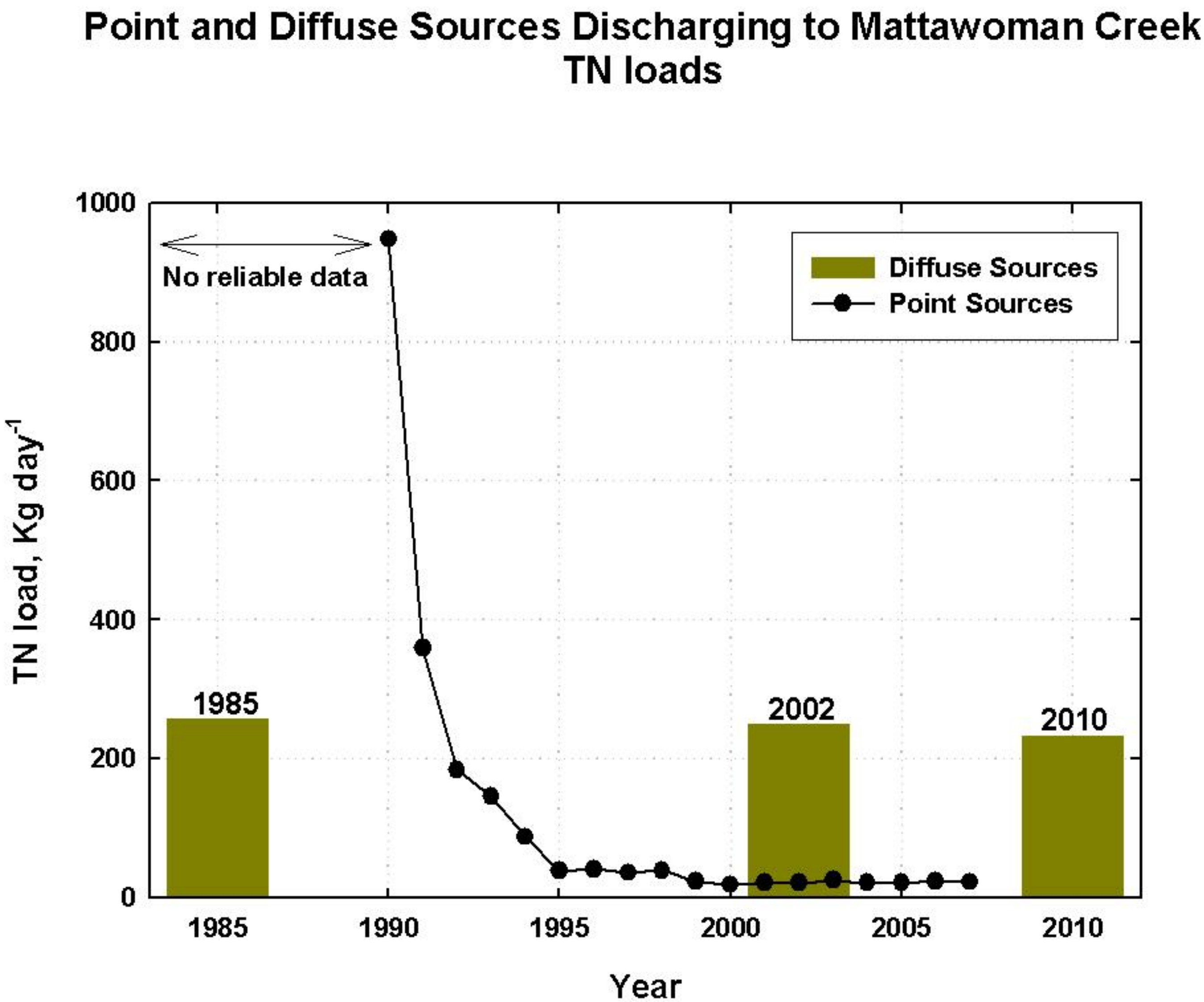

Figure 5-19. Plots of annual point and diffuse TN loads to Mattawoman Creek. Point source loads were from the CBP database:
http://www.chesapeakebay.net/data/downloads/bay_program_nutrient_point_source_database
Diffuse source loads were estimated from the Chesapeake Bay Program land use model (G. Shenk, pers. comm.).

5-9.2 Internal Nutrient Sinks

In nutrient budget evaluations such as this one for Mattawoman Creek we must account for internal losses of N. Generally, the two major losses include net denitrification (i.e., the net difference between N-fixation and denitrification) and long-term burial of N (mainly particulate organic N) in the accreting sediments of the estuary. Fisheries extractions (landings) and fish migrations are considered in some budgets but were not included here because there are no site-specific data available (generally, but not always, these are small terms in the budgets). Unfortunately, there are no direct measurements of either of the major internal loss rates available for Mattawoman Creek. However, during the last decade there have been an increasing number of these measurements made in shallow estuarine systems and many of these measurements have been summarized by Greene (2005a and 2005b). To make preliminary estimates of net denitrification and long-term N burial we used annual average rates of 47 µmoles N m^{-2} hr^{-1} and 6.0 g N m^{-2} yr^{-1}, respectively. These rates represent substantial fractions of the N inputs to Mattawoman Creek and the implications of this are described in the Nutrient Synthesis section of this report.

5-9.3 Exchanges with Potomac River

The final, and most difficult, component of this mass balance evaluation concerns N exchanges with the adjacent Potomac River estuary. These systems are connected via tidal water transport between the creek and Potomac as well as freshwater flows from the basin. These processes vary on many time scales (hourly to inter-annual) and are also influenced by local and regional storm events. In addition to water transport, nutrient exchanges between the Potomac and Mattawoman Creek are also governed by nutrient concentration differences in these two systems. In many cases, the more seaward system (the Potomac in this case) has lower nutrient concentrations than the landward system so the nutrient concentration gradient favors export from the landward to seaward system. That is not generally the case for these systems. In several previous case studies (e.g., Corsica River and Back River; Boynton *et al.* 2009; Boynton *et al.* 2011) we used relatively simple salt and water box model results coupled to nutrient concentrations to estimate net nutrient flux into or out of these small estuarine systems. However, there is rarely any measurable salinity in Mattawoman Creek so that approach is not possible (i.e., the site lacks a conservative tracer….salt).

To have an estimate of Potomac – Mattawoman nutrient interaction we obtained output from the Chesapeake Bay Program water quality model wherein the Bay Program modeling staff computed the net N flux across the mouth of Mattawoman Creek (average monthly flux) for the period 1991-2000. These dates do not coincide with the more recent data primarily used in our current effort but we used these data as the best quantitative estimate currently available. The model results indicate some NO_{23} net transport from Mattawoman Creek to the Potomac during winter or spring and the opposite during summer-fall. Averaged over all years the net NO_{23} flux was about 102 Kg N day^{-1} directed into Mattawoman Creek from the Potomac River. We also had estimates of TN (Total Nitrogen) flux and in this case the annual average for the period 1991-2000 was very small (0.4 Kg N day^{-1}) and this was directed out of Mattawoman Creek to the Potomac River. This result suggests that Mattawoman Creek consumes NO_{23} (and perhaps other forms of DIN) but exports almost no TN…the creek system thus acts as a nitrogen sink. While it was useful to have the model results, we also wanted to compare the direction of net N flux (into or out of the creek based on model results) with the N concentration gradient computed from NO_{23} data collected by the monitoring program inside the creek and in the adjacent Potomac River. If the N concentration in the creek was higher than the in the Potomac we would expect export of N and the opposite if Potomac N concentrations were higher. Concentration gradients agreed with model results during 88 of the 120 monthly comparisons (73% agreement). The main periods of disagreement were during the winter-spring period when the model indicated N export from the creek but concentration gradients indicated import of N to the creeks. Computing net flux of anything at the mouth of an estuary is difficult as has been noted by many investigators (e.g., Boynton *et al.* 1995). At this point it seems like Mattawoman Creek imports NO_{23} from the Potomac at modest rates but exports very little TN. If water quality in the Potomac continues to improve we would suggest that NO_{23} import to Mattawoman Creek would decrease and this would have a positive effect on creek water quality.

5-9.4 Nutrient Synthesis

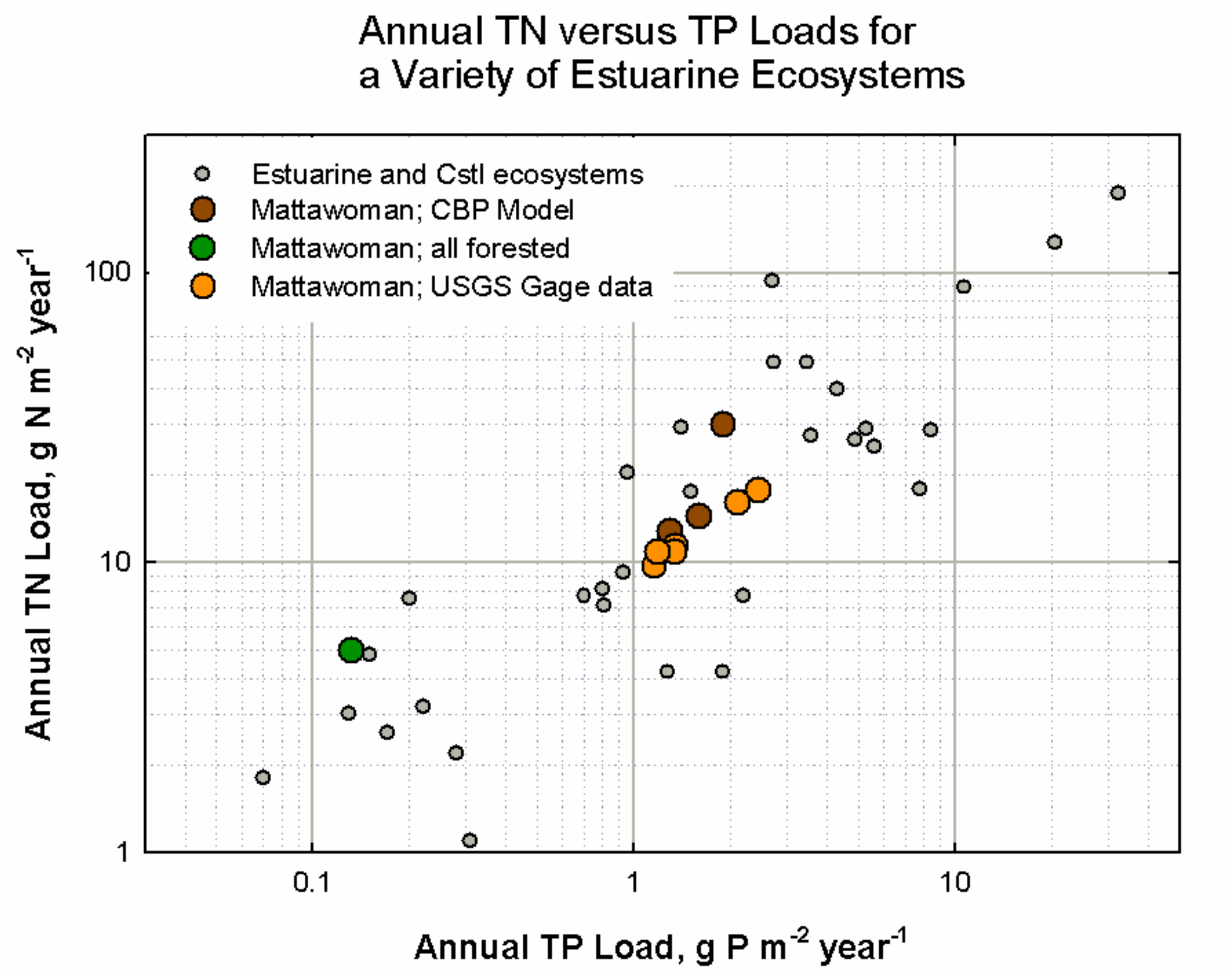

Figure 5-20. Scatter plot of annual TP versus TN loads for a variety of estuarine and coastal marine ecosystem (small gray circles; see Boynton *et al.* 1995 for references for these sites). TP and TN loads for Mattawoman Creek based on the land use model (1985, 2002 and 2010) are shown as dark brown circles. Loads based on USGS gauge data are shown as light brown circles (2005-2011). The green circle represents an estimate of TN and TP loads from a fully forested basin with no atmospheric deposition of N or P.

One useful metric to consider in case studies such as this one is to compare the N and P loading rates of Mattawoman Creek with those of other estuarine ecosystems. In effect we ask here …where does this system sit relative to many other generally similar ecosystems? We have compiled N and P loading rates for many such systems and added Mattawoman Creek data for several time periods and from several data sources to this analysis (Fig. 5-20). Several points are immediately clear and include the following: 1) N and P loads prior to WWTP modifications were much higher than at present but not, even then, as high as they are in very heavily loaded systems; 2) there was a significant decline in N and P loading rates associated with WWTP modifications beginning in the early 1990s; 3) N and P loads still exhibit considerable inter-annual variability related to wet and dry years; 4) loading rate data from gauges and from models agreed quite well; 5) loading rates for the completely forested watershed (with no atmospheric deposition) were about half what they are now during dry years.

One of the major emerging features of Mattawoman Creek is the expansion, and persistence of vast SAV communities covering about half the surface area of the tidal creek. A great deal has been learned and speculated concerning relationships between nutrient loading rates and SAV community health (e.g., Kemp *et al.* 2005). In general, it is thought that SAV communities are not competitive in environments having large nutrient loads. In fact, Orth *et al.* (2010) have shown that SAV resurgence in several areas of Chesapeake Bay was related to decreased N loading. In Mattawoman Creek SAV were largely absent when N loading rates were in the range of 30 g N m^{-2} yr^{-1}. When loading rates decreased to about 10 g N m^{-2} yr^{-1} SAV re-invaded the creek system. So, the Mattawoman Creek example appears to be generally consistent with other examples in Chesapeake Bay. In addition, Latimer and Rego (2010) examined many SAV communities in southern new England for relationships to N loading rates and found SAV to be healthy when loading rates were about 5 g N m^{-2} yr^{-1}, less robust when loading rates were about 10 g N m^{-2} yr^{-1}, and generally absent when loads exceeded this amount. Mattawoman Creek loads are in the upper portion of the range of “SAV-friendly” loads reported by Latimer and Rego (2010).

We have described above all the major inputs, exports and internal loss terms for the creek system and we now need to put these together as a synthesis of the budget pieces. One test for the veracity and consistency of this analysis is to examine the overall balance between inputs and outputs of TN mass. Nutrient budgets in the context of estuarine research are an application of the conservation of mass laws. The constraint that mass must be conserved is the foundation of budget analysis. Budgets that balance (inputs = outputs) lead us to believe we have a solid understanding of the system while those that do not balance by a wide margin send us back to the drawing boards because we have missed something important. A nitrogen budget for Mattawoman Creek is provided in Figure 5-21 where the red circles indicate various N inputs to the ecosystem and the red arrows represent nitrogen losses from the system. This budget assumes (1) steady state, (2) completeness (i.e., there are no missing terms in the budget), and (3) internal storages that are unaccounted for are not substantially changing from year to year. There are several key issues that are resolved by inspecting this synthesis diagram.

First, the budget does not currently balance. TN inputs (385 kg N day^{-1}; including DIN inputs from the Potomac) are considerably larger than estimated nitrogen losses (239 kg N day^{-1}) indicating that one or more major processes have not been adequately considered. One likely explanation for this is that we were not able to assign specific denitrification or nutrient burial rates to either the SAV or fringing tidal wetland communities. Direct measurements of these rates in tidal freshwater marshes of the Corsica River yielded rates three times the rates measured in open waters of the Corsica. If we adjusted Mattawoman internal loss rates so that

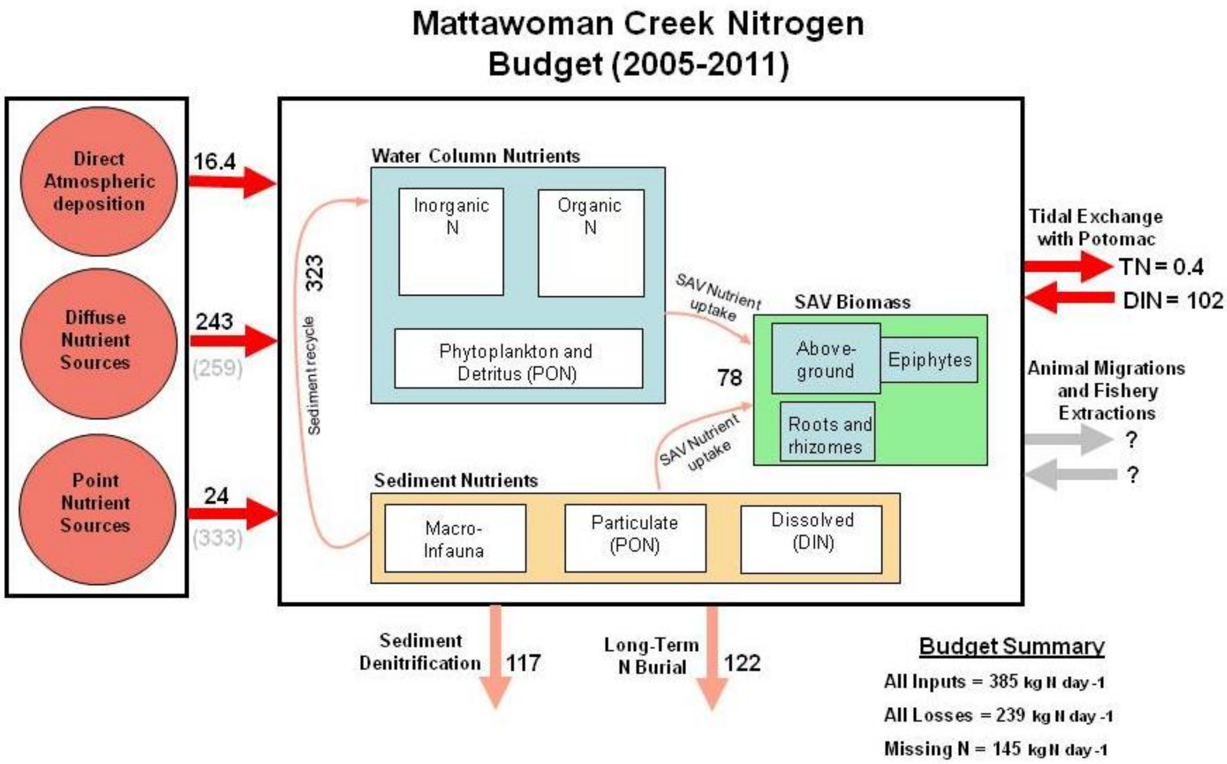

Figure 5-21. A schematic diagram of a nutrient budget (TN) model developed for Mattawoman Creek. Nutrient sources are shown on the right (from point, diffuse and atmospheric sources) and left (exchange with the Potomac River). Internal loss terms are shown on the bottom of the diagram. Several internal nutrient re-cycling processes are also indicated. Internal stocks were not evaluated because data were not available and such data are not essential for construction of a mass balance model. Bright arrows indicate data specific to Mattawoman Creek were used; light red arrows indicate data from the Chesapeake Bay region were used; gray arrows indicate no data were available and an estimate was not attempted. The light gray numbers in parentheses indicate diffuse and point source N loads prior to WWTP modifications.

losses were higher in SAV and fringing tidal marsh communities the budget comes much closer to balancing. It may well be worth supporting a measurement program to better quantify N losses in these communities; the CBP water quality modeling group is considering adding such a component to the Bay model (C. Cerco, pers. comm.; Boynton *et al.* 2008). The second point is that diffuse sources are the most important nitrogen source. Efforts to further improve water quality will likely fail unless this term is considered and acted on; if this term increases because of changes in land use water quality will likely degrade. Third, the two major nitrogen loss terms are both important. However, both of these were estimated based on literature values. If an improved budget is to be developed, these rates need to be measured locally and special attention needs to be directed at measuring N losses in the SAV and fringing wetland communities because rates in these areas are often very high (Boynton *et al.* 2008). Fourth, the TN export/import term associated with exchanges with the Potomac River needs more examination. At present, model results indicate almost no net exchange of TN between the Potomac and Mattawoman Creek but also indicate a substantial input of DIN, almost all as NO_{23}, into the creek from the Potomac. This suggests that the creek acts as an N sink for the Potomac. This needs to be confirmed. During most of the year NO_{23} concentration in the Potomac were higher than in the creek so the direction of net transport was largely consistent with model results.

However, there is reason for concern here. Should nutrient concentrations in the Potomac increase further, or if remediation (and large SAV communities) further reduces N concentrations in Mattawoman Creek, the magnitude of DIN import to the creek could increase; the creek could become more nutrient enriched. DIN flux from large to smaller systems has already been documented for the Patuxent and Corsica estuaries in some summer and fall months (Boynton *et al.* 2008). Also, we should note that model results were only available for an earlier period of time (1990-2000) and thus do not temporally coincide with more recent conditions in Mattawoman Creek. Finally, we were able to add a few internal nutrient-cycling terms to the budget analysis. Uptake of N from sediments and the water column by SAV serves as a seasonal-scale (i.e., SAV growing season; Apr-Oct) nutrient loss term as N is incorporated into plant tissue. We estimated this rate by using data from the VIMS aerial SAV survey (http://web.vims.edu/bio/sav/), SAV biomass papers by Moore *et al.* (2000) and estimates of the % N content of SAV (Moore pers. comm., Abbasi *et al.* 1990, Yu *et al*. 2010, Mukherjee *et al.* 2008, Zimba *et al.* 1993, Spencer *et al*. 1992, Best *et al*. 2001, Shields *et al*. (in press), Shields 2008, Rybicki *et al.* 2001, Mony *et al.* 2007, Spencer and Ksander 1999, Dorenbosch and Bakker 2011, Grance and Wetzel 1978, Madsen 1997, Titus and Adams 1979 and Barko 1983). The results indicate a modest seasonal-scale buffering of nutrients by the SAV community. It is likely that SAV nutrient buffering via enhanced denitrification and burial of PON, as indicated above, is considerably greater than the estimate we were able to generate with available data. We also examined sediment flux data from many small Chesapeake Bay tributaries and from these data estimated sediment NH_4 releases in Mattawoman Creek. These are substantial…in fact the largest single term in the budget. This result has also been repeatedly observed in other systems (Boynton *et al.* 1996; Boynton and Kemp 2008) and indicates the importance of sediment nutrient sources in shallow systems.

At this point the message from the preliminary nutrient budget seems clear. Maintenance of the currently successful restoration depends on maintaining the sharp reductions in point sources that have been achieved. Additional water quality and habitat improvements will need to focus on reductions of diffuse source nutrient inputs and better estimation and understanding of the nutrient inputs from the Potomac River estuary.

5-10 Ecosystem Linkages: Nutrient Cause-Effect Chain for Mattawoman Creek

5-10.1 Nutrient Loads and Algal Biomass

In many estuarine ecosystems, excessive nutrient loading is the primary cause of rapid algal growth and biomass accumulation (algal blooms measured as chlorophyll-*a* concentration) and that seems to be the case in Mattawoman Creek. This relationship between nutrient loads from all sources and algal responses (in terms of chlorophyll-*a* concentration) is a critical starting point for the analyses that follow. Essentially, a simple cause-effect chain is developed, where the nutrient loading from drainage basins is linked to estuarine chlorophyll-*a*, which is subsequently linked to summer water clarity and hypoxia. These linkages of key water quality issues to nutrient loads and chlorophyll-*a* will allow for estimates of the likely magnitude of estuarine responses to nutrient load reductions or future nutrient load increases. In developing these relationship data from several shallow estuarine systems were used (a comparative analysis approach) to increase the signal to noise (i.e., normal estuarine variability) ratio.

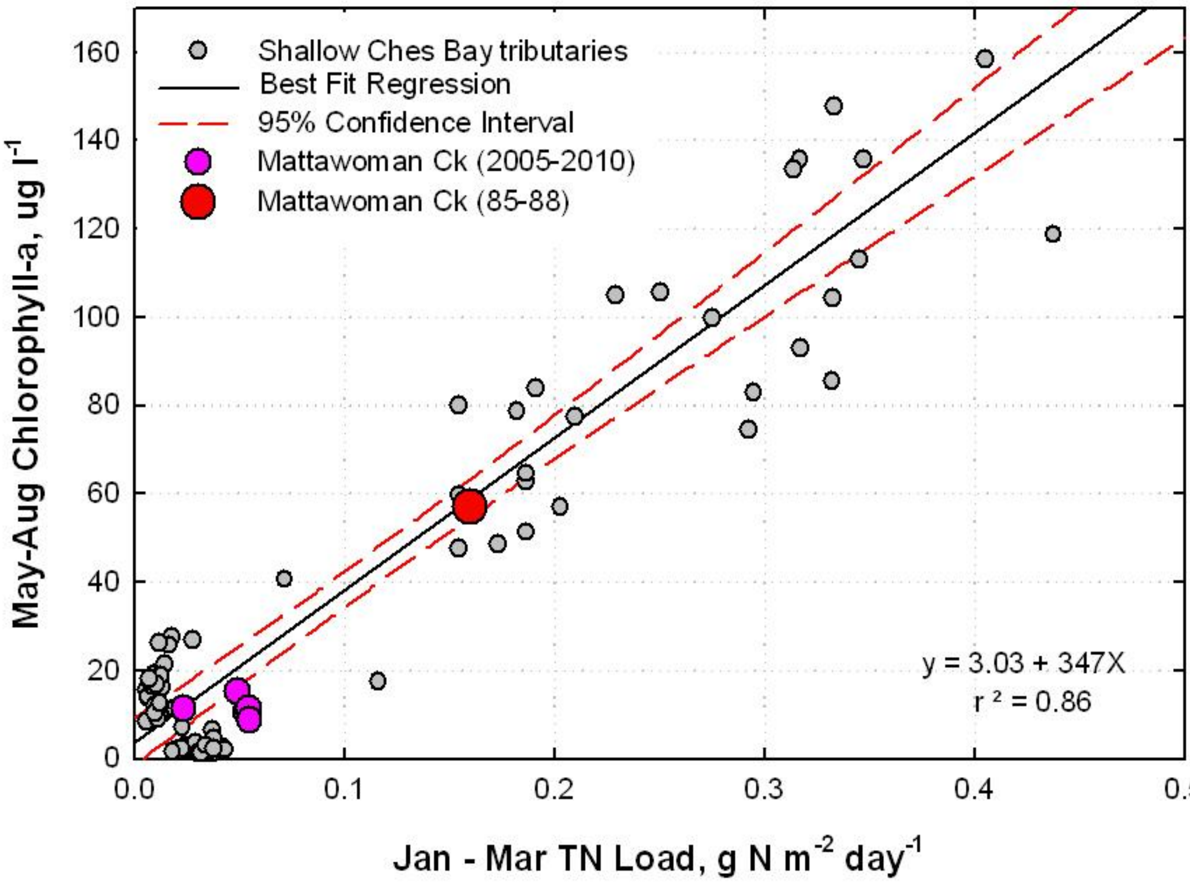

Figure 5-22. A scatter plot of TN load versus chlorophyll-*a* concentration developed for Mattawoman Creek and other shallow Chesapeake Bay tributaries. The large decrease in nitrogen loading was accompanied by a similar and large reduction in chlorophyll-*a* concentration. Data for the other Chesapeake bay systems was from Boynton *et al.* (2009).

Intensive (ConMon) and repeated (Biomonitoring Program) measurements of chlorophyll-*a* distributions at several locations in Mattawoman Creek indicated elevated summer concentrations. Winter-spring algal blooms may deposit labile organic material onto sediments which are not decomposed until early-to-mid summer when elevated temperature stimulates bacterial activity. Respiration of such material releases nutrients to the water column during summer and these nutrients, in addition to spring nutrient inputs, help stimulate the summer blooms in the creek. The connection of winter-spring nutrient loads to summer blooms is well described in Chesapeake Bay and its tributaries and is also reflected in data for several shallow estuaries connected or adjacent to Chesapeake Bay region. In a multi-system comparison of shallow, eutrophic estuaries in the region of Chesapeake Bay, spring N loading and summer chlorophyll-*a* were found to be highly correlated, and available data for Mattawoman Creek fit the general pattern (Fig. 5-22). Correlations between winter-spring nutrient load and summer chlorophyll-*a* appears to be linear and indicates the potential for large changes in chlorophyll-*a* in response to nitrogen load changes. Several annual observations were available for the creek system including one set of observations from the 1985-1988 periods when nutrient loading rates were much higher and a set of more recent observations (2005-2010) collected when nutrient loading rates were much lower. Both data sets conformed to the general relationship. Using comparative data a strong relationship was observed between nitrogen loading rates (winter-spring period) and summer algal bloom intensity. A factor of 5 reduction in nutrient loads resulted in somewhat more than a factor of 5 reduction in chlorophyll-*a* concentration. This suggests this system is quite responsive to nutrient load rate changes.

5-10.2 Algal Biomass and Water Clarity

Water clarity determines how much sunlight penetrates through the water and is available for photosynthesis by phytoplankton in the water column and by SAV and benthic algae growing in the sediments. Water clarity is typically reduced in estuaries when the concentration of algae, sediments, and other particles increases in the water column and that was clearly the case in Mattawoman Creek during earlier years.

A Secchi Disk was used to measure water clarity at several long-term monitoring sites and the site in the lower estuary was used in the following analysis because it represented much of the open water portion of the system. These measurements revealed distinct patterns in water clarity, the main ones being that Secchi Disk depths varied seasonally during any one year and water clarity has improved since 2000 (Fig. 5-8). Using Secchi Disk information, it is possible to estimate the water depth to which 1% of surface light penetrates. This is of significance because 1% of surface light is the minimum light level needed to support growth of benthic micro-algae (e.g., benthic diatoms). Growth of these algae on the sediment surface can reduce nutrient flux from sediments (good for water quality) to the water column and also suppress sediment resuspension (good for water transparency). Using the mean depths of the estuary along its axis and the depth of 1% light penetration, it is clear that prior to 2000, 1% light level occurred at depths of about 1.1m; much of the creek is deeper than that. In more recent years Secchi Disk depths have increased (~1.1 m during 2009) and the 1% light depth increased to 3 m, greater than the average depth of the creek.

Correlations between Secchi Disk depth and both chlorophyll-*a* and total suspended solids (TSS) suggest that both contribute to light attenuation in the creek, but chlorophyll-*a* exerts a stronger control on water clarity and that reductions in chlorophyll-*a* via nutrient load reductions would result in increased water clarity in the Corsica estuary. To continue with examination of the cause-effect chain described earlier, chlorophyll-*a* and Secchi Disk data from Mattawoman Creek and from several other small tributary rivers were combined and a strong relationship was observed (Fig. 5-23). As indicated earlier, SAV were generally absent from this system when Secchi Disk depths were less than 0.5 m or when chlorophyll-*a* concentrations were greater than about 18 µg L^{-1}. In recent years Secchi Disk depths were >0.6 m and chlorophyll-*a* concentrations less than 10 µg L^{-1}; SAV communities in recent years have been dense and expansive.

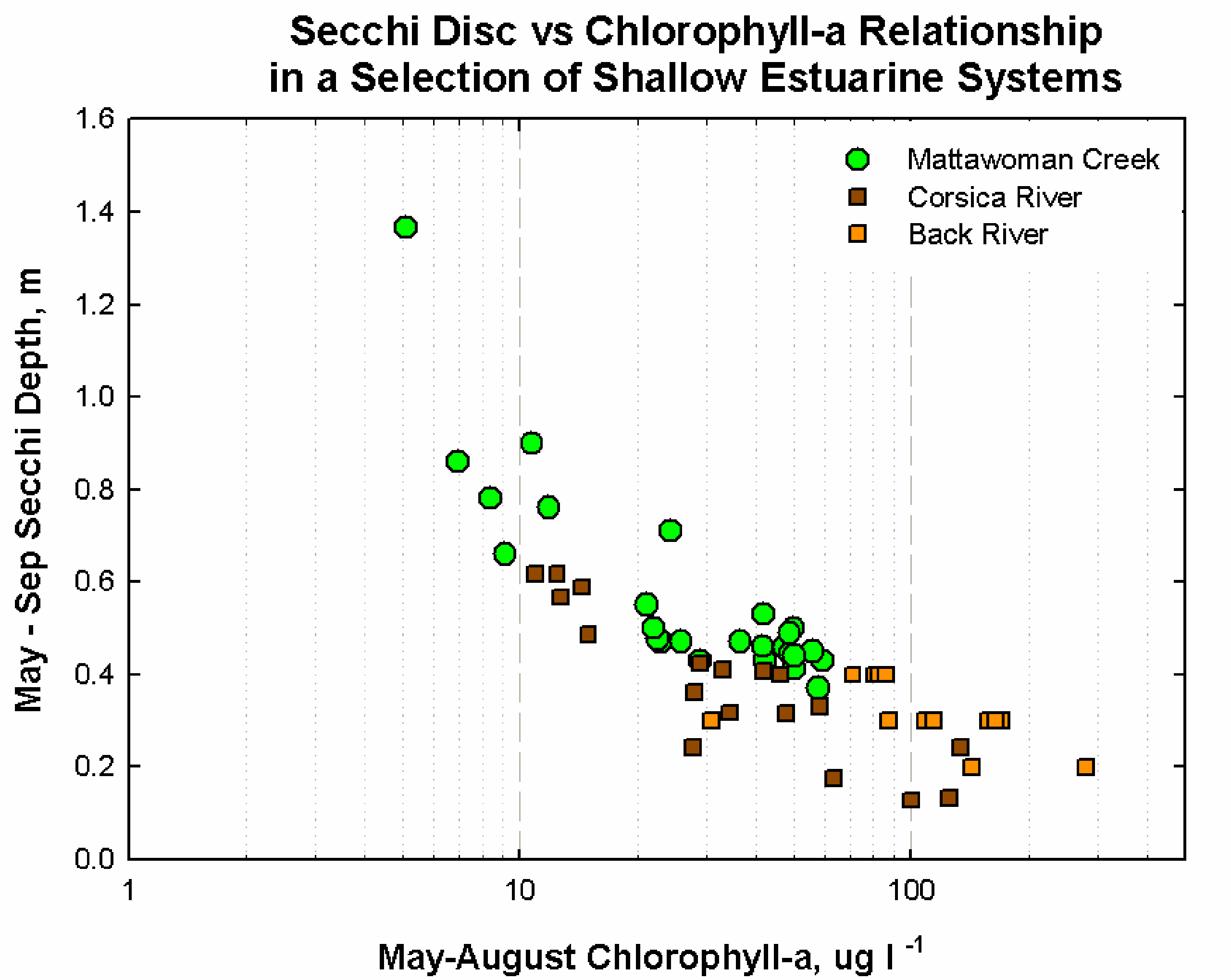

Figure 5-23. A scatter plot of chlorophyll-*a* versus Secchi Disk depth developed for Mattawoman Creek and two other shallow Chesapeake Bay systems. Data for the other shallow systems were from Boynton *et al.* (2009).

5-10.3 Algal Biomass and Hypoxia

The final link between nutrient loads and water quality involves establishing significant, quantitative relationships for phytoplankton chlorophyll-*a* and hypoxia (low dissolved oxygen conditions) which are harmful to animal life. There were a large number of oxygen measurements made since 2004 at one primary location in the creek using a multi-probe sensor system that recorded dissolved oxygen and other variables every 15 minutes between April and October of each year. Thus, at this site approximately 21,000 O_2 measurements were collected each year (2004-2010). In many shallow estuarine systems large phytoplankton blooms occurring during summer cause large dawn to dusk changes (10-20 mg L^{-1}) in O_2 during a single day. Such large daily swings in O_2 are caused by high rates of algal photosynthetic activity during the day (fueled by high nutrients), followed by high respiration rates (fueled mostly by simple carbohydrates produced by the day's photosynthesis) during the night. Night time respiration is often high enough in some small estuaries to cause hypoxia ($O_2 < 3.2$ mg L^{-1}) in the entire water column. For example, Corsica River data were analyzed to record all occurrences of hypoxia and the most frequent and intense hypoxic events occurred at Sycamore Point in the upper estuary and were clearly related to chlorophyll-*a* concentrations (Boynton *et al.* 2008).

We developed a similar analysis relating chlorophyll-*a* concentration to hypoxia using data from both Mattawoman Creek and the Corsica River (Fig. 5-24). In this case Mattawoman Creek data generally conformed to data from the Corsica River with increasing cumulative hours of hypoxia associated with high algal biomass. In other words, summer chlorophyll-*a* concentration was

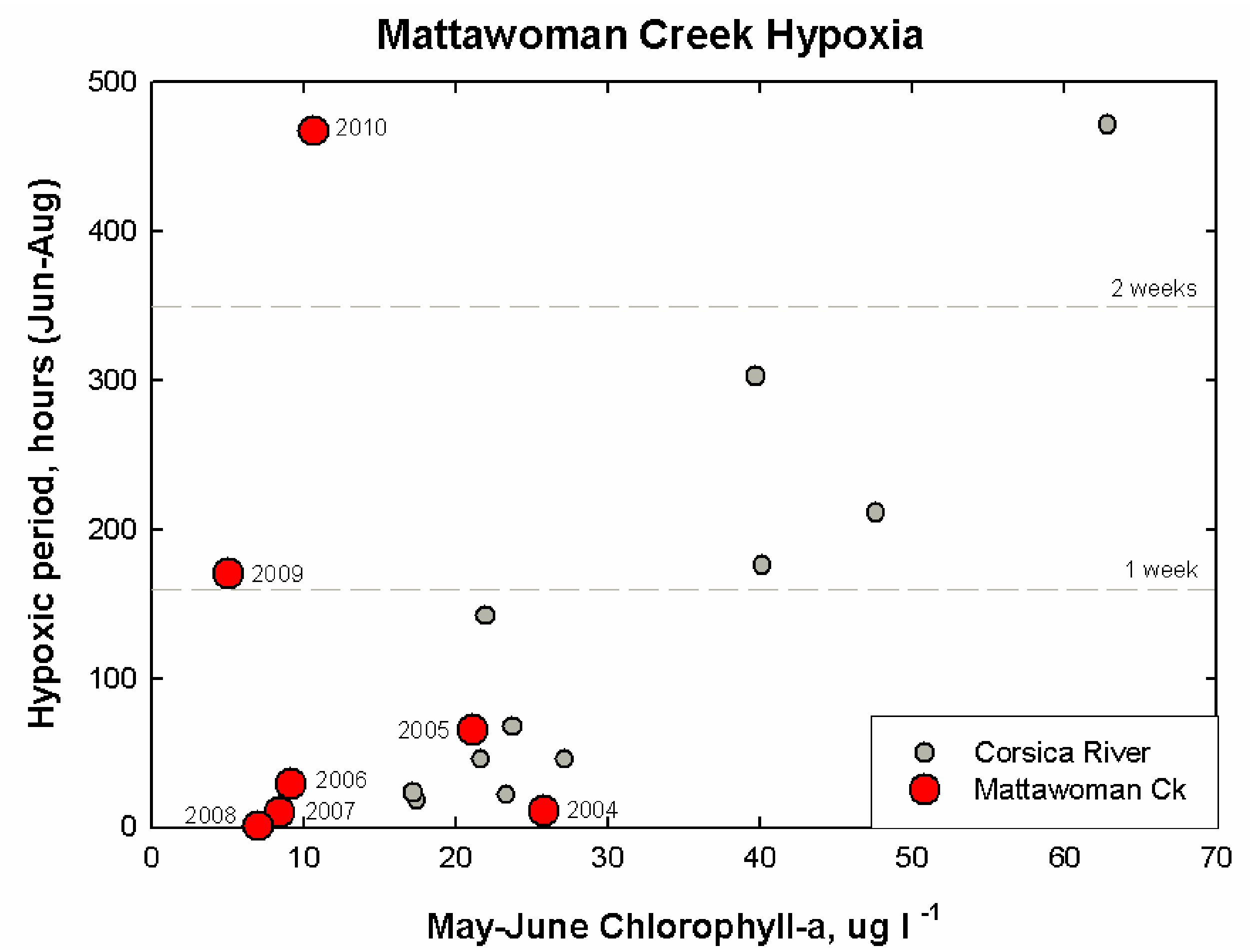

Figure. 5-24. A scatter plot of chlorophyll-*a* versus hypoxic period (DO < 3.2 mg L^{-1}) developed for Mattawoman Creek and one other shallow Chesapeake Bay systems. Data for the other shallow system was from Boynton *et al.* (2009).

highly correlated with summer cumulative duration of hypoxia suggesting a strong link between nutrient loading, algal biomass accumulation, and hypoxia. Consequently, it is anticipated that further reductions in nutrient loads will decrease (or increase if loads increase) the cumulative duration of hypoxic events in these shallow estuaries if summer chlorophyll-*a* concentrations continue to decrease. We suspect that hypoxia was more prevalent in the creek in earlier years when chlorophyll-*a* concentration was much higher. However, the high frequency DO data needed for this analysis were not collected prior to 2004.

It is also clear in Figure 5-24 that not all data from Mattawoman Creek conformed to this relationship. Specifically, hypoxic hours increased during 2009 and further increased during 2010; these increases were not associated with substantially increased chlorophyll-*a* concentration in either year. We indicated earlier that this departure from an expected pattern may have been the result of SAV bed location and bed effects on very local water circulation patterns. We do not know if this is true and so this explanation must be regarded as a hypothesis. It would be helpful if field notes were available in the future to assist in understanding this departure from an expected pattern. However, an additional concern was do these hypoxic measurements at the ConMon site represent a declining DO condition covering the more open waters of Mattawoman Creek. Bottom water DO measurements (2 measurements per month) during July-September, 2004-2011 at 4 open water sites in the lower portion of the creek (DNR Trawl sites 1-4) do not indicate worsening DO conditions. To the contrary, bottom water DO concentration remained in excess of 5 mg $L^{-1}$and was often higher during summer periods. Thus, it does not seem like a summer season creek-wide DO depletion pattern is developing (M. McGinty, pers. comm.).

5-11 Future Expectations

Given the patterns that emerged from this cause-effect chain analysis, can we use these data to anticipate future water quality and habitat conditions in the creek? What will likely happen if nutrient load reductions continue or if loads increase due to changes in basin land-uses or water quality conditions in the Potomac River? We have attempted a synthesis of the cause-effect chain analysis towards this end and summarized results in graphical form in Figure 25. At the start a word of caution is needed. The analyses we have conducted are of a statistical nature. Given that form of analysis it is not prudent to project outcomes far beyond observed values used in the original statistical analysis. Furthermore, there may well be some ecosystem-level interactions that develop that can either promote restoration or degradation beyond those expected from the type of analysis used here (e.g., Kemp and Goldman 2008).

Due to the nature of the relationships and the constraints on potential load reductions, our predictions include both positive (improved water quality) and negative (degraded water quality) effects (Fig. 5-25). Both effects appear to be relatively linear responses in Mattawoman Creek. Given the range of nutrient loading and chlorophyll-*a* observed during the period pre-2002-2010, we found a factor of three reduction in nutrient loading (from about 30 to 10 g N m^{-2} $year^{-1}$) and this load reduction yielded a factor of 4 reduction in summer chlorophyll-*a*. Such a reduction in chlorophyll-*a* was associated with about a doubling in water clarity (Secchi Disk Depth), which would lead to an increase in the area of creek sediments that could support benthic algae and SAV habitat (where light at the sediment surface exceeds 15% of that at the water surface). This analysis suggests the creek is very responsive to nutrient load changes (both increases and decreases) relative to chlorophyll-*a* l conditions.

If we consider the historic distribution of SAV in the creek we see that SAV were generally absent from 1975-1989. SAV represented a small component of the system during the period 1990-2000 when point source nutrient reductions first began but it is important to note that initial SAV resurgence appeared to be coupled with nutrient load reductions. Point source load reductions were largely complete by 1995 and SAV began expanding with high levels of SAV coverage evident by 2002. Our analysis suggests further SAV expansion should nutrient loading rates decrease beyond present levels but the degree of further expansion would likely be space limited. In this analysis healthy SAV communities were not associated with nutrient loading rates in excess of about 10 g N m^{-2} $year^{-1}$. Should loads increase from either the Mattawoman watershed or from the Potomac River we would expect contraction of SAV coverage.

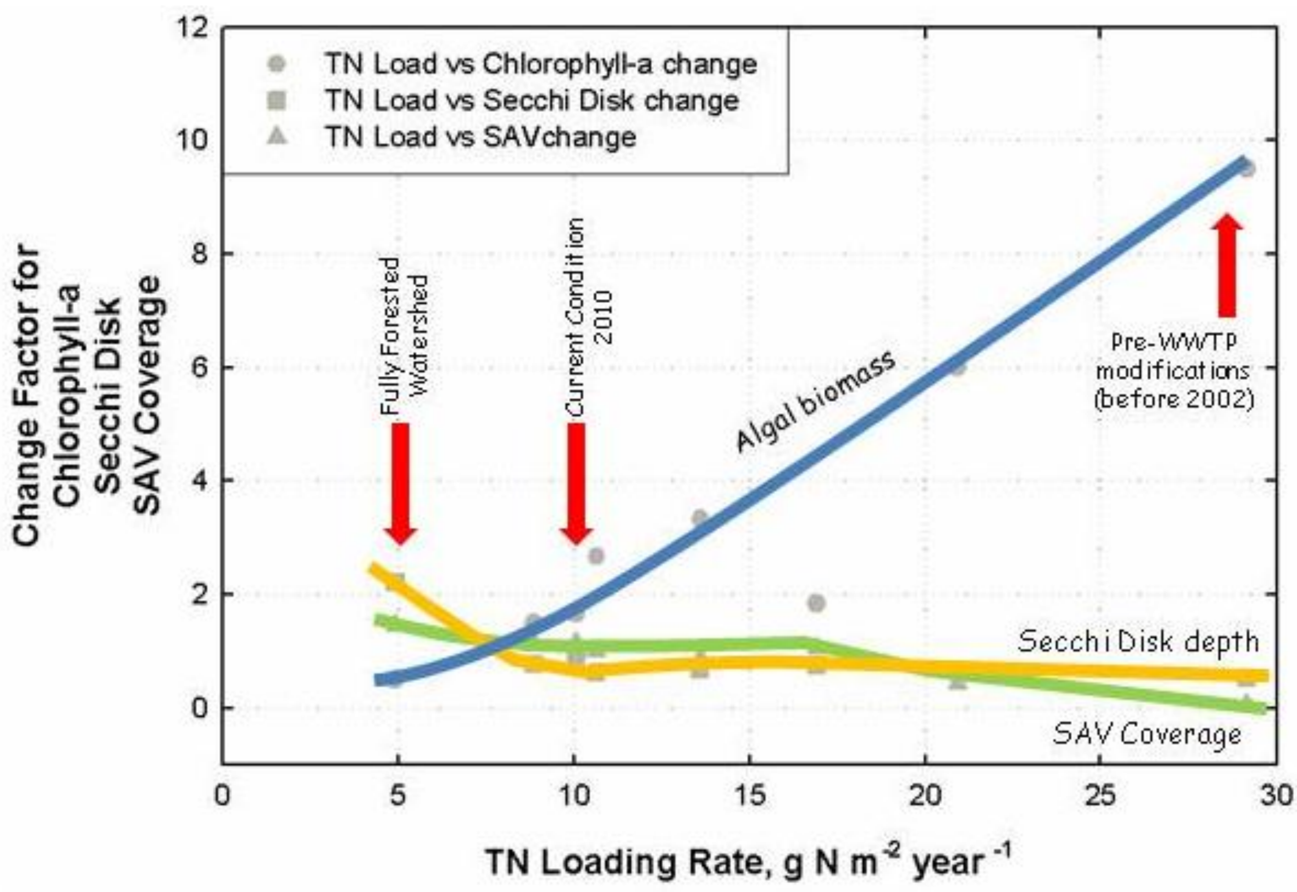

Figure 5-25. A semi-quantitative plot of changes in algal biomass (indexed as chlorophyll-*a*), water clarity (indexed as Secchi Disk depth) and SAV coverage as a function of TN loading rate. The extended portions of these plots are based on best professional judgment; the directions of these trends are well supported in the literature but the magnitude of change is speculative. All data sources used in this figure have previously been cited.

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Chapter 6

Modeling ConMon Data for Dissolved Oxygen Criteria Assessment

Y. Lee and W.R. Boynton

6-1	INTRODUCTION	1
6-2	INITIAL APPROACH TO THE PROBLEM	2
6-3	DATA SOURCES AND ANALYTICAL METHODS	3
6-4	PRELIMINARY RESULTS	3
6-5	NEXT STEPS	5

6-1 Introduction

The dissolved oxygen criteria for Chesapeake Bay were developed after the current traditional fixed channel station monitoring program was adopted and for which there is now a 26 year record for many sites (~150) in Chesapeake Bay and tributary rivers. Details of the dissolved oxygen (DO) criteria were presented earlier (Chapter 2) but here the central issue is that two additional monitoring programs were initiated after the criteria were established, in part to deal with known and suspected spatial variability (Dataflow© monitoring system) in DO concentration (as well as variability in other important water quality variables) as well as temporal variability (ConMon monitoring system) in DO concentration and other water quality variables.

Several DO criteria require measurements of DO conditions at temporal scales much shorter (7 day mean and instantaneous) than provided by the long-term monitoring program and also require consideration of spatial aspects of DO conditions. In the case of Dataflow©, this monitoring system supplies spatially intensive measurements (suite of water quality measurements every 30-60 m) coving a full tributary system usually within a few hours. These measurements are repeated every month for seven months (Apr-Oct) and the work is repeated for three years at each tributary (or mainstem sector) location. Thus, spatial water quality coverage of surface water conditions has been much enhanced since this program began. However, as with all monitoring approaches this one program can not fill all needs. In this case spatial coverage is very intensive but temporal coverage is restricted to monthly time intervals. The second monitoring program (ConMon) provides high frequency shallow water DO (and other water quality variables) measurements (15 minute intervals) at fixed locations (usually the end of a pier) at many locations in the Bay and tributary rivers for the period April – October (at most sites). This measurement scheme is repeated for three years at each site but at a few selected sites (referred to as sentinel sites) ConMon measurements are continued for many years for purposes of more fully capturing scales of inter-annual variability and for detection of water quality trends. As was the case with Dataflow measurement system, ConMon also has limitations and in this case extrapolating data collected at a fixed site to larger estuarine areas remains a challenge.

This EPC analysis represents an initial effort to address the issues of time and space in DO monitoring. Specifically we asked… "can we use ConMon data to develop a statistical model of daily-scale DO dynamics (i.e., the range of DO observed) using a small suite of variables and then apply this model to Dataflow spatial DO distributions thus better characterizing both the spatial and high frequency characteristics of shallow water DO conditions?" This effort is exploratory and focused on the possibility of developing a simple statistical model of day-scale DO patterns based on ConMon data from a few sites.

6-2 Initial Approach to the Problem

We examined the relationship between the range of daily DO concentration and water quality variables in order to estimate the variability of daily DO range using water column properties such as chlorophyll-a concentration, water temperature, PAR, and salinity at three sites in the Maryland Coastal Bays.

The analysis was based on the continuous monitoring (ConMon) data available at the 'Eyes On the Bay' website (http://mddnr.chesapeakebay.net/eyesonthebay/index.cfm) maintained by the Maryland Department of Natural Resources (DNR). Data from three stations were selected for this analysis (Bishopville Prong: station ID XDM4486; Turville Creek station ID TUV0021; Public Landing station ID XBM8828), since they exhibited different degrees of eutrophication. The Bishopville Prong is the most eutrophicated site and the Public Landing the least among the three sites (Fig. 6-1).

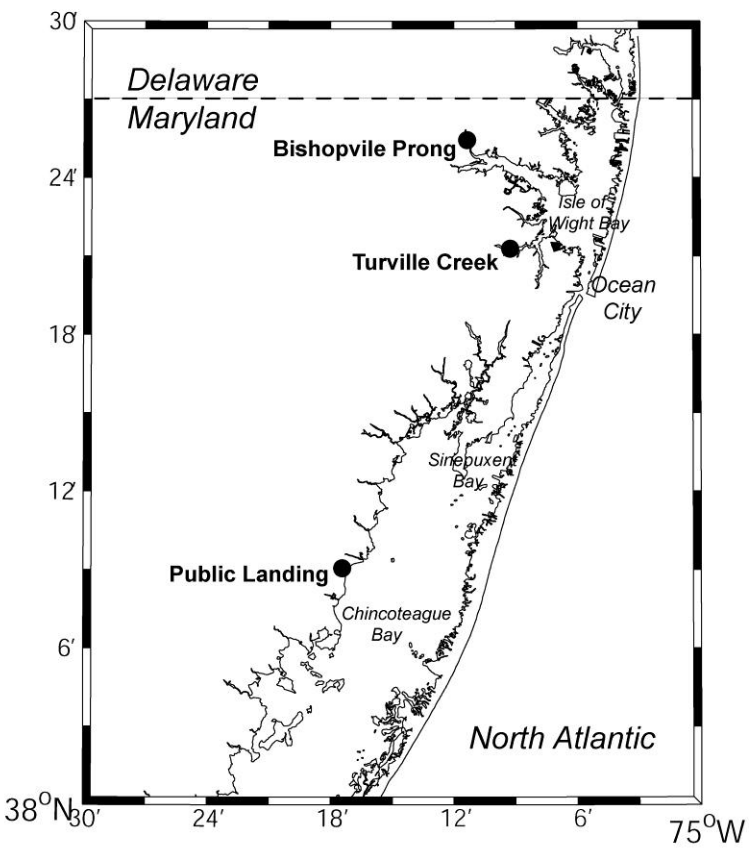


Figure 6-1. Continuous monitoring ConMon station locations (●) in the Maryland Coastal Bays.

6-3 Data Sources and Analytical Methods

The ConMon stations measure water column properties including water temperature, salinity, DO, pH, turbidity, and chlorophyll-a at 15-minute intervals. The analysis was performed using the data from three different locations during the summer (July-August) season using data collected during 2005-2007. The 15-minute data were averaged for a day (midnight to midnight) in each station and data from all three sites were used in this analysis. Daily DO range was also calculated as the difference between the minimum and maximum DO concentration in a 24 hour period. Daily PAR (Photosynthetically Active Radiation) measurements were acquired from Horn Point Laboratory in Cambridge, Maryland. All data, including the calculated daily DO range data) were averaged for every 5-day period and used in subsequent analyses. Previous analyses indicated that basing a statistical model on single day values was not productive because of considerable unexplained variability. Based on the 5-day averaged data, cross-correlation analysis was conducted to estimate the degree to which any two series were correlated. Also, a linear regression method was used to quantify relationships between a dependent variable (DO range) and a selection of independent variables (water column properties and PAR).

6-4 Preliminary Results

We found the 5-day mean of daily DO range during summer was significantly correlated with three water column properties including temperature, salinity, and chlorophyll-a (Table 6-1). Generally, temperature and chlorophyll-a concentration was positively related with the DO range while salinity exhibited a negative relationship. No significant temporally lagged relationships were observed. The strongest correlation coefficient (r) was observed with 5 day averaged chlorophyll-a concentration in 2005 and 2006, but with temperature in 2007, suggesting that there may be inter-annual variability of factors having the most influence on 5 day averaged DO range.

Table 6-1. Correlation coefficients (*r*) between 5-day mean of daily DO range and water column properties including PAR, temperature, salinity, turbidity and chlorophyll-a. Statistically significant relationships are indicated by bold values.

Variables	Year		
	2005	2006	2007
PAR	0.17	-	0.10
Temperature	**0.63**	**0.48**	**0.56**
Salinity	**-0.71**	**-0.54**	**-0.52**
Turbidity	-0.13	0.16	0.22
Chlorophyll-a	**0.80**	**0.85**	**0.39**

-: insufficient PAR data

These initial results suggested that a multiple linear regression method should be applied in each year using three independent variables (chlorophyll-a, temperature, and salinity). Figure 6-2 indicates that the regression model produced a significant result with observed DO range. However, the model exhibits multicollinearity since independent variables are inter-correlated except during 2007 when the variability of salinity shows no significant relationship with temperature and chlorophyll-*a*.

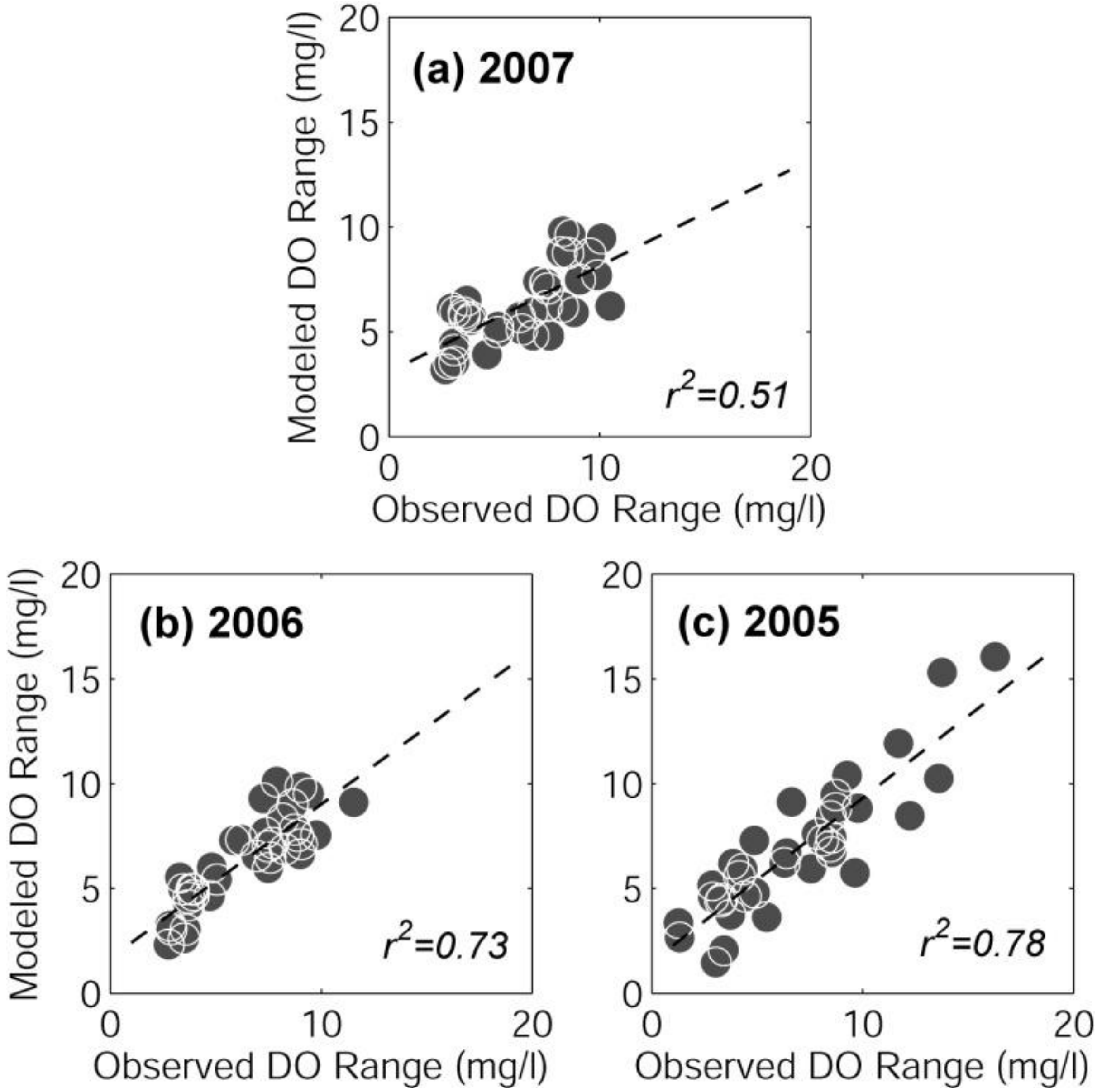


Figure 6-2. Multiple linear regression model results for: (a) 2007, (b) 2006, and (c) 2005. Observed DO range is the dependent variable representing 5-day averaged daily dissolved oxygen (DO) concentration range during the July-August period. Independent variables are the 5-day average of daily mean temperature, salinity, and chlorophyll-*a* concentration.

In addition, we applied different numbers of independent variables instead of combining all three variables in a single model and Table 6-2 shows the results of linear regression analyses based on one and two independent variable models. During 2005 and 2006, the DO range could be primarily explained by chlorophyll-a concentration alone. However, in 2007, salinity and temperature played a role in predicting averaged DO range and these variables were not correlated with each other. This suggests inter-annual variability of DO range might be caused by different forcing influences among years or it may be because sampled data were corrupted. Further investigation of these two ideas is warranted.

Table 6-2. The coefficient of determination (r^2) between observations and linear regression models using different numbers of independent variables during 2005, 2006, and 2007.

Independent Variables	Year		
	2005	2006	2007
Chlorophyll-a	0.64	0.72	0.15
Salinity	0.51	0.30	0.27
Temperature	0.40	0.23	0.32
Salinity + Temperature	0.68	0.38	0.50
Chlorophyll-a + Salinity	0.70	0.72	0.33
Chlorophyll-a + Temperature	0.73	0.73	0.36
Chlorophyll-a + Salinity + Temperature	0.78	0.73	0.51

6-5 Next Steps

At this point it seems possible to develop relatively simple models of DO concentration range using ConMon data. These preliminary results need to be further tested by using data from a larger selection of ConMon sites, examining the utility of including other variables in statistical models, investigating the use of alternative time-averaging schemes, and considering other statistical models (e.g., GAM or others). It remains an unanswered question as to whether further efforts will produce a stronger (i.e., more predictive) or more variable model or models. However, seeking to join the strongest features of two monitoring methodologies for DO criteria evaluation seems to be a useful goal. It also seems possible, and very likely easier, to place this model (or a refinement of this model) into GIS grid cells for tributary rivers and using data collected by the Dataflow© system produce maps of predicted day-scale DO concentration range at the full system spatial scale.

Chapter 7

Upper Patuxent River Estuary: Macrophyte Re-Establishment Case Study

W.R. Boynton and Younjoo Lee

7-1	INTRODUCTION	1
7-2	DESCRIPTION OF PATUXENT RIVER WATERSHED AND ESTUARY	2
7-3	PREVIOUS STUDIES IN THIS ESTUARY	4
7-4	THE UPPER PATUXENT RIVER ESTUARY AS A THRESHOLD SITE	5
7-5	APPROACH OF THIS EVALUATION	5
7-6	PATUXENT RIVER SAV HISTORY	6
7-7	EVALUATION OF INPUTS	7
7-7.1	FALL LINE FLOWS AND LOADS	8
7-7.2	LOCAL POINT SOURCES OF N AND P	11
7-8	SAV AND WATER QUALITY IN THE TIDAL FRESH (TF) AND OLIGOHALINE (OH) ZONES	12
7-8.1	SAV DISTRIBUTIONS	12
7-8.2	TEMPERATURE	14
7-8.3	SALINITY	14
7-8.4	WATER COLUMN TSS AND TURBIDITY	14
7-8.5	NUTRIENT CONCENTRATION	16
7-8.6	CHLOROPHYLL-*A* CONCENTRATION	17
7-9	LONG-TERM WATER QUALITY TRENDS	18
7-10	NUTRIENT SIGNALS IN ESTUARINE SEDIMENTS	21
7-11	A RECORD OF COMMUNITY METABOLISM	24
7-12	STATISTICAL MODELING OF WATER QUALITY AND SAV DISTRIBUTIONS	27
7-12.1	RIVER FLOW VERSUS WATER QUALITY	27
7-12.2	WATER QUALITY RELATIONSHIPS AMONG SITES IN THE SAV RE-GROWTH AREAS	28
7-12.3	SAV THRESHOLD RESPONSES	29
7-12.4	INTER-ANNUAL VARIABILITY IN THE POST-RECOVERY PERIOD (1994-2007)	32
7-13	LESSONS FOR MANAGEMENT	34
7-14	FINAL SUMMARY COMMENTS AND SPECULATION	36
7-15	REFERENCES	39

7-1 Introduction

During the past few years there has been a growing interest in the water quality management community for analyses directed toward understanding ecosystem responses to management actions. This type of analysis has been a tradition in fisheries management but such analyses have not been anywhere near as common in the water quality management community. Recent case studies of water quality management and ecosystem responses include the analysis completed for the Corsica River system (Boynton *et al.* 2009), and two analyses relating nutrient conditions to seagrass community status and trends (Orth *et al.* 2010; Ruhl and Rubicki, 2010).

The analysis provided here focused on the upper Patuxent River estuary, a eutrophic system on the Western shore of Chesapeake Bay. This site was selected for analysis because there have been, and continue to be, strong management actions taking place to improve water and habitat quality of the ecosystem. Specifically, there have been major upgrades to the waste water treatment plants (WWTP) that discharge into the upper portion of this system. These up-grades continue and will likely be completed during 2012-2013.

During 2010 the Chesapeake Bay Program and Maryland Sea Grant provided some "seed" funding to support analyses aimed at detecting and understanding the nature of ecosystem responses to management actions. The upper Patuxent River estuary was selected as one of several sites. We have included a draft version of the Patuxent analysis in the 2011 Ecosystem Processes Component (EPC) Interpretive Report because a substantial fraction of this analysis was completed in concert with other EPC work and because Maryland DNR has a keen interest in the results of case studies associated with management actions. A full report, including three other case studies is expected to be completed during 2012.

7-2 Description of Patuxent River Watershed and Estuary

The Patuxent River basin encompasses 2256 km^2 of land, 143 km^2 of open tidal waters and 29 km^2 of tidal marshes; intertidal area is very small. The Patuxent ranks sixth in drainage basin size, sixth in estuarine volume and seventh in freshwater inflow among the tributaries of the Chesapeake system (Cronin and Pritchard 1975). It is among the better known and studied because of a long history of management debate, court cases and eventual management actions aimed at water quality and habitat restoration (Mihursky and Boynton 1978; Heinle *et al.* 1980; Malone *et al.* 1993; D'Elia *et al.* 2003; Fisher *et al.* 2006; Testa *et al.* 2008; and others).

Human population in the Patuxent basin was about 30,000 (13 km^2) in 1900. The basin remained very rural until about 1960 when rapid population growth began, a trend that continues to the present. During a recent 10 year period (1992-2001) population increased by 36, 14 and 50% in the upper, mid and lower basins, respectively. Population density in 2001 was highest in the upper basin (356 km^2) and less than half that in the mid (154 km^2) and lower (157 km^2) basins. Population density in mid-Atlantic basins averaged 317 km^2 in 1990, similar to the density of the upper Patuxent basin (Basta *et al.* 1990).

The dominant land use in the Patuxent basin as of 2001 (Homer *et al.* 2004) was forested lands (40%); the percentage decreased from the lower (47%) to upper basin (26%). Agricultural land uses accounted for 44% of the upper watershed and a smaller proportion in the middle and lower basins. Urban, suburban and other developed land uses were highest in the upper basin (22%) and lower elsewhere. These data reflect ongoing conversion of forest and agricultural land to residential and urban uses.

The Patuxent River and estuary are about 170 km in total length; the lower 95 km are tidal (Fig. 7-1**).** The upper portion of the tidal estuary, from river kilometer (rkm) 40 to 95, is narrow (50 – 300 m wide), very turbid (K_d =3.0 m^{-1}), vertically well-mixed, and has a tidal range of 0.5 - 1.0 m and an average depth of 1.1 m. In addition, this portion of the estuary is flanked by extensive tidal freshwater and salt marshes with the ratio of marsh area to river distance ranging from 0.4 to 0.8 km^2 of river. The surface area of the upper estuary is 26 x 10^6 m^2. The lower estuary (rkm 40 to mouth at Chesapeake Bay) is much wider (1 to 5 km), deeper (mean depth = 5.4 m), clearer (K_d = 0.9 m^{-1}) and seasonally stratified. The surface area of the lower estuary is 117 x 10^6 m^2.

The Maryland Department of Natural Resources maintains a web page with detailed statistics concerning the watershed, river and estuary (http://mddnr.chesapeakebay.net/wsprofiles).

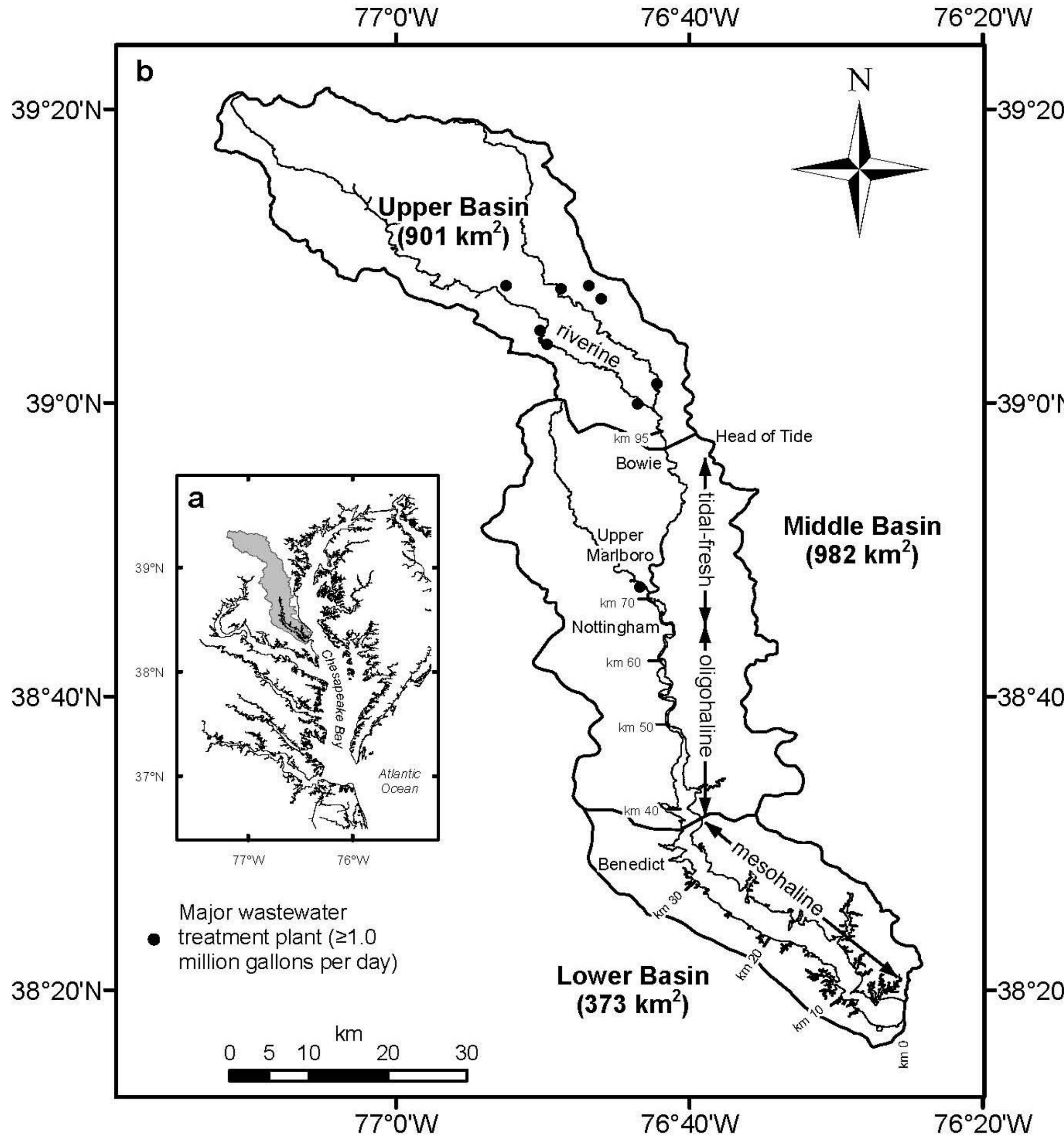


Figure 7-1. Map showing regional location (a) and spatial details (b) of the Patuxent basin and Patuxent River estuary. This study focuses on the tidal freshwater (TF) and oligohaline (OH) zones of the estuary located between river kilometers (rkm) 45 and 70.

7-3 Previous Studies in this Estuary

The Patuxent estuary and, to a lesser extent, the surrounding watershed has a long history of environmental monitoring and research. The record of early measurements, mainly focused on water quality, began in the 1930s and most of this information was summarized by Mihursky and Boynton (1978). However, much of this work lacked continuity regarding location of sampling sites, variables measured, analytical methods used and study duration. As a result, it was difficult to detect water quality trends occurring during the pre-Bay Program period (before 1985) with a few notable exceptions (Heinle *et al.* 1980; Cory 1974). One exception that is particularly relevant here is the work of Cory and colleagues who maintained an automated and near-continuous water quality monitoring site at Benedict, Maryland (rkm 35) from 1964 through 1969. Additional and very similar high frequency measurements were again instituted during 1992, 1996-1998 and 2003-2005. These data were used to compute estimates of community production and respiration and are examined in this case study. Despite all the shortcomings of early monitoring programs, it did appear that water clarity was greater and bottom water dissolved oxygen concentration higher during the earlier periods. Finally, during the mid to late 1960s SAV disappeared from both the upper and lower estuary (Stankelis *et al.* 2003).

Beginning in 1968 Flemer *et al.* (1971) began a modern water quality study of the estuary and a few years later Maryland DNR initiated a similar program. Both lasted for 2-3 years and were terminated. We have yet to obtain the DNR data but much of the Flemer *et al.* (1970) data have been converted to electronic format. In 1984 the US EPA Chesapeake Bay Program initiated a long-term Biomonitoring program and 13 water quality stations were established along the axis of the tidal portion of the Patuxent. In addition, this estuary also had a USGS River Input station located above the head of tide (http://va.water.usgs.gov/chesbay/RIMP/dataretrieval.html). There were also benthic, zooplankton, phytoplankton and sediment processes monitoring programs at selected sites in this estuary; all but the benthic program were terminated in the past 2-8 years. Fortunately, an annual aerial-based SAV monitoring program remains active. All of these data are available from the Chesapeake Bay Program web site (http://www.chesapeakebay.net/) or linked sites.

Because SAV are a focal point of this case study, we reviewed a considerable literature concerning SAV in the Patuxent. This literature will be used in more detail later but included work on the long-term (paleoecological) history of SAV in the estuary, field observations during the 1960s-1970s, reconstruction of SAV distribution based on historical aerial photographs, and the more recent and comprehensive SAV aerial monitoring work that was initiated in 1985 as part of the USEPA Chesapeake Bay Program.

Very recently an analysis of SAV trends in Chesapeake Bay related to water quality was published (Orth *et al.* 2010) and this work included an analysis of SAV trends in the upper Patuxent River. Results for the Patuxent indicated an important role for local point sources of N and P and water clarity in determining SAV coverage. To some extent this published work covers the same issues addressed here. However, there are several important differences. First, the recent evaluation focused on the resurgence of SAV in the Patuxent; we will examine initial SAV decline as well as resurgence and recent inter-annual variability and attempt to relate both to water quality using the same data sets used in the recent study as well as a few that were not included in the recent study. Second, we have also developed a long-term record of community metabolism for this system (1964-2005) in a portion of the estuary just downstream of the area of SAV resurgence and the community metabolism time-series will be examined relative to threshold responses as well.

7-4 The Upper Patuxent River Estuary as a Threshold Site

As with all the sites considered for ecological threshold analysis, the upper Patuxent estuary was selected because there have been strong management actions achieved in this system. In the case of the Patuxent, the major management actions were related to a three-step sequence of improvements in WWTP discharges. The first involved a P-ban on detergents and P removal at WWTPs (completed by 1986), the second involved seasonal (warm seasons) removal of N using biological nitrogen removal (BNR) technology (completed by 1993), and the third involved adoption of continuous enhanced nutrient removal (ENR) at the Western Branch WWTP (2004). Point source TP loads declined from about 250 kg P day^{-1} before upgrades to about 60 kg P day^{-1} following upgrades. Point source TN loads (from all WWTPs above the fall line) ranged from 1,200 – 1,900 kg N day^{-1} prior to upgrades and declined to about 500-600 kg N day^{-1} during summer periods after upgrades. Nitrogen loads at the Western Branch WWTP (located below the fall line adjacent to SAV communities) were about 900 kg N day^{-1} in 1991, decreased to an annual average of about 500 kg N day^{-1} from 1992–2004, and then decreased further to about 150 kg N day^{-1} in 2004. Many details concerning nutrient loads and ecological consequences have been addressed by D’Elia *et al.* (2003), Fisher *et al.* (2006), Boynton *et al.* (2008), Testa *et al.* (2008), and Testa and Kemp (2008). The CBP Biomonitoring Program’s SAV aerial surveys indicated a dramatic response by the SAV community following nitrogen removal at WWTPs in this system and this response indicated the upper Patuxent would be a useful system to examine for ecological thresholds.

Additionally, we have developed a near-continuous record of dissolved oxygen, temperature and salinity from 1964–1969, a period encompassing the SAV decline in this estuary (Cory 1974). We also have the same type of data record for the periods 1992, 1996-1998 and 2004-2005 covering the period following WWTP upgrades and SAV re-emergence in this system. We have used these data to compute various measures of community production and respiration and we examined this data set for indications of threshold responses to management actions.

7-5 Approach of this Evaluation

The focus of this threshold analysis concerns SAV communities in the upper tidal portion of the Patuxent River estuary. Specifically, the focus is on SAV responses in the tidal freshwater (TF) and oligohaline (OH) zones of the estuary. A wide variety of factors potentially controlling or influencing SAV distribution were considered and these included water column characteristics (e.g., temperature, salinity, water clarity and dissolved and particulate nutrient concentrations), sediment characteristics (e.g., sediment C, N and P concentrations, sediment-water nutrient exchanges) and nutrient loading rates from both proximal (e.g., local point source discharges) and more distal (river inputs at the Patuxent fall line) sources. Information concerning SAV distribution was available only on an annual basis and the modern record began in 1985 with less quantitative data available from earlier periods of the 20th century and before. We have examined potential factors influencing SAV re-emergence in this system, and recent inter-annual variation in SAV distribution, using a variety of time-scales ranging from monthly to multi-year averages.

Finally, we have also examined a rare, high frequency time-series of dissolved oxygen, salinity and temperature collected at Benedict, MD from 1964-1969, 1992, 1996-1998, and 2003-2005. Based on these data we have estimates community production and respiration and examined these data as well for threshold-like responses.

7-6 Patuxent River SAV History

There have been many observations made concerning SAV in the Patuxent estuary including those of Academy of Natural Sciences of Philadelphia (ANSP,1965), Anderson *et al.* (1965), Anderson (1966), Haney (1966), Anderson and Rappleye (1969), Anderson (1969), Mihursky (1969), Rawls *et al.* (1975), and Kerwin *et al.* (1976). More recent observations include those of Stankelis *et al.* (2003) and the long-term SAV biomonitoring work done by the Chesapeake Bay Program (http://web.vims.edu/bio/sav/). In addition, there are several paleoecological studies of the Patuxent estuary, including SAV analyses (Brush *et al.* 1980; Brush and Davis 1984; and Brush and Hilgartner 2000). Thus, there is a diverse record of SAV characteristics in this estuary from pre-colonial periods to the present.

The paleo-record for SAV communities suggests the following. In the tidal fresh area six SAV species were typically present prior to 1840 and after this date several more pollution-tolerant species appeared. Seeds of these and several other species disappeared from the record, likely during the 1960-1970s (Brush *et al*. 1980). Brush and Davis (1984) noted that there was a marked decrease in the number of SAV seeds deposited since the 1960s and further concluded that the mesohaline SAV were less affected by pollution than were SAV in the OH and TF portions of the system. Finally, Brush and Hilgartner (2000) reported local extinctions of two pollution tolerant SAV species from the upper Patuxent during the late 1960s.

The earliest direct reference to Patuxent SAV we have found comes from a book published by Hungerford (1859), who owned a large farm in Calvert County, MD. He describes, in poetic terms, SAV growing in the lower mesohaline estuary in waters likely to have been 4-6 m in depth during the summer of 1832. He states being able to clearly see… the finny tribes (fish schools) moving through the seaweed forest (SAV communities) and gliding over the pearly sands. Observations from the early to mid-1960s leave the impression of SAV beds being in a healthy condition. For example, ANSP (1965) reported SAV beds extending out about 300 m from the shore in the vicinity of Benedict, MD (river kilometers, rkm 35) and SAV were reported to be common in all tidal areas of the river system. During 1963-1964 Anderson *et al.* (1965) conducted a survey of SAV and other aquatic plants in the river from near the head of tide (rkm 90) to Sheridan Point (rkm 30) at the head of the mesohaline estuary. Again, in the Benedict area, SAV coverage was extensive although there seems to have been less SAV upriver of Summerville Creek (rkm 45). A summer ANSP survey in 1966 indicated SAV were less abundant than in earlier years (1962-1964) in the vicinity of Sheridan Point. Mihursky (1969) reported that by the late 1960s SAV did not extend upriver of Holland Cliff (rkm 50). In 1966 Haney (1966) conducted a brief SAV survey in the vicinity of Solomons, MD and reported dense beds of SAV. By the mid-1970s SAV were reported to be present only in small and scattered pockets (Rawls *et al.* 1975). Anecdotal observations by staff at the Chesapeake Biological Laboratory report that SAV were gone from the Solomons area by about 1970 and have not returned. Approximately 50 sites were visited annually (Aug-Sep) in the Patuxent from 1971 - 1974 and the percentage of sites having SAV ranged from 0.0 -4.3%. This SAV frequency of occurrence was among the lowest of the 27 Bay systems sampled (Kerwin *et al.* 1976). Stankelis *et al.* (2003) utilized early aerial photographs to characterize SAV coverage in the mesohaline Patuxent. These photographs clearly indicated dense SAV communities were present during the late 1930s. Coverage was similar or slightly greater in the mesohaline region during the mid-1950s. By the mid-1970s coverage had decreased by an order of magnitude and by the mid-1990s had declined by another order of magnitude.

From these sources it is possible to chart a semi-quantitative time-line of SAV community demise in this estuary. It appears from the paleo-ecological data that SAV were present as a time-varying cluster of species prior to European settlement through about 1840. After this date more pollution-tolerant species were added to the mix. Furthermore, there is evidence of diverse SAV communities in the tidal freshwater and mesohaline zones as late as the 1950s and early 1960s. Direct observations seem to indicate a decline in mesohaline SAV communities during the mid-to-late 1960s, long before the arrival of Tropical Storm Agnes in June of 1972. There is considerable evidence that SAV were largely gone from the mesohaline region by 1970 and have not returned. The time-line for the oligohaline and tidal freshwaters is not quite as clear. There were indications of SAV decline in the oligohaline region by the mid-to-late 1960s and one observer noted that SAV were gone from the oligohaline and tidal freshwater regions by 1969. These communities, once massive, likely did not disappear during a single year. Evidence, though not conclusive, indicates substantial stress on these communities by the mid-1960s and local extermination before 1970. It is interesting to note that Flemer *et al*. (1971) conducted the first modern water quality study in the upper estuary during 1968-1970 and did not mention SA, likely because they were already gone from this zone of the estuary.

7-7 Evaluation of Inputs

The Patuxent River estuary has been monitored and studied for a considerable number of years, as indicated by Mihursky and Boynton (1978). One of the benefits of this monitoring is that Hagy *et al*. (1998) were able to reconstruct N and P inputs to the estuary at the head of tide near Bowie, MD, a site that drains about 40% of the watershed and which has been the most intensively developed since about 1960. The reconstructed loads generated by Hagy *et al*. (1998) were spliced to the more modern measurements developed by the USGS at the Bowie site from 1978-2009. One especially relevant aspect of these measurements is that they encompass the period prior to massive basin development (1960-1965), the period of rapid and ill-regulated growth and the early period of WWTP discharges (1966-1985), and the period when growth controls began to be implemented (e.g., MD Critical Area Law) and WWTP operations began removing phosphorus (1986) and later nitrogen (1992-1993 and again in 2004). Time-series plots for a 49 year period of freshwater flow, TN, TP, NO23 and PO4 loads are provided in Figures 7-2 to 7-6.

7-7.1 Fall Line Flows and Loads

It is clear that freshwater flows varied substantially both within and among years (Fig. 7-2). In general, flows were high during winter-spring and low during summer-fall. Of course, there were exceptions and these were also clear. For example, the highest flow on record (~1600 cubic feet per second, cfs) was associated with Tropical Storm Agnes in June1972. There were several dry periods and these were also evident, especially during the period 1962-1969. Likewise, wet periods were evident during the late 1970s and the mid-1990s.

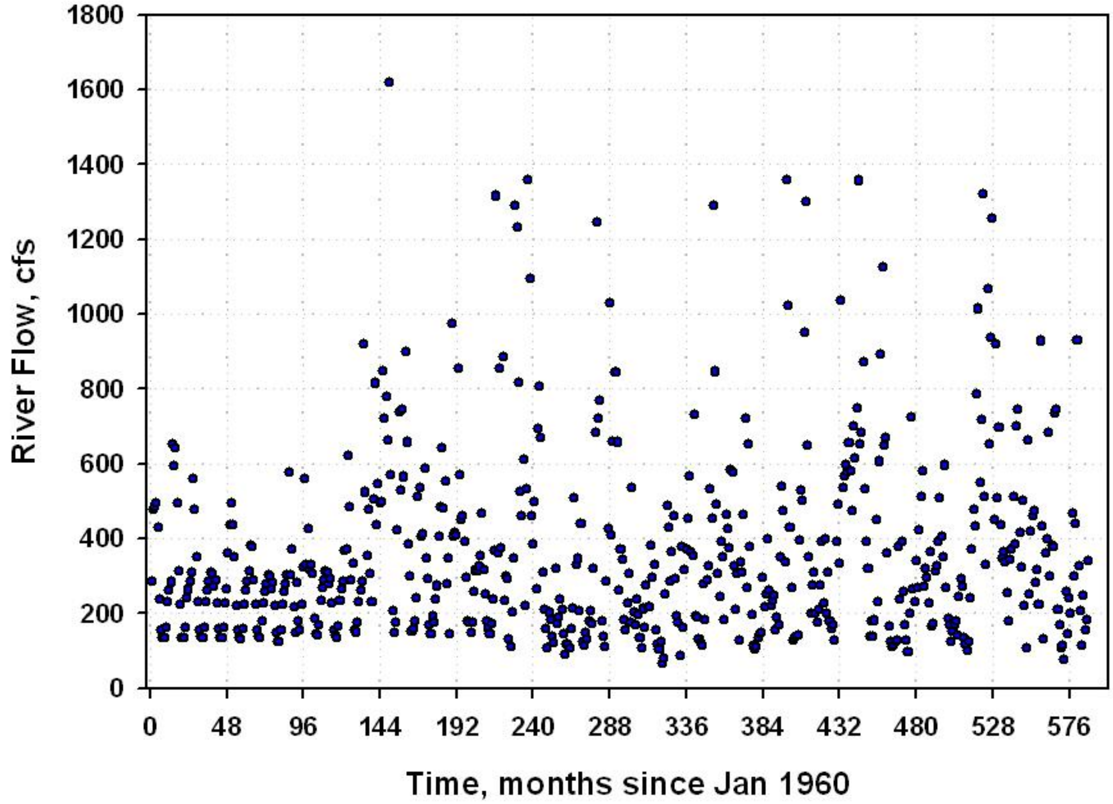


Figure 7-2. A time-series plot of freshwater flow (cfs) of the Patuxent River measured at the USGS gage located at Bowie, MD. The record high flow in June 1972 was associated with Tropical Storm Agnes.

Total nitrogen (TN) loads measured at the fall line (Bowie, MD; 40% of the basin) were also quite variable within and among years, generally following the pattern of flow (Fig. 7-3). The single highest TN load was associated with Tropical Storm Agnes (June, 1972). There was a distinct pattern in TN loads during the full period with loads increasing from 1960 until the mid-1980s and then decreasing through the present time, except for a temporary increase during the very wet years of 2003 and 2004. Annual average loads during the last few years were comparable to those observed during the mid-1960s.

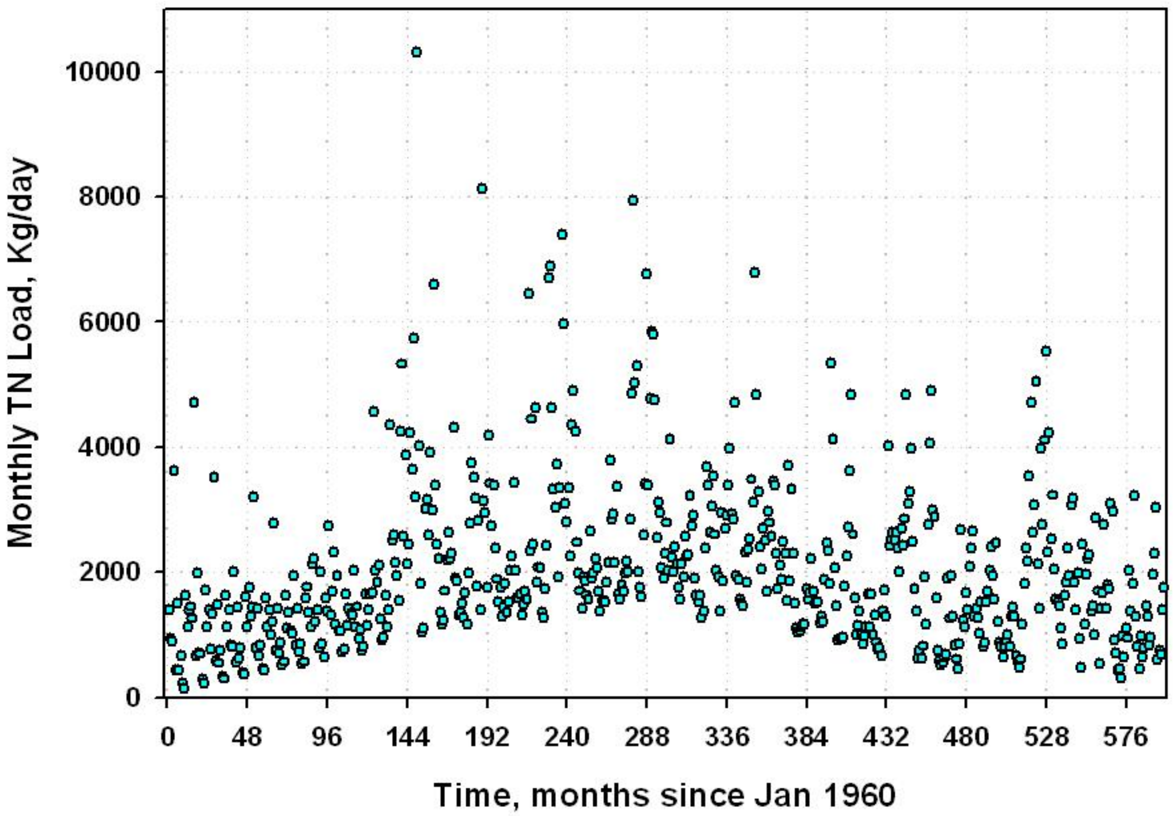


Figure 7-3. A time-series plot of total nitrogen (TN) load to the Patuxent River measured at the USGS gage located at Bowie, MD. This gage estimates flow and loads from about 40% of the Patuxent basin.

There were some remarkable changes in TP loads measured at the Patuxent fall line (Fig. 7-4). Occasional loads were quite high during the 1960s, increased to maximum values between 1978 and 1984 and then were sharply reduced after 1986, associated with the P-ban in detergents and P removal at WWTP. Very few high P loads have been observed since 1986, even during the wet years of 2003 and 2004, and that suggests better control of storm-related P inputs.

Loads of biologically reactive NO_{23} and PO_4 also exhibited considerable within and among year variability (Figs. 7-5 and 7-6). In the case of NO_{23}, loads slowly increased during the 1960s, increased more rapidly from 1970 through the late 1980s and then began decreasing until the present time. Higher loads were evident during the wet years of 2003 and 2004. The loads of PO_4 reached a maximum in 1979 and then began to decrease, reaching much lower levels after 1986 and maintaining these low load levels through 2009.

The general picture that emerges from this 49 year time series is one of varying periods of high and low flow, increased loads of both N and P early in the record and then sharply decreasing P loads followed by less sharply decreasing N loads. It is important to remember that the Bowie, MD site monitors only 40% of the drainage basin so these loads do not represent the full water flows or nutrient loads to this estuary but they do represent a large fraction of basin drainage to the TF and OH zones of the estuary where SAV have again become a part of this system.

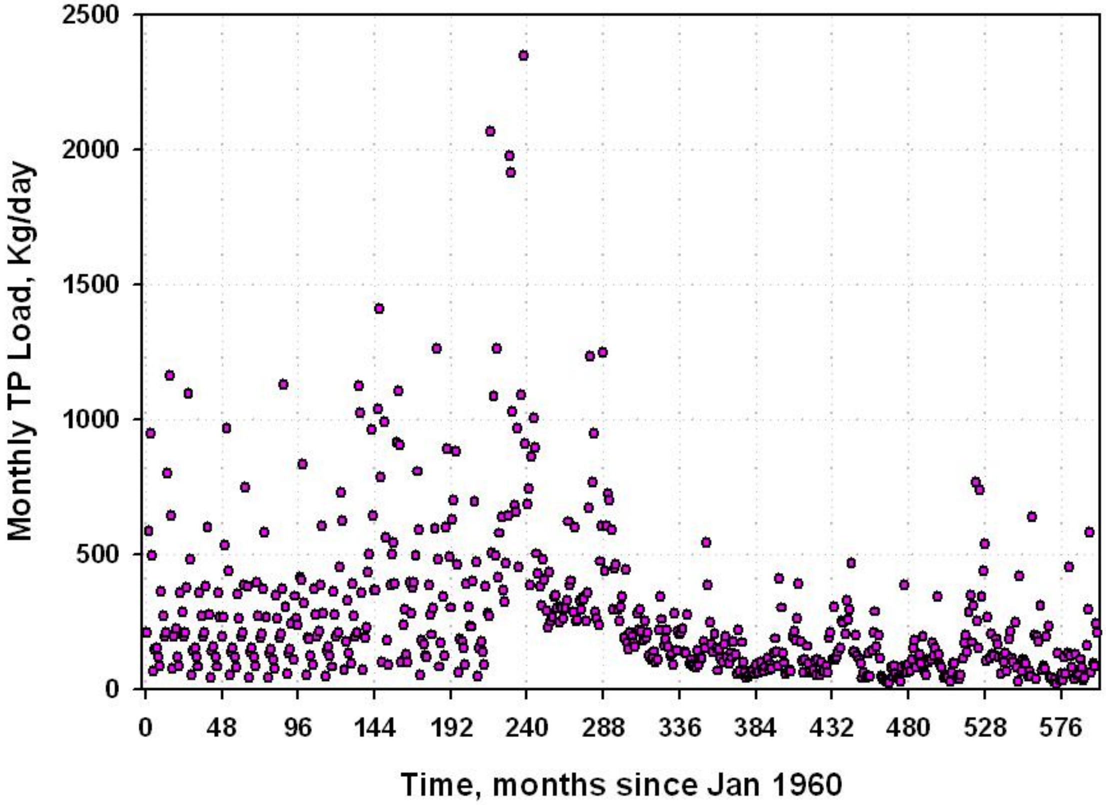


Figure 7-4. A time-series plot of total phosphorus (TP) load to the Patuxent River measured at the USGS gage located at Bowie, MD. This gage estimates flow and loads from about 40% of the Patuxent basin.

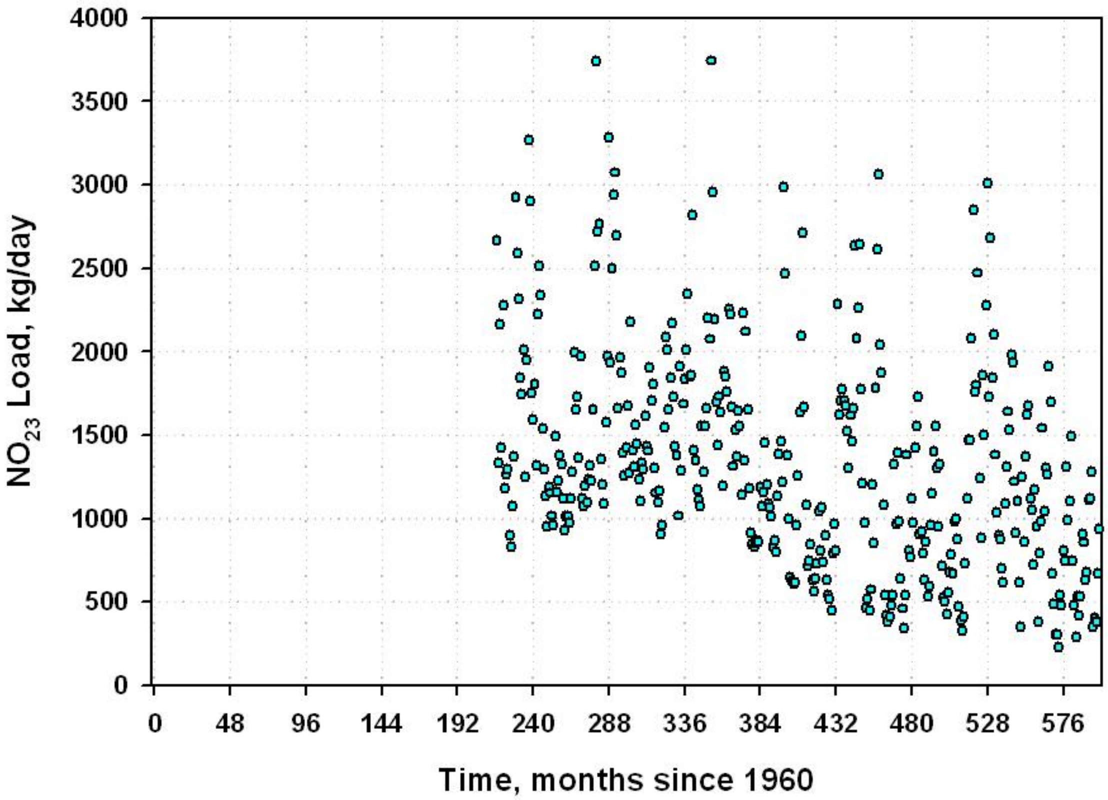


Figure 7-5. An incomplete time-series plot of nitrate plus nitrite load to the Patuxent River measured at the USGS gage located at Bowie, MD. This gage estimates flow and loads from about 40% of the Patuxent basin. Data from the earlier period need to be located but do exist.

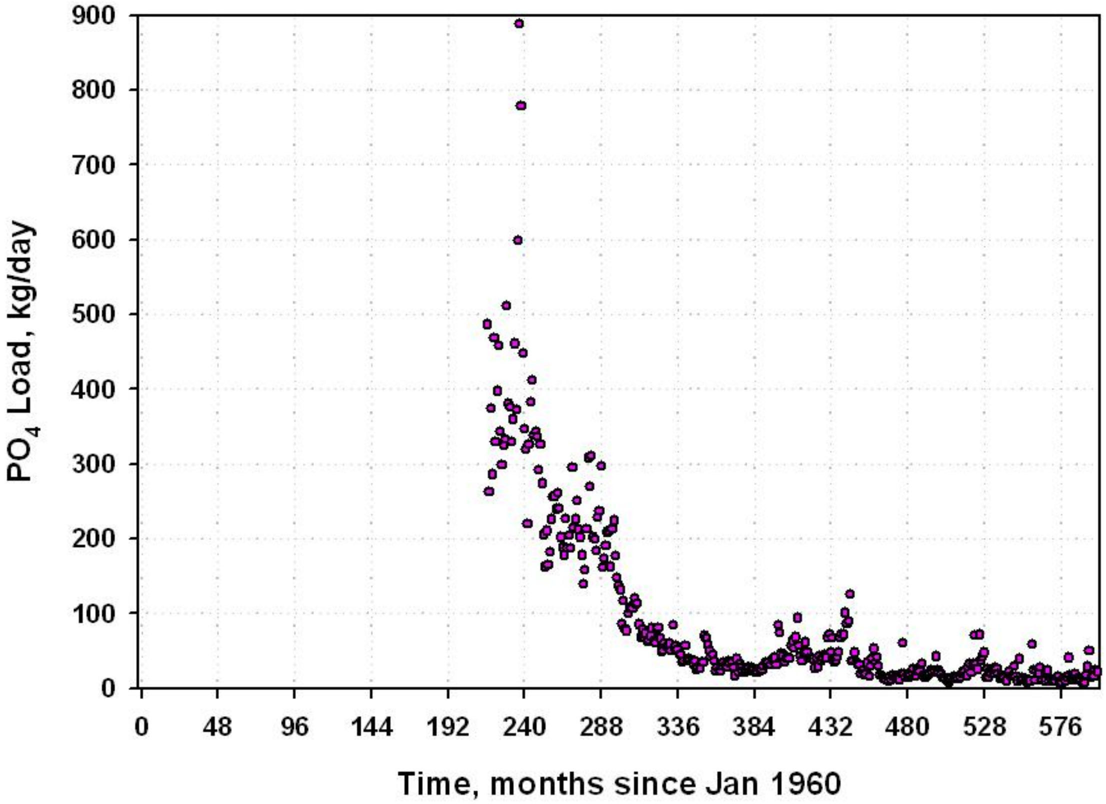


Figure 7-6. An incomplete time-series plot of dissolved inorganic phosphorus (PO_4) load to the Patuxent River measured at the USGS gage located at Bowie, MD. This gage estimates flow and loads from about 40% of the Patuxent basin. Data from the earlier period need to be located but do exist.

7-7.2 Local Point Sources of N and P

The fall line loads reported above include both point and non-point N and P sources. However, there is one large WWTP (22 mgd discharge in recent years) located below the fall line (Western Branch WWTP) and this discharge is in the vicinity of SAV resurgence. Because of the large size and location of this facility, flow and N and P load data are included here.

We have also located some data concerning the initial operations of WWTP in the Patuxent Basin. Domotor *et al.* (1989) reported the first WWTP discharges started in 1963 (flow = 2.5 mgd; TN load = 200 kg day^{-1}; TP load = 50 kg day^{-1}) and rapidly increased for the next 15 years after which increases continued at a slower rate. Of particular interest here is that estimates of WWTP discharge and loads during 1967 were 12 mgd, 850 kg N day^{-1} and 210 kg P day^{-1}. This is the period during which SAV were being lost from this section of the estuary.

Flows and loads from the Western Branch WWTP are shown in Figure 7-7 for the period 1985-2008. This time-series clearly shows the large reduction in P loads during 1986 and reasonably constant P loads (~50 kg P day^{-1}) through 2008. The TN load sequence is more complex. TN loads increased steadily from 1985-1992 and then began seasonally varying with high (and also increasing) loads during the cool periods (800-1200 kg N day^{-1}) and much lower loads during warm periods (100-150 kg N day^{-1}). Finally, in 2004, ENR was instituted at this plant on an annual basis and TN loads stabilized at about 100-150 kg N day^{-1} throughout the year. The TN load to the Patuxent in the vicinity of the SAV resurgence is about half that of the point source N load from all upper basin WWTPs in 1967.

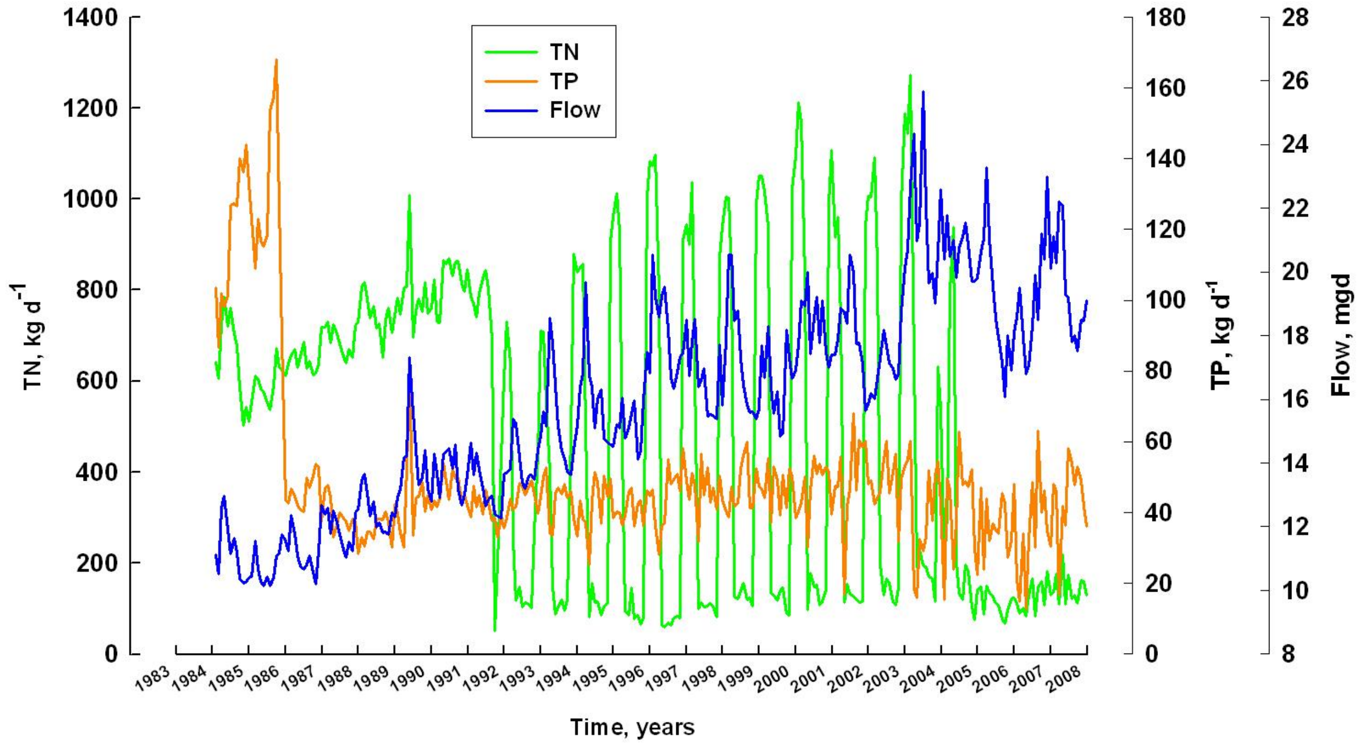


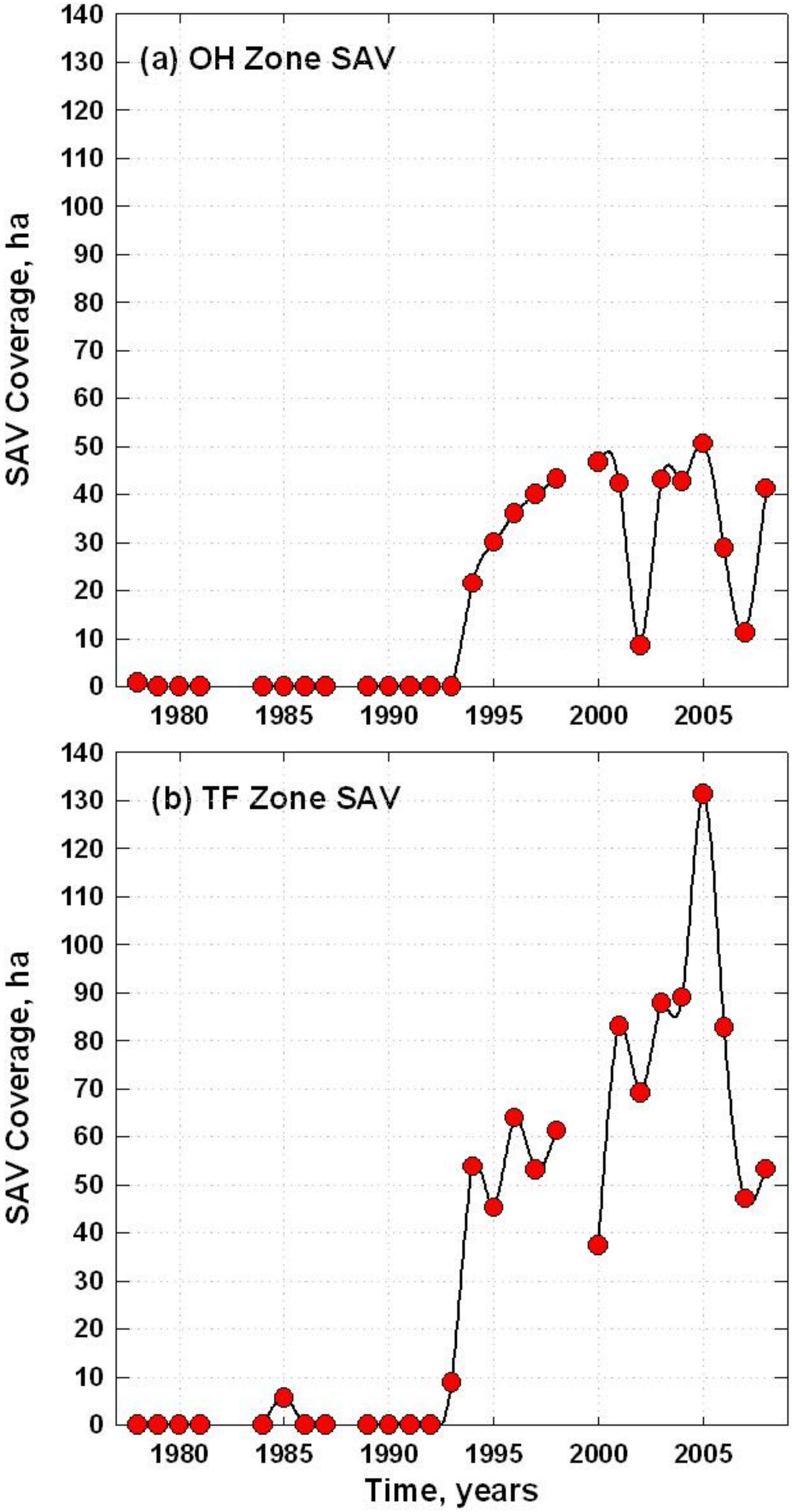
Figure 7-7. A time-series plot of discharge flow, TN and TP loads from the Western Branch WWTP. The WWTP discharge is located at river kilometer (rkm) 72.

7-8 SAV and Water Quality in the Tidal Fresh (TF) and Oligohaline (OH) Zones

In this section we describe patterns of SAV distribution and a variety of water quality conditions that could influence SAV distribution. These data are qualitatively described here and later used, along with other data, in statistical models focused on better understanding SAV resurgence and inter-annual SAV variability in this system. There are two water quality stations in this sector of the estuary (TF 1.3 and TF 1.5). In the descriptions of water quality that follow the data from the upper river site (TF 1.3) are emphasized because these are more closely related to the largest areas of SAV distribution. Differences between these sites are noted.

7-8.1 SAV Distributions

Qualitative descriptions of SAV distribution in the Patuxent were provided earlier in this report. This section focuses on SAV distribution in the OH and TF portions of the estuary and is based on aerial monitoring work (http://web.vims.edu/bio/sav/).



The monitoring record for the OH zone indicates absence of SAV from 1978 through 1993, after which there was a sharp increase (0 to 45 ha) through 2000-2001 (Fig. 7-8a). SAV distribution sharply decreased during 2002 (a low flow year), re-bounded to former levels during 2003-2005 (45-50 ha), dipped again during 2007 and rebounded during 2008. There were no dips in SAV distribution since 1994 associated with high flow years (e.g., 1996 and 2003) but declines in SAV distribution were associated with low flow years (e.g., 2002 and 2007).

The pattern of SAV resurgence in the TF zone was similar to that observed in the OH zone except that the spatial extent was considerably greater (50-130 ha). Aerial surveys found almost no SAV from 1978-1992. During 1993 there was a small amount recorded (10 ha) after which coverage expanded to 45-64 ha through 1998. Coverage increased again from 2001 through 2006 (70-130 ha) and then decreased to levels observed during the mid-late 1990s (Fig. 7-8b).

Figure 7-8. Time-series plots of SAV coverage (ha) in the OH (a) and TF (b) zones of the Patuxent River estuary from 1978-2008. Data were from the Chesapeake Bay Biomonitoring Program (http://web.vims.edu/bio/sav/).

7-8.2 Temperature

Water temperature at both sites exhibited an expected range through all years of record (~3 to 26 °C). However, there were some substantial inter-annual differences between maximum and minimum temperature. For example, near-zero temperatures were recorded during the late 1980s while minimum temperatures were about 5 °C during the late 1990s. Similarly, maximum summer temperatures reached 28-29 °C several times prior to 1996 and were as low as 23 °C during the early 2000s. There were very few extreme temperatures recorded since SAV recovery started in 1993.

7-8.3 Salinity

There was no measureable salinity at the upper river site (TF 1.3) but salinities of up to 2.5 ppt were observed at a more downriver site (TF 1.5) and were higher still at TF 1.7 near the downriver boundary of SAV re-growth. Measureable salinity always occurred during summer and fall (low river flow periods) when SAV are growing and highest salinity was associated with drought years (e.g., 1986, 1999, 2001 and 2002). Since the SAV growing in this zone of the river are basically freshwater species, the possibility of salt stress during low flow summer periods exists and may be partly responsible for observed inter-annual variability in SAV coverage in the OH zone.

7-8.4 Water Column TSS and Turbidity

Concentrations of TSS were generally higher at TF 1.5 (20-60 mg L^{-1}) than at TF 1.3 (10-50 mg L^{-1}) but these are all high concentrations. Both sites exhibited frequent spikes in TSS concentration between 60 and 100 mg L^{-1} (Fig. 7-9a). Water clarity at both sites was quite poor with Secchi Disk depths ranging from 0.1 to 0.5 m at TF 1.5 and (with a shorter record) from 0.2 to about 1.0 m at TF 1.3 (Fig. 7-9b). There were no obvious long-term (1985-2007) trends at TF 1.5 or at TF 1.3.

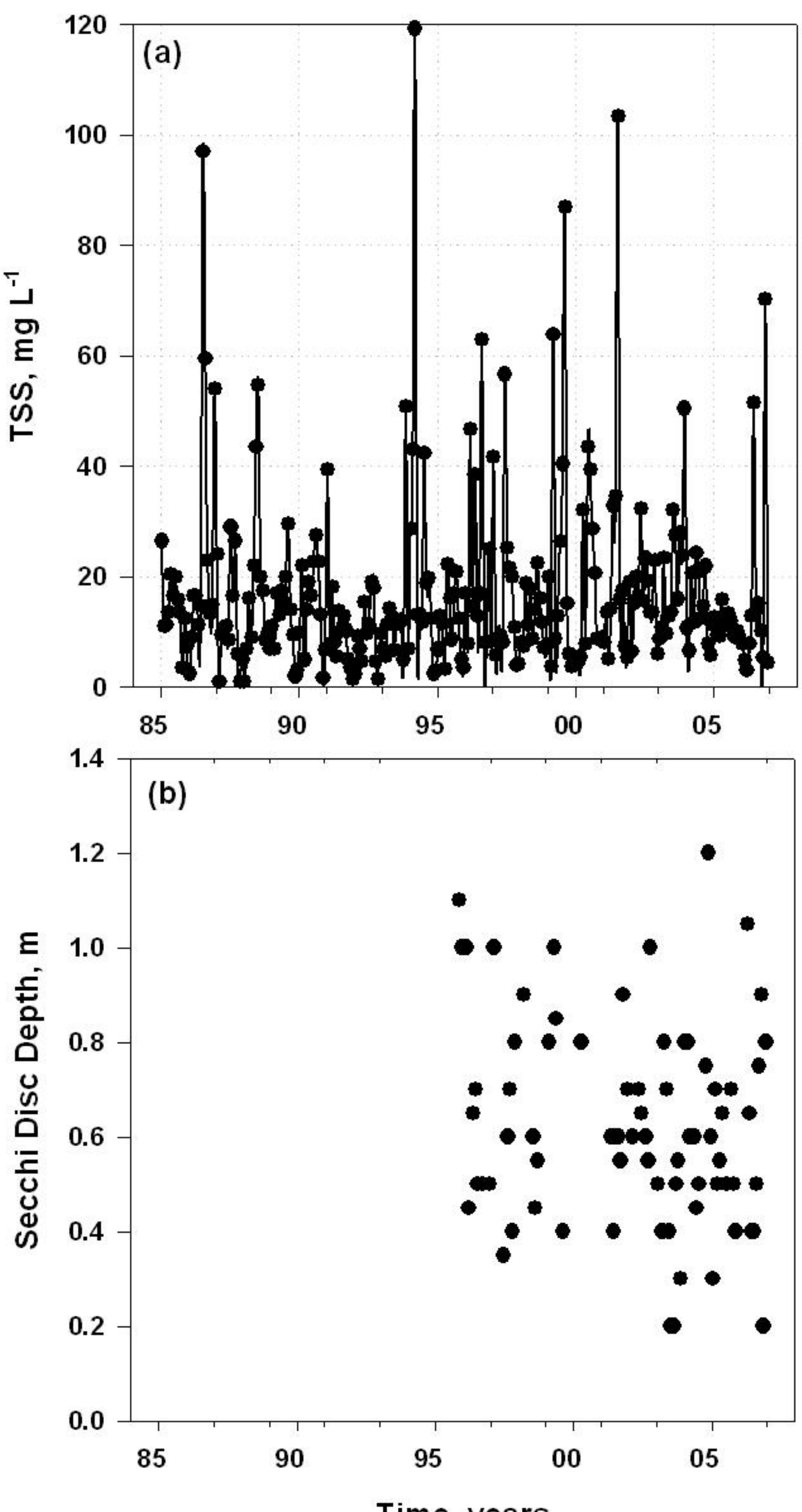
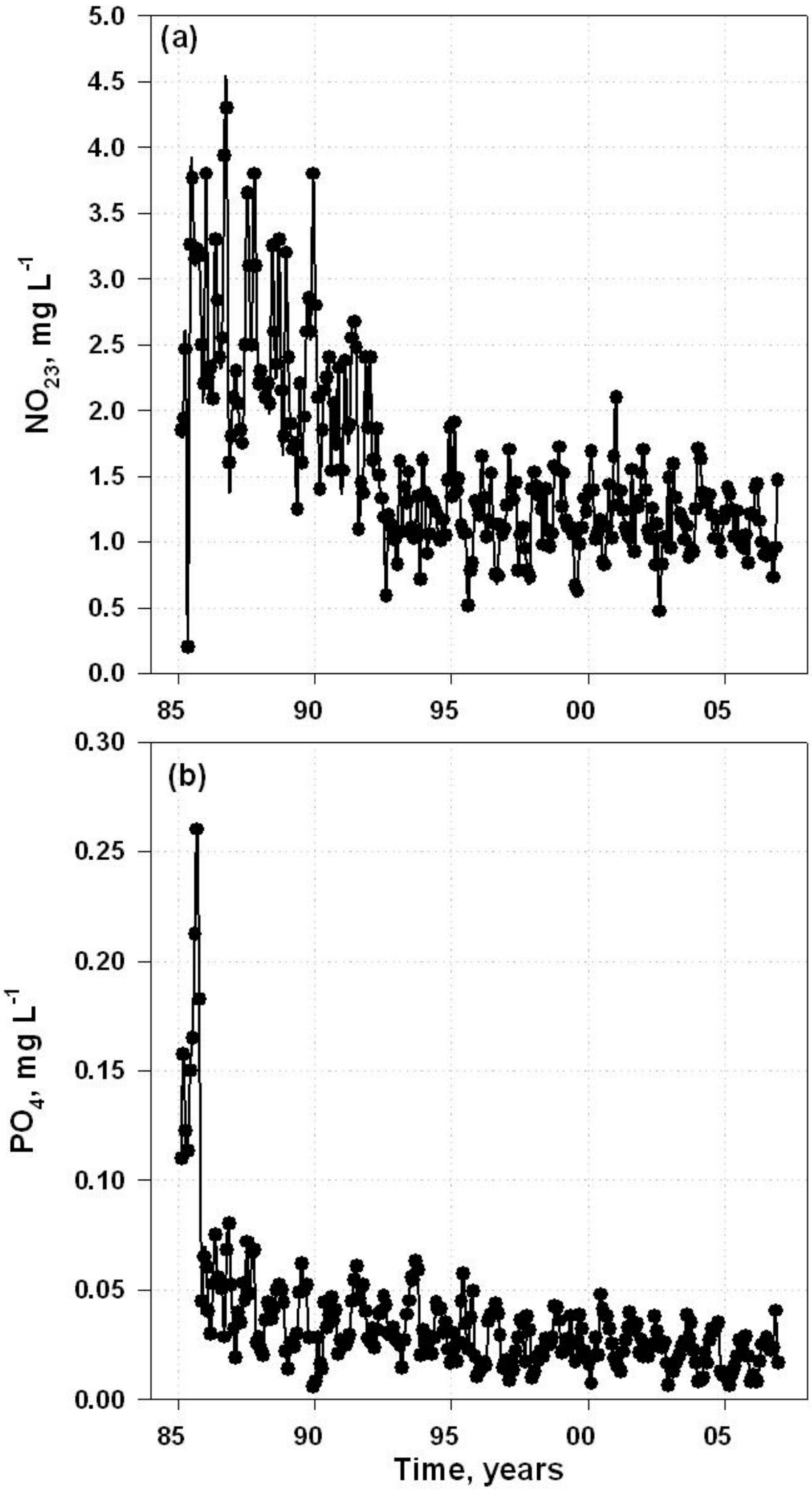


Figure 7-9. Time-series plot of TSS (a) and Secchi Disk depth (b) at station TF 1.3 in the Patuxent River estuary (1985-2007). Data were from the Chesapeake Bay Program data hub (http://www.chesapeakebay.net/).



7-8.5 Nutrient Concentration

Concentrations of NO_{23} exhibited distinct seasonal and inter-annual patterns. NO_{23} concentrations were higher during winter than summer (at both sites). Concentrations during the mid-1980s ranged between 1.5 to 4.2 mg L^{-1} (Fig. 7-10a). Following N-removal at WWTPs concentrations decreased sharply to annual means of about 1.3 mg L^{-1} with a range of 0.5 to 1.7 mg L^{-1}. Particularly low NO_{23} concentrations (0.5-0.7 mg L^{-1}) were observed during drought years (e.g., 1995, 1999, 2002). TN concentrations followed the same seasonal and inter-annual pattern.

Concentration of PO_4 exhibited very sharp decreases after 1985 associated with the P-ban in detergents and P-removal at WWTPs (Fig. 7-10b). Prior to the ban, peak concentrations were about 0.25 mg L^{-1}. For the remainder of the record concentrations continued to decrease slowly with peak concentrations about 0.04-0.07 mg L^{-1}. In sharp contrast to NO_{23} concentration, PO_4 concentrations were maximal during the warm seasons of the year. TP concentration followed a very similar long-term pattern but with occasional large spikes in concentration, likely associated with storm events.

Figure 7-10. Time-series plot of NO_{23} (a) and PO_4 (b) at station TF 1.3 in the Patuxent River estuary (1985 – 2007). Data were from the Chesapeake Bay Program data hub (http://www.chesapeakebay.net).

7-8.6 Chlorophyll-*a* Concentration

Despite the turbid conditions in this zone of the estuary, chlorophyll-*a* concentration was very high at times with peak concentrations often above 50 µg L^{-1}. Chlorophyll-*a* concentration was generally highest during the low flow summer-fall period and lowest during winter-spring periods (Fig. 7-11). Short water residence time in this zone of the estuary during the high flow winter-spring period likely limited biomass accumulation during these seasons. Chlorophyll-*a* concentration tended to be higher at the more down-river site, particularly early in the time-series. At all monitoring sites in the upper estuary, annual average concentration and peak seasonal concentration were reduced after 2002 (except for one very high measurement at TF 1.3 during 2006). This pattern could be the combined result of strong river flows throughout 2003 and sharply reduced N-loads from the Western Branch WWTP beginning in 2004.

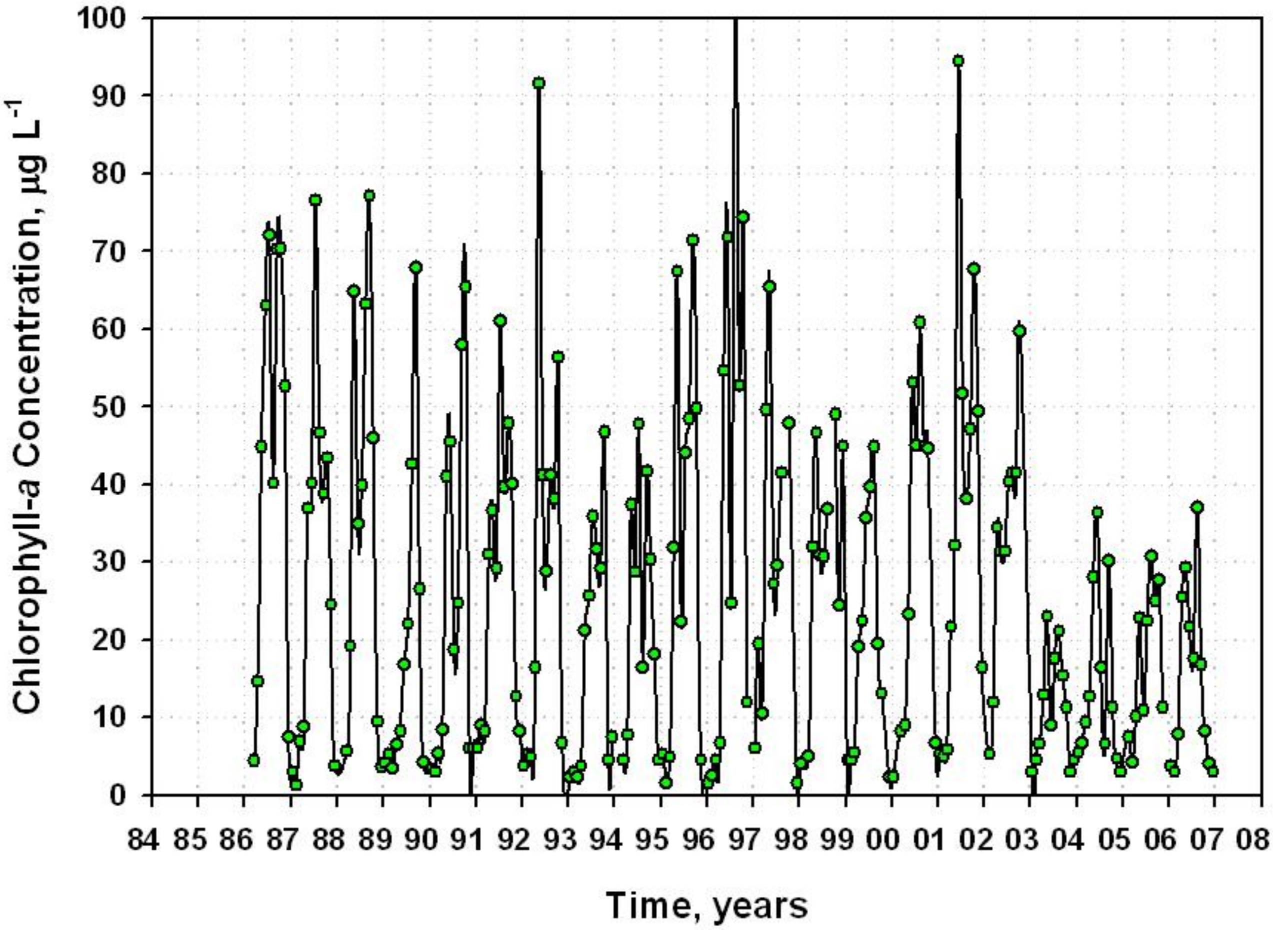


Figure 7-11. Time-series plot of chlorophyll-*a* concentration at station TF 1.5 in the Patuxent River estuary (1985-2007). Data were from the Chesapeake Bay Program data hub (http://www.chesapeakebay.net).

7-9 Long-Term Water Quality Trends

To provide some additional perspective on longer term water quality in the upper estuary we searched older reports (e.g., Mihursky and Boynton 1978, Flemer *et al.* 1971, Bostater *et al.* 1983 and others) for measurements made prior to the beginning of the current monitoring program. In a few cases we found information from the mid-1930s and in other cases to the mid to late 1960s. All early measurements, except those of Cory (1974), were limited. The most common place in the upper estuary where measurements were made was in the vicinity of Lower Marlboro (rkm 55) and those measurements have been organized into two time-series (Figs. 7-12 and 7-13).

All data available for this site were plotted as a function of time, in some cases back to 1936 (Fig. 7-12). Nitrate concentration was apparently very low from the mid-1930s through the early 1960s but increased substantially by 1969-1970 (peak concentrations 0.7 mg L^{-1}[~50 µM]). Maximum concentrations were measured during the late 1980s and early 1990s after which concentrations decreased, probably in response to N-removal at WWTPs. In more recent years peak nitrate concentrations were about 1.05-1.40 mg L^{-1} (75-100 µM) or about half the earlier maximum concentration. Secchi Disk depth and seston concentrations also decreased during the period of record. Why seston concentrations were so high during the 1970 period remains a mystery. Current Secchi Disk depths generally range between 0.2 and 0.5 m, indicative of very turbid water. Dissolved inorganic phosphorus (PO_4) concentrations may or may not have exhibited a more complex pattern. Observations from earlier times (1930s) are very limited. In any case, concentrations at the earliest date were surprisingly high (0.05 mg L^{-1} [1.5 µM]). Concentrations reached maximum levels during 1970 (prior to any P removal at WWTPs) and have since declined. Finally, chlorophyll-*a* concentration ranged from about 4-20 µg L^{-1} during 1964, increased to about 50 µg L^{-1} during 1969-1970 and reached maximum concentrations during the late 1980s and early 1990s (75-100 µg L^{-1}). In more recent years peak concentrations have decreased to about 35 µg L^{-1}.

To focus more on water quality during the SAV growing period we parsed the time-series data set to include only summer (Jun-Sep) measurements (Fig. 7-13). The general temporal patterns were the same as seen in the full data set but maximum values of nitrate, seston and chlorophyll-*a* were lower by varying amounts.

One useful result of organizing these time-series is that we can make a few qualitative comparisons among water quality variables and SAV distribution in the past and during the time of resurgence. The summer concentration of nitrate was very low (<2 µM; <0.03 mg L^{-1}) during the late 1930s and the early 1960s, both times when SAV were reported to be thriving in this portion of the estuary. However, summer concentrations had increased to about 0.14-0.21 mg L^{-1} (10-15 µM0.14-0.21 mg L^{-1}) by 1970 a time when most reports indicate SAV were gone or severely reduced in this portion of the system. In the past decade summer nitrate concentrations have decreased and in some years (e.g., drought years of 1999, 2001) to levels below those found when SAV were declining or gone in 1970. In recent years (with one exception) summer nitrate concentrations have been below 0.28 mg L^{-1} (20 µM). Similarly, chlorophyll-*a* concentrations were low during the early 1960s (<20 µg L^{-1}), increased to maximum values of about 60 µg L^{-1} from the mid-1980s through about 2000 and then decreased to levels comparable to those observed in the mid-1960s. These trends are consistent with improved sewage treatment at WWTPs and with a system becoming less eutrophic and more amenable to SAV community resurgence and growth.

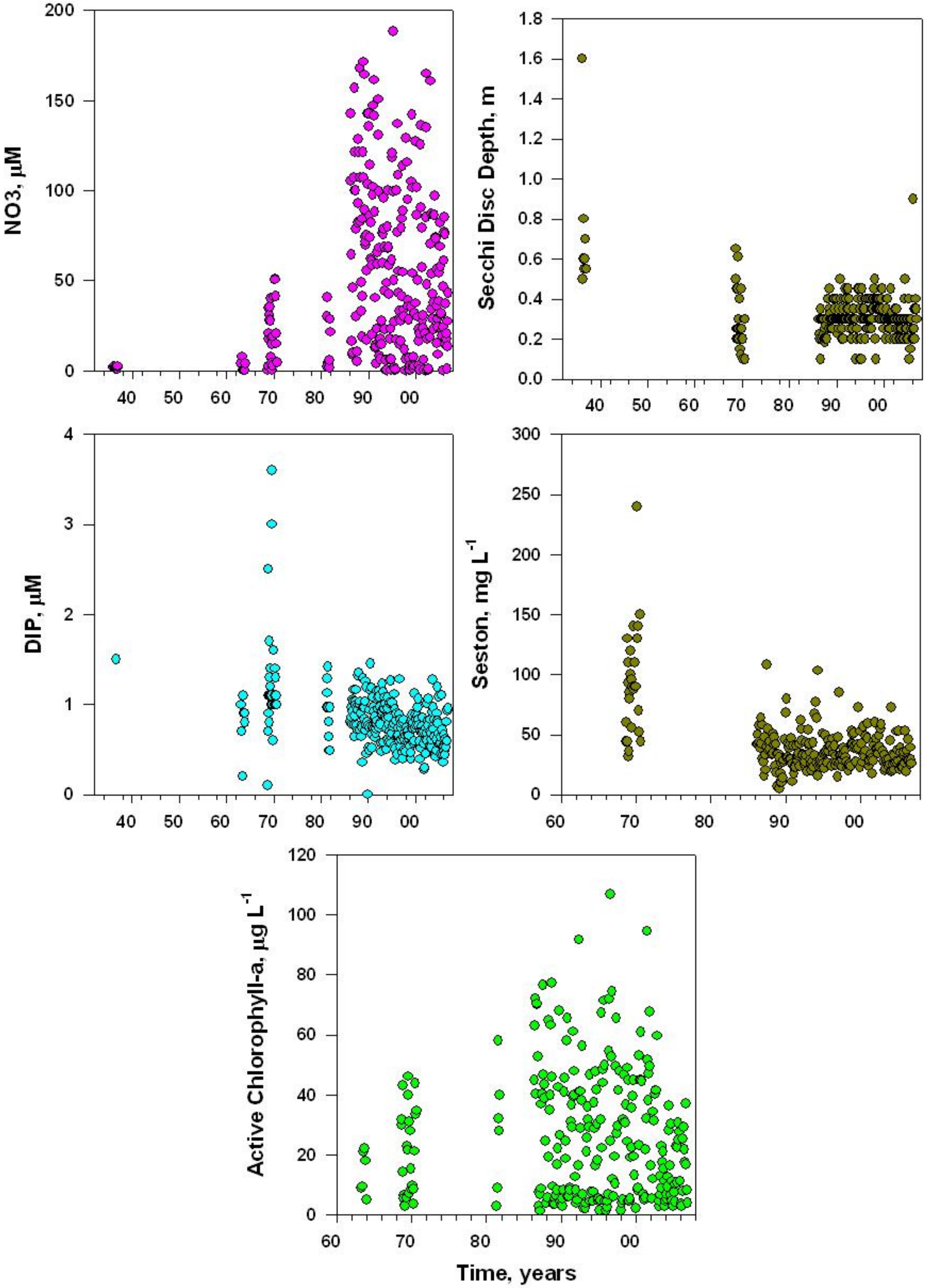


Figure 7-12. Time-series plots of a selection of water quality variables taken from both the historical record and the more recent monitoring program data base (1985-2007). All data were collected in the vicinity of Lower Marlboro (rkm 55). Data from before 1967 were contained in Mihursky and Boynton (1978), data from 1968-1970 are from Flemer *et al.* (1971), and data from the early 1980s are from Bostater *et al.* (1983). Data from all seasons of the year were included in these plots.

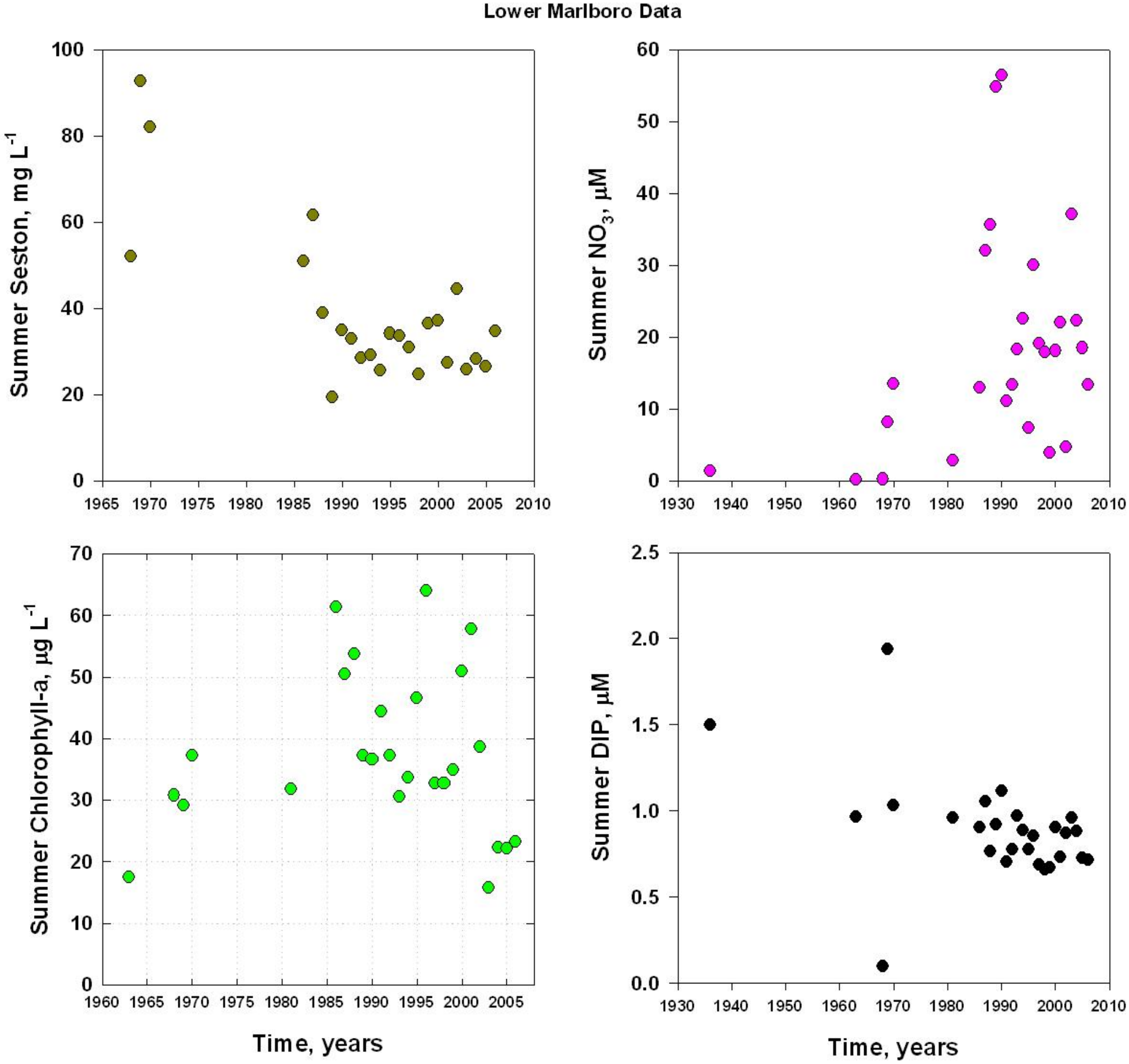


Figure 7-13. Time-series plots of a selection of water quality variables taken from both the historical record and the more recent monitoring program data base (1985-2007). All data were collected in the vicinity of Lower Marlboro (rkm 55). Data from before 1967 were contained in Mihursky and Boynton (1978), data from 1968-1970 are from Flemer *et al.* (1971), and data from the early 1980s are from Bostater *et al.* (1983). Data from just the summer season (Jun-Sep) were included in these plots.

7-10 Nutrient Signals in Estuarine Sediments

Sediment nutrient storage and subsequent release of dissolved nutrients to the water column can play an important role in influencing water and habitat quality. In a small and very eutrophic Chesapeake Bay tributary (Back River) there was considerable (2-4 years) nutrient memory contained in surface sediments of the estuary. In the case of SAV resurgence in the tidal freshwater and oligohaline portions of the Patuxent estuary it is very tempting to limit investigation of causal mechanisms to the major events associated with WWTP upgrades removing N from discharges. However, a similar and abrupt change in point source loads of P from WWTPs, beginning in 1986, also occurred. Immediately following this major management action there was no response from SAVs in any part of the estuary. Is there a lag involved wherein sediments were enriched in P because of several decades of WWTP inputs and the P stored in sediments was gradually released to the water column and this source of enrichment was blocking SAV recovery?

The historical record of water quality and other ecological variables in the Patuxent is far richer than in most other estuaries, but still weak in some respects. With regards to sediment storage of nutrients there is a partial record for particulate phosphorus (PP) and particulate nitrogen (PN) collected from one site from 1978-1980 and from 1985-2003. Fortunately, this record spans the period of SAV absence through SAV resurgence and also includes the period of sharp declines in WWTP discharges of P (1986) and N (1992-1994). However, this site (Buena Vista; BUVA), located about one kilometer north of the MD Route 231 Bridge adjacent to Benedict, MD (rkm 40), is downstream (~10-20 km) of the areas where SAV re-growth has occurred. There are no long-term records for these variables from sites farther upstream.

The sediment PP data collected at BUVA exhibited a clear downward trend in concentration between 1979 and 2003 (Fig. 7-14). Values decreased from about 0.20 % of sediment dry weight

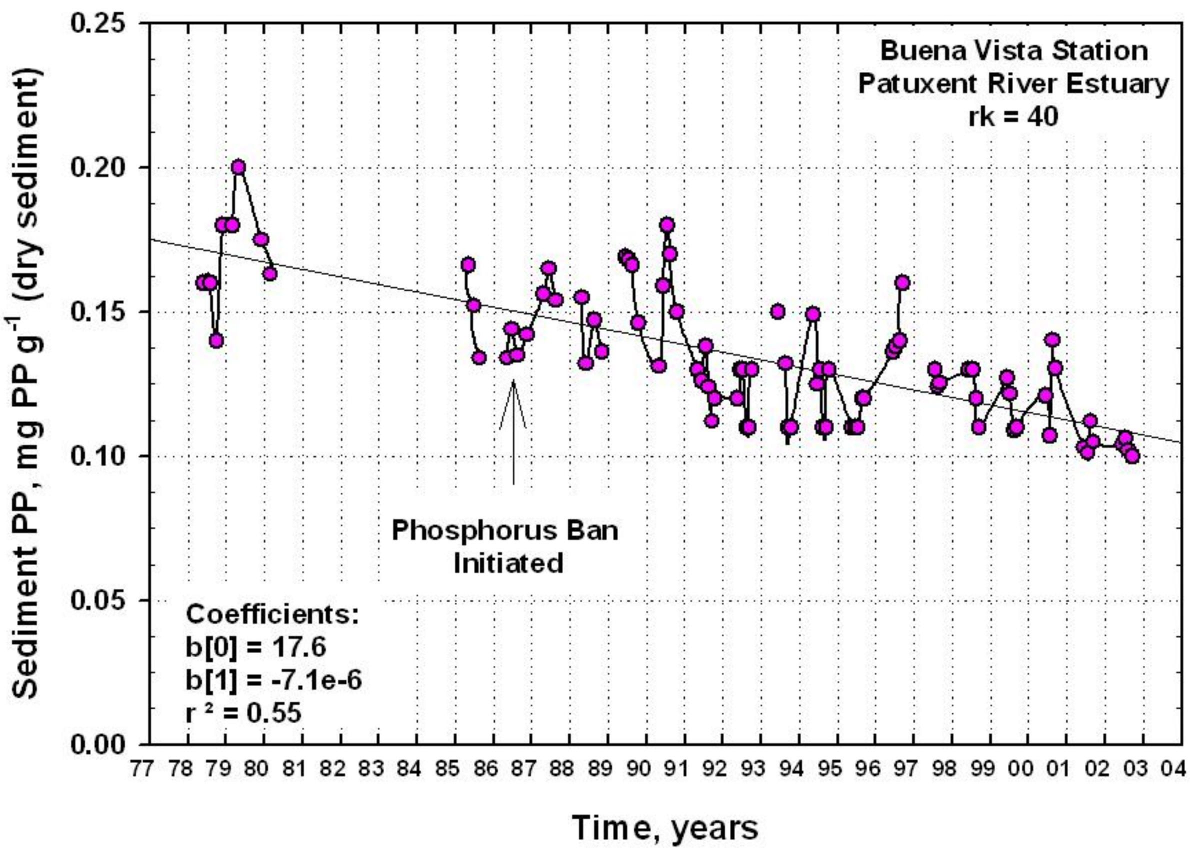


Figure 7-14. A time-series plot of surficial sediment particulate phosphorus (PP) concentration collected at the Buena Vista station (rkm 40) between 1978 and 2002. Data were from Boynton and Bailey (2008).

to about 0.10% P of sediment dry weight during this 25 year period. The decline showed some within year variability (highest values in spring and summer) but this did not obscure the seemingly linear decline with time (r^2 = 0.55). Similar data for sediment PN did not exhibit a decreasing trend at this site. Concentrations remained remarkably constant from 1985-2002 at about 0.35% N of sediment dry weight.

Net dissolved oxygen and nutrient exchanges between estuarine sediments and the water column were monitored in portions of the Patuxent River estuary for a considerable period of time. Initial measurements were made during 1979 and 1980 at a few sites in the Benedict area (rkm 40). Routine measurements were initiated in 1985 at 4-6 locations (4 to 6 measurements per year) in the estuary and these measurements continued until the fall of 2002. Unfortunately, no stations were located in the area of the estuary where SAV resurgence has occurred (TF and OH zones). However, since sediment nutrient releases and oxygen consumption rates are often substantial in shallow estuarine systems and since nutrients appear to be tied to SAV dynamics (e.g., Orth *et al.* 2010) we examined these rates from a site adjacent (downstream) to SAV resurgence areas. Additionally, these sediment flux measurements spanned the period of time when improvements occurred at the WWTPs discharging to this estuary and the magnitude of sediment fluxes were considered with those modifications in mind.

Sediment oxygen demand (SOD) data are shown in Figure 7-15. Fluxes ranged from about 0.5 to 3.8 g O_2 m^{-2} day^{-1}. These are substantial rates when compared to those measured in other estuarine systems (Boynton and Kemp 2008). There do not appear to be consistent sediment flux responses related to patterns of river flow (and associated diffuse source nutrient loads). However, there does appear to be a general trend towards lower SOD rates beginning about 1990 and continuing through 2002. Sediment phosphorus fluxes ranged from zero to about 150 µmoles P m^{-2} hr^{-1} and many of these measurements represent large P releases from sediments (Fig. 7-15). There appeared to be a general decline in sediment P releases between 1993 and 2002. The years 1999-2002 were all low flow years and this may have influenced these rates. Sediment NH_4 fluxes at this site were also substantial, often greater than 500 µmoles N m^{-2} hr^{-1}. A sampling of summer NH_4 fluxes from 48 estuaries yielded a median NH_4 flux of about 100 µmoles N m^{-2} hr^{-1} (Boynton and Kemp 2008). In this case there was not a decreasing trend of sediment N flux (Fig. 7-15). Rather, with a good deal of variability, there appeared to be a general increase beginning in about 1988 and continuing through most of the record. Sediment fluxes of NO_2+NO_3 were directed both out of and into sediments, the former being a good indication of sediment nitrification activity (and oxidized surface sediments) and the latter likely an indication of nitrate uptake leading to denitrification. These fluxes were considerably smaller than those of ammonium (Fig 7-15).

Relationships between sediment fluxes of O_2, N and P and potential causal variables (e.g., nutrient loading rate, sediment N and P characteristics) were not very clear at this site. SOD and P fluxes tended to decrease while N fluxes possibly increased. The sediment P flux decrease is consistent with decreasing P content of sediments but the temporal link between P reductions at WWTPs and decreased sediment flux was lagged by almost a decade and some of the highest sediment P fluxes in the record occurred after WWTP upgrades for P removal.

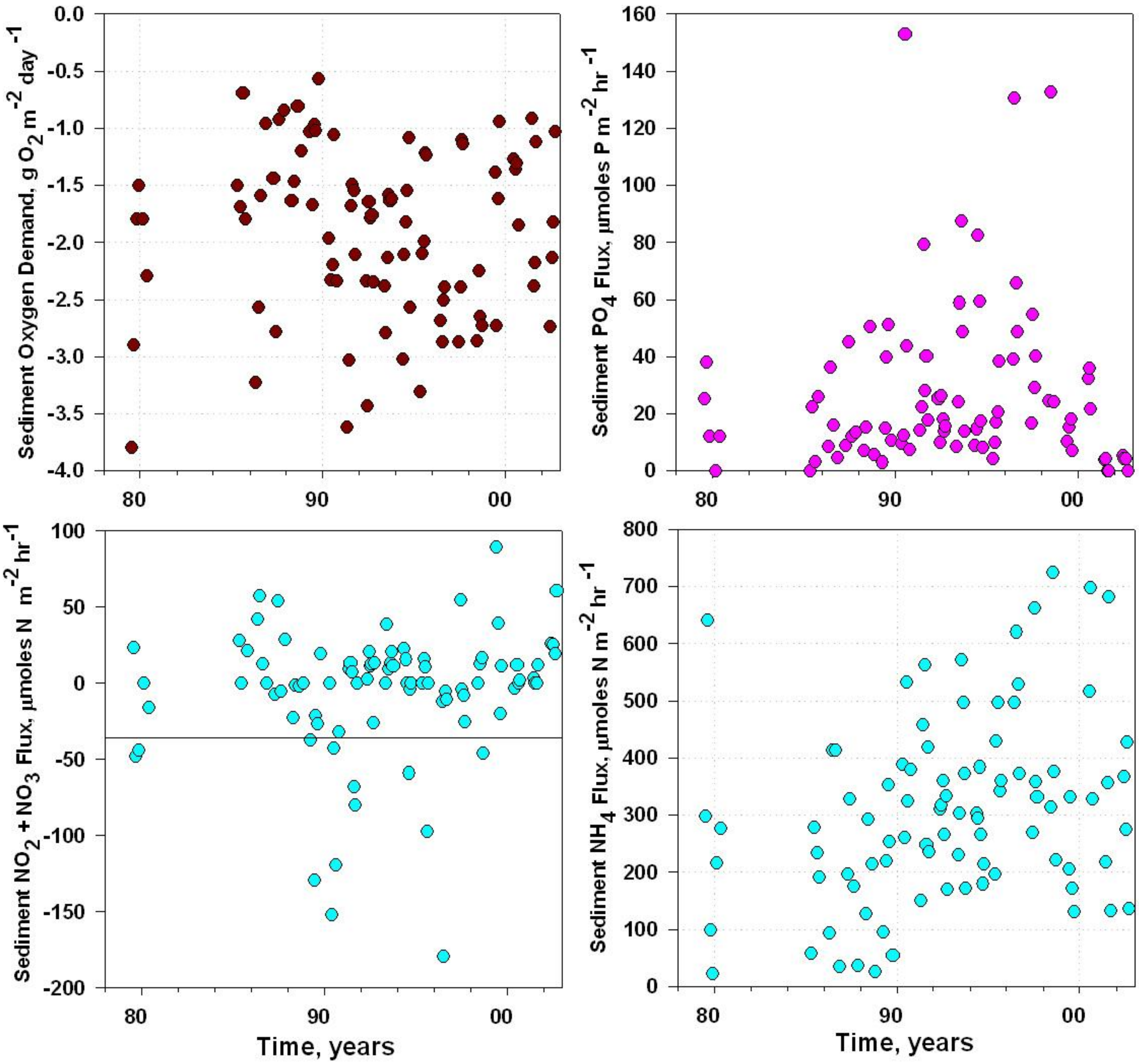


Figure 7-15. Time-series plots of net sediment-water exchange rates collected at Buena Vista (rkm 40) in the Patuxent River estuary. Data were from Boynton and Bailey (2008).

7-11 A Record of Community Metabolism

The construction of a steam electric power plant near Benedict, MD in the early 1960s stimulated a variety of measurements in the estuary near the power plant to assess potential environmental impacts. This plant initially used "once-through" cooling water taken from and then released back to the Patuxent estuary. In later times, cooling towers were added to this facility. Environmental concerns focused on possible detrimental effects on the local estuarine system from entrainment and associated mortality of animal populations and elevated temperature in discharge water. One of the measurements systems deployed at this time was a series of sensors, suspended off the center platform of the MD Route 231 Bridge crossing the Patuxent River at Benedict, MD, several kilometers downstream of the power plant location. This early sensor system measured temperature, conductivity, dissolved oxygen, tidal height, and water clarity. Sensor signals were recorded on large format strip charts. Fortunately, the investigator responsible for this system was very focused on maintaining the system in good order and calibrating sensors on a frequent basis (Cory 1974). The system record began in the fall of 1963 and continued, with few interruptions, through 1969. Portions of these data have been converted to electronic format and are available on a web site (www.gonzo.cbl.umces.edu). Modern sensor systems were re-installed on the Benedict Bridge during 1992, 1996-1998 and 2003-2005. Thus, there is a record of these variables (and additional variables in the most recent deployments) for a considerable period of time. The early data had a measurement interval of one hour while all the measurements after 1991 had measurement intervals of 15 minutes.

The measurements of dissolved oxygen from this data set have recently been used in assessing dissolved oxygen compliance with Chesapeake Bay DO criteria (Boynton *et al.* 2011). In addition, these data can be used to estimate several fundamental ecological rate processes and these include community production and community respiration. We have used an automated version of the open water dissolved oxygen technique developed by Odum and Hoskin (1956) to make these estimates. The basic idea of this technique is to use daytime increases in dissolved oxygen concentration to estimate net community photosynthesis and nighttime decreases in dissolved oxygen to estimate community respiration, with both rates corrected for dissolved oxygen diffusion across the air-water interface.

We recently had the opportunity to use these data and developed estimates of gross community production (Pg*) and community respiration (Rn) for the period of record (Fig. 7-16). Production rates (averages for the April – October period for each year) early in the record were about 3.5 g O_2 m^{-3} day^{-1}. These rates had doubled by the mid-1990s and increased again to between 8-9 g O_2 m^{-3} day^{-1} in the early 2000s. Community respiration also increased between the early and late 1960s, remained about the same in the early 1990s and the increased during the late 1990s and early-2000s. The overall change in community respiration was by about a factor of two, less than that observed for community production, but with a similar long-term pattern.

These rates need to be placed in some context to be more useful. We had the opportunity to compute these rates for many areas of Chesapeake Bay. At the least enriched sites, rates of Pg* of about 5 -6 g O_2 m^{-3} day^{-1} were common. At sites identified as occasionally failing various water quality criteria rates were larger, often in the range of 8-12 g O_2 m^{-3} day^{-1}. At sites experiencing severe eutrophication (e.g., Back River, Corsica River) rates were very large, often exceeding 15 g O_2 m^{-3} day^{-1} and occasionally exceeding 20 g O_2 m^{-3} day^{-1}. We have yet to find rates based on the Chesapeake Bay ConMon data set as low those recorded at the Benedict site in the mid-1960s.

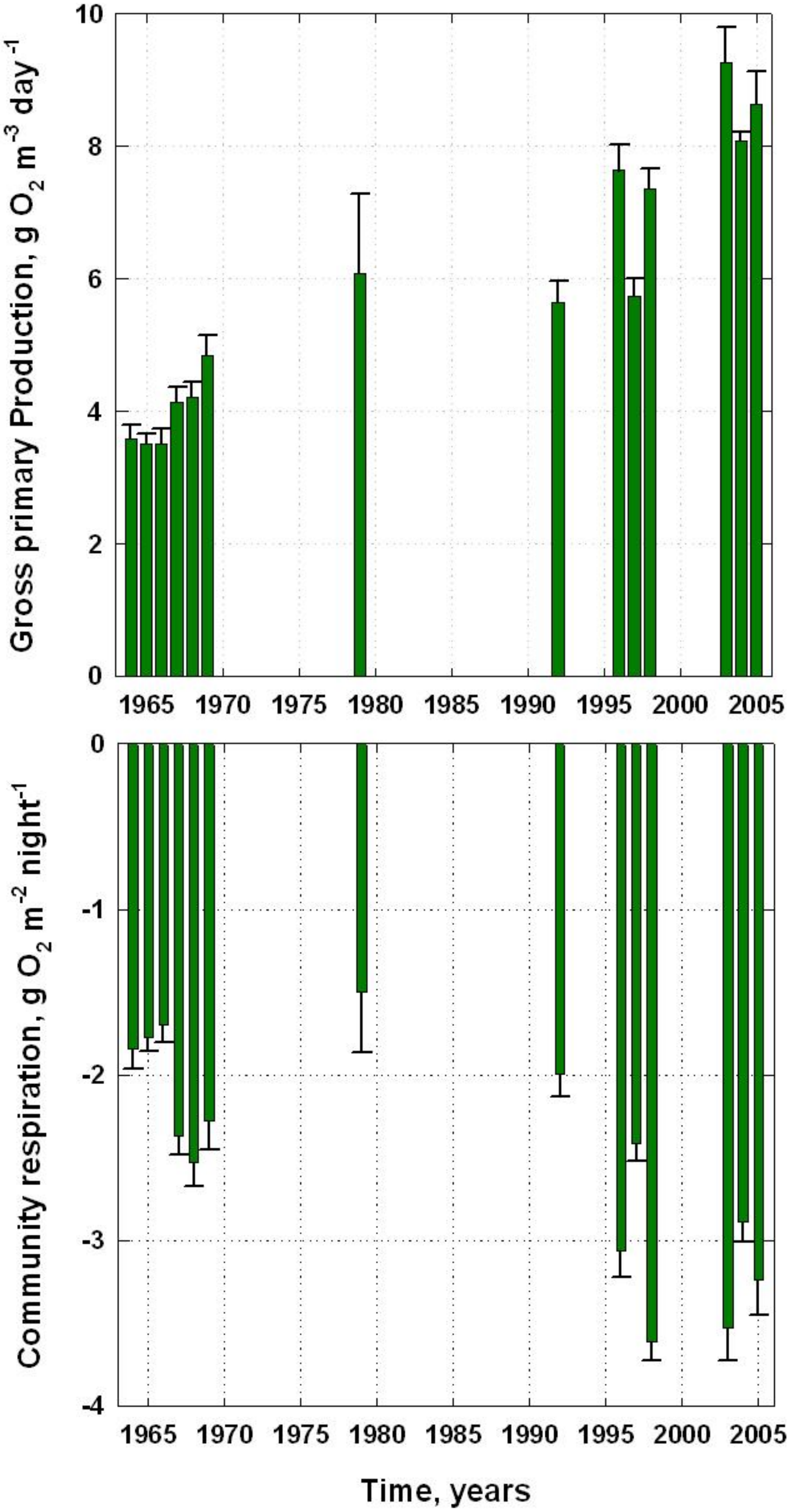


Figure 7-16. Vertical bar chart of annual (April-October) gross primary production (Pg*) and community respiration (Rn) for the period 1964-2005. Data were collected at the Benedict, MD Route 231 bridge.

As expected, the 1960s rates at the Benedict site were low, consistent with low nutrient loading rates as reported by Hagy *et al.* (1998). Rates in the late-1960s began to increase and this is consistent with increasing sewage disposal into the river (Domotor *et al*. 1989) and increased development of the upper watershed. The rates computed for the most recent years (2003-2005) were the highest observed and were consistent with moderately high nutrient loading rates, especially from diffuse sources in the middle portion of this basin (Boynton *et al*. 2008). These time-series plots suggest a slow but continuous increase in metabolic rates associated with increased nutrient loading during a 41 year period. In this interpretation there is no evidence of a threshold response either to increased or decreased nutrient loading rates. Rather, Pg* increased at an average rate of about 1.3 g O_2 m^{-3} day $^{-1}$ per decade. However, it is also obvious that there are many years during this 4 decade time span when there were no measurements available and it is possible that threshold-like responses occurred during those times. For example, nitrogen concentrations likely peaked in this zone of the river during the late-1970s and early-1980s as did chlorophyll-*a* concentrations, a period when only 2 metabolism measurements are available. In fact, the measurements made during 1979 had increased sharply compared to the last of the 1960s measurements. The lower metabolic rates measured during 1992 were associated with a severe drought year and might have been depressed because of a weather-induced reduction in nutrient supply. It is possible, given these observations, that metabolic rates reached maxima during the period between the late-1970s and had started to decline during the late-1990s. It is also interesting to note that there was not a decline in rates following N and P load reductions at WWTPs (1986 for P and 1992-1993 and again in 2004 for N). There are clear signs of this expressed in nutrient concentrations and chlorophyll-*a* farther upstream in the estuary closer to these point sources. However, metabolic rates in the late 1990s were lower than rates during the mid-2000s. Nutrient budgets developed by Boynton *et al.* (2008) indicated that since the initial point source nutrient reductions, diffuse sources have become dominant in this basin and coupled with wet years (e.g., 2003) may have actually increased nutrient loads (at least in wet years) to this portion of the estuary.

7-12 Statistical Modeling of Water Quality and SAV Distributions

The statistical modeling of SAV distributions, and the search for threshold triggers, is divided into four sections including the following: 1) relationships between river flow and water quality at long-term monitoring stations within the SAV re-growth area; 2) examination of similarities or differences among the long-term water quality monitoring sites in the SAV re-growth area of the estuary; 3) statistical modeling of combined tidal freshwater (TF) and oligohaline (OH) SAV focused on the observed threshold responses of these communities; and 4) statistical modeling of the inter-annual variability of the SAV communities following re-growth in the TF and OH zones of this estuary.

7-12.1 River Flow Versus Water Quality

We examined the data set (1985-2007) for relationships between river flow and associated nutrient load and water quality observed at three long-term monitoring sites (TF 1.3, TF 1.5 and TF 1.7) located in the zones of the estuary where SAV re-growth has occurred. River flows, and associated variables, have repeatedly been shown to play a strong role in influencing water quality and habitat conditions in coastal plain estuaries (Boynton and Kemp 2000). In these analyses (correlation analysis with and without lag times; month-scale data for all variables) a few expected relationships emerged. For example, there was an inverse relationship between salinity and flow at station TF 1.7 (most downstream site), positive relationship between flow and nitrate concentration, negative correlation between flow and chlorophyll-*a* and a negative relationship between flow and temperature at the most up-river site (TF 1.3).

However, many of these correlations were weak (but significant at the $p < 0.05$ level) and even more were not significant or had unexplainably long lag times. In some ways this is not surprising because of the substantial distance between the USGS gage at Bowie, MD (rkm 90) where these loads are estimated and the SAV re-growth areas (rkm 50-65). In this zone of the estuary there are extensive tidal marshes that have been shown to have strong effects on nutrient conditions (Fisher *et al.* 2006; Boynton *et al.* 2008) probably more so during the warm seasons of the year when tidal marshes are metabolically very active. In any case, there were not especially strong relationships between river flow and water quality variables in the SAV re-growth zone. As we will show later, it appears that more proximal nutrient sources play a stronger role in influencing SAV distribution than the more distal source represented by the USGS gauge at Bowie, MD.

7-12.2 Water Quality Relationships among sites in the SAV Re-Growth Areas

Because SAV re-growth occurred in two zones of the estuary (TF and OH) we examined water quality conditions among stations in these zones to see if there were strong differences or similarities among water quality variables that could influence SAV communities. We used correlation analysis for the full data set (monthly averages for the period 1985-2007) and compared data from TF 1.3 with data from TF 1.4, TF 1.5, TF 1.6 and TF 1.7. Variables included water temperature, salinity, Secchi Disk depth, and chlorophyll-*a*, nitrate plus nitrite, ammonium, dissolved inorganic phosphorus, TN, TP and TSS concentrations. In general there was very strong correspondence of temperature, dissolved nitrogen and phosphorus and TN, TP and TSS between station TF 1.3 and station TF 1.4. Maximum correlation coefficients indicated no temporal lags although there were significant correlations with 1-3 month lags for many water quality variables. Correlation coefficients for N and P variables all exceeded 0.95 in this adjacent station comparison. As the distance between TF 1.3 and more down-river sites increased, correlation coefficients also decreased but temperature and N and P variables still remained highly correlated ($r > 0.5$; significant at $p = 0.05$). Correlations between TSS at TF 1.3 and more down-river sites were not significant. This likely resulted because the estuarine turbidity maximum occurs within this span of sampling sites (and shifts within this span of sites seasonally and inter-annually) and thus TSS concentrations were quite variable.

There were no among site significant correlations between chlorophyll-*a*, Secchi Disk depth and salinity. Because most of these sites had salinity of zero (or near-zero) this is not surprising. Secchi Disk depths were low at all sites so the lack of significant correlations for this variable is also not surprising. The lack of significant chlorophyll-*a* correlation among these sites was not expected. Our impression was that chlorophyll-*a* concentration was low during high flow seasons (winter-spring) and higher during the low flow seasons (summer-fall). Apparently, there was sufficient variability in chlorophyll-*a* concentration to weaken this qualitative pattern to the point of statistical non-significance.

Finally, there were some weak suggestions of lag times between some variables at the extreme ends of this span of sampling sites (i.e., TF 1.3 versus TF 1.7). For example, nitrate plus nitrite, ammonium and dissolved inorganic phosphorus concentrations at TF 1.7 were highly correlated with those concentrations at TF 1.3 but with 1 to 3 month lags. It is not clear if this was a result of simple water transport lag times or a more complex set of processes involving transport and nutrient re-cycling along the axis of the estuary. Despite the emergence of some temporal differences in water quality conditions at the extreme ends of the station span, the general picture that emerges is one of similar, rather than very different, conditions in the TF and OH zones of the estuary.

7-12.3 SAV Threshold Responses

This section focuses on developing statistical analyses that point toward threshold levels of variables likely controlling SAV re-growth in the TF and OH areas of the estuary.

We began this analysis by examining correlations (with and without time lags) of water quality variables at all stations within the TF and OH zones with SAV distribution. We chose to examine water quality during the May-July period, rather than other periods of the year, because this time period captures the period of rapid SAV growth and maximum spatial coverage. At the up-river two stations (TF 1.3 and TF 1.4) there were significant negative correlations of SAV coverage with water temperature (no lag) but this correlation was absent at all other sites. At all sites there were significant negative correlations of SAV coverage with nutrients (both N and P) and these were strongest at the upper river sites (TF 1.3-TF 1.5) but still significant at the lower river site (TF 1.5-TF 1.7), often with a lag of one or more months at the lower river sites. Correlations were generally stronger for N than for P.

We next examined input data (loads of N and P) from the USGS gage station at Bowie, MD and from the much closer Western Branch WWTP discharge located at the head of the SAV re-growth zone of the estuary (rkm 72). In the case of loads from the Patuxent River, weak but significant negative correlations were found for nitrate plus nitrite, dissolved inorganic phosphorus and TN. Correlations for TP, TSS and flow were all non-significant with or without lags. However, significant correlations all involved a lag between SAV re-growth and declines in loads of three years (i.e., loads decreased three years before SAV re-growth). Correlation between N discharge (average for the May-July period) from the western Branch WWTP and SAV coverage were very strong (r ~0.8) and correlations with P discharge non-significant. The significant correlations with N discharge were maximal with a two year lag but still very significant with one and zero year lags. The strongest correlation with SAV coverage found in this portion of the analysis was with nitrate plus nitrite discharge from Western Branch WWTP with a two year lag.

These results suggested that proximal (i.e., Western Branch WWTP) rather than distal (i.e., USGS gage at Bowie, MD) nutrient sources were more important, that nitrate plus nitrite rather than TN or P compounds were also more important and that there was a time lag of several years between load reduction and substantial SAV response as expressed as increased areal coverage. Given these results, we developed several multiple linear regression models using estimates of annual SAV coverage as the dependent variable and concentrations of nitrate plus nitrite from long-term monitoring sites and nitrate plus nitrite loads from the Western Branch WWTP and from the USGS gage at Bowie, MD. Results from three of these analyses are shown in Figure 7-17 as conventional and as natural log transformed scatter plots. The statistical models used May-July average nitrate plus nitrate concentration or load with 0, 1 and 2 year lag times and no other variables. River flow did add to the amount of variance explained by the model but the addition was small and we opted for using the simplest model that explained most of the variance. The results were similar for the models using nitrate plus nitrite concentration in the river and Western Branch WWTP nitrate plus nitrite load (r^2 = 0.74 – 0.78).

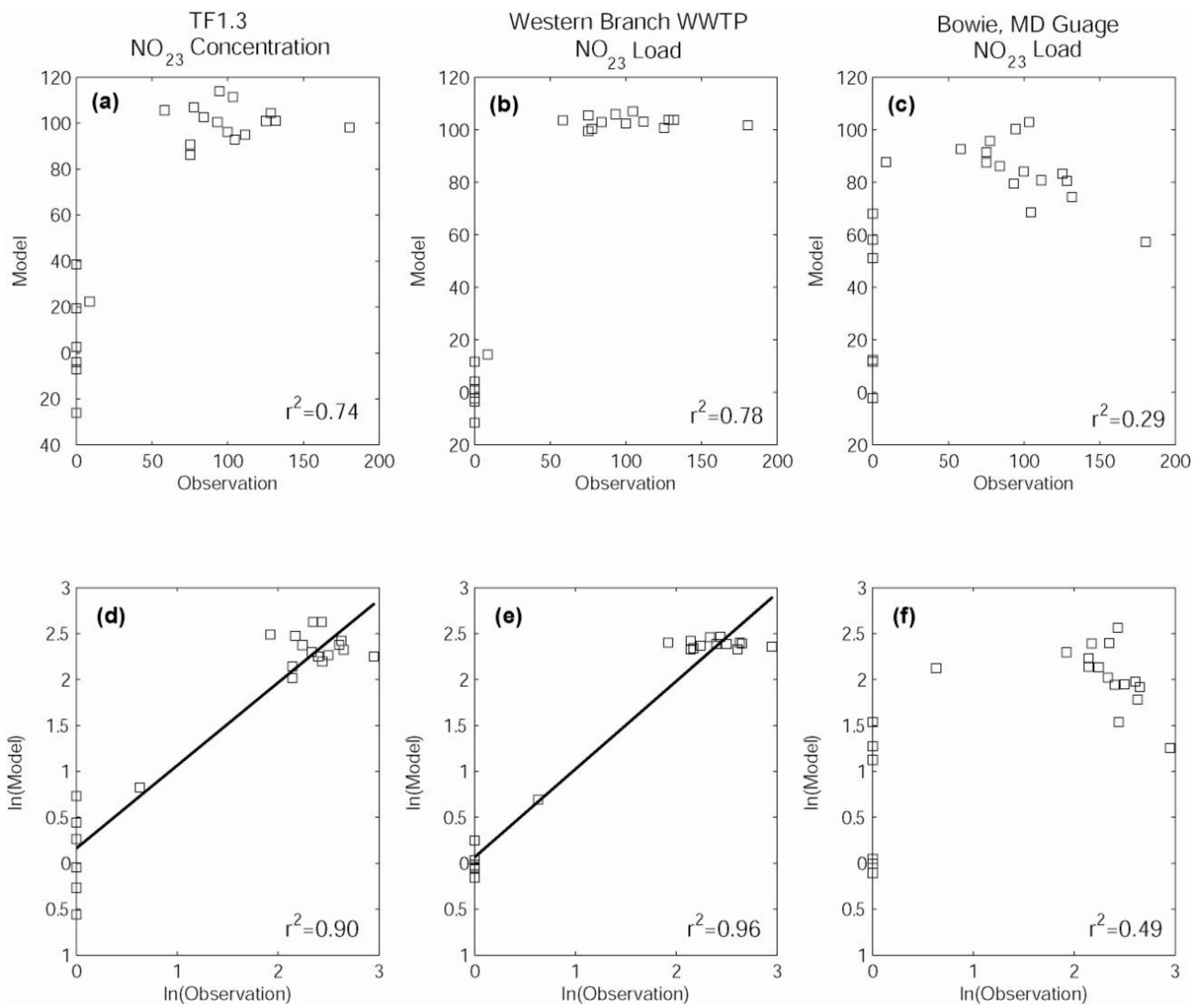


Figure 7-17. Multiple scatter plots of observed SAV coverage versus modeled SAV coverage for the combined TF and OH zones of the Patuxent River estuary. Each multiple regression model used either NO_{23} concentration in the river (from Station TF 1.3) or NO_{23} load from the Western Branch WWTP or from the Bowie, MD USGS gage. Each model included zero, 1 and 2 year lag times of the NO_{23} variable (i.e., concentration or load). Concentration and loads were averaged for the May-July period of each year. Data from 1985 – 2009 were used in these analyses.

However, the model using the nitrate plus nitrite load estimated at the USGS gage at Bowie, MD was only marginally significant, again suggesting that local loads or concentration were more influential than distal loads. Natural log transformation of model results very substantially increased r^2 values, especially for the model using Western Branch WWTP discharge of nitrate plus nitrite (r^2 = 0.96). Thus, we have two relatively simple models that largely capture the substantial jump in SAV coverage in this portion of the Patuxent estuary.

If, indeed, nitrate plus nitrite load or concentration was critical to SAV re-growth, it is useful to have an estimate of the concentration or load at which the change took place. We plotted measured loads and concentration versus SAV coverage in the TF and OH zones of the estuary for the period of record (Fig. 7-18 and Fig. 7-19). In the case of loads, SAV were not present when loads from the Western Branch WWTP were greater than 90-100 kg N day^{-1}. SAV began re-growth during 1993, a year of especially low loads (~50 kg N day^{-1}). Since that time loads have not exceeded 90 kg N day^{-1}. However, there does not appear to be any further temporal

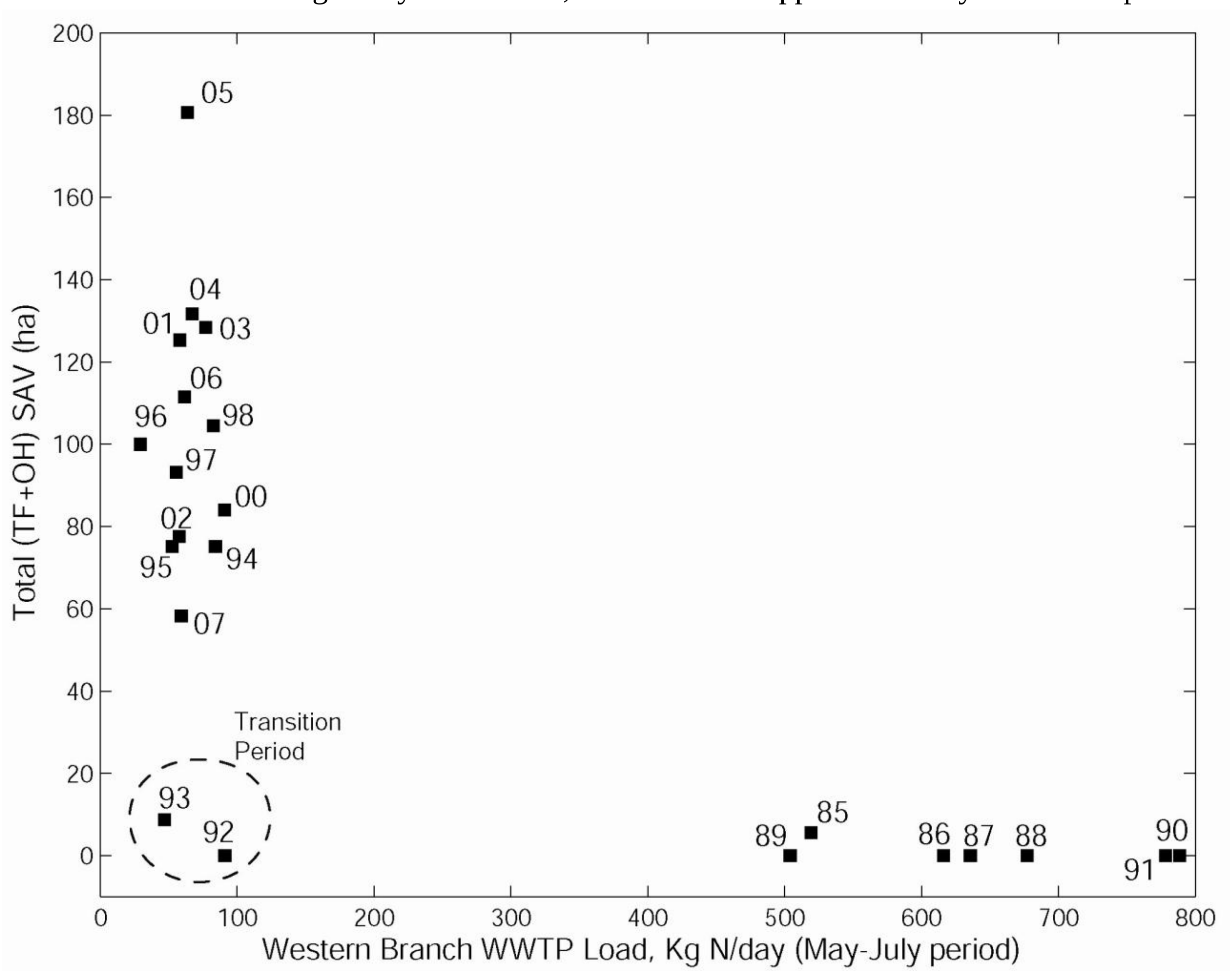


Figure 7-18. Scatter plot of Western Branch WWTP NO_{23} load versus annual SAV coverage (ha) in the combined TF and OH portion of the estuary. Loads were averaged for the May-July portion of each year.

relationship between proximal loads and SAV coverage. Since 1994 SAV coverage has ranged from a low of about 60 ha to a high of about 180 ha. Once the threshold load was reached, something else was apparently regulating SAV coverage. A similar story emerges when nitrate plus nitrite concentration is considered. There was some small amount of SAV coverage observed during 1993 when nitrate plus nitrite concentration was 1.3 mg L^{-1}. At about the same concentration SAV coverage expanded in 1994 and reached maximum coverage during 2005. As in the load model, after a critical threshold was reached, no further relationship between SAV coverage and nitrate plus nitrite concentration was evident.

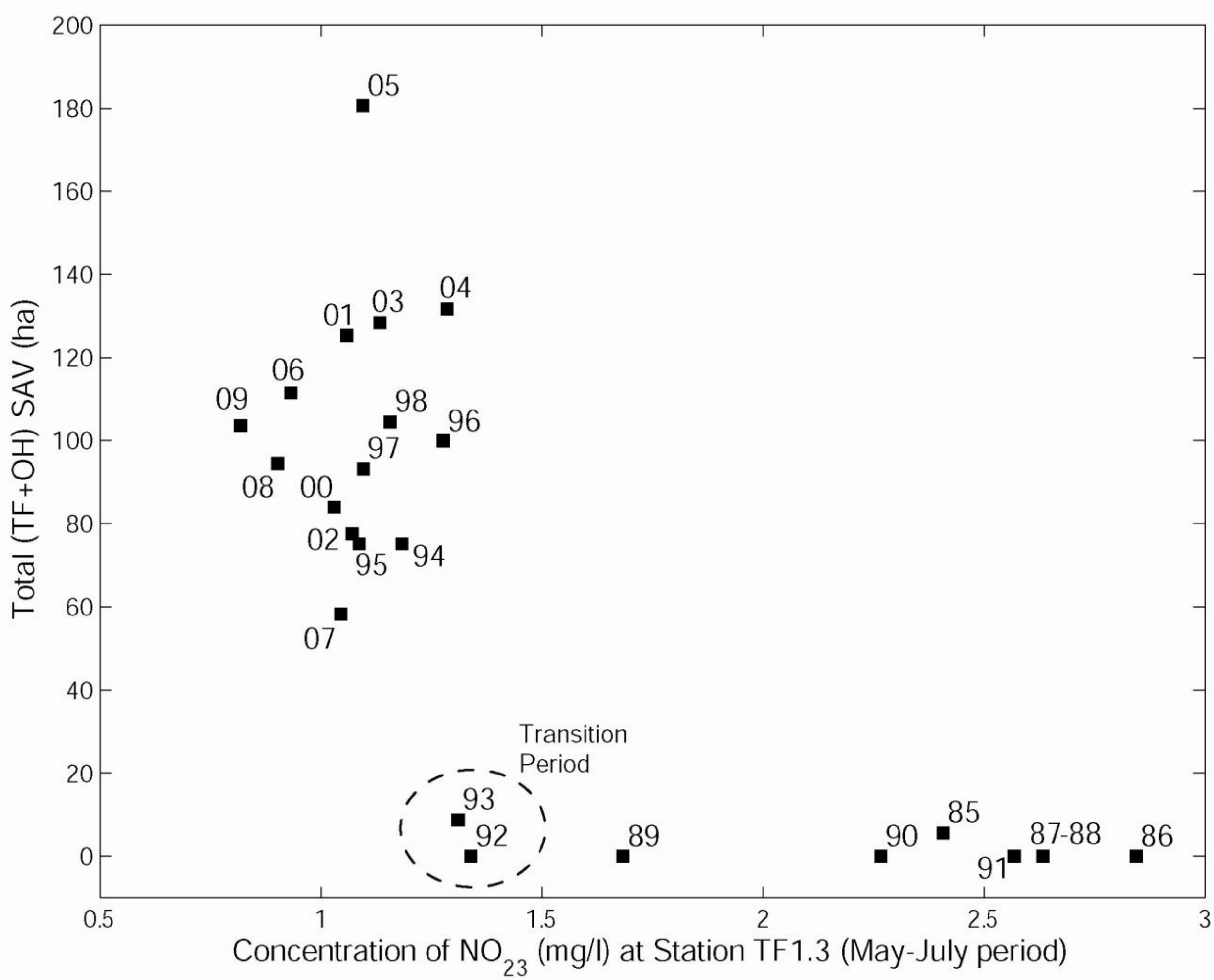


Figure 7-19. Scatter plot of NO_{23} concentration at Station TF 1.3 versus annual SAV coverage (ha) in the combined TF and OH portion of the estuary. Concentration of NO_{23} were averaged for the May – July portion of each year.

7-12.4 Inter-Annual Variability in the Post-Recovery Period (1994-2007)

To model inter-annual variability in SAV coverage during the post-recovery period we used the same general approach as described above. We first examined correlations (with and without lags) between water quality variables and loads using sequentially averaged monthly values (i.e., Jan-Mar, Feb-Apr, Mar-May) and SAV coverage separately for the TF and OH zones. These results indicated that the TF and OH SAV zones needed to be treated separately because a different suite of variables were indicated as important in these different zones.

In the TF zone the strongest multiple linear regression model used water quality (from Station TF 1.3) and load data from the May-June period of each year. Variables useful for accounting for inter-annual variability included Western Branch WWTP loads of dissolved inorganic phosphorus, nitrate plus nitrite, and water column dissolved inorganic phosphorus and water temperature (Station TF 1.3). A plot of observed versus modeled results (ln transformed) indicates a linear fit with an r^2 value of 0.71 (Fig. 20a). While this is a “messy model” (i.e., one having quite a few variables) it also has the important characteristic of being free from lags which seems an important feature of a model addressing inter-annual scales of variability.

The multiple linear regression model accounting for inter-annual variability in SAV coverage in the OH section of the river was both simple and explained much of the inter-annual variability. In this case just two independent variables (water temperature and salinity; zero lag times) computed for the May – July period explained 81% of the variability after natural log transformation of the data (Fig. 20b). We were unable to find other combinations of variables that would further account for inter-annual variability of SAV coverage in the TF and OH zones of the river and still be consistent with our current understanding of SAV dynamics.

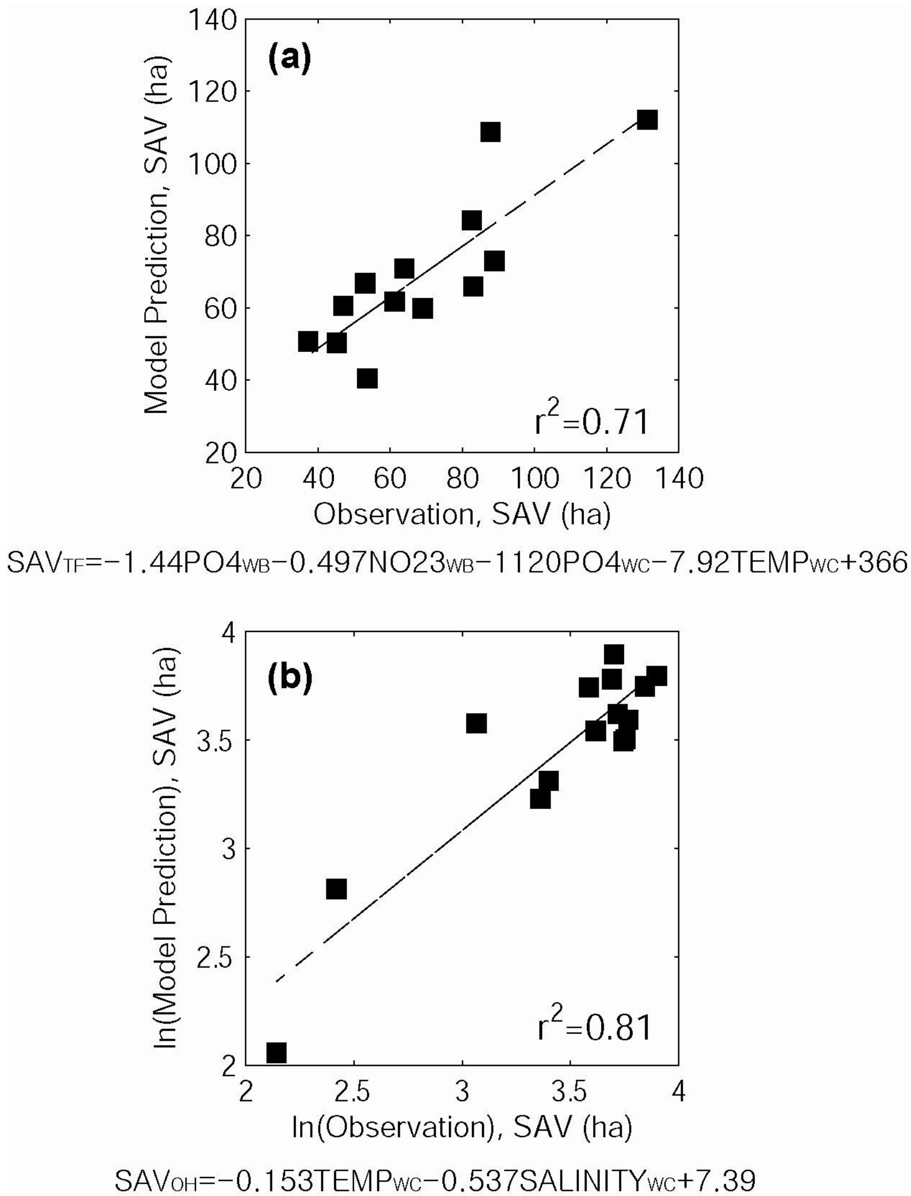


Figure 7-20. Scatter plots of observed versus modeled SAV coverage in the TF (a) and OH (b) zones of the estuary for the post-recovery period (1994-2007). Regression equations with coefficients are provided below each panel. The following abbreviations were used for the TF regression model: $PO4_{WB}$ = PO_4 load (May-July average) from the Western Branch WWTP; $NO23_{WB}$ = NO_{23} load (May-July average) from the Western Branch WWTP; $TEMP_{WC}$ = water temperature (May-July average) at Station TF 1.3. The following abbreviations were used for the OH regression model: $TEMP_{WC}$ = water temperature at Station TF 1.7 (May-July average); $SALINITY_{WC}$ = salinity at Station TF 1.7 (May-July average).

7-13 Lessons for Management

Several lessons for environmental managers emerged from this analysis. First, WWTPs were compelled to reduce loads of N and P in the mid-1980s and early 1990s, respectively (D'Elia *et al.* 2003). There was no response in SAV growth to the P load reductions. Several years after N loads were reduced there was a very substantial re-growth of SAV in the TF and OH regions of the estuary. This came as a surprise to many because this zone of the estuary remained very turbid and nutrient loads and concentrations remained relatively high. What has not occurred to any significant extent is making the public fully aware of the success, limited though it might be, of this SAV recovery. We should celebrate successes!

Second, the SAV recovery story in the TF and OH areas of the Patuxent appears to be based on nitrogen rather than phosphorus. In some ways this runs counter to our current understanding of nutrient limitation in these systems. As a general rule, P tends to be more limiting in the freshwater zones and N more limiting in the saltier zones (Fisher *et al.* 1999). In this case, the threshold for SAV recovery seemed to depend on lowering nitrate plus nitrite concentration or load or both and lowering these variables proximal to SAV re-growth areas. There is clearly a need for improved understanding of the roles of N and P in these systems. In addition, there is some conflict within our analytical results. In the TF zone, SAV coverage after the N threshold was passed seemed to depend on a number of variables including both loads and concentration of N and P as well as water temperature. However, this was the only place where P seemed to play an important role in any of our analyses.

Third, we learned at the Thresholds workshop (Kemp and Goldman 2008) that some European colleagues found TF and OH regions of estuaries to be most responsive to management actions (Jeppesen 2008) while mesohaline and polyhaline regions appeared to be less responsive (Conley 2008). So, do TF and OH zones respond more rapidly to nutrient reductions than other estuarine salinity zones even though they often remain nutrient rich after nutrient reductions are accomplished? We have no general answer to this question. However, we do have several observations that may provide some local answers and suggest directions for future research. In the Patuxent, SAV recovered despite continued conditions of high turbidity and high nutrient concentrations. In fact, the nitrate plus nitrite concentration at the threshold was about 1.3 mg N L^{-1} (93 µM); this is a huge concentration in estuarine waters. However, in this system SAV grow in both the TF and OH zones in very shallow (<0.5 m at high tide) water and most species are canopy forming types. It may be the physical morphology of the upper Patuxent (very shallow shoal areas) favors SAV growth even when the water remains turbid. We also had a limited opportunity to evaluate epiphytic accumulation rates along the salinity, turbidity and nutrient gradients of this estuary. Epiphytic growth on SAV leaves can constitute a severe light stress on these plants. Mylar strips suspended in the water column were the basic tool used in this limited exercise to estimate epiphyte growth rates. At the same light levels epiphyte accumulation rates were much higher in the salty portions of the estuary than in the TF zone, despite the fact that nutrient concentrations were far higher in the TF zone (Stankelis, pers. comm.). The reason for these differences remains unclear but slower epiphytic fouling of plant leaves would be favorable for SAV growth. Finally, the nutrient concentrations used in this analysis were based on samples collected in the main channel of the river and not from the waters surrounding SAV plants growing in the shallows. It may be that a combination of sediment uptake of nitrate (denitrification) and SAV, benthic diatom and epiphyte utilization of nitrate reduced concentrations in these waters and perhaps reduced concentrations to levels more commonly associated with SAV communities. Gruber and Kemp (2010) reported multiple positive feedback effects of a canopy-forming SAV community including improvement of light available for SAV growth. At this point we have no direct measurements of any dissolved inorganic nutrients or light conditions from within these communities. However, detailed aerial photographs taken as part of the SAV monitoring program (http://web.vims.edu/bio/sav/) show very clear water associated with the SAV beds in some portions of the river indicating that the sediment capturing and water clarification roles played by SAV communities are active here. These communities may also reduce nutrient concentrations to levels associated with healthy SAV growth.

7-14 Final Summary Comments and Speculation

Based on the paleoecological record, and a few historical observations, it is clear that SAV were an important component of this system for many years pre-European settlement and for about 330 years after establishment of the first Maryland colony at St Mary's City in 1634.

Despite the fact that SAV disappearance is a relatively recent event we are less certain about the fine-scale temporal sequence of SAV demise in this estuary. There is strong evidence that SAV were abundant in all salinity regions during the 1950s and into the early to mid-1960s. It is less clear when SAV left the TF and OH zones but it seems like they were in retreat during the late-1960s and largely gone by about 1970. SAV in the mesohaline region seemed to have disappeared about the same time, with just isolated patches remaining into the early-1970s. This decline in SAV coverage was not limited to the Patuxent but occurred throughout the Bay area, more severely in some places than in others (Kemp *et al.* 2005). In the Patuxent the decline in SAV coverage was correlated with increases in developed land, especially in the upper basin, increased discharge of sewage from WWTPs located in the upper and middle portions of the basin. The use of agricultural fertilizers also increased during this time period. All of these activities tended to increase the loading rate of nutrients, sediments and various contaminants to the estuary. Loss of SAV and increased volumes of hypoxic water during warm periods of the year were two results of several decades of greatly increased and only mildly regulated human activity in the watershed.

There has been some untold number of restoration projects conducted in the Patuxent River basin by a variety of large and small organizations, Federal and State agencies, County governments, farmers and individual home and business owners. It's fair to say that the net effect of all these actions is generally unknown, as is the effectiveness of most of these restoration activities. However, there have been three management actions in the Patuxent basin that have been carefully measured and which seem to play a central role in the re-growth of SAV communities in the TF and OH regions of the estuary. All three actions involved up-grading treatment levels associated with WWTP operations. The first occurred in about 1986 and involved P removal at all 9 major WWTPs in the Patuxent basin (along with a ban on P in laundry detergents), the second involved warm season reduction (May-October) of N from all 9 major WWTPs in the basin and the third involved the initiation of Enhanced Nutrient Removal (ENR) at the largest WWTP in the basin. This final action involved a further lowering of N concentration in WWTP discharge and maintenance of these low concentrations (3-4 mg N L^{-1}) throughout the year. Reductions such as these are planned for the remaining 8 major WWTPs in the basin and up-grades are expected to be completed during the period 2012-2014.

It is possible to place these major management actions in some perspective. Recently, Boynton *et al.* (2008) completed a nutrient budget for the Patuxent River estuary and nutrient sources were consider based on location (above or below the fall line) and by type (diffuse or point). Prior to

WWTP upgrades N loads at the fall line (Bowie, MD) averaged about 1600 kg N day^{-1} while after WWTP upgrades N loads were about 740 kg N day^{-1}. Below fall line WWTP N loads were 740 and 450 kg N day^{-1} before and after upgrades. Thus, total WWTP loads before and after improved N removal were about 2400 and 1200 kg N day^{-1}, respectively. Even the lower loading rate is well above the point source N loading rate reported by Domotor *et al.* (1989) for the early 1970s (~770 kg N day^{-1}) when SAV were gone from the TF and OH region of the estuary. However, statistical analyses presented earlier point toward local (below fall line in this case) N sources being more important than distal (above fall line) in regulating SAV coverage. Reported N loads from the Western Branch WWTP during SAV recovery were about 450 kg N day^{-1}, much less than the estimated load during 1967 when SAV were very stressed and disappearing from this zone of the estuary. However, summer N concentration (NO_{23}) in the SAV re-growth area still exceed those measured prior to and during the period of SAV decline. It remains unclear why SAV declined at lower nitrate concentrations and recovered at higher nitrate concentrations.

The upper Patuxent SAV recovery has been associated with reductions in WWTP discharges. A similar story has emerged for SAV in the upper (TF) portions of the Potomac River estuary (Ruhl and Rybicki 2010). A central feature of these SAV responses to WWTP discharge reductions is that reductions were limited to the warm seasons of the year in both systems, until recently. Nitrogen inputs to both systems are dominated by diffuse sources on an annual basis and these have not changed much during the SAV re-growth period. Despite this, SAV have recovered in the TF and OH zones. However, diffuse sources are much higher during the winter and spring than during the summer and fall. During summer months point sources become relatively more important and it is during this time that N discharge reductions at WWTP impact seasonal nutrient loads. SAV in the upper Patuxent flourish during the late-spring through summer, begin to die during late summer and fall and are not present as above-ground structures during winter and early spring. Thus, it may be that the large diffuse nutrient loads associated with winter-spring conditions simply "pass through" the TF and OH region of the Patuxent and have little impact on SAV habitat conditions. Furthermore, sharp reductions in point source loads, especially local point source loads during spring-fall, apparently reached a level amenable for SAV re-growth.

Seasonal variation in the magnitude of nutrient loads is also consistent with the observations that annual-scale loads to the lower estuary have not changed much and that SAV in the lower estuary have not recovered. It is possible that the large winter-spring nutrient load rapidly transits the TF and OH zones but is used to support the winter-spring diatom bloom in the deeper and clearer mesohaline estuary and is, in this way, retained in the estuary. As temperatures warm, this organic material is re-mineralized and nutrients again become available to support phytoplanktonic and epiphytic algal growth, both of which can have severe light-limiting effects on SAV. In a sense, nutrient loads in the upper estuary have been reduced (during critical times of the year) while loads to the lower estuary have not been reduced to a point amenable for SAV growth. Boynton *et al.* (2008) have estimated a whole-system N loading rate of about 16 g N m^{-2} yr^{-1}. Latimer and Rego (2010) reported that SAV communities were absent from small to medium sized southern New England estuaries with N loading rates greater than 10 g N m^{-2} yr^{-1}. While there are always serious uncertainties associated with comparisons between systems separated by substantial distances, as well as a host of other things, the lack of SAV in the lower

Patuxent is consistent with results from the New England study. Boynton *et al.* (2008) used two empirical approaches to estimate N loads when SAV communities were still abundant in the lower Patuxent estuary. Both approaches indicated average annual N loading rates during the early-1960s to be about half (~8 g N m^{-2} yr^{-1}) present loading rates and within the range of rates reported by Latimer and Rego (2010) associated with some SAV presence. It is not clear if the TMDL process currently beginning in the Patuxent will achieve nutrient load reductions to levels associated with healthy SAV communities in the masohaline zone.

Finally, there is the question of where propagules came from to start the re-growth of SAV in the upper Patuxent. SAV were likely absent from the TF and OH zones for 20-30 years (late 1960s-1993) so a very local source (i.e., small pockets of SAV along the main channel of the river) seems unlikely, but possible. Since these are freshwater species, a reasonable answer to this question involves downstream transport of seeds or other viable plant parts to the TF and OH zones from more up-stream sources. While this may have happened we also know that a variety of freshwater species were actively growing in the middle and upper reaches of creeks that drain into the TF and OH zones of the Patuxent. Repeated canoe trips into these creeks during the mid and late-1980s revealed a total of 8 species of SAV were present (Garber and Boynton, pers. comm.). Many of these SAV communities were very lush and surrounding water was exceptionally clear. Thus, it is possible, maybe likely, that there was in fact a very local source of SAV material to start the re-growth process in the mainstem of the TF and OH regions of the Patuxent.

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