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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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Executive Summary 2010

DATAFLOW© *Spatial Analyses*

- Spatial analysis showed that cross-channel gradients in salinity formed periodically in all four estuaries examined. These results suggest that water quality and habitat drivers may differ substantially on different sides of the estuary. In the western shore tributaries it appears that northern and eastern sides of the estuary may be more strongly influenced by Bay water, particularly during the spring. Salinity did not consistently explain patterns of chlorophyll-*a*, however the analysis also showed that **areas with elevated chlorophyll-*a* were not randomly distributed in the estuaries** and that patterns suggested that local sources of nutrients were likely to be important in driving some elevated concentrations in chlorophyll-*a*.

- The strongest correlation we found at the whole-estuary level was that **persistently elevated chlorophyll-*a* were negatively correlated with SAV abundance**, as expected. Summer salinity was another variable that was significantly inversely correlated with SAV and positively correlated with elevated chlorophyll-*a*. Yet, a variety of seemingly obvious drivers such as modeled TN loads and watershed land use were not significantly correlated with system responses of SAV and extent and frequency of elevated chlorophyll-*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.

- **DATAFLOW© data have limitations for distinguishing where SAV will or will not occur.** Chlorophyll-*a* data collected when SAV was not present were not able to strongly distinguish between areas that would or would not support SAV. Our analysis suggests that elevated chlorophyll-*a* is not randomly distributed in the estuary, suggesting that it should be possible to differentiate areas that are better or worse habitat quality for SAV, but the data may need to be evaluated differently or additional sampling in shallow areas may be needed to screen sites based on water quality.

High Frequency Analysis with ConMon Data: DO Criteria and Metabolism

- ConMon 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 during early morning hours. These data can also 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. These data can be used in DO criteria assessments for shallow open water sites and for developing a better understanding of the relationship between average DO conditions and DO variability, especially on daily time--scales. We found many DO criteria failures at sites in Maryland tributary rivers and at some sites in the MD Coastal Bays. It appears that the **CBP “Umbrella Criteria” for shallow waters does not provide sufficient protection.** Continuous periods of low DO conditions (low DO duration) are common and can be as long as 121 hours at very nutrient enriched sites. Low DO duration events appear to be well correlated with the degree of DO criteria failure.

- Community production and respiration are responsive to nutrient enrichment in Chesapeake Bay tributaries. Rates were very high at sites with severe nutrient enrichment (~20 g O_2 m^{-3} day^{-1}) and much lower at less impacted sites (~5 g O_2 m^{-3} day^{-1}) and at the historical data site in the Patuxent River. At sites in the Maryland Coastal Bays where estimates of local nitrogen loadings were available, a strong relationship between load and metabolism emerged. **Linking loads to estuarine performance measures such as production and respiration should be continued.**

- We have completed examination of water quality data collected at a site in the Patuxent River estuary during the early 1960s, a period prior to extensive and severe eutrophication of that estuary. During 1964 Pg* rates reached maximum values in spring (May-June) and lower rates during summer and fall. Winter rates were very low. We interpreted this pattern as being associated with the spring freshet when “new” nutrients were delivered to the estuary and were available to support primary production. Summer rates at that time were limited by low additions of nutrients from the drainage basin. As nutrient loads to the Patuxent increased through the late 1960s the temporal pattern of Pg* changed wherein the spring pulse in production was subsumed by rates that continued to increase through the summer until reaching maximum values in August or early September. We suggest this is the eutrophic production pattern (i.e., elevated rates and peak rates during the summer period). All of the most eutrophic sites on the Potomac exhibited this pattern. Less eutrophic sites exhibited peak rates of Pg* earlier in the summer or late spring. The eutrophic pattern of production likely results from large nutrient additions during the spring freshet, lower but still enhanced nutrient additions during late spring and early summer and more efficient recycle of nutrients. In the current condition of Chesapeake Bay there is little nutrient buffering from SAV communities, denitrification is severely compromised during the extensive hypoxic period and nutrient storage in longer-lived plants and animals (e.g. SAV, large benthic infauna) has also been sharply reduced. Thus, nutrients are more available for re-use in support of elevated rates of production, largely by phytoplanktonic algae. **We suggest that if nutrient loads are reduced, the magnitude of Pg* should also be reduced and the temporal pattern of production shift from a very high summer peak to a smaller spring peak.**

Back River Results for Management

- In conjunction with other sources of funding (US EPA Bay Program and Maryland Sea Grant) we developed a case study of the Back River estuary, a heavily nutrient enriched system located adjacent to Baltimore, MD. **The analysis was geared toward providing management with guidance relative to the timing and magnitude of estuarine responses to significant management actions** (reductions in sediment, BOD, N and P loading).

- **We found that point sources of N were reduced by more than a factor of two and P reductions were even greater.** However, there was no indication of diffuse N or P load reductions during the period of record (1985 – 2007). Nitrogen concentrations were reduced to limiting levels following WWTP upgrades to remove N. The frequency of very high chlorophyll-*a* concentration also decreased following WWTP upgrades.

- We also constructed a simple box model to examine flows of water and nutrients between the Back River and Chesapeake Bay. The model indicated that **the N exchange between the river and Bay was small and represented an export of N from the river to the Bay.** This indicates that much of the hyper-eutrophication in the Back River is a local problem and can be solved with local efforts to reduce point and diffuse nutrient inputs. In general, chlorophyll-*a* concentration has been reduced from an annual average of about 120 µg L^{-1} to about 70 µg L^{-1}. **This WWTP is currently being upgraded again and it would be prudent to consider enhanced monitoring of this small system when those activities are complete.**

Hypoxia Forecasting Tool

- In conjunction with funding from NOAA, we developed an approach for forecasting summertime hypoxic conditions for the mainstem of Chesapeake Bay. Details of this model are currently being prepared for publication. This Chapter presents "cookbook" directions for using the hypoxia forecasting model and we suggest it might be a candidate for inclusion on the Eyesonthebay web page. **This model allows summer forecasts (June-September) to be made as soon as the April mainstem Bay chlorophyll data can be obtained.**

Chapter 1

Introduction and Objectives

W.R. Boynton, L.A. Wainger, E.M. Bailey, A.R. Bayard and C.L. Sperling

1-1 Background

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

The first phase of the Chesapeake Bay Program was undertaken during a period of four years (1984 through 1987) and had as its goal the characterization of the existing state of the bay, including spatial and seasonal variation, which were keys to the identification of problem areas. During this phase of the program, the Ecosystems Processes Component (EPC) measured sediment-water oxygen and nutrient exchange rates and determined the rates at which organic and inorganic particulate materials reached deep waters and bay sediments. Sediment-water exchanges and depositional processes are major features of estuarine nutrient cycles and play an important role in determining water quality and habitat conditions. The results of EPC monitoring have been summarized in a series of interpretive reports (Boynton *et al.* annually from 1984 through 2010; and Bailey *et al.* 2008). The results of this characterization effort have confirmed the importance of deposition and sediment processes in determining water quality and habitat conditions. Furthermore, it is also now clear that these processes are responsive to changes in nutrient loading rates (Boynton and Kemp 2008). Much of these data played a key role in formulating, calibrating and verifying Chesapeake Bay water quality models and these data are continuing to be used as the "gold standard" against which the sediment model is further tested and refined. We have also created a web-accessible and complete Chesapeake Bay sediment flux data base that is available to all interested parties (www.gonzo.cbl.umces.edu).

The second phase of the program effort, completed during 1988 through 1990, identified interrelationships and trends in key processes monitored during the initial phase of the program. The EPC was able to identify trends in sediment-water exchanges and deposition rates. Important factors regulating these processes have also been identified and related to water quality conditions (Boynton and Kemp 2008, Kemp and Boynton 1992; Boynton *et al.* 1991).

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

During the latter part of 1997 the Chesapeake Bay Program entered another phase of re-evaluation. Since the last evaluation, programs had collected and analyzed additional information, nutrient reduction strategies had been implemented and, in some areas, habitat improvements have been accomplished. The overall goal of the 1997 re-evaluation was the assessment of the progress of the program and the implementation of necessary modifications to the difficult process of restoring water quality, habitats and living resources in Chesapeake Bay. During this portion of the program, EPC has been further modified to include 1) development of intensive spatial water quality mapping; 2) intensive examination of SAV habitat conditions in major regions of the Chesapeake Bay and development of a high frequency shallow water monitoring protocol (ConMon) that has been extensively implemented in many regions of the Bay and tributary rivers.

Chesapeake 2000 involved the commitment of the participants *“to achieve and maintain the water quality necessary to support aquatic living resources of the Bay and its tributaries and to protect human health."* More specifically, this Agreement focuses on: 1) living resource protection and restoration; 2) vital habitat protection and restoration; 3) water quality restoration and protection; 4) sound land use and; 5) stewardship and community engagement. The current EPC program has activities that are aligned with the habitat and water quality goals described in this agreement.

During the past several years (2008-2010) the EPC of the Biomonitoring Program has further evolved to focus on data analysis of water quality issues. Specifically, the EPC has examined the following: 1) rescued a rare, high quality, near-continuous and long-term water quality data set collected in the mesohaline portion of the Patuxent estuary from 1963-1969 and made this data set generally available; 2) examined multiple sites using dataflow results for a better understanding of the spatial features of water quality and factors, both local and remote, influencing these water quality distributions; 3) used ConMon data sets to assess DO criteria attainment and duration of low DO events in near-shore areas using a variety of computational approaches; and 4) developed an algorithm for computing community-scale primary production and respiration using ConMon data for purposes of developing another metric of water quality and relating these fundamental ecosystem processes to important controlling factors such as nutrient loading rates.

The Chesapeake Bay Water Quality Monitoring Program was initiated to provide guidelines for restoration, protection and future use of the mainstem estuary and its tributaries and to provide evaluations of implemented management actions directed towards alleviating some critical pollution problems. A description of the complete monitoring program is provided in the following documents:

Magnien *et al.* (1987),

Chesapeake Bay program web page http://www.chesapeakebay.net/monprgms.htm

DNR web page http://www.dnr.state.md.us/bay/monitoring/eco/index.html

In addition to the EPC program portion, the monitoring program also has components that measure:

1. Freshwater, nutrient and other pollutant input rates.
2. Chemical, biological and physical properties of the water column.
3. Phytoplankton community characteristics (this program has been much reduced since 2009).
4. Benthic community characteristics (abundances, biomass and indices of health).

1-2 Objectives of the Water Quality Monitoring Program

The EPC has undergone multiple and significant program modification since its inception in 1984 but its overall objectives have remained consistent with those of other Monitoring Program Components. The objectives of the 2010 EPC program were as follows:

- **SPATIAL ANALYSIS USING DATAFLOW© DATA**
  Our key objective in evaluating the spatial patterns of water quality was to understand the spatial structure of the littoral environment in ways that are relevant to aquatic organisms and habitat restoration goals. A specific application was understanding where there are opportunities for and limitations to SAV restoration, in order to target restoration resources most effectively. Although water mixes in estuaries, it is also clear that estuaries are not homogenous in their water quality because of stratification, Coriolis forces, aquatic vegetation patterns, and the location and timing and quality of freshwater inputs. These biophysical forces create spatial structure in the environment that controls habitat for aquatic plants and animals. Whether such structures are persistent through time is further relevant to understanding the drivers of local water quality conditions and the degree of stress or benefit they provide to aquatic organisms. During this period we used GIS and statistical approaches on DATAFLOW© and complementary data to evaluate conditions within four shallow subestuaries: Bush, Corsica, Magothy and Severn and to test the applicability of analysis methods and available data to address research questions.

- **HIGH FREQUENCY ANALYSIS USING CONMON DATA: DO criteria assessment, low DO event duration, community metabolism and DO dynamics**

The ConMon Program (high frequency fixed station monitoring) has been in existence for more than a decade and has accumulated a large data set of water quality variables from a set of about 60 locations in the Maryland tributaries. The EPC began last year to examine these data with several general goals in mind and we have continued and expanded this effort. First, we developed an algorithm to survey ConMon DO data for compliance with dissolved oxygen (DO) criteria for shallow water habitats. We also developed a computational package to compute the duration of low DO events. In both of these analyses we selected sites ranging from very impacted to modestly impacted. Second, we used a modification of the first algorithm to compare historical Patuxent River DO data with Patuxent ConMon DO data collected at the same location to develop a quantitative estimate of change regarding DO criteria attainment between the pre-eutrophication period (1960s) and current times. Third, we developed another algorithm to compute primary production and community respiration using ConMon data, again for a selection of sites ranging from very impacted to modestly impacted. These rates are fundamental properties of all ecosystems and as such are important in understanding ecosystem performance. In the specific case of the Bay Program goals, primary production is linked to nutrient loads and produces labile organic matter available for food webs and bacterial decomposition which often becomes excessive and leads to hypoxia and anoxia. Community respiration is a direct measure of the extent of DO consumption by shallow water communities. We hope to add these computations to bay area web pages (e.g. www.eyesonthebay.net), some of which could be operated in near-real time. Finally, we developed another computational scheme wherein the relationship between mean daily DO conditions and daily DO variability (short-term DO dynamics) could be examined relative to DO criteria assessment. This effort may provide additional approaches for DO criteria assessment in areas of the Bay not having ConMon data sets.

- **RECONSTRUCTION OF HISTORIC PATUXENT RIVER HIGH FREQUENCY DATA SET**

  The EPC Program came into possession of a historically significant water quality data set collected in the mesohaline portion of the Patuxent River. These data were collected from October 1963 through December, 1969, a period of time preceding large-scale watershed development (1963-1966) and then including the initial period of intensive development, land clearing and sewage treatment plant operations. Data were collected using a variety of early sensor systems and recorded on large format plot recorders. Calibration of sensors was a high priority and a series of reports developed by Cory and colleagues clearly described all procedures. The EPC invested considerable effort to convert data contained on these strip charts to digital data. Water quality data included temperature, salinity and dissolved oxygen. Data were recorded at one hour intervals for the entire time period. This data set constitutes one of the only intensive set of observations in the Bay system prior to serious eutrophication of these estuaries and thus serves as a benchmark data set. This data set is now available to all interested groups.

- **CASE STUDY OF THE BACK RIVER ESTUARY**

In conjunction with other sources of funding (US EPA Bay Program and Maryland Sea Grant) we developed a case study of the Back River estuary, a heavily nutrient enriched system located adjacent to Baltimore, MD. This site was selected because strong management actions have been taken at the large WWTP located at the head of this estuary. Actions included reductions in sediment, BOD, N, and P loading. The analysis was geared toward providing management with guidance relative to the timing and magnitude of estuarine responses to these significant management actions.

- **HYPOXIA FORECASTING FOR THE MAINSTEM CHESAPEAKE BAY**
  In conjunction with funding from NOAA, we report here an approach for forecasting hypoxic conditions in the mainstem of Chesapeake Bay. The approached used a statistical model utilizing many monitoring program components (River input monitoring, WQ data) as well as wind conditions. This Chapter presents "cookbook" directions for using the hypoxia forecasting model. Details of this model are currently being prepared for publication.

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Chapter 2

Dissolved Oxygen Criteria Assessments: Utility of ConMon Data for Assessing Shallow Water Habitats

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

2-1 Introduction and Objectives

Until the last decade, 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. 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, a single measurement scheme is not adequate for addressing all questions. Thus, 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-***Con***tinuous ***Mon***itoring 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 main channel 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 about 60 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).

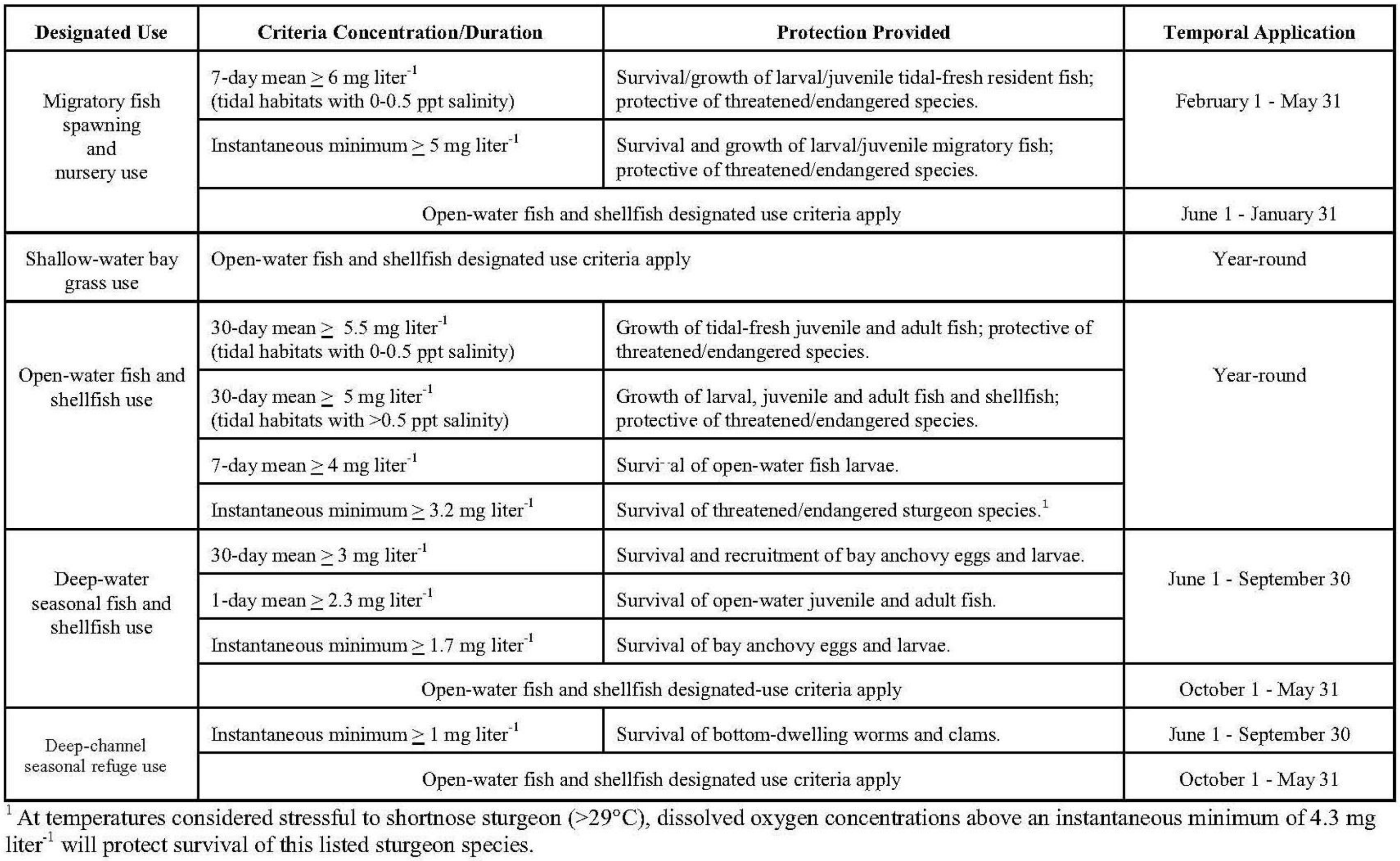
It is the second and fourth ConMon uses that are the focus of this chapter and we approach this issue in three ways. First, we provide examples of DO criteria % non-attainment for several sites in the Bay system. It remains unclear as to which of several approaches best captures meaningful DO non-attainment; we present results of all approaches in this section. 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 focus on the interaction of day-scale DO mean concentration and diel variability, again as a tool for considering how to generally address DO criteria attainment or non-attainment.

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).

Based on these USEAP 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 2-1. 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.

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 (Bailey, Wainger, Perry and Hall pers. comm. 2010) used to import, clean and select ConMon data:

```
libname conmon 'C:\Documents and Settings\boynton\My Documents\My SAS
Files\9.2\ConMon';
run;

data conmon.cleanBen2;
set SASUSER.Ben;
%let InvalidCodes =
'GBO','GNV','GPC','GPF','GSC','GWL','GWM','NIR','NIS','NND','NNF','NOW','NPF'
,'NQR','PDP','PSW';
if TOTAL_DEPTH_A in (&InvalidCodes) then DELETE;
if BATT_A     in (&InvalidCodes) then DELETE;
if WTEMP_A    in (&InvalidCodes) then DELETE;
if SPCOND_A    in (&InvalidCodes) then DELETE;
if SALINITY_A   in (&InvalidCodes) then DELETE;
if DO_SAT_A    in (&InvalidCodes) then DELETE;
if DO_A      in (&InvalidCodes) then DELETE;
if PH_A      in (&InvalidCodes) then DELETE;
if TURB_NTU_A   in (&InvalidCodes) then DELETE;
if FLUOR_A    in (&InvalidCodes) then DELETE;
if TCHL_PRE_CAL_A in (&InvalidCodes) then DELETE;
if CHLA_A     in (&InvalidCodes) then DELETE;
l_CHLA_A = LENGTH(CHLA_A);

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;

data metabdata_year;
set conmon.cleanBen2;
Year = substr(SAMPLE_DATE, 1, 4);
run;

data metabdataxed;
set metabdata_year;
where STATION = 'XED0694' and YEAR = '2005';
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

Attainment of the instantaneous (DO < 3.2 mg L^{-1}) and 30-day mean minimum DO criteria (DO < 5.0 mg L^{-1}) as a function of % non-attainment and maximum low DO duration (event) was evaluated by the following procedures.

Data from 11 ConMon stations (Table 2-4 and Figure 2-1) 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.

1	Site	Code	Year
	Benedict	XED0694	2003
	Benedict	XED0694	2004
	Benedict	XED0694	2005
2	Bishopville Prong	XDM4486	2003
	Bishopville Prong	XDM4486	2004
	Bishopville Prong	XDM4486	2005
	Bishopville Prong	XDM4486	2006
	Bishopville Prong	XDM4486	2007
	Bishopville Prong	XDM4486	2008
3	CBL	XCF9029	2003
	CBL	XCF9029	2004
	CBL	XCF9029	2005
4	Fenwick	XFB0231	2004
	Fenwick	XFB0231	2005

	Fenwick	XFB0231	2006
	Fenwick	XFB0231	2007
	Fenwick	XFB0231	2008
5	Fort McHenry	XIE5748	2004
	Fort McHenry	XIE5748	2005
	Fort McHenry	XIE5748	2006
	Fort McHenry	XIE5748	2007
	Fort McHenry	XIE5748	2008
6	Jug Bay	PXT0455	2003
	Jug Bay	PXT0455	2004
	Jug Bay	PXT0455	2005
	Jug Bay	PXT0455	2006
	Jug Bay	PXT0455	2007
	Jug Bay	PXT0455	2008
7	Pin Oak	XDE4587	2003
	Pin Oak	XDE4587	2004
	Pin Oak	XDE4587	2005
	Pin Oak	XDE4587	2006
	Pin Oak	XDE4587	2007
8	Public Landing	XBM8828	2005
	Public Landing	XBM8828	2006
	Public Landing	XBM8828	2007
	Public Landing	XBM8828	2007
9	St. George's Island	XBF7904	2006
	St. George's Island	XBF7904	2007
	St. George's Island	XBF7904	2008
10	Sycamore Point	XHH3851	2005
	Sycamore Point	XHH3851	2006
	Sycamore Point	XHH3851	2007
	Sycamore Point	XHH3851	2008
11	Turville Creek	TUV0021	2003
	Turville Creek	TUV0021	2004
	Turville Creek	TUV0021	2005

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).

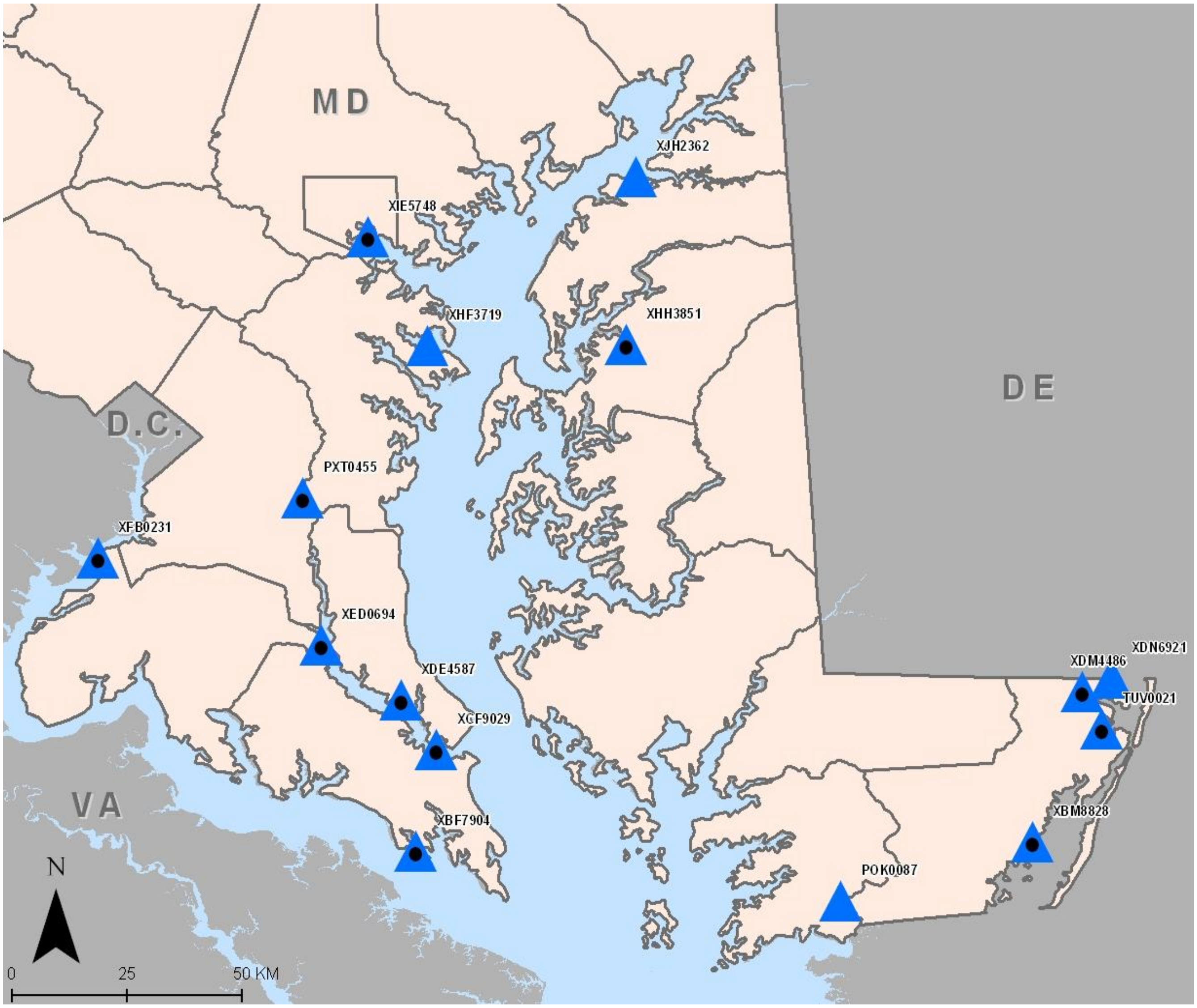


Figure 2-1. Location of ConMon stations used in DO criteria analyses. Stations marked with blue triangles (▲) were used for mean and range analysis and stations marked with black circles (●) were used for % attainment and duration calculations.

2-3.3 DO Mean Versus DO Range Analysis

Attainment of the instantaneous minimum DO criteria (DO < 3 .2mg L^{-1}) as a function of daily mean DO levels (mg L^{-1}) and daily range of DO levels (mg L^{-1} day^{-1}) was evaluated using the following procedures.

A total of 15 ConMon stations (Table 2-5 and Figure 2-1) were QA/QC'd using the SAS program before being processed with MATLAB, which generated a total of 30 graphic analyses. ConMon data was imported into MATLAB 7.3.0.267 (R2006b) in tab delimited format for each

ConMon station and year. In this instance "year" can include data for the full calendar year (in a few instances) but more generally for the period April – October.

Several functions were created in order to prepare the ConMon data to be manipulated/plotted in MATLAB. First function converted the date and time format into vector format. The second function performed the mean, range and threshold test calculations on daily DO values. Threshold test calculated failure when DO was < 3.2 mg L^{-1}. Limits were applied so that days must have 96 data points (complete collection days = 96 measurements in a 24 hour period) to be included in the analysis. Daily failures were tallied and divided by 96 to give daily threshold of failure (%). Threshold color schemes were applied for different degrees of DO criteria failure (blue = 0%, yellow > 0% - 10%, red > 10%). The next function was to extract summer months (June-August) from the ConMon data set. Finally two graphs were generated for each station, all months and summer months only. Linear regression equation model was fit to threshold data (> 0 – 10% failure; yellow data) for each graph.

All Betterton Beach DO data were within 0% frequency for minimum DO Criteria failure, which generated all blue threshold data points and the linear regression equation model was not applied. Grey's Creek only had 2008 data giving only 41 complete data collection days (complete = 96 readings/day) and in turn the Summer months gave only 8 data points. The Rehobeth station (Pocomoke River) was QA/QC'd manually and the MATLAB functions described above were applied).

Table 2-5. List of ConMon stations used in DO mean vs. DO range analysis.

1	**Site**	**Code**	**Year**
	Benedict	XED0694	2003
	Benedict	XED0694	2004
	Benedict	XED0694	2005
2	Betterton Beach	XJH2362	2006
	Betterton Beach	XJH2362	2007
	Betterton Beach	XJH2362	2008
3	Bishopville Prong	XDM4486	2003
	Bishopville Prong	XDM4486	2004
	Bishopville Prong	XDM4486	2005
	Bishopville Prong	XDM4486	2006
	Bishopville Prong	XDM4486	2007
	Bishopville Prong	XDM4486	2008
4	CBL	XCF9029	2003
	CBL	XCF9029	2004
	CBL	XCF9029	2005
5	Fenwick	XFB0231	2004
	Fenwick	XFB0231	2005
	Fenwick	XFB0231	2006
	Fenwick	XFB0231	2007

	Fenwick	XFB0231	2008
6	Fort McHenry	XIE5748	2004
	Fort McHenry	XIE5748	2005
	Fort McHenry	XIE5748	2006
	Fort McHenry	XIE5748	2007
	Fort McHenry	XIE5748	2008
7	Grey's Creek	XDN6921	2008
8	Jug Bay	PXT0455	2003
	Jug Bay	PXT0455	2004
	Jug Bay	PXT0455	2005
	Jug Bay	PXT0455	2006
	Jug Bay	PXT0455	2007
	Jug Bay	PXT0455	2008
9	Pin Oak	XDE4587	2003
	Pin Oak	XDE4587	2004
	Pin Oak	XDE4587	2005
	Pin Oak	XDE4587	2006
	Pin Oak	XDE4587	2007
10	Public Landing	XBM8828	2005
	Public Landing	XBM8828	2006
	Public Landing	XBM8828	2007
	Public Landing	XBM8828	2007
11	Rehobeth	POK0087	2002
12	St. George's Island	XBF7904	2006
	St. George's Island	XBF7904	2007
	St. George's Island	XBF7904	2008
13	Stonington	XHF3719	2001
	Stonington	XHF3719	2002
	Stonington	XHF3719	2003
14	Sycamore Point	XHH3851	2005
	Sycamore Point	XHH3851	2006
	Sycamore Point	XHH3851	2007
	Sycamore Point	XHH3851	2008
15	Turville Creek	TUV0021	2003
	Turville Creek	TUV0021	2004
	Turville Creek	TUV0021	2005

2-4 Results and Discussion

2-4.1 Testing Shallow Water Sites for DO Criteria Attainment and Low DO Duration

High frequency DO data was analyzed from 11 ConMon stations (Figure 2-1 and Table 2-3) 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 2003 to 2008 and generally most data sets included data from March to December.

The total hours (per year) of DO collection at each station ranged from ~ 2700 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 121 hours (~5 days).

Table 2-6. 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}).

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

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
		2004	3/11 to 12/21	6802	384	6	17	1392	20	37
		2005	3/2 to 12/20	6214	299	5	13	1057	17	21
Public Landing		2005	3/2 to 12/20	6303	20	0	4	330	5	35
		2006	3/15 to 12/20	4768	68	1	10	733	15	41
		2007	3/15 to 12/12	5492	14	0	4	384	7	17
		2008	3/19 to 12/10	5688	22	0	8	424	7	18

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
Fort 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
St. George's Island	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
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

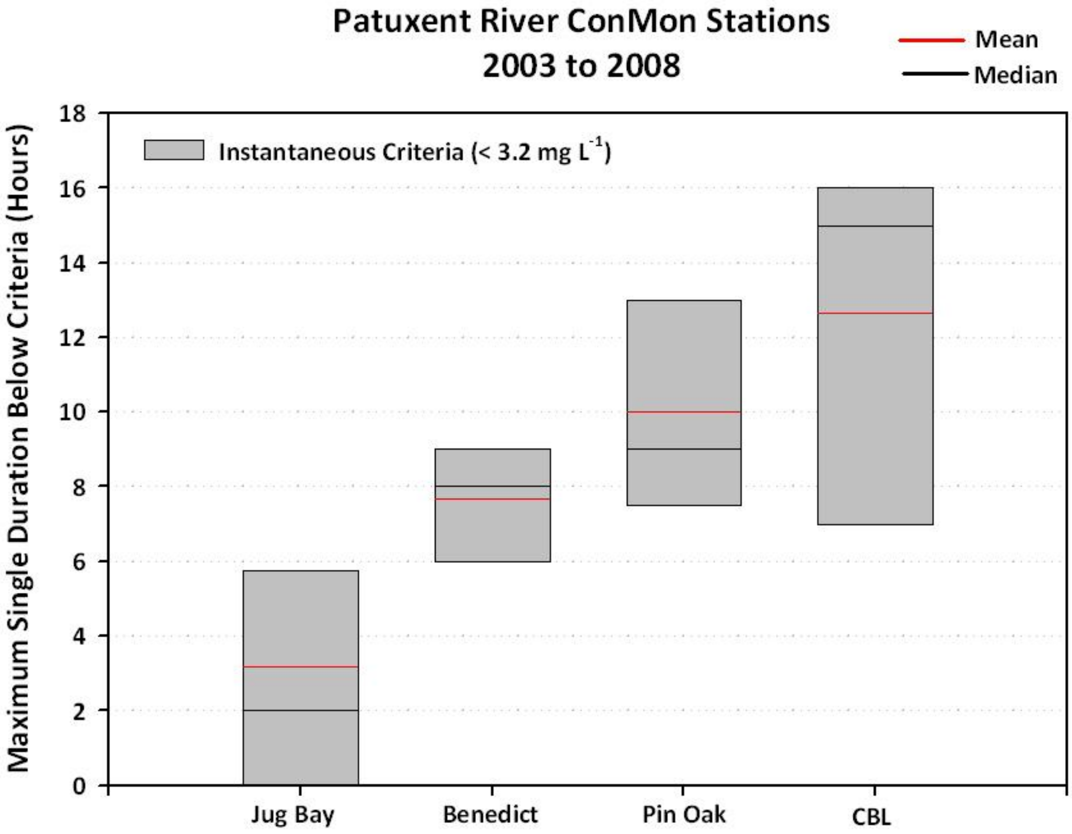


Figure 2-2. Box plots of maximum single duration (hours) of below criteria DO events at Patuxent River ConMon stations (2003-2008). Boundary of box indicates 25th and 75th percentiles. Red lines indicate mean value and black lines indicate median value.

Along an estuarine gradient in the Patuxent River maximum duration of low DO events (instantaneous criteria < 3.2 mg L^{-1}) increased with increasing salinity (Figure 2-2). Even though the Jug Bay station is located closer to nutrient sources this station has a well-mixed and shallow water column resulting in short residence times for water masses. We suspect that this is the reason for the short duration (averaging 3.5 hours) of low DO events at this station.

Criteria failure (Table 2-6) was highest at highly enriched sites (Ft. McHenry, Patapsco River and Bishopville Prong, MD Coastal Bays). At Ft. McHenry DO % failure ranged from 7- 18% (instantaneous) and 14 – 37% (30-day) overall with maximum duration of low DO events ranging from 14 – 23 hours (< 3.2 mg L^{-1}) and 47 to 112 hours (< 5 mg L^{-1}). In the Maryland Coastal Bays (Figure 2-3) criteria failure followed a nutrient enrichment gradient similar to that found in our metabolism analyses (Chapter 3, this report) with the highest failures occurring at the Bishopville Prong station (8 - 44%) and lowest at Public Landing (0-15%). For instantaneous criteria (< 3.2 mg L^{-1}) the Bishopville Prong station experienced failure events on average lasting 30 hours (Figure 2-3). In contrast the Public Landing site rarely had events longer than 4 hours.

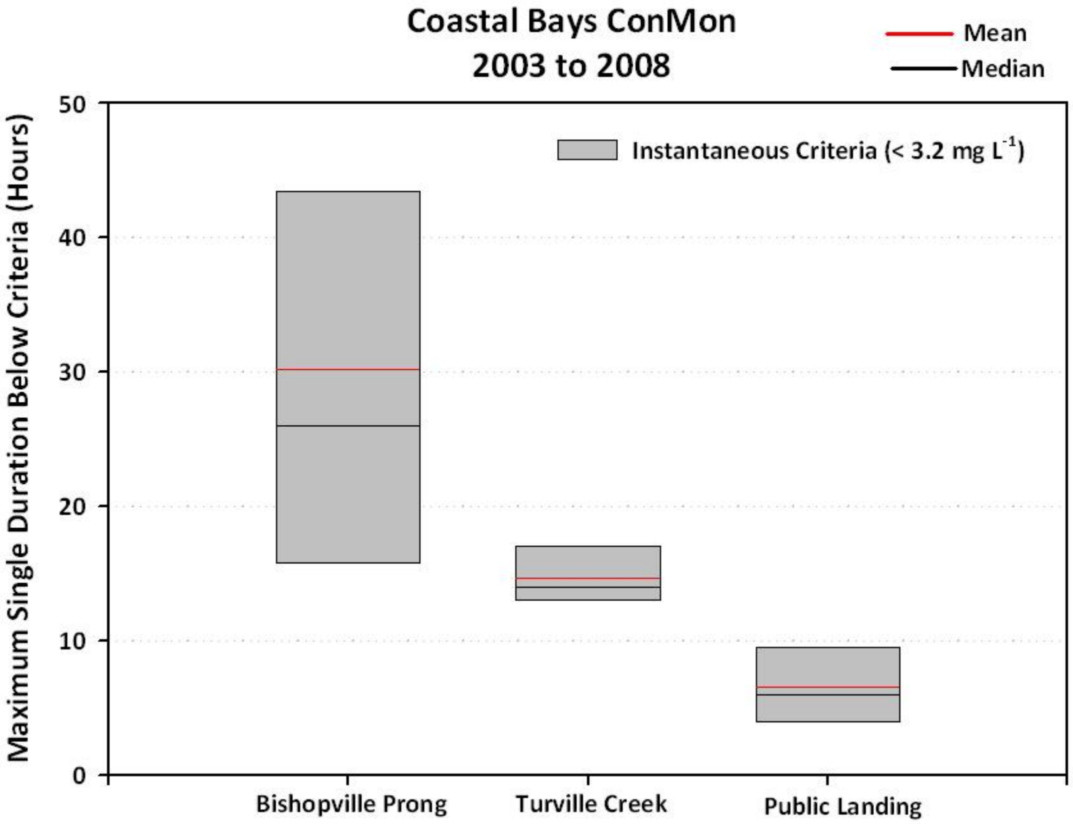


Figure 2-3. Box plots of maximum single duration (hours) of below criteria DO events at Maryland Coastal Bays ConMon stations (2003-2008). Boundary of box indicates 25th and 75th percentiles. Red lines indicate mean value and black lines indicate median value.

In most cases the frequency of duration of low DO events varied little inter-annually for the years we examined. We did not examine the frequency distribution of when these events occurred seasonally (this would be a good topic to expand on in further analysis) rather just the overall frequency of duration length in a given year's data set. For example (Figure 2-4) at the Ft. McHenry site (Patapsco River) the two years of highest % failure of the 30 day DO criteria (< 5 mg L^{-1}) were 2007 (37%) and 2008 (38%). In 2007 there were 492 separate failure events with 414 of those events lasting only 15 minutes. At first glance this seems like something most biota could tolerate. However, that same year this site also experienced DO under 5 mg L^{-1} 40 different times, 30 events lasting 12 to 24 hours, 7 events lasting 1-3 days and 3 events lasting 3 to 4 days. In 2008 Ft. McHenry only experienced a 3 - 4 day event one time.

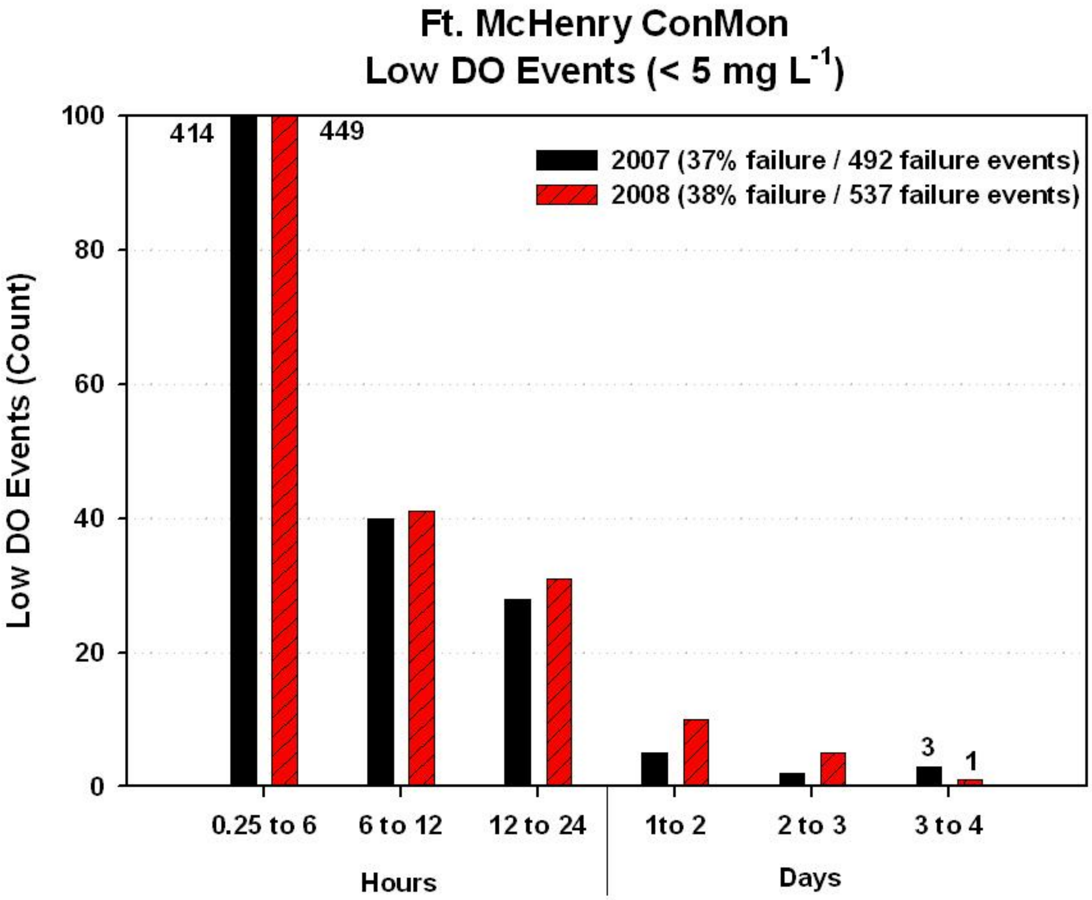


Figure 2-4. Low DO events (frequency and duration) at the Ft. McHenry (Patapsco River) ConMon station. Percent failure and total low DO event count shown for each entire year data set.

2-4.2 Exploring DO Dynamics in Shallow Water Habitats

In the previous section, high frequency DO data were examined for DO criteria attainment or failure and for low DO duration for selected shallow water areas of the Bay, with a focus on tributary locations. While there have been a large number of ConMon sites located throughout the Maryland Bay and tributaries (~60; some active now and some no longer active) it is not reasonable to expect that every sector of the Bay will have ConMon data sufficient to fully address DO criteria attainment. Thus, a better understanding of DO dynamics in shallow waters is an appropriate topic because such examination may allow for reasonable extrapolation (i.e., via statistical modeling) of DO data from ConMon sites to sites that need criteria evaluation but do not have the high frequency DO data available.

To put the need for careful evaluation of shallow water environments of the Bay into context, the area of water less than 2 m in depth in the 92 Bay Program segments is summarized in Table 2-7. About 50 % of Bay Program segments have greater than 50 % of area less than 2 m in depth (the 2 m depth contour is a generally accepted definition for shallow waters). Of the total open water area (~2.8 x 10^6 m^2), about 25 % is shallow water. Thus, at the largest spatial scale (open waters for the full Bay system) shallow waters represent a modest proportion of the area. However, at the segment scale (the scale at which DO criteria are applied) shallow waters play a dominant role in many segments.

Table 2-7. A summary of total area, area less than 2 meters in depth and percent of shallow area for all 92 EPA Chesapeake Bay Program segments. Data were from P. Tango (pers. comm.).

CBP segment	STATE	Segment Area, acres	Segment Area < 2m	%Shallow (<=2m)
CB8PH	VA	101,913	2,239	2.2
PAXTF	MD	1,089	54	5.0
CB6PH	VA	183,686	9,256	5.0
CB5MH	MD	227,457	19,566	8.6
CB3MH	MD	89,350	9,211	10.3
CB4MH	MD	224,581	25,230	11.2
CB7PH	VA	375,792	42,441	11.3
CB5MH	VA	136,931	15,610	11.4
C&DOH	DE	264	37	14.1
POTOH	MD	43,262	6,580	15.2
CB2OH	MD	68,013	10,356	15.2
JMSPH	VA	18,919	3,022	16.0
POTMH	MD	198,723	32,356	16.3
PATMH	MD	23,130	4,814	20.8
C&DOH	MD	617	133	21.6
PMKOH	VA	3,483	807	23.2
POTTF	MD	22,978	5,961	25.9
ELIPH	VA	5,227	1,461	28.0
MPNOH	VA	1,965	554	28.2
SEVMH	MD	7,262	2,109	29.0
PAXMH	MD	26,573	8,797	33.1
TANMH	VA	78,278	26,419	33.8
MAGMH	MD	6,556	2,254	34.4
TANMH	MD	143,605	49,685	34.6
JMSOH	VA	31,567	10,949	34.7
ANATF	DC	787	273	34.7
CHOMH1	MD	59,814	20,903	34.9
EASMH	MD	57,957	20,948	36.1
CHOMH2	MD	18,335	6,836	37.3
RPPMH	VA	80,020	30,517	38.1
CHSMH	MD	29,477	11,505	39.0
NANTF	MD	286	113	39.3
SOUMH	MD	5,926	2,339	39.5
MOBPH	VA	84,682	34,639	40.9
JMSMH	VA	75,180	30,924	41.1
YRKPH	VA	16,906	7,001	41.4
POTTF	DC	3,283	1,467	44.7
CRRMH	VA	5,803	2,612	45.0
SASOH	MD	8,176	3,714	45.4
CHOOH	MD	3,716	1,737	46.7

PIAMH	VA	17,242	8,123	47.1
RHDMH	MD	2,251	1,086	48.2
SBEMH	VA	2,074	1,007	48.6
RPPTF	VA	9,020	4,514	50.0
NANOH	MD	4,066	2,055	50.5
RPPOH	VA	4,828	2,511	52.0
YRKMH	VA	23,374	12,721	54.4
JMSTF	VA	18,282	9,952	54.4
ELKOH	MD	9,210	5,027	54.6
JMSTF	VA	5,268	2,890	54.9
LCHMH	MD	22,134	12,378	55.9
POTOH	MD	1,909	1,079	56.5
WSTMH	MD	2,793	1,600	57.3
POCMH	VA	34,293	19,741	57.6
CB1TF	MD	37,466	21,595	57.6
PAXOH	MD	3,520	2,073	58.9
BSHOH	MD	7,547	4,607	61.0
MPNTF	VA	2,293	1,409	61.5
MIDOH	MD	4,007	2,480	61.9
POCMH	MD	12,870	7,981	62.0
CHSOH	MD	3,655	2,309	63.2
NANMH	MD	11,949	7,715	64.6
BOHOH	MD	2,947	1,905	64.6
CHKOH	VA	6,911	4,504	65.2
POTMH	VA	20,673	13,487	65.2
FSBMH	MD	20,635	13,649	66.1
PMKTF	VA	4,010	2,653	66.2
HNGMH	MD	24,147	16,463	68.2
CHOTF	MD	2,201	1,521	69.1
NORTF	MD	3,909	2,743	70.2
BIGMH	MD	7,182	5,071	70.6
MANMH	MD	15,021	10,705	71.3
BACOH	MD	3,997	2,861	71.6
GUNOH	MD	10,279	7,362	71.6
WICMH	MD	8,677	6,385	73.6
WBEMH	VA	1,484	1,109	74.7
POCTF	MD	988	748	75.7
POCOH	MD	1,595	1,210	75.8
EBEMH	VA	1,427	1,083	75.9
MATTF	MD	1,799	1,389	77.2
APPTF	VA	1,980	1,604	81.0
LYNPH	VA	4,845	3,943	81.4
CHSTF	MD	1,009	870	86.2
LAFMH	VA	1,421	1,246	87.7
NANTF	DE	852	774	90.8
POTTF	VA	10,912	10,082	92.4
POTOH	VA	5,194	4,853	93.4

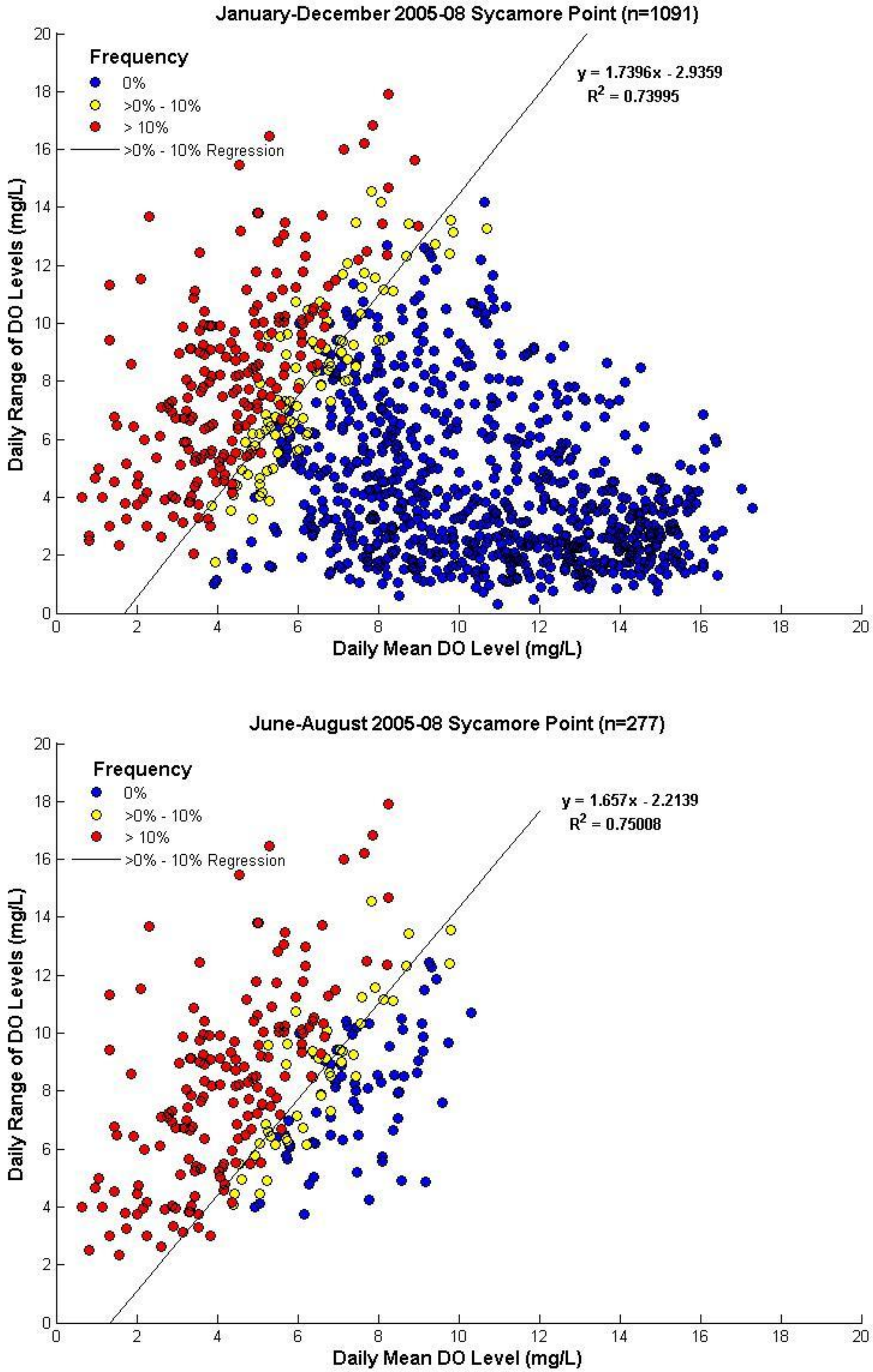
POCOH	VA	1,820	1,714	94.2
POTOH	MD	2,754	2,688	97.6
PISTF	MD	917	914	99.8
ANATF	MD	54	54	100.0

The analysis presented here was explained in the Methods section of this Chapter but for the sake of clarity the essential pieces of this analysis are summarized here. In the following color-coded figures each dot represents one day of data from a specific ConMon site. In this case, all dots are represented by 96 DO measurements (one measurement per fifteen minutes for 24 hours). Based on these 96 measurements, a daily mean was computed (x-axis) and a daily range (max DO – min DO) was computed (y-axis). If, for a 24 hour period, there were no values below the instantaneous DO criteria (> 3.2 mg L^{-1}) then that day was recorded as a blue dot. If there were fewer than 10% of observations below the criteria concentration, then that day received a yellow dot. If there were more than 10% of observations below the criteria value, that day received a red dot. The regression line indicated in each diagram was developed using the yellow-coded data and serves to separate the serious criteria failures (red dots) from those passing the criteria (blue dots). The paired graphs were developed using all data available for a year or multiple years (top panel) and using data just from the critical DO period (June – August), again using data for all years available (bottom panel). If a daily set of observations were missing one or more observations (i.e., there were less than 96 observations for a specific day), data from that day were not used. Finally, all graphs have the same axis ranges (0 - 20 mg O_2 L^{-1}) so all are directly comparable. We have adapted this analysis from previous work done by Claire Buchanan from the Interstate Commission on the Potomac River Basin (pers. comm.).

We selected a series of sites for this analysis and these included the following: several very nutrient enriched sites (Sycamore Point in the upper Corsica River and Fort McHenry in the Patapsco River); two estuarine systems offering a gradient of nutrient enrichment (Maryland Coastal Bay sites of Bishopville Prong, Turville Creek and Public Landing) and four sites along the axis of the Patuxent River (Jug Bay, Benedict, Pin Oak and CBL). Finally, we selected a series of ConMon locations that were at more “exposed” sites along the shore (or near the shore) of the Chesapeake mainstem (Betterton Beach & Stonington) or the Potomac mainstem (Fenwick) and one black water (naturally high in dissolved organic carbon) site (Rehobeth).

We start this discussion with analyses based on data collected at two highly enriched sites (Sycamore Point in the upper Corsica River and Fort McHenry in the Patapsco River). Perhaps the most striking feature of these analyses is the extreme range observed in both mean daily DO concentration (~1 to 17 mg L^{-1}) and daily DO range (~2 to 18 mg L^{-1} day^{-1}). Very large day-scale variation in mean and range seem to be characteristic of nutrient enriched sites as we have suggested in earlier reports (Figs 2-5a and 2-5b). At both sites there were frequent DO criteria failures and, by comparing the top and bottom panels, it is clear the vast majority of failures (red dots) and partial failures (yellow dots) occurred during summer periods (June – August). It is also clear that DO criteria failures occurred when daily mean DO was both high and low. Low daily mean DO values were often, but not always, associated with lower DO ranges. There are also many observations of very high mean DO (> 12 mg L^{-1}) at both sites and these observations were associated with relatively small daily DO ranges. The majority of these data are from fall through late spring periods. Finally, there is a surprisingly clear delineation among days failing and passing DO criteria as indicated by a simple regression model based on marginal failure days (yellow dots). In effect, this indicates that both the mean and the diel variation in DO plays into criteria pass or fail. It is interesting to note that the slope of this regression model is quite similar between sites and between models using all data available and just summer data. If a daily DO mean is used as an indicator of DO criteria attainment, a mean daily value of about 9 mg L^{-1} is needed to assure complete compliance. During summer periods this concentration is well above DO saturation values and is indicative of the importance of biological processes (P and R) in determining DO concentrations in these shallow water environments.

A


January-December 2005-08 Sycamore Point (n=1091)
Frequency
0%
>0% - 10%
> 10%
>0% - 10% Regression
y = 1.7396x - 2.9359
R^2 = 0.73995
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20
June-August 2005-08 Sycamore Point (n=277)
Frequency
0%
>0% - 10%
> 10%
>0% - 10% Regression
y = 1.657x - 2.2139
R^2 = 0.75008
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20

B

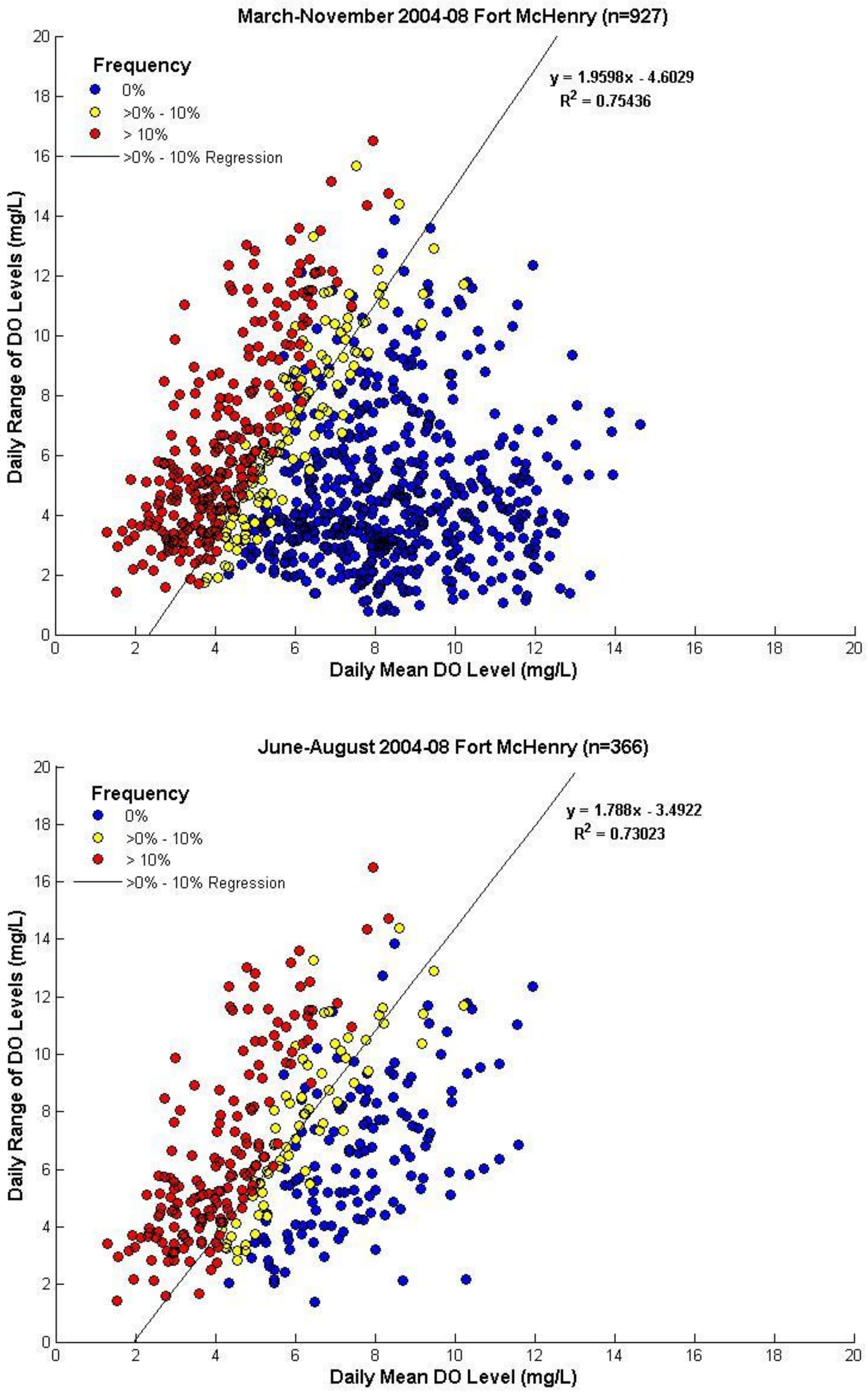
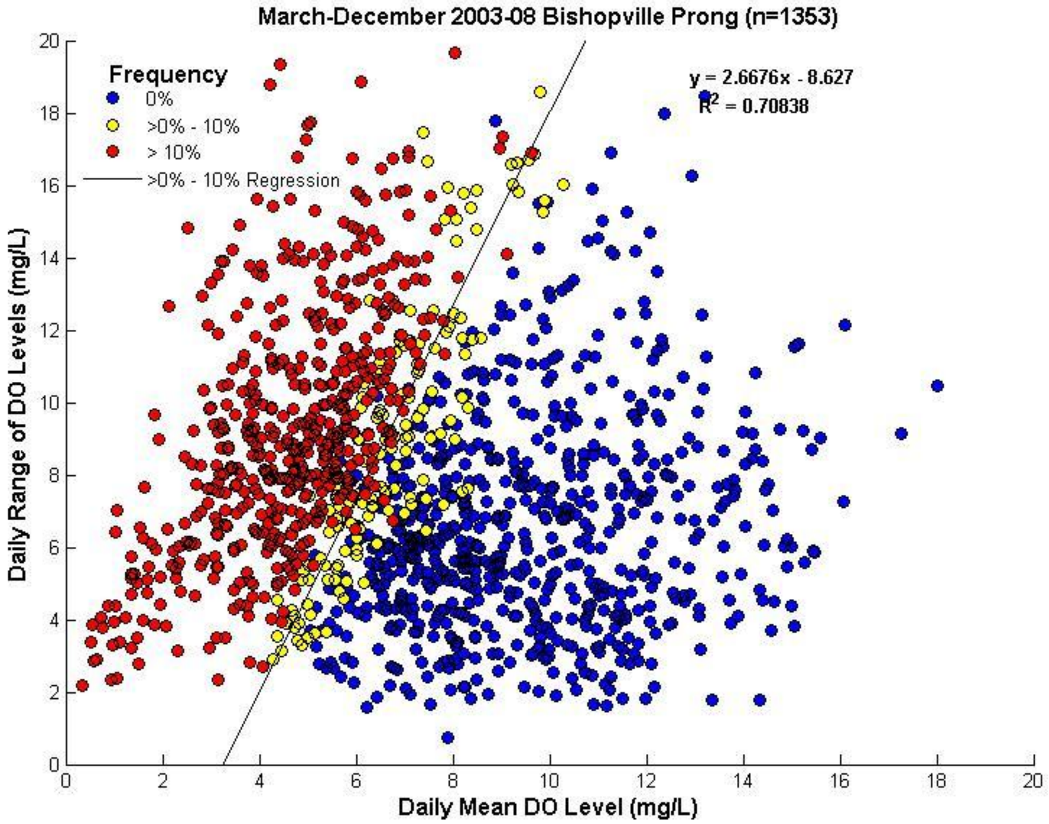
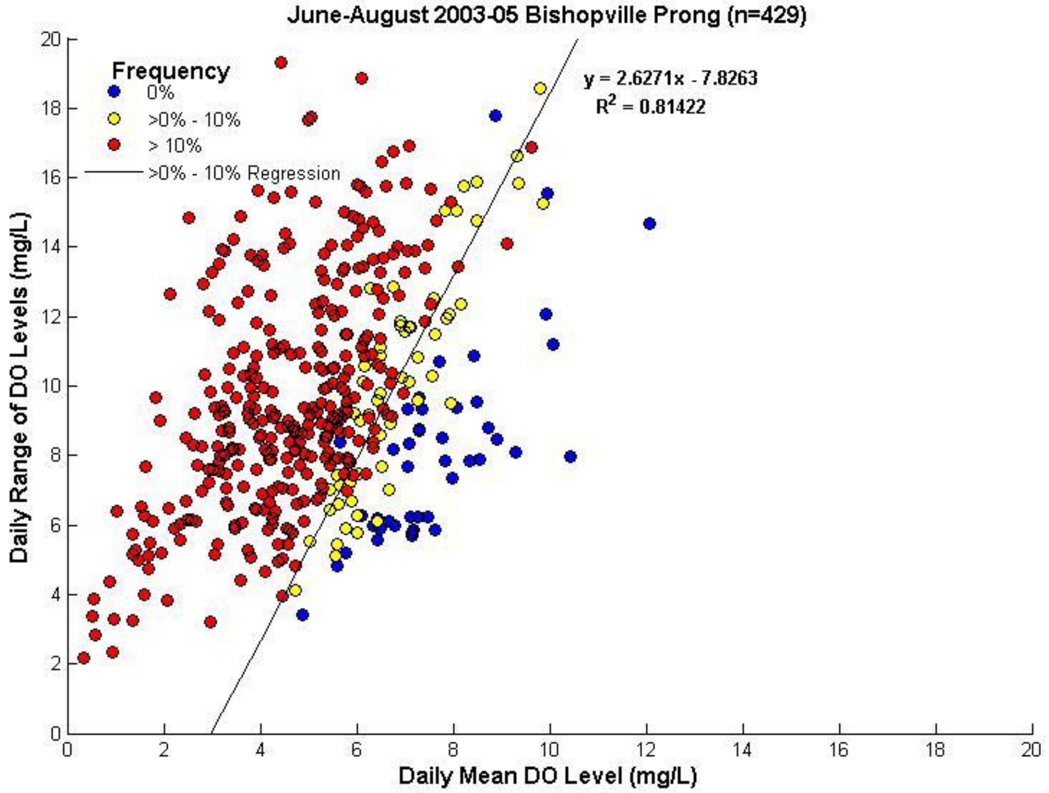

March-November 2004-08 Fort McHenry (n=927)
Frequency
0%
>0% - 10%
> 10%
>0% - 10% Regression
y = 1.9598x - 4.6029
R^2 = 0.75436
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20
June-August 2004-08 Fort McHenry (n=366)
Frequency
0%
>0% - 10%
> 10%
>0% - 10% Regression
y = 1.788x - 3.4922
R^2 = 0.73023
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20

Figure 2-5 (previous 2 pages). Scatter plots of daily mean DO concentration versus daily DO range (max DO – min DO) based on ConMon data from (A) Sycamore Point in the Corsica River and (B) Fort McHenry in the Patapsco River. Blue dots represent days when there were no instantaneous DO criteria (DO < 3.2 mg l^{-1}) failures, yellow dots indicate days when there were less than 10% DO criteria failures and red dots indicate >10% DO criteria failures during a single 24 hour period. The regression line is based on the yellow dots and serves to separate the days passing and failing instantaneous DO criteria. The top panel includes all data during each year of measurement; the bottom panel only has data from June – August. In both cases, only days with a complete set of observations (n = 96) were used in these analyses.

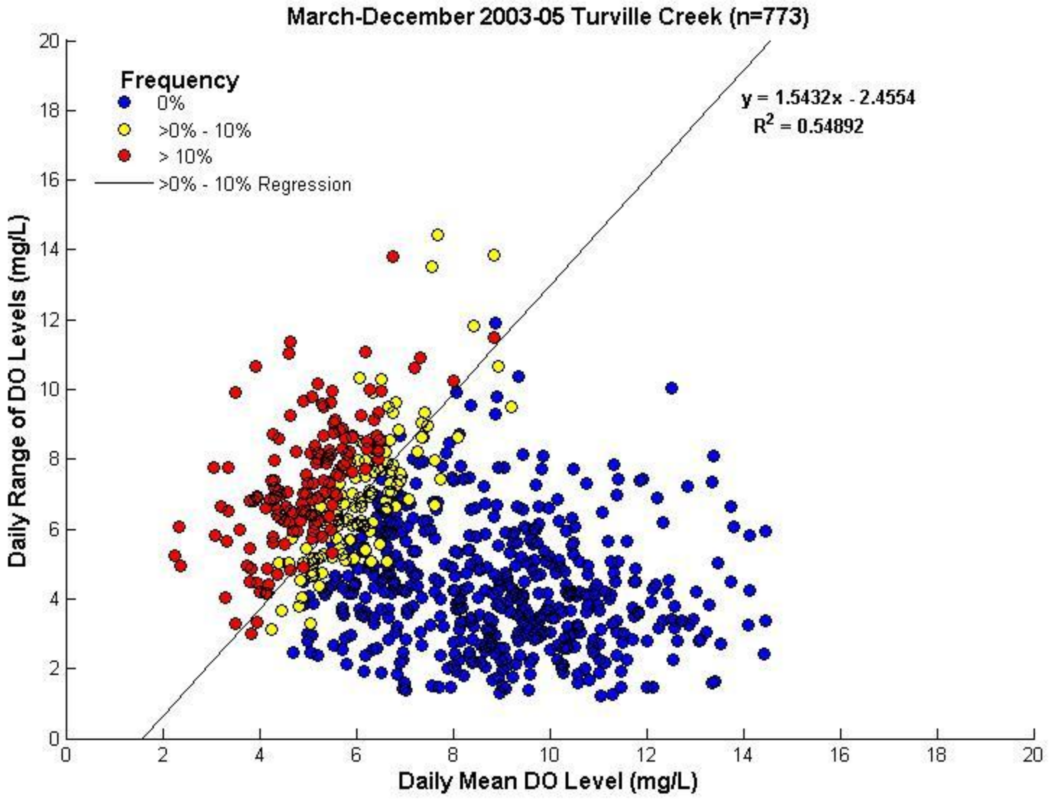
The second set of analyses used sets of ConMon sites where a gradient in nutrient enrichment was present (Figs. 2-6 a-c and Figs. 2-7 a-d). The first of these was in the Maryland Coastal Bays and included ConMon stations in highly enriched (Bishopville Prong), moderately enriched (Turville Creek) and relatively non-enriched (Public Landing) locations. Estimates of nitrogen loads to these locations were previously reported by Boynton et al (1996) and confirm the N-load gradient indicated here. At the most enriched site, ranges in DO daily means and ranges was again extreme and virtually all DO criteria failures occurred during summer. In fact, there were relatively few days during summer when DO criteria were achieved (Fig. 2-6a; bottom panel). At this extreme site there were numerous days when the DO range was huge (> 12 mg L^{-1} day^{-1}) but DO criteria were still achieved because the mean daily DO was so high. It is also interesting to note that the regression model slope (based on yellow dots) was considerably steeper than in previous examples and may be a characteristic of extreme sites. At the moderately enriched site (Turville Creek; Fig. 2-6b) and at the less enriched site (Public Landing; Fig. 2-6c) the patterns are quite different. Failure rates at Turville Creek were less frequent and the daily DO range was considerably smaller. At the Public Landing site DO criteria failures were rare and the daily DO range and daily DO means were even less extreme. Again, it is useful to note that the slope of the regression model decreased as the DO criteria failure rate decreased.

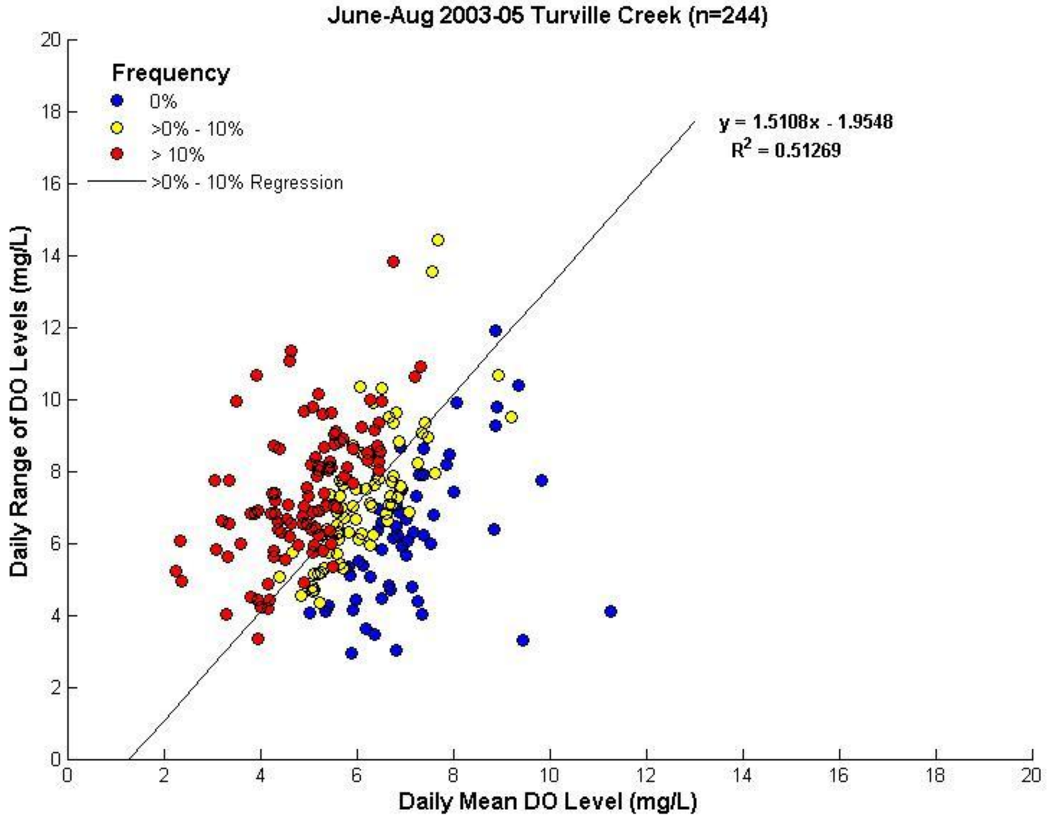
A


March-December 2003-08 Bishopville Prong (n=1353)
Frequency
0%
>0% - 10%
> 10%
>0% - 10% Regression
y = 2.6676x - 8.627
R2 = 0.70838
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20


June-August 2003-05 Bishopville Prong (n=429)
Frequency
0%
>0% - 10%
> 10%
>0% - 10% Regression
y = 2.6271x - 7.8263
R2 = 0.81422
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20

B


March-December 2003-05 Turville Creek (n=773)
Frequency
● 0%
● >0% - 10%
● > 10%
>0% - 10% Regression
y = 1.5432x - 2.4554
R2 = 0.54892
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20


June-Aug 2003-05 Turville Creek (n=244)
Frequency
● 0%
● >0% - 10%
● > 10%
>0% - 10% Regression
y = 1.5108x - 1.9548
R2 = 0.51269
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20

C

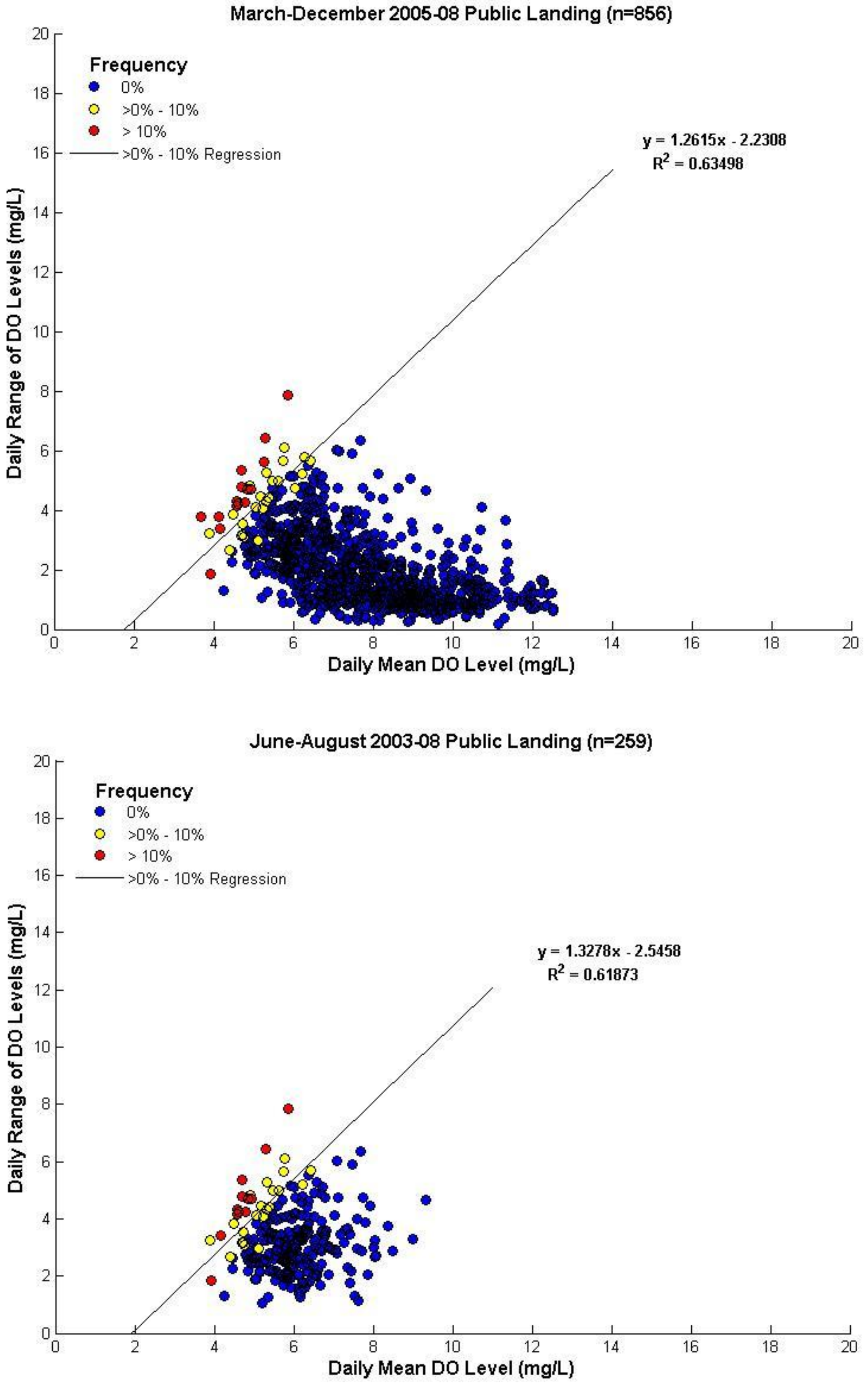
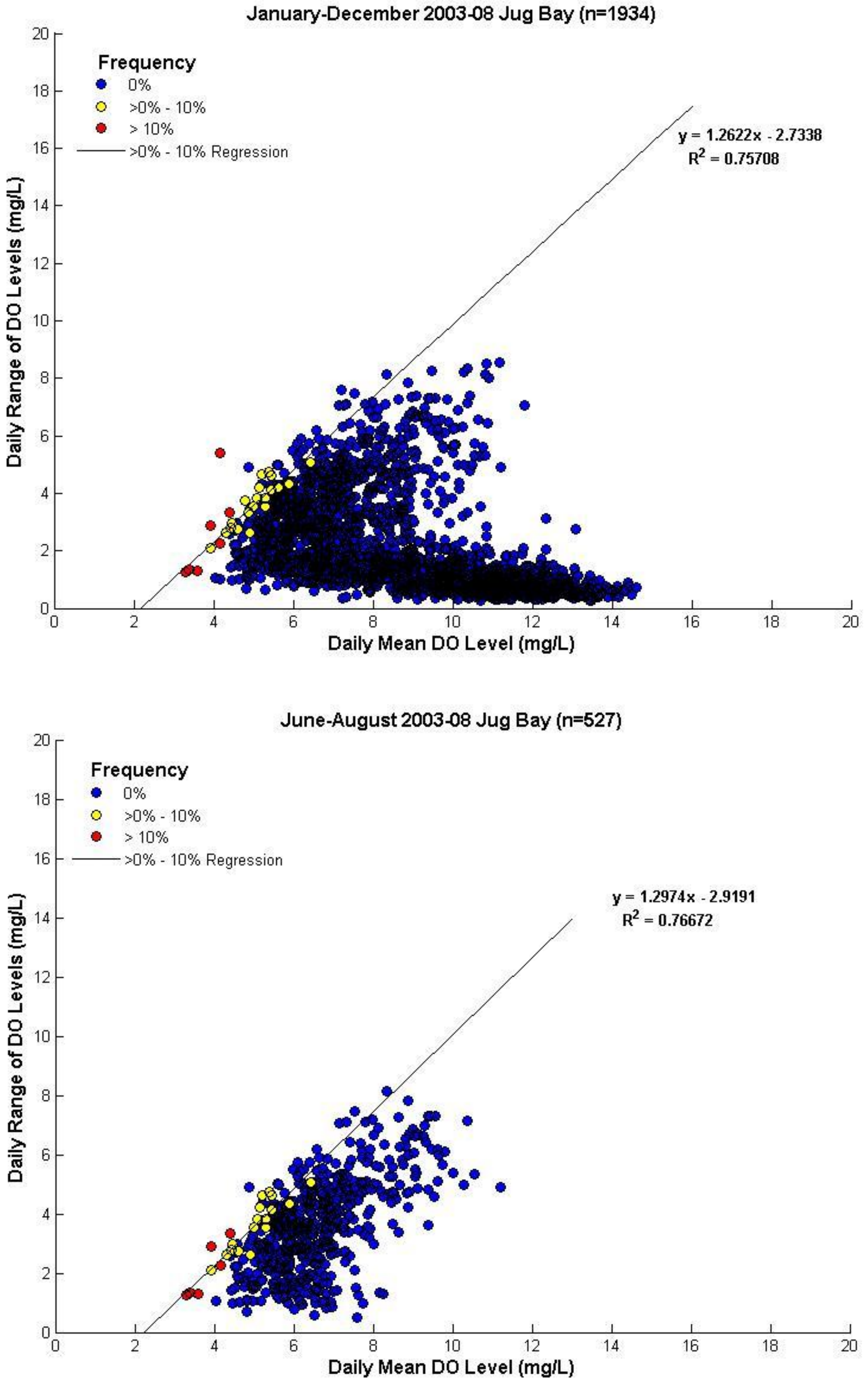

March-December 2005-08 Public Landing (n=856)
Frequency
● 0%
● >0% - 10%
● > 10%
>0% - 10% Regression
y = 1.2615x - 2.2308
R² = 0.63498
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20
June-August 2003-08 Public Landing (n=259)
Frequency
● 0%
● >0% - 10%
● > 10%
>0% - 10% Regression
y = 1.3278x - 2.5458
R² = 0.61873
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20

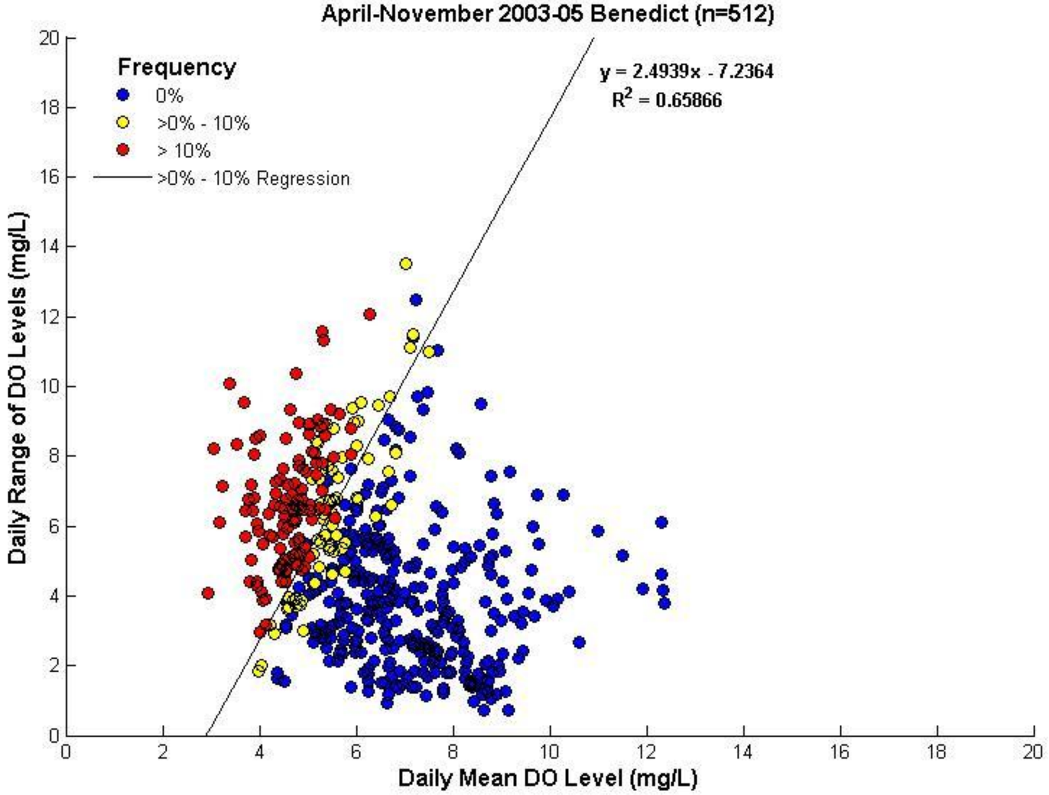
Figure 2-6 (previous three pages). Scatter plots of daily mean DO concentration versus daily DO range (max DO – min DO) based on ConMon data collected along a eutrophication gradient from the Maryland Coastal Bays (A) Bishopville Prong (B) Turville Creek (C) Public Landing. Blue dots represent days when there were no instantaneous DO criteria (DO < 3.2 mg L^{-1}) failures, yellow dots indicate days when there were less than 10% DO criteria failures and red dots indicate >10% DO criteria failures during a single 24 hour period. The regression line is based on the yellow dots and serves to separate the days passing and failing instantaneous DO criteria. The top panel includes all data during each year of measurement; the bottom panel only has data from June – August. In both cases, only days with a complete set of observations (n = 96) were used in these analyses.

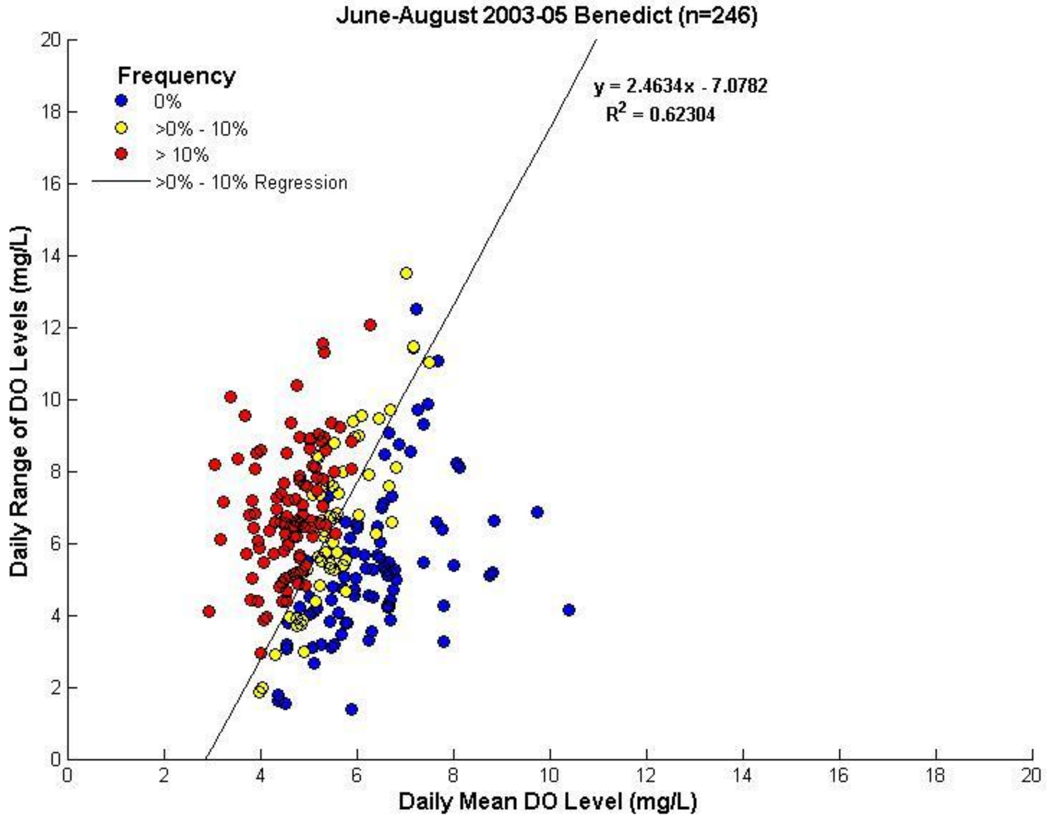
The second location with a series of ConMon sites located along a nutrient enrichment gradient is the Patuxent River where data from four ConMon sites were available ranging from tidal freshwater (Jug Bay), to low mesohaline (Benedict), to mesohaline (Pin Oak) to the mouth of the Patuxent (CBL). These data present a sharp contrast to the previous set of analyses (Figs 2-7a - d). In the upper Patuxent there were very few DO criteria failures, a strong indication of high DO values associated with small diel variation in DO concentration and DO failures mainly associated with low daily mean DO values with small diel variability. This zone of the estuary has high N and P concentrations and is in close proximity to a large WWTP point source discharge. It’s fair to ask why there are so few DO criteria failures at this site? At Jug Bay the water column is relatively shallow and well mixed (the tidal height maximum is close by), features that would tend to suppress large DO ranges due to effective re-aeration from the atmosphere. Second, this section of the estuary is very turbid (secchi disk < 0.5 m) and the water residence time is short (hours to days). Both of these factors would not favor phytoplankton accumulation and the effect algae have on DO dynamics. Finally, this zone is surrounded by tidal marshes and organic matter export from these marshes can be substantial. When this decomposing material gets to the river it would exert a DO demand and tend to reduce DO concentrations. A mean daily DO of only 5 mg L^{-1} would generally assure instantaneous DO criteria compliance at this site. At the Benedict site the estuary is deeper, wider and clearer. Phytoplankton production is enhanced at this site as we have shown in the Community Metabolism Chapter of this report. DO criteria failures are far more frequent, daily mean DO concentrations associated with criteria failures are also quite low (< 6 mg L^{-1}) and daily DO ranges were much larger than at the tidal freshwater site. The effect of phytoplankton communities (enhanced via nutrient enrichment) can be clearly seen in the increased daily DO ranges observed. At this site a daily DO mean of about 6 mg L^{-1} would generally ensure DO criteria compliance. At both the mesohaline (Pin Oak) and Patuxent mouth (CBL) sites DO criteria failure rates decline markedly (Figs 2-7c, d). The daily DO range was also considerably less than at Benedict at both Pin Oak and CBL. We interpret these results as indicating that the degree of enrichment, and associated phytoplankton biomass, was less than at Benedict resulting in generally better DO conditions.

A

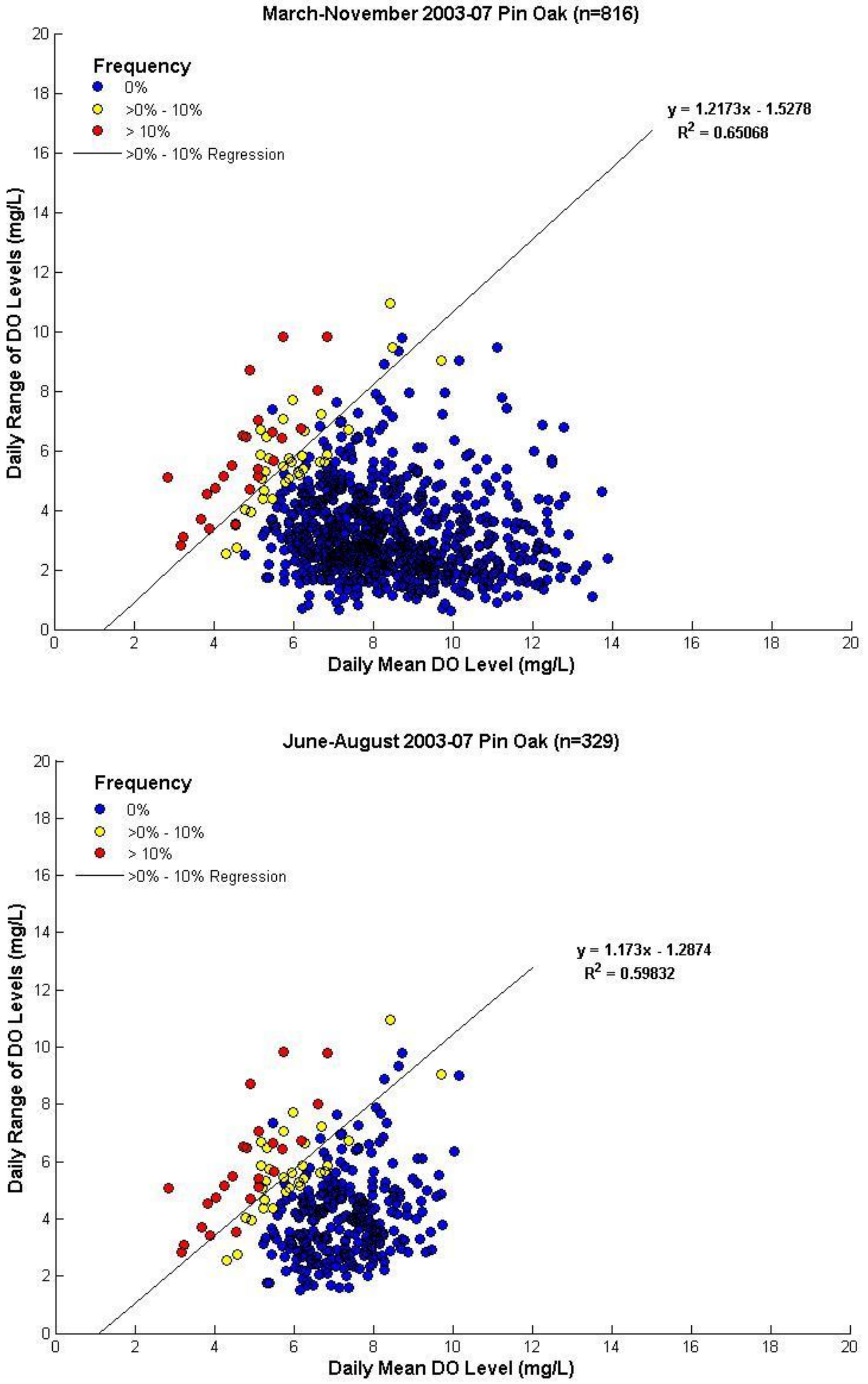

January-December 2003-08 Jug Bay (n=1934)
Frequency
● 0%
● >0% - 10%
● > 10%
>0% - 10% Regression
y = 1.2622x - 2.7338
R^2 = 0.75708
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20
June-August 2003-08 Jug Bay (n=527)
Frequency
● 0%
● >0% - 10%
● > 10%
>0% - 10% Regression
y = 1.2974x - 2.9191
R^2 = 0.76672
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20

B

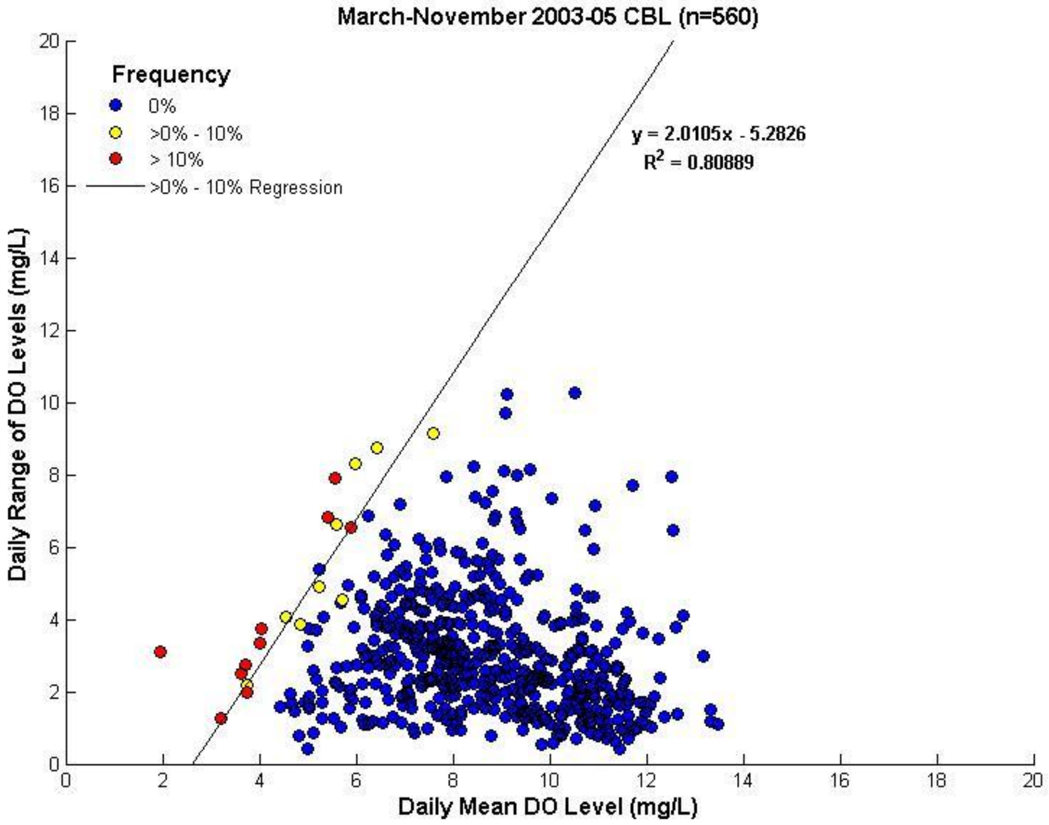

April-November 2003-05 Benedict (n=512)
Frequency
● 0%
● >0% - 10%
● > 10%
>0% - 10% Regression
y = 2.4939x - 7.2364
R² = 0.65866
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20


June-August 2003-05 Benedict (n=246)
Frequency
● 0%
● >0% - 10%
● > 10%
>0% - 10% Regression
y = 2.4634x - 7.0782
R² = 0.62304
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20

C


March-November 2003-07 Pin Oak (n=816)
Frequency
● 0%
● >0% - 10%
● > 10%
>0% - 10% Regression
y = 1.2173x - 1.5278
R^2 = 0.65068
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20
June-August 2003-07 Pin Oak (n=329)
Frequency
● 0%
● >0% - 10%
● > 10%
>0% - 10% Regression
y = 1.173x - 1.2874
R^2 = 0.59832
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20

D


March-November 2003-05 CBL (n=560)
Frequency
● 0%
● >0% - 10%
● > 10%
——— >0% - 10% Regression
y = 2.0105x - 5.2826
R² = 0.80889
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20

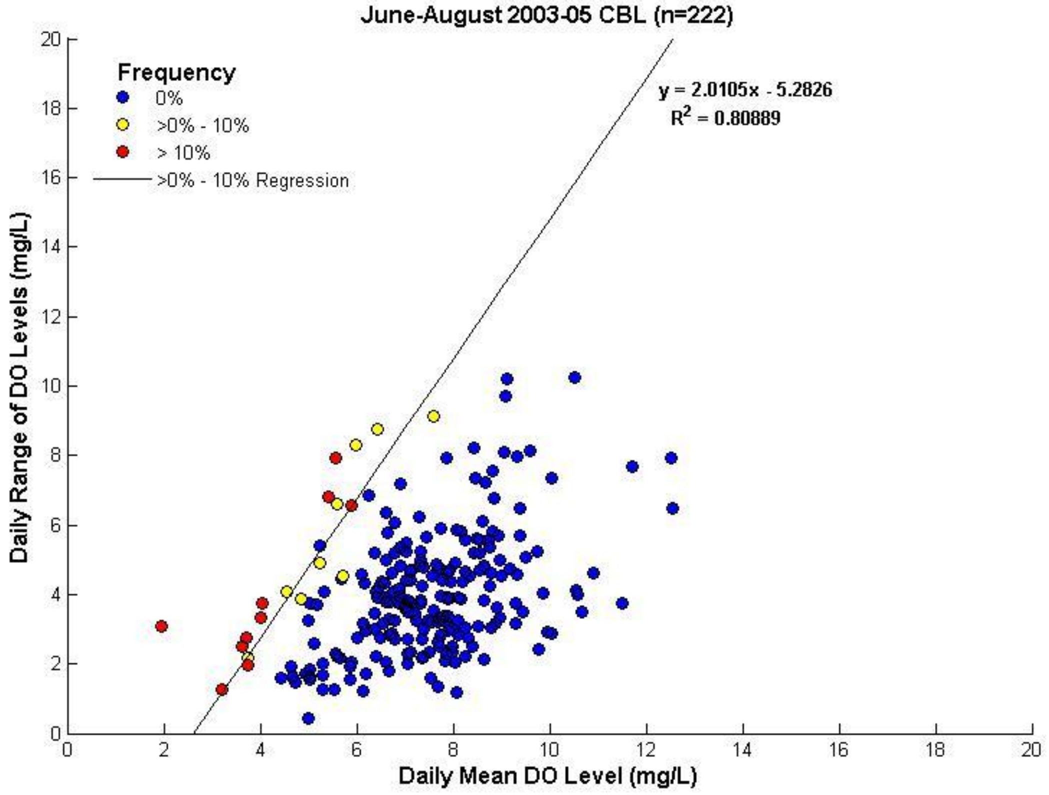
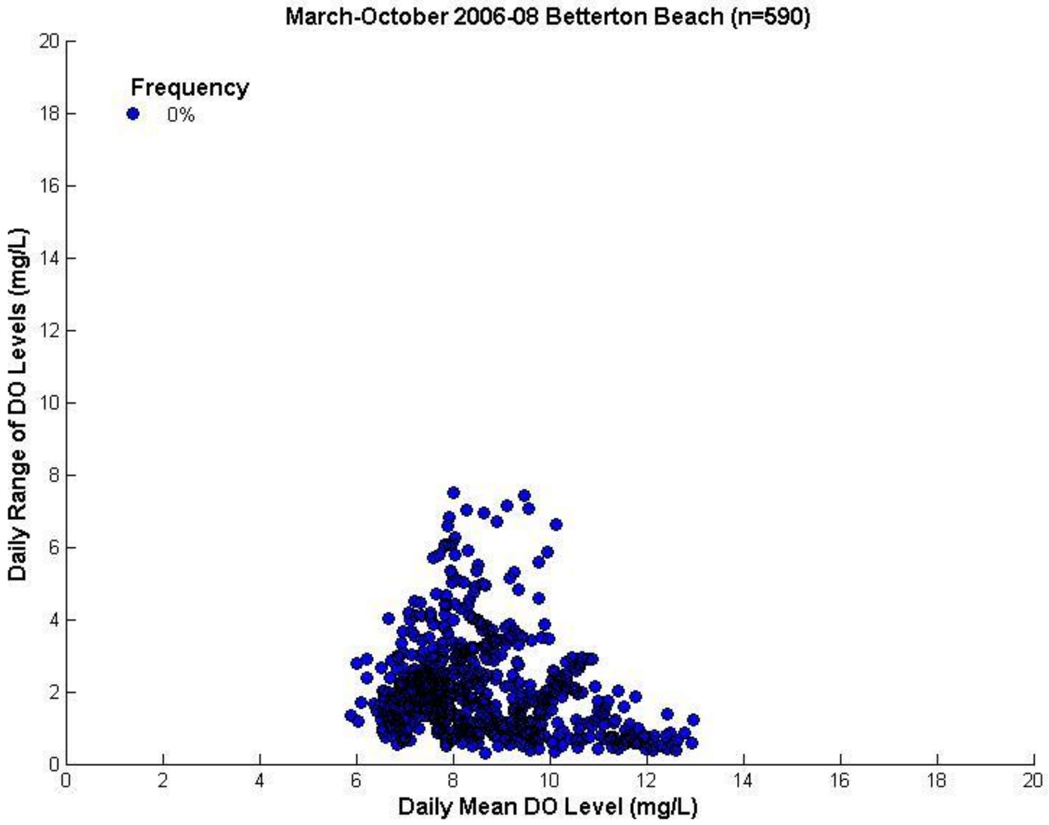
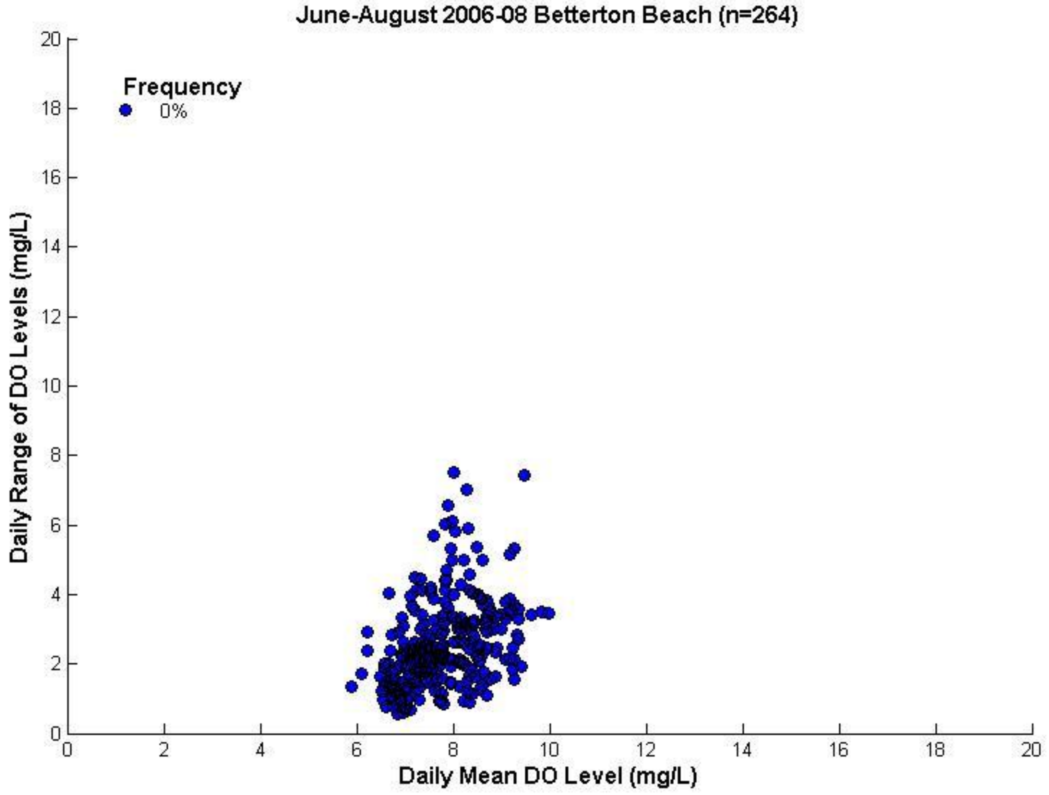

June-August 2003-05 CBL (n=222)
Frequency
● 0%
● >0% - 10%
● > 10%
——— >0% - 10% Regression
y = 2.0105x - 5.2826
R² = 0.80889
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20

Figure 2-7 (previous three pages). Scatter plots of daily mean DO concentration versus daily DO range (max DO – min DO) based on ConMon data collected along a eutrophication gradient along the axis of the Patuxent River estuary (A) Jug Bay (B) Benedict (C) Pin Oak and (D) CBL. Blue dots represent days when there were no instantaneous DO criteria (DO < 3.2 mg L^{-1}) failures, yellow dots indicate days when there were less than 10% DO criteria failures and red dots indicate >10% DO criteria failures during a single 24 hour period. The regression line is based on the yellow dots and serves to separate the days passing and failing instantaneous DO criteria. The top panel includes all data during each year of measurement; the bottom panel only has data from June – August. In both cases, only days with a complete set of observations (n = 96) were used in these analyses.

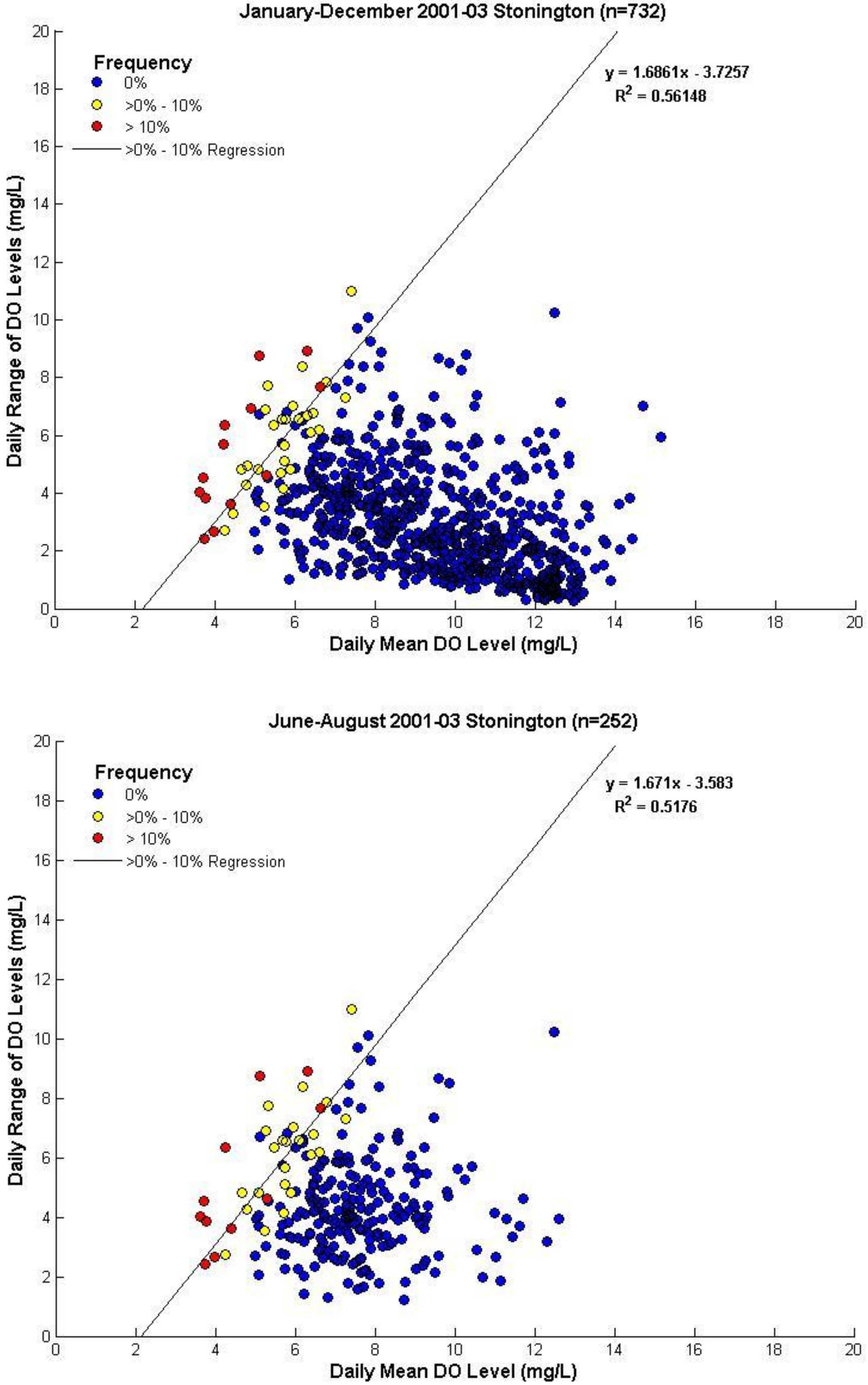
The final selection of sites (Figs. 2-8a-d) aimed to examine DO dynamics under a variety of conditions including exposure to large, open water bodies (as opposed to sites in small tributaries of tributaries) and from sites having unusual characteristics (e.g., high, naturally occurring organic matter content). The first two sites were located in the upper Bay near the mouths of the Sasafrass (Betterton beach) and Magothy (Stonington) rivers and the third site (Fenwick) was located along an exposed shoreline of the upper Potomac River. These sites represent “more exposed” shallow water sites and all are proximal to major nutrient sources (Susquehanna and Potomac River outlets). DO mean and diel range at the Betterton Beach site ranged from about 6 to 13 mg L^{-1} and less than 1 to 8 mg L^{-1} day^{-1}, respectively. Mean DO values did not depart markedly from saturation values at this site. There were no instantaneous DO criteria failures at this site during the 2006-2008 periods. Data from this site, especially during the critical summer period (bottom panel; Fig. 2-5a) conform to the expected distribution of a “lean” versus an “obese” site relative to nutrient enrichment. Specifically, DO mean values did not depart markedly from saturation values and diel DO ranges were modest (most < 4 mg L^{-1} day^{-1}). While there were some DO criteria failures at the Stonington and Fenwick sites, failures were relatively rare (Figs. 2-5b and 5c). The final site selected was from Rehobeth in the Pocomoke River. This is an unusual site in that the Pocomke River is naturally high is dissolved organic carbon (DOC); it is one of the few “black water” rivers of the Chesapeake system. The DO characteristics of this site also differ from other sites examined in several ways. First, mean daily DO concentrations were often low, sometimes very low. The range in mean concentration ranged from less than 2 mg L^{-1} to 8 mg L^{-1} during the period May – November. In addition, the daily range in DO concentration was quite small, mainly < 4 mg L^{-1} day^{-1} even during summer periods. Finally, serious DO criteria failures were almost always associated with both very low mean DO values and with very small daily DO ranges. This is the type of “DO profile” we would expect from an organic-rich system rather than a nutrient enriched system. A variety of forested wetland systems fringe the Pocomoke and supply large amounts of DOC to the river. DOC concentrations in this river are the highest observed in the Bay system. In the case of the Pocomoke it appears that respiratory processes dominate (i.e., the system is strongly heterotrophic), leading to relatively low DO mean values. In addition, phytoplankton communities are not a large feature of black water rivers probably because of light limitation and possibly because of deep vertical mixing in the Pocomoke. Because of limited phytoplankton activity the diel DO range is suppressed as well. This may be one of those cases where low DO is the norm and little can be done to change the DO signature of this system.

A

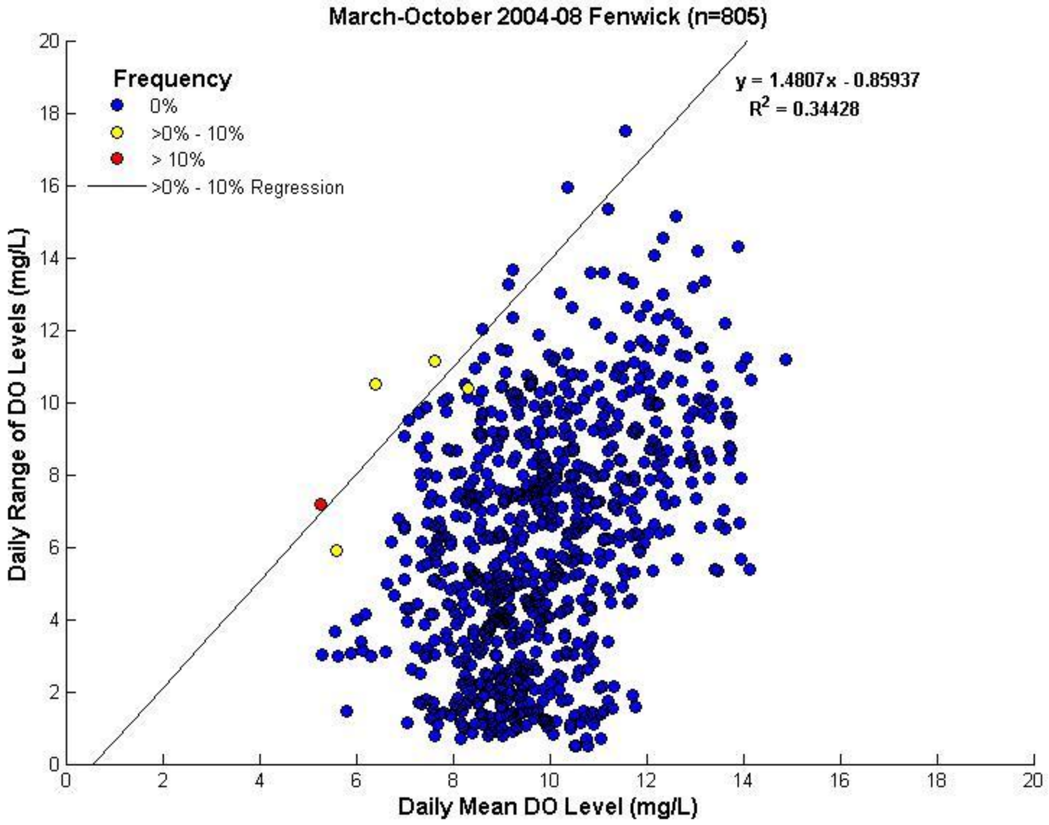

March-October 2006-08 Betterton Beach (n=590)

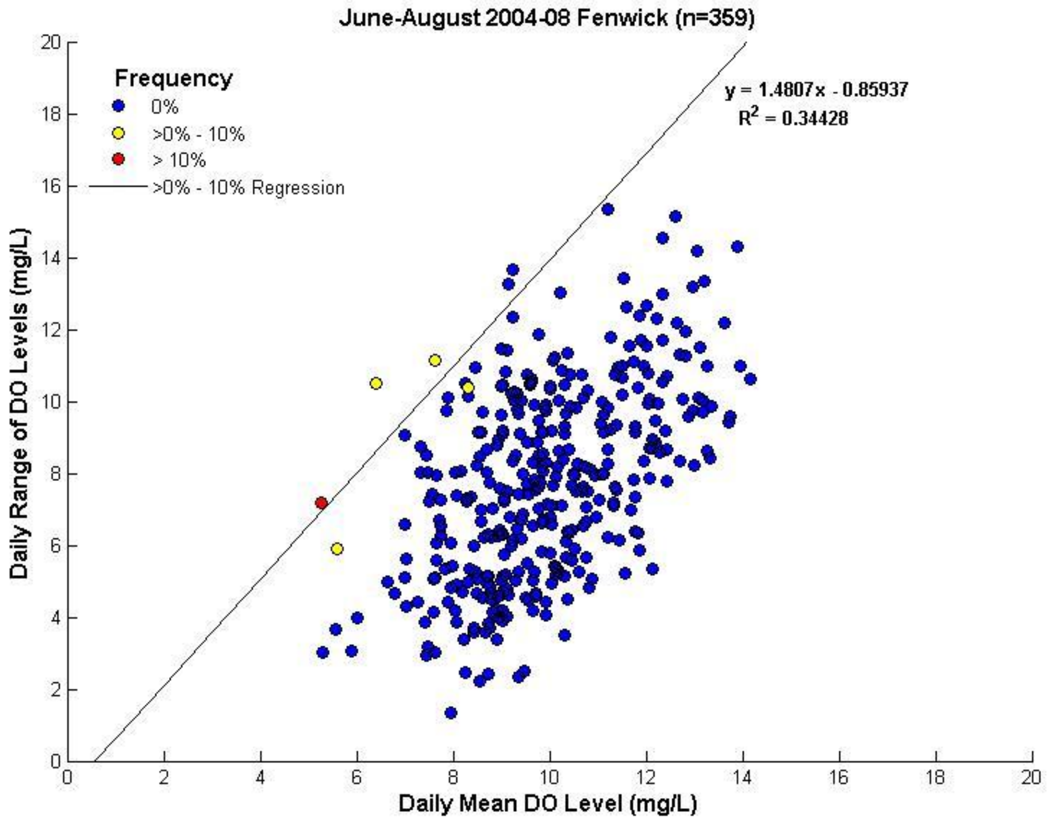

June-August 2006-08 Betterton Beach (n=264)

B

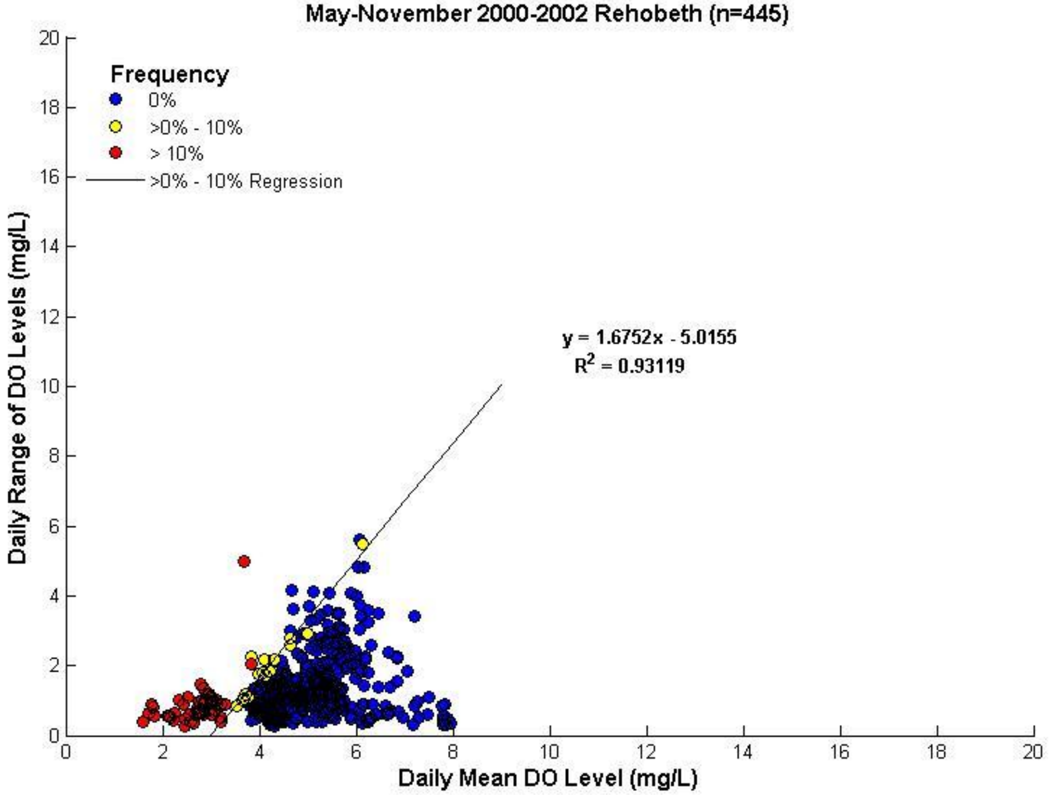

January-December 2001-03 Stonington (n=732)
Frequency
0%
>0% - 10%
> 10%
>0% - 10% Regression
y = 1.6861x - 3.7257
R2 = 0.56148
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20
June-August 2001-03 Stonington (n=252)
Frequency
0%
>0% - 10%
> 10%
>0% - 10% Regression
y = 1.671x - 3.583
R2 = 0.5176
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20

C


March-October 2004-08 Fenwick (n=805)
Frequency
0%
>0% - 10%
> 10%
>0% - 10% Regression
y = 1.4807x - 0.85937
R^2 = 0.34428
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20


June-August 2004-08 Fenwick (n=359)
Frequency
0%
>0% - 10%
> 10%
>0% - 10% Regression
y = 1.4807x - 0.85937
R^2 = 0.34428
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20

D


May-November 2000-2002 Rehobeth (n=445)
Frequency
0%
>0% - 10%
> 10%
>0% - 10% Regression
y = 1.6752x - 5.0155
R2 = 0.93119
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20

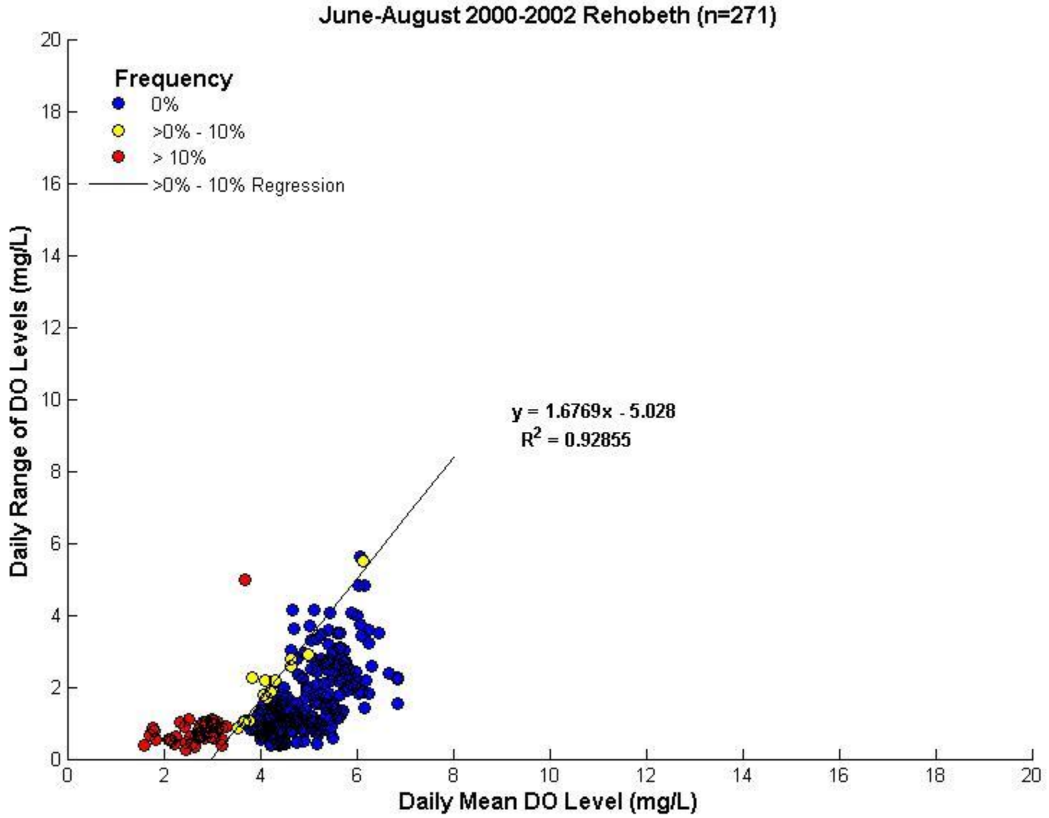

June-August 2000-2002 Rehobeth (n=271)
Frequency
0%
>0% - 10%
> 10%
>0% - 10% Regression
y = 1.6769x - 5.028
R2 = 0.92855
Daily Range of DO Levels (mg/L)
Daily Mean DO Level (mg/L)
0
2
4
6
8
10
12
14
16
18
20

Figure 2-8 (previous four pages). Scatter plots of daily mean DO concentration versus daily DO range (max DO – min DO) based on ConMon data collected from a selection of more exposed sites in Chesapeake Bay (A) Betterton Beach at the mouth of the Sasafrass River (B) Stonington at the mouth of the Magothy River (C) Fenwick in the upper Potomac River and (D) Rehobeth in the upper Pocomoke River. Blue dots represent days when there were no instantaneous DO criteria (DO < 3.2 mg L^{-1}) failures, yellow dots indicate days when there were less than 10% DO criteria failures and red dots indicate >10% DO criteria failures during a single 24 hour period. The regression line is based on the yellow dots and serves to separate the days passing and failing instantaneous DO criteria. The top panel includes all data during each year of measurement; the bottom panel only has data from June – August. In both cases, only days with a complete set of observations (n = 96) were used in these analyses.

Thus far we have examined DO dynamics at the daily time scale for a variety of Chesapeake sites. In the next year we will continue examining additional sites and focus more on examining the "sentinel sites" where a longer time-series of high frequency data have been accumulated. However, further examination of sites will likely produce few new insights. What appears to be needed are abilities to both generalize the shallow water DO data and to link the DO data collected at ConMon sites to management actions aimed at nutrient load reduction.

We have initiated work on both of these issues. First, we have started to develop a statistical model that will generate estimates of daily mean and range in DO for Bay locations not having ConMon sites. From inspection of data from many ConMon sites it seems that several (2-3) independent variables may be sufficient to develop such a model. First, it appears that large diel DO ranges are associated with large algal (or SAV) communities. Thus, water column chlorophyll-*a* seemed like a very likely candidate as an important independent variable in a model. In earlier work, using data far less refined than ConMon data, Boynton et al (1996) found a strong relationship between daily average chlorophyll-*a* and average daily DO rate of change (Fig. 2-9). This analysis utilized data collected along nutrient enrichment gradients in the Maryland Coastal Bays and strongly supports using chlorophyll-*a* in a statistical model. An additional arguement for selecting chlorophyll-*a* as an important variable is that chlorophyll-*a* data are available from many locations in the Bay and tributary rivers, has been collected for many decades and is available from several measurement platforms, each having distinctive time and space-scales of collection (e.g., fixed station, ConMon, Dataflow).

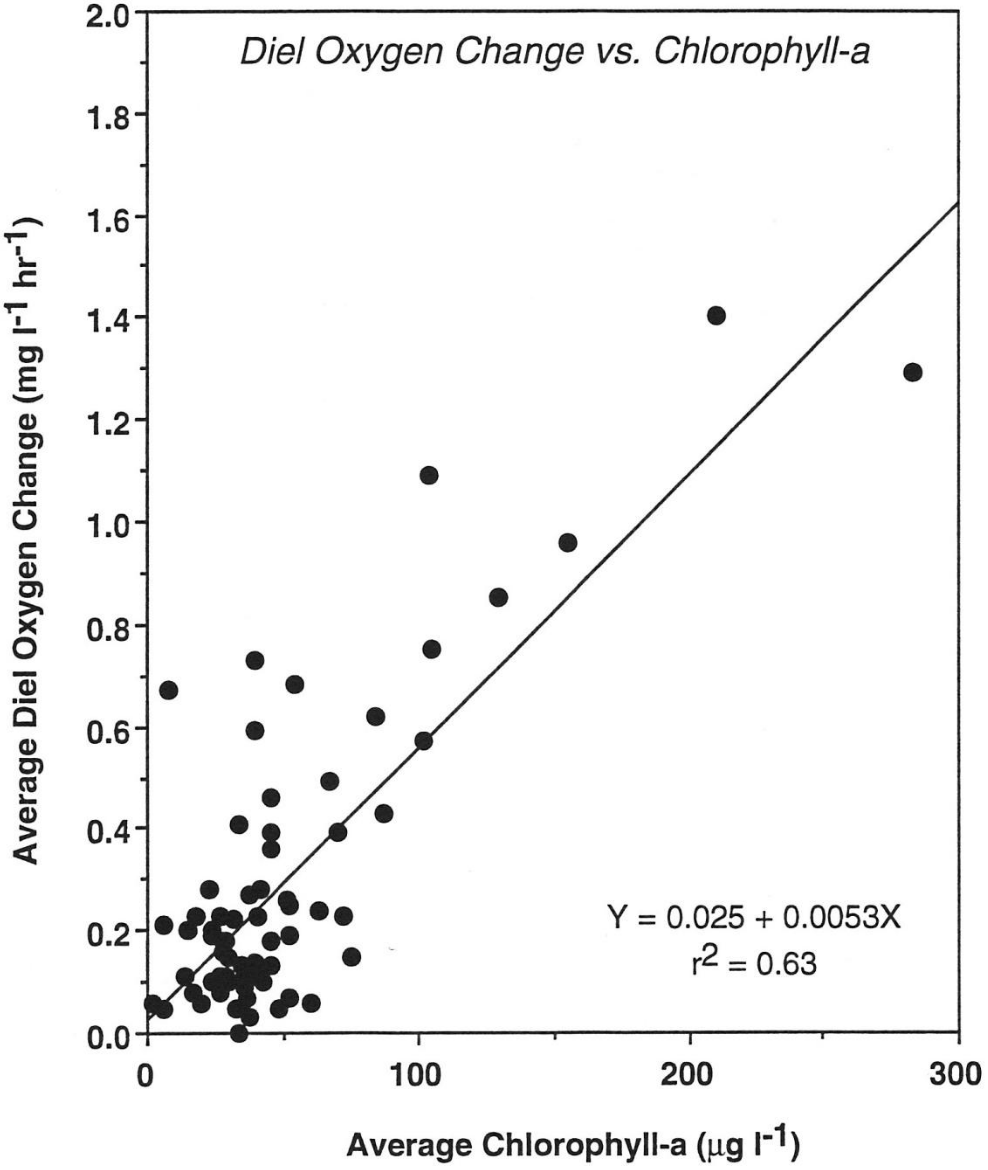


Figure 2-9. A scatter plot of average daily chlorophyll-*a* versus average daily DO rate of change. Data were collected from a variety of sites in the Maryland Coastal Bays. Figure was taken from Boynton *et al.* (1996).

A second variable, important at the daily time-scale, appears to be light availability or photosynthetically active radiation (PAR). The diel DO range appears to be sensitive to PAR in that on cloudy days light is not sufficient to generate high rates of photosynthesis and the diel DO range is suppressed. We have initiated inspection of diel DO range and PAR and it appears that PAR has a strong influence of DO range (Fig. 2-10). In this figure PAR was relatively high for 6 successive days and DO range was also relatively high. However, PAR dropped by over a factor of 2 on the seventh day and DO diel range also decreased. We have seen similar patterns in other data sets sufficient to conclude that PAR needs to be part of the statistical model. Finally, it appears that temperature needs to be included, in part because of effects on both respiration (DO loss from the water column) and photosynthesis (DO gain in the water column) and as a variable tracking time of year. For the most part, large diel DO ranges are not observed during spring, fall or winter. A temperature variable would track that effect in the model. The advantages of producing such a model include the before mentioned ability to generalize ConMon data to many sites in the Bay. Second, the model, if successful, is relatively simple and there are good conceptual reasons for using the variables selected. However, there are issues to be considered. This effort would require considerable ConMon data manipulation in production of daily temperature and chlorophyll-*a* means and ranges, matching PAR data with appropriate days and deciding on a smoothed (solid line in Fig. 2-7) or empirical (open circles in Fig 2-10) approach to estimating DO diel range. It seems that all of these issues can be resolved and we plan to continue work on this model.

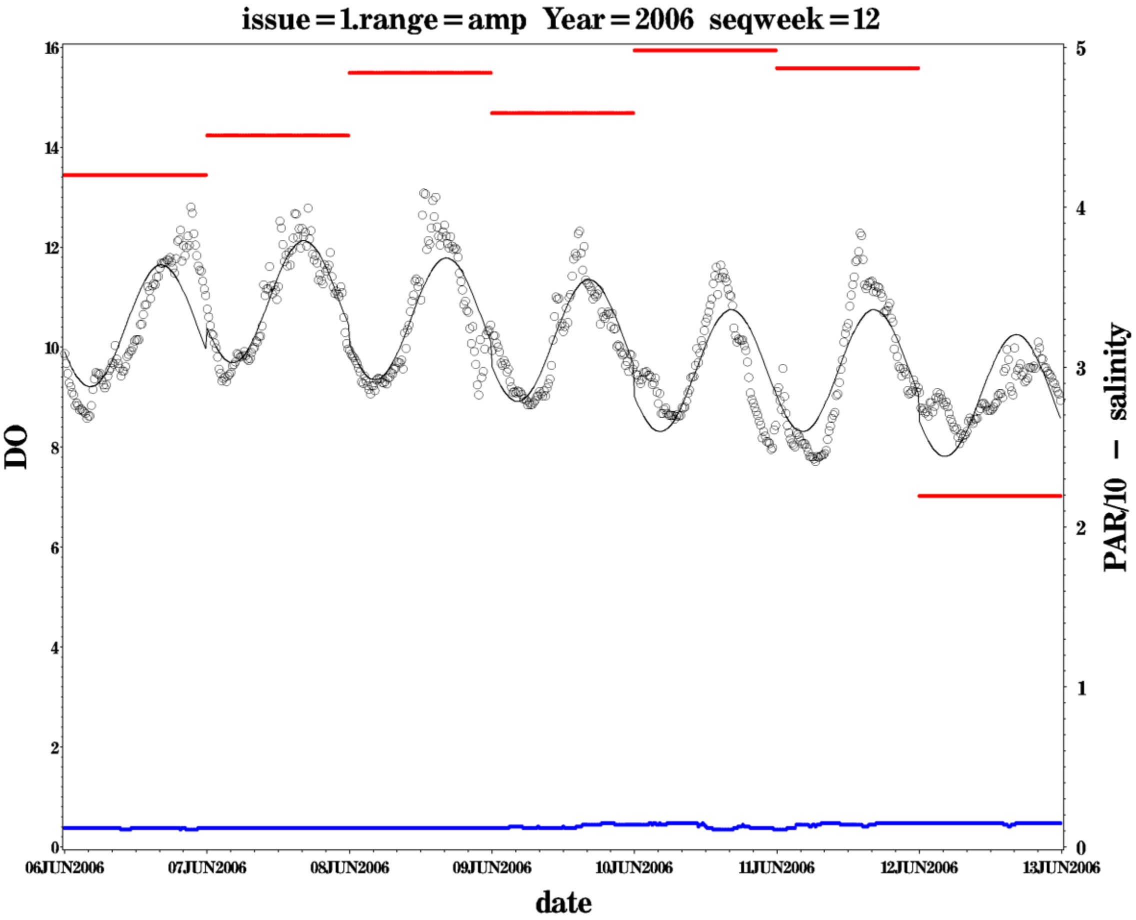


Figure 2-10. A time-series plot of high frequency (15 minute intervals) DO measurements (open circles) and modeled diel DO concentrations (solid line) for a 7 days period. Photosynthetically active radiation (PAR) for each day is shown as a red bar at the top of the figure. Data were from the Public Landing ConMon site in the Maryland Coastal Bays.

Finally, this model has only an indirect link to the key management effort in the Bay…nutrient loads and load reductions. As the model stands now, chlorophyll-*a* is the link to the key management issue and it is indirect to the extent that enhanced nutrient availability promotes production of more chlorophyll-*a*. The model does not have a nutrient availability term. One way to better confirm the nutrient - chlorophyll relationship would be to also pursue a nutrient-chlorophyll-*a* model as was done in the recent Corsica River work (Boynton *et al*. 2009) and other similar efforts (Testa *et al.* 2008) in the Patuxent River.

The recent TWAW workshop and TMAW deliberations in the past year have also struggled with how shallow water DO dynamics could be coupled with the CBP Open Water designated area. This remains an open question and is an important issue. The few comparisons that have been made between adjacent shallow water and open water sites seem to indicate that diel-scale DO ranges are larger in shallow waters than in adjacent open waters. This has important habitat implications and eventually needs to be resolved.

2-5 Future Analyses and Issues

During the next contract period we will continue to examine the following issues:

1. Additional ConMon sites will be examined for DO criteria compliance/failure and we will continue to examine relationships between DO average values and diel range as an additional criteria assessment tool.
2. We will continue to develop a DO assessment based on a statistical model using several readily available variables. Considerable progress on this issue has already been achieved.
3. We hope that the statistical model will play a role in unifying shallow water and open water assessment tools (i.e., modeled DO results can then be used in locations not having ConMon data).
4. Perhaps the most challenging issue will involve more directly relating DO criteria assessments to nutrient loading rates and changes in nutrient loading rates associated with management actions. We fully appreciate the need to detect changes in water quality associated with management actions during the early phases of the Chesapeake Bay TMDL effort.

2-6 References

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Chapter 3

Community Metabolism: Use of ConMon Data for Understanding the Degree of Remediation Needed for Improved Water Quality

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

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 2008). 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 and from several sites fronting on the open Bay or along the shoreline of major 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. An analysis of patterns of community production (P) and respiration (R) for a mesohaline site on the Patuxent River; the historical nature of this data set serves as a reference condition type of analysis as well as for DO criteria assessment work.
2. An analysis of community rates of P and R along two estuarine eutrophication gradients, suggesting approaches for directly linking these rates to nutrient loading rates
3. A summary of mean rates of community P and R for multiple sites in the Maryland Bay and tributary rivers, again qualitatively relating these rates to nutrient loading rates
4. Consideration of the effects of location (exposed shorelines of large systems versus shorelines of smaller and less physically exposed sites) on rates of Community P and R
5. Continue work on a format for translating these data to a web page geared to be used 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 at least a 24 hour period. During hours of daylight, oxygen increases in the water due to the release of O_2 as a by-product of photosynthesis. During hours of darkness, O_2 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 (Personal Communication). 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”.

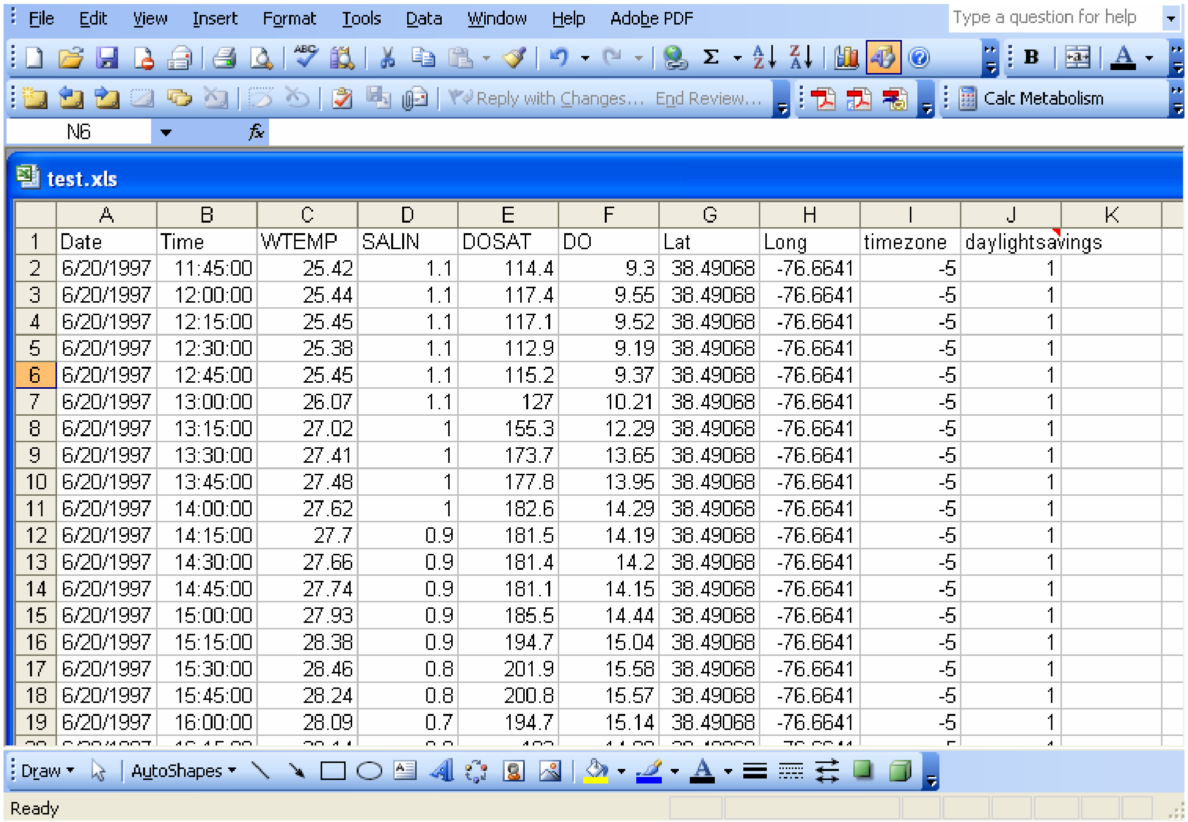
10. Sums of the changes in DO, time, air/sea exchange and DO flux (step 8) are calculated for each metabolic day for the periods between sunrise and metabolic dawn, metabolic dawn and metabolic dusk, metabolic dusk and sunset, and sunset and the following sunrise.

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_2m^{-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 requisite input format needed by 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

There were 4 sources of data used in this section. The first was the historic Cory data set collected from 1963 through 1969 at the Maryland Route 231 Bridge near Benedict, MD (Fig. 3-1). These data (surface water temperature, salinity and dissolved oxygen) were converted from graphic (strip chart) to electronic format (.pdf and .xls) and the details of that process were provided in our last Interpretive Report (Boynton et al 2010). The second and third sources of data were collected from the same location (MD Route 231 Bridge) by Sweeney (1995) and Burger and Stankelis (1996-1998). These data (.xls format) were converted and QA/QC'd for use in the Metabolism macro. All of these data sets (.xls format) are now available at the following web site (http://www.gonzo.cbl.umces.edu) as are all of the original Cory strip-chart data (.pdf format). All other data (ConMon) were obtained from from the Maryland DNR (MDDNR) ConMon program (Cole 2010) and are available from the following website (http://mddnr.chesapeakebay.net/eyesonthebay/index.cfm). The MDDNR ConMon location was located at the same point as the bridge along the axis of the estuary, but was located on a pier on the western shore of the river rather than on the central platform of the bridge. The original site and the ConMon site are about 500 m apart. Data source information and quality assurance information are provided in Table 3-1.

Table 3-1. Data source reference, quality assurance information and descriptions.

Source	Reference	Description and Quality Assurance Information
Cory	Cory and Nauman (1967, 1968, & 1971)	The 1963-1969 Cory data sets were sorted to find missing values. Any row with a missing value was deleted entirely to avoid errors in the macro. A –met file was created for the "clean" data to run through the Metabolism Macro.
Sweeney	Sweeney (1995)	The 1992 data came from Brendan Sweeney's master's thesis. The parameters needed to run the Metabolism Macro, were copied into a new excel spreadsheet. For QA/QC the data was sorted to find missing values. Any row with a missing value was deleted entirely to avoid errors in the macro.
Burger & Stankelis	Burger and Stankelis (1996, 1997 and 1998)	The parameters needed to run the Metabolism Macro, were copied into a new excel spreadsheet. For QA/QC the data was sorted to find missing values. Any row with a missing value was deleted entirely to avoid errors in the macro.
ConMon	www.eyesonthebay.net and B. Cole (2010)	Continuous monitoring data acquired in electronic format from Maryland DNR (Cole 2010) was sorted (using SAS©) according to codes outlined in the MDDNR QAPP (2009). Any fields with failing error codes were not used in the Metabolism Macro. In addition, any rows with missing values were deleted.

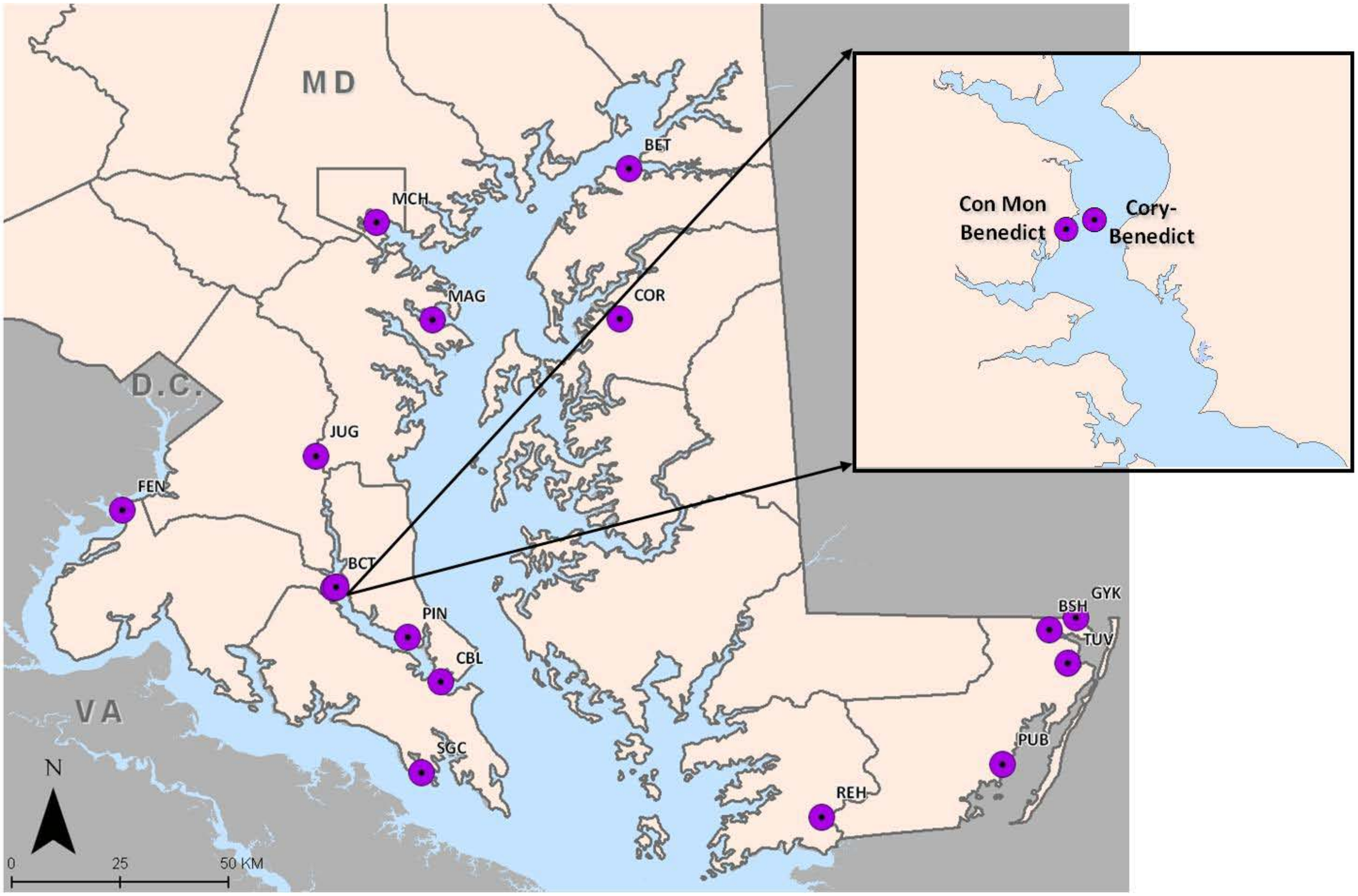


Figure 3-2. A map showing the location of ConMon sites used in all analyses in this report. The location of the historical data collection site on the Maryland Route 231 bridge near Benedict, Maryland is also shown (Patuxent River estuary).

3-4 Results and Discussion

3-4.1 Historical Record of Metabolism: Patuxent River Estuary

We start this discussion by presenting the annual-scale results of community metabolism measurements made in the mesohaline region of the Patuxent River estuary at the Maryland Route 231 Bridge at Benedict, MD from 1964 – 2005 (41 year period). The significance of this data set is that it covers the period before major developments started in this river basin (1964-1966), the period of rapid and largely uncontrolled development (1968 – 1990) and the recent period of more development (1991 – 2005) but with more environmental controls in place (e.g., WWTP upgrades, critical area laws in place, generally more sediment erosion controls, placement of riparian buffers).

Rates of gross community production (Pg*) increased from about 3.7 g O_2 m^{-3} day^{-1} in 1964-1965 to almost 9 g O_2 m^{-3} day^{-1} in 2003, an increase by a factor of 2.4. Thus, the rate of carbon fixation, mainly by planktonic algae, has more than doubled during these multiple decades (Fig. 3-3a). A simple regression analysis indicates a linear increase in production of about 0.11 g O_2

m^{-3} $year^{-1}$ ($r^2 = 0.91$; $p < 0.05$). Rates at the beginning of this period were modest, typical of a healthy estuarine system. However, by the 2004 – 2005 period rates were more typical of a eutrophic system, although they were less than those measured in very eutrophic systems such as the Back River or upper portions of the Corsica River estuaries (Boynton et al. 1998 and Boynton et al. 2009).

All measurements used in this analysis were made at the same location (on or adjacent to the Benedict Bridge) using the same measurement approach (high frequency measurements of salinity, temperature and dissolved oxygen in surface water) except the measurements made during the late 1970s. These metabolic estimates were based on light/dark bottle measurements and were made several kilometers upriver of the Benedict Bridge. In addition, there were relatively few measurements made and these were largely made during summer periods. Thus, these data need to be used cautiously, if at all. One reason for including these here is that if they are roughly comparable to the rest of the data set, they suggest a period of rapidly increasing community metabolism during the period when loading rates were sharply increasing (Hagy et al. 1998) and before any of the WWTPs had adopted either P (1986) or N (1992-1993) removal procedures. The late 1970s data, as well as the increased rates indicated in the late 1960s (a period when nutrient loads first began increasing), suggest this system to be responsive to nutrient loading rate modifications. This is an important management-relevant conclusion.

Community respiration rates (Fig. 3-3b) exhibited a similar pattern of increase. Rates during the early 1960s were about 1.8 g O_2 m^{-3} $night^{-1}$ and increased to about 3.5 g O_2 m^{-3} $night^{-1}$ by the mid-2000s, an increase of about a factor of two. The limited data from the late 1970s did not exhibit an increase and is perhaps another reason to view data from that period with caution. The rates of community respiration observed during the mid-2000s are indicative of those observed in other eutrophic systems. Boynton and Bailey (2008) have compiled water column respiration data for a large number of Chesapeake Bay systems and these data support the above conclusion.

To better view the changes in community metabolic rates during the period of record we have plotted data from the beginning (1964) and end (2005) of the record on the same figure, one averaged to month (Fig 3-4a) and one to weeks (Fig 3-4b). In this format the changes become even more apparent. In addition, there appears to have been a change in the seasonal pattern of production (Pg*) between the early and recent data sets. During the early period, peak rates were observed earlier in the year (May – July) and declined thereafter. However, in the recent data set rates peaked in July but were also very high in both June and August. Thus, the period of very high production has shifted to later in the summer. In fact, the most extreme difference between the early and recent data sets occurred during August where recent rates were 3.4 times larger than early rates. We suggest that the shifting of periods of peak production from spring-early summer to mid-late summer is another metric of nutrient enrichment. The rich ConMon data set could be used, over time and in association with management actions, to assess periods of peak production and watch for shifts back to late spring periods of peak production. While we have no experimental data to explain the time shift in peak production, it does appear that this shift is

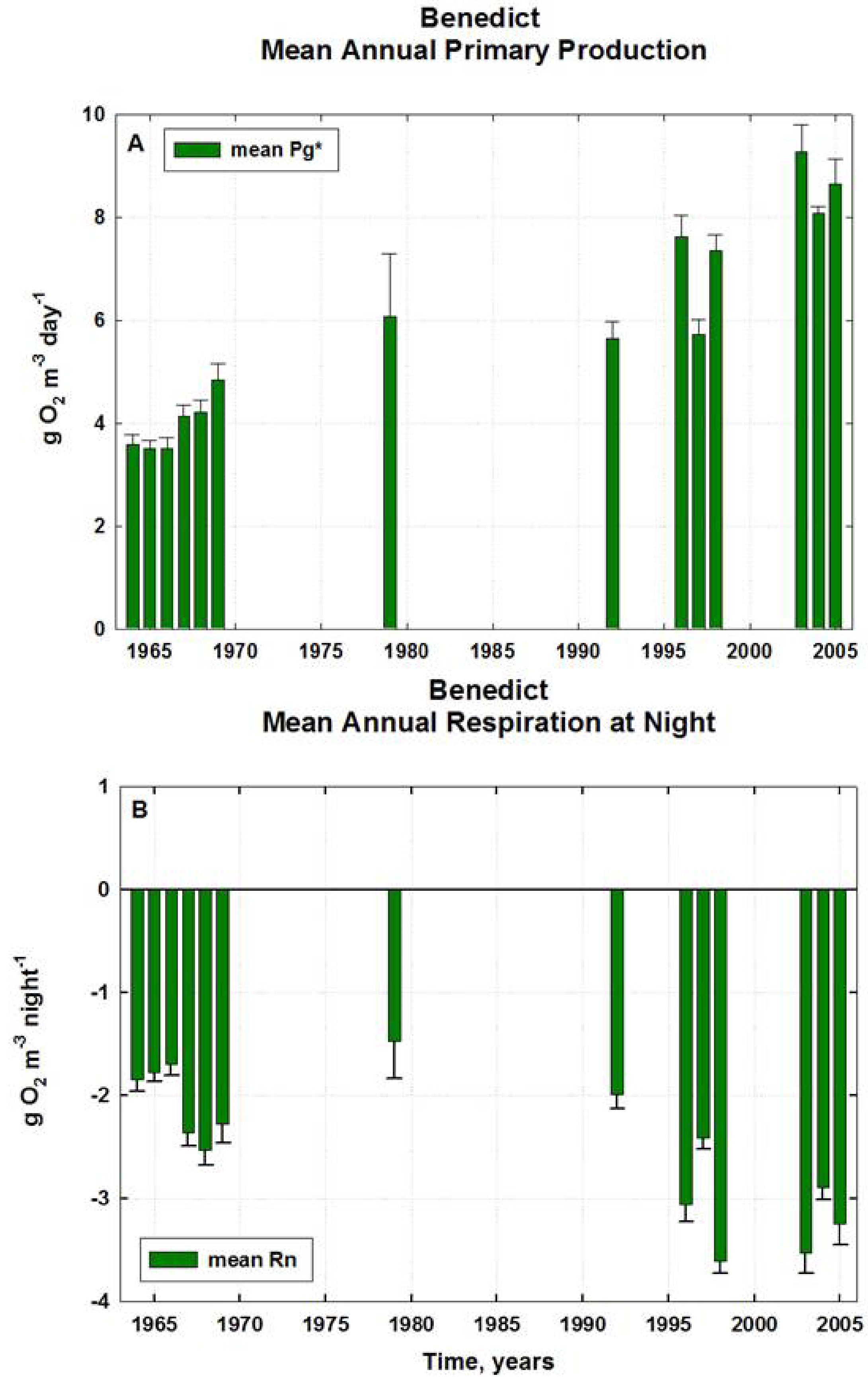


Figure 3-3. Bar graphs (mean and standard error) of mean annual gross primary production (a) and community respiration (b) collected at the Benedict Bridge site between 1964 and 2005. For most years data were available from April – October. Data from the late 1970s were collected using a light-dark bottle approach; the late—1970s data need to be viewed with caution.

consistent with the degradation trajectory shown in Figure 1-1 (Chapter 1). In this case, during the low load period, nutrients associated with spring run-off were supported by the spring diatom bloom and were thus retained, as particulate organic nitrogen, within the estuary. Some fraction of this diatom bloom was assimilated by long-lived organisms (fish and benthic communities), some was denitrified (sediments were well oxygenated during the early period and were good sites for this N-loss process), some was used by SAV and benthic micro-algal communities and some N was recycled to support summer production. However, the summer recycle was limited because of the other loss processes. By mid to late-summer N reserves from the spring freshet were depleted and production rates decreased. However, in recent times, all of the N loss processes in the mesohaline estuary have been severely compromised or lost. There is a good deal of nitrogen retained in the system from the now much larger diatom bloom during late winter and spring and there is also much more N available for recycling during the warm summer periods. As a result, high rate of community production can be maintained from July through September. If nutrient loads decrease substantially, we would predict both a decrease in rates and a shift in peak rates back toward spring periods.

To provide a visual and quantitative example of the restoration task before us, we have taken the community metabolism rates measured during 1964 at Benedict, Maryland and overlaid these on metabolism rates measured in a very nutrient enriched system (upper Corsica River station Sycamore Point; Fig 3-5). Rates of both production and respiration at Sycamore Point greatly exceed those measured at Benedict during the early 1960s, often by almost an order of magnitude. In addition, it is clear here, even in this small tributary system, very high rates, some greater than 20 g O_2 m^{-3} day^{-1}, occur during a broad portion of the year (June – August) with sustained high rates in July and August. In sharp contrast, the low nutrient load conditions at Benedict during 1964 resulted in much lower rates with peak rates occurring earlier in the year (May – July).

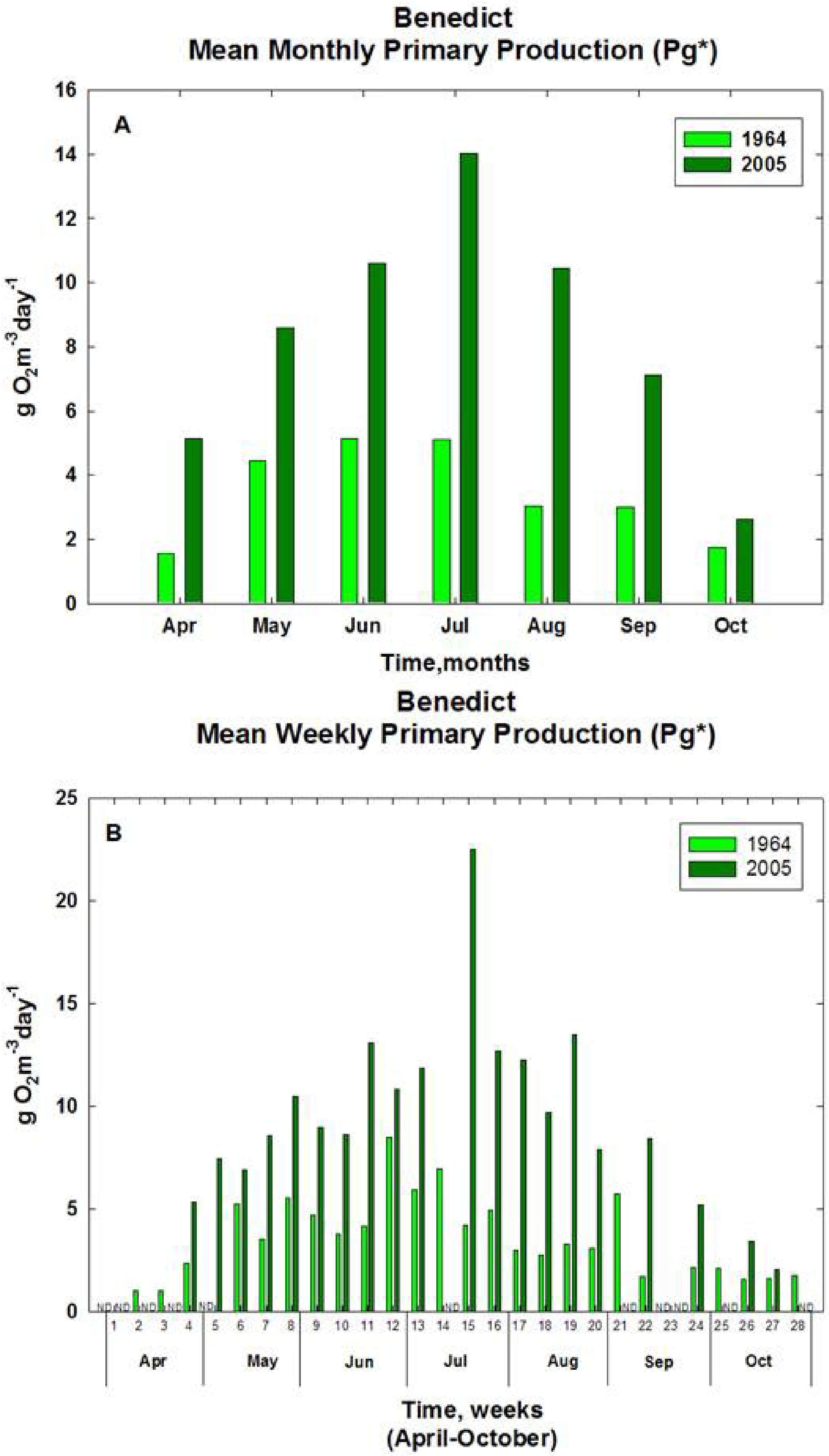


Figure 3-4. Bar graphs of mean monthly gross primary production (A) and weekly mean gross primary production (B) for 1964 and 2005. Data were collected at the Benedict Bridge site on the Patuxent River estuary.

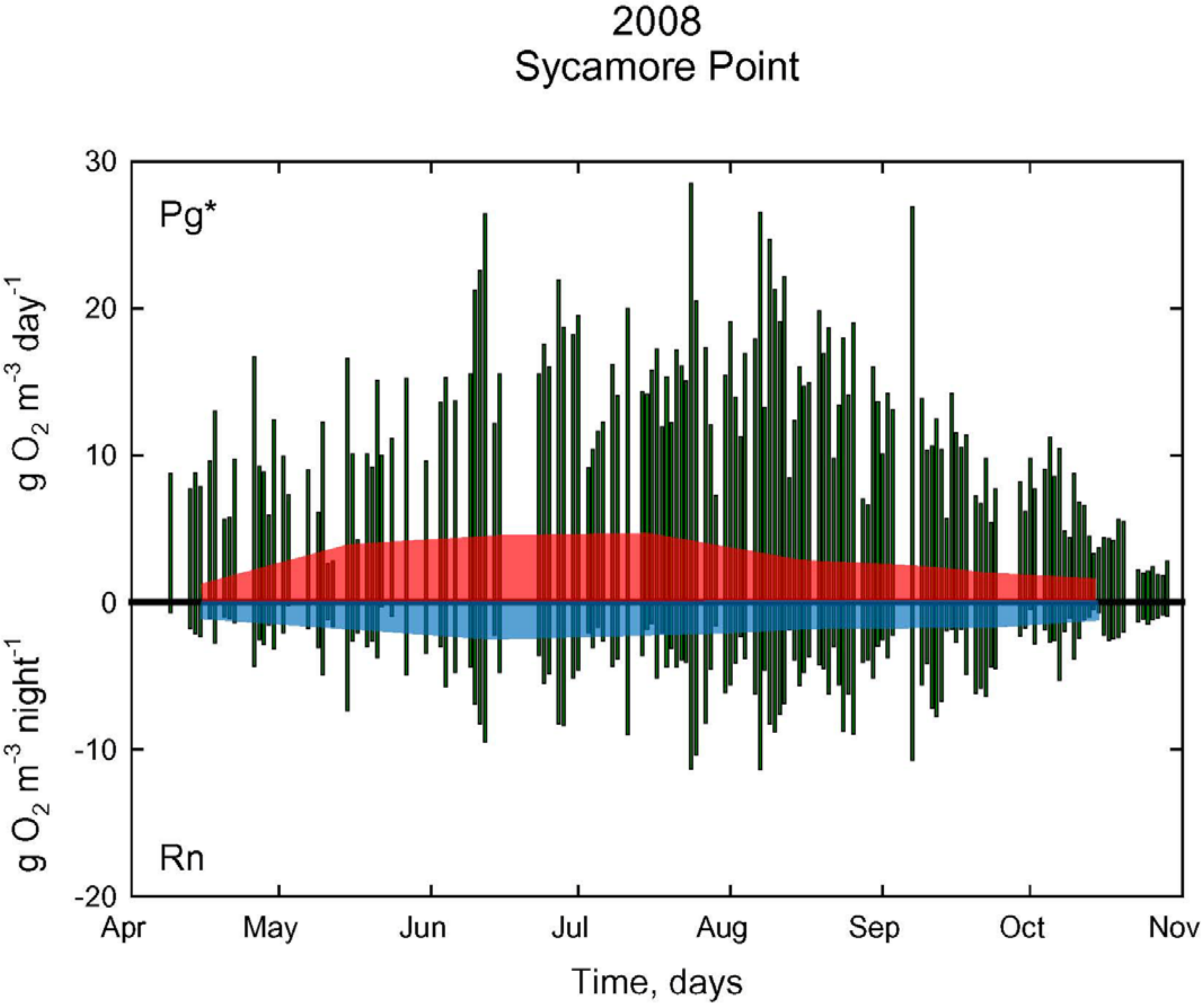


Figure 3-5. A bar graph showing daily rates of gross primary production based on ConMon data collected during 2008 at the Sycamore Point site in the upper Corsica River estuary. The red and blue shaded areas show average monthly rates of gross primary production (Pg*) and community respiration (Rn) measured at the Benedict Bridge site during 1964 and serve as an indicator of rates observed under more pristine conditions.

Finally, we have referred to lower nutrient loading rates at the Benedict site several times in this report. A good question is…"what were these lower loading rates?" The answer is, unfortunately, not simple. We have reconstructed the nutrient loading rate from 1960 through to the time when USGS began routine monitoring of the Patuxent at the gauge at Bowie, Maryland (1978) and have earlier reported the results of this analysis (Hagy et al. 1998). Loads were much lower at the fall line during the early 1960s than they were during the mid-1980s and later. However, the USGS gauge at Bowie monitors nutrient loads from about 30% of the basin. A larger portion of the basin is not gauged between Bowie and Benedict and thus there are no direct measurements of loads from about 40% of the basin upstream of Benedict. However, Boynton et al (2008) developed a nutrient budget for the Patuxent estuary and made some estimates of early and more recent loads to this section of the estuary. Using reconstructed fall line loads and land use modeling results they estimated that loads during the 1960s were about half of what they are now (~3000Kg N day^{-1} in 1960s versus about 6600 Kg N day^{-1} during early 2000s). It is useful to note that metabolic rates during 2003-2005 were about twice what they were during the early 1960s and, if there is proportionality between loads and metabolic rates, we should expect decreasing rates as TMDL mandated nutrient load reductions occur in the next several years.

3-4.2 Comparisons of Metabolism Along Estuarine Nutrient Enrichment Gradients

One of the things becoming clear, based on inspection of data from the many ConMon sites, is the substantial diversity in water quality conditions among sites. These range from severely enriched to relatively healthy and thus offer many alternative analyses. To examine community metabolism responses to proximity of nutrient sources we selected two general locations where ConMon data were available along a gradient of nutrient enrichment. The first of these areas was the Maryland Coastal Bays where three sites could be identified along an enrichment gradient and these, in order of enrichment, were Bishopville Prong, Turville Creek and Public Landing. The second region was the Patuxent River where ConMon sites were located in the tidal freshwater upper estuary at Jug Bay, Benedict at the upper end of the mesohaline estuary, Pin Oak in the middle of the mesohaline estuary and CBL at the mouth of the Patuxent estuary. While proximity to major nutrient sources was a critical factor in selecting sites we recognize that other important variables are also changing along these nutrient enrichment gradients (e.g., water clarity, water residence times, depth, nutrient concentrations and others) and also influencing metabolic rates.

Monthly average production and respiration rates (Pg* and Rn) are shown for Coastal Bay sites in Figure 3-6a-c. Rates at the most enriched site ranged from about 10 to 28 g O_2 m^{-3} day^{-1} and were often between 15 and 20 g O_2 m^{-3} day^{-1}. These are very large rates and are indicative of extreme nutrient enrichment. Rates of Pg* at Turville Creek were lower (but still substantial) ranging from 5 to 15 g O_2 m^{-3} day^{-1}. Rates of Rn at all sites exhibited similar patterns. Finally, at the least enriched site (Public Landing in the center of Chincoteague Bay) rates of Pg* ranged from about 2 to 8 g O_2 m^{-3} day^{-1} and were often less than 5 g O_2 m^{-3} day^{-1}. While rigorous nutrient loading rates are not known, Boynton et al (1996) made estimates based on land use yield coefficients and reported N loading rates of about 40, 16 and 3 g N m^{-2} year^{-1} at the three sites. Combining these loading rates with average values of Pg* in a simple regression analysis yields a highly significant result ($r^2 = 0.95$; $p > 0.05$; $Y_{Pg*} = 4.6 + 0.35X_{N\ load}$). The slope of the regression model indicates an increase of 0.35 g O_2 m^{-3} day^{-1} per unit of increase in N loading rate. While this is hardly a rigorous model (based on only 3 sets of observations) it is suggestive of a strong response of community production rates to nutrient loads and as such as important management implications. Finally, there have been some management actions taken in the Bishopville Prong drainage basin to reduce nutrient loading rates (C. Wazniak pers. comm.) and the magnitude of Pg* rates at this site seem to reflect those actions.

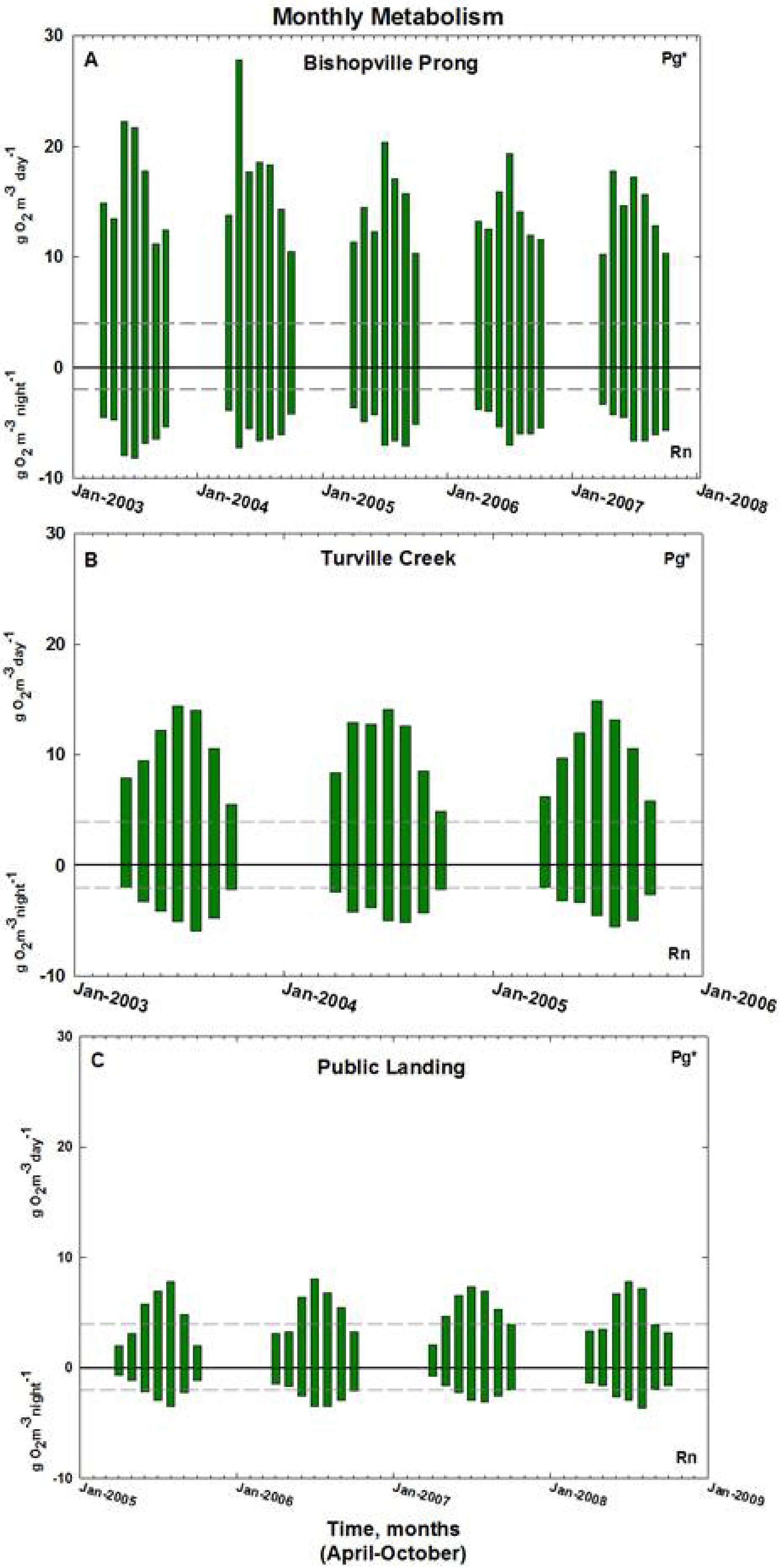


Figure 3-6. Bar graphs of mean monthly (April–October) rates of gross primary production (Pg*) and community respiration (Rn) along a eutrophication gradient in the Maryland Coastal Bays: (A) Bishopville Prong (most enriched); (B) Turville Creek (moderately enriched); (C) Public Landing (least enriched). The dashed horizontal lines represent mean rates of Pg* and Rn collected during 1964 at the Benedict Bridge site on the Patuxent River estuary and represent a more pristine condition.

Rates measured during 2003 and 2004 were higher than those measured during 2006 and 2007. It might well be worth maintaining this ConMon station as a sentinel site, especially if more could be discovered concerning nutrient load reductions in this basin. In any case, we see here clear responses of a very important ecosystem process (production of labile organic matter) to nutrient load conditions.

Monthly average production and respiration rates (Pg* and Rn) are shown for Patuxent River sites in Figure 3-7a-d. Rates ranged from 2-8, 3-14, 3-11, and 3-11 g O_2 m^{-3} day $^{-1}$ at Jug Bay, Benedict, Pin Oak and CBL, respectively. Summer averages were about 5, 9, 6, and 6 g O_2 m^{-3} day^{-1}, respectively. Relating these rates to nutrient loads is not such a simple matter and we have not yet determined a defensible approach for doing this in the Patuxent. In this case we do have excellent nutrient load estimates but it is not clear how to partition or assign these loads along the axis of the estuary. In any case, several useful features were still apparent. First, peak rates were observed at the Benedict site which is at the head of the mesohaline estuary. It is at this point along the estuarine axis that water clarity improves markedly. Routine water quality measurements also indicate that essential nutrient (N and P) concentrations remain high. So, it is not surprising that rates of Pg* reach a maximum at this point. Rates at Pin Oak and CBL remain substantial but less than those measured at Benedict. It is likely that nutrient limitation is regulating these rates. The CBL rates may also be influenced by N coming from the mainstem Chesapeake Bay during summer periods (Boynton et al. 2008) and explain why rates furthest from the main basin sources of N are comparable to the Pin Oak rates which are closer to the main riverine N sources. Finally, rates of both Pg* and Rn at the Jug Bay site were lower during 2003 than in other years. It is likely this is related to river flow and resultant effects on water residence times. River flow during 2003 was unusually high during most of the year, especially during the summer periods…this was the summer when the rain simply did not let up. As a result of high river flow (and shorter water residence times) phytoplankton communities had little chance to become established in the upper estuary and hence rates of Pg* were lower than normal. This “wash-out” effect has been shown previously for this portion of the river (Boynton et al. 2008). However, at the Benedict and Pin Oak sites, where river flow has much less effect on water residence times (but still has a large effect on nutrient loading rates), we see that rates of Pg* were enhanced during 2003. These observations, all based on ConMon data sets, suggest that key rates influencing water quality (Pg* and Rn) are quite responsive to nutrient loading rates. We can confidently assume that when these loads are reduced, as specified by the TMDL, there will also be reductions in key community production and respiration rates. It is also useful to note that loading rates alone are generally not sufficient to predict production rates, as we have shown in this Patuxent River example.

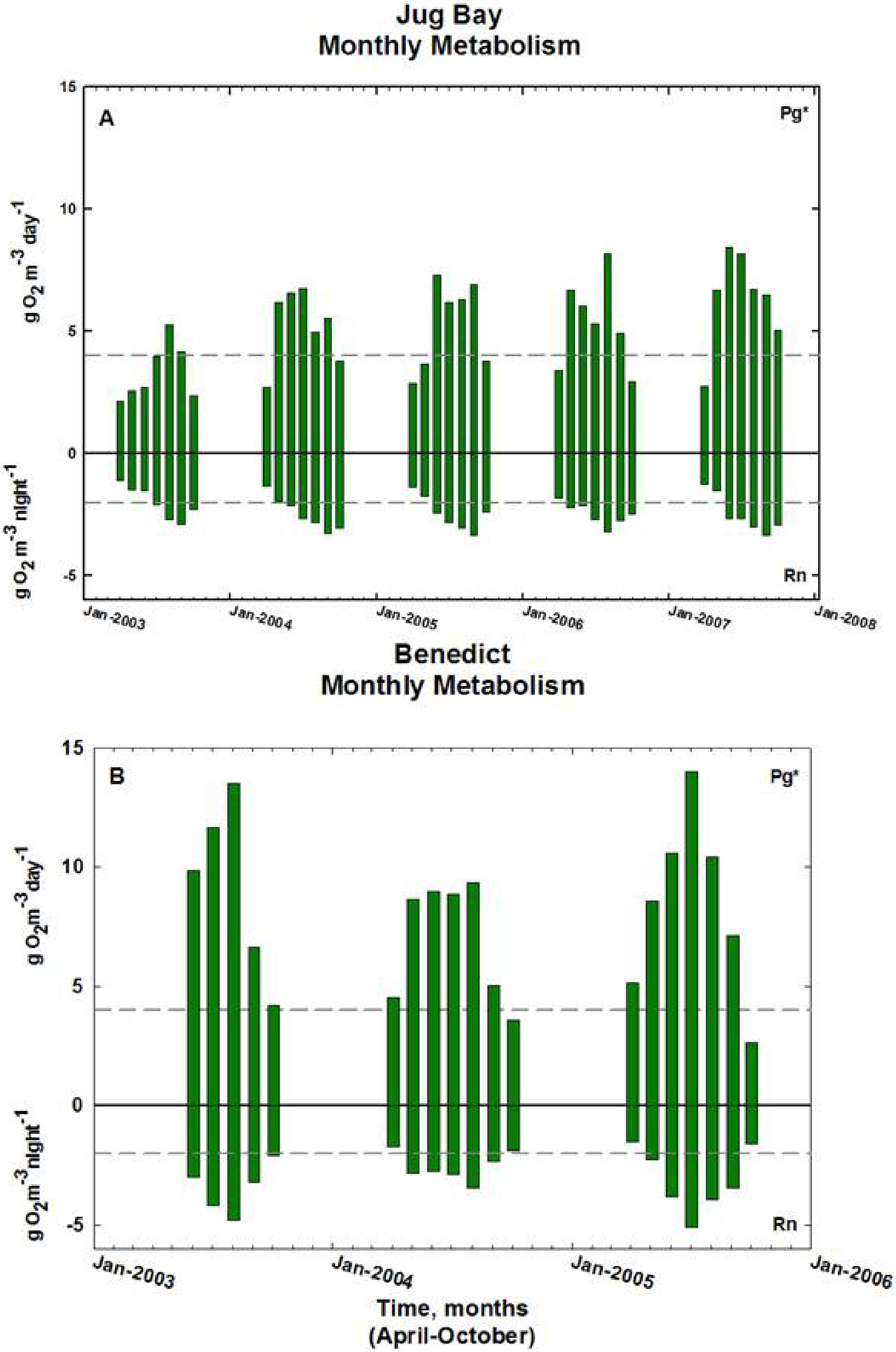
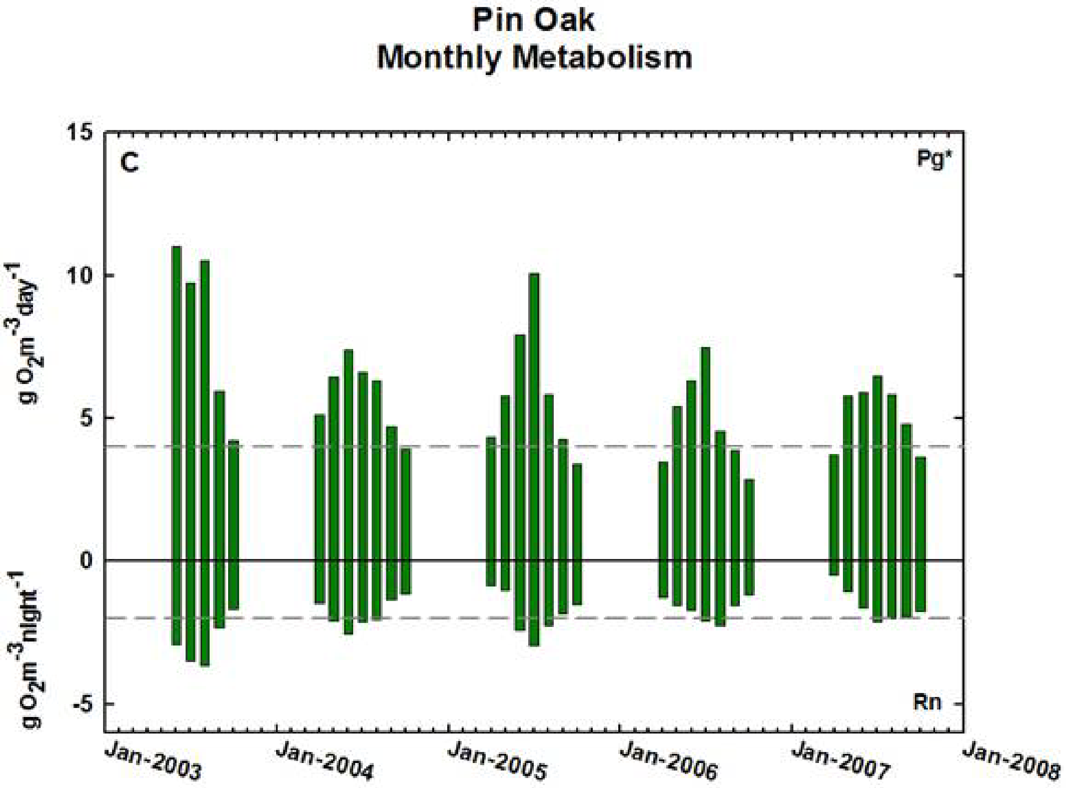


Figure 3-7. Bar graphs of mean monthly (April–October) rates of gross primary production (Pg*) and community respiration (Rn) along a eutrophication gradient in the Patuxent River estuary: (A) Jug Bay (tidal freshwater site); (B) Benedict Bridge (moderately enriched site);


Pin Oak
Monthly Metabolism

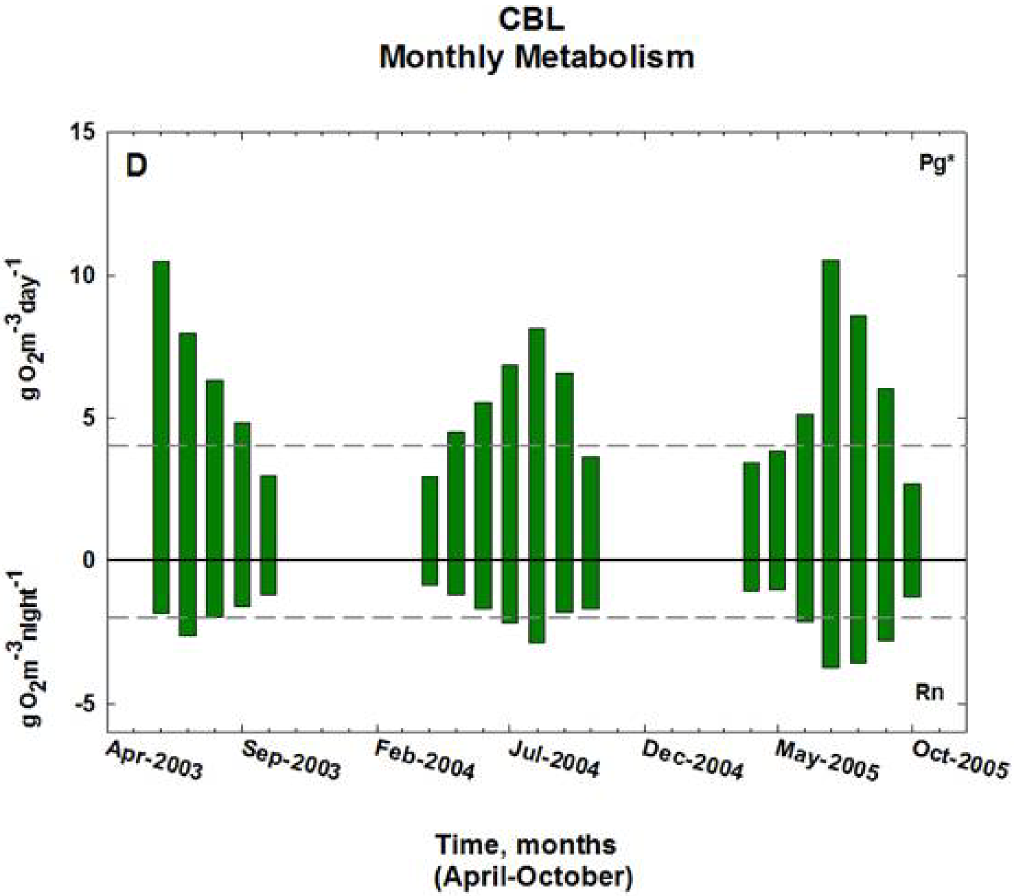

CBL
Monthly Metabolism

Figure 3-7 (continued): (C) Pin Oak (mesohaline site); (D) CBL (site at mouth of Patuxent River estuary). The dashed horizontal lines represent mean rates of Pg* and Rn collected during 1964 at the Benedict Bridge site on the Patuxent River estuary and represent a more pristine condition.

3-4.3 Production and Respiration at Exposed Shoreline Sites

Many of the ConMon sites we have used in the above analyses were located in tributary rivers (e.g., upper Patuxent) or even in small tributaries of tributary rivers (e.g., Corsica River). In a sense, many of these data come from sites that are not a direct part of a big system (e.g., mainstem Chesapeake Bay or Potomac River) and generally do not have exposure to wind, current or wave conditions typical of shoreline sites in these bigger systems.

We chose to examine, as a first step, two shoreline ConMon sites having exposure to “big system” conditions. These sites were Betterton Beach in the upper Bay and Fenwick in the upper Potomac. Rates of Pg* and Rn are shown in Figure 3-8a and 3-8b. Both sites are reasonably close to major nutrient sources (Fenwick is near Washington, DC and betterton is near the mouth of the Susquehanna River). However, rates of Pg* and Rn are radically different at these sites. The rates at Fenwick are very large, indicative of high nutrient loading rates while values at Betterton are quite modest. Clearly, more work needs to be done to understand such strong differences. One aspect of data from these sites, and indeed all the ConMon sites we have examined, is that rates of Pg* and Rn strongly tend to be low during April and again in October. Across sites with very different water quality conditions low rates prevail during these periods of time. We suspect that low temperature and limited sunlight play a strong role in maintaining rates at low values. From a monitoring perspective, it appears we learn little by having sites active in either April or October…the action, in terms of metabolism and in terms of DO criteria attainment (or non-attainment), appears to be between May and September.

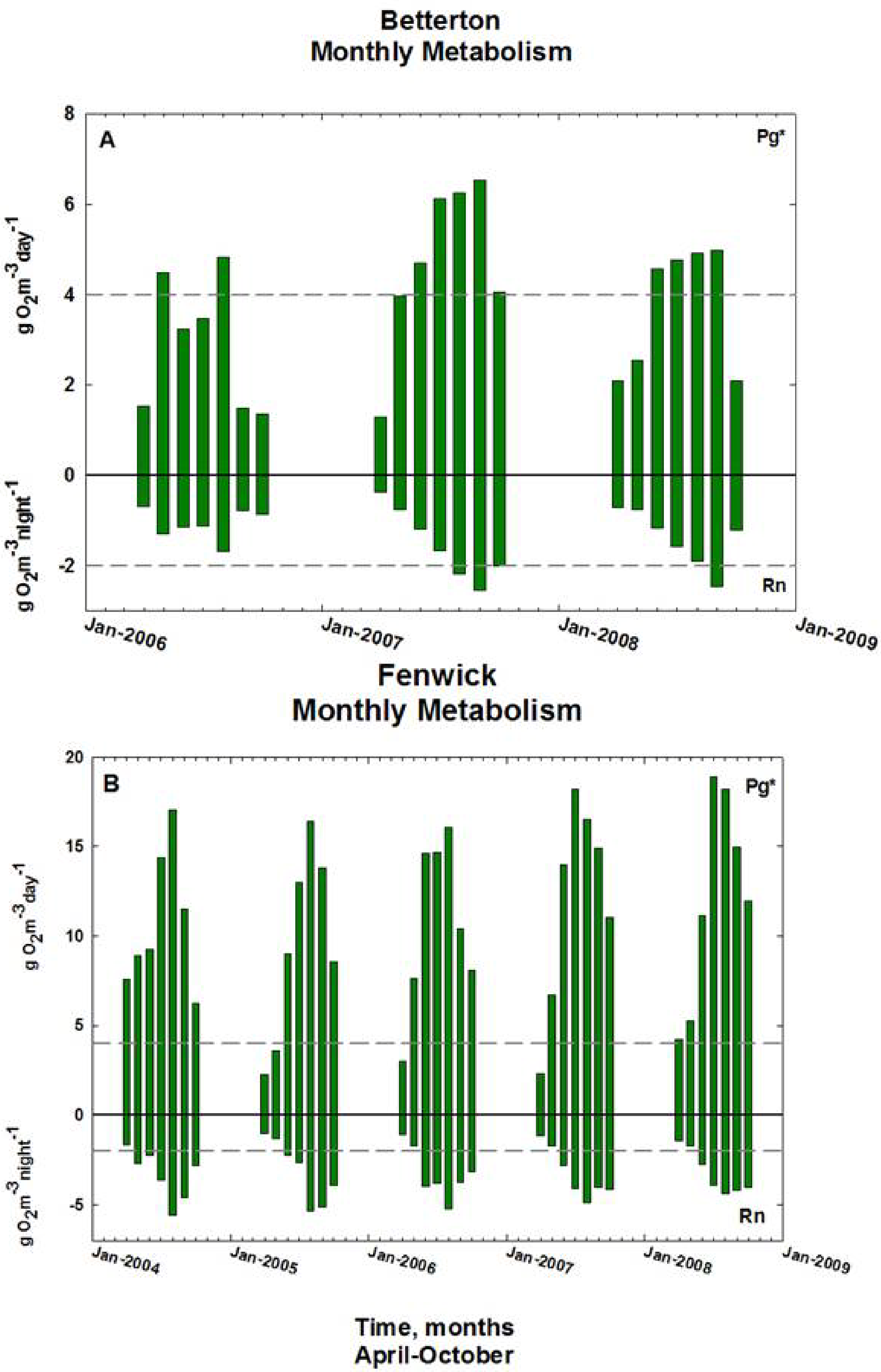


Figure 3-8. . Bar graphs of mean monthly (April – October) rates of gross primary production (Pg*) and community respiration (Rn) at ConMon sites located along exposed shoreline of large estuarine systems (a) Betterton in the upper Chesapeake Bay eastern shore; (b) Fenwick on the eastern shore of the upper Potomac River estuary. The dashed horizontal lines represent mean rates of Pg* and Rn collected during 1964 at the Benedict Bridge site on the Patuxent River estuary and represent a more pristine condition.

3-5 Summary of Summer Rates of Pg* and Rn

We have summarized metabolism data from all sites where these computations were completed in Table 3-2. We chose to focus on summer rates (June – August) because it is now clear that these rates are at annual 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 categories and these include the following: a) enriched sites; b) less enriched sites; c) coastal bays nutrient enrichment gradient; d) Patuxent River enrichment gradient; and e) historical time-series at Benedict, Maryland.

At the enriched sites (those located near large point and diffuse nutrient sources) rates of Pg* are exceptionally large. Average summer rates ranged, across all sites, from about 11 to 16 g O_2 m^{-3} day^{-1}. If these rates were converted to carbon equivalents (using a photosynthetic quotient of 1.25) rates would be between 3.5 and 5.0 g C m^{-3} day^{-1}. These rates constitute a very large source of labile organic matter which, when decomposing, exert a large oxygen demand. In addition, the P:R ratio at these enriched sites averaged about 3.0 for summer periods. This suggests these shallow water systems are very autotrophic (i.e., more organic matter is produced than consumed). This, in turn, indicates that these systems serve as an organic matter source to adjacent systems. It may be that the very high production of these shallow waters provides organic matter which fuels oxygen depletion in adjacent, deeper waters.

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 into categories and this is indicated at the top of each table.

Summer Metabolism (June-August)

Enriched Sites

Corsica River

Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Sycamore Point	2005	-4.3	2.7	14.4	6.2	3.3
Sycamore Point	2006	-3.7	1.7	10.6	4.3	2.9
Sycamore Point	2007	-4.7	2.0	13.7	5.7	2.9
Sycamore Point	2008	-5.2	2.4	16.0	4.5	3.1

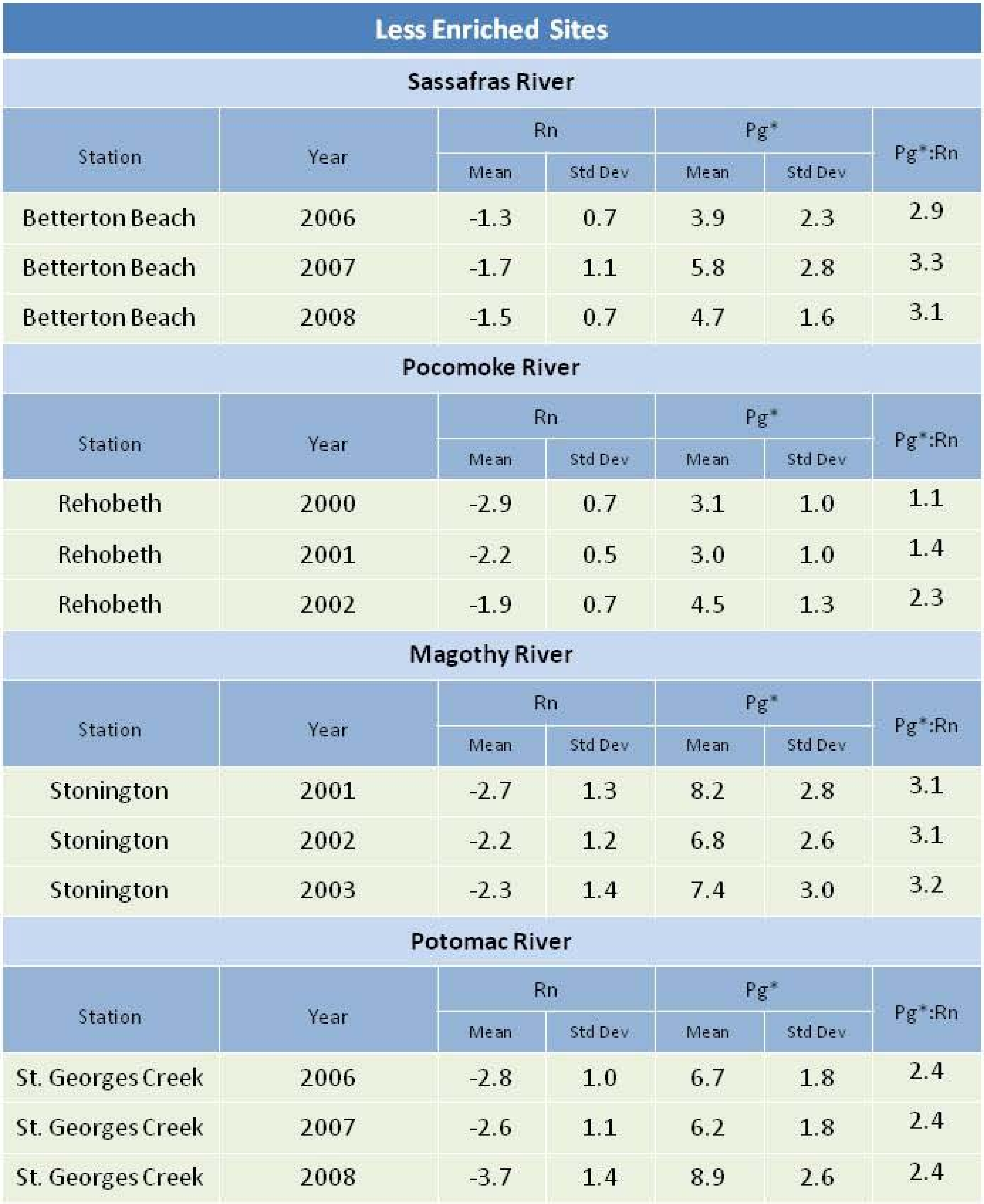
Patapsco River

Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Fort McHenry	2004	-3.8	2.0	10.1	4.4	2.7
Fort McHenry	2005	-3.6	1.7	11.7	4.3	3.3
Fort McHenry	2006	-3.5	1.9	10.3	4.6	2.9
Fort McHenry	2007	-4.7	1.9	13.0	5.7	2.8
Fort McHenry	2008	-5.0	2.3	13.0	5.9	2.6

Potomac River

Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Fenwick	2004	-3.9	2.0	13.7	5.1	3.5
Fenwick	2005	-3.6	2.0	13.2	4.6	3.7
Fenwick	2006	-4.3	1.6	15.2	3.3	3.5
Fenwick	2007	-4.2	1.6	16.3	4.9	3.9
Fenwick	2008	-3.6	1.5	15.9	5.0	4.4

Summer Metabolism (June-August)



Less Enriched Sites

Sassafras River

Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Betterton Beach	2006	-1.3	0.7	3.9	2.3	2.9
Betterton Beach	2007	-1.7	1.1	5.8	2.8	3.3
Betterton Beach	2008	-1.5	0.7	4.7	1.6	3.1

Pocomoke River

Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Rehobeth	2000	-2.9	0.7	3.1	1.0	1.1
Rehobeth	2001	-2.2	0.5	3.0	1.0	1.4
Rehobeth	2002	-1.9	0.7	4.5	1.3	2.3

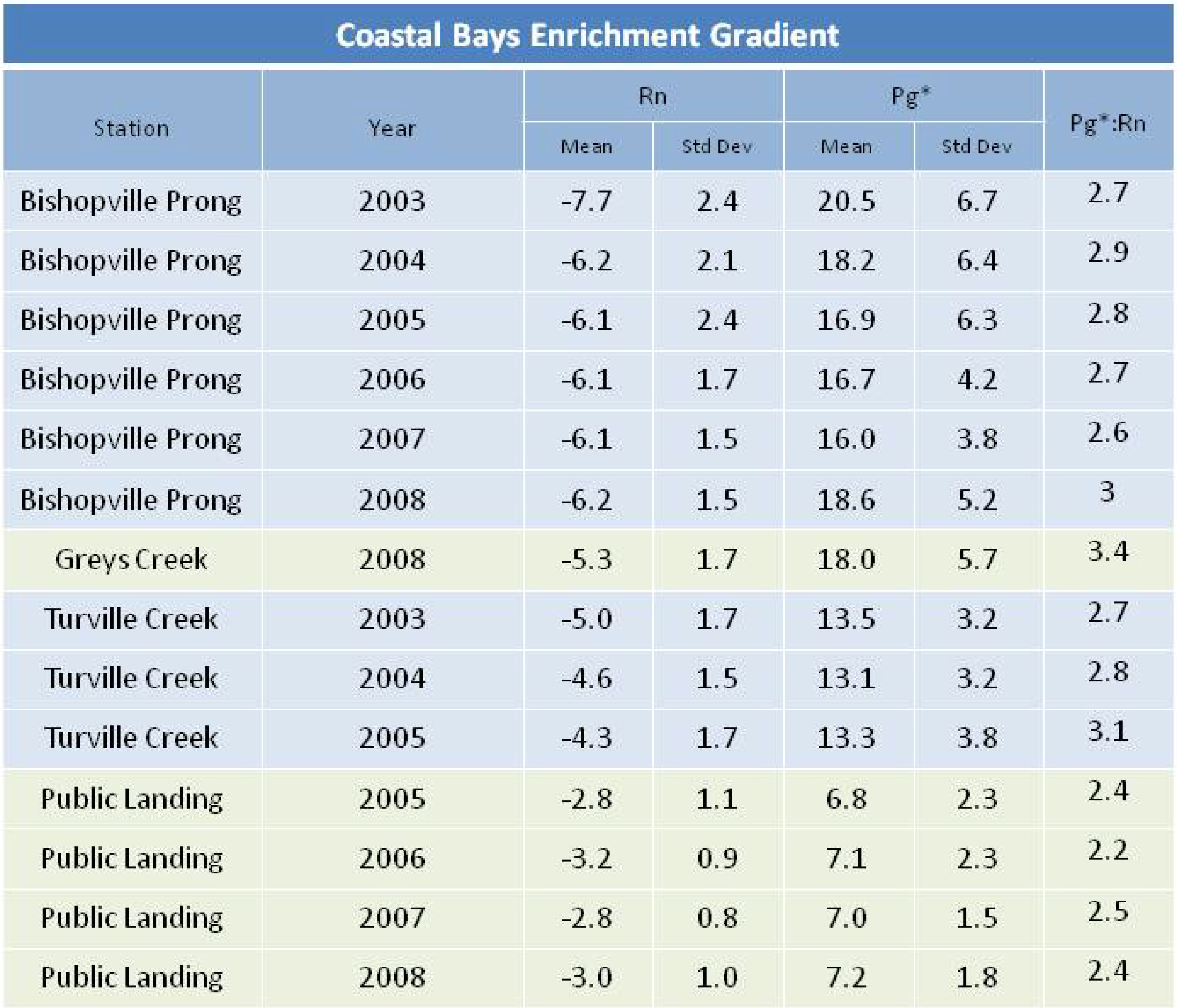
Magothy River

Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Stonington	2001	-2.7	1.3	8.2	2.8	3.1
Stonington	2002	-2.2	1.2	6.8	2.6	3.1
Stonington	2003	-2.3	1.4	7.4	3.0	3.2

Potomac River

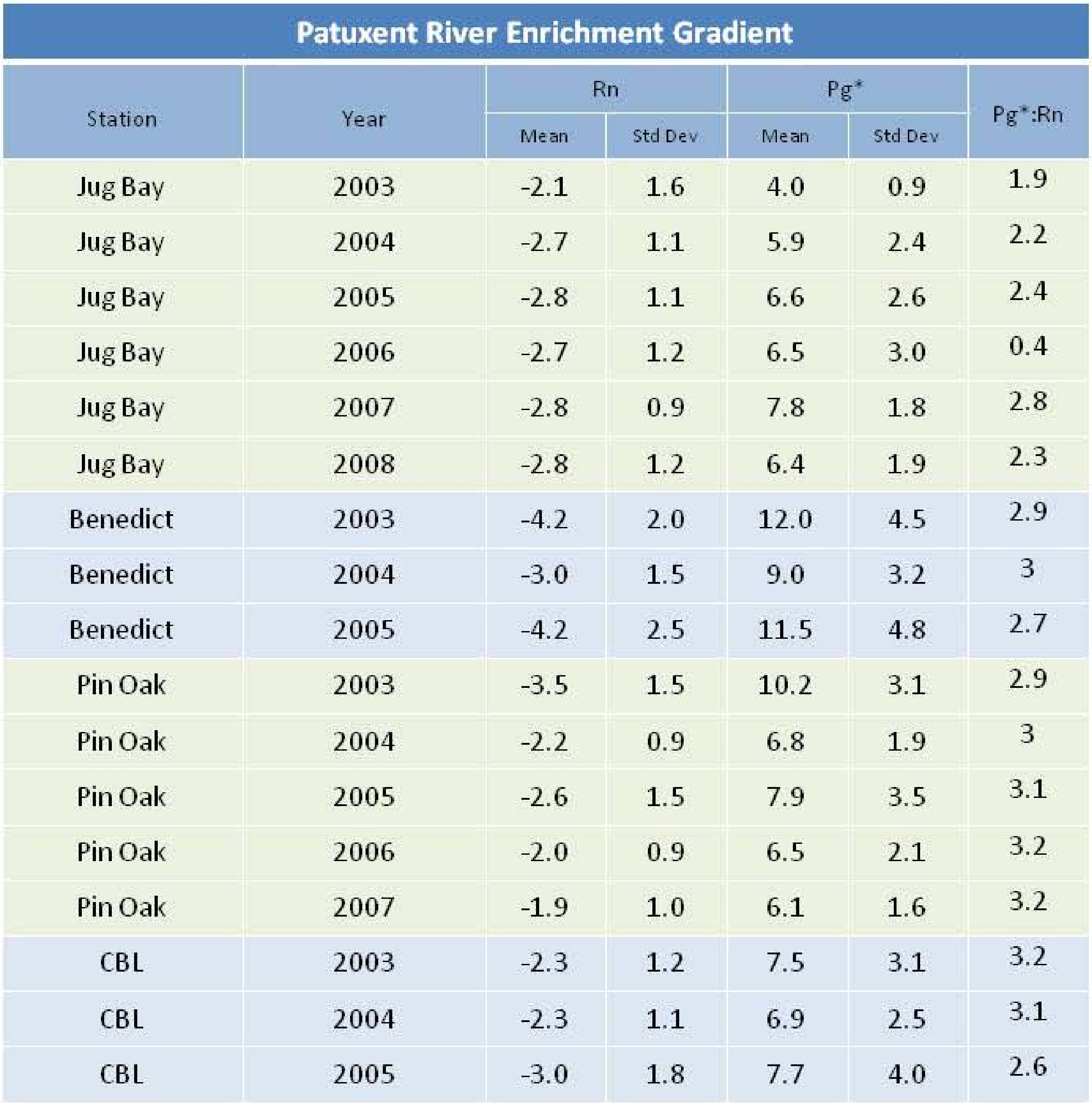
Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
St. Georges Creek	2006	-2.8	1.0	6.7	1.8	2.4
St. Georges Creek	2007	-2.6	1.1	6.2	1.8	2.4
St. Georges Creek	2008	-3.7	1.4	8.9	2.6	2.4

Summer Metabolism (June-August)



Coastal Bays Enrichment Gradient						
Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Bishopville Prong	2003	-7.7	2.4	20.5	6.7	2.7
Bishopville Prong	2004	-6.2	2.1	18.2	6.4	2.9
Bishopville Prong	2005	-6.1	2.4	16.9	6.3	2.8
Bishopville Prong	2006	-6.1	1.7	16.7	4.2	2.7
Bishopville Prong	2007	-6.1	1.5	16.0	3.8	2.6
Bishopville Prong	2008	-6.2	1.5	18.6	5.2	3
Greys Creek	2008	-5.3	1.7	18.0	5.7	3.4
Turville Creek	2003	-5.0	1.7	13.5	3.2	2.7
Turville Creek	2004	-4.6	1.5	13.1	3.2	2.8
Turville Creek	2005	-4.3	1.7	13.3	3.8	3.1
Public Landing	2005	-2.8	1.1	6.8	2.3	2.4
Public Landing	2006	-3.2	0.9	7.1	2.3	2.2
Public Landing	2007	-2.8	0.8	7.0	1.5	2.5
Public Landing	2008	-3.0	1.0	7.2	1.8	2.4

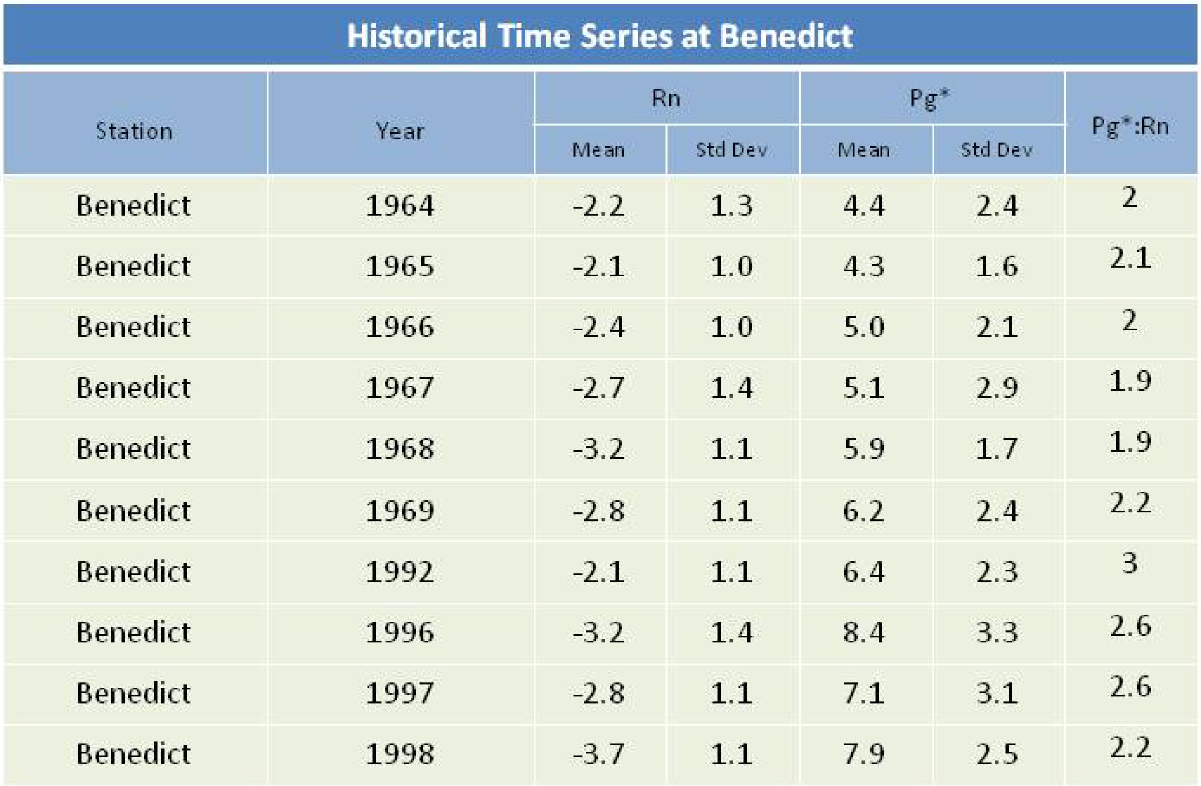
Summer Metabolism (June-August)



Patuxent River Enrichment Gradient

Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Jug Bay	2003	-2.1	1.6	4.0	0.9	1.9
Jug Bay	2004	-2.7	1.1	5.9	2.4	2.2
Jug Bay	2005	-2.8	1.1	6.6	2.6	2.4
Jug Bay	2006	-2.7	1.2	6.5	3.0	0.4
Jug Bay	2007	-2.8	0.9	7.8	1.8	2.8
Jug Bay	2008	-2.8	1.2	6.4	1.9	2.3
Benedict	2003	-4.2	2.0	12.0	4.5	2.9
Benedict	2004	-3.0	1.5	9.0	3.2	3
Benedict	2005	-4.2	2.5	11.5	4.8	2.7
Pin Oak	2003	-3.5	1.5	10.2	3.1	2.9
Pin Oak	2004	-2.2	0.9	6.8	1.9	3
Pin Oak	2005	-2.6	1.5	7.9	3.5	3.1
Pin Oak	2006	-2.0	0.9	6.5	2.1	3.2
Pin Oak	2007	-1.9	1.0	6.1	1.6	3.2
CBL	2003	-2.3	1.2	7.5	3.1	3.2
CBL	2004	-2.3	1.1	6.9	2.5	3.1
CBL	2005	-3.0	1.8	7.7	4.0	2.6

Summer Metabolism (June-August)



Historical Time Series at Benedict

Station	Year	Rn		Pg*		Pg*:Rn
		Mean	Std Dev	Mean	Std Dev	
Benedict	1964	-2.2	1.3	4.4	2.4	2
Benedict	1965	-2.1	1.0	4.3	1.6	2.1
Benedict	1966	-2.4	1.0	5.0	2.1	2
Benedict	1967	-2.7	1.4	5.1	2.9	1.9
Benedict	1968	-3.2	1.1	5.9	1.7	1.9
Benedict	1969	-2.8	1.1	6.2	2.4	2.2
Benedict	1992	-2.1	1.1	6.4	2.3	3
Benedict	1996	-3.2	1.4	8.4	3.3	2.6
Benedict	1997	-2.8	1.1	7.1	3.1	2.6
Benedict	1998	-3.7	1.1	7.9	2.5	2.2

Rates of Pg* and Rn were considerably lower at the less enriched sites. At present we are using the terms enriched and less enriched in a qualitative fashion. For most of these systems we have not been able to determine a way to quantitatively estimate nutrient loading rates at the size—scale of a ConMon site. We have made numerous estimates of nutrient loading at the scale of whole estuaries and we have used these estimates as a guide in the classification used here (Boynton et al 2008). At the less enriched sites rates of Pg* ranged from about 3 to 9 g O_2 m^{-3} day^{-1} and most values were between 4 and 7 g O_2 m^{-3} day^{-1}. It is useful to note that P:R ratios at these sites were also much lower than at the enriched sites ranging here from about 1 to 3. This suggests less export of labile organic matter to adjacent systems.

Two systems were examined for metabolic responses along nutrient enrichment gradients (Maryland Coastal Bays and the Patuxent River). At Coastal Bay sites rates of Pg* ranged from about 20 g O_2 m^{-3} day^{-1} at the most enriched site to about 7 g O_2 m^{-3} day^{-1} at the least enriched site. This is one of the few locations where sub-estuary nutrient loading rates were available and as we reported earlier in this Chapter there is a significant correlation between loads and metabolic rates. This indicates that these rates are responsive to nutrient loads and, importantly, to management-induced nutrient load reductions. In the simple statistical model reported earlier there was no indication of temporal lags (i.e., loads from a previous year influencing rates in the current year) between loads and metabolic rates. This suggests that load reductions, at least in this system, should be rapidly followed by expected system responses. The metabolic pattern observed in the Patuxent was more complex and, perhaps, more interesting. In this case rates were relatively low in the upper estuary, peaked in the upper mesohaline region and were lower in the lower estuary. Our interpretation of this pattern is that rates were light and residence time limited in the upper estuary, responsive to large nutrient supplies in the upper mesohaline region and nutrient limited in the lower estuary. It is also useful to note that rates of Pg* were lowest in the upper estuary (Jug Bay) during 2003 (a year of sustained high river flow) suggesting that short water residence times (and possibly enhanced turbidity) were suppressing production. During this same year rates of Pg* were highest at Benedict and Pin Oak, suggesting a simulating effect of large nutrient loads associated with a wet year. This again indicates that these rates are responsive, on an annual basis, to changes in nutrient loading rates.

Finally, Pg* and Rn data from the historical site at Benedict (upper mesohaline region of the Patuxent River) were summarized for summer periods for the pre-ConMon period (1964-1998). Metabolic rates during this time interval doubled and this is correlated with a large increase in nutrient loading rates to this system between the mid-1960s and the late 1990s. Consistent with this degree of increase in Pg* rates, Boynton et al (2009) estimated that nutrient loads in this system would need to be reduced by a factor of 2 to achieve water quality and habitat conditions characterized by healthy SAV communities and very limited hypoxia during summer.

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Chapter 4

Spatial Analysis of Water Quality Conditions

L.A. Wainger and A.R. Bayard

4-1 Goals of This Analysis

This analysis explored seasonal and annual spatial variability of water quality conditions in four shallow sub-estuaries of the Chesapeake Bay to promote two primary goals. The first set of goals was to characterize the extent and distribution of adverse water quality conditions (elevated chlorophyll-*a*) including evaluating the persistence of such conditions through time and developing hypotheses regarding the drivers of localized differences in water quality. The last goal was served by conducting an Integrated Ecosystem Assessment of the estuaries that explored a wide range of stressors effects on selected estuarine responses. A second and related goal was to evaluate the potential for DATAFLOW$^{©}$ data to be used to identify heterogeneity in SAV habitat quality and restoration potential. For this analysis, we tested whether water quality conditions in areas where SAV grew were significantly different from areas that had historically supported SAV but which did not support SAV in a given year.

4-2 Introduction

Spatial patterns of water quality have the potential to reveal drivers of water quality conditions and their relevance to aquatic organisms. For example, if water quality degradation is consistently localized near sources of concentrated nutrient and sediment runoff, the pattern suggests the need to investigate whether those sources are a primary cause of the local impairment. On the other hand, if poor water quality is uniformly or randomly distributed, such patterns suggest that causes may be diffuse or distant. Further, we understand that uniformly poor water quality conditions (such as low oxygen or high suspended solids) present a different level of stress to living resources than patchy conditions, because organisms will have limited refuge available to them. Therefore, understanding the spatial distribution of water quality conditions in the Bay provides an opportunity to develop hypotheses regarding drivers of change and their impacts on living resources.

Because the methods for analyzing spatially and temporally-detailed dataset have not been well-developed, we evaluated several alternatives for characterizing patterns of water quality in ways that are meaningful for understanding effects on living resources. A companion study that we are conducting is using the spatial data and pattern analysis results to statistically test alternative drivers of water quality condition.

As part of the spatial analysis, we consider the persistence and pattern of conditions through time and the aggregate responses of chl-*a* (chlorophyll-*a*) and SAV responses to a variety of water quality conditions, watershed stressors and physical relationships that may structure estuarine response to stressors. These aggregate response indicators and potential stressors are then used in an integrated assessment of key system relationships. The DATAFLOW$^{©}$ datasets are able to reveal fine to coarse scale variability or patchiness in water quality conditions (temperature, turbidity, salinity and chl-*a*) and, since data are sampled semi-monthly to monthly, they also provide some information about the persistence of characteristics through time. However, a major challenge to interpreting the data is the highly variable nature of the estuaries we are examining. The more variable a system is, the more data are needed to be able to statistically detect patterns (i.e., see the signal amidst the noise). DATAFLOW$^{©}$ datasets typically cover 3 years, which provides enough data to begin to hypothesize drivers of patterns, but, because of high variability in these estuaries, data analysis can only provide preliminary support for such hypotheses.

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 year and used the repeated sampling to evaluate the frequency with which the elevated chl-*a* occurred. We chose a threshold of 15 $\mu g\ L^{-1}$ for this exercise to define “elevated” because this has been suggested to be relevant to SAV habitat quality (Batiuk *et al.* 2000). Also, we evaluated whether the chl-*a* was elevated at least 20% of the time. Other management-relevant thresholds might also be used. For example, in implementing the Neuse Estuary TMDL, the regulations state that no more than 10% of samples can exceed 40 µg L^{-1}. The analysis of the DATAFLOW© data is easily tailored to fit any particular threshold.

We further considered the utility of DATAFLOW© for identifying SAV restoration areas. SAV restoration efforts in the Chesapeake Bay have historically included the direct seeding of new beds and areas to be seeded are selected, in part, based on analysis of available water quality data (L. Karrh, pers comm.). However, SAV planting has had mixed success in generating persistent beds and inducing new bed development (Orth *et al.* 2010), suggesting the need for a more thorough look at the causes of success and failure in particular locations. Water quality is hypothesized to be an important driver of SAV bed establishment and persistence, in conjunction with other hydrodynamic and sediment conditions (Batiuk *et al.* 2000, Koch *et al.* 2010).

To evaluate the usefulness of DATAFLOW© data for differentiating areas that were more or less likely to support SAV, we tested whether areas that supported SAV had significantly different levels of chl-*a* than areas that did not support SAV, even though they had supported beds in the past. Chl-*a* values were used as an integrative proxy for habitat-relevant water quality conditions, in part, because direct measures of nutrients are not available in the DATAFLOW© data. This analysis is just one approach to help inform which areas might be appropriate for targeting for direct SAV planting.

4-2.1 Study Area

Four case study subestuaries were chosen for analysis because their relatively small size and shallowness was expected to make them more responsive to watershed inputs than the larger estuaries that we previously studied (Patuxent and Potomac, EPC Report #27 (Boynton *et al.* 2010). The estuaries fall within the upper half of the Bay and include the Corsica, Severn and Magothy rivers in the mesohaline zone (5-18 ppt) and the Bush river in the oligohaline zone (0.5-5 ppt) of the Chesapeake Bay (Figure 4-1). The watersheds have variable land cover, with the Corsica representing the most heavily agricultural watershed (67%), whereas the other watersheds have a mix of suburban, agricultural and natural land uses/covers (based on data from USGS 2006, Table 4-2). The Bush and the Severn both have roughly equal proportions of forest and residential (32-39%). However, the Bush also has over 20% agricultural compared to 6% in the Severn (Table 4-1). The Magothy, like the Severn has little agricultural cover and developed land is the dominant land use at just under 45% of the watershed; forest is just over 30%. 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.

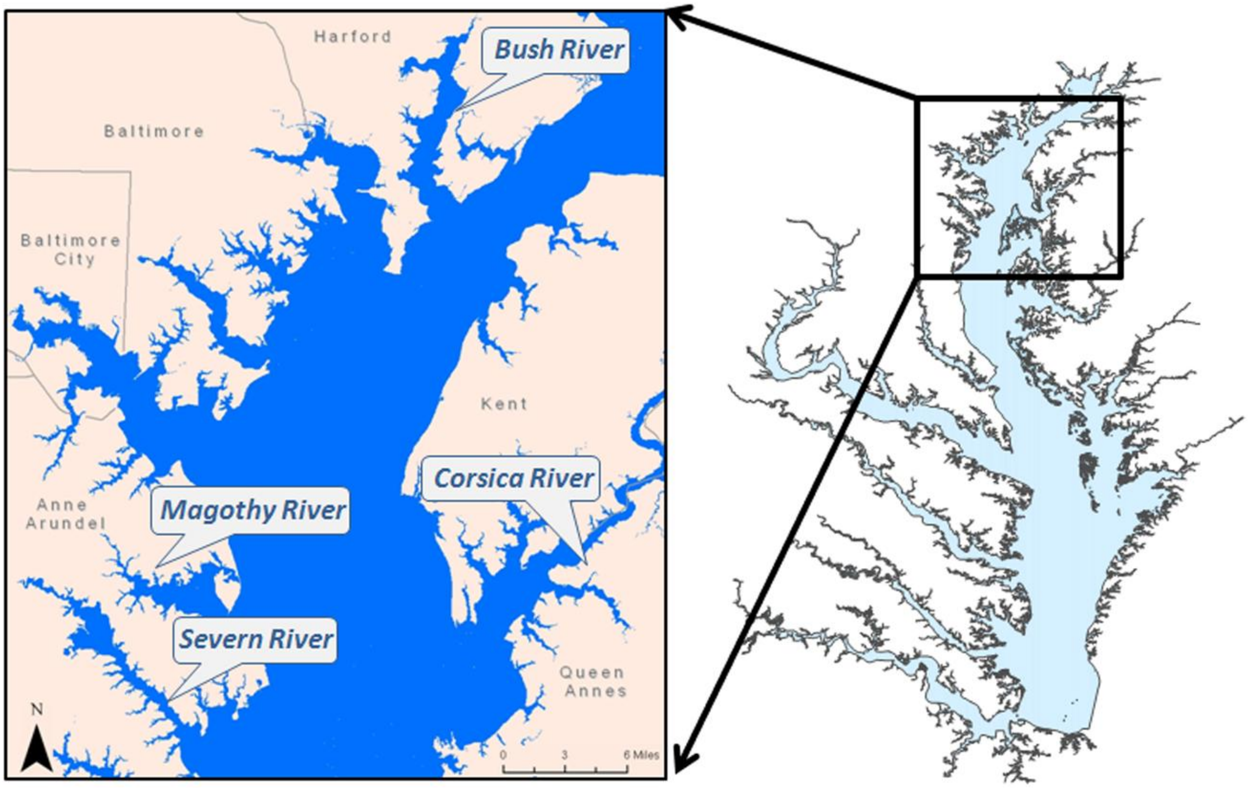


Figure 4-1. Location of four case study subestuaries.

Table 4-1. Land use distributions for case study watersheds.

AGGREGATED CLASSES	DETAILED CLASSES	Bush		Corsica		Magothy		Severn	
		Acres	%	Acres	%	Acres	%	Acres	%
Agriculture	Cultivated crop	11,480	11	12,803	53	416	2	2,268	5
	Pasture/ Hay	11,952	11	3,486	14	45	0	360	1
	Total Agriculture	**23,432**	**22**	**16,289**	**67**	**461**	**2**	**2,628**	**6**
Developed	Low Intensity	12,647	12	719	3	5,824	25	8,569	19
	Medium Intensity	5,327	5	182	1	1,226	5	2,987	7
	High Intensity	2,286	2	75	0	219	1	997	2
	Developed Open Space	13,446	12	226	1	2,815	12	4,266	9
	Barren	556	1	68	0	9	0	133	0
	Unconsolidated shoreline	25	0	4	0	18	0	92	0
	Grassland Herbaceous	864	1	32	0	35	0	180	0
	Total Development	**35,153**	**32**	**1,307**	**5**	**10,145**	**44**	**17,224**	**38**
Forest	Deciduous	32,114	30	3,580	15	4,942	21	12,174	27
	Evergreen	511	0	20	0	335	1	1,143	3
	Mixed	3,416	3	159	1	1,627	7	3,271	7
	Shrub Scrub	4,380	4	359	1	351	2	966	2
	Total Forest	**40,422**	**37**	**4,117**	**17**	**7,255**	**31**	**17,554**	**39**
Wetlands	Woody	7,907	7	2,390	10	5,090	22	7,506	17
	Emergent	1,269	1	153	1	99	0	195	0
	Total Wetlands	**9,176**	**8**	**2,543**	**10**	**5,188**	**23**	**7,701**	**17**
	TOTALS	**108,182**	**100**	**24,256**	**100**	**23,049**	**100**	**45,107**	**100**

4-3 Methods

4-3.1 Integrated Spatial Assessment

A variety of techniques have been applied to evaluate water quality drivers using the DATAFLOW© data using both the raw data points and the output of kriging. We use the data in both forms to take advantage of their strengths. Direct analysis of the raw data points prevents interpolation techniques from introducing error or bias into the results, but the kriging allows for a more even and comprehensive analysis of the estuary because it estimates data in unsampled areas and reduces the bias due to the sampling pattern for some calculations.

4-3.2 Data Sources

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). Other data used and their sources are summarized in Table 4-2.

4-3.2.1 Kriging Techniques

To inform hypotheses regarding drivers of water quality, 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 Bush 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

Table 4-2. Sources of original and derived datasets.

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. ; website: http://viewer.nationalmap.gov/viewer/ [Last Accessed: 06/20/11]
Impervious surface	MRLC NLCD Percent Developed Imperviousness, 2006, http://seamless.usgs.gov/website/seamless/viewer.htm
Landcover	USGS Chesapeake Bay Land Cover Data (CBLCD) Series, 2006, http://www.chesapeakebay.net/dataandtools.aspx?menuitem=14872#databases
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/finddata.html
Population density (Calculated using US Census SF1 2000 file: Block Groups)	U.S. Census Bureau, 2000, data product SF1, http://factfinder.census.gov/servlet/DownloadDatasetServlet?_lang=en
Precipitation	NOAA Annual Climatological Summary, http://www.ncdc.noaa.gov/oa/climate/climatedata.html
Septic system density	R. Pellicano, 2010 (MD Dept.Environment) 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/sav09 [last accessed: 06/13/2011].
Susquehanna flow	USGS National Water Information System, http://waterdata.usgs.gov/nwis
Total Nitrogen delivered from watershed	US EPA Chesapeake Bay Program; Gary Shenk and Jing Wu Pers. Comm.
Watershed Surface Area: Estuary surface Area ratio (Calculated in GIS using files: P5_LSEGS_Nov18_09.shp and NHD Area)	Chesapeake Bay Program, Land Segments (2009) and USGS National Hydrography Dataset (2009), http://www.chesapeakebay.net/dataandtools.aspx?menuitem=14872#databases , http://nhd.usgs.gov/

4-3.2.2 Data Summary Techniques

A variety of GIS techniques available within ArcMap 9.3.1 were used to evaluate the spatial data and 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. Summaries of whole-estuary conditions for elevated chl-*a* were calculated using the Spatial Analyst tools, including reclass as well as arithmetic and conditional statements. Area of SAV was calculated by summarizing polygon areas from the shapefiles of mapped SAV (Orth, *et al.* 2010). Cross-sections of kriged 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, animations of data were developed using freeware software (Picasa 2011) to examine changes through time (see Appendix 4-1 for viewing instructions).

4-3.2.2.1 Spatio-Temporal Summaries

Spatio-temporal summaries were used to characterize the percentage of each estuarine area that contained elevated chl-*a* over a given time period (annual or multi-year). To create these analyses, we defined thresholds to apply to spatial and temporal data and conditional statements to summarize conditions within a set of kriged output data that exceeded the defined thresholds. We chose a threshold of 15 µg L^{-1} in order to define "elevated" conditions in the spatial data because this has been suggested to be relevant to SAV habitat quality. And we evaluated whether the chl-*a* was elevated at least 20% of the time. Note that the actual percent of time measured by the datasets will vary by the number of samples available in a given year. The temporal detail that we were able to represent was dependent on the number of DATAFLOW© sampling cruises which varied from semi-monthly to monthly in our datasets.

4-3.3 SAV Habitat Analysis

For the SAV Habitat analysis, we tested the hypothesis that chl-*a* concentrations would be different in areas where SAV established compared to potential habitat where it did not establish, in a given year. We expected that if the chl-*a* values were different, it would be lower in areas that supported SAV. We tested this hypothesis by subsampling the DATAFLOW© observations and statistically comparing the values between these two zones. The major steps of our analysis were to:

1. Delineate SAV zones of potential growth areas and beds that established in a given year
2. Subsample DATAFLOW© data to create balanced datasets
3. Statistically compare the water quality data in the two zones

4-3.3.1 Delineating SAV Zones of Existing and Potential SAV

Delineating and mapping the two SAV zones (existing and potential) was based on historic SAV survey data developed by VIMS (Orth *et al.* 2010b). Using GIS tools, we created a GIS map of historic bed distribution by merging all available SAV distribution data mapped between 1971 to the present. This amalgam of historic distribution was used to define the area of *potential* SAV habitat. Although we recognize that changes in water quality and bed conditions since the 70s can limit the relevance of this historic distribution to current restoration targeting, for this exercise, based on our experience evaluating conditions within recently mapped SAV beds, the historical distribution is a more accurate method for delineating potential habitat than delineations based on available data for depth and sediment type. Existing SAV distribution was developed from the VIMS data that corresponded to the DATAFLOW© cruise year. The existing SAV distribution data were intersected with the zones of potential distribution to create a unique potential distribution map for a given estuary and year.

A different technique was used to create the potential SAV zone in the Bush River because existing SAV distribution in each year was almost identical to the historic potential distribution. For this estuary we performed a slightly different test and compared water quality in the areas that supported beds to water quality in shallow areas (< 2 m) that did not support SAV. We expected that factors other than water quality were likely to be limiting growth in this version of "potential" area but included this test for completeness.

4-3.3.2 Subsampling Data in Existing and Potential Zones

Because the DATAFLOW© field crew avoids entering SAV beds, we were limited to using data points from months when SAV was absent, newly emerged, or senescent. In essence, **we were testing whether water quality conditions in spring and fall could be used to predict where grasses emerge and persist during the late spring and summer**. This is a completely different analysis from other studies that have evaluated the improvement that SAV has on water quality by promoting settling of fine particles (Gruber *et al.* 2011) because it is intending to inform predictions of where SAV beds may thrive.

Once the SAV zones were established, we subsampled an equal number of points from each zone using GIS techniques (Figure 4-2). The Hawth's tools extension for ArcGIS, was used to randomly subsample 25 points from the existing and potential zones throughout the estuary in a given year using the "random selection within subsets" option of Sampling Tools (Beyer 2004). The sample size per date was small because few points occur within areas of SAV growth and so a small sample size was needed to retain as many sampling dates as possible and maintain balance between the existing and potential zones. However, because multiple sampling dates were pooled, the sample sizes used in the statistical tests were quite respectable (e.g., 100-375 points for the multi-year tests).

In some cases, the number of points in a zone was insufficient to balance the sample. In these cases, a buffer of 30 m was applied to either the existing or potential SAV zone and points were randomly selected within the buffer. This approach was considered acceptable because water is not completely stationary but, rather, would be expected to move on the order of 5-10 km laterally on a tidal cycle (W. Boynton, pers comm.). Therefore, a 30m buffer was judged to be reasonable.

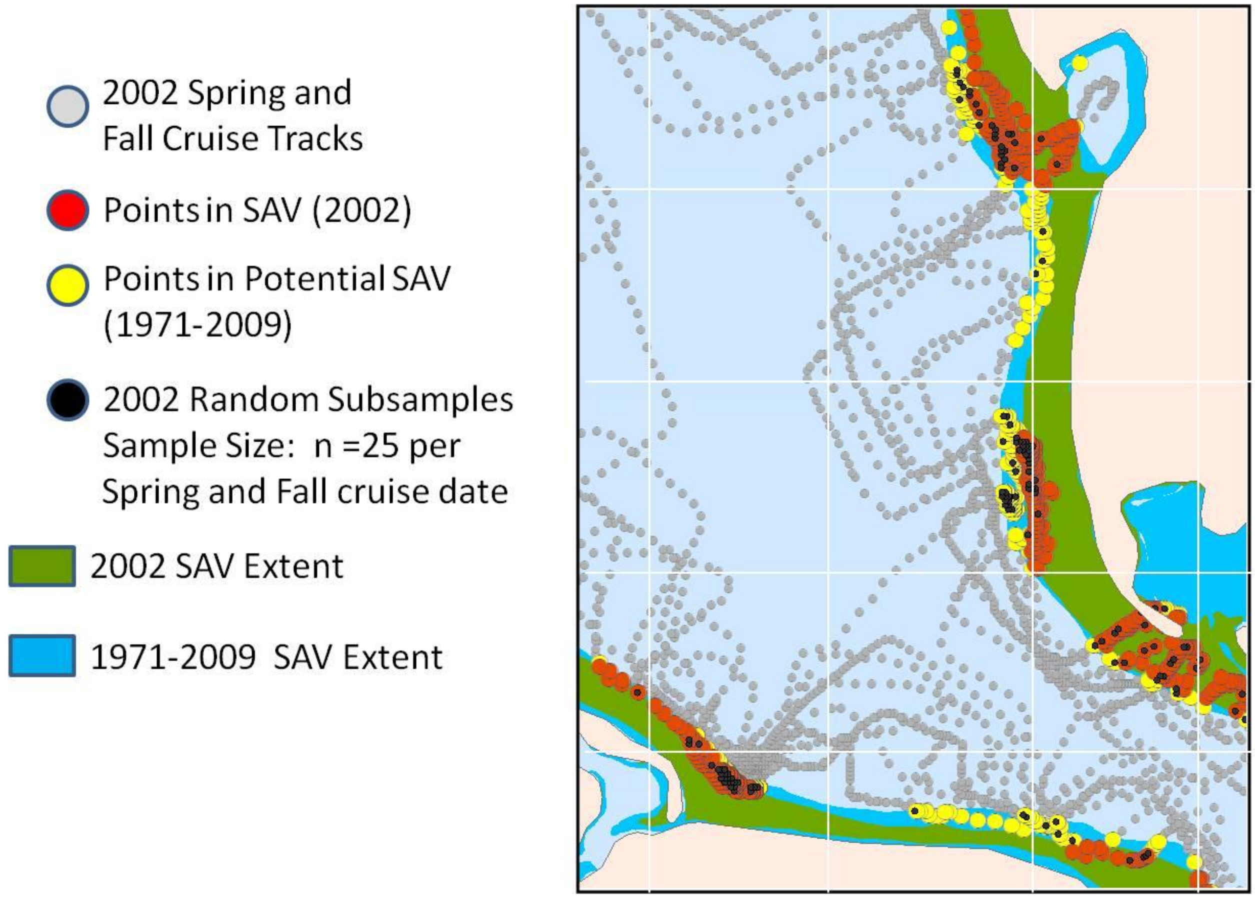


Figure. 4-2. Example of data subsampling by SAV zone in the Severn 2002. Data points were subsampled from the available data by randomly subsampling the points that fell with existing and historic SAV beds.

4-3.3.3 Statistical Test of Differences Between Groups

The zones with and without SAV were compared statistically using chl-*a* values. The chl-*a* values were compared for differences using the non-parametric Mann-Whitney U test available in Systat 13 (Systat, 2009). A non-parametric test was needed because the temporal and spatial autocorrelation of the data would bias the significance of a parametric test. The Mann-Whitney test uses comparative rankings to compare sets of data values to test the null hypothesis that the randomly subsampled points from each zone are not different – i.e., that they come from the same population. The test is sensitive to outliers, so the data were logged (ln) to create normal distributions of values.

4-4 Results and Discussion

4-4.1 Kriging Results and Spatial Patterns

4-4.1.1 Salinity Patterns

A somewhat surprising result of the spatial analysis was that it showed marked salinity gradients occur periodically across the channel of these small shallow estuaries. This was unexpected because cross-channel gradients are often the result of Coriolis forces which are not thought to act at these scales. Coriolis forces are known to be important in larger and deeper estuaries and they have traditionally been modeled for the Bay mainstem (Xu *et al.* 2002, Valle-Levinson and Atkinson 1999). Other drivers could explain the salinity gradient such as wind, density-driven flow, or rainfall patterns. However, it is interesting to note that the salinity gradient, when detected in western shore tributaries, was often consistent with counterclockwise flow of water entering the mouth of the sub-estuary from the Bay's mainstem and the gradients were often strongest near the mouth of the small estuaries (Figures 4-3 and 4-4). The Corsica showed cross-channel gradients but they were not consistent with water flowing in a counter-clockwise pattern after entering the mouth.

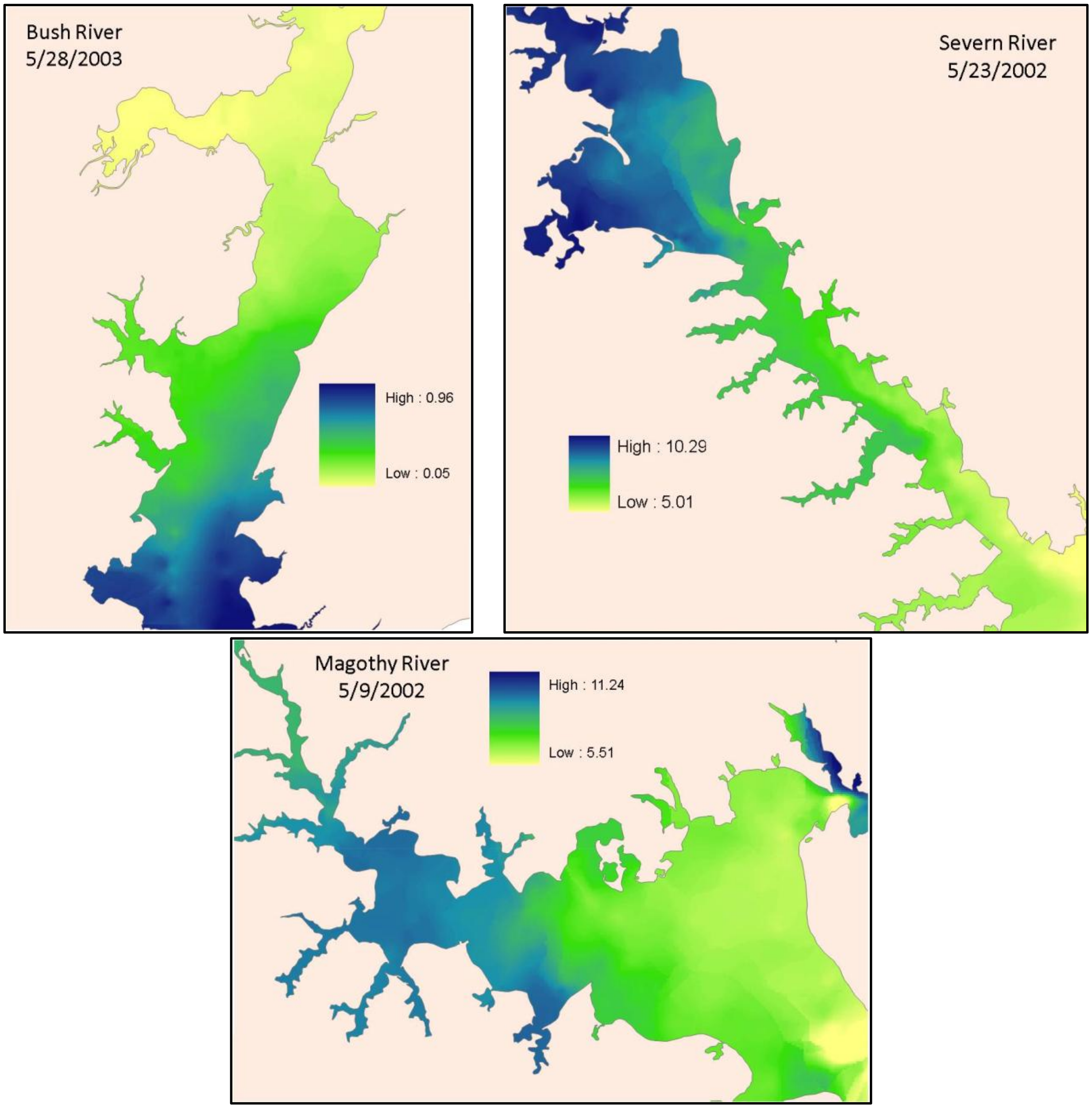


Figure. 4-3. Examples of cross-channel salinity gradients for individual cruise dates.

The salinity patterns also show that the water entering the mouths of small western shore sub-estuaries from the Bay can be either fresher or saltier than water in the mid-estuary, and the *down-estuary* gradient can vary seasonally (Figure 4-5). In addition, the slope of the salinity gradient *across* the mouth of the estuary reverses directions seasonally in all four of the estuaries we examined (Figure 4-4). The spring direction of the gradient (fresher on the N or E side of western shore tributaries; fresher on the S side of the Corsica) can be associated with freshwater inflow from the mainstem that can occur in the spring. The best example of freshwater entering from the mainstem is in the Magothy, which has a consistent pattern of peak salinity near the top of the wide portion of the estuary and lower salinity both towards the mouth and headwaters, throughout the spring (Figure 4-5c). The gradients across the mouth and down-estuary that are apparent in the western shore tributaries, suggest that these sub-estuaries are seasonally influenced by freshwater originating from the Susquehanna or other western shore tributaries since large influxes of freshwater from the Susquehanna or northwest tributaries would tend to hug the western shore and enter the case study tributaries.

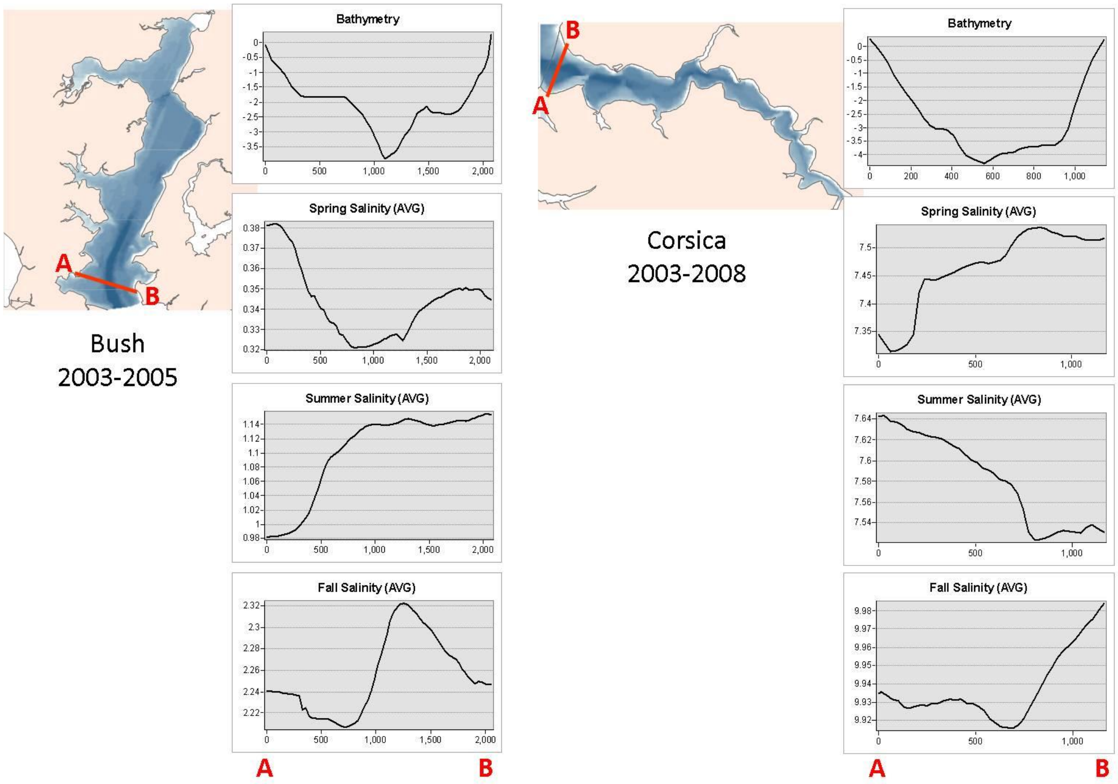


Figure. 4-4. Cross section profiles of average spring, summer and fall salinity for the Bush and Corsica Rivers. Average salinity was calculated at each cell location using interpolated maps (kriging) of all sample dates in the sampling period.

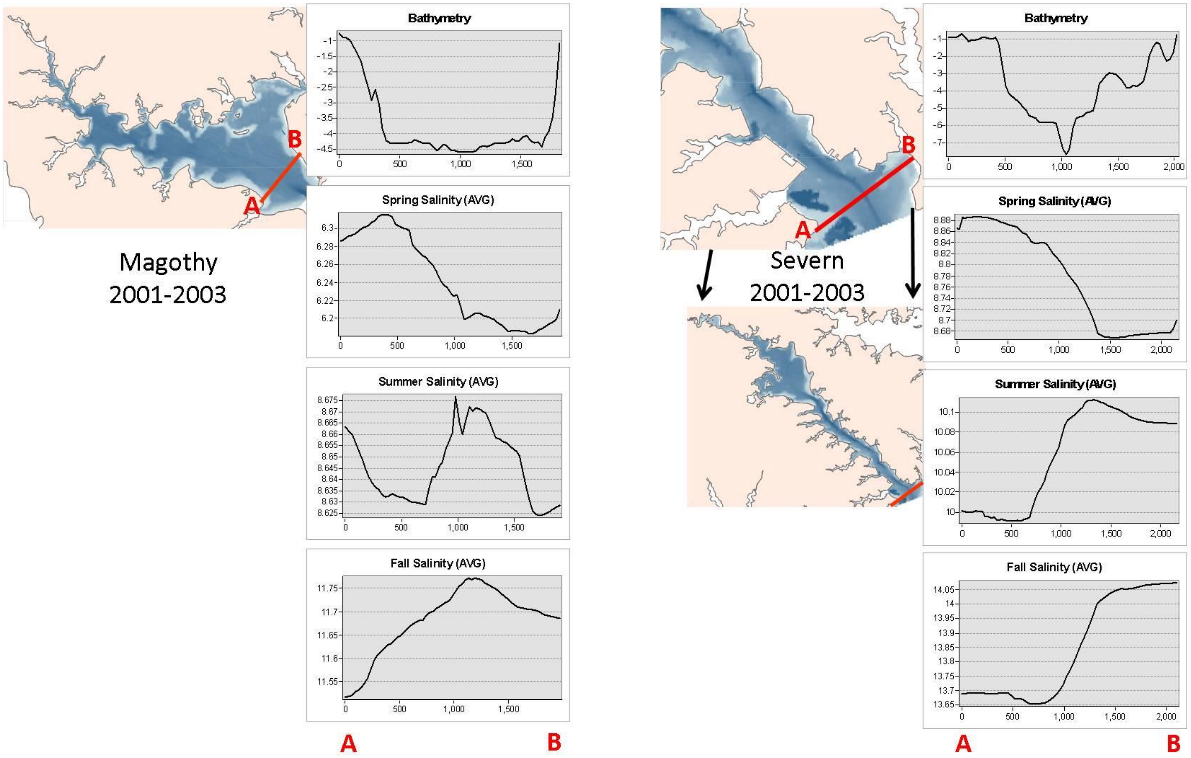
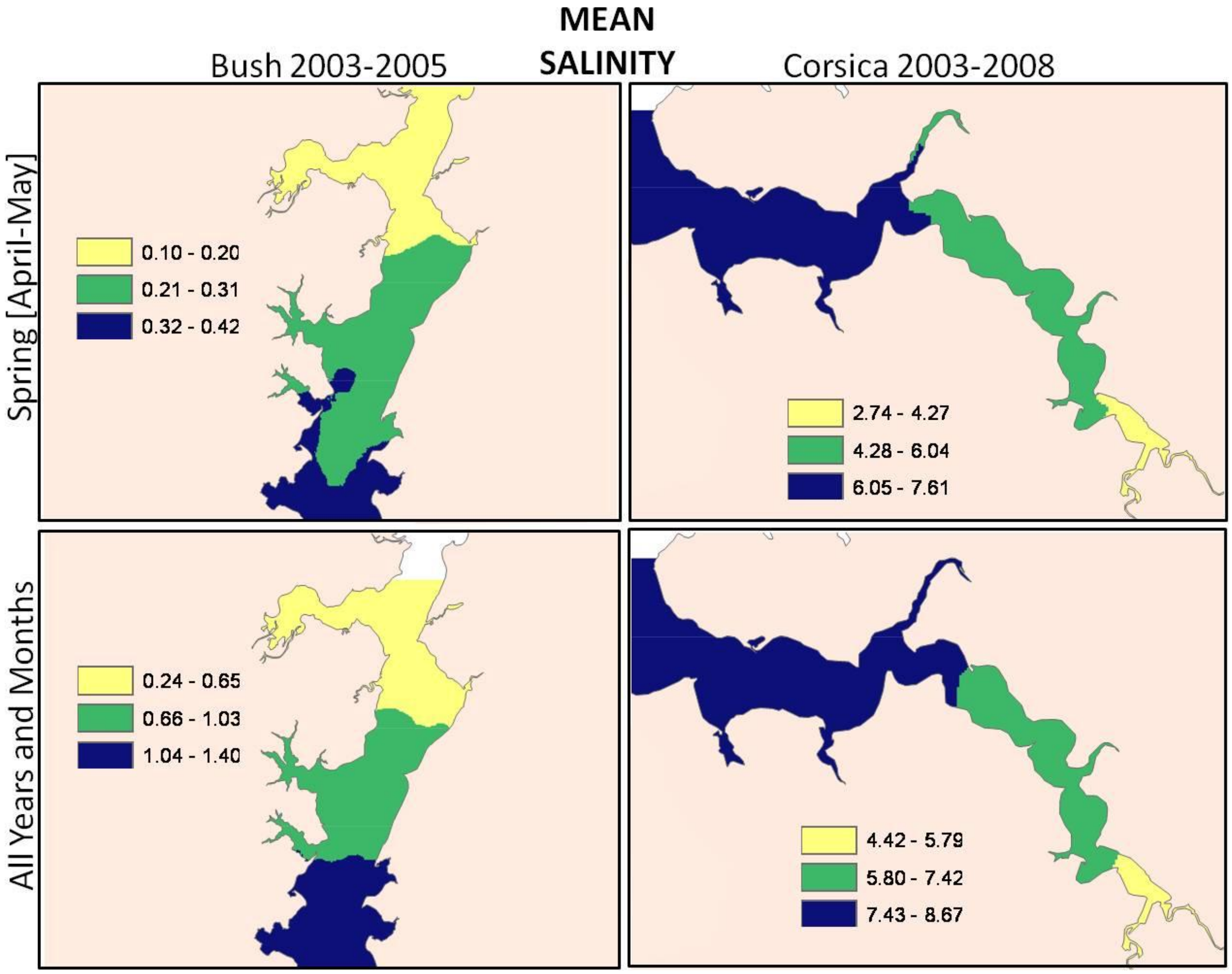


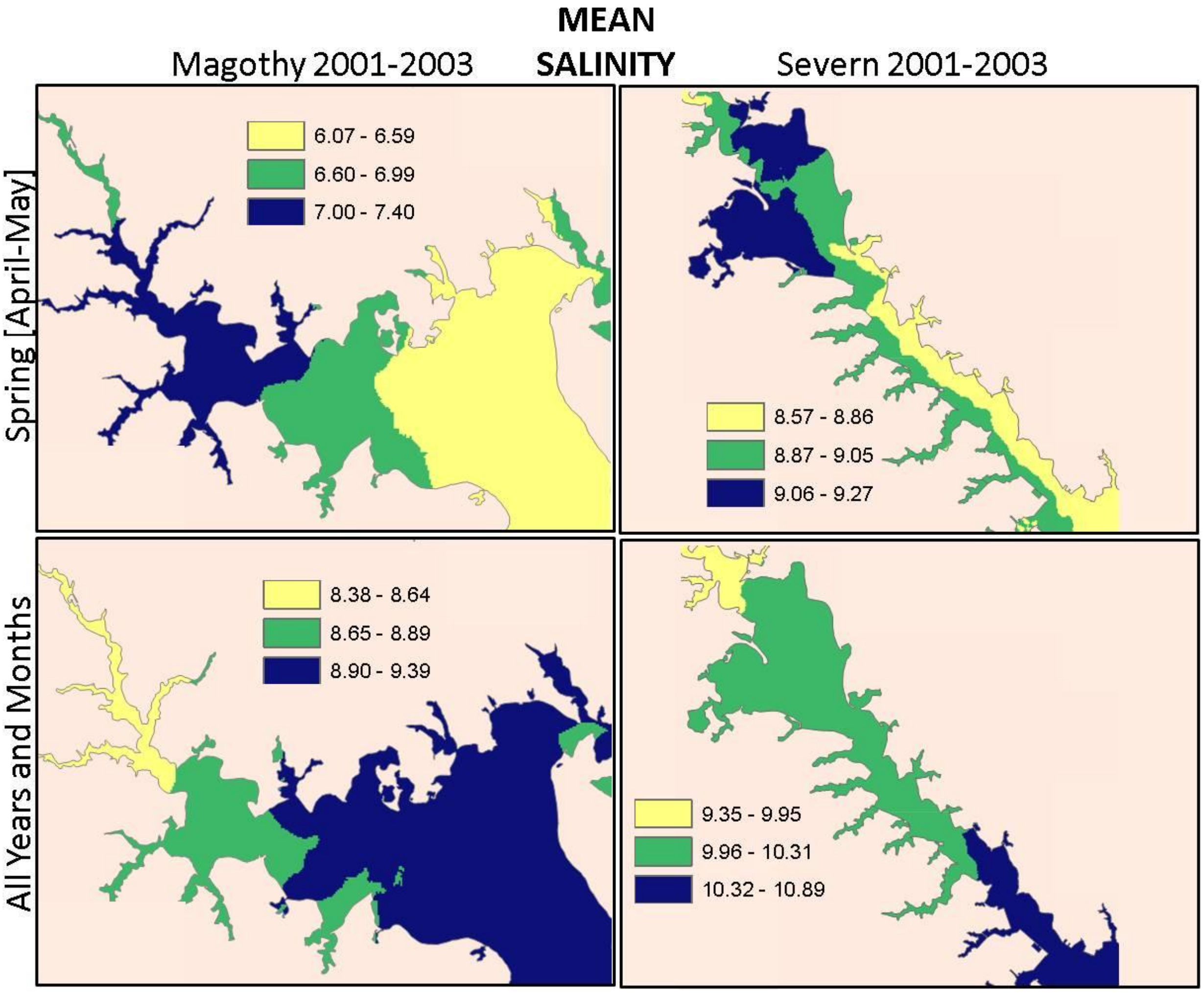
Figure. 4-4(continued). Cross section profiles of average spring, summer and fall salinity for the Magothy and Severn. Average salinity was calculated at each cell location using interpolated maps (kriging) of all sample dates in the sampling period.

Despite these seasonal differences, overall, the kriged results show that salinity is highly dynamic and, on average, across the sampling dates, the pattern is one of a down-estuary salinity gradient from low to high salinity (Figure 4-5). An exception is the Severn which shows a higher average salinity on the NE side when all dates are examined (Figure 4-5d). Intriguingly, the standard deviation of salinity values suggests that areas on the North and East sides of the western shore tributaries experience greater variability of salinity that may be a result of periodic Bay-water incursions into the N and E sections of the sub-estuaries (Figure 4-6).



(a) **(b)**

Figure. 4-5. Average spring salinity pattern for the Bush and Corsica Rivers. The salinity was divided into three categories to identify broad trends. The Jenks natural breaks classification was used to determine the divisions. This classification groups similar values within classes and maximizes differences between classes.



(c) **(d)**

Figure. 4-5 (continued). Average spring salinity pattern for the Magothy and Severn Rivers. The salinity was divided into three categories to identify broad trends. The Jenks natural breaks classification was used to determine the divisions. This classification groups similar values within classes and maximizes differences between classes.

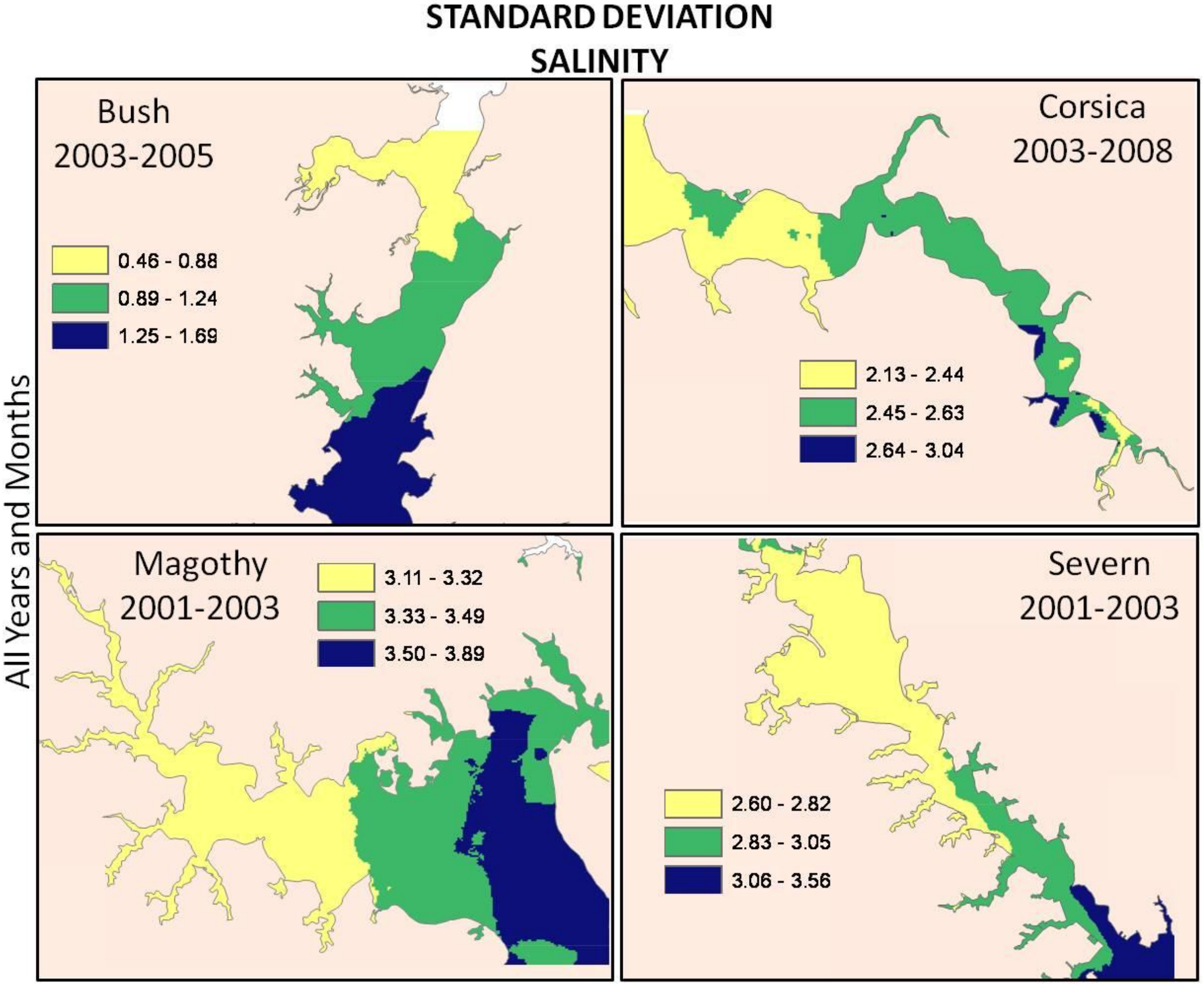


Figure. 4-6. Standard deviation of spring salinity for case study estuaries. The salinity was divided into three categories to identify broad trends. The Jenks natural breaks classification was used to determine the divisions. This classification groups similar values within classes and maximizes differences between classes.

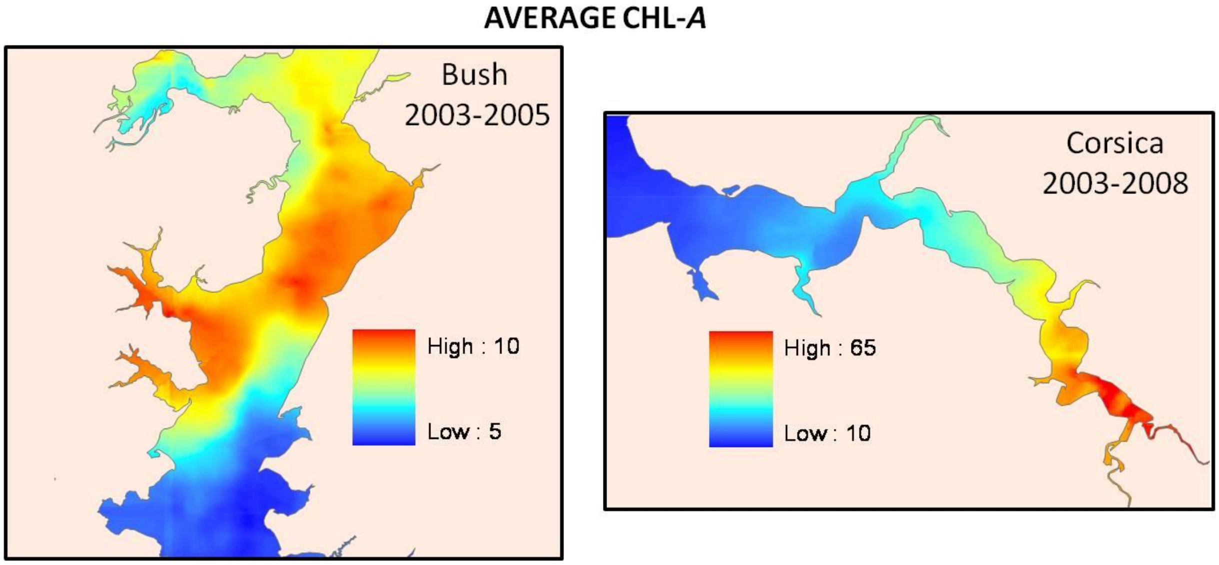
4-4.1.2 Chlorophyll-*a* Patterns

Unlike salinity, chl-*a* concentration does not show consistent patterns across estuaries. The simplest pattern was seen in the Corsica which shows a down-estuary decline in average chl-*a* concentration (Figure 4-7b). Another striking pattern was seen in the Severn, which showed substantially higher chl-*a* concentrations on the NE side of the estuary below Round Bay than on the SW side (Figure 4-7d). Both the Bush and the Magothy showed a peak in average chl-*a* concentration near the middle of the estuary (Figures 4-7-a and c), but the pattern in the Magothy defies simple description since patches of elevated average chl-*a* occur scattered throughout the estuary.

The results from individual cruises (Appendix 4-1) show that elevated chl-*a* can be widespread in the estuary on a given date and that, across the sampling season, large portions of the Corsica, Magothy and Severn have consistently high chl-*a* (Figure 4-8). However, the greatest variability in conditions over time tends to be more localized (Figure 4-9).

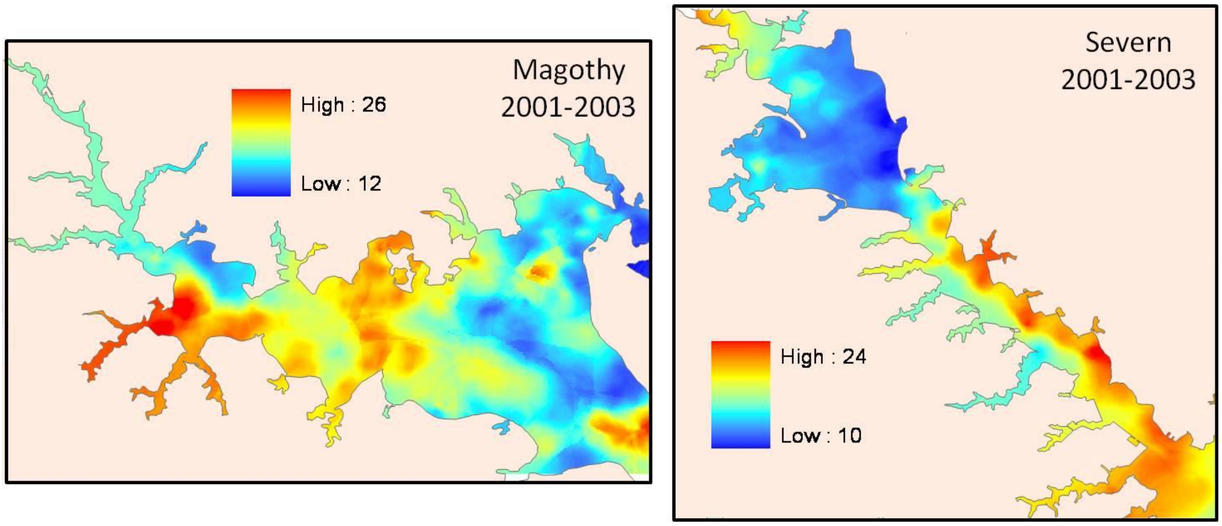
The variability of chl-*a* conditions can suggest where drivers of blooms change the most over time (e.g., seasonally or with weather patterns). Chl-*a* concentration was highest in the middle portions of the estuary for the Severn and the Bush. However, in the Corsica, variability was highest at the head of the estuary and, for the Magothy, variability is high near the mouth and at scattered sites throughout the estuary. There is a weak tendency towards higher variability in chl-*a* on the S and W portions of these estuaries, although for the Severn this is only apparent in Round Bay and not in the lower estuary (Figure 4-9).

These patterns of average concentration and variability suggest some potential explanations of chl-*a* conditions that are specific to each estuary, but they would need to be confirmed by evaluating hydrodynamics of each estuary. For example, the Corsica (Figure 4-7b) pattern suggests that blooms are being driven by nutrient input from the watershed, which would vary with rainfall, and that dilution, flushing or removal of the nutrients occurs as the water moves out to the Bay, thereby reducing the prevalence of blooms near the mouth. Given the high proportion of agricultural land use in the watershed, this is not a surprising pattern, but it is also possible that other forces are at work. The cross-channel gradient in the lower Severn (Figure 4-7d) is difficult to explain, but, because chl-*a* concentrations are generally higher near the mouth, it suggests that the Bay is a major source of nutrients that are promoting the blooms. However, this pattern might also be explained by phytoplankton dynamics or spatial variability of other sources of nutrients such as land use patterns and groundwater inputs. The pattern of chl-*a* hotspots observed just below the mouths of small tributaries on the NE side of the Severn (Figure 4-7d) suggests local inputs may be important in determining these patterns.



(a) **(b)**

Figure. 4-7. Average of chl-*a* for the Bush and Corsica Rivers.



(c) (d)

Figure. 4-7 (continued). Average of chl-*a* for the Magothy and Severn Rivers.

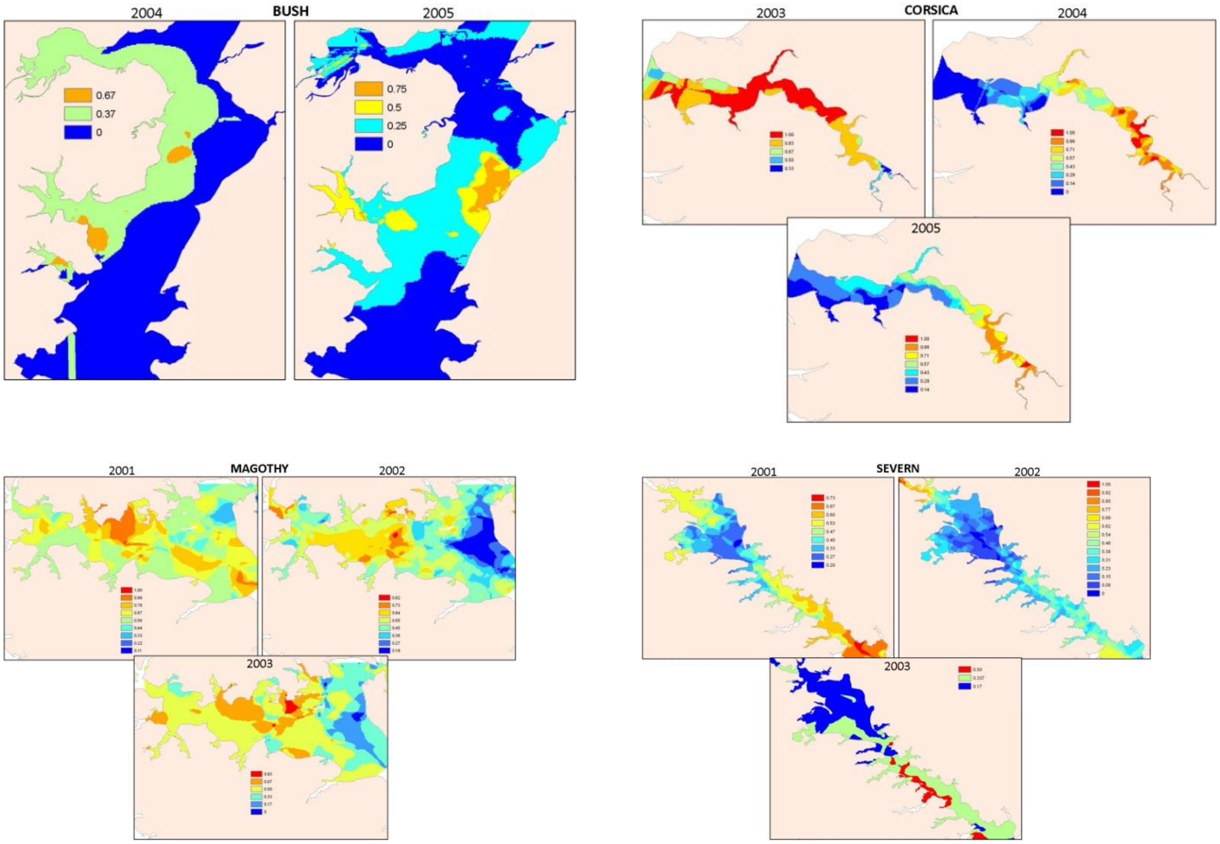


Figure. 4-8. Frequency of elevated chl-*a* for the case study estuaries.

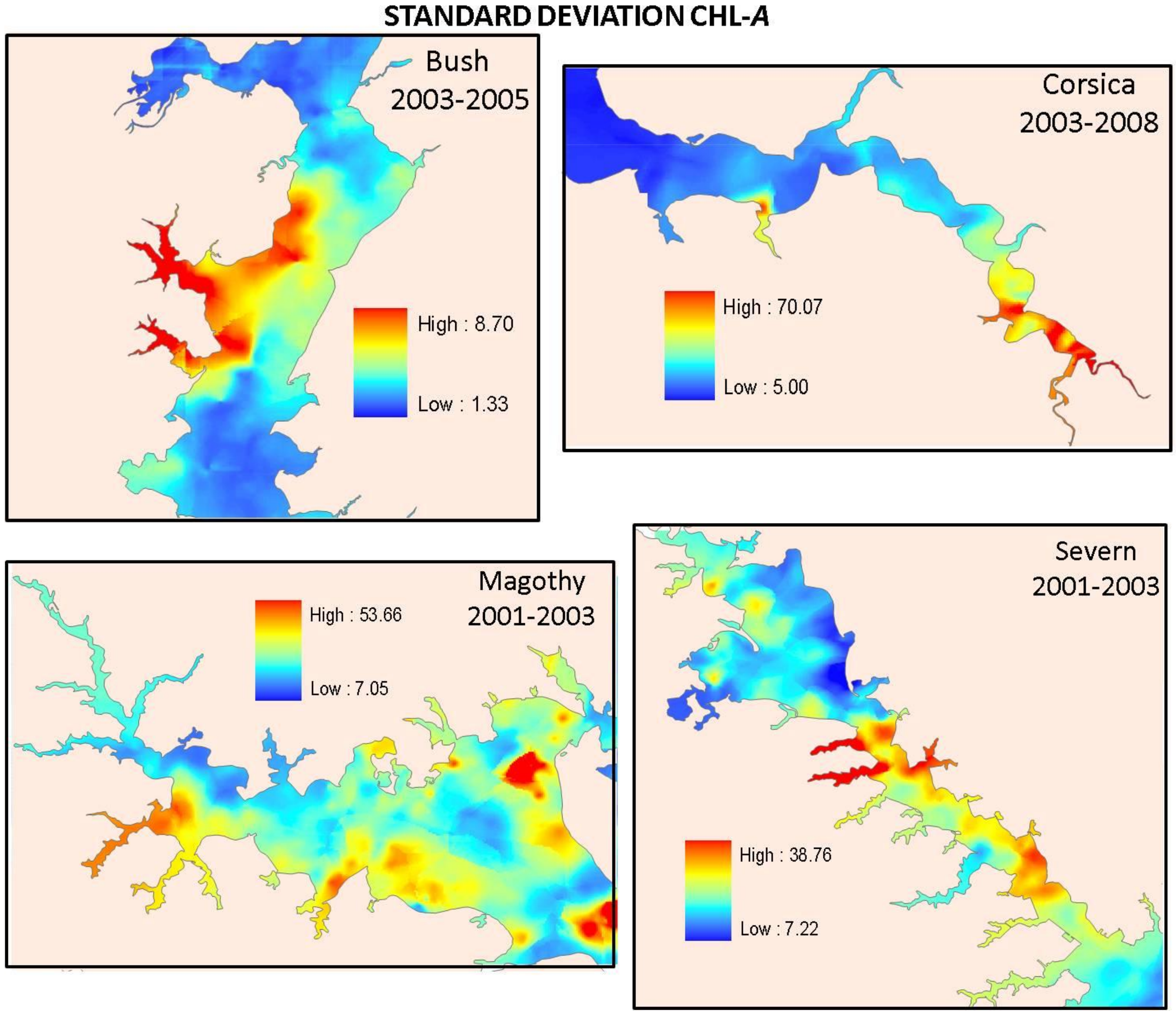


Figure. 4-9. Standard deviation of chl-*a*.

4-4.1.3 Relationships between Salinity and Chlorophyll-*a* Patterns

To further explore drivers of chl-*a* concentration, we evaluated whether observed salinity gradients might be associated with differences in chl-*a* conditions using several methods. First we evaluated the average over time of spatial patterns and found that these *average* conditions of salinity and chl-*a* within a given estuary did not correspond. However, when we looked at the spatial pattern of individual cruises, we found that patterns of chl-*a* and salinity could be correlated, but the pattern was not consistent enough to draw conclusions. We further explored a relationship we found in some estuaries, where peaks in chl-*a* concentration (blooms / hotspots) tended to form near the mouths of small tributaries. We examined whether these blooms might be associated with freshwater fluxes from these tributaries. We evaluated salinity within and near these blooms and, in every bloom we examined, we found that salinity varied little across these blooms indicating that any local watershed contributions to these blooms could not be determined from DATAFLOW© data.

4-4.2 Integrated Ecosystem Assessment

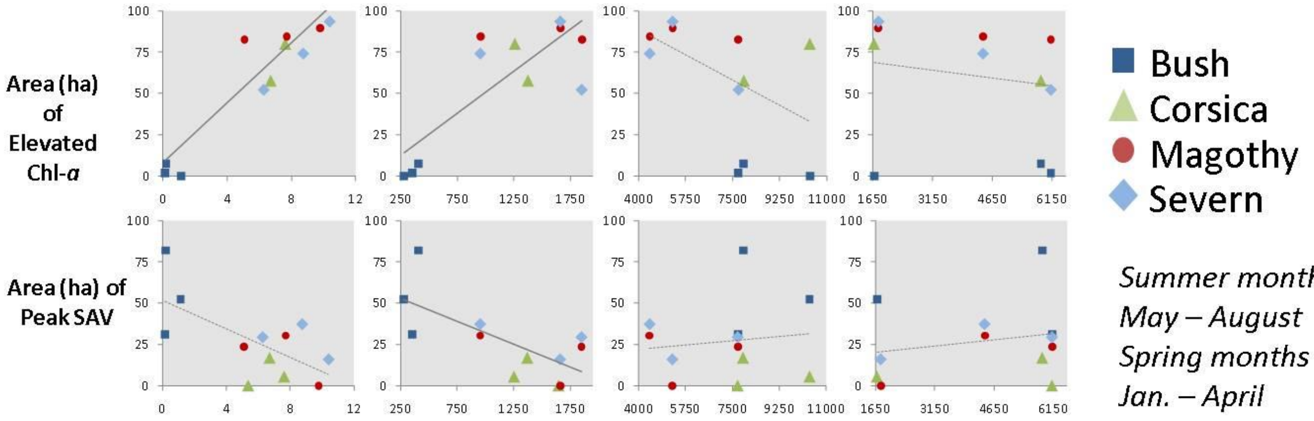
An integrated ecosystem assessment (IEA) is a tool to support ecosystem-based management of coastal ecosystems (Levin *et al.* 2009). We conducted a limited IEA of our sub-estuaries to reveal underlying links and feedbacks between stressors and key ecosystem responses. Through our data assessment, we have found some expected and unexpected relationships between stressor and response variables. The analysis suggests that some variables are well-correlated with SAV abundance and intensity of elevated chl-*a*. However, the weakness of some relationships (and/or unexpected directions of correlation) between outcomes and variables that would generally be considered to be obvious drivers of condition, reflects both the complexity of estuarine systems and the difficulty of identifying individual explanatory factors in the face of complex interactions of system variables. It can be particularly challenging to identify simple relationships for sub-estuaries that share a common water body connection compared to studies that evaluate whole estuaries. Summaries of conditions within whole estuaries are better able to represent how different levels of overall stress (e.g., watershed development) relate to in-water habitat. For sub-estuaries, it tends to be more challenging to explain localized variability, but also more meaningful for management, when variables can be identified that can explain differences in localized levels of system response.

Figure 4-10 shows a set of scatter plots each representing the relationship between one driver and one system response variable. The points represent three annual assessments for the four case study estuaries (n = 12). 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.

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.

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. 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. As expected, the Spearman Rank correlation coefficient between these two variables is negative ($\rho = -0.78$) and significant at the 0.05 level.

Of the drivers we tested, the following were significantly correlated with chl-*a*: summer salinity ($\rho = 0.66$), precipitation ($\rho = 0.60$), and septic density in the 1000’ buffer ($\rho = -0.65$) (Table 4-4 and Figure 4-10). However, other correlations that were close to being significant were spring Susquehanna flow ($\rho = -0.53$), depth ($\rho = 0.52$), and total nitrogen load ($\rho = -0.51$). Septic density, spring Susquehanna flow and nitrogen loads showed the opposite sign of the expected correlation.



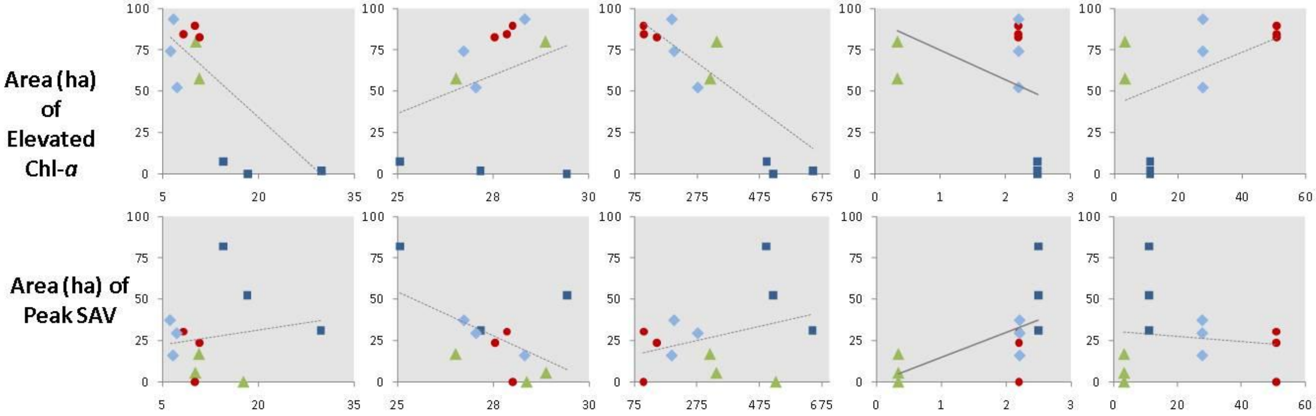
Summer months include May – August
Spring months include Jan. – April
**value X 1,000*

(a) Median Summer Salinity

(b) Total Summer Rainfall (m3*) per watershed*

(c) Spring Susquehanna flow (cfs*)

(d) Summer Susquehanna flow (cfs*)



(e) Turbidity (NTU)

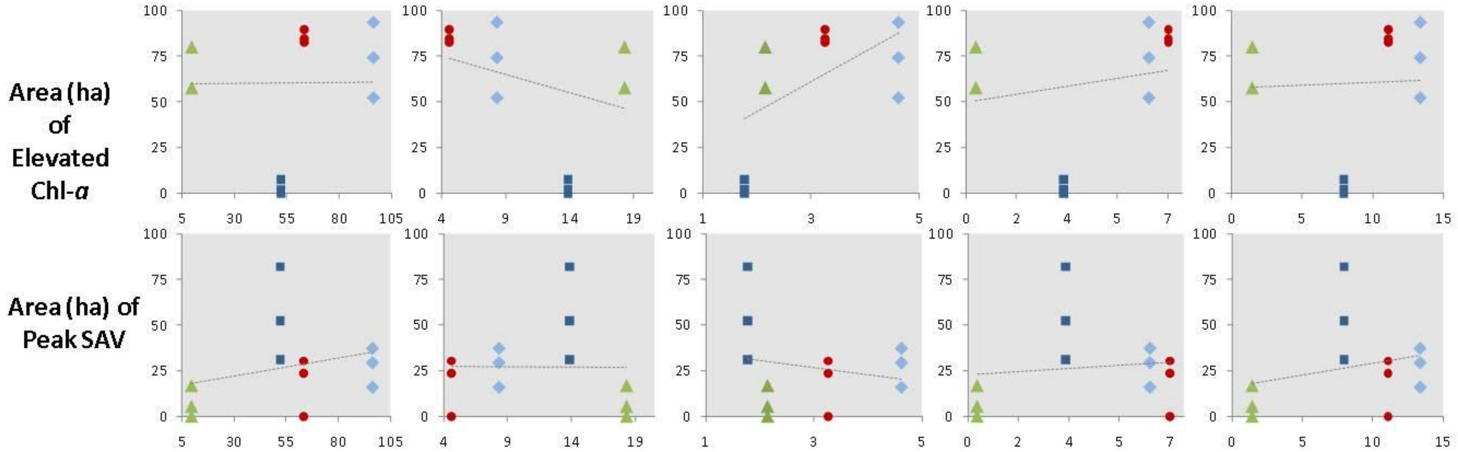
(f) August Temperature (°C)

(g) Modeled TN

(h) Septic Density within 1,000 ft Buffer

(i) Distance Weighted Density of Septics

Figure. 4-10. Scatter plots representing univariate correlations between stressor and response variables.



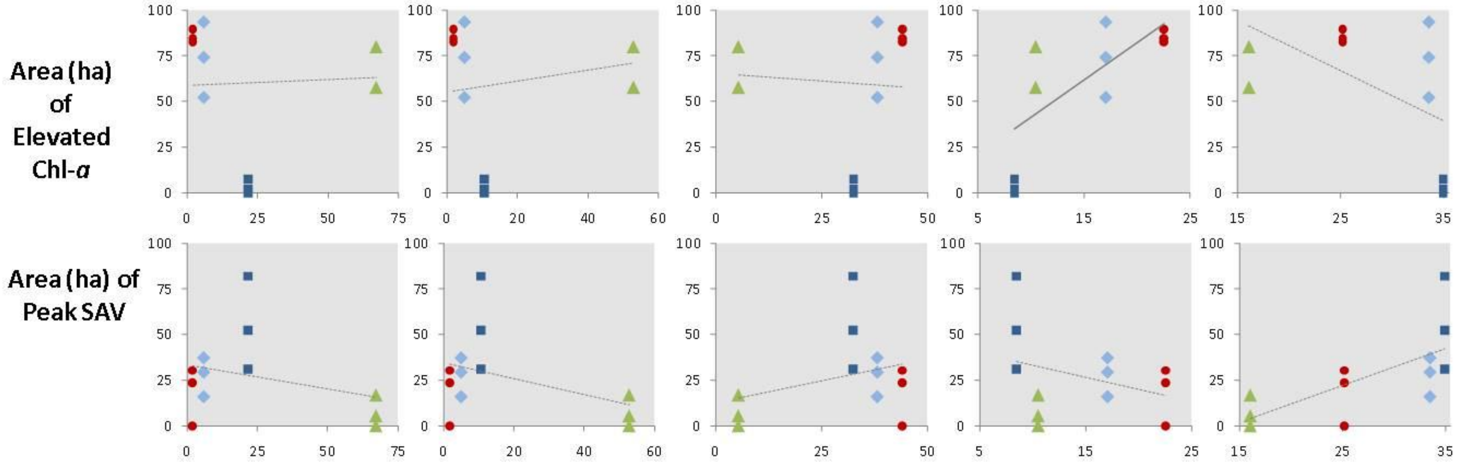
(j)
Estuary Volume
(m^3 x 10^6)

(k)
Watershed Surface Area: Estuary Surface Area Ratio

(l)
Mean Depth
(m)

(m)
Population Density
(people/HA)

(n)
Impervious Surface



(o)
Ag
(%)

(p)
Row Crop
(%)

(q)
Development
(%)

(r)
Wetlands
(%)

(s)
Forest Cover
(%)

Figure. 4-10 (continued). Scatter plots continued.

Table 4-3. Spearman correlation coefficients between stressor and response variables. Asterisk (*) represents a significant relationship at α 0.05.

VARIABLES	Area Elevated Chl-*a*	Area Peak SAV	Median Summer Salinity	Spring Susquehanna Flow	Summer Susquehanna Flow	Total Summer Rainfall	Watershed : Estuary Surface Area Ratio	Estuary Volume	Mean Depth	Turbidity
Area Elevated Chl-*a*	1.000									
Area Peak SAV	-0.7776*	1.000								
Median Summer Salinity	0.6573*	-0.504	1.000							
Spring Susquehanna Flow	-0.531	0.104	-0.5880*	1.000						
Summer Susquehanna Flow	-0.043	0.047	-0.502	-0.118	1.000					
Total Summer Rainfall	0.5975*	-0.6637*	0.422	-0.267	0.296	1.000				
Watershed: Estuary Surface Area Ratio	-0.259	-0.076	-0.367	0.7085*	0.000	-0.358	1.000			
Estuary Volume	0.130	0.195	0.453	-0.7085*	0.000	0.358	-0.8000*	1.000		
Mean Depth	0.518	-0.292	0.7557*	-0.7085*	0.000	0.6837*	-0.6000*	0.8000*	1.000	
Turbidity	-0.406	0.172	-0.8531*	0.6239*	0.265	-0.453	0.475	-0.7125*	-0.9069*	1.000
August Temperature	0.343	-0.434	0.175	0.165	-0.524	-0.004	0.086	-0.194	-0.086	0.161
Total Nitrogen	-0.511	0.266	-0.7273*	0.6310*	0.230	-0.552	0.8205*	-0.6262*	-0.7341*	0.7203*
Population Density	0.259	0.076	0.367	-0.7085*	0.000	0.358	-1.000	0.8000*	0.6000*	-0.475
Impervious Surface	0.130	0.195	0.453	-0.7085*	0.000	0.358	-0.8000*	1.0000*	0.8000*	-0.7125*
Forest	-0.324	0.552	0.065	-0.354	0.000	-0.065	-0.400	0.8000*	0.400	-0.389
Agriculture	-0.259	-0.076	-0.367	0.7085*	0.000	-0.358	1.0000*	-0.8000*	-0.6000*	0.475
Developed	0.259	0.076	0.367	-0.7085*	0.000	0.358	-1.000	0.8000*	0.6000*	-0.475
Wetland	0.6478*	-0.411	0.6693*	-0.7085*	0.000	0.6837*	-0.8000*	0.6000*	0.8000*	-0.6693*
Septic Density within 1,000 ft. Buffer	-0.6478*	0.7895*	-0.410	0.000	0.000	-0.488	-0.200	0.400	-0.200	0.173
Distance Weighted Density of Septics	0.259	0.076	0.367	-0.7085*	0.000	0.358	-1.000	0.8000*	0.6000*	-0.475

Table 4-3 (continued).

VARIABLES	August Temperature	Total Nitrogen	Population Density	Impervious Surface	Forest	Agriculture	Developed	Wetland	Septic Density within 1,000 ft. Buffer	Distance Weighted Density of Septics
August Temperature	1.000									
Total Nitrogen	0.063	1.000								
Population Density	-0.086	-0.8205*	1.000							
Impervious Surface	-0.194	-0.6262*	0.8000*	1.000						
Forest	-0.259	-0.086	0.400	0.8000*	1.000					
Agriculture	0.086	0.8205*	-1.000	-0.8000*	-0.400	1.000				
Developed	-0.086	-0.8205*	1.0000*	0.8000*	0.400	-1.000	1.000			
Wetland	0.022	-0.9284*	0.8000*	0.6000*	0.000	-0.8000*	0.8000*	1.000		
Septic Density within 1,000 ft. Buffer	-0.216	0.259	0.200	0.400	0.8000*	-0.200	0.200	-0.400	1.000	
Distance Weighted Density of Septics	-0.086	-0.8205*	1.0000*	0.8000*	0.400	-1.000	1.0000*	0.8000*	0.200	1.000

The variables that were correlated with SAV were similar, but the direction of correlation was reversed from our hypothses: precipitation (ρ = -0.66) and septic density in the 1000' buffer (ρ = 0.79). Salinity was near significant (ρ = -0.50) as was % forest (ρ = 0.55) and both had the expected signs. Another surprising result was that the modeled TN load was inversely correlated with chl-*a*, although the relationship was not significant.

The fact that density of septics in the thousand foot buffer was negatively correlated with chl-*a* and positively correlated with SAV abundance does not exonerate septics as a source of nutrients. However, this result probably indicates that the level of agriculture or other variables must be controlled for, before establishing the effect of septic density in the riparian buffer.

Other relationships between variables are intriguing but more data are needed to confirm these relationships. Of particular interest is the negative but insignificant correlation (ρ = -0.43) between average August water temperature (as measured by one or more DATAFLOW© datasets) and SAV abundance (Figure 4-10f). If we examine the relationship between these variables for individual estuaries (by isolating the three points for each estuary shown in Figure 4-10) the Bush, Severn and Corsica and the four estuaries individually and when combined, all suggest that higher August temperature, as measured by DATAFLOW©, is associated with lower abundance of SAV. If this relationship could be demonstrated across many estuaries, it would provide support for hypotheses that summer temperature is a critical SAV habitat variable (Moore and Jarvis 2008, Jarvis and Moore 2010) and that warming temperatures could pose a risk to SAV survival in the long term.

Work in our companion project will evaluate whether multi-variate statistical analyses can improve the ability to use these types of variables to explain variability among estuaries. For this univariate analysis, it is useful to note that the Bush estuary, which is the only oligohaline estuary in the group, often drives the direction of the correlation between stressors and responses (Figure 4-10). For example, if we look at the graphs of total summer rainfall vs. chl-*a* and vs SAV abundance, we see that if we remove the Bush data, we would not have a clear correlation between stressors and responses (Figure 4-10b). This is not universally true, but it is clear that more data need to be analyzed to confirm these relationships.

4-4.3 Differences in Existing vs. Potential SAV Zone Water Quality

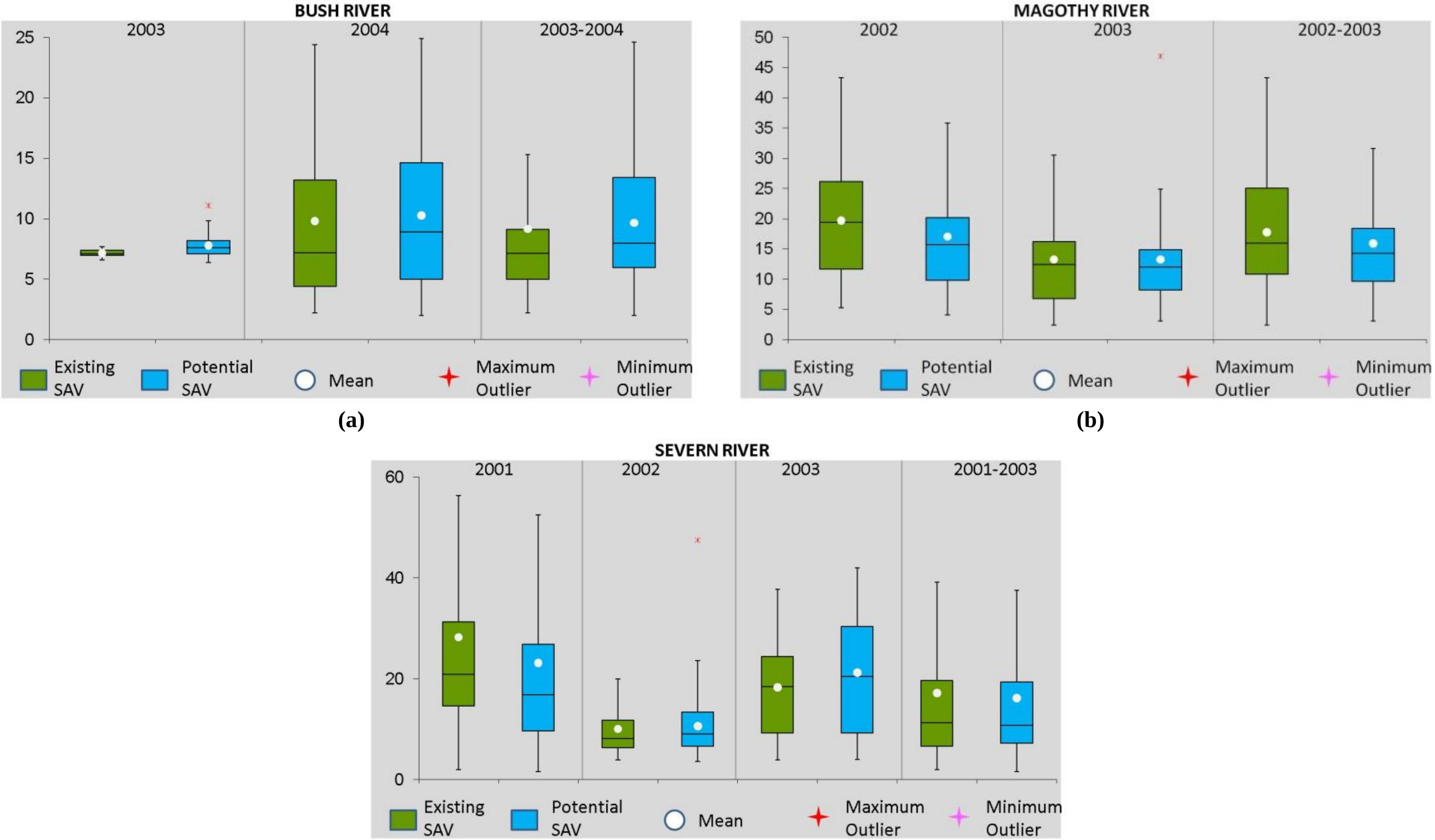
The results of the comparison of the water quality variable chl-*a* in areas of existing and potential SAV beds showed that this variable was not a strong predictor of whether SAV would thrive in a given location in a given year for any of the three western shore tributaries we evaluated. (The Corsica estuary was omitted from this analysis because we did not have sufficient DATAFLOW© data in the areas where SAV emerged.) Although some significant differences were found between areas that supported SAV and areas that had historically supported SAV, chl-*a* was not consistently higher or lower in the areas that supported SAV.

Our initial assessment of the data compared means and standard deviations of multiple variables between the sets of observations for the two SAV zones (Table 4-4). The sets of observations were similar for temperature, turbidity, and chl-a, but, on average, the set of points subsampled from potential SAV zones had an average depth that was 0.5 – 1 m deeper than the those drawn from areas that supported SAV beds. This difference in the two sets is an unintended side effect of the differential loss of SAV in deeper areas of former habitat.

We further compared the distributions of chl-*a* using box and whiskers plots which showed that differences between the chl-*a* values for the DATAFLOW© points drawn from inside a given year's beds and points drawn from potential habitat outside those beds were often similar. When data were combined across all years (Figures 4-11 a - c) the same pattern was seen. Using the Mann-Whitney non-parametric statistical test, we tested for significant differences between these two groups of data points. For the Severn, the two groups (existing and potential SAV areas) were not significantly different for any of the individual years 2001-2003 or for all years combined (Figure 4-11c). For the Magothy, the test suggested that the groups were significantly different ($p<0.05$) for 2002 and when all years were combined (Figure 4-11b). However, the chl-*a* was *higher* inside the areas that supported SAV, rather than outside, an unexpected result. For the Bush, the test showed significant differences for 2003 only and not for 2004 or for both years combined (Figure 4-11a). In 2003, chl-*a* was higher outside the beds, as expected. The test in the Bush, which is the only one that showed the expected relationship, was a slightly different test from the other estuaries, because, as discussed in the methods section, the potential SAV zone includes areas less than 2 m that have not historically supported SAV (Figure 4-11a). In all cases, the data used in the statistical analysis included only spring and fall sampling dates. Median chl-*a* values and statistical significance between sampled points in existing and potential SAV zones (based on the Mann-Whitney test) for each estuary are summarized in Table 4-5.

Table 4-4. Comparison of summary statistics for depth and water quality within existing and potential SAV zones for the Bush, Magothy and Severn estuaries.

Estuary	Existing SAV	Potential SAV
	Mean (Standard Deviation)	
Bush (2003-2004) / n=100		
Depth (m)	0.77 (0.38)	1.24 (0.26)
Temperature (°C)	19.30 (6.28)	19.36 (6.48)
Turbidity (NTU)	44.73 (31.80)	30.48 (32.44)
Chla (µg/L)	9.16 (6.27)	9.66 (5.33)
Magothy (2002-2003) / n=250		
Depth (m)	0.84 (0.61)	1.77 (1.47)
Temperature (°C)	20.27 (5.07)	19.73 (4.73)
Turbidity (NTU)	9.43 (6.36)	10.24 (4.98)
Chla (µg/L)	17.79 (9.62)	15.91 (9.77)
Severn (2001-2003) / n=375		
Depth (m)	1.6 (1.10)	2.25 (2.11)
Temperature (°C)	21.27 (4.21)	20.49 (4.87)
Turbidity (NTU)	6.77 (6.17)	5.86 (4.09)
Chla (µg/L)	17.22 (17.95)	16.2 (17.67)



(a)

(b)

(c)

Figure 4-11. Box and whisker plots of chl-*a* values by SAV zone for three estuaries. The 25th-75th percentile and the median values are represented as horizontal lines of the box. The mean value is shown as a dot. The whiskers around the box represent the tails of the distribution, where the upper whisker represents the maximum value and the lower whisker is the minimum value, with outliers excluded. Some maximum and minimum outliers are excluded to improve visual display and interpretation. Note: Bush 2003 variability is low because the sample size was lower (n=25) than 2004 (n=75).

Table 4-5. Summary statistics of sampled SAV points. Rows highlighted in light grey indicate significant difference in medians.

SUB-ESTUARY / YEAR	MEDIAN CHL*A* (actual SAV)	MEDIAN CHL*A* (potential SAV)	P-value
SEVERN 2001	20.9	16.8	0.150
SEVERN 2002	8.2	9.1	0.314
SEVERN 2003	18.5	20.5	0.260
SEVERN 2001-2003	11.3	10.8	0.868
MAGOTHY 2002	19.4	15.7	0.001
MAGOTHY 2003	12.4	12	0.838
MAGOTHY 2002-2003	16.0	14.3	0.005
BUSH 2003	7.1	7.6	0.021
BUSH 2004	7.2	8.9	0.385
BUSH 2003-2004	7.2	8.0	0.109

Our test demonstrated that the chl-*a* values collected by DATAFLOW© during times when SAV were not present were not able to strongly distinguish between areas that would or would not support SAV. The surprising result that chl-*a* values were often *higher* inside the areas that supported SAV than in areas outside the beds is more likely the result of random variation in the data than a significant finding. This result does not contradict other work that finds better water quality within SAV beds than outside of beds because we tested whether water quality was persistently better or worse during times when SAV was *not present* (i.e. spring and fall DATAFLOW© cruises.

Although we aimed, with our subsampling techniques, to create balanced data sets for the 2 SAV zones, it is likely that the two data sets do not share identical distributions of water quality characteristics. These differences could bias the test of significant differences but is difficult to avoid; as water quality has declined, the distribution of SAV beds have become restricted to shallower depths. As a result, a random selection of points from the areas that historically supported SAV beds is inevitably going to have a higher average depth than points randomly selected from existing beds.

Other factors are likely to have limited the strength of this test of the potential influence of local drivers of SAV establishment. First, chl-*a* was selected as a proxy for nutrient and light conditions, but it may not be the best variable to use to predict water quality important to SAV, particularly since the duration of elevated chl-*a* is not well-captured through this particular sampling technique. Second, the sampling of existing and potential beds is not completely random. Third, the sample size was small when compared to the high variability of these estuaries in space and time. Therefore, these results are somewhat preliminary since more areas should be tested before drawing strong conclusions.

4-5 Conclusions and Management Implications

The fine scale data provided by DATAFLOW© revealed some important estuarine dynamics. Some intriguing results were that salinity gradients were periodically strong across the channel of all of these small shallow estuaries. This was somewhat unexpected given that Coriolis-driven flow, which is one possible explanation for this gradient, is typically only associated with larger and deeper water bodies. However, other conditions could create this gradient, such as wind driven flow, groundwater inputs, among other factors. Also, further testing is needed before attributing these gradients to Coriolis forces since such forces may not be physically possible at this scale. Nonetheless, these cross-channel gradients suggest that water quality and habitat drivers may differ substantially on alternate sides of the estuaries, at least during some seasons.

The salinity gradients seen across the channels were most common during the spring and were not consistent through the year. In the western shore tributaries, it appears that northern and eastern sides of the estuary may be more strongly influenced by Bay water in the spring. The water on the N and E sides may be fresher than resident water and the evidence that it is coming from the Bay is that the zone of fresher water appears as a lobe entering from the mouth that does not extend up the full extent of the estuary. However, an annual average of data shows the typical down-estuary salinity gradient, since these patterns did not persist throughout the year.

The chl-*a* concentrations also displayed gradients or distinct areas of elevated chl-*a* when averaged over time suggesting that chl-*a* concentrations are not homogenous or random in the estuaries. For example, in the Severn, average chl-*a* shows a cross-channel gradient that is similar to spring salinity, with chl-*a* being higher on the NE side of the lower estuary. A finding from the spatial data that is important for characterizing habitat quality, is that the extent and persistence of elevated chl-*a* in the Corsica, Magothy and Severn suggest that SAV (and other living resources that respond negatively to high chl-*a*) will have difficulty finding relief from adverse conditions (Figure 4-8). In contrast, elevated chl-*a* was much less extensive and frequent in the Bush, and this was reflected in a higher abundance of SAV in this estuary. Almost all of the historic SAV habitat in the Bush was populated with plants during all three years of sampling.

The chl-*a* patterns suggest that local drivers are contributing to variability of conditions. Some consistent hotspots of elevated chl-*a* near small tributaries (particularly in the Severn) need to be further explored, as these suggest localized sources of nutrients. The Corsica’s pattern of higher chl-*a* and higher variability in the upper estuary contrasts with the western shore estuaries which tend to have higher chl-a near the mouth or near the middle of the estuary. The Corsica’s pattern suggests that conditions there are more likely to be driven by local watershed nutrient sources.

The integrated ecosystem assessment revealed that individual variables are generally poor predictors of estuarine responses. This has been demonstrated to be the case for estuaries and other highly variable aquatic ecosystems. For example, limnologists have developed several robust statistical models relating chl-*a* to nutrient loads (mainly phosphorous). They found generally weak relationships between these variables until additional variables were added to the model. Example of additional variables include, system depth characteristics and water residence time (Vollenweider 1976). A similar analysis for the Chesapeake Bay also yielded weak nutrient-chl-*a* relationships until additional variables were added, much like the limnologists' experience (Boynton and Kemp 2000). Similarly, examination of causes of Bay hypoxia were initially considered using a single causative variable (N load) (Hagy *et al.* 2004), but more recent examinations have invoked stratification, algal biomass and wind speed and direction as additional important factors (Skully 2010; Younjoo Lee, Pers Comm). These studies suggest that multivariate causal connections appear to be the norm, rather than the exception.

The strongest correlation we found at the whole-estuary level was that persistently elevated chl-*a* was negatively correlated with SAV abundance. Summer salinity was another variable that was significantly inversely correlated with SAV and positively correlated with elevated chl-*a*. Interestingly, higher local precipitation was correlated with elevated chl-*a* and lower SAV, as expected, but the watershed land cover variables that would be an associated driver with precipitation (e.g., % agriculture) were not significant. Similarly, precipitation was highly correlated with turbidity, but turbidity was not correlated with chl-*a* or SAV. These results suggest that further work needs to be done to identify combinations of conditions that can explain system variability.

Our test of whether DATAFLOW© data could be used to project SAV habitat quality, showed that the data have limitations for targeting SAV restoration. A statistical analysis demonstrated that the chl-*a* data collected during times when SAV was not present were not able to strongly distinguish between areas that would or would not support SAV. The choice of chl-*a* as the explanatory variable is vindicated, to some extent, by the high negative correlation of chl-*a* and SAV seen at the estuarine scale. Yet, using the available data, we could not detect this relationship at the site scale. Only one test of significance difference, in the Bush, had the expected negative relationship between chl-*a* and SAV. Otherwise, when a significant difference was found between zones, the chl-*a* was higher in areas that did not support SAV. In many cases, the differences were not significant.

The inability to use chl-*a* measurements to distinguish water quality differences at the site scale should not be seen as an indictment of this test in general. First, our analysis suggests that elevated chl-*a* is not randomly distributed in the estuaries, suggesting that it should be possible to differentiate areas that are better or worse habitat quality for SAV. However, it appears that more data are needed in shallow areas because we were not able to comprehensively test shallow areas throughout the estuary due to sampling patterns and water depth at sampling locations. The temporal frequency of the DATAFLOW© data appeared to be sufficient to establish where chl-*a* patterns are persistent, but longer time series would help to verify that result.

4-5.1 Future Directions

Many other techniques might be used to gain understanding about spatial variability of water quality and its relevance to aquatic habitat. Other drivers of SAV habitat quality and bed condition include temperature and spring turbidity which may be important to include in future statistical analyses. Further, some questions about drivers, such as sources of fresh water may be more directly examined by considering additional complementary datasets. Data from ConMon are another source of complementary data which could prove useful for estimating duration of water quality conditions. Additional investigations with DATAFLOW© datasets will include lagged effects of water quality and tests in more estuaries to expand the range of water quality conditions to better establish drivers.

4-5.2 Recommendations for Future Sampling

A key finding was that variability in estuaries confounds simple tests of drivers and responses. Although DATAFLOW©'s strength is capturing instantaneous spatial variability, the results of this analysis suggest that persistence and duration of timing of water quality stress may be more important than spatial variability. Therefore, to make progress, longer time series of data will be needed to evaluate the patterns of water quality dynamics and evaluate potential drivers. To address this need, it would be appropriate to pick sentinel estuaries for long-term monitoring with DATAFLOW©. Further, to better understand the relative influence of local drivers, it would be helpful to look for experiments of opportunity such as WWTP upgrades or widespread implementation of BMPs in a watershed, in order to monitor conditions before and after a change. As with other estuarine analyses, multi-year monitoring will generally be needed to overcome high natural variability in these systems.

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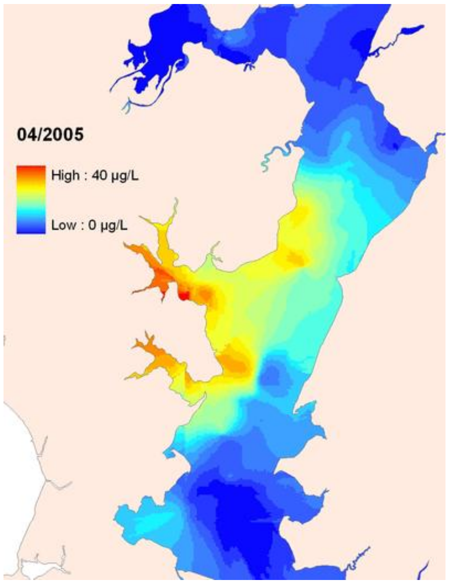
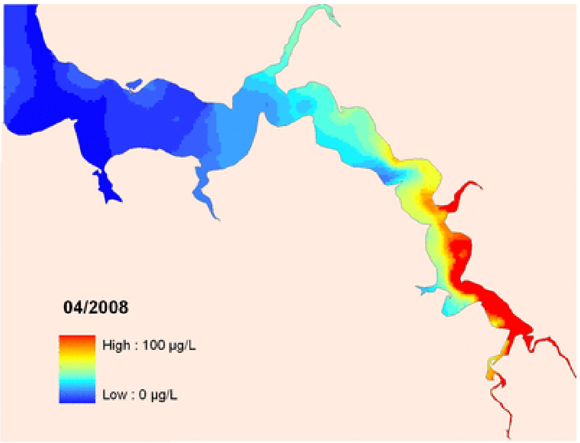
Appendix 4-1. Animations showing chl-*a* patterns through the sampling season for the case study estuaries. NOTE: In the folder labeled "Animations070611", right click on each GIF file, select Open With, and select a web browser such as Firefox or Internet Explorer to view the animations.

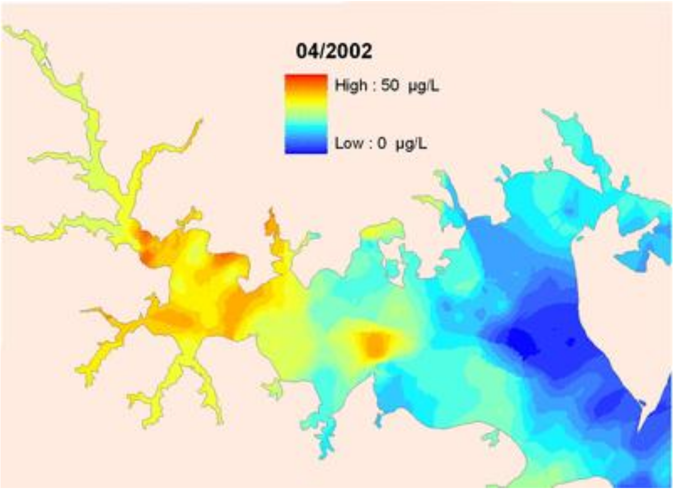
Bush River 2005



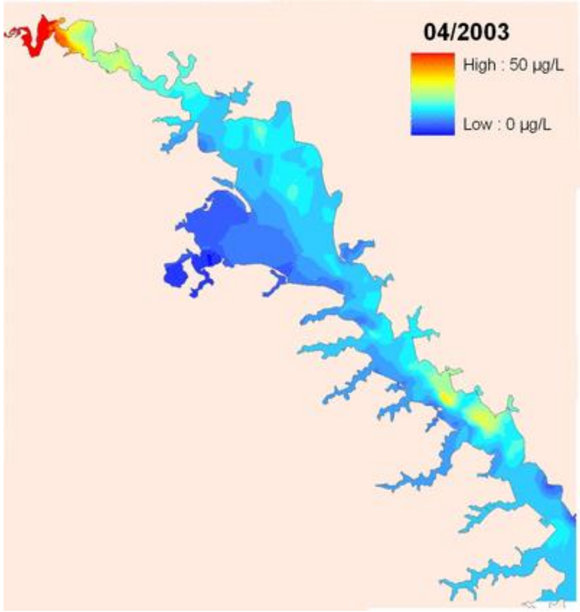
Corsica River 2008



Magothy River 2002



Severn River 2003



Chapter 5

A Case Study of a Chesapeake Bay Tributary System: Back River Estuary

W.R. Boynton, Y. Lee, M. Brooks and W.M. Kemp

5-1 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 and Ruhl and Rubicki, 2010).

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

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 Back River estuary was selected as one of several sites. We have included a draft version of the Back River analysis in the 2010 Ecosystem Processes Component (EPC) Interpretive Report because a substantial fraction of the Back River 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 by late spring, 2011.

5-2 Description of Back River Watershed and Estuary

The Back River watershed and associated estuary represent a truly urban estuarine system. The watershed and estuary lie immediately north of Baltimore City and the Patapsco River estuary and both areas have been intensely urban and industrial for at least the past one hundred years (Capper et al. 1983). The Back River watershed is relatively small (34, 882 acres; 14,116 ha). About 77% of the land is classified as urban (population density = 5.3 people/acre) and, associated with this intense development, about 41% of the land is in impervious cover. Typically, when impervious cover exceeds about 10% of a drainage basin, significant water quality impacts generally emerge. The basin is drained by about 73 miles (118 km) of streams. There have also been substantial historic wetland losses (7011 acres; 2837 ha) associated with development in this basin. Other details concerning the Back River watershed are provided in Table 5-1.

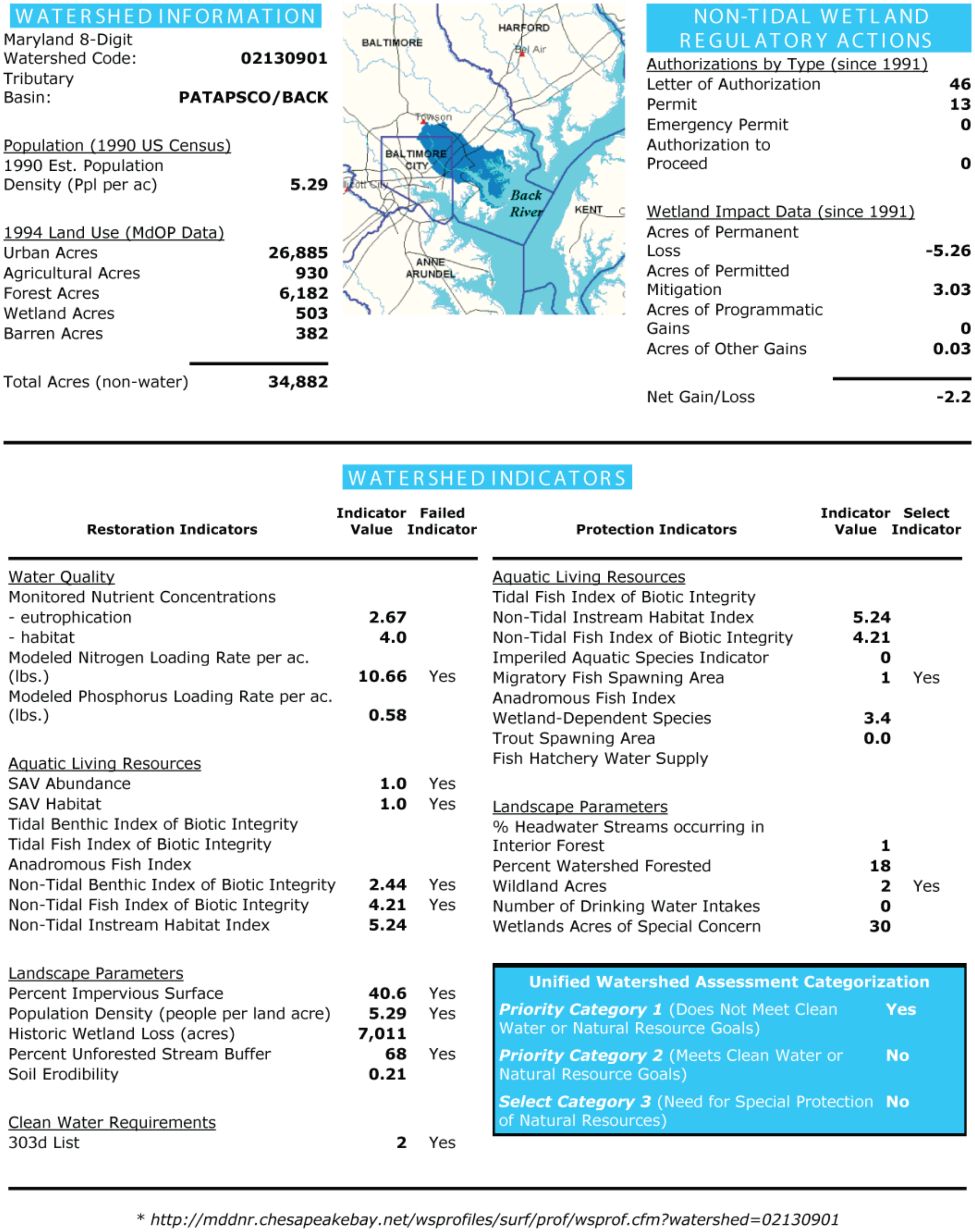
As might be expected, the general ecological condition of the Back River estuary can be described as very eutrophic. The estuary fails many indicators of water quality as measured by Maryland DNR and seagrasses, often used as an indicator of good water quality, are not present in this system (Table 5-1). Perhaps the most obvious sign of the eutrophic nature of the Back River ecosystem are the persistent algal blooms that occur at any time of the year (most frequently during summer) when chlorophyll-a concentrations often exceed 100 ug l-1. The water has a distinct green color and water clarity is seriously compromised (secchi disk depth < 0.5 m). There was no evidence of anoxia based on the Biomonitoring Program measurements but diel-scale hypoxia is common during summer periods (Boynton et al. 1998).

The estuary has a volume of about 56.9 x 106 m3, a surface area of 28.9 x 106 m2, an average depth of about 2 m, and is 8.5 nautical miles (14 km) in length. The ratio of basin area to estuarine surface area (often referred to as the dilution ratio) is about 5, a value much lower than those for other, larger tributaries of Chesapeake Bay and for the full Chesapeake Bay system (~14).

5-3 Previous Work in the Back River

The Back River is one of the many tributaries of Chesapeake Bay having but a single Maryland Long-Term Biomonitoring site located about mid-way along the axis of the estuary (Station WT4.1; Fig. 1). A variety of physical, chemical and a few biological variables have been routinely measured at this site (14 to 20 times per year since 1984) in both surface and bottom waters. These data and associated metadata are available from the Chesapeake Bay Program web page under the "Data Hub" tab (http://www.chesapeakebay.net/data_waterquality.aspx). In addition, sewage treatment discharge data are available for the same time period from the same source. Non-point inputs, based on a version of the Chesapeake Bay Program land-use model, were provided to us by the CBP (G. Shenk, pers. comm.). At present, there are no ConMon (high frequency water quality measurements made at fixed sites) or DataFlow (high spatial resolution measurements made at full estuary scale on a monthly basis) measurements available for the Back River although there are plans to initiate these measurements in the future (B. Michael, MD-DNR pers. comm.). There were a few short-term (Jun - Sep) high frequency measurements made at two locations in the Back River during 1997 and these data are described later in this report (Boynton et al. 1998). Finally, measurements of net sediment-water oxygen and nutrient exchange and sediment composition were made at several locations in the Back River, mainly during summer periods (Boynton et al. 1998). These are also presented later in this report.

Table 5-1. A detailed summary of watershed information and watershed indicators concerning the Back River watershed. This information is from the Maryland Department of Natural Resources web page (http://mddnr.chesapeakebay.net/wsprofiles/surf/prof/prof.html).



WATERSHED INFORMATION

Maryland 8-Digit Watershed Code:	**02130901**
Tributary Basin:	**PATAPSCO/BACK**
Population (1990 US Census)	
1990 Est. Population Density (Ppl per ac)	**5.29**
1994 Land Use (MdOP Data)	
Urban Acres	**26,885**
Agricultural Acres	**930**
Forest Acres	**6,182**
Wetland Acres	**503**
Barren Acres	**382**
Total Acres (non-water)	**34,882**

NON-TIDAL WETLAND REGULATORY ACTIONS

Authorizations by Type (since 1991)	
Letter of Authorization	**46**
Permit	**13**
Emergency Permit	**0**
Authorization to Proceed	**0**
Wetland Impact Data (since 1991)	
Acres of Permanent Loss	**-5.26**
Acres of Permitted Mitigation	**3.03**
Acres of Programmatic Gains	**0**
Acres of Other Gains	**0.03**
Net Gain/Loss	**-2.2**

WATERSHED INDICATORS

Restoration Indicators	**Indicator Value**	**Failed Indicator**
Water Quality		
Monitored Nutrient Concentrations		
- eutrophication	**2.67**	
- habitat	**4.0**	
Modeled Nitrogen Loading Rate per ac. (lbs.)	**10.66**	Yes
Modeled Phosphorus Loading Rate per ac. (lbs.)	**0.58**	
Aquatic Living Resources		
SAV Abundance	**1.0**	Yes
SAV Habitat	**1.0**	Yes
Tidal Benthic Index of Biotic Integrity		
Tidal Fish Index of Biotic Integrity		
Anadromous Fish Index		
Non-Tidal Benthic Index of Biotic Integrity	**2.44**	Yes
Non-Tidal Fish Index of Biotic Integrity	**4.21**	Yes
Non-Tidal Instream Habitat Index	**5.24**	
Landscape Parameters		
Percent Impervious Surface	**40.6**	Yes
Population Density (people per land acre)	**5.29**	Yes
Historic Wetland Loss (acres)	**7,011**	
Percent Unforested Stream Buffer	**68**	Yes
Soil Erodibility	**0.21**	
Clean Water Requirements		
303d List	**2**	Yes

Protection Indicators	**Indicator Value**	**Select Indicator**
Aquatic Living Resources		
Tidal Fish Index of Biotic Integrity		
Non-Tidal Instream Habitat Index	**5.24**	
Non-Tidal Fish Index of Biotic Integrity	**4.21**	
Imperiled Aquatic Species Indicator	**0**	
Migratory Fish Spawning Area	**1**	Yes
Anadromous Fish Index		
Wetland-Dependent Species	**3.4**	
Trout Spawning Area	**0.0**	
Fish Hatchery Water Supply		
Landscape Parameters		
% Headwater Streams occurring in Interior Forest	**1**	
Percent Watershed Forested	**18**	
Wildland Acres	**2**	Yes
Number of Drinking Water Intakes	**0**	
Wetlands Acres of Special Concern	**30**	

Unified Watershed Assessment Categorization	
Priority Category 1 (Does Not Meet Clean Water or Natural Resource Goals)	**Yes**
Priority Category 2 (Meets Clean Water or Natural Resource Goals)	**No**
Select Category 3 (Need for Special Protection of Natural Resources)	**No**

** http://mddnr.chesapeakebay.net/wsprofiles/surf/prof/wsprof.cfm?watershed=02130901*

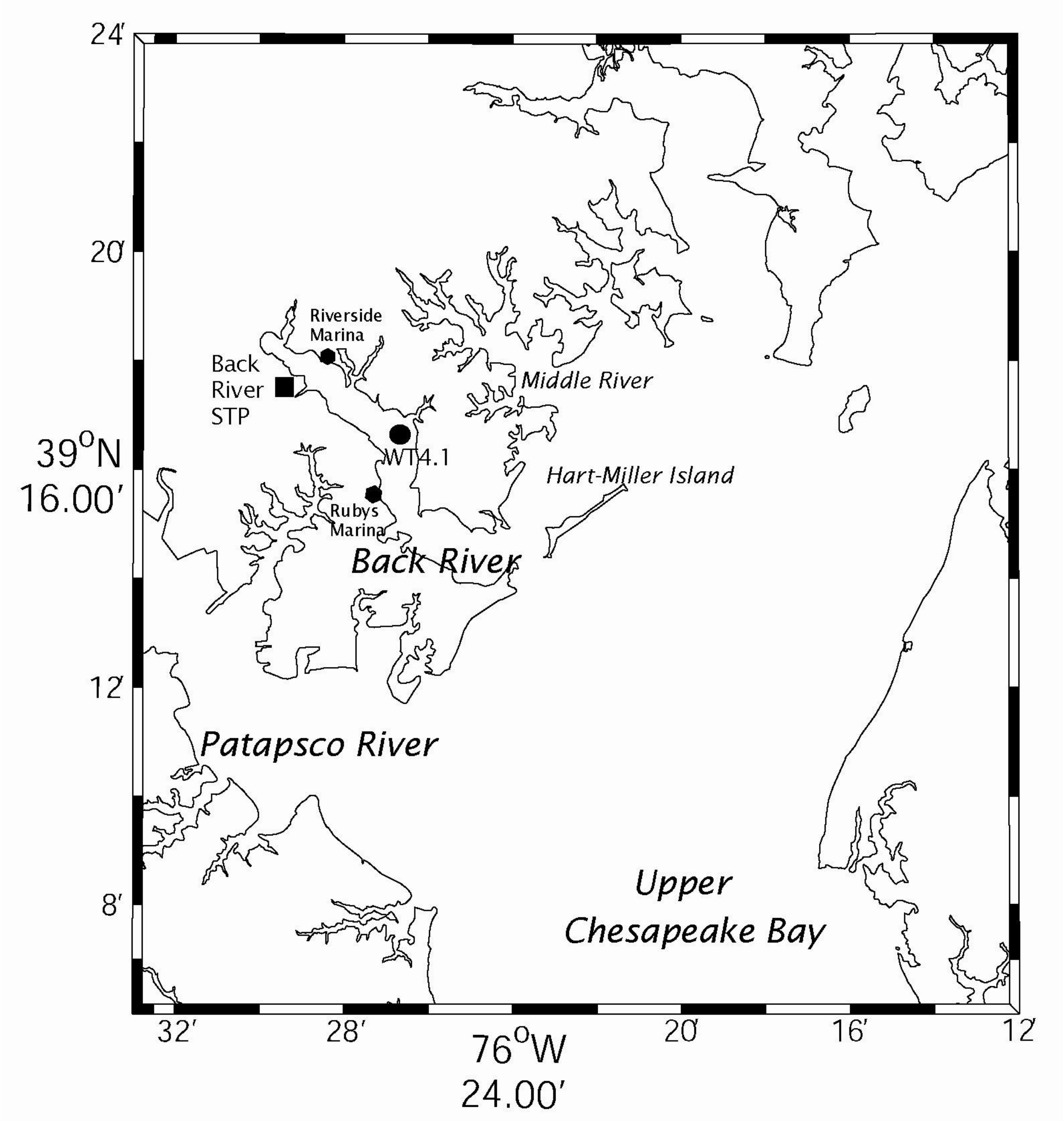


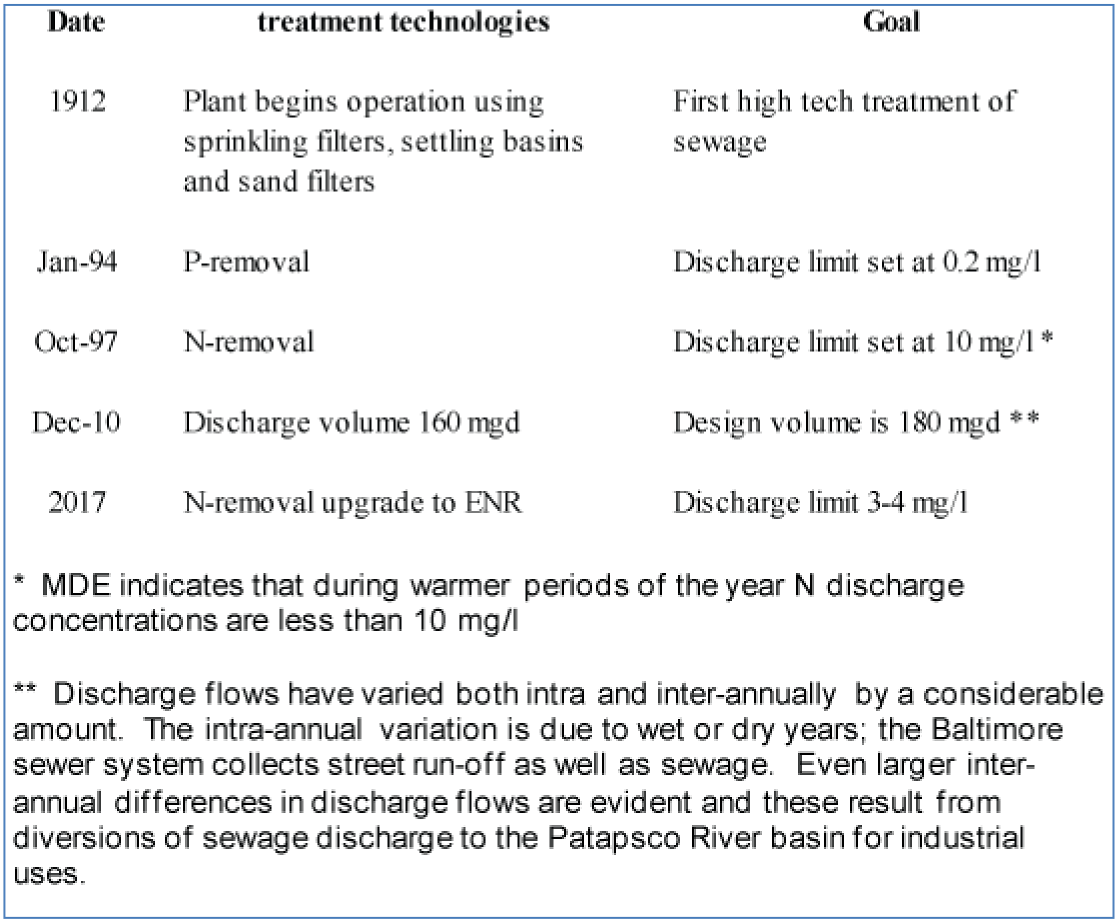
Figure 5-1. A map of the Back River, Maryland showing the locations of sampling stations and locations mentioned in the text.

5-4 Back River Estuary as a Threshold Site

This heavily enriched, small estuary may seem an unlikely choice as a candidate system possibly exhibiting threshold responses. However, one of the criteria used in selecting sites was the site needed to have been subjected to a major management action. Such is the case in the Back River. This system receives discharge from a major WWTP (Back River Plant) servicing a portion of the Baltimore metropolitan area. In recent years the plant has been upgraded to reduce BOD, sediment, phosphorus and nitrogen releases and work continues on upgrades, particularly those related to nitrogen removal. Table 5-2 provides a timeline of selected and major upgrades in plant operations.

The Back River facility has been in operation for almost 100 years and because of this long history a few words concerning the facility and sewage treatment in general are warranted. By the end of the Civil War some creeks in the Bay area receiving untreated sewage had become “offensive to the senses”. Included in these areas were the Patapsco, upper Potomac near Washington, DC and Hampton Roads, Virginia. Sewage Commissions were appointed to study and make recommendations concerning the situation in Baltimore in 1862, 1883 and 1893 and a report submitted in 1897 recommended building a sewer pipe that would discharge untreated wastes from 350,000 Baltimore residents directly into the Bay; the approach was labeled the “water carriage-dilution method”.

Table 5-2. A summary of operations and selected technical upgrades at the Back River WWTP between 1912 and 2010. Information is from Clapper et al. 1983 and Maryland Department of Environment (W. Saffouri, pers. comm.).



Date	treatment technologies	Goal
1912	Plant begins operation using sprinkling filters, settling basins and sand filters	First high tech treatment of sewage
Jan-94	P-removal	Discharge limit set at 0.2 mg/l
Oct-97	N-removal	Discharge limit set at 10 mg/l *
Dec-10	Discharge volume 160 mgd	Design volume is 180 mgd **
2017	N-removal upgrade to ENR	Discharge limit 3-4 mg/l

* MDE indicates that during warmer periods of the year N discharge concentrations are less than 10 mg/l

** Discharge flows have varied both intra and inter-annually by a considerable amount. The intra-annual variation is due to wet or dry years; the Baltimore sewer system collects street run-off as well as sewage. Even larger inter-annual differences in discharge flows are evident and these result from diversions of sewage discharge to the Patapsco River basin for industrial uses.

Another writer at the time indicated that the approach represented “construction of a great artificial intestine and anus” for the city of Baltimore (Capper et al. 1983). The sewage commissions viewed sewage wastes (organic matter, nitrogen and phosphorus) as good for augmenting oyster growth, an opinion held by some Maryland environmental officials at least until the early 1970s. However, concern for human health via eating contaminated oysters carried the day and the Bay discharge plan was finally abandoned. However, nothing was done with Baltimore sewage until 1904, the year of the great Baltimore fire, when the way was cleared (literally) for construction of a city-wide sewer system and the Back River was selected as the site for a new sewage treatment facility. The plant began operations in 1912 using sprinkling filters, settling basins and sand filters and was considered "one of the engineering wonders of the modern world" (Cappers et al. 1983).

In recent years the plant has been upgraded to remove more BOD5, sediment, phosphorus and, more recently, nitrogen. The more recent nitrogen load reduction was largely the reason the Back River was selected as a site for threshold examination.

5-5 Approach of this Evaluation

The choice of variables to be considered in the search for threshold responses is, of course, limited by data availability. In the case of the Back River, algal biomass (as indexed by chlorophyll-a concentration) appeared to be the best choice as the prime response variable. There is an obvious issue in this tributary with excessive algal growth and biomass accumulation. There is a solid record of this variable based on CBP Biomonitoring data collections (1984 – 2009). Furthermore, there is a strong causative link between nutrient supply and algal biomass conditions supported by an extensive literature in both lakes (e.g., Vollenweider 1976) and estuaries (Boynton et al. 1982; Nixon 1995). Other candidate variables were not considered for a variety of reasons (e.g., SAV are not present in this system; high frequency dissolved oxygen (DO) data are very limited).

We did have available a number of variables that could influence chlorophyll-a concentrations (and potential threshold responses) and these included excellent time-series measurements of both point and diffuse nutrient inputs, freshwater flow and sediment loads, nutrient, sediment and organic matter concentrations in the water column, temperature, salinity and water clarity. In addition, we had some measurements of high frequency DO patterns (15 minute interval measurements) for one summer period and sediment nutrient concentrations and net sediment – water exchanges of oxygen and dissolved nutrients for portions of several years. These latter measurements were organized because they may be useful in understanding long-term chlorophyll-a patterns and chlorophyll-a responses to load modifications. The time-scale of this evaluation involved examination of monthly time-series for a period of 23 years (1985 – 2007) and various longer-term averages (i.e., seasonal, annual and multi-annual) based on these monthly measurements.

5-6 Evaluation of Inputs

The major management action in this watershed concerned the upgrading of treatment levels at the Back River WWTP. In this section time-series of freshwater inflows (from both the WWTP and the watershed) and point and diffuse nutrient loads are presented.

5-6.1 Freshwater Input (Point and Diffuse Sources)

Monthly estimates of freshwater flow from the WWTP plant are shown in Figure 5-2. Flows have varied a good deal (2 to 8 m3 sec -1) between 1984 and 2007. The intra-annual variability can be attributed to seasonal run-off, mainly during the winter and spring period. There is infiltration and exfiltration and as a result there is excessive flow to WWTP during storm events.

Flows during the severe drought of 1999 were much reduced because of reduced surface run-off collected by the sewer system. The longer scale variability is caused by inter-basin transfer of

WWTP discharges; at times some flow is transferred to the Patapsco basin and used in industrial processes. Annual average freshwater flows (including both point source discharges and stream runoff) ranged from just over 3 m3 sec -1 to about 7 m3 sec -1. Flows between 2003 and 2005 have been among the highest since 1985 (Fig. 5-3). To place flow into context with receiving waters, the hydraulic water residence time can be estimated (time needed for freshwater inflow to replace the volume of water in the estuary). In the case of the Back River estuary the hydraulic residence time ranged between 220 (input = 3 m3 sec -1) and 93 days (input = 7 m3 sec -1). The mainstem Bay and larger tributary rivers have hydraulic residence times closer to one year (Boynton et al. 1995).

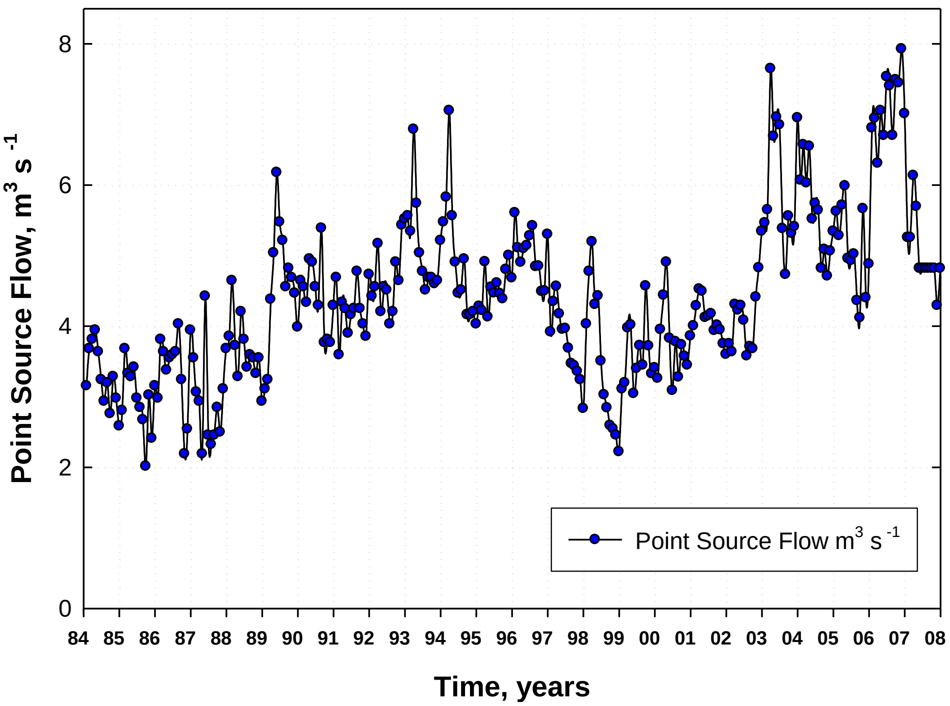


Figure 5-2. A time-series plot of the average freshwater flow to the Back River from the WWTP plant during the period from 1984–2007. Data were from the Chesapeake Bay Program data hub (point sources).

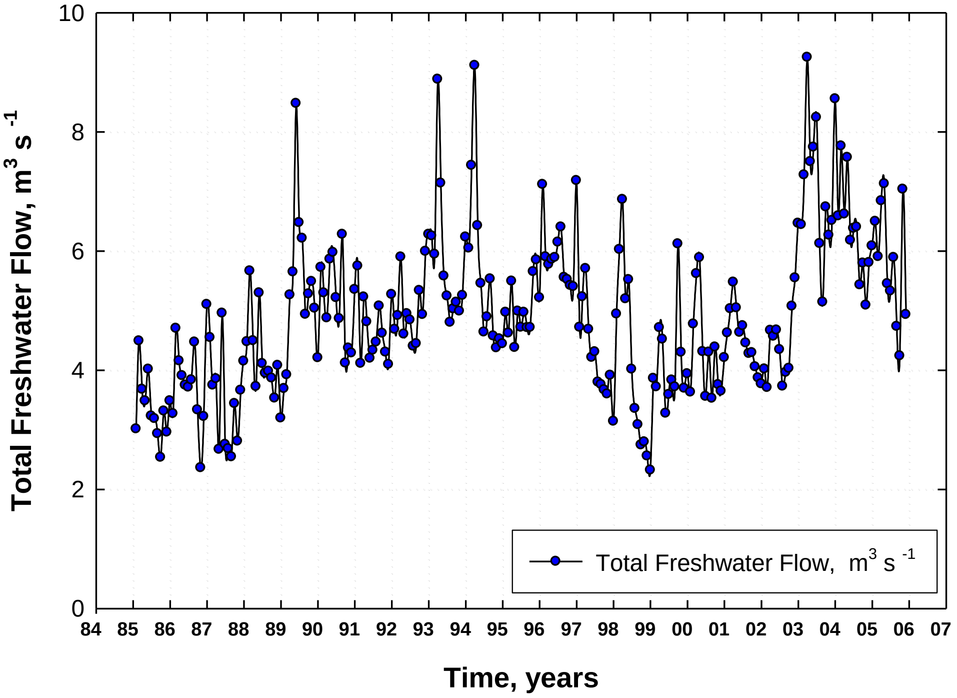


Figure 5-3. A time-series plot showing the average monthly freshwater flow to the Back River from both point and diffuse sources for the period 1985-2005. Point source data were from the sources listed in Figure 5-2; diffuse source freshwater flows were based on landscape model estimates from the Chesapeake Bay Program (Gary Shenk, pers. Comm.).

5-6.2 Point Sources of Sediment, BOD, Nitrogen and Phosphorus

The changes in WWTP discharges during the late 1980s and early 1990s were large and occurred during a relatively short period of time. For example, TSS discharge was reduced by about a factor of 15 and BOD by a similar amount (Figs. 5-4 and 5-5). Discharge of ammonium decreased as the plant began more effective nitrification while nitrate discharge increased and then later decreased when denitrification was added to the treatment process (Fig. 5-6). DIN and TN inputs during the period 1984–1996 were about twice those since 1996 (Figs. 5-7 and 5-8). In recent years (2003–2007) nitrate, DIN and TN discharges have again increased associated with increased flow from the WWTP. Both reactive (PO4) and total phosphorus (TP) exhibited dramatic reductions after 1993 (Figs. 5-9 and 5-10). For example, TP loads were about 700 kg day-1 in 1984 while after 1993 loads were reduced to less than 100 kg day-1. Further, TP loads increased only slightly during higher WWTP discharge flows during recent years. The ratio of DIN to PO4 has often been used to indicate which nutrient (N or P) would first become limiting to algal growth (Fig. 5-11). This ratio is based on the observation that algae typically use N and P in a ratio of 16:1. Thus, if the N:P ratio is less than 16:1, N would become limiting before P. Conversely, ratios greater than 16:1 would suggest a possibility of P limitation before N limitation. It must be remembered that if N and P supply or concentrations are high, neither would be limiting, regardless of their relative abundance. Between 1984 and 1992 the N: P ratio was generally low but only on a few occasions approached 16 or less. Following P removal at the plant the N:P ratio increased to over 200 and has remained at high to very high values.

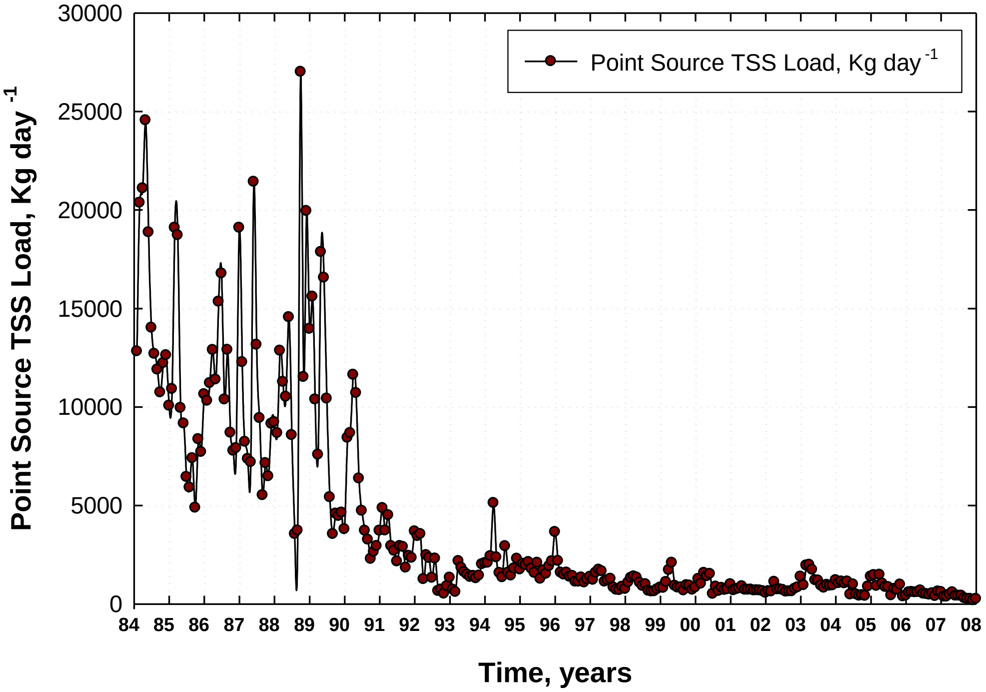


Figure 5-4. A time-series plot of the average (month) total suspended solids loads to the Back River from the Back River WWTP for the period 1984–2007. Data were from the Chesapeake Bay Program data hub (point sources).

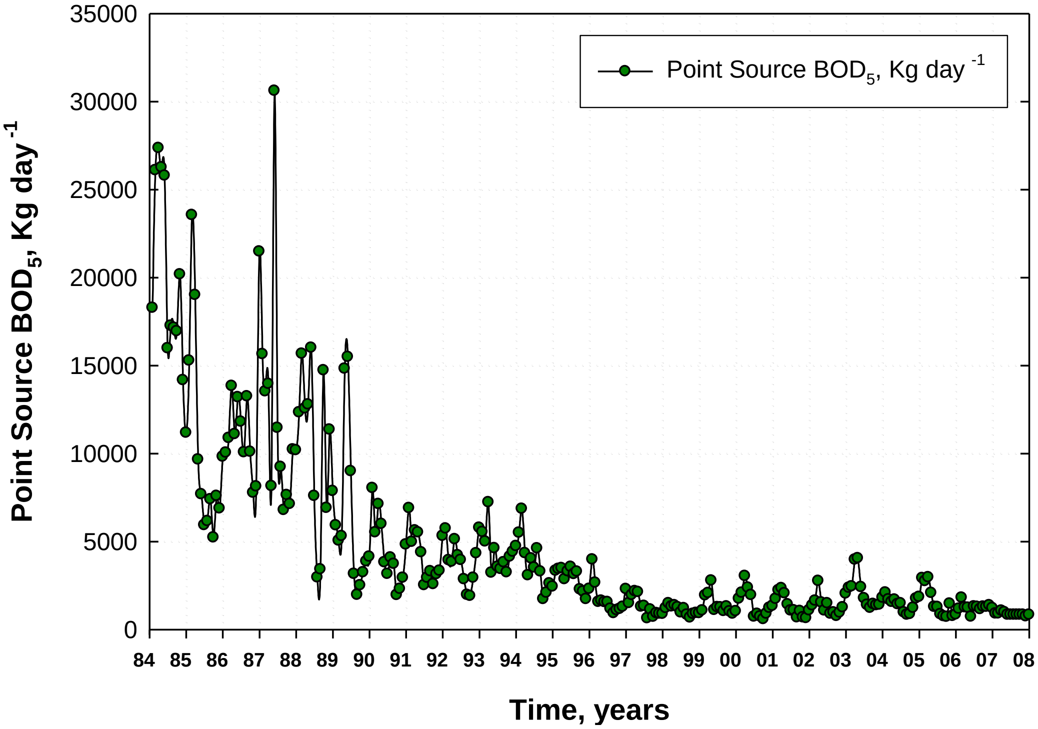


Figure 5-5. A time-series plot of the average (month) BOD$_5$ day$_y$ loads to the Back River from the Back River WWTP for the period 1984–2007. Data were from the Chesapeake Bay Program data hub (point sources).

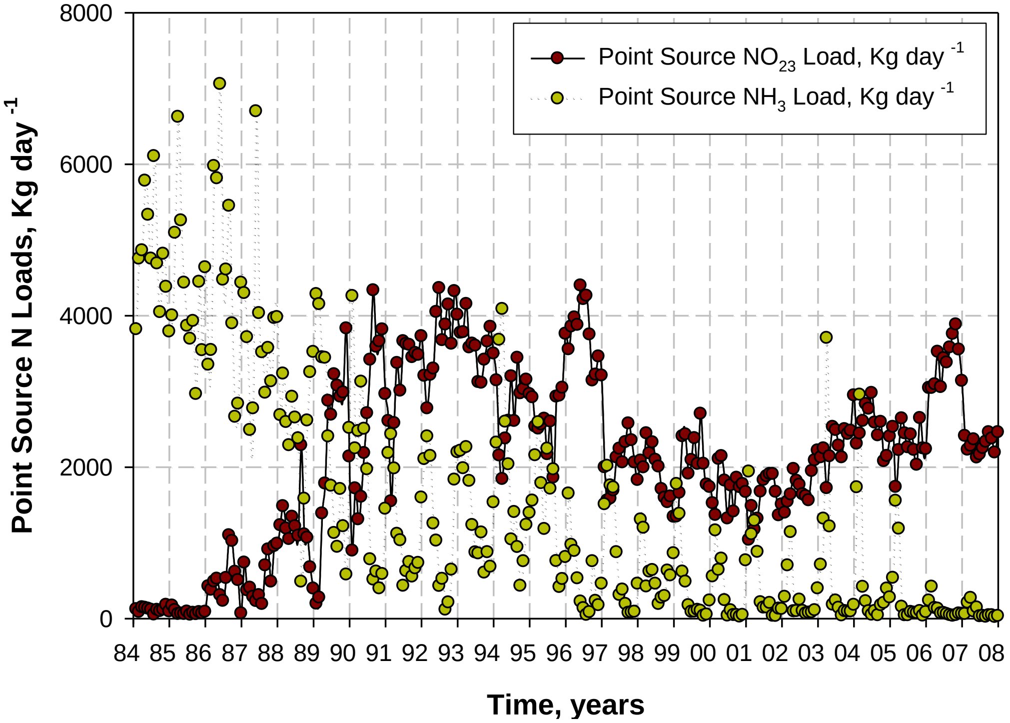


Figure 5-6. A time-series plot of the average (month) nitrate and ammonium loads to the Back River from the Back River WWTP for the period 1984–2007. Data were from the Chesapeake Bay Program data hub (point sources).

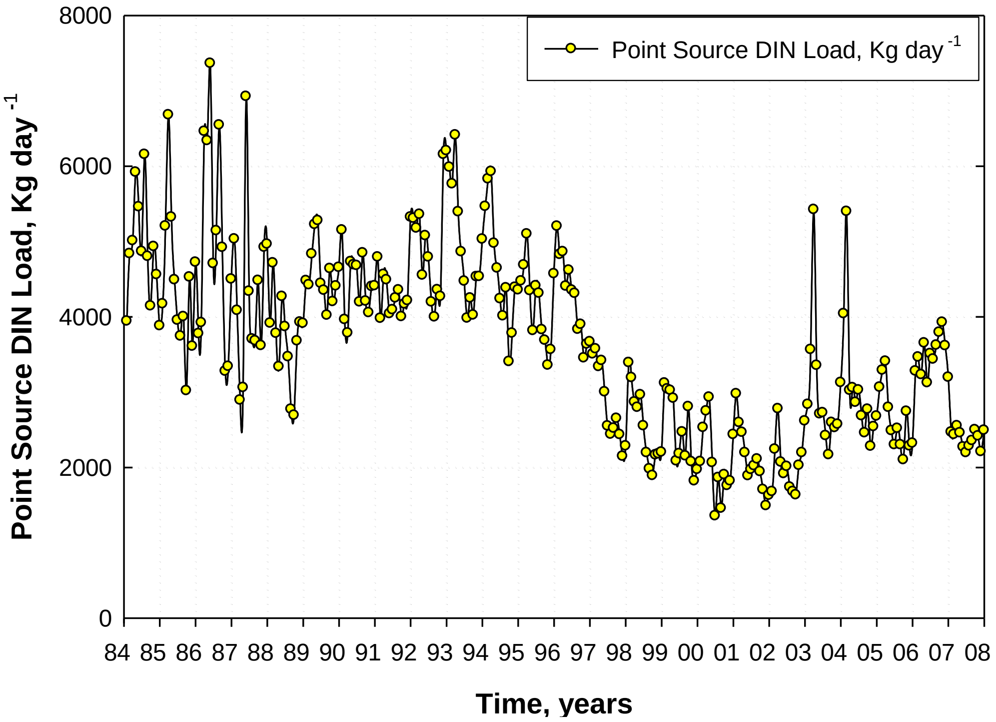


Figure 5-7. A time-series plot of the average (month) dissolved inorganic nitrogen (DIN) loads to the Back River from the Back River WWTP for the period 1984–2007. Data were from the Chesapeake Bay Program data hub (point sources).

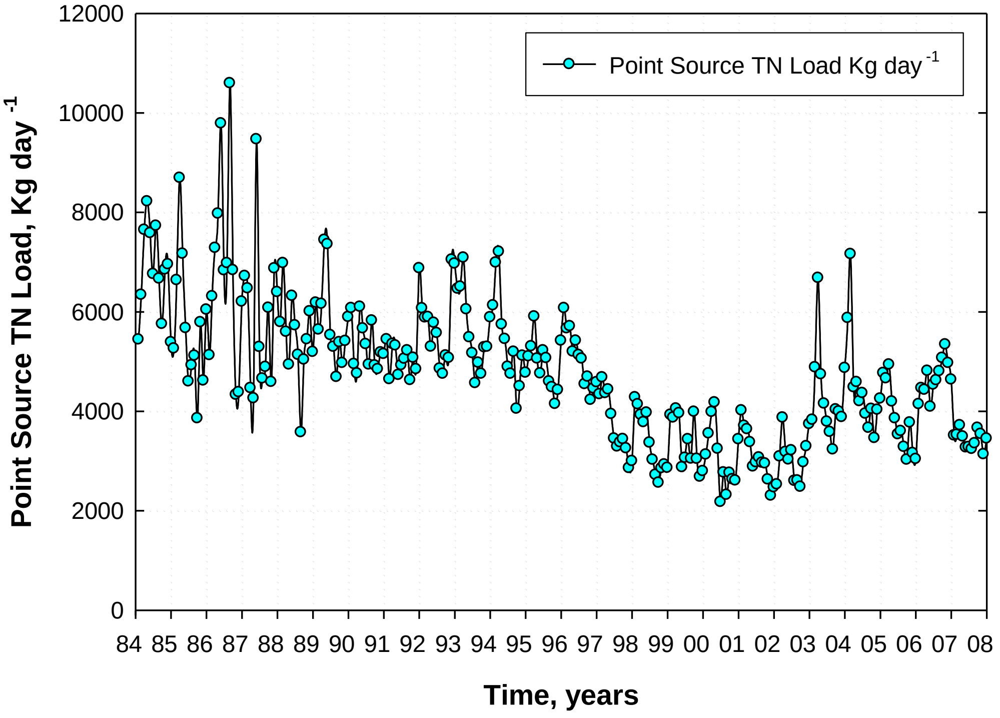


Figure 5-8. A time-series plot of the average (month) total nitrogen (TN) loads to the Back River from the Back River WWTP for the period 1984–2007. Data were from the Chesapeake Bay Program data hub (point sources).

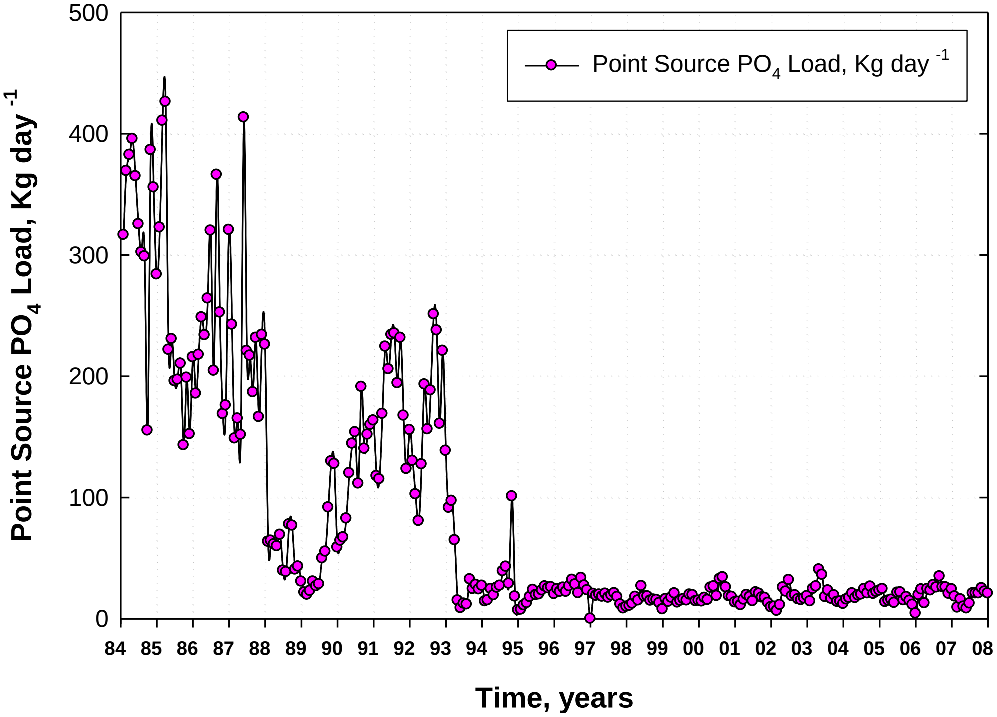


Figure 5-9. A time-series plot of the average (month) dissolved inorganic phosphorus (PO_4) loads to the Back River from the Back River WWTP for the period 1984–2007. Data were from the Chesapeake Bay Program data hub (point sources).

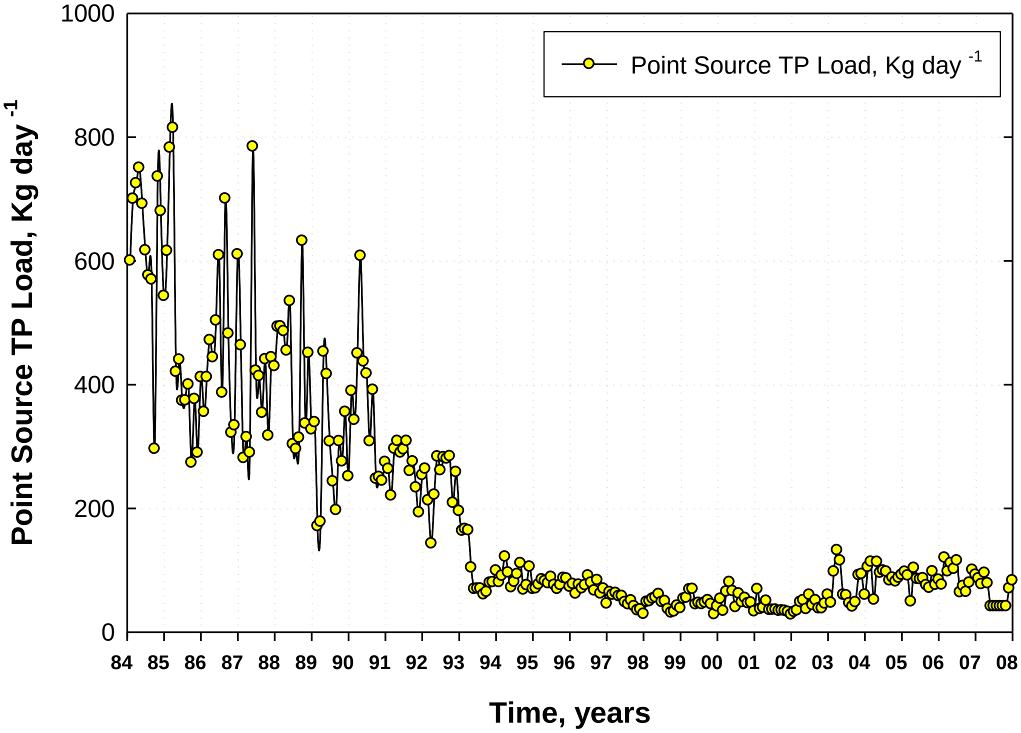


Figure 5-10. A time-series plot of the average (month) total phosphorus (TP) loads to the Back River from the Back River WWTP for the period 1984–2007. Data were from the Chesapeake Bay Program data hub (point sources).

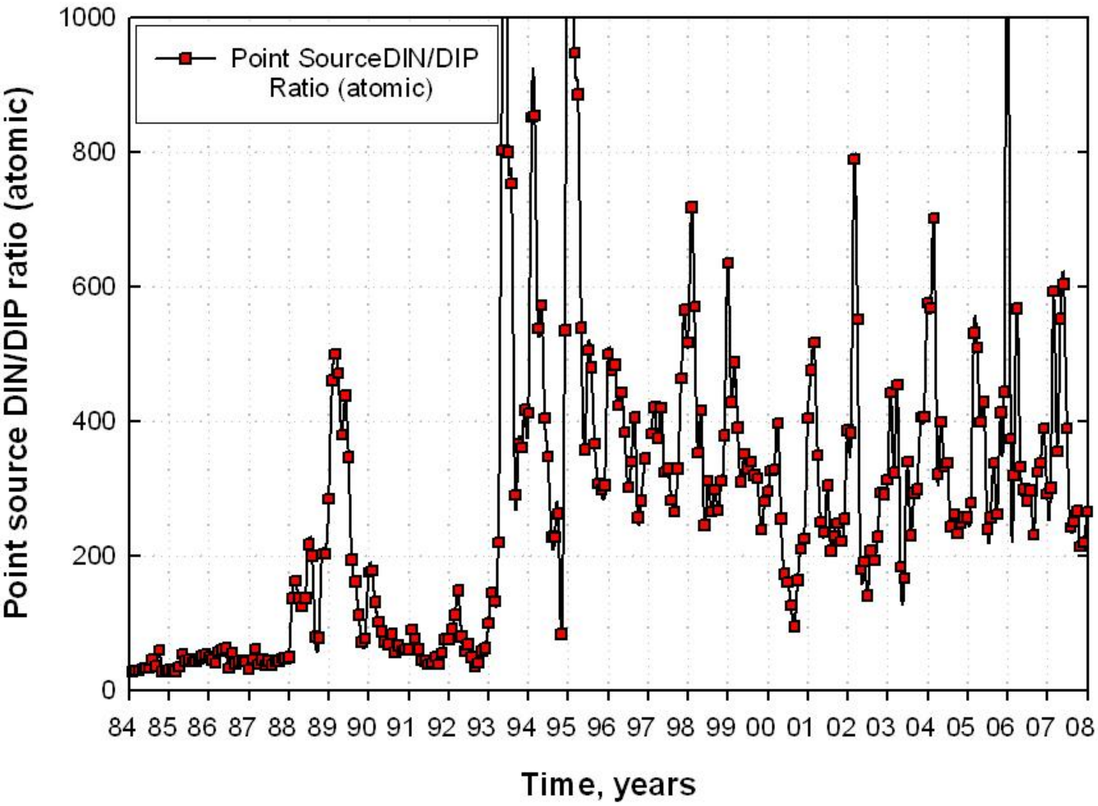


Figure 5-11. A time-series plot of the average (month) point source DIN:DIPload ratio (atomic basis) from the Back River WWTP for the period 1984–2007. Data were from the Chesapeake Bay Program data hub (point sources).

5-6.3 Diffuse Sources of Nitrogen and Phosphorus

Diffuse sources of N and P coming from the Back River watershed were based on landscape model estimates and included the time period 1985–2005. Estimates were not available for earlier or more recent years. Diffuse source DIN loads (monthly) ranged from about 900 to 4500 kg N day-1, considerably smaller than the point source load during the earlier period (1985–1996) and about the same as the point source load during the more recent period (Fig. 5-12). Current TN diffuse loads are about the same as point source TN loads in recent years (Fig. 5-13). One clear difference between point and diffuse nutrient sources is the inter-annual pattern. Point sources are reasonable constant through the year with some exceptions associated with very wet years (e.g., 2003). In contrast, diffuse loads were typically quite low during summer and fall and much higher during winter and spring. For example, during the wet year of 2003 peak loads reached 7000 kg N day-1 during winter but were reduced to 2200 kg N day-1 during summer. Thus, a portion of the nutrient load to the Back River is relatively constant while the other portion of the load varies widely with season and among years. Diffuse loads of PO4 and TP (Fig. 5-14) also exhibited similar seasonal and inter-annual differences in patterns. In the case of TP, diffuse sources for most of the period of record were higher than point source TP inputs. Finally, while there were large to very large reductions in point source discharges, there were no readily apparent long-term trends in diffuse source nutrient inputs to the Back River.

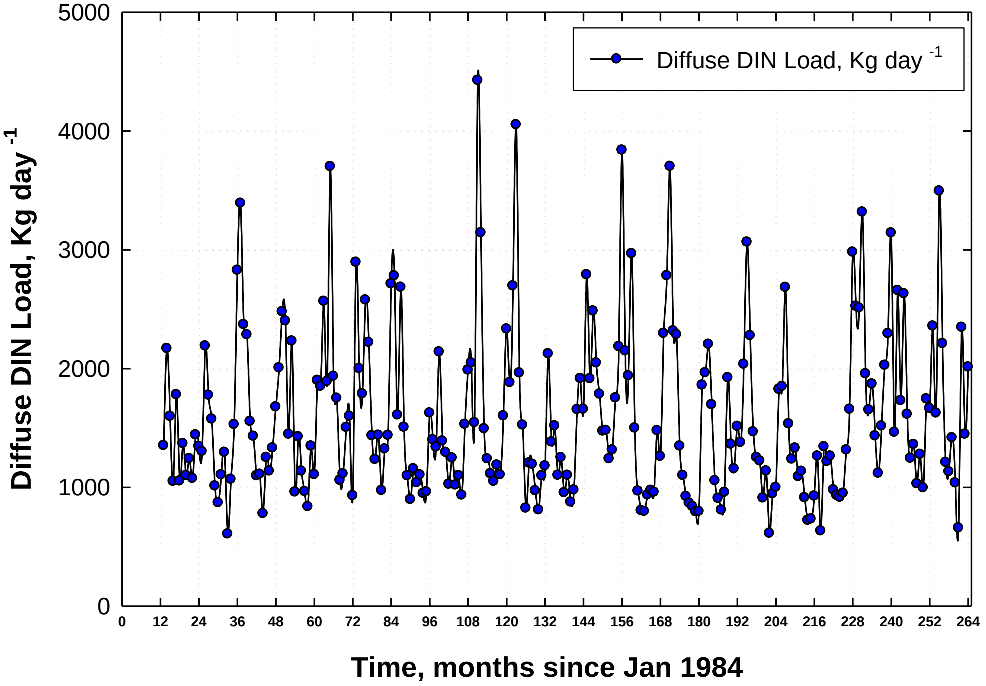


Figure 5-12. A time-series plot of the average (month) diffuse source DIN load to the Back River for the period 1985–2005. Data were from the Chesapeake Bay Program landscape model (Gary Shenk, pers. comm.).

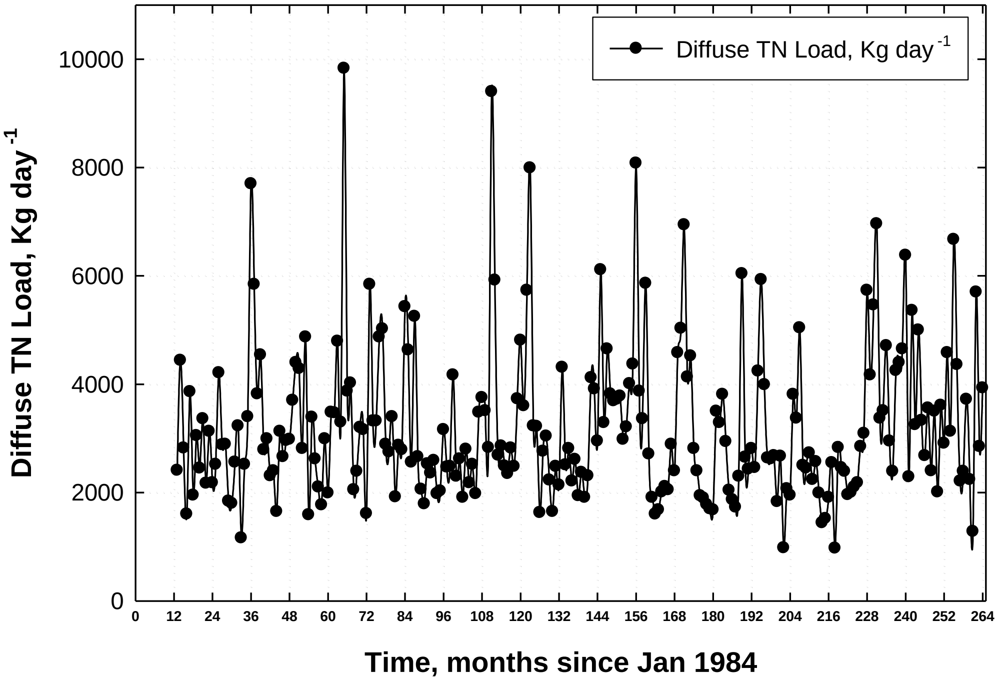


Figure 5-13. A time-series plot of the average (month) diffuse source TN load to the Back River for the period 1985–2005. Data were from the Chesapeake Bay Program landscape model (Gary Shenk, pers. comm.).

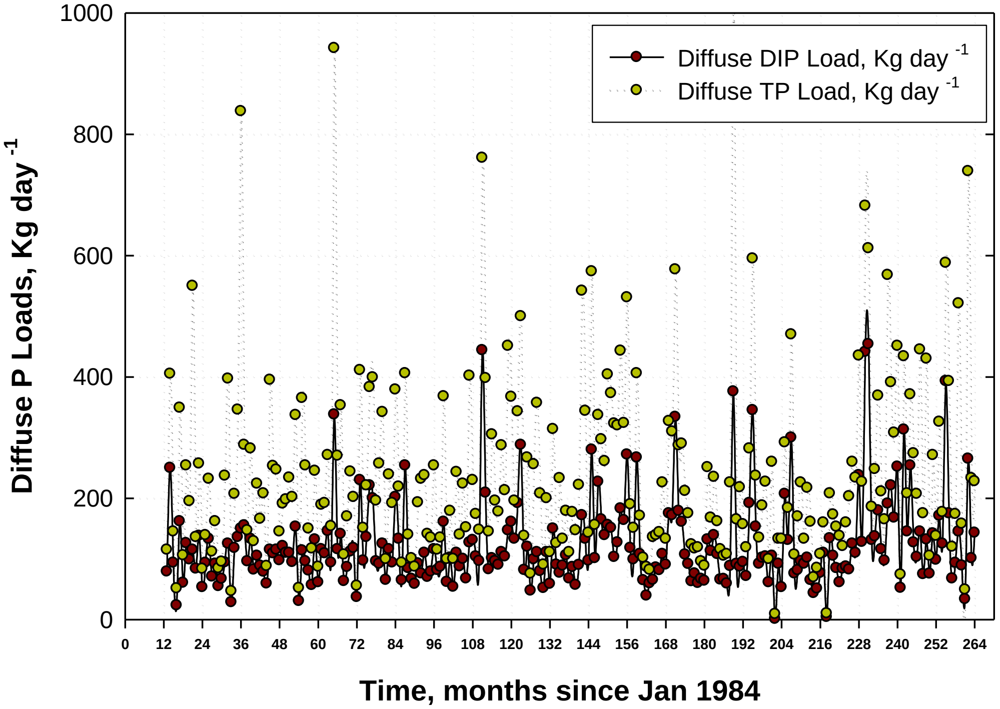


Figure 5-14. Time-series plots of the average (month) diffuse source DIP and TP loads to the Back River for the period 1985–2005. Data were from the Chesapeake Bay Program landscape model (Gary Shenk, pers. comm.).

5-6.4 Comparison of Back River Nutrient Loads with Other Estuaries

Back River point and diffuse nutrient loads were combined and averaged to annual rates to make direct comparisons with other estuarine systems. Combined point and diffuse TN loads on an annual average basis indicated sustained high load rates from 1985–1993 (~ 9000 kg N day-1) and then a generally progressive decrease in loads through 2002 (~6000 kg day-1). TN loads again increased in 2003–2005, likely in response to wet weather (2003) and higher WWTP discharges (Fig. 5-15). Annual average TP loads exhibited much larger declines beginning in 1986 and continuing through 2002. Loads decreased from about 700 kg P day-1 to about 200 kg P day-1 in 2002. TP loads again increased during 2003–2005 in response to wet weather conditions (Fig. 5-16).

Of the 35 estuarine systems shown in Figure 5-17, both N and P loads to the Back River were especially high. In the Back River case, multiple years are indicated and there is a substantial decrease in TP loads and a more modest decrease in TN loads. Multi-year load data are also shown for the Patuxent River estuary for the sake of comparison. Nutrient load reductions from point sources were also made in the Patuxent but most of the inter-annual variations in load were caused by inter-annual variation in local weather conditions and the important effect this has on diffuse loads (i.e., wet and dry years; Boynton et al. 2008). To place N and P loads to the Back River is another perspective, we can compare this system to others that have seagrass communities present. Lattimer and Rego (2010) recently reported that southern New England estuaries with healthy seagrass communities had nitrogen loads generally less than 10 g N m-2 year-1. Closer to the Back

River, the portions of the Maryland Coastal Bays having seagrass communities also have modest N loads, generally less than 5 g N m-2 year-1 (Boynton et al. 1996).

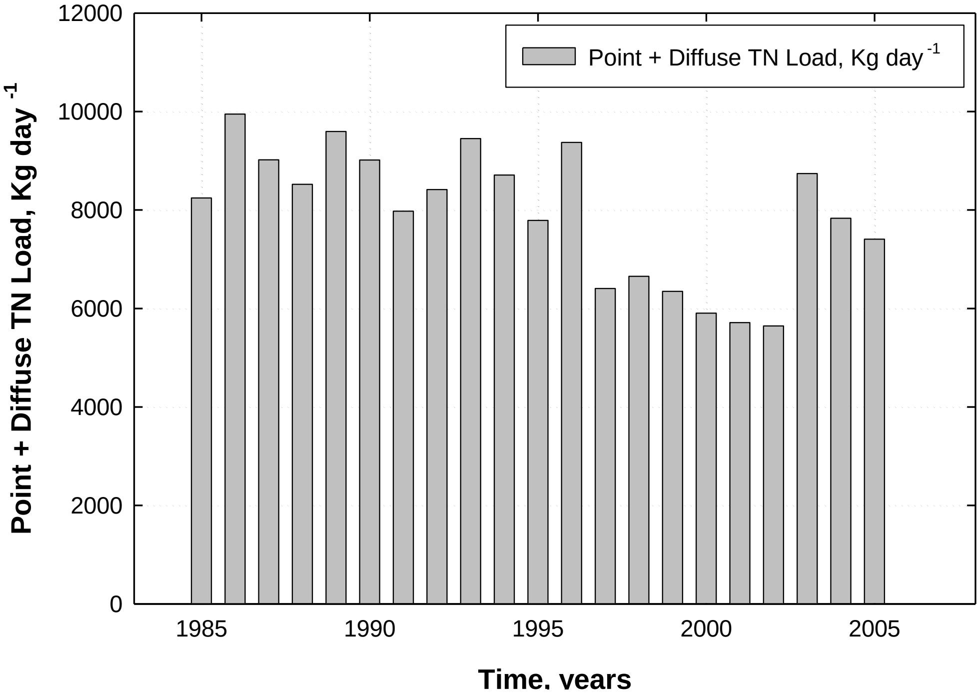


Figure 5-15. Time-series plot of the average annual point plus diffuse source TN loads to the Back River for the period 1985–2005. Data were from the Chesapeake Bay Program landscape model (Gary Shenk, pers. comm.) and from the Chesapeake Bay Program data hub (point sources).

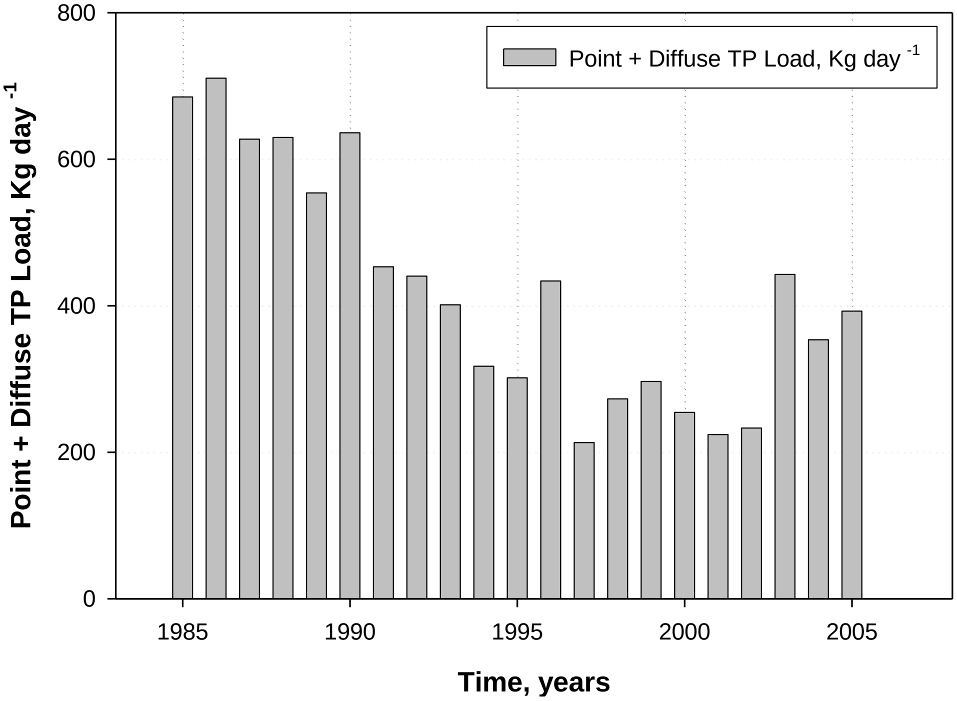


Figure 5-16. Time-series plot of the average annual point plus diffuse source TP loads to the Back River for the period 1985–2005. Data were from the Chesapeake Bay Program landscape model (Gary Shenk, pers. comm.) and from the Chesapeake Bay Program data hub (point sources).

5-7 Water Quality Characteristics

We had access to the normal suite of water quality characteristic measured by the routine Biomonitoring program, generally on a monthly basis, since 1985. The record we considered spanned the period 1985-2007. In the sections below we briefly characterize key variables with emphasis on the possible role these may play in threshold dynamics. Finally, while surface and bottom measurements were routinely made, we have worked with surface values because there were so few differences between surface and bottom values in this vertically well-mixed system.

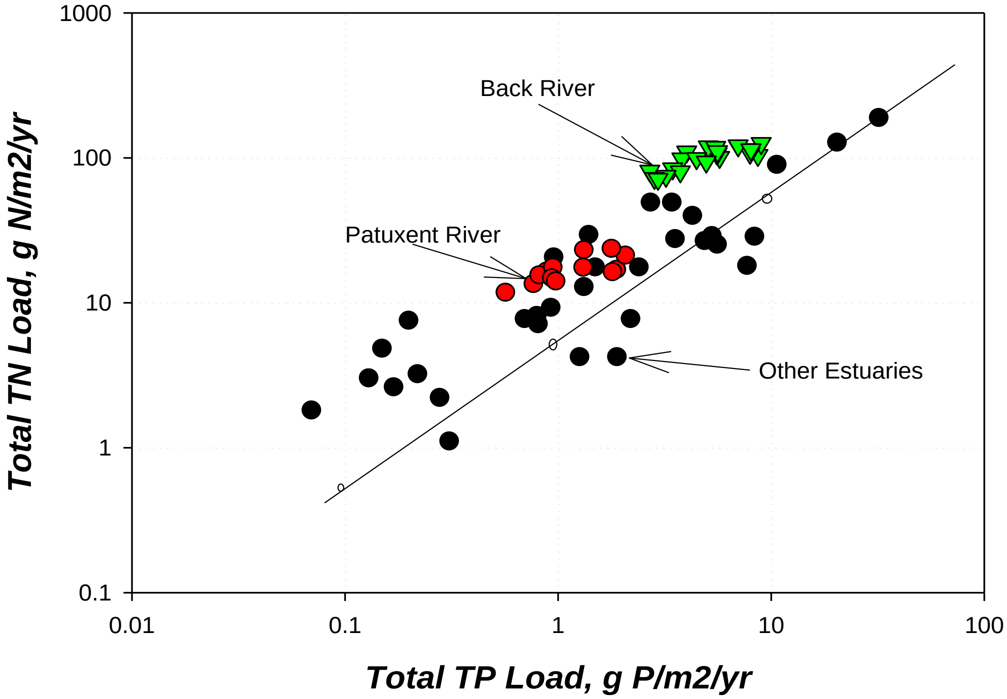


Figure 5-17. A scatter diagram showing annual TN and TP loading rates to a selection of coastal, estuarine and lagoon ecosystems. Loads to the Back River estuary are for the years 1985-2005 and to the Patuxent River estuary for the years 1985–1997. The solid diagonal line represents the Redfield ratio of TN:TP inputs (mass basis). Data sources for all estuaries except the Back River are provided in Boynton et al. (2008).

5-7.1 Temperature, Salinity and pH

As expected, water temperature exhibited a very regular pattern with highest values during late summer and lowest values during mid-winter (Fig. 5-18). However, there was considerable range in maximum and minimum annual values. For example, winter low temperature was typically about 4 ºC but reached 1 ºC during 2002. Highest temperatures were generally between 27–28 ºC but reached 31 ºC during 1988. A CART analysis (presented in detail later) found temperature to be the strongest “splitting” variable separating low from high chlorophyll-a concentration during the full period of record.

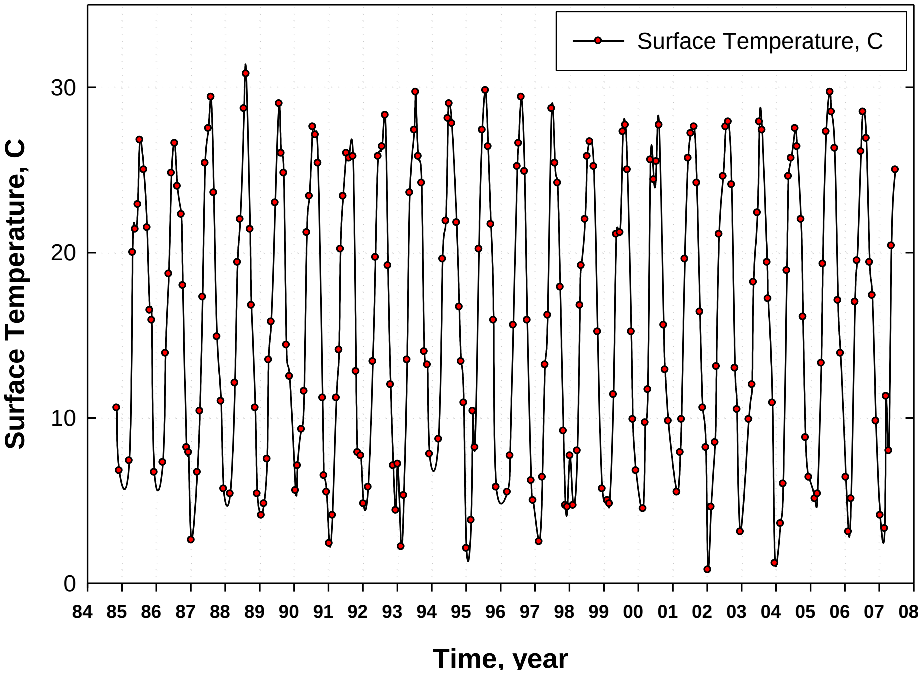


Figure 5-18. Time-series plot of the average monthly surface water temperature (°C) from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

Salinity in the Back River was typical of an oligohaline site in the upper Chesapeake Bay with values ranging between zero during wet periods and 8 following a severe drought during the summer of 2002 (Fig. 5- 19). There is considerable complexity in average monthly salinity in this tributary because of the influence of three relatively important freshwater sources (Chesapeake Bay, sewage treatment plant discharge and watershed runoff). Later we used salinity and freshwater flow from these sources to estimate net exchanges of water and nutrients between the Back River and Chesapeake Bay. Examination of surface and bottom water salinity values revealed very small differences on almost all sampling occasions, indicating a well mixed water column.

There were several very distinct patterns in pH observed in the record. First, pH was low (~ 7.5) early in the record (1985-1988), increased dramatically during the 1990s, decreased during the early 2000s and then began increasing again at the end of the record (Fig. 5-20). Maximum values during the late 1990s were close to 10 and often exceeded 9. Highest values were almost always associated with high chlorophyll-a concentration during summer periods. The long-term pH pattern closely resembled the long-term pattern of both freshwater flow and DIN loading. At pH values in excess of 9, phosphorus is known to be released from iron-rich sediments and thus become available to the phytoplankton community. There is no time-series record of sediment P exchanges from the Back River but there are some measurements and these do indicate high rates of sediment P release during summer periods (Boynton et al. 1998).

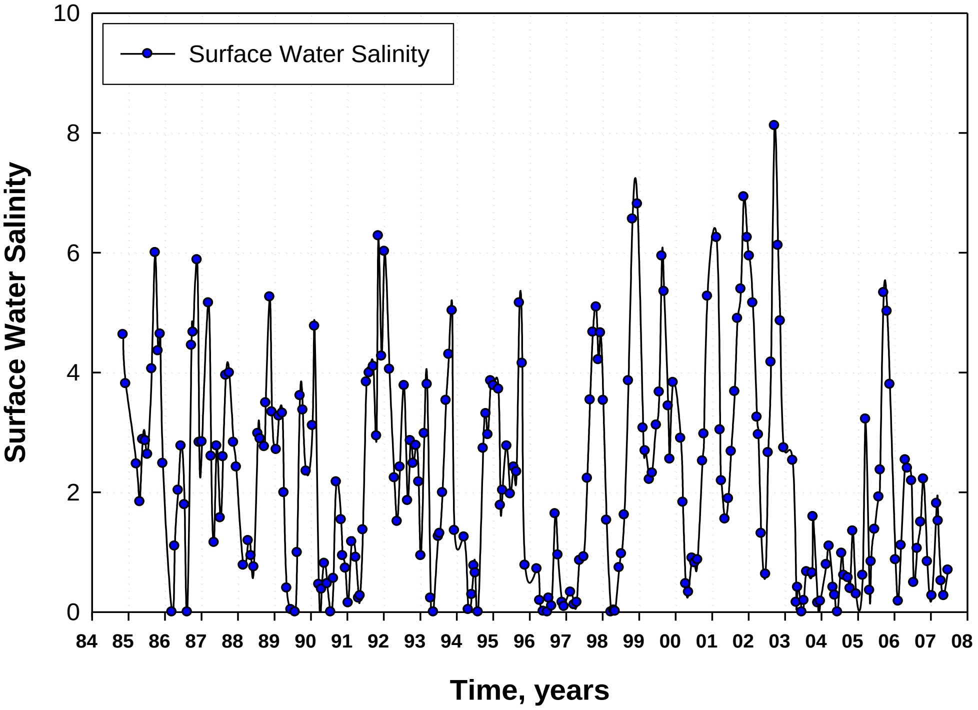


Figure 5-19. Time-series plot of the average monthly surface water salinity from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

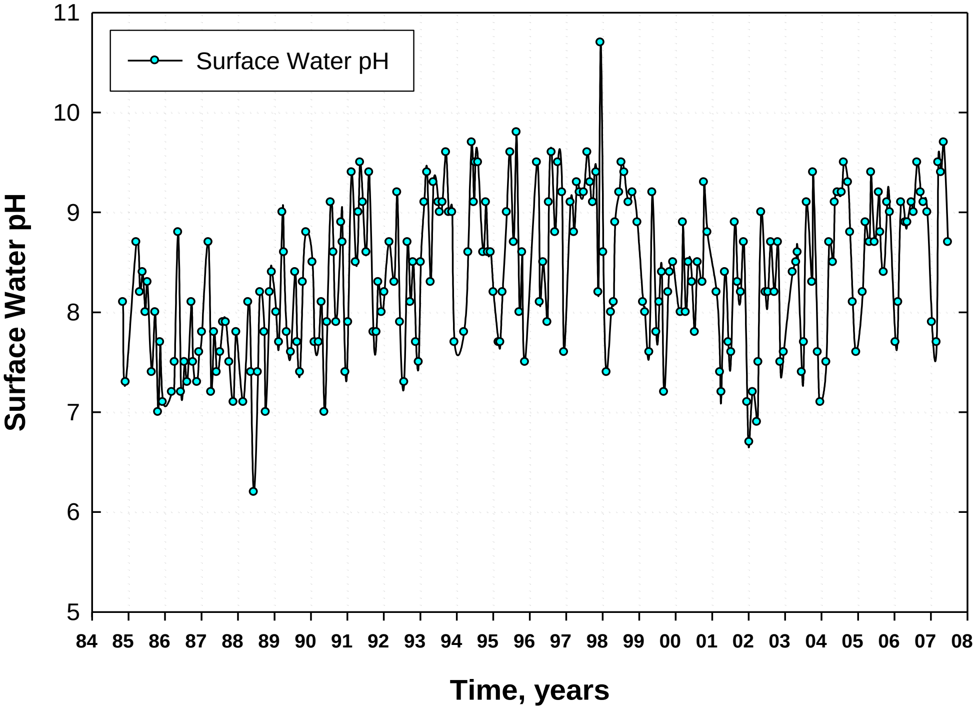


Figure 5-20. Time-series plot of the average monthly surface water pH from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

5-7.2 Secchi Depth and TSS Concentration

The Back River is an extremely turbid and sediment rich system. Secchi depths were generally between 0.3 and 0.4 m which equates to a 1% light depth of only 0.8 to 1.0 m, far less than the average depth of 2 m. Despite large reductions in TSS discharge from the sewage plant there do not appear to be any long-term trends in water transparency (Fig. 5-21). On the rare occasions when secchi depth exceeded 0.8 m, all were recorded during winter periods.

Surface water TSS concentration typically ranged between 15 to 60 mg l -1. High and low values were observed during all portions of the year. There were no significant long-term trends in TSS concentration, despite TSS load reductions associated with WWTP operations (Fig. 5-22).

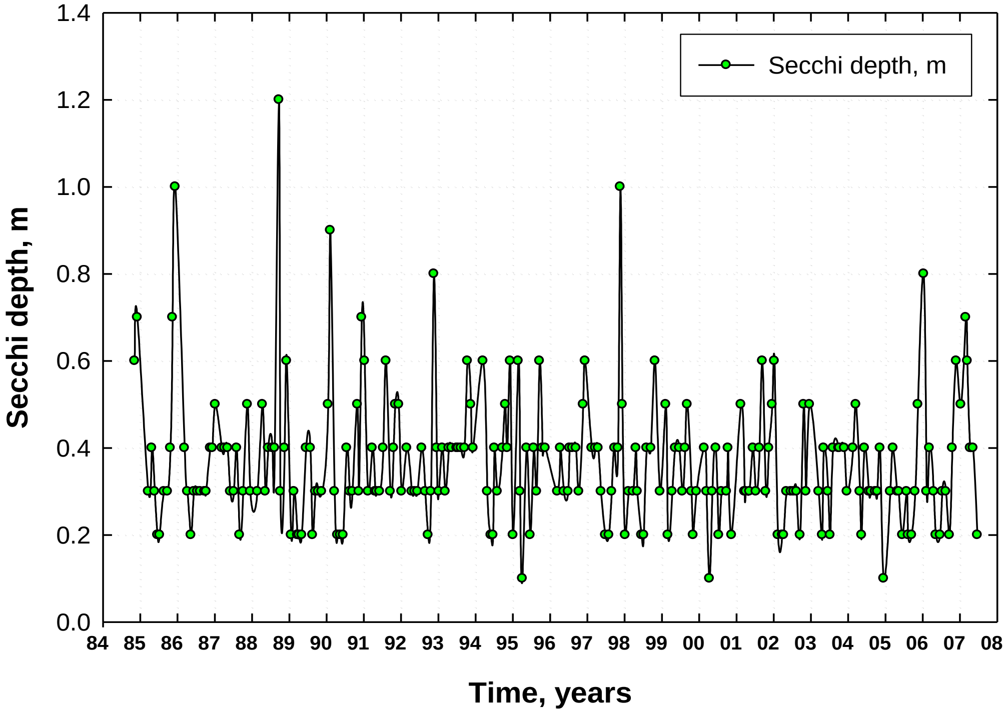


Figure 5-21. Time-series plot of the average monthly surface water secchi disk depth from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

5-7.3 Nutrient Concentration

A series of dissolved nutrient concentrations are presented in the following section. In general, N and P concentrations were high to very high, exhibited strong seasonal-scale changes in concentration and followed long-term patterns reflective of loading changes associated with WWTP operations.

Nitrate concentrations ranged from below detection to 5 mg l-1 and most values were between 0.5 and 2 mg l-1, a concentration range far in excess of limiting levels (Fig. 5-23). Virtually all high

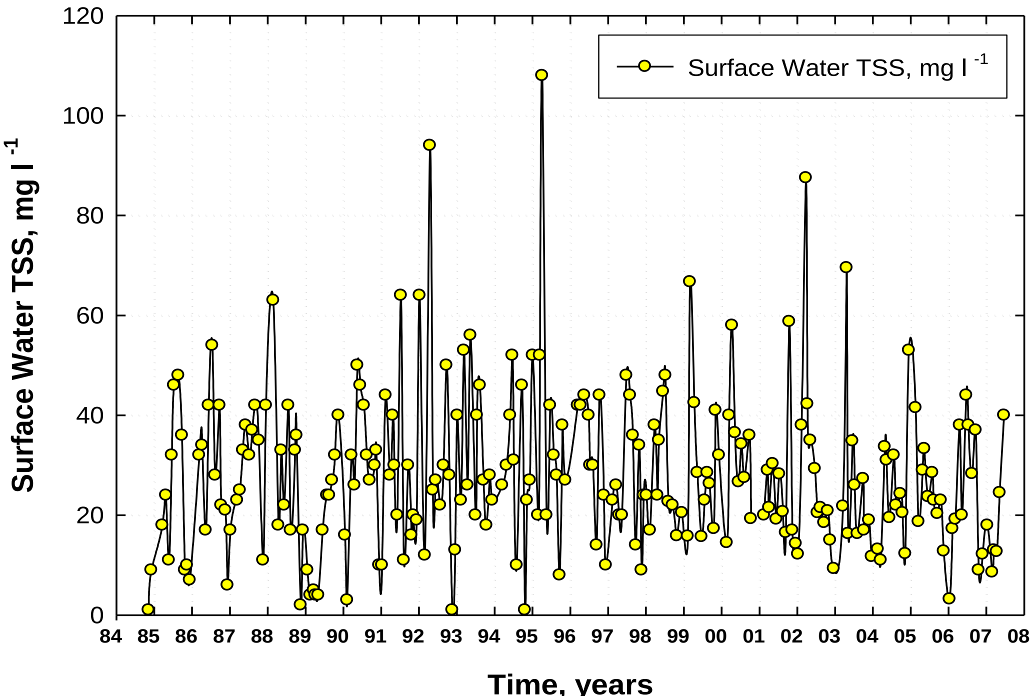


Figure 5-22. Time-series plot of the average monthly surface water TSS concentration from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

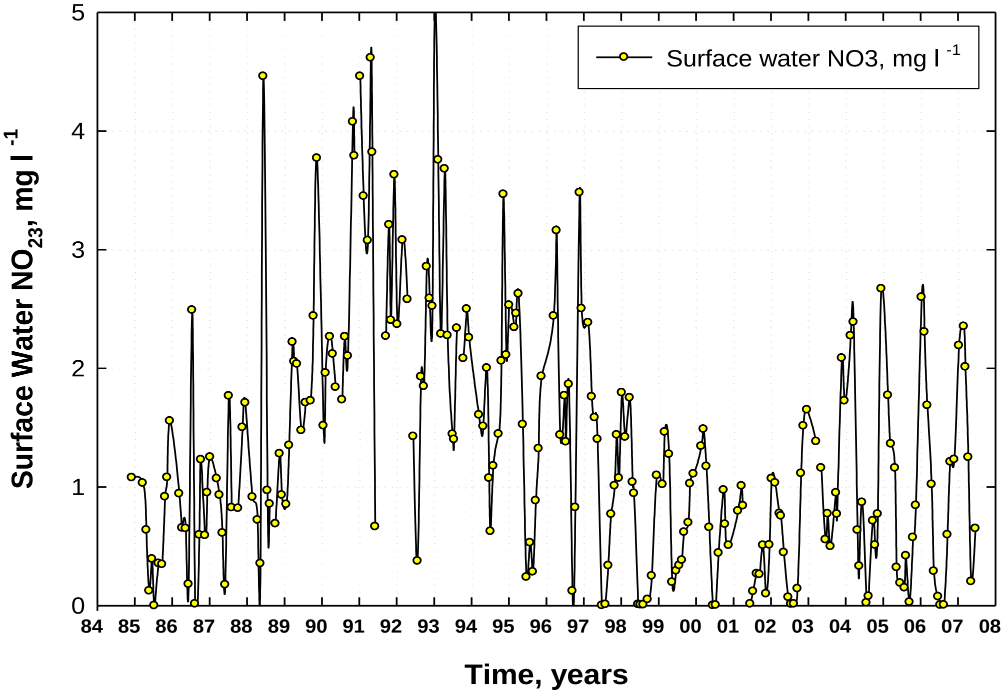


Figure 5-23. Time-series plot of the average monthly surface water NO$_{23}$ concentration from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

nitrate values occurred during winter-spring and low values during summer-fall. Water column concentrations of nitrate closely followed WWTP nitrate discharge rates although the seasonal oscillations in water column concentrations were much larger than those associated with WWTP loads. Nitrate concentrations steadily decreased beginning in 1993 through 2002 and then increased again, but not to levels recorded during the early 1990s.

Water column ammonium concentrations exhibited a remarkable change during the period of record (Fig. 5-24). Prior to 1990, concentrations were very high, often exceeding 2 mg l-1. After 1990 concentrations rarely exceeded 2 mg l-1 (only during winter) and were commonly less than 0.2 mg l-1. Except on rare occasions, ammonium concentrations were above those generally thought to limit phytoplankton growth. The long-term pattern of DIN concentration shows an increase in concentration from 2003–2007 (Fig. 5-25).

Other forms of nitrogen (DON and PON) exhibited little change during the period of record (DON; concentration range 0.5–1.5 mg l-1) or were not sampled for significant periods of time (PN).
Water column concentrations of PO4 (DIP) exhibited temporal patterns very different than those for DIN (Fig. 5-26). During the period of record DIP concentrations generally increased from 1985-1996 and then decreased through 2007. In addition, peak concentrations almost always occurred during summer, the same time that chlorophyll-a concentration reached maximum values and DIN reached minimum values. About 70% of all DIP concentrations were less than 0.03 mg l-1 (1 µM), a concentration still in excess of values thought to limit phytoplankton growth rates.

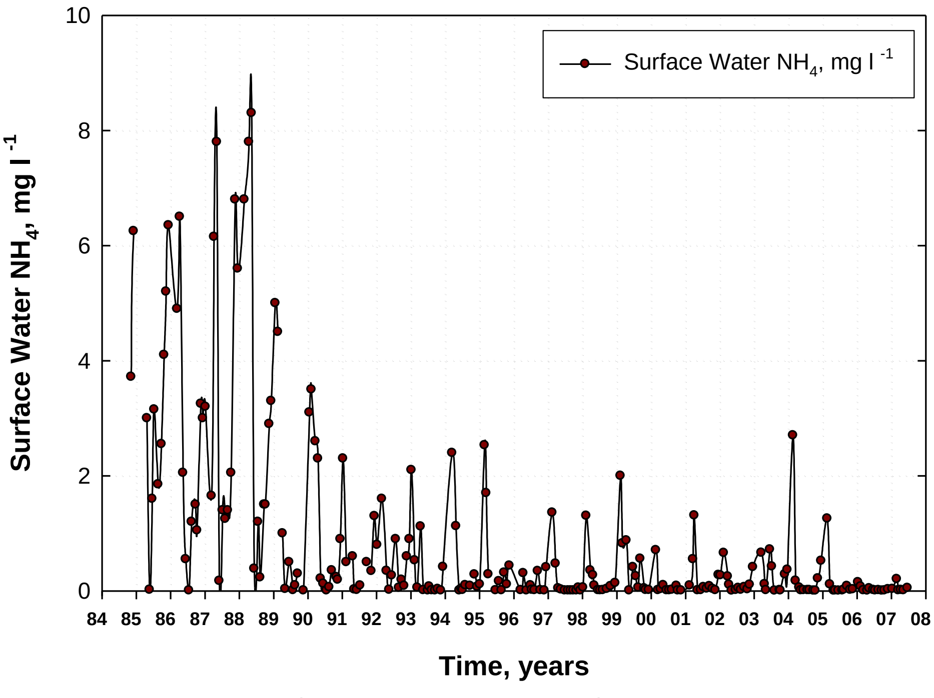


Figure 5-24. Time-series plot of the average monthly surface water NH_4 concentration from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

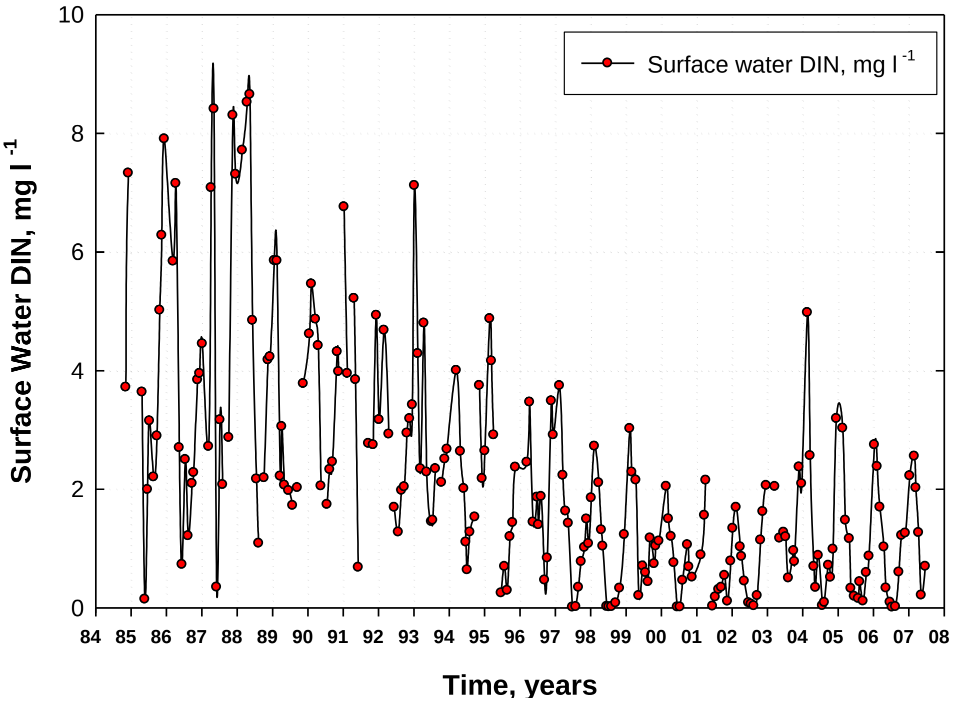


Figure 5-25. Time-series plot of the average monthly surface water dissolved inorganic nitrogen concentration (NH_4+NO_{23}) from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

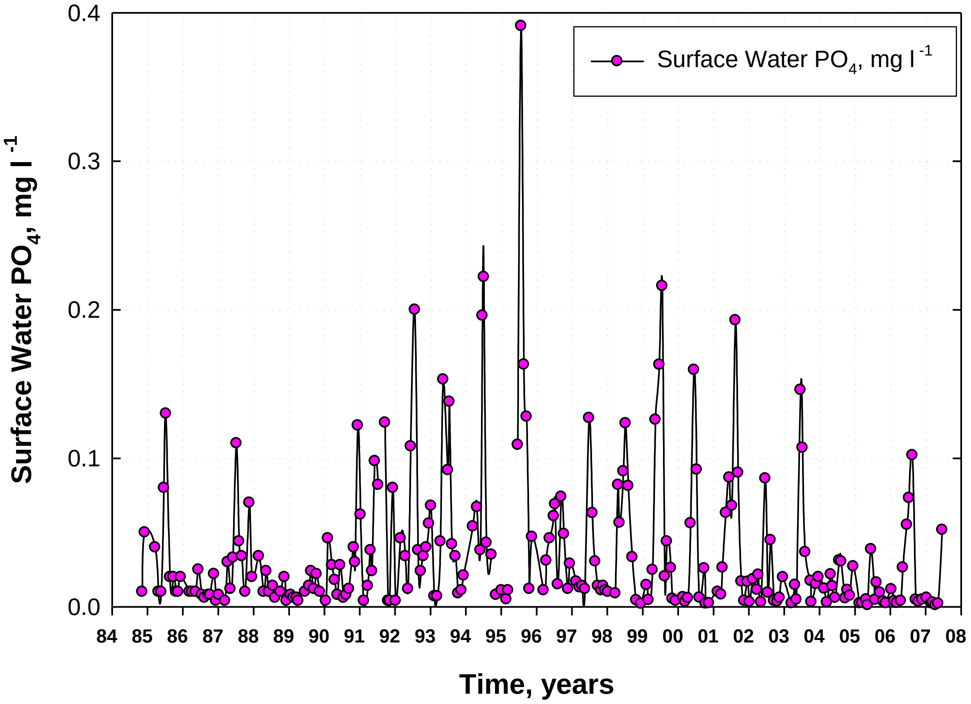


Figure 5-26. Time-series plot of the average monthly surface water dissolved inorganic phosphorus concentration (PO_4) from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

Particulate phosphorus (PP) concentrations exhibited a small but steady decrease in concentration during the period of record (Fig. 5-27) and also exhibited far less seasonal variation. Typical concentrations ranged from about 0.1 to 0.3 mg l-1.

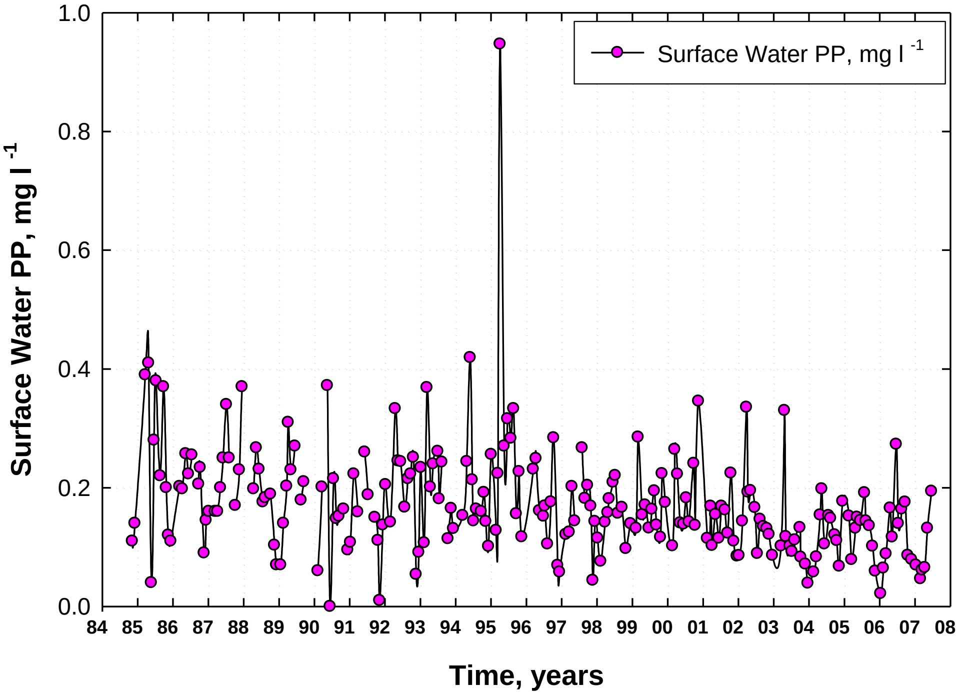


Figure 5-27. Time-series plot of the average monthly surface water particulate phosphorus concentration (PP) from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

5-8 Chlorophyll-a Concentration

Since chlorophyll-a is a target variable in this analysis, several different time-scales of data are presented (monthly, seasonal [Winter:Dec-Feb; Spring:Mar-May; Summer:Jun-Aug; Fall:Sep-Nov], and annual). Perhaps the most obvious characteristic of the monthly scale chlorophyll-a data is the variability among sampling periods (Fig 5-28). Concentrations ranged from a few micrograms per liter to over 300 µg l -1. In general, highest chlorophyll-a concentrations were measured prior to sewage plant N removal beginning by 1997, but there were a few exceptions to this pattern. For example, WWTP N loads also increased towards the end of the record and chlorophyll-a concentrations also increased. There was also a period (summer 1991 to spring 1995) when chlorophyll-a concentration was consistently lower than during preceding or subsequent years. The reason(s) for these low values remains obscure.

Seasonal chlorophyll-a concentrations also exhibited strong variability with values ranging from about 10 µg l -1 to about 225 µg l -1. In general, highest seasonal mean values were recorded prior to WWTP N removal (pre-1997) operations (Fig. 5-29). As WWTP N loads began to increase during 2003, chlorophyll-a concentrations also tended to increase. The period of relatively low chlorophyll-a concentration measured during summer 1991 to spring 1995 were also evident at the seasonal scale

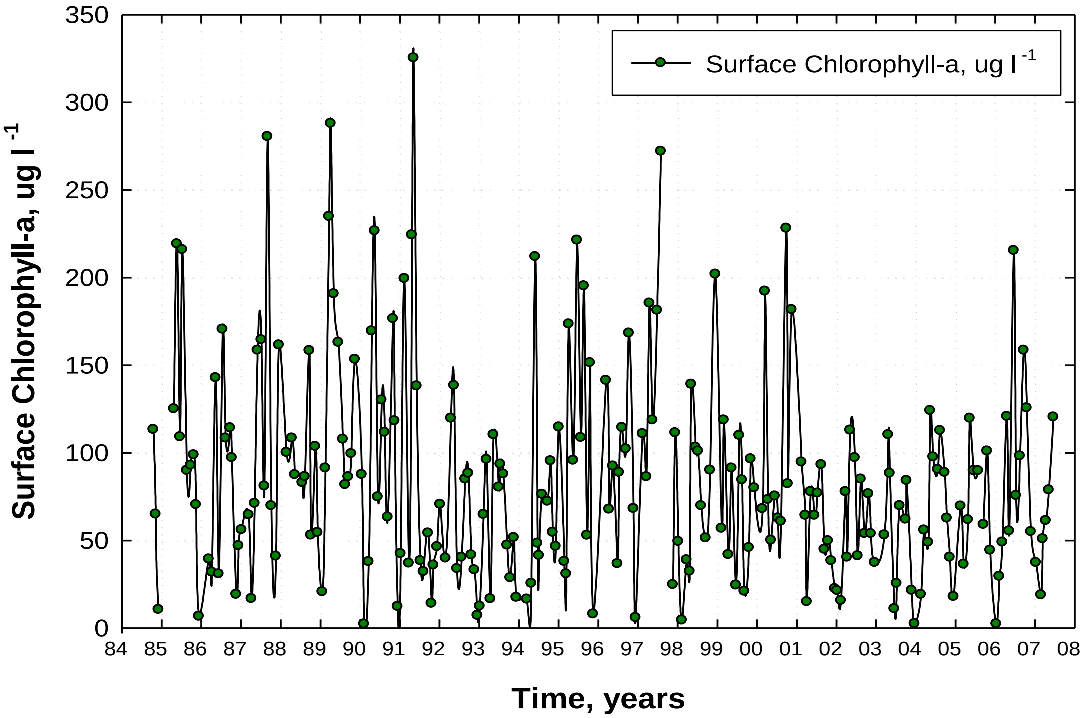


Figure 5-28. Time-series plot of the average monthly surface chlorophyll-a concentration from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

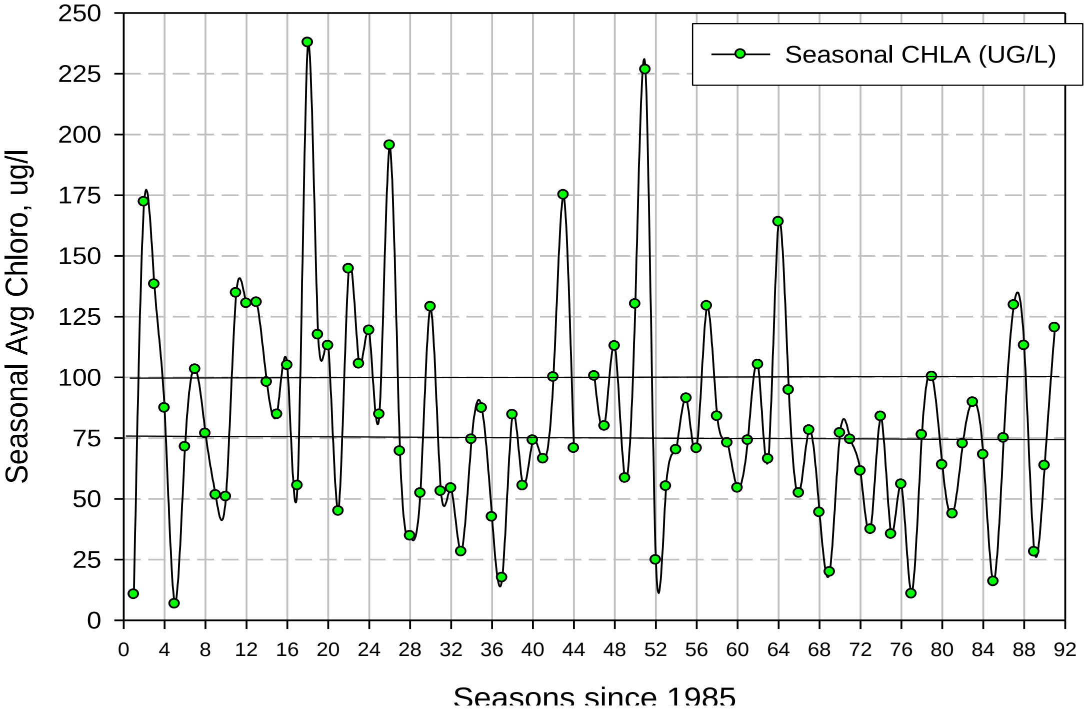


Figure 5-29. Time-series plot of the seasonal average surface chlorophyll-a concentration from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

and were more clearly the result of a lack of very high values rather than an increase in the number of very low concentrations measured during this period.

At the annual time scale chlorophyll-a concentrations ranged from about 50 to 140 µg l-1. The period 1992-1994 exhibited relatively low concentrations even though N loads from the WWTP remained quite high, as did P loads for the first portion of this time period (Fig. 5-30). Following very high annual concentrations during 1997, chlorophyll-a values decreased as did WWTP N loads through 2003 and then began increasing, as did WWTP N loads.

To better examine seasonal patterns of chlorophyll-a, a frequency histogram was constructed showing the season in which either maximum or minimum chlorophyll-a concentrations were measured (Fig. 5-31). In all but 5 years, peak chlorophyll-a concentration was observed during spring or summer (mainly summer) while minimum concentrations were measured during winter or fall (mainly winter).

Another histogram was developed showing the frequency of occurrence of chlorophyll-a concentration either above 100 µg l -1 or below 75 µg l -1 for four year periods (3 year period for final grouping). In this case there is a distinct pattern of decreasing very high concentrations (>100 µg l -1) during the period of record and a corresponding increase in the frequency of occurrence of somewhat lower concentrations (< 75 µg l -1) later in the record (Fig 5-32). Major blooms were very infrequent during the 2001-2004 period despite a wet year during 2003 (larger loads were associated with wet years).

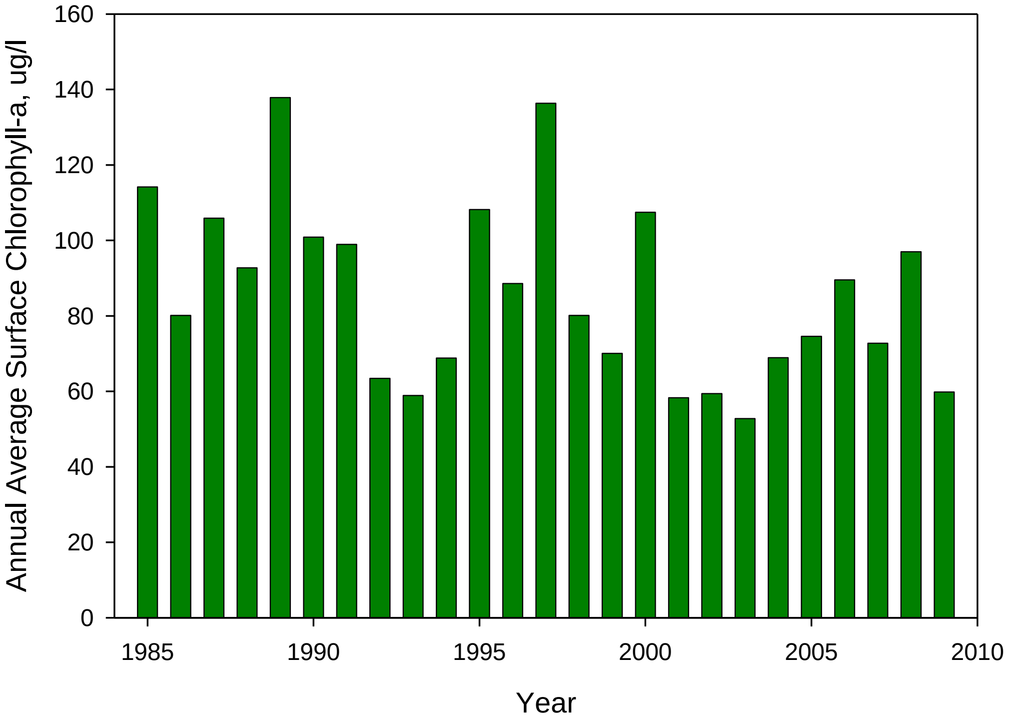


Figure 5-30. Time-series plot of the annual average surface chlorophyll-a concentration from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

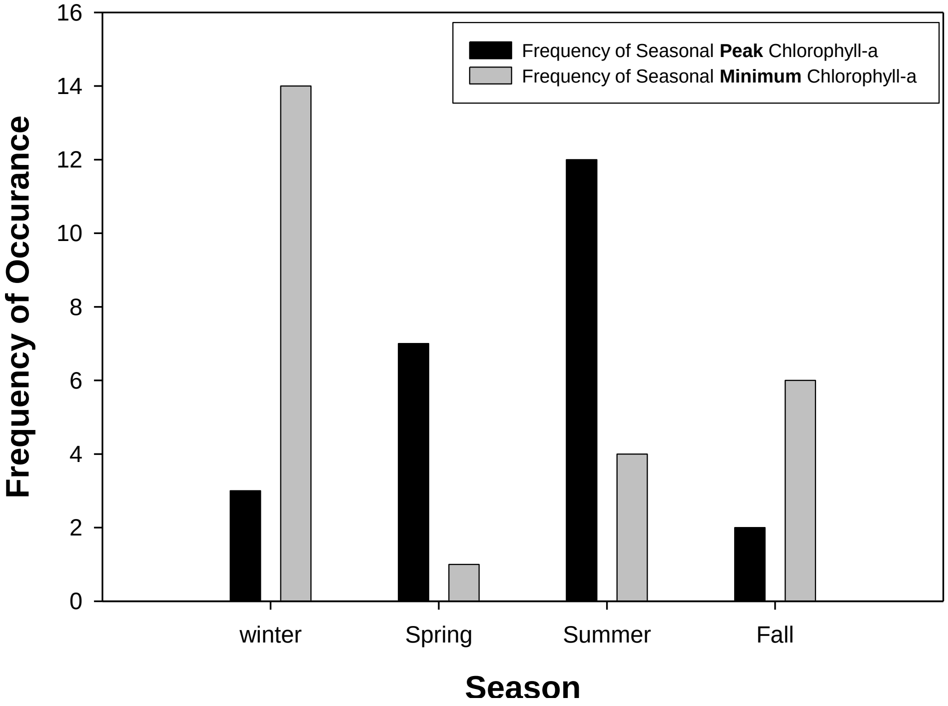


Figure 5-31. Frequency histogram of seasonal maximum and minimum surface water chlorophyll-a concentrations collected from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

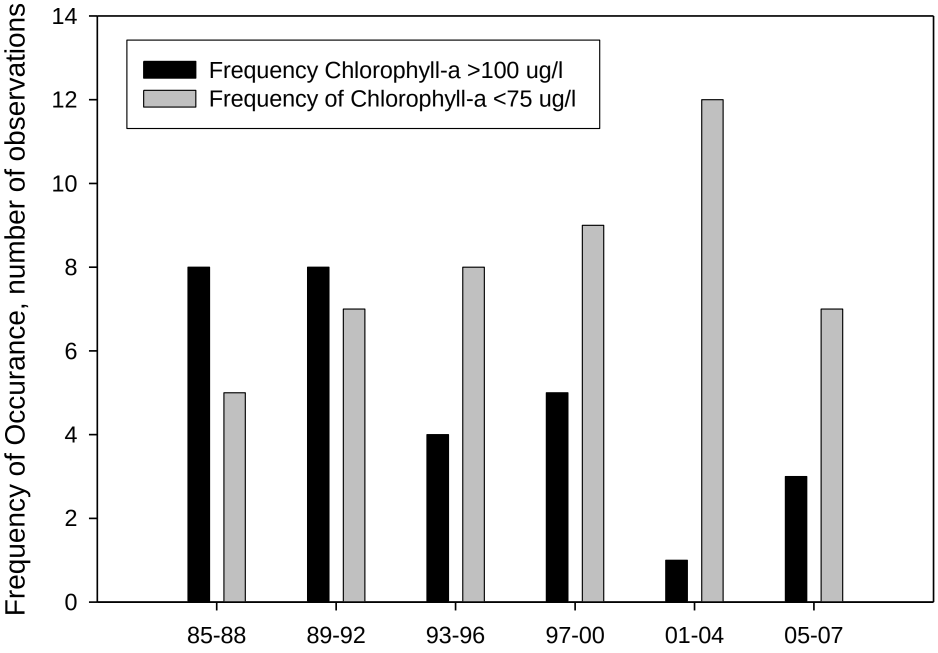


Figure 5-32. Frequency histogram of surface water chlorophyll concentrations greater than 100 µg l^{-1} and less than 75 µg l^{-1} arranged by 4-year time periods based on data collected from the Back River estuary (Biomonitoring Station WT 4.1) during the period 1985–2007. Data were from the Chesapeake Bay Program data hub (water quality).

Finally, we also located some chlorophyll-a measurements made at two locations in the estuary at higher frequencies (weekly) during summer 1997 (Boynton et al. 1998). One site was located on the north side of the Back River very close to the WWTP discharge (located on the south side of the estuary) while the other site was located nearer the junction of the Back River with upper Chesapeake Bay. The Biomonitoring site is located between these two sites (Fig 5-1). Several things are apparent in these data. First, there is often a very large difference between chlorophyll-a concentrations at the two sites sampled during 1997 (Fig 5-33). Concentrations were almost always lower at the site closer to the Bay. During 6 of 11 sampling periods during June to August the differences between sites were large (> 50 µg l-1). Differences were still apparent during September but were not as large as during the summer period. Finally, there were only 2 observations available from the Biomonitoring program during this time period (summer 1997) and both were very similar to those measured at the site closest to the Bay. These data suggest several things. First, the Biomonitoring data may better represent the "outer Back River" than the inner portion of the estuary most heavily impacted by WWTP discharges. Second, and of particular concern here, is that the Biomonitoring site may be mainly representative of the "outer Back River" but may, at times, be representative of the hyper-eutrophic inner estuary. For example, if routine sampling took place during the final stages of a strong ebb tide chlorophyll-a concentration at the Biomonitoring site might reflect conditions in the inner estuary while the last part of a strong flood tide might reflect chlorophyll-a conditions in the outer Back River or upper Chesapeake Bay. Some of the extreme variability exhibited in monthly chlorophyll-a measurements may be the result of a very strong (but very poorly described) chlorophyll-a gradient in this system.

5-9 Additional Data Sets of Interest

Because of environmental impacts caused by the large WWTP located at the head of the Back River estuary there have been a few additional data collection programs active, generally for very short periods of time, in this system. Selected portions of these data are summarized here.

Sediment particulate organic carbon (POC), nitrogen (PON), particulate phosphorus (PP) and chlorophyll-a concentrations were measured at several locations in the Back River during the mid-1990s in conjunction with measurements of sediment oxygen and nutrient exchanges (Fig. 5-34). As might be expected in a system subjected to very high nutrient loading rates, POC, PON, PP and chlorophyll-a concentration were very high. Median concentrations of POC, PON, PP and chlorophyll-a were 5.4, 0.61, 0.32 percent dry weight and 170 mg m-2, respectively. To place these values in perspective with other oligohaline sites in the Bay system we have also assembled data from the Fluxzilla sediment flux database (Boynton et al. 2008). Relative to other comparable sites, Back River POC, PON, PP and chlorophyll-a concentration were 1.6, 2.2, 3.2 and 2.6 times greater, respectively. In all but a few cases, even extreme values from other oligohaline sites were not even as large as the median concentration in the Back River. Sediment values for PP were especially high in the Back River. These data strongly suggest a very substantial internal storage of C, N, P and labile organic matter in surficial sediments (i.e., top several centimeters). Another way to consider these sediment N and P values is to convert them to units typically used in a mass balance analysis (areal units of mass per area). While we did not have all of the data needed for this conversion, we did have percent N and P content of sediments and we estimated percent water to be about 70% (averaged over a sediment column of 10 cm) and a sediment density of 2.5 g cm-3. Using these values we estimate N and P mass in the top 10 cm of the sediment column to be about 230

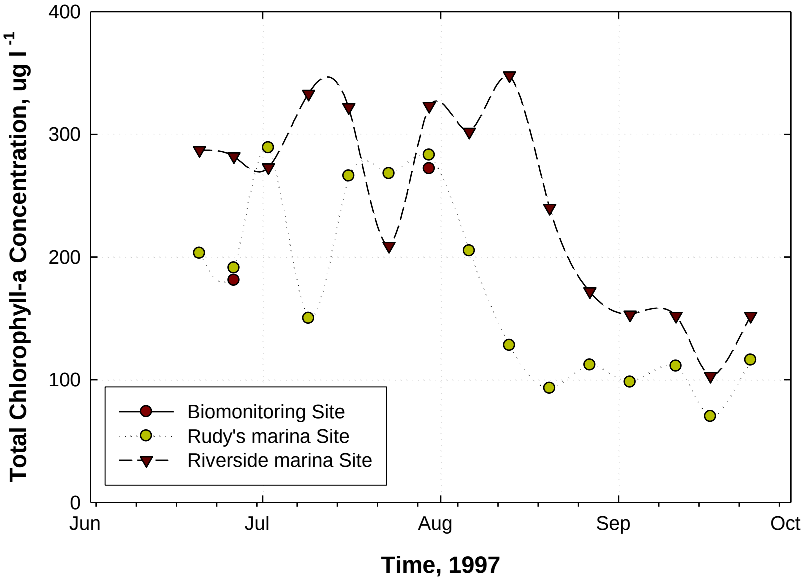


Figure 5-33. Time series plot of surface water chlorophyll-a concentrations collected from three sites in the Back River estuary during summer, 1997. Data were from the Chesapeake Bay Program data hub (water quality) and from Boynton et al. (1998).

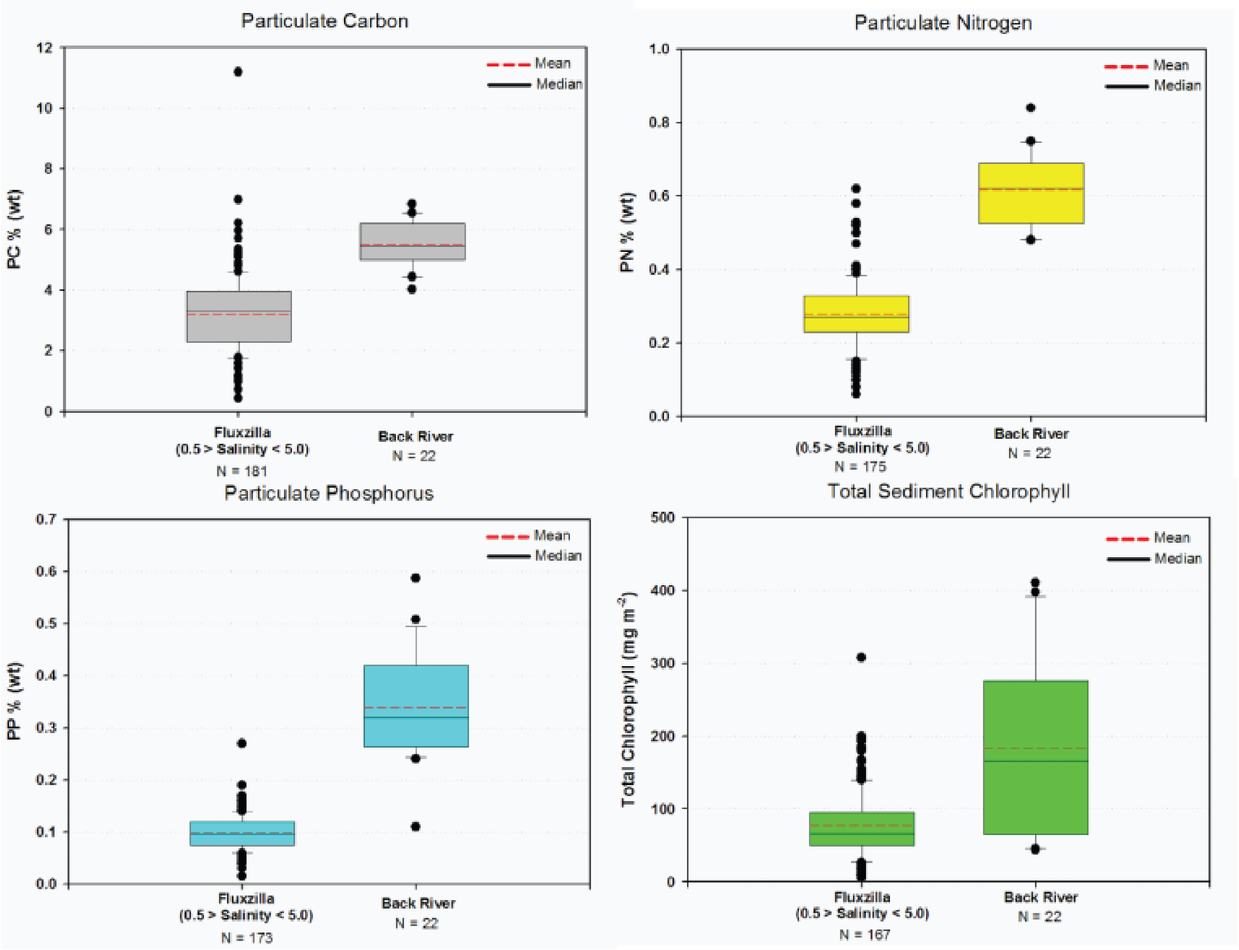


Figure 5-34. Box and whisker plots of sediment particulate carbon, nitrogen, phosphorus and total chlorophyll-a concentration (surface 1 cm). Fluxzilla data are from other areas of Chesapeake Bay collected in zones having salinity between 0-5 and temperature > 20 ºC. Back River data are from Boynton et al. (1998) and Fluxzilla data are from Boynton and Bailey (2008).

and 12 g N or P m-2, respectively. The particulate N and P stored in sediments represents more than two years of input for N and 4 to 5 years of input for P from the point source in this system. We can also place these sediment N and P storages in the context of the whole estuary and compare the storage of N and P in sediments throughout the estuary with total external loads of N and P. In this case, whole estuary N and P storage is estimated to be about 6.7 x 106 kg N and 0.35 x 106 kg P, respectively. Whole estuary loads of N and P for recent years were about 2.6 and 0.11 x 106 kg year -1 N or P, respectively. These storages exceed annual new N and P inputs by factors of 2.6 and 3.2, respectively. Both of these simple calculations indicate the real possibility of "system nutrient memory" based on these internal nutrient storages. In other words, if nutrient inputs were to decline, in-situ nutrient concentrations and other nutrient-dependent processes might not respond immediately because nutrient needs could be supplied from sediment storages.

Sediment exchanges of nitrogen (ammonium and nitrate plus nitrite) and phosphorus (dissolved inorganic phosphorus) were also measured at a few sites in the Back River during the mid-1990s, mainly during warm portions of the year (Fig. 5-35). The measurements made at sites in the Back

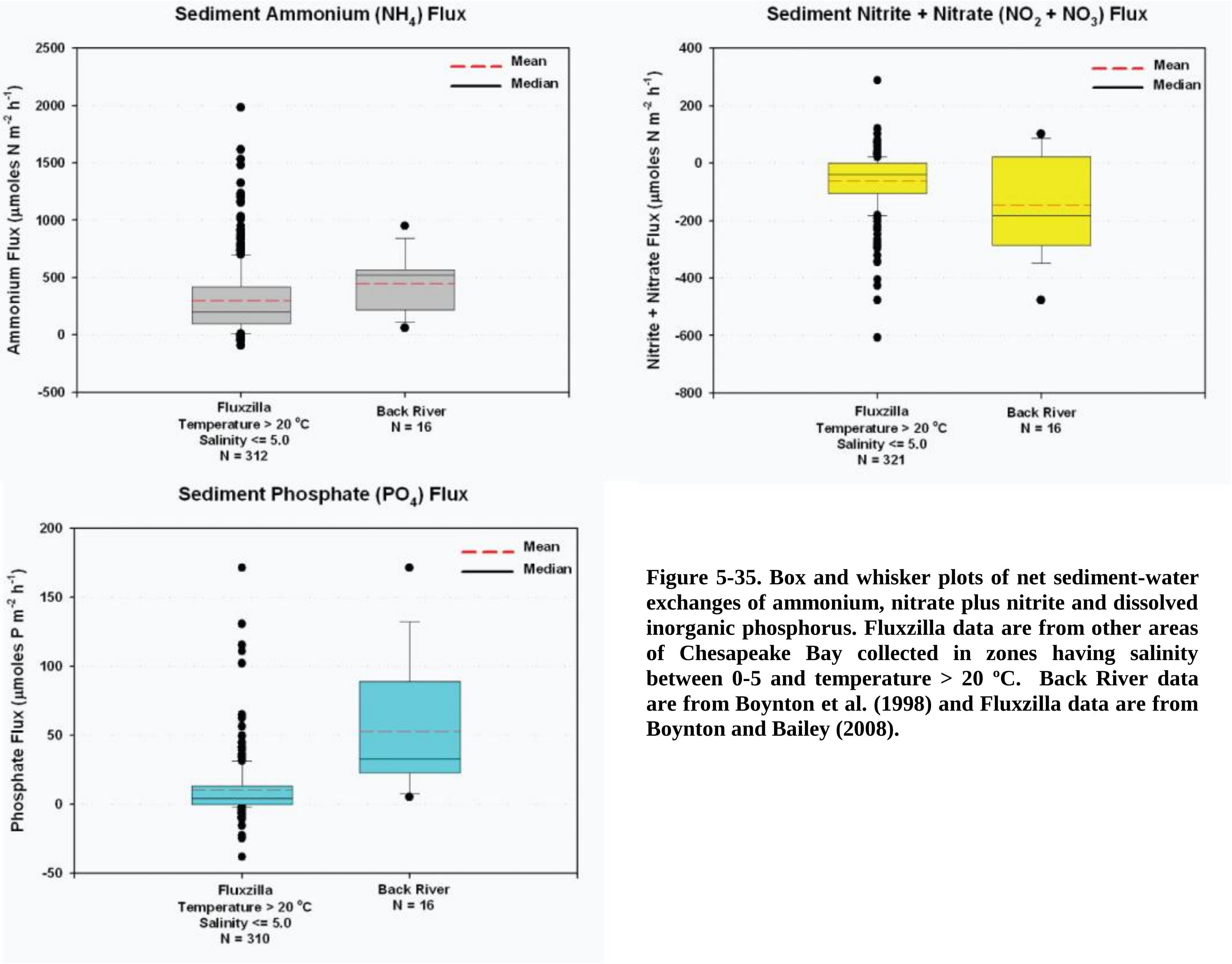


Figure 5-35. Box and whisker plots of net sediment-water exchanges of ammonium, nitrate plus nitrite and dissolved inorganic phosphorus. Fluxzilla data are from other areas of Chesapeake Bay collected in zones having salinity between 0-5 and temperature > 20 °C. Back River data are from Boynton et al. (1998) and Fluxzilla data are from Boynton and Bailey (2008).

River were also compared to measurements made in other areas of Chesapeake Bay at similar temperature and salinity. Boynton and Bailey (2008) examined many sediment flux measurements made in the Bay system and found a strong and positive relationship between flux magnitude and nutrient loading rate. Thus, given the high loading rates associated with the Back River, high sediment exchanges of N and P would be expected and that appears to be the case. Median ammonium and nitrate fluxes in the Back River were about 3 times those measured in other low salinity locations and median phosphate fluxes were even larger in the Back River than in other low salinity sites (~8X). To place the importance of these sediment releases of P, especially during summer seasons when algal biomass accumulation is typically largest, P from sediments can be compared to P from combined point and diffuse sources. During the late 1990s (time period when sediment P exchanges were measured) external inputs of TP ranged from 200-400 kg P day-1. If the median sediment P release flux is used (50 µM P m-2 hr-1) and converted to the same units as inputs, sediments supply P at a rate of about 1000 kg day-1, 2.5 to 5 times the rate of “new” P inputs from point and diffuse sources. This suggests a large internal storage of P (and likely N) that could serve to support high algal biomass for some period of time after external loads are reduced.

There were also a few summer season (June to September) measurements of community production and respiration available from two sites in the Back River (Fig. 5-36). These measurements were made by collecting high frequency dissolved oxygen, temperature and salinity data using in-situ data sondes (15 minute intervals) and applying the Odum and Hoskin (1958) open water method for computing these community rate measurements. In general, these rates are very high, consistent with very high nutrient loading rates and very large algal stocks. Rates were somewhat higher at the site closest to the WWTP discharge (Riverside Marina) than at Rudy’s marina, although it is not likely that the two sites were statistically different (see Fig. 5-1 for site locations). These community-level rates have been computed for a variety of Bay locations and these are among the highest yet recorded.

5-10 Box Model Budget for the Back River Estuary

One approach to developing understanding of water quality status of an estuary is to estimate nutrient budgets. These budgets, in their simplest form, are mass balances wherein inputs are balanced against the sum of exports, internal losses and changes in system storages. In the case of the Back River we do not have sufficient measurements to construct such a detailed evaluation as has been done for the mainstem Chesapeake Bay for organic matter (Kemp et al. 1997) and other Bay estuaries for nitrogen and phosphorus (Boynton et al. 1995 and 2008). However, a simpler mass balance can be constructed using a box model approach following the method of Hagy et al. 2000. In this case the box model consists first of a salt and water balance which is used to compute net water transfers between the Back River and Chesapeake Bay. Nutrient inputs (from both diffuse and point sources) are organized as are nutrient concentrations measured in the estuary and at the confluence of the estuary and Chesapeake Bay. Coupled with water transport computed from the box model, net transport of nutrients between the estuary and the Bay can be estimated as can the net production (or loss) of nutrients within the estuary. This approach was utilized for the Back River on an annual basis (with 1998 data) and considered the following nutrient groups: TN, DIN (all forms of dissolved inorganic nitrogen), TP and DIP (dissolved inorganic phosphorus).

Several clear results emerged from this useful synthesis based on typical monitoring data sets (Fig. 5-37). First, the inputs are very large for both N and P as we indicated earlier and a large fraction (> 50%) of both N and P inputs are in chemical forms directly available to phytoplankton. Second, a very small fraction of these inputs are exported to the Chesapeake Bay and that suggests important internal nutrient sinks (e.g., denitrification, burial and storage of PN and PP) are very important in the nutrient economy of this estuary. It is also useful to note that the net nutrient flux for both N and P on an annual basis is from the Back River to the Bay rather than in the other direction. Given that the Bay has substantial nutrient concentrations and that there is an open, tidally-driven connection between the Bay and the Back River the opposite condition could have resulted. The net fluxes of nutrients from the Back River to the Bay suggests that considerable water quality remediation could be achieved by local actions directed at reducing both point and diffuse sources located within the Back River watershed.

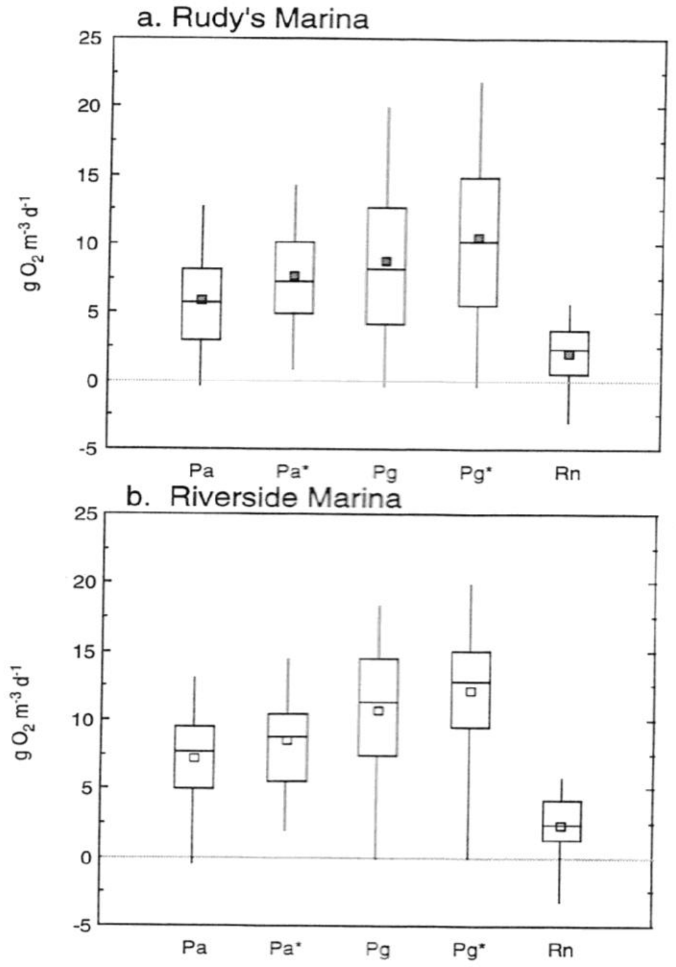


Figure 5-36. Box and whisker plots of several measures of net (Pa and Pa*) and gross (Pg and Pg*) phytoplankton production and community respiration (Rn) developed from high frequency data (15 minute intervals) collected at two sites in the Back River (see Fig V-1 for station locations) during summer (June-September) 1997. Data are from Boynton et al. (1998). The Rn values are plotted as positive numbers but actually represent dissolved oxygen consumption. Metabolism variables are defined on page 3-4 of this report. Details of the calculation of all metabolic variables are provided in Boynton et al. (1998).

There are relatively large negative net nutrient production fluxes for both N and P. These negative values indicate losses of nutrients within the system. Likely pathways of loss include denitrification and long-term burial of both PN and PP. These pathways have not been directly measured but the very high nutrient concentrations in sediments suggest an important role for burial and the very high rates of nitrate uptake by sediments suggests active denitrification. Finally, these N and P box model budgets do not balance (i.e., inputs = exports + internal losses + changes in internal storages), but they are close, especially for the biologically active components (DIN and DIP). A more refined box model budget could have been constructed if there were more than one water quality site in the Back River and if there had been a Bay monitoring site closer to the mouth of the Back River. Both of these factors might have contributed to a budget that more fully captured the inputs and fates of N and P in this system. Despite these shortcomings the message seems clear. The Back River still has exceptionally high N and P loading rates, export to the Bay is small and internal losses are very important. This system retains N and P rather than acting as an active transporter of these materials. It would appear that the main pollution issues are internal to the Back River and that solutions to these issues can be addressed by reducing local point and diffuse inputs.

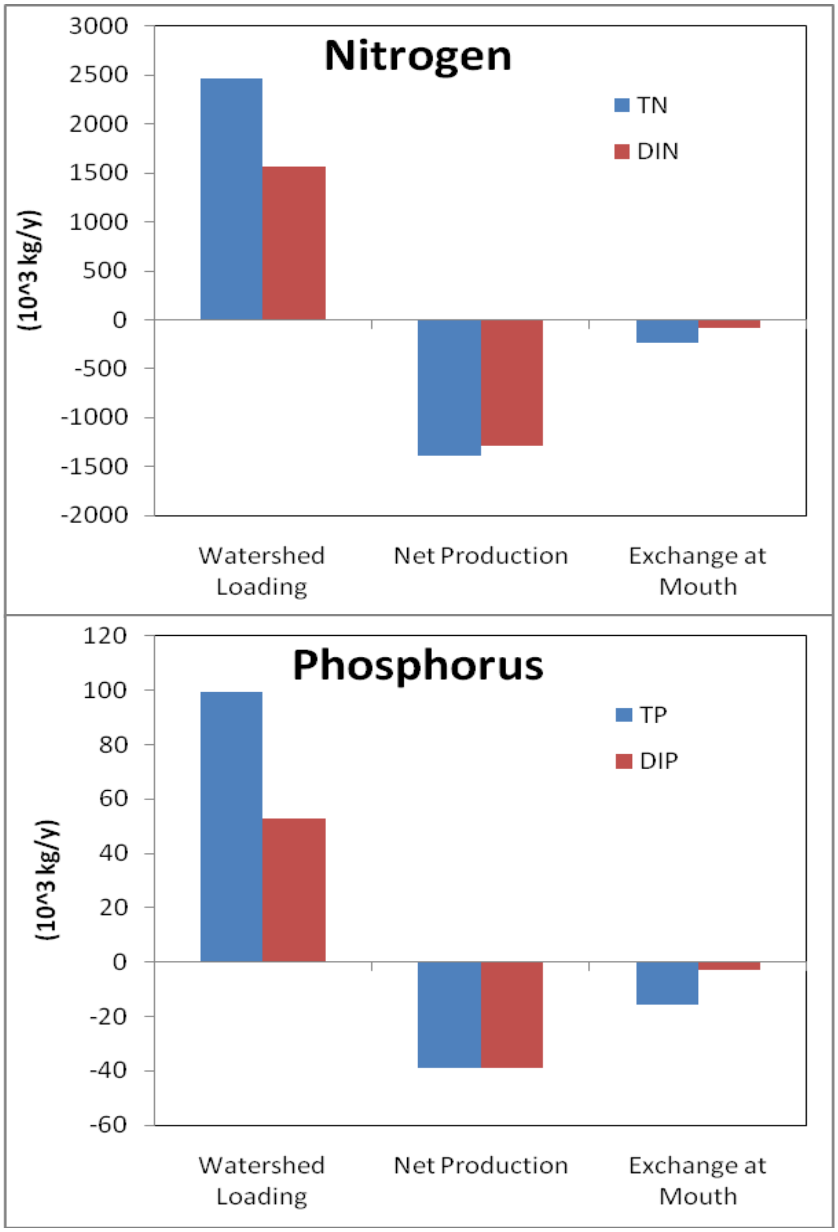


Figure 5-37. Results of an annual-scale box model analysis of nitrogen and phosphorus inputs, exports and net production (internal gains and losses) for the Back River estuary. This computation was based on data from 1998 because all data needed for the computation were available.

5-11 Statistical Analysis of Back River Data

Work focused on threshold descriptions were conducted using several different time-scales of averaged data sets and utilized several different statistical techniques. In all cases the dependent response variable was water column chlorophyll-a concentration.

More specifically, correlation analysis was conducted using monthly, seasonal and annual data, CART analysis utilized monthly data, linear regression utilized all time scales (with and without lags) and multiple regression techniques used annual-scale and multi-annual scale averaged data, also with and without temporal lags. As it turns out, time lags between dependent and independent variables and averaging all data to longer time scales was critically important in finding reasonable relationships between nutrient loads and algal biomass.

5-11.1 Correlation Analysis: A Simple Survey of the Data Set

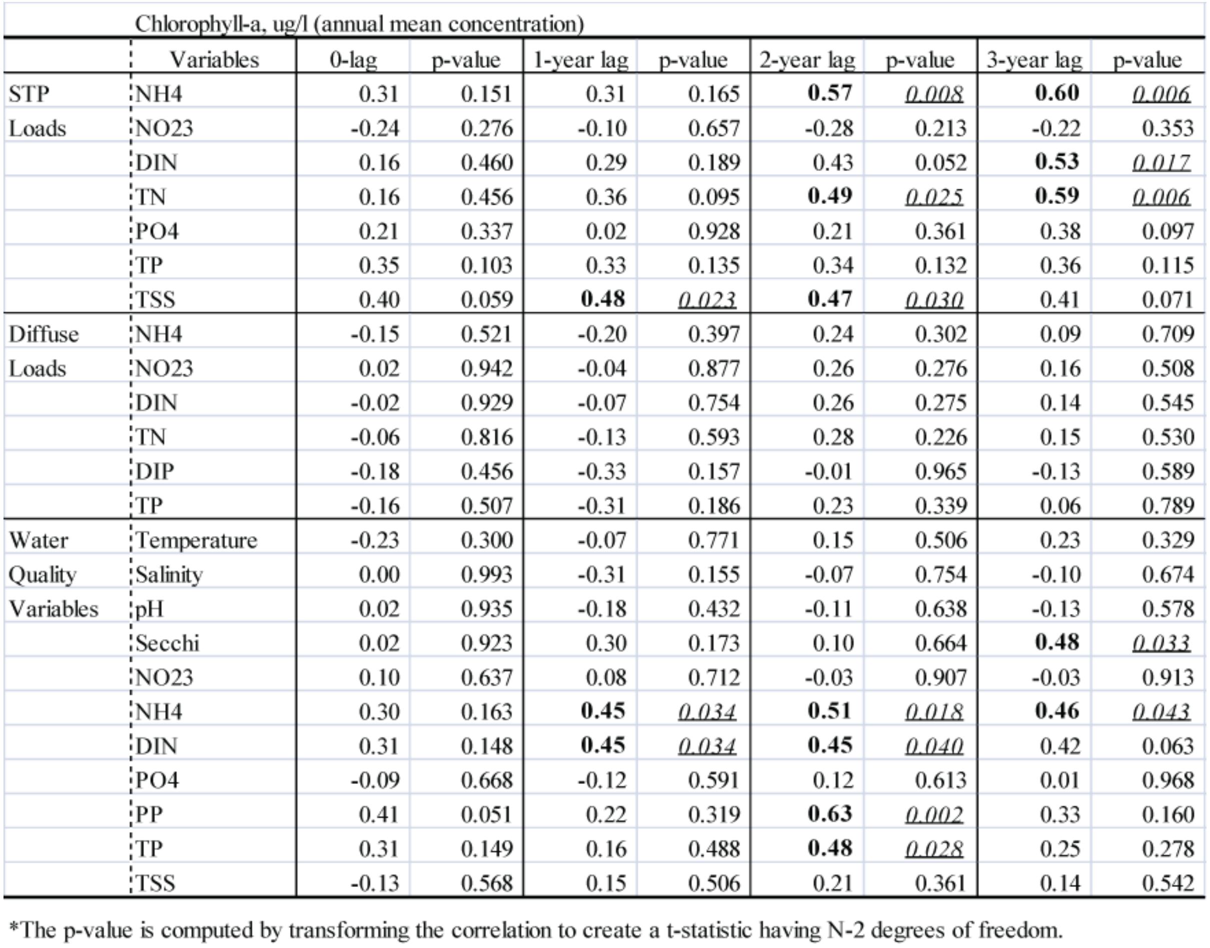
Cross-correlation analysis was used to estimate the degree to which two data series were correlated as for example between a specific nutrient input (e.g., TN) and a water column property (e.g., chlorophyll-a). We used the Matlab software package to perform the statistical analysis. Matlab software also allows for calculation of time lags between variables and this function was used as well.

We began by conducting simple correlation analyses between all variables (nutrient sources plus all water column variables for which there was a 23 year time-series). At the monthly time scale there were a large number of statistically significant correlations, many of them expected. Of 300 possible results, 174 (58%) were significant at the 0.05 probability level. However, these r values, while significant, were generally low and provided little explanatory power. In general, correlations among nutrient species in both point and diffuse loads were relatively high. Similarly, correlations between point and diffuse loads and water column nutrients (and other water column variables) were also frequently observed but r values tended to be much lower. The only strong correlation between water column chlorophyll-a and any other variable was with water column PP concentration (r= 0.52). At the seasonal time scale results of correlation analysis were similar but because the number of observations were reduced by a factor of four via the averaging scheme fewer significant correlations were observed. Again, of a possible 300 results, 152 (51%) were significant. The strongest correlations with chlorophyll-a were with water column PP (r=0.58) and TP (r=0.50).

Because statistical measures explaining variability (r values) were generally quite low using monthly and seasonal-scale data, we focused much of our effort on analysis of annual and multi-annual-scale data following the suggestions of Li et al. 2010. Of 300 possible annual-scale correlations, 27% yielded significant results, again with mostly low, but statistically significant, r values. Several features of this analysis were of interest. There were many significant relationships among the various nutrient species of the point source loads. For example, TN and DIN point source loads were highly correlated (r = 0.96) as were other load components. Similarly, diffuse load components were also significantly correlated. All of these correlations were expected. However, there were no significant and readily understandable correlations between diffuse source loads and either point source loads or water column concentrations of these same variables. The lack of any correlations between diffuse source loads and water column variables was a surprise because diffuse source loads represent a substantial fraction (25-35 %) of the total loads to the system from external sources. This result suggests an overwhelming importance of point sources in this system. In sharp contrast, point source loads were correlated with water column concentrations in many instances. For example, DIN loads were correlated with water column DIN concentrations. Similarly, water column phosphorus concentrations were correlated with point source phosphorus inputs.

Perhaps the most surprising results from these survey analyses is that annual mean water column chlorophyll-a concentrations were not significantly correlated with any concurrently measured point or diffuse load variable or with any water column variable except with water column PP and TP concentration. This suggests that there are more complex relationships involved in controlling chlorophyll-a concentrations in this system. For example, and as indicated earlier, peak chlorophyll-a concentration mainly occurred during spring or summer but also occurred, at lower frequencies, during winter and fall. In addition, sediment data presented earlier (i.e., sediment fluxes of N and P and sediment N and P content) suggest a very nutrient-rich system with large internal storages of N and P. The box model results also indicated strong internal sinks for N and P.

Table 5-3. A summary of correlation coefficients obtained from an analysis relating water column chlorophyll-a concentrations to a variety of nutrient loads (point and diffuse loads) and water column variables. The analysis was conducted using 0, 1, 2 and 3 year lags between the dependent and independent variables. Statistically significant results (p< 0.05) are indicated with bold fonts.



	Chlorophyll-a, ug/l (annual mean concentration)								
	Variables	0-lag	p-value	1-year lag	p-value	2-year lag	p-value	3-year lag	p-value
STP	NH4	0.31	0.151	0.31	0.165	**0.57**	*0.008*	**0.60**	*0.006*
Loads	NO23	-0.24	0.276	-0.10	0.657	-0.28	0.213	-0.22	0.353
	DIN	0.16	0.460	0.29	0.189	0.43	0.052	**0.53**	*0.017*
	TN	0.16	0.456	0.36	0.095	**0.49**	*0.025*	**0.59**	*0.006*
	PO4	0.21	0.337	0.02	0.928	0.21	0.361	0.38	0.097
	TP	0.35	0.103	0.33	0.135	0.34	0.132	0.36	0.115
	TSS	0.40	0.059	**0.48**	*0.023*	**0.47**	*0.030*	0.41	0.071
Diffuse	NH4	-0.15	0.521	-0.20	0.397	0.24	0.302	0.09	0.709
Loads	NO23	0.02	0.942	-0.04	0.877	0.26	0.276	0.16	0.508
	DIN	-0.02	0.929	-0.07	0.754	0.26	0.275	0.14	0.545
	TN	-0.06	0.816	-0.13	0.593	0.28	0.226	0.15	0.530
	DIP	-0.18	0.456	-0.33	0.157	-0.01	0.965	-0.13	0.589
	TP	-0.16	0.507	-0.31	0.186	0.23	0.339	0.06	0.789
Water	Temperature	-0.23	0.300	-0.07	0.771	0.15	0.506	0.23	0.329
Quality	Salinity	0.00	0.993	-0.31	0.155	-0.07	0.754	-0.10	0.674
Variables	pH	0.02	0.935	-0.18	0.432	-0.11	0.638	-0.13	0.578
	Secchi	0.02	0.923	0.30	0.173	0.10	0.664	**0.48**	*0.033*
	NO23	0.10	0.637	0.08	0.712	-0.03	0.907	-0.03	0.913
	NH4	0.30	0.163	**0.45**	*0.034*	**0.51**	*0.018*	**0.46**	*0.043*
	DIN	0.31	0.148	**0.45**	*0.034*	**0.45**	*0.040*	0.42	0.063
	PO4	-0.09	0.668	-0.12	0.591	0.12	0.613	0.01	0.968
	PP	0.41	0.051	0.22	0.319	**0.63**	*0.002*	0.33	0.160
	TP	0.31	0.149	0.16	0.488	**0.48**	*0.028*	0.25	0.278
	TSS	-0.13	0.568	0.15	0.506	0.21	0.361	0.14	0.542

*The p-value is computed by transforming the correlation to create a t-statistic having N-2 degrees of freedom.

These observations suggested we needed to look more carefully at lag times between water column chlorophyll-a and what are typically thought of as controlling variables of chlorophyll-a concentration.

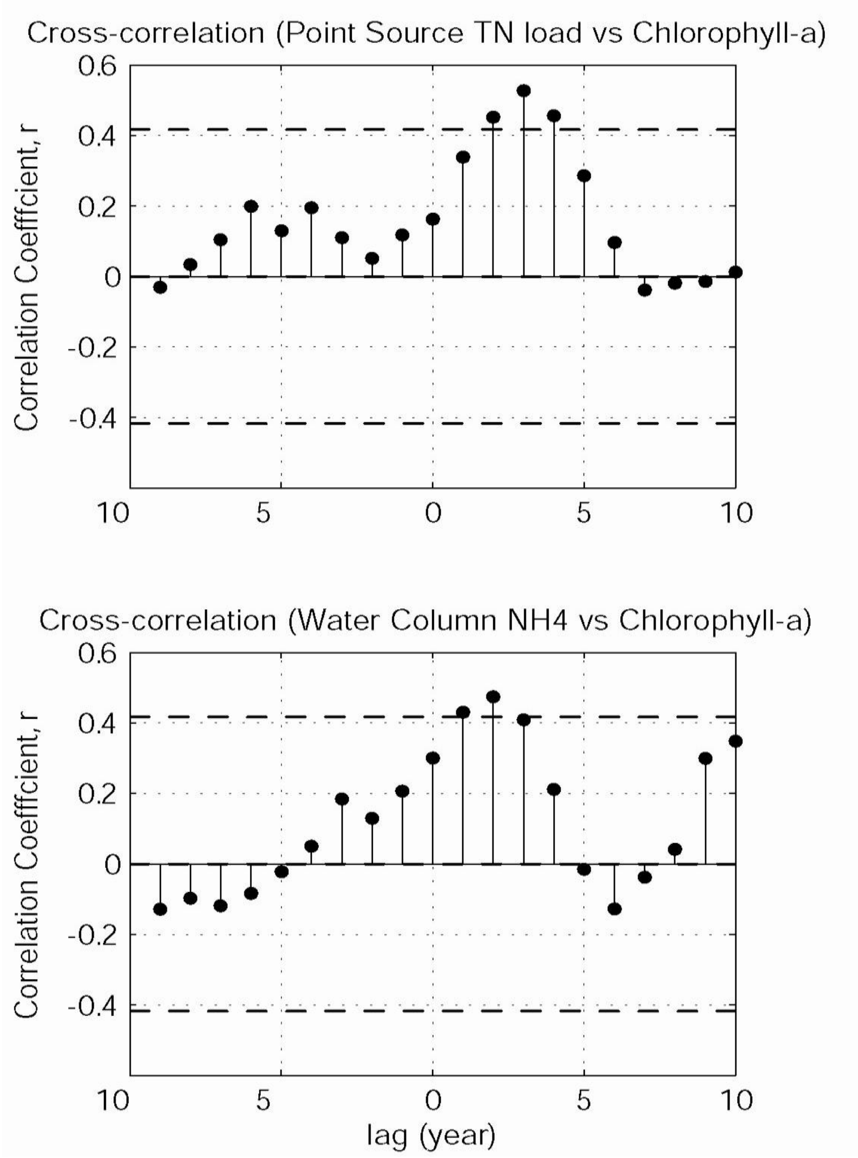


Figure 5-38. An example of results of correlation analysis for two variables (STP TN; point source total nitrogen load and WQ NH4; water column NH4 concentration) indicating the importance to temporal lags of 1 to 3 years. Horizontal lines in both panels indicate correlation coefficient values needed for significance at the $p < 0.05$ level.

We again used a Matlab routine to explore the data set for correlations between chlorophyll-a and point source loads (7 variables), diffuse sources (6 variables) and water column conditions (11 variables) using annual-scale data and 0, 1, 2, and 3 year lags (Table 5-3). As reported above, with no lags there were no significant correlations between chlorophyll-a and point, diffuse or water column variables. There was a marginally significant relationship between chlorophyll-a and water column PP concentration. However, when multi-year lags were introduced more significant relationships appeared, especially with 2 and 3 year lags (Figure 5-38). Most significant results involved N loads or N concentrations and this is readily understandable. It is interesting to note again that there were no significant relationships between chlorophyll-a and any variable associated with diffuse loads. While these results were encouraging and seemingly consistent with an emerging understanding of how this system may operate (i.e., large internal nutrient storages are important) there was still little explanatory power gained from these analyses. However, the possible importance of substantial lag times was noted.

5-11.2 Linear Regression Analyses: With and Without Time Lags

Linear regression methods available in Matlab were used to quantify the relationship between a dependent variable (e.g., chlorophyll-a) and an independent variable (nutrient inputs and water column properties). Linear regression analyses, with and without time lags, were conducted using monthly, seasonal and annual-scale data. These analyses were designed to explore the data set for relationships between water column chlorophyll-a and various water column and nutrient load variables. Independent variables included all forms of N and P associated with point and diffuse sources as well as TSS loads and all of the routinely measured water column variables (e.g., temperature, salinity, pH, water clarity, TSS and all N and P nutrient species). Results for all temporal averaging schemes indicated that no one water column or load variable explained much of the variability in water column chlorophyll-a concentration and the majority were non-significant. Regression models attempting to relate point, diffuse and total loads to chlorophyll-a concentration with various lag times (one month for monthly data, one season for seasonal data and one year for annual data) were also not successful. For example, a regression model of seasonal total DIN load versus seasonal chlorophyll-a with a one season lag yielded an r2 value of 0.06. Of some 60 different seasonal-scale regressions using various forms of nutrient load and chlorophyll-a, almost none yielded significant results and none exhibited useful explanatory power. Again, these results likely indicate a complex set of conditions regulating chlorophyll-a concentrations in this system.

5-11.3 CART Analysis: Additional controlling variables

Classification and regression trees (CART) is a non-parametric decision tree learning technique that produces either classification or regression trees, depending on whether the dependent variable is categorical or numeric, respectively. Classification tree analysis is used when the predicted outcome is the class to which the data belongs. Regression tree analysis is used when the predicted outcome is a quantitative value, which is the case here (i.e., chlorophyll-a concentration). CART analysis constructs a set of decision rules that identify homogeneous groups of the response variable as a function of a set of explanatory variables. During each recursion, splits for each explanatory variable are examined, and the split that maximizes the homogeneity of the two resulting groups with respect to the dependent variable is chosen. To avoid over-fitting of the data, algorithms used in CART usually simplify or “prune” the tree that contains all possible splits of the data to an optimal tree that contains a sufficient number of splits to describe the data.

The lack of success in the simple regression analyses (with and without lags) suggests a more complex set of factors regulating the dependent variable are involved and, hence, CART analysis was thought to be a useful tool because this approach explicitly considers multiple independent variables.

Several CART analyses were conducted, some yielding seemingly spurious results and others yielding very short trees. These results suggested a pruning of the data set (eliminating some variables) was needed. After this step was completed, analyses made more sense. Results of the final CART analysis are shown in Figure 5-39. In this case average monthly chlorophyll-a concentration was the response variable and monthly water column concentrations and month-scale diffuse and point source loads were the independent variables. The CART analysis produced the first split of the tree based on water temperature above and below 11 ºC. This is consistent with an earlier observation that maximum chlorophyll-a concentration generally (but not always) occurred

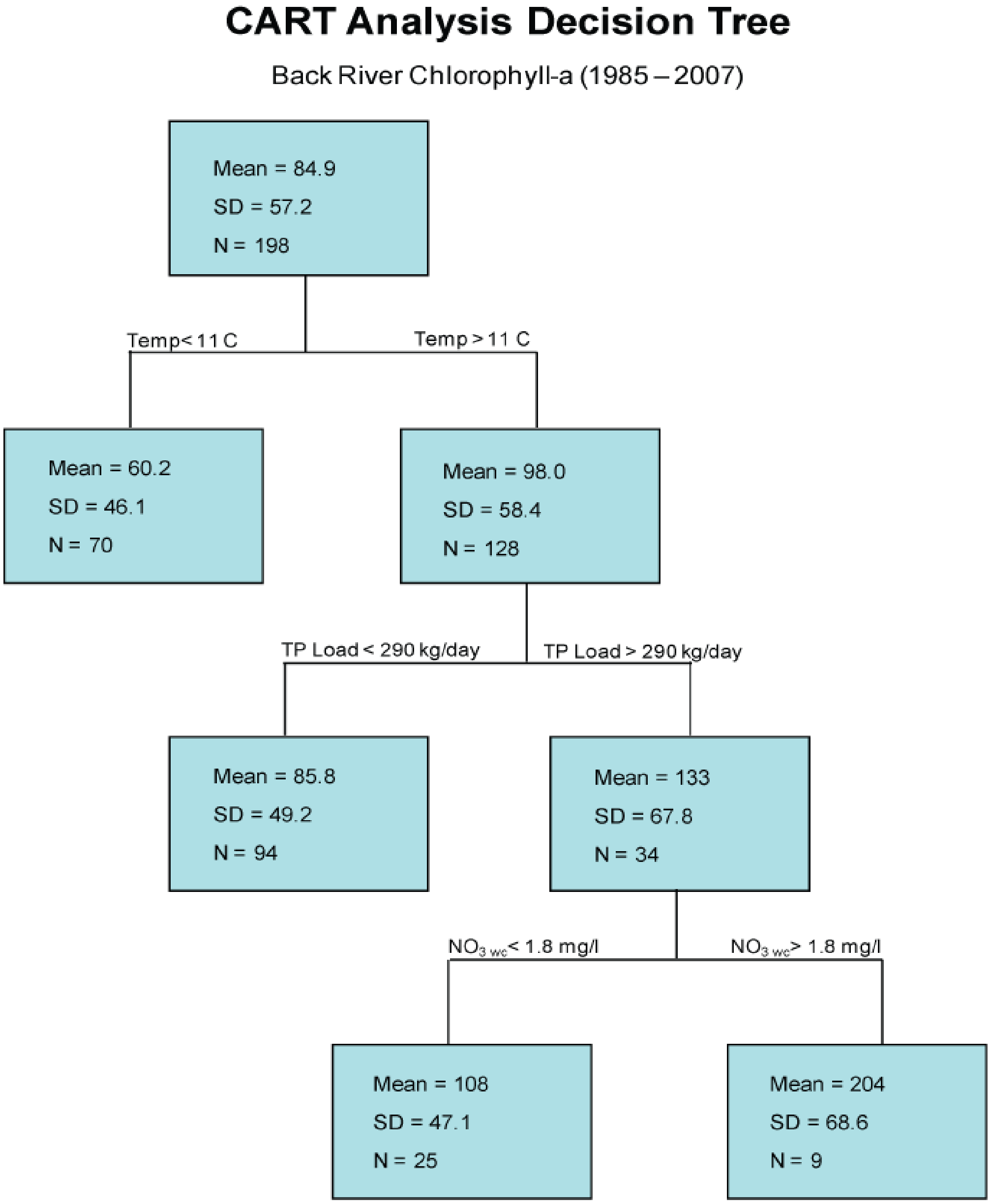


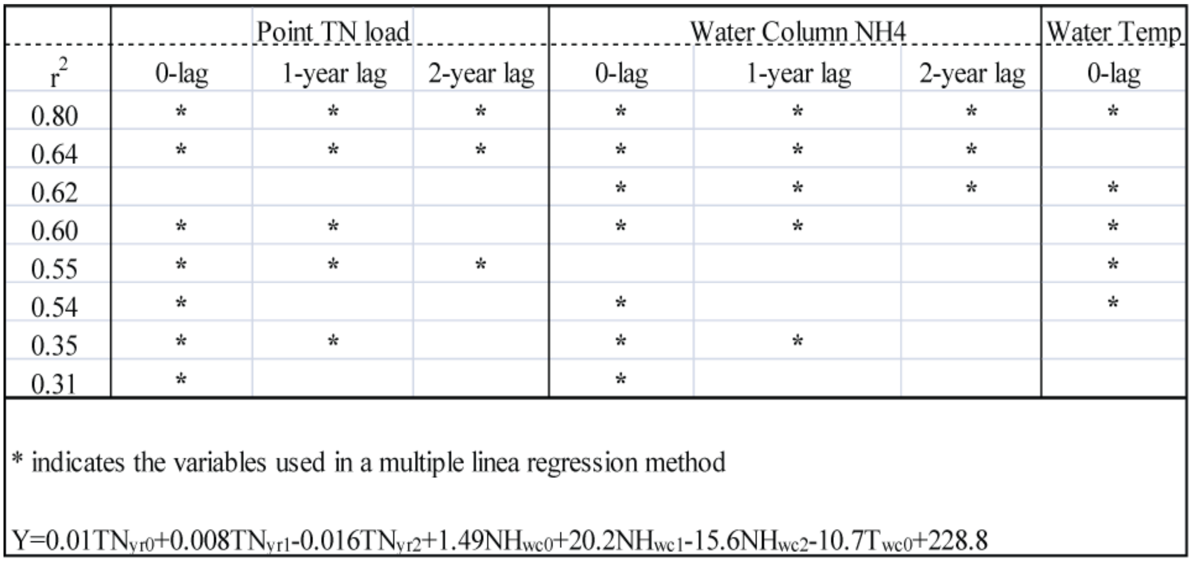
Figure 5-39. Graphic display of CART analysis based on monthly data obtained from the Back River estuary for the period 1985-2007. Water column chlorophyll concentration was the dependent variable in this analysis. Independent variables included most of the variables indicated in Table 5-3.

during the warmer periods of the year. The second split was based on TP loads less than or greater than 290 kg P day -1. In this low salinity system the importance of phosphorus as an important variable regulating phytoplankton dynamics is consistent with previous work that reported that in the low salinity areas of the Bay and tributary rivers P and light were most often the factors limiting algal growth rates (e.g., Fisher et al. 1999). The final split was based on water column NO23 concentrations less than or greater than 1.8 mg l -1. This final split is more difficult to explain. Concentrations of NO23 in excess of 1.8 mg l-1 are very high, far in excess of any physiological limiting concentration. Nevertheless, on a few occasions (n = 9) extreme chlorophyll-a concentrations (mean = 204 µg l-1) were measured when nitrate concentrations were also very high. Thus, this exploratory analysis suggests multiple factors play a role in determining chlorophyll-a concentrations.

5-12 Modeling Chlorophyll-a: Multi-year Time Lags

Analyses to this point indicated the following: 1) this system, even after WWTP upgrades, was heavily loaded with N and P from point sources; 2) diffuse sources were a smaller fraction of the N and P load and exhibited little relationship to measured water quality variables; 3) N and P were not rapidly transported to Chesapeake Bay but retained in the system resulting in high N and P concentrations in sediments as well as the water column. The last point suggested that there might be a good deal of “nutrient inertia” in this system and that lags needed to be seriously considered in modeling efforts. In Table 5-4 the results of “building up” a multiple regression model are summarized. This model started with two variables identified as somewhat important in earlier analyses (point source TN load and water column NH4 concentration). As time lags were added to the model, r2 values increased to 0.64. The residuals of this model were then regressed against all loading and water quality variables and a significant relationship with water temperature emerged; all others were either not significant or only marginally significant. When temperature was added to the model, the r2 value increased to 0.80 (Figure 5-40). We were unable to improve the explanatory power of this analysis further. This model has both strong and weak aspects. Perhaps the least attractive aspect is that the final model has seven variables, probably too many to justify in a strict statistical fashion. On the other hand, this model does relate algal biomass to both N inputs and N concentrations in the water and this is consistent with current understanding of algal/nutrient dynamics (Smith 2006). Furthermore, the water column nutrient of importance (NH4) is preferred

Table 5-4. A summary of results for a variety of multiple linear regression models relating nutrient loads (point source TN load) and water column variables (NH_4 concentration and temperature) to chlorophyll concentration. Annually averaged data were used in this analysis and 0, 1 and 2 year time lags were considered. The equation (with coefficients) at the bottom of the table is for the full regression model (r^2 = 0.80).



	Point TN load			Water Column NH4			Water Temp
r^2	0-lag	1-year lag	2-year lag	0-lag	1-year lag	2-year lag	0-lag
0.80	*	*	*	*	*	*	*
0.64	*	*	*	*	*	*	
0.62				*	*	*	*
0.60	*	*		*	*		*
0.55	*	*	*				*
0.54	*			*			*
0.35	*	*		*	*		
0.31	*			*			

* indicates the variables used in a multiple linea regression method

$$Y=0.01TN_{yr0}+0.008TN_{yr1}-0.016TN_{yr2}+1.49NH_{wc0}+20.2NH_{wc1}-15.6NH_{wc2}-10.7T_{wc0}+228.8$$

by many algal species and, perhaps more importantly, it is the first inorganic N product of organic matter re-mineralization. This latter point is consistent with the idea of large internal storages of nutrients being important in influencing algal biomass conditions in this estuary. While a bit cumbersome, a model has been produced using reasonable variables and accounts for much of the inter-annual variability in water column chlorophyll-a during a 23 year record, including and a period of change in nutrient loading rates related to management actions.

5-13 Modeling Chlorophyll-a: Multi-year Time Averaging and Lags

In a recent paper Li et al. 2010 examined several nutrient/chlorophyll-a data sets collected during various time intervals ranging from hourly to decadal. Among other things, they reported stronger relationships between nutrients and chlorophyll-a when these variables were averaged over longer time periods. We adopted a version of this approach in this final set of regression models for the Back River estuary. In this case monthly data for all variables were averaged to annual scales and then averaged into 2, 3 and 4-year averages. In the simplest model, 3-year averaged chlorophyll-a was regressed against 3-year averaged point source TN load with no temporal lags (Fig. 5-41). The resultant regression was not significant but did show a general, but irregular, decline in chlorophyll-a associated with point source load reductions.

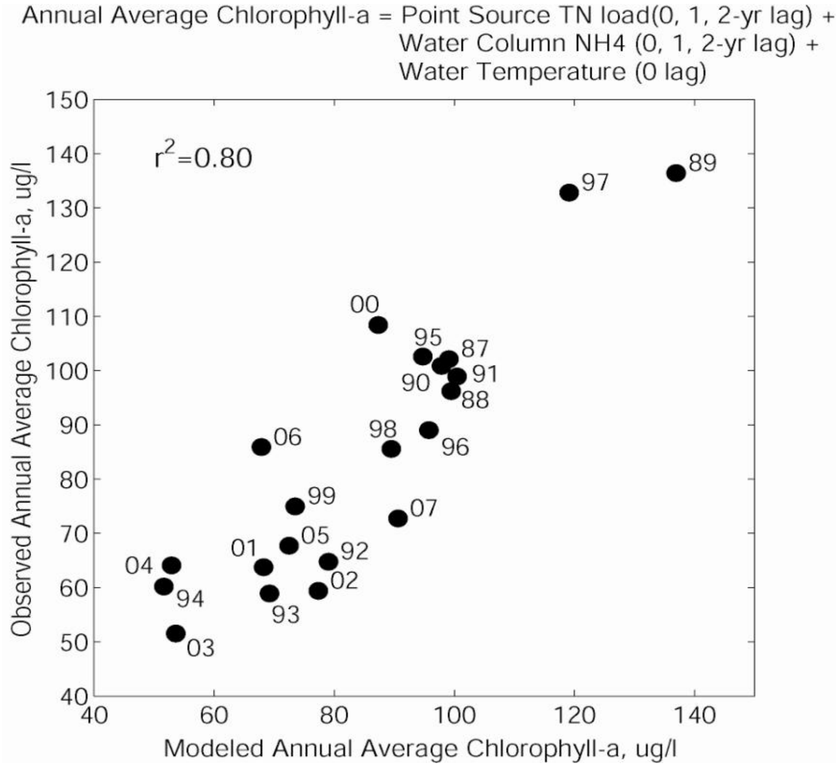


Figure 5-40. A scatter plot showing the relationship between observed average annual chlorophyll-a concentration and modeled chlorophyll-a concentration for a 23 year period of record in the Back River estuary. The component pieces of the regression model are shown and model coefficients are provided in Table 5-4.

We next considered the notion of lags and reasoned that in this nutrient-rich system it would not be surprising if the signal from averaged inputs extended over a number of years. The next model lagged nutrient inputs by one time unit (Fig 5-42) and the resultant regression was significant even though the number of observations were reduced from 8 to 7 because of the lag required. Regression models using 2 or 4 year averaging and similar lag times were not as significant as efforts using 3-year averaging schemes. In this model there was also a far clearer response to nutrient load reductions (96-98 to 02-04) and also a clear response to more recent nutrient load increases from the WWTP (02-04 to 05-07).

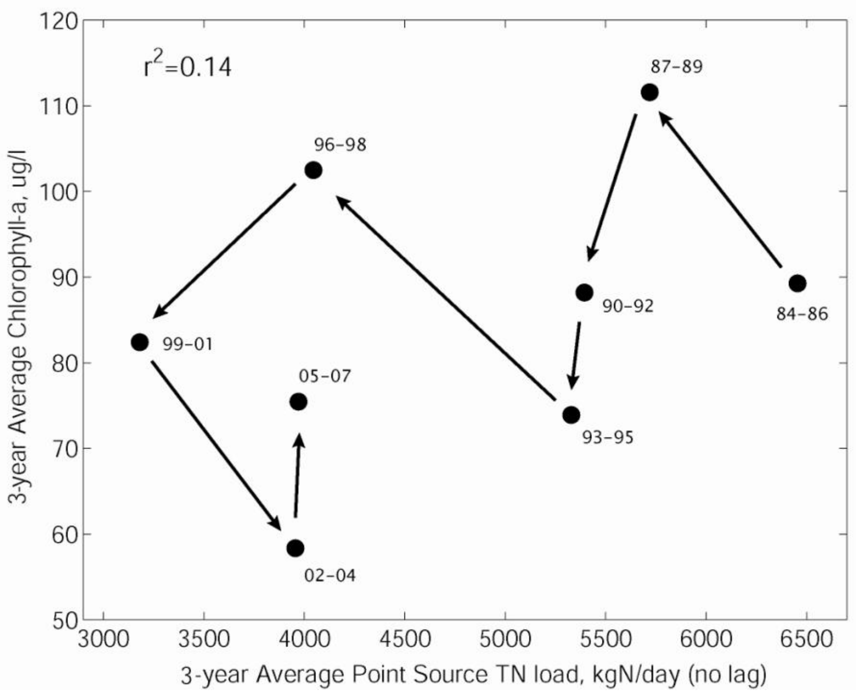


Figure 5-41. A scatter plot showing the relationship between observed 3 year average chlorophyll-a concentration and 3 year average point source TN load (with no lags). The arrows on the diagram indicate the temporal trajectory of changes in chlorophyll-a concentration. This regression model is not significant at the 0.05 probability level.

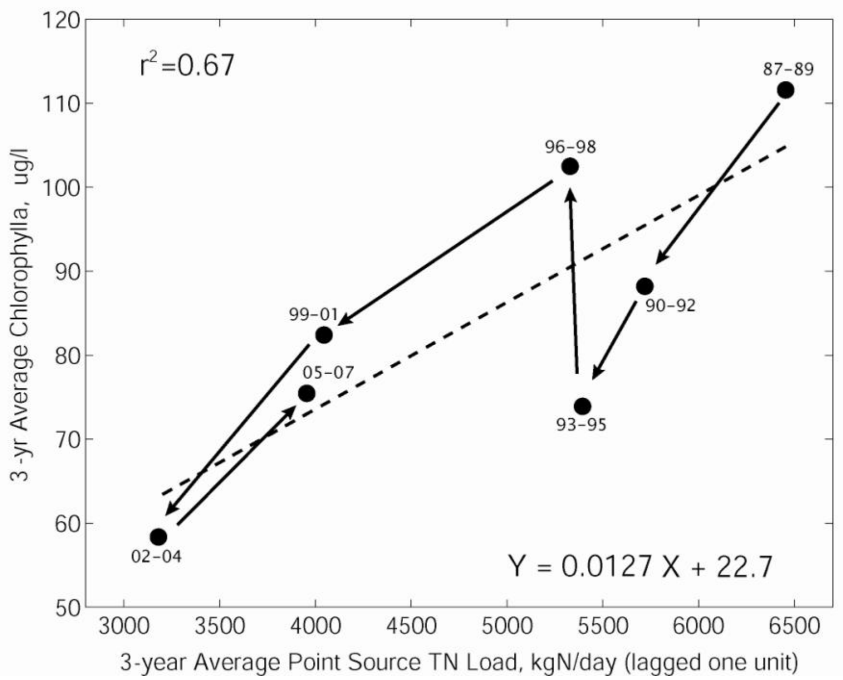


Figure 5-42. A scatter plot showing the relationship between observed 3 year average chlorophyll-a concentration and 3 year average point source TN load (with one unit lag time). The arrows on the diagram indicate the temporal trajectory of changes in chlorophyll-a concentration. The dashed line is the regression line. This regression model is significant at the 0.05 probability level.

We then combined components of the previous two models (independent variables included 0 lag 3-year averaged TN load plus 1-lag 3-year averaged TN load) and obtained an improved result where about 88% of the variability in the record was accounted for using these two independent variables (Fig. 5-43). In simple terms, this model suggests that both recent and less recent nitrogen loads were important in determining multi-year averaged chlorophyll-a concentrations.

In this analysis we have focused on nitrogen as a key regulating factor. We recognize that other factors, such as light availability, flushing rate, phosphorus dynamics and others, could also play a role. Analysis of the data set does not support a strong role for either flushing or water clarity. In the case of flushing (as indexed by freshwater input rates) there was inter-annual variability but not any consistent relationships with chlorophyll-a. In the case of water clarity there was very little inter-annual variability. The estuary was always quite turbid. The case for phosphorus is not so clear. At the annual time scale there was a significant correlation between DIP concentration and chlorophyll-a but the correlation had little explanatory power. When we examined the data set for chlorophyll-a relationships to P we found that significance indicators increased but only with time lags of 4 to 5 years and these were not as strong as were indicators for N at shorter time lags. With long P lag times the number of observations are reduced (because of the time lags) and lag analyses did not produce results as strong as those for nitrogen. As the Back River time-series lengthens it would be prudent to re-visit the phosphorus analysis. Because of this uncertainty in the Back River analysis and because other low salinity sites have clear signals indicating the importance of

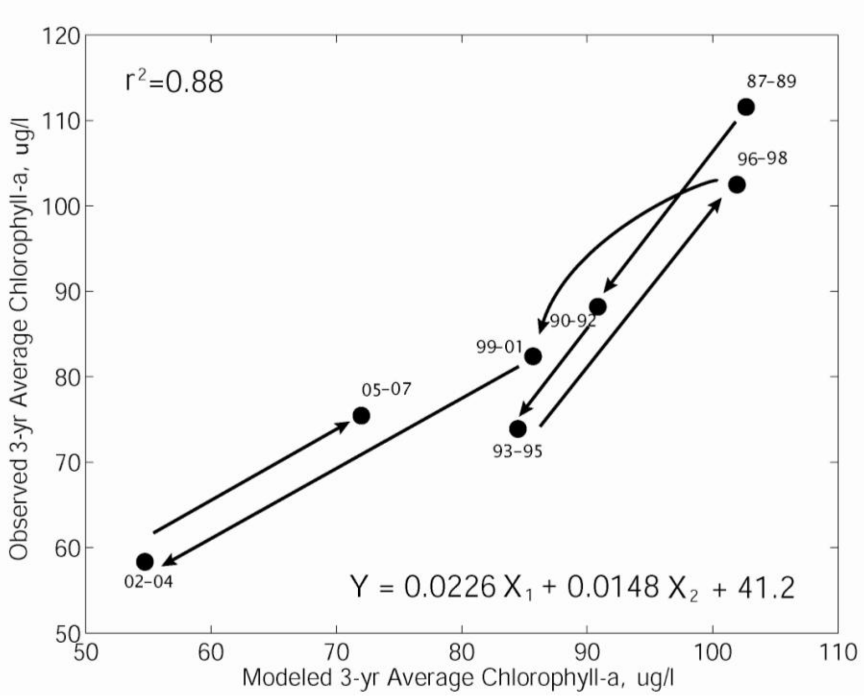


Figure 5-43. A scatter plot showing the relationship between observed (3-year average chlorophyll-a concentration) and modeled 3 year average chlorophyll-a concentration (with 0 and one unit lag time of point source TN load). The arrows on the diagram indicate the temporal trajectory of changes in chlorophyll-a concentration. This regression model is significant at the 0.05 probability level.

phosphorus dynamics (see Case Study #1), the dual nutrient reduction strategy of the Bay Program seems well justified.

5-14 Summary: Thresholds or Lagged Responses

So, what does all this analysis tell us about the reaction of the Back River estuary to a strong management action? The bulk of this analysis suggests the Back River has been and still is a very heavily loaded system, especially for nitrogen. However, WWTP upgrades have caused a substantial decrease in load, especially during the period 1995–2002. After this period, loads again increased because of increased flow from the WWTP but not to the levels recorded prior to 1995. Our examination of the data sets did not reveal any dramatic “threshold-like” responses related to chlorophyll-a or any other water quality variables. Rather, there did appear to be important responses of longer-term (3 year average) chlorophyll-a to time averaged and lagged point source N loading. In the simplest of these averaged and lagged regression models (Fig. 5-42) there is about a factor of two reduction in chlorophyll-a concentration between 1987-1989 and 2002-2004 associated with about a factor of two reduction in point source N loads. Additionally, there was a proportional increase in chlorophyll-a concentration associated with N load increases towards the end of the time-series. So, a key here, if we are viewing these data correctly, is that this system has substantial nutrient memory relative to nitrogen and possibly phosphorus as well. The idea of system memory is well documented feature of lakes recovering from eutrophication and typically is focused on phosphorus. In the case of the Back River sewage disposal began about 100 years ago. The sediments in this system are as rich or richer in both N and P as any site in Chesapeake Bay (Boynton and Bailey 2008). Thus, there is clear evidence of nutrient storage, a requirement for invoking any sort of nutrient memory. Given that the strongest predictor of chlorophyll-a involved 3 year averaged N loading and a one 3-year lag of N loading we would expect this system to respond slowly (3-6 year lag) to further nutrient load reductions, which are planned for this WWTP facility.

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Chapter 6

Summer Hypoxia Model and Prediction: Directions for Use

W.R. Boynton and Y. Lee

6-1 Introduction

Since summer hypoxia is shown to be more related to late winter-spring processes than summer stratification, we have developed an effective tool to predict summer hypoxia based on river discharge, nutrient load, algal biomass, and wind conditions. The Susquehanna River contributes both nutrient loads from the land and buoyancy effects on estuarine dynamics. The concentration of spring chlorophyll-a, a proxy for spring algal biomass, is also associated with the initiation and duration of hypoxia. Moreover, cross-bay wind is significantly correlated with summer hypoxia, which is influenced by regional climate. Table 6-1 lists the variables used in a multiple linear regression analysis for predicting summer hypoxia (June-September).

Table 6-1. The variables used in the multiple linear regression model.

Variables		Period	Unit
Dependent (predictand)	Mean Summer Hypoxic Volume (Hypoxia)	Jun-Sep	km^3
Independent (predictor)	Mean Susquehanna River Discharge (River)	Jan-May	m^3/sec
	Mean Total Nitrogen Load from the Susquehanna River (TN)	Jan-May	kg/day
	Mean Concentration of Chlorophyll-a in the Mid-Bay (Chla)	Feb-Apr	µg/l
	Mean Cross-bay Wind at Patuxent River Naval Air Station (Uwind)	Feb-Apr	m/s

6-2 Data

6-2.1 Hypoxic Volume

Dissolved oxygen (DO) concentrations were interpolated to calculate hypoxic volume (DO < 2.0 mg L^{-1}) using the Chesapeake Bay Program data (1985-2010). The hypoxic volume was then averaged for June-September, representing summer hypoxia. The Chesapeake Bay Program also provides the hypoxic volume datasets which can be applied to a multiple linear regression model (ftp://ftp.chesapeakebay.net/Monitoring/HypoxicVolumeDatasets_1985-2010).

6-2.2 Freshwater Flow and Total Nitrogen Load

The Susquehanna River discharge is the largest single source to Chesapeake Bay and it is considered to represent a good estimate of freshwater flow and total nitrogen (TN) load into the Bay. Monthly freshwater flow and TN load were retrieved from two websites (Table 6-2) and averaged for January-May.

Table 6-2. Websites used to retrieve monthly freshwater flow and TN load.

Website	Data Information
http://waterdata.usgs.gov/usa/nwis/uv?01578310	Monthly freshwater flow
http://va.water.usgs.gov/chesbay/RIMP/loads.html	TN Load

6-2.3 Chlorophyll-*a* Concentration

Since phytoplankton production provides the main organic matter sources for oxygen consumption in the Bay, depth-averaged chlorophyll-a concentration is used as a proxy for water column algal biomass. Chlorophyll-a concentration were averaged for the monitoring stations in the mid-bay region (i.e., stations CB3.3C, CB4.1C, CB4.2C, CB4.3C, CB4.4, CB5.1, CB5.2, and CB5.3) during the spring bloom period between February and April.

6-2.4 Cross-bay Wind

Wind data were obtained from the Naval Air Station (NAS) near the mouth of the Patuxent River, a centrally located position in Chesapeake Bay (http://www4.ncdc.noaa.gov/cgi-win/wwcgi.dll?wwDI~StnSrch~StnID~20009429). The wind vector (wind speed and direction) can be expressed as two velocity components; Uwind (cross-bay wind towards East) and Vwind (along-bay wind towards North):

$$U_{wind} = -W_{spd} \times \sin\left[\frac{\pi}{180} \times W_{dir}(deg)\right] \quad \text{------ (1)}$$

$$V_{wind} = -W_{spd} \times \cos\left[\frac{\pi}{180} \times W_{dir}(deg)\right] \quad \text{------ (2)}$$

where, Wspd is a wind speed and Wdir is a meteorological wind direction, i.e., the direction from which the wind is blowing. Then, hourly Uwind is averaged for February-April during the spring bloom.

6-3 Models

A multiple linear regression is used to model the relationship between a dependent variable (predictand) and independent variables (predictors). It is based on a least square method to minimize the sum-of-squares of differences between observed and predicted values. The model expresses the values of a summer hypoxic volume as a linear function of four independent variables as listed in Table 6-1 and an error term:

$$Hypoxia = a_0 + a_1 \cdot River + a_2 \cdot TN + a_3 \cdot Chla + a_4 \cdot Uwind + e \quad \text{------ (3)}$$

where, a0 is regression constant, a1, a2, a3, and a4 are coefficients on the independent variables, and e is an error term. The model (3) is estimated by least squares, which yields parameter estimates denoted as '^' in (4). The resulting prediction equations is:

$$Hypoxia_{predicted} = \hat{a}_0 + \hat{a}_1 \cdot River + \hat{a}_2 \cdot TN + \hat{a}_3 \cdot Chla + \hat{a}_4 \cdot Uwind \quad \text{------ (4)}$$

One measure of goodness-of-fit is to calculate the coefficient of determination (R^2) which indicates how closely predicted values obtained from a regression model match the dependent variable that the regression model is intended to predict:

$$R^2 = 1 - \frac{\sum \left(Hypoxia_{observed} - Hypoxia_{predicted}\right)^2}{\sum \left(Hypoxia_{observed} - \overline{Hypoxia}_{observed}\right)^2} \quad \text{------ (5)}$$

6-4 Example

A multiple linear regression method was applied using the data from 1985 to 2009. Figure 6-1 shows that the regression model produced a significant result with observed hypoxic volume (R^2=0.70).

Table 6-3. Estimated coefficients in the multiple linear regression model using the data between 1985 and 2009.

Estimated Coefficient	Value
â0	3.19
â1	2.56×10-3
â2	-1.02×10-5
â3	6.44×10-2
â4	-1.02

Based on the estimated coefficients from the multiple linear regression model (Table 6-3), summer hypoxia in 2010 was predicted as 4.46 km^3 which less than the observed hypoxic volume of 5.19 km^3 in 2010. Average summer hypoxia during 2010 was in the lower 2/3 of all years between 1985 and 2010.

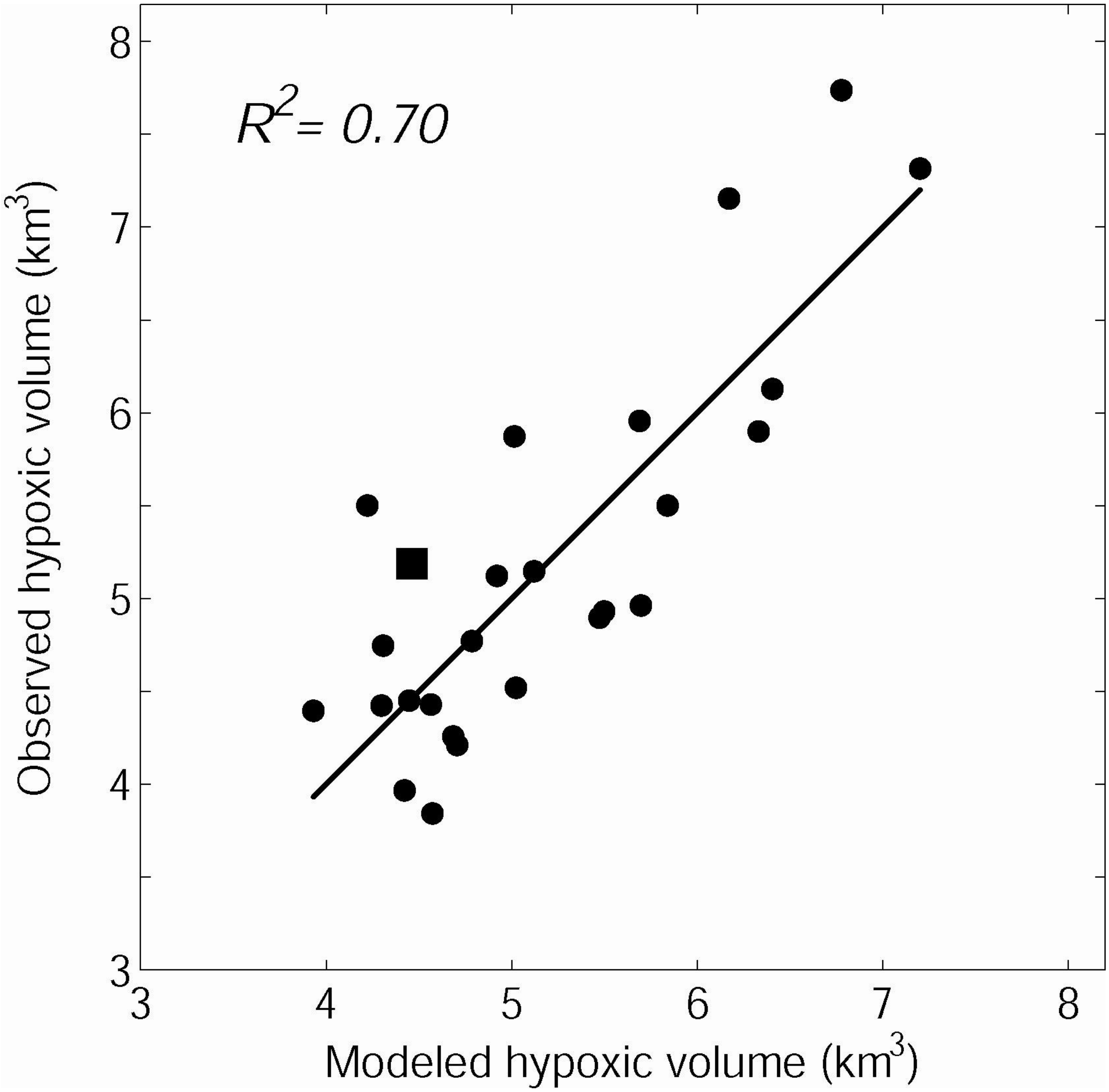

Figure 6-1. The multiple linear regression analysis of the mean summer hypoxic volume (June-September) of Chesapeake Bay with the independent variables, i.e., mean freshwater flow and TN load from the Susquehanna River (January-May), mean chlorophyll-a concentration in the mid-bay (February-April), and mean cross-bay wind component (February-April). A square marker (■) indicates the predicted hypoxic volume in 2010 based on the estimated coefficients shown in Table 6-2.