# Multi-decade Responses of a Tidal Creek System to Nutrient Load Reductions: Mattawoman Creek, Maryland USA

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**Abstract** We developed a synthesis using diverse monitoring and modeling data for Mattawoman Creek, Maryland, USA to examine responses of this tidal freshwater tributary of the Potomac River estuary to a sharp reduction in point-source nutrient loading rate. Oligotrophication of these systems is not well understood; questions concerning recovery pathways, threshold responses, and lag times remain to be clarified and eventually generalized for application to other systems. Prior to load reductions Mattawoman Creek was eutrophic with poor water clarity (Secchi depth <0.5 m), no submerged aquatic vegetation (SAV), and large algal stocks (50–100 μg L$^{-1}$ chlorophyll-$a$). A substantial modification to a wastewater treatment plant reduced annual average nitrogen (N) loads from 30 to 12 g N m$^{-2}$ year$^{-1}$ and phosphorus (P) loads from 3.7 to 1.6 g P m$^{-2}$ year$^{-1}$. Load reductions for both N and P were initiated in 1991 and completed by 1995. There was no trend in diffuse N and P loads between 1985 and 2010. Following nutrient load reduction, $NO_2+NO_3$ and chlorophyll-$a$ decreased and Secchi depth and SAV coverage and density increased with initial response lag times of one, four, three, one, and one year, respectively. A preliminary N budget was developed and indicated the following: diffuse sources currently dominate N inputs, estimates of long-term burial and denitrification were not large enough to balance the budget, sediment recycling of $NH_4$ was the single largest term in the budget, SAV uptake of N from sediments and water provided a modest seasonal-scale N sink, and the creek system acted as an N sink for imported Potomac River nitrogen. Finally, using a comparative approach utilizing data from other shallow, low-salinity Chesapeake Bay ecosystems, strong relationships were found between N loading and algal biomass and between algal biomass and water clarity, two key water quality variables used as indices of restoration in Chesapeake Bay.

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## Introduction

Understanding the causes and consequences of eutrophication in lagoons, bays, estuaries, and near-coastal waters has been the focus of much research during the last five to six decades, starting in the USA with the work of Ryther (1954) on Moriches Bay, NY where duck wastes were linked to intense algal blooms, a reduction in oyster production, and aesthetic impacts. Since that beginning our understanding of eutrophication now includes a useful definition (Nixon 1995), general conceptual models (Cloern 2001), more specific models of shallow (Nixon *et al.* 2001) and river-dominated (Kemp *et al.* 2005) systems, reviews of nitrogen (N) versus phosphorus (P) limitation (Rabalais 2002; Howarth and R. Marino 2006; Smith *et al.* 2006; Paerl 2009), and consideration of thresholds (Conley *et al.* 2009) and other feedback processes that can exacerbate or suppress eutrophication (Kemp *et al.* 2005; Conley *et al.* 2007; Gruber and Kemp 2010).

During the past decade, there has been a growing interest in estuarine science and water quality management communities to understand how the ecosystem responds to nutrient load reductions or in the terms of Nixon (2009), the oligotrophication of these systems. This represents a change in focus but is understandable because large amounts of public funds are being devoted to restoration efforts. Interest in oligotrophication has stimulated thinking, speculation, and synthesis on the likely responses of these important resources to reduced nutrient loading rates. For example, Duarte *et al.* (2009) reviewed responses of several systems and found convoluted trajectories that failed

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to return to pre-eutrophication conditions. Kemp *et al*. (2009) examined response trajectories related to hypoxia reduction in 24 coastal ecosystems and found about half displayed remediation trajectories that mirrored the degradation trajectory while the remainder displayed complex patterns similar to those reported by Duarte *et al*. (2009). Studies of ecosystem responses to load reductions in Chesapeake Bay and elsewhere are relatively rare. Available sources indicate a variety of responses including no or limited response (Conley *et al*. 2002; Kronvang *et al*. 2005; Boynton *et al*. 2009), gradual improvement (Jeppesen *et al*. 2005; Murphy *et al*. 2011), rapid responses (Rask *et al*. 1999), delayed algal biomass reduction (Yamamoto 2003; Boynton *et al*. 2011), and several threshold-like responses involving submerged aquatic vegetation communities (Johansson 2002; Orth *et al*. 2010; Ruhl and Rubicki 2010). Given the large financial costs associated with restoration programs in the Chesapeake and elsewhere it is important to develop a better understanding of system responses to these actions.

This analysis focuses on Mattawoman Creek, an oligohaline/tidal freshwater tributary of the upper Potomac River estuary. This site was selected for analysis for several reasons. First, between the 1970s and mid 1990s, this system was very eutrophic, having large algal blooms and lacking submerged aquatic vegetation. A major reduction of point source nutrient loads was achieved during the early 1990s. Second, this system has been the focus of study and interest by federal, state, and local volunteer organizations interested in preserving and improving habitat quality in the face of growing development. Hence, there is a diverse and long-term data set available for examination. Finally, this system is connected via tidal exchanges with the upper Potomac estuary. While nutrient load reductions have been achieved in the upper Potomac estuary, loads remain high and nutrient concentrations in the Potomac adjacent to Mattawoman Creek are higher than those within the creek. Thus, it is likely Mattawoman Creek receives nutrient loads from the local watershed, atmosphere, and the adjacent tidal Potomac. Understanding the influence of downstream waters (tidal Potomac in this case) on upstream waters receiving management actions is of particular interest to the management community.

The focus of this analysis concerns water quality (chlorophyll-*a* and nutrient concentrations and water clarity) and habitat conditions (SAV communities) in Mattawoman Creek. We examine how these features have responded to past and current management actions and speculate how the creek may respond to future land use and nutrient load alterations. Specifically, we summarize information concerning nutrient loading rates from the surrounding basin, the atmosphere, and the adjacent Potomac River for several time periods and compare these with other estuarine systems. We then examine time series data sets of water quality and habitat condition, largely from 1986 to 2010. Using both local information and

literature sources a nitrogen budget was developed which placed nutrient sources and sinks in perspective, an exercise useful for future nutrient management decisions. Finally, we develop a "cause–effect" chain relating nutrient loads to algal biomass, water clarity, and SAV community status using a comparative approach wherein data from other small, shallow, estuarine systems are combined in order to develop robust relationships among variables and test the generality of results (Kemp and Boynton 2012).

Mattawoman Creek Watershed and Estuary

The Mattawoman Creek watershed encompasses 245 km$^2$ of land, 7.4 km$^2$ of open tidal waters, and 2.5 km$^2$ of wetlands; intertidal area is very small (Fig. 1). The watershed to estuarine surface area ratio is about 33, a value higher than 60 % of USA estuarine systems, and much higher than the Chesapeake Bay system which has a ratio of 14 (Bricker *et al*. 1999). The significance of this ratio (often called a dilution ratio) is a qualitative index of the potential influence of adjacent land on receiving waters. The high ratio for Mattawoman Creek indicates an elevated potential for pollution effects. The shallow nature of this system further exacerbates this effect because there is not much water to dilute the effects of land-derived nutrients, sediments, or other pollutants.

The dominant land use in Mattawoman Creek basin in 2010 was forested lands (54 %); agricultural land uses accounted for 9.3 % of watershed land uses (Table 1). Urban, suburban, and other developed land uses occupied 35 % of the basin land area. Between 1973 and 2010, urban lands increased by about a factor of three (12 to 35 %) while agricultural and forested lands both decreased. Changes in barren land and wetlands were very small. Estimates of impervious surfaces in the watershed increased linearly from less than 2 % in 1950 to 5 % by mid 1980 (MDNR 2012; MDP 2012). The rate of change of impervious surface cover increased during the mid 1980s and by 2010 was just over 10 %. As a rule of thumb, small basins with impervious cover greater than 10 % often have impaired waterways (Schueler 1994; Allan 2004; Holland *et al*. 2004).

Mattawoman Creek is typical in size and volume to many of the small tributaries of Chesapeake Bay and the Potomac River estuary (Cronin and Pritchard 1975). Mattawoman Creek is about 50 km in total length; the lower 20 km are tidal (Fig. 1). The upper portion of the tidal estuary is narrow and meandering (25–100 m wide) and turbid. The lower portion of the creek is much wider (1–3 km), deeper (mean depth ~1.5 m), clearer, and vertically well-mixed most of the time. The surface area and volume of the tidal estuary is $7.4 \times 10^6$ m$^2$ and $10 \times 10^6$ m$^3$, respectively. SAV are currently a prominent feature of this system covering about 3.5 km$^2$ of estuarine bottom area in 2010 (~47 % of creek surface area).
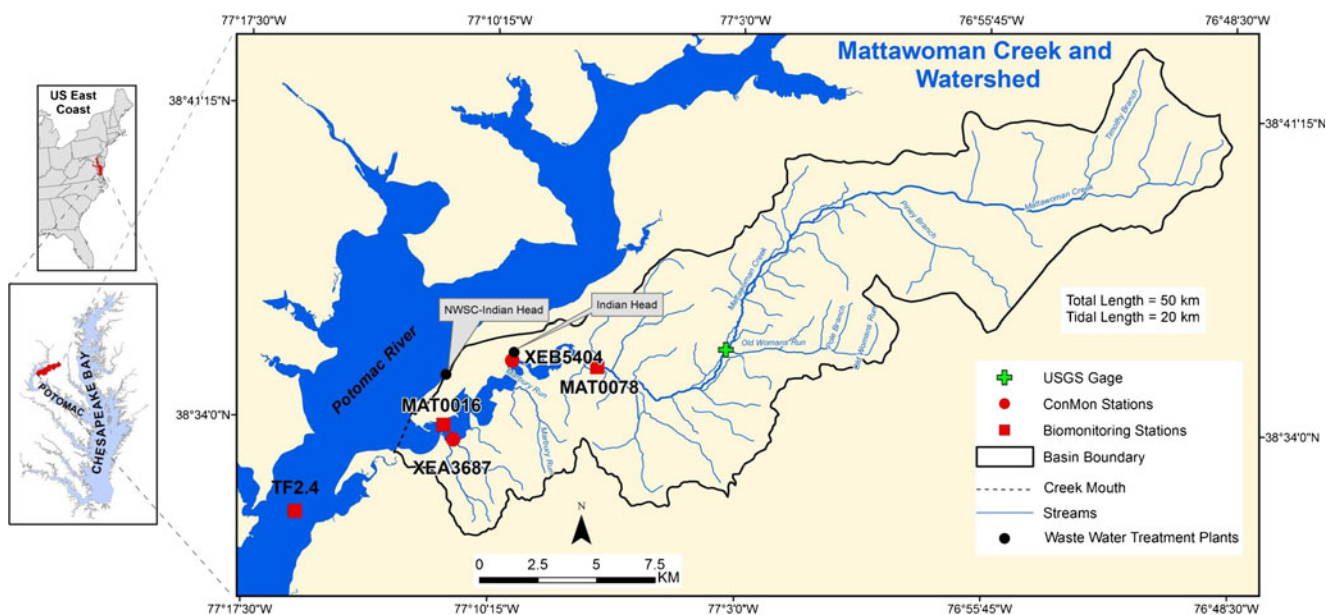
**Fig. 1** A map of Mattawoman Creek and watershed showing locations of stream network, water quality sampling sites, location of USGS flow gauge, and the cross-section of the creek mouth (*dashed line*) where net nutrient fluxes were estimated. The NWSC-Indian Head WWTP facility discharges into the Potomac and the Indian Head facility was upgraded with new nutrient removal technologies several times. *Insets* show the location of Chesapeake Bay on the east coast of the USA and the location of Mattawoman Creek on the Potomac River estuary

## Data Sources and Analytical Approaches

All data used in this analysis are listed in Table 2. Concise descriptions of variables, information regarding sampling sites, period of the data record, measurement frequency, and analytical technique used are also provided, as are references to data sets and more detailed descriptions of methodologies. All water quality data were averaged to a monthly or annual basis even though some data were available on a bi-weekly basis. Differences between surface and bottom water concentrations were examined; differences were negligible at all sites during all seasons so surface water samples were used in this analysis.

In addition to monthly water quality sampling, two high-frequency monitoring sites (Maryland Dept. Natural resources ConMon Program) were also established (Fig. 1) and these provided water quality measurements at 15-min intervals from April to October from 2004 to 2010. Data collected included temperature, salinity, pH, water clarity (as NTUs), dissolved oxygen, and chlorophyll-*a* concentration. High-frequency water quality data (ConMon Program; Table 2) are ideal for computing rates of community production (photosynthesis) which is a basic property of all ecosystems. We adapted the Odum and Hoskin (1958) approach to computing community metabolism and adopted air–water dissolved oxygen flux corrections as suggested by Caffrey (2004). In brief, community production is inferred from the daytime increase in DO concentration. Community production rates are corrected for oxygen diffusion between the water and atmosphere which, in turn, is governed by water temperature and salinity effects on dissolved oxygen saturation in water.

The phase 5.3 Chesapeake Bay Watershed Model is an application of the Hydrologic Simulation Program-Fortran (HSPF; Bicknell *et al.* 2005; Linker *et al.* 2008). The segmentation scheme divides the Chesapeake Bay watershed into more than 1,000 segments/subbasins (including Mattawoman Creek basin), uses 280 monitoring stations throughout the Bay watershed for calibration of hydrology and 200 monitoring stations to calibrate water quality. The model simulates on a 1-h time step and we used output on an annual basis. Nutrient

**Table 1** A summary of land use/land cover in the Mattawoman Creek watershed for three time periods (1973, 2002, and 2010)

| Land use/land cover type | 1973 | | 2002 | | 2010 | |
|---|---|---|---|---|---|---|
| | ha | % | ha | % | ha | % |
| Agriculture | 3,951 | 16.2 | 2,901 | 11.9 | 2,280 | 9.3 |
| Barren | 0 | 0 | 48 | 0.2 | 243 | 1.0 |
| Forest | 17,193 | 70.4 | 14,477 | 59.2 | 13,142 | 53.8 |
| Urban | 3,053 | 12.5 | 6,672 | 27.3 | 8,447 | 34.6 |
| Water | 69 | 0.3 | 88 | 0.4 | 87 | 0.4 |
| Wetlands | 184 | 0.6 | 263 | 1.1 | 252 | 1.0 |
| Total area | 24,450 | 100 | 24,449 | 100 | 24,451 | 100 |

Areas are in hectares (ha) and numbers in parentheses represent percent of cover by category for each time period. Data are from Maryland Department of Planning (2012)

**Table 2** A summary of the types of data used in this analysis, sampling locations, period of data record, measurement frequency, analytical methods used, and data sources

| Data types | Location or sampling sites | Period of record | Measurement frequency | Approach/technique | Data sources/technique details |
|---|---|---|---|---|---|
| **System-scale data** | | | | | |
| Land uses | Full watershed; 6 land covers | 1973, 2002, 2010 | Selected years | Aerial photos; GIS | Maryland Dept Planning (2012) |
| Impervious surfaces | Full watershed | 1950–2010 | Mainly annual | Aerial photos; GIS | Maryland Dept. Planning (2012) |
| Creek dimensions | NA | 1975 | NA | Bathemetric surveys | Cronin and Pritchard 1975 |
| Freshwater input | One; see Fig. 1 | 2005–2011 | Daily | USGS gauge site | USGS (2011) |
| **Nutrient loads** | | | | | |
| Measured | One; see Fig. 1 | 2005–2011 | Monthly average | USGS standard computation | USGS (2011) |
| Model estimates | Full watershed | 1985, 2002, 2010 | Monthly average | Ches. Bay Prog. HSPF model | Shenk and Linker (2013); Linker et al. 2000 |
| Point source | Several discharges | 1986–2010 | Monthly average | Grab samples | Ches. Bay Program (2011) |
| Atmos deposition | Surface area of creek | 1984–1999 | Annual | NADP and local data collection | Boynton et al. 2008 (Table 2) |
| **Net nutrient exchange with Potomac River** | | | | | |
| | Mouth of creek; see Fig. 1 | 1991–2000 | Monthly average | Ches. Bay Prog. water quality model | Cerco et al. 2010 and Linker (personal communication) |
| **Water quality data** | | | | | |
| Nutrient conc | 3; see Fig. 1 | 1986–2011 | 1 or 2 per month | Standard chemical techniques | Ches. Bay Water Quality Monitoring Program (2012) |
| Chlorophyll-a | 3; see Fig. 1 | 1986–2011 | 1 or 2 per month | " | Nutrient Analytical Services Lab (2012) |
| Secchi depth | 3; see Fig. 1 | 1986–2011 | 1 or 2 per month | " | As above for nutrients and chlorophyll-a |
| High frequency data | 2; see Fig. 1 | 2004–2010 | 15 min; Apr–Oct | In situ Sondes; 1 m depth | Maryland Department of Natural Resources (2012) |
| **Habitat data** | | | | | |
| SAV coverage | Full creek area | 1994–2010 | Annual | Aerial photographs; GIS | Virginia Institute of Marine Sciences (2011) |
| **DO concentration** | | | | | |
| Monthly | 2; see Fig. 1 | 1986–2011 | 1 or 2 per month | Sonde; water column profiles | Ches. Bay Water Quality Monitoring Program (2012) |
| High frequency | 2; see Fig. 1 | 2004–2010 | 15 min; Apr–Oct | In situ Sondes; 1 m depth | Maryland Dept of Natural Resources (2012) ConMon Program |
| **Nitrogen rate data** | | | | | |
| Denitrification | Variety of shallow Ches Bay sites | 2005–2012 | Month–season | $N_2$-argon technique | Greene (2005a, b); Boynton et al. (2009); Gao et al. (2012); Cornwell (personal communication) |
| Long-term N burial | Variety of shallow Ches Bay sites | 1999–2005 | Annual | Pb-210 dating; PN analysis of sediment cores | Greene (2005a); Merrill (1999) |
| Sediment N flux | Variety of shallow Ches Bay sites | 1986–2007 | Month–season | Shipboard incubation of intact cores | Bailey et al. (2005); http://www.gonzo.cbl.umces.edu/data.htm |

Additional details are contained in the text

input loads are from atmospheric deposition, fertilizers and manures, and other smaller sources. Municipal and industrial wastewater treatment and discharging facilities and onsite wastewater treatment system (septic system) nitrogen, phosphorus, and sediment contributions are also included in the model (USEPA 2010).

The Bay Water Quality Model combines a three-dimensional hydrologic transport model (CH3D) with a eutrophication model (CE-QUAL-ICM) to predict water quality conditions in the Bay resulting from changes in loads from the contributing basin areas. The hydrodynamic model computes transport using a three-dimensional grid framework (Cerco et al. 2010). The

hydrodynamic model was calibrated for the period 1991–2000 and verified against the large number of observed tidal elevations, currents, and densities available for the Bay. The eutrophication (water quality) model computes algal biomass, nutrient concentration, nutrient cycling rate, and DO concentration and other constituents and processes using a 15-min time step (Cerco and Noel 2004). The model also incorporates a sediment diagenesis component which simulates the chemical and biological processes at the sediment–water interface (DiToro 2001). We used estimated net flux of N and P compounds across the mouth of Mattawoman Creek at monthly time scales for the period 1992–2000. Detailed documentation of the Chesapeake Bay Water Quality and Sediment Transport Model can be found at http://www.chesapeakebay.net/content/publications/cbp_26167.pdf.

We did not have local measurements of denitrification or long-term burial rates of particulate nitrogen (PN) and particulate phosphorus (PP). However, for the purposes of the preliminary nutrient budget that was developed, we used an average of denitrification values measured in similar shallow water, nitrate-rich tributaries of Chesapeake Bay that also had oxidized surface sediments likely to support active nitrification. Burial estimates were from similar environments and we chose to use a value at the conservative end of Chesapeake Bay values currently available (Table 2).

For this analysis, we used the correlation and linear regression analysis package available in MatLab. Where appropriate, we manually examined various time scales (monthly, seasonal, and annual; with and without lags) for exploring relationships between nutrient loading rates and water quality and habitat conditions.

## Results and Discussion

### Current and Historical Nutrient Sources

The USGS maintained a gauging station in the Mattawoman watershed from 2005 to 2011. This site monitored water, nutrient, and sediment discharges from 59 % of the basin land area. Water flow and TN and TP loads varied seasonally as well as inter-annually (Fig. 2). During 4 of the 6 years of record, flow and loads were highest during winter–spring and much lower during summer and fall, a pattern typical of other tributaries of the Chesapeake Bay (Boynton et al. 2008). However, during the spring and fall of 2006 and fall of 2011 tropical storms (Alberto, Ernesto, and Irene, respectively) passed through the area, and flow and loads exhibited large but temporary increases. On an inter-annual basis, diffuse N loads varied by a factor of two (180–343 kg N day$^{-1}$) and P loads by just over a factor of two (23.5 to 49.7 kg P day$^{-1}$). This scale of inter-annual variability has been reported for other Chesapeake Bay systems (Hagy et al. 2004). Direct

inputs of N to tidal waters from groundwater were not directly evaluated. However, groundwater N inputs were included in the USGS estimates of loads from 59 % of the drainage basin located above the gauge (Fig. 1). We prorated these loads to the entire basin and, because of this, we have included an estimate of groundwater inputs to the tidal portion of the basin.

Diffuse source loads were also estimated based on the Chesapeake Bay Program land use model and those estimates (1985, 2002, and 2010) were similar to those derived from the USGS stream monitoring data and they served to extend temporal coverage of diffuse inputs to this system (Table 3). Based on both data sets it does not appear that diffuse loads have changed much between 1985 and the present time.

The major change in nutrient input to Mattawoman Creek was related to point source reductions. During 1990 point source loads were about 360 kg N day$^{-1}$ and were a much larger source than diffuse loads. Point source loads declined very sharply to about 50 kg N day$^{-1}$ by 1995 and then decreased again beginning in 2000. Point source loads have been very low since then and now represent a small fraction of the total nutrient load to the system.

Direct deposition of N to the surface waters of Mattawoman Creek represents another nutrient source. We used atmospheric deposition data from Boynton et al. (2008) that included all forms of N in both wet and dry deposition (0.81 mg N L$^{-1}$ as an annual average concentration). Given precipitation averages about 1 m year$^{-1}$, direct atmospheric deposition to surface waters of the creek contributed about 6,000 kg N year$^{-1}$ or about 16 kg N day$^{-1}$ to the creek system. Thus, direct N deposition is a small component of the N budget for this system. However, rain (and dry deposition) falls on the full basin and all this rain contains N compounds. In the Chesapeake Bay basin, Fisher and Oppenheimer (1991) and more recently Castro et al. (2003) estimated that about 25 and 22 %, respectively, of atmospheric N deposition to the landscape is exported to streams and estuarine waters. While direct measurements are not available for the Mattawoman basin, applying the most recent estimate of 22 % suggests that about 120 kg N day$^{-1}$ would reach estuarine waters as a component of diffuse source loading, or about 49 % of the total diffuse source load. In this larger view, atmospheric deposition is a very important part of the N input signature for this system. If this estimate proves to be correct, continuing emphasis on decreasing atmospheric deposition of N is an important management objective and one where there has been progress on a regional scale during the last 20 years (Burns et al. 2011).

One useful nutrient enrichment metric to consider is TN and TP loading rates to Mattawoman Creek compared with those of other estuarine ecosystems. To compare nutrient loading in this system to loading at other coastal and estuarine locations, we compiled N and P loading rates for many such systems and added Mattawoman Creek data for several time periods (Fig. 3). Several points are clear and include the

**Fig. 2** A time series of TN, TP, and water flows based on data collected at the USGS gauge on Mattawoman Creek (USGS 016558000; Fig. 1). There were some gaps in the load record and these were estimated using a linear flow–load relationship developed with these data. The intensity of measurements used to develop load estimates is also shown in the figure. Data were from the US Geological Survey (2011)
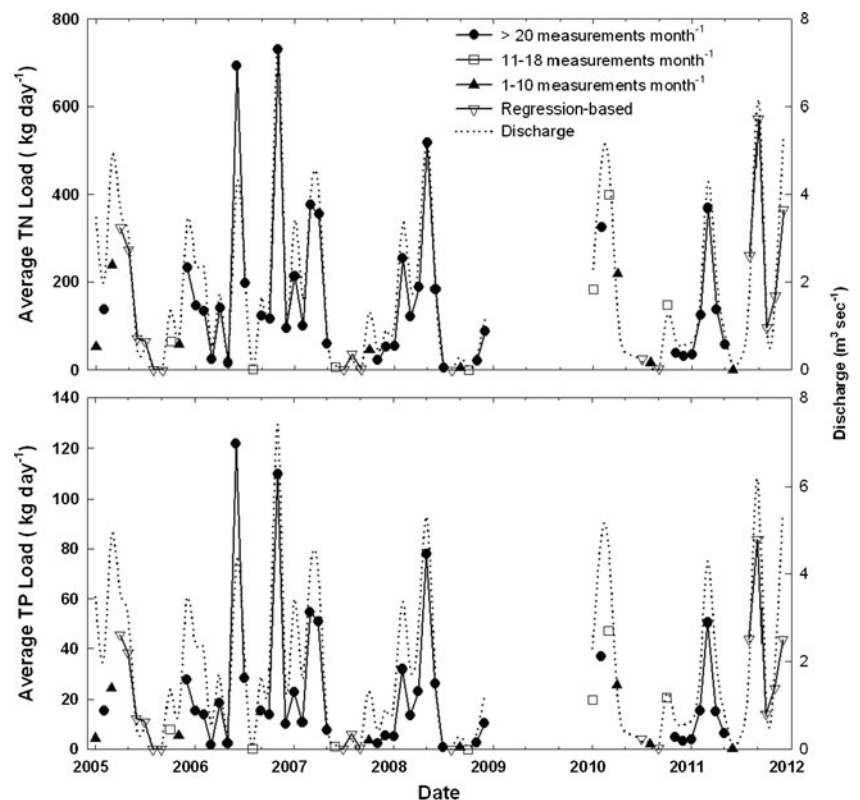


following: (1) N and P loads prior to wastewater treatment plant (WWTP) modifications were higher than at present but not, even prior to WWTP modifications, as high as they are in very heavily loaded systems; (2) there was a significant decline in N and P loading rates associated with WWTP modifications beginning in the early 1990s (TN and TP loads from all sources decreased by factors of factors of 1.9 and 2.4, respectively); (3) diffuse N and P loads exhibited considerable inter-annual variability related to wet and dry years; (4) loading rate estimates from gauges and from models agreed quite well in this system; (5) loading rates for the completely forested watershed (with no atmospheric deposition) were

**Table 3** Multiple estimates of annual diffuse source total nitrogen (TN) and total phosphorus (TP) loads to Mattawoman Creek

| Data source/condition | Year | Annual TN load (Kg N day$^{-1}$) | Annual TP load (Kg P day$^{-1}$) | Reference |
|---|---|---|---|---|
| CBP landscape model estimates | 1985 | 260 | 39.2 | Linker *et al.* 2000 |
| | 2002 | 251 | 33.0 | and G. Shenk( personal communication) |
| | 2010 | 233 | 26.1 | |
| USGS river input monitoring | 2005 | 216 | 27.3 | USGS (2011) |
| | 2006 | 343 | 49.7 | |
| | 2007 | 180 | 23.5 | |
| | 2008 | 204 | 27.2 | |
| | 2009 | No data | No data | |
| | 2010 | 204 | 24.1 | |
| Pristine forested basin | Pre-European settlement | 100 | 2.7 | Boynton *et al.* 1995 |

Direct atmospheric deposition of N to surface waters of the creek was included in the diffuse source estimates. TN and TP exchange with the Potomac River estuary are not included here, but are considered later. USGS data for the gauged portion of the watershed were scaled up to the full watershed using a linear ratio of gauged to non-gauged areas be comparable with data from the Chesapeake Bay Program model estimates (USGS (2011)). Chesapeake Bay Program estimates were from G. Shenk (personal communication). Estimate of TN and TP loads from a pristine forested basin with no atmospheric deposition of N or P were based on basin area and used forest yield coefficients of 0.15 g N m$^{-2}$ year$^{-1}$ and 0.004 g P m$^{-2}$ year$^{-1}$ (Boynton *et al.* 1995)
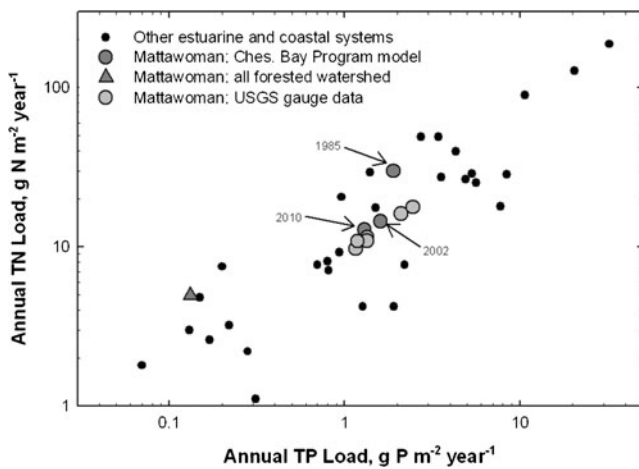
Fig. 3 Scatter plot of annual TP versus TN loads for a variety of estuarine and coastal marine ecosystem (*small gray circles*; see Boynton *et al.* 1995 for references for these sites). TP and TN loads for Mattawoman Creek were from several sources including the Chesapeake Bay Program land use model (1985, 2002, and 2010) and the USGS gauge data (2005–2011). An estimate of TN and TP loads from a fully forested pristine basin with no atmospheric deposition of N or P is also shown and was developed using forest yield coefficients of 0.15 g N $m^{-2}$ $year^{-1}$ and 0.004 g P $m^{-2}$ $year^{-1}$ (Boynton *et al.* 1995)

about half what they are now during dry years and about four times lower than in recent wet years.

Nutrient Exchanges with the Potomac River

The final component of this evaluation of nutrient inputs to Mattawoman Creek concerned nutrient exchanges with the adjacent Potomac River estuary. These systems are connected via tidal water transport between the creek and Potomac River. These processes vary in magnitude on many time scales (hourly to inter-annual) and are also influenced by local and larger storm events.

In several previous studies of Chesapeake Bay tributaries, we used salt and water box model results, coupled to nutrient concentrations, to estimate net nutrient flux into or out of these small estuarine systems (e.g., Hagy *et al.* 2000; Boynton *et al.* 2009; Boynton *et al.* 2011). However, there is rarely any measurable salinity in Mattawoman Creek, rendering that approach impossible due to the lack of a conservative tracer. To estimate Potomac–Mattawoman nutrient exchange, we obtained output from the Chesapeake Bay Program water quality model for net monthly N and P flux across the mouth of Mattawoman Creek for the period 1991–2000 (Cerco and Noel 2004). Model results indicated some DIN net transport from Mattawoman Creek to the Potomac during winter or spring and the opposite during summer–fall (Fig. 4). Averaged over all years, the net DIN flux was about 102 kg N $day^{-1}$ directed into Mattawoman Creek from the Potomac River. We also had estimates of TN flux and the average multi-year flux was very small (0.4 kg N $day^{-1}$) and was directed from Mattawoman Creek to the Potomac River.
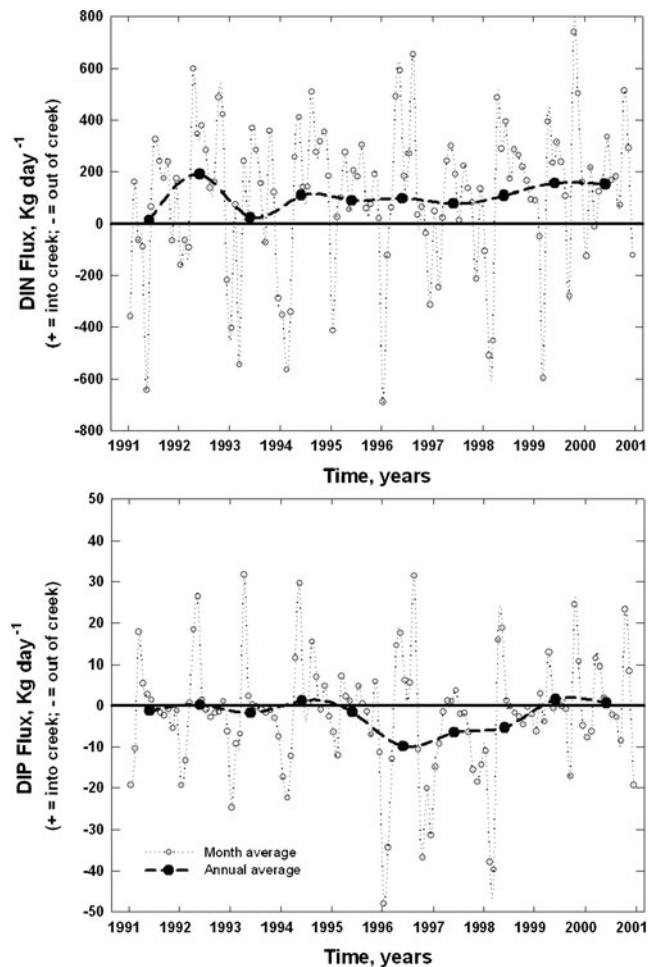


Fig. 4 A time series of monthly and annual net DIN and DIP exchanges between Mattawoman Creek and the Potomac River estuary for the period 1991–2001. The decade-long annual average exchange rates were 102 Kg N $day^{-1}$ (into the creek) and −2.3 kg P $day^{-1}$ (out of the creek). These estimates were generated from the Chesapeake Bay Program water quality model (Cerco *et al.* 2010; G. Shenk, personal communication)

Dissolved inorganic phosphorus (DIP) appeared to be exported from the creek during winter and imported from the Potomac during summer; the multi-year average was an export from the creek of 2.3 kg P $day^{-1}$ (Fig. 4). These results suggest Mattawoman Creek consumes DIN but exports little TN and seasonally imports or exports small amounts of DIP. The creek system acts as a sink for Potomac River N and both a small source and sink for Potomac river DIP.

Water Quality Patterns and Trends

*Nutrient Concentrations* Nitrate plus nitrite ($NO_{23}$) and phosphate ($PO_4$) are essential plant nutrients, the excessive supply of which is often a root cause of estuarine eutrophication. Concentrations of $NO_{23}$ ranged from 0.003 to about 3 mg $L^{-1}$ and were uniformly higher at the downstream site than at the upstream site throughout the period of record (Fig. 5). This sharply contrasts with most estuarine sites wherein nutrient

**Fig. 5** Annual average time series data for water quality variables measured in surface waters at two sites in Mattawoman Creek (MAT 0078 and MAT0016) and one site in the adjacent Potomac River (TF2.4) for the period 1986–2010. Data are from Chesapeake Bay Water Quality Monitoring Program (2012)
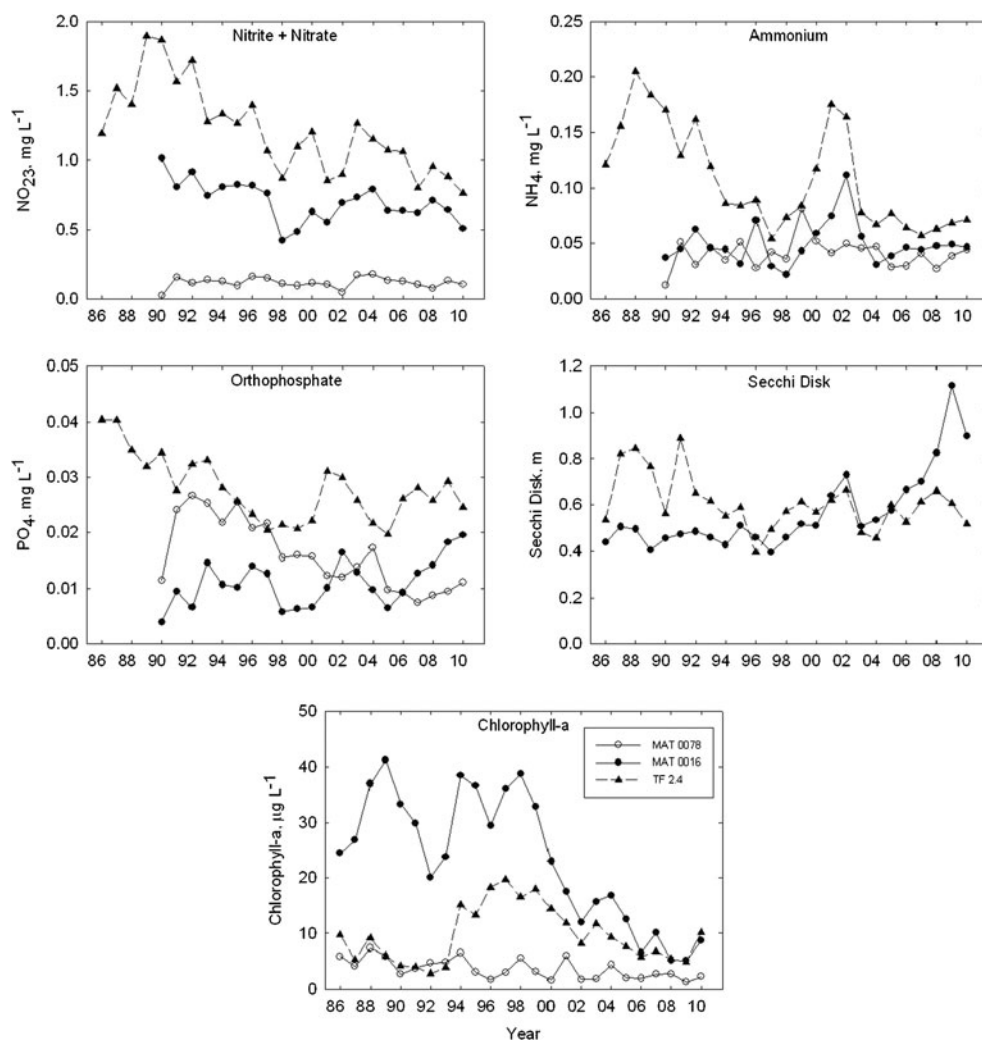
concentrations decrease with distance from riverine (upstream) sources (Boynton and Kemp 2008). In Mattawoman Creek, the higher $NO_{23}$ concentrations at the downstream site likely reflect proximity to the Potomac River which has elevated $NO_{23}$ and $NH_4$ concentrations for much of the year. Highest $NO_{23}$ concentrations occurred during winter–spring, coincident with periods of high Potomac and local river flow. Concentrations were at times 2 orders of magnitude lower during the warm periods of the year coincident with rapid SAV and phytoplankton biomass accumulation and periods of the year when denitrification rates were also likely highest (Greene 2005b). During summer periods $NO_{23}$ concentrations were frequently below N half-saturation ($k_s$) values for estuarine phytoplankton (<0.035 mg $L^{-1}$; Parsons *et al.* 1984; Sarthou *et al.* 2005) but the frequency of low values did not increase after WWTP modifications in 1996. $NO_{23}$ concentrations at the downstream site and at the Potomac River site have decreased over time (0.014 and 0.03 mg N $L^{-1}$ year$^{-1}$, respectively) possibly as a result of Potomac River and Mattawoman Creek WWTP modifications. No trends in $NO_{23}$ concentration were evident at the

upstream site. Ammonium concentrations were generally an order of magnitude lower than $NO_{23}$ concentrations and were always higher in the Potomac than in Mattawoman Creek and did not exhibit strong temporal patterns at either sampling site in Mattawoman Creek (Fig. 5).

The time series of $PO_4$ concentrations in Mattawoman Creek indicate a complex pattern (Fig. 5). Concentrations ranged from 0.005 to 0.08 mg $L^{-1}$ at the upstream site and from about 0.002 to 0.06 mg $L^{-1}$ at the downstream site. These are typical values for a low-salinity estuarine ecosystem (Boynton and Kemp 2008). In this case, $PO_4$ concentrations were higher at the upstream site, as expected, during the early portion of the record (1991–2004) and then declined to levels lower than those at the downstream site. Since 2005, $PO_4$ concentrations at the downstream site have been increasing, possibly because of seasonal N limitation. TP concentrations were generally similar between upstream and downstream sites for the period of record and ranged from the level of detection ($\sim$0.01 mg $L^{-1}$) to about 0.3 mg $L^{-1}$. Highest TP values consistently occurred during the warmer portions of the year, a pattern frequently observed

in shallow estuarine environments (Boynton and Kemp 2008), and caused by active sediment releases of P at a time of the year when autotrophic growth is limited by N.

*Water Clarity* Secchi disk data are only available for the downstream station for the period 1986–2010 (Fig. 5). Measurements ranged from about 0.2 to 2.7 m. There was a clear trend in water clarity with values of about 0.5 m early in the record and then increasing sharply after 2004 to an annual average of about 1.1 m during 2009. Water clarity is a key issue regulating SAV community health. In the adjacent Potomac, Ruhl and Rybicki (2010) reported strong correlations between water clarity and SAV community density, coverage, and species composition. At those sites, Secchi values in excess of 0.65 m were associated with bed expansion, increased plant density, and a return of native species. The measurements reported here were made at sites along the main channel of Mattawoman creek rather than in SAV beds. It may be that these values underestimate water clarity in the SAV beds as shown by Gruber and Kemp (2010) based on detailed water clarity and other measurements inside and outside SAV beds in the mesohaline Chesapeake Bay. Conversely, measurements in the channel might also be higher than normal because SAV beds line much of the shoreline and tend to suppress shoreline erosion, sediment resuspension, and efficiently trap sediments (Ward *et al.* 1984).

*Algal Biomass* Chlorophyll-*a* concentrations varied between 0.3 and 110 μg L$^{-1}$ at the downstream site and from 0.15 to 30 μg L$^{-1}$ at the upstream site (Fig. 5). Typical values at the downstream site were higher, at times an order of magnitude higher, than at the upstream site. It is likely that a combination of limited light and shorter water residence time both contributed to lower algal biomass at the upstream site. There did not appear to be any long-term trend in chlorophyll-*a* concentration at the upstream site. However, there were several distinctive temporal trends at the downstream site. Chlorophyll-*a* concentrations were generally high (annual average concentration 20–40 μg L$^{-1}$) from 1986 to 1998. Concentrations then steadily declined through 2010 to between 5 and 10 μg L$^{-1}$. The decline in algal biomass is likely caused by nutrient load reductions associated with WWTP operations both in Mattawoman Creek and the Potomac River. The general picture of water quality conditions that emerges from these data indicates an increase in water quality associated with changes in WWTP operations in both Mattawoman Creek and in the adjacent Potomac River. Water column pH (not shown), $NO_{23}$, $PO_4$, and chlorophyll-*a* concentration all declined and water clarity and SAV community metrics increased.

Community Production

We did not have high-frequency water quality data for the period prior to WWTP load reductions so it was not possible to compare community production rates in the creek before and after load reductions. However, we did have estimates of community gross production (but not for total community respiration) from a variety of shallow Chesapeake Bay sites. Community gross photosynthesis (Pg) rates in Mattawoman Creek ranged from about 2 to 11 g $O_2$m$^{-3}$ day$^{-1}$ (Fig. 6). Rates were lower during spring (Apr–May) and fall (Sep–Oct) and highest during Jun–Aug, particularly during July. Summer average rates were relatively low during 2004–2005, increased during 2006, and then declined through 2010. To place these in perspective we compared Pg rates for a variety of Chesapeake Bay systems ranging from very nutrient enriched to less enriched (Table 4). In general, rates were proportional to one index of enrichment (chlorophyll-*a* concentration; Pg=5.8+0.15 chlorophyll-*a*; $p<0.05$; $n=10$; $r^2=0.55$) as Caffrey (2004) reported earlier. Rates in Mattawoman Creek tended to be low compared with rates measured in heavily enriched (e.g., upper Potomac and Corsica Rivers) ecosystems. These results are consistent with several observations: nutrient loading rates to Mattawoman Creek were sharply reduced during the time period when these measurements were made; nutrient and chlorophyll-*a* concentrations also decreased and SAV became abundant in Mattawoman Creek and such communities are not usually associated with heavily enriched systems (Latimer and Rego 2010; Orth *et al.* 2010). Modest Pg rates support the idea that considerable oligotrophication of this system has occurred.

SAV in Mattawoman Creek

The resurgence of SAV in Mattawoman Creek represents one of a limited number of restoration successes in the Chesapeake Bay region (Orth *et al.* 2010). It appears that substantial
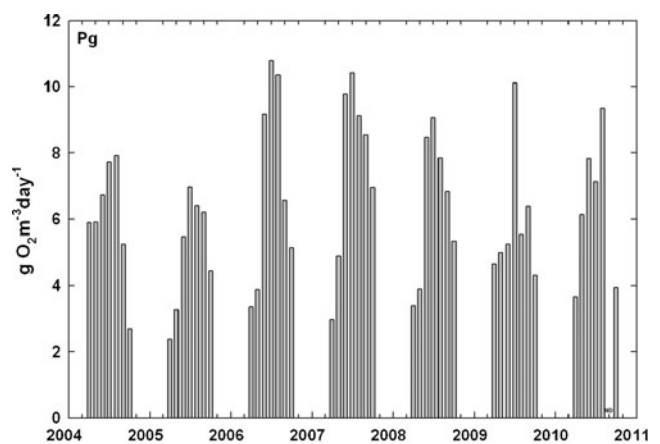


Fig. 6 Mean monthly (Apr–Oct) estimates of community gross primary production (*Pg*; g $O_2$m$^{-3}$ day$^{-1}$) for the period 2004–2010. These estimates were generated following the technique of Odum and Hoskin (1958). Data used in these computations were from ConMon site XEA3687 in Mattawoman Creek (Fig. 1). Data used in these computations are available at Maryland Department of Natural Resources (2012)

**Table 4** A selection of community gross primary production rates from very enriched and less enriched Chesapeake Bay tributary sites

Since estimates of nutrient loads were not available for all sites for summer seasons (June–August) chlorophyll-*a* concentration was used as an indicator of nutrient enrichment. Details of the method for computing oxygen-based production are given in Hodgkins *et al.* (2012). Data for these estimates were from Maryland Department of Natural Resources (2012)

| Nutrient enrichment status | System or location | Summer average gross primary production (G $O_2 m^{-3}$ day$^{-1}$) | Summer average chlorophyll-*a* |
|---|---|---|---|
| Very enriched | Bishopville (MD Cstl Bays) | 17.0 | 70.2 |
| | Turville Ck (MD Cstl Bays) | 13.0 | 28.4 |
| | Piscataway Ck (Upper Potomac) | 16.0 | 28.8 |
| | Upper Corsica River | 12.3 | 45.7 |
| | Back River | 14.3 | 60.0 |
| | Average | 14.5 | 46.6 |
| Less enriched | St. Georges Ck (Lower Potomac) | 7.3 | 5.9 |
| | Stonington (Magothy) | 7.5 | 23.5 |
| | Mattawoman (Upper Potomac) | 8.1 | 8.0 |
| | Betterton Beach (Sassafras) | 4.8 | 29.0 |
| | Piney Pt (Lower Potomac) | 5.0 | 10.3 |
| | Average | 6.5 | 15.3 |

nutrient reductions from point sources within Mattawoman Creek and the mainstem Potomac initiated a cascade of events leading to water quality conditions supportive of SAV growth (Fig. 7). Before 1977, SAV were absent from the creek system. Beginning in 1989, SAV reappeared and covered a small percentage of creek bottom area (~5 %) through 1997. After 1997, there was a very rapid increase in SAV coverage and beds were quite dense. By 2002, SAV beds covered about 40–50 % of the surface area of the creek and have become an important component of this tributary system. The spatial pattern of SAV community recovery was also distinctive. Beginning in 1996, SAV appeared in the upper portions of the creek and began to extend downstream through 2000. By 2002, SAV had spread along both the north and south shores to the creek mouth. In more recent years (2005–2010), SAV has extended to deeper water along both shores of the creek. This pattern of resurgence, beginning in the upstream areas of the creek, is similar to the pattern observed in other shallow,

low-salinity Chesapeake Bay tributaries (Boynton *et al.* 2011). It may be that these areas are re-colonized first because they are proximal to seed and other vegetative propagules surviving in small streams of the watershed.

The emerging understanding of SAV resurgence seems to be related to a chain of cause–effect events. It appears resurgence follows nutrient input reductions. In some cases, P seems to be the key element (e.g., Gunston Cove, a Potomac tributary; C. Jones, personal communication) and resurgence is preceded by a considerable lag period likely caused by the effects of excess P slowly purging from estuarine sediments. In other cases, there appears to be minimal lag and N seems to be the key element (e.g., upper Patuxent; Boynton *et al.* 2011). Algal biomass declined and water clarity increased as nutrient inputs to Mattawoman Creek declined. We examined the Mattawoman data set for possible threshold responses relative to SAV resurgence (Fig. 8). The clearest of these appears to be related to water column chlorophyll-*a* concentration. When

**Fig. 7** Annual summary of SAV coverage (ha), water clarity (Secchi disk depth) and algal biomass (chlorophyll-*a* concentration) for the period 1986–2010 in Mattawoman Creek. Note the large change in SAV coverage and water clarity associated with the large decline in algal biomass. All data sources have been previously described
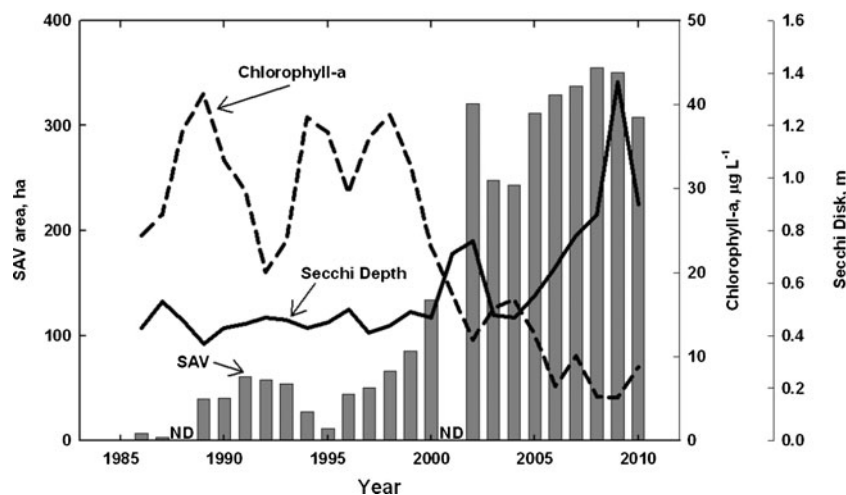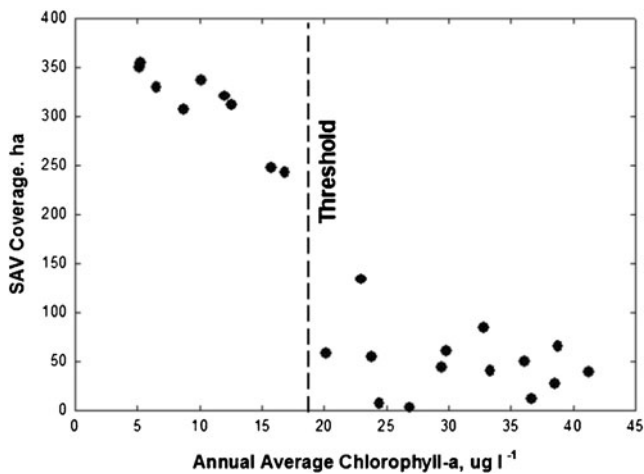
**Fig. 8** Scatter plots of average annual chlorophyll-*a* concentration versus SAV coverage for Mattawoman Creek. Data indicate a large change in SAV coverage associated with a chlorophyll-*a* threshold of about 18 μg L$^{-1}$ and Secchi disk depth of about 0.5 m (not shown). Data sources have been previously noted

annual average chlorophyll-*a* concentration was in excess of about 18 μg L$^{-1}$, SAV coverage was either close to zero or minimal. In contrast, when chlorophyll-*a* concentration dropped below 18 μg L$^{-1}$ SAV coverage expanded very quickly; below this chlorophyll-*a* "threshold value", some other factor or factors apparently regulate inter-annual variability of SAV coverage. There was also some indication of threshold behavior related to water clarity where SAV coverage increased when Secchi disk depths exceeded about 0.5 m. Ruhl and Rybicki (2010) reported a similar response in the adjacent tidal freshwater Potomac River although the "critical" Secchi disk depth was slightly higher (0.65 m). There now appear to be a number of cases in the Chesapeake system (in both small and large low-salinity regions) where nutrient load reductions were followed by SAV resurgence and rapid bed expansion. It still remains uncertain what factors regulate lag times (when they occur) and under what conditions N or P load reductions might be the key element initiating the resurgence process.

A general understanding is emerging concerning relationships between nutrient loading rates and SAV community health (e.g., Kemp *et al.* 2005). In general, it is thought that SAV communities are not competitive in environments having large nutrient loads (e.g., Valiela *et al.* 1997). Orth *et al.* (2010) have shown that SAV resurgence in several areas of Chesapeake Bay was related to decreased N loading. In Mattawoman Creek, SAV were largely absent when N loading rates were in the range of 30 g N m$^{-2}$ year$^{-1}$. When loading rates decreased to about 10–12 g N m$^{-2}$ year$^{-1}$, SAV recolonized the creek system. In addition, Latimer and Rego (2010) examined many SAV communities in southern New England for relationships to N loading rates and found SAV to be healthy when loading rates were about 5 g N m$^{-2}$ year$^{-1}$,

less robust when loading rates were about 10 g N m$^{-2}$ year$^{-1}$, and generally absent when loads exceeded this amount. Mattawoman Creek loads are in the upper portion of the range of "SAV-friendly" loads reported by Latimer and Rego (2010).

Nitrogen Budget for Mattawoman Creek

A nitrogen budget for Mattawoman Creek is provided in Fig. 9; red circles indicate external N sources to the ecosystem, red arrows represent nitrogen inputs to and exports from the system, pink arrows represent estimated internal losses, and gray arrows represent animal migrations that we did not attempt to quantify. In addition, water column, sediment, and SAV N storages are indicated (but not evaluated) as are two internal nutrient pathways (sediment N re-cycling and net SAV N uptake). This annual time-scale budget assumes (1) completeness (i.e., there are no important missing terms in the budget) and (2) internal storages of N are not substantially changing from year to year. The attraction of a mass balance is as a quantitative framework against which we can test our understanding of system-scale nutrient dynamics (Boynton and Nixon 2012). If the budget balances (within reason), we conclude that all important processes were included and properly evaluated. However, if the budget does not balance then we know we have made an important error or neglected critical processes. Finally, reasonably balanced budgets allow us to separate large from small processes and this is an important step in choosing effective management actions.

External N inputs amounted to 385 kg N day$^{-1}$ and were dominated by diffuse sources (63 %) followed by net inputs from the Potomac River (26 %). Point sources and direct atmospheric deposition of N were small (6 and 4 %, respectively). Prior to WWTP modifications, total N inputs were much larger (737 kg N day$^{-1}$) and point sources were the dominant source (47 %). The two major internal losses include net denitrification and long-term burial of N (mainly particulate organic N) in the accreting sediments of the estuary. Unfortunately, there are no direct measurements of either of the major internal loss rates available for Mattawoman Creek. However, during the last decade there have been an increasing number of these measurements made in shallow estuarine systems and many of these measurements have been summarized by Greene (2005a and 2005b), Pina-Ochoa and Alverez-Cobelas (2006), and Mullholland *et al.* (2008). To make preliminary estimates of net denitrification and long-term N burial we reviewed these values and other local values (Gao *et al.* 2012 and Cornwell, personal communication) and used annual average rates of 47 μmoles N m$^{-2}$ h$^{-1}$ and 6.0 g N m$^{-2}$ year$^{-1}$ for denitrification and long-term burial, respectively.

First, the proposed budget does not balance. TN inputs (385 kg N day$^{-1}$; including DIN inputs from the Potomac) are larger than estimated nitrogen losses (317 kg N day$^{-1}$;
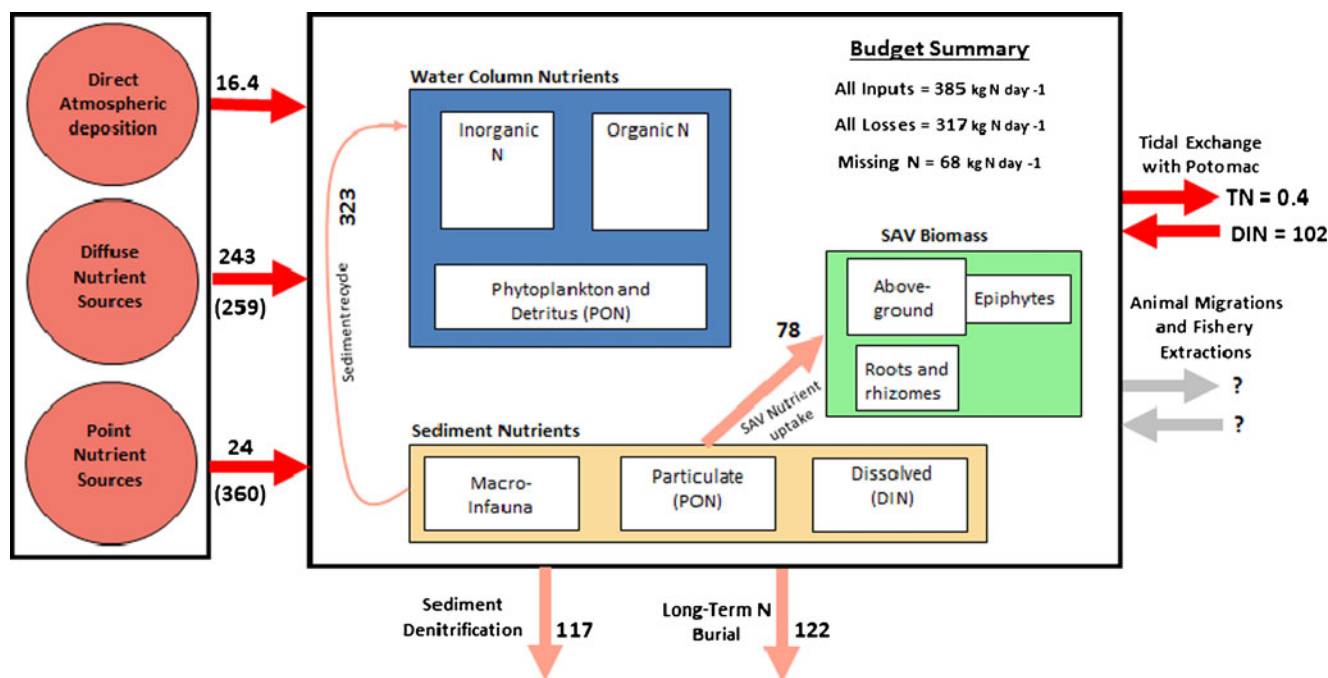
**Fig. 9** A schematic diagram of a nutrient budget (*TN*) model developed for Mattawoman Creek for the 2005–2010 time period. Nutrient sources are shown on the *right* (point, diffuse, and atmospheric) and *left* (exchange with the Potomac River). Internal loss terms are shown at the bottom of the diagram (denitrification, long-term burial) and within the diagram (SAV uptake). One internal nutrient re-cycling process is also shown (sediment recycle). We recognize that SAV uptake of N is a less permanent loss than either burial or denitrification but have included it as a loss term because of the long SAV growing season. Internal stocks were not evaluated because data were not available. *Bright arrows* indicate data specific to Mattawoman Creek were used; *light red arrows* indicate data from the Chesapeake Bay region were used; *gray arrows* indicate no data were available and no estimate was attempted. The *numbers in parentheses* indicate diffuse and point source N loads prior to WWTP modifications

including SAV N uptake) indicating that one or more major processes have not been adequately considered. One likely explanation for this is that we were not able to assign specific denitrification or nutrient burial rates to either the SAV or fringing tidal wetland communities. Direct measurements of these rates in tidal freshwater marshes of the Corsica River yielded rates three times the rates measured in open waters of the Corsica. If we adjusted Mattawoman internal loss rates so that N losses were higher in SAV and fringing tidal marsh communities, the budget readily balances. It may be worth supporting a measurement program to better quantify N losses in these communities. Boynton *et al*. (2008) found fringing tidal marshes to be a very large N and P sink in the tidal freshwater portions of the Patuxent River estuary. The second point is that diffuse sources are the most important nitrogen source. Efforts to further improve water quality will likely fail unless this term is considered and acted on; if this term increases because of changes in land use water quality will likely degrade. Third, the TN export/import term associated with exchanges with the Potomac River needs more examination. At present, model results indicate almost no net exchange of TN between the Potomac and Mattawoman Creek but also indicate a substantial input of DIN, almost all as $NO_{23}$, into the creek from the Potomac. This suggests that the creek acts as an N sink for the Potomac. During most of the year $NO_{23}$

concentration in the Potomac was higher than in the creek so the direction of net transport was largely consistent with model results. Should nutrient concentrations in the Potomac increase further, the magnitude of DIN import to the creek could increase and the creek could become more nutrient enriched. DIN flux from large to smaller systems has already been documented for the Patuxent and Corsica estuaries (both Chesapeake Bay tributaries) in some summer and fall months (Boynton *et al*. 2008). Finally, we were able to add a few internal nutrient-cycling terms to the budget analysis. Uptake of N from sediments and the water column by SAV serves as a seasonal-scale (i.e., SAV growing season; Apr–Oct) nutrient loss term as N is incorporated into plant tissue. We estimated this rate by using data from aerial SAV surveys (VIMS 2011), SAV biomass as suggested by Moore *et al*. (2000), and estimates of the % N content of SAV from a variety of sources (e.g., Abbasi *et al*. 1990; Yu *et al*. 2010; Mukherjee *et al*. 2008). The results indicate a modest seasonal-scale buffering of nutrients by the SAV community. It is likely that SAV nutrient buffering via enhanced denitrification and burial of PON, as indicated above, is considerably greater than the estimate we generated with available data from non-vegetated sediments. We also examined sediment flux data from many small tidal freshwater Chesapeake Bay tributaries (Bailey 2005) and estimated sediment $NH_4$ releases in

Mattawoman Creek. These were substantial and were the largest single term in the budget. This result has been observed in other systems (Boynton *et al.* 1995; Boynton and Kemp 2008) and indicates the importance of sediment nutrient sources in sustaining autotrophic production in shallow systems, especially during warmer months of the year when sediment processes are most active. However, we also know that sediment releases of $NH_4$ are sensitive to the supply of labile organic matter to the sediment surface (Cowan and Boynton 1996). The supply rate of such material likely decreased following large reductions in WWTP discharges and the magnitude of sediment nutrient releases probably also declined.

Nutrient Cause–Effect Chains

In many estuarine ecosystems, excessive nutrient loading is the primary cause of rapid algal growth and biomass accumulation and that seems to be the case in Mattawoman Creek. The relationship between nutrient loads from all sources and algal responses (chlorophyll-*a* concentration) is the starting point for the following analyses. Essentially, we attempted to link nutrient loading from drainage basins to estuarine chlorophyll-*a* concentration, and subsequently link algal stocks to summer water clarity. Linkages of key water quality variables to nutrient loads will allow for preliminary estimates of the magnitude of estuarine responses to future nutrient load reductions or increases. In developing these relationships, data from several shallow estuarine systems were used in a comparative analysis approach to increase the signal to noise ratio and to examine the robustness or generality of results (Kemp and Boynton 2012).

Many measurements of chlorophyll-*a* from several locations in Mattawoman Creek indicated elevated summer concentrations. Cold season algal blooms also occurred and likely deposited labile organic material onto sediments which are not decomposed until early-to-mid summer when elevated temperature stimulates sediment bacterial activity. Respiration of such material releases nutrients to the water column during summer and these nutrients, in addition to spring nutrient inputs, help stimulate the large summer blooms in the creek. The connection of winter–spring nutrient loads to summer blooms is well described in Chesapeake Bay and its tributaries and is also reflected in data for several shallow estuaries connected or adjacent to Chesapeake Bay region (Boynton *et al.* 1995; Boynton and Kemp 2000) and elsewhere (Nixon 1988). Using a multi-system comparison of shallow, mildly to very eutrophic estuaries in the Chesapeake Bay region, winter–spring N loading and summer chlorophyll-*a* were found to be highly correlated, and data for Mattawoman Creek fit the general pattern (Fig. 10). The relationship appears to be linear and indicates the potential for large changes in chlorophyll-*a* in response to nitrogen load changes. Several annual observations
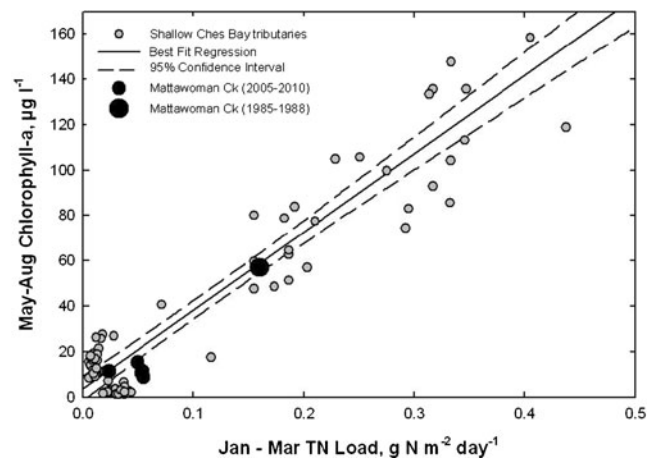


**Fig. 10** A scatter plot of winter–spring TN load versus chlorophyll-*a* concentration developed for Mattawoman creek and other shallow Chesapeake Bay tributaries. The large decrease in nitrogen loading was accompanied by a large reduction in chlorophyll-*a* concentration. Data for the other Chesapeake Bay systems was from Boynton *et al.* (2009)

were available for Mattawoman Creek including one set of observations from the 1985 to 1988 periods when nutrient loading rates were much higher and a set of more recent observations (2005–2010) collected when nutrient loading rates were much lower. Both data sets conformed to the general relationship. A factor of about four reduction of nutrient loading rate resulted in about a factor of five reduction in chlorophyll-*a* concentration, suggesting this system is responsive to nutrient load changes.

Water clarity determines how much light is available for photosynthesis by phytoplankton in the water column and by SAV and benthic algae growing at the sediment surface. Water clarity is typically reduced in estuaries when the concentration of algae, sediments, colored dissolved materials, and other particles increases in the water column, and that was the case in Mattawoman Creek during earlier years. Secchi disk measurements revealed distinct patterns in water clarity, the main ones being that depths varied seasonally during any 1 year (not shown) and water clarity has improved since 2000 (Fig. 5). Using Secchi disk data, we estimated the water depth to which 1 % of surface light penetrated (minimum light needed for benthic diatom growth). Growth of these algae on the sediment surface can reduce nutrient flux from sediments to the water column and also suppress sediment re-suspension. It is clear that prior to 2000, 1 % light reached depths of about 1.1 m while during more recent years, Secchi depths increased and the 1 % light depth increased to 3 m, considerably greater than the average depth of the creek.

Correlations between Secchi depth and both chlorophyll-*a* and total suspended solids indicated that both contributed to light attenuation in the creek, but chlorophyll-*a* in this case was more strongly correlated with water clarity ($r^2 = 0.80$, $p < 0.01$) and this, in turn, suggested that reductions in chlorophyll-*a* via

nutrient load reductions would result in increased water clarity. To continue examination of the cause–effect chain described earlier, chlorophyll-*a* and Secchi depth data from Mattawoman Creek and from several other small tributary rivers were combined in a comparative analysis and a strong relationship was again observed (Fig. 11). SAV were absent from this system when Secchi depths were less than 0.5 m or when chlorophyll-*a* concentrations were greater than about 18 μg $L^{-1}$.

Summary and Future Investigations

Substantial point source nutrient (N and P) reductions in the system resulted in large reductions in algal biomass, large increases in SAV coverage and density, and modest increases in water clarity. Initial responses to nutrient load reductions occurred relatively quickly (1–4 years) but more "steady-state" conditions took longer to emerge. For example, N and P load reductions were initiated during 1991 and were largely completed by 1995; algal chlorophyll began declining during 1999 but did not reach low and stable levels until 2006, 11 years after input reductions were complete. A similar, but shorter, response pattern was evident with SAV wherein bed expansion started the year after load reductions were completed but did not reach a higher and more stable condition until 2003. Thus, system responses ranged from annual to decadal depending on the component being considered, a finding similar to those reported by Borja *et al*. (2010). We suggest that researchers clearly indicate the temporal sequence of nutrient load reductions as this clearly has implications for determining response lag times.

Duarte *et al*. (2009) considered the notion that appropriate nutrient load reductions would return impaired ecosystems to their original or baseline condition. In their evaluation of four
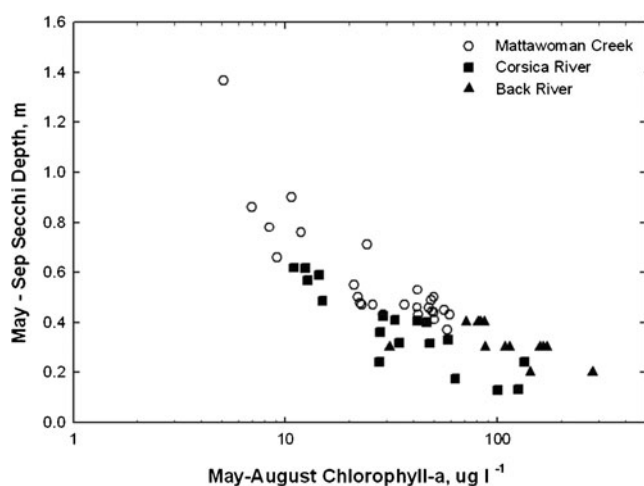


**Fig. 11** A scatter plot of chlorophyll-*a* versus Secchi disk depth developed for Mattawoman Creek and two other shallow Chesapeake Bay systems. Data for the other shallow systems were from Boynton *et al*. (2009) and Mattawoman Creek data were from the Chesapeake Bay Water Quality Monitoring Program (2012)

systems they found complex restoration trajectories and each system failed to return to an earlier reference condition. This is depressing news for those charged with restoration and responsibility for the expenditure of public funds. In the case of Mattawoman Creek, we can make several observations relevant to the Duarte *et al*. (2009) results. While we do not have a reference condition with which to compare the current status of the creek, we do know several things. SAV were abundant in the upper Potomac, including Mattawoman and other small tributaries, prior to 1940 (Carter *et al*. 1994). After that time, water quality and habitat conditions seriously deteriorated through the early 1970s; huge algal blooms were common, DO concentrations declined and SAV were largely absent (Jaworski *et al*. 2007). We found a few chlorophyll measurements for Mattawoman Creek from the 1970s exceeding 100 μg $L^{-1}$ and aerial photographs indicated SAV were absent. Thus, there is qualitative information suggesting an earlier state of clearer water, low algal stocks, and abundant SAV followed by a 40-year period of poor water and habitat quality. The current condition in Mattawoman Creek tends to resemble the pre-1940s condition including clearer water, a vibrant SAV community (with invasive species included), relatively low algal stocks, and a "world-class" largemouth bass fishery. Mattawoman Creek may not have returned to a baseline condition (*Neverland* in Duarte's terms) but, from the point of view of water quality managers and those who recreate in this system, it is vastly improved and it seems reasonable to call this a successful restoration. Similar results have been reported for Tampa Bay (Greening and Janicki 2006), and multiple SAV sites in Chesapeake Bay (Orth *et al*. 2010).

The nutrient budget did not balance and that might be grounds for not reporting results. However, imperfect budgets can still be very useful thinking and organizational tools. First, nutrient input data were available and these indicated the current importance of diffuse sources and the large role atmospheric deposition plays in this load component. Without a budget framework these conclusions would not have been evident. In addition, use of water quality model results indicated nitrogen was imported from the Potomac to Mattawoman Creek, constituting another source. The budget framework allowed us to conclude this source represented about 25 % of the annual N load. We often think of nutrient loads coming from the surrounding basin, atmospheric deposition and point sources but this observation indicates downstream sources can be important as well. A similar result has been reported for the Patuxent River estuary (Boynton *et al*. 2008). Water quality managers need to know if enrichment problems are caused by local, downstream, or some combination of both sources. We also considered why the budget did not balance and where the missing N sinks might be located. Work by others in Chesapeake Bay have indicated elevated denitrification rates in fringing wetlands, oyster reefs, and SAV communities (Gao *et al*. 2012; Boynton *et al*. 2008;

Greene 2005a). Our estimates indicate even slightly elevated rates would lead to a balanced N budget.

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