Aquat Geochem (2014) 20:291–323
DOI 10.1007/s10498-014-9226-y

ORIGINAL PAPER

From Headwaters to Coast: Influence of Human Activities on Water Quality of the Potomac River Estuary

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Received: 4 June 2013 / Accepted: 3 February 2014 / Published online: 26 February 2014

Abstract The natural aging process of Chesapeake Bay and its tributary estuaries has been accelerated by human activities around the shoreline and within the watershed, increasing sediment and nutrient loads delivered to the bay. Riverine nutrients cause algal growth in the bay leading to reductions in light penetration with consequent declines in sea grass growth, smothering of bottom-dwelling organisms, and decreases in bottom-water dissolved oxygen as algal blooms decay. Historically, bay waters were filtered by oysters, but declines in oyster populations from overfishing and disease have led to higher concentrations of fine-sediment particles and phytoplankton in the water column. Assessments of water and biological resource quality in Chesapeake Bay and tributaries, such as the Potomac River, show a continual degraded state. In this paper, we pay tribute to Owen Bricker's comprehensive, holistic scientific perspective using an approach that examines the connection between watershed and estuary. We evaluated nitrogen inputs from Potomac River headwaters, nutrient-related conditions within the estuary, and considered the use of shellfish aquaculture as an in-the-water nutrient management measure. Data from headwaters, nontidal, and estuarine portions of the Potomac River watershed and estuary were analyzed to examine the contribution from different parts of the watershed to total nitrogen loads to the estuary. An eutrophication model was applied to these data to evaluate eutrophication status and changes since the early 1990s and for comparison to regional and national conditions. A farm-scale aquaculture model was applied and results scaled to the estuary to determine the potential for shellfish (oyster) aquaculture to mediate

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eutrophication impacts. Results showed that (1) the contribution to nitrogen loads from headwater streams is small (about 2 %) of total inputs to the Potomac River Estuary; (2) eutrophic conditions in the Potomac River Estuary have improved in the upper estuary since the early 1990s, but have worsened in the lower estuary. The overall system-wide eutrophication impact is high, despite a decrease in nitrogen loads from the upper basin and declining surface water nitrate nitrogen concentrations over that period; (3) eutrophic conditions in the Potomac River Estuary are representative of Chesapeake Bay region and other US estuaries; moderate to high levels of nutrient-related degradation occur in about 65 % of US estuaries, particularly river-dominated low-flow systems such as the Potomac River Estuary; and (4) shellfish (oyster) aquaculture could remove eutrophication impacts directly from the estuary through harvest but should be considered a complement—not a substitute—for land-based measures. The total nitrogen load could be removed if 40 % of the Potomac River Estuary bottom was in shellfish cultivation; a combination of aquaculture and restoration of oyster reefs may provide larger benefits.

Keywords Nutrients · Eutrophication · Nitrogen load · Headwater streams · Shellfish aquaculture · Nutrient bioextraction

1 Introduction

Chesapeake Bay, located in the mid-Atlantic region of the United States, is a classic drowned river valley, formed when the Atlantic Ocean, rising in response to melting Pleistocene glaciers, flooded the river valleys that drained the North American continent (Pritchard 1967). As a result, the bay receives inputs of freshwater and associated sediment and solutes from the watershed, as well as oceanic inflows through the mouth of the estuary (Meade 1981; Guilcher 1967). Physical and chemical weathering of bedrock in the watershed produces sediment and solutes that are transported downstream, some of which eventually enter the estuary (Bricker et al. 2003a). During transport, sediment and solutes are transformed and concentrations changed as a result of chemical and physical processes; the effect is cumulative in the downstream direction. While natural weathering processes have always been a source of sediment and nutrients, during the past 200 years human population growth and activities have caused increased river loads in the Chesapeake region, as in many other places, to several times the levels that occur naturally (Meybeck 1982; Garrels and Mackenzie 1971; Garrels et al. 1973). Human-influenced increases in nutrient loads include discharge from sewage treatment plants, atmospheric deposition onto terrestrial and aquatic surfaces, and runoff from urban and agricultural land uses.

The progression of ecological impacts associated with excess nutrient discharges (called eutrophication) to coastal waters is well described and follows a fairly predictable sequence that has been observed in estuaries and coastal water bodies worldwide (Bricker et al. 2007; Garmendia et al. 2012; Ferreira et al. 2011b; and others in Bricker and Devlin 2011). Briefly, nutrients cause eutrophication through stimulation of algal blooms, which, if excessive, may lead to reductions in water transparency and subsequent loss of sea grass habitat and fisheries (NRC 2000; Glibert et al. 2010). Other goods and services provided by the estuary also may be impacted causing economic losses, for example, through decreased fish catch and losses of tourism (Lipton and Hicks 1999, 2003; Bricker et al. 2006; Daily 1997). Although sediments and nutrients (nitrogen and phosphorus) were named as the top pollutants in Chesapeake Bay in the 2009 Executive Order (EO 13508), analysis of sediments and phosphorus is beyond the scope of this paper. We focused on nitrogen because it is most often, though not exclusively (e.g., Malone et al. 1996; Kemp et al. 2005), the limiting nutrient in estuarine waters.

Eutrophication is globally recognized as a threat to coastal water quality and to the services provided by coastal ecosystems (Bricker et al. 2007; Zaldivar et al. 2008; Diaz and Rosenberg 2008; Foden et al. 2011). Chesapeake Bay and tributary estuaries such as the Potomac River, as with other water bodies worldwide, have experienced nutrient-related water-quality degradation for decades with consequent impacts to living resources such as sea grasses and cascading impacts on fisheries (e.g., Orth and Moore 1984; Breitburg 2002; Breitburg et al. 2009a, b; Lipton and Hicks 1999, 2003; Mistiaen et al. 2003). Eutrophication and overall health assessments of the Potomac River Estuary (PRE) show that water quality has been degraded by nutrient inputs (e.g., UMCES 2011; Chesapeake Bay Foundation 2012; Bricker et al. 1999, 2007). Concern about these conditions has led to legislation (Table 1; Boesch et al. 2001) to reduce nutrient pollution and restore water quality to acceptable standards. Legislative mandates have required implementation of management measures to reverse coastal eutrophication, mostly focusing on the reductions in land-based sources of nutrients, such as fertilizer application and wastewater treatment plant discharges. Records of long-term discharges from Potomac River Publically Owned Treatment Works (POTW; Jaworski et al. 2007) indicate that nutrient reductions have had an effect (Fig. 1). There is increasing recognition, however, that returns on investment in both point- and nonpoint-source controls are diminishing, that additional management will not lead to significantly greater reduction in nutrient loads, and that at some point further reductions are not cost-effective (Stephenson et al. 2010). This is particularly the case for nonpoint sources, where control is difficult technically, but also is a problem for point sources once nutrients are reduced to concentrations at the limit of technological feasibility. Another argument for finding alternate management measures is highlighted by the different (and unpredictable) recovery paths that can occur, in contrast to a management model based on the presumption that the recovery path is the reverse of the impact path, and that both are linear (Duarte et al. 2009).

In the 1800s, Maryland produced about 40 % of the total US oyster harvest with the principal Chesapeake Bay oyster beds located in the PRE. Harvest has been significantly reduced, however, due to overfishing and disease (Figs. 2, 3; Livings 2011; Keiner 2009; Churchill Jr 1920). In-the-water methods that remove nutrients, chlorophyll *a,* and particulate matter once they are in the water body have an immediate positive effect by directly removing the symptoms of eutrophication. Measures such as shellfish aquaculture are especially important to reductions in nonpoint-source inputs, which are the most difficult to control and regulate. Oyster aquaculture is of particular relevance in Chesapeake Bay and the PRE, since by the early 1900s, oyster populations already were recognized for their integral part in maintenance of good water quality due to their filtration (Rothschild et al. 1994; Keiner 2009). Presently, there is mounting evidence that "bioextraction," an environmental management strategy by which nutrients are removed from an aquatic ecosystem through the harvest of enhanced biological production, including the aquaculture of suspension-feeding shellfish, could play an important role in restoration of coastal water quality (Lindahl et al. 2005; Nobre et al. 2010; Ferreira et al. 2011a, 2012; Burkholder and Shumway 2011; Lindahl 2011). Additional benefits of this alternative management practice is that it can address legacy pollution in the water column and sediments, provide marketable seafood product, and potentially supply growers with additional income in a nutrient-trading program. The present-day question is whether enhanced oyster

Table 1 Acts, policies, and partnerships to prevent pollution impacts in Chesapeake Bay

Year(s)	Act, policy, or partnership
1940	Interstate Commission on the Potomac River Basin
1963	Clean Air Act (amendments 1970, 1977, 1990)
1969	National Environmental Policy Act
1972	Clean Water Act (amendments 1977, 1983, 1985, 1987)
1983, 1987, 2000	Chesapeake Bay Agreements developed Chesapeake Bay Program partnership
2008	Bay Action Plan
2009	Executive Order 13508 Chesapeake Bay Protection and Restoration

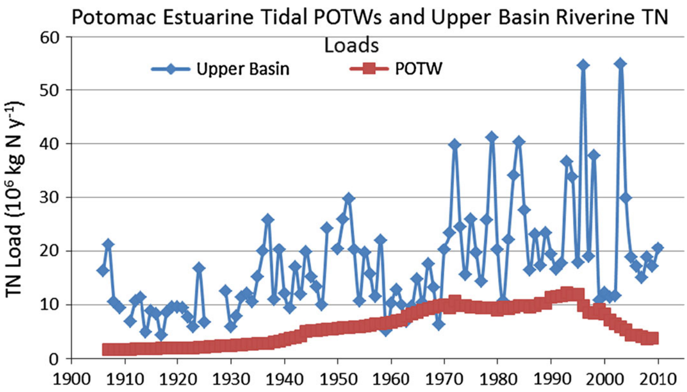


Fig. 1 Potomac River Upper Basin TN loads 1900–2010 and Tidal Publically Owned Treatment Works (POTW) TN load 1900–2010 (N. Jaworski, retired, USEPA, pers. comm., 2013; Jaworski et al. 2007)

populations can improve water quality in the Chesapeake Bay and PRE. Several studies have investigated this possibility (Cerco and Noel 2007; Scientific and Technical Advisory Committee (STAC) 2013; Kellogg et al. 2013).

The focus of our study is the continuing challenge of eutrophication impacts originating throughout the PRE watershed. Building on Owen Bricker's legacy of a holistic scientific perspective, we aim to provide a comprehensive picture of headwater contributions, eutrophication impacts within the estuary, and whether bioextraction is a potential solution by:

1. evaluating the nitrogen load from Potomac River headwater streams to the PRE. Although previous studies have identified the major sources as originating upstream of the Fall Line, the significance of headwater streams to that load has not been quantified. We used data collected from three small, forested watersheds by the US Geological Survey (USGS) in the Catoctin Mountain headwaters of the Potomac River from 1990 to 1994 (Rice et al. 1996; Rice and Bricker, unpublished data) to determine the significance of this source and the implication to management within the entire watershed;
2. updating the eutrophication status of the PRE to reflect present conditions by applying the Assessment of Estuarine Trophic Status (ASSETS) eutrophication assessment

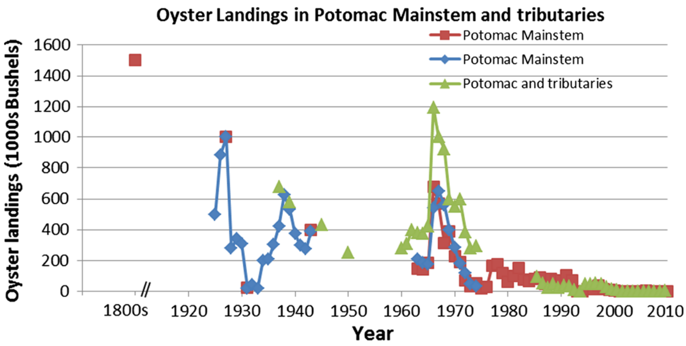


Fig. 2 Oyster landings in Potomac River mainstem and tributaries [data sources: *red squares*, E. Crosby, Potomac River Fisheries Commission, pers. comm., 2013; *blue diamonds*, Haven (1976); *green triangles*, 1937–1973, Haven (1976) and 1989–2011, Tarnowski (2012)]

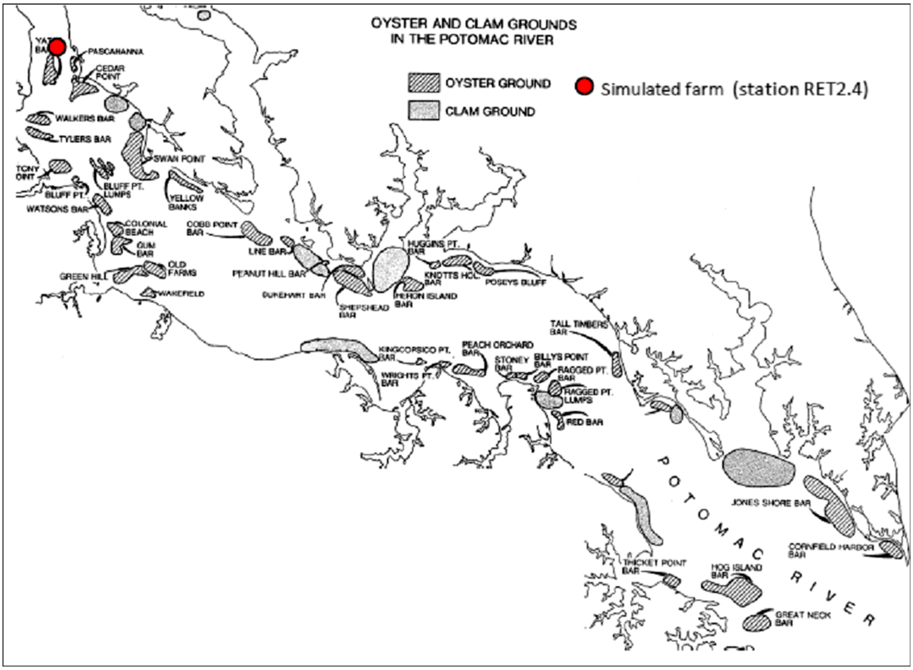


Fig. 3 Map of natural oyster grounds in Potomac River and location of simulated farm at site of MD DNR sampling station RET2.4

model (Bricker et al. 2003b) to recent water-quality data (2009–2011). Nutrient-related conditions in the PRE previously were evaluated as part of the National Estuarine Eutrophication Assessment (NEEA) in the early 1990s (Bricker et al. 1999, 2003b) and again in the early 2000s (Bricker et al. 2007, 2008). These previous studies provide a baseline to which results of this study can be compared, placing the PRE into context on regional and national geographical scales; and

3. investigating the use of aquaculture of the Eastern oyster (*Crassostrea virginica*) as an alternate management measure to complement traditional nutrient management strategies. We quantified nitrogen removal in the PRE by shellfish cultivation and harvest at a simulated farm by application of the Farm Aquaculture Resource Management (FARM, Ferreira et al. 2007b, 2009; Fig. 3) model for both present densities being cultivated and for potential expansion of aquaculture.

Our goal is to provide insights that will be useful in ongoing discussions and planning for future management measures to protect and remediate nitrogen impacts in the PRE, serving as a prototype for other eutrophic estuaries.

2 Potomac River Watershed and Estuary: The Setting and Background

2.1 Physical, Hydrological, and Watershed Characteristics

2.1.1 Potomac River Estuary

The PRE is the largest of the nine major tributary estuaries of Chesapeake Bay and is the second largest tributary after the Susquehanna River. It is located on the western shore, along with the Patuxent, Rappahannock, York, and James River Estuaries, while the Choptank, Tangier/Pokomoke, Chester, and Nanticoke River Estuaries are located on the eastern shore of Chesapeake Bay. The 37,995-km^2 Potomac River Basin includes parts of Maryland, Pennsylvania, Virginia, West Virginia, and the District of Columbia; 29,940 km^2 is considered Upper Basin and 8,055 km^2 is considered Lower Basin (Jaworski et al. 2007), with the boundary being roughly at the location of the Fall Line at Great Falls, Virginia, about 23 km upstream of Washington, DC (Fig. 4). The Basin is classified into seven physiographic provinces—six in the Upper Basin: the Appalachian Plateau, Valley and Ridge, Great Valley, Blue Ridge, Piedmont, and Triassic Lowlands ; one in the Lower Basin: Coastal Plain Basin (Blomquist et al. 1996). The geology of the Potomac River Basin, greatly simplified, can be roughly categorized into four groups: unconsolidated sediment, carbonate sedimentary rocks, siliciclastic sedimentary rocks, and crystalline rocks (Blomquist et al. 1996).

Potomac River discharge has been measured by the USGS at a long-term streamgage at Chain Bridge near Washington, DC, which represents drainage from most of the Upper Potomac River Basin. Annual mean discharge from 1895 to 2002 at Chain Bridge was 11,350 cubic feet per second (cfs) (321 m^3 s^{-1}; Jaworski et al. 2007), but it is highly variable from year to year, as can be seen by part of the record representing the years relevant to this study (Table 2). Within the estuary, there is net downstream transport based on the comparison of net nontidal upstream and downstream velocities (Lippson et al. 1979).

The 1,260-km^2 surface area of the estuarine portion of the Potomac River, from the head of tide to the mouth, is long and narrow (189-km long; average width 0.5 km) with water-retention times in the upper, middle, and lower zones of 16, 64, and 311 days, respectively (Fig. 4; Jaworski et al. 2007). The average depth is 5.1 m, with mid-channel depths ranging from 6.6 to 26.5 m. Tidal heights are 0.3 m at the mouth and 0.88 m near the head of tide (average 0.48 m), and salinity varies from 0 practical salinity units (psu) at the head of tide to >18 psu at the mouth (average 11 psu; Lippson et al. 1979). Tidal ebb velocities (average = 40 m s^{-1}, median = 36 m^3 s^{-1}) are slightly faster than tidal flood velocities

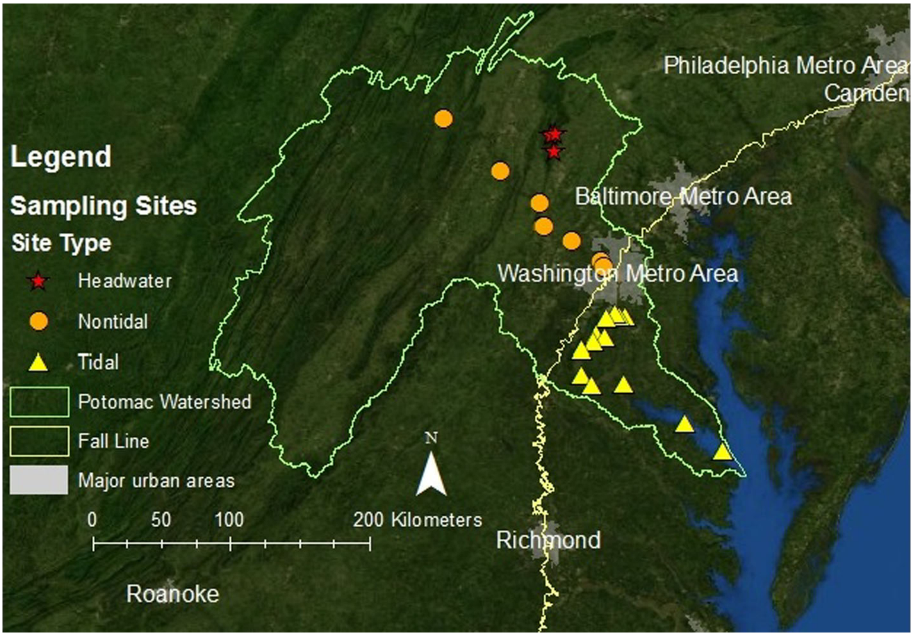


Fig. 4 Location of the Potomac River Estuary tidal, nontidal, and headwater sampling sites (map credit: D. Whitall and J. Pope)

(average = 36 m s^{-1}, median = 31 m^3 s^{-1}; Lippson et al. 1979). For this study, the PRE below the Fall Line was divided into a tidal fresh zone (183 km^2), where annual average water column salinity is 0–0.5 psu, and a mixing zone (1,077 km^2), where salinities are 0.5–25 psu.

Among the Chesapeake Bay tributaries, the PRE watershed has the highest mean (330 m) and maximum (1,433 m) elevation, the largest population, and is second highest in population density (123 people km^{-2}; Patuxent River watershed population density is 181 people km^{-2}; Bricker et al. 2007). Population in the PRE watershed in 2010 was about 6.11×10^6, with 88 % of those people located in the Washington metropolitan area (http://www.potomacriver.org/facts-a-faqs). Forest is the largest land use in the watershed as a whole, whereas agriculture and urban are the second highest land uses in the Upper and Lower Basins, respectively (Table 3).

2.1.1.1 Nutrients and Eutrophication in the Potomac River Estuary Long-term water-quality monitoring and studies of the Potomac River have provided understanding of nitrogen budgets (sources, losses, storage), cycling, and downstream exchange, as well as documentation of long-term impacts on water quality, habitat, and fisheries [Jaworski et al. 2007; Kemp et al. 2005; Boynton et al. 1995; Boesch et al. 2001; MD DNR Eyes on the Bay (www.mddnr.chesapeakebay.net/eyesonthebay/); USGS Water-Quality Loads and Trends at Nontidal Monitoring Stations in the Chesapeake Bay Watershed (www.cbrim.er.usgs.gov)]. Jaworski et al. (2007) provide a detailed history of nutrient loading, causes of changes, and resultant water-quality consequences since 1895. Together, these studies have shown that the dominant sources of nitrogen (as total nitrogen, TN) are nonpoint, about twice that of point sources, and mostly originate upstream of the Fall Line. Direct

Table 2 Annual mean and maximum and minimum daily mean discharge of Potomac River near Washington, DC, for calendar years 1990 through 2011

Year	Annual mean discharge (cfs/m^3 s^{-1})	Daily mean discharge	
		Maximum (cfs/m^3 s^{-1})	Minimum (cfs/m^3 s^{-1})
1990	11,056/313	84,500/2,393	2,130/60
1991	9,897/280	97,100/2,750	1,280/36
1992	10,352/293	93,500/2,648	2,130/60
1993	17,560/497	179,000/5,069	1,600/45
1994	17,285/490	140,000/3,965	2,190/62
1995	9,888/280	85,700/2,427	1,780/50
1996	28,410/805	327,000/9,261	6,210/176
1997	10,778/305	109,000/3,087	1,810/51
1998	18,830/533	141,000/3,993	1,590/45
1999	7,030/199	45,200/1,280	958/27
2000	8,219/233	58,900/1,668	2,330/66
2001	7,673/217	65,200/1,846	1,450/41
2002	7,428/210	49,100/1,391	995/28
2003	26,148/740	151,000/4,276	2,800/79
2004	16,111/456	109,000/3,087	2,530/72
2005	11,203/317	133,000/3,767	1,620/46
2006	10,285/291	77,200/2,186	1,700/48
2007	9,222/261	112,000/3,172	1,350/38
2008	11,520/326	128,000/3,625	1,740/49
2009	10,884/308	118,000/3342	1,810/51
2010	12,069/342	193,000/5,466	1,170/33
2011	17,261/489	163,000/4,616	2,030/57

Data from USGS National Water Information System for gage number 01646502 at Chain Bridge

cfs cubic feet per second, m^3 s^{-1} cubic meters per second

Table 3 Land use (as % total watershed) in Upper and Lower Potomac Basin (Jaworski et al. 2007)

Land-use type	Upper basin 29,940 km^2	Lower basin 8,060 km^2
Agricultural	34.6	16.0
Forest	60.8	38.0
Urban	2.6	19.0
Water	1.2	23.0
Other	0.8	4.0

atmospheric deposition to the PRE surface is estimated by some studies to account for 2–5 % of TN inputs (Jaworski et al. 2007; Boynton et al. 1995). Other studies show considerably higher contributions, but SPARROW includes indirect contributions to the estuary (30 %, Smith et al. 1997) and Blomquist and Fisher (1994) estimate indirect deposition to the watershed only (22 %).

Several authors have estimated the TN load for various time frames. Boynton et al. (1995) estimate loads of about 35.5 $\times$ 10^6 kg N $year^{-1}$ for 1985–1986, which is consistent with the 38.7 × 10^6 N kg year^{-1} estimate of Jaworski et al. (2007) for the same time period, and the 32.6 × 10^6 kg N year^{-1} for 2002 from SPARROW (A. Hoos, USGS, pers. comm., 2013). The long-term record shows that riverine inputs are the primary source of TN to the Upper PRE (average of 83 % of TN loads for 1895–2005, Jaworski et al. 2007; Fig. 1). Loads to the total estuary during the time frames of interest to this study were 44.2 × 10^6 kg N year^{-1} in 1990–1994 and 27.7 × 10^6 kg N year^{-1} in 2008–2009, the most recent estimates available (N. Jaworski, retired, USEPA, pers. comm., 2013).

TN loads have decreased since the mid-1990s and have continued to decrease largely as a result of further declines in POTW discharges and declines in atmospheric deposition (Fig. 1; Linker et al. 2013; Eshleman et al. 2013). Beginning in the late 1970s, the reduction in TN loads led to water-quality improvements in the Upper PRE, such as decreased chlorophyll *a* concentrations and increases in bottom-water dissolved oxygen (Kemp et al. 2005; Jaworski et al. 2007). Improvements have not been system wide, however, as Lower PRE monitoring stations continue to have high summer chlorophyll *a* concentrations (10–30 μg L^{-1}), and bottom-water dissolved oxygen concentrations are seasonally hypoxic [Jaworski et al. 2007; MD DNR Eyes on the Bay (www.mddnr.chesapeakebay.net/eyesonthebay/)].

2.1.2 Headwater Stream Sites

Headwater stream sites used in this analysis are located on Catoctin Mountain in north-central Maryland (Fig. 4; Rice and Bricker 1995). We used data collected from 1990 to 1994, the most recent data available from three small watersheds, which are as follows: 1) Hauver Branch, 2) Bear Branch, and 3) Fishing Creek Tributary (Table 4). These watersheds are located in a forested area that is primarily used for hiking and recreation, with no permanent residents. Mirroring flow in the Potomac River, average annual discharge at these sites is variable (Table 5). The combined area of the watersheds (~7 km^2) is about 0.02 % of the Potomac River watershed. The combined averages of discharge for the three headwater streams (4.95 cfs) was about 0.04 % of the average measured Potomac River flow (13,231 cfs) for 1990–1994.

3 Methods

3.1 Nitrogen Load from Headwater Streams

Data used for the analysis of nitrogen contributions from headwater streams were collected from the three sites described in Section 2.1.2 (Fig. 4; Table 6). The analysis was restricted to nitrate nitrogen (hereafter NO_3^-); concentrations of other nitrogen species in headwater streams were not measured because they typically are near zero. There are no recent data for these headwater stations; thus, data from 1990 to 1994 were used for the analysis (Rice et al. 1996; Rice and Bricker, unpublished data). Estimates of load from headwater streams were made using an averaging approach where the average concentrations for 1990–1994 were multiplied by the average stream discharge during the same period at each site (Richards 2003). Loads from the three sites were summed to provide an estimate of the input from those headwater streams. The 90th percentile of annual NO_3^- concentration data was calculated for comparison to downstream nontidal and tidal concentrations.

Table 4 Catoctin Mountain headwater stream sites attributes

Attribute	Hauver Branch	Bear Branch	Fishing Creek Tributary
Watershed area (km^2)	5.5	0.98	1.04
Bedrock (Fauth 1977)	Catoctin formation	Weverton formation, lower unit	Weverton formation, upper unit
Soils (Matthews 1960)	Highfield series, medium textured, well-developed and generally well-drained soils	Edgemont-Chandler very stony loams, 20–60 % slopes	Edgemont-Chandler very stony loams, 20–60 % slopes; Braddock gravelly and cobbly loams, 8–15 % slopes, moderately eroded
Land use	Forested—recreational hiking, and camping	Forested; recreational hiking	Forested—recreational hiking
Watershed relief (m)	260	267	226
Stream order	Second	First	First

See Table 6 for site latitude and longitude

Table 5 Annual mean discharge of headwater stream sites at Catoctin Mountain for calendar years 1990 through 1994

Year	Hauver Branch Gage number 01640965 (cfs/m^3 s^{-1})	Bear Branch Gage number 01640980 (cfs/m^3 s^{-1})	Fishing Creek Tributary Gage number 01641510 (cfs/m^3 s^{-1})
1990	3.29/0.093	–	0.509/0.014
1991	2.46/0.070	0.448/0.013	0.458/0.013
1992	3.97/0.112	0.712/0.020	0.714/0.020
1993	4.88/0.138	0.848/0.024	0.870/0.025
1994	–	0.669/0.019	0.580/0.016
Average of years available	3.65/0.103	0.669/0.019	0.626/0.018
Sum of annual averages across all sites			4.95 cfs; 0.140 m^3 s^{-1}

Data from USGS National Water Information System

cfs cubic feet per second, m^3 s^{-1} cubic meters per second

–, data for full calendar year not complete

3.2 Eutrophication Assessment

Eutrophication indices provide a simplified way to evaluate nutrient-related water-quality conditions, to link the nutrient sources that are probable causes of degradation, and to predict future conditions as a means of informing development of successful management measures. Important considerations for these indices are to relate the level of nutrient input to observed water-quality conditions, to accurately represent conditions based on published methods and data, and to provide results that are understandable to all users. There are several multi-metric aggregated indicator eutrophication assessment methods available, each using a variety of indicators, time frames, and statistical methods analyses (e.g., ASSETS, Bricker et al. 2003b; EPA NCA, USEPA 2008; TRIX, Vollenweider et al. 1998; WFD-BC, Garmendia et al. 2012; WFD-UK, Devlin et al. 2011, Foden et al. 2011; others

Table 6 Potomac River Basin station names and locations used in the analysis

Station name	Latitude	Longitude
Headwater streams		
Hauver Branch	39.61944	−77.46667
Fishing Creek Tributary	39.53583	−77.44667
Bear Branch	39.62083	−77.44000
Nontidal stations		
POT1472	39.15551	−77.52232
POT1471	39.15441	−77.52125
SEN0008	39.07958	−77.33964
CJB0005	38.97344	−77.14884
POT1184	38.94821	−77.12733
POT2386	39.69741	−78.17630
POT1830	39.43507	−77.80266
POT1595	39.27347	−77.54367
Tidal stations		
PIS0033	38.69841	−76.98673
XFB1986	38.69786	−77.02317
TF2.1	38.70664	−77.04876
TF2.2	38.69067	−77.11111
TF2.3	38.60822	−77.17397
MAT0078	38.58852	−77.11865
MAT0016	38.56508	−77.19345
TF2.4	38.53006	−77.26537
RET2.1	38.40347	−77.26909
RET2.2	38.35253	−77.20508
RET2.4	38.36259	−76.99063
LE2.2	38.15760	−76.59803
LE2.3	38.01421	−76.34615

Nontidal and tidal stations from MD Department of Natural Resources Monitoring Program

Only tidal stations were used for the ASSETS analysis with tidal fresh zone represented by stations PIS0033 – TF2.4 and mixing zone represented by stations RET2.1 – LE2.3

in Borja et al. 2008 and Zaldivar et al. 2008). It is beyond the scope of this paper to make a detailed evaluation of different methods (for a review, see Zaldivar et al. 2008; Borja et al. 2008, 2012; Devlin et al. 2011; Garmendia et al. 2012). There are several assessments specific to Chesapeake Bay (e.g., UMCES 2011; Chesapeake Bay Foundation 2012), but we focused on an assessment method that has been applied to the PRE, other Chesapeake region estuaries, and other US estuaries (ASSETS; Bricker et al. 2003b).

The PRE was one of 141 water bodies evaluated in the National Estuarine Eutrophication Assessment (NEEA) using ASSETS. The assessment was conducted twice, with the PRE receiving an overall score of high-level eutrophication both times, with human-related loads considered high (Bricker et al. 1999, 2007). The ASSETS assessment has overall scores for the PRE and other Chesapeake region estuaries. It is applied by salinity zone, with separate results generated for the mixing and tidal fresh zones (see Table 6 for list of stations used in the analysis) that are then area weighted to provide a system-wide score; thus, differences and trends between the upper estuary and lower estuary can be distinguished.

We applied ASSETS to data for the PRE below the Fall Line to evaluate conditions in 2009–2011. The ASSETS tool is straightforward in both required parameters and calculations (see Table 7 for required data; an automated version is available at www.eutro.org/register), and it is designed to provide broad management-level guidance, including for poorly sampled coastal systems. It was originally developed for US coastal system assessment (Bricker et al. 2003b, 2007) and has been tested extensively in European systems (e.g., Ferreira et al. 2003, 2007a; Devlin et al. 2011; Garmendia et al. 2012) and in China (e.g., Xiao et al. 2007). The two previous NEEA reports summarize results by region and nationally so that PRE results can be placed into context on these scales.

The ASSETS tool evaluates three components of eutrophication: (1) *Influencing Factors* combines natural susceptibility and human-related nutrient inputs; (2) *Eutrophic Condition* estimates the level of impact based on five indicators, which also are used in the required 303(d) state water monitoring and assessment program (http://water.epa.gov/type/watersheds/monitoring/elements.cfm) and are used to measure progress toward Chesapeake Bay Program water-quality goals; and (3) *Future Outlook* evaluates potential changes that may occur based on natural susceptibility and expected changes in nutrient load (Table 7). The final step combines the categorical (i.e., high, moderate, and low) results for the three components into a single overall rating (Bricker et al. 2003b, 2008; Whitall et al. 2007). The assessment uses quantitative and qualitative data to determine trophic status. For example, to evaluate the "typical" extreme concentrations over the annual cycle, algal bloom concentrations are represented as the 90th percentile of annual chlorophyll *a* data. The 90th percentile of annual chlorophyll *a*, total suspended solids (TSS), and NO_3^- data for nontidal and tidal stations was calculated for 1993–1994 and 2009–2011. The 90th percentile of annual NO_3^- data for headwater stations was calculated for 1993–1994. These values were used to support the eutrophication assessment, to determine whether changes occurred between the two periods, and to facilitate the interpretation of assessment results.

3.3 Eutrophication and Shellfish Aquaculture

Shellfish aquaculture has shown promise in reducing eutrophication impacts. Shellfish filter phytoplankton and detritus from the water, thereby reducing eutrophication by short-circuiting organic degradation and consequent effects on bottom-water dissolved oxygen, and remove nutrients through harvest (Ferreira et al. 2007b, 2011a; Cerco and Noel 2007; Kellogg et al. 2013; Rothschild et al. 1994; Keiner 2009). Shellfish cultivation and harvest are currently being promoted as a means of national sustainable domestic production by NOAA's 2009 Aquaculture Policy and National Shellfish Initiative, and locally through the 2009 Maryland Shellfish Aquaculture Plan. Both the national policy and the local initiative acknowledge the water-quality benefits provided in addition to seafood production. Similar benefits have been noted from the restoration of oyster reefs as a result of increased denitrification (Kellogg et al. 2013; Cerco and Noel 2007).

We used a modeling approach to estimate the potential use and benefit of shellfish in nutrient remediation without the cost and time required for implementation. Specifically, we used the well-tested Farm Aquaculture Resource Management (FARM) model to evaluate the potential for the cultivation of Eastern oyster (*Crassostrea virginica)* to reduce eutrophic symptoms. This model has been tested in the EU, China, Ireland, and Northern Ireland (Ferreira et al. 2007b, 2009, 2011a, 2012; Nunes et al. 2011). The FARM model combines physical and biogeochemical models, shellfish growth models, and screening models at the farm scale for the determination of shellfish production and for the assessment of water-quality changes on account of shellfish cultivation. The model is useful for decision support for aquaculture siting (e.g., Silva et al. 2011; Ferreira et al. 2012), because it also evaluates farm-related impacts on benthic processes through biodeposition. It can be

Table 7 Data required for the application of ASSETS eutrophication assessment (Bricker et al. 2003b) and FARM (Ferreira et al. 2007a) models and data sources

Model/ component	Data required	Data source
Assessment of Estuarine Trophic Status (ASSETS) Model		
Influencing factors	Depth, area, volume, tidal range, degree of stratification, freshwater inflow, nitrogen concentration in riverine and oceanic end points, and total nitrogen load	Bricker et al. 2007; MD DNR Monitoring Program (R. Karrh, MD DNR, pers. comm., 2013; P. Tango, USGS CBPO, pers. comm., 2013); MD DNR Eyes on the Bay (http://mddnr.chesapeakebay.net/eyesonthebay/)
Eutrophic condition (based on annual data)	Chlorophyll *a*[a] (90th percentile), macroalgal abundance, dissolved oxygen[a] (10th percentile), nuisance and toxic bloom occurrence, changes in sea grass spatial coverage	MD DNR Monitoring Program (R. Karrh, MD DNR, pers. comm., 2013; P. Tango, USGS CBPO pers. comm., 2013); MD DNR Eyes on the Bay (http://mddnr.chesapeakebay.net/eyesonthebay/)
Future outlook	Estuary volume, tidal range, degree of stratification, freshwater inflow, expected future change in nutrient load	Bricker et al. 2007; MD DNR Monitoring Program (R. Karrh, MD DNR, pers. comm., 2013; P. Tango, USGS CBPO, pers. comm., 2013); MD DNR Eyes on the Bay (http://mddnr.chesapeakebay.net/eyesonthebay/)
Farm Aquaculture Resource Management (FARM) Model		
Farm layout and culture practice	Farm width, length, depth, number of sections, section volume, total animals, species, cultivation period, density, mortality	B. Russell, Shore Things Shellfish, pers. comm., 2013 and http://www.shorethingshellfish.com/about-us.html), D. Webster, UMD, pers. comm., 2013
Environment	Water temperature, current speed, chlorophyll *a*, particulate organic matter, total suspended solids, dissolved oxygen, wind speed	data for Potomac River Estuary station RET2.4 http://mddnr.chesapeakebay.net/eyesonthebay/); current and wind speed from S. Gill, NOAA CO-OPS, pers. comm., 2013

[a] The percentile of annual measures is used to represent the "typical" extreme concentrations (i.e., highest for chlorophyll, lowest for dissolved oxygen) observed during the year (Bricker et al. 2003a, b) see text

used for marginal analyses of farm production potential and profit maximization, while assessing potential credits for carbon and nitrogen trading (Ferreira et al. 2007a, 2009, 2011a; www.farmscale.org). The model converts estimated nitrogen removed by the oysters to human population equivalents and calculates the potential value of the ecosystem service represented, providing a substitution or "avoided" cost of land-based nutrient removal that would serve as additional revenue to the farmer in a nutrient-trading program. Also evaluated are changes in eutrophication indicators, chlorophyll *a* and dissolved oxygen, that result from the filtration of the oysters during the culture period using components of ASSETS (Bricker et al. 2003b). The general layout for the model is shown (Fig. 5) and is applicable to suspended culture from rafts or longlines, as well as to bottom culture. Inputs for shellfish modeling include data on culture practice (e.g., farm layout, species, and stocking densities) and environmental parameters, including shellfish food particles in the water column (i.e., phytoplankton and detritus, Table 7). The model output of interest here is the mass of nitrogen removed through uptake of phytoplankton and detritus by shellfish filtration.

Although presently there are no aquaculture leases in the PRE mainstem (K. Greenhawk, MD DNR, pers. comm., 2013), we simulated a farm in the mid-Potomac River mainstem using data from station RET2.4, which is above the area of seasonal hypoxia (MD Department of Natural Resources http://mddnr.chesapeakebay.net/eyesonthebay/) and is located near a natural oyster bar (Fig. 3). We used 2010 data for water-quality drivers (temperature, salinity, particulate organic matter, chlorophyll *a*, and TSS). Typical "extensive" spat-on-shell bottom culture practices for Eastern oyster that are employed by Chesapeake Bay region growers were used for the simulation. We used seeding density of 100 oysters m^{-2}, the typical oyster density at "healthy" bay sites (Greenhawk et al. 2007) and less than densities supported at a restored reef (131 oysters $m^{-2;}$ Kellogg et al. 2013). Mortality is highly variable from year to year; we used 40 % mortality per cycle, for a 3-year cycle on a 4-acre farm where 3 acres are in cultivation (Table 7).

In addition to the removal of nitrogen by oyster aquaculture, the avoided cost of wastewater treatment, the ecosystem service that is provided by the shellfish filtration, was estimated based on a substitution value of \$12.40 to \$14.40 kg^{-1} for the estimated removed nitrogen (Lindahl et al. 2005). People equivalents were calculated based on an annual per person N load of 3.3 kg (Ferreira et al. 2007b). The results for the simulated farm were scaled up to evaluate potential removal using (1) the existing acres of oyster habitat reported in the PRE assuming they were cultivated rather than natural (3.72×10^3 acres; Greenhawk et al. 2007) and (2) the total area suitable for extensive bottom aquaculture in the PRE based on legal, policy, and environmental criteria (N. Carlozo, MD DNR, pers. comm., 2013) in the manner of Silva et al. (2011) to provide insight about the potential improvement of water quality should leases be allowed in the future. There are an estimated 112×10^3 acres that are suitable for bottom culture. Of this, we used 50 % of the area for the upscaling calculation given potential exclusions for depth and bottom type that are not included in the present estimate of the oyster aquaculture targeting study (N. Carlozo, MD DNR, pers. comm., 2013).

4 Results

4.1 Headwater Stream Contribution of Nitrogen to the Potomac River Estuary

Mean NO_3^- concentrations for 1990–1994 were 1.74, 1.19, and 0.54 mg NO_3^- L^{-1} for Hauver Branch, Bear Branch, and Fishing Creek Tributary, respectively (Table 8). Seasonal variability of NO_3^- in all three Catoctin Mountain headwater streams was observed, with lowest concentrations in summer and early fall, during maximum biological activity, and highest concentrations in winter, reflective of vegetative dieback (Fig. 6). The contribution from headwater streams to the PRE nitrogen load was determined for 1990–1994 by comparing the discharge from the three stream sites to the TN loads to the Potomac River estimated by Jaworski et al. (2007 includes riverine, POTW, and atmospheric inputs). The combined TN load from the three headwater streams (6.69×10^3 kg N $year^{-1}$) was less than 0.02 % of the total estimated load to the Potomac River for the same period (44.2×10^6 kg N $year^{-1}$). Mean concentrations (Table 8) of NO_3^- in the headwater streams were as high as nontidal and upper tidal estuary stations (see section 4.2), and 90th percentile concentrations were within the same range as concentrations observed in PRE samples during the same period (Fig. 7a). Mean concentrations of NO_3^- in three forested headwater streams in Shenandoah National Park, Virginia, were similar to the Catoctin Mountain streams (Table 8).

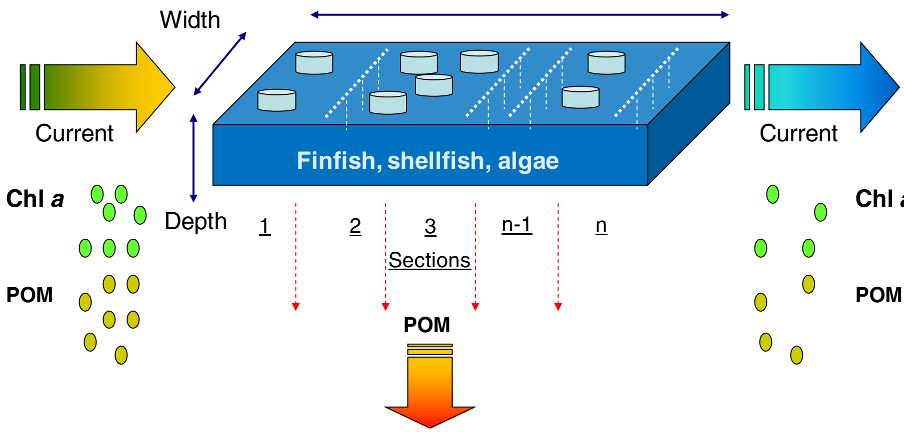


Fig. 5 Farm layout (rope and bottom culture) used in FARM model (adapted from Ferreira et al. 2007a, b)

4.2 Eutrophication Condition of the Potomac River Estuary

Results of the 90th percentile concentrations for NO_3^-, chlorophyll *a*, and TSS for nontidal and tidal stations and for time frames 1993–1994 and 2009–2011 (see Fig. 4 for locations) are shown in Fig. 7 a, b, c. The distribution of NO_3^- along mile points of the PRE showed a general decrease in concentrations from the nontidal to the tidal zone for both time frames, though the data were highly variable. The plots of chlorophyll *a* and TSS did not

Table 8 Headwater stream sites average NO_3^- concentrations, flow, and nitrate nitrogen load, 1990–1994, and NO_3^- concentrations at selected nontidal and tidal stations in the Potomac River, 1993–1994

	Average NO_3^- concentration (mg L^{-1})	Flow (m^3 s^{-1})	NO_3^- load (kg $year^{-1}$)
Headwater Sites (Potomac)			
Hauver Branch	1.74	0.103	5672
Bear Branch	1.19	0.019	712
Fishing Creek Tributary	0.54	0.018	304
Other headwater sites (Potomac and Rappahannock)			
Paine Run	0.84	–	–
Staunton River	0.27	–	–
Piney River	1.16	–	–
Nontidal and tidal stations			
CJB0005	1.28	–	–
TF2.3	1.48	–	–
RET2.2	0.94	–	–
LE2.2	0.40	–	–

Nontidal and tidal station data from MD Department of Natural Resources monitoring program; other headwater sites data from Shenandoah Watershed Study at University of Virginia

Average total nitrogen load to Potomac River Estuary 1990–1994 was 44.2 × 10^6 kg N $year^{-1}$ (Jaworski et al. 2007)

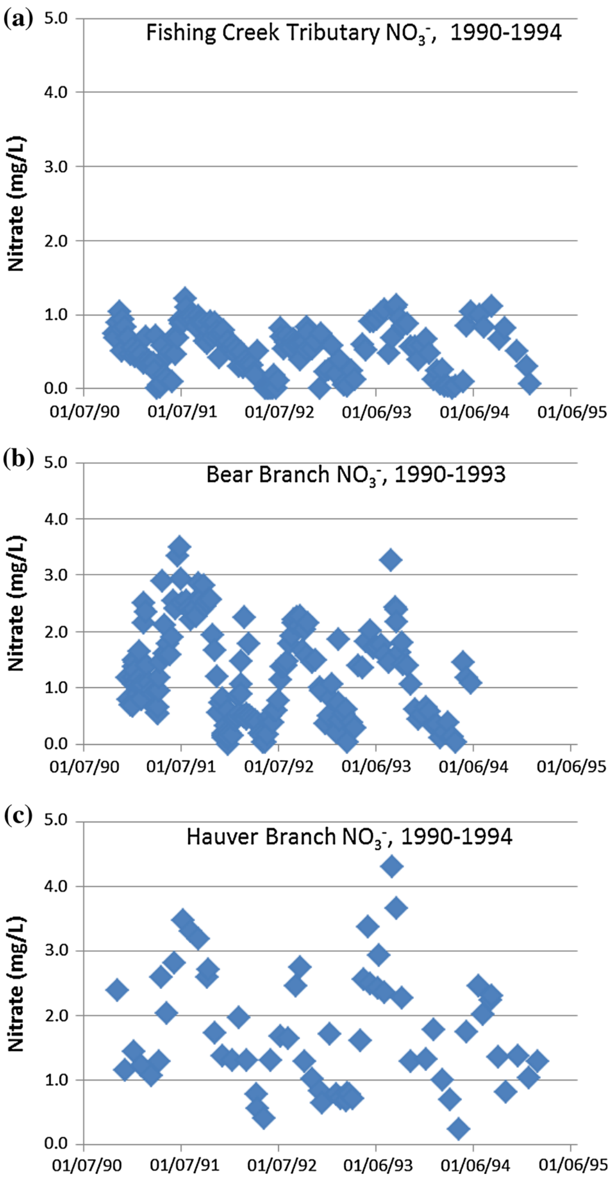
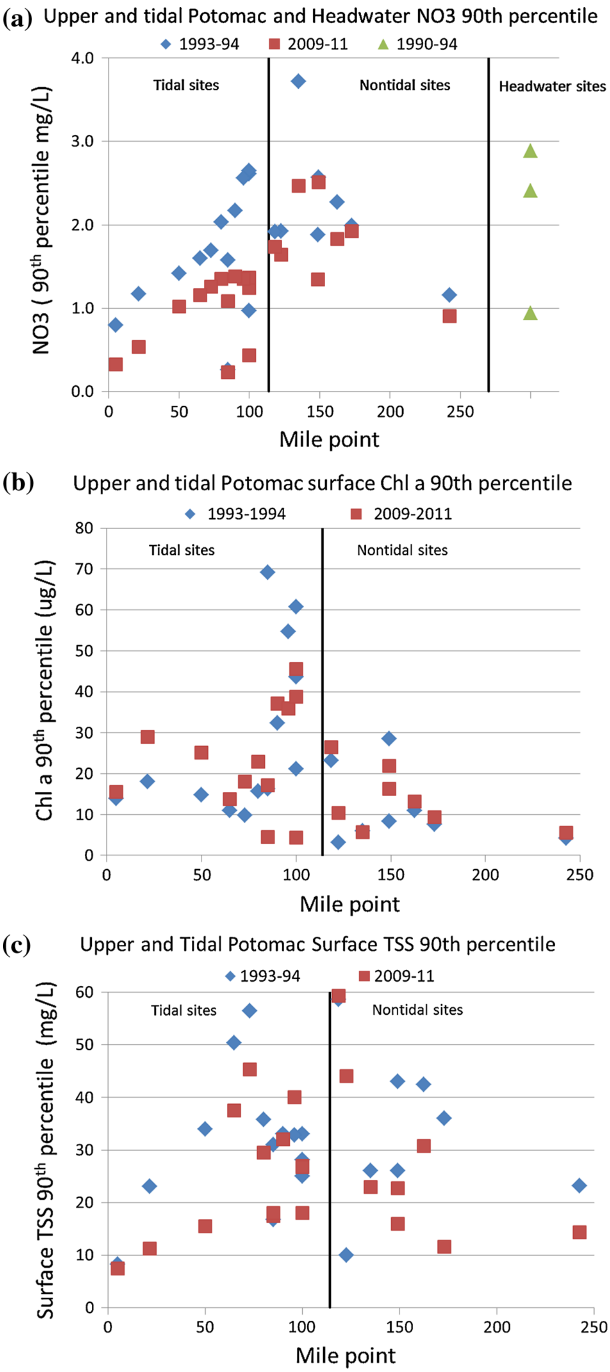


Fig. 6 Nitrate nitrogen concentrations in Catoctin Mountain headwater streams for 1990–1994 at **a** Fishing Creek Tributary, **b** Bear Branch, and **c** Hauver Branch. (*Data sources* **a** and **b**, Rice et al. (1996); **c**, Rice and Bricker (unpublished data)

show any discernible pattern, though both showed highest variability among stations between mile points 60 and 100.

We used Kruskal–Wallis and subsequent multiple comparison tests (Zar 1999) to compare means of the 90th percentile concentrations in the nontidal zone with those in salinity zones within the tidal portion of PRE (tidal fresh and mixing zones; see Table 6 for groupings). Differences in the means of 90th percentile concentrations grouped by time frames 1993–1994 and 2009–2011 in nontidal and the two salinity zones also were tested (Fig. 8).

Fig. 7 Percentile 90 of annual data 1993–1994 and 2009–2011 for **a** nitrate nitrogen; **b** chlorophyll *a*; and **c** total suspended solids at headwater (nitrate nitrogen only), nontidal, and tidal stations. Mile point 0 is the mouth of the estuary



Total suspended solids (TSS) 90th percentile concentration data show for both time frames that there were no significant differences among the nontidal and tidal fresh or mixing zones and no significant changes between time frames. The 90th percentile NO_3^-

Fig. 8 Means and standard errors of 90th percentile concentrations in nontidal, tidal fresh, and mixing zone stations in Potomac River Estuary for **a** nitrate nitrogen, **b** chlorophyll *a*, and **c** total suspended solids for 1993–1994 and 2009–2011 from Kruskal–Wallis and multiple comparison test (Zar 1999)

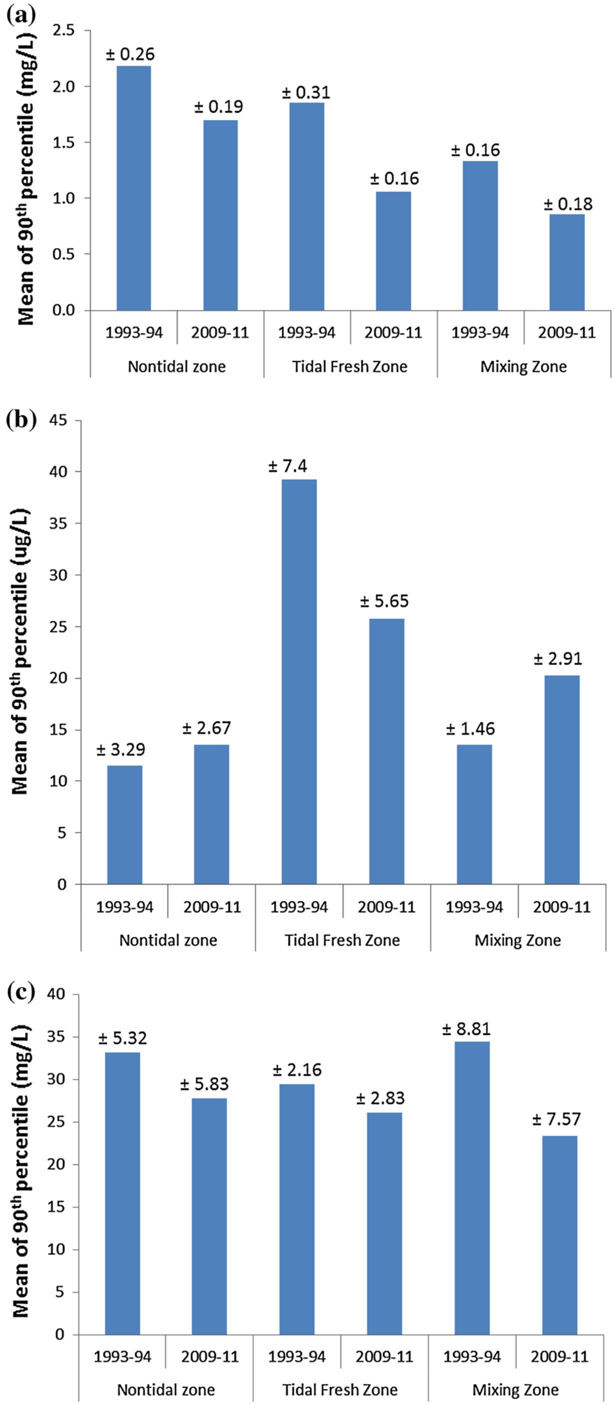


Table 9 Details of results for the ASSETS analysis for tidal fresh zone, mixing zone, and system wide for the Potomac River Estuary for 2009–2011 (total nitrogen load from 2008 to 2009, the most recent estimates available, N. Jaworski, retired, USEPA, pers. comm., 2013)

Component or indicator	Tidal fresh zone ($183\ km^2$)	Mixing zone ($1,077\ km^2$)	System wide ($1,260\ km^2$)
Influencing factors	High		
Susceptibility	Moderate (high dilution potential, low flushing potential)		
Total nitrogen load	High (27.7×10^6 kg N year^{-1})		
Eutrophic condition	High	High	High
Chlorophyll *a* (90th percentile)	High (35.2 μg L^{-1})	High (20.5 μg L^{-1})	High (22.6 μg L^{-1})
Macroalgae	No problem	No problem	No problem
Bottom-water dissolved oxygen (10th percentile)	No problem (5.8 mg L^{-1})	High (0.8 mg L^{-1})	Moderate (1.5 mg L^{-1})
Nuisance/toxic blooms	Moderate	High	High
Sea grasses	Low	Low	Low
Future outlook	Improve low		
Future nitrogen load	Decrease		
ASSETS score	Bad		

Susceptibility is the measure of the sensitivity of the system to nutrient loads due to the combination of dilution and flushing potential. Potomac River Estuary is moderately sensitive. For all indicators and for eutrophic condition, a rating of high indicates the worst or most severe eutrophic impact. See Table 6 for stations used in the analysis

concentrations were significantly higher in the nontidal zone than in the tidal fresh and mixing zones in both time frames. Decreases in the 90th percentile concentrations of NO_3^- in the nontidal zone (~17 %) were not significant but decreases in the tidal fresh (~43 %) and mixing (~35 %) zones were deemed significant. It should be noted that the data were not flow weighted, but since flows were ~23 % higher in 1993–1994, it is possible that decreases were greater given the potential dilution due to the higher flow. The 90th percentile concentrations of chlorophyll *a* were highest in the tidal fresh zone for both time frames, though not significantly different from the nontidal zone in 1993–1994. Decreases in the 90th percentile concentrations of chlorophyll *a* in the nontidal zone (~18 %) were not considered significant, whereas the estimated decrease in the tidal fresh zone (~34) and increase in the mixing zone (~50 %) were considered significant.

The ASSETS eutrophication assessment results shown in Table 9 indicate that for the system overall, influencing factors are high due to moderate susceptibility and high nutrient loads. The eutrophic conditions also are considered high (worst case or most severe impact), with consistent results for both zones, except for dissolved oxygen. There is greater oxygen depletion in the lower part of the PRE, the mixing zone, than in the tidal fresh zone. Comparison to results from previous assessments suggests that overall conditions have not changed since the early 1990s. Some improvements, however, were observed, such as the change in dissolved oxygen from high to moderate impact and of seagrasses from moderate to low impact (Table 10).

The 90th percentile of chlorophyll *a* concentrations, considered high, the extensive area over which they are observed, and the seasonal occurrence of high concentrations led to a rating of high, indicating significant nutrient-related impacts. Macroalgal blooms are not a problem in the PRE. There were indications of nuisance and toxic boom occurrences

Table 10 Potomac River Assessment of Estuarine Trophic Status (ASSETS) Eutrophication Assessment early 1990s (Bricker et al. 1999, 2003b), early 2000s (Bricker et al. 2007, 2008), and 2009–2011 (this study)

Assessment component	Early 1990s	Early 2000s	2009–2011
Influencing factors	High	High	High
Dilution potential	High		
Flushing potential	Low		
Susceptibility	Moderate		
Nitrogen load (10^6 kg year^{-1})	High (33.6[a])	High (33.8[b])	High (26.9[c])
Eutrophic condition	High	High	High
Chlorophyll	High	High	High
Macroalgae	Low	Unknown	Low
Dissolved oxygen	High	Moderate	Moderate
Sea grasses	Moderate	Low	Low
Nuisance/toxic blooms	Moderate	High	High
Future outlook	Worsen low	Worsen low	Improve low
Future nitrogen loads	Increase	Increase	Decrease
ASSETS score	Bad		

Susceptibility is the measure of the sensitivity of the system to nutrient loads due to the combination of dilution and flushing potential. Potomac River Estuary is moderately sensitive. For all indicators and for eutrophic condition, a rating of high indicates the worst or most severe eutrophic impact. Dilution and flushing potentials and susceptibility are the same for all periods since they are determined from long-term hydrological data. Unknown indicates inadequate data for the analysis

[a] Bricker et al. (1999) using SPARROW model estimates from base year 1987 (Smith et al. 1997)

[b] Bricker et al. (2007), N load is Fall Line point- and nonpoint-source loads plus below Fall Line point-source loads. Mean monthly effluent loads and flows are available from the CIMS data base, which is maintained by Chesapeake Bay Program (http://www.chesapeakebay.net)

[c] For 2008–2009, the most recent estimates available include atmospheric, POTW, and riverine input (N. Jaworski, retired, USEPA, pers. comm., 2013)

with *Microcystis aeruginosa* blooms causing skin rashes and nausea in humans (http://mddnr.chesapeakebay.net/hab/news_080211.htm) observed in 2009 and 2011. They lasted for weeks to months each time and occurred in both tidal fresh and mixing zones. Likewise, a *Prorocentrum minimum* bloom occurred in the mixing zone for months in 2009 and 2011 (http://mddnr.chesapeakebay.net/hab/HAB_archive.cfm#picview); thus, the impact of nuisance and toxic blooms was rated as high. Sea grasses decreased for 2 years in a row (2009–2011) as a result of storms and conditions unrelated to human nutrient inputs. Despite the declines, sea grass coverage in the Potomac River mixing zone met or exceeded restoration targets in 2010, due, in part, to sewage treatment plant upgrades and long-term reductions in nutrients entering the water (http://www.dnr.state.md.us/bay/sav/news/bgic_2010.asp; http://web.vims.edu/bio/sav/maps.html). Thus, the sea grass component received a rating of low indicating a low level of impact. Future outlook (Table 9) was rated as "improve low," a combination of the expected future decrease in nitrogen loads (e.g., Shenk and Linker 2013) and the moderate susceptibility of the PRE. The Potomac River watershed includes many new suburban communities that are expected to continue to experience rapid growth, thus potentially increasing nitrogen loads. Additionally, population growth in the Maryland part of the watershed alone is projected to increase approximately 1 % each year. Even though nitrogen and sediment loads to the estuary are decreasing and are expected to continue to do so, significant population increases and development in the watershed may offset some of those decreases.

4.3 Nutrient Removal by Shellfish Aquaculture: Can Shellfish Aquaculture Save the Potomac River Estuary?

4.3.1 FARM Simulation for the Potomac River Estuary

FARM model results for the simulated farm (Figs. 3, 9) suggest that bottom cultivation of Eastern oysters at a density of 100 oysters m^{-2} on a 4-acre farm (3 acres in cultivation) could remove 0.690×10^3 kg N $year^{-1}$ through filtration and harvest. The N removed is equivalent to nutrient treatment for 209 people, and if growers were included in a nutrient-trading program, it could provide additional income of about \$8,400 $year^{-1}$ for the value of the avoided cost of nutrient treatment (Fig. 9). Simulated changes in environmental effects due to oyster growth showed that chlorophyll *a* concentrations would decrease and dissolved oxygen would remain the same. However, the change in chlorophyll *a* is not enough to change the ASSETS rating to a lower category. These are results for a single farm, which, when scaled up to the total acres of suitable bottom, or potential lease area, in the PRE, become more significant.

4.3.2 Upscaling Farm-scale Results

Although leases are not currently allowed for oyster cultivation in the PRE, there are a reported 3.72×10^3 acres of oyster habitat (Greenhawk et al. 2007) and 112×10^3 acres of bottom have been estimated to be suitable for cultivation (N. Carlozo, MD DNR, pers. comm., 2013). The upscaling calculation requires several assumptions: there are no

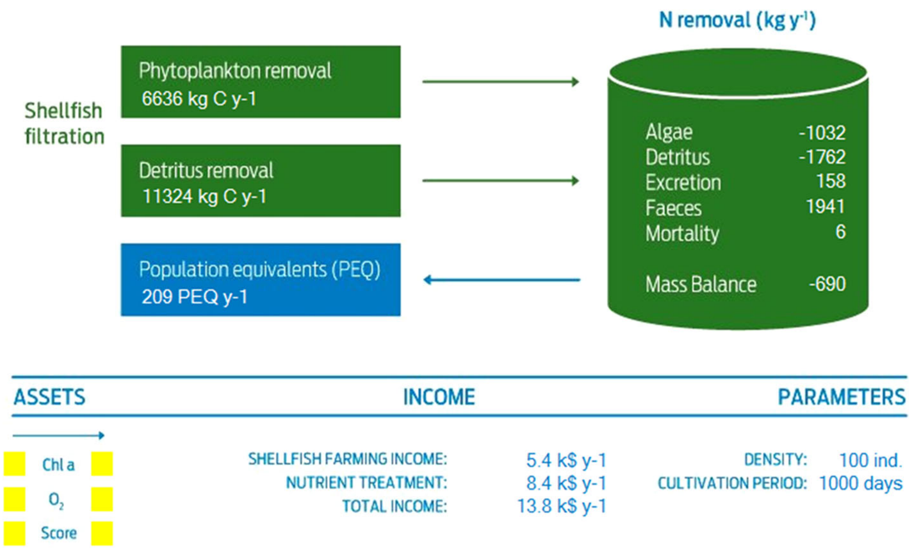


Fig. 9 Mass balance of nutrients, eutrophication, and economic assessment from Farm Aquaculture Resource Management (FARM) model analysis for a 4-acre simulated farm at Station RET2.4 and 100 individuals m^{-2}. The mass balance is created automatically by the model

additional reasons that the bottom area could not be leased; all lease areas have the same removal rates, despite potential differences in water quality among farm locations; and there is no interaction, i.e., food depletion, among adjacent farms. The reported area of existing habitat was used for the upscaling to simulate present removal. To illustrate the potential nutrient removal through expanded aquaculture harvest on suitable bottom area, we assumed that 50 % of the area is under cultivation since there are potential exclusions due to bottom type, depth, and unforeseen criteria. Given these caveats, the oyster habitat area, if cultivated, would remove 0.856×10^6 kg N year^{-1}, equal to nutrient treatment for 0.259 x 10^6 people. The suitable bottom area, if cultivated, would remove almost 13×10^6 kg N year^{-1} representing an ecosystem service of nutrient treatment for about 3.9×10^6 people. If growers were included in a nutrient-trading program, the value of the ecosystem service for the larger area would result in additional revenue to growers of about $\$157 \times 10^6$.

5 Discussion

5.1 Headwater Stream Contribution of Nitrogen to the Potomac River Estuary

Mean concentrations (Table 8) of NO_3^- in the headwater streams were as high as nontidal and upper tidal estuary stations, and 90th percentile concentrations were within the same range as concentrations observed in Potomac River samples during the same period (Fig. 7a). There is a positive relationship between discharge and NO_3^- concentration, suggesting that NO_3^- is exported during high flows; this indicates that groundwater NO_3^- concentrations in these headwater streams are low. The high NO_3^- concentrations in the headwater streams may be explained by the downwind location of the headwater sites relative to the Ohio River Valley, a major regional source of atmospheric pollution. The upland location of the headwater streams tends to focus atmospheric deposition, making them a target for elevated concentrations of NO_3^-. Although headwater stream concentrations were as high as in the estuary, headwater stream loads were much lower because their combined discharge is only a fraction of that of the PRE.

Because the watersheds are forested and uninhabited, this load represents the human influence of atmospheric deposition not taken up by vegetation. From the standpoint of nutrient management, this contribution might not compel the implementation of additional management measures to address impacts in the estuary. Funding for long-term monitoring of these sites was suspended in the mid-1990s. Given the small increase in atmospheric deposition of nitrogen in the form of ammonia to the PRE in recent decades (e.g., Linker et al. 2013), however, it may be worthwhile to re-institute a regular monitoring program in such headwater streams to track future changes so that, if necessary, management measures can be applied in an appropriate manner (Lovett et al. 2007). There is no substitute for long-term monitoring; it is needed for tracking performance of implemented management measures and for comparison to changes in directly measured values (e.g., via the National Atmospheric Deposition Program).

If all 837 km^2 of Potomac headwater area (VA DEQ 2012) were considered, assuming that the NO_3^- loading rate is consistent across that area, the contribution would represent about 2 % of total N inputs to the PRE. This may be an underestimate since it is for NO_3^- only. Additionally, this analysis applies only to forested areas and thus may not be representative of all headwater areas but indicates that loadings of NO_3^- from forested headwaters are low. These results are in contrast to Alexander et al. (2007) who show that headwater streams of various land uses account for a significant flux (i.e., greater than 45 %) of nitrogen to downstream tributaries.

5.2 Eutrophication Condition of the Potomac River Estuary

The results of the 90th percentile calculation for NO_3^-, chlorophyll *a*, and TSS shown in Figs. 7a–c and 8a–c represent the typical higher concentrations or "worst case" seen during the annual cycle. Higher concentrations of NO_3^- in nontidal stations and in the tidal fresh or upper zone of the estuarine portion of PRE are consistent with distributions reported for many estuaries (Boynton and Kemp 2008). This pattern reflects a gradient where concentrations are higher closer to the main nitrogen source, as well as dilution with increased water volume and biological processing (e.g., via uptake by phytoplankton and/or denitrification via anammox bacteria; Kuenen 2008) as water travels downstream. This pattern also might be expected for chlorophyll *a* and TSS concentrations. However, the distribution of TSS shows no trend, though lower concentrations were seen in the upper-most nontidal station and the stations near the mouth of PRE (Figs. 7c, 8c). The highest 90th percentile concentrations of chlorophyll *a* were seen in the middle of the PRE from about mile point 60 to mile point 100. This is considered the "transition zone," which coincides with the turbidity maximum, a length of the PRE that is characterized by mixing of riverine and oceanic waters (e.g., Housman 2009; Herman and Friedrichs 2010). These areas typically have high biological production and high suspended sediment compared to the rest of the water body (Postma 1967) and is likely the explanation for the higher concentrations of chlorophyll *a* in these stations. Lower concentrations in the tidal stations may be the result of higher flushing (e.g., 16-day residence time), whereby blooms are less likely to occur in the nontidal portion given higher flow of the river; in the tidal portion, algae have a greater opportunity to bloom because of tidal re-entrainment and longer retention of the water mass (i.e., 34- to 311-day residence times; Ferreira et al. 2005; Bricker et al. 2008). Although there is water exchange with the Chesapeake Bay mainstem, it is not clear how much Lower PRE stations are impacted by potential inputs from the mainstem compared to TN loads from upstream, since studies show that there is net TN export from the PRE to the mainstem (Boynton et al. 1995; N. Jaworski, retired, USEPA, pers. comm., 2013).

Figures 7a and 8a show that 90th percentile NO_3^- has declined in all zones of the PRE, consistent with measurable declines in POTW (Jaworski et al. 2007) with significant decreases in both the tidal fresh (43 %) and mixing (36 %) zones. Changes in the concentrations of chlorophyll *a* (Figs. 7b, 8b) show significant decreases (34 %) in the tidal fresh zone but significant increases (50 %) in the mixing zone. A reverse pattern is seen for bottom-water dissolved oxygen concentrations: significant increases in the tidal fresh and significant decreases in the mixing zone. Decreases in TSS are not statistically significant. These results suggest that despite improvements in NO_3^-, bottom-water dissolved oxygen, and chlorophyll *a*, there are still significant eutrophication impacts, particularly in the lower estuary. Lag times of improvements after load reductions have occurred have been noted in other systems (e.g., Tampa Bay, Greening and Janicki 2006; Gunston Cove, Jones and Kraus 2009) and may also be occurring here. The continued eutrophication of the Lower PRE suggests that influx from the mainstem may contribute to the observed impacts.

The ASSETS results (Table 9) are consistent with those from the two earlier assessments showing that eutrophic conditions have not changed since the early 1990s, despite the nearly 50 % reduction in nitrogen load, decreases in chlorophyll *a*, and improvement of dissolved oxygen concentrations in the tidal fresh zone over the same period. Estuarine water-quality response to changes in nutrient loads is complex and nonlinear. Previous studies of the PRE suggest that we should not expect a one-to-one improvement of water quality with decreases in nutrient inputs (Boynton et al. 1995) and that a system's recovery should not be expected to be identical but opposite to the degradation path (Duarte et al. 2009). There may be a lag in response to nutrient reductions as noted above. Alternatively, improvements in one part of the estuary were not adequate to change the system-wide assessment rating. For example, the increase in bottom-water dissolved oxygen (i.e., change from high to moderate impact) and continued regrowth of sea grasses are signs that conditions improved slightly but not to the extent to move the overall eutrophication status into the next category (Table 10). Likewise, the decrease in chlorophyll *a* in the tidal fresh zone from 39.2 to 25.8 μg L^{-1} is significant but does not move the ASSETS rating from the High category, which is any concentration >20 μg L^{-1} .

5.2.1 *Comparison of the Potomac River Estuary to Chesapeake Bay Region and Other estuaries*

Potomac River Estuary results are representative of other estuaries within the Chesapeake Bay region where population density is high, causing human-related nutrient loads to be high and promoting development of eutrophic conditions (Glibert et al. 2010; Bricker et al. 2008). The assessment results for the PRE also are representative of the majority of US estuaries assessed in the NEEA, where 65 % of estuaries in both studies reported moderate- to high-level eutrophic symptoms. The larger dataset shows that land use is highly related to eutrophication status; systems with >40 % of the watershed in combined urban and agricultural land use are those that show moderate-high to high eutrophication (Glibert et al. 2010).

Several attempts have been made to develop groupings of estuaries through the analysis of susceptibility to predict the magnitude of eutrophication symptoms that might be expected for a given nutrient load. The rationale is that water-quality impairments in systems of the same type could be addressed with similar management approaches, permitting the transfer of knowledge and experience to facilitate successful management (Glibert et al. 2010; Kurtz and Hagy 2012). Some of these analyses used geospatial clustering applications (e.g., LOICZ, DISCO; Buddemeier et al. 2007; Kurtz et al. 2006) to biogeochemical databases, while others used a narrative top-down approach with similar datasets (Alexander and Bricker 2003). We used the top-down approach, categorizing the 141 NEEA estuaries into four types: coastal embayments, fjords, lagoons, and river-dominated systems.

The NEEA ratings for the water-quality indicators are not measured values but rather categorical assessment results; thus, a frequency distribution was used to examine relations among the different groups. We used chlorophyll *a* as the response of interest because algae respond directly to nutrient loads and thus are the first indication of nutrient over enrichment. Analysis of the four estuary types shows that while all have some systems with high impacts, the river-dominated systems are the most highly impacted (Fig. 10a). The same pattern is seen for eutrophic condition; more than 50 % of estuaries with high chlorophyll *a* also have high overall eutrophication impacts (Bricker et al. 2003b).

Because the PRE and the other Chesapeake Bay region estuaries are all river-dominated, this group was further divided into fast, moderate, and slow/very slow flow to evaluate the effect of flow rate on chlorophyll *a* impact development. Slower flow rates allow more time for phytoplankton to grow and previously were shown to be related to the occurrence of nuisance, and in particular, toxic, algal blooms. Toxic species often have much slower growth rates than nontoxic species and are flushed from estuaries with faster flow rates (Ferreira et al. 2005; Bricker et al. 2008). The slow/very slow flow-rate systems

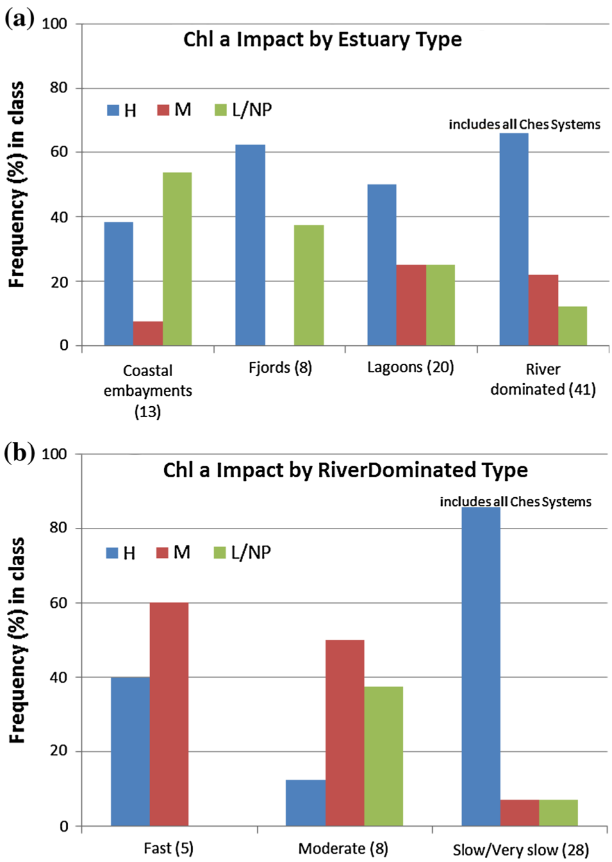


Fig. 10 Chlorophyll *a* impact by **a** types of systems, and **b** flow rate in river-dominated systems where *H* high, *M* moderate, *L/NP* low/no problem. Note that for fast-flow systems, there are no high impacts (from: Bricker et al. 2007) among the river-dominated: fast = hours to 2 days; moderate = 3 days to 1 week; slow/very slow ≥10 days (*Numbers in parentheses* in both **a** and **b** are the number of systems included in the group)

have the highest impacts for both chlorophyll *a* and eutrophic condition (Fig. 10b). There was a distinct pattern of residence times among the three different flow-rate groups for which chlorophyll *a* impacts were rated as moderate or high. The fast-flow systems had residence times of two days or less, moderate flow systems had rates from three to seven days, and slow/very slow flow systems had residence times greater than ten days.

There were not adequate data to evaluate the relationship to TN loading among these same systems but there was a pattern among population densities, which can be used as a proxy for load; we used the number of people per km^2 of water area. The fast-flow systems had an average population density of nine people per km^2 of estuarine water area. Moderate flow systems had a population density of 28 per km^2, and slow/very slow flow systems had a population density of 100 per km^2 or greater. While not statistically tested, and with the caveat that results may be confounded by systems of low population density and high agricultural load, results of our analysis appear to provide a rough predictor of when moderate-to-high chlorophyll *a* impact (and by extension, high eutrophic impact) might be expected for river-dominated systems: slow flow (residence times greater than ten days) and high population densities (>100 people per km^2 estuarine area). Our results are particularly relevant to Chesapeake Bay region estuaries, which have high watershed population density and for which most (i.e., 7 of 9 systems) have land use that is >40 % urban and agricultural.

5.3 Nutrient Removal by Shellfish Aquaculture: Can Shellfish Aquaculture Save the Potomac River Estuary?

To meet water-quality criteria established for the PRE, Jaworski et al. (2007) estimate that a reduction of 50 % of 1985 base year TN loads would be required and that nonpoint sources would require reductions of 54–65 %, in addition to the continued reductions of TN from POTW effluent. Point-source improvements become increasingly more expensive as limits of technology are approached, i.e., hundreds of millions of dollars for a single plant (Rose et al. 2014). For this reason, alternative cost-effective nutrient management measures are being explored to complement traditional land-based measures, shellfish aquaculture among them (Stephenson et al. 2010; Rose et al. 2014).

5.3.1 FARM Simulation for the Potomac River Estuary

A 2007 study by the Maryland Oyster Advisory Commission led to the 2009 passage of the Maryland Shellfish Aquaculture Plan to promote a sustainable shellfish industry and to facilitate the expansion of aquaculture (MD Senate Bill 271; MD House Bill 312). There also is an ongoing effort to restore oyster reefs in the Chesapeake Bay region (i.e., MD Native Oyster Restoration Plan). While oyster cultivation is not yet part of the nutrient-trading program in Chesapeake Bay, there is discussion and research on the use of oyster aquaculture as a best management practice [Scientific and Technical Advisory Committee (STAC) 2013], and the USEPA Regional Ecosystem Services Program and NOAA are supporting research to investigate the potential removal of nitrogen through oyster harvest. Further, in the 2012 Session of the General Assembly of Virginia, aquaculture was added to the list of potential nutrient controls or removal practices that could receive nutrient credit certification (House Bill No. 176, referred to Committee on Agriculture, Chesapeake and Natural Resources).

Results of the FARM model application at PRE station RET2.4 show removal rates (0.23×10^3 kg N removed acre^{-1} year^{-1}) similar to removal rates reported in other studies of N removal through aquaculture harvest ($0.285–1.984 \times 10^3$ kg N removed acre^{-1} year^{-1}; Rose et al. 2014; NB: the shellfish remove N as particulate N). Upscaled to existing oyster habit, harvest would remove 0.856×10^6 kg N year^{-1}, while the cultivation of half of the estimated suitable bottom area would result in removal of 13×10^6 kg N year^{-1}. The upscaled N removal for the larger area is equal to an ecosystem service value for nutrient treatment for 3.9×10^6 people, more than half of the present population of the PRE watershed, and is equal to almost 50 % of the TN input to the PRE (27.8×10^6 kg N year^{-1} in 2008–2009). To remove the total current nitrogen load to the PRE would require cultivation at the same oyster densities of 120×10^3 acres (39% estuarine area), close to the estimated area of bottom that is considered suitable for cultivation (112×10^3 acres; N. Carlozo, MD DNR, pers. comm., 2013). A cultivated area of 8.7×10^3 acres, an area 2–3 times the existing oyster habitat area, could completely remove the estimated 2.00×10^6 kg N year^{-1} of N expected to be discharged from POTWs (i.e. not including other sources) once they are operationally "fine tuned" (Jaworski et al. 2007).

These model results have not been compared to reported harvests; however, the results for the simulated farm are similar to typical harvests seen in the Chesapeake region and elsewhere (D. Webster, UMD, pers. comm., 2013). Likewise, the N removal rates are within ranges of removal seen elsewhere (Rose et al. 2014). Additionally, the removal rates are compared only to TN inputs from upstream (riverine and POTW inputs) and may overestimate the total removal if there are influxes of TN from the Chesapeake Bay mainstem. Finally, there are no leases at present, though there may be in the future, but it is unlikely that such a large area of the PRE would ever be cultivated because of conflicting uses.

It is useful to compare these model results to estimated removal rates through denitrification, one of the benefits of restoring oyster reefs, either through increasing substrate available for the settlement of oyster larvae or transplanting oysters that were produced and set on shell at a hatchery to suitable bottom areas. A recent study of measured denitrification rates in a restored oyster reef in the Choptank River shows N removal through denitrification of 0.23×10^3 kg N acre^{-1} of restored reef year^{-1} (Kellogg et al. 2013), the same as our estimated removal rates through harvest. If we assume that the bottom culture with no gear is, for the approximate 3-year culture cycle duration, similar to the action of a restored reef, we can expect that the area in cultivation would also provide N removal through denitrification. Thus, the upscaled area of 56.0×10^3 acres would remove 13×10^3 kg N year^{-1}. Combined with the N removed through harvest, the cultivated area would remove a total of about 26×10^3 kg N year^{-1}, 93 % of the estimated load to the PRE from upstream sources; note that this may overestimate the removal from the total load if there are influxes from the Chesapeake Bay mainstem.

Additional studies are needed to confirm and refine our modeling results. We have not considered the potential impact of sea-level rise, which modeling indicates would increase salinity in Chesapeake Bay estuaries (e.g., Rice et al. 2012). The Eastern oyster has a wide salt tolerance ($\sim$4 to 27 psu; Loosanoff 1952); thus, increased salinity might improve their growth and extend their range farther upstream with salinity intrusion into freshwater areas where they presently cannot grow. This suggests that in the PRE, oyster habitat might expand, thus increasing the future ecosystem services provided by natural, cultivated, and restored oysters.

On the basis of our results, the most expedient way to reduce eutrophication in the PRE would be continued land-based reductions complemented by a combination of aquaculture and restoration of oyster reefs. This combination could provide significant removal of nutrients and eutrophication impacts directly from the water column, boosting the effect of traditional land-based management measures and offering innovative solutions to long-term and persistent nutrient-related water-quality problems, as well as providing oyster product.

6 Conclusions

- Contribution to total nitrogen loads to the Potomac River Estuary, taking into account all headwater area, represents about 2 % of loads in the early 1990s. Although the percentage estimated is small, the current loads from forested headwaters are not known because monitoring of headwater streams for nitrogen loads has been discontinued in forested watersheds except in Shenandoah National Park; hence, data are lacking to track changes in water quality related to atmospheric deposition. Restarting monitoring at discontinued sites and continued monitoring at active sites would provide the data needed for assessing nitrogen inputs.

- Eutrophication status of the Potomac River Estuary shows that high nitrogen loads and high-level impacts have not changed overall since the early 1990s, though there are some signs of improvement (i.e., increased dissolved oxygen and decreased chlorophyll *a* in the tidal fresh zone; continued regrowth of sea grasses).
- The Potomac River Estuary eutrophication assessment results are representative of Chesapeake Bay region estuaries and US estuaries; more than half of US estuaries have moderate- to high-level eutrophication.
- River-dominated systems with high watershed population density (i.e., >100 people per km^2) with slow flow (>10 days residence time) and >40 % of land in urban and agricultural uses, such as the Potomac River Estuary, are likely to have high eutrophication impact.
- A simulation of shellfish aquaculture at a site in the lower Potomac River Estuary showed that shellfish filtration in a simulated farm could remove less than 2 % of total nitrogen inputs but expansion to one-half of suitable bottom area would remove an equivalent (13×10^6 kg N $year^{-1}$) to an ecosystem service of nutrient treatment for 3.9×10^6 people, more than 50 % the present population of the Potomac River Estuary watershed.
- Cultivation or restoration of about 40 % of the bottom area of Potomac River Estuary, almost all of which is deemed suitable to bottom culture, would remove the current estimated total nitrogen load.
- Using recently measured rates of denitrification in another Chesapeake tributary, a calculation of potential N removal from one-half of estimated suitable bottom area suggests that an additional 13 x 10^6 kg nitrogen $year^{-1}$ would be removed assuming bottom oyster culture acts the same as a restored reef.
- An area 2–3 times the area of existing oyster habitat could remove the total nitrogen input from Privately Owned Treatment Works once the treatment works are "fine tuned."
- These results are promising with respect to nitrogen removal via aquaculture and reef restoration, which would be most significant if applied together; additional research is needed for the confirmation of results and for the exploration of capacity to transfer to other water bodies.

Acknowledgments We dedicate this paper to Owen P. Bricker III, father (SBB), dear friend, and wonderful mentor (KCR), a great teacher and scientist, oyster lover, champion of the Chesapeake Bay, and one of the originators of the present Chesapeake Bay Program. Although he died in March 2011, his legacy remains in those of us whose lives he helped shape. We would like to thank the following people for their generous sharing of data and information that made our analyses possible: Renee Karrh (MD DNR), Peter Tango (USGS CBPO), Steve Gill (NOAA CO-OPS), Bob Paul (St. Mary's College of Maryland) and Norb Jaworski (retired, USEPA) for data and insight; Brian Russell, Mandy Burch, Kevin Boyle and Sheldon Russell (Shore Thing Seafood) and Kelly Greenhawk, Maude Livings, Katie Busch (MD DNR), Don Webster (UMD) for culture practice information; Joao Ferreira (Longline Environment, Ltd.) for use of the FARM model; Greg Piniak and Julie Rose (NOAA), and Jason Price (Millersville University) for review comments; Dave Whitall (NOAA) for review comments and map construction; Jason Pope (USGS) for map data; Erik Davenport and Annie Jacob (NOAA) for statistical analyses; and Nicole Carlozo (MD DNR) for bottom culture suitable area estimates. We also thank the insightful reviewers who greatly improved the paper. Thanks to our editor-in-chief, George Luther, III. Special thanks to our handling and Associate Editor, Fred Mackenzie for helpful guidance and for suggesting this tribute volume.

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