Maryland’s Upper Eastern Shore Summary: A summary of trends in tidal water quality and associated factors, 1985-2018.

June 7, 2021

Prepared for the Chesapeake Bay Program (CBP) Partnership by the CBP Integrated Trends Analysis Team (ITAT)

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This tributary summary is a living document in draft form and has not gone through a formal peer review process. We are grateful for contributions to the development of these materials from the following individuals: Jeni Keisman, Rebecca Murphy, Olivia Devereux, Jimmy Webber, Qian Zhang, Meghan Petenbrink, Tom Butler, Zhaoying Wei, Jon Harcum, Renee Karrh, Mike Lane, and Elgin Perry.

Contents

1. Purpose and Scope

The Maryland Upper Eastern Shore Tributary Summary outlines change over time in a suite of monitored tidal water quality parameters and associated potential drivers of those trends for the time period 1985 – 2018, and provides a brief description of the current state of knowledge explaining these observed changes. Water quality parameters described include surface (above pycnocline) total nitrogen (TN), surface total phosphorus (TP), spring and summer (June, July, August) surface chlorophyll *a*, summer bottom (below pycnocline) dissolved oxygen (DO) concentrations, and Secchi disk depth (a measure of water clarity). Results for annual surface water temperature, bottom TP, bottom TN, surface ortho-phosphate (PO4), surface dissolved inorganic nitrogen (DIN), surface total suspended solids (TSS), and summer surface DO concentrations are provided in an Appendix. Drivers discussed include physiographic watershed characteristics, changes in TN, TP, and sediment loads from the watershed to tidal waters, expected effects of changing land use, and implementation of nutrient management and natural resource conservation practices. Factors internal to estuarine waters that also play a role as drivers are described including biogeochemical processes, physical forces such as wind-driven mixing of the water column, and biological factors such as phytoplankton biomass and the presence of submersed aquatic vegetation. Continuing to track water quality response and investigating these influencing factors are important steps to understanding water quality patterns and changes in Maryland's Upper Eastern Shore.

2. Location

The Maryland Upper Eastern Shore watershed covers approximately 1.5% of the Chesapeake Bay watershed. It's watershed is approximately 2,441 km^2 (Table 1.) and is contained within two states: Maryland and Delaware (Figure 1).

Tributary Name	Watershed Area km2
MARYLAND MAINSTEM	71967
POTOMAC	36611
JAMES	25831
YORK	6537
RAPPAHANNOCK	6530
LOWER EASTERN SHORE	4532
MARYLAND UPPER EASTERN SHORE	2441
PATUXENT	2236
VIRGINIA MAINSTEM	2052
CHOPTANK	1844
PATAPSCO-BACK	1647
MARYLAND UPPER WESTERN SHORE	1523
MARYLAND LOWER WESTERN SHORE	439

Table 1. "Watershed areas for each of the thirteen tributary or tributary groups for which Tributary Trends summaries have been produced. All of the tributary summaries can be accessed at the following link: https://cast.chesapeakebay.net/Home/TMDLTracking#tributaryRptsSection".

2.1 Watershed Physiography

The Maryland Upper Eastern Shore watershed extends across two major physiographic regions, Coastal Plain and Piedmont (Bachman *et al.*, 1998) (Figure 1). This Coastal Plain physiography covers lowland, dissected upland, and upland areas. The Piedmont physiography includes crystalline areas. Implications of these physiographies for nutrient and sediment transport are summarized in Section 5.1.1.

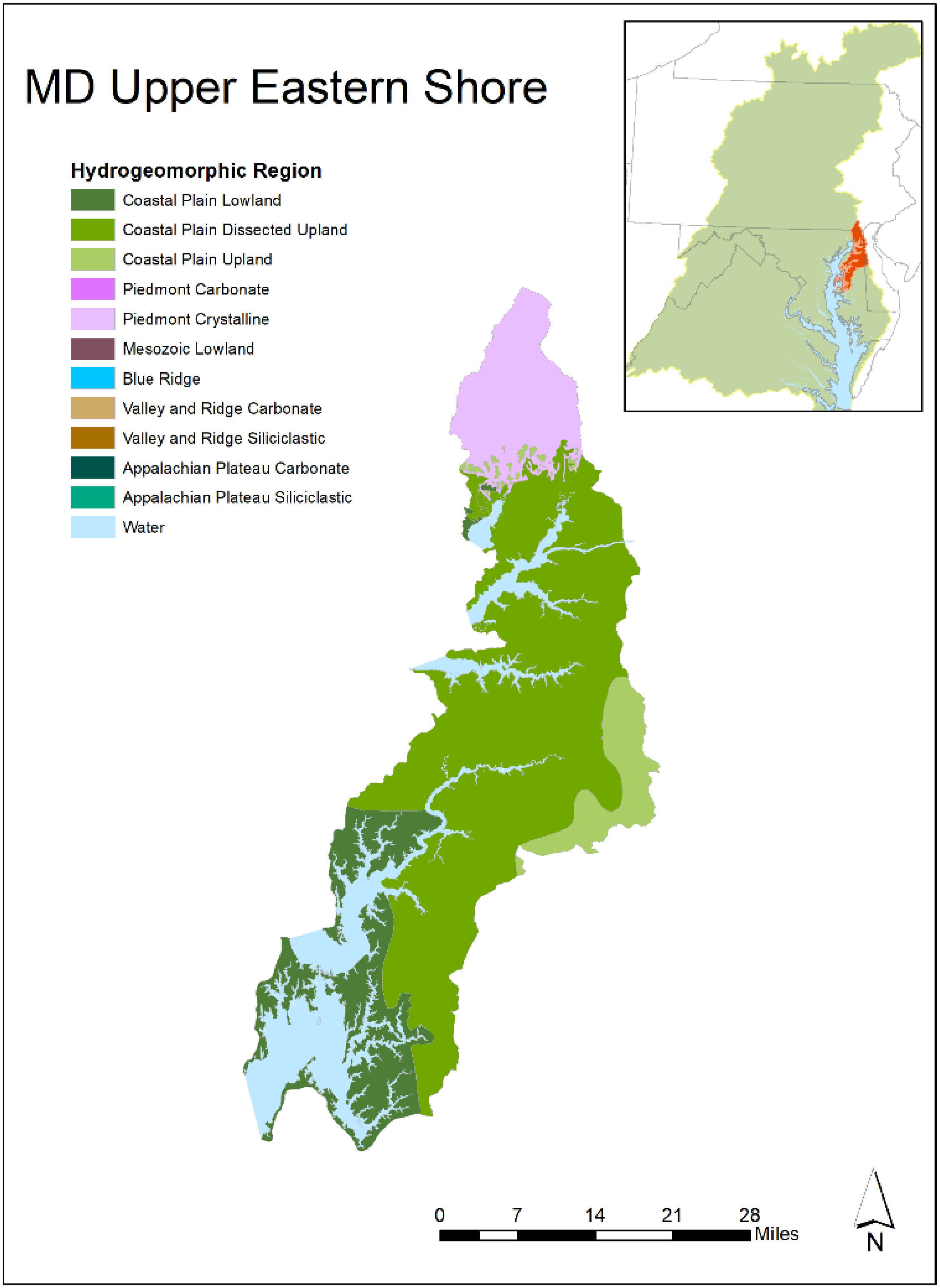

Figure 1. Distribution of physiography in the Maryland Upper Eastern Shore watershed. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

2.2 Land Use

Land use in the Maryland Upper Eastern Shore watershed is dominated (48%) by agriculture areas. Urban and suburban land areas have increased by 39,362 acres since 1985, agricultural lands have decreased by 24,822 acres, and natural lands have decreased by 14,439 acres. Correspondingly, the proportion of urban land in this watershed has increased from 9% in 1985 to 15% in 2019 (Figure 2).

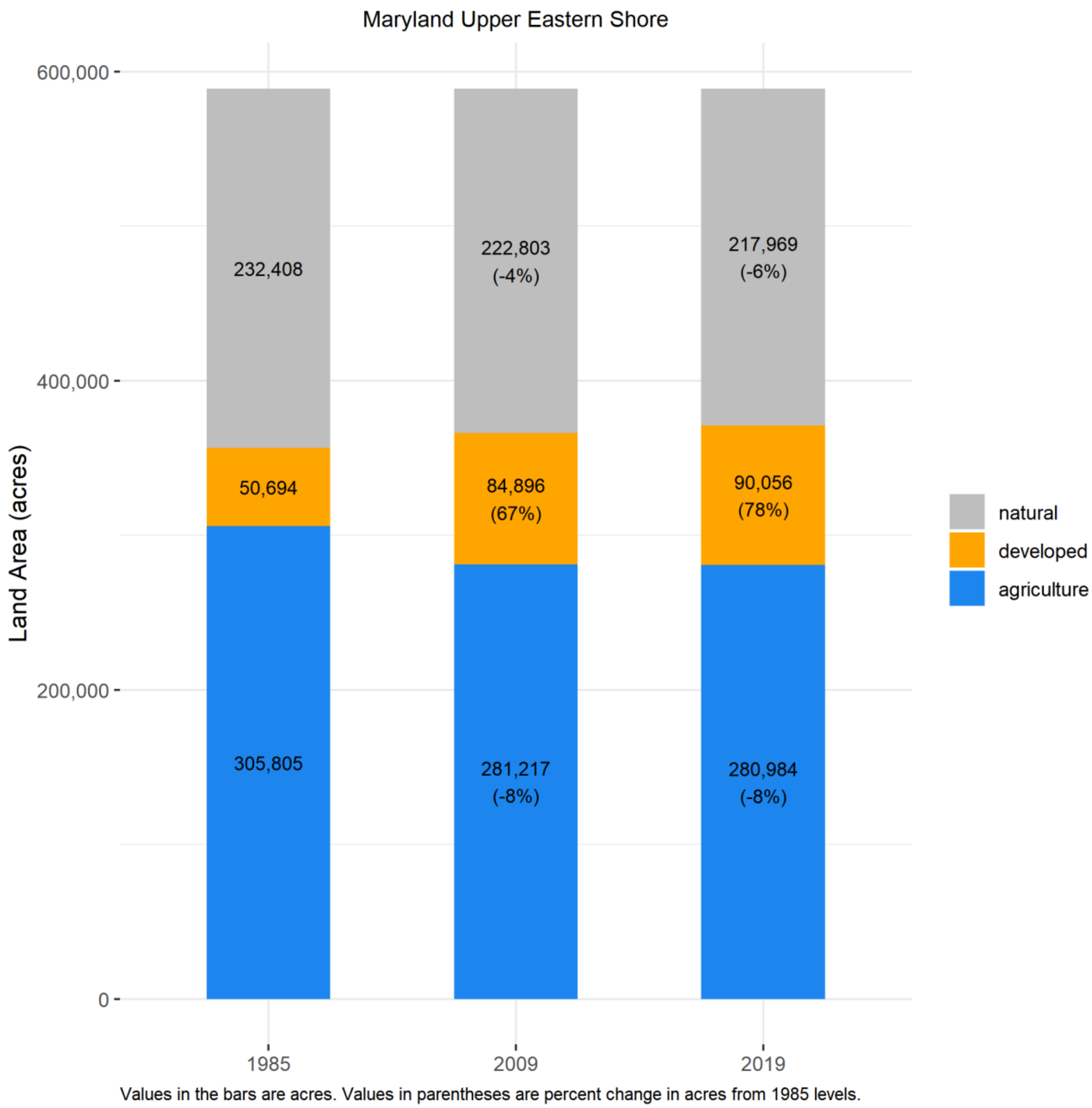

Figure 2. Distribution of land uses in the Maryland Upper Eastern Shore watershed. Percentages are the percent change from 1985 for each source sector.

In general, developed lands in the 1970s were concentrated within towns and major metropolitan areas. Since then, developed and semi-developed lands have increased around these areas, as well as expanding into previously undeveloped regions (Figure 3). The impacts of land development differ depending on the use from which the land is converted (Keisman *et al.*, 2019; Ator *et al.*, 2019). Implications of changing land use for nutrient and sediment transport are summarized in Section 5.1.3.

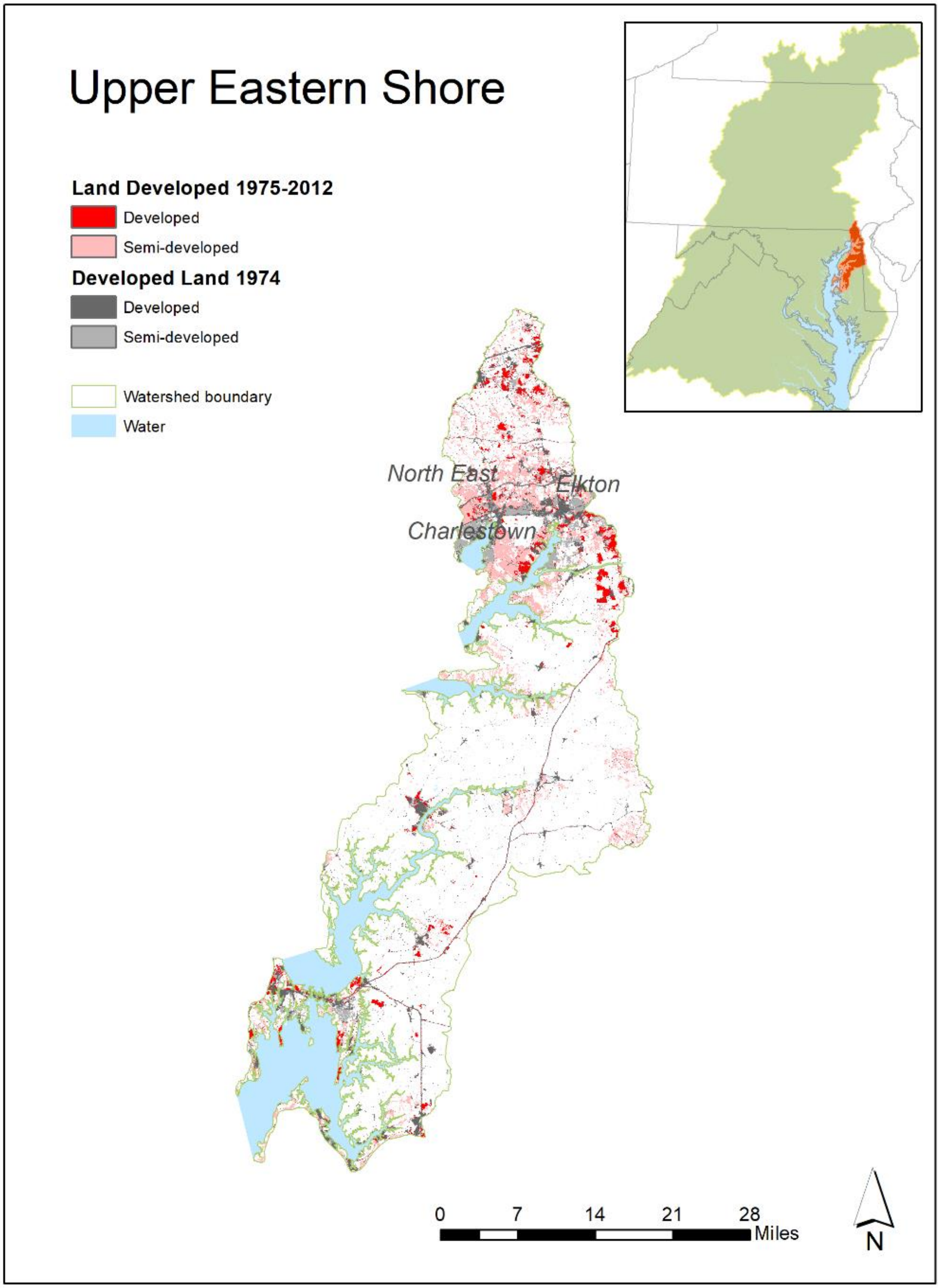

Figure 3. Distribution of developed land in the Maryland Upper Eastern Shore rivers watershed. Derived from Falcone (2015). Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

2.3 Tidal Waters and Stations

For the purposes of water quality standards assessment and reporting, the tidal portions of the MD Upper Eastern Shore Tributaries are divided into 10 segments (U.S. Environmental Protection Agency, 2004) (Figure 4). The Chesapeake & Delaware canal is split into Maryland and Delaware segments (C&DOH_MH and C&DOH_DE). The Tidal Fresh Northeast River (NORTF), Oligohaline Bohemia River (BOHOH), Oligohaline Elk River (ELKOH), and Oligohaline Sassafras River (SASOH) are all grouped in the northern part of this region. The Chester River is split into three segments: Tidal Fresh (CHSTF), Oligohaline (CHSOH) and Mesohaline (CHSMH). And south of the Chester River, Eastern Bay is included in this group with one Mesohaline segment (EASMH).

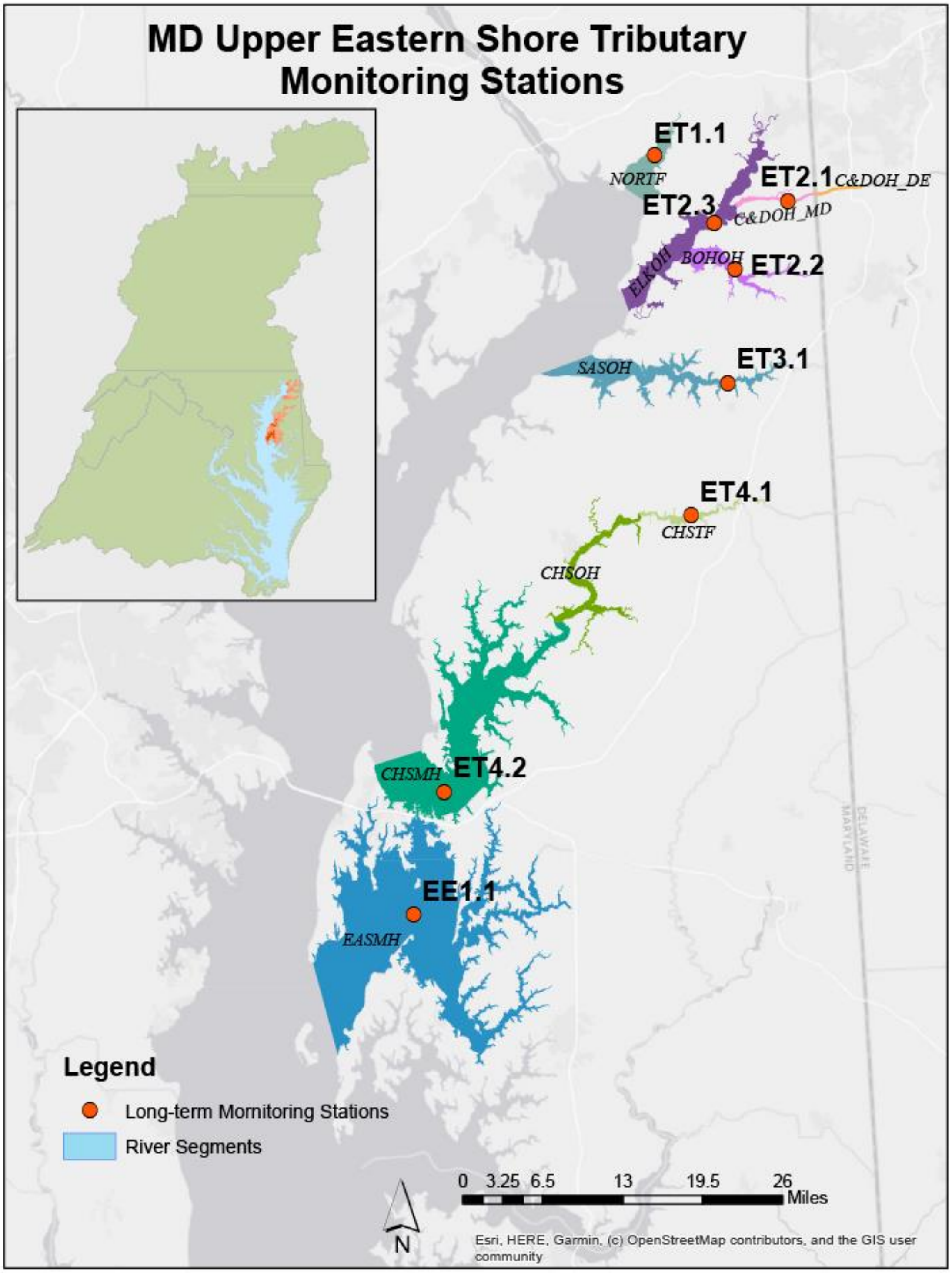

Figure 4. Map of tidal MD Upper Eastern Shore river segments and long-term monitoring stations. Base map credit Esri, HERE, Garmin, (c) OpenStreetMap contributors, and the GIS user community, World Geodetic System 1984.

Long-term trends in water quality are analyzed by MDDNR at eight stations with one in each of the segments except for the Delaware portion of the canal and the Chester oligohaline (Figure 4). Water quality data at these stations are also used to assess attainment of dissolved oxygen (DO) water quality criteria. All tidal water quality data analyzed for this summary are available from the Chesapeake Bay Program Data Hub (Chesapeake Bay Program, 2018). Other shorter-term monitoring has been conducted in these tributaries by MDDNR over time, but is not included in the long-term trend graphics because of the shorter durations.

3. Tidal Water Quality Dissolved Oxygen Criteria Attainment

Multiple water quality standards were developed for the Upper Eastern Shore tributaries to protect aquatic living resources (U.S. Environmental Protection Agency, 2003; Tango and Batiuk, 2013). These standards include specific criteria for dissolved oxygen (DO) and water clarity/underwater bay grasses. For the purposes of this summary, a record of the evaluation results indicating whether each of the tributary segments have met or not met a 30 day or instantaneous Open Water (OW), Deep Water (DW), and Deep Channel (DC) DO criteria over time is shown below (Zhang *et al.*, 2018a; Hernandez Cordero *et al.*, 2020). While analysis of water quality standards attainment is not the focus of this summary, the results (Tables 2 and 3) provide context for the importance of understanding factors affecting water quality trends. For more information on water quality standards, criteria, and standards attainment, visit the CBP's "Chesapeake Progress" website at www.chesapeakeprogress.com. In the recent period (2016-2018), only two of the 10 segments did not met the 30-day mean OW summer DO requirements, both in the Chester River (CHSTF and CHSOH). Neither of the segments with DW or DC requirements met them (Zhang *et al.*, 2018b).

Table 2. Open Water summer DO criterion evaluation results (30-day mean June-September assessment period). Green indicates that the criterion was met. White indicates that the criterion was not met.

time period	NORTF	ELKOH	C&DOH _DE	C&DOH _MD	BOHOH	SASOH	CHSTF	CHSOH	CHSMH	EASMH
1985-1987	Green	Green	Green	Green	Green	White	Green	Green	Green	Green
1986-1988	Green	Green	Green	Green	Green	White	Green	Green	Green	Green
1987-1989	Green	Green	Green	Green	Green	White	White	Green	Green	Green
1988-1990	Green	Green	Green	Green	Green	White	White	Green	Green	Green
1989-1991	Green	Green	Green	Green	Green	White	White	Green	Green	Green
1990-1992	Green	Green	Green	Green	Green	White	Green	Green	Green	Green
1991-1993	Green	Green	Green	Green	Green	White	Green	Green	Green	Green
1992-1994	Green	Green	Green	Green	White	White	Green	Green	Green	Green
1993-1995	Green	Green	Green	Green	White	White	Green	Green	White	Green
1994-1996	Green	Green	Green	Green	White	White	White	Green	Green	Green
1995-1997	White	Green	Green	Green	Green	White	White	Green	Green	Green
1996-1998	White	Green	Green	Green	Green	White	White	Green	Green	Green
1997-1999	White	Green	Green	Green	White	White	Green	Green	Green	Green
1998-2000	Green	Green	Green	Green	White	White	White	White	Green	Green

1999-2001										
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2014-2016										
2015-2017										
2016-2018										

Table 3. Deep Water summer DO (30-day mean) criteria evaluation results. Green indicates that the criterion was met. White indicates that the criterion was not met.

time period	Deep Water		Deep Channel	
	CHSMH	EASMH	CHSMH	EASMH
1985-1987				
1986-1988				
1987-1989				
1988-1990				
1989-1991				
1990-1992				
1991-1993				
1992-1994				
1993-1995				
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2005-2007				
2006-2008				
2007-2009				
2008-2010				

2009-2011				
2010-2012				
2011-2013				
2012-2014				
2013-2015				
2014-2016				
2015-2017				
2016-2018				

Comparing trends in station-level DO concentrations to the computed DO criterion status for a recent assessment period can reveal valuable information, such as whether progress is being made towards attainment in a segment that is not meeting the water quality criteria, or conversely the possibility that conditions are degrading even if the criteria are currently being met. To illustrate this, the 2016-2018 attainment status for the OW summer and DC instantaneous DO criteria shown in Tables 2 and 3 are overlain with the 1985-2018 change in summer surface DO concentration and the 1985-2018 change in bottom summer DO concentrations, respectively (Figure 5). The 30-day mean OW summer DO criterion was met in the northern group of segments where the surface DO trends are all either improving or not changing. The Chester River tidal fresh and oligohaline segments are not meeting the 30-day mean OW summer DO criterion, and the DO is degrading. In the Eastern Bay mesohaline segment (EASMH) the OW criterion is met, but the surface DO is degrading. In addition, the two mesohaline segments are not meeting the DC instantaneous criterion and the bottom DO concentrations are either not changing (CHSMH) or degrading (EASMH). Taken together, these results for the Chester River and Eastern Bay segments suggest DO conditions are degrading in that region.

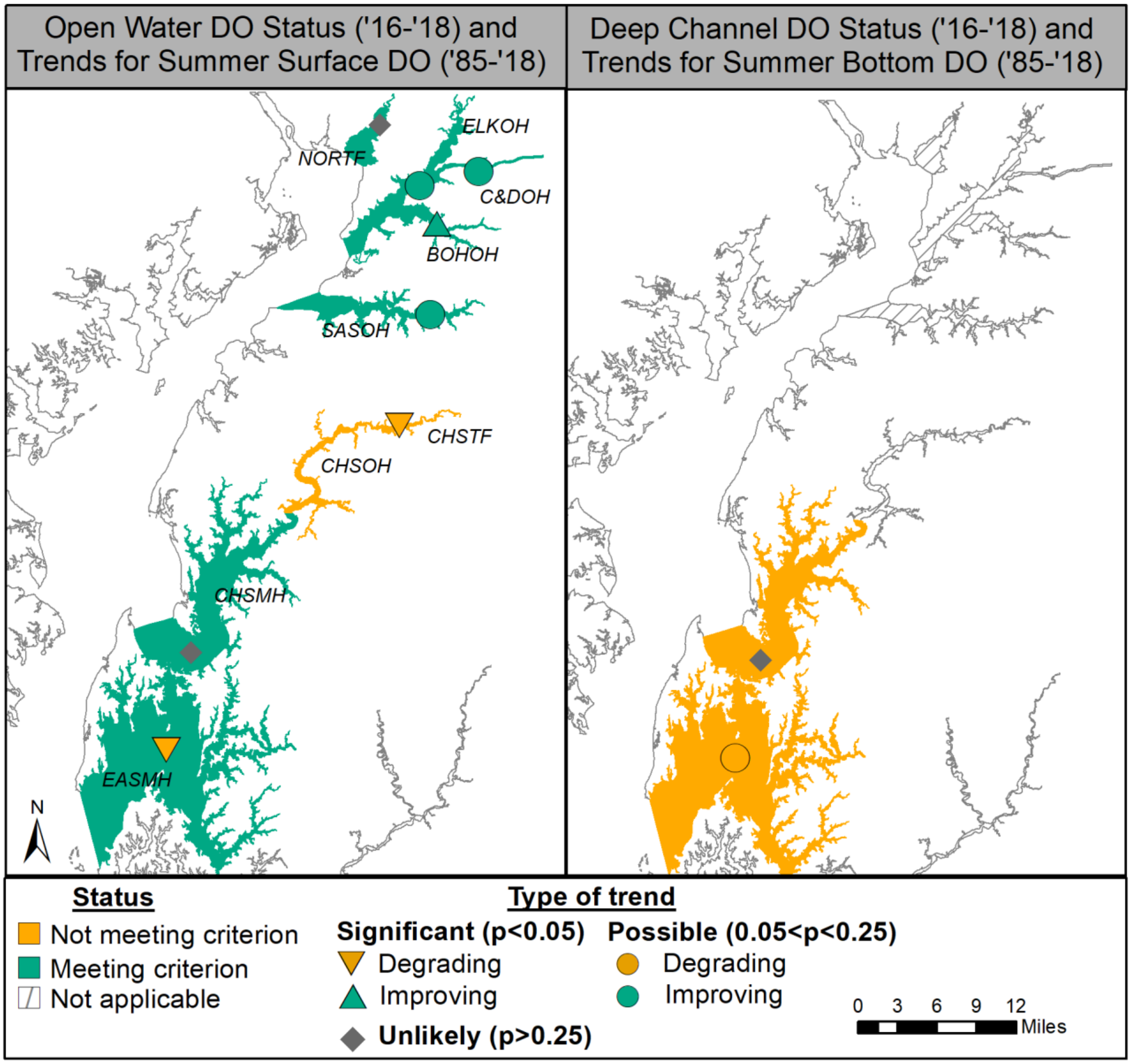


Figure 5. Pass-fail DO criterion status for 30-day OW summer DO and instantaneous DC summer DO designated uses in Upper Eastern Shore segments along with long-term trends in DO concentrations. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

4. Tidal Water Quality Trends

Tidal water quality trends are computed by fitting generalized additive models (GAMs) to the water quality observations that have been collected one or two times per month since the 1980s at the eight Upper Eastern Shore tidal stations labeled in Figure 4. For more details on the GAM implementation that is applied each year by MD Department of Natural Resources for these stations in collaboration with the Chesapeake Bay Program and Virginia analysts, see Murphy *et al.* (2019).

Results shown below in each set of maps (e.g., Figure 6) include those generated using two different GAM fits to each station-parameter combination. The first approach involves fitting a GAM to the raw observations to generate a mean estimate of the concentrations over time, as observed in the estuary. The second approach involves including monitored river flow or *in situ* salinity (as an aggregated measure of multiple river flows) in the GAM to explain some of the variation in the water quality parameter. From the results of this second approach, it is possible to estimate the "flow-adjusted" change over time, which gives a mean estimate of what the water quality parameter trend would have been if river flow had been average over the period of record. Note that depending on station and parameter, sometimes gaged river flow is used for this adjustment and sometimes salinity is used, but we refer to all these results as "flow-adjusted" for simplicity.

To determine if there has been a change over time (i.e., a trend) at a particular station for a given parameter, we compute a percent change between the estimates at beginning and end of a period of interest from the GAM fit. For each percent change computation, the level of statistical confidence can be computed as well. Change is called significant if $p < 0.05$ and possible if the p-value is up to 0.25. That upper limit is higher than usually reported for hypothesis tests but allows us to provide a more complete picture of the results, identifying locations where change might be starting to occur and should be investigated (Murphy *et al.*, 2019). In addition to the maps of trends, for each parameter, there is a set of graphs (e.g., Figure 7) that include the raw observations (dots on the graphs) and lines representing the mean annual or seasonal GAM estimates, without flow-adjustment. The flow-adjusted GAM line graphs are not shown.

4.1 Surface Total Nitrogen

Annual total nitrogen (TN) trends have improved over the long-term at four of the eight tidal Upper Eastern Shore stations, using non-flow-adjusted results (Figure 6). When the trends are flow-adjusted, five stations have improving TN trends. Over the short-term, some degrading trends appear in the northern cluster of stations while possible improvements are occurring at ET4.1 and ET4.2 in the Chester River after flow-adjustment.

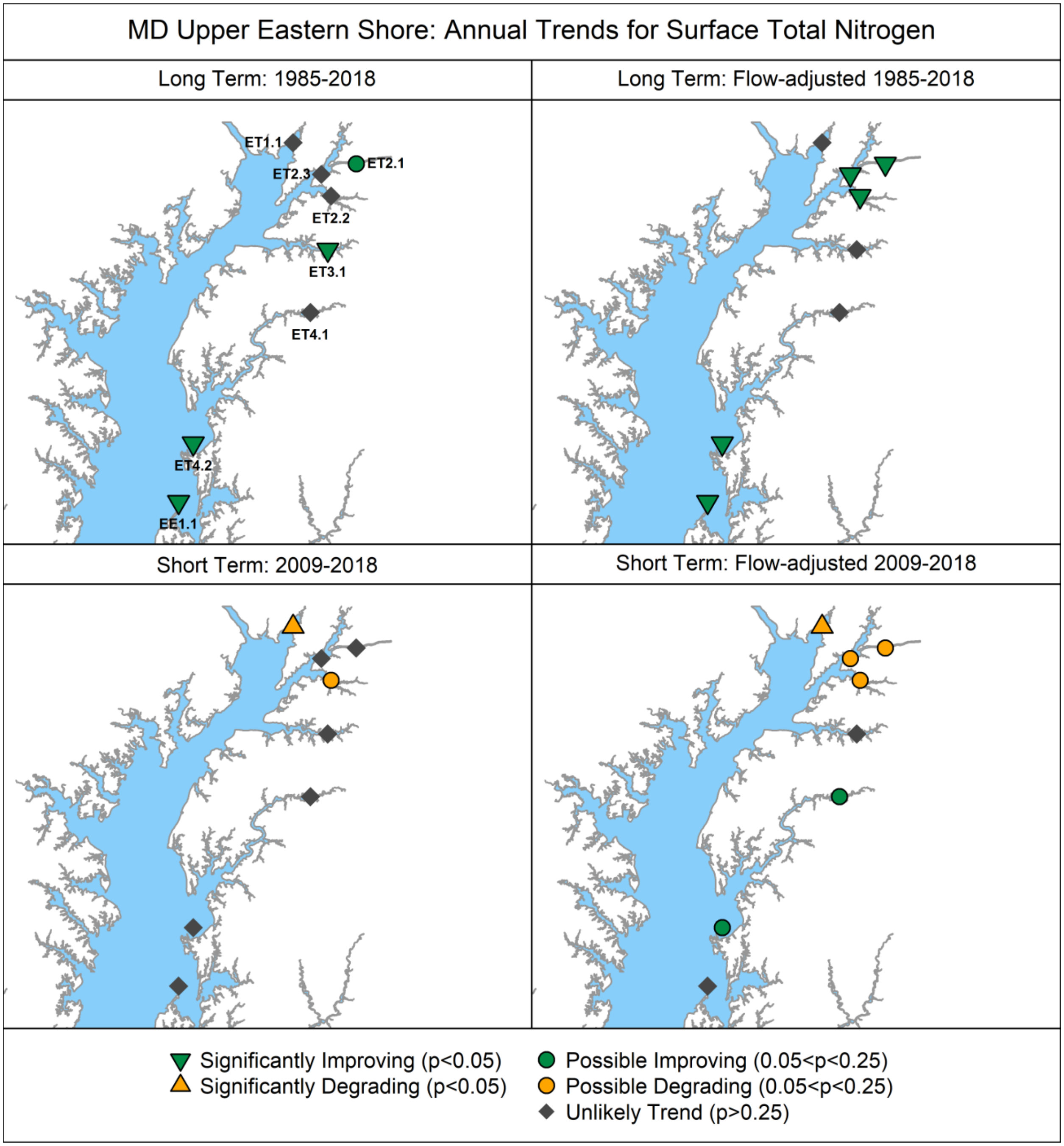


Figure 6. Surface TN trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

Long-term TN trends appear slight in the graphs of the data and the non-flow-adjusted mean annual GAM estimates presented in Figure 7, and variability due to river flow is evident in the time series. A slight upswing at the end of some of the timeseries is responsible for the short-term degrading trends in Figure 6. Vertical blue dotted lines represent a laboratory and method change (May 1, 1998) that was tested for its impact on data values. A statistical intervention test within the GAM models showed that these changes were significant at most stations. This is evident by the vertical jump in the mean annual GAM estimates shown with the lines. With this technique, we can estimate long-term change after accounting for the artificial jump from the method change (Murphy *et al.*, 2019).

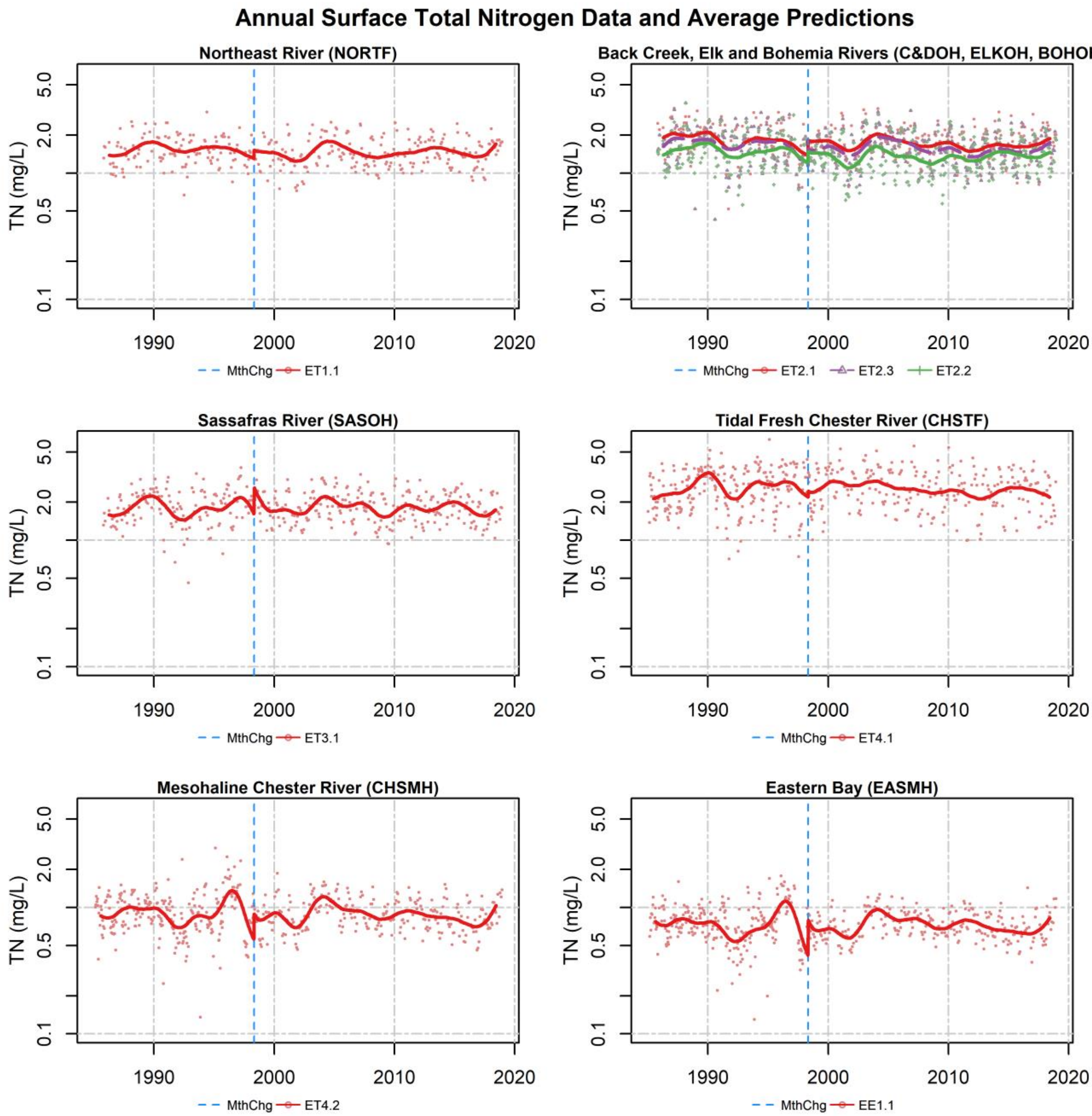


Figure 7. Surface TN data (dots) and average long-term pattern generated from non-flow adjusted GAMs. Colored dots represent data corresponding to the monitoring station shown indicated in the legend; colored lines represent mean annual GAM estimates for the noted monitoring stations.

4.2 Surface Total Phosphorus

Surface total phosphorus (TP) trends are improving at all stations over the long term without flow-adjustment (Figure 6). After adjustment, the majority of the stations still show improving TP trends. In the short-term, there is only one improving trend (ET1.1) and several degrading trends with and without flow adjustment.

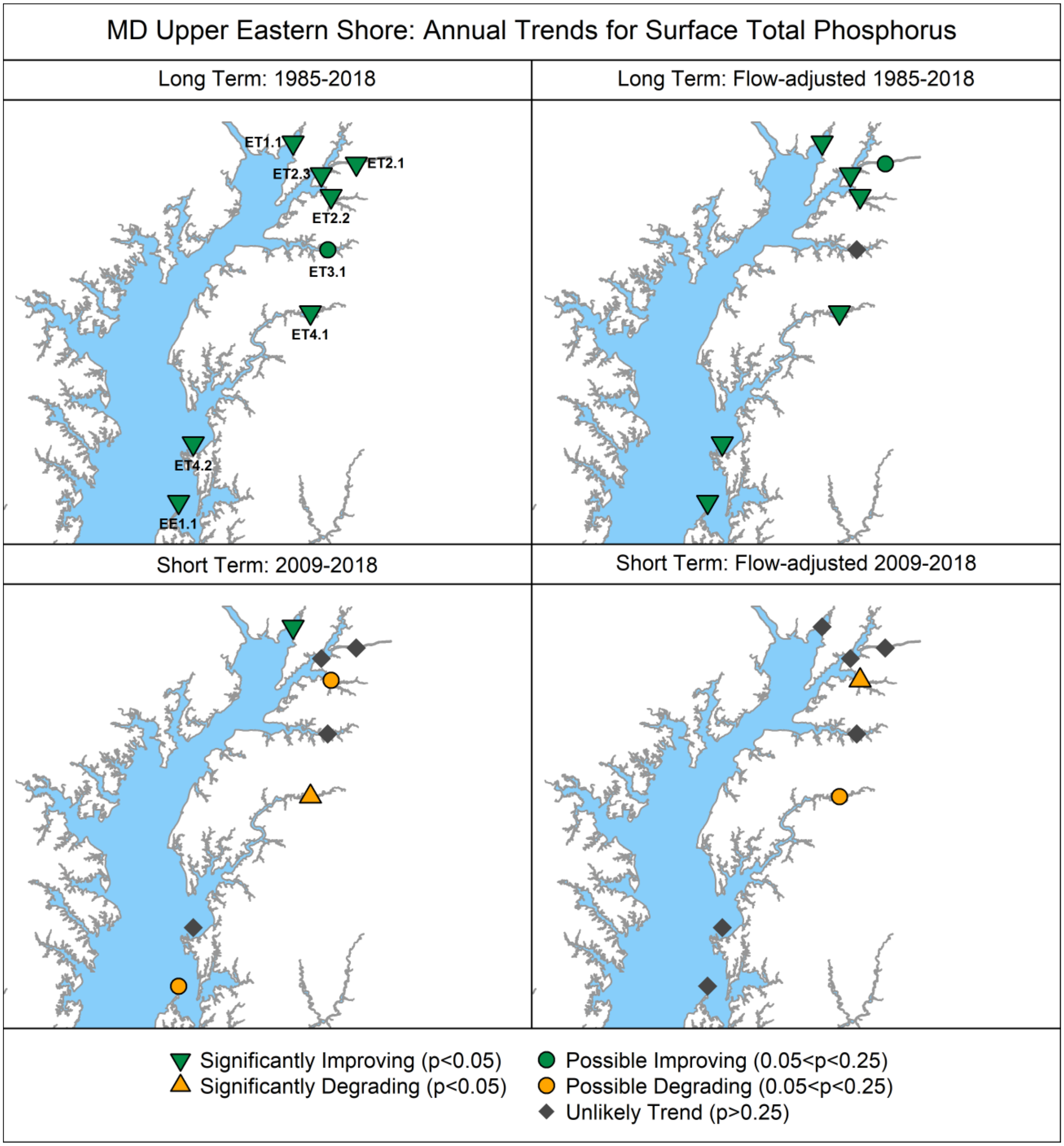


Figure 8. Surface TP trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

A decrease in concentrations throughout the entire record is clear at ET1.1 in the Northeast River (Figure 9) where long- and short-term improving trends were found (Figure 8). The long-term improvements at most other stations appear to be due to decreasing concentrations in the early part of the record (Figure 9). The recent degrading trends at ET2.2 and ET4.1 can be seen as a change in direction of the long-term GAM patterns at both those stations

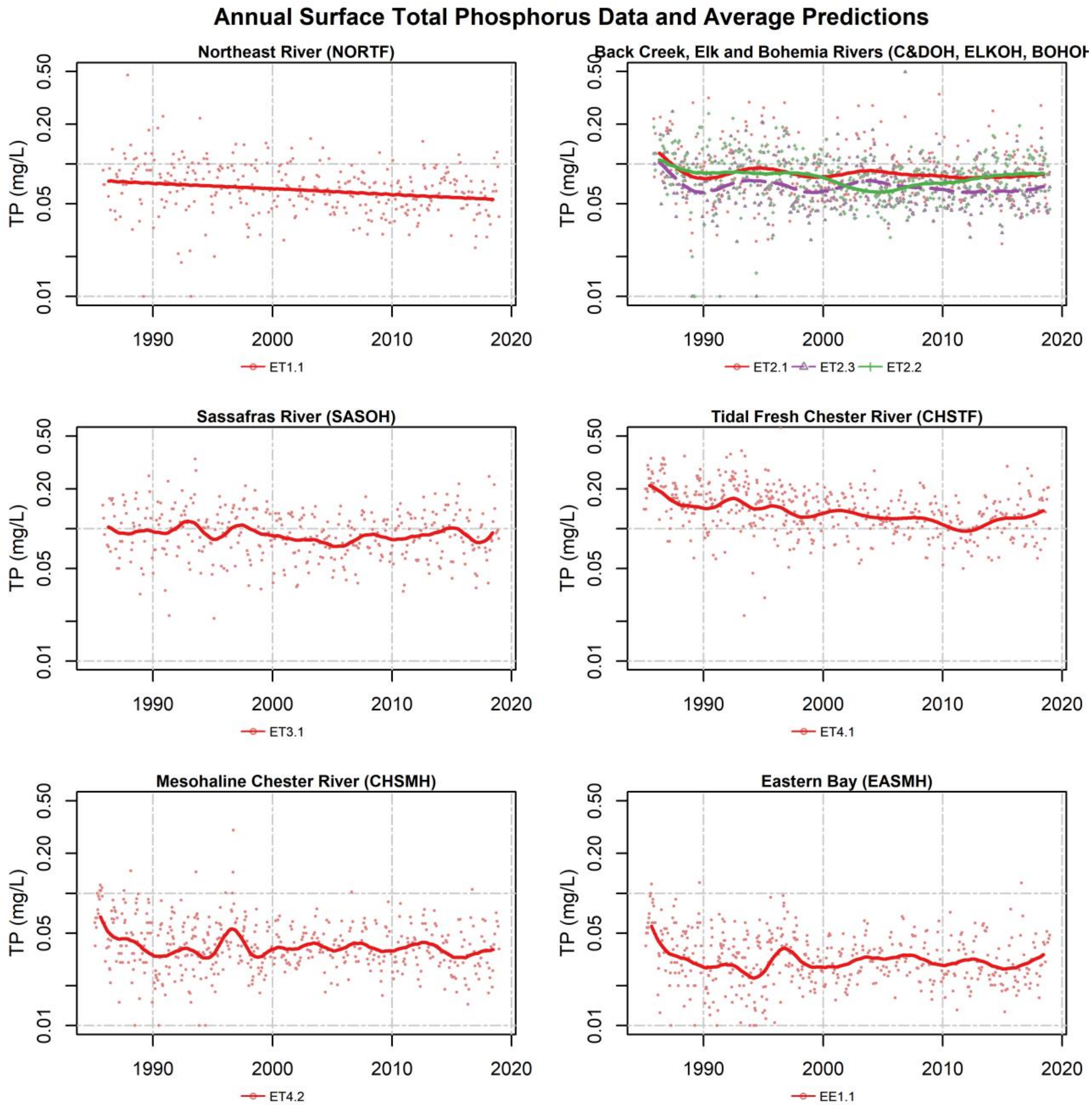


Figure 9. Surface TP data (dots) and average long-term pattern generated from non-flow adjusted GAMs. Colored dots represent data corresponding to the monitoring station shown indicated in the legend; colored lines represent mean annual GAM estimates for the noted monitoring stations.

4.3 Surface Chlorophyll *a:* Spring (March-May)

Trends for chlorophyll *a* are split into spring and summer to analyze chlorophyll *a* during the two seasons when phytoplankton blooms are commonly observed in different parts of Chesapeake Bay (Smith and Kemp, 1995; Harding and Perry, 1997). Spring trends (Figure 10) are mixed, with more improving than degrading trends. There are very few differences in the trends after flow-adjustment. In the short-term, ET4.2 is significantly improving, otherwise there are several possible improvements and mostly no trends at these stations.

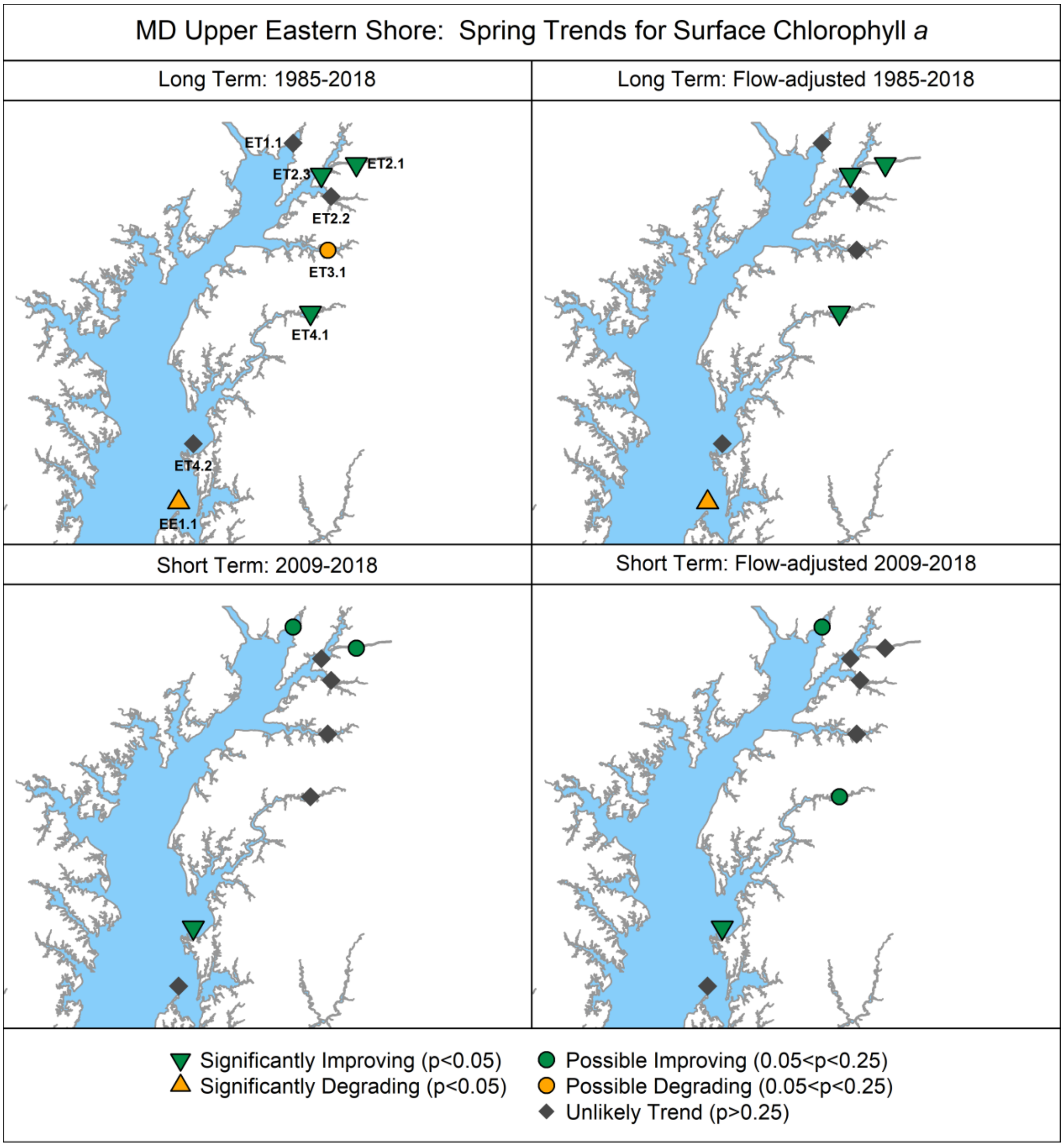


Figure 10. Surface spring (March-May) chlorophyll *a* trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

The long-term patterns in spring chlorophyll *a* vary greatly among these stations (Figure 11), following from the mixed long-term trends (Figure 10). Some clear patterns are a recent step-down in concentrations and GAM estimates at ET4.1 in the tidal fresh Chester River, and a steady long-term increase at EE1.1 in Eastern Bay.

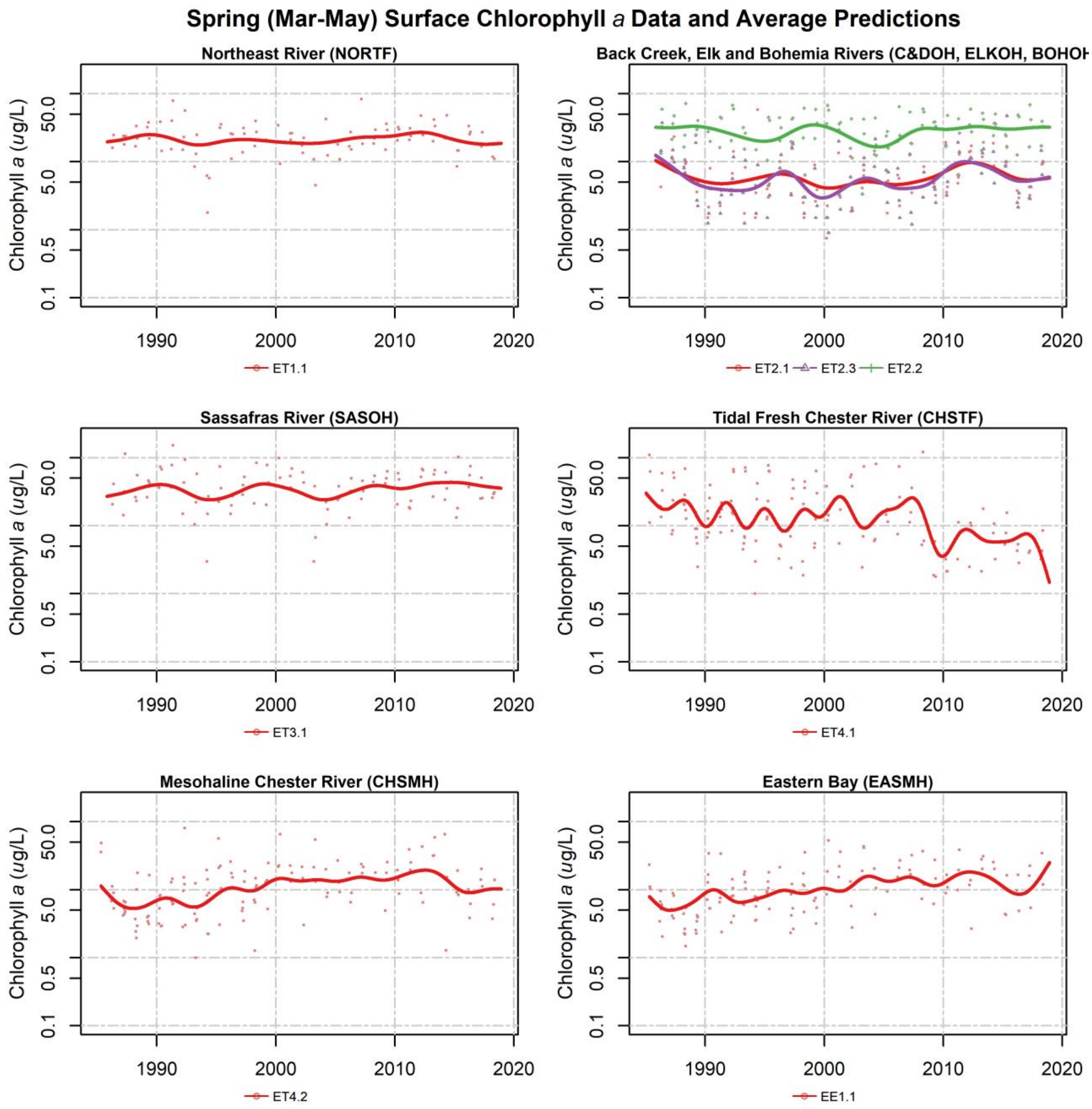


Figure 11. Surface spring chlorophyll *a* data (dots) and average long-term pattern generated from non-flow adjusted GAMs. Colored dots represent March-May data corresponding to the monitoring station

indicated in the legend; colored lines represent mean spring GAM estimates for the noted monitoring stations.

4.4 Surface Chlorophyll *a:* Summer (July-September)

The spatial patterns in summer long-term chlorophyll *a* trends (Figure 12) are fairly similar to spring trends (Figure 10). There is a mixture of improving and degrading trends over the long-term without flow-adjustment. With flow-adjustment the degrading trends are mostly explained. Over the short-term, there are several possible improvements, including the tidal fresh Chester River (ET4.1).

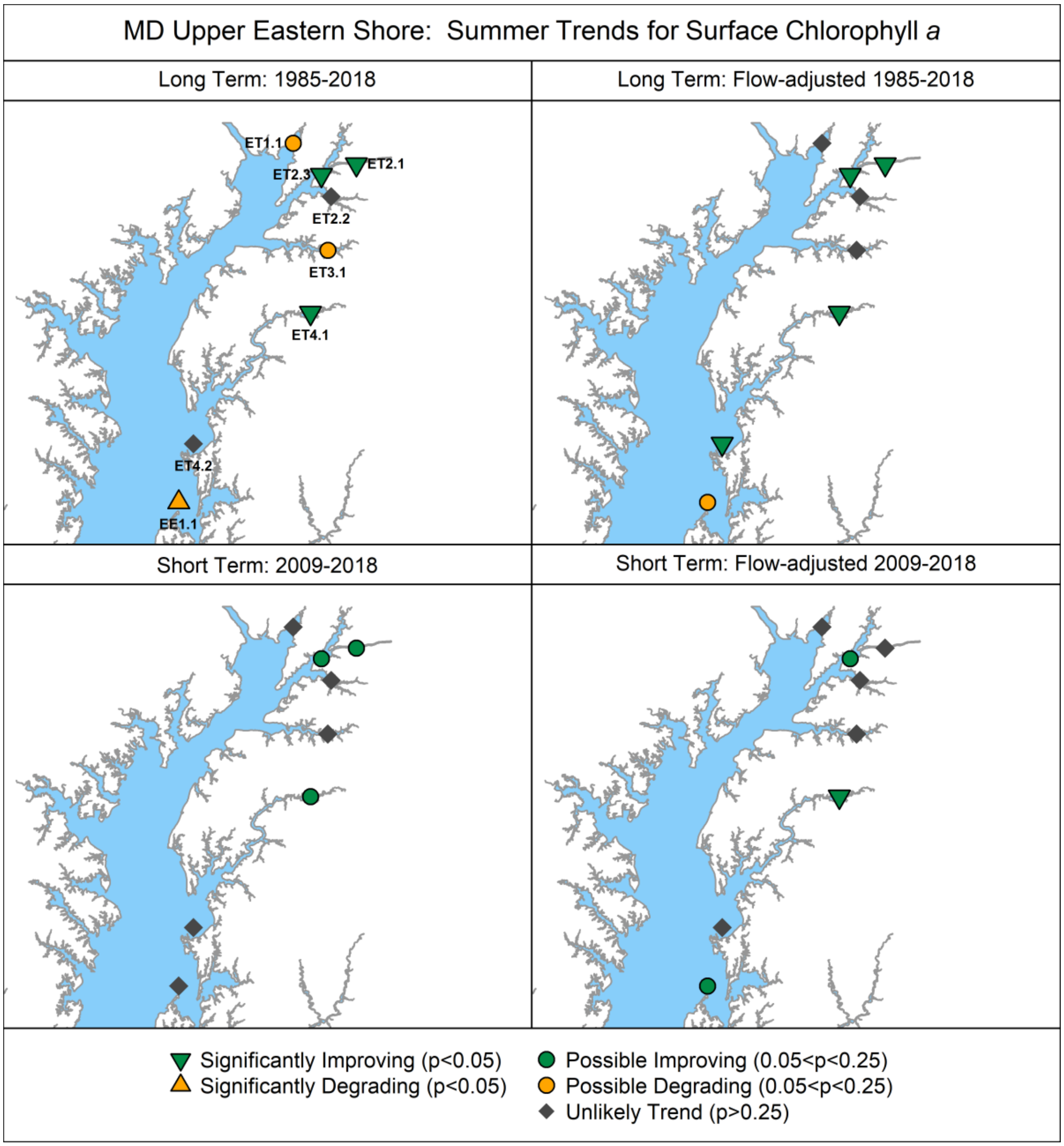


Figure 12. Surface summer (July-September) chlorophyll *a* trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

The patterns in mean summer GAM estimates (Figure 13) are very similar to the spring patterns (Figure 11) although the summer concentrations are higher than the spring concentrations at the majority of the stations. Most of the stations with summer chlorophyll *a* trends (Figure 12) appears to have fairly small changes in concentrations causing those trends (Figure 13). Exceptions are at the same stations mentioned above for the spring. There is a large decrease in the tidal fresh Chester River (ET4.1) and a fairly clear increase in Eastern Bay (EE1.1) over time.

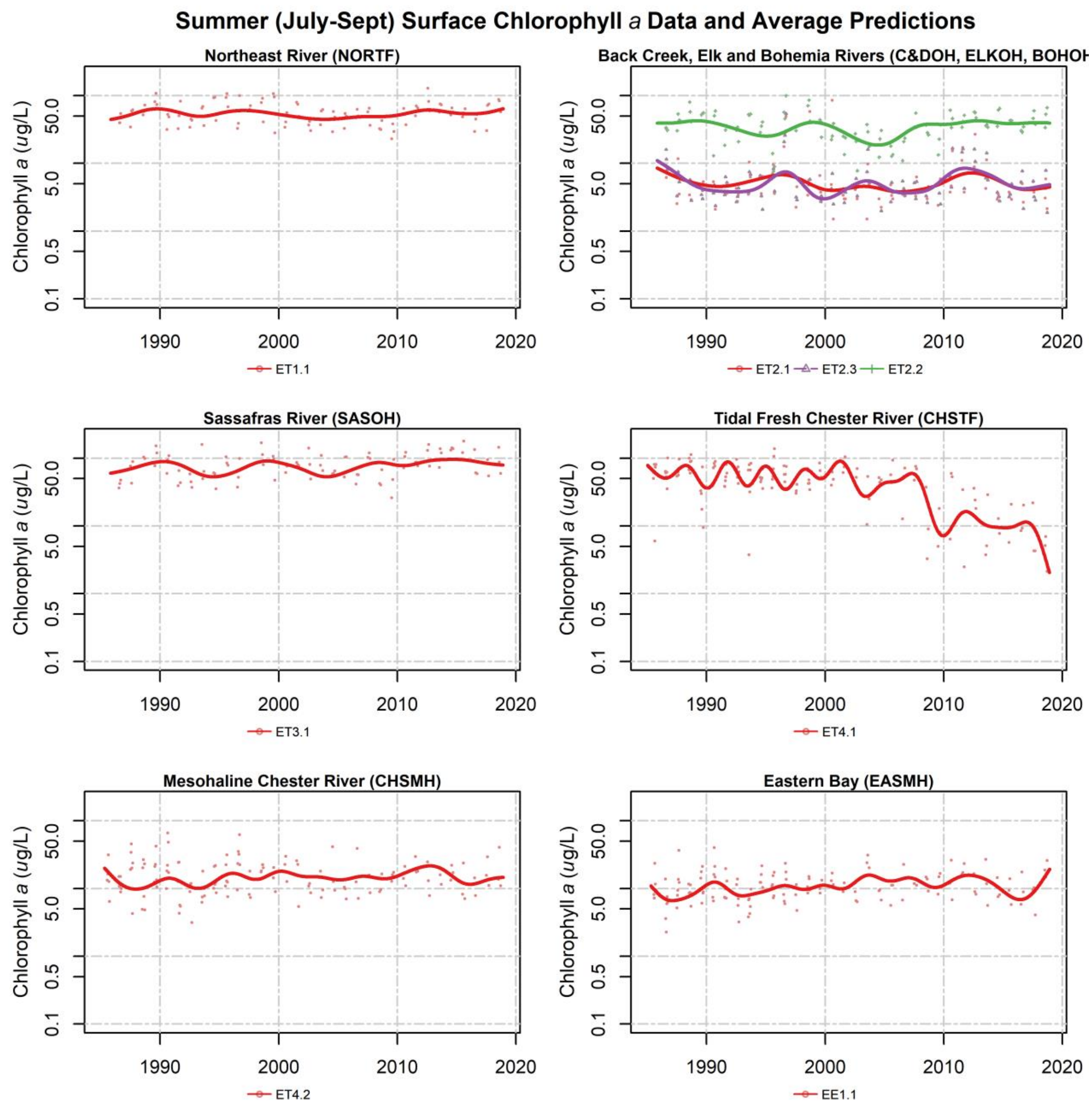


Figure 13. Surface summer chlorophyll *a* data (dots) and average long-term pattern generated from non-flow adjusted GAMs. Colored dots represent July-September data corresponding to the monitoring station indicated in the legend; colored lines represent mean summer GAM estimates for the noted monitoring stations.

4.5 Secchi Disk Depth

Trends in Secchi disk depth, a measure of visibility through the water column, are degrading over the long-term at ET4.2 (Chester River mesohaline) and EE1.1 (Eastern Bay), with and without flow-adjustment. The trend at ET4.1 (tidal fresh Chester River) is improving over the long-term, but otherwise there are no long-term Secchi trends. Over the short-term, there are a few possible trends but mostly no trends at the Upper Eastern Shore stations.

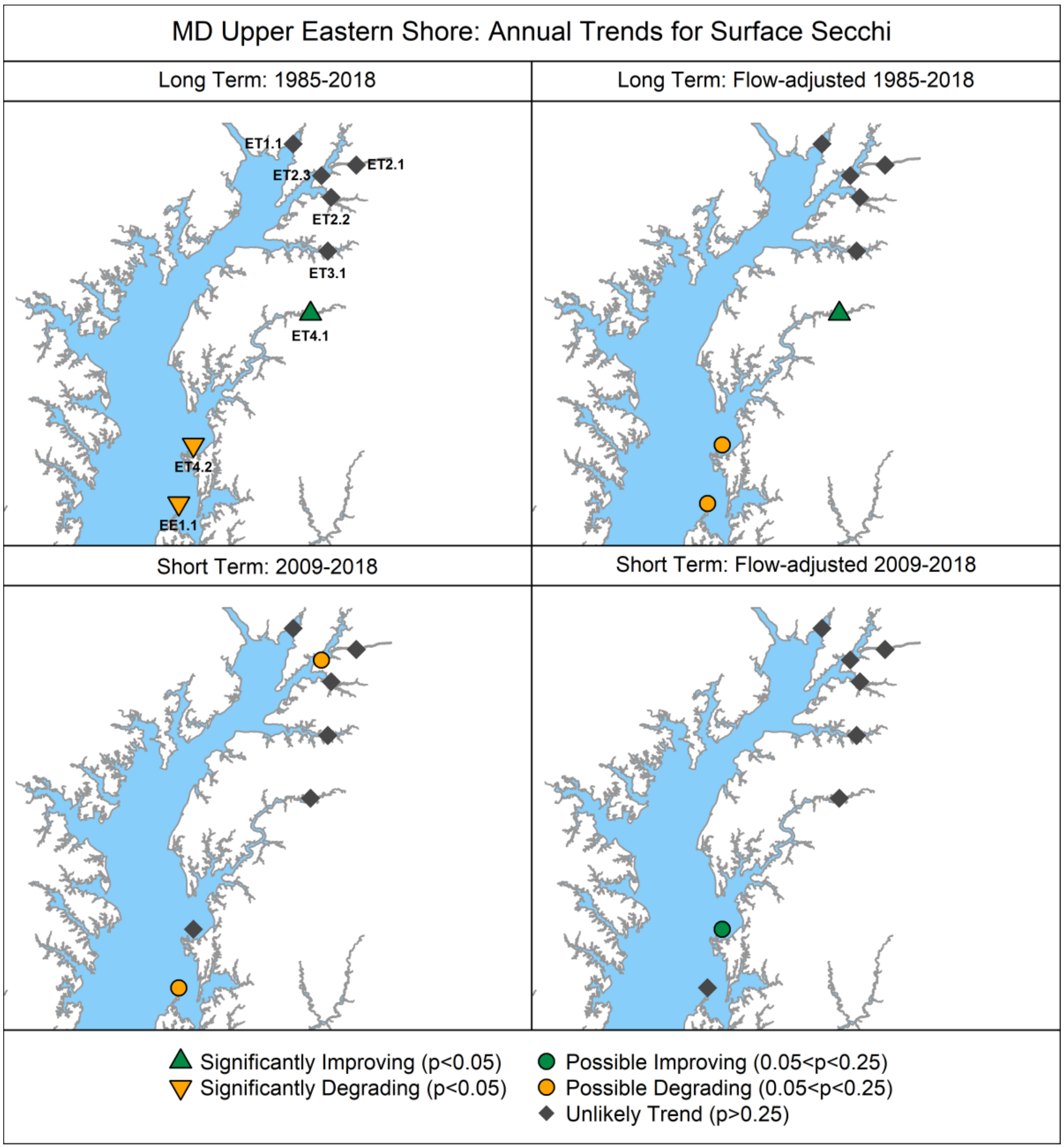


Figure 14. Annual Secchi depth trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

Secchi depth is generally much shallower in the tidal fresh and oligohaline Upper Eastern Shore stations, than in the mesohaline stations (Figure 15). Notably the mesohaline stations (ET4.2 and EE1.1) are the ones with long-term degrading trends, which appear to be mostly due to a decrease from large Secchi observations in the early part of the record. The improvement at ET4.1 (tidal fresh Chester) is clear with an increase in the mean annual GAM estimates, despite the very low magnitude of the observations at that station.

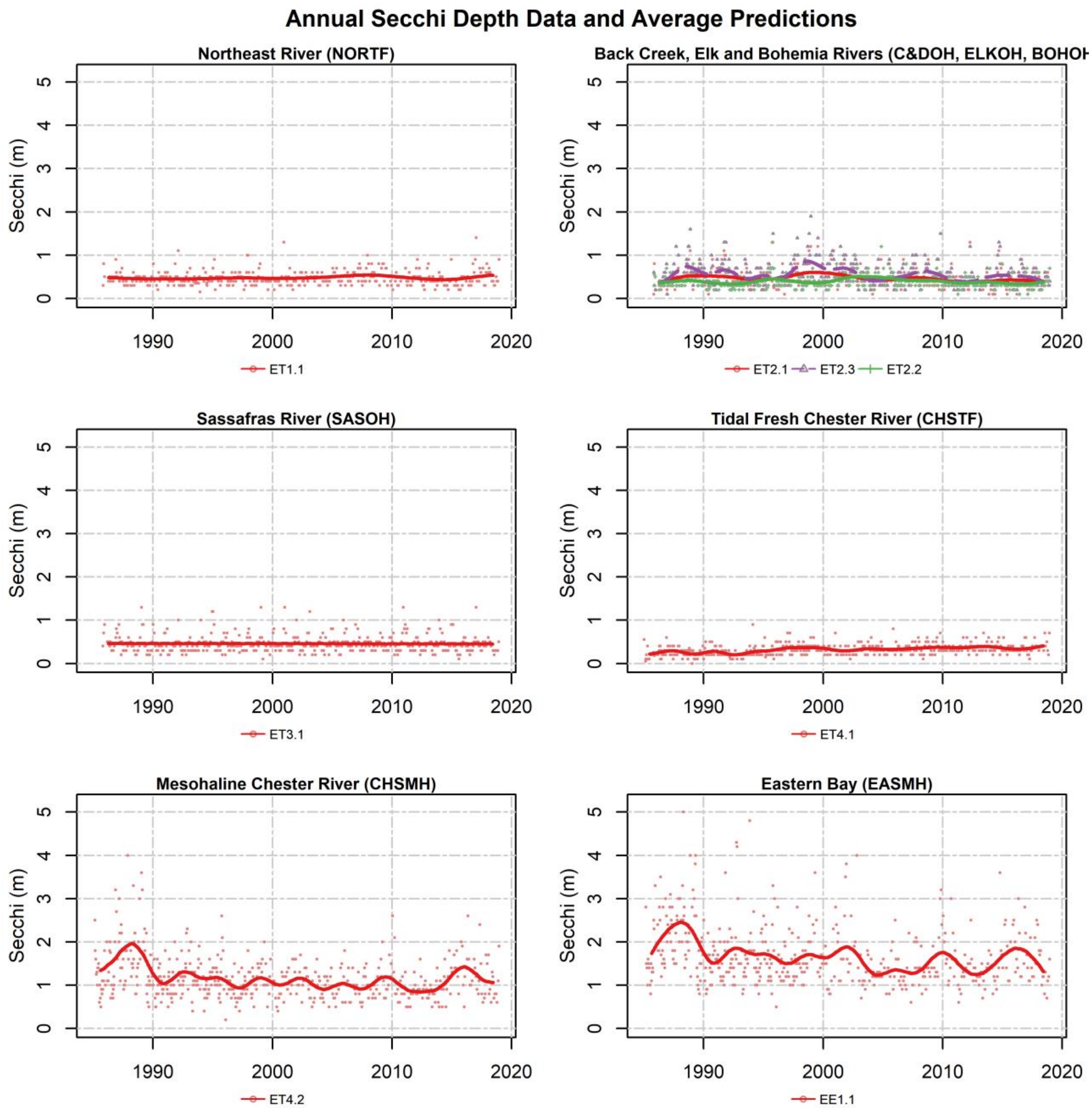


Figure 15. Annual Secchi depth data (dots) and average long-term pattern generated from non-flow

adjusted GAMs. Colored dots represent data corresponding to the monitoring station shown indicated in the legend; colored lines represent mean annual GAM estimates for the noted monitoring stations.

4.6 Summer Bottom Dissolved Oxygen (June-September)

Bottom summer oxygen trends are mixed with some improving trends in the northern tributaries over the long-term and some degrading trends in the more southern stations in this group (Figure 16). Short-term improvements without flow-adjustment are clustered in the northern tributaries and ET4.2, and a short-term possible degradation has occurred at ET4.1.

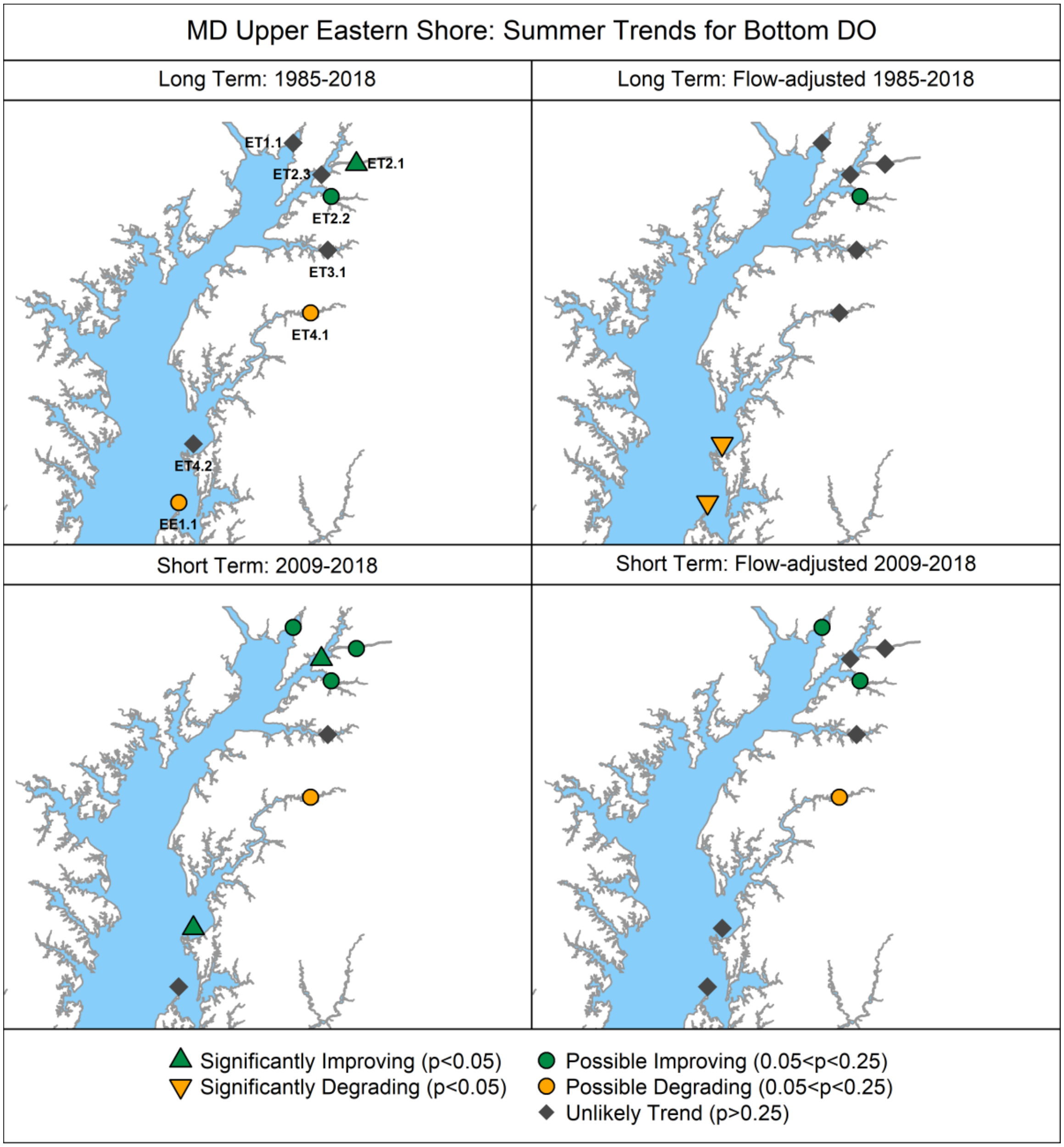


Figure 16. Summer (June-September) bottom DO trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

Plots of the summer data and average summer GAM estimates demonstrate the spatial variability in bottom DO concentrations at these stations (Figure 17). Concentrations at the tidal fresh and oligohaline stations are much higher than the mesohaline stations. The short-term decrease at ET4.1 is clear as a dip in concentrations at that station. This is consistent with the sharp decrease in spring and summer chlorophyll *a* concentrations at this station at the same time (Figures 11 and 13) and does suggest an event occurred to impact water quality near that station. The two deeper water stations clearly show a decline over the long-term, although the mesohaline Chester River DO concentrations are increasing in the last few years (Figure 17).

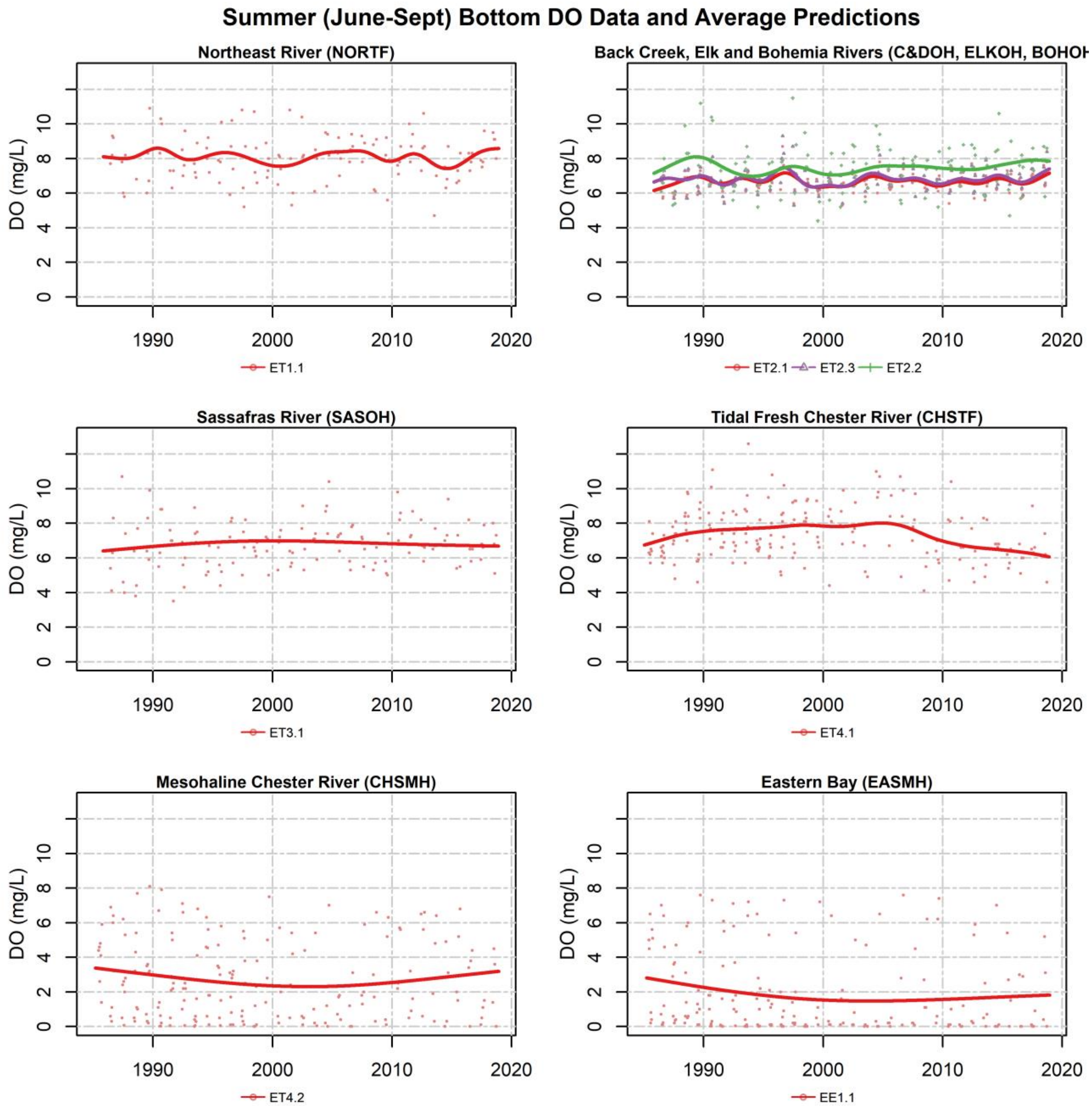


Figure 17. Summer (June-September) bottom DO data (dots) and average summer long-term pattern generated from non-flow adjusted GAMs. Colored dots represent data corresponding to the monitoring

station shown indicated in the legend; colored lines represent mean annual GAM estimates for the noted monitoring stations.

5. Factors Affecting Trends

5.1 Watershed Factors

5.1.1 Effects of Physical Setting

Large nitrogen and phosphorus loads occur throughout the Eastern Shore because unique combinations of hydrogeology, topography, and soils promote the efficient transport of agricultural-associated nutrient applications to streams and tidal waters (Figure 18) (Brakebill *et al.*, 2010; Ator *et al.*, 2011; Ator *et al.*, 2019; Ator *et al.*, 2020; Noe *et al.*, 2020). Sediment loads are typically low throughout the Eastern Shore because of the relatively flat topography of the Atlantic Coastal Plain.

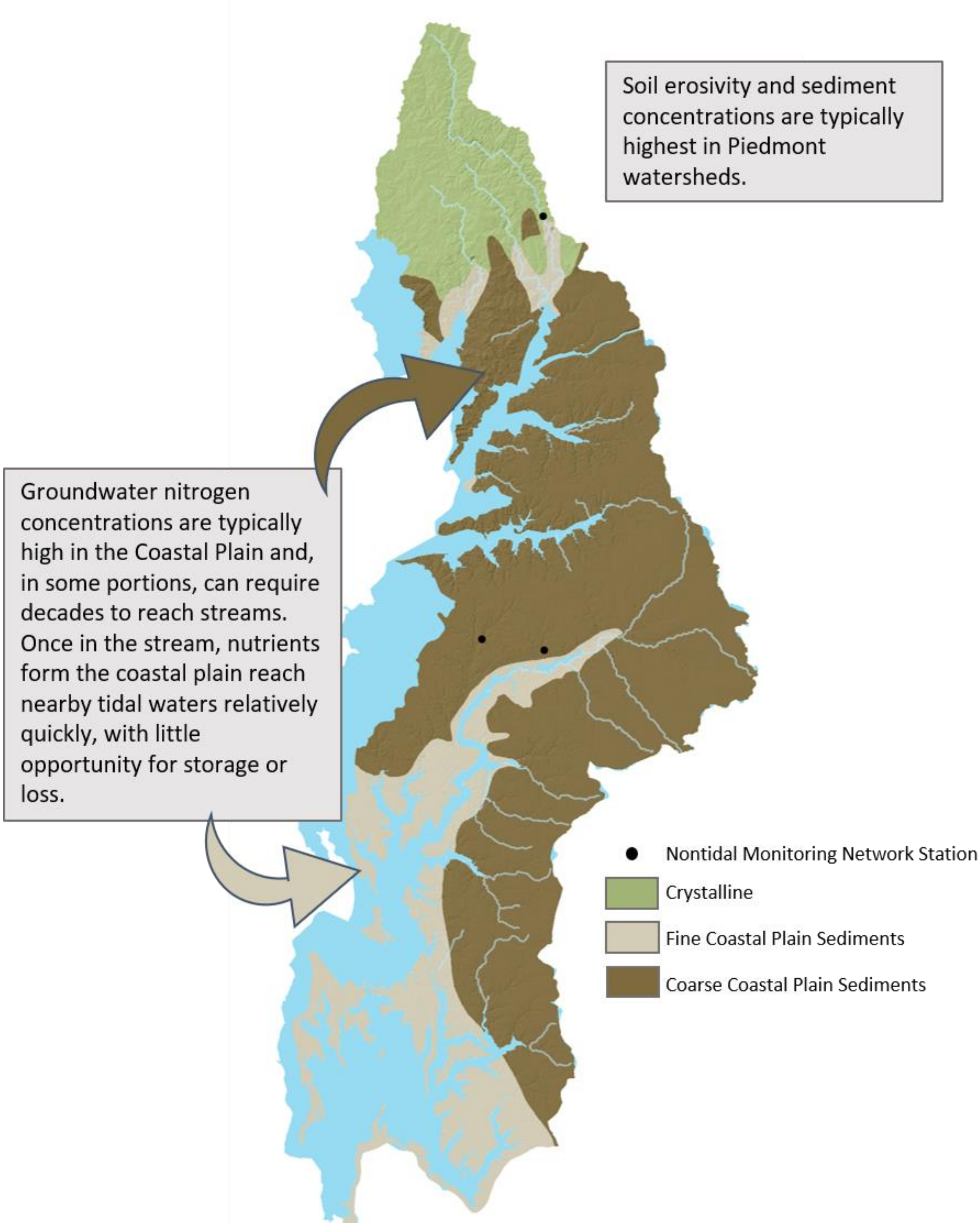


Figure 18. Effects of watershed hydrogeomorphology on nutrient transport to freshwater streams and tidal waters. Base map modified from King *et al.* (1974) and Ator *et al.* (2005), North American Datum 1983.

Nitrogen

Groundwater is an important delivery pathway of nitrogen, as nitrate, to most streams in the Chesapeake Bay watershed (Ator and Denver, 2012; Lizarraga, 1997) and contributes about 70% of the nitrogen to Eastern Shore streams (Ator and Denver, 2012; Domagalski and others, 2008). Some of the highest concentrations of groundwater nitrogen in the Bay watershed are present in portions of the Eastern Shore where oxygen-rich groundwater limits denitrification (Debrewer and others, 2008; Greene and others, 2005). Eastern Shore denitrification rates are low and nitrate concentrations are high in sandy soils and sediments (Böhlke and Denver, 1995; Denver and others 2004), in soils that have been drained to support agricultural activities (Staver and Brinsfield, 2001), and in areas underlain by a thick surficial aquifer that prevents contact with deeper, anoxic groundwater (Böhlke and Denver, 1995). These features vary substantially from place to place throughout the Delmarva Peninsula, but conditions limiting denitrification are common throughout the Upper Eastern Shore. Some of the highest Eastern Shore nitrogen concentrations are found in streams of this region (Ator and others, 2011) because of such conditions favorable to nitrogen transport (Ator and Denver, 2015). In general, the lowest Eastern Shore nitrate concentrations discharge to streams along the perimeter of the Delmarva Peninsula, where less permeable soils and a thinner surficial aquifer result in groundwater flowpaths that are more likely to encounter anoxic conditions (Ator and Denver, 2015). Most Eastern Shore streamflow is generated from groundwater that discharges from the uppermost few meters of a shallow, surficial unconfined aquifer (Cushing and others, 1973, Sanford and others, 2012). More than half of the groundwater discharging to streams is older than thirteen years (Sanford and Pope, 2013), so the high concentrations of nitrate that have increased in portions of the Eastern Shore aquifer (Debrewer and others, 2008), will likely contribute to streams for decades.

Phosphorus

Eastern Shore phosphorus concentrations are higher than most other regions of the Chesapeake Bay watershed (Ator *et al.*, 2011) because phosphorus concentrations are high in soils underlaying agricultural watersheds. Phosphorus applications have exceeded Eastern Shore cropping needs and have accumulated in such soils for decades (Staver and Brinsfield, 2001; Ator and Denver, 2015). Such conditions can increase the amount of sediment-bound and dissolved phosphorus carried in runoff (Heckrath *et al.*, 1995). Sandy soils common throughout the Eastern Shore can become fully phosphorus saturated relatively quickly because of their low phosphorus sorption capacity (Sharpley, 1980). As a result of such conditions, phosphorus can also be exported to streams from shallow soils and groundwater (Staver and Brinsfield, 2001). Reducing soil phosphorus concentrations can take a decade or more (Kleinman *et al.*, 2011) and, until this occurs, watershed phosphorus loads may be unresponsive to management practices (Jarvie *et al.*, 2013; Sharpley *et al.*, 2013).

Sediment

Despite increased sediment erosion associated with agricultural land uses, Eastern Shore sediment loads are typically as low as some undeveloped regions of the Bay watershed (Brakebill *et al.*, 2010) because of the relatively flat topography of the Atlantic Coastal Plain. The sediment load of a given stream reach is a balance of sediment eroded from uploads and streambanks and sediment stored in floodplains and stream channels. Eastern Shore streambank erosion rates are reduced in areas with low topographic gradient, but are also affected by watershed drainage area (Gellis and Noe, 2013; Gellis *et al.*, 2015; Gillespie *et al.*, 2018; Hopkins *et al.*, 2018), bank sediment density (Wynn and Mostaghimi, 2006), vegetation (Wynn and Mostaghimi, 2006), and other stream valley geomorphic properties (Hopkins *et al.*, 2018). The largest Eastern Shore sediment concentrations typically occur in streams draining portions of the Upper Delmarva Peninsula where land-surface slopes are the steepest (Ator and Denver, 2015).

Delivery to tidal waters

The delivery of nitrogen, phosphorus, and sediment in non-tidal Eastern Shore streams to tidal waters varies based on physical and chemical factors that affect in-stream retention, loss, or storage. In general, the proximity of much of the Eastern Shore to tidal waters limits opportunities for in-stream denitrification (Staver and Brinsfield, 2001). There are no natural chemical processes that remove phosphorus from streams, but sediment, and associated phosphorus, can be trapped in floodplains before reaching tidal waters. High rates of sediment trapping by Coastal Plain nontidal floodplains and head-of-tide tidal freshwater wetlands creates a sediment shadow in many tidal rivers and limits sediment delivery to the bay (Noe and Hupp, 2009; Ensign *et al.*, 2014). Shoreline erosion can be larger source of sediment delivered to Eastern Shore estuaries than upland runoff or streambank erosion because of such trapping and because of the low relief of the Atlantic Coastal Plain (Yarbro *et al.*, 1983).

5.1.2 Estimated Nutrient and Sediment Loads

Estimated loads to tidal portions of the Maryland Upper Eastern Shore Tributaries are a combination of simulated non-point source, atmospheric deposition, and reported point-source loads. These loads were obtained from the Chesapeake Bay Program Watershed Model's progress runs specific to each year from 1985 and 2018 (https://cast.chesapeakebay.net/). Nonpoint source loads were adjusted to reflect actual hydrology using the method of the Chesapeake Bay Program's Loads to the Bay indicator (see https://www.chesapeakeprogress.com/clean-water/water-quality). Over the period of 1985-2018, 0.14, 0.0096, and 12 million tons of nitrogen, phosphorus, and suspended sediment loads were exported from this watershed, respectively (Figure 19).

Mann-Kendall trends and Sen's slope estimates are summarized for each loading source in Table 4.

Nitrogen

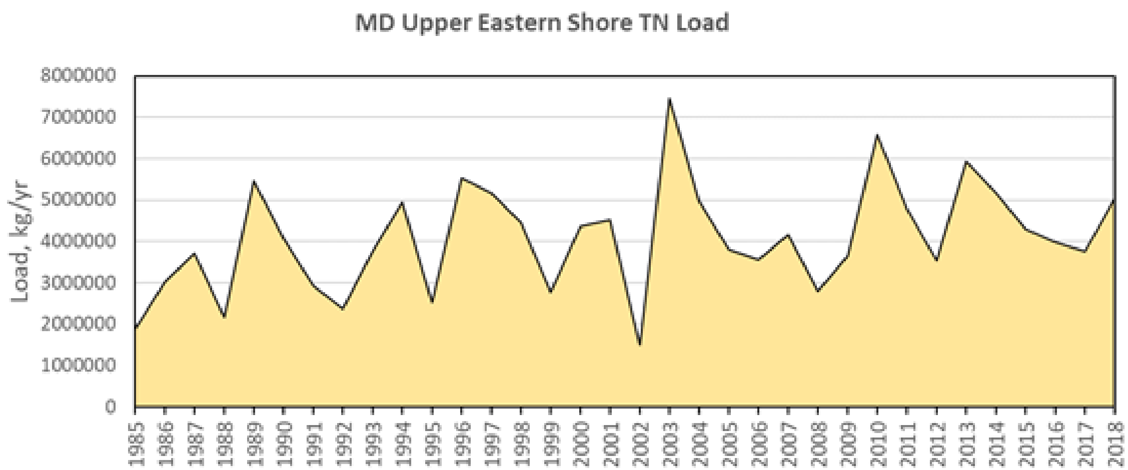
Estimated TN loads showed an overall increase of 48 ton/yr in the period between 1985 and 2018, although it is not statistically significant ($p = 0.08$). Long-term declines were observed with both point sources (-1.1 ton/yr, $p = 0.07$) and atmospheric deposition to the tidal waters (-3.0 ton/yr, $p < 0.01$). By contrast, the nonpoint sources showed a long-term increase in this period (50 ton/yr), although it is not statistically significant ($p = 0.05$). The significant point source reductions in TN are a result of substantial efforts to reduce nitrogen loads from major wastewater treatment facilities by implementing biological nutrient removal (Lyerly *et al.*, 2014; Fisher *et al.*, 2021). The significant decline in atmospheric deposition of TN to the tidal waters is consistent with findings that atmospheric deposition of nitrogen has decreased due to benefits from the Clean Air Act implementation (Eshleman *et al.*, 2013; Lyerly *et al.*, 2014).

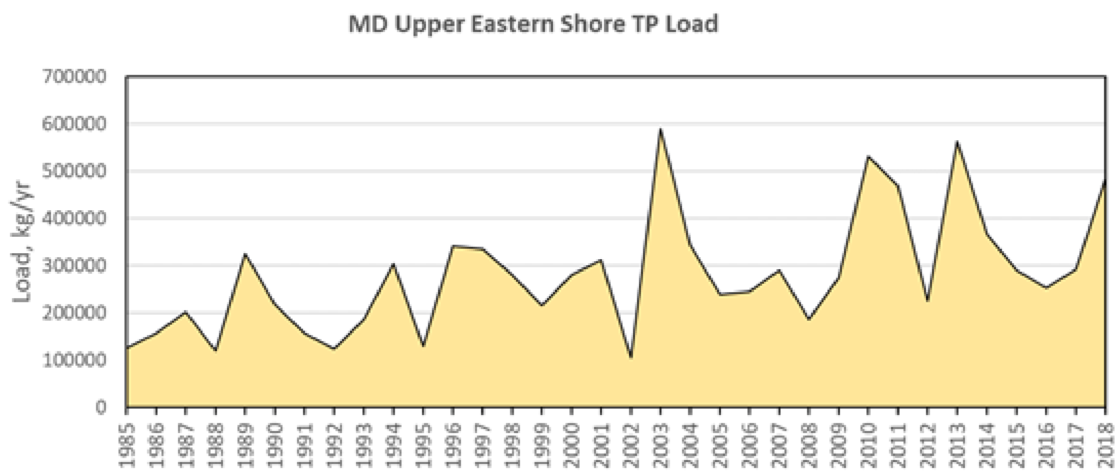
Phosphorus

Estimated TP loads showed an overall increase of 5.9 ton/yr in the period between 1985 and 2018, which is statistically significant ($p < 0.01$). This increase is entirely driven by nonpoint sources (6.2 ton/yr, $p < 0.01$). By contrast, point sources showed a statistically significant decline (-0.26 ton/yr, $p < 0.01$). This TP point source load reduction has also been attributed to significant efforts to reduce phosphorus in wastewater discharge through the phosphorus detergent ban in the early part of this record, as well as technology upgrades at wastewater treatment facilities (Lyerly *et al.*, 2014; Fisher *et al.*, 2021).

Sediment

Estimated suspended sediment (SS) loads showed an overall increase of 2,247 ton/yr in the period between 1985 and 2018, which is statistically significant ($p < 0.01$). This increase is entirely driven by nonpoint sources (2,252 ton/yr, $p < 0.01$). Like TP and TN, point source load of SS showed a statistically significant decline in this period (-1.5 ton/yr; $p < 0.01$).





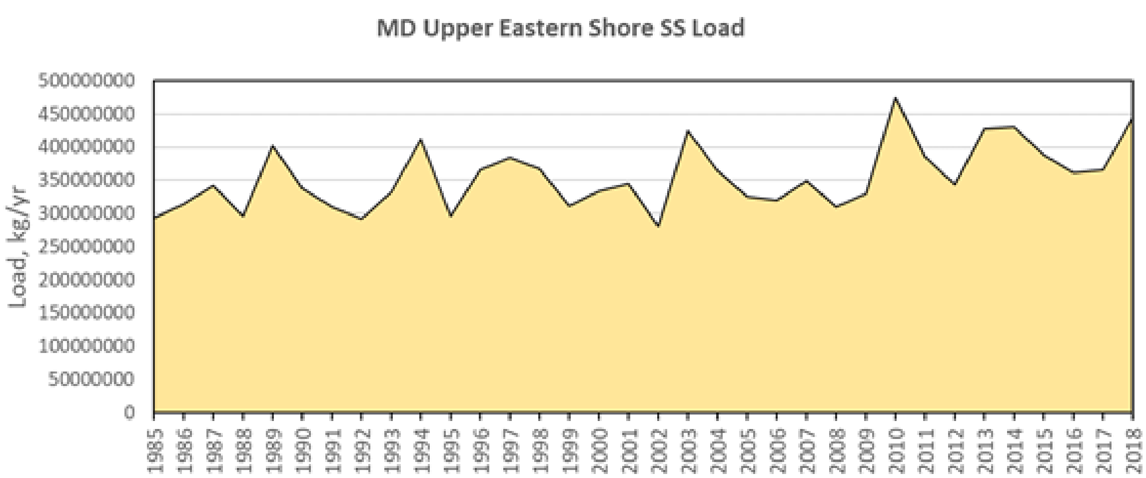


Figure 19. Estimated total loads of nitrogen (TN), phosphorus (TP), and suspended sediment (SS) to the Maryland Upper Eastern Shore Tributaries.

Table 4. Summary of Mann-Kendall trends for the period of 1985-2018 for total nitrogen (TN), total phosphorus (TP), and suspended sediment (SS) loads from the Maryland Upper Eastern Shore watershed.

Variable	Trend, metric ton/yr	Trend p-value
TN		
Total watershed[1]	48	0.08
Point source	-1.1	0.07
Nonpoint source[2]	50	0.05
Tidal deposition	-3.0	< 0.01
TP		
Total watershed	5.9	< 0.01
Point source	-0.26	< 0.01
Nonpoint source	6.2	< 0.01
SS		
Total watershed	2,247	< 0.01
Point source	-1.5	< 0.01
Nonpoint source	2,252	< 0.01

[1] Loads from the different sources were obtained from the Chesapeake Bay Program Watershed Model progress runs specific to each year from 1985 and 2018, (https://cast.chesapeakebay.net/).
[2] Nonpoint source loads were adjusted to reflect actual hydrology using the method of the Chesapeake Bay Program's Loads to the Bay indicator (see https://www.chesapeakeprogress.com/clean-water/water-quality). The adjustment factor for each year is defined as the ratio between monitored load and watershed model simulated load for an applicable USGS River Input Monitoring (RIM) station. Because the Maryland Upper Eastern Shore Tributaries do not have RIM stations, adjustment factors need to be transferred from a different tributary that has a RIM station. In this regard, the Choptank River was selected for two reasons: (1) it is geographically proximate to the Maryland Upper Eastern Shore Tributaries, and (2) it is hydrologically similar to the Maryland Upper Eastern Shore Tributaries based on an analysis of annual riverflow anomalies.

5.1.3 Expected Effects of Changing Watershed Conditions

According to the Chesapeake Bay Program's Watershed Model known as the Chesapeake Assessment Scenario Tool (CAST; https://cast.chesapeakebay.net, version CAST-2019), changes in population size, land use, and pollution management controls between 1985 and 2019 would be expected to change long-term average nitrogen, phosphorus, and sediment loads to the tidal Maryland Upper Eastern Shore River by -18%, -44%, and -9%, respectively (Figure 20). In contrast to the annual loads analysis above, CAST loads are based on changes in management only and do not include annual fluctuations in weather. CAST loads are calculated without lag times for delivery of pollutants or lags related to BMPs becoming fully effective after installation. In 1985, agriculture and developed were the two largest sources of nitrogen loads. By 2019, agriculture and developed remained the two largest sources of nitrogen loads. Overall, decreasing nitrogen loads from agriculture (-27%), natural (-7%), stream bed and bank (-18%), and wastewater (-48%) sources were partially counteracted by increases from developed (70%) and septic (50%) sources.

The two largest sources of phosphorus loads as of 2019 were the agriculture and shoreline sectors. Overall, expected declines from agriculture (-66%), natural (-11%), septic (-75%), stream bed and bank (-51%), and wastewater (-80%) sources were partially counteracted by increases from developed (35%) sources.

For sediment, the largest sources are shoreline and stream bed and bank areas: these two sources changed by 0% and -31%, respectively between 1985 and 2019. Sediment loads from the agriculture sector changed by -52%, whereas sediment load from developed areas changed by 57%.

Overall, changing watershed conditions are expected to result in the agriculture, natural, stream bed and bank, and wastewater sectors achieving reductions in nitrogen, phosphorus, and sediment loads between 1985 and 2019, whereas the developed sectors are expected to increase in nitrogen, phosphorus, and sediment loads.

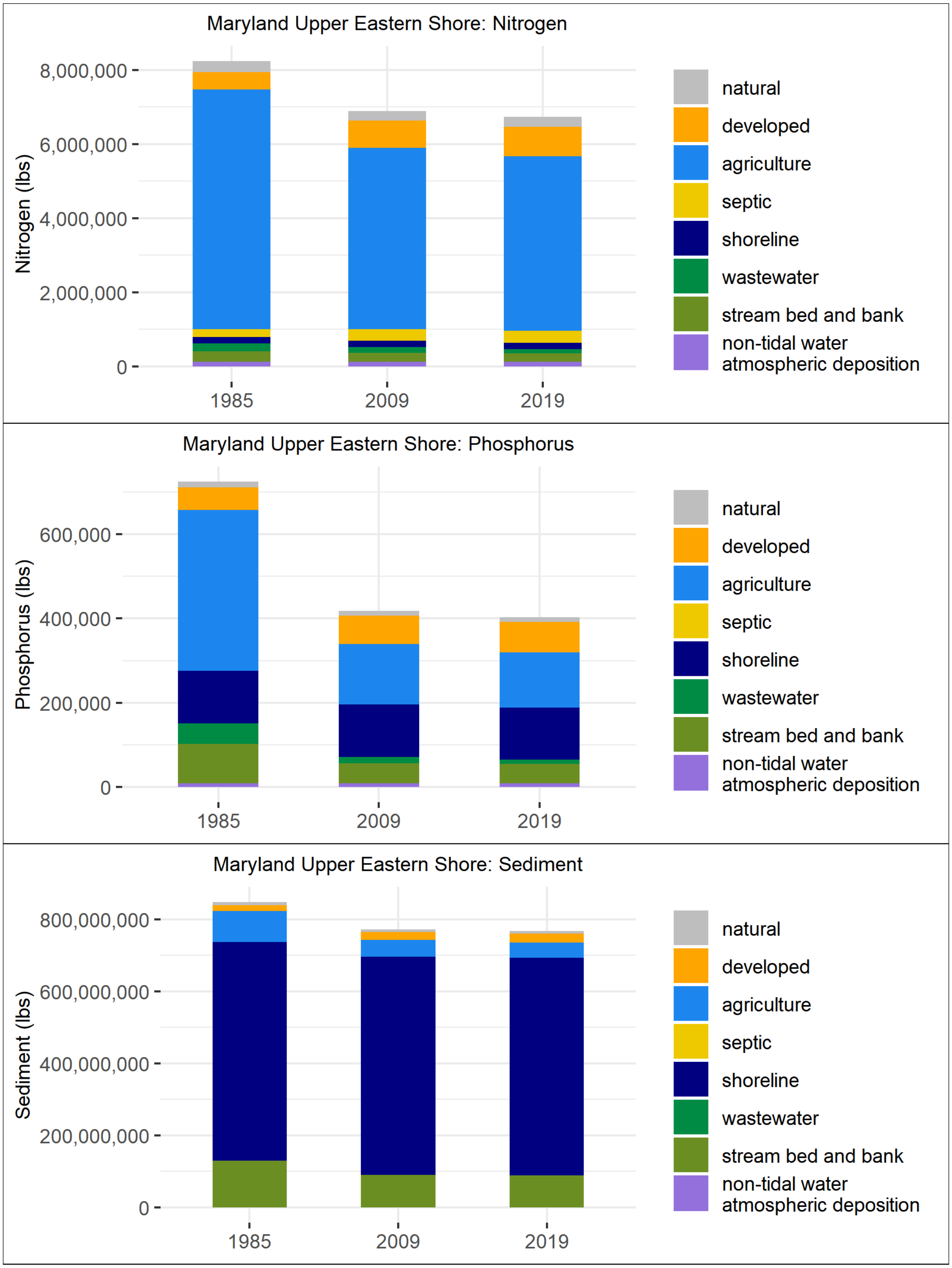


Figure 20. Expected long-term average loads of nitrogen, phosphorus, and sediment from different sources to the tidal Chester, as obtained from the Chesapeake Assessment Scenario Tool (CAST-19). Data shown are time-average delivered loads over the average hydrology of 1991-2000, once the steady

state is reached for the conditions on the ground, as obtained from the 1985, 2009, and 2019 progress (management) scenarios.

5.1.4 Best Management Practices (BMPs) Implementation

Data on reported BMP implementation are available for download from CAST (https://cast.chesapeakebay.net, version CAST-2019). Reported BMP implementations on the ground as of 1985, 2009, and 2019 are compared to planned 2025 implementation levels in Figure 21 for a subset of major BMP groups measured in acres. As of 2019, tillage, cover crops, pasture management, forest buffer and tree planting, stormwater management, agricultural nutrient management, and urban nutrient management were credited for 188, 129, 2.5, 0.2, 5, 506, and 45 thousand acres, respectively. Implementation levels for some practices are already close to achieving their planned 2025 levels: for example, 64% of planned acres for pasture management had been achieved as of 2019. In contrast, about 91% of planned commodity & cover crops implementation had been achieved as of 2019.

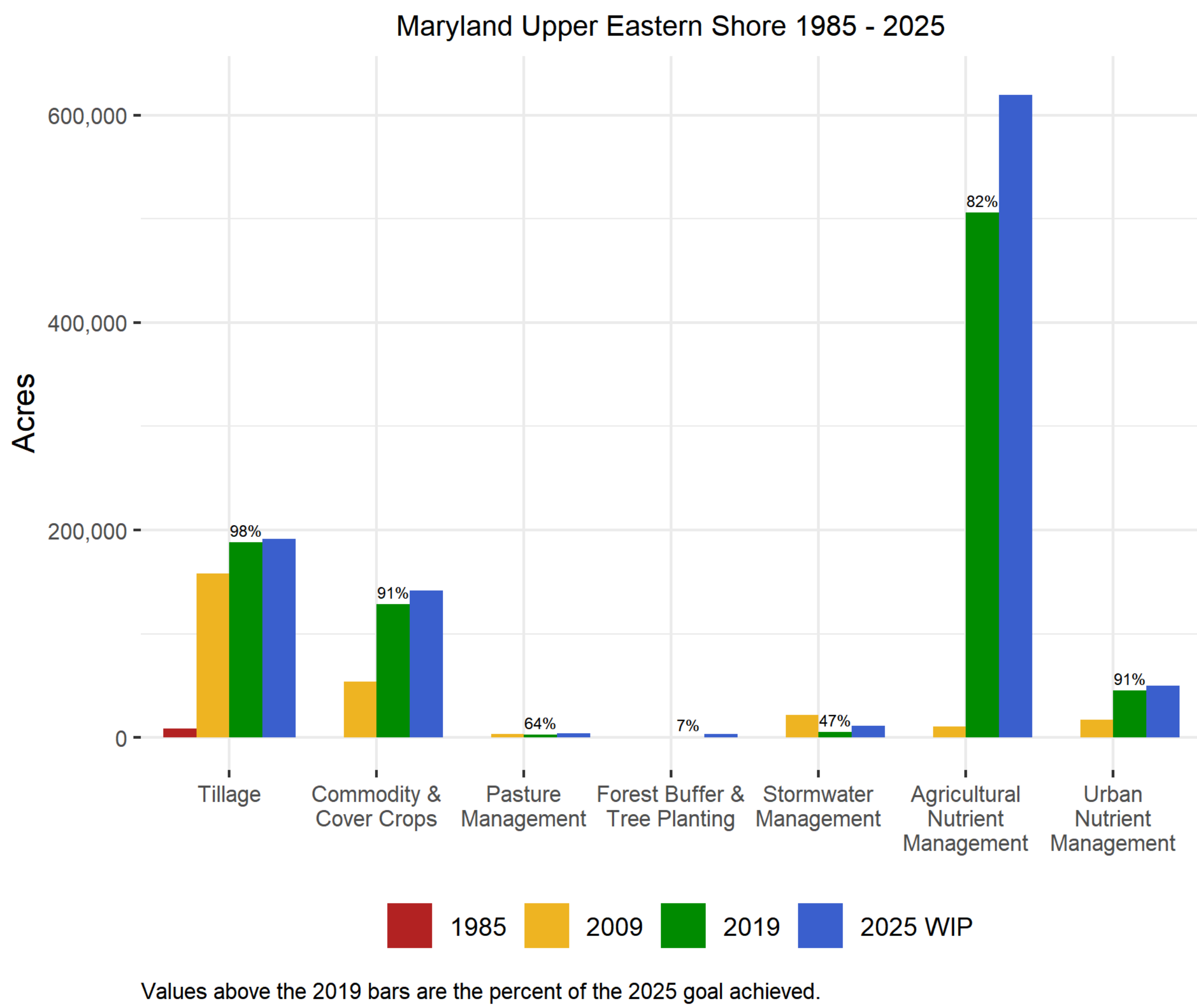


Figure 21. BMP implementation in the Maryland Upper Eastern Shore watershed

Stream restoration and animal waste management system systems are two important BMPs that cannot be compared directly with those above because they are measured in different units. However, progress towards implementation goals can still be documented. Stream restoration (agricultural and urban) had increased from 0 feet in 1985 to 3,222 feet in 2019. Over the same period, animal waste management systems treated 399 animal units in 1985 and 179,839 animal units in 2019 (one animal unit represents 1,000 pounds of live animal). These implementation levels represent 7% and 89% of their planned 2025 implementation levels, respectively.

5.1.5 Flow-Normalized Watershed Nutrient and Sediment Loads

Flow normalization can better reveal temporal trends in river water quality by removing the effect of inter-annual variability in streamflow. Flow-normalized trends help scientists evaluate changes in load resulting from changing sources, delays associated with storage or transport of historical inputs, and/or implemented management actions. Flow-normalized nitrogen, phosphorus, and sediment trends have been reported for the short term (2009-2018) at nontidal network stations throughout the watershed (Moyer and Langland, 2020) (Table 5). These trends result from variability in nutrient applications, the delivery of nutrients and sediment from the landscape to streams, and from processes that affect in-stream loss or retention of nutrients and sediment.

Table 5. Short-term trends (2009 - 2018) of flow-normalized total nitrogen (TN), total phosphorus (TP), and suspended sediment (SS) loads for the nontidal network monitoring location in the Maryland Upper Eastern Shore watershed. A more detailed summary of flow-normalized loads and trends measured at all USGS Chesapeake Bay Nontidal Network stations can be found at https://cbrim.er.usgs.gov/summary.html.

USGS Station ID	USGS Station Name	Trend start water year	Percent change in FN load, through water year 2018		
			TN	TP	SS
01495000	BIG ELK CREEK AT ELK MILLS, MD	2009	-4.4	-7.5	-4.9

Decreasing trends listed in green, increasing trends listed in orange, results reported as "no trend" listed in black. TN = total nitrogen, TP = total phosphorus, SS = suspended sediment

5.2 Tidal Factors

Once pollutants reach tidal waters, a complex set of environmental factors interact with them to affect key habitat indicators like algal biomass, DO concentrations, water clarity, submerged aquatic vegetation (SAV) abundance, and fish populations (Kemp *et al.*, 2005; Testa *et al.*, 2017) (Figure 22). For example, phytoplankton growth depends not just on nitrogen and phosphorus (Fisher *et al.*, 1992; Kemp *et al.*, 2005; Zhang *et al.*, 2021), but also on light and water temperature (Buchanan *et al.*, 2005; Buchanan, 2020). In general, the saline waters of the lower Bay tend to be more transparent than tidal-fresh regions, and waters adjacent to nutrient input points are more affected by these inputs than more distant regions (Keisman *et al.*, 2019; Testa *et al.*, 2019). Dissolved oxygen concentrations are affected by salinity- and temperature-driven stratification of the water column, and conversely by wind-driven mixing, in addition to phytoplankton respiration and decomposition (Scully, 2010; Murphy *et al.*, 2011). When anoxia occurs at the water-sediment interface, nitrogen and phosphorus stored in the sediments can be released through anaerobic chemical reactions (Testa and Kemp, 2012). When low-oxygen water and sediment burial suffocate benthic plant and animal communities, their nutrient consumption and water filtration services are lost. Conversely, when conditions improve enough to support abundant SAV and benthic communities, their functions can sustain and even advance progress towards a healthier ecosystem (Cloern, 1982; Phelps, 1994; Ruhl and Rybicki, 2010; Gurbisz and Kemp, 2014).

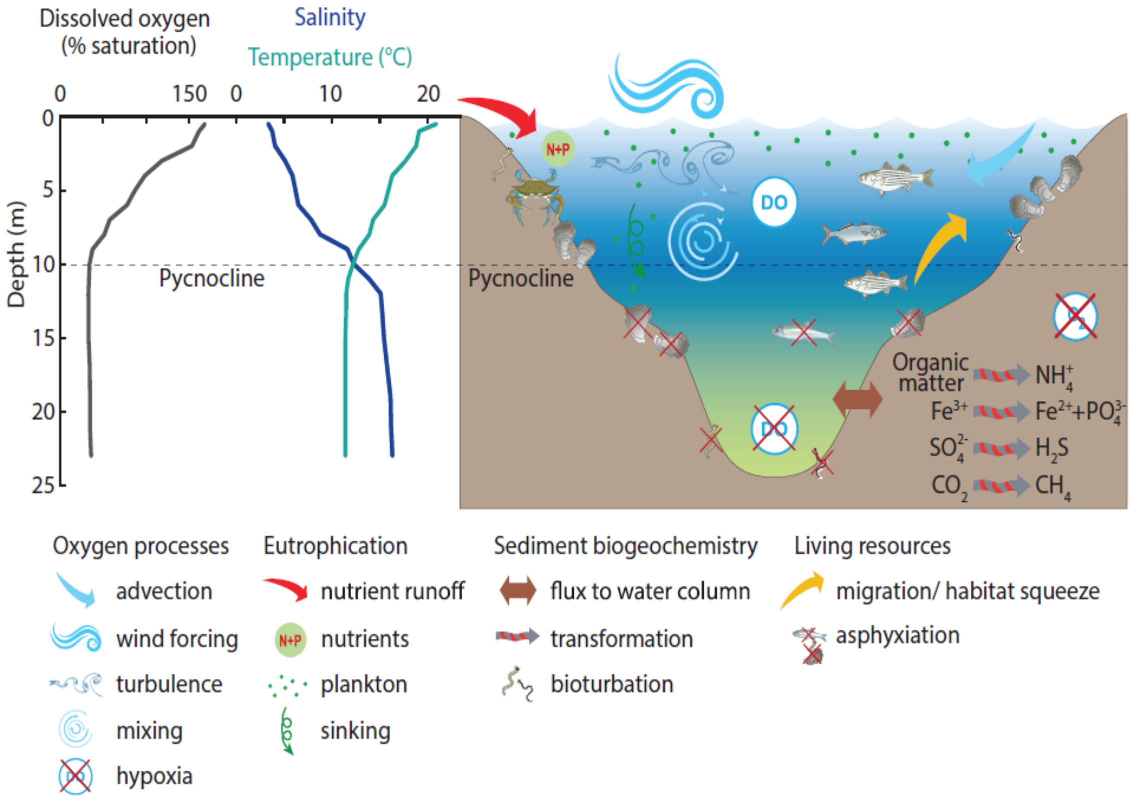


Figure 22. Conceptual diagram illustrating how hypoxia is driven by eutrophication and physical forcing, while affecting sediment biogeochemistry and living resources. From Testa *et al.* (2017).

High nutrient loads relative to tidal river size are indicative of areas that are more susceptible to eutrophication (Bricker *et al.*, 2003; Ferreira *et al.*, 2007). The relationship between watershed area and tidal river size may also be an important indicator of eutrophication potential, however there are competing effects. A large watershed relative to the volume of receiving water would likely correlate with higher nutrient loads, however it would also correlate with a higher flow rate and decreased flushing time (Bricker *et al.*, 2008). Figure 23 is a comparison of watershed area versus estuarine volume for all estuaries and sub-estuaries identified in the CBP monitoring segment scheme. Larger estuaries will contain multiple monitoring segments and, in many cases, sub-estuaries. For example, the Potomac River contains monitoring segments in the tidal fresh, oligohaline, and mesohaline sections of the river as well as the entire Anacostia River and other sub-estuaries. Figures 24 and 25 are comparisons of estimated annual average nitrogen and phosphorus loads, respectively, for the 2018 progress scenario in CAST versus the estuarine volume for the same set of estuaries and sub-estuaries.

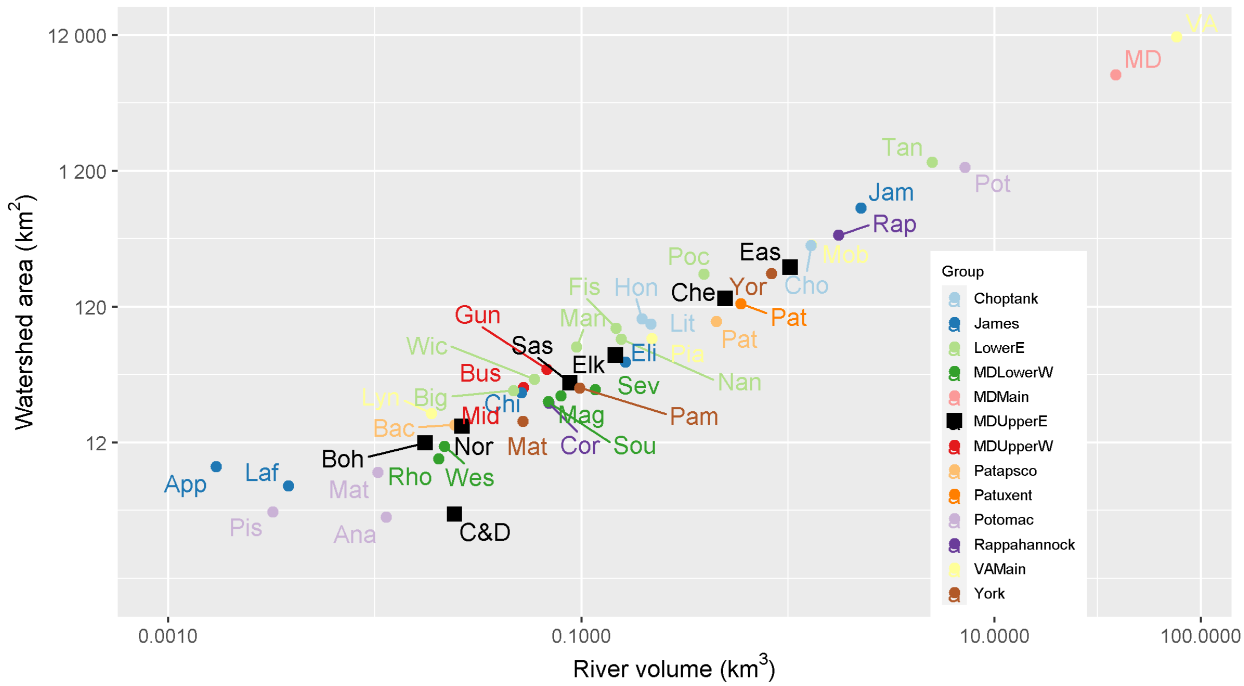


Figure 23. Watershed area vs estuarine volume.

Abbreviated tributary name	Full tributary name	Abbreviated tributary name	Full tributary name
Ana	Anacostia River	Mat	Mattaponi River
App	Appomattox River	MD	MD MAINSTEM
Bac	Back River	Mid	Middle River
Big	Big Annemessex River	Mob	Mobjack Bay
Boh	Bohemia River	Nan	Nanticoke River
Bus	Bush River	Nor	Northeast River
C&D	C&D Canal	Pam	Pamunkey River
Che	Chester River	Pat	Patapsco River
Chi	Chickahominy River	Pat	Patuxent River
Cho	Choptank River	Pia	Piankatank River
Cor	Corrotoman River	Pis	Piscataway Creek
Eas	Eastern Bay	Poc	Pocomoke River
Eli	Elizabeth River	Pot	Potomac River
Elk	Elk River	Rap	Rappahannock River
Fis	Fishing Bay	Rho	Rhode River
Gun	Gunpowder River	Sas	Sassafras River
Hon	Honga River	Sev	Severn River
Jam	James River	Sou	South River
Laf	Lafayette River	Tan	Tangier Sound
Lit	Little Choptank River	VA	VA MAINSTEM
Lyn	Lynnhaven River	Wes	West River
Mag	Magothy River	Wes	Western Branch (Patuxent River)
Man	Manokin River	Wic	Wicomico River
Mat	Mattawoman Creek	Yor	York River

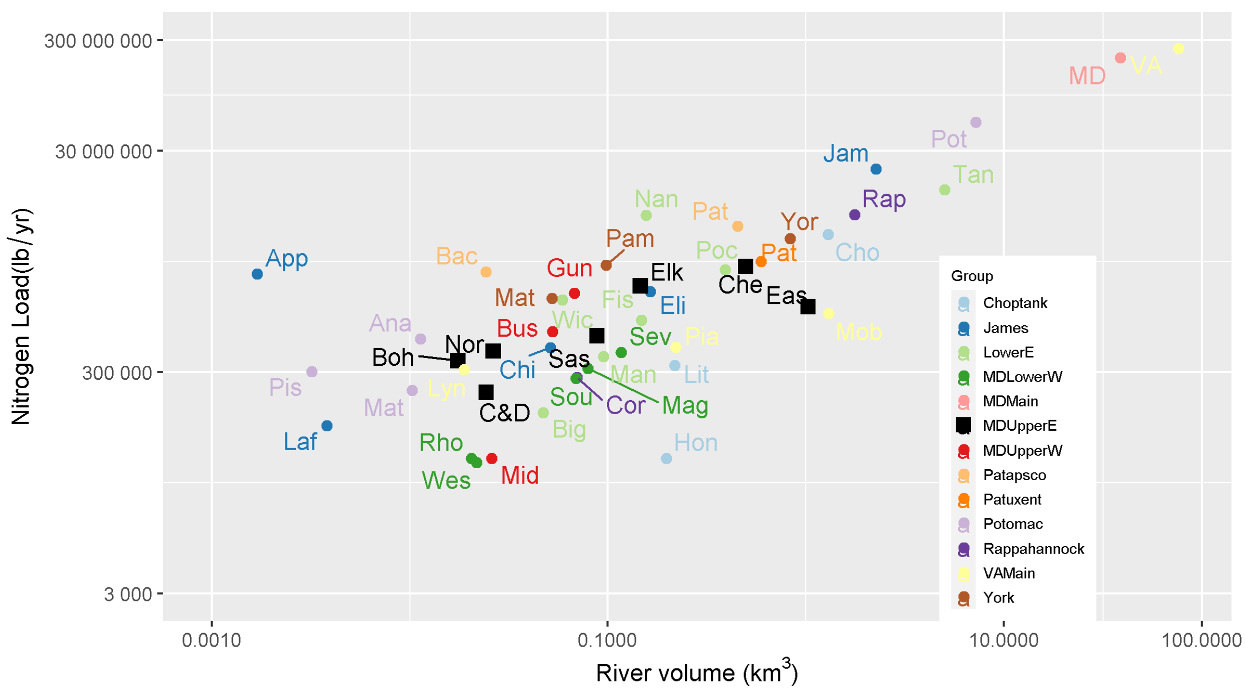


Figure 24. Annual average expected nitrogen loads versus estuarine volume. Nitrogen loads are from the 2018 progress scenarios in CAST (Chesapeake Bay Program, 2020), which is an estimate of nitrogen loads under long-term average hydrology given land use and reported management as of 2018.

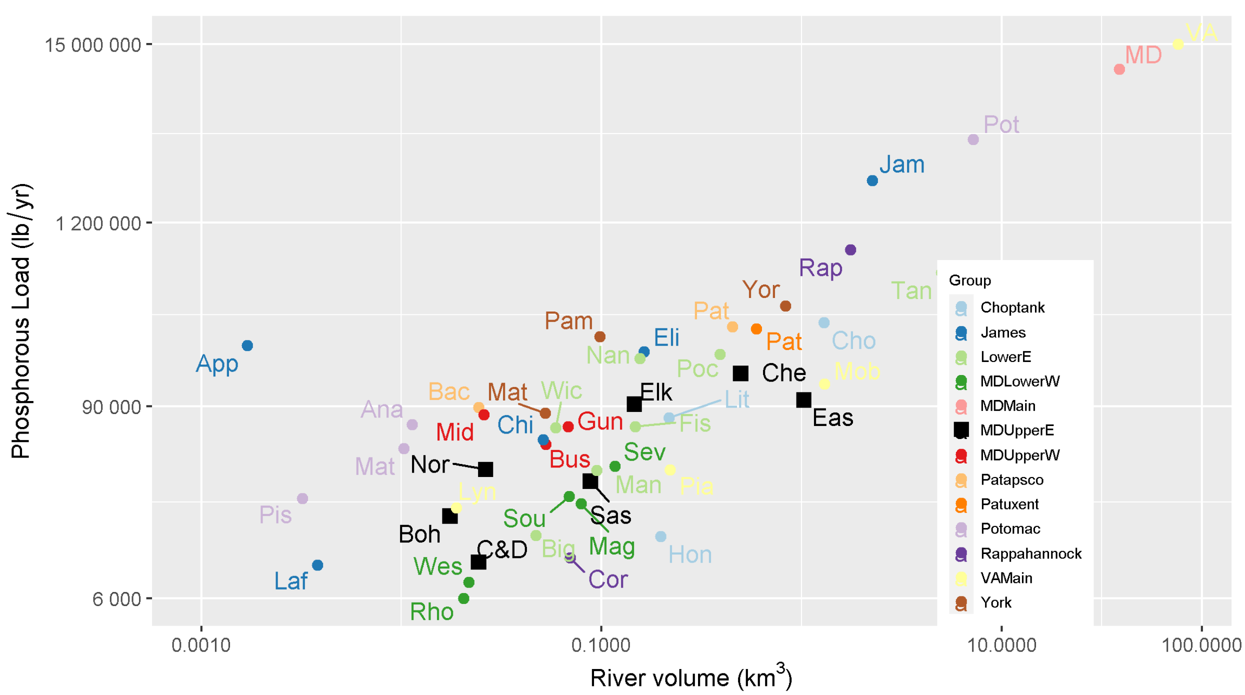


Figure 25. Annual average expected phosphorus loads versus estuarine volume. Phosphorus loads are from the 2018 progress scenarios in CAST (Chesapeake Bay Program, 2020), which is an estimate of

phosphorus loads under long-term average hydrology given land use and reported management as of 2018.

The Maryland Upper Eastern Shore estuary volume and watershed contain approximately 2% of the total volume and watershed of the Chesapeake Bay. This ranks the Maryland Upper Eastern Shore as the 6th largest volume and 7th largest watershed area aggregated tributary in this summary (Figures 23, 24, and 25). The ratios of watershed area, nitrogen loading, and phosphorus loading to estuarine volume are generally consistent with other estuaries in the Chesapeake system, indicating a moderate level of susceptibility to eutrophication. The smaller tributaries within the Maryland upper Eastern Shore system, including the Chester river, Eastern Bay, Elk river, Bohemia river, Northeast river, and Sassafras river all drive this consistent relationship. The C&D Canal however has a lower watershed area relative to its estuarine volume indicating lower susceptibility to eutrophication. Eastern Bay has lower phosphorous and nitrogen load relative to river volume while the remaining tributaries are all consistent with other tributaries.

5.3 Insights on Changes in the Upper Eastern Shore

Completion of Section 5.3 is contingent upon stakeholder interest and availability of resources. It requires:

- *Synthesis of the information provided in previous sections and of the recent literature on explaining trends in general and any work conducted on this tributary in particular;*
- *Discussion with local technical experts to clarify insights and vet hypotheses and preliminary findings.*

6. Summary

Completion of Section 6 is contingent upon completion of Section 5.3.

References

Ator, S. W., J. D. Blomquist, J. S. Webber and J. G. Chanat, 2020. Factors driving nutrient trends in streams of the Chesapeake Bay watershed. *J. Environ. Qual.* 49:812-834, DOI: 10.1002/jeq2.20101.

Ator, S. W., J. W. Brakebill and J. D. Blomquist, 2011. Sources, fate, and transport of nitrogen and phosphorus in the Chesapeake Bay watershed: An empirical model. U.S. Geological Survey Scientific Investigations Report 2011-5167, Reston, VA, p. 27. http://pubs.usgs.gov/sir/2011/5167/.

Ator, S. W. and J. M. Denver, 2015. Understanding nutrients in the Chesapeake Bay watershed and implications for management and restoration: the Eastern Shore. U.S. Geological Survey U.S. Geological Survey Circular 1406, p. 72. http://dx.doi.org/10.3133/cir1406.

Ator, S. W., J. M. Denver, D. E. Krantz, W. L. Newell and S. K. Martucci, 2005. A Surficial Hydrogeologic Framework for the Mid-Atlantic Coastal Plain. U.S. Geological Survey U.S. Geological Survey Professional Paper 1680. https://pubs.usgs.gov/pp/2005/pp1680/.

Ator, S. W., A. M. García, G. E. Schwarz, J. D. Blomquist and A. J. Sekellick, 2019. Toward explaining nitrogen and phosphorus trends in Chesapeake Bay tributaries, 1992–2012. *J. Am. Water Resour. Assoc.* 55:1149-1168, DOI: 10.1111/1752-1688.12756.

Bachman, L. J., B. Lindsey, J. Brakebill and D. S. Powars, 1998. Ground-water discharge and base-flow nitrate loads of nontidal streams, and their relation to a hydrogeomorphic classification of the Chesapeake Bay Watershed, middle Atlantic coast. US Geological Survey Water-Resources Investigations Report 98-4059, Baltimore, MD, p. 71. http://pubs.usgs.gov/wri/wri98-4059/.

Brakebill, J. W., S. W. Ator and G. E. Schwarz, 2010. Sources of suspended-sediment flux in streams of the Chesapeake Bay watershed: A regional application of the SPARROW Model. *J. Am. Water Resour. Assoc.* 46:757-776, DOI: 10.1111/j.1752-1688.2010.00450.x.

Bricker, S. B., J. G. Ferreira and T. Simas, 2003. An integrated methodology for assessment of estuarine trophic status. *Ecol. Model.* 169:39-60, DOI: 10.1016/s0304-3800(03)00199-6.

Bricker, S. B., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks and J. Woerner, 2008. Effects of nutrient enrichment in the nation's estuaries: A decade of change. *Harmful Algae* 8:21-32, DOI: 10.1016/j.hal.2008.08.028.

Buchanan, C., 2020. A water quality binning method to infer phytoplankton community structure and function. *Estuaries Coasts* 43:661-679, DOI: 10.1007/s12237-020-00714-3.

Buchanan, C., R. V. Lacouture, H. G. Marshall, M. Olson and J. M. Johnson, 2005. Phytoplankton reference communities for Chesapeake Bay and its tidal tributaries. *Estuaries* 28:138-159, DOI: 10.1007/bf02732760.

Chesapeake Bay Program, 2018. Data Hub.

Chesapeake Bay Program, 2020. Chesapeake Assessment and Scenario Tool (CAST) Version 2019.

Cloern, J. E., 1982. Does the benthos control phytoplankton biomass in South San Francisco Bay? *Mar. Ecol. Prog. Ser.* 9:191-202, DOI: 10.3354/meps009191.

Ensign, S. H., C. R. Hupp, G. B. Noe, K. W. Krauss and C. L. Stagg, 2014. Sediment accretion in tidal freshwater forests and oligohaline marshes of the Waccamaw and Savannah Rivers, USA. *Estuaries Coasts* 37:1107-1119, DOI: 10.1007/s12237-013-9744-7.

Eshleman, K. N., R. D. Sabo and K. M. Kline, 2013. Surface water quality is improving due to declining atmospheric N deposition. *Environ. Sci. Technol.* 47:12193-12200, DOI: 10.1021/es4028748.

Falcone, J. A., 2015. U.S. conterminous wall-to-wall anthropogenic land use trends (NWALT), 1974–2012. U.S. Geological Survey Data Series 948, Reston, VA. https://doi.org/10.3133/ds948.

Ferreira, J. G., S. B. Bricker and T. C. Simas, 2007. Application and sensitivity testing of a eutrophication assessment method on coastal systems in the United States and European Union. *J. Environ. Manage.* 82:433-445, DOI: 10.1016/j.jenvman.2006.01.003.

Fisher, T. R., R. J. Fox, A. B. Gustafson, E. Koontz, M. Lepori-Bui and J. Lewis, 2021. Localized water quality improvement in the choptank estuary, a tributary of Chesapeake Bay. *Estuaries Coasts* 44:1274-1293, DOI: 10.1007/s12237-020-00872-4.

Fisher, T. R., E. R. Peele, J. W. Ammerman and L. W. Harding, 1992. Nutrient limitation of phytoplankton in Chesapeake Bay. *Mar. Ecol. Prog. Ser.* 82:51-63, DOI: 10.3354/meps082051.

Gellis, A. C. and G. B. Noe, 2013. Sediment source analysis in the Linganore Creek watershed, Maryland, USA, using the sediment fingerprinting approach: 2008 to 2010. *J. Soils Sed.* 13:1735-1753, DOI: 10.1007/s11368-013-0771-6.

Gellis, A. C., G. B. Noe, J. W. Clune, M. K. Myers, C. R. Hupp, E. R. Schenk and G. E. Schwarz, 2015. Sources of fine-grained sediment in the Linganore Creek watershed, Frederick and Carroll

Counties, Maryland, 2008–10. U.S. Geological Survey Scientific Investigations Report 2014–5147, Reston, VA, p. 56. http://dx.doi.org/10.3133/sir20145147.
Gillespie, J. L., G. B. Noe, C. R. Hupp, A. C. Gellis and E. R. Schenk, 2018. Floodplain trapping and cycling compared to streambank erosion of sediment and nutrients in an agricultural watershed. *J. Am. Water Resour. Assoc.* 54:565-582, DOI: 10.1111/1752-1688.12624.
Gurbisz, C. and W. M. Kemp, 2014. Unexpected resurgence of a large submersed plant bed in Chesapeake Bay: Analysis of time series data. *Limnol. Oceanogr.* 59:482-494, DOI: 10.4319/lo.2014.59.2.0482.
Harding, J. L. W. and E. S. Perry, 1997. Long-term increase of phytoplankton biomass in Chesapeake Bay, 1950-1994. *Mar. Ecol. Prog. Ser.* 157:39-52, DOI: 10.3354/meps157039.
Heckrath, G., P. C. Brookes, P. R. Poulton and K. W. T. Goulding, 1995. Phosphorus leaching from soils containing different phosphorus concentrations in the broadbalk experiment. *J. Environ. Qual.* 24:904-910, DOI: 10.2134/jeq1995.00472425002400050018x.
Hernandez Cordero, A. L., P. J. Tango and R. A. Batiuk, 2020. Development of a multimetric water quality indicator for tracking progress towards the achievement of Chesapeake Bay water quality standards. *Environ. Monit. Assess.* 192:94, DOI: 10.1007/s10661-019-7969-z.
Hopkins, K. G., G. B. Noe, F. Franco, E. J. Pindilli, S. Gordon, M. J. Metes, P. R. Claggett, A. C. Gellis, C. R. Hupp and D. M. Hogan, 2018. A method to quantify and value floodplain sediment and nutrient retention ecosystem services. *J. Environ. Manage.* 220:65-76, DOI: 10.1016/j.jenvman.2018.05.013.
Jarvie, H. P., A. N. Sharpley, B. Spears, A. R. Buda, L. May and P. J. Kleinman, 2013. Water quality remediation faces unprecedented challenges from "legacy phosphorus". *Environ. Sci. Technol.* 47:8997-8998, DOI: 10.1021/es403160a.
Keisman, J., C. Friedrichs, R. Batiuk, J. Blomquist, J. Cornwell, C. Gallegos, S. Lyubchich, K. Moore, R. Murphy, R. Orth, L. Sanford, P. Tango, J. Testa, M. Trice and Q. Zhang, 2019. Understanding and explaining 30 years of water clarity trends in the Chesapeake Bay’s tidal waters. Chesapeake Bay Program Scientific and Technical Advisory Committee STAC Publication Number 19-004, Edgewater, MD, p. 25. http://www.chesapeake.org/pubs/411_Keisman2019.pdf.
Kemp, W. M., W. R. Boynton, J. E. Adolf, D. F. Boesch, W. C. Boicourt, G. Brush, J. C. Cornwell, T. R. Fisher, P. M. Glibert, J. D. Hagy, L. W. Harding, E. D. Houde, D. G. Kimmel, W. D. Miller, R. I. E. Newell, M. R. Roman, E. M. Smith and J. C. Stevenson, 2005. Eutrophication of Chesapeake Bay: Historical trends and ecological interactions. *Mar. Ecol. Prog. Ser.* 303:1-29, DOI: 10.3354/meps303001.
King, P. B., H. M. Beikman and G. J. Edmonston, 1974. Geologic map of the United States (exclusive of Alaska and Hawaii). U.S. Geological Survey. https://doi.org/10.3133/70136641.
Kleinman, P., A. Sharpley, A. Buda, R. McDowell and A. Allen, 2011. Soil controls of phosphorus in runoff: Management barriers and opportunities. *Can. J. Soil Sci.* 91:329-338, DOI: 10.4141/cjss09106.
Lyerly, C. M., A. L. H. Cordero, K. L. Foreman, S. W. Phillips and W. C. Dennison, 2014. New insights: Science-based evidence of water quality improvements, challenges, and opportunities in the Chesapeake. Annapolis, MD, p. 47. http://ian.umces.edu/pdfs/ian_report_438.pdf.
Moyer, D. L. and M. J. Langland, 2020. Nitrogen, phosphorus, and suspended-sediment loads and trends measured at the Chesapeake Bay Nontidal Network stations: Water years 1985-2018. Accessed https://doi.org/10.5066/P931M7FT.
Murphy, R. R., W. M. Kemp and W. P. Ball, 2011. Long-term trends in Chesapeake Bay seasonal hypoxia, stratification, and nutrient loading. *Estuaries Coasts* 34:1293-1309, DOI: 10.1007/s12237-011-9413-7.

Murphy, R. R., E. Perry, J. Harcum and J. Keisman, 2019. A generalized additive model approach to evaluating water quality: Chesapeake Bay case study. *Environ. Model. Software* 118:1-13, DOI: 10.1016/j.envsoft.2019.03.027.

Noe, G. B., M. J. Cashman, K. Skalak, A. Gellis, K. G. Hopkins, D. Moyer, J. Webber, A. Benthem, K. Maloney, J. Brakebill, A. Sekellick, M. Langland, Q. Zhang, G. Shenk, J. Keisman and C. Hupp, 2020. Sediment dynamics and implications for management: State of the science from long-term research in the Chesapeake Bay watershed, USA. *Wiley Interdisciplinary Reviews: Water* 7:e1454, DOI: 10.1002/wat2.1454.

Noe, G. B. and C. R. Hupp, 2009. Retention of riverine sediment and nutrient loads by coastal plain floodplains. *Ecosystems* 12:728-746, DOI: 10.1007/s10021-009-9253-5.

Phelps, H. L., 1994. The asiatic clam (Corbicula fluminea) invasion and system-level ecological change in the Potomac River Estuary near Washington, D.C. *Estuaries* 17:614-621, DOI: 10.2307/1352409.

Ruhl, H. A. and N. B. Rybicki, 2010. Long-term reductions in anthropogenic nutrients link to improvements in Chesapeake Bay habitat. *Proc. Natl. Acad. Sci. U. S. A.* 107:16566-16570, DOI: 10.1073/pnas.1003590107.

Scully, M. E., 2010. Wind modulation of dissolved oxygen in Chesapeake Bay. *Estuaries Coasts* 33:1164-1175, DOI: 10.1007/s12237-010-9319-9.

Sharpley, A., H. P. Jarvie, A. Buda, L. May, B. Spears and P. Kleinman, 2013. Phosphorus legacy: Overcoming the effects of past management practices to mitigate future water quality impairment. *J. Environ. Qual.* 42:1308-1326, DOI: 10.2134/jeq2013.03.0098.

Sharpley, A. N., 1980. The enrichment of soil phosphorus in runoff sediments. *J. Environ. Qual.* 9:521-526, DOI: 10.2134/jeq1980.00472425000900030039x.

Smith, E. M. and W. M. Kemp, 1995. Seasonal and regional variations in plankton community production and respiration for Chesapeake Bay. *Mar. Ecol. Prog. Ser.* 116:217-231, DOI.

Staver, K. W. and R. B. Brinsfield, 2001. Agriculture and water quality on the Maryland eastern shore: Where do we go from here? *Bioscience* 51:859-868, DOI: 10.1641/0006-3568(2001)051[0859:Aawqot]2.0.Co;2.

Tango, P. J. and R. A. Batiuk, 2013. Deriving Chesapeake Bay water quality standards. *J. Am. Water Resour. Assoc.* 49:1007-1024, DOI: 10.1111/jawr.12108.

Testa, J. M., J. B. Clark, W. C. Dennison, E. C. Donovan, A. W. Fisher, W. Ni, M. Parker, D. Scavia, S. E. Spitzer, A. M. Waldrop, V. M. D. Vargas and G. Ziegler, 2017. Ecological forecasting and the science of hypoxia in Chesapeake Bay. *Bioscience* 67:614-626, DOI: 10.1093/biosci/bix048.

Testa, J. M. and W. M. Kemp, 2012. Hypoxia-induced shifts in nitrogen and phosphorus cycling in Chesapeake Bay. *Limnol. Oceanogr.* 57:835-850, DOI: 10.4319/lo.2012.57.3.0835.

Testa, J. M., V. Lyubchich and Q. Zhang, 2019. Patterns and trends in Secchi disk depth over three decades in the Chesapeake Bay estuarine complex. *Estuaries Coasts* 42:927-943, DOI: 10.1007/s12237-019-00547-9.

U.S. Environmental Protection Agency, 2003. Ambient water quality criteria for dissolved oxygen, water clarity and chlorophyll-a for the Chesapeake Bay and its tidal tributaries. USEPA Region III Chesapeake Bay Program Office EPA 903-R-03-002, Annapolis, Maryland.

U.S. Environmental Protection Agency, 2004. Chesapeake Bay Program analytical segmentation scheme: Revisions, decisions and rationales 1983-2003. USEPA Region III Chesapeake Bay Program Office EPA 903-R-04-008, Annapolis, Maryland, p. 64.

Wynn, T. and S. Mostaghimi, 2006. The effects of vegetation and soil type on streambank erosion, southwestern Virginia, USA. *J. Am. Water Resour. Assoc.* 42:69-82, DOI: 10.1111/j.1752-1688.2006.tb03824.x.

Yarbro, L. A., P. R. Carlson, T. R. Fisher, J. P. Chanton and W. M. Kemp, 1983. A sediment budget for the Choptank River estuary in Maryland, U.S.A. *Estuar. Coast. Shelf Sci.* 17:555-570, DOI: 10.1016/0272-7714(83)90007-0.

Zhang, Q., T. R. Fisher, E. M. Trentacoste, C. Buchanan, A. B. Gustafson, R. Karrh, R. R. Murphy, J. Keisman, C. Wu, R. Tian, J. M. Testa and P. J. Tango, 2021. Nutrient limitation of phytoplankton in Chesapeake Bay: Development of an empirical approach for water-quality management. *Water Res.* 188:116407, DOI: 10.1016/j.watres.2020.116407.

Zhang, Q., R. R. Murphy, R. Tian, M. K. Forsyth, E. M. Trentacoste, J. Keisman and P. J. Tango, 2018a. Chesapeake Bay's water quality condition has been recovering: Insights from a multimetric indicator assessment of thirty years of tidal monitoring data. *Sci. Total Environ.* 637-638:1617-1625, DOI: 10.1016/j.scitotenv.2018.05.025.

Zhang, Q., P. J. Tango, R. R. Murphy, M. K. Forsyth, R. Tian, J. Keisman and E. M. Trentacoste, 2018b. Chesapeake Bay dissolved oxygen criterion attainment deficit: Three decades of temporal and spatial patterns. *Frontiers in Marine Science* 5:422, DOI: 10.3389/fmars.2018.00422.

Appendix

Additional tidal trend maps and plots are in a separate Appendix document for:

- Bottom Total Nitrogen
- Bottom Total Phosphorus
- Surface Dissolved Inorganic Nitrogen
- Surface Orthophosphate
- Surface Total Suspended Solids
- Summer Surface Dissolved Oxygen
- Surface Water Temperature