

Choptank Tributary 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)



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.

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1. Purpose and Scope

The Choptank 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 the Choptank River.

2. Location

The Choptank River watershed covers approximately 1.1% of the Chesapeake Bay watershed. Its watershed is approximately 1,844 km² (Table 1.) and is contained within parts of 2 states: Delaware and Maryland (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 Choptank River watershed is entirely located in the Coastal Plain region (Figure 1). This physiography covers lowland, dissected upland, and upland areas. Implications of these physiographies for nutrient and sediment transport are summarized in Section 5.1.1.

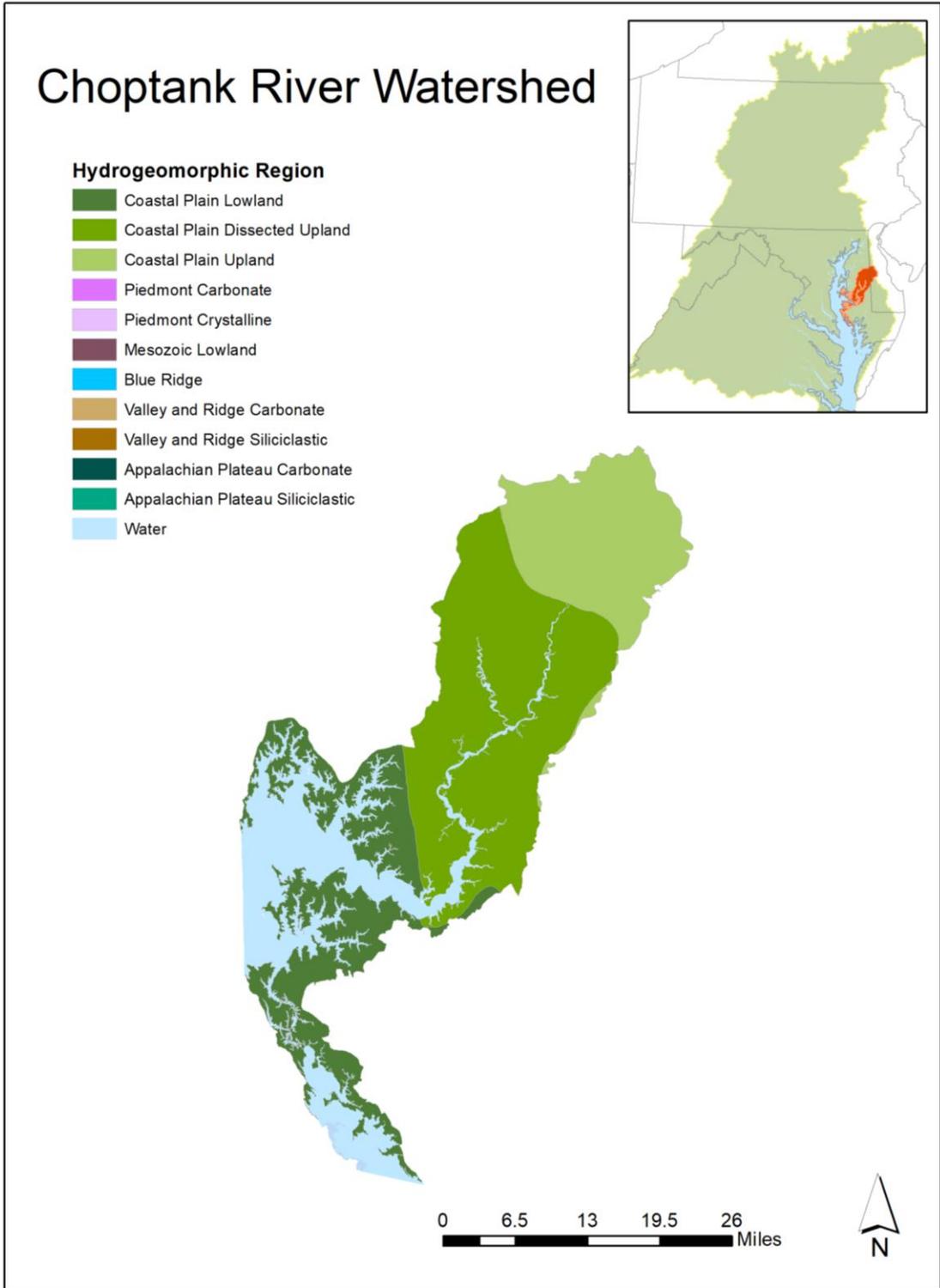


Figure 1. Distribution of physiography in the Choptank River watershed. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

2.2 Land Use

Land Use

Land use in the Choptank watershed is dominated (52%) by agriculture areas. Urban and suburban land areas have increased by 17,448 acres since 1985, agricultural lands have decreased by 9,708 acres, and natural lands have decreased by 7,887 acres. Correspondingly, the proportion of urban land in this watershed has increased from 8% in 1985 to 12% in 2019 (Figure 2).

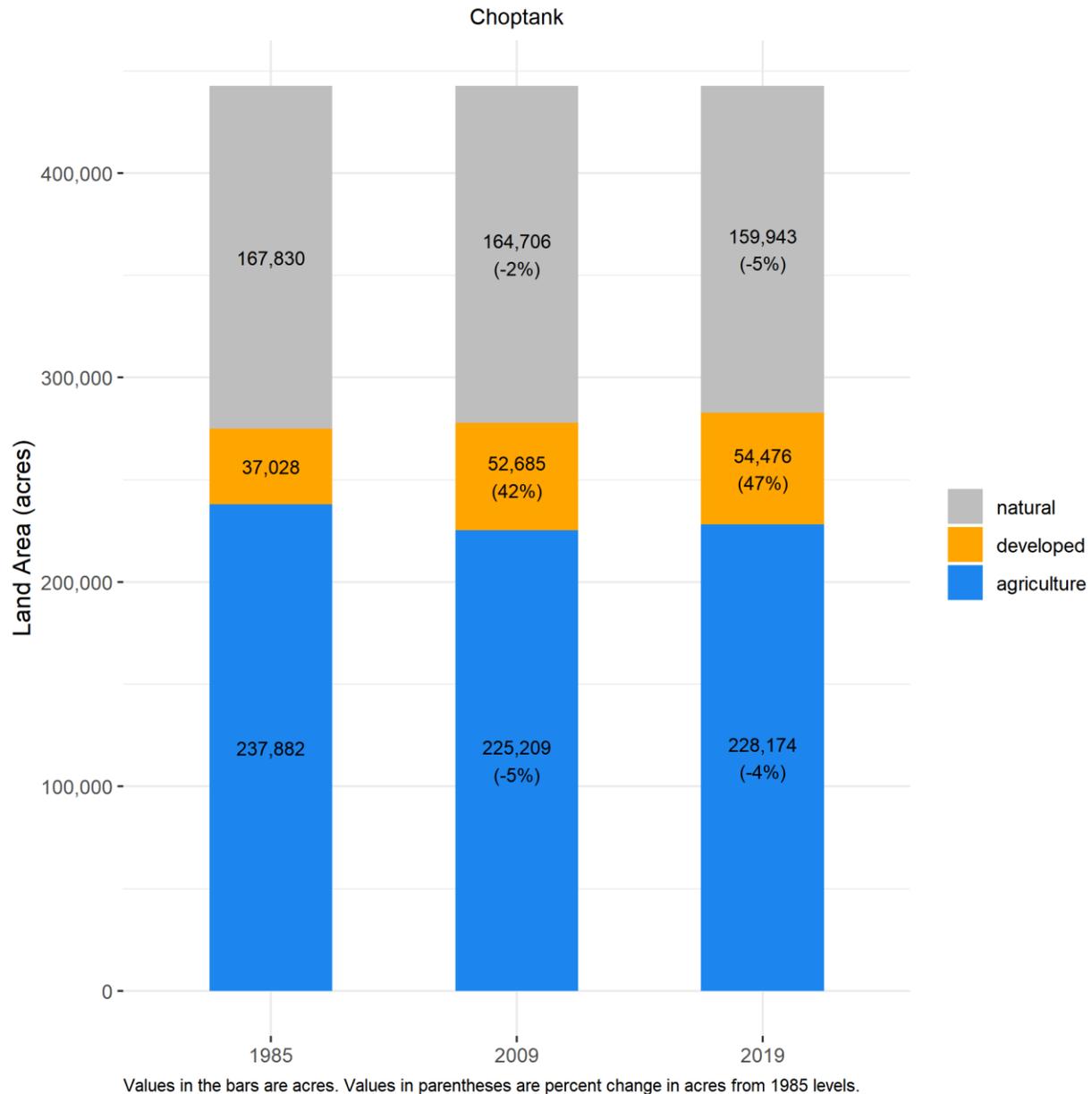


Figure 2. Distribution of land uses in the Choptank watershed. Percentages are the percent change from 1985 for each source sector.

In general, developed lands in the 1970s were more concentrated within towns and major metropolitan areas. Since then, developed and semi-developed lands have expanded around these areas, as well as

extending 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.*, 2018; Ator *et al.*, 2019). Implications of changing land use for nutrient and sediment transport are summarized in Section 5.1.3.

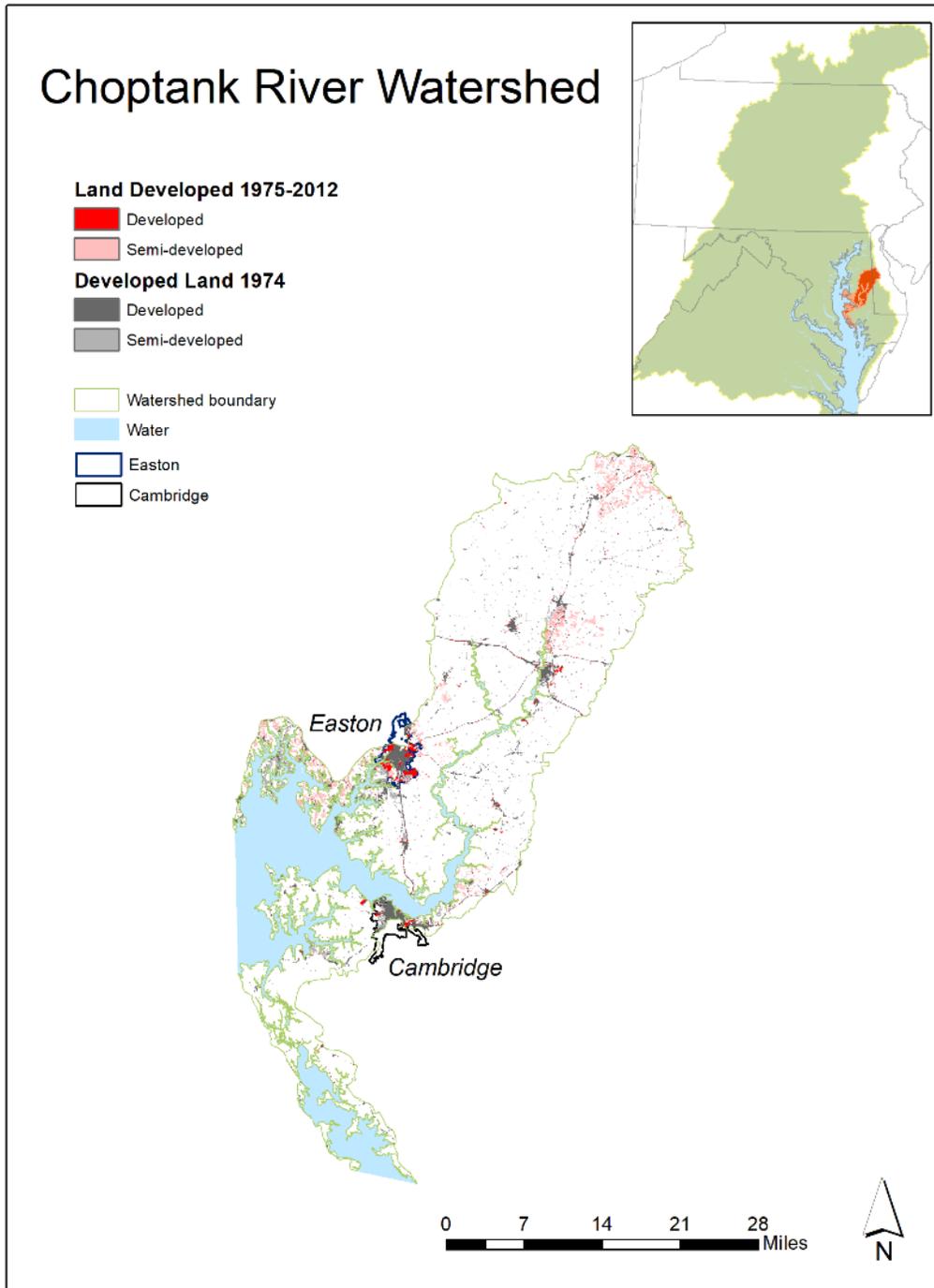


Figure 3. Distribution of developed land in the Choptank River watershed. Derived from Falcone (2015). Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

2.3 Tidal Waters and Stations

For the purposes of water quality standards assessment and reporting, the tidal portions of the Choptank River and nearby tributaries are divided into multiple segments (U.S. Environmental Protection Agency, 2004): Tidal Fresh Choptank (CHOTF), Oligohaline Choptank (CHOOH), and Mesohaline Choptank, Little Choptank River and Honga River (CHOMH2, CHOMH1, LCHMH, and HNGMH) (Figure 4).

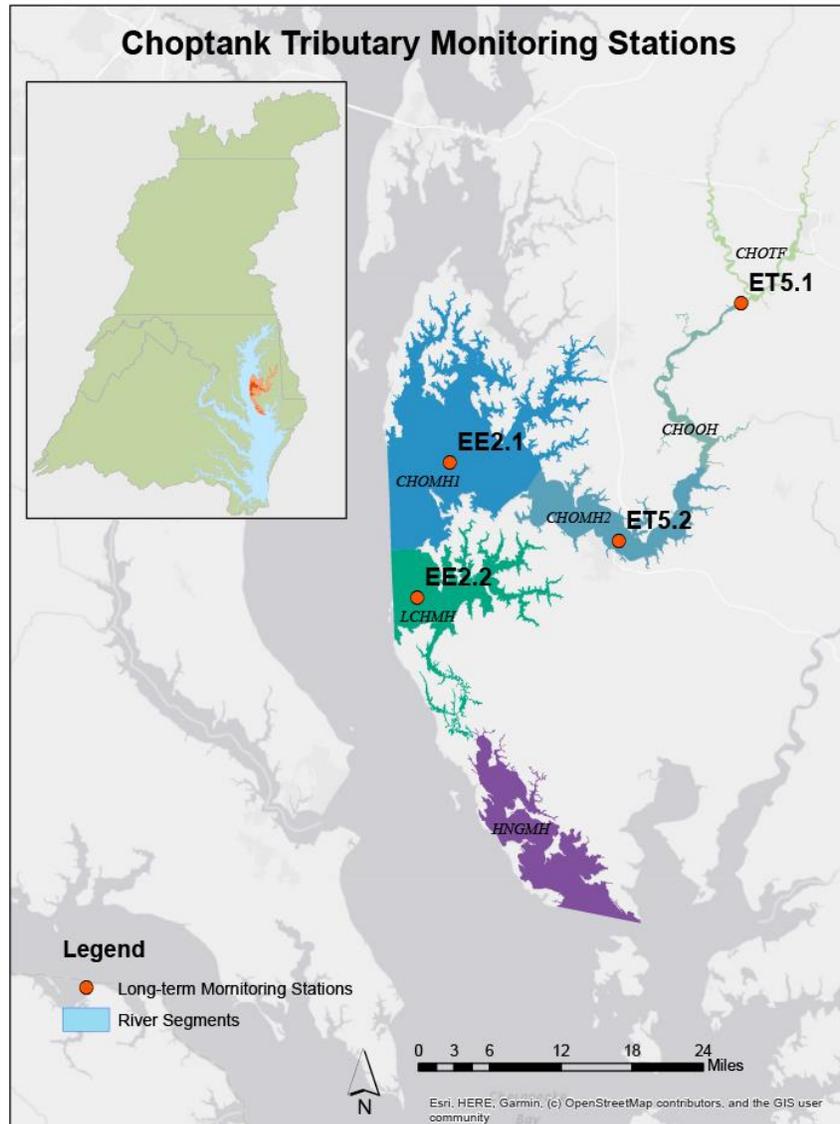


Figure 4. Map of tidal Choptank River and nearby segments and long-term monitoring stations. Base map credit Esri, HERE, Garmin, (c) OpenStreetMap contributors, and the GIS user community, World Geodetic System 1984.

Long-term trends in water quality are analyzed by MDDNR at four stations in the Choptank and Little Choptank Rivers (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). There is currently no long-term monitoring station in the Honga River, but some sampling activities have occurred in that segment in the past. In addition, shallow-water monitoring has been conducted in this region that can be included in the water quality criteria evaluation but not shown in the long-term trend graphics in subsequent sections.

3. Tidal Water Quality Dissolved Oxygen Criteria Attainment

Multiple water quality standards were developed for the Choptank, Little Choptank and Honga 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 the different tributaries have met or not met one of the Open Water (OW) 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 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 the Little Choptank mesohaline segment (LCHMH) met the 30-day mean OW summer DO requirements, while the Choptank River segments did not, and the Honga segment did not have sufficient data for evaluation (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. “ND” indicates no data.

time period	CHOTF	CHOOH	CHOMH2	CHOMH1	LCHMH	HNGMH
1985-1987						ND
1986-1988						ND
1987-1989						ND
1988-1990						ND
1989-1991						ND
1990-1992						ND
1991-1993						ND
1992-1994						ND
1993-1995						ND
1994-1996						ND
1995-1997						ND
1996-1998						ND
1997-1999						ND

1998-2000						ND
1999-2001						ND
2000-2002						ND
2001-2003						ND
2002-2004						ND
2003-2005						ND
2004-2006						ND
2005-2007						ND
2006-2008						
2007-2009						
2008-2010						
2009-2011						
2010-2012						
2011-2013						ND
2012-2014						ND
2013-2015						ND
2014-2016						ND
2015-2017						ND
2016-2018						ND

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 DO criteria shown in Table 2. is overlain with the 1985-2018 change in summer surface DO concentration (Figure 5). In this region, a mixture of trends in surface DO and criterion status exists. None of the Choptank segments met the 30-day mean OW summer DO criterion, but the direction of the surface DO trends range from degrading in the tidal fresh, to no trend at one mesohaline station and improvement at the other. The Little Choptank segment, which is meeting the OW criterion, has no change in surface DO.

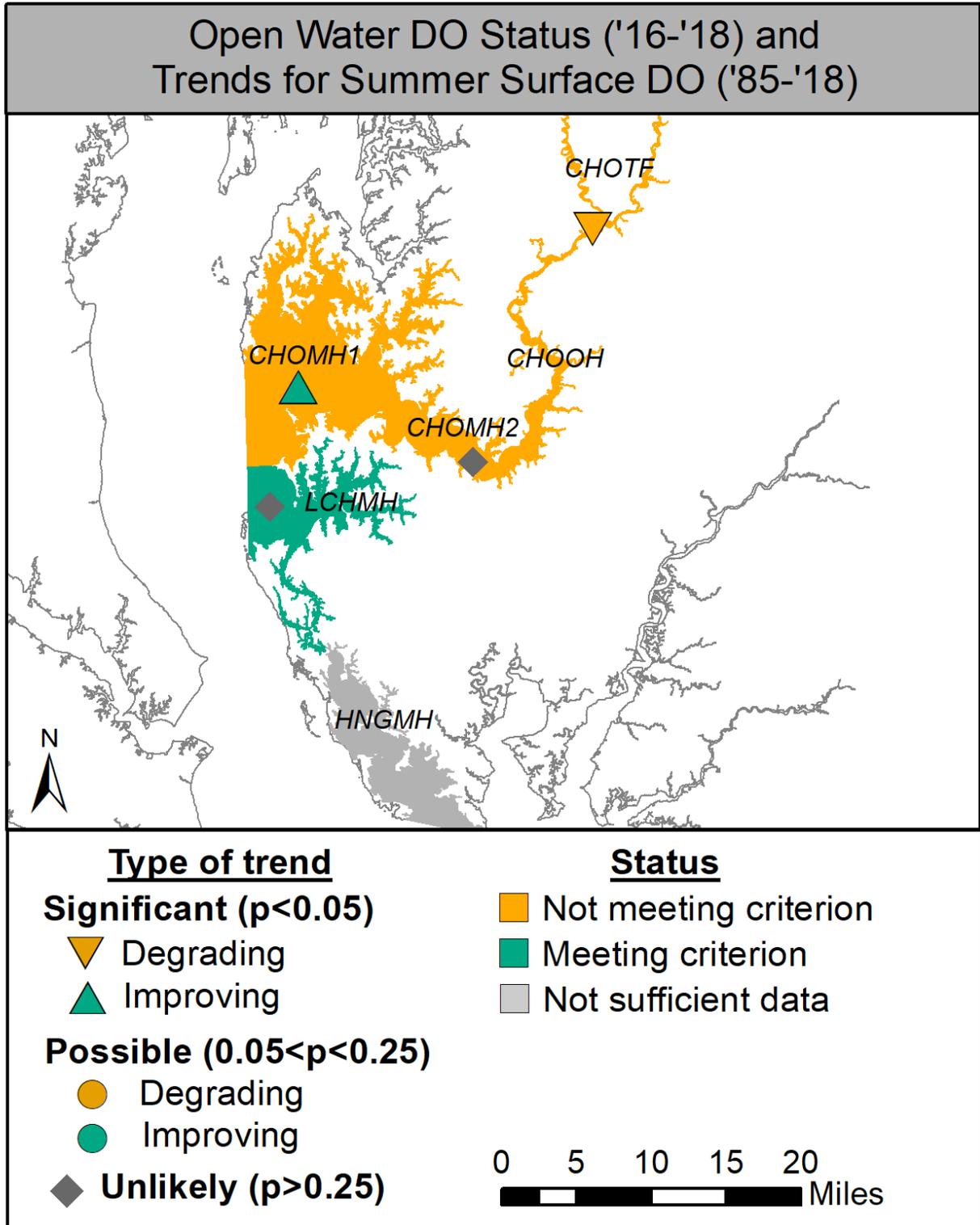


Figure 5. Pass-fail DO criterion status for 30-day OW summer DO designated use in Choptank segments along with long-term trends in DO concentrations. Base map credit Chesapeake Bay Program, **Error! Hyperlink reference not valid.**, North American Datum 1983.

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 four 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 the location in the Choptank or Little Choptank Rivers, sometimes gaged river flow is used for this adjustment and sometimes salinity is used, but we refer to all of 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 statistical 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.

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

4.1 Surface Total Nitrogen

Annual total nitrogen (TN) concentrations have improved (decreased) from 1985 to 2018 at the three mesohaline Choptank and Little Choptank stations, using both trends on concentration data alone and adjusting for flow (Figure 6). In the past 10 years, these improving trends have leveled out to be no change, with one possible degradation and one possible improvement (bottom panels Figure 6). On the other hand, the trend at the oligohaline Choptank River station (ET5.1) for both time periods, with and without flow-adjustment is degrading.

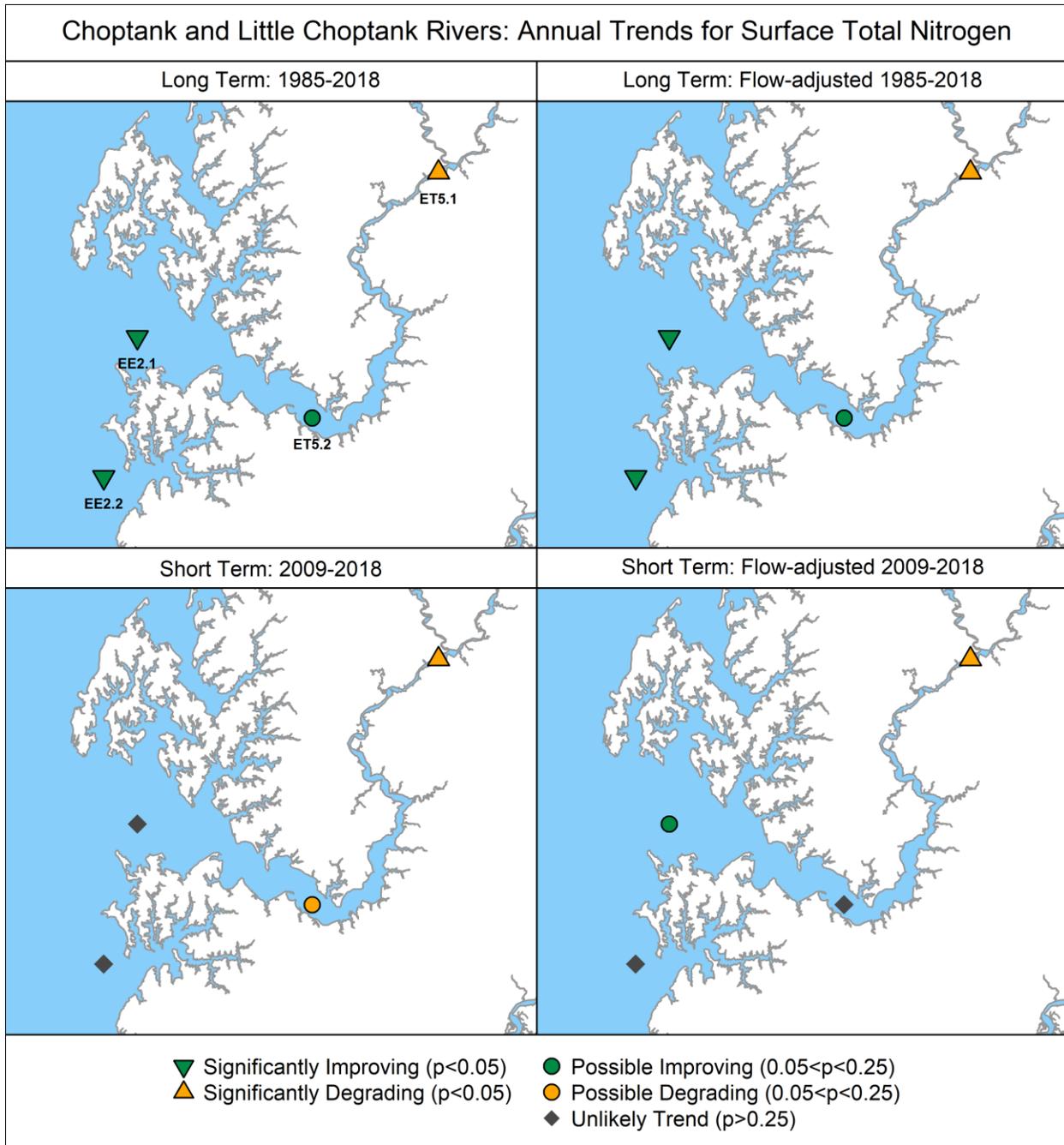


Figure 6. Surface TN trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

The long-term increase in TN at the oligohaline station (ET5.1) is evident in both the data and the non-flow-adjusted mean annual GAM estimates presented in Figure 7 (top left panel). The patterns at the other stations are less clear with more variability that is likely due to intra-annual freshwater flow variability. This is evident in the upswing in TN concentrations in 2018 which was a year with high freshwater flows. The magnitude of the observed TN is also much higher at the oligohaline station than

at the other three. 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).

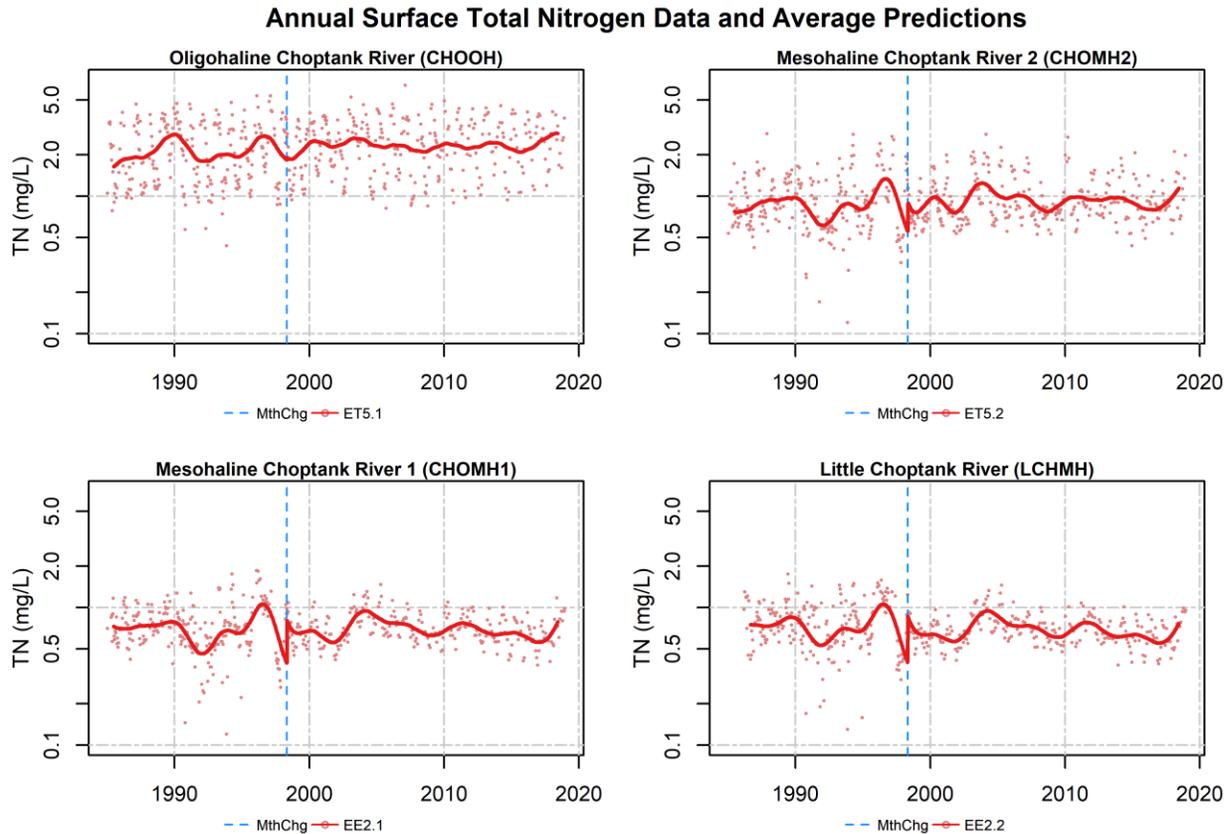


Figure 7. Surface TN data (dots) and average long-term pattern generated from non-flow adjusted GAMs. Colored dots represent data corresponding to the monitoring station shown indicated in the legend; colored lines represent mean annual GAM estimates for the noted monitoring stations. Vertical blue dotted lines represent timing of changes in laboratory and/or sampling methods.

4.2 Surface Total Phosphorus

Surface total phosphorus (TP) concentrations have improved at all the mesohaline stations over the long-term, both with and without flow-adjustment (Figure 8). The oligohaline station (ET5.1) shows a possible decrease in the observed data, but no change with flow-adjustment. In the short-term, there are no significant or possible changes at any stations in TP, indicating a leveling-out of any trends that were observed over the longer-term.

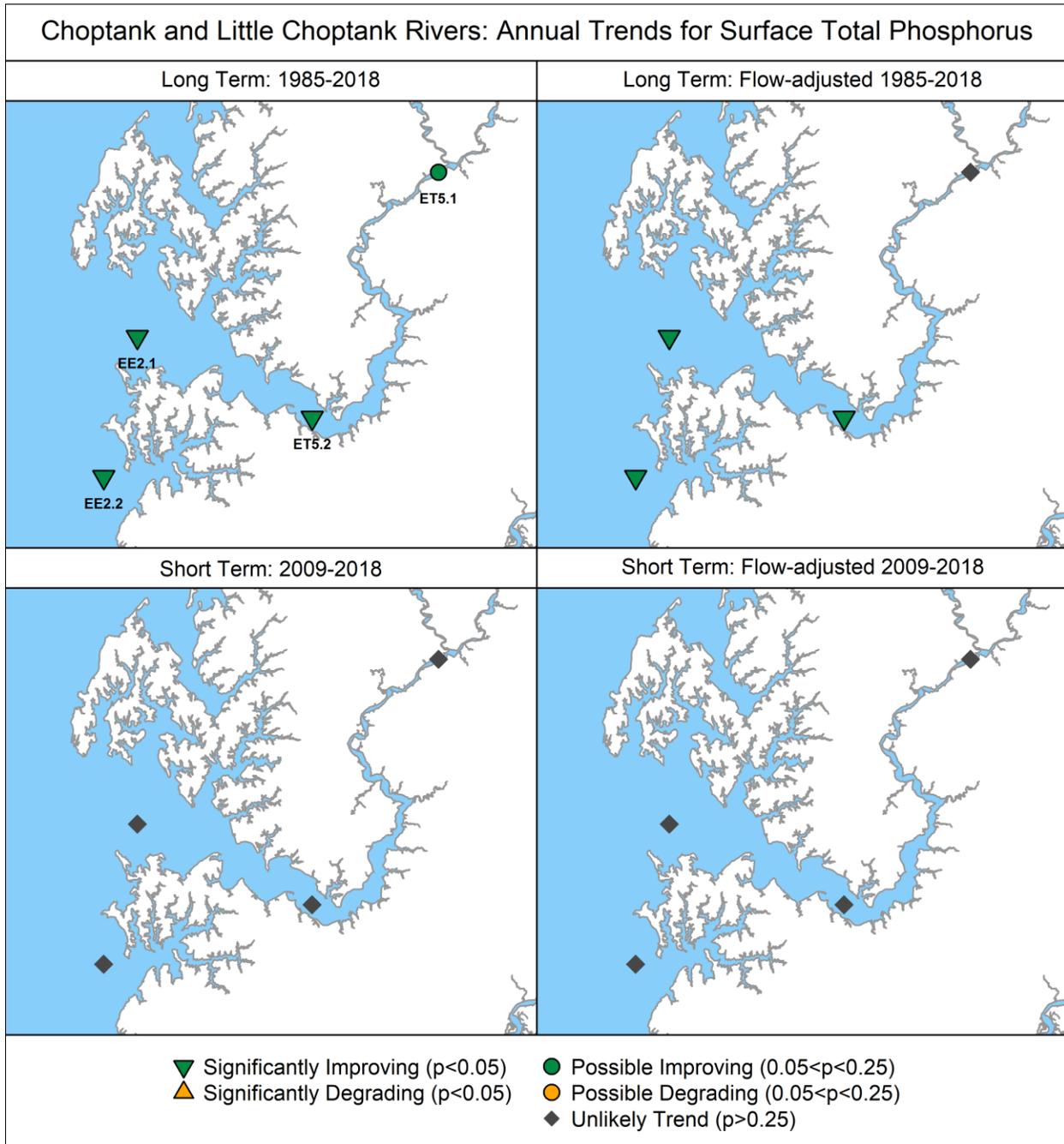


Figure 8. Surface TP trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

A noticeable TP decrease occurs at the beginning of the record at all four stations, with much more level concentrations in the second half of the record (Figure 9). The observed TP concentrations are higher at ET5.1 than at the other Choptank stations (Figure 9), which is similar to the comparison between this station and others for TN (Figure 7).

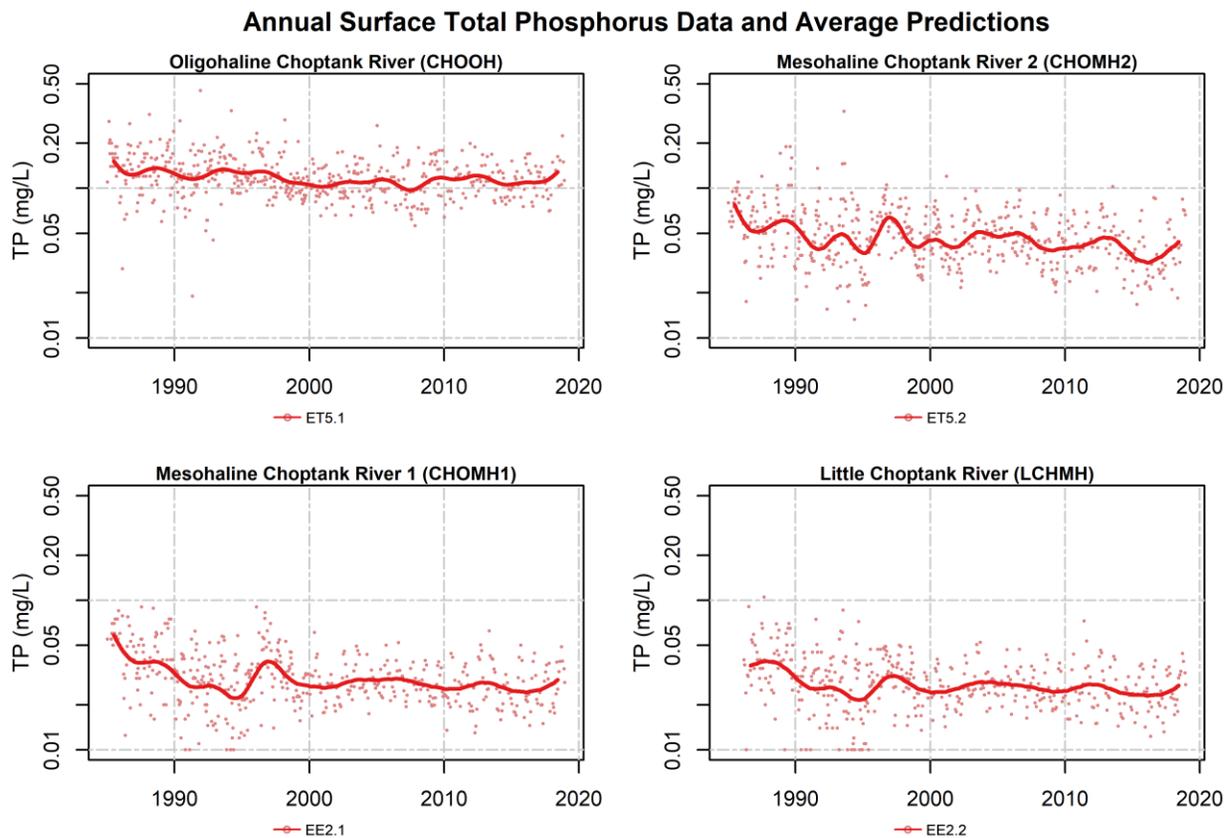


Figure 9. Surface TP data (dots) and average long-term pattern generated from non-flow adjusted GAMs. Colored dots represent data corresponding to the monitoring station shown indicated in the legend; colored lines represent mean annual GAM estimates for the noted monitoring stations.

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 are distinctly different at the oligohaline station (ET5.1) and the rest of the Choptank and Little Choptank stations (Figure 10). Long-term trends are degrading at the three mesohaline stations and possibly improving at the oligohaline station without flow-adjustment, but the two more upstream stations show no trend after flow-adjustment. Over the short-term, only the oligohaline station shows a likely degrading trend.

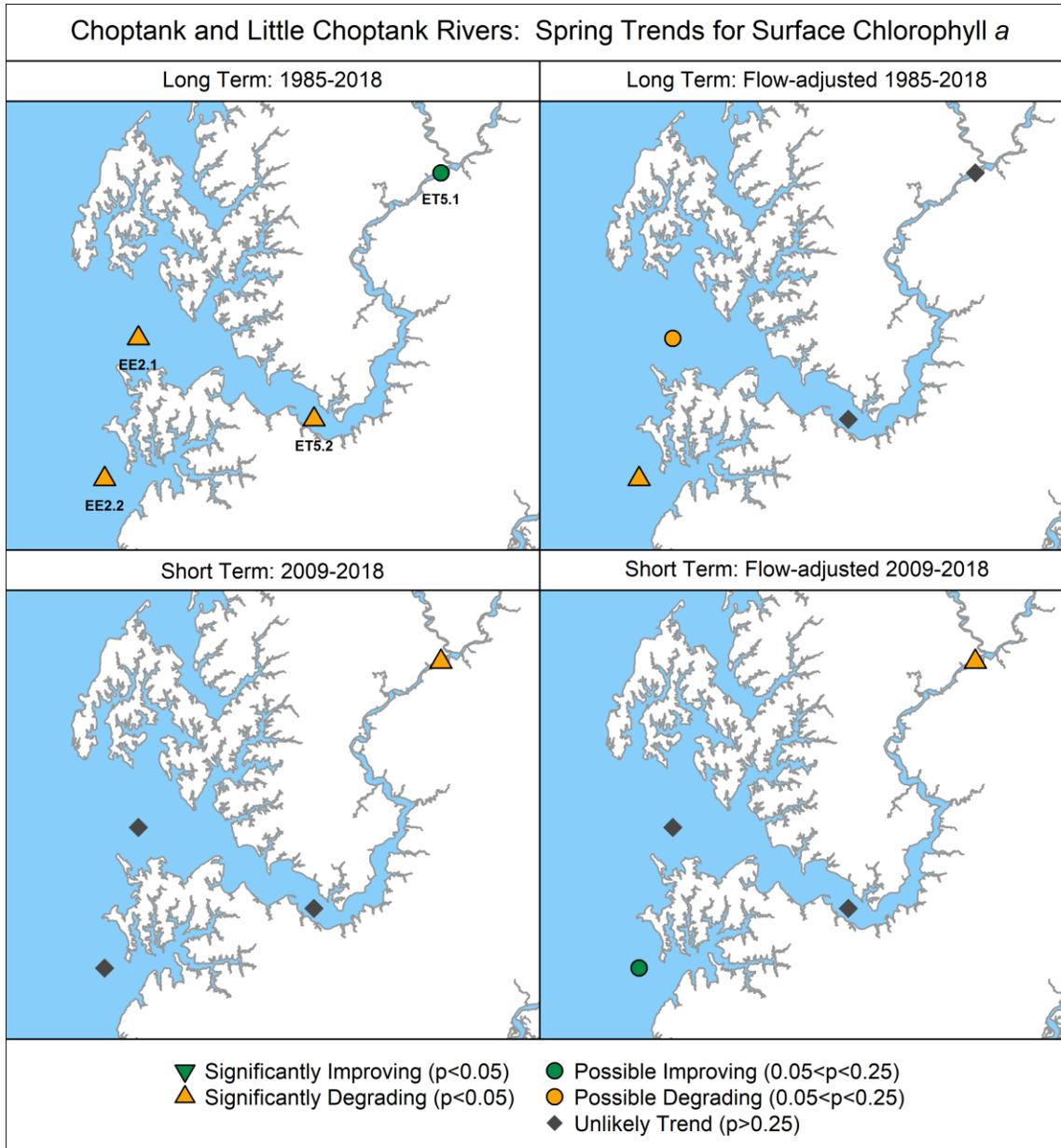


Figure 10. Surface spring(March-May) chlorophyll *a* trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

Long-term increases are apparent at the three mesohaline stations in both the spring chlorophyll *a* data sets and average spring GAM estimates (Figure 11). The oligohaline station's pattern is much more variable.

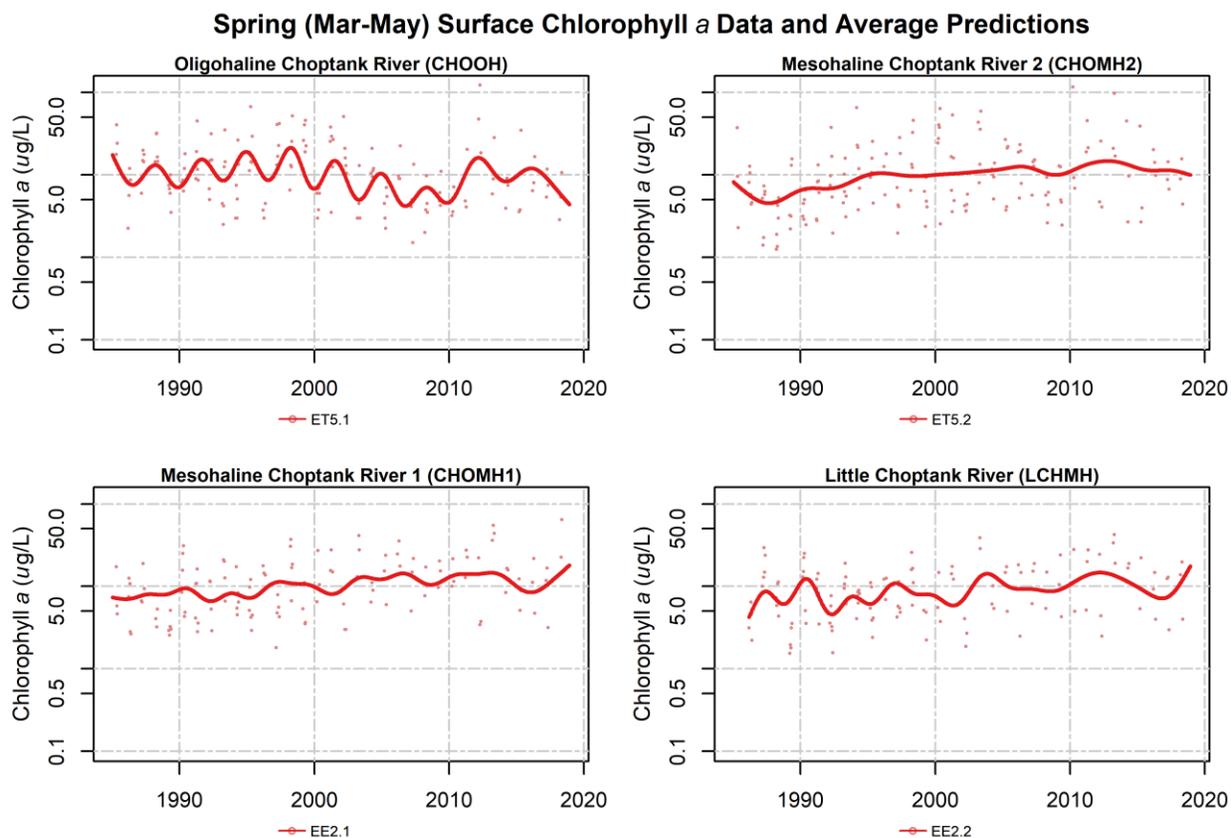


Figure 11. Surface spring chlorophyll *a* data (dots) and average long-term pattern generated from non-flow adjusted GAMs. Colored dots represent March-May data corresponding to the monitoring station shown indicated in the legend; colored lines represent mean spring GAM estimates for the noted monitoring stations.

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

The spatial pattern in summer long-term chlorophyll *a* trends is fairly similar to the spring pattern. In the summer, there is long-term degradation at the mesohaline stations and improvement at the more upstream locations (Figure 12). Over the short-term, there are no significant trends in the data and two possible improvements with flow-adjustment.

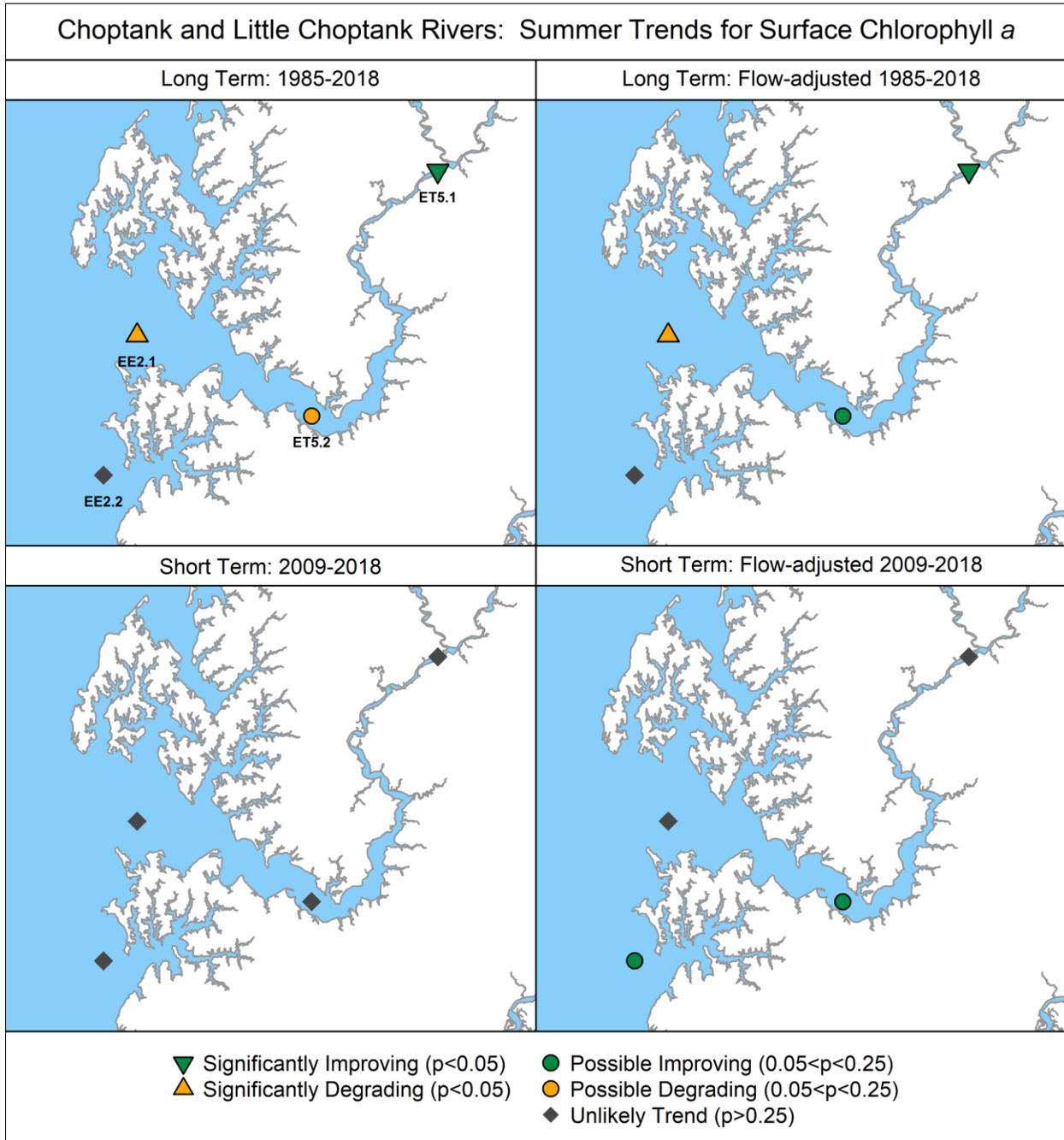


Figure 12. Surface summer (July-September) chlorophyll *a* trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

The magnitude of the summer chlorophyll *a* concentrations (Figure 13) in the oligohaline is higher than in the spring (Figure 11), although all of the patterns over time look similar between the spring and summer.

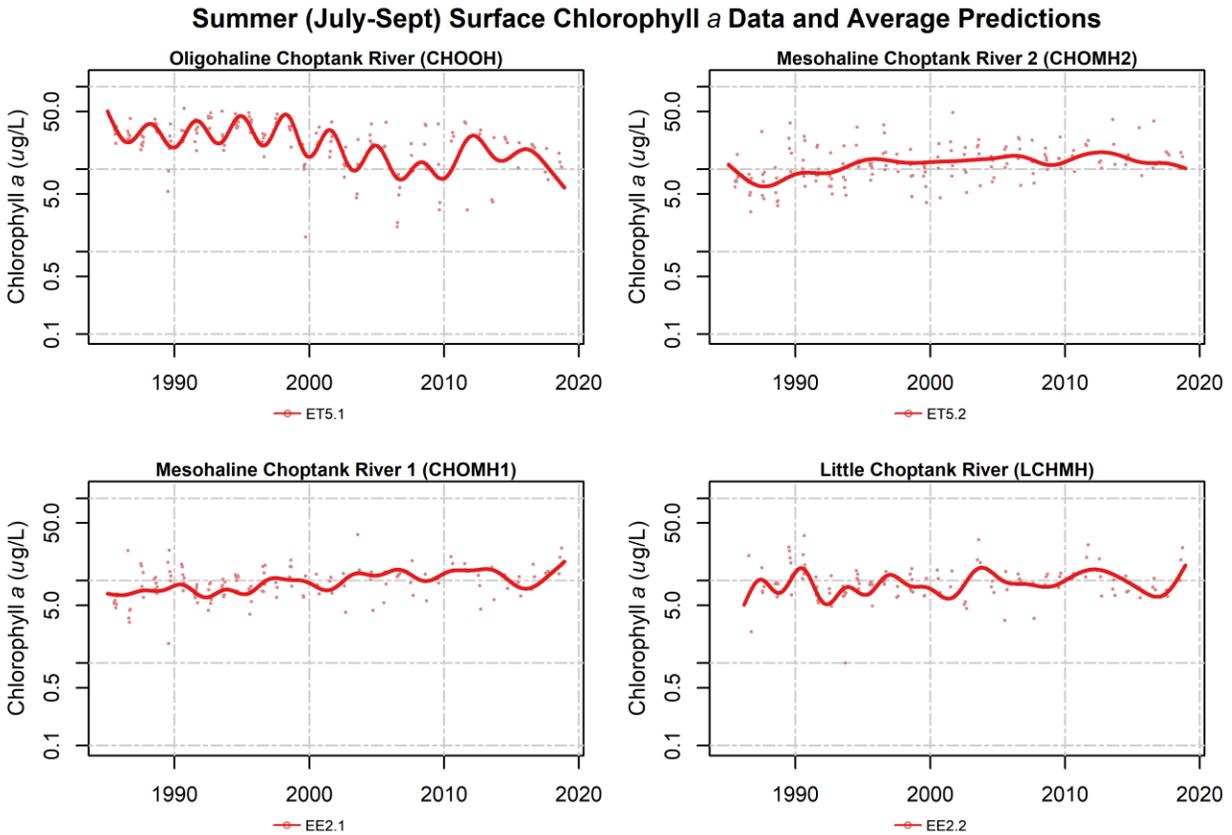


Figure 13. Surface summer (July-September) chlorophyll *a* data (dots) and average long-term pattern generated from non-flow adjusted GAMs. Colored dots represent July-September data corresponding to the monitoring station shown indicated in the legend; colored lines represent mean summer GAM estimates 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 the mesohaline station and improving at the oligohaline station without flow-adjustment (Figure 14). With flow adjustment, some of those stations are showing no trend. Over the short-term, there are also fewer Secchi depth trends.

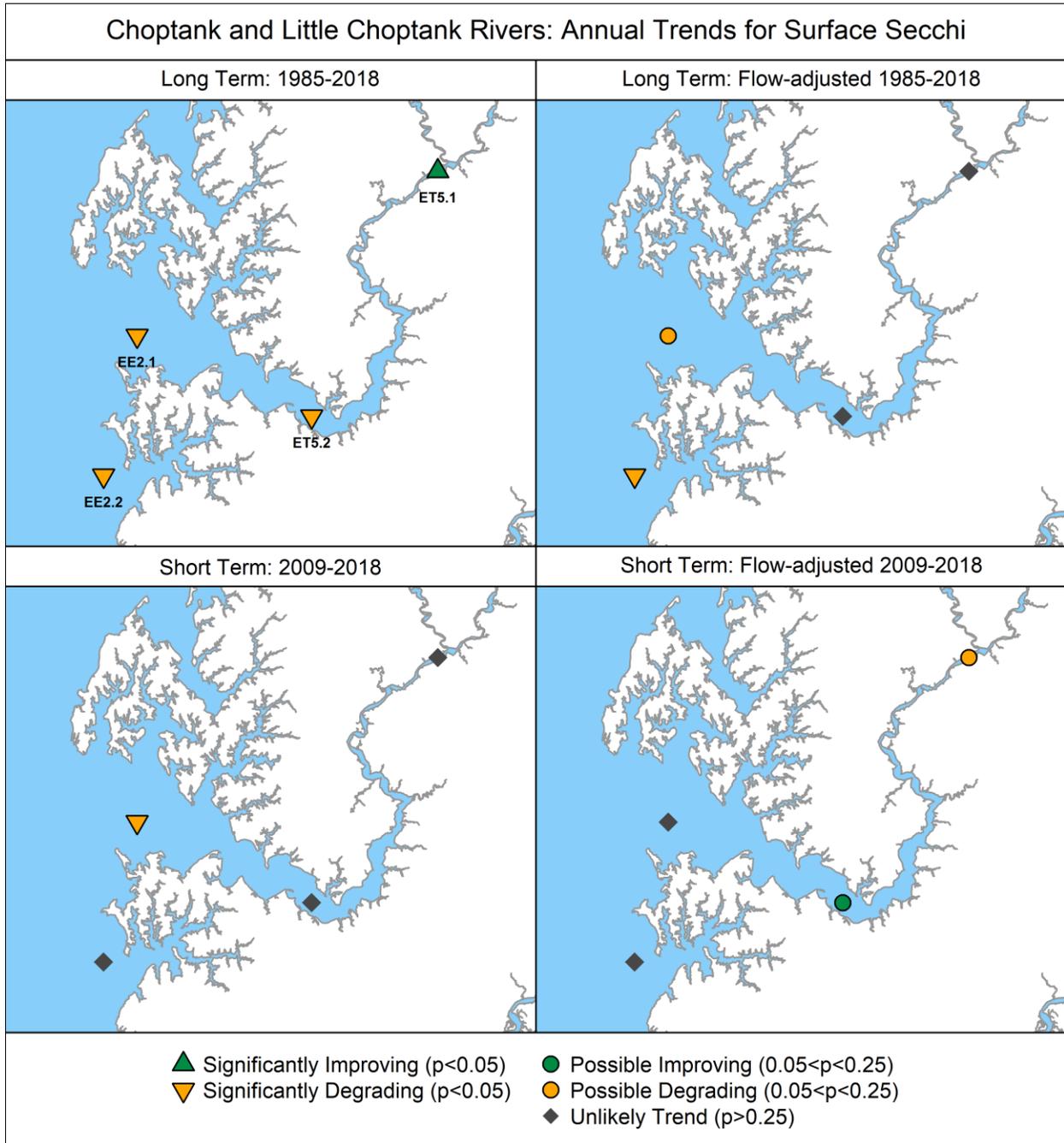


Figure 14. Annual Secchi depth trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

Secchi depth is much shallower at the oligohaline station than at the remaining mesohaline stations (Figure 15). Thus it is difficult to discern the slight increase in Secchi data set (dots on the graph) and average annual GAM estimates (line on the graph) at ET5.1, but the decrease in average annual Secchi at the other stations is clear.

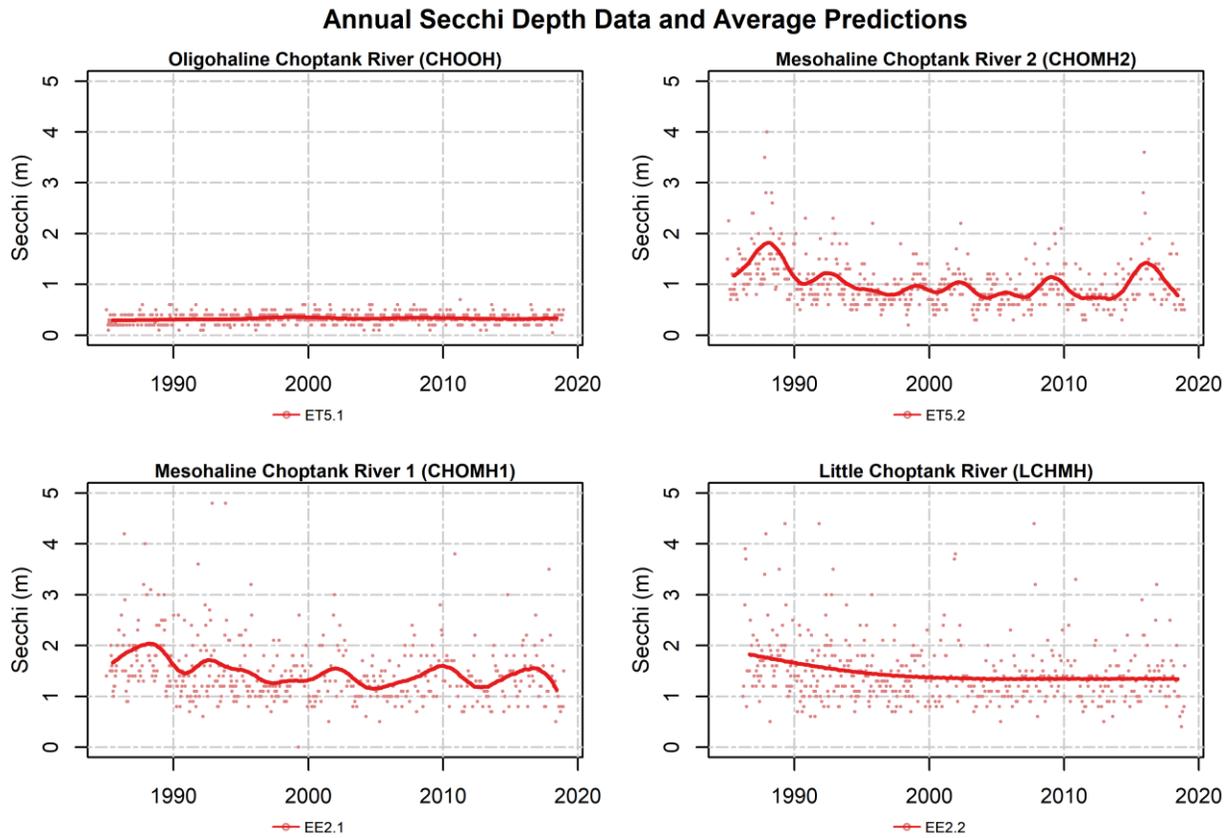


Figure 15. Annual Secchi depth data (dots) and average long-term pattern generated from non-flow adjusted GAMs. Colored dots represent data corresponding to the monitoring station shown indicated in the legend; colored lines represent mean annual GAM estimates for the noted monitoring stations.

4.6 Summer Bottom Dissolved Oxygen (June-September)

Bottom dissolved oxygen long-term trends are degrading at the two upstream stations (ET5.1 and ET 5.2) without flow-adjustment, but with flow-adjustment possible degradation is instead occurring at the other two stations (EE2.1 and EE2.2) closer to the mainstem bay (Figure 16). Over the short-term there are improvements at the two stations near the mainstem, which change to a possible degradation and no trend with flow-adjustment (Figure 16).

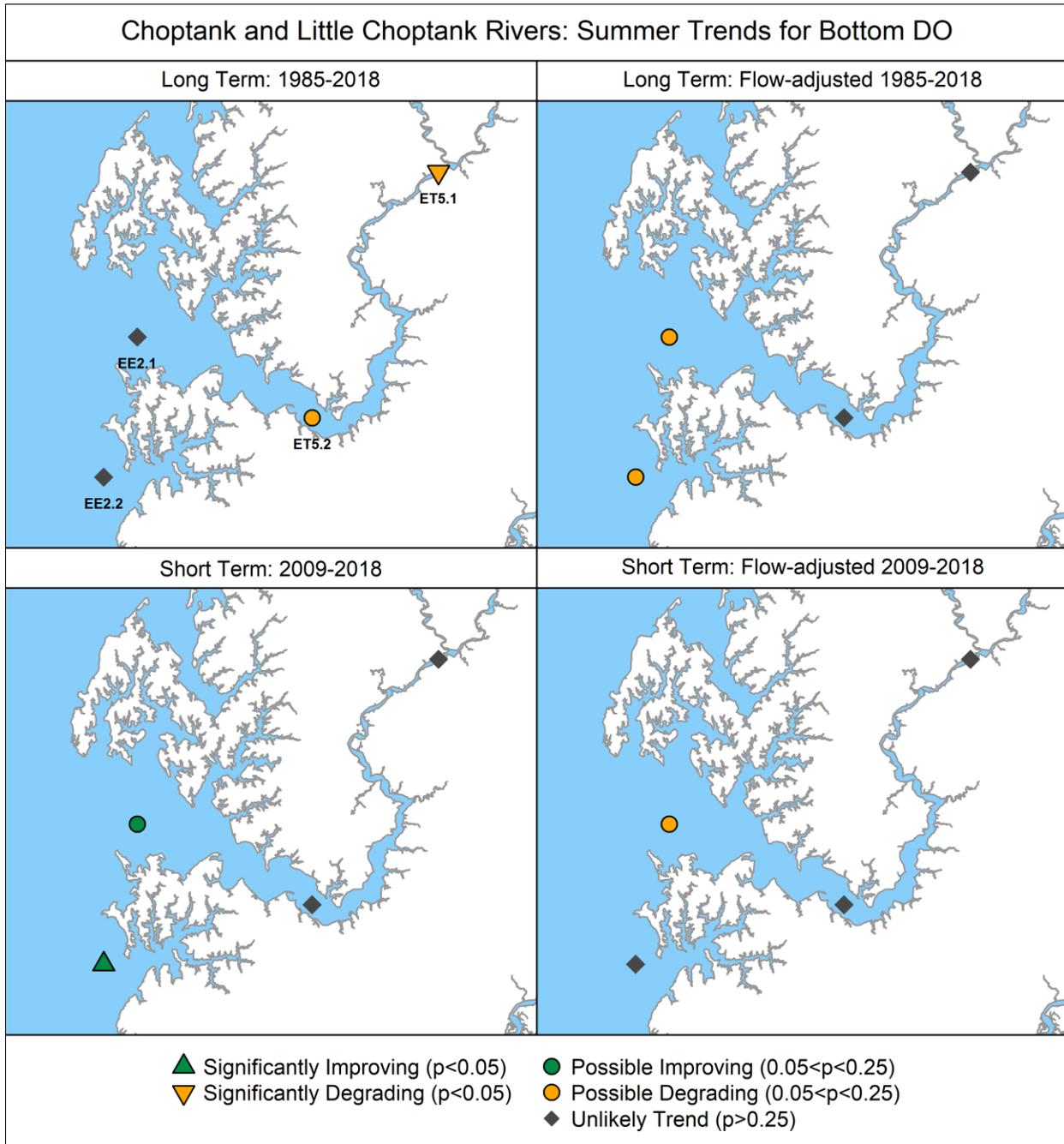


Figure 16. Summer (June-September) bottom dissolved oxygen (DO) trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

Plots of the summer data and average summer GAM estimates demonstrate the spatial variability in bottom DO concentrations (Figure 17). DO concentrations at the Choptank stations are higher than the Little Choptank station, but still frequently go below the 5 mg/L summer Open Water 30-day mean DO criterion. Bottom DO concentrations are substantially lower at the Little Choptank station (EE2.2) but appear to be improving in recent years.

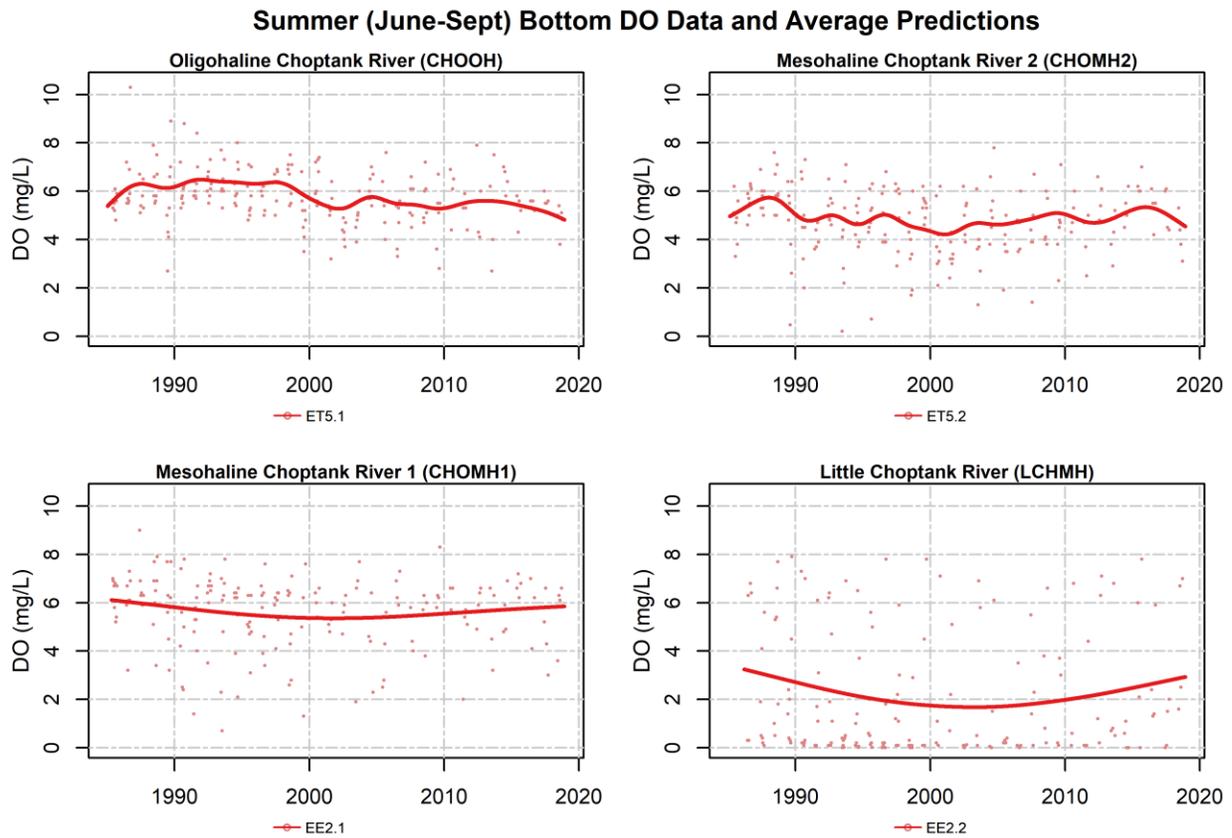


Figure 17. Summer (June-September) bottom dissolved oxygen (DO) data (dots) and July 1 long-term pattern generated from non-flow adjusted GAMs. Colored dots represent June-September data corresponding to the monitoring station shown indicated in the legend; colored lines represent mean summer GAM estimates 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.*). Sediment loads are typically low throughout the Eastern Shore because of the relatively flat topography of the Atlantic Coastal Plain.

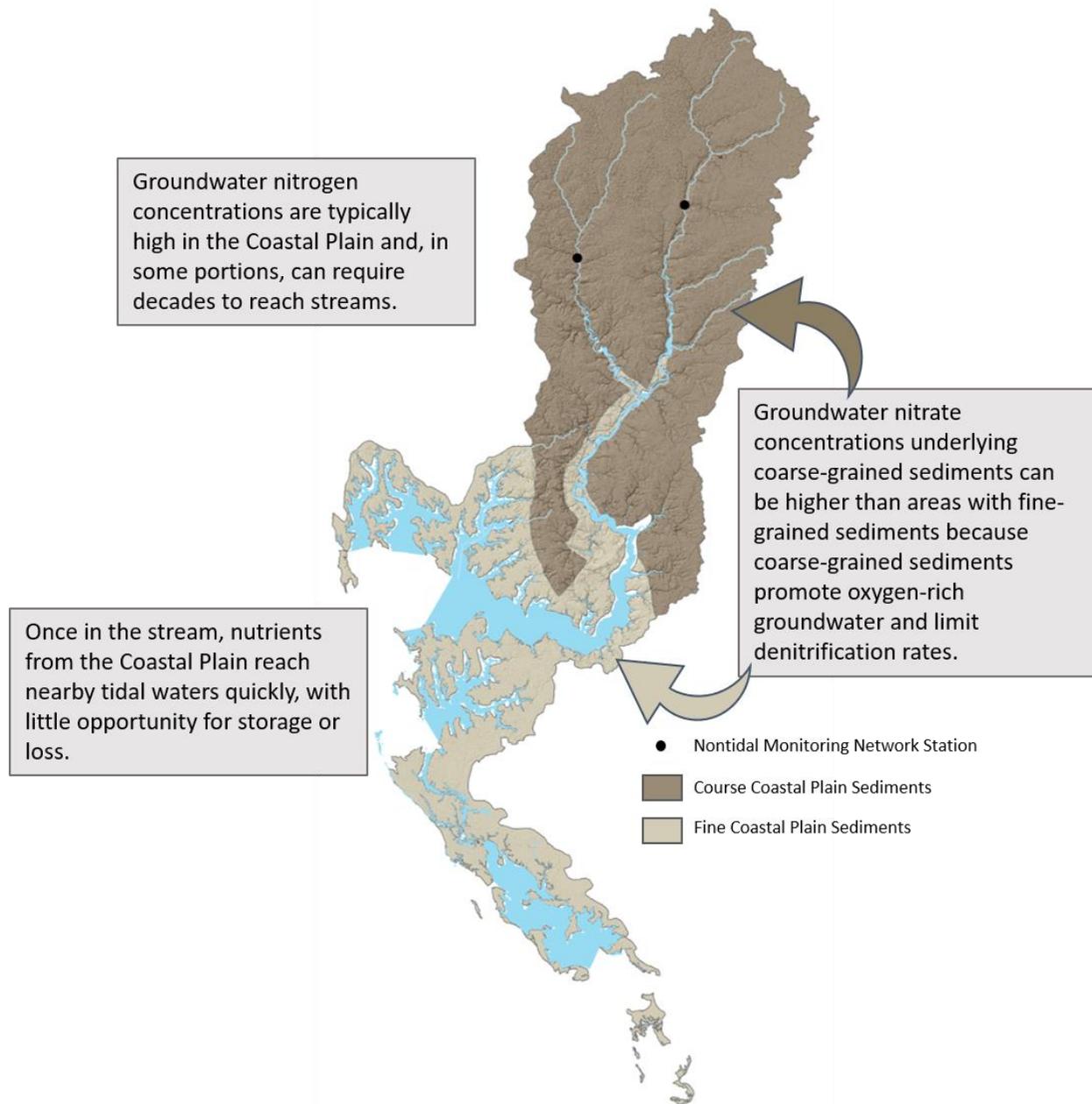


Figure 18. Effects of watershed hydrogeomorphology on nutrient transport to freshwater streams and tidal waters. Base map modified from King *et al.* (1974) and Ator *et al.* (2005), North American Datum 1983.

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 Eastern Shore, but conditions limiting denitrification are common. 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. Groundwater is the primary delivery pathway of nitrogen, as nitrate, to most streams in the Chesapeake Bay watershed (Lizarraga, 1997; Bachman *et al.*, 1998; Ator and Denver, 2012) and contributes about 70% of the nitrogen to Eastern Shore streams (Domagalski *et al.*, 2008; Ator and Denver, 2012). 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 (Greene *et al.*, 2005; Debrewer *et al.*, 2008). Eastern Shore denitrification rates are low and nitrate concentrations are high in sandy soils and sediments (Böhlke and Denver, 1995; Denver *et al.*, 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 Eastern Shore, but conditions limiting denitrification are common. 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 *et al.*, 1973; Sanford *et al.*, 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 *et al.*, 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 underlying 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 uplands 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).

Delivery to tidal waters from the non-tidal watershed

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 Chesapeake Bay tributaries are a combination of monitored fluxes from U.S. Geological Survey (USGS) River Input Monitoring (RIM) stations located at the nontidal-tidal interface and below-RIM simulated loads from the Chesapeake Bay Program Watershed Model. Nitrogen, phosphorus, and suspended sediment loads to the tidal Choptank were primarily from the below-RIM areas (Figure 19). Over the period of 1985-2018, 0.12, 0.0090, and 13.4 million tons of nitrogen, phosphorus, and suspended sediment loads were exported through the Choptank River watershed, with 93%, 94%, and 99% of those loads from the below-RIM areas, respectively.

Mann-Kendall trends and Sen's slope estimates are summarized for each loading source in Table 3.

Nitrogen

Estimated TN loads showed an overall increase of 39 ton/yr in the period between 1985 and 2018, which is not statistically significant ($p = 0.07$). This increase reflects a combination of increases in RIM loads (3.9 ton/yr; $p < 0.05$) and below-RIM loads (35 ton/yr; $p = 0.09$). The below-RIM increase is entirely driven by below-RIM nonpoint sources (40 ton/yr, $p < 0.05$). In contrast, long-term declines were observed with the below-RIM point sources (-3.7 ton/yr, $p < 0.01$) and the atmospheric deposition to the

tidal waters (-3.5 ton/yr, $p < 0.01$). The significant below-RIM point source reductions in TN are a result of substantial efforts to reduce nitrogen loads from several major wastewater treatment 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 4.3 ton/yr in the period between 1985 and 2018, which is statistically significant ($p < 0.01$). This increase reflects a combination of increases in RIM loads (0.50 ton/yr; $p < 0.01$) and below-RIM loads (3.8 ton/yr; $p < 0.01$). Within the below-RIM load, point sources showed a statistically significant decline (-1.0 ton/yr; $p < 0.01$), whereas nonpoint sources showed a statistically significant increase (4.6 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 523 ton/yr in the period between 1985 and 2018, which is statistically significant ($p < 0.05$). This increase is largely driven by below-RIM loads (477 ton/yr; $p < 0.05$), which in turn is driven by below-RIM nonpoint sources (483 ton/yr; $p < 0.05$). Like TP and TN, the below-RIM point source load of SS showed a statistically significant decline in this period (-4.0 ton/yr; $p < 0.01$).

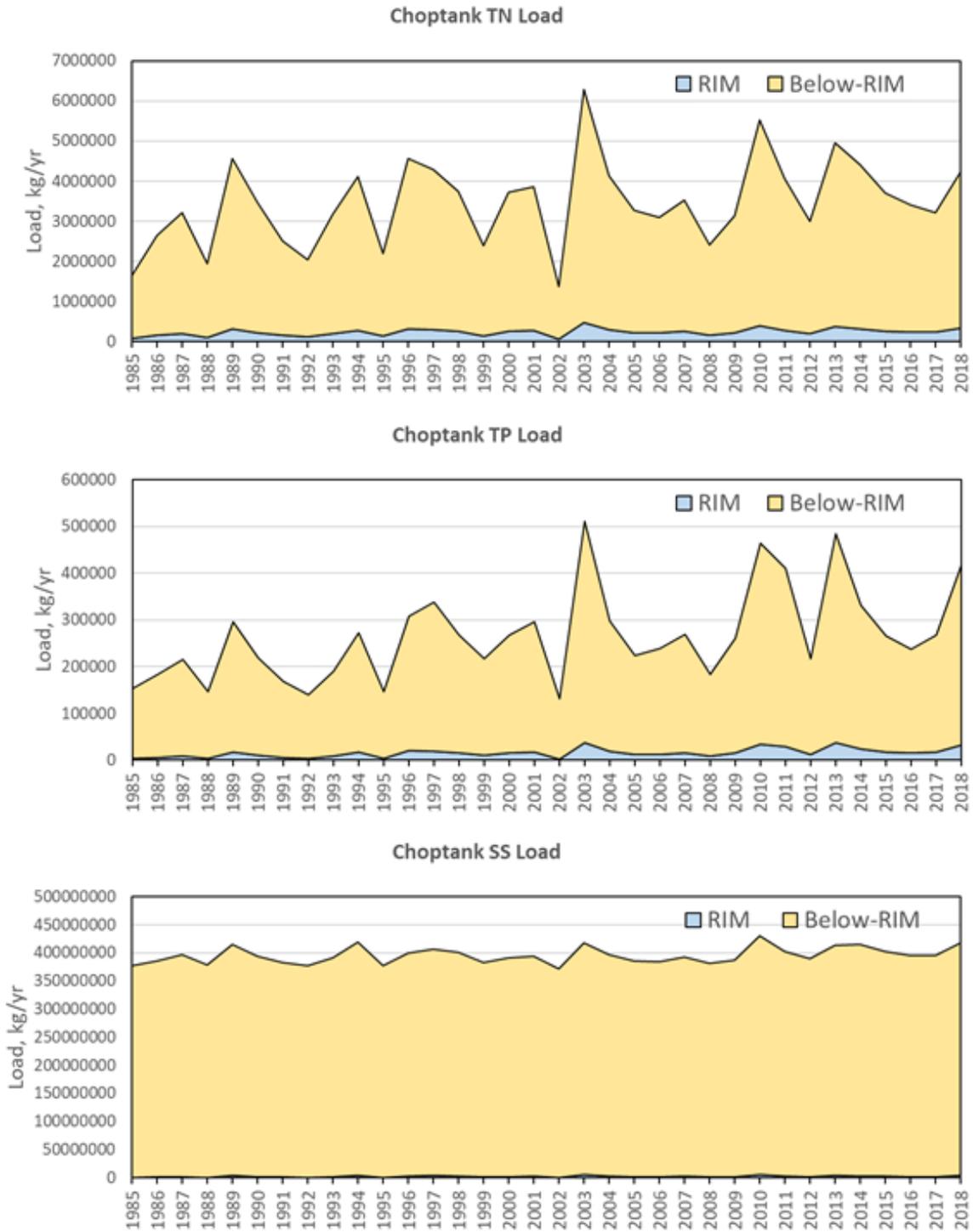


Figure 19. Estimated total loads of nitrogen (TN), phosphorus (TP), and suspended sediment (SS) from the RIM and below-RIM areas of the Choptank River. RIM refers to the USGS River Input Monitoring site located just above the head of tide of this tributary, which includes upstream point source loads. Below-RIM estimates are a combination of simulated non-point source, atmospheric deposition, and reported point-source loads.

Table 3. 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 Choptank River watershed.

Variable	Trend, metric ton/yr	Trend p-value
TN		
<i>Total watershed</i>	39	0.07
<i>RIM watershed</i> ¹	3.9	< 0.05
<i>Below-RIM watershed</i> ²	35	0.09
<i>Below-RIM point source</i>	-3.7	< 0.01
<i>Below-RIM nonpoint source</i> ³	40	< 0.05
<i>Below-RIM tidal deposition</i>	-3.5	< 0.01
TP		
<i>Total watershed</i>	4.3	< 0.01
<i>RIM watershed</i>	0.50	< 0.01
<i>Below-RIM watershed</i>	3.8	< 0.01
<i>Below-RIM point source</i>	-1.0	< 0.01
<i>Below-RIM nonpoint source</i>	4.6	< 0.01
SS		
<i>Total watershed</i>	523	< 0.05
<i>RIM watershed</i>	36	0.13
<i>Below-RIM watershed</i>	477	< 0.05
<i>Below-RIM point source</i>	-4.0	< 0.01
<i>Below-RIM nonpoint source</i>	483	< 0.05

¹ Loads for the RIM watershed were estimated loads at the USGS RIM station 01491000 (Choptank River near Greensboro, Md.; https://cbrim.er.usgs.gov/loads_query.html).

² Loads for the below-RIM watershed were obtained from the Chesapeake Bay Program Watershed Model (<https://cast.chesapeakebay.net/>).

³ Below-RIM nonpoint source loads were obtained from the Chesapeake Bay Program Watershed Model's progress runs specific to each year from 1985 and 2018, which 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>).

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 Choptank River by -17%, -44%, and -6%, 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 (-22%), natural (-5%), stream bed and bank (-16%), and wastewater (-69%) sources were partially counteracted by increases from developed (46%) and septic (37%) sources.

The two largest sources of phosphorus loads as of 2019 were the shoreline and agriculture sectors. Overall, expected declines from agriculture (-61%), natural (-6%), stream bed and bank (-54%), and wastewater (-94%) sources were partially counteracted by increases from developed (22%) sources.

For sediment, the largest sources are shoreline and stream bed and bank areas: these two sources changed by 0% and -53%, respectively between 1985 and 2019. Sediment loads from the agriculture sector changed by -62%, whereas sediment load from developed areas changed by 47%.

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.

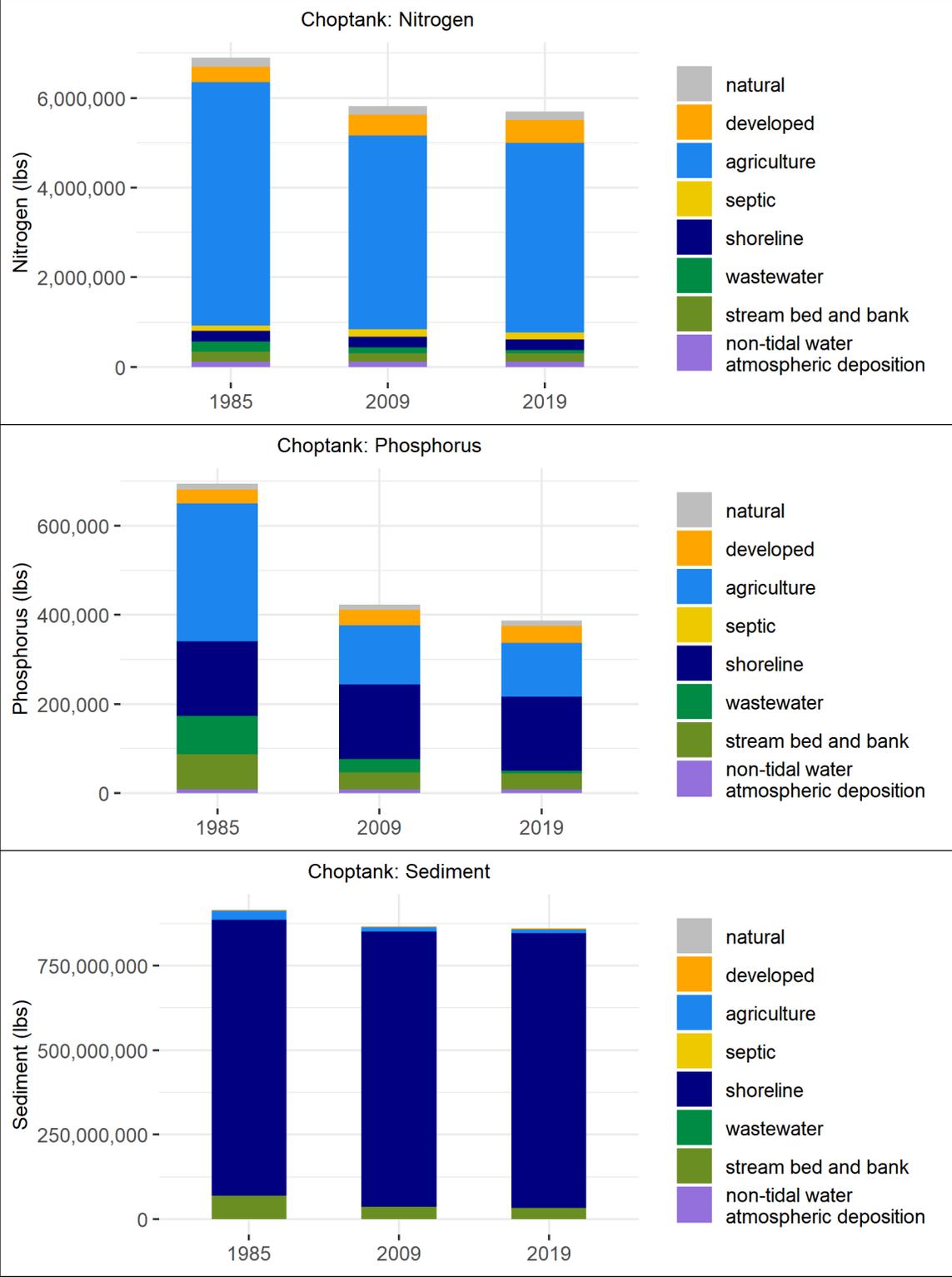
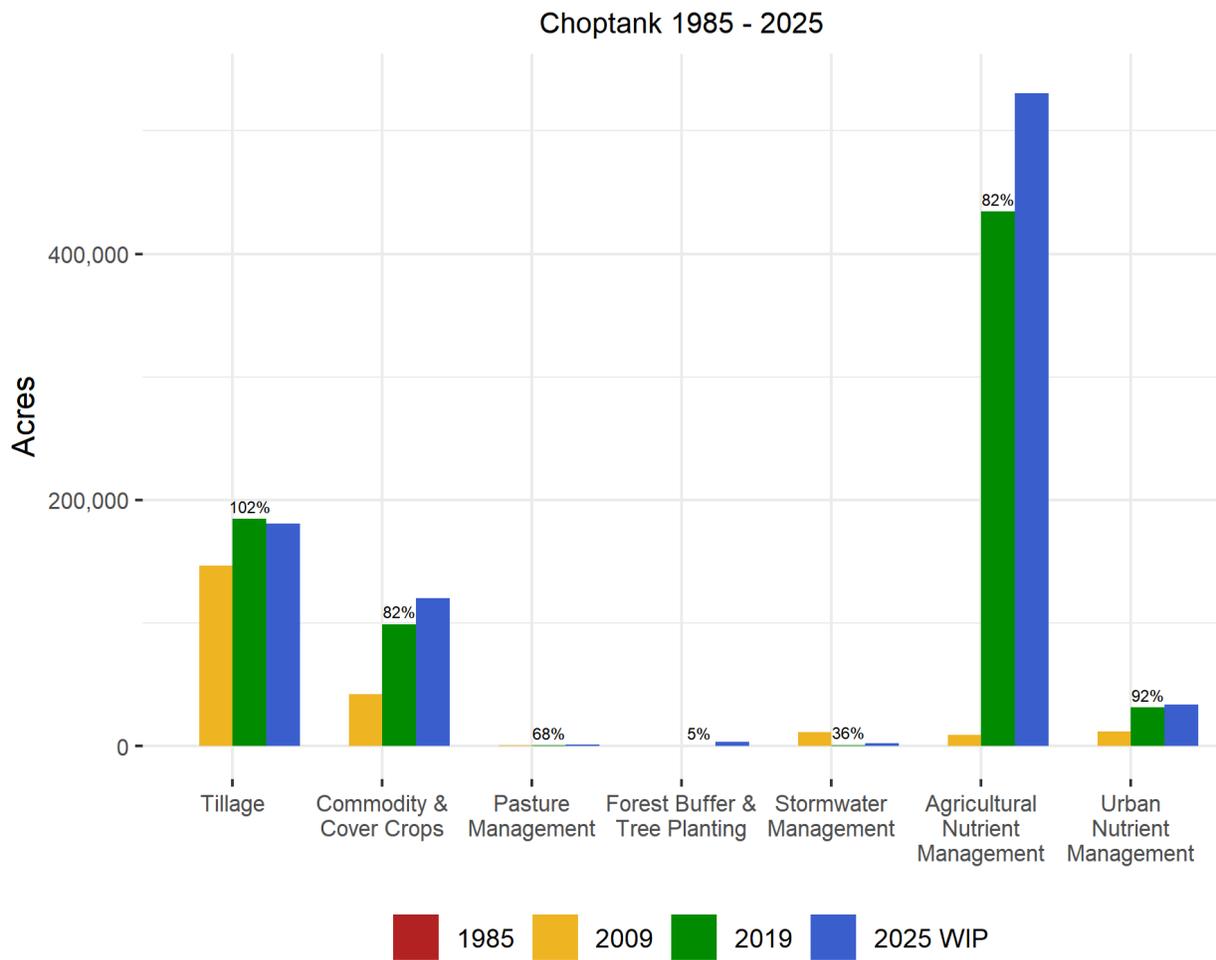


Figure 20. Expected long-term average loads of nitrogen, phosphorus, and sediment from different sources to the tidal Choptank, as obtained from the Chesapeake Assessment Scenario Tool (CAST-19). Data shown are time-average delivered loads over the average hydrology of 1991-2000, once the steady

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 185, 99, 0.7, 0.2, 0.9, 434, and 31 thousand acres, respectively. Implementation levels for some practices are already close to achieving their planned 2025 levels: for example, 102% of planned acres for tillage had been achieved as of 2019. In contrast, about 82% of planned commodity & cover crops implementation had been achieved as of 2019.



Values above the 2019 bars are the percent of the 2025 goal achieved.

Figure 21. BMP implementation in the Choptank 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 2,024 feet in 2019. Over the same period, animal waste management systems treated 7,078 animal units in 1985 and 487,542 animal units in 2019 (one animal unit represents 1,000 pounds of live animal). These implementation levels represent 22% and 95% 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 long term (1985-2019) and short term (2009-2018) at nontidal network stations throughout the watershed (Moyer and Langland, 2020) (Table 4). 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 4. Long-term (1985 - 2018) and short-term trends (2009 - 2018) of flow-normalized total nitrogen (TN), total phosphorus (TP), and suspended sediment (SS) loads for nontidal network monitoring locations in the Choptank River 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
01491000	CHOPTANK RIVER NR	1985	4.6	68.8	-25.9
	GREENSBORO, MD	2009	4.7	37.4	11.1
01491500	TUCKAHOE CREEK NEAR RUTHSBURG, MD	2009	-2.4	30.8	15.9

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).

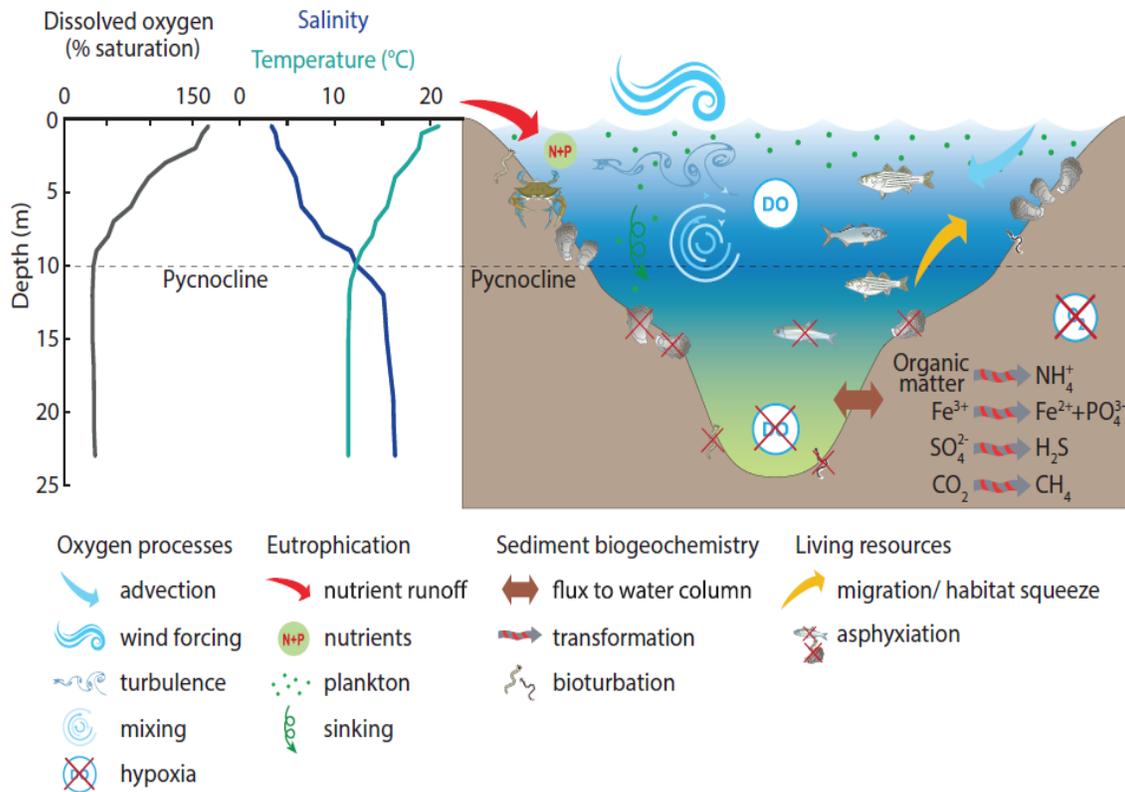


Figure 22. Conceptual diagram illustrating how hypoxia is driven by eutrophication and physical forcing, while affecting sediment biogeochemistry and living resources. From Testa *et al.* (2017).

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.

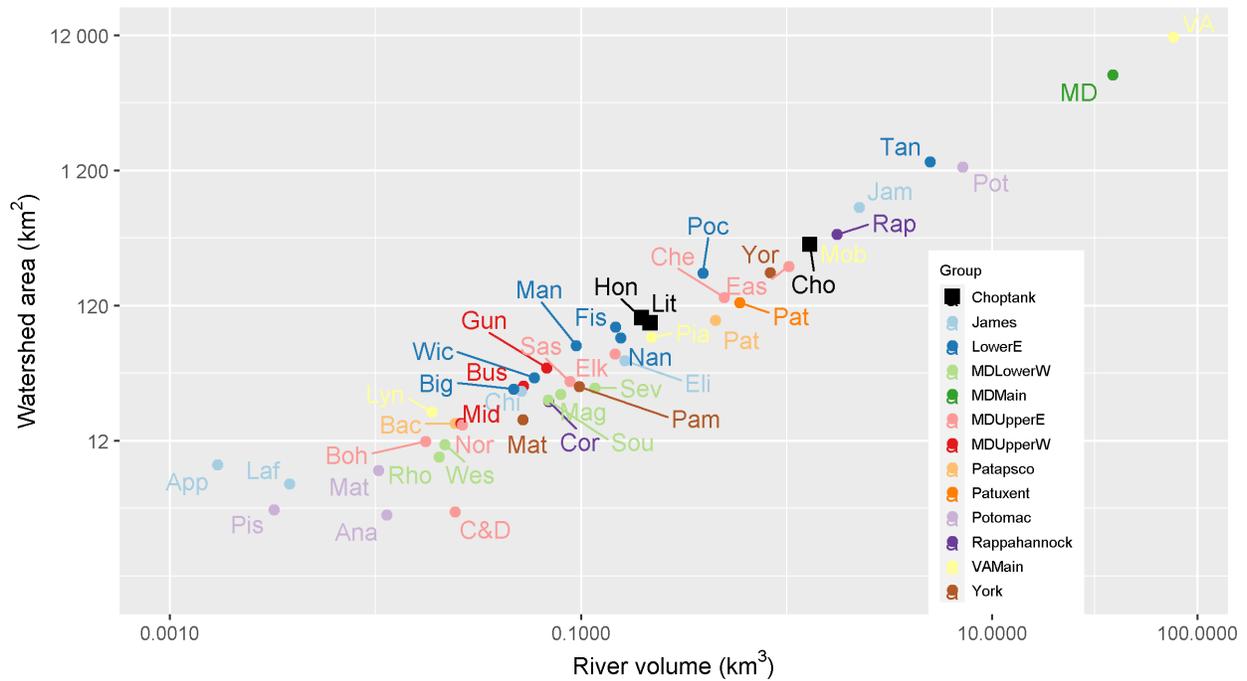


Figure 23. Watershed area vs estuarine volume.

<u>Abbreviated tributary name</u>	<u>Full tributary name</u>	<u>Abbreviated tributary name</u>	<u>Full tributary name</u>
Ana	Anacostia River	Mat	Mattaponi River
App	Appomattox River	MD	MD MAINSTEM
Bac	Back River	Mid	Middle River
Big	Big Annessex 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

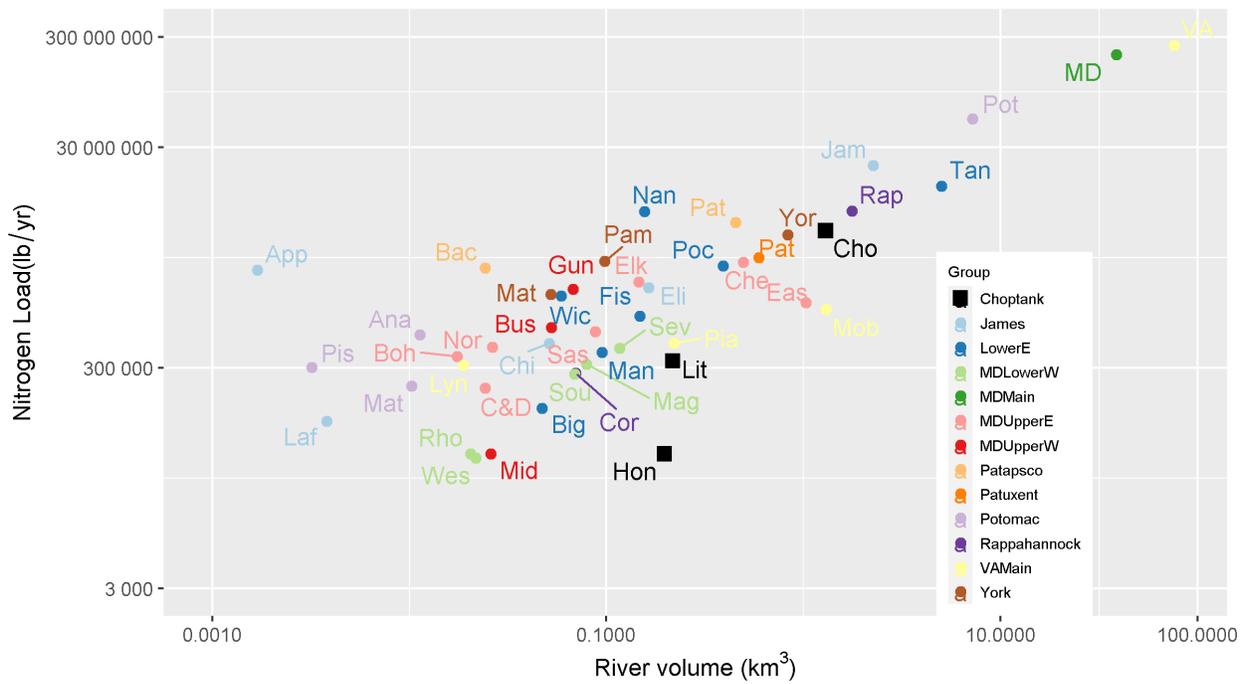


Figure 24. Annual average expected nitrogen loads versus estuarine volume. Nitrogen loads are from the 2018 progress scenarios in CAST (Chesapeake Bay Program, 2020), which is an estimate of nitrogen loads under long-term average hydrology given land use and reported management as of 2018.

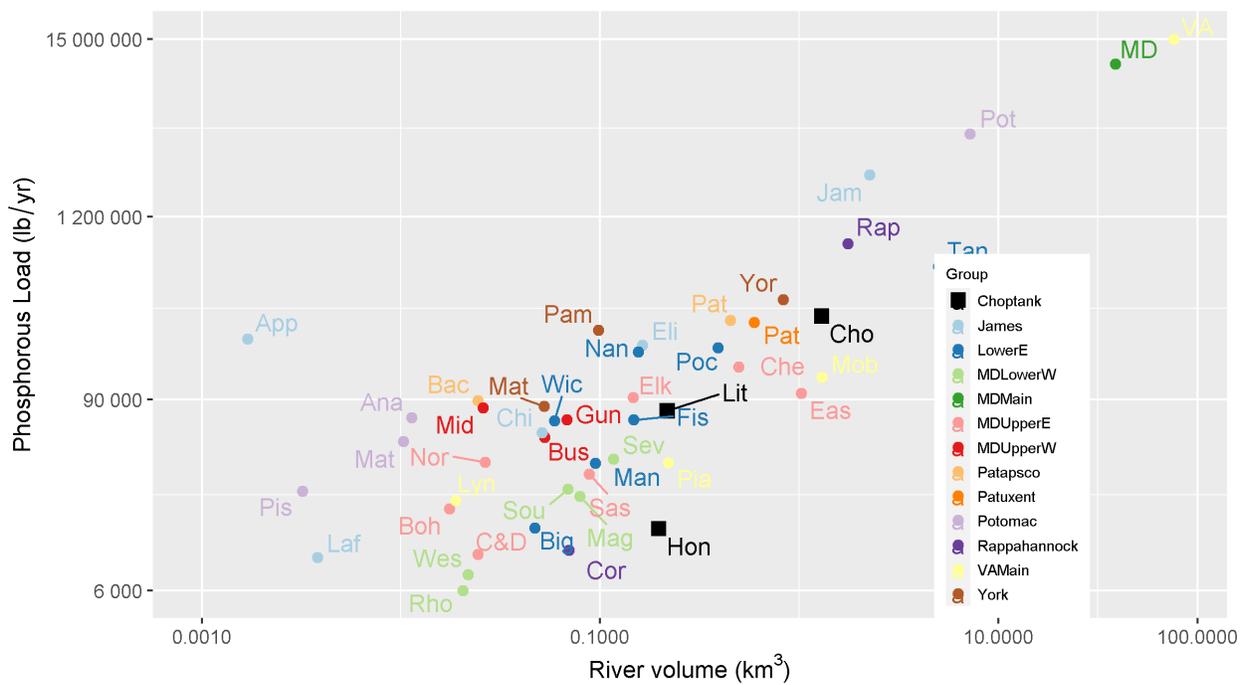


Figure 25. Annual average expected phosphorus loads versus estuarine volume. Phosphorus loads are from the 2018 progress scenarios in CAST (Chesapeake Bay Program, 2020), which is an estimate of

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

The Choptank river estuary volume and watershed contain approximately 2 and 1% of the total volume and watershed of the Chesapeake Bay. This ranks the Choptank as the 8th largest volume and 10th 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 consistent with other estuaries in the Chesapeake system, indicating a moderate level of susceptibility to eutrophication. The smaller tributaries within the Choptank system, the Honga river and Little Choptank river follow a similar trend. The Honga river has a reduced load of phosphorus relative to its estuarine volume. Both the Honga river and Little Choptank river also have low relative loads of nitrogen.

5.3 Insights on Changes in the Choptank

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.

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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