James 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)

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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 James 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 submerged 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 James River.

2. Location

The James River watershed covers approximately 15.7% of the Chesapeake Bay watershed. Its watershed is approximately 25,831 km^2 (Table 1.) and is contained within the state of Virginia (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 James River watershed stretches across five major physiographic regions, namely, Valley and Ridge, Blue Ridge, Piedmont, Mesozoic Lowland, and Coastal Plain (Bachman *et al.*, 1998) (Figure 1). The Valley and Ridge physiography covers both carbonate and siliciclastic areas. The Piedmont physiography covers both carbonate and crystalline areas. The Coastal Plain physiography covers lowland, dissected upland, and upland areas. Implications of these physiographies for nutrient and sediment transport are summarized in Section 5.1.1.

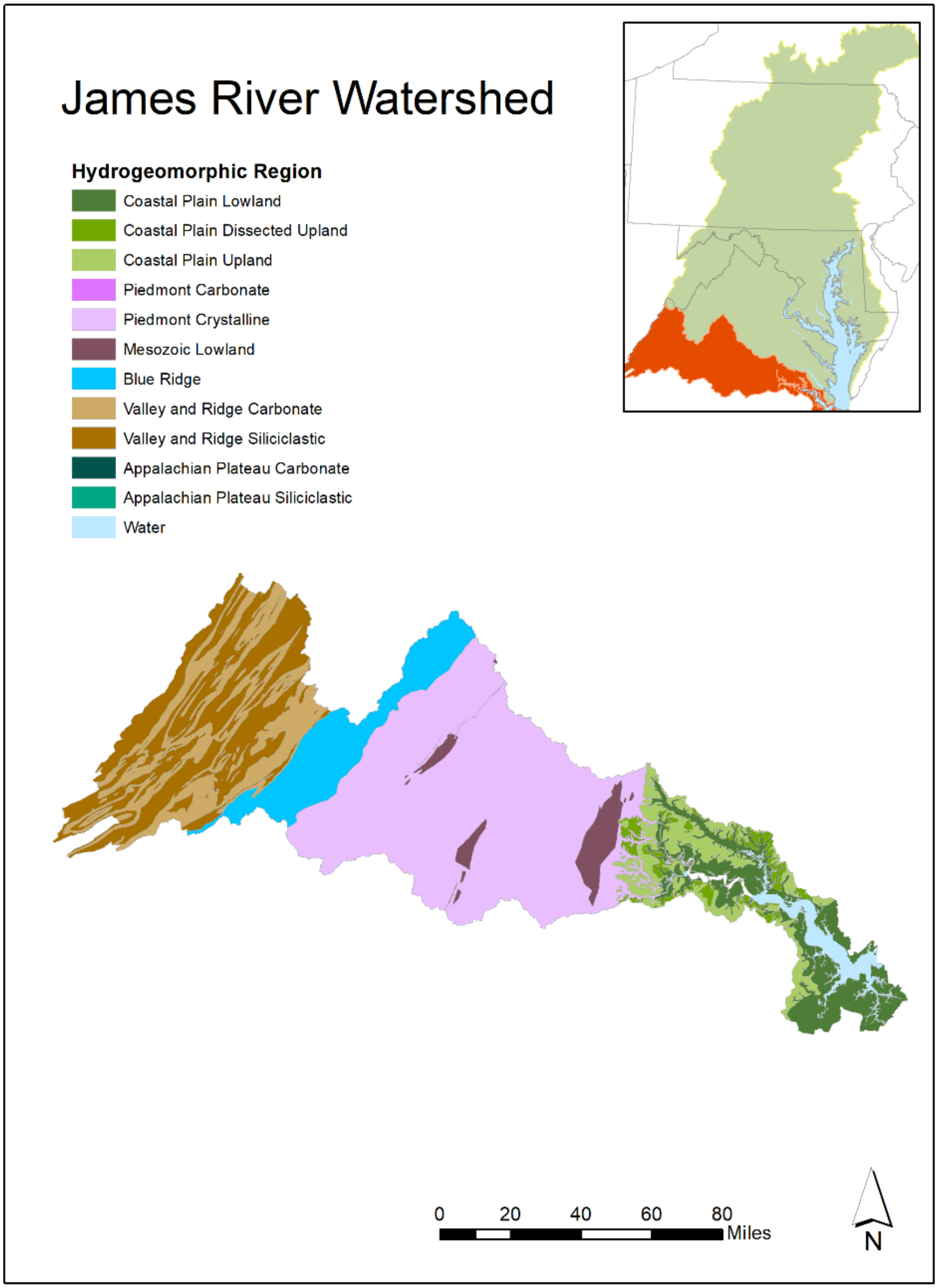

Figure 1. Distribution of physiography in the James River watershed. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

2.2 Land Use

Land use in the James watershed is dominated (75%) by natural areas. Urban and suburban land areas have increased by 300,688 acres since 1985, agricultural lands have decreased by 85,328 acres, and natural lands have decreased by 212,805 acres. Correspondingly, the proportion of urban land in this watershed has increased from 7% in 1985 to 12% in 2019 (Figure 2.).

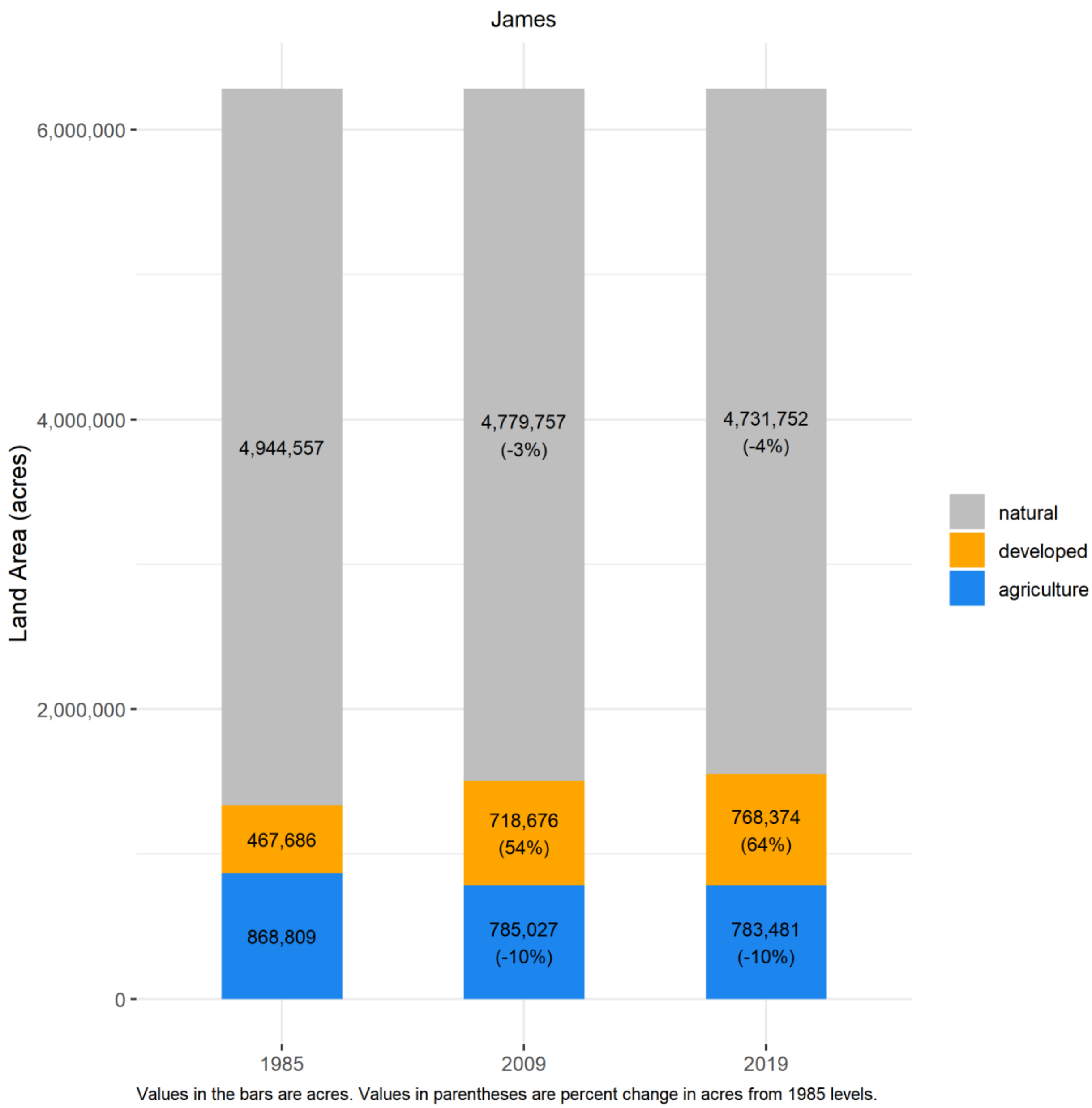

Figure 2. Distribution of land uses in the James watershed. Percentages are the percent change from 1985 for each source sector.

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.

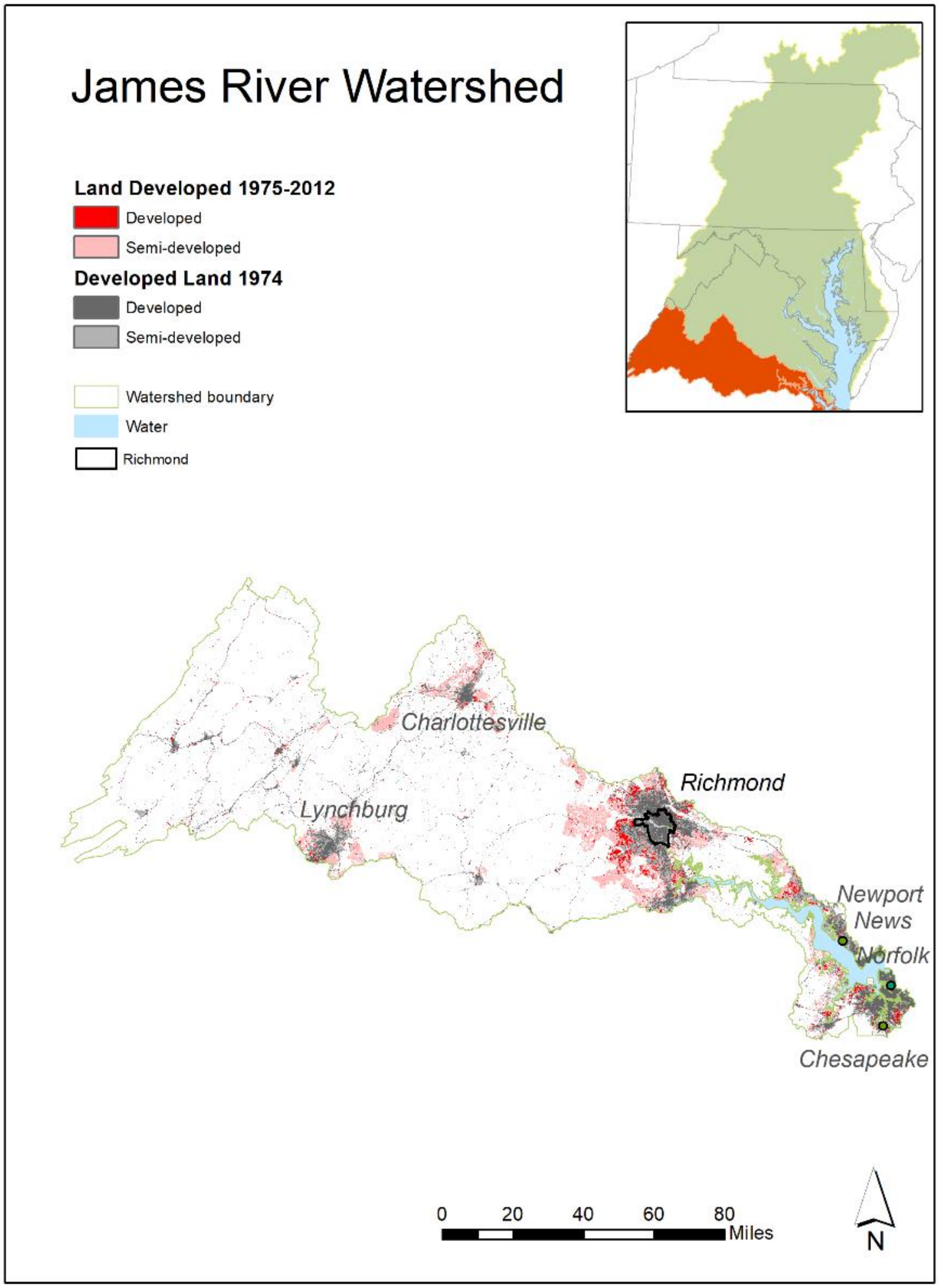

Figure 3. Distribution of developed land in the James River watershed. Derived from Falcone (2015). Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

2.3 Tidal Waters and Stations

For the purposes of water quality standards assessment and reporting, the tidal portion of the James Rivers is split into the following segments (U.S. Environmental Protection Agency, 2004): Tidal Fresh (JMSTF1, JMSTF2), Oligohaline (JMSOH), Mesohaline (JMSMH) and Polyhaline (JMSPH). The Tidal Fresh Appomattox River (APPTF) and Oligohaline Chickahominy (CHKOH) are also included in this group, along with segments in the Elizabeth River and its branches (EBEMH, ELIPH, SBEMH, and WBEMH) (Figure 4).

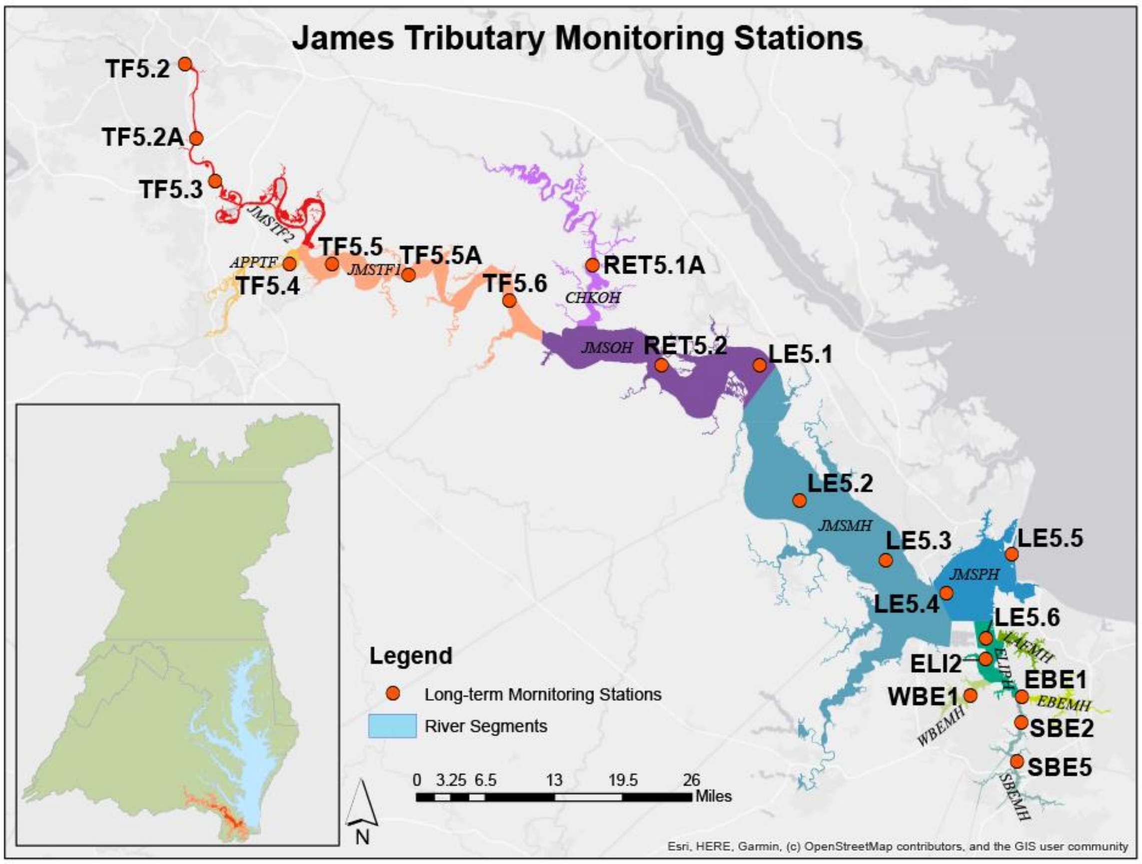

Figure 4. Map of tidal James River segments and long-term monitoring stations. Base map credit Esri, HERE, Garmin, (c) OpenStreetMap contributors, and the GIS user community, World Geodetic System 1984.

Long-term trends in water quality are analyzed by VADEQ at 20 stations stretching from the Appomattox River and Tidal Fresh James to the mouth of the James River, including the Elizabeth River system, flowing into Chesapeake Bay (Figure 4). Water quality data at these stations are also used to assess attainment of dissolved oxygen (DO) and chlorophyll *a* 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 extensive monitoring has been done in the James and Elizabeth Rivers, especially to analyze chlorophyll *a* spatial and temporal dynamics. Those observations are not included in subsequent trend graphics, with the focus here primarily on the long-term monitoring at fixed stations.

3. Tidal Water Quality Dissolved Oxygen Criteria Attainment

Multiple water quality standards were developed for the James River, Elizabeth River, and their 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), chlorophyll *a*, and water clarity/underwater bay grasses. Note that chlorophyll *a* criteria exist for the segments in the James and Elizabeth Rivers, and progress on that is tracked closely by VADEQ. For consistency with other tributary summaries, a record of the evaluation results indicating whether each of the segments have met or not met a subset of 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), the James and Appomattox River segments met the 30-day mean OW summer DO requirements, while the segments in the Chickahominy River, Elizabeth River and its branches, and Lafayette River did not meet the 30-day mean OW summer requirements. DW and DC criteria are only applicable in the Elizabeth polyhaline and Southern Branch (for DW), where the requirements were met in the most recent period (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	APPTF	JMSTF2	JMSTF1	CHKOH	JMSOH	JMSMH	JMSPH	ELIPH	WBEMH	SBEMH	EBEMH	LAFMH
1985-1987									ND	ND	ND	ND
1986-1988									ND	ND	ND	ND
1987-1989												
1988-1990												
1989-1991												
1990-1992												
1991-1993												ND
1992-1994												ND
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2013-2015												
2014-2016												
2015-2017												
2016-2018												

Table 3. Deep Water summer DO (30-day mean) and Deep Channel (Instantaneous) DO criteria evaluation results. Green indicates that the criterion was met. White indicates that the criterion was not met. (Note: this table is shorter than Table 1 due to lack of computations for the earlier years.)

time period	Deep Water		Deep Channel
	ELIPH	SBEMH	ELIPH
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 DW summer 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 bottom depths at each of these stations is different due to varying bathymetry, but the bottom DO trends at these stations are expected to represent water in the DW designated use. Notably the DO concentrations both in the surface and the bottom are improving at most of the Elizabeth River region stations, suggesting progress even in segments not meeting the OW summer 30-day mean DO criterion.

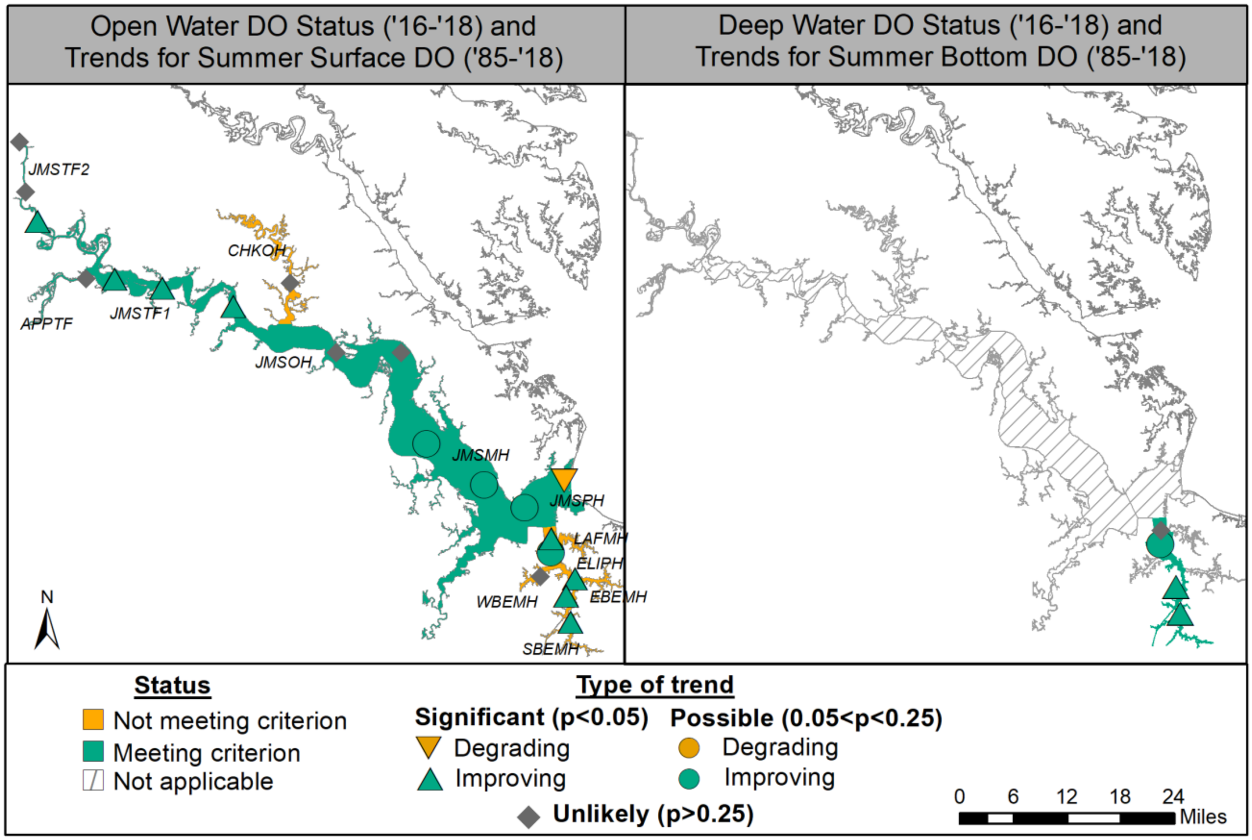


Figure 5. Pass-fail DO criterion status for 30-day OW summer DO and DC summer DO designated uses in MD mainstem segments along with long-term trends in DO concentrations. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

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 20 tidal stations labeled in Figure 4. For more details on the GAM implementation that is applied each year by VA Department of Environmental Quality and Old Dominion University for these stations in collaboration with the Chesapeake Bay Program and Maryland 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 James, Appomattox, Chickahominy or Elizabeth 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 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) concentrations have improved at all James River and tributary stations over the long-term, using both trends on concentration data alone and adjusting for flow (Figure 6). Over the short-term, a few stations switched to possible trends or no trend, but mostly the significant improving trends continue.

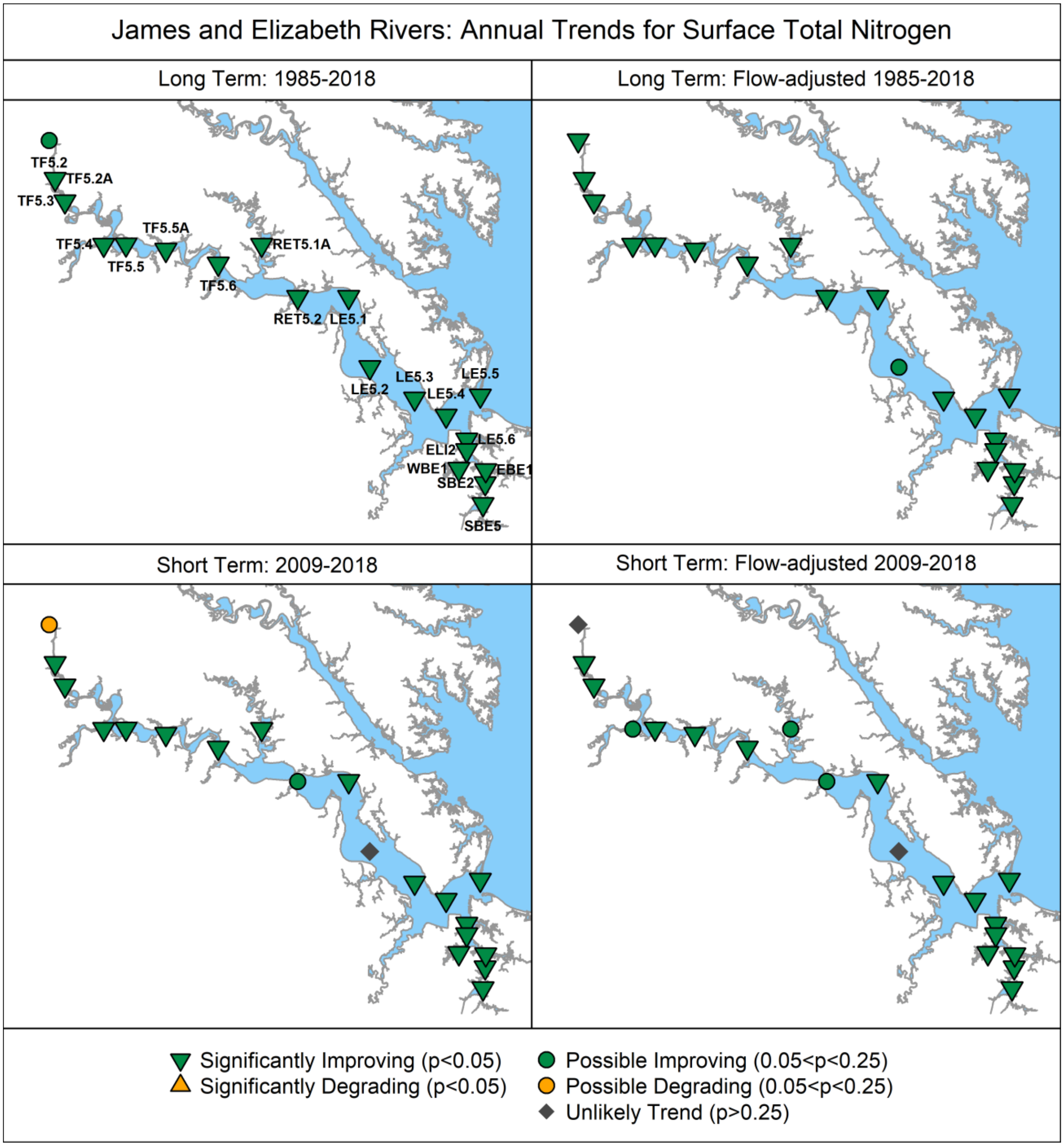


Figure 6. Surface TN trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

Decreasing trends are evident in both the data and the non-flow adjusted mean annual GAM estimates presented in Figure 7. The magnitude of the TN concentrations is highest at the lower tidal fresh James and mesohaline Elizabeth branch stations and decreases with distance from those stations. Those stations with the highest initial concentrations also have some of the strongest decreases in TN over time. For TN at most of the VA tributary stations, the records before 1994 contain too many values below the detection limits to accurately model the patterns, therefore, many of the time series start in 1994. All stations in the Elizabeth River began sampling in 1989 (Murphy *et al.*, 2019).

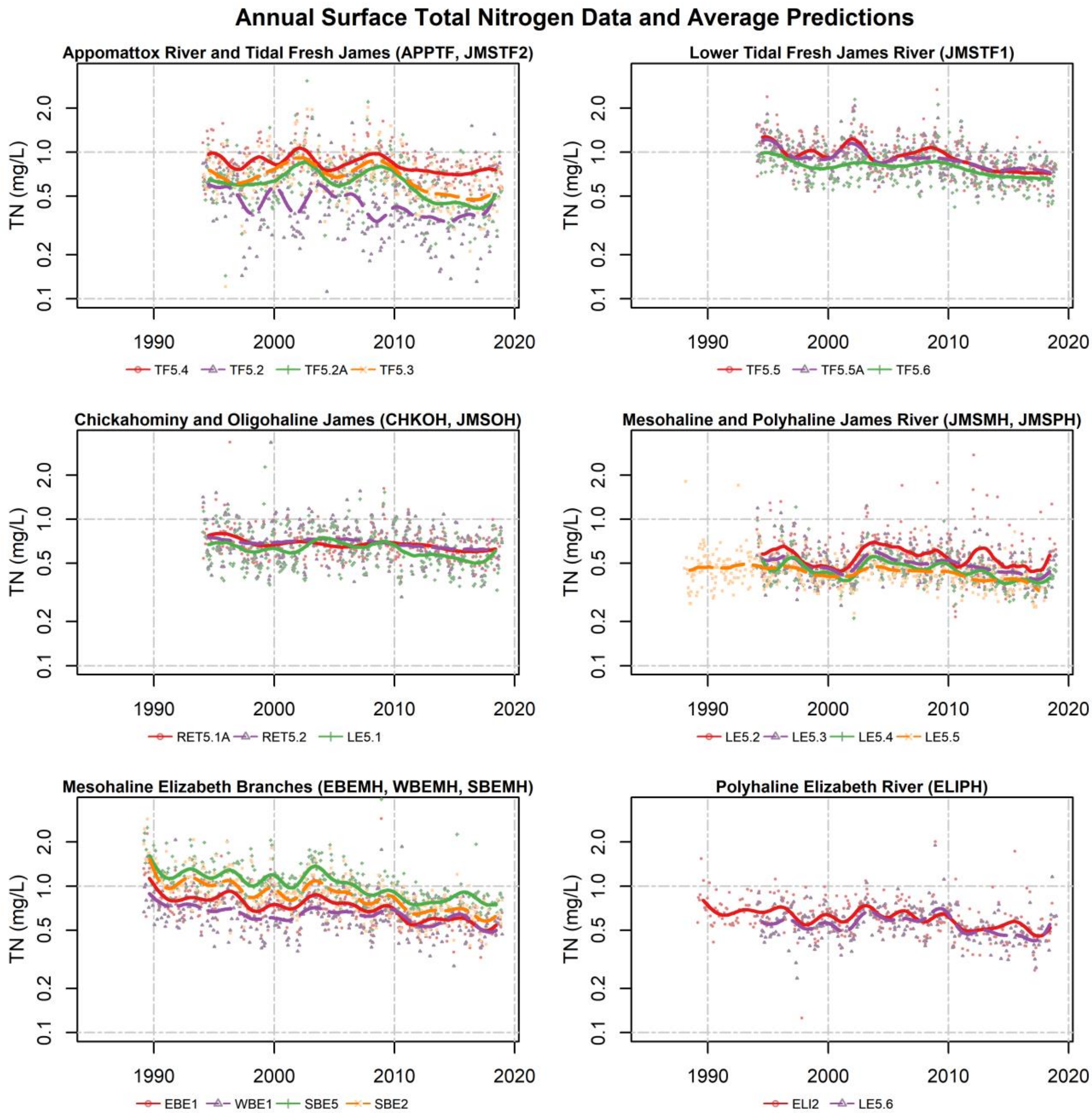


Figure 7. Surface TN data (dots) and average long-term pattern generated from non-flow-adjusted GAMs. Colored dots represent data corresponding to the monitoring station shown indicated in the legend; colored lines represent mean annual GAM estimates for the noted monitoring stations.

4.2 Surface Total Phosphorus

Annual total phosphorus (TP) is also improving at almost all James River and tributary stations over the long-term, using both trends on concentration data alone and adjusting for flow (Figure 8). Over the short-term, some stations switched to possible trends or no trend, but the improving trends continue at many stations, especially with flow-adjustment.

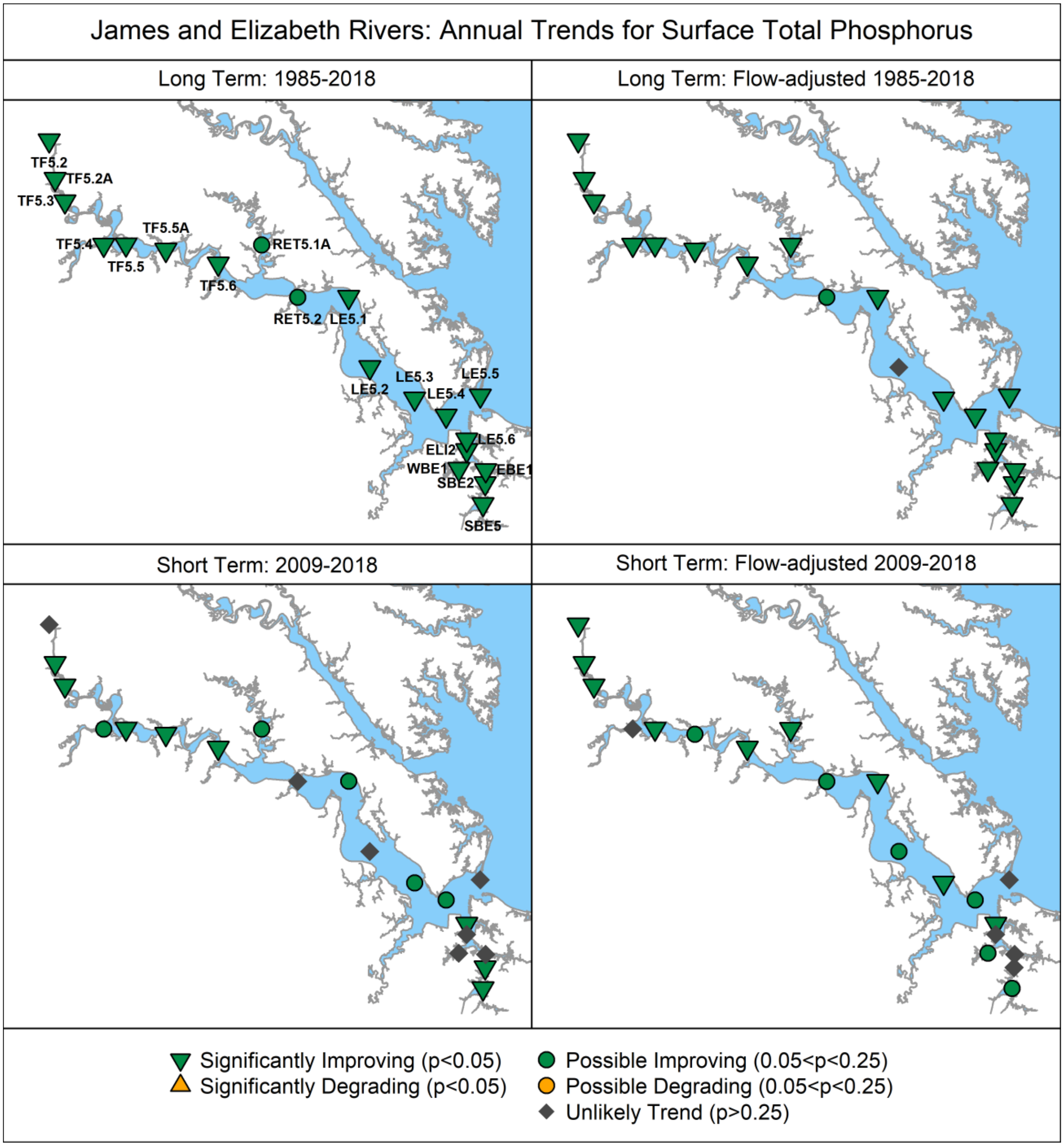


Figure 8. Surface TP trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

Long- and short-term decreases are very clear in the tidal fresh James station data and the non-flow adjusted mean annual GAM estimates presented in Figure 9 (top two panels). Long-term decreases are also apparent at the stations in saltier regions, but TP concentrations at most of those stations appear to level out in the recent decades.

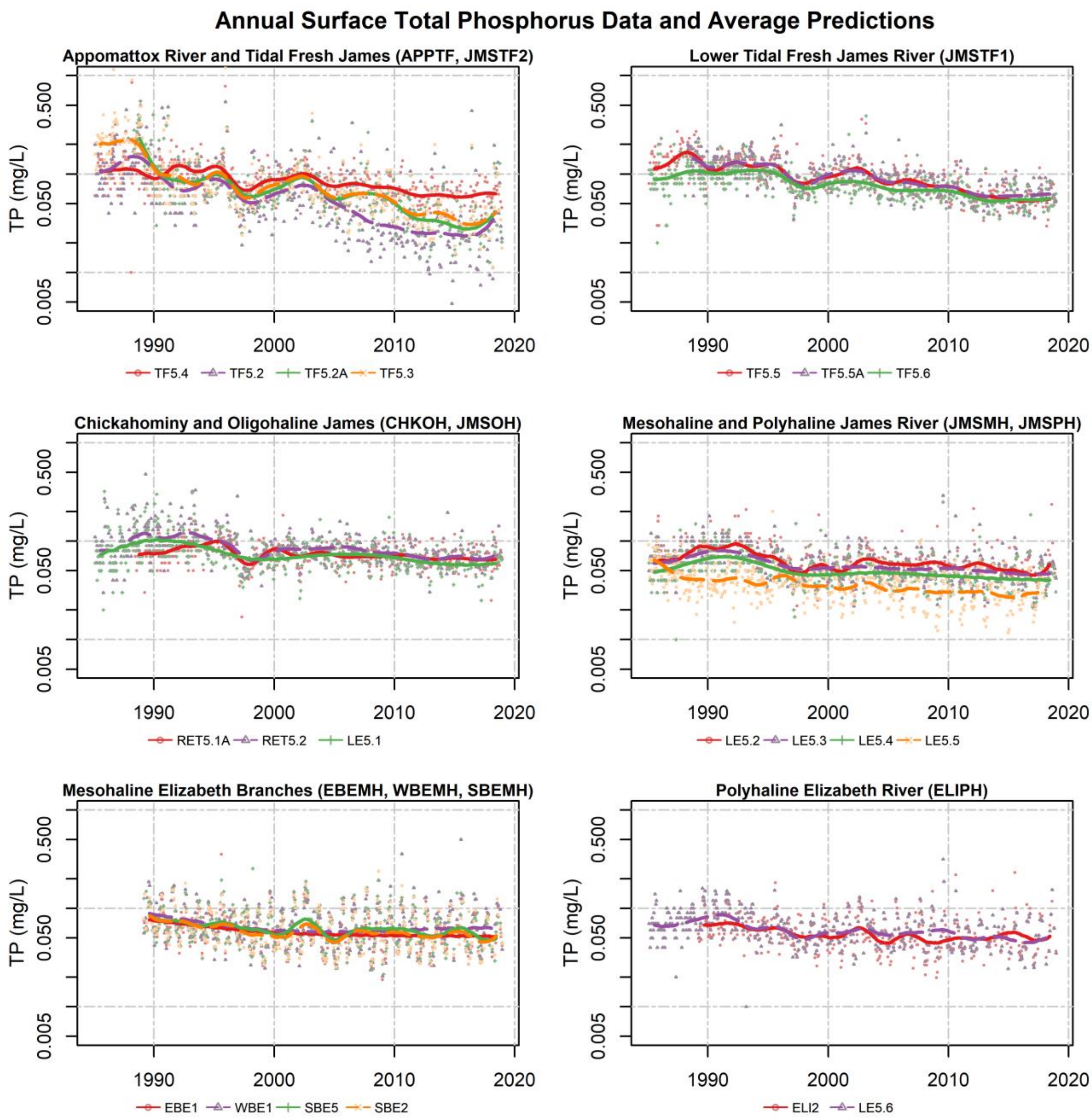


Figure 9. Surface TP data (dots) and average long-term pattern generated from non-flow adjusted GAMs. Colored dots represent data corresponding to the monitoring station shown indicated in the legend; colored lines represent mean annual GAM estimates for the noted monitoring stations.

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). Long-term spring trends are improving at about half of the James River and tributary stations, with mostly no trend at the remaining stations. Improvements are clustered in the tidal fresh and polyhaline/Elizabeth River regions. Over the short-term, the Elizabeth River improvements persist, but the lower tidal fresh and oligohaline stations show some possible degradation.

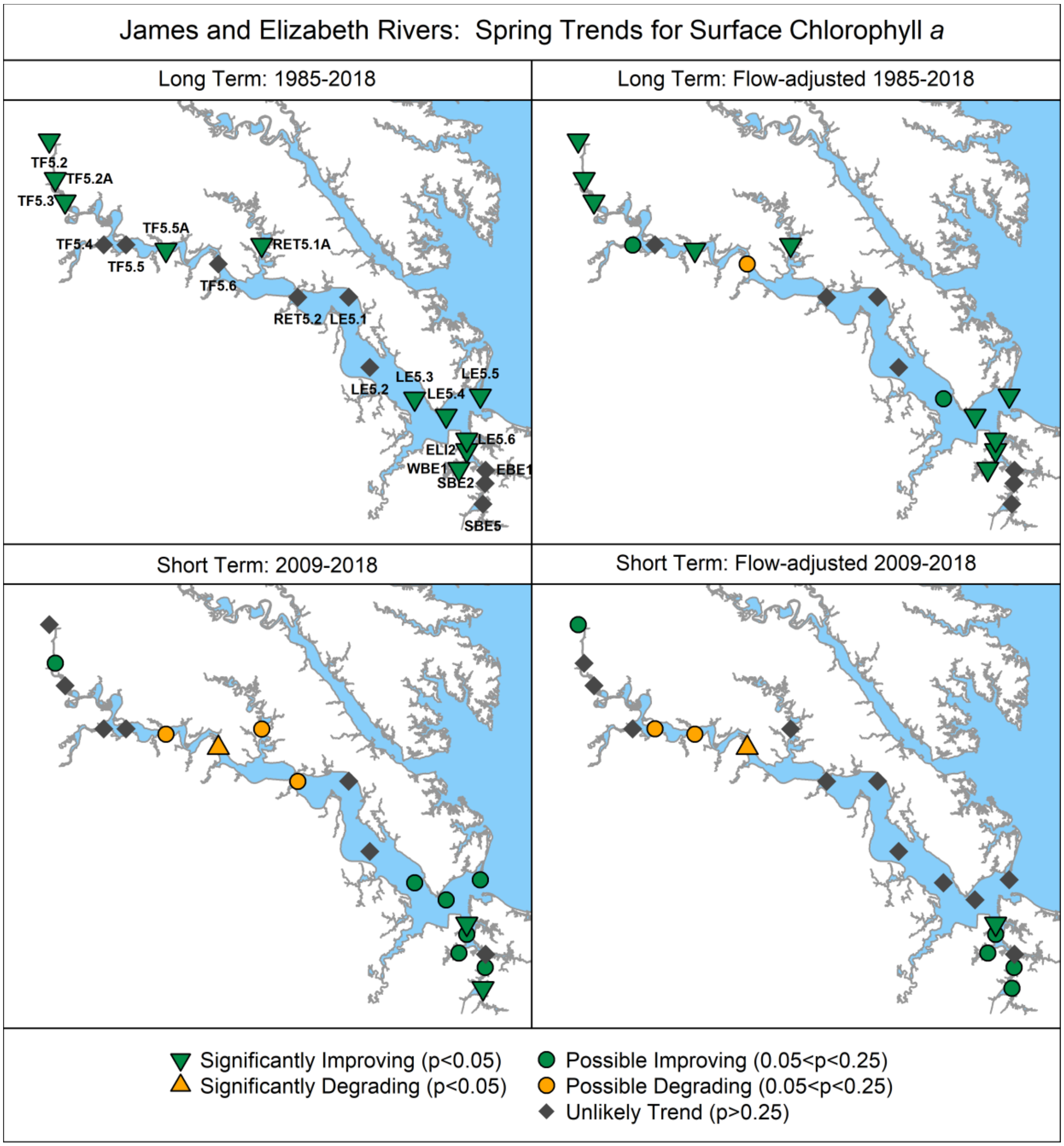


Figure 10. Surface spring (March-May) chlorophyll *a* trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

A high amount of variability exists in the long-term patterns of some of the chlorophyll *a* data sets and average spring GAM estimates (Figure 11). Notably, in all graph panels, it is clear that very high spring chlorophyll *a* concentrations that were measured in the 1980s and 1990s have not been observed in recent decades. In the last decade, leveling out of trends or slight increases are apparent in the James River stations while the Elizabeth River trends have decreased continuously through the record.

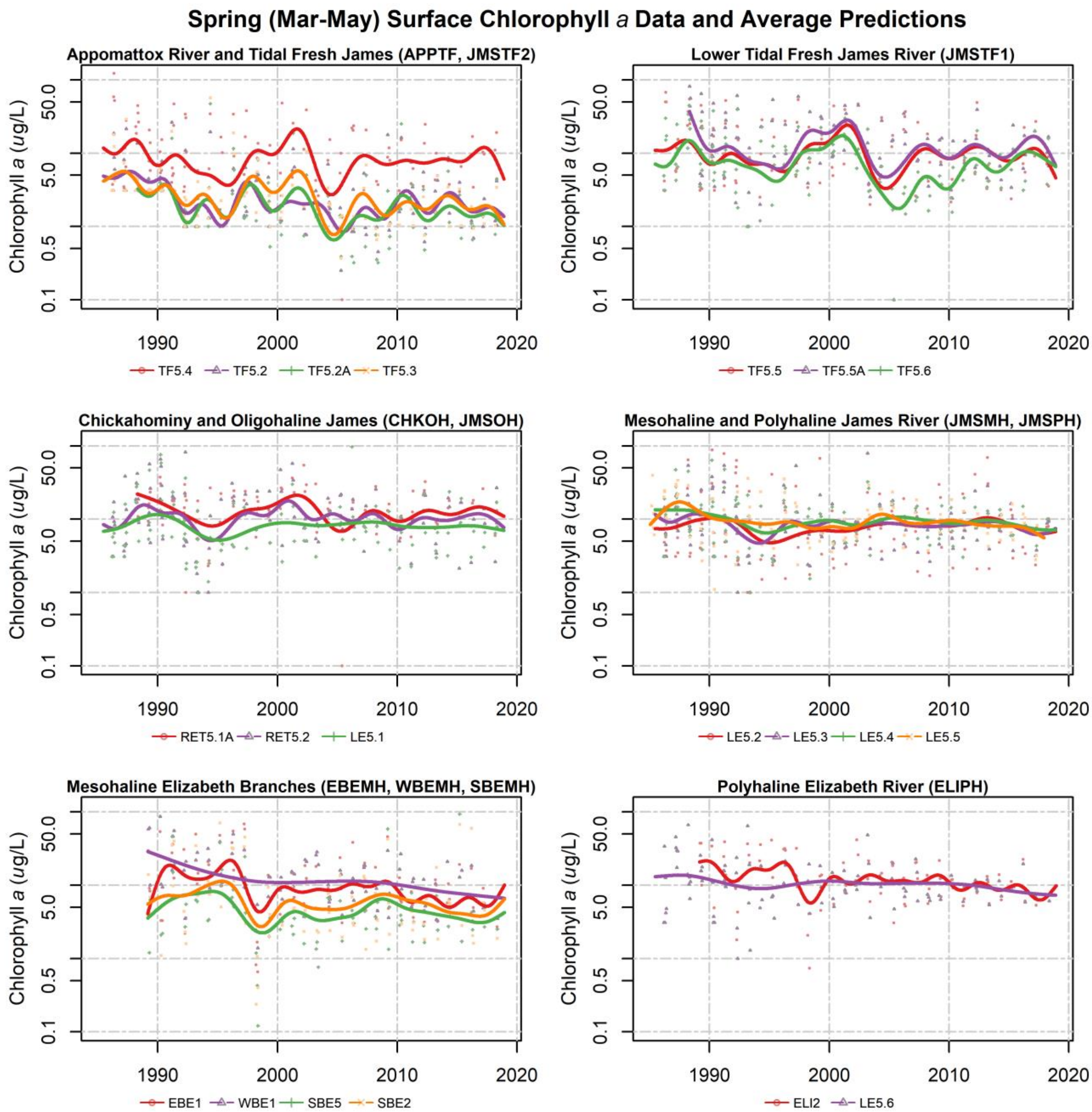


Figure 11. Surface spring chlorophyll *a* data (dots) and average long-term pattern generated from non-flow adjusted GAMs. Colored dots represent March-May data corresponding to the monitoring station

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)

Long-term trends in summer chlorophyll *a* (Figure 12) differ from spring long-term trends (Figure 11) at the mesohaline and polyhaline stations. In the summer, there are long-term degradations in those regions at many stations, while they were mostly improving in the spring. Other patterns are more similar between the seasons including long-term improvements at tidal fresh stations and the spatial distribution of short-term trends.

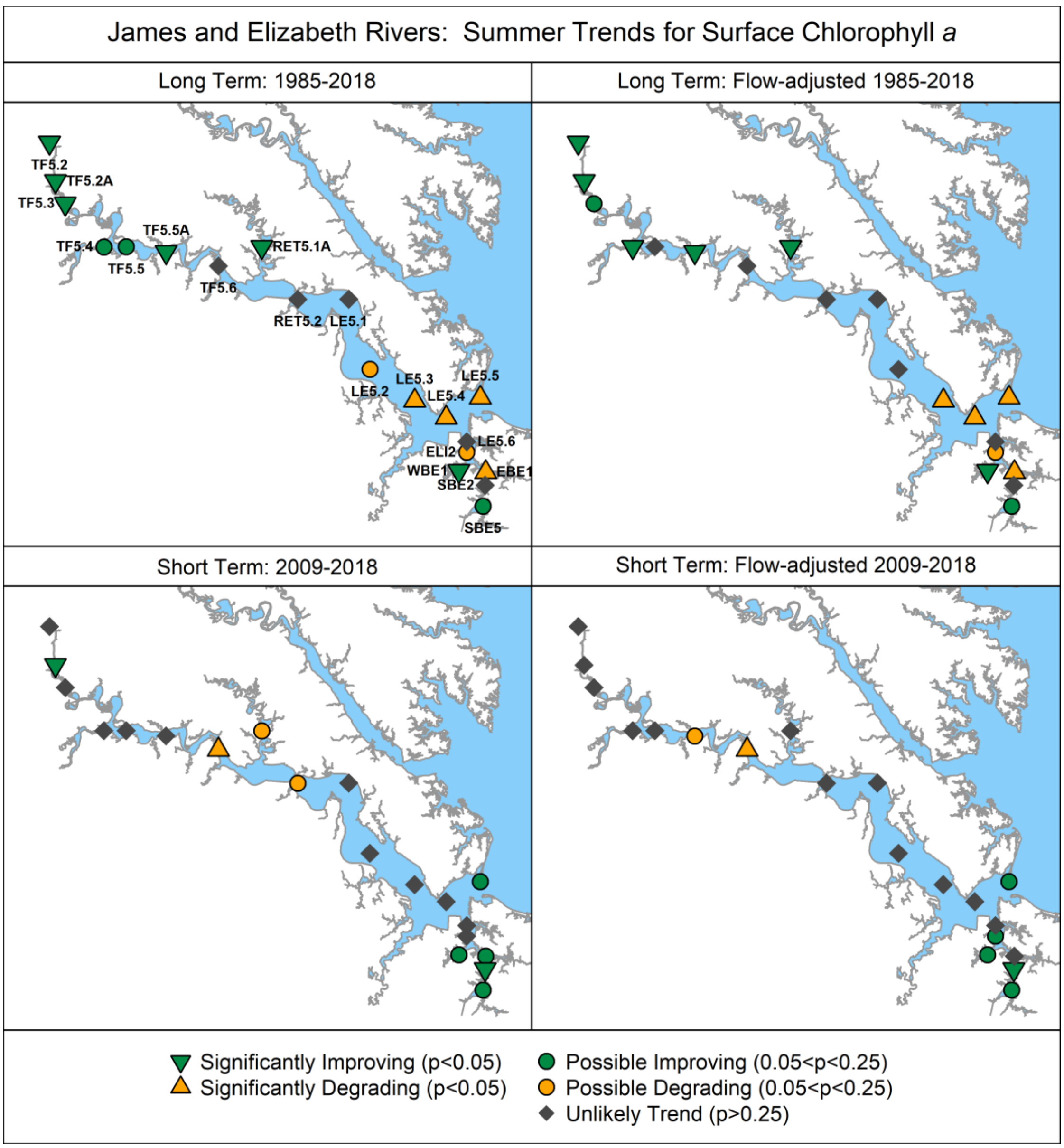


Figure 12. Surface summer (July-September) chlorophyll *a* trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

The increases in the mesohaline and polyhaline James and polyhaline Elizabeth stations over the long-term are clear in the data and average summer GAM estimates (Figure 13). These increases do level out over the last decade, but the difference in the bottom and middle right panels of Figure 13 and Figure 11 is notable and worth further investigation. Other summer chlorophyll *a* trends are variable with fresher stations' data and GAM estimates showing long-term decreases or no trend.

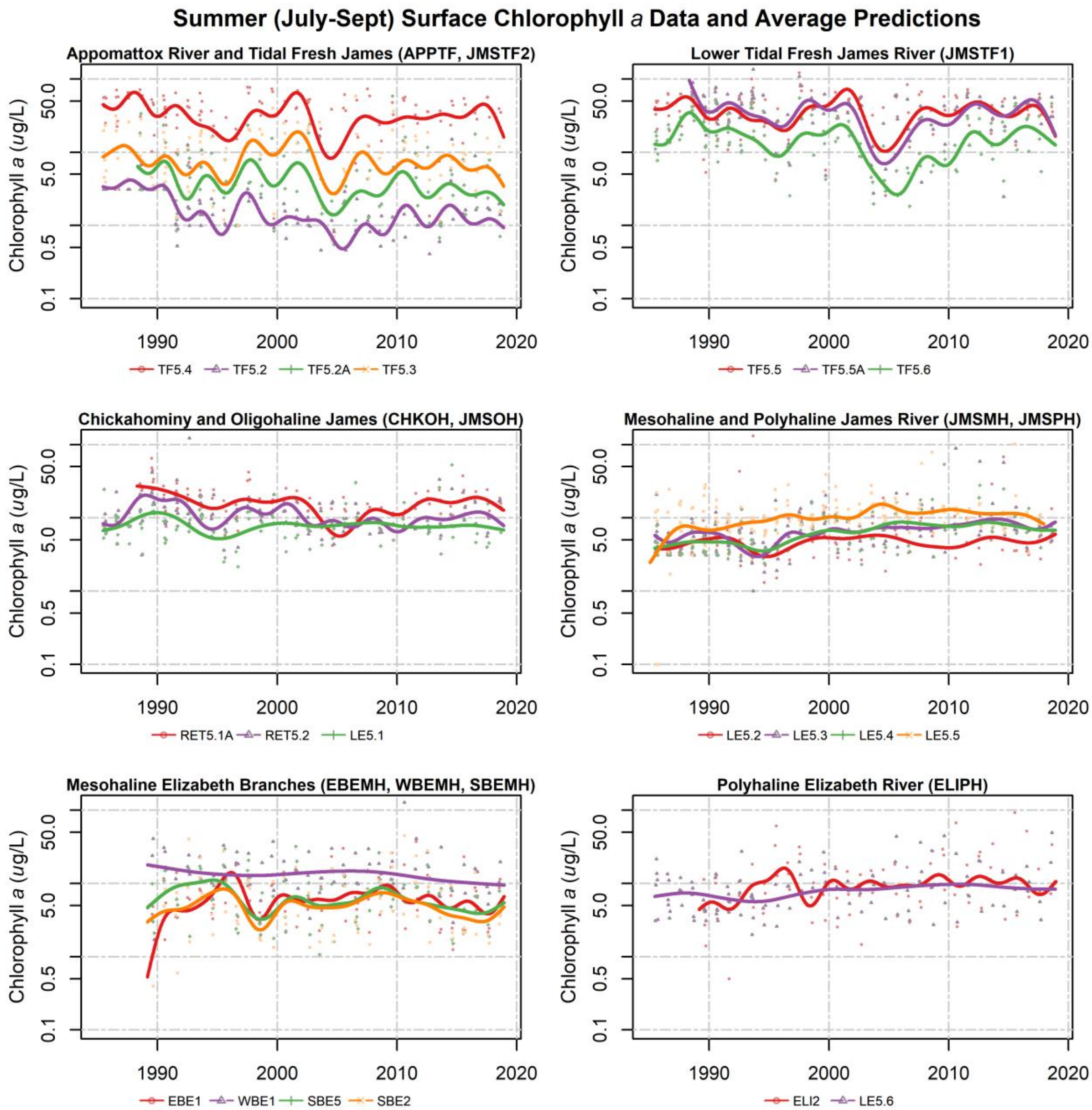


Figure 13. Surface summer chlorophyll *a* data (dots) and average long-term pattern generated from non-flow adjusted GAMs. Colored dots represent July-September data corresponding to the monitoring station shown indicated in the legend; colored lines represent mean summer GAM estimates for the noted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

4.5 Secchi Disk Depth

Trends in Secchi disk depth, a measure of visibility through the water column, are degrading at many of the tidal fresh stations and the Chickahominy River station (RET5.1A) (Figure 14). On the other hand, the Secchi trends are mostly improving at the mesohaline and polyhaline stations (Figure 14). These patterns are consistent over the long- and short-term, both for non-adjusted and flow-adjusted results.

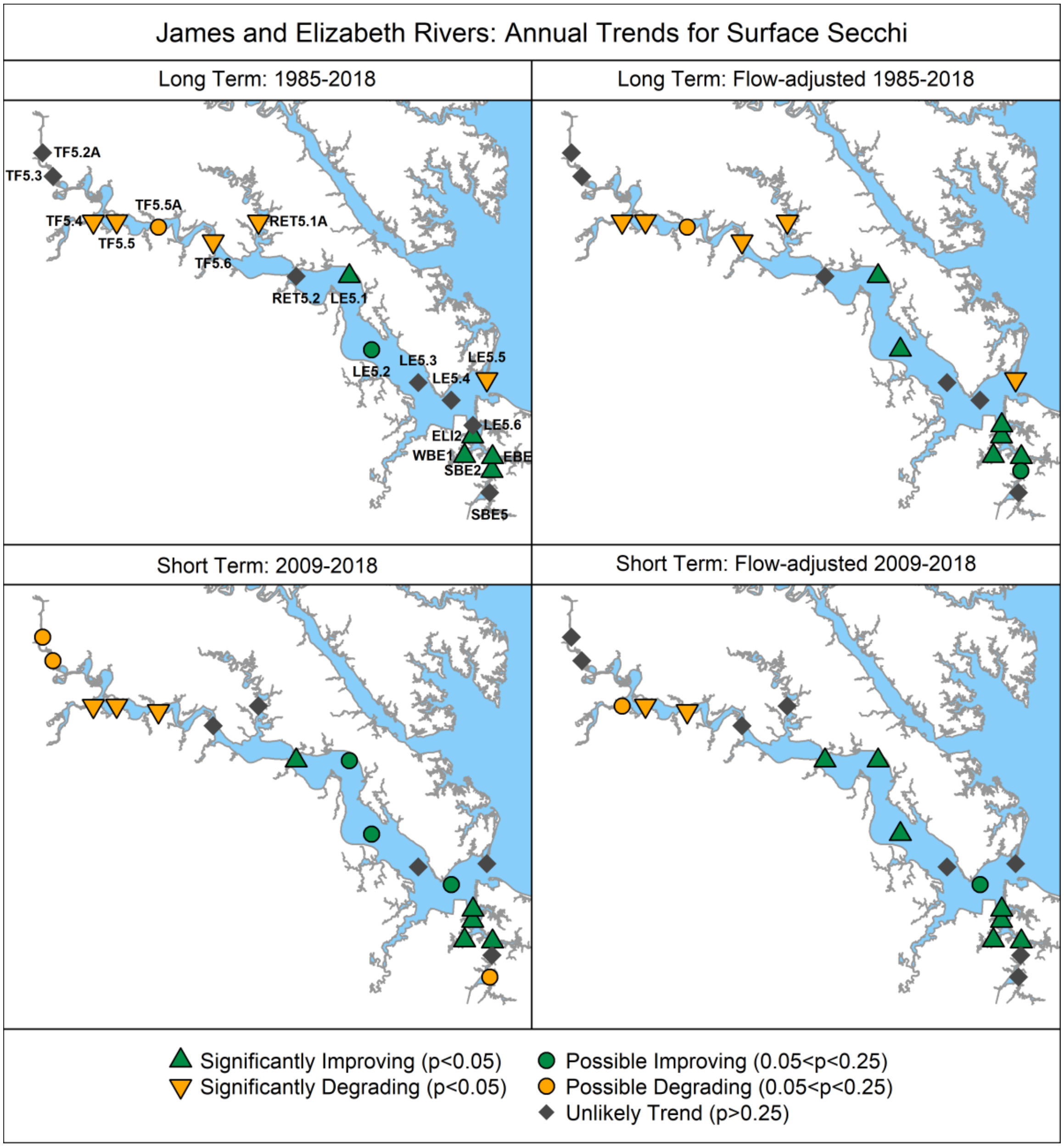


Figure 14. Annual Secchi depth trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

Secchi depths are almost all less than 1 meter at the stations with degrading trends, which are in the Appomattox (TF 5.4), lower tidal fresh (top right panel) and Chickahominy (RET5.1A) (Figure 15). The shallow depths make it hard to decipher the decreasing trends at these stations in the data values and mean annual GAM estimates. The lower James stations have slightly deeper Secchi depth values and are mostly increasing over time, especially in the Elizabeth River (bottom two panels, Figure 15).

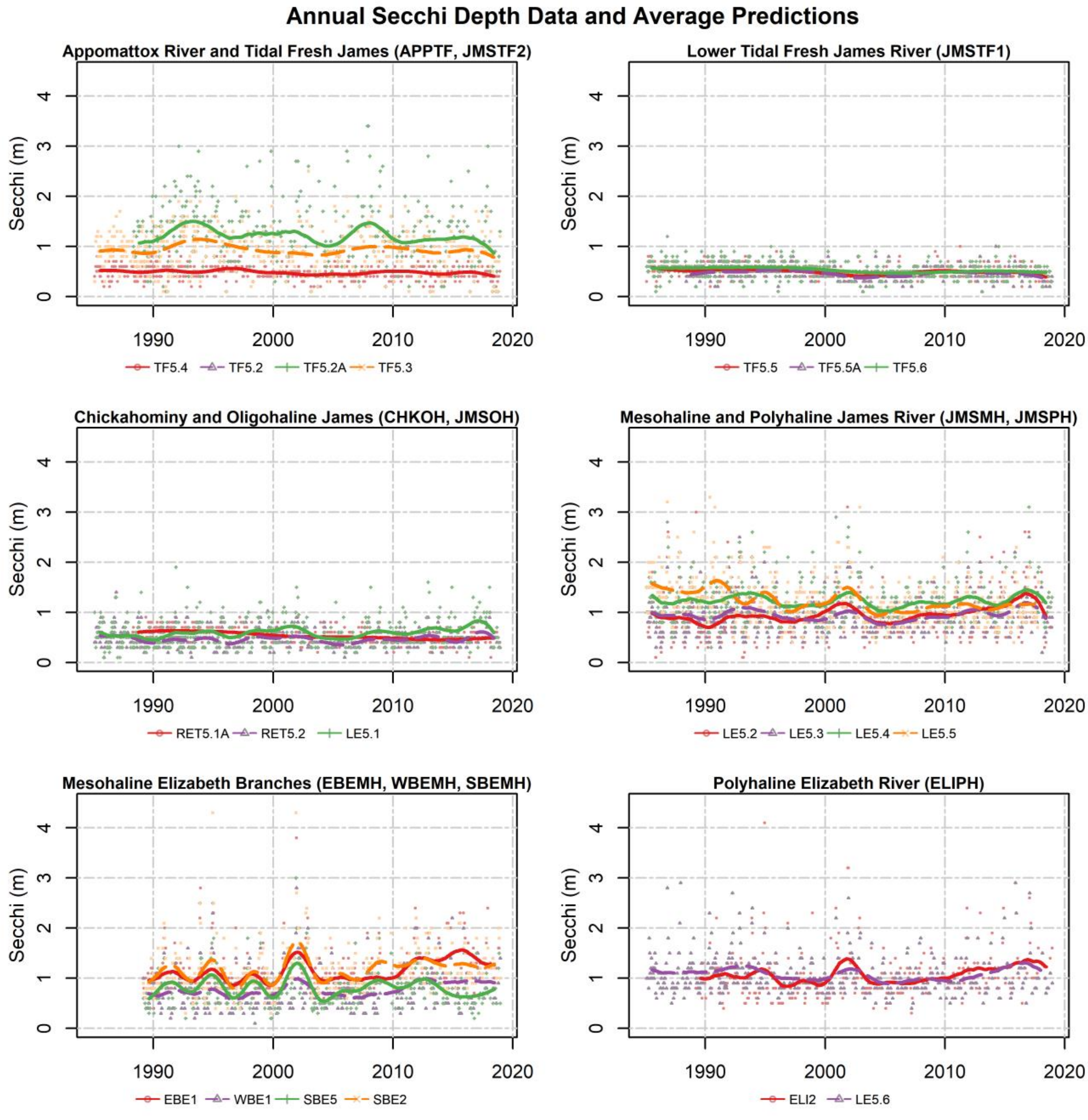


Figure 15. Annual Secchi depth data (dots) and average long-term pattern generated from non-flow adjusted GAMs. Colored dots represent data corresponding to the monitoring station shown indicated in the legend; colored lines represent mean annual GAM estimates for the noted monitoring stations.

4.6 Summer Bottom Dissolved Oxygen (June-September)

James River and tributary summer bottom oxygen trends are fairly mixed, with a cluster of improvement in the Elizabeth River over the long-term (Figure 16). TF5.6 also stands out as improving across all panels.

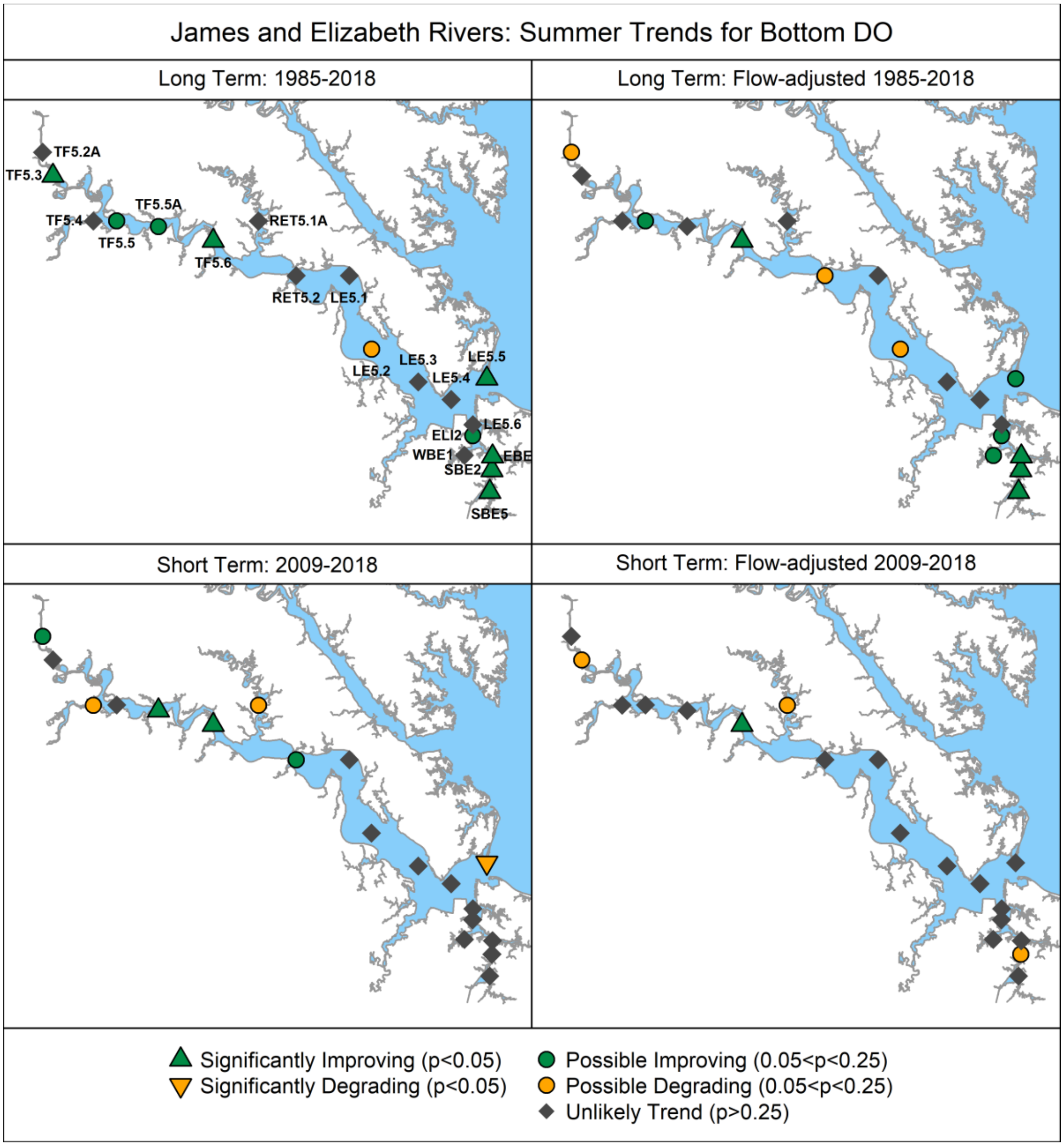


Figure 16. Summer (June-September) bottom DO trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

Summer bottom DO concentrations do not generally get as low in the James River as some other regions in Chesapeake Bay. Observations and mean summer GAM estimates are mostly above the 5 mg/L Open Water 30-day mean criterion that apply to the James segments (Figure 17). Some of the Elizabeth River segments also have the deep water designated use, and one is deep channel. Those stations show an increase over time in both observed DO and mean summer GAM estimates (Figure 17, bottom two panels)

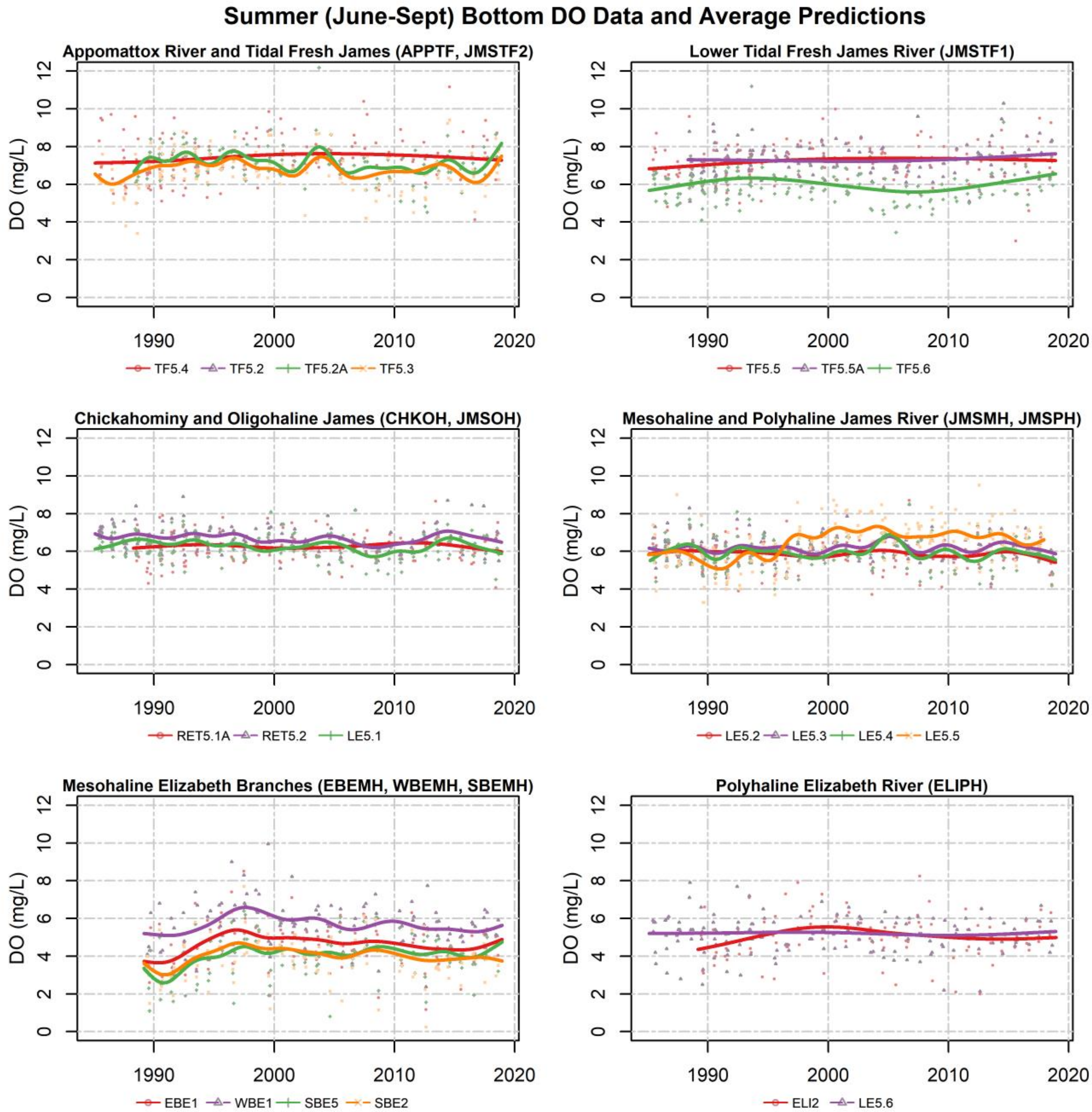


Figure 17. Summer (June-September) bottom DO data (dots) and average summer long-term pattern generated from non-flow adjusted GAMs. Colored dots represent June-September data corresponding to the monitoring station shown indicated in the legend; colored lines represent mean summer GAM estimates for the noted monitoring stations.

5. Factors Affecting Trends

5.1 Watershed Factors

5.1.1 Effects of Physical Setting

The geology of the James River watershed and its associated land use affects the quantity and transmissivity of nitrogen, phosphorus, and sediment delivered to non-tidal and tidal streams (Figure 18) (Brakebill *et al.*, 2010; Ator *et al.*, 2011; Ator *et al.*, 2019; Ator *et al.*, 2020; Noe *et al.*, 2020).

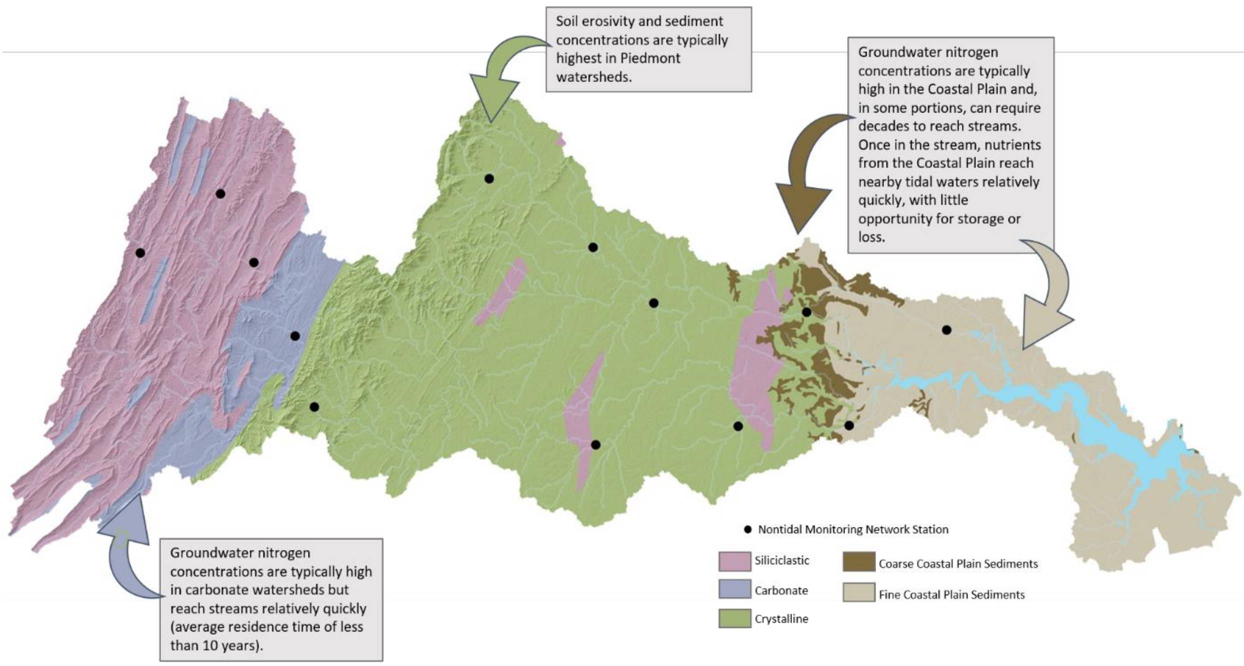


Figure 18. Effects of watershed hydrogeomorphology on nutrient transport to freshwater streams and tidal waters. Base map modified from King *et al.* (1974) and Ator *et al.* (2005), North American Datum 1983.

<u>Nitrogen</u>

Groundwater is an important delivery pathway of nitrogen, as nitrate, to most streams in the Chesapeake Bay watershed (Ator and Denver, 2012; Lizarraga, 1997). Concentrations of groundwater nitrogen, as nitrate, are typically highest in headwater portions of the James River watershed where carbonate rocks underlay some of the Valley and Ridge physiographic province (Greene and others, 2005; Terziotti and others, 2017). The geology of these areas provides suitable land for agriculture, but little potential for denitrification (Böhlke and Denver, 1995; Lizarraga, 1997; Miller and others, 2007; Sanford and Pope, 2013), so nitrogen that isn't removed by plants or exported in agricultural products can move relatively efficiently to groundwater. Carbonate rocks only compose a small area of the James River watershed, with most streams underlain by Piedmont crystalline rocks or Coastal Plain sediments.

The typical residence time of groundwater delivered to streams in the Chesapeake Bay watershed is about 10 years, but ages vary from less than one year to greater than 50 years based on bedrock structure, groundwater flow paths, and aquifer depths (Lindsey and others, 2003). In general, groundwater ages tend to be relatively short (0-10 years) in carbonate settings, where permeable soils and solution-enlarged fractures enhance groundwater connectivity (Lindsey and others, 2003). Groundwater represents about 50% of streamflow in most Chesapeake Bay streams, with the other half composed of soil moisture and runoff, which have residence times of months to days (Phillips, 2007).Groundwater is the primary delivery pathway of nitrogen to most streams in the Chesapeake Bay watershed (Lizarraga, 1997; Bachman *et al.*, 1998; Ator and Denver, 2012). Concentrations of groundwater nitrogen, as nitrate, are typically highest in headwater portions of the James River watershed where carbonate rocks underlay some of the Valley and Ridge physiographic province (Greene *et al.*, 2005; Terziotti *et al.*, 2017). The geology of these areas provides suitable land for agriculture, but little potential for denitrification (Böhlke and Denver, 1995; Lizarraga, 1997; Miller *et al.*, 1997; Sanford and Pope, 2013), so nitrogen that isn't removed by plants or exported in agricultural products can move relatively efficiently to groundwater (Tesoriero *et al.*, 2015). Carbonate rocks only compose a small area of the James River watershed, with most streams underlain by Piedmont crystalline rocks or Coastal Plain sediments. The typical residence time of groundwater delivered to streams in the Chesapeake Bay watershed is about 10 years, but ages vary from less than one year to greater than 50 years based on bedrock structure, groundwater flow paths, and aquifer depths (Lindsey *et al.*, 2003). In general, groundwater ages tend to be relatively short (0-10 years) in carbonate settings, where permeable soils and solution-enlarged fractures enhance groundwater connectivity (Lindsey *et al.*, 2003). Groundwater represents about 50% of streamflow in most Chesapeake Bay streams, with the other half composed of soil moisture and runoff, which have residence times of months to days (Phillips, 2007).

<u>Phosphorus</u>

Phosphorus binds to soil particles and most phosphorus delivered to the Bay is attached to sediment (Zhang *et al.*, 2015); however, once fully phosphorus saturated, soils will not retain new applications and export of dissolved phosphorus to streams, from shallow soils and groundwater, will increase (Staver and Brinsfield, 2001). Phosphorus sorption capacity varies based on soil particle chemical composition and physical structure with clays typically having the greatest number of sorption sites and highest average phosphorus concentrations (Sharpley, 1980). The highest soil phosphorus concentrations occur in the headwaters of the James River watershed where inputs of manure and fertilizer applied to agricultural fields exceed crop needs. 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).

<u>Sediment</u>

The delivery of sediment from upland soil erosion, streambank erosion, and tributary loading varies throughout the James River watershed, but in-stream concentrations are typically highest in streams above the fall line that drain Piedmont geology (Brakebill *et al.*, 2010). The erosivity of Piedmont soils results from its unique topography and from the prevalence of agricultural and urban land uses in these areas (Trimble, 1975; Gellis *et al.*, 2005; Brakebill *et al.*, 2010). Factors affecting streambank erosion are highly variable throughout this watershed and include 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), stream valley geomorphology (Hopkins *et al.*, 2018), and developed land uses (Brakebill *et al.*, 2010).

<u>Delivery to tidal waters from the non-tidal watershed</u>

The delivery of nitrogen, phosphorus, and sediment in non-tidal streams to tidal waters in the James River watershed shore varies based on physical and chemical factors that affect in-stream retention, loss, or storage. In general, nutrient and sediment loads in tidal waters are most strongly influenced by conditions in proximal non-tidal streams that have less opportunity for denitrification and floodplain trapping of sediment associated phosphorus. In-stream denitrification rates vary spatially with soil moisture and temperature (Pilegaard, 2013) and are typically higher portions of in the James River watershed than in more northern Bay regions because of a warmer climate. More than half of the nitrogen in the uppermost reaches of the James River are removed via denitrification before reaching tidal waters (Ator *et al.*, 2011). 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). The average age of sediment stored in-channel is typically assumed to be less than a year (Gellis *et al.*, 2017), but delivery to tidal waters can be exponentially longer as sediment moves in and out of different storage zones during downstream transport.

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 load to the tidal James were primarily from the below-RIM areas, whereas phosphorus, and suspended sediment loads were primarily from the RIM areas (Figure 19). Over the period of 1985-2018, 0.51, 0.067, and 42 million tons of nitrogen, phosphorus, and suspended sediment loads were exported through the James River watershed, with 34%, 57%, and 61% of those loads from the RIM areas, respectively.

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

<u>Nitrogen</u>

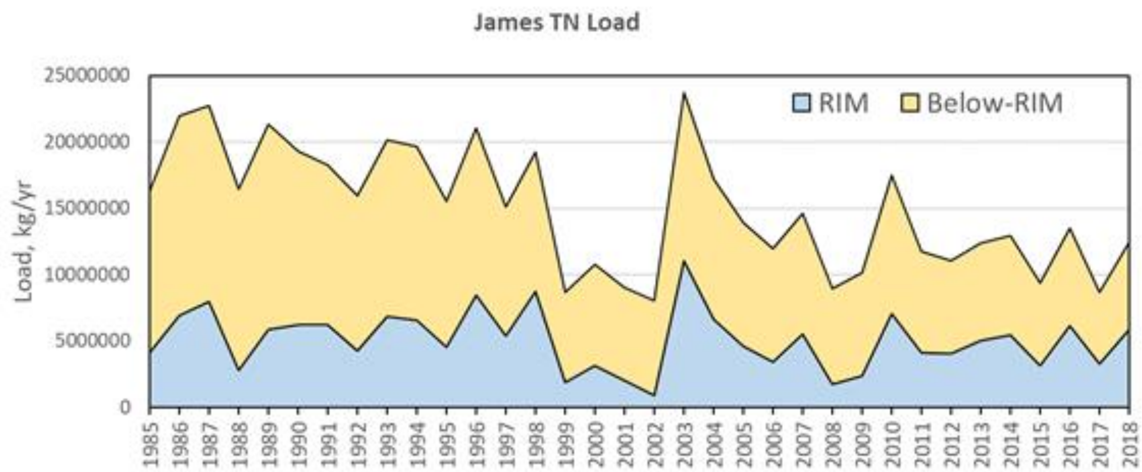
Estimated TN loads showed an overall decline of 299 ton/yr in the period between 1985 and 2018, which is statistically significant ($p < 0.01$). This reduction is largely driven by reductions in below-RIM loads (-250 ton/yr, $p < 0.01$), with a smaller contribution by the RIM loads (-42 ton/yr, $p = 0.28$). The below-RIM decline is largely driven by below-RIM point sources (-233 ton/yr, $p < 0.01$). This significant reduction in below-RIM point sources is a result of substantial efforts to reduce nitrogen loads from major wastewater treatment facilities by implementing biological nutrient removal (Lyerly *et al.*, 2014).

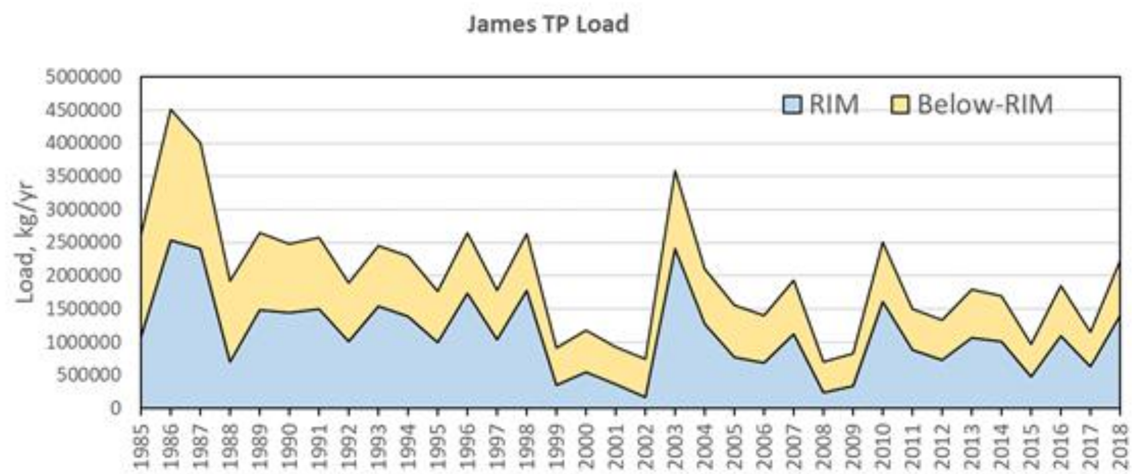
<u>Phosphorus</u>

Estimated TP loads showed an overall decline of 40 ton/yr in the period between 1985 and 2018, which is statistically significant ($p < 0.01$). This reduction reflects a combination of reductions in RIM loads (-20 ton/yr, $p < 0.05$) and below-RIM loads (-20 ton/yr, $p < 0.01$). The below-RIM decline is entirely driven by below-RIM point sources (-20 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). By contrast, the below-RIM nonpoint sources showed a long-term increase in this period (4.4 ton/yr, $p = 0.10$).

Sediment

Estimated suspended sediment (SS) loads showed an overall decline of 11,355 ton/yr in the period between 1985 and 2018, although it is not statistically significant ($p = 0.31$). Both the RIM and below-RIM loads showed long-term declines, but both are not statistically significant. Like TP and TN, the below-RIM point source load of SS showed a statistically significant decline in this period (-196 ton/yr; $p < 0.01$).





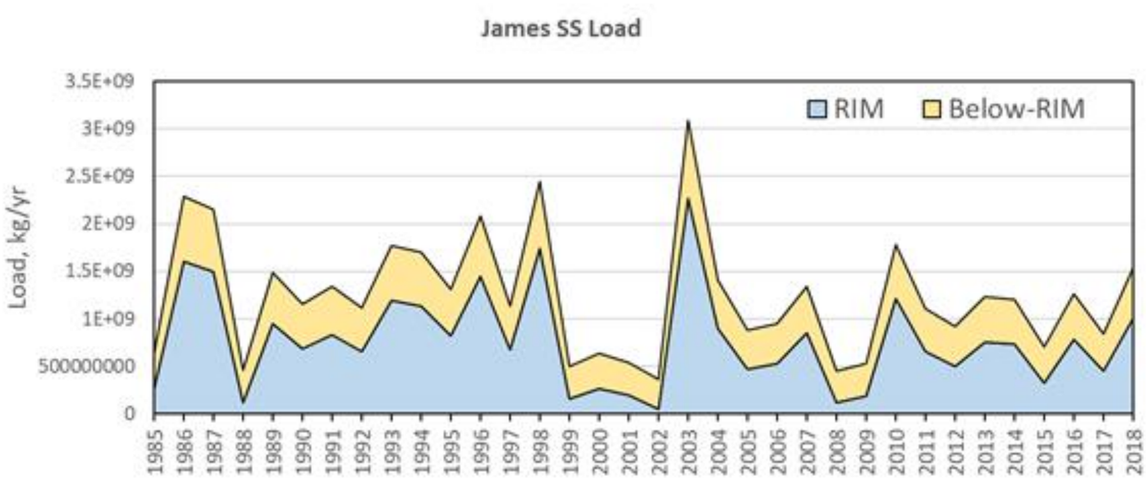


Figure 19. Estimated total loads of nitrogen (TN), phosphorus (TP), and suspended sediment (SS) from the RIM and below-RIM areas of the James River. RIM refers to the USGS River Input Monitoring site located just above the head of tide of this tributary, which includes upstream point source loads. Below-RIM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

Table 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 James River watershed.

Variable	Trend, metric ton/yr	Trend p-value
TN		
Total watershed	-299	< 0.01
RIM watershed [1]	-42	0.28
Below-RIM watershed [2]	-250	< 0.01
Below-RIM point source	-233	< 0.01
Below-RIM nonpoint source [3]	-12	0.59
Below-RIM tidal deposition	-2.0	0.12
TP		
Total watershed	-40	< 0.01
RIM watershed	-20	< 0.05
Below-RIM watershed	-20	< 0.01
Below-RIM point source	-20	< 0.01
Below-RIM nonpoint source	4.4	0.10
SS		
Total watershed	-11,355	0.31
RIM watershed	-8,709	0.35
Below-RIM watershed	-2,417	0.25
Below-RIM point source	-196	< 0.01
Below-RIM nonpoint source	-2275	0.30

[1] Loads for the RIM watershed were estimated loads at the USGS RIM station 02035000 (James River at Cartersville, Va.; 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 James River by -43%, -70%, and -4%, 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, wastewater and agriculture were the two largest sources of nitrogen loads. By 2019, wastewater and agriculture remained the two largest sources of nitrogen loads. Overall, decreasing nitrogen loads from agriculture (-23%), natural (-9%), stream bed and bank (-4%), and wastewater (-70%) sources were partially counteracted by increases from developed (54%) and septic (57%) sources.

The two largest sources of phosphorus loads as of 2019 were the wastewater and stream bed and bank sectors. Overall, expected declines from agriculture (-44%), natural (-7%), stream bed and bank (-25%), and wastewater (-89%) sources were partially counteracted by increases from developed (65%) sources.

For sediment, the largest sources are stream bed and bank and shoreline areas: these two sources changed by -3% and -1%, respectively between 1985 and 2019. Sediment loads from the agriculture sector changed by -45%, whereas sediment load from developed areas changed by 29%.

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.

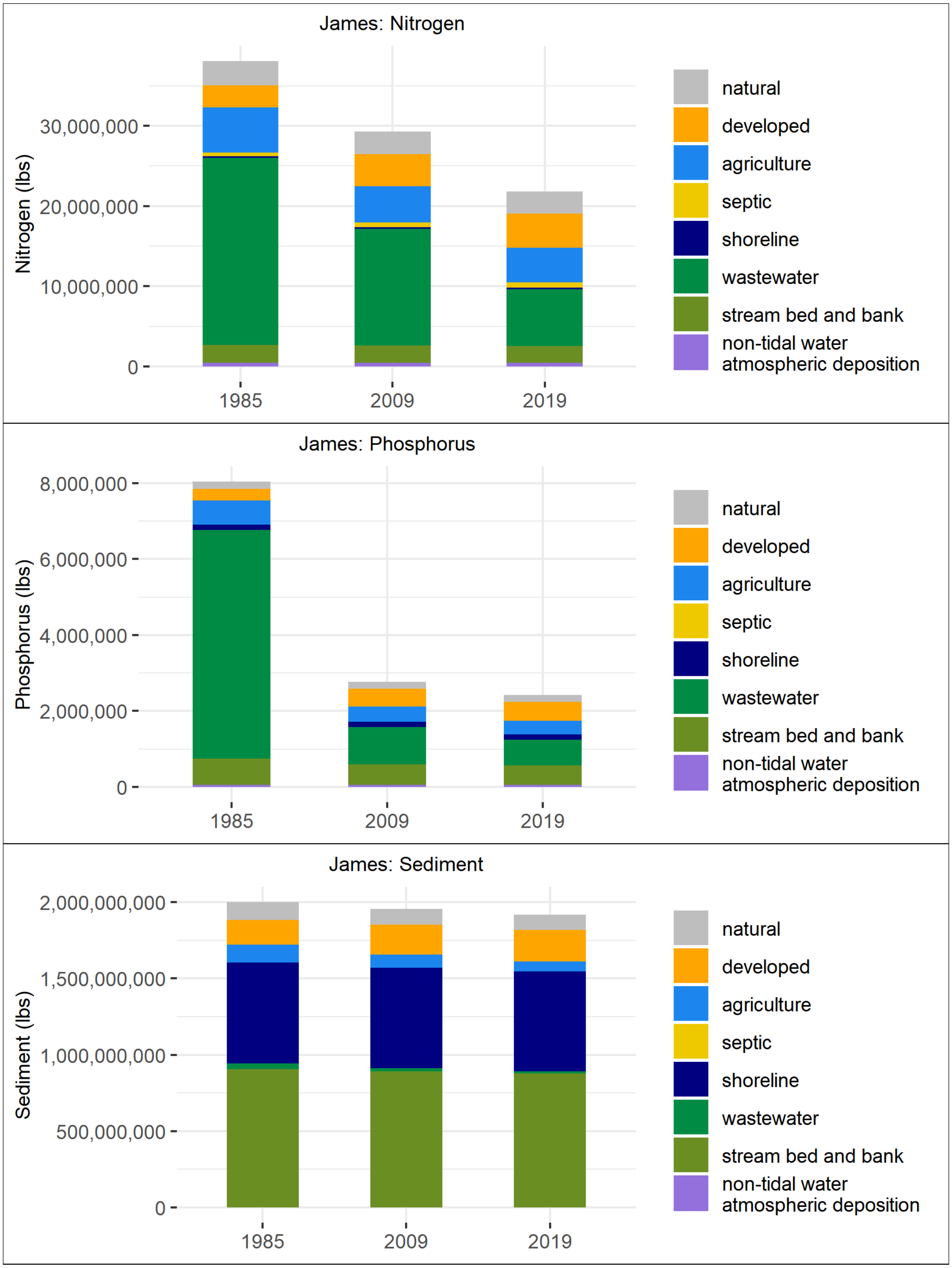


Figure 20. Expected long-term average loads of nitrogen, phosphorus, and sediment from different sources to the tidal James, as obtained from the Chesapeake Assessment Scenario Tool (CAST-19). Data shown are time-average delivered loads over the average hydrology of 1991-2000, once the steady state

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 145, 37, 131, 0.1, 86, 238, and 17 thousand acres, respectively. Implementation levels for some practices are already close to achieving their planned 2025 levels: for example, 104% of planned acres for tillage had been achieved as of 2019. In contrast, about 7% of planned urban nutrient management implementation had been achieved as of 2019.

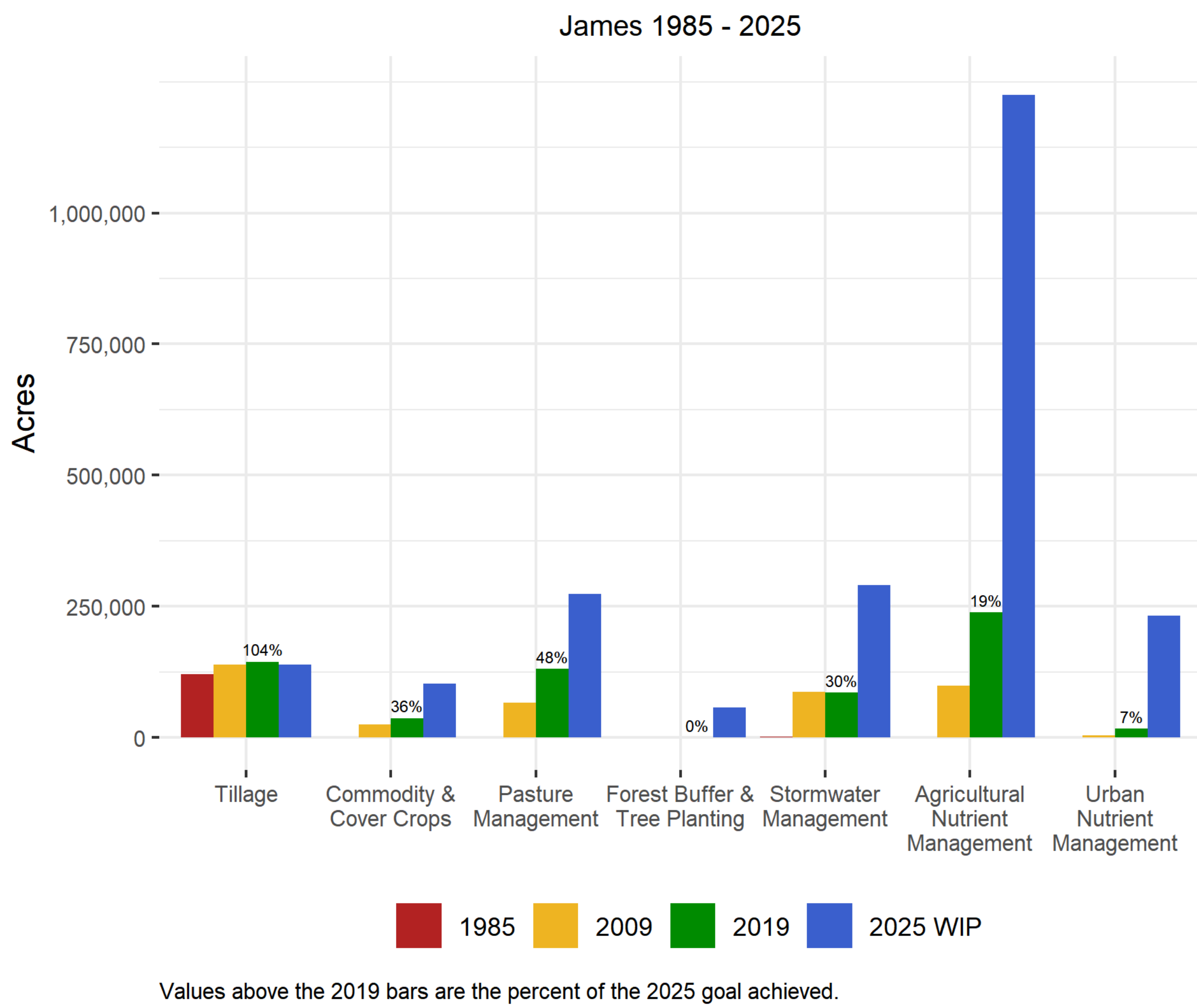


Figure 21. BMP implementation in the James watershed

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 85,175 feet in 2019. Over the same period, animal waste management systems treated 68 animal units in 1985 and 115,747 animal units in 2019 (one animal unit represents 1,000 pounds of live animal). These implementation levels represent 16% and 29% 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 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. 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 James 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
02011500	BACK CREEK NEAR MOUNTAIN GROVE, VA	1985	**-7.1**	-	-
		2009	**5.7**	-	-
02015700	BULLPASTURE RIVER AT WILLIAMSVILLE, VA	1985	**29.8**	-	-
		2009	8.1	-	-
02020500	CALFPASTURE RIVER ABV MILL CREEK AT GOSHEN, VA	2009	**28.4**	-	-
02024000	MAURY RIVER NEAR BUENA VISTA, VA	1985	**25.6**	-	-
		2009	0.3	-	-
02024752	JAMES RIVER AT BLUE RIDGE PKWY NR BIG ISLAND, VA	2009	0.1	**-7.9**	12.1
02031000	MECHUMS RIVER NEAR WHITE HALL, VA	1985	**-9.7**	-	-
		2009	**13.6**	-	-
02034000	RIVANNA RIVER AT PALMYRA, VA	1985	-1.7	-	-
		2009	-2.5	-	-
02035000	JAMES RIVER AT CARTERSVILLE, VA	1985	**-14.6**	**-35.7**	23.4
		2009	**-3.1**	**-7.8**	**-7.0**
02037500	JAMES RIVER NEAR RICHMOND, VA	1985	4.7	-	-
		2009	4.5	3.0	21.5
02039500		1985	**25.2**	-	-

	APPOMATTOX RIVER AT FARMVILLE, VA	2009	**27.4**	-	-
02041000	DEEP CREEK NEAR MANNBORO, VA	2009	**-6.4**	-	-
02041650	APPOMATTOX RIVER AT MATOACA, VA	1985	**0.1**	**64.7**	6.7
		2009	**10.8**	**13.4**	**19.9**
02042500	CHICHAMOMINY RIVER NEAR PROVIDENCE FORGE, VA	1985	**-16.4**	-	-
		2009	0.8	**5.6**	**-5.3**

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 (Figure 22) (Kemp *et al.*, 2005; Testa *et al.*, 2017). 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).

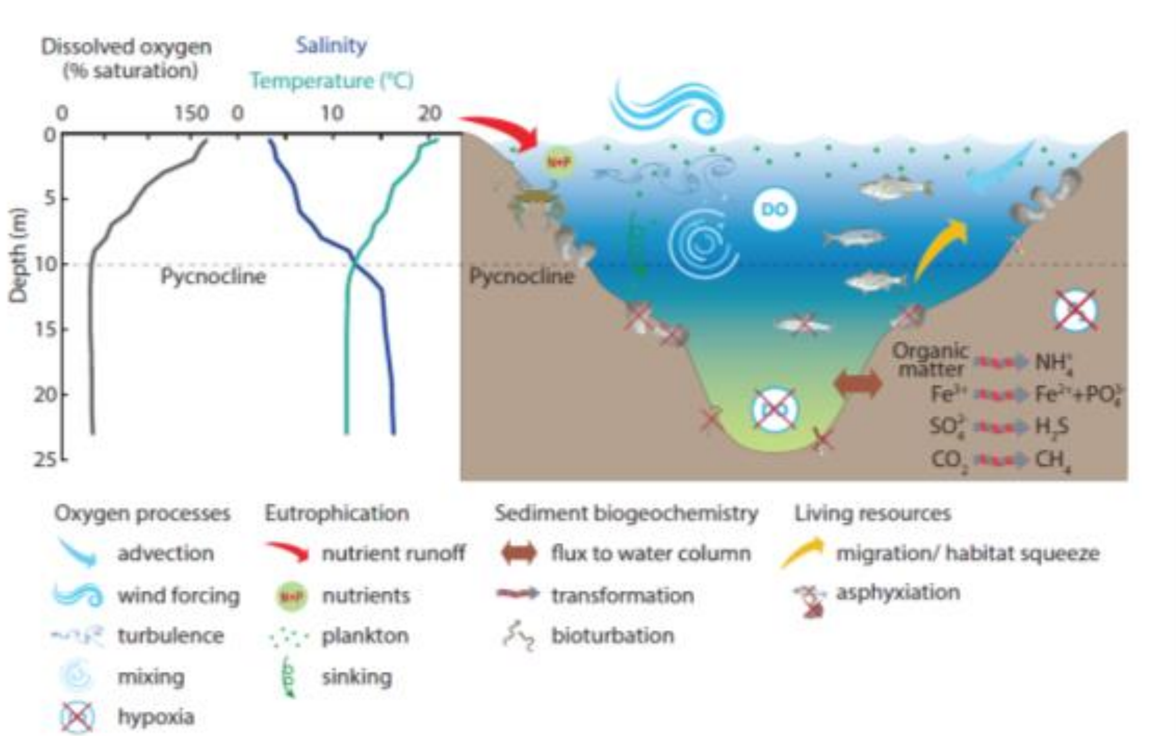


Figure 22. Conceptual diagram illustrating how hypoxia is driven by eutrophication and physical forcing, while affecting sediment biogeochemistry and living resources. From Testa *et al.* (2017).

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.

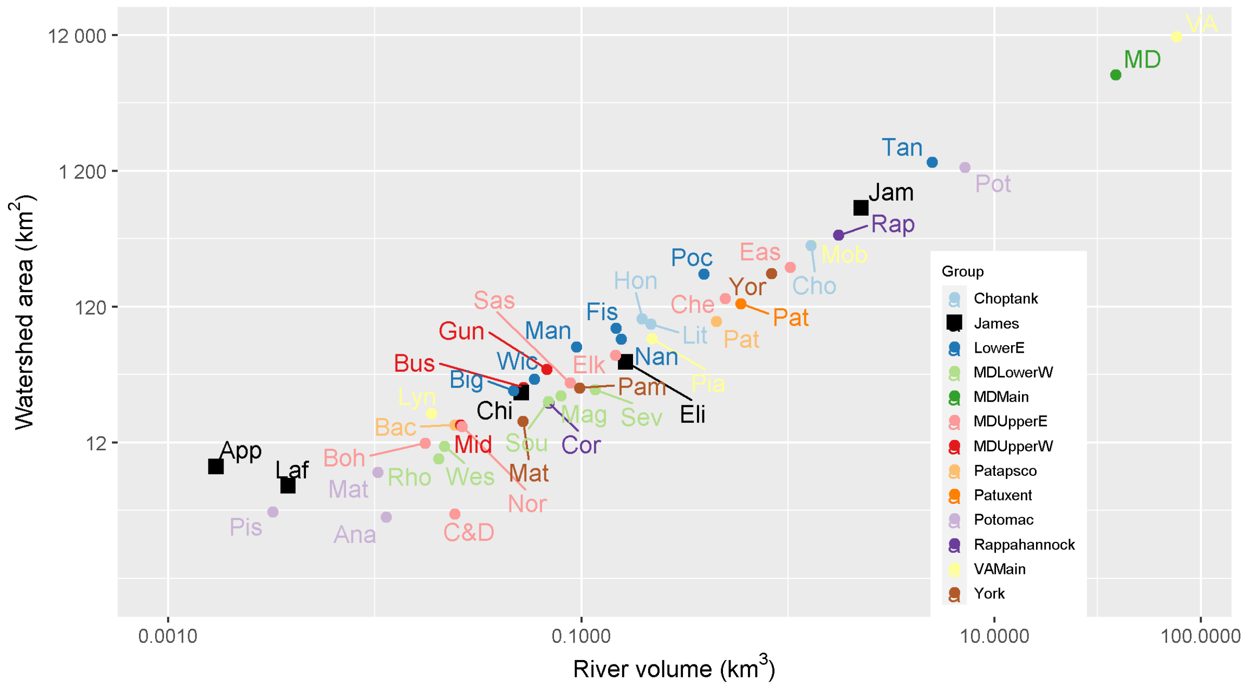


Figure 23. Watershed area vs estuarine volume.

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

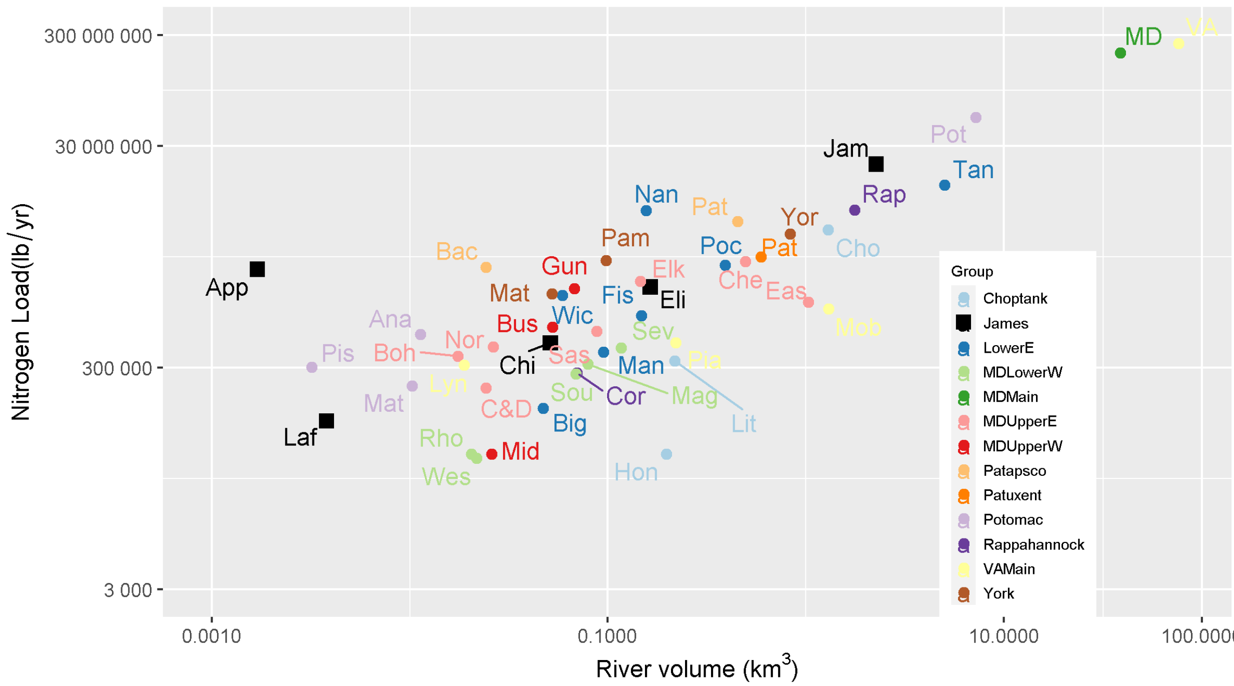


Figure 24. Annual average expected phosphorus loads versus estuarine volume. Phosphorus loads are from the 2018 progress scenarios in CAST (Chesapeake Bay Program, 2020), which is an estimate of phosphorus loads under long-term average hydrology given land use and reported management as of 2018.

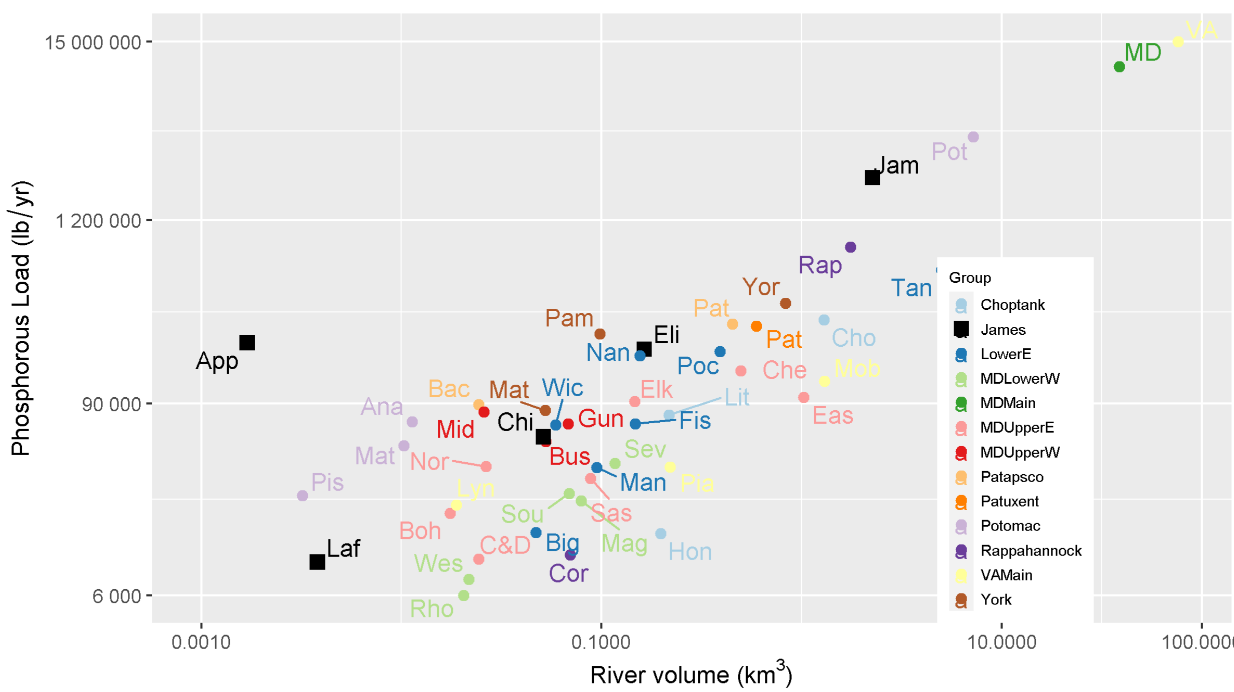


Figure 25. Annual average expected phosphorus loads versus estuarine volume. Phosphorus loads are from the 2018 progress scenarios in CAST (Chesapeake Bay Program, 2020), which is an estimate of

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

The James river estuary volume and watershed contain approximately 3 and 16% of the total volume and watershed of the Chesapeake Bay. This ranks the James as the 5th largest volume and 3rd 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. Several of the smaller tributaries within the James system, the Elizabeth river, Chickahominy river, and Lafayette river, all follow the same trend as the main James river. The Appomattox river appears to have a higher watershed area relative to its estuarine volume, indicating potentially high susceptibility. The Appomattox river also has an elevated load of phosphorus relative to its estuarine volume. The Appomattox river have high relative loads of nitrogen, while nitrogen loads in the Elizabeth river, Chickahominy river, and Lafayette river, also are more moderate.

5.3 Insights on Changes in the James

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