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

2. Location

The Maryland Upper Western Shore watershed covers approximately 0.9% of the Chesapeake Bay drainage basin. Its watershed is approximately 1,523 km^2 (Table 1.) and is contained within one state, 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 Maryland Upper Western Shore watershed stretches across two major physiographic regions, namely, Piedmont and Coastal Plain (Bachman *et al.*, 1998) (Figure 1). The Piedmont physiography covers primarily 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.

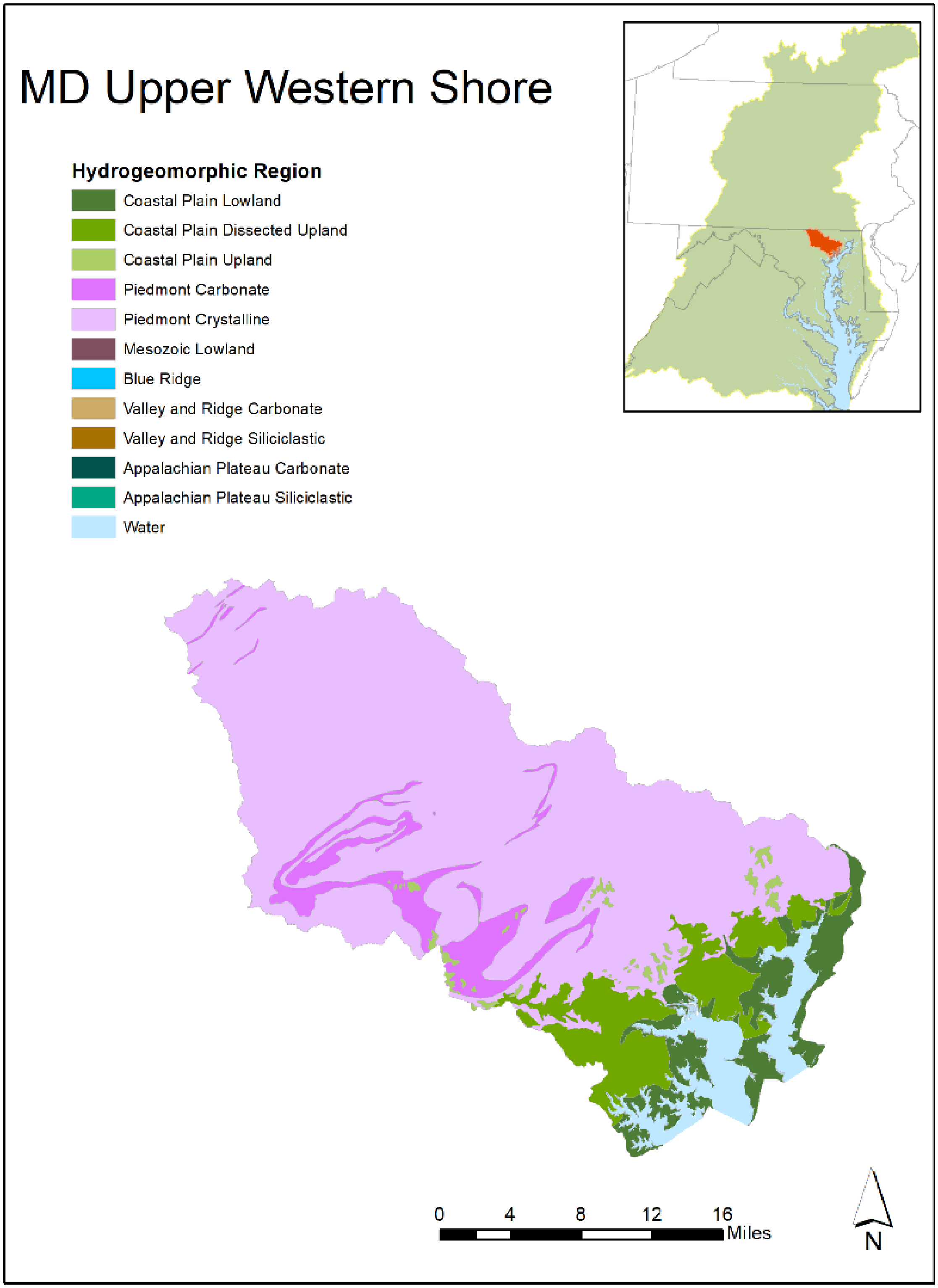

Figure 1. Distribution of physiography in the Maryland Upper Western Shore watershed.

2.2 Land Use

Land use in the Maryland Upper Western Shore watershed is dominated (46%) by natural areas. Urban and suburban land areas have increased by 45,773 acres since 1985, agricultural lands have decreased by 28,078 acres, and natural lands have decreased by 17,698 acres. Correspondingly, the proportion of urban land in this watershed has increased from 20% in 1985 to 32% in 2019 (Figure 2).

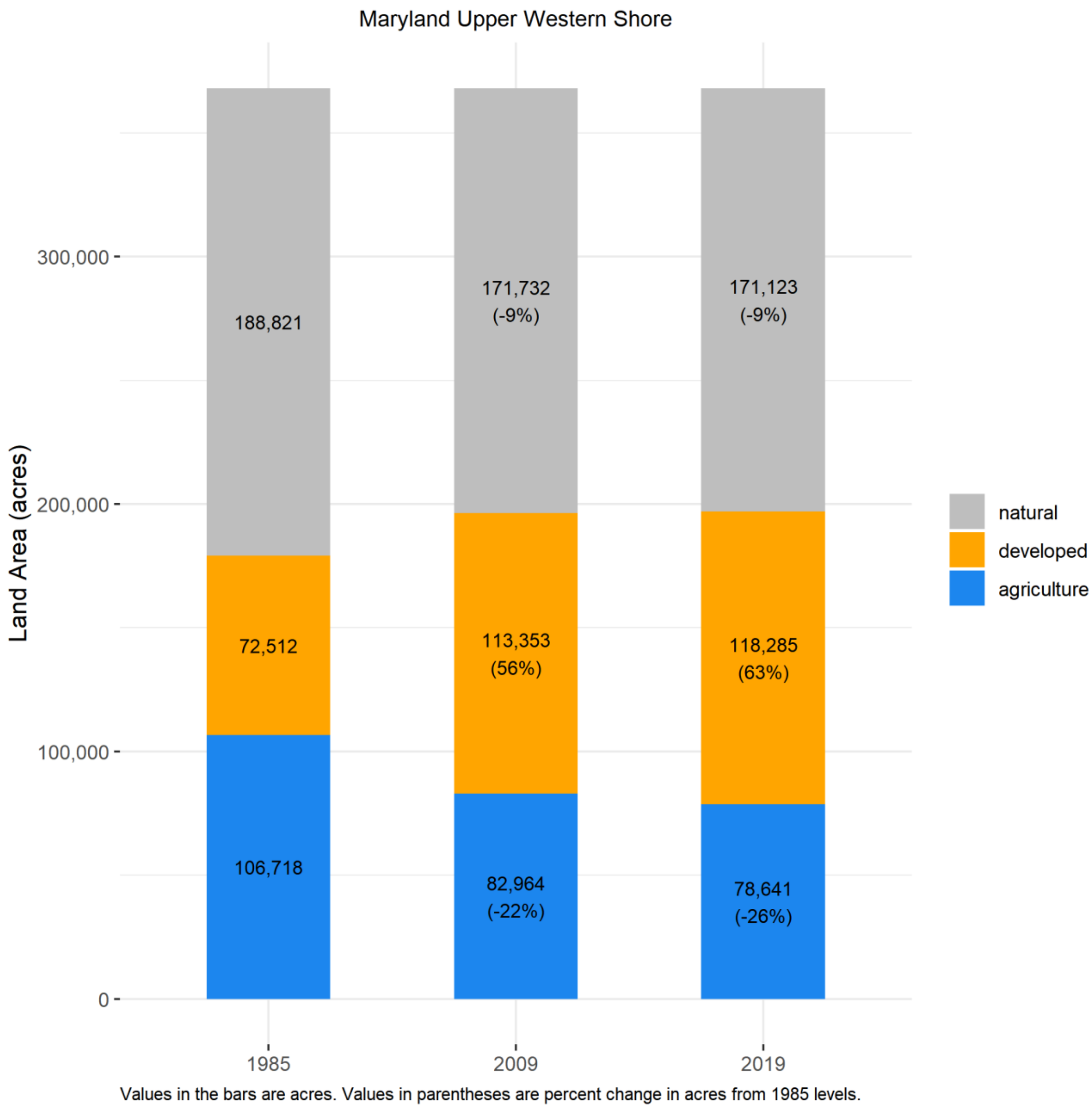

Figure 2. Distribution of land uses in the Maryland Upper Western Shore watershed. Percentages are the percent change from 1985 for each source sector.

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

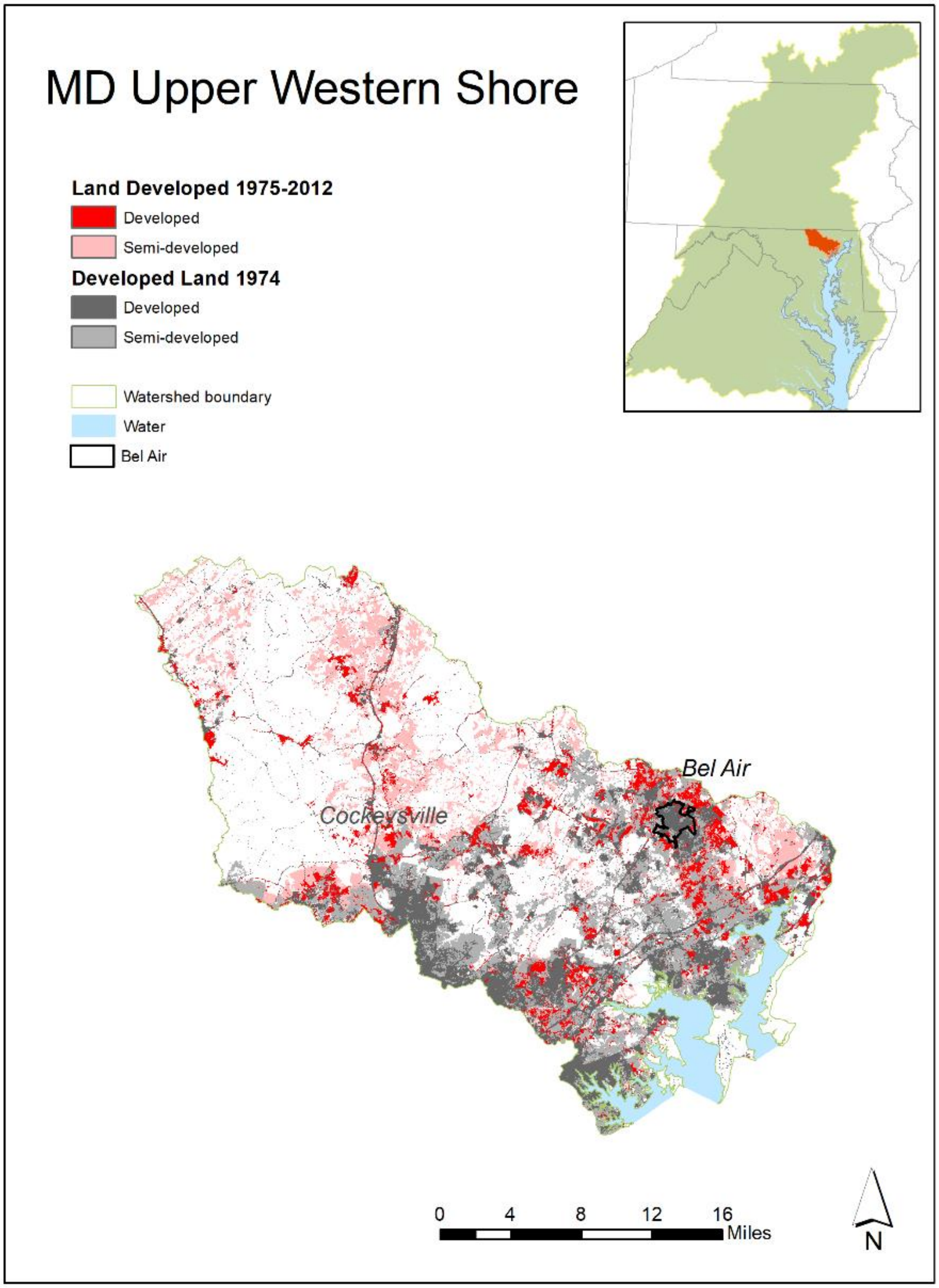

Figure 3. Distribution of developed land in the Maryland Upper Western Shore watershed. Derived from Falcone (2015). Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

2.3 Tidal Waters and Stations

For the purposes of water quality standards assessment and reporting, the tidal waters associated with the Maryland Upper Western Shore Tributaries are divided into three segments (U.S. Environmental Protection Agency, 2004). The segments are all oligohaline and are in the Bush River (BSHOH), Gunpowder River (GUNOH), and Middle River (MIDOH) (Figure 4).

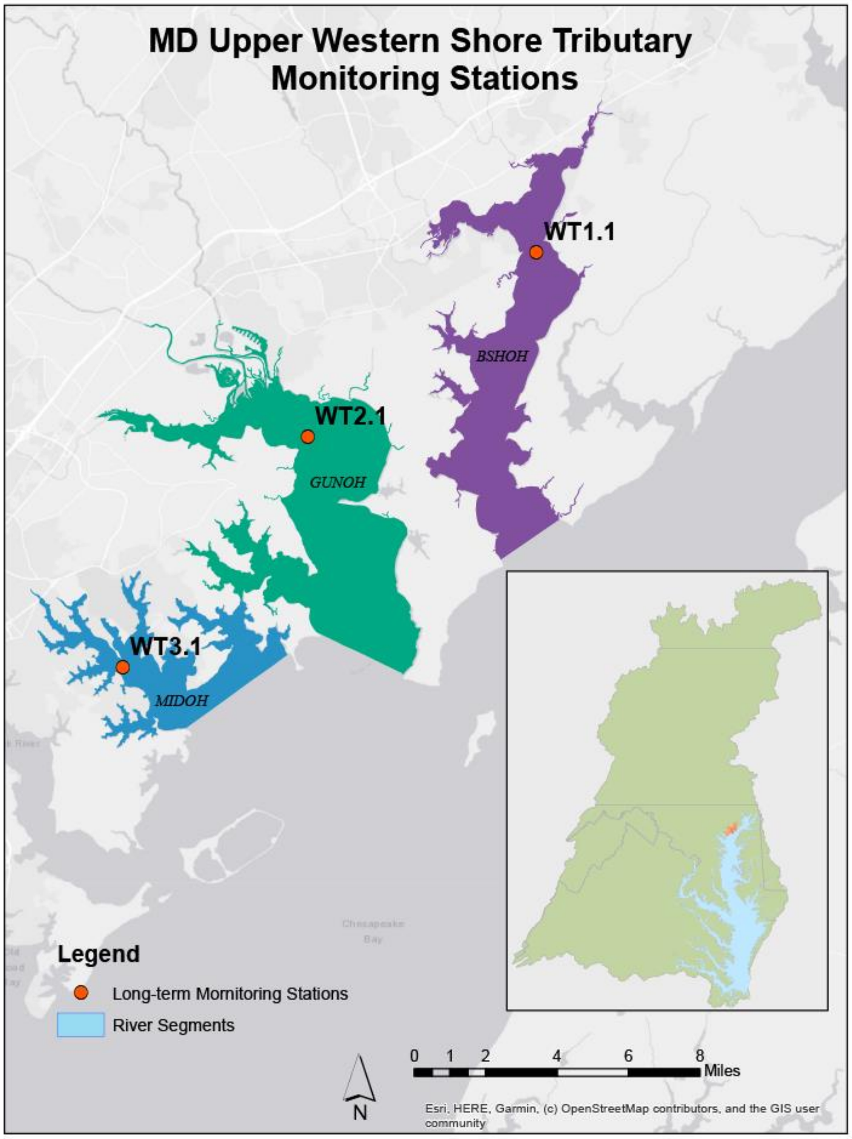

Figure 4. Map of tidal Maryland Upper Western Shore River segments and long-term monitoring stations. Base map credit Esri, HERE, Garmin, (c) OpenStreetMap contributors, and the GIS user community, World Geodetic System 1984.

Long-term trends in water quality are analyzed by the MD Department of Natural Resources at three stations in this group, one in each of the tributaries (Figure 4). Water quality data at these stations are also used to assess attainment of dissolved oxygen (DO) water quality criteria. All tidal water quality data analyzed for this summary are available from the Chesapeake Bay Program Data Hub (Chesapeake Bay Program, 2018). Other shorter-term monitoring has been conducted in these tributaries but is not included here for long-term trend analysis.

3. Tidal Water Quality Dissolved Oxygen Criteria Attainment

Multiple water quality standards were developed for the Maryland Upper Western Shore tributaries to protect aquatic living resources (U.S. Environmental Protection Agency, 2003; Tango and Batiuk, 2013). These standards include specific criteria for dissolved oxygen (DO) and water clarity/underwater bay grasses. For the purposes of this summary, a record of the evaluation results indicating whether 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 (Table 2) 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 Gunpower River and Middle River oligohaline segments (GUNOH and MIDOH) met the 30-day mean OW summer DO requirements, while the Bush River oligohaline segment (BSHOH) did not (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	BSHOH	GUNOH	MIDOH
1985-1987			
1986-1988			
1987-1989			
1988-1990			
1989-1991			
1990-1992			
1991-1993			
1992-1994			
1993-1995			
1994-1996			
1995-1997			
1996-1998			
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2000-2002			
2001-2003			
2002-2004			

2003-2005			
2004-2006			
2005-2007			
2006-2008			
2007-2009			
2008-2010			
2009-2011			
2010-2012			
2011-2013			
2012-2014			
2013-2015			
2014-2016			
2015-2017			
2016-2018			

Comparing trends in station-level DO concentrations to the computed DO criterion status for a recent assessment period can reveal valuable information, such as whether progress is being made towards attainment in a segment that is not meeting the water quality criteria, or conversely the possibility that conditions are degrading even if the criteria are currently being met. To illustrate this, the 2016-2018 attainment status for the OW summer DO criteria shown in Table 2 is overlain with the 1985-2018 change in summer surface DO concentration (Figure 5). In the three tributaries, a mixture of trends in surface DO and criterion status exists. The Bush River surface DO is not trending, and the OW summer DO criterion was not met in the recent period. On the other hand, the Gunpowder River station has an improving DO trend, while the segment is meeting the criterion. Finally, the Middle River surface DO is possibly degrading but the segment met the OW summer DO criterion in the recent period.

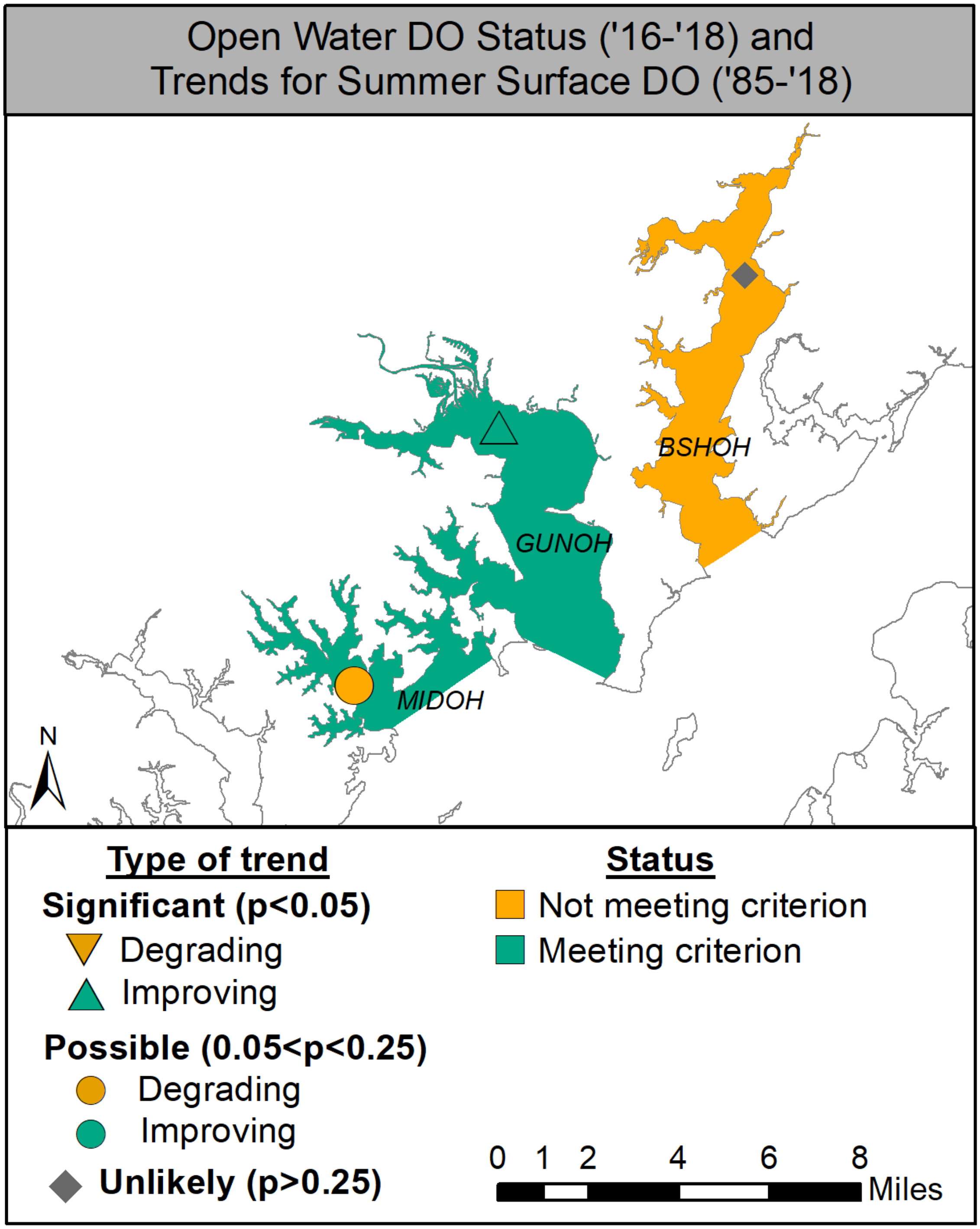

Figure 5. Pass-fail DO criterion status for 30-day OW summer DO designated uses in Maryland Upper Western Shore segments along with long-term trends in DO concentrations. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

4. Tidal Water Quality Trends

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

Results shown below in each set of maps (e.g., Figure 6) include those generated using two different GAM fits to each station-parameter combination. The first approach involves fitting a GAM to the raw observations to generate a mean estimate of the concentrations over time, as observed in the estuary. The second approach involves including monitored river flow or *in situ* salinity (as an aggregated measure of multiple river flows) in the GAM to explain some of the variation in the water quality parameter. From the results of this second approach, it is possible to estimate the “flow-adjusted” change over time, which gives a mean estimate of what the water quality parameter trend would have been if river flow had been average over the period of record. Note that depending on station and parameter, sometimes gaged river flow is used for this adjustment and sometimes salinity is used, but we refer to all 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 station WT1.1 in the Bush River consistently using both trends on concentration data alone and adjusting for flow over the long- and short-term (Figure 6). The Gunpowder River station (WT2.1) shows a possible improvement using the non-adjusted data over the long-term and the Middle River station (WT3.1) has a significant long-term improvement with flow-adjustment. Over the short-term only WT1.1 has a trend.

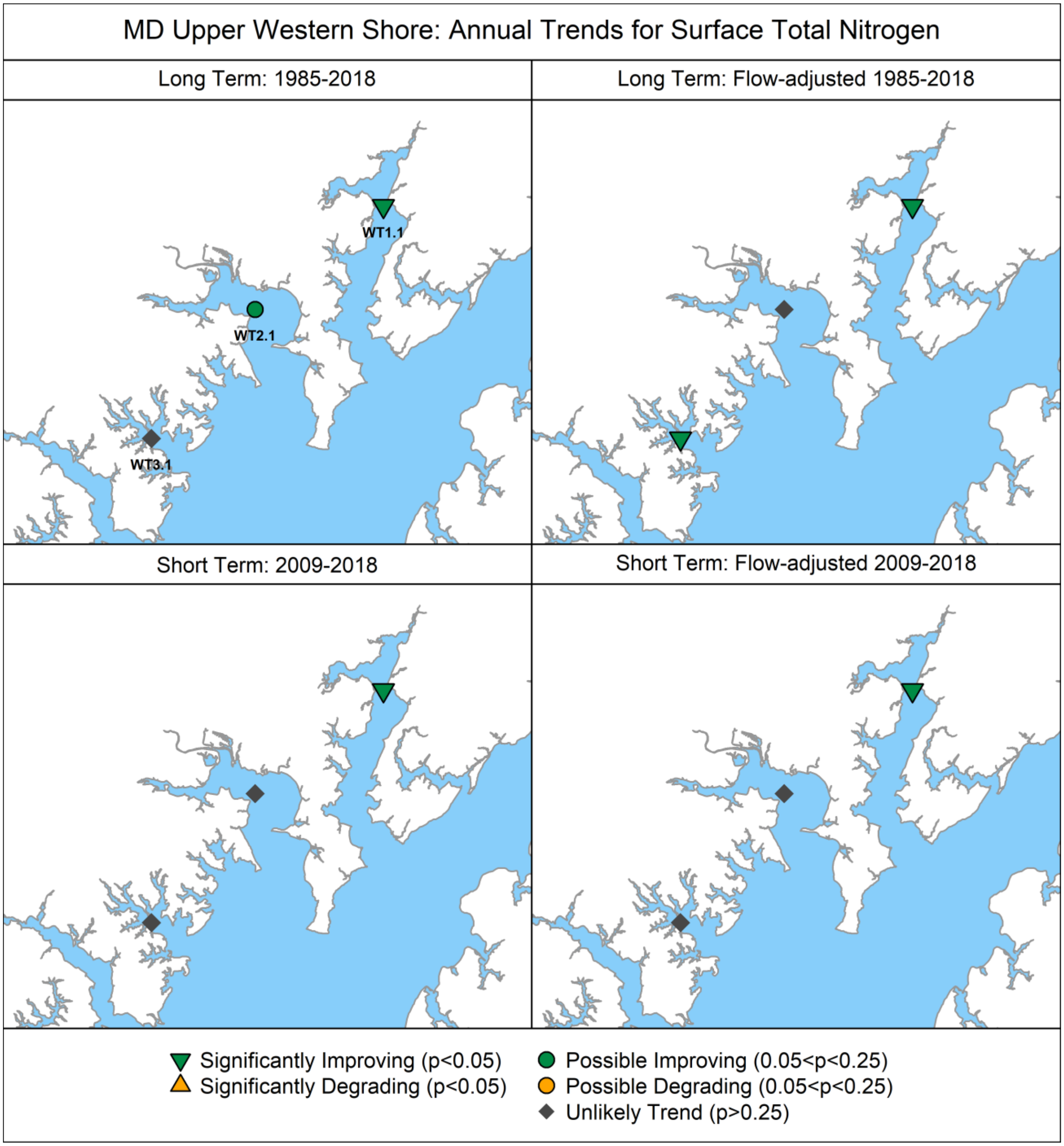

Figure 6. Surface TN trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

The long-term TN decrease in the Bush River is a result of clearly lower concentrations in the second half of the record than the first (Figure 7). Otherwise, the non-flow-adjusted mean annual GAM estimates presented in Figure 7 show variable TN concentrations over time. 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 this change was 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).

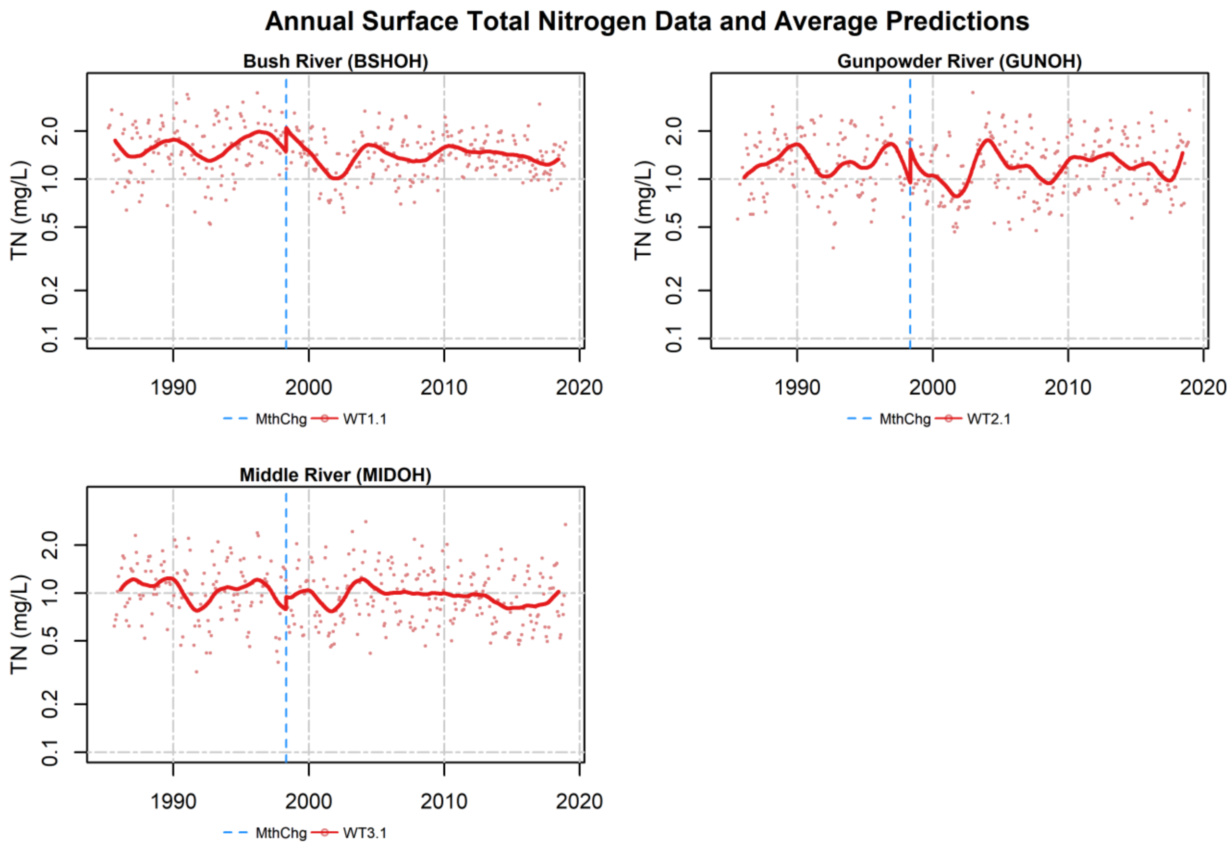

Figure 7. Surface TN data (dots) and average long-term pattern generated from non-flow-adjusted GAMs. Colored dots represent data corresponding to the monitoring station indicated in the legend; colored lines represent mean annual GAM estimates for the noted monitoring stations. Vertical blue dotted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

4.2 Surface Total Phosphorus

Surface total phosphorus (TP) trends are possibly improving over the long-term at all three of the Upper Western Shore stations, although with flow-adjustment, two of the stations show no trend instead (Figure 8). Over the short-term, there is only one improving trend (WT1.1) without flow-adjustment.

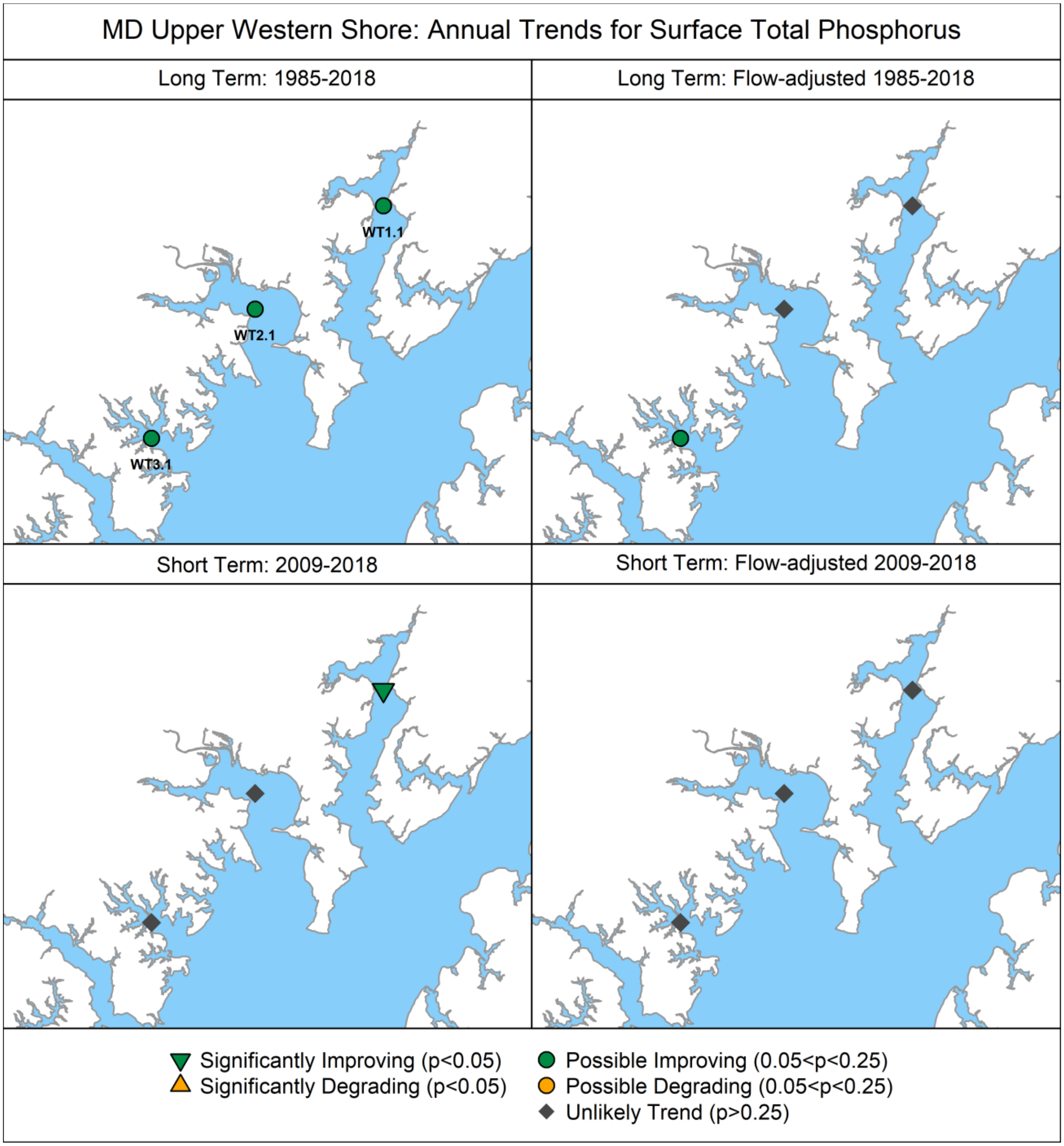

Figure 8. Surface TP trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

The long-term decrease at each of these stations is slight, but still apparent in the data values and mean annual GAM estimates (Figure 9).

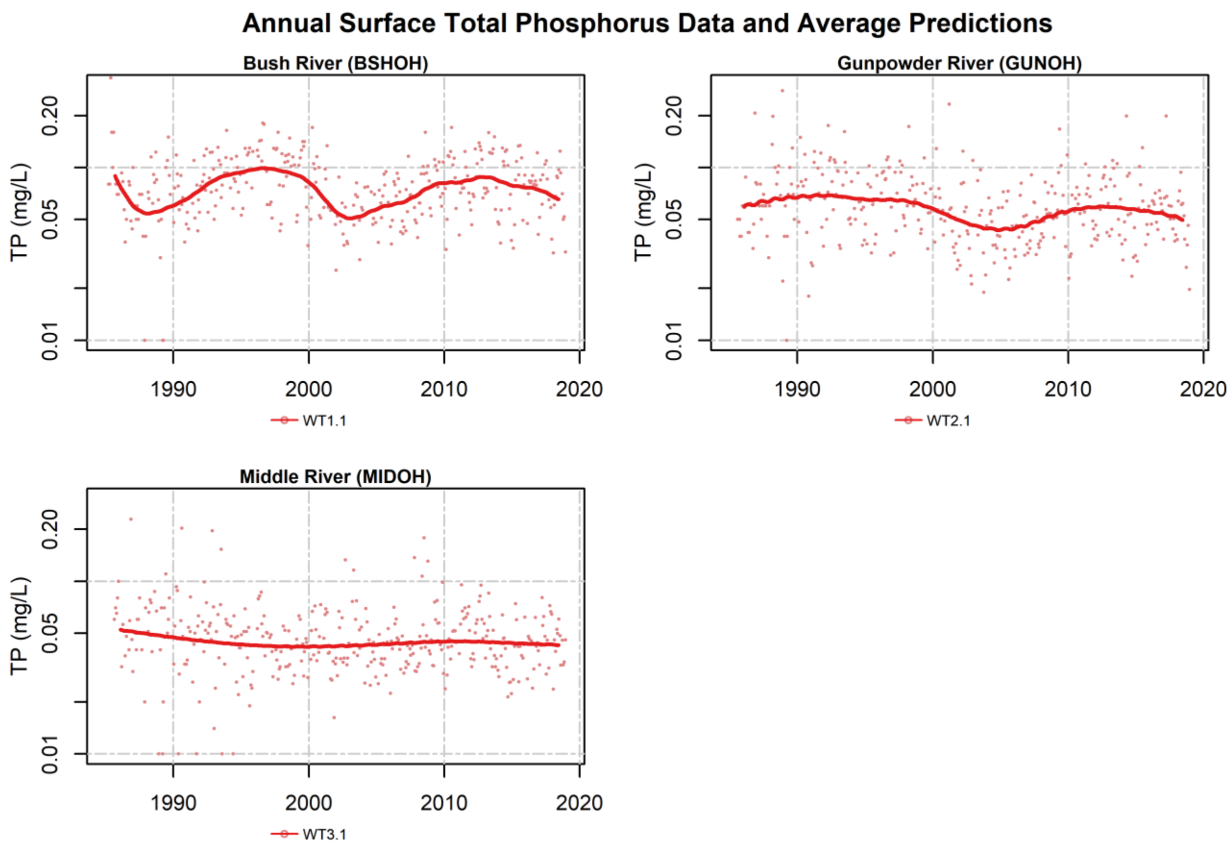

Figure 9. Surface TP data (dots) and average long-term pattern generated from non-flow adjusted GAM. Colored dots represent data corresponding to the monitoring station indicated in the legend; colored lines represent mean annual GAM estimates for the noted monitoring stations.

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 chlorophyll *a* is degrading over the long-term at the Bush River (WT1.1) and the Gunpowder River (WT2.1) stations, with and without flow-adjustment (Figure 10). The Middle River station shows no trend over the long-term, and over the short-term there are no spring chlorophyll *a* trends at these stations.

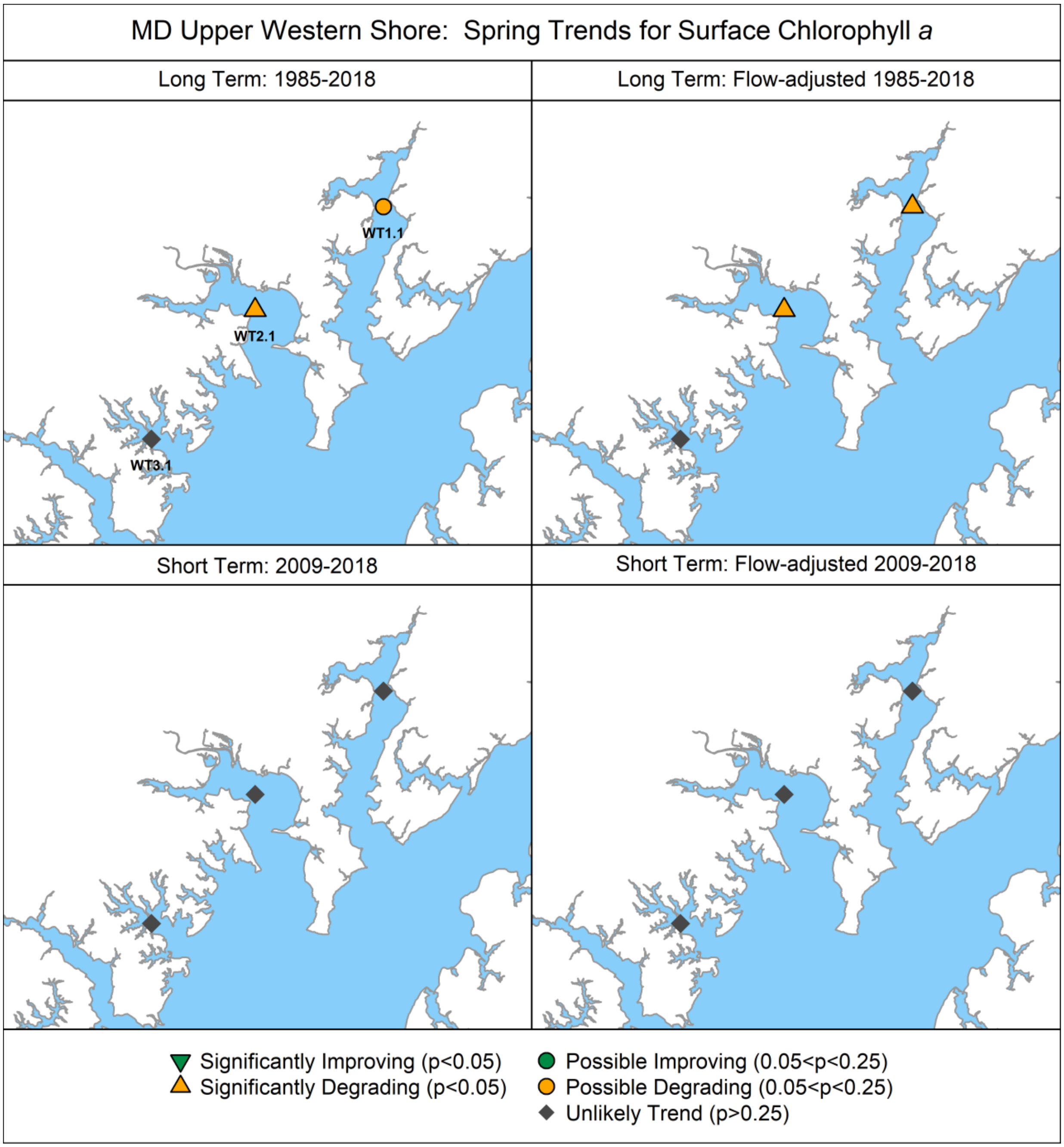

Figure 10. Surface spring (March-May) chlorophyll *a* trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

The observed chlorophyll *a* concentrations and average spring GAM estimates are increasing at the Bush and Gunpowder stations, and fairly level at all three stations over the last 10 years (Figure 11).

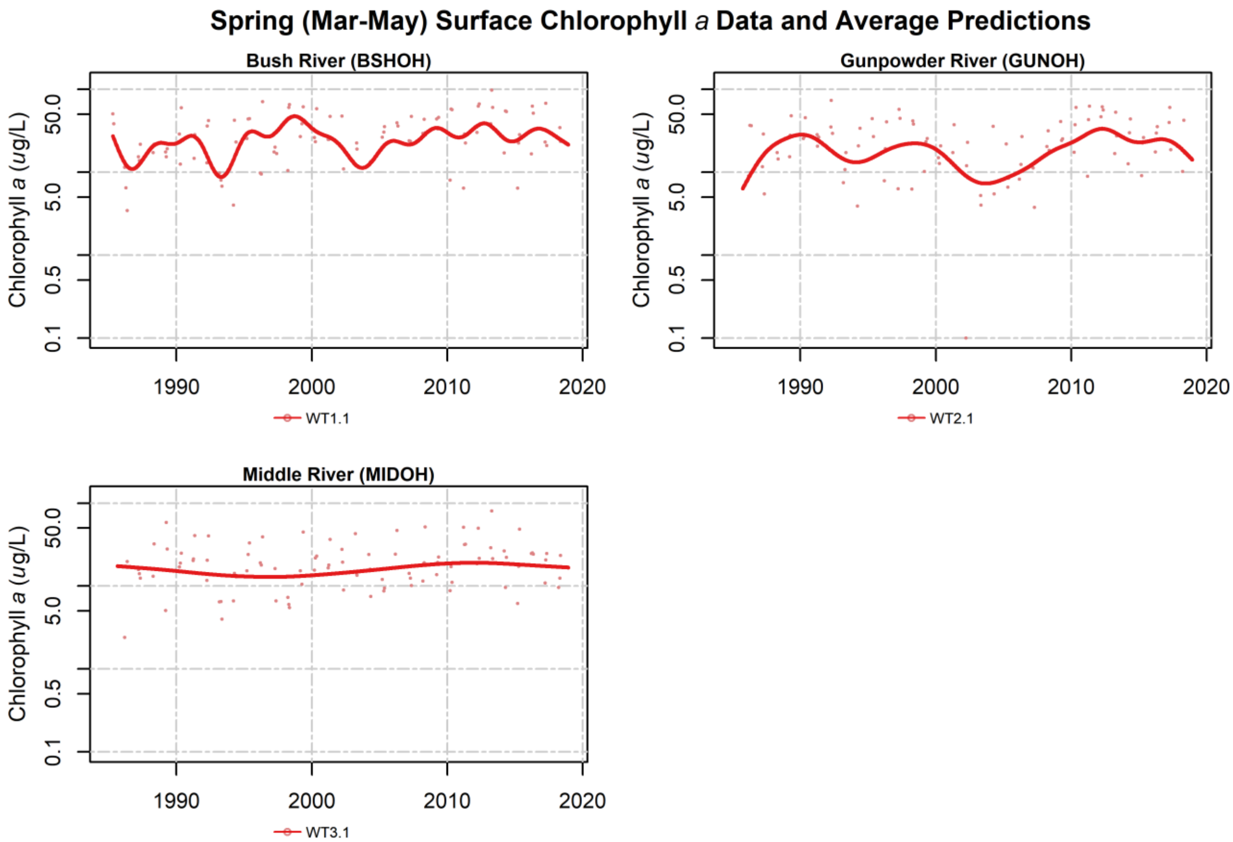

Figure 11. Surface spring chlorophyll *a* data (dots) and average long-term pattern generated from non-flow adjusted GAM. Colored dots represent March-May data corresponding to the monitoring station indicated in the legend; colored lines represent mean spring GAM estimates for the noted monitoring stations.

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

Summer long-term chlorophyll *a* trends are degrading at the Bush River station (WT1.1) and the Gunpowder River station (WT2.1), with and without flow-adjustment (Figure 12), similar to the spring chlorophyll *a* trends. The difference between spring and summer chlorophyll *a* is an improving summer trend at the Middle River station (WT3.1) which continues over the short-term with and without flow-adjustment (Figure 12).

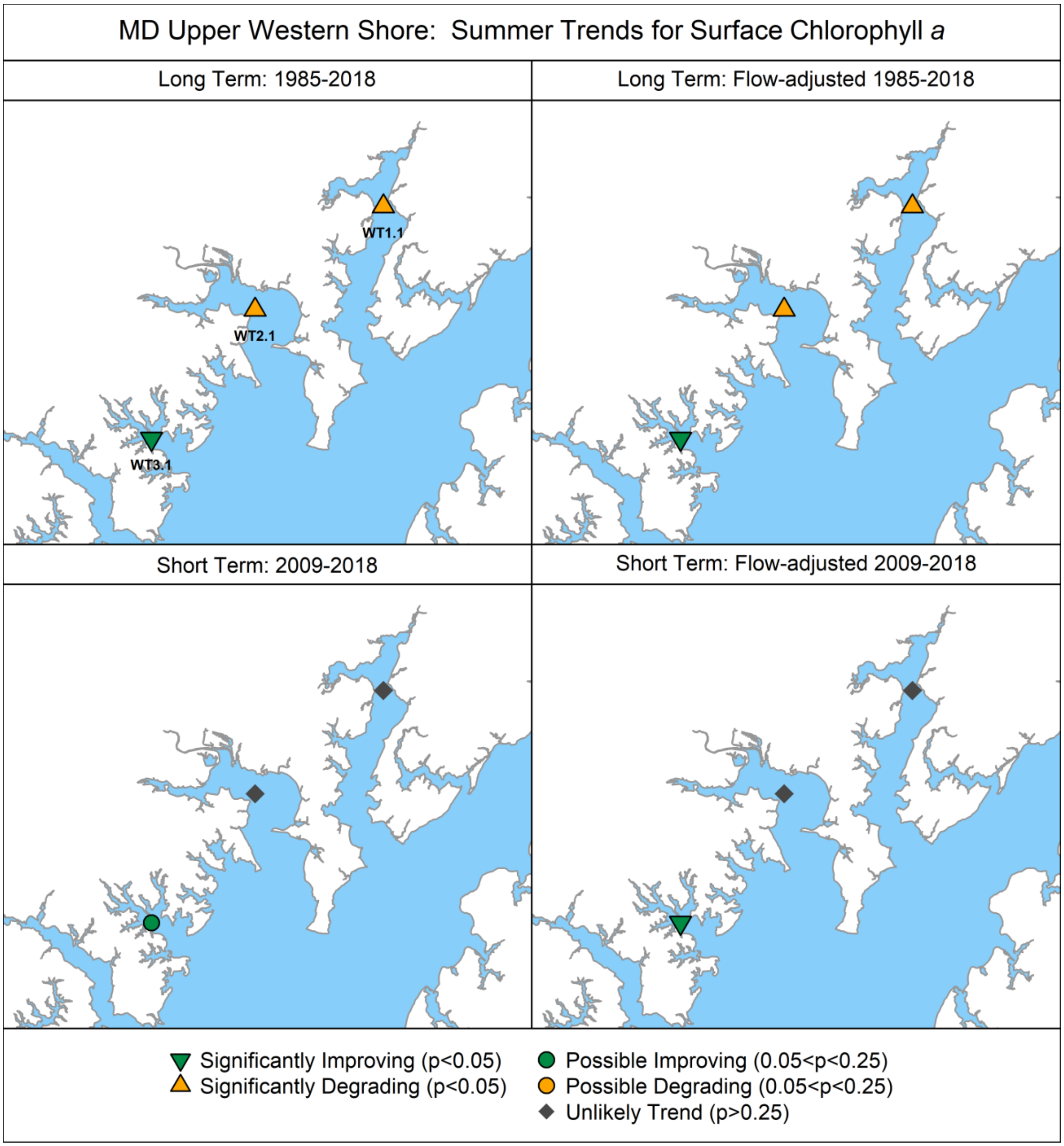

Figure 12. Surface summer (July-September) chlorophyll *a* trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

The observed chlorophyll *a* concentrations and average seasonal GAM estimates are very similar at WT1.1 and WT2.1 in the summer (Figure 13) compared to the spring (Figure 11). The summer concentrations and mean summer GAM estimates at WT3.1 are lower than the other two stations and clearly decreasing (Figure 13).

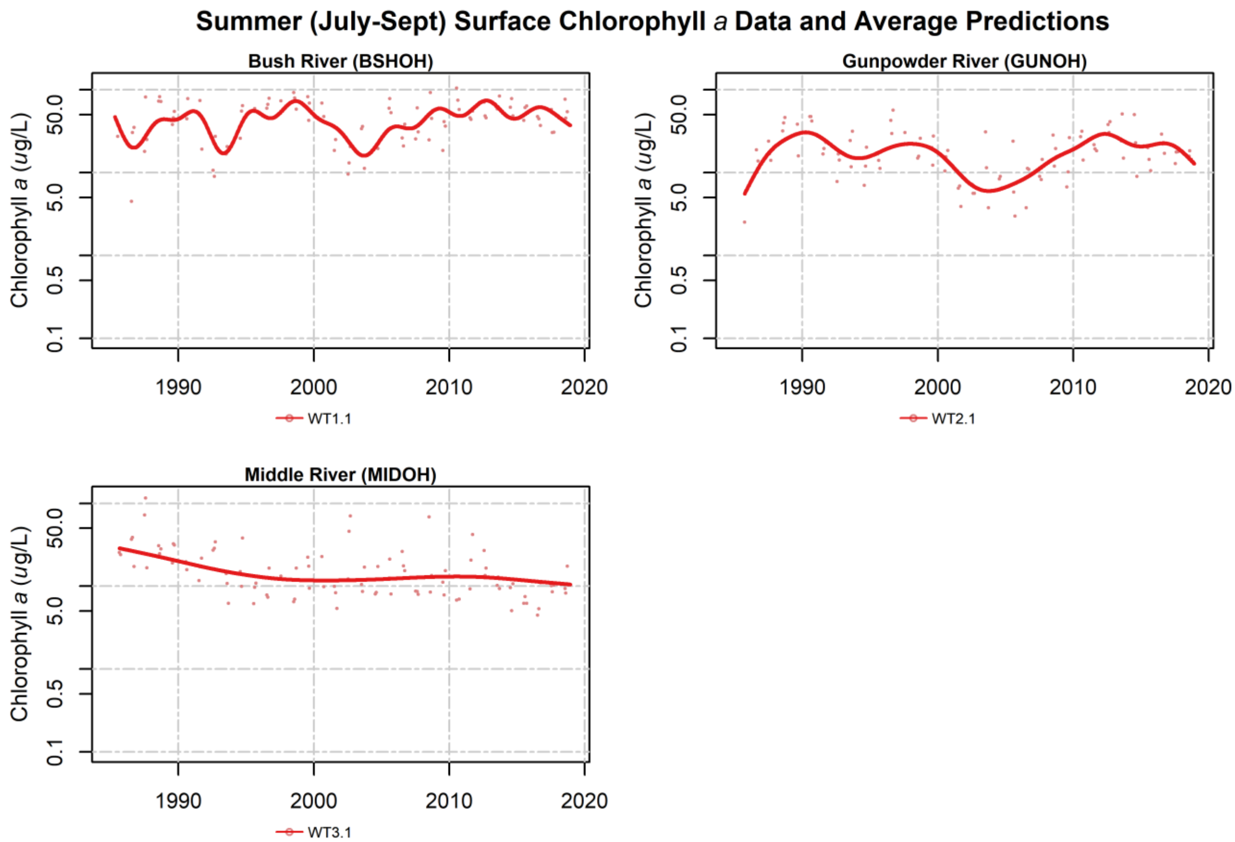

Figure 13. Surface summer chlorophyll *a* data (dots) and average long-term pattern generated from non-flow adjusted GAM. Colored dots represent July-September data corresponding to the monitoring station indicated in the legend; colored lines represent mean summer GAM estimates for the noted monitoring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

4.5 Secchi Disk Depth

Trends in Secchi disk depth, a measure of visibility through the water column, are degrading at two of the stations over the long-term both with and without flow-adjustment (Figure 14). Over the short-term, without flow adjustment, Secchi trends are improving at all three stations and with flow adjustment possibly improving at one.

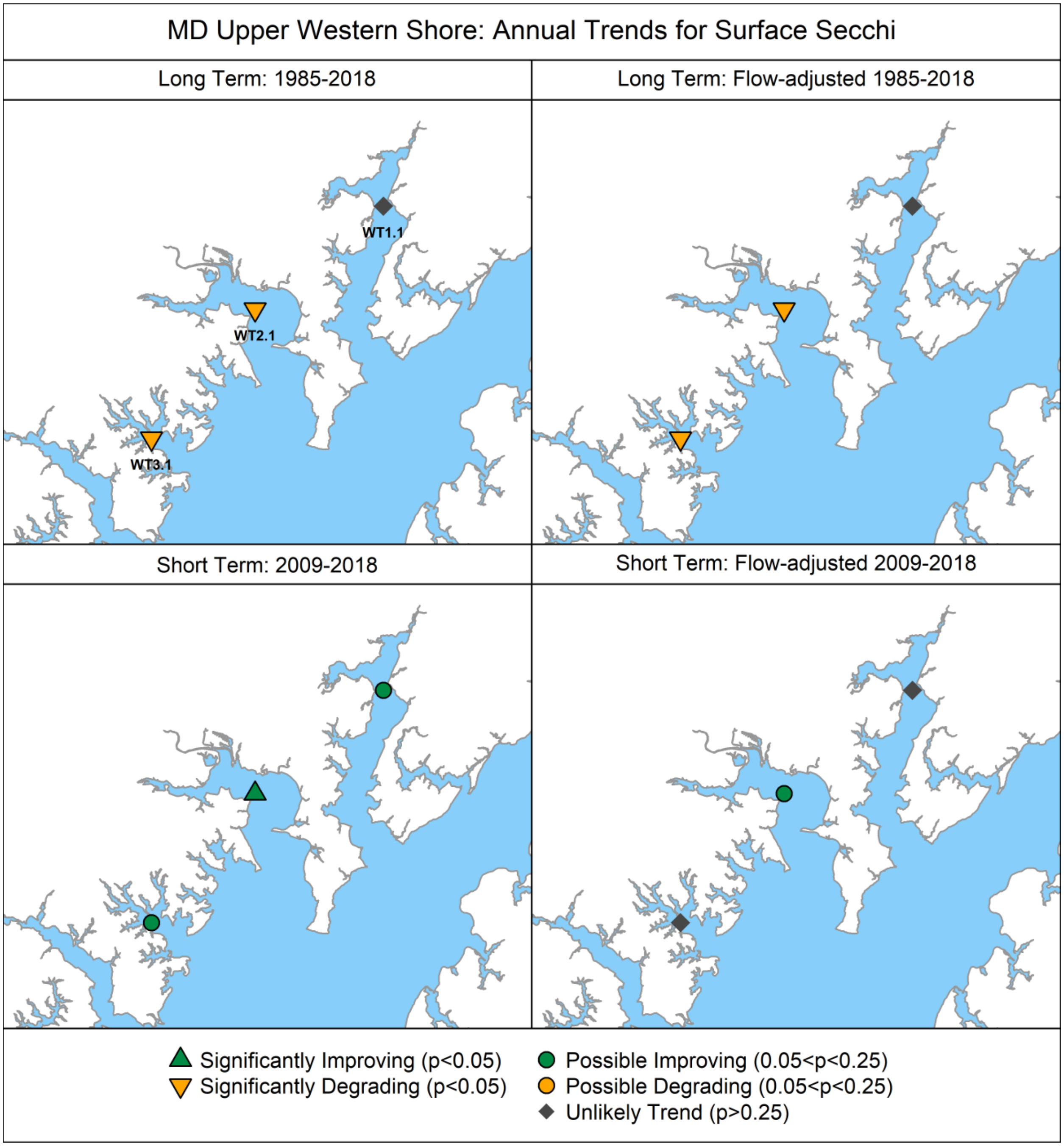

Figure 14. Annual Secchi depth trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

Secchi depth is on average shallower at the Bush and Gunpowder River stations than the Middle River (Figure 15). The observations are much more variable at the Gunpowder and Middle stations which also show decreases in mean annual GAM estimates over time. The short-term increases at all three stations are small, but evident in the graphs.

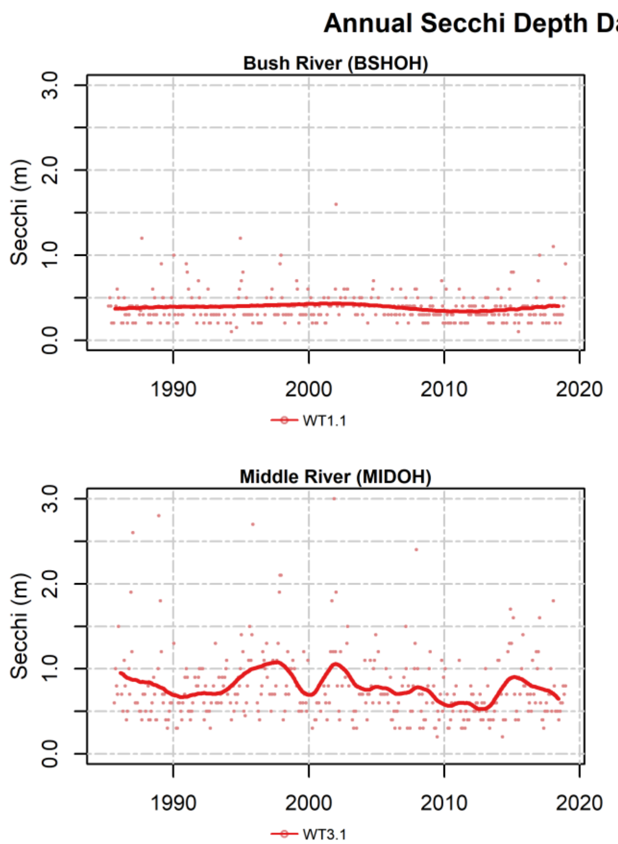

Figure 15. Annual Secchi depth data (dots) and average long-term pattern generated from non-flow adjusted GAM. Colored dots represent data corresponding to the monitoring station indicated in the legend; colored lines represent mean annual GAM estimates for the noted monitoring stations.

4.6 Summer Bottom Dissolved Oxygen (June-September)

Bottom summer DO trends are improving consistently over the long- and short-term with and without flow adjustment at WT2.1 (Gunpowder River) (Figure 16). Bottom DO trends are mixed at the other two upper western shore stations with some possible trends (Figure 16).

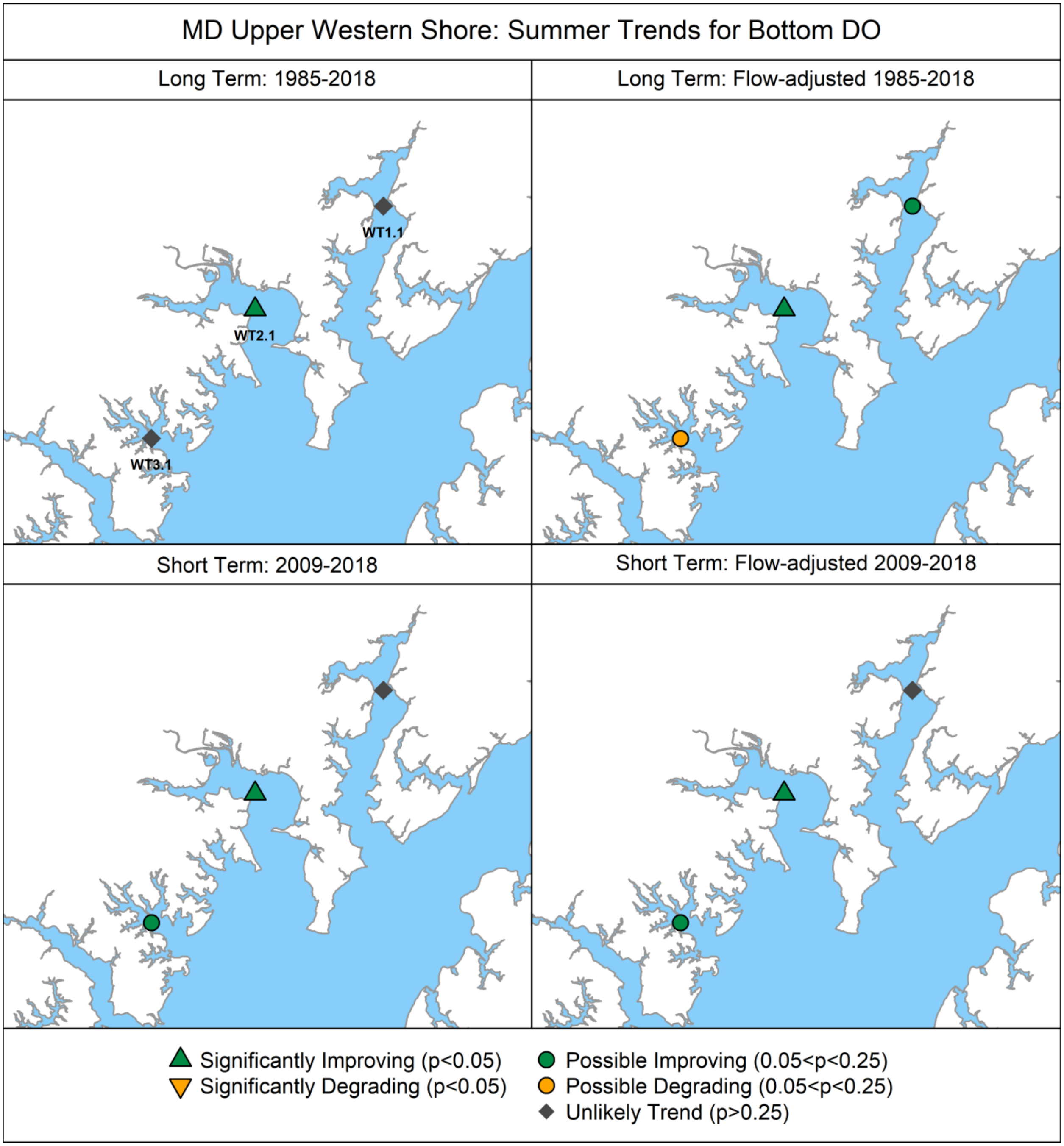

Figure 16. Summer (June-September) bottom DO trends. Base map credit Chesapeake Bay Program, www.chesapeakebay.net, North American Datum 1983.

Summer bottom DO concentrations do not get as low at these upper western shore stations as they do in other regions of the Chesapeake Bay. The 5 mg/L summer Open Water 30-day mean DO criterion applies to these tributaries, but not the deeper water criteria. An increase in DO concentrations and mean summer GAM estimates over the last half of the record is apparent at WT2.1 (Figure 17) which is the station showing consistently improving trends in Figure 16. The other two stations' patterns are relatively flat over time.

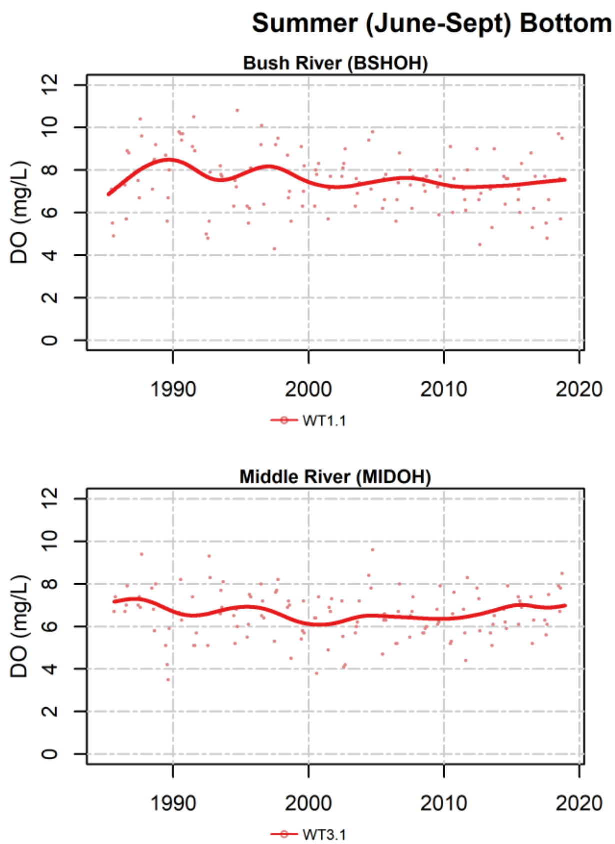

Figure 17. Summer (June-September) bottom DO data (dots) and mean summer long-term pattern generated from non-flow adjusted GAMs. Colored dots represent July-September data corresponding to the monitoring station indicated in the legend; colored lines represent mean summer GAM estimates for the noted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

5. Factors Affecting Trends

5.1 Watershed Factors

5.1.1 Effects of Physical Setting

The geology of the Maryland's upper western shore 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).

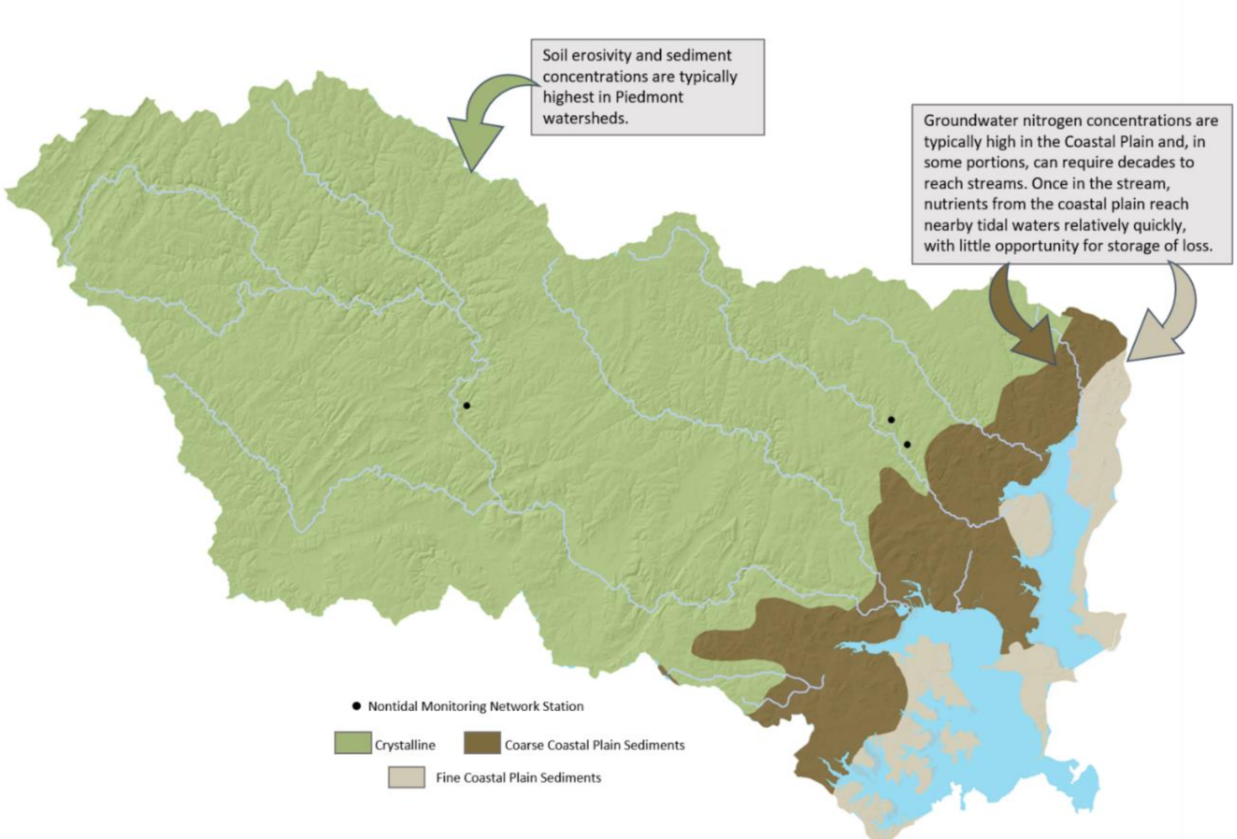

Figure 18. Effects of watershed hydrogeomorphology on nutrient transport to freshwater streams and tidal waters. Base map modified from King *et al.* (1974) and Ator *et al.* (2005), North American Datum 1983.

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). Groundwater nitrate concentrations are high in Maryland's upper western shore and are greatest in streams above the fall line that drain agricultural land uses from the Piedmont physiographic province (Greene and others, 2005; Terziotti and others, 2017). Most of Maryland's western shore lies in the Piedmont where aquifers are composed of igneous and metamorphic rocks. Crystalline rocks such as these typically have low porosity and are set on steep slopes, so less water infiltrates to groundwater (Lindsey and others, 2003), however, they can contain large amounts of oxic groundwater (Tesoriero and others, 2015), promoting nitrate transport. 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). A similar range of water ages has been measured from Piedmont crystalline springs (0 – 33 years, Phillips and others, 1999). 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).

Phosphorus

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 in Maryland's upper western shore typically occur in agricultural areas where inputs of manure and fertilizer 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).

Sediment

The delivery of sediment from upland soil erosion, streambank erosion, and tributary loading varies throughout Maryland's upper western shore, but in-stream concentrations are typically highest in the upper portion of watershed that drains 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 the upper western shore 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).

Delivery to tidal waters

The delivery of nitrogen, phosphorus, and sediment in non-tidal streams to tidal waters on Maryland's upper western 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. 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 contributes more fine-grained sediment to estuarine waters in Maryland's western shore than is delivered from the watershed (Langland and Cronin, 2003), likely as a result of such trapping and relatively small upland watershed areas.

5.1.2 Estimated Nutrient and Sediment Loads

Estimated loads to tidal portions of the Maryland Upper Western Shore Tributaries are a combination of simulated non-point source, atmospheric deposition, and reported point-source loads. These loads were obtained from the Chesapeake Bay Program Watershed Model's progress runs specific to each year from 1985 and 2018 (https://cast.chesapeakebay.net/). Nonpoint source loads were adjusted to reflect actual hydrology using the method of the Chesapeake Bay Program's Loads to the Bay indicator (see https://www.chesapeakeprogress.com/clean-water/water-quality). Over the period of 1985-2018, 0.040, 0.0035, and 2.6 million tons of nitrogen, phosphorus, and suspended sediment loads were exported from this watershed, respectively (Figure 19).

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

Nitrogen

Estimated TN loads showed an overall decline of 11 ton/yr in the period between 1985 and 2018, which is statistically significant ($p < 0.05$). Long-term, statistically significant declines were observed with both point sources (-0.84 ton/yr, $p < 0.01$) and atmospheric deposition to the tidal waters (-0.60 ton/yr, $p < 0.01$). In addition, nonpoint sources also showed a decrease in this period (-5.7 ton/yr), although it is not statistically significant ($p = 0.19$). The significant point source reductions in TN are a result of substantial efforts to reduce nitrogen loads from major wastewater treatment facilities by implementing biological nutrient removal (Boynton *et al.*, 2008; Lyerly *et al.*, 2014). 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 decrease of 0.40 ton/yr in the period between 1985 and 2018, although it is statistically significant ($p = 0.42$). Point sources showed a long-term decline (-0.15 ton/yr, $p < 0.01$), whereas nonpoint sources showed a long-term increase (0.31 ton/yr, $p = 0.48$). 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 (Boynton *et al.*, 2008; Lyerly *et al.*, 2014).

Sediment

Estimated suspended sediment (SS) loads showed an overall increase of 849 ton/yr in the period between 1985 and 2018, which is statistically significant ($p < 0.05$). This increase is entirely driven by nonpoint sources (849 ton/yr, $p < 0.05$). Like TP and TN, point source load of SS showed a statistically significant decline in this period (-0.27 ton/yr; $p < 0.01$).

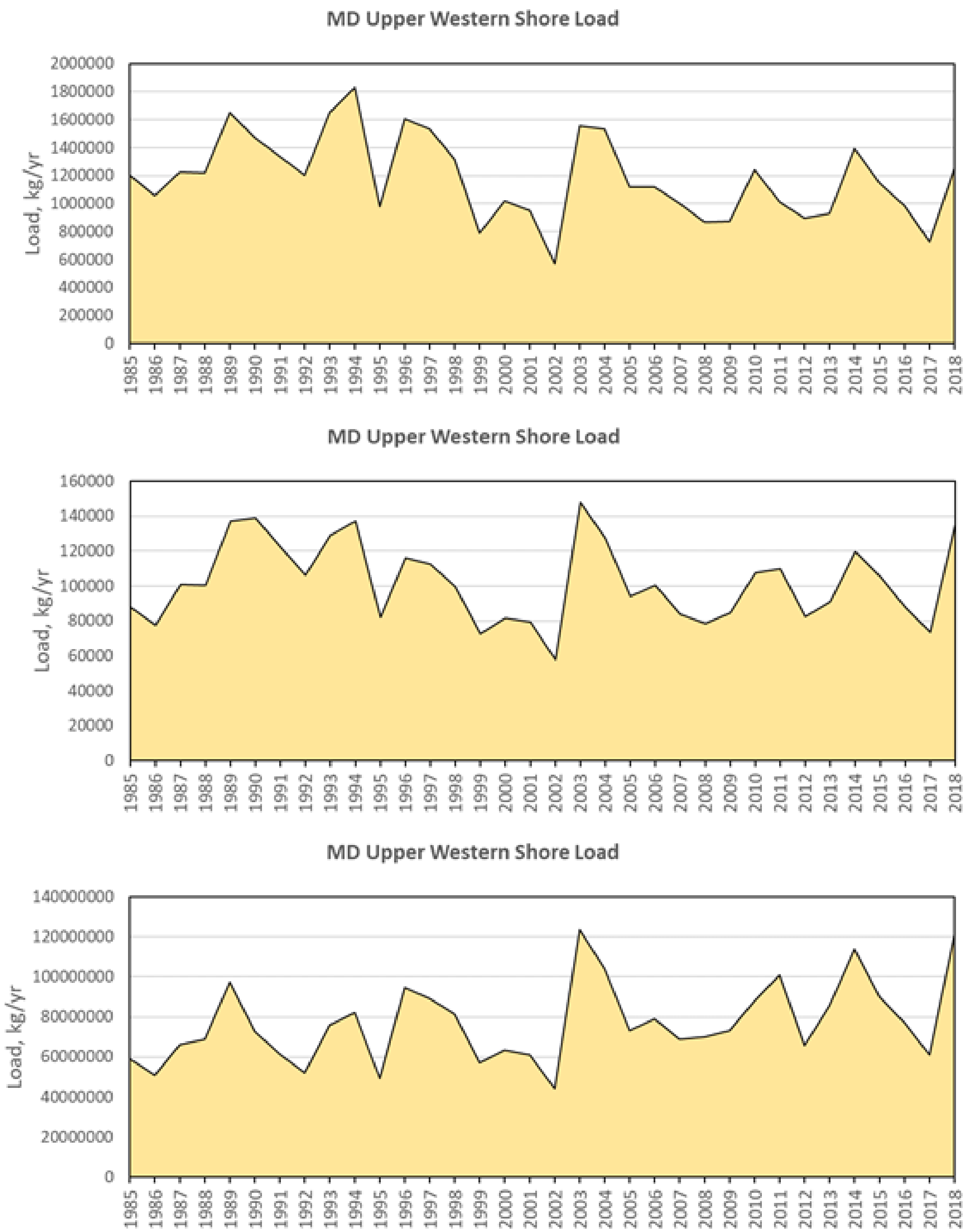

Figure 19. Estimated total loads of nitrogen (TN), phosphorus (TP), and suspended sediment (SS) to the Maryland Upper Western Shore Tributaries.

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 Maryland Upper Western Shore watershed.

Variable	Trend, metric ton/yr	Trend p-value
TN		
Total watershed[1]	-11	< 0.05
Point source	-0.84	< 0.01
Nonpoint source[2]	-5.7	0.19
Tidal deposition	-0.60	< 0.01
TP		
Total watershed	-0.40	0.42
Point source	-0.15	< 0.01
Nonpoint source	0.31	0.48
SS		
Total watershed	849	< 0.05
Point source	-0.27	< 0.01
Nonpoint source	849	< 0.05

[1] Loads from the different sources were obtained from the Chesapeake Bay Program Watershed Model progress runs specific to each year from 1985 and 2018, (https://cast.chesapeakebay.net/).

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

5.1.3 Expected Effects of Changing Watershed Conditions

According to the Chesapeake Bay Program's Watershed Model known as the Chesapeake Assessment Scenario Tool (CAST; https://cast.chesapeakebay.net, version CAST-2019), changes in population size, land use, and pollution management controls between 1985 and 2019 would be expected to change long-term average nitrogen, phosphorus, and sediment loads to the tidal Maryland Upper Western Shore River by -20%, -18%, and -3%, 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 (-37%), natural (-10%), stream bed and bank (-11%), and wastewater (-77%) sources were partially counteracted by increases from developed (48%) and septic (37%) sources.

The two largest sources of phosphorus loads as of 2019 were the wastewater and developed sectors. Overall, expected declines from agriculture (-76%), natural (-18%), and stream bed and bank (-42%) sources were partially counteracted by increases from developed (22%) and wastewater (5%) sources.

For sediment, the largest sources are shoreline and stream bed and bank areas: these two sources changed by 0% and 1%, respectively between 1985 and 2019. Sediment loads from the agriculture sector changed by -62%, whereas sediment load from developed areas changed by 53%.

Overall, changing watershed conditions are expected to result in the agriculture and natural 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.

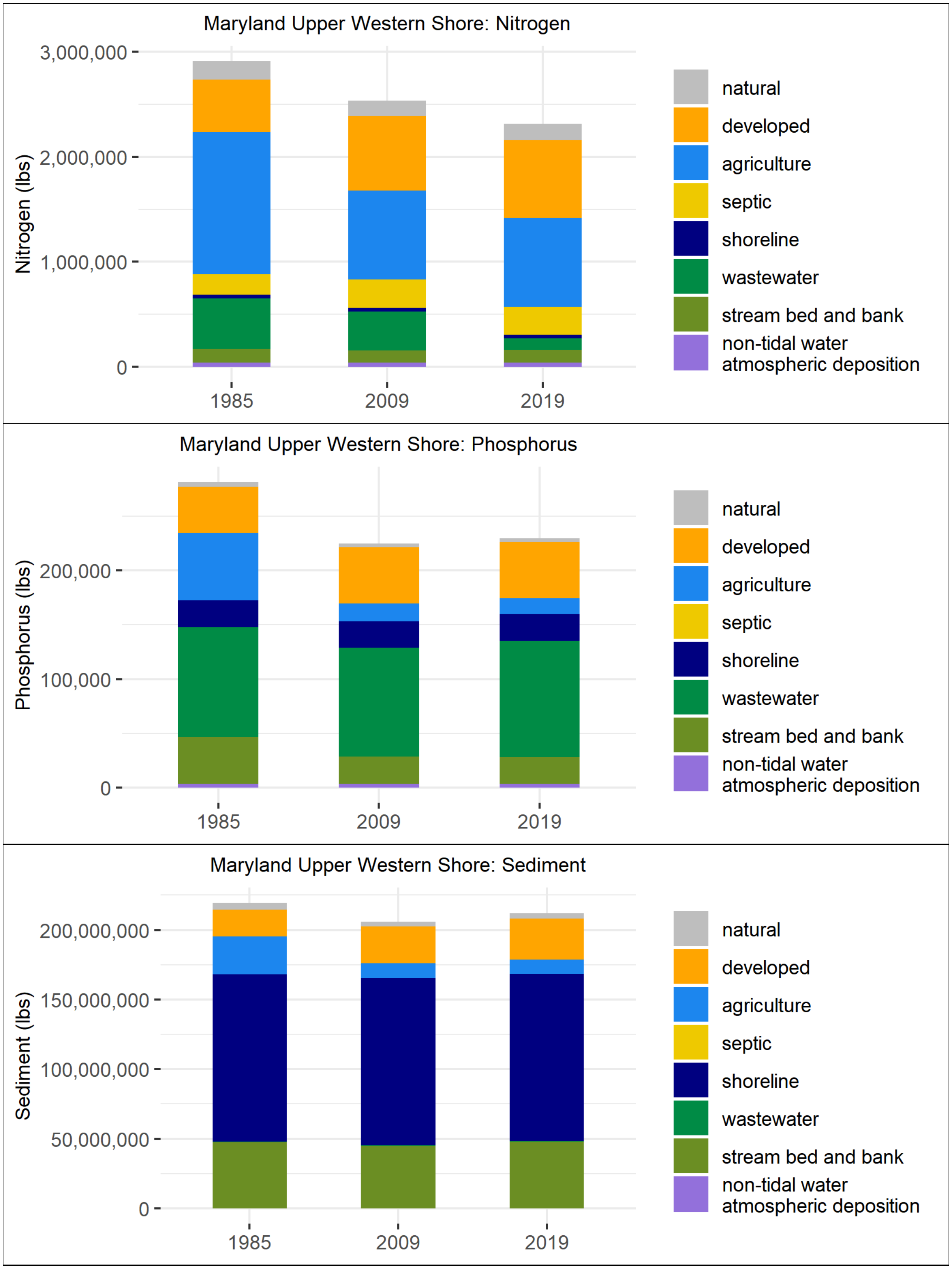

Figure 20. Expected changes in nitrogen, phosphorus, and sediment loads from different sources to the tidal portions of Maryland Upper Western Shore rivers, as obtained from the Chesapeake Assessment Scenario Tool (CAST). Data shown are time-average delivered loads over the average hydrology of 1991-

2000, once the steady state is reached for the conditions on the ground, as obtained from the 1985, 2009, and 2018 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 38, 16, 3.7, 0.1, 10, 104, and 71 thousand acres, respectively. Implementation levels for some practices are already close to achieving their planned 2025 levels: for example, 150% of planned acres for pasture management had been achieved as of 2019. In contrast, about 39% of planned stormwater management implementation had been achieved as of 2019.

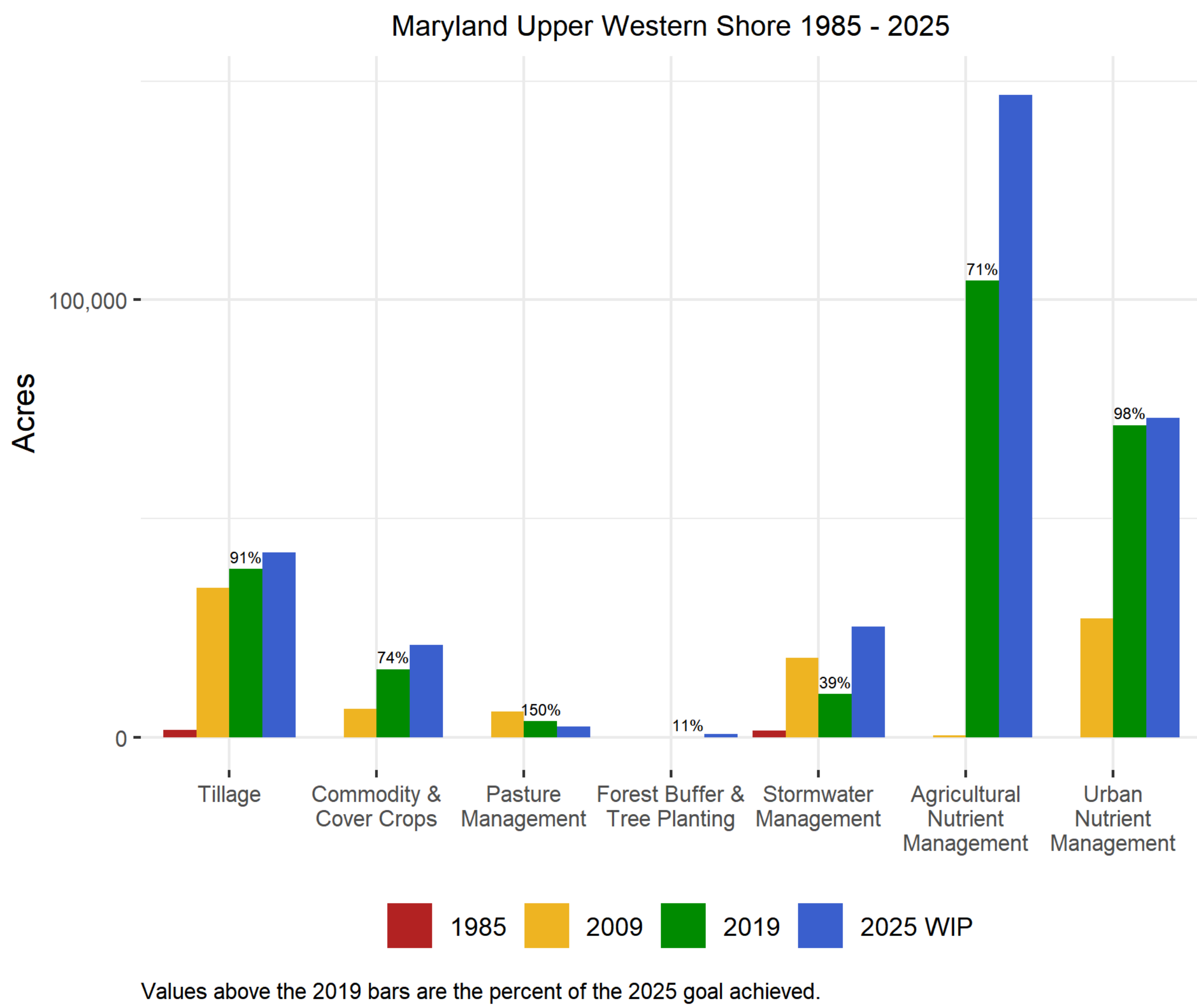

Figure 21. BMP implementation in the Maryland Upper Western Shore watershed

Stream restoration and animal waste management system systems are two important BMPs that cannot be compared directly with those above because they are measured in different units. However, progress towards implementation goals can still be documented. Stream restoration (agricultural and urban) had increased from 0 feet in 1985 to 24,626 feet in 2019. Over the same period, animal waste management systems treated 192 animal units in 1985 and 4,355 animal units in 2019 (one animal unit represents 1,000 pounds of live animal). These implementation levels represent 6% and 36% 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 the nontidal network monitoring location in the Maryland Upper Western Shore watershed. A more detailed summary of flow-normalized loads and trends measured at all USGS Chesapeake Bay Nontidal Network stations can be found at https://cbrim.er.usgs.gov/summary.html.

USGS Station ID	USGS Station Name	Trend start water year	Percent change in FN load, through water year 2018		
			TN	TP	SS
01582500	GUNPOWDER FALLS AT GLENCOE, MD	1985	**6.7**	-	-
		2009	**-6.6**	**65.7**	**134.0**

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

5.2 Tidal Factors

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

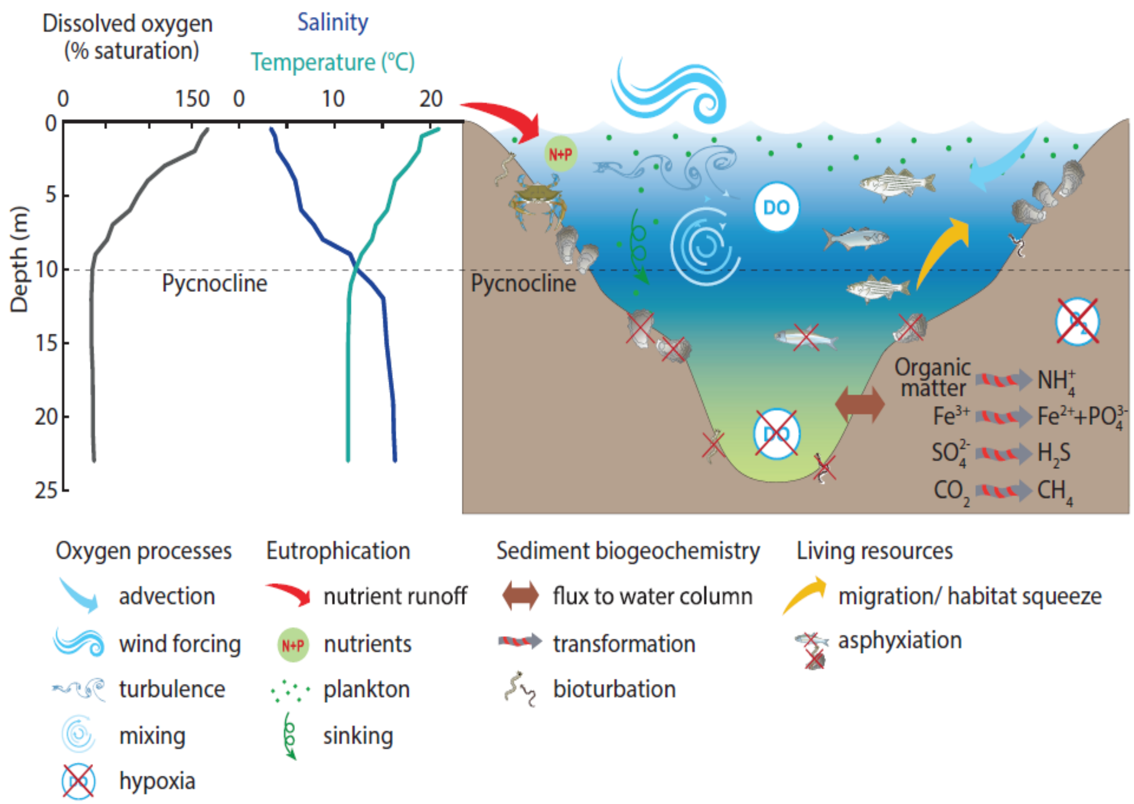

Figure 22. Conceptual diagram illustrating how hypoxia is driven by eutrophication and physical forcing, while affecting sediment biogeochemistry and living resources. From Testa *et al.* (2017).

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

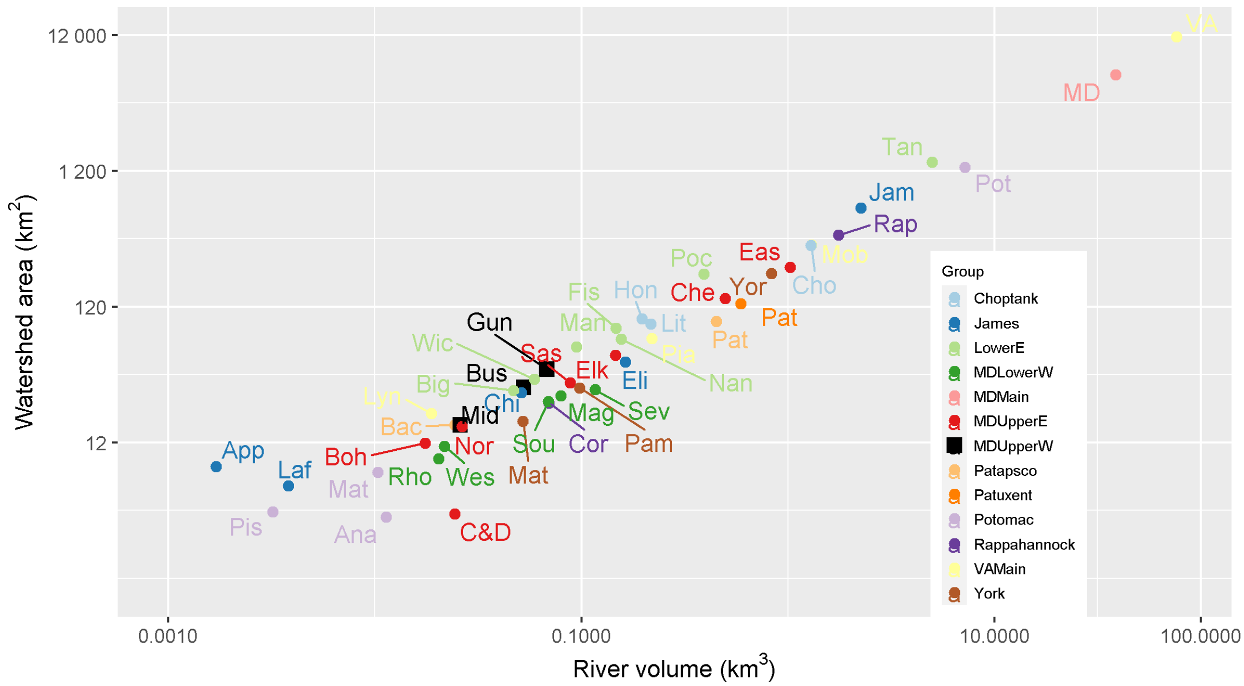

Figure 23. Watershed area vs estuarine volume.

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

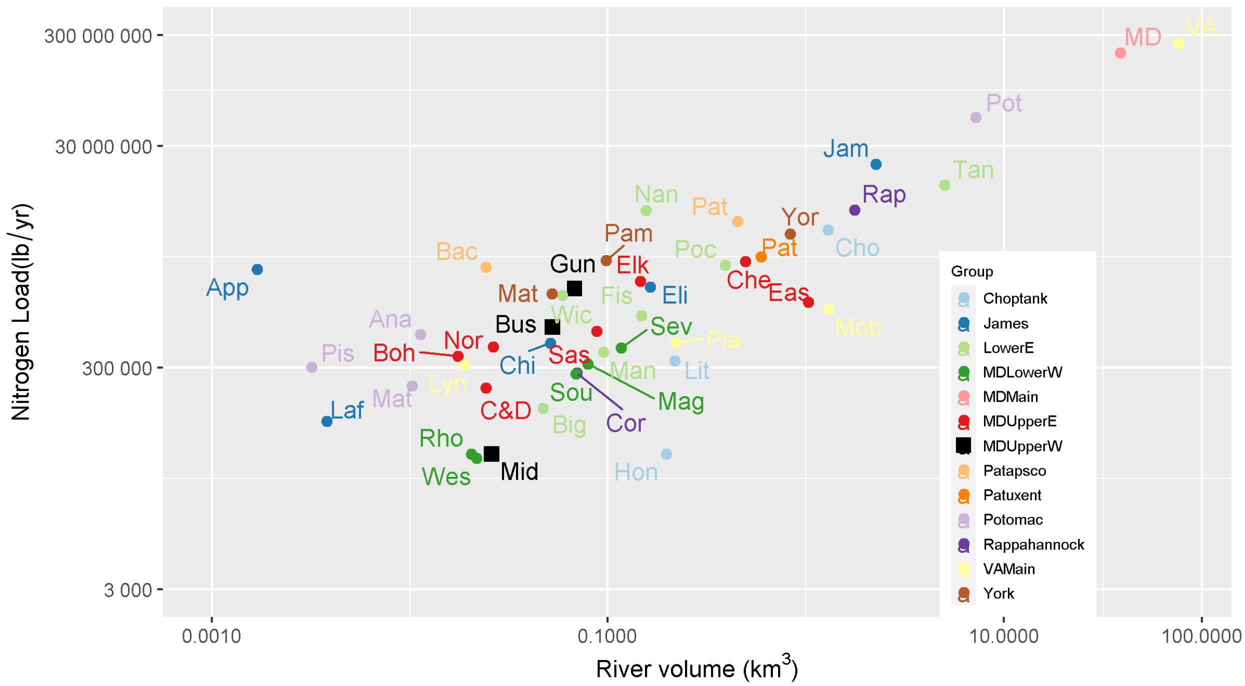

Figure 24. Annual average expected nitrogen loads versus estuarine volume. Nitrogen loads are from the 2018 progress scenarios in CAST (Chesapeake Bay Program, 2020), which is an estimate of nitrogen loads under long-term average hydrology given land use and reported management as of 2018.

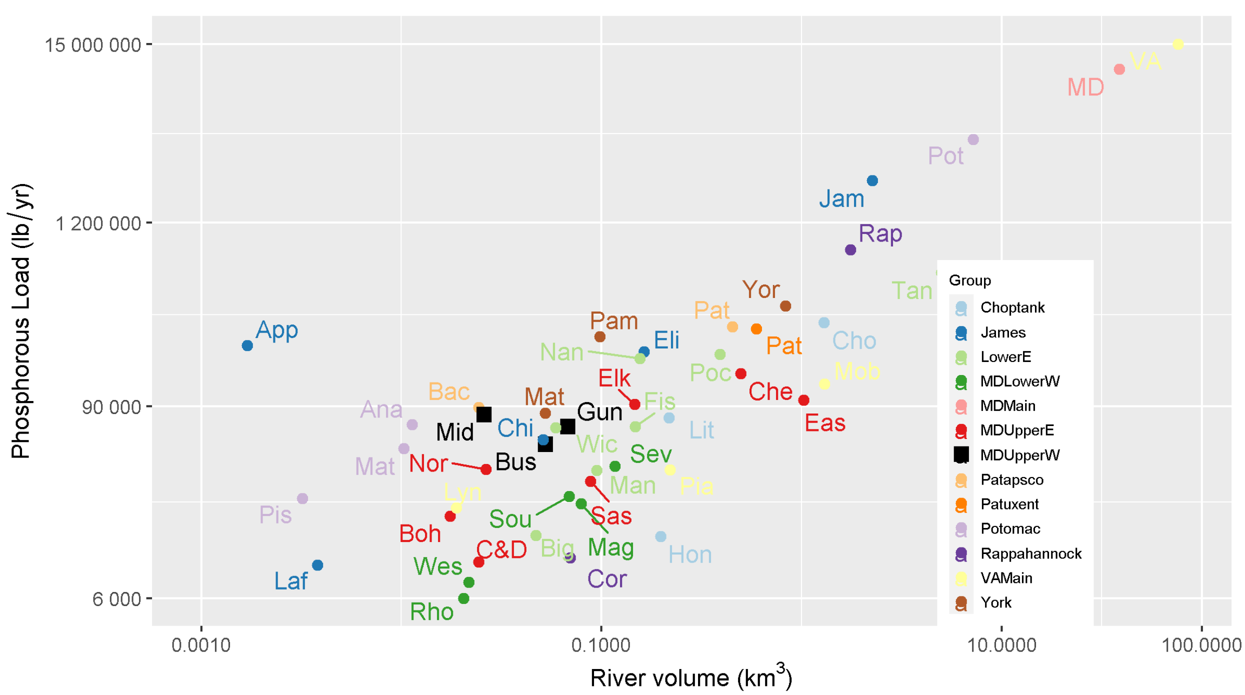

Figure 25. Annual average expected phosphorus loads versus estuarine volume. Phosphorus loads are from the 2018 progress scenarios in CAST (Chesapeake Bay Program, 2020), which is an estimate of

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

The Maryland Upper Western Shore estuary volume and watershed contain approximately 0.2 and 0.9% of the total volume and watershed of the Chesapeake Bay. This ranks the Maryland Upper Western Shore as the 13th largest volume and 12th 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 Maryland Western Shore system, the Bush, Gunpowder and Middle all drive this trend. The Middle river has an elevated load of phosphorus relative to their estuarine volume. Middle river has a lower relative load of nitrogen while the Bush and Gunpowder river loads are more moderate.

5.3 Insights on Changes in the Upper Western Shore

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

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

6. Summary

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

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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