Progress toward the Restoration of Chesapeake Bay in Time and Space

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Summary

Three decades of monitoring in Chesapeake Bay and tributary rivers has allowed for an examination of the spatial and temporal patterns of water quality change in response to watershed restoration activities. This review of past monitoring data has revealed clear signs of successful water quality remediation in some Chesapeake regions. Upgrades to waste water treatment plants (WWTP) have led to measurable reductions in nutrient concentrations and algal biomass with associated recoveries of submerged aquatic vegetation and reductions in sediment and nutrient levels. Point-source related improvements were observed in waters local to the WWTP facility, which are generally in oligohaline and tidal freshwater regions of tributaries. Reductions in atmospheric deposition of nitrogen within the Bay watershed has resulted in marked reductions in nitrogen inputs from the Susquehanna and several other tributaires, and these reductions in watershed input have resulted in lower concentrations within the estuary. Coastal plain watersheds with high agricultural intensity continue to yield high amounts of nutrients, and water quality has not improved in the receiving waters of many of these tributaries. Signs of eutrophication remediation are clearest where nutrient load reductions were large and local. In more seaward estuarine reaches, recovery from eutrophication appears to be season- and region-specific, where the late growing season period in high-salinity waters, which is most vulnerable to nutrient limitation and oxygen replenishment, appear to have recovered first. These findings suggest a refinement of our existing conceptual models of the eutrophication process in Chesapeake Bay, where time of year and proximity to nutrient sources are important to understanding spatial and temporal variation in recovery.

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1. Introduction

History of Water Quality and Habitat Degradation: During the last four centuries the Chesapeake Bay ecosystem has changed dramatically, and these changes, some subtle and others very large, began as far back in time as the 17^{th} and 18^{th} centuries. Examination of sediment cores, which capture past events in the Bay, indicate increasing sedimentation rates due to land clearance for agricultural purposes and increased burial of silica-rich diatoms, an early indicator of eutrophication. The sediment record also shows evidence of eutrophication trends in the late 19^{th} century when the ratio of pelagic (centric) to pinnate (benthic) diatoms increased, suggesting an increase in water column turbidity. Direct measurements of water clarity (using a Secchi disk) made between 1940 and 1970s indicate a sharp increase in turbidity, especially during spring and summer months. Phytoplankton biomass (as chlorophyll-a) has also increased post-1950, where increases were especially large in the meso and polyhaline regions of the bay. This increase was correlated with increased nutrient loads to the Bay and there has been a shift in phytoplankton community composition towards smaller cells and a general increase in harmful algal blooms (HABs) often associated with eutrophic conditions.

The sediment record also suggests wide-spread summer season bottom water hypoxia and anoxia in the deeper areas of the bay and tributary rivers was a more recent phenomenon beginning during the mid-20^{th} century. Prior to this period, direct measurements of hypoxia were made in the early 1900s in the Potomac and in the 1930s in Chesapeake Bay. Hypoxia and anoxia became chronic seasonal features of the Bay by the early 1970s and exhibited large inter-annual variability associated with changes in stratification strength and nutrient loadings caused by differences in freshwater flow rates that occur during wet and dry years. Variations in prevailing summer and winter wind conditions also impacted hypoxic volume by altering circulation and vertical mixing that either replenish bottom-waters with – or deprive them of – dissolved oxygen. In addition to the large, seasonal hypoxic volume in the mainstem of Chesapeake Bay, larger tributaries were also found to experience seasonal hypoxia (e.g., Potomac, York, Patuxent), while many smaller, shallow and nutrient-rich habitats experienced hypoxia, sometimes severe, at night.

Impacts on living resources have also been associated with Bay eutrophication. A formerly diverse seagrass community was dramatically affected by both nutrient load increases beginning in the mid-20^{th} century and associated reductions in water clarity. Seagrasses disappeared from the upper Potomac estuary during the 1940s and were gone from the Patuxent by 1970. In the upper Bay seagrasses also began to decline in the 1940s, experienced a bloom of an invasive species during the early 1960s and then largely disappeared by the mid-1970s, in part due to the dramatic scour and sediment loading associated with Hurricane Agnes in 1972. Benthic communities in deeper regions of the Bay have been severely depleted with loss of longer-lived species and large declines in benthic biomass. Multi-decade fisheries data suggest a gradual shift in catch composition from demersal species to more pelagic species and a general reduction in trophic transfer efficiencies in the estuarine food web.

Finally, it has become clear that both the degradation and restoration phases of eutrophication are not linear ecosystem-level processes, but rather have a complex assemblage of positive and negative feedback mechanisms that can serve to accelerate either restoration processes or eutrophication severity.

Chesapeake Bay Restoration Plan and the TMDL: The Chesapeake Bay Program has a 40 year history beginning in the late 1970s when Maryland's U.S. Senator Charles Mathias secured funding for a five-year study to analyze the Bay's rapid loss of high quality habitat with particular emphasis on effects of nutrient additions on Bay ecology, loss of seagrasses and distribution of toxic contaminants. The study identified excessive nutrient inputs as the main source of the Bay's degradation. These initial research findings led to the formation of the Chesapeake Bay Program as the means to restore the Bay.

The original 1983 Chesapeake Bay Agreement was a simple, one-page document that recognized a cooperative approach was necessary to address the Bay's pollution problems and it importantly was a Federal-regional approach with leadership including the governors of Maryland, Pennsylvania and Virginia, the mayor of the District of Columbia, the administrator of the U.S. Environmental Protection Agency (EPA), and the chair of the Chesapeake Bay Commission. Importantly, a comprehensive estuarine monitoring program was developed at this time and included monitoring of major river inputs of nutrients and other pollutants, Bay water quality, habitat and living resources. This program, with modifications, remains in place today and serves as the "gold standard" for estuarine status and trend analyses.

The first agreement was followed by the 1987 Chesapeake Bay Agreement where numeric goals were first established to reduce nutrient loading rates to the Bay by 40 %. In 2000, a more comprehensive document was developed, called Chesapeake 2000, with over 100 goals included to reduce pollution, restore habitats, protect living resources, promote sound land use practices and engage the public in Bay restoration through 2010. Also in 2000, the Bay's "headwater states" of Delaware, New York and West Virginia joined the Bay Program's restoration efforts. While restoration progress was made in some areas, attempts at significant nutrient load reductions were less widespread.

In 2010, the EPA established the landmark Chesapeake Bay Total Maximum Daily Load (TMDL). The TMDL transitioned the restoration program from largely voluntary and collaborative to a required "pollution diet" that sets limits on the amount of nutrients and sediment that can enter the Bay and its tidal rivers to ensure that water quality goals can be met. All seven Bay jurisdictions developed a Watershed Implementation Plan (WIP) with detailed and specific steps each jurisdiction must take to meet pollution reduction goals by 2025. Federal, state and local governments all participated in developing the WIPs. The WIPs guide Bay restoration efforts through the next decade and beyond and jurisdictions will use two-year milestones to track and assess progress toward completing the restoration actions in their WIPs.

The most recent agreement was signed by all the watershed states in June 2014 (Chesapeake Bay Watershed Agreement). This agreement contains ten interrelated goals aimed at restoration and protection of the Bay, its tributaries and surrounding watersheds. This agreement required a mid-point assessment of progress toward restoration goals due at the end of 2017. Toward that goal, this white paper serves to review the status of change in the Chesapeake Bay and its tributaries with respect to restoration efforts within the watershed, complementing the assessment of progress in the implementation of restoration projects.

Conceptual Model of Eutrophication: Water-quality restoration is the tool used by the US EPA to restore Chesapeake waters to "swimmable" and "fishable" conditions, as mandated by Section 101{a}{2} of the Clean Water Act. Nutrient and sediment inputs to Chesapeake waters compromise the ability to swim or fish by (a) stimulating Harmful Algal Blooms (HABs) that kill fish or have adverse effects on humans and wildlife, (b) generating hypoxia that damages food webs, kills fish, or limits fish habitat, and (c) limiting submerged aquatic vegetation (seagrass or SAV) habitat that provides a refuge and food source for many important estuarine organisms. Linkages between these conditions and nutrients loads are an integral part of our scientific understanding of the estuary, as well as the conceptual models used to organize knowledge of tidal water ecosystems.

Restoration activities in Chesapeake Bay have been geared toward reductions in watershed inputs of sediments, nitrogen, and phosphorus to the estuary. Research documenting quantitative relationships between nutrient availability and the accumulation of algal biomass led to the development of a conceptual model that linked watershed nutrient inputs to elevated algal biomass that sinks to deep waters at high rates, and whose decomposition leads to oxygen depletion (hypoxia) or the near-complete absence of oxygen (anoxia). This model has been reinforced by the association of growing hypoxic zones around the world with regions of high human-impact on the landscape and associated high nutrient loads to estuaries. Reductions in suitable habitat for SAV are also included in this model, where high nutrient concentrations stimulate high algal biomass in the water column and on SAV leaves, which in combination with elevated sediment loads, limit light availability for SAV growth and survival.

This "bottom-up" perspective recognizes water quality as a control of algal biomass by processes related to nutrient availability. There are also clear controls on algal biomass by removal via "top-down" processes associated with grazing by higher trophic levels. Water-column organisms, such as microscopic ciliates, larger zooplankton ('the bugs of the sea'), jellyfish, and filter feeding fishes (e.g., menhaden) form a complex network of feeding that consume algae and can thus influence algal biomass. Benthic organisms, such as oysters, clams, worms, and other benthic invertebrates filter algae out of the water column and, when abundant, can keep algae at low levels. One of the biggest challenges in understanding long-term patterns of algal biomass is unraveling the relative, and possibly changing, importance of such factors as grazing, nutrient inputs or other processes controlling algal biomass.

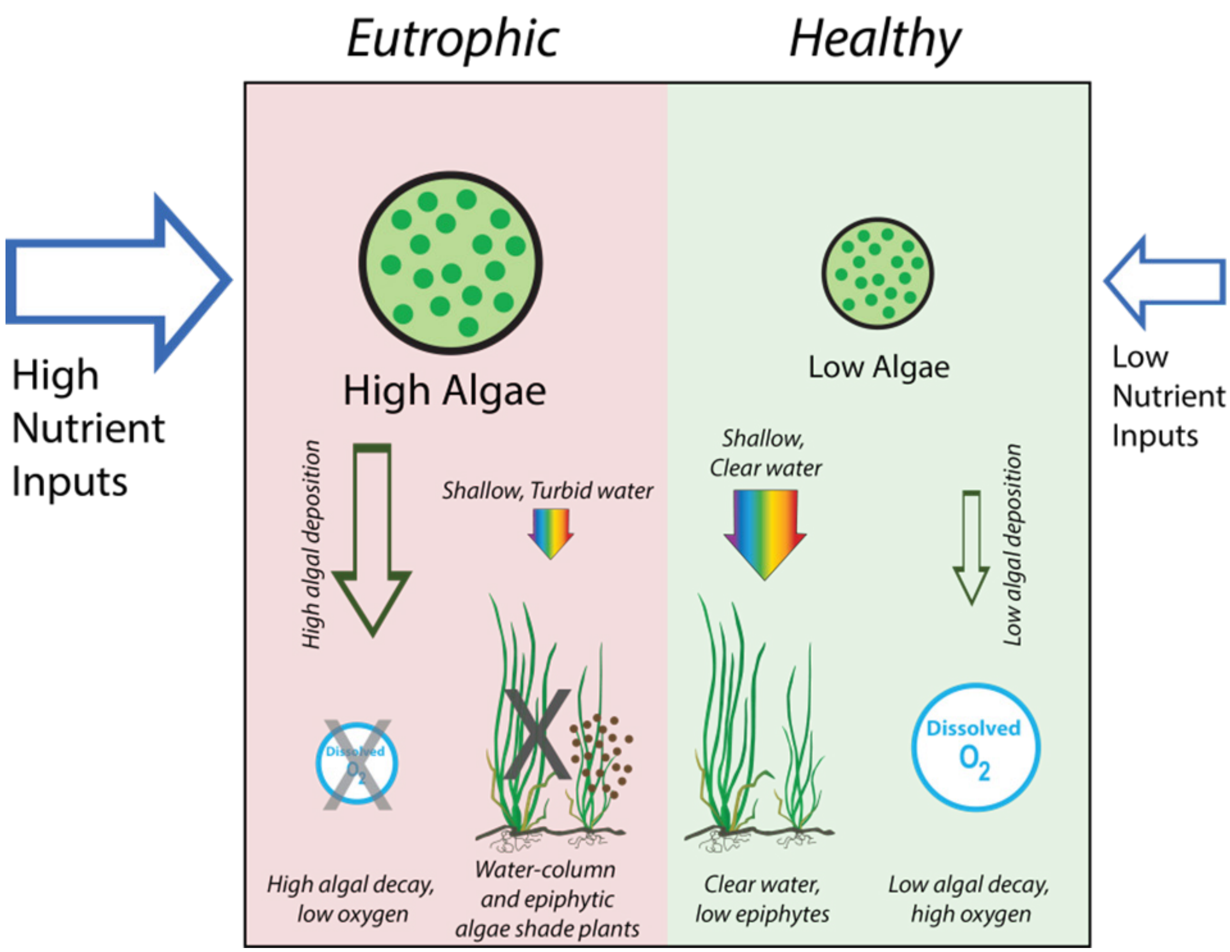

Figure 1: Simple conceptual diagram that illustrates the primary degradation factors for water quality that result from eutrophic (nutrient-rich) conditions in contrast to a "healthy" (low-nutrient) ecosystem. While many other parts of the ecosystem are impacted by eutrophication, the processes illustrated above are of most relevance to Chesapeake Bay water quality management.

Physical processes, such as winds, sunlight, precipitation, circulation, and temperature are also represented in our conceptual models of the Chesapeake estuary. These forces influence the estuary in the short-term on the scale of days to weeks ('weather'), in the medium term on the scale of seasons (months), and in the longer term on the scale of years, decades, or centuries 'climate'). Each of these physical processes influences either the delivery of sediment and nutrients to the estuary or the manifestation of negative effects caused by sediments and nutrients and all of these factors interact in significant ways. For example:

- Water clarity and light availability is the ultimate driver of the photosynthesis that supports algae growth, but precipitation leads to more nutrient and sediment delivery to the estuary (either stimulating or repressing photosynthesis).
- Winds and tides mix the water to force algae into deeper, darker waters (and limit photosynthesis), but also mix nutrient-rich water from deep basins into the shallow regions (to stimulate photosynthesis).

- High riverine inputs deliver nutrients to increase algal growth, but they also reduce the residence time of water and algae in the estuary, physically flush algae away, and deliver sediments that reduce light availability to algae.
- The ability of water to hold oxygen gas declines at higher temperatures, while the bacterial processes that consume oxygen are stimulated at high temperatures.
- Strong winds and tides can mix estuarine waters and deliver atmospheric oxygen to deeper waters (relieving hypoxia), while weak winds or the absence of strong tidal mixing allows for stability in the water and continued oxygen depletion.
- Strong riverine flows increase the physical separation of surface and bottom waters to support hypoxia development, but large flows can also stimulate strong up-Bay flows in deep waters that replenish oxygen levels.

A rich scientific literature exists to describe these complex and dynamic interactions and feedbacks between physical, chemical, and biological processes in estuaries that influence water quality (see Section 7).

2. Factors controlling patterns of estuarine water quality

Estuarine water quality (i.e., dissolved oxygen concentrations, turbidity/clarity, hypoxic volume, nutrient concentrations, algal blooms) varies as a complex function of pollutant type and loading rates, geophysical factors, biological factors, and climatic factors - the latter operating at timescales from intra-seasonal to multi-decadal. In this section we consider how spatial and temporal patterns in estuarine water quality are influenced by (a) watershed processes, (b) climatic effects, and (c) estuarine processes.

Watershed factors: Here, we focus on *nutrient* delivery from watersheds to Chesapeake Bay tidal waters; a separate effort is addressing *sediment* inputs and associated water clarity. The primary driver of variation in total nitrogen (TN) and phosphorus (TP) outputs from watersheds to adjacent estuarine water bodies is linked to the magnitude of nutrient input on land (Nixon et al. 1996). This pattern holds when comparing across watersheds regionally or globally, but locally the primary factor for delivering nutrients on a seasonal or yearly basis is the amount of precipitation that falls in the watershed and is transported to the estuary via streams and rivers. The area of a watershed can be used as a proxy for this inflow rate when considering watersheds/estuaries with similar hydroclimatic conditions, thus larger watersheds tend to yield larger inputs of freshwater and nutrients to estuaries, given similar land-uses. These facts hold true for one of the two general sources of nutrients to estuaries, called *non-point* sources, which include inputs that are spread out across the watershed, such as atmospheric deposition, fertilizer and manure applications, and the fixation of nitrogen gas into dissolved (useable) forms. The other category of nutrient input is called *point-source*, which includes direct discharges of nutrients to estuaries from municipal wastewater treatment facilities and commercial entities. Given that most non-point and point-sources of nutrients arise from human activity, it is not surprising that variations in annual riverine TN loads across the North Atlantic region were explained by human population density (Peierls et al. 1991) and the activities associated with human populations, agriculture, and industry (Howarth et al. 1996).

There are multiple pathways by which *non-point* nutrient inputs can be transported to the estuary, including (1) leaching of dissolved nutrients directly to surface waters, (2) "delayed" transport of leached nutrients to groundwater and later to surface waters, (3) incorporation into crops, consumption by livestock and humans, and export to streams and rivers as waste products, (4) erosion of nutrient-laden sediments from land into streams and rivers, and (5) volatilization (conversion to gas) of nutrients from waste products, followed by redeposition onto the land surface, and either direct or delayed transport to surface waters. The relative importance of these factors is different for nitrogen and phosphorous, where mechanisms #1, 2, 3, and 5 are important for nitrogen, but only #1 and 4 are important for phosphorus. Nutrients delivered to watersheds via atmospheric deposition following fossil fuel combustion are an important non-point source to watersheds, especially for nitrogen. Howarth et al. (1996) found that atmospheric nitrate (a key dissolved form of nitrogen) deposition was strongly and linearly correlated with riverine total nitrogen inputs, but more recent analyses using data from many other watersheds revealed exponential relationships between nitrate deposition and TN loads—suggesting that leakage of nitrate from watersheds may increase exponentially in respond to higher nitrate deposition (Howarth et al. 2012; Eshleman and Sabo 2016).

Chesapeake Bay has a watershed comprised of eleven hydrogeomorphic regions (Bachman et al. 1998) draining to nine major tributaries and many more small creeks and embayments, all combining into a mainstem that stretches about 200 miles from its freshwater origins near Havre de Grace, MD to the Atlantic ocean near Virginia Beach. Clearly, there is much variability even within Chesapeake Bay with respect to the relative importance of changes in land use, population size, implementation of management practices, and associated nutrient inputs. The geomorphology of a given drainage basin controls the amount of nutrients and sediment reaching tidal waters and how quickly they get there (Ator and Garcia 2016, Ator and Denver 2015). While these differences within the watershed create a challenge for a "one-size fits all" management approach, consistent relationships between nutrient inputs, physical forces, and biological and chemical processes allow for similar management measures across tributaries.

Climatic Factors: Seasonal, inter-annual, and long-term variability in weather and climate patterns mediate the relationship between nutrient and sediment inputs from the watershed and estuarine productivity and hypoxia. While our understanding of these patterns and their drivers is still evolving, it is clear that seasonal and inter-annual variability in temperature, precipitation, and wind conditions mediate the effects of nutrient and sediment inputs from the watershed on estuarine dissolved oxygen, nutrient, and chlorophyll a concentrations, as well as water clarity and the living resources whose health and productivity are dependent upon water quality. While we summarize these interactions in the conceptual models described above in Section 1, we review a variety of important climatic factors in the section below.

There are a variety of climatic patterns, cycles, and long-term trends that influence the coupled watershed-estuarine ecosystem of Chesapeake Bay. The Bay resides in a temperate climate, where strong seasonal changes in temperature, sunlight, precipitation, and wind stress influence the major physical and biological dynamics of the estuary. Seasonal precipitation patterns generally result in a strong winter-spring peak in freshwater input the estuary, followed by a secondary peak in fall that may include large flow events associated with the passing of tropical storms. High winter-spring flows deliver the nutrients that support a typical spring peak in algal growth, which also corresponds to early season increases in the intensity of sunlight. These flows also interact with spring increases in temperature to generate early summer peaks in stratification – the layering of surface and bottom waters – that set the stage for seasonal low-oxygen zones in deep waters. Warm temperatures and low-wind speeds during summer allow for the maintenance of low-oxygen conditions in bottom waters, while associated peaks in sunlight allow for high algal growth in surface waters. In low-salinity regions of the tributaries, algal biomass often peaks in summer when residence times and light availability are higher. In fall, increasing wind speeds and surface cooling promote the mixing of deeper waters and the relief of hypoxia and the delivery of nutrients to surface waters that can stimulate fall blooms. Cold winter temperatures limit the rate of most biological processes, such as oxygen consumption, grazing on algae, and nutrient remineralization, but high rates of primary production and algal growth have been noted in some regions.

These typical seasonal phenomena occur along with longer-term changes in regional climate patterns. The first of these patterns include decade-scale climate cycles, such as the North Atlantic Oscillation (NAO), the Atlantic Multidecadal Oscillation (AMO), and the El Niño-Southern Oscillation (ENSO), which influence temperature, precipitation and other climatic variations over 10-30 cycles that shift between different "states". While the mechanistic linkages between these cycles and the climate and circulation of Chesapeake Bay are still not fully understood, recent analyses have associated large-scale climate cycles with freshwater inputs to Chesapeake Bay (ENSO), patterns of wind fields (NAO), salinity (NAO), and conditions in the adjacent Atlantic Ocean (AMO). Long-term trends in estuarine dynamics are predicted for Chesapeake Bay associated with fossil-CO_2 driven climate change (Johnson et al. 2016), but the net effect of these changes is uncertain. Elevated temperature is expected over the next century, which will decrease the solubility of oxygen in water, speed up rates of oxygen consumption, and increase stratification. Combined with expected increase in precipitation, it could be expected that hypoxic zones will increase. However, the impact of elevated temperatures on future algal growth and biomass is less clear, so the production of fuel that ultimately sustains hypoxia may not increase. Increases in sea level could increase stratification or vertical mixing (with opposing effects on hypoxia), and future wind patterns are difficult to predict, so the physical conditions that support hypoxia may be enhanced or decreased. In short, it is difficult to predict how climate cycles and long-term changes will impact Chesapeake Bay.

Estuarine factors: Estuary and watershed size, shape, geomorphology, and hydrodynamics play a fundamental role in shaping spatial and temporal patterns in estuarine water quality. While Chesapeake Bay is the largest estuary in the United States, its shallow nature, dendritic shoreline, and large watershed size results in a system that is very sensitive to increasing pressures of human activity across the landscape (Kemp et al. 2005). Chesapeake Bay tends to have a long residence time (6-12 months) and a circulation pattern that allows for trapping and recycling of watershed nutrients, leading to a naturally high productivity system that is easily pushed into a nutrient-enriched state. While tidal ranges in Chesapeake Bay are not exceptional, tidal mixing can be an important factor in the delivery of nutrients to surface waters and relief of bottom water oxygen conditions (Li and Zhong 2009). Although wind speed and direction vary greatly over space and time, the relatively shallow nature of Chesapeake Bay makes it particularly sensitive to wind-induced circulation and mixing that strongly influence algal growth and oxygen depletion/replenishment. Many recent papers have addressed the role of winter-spring winds in transporting algae to locations that favor oxygen depletion (Lee et al. 2013), the role of summer winds in either enhancing hypoxia (westerly) or reliving hypoxia (southerly; Du and Shen 2015, Scully 2010), the interaction of wind direction and bathymetry in driving estuarine circulation (Wang et al. 2016), and long-term changes in stratification (Murphy et al. 2011).

Estuarine water quality is also affected by location. Upper portions of tributaries, including tidal freshwaters, are closely coupled to inputs from the watershed and tend to be close to WWTPs, so there is a reason to conclude that these waters would be more susceptible to eutrophication. The same factors that deliver nutrients to these low-salinity regions and cause them to be nutrient-rich, however, also deliver sediments that restrict the light available for photosynthesis and the associated accumulation of algal biomass, at least during periods of high river flow. This latter point is underscored by the fact that low-salinity regions of the Chesapeake Bay mainstem and most large tributaries also have hydrodynamically-controlled high-turbidity regions (known as estuarine turbidity maximums, or ETMs), that serve as sediment trapping locations. In contrast, higher-salinity, downstream regions of the estuary that are less connected to watershed nutrient and sediment inputs tend to have lower nutrient concentrations and better light conditions, making them more vulnerable to nutrient limitation. Thus, two open questions related to the Bay's recovery are (1) Where should the impacts of eutrophication be most severe? and (2) Do we expect recovery from eutrophication to occur first in the high-impact, up-estuary regions, or rather in the relatively low-impact, but high-vulnerability down-estuary regions? One key aspect of this question is the extent to which WWTP nutrient inputs, which are largely discharged to up-estuary regions , will primarily impact local waters or will influence higher-salinity waters far downstream. Water quality in Chesapeake Bay has been dramatically altered by increased loading of nutrients and sediment due to human activities on the landscape, but here we seek to understand where watershed-estuary connections are most intense and what regions of the estuary appear to be recovering most rapidly. We address these questions via a basin-wide assessment of changes in watershed nutrient load and estuarine nutrient concentration in combination with case studies illustrating specific examples of pollution mitigation strategies and their impact on nutrient inputs and estuarine water quality.

3. Nutrient Loading and Concentration Changes in Chesapeake Bay

Inputs of nitrogen and phosphorus have been routinely monitored over the past several decades and these data offer insights into the relative importance of local sources (land areas adjacent to the estuary, hereafter Below Fall Line (BFL) loads) vs. distant sources (from watershed areas extending Above the Fall Line; AFL). These inputs determine nutrient availability within Chesapeake Bay (CB) and its tributaries, and associated water quality conditions. In order to evaluate Bay-wide changes in nutrient inputs, we compiled data from two primary sources: (a) tributary inputs *monitored* at the Fall Line of the nine major rivers entering the Bay, and (b) below Fall Line inputs *modeled* based on land use practices for watershed regions draining directly to estuarine waters, which are matched to the tidal water segments into which they drain. Below Fall Line inputs include point source discharges.

Spatial Pattern of Loading Changes: Direct measurements of total nitrogen inputs at nine regularly monitored stations indicate that total nitrogen inputs have declined Bay-wide, with seven stations showing declining inputs from 1985 to 2015 and only one station showing an increase (Choptank River). At the many stations monitored upstream of these nine sites, twice as many sites have shown improving trends in load (due to reduced in-river concentrations) as opposed to degrading trends (Moyer and Blomquist 2016). This contrasts with total phosphorus loads, where long-term declines are only apparent in four rivers (Potomac, James, Patuxent, Mattaponi), and four others (Choptank, Susquehanna, Appomattox, and Pamunkey) show increases (Moyer and Blomquist 2016). While total phosphorus inputs from many of the rivers have increased, dissolved, bioavailable phosphorus (the form of phosphorus directly available to algae) loads have generally declined across the Chesapeake Bay (Zhang et al. 2015). Additionally, over three times as many stations monitored upstream of the nine large river sites have shown phosphorus declines relative to increases (Moyer and Blomquist 2016), suggesting that a limited area the of watershed is responsible for elevated TP load. In many of the rivers, phosphorus increases are primarily associated with increases in the particulate fraction.

In order to quantify how nutrient inputs to Chesapeake Bay have changed during the past thirty years, we compiled estimates of measurement-based nutrient loads (TN and TP) from nine fall line monitoring stations (AFL; https://cbrim.er.usgs.gov/) with model-based estimates of nutrient inputs from the BFL watershed from the Chesapeake Bay Program Watershed Model Phase 6 draft simulations. The BFL estimates included model output for both point and non-point sources. Results from this analysis indicate that TN and TP loads to the Bay have declined by 27% and 23% (respectively) during the ~25-year span. Declines in AFL and BFL loads have contributed approximately equally to the overall change (Table 1), though proportional declines were greater for BFL inputs (TN=37%, TP=32%) than for AFL inputs (TN=21%, TP=17%). Reductions in point sources accounted for the bulk (>80%) of BFL load declines

Table 1. Annual average Above Fall Line (AFL) and Below Fall Line (BFL) loads to Chesapeake Bay during historical and recent periods (units are kg/y; multiply by $2.2e^{-6}$ for million lbs/yr). Statistical significance of loading trends based on paired t-tests (recent vs. historical; $p < 0.05$ is significant) for tributary monitoring stations (AFL loads) and segments (BFL loads).

	AFL		BFL (Total)		BFL Non-PS		BFL PS	
	TN	TP	TN	TP	TN	TP	TN	TP
1989-1991	95,073,299	5,076,407	58,114,831	3,720,981	29,177,948	2,156,943	28,936,884	1,564,038
2012-2014	74,731,516	4,234,346	36,487,174	2,527,940	25,230,005	1,900,122	11,257,169	627,818
Δ Annual Load	20,341,783	842,061	21,627,658	1,193,041	3,947,943	256,821	17,679,715	936,220
% Decline	21%	17%	37%	32%	14%	12%	61%	60%
p	0.13	0.15	0.010	0.001	0.001	0.004	0.039	0.001

Multiple factors are involved in observed nitrogen loading declines, but the two primary causes are reductions in atmospheric deposition associated with the Clean Air Act and improvements in sewage treatment processes at major wastewater treatment plants (WWTPs) in Chesapeake Bay. Atmospheric deposition has declined steadily since the late 1980s, while fertilizer and manure nitrogen applications on agricultural fields have remained fairly constant (Fig. 2). Crop yields have increased in many regions of the watershed, but we were not able to characterize the impact of crop uptake on the net nutrient balance for agriculture. Expensive efforts to upgrade sewage treatment processes to reduce nitrogen inputs have been highly successful, with substantially reduced nutrient inputs from major facilities in the James, Potomac, Patuxent, Back and Patapsco basins (Fig. 3; Testa et al. 2008, Boynton et al. 2014). As a consequence, TN concentrations in most CB rivers have declined, resulting in a lower dissolved nitrogen input at a given freshwater flow for some rivers (Fig. 2, bottom panel). Explanations for phosphorus loading changes are more complicated. While reduced phosphorus fertilizer applications, the phosphate detergent ban, removal of phosphorus at sewage treatment plants, and improved sediment

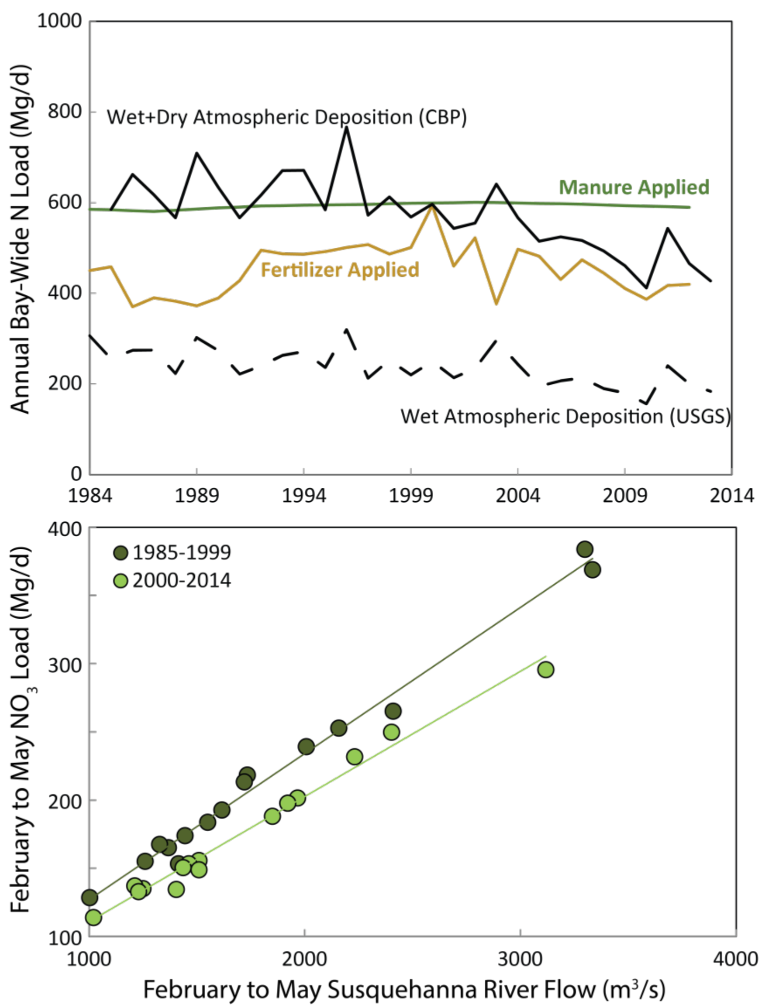

Figure 2: (top) N inputs to the Chesapeake Bay watershed from atmospheric deposition, fertilizer, and manure applications (data from Chesapeake Bay Program and USGS). Note: changes in crop uptake are not included in these values. (bottom) Relationship of February to April nitrate load versus discharge (flow) from the Susquehanna River over historic (1985-1999) and recent (2000-2015) timeframes.

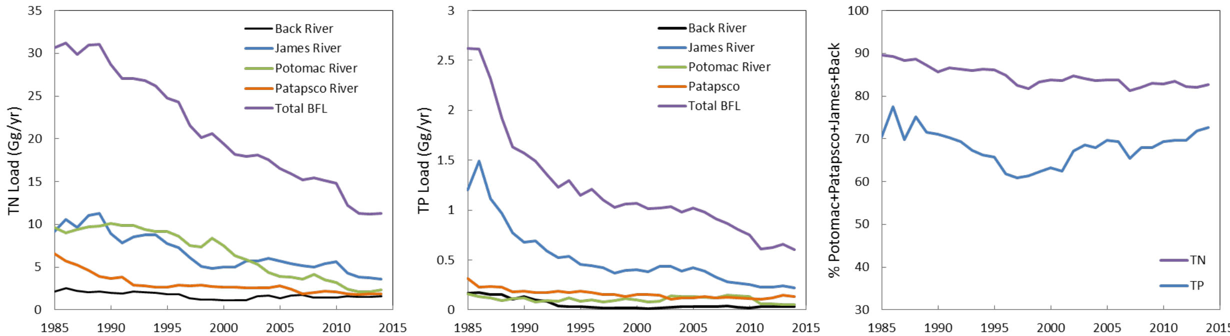

Figure 3: Long-term WWTP nutrient inputs to below fall line waters for TN (left) and TP (middle) for 4 major tributaries and the overall Chesapeake Bay. (right) %of total Bay WWTTP loads from the James, Potomac, Back and Patapsco Rivers.

management have helped reduced phosphorus load, other processes have led to elevated phosphorus inputs. Coastal plain watersheds, especially on the eastern shore of Maryland, have generally seen phosphorus loading inputs increase (Zhang et al. 2015), which have been associated with soil P saturation (Kleinman 2017) and other factors. In the case of the Susquehanna basin, the infilling of three dams near the mouth of the Susquehanna River has resulted in less trapping of sediment, leading to increased net export of sediment and total phosphorus (e.g., Zhang et al. 2015). A large fraction of the net phosphorus input increases are sediment-associated, and because a large fraction of sediments are deposited or trapped in the upper Chesapeake Bay (e.g., Sanford et al. 2001), their impact on overall Bay water quality is likely limited. Due to the intricate array of potential factors involved in phosphorus input changes associated with each watershed and the land use within it, each watershed must be analyzed individually to understand long-term change.

Linkages between Nutrient Loading and Availability: Our conceptual model assumes that there is a relationship between nutrient input to the estuary and nutrient concentrations within the estuary, and by extension, that when nutrient inputs go up or down, estuarine nutrient concentrations in receiving waters should follow. We sought to test the hypothesis that spatial variation in TN and TP concentration changes among estuarine monitoring stations could be explained by changes in TN and TP watershed loads that are aggregated into each of 92 CB segments (Fig. 4). To quantify long-term changes, we computed average annual loads (kg/y) and estuarine concentrations (mg/L) for two periods (1989-1991 and 2012-2014). These periods were selected to represent the historic and recent portions of the historical record and because they had similar average annual flows (means = 78,667 versus 79,100 cfs) that were representative of long-term average conditions (78,563 cfs; Moyer et al. 2016). Thus, we effectively removed the effects of variable river flow from the comparison of recent and historical loads. Concentrations of TN and TP in the estuary were derived from the Chesapeake Bay Monitoring program database and corrected for methodological and laboratory changes, while loading estimates were as described in the previous section. We used individual 1-way ANOVAs to identify segment loads and estuarine stations showing significant changes in nutrient loads and concentrations and we related those changes in load and concentration to each other statistically (direction of change) and quantitatively (magnitude of change) to determine if both load and concentration declined in a given region.

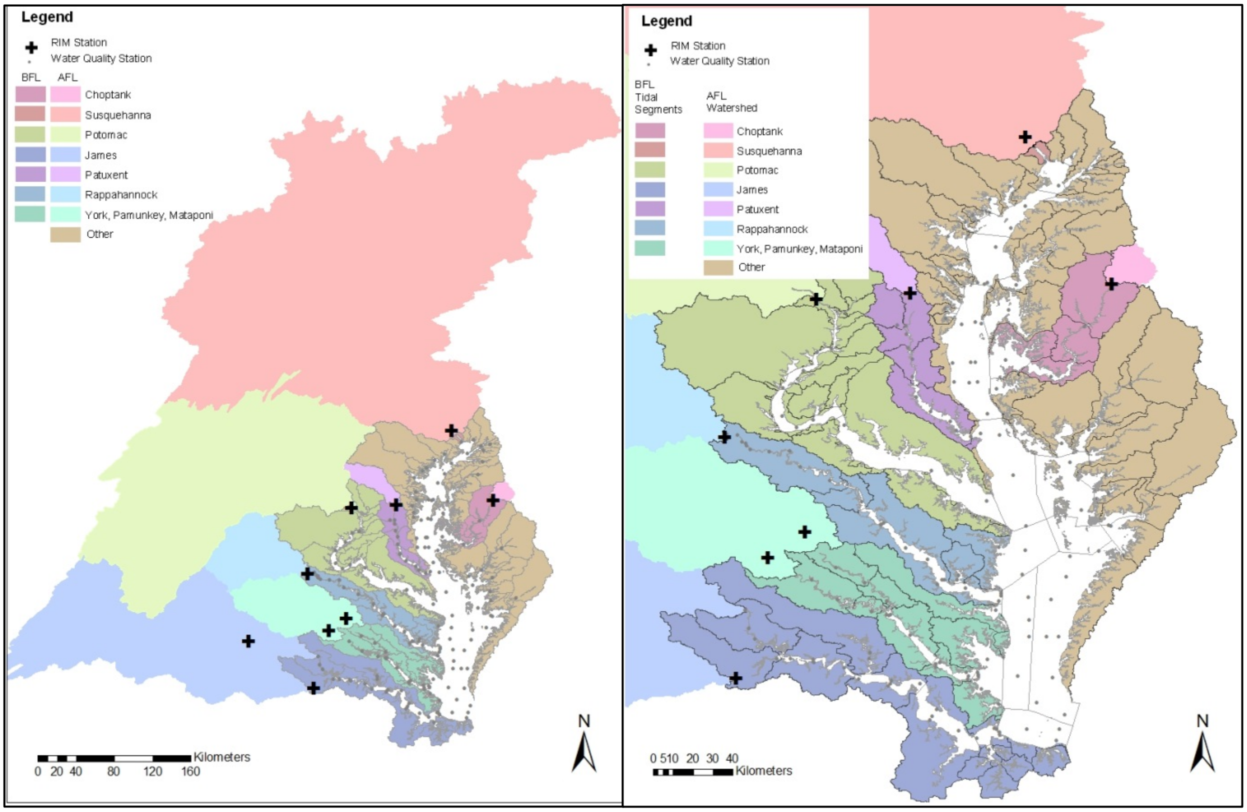

Figure 4: (left) Map of Chesapeake Bay watershed, with AFL and BFL watershed areas defined, as well as AFL monitoring stations (crosses), estuary monitoring stations. (right) Same map as left, but zoomed in to highlight boundaries of watershed segments where BFL l**o**adings were calculated.

The results show that the majority of sites exhibited significant decreases between the historic and recent BFL segment loads and estuarine concentrations (79% and 62% of all sites for TN and TP, respectively; Table 2, Fig. 5). Within this group, the average decline in segment loads (TN = 31%, TP = 34%) was comparable to the average decline in estuarine concentrations (TN = 34%, TP = 24%). A small subset of sites instead exhibited significant increases in both segment BFL load and estuarine concentration (8% and 4% for TN and TP, respectively). Combining these results, we can report that segment loading changes correctly predicted estuary changes in 88% (TN) and 66% (TP) of cases. Focusing on the subset of estuarine stations exhibiting significant declines in estuarine concentration (N = 116 and 61 for TN and TP, respectively), there was a high concordance with declines in segment loads (87% and 84% agreement for TN and TP, respectively). Instances of stations showing significant increased estuarine concentrations and loads were rare by comparison (<5 stations). There was only one instance where a significant increasing change was observed at a station where segment loads declined, but there were a number of stations showing declines in concentrations despite an increase in segment load.

Table 2. Comparison of changes in below Fall Line inputs of TN and TP with changes in estuarine concentrations of TN and TP across **all** stations monitoring in the receiving waters of Chesapeake Bay. Significant differences are noted for p-values less than 0.05) in the means comparison.

Segment Load Trend	Estuarine Conc. Trend	# Station TN	# Station TP	Mean Load Change (%) TN	Mean Load Change (%) TP	Mean Concentration Change (%) TN	Mean Concentration Change (%) TP	Sig. Trend (p<0.05) TN	Sig. Trend (p<0.05) TP
Decrease	Decrease	110	87	-31%	-34%	-34%	-24%	96	49
	Increase	3	26	-40%	-17%	30%	13%	1	0
Increase	Decrease	16	21	14%	12%	-28%	-17%	14	10
	Increase	11	6	12%	7%	21%	14%	5	2

We also determined whether changes among stations were related to changes in loading at the whole tributary scale. Input data for these analyses were average annual TN and TP loads (kg/y) inclusive of the AFL and BFL inputs. Segment-specific BFL loads (from the Bay watershed model; see Part 1) and tributary–specific AFL loads were combined by aggregating at the tributary scale. Recent (2012-14) and historical (1989-91) loads were derived for the Bay as a whole and for 8 tributaries: Choptank, James (including Appomattox), Patuxent, Potomac, Rappahannock, Susquehanna and York (including Pamunkey and Mattaponi). The % change in the combined AFL and BFL loads for each tributary was used as a predictor of changes among stations located within that tributary.

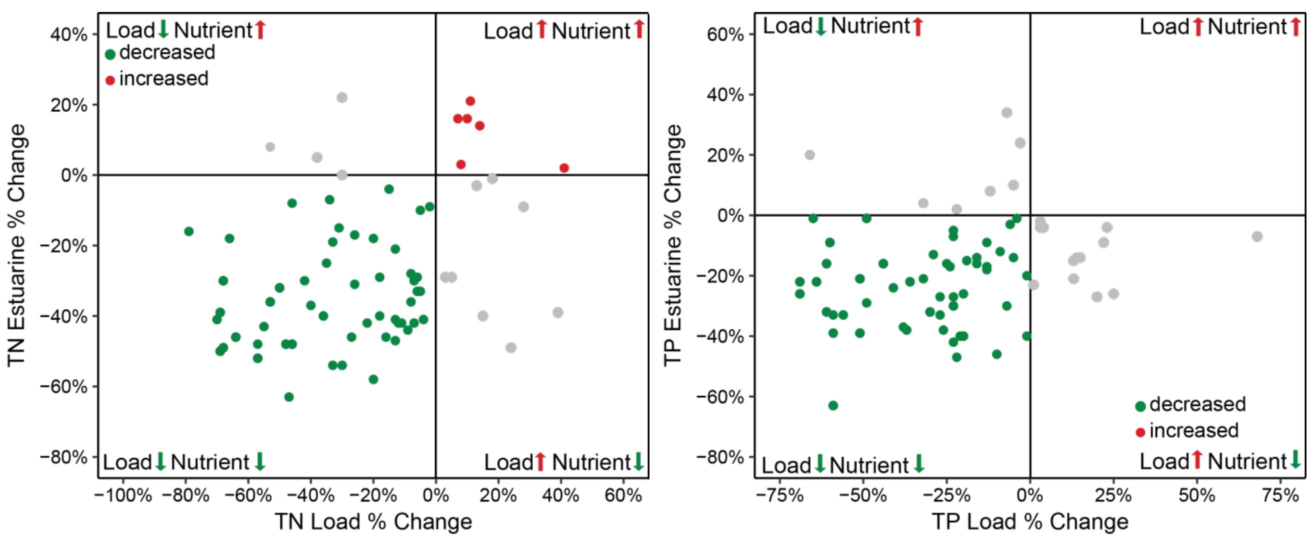

Figure 5. Variation in the percentage change in estuarine TN and TP concentrations as a function of the percentage change in segment BFL loads. Note that these data differ from Table 2, in that % changes for stations are aggregated to segments. Statistical analysis of these models is complicated by the fact that individual segments are represented by multiple stations, raising concerns about independence of observations (i.e., the dataset includes multiple *y* estimates associated with a single *x* value). Therefore no statistical assessment of changes is provided.

For TN, the largest combined reductions were observed in the James (-41%), Patuxent (-31%) and Susquehanna (-26%) tributaries (Fig. 6). Only the Rappahannock (+8%) exhibited an increase in TN loading during this interval. For the Bay overall, TN loads declined by 25%, with nearly equal contributions from declines in AFL and BFL loads. The 5 tributaries showing the largest declines in loads (James, Patuxent, Choptank, Susquehanna, Potomac) also exhibited the highest proportion of sites with statistically significant declines in TN concentrations (range = 50 to 100%) and the greatest proportional change in estuarine TN concentrations (range = 23 to 46%). The two tributaries exhibiting little change in loads (York and Rappahannock), saw a smaller proportion of stations showing a significant decline in concentrations, and a near-zero change in estuarine TN concentrations. The largest declines in TP loads were observed in the James (-49%) and Choptank (-34%) tributaries, while the Potomac (+5%) and Rappahannock (+29%) showed increasing TP. A relationship between TP loads and changes in estuarine TP was not observed when all stations were aggregated with total TP inputs (AFL+BFL). One caveat of this analysis is that we consider loads at the Fall Line to impact all estuarine stations similarly, which would not be true given that some N or P entering an estuary is sequestered after input.

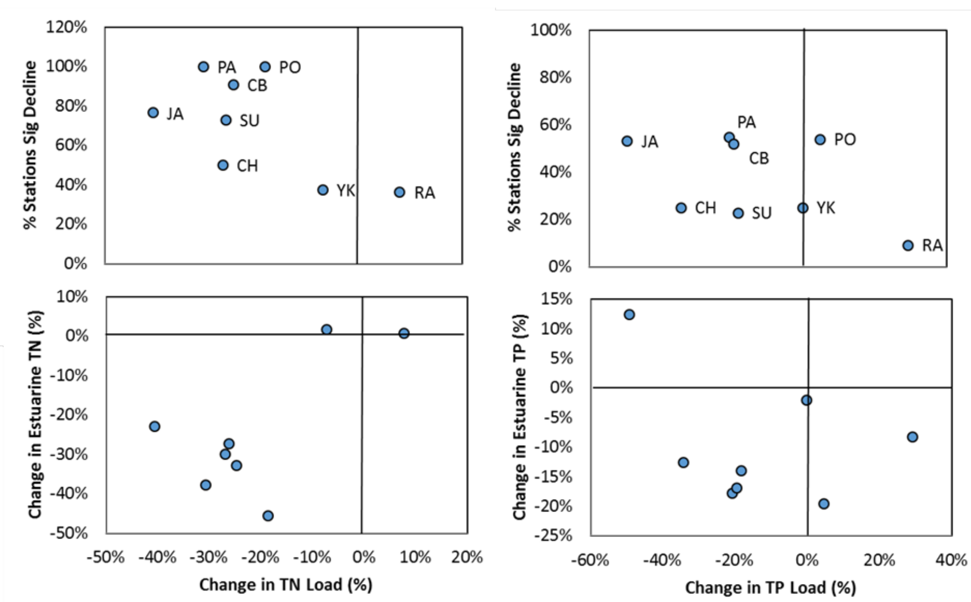

Figure 6: (top panels) % stations in each tributary with a significant decline related to % changes in total TN and TP loads to the estuaries. (bottom panel) Relationship between basin-averaged % changes in total phosphorus and nitrogen loadings and concentration between 1989-1991 and 2012-2014. JA = James, PA = Patuxent, CB = overall Chesapeake Bay, SU = Susquehanna, CH = Choptank, YK = York, RA = Rappahannock, PO = Potomac

The most compelling finding from this analysis was that in almost 90% of cases, segments that saw reduced BFL TN and TP loads saw decreased TN and TP concentrations, illustrating that nutrient input reductions lead to nutrient availability reductions. For TN, this "reduced load, reduced concentration" pattern prevails when both fall line and below fall line loads are considered (Fig. 7). For TP, this "reduced load, reduced concentration" pattern *did not* prevail when both fall line and below fall line loads were considered (Fig. 6), which is likely related to the fact that TP entering at the fall line is sequestered in upper regions of tributaries due to particulate (sediment) phosphorus trapping and sedimentation, and does not strongly link to lower regions of the estuaries. Spatial mapping of changes in AFL and BFL loads to concentration changes reveal that these changes do not always match (Fig. 7).

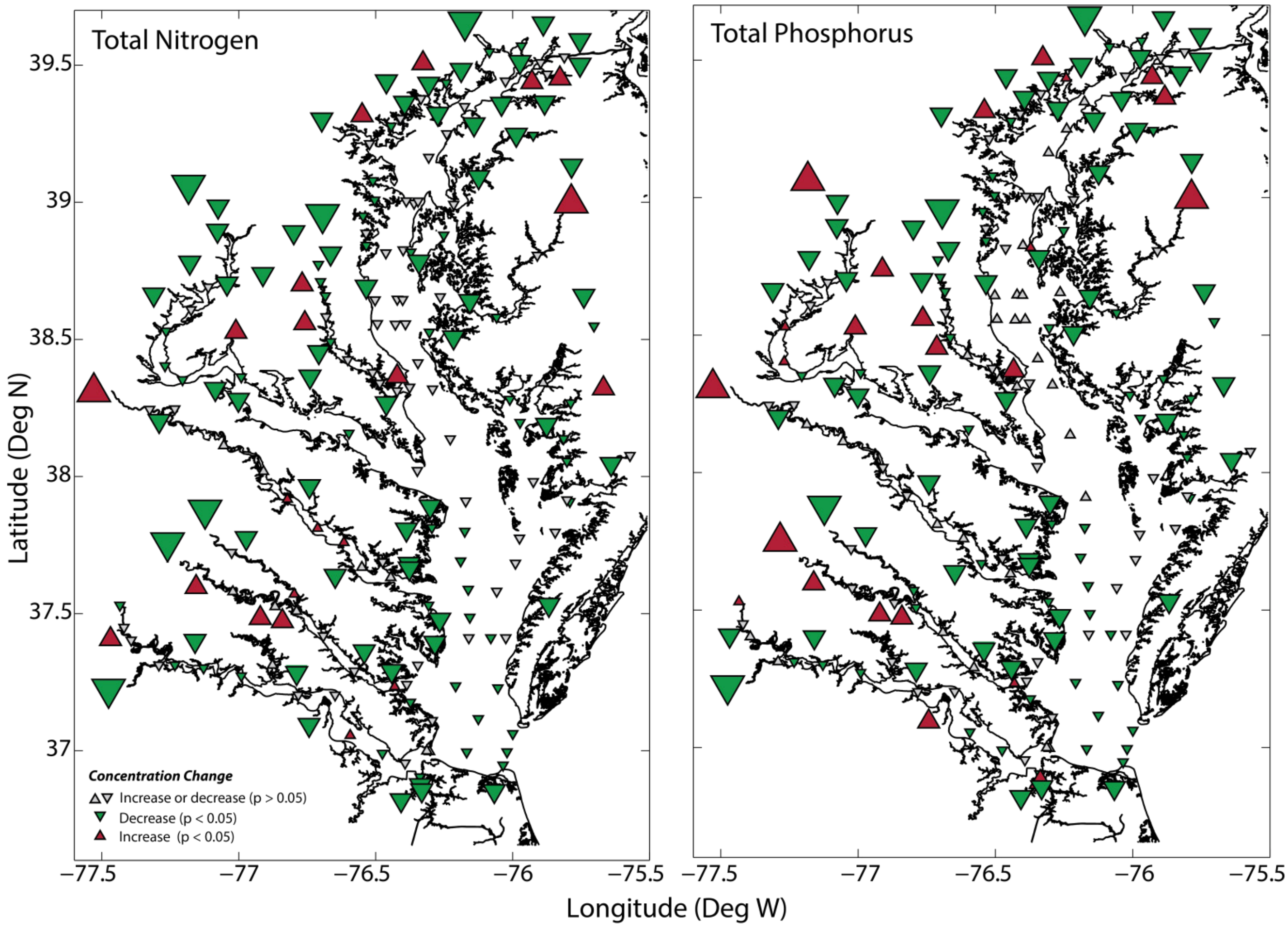

Figure 7: Summary of changes to AFL loads (large triangles), BFL loads (medium triangles) and estuary concentrations (small triangles) for TN (left) and TP (right) between 1989-1991 and 2012-2014. For concentrations, green indicates a significant decrease, while red indicates a significant increase. Grey triangles are non-significant changes.

Summaries of long-term change in TN, TP and chlorophyll for major Bay regions: Below we summarize the long term changes in loading and concentration for specific large tributaries and Bay regions. We do not include the Potomac and James Rivers, which are highlighted in ***Section 4***, ***Case Studies.***

Upper Western Shore MD tributaries: These tributaries include the Bush, Gunpowder, Magothy, Rhode, Severn, South, and West Rivers. All of these tributaries experienced declines in TN and TP concentrations between 1989-1991 and 2012-2014 (mean of 36% for TN and 26% for TP). These reductions occurred despite somewhat smaller percent declines in TN and TP inputs from BFL sources (6 and 15 for TN and TP, respectively). In fact, total BFL TN and TP loads increased to the Gunpowder River and TN loads for the Rhode River. Chlorophyll-a concentrations were either unchanging or increasing in all but one (Magothy) of these tributaries from 1999-2015, while Secchi depth was unchanged or declined in all but two (Severn, South) of these systems. In summary, although nutrient concentrations have declined over the past three decades, water quality has been stable since 1999 in the majority of these tributaries.

Upper Eastern Shore MD: These tributaries include the Elk, Bohemia, North East, Sassafras, and Chester Rivers, as well as Eastern Bay and the Chesapeake and Delaware Canal. All of these tributaries experienced declines in TN and TP concentrations between 1989-1991 and 2012-2014 (mean of 34% for TN and 21% for TP). Loads were also lower in recent years for all rivers, except the Elk River, where TP load increased by 23% and TP concentration only declined by 4% and the North East River, where TP loads increased by 25%. However, recent patterns of change in chlorophyll-a and Secchi depth since 1999 indicate degrading or unchanged conditions, consistent with TP and chlorophyll-a increases in the adjacent mainstem Chesapeake Bay. A stand-along river is the Chester, where chlorophyll-a and Secchi depth have recently improved (since 1999, chlorophyll-a decline in upper Chester only). While this surprising increase in the upper Chester has yet to be fully explained, the reduction in chlorophyll-a was accompanied by reductions in other particulate forms (TSS, particulate phosphorus, carbon, and nitrogen), indicative of an internal ecological change.

Lower Eastern Shore MD: These tributaries include Fishing Bay and the Little Choptank, Big Annemessex, Pocomoke River, Nanticoke, Manokin, and Wicomico Rivers. BFL loadings of both TN and TP declined in all of these regions, with and average decline of 46 and 49%, respectively across all rivers. In response, TN and TP concentrations have declined by 35 and 38% respectively, across all rivers, but some stations displayed small increases (TN for the Mesohaline Nanticoke) or minimal changes (TN in tidal fresh Nanticoke, TP in mesohaline Pocomoke). Recent (1999-2015) trends in TN, TP, and chlorophyll-a have not been significant, except for TP in some regions, but Secchi depths have increased at many stations (see Section 5).

Patapsco/Back: The Patapsco and Back River both experienced large declines (>25%) in both TN and TP load and concentration between 1989-1991 and 2012-2014. Chlorophyll-a and Secchi depth have not changed significantly since 1999, but these two tributaries rank among the Bay's most polluted, given their proximity to Baltimore City. The Back River is described in more detail in Section 4i.

Patuxent River: The Patuxent River experienced declines of 17% for TN and TP loads from BFL sources between 1989-1991 and 2012-2014 and additional reductions in AFL loads. As a result TN and TP concentrations have declined throughout the estuary, including by > 50% for both TN and TP in Western Branch, where a large WWTP was upgraded in the 1990s. Although loads during 2011 and 2014 were low, annual BFL loads from non-point sources do appear to be trending upward over 1985-2014. As a consequence, chlorophyll-a, Secchi depth, and TN and TP have not changed or are degrading in the mesohaline sections of the estuary since 1999.

Rappahannock River: We analyzed the Rappahannock River segments and the adjacent Piankatank and Corrotoman Rivers. With the exception of the tidal fresh segment of the Rappahannock River, TN and TP loads increased in these waters or only changed slightly. TN and TP concentrations declined overall (16% or less on average), but TN increased in the lower reaches of the Rappahannock. Consequently, the tidal fresh regions of the Rappahannock River saw improved chlorophyll-a and Secchi depth conditions since 1999, but the oligohaline and mesohaline reaches (including Piankatank and Corrotoman) saw stable or degraded conditions.

York River: We analyzed the York River segments, as well as the Mattaponi and Pamunkey Rivers upstream and Mobjack Bay downstream. On average, TN and TP loads did not change in this system, but loads increased to all segments except Mobjack Bay and the oligohaline Pamunkey River. As a consequence, TN concentrations increased in all but 2 segments within the York system, TP concentrations dropped minimally, and chlorophyll-a and Secchi depth have been unchanged since 1999.

Choptank River: While BFL TN and TP loads declined in the Choptank River between 1989-1991 and 2012-2014 (mean of 38% for TN and 44% for TP), AFL loads have been stable and have shown periods of increase since 1985. Thus, while TN concentration declined in the lower mesohaline Choptank, TP concentrations have generally increased and chlorophyll-a and Secchi depth remain degraded.

Tangier Sound: Tangier Sound displayed reductions of ~40% in TN BFL load and concentration and 16% reductions in TP BFL load and concentration over the 30-year period. As with the Lower Eastern Shore of Maryland, recent (1999-2015) trends have not been significant although Secchi depth has increase recently.

4. Case Studies of Restoration in Chesapeake Bay

Looking beyond large scale analyses of changes in loads and concentrations across the Bay watershed and estuary, we present below a series of tributary-specific “case studies” that highlight restoration successes. There are abundant examples of restoration “success” for various metrics across many habitats within Chesapeake Bay, and here we summarize the key factors associated with those successes, as well as examples of delayed or absent recovery.

Advances in Wastewater Treatment Many coastal regions experiencing urbanization as a result of the industrial revolution follow a familiar track record in regards to wastewater management. This is certainly the case in the Chesapeake Bay where poor water quality and outbreaks of disease were leapfrogged by technological upgrades in WWTPs during the past one hundred years. In many cities located in the upper portions of our tidal tributaries, early sewage systems discharged directly to tidal waters in the early 1900s, including both Baltimore and the District of Columbia (D.C.). In applying the TMDL for the Chesapeake Bay, targeting point sources such as those from WWTPs was an obvious first step in tackling the challenging problems associated with anthropogenic nutrient inputs. Largely implemented through the National Pollutant Discharge Elimination System (NPDES) permitting program administered by the EPA, the most recent efforts to manage nutrient pollution have focused on removing nitrogen through Enhanced Nitrogen Removal (ENR) processes, sometimes categorized as “tertiary” treatment. In the vast majority of the watershed, WWTPs have been upgraded to other forms of advanced treatment, including Biological Nitrogen Removal (BNR) and other processes. For the purpose of this work, we identified WWTPs that discharge ***directly*** to Chesapeake Bay tidal waters and identified 80 total facilities. Of these, 45 are major (i.e. average flow > 500,000 gallons per day) and 35 are minor (Table 3).

Table 3: Number of WWTPs in three segments of the Chesapeake Bay watershed, including those that are undergoing (or have completed) major tertiary upgrades. (Major WWTPs = Flows > 500,000 gallons per day). ND = no data. Data were collected from the NPDES system and are current through 2016. A survey was undertaken in Virginia to confirm plant status. Nevertheless, further efforts to create a WWTP database or the Bay would be well appreciated. Maryland has more complete data on discharging WWTPs, but in order to align data sources across both MD and Virginia, we chose data sources from EPA NPDES.

	CB_WWTPs_Tidal		ENR Complete		ENR Partial		ENR Targeted	
	Major	Minor	Major	Minor	Major	Minor	Major	Minor
Washington DC	1	0	1	0	0	0	0	0
Maryland	23	27	14	7	6	4	0	5
Virginia	21	8	ND	ND	ND	ND	ND	ND
Total	**45**	**35**	15	7	6	4	0	5

As of 2016, The Chesapeake Bay Restoration Fund supported ENR upgrades for 67 WWTPs in Maryland (this includes those discharging directly to tidal waters as well as other WWTPs in the watershed). ENR improves upon the nitrogen and phosphorus reductions achieved through biological nutrient removal (BNR). Total estimated reductions from ENR upgrades for the 31 WWTPs in Maryland that discharge directly to tidal waters is 8,103,391 lbs/yr for nitrogen and 686,836 lbs/yr for phosphorus. While wastewater contributes less than 10% of the total nitrogen budget for the Chesapeake Bay, these point sources can have significant impacts on local receiving waters. This is reflected in the finding that point source reductions accounted for the bulk of the Below Fall Line load reductions (see above). Given the great progress made in reducing these point sources, we present case studies from five systems in the Chesapeake Bay where notable changes in water quality have been noted and attributed to changes in WWTP nutrient loads.

i. Back River Case Study

The Back River represents a truly urban estuarine system. The watershed and estuary lie immediately north of Baltimore City and the Patapsco River estuary and both areas have been intensely urban and industrial for the past one hundred years (Capper et al. 1983). The estuary failed many indicators of water quality. For example, SAV presence, often used as an indicator of good water quality, were not present in the Back River and persistent algal blooms occurred at all times of the year (most frequently during summer) when chlorophyll-a concentrations often exceeded 100 $\mu g\ l^{-1}$. The water often had a distinct green color and water clarity was seriously compromised (Secchi depth < 0.4 m). There was no evidence of anoxia but night-time hypoxia was common during summer periods (Boynton et al. 1998). The Back River sewage treatment facility has been in operation for 104 years and in recent years the plant has been upgraded to remove more sediment, phosphorus and nitrogen with additional nitrogen removal scheduled to begin during 2017. As a result, the changes in WWTP discharges during the late 1980s through the late 1990s were large. DIN and TN inputs during the period 1984–1996 were about twice those since 1996. In more recent years, nitrate, dissolved inorganic nitrogen (nitrate+nitrite+ammonium) and TN discharges have again increased associated with increased flow from the WWTP, but both are expected to decline again when the upgrades to advanced wastewater treatment are completed during 2017. Both reactive and total phosphorus exhibited dramatic reductions after 1993. For example, TP loads were about 700 kg day^{-1} in 1984 while after 1993 loads were reduced to less than 100 kg day^{-1}.

A number of water quality variables responded to reduced point source nutrient loading during the 1990s. WWTP discharges of TSS, BOD5, and NH_4^+ sharply decreased. Nitrate, ammonium, and phosphate concentrations have also declined, some by a large amount. Surface sediment concentrations (top 1 cm) of particulate carbon, nitrogen, phosphorus and chlorophyll-*a* were all lower following WWTP load reductions. In addition, releases of both ammonium and phosphorus from sediment to the water-column have substantially declined and likely contributed to the observed reduction in water column chlorophyll-a concentration (Boynton et al. 2015). While still highly eutrophic, water quality has increased overall in response to sewage treatment upgrades in the Back River. With expected future nutrient load reductions (2017) it is reasonable to expect further reductions in algal stocks, reduced diel oxygen excursions, lower nitrogen and phosphorus concentrations in the water column and possible further reductions in sediment nutrient releases. However, while all these changes will move the system towards a less eutrophic state, the Back River will still likely remain an overly enriched system. This tributary stands out as one that will likely take time to recover, given extremely high nutrient concentrations, a very high human population in the watershed, a large storage of nutrients in sediments, and its long history of degradation.

ii. Potomac River Case Study

The Potomac River is the second largest source of nutrients and freshwater to Chesapeake Bay and is the largest tributary estuary within the ecosystem. It has also been a key study site in water quality research, beginning with work to address public health concerns such as safe drinking water and waterborne disease during periods of population growth and urbanization at the turn of the 20th century. In many ways, the management actions related to management of wastewater and the trajectory of technological advancements in WWTP engineering nationwide are distilled in the local history of the iconic Blue Plains WWTP that serves the metropolitan DC region in the upper portion of the estuary. Blue Plains continues to represent the largest point source of nutrients to the Chesapeake Bay, but its advances in nutrient removal have been at the vanguard of technology for a WWTP of its size. Primary treatment was the first development to come to Blue Plains to reduce biological oxygen demand, with P removal occurring in the late 1970s. At the turn of the 21st century, nitrogen reducing technologies were added and ENR has reduced N loads significantly. Today, Blue Plains discharges an average of 3.6×10^6 kg of N per year (DC WATER 2010). A recent study by

Pennino et al. (2016) to track the fate of wastewater N in the Potomac as it transitions through the estuary to the mainstem of the Chesapeake Bay indicates a seasonal trend of net export (Table 4). The greatest loads to the mainstem occur in the spring, while other seasons exhibit intense internal cycling and attenuation of N due to burial and denitrification.

Table 4. Seasonal inputs and exports of total nitrogen for the Potomac estuary

	Total Potomac Inputs (kg/day)	% Inputs from Blue Plains*	Net Export to Bay (kg/day)	% Blue Plains Inputs Exported
Winter	49,150 ± 30,323	10 ± 13	19,844 ± 13,728	3.7 ± NA
Spring	135,317 ± 14,614	8 ± 0.8	68,431 ± 48,060	71 ± 20
Summer	13,888 ± 596	38 ± 3	4,853 ± 8,326	19 ± 11
Fall	15,334 ± 3,700	47 ± 13	-1,613 ± 12,124	18 ± 10

The impact of these reductions have been evident in water quality improvements measured in the tidal fresh portion of the Potomac estuary. These improvements began when primary treatment was added to Blue Plains. Jaworski et al. (2007) report historic data documenting improvements in dissolved oxygen and BOD in the 1960s and 1970s. More recently, Bricker et al. (2014) evaluated "eutrophication condition" based on records of nitrate, chlorophyll *a* and TSS to show that tidal fresh nitrate and chlorophyll *a* concentrations have decreased in recent years. Ruhl and Rybicki (2010) undertook a statistical analysis to link changes in nutrient concentration, WWTP loads, and TSS conditions to improvements in submerged aquatic vegetation in the tidal fresh Potomac, a region with marked expansion of SAV in the past 10 years. Unfortunately, these effects have not been evident in the lower portion of the Potomac, where the influence of large watershed nutrient inputs, climate fluctuations, and mainstem nutrient and dissolved oxygen concentrations may be more influential than elsewhere in the tributary.

iii. Mattawoman Creek Case Study

Mattawoman Creek is a small tributary branching off the upper Potomac River that has shown a dramatic restoration in recent decades. Nutrient levels were historically high in this shallow tributary due to high nutrient inputs from the adjacent Potomac River and smaller WWTPs within the watershed of Mattawoman Creek. As a result, water clarity was poor (< 0.6 meters), SAV were absent, and the waters were filled with an abundance of microscopic algae. In the mid-late 1990s, nitrogen reductions began in earnest leading to declining dissolved nitrogen levels and by 2000 an abrupt increase in water clarity and SAV coverage occurred (Fig. 8; Boynton et al. 2014). The improvements in this shallow ecosystem following sustained nutrient reductions reveal how rapid recovery can occur in regions of Chesapeake Bay where modest improvements in water clarity can support SAV expansion. This story also highlights how climatic conditions can support recovery, as the SAV recovery coincided with an extended drought from 1999-2002, as well as the role of non-native species, where *Hydrilla* was the

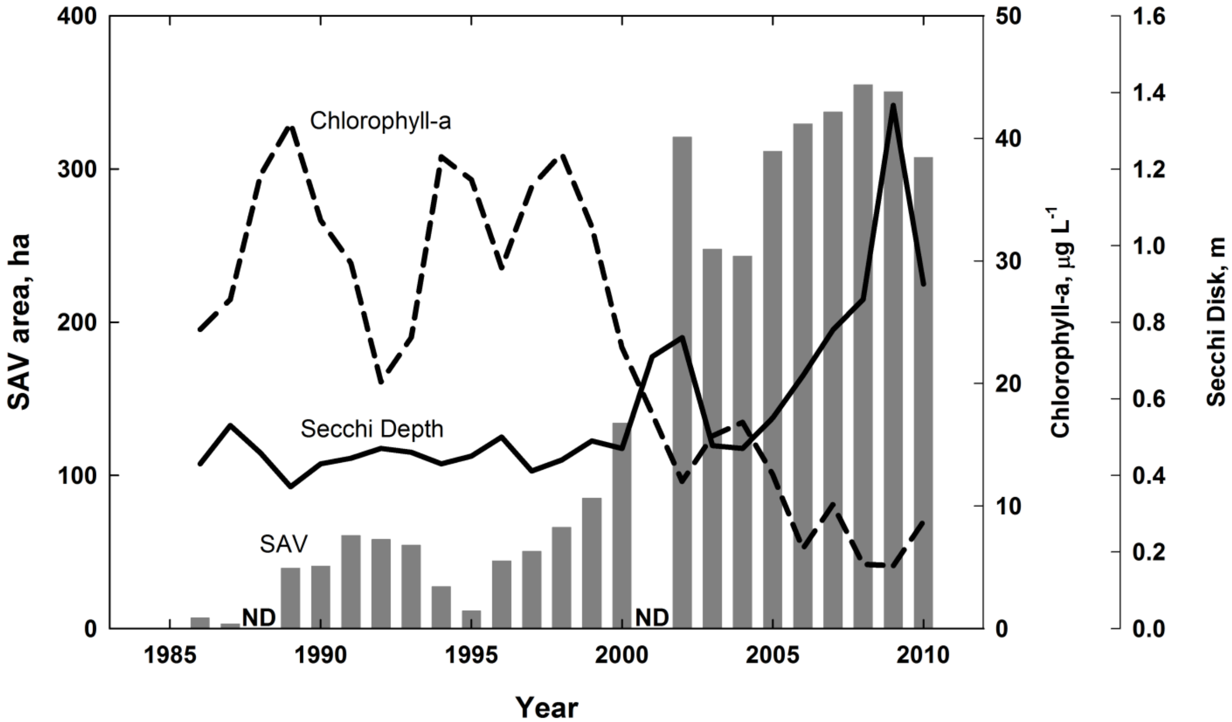

Figure 8: Long-term patterns of chlorophyll-*a*, Secchi depth, and SAV coverage in Mattawoman Creek over the 1985 to 2010 period. Figure from Boynton et al. (2014).

primary species involved in the initial recovery.

iv. Patuxent River Case Study

Unlike the broader Bay watershed, nutrient inputs from the Upper Patuxent River watershed have historically been dominated by WWTP inputs associated with a high human population. In response to these inputs, large TP (initially) and TN (by mid-1990s) load reductions from WWTPs served to reduce the overall nutrient input to the upper estuary, but watershed-wide nutrient inputs did not decline (Boynton et al. 2008), primarily due to high flows (and loads) in the 1990s and exchange with mainstem Chesapeake Bay. In the upper Patuxent River, nitrogen reductions associated with BNR in the 1990s were associated with elevated SAV colonization in the tidal freshwater and low salinity regions of the estuary, as well as some improvements in summer dissolved oxygen conditions (Kemp et al. 2009). Thus, the local effects of nutrient input reductions appeared to be positive. However, in the higher salinity reaches of the estuary (i.e., the lower estuary) water quality continued to show eutrophic conditions even after nutrients were reduced, which underscored the effects of exchange with the mainstem Bay in this region and the potential for changes in "top-down" control (i.e., reduced grazing; Testa et al. 2008). Given the relative stability of the magnitude of watershed-wide nutrient inputs, the Patuxent estuary overall continues to receive water quality scores that are lower than many adjacent systems (Orth et al. 2017).

v. James River Case Study

The James is the southernmost and third largest tributary of the Bay. The James receives nutrient inputs from a large contributing area that includes much of the Commonwealth of Virginia, as well as from local point sources associated with major metropolitan areas in Richmond and Norfolk. As a result, the estuary exhibits high levels of algal production, and is considered impaired on the basis of exceeding site-specific chlorophyll standards. Of particular concern are the occurrence of harmful algal blooms which include periodic outbreaks of dinoflagellates in the lower, saline portions of the James, and the persistent occurrence of harmful cyanobacteria in the upper, freshwater segments. The tidal fresh segment of the James exhibits the highest annual average chlorophyll values within the Bay, and these are associated with the presence of cyanotoxins in water, sediments, fish and shellfish (Wood et al. 2014). Recent work has also shown that these algal toxins are propagated into riparian food webs via emerging insects (Moy et al. 2016). High chlorophyll concentrations in the tidal fresh segment are attributed to natural and anthropogenic factors. A transition in channel geomorphometry from a deep, constricted channel, to a broad, shallow channel stimulates phytoplankton growth due to longer water residence time, and because algae spend more time closer to the surface at high light intensities (Bukaveckas et al. 2011). The partial release from light limitation allows algae to more efficiently utilize nutrient inputs from riverine and local point sources. Recent improvements in wastewater treatment plants have raised expectations for associated improvements in water quality, and specifically, mitigation of algal blooms. Progress is apparent in that a comparison of the most recent 8-y period (2007-2014) with the historic portion of the monitoring record (1985-1992) shows that point source inputs of TN declined from 20,000 to 7,000 kg d^{-1} and for TP, from 2,000 to 250 kg d^{-1}. Despite reductions in nutrient loads, there is little evidence for mitigation of algal blooms, based on long-term monitoring of chlorophyll. The lack of response to reductions in TN and TP loads is of concern, though recent work suggests that summer N and P concentrations are declining, and the severity of phytoplankton nutrient limitation is increasing (Wood and Bukaveckas 2014). The tidal fresh segment is highly effective in trapping N- and especially P- rich particulate matter, which support internal cycles, and may delay recovery in response to nutrient load reductions (Bukaveckas & Isenberg 2013). Studies of N cycling in the James show that internal regeneration was sufficient to fully meet the daily algal N demand in this co-limited (N and P) system (Wood et al. 2016). Overall, these findings suggest that there may be appreciable time lags to improvements in water quality, despite large reductions in point source nutrient inputs.

vi. Mainstem Chesapeake Bay

The mainstem body of Chesapeake Bay is a large and complex ecosystem that includes waters with a wide-range of salinities and nutrient levels. Despite this complexity, recent changes in nutrient concentrations, chlorophyll-a, and dissolved oxygen are discernable.

Total Nitrogen, Reductions The most consistent evidence of restoration in Chesapeake Bay is the widespread reduction in total nitrogen concentrations in the tributary-estuary system. Recent trend analyses at a large collection of monitoring stations has indicated long-term declines in TN concentration over the past 15 years at the majority of the stations monitored. Total Nitrogen (TN) represents the sum of all forms of the key limiting nutrient nitrogen (both dissolved and particulate) and is tightly linked to the input of nitrogen from the watershed (Boynton and Kemp 2000). Such widespread reductions in TN are primarily associated with reductions in inputs (Fig. 2 & 3), particularly the long-term decline in atmospheric deposition and reductions in inputs from many of the watershed's sewage treatment plants, including all of the largest facilities. In general, low-salinity regions of tributaries and the mainstem stations displayed consistent declines in TN concentration, while the moderate salinity regions of the tributaries did not change substantially (Fig. 11). These concurrent declines in both TN input and concentrations display clear evidence that efforts to reduce inputs have direct consequences for the estuary.

Total Phosphorus Reductions The concentration of phosphorus in the Chesapeake Bay has also declined across large regions of the Bay (Fig. 11). Reductions in phosphorus are most evident in western shore tributaries and high-salinity Bay waters, primarily due to improvements in WWTP loads and reductions in dissolved phosphorus loads (Fig. 2 & 3). In contrast, phosphorus concentrations in the middle and upper regions of Chesapeake Bay appear to be stable or increasing, which is likely related to increases in loads from some coastal plain watersheds (e.g., Choptank), elevated retention of phosphorus in the region associated with increasing winter algal blooms (Fig. 9), and reduced trapping of particulate phosphorus behind Conowingo Dam (a large fraction of phosphorus inputs are associated with sediments).

Chlorophyll-a Reductions Trends in chlorophyll-*a* paint a much more complicated picture (Fig. 9 &11). While surface chlorophyll-*a* values are decreasing in many of the high-salinity and southern tributary regions (consistent with TN and TP declines), chlorophyll-*a* is increasing in the low-salinity regions of Chesapeake Bay and in low-salinity reaches of the Patuxent and Potomac low-salinity regions (Fig. 9). In the mainstem, the upper Bay chlorophyll-*a* increases are primarily occurring in cold months (February to March), while the lower Bay declines are primarily occurring in April and May (Fig. 9; Testa et al., in prep). In response, the accumulation of bioavailable ammonium resulting from the breakdown of algal material has increased in the upper Bay bottom waters, but declined in the lower Bay waters (Fig. 9). Thus, the long-term changes in chlorophyll-*a* and nutrient accumulation appear to be season- and region-specific.

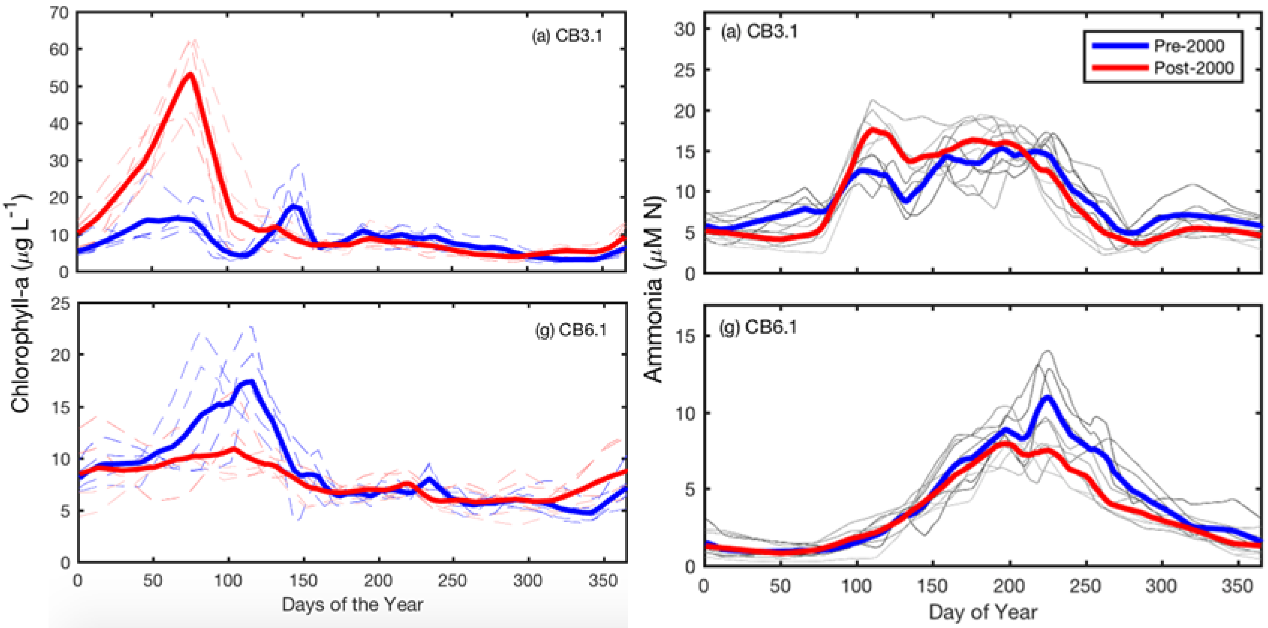

Figure 9: Seasonal cycles of vertically-integrated chlorophyll-a (left) and bottom-water ammonium (right) for an upper Bay station (CB3.1) and a lower Bay station (CB6.1). Bold lines indicate long-term averages over historic (1985-1999; blue lines) and recent (2000-2015; red lines) periods. Muted lines represent all individual years of data.

Declines in Anoxic Volume Anoxia is the condition of near zero oxygen levels in the water-column that can kill or force migration of most multi-celled organisms and leads to changes in Bay chemistry that favors the recycling of nutrients over permanent loss. The volume of anoxic water in Chesapeake Bay has been linked to both freshwater and nutrient inputs (e.g., Hagy et al. 2004) and can therefore vary substantially from year to year. Recent analyses of changes in the anoxic volume over the past 30 years have indicated that volumes in the later summer (mid-July to September) have been steadily decreasing over time, with extremely low volumes in recent years (Fig. 10; http://dnr.maryland.gov/waters/bay/Documents/HypoxiaReports/2016_lateAugust.pdf; Testa et al. 2017). Although low oxygen "hypoxic" volumes have persisted in recent years, the lower levels of oxygen-free water are consistent with reduced nitrogen inputs and chlorophyll-a concentrations (Figs. 9 & 11).

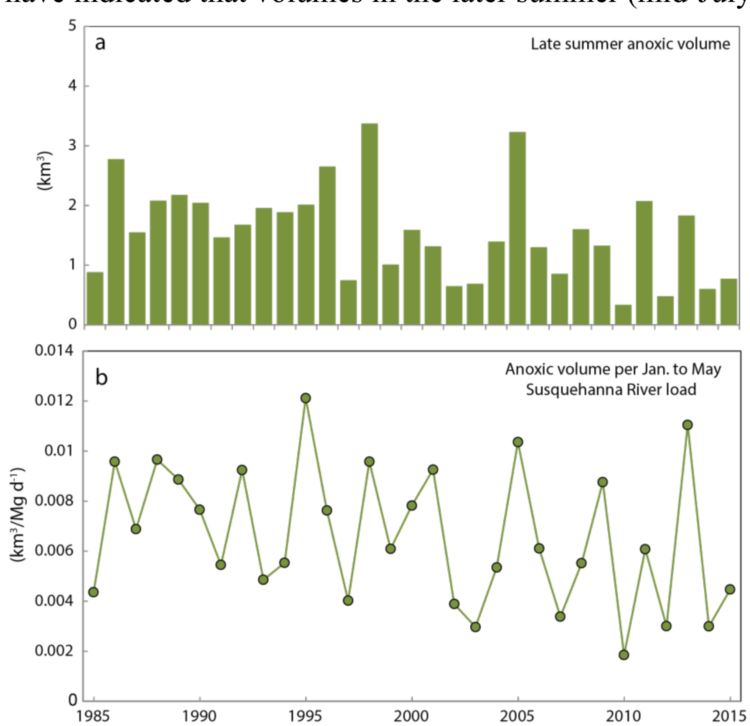

Figure 10: (top) Time-series of late-July to September anoxic volume in Chesapeake Bay and (bottom) volume of late summer anoxic volume per unit Susquehanna River flow.

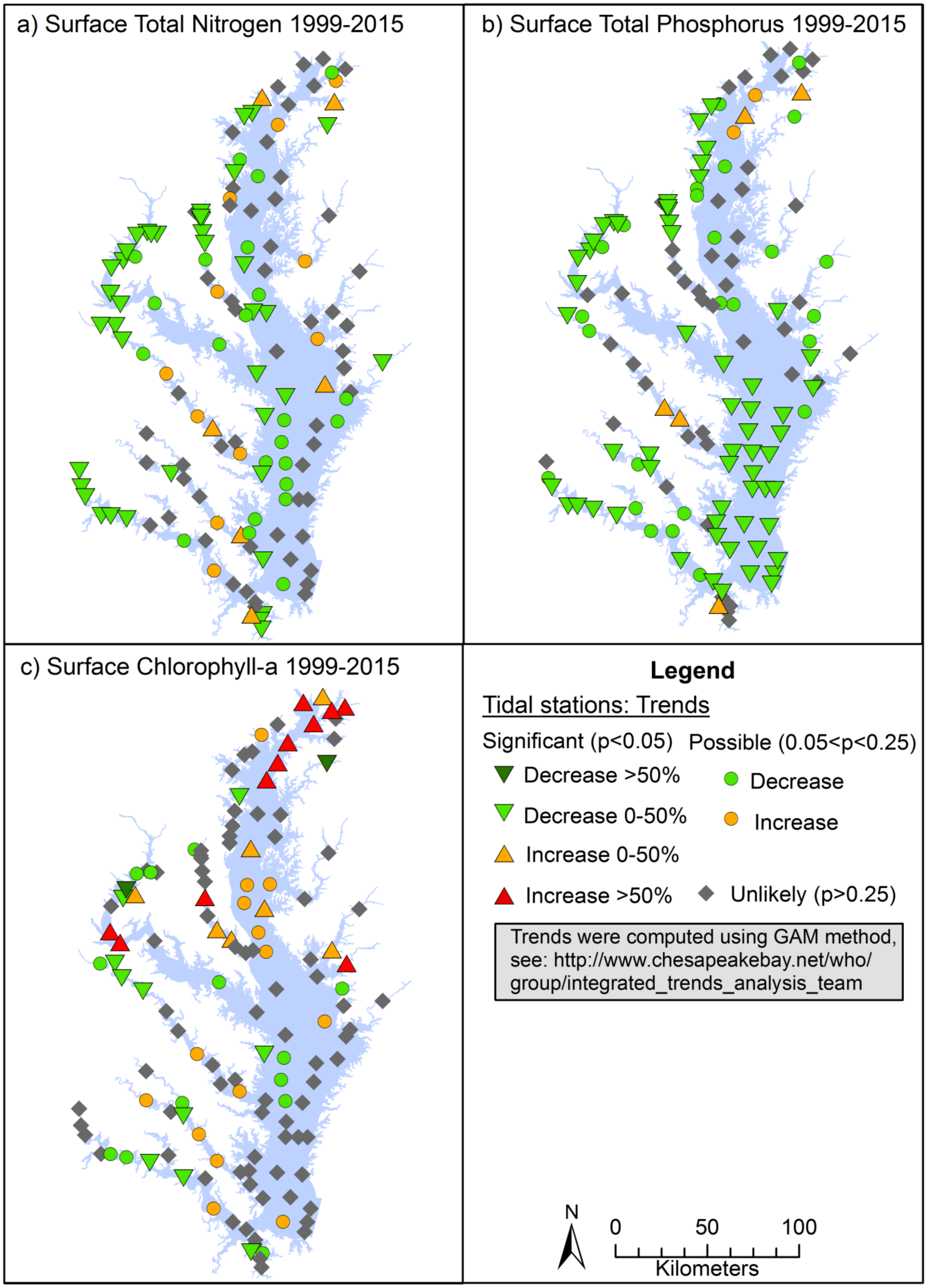

Figure 11: (top) Map of magnitude and direction of trend for total nitrogen (TN), total phosphorus (TP), and chlorophyll-a concentrations in Chesapeake Bay and its tributaries over 1999-2015.

vi. Resistance to Change

Despite the steady accumulation of evidence that displays gradual reductions in the eutrophication of Chesapeake Bay and its tributaries, there are several factors causing resistance to recovery in the estuary and here we provide examples of recalcitrant water-quality.

Water Clarity: Clear water is necessary for SAV growth and the amount of light reaching SAV beds is dependent on the concentration of algae, sediments, and colored dissolved material in the water-column, as well as the growth of epiphytic algae on SAV leaves. While many of these metrics have not recovered to historic levels (e.g., Harding et al. 2016b), tidal water quality trend analyses suggest that over the last 10 years, there has been a reduction in chlorophyll-*a* (an indicator of phytoplankton biomass) and an associated increase in water clarity (as measured with Secchi depth) in much of the mainstem Chesapeake Bay (Figs. 12 & 13). While poor water clarity remains a problem in large regions of Chesapeake Bay, these recent patterns in improvements are consistent with reductions in TN concentrations, declines in chlorophyll-a, and the resurgence of submerged aquatic vegetation in many regions of the Bay (see Fig. 14 and Orth et al. 2017).

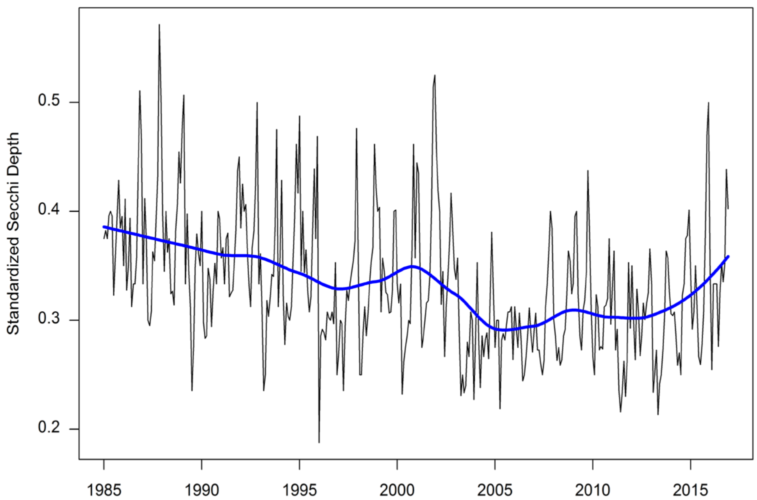

Figure 12: Median of monthly Secchi depths from 133 stations, standardized by the maximal monthly average Secchi depth observed at each station. The superimposed blue line is local regression smoothing (loess).

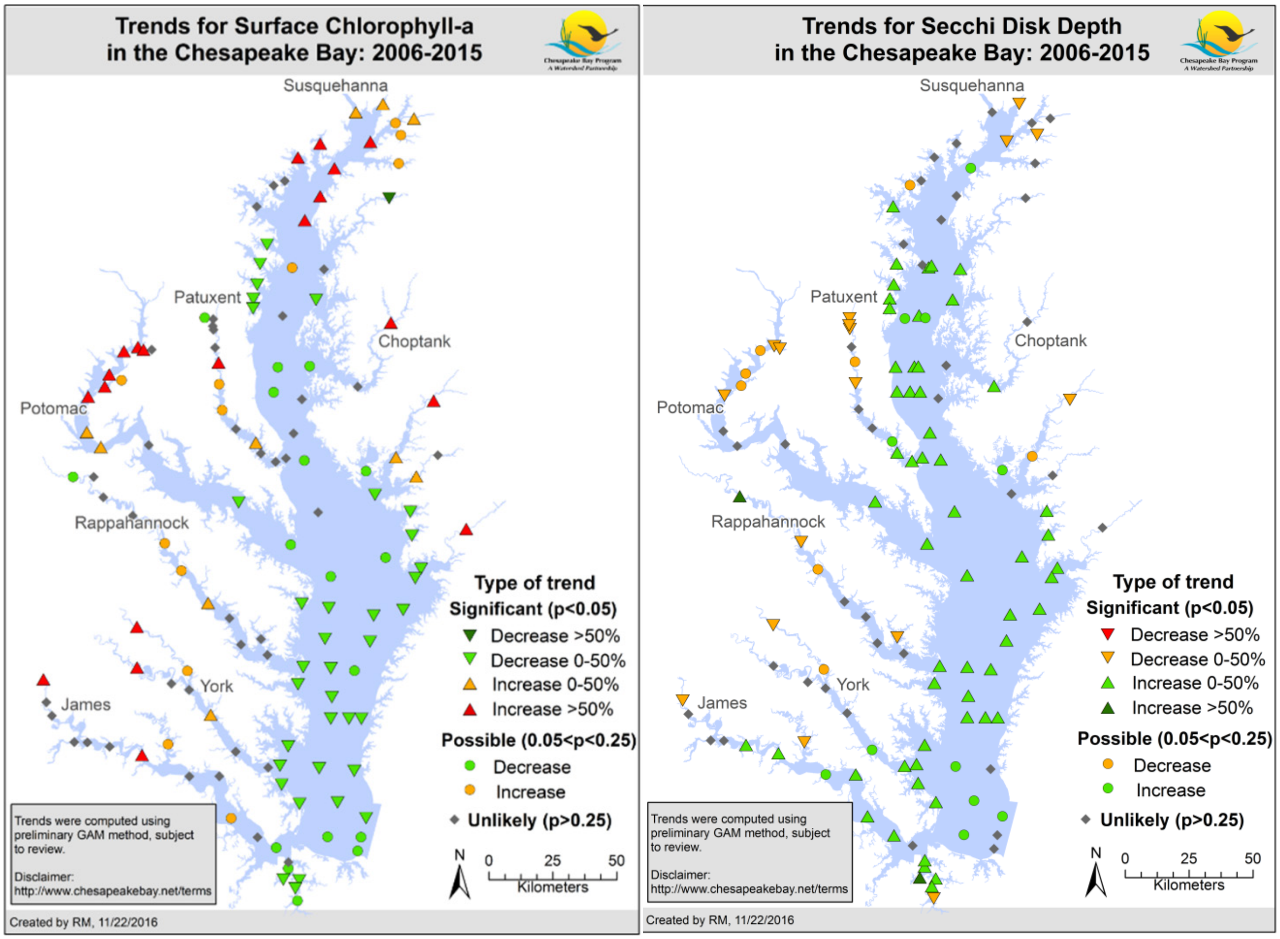

Figure 13: (left) Trend results for annual surface chlorophyll-a (right) and annual Secchi depth for the period between 2006-2015. Arrows indicate direction and color indicates trend significance.

Submerged Aquatic Vegetation and Seagrass (SAV): SAV has shown clear recoveries in many of the low salinity regions of the Bay, including Susquehanna Flats in the upper Bay near the Susquehanna River mouth (Orth et al. 2017). These increases have been associated with recent reductions in nutrient inputs and concentrations (Lefcheck et al. *in review*) and large increases in many locations followed the prolonged drought between 1999 and 2002 (Gurbisz and Kemp 2014). In the middle regions of the Bay, including mesohaline regions, SAV change has been variable and without a specific trend, but recent years of expanding SAV beds in this region are consistent with increased in Secchi depth over the past decade (Fig.13). In lower Bay, polyhaline regions, trajectories have been toward stable or lower coverage. While some of these changes in polyhaline regions are linked to elevated temperatures (Fig. 14), reductions in the availability of light and potential depth distribution of SAV has limited habitat for *Zostera* (Lefcheck et al. 2017). More comprehensive discussions of the changes in SAV over the past three decades are available elsewhere (e.g., Orth et al. 2017)

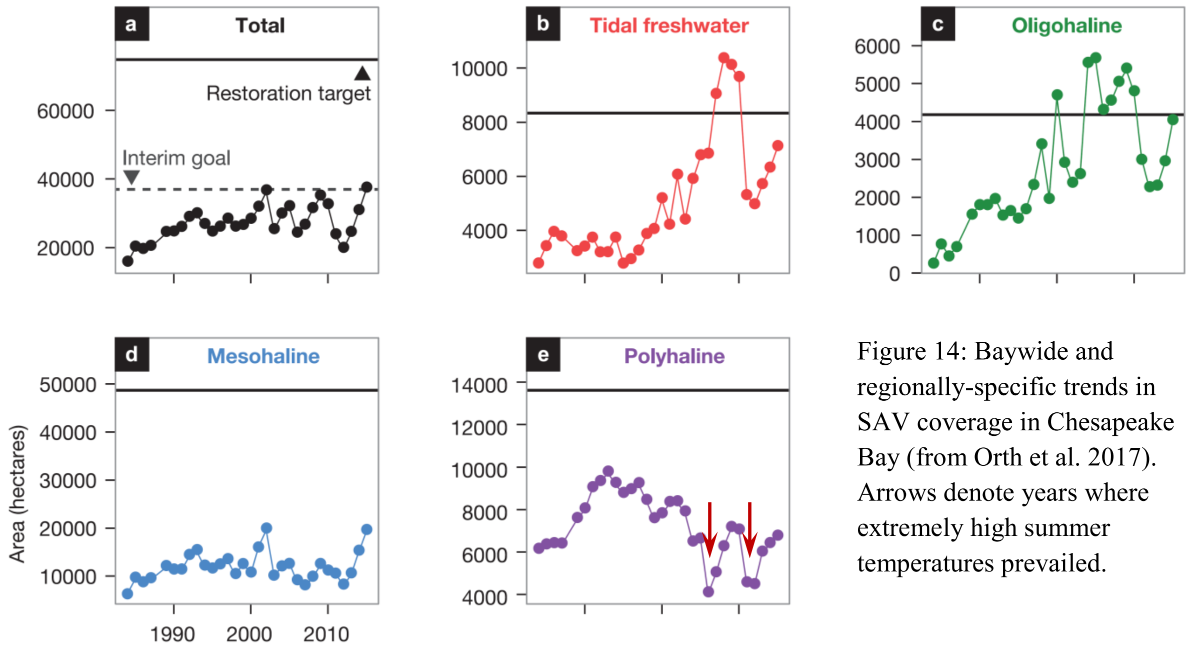

Figure 14: Baywide and regionally-specific trends in SAV coverage in Chesapeake Bay (from Orth et al. 2017). Arrows denote years where extremely high summer temperatures prevailed.

New and Persistent Nutrient Inputs: As the Conowingo reservoir has filled in to reach a state of "dynamic equilibrium", reduced particle trapping has led to increases in particulate nitrogen and phosphorus loading (Zhang et al. 2016). While the impacts of these additional inputs originating from the reservoir have marginal impacts on baywide oxygen conditions, they do result in small and local reductions in oxygen concentrations (Cerco 2016). In addition, chlorophyll-*a*, TN and TP appear to be increasing in some regions of the upper Bay (Fig. 11). Reductions in nutrient inputs from agriculture have been difficult to achieve for a number of reasons, and some eastern shore tributaries fail to show signs of recovery despite BMP implementation (e.g., Choptank River, Corsica). Groundwater in these coastal plain watersheds has long residence times and may harbor nutrients for decades to come.

Climatic Changes: Temperature and sea level continue to rise (Orth et al. 2017, Ding and Elmore 2015), which will have uncertain future impacts on the Bay, but certainly some effects that inhibit restoration. Chesapeake Bay is perhaps most strongly impacted by freshwater inputs from major rivers (e.g., Harding et al. 2015a), and any future changes in river flow will determine the potential for high algal growth and hypoxic volumes. Recent positive signs in the estuary have come during an extended period of moderate to low freshwater and nutrient inputs and despite the fact that in-river dissolved nutrient concentrations are declining, future periods of elevated freshwater inputs may deliver high amounts of nutrients and support water quality degradation.

Sediment Memory: Estuarine sediments act as nutrient reservoirs storing decades of past nutrient inputs. Prolonged "internal loading" via nutrient release from sediments may delay recovery following watershed input declines (e.g., Søndergaard et al. 2003). Delayed recovery may be particularly important in the upper, freshwater segments of the Bay and its tributaries, as these sites retain a large fraction of N and P input loads (Bukaveckas et al. 2017). Algal blooms may exacerbate nutrient release from sediments by altering pH and dissolved oxygen conditions in water overlying sediment (Seitzinger 1991; Zilius et al. 2014). However, much of the mainstem Bay, particularly the expansive mesohaline region, may be less likely to suffer from sediment nutrient "memory", given the predominance of sandy (low nutrient) sediments. In these regions we would expect rapid sediment recovery following nutrient load reductions (Tucker et al. 2014) and more immediate coupling between nutrient deposition and sediment-water fluxes (i.e., "short memory"; Boynton et al. 2017).

5. Where has success been achieved in other coastal ecosystems?

Clear examples of both restoration success and degradation inertia are present in the Chesapeake Bay ecosystem, but how have other systems responded to eutrophication abatement efforts? A useful comparison can be made between the Chesapeake and the coastal waters of Denmark, where (a) similar degradation patterns emerged following the second World War, (b) a long-term monitoring program was established for a diversity of coastal ecosystems (Conley et al. 2002), (c) large social-economic commitments were made to restoration efforts in the watershed, and (d) successful restoration stories emerged. Perhaps the primary difference between Denmark and the Chesapeake is that Denmark aggressively reduced nutrient inputs to many estuaries (e.g, Riemann et al. 2015), resulting in sometimes dramatic declines in nutrient load, in some cases approaching 70% reductions. As a result, clear reductions in chlorophyll-*a* and nutrient concentrations were evident throughout the coastal waters of Denmark and numeric water-quality criteria were often met (Staehr et al. 2016). Other global reviews have indicated that measureable improvements in water quality were documented only in those ecosystems where substantial nutrient reductions were achieved, especially for dissolved oxygen recoveries in systems highly enriched with sewage treatment plant outflows (e.g., Delaware, NYC Metropolitan area, Thames, Scheldt; Kemp et al. 2009). Yet other ecosystems experienced recoveries in submerged aquatic vegetation (e.g., Tampa Bay; Greening and Janicki 2006) and benthic communities (Boston Harbor; Taylor et al. 2011) after substantial point source reductions. This is consistent with the fact that many of the clear 'success stories" in the Chesapeake region are associated with similar large reductions in WWTP loads (see Section 4).

One of the more daunting challenges for the management of coastal ecosystem water quality is the successful reduction of nutrient inputs from *non-point* sources, such as atmospheric deposition, agricultural sources, and storm water flows. Several large ecosystems have seen measureable recoveries in water quality associated with reduced non-point source loads, but they are the exception and not the rule. For example, agricultural subsidies provided to eastern European countries in the Danube River basin by the former Soviet Union lead to large applications of fertilizers in those watersheds and associated growth of the hypoxic zone in the Black Sea waters that directly receive Danube River discharges. Following the collapse of the Soviet Union, the subsidies ended and fertilizer application declined precipitously, leading to a rapid decline in the extent of hypoxia on the northwester shelf of the Black Sea (Mee 2006, Kemp et al. 2009). In the Baltic Sea, a large estuarine basin that has been affected by eutrophication over the past century, recent evaluations of the eutrophication status of the system have indicated marked improvements in many of the Sea's basins in response to persistent reductions in overall nitrogen and phosphorus loads to the estuary (Andersen et al. 2015).

Measurable improvements in dissolved oxygen levels, water clarity, or chlorophyll-*a* have been more difficult to achieve in many other coastal ecosystems. In these cases where some nutrient load reductions have been achieved but improvements have yet to be observed, the stagnation of poor water quality under reduced nutrient loads has been termed a "Return to Neverland" (Duarte et al. 2009). The "Return to Neverland" idea suggests that some aspect of the ecosystem has been altered by the degradation of water quality, which has altered the ecosystem's ability to reach restored conditions. For example, if degradation was accompanied by the removal of filter feeding benthic communities, either through harvest and disease (oysters) or habitat degradation (e.g., clams), much more algal biomass would be able to accumulate for a given nutrient loading because it is no longer being removed by filter feeders. Another example is that the existence of low-oxygen conditions favors the recycling of nutrients over the removal or burial in sediments, so the hypoxia and anoxia that develops under high nutrient loading and algal biomass would help to keep the ecosystem in a high-recycling – high algal biomass state. While many ecosystems around the world have persistently low-water quality despite large efforts to reduce nutrients, the continued poor water quality may exists because the effective nutrient loads to the system have yet to go down. In some cases, where tributary estuaries are connected to high-nutrient waters downstream, tidal nutrient and chlorophyll-*a* inputs can help sustain poor water quality (lower Patuxent Estuary; Testa et al. 2008). Finally, even in systems where large nutrient reductions from sewage treatment plants were made to successfully reduce eutrophication, like in Lake Erie in the 1970s, the persistence and increase in non-point inputs led to a re-eutrophication of this large ecosystem (Scavia et al. 2014).

6. Concluding Comments: Key Themes to Success and Managing Expectations

We have reviewed decades of scientific studies in combination with new data analysis to assess trajectories of water quality in Chesapeake Bay and its tributaries with respect to restoration efforts. Our efforts highlight management features most commonly associated with restoration success, while emphasizing the impediments to water quality improvements associated with inertia in nutrient reductions, climatic change and variability, and lags in the restoration of key habitats and organisms. Below we summarize the key conclusions from our synthesis.

Sewage Treatment Plant Upgrades Work. Our synthesis, combined with prior studies in the Chesapeake Bay and around the globe, indicated that sewage treatment plant upgrades are an effective means to reduce eutrophication in places where they are the dominant nutrient source. These upgrades have a clear local effect, and help reduce eutrophication effects during periods where watershed freshwater and nutrient inputs are low.

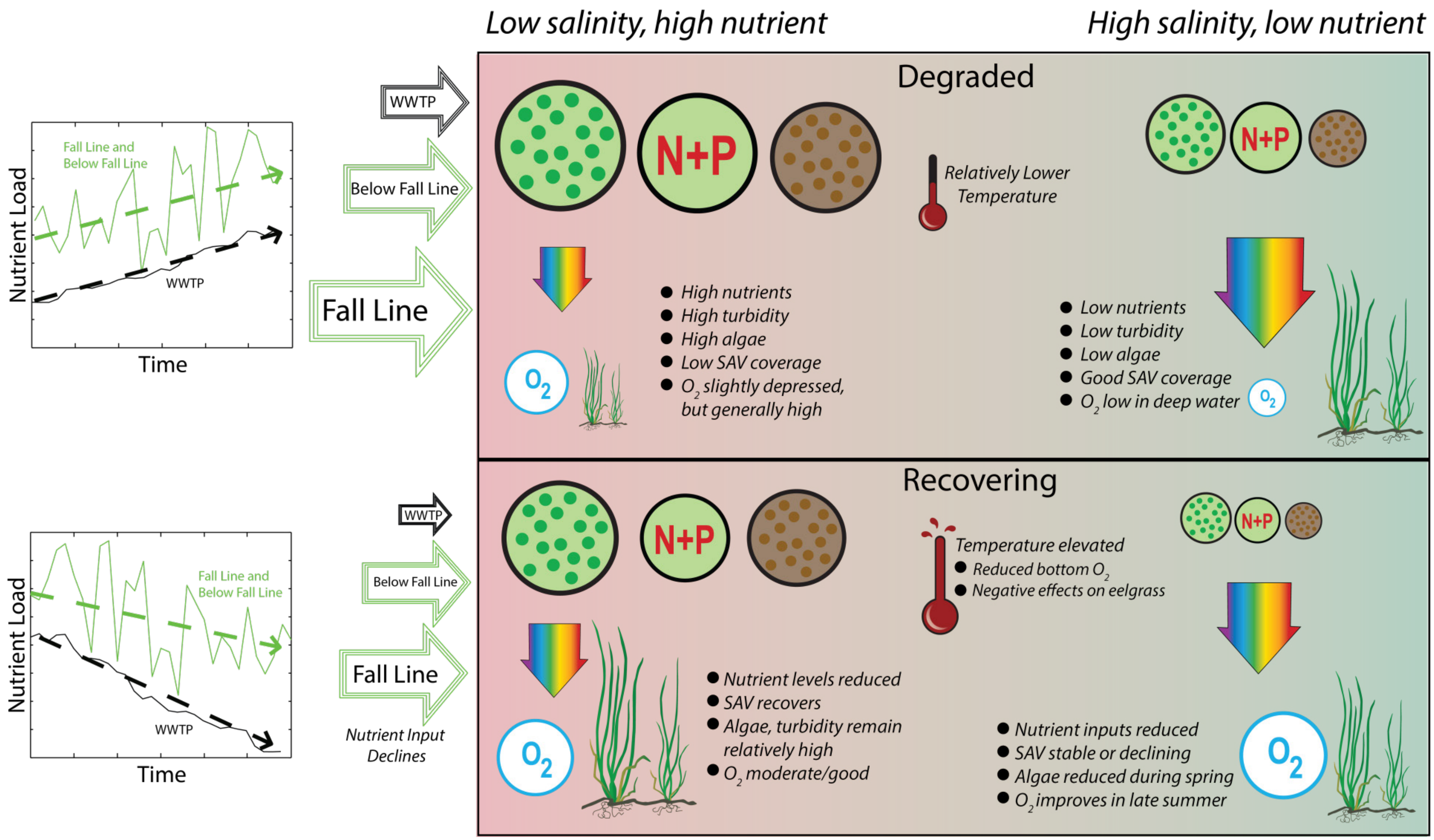

Figure 15: Expanded conceptual model of water quality change in Chesapeake Bay. The diagram emphasizes that low-salinity regions of the estuary, which are closer to watershed nutrient sources, have higher nutrient levels than saltier regions and tend to support higher algal biomass, even under reduced nutrient loads. Thus, when nutrient loads increase (top), the algal biomass in low-salinity regions remains high and the increase nutrient inputs leads to elevated nutrient transport to high-salinity waters, stimulating algal biomass. As a consequence, when nutrient loads are reduced (bottom), reduced nutrient concentrations and algal biomass are realized first in higher-salinity waters, and lower algal biomass generally leads to oxygen replenishment in deep water. SAV has increased in low-salinity regions due to nutrient input reductions, but SAV is stable or declining in high-salinity waters, as elevated temperature and reduced water-clarity limit growth of the lower Bay (i.e., high-salinity) species.

BOTH N and P remain important. Estuaries are complex environments where both nitrogen and phosphorus may be important limiting, or co-limiting, nutrients for algal growth. This fact was recognized early in Chesapeake Bay eutrophication research and it remains true today. Improved water quality and SAV recovery have been realized throughout Chesapeake Bay, underscoring the relevance for both nutrients, despite a historical focus on *either* N or P.

Tributary-Bay Interactions. Chesapeake Bay is a network of interacting tributaries that exchange nutrients across their boundaries. While we often think of nutrient inputs as originating from the local watershed, tributary estuaries can also receive nutrients at the salty seaward boundaries, complicating what we can expect from watershed nutrient reductions. The degree to which these *downstream* nutrient inputs are important must be evaluated on a case-by-case basis.

Sediment Nutrient 'Memory'. Estuarine sediments were long thought to be reservoirs that stored decades of legacy nutrients. Although watershed nutrient inputs might be reduced, the sediments were suspected to continually release the legacy nutrients for long periods, acting as a source of 'internal nutrient loading'. While this 'internal loading' may be important seasonally in estuaries and may cause inertia in lakes or in regions with histories of exceptionally high nutrient inputs, sediment studies indicate that a 'memory' of past nutrient inputs is likely quite short (one to a few years) in the vast majority of Chesapeake Bay sediments.

The *Space and Time* of 'oligotrophication'. It has become increasingly clear that the response of an estuarine region to restoration depends on its location along the estuarine salinity gradient. While TN and TP loads and concentrations have generally declined throughout Chesapeake Bay, only a subset of Bay regions have shown evidence for clear recovery. The first of these locations is the tidal fresh and oligohaline regions of tributaries, where SAV recoveries have accelerated in recent years associated with reduced nutrient inputs, but also the expansion of invasive species. Algal biomass has also declined in some of these low-salinity regions, but primarily where substantial reductions of WWTP inputs have occurred. We might expect recovery in these low salinity regions first, given that they are closest to the watershed nutrient source. However, the other regions where recovery appears to have occurred is within the higher-salinity regions, especially during late summer. Nutrient limitation is at its most severe in Chesapeake Bay in this region and season, and we should also expect that recovery from eutrophication (reduced algal biomass, elevated oxygen) must first occur at the times and places most vulnerable to nutrient poverty. It is clear that our ability to measure a response to eutrophication reduction is dependent upon the time of year and region of the ecosystem when we look. These two features of eutrophication response demand a more complicated conceptual model of eutrophication in Chesapeake Bay, but are consistent with our basic understanding of nutrient impacts on estuaries.

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