# Scientific and Technical Advisory Committee: Evaluation of Management Efforts to Reduce Nutrient and Sediment Contributions to the Chesapeake Bay Estuary


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## About the Scientific and Technical Advisory Committee

The Scientific and Technical Advisory Committee (STAC) provides scientific and technical guidance to the Chesapeake Bay Program (CBP) on measures to restore and protect the Chesapeake Bay. Since its creation in December 1984, STAC has worked to enhance scientific communication and outreach throughout the Chesapeake Bay Watershed and beyond. STAC provides scientific and technical advice in numerous ways, including (1) technical reports and papers, (2) discussion groups, (3) assistance in organizing merit reviews of CBP programs and projects, (4) technical workshops, and (5) interaction between STAC members and the CBP. Through professional and academic contacts and organizational networks of its members, STAC ensures close cooperation among and between the various research institutions and management agencies represented in the Watershed. For additional information about STAC, please visit the STAC website at www.chesapeake.org/stac.

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## 1. Introduction

This document examines the watershed system's response to the Chesapeake Bay Program's efforts to achieve the nutrient and sediment reduction goals thought necessary to achieve water quality standards. We focus on response gaps and uncertainties about how the watershed system has responded to management efforts meant to reduce nitrogen (N), phosphorus (P), and sediment loads reaching the Chesapeake Bay. Our discussion focuses primarily on the management of agricultural and urban nonpoint source pollution. Significant resources have been devoted to controlling both agricultural and urban nonpoint source pollution, but these efforts have not yet generated the expected system response and associated water quality improvements. Based on a synthesis of watershed response studies, we identify a number of possible reasons for the system response gap and discuss possible management actions to further pollution reduction efforts and improve system response.

We conclude that while there is substantial agreement on several major causes for the system response gaps, adequate remedies are confronted with uncertainties and/or difficult trade-offs. Wider best management practice (BMP) implementation/adoption (as is commonly acknowledged) can help to achieve reduction goals along with programmatic changes to improve the effectiveness of management actions. Improving system response to management actions will be challenging given the uncertainties associated with how the physical system responds to different management actions (BMPs) as well as how people manage nutrients and respond to policies designed to reduce pollution. Enhanced adaptive management approaches could be developed. Multiple opportunities to test, learn, and adapt to uncertainties associated with management could be evaluated. Additionally, the Chesapeake Bay Program (CBP) could make greater use of evaluation tools to systematically identify and evaluate key uncertainties when designing and implementing active adaptive management to address pollution.

## 2. The Nonpoint Source Challenge

The total maximum daily load (TMDL) established nitrogen (N), phosphorus (P), and sediment load targets of 215, 13.3, and 18,600 million pounds per year, respectively. The CBP's watershed model (CBP, 2020), the Chesapeake Assessment Scenario Tool (CAST)[1] predicts that management efforts implemented through 2021 are sufficient to achieve the sediment goal. By 2021, P loads stand at an estimated 14.7 million lb/yr, 1.4 million lb/yr above than the 2025 goal. The nitrogen reduction goal is thought to be the most difficult pollutant goal to achieve. The CAST model estimates that 2021 N loads were 258 million lb/yr, or 43 million lb/yr short of

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[1] The CAST model estimates loads at many locations and over time based on a mathematical model of watershed processes driven by landscape characteristics, and the spatial and temporal distribution of various land uses and control measures. It includes nutrient inputs (including atmospheric deposition), land use conditions, point source loads, nonpoint source loads, control practices (BMPs), and transport/delivery to calculate the loads delivered to the Chesapeake Bay.

the 2025 TMDL target. As a relative comparison, CAST calculates 40 million lb/yr of N reductions were achieved between 2009 and 2021.

Reducing nutrient loads to meet the Chesapeake Bay TMDL water quality goals depends largely on reducing nonpoint source pollution. The CAST model calculates that nonpoint sources (agricultural and urban combined) contribute 78% and 74% total controllable (loads from anthropogenic sources) N and P loads to the Chesapeake Bay, respectively (Chesapeake Progress, 2023).

Nonpoint source loads, however, have proven challenging to reduce. Between 2009 and 2021, the vast majority of nutrient reductions have come almost exclusively from upgrading point source wastewater treatment plants and reduced amounts of atmospheric nitrogen deposition (Entringer & Howarth, 2009; Chesapeake Progress, 2023). These data show that, from 2009 through 2021, wastewater was responsible for 65% of N reductions and 76% of P reductions. Atmospheric sources were responsible for 25% of N reductions. In comparison, N from urban and agricultural sources (combined) were responsible for 8% of N and 12% of P reductions (3 million lb/yr reduction in N and 0.3 million lb/yr reduction in P between 2009 and 2021). Given that many large wastewater treatment plants are operating near the limits of treatment technology, only modest additional nutrient reductions can be achieved from existing point sources, and only at considerable cost.

Finally, the reported CAST model loads underestimate the total nutrient reductions needed to meet the Chesapeake Bay TMDL and water quality standards. The CAST nutrient load estimates above do not include the additional nutrient loads needed to offset the early infill of Conowingo reservoir (6 million lb/yr of N). In addition, the CBP has recently discovered unaccounted for sources of nutrients (undercounting millions of animals and missing fertilizer sales, discussion below) that will add millions of pounds of additional nutrient loads to CAST estimates. Lastly, the CBP estimates that the TMDL nutrient targets will also need to be reduced in order to meet the Chesapeake Bay dissolved oxygen water quality criteria because of climate change.

Over the past several decades, states and municipalities have made considerable investments to reduce pollution loads. The federal government alone has spent roughly $500M per year on both point and nonpoint source controls, and on ecosystem restoration to restore the Chesapeake Bay (OMB, 2019). The costs of achieving additional nutrient and sediment reductions, particularly from urban stormwater sources, is substantial. For example, Maryland estimates that for municipal stormwater systems to achieve the required additional reduction of 85,000 lb/yr of N and 43,000 lb/yr of P will cost $1.195B (MDE, 2019). However, this planned billion-dollar investment will only contribute 0.2% N and 7% P to the additional load reductions needed to achieve TMDL goals (based on CAST simulations).

From a management perspective, it is not clear whether additional investments to reduce nutrient and sediment loads to the Chesapeake Bay will produce the desired outcome. Monitored nutrient and sediment loads have not declined in all areas of the watershed and do not consistently align with expected management practice effects (Ator et al., 2019; Fanelli et al., 2019; Keisman et al., 2018a; Kleinman et al., 2019; Mason et al., 2023; Noe et al., 2020a).

Challenges exist for soliciting sufficient behavioral change at the scale needed to achieve nutrient and sediment reduction goals. If there are gaps between existing management actions and system response, what changes in management (both actions on the landscape and behavioral change) are needed to attain the desired water quality outcomes? The challenge confronting the CBP partnership is how to answer this question in such a complex system (Hershner, 2011; Shabman et al., 2007).

### 2.1. Trends in nutrient and sediment loads and effectiveness of reduction efforts

Nutrient and sediment loads and trends are computed from data collected from river and stream monitoring stations (the nontidal network) throughout the Chesapeake Bay watershed. Nine of these stations have been monitored since 1985 on the largest rivers in the Chesapeake Bay watershed. These nine stations are collectively referred to as River Input Monitoring (RIM) stations and represent loads delivered from 78 percent of the Chesapeake Bay watershed (Mason and Soroka, 2022). Table 1 illustrates the long-term (1985–2021) average N, P, and sediment loads and load trends at the RIM stations. Long-term trends show N loads declining at six stations, including the four largest rivers (the Susquehanna, Potomac, James, and Rappahannock). Long-term trends in P and sediment show P and sediment loads have only declined in three rivers. The three largest rivers (Susquehanna, Potomac, James) are showing either decreasing loads or no short-term trends for nutrients. The rivers that show the most consistent and sustained decreasing trends in nutrient loads are the James, Potomac, and Patuxent rivers. These three tributaries also had the highest initial proportion of nutrient loads coming from point source loads (USGS, 1999). Trends in N load have been computed from 89 stations and trends in P and sediment load has been computed from 70 stations throughout the Chesapeake Bay watershed (Mason and Soroka, 2022). Nutrient and sediment trends have not declined at most of these stations in recent years. N and P loads declined at about 40 percent of stations and sediment loads declined at 18 percent of stations from 2011 through 2020.

Analysis of monitored water quality data generally finds mixed evidence of pollution reduction effectiveness (table 1 and figure 1). While nutrient and sediment load reductions have occurred in some agricultural and urban watersheds, the drivers of such changes, including the role of BMPs, are uncertain. In some Chesapeake Bay watersheds, reductions in flow-normalized total nitrogen loads from 2007 through 2018 were associated with reductions in the surplus amount of nitrogen inputs on agricultural land (Zhang et al., 2022). However, agricultural surpluses (the difference between nitrogen inputs and cropland nitrogen removal) have been increasing in many areas of the watershed in recent years (Sabo et al., 2022). A large number of BMPs were installed in the agricultural Choptank River watershed, on Maryland's Eastern Shore, from 2003 through 2014 (Fox et al., 2021); however, monitored nutrient and sediment conditions did not consistently improve (figure 1). In fact, flow-normalized TP loads in the Choptank have nearly doubled since the mid-1990s (figure 1). Empirical Spatially Referenced Regression on Watershed Attributes (SPARROW) models that were extended to include a temporal component found that flow-normalized nutrient yields from agricultural areas have not changed substantially in most areas of the Chesapeake Bay watershed in recent decades (Ator et al., 2019; Chanat and Yang, 2018). Another SPARROW analysis of monitoring data found that while P loads are declining in some regions of the Chesapeake Bay watershed, those improvements were offset by increases in agricultural P sources in other areas (Fanelli et al., 2019). It is important to note that SPARROW uses the flow-normalized loads at many sites in the watershed coupled with statistical means to apportion these loads across either land use categories, watersheds, or political jurisdictions. Furthermore, a STAC report (Keisman et al., 2018a) summarized that “current research suggests that the estimated effects of conservation practices [BMPs] have not been linked to water quality improvements in most streams.”

**Table 1: Average loads (million lb/yr, 1985–2021) and long-term flow-normalized load trends (shaded) of total nitrogen, total phosphorus, and total suspended sediment at RIM stations (Source: Mason and Soroka, 2022).**

| | Total Nitrogen | Total Phosphorus | Total Suspended Sediment |
|---|---|---|---|
| Susquehanna River at Conowingo MD | 135 | 5.44 | 3,533 |
| Potomac River, Chain Bridge, Washington, DC | 48.6 | 3.27 | 2,452 |
| James River at Cartersville, VA | 11.0 | 2.21 | 1,543 |
| Rappahannock River, near Fredericksburg, VA | 4.32 | 0.64 | 478 |
| Appomattox River at Matoaca, VA | 1.46 | 0.14 | 39.4 |
| Pamunkey River near Hanover, VA | 1.41 | 0.16 | 85.0 |
| Mattaponi River near Beulahville, VA | 0.65 | 0.06 | 14.8 |
| Patuxent River near Bowie, MD | 1.51 | 0.11 | 49.0 |
| Choptank River near Greensboro MD | 0.55 | 0.04 | 5.12 |
| Green shaded cells indicate long-term declining loads; red shaded cells indicate increasing loads; and yellow shaded cells indicate no statistical trend in loads. | | | |

Ator et al. (2020) presented a comparison between N and P loads delivered to the Chesapeake Bay estimated by CAST and by a SPARROW model (figure 2). The CAST and SPARROW estimates generally agree that N delivery to the Chesapeake Bay is declining, although CAST generally predicts larger reductions from agricultural sources (crop and pasture) than estimated by SPARROW. However, P delivered loads estimated by CAST and those derived from measured data differ markedly. CAST predicts relatively consistent reductions in delivered P loads, with the reductions occurring relatively consistently across all source sectors (figure 2), while SPARROW estimates that delivered P loads are increasing, attributable largely to agricultural and urban nonpoint source pollution (Ator et al., 2020).

The P results have policy relevant implications since state and local management decisions designed to meet the water quality standards are generally based on CAST results. Historically, CAST has indicated that the gap between existing and target loads is largest for N. The CAST model estimates that sediment reduction targets were achieved in 2019, and P reduction targets are closer to being achieved than N reduction targets. The recent SPARROW estimates suggest that achieving P and sediment reduction goals will be more challenging than expected (Ator et al., 2020). Addressing the divergence of CAST with empirical results is important because CAST is the primary tool for guiding management.

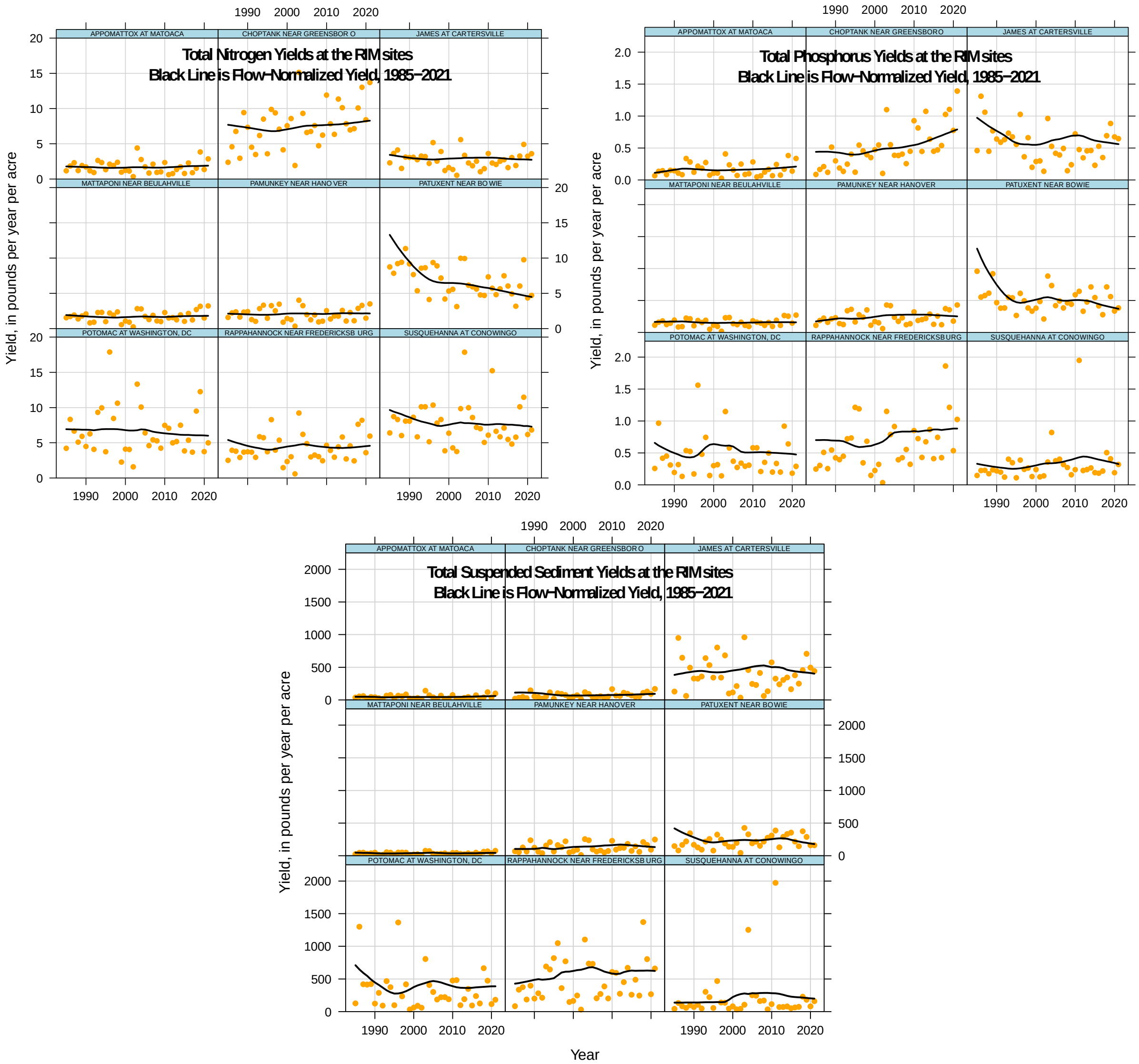


**Figure 1. Flow-normalized total nitrogen, total phosphorus, and total suspended sediment yields at River Input Monitoring (RIM) stations for the period 1985–2020. Figure created by Chris Mason, U.S. Geological Survey Virginia and West Virginia Water Science Center, Richmond, VA. (Source: Mason and Soroka, 2022).**

Uncertainty also surrounds attainment of sediment reduction targets. While CAST estimates that sediment load reduction targets have been achieved, suspended sediment loads have increased in five RIM monitoring stations from 1985 through 2021 (table 1; Mason and Soroka, 2022) and in 46 percent of all monitoring stations in the Chesapeake Bay watershed from 2011 through 2020 (Mason et al., 2023). However, even with ongoing land-based erosion and streambank erosion contributing to watershed sediment yield, contemporary sediment yields are lower than they were during the period of most intensive land disturbance in the 19th century (Brush, 2009; Langland, 2015; LSRWA, 2015; Noe et al., 2020a; Wolman, 1967). Previous reports have concluded that sediment itself is not the primary problem for attainment of Bay water quality goals and nutrients should be the focus of the pollution reduction efforts (Miller et al., 2019). While having many potential adverse impacts on Bay habitats and water quality (Noe et al., 2020b), sediment impacts are arguably more pronounced in Bay watershed tributaries than in the Chesapeake Bay itself. Sediment can also facilitate increased nutrient transport and loading, although the extent to which this is a major driver of changes in N and P loads is highly variable and episodic (Ator et al., 2020).

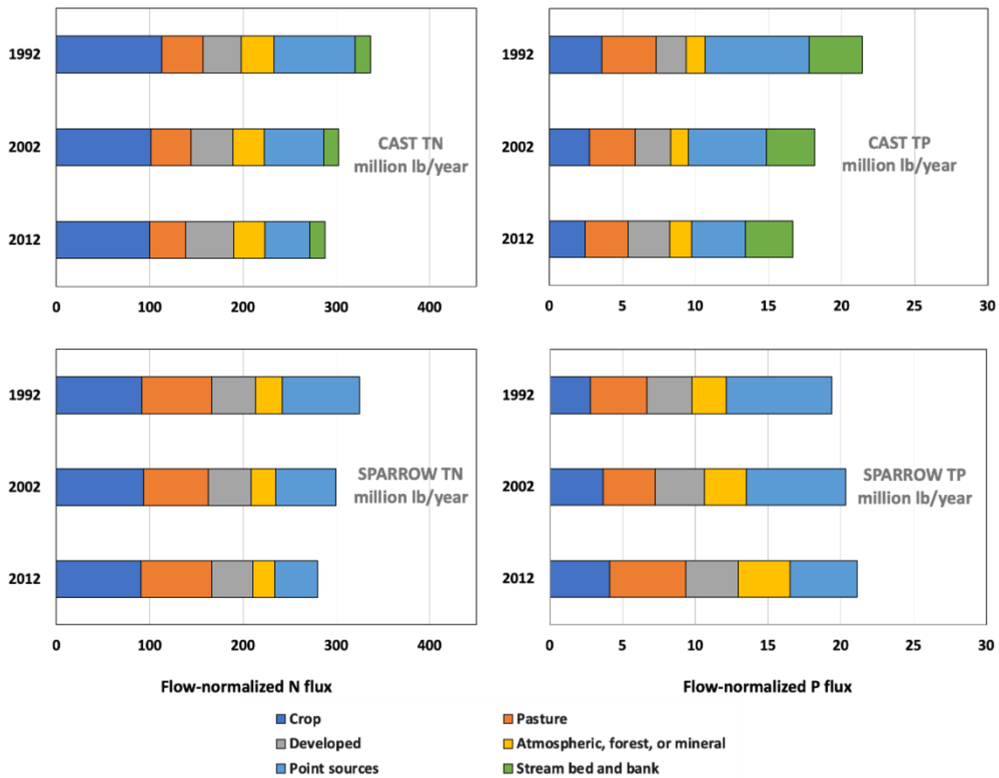


**Figure 2. Estimated flow-normalized total and source sector total nitrogen (TN) and total phosphorus (TP) fluxes to the Chesapeake Bay for selected years for the CAST and SPARROW models (Source: adapted from Ator et al. [2020]).**

The challenge of realizing and sustaining large reductions in nonpoint source loads is not unique to the CBP. Studies of individual BMPs, or studies of BMP nonpoint source reduction efforts conducted at a fine scale (i.e., edge of field, headwater basin) with intensive monitoring, have shown BMP implementation can reduce nutrient and sediment loads to streams. Ator et al. (2020) cite numerous examples of BMPs and management actions effective at reducing nutrients at fine scales, including (1) declining nutrients in streams or contributing groundwater observed in response to reductions in reduced agricultural inputs, such as through animal waste removal (Ferris et al., 2010) or declining fertilizer applications (Böhlke and Denver, 1995; Denver et al., 2010); (2) reduction of N contribution to shallow groundwater from cover crops (Staver and Brinsfield, 1998); (3) phytase and composting or pelletizing of poultry manure may be particularly effective at reducing P mass imbalances (Ward and Ritter, 2003); and (4) urban BMPs that increase infiltration or detain stormwater have been effective in reducing nutrient loads (Jefferson et al., 2017; Li et al., 2017). However, demonstrating the effectiveness of control efforts at larger scales has proven more difficult (Lintern et al., 2020; Osmond et al., 2012; Sprague and Gronberg, 2012; Tomer and Locke, 2011). The lack of observable reductions in nonpoint source loads is one of the most fundamental and common challenges confronting large-scale water quality programs (Boesch, 2019).

A number of explanations have been offered for the limited water quality response to nonpoint source reduction efforts (referred to here as BMP implementation). Two possibilities often cited are: (1) BMP implementation is working effectively but the results have not yet been fully detected, and (2) BMP implementation and related load reduction policies are not working as expected (Ator et al., 2020; Lintern et al., 2020; Osmond et al., 2019; Tomer and Locke, 2011). What follows is a discussion of the possible reasons and challenges that might explain why pollution reduction efforts to date, chiefly BMP implementation, have not produced the expected system response and associated water quality improvements.

## 2.2. Control efforts are effective, but response has not yet been detected

Agricultural and urban landscapes are complex systems, composed of a mosaic of land uses, landscape characteristics and hydrologic settings that may delay the expected BMP effects. Agricultural and urban BMPs are typically designed to use natural hydrologic and chemical processes to reduce pollution by reducing/altering pollutant inputs (e.g., decrease fertilizer use, store manure), altering pollutant transport pathways (e.g., increase infiltration and percolation), or promoting pollutant treatment/transformation (e.g., denitrification, phosphorus sorption, sediment trapping). These BMP reduction processes can take time to be measurable, depending on numerous factors, including pollutant travel times, physicochemical and biogeochemical processes that alter pollutant concentrations during transit, groundwater residence times and pathways, the time it takes for BMPs to become fully effective, and challenges related to monitoring networks and trend analyses. The result of these interacting factors makes detecting reductions in pollutant loads due to BMP implementation difficult, especially in higher order streams and riverine systems.

### *Legacy nutrients and sediment and lag times mask BMP effectiveness*

A National Research Council report cautioned that achieving Bay water quality goals could be significantly delayed by legacy nutrients (NRC, 2001). The accumulated stores of legacy nutrients and sediment have been identified as an important factor that accounts for the lack of observable water quality improvement (Chang et al., 2021; Kleinman et al., 2019; Noe et al., 2020b; Sharpley et al., 2013; Stackpoole et al., 2019). Legacy nutrients—resulting from past human activity and subsequent storage in soil, sediment, or groundwater—introduce a lag in time between present-day changes in nutrient inputs and observable reductions in loads delivered to downstream waters, often preventing the attainment of water quality improvement goals (Sharpley et al., 2013). In many areas of the watershed, N inputs to the landscape continue to exceed crop needs, increasing the storage (or load) of N in the environment (Sabo et al., 2022). In fact, U.S. Geological Survey (USGS) monitoring suggests, groundwater nitrate and orthophosphate concentrations have not declined in the Potomac River watershed or in the Delmarva Peninsula from late 1980s/early 1990s through the early 2000s/2010s (Lindsey et al., 2020; Lindsey and Rupert, 2012). Reducing nutrient inputs and implementing BMPs designed to treat or transform nutrients may produce rapid reductions at the point of BMP implementation, but their benefit can take years or even decades to propagate through the coupled surface water-soil-groundwater system, resulting in significant lag times between BMP implementation and downstream water quality response (Böhlke, 2002; Hirsch et al., 2013). Thus, legacy nutrients continue to be a source of nutrients to surface water bodies even as contemporary nutrient loads are reduced or eliminated (presumably due to BMP implementation).

Legacy N exists in both groundwater and soils. Groundwater modeling within the Chesapeake Bay watershed shows elevated concentrations of nitrogen (in the form of nitrate) stored in groundwater in several regions (figure 3). A recent study found that legacy N is an important element of the contemporary N load (Van Meter et al., 2017), responsible for supplying about half of all water and N delivered to streams in the Chesapeake Bay watershed (Focazio et al., 1998; Phillips and Lindsey, 2003). For instance, in the Susquehanna River Basin, legacy N (> 1 year residence time) in groundwater was found to contribute nearly 50% of the N load entering the Chesapeake Bay, while the remaining N losses were attributed to faster flow pathways (< 1 year residence time) that wash off recent N fertilizer applications and mobilize short-term storages of N in shallow groundwater (Van Meter et al., 2017). Because it is relatively mobile in the soil-water system, nitrate can move quickly through the soil profile into groundwater where it can reside for decades, creating a lagged response to BMP implementation (Sanford and Pope, 2013; Sanford et al., 2012), although Chanat and Yang (2018) noted that over a multidecadal period the average effects of legacy N may not outweigh the importance of contemporaneous changes in inputs and climatic effects. In the Chesapeake Bay watershed, the transit time for groundwater discharged into surface water ranges from less than a year to more than 50 years (Lindsey et al., 2003; Meals et al., 2010; Phillips and Lindsey, 2003; Sanford and Pope, 2013; STAC, 2013). In fact, some of the longest lag times are in the most intense agricultural regions (Delmarva, Shenandoah Valley, Southern PA). These areas, that have the highest nutrient yields, may be most affected by lag times.

The CAST model does not explicitly attempt to model groundwater lags, which may account for some of the differences between modeled nutrient load estimates and monitored observations. Different modeling approaches have been tried in the Chesapeake Bay watershed and elsewhere (Basu et al., 2022; Harman et al., 2016), but spatial and temporal variations of hydrologic flow paths and travel times have been difficult to characterize.

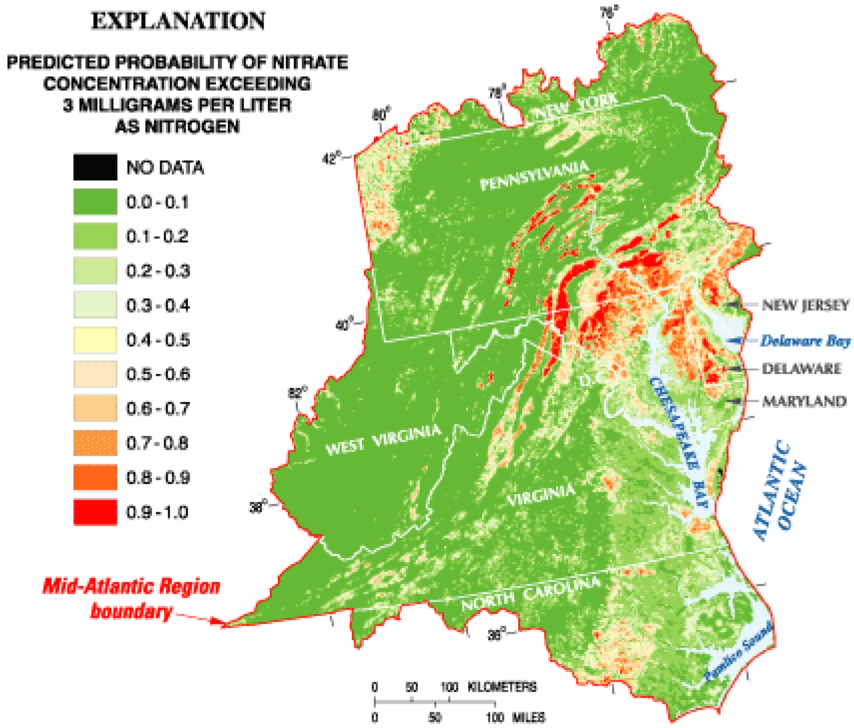


**Figure 3. Predicted probability of groundwater nitrate levels exceeding 3 mg/L (Source: Greene et al., 2004).**

Excess N inputs can also cause gradual accumulation of N in solid organic matter in soils (Sebilo et al., 2013; Van Meter et al., 2016). Reversal of such trends is possible, but the accumulated soil N can continue to contribute to groundwater and stream export for years. Measurements and predictions of changes in soil N reservoirs are highly uncertain; estimates for agricultural soils in the Mississippi basin suggest that it would take three decades to draw down soil organic N, even assuming the complete cessation of N application (Van Meter et al., 2016). Similarly, legacy sediment stored in riparian areas contains large pools of N, the bioavailability of which varies with surrounding land use (Noe et al., 2013; Weitzman et al., 2014).

Legacy P presents a more persistent challenge to water quality management and protection (Kleinman et al., 2019; Staver et al., 2014). In many areas, legacy P loads may overwhelm the P loads from other sources (Kleinman et al., 2019). Legacy P is stored primarily in soils, and depending on the soil characteristics and management, P loss can occur by both surface and subsurface pathways. In areas with intensive livestock production (e.g., poultry, dairy), animal manures are typically land-applied as fertilizer. Animal manures are high in P relative to other plant nutrients, and, as a result, P has been historically applied at rates that exceed crop needs, creating a buildup of P in soils. Historically, P management focused on so-called “soil conservation” strategies, as P is typically tightly bound to sediment, and P is lost to surface water via eroded sediments in runoff. In high P soils, however, P loads to surface water can be both sediment-bound and in biologically available dissolved forms (Kleinman et al., 2019). The increasing importance of dissolved P losses from legacy P in soils creates challenges when P management strategies focus solely on erosion and runoff control. The challenge of remediating legacy P is that significant P stores in soils can serve as a constant source contributing to dissolved P losses in surface runoff and shallow subsurface flow (Kleinman et al., 2019; Sharpley et al., 2013). This background P loss makes detecting the impact of BMPs implemented on the landscape difficult.

Drawing down high P soils is a critical strategy for reducing legacy P in soils, but studies suggest this will take time. For instance, Fiorellino et al. (2017) estimated that for high P soils (Mehlich-3 P > 200 ppm), it would take between 18 and 44 years for P levels to reach optimal agronomic levels (Mehlich-3 P < 150 ppm), assuming no additional P application. Lower antecedent soil P levels and specific cropping rotations, however, could result in a more rapid P drawdown. Yet, given that high P soils tend to occur in areas with intensive livestock production, and given the amount of manure generated and the relatively high cost of transporting manure, continued, localized manure applications will likely extend the time needed to reduce the impact of legacy P. In addition, tracking changes in soil P is made more difficult given confidentiality and privacy issues associated with reporting soil test P levels.

Phosphorus is also stored in eroded legacy sediments in riparian areas, which, through erosion and exchange processes, can generate large continued P loads downstream (Noe et al., 2022). These sediments tend to have relatively low to moderate total nutrient concentration levels in comparison to surface soils (Inamdar et al., 2020; Noe et al., 2020b), although there are watersheds in areas of intensive agriculture with potentially large concentrations of total P in streambank sediments. There is considerable uncertainty about the importance of this P source for Bay water quality when compared with other sources or forms of P. The relative bioavailability of P stored in legacy sediment compared to other sediment-associated P, and changes in P bioavailability over time in sediment storage zones, needs to be resolved. Furthermore, sediment (including legacy) can remove or release phosphate from the stream water column through reversible sorption processes depending on sediment characteristics and phosphate concentrations of the sediment and water (Inamdar et al., 2020).

Sediment management efforts also face the challenge introduced by legacy sediment, which also introduces potential lag times in system response. Legacy sediment is defined as sediment from erosion from land disturbing activities that has become stored in uplands, flood plains, and/or stream channels (e.g., sediment trapped behind mill dams) (Miller et al., 2019). It is estimated that historical erosion rates prior to World War II were considerably higher than estimated contemporary erosion rates (Portenga et al., 2019). A large portion of that legacy sediment is still stored on hillslopes, footslopes, and valley floors throughout the Chesapeake Bay watershed (Jacobson and Coleman, 1986; Smith and Wilcock, 2015). For example, in a study of a 155 $km^2$ watershed in the piedmont of Maryland, Costa (1975) estimated that 52% of material eroded following European settlement was still stored on hillslopes, 14% was stored in floodplains, and 34% had been exported from the watershed.

The extent to which legacy sediment can be remobilized and transported to the Chesapeake Bay is highly variable in both time and space (Miller et al., 2019). Stored sediment may experience long lag times (years to millennia) before exiting the watershed, and it is possible that legacy sediments may be eroded and deposited more than once, with typical event transport length scales for sand and finer particles in the mid-Atlantic region ranging from 4 to 60 km and 0.4 to 113 km, respectively (Pizzuto et al., 2014). In some cases, this may mean that continued erosion of legacy sediment will generate elevated sediment loads well into the future, despite reduced sediment inputs from upland areas (Jackson et al., 2005). Uncertainty about sediment delivery lag times has implications for the time scale on which one might expect to see downstream impacts of upstream sediment control efforts.

In some locations, streambank erosion of legacy sediment stored in valley deposits can be a major contributor to downstream sediment load (Noe et al., 2020a). There are documented examples where breached dams have led to rapid headward (upstream) incision and evacuation of stored sediment in just a few years (Miller et al., 2019). The effectiveness of practices to address legacy sediments is subject of ongoing study. One approach gaining increasing attention involves removal of riparian area legacy sediments to prevent downstream transport (Forshay et al., 2022; Langland et al., 2020). Such projects have demonstrated water quality improvements. Sediment removal in Big Spring Run in Pennsylvania, for instance, demonstrated nitrate concentration reductions of 2-3 mg/L, but these reductions attenuated quickly downstream of the site (Forshay et al., 2022). There is not yet enough information to quantify long-term engineering reliability or accumulated benefits across the Chesapeake Bay watershed (Miller et al., 2019).

It is important to note that uncertainties associated with legacy sediment should not divert attention from contemporary control of erosion (Miller et al., 2019). There is empirical evidence to suggest that even large watersheds may respond on shorter time scales to changes with the adoption of land use practices designed to control contemporary erosion. For example, for the Susquehanna River, Langland (2015) estimated that annual sediment load upstream of the three reservoirs along the lowermost portion of the river declined from 8.5 million tons/yr for the period 1941–1950 to 3.5 million tons/yr for the period 1991–2012. This trend was attributed to implementation of soil conservation practices and diminished mining activity. Preliminary analysis comparing post-1985 flow-normalized loads with decadal load averages on the Susquehanna at the Marietta gages suggests that reductions have indeed occurred on a 50-year time scale.

*Existing level/scale of monitoring is insufficient to detect a signal*

Another potential explanation for the seeming lack of response to management efforts is that current monitoring networks and available analytical tools may not be capable of detecting subtle water quality changes that result from BMP implementation. For instance, water quality trends are derived from monitoring data collected at 123 nontidal stations in the Chesapeake Bay watershed. These stations are mostly high order streams and rivers; potential BMP effects in upstream areas may be difficult to detect at the outlet of these large watersheds. Traditionally, N, P, and sediment concentrations are measured by a combination of periodic sampling (e.g., monthly, bi-weekly) and samples targeted to capture high flows, and subsequent laboratory analyses. More recently, high-frequency water quality sensors (Burns et al., 2019; Duncan et al., 2017; Pellerin et al., 2016; Rode et al., 2016) have demonstrated that this periodic sampling may underrepresent temporal nutrient and sediment dynamics. Nutrient and sediment loads are commonly computed with regression-based techniques by combining daily streamflow data with concentration measurements (Hirsch et al., 2010). For example, Weighted Regressions on Time, Discharge, and Season (WRTDS) (Hirsch et al., 2010) and similar methods were developed to exploit legacy data sets, and loads computed from such methods contain some statistical uncertainty, which may affect trend interpretations and an ability to identify BMP effects. If sensors of the constituent of interest, or of surrogates for those constituents (e.g., turbidity for suspended sediment), are used, the ability to detect and describe trends can be greatly enhanced, but there must be investment in modern instrumentation and development of appropriate statistical protocols for this new type of data to yield reliable and defensible water quality trends.

The signal of BMP effects could also be overwhelmed by hydrologic signals or “noise” in the system. Streamflow varies greatly at time scales ranging from hours to days (precipitation events) to months (seasonal water balance, weather patterns), to years and decades (climate change, consumptive withdrawals). Constituent concentrations also vary on similar time scales and can be strongly influenced by streamflow. Relationships between pollutant or constituent concentrations and streamflow (so called C-Q relations) can be complex, and they differ from place to place and for different constituents (Burns et al., 2019). For example, sediment and sediment-bound P concentrations are positively correlated with flow, such that a few relatively high-flow events can dominate the annual P export from some watersheds. Trend methods attempt to minimize interannual streamflow variability by modeling C-Q relations (referred to as flow-normalization), but the resulting trends may not fully remove such effects (Hirsch et al., 2010). Typically, trends in N, P, and sediment loads cannot be detected on time scales less than 10 years, especially if anomalous hydrologic extremes (e.g., large floods or long-duration droughts) only happen once during the monitoring period, making it impossible to evaluate how the water quality response to these extremes may have changed over the monitoring period. Variability in hydrologic conditions, another source of noise, can also make it difficult to detect a BMP implementation signal. For example, discharge of 1000 cfs at a monitoring site may be associated with snow melt one day, a precipitation event on another day, and the contribution by groundwater on yet another. For each of these situations, the water quality may be quite different. Even a Before-After-Control-Impact sampling design may not be able to detect long-term cumulative reduction in nutrient and sediment concentrations smaller than 20% (Liang et al., 2019). New methods that account for source and transport variability can help to better detect BMP signals and water quality trends (Hirsch et al., 2010).

One potential means to determine BMP effectiveness is to focus on smaller, more intensively managed systems where the BMP footprint occurs across a larger area of the watershed. While larger streams are somewhat less noisy than smaller streams with respect to variations in flow and load, BMP implementation typically occupies a small area of the watershed. To detect a change in water quality, nonpoint source control actions must be implemented in a large fraction of the watershed. If only small parts of the watershed have had BMPs implemented, it will be virtually impossible to detect the BMP water quality signal, no matter how sophisticated the analysis. As such, water quality improvements stemming from BMP implementation are likely to be more reliably detected at finer scales such as edge-of-field and lower-order, headwater streams (Bishop et al., 2005).

## 2.3. Nonpoint source BMPs and policies are less effective than predicted

If legacy nutrients/sediments and lag times are primarily responsible for the limited detection of reductions achieved to date, then one policy response would be to simply extend expectations for when the pollutant reduction goals will be achieved and continue with existing implementation. A more fundamental management concern, however, relates to whether policy and agricultural and urban BMPs are less effective than predicted. Several hypotheses may account for the response gap that exists between predicted and observed load reductions in the agricultural and urban sectors, including: (1) incomplete accounting of the distribution and quantity of surplus nutrient inputs across the watershed, (2) overestimation of BMP pollutant control effectiveness, (3) misalignment between BMP effort and watershed areas with high loads, and (4) incomplete understanding of behavioral response to load reduction policy.

### *Regional nutrient mass imbalances*

From a system perspective, nutrient mass balances are critical for determining the nutrient status of a given region. Nutrient mass balances quantify the input of nutrients to a system (i.e., livestock feed, fertilizer, atmospheric deposition), the outputs of nutrients from the system (i.e., agricultural products harvested, losses to water or air), and changes in storage pools (i.e., soil, groundwater, plants). When nutrient inputs exceed exports, excesses must either accumulate in the system (e.g., increased soil P levels or higher concentrations of N in groundwater) or be lost to surface runoff, groundwater discharge, or volatilization to the atmosphere. Large nutrient mass imbalances exist in several regions of the Chesapeake Bay watershed, and in most of these regions these imbalances are directly associated with increasing nutrient losses. While mass imbalance issues can also be high in many urban areas, the most severe and extensive cases of mass imbalances are associated with regions dominated by intensive agriculture (Ator et al., 2011; Bryant et al., 2022; Keisman et al., 2018b; Sabo et al., 2019, 2021). While the general location of these regions is well known, uncertainties surround the quantity, distribution, and use of manure and commercial fertilizers in these areas. Substantial and sustained reduction in these areas is unlikely unless these regional mass imbalances can be corrected.

The expansion of Bay hypoxia is also closely tied to the nutrient mass balance problem. The hypoxic volume in the Chesapeake Bay began to expand steadily in the 1950s (Kemp et al., 2005). This expansion directly coincides with the significant increase in imported commercial fertilizer for crop production and feed for commercial livestock in the post-war period (Kemp et al., 2005). Between 1950 and 1982, soil N and P inputs from commercial fertilizer and manure increased 90% and 13%, respectively (Keisman et al., 2018b). Since 1982, modest declines in overall soil N inputs and larger reductions in P inputs have occurred, driven largely by reductions in commercial fertilizer use. However, livestock manure use as a fertilizer has gradually increased and the distribution of livestock has become more concentrated. Across the Chesapeake Bay watershed, livestock produce about 10 times the total volume of excrement when compared to the total human population, and many times more nutrients (Ator and Denver, 2015; Kleinman et al., 2012). Human waste is treated to remove the majority of nitrogen and phosphorus (among other constituents), with vast systems to transport, treat, and dispose of this waste; however, these processes are not currently implemented for livestock waste.

Nutrient inputs are unevenly distributed, and in many places, and nutrients have become more concentrated over time (Keisman et al., 2018b). Between 1982 and 2012, nutrient inputs increased in regions such as the lower Susquehanna valley, Delmarva Peninsula, and Potomac/Shenandoah basins. Increases in these areas that already receive higher than average nutrient inputs are being driven largely by increases in livestock numbers (Ator and Denver, 2015; Kleinman et al., 2012) and agricultural intensification. According to the U.S. Department of Agriculture (USDA) National Agricultural Statistics Service (NASS) (USDA NASS, 2017), poultry numbers in the Lower Susquehanna and on the Delmarva increased from 2002 to 2017 by 20.5 million (from 37 million, a 64% increase) and 70.8 million (from 143 million, a 66% increase), respectively. In these regions, the volume of manure and fertilizer inputs exceed the localized crop needs, contributing to legacy nutrient buildup. In fact, Sabo et al. (2022) found surplus amounts of nutrients in agricultural areas of the Chesapeake Bay watershed have increased from 2009 through 2019. These recent increases reverse longer-term patterns, where surplus inputs in 2019 were lower than in 1985.

In areas with nutrient mass imbalances, the extent to which conventional on-farm BMPs (e.g., cover crops, tillage practices, buffers) can make substantial long-term nutrient reductions without addressing the mass imbalance itself is limited (Ator et al., 2020; Beegle, 2013). Many conventional agricultural BMPs that are integral elements to watershed implementation plans do not appreciably change the mass balance. For instance, P removal efficiencies for cover crops, conservation tillage, and riparian buffers tend to be a result of altering transport pathways, increasing temporary nutrient storage, or altering the form of nutrients. In areas with mass imbalances, some BMPs (e.g., conservation tillage and no-till practices) can accelerate nutrient losses, particularly highly reactive soluble forms (Duncan et al., 2019; Kleinman et al., 2019). For instance, the widespread adoption of conservation tillage in the Lake Erie watershed in the 1980s has been cited as a principal driver of increased dissolved P losses from agriculture (Joosse and Baker, 2011; Richards et al., 2002), and has contributed, in part, to the re-eutrophication of Lake Erie (Watson et al., 2016). Analysis of water quality trends in 53 monitoring stations found that regions with high and increasing trends in biologically available forms of P were associated with agricultural areas with manure applications and conservation tillage (Fanelli et al., 2019). Addressing existing mass imbalances would require reductions in the animal numbers in the watershed, reductions the import of feed and fertilizer, increases in the export of excess nutrients out of the region/watershed, such as hauling manure out of the watershed (Spiegal et al., 2020), or the use of manure conversion technologies (Kleinman et al., 2012; Sharpley et al., 2013). Precision feed management, practices intended to optimize livestock diets to reduce overfeeding of nutrients and to maximize use of feed grown on farms, has shown consistent reductions in manure nutrient content (Cerosaletti et al., 2005; Plumstead et al., 2007). Population growth and consumer preferences also drive animal intensification in the region.

Considerable uncertainty still exists regarding the quantity and spatial distribution of manure/fertilizer applications and their impacts on nutrient losses in areas with nutrient mass imbalances. In general, fertilizer inputs and manure production and application data are available on larger (more coarse) spatial scales (i.e., state and county levels). The quantity of manure produced in the watershed is based on estimates of animal numbers reported from NASS surveys, which may undercount total livestock populations. The actual distribution of nutrient inputs significantly impacts loading rates and BMP effectiveness. However, there is limited information on how, when, and where land managers actually apply fertilizer and manure in the Chesapeake Bay watershed at the farm- or field-scale (Yagow et al., 2016). A recent study of dairy farms in Virginia's Shenandoah Valley showed that the range of fertilizer and manure application rates can vary substantially across farms (Pearce and Maguire, 2020). The study, which estimated P mass balance for 58 dairy farms (nonrandom sample), showed that surplus P (P imports minus P exported in products and manures) ranged from -30.9 to +97.6 kg/ha. The median positive P imbalance (i.e., surplus) of 12.4 kg/ha is directly associated with increasing soil P levels in most soils. The CBP must make assumptions about how manure and fertilizer is then distributed at finer spatial scales (i.e., within a county). The CAST model assumes nutrients are applied according to crop needs and nutrient management plans. These assumptions reflect best case scenarios with respect to nutrient losses, particularly when it comes to manure which is costly to transport. Assumptions about nutrient use and distribution are a primary reason CAST is unable to capture observed trends, particularly for P.

Data describing urban area nutrient use and inputs are even scarcer, but there are areas where better data or accounting could improve the understanding of the contribution of urban landscapes to water quality. For instance, fertilizer use in urban areas is highly variable, and often the data do not reflect differences attributable to application rates made by homeowners versus landscapers (Carrico et al., 2018). Likewise, data often do not reflect recent legislative bans of residential P fertilizer, although the limited evidence that does exist suggests fertilizer bans provide, at best, limited water quality improvements (Hochmuth et al., 2012). Finally, given the mosaic of land uses in urban areas, the contributions from sources such as wastewater inputs (septic effluent, sanitary sewer line exfiltration) and industrial/commercial sources make inputs difficult to quantify.

*BMP performance effectiveness estimates*

BMP effectiveness assumptions used in watershed models (Liu et al., 2017; Y. Liu et al., 2018) may partially explain why predicted water quality improvements may not be reflected in monitoring data. This issue is not unique to the CBP. Osmond et al. (2012) noted that models used in USDA's Conservation Effects Assessment Project (CEAP) consistently overestimated BMP effectiveness, and recent review articles by Lintern et al. (2020) and Kleinman et al. (2022) support this assertion, noting that the greatest proportion of studies showing improvements in water quality from BMPs emanated from the modeling literature, with field and watershed monitoring studies showing mixed or little to no improvement due to BMP implementation.

The CBP uses a partnership-approved process to develop BMP nutrient/sediment removal estimates that are then used in CAST to predict nutrient/sediment load reductions associated with BMP implementation. In this process, panels of experts typically rely on a mixture of existing literature (peer-reviewed and grey) and best professional judgment to estimate BMP performance, typically defined as a percentage (or fractional) removal efficiency. For most BMPs, the CBP program then assigns a single nutrient/sediment removal efficiency estimate to be used across the watershed. In assigning BMP removal effectiveness, the CBP does not systematically assess or document the sources and relative magnitude of BMP performance uncertainties (Stephenson et al., 2018) or the range of possible effectiveness.

Aggregating and generalizing isolated field-level studies into aggregated estimates of removal effectiveness is analytically challenging and uncertain (Lintern et al., 2020). BMP studies may only focus on specific nutrient and/or sediment removal processes and may not fully evaluate the impacts of alternative removal and/or sequestration pathways (Allaire et al., 2015; Banaszuk et al., 2013; Heeren et al., 2010; Vellidis et al., 2001). For instance, the majority of BMP performance studies focus on surface water impacts and rarely consider impacts on groundwater. Studies that evaluate BMP effectiveness by measuring edge of field water quality typically focus on surface losses, which can neglect to account for the export of nutrients by subsurface pathways; these pathways are often responsible for large contaminant fluxes, particularly for N. This is one reason edge of field studies tend to overestimate the effectiveness of BMPs in the literature. Expert panels that rely on this literature must evaluate and interpret literature where percent removal rates can range from 90 percent to less than zero percent (i.e., the BMP produces pollutants). BMP efficiency estimates often have multiple embedded assumptions (e.g., hydrologic setting, site-specific conditions, maintenance levels) that may not reflect real world conditions (Aguilar and Dymond, 2019; Stephenson et al., 2018; Strecker et al., 2001). As a result, expert panels must often extrapolate beyond individual, site-specific, field-scale studies and rely on best professional judgment to develop nutrient/sediment reduction efficiency estimates.

Further, the existing literature, which forms the basis for establishing efficiency estimates, may not be reflective of overall BMP performance (Liu et al., 2017), and is unlikely to be representative of actual implementation conditions for the wide range of physiographic/ management settings in the Chesapeake Bay watershed. For many BMPs, the available literature often focuses on evaluating performance under highly controlled, site-specific conditions. In addition, the published literature may underrepresent instances when BMP studies report findings of no or negative pollutant control effectiveness because such studies are more difficult to publish.

Scientific evaluations of BMP performance typically assess performance a few years after installation and under well-maintained conditions. For certain types of BMPs, pollutant control performance will likely vary over time. Many common BMPs (e.g., many structural urban stormwater BMPs) function by temporarily storing nutrients or sediment, either in the soil or in plant biomass. Yet the storage capacity of a BMP is ultimately limited, and the fate of those stored nutrients/sediments over time is not well characterized. If stored nutrients are more likely to be mobilized as the BMP matures, the pollutant reduction performance of the BMP will be overestimated over time (Hopkins et al., 2022). Most structural BMPs require some type of routine maintenance to sustain their performance. BMPs subject to scientific study are likely to be subject to scheduled maintenance, whereas BMPs implemented in real world settings may not. Additional research could help to better understand structural BMP performance over time and under varying levels of maintenance representative of observed maintenance levels (Liu et al., 2017). A related question involves the effect of reservoirs (particularly the three reservoirs at the downstream end of the Susquehanna watershed) that act as huge retention BMPs storing and cycling nutrients and sediments. How these reservoirs act to cycle and store nutrients and sediments is, at best, poorly understood; thus effort to understand their effectiveness as a trap for N, P, and sediment is likely to evolve over time. Also, knowledge of how physical manipulations of these systems could potentially enhance their ability to trap N, P, and sediment in the future is needed.

Expert interpretation of existing literature may also be prone to instances of BMP effectiveness overestimation. Behavioral research finds that even experts can often be prone to errors that systematically err on the side of overconfidence, particularly when risk and uncertainty are involved (Stephenson et al., 2018). For instance, people tend to assign causal explanations to randomly produced outcomes. This possibility is increased in data-poor settings often faced by expert panels.

**Text Box 1: Septic systems, P and the Chesapeake Bay TMDL**

The most common form of onsite wastewater treatment is the septic system, consisting of a primary treatment component (e.g., settling tank, mechanical mixing, aeration) and a soil adsorption field where effluent is dosed. General design requirements for conventional soil adsorption systems specify that there should be at least 1 m between the septic effluent distribution pipes and seasonal high groundwater depth (Otis, 1980a, 1980b). Adequate soil depth, adequate aeration, and distance from water sources are critical features of well-functioning systems. In the Chesapeake Bay TMDL crediting system, N reductions attributable to well-functioning septic systems range from 20–70%, depending on level of treatment employed; however, P treatment is assumed to be complete (i.e., 100% removal) (Adler et al., 2014).

Septic performance in practice is highly variable and can generate substantial losses, even for P. Even in properly sited and functioning systems, the risk of offsite transport is substantial (Meeroff et al., 2008; Ouyang & Zhang, 2012; Robertson, 1995; Robertson & Harman, 1999). Phosphate plume concentrations in groundwater have exceeded drinking water standards (2 mg/L) at documented distances greater than 25 m from septic disposal fields (Wilhelm et al., 1994). Corbett et al. (2002) found P concentrations 2.5 times background levels in septic plumes >50m from the drain field. In regions of the Chesapeake Bay watershed with unsuitable site conditions (shallow soils, high water table), export of P can be even greater (Collick et al., 2006). Assuming that septic systems provide 100% P removal efficiency introduces a discrepancy that may account for some of the divergence between CAST predictions and observed water quality trends, especially in areas with rapid suburban growth. As an aside, CAST also assumes zero leakage from sanitary sewer lines, which can be an important source of nutrients in urban areas (Kaushal et al., 2011).

Further, using the existing literature, it is often difficult to assess BMP performance under extreme conditions. BMPs are not typically designed to accommodate large (i.e., less frequent, longer return period) storm events despite the fact that most nutrient/sediment movement (loss) occurs during these large events (Hopkins et al., 2022; Selbig and Bannerman, 2008). As a case in point, Hirsch (2012) estimated that N, P, and sediment loads from the remnants of Tropical Storm Lee (September 7–15, 2011) comprised 31, 61, and 78% of the total N, P, and sediment loads delivered to the Chesapeake Bay in water year 2011 (October 1, 2010 to September 30, 2011). Notably, the return period for Tropical Storm Lee on the Susquehanna River was roughly 20 years using the maximum daily discharge on September 9, 2011 (Hirsch, 2012). The impact of large storm events like Tropical Storm Lee on BMP performance and nutrient/sediment loads reaching the Chesapeake Bay are not typically considered in CBP management decisions.

*Spatial distribution of BMPs with respect to pollutant sources*

Numerous studies have found that spatial targeting BMP implementation to sites with higher pollution potential can improve effectiveness and reduce costs of pollution reduction efforts (Giri et al., 2012; Kast et al., 2021; Khanna et al., 2003; Lintern et al., 2020; Xu et al., 2019; Yang and Weersink, 2004). Researchers have noted that areas of high nutrient and sediment loss are site-specific and highly localized (Easton et al., 2017). Inadequate BMP implementation in areas of high nutrient loss may fail to produce expected load reductions. Many studies suggest that 5–20% of the land area generates 50–90% or more of runoff and loads, particularly P and sediment loads (Bello et al., 2019; Easton et al., 2007, 2008a,2008b; Heathwaite et al., 2000; Qui, 2009; Rao et al., 2009, 2012; Wagena and Easton, 2018; White et al., 2009; Xu et al., 2019). Within agricultural fields and urban areas, nutrient losses may be confined to relatively small, hydrologically active areas that, with targeted BMP adoption, might be more effectively reduced. Nutrient loss rates also vary across land managers (Pearce and Maguire, 2020).

While CAST can identify high loading areas at a relatively coarse spatial scale, it does not reflect localized high-loss areas at finer spatial scales that could potentially benefit from more targeted BMP implementation (Easton et al., 2020; Lintern et al., 2020). Within CAST, most nutrient and sediment reductions are calculated based on multiplying estimates of nutrient load (lb/ac) from a land use by the BMP removal efficiency and the number of acres served by the BMP (figure 4), over a land river segment which typically ranges from 9,000 to 12,000 acres. Recently, SPARROW has been used to estimate areas of the watershed with disproportionally large nutrient losses (Ator and Garcia, 2016), which could inform targeted BMP implementation.

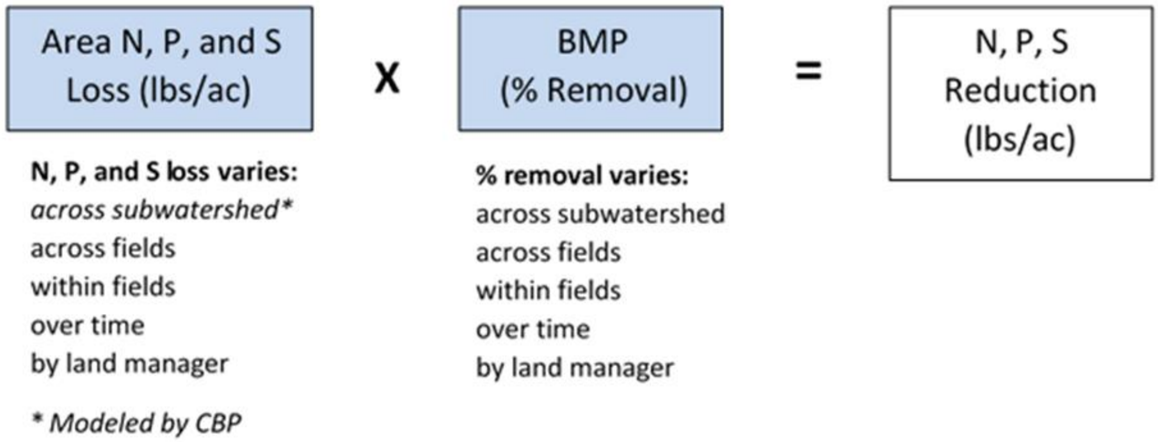


**Figure 4. Conceptual model of how the Chesapeake Assessment Scenario Tool (CAST) estimates Best Management Practice (BMP) load reductions.**

Presently, the CBP does not incentivize jurisdictional partners to adopt and potentially benefit from targeted BMP implementation. Given existing crediting, Bay jurisdictions cannot claim additional reduction credits by identifying localized high-loss areas within a specific land use and geographic region. Bay jurisdictions have no programmatic-oriented incentive to identify land managers generating disproportionate loads because within the CBP accounting framework all land within a specific geographic area (i.e., land river segment) are assumed to produce the same nutrient and sediment loss. Since a given BMP generally counts the same regardless of where it is placed within a subwatershed, state and local jurisdictions may use other criteria to place BMPs on the landscape, such as convenience and ease of implementation or greater co-benefits. This often means that, within agricultural areas, service providers such as Natural Resources Conservation Service (NRCS) or Soil and Water Conservation Districts (SWCDs) may only work with cooperative land managers, which may, as a group, already have relatively low nutrient/sediment loss rates. In the case of urban stormwater BMPs, “retrofit” intervention measures and urban stream restoration are often implemented on public property (e.g., schools, libraries, parks) or land with lower density development where there are few conflicts with public utilities and other infrastructure. This type of BMP placement does not reflect the diversity of contaminant sources and pathways in the watershed and may leave large pollutant reductions unrealized.

Targeting sediment reduction BMPs is challenging because the magnitude and sources of sediment in specific locations can be very uncertain (Noe et al., 2020a). Any assessment of the relative importance of different sediment sources contributing to watershed sediment loads requires that the different types of sources (various upland as well as alluvial sources in the stream valley) and storage zones are quantified. This information is needed in order to determine which mitigation or management measures are likely to be most effective and where they should be located. Sediment budgets of Chesapeake watersheds have indicated highly variable upland sediment yields delivered to streams among different basins (Allmendinger et al., 2007; Hopkins et al., 2022; Noe et al., 2020b; Smith and Wilcock, 2015). It is much easier to measure rates of bank erosion and evaluate the contribution to watershed sediment budgets from stream banks than is the case for the more diffuse sources of surficial or upland erosion. The fact that the resulting value represents a large fraction of watershed sediment yield has sometimes been interpreted as indicating that bank erosion is indeed the dominant source, but as indicated above this may not be a safe assumption if upland erosion is not also considered.

**Text Box 2: Predicting sediment erosion, transport, and fate is challenging**

Sediment reduction targets in the Chesapeake Bay TMDL are predicated, like those for total N and P, on the predictions of CAST. As documented in the review of the Phase 6 model (Easton et al., 2017) and in the Visioning Workshop on Chesapeake Bay Modeling in 2025 and Beyond (Hood et al., 2019), the sources, erosion rates, transport pathways, storage locations and residence times, rates of remobilization from storage, and ultimate fates of sediment are subject to many sources of uncertainty. While predicting sediment transport in river channels is notoriously difficult, it is only a small part of the problem of making predictions that quantify upland erosion, rates of transport, and travel times from source to sink.

The team of scientists conducting the Phase 6 model review called for major improvements to the simulation of sediment dynamics by the Chesapeake Bay model. Notably, the team stated in its recommendations: "Current scientific understanding is not sufficient to accurately quantify the relevant processes, for example, to make predictions of lag times and delivery rates for sediments at the watershed scale with a reasonable degree of confidence. Therefore, the Phase 6 modeling approach should be regarded as an interim solution with the expectation that improved scientific understanding will allow a more comprehensive approach in Phase 7." This finding was echoed in the recommendations from the Visioning Workshop (Hood et al., 2021), which included the following: "Design and implement a replacement model that better represents a new understanding of sediment dynamics. This process should start with the development of a conceptual model to rank governing processes (and how they change across time and space) in the watershed. This would provide guidance for the development of a new computational model that would represent sediment processes and time scales."

### *Uncertainty in behavioral response to nonpoint source control measures*

To a significant degree, Bay jurisdictions rely on voluntary incentive-based programs to induce agricultural nutrient load reductions. Financial incentives aim to encourage adoption of specific practices ("practice-based" incentive programs) by sharing the costs of BMP installation. The behavioral response to agricultural policies can have implications for the overall effectiveness of achieving load reductions. Relatively little research has been conducted on how participation in conservation planning varies across agricultural land managers (Patterson et al., 2013; Reimer and Prokopy, 2014). Voluntary, incentive-based agricultural BMP implementation programs are, by definition, self-selected by participants in the programs. To accurately estimate the influence of BMP effectiveness, it is necessary to understand how these incentives shape adoption behavior. The type and degree of participation, however, may produce systematic overestimation of the nutrient control effectiveness of BMPs. There is also the argument, though, that BMP adoption is systematically undercounted because there may be participants who do not seek incentives for adoption. Indeed, Nelson and Spies (2013) found that 38% of BMPs in the Upper Chester, Maryland watershed were not funded with cost sharing over the 2010–2012 period.

Typically, voluntary, incentive-based financial assistance programs only partially compensate land managers for the costs of installing and maintaining/operating BMPs. The current structure of the voluntary programs is more likely to engage specific segments of land managers, and the composition of participating land managers would likely be different under alternative incentive programs (Shortle and Horan, 2017; Talberth et al., 2015). For example, research generally finds that conventional financial assistance programs encourage the adoption of practices with significant private benefits and low upfront costs, for example conservation tillage and cover crops (Claassen et al., 2014; Fox et al., 2021; Pineiro et al., 2021). Given that many practices produce net costs, existing programs tend to solicit the participation of land managers with strong social and ethical motivations (Prokopy et al., 2019; Ribaudo, 2015). These managers may be more willing and able to incur a portion of the costs to improve public water resources. Land managers motivated more by financial considerations may be less likely to participate in voluntary, incentive-based BMP programs. In other instances, largely because of cultural norms, several animal-intensive production regions in the Chesapeake Bay watershed have relatively high numbers of land managers (e.g., plain sect communities in Pennsylvania and Virginia) who do not accept the government assistance payments or cost sharing that are part of the existing incentive programs (Brock et al., 2018). Farms that are land constrained, small, or resource-limited in these areas could have systematically lower BMP adoption rates (Prokopy et al., 2019). Furthermore, a significant amount of farmland is rented, and BMP adoption rates tend to be lower in such situations (Ranjan et al., 2019). Arguably, our understanding of which BMPs can potentially provide exceptional benefits and where those BMPs should be located exceeds our understanding of landowner concerns and adoption behavior (Reimer et al., 2014).

If BMPs tend to get clustered on lands with high adoption rates and high nutrient use efficiency, then BMPs can exhibit declining incremental nutrient removal effectiveness. BMPs placed on land already treated by a BMP or on well-managed land with below-average pollutant losses may yield less pollutant load reduction than expected from application of a BMP under assumed "average" conditions. To the extent this occurs, overall BMP effectiveness can be overestimated given how CAST accounts for BMPs.

Effectiveness of BMPS also depends on BMP maintenance (Aguilar and Dymond, 2019; Hood et al., 2019). However, relatively little is known about how BMP maintenance occurs in practice. In theory, structural BMPs (e.g., livestock exclusion fencing, buffers, animal waste facilities) installed under an incentive-based program contract are maintained throughout the life of that contract. Compliance with the terms of such a contract are typically verified via an assessment by the contracting entity through some sort of periodic spot checking of a random subsample. However, the maintenance of many critically important BMPs is difficult to monitor over the long term.

Source control BMPs, like nutrient management, conservation planning, and cover crops can be used to achieve nutrient load reductions. These practices rely heavily on assumptions about behavioral compliance, and verifying the nature of behavioral compliance is challenging. For example, cover crops are effective at reducing nutrients by scavenging residual nutrients, but effectiveness may be limited if farmers add additional fertilizer to boost yield or till the cover crop, mobilizing sediment. Several studies suggest that a relatively small percentage of farmers completely follow nutrient management plans (Osmond et al., 2015; Ulrich-Schad et al., 2017). Claassen et al. (2014) compared farms with and without nutrient management plans (NMPs) and found statistical evidence that having a NMP reduces fall fertilizer application rates, but there was no conclusive statistical evidence of overall differences in fertilizer use between the two groups.

With respect to maintenance of structural BMPs, research in other regions of the country suggests that the lack of sustained BMP maintenance is a relatively frequent problem that can adversely affect BMP performance (Jackson-Smith et al., 2010). Similarly, relatively little is known about the maintenance of urban stormwater BMPs. While technically subject to periodic inspection, many municipal stormwater programs face challenges with tracking, inspecting, and enforcing BMP maintenance (Aguilar and Dymond, 2019). Studies that do exist generally have found that structural urban stormwater control practices have a host of maintenance needs (Hirschman et al., 2009; Li, 2015), and Li (2015) found that over half of all discharges from 279 stormwater control BMPs along a regional Maryland highway system were untreated.

### 2.4. Closing the implementation gap

The above discussion highlights some possible reasons why predicted pollution reduction efforts are not reflected in observed instream nutrient and sediment loads. Yet, even if BMP performance perfectly matched our predictions, significantly higher levels of BMP implementation would be needed to achieve and maintain the nutrient and sediment reductions called for in the Chesapeake Bay TMDL. A key question remains: To what degree can our current implementation programs generate the level and type of participation needed to achieve the reduction goals specified in the TMDL? This question is particularly challenging for programs that depend largely on voluntary participation, as in the unregulated agricultural and urban sectors.

There is limited empirical evidence about the levels and limits of participation in existing incentive-based (cost-share) BMP implementation programs (Reimer and Prokopy, 2014). While local success stories exist, voluntary, incentive-based programs targeting the agricultural sector have not consistently generated the level (or type) of BMP implementation sufficient to produce the reductions needed to achieve water quality standards (Prokopy et al., 2019; Ribaudo and Shortle, 2019). The BMP adoption literature consistently refers to the challenges in identifying factors that explain adoption of BMPs, or lack thereof (Prokopy et al., 2019; Ranjan et al., 2019; Reimer et al., 2014b). A number of factors have been proposed to increase adoption within the structure of existing voluntary, incentive-based agricultural programs. Research indicates that BMP adoption is often dependent on personal relationships between land managers and conservation staff (T. Liu et al., 2018). To encourage BMP adoption, federal and state incentive-based agricultural pollution control programs rely on technical staff to work with land managers on developing site-specific conservation. The lack of funding for technical service providers and financial assistance (cost sharing) has been identified as a significant challenge to achieving the degree and type of BMP implementation necessary to achieve further pollutant load reductions (Chesapeake Bay Commission, 2017). The CBP has also sought to increase BMP implementation by highlighting that nonpoint source BMPs often produce multiple benefits to local citizens, so called "co-benefits" (McGee et al., 2017; Wainger et al., 2013). For instance, stream restoration and urban tree canopy offer aesthetic, recreational, and property value benefits, and adoption of nutrient reduction practices can produce local water quality improvements. The extent to which these potential co-benefits can translate into behavior change and increased rates of BMP adoption is an area of ongoing area of study.

Despite these efforts, little historical evidence exists in the Chesapeake Bay watershed or elsewhere to show that conventional, voluntary, incentive-based programs can generate and sustain large-scale nutrient load reductions (Prokopy et al., 2019; Ribaudo and Shortle, 2019; Shortle et al., 2012, 2021; Stephenson et al., 2022). One challenge is the structure of the incentive-based program can limit adoption of cost-effective BMPs. Conventional incentive-based programs are designed to induce land managers to adopt BMPs by paying for a portion of the cost to install a BMP. Farmers tend to adopt BMPs that also generate farm-level benefits (e.g., reduced input costs, improved soil conditions, better animal health, etc.) more than BMPs that do not. Yet, BMPs with few financial benefits can potentially generate large, low-cost pollutant reductions. Practices that reduce the mass of nutrients applied in a watershed (such as manure transport, nutrient management practices, residential fertilizer restrictions) are likely to be more effective than BMPs that attempt to control nutrient transport (Ator et al., 2020; Bryant et al., 2022), but these BMPs can be more expensive or provide little additional benefit. Riparian buffers and denitrifying bioreactors are also BMPs that offer the potential for large nutrient reductions but suffer from high cost and little additional benefit. Furthermore, the structure of existing incentive-based programs is not designed to identify and target BMP implementation on lands with the highest reduction potential or the lowest abatement costs.

Multiple policy alternatives that could be pursued to potentially alter behavior and improve the effectiveness of voluntary incentive-based programs do exist (Ribaudo, 2015; Shortle et al., 2012). For example, pay-for-performance or pay-for-success programs pay land managers for the nutrient/sediment reductions achieved. Unlike existing incentive-based programs which pay a portion of the cost to install a BMP, a pay-for-performance program would pay land managers for the mass of pollutant reduction achieved. Conceptually, pay-for-performance programs would allow land managers to profit from pollution reduction efforts if reductions can be provided at a cost lower than the performance payment. Such a system would likely generate a different adoption profile than conventional incentive-based programs by encouraging participation from high-loss, low-cost land managers and encourage the adoption of practices with significant public water quality benefits. There are numerous ways such programs can be designed (Easton et al., 2020; Fleming et al., 2022), including different payment options and different approaches to quantifying outcomes (combinations and types of monitoring and modeling approaches). The limited application of pay-for-performance programs in existing water quality management programs, however, makes estimating the actual behavioral outcomes from implementing pay-for-performance programs uncertain.

Despite numerous approaches to boost participation in incentive-based programs, some policy analysts have questioned whether existing voluntary, incentive-based programs can produce the needed pollution reduction goals (Ribaudo and Shortle, 2019; Shortle et al., 2012; Stephenson et al., 2022). In light of the shortcomings of existing programs, the role of mandatory programs to control nutrient loads has also received attention given the limited progress achieved through voluntary cost-share based programs (Shortle et al., 2012). Given the diversity of agriculture in the Chesapeake Bay watershed, any mandatory BMP implementation effort would need to be tailored to accommodate specific circumstances. In the urban sector, recent bans or restrictions on residential phosphorus uses offer promise, but the effectiveness of these bans depends partly on rates of compliance (Hochmuth et al., 2012; Smidt et al., 2022).

## 3. Delivery of Nutrients and Sediments to the Chesapeake Bay

Some emerging issues related to the characteristics of nutrient and sediment delivery through the river system to the Chesapeake Bay have potential implications for achievement of water quality standards. These include the changing forms of nutrients and the impact of major impoundments in the system.

Nutrient speciation (i.e., the chemical composition and bioavailability of nutrients delivered to the Chesapeake Bay) has important implications for both the amount and type management effort necessary to achieve water quality standards. Not accounting for the differential impact that different nutrient species have on the ambient water quality response in the Chesapeake Bay could mean that the total nitrogen (TN) and total phosphorus (TP) reduction targets in the TMDL could be met while the dissolved oxygen (DO), water clarity, and/or chlorophyll *a* (Chl *a*) endpoints are not. Even if TN and TP reduction goals are met, additional management efforts may be necessary to reduce biologically available forms of nutrients. There may also be implications for the type of management efforts needed to adequately manage increasing bioavailable forms of nutrients. Efforts to more clearly define the eutrophication potential of different forms of N and P have the potential to improve the effectiveness of management efforts. Yet, our current understanding of where, when, and how much biogeochemical processing affects the fate and transport of the various forms of N and P is rudimentary. Additional research could help to more clearly understand nutrient speciation, fate, and transport (Shenk et al., 2020).

The nutrient reduction targets in the Chesapeake Bay TMDL focus on reducing TN and TP, yet the bioavailable forms of these nutrients pose the most significant challenge to achieving water quality standards (Shenk et al., 2020). Tributaries that deliver large N loads are generally dominated by bioavailable nitrate, while tributaries that deliver lower N loads tend to have a larger fraction of lesser bioavailable dissolved organic nitrogen (DON) (Schmadel et al., 2019; Zhang et al., 2018). From 2011 through 2020, TN loads increased at 42 percent of Chesapeake Bay nontidal monitoring stations while nitrate loads increased at 69 percent of stations. A similar pattern was observed between TP and ortho P load trends: TP loads increased at 23% of stations and ortho P loads increased at 21% of stations (Mason et al., 2023). In many tributaries, the level and proportion of P that is entering in a bioavailable dissolved form (ortho P) is increasing (Shenk et al., 2020). For example, particulate P represents a large portion of the P delivered to the Chesapeake Bay. However, dissolved P, which represents an estimated 15–20% of total P load, drives phytoplankton growth (Shenk et al., 2020). Increases in ortho P levels come primarily from agriculture (Fanelli et al., 2019). Phosphorus fractionation is an important issue for understanding P bioavailability, P transport, and P measurement. It is difficult to measure sediment-bound P, and each method selectively extracts different forms of P. Some analyses report *total P* using partial extraction techniques that do not represent all forms of P. The differing partial extraction methods for bioavailable P extract a different combination of forms of P, adding to uncertainty. Furthermore, P compounds in the extractable P fractions undergo dynamic biogeochemical and physical processes that can change their bioavailability.

Reservoirs also play a critical role in form, timing, and delivery of nutrients and sediment to the Chesapeake Bay. The Conowingo Dam, the most downstream dam and largest reservoir on the Susquehanna River, illustrates some of these challenges. Reservoirs offer substantial, but finite, sediment and nutrient trapping potential. In the case of Conowingo, the sediment and nutrient trapping potential was reached earlier than expected (Cerco, 2016; Hirsch, 2012; Langland, 2015). The cycling of some forms of nutrients, like dissolved P, within the reservoir and their release from the reservoir is an emerging concern, but is poorly understood. The quantity and form of nutrients and sediments released by Conowingo may necessitate additional nutrient and sediment pollution control efforts.

**Text Box 3: The case of Conowingo**

Conowingo Dam is the furthest downstream of three dams that were built along the lower Susquehanna River between 1910 and 1931. It occupies a unique position among all such structures in the Chesapeake Bay watershed because it modulates the flux of water, sediment, and nutrients from nearly half of the land area draining to the Chesapeake Bay. For decades the Conowingo Dam and reservoir trapped (reduced) nutrient and sediment loads from the Susquehanna River watershed, functioning as a BMP. However, since the reservoir is essentially filled, nutrients and sediment are no longer being trapped, but are passed downstream to the Chesapeake Bay. The response of the CBP to the recognition of reduced Conowingo reservoir storage capacity and associated increase in nutrient and sediment loads to the Chesapeake Bay has been to develop a separate Watershed Implementation Plan (WIP) to compensate for the increased loads and mitigate the water quality impacts.

Hydrologists have long understood that the Conowingo storage capacity was approaching dynamic equilibrium, a condition where the mass of sediment entering the reservoir would be the same as the mass exiting when averaged over a period of years (figure 5). Evidence published over the past decade shows that the Conowingo has reached this condition (Hirsch, 2012; Langland, 2015). Figure 5 shows that over the period from 1995 to 2011 as sediment storage was continuing to increase in the reservoir the TP trapping efficiency declined, leading to an increase in flow-normalized TP load. In 2011, Tropical Storm Lee caused a significant amount of scour and export, thereby increasing the trapping efficiency and lowering the flow-normalized TP load from the reservoir over at least the next decade. This is the type of variability that is expected from dynamic equilibrium, a sequence of multi-year periods of filling resulting in increased TP export, followed by a major scour event and improved trapping, ultimately leading to another period of increasing loads until such time as another major scour event takes place. Another factor that is likely contributing to the current trend is that flow-normalized loads from upstream (Susquehanna River at Marietta, Pennsylvania) have declined by 13% over the 2011–2020 period. Also, contributing to this downward trend are the results from the Conestoga River at Conestoga, Pennsylvania, the largest monitored tributary of the Susquehanna between Marietta and Conowingo, which show an estimated decrease of 3% over the last decade. The Lower Susquehanna River Watershed Assessment (LSRWA, 2015) examined multiple sediment management options and concluded that sources of nutrients upstream of the reservoir have far more impact on the Chesapeake Bay than sequestered nutrients associated with sediments in the reservoir.

Palinkas et al. (2019) investigated the potential impacts of Conowingo infill on the Chesapeake Bay and concluded that nutrients produced by large rainfall events (scour) would have only marginal impacts on dissolved oxygen in the Bay. The findings of Palinkas et al. (2019) become significant here in that the P trapped during these filling phases may ultimately move out of the estuary relatively rapidly with these large pulses, minimizing the impact on Bay DO. Accounting for these scour and fill processes in the CBP program can help in evaluating the impacts of future control strategies in the Susquehanna River watershed, supported by an active monitoring and research effort on the inputs to, internal processes of, and outputs from the reservoirs.

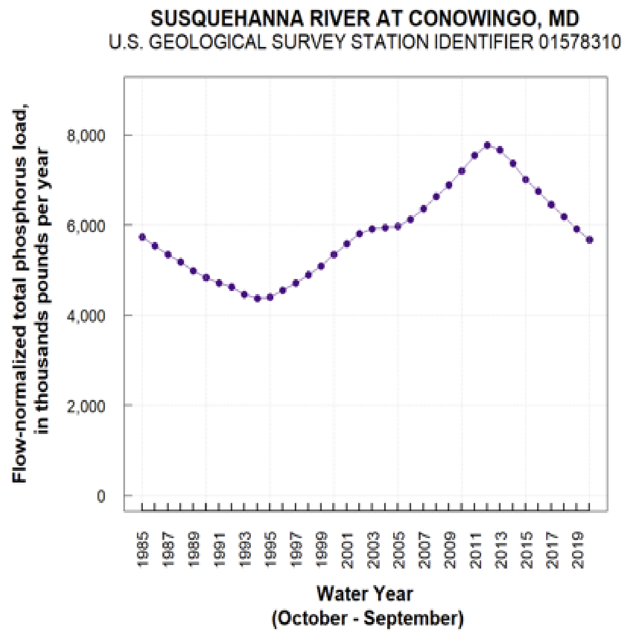


**Figure 5. Flow-normalized total phosphorus (TP) loads at Conowingo, Maryland for the period 1985–2020 (USGS, 2023).**

## 4. Confronting the Future: Challenges and Uncertainties in Managing Nutrient and Sediment Loads

Future external factors and trends such as climate change, population growth, increasing urbanization, and agricultural intensification may have a large, but uncertain, impact on the nutrient and sediment loads coming from the Chesapeake Bay watershed. For instance, accelerated conversion from fossil fuels to renewables, especially in the transportation sector, could further decrease atmospheric N deposition. Changing consumer preferences for protein and dairy products could produce shifts in agricultural production, a major source of nutrient loads. Increases in storm intensity from climate change could lead to increased pollutant loads to the Chesapeake Bay. Long-term changes create uncertainty about future Bay pollutant load trajectories and about the type and level of management needed to address pollutant loads going forward. A comprehensive review of these uncertainties is beyond the scope of this report, but can be illustrated through discussion of the potential impact of climate change on nutrient and sediment controls.

Climate change poses an array of challenges to meeting the Chesapeake Bay TMDL nutrient/sediment reduction targets. Some of these challenges, such as increased streamflow, are widely recognized for their potential to increase nutrient/sediment delivery to the

Chesapeake Bay. Indeed, the TMDL Phase III Watershed Implementation Plans (WIPs) now require all Bay jurisdictions to account for the additional nutrient and sediment loading expected from climate change through 2025. In order to understand how climate change is affecting system response at the Chesapeake Bay level, it is important to characterize the ways in which climate-induced changes to the watershed may be benefitting, offsetting, or even negating the effects of management actions implemented to achieve the Chesapeake Bay TMDL.

The primary climate-related drivers affecting the Chesapeake Bay watershed are air temperature, precipitation, and sea-level rise. Changes in these drivers are expected to alter key processes within the Chesapeake Bay and its watershed, including evapotranspiration, soil moisture, streamflow, terrestrial and aquatic biogeochemistry, water temperature, salinity, estuarine circulation, and water quality variables such as water clarity, Chl *a*, and DO (Najjar et al., 2010). Climate change can also affect watershed water quality by indirect means, such as by increasing the length of the growing season, which can result in changes in agricultural land use, and increasing the opportunity for agricultural intensification, such as double cropping. This could fundamentally alter the nutrient mass balance and, as a result, the cycling and export of nutrients in ways we do not fully understand. Increased air temperature and precipitation are already thought to have decreased flow-normalized N loads from the watershed over a multidecadal period, possible because of increased denitrification (Chanat and Yang, 2018).

Precipitation is one of the key climatic variables that not only controls watershed discharge, but also influences internal nutrient cycling processes, and the potential for increased nutrient/sediment export from the watershed. According to Easterling et al. (2017), mean annual precipitation in the mid-Atlantic region increased by 5–10% from the historical period (1901–1960) to the 1986–2015 period. These findings dovetailed with recent observations by Rice et al. (2017) showing that precipitation increased throughout the Chesapeake Bay watershed from 1927–2014, with northern regions of the watershed exhibiting increases on the order of 6–15%. Notably, studies by Sinha and Michalak (2016) and Ballard et al. (2019) indicated strong linkages between increasing precipitation and N export to the Chesapeake Bay. Moreover, a study by Ryberg et al. (2018) suggested that annual precipitation was a key driver of P loads to the Chesapeake Bay, and that increases in precipitation could already be offsetting management actions to reduce P loss. Indeed, studies in other regions of the United States also suggest that annual precipitation is an important control on N (Bowles et al., 2018; Sinha et al., 2017), P (Ockenden et al., 2016, 2017), and sediment (Vidon et al., 2013) export. In cases where increasing precipitation is increasing nutrient loading to the Chesapeake Bay (Ballard et al., 2019), jurisdictions might need to implement additional management practices to mitigate these trends (Rice et al., 2017; Ryberg et al., 2018), as indicated in the Phase III WIPs.

It is important to note, however, that aggregate trends in precipitation do not reflect changes in rainfall distributions, particularly the duration, frequency, and magnitude of extreme events. Such changes are of concern when it comes to nutrient and sediment loss. For example, precipitation intensity has been on the rise in the United States (Mallakpour and Villarini, 2017) and throughout the Northeast (Huang et al., 2017). According to Easterling et al. (2017), the amount of annual precipitation falling in the heaviest 1% of daily events increased by 55% in the northeastern United States, faster than any other region in the nation. These changes in precipitation intensity can affect patterns of nutrient and sediment loss. Not surprisingly, nutrient losses from extreme precipitation have important implications for agricultural BMP performance (Renkenberger et al., 2017). Extreme rainfall is also problematic in urban areas, as impervious areas amplify runoff generation, which then overwhelms stormwater infrastructure (Moglen and Vidal, 2014) and potentially diminishes nutrient load reductions from urban BMPs (Hopkins et al., 2022; Selbig and Bannerman, 2008). Increased water and air temperatures can also impact nutrient (and sediment) cycling and transport, with increased stream temperatures resulting in greater ortho P and nitrate concentrations (Rice et al., 2017). Conversely, warmer air temperatures can improve BMP nutrient removal rates (Lintern et al., 2020), although there are many complicated interactions among climate and nutrient/sediment cycling and transport that make extrapolating conclusive results exceedingly difficult.

The uncertainty of BMP performance with climate change is a critical factor in understanding future changes in Bay water quality. BMPs are designed and implemented with consideration of expected patterns of precipitation variability, including extreme events. In the decades to come, changes in climate present additional uncertainty and risk to BMP performance. Long-term changes in climate and extreme weather can have implications for BMP siting, design, and maintenance strategies that seek to minimize a BMP's vulnerability to structural failure during its design life (Johnson et al., 2018). BMPs function through a variety of mechanisms, including physical retention (storage), filtration, chemical conversion, and biological uptake. These mechanisms determine the sensitivity of BMPs to different climate drivers (e.g., rainfall volume and intensity, temperature, soil moisture, etc.). Higher pollutant loading from urban and agricultural lands to BMPs could alter BMP pollutant removal effectiveness, requiring resizing/redesign, or the need for additional BMPs to meet water quality goals (Hanson et al., 2022; Wagena and Easton, 2018). Climate change could also alter physical and biological processes (e.g., denitrification) affecting the ability of BMPs to reduce pollutant loading.

Additional research could help to inform the selection, design, and siting of cost-effective BMPs that are resilient to anticipated long-term changes in hydroclimatic conditions (Bosch et al., 2018; Hanson et al., 2022; Williams et al., 2017; Xu et al., 2019). Specifically, areas for advancement include: (1) design guidance to increase BMP resilience (e.g., standards for considering the impacts associated with extreme weather and climate into BMP siting and design); (2) improved simulation modeling capabilities for BMPs; (3) targeted research to quantify the impacts of climate change on BMP effectiveness; and (4) improved methods to evaluate siting and design considerations within the watershed context, in addition to site-level assessment needs (e.g., including BMP cost-effectiveness and co-benefits) (Johnson et al., 2018).

## 5. The Adaptive Management Framework

An adaptive management approach is designed to address uncertainties in water quality management, particularly with respect to pollution (Freedman et al., 2008; NRC, 2001). The CBP has implemented a decision framework intended to continuously monitor and evaluate progress toward achievement of specific CBP program goals and adjust implementation based on these assessments. The CBP's Goal Implementation Teams (GITs) are responsible for implementing the decision framework. More generally the GITs are responsible for overseeing and promoting the implementation of plans to achieve the goals of the CBP. The Water Quality GIT, responsible for water quality and TMDL implementation, consists of state and federal agency staff and technical experts.

Fully realizing the decision framework is an iterative, ongoing process. Over the past several years, significant progress has been made in defining goals, describing factors influencing goal attainment, and assessing management gaps. GIT work has focused on identifying metrics for documenting strategy action implementation (e.g., tracking and verifying BMPs). In the most recent reporting of the Chesapeake Bay decision framework, the Water Quality GIT has identified many factors influencing water quality goal attainment including improving nutrient/sediment source identification, increased monitoring, the influence of climate change, prioritization of research needs, and improved understanding of uncertainty in CAST predictions.

Several important challenges remain in the CBP efforts to implement the adaptive management framework. The challenges focus on development of specific steps of the adaptive management framework: monitoring, assessing performance and strategy development, and adaptation. An important limitation of existing adaptive management is management-related decision-making under uncertainty. This includes the analytical tools and processes needed to identify decision-relevant uncertainties and the "most-probably-effective" actions considering uncertainty. Because the resources for pollution control action implementation are limited, strategy development also requires prioritization of uncertain actions with the objective of maximizing potential outcomes at lowest possible cost.

Another gap in the CBP adaptive management process is more effective assessment of management/BMP efficacy. Here the question is not simply: Are we undertaking the planned actions? Rather: Are the actions producing the anticipated outcomes? There is widespread acknowledgment that additional monitoring of ambient water quality is necessary, but an effective adaptive management process needs to identify appropriate monitoring metrics to assess system response and inform future actions. This is difficult because most strategies incorporate multiple actions, and determining what works and what does not work in a compendium of actions is not always straightforward. Determining what monitoring reveals about current program efficacy, and what the implications are for future program goals, is both a critical and difficult undertaking. Ultimately, there is a need for objective expert advice in the assessment step of the decision framework.

Finally, the adaptive management process requires not only assessing the efficacy of existing practices and programs, but also developing and testing alternatives that address uncertainties and improve program outcomes. To date, the CBP decision framework operates within the confines of the existing management programs and CBP accounting framework. Yet, new modeling tools to improve identification and treatment of nutrient/sediment sources and inform different management programs and incentive policies that can induce more effective behavioral change are all potential improvements to program effectiveness. These alternatives offer new opportunities to advance learning and improve program effectiveness but may represent different approaches, tools, and programs than those currently used by the CBP.

The CBP faces several challenges in implementing the evolutionary dimension of adaptive management. The CBP TMDL accounting framework serves many useful purposes but the framework itself generates its own internal set of behaviors and incentives. The accounting framework tracks BMP implementation and credits progress toward meeting TMDL nutrient/ sediment reduction targets through the CAST framework. There is a very limited ability to assign differential credit of BMPs implemented in high loss areas. Since BMP efficiencies are generally established as a default estimate, there are limited incentives to alter (increase or decrease) removal efficiencies of BMPs installed on the landscape. Finally, the accounting framework focuses CBP partners' attention on counting BMPs at the expense of monitoring to evaluate BMP performance and assess water quality outcomes.

A commitment to more active adaptive management includes budgetary and management commitments. The participants primarily responsible for implementing the CBP adaptive management process do not typically have the authority or incentive to make budgetary or program commitments. This represents a current gap and future opportunity for collaboration in developing a more effective adaptive management process.

## 6. Conclusions and Addressing Challenges

Achieving CBP nutrient/sediment reduction goals depends on whether nonpoint source load reduction targets can be achieved. Nonpoint source loads are generated from the decisions and behavior of millions of individuals living in the watershed. Crafting policy to substantially alter those behaviors to effectively reduce diffuse loads is challenging. Monitoring to assess the effectiveness of BMPs implemented to reduce pollution is confounded by many factors, including temporal factors such as how BMPs perform over very brief intervals during and after storm events, the mechanisms by which BMPs sequester or reduce pollutant loads, the complexity of the environment in which BMPs are implemented, and the maintenance of implemented BMPs. Other confounding factors, such as legacy nutrients, current landscape activities, and climate change further impact achieving applicable water quality standards.

Achieving needed nutrient/sediment reduction targets, from both the agricultural and urban sectors, may face many challenges. In recent years, the CBP partnership has focused on achieving N reductions. According to CAST estimates, an additional 43 million pounds of N reductions are needed, primarily from nonpoint sources. As a relative comparison, the CBP estimates that nonpoint source N loads have been reduced by 3 million lb/yr between 2009 and 2021. Empirical estimates from the SPARROW model suggest that most N load delivered to the Chesapeake Bay is from nonpoint source pollution sources (Ator et al., 2020). While empirical estimates from the SPARROW model and CAST both suggest that N loads delivered to the Chesapeake Bay have declined in recent decades, these models disagree about changing P loads. The CAST model estimates decreasing P loads while the SPARROW model estimates increasing P loads (Ator et al., 2020). Based on water quality data from 2011 through 2020, TN, TP, and suspended sediment loads did not decline in most monitoring stations throughout the Chesapeake Bay watershed (Mason et al., 2023). Another consideration for the CBP is nutrient speciation, particularly for P, where an increase in the dissolved P fraction has the potential to degrade water quality.

Uncertainties surrounding achieving required load reductions fall into two categories: (1) Are the BMPs that are being implemented to reduce loads as effective as expected? And (2) Can existing policies generate the type and level of behavioral response needed to achieve sufficient BMP implementation to close the load reduction response gap?

The limited load reductions to date may be due to a variety of reasons, including lag times in both BMP maturation and the release of legacy nutrients/sediments in ways not controlled by BMP implementation, the inability of the existing monitoring networks to detect reductions that are the result of BMP implementation, the overestimation of the effectiveness of BMPs, and an incomplete accounting of the magnitude and distribution of nutrient/sediment inputs to the system. Isolating the reasons why observed water quality indicators are not responding to load reduction efforts is difficult, complex, and uncertain. From a management perspective, the best-case scenario is that BMPs are working, but lag times and monitoring limitations are delaying and/or masking a water quality response. However, the evidence suggests that BMPs and policies designed to implement those BMPs are not as effective as expected.

Whether existing programs can generate the level of implementation needed to meet and maintain reduction goals is uncertain. To put the nutrient load reduction response gap into perspective, CAST estimates that it took nearly 35 years (1985 to 2019) to achieve 25 million pounds of N reduction in loads. To meet the TMDL N load reduction target, CAST estimates the CBP partnership must have enough practices in place by 2025 to reduce an additional 43 million pounds of N, although that reduction is assumed to occur at some point after 2025. The BMP adoption challenge is particularly acute for the agricultural sector, which still largely relies on voluntary, incentive-based programs to achieve BMP implementation targets. There is little evidence to suggest that these voluntary, incentive-based programs have produced the magnitude and scale of reductions called for under the Chesapeake Bay TMDL or, for that matter, other similar, large-scale water quality restoration efforts (e.g., Gulf of Mexico hypoxia and Great Lakes eutrophication).

The cumulative evidence suggests that continuation of existing policies alone is not likely to produce the nutrient/sediment reductions needed to attain Bay water quality standards. There is, however, evidence of ways that policies and programs could be altered to achieve additional pollution reduction. The analytical and policy challenge is to identify and explore plausible programmatic alternatives, even though there is uncertainty about how they may translate into water quality improvements, and then to use the information gained from implementation to improve program effectiveness. There are several areas where evidence suggests concerted efforts could be made:

*The mass balance issue*: Without adequate policies to reduce nutrient inputs or substantially change exports, nonpoint source load reduction efforts may be of limited effectiveness. More effective and systematic approaches to addressing nutrient mass balance issues offer opportunities for substantial, sustained reductions in nutrient loads. A mass balance approach describes inputs (e.g., fertilizer and feed) to and outputs (e.g., grain or meat export, loss to water bodies) from the system, reactions or transformations (e.g., denitrification), and storages (e.g., buildup of soil P) in the system. Most BMPs do not substantially alter mass balances. Evidence suggests that policies designed to alter regional mass balances have proven particularly effective in improving water quality. The P detergent ban, the Clean Air Act, and wastewater treatment technology investments to increase denitrification are examples from point source management. Nonpoint source reduction BMPs designed to limit the amount of nutrients added to the landscape are likely to be more effective at achieving nutrient reductions than those that attempt to control the movement of nutrients from the landscape to streams (Ator et al., 2020).

Another challenge faced by the CBP is addressing existing mass balances associated with intensive animal agriculture to reduce agricultural loads. Evidence suggests that increasing P loads are closely linked to intensive livestock operations, and given the trends towards more intensive livestock operations, traditional BMPs (e.g., cover crops, conservation-tillage, nutrient management plans) that do not substantially alter inputs or transformations are unlikely to alter outputs (e.g., nutrient losses). Although any efforts that more consistently match nutrient applications with plant requirements can optimize nutrient uptake, sequestration and harvest can be effective in moderating mass imbalances. Potential mechanisms to address the nutrient mass balance issues include:

- Reducing, or optimizing, the nutrient content of livestock feed. Evidence suggests that efforts like this, commonly known as precision feed management, have resulted in greater than 30% reduction in manure nutrient content, ultimately reducing the mass of nutrients available for loss.
- Manure utilization, transport, treatment, and conversion technologies (e.g., thermochemical or microbial transformation) offer the potential to significantly alter the mass balance by transforming manure-derived nutrients into forms that are either more recalcitrant or more easily transportable. This will likely require significant and creative investment to develop centralized manure transport and treatment systems.
- Increasing on-farm manure storage. Even though increased on-farm storage would not alter the overall nutrient mass balance, except for some chemical species conversions, increased on-farm manure storage could substantially improve water quality by preventing manure application to frozen or saturated soils or at times of high runoff risk.

*Targeting BMP investments*: Existing voluntary BMP implementation programs typically rely on the willingness of landowners and financial incentives to drive participation. These voluntary, incentive-based programs are not amenable to targeted implementation. However, evidence suggests that implementing BMPs on areas of the landscape that produce the most pollution is essential to achieve pollutant reduction goals. In this sense, *targeting* is used broadly to include both the identification of high-loss areas (due to both landscape features and individual land manager behavior) and site-specific selection of BMPs (i.e., some mechanism to more effectively direct who implements what BMPs and where). Implementing more effective targeting may require changes in the CBP partnership's policies, programs, and incentives and the CBP modeling tools and associated accounting protocols.

- The CBP may consider developing mechanisms to allow and encourage differential crediting of BMP efficiency, incentivizing the identification and treatment of high-loss areas and agricultural operations. The CBP accounting framework counts and credits loads based on averages (i.e., average loading rates, average BMP effectiveness, average nutrient application rates). Differential crediting could be accomplished in a number of ways including a finer-scale targeting/modeling system, or by granting flexibility to state and local partners to develop, test, evaluate, and quantify the performance of BMP alternatives that target high-loss areas. This will necessitate finer-scale modeling capabilities that can identify areas of the landscape contributing disproportionate nutrient and sediment loads.
- The CBP may consider developing and evaluating new incentive mechanisms to encourage land managers to identify low-cost, high-impact practices and/or behavioral traits conducive to positive change. New incentives and programs can help to encourage and engage land managers to incentivize the adoption of cost-effective practices in high nutrient/sediment loss areas. These include payment for environmental service programs that compensate land managers for nutrient/sediment reductions achieved and ambient bonus systems that reward achievement of observable ambient benchmarks (e.g., soil nutrient levels, ambient water quality conditions, etc).

*Strengthening adaptive management*: The CBP may consider more explicitly recognizing, managing, and seeking to reduce the uncertainty surrounding management practices and behavior. Active adaptive management can be strengthened to enhance the ability to learn how BMP performance and outcomes can be improved while still moving forward with BMP implementation. Strengthening adaptive management could include:

- Addressing divergence between CAST simulated TP loads and trends in observed TP loads (figure 2) is critical to trust in the model. Adaptive management strategies can be improved by making use of the empirical evidence to continually adjust CAST to be consistent with observations.
- Incorporating tools and processes to identify and reduce decision-relevant uncertainties. Tools and techniques are available to reduce decision-relevant uncertainty in management choices. For example, the expected value of information can be used to identify which uncertainties in management are most important to resolve to improve water quality. Such approaches aim to identify those uncertainties that pose the greatest risk to not achieving management objectives and identify how much a given outcome could be improved if a given uncertainty was resolved. Robust decision-making tools seek to identify solutions and management strategies that perform well under many possible assumptions, rather than optimal solutions that minimize risk given a stringent set of assumptions.
- Designing appropriately scaled and sized implementation trials that could provide a platform to test and evaluate alternative BMP implementation program designs in an effort to reduce uncertainty. Selecting appropriately sized and located watersheds

includes both physical and behavioral considerations. Appropriately selected subwatersheds can provide opportunities to investigate processes of scaling up effectiveness. Consideration may also need to be given to ensure appropriate socio-economic and behavioral profiles that respond differently to different policy or incentive designs are present in test bed locations. Implementation trials should include testing and evaluation of alternative policies designed to improve BMP performance or behavioral change.

- Improving implementation trials by including appropriately designed, finer-scale monitoring capable of assessing the targeted uncertainties and outcomes associated with BMP performance. Monitoring requires both ambient water quality assessment as well as finer-scale monitoring of intermediate indicators of effectiveness (i.e., groundwater, soil nutrient levels, etc.). For example, the partnership could make better use of soil nutrient testing (especially for P) to assure that application of additional nutrients does not further contribute to mass imbalances. Monitoring of implementation trials could also include systematic quantification of BMP implementation-related behaviors.
- Granting Bay jurisdictions the flexibility to deviate from TMDL accounting rules when deploying implementation trials. A substantial potential benefit of implementation trials is to create the opportunity to test the efficacy of different approaches to accounting, tracking, and incentivizing management, and this may not be fully realized without variances from existing crediting protocols. New incentives could help the partnership to facilitate the last step in the CBP's adaptive management process: revise programs, models, and monitoring strategies to improve program performance.

*Nonpoint source policy adaptations*: Coupled with strengthening implementation of adaptive management, the CBP may explore alternative policies, especially those targeting the agricultural sector. Alternative policies might include:

- Alternative policies may be developed to more effectively reduce agricultural sector loads. Given the limits of voluntary, incentive-based BMP implementation programs, the public policy tradeoff increasingly appears to be to implement additional requirements on the agricultural sector or redefine TMDL nutrient/sediment reduction goals. Given the diversity of agricultural operations, additional mandatory requirements do not necessarily need to be broad, inflexible, or exceedingly costly. Performance-based regulatory programs that incentivize pollutant load reductions, for instance, can focus on large scale agricultural production systems (e.g., vertically integrated livestock production) that have scale advantages that may allow them to address mass imbalances.
- Voluntary incentive-based agricultural BMP implementation conservation programs can be strengthened in a number of ways. Voluntary adoption requires effective conservation professionals, and a scaling up of adoption of voluntary programs may require additional technical service providers. To increase program effectiveness, states may consider supplemental financial assistance programs that allow more flexibility for how funds are spent to incentivize adoption in ways that complement federal programs. Historically, federal financial assistance (cost-share) programs have been a primary

source of conservation funding in the Chesapeake Bay watershed, but the form of assistance (cost sharing) and associated requirements may act as a barrier to implementing certain types of cost-effective practices (practices with large upfront costs and limited agronomic benefits) and inducing adoption from some types of farm operators.

*Future challenges*: The future of the Conowingo Dam and climate change are among the many challenges that can impact nutrient and sediment reduction efforts.

- The CBP may consider evaluating and preparing for increased scour and fill activity in watershed reservoirs, in particular for Conowingo. This scour and fill process and the related impact on Bay water quality are not well understood; the stochasticity of climate change-driven extreme events, such as Tropical Storm Lee, occurring in September 2011 and responsible for the last large scour event, means they may bring entirely different Bay water responses if they occur under different hydroclimatic conditions.
- With respect to climate change, efforts could help to further understand how a wetter, warmer, and more intense climate influences hydrologic and biogeochemical cycling and transport in the watershed. Building on that understanding, research could help to inform the selection, design, and siting of cost-effective BMPs that are resilient to anticipated long-term changes in hydroclimatic conditions.

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