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CELEBRATING THE 350TH ANNIVERSARY OF DISCOVERING PHOSPHORUS—FOR BETTER OR WORSE

# Phosphorus and the Chesapeake Bay: Lingering Issues and Emerging Concerns for Agriculture

Peter J. A. Kleinman,* Rosemary M. Fanelli, Robert M. Hirsch, Anthony R. Buda, Zachary M. Easton, Lisa A. Wainger, Chris Brosch, Martin Lowenfish, Amy S. Collick, Adel Shirmohammadi, Kathy Boomer, Jason A. Hubbart, Ray B. Bryant, and Gary W. Shenk

## Abstract

Hennig Brandt's discovery of phosphorus (P) occurred during the early European colonization of the Chesapeake Bay region. Today, P, an essential nutrient on land and water alike, is one of the principal threats to the health of the bay. Despite widespread implementation of best management practices across the Chesapeake Bay watershed following the implementation in 2010 of a total maximum daily load (TMDL) to improve the health of the bay, P load reductions across the bay's 166,000-km$^2$ watershed have been uneven, and dissolved P loads have increased in a number of the bay's tributaries. As the midpoint of the 15-yr TMDL process has now passed, some of the more stubborn sources of P must now be tackled. For nonpoint agricultural sources, strategies that not only address particulate P but also mitigate dissolved P losses are essential. Lingering concerns include legacy P stored in soils and reservoir sediments, mitigation of P in artificial drainage and stormwater from hotspots and converted farmland, manure management and animal heavy use areas, and critical source areas of P in agricultural landscapes. While opportunities exist to curtail transport of all forms of P, greater attention is required toward adapting P management to new hydrologic regimes and transport pathways imposed by climate change.

## Core Ideas

- At the midpoint of the Chesapeake TMDL, dissolved P is increasing in some tributaries.
- Lingering concerns include legacy P, artificial drainage, animal heavy use areas.
- Extreme events represent an acute risk to water quality.



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*Corresponding author (peter.kleinman@ars.usda.gov).

Excessive phosphorus (P) loading is a principal driver of degraded aquatic health in the Chesapeake Bay, the largest estuary in North America, stimulating algal growth and persistent "dead zones" with low oxygen, promoting harmful algal blooms, and altering the structure of aquatic communities (Boesch et al., 2001; Heisler et al., 2008; Testa et al., 2017). In the centuries since Hennig Brandt in 1669 extracted 120 g of P from 5500 L of urine and set the stage for the transformation of the earth's biogeochemical cycles through agricultural and industrial revolutions, P has gone from an element of scarcity in most environments to one of excess, albeit bolstered by finite reserves (Sharpley et al., 2018). So profound has this transformation of P availability become that Brandt's 120 g of P, his life's great achievement, can now be purchased by farmers for less than US$0.50. By way of comparison, in 2017, approximately 1.7 million kg of P was discharged from wastewater treatment plants and combined sewer systems in the Chesapeake Bay watershed (Chesapeake Bay Program, 2018b), nearly 14 million times Brandt's yield.

Efforts to improve the health of the Chesapeake Bay have been organized by the Chesapeake Bay Program, a partnership of seven major jurisdictions encompassing the bay's watershed (Delaware, Maryland, New York, Pennsylvania, Virginia, West Virginia, and the District of Columbia) and the US federal government. The seven jurisdictions have worked with a broad array of local, federal, and nongovernmental partners in the bay's 165,000-km$^2$ watershed to reduce P, nitrogen (N), and sediment loads to bay waters through a series of watershed implementation plans that have been reviewed by the Chesapeake Bay Program using the Chesapeake Bay Watershed Model, which serves as the central decision support system for watershed

P.J.A. Kleinman, A.R. Buda, and R.B. Bryant, USDA-ARS, Pasture Systems and Watershed Management Research Unit, University Park, PA 16877; R.M. Fanelli, USGS, MD–DE–DC Water Science Center, Baltimore, MD, 21228; R.M. Hirsch (emeritus), USGS National Center, Reston, VA, 20192; Z.M. Easton, Dep. of Biological Systems Engineering, Virginia Tech, Blacksburg, VA 24061; L.A. Wainger, Chesapeake Biological Laboratory, Univ. of Maryland Center for Environmental Science, Solomons, MD 20688; C. Brosch, Delaware Dep. Agriculture, Delaware Nutrient Management Program, Dover, DE 19901; M. Lowenfish, USDA-NRCS, Conservation Planning and Technical Assistance Division, Washington, DC; A.S. Collick, Dep. Agriculture, Univ. of Maryland Eastern Shore, Princess Anne, MD 21853; A. Shirmohammadi, Dep. of Environmental Science and Technology, Univ. of Maryland, College Park, MD 20742; K. Boomer, The Nature Conservancy, MD/DC Field Office, Bethesda, MD 20814; J.A. Hubbart, Institute of Water Security and Science, Univ. of West Virginia, Morgantown, WV 26506; G.W. Shenk, USGS, Chesapeake Bay Program Office, Annapolis, MD 21403. Assigned to Associate Editor Heidi Peterson.

**Abbreviations**: TMDL, total maximum daily load.

mitigation activities. The USEPA's 2010 total maximum daily load (TMDL) established a goal of implementing practices that would, according to the 2010 version of the Chesapeake Bay Watershed Model (Phase 5.3), reduce annual total P loads under long-term average hydrologic conditions to the Chesapeake Bay and its tidal tributaries by 25% between the TMDL baseline year of 2009 and 2025 (USEPA, 2010).

At the TMDL midpoint in 2017, 60% of the mitigation activities identified under each jurisdiction's watershed implementation plans were required to be carried out. That goal was surpassed for P (87% of planned mitigation activities were implemented), to a large degree through upgrades to wastewater treatment plants and controls on other point sources. As predicted by the version of the Chesapeake Bay Watershed Model that was used at the TMDL midpoint (Phase 5.3.2), total P loads to the bay from activities related to major nonpoint sources of P (urban runoff, agricultural sources, and natural areas) have dropped by 16% (1.12 million kg total P) from 2009 to 2017 (Chesapeake Bay Program, 2018a, 2018b). Concurrently, early signs of improvements to the ecological health of the Chesapeake Bay have also been reported, including increased coverage of submerged aquatic vegetation and higher dissolved oxygen levels (Lefcheck et al., 2018; Zhang et al., 2018). As the TMDL proceeds into its later stages, challenges remain in the area of reducing nonpoint source loads from agricultural and suburban landscapes.

Progress toward P mitigation has not been universal across the Chesapeake Bay watershed. The agricultural sector is the largest source of P, estimated by the version of the Chesapeake Bay Watershed Model used for the TMDL midpoint assessment (Phase. 5.3.2; Chesapeake Bay Program, 2018b) to contribute 56% of the total P load to the Chesapeake Bay, followed by urban sources (18% of total P load), wastewater treatment plants (15% of total P load), and natural sources (11% of total P load). At the time of the 2017 midpoint for the TMDL process, all states involved in the Chesapeake Bay cleanup had met the 2017 midpoint goals of their watershed implementation plans related to P mitigation activities for agriculture, with the notable exception of Pennsylvania, which failed to meet these specific goals (Chesapeake Bay Program, 2018a). Mixed progress was reported across other sectors too. For instance, while the majority of jurisdictions met the 2017 midpoint TMDL goals for P mitigation practices from wastewater sources, most jurisdictions fell short on midpoint goals for P mitigation activities related to urban sources.

Equating mitigation activities to water quality improvement presents many challenges, from the complexity of accounting to scaling fate-and-transport processes to differentiating between the impacts of various forms of P. While all seven jurisdictions working on the restoration, except Pennsylvania, met the midpoint goals for implementing agricultural P mitigation practices in 2017, water quality improvements have been especially lacking in agricultural watersheds (Moyer and Blomquist, 2017; Fanelli et al., 2019). Furthermore, based on the version of the Chesapeake Bay Watershed Model used at the time of the 2017 TMDL midpoint (Phase 5.3.2), the six Chesapeake Bay watershed states will need to achieve an additional 0.46 million kg of P reductions per year from agriculture to meet the TMDL goals for 2025 (Chesapeake Bay Program, 2018b). Given the precedent-setting nature of the Chesapeake Bay's restoration efforts and the 350th anniversary of Brandt's discovery of P, we reflect on the challenges P management presents to agriculture in the Chesapeake Bay watershed, highlighting hurdles and opportunities as the Chesapeake Bay TMDL enters into the second half of its 25-yr implementation period.

## Societal Investment

Current efforts to improve the health of the Chesapeake Bay build on a history of public investment in reducing loads through point-source pollution controls, public education, and implementation of best management practices by landowners. Simultaneously, research investments have refined our understanding of load sources and effective controls. In 2017, it was estimated that nearly US$2 billion was spent by federal ($1.41 billion) and state ($0.57 billion) governments in support of Chesapeake Bay restoration activities to date (US Office of Management and Budget, 2017). These activities have been far reaching, including stabilizing streambanks, urban programs promoting pet waste cleanup, roadside ditch and highway runoff treatment practices, wastewater treatment plant upgrades, separating sanitary and stormwater drainage systems to prevent storm-driven releases of raw sewage, and adding advanced nutrient removal systems to existing wastewater treatment plants for nonagricultural sources of P.

With over 80,000 working farms managing 5 million ha of farmland in the Chesapeake Bay watershed, the agricultural sector has required, and continues to require, major investment related to watershed mitigation. At the federal level, the USDA provided nearly $1 billion between 2009 and 2018 to support the implementation of conservation systems on over 1.5 million ha (USDA, 2018). The list of agricultural practices tied to P mitigation is expansive, from those related to the farmstead stormwater and animal waste infrastructure to strategies and technologies related to nutrient management, soil conservation, and drainage water management. In 2016, nearly 90,000 ha of agricultural land in the Chesapeake Bay watershed was enrolled in USDA's Conservation Reserve Program, including approximately 36,000 ha of riparian buffers (USDA, 2017).

State and local programs have been essential to agricultural mitigation strategies. Notable examples include cover crop programs in Maryland (Maryland Department of Agriculture, 2017) and streambank fencing programs in Virginia (Chesapeake Bay Funders Network, 2010). Since the late 1990s, all six Chesapeake Bay states have implemented nutrient management programs consistent with land grant university recommendations for application of manures and fertilizers. All of the Chesapeake Bay states are emphasizing the use of the P index site assessment tool to provide guidance to farmers about the need to apply P to farm fields (Sharpley et al., 2017). While state P indices differ across the Chesapeake Bay watershed, there have been regular efforts to coordinate P Index updates to promote consistent assessment of P runoff potential from agricultural fields and prioritize recommendations made by this decision support tool (Sharpley et al., 2017; Drohan et al., 2019).

## Mixed Progress at the TMDL Midpoint

Assessment of the state of P mitigation across a 166,000-$km^2$ watershed is challenging, pushing the limits of monitoring and modeling and requiring adaptive management in these areas to incorporate emerging information. Although widespread progress has been made in the implementation of P mitigation practices for both point and nonpoint sources, it is often unclear the extent to which these practices are improving water quality.

## Tracking Progress through the Chesapeake Bay Program Partnership's Modeling Suite

The Chesapeake Bay Program Partnership's modeling suite serves a central role in forecasting the outcomes of watershed mitigation activities on Chesapeake Bay health. This decision support system has evolved over time into a complex set of computational models that includes watershed, estuarine, and land change components. As the TMDL has progressed, various versions of the Watershed Model component of the modeling suite have provided predictions of how mitigation activities under the watershed implementation plans will alter P loads to the bay (Table 1). The watershed output is presented on an average-hydrology basis to represent the changes due to management actions rather than weather or climate and can be scaled down to roughly a 100-km$^2$ level (Shenk and Linker, 2013). Notably, the Chesapeake Bay Watershed Model is calibrated to monitoring data from the 1990s, providing predictions of water quality (e.g., P loadings) for later years. The Chesapeake Bay Watershed Model has been the focus of litigation, challenging the TMDL's requirements of the agricultural sector (Fears, 2016), but, it has withstood such challenges and persists as the framework in which policy and science around P mitigation have collectively advanced (Easton et al., 2017b).

The evolution of the Chesapeake Bay Watershed Model warrants particular discussion here, as it serves as both a point of confusion and a point of contention. At the time of the implementation of the Chesapeake Bay TMDL, Phase 5.3 of the Chesapeake Bay Watershed Model was used to estimate watershed and state loadings (USEPA, 2010), followed by Phase 5.3.2, which was used from 2011 through 2017 (Chesapeake Bay Program, 2018b), and the currently used Phase 6.0 (Chesapeake Bay Program, 2017). Each revision of the Watershed Model has incorporated new science and new capacity to support the watershed implementation plans of states and to achieve other objectives. Major differences between Phase 5.3 and 5.3.2 include improvements in land use mapping and updates to estimates of the effectiveness of nutrient management practices, explaining the greater prediction of P loadings to the bay under Phase 5.3.2 (Table 1). Most recently, Phase 6 of the model incorporated a major revision of the mechanisms of P transport in response to input from the scientific community, as well as significant structural changes in the modeling system, updates to input data, and notably, the inclusion of streambed, streambank, and tidal shoreline loads and improved data, particularly in the coastal plain (Chesapeake Bay Program, 2017). These changes, particularly the addition of fluvial and coastal routines, enabled better targeting of associated mitigation activities. However, one consequence of breaking out additional sources was that the relative contribution of other sources, including agriculture, diminished under Phase 6 compared with Phase 5.3 and 5.3.2 (Table 1).

According to Phase 6 of the Chesapeake Bay Watershed Model (Chesapeake Bay Program, 2017), total P loads are predicted to have decreased 13%, from 7.7 million kg yr$^{-1}$ in 2009 to 6.7 million kg yr$^{-1}$ in 2017, or about 77% of the total P load reduction needed to achieve TMDL goals (Table 1). Previously, Phase 5.3.2 had predicted that implementation of mitigation practices should have achieved a 21% reduction in total P loadings, surpassing the midpoint goal for total P load reductions and approaching 87% of the final goal for 2025. Note that these goals have been adjusted as the Chesapeake Bay Watershed Model has been updated. Again, these conflicting assessments by different versions of the model highlight an evolving science, difficulty in scaling processes of P fate-and-transport over 166,000 km$^2$, uncertainty over the efficacy of practices implemented under the TMDL, and limited access to soil P data by the Chesapeake Bay Program as a result of privacy concerns by soil testing laboratories, among other things (Radcliffe et al., 2009; Sharpley et al., 2013; Easton et al., 2017b; Harrison et al., 2019).

## Trends Observed by the Chesapeake Bay Program's Nontidal Monitoring Network

In comparison with the Chesapeake Bay Watershed Model projections, monitoring data from 115 water quality and streamflow monitoring nontidal river stations within the nontidal network present a picture of mixed trends in P loads across the Chesapeake Bay watershed (Moyer and Blomquist, 2017). The nontidal network is operated by many Chesapeake Bay Program partners, with data management and analysis provided by the USGS. Of the 115 stations within the nontidal network, 66 stations have sufficient total P data and discharge data to allow for analysis of trends in total P loads from 2007 to 2016 (Fig. 1). While reductions of total P loadings were observed over this period in 38% of Chesapeake Bay tributaries (26 of the 66

**Table 1. Phosphorus loads predicted by different versions of the Chesapeake Bay Watershed Model.**

| Chesapeake Bay Program decision† | Years used | Model Version | Reference | Source | 2009 load | 2017 load | 2025 goal |
|---|---|---|---|---|---|---|---|
| | | | | | million kg yr$^{-1}$ | | |
| TMDL and Phase I WIPs | 2010–2011 | Phase 5.3 | USEPA (2010, Appendix J) | All sources | 7.47 | N/A‡ | 5.68 |
| | | | | Agriculture | 3.30 | N/A‡ | 2.81 |
| Phase II WIPs | 2011–2018 | Phase 5.3.2 | Chesapeake Bay Program (2018b) | All sources | 8.72 | 6.85 | 6.56 |
| | | | | Agriculture | 4.78 | 3.80 | 3.35 |
| Phase III WIPs | 2017–? | Phase 6 | Chesapeake Bay Program (2017) | All sources | 7.74 | 6.73 | 6.43 |
| | | | | Agriculture | 2.02 | 1.87 | N/A§ |

† WIPs, watershed implementation plans developed by Chesapeake Bay watershed jurisdictions to document how they will meet total maximum daily load (TMDL) goals.

‡ Not applicable; Phase 5.3 was not in use after 2011 and so no estimate was made using data from 2017.

§ Not applicable; as of this writing, the Phase III WIPs have not been finalized and so the 2025 goal is not yet broken out by source.

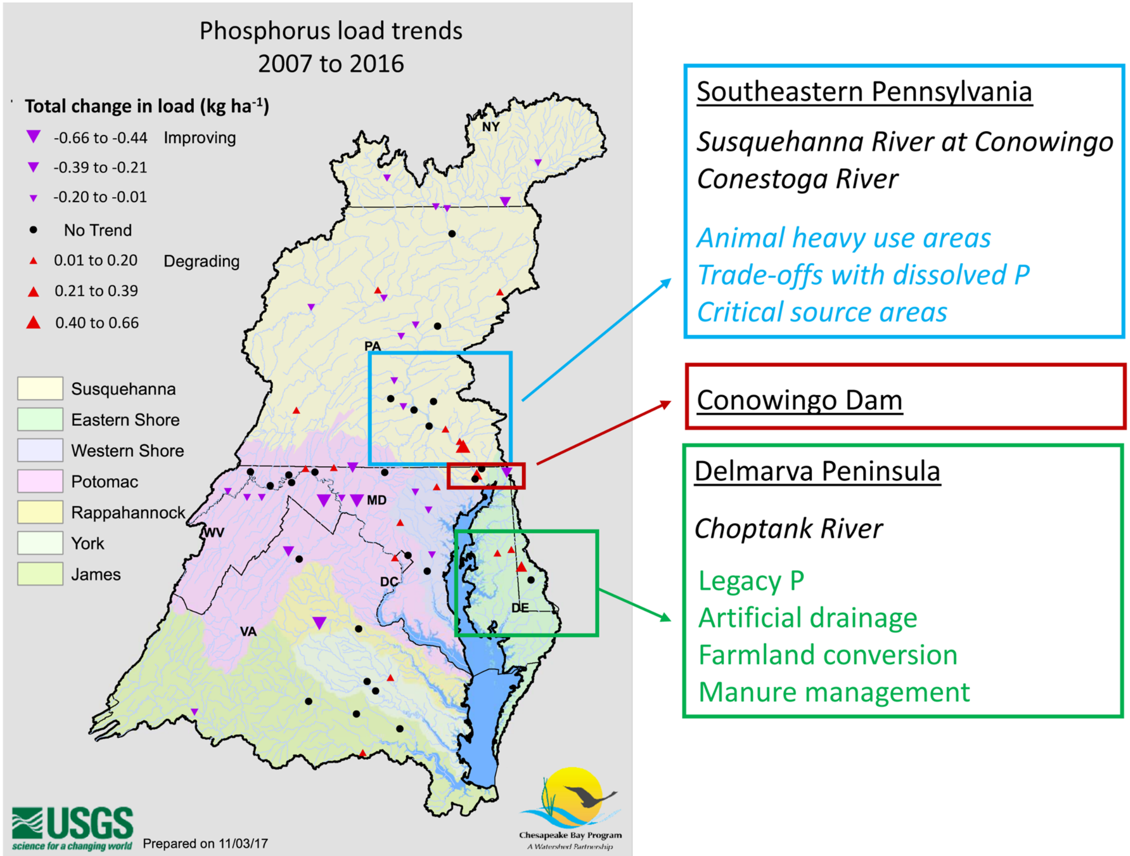



**Fig. 1. Trends in total P losses (kg ha$^{-1}$) for select USGS monitoring sites in the Chesapeake Bay watershed. Map derived from USGS (2017).**

monitoring stations), 26% of the monitoring stations (17 out of 66) witnessed an increase in total P loads, and the remaining 36% of tributary monitoring stations (23 stations) showed no conclusive increase or decrease in total P loads over the 2007 to 2016 period (see trend arrows in Fig. 1; the Supplemental Material provides information on how trends were evaluated). Of the 17 monitoring stations with increasing total P trends, 12 stations also have increasing dissolved P trends. A recent analysis examining P trends during a slightly different time period (2006–2014) found upward total P trends were primarily driven by increasing dissolved P in agricultural watersheds of the region (Fanelli et al., 2019).

Trends in water quality from USGS riverine monitoring sites on the Delmarva Peninsula and southeastern Pennsylvania provide compelling evidence of the conditions under which P loadings have been increasing over the past decade. Two of these monitoring sites (Choptank and Conestoga) have a significant catchment area in agriculture (41 and 28%, respectively) and export the second- and eighth-largest dissolved reactive P yields in the monitoring network (Fanelli et al., 2019). Both the Choptank and Conestoga Rivers have increasing total P and dissolved reactive P (an estimate of inorganic P that is not associated with sediments) trends over the most recent 10-yr period, with dissolved reactive P representing over half the total P load at Conestoga (Table 2). Dissolved reactive P exports from the Choptank and Conestoga Rivers have increased by 49 and 30% over the last 10 yr, respectively (Table 2). Of note, monitoring at the Conestoga River indicates that particulate P loads have declined 11% during this time period while dissolved reactive P loads have increased. These patterns are widespread across these two regions. Large dissolved reactive P exports are observed from Pequea Creek, Swatara Creek, and West Conewago Creek in southeastern Pennsylvania, as well as Marshyhope and Tuckahoe Creeks in the Delmarva Peninsula (Fanelli et al., 2019).

## The Delmarva Peninsula

Upward trends in dissolved reactive P loads from the Choptank watershed (Fig. 1, Table 1) and in other streams in the Delmarva Peninsula (Fanelli et al., 2019) confirm the persistence of P management concerns in the Atlantic Coastal Plain of the Chesapeake Bay watershed. Historically, there has been relatively limited monitoring of the Delmarva's riverine systems. In addition, only recently has there been widespread recognition that computational models have poorly predicted P fate and transport in flat landscapes with extensive artificial drainage—a problem not restricted to the Chesapeake Bay but also common to the watersheds of Lake Erie, the Mississippi River basin, the Baltic Sea, and other water bodies affected by P (King et al., 2015; Kleinman et al., 2015a; Radcliffe et al., 2015).

The Delmarva Peninsula has long been the focus of studies on agriculture and P, largely due to the concentration of poultry production (estimated at 8% of the US broiler industry). Originally, analyses of P in Delmarva cropping systems found excessive accumulations of P in soils receiving large amounts of poultry litter, thus highlighting the potential for P to be exported to Delmarva tributaries by artificial drainage (e.g., Sims et al., 1998; Maguire and Sims, 2002). These concerns were confirmed by monitoring runoff from field soils (Staver and Brinsfield, 2001) and ditch

Table 2. Flow normalized (FN) flux and trend results for different P constituents at three USGS water quality monitoring stations in the Chesapeake Bay watershed. The time period for reporting trend and changes in flux results is 2007–2016. The location of the monitoring sites is highlighted in Fig. 1.

| | Choptank River, Greensboro, MD | Susquehanna River, Conowingo, MD | Conestoga River, Conestoga, PA |
|---|---|---|---|
| Dissolved reactive P | | | |
| 2016 FN-flux, $10^3$ kg† | 6.1 | 660 | 110 |
| 2016 FN-yield, kg ha$^{-1}$ yr$^{-1}$ | 0.21 | 0.09 | 0.92 |
| Change in FN-flux, $10^3$ kg | 2.0c | 292b | 26c |
| Percentage change in FN-flux, % | 49 | 80 | 30 |
| Particulate P + dissolved organic P‡ | | | |
| 2016 FN-flux, $10^3$ kg† | 13 | 3900 | 87 |
| 2016 FN-yield, kg ha$^{-1}$ yr$^{-1}$ | 0.45 | 0.56 | 0.71 |
| Change in FN-flux, $10^3$ kg | 1.6 na | 1430 na | −11 na |
| Percentage change in FN-flux, % | 14 | 58 | −11 |
| Total P | | | |
| 2016 FN-flux, $10^3$ kg† | 19 | 4600 | 200 |
| 2016 FN-yield, kg ha$^{-1}$ yr$^{-1}$ | 0.66 | 0.65 | 1.6 |
| Change in FN-flux, $10^3$ kg | 3.6c | 1720c | 15c |
| Percentage change in FN-flux, % | 23 | 61 | 8 |

† Likelihood of the trend direction designated as follows: a = likely; b = very likely; c = extremely likely; na = not applicable; trends were not assessed on the constituent. See Hirsch et al. (2015) and https://cbrim.er.usgs.gov for more details.

‡ Calculated by taking the difference between total P and ortho-P.

drainage (Kleinman et al., 2007), the latter documenting loads greater than 20 kg ha$^{-1}$, with more than 90% of the P traveling through shallow groundwater to the region's open drainage ditches. Indeed, increasing dissolved P trends are even evident in low flows (e.g., fifth percentile flows) in some of the Chesapeake Bay's agricultural watersheds (Fanelli et al., 2019). Recognizing the unique nutrient management and P transport conditions of flat, intensively drained landscapes is key to advancing mitigation strategies as reflected by revision of Maryland's Phosphorus Management Tool (Shober et al., 2017).

## Legacy Phosphorus

Research from the Delmarva Peninsula has highlighted very high P losses in runoff from agricultural soils, sometimes from cropland that currently complies with fertilizer and manure standards. High P losses in runoff undoubtedly reflect historical application of poultry manure at rates that greatly exceeded crop P demand, augmenting reserves of P in soil and sediments, increasing soil or sediment P sorption saturation (Kleinman, 2017), and creating legacy sources whose contribution to runoff P losses today may overwhelm the signature of other sources (Kleinman et al., 2007; Church et al., 2010). Indeed, the largest increases in dissolved P concentrations in the USGS water quality monitoring network (Fig. 1) were found in association with increases in manure application rates in the Choptank and Marshyhope watersheds, both on the Delmarva Peninsula (Fanelli et al., 2019). Legacy P is by no means a phenomenon restricted to the Delmarva Peninsula, and its role in undermining watershed mitigation programs has been illustrated in many case studies outside of the region (Sharpley et al., 2013).

Legacy P is particularly difficult to manage as it contributes dissolved forms of P to the environment, rendering ineffective traditional soil conservation practices that are often recommended for P mitigation because those practices are designed to reduce soil loss, thereby reducing the loss of particulate P. When P accumulates to very high levels in soils, it can take decades to draw down using conventional farming techniques (Fig. 2). Few practices are designed to address legacy P in a timely, cost-effective fashion. Phytomining by growing and harvesting crops without P fertilization, as illustrated in Fig. 2, is slow and dependent on antecedent soil P levels (Schelfhout et al., 2018). Gypsum, particularly FGD gypsum that it is derived as an inexpensive by-product of scrubbing sulfur from coal burning generators, has gained widespread interest within the Delmarva farming

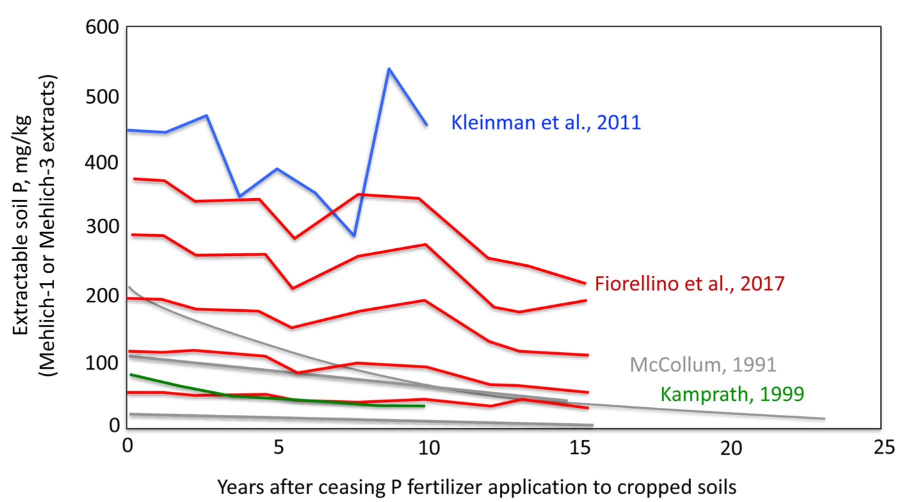



Fig. 2. Observed and modeled trends in soil P (Mehlich-1 or Mehlich-3) related to the cessation of fertilizer P application to cropped soils and subsequent phytomining with crop removal. Trend lines include the studies reported by McCollum (1991), Kamprath (1999), Kleinman et al. (2011), and Fiorellino et al. (2017). Adapted from A. Shober, University of Delaware.

community as a means of reducing legacy P solubility, especially as it offers other benefits (e.g., as a treatment for sea salt-impacted soils and as a source of S; Murphy and Stevens, 2010). Practices that do not confer other agronomic benefits (e.g., deep tillage that dilutes surface soils with subsurface soils [Sharpley, 2003], and amendments that reduce the solubility of P in surface soils [Callahan et al., 2002]) have not gained traction due to cost and concern over potential adverse impacts.

It has long been understood that from a mass balance standpoint, areas of concentrated poultry production on the Delmarva Peninsula are P hot spots within the Chesapeake Bay watershed (Russell et al., 2008). High levels of soil and sediment P have been documented in various studies, including those summarizing subsets of agronomic soil samples in Delaware and Maryland (Pautler and Sims, 2000; Bryant et al., 2012). However, there are few current datasets that provide insight into the distribution of soil P of the Delmarva, let alone the Chesapeake Bay watershed. While Maryland completed a sampling of field soils across the state in 2016 (Maryland Department of Agriculture, 2016), this was a unique effort. Comprehensive identification of legacy P sources across the Chesapeake Bay watershed has been hampered by privacy considerations in soil testing. As a result, the Chesapeake Bay Watershed Model must use indirect means, specifically through the Annual Phosphorus Loss Estimator (APLE) model (Vadas, 2017) to infer soil P levels at the county scale (Chesapeake Bay Program, 2018a).

### Farmland Conversion

Given the likely extent of legacy P in the Delmarva's poultry production region, the increasing conversion of farmland to buildings, including human dwellings and animal housing, raises concerns that stormwater from these facilities will interact with legacy P to exacerbate P losses. For instance, there has been a trend to consolidate poultry housing, with new farms often including eight or more houses (Kobell, 2015). Between 2010 and 2017, 412 new poultry houses were constructed on the Delmarva Peninsula (increasing from 4679 to 5091), corresponding with an increase from approximately 560 million birds in 2010 to 600 million birds in 2017 (Delmarva Poultry Inc., 2019). This new construction has placed greater pressure on state and federal agencies to approve and even design stormwater management plans, but these plans do not account for the potential mobilization of legacy P sources in soils where construction and development have recently taken place. As illustrated in a hypothetical example from Maryland's Eastern Shore, addition of four poultry barns to an 11 ha field, can substantially increase stormwater runoff potential (Fig. 3). The greater prevalence of impervious surfaces and associated stormwater flows through soils rich with legacy P have the potential to contribute significant P loads in drainage water (Kleinman et al., 2007; Church et al., 2010). Monitoring and mitigation of stormwater from such developments should be treated as a priority water quality concern as well as a priority research topic. Practices such as the use of P sorbing materials in runoff detention/retention basins or P-rich soils around poultry houses all show potential (Buda et al., 2012 Bryant et al., 2012; Penn et al., 2017).

## Southeastern Pennsylvania

Southeastern Pennsylvania represents another hot spot of P in the Chesapeake Bay watershed, characterized by high concentrations of small farms, including many farms that have historically had limited interaction with conservation programs, from Amish operations to hobby farms (horses account for about 8% of manure dry matter generated across the Chesapeake Bay watershed; Kleinman et al., 2012). The emergence of dissolved P in watersheds such as the Conestoga (Fig 1; Table 1) reflects an array of recalcitrant sources of P, exacerbated by increased infrastructure needs of small livestock operations (Kleinman et al., 2012). Although conservation tillage is widespread in the region (roughly two-thirds of the Bay watershed's farm soils are in no-till or some type of perennial forage), adoption of cover crops in Pennsylvania is not as extensive as in Maryland, where

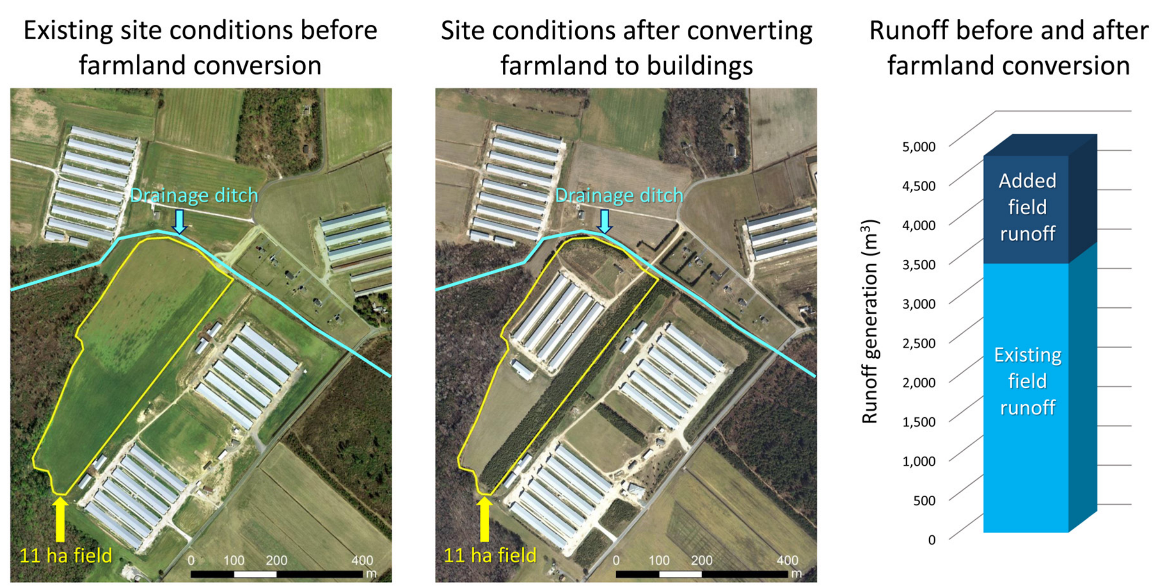



**Fig. 3. The potential effects of farmland conversion on runoff generation. Orthophotos illustrate site conditions before (left panel) and after (middle panel) converting farmland to poultry houses on Maryland's lower Eastern Shore. Using the Curve Number method for a 2-yr, 24-h storm event, differences in runoff generation are shown for an 11-ha field before and after farmland conversion (right panel). In this case, farmland conversion increases existing site runoff by 40% (from 3425 m³ before conversion to 4794 m³ after conversion).**

it has been heavily subsidized (13 vs. 29% of cropland area in Pennsylvania and Maryland, respectively, in 2017), and soil erosion remains a dominant concern of nutrient management specialists and resource conservation programs (Cela et al., 2016; Bryant, 2019). Even so, the emergence of dissolved P as a concern may also reflect the success of soil conservation programs, which struggle to deal with the persistent issue of vertical soil P stratification (Baker et al., 2017).

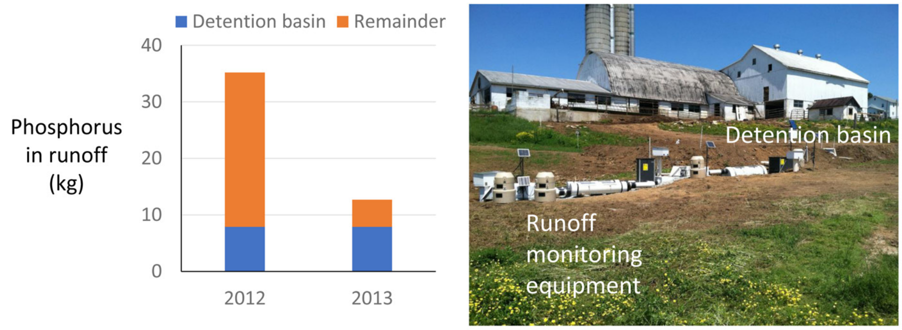



**Fig. 4. Phosphorus loads (kg) in runoff from a dairy barnyard equipped with a detention basin, Lancaster County, PA. Kleinman et al. (2016).**

## Animal Heavy Use Areas

The small farms of southeastern Pennsylvania frequently include livestock with unimproved barnyards, turn-out areas, and loafing areas that lack structures to isolate them from runoff or stormwater flow pathways and are in direct connection with concentrated flow or streams. The combination of livestock dung and compaction of soil by hooves creates conditions that can produce disproportionately high runoff loads of P. As illustrated in Fig. 4, a study in Lancaster County, PA, observed annual P losses in runoff of 13 to 35 kg from a <0.1-ha barnyard on a small plain sect dairy farm. Often, traditional mitigation options for animal heavy use areas are seen as too expensive for small operations, particularly if they involve installation of concrete infrastructure. However, less costly approaches can be taken to reroute stormwater runoff from these areas, reduce the buildup of dung sources in areas that yield large amounts of runoff, and trap sediment and P. In fact, a simple detention basin installed on Lancaster County farm was able to trap 22 to 62% of P in yearly runoff from the unimproved barnyard (Fig. 4). Woodchip pads have likewise shown promise as a low-cost, stormwater treatment alternatives to traditional barnyard systems (Faulkner, 2017).

## Critical Source Areas

Variable source area hydrology—expanding and contracting zones of surface runoff generation created by water-logged soils—dominates the foot slopes of the Chesapeake Bay watershed's upland landscapes and is central to many nonpoint source P mitigation strategies. Research demonstrates that when these areas overlap with elevated concentrations of available P, they serve as critical source areas, disproportionately influencing watershed P loads and therefore serving as priorities for P mitigation (Sharpley et al., 2011).

Given historical preferences in agricultural development (e.g., farmsteads often located close to water sources in valley bottoms), limited land availability, and a high density of farming operations in areas with the best soils, many riparian soils have built legacy P reserves, although perhaps not as extreme as in the Delmarva's poultry-producing counties. As demonstrated by Buda et al. (2009), even modest elevations of soil P can produce disproportionate yields of P in runoff from hydrologically active riparian soils (Fig. 5). Increasingly, there is recognition that riparian buffer programs, which generally prohibit biomass harvest and therefore preclude phytomining of legacy P, must incorporate critical source area considerations in their management guidelines.

To identify such critical source areas, site assessment tools have played a central role in nutrient management programs of the region. Numerous studies have shown that targeting mitigation practices to sites with higher pollution potential can improve cost effectiveness of pollution reduction efforts (Khanna et al., 2003; Yang and Weersink, 2004; Wagena and Easton, 2018; Xu et al., 2019). Studies have shown that targeting practices by flow paths, subwatersheds, soil erodibility, or

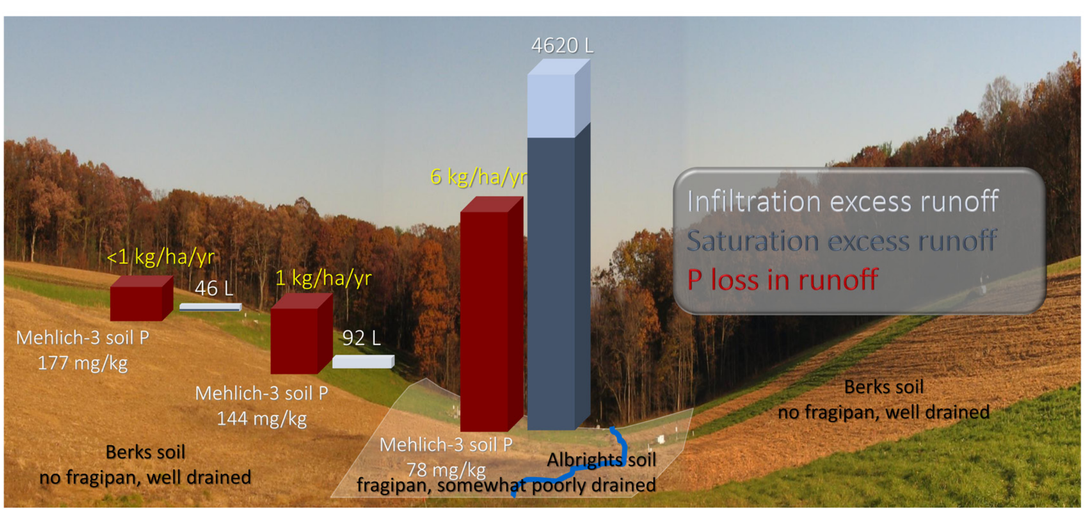



**Fig. 5. Contribution of well-drained and somewhat poorly drained soils to surface runoff and total P loss in surface runoff over 2.5 yr of monitoring. Adapted from Buda et al. (2009).**

other land and soil characteristics instead of applying practices randomly or uniformly can reduce costs of meeting a given water quality goal (Yang and Weersink, 2004; Veith et al., 2004).

Debate persists over whether legacy P evaluation tools, such as the P index, are sufficiently stringent in their assignment of risk and whether their use has led to significant water quality improvements (Sharpley et al., 2017). End users have reported dissatisfaction with the general nature of nutrient management recommendations (Osmond et al., 2012), which often do not connect to specific best management practices and do not support daily, operational decisions that reflect antecedent conditions and weather (Buda et al., 2013). Persistent improvements to Chesapeake Bay state P indices (Sharpley et al., 2017), new conservation planning tools that target best management practices to vulnerable areas in watersheds (Tomer et al., 2015), and new decision support tools that use short-term weather forecasts (Easton et al., 2017a; Sommerlot et al., 2017) all support improved assessment and management of P critical source areas.

### Dissolved Phosphorus: A Source of Conservation Program Trade-Offs

Some of the largest increasing total P trends in the Chesapeake Bay watershed are driven by increasing dissolved P (Table 1; Fanelli et al., 2019). Despite an abundance of conservation practices to protect soils from erosion, there remain limited options for controlling dissolved forms of P in runoff from agricultural land. Too often, important conservation practices, such as no-till and cover crops, are implemented for P mitigation, while trade-offs related to their effect on dissolved P are overlooked or ignored (Kleinman et al., 2015b; Jarvie et al., 2017). Indeed, these practices, although effective at decreasing particulate P loss, are regularly shown to exacerbate dissolved P losses from soils across the Chesapeake Bay watershed (Staver and Brinsfield, 1991, 2001; Verbree et al., 2010; Fanelli et al., 2019). In evaluating short-term trends in P monitored at USGS gauges in the Chesapeake Bay watershed, Fanelli et al. (2019) observed an increase in dissolved P associated with areas where increased implementation of conservation tillage had been reported. The watersheds with the largest increases in percentage agricultural land under conservation tillage from 2002 to 2012 were located in southeastern Pennsylvania (Sekellick, 2018). Although the goal of the TMDL is to reduce total P loads, a shift from particulate P to dissolved P can negatively affect freshwater ecosystems (including northern sections of the bay), given that dissolved P is more bioavailable than particulate P.

In cropped soils, P accumulation on the soil surface can occur very rapidly, with severe vertical stratification occurring even a few years after no-till has been implemented (Baker et al., 2017). Phosphorus on or near the soil surface is generally the principal source of P in surface runoff and drainage water (Sharpley, 1985; Kleinman et al., 2015a,c). When vertical P stratification occurs in hydrologically active soils (e.g., the somewhat poorly drained soils depicted in Fig. 5), neither no-till nor cover crops can be expected to curtail dissolved P losses. In fact, because landscape processes are the primary control of surface runoff generation, producing "saturation excess" runoff (Easton et al., 2008; Buda et al., 2009), improvements to soil infiltration properties expected of no-till and cover crops will not significantly influence runoff generation (although these practices can reduce runoff from well-drained soils where "infiltration excess" runoff is most important).

From the standpoint of crop production, continued application of P to the soil surface without subsequent incorporation into the soil is an extremely inefficient means of fertilizing crops, as it places P above the root zone and the majority of P is retained in the vicinity of application (Smith et al., 2017). Improvements in P use efficiency are frequently used to promote the adoption of P mitigation practices, especially in plant nutrition. However, losses of P to runoff are of little economic consequence to crop production, as only very small fractions of P applied as fertilizer and manure (<5%) are lost in runoff, even under extreme conditions. Conservation programs need to prioritize dissolved P as a concern, recognizing the unintended consequences of philosophies that promote "never-till" and promoting practices that minimize or correct vertical stratification of P (Liu et al., 2014; Jarvie et al., 2017).

## The Susquehanna River and Conowingo Dam

The southeastern Pennsylvania watersheds discussed above drain into the Susquehanna River (watershed area: 70,200 km$^2$), which passes over the Conowingo Dam before discharging to the Chesapeake Bay. The Susquehanna River contributes approximately half of the bay's fresh water (Bue, 1968) and has a dominant influence in the bay's main stem from the head of the Bay though the confluence with the Potomac River estuary (Pritchard, 1952). Completed in 1928, the Conowingo Dam once served as a major sink of sediment from the Susquehanna, estimated at 200 million Mg yr$^{-1}$ in 2008 (Langland, 2009). More recently, however, the sediment storage capacity of the dam has been exhausted (Zhang et al., 2016; Linker et al., 2016). As a result, the long-term net export of sediment and most forms of P from the Susquehanna is increasing (Table 1). The condition of the Conowingo Dam is germane to this discussion of agriculture and P in the Chesapeake Bay region because the Conowingo Reservoir is no longer a significant trap for particulate P from the Susquehanna River watershed. Moreover, it is possible that internal processes within the Conowingo Reservoir could contribute to the increasing dissolved P trend in the Susquehanna River. So, achieving the desired reductions in loads from all sources in the watershed will need to depend on an even greater level of effort than was anticipated at the time when the TMDL requirements were established in 2010 (Linker et al., 2016).

## Climate Change

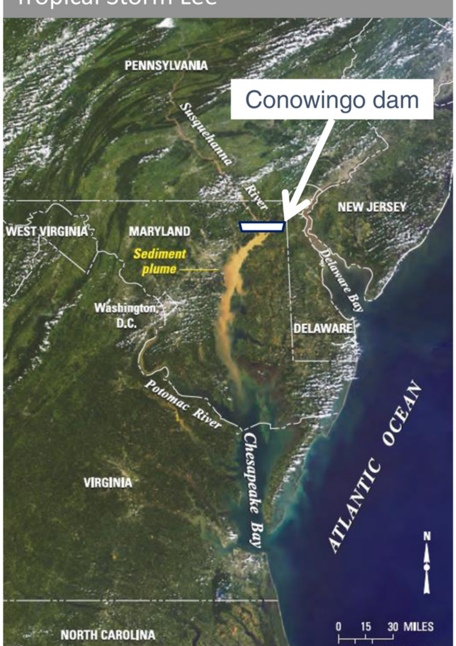
Climate change is expected to bring an array of even greater challenges to P management in the Chesapeake Bay region. Some of these challenges, such as increased total precipitation and increased incidence of extreme precipitation events (Easterling et al., 2017), are widely acknowledged for their potential to increase P losses from agriculture (Daloğlu et al., 2012; Ockenden et al., 2016). Other challenges, such as the potential effects of saltwater intrusion on soil P cycling in low-lying coastal areas (Tully et al., 2019b), are only beginning to emerge. Integrating the multifaceted impacts of climate change on P loss along with future changes in conservation practice adoption and agronomic P management will be critical to predicting whether the P reduction goals set by the Chesapeake Bay TMDL can be sustained.

Increases in precipitation extremes represent an acute risk to water quality in the Chesapeake Bay. In the northeastern United States, the magnitude and frequency of extreme precipitation events have risen markedly (Mallakpour and Villarini, 2017; Huang et al., 2017), with the amount of annual precipitation falling in the heaviest 1% of daily events increasing by 55%, faster than any region in the United States (Easterling et al., 2017). While links between total precipitation and P loss are generally well established (Ockenden et al., 2016, 2017), study of the role of extreme precipitation in P export is relatively nascent. In a recent study, Carpenter et al. (2018) found a strong connection between extreme daily precipitation and daily P loads, noting that expected rises in the frequency of such extreme events would have a disproportionately large effect on annual P loads. In the Chesapeake Bay watershed, the one-two punch of Hurricane Irene and Tropical Storm Lee resulted in extreme nutrient and sediment losses (Hirsch, 2012; Vidon et al., 2017, 2018), with P loads from Tropical Storm Lee accounting for 60% of the P loss in 2011 and 22% of the P loss over the previous decade (Fig. 6; Hirsch, 2012). As the aftermath of Hurricane Irene and Tropical Storm Lee makes clear, successive extreme events, while rare, can intensify nutrient export from watersheds (McMillan et al., 2018). Better knowledge of the hydrometeorological and land management conditions that make these extreme events more likely would improve efforts to curtail P losses in the Chesapeake Bay watershed in a changing climate (Michalak, 2016).

In low-lying coastal areas of the Chesapeake Bay, there is growing concern that climate-change-induced sea level rise could render agricultural soils in these regions more vulnerable to P loss due to saltwater intrusion. While the risk of enhanced P mobilization with increased salinity is well understood (Jordan et al., 2008; Hartzell and Jordan, 2012; Upreti et al., 2015), the potential for saltwater intrusion to enhance legacy P losses from agriculture is receiving increasing attention (Tully et al., 2019a, b). Indeed, a recent study on Maryland's Lower Eastern Shore found increasing soil P concentrations along saltwater intrusion gradients in several farm fields (Tully et al., 2019b), while a laboratory study using coastal wetland soils from Florida showed enhanced P losses with increased salinization (Steinmuller and Chambers, 2018). Saltwater intrusion can arise during drought periods, when freshwater gradients slacken, allowing saline water to move landward (Ardón et al., 2013). Climate change is projected to increase the frequency and magnitude of droughts in the mid-Atlantic region (Wehner et al., 2017). Additionally, high tide flooding, especially during storms, can push saline waters well inland, and these impacts can be magnified in artificially drained landscapes (like the Delmarva) where intense ditching and tiling increases hydrological connectivity (Bhattachan et al., 2018). High tide flooding is also expected to worsen with sea level rise. The mid-Atlantic coast, including the Chesapeake Bay, is already experiencing rates of sea level rise that are three to four times the global average (Dupigny-Giroux et al., 2018), and high tide flooding events that occur roughly 6 to 10 d per year now could happen almost daily by the end of the century (Sweet et al., 2018). As such, the risk of legacy P transfers from saltwater intrusion events in coastal agricultural regions of the Chesapeake Bay appears likely to rise in concert with continued climate change.

In light of the potential for climate change to enhance watershed P losses, a growing number of modeling studies have begun to examine whether nutrient management efforts will be sufficient to maintain the 2025 Chesapeake Bay TMDL goals. For instance, Wagena and Easton (2018) incorporated projections from six global climate models into the Soil and Water Assessment Tool–Variable Source Area (SWAT–VSA) to

a. Satellite image of Chesapeake Bay Region showing sediment plume from Tropical Storm Lee




b. Flow, total phosphorus, and suspended sediment contributions from Tropical Storm Lee relative to different monitoring periods

| | 2011 | 2002-2011 | 1978-2011 |
|---|---|---|---|
| Flow | 12% | 1.8% | 0.6% |
| Total phosphorus | 61% | 22% | 9% |
| Suspended sediment | 78% | 39% | 22% |

c. Aerial photograph of discharge from the Conowingo Dam following Tropical Storm Lee

**Fig. 6. The effects of Tropical Storm Lee (2011) on total P and suspended sediment loss in the Susquehanna River basin, including (a) a satellite image showing the sediment plume from Tropical Storm Lee (image credit: NASA), (b) flow, total P, and suspended sediment loads from Lee as a percentage of different monitoring periods (Hirsch, 2012), and (c) an aerial photo of discharge from the Conowingo Dam following Tropical Storm Lee. (Image credit: Wendy McPherson, USGS.)**

simulate the effects of climate change on hydrology and nutrient export in the Susquehanna River basin. Even though river flow was projected to increase by 4.5% toward the end of the century, total P export was expected to decline by roughly 2.6%, although two of the six models suggested that total P loads might increase. Studies of smaller tributaries, on the other hand, have tended to indicate a stronger likelihood of increased watershed P losses with climate change. On Maryland's Lower Eastern Shore, companion SWAT modeling studies by Renkenberger et al. (2016; 2017) in a 298-km$^2$ agricultural basin indicated a two- to threefold expansion of critical source areas of P loss with climate change that led to an 80% increase in P export relative to current conditions. Elsewhere, a SWAT–VSA modeling study by Wagena et al. (2018) in a sloping 7.3-km$^2$ agricultural basin in east-central Pennsylvania suggested annual total P export could increase by up to 11%, with the majority of the load increase driven by higher wintertime stream flows. Taken together, these studies found that more efficient and additional best management practices above and beyond those currently planned would be needed to maintain the P loading targets specified by the Chesapeake Bay TMDL in a changing climate.

# Opportunities

Just as we envision that Hennig Brandt marveled at the glow of white P he had distilled, so too can an observer marvel at the P mitigation achievements in the Chesapeake Bay watershed. However, there is little doubt that excess P in waterways is a contemporary problem, the product of our success in converting Brandt's discovery of P into a global resource for agriculture. At the midpoint of the Chesapeake Bay TMDL, implementation of agricultural P mitigation practices across the Chesapeake Bay watershed has affected short- and long-term processes controlling P transfers from agriculture. For each challenge reviewed here, opportunities exist to improve the practices and strategies of P management (Table 3). Ultimately, to be sustainable, P management in the Chesapeake Bay region will need to balance the need for profitable agriculture with the goal of restoring the health of the bay and sustaining a safe environment for seafood production and recreation for future generations.

## Supplemental Material

An expanded description of the analytical technique (Weighted Regressions on Time, Discharge, and Season) used to depict water quality trends in Figure 1 and Table 2 is provided in the supplemental material.

## Conflict of Interest

The authors declare no conflict of interest.


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**Table 3. Challenges and opportunities for P mitigation in the Chesapeake Bay watershed.**

| Challenge | Opportunities |
|---|---|
| Decision support | · Improved reliability of site assessment tools in artificially drained settings.<br>· Runoff forecasting tools to guide daily decisions on when and where P is applied. |
| Legacy P | · Cost-effective technologies that rapidly draw down legacy P or sequester it in relatively immobile forms.<br>· Greater consideration of legacy P in agronomic and conservation planning (e.g., promoting drawdown of soil P prior to establishment of riparian buffers).<br>· Improving availability of legacy P data (soils and sediments) in a fashion that adequately protects privacy of landowners. |
| Artificial drainage | · Drainage water management systems that conserve water and nutrient resources within farming operations.<br>· Filter systems to remove P from drainage waters (e.g., P removal structures, in-ground reactors).<br>· Field management strategies that minimize P availability to drainage water (e.g., setbacks, variable rate application).<br>· Improved availability of data on the distribution of drainage ditches and tile drains in a fashion that adequately protects privacy of landowners. |
| Farmland conversion | · Improved stormwater management systems that account for legacy P sources and promote on-farm P retention.<br>· Better coordination between integration of farm and nonfarm stormwater management systems (e.g., highway and farm drainage systems). |
| Animal manure | · Farm and watershed strategies that promote annual P balances.<br>· P recovery systems for manures that improve N/P ratio for agronomic management.<br>· Manure transport systems that enable P to be relocated from areas of surfeit to areas of deficit. |
| Animal heavy use areas | · Stormwater management systems to disconnect animal heavy use areas from runoff.<br>· Low-cost barnyard alternatives (e.g., woodchip pads).<br>· Filter systems to remove P from runoff (e.g., P removal structures, in-ground reactors). |
| Critical source areas | · Continued adaption of "4R" nutrient stewardship strategies to the production systems of the Chesapeake Bay region.<br>· Improved guidelines for winter manure management.<br>· Improved decision support. |
| Dissolved P trade-offs | · Perennialization of cropping systems to promote soil conservation and control particulate P loss.<br>· Greater management options to prevent vertical stratification of P in soils and better consideration of vertical stratification by soil health programs. |
| Climate change | · Enhancing the resilience of agricultural landscapes to extreme rainfall events using practices that control runoff and reduce erosion.<br>· Improving drainage water management techniques in coastal settings to mitigate the effects of salt water intrusion on legacy P losses from drained soils.<br>· Better targeting of best management practices to account for the projected expansion of critical source areas of P loss with climate change. |

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