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RESEARCH ARTICLE

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Evaluating water-quality trends in agricultural watersheds prioritized for management-practice implementation

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Abstract

Many agricultural watersheds rely on the voluntary use of management practices (MPs) to reduce nonpoint source nutrient and sediment loads; however, the water-quality effects of MPs are uncertain. We interpreted water-quality responses from as early as 1985 through 2020 in three agricultural Chesapeake Bay watersheds that were prioritized for MP implementation, namely, the Smith Creek (Virginia), Upper Chester River (Maryland), and Conewago Creek (Pennsylvania) watersheds. We synthesized patterns in MPs, climate, land use, and nutrient inputs to better understand factors affecting nutrient and sediment loads. Relations between MPs and expected water-quality improvements were not consistently identifiable. The number of MPs increased in all watersheds since the early 2010s, but most monitored nutrient and sediment loads did not decrease. Nutrient and sediment loads increased or remained stable in Smith Creek and the Upper Chester River. Sediment loads and some nutrient loads decreased in Conewago Creek. In Smith Creek, a 36-year time-series model suggests that changes in manure affected flow-normalized total nitrogen loads. We hypothesize that increases in nutrient applications may overshadow some expected MP effects. MPs might have stemmed further water-quality degradation, but improvements in nutrient loads may rely on reducing manure and fertilizer applications. Our results highlight the importance of assessing MP performance with long-term monitoring-based studies.

KEYWORDS
Chesapeake Bay, agriculture, nonpoint sources, nutrients, sediment, management practices

1 | INTRODUCTION

Excess nutrient and sediment loads have degraded the water-quality conditions and biological communities of many aquatic ecosystems (Boesch, 2019; Cloern, 2001; Noe et al., 2020; Scavia & Bricker, 2006). The restoration of these ecosystems often relies on the voluntary use of management practices (MPs) to lower amounts of nonpoint source pollution but expected water-quality improvements are not always

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Research Impact Statement

Monitoring data suggest that agricultural activities and natural factors may have offset expected nutrient and sediment load reductions in watersheds targeted for management-practice implementation.

apparent in monitored responses. The Chesapeake Bay and its watershed is a notable example where there are ongoing efforts to reduce nonpoint source pollution. Excess nutrients and sediment have contributed to ecological declines in the Chesapeake Bay since at least 1950 (Boynton et al., 2008; Kemp et al., 2005). Since this time, the human population of the Chesapeake Bay watershed has more than doubled (Chesapeake Bay Program, 2023), causing changes in agricultural and urban activities that have increased the amount of nutrients and sediment in streams (Ator et al., 2020; Lyerly et al., 2014; Sabo et al., 2022).

Substantial nutrient and sediment reductions in the Chesapeake Bay watershed have resulted from decades of active management, but recent nutrient reductions have been insufficient to meet water-quality goals (Ator et al., 2020; Chesapeake Bay Program, 2021; Scientific and Technical Advisory Committee, 2023). Decreased point-source nutrient loads and atmospheric nitrogen loads have likely contributed to water-quality improvements (Burns et al., 2021; Clune et al., 2021; Eshleman et al., 2013; Lyerly et al., 2014; Shenk & Linker, 2013), but these combined sources contribute a smaller fraction of nutrients to the Bay than agricultural lands (Ator et al., 2019). While nutrient loads have decreased in some agricultural watersheds (Mason et al., 2023), the net amount of nutrients from agricultural lands throughout the Chesapeake Bay watershed has not substantially decreased in recent decades (Ator et al., 2020; Boesch, 2019; Brakebill et al., 2010; Shenk & Linker, 2013; Stephenson et al., 2022; Zhang et al., 2023). Some commonly used agricultural MPs in the Chesapeake Bay watershed include nutrient management plans, tillage management, cover crops, animal waste management, pasture management, and buffers (Chesapeake Assessment Scenario Tool, 2023a; Sekellick et al., 2019). Meeting Chesapeake Bay water-quality goals likely relies on the voluntary use of these and other MPs to reduce nonpoint source agricultural water-quality loads (Boesch, 2019; Stephenson et al., 2022).

The ability of agricultural MPs to reduce nutrient and sediment loads varies widely. Some studies have associated water-quality improvements with MPs (Fanelli et al., 2019; Hively et al., 2020; Law et al., 2020; Makarewicz et al., 2009; Moriasi et al., 2020) while others have not (Ator et al., 2020; Fisher et al., 2021; Jain & Singh, 2019; Kroll & Oakland, 2019; Lintern et al., 2020; Melland et al., 2018). Variability in MP performance is often attributed to the effects of practice type, location, and maintenance; watershed physiography, land use, and size; and study duration, design, and methodology (Duriancik et al., 2018; Kroll & Oakland, 2019; Lintern et al., 2020; Melland et al., 2018; Tomer & Locke, 2011). Nutrient inputs have been identified as a driver of changing nutrient loads in the Chesapeake Bay watershed (Eshleman et al., 2013; Eshleman & Sabo, 2016; Sabo et al., 2022; Zhang et al., 2022). Therefore, load reductions may be more likely to result from MPs that lower surplus nutrient inputs rather than MPs that attempt to control the delivery of landscape nutrient inputs to streams (Ator et al., 2020; Boesch, 2019). However, MP effects in the Chesapeake Bay watershed may be affected by increases in water temperature and increases in the volume and intensity of precipitation (Fleming et al., 2021; Hanson et al., 2022; Lintern et al., 2020; Rice & Jastram, 2014). Temperature and precipitation can change the amount of nutrients and sediment delivered to streams by altering streamflow volumes, sediment runoff, nutrient biogeochemistry, and agricultural production (Ator et al., 2022; Chanat & Yang, 2018; Hanson et al., 2022; Najjar et al., 2010; Ryberg et al., 2018). Evidence of MP benefits are most consistently demonstrated from field- and plot-scale studies (Duriancik et al., 2018; Hanrahan et al., 2021; Jordan et al., 1993; McDowell et al., 2009; Smith et al., 2001; Thapa et al., 2018; Tomer et al., 2014), and there is considerable uncertainty about the collective effects of MPs in larger watersheds. In the current state-of-the-science, MP load reductions are more likely to be identified by studies that use modeled predictions than those that assess monitored water-quality responses (Lintern et al., 2020; Scientific and Technical Advisory Committee, 2023).

Insights about the water-quality effects of agricultural MPs are needed from long-term monitoring studies to inform watershed restoration efforts. Producing such insights is challenging because it requires (1) robust monitoring to accurately characterize nutrient and sediment loads, (2) an understanding of how loads respond to multiple environmental factors, (3) statistical approaches to identify direct cause and effect relations, (4) tracking of pollution sources and MPs, and (5) enough time to detect potential MP benefits in streams and rivers (Duriancik et al., 2018; Melland et al., 2018). To address these challenges, a 2010 Presidential Executive Order directed a federal partnership to "…establish showcase projects in small watersheds to test and monitor the benefits of a focused, highly partnered, voluntary approach to conservation" (Federal Leadership Committee, 2010). The U.S. Department of Agriculture selected three agricultural showcase watersheds, namely, the Smith Creek watershed in Virginia, the Upper Chester River watershed in Maryland, and the Conewago Creek watershed in Pennsylvania. Since 2010, these Chesapeake Bay watersheds received targeted MP investments and intensive amounts of water-quality monitoring.

The objectives of this paper are to describe how nutrient and sediment loads changed over time in the three Chesapeake Bay showcase watersheds and to evaluate the water-quality effects of natural and anthropogenic factors. This paper synthesizes data representing MPs, land use, nutrient sources, and climatic conditions in each watershed and uses these data to evaluate the drivers of monitored water-quality conditions over time. This unique interpretation of multiple datasets may help identify the effect of MPs on nutrient and sediment loads. This paper primarily focuses on monitoring data collected from the early 2010s through 2020 and builds upon previous research in the showcase watersheds, which documented water-quality conditions, nutrient sources, transport factors, and MP implementation patterns after 3 years of monitoring (Hyer et al., 2016).

2 | METHODS

2.1 | Description of study watersheds

The Smith Creek, Upper Chester River, and Conewago Creek watersheds were selected as showcase watersheds because they had public and private sector groups working together to address natural resource issues and contain agricultural land uses and geologic settings common to the Chesapeake Bay watershed. Research about water-quality responses and MP effects in these watersheds may thus help inform agricultural management throughout the region. A brief description of the stream network, agricultural activities, and geology of each watershed is provided below; a more detailed summary can be found in Hyer et al. (2016).

Smith Creek is a 273 km^2 watershed within the Ridge and Valley physiographic province in Virginia's Shenandoah Valley that drains north to the North Fork of the Shenandoah River. The southern portion of the watershed has mostly carbonate geology, primarily dolomite and limestone, and many headwater springs and sinkholes. The northern portion of the watershed is mostly underlain by shale and sandstone. The eastern side of the watershed contains forested land along the Massanutten Mountain ridge, agricultural land is common elsewhere in the watershed (Figure 1a). Crop and pasture land covered 40% of the watershed in 2018; cattle and poultry production are the dominant agricultural activities (Chesapeake Bay Program Office, 2022).

The Upper Chester River watershed is a 95 km^2 area within the Coastal Plain physiographic province in Maryland on the Delmarva Peninsula. There are minimal differences in elevation throughout the Upper Chester River watershed. The watershed contains nontidal rivers that flow north and south to the Chester River; the Chester River bisects the area and flows from east to west (Figure 1b). Sandy soils are common throughout the area, with well-drained soils north of the Chester River that are predominantly used for row-crop agriculture. Undeveloped areas and wetlands are common south of the Chester River, where some soils are more poorly drained. Cropland covered 53% of the area in 2017; corn and soybean production are the dominant agricultural activities (Chesapeake Bay Program Office, 2022). A large plant nursery is located in the headwaters of Chesterville Branch, a tributary to the Chester River with a streamgage monitoring station. Activities at this nursery may affect Chesterville Branch water-quality conditions.

Conewago Creek is a 136 km^2 watershed within the Piedmont physiographic province in southeastern Pennsylvania that drains to the Susquehanna River. Sedimentary rocks including sandstone, siltstone, and mudstone represent most of the geologic features, with some intrusions of basalt, primarily along the southern watershed boundary. The watershed contained a mixture of forest (38%) and cropland (33%) in 2017 (Chesapeake Bay Program Office, 2022). Agricultural land use, a combination of crop and pastureland, is common in the central and downstream portions of the watershed (Figure 1c). Forested land is common on elevated ridges in the southern border of the watershed and in the watershed headwaters.

2.2 | Streamflow and water-quality monitoring data

Streamflow was measured every 15 min from streamgages within each showcase watershed (Figure 1; Table 1) following standard U.S. Geological Survey (USGS) methods (Rantz, 1982). The Smith Creek and Upper Chester River watersheds each have one streamgage. Two streamgages are in the Conewago Creek watershed: one in the upper third of the watershed (Bellaire) and one near the watershed outlet (Falmouth). Monthly and high-streamflow water-quality samples were collected from each streamgage since approximately 2011 and were analyzed for concentrations of nitrogen, phosphorus, and suspended sediment (SS). Nitrogen, phosphorus, and SS concentrations were analyzed at laboratories using EPA standard method 4500, EPA method 365.1, and the American Society for Testing and Materials filtration method D3977-97, respectively. At the Smith Creek streamgage, nitrate isotopes of nitrogen and oxygen were collected since 2011 to help characterize nitrogen sources and monthly samples of nitrogen were collected since 1985. Baseflow water-quality samples were collected about four times per year between 2012 and 2020 from Smith Creek monitoring stations that represent discharge from springs (Figure 1a). At Chesterville Branch (the streamgage within the Upper Chester River watershed), six nitrate plus nitrite (NO_x) samples were collected in 1990 and 1991 and 145 NO_x samples were collected from 1996 through 2002. All water-quality samples were collected in accordance with standard methods (U.S. Geological Survey, n.d., variously dated). All streamflow and water-quality data are available from the USGS National Water Information System (U.S. Geological Survey, 2023).

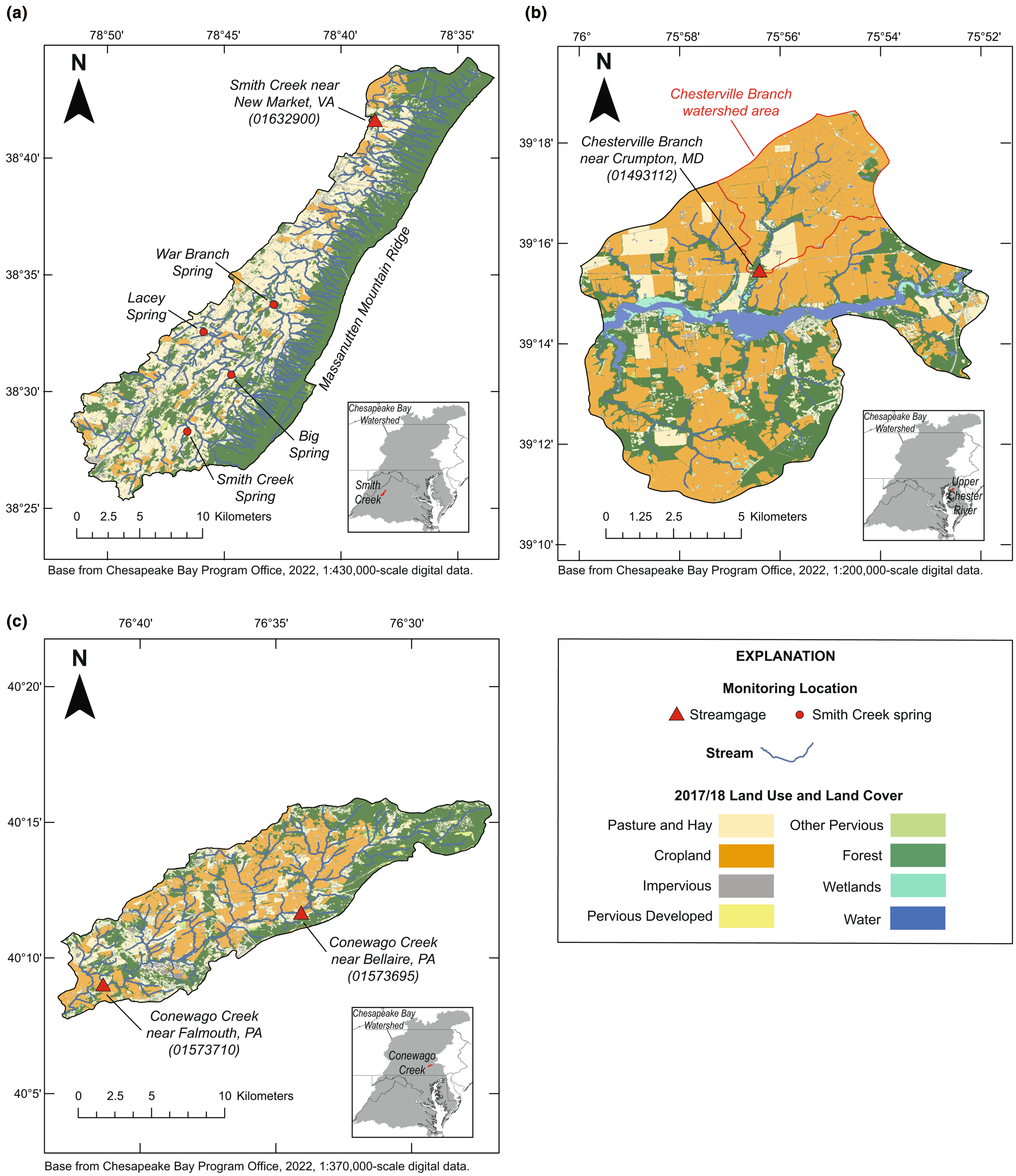

FIGURE 1 Monitoring stations and land use within the (a) Smith Creek, (b) Upper Chester River, and (c) Conewago Creek watersheds (U.S. Geological Survey, 2023).

2.3 | Compilation of water-quality predictor datasets

A suite of spatiotemporal predictor data representing hypothetical water-quality drivers was assembled on annual time-steps for each study watershed. These data include measures of precipitation, air temperature, land use and land cover, MPs, nutrient crop uptake, and values representing source-specific, total, and surplus nutrient inputs. Patterns in each dataset were summarized from 1985 through 2020. This time

TABLE 1 Study watershed streamgage information (U.S. Geological Survey, 2023).

	Streamgage			
Watershed	USGS station number	Name	Drainage area (km^2)	Year established
Smith Creek	01632900	Smith Creek near New Market, VA	242	1963
Upper Chester River	01493112	Chesterville Branch near Crumpton, MD	16	2011
Conewago Creek	01573695	Conewago Creek near Bellaire, PA	53	2012
	01573710	Conewago Creek near Falmouth, PA	123	2011

period provides a multi-decadal perspective on changing watershed conditions and overlaps with water-quality sampling years in the study watersheds. Annual climate data were summarized from weather monitoring stations near the study watersheds (Table S1; data available online: https://www.ncdc.noaa.gov/cdo-web/; access date: 10/01/2023). Annual estimates of land use, land cover, and nutrient inputs and cropland nutrient uptake were summarized from Chesapeake Bay Program's Watershed Model (CBP-WSM) data published by Devereux (2022a). Methods used by the CBP-WSM to produce these datasets are published online (Chesapeake Assessment Scenario Tool, 2023b). Additional details about all water-quality predictor datasets used by this paper are described in Supporting Information.

A comprehensive assessment of the amount, type, location, and timing of agricultural MPs in each study watershed was characterized from water years 2007 through 2020 (October 2006 through September 2020) using datasets provided by federal and state partners. Water years will be referred to as "years" in the remainder of this paper. Federal and state MP datasets were provided through data sharing agreements that protect landowner privacy. Analyses in this paper were completed with all MP data; however, raw MP data are not publicly available and only MPs that met data aggregation limits are reportable (Hively et al., 2018). MPs considered in this paper represent structural or non-structural agricultural practices that received funding or technical assistance from the Natural Resources Conservation Service (NRCS), the Virginia Agricultural Cost-Share Program, or the Maryland Agricultural Water-Quality Cost-Share Program. For Conewago Creek, MPs implemented on conservation easements were also considered. Data representing MPs funded solely by landowners or nonprofit organizations were not included in this paper. Such MPs may represent a large amount of implementation. Based on a 2011 farmer assessment survey, 38% of all MPs within the Upper Chester River watershed were non-cost shared (Nelson & Spies, 2013). Some MPs (such as fencing) are assumed to function for multiple years after installation while others (such as cover crops) have single-year lifespans. MP practice counts and the additional MP characterizations described below represent these single- or multi-year lifespans.

Additional MP characterizations were provided by NRCS expert opinion and the CBP-WSM. Devereux (2022b) reported an integer rating for each MP based on NRCS expert opinion about the ability of MPs to reduce nutrient and (or) sediment loads. MPs rated as a 4 or 5 are expected to most effectively reduce nutrient and (or) sediment loads and were defined in this paper as "high-impact" load-reduction MPs. For each watershed, the expected mass of total nitrogen (TN), total phosphorus (TP), and total SS reduced by nonpoint source MPs from 1985 through 2020 was estimated by the CBP-WSM (Devereux, 2022c). These data represent nutrient and sediment load differences between model scenarios that include and exclude MP effects. Loads modeled by scenarios with MP effects are always lower than the alternative scenario because the CBP-WSM assigns a load-reduction credit to each MP. Modeled MP load reductions represent long-term (eventual) changes as opposed to real-time changes because the CBP-WSM does not model lag times. While the characterization of MP implementation is based on location-specific data provided by federal and state partners, modeled MP load reductions are based on county-scale data reported to the CBP. Devereux (2022c) used agricultural land area to disaggregate MP data from county- to watershed-scales. Therefore, modeled MP load-reduction estimates assume that the amount, type, and timing of MPs in the study watersheds match county-scale patterns.

2.4 | Trend computation

Trends in TN, NO_x, TP, orthophosphate (PO_4), and SS load were computed at the five study-watershed streamgages using monitored water-quality data. These nutrient and sediment loads were reported for all study watershed streamgages by Mason et al. (2023) because the stations are part of the 123-station Chesapeake Bay nontidal monitoring network. Loads and trends in flow-normalized (FN) load were computed with a regression-based approach called the Weighted Regressions on Time, Discharge, and Season (WRTDS) model (Chanat et al., 2015; Hirsch et al., 2010). WRTDS is commonly used to compute water-quality trends in the Chesapeake Bay watershed and other settings (Chanat et al., 2015; Oelsner & Stets, 2019; Stackpoole et al., 2017; Zhang & Blomquist, 2018) and was used in this paper to be consistent with previously published trend results (Mason et al., 2023). A block bootstrap method was used to compute a likelihood statistic for each trend result (Hirsch et al., 2015). The likelihood statistic provides information on whether a trend occurred over the period of record. FN trends remove most variability in water-quality loads associated with year-to-year differences in streamflow. Therefore, trends in FN load are expected to be more closely associated with landscape factors, including MP effects, than streamflow variability. However, long-term increases or decreases in streamflow violate an assumption of WRTDS (Hirsch et al., 2010) and may cause some amount of a FN trend to be caused by changes in streamflow (Choquette et al., 2019). Mason et al. (2023) reported trends in nutrient and sediment load for the Smith Creek streamgage from 2011 through 2020 and, for nitrogen only, from 1985 through 2020. In this paper, we computed FN trends in nutrient and sediment load in Chesterville Branch and Conewago Creek from 2012 and 2013 through 2020, respectively. Trends computed from FN loads with less than 10-years of data can produce uncertain results (Chanat et al., 2015). However, this analysis supported an evaluation of water-quality changes and drivers in two of the showcase watershed study areas: the Chesterville Branch and Conewago Creek watersheds.

Trends in streamflow, air temperature, and precipitation were computed with a Mann-Kendall test. The significance of these trend results was based on an alpha value of less than or equal to 0.05. The *p*-value for the Mann-Kendall tests was computed using an adjustment for serial correlation (Wang & Swail, 2001). Trends were computed on streamflow data measured from each study watershed streamgage. Trends in annual air temperature and precipitation data were computed for each study watershed.

2.5 | Evaluating drivers of water-quality trends

The drivers of nutrient and sediment trends, including MP effects, were evaluated using monitored loads from an 8- to 10-year period (2011, 2012, or 2013 through 2020). Trends in total nutrients, dissolved nutrients, and SS were compared with one another. This comparison can help identify relations between particulate and dissolved water-quality constituents. Changes in load during low- and high-streamflow conditions were computed using methods described by (Zhang et al., 2020). Understanding how loads change under various hydrologic conditions can help evaluate and inform MP actions. Trends in load in the study watersheds were compared with published trends in surrounding Chesapeake Bay agricultural watersheds (Mason et al., 2023). This comparison helps address the overall objective of this paper because unique trends in the study watersheds may suggest that targeted MP implementation contributed to water-quality improvements. Finally, trends in load were evaluated against changes in (1) MP implementation, (2) modeled MP effects, and, for nitrogen and phosphorus, (3) nutrient inputs. A strong correspondence between changes in nutrient loads, MPs, and nutrient inputs may help evaluate effects of these factors.

TN loads have been monitored from the Smith Creek watershed since 1985, a long-term dataset that is not available in the other showcase watersheds. Analysis of this long-term record can provide important insights about water-quality conditions in the Smith Creek watershed. A unique method was thus applied to Smith Creek to evaluate changes in TN load over this long-term monitoring period. A time-series regression model was used to evaluate how water-quality predictor variables (streamflow, air temperature, precipitation, land use, nutrient terms, and MPs; Table S2) explained multi-decadal changes in annual FN TN loads at the Smith Creek streamgage. This modeling approach was designed to identify potential drivers of load while addressing the statistical and theoretical complexities of working with time-series water-quality data. A discussion of methods related to this modeling approach is provided in Supporting Information.

3 | PATTERNS IN POTENTIAL NUTRIENT AND SEDIMENT DRIVERS

3.1 | MP implementation

Patterns in MP implementation were evaluated to better understand how the number of MPs changed over time and how the area of MPs related to agricultural land area. The number of MPs increased from 2007 through 2020 in each watershed (Figure 2a). Over these years, the number of MPs increased by approximately 1200 in Smith Creek, 250 in the Upper Chester River and 620 in Conewago Creek. Higher MP implementation rates occurred after 2010, likely because of funding and outreach efforts associated with the showcase watershed study and other efforts to meet Chesapeake Bay water-quality goals. Other agricultural regions of the Chesapeake Bay watershed have seen substantial increases in MPs since the early 2000s (U.S. Department of Agriculture, 2013). The area of MPs, representing MPs reported in or converted to square kilometers, (1) was highest in the Upper Chester River and increased by about 20% through 2020, (2) increased the most in Conewago Creek through 2020, and (3) was lowest in Smith Creek during most years, but doubled through 2020. The area of MPs exceeded agricultural land area in the Upper Chester River in all years and, after 2015, in Conewago Creek (Figure 2b). MP area can exceed agricultural land area when multiple MPs are used in the same field or pasture. MP area in Smith Creek represented about 20% to 30% of agricultural land area. The percentage is likely lower in Smith Creek than the other watersheds because Smith Creek contains more pasture land. Unlike crop fields, there are fewer MPs that apply to entire pastures.

The suite of MPs in each watershed was designed to achieve a variety of environmental, agronomic, and economic benefits, with some MPs expected to reduce nutrient and sediment loads. Most MPs in the Upper Chester River watershed were expected to have a high impact on nutrient- and (or) sediment-load reductions, about 50% and 30% of practices in the Conewago Creek and Smith Creek watersheds had similar

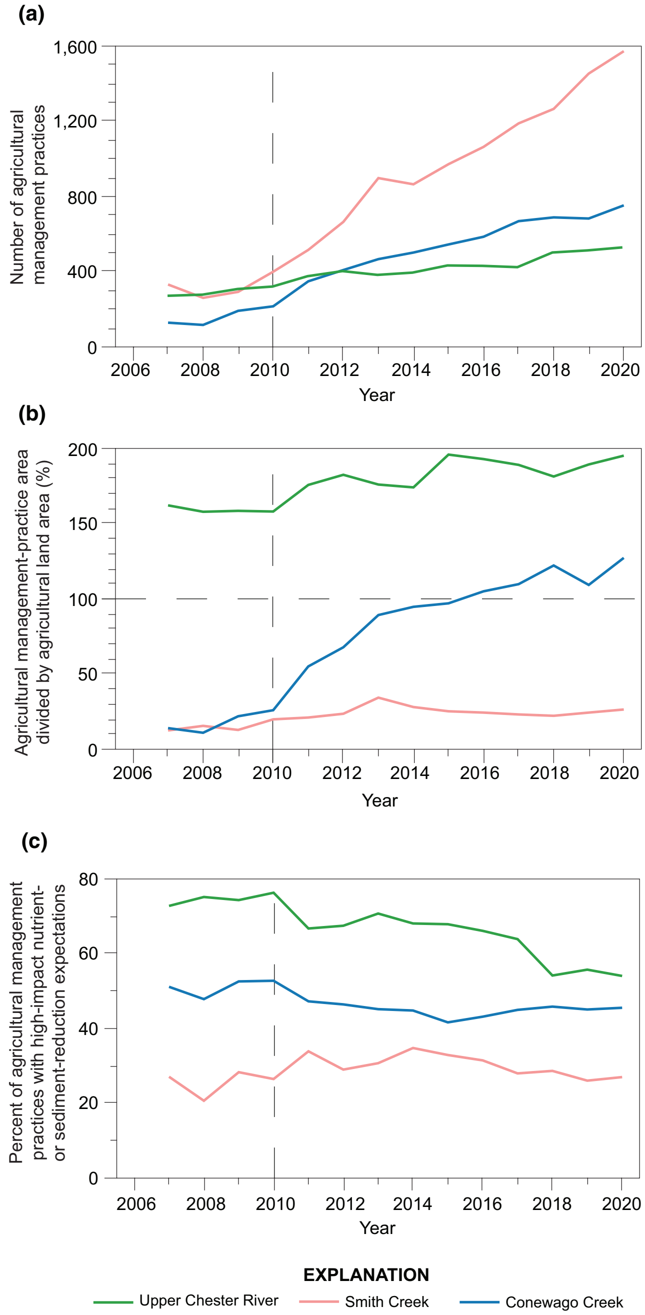

FIGURE 2 Agricultural management practice (MP) (a) number, (b) area as a percentage of agricultural land area, and (c) percentage with high-impact nutrient- or sediment-reduction expectations in the study watersheds from 2007 through 2020; vertical dashed lines on panels (a–c) indicate the designation of the showcase watershed study in 2010; a horizontal dashed line on panel (b) is included to highlight 100%.

expectations, respectively (Figure 2c). The percentage of high-impact MPs remained stable or slightly decreased over time in each watershed. Therefore, the increased number of MPs from 2007 through 2020 did not represent a proportional shift towards the use of high-impact nutrient or sediment practices.

There were eight types of high-impact nutrient or sediment load-reduction MPs implemented in the study watersheds (Table 2). High-impact nutrient load-reduction MPs were generally designed to reduce or prevent the delivery of landscape-applied nutrients to streams through the use of (1) vegetated areas, (2) manure-storage facilities, or (3) fencing to exclude livestock from streams. Other than nutrient-management plans, high-impact nutrient MPs were not designed to directly lower landscape nutrient applications. High-impact sediment load-reduction MPs were generally designed to slow surface runoff, retain soil on agricultural fields, and (or) lower rates of streambank erosion.

The most commonly implemented MPs represent the agricultural activities and resource-management priorities in each watershed. In the Conewago Creek watershed, common MPs included tillage management, filtering, terracing, fencing, and underground outlets (Figure S1A).

TABLE 2 Description of MPs installed in the study watersheds from 2007 through 2020 with an expected high impact on nutrient and or sediment reductions.

	Expected high-impact load-reduction effect		
Practice type	**Nutrient**	**Sediment**	**Description**
Buffers	Yes	Yes	Near-stream vegetated areas that help control nutrients, sedimentation, erosion, and other environmental concerns
Conservation cover	Yes	Yes	Practices that establish and maintain vegetative cover to protect soils and water resources
Stream fencing	Yes	Yes	Fencing along streams to reduce erosion, sedimentation, and pollution from agricultural nonpoint sources
Nutrient management	Yes	No	Practices that manage the amount, source, placement, and timing of plant nutrients and soil amendments
Watering systems	Yes	No	Practices designed to provide an alternative to watering livestock directly from streams, rivers, or lakes
Animal waste management	Yes	No	Practices designed to store or treat manure and other agricultural waste products to promote more flexible waste utilization
Filtering	No	Yes	Vegetated areas designed to remove sediment and other pollutants from surface-water runoff
Tillage management	No	Yes	Practices that address the amount, orientation, and distribution of crop and other plant residue on the soil surface

Filtering and tillage management MPs have a high-impact sediment-load reduction expectation, the number of these MPs increased from 2007 through 2020. Terracing and underground outlets may also reduce sediment loads by lowering rates of surface water runoff and managing flooding.

A diverse mixture of MPs were commonly used in the Smith Creek watershed, reflecting the combination of animal and cropland agricultural activities in the watershed (Figure S1B). Fencing, watering systems, and livestock pipelines increased from 2007 through 2020; a suite of MPs that provide livestock with alternative sources of water when they are excluded from streams. There were 56 animal-waste-management systems in 2020, a high-impact nutrient load-reduction MP that allows farmers to store or treat manure. Common cropland MPs in Smith Creek included cover crops and tillage management. Cover crops peaked in 2013 and decreased through 2020 whereas tillage management increased over time.

The most commonly implemented MPs in the Upper Chester River watershed were cover crops, nutrient-management plans, tillage management, and filtering (Figure S1C). About 55% of cropland was enrolled in cover crops in 2020, a value that tripled from 2007 through 2020. Between 74% and 91% of cropland was estimated to have a nutrient-management plan, a high-impact nutrient load-reduction MP. Between 67% and 79% of cropland was estimated to use tillage management, a high-impact sediment load-reduction MP. Filtering MPs also have a high-impact sediment load-reduction expectation and increased over time in the Upper Chester River watershed.

MPs were expected to reduce nutrient and sediment loads in each watershed. As estimated by the CBP-WSM, the amount of nutrient and sediment loads reduced by MPs generally increased from 1985 through 2020 in all watersheds (Figure 3). Changes in modeled MP load reductions result from changes in the number and type of MPs reported to the CBP. These modeled reductions represent long-term (eventual) changes as opposed to real-time changes because the CBP-WSM does not model lag times. The largest modeled MP load reductions, expressed as a percentage of load per year, were estimated for Chesterville Branch. Modeled loads in the early 2010s would have been 30% to 50% higher if no MPs were implemented in the Chesterville Branch watershed. In the Smith Creek and Conewago Creek watersheds, modeled loads would have been about 15% to 30% higher without MPs. Modeled estimates of MP effects may help explain changes in monitored nutrient and sediment loads, relations that are evaluated in the Section 4.3.

3.2 | Climate and streamflow

Long-term trends and annual variability in climatic conditions can affect the sources and amounts of nitrogen, phosphorus, and sediment delivered to streams. There was a significant increasing trend in average air temperature from 1985 through 2020 in the Upper Chester River ($p=0.0034$) and Conewago Creek ($p=0.0031$) watersheds (Figure S2A). Average air temperature increased by 0.03°C per year in these watersheds, similar to reported annual air temperature increases of 0.02°C from 1960 through 2020 in the mid-Atlantic region, USA (Rice

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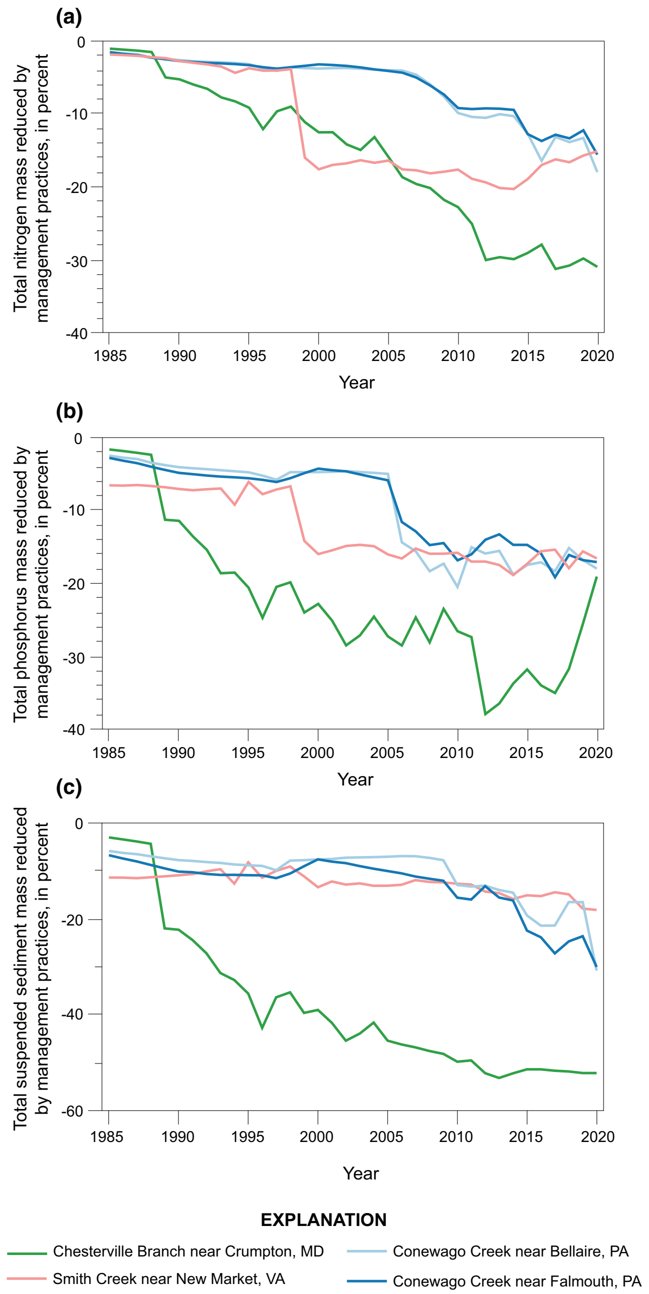

FIGURE 3 Modeled MP percent reductions of (a) total nitrogen (TN), (b) total phosphorus (TP), and (c) total suspended sediment (SS) in the study watersheds from 1985 through 2020 (Devereux, 2022c).

& Jastram, 2014). Average air temperature decreased slightly since 1985 in the Smith Creek watershed but this change was not statistically significant. Years with the most precipitation varied among the watersheds (Figure S2B); however, trends from 1985 through 2020 were not significant in any watershed.

Streamflow is an important driver of water-quality conditions and can be affected by patterns in precipitation and temperature. Annual amounts of streamflow at the Smith Creek streamgage varied from 1985 through 2020, but did not consistently increase or decrease over this period. The Smith Creek streamgage had no significant trends in streamflow from 1970 through 1999 or from 2000 through 2018 (Mason et al., 2021). Long-term streamflow data were not available at the Conewago Creek or Chesterville Branch streamgages. Streamflow generally increased from 2011, 2012, or 2013 through 2020 at all streamgages, but most of these changes were not significant (Figure S3).

3.3 | Nutrient inputs and cropland nutrient balances

Evaluating the magnitude and changes in agricultural nutrient inputs is important to understand water-quality responses in the study watersheds. In 2020, agricultural nutrient inputs (manure, fertilizer, and nitrogen crop fixation) provided more than 75% of total nutrient inputs in the study watersheds (Figure 4). Manure-nutrient inputs considered in this paper are defined as the amount of nutrients from farm-animal manure that are applied as fertilizer to agricultural lands. These inputs do not include manure directly deposited onto pasture. Manure represented more than half of all nitrogen and phosphorus inputs in Smith Creek and was the largest nutrient input in Conewago Creek. Fertilizer and crop nitrogen fixation were the largest nutrient inputs in the Upper Chester River watershed. Non-agricultural nutrient inputs included atmospheric deposition, fertilizer applied to turfgrass, and point and non-point wastewater inputs. In 2020, atmospheric deposition was the largest percentage of non-agricultural nutrient inputs in each study watershed (Figure 4). Permitted wastewater point sources represented less than 4% of all nutrient inputs in Smith Creek and Conewago Creek and were not present in the Upper Chester River watershed from 1985 through 2020. Even small point source inputs can have important water-quality impacts because, unlike nonpoint sources, point sources are discharged directly to streams.

Non-agricultural nutrient inputs decreased in most study watersheds, mostly because of atmospheric deposition reductions (Tables S4 and S5). Decreases in atmospheric deposition have been observed throughout the Chesapeake Bay watershed and have largely resulted from Clean Air Act amendments in the 1990s (Burns et al., 2021; Eshleman & Sabo, 2016; Sabo et al., 2022). Decreased atmospheric deposition inputs, however, were typically offset by increased agricultural inputs resulting in amounts of nitrogen and phosphorus that were higher in 2020 than 2011 or 1985 in most study watersheds (Figure 5) Agricultural nutrient inputs reflect changes to animal populations and crop activities. Increased agricultural nutrient inputs were not related to differences in land use because the amount of agricultural land in each watershed was mostly unchanged over time (Table S6). Manure nutrient inputs were about 40% to 80% higher in 2020 than 1985 in all watersheds (Tables S4 and S5). Larger poultry populations accounted for most of the increased manure inputs in the Upper Chester River watershed; poultry and cattle populations both contributed to increased manure in the Smith Creek and Conewago Creek watersheds (Figure S4). The harvested yield of corn and soybeans was two to three times higher in 2017 than in 1987 in counties that contain the study watersheds (Figure S5). As a result of soybean increases, the amount of atmospheric nitrogen fixation increased (Table S4), although some of these increases were offset by crop nitrogen removal. Nitrogen from agricultural fertilizer increased from 1985 through 2020 in the Upper Chester River watershed, but, for nitrogen and phosphorus, decreased in the Smith Creek and Conewago Creek watersheds (Tables S4 and S5). In these two watersheds, decreases in fertilizer may have resulted from the increased use of manure to meet crop nutritional requirements and (or) MPs that addressed fertilizer application rates.

Nonpoint source nutrient inputs can be removed from the landscape by agricultural crop and animal activities and never reach streams. However, nutrient inputs that exceed agricultural demand ("surplus inputs") have been documented throughout the Chesapeake Bay watershed and can cause water-quality impairments (Chang et al., 2021; Dupas et al., 2020; Sabo et al., 2022; Zhang et al., 2022). The calculation of surplus nutrient inputs (which is fully described in Supporting Information) likely underrepresents the complexity of nutrient cycling in agricultural landscapes. For example, estimated amounts of nutrients removed by livestock and poultry were not available and not represented in this calculation. From 1985 through 2020, all study watersheds had a surplus amount of nitrogen applied to cropland, amounts that increased over time (Figure S6A). The Smith Creek and Conewago Creek watersheds had surplus phosphorus inputs, amounts that increased from 1985 through 2020 (Figure S6B). In the Upper Chester River watershed, surplus phosphorus inputs decreased since 1985 and, by 2003, the estimated mass of phosphorus removed by crops exceeded cropland phosphorus inputs. Although these changes might improve water-quality conditions in the Upper Chester River watershed, it can take decades to overcome the effects of legacy phosphorus stored in soils (Kleinman et al., 2019).

4 | NUTRIENT AND SEDIMENT RESPONSES AND DRIVERS

4.1 | Nutrient and sediment yields

Nutrient and sediment yields represent annual loads divided by watershed area; yields are helpful for comparing loads across watersheds of varying size. The yields described below represent the average of annual conditions monitored from 2016 through 2020. Nutrient and sediment yields differed among the study watersheds (Table 3). Chesterville Branch TN and NO_x yields were more than twice as high as any other study watershed and were higher than any other station in the Chesapeake Bay nontidal monitoring network (Mason et al., 2023). Differences in TN yield among the study watersheds were strongly correlated with nitrogen inputs: watersheds with higher area-normalized nitrogen inputs had higher TN yields (Figure S7A). Based on this relation, reducing nitrogen inputs in the study watersheds may be an important step toward lowering nitrogen loads.

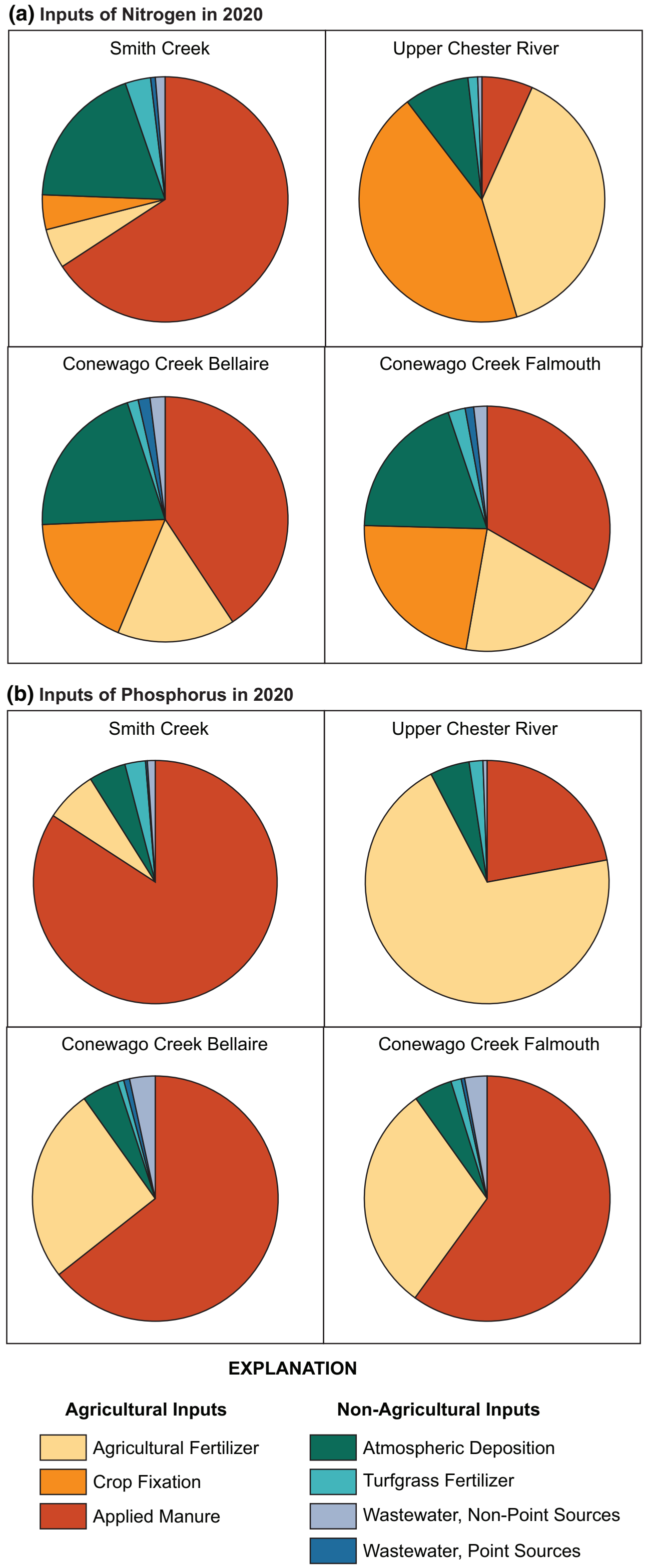

FIGURE 4 Agricultural and non-agricultural inputs of (a) nitrogen and (b) phosphorus in the study watersheds in 2020 (Devereux, 2022a).

Yields of TP and PO_4 at Conewago Creek Falmouth were highest among the study watersheds (Table 3) and all Chesapeake Bay non-tidal monitoring network stations (Mason, Colgin, and Moyer 2023). PO_4 represented about 66% of the TP yield in Conewago Creek and only about 20% of the TP yield in the Smith Creek and Chesterville Branch watersheds. Elevated PO_4 yields in Conewago Creek may result from high soil-phosphorus concentrations following years of elevated phosphorus inputs in southeastern Pennsylvania (Sabo et al., 2021). Reducing soil phosphorus concentrations may be important for lowering phosphorus loads in Conewago Creek. Differences in TP yield

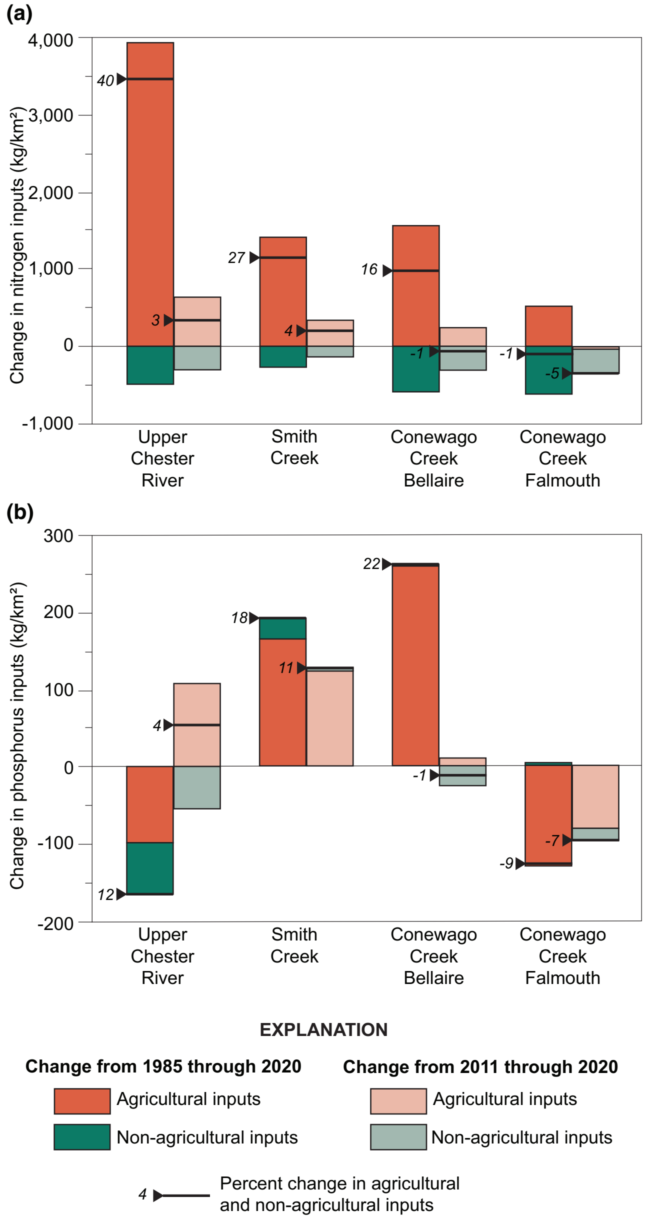

FIGURE 5 Changes in (a) nitrogen and (b) phosphorus nutrient inputs from 1985 through 2020 and 2011 through 2020 in the study watersheds (Devereux, 2022a).

TABLE 3 TN, NO_x, TP, PO_4, and SS yields in the study watersheds, representing the average annual conditions from 2016 through 2020.

Watershed	Nutrient yield (kg/km^2) TN	NO_x	TP	PO_4	SS
Smith Creek	877	656	78	15	54,922
Chesterville Branch	4326	3833	167	35	61,871
Conewago Creek, Bellaire	1603	1132	152	100	33,401
Conewago Creek, Falmouth	1973	1356	230	150	60,414

among the study watersheds were not related to differences in area-normalized watershed inputs (Figure S7B). In addition to legacy phosphorus in soils, TP yields may have been affected by SS, which plays a critical role in the delivery of TP to streams (Ator et al., 2011; Sharpley et al., 2013).

SS yields were lowest in Conewago Creek Bellaire and were approximately twice as high in the Smith Creek, Chesterville Branch, and Conewago Creek Falmouth watersheds (Table 3). The Conewago Creek Bellaire watershed has the least amount of agricultural and urban land, areas that can contribute high amounts of sediment throughout the Chesapeake Bay watershed (Brakebill et al., 2010). Streambanks are likely the dominant source of SS in the Smith Creek watershed (Gellis & Gorman-Sanisaca, 2018). Reducing SS loads in Smith Creek may therefore require MPs that lower rates of streambank erosion (Clifton et al., 2022). Streambanks may be an important SS source in the other study watersheds along with upland soil erosion, which often contribute SS in agricultural areas (Noe et al., 2020). Without knowledge of specific SS sources, lowering SS yields in the Chesterville Branch and Conewago Creek watersheds may require a suite of MPs that address both streambank and upload sediment sources.

4.2 | Nutrient and sediment trends

Monitored trends in FN nutrient and sediment loads provide an assessment of how water-quality conditions are changing over time after removing interannual streamflow differences. Approximately one-third of all nutrient and sediment trends decreased from the early 2010s through 2020. All loads in the Conewago Creek Falmouth watershed decreased, along with PO_4 in Smith Creek and SS in Conewago Creek Bellaire. Nearly half of all trends increased over time. Although the use of MPs in the study watersheds was expected to reduce nutrient and sediment loads, monitored trends in load did not consistently decrease.

TN and NO_x loads increased in Smith Creek and Conewago Creek Bellaire, decreased in Conewago Creek Falmouth, and had no trend in Chesterville Branch (Table 4). NO_x is the largest fraction of TN load in all watersheds; however, MPs that control the delivery of particulate nitrogen, organic nitrogen, and ammonia may be important for reducing nitrogen loads. These non-NO_x fractions of TN increase with streamflow (Figure S8) because of their association with SS. In all watersheds, nitrogen loads had the largest changes on days with above-average streamflow (Figure S9A). Therefore, increases in TN in Smith Creek and Chesterville Branch may have been related to increased SS loads. Conversely, decreased SS loads in Conewago Creek may have helped lower TN loads. Conewago Creek SS decreases in the downstream watershed (Falmouth) were approximately three times larger than the upstream watershed (Bellaire); TN loads decreased in the downstream watershed and increased in the upstream watershed. These patterns highlight the potential TN benefits that that could be associated with MPs that address SS loads.

TABLE 4 Nutrient and sediment trends from 2011, 2012, or 2013 through 2020 in the study watersheds.

Watershed	Parameter	Trend period	Trend direction	Change in load (kg/km^2)	Change in load (%/year)
Smith Creek	TN	2011–2020	Increasing	61	1
	NO_x		Increasing	44	1
	TP		Increasing	26	3
	PO_4		Decreasing	−4.2	−2
	SS		Increasing	41,136	7
Chesterville Branch	TN	2012–2020	No trend	−17	−0.0
	NO_x		No trend	−29	−0.1
	TP		Increasing	73	5
	PO_4		Increasing	6.0	2
	SS		Increasing	40,498	8
Conewago, Bellaire	TN	2013–2020	Increasing	219	2
	NO_x		Increasing	217	3
	TP		No trend	−3.6	0.3
	PO_4		No trend	0.1	0.0
	SS		Decreasing	−20,769	−4
Conewago, Falmouth	TN	2013–2020	Decreasing	−154	−1
	NO_x		Decreasing	−41	−0.4
	TP		Decreasing	−35	−2
	PO_4		Decreasing	−37	−3
	SS		Decreasing	−67,075	−6

TP loads increased in Smith Creek and Chesterville Branch, decreased in Conewago Creek Falmouth, and had no trend in Conewago Creek Bellaire (Table 4). In all watersheds, most changes in TP load occurred on days with above-average streamflow (Figure S9B). This pattern highlights the importance of MPs that reduce TP during elevated streamflow, which could likely be achieved by controlling SS. Such MPs may be most beneficial in Smith Creek and Chesterville Branch, where particulate phosphorus concentrations were the largest fraction of TP concentrations (Figure S10). Conewago Creek SS decreases were larger downstream (Falmouth) than upstream (Bellaire), which may have contributed to downstream TP load reductions. In addition to MPs that reduce SS, MPs that address PO_4 loads may be required to achieve larger and (or) sustained TP reductions in Conewago Creek.

Trends in nutrient and sediment loads in the study watersheds can be compared to trends in nearby agricultural watersheds. Most of these comparison watersheds are larger than the study watersheds but have similar land uses as the study watersheds (Table S7). Assuming climatological and nutrient input patterns were similar between the study and comparison watersheds, trend differences may be related to the targeted use of MPs in the study watersheds. Such interpretation, however, is limited because MP patterns were not summarized in the comparison watersheds. Nutrient and sediment trends were similar between most study and comparison watersheds (Figure 6). From 2011 through 2020, TN loads in Shenandoah Valley watersheds changed by less than 2% per year, similar to the 0.8% annual TN increase in Smith Creek. The Smith Creek increase in TP and SS load, however, differed from decreasing TP and SS loads in two larger watersheds on the North and South Fork of the Shenandoah River. From 2012 through 2020, TN loads in Chesterville Branch and at five of the six comparison watersheds on the Delmarva Peninsula changed by less than 1% per year. TP and SS loads increased in Chesterville Branch and in most comparison watersheds. From 2013 through 2020, TN loads in southern Pennsylvania watersheds that drain to the Susquehanna River typically changed by less than 2% per year. These changes were similar to Conewago Creek TN load trends. TP and SS loads were lower in 2020 than in 2013 in Conewago Creek; similar decreases were observed in about half of the comparison watersheds. These comparisons provide inconclusive evidence that the targeted use of MPs in the study watersheds contributed to load reductions. Rather, trend similarities between study and comparison watersheds may suggest some regional consistency in factors affecting trends in agricultural Chesapeake Bay watersheds.

4.3 | Potential nutrient and sediment drivers

Contemporary and historical anthropogenic activities affect water-quality responses in agricultural Chesapeake Bay watersheds (Ator et al., 2020; Chang et al., 2021; Clune et al., 2021; Fanelli et al., 2019; Noe et al., 2020; Ryberg et al., 2018; Zhang et al., 2022) and likely contributed to changing nutrient and sediment loads in the study watersheds. There were minimal land use changes in each watershed in recent decades (Table S6) but, in general, the number of MPs and amount of nutrient inputs increased. The potential water-quality effects of MPs and nutrient inputs are described below.

During the period of water-quality trend analysis (early 2010s through 2020), the number of MPs and high-impact load-reduction MPs increased in all watersheds (Table 5). Normalized by watershed area, increases in MPs and high-impact nutrient MPs were larger in Smith Creek than the other study watersheds. From 2011 through 2020, 152 high-impact nutrient practices were installed in Smith Creek, including watering systems and animal-waste MPs. These MPs are designed to reduce nutrient loads by reducing the amount of manure nutrients delivered to streams (Table 2), the largest source of nitrogen in Smith Creek (Hyer et al., 2016). Despite the installation of such MPs, TN, NO_x, and TP loads increased in Smith Creek (Table 4). Smith Creek PO_4 loads decreased. The cause of changing PO_4 loads in Smith Creek is uncertain but may include the amount of nutrients discharged from a permitted poultry-processing facility, a point source near the Smith Creek streamgage.

MPs, including those with high-impact load reduction expectations, were used throughout the Upper Chester River watershed (Figure 2), but nutrient and sediment loads in Chesterville Branch did not decrease from 2012 through 2020 (Table 4). Cover crop and nutrient management plans were common in the Upper Chester River watershed and can address fertilizer inputs, the largest source of nutrients in the watershed (Hyer et al., 2016). Cropland MPs such as cover crops and nutrient management have been shown to reduce nutrient and sediment loads (Blanco-Canqui, 2018; Hanrahan et al., 2021; Hively et al., 2020; Kroll & Oakland, 2019; McCarty et al., 2008; Thapa et al., 2018). Tillage management was also common in the Upper Chester. Tillage management can reduce sediment loads but have also been associated with increased dissolved nutrient loads (Baker et al., 2017; Fanelli et al., 2019; Sharpley et al., 1994; Tuppad et al., 2010).

Increases in high-impact sediment MPs were larger in Conewago Creek than the other study watersheds (Table 5). There were 110 such MPs installed in Falmouth watershed from 2013 through 2020, mostly filtering and tillage management MPs. Conewago Creek was the only study watershed where SS loads decreased (Table 4). The ability of tillage management to reduce soil runoff and sediment erosion has been well established (Baker & Laflen, 1983; Shipitalo & Edwards, 1998) and MPs that trap agricultural runoff (such as vegetated filter strips, buffers, and grassed waterways) can reduce sediment loads (Dillaha et al., 1988; Gall et al., 2018; Liu et al., 2008). In all watersheds, factors related to climatic variability, landscape conditions, and human activities may be offsetting or delaying water-quality responses to increased MP implementation.

Nutrient load trends and expected MP effects were likely affected by changing nutrient inputs. During water-quality trend years, nitrogen and phosphorus inputs increased in Smith Creek and decreased in Chesterville Branch and Conewago Creek (Figure 7). In all watersheds,

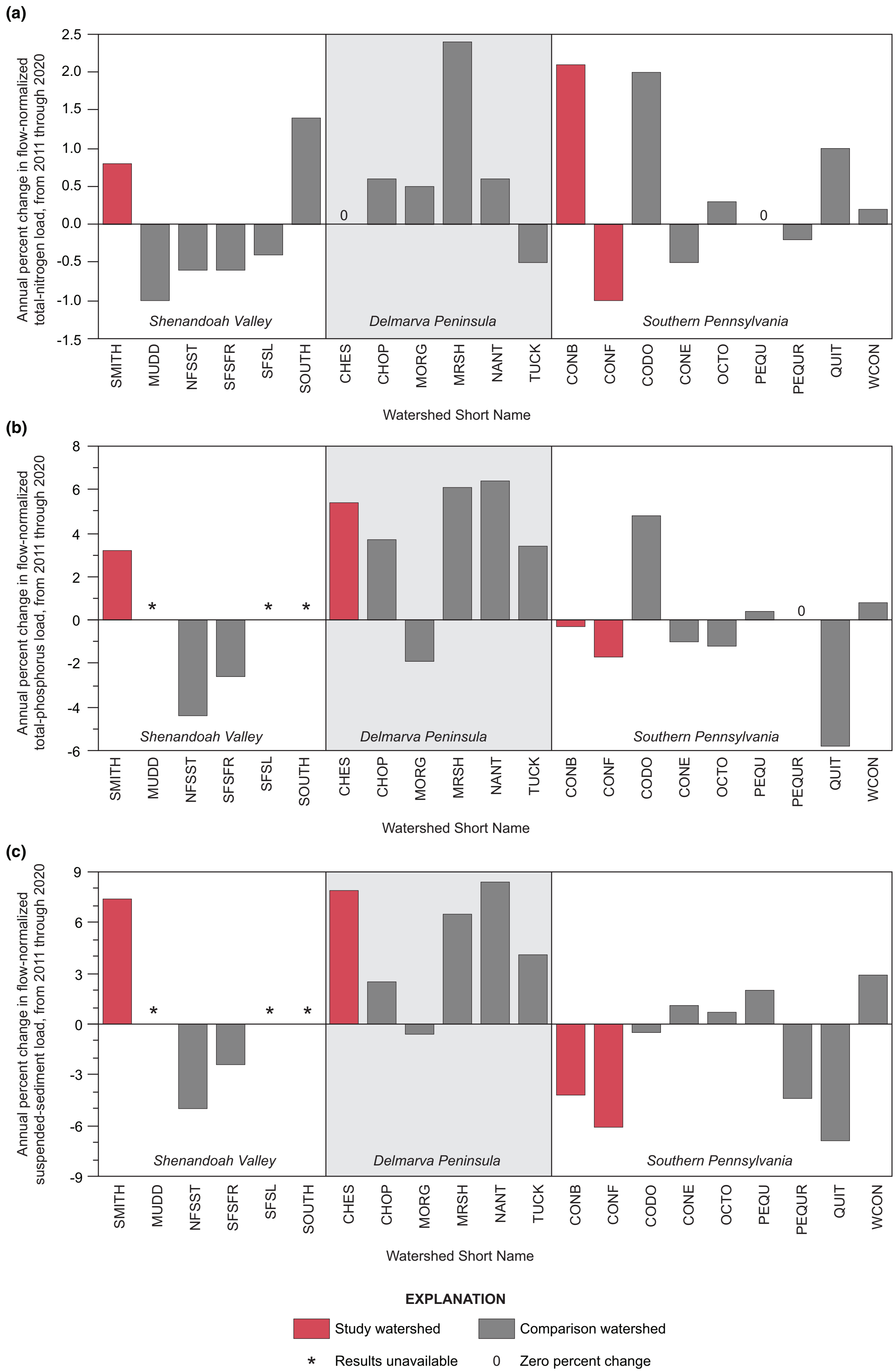

FIGURE 6 Annual percent changes of (a) TN, (b) TP, and (c) SS at study watershed streamgages and comparison nontidal network monitoring stations (Mason et al., 2023). Watershed short names are defined in Table S7.

TABLE 5 Change in the number of area-normalized MPs in the study watersheds during water-quality trend years.

Watershed	Trend period	Watershed area (km^2)	Change in MPs (n/km^2)		
			All MPs	High-impact nutrient MPs	High-impact sediment MPs
Smith Creek	2011–2020	242	4.0	0.6	0.4
Chesterville Branch	2012–2020	16	1.7	0.0	0.2
Conewago, Bellaire	2013–2020	53	1.5	0.1	0.6
Conewago, Falmouth	2013–2020	123	2.1	0.2	0.9

nutrient input changes were similar to changes in the surplus amount of nutrients applied to cropland. In Smith Creek, monitored TN and TP loads increased from 2011 through 2020, possibly in response to increased nutrient inputs. These results suggest that reducing nutrient loads in Smith Creek may rely on reducing nutrient inputs to align with crop nutritional requirements. Proper methods and timing of nutrient inputs are also likely needed to prevent excess nutrient delivery to surface waters.

In Conewago Creek Falmouth, monitored TN and TP loads decreased from 2013 through 2020, possibly in response to decreased nutrient inputs (Figure 7). In Conewago Creek Bellaire, TP had a similar pattern of decreased loads and inputs; however, TN loads increased. The upstream and downstream Conewago Creek watersheds may have different TN trend directions because of changing point-source inputs. In Conewago Creek, nitrogen point-source inputs decreased by about 50% from 2018 through 2020. Most reductions were from two permitted facilities: a travel center and an industrial park (Figure S11A). The travel center discharges to a Conewago Creek tributary that flows through a wetland complex before reaching the Bellaire streamgage. The industrial park discharges directly to Conewago Creek upstream of the Falmouth streamgage. Given this close hydrologic connectivity, point-source reductions from the industrial park may have contributed to lower amounts of nitrogen in the Falmouth watershed after 2018, as observed by sampled TN concentrations (Figure S11B). Conversely, point-source reductions from the travel center may be less apparent in the Bellaire watershed. Even though point-source inputs represent a small fraction of total nutrient inputs in Conewago Creek, these results highlight the importance of managing point-source loads.

In Chesterville Branch, monitored TN loads had no trend and TP loads increased despite nutrient inputs and surplus nutrient applications to cropland that were lower in 2020 than 2012 (Figure S7). A large plant nursery in the headwaters of Chesterville Branch may have affected these monitored loads; activities at this nursery are likely not well characterized by available nutrient-input data. In Chesterville Branch and in the other study watersheds, nutrient loads may reflect agricultural activities from previous decades. Surplus nitrogen inputs to cropland increased from 1985 through 2020 in the Upper Chester River watershed, likely because of increasing poultry populations (Figure S4B) and crop production (Figure S5). Possibly in response to these activities, groundwater concentrations of dissolved nitrogen increased throughout the Delmarva Peninsula in recent decades (Fleming et al., 2017; Lindsey & Rupert, 2012) and NO_x concentrations in Chesterville Branch were higher since 2011 than the early 1990s and 2000s (Figure S12). Although NO_x delivered by groundwater to Chesterville Branch can be decades old (Hyer et al., 2016), the stable trends in TN load monitored in recent years may, in part, reflect the chemistry of stormwater runoff. About 20% of streamflow in this watershed is produced from stormwater runoff, which is, on average, hours to months old (Sanford & Pope, 2013). Nitrogen in stormwater runoff may be more responsive to recent MP effects than nitrogen delivered through groundwater.

Monitored phosphorus loads increased in Chesterville Branch from 2012 through 2020 despite decreases in phosphorus inputs and surplus phosphorus applications to cropland from 1985 through 2020. In addition to the previously described linkage between SS and TP loads, increased TP loads in Chesterville Branch may result from high soil-phosphorus concentrations caused by decades of surplus agricultural inputs (Ator & Denver, 2015). Even when phosphorus inputs are reduced, it can take decades to reduce legacy soil-phosphorus concentrations without mining down of soil phosphorus, challenging the ability of MPs to quickly improve water-quality conditions (Kleinman et al., 2019; Sharpley et al., 2013). Considering the history of intensive agricultural activities in the Upper Chester River watersheds, MPs may have helped prevent larger nutrient load increases; however, confidently identifying MP effects remains a challenge for this and similar monitoring-based water-quality studies.

Estimates from the CBP-WSM can help evaluate how modeled MP load reductions relate to changes in nutrient inputs and monitored trends in load. For most nutrient and sediment loads, modeled MP load reductions were larger in 2020 than 2011, 2012, or 2013 (represented as negative values in Figure 7). These modeled effects are plausible, as the number of MPs increased in all watersheds over this time period (Figure 2a). The four improving trends in monitored load (TN, TP, and SS at Conewago Creek Falmouth and SS at Conewago Creek Bellaire) all had larger modeled MP load reductions in 2020 than 2013. However, modeled MP load reductions did not correspond with improvements in other monitored loads. These relations demonstrate that expected MP load reductions are not consistently observed by monitoring data. Except for TN in Conewago Creek Bellaire, changes in nutrient inputs were always much larger than changes in modeled MP effects. Although not all nonpoint source inputs reach streams, these patterns show how large increases in nutrient inputs can contribute to the challenge of identifying the water-quality effects of MPs.

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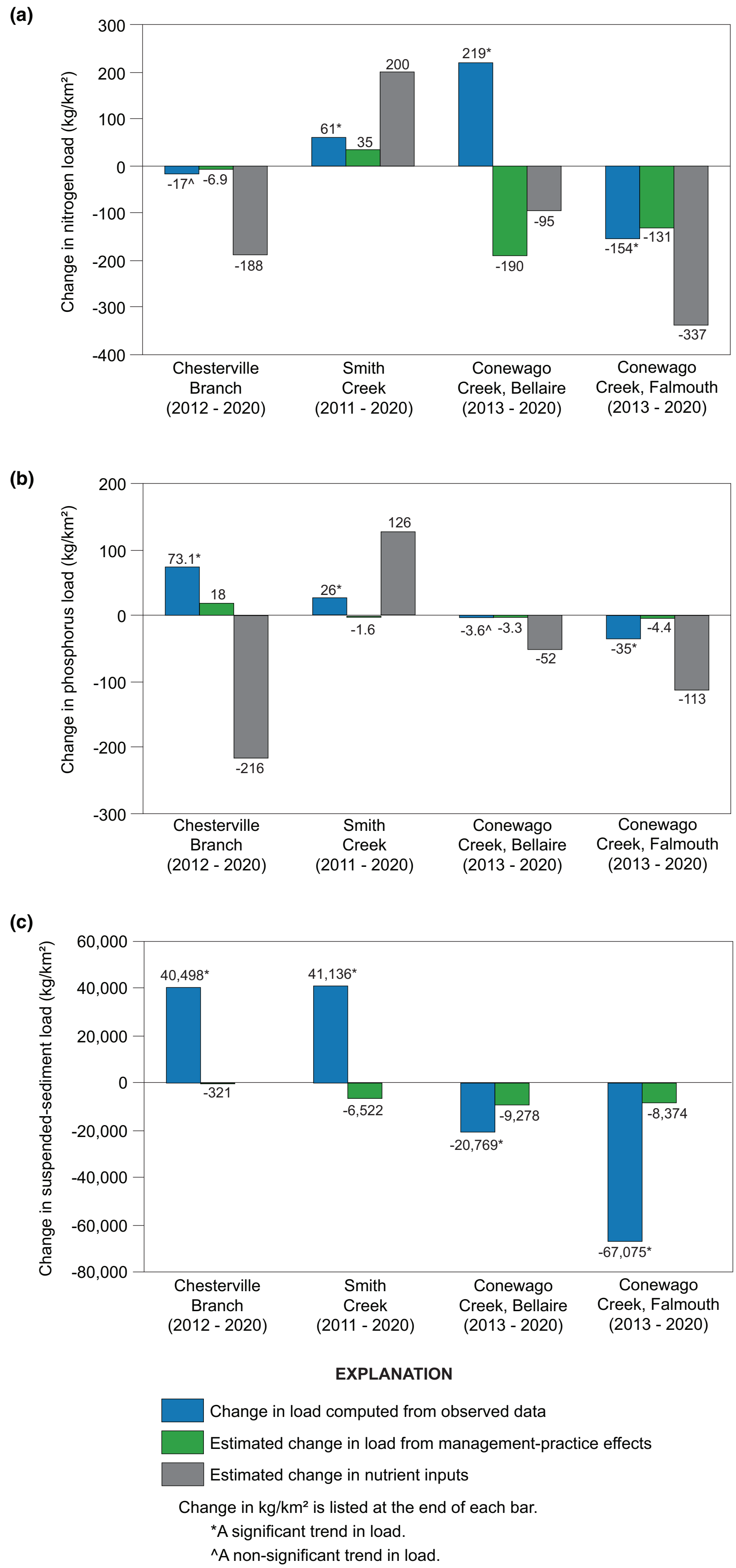

FIGURE 7 Graphs showing changes in flow-normalized (FN) load (Mason et al., 2023), estimated MP load reductions (Devereux, 2022c), and estimated nutrient inputs (Devereux, 2022a) in the study watersheds for (a) nitrogen, (b) phosphorus, and (c) SS.

5 | NITROGEN RESPONSES AND DRIVERS IN THE SMITH CREEK WATERSHED

5.1 | Smith Creek nitrogen responses

Nitrogen has been measured at the Smith Creek streamgage since 1985, a long-term water-quality record that is not available for the other study watersheds. Insights about water-quality responses and drivers over a multidecadal period can help inform recent management activities; therefore, the following discussion provides a detailed analysis of long-term nitrogen conditions at the Smith Creek streamgage. FN loads of TN increased by about 13,000 kg, or 7%, from 1985 through 2020 at the Smith Creek streamgage (Figure 8a). Over this same 36-year period, FN NO_x loads increased by about 4000 kg, but this 3% change was not a statistically significant trend (Figure 8b). Changes in FN TN and NO_x load were nonlinear. Notably, there was a localized peak in the early 2000s and increases since about 2010.

Trends in Smith Creek nitrogen load differed by hydrologic condition. On average, 80% to 85% of the TN and NO_x load was delivered to Smith Creek during high flows (days when daily flow exceeded the 1985 through 2020 median value). For both TN and NO_x, high-flow changes over time were similar to overall trends, reflecting the strong influence these conditions have on nitrogen delivery to streams (Figure 8). Previous research in Smith Creek found that most NO_x load was delivered by groundwater (Miller et al., 2016). High streamflow conditions increase the amount of nitrogen delivered by groundwater and the amount of nitrogen in runoff. Therefore, MPs that lower nitrogen in both groundwater and runoff may be important for reducing loads in Smith Creek. TN and NO_x loads delivered during low flows were unchanged for most of the record but increased from 2004 through 2020. Increases in low-flow nitrogen load may indicate that groundwater nitrogen concentrations have increased in Smith Creek since 2004.

TN concentrations measured from four springs in the carbonate headwaters of Smith Creek provide additional insights about groundwater nitrogen concentrations in the watershed (Figure 1a). Between 2012 and 2020, TN concentrations did not decrease at these monitoring

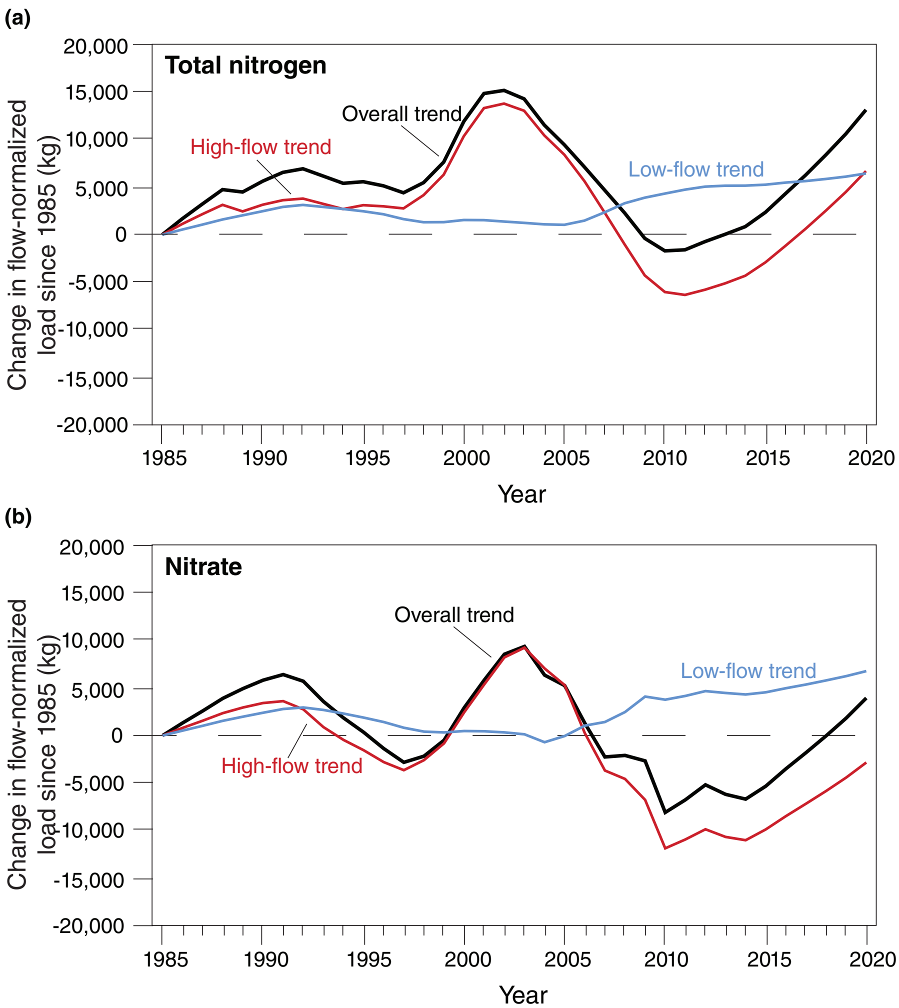

FIGURE 8 Annual changes in FN (a) TN and (b) NO_x loads during high- and low-streamflow conditions since 1985 at the Smith Creek streamgage.

locations (Figure S13). In 2012, TN concentrations sampled from Big Spring were lower than the other springs but increased over time. Sampled TN concentrations in the War Branch, Smith Creek, and Lacey Springs were two to three times higher than Big Spring and either increased or remained stable. These patterns suggest that groundwater nitrogen concentrations in Smith Creek did not decrease between 2012 and 2020. MPs can achieve nutrient reductions by lowering the amount of nitrogen delivered to and stored in groundwater, but such changes were not apparent in Smith Creek. Groundwater lag times are often cited as a factor that complicates the ability to identify MP effects (Ator et al., 2020; Lintern et al., 2020). In this study, however, the 8-year monitoring record may have exceeded the average age of groundwater discharged from the Smith Creek springs. Environmental tracers measured in May 2014 indicated that the streamflow from these springs likely includes a combination of days-old water that passes quickly through epikarst layers to water stored for up to 10 years. These estimates are similar to groundwater ages previously reported throughout Virginia's Shenandoah Valley (Yager et al., 2013). Some of these springs may respond to MP effects fairly quickly. For example, the water chemistry of Lacey Spring may be closely related to landscape activities (Doctor et al., 2014) and turbid water can be discharged from Lacey Spring during storm events. Therefore, additional MPs that limit nitrogen delivery to the carbonate headwaters of Smith Creek may be needed to reduce groundwater nitrogen concentrations in the watershed.

5.2 | Smith Creek TN drivers

A time-series regression model was used to identify factors that may have affected trends in FN TN load at the Smith Creek streamgage from 1991 through 2016. Using a best-subsets model-selection procedure, models with the lowest Akaike's information criterion (AICc; Sugiura, 1978) all included a parameter representing nitrogen from pasture-applied manure, lagged by 4 years (Figure S14). The pasture-applied manure term represents nutrient inputs from farm-animal manure that are applied to pasture lands. This term does not include manure directly deposited onto pasture. The input of nitrogen from pasture-applied manure was correlated with total inputs of nitrogen (Table S3), but models that replaced the pasture-applied manure term with total inputs of nitrogen had worse performance. Models with the lowest AICc also included two autoregressive terms, which produced a white-noise model error term (Figure S15). A three-parameter model with pasture-applied manure and two autoregressive terms had one of the lowest AICc values. Models with additional parameters did not offer meaningful AICc improvements, so the three-parameter model was selected for interpretation.

A model of 4-year lagged nitrogen input from pasture-applied manure (dMAN.PAS.lag4) and first- and second-order autoregressive terms (AR1 and AR2) explained 83% of the variability in annual FN load changes (Table 6). Annual FN loads predicted from this model aligned well with observed values (Figure 9). FN TN loads followed a similar trajectory as pasture-applied manure from 1985 through 2020, with changes in load appearing to lag the input of manure by about 4 years. Although manure runoff to streams can immediately affect river loads, the lagged manure term may reflect the importance of groundwater nitrogen delivery. This model likely underrepresents the fraction of nitrogen delivered to Smith Creek through quick flowpaths. Karst systems can quickly intercept, store, and transport NO_x to streams hours to days after watershed applications (Husic et al., 2020). These quick flowpaths may be important in the upper watershed, but less influential at the Smith Creek streamgage. The modeled manure coefficient (0.04) means that, on average, an increase of 100 kg of nitrogen from pasture-applied manure was estimated to result in, after 4 years, about a 4 kg increase in FN TN load at the Smith Creek streamgage. Although this analysis is specific to Smith Creek, this estimated rate of manure delivery is similar to other empirical models that considered the capacity of nitrogen inputs to export FN TN loads. Ator et al. (2011) found that about 6% of manure nitrogen inputs reach Chesapeake Bay streams. Chanat and Yang (2018) reported that, from 1990 through 2010, an increase in manure nitrogen inputs resulted in about a 7% increase in FN TN load throughout the Chesapeake Bay watershed.

Manure is the largest nitrogen input in Smith Creek (Figure 4) and has previously been described as an important nitrogen source in the watershed and throughout the Shenandoah Valley (Hyer et al., 2016; Schaeffer et al., 2017). Hyer et al. (2016) reported that the NO_x isotope signature of water samples collected at the Smith Creek streamgage from 2011 through 2013 was likely indicative of manure sources. Nitrate isotopes measured from 2014 through 2018 continue to show that manure is the dominant source of nitrogen in Smith Creek (Figure S16). Pastureland (areas of low vegetation that include hay production fields) is five times more abundant than cropland in Smith Creek and receives most of the manure applied in the watershed. Pasture manure applications may exceed vegetative nutrient requirements, as estimated for Smith Creek cropland (Figure S6). Similar patterns have been observed in other areas where concentrated livestock and poultry operations

TABLE 6 Regression model results for predicting annual changes in FN TN load at the Smith Creek streamgage from 1991 through 2016.

Term	Estimate	Standard error	*p*-Value
dMAN.PAS.lag4	0.04	0.01	0.0005
AR1	1.26	0.15	<0.0001
AR2	−0.59	0.15	<0.0001
Model R^2 = 0.83			

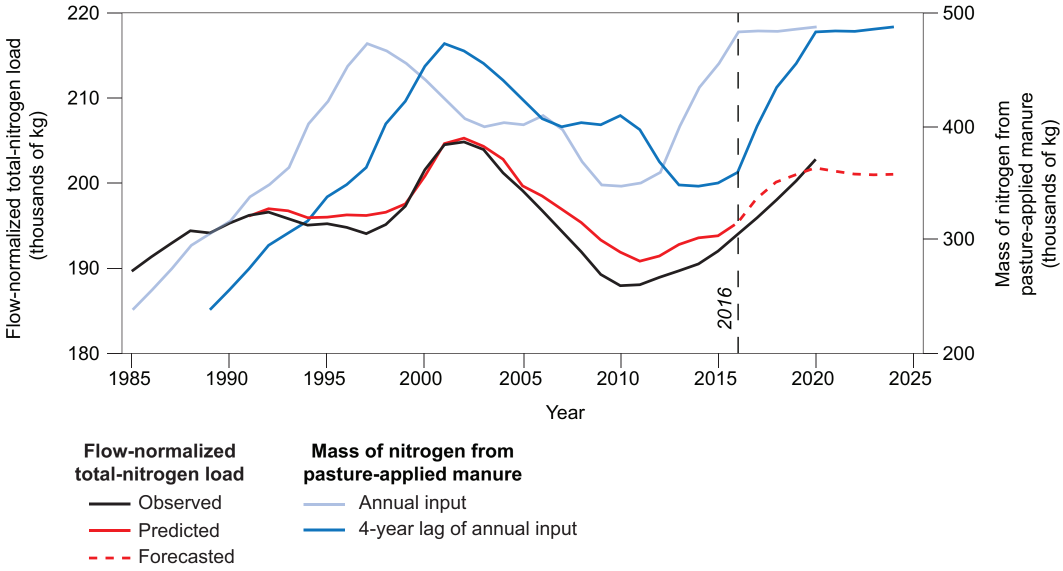

FIGURE 9 Plot of observed FN TN loads (Mason et al., 2023) and predicted FN TN loads at the Smith Creek streamgage and the input of nitrogen from pasture-applied manure (Devereux, 2022a), from 1985 through 2020.

have resulted in large manure nutrient surpluses (Bryant et al., 2022; Sabo et al., 2021). The fate of nitrogen in landscape applied manure is complicated and can be affected by management actions related to the application, storage, and treatment of manure (Chambers et al., 2000; Clune et al., 2021; Szogi et al., 2015). In general, nutrients that are not assimilated by vegetation, removed by agricultural products, or transformed to atmospheric nitrogen can eventually leach to groundwater and streams. Most manure produced in Smith Creek is likely applied in the watershed; the broader Shenandoah Valley produces a large quantity of manure and economic constraints can prevent long-haul manure transport (Bryant et al., 2022; Kleinman et al., 2012). Given these challenges, effective nitrogen management in the Smith Creek watershed may need to focus on limiting the amount of manure generated in the watershed and minimizing the delivery of manure to streams.

Although MPs that store animal waste and exclude livestock from streams are used throughout the Smith Creek watershed, a predictor variable representing expected MP load reduction was not selected by any top-performing Smith Creek regression models. As modeled by the CBP-WSM, MPs were expected to reduce TN loads by up to 20% per year from 1985 through 2020 (Figure 3a); however, observed FN loads increased by 7% over these same years (Figure 8a). The relation between observed FN loads and modeled MP load reductions was nonlinear from 1985 through 2020 (Figure 10). In Figure 10, estimated MP load reductions from 1999 through 2020 were adjusted by 36,600 kg to offset a large step change associated with poultry waste management systems reported in Shenandoah and Rockingham Counties. A similar step change was not observed in FN TN loads, possibly because these MPs were not installed in the Smith Creek watershed. There was a weak correlation between observed FN TN loads and modeled MP load reductions from 1985 through 1998. After these years, however, observed FN TN loads were lower in years with larger modeled MP load reductions. This relation suggests that MPs may explain some variability in observed FN loads in more recent years. The relation between observed FN loads and modeled MP load reductions may be stronger in more recent years because (1) MPs are being reported more accurately to the CBP, (2) MPs are being used that more effectively reduce TN loads, and (or) (3) there is a spurious correlation between MPs and other factors affecting TN load.

6 | IMPLICATIONS FOR CONSIDERATION BY RESOURCE MANAGERS

The results of this work highlight important considerations for managing agricultural nutrient and sediment loads and for evaluating the water-quality effects of MPs:

1. Nutrient load reductions may not occur until manure and fertilizer inputs are lowered to align with local crop nutritional requirements, changes that would reduce surplus nutrient inputs and could prevent the accumulation of legacy nutrients stored in the environment. Most MPs are designed to control the movement of nutrients from the landscape to streams, not to reduce the amount of nutrients applied to the landscape. However, the use of nutrient management plans and technologies that support precision agriculture is increasing throughout the Chesapeake Bay watershed (Clune et al., 2021). MPs and technologies like these may reduce nutrient delivery to streams by aligning nutrient inputs with crop requirements and adjusting the timing and method of nutrient applications.

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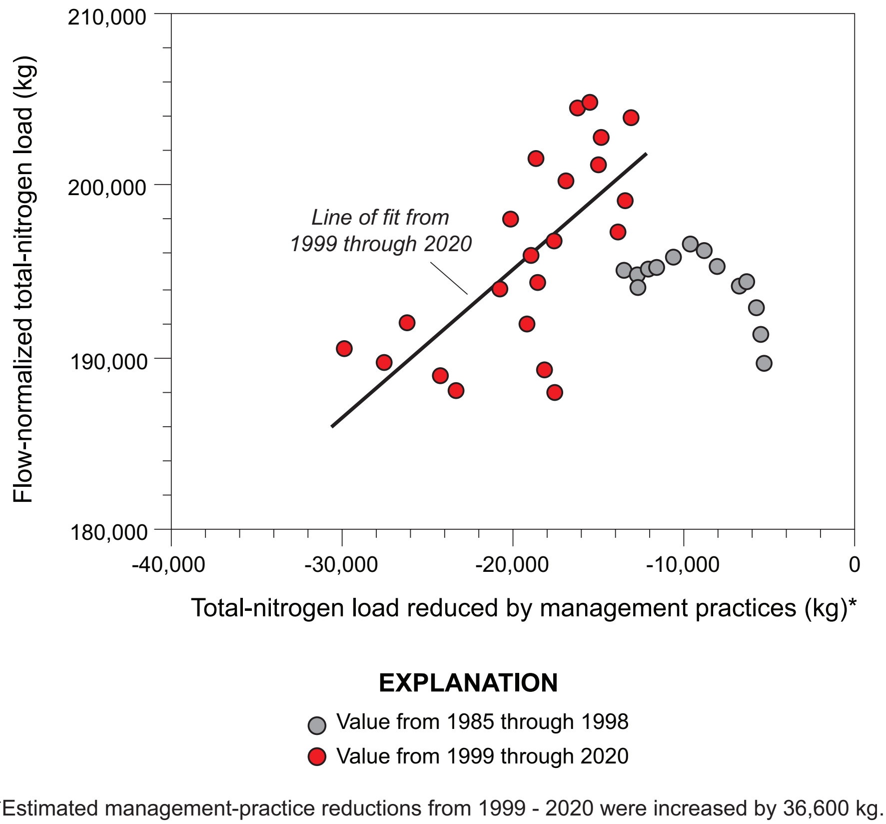

FIGURE 10 Graph showing the relation of FN TN load to estimated TN loads reduced by MPs from 1985 through 2020 in Smith Creek.

2. MPs that control the delivery of nutrients and sediment during periods of high streamflow may help reduce loads. Streamside vegetative buffers and MPs designed to store water on the landscape can help reduce stormflow runoff and associated nutrient and sediment loads; however, the performance of these and other MPs can diminish with increasing storm size (Gall et al., 2018; Liu et al., 2017; Raisin et al., 1997; Xie et al., 2015). A large percentage of loads can be delivered during periods of high streamflow through surface water runoff and, for nutrients, groundwater pathways. Compared to longer groundwater residence times, surface water runoff is often hours, days, or months old. Therefore, controlling nutrient loads delivered with surface water runoff may help shorten the potential lag between MP implementation and water-quality response.
3. MPs that reduce sediment loads may help reduce nutrient loads. Particulate phosphorus can be the largest fraction of TP load, can affect trends in TP load, and is often associated with sediment delivery to streams. NO_x is often the largest fraction of TN load; however, reduced sediment loads may help lower transport of non NO_x forms of nitrogen that contribute to TN loads. In the Chesapeake watershed, 73% of TP loads and 18% of TN loads transported to the Bay is typically attached to sediment (Noe et al., 2020). MPs that reduce streambank and upland soil erosion may be important for reducing sediment loads in agricultural watersheds (Noe et al., 2020).
4. Point-source discharges can affect nutrient loads in agricultural watersheds, even when point-source nutrient inputs are much smaller than non-point source nutrient inputs from fertilizer and manure. Agricultural watersheds (especially those with animal feeding operations) can contain many permitted facilities that discharge nutrients to streams. In such watersheds, nutrient concentrations and loads can decrease in response to reduced point-source inputs.
5. Sustained water-quality monitoring, advancements in statistical tools, and collaborative partnerships would increase understanding of how agricultural nutrient and sediment loads respond to MPs. Monitoring data provide an accurate assessment of water-quality conditions, but it is a challenge to directly associate monitored responses with MP effects. This challenge could be reduced through long-term water-quality monitoring studies in small agricultural watersheds (zero- to third-order streams) with targeted MP implementation. Long-term monitoring studies may be particularly important for evaluating MP effects because multiple years of data are needed to quantify water-quality trends and to statistically associate water-quality changes with drivers. Investments to advance empirically based statistical tools could better identify and explain the water-quality effects of various natural and anthropogenic factors. Additionally, an exchange of data and expertise among a multiagency partnership of researchers, watershed managers, and local stakeholders would help develop a comprehensive understanding of landscape activities, MPs, and water-quality responses.

Although this research focused on study areas in the Chesapeake Bay watershed, these implications may help inform other watersheds that rely on voluntary conservation efforts to reduce agricultural nonpoint source loads.

7 | CONCLUSIONS

In three agricultural Chesapeake Bay watersheds that were targeted for MP implementation, we found that most nutrient and sediment loads did not decrease since the early 2010s and that trends in load were likely affected by a combination of natural and anthropogenic factors. The number of agricultural MPs increased in each watershed since the early 2010s; however, expected MP load reductions were not consistently observed in monitored responses. Some water-quality benefits may have been related to MPs. Sediment-reducing MPs may have helped decrease SS loads in the Conewago Creek watershed. The management of point sources may have helped decrease nutrient loads in one Conewago Creek monitoring station. Cropland MPs may have helped stabilize TN loads in Chesterville Branch following decades of intensifying agricultural activities and increased nitrogen concentrations. From 1999 through 2020, modeled MP reductions of TN load in the Smith Creek watershed were consistent with changes in monitored TN load. In all watersheds, however, trends in nutrient and sediment load were similar to trends in some surrounding agricultural watersheds that were not specifically targeted for MP implementation.

The amount of nutrients applied to the landscape and corresponding changes in surplus nutrient inputs may be a particularly important factor affecting agricultural water-quality responses. Since 1985, the study watersheds received a surplus amount of nitrogen and phosphorus inputs to cropland, and agricultural nutrient inputs typically increased through 2020. In Smith Creek, increases in manure applications likely affected changes in FN loads of TN since the early 1990s. Although most nonpoint source inputs do not reach streams, changes in nutrient inputs since the early 2010s were almost always much larger than modeled changes in nutrient load attributed to MPs. These patterns suggest that increases in nutrient inputs might overshadow expected MP load reductions. This work can help inform management of agricultural nutrient and sediment loads in the Chesapeake Bay watershed and in other watersheds that rely on voluntary conservation efforts to agricultural reduce nonpoint source loads. Collectively, these results highlight the complexity of evaluating watershed responses to MPs and the importance of assessing MP performance with long-term monitoring-based studies.

AUTHOR CONTRIBUTIONS

James Webber: Conceptualization; data curation; formal analysis; methodology; visualization; writing – original draft; writing – review and editing. **Jeffrey Chanat:** Conceptualization; formal analysis; methodology; visualization; writing – review and editing. **John Clune:** Conceptualization; methodology; visualization; writing – review and editing. **Olivia Devereux:** Conceptualization; data curation; methodology; visualization; writing – review and editing. **Natalie Hall:** Conceptualization; methodology; visualization; writing – review and editing. **Robert D. Sabo:** Methodology; writing – review and editing. **Qian Zhang:** Methodology; writing – review and editing.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflict of interest related to the preparation or publication of this manuscript.

DATA AVAILABILITY STATEMENT

Water-quality and streamflow data are available at https://doi.org/10.5066/F7P55KJN. Calculated nutrient and sediment loads are available at https://doi.org/10.5066/P96H2BDO. Inputs to the Chesapeake Bay Program's watershed model are available at https://doi.org/10.5066/P93SVYQG. Estimated nutrient and sediment management-practice load reductions are available at https://doi.org/10.5066/P95WG7G0. Expected physical effects of agricultural conservation practices are available at https://doi.org/10.5066/P9VY95KT.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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