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The Quality of Our Nation's Waters

Nitrogen in the Chesapeake Bay Watershed
A Century of Change 1950-2050

National Water-Quality Program
National Water-Quality Assessment Project

Circular 1486

U.S. Department of the Interior
U.S. Geological Survey

Cover: Wetlands near the Elk River in Cecil County, Maryland.
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Nitrogen in the Chesapeake Bay Watershed
A Century of Change 1950-2050

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Agriculture in Lancaster County, Pennsylvania. Courtesy of the Chesapeake Bay Program, used with permission.

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Agriculture in Lancaster County, Pennsylvania. Courtesy of the Chesapeake Bay Program, used with permission.

Foreword

Sustaining the quality of the Nation's water resources and the health of our diverse ecosystems depends on the availability of sound water-resources data and information to develop effective, science-based policies. Effective management of water resources also brings more certainty and efficiency to important economic sectors. Taken together, these actions lead to immediate and long-term economic, social, and environmental benefits that will make a difference to the lives of the almost 400 million people projected to live in the United States by 2050.

The Chesapeake Bay is the largest and most productive estuary in the United States and is a vital environmental and economic resource. Approximately half of the water volume of the Chesapeake Bay originates from streams and rivers that drain the 64,243 mi^2 Chesapeake Bay watershed. The Bay and its tributaries have been degraded by excessive nutrients, such as nitrogen, from contributing watersheds. Inputs of nitrogen to the Bay lead to increased algal growth, decreased dissolved oxygen, and declining fisheries. In 2000, the Chesapeake Bay was listed as impaired under the Clean Water Act and Total Maximum Daily Loads (TMDLs) for nutrients and sediment have been established to assist with management actions aimed at nutrient reductions. Effective nutrient management requires an understanding of past, present, and future nutrient sources, fate, and transport in the watershed.

The Chesapeake Bay community has been a pioneer in science, management, and regulation to improve water quality. Factors like climate, hydrology, source inputs, and management controls play a vital role in determining the delivery and magnitude of nitrogen inputs to the Bay. Science in the form of monitoring data, predictive tools, and interpretive reports can help inform decisions to better balance the use and control of nitrogen in coastal areas. The findings in this report can contribute to effective management of the Bay and its watershed by providing a synthesis of the understanding of how human activities and environmental change in the watershed in the past, present, and future will influence the export of nitrogen to the Bay.

We hope this publication will provide you with insights and information to meet your water-resource needs and will foster increased civilian awareness and involvement in the protection and restoration of our Nation's waters. The information in this report is intended primarily for those interested or involved in resource management and protection, conservation, regulation, and policymaking at the regional and national levels.

Dr. Donald W. Cline
Associate Director for Water
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Breton Bay in St. Mary's County, Maryland. Courtesy of the Chesapeake Bay Program with aerial support by LightHawk, used with permission.

Abbreviations

BMP	best management practice
CAST	Chesapeake Assessment Scenario Tool
CMAQ	Community Multiscale Air Quality Model
EPA	U.S. Environmental Protection Agency
ICPRB	Interstate Commission on the Potomac River Basin
MGD	million gallons per day
NADP	National Atmospheric Deposition Program
NPDES	National Pollutant Discharge Elimination System
RCP	representative concentration pathway
SAV	submerged aquatic vegetation
SPARROW	Spatially Referenced Regression on Watershed Attributes
TMDL	total maximum daily load
WWTP	wastewater treatment plant

Agriculture in Lancaster County, Pennsylvania. Courtesy of the Chesapeake Bay Program, used with permission.

Overview of Major Findings

By Paul D. Capel[1] and John W. Clune[1]

This section presents an overview of the major findings of this circular. The succeeding sections provide background on the environmental, nitrogen, and historical setting of the Chesapeake Bay watershed as a precursor for the main chapters and final thoughts.

> ***"The Chesapeake Bay watershed includes 166,000 square kilometers of mixed land uses, multiple nutrient sources, and variable hydrogeologic, soil, and weather conditions, and Bay restoration is complicated by the multitude of nutrient sources and complex interacting factors affecting the occurrence, fate, and transport of nitrogen and phosphorus from source areas to streams and the estuary." (Ator and others, 2011)***

Nitrogen, a critical element in all forms of life, is continuously being passed from nonliving to living matter and then back again, but an excess of this nutrient can have adverse effects on aquatic environments. An understanding of the past, present, and future sources, movement, and fate of nitrogen in the Chesapeake Bay watershed can help inform efforts to bring this cycle back into balance (fig. OV.1). A substantial shift in the nitrogen balance has occurred since the 1940s, when industrially produced nitrogen fertilizer became widely available. Fertilizer application to crops increased substantially in succeeding decades in order to meet the demand for food by a growing population. During this period, the loading of excess nitrogen took a toll on aquatic ecosystems and the related economies that are dependent on the Chesapeake Bay (also referred to as the Bay). Increased algal productivity from nutrient enrichment, both nitrogen and phosphorus, has resulted in the expansion of low oxygen (hypoxia) conditions, also known as "dead zones," in the Bay. Substantial legislative, management, and grass roots initiatives have been developed over recent decades to combat the increasing impacts of excessive nutrients in the Chesapeake Bay watershed. In 2010, regulatory pollution limits were developed for the Chesapeake Bay creating the largest and most complex total maximum daily loads in the Nation for nitrogen, phosphorus, and sediment. Over this same time period, the monitoring and modeling of water quality has expanded to provide a better understanding of the fate and transport of nitrogen in the watershed. Our ability to improve the nitrogen balance in the Chesapeake Bay is dependent on our understanding of how a changing climate, population, land use, and decisions at all levels of government impact the health of the Bay. The significance and magnitude of the management and scientific efforts to restore the Chesapeake Bay have provided a template for restoration and serve as a model for the Nation and the world.

Photograph of Wicomico River in St. Mary's County, Maryland. Courtesy of the Chesapeake Bay Program, with aerial support by LightHawk, used with permission.

[1]U.S. Geological Survey.

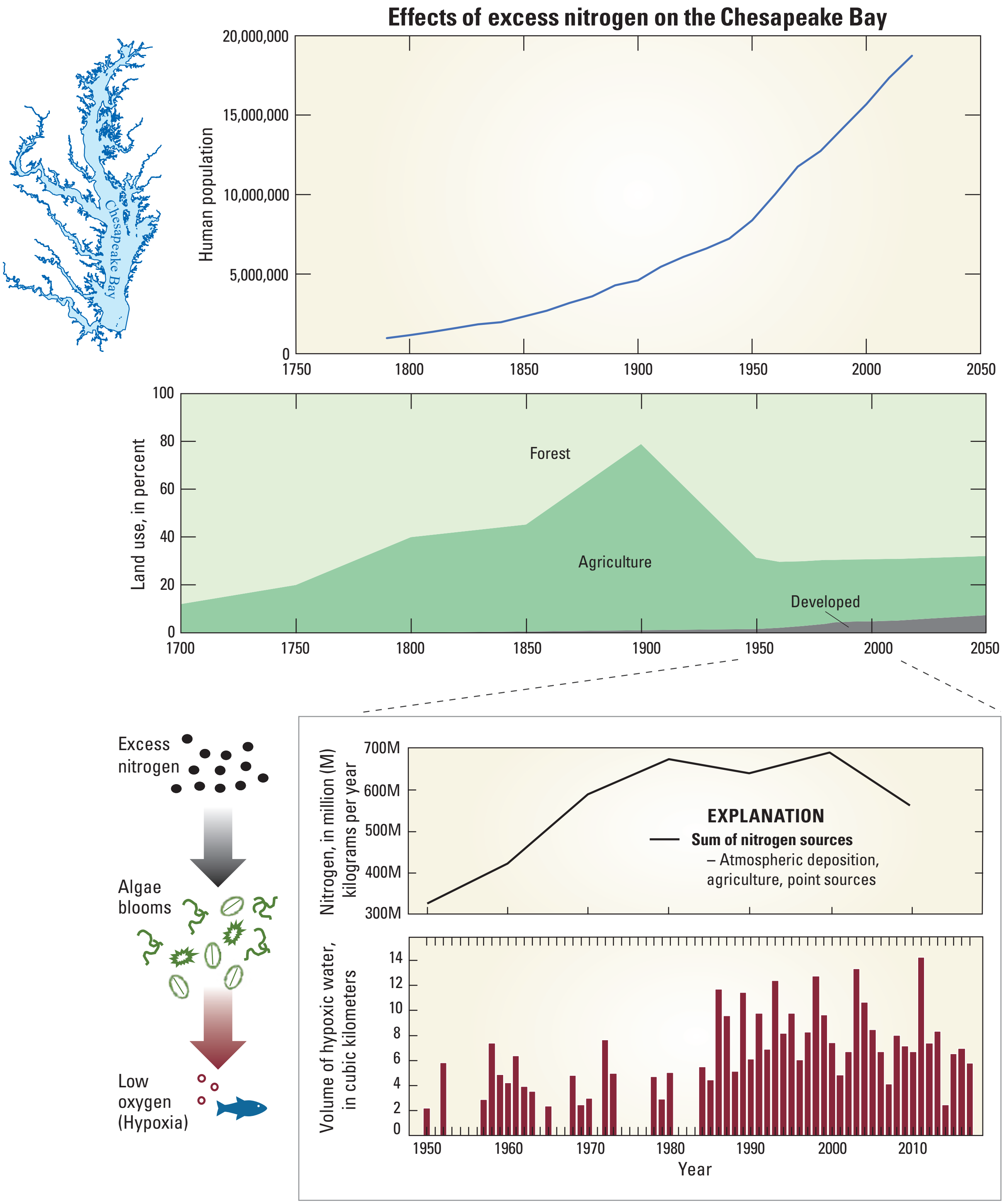


Figure OV.1. As human population has increased, land use has changed from primarily undeveloped (forested) to agricultural and developed land (Hopple and others, 2021), these changes have led to increases in nitrogen inputs that result in water quality impairments such as low oxygen or hypoxic water (dissolved oxygen < 2 milligrams per liter) in the Chesapeake Bay (Hagy and others, 2004; Murphy and others, 2011; Schulte, 2017; Testa and others, 2018).

The goal of this report is to provide a century-long understanding of how the long-term human activities and environmental changes in the watershed have influenced the export of nitrogen to the Bay and to make forecasts of future effects. Previous studies have improved our understanding of nitrogen sources and processes, but this report provides a unique synthesis of the story of nitrogen since early European settlement, with a particular focus on the past and future changes in the nitrogen cycle in the Chesapeake Bay watershed for the 100-year time period from 1950 to 2050. Climate and hydrology play a vital role in determining the delivery mechanisms and magnitude of nitrogen export from the watershed to the Bay (see chaps. 3 and 4). Nitrogen is introduced to the landscape each year through atmospheric deposition and human land use (see chaps. 5 and 6). Management controls are used to mitigate the amount of nitrogen exported from agricultural and urban areas (see chaps. 7 and 8). Summaries and analysis of past monitoring data provide perspective on relatively recent historical trends and are used as the foundation for past predictions (hindcasts) back to the year 1950 and future projections (forecasts) to 2050 to describe a century of change in the Chesapeake Bay watershed. This report presents possible future scenarios of sources and export of nitrogen to the watershed and Bay that provides a long-term perspective to help inform decisions to better balance the use, release, and management of nitrogen in the Chesapeake Bay and other coastal areas (see chaps. 9 and 10).

Changes Impacting Nitrogen Export in Chesapeake Bay and its Watershed

Before the arrival of Europeans, the Chesapeake Bay and its watershed was a diverse and resilient ecosystem that could adapt to ecological disturbances both natural and human. Much of the watershed was covered by a vast wilderness with an abundant array of plant and animal species, and many areas were managed by a sparse population of Native Americans who hunted and cultivated the landscape. Early colonists in the 1600s adopted low-impact indigenous farming methods but, by the 18th century, the first major human-induced changes in nutrient water quality became apparent with the introduction of more intensive (plow) agriculture and the precipitous extraction of beaver, lumber, and coal (Miller, 1986). As an agrarian society developed, forests and wetlands that once provided buffering capacity against ecological disturbances were lost, and the cultivation of the land introduced more nitrogen to the ecosystem (Galloway and others, 2004). As the Nation developed further in the 19th and early 20th centuries, changes in land use and a rising population contributed to an accelerated loss of soil and nutrients (phosphorus and nitrogen) into waterways from agricultural erosion and untreated sewage runoff. The newly developed capacity to synthetically create inorganic fertilizer produced a substantial increase in nitrogen inputs, while the combustion of fossil fuels (primarily coal) further introduced new sources of nitrogen into the environment (Haber, 1920; Erisman and others, 2008). The post-World War II era brought extensive suburban development that increased sewage discharges and initiated fertilizer use on lawns. Overall, inputs of sediment, nutrients, and toxins increased several-fold during the 18th through 20th centuries compared to pre-European settlement and have taken a toll on ecological and economic aspects of the Chesapeake Bay and surrounding watershed. By the mid- to late-20th century, substantial degradation of water quality compelled bold measures, including establishing the Chesapeake Bay Program partnership in 1983, which has the goal to reduce nutrients to the Bay. Regulation, management, and monitoring of nitrogen have shown some water-quality improvement, but not enough change to reach the overall water-quality goals for the Chesapeake Bay. The slow voluntary progress to reduce nutrients and the continued poor water-quality in the Bay led to the development of a Chesapeake Bay total maximum daily load (TMDL) in 2010. The Bay TMDL requires states to have practices in place by 2025 to reduce nutrients and improve water-quality conditions in the Bay. Additional outcomes include attainment of water-quality standards that are important for fisheries in the Bay, such as dissolved oxygen and water clarity, although there is no deadline for when the standards must be achieved.

> ***"There is but one entrance by sea into this country, and that is at the mouth of a very goodly bay … all along the shores rest plenty of pines and firs ... Within is a country that may have the prerogative over the most pleasant places known, for large and pleasant navigable rivers, heaven and earth never agreed better to frame a place for man's habitation." —John Smith***

Management Capacity in Controlling Change and the Movement of Nitrogen

Natural and human activities directly and indirectly affect the export of nitrogen from the watershed to the Chesapeake Bay. Those responsible for management decisions at the regional, state, and Federal levels have varying degrees of influence on the changes that control the movement of nitrogen to the Bay (fig. OV.2). Options for controlling nitrogen exports to the Bay include long-term planning, implementation of technologies, economic incentives, and informing societal decisions. Similarly, regulatory and voluntary actions are important components of a successful nutrient-reduction strategy.

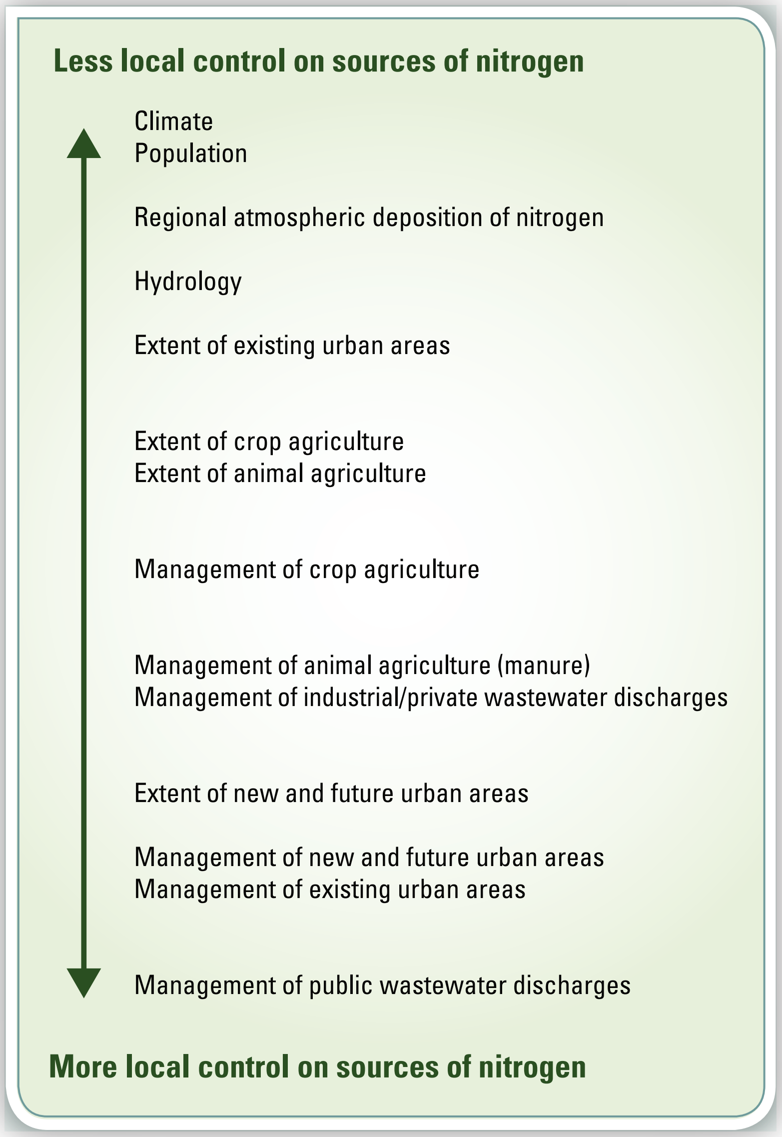


Figure OV.2. Local water quality managers have varying degrees of control on the movement of nitrogen to the Chesapeake Bay and its watershed.

Factors that are Less Manageable at the Regional Chesapeake Bay Watershed Level (Climate, Hydrology, Atmospheric Deposition, Population)

The climate of the Chesapeake Bay watershed is controlled at the global scale and over the past few decades has been experiencing a warming trend like much of the planet. Annual mean air temperature in the Bay watershed is forecasted to rise by 2.0 °C and precipitation is generally predicted to increase in most areas of the watershed on average by 6.3 percent (2050 compared to 1995) (Bureau of Reclamation, 2013; Chesapeake Bay Program, 2017). Much of the increase in precipitation is forecasted to be in the form of more extreme storm events. Generally, hydrologic shifts to drier summer and fall seasons followed by wetter winter and spring seasons are expected.

The storage, transformation, and transport of nitrogen within the watershed and to the Bay is largely controlled by the movement of water through hydrologic compartments including precipitation, evapotranspiration, runoff, groundwater, lakes, and streams. The rain that falls on the Chesapeake Bay watershed can evaporate to the atmosphere, run off the land to streams, or infiltrate into the ground and eventually be discharged to local streams and coastal areas. Nitrogen movement is influenced by surface water retention and release, but also by groundwater storage that delays the transport of nitrogen. Each dynamic hydrologic compartment has different characteristics, such as volume and residence time, that may be impacted by human development and changes in climate. Changes in the landscape prior to 1950 (for example, deforestation, dam creation, early urbanization) had a significant impact on the hydrologic cycle and the load of nitrogen and these changes have accelerated as the population in the watershed has more than doubled from 1950 to 2017 (Hopple and others, 2021). Future predictions of climate and land-use changes indicate that runoff will generally increase throughout the watershed, but with varying spatial patterns.

The deposition of nitrogen from the atmosphere to the watershed and the Bay is another example of nitrogen movement not primarily governed by state or regional controls, but rather by clean air regulations enacted at the national scale. Emissions from industrial, transportation, and agricultural activities from as far as North Carolina in the south to New England in the north, and to the Ohio River valley and beyond in the west, contribute about three-quarters of the nitrogen deposited from the atmosphere to the Bay watershed. The remaining quarter of the nitrogen load is from the rest of the North American continent and the world (U.S. Environmental Protection Agency, 2010b; Linker and others, 2013b). Early studies started in the 1980s raised awareness of the role of atmospheric nitrogen deposition as a significant contributor to eutrophication in the Bay (Tyler, 1988; Fisher and Oppenheimer, 1991).

Implementation of the Clean Air Act has resulted in gradual decreases in atmospheric nitrogen deposition beginning in the 1970s and even sharper decreases since the 1990s. Atmospheric deposition as a nitrogen source was included as part of the TMDL load reduction goals in 2010, despite the challenge that the source emissions airshed extends outside the watershed boundary. Oxidized forms of atmospheric nitrogen deposition (NOx or nitrate deposition) have decreased by more than 50 percent since the 1990s, whereas reduced nitrogen (NH_4+ or ammonia) deposition has shown little change over the same period. Modeling results indicate that decreases in atmospheric nitrogen deposition have been the second most important contributor to decreases in nitrogen loads to the Bay during the 1992 to 2012 period. Future projections based on current air quality regulations indicate that further modest decreases in atmospheric nitrogen deposition will continue and then reach a plateau in about 2030. However, the emissions and deposition of ammonia, a reduced form of nitrogen, has slightly increased since 1990, primarily around areas of dense animal agriculture. The Chesapeake Bay provides an example of how joint regulatory implementation of the Clean Air and Clean Water Acts was mostly effective in addressing estuarine eutrophication, with the exception of ammonia emissions, which remain uncontrolled.

From 1950 to 2017 the human population in the Chesapeake Bay watershed increased by about 117 percent (9.8 million), from 8.4 to 18.2 million (fig. OV.1). The increased population led to more houses, roads, towns, wastewater, fossil fuel consumption, agricultural intensity, and resource extraction. Compounding these development pressures, the growing populated areas have tended to be close to the Bay in urban centers such as Baltimore, Maryland; Richmond, Virginia; and Washington, D.C. The upward population trajectory is closely linked to nitrogen inputs and has altered the hydrologic cycle, water quality, and transport of nutrients, all contributing to the decline in health of the Bay.

Factors that are More Manageable at the Local Level (Developed Areas and Agriculture)

The substantial continued efforts of management, regulation, and conservation in the Chesapeake Bay watershed have had varying degrees of success in controlling or managing nitrogen inputs and exports. Developed areas, although a minor fraction of the overall land use in the Chesapeake Bay watershed, are a unique and complex component of the landscape. Historically, developed areas contributed a large fraction of nitrogen to Chesapeake Bay through both point sources (wastewater) and nonpoint sources (atmospheric deposition, residential lawn fertilizer, and septic systems). National and regional regulations have changed the way urban nitrogen is managed, including regulations on wastewater and on stormwater management. Consequently, the contributions of nitrogen from point sources to Chesapeake Bay have declined. Improvements in wastewater treatment, in particular, have been identified as a major driver of declining nitrogen export in major tributaries to the Bay. As the population increased by 35 percent (4.7 million persons) from 1985 to 2017, nitrogen discharges from wastewater treatment plants (WWTPs) were reduced by 50 percent (–16.8 million kilograms) mainly from technological improvements and regulatory controls (Hopple and others, 2021). Point sources, managed through regulations, will likely continue to play a major role in controlling nitrogen loads in urban streams and rivers. Management of urban nonpoint sources of nitrogen has been more of a recent focus (since 1987) and will likely continue as urban centers continue to expand across the watershed in the coming decades, but the collective effectiveness of urban best management practices (BMPs) represents a sufficient knowledge gap where further research is warranted. Although implementation of urban management practices to reduce nonpoint sources of nitrogen will continue with new development, the potential nutrient reductions of those practices in the future are uncertain.

Agriculture is an important part of the economy and heritage of the Chesapeake Bay region, but controlling legacy and current nonpoint nutrient issues is a major challenge for resource managers. Agriculture tends to be concentrated on the Coastal Plain and in areas with fertile soils, such as the Eastern Shore of the Chesapeake Bay and southeastern Pennsylvania. Unfortunately, many of these regions also tend to have geologic settings (for example, sand/gravel, carbonate aquifers) that facilitate nutrient transport to groundwater and eventually streams. Nitrogen and phosphorus inputs from agriculture are the largest source of nutrient inputs to the landscape in the Chesapeake Bay watershed (Boesch and others, 2001; U.S. Environmental Protection Agency, 2010a) and eventually exported to Chesapeake Bay (Ator and others, 2011). Since the first nutrient pollution reduction goals in the 1987 Chesapeake Bay Agreement, significant investments have been made to both restore and preserve water quality in the Bay, making the region a national leader in adoption of conservation practices. Across the entire Chesapeake Bay watershed, nitrogen loads were estimated to be reduced by 11 percent (22 million kilograms or 49 million pounds) from 1985 to 2014 owing to the implementation of conservation practices (Sekellick and others, 2019) relating mostly to land retirement, animal waste management systems, and conservation tillage, but also bioretention by ponds and wetlands. The delayed travel time of water and nitrogen moving through streams and groundwater following decades of past conservation practices may have slowed the full water-quality response to management efforts as measured nitrogen loads and yields from most agricultural areas have not changed substantially in the Bay watershed.

A Past and Future Perspective for Predicting Nitrogen Export to the Bay

Monitoring data provide critical quantitative information on past and present water-quality conditions and can be used to help predict future scenarios. These data can be used to track changes in water quality in response to past and present natural and human-induced change, infer the effectiveness of regulatory and management changes, and inform the development of new strategies and policies. Sustained monitoring provides insights into long-term trends and can provide understanding that may be obscured in short-term monitoring efforts because of changing weather, hydrology, and annually variable inputs of nitrogen to the watershed. Ongoing monitoring provides the best assurance for early recognition of trends, which is critical for implementing changes to prevent continued deterioration of water resources. Monitoring data are also the foundation for the development of statistical and process-based mathematical models that predict and improve understanding of the complex interactions within the nitrogen and hydrologic cycles. Models can estimate the magnitude of different nitrogen sources and make predictions of how these sources may change in the future under different assumptions of human activities, land use change, and climate. Additionally, models can be used to identify which types of data are most important to collect and where monitoring gaps may exist. High quality monitoring data for nitrogen in streams, groundwater, and precipitation in the Chesapeake Bay watershed are available beginning in the mid-1980s. Before this period, hindcast modeling is needed to estimate past changes in water quality in the hydrologic system.

There are multiple sources of nitrogen and different processes that influence the delivery of nitrogen to streams and receiving waters such as the Chesapeake Bay. Watershed models, such as the Spatially Referenced Regression on Watershed Attributes (SPARROW) model, are commonly developed and applied to estimate how much nitrogen, originating from each source and from all sources combined, is delivered to streams and the Bay. This information is useful for developing strategies to mitigate the impacts of nitrogen on aquatic ecosystems. For example, a recent SPARROW model (Ator and others, 2011; Sekellick and others, 2021) shows crop fertilizer as the dominant source (45 percent of the total load) of nitrogen to the Chesapeake Bay and suggests that land-use change from agricultural land receiving fertilizer application to undeveloped or developed land has great potential for reducing nitrogen loading to streams in the Chesapeake Bay watershed. Additional model results indicate that implementation of agricultural and urban BMPs is effective for mitigating nitrogen loads (Chesapeake Bay Program, 2017). Effective management requires estimates of total nitrogen response to changing land use, and management practices and models provide an opportunity to make these assessments.

Future Scenarios of Nitrogen in the Chesapeake Bay Watershed

Some of the possible future scenarios affecting nitrogen in the waters of the Chesapeake Bay watershed are relatively fixed, but many others are dependent on management decisions. Population growth and urban development are certain to continue, but planning for urban development can include sustainable practices that decrease the export of nitrogen. Deposition of nitrogen from atmospheric emissions is controlled on regional, state, and national levels, whereas wastewater treatment and agriculture management are largely managed on a local scale by the people who live in the watershed. Although the future is unknown, examination of many possible scenarios can be informative for current decision-making and planning processes. As illustrative examples, the same trajectories of change for climate, population, land use, and atmospheric deposition of nitrogen have been combined with selected scenarios for future changes in agricultural activities and wastewater discharges (fig. OV.S1). These scenarios are meant to envelop the extremes of realistic changes that might occur in agriculture and wastewater discharges in the Chesapeake Bay watershed over the next few decades (see sidebar–Select Future Nitrogen Scenarios for the Chesapeake Bay Watershed).

Historical nitrogen input data (1950–2012) from various sources to the watershed were gathered or estimated using hindcast models (Hopple and others, 2020, 2021) and combined with nitrogen inputs estimated for possible future scenarios (fig. OV.S1). The relative inputs of the nitrogen sources have varied over time (fig. OV.3), often independently of each other. Nitrogen from atmospheric deposition increased

Select Future Nitrogen Scenarios for the Chesapeake Bay Watershed

There are countless environmental and human changes that will influence the future export of nitrogen to the Chesapeake Bay, with some changes more likely than others. For each future model scenario developed, the same trajectories of change were used for climate, population, land use, and atmospheric deposition of nitrogen; future projected changes in nitrogen from agricultural activities and wastewater discharges varied across scenarios (fig. OV.S1). These scenarios provide illustrative examples of how future policy and management decisions could affect the estimated export of nitrogen to the Bay.

This report uses climate predictions (2050 compared to 1995) that forecast warmer temperatures (rise of 2.0°C) and slight increases in precipitation (6.3 percent) and corresponding changes in streamflow and recharge (see chap. 3; Bureau of Reclamation, 2013; Chesapeake Bay Program, 2017). Atmospheric deposition of nitrogen to the Bay watershed is controlled by activities throughout an airshed that is much larger than the watershed itself and the regional atmospheric deposition of nitrogen estimates in the forecasting model for this report use predictions that suggest a decrease and plateau in the future (see chap. 5). By 2050, the population living in the Chesapeake Bay watershed is expected to be 22.5 million, an increase of 24 percent from 2017, and the urban growth model in this report assumes that future development will progress similarly to past development (see chap. 6).

In the past few decades, improved wastewater treatment using enhanced nitrogen removal technologies has substantially reduced the export of nitrogen to the Bay. Wastewater treatment technology was considered to improve or remain unchanged in the modeled scenarios (fig. OV.S1). Population growth was assumed to increase the amount of nitrogen delivered from wastewater treatment plants. One future scenario assumes no further investment in upgrading wastewater treatment, while the other assumes that enhanced nitrogen removal will be implemented at all remaining large wastewater treatment plants in the watershed.

Agriculture nitrogen sources in the modeled scenarios were considered to either increase, decrease or remain constant (fig. OV.S1). This range is meant to bracket the extremes of realistic changes that might occur in agriculture in the Chesapeake Bay watershed over the next few decades. One agricultural scenario reflects a 10-percent decrease in nitrogen inputs to the Bay watershed owing to a combination of possible changes including (1) increased numbers and efficiencies of management practices implemented, (2) increased areas of cropland in conservation programs or otherwise retired from production, (3) decreased application of fertilizer, and (4) decreased manure production (and release to fields) from decreased numbers of animals and (or) the development of new technologies that decrease the amount of nitrogen available to the Bay. Three of the agricultural scenarios reflect a 10-percent increase in nitrogen inputs to the Bay watershed owing to future changes in agriculture that could occur, such as (1) intensified crop or animal agriculture, (2) increased fertilizer use owing to increased crops, and (3) changes in crop types due to changes in climate. The 10-percent increase or decrease in nitrogen inputs to the Bay watershed were chosen to approximate upper and lower bounds of reasonable change over the next few decades and to show how changes in nitrogen released from agricultural areas could influence the total load to the Bay.

1. *Wastewater treatment technology does not improve*

Wastewater treatment technology does not improve and remains unchanged from 2012 (as population increases, the export of nitrogen from wastewater increases proportionally).

Manure production and *chemical fertilizer* use for agriculture remain constant compared to 2012.

2. *Wastewater treatment technology improves*

Wastewater treatment technology improves from 2012 (enhanced nitrogen removal is implemented in all population centers with wastewater treatment plants that discharge greater than 0.2 MGD (million gallons per day).

Manure production and *chemical fertilizer* use for agriculture remain constant compared to 2012.

3. *Crop and animal agriculture decreases*

Wastewater treatment technology does not improve and remains unchanged from 2012 (as population increases, the export of nitrogen from wastewater increases proportionally).

Changes in both *manure* production and *chemical fertilizer* use and management result in a cumulative annual decrease of 10 percent in nitrogen inputs between 2013 and 2050 (for example, due to better nutrient management, conservation programs and advances in technology).

4. *Crop agriculture increases*

Wastewater treatment technology does not improve and remains unchanged from 2012 (as population increases, the export of nitrogen from wastewater increases proportionally).

Crop production increases and changes in management practices result in a cumulative annual increase of 10 percent in nitrogen inputs from *chemical fertilizer* use between 2013 and 2050. (for example, due to a changes in crop types, yields and climate, and land removed from conservation programs).
Manure produced by animal agriculture is assumed to stay constant compared to 2012.

5. *Animal agriculture increases*

Wastewater treatment technology does not improve and remains unchanged from 2012 (as population increases, the export of nitrogen from wastewater increases proportionally).

Animal production will result in a cumulative annual increase of 10 percent of nitrogen inputs from the the production of *manure* between 2013 and 2050 (locations of animal agriculture and the ratios of animals for a given location are assumed constant compared to 2012.

6. *Crop and animal agriculture increases*

Wastewater treatment technology does not improve and remains unchanged from 2012 (as population increases, the export of nitrogen from wastewater increases proportionally).

Changes in both *manure* production and *chemical fertilizer* use result in a cumulative annual increase of 10 percent between 2012 and 2050 described in the two scenarios above.

Figure OV.S1. Select future scenarios of nitrogen sources and inputs.

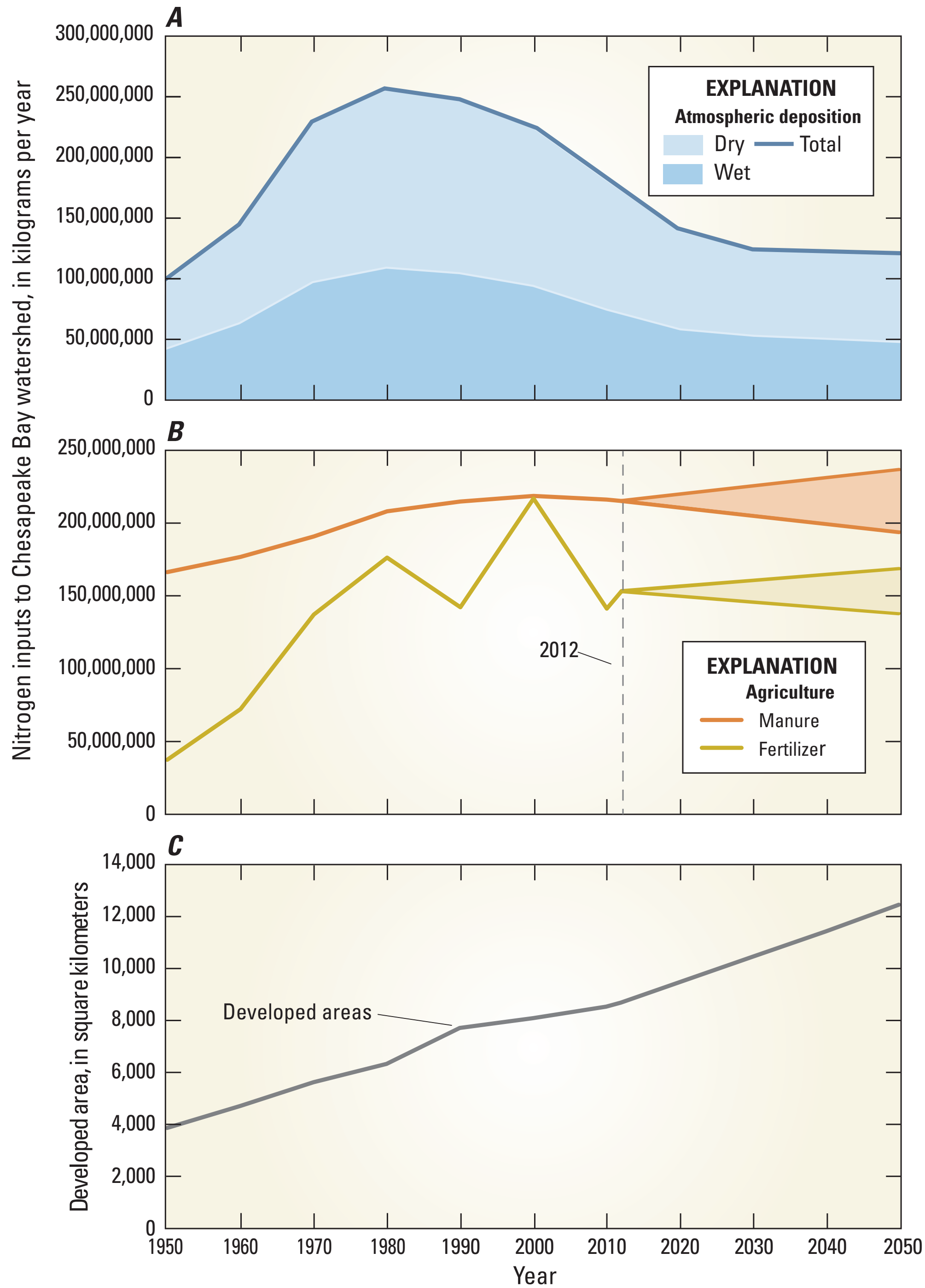


Figure OV.3. Annual nitrogen inputs to the Chesapeake Bay watershed, 1950–2050, from ***A***, atmospheric deposition; ***B***, fertilizer and manure from agriculture (two future agricultural scenarios are shown, figure OV.S1); and ***C***, developed areas (Hopple and others, 2020, 2021).

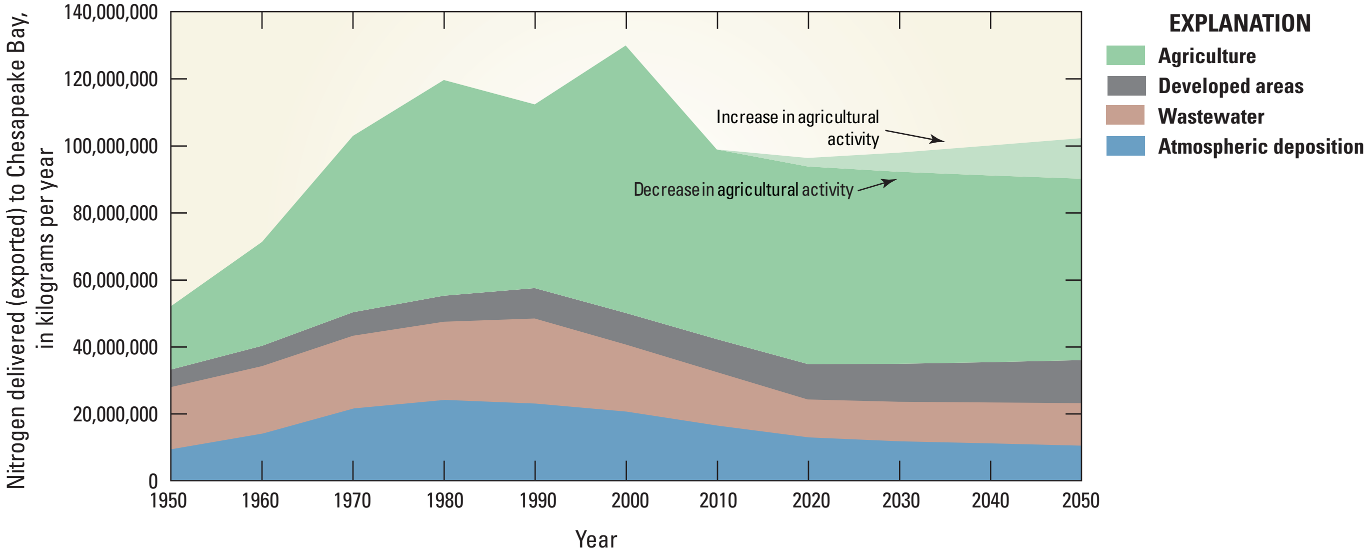


Figure OV.4. Annual nitrogen loads exported to the Chesapeake Bay by source, 1950–2050. Fertilizer and manure are combined into a single agricultural source for the modeled time period. After 2010, the two future agricultural scenarios represent (1) increased intensity of both crop and animal agriculture, and (2) decreased intensity of both crop and animal agriculture (figure OV.S1). Only the future scenario for constant wastewater treatment technology is shown (Hopple and others, 2020, 2021).

substantially from 1950 to the early 1980s owing to increased emissions from the transportation and industrial sectors (fig. OV.3*A*; see chap. 5). Since the 1980s, nitrogen from atmospheric deposition has decreased owing to the effects of the Clean Air Act and is projected to continue to decrease for the next few decades. Nitrogen from manure has continually increased from 1950 to 2012 owing to the intensification of animal agriculture in some areas of the watershed (fig. OV.3*B*; see chap. 7). Nitrogen from chemical fertilizer has increased over this same period, but there is considerable variability during the past few decades owing to changes in crops (both type and yield), weather, and economics (demand for the crops grown in the watershed) (fig. OV.3*B*). The variety and quantity of conservation practices to reduce nitrogen has also increased substantially over this time period and across the watershed (see chap. 7). Developed areas have increased and this trend is projected to continue (fig. OV.3*C*). Nitrogen from wastewater treatment plants has increased from 1950 to the early 1990s owing to increases in population, but since about 1990, nitrogen sources from wastewater have decreased owing to the implementation of enhanced nitrogen removal technologies in many developed areas (see chap. 8).

The nitrogen delivered from the watershed to the Bay (exports) were modeled for 1950–2050, based on measured data through 2012 and projected nitrogen sources to the watershed (fig. OV.4). The nitrogen load from agriculture peaked in 2000 and decreased through 2012. The sum of all sources of nitrogen to the Bay increased substantially from 1950 until the 1980s. Many of the current water-quality problems in the Bay started or expanded during this time period. Between the 1980s and 1990s, the export of nitrogen fluctuated around 120 million kilograms per year, depending on the amount of fertilizer applied. Starting in the year 2010, the SPARROW model used in this report (see chaps. 9 -) suggests there has been a decrease in the export of nitrogen to the Bay owing to a general decrease or fluctuation in use of fertilizer, and possibly the combined effects of nitrogen-reduction strategies initiated by the TMDL. Both future scenarios shown in figure OV.4, which bracket the realistic changes in agriculture in the watershed, suggest an annual export of nitrogen that presents a challenge to meeting nitrogen load targets for the current TMDL. This suggests that an increased effort to reduce nitrogen from agriculture, even more than the 10-percent reduction of nitrogen inputs from this sector used in this model, may need to be considered into the future. The sources of nitrogen from wastewater and developed areas will likely continue to gradually increase based on the rising population projected for the watershed unless improved wastewater technology can reverse this expected pattern.

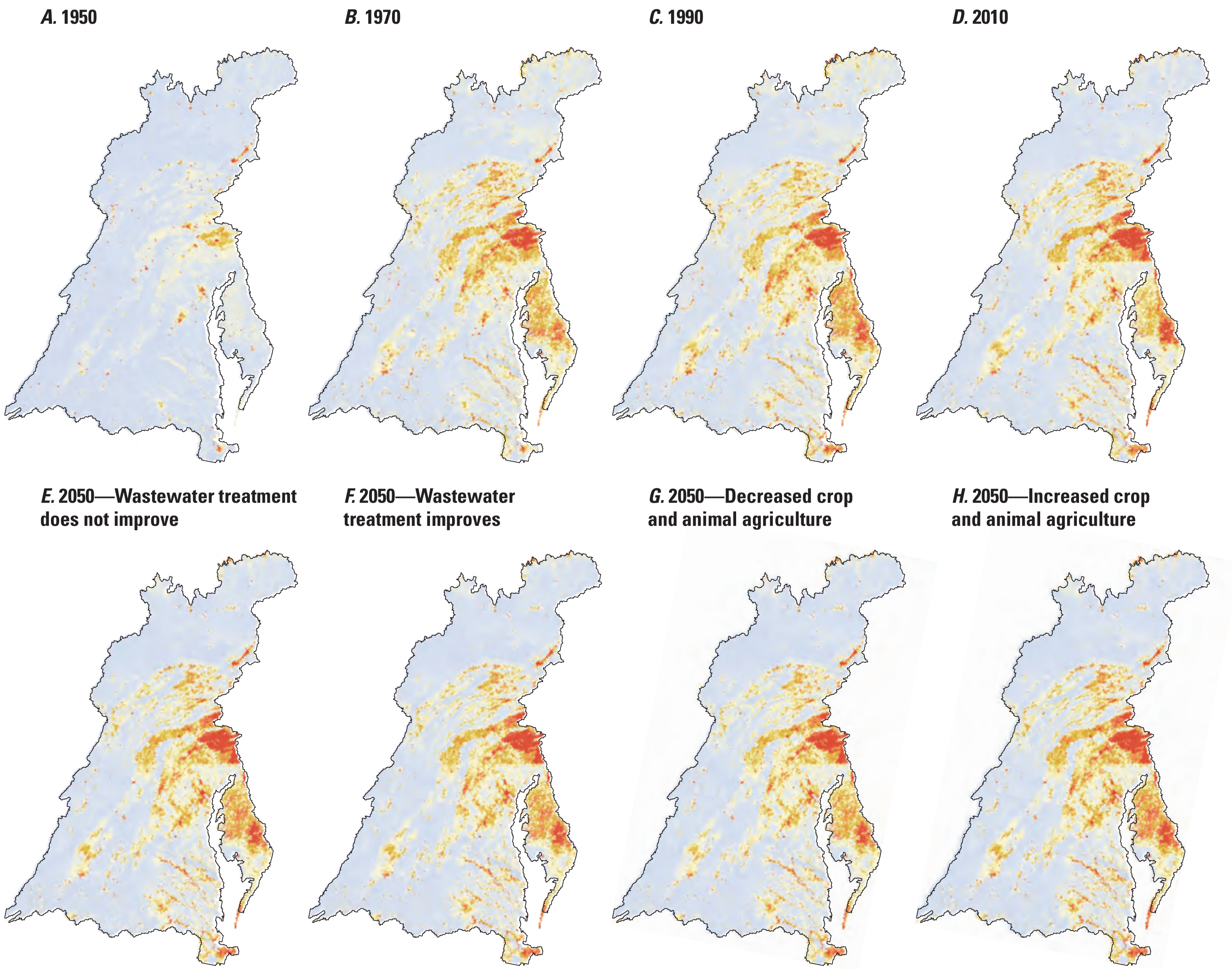


Figure OV.5. The spatial distribution of nitrogen yields from individual catchments in the Chesapeake Bay watershed for ***A***, 1950; ***B***, 1970; ***C***, 1990; ***D***, 2010; ***E***, 2050, where wastewater treatment does not improve (beyond the improvements implemented prior to 2015); ***F***, 2050, where wastewater treatment does improve; ***G***, 2050, where crop and animal agriculture decreases; and ***H***, 2050, where crop and animal agriculture increases (Hopple and others, 2020, 2021).

The estimates of past and future exports can also be viewed from a spatial perspective (fig. OV.5). Nitrogen exports are expressed as yields (kilograms of nitrogen per square kilometer) to allow for a direct comparison and the increase in yields from 1950 to 2010 are shown in figure OV.5A–*D*. In 1950, the nitrogen yields per square kilometer were greatest in the developed areas owing to wastewater discharge. Over the years, the areas that contribute nitrogen to the Bay increased as agriculture intensified in the watershed. Agricultural activities in areas with prime soils and significant groundwater-surface water connections, such as the Coastal Plain and areas underlain by carbonate aquifers, have the highest nitrogen yields. There are many undeveloped (forested) areas of the watershed that have contributed and will continue to contribute only minimal yields of nitrogen.

Figure OV.5E–*H* shows the nitrogen yields projected to 2050 under four of the future scenarios (fig. OV.S1). None of the results for future scenarios presented here show major spatial differences in 2050 compared to 2010. The current (2010) high (agricultural and urban areas) and low (undeveloped lands) nitrogen yield areas are generally expected to continue into 2050. At the scale of these maps, changes in land use (undeveloped to urban, agricultural to urban, undeveloped to agriculture) are too small to be observed, but nevertheless are important. The maps illustrate subtle, but real, differences in agricultural areas. For example, the differences in the estimated yields of nitrogen from some parts of the Delmarva Peninsula can be observed when comparing figure OV.5*G*, *H*. These two maps show the two agricultural scenarios that are most different (using a 10-percent increase and 10-percent decrease in the amount of nitrogen inputs from agricultural activities).

Implications for Managing Nitrogen into the Future

The water-quality problems from excess nitrogen that were observed in the Bay in the last few decades of the 20th century have led to the current approaches to regulation and management. The foundation of the water-quality problems is rooted in the human activities and population growth that occurred during the previous century but accelerated in the 1950s and beyond. Human activities related to urbanization, agriculture, and industry provided excess nitrogen to streams, groundwater, and the atmosphere. Over the past few decades, Federal, state, and local decision makers have set in motion different types of controls to reduce excess nitrogen in the environment. The multiagency partnership of the Chesapeake Bay Program has set a series of nitrogen reduction goals and the regulatory framework embodied by the landmark Chesapeake TMDL, with the completion of management practices scheduled by 2025. These efforts are coordinated with Federal, state, and local level nitrogen reduction targets. Many of the final, practical, everyday decisions are made by resource managers, agricultural producers, and residents of rural and urban environments. The decisions made for their lawns, fields, livestock, pets, and maintenance on septic systems are important for reaching the goals of healthy and sustainable ecosystems in the Bay.

The ability to analyze inputs of nitrogen to the watershed and its export to the Chesapeake Bay over a century of time through modeling provides a powerful tool for resource managers. Modeled results suggest that the water-quality impairments of the Bay will continue to be a concern into the future, as quantified by projections that indicate reaching and maintaining the nitrogen load targets of the current TMDL will be challenging. Reductions in nitrogen loads owing to decreasing atmospheric deposition and wastewater discharges have largely been achieved. However, the population and developed areas will continue to grow and increasing nitrogen loads are a likely result. Agriculture is the largest contributor of excess nitrogen to the Bay and, even with the most environmentally favorable future scenario (10-percent decrease in nitrogen inputs from agriculture), the water-quality goals of the TMDL will remain a challenge to achieve over time (fig. OV.4). The question then becomes, at the time horizon of decades, what are the innovative changes that can be implemented to reach the water-quality goals for lowering nutrient exports to the Bay while also achieving water-quality standards for dissolved oxygen and recovery of submerged aquatic vegetation? If further reductions in nitrogen inputs are needed to reach nitrogen export targets, additional conservation measures will be important such as (1) further decreases in application rates of fertilizer, (2) continued efforts, increased efficiencies, and proper placement of agricultural best management practices, (3) increased areas of cropland moved to conservation programs or otherwise land retired from production, (4) development of new technologies that decrease the export of nitrogen from fields, (5) decreased numbers of animals, and (6) the development and implementation of technologies to remove nitrogen from manure before field application. The specifics of the decision-making process that will reduce excess nitrogen originating from urban and agricultural areas in future decades will be extremely challenging, just as difficult decisions that have been made over the past several decades to address the TMDL goal. Nevertheless, if society aims to reach these nitrogen reduction goals, these decisions need to be made and implemented for the ecological and economic benefit of the Bay and for the residents of the watershed (Morgan and Owens, 2001; Chesapeake Bay Foundation, 2012).

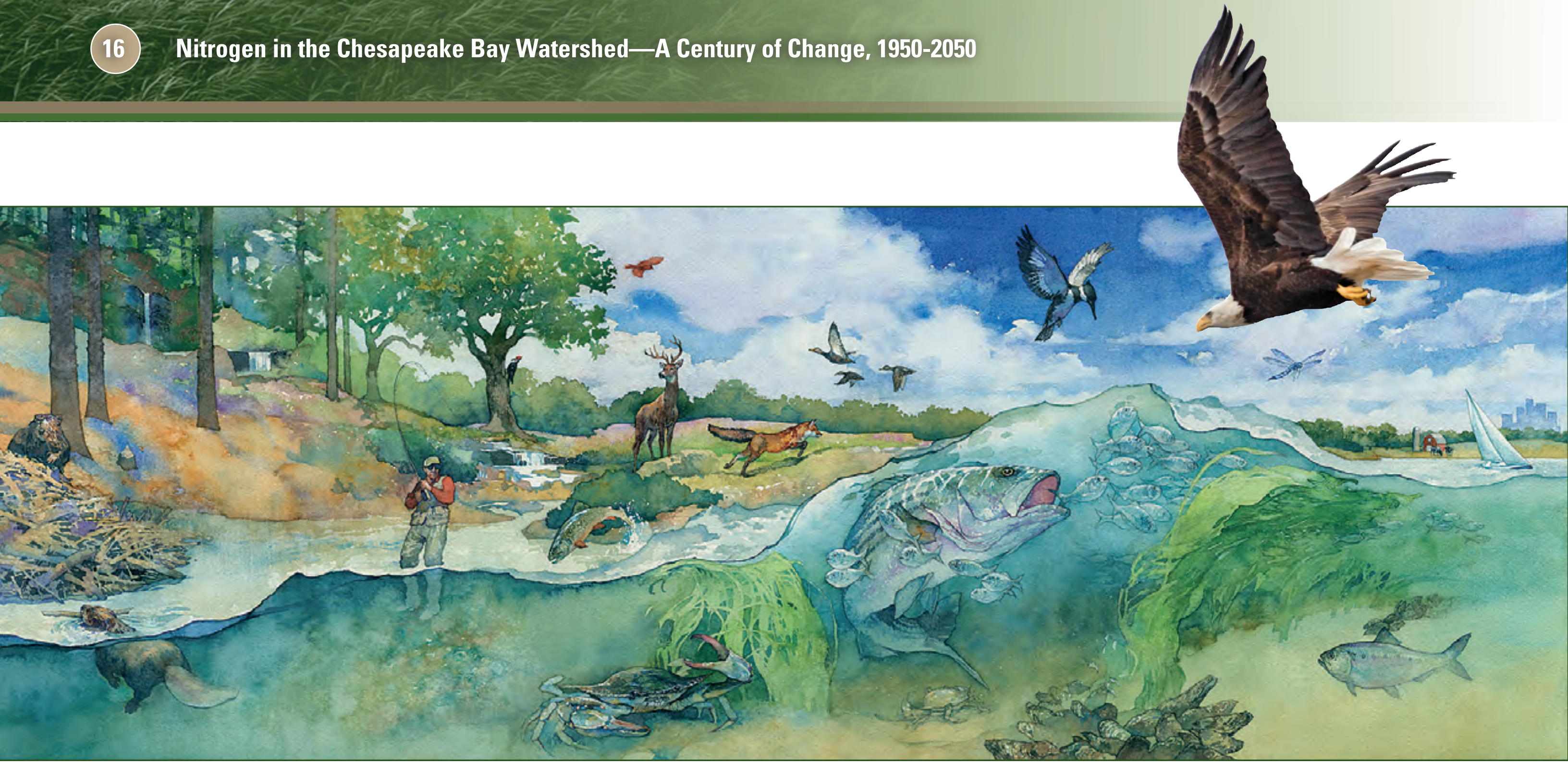
Dairy farm in Lancaster County. Courtesy of the Chesapeake Bay Program, used with permission.

Environmental Setting of the Chesapeake Bay Watershed

By Silvia Terziotti[1] and Andrew J. Sekellick[1]

The Chesapeake Bay is the Nation's largest estuary and its 64,243 square-mile (166,389 square kilometers) watershed drains portions of six states (New York, Pennsylvania, West Virginia, Maryland, Delaware, and Virginia) and the District of Columbia. As of 2017, about 18.2 million people live in the watershed, including the population centers of Washington, D.C.; Baltimore, Maryland; Richmond, Virginia; and Harrisburg, Pennsylvania. The varied climate, land uses, and geology support an array of ecological, economic, and recreational sectors. As one of the fastest growing population and economic centers in the country, this vast watershed is a national treasure, but resource managers face the challenge of restoring the Bay's water quality. U.S. Geological Survey, Chesapeake Bay: A Landsat 8 Surface Reflectance Mosaic (2014).

[1]U.S. Geological Survey.



Illustration, eagle and wetland images, Courtesy of the Chesapeake Bay Program, used with permission.

Ecology and Natural Resources

The Chesapeake Bay watershed is renowned for its abundance of aquatic life, migratory birds, and terrestrial wildlife. The estuary provides habitat to a diverse set of wildlife such as striped bass, blue crabs, oysters, dolphins, black duck, and river otters with occasional visits from manatees and humpback whales. Much of the forested mountain areas are still home to common native species such as eastern hemlock, white oak, white-tailed deer, black bear, beaver, and brook trout, but also reintroduced wildlife such as elk, peregrine falcon, and the bald eagle. At least five endangered species are native to the watershed, including the Atlantic sturgeon and Virginia big-eared bat (Chesapeake Bay Program, 2019d). Unfortunately, nonnative species such as phragmites, Japanese knotweed, didymo, emerald ash borer, nutria, invasive catfish, and northern snakehead have invaded and decreased ecosystem diversity in many areas.

Wetlands

Wetlands are a critical component of the ecosystem of the Chesapeake Bay. Tidal (estuarine) and freshwater nontidal (palustrine) wetlands are prevalent throughout the Bay watershed. These marshy areas provide food and habitat for migratory birds in the winter, and year-round protection for thousands of animals and aquatic species. They provide important spawning and nursery grounds for shellfish and many commercially valuable fish. Wetlands are also critical in flood and erosion control, as well as sediment and nutrient retention. Thick vegetation within wetlands slows the movement of surface and groundwater to rivers and tidal waters, providing a buffer and filter for pollutants. There has been widespread recognition of the importance of protecting wetlands from development and sea level rise in order to protect water quality (Chesapeake Bay Program, 2019c).

Wetlands near Nanticoke Wildlife Management Area in Wicomico County, Maryland. Courtesy of the Chesapeake Bay Program, used with permission.

Cultural and Recreational Opportunities

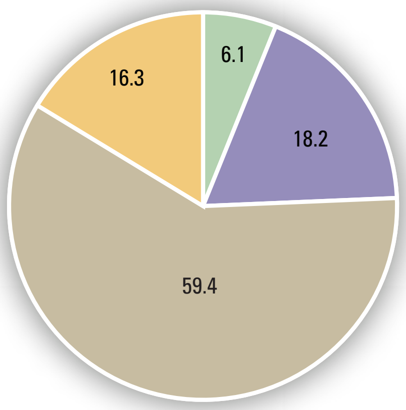
The Chesapeake Bay's long human history is reflected in the many cultural and historically significant sites throughout the watershed. The widespread Native American footprint can be seen through the many names given to streams and rivers such as the Susquehanna. The rich heritage of the watershed includes the first permanent European settlement in Jamestowne, important events of the Revolutionary and Civil Wars, and the eventual home of our Nation's capital. Recreational opportunities abound from the mountains to the coast with 55 national parks, 16 national refuges, 2 national forests, 5 national trails, and many other natural protected areas. These recreational areas contribute to the economy with tourism dollars, sales, and services (U.S. Army Corps of Engineers, 2015). As recently as 2011, saltwater recreational fisheries alone were reported to contribute $1.6 billion in sales, and are estimated to stimulate roughly $800 million of additional economic activity and support about 13,000 jobs (Chesapeake Bay Program, 2019a). Additionally, freshwater fishing along with hunting provide an economic boost to local rural economies.

Fisherman in the Chesapeake Bay watershed. Courtesy of the Chesapeake Bay Program, used with permission.

Visitors to Fletcher's Cove in Washington, D.C. Courtesy of the Chesapeake Bay Program, used with permission.

Employment and the Economy

More than 18 million people live and work in the Chesapeake Bay watershed (Phillips and others, 2017; Hopple and others, 2021). The Bay's regional economy provides 8.3 million jobs and an annual income of almost $400 billion (DiPasquale, 2017). Sales and services, followed by construction and manufacturing, are the top industries. The public sector follows closely behind, with more than 16 percent of workers within the Chesapeake Bay watershed employed by the government (fig. ES.1; McKendry, 2009).



EXPLANATION

Percent of earnings by industry

- Sales and services
- Government
- Construction and manufacturing
- Agriculture and natural resources

Figure ES.1. Employment by sector, in percent, in the Chesapeake Bay watershed (McKendry, 2009).

Federal, state, and local governments have major economic impacts on the Chesapeake Bay region. Government agencies provide jobs directly and indirectly to contractors, as well as sales and service sectors, and are also a key source of expenditures. In 2005, one-fifth of the counties in the Bay watershed received more than $10,000 per person in federal expenditures (McKendry, 2009). The farming industry includes 83,000 farms and an annual agricultural production of $10 billion (Natural Resources Conservation Service, 2018). In 2009, the commercial seafood industry in Maryland and Virginia contributed $890 million in income and almost 34,000 jobs to the local economy (Chesapeake Bay Foundation, 2012). Investments in the restoration of the Bay are supported by approximately $1.7 billion annually (Phillips and others, 2017). Recreational fishing in the streams and rivers of the watershed contribute more than $1.6 billion to the economy (Chesapeake Bay Program, 2019a).

Agriculture and natural resources

Construction and manufacturing

Government

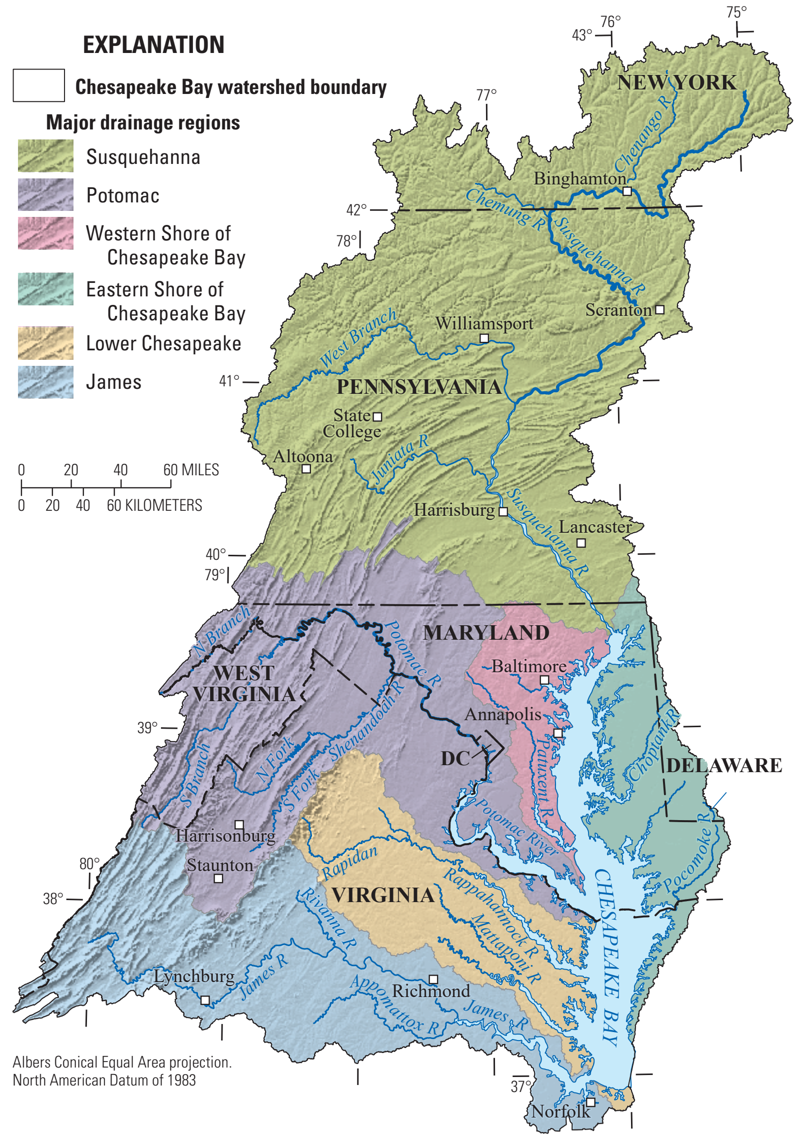
Sales and services

Images courtesy of the Chesapeake Bay Program, used with permission.

Major River Drainage Regions

The Chesapeake Bay is the largest estuary in the United States. The watershed that drains to the Bay covers 64,243 square miles (mi^2) (166,389 square kilometers [km^2]) and drains portions of six states (New York, Pennsylvania, West Virginia, Maryland, Delaware, and Virginia) and the District of Columbia (fig. ES.2*A*). More than 18 million people live in the watershed, which includes the population centers of Washington D.C.; Baltimore, Maryland; Norfolk and Richmond, Virginia; and Harrisburg, Pennsylvania. Water is both abundant and adversely affected by the many human activities that occur within the watershed. Almost all of the water that enters the Chesapeake Bay comes from the six major drainage regions that surround the Bay, including the Susquehanna River, Potomac River, Eastern Shore, Western Shore, Lower Chesapeake, and James River (fig. ES.2*B* and ES.2*C*). The Susquehanna River basin is the largest in the Bay watershed (43 percent), supplying 51 percent of the freshwater inflow. The Potomac River basin covers 22 percent of the Bay watershed and contributes 19 percent of the streamflow. The James River basin follows in size, contributing 13 percent of streamflow. The smaller drainage regions—the Lower Chesapeake, Western Shore, and Eastern Shore of the Chesapeake Bay—make up the remaining 19 percent of the area and contribute almost 17 percent of freshwater streamflow. Overall, the Chesapeake Bay watershed has 150 major rivers and streams fed by more than 100,000 smaller tributaries (Chesapeake Bay Program, 2019b).

A



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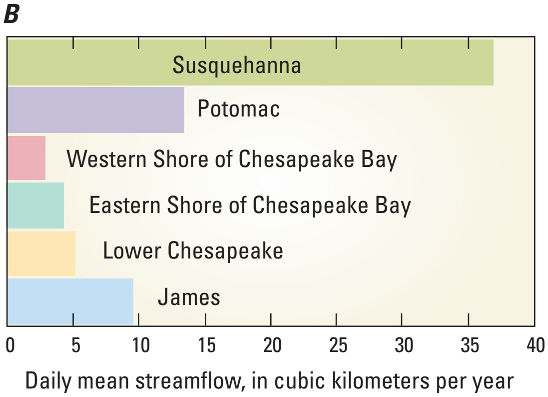

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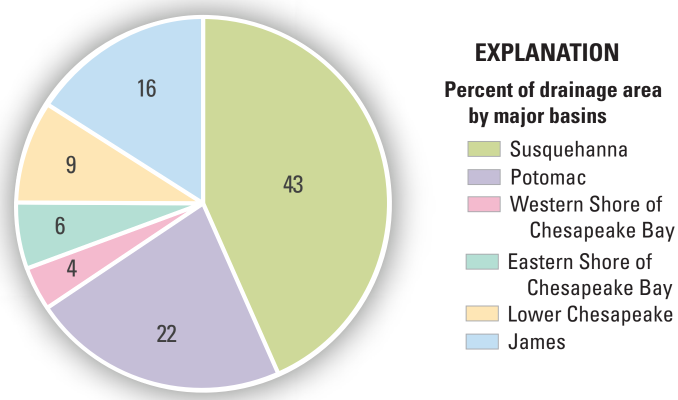


Figure ES.2. The Chesapeake Bay watershed, including ***A***, the location of major political boundaries, cities, drainage regions, and rivers; ***B***, average annual streamflow of the major river drainage regions and Eastern Shore (Chesapeake Bay Program, 2017); and ***C***, areal distributions of the major drainage basins of the Chesapeake Bay watershed, in percent (Chesapeake Bay Program, 2017).

Climate

Climate is varied throughout the watershed. The higher elevations in the north and west have cooler average temperatures and generally receive less precipitation, except where air moves from low to high elevations and cools more rapidly, such as the Blue Ridge and parts of the Allegheny Plateau (figs. ES.3A,B; ES.4A).

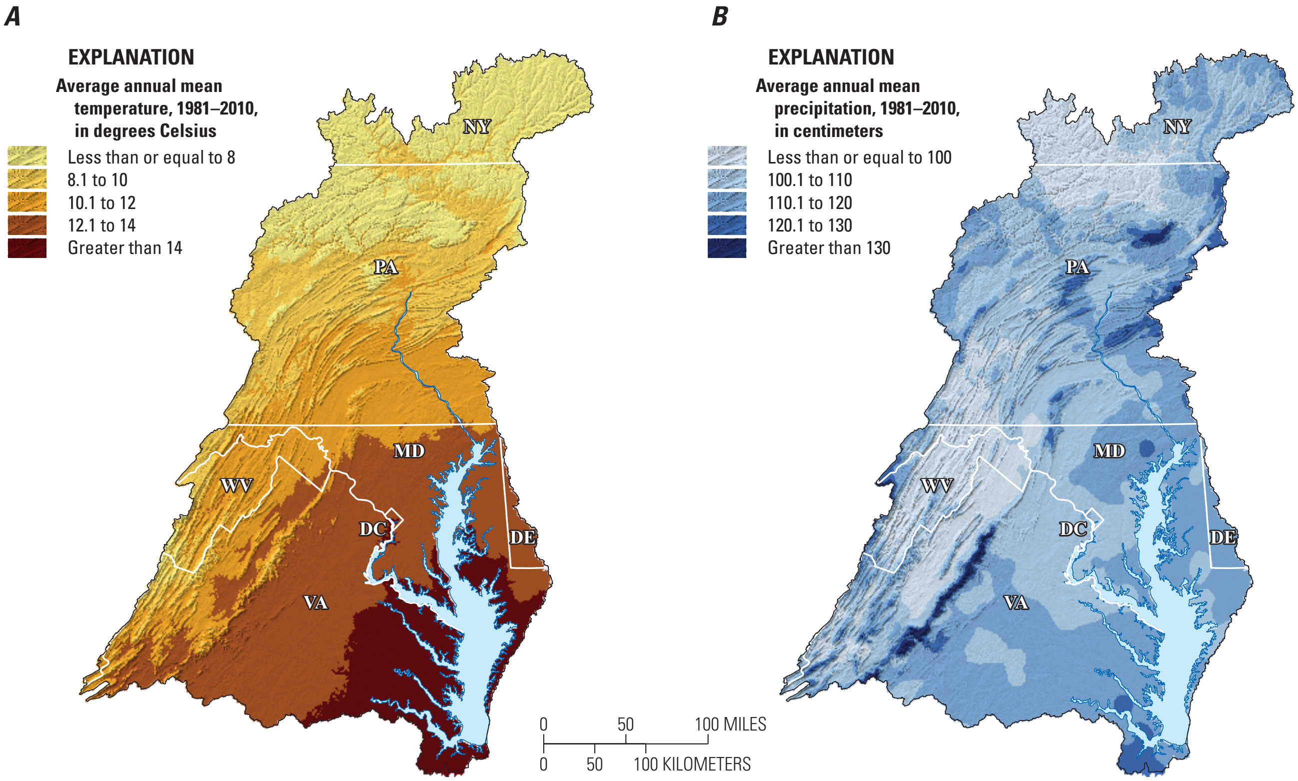


Figure ES.3. Spatial distribution of the (***A***) average annual mean air temperature and (***B***) average annual mean precipitation for 1981–2010 across the Chesapeake Bay watershed (PRISM Climate Group, 2017). DC, Washington, D.C.; DE, Delaware; MD, Maryland; NY, New York; PA, Pennsylvania; VA, Virginia; WV, West Virginia.

Geology and Groundwater

The five characteristic geologic settings (physiographic provinces) in the Chesapeake Bay watershed are the Appalachian Plateau, Blue Ridge, Piedmont, Valley and Ridge, and Coastal Plain (fig. ES.4*A*). These physiographic provinces are partly defined by the underlying rock types, primarily siliciclastic, crystalline, carbonate, and unconsolidated (sand/gravel) units. The geologic setting has a strong influence on water quality and transport of nitrogen as water travels through the rocks to streams and eventually to the Bay (fig. ES.4*B*; Bachman and others, 1998). Overall, groundwater accounts for more than half of the total flow to streams in the Chesapeake Bay watershed. The time it takes for groundwater and pollutants to reach streams (lag time) can range from days to decades. This means that, in some areas, the effect of management actions or additional sources of nitrogen may not be realized for decades to come.

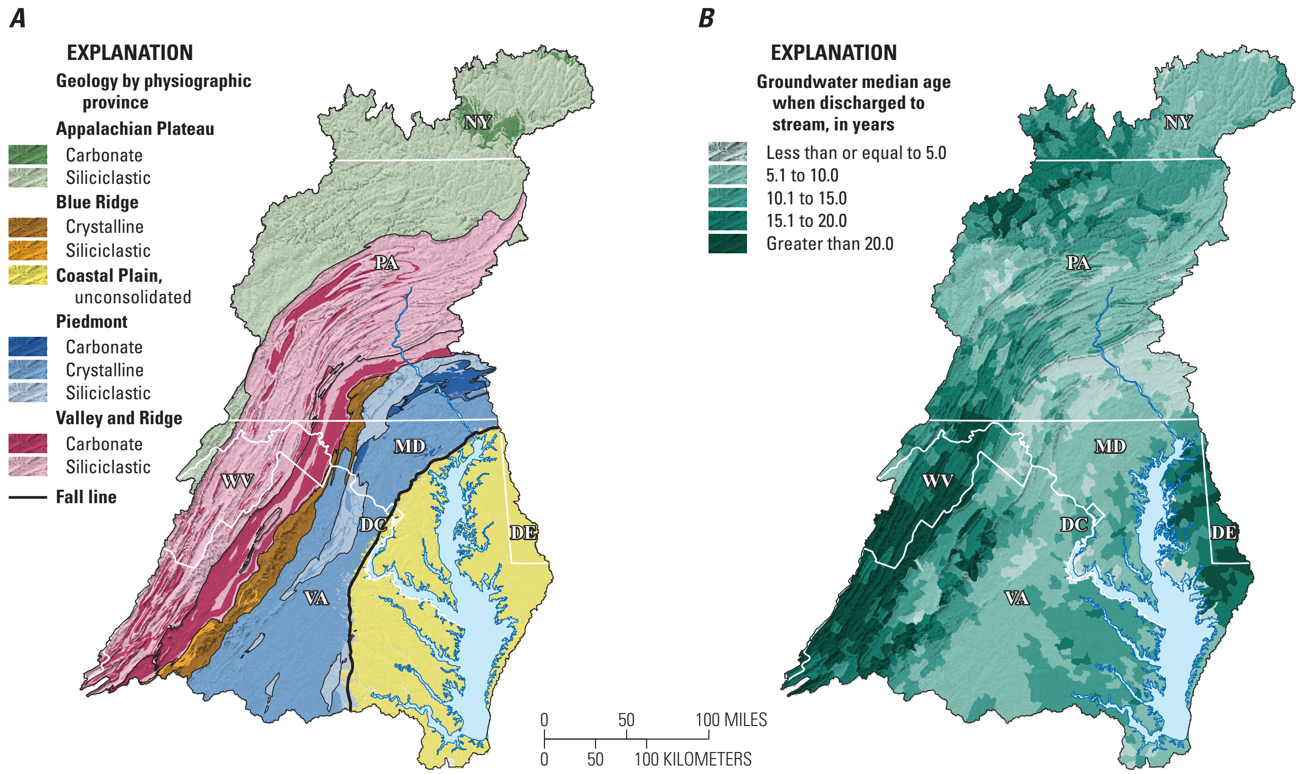
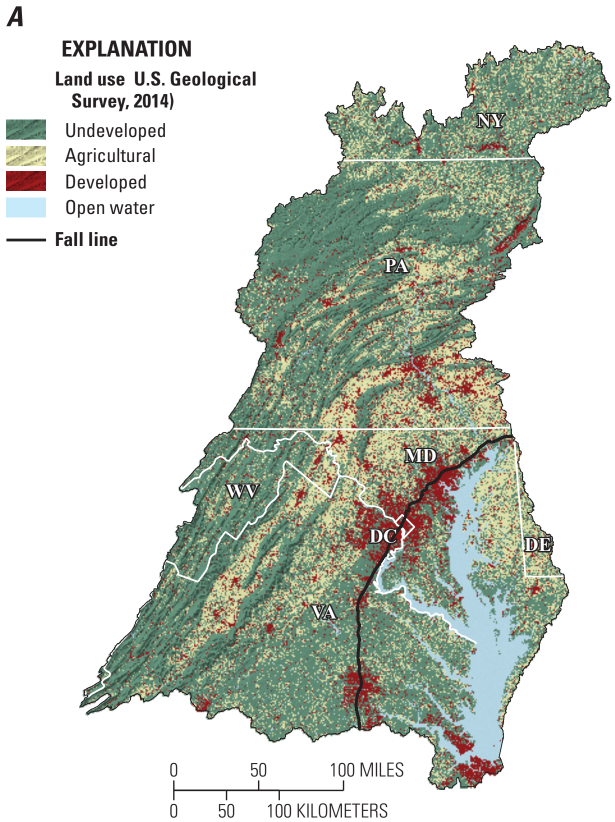


Figure ES.4. The distribution of geology and groundwater age in the Chesapeake Bay watershed. ***A***, Major physiographic provinces and bedrock geology, and ***B***, spatial distribution of the median ages of groundwater discharged to streams in the Chesapeake Bay watershed. DC, Washington, D.C.; DE, Delaware; MD, Maryland; NY, New York; PA, Pennsylvania; VA, Virginia; WV, West Virginia.

Land Use

Land use in the Chesapeake Bay watershed is predominantly undeveloped (64 percent), along with agricultural (25 percent) and developed areas (11 percent). (fig. ES.5). Population centers tend to be close to the Bay, where many original native communities were located and where permanent European settlements developed. Prime agricultural land can be found in much of the Coastal Plain areas containing fertile soils and carbonate valleys. Although the watershed remains predominately undeveloped and agricultural land, the developed land has increased in size and intensity.



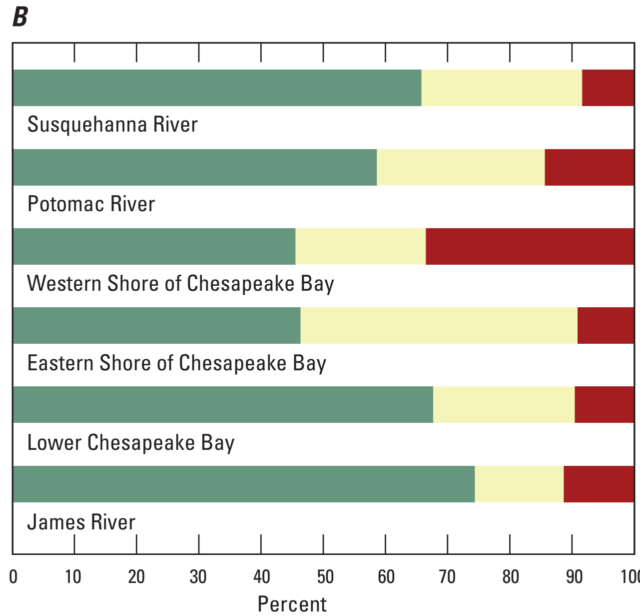
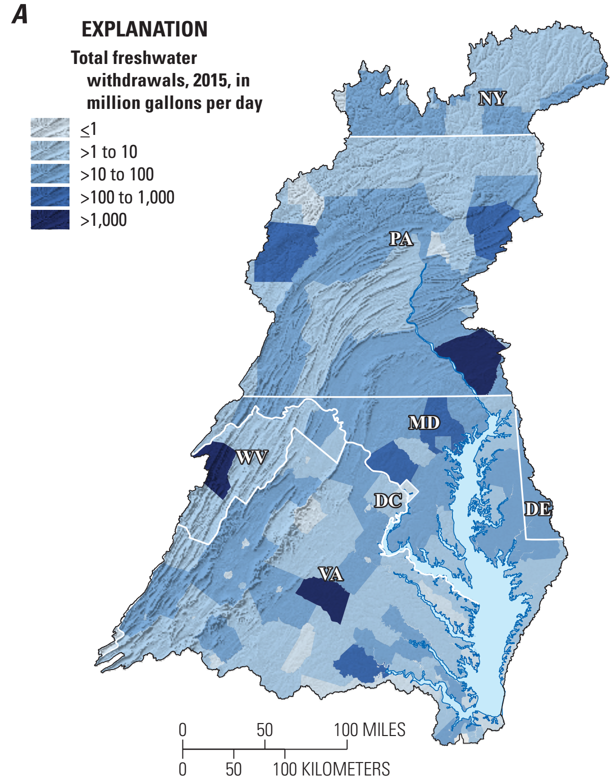


Figure ES.5. Land use in the Chesapeake Bay watershed is predominantly undeveloped (64 percent), along with agricultural (25 percent) and developed areas (11 percent). ***A***, Spatial distribution of land use in the Chesapeake Bay watershed, and ***B***, proportion of land use in the six major drainage regions. DC, Washington, D.C.; DE, Delaware; MD, Maryland; NY, New York; PA, Pennsylvania; VA, Virginia; WV, West Virginia.

Water Use

Almost 12,000 million gallons (45.4 million cubic meters) of fresh water are withdrawn each day throughout the Chesapeake Bay watershed for a variety of purposes, including power generation, drinking water, irrigation for farms and golf courses, fish hatcheries, mining, oil and gas extraction, and industrial applications (fig. ES.6*A*). About 90 percent of water use comes from surface water, with the remainder from groundwater sources (Dieter and others, 2018). Drinking water in the Chesapeake Bay watershed is primarily from surface water sources. For more populated regions, public suppliers provide drinking water from both surface water sources (86 percent) and groundwater (14 percent) (fig. ES.6*B*). In rural areas, about 13 percent of drinking water comes from private groundwater wells.

Water is used at many power plants to generate steam and provide cooling. In 2015, thermoelectric power generation was responsible for 59 percent of all freshwater withdrawals, although only 2 percent is consumptive and removed from the immediate environment. Public supply and domestic uses account for 26 percent of total freshwater withdrawals. Some sectors, like irrigation, are increasing with changing food supply demands (Dieter and others, 2018)



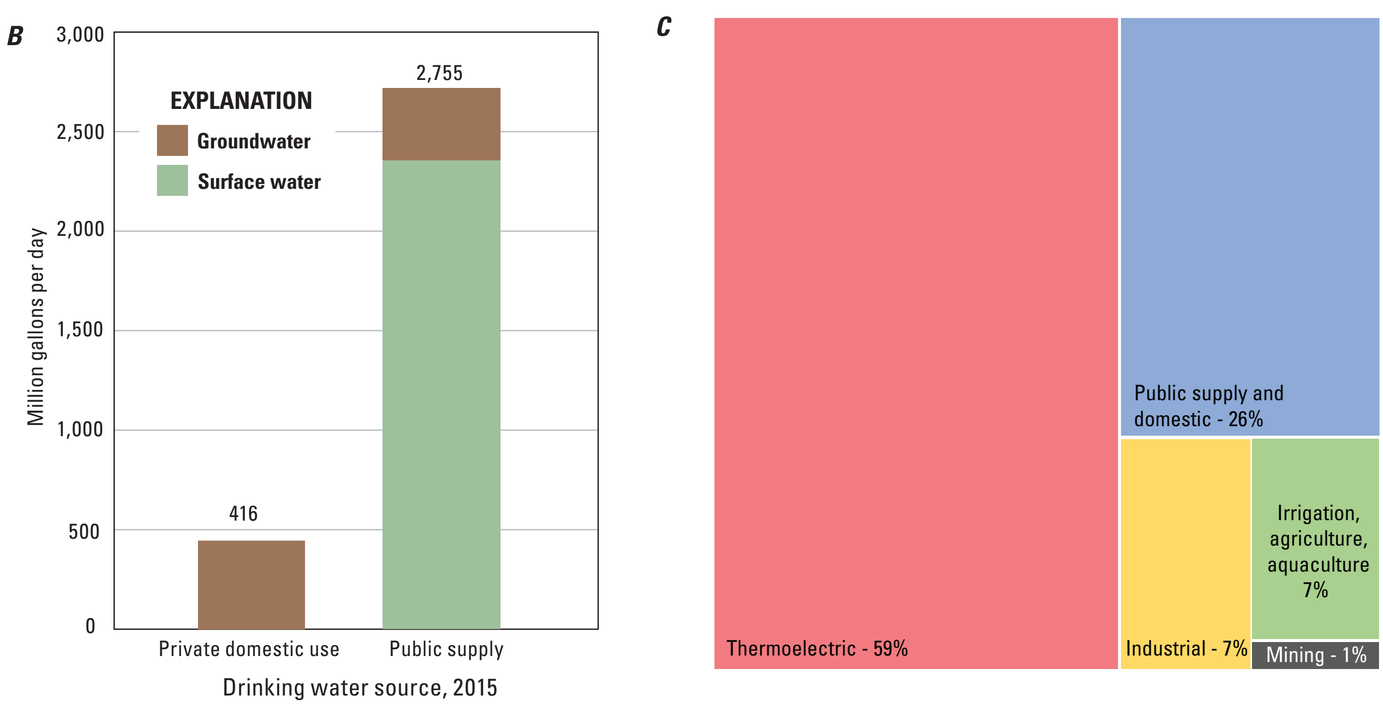


Figure ES.6 Water use in the Chesapeake Bay watershed. ***A***, spatial distribution of total freshwater withdrawals for drinking water in 2015, ***B***, private domestic and public supply water use in 2015, and ***C***, usage of freshwater, in percent (Dieter and others, 2018). DC, Washington, D.C.; DE, Delaware; MD, Maryland; NY, New York; PA, Pennsylvania; VA, Virginia; WV, West Virginia.

Breton Bay in St. Mary's County, Maryland. Courtesy of the Chesapeake Bay Program, with aerial support by LightHawk, used with permission.

Nitrogen Setting of the Chesapeake Bay Watershed

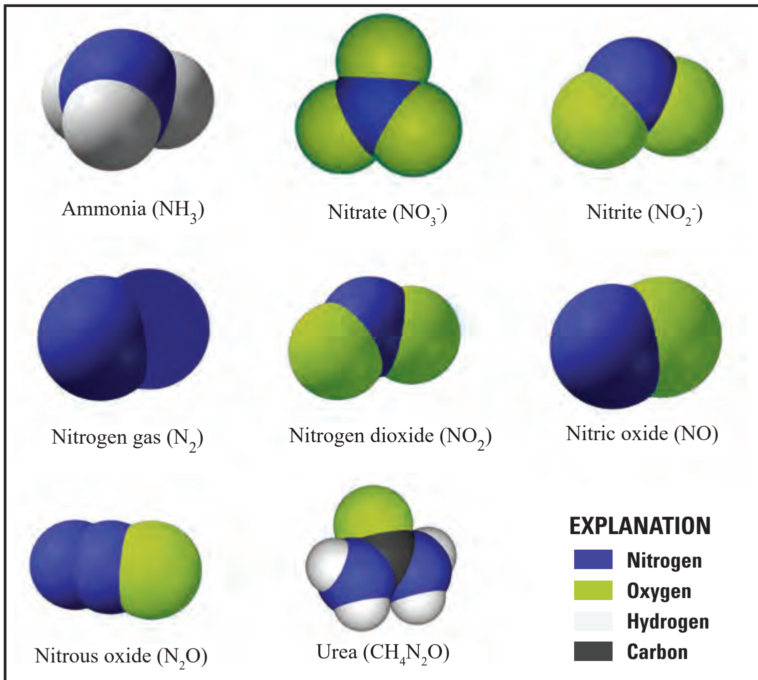
By Paul D. Capel[1] and John W. Clune[1]

Nitrogen may occur in many chemical forms as it cycles through the environment (see Sidebar: Reactive [Bioavailable] Nitrogen). It is a critical nutrient for all forms of life and a frequent component of human, animal, agricultural, and industrial waste. The major nitrogen sources to the Chesapeake Bay watershed are the application of crop fertilizers and livestock waste (manure) to the land, wastewater discharged to streams, runoff from developed areas, and atmospheric deposition (primarily from combustion and agricultural sources for oxidized nitrogen and ammonia, respectively). Other additional nitrogen sources such as biological fixation and industrial discharge are assumed to be minor. Nitrogen imports from food are reflected in the sources to the landscape, applied fertilizer, manure, and wastewater. Nitrogen is transported from the watershed to the Bay through streams, groundwater, and wastewater discharges. A summary of nitrogen forms (see sidebar–Forms of Nitrogen) as well as graphs and tables of the common transformations, sources to the watershed, and export from the watershed to the Bay are included in figures NS.1–NS.8.

Total reactive nitrogen in water includes only the reactive forms of nitrogen that affect water quality and for the purposes of this report are the sum of ammonia, nitrate, nitrite, and organic nitrogen (dissolved and particulate). Because there are other elements contained in the various forms of nitrogen, including oxygen, hydrogen, and carbon, the concentration of total nitrogen is usually given in terms of the nitrogen content only and commonly reported in units of milligrams per liter (mg/L) as nitrogen. For example, a solution which contained 10 milligrams of urea dissolved in water would have a concentration of 10 mg/L as urea and a concentration of 2.3 mg/L as nitrogen, because urea contains 23 percent (by weight) nitrogen.

Photograph of power plant in Snyder County, Pennsylvania. Courtesy of the Chesapeake Bay Program, with aerial support by LightHawk, used with permission.

[1]U.S. Geological Survey.



	Nitrate (NO_3^-)	Nitrite (NO_2^-)	Ammonia (NH_3 / NH_4^+)	Dissolved organic nitrogen	Particulate organic nitrogen	Urea (CH_4N_2O)	Nitrogen gas (N_2)	Nitrogen dioxide (NO_2)	Nitric oxide (NO)	Nitrous oxide (N_20)
Included in "Total nitrogen in water" in water in this report	X	X	X	X	X					
Forms of nitrogen that are important in different environments										
Surface water	X	X	X	X			X			
Groundwater	X						X			
Atmosphere					X		X			X
Atmospheric deposition	X		X	X	X					
Wastewater plant effluent			X	X						
Environmental benefits and impacts										
Chemical fertilizers	X		X			X				
Manure	X		X	X	X					
Toxic to humans	X		X							
Toxic to aquatic biota			X							
Eutrophication	X			X						
Climate change (greenhouse gas)										X
Acidic precipitation	X								X	
Chemical state in different environments										
Exist as a gas (in air) or a dissolved gas (in water)			X				X	X	X	X
Exist as an ion (charged chemical) in water	X	X	#	#	#					
Commonly attach to particles in water and soil			X			X				

Figure NS.1. Elemental configurations of common forms of nitrogen found in air, water, and soil. The various forms of nitrogen are important in different types of environments, have different environmental benefits and impacts, and exist in different chemical states. "X" denotes that this form of nitrogen is included in this group. "#" denotes that the chemical can exist in both charged (ion) and noncharged (neutral) forms in water.

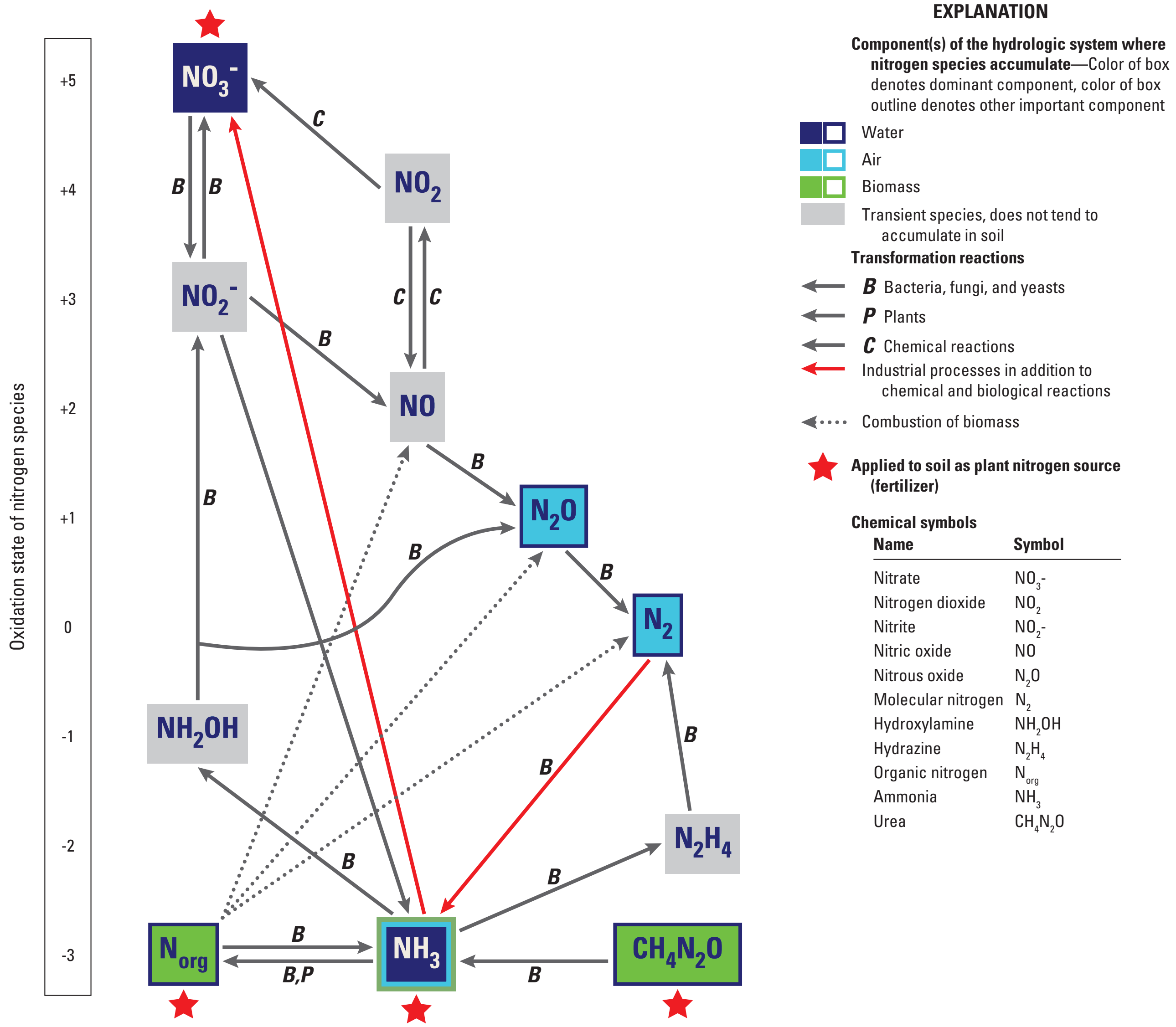


Figure NS.2. Natural chemical and biological reactions transform nitrogen among the various forms that are observed in the environment. These transformations create a natural nitrogen cycle in which there is no permanent accumulation of one of the forms. In addition to the natural transformation processes, industrial transformation processes are used to make nitrogen fertilizers. The biologically mediated process by which nitrate (NO_3-) is transformed to nitrite, nitrogen oxides, nitrogen gas, or ammonia is called denitrification. The inverse biologically mediated process, nitrification, transforms ammonia to nitrite and, then, to nitrate. Nitrogen mineralization is the conversion of organic N to ammonium. Modified from Capel and others (2018a).

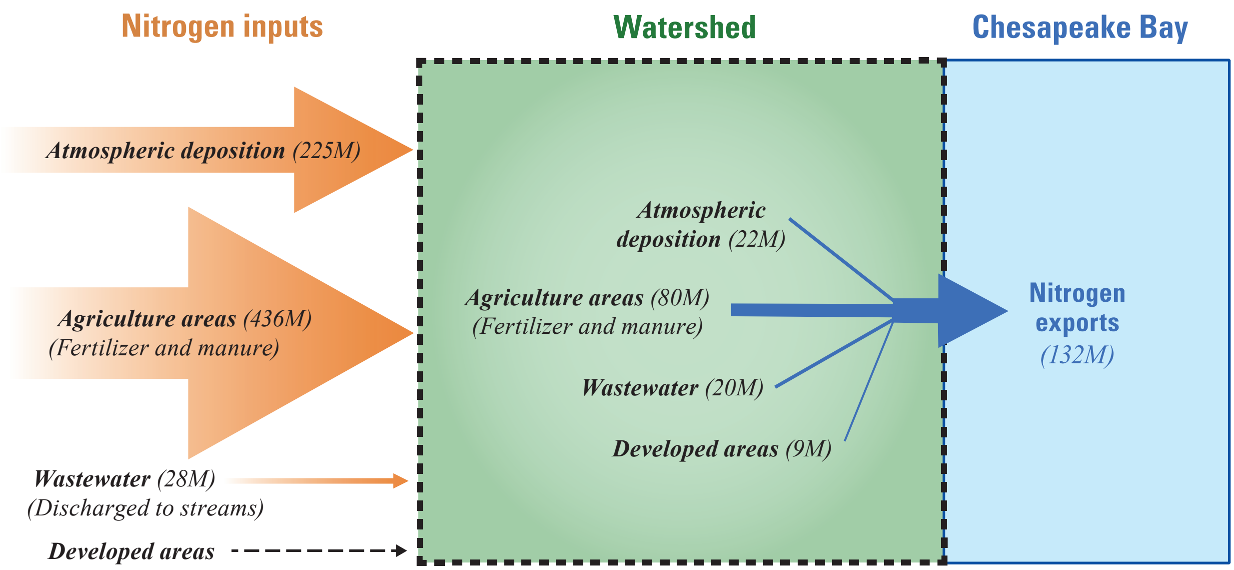


Figure NS.3. The annual major inputs (orange arrows) to the Chesapeake Bay watershed and tributaries and annual nitrogen exports (blue arrows), by source, delivered to the Chesapeake Bay, 2010. The size of the arrows is proportional to the mass of nitrogen shown in parenthesis as millions (M) of kilograms per year. Developed areas are sources of nitrogen to the watershed, but not proportional to the mass of nitrogen because inputs, which are presented in square kilometers (Hopple and others, 2020, 2021).

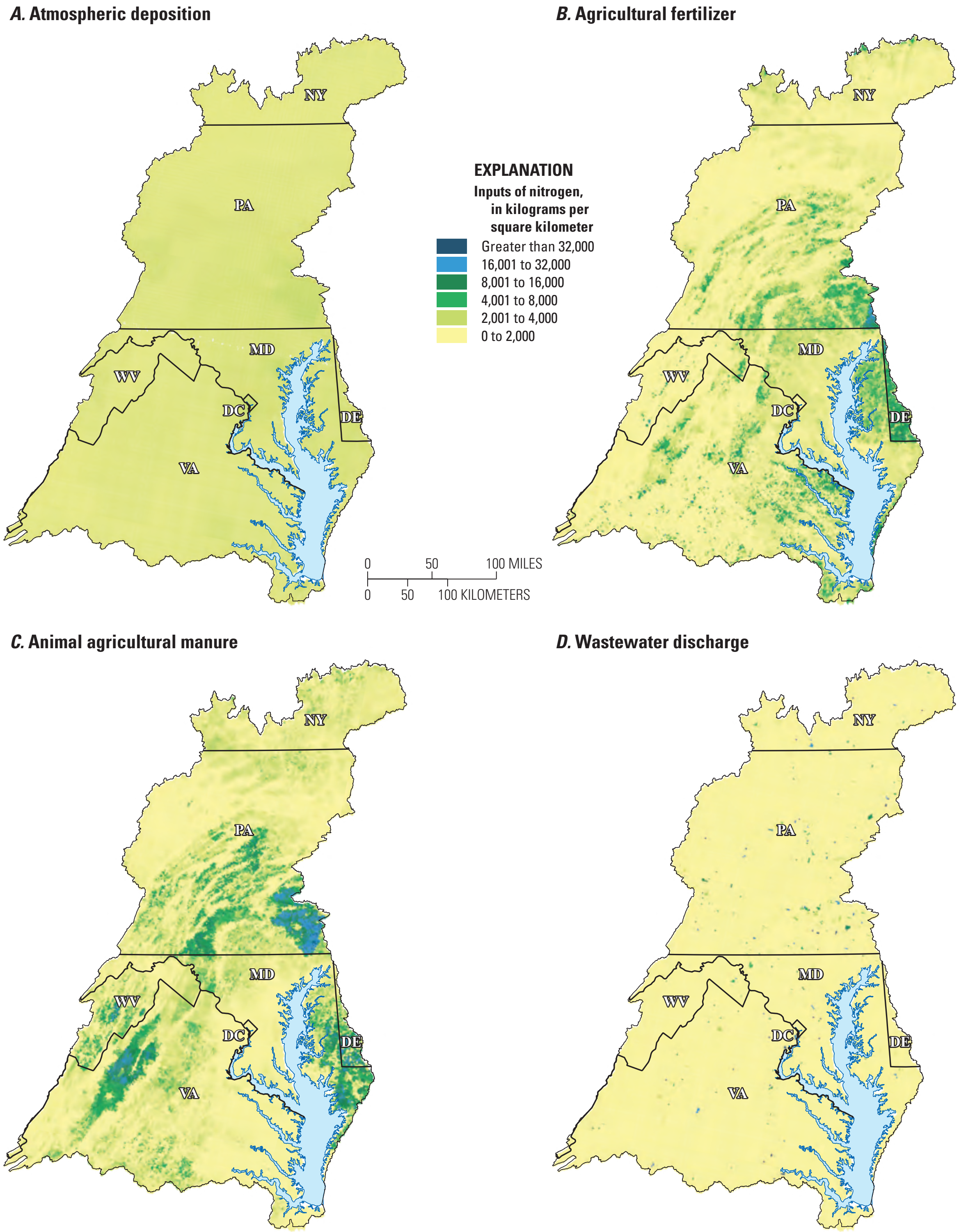


Figure NS.4. The spatial distribution of various nitrogen sources in individual catchments in the Chesapeake Bay watershed for 2010. ***A***, atmospheric deposition, ***B***, fertilizer used in agriculture, ***C***, manure from animal agriculture, and ***D***, wastewater discharge (Hopple and others, 2020, 2021). DC, Washington, D.C.; DE, Delaware; MD, Maryland; NY, New York; PA, Pennsylvania; VA, Virginia; WV, West Virginia.

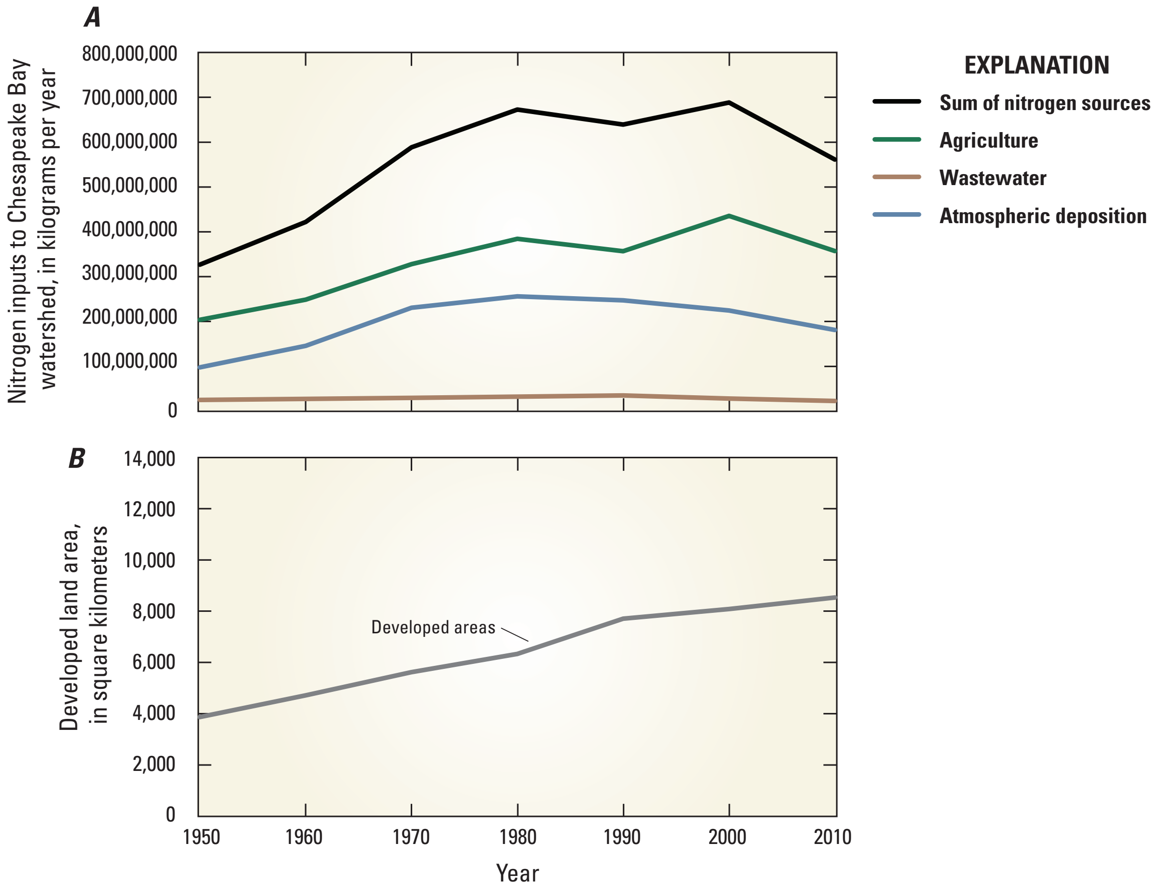


Figure NS.5. The annual nitrogen source inputs to the Chesapeake Bay watershed, 1950–2010, from ***A***, atmospheric deposition, agriculture (fertilizer and manure combined), and wastewater discharges; and ***B***, total developed land area in square kilometers (Hopple and others, 2020, 2021).

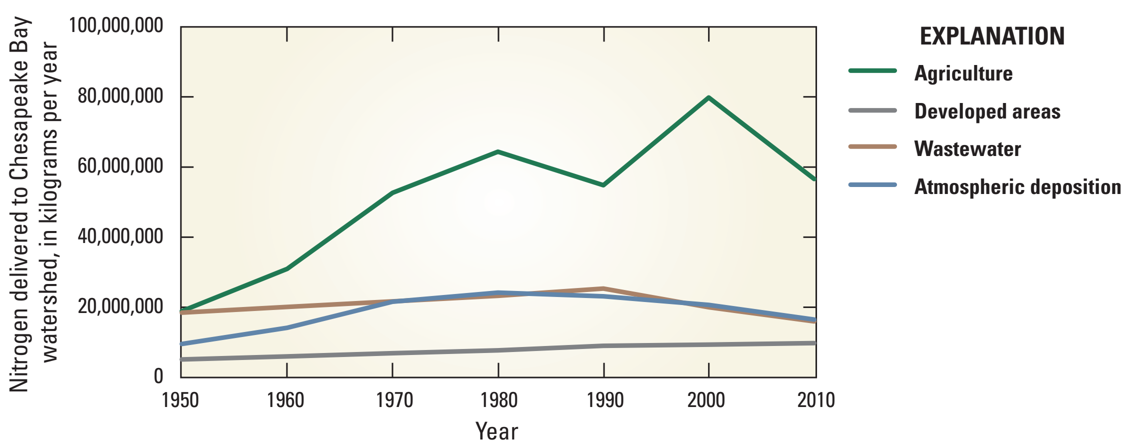


Figure NS.6. The annual exports of nitrogen from major watershed sources delivered to the Chesapeake Bay, 1950–2010 (Hopple and others, 2020, 2021).

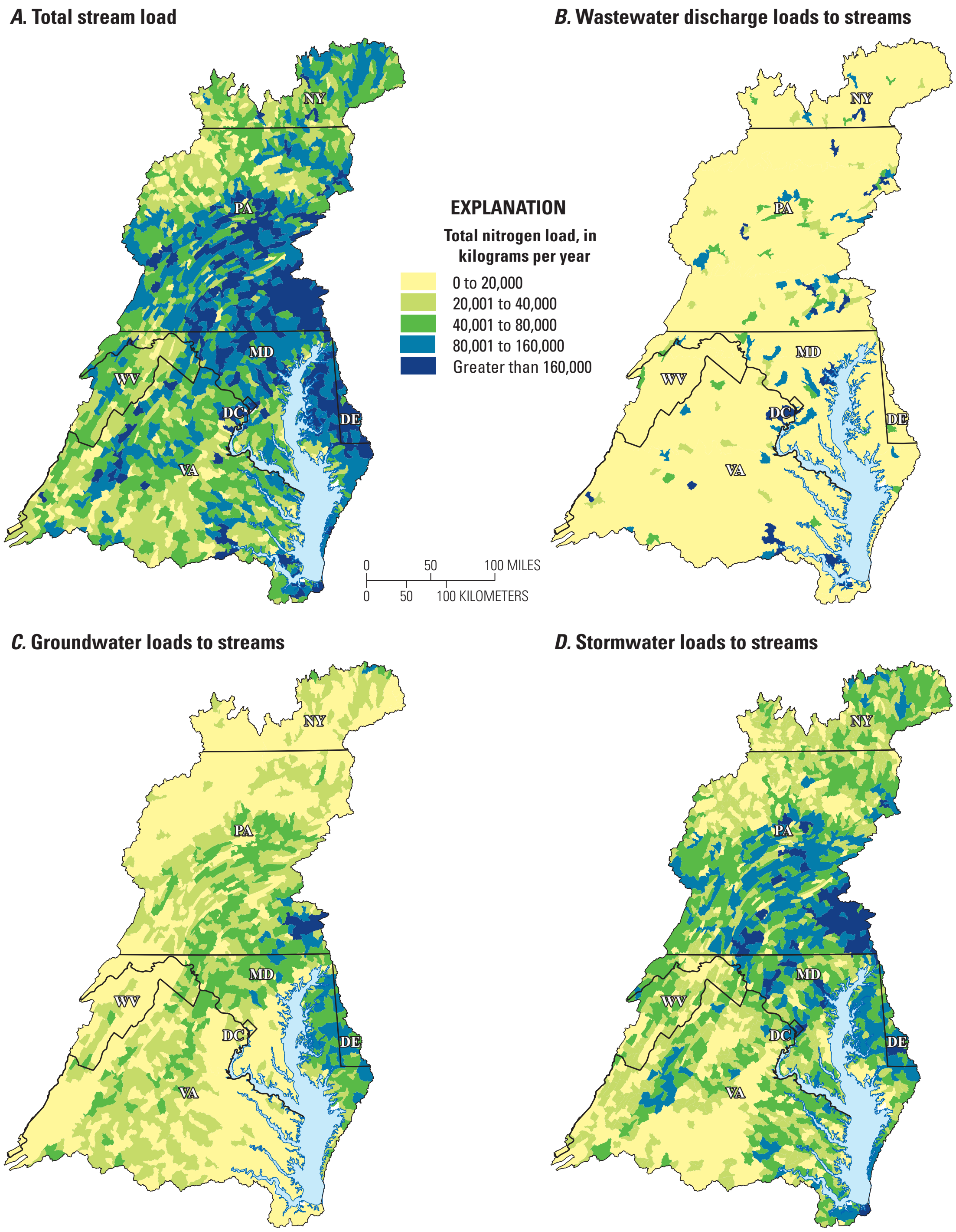


Figure NS.7. Annual loads of nitrogen to the local streams by water flow path, including ***A***, total stream ***B***, wastewater discharge (Ator and others, 2011), ***C***, groundwater (Terziotti and others, 2018) and ***D***, stormwater runoff calculated as a fraction of the total stream load that is not attributed to either groundwater discharge or wastewater discharge (Ator and others, 2011; Terziotti and others, 2018). Mapped to 12-digit hydrologic units (HU12) within the Chesapeake Bay watershed. The total stream loads of nitrogen were estimated independently from the groundwater loads of nitrogen to streams. DC, Washington, D.C.; DE, Delaware; MD, Maryland; NY, New York; PA, Pennsylvania; VA, Virginia; WV, West Virginia.

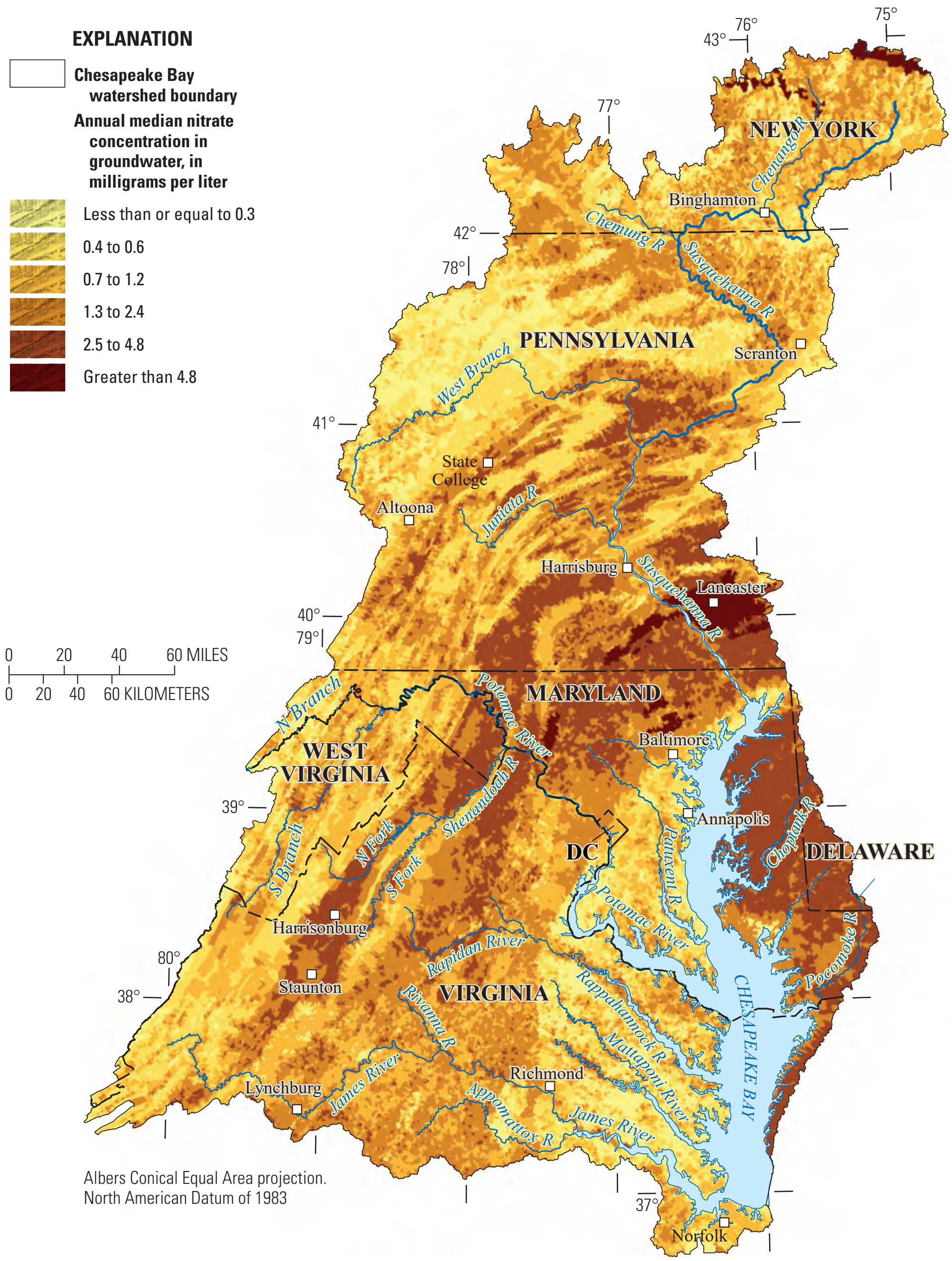


Figure NS.8. The spatial distribution of median nitrate concentrations in groundwater in 2002 (Terziotti and others, 2018).

Reactive (Bioavailable) Nitrogen

Nitrogen is an essential part of the biomolecules (for example, proteins and nucleic acids such as deoxyribonucleic acid [DNA]) in all living matter. Even though nitrogen gas composes 78 percent of the atmosphere, it is in a nonreactive, diatomic form (N_2) that cannot be utilized directly by animal and plant life (fig. NS.S1). Besides nitrogen fixation by lightning, which has enough energy to break the strong N_2 bonds, the major natural process for nitrogen to be converted (fixed) into a bioavailable (reactive) form is by specific species of bacteria that reside in the soil, water, and nodules of legumes. A series of different bacteria convert nitrogen gas to ammonia, then into ammonium (ammonification), and then again into nitrate (nitrification). Nitrate and ammonia are forms of nitrogen that can be readily used (assimilated) by plants for growth. Once taken up by bacteria or plants, the organic nitrogen becomes part of the food chain and may be consumed by living organisms. Nitrogen is later conveyed as waste or through decay, where it may be converted by bacteria again into nitrate (decomposition). If not recycled through the food chain, nitrate may be converted by bacteria to nitrogen gas (denitrification). Artificial processes (the Haber-Bosch process) have been developed for the industrial fixation of ammonia from nitrogen gas and are used in the manufacturing of fertilizers (Haber, 1920; Erisman and others, 2008). Fertilizers are commonly applied in the form of ammonium and then converted by bacteria to nitrate for plant use.

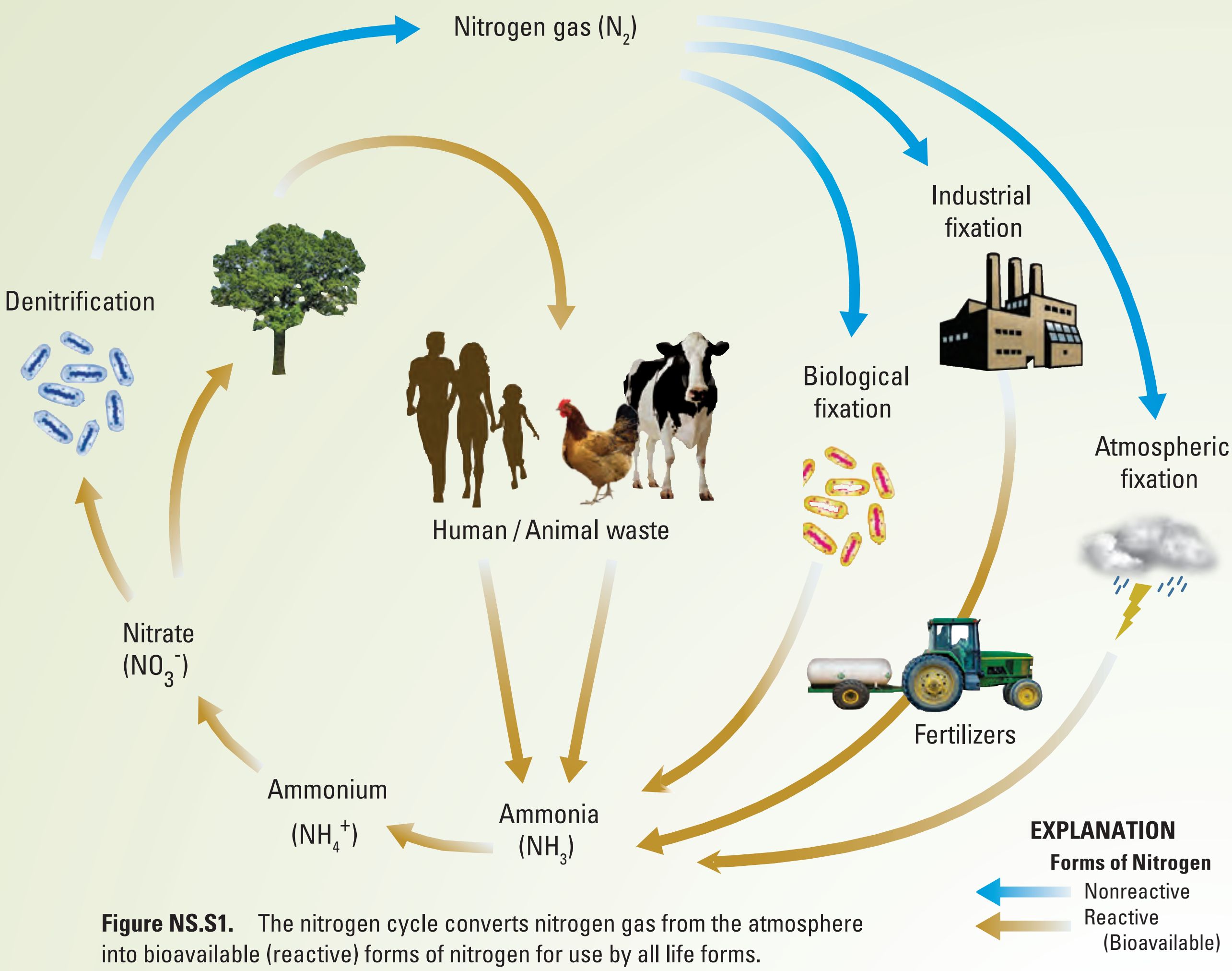


Figure NS.S1. The nitrogen cycle converts nitrogen gas from the atmosphere into bioavailable (reactive) forms of nitrogen for use by all life forms.

Globally, the principal bioavailable forms of nitrogen have changed over time (Galloway and others, 2004). Natural fixation of nitrogen on land has slightly decreased from 1860 and is expected to continue decreasing in the future owing to land use conversion from forest to agricultural and developed areas. Anthropogenic (originating from human activity) fixation of nitrogen has increased and is projected to continue to significantly increase to meet the world's food demand through industrial fixation and biological fixation from more cultivated land (Galloway and others, 2004).

Forms of Nitrogen

Nonreactive Nitrogen

Nitrogen gas (N_2) is the main component of the atmosphere, but is not available for use by organisms other than certain groups of specialized bacteria. It is nonreactive when dissolved in water.

Reactive (Bioavailable) Nitrogen

Water

Total reactive nitrogen in water (N_r) contains the sum of only the reactive forms of nitrogen that affect water quality. For the purposes of this report, *total reactive nitrogen in water* is the sum of inorganic (ammonia, nitrate, and nitrite) and organic nitrogen (dissolved and particulate). *Inorganic nitrogen* species are important because they are available to living organisms like plants and algae.

Ammonia (NH_3 and NH_4^+) for the purposes of this report, ammonia represents total ammonia and is a weak acid in water that exists in two forms as ammonium ion (NH_4^+) and to a lesser extent ammonia (NH_3). The relative amount of these two forms is dependent on the pH of the water. Ammonia is not stable, is easily oxidized to nitrate, and contributes little to the total nitrogen load in streams but in elevated amounts can be toxic to fish.

Nitrate (NO_3^-) is a strong acid in water, and thus exists in only the nitrate form. In the atmosphere, nitrate can exist as molecular nitric acid (HNO_3) in the gas phase or as nitrate ion (NO_3^-) in the water phase (for example, raindrops). Because nitrate is stable in aquatic environments, it is often a significant contributor to the total nitrogen loads of streams.

Nitrite (NO_2^-) is usually only detected in low amounts, and because it is not stable, it is readily oxidized to nitrate. Therefore, nitrite is often a very negligible contributor to the total nitrogen load in streams.
Organic nitrogen (N_{org}) compounds are chemicals that contain both nitrogen and carbon. Most of these compounds in the environment are intermediate chemicals formed during the decay of biomass. In water, organic nitrogen compounds can exist in both the dissolved and particulate forms. Organic nitrogen is often in the form of plant material or organic contaminants and can be a significant contributor to the total nitrogen loads in streams.

Atmosphere

Nitrogen dioxide (NO_2) is produced during combustion. NO_2 can react in the atmosphere to create ozone, which causes eye irritation and exacerbates respiratory conditions. It is also a component of atmospheric dry deposition.

Nitric oxide (NO) can transform into nitrogen dioxide that can produce nitric acid, which contributes to acidic precipitation.

Nitrous oxide (N_2O) is a product of incomplete denitrification in soil and water and is a potent greenhouse gas.

Animal Waste/Fertilizer

Urea (CH_4N_2O) is a nitrogen-containing organic compound that is often used as a nitrogen fertilizer.

Historical Setting of the Chesapeake Bay Watershed

By Richard H. Coupe[1]

In 1608, Captain John Smith described the Potomac as "… frequented by otters, beavers, martens, and sables. Neither better fish, more plenty, nor more variety for small fish had any of us ever seen in a place." (Interstate Commission on the Potomac River Basin, 2019)

Prior to the successful establishment of the Jamestown settlement by Europeans in 1607 and successful exploration of the Chesapeake Bay by John Smith 5 years later (fig. H.1), Native Americans had been living in the watershed for about 10,000 years. The name chosen for the largest estuary in North America is so aptly derived from the Algonquin word K'che-se-piak meaning "land along the big river" (Orth and others, 2017). Much of the watershed was covered in old growth forest with a continuous, closed canopy shading streams and protecting the soil from runoff. The hydrology of most streams was dominated by beavers who built dams that slowed and retained water, increased infiltration to groundwater, stored sediments, and created wetland habitat (Brush, 2009). By the cutting of trees and shrubs for food and dam building, beavers had a substantial impact on the structure, stability, and productivity of streams and their riparian zones (Naiman and others, 1988). Beavers were later trapped for their furs to outfit European fashion and were quickly eliminated from the landscape, extensively altering a natural wetland mosaic riverine system to the more free-flowing streams seen today.

Historical photograph, courtesy of the Chesapeake Bay Maritime Museum, used with permission.

[1]U.S. Geological Survey.

Figure H1. Published in 1671 by John Ogilby, an early colonial map of the Chesapeake Bay region. Map courtesy of the Maryland State Archives Special Collections (William T. Snyder Map Collection).

"Oysters lay as thick as stones." – Captain John Smith

Native Americans managed parts of the landscape with fire for agriculture and to drive game, which had less impact than modern agriculture on the overall ecology of the Chesapeake Bay and its watershed. The Bay was a pristine estuarine ecosystem dominated by bottom-dwelling organisms, teeming with submerged aquatic vegetation, oysters, and hundreds of native species of birds and fish. Oyster beds covered extensive bottom areas throughout the mid- and lower-Chesapeake Bay and in some places broke the water surface during low tide. These beds could pose a threat to ships. Oysters obtain their food by filtering organic material (such as algae) suspended in the water through their gills. It is estimated that prior to 1870, oysters were so numerous that they could filter the entire Bay in a few days, but by 1988 the oyster stocks were so depleted that it would take close to a year to filter the Bay (Newell, 1988).

European Arrival and Ecologic Change: 1607–1750

The impact of colonial settlements on the ecology of the Chesapeake Bay manifested slowly and was only localized during the early 17th century. The permanent European settlements brought two major changes not seen before in the Chesapeake Bay watershed: (1) a land ethic centered on private property and ownership of natural resources for one's benefit, and (2) access to sell these resources to a commercial market. "To tie the Chesapeake Bay to external markets was to change its ecological dynamics" (Cronon, 2001).

During this time the colonists used slash and burn agriculture similar to the Native Americans (Miller, 1986). This method provided protection from soil erosion and helped the soil quality. Many early colonists grew corn for their domestic use, whereas tobacco was the main commercial crop. After several growing seasons of use, the fields were left fallow for many years to recover their fertility.

Although there was a minimal effect on the ecology of the Bay from the early colonists, the new processes that would eventually impact the water quality of the Bay were already set in motion. The introduction of domestic grazing animals such as horses, pigs, cows, and sheep that roamed freely on farms, compacted the soil, fed on vegetation, and degraded riverbanks brought additional pressure to the cleared land. The arrival of African slaves in Virginia and Maryland around 1700 increased the overall population and changed the land to labor ratio, allowing more forest clearing for agriculture (Silver, 2001). The introduction of nonnative plant species and animals caused irreversible changes to the ecology of the watershed, as nonnative species outcompeted endemic species for resources. In addition, the economy was focused on exports such as wood to the Caribbean and tobacco to Europe, which caused a change in the ecological balance as natural resources were now exported out of the watershed.

America is Born and Transitions from a Colony to an Independent Nation: 1750–1820

Early settlement patterns in the Chesapeake Bay watershed created port cities such as Washington, Baltimore, and Richmond, that were often developed at the furthest extent that sea vessels could travel, known as the "Fall Line" or where the edge of the Coastal Plain meets rocky upland terrain (figs. ES.4*A* and ES.5). During the late 17th and early 18th centuries, the colonial economy was focused on exports, but there were growing internal markets as the population increased. People began to move further inland away from the Bay and the first wave of natural forest clearcutting, wetland draining, and crop planting on hilly lands followed. By 1750, plantations for tobacco in southern states began to spread and slaves were almost 44 and 31 percent of the populations of Virginia and Maryland, respectively. In the northern part of the watershed, a large wave of immigrants moved into the watershed during this time, clearing forests for agricultural land and for building houses. Additionally, trees were clear cut in upper parts of the watershed in places like Pennsylvania and used for charcoal to operate iron furnaces and make way for agricultural and industrial progress (fig. H.2; DeCoster, 1995).

Figure H.2. An 1856 painting by George Inness called "The Lackawanna Valley" depicts a northern region of the Chesapeake Bay watershed, where a Pennsylvania forest landscape was profoundly changed in a short period of time for industrial progress. Image courtesy of National Gallery of Art.

Early agricultural remnants can be seen in the record of bed sediment in the Chesapeake Bay starting about 1760. This "agricultural horizon" is quantified by the increased amount of ragweed pollen in the sediment layers over time and corresponds to the increased percentage of land cleared for agriculture (Cooper and Brush, 1991). During the late 18th century, agriculture changed to farming techniques imported from Europe, with a more clean and plowed landscape, and moved away from the Native American style of rotation of fallow fields. These new farming methods originated from areas of western Europe with a climate of gentle rainfall over long periods of time but were less suited to the more intense storms of short duration typical of the continental United States (Earle and Hoffman, 2001). Additionally, immigrant labor became more abundant and grain prices rose in response to European wars. Together, these changes created an increased demand for agricultural goods (grain), resulting in widespread disturbances of the soil that washed extensive sediment (eroded soil) to the Bay during this period. Runoff and sediment delivery increased in the late 1700s, as compared to precolonial times, and filled in many navigation channels and colonial harbors such as Port Tobacco, Piscataway, and Joppa Town, Maryland (Miller, 1986).

During this period, the human-induced environmental effects of forest clearing for agriculture became apparent (Miller, 1986). The hydrology was altered as a result of these new changes to the landscape. Trees were often cut from the banks and slopes adjacent to streams, which eliminated the canopy and led to warmer water and increased evaporation. Absent of trees to control and moderate sunlight, the newly cut fields were subject to extreme variations in temperature. Exposed soils produced excess runoff and resulted in less water percolating into the ground. The reduction in the forest canopy also caused higher streamflow in the spring, but because there was less recharge, flow became lower during the summer. It is estimated that peak streamflows have increased by 25–30 percent and low flows have decreased by 10–15 percent since the arrival of the Europeans (Biggs, 1981). This may have caused a shift in the salinity of the estuary, which would have affected the range of salt-tolerant plants and animals, forcing them further south in the estuary.

Economic Boom Brings Environmental Impacts: 1820–1900

Most of the 19th century was characterized by the massive growth of internal markets, developed areas, and transportation corridors. There was an explosion in technological innovation and industrial manufacturing. Railroads opened up fertile land in the Midwest and farmers in the Chesapeake Bay watershed had to diversify to compete. The highest percentage of agricultural land (cleared land) and the lowest percentage of forested land occurred throughout the watershed during this period (fig. OV.1). This period also brought the maximum harvests of fish, oysters, and birds from the Chesapeake Bay. Some bird species, such as the Passenger Pigeon, became extinct as a result of overhunting and habitat loss (Guiry and others, 2020).

Serious environmental issues began to develop in areas that were settled first. Lands in Cumberland County, Virginia, in 1838 were described as "… naturally the best in the world have become worn and exhausted by the culture of tobacco. The bitter weed has laid the forest low and left us with nothing but galls and gullies and dwarf pine. Our ridges have become so barren, that they do not afford cover for the partridges and they have followed the soil down branches and creeks, hovering in the flats, Virginia's forest has been swept away and her great men of genius and worth together with the hard cultivators of the soils, the bone and sinew of the land, have, by thousands and tens of thousands, been driven out of the state in search of better lands" (Trimble, 1974).

This period also marked a time when increased levels of deforestation and natural resource extraction occurred within the Chesapeake Bay watershed. Massive deforestation was occurring in the uplands of the watershed in places like Pennsylvania, where entire forests were used to fuel iron furnaces. In 1845, there were 145 charcoal furnaces in Pennsylvania and 20,000–35,000 acres of trees were required to sustain each furnace

Baltimore steam packet company. Courtesy of the Chesapeake Bay Maritime Museum, used with permission.

(DeCoster, 1995). In addition, the Chesapeake Bay's abundant natural resources such as chrome, iron, and coal, increased in importance locally and internationally. As an example, Isaac Tyson, a Quaker and French-trained geologist and chemist from Maryland, held a virtual monopoly on the world's supply of chrome in the middle 19th century.

Pioneering efforts in conservation also began during this time period to prevent further destruction of the environment and enhance its economic value. Gifford Pinchot, born in 1865, served as the first chief of the U.S. Forest Service and later as Governor of Pennsylvania and promoted the scientific management of forests for the benefit of mankind. Edmund Ruffin, a Virginia planter, known as the father of soil science in the United States (Matthew, 1988), began his pioneering work on the restoration of tobacco fields that were depleted of nutrients after 100 years or more of tobacco monoculture. Guano, the accumulated excrement of seabirds and bats, was imported from South America and used as fertilizer to help counteract the decline of crop yields on the exhausted fields.

By the 19th century, oyster population declines were observed, and this period was the start of oyster regulation and conservation (Schulte, 2017). As early as 1810, Virginia banned dredging for oysters, but the protection of oysters proved to be an elusive goal. With the improvements in harvesting technology, roads, and food preservation methods, Chesapeake Bay oysters became an important source of food and wealth. It also helped create a frontier atmosphere on the Chesapeake Bay, which saw the Governor of Virginia hunting oyster pirates from an armed tugboat, the creation of a so-called Oyster Navy in Maryland, and pitched battles between foes to determine who would have access to the oyster beds. Almost 50,000 metric tons of oysters were harvested from the Chesapeake Bay in 1884, but just 5 years later, the harvest had declined by almost a third. The oyster harvest continued to decline precipitously thereafter. In the 1950s, oysters were subjected to several insidious and debilitating diseases that further decreased their populations (Kennedy and Mountford, 2001).

Increasing Population Pressure on Aquatic Ecosystems: 1900–1950

The early to middle 20th century is notable for the continued decline of the oyster in the Chesapeake Bay and a precipitous decline in submerged aquatic vegetation (SAV). These plants provide many ecological services such as removing excess nutrients, trapping sediments, providing hiding places and breeding habitats for fish and invertebrates, and serving as food for waterfowl. Estimates of the areal coverage of SAV in the Chesapeake Bay put the total at more than 772 square miles (2,000 square kilometers) when Europeans first arrived (Lynch, 2001). SAV was still widespread in the Bay in 1937, but increased nutrients from wastewater, animal manure, and agricultural runoff from developed land led to a shift from rooted SAV to free-floating algae (phytoplankton) as the Bay's dominant source of aquatic photosynthesis, which caused a loss of water clarity. Light penetration was so reduced that many species of SAV were unable to survive. The reduction in SAV was first reported in the 1930s and 1940s on the Potomac estuary below Washington, D.C. (Lynch, 2001). By 1960, SAV loss was evident in the main stem of the Chesapeake Bay. In 1991, areal coverage of SAV was just 10 percent of the original estimates (Lynch, 2001).

There were many notable changes in the Chesapeake Bay watershed during the early and middle 20th century—the same changes that occurred at many locations in the United States. There was a dramatic rise in population especially after World War II (WWII) and an associated rise in the consumption of fossil fuels. With the development of highway infrastructure, communities were no longer clustered along the railroad networks, but more people moved into rural areas and were able to commute into the cities for work. The boundaries between cities and countryside became blurred, and the population of the watershed doubled between 1900 and 1950 (fig. OV.1; Kemp and others, 2005).

Dams had been constructed on the rivers and streams of the Chesapeake Bay watershed—almost from the first landings of the Europeans in the 17th century—and many were used to power grist mills or sawmills. Others were used for navigation, for ferries to cross or to keep water in the canals used for transportation. Dams altered the hydrology of the system and blocked the access of anadromous fish to their spawning habitat; for example, American shad were found at Binghamton, New York some 300 miles from the mouth of the Susquehanna River, but by 1899 American shad could only be found about 80 miles up the river. The early years of the 20th century saw the construction of large dams in the upper watershed and on the main stem of the Susquehanna (such as the Conowingo Dam) for hydroelectric power and flood control. Many of these dams are still in operation.

Legislation was passed to address increasing concerns over the impact of humans on the Bay aquatic ecosystems. In 1913, the Federal Migratory Bird Law gave the Federal government authority over hunting of migratory birds and the first migratory bird hunting regulations were adopted (U.S. Fish and Wildlife Service, 2019). Federal legislation created Soil and Water Conservation Districts to oversee and implement mitigation strategies to control soil erosion (Natural Resources Conservation Service, 2019). Between the mid-1930s and 1950, 20 Districts, which covered 95 percent of the Potomac River Basin, were established (see sidebar– Changes in the Potomac River in the 20th Century). In addition, the Federal Water Pollution Control Act of 1948 was enacted to address water pollution (U.S. Environmental Protection Agency, 2019a). This was the first major U.S. law to protect water supplies and to decrease the impact of wastewater.

By the end of the 1940s, more than 300 years after Europeans landed at Jamestown, the landscape of the Chesapeake Bay watershed was a shadow of its former self; the landscape had been substantially altered and every ecosystem service had been affected. Although forest regeneration has occurred throughout much of the watershed and the landscape has begun to heal from the early extraction of natural resources, the overall ecosystem has been altered. The Chesapeake Bay and its watershed continue to provide value and services, but the services are more vulnerable to degradation. Natural processes that once buffered the system from disruption and had developed over millennia (forests, SAVs, oysters, and so forth) were changed over decades and no longer function as they once did to process nitrogen. The main source of available nitrogen before the arrival of Europeans was through natural biological fixation, which was in balance with the environment and returned to the atmosphere through denitrification (Richter and Markewitz, 2001; Brush, 2009). The amount of nitrogen imported to the Bay watershed and exported to the Bay itself increased as the population of humans and animals increased (see chap. 2). With the availability and increasing use of synthetic fertilizers after WWII, the nitrogen load to the Bay has increased substantially. Altogether, the Chesapeake Bay and its ecosystem have been adversely affected by many factors, not the least of which is excess nitrogen.

Changes in the Potomac River in the 20th Century

In 1920, there were about equal numbers of people in rural and developed areas in the Potomac River Basin, but by 1943 the population in developed areas far outnumbered that in rural areas and the value added to the economy from manufacturing far exceeded that from agriculture. In the mid-1940s, there were 53 communities of more than 500 people with no public sewers. There were another 39 communities that had public sewers, but that discharged raw waste directly to the river (Interstate Commission on the Potomac River Basin, 1945, 2010). Concerns over the quality of water in the Potomac River led to the formation of the Interstate Commission on the Potomac River Basin (ICPRB) in 1940. The ICPRB determined the major contributors to the impairment of the Potomac River, included agriculture (such as soil erosion), industry (mining, paper mills, and so forth), and wastewater (Interstate Commission on the Potomac River Basin, 1945, 2010).

The effluent from wastewater had a detrimental effect on the water quality of the Potomac River. For example, a middle 19th century wastewater treatment plant on the Anacostia River, which served 100,000 people, performed only primary treatment on about half of its flow, the rest was discharged as raw sewage. Nearby, the City of Alexandria, with a population of 50,000, discharged all of its raw sewage into the Potomac River. About one-third of the sewage from Arlington County bypassed the wastewater treatment plant and discharged directly into the Potomac River or one of its tributaries (Interstate Commission on the Potomac River Basin, 1945, 2010).

Similar to other facilities at the time, when the Blue Plains Advanced wastewater treatment plant opened in 1938, it offered only primary treatment (settling out of solids). It treated 100 million gallons per day (MGD) (380 million liters per day) and served about 650,000 people. The plant's capacity was quickly reached as the population grew and primary treatment was found to be inadequate. The plant was expanded in 1959 to 240 million gallons per day (910 million liters per day) and provided secondary treatment (removal of soluble matter) to the wastewater. By 1969, the Blue Plains facility had reached capacity and from 1970 to 1983 underwent expansion to become an advanced treatment plant with increased capacity. A new nitrification-denitrification system was installed between 1998 and 2001. The treatment levels were greatly improved and helped to restore the health of the Potomac River (Ruhl and Rybicki, 2010). The Blue Plains Advanced Wastewater Treatment Plant has become one of the largest advanced wastewater treatment plants in the world with the ability to treat more than 4-hour peak flows of 555 MGD through complete treatment and 225 MGD through wet weather treatment. It serves more than 2 million customers in the region (see chap. 8 sidebar–Blue Plains Advanced Wastewater Treatment Plant; DC Water, 2019).

Changes in Nitrogen, Water Quality, and Management

Chapter 1

By John W. Clune[1] and
Gary W. Shenk[2]

West Branch Susquehanna River in Clinton County, Pennsylvania.
Courtesy of the Chesapeake Bay Program, used with permission.

Past Increases of Nitrogen in the Environment Cause Ecological and Economic Impacts that Prompt Regulatory and Management Action

In the early (pre-colonial) environment of the Chesapeake Bay watershed, the main source of nitrogen available for plants and animals was converted (biological fixation) from the atmosphere by bacteria, recycled, and released back into the environment in forms that could be used by other living organisms. With the transition to an agrarian society, cultivation began to introduce modest amounts of nitrogen from biological fixation by legumes (Galloway and others, 2004). As the nation developed further in the 19th and early 20th century (see Historical Setting), deforestation and a rising population contributed to an accelerated loss of soil, phosphorus, and nitrogen to waterways from untreated wastewater and agricultural erosion (fig. 1.1). A substantial increase in the input of nitrogen into terrestrial and aquatic environments came about in the middle 20th century (fig. OV.1), when an industrial process to transform nitrogen gas in the atmosphere to ammonia was developed and provided an abundant supply of chemical fertilizer to be used on crops to meet the growing population's demand for food (Haber, 1920; Erisman and others, 2008). The combustion of fossil fuels in support of the country's energy demands began emitting nitrogen into the atmosphere. Additionally, the post-World War II era brought extensive suburban development that increased the volume of wastewater and introduced fertilizer use to lawns.

[1]U.S. Geological Survey.

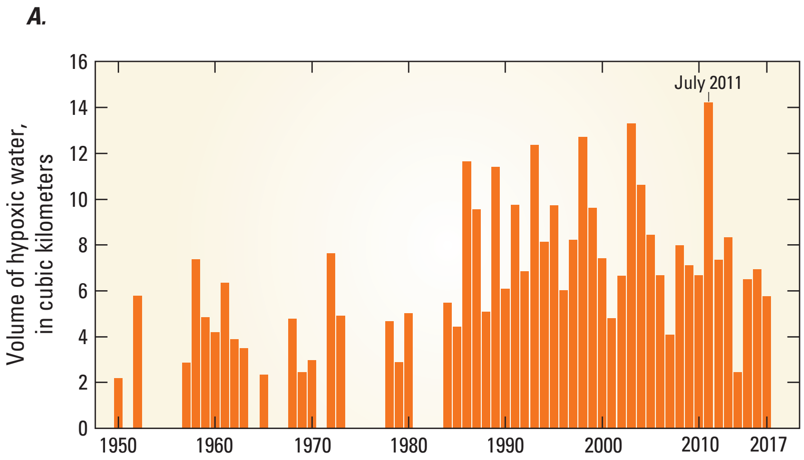
[2]U.S. Geological Survey - Chesapeake Bay Program Office.

Figure 1.1. The Chesapeake Bay watershed was largely undisturbed until early extraction of natural resources, cultivation of arable land, and modern development introduced an excess of nutrients like nitrogen into the waterways. ***A***, Painting of the Juniata River by Thomas Moran (National Gallery of Art, used with permission), ***B***, photograph of deforestation in Pine Creek watershed, Pennsylvania, (WikiCommons/Public domain), and ***C***, photograph of suburbs and agriculture (courtesy of the Chesapeake Bay Program, used with permission).

By the 1950s, nitrogen exported to streams had taken a toll on the ecologic and economic aspects of the Chesapeake Bay and surrounding watershed. Excessive nutrients entering waterways stimulate algae growth, and during algal decay the respiration of decomposing bacteria depletes the oxygen that aquatic life depends on (Wetzel, 2001). This increased productivity from nutrient enrichment (eutrophication) resulted in the expansion of low oxygen (hypoxia) conditions, also known as dead zones, in the Chesapeake Bay (fig. 1.2; Hagy and others, 2004; Murphy and others, 2011; Testa and others, 2017) and contributed to a widespread decline of submerged aquatic vegetation (SAV; Gurbisz and Kemp, 2014; Lefcheck and others, 2019). Hypoxic conditions provoke stress and mortality among sensitive species that are not able to move away from areas of depleted oxygen and shift the overall community structure within the underwater ecosystem (National Research Council, 2000; Breitburg and others, 2015). The fast-growing free-floating algae in the water (phytoplankton) reduce water clarity and deprive slower growing SAV, which are an essential habitat for juvenile fish and crabs, of available light (Lipcius and others, 2005; Orth and others, 2017). Chronic nutrient rich conditions have also caused harmful algal blooms in the Chesapeake Bay that discolor the water (for example, red tides and blue-green algae) and produce potent toxins that can contaminate drinking water and fish used for human consumption (Glibert and others, 2001).

Poor water quality owing to eutrophic conditions also has economic impacts on property values, commercial fishing, recreation, tourism, and other related regional industries that depend on the health of the Bay (Morgan and Owens, 2001). Costs to society include increased maintenance of infrastructure and additional investments in water treatment. For example, if high nitrate in drinking water is not properly treated before it is consumed, it can restrict oxygen transport in the bloodstream and can be fatal for infants (Bouchard and others, 1992). The Chesapeake Bay, already declining ecologically and economically, received a catastrophic blow from tropical storm Agnes in 1972, which delivered the maximum amount of streamflow ever recorded to the Bay together with large amounts of nitrogen, phosphorus, and sediment. The ecosystem and economy required many years to recover from that storm event (fig. 1.3; Chesapeake Research Consortium, 1976).

During the middle to late 20th century, pioneering legislation, watershed management efforts, and grass roots movements were initiated to combat the increasing observable effects of excessive nutrients in the Chesapeake



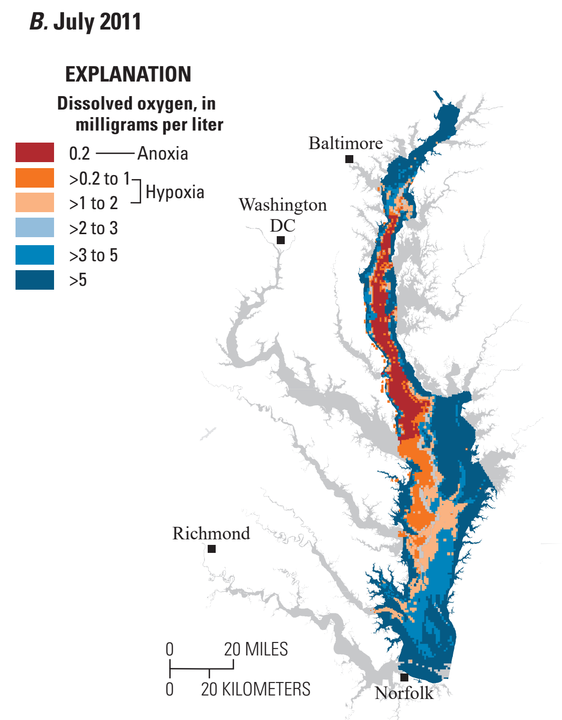


Figure 1.2. Excessive nitrogen loads have been a leading contributor to the ***A,*** volume of hypoxic water (dissolved oxygen <2 milligrams per liter [mg/L]) over time (Hagy and others, 2004; Murphy and others, 2011; Testa and others, 2018), and ***B***, the extent of low oxygen conditions, as seen in the Chesapeake Bay in July 2011 (Testa and others, 2017).

Figure 1.3. The slow ecologic and economic decline of the Chesapeake Bay received a catastrophic blow from tropical storm Agnes in 1972, as shown in Richmond, Va. Photograph from the Library of Virginia, used with permission.

Bay watershed (fig. 1.4). Conservation efforts were started in the forestry and agricultural sectors to reduce the loss of soil and nutrients into waterways (Bennett, 1939). Organizations and partnerships like the Chesapeake Bay Foundation, the Alliance for the Chesapeake Bay, and the Susquehanna River Basin Commission were formed to coordinate efforts to better address the environmental problems in and around the Bay. In 1972, Congress enacted the Clean Water Act, which provided a framework for assessment and regulation "to restore and maintain the chemical, physical, and biological integrity of the Nation's water" (Clean Water Act, 33 U.S.C. § 1251 et seq.; 40 C.F.R. §§ 104.1). Regulation and assessment were based on the societal use and value of a waterbody (for example, public water supplies, propagation of fish and wildlife, recreation, agriculture, industry, and navigation). Waterbodies like the Chesapeake Bay that did not meet the water-quality standards for their designated use were placed on the U.S. Environmental Protection Agency (EPA) 303d list and a pollution budget was established outlining the maximum amount of the pollutant (for example, nitrogen) that a waterbody can receive (see sidebar–Largest Total Maximum Daily Load in the Nation). During this period, some reductions from direct nitrogen loading to streams came with improvements in sewage treatment.

Largest Total Maximum Daily Load in the Nation

The Clean Water Act requires that a list of waterways is assessed in each state every 2 years to evaluate compliance with water-quality standards. If the waterways are impaired, regulatory pollution limits or TMDLs are developed. In 2010, the largest and most complex TMDL in the Nation was developed for the Chesapeake Bay for nitrogen, phosphorus, and sediment. These pollution allocations were further divided by major river basins and states (fig. 1.S1). Management practices expected to eventually reduce nitrogen by 25 percent are set to be implemented in the Bay watershed by 2025 (Linker and others, 2013a).

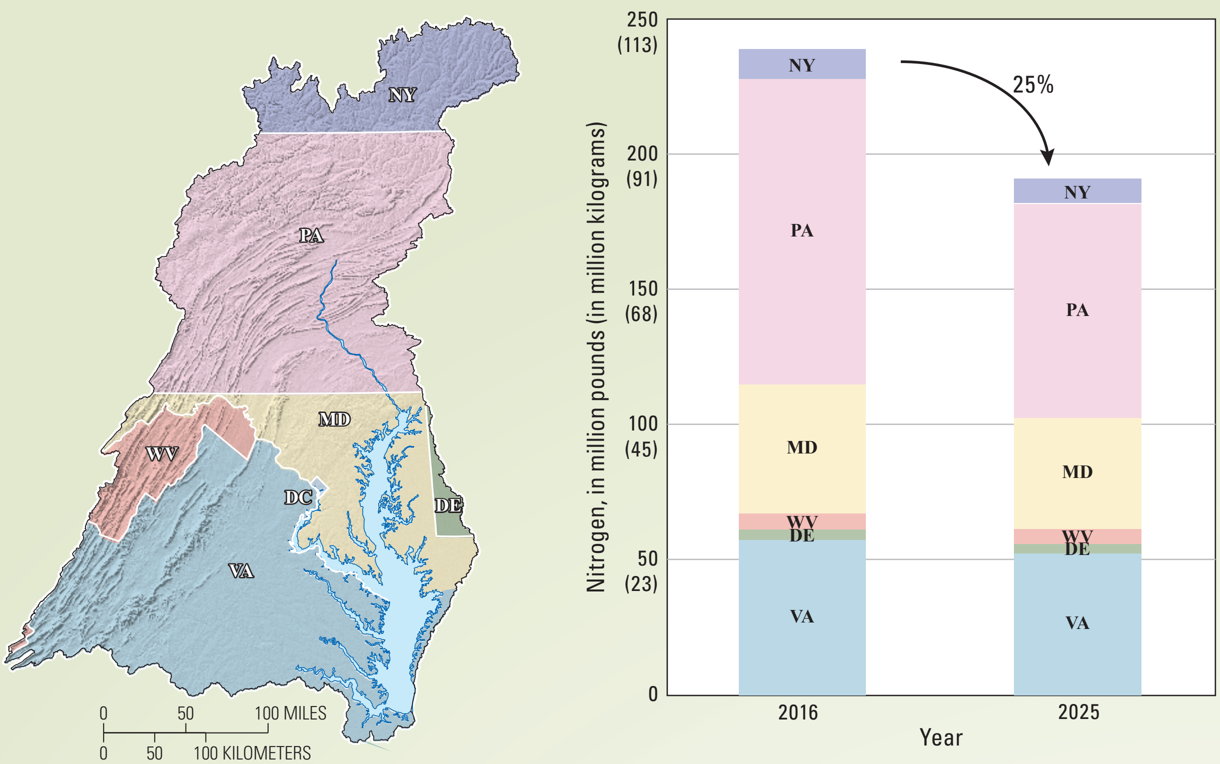


Figure 1.S1. Statewide management practices expected to eventually reduce nitrogen by 25 percent are set to be implemented in the Bay watershed by 2025 (Chesapeake Bay Program, 2017). DC, Washington, D.C.; DE, Delaware; MD, Maryland; NY, New York; PA, Pennsylvania; VA, Virginia; WV, West Virginia.

Watershed implementation plans specify how the jurisdictions in the Bay watershed will reach pollution allocations (U.S. Environmental Protection Agency, 2019a). By the midway point of 2017, management actions were to have been in place to achieve 60 percent of the necessary pollution reductions, with all pollution measures in place by 2025. Each state will need to implement management strategies that prioritize resources and target sources that are delivering the highest loads to the Bay (fig. 1.S2)

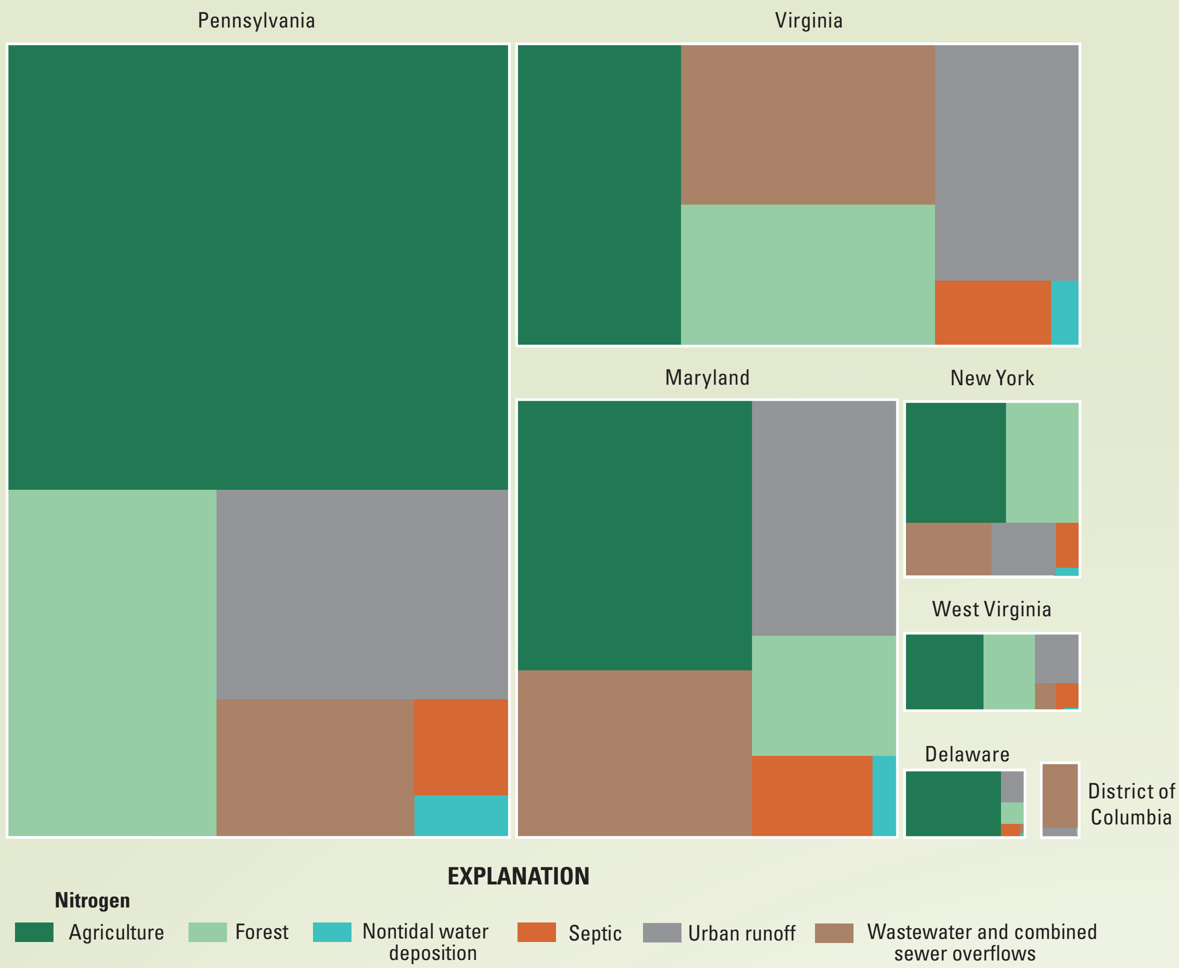


Figure 1.S2. Proportional sources of nitrogen load delivered to the Chesapeake Bay by state (Chesapeake Bay Program, 2017).

Starting in the mid-1980s, nationwide government programs such as the U.S. Department of Agriculture Conservation Reserve Program provided further incentives to the agricultural community to adopt conservation initiatives in order to improve water quality (U.S. Department of Agriculture, 2019a). Regionally, the Chesapeake Bay Commission was formed to coordinate policy across state lines, a series of Chesapeake Bay Agreements were signed with the goal to lower nutrient loads, and the Chesapeake Bay Program was established to coordinate restoration efforts across the watershed (U.S. Environmental Protection Agency, 2010b). Over the past few decades, the monitoring and modeling of water quality has been expanded (see sidebar–The Chesapeake Bay Program Watershed Model) to provide a better understanding of the fate and transport of nitrogen in the watershed (Ator and others, 2011; Chesapeake Bay Program, 2017). For instance, recent water-quality trends show that nitrogen concentrations have decreased at many

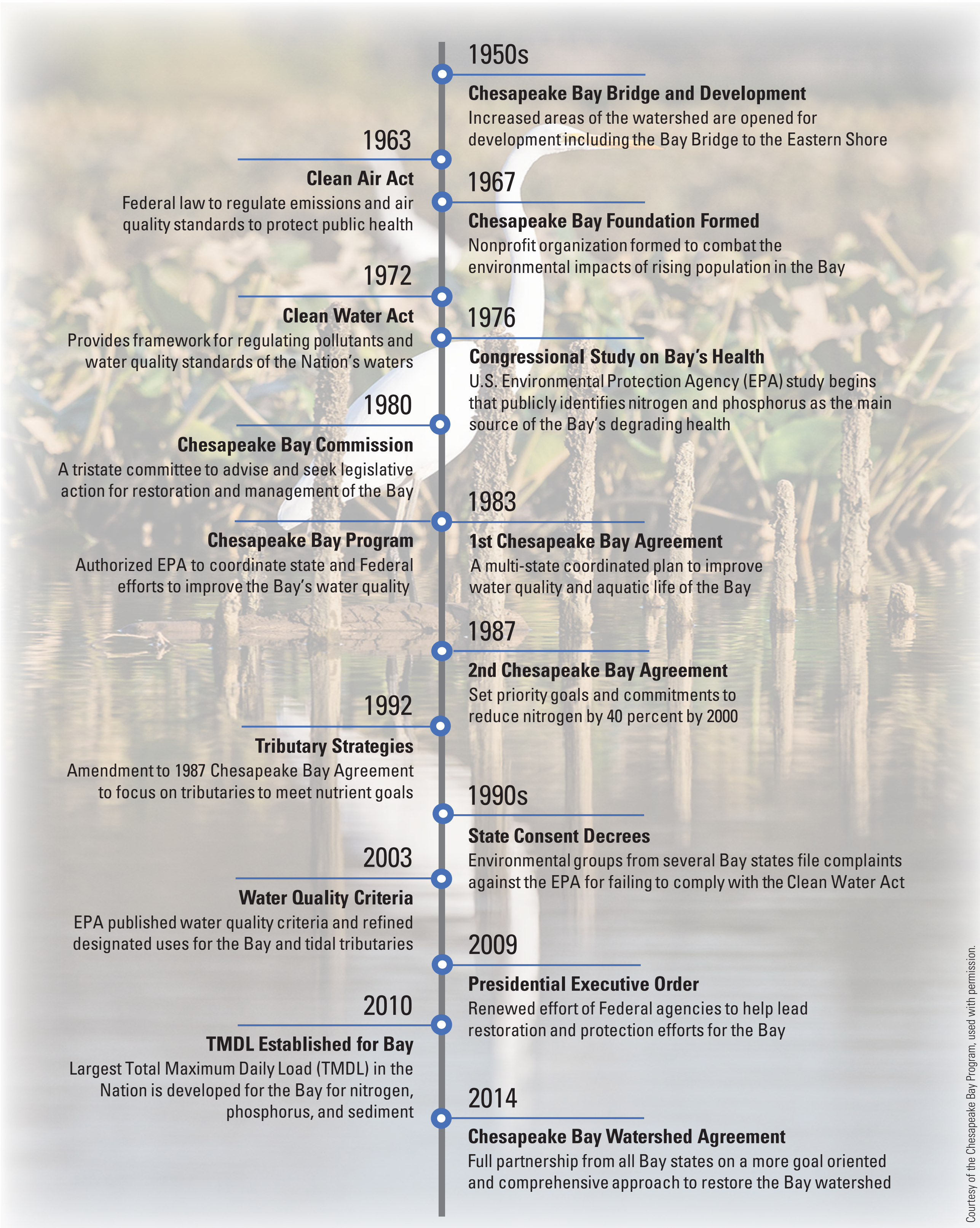


Figure 1.4. Timeline of selected regulatory, policy, and management milestones affecting nitrogen in the Chesapeake Bay watershed since 1950.

The Chesapeake Bay Program Watershed Model

The Chesapeake Bay Program partnership uses computer models of the Chesapeake Bay watershed and estuarine system to provide decision makers with expected environmental responses to management actions that affect streamflow, nitrogen, phosphorus, and sediment. A series of linked watershed and estuarine models, periodically updated, has been used in decision making since the late 1980s, including in the development of the 2010 TMDL (Linker and others, 2013a). The most recent version of the watershed model, known as Phase 6, was used in the 2017 Midpoint Assessment of the TMDL undertaken by the Chesapeake Bay Program (U.S. Environmental Protection Agency, 2017).

The Phase 6 model consists of two parallel models: a time-averaged model and a dynamic model, which is constrained to match the time-averaged model over the long term. The time-averaged model, known as the Chesapeake Assessment Scenario Tool (CAST), is used as the primary model for decision making. Stakeholders and other users access CAST through a dynamic web interface (Chesapeake Bay Program, 2017). By using CAST, stakeholders and other users can quickly run user-defined scenarios using the official model used for TMDL calculations. The dynamic model is used in calibration of the Phase 6 system, to translate CAST scenarios into hourly loads of nutrients and sediment for the estuarine model, and to perform research.

Physical processes in CAST have been simplified to allow for better stakeholder understanding and participation in the process of building the model. The Phase 6 model received scientific and stakeholder input from standing committees within the Chesapeake Bay Program partnership, including more than 300 members representing academic institutions, government agencies, and private interests. Three themes emerged during discussions: incorporating multiple lines of evidence, improving data sources, and increasing comprehension of the model outputs.

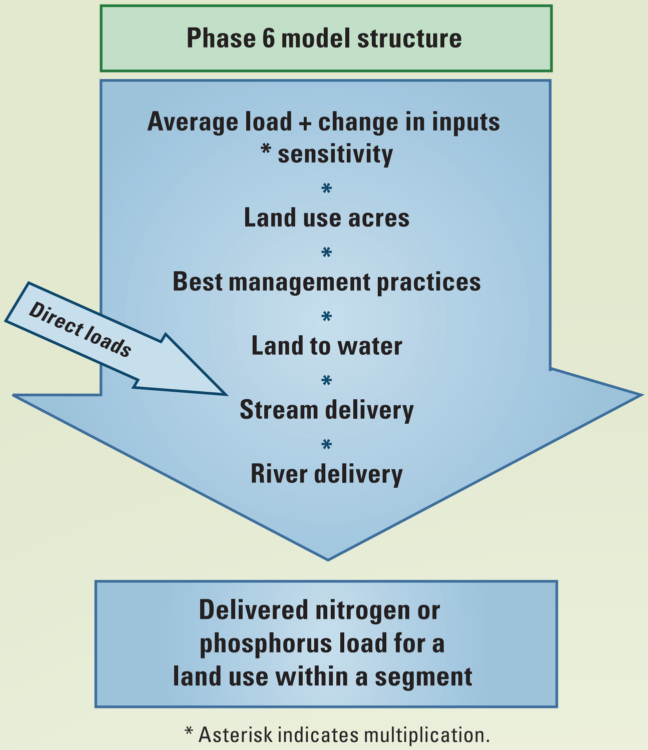


Figure 1.S3. The Phase 6 Chesapeake Bay Watershed Model provides decision makers with expected environmental responses to management actions that affect streamflow, nitrogen, phosphorus, and sediment to the Bay.

The load of nitrogen, phosphorus, or sediment for a given land use and geographic area is evaluated with nine factors in CAST (fig. 1.S3). Loads exported to a stream consider local applications (inputs) of nutrients but not local watershed conditions. Nutrient and sediment loads are then multiplied by land use area and the effect of local conservation practices. Lastly, three factors are used to represent delivery characteristics of the watershed. Full documentation of CAST is available online (Chesapeake Bay Program, 2017).

locations from 1985 to 2008, but not to the level needed to meet water-quality goals (Moyer and Blomquist, 2017). These scientific data have helped scientists and managers evaluate and better understand the effect that increased nitrogen loads have had on downstream aquatic resources such as SAV (Lefcheck and others, 2018). Also, research on the spatial and temporal patterns of nitrogen helps with targeting, planning, and evaluating the effectiveness of conservation practices for development and agriculture (U.S. Department of Agriculture, Natural Resources Conservation Service, 2013; Hyer and others, 2016; Hopkins and others, 2017).

Current Water-Quality Conditions Require Continued Regulation and Management of Nitrogen into the Future

The current nitrogen cycle for the Chesapeake Bay is depicted in figure 1.5. As previously mentioned, biological fixation of nitrogen gas from the atmosphere provided the bulk of natural inputs to the Chesapeake Bay before the 1940s, but industrial fixation of nitrogen has far exceeded natural inputs to the watershed in more recent decades. In natural ecosystems, denitrification is in balance with the

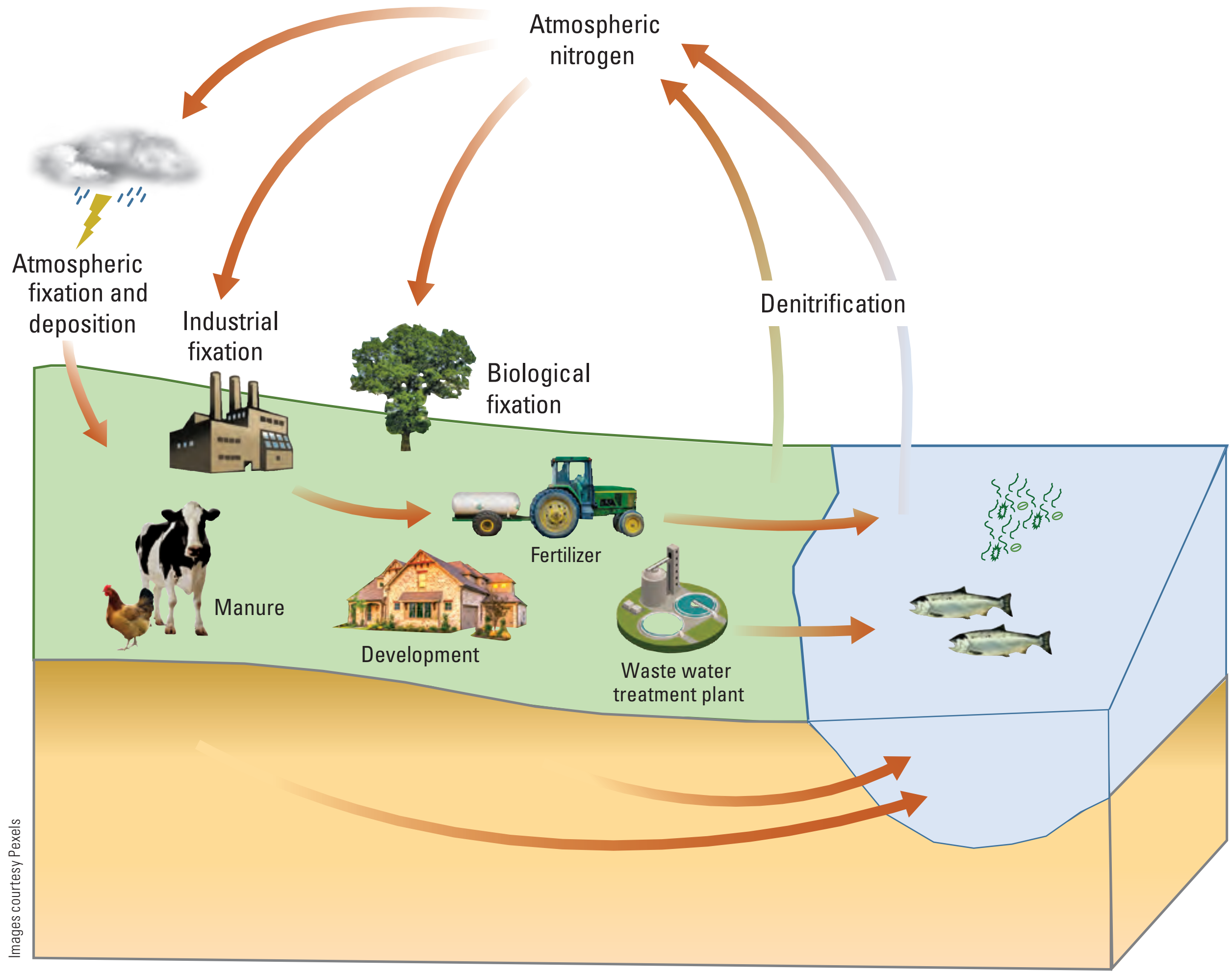


Figure 1.5. The current nitrogen cycle in the Chesapeake Bay.

biological fixation of nitrogen (Galloway and others, 2004). Under current conditions in the Bay with industrial fixation, the capacity for denitrification to convert nitrogen back to the atmosphere is exceeded and a substantial portion of the surplus nitrogen often moves to surface water and groundwater. This movement of bioavailable nitrogen (see sidebar– Reactive [Bioavailable] Nitrogen) has local and far reaching effects on waterways. Many streams within the Chesapeake Bay watershed are not meeting their designated uses and are considered impaired for nutrients such as nitrogen. Excess nitrogen in local streams and lakes reduces biodiversity owing to eutrophication and acidification (Galloway and others, 2004). High levels of nitrate and associated constituents in groundwater can also be a threat to rural drinking water supplies (Loper and others, 2009; Clune and Cravotta, 2019, 2020). Once excess nitrogen reaches the Bay, hypoxic conditions can occur and can cause significant ecologic and economic impairments (Officer and others, 1984). Reversing the impacts of nitrogen loading will take time, but recent water-quality indicators offer some evidence that the health of the Bay may be improving (Zhang and others, 2018).

The future of excess nitrogen in the Chesapeake Bay is dependent on many legacy issues, but also some new challenges. Land use, food production demand, and fossil fuel combustion are expected to continue to increase owing to rising populations and consumption rates. In the Chesapeake Bay watershed, the population is expected to increase by 16 percent by 2050 (fig. OV.1; see chap. 6). Climate change is expected to bring rising temperatures, more precipitation, and sea level rise that will likely change the transport and fate of nitrogen. For instance, increases in temperature will change the timing and length of agricultural growing seasons and how much and when fertilizer is applied. Increased storm events could bring pulses of higher nutrient loads to the Bay from upstream, especially where impoundments such as the Conowingo Dam have reached their capacity for holding back nutrients and sediments (Langland, 2015; Zhang and others, 2016).

The continued efforts of management, regulation, and conservation in the watershed may reveal long-awaited improvements for the health of the Bay. Enacted air and water-quality standards should continue to decrease the nitrogen load. Watershed implementation plans to meet total maximum daily load (TMDL) goals will have far reaching effects across the entire watershed. Degraded conditions in the Bay may improve over time as water moving through groundwater and in streams reflecting decades of conservation practices contributes to better water quality. (see chap. 4; Clune and Denver, 2012). The significance and magnitude of efforts to restore this national treasure may provide a template that could be replicated and serve as a model to the Nation and the world.

Major Factors that Influence Nutrients in Waters of the Chesapeake Bay Watershed

Both nitrogen and phosphorus are essential nutrients that all living things need to survive and that are a natural part of the Chesapeake Bay ecosystem. Excess amounts of nitrogen and phosphorus from common sources have contributed to water-quality degradation of the Bay, but the environmental behaviors of nitrogen and phosphorus are very different as they move through the watershed. Phosphorus tends to associate with particles, such that it can have long-term storage in stream and reservoir bed sediments and floodplains. In contrast, nitrogen largely stays dissolved in water as it is transported, leading to a much faster and predictable response time between nitrogen input into the watershed and nitrogen transport to the Bay, compared to that of phosphorus. This report focuses on nitrogen, but a parallel century-of-change story could be written for phosphorus. Controlling both nutrients is inherent in the Chesapeake Bay TMDL.

This report aims to provide an understanding of how human activities and environmental change in the watershed in the past, present, and future influence the export of nitrogen to the Bay. The spatial and temporal aspects of the interconnected and changing physical factors, source inputs, and management actions that control the current status and future trends of nitrogen are discussed in this report (fig. 1.6).

Physical factors like climate and hydrology play a vital role in determining the delivery mechanisms and magnitude of nitrogen inputs to the Bay (chaps. 3 and 4). Source inputs of nitrogen are introduced to the landscape each year through atmospheric deposition and human activities that vary with land use (chaps. 5 and 6). Management controls are used to mitigate the amount of nitrogen that is exported out of agricultural and developed areas (chaps. 7 and 8). The analysis of past monitoring data provides perspective on relatively recent historical trends (chap. 2) and can be used as the foundation for predictive tools to provide hindcasts and forecasts to describe a century of change in the Chesapeake Bay watershed (chaps. 9 and 10). This long-term perspective can help inform decisions that better balance the application, production, and control of nitrogen in coastal areas.

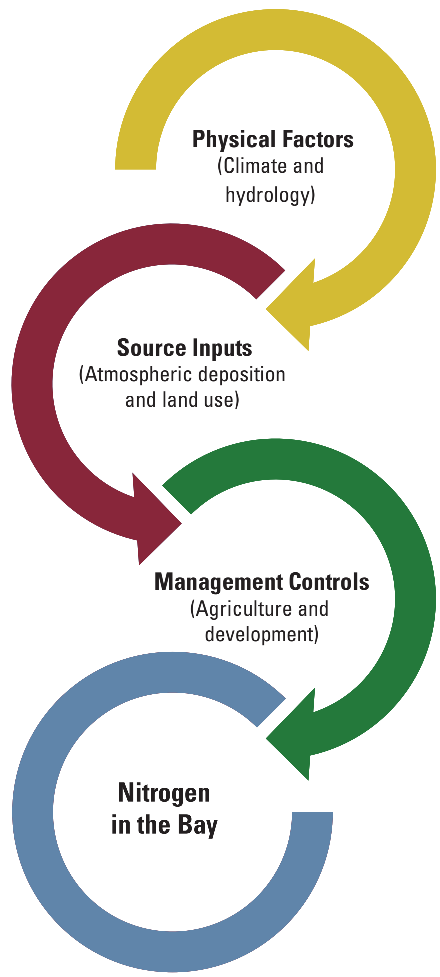


Figure 1.6. Interconnected physical factors, source inputs, and management controls drive current trends and future scenarios of nitrogen in the Chesapeake Bay watershed.

Nitrogen in Streams and Groundwater

Chapter 2

By Paul D. Capel[1],
Joel D. Blomquist[1],
and John W. Clune[1]

A restored section of Whitethorn Creek in West Virginia, in the Chesapeake Bay watershed. Courtesy of the Chesapeake Bay Program, used with permission.

Nitrogen has been sporadically measured in the streams and groundwater of the Chesapeake Bay watershed for decades (fig. 2.1). Some of the earliest measurements were in the Potomac and James Rivers in the late 1920s (Stets and others, 2015). Early measurements of nitrogen in groundwater did not occur until much later. These measurements were made by a wide range of Federal, state, and local agencies, universities, and wastewater treatment dischargers to fulfill regulatory mandates, protect human and aquatic health, and advance the understanding of the sources of excess nitrogen and its movement, behavior, and effects. Before the more extensive human impacts of deforestation, agriculture, and industrialization on the landscape, there were low levels of nitrogen in streams. The natural background concentration of nitrogen in streams has been estimated for the inland to be around 0.15 to 0.42 milligrams per liter (mg/L) and for coastal areas about 0.5 mg/L as nitrogen, respectively (Smith and others, 2003; Clune and others, 2020a).

Before the early 1980s, measurements of nitrogen in streams were insufficient to allow for generalizations over time and space. After the Clean Water Act was enacted and the establishment of the Chesapeake Bay Program in the 1980s, programs of systematic monitoring of nitrogen in selected streams were initiated (fig. 1.4). This systematic monitoring marked the beginning of the ability to assess the water-quality conditions with regards to nitrogen, and to examine the trends in concentrations

[1]U.S. Geological Survey.

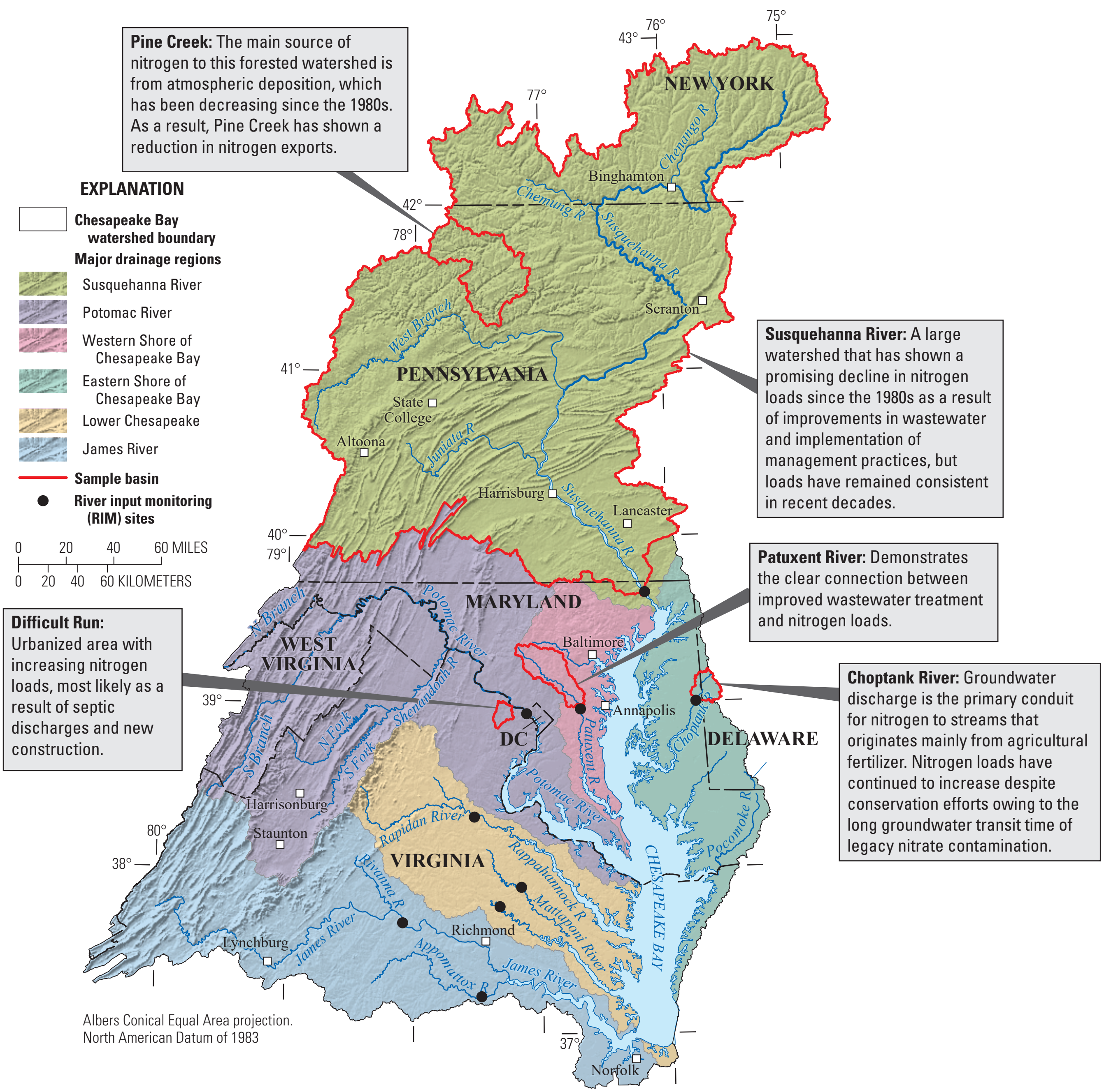


Figure 2.1. Map of the major rivers and their watersheds used as examples in this chapter.

and loads (see sidebar–Quantifying Nitrogen: Concentrations, Loads, and Yields). There has never been a systematic program for monitoring nitrogen in groundwater in the Chesapeake Bay watershed, but numerous measurements, for a range of purposes, have been a part of local and regional studies (Debrewer and others, 2008). Analysis of available groundwater data has shown spatial patterns of nitrogen in groundwater (Greene and others, 2005; Terziotti and others, 2018), but changes in groundwater concentrations over time are less well understood (fig. NS.8).

Quantifying Nitrogen: Concentrations, Loads, and Yields

Concentration is the fundamental way that a constituent in water is expressed. The concentration of nitrogen, for example, is simply the mass of nitrogen in a known volume of water (fig. 2.S1). Generally, concentrations are standardized and expressed as milligrams (mg) of nitrogen (N) in a liter (L) of water (mg N/L). Nitrogen pollution can impact aquatic biota growth and health. For example, high concentrations of nitrate can act as a fertilizer in water and contribute to algal blooms that, upon decay, reduce or eliminate oxygen, leading to fish illness and death. Further, high concentrations of ammonia that are toxic to fish may occur in water when a bloom dies off and dead algae undergo decomposition.

Load is the expression of the mass of nitrogen moving past a location over a given length of time. For an annual period, load is simply the mass of nitrogen per year (for example, kilograms per year or kg/yr). Load is calculated as the concentration multiplied by the volume of water moving past a location. Nitrogen loads are used by water quality managers when considering the effects on downstream water bodies, such as in the Chesapeake Bay TMDL (see chap. 1 sidebar–Largest Total Maximum Load in the Nation). Annual nitrogen loads are dependent on both concentration and climate variability and can fluctuate considerably from year to year. The impacts of season and weather are considered by resource managers as they assess nitrogen trends and how management practices affect stream water quality.

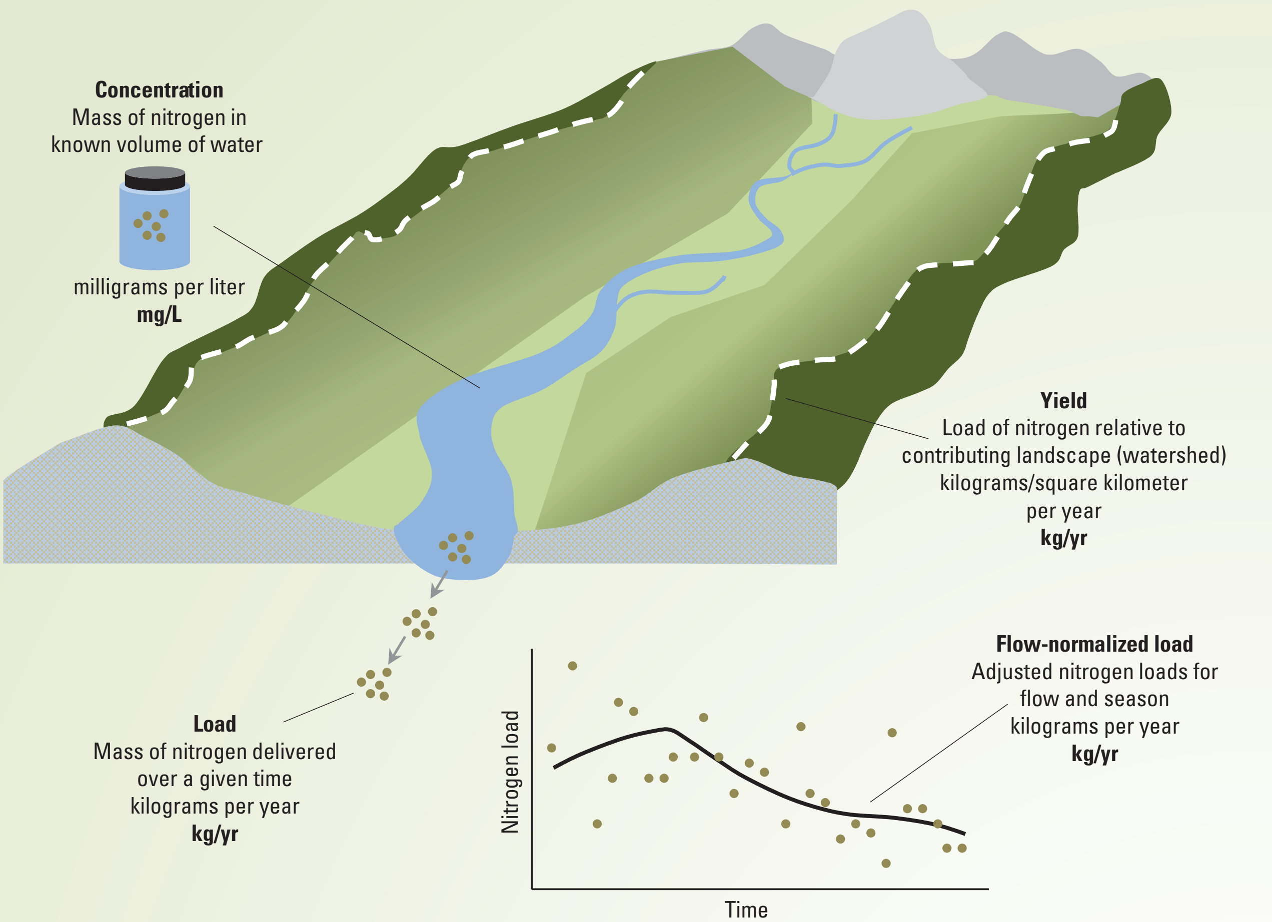


Figure 2.S1. Understanding nitrogen in terms of concentration, load, yield, and flow-normalized load is important for assessing the impacts and management of nutrients to the Chesapeake Bay and its tributaries. Graphic modified from Long Tom Watershed Council (https://www.longtom.org/about-ltwc/watershed-diagram/).

Yield is the expression of the nitrogen load relative to the area of the landscape from which it is derived. It is simply load divided by watershed area, expressed in this report as kilograms (kg) of nitrogen (N) per square kilometer (km^2) per year (yr) (kg N/km^2/yr). The yield of nitrogen from the landscape is analogous to the yield of a crop from the landscape (kilograms of nitrogen per square kilometer per year compared to tons of grain per acre per year).

Flow-normalized loads have been developed using a statistical method that removes the influence of seasonal and streamflow effects and provides another way to view nitrogen trends in streams and the Bay (fig. 2.S1; Hirsch and others, 2010). Understanding nitrogen in terms of concentrations, loads, yields, and flow-normalized loads is important because the amount of nitrogen entering the Bay is strongly affected by major storms over the historical record (fig. 2.S2).

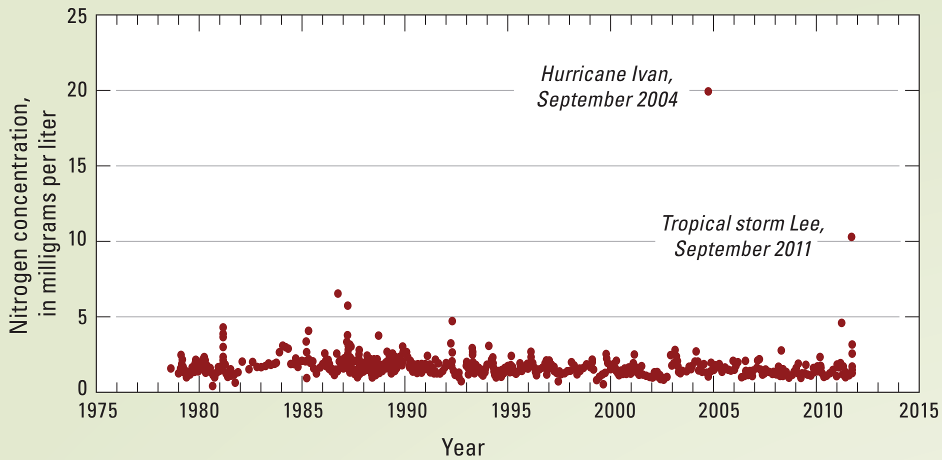


Figure 2.S2. Total-nitrogen concentration as a function of time for the Susquehanna River at Conowingo, Maryland (Hirsch, 2012).

Export of Nitrogen from the Watershed to the Bay

Starting in 1985 and enhanced in 2004, sufficient measurements of nitrogen and streamflow during a range of conditions have been made in 123 nontidal streams and rivers to enable the calculation of annual nitrogen loads to the Bay (Moyer and Blomquist, 2020a). Nine of these locations, referred to as river input monitoring (RIM, fig. 2.1) sites, are near the downstream end (Moyer and Blomquist, 2019) of the nontidal zone and provide the ability to quantify the concentrations and loads of nitrogen that are exported directly to the Bay from major tributaries. The trends in nitrogen loads and flow-normalized nitrogen loads for the river input monitoring sites during the years 1985 to 2019 (for an explanation of flow-normalization, see sidebar–Quantifying Nitrogen: Concentrations, Loads, and Yields) are shown in figure 2.2. Over this time period, the highest flow-normalized loads occurred at the beginning of the monitoring period (in other words, 1985) for the Susquehanna, James, and other rivers (fig. 2.2). The total export of nitrogen from the major tributaries to the Bay decreased by 19 percent over the 34-year period from 1985 to 2019.

The general downward trend in the export of nitrogen to the Bay is a result of both local and regional management of nitrogen. There has been a regional decreasing trend in atmospheric deposition of nitrogen to the watershed since the mid-1980s owing to the enactment of the Clean Air Act (fig. NS.5A; see chap. 5). Since the beginning of the Bay total maximum daily load (TMDL) in 2010, there has been a coordinated effort to better manage nitrogen within the watershed through the implementation of agricultural (see chap. 7) and urban management practices, and upgrades of wastewater treatment facilities (see chap. 8).

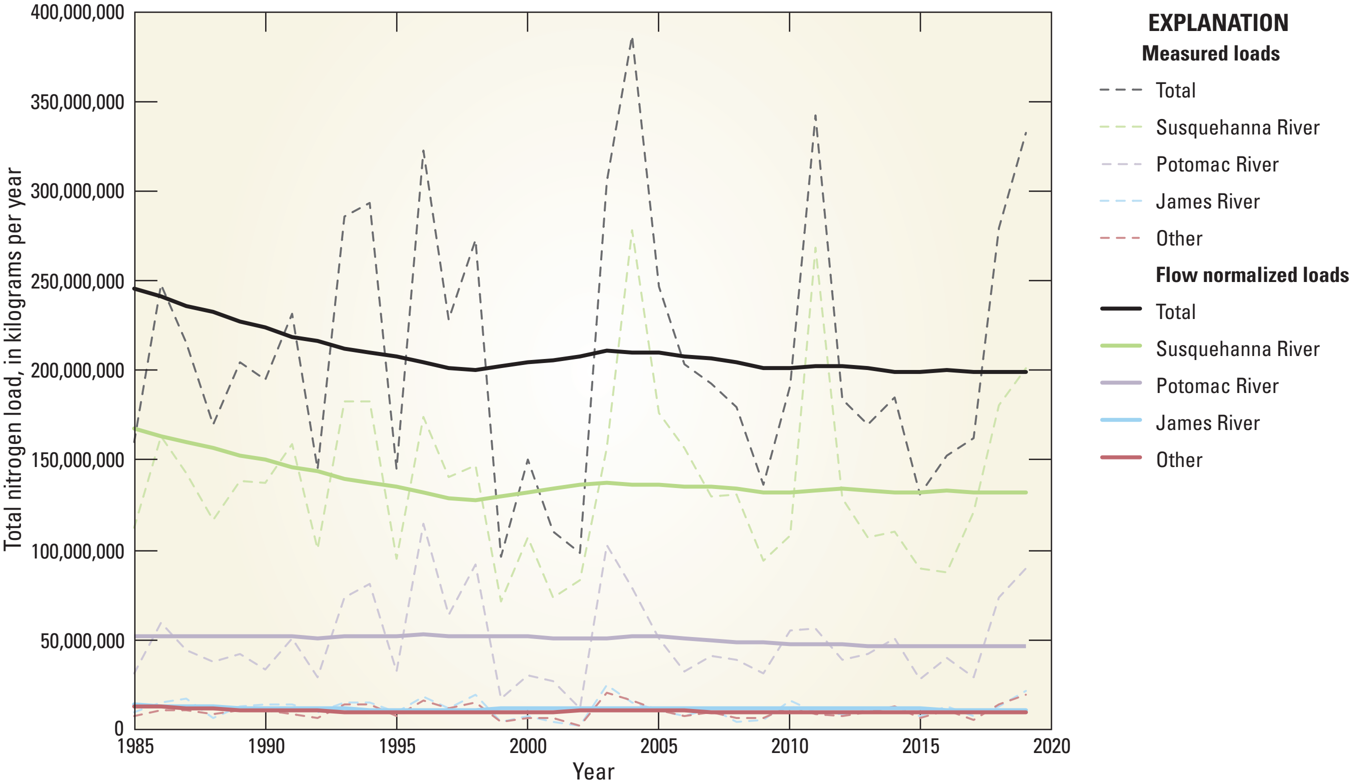


Figure 2.2. Estimated loads of nitrogen exported to the Chesapeake Bay through major tributaries, 1985–2019 (Moyer and Blomquist, 2020). The bold lines are the flow-normalized loads, which remove the effect of year-to-year variability in climate. The dotted lines are the actual measured loads exported to the Bay. The black line is the sum of the loads from all nine tributaries. The loads from the Susquehanna, Potomac, and James Rivers, are shown separately. The "Others" group combines the loads from the Rappahannock, Appomattox, Pamunkey, Mattaponi, Patuxent, and Choptank Rivers. For explanation of flow-normalized loads, see sidebar–Quantifying Nitrogen: Concentrations, Loads, and Yields.

Water-quality models are used to estimate the movement of nitrogen within the watershed and exports to the Bay. Modeling results provide spatial and temporal insights where there are limited measurements of nitrogen. The measured values (observed in streams) are used to calibrate models. The annual flow-normalized loads based on observed values provide insight to the long-term trajectory of nitrogen delivered to the Bay (fig. NS.6).

Nitrogen export from the nontidal portions of the major rivers differed across the Bay watershed. Nitrogen loads decreased in the James River by 21 percent, in the Potomac River by 11 percent, and in the Susquehanna River by 21 percent. The other six major tributaries discharging directly to the Bay (fig. 2.2) had a combined decrease of 22 percent. These changes can be put into context by comparing the relative amounts that each of these rivers contribute to the combined total load of nitrogen exported to the Bay. Over this 34-year period, the average percentage of the total load from the nontidal portions of major rivers to the Bay was 6, 24, 66, and 5 percent for the James, Potomac, and Susquehanna Rivers, and the sum of the other six major tributaries, respectively. Nutrient management in the Susquehanna River, with the largest annual load and smallest decrease in loads over time, continues to be a major challenge with respect to reaching the goal of the TMDL.

Spatial Distribution of Nitrogen in Streams, Groundwater, and Tidal Areas

Nitrogen measured in streams and groundwater, in conjunction with hydrologic models that simulate the watershed, provides a spatial perspective on the movement of nitrogen though the watershed. The spatial patterns of nitrogen loads observed in streams correspond to an understanding of nitrogen sources to the watershed (fig. NS.7). The lighter shaded portions are areas that are predominately forested, where there are few sources of nitrogen to the watershed other than deposition from the atmosphere. The darker shaded portions of the maps, with higher loads of nitrogen, are population centers (Washington, D.C.; Baltimore, Maryland; Norfolk, Virginia) and areas of intense agriculture (southeastern Pennsylvania, Delmarva Peninsula). These areas have many sources of nitrogen including urban runoff, agricultural fertilizer, agricultural manure, wastewater discharge, septic system releases, and atmospheric deposition.

The concentrations of nitrogen in groundwater and loads to streams from groundwater have a similar spatial pattern to that of nitrogen load in streams (fig. NS.7 and NS.8). There are large areas of the watershed where the groundwater concentrations and loads to streams are very low (lighter shades) in aquifers that underlie forested areas with limited sources of new nitrogen each year. Relatively high nitrogen concentrations in groundwater are generally observed in areas of the watershed (1) with intense agriculture (crop and [or] animal) in the Coastal Plain, which contains sandy permeable soils and sand and gravel aquifers (Delmarva Peninsula), or (2) underlain by carbonate rock aquifers that allow water and nitrogen to move quickly through the cracks and channels in the rock (see sidebar–Legacy of Nitrogen in Groundwater). A statistical analysis showed that the highest base flow nitrate concentrations were associated with intensive agricultural land use and carbonate geology, but also areas that lacked riparian canopy (Wherry and others, 2020).

Human activities control the mass of excess nitrogen introduced in the landscape each year. Some sources, such as atmospheric deposition, are regional, but most are local to an urban or agricultural location (city block or farm field). Wastewater treatment facilities collect human waste from delineated developed areas and discharge some fraction of this nitrogen directly to streams after treatment, whereas the nitrogen released from on-lot septic systems usually goes to the soil and groundwater. The Chesapeake Bay Program has developed an estimate of the contributions of nitrogen from septic systems. Agriculture (fertilizer and manure from animal production) is the largest source of nitrogen, followed by atmospheric deposition, wastewater discharges, and developed areas. Many of the largest urban areas in the watershed are located near the Bay itself and their wastewater discharges are input into the tidal zones of the rivers. From 1984 to 1989, 67 percent of the nitrogen from wastewater treatment facilities was discharged into the tidal zones, but this has been substantially reduced in recent years (2010–2015) when many of these wastewater treatment plants have been upgraded with new technologies (see chap. 8 sidebar–Blue Plains Advanced Wastewater Treatment Plant; Chesapeake Bay Program, 2017; Hopple and others, 2020b).

Legacy of Nitrogen in Groundwater

Precipitation that infiltrates through the soil and rocks becomes groundwater (see chap. 4). Along the way, water picks up natural and anthropogenic (generated by human activities) contaminants on the land surface or while traveling through the soil and underlying geology. Nitrogen in the form of nitrate is not found naturally in high concentrations in groundwater, but is often leached from the surface inputs of fertilizer, manure, and sewage/septic systems.

Groundwater is a vital resource for both humans and aquatic life. Many residents in the Chesapeake Bay watershed depend on groundwater as their source of drinking water. Excessive nitrate (>10 milligrams per liter [mg/L]) in drinking water is considered a health risk and is regulated to protect public drinking supplies (U.S. Environmental Protection Agency, 2012). Nitrogen in groundwater that eventually discharges to streams can raise stream levels above background concentrations and stimulate algal growth and decay that impairs aquatic ecosystems (Dubrovsky and others, 2010; Clune and others, 2020a,b). The movement of groundwater can take days to thousands of years before discharging as surface water and this creates a lag in the movement of nitrate to streams (fig. 2.S3). Scientists and resource managers find it challenging to assess if recent management actions are improving water quality while legacy nitrate contamination from groundwater is still advancing to the Chesapeake Bay and its tributaries.

In the Chesapeake Bay region, the vulnerability of nitrate contamination in groundwater is associated with intensive agriculture, especially in areas overlying carbonate rocks or coarse sand deposits such as in southeastern Pennsylvania and along the Eastern Shore of the Chesapeake Bay (fig. 2.S4; Greene and others, 2005). Increases in fertilizer use since the 1950s have led to increased storage of nitrate in groundwater across the watershed, but the extent of contamination in groundwater can vary spatially depending on local agricultural practices, residence times, and geology. For example, the fate of nitrate in two adjacent watersheds having similar agricultural practices on the Eastern Shore of the Chesapeake Bay watershed were shown to be very different owing to the underlying groundwater-flow system (Böhlke and Denver, 1995). As shown in figure 2.S5, groundwaters of similar age (determined through isotopic analysis) were shown to have comparable nitrate concentrations when they first entered (recharged) the aquifer. However, over time nitrate levels in the groundwater of Chesterville Branch were shown to be more stable and elevated (10 mg/L) compared to groundwater of the nearby Morgan Creek, where the anoxic conditions allow for denitrification and lower nitrate concentrations (3 mg/L).

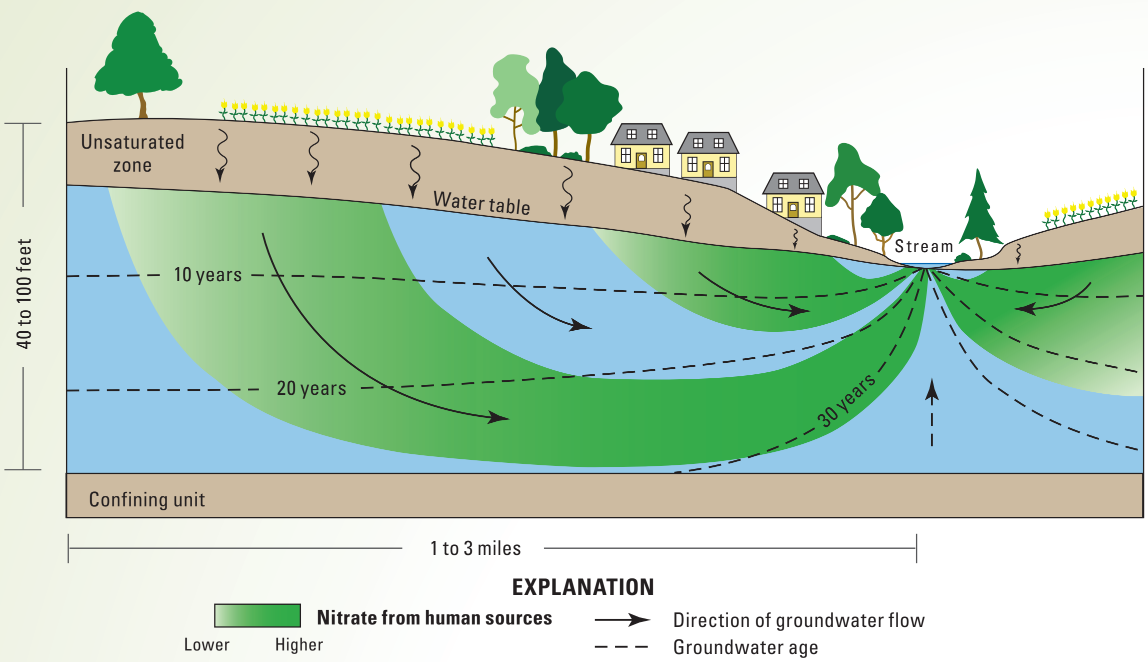


Figure 2.S3. The slow movement of groundwater creates a delay (lag time) for which implementation of management practices may produce a subsequent response in the nutrient concentrations in streams. Adapted from Terziotti and others (2018) and Shedlock and others (1999).

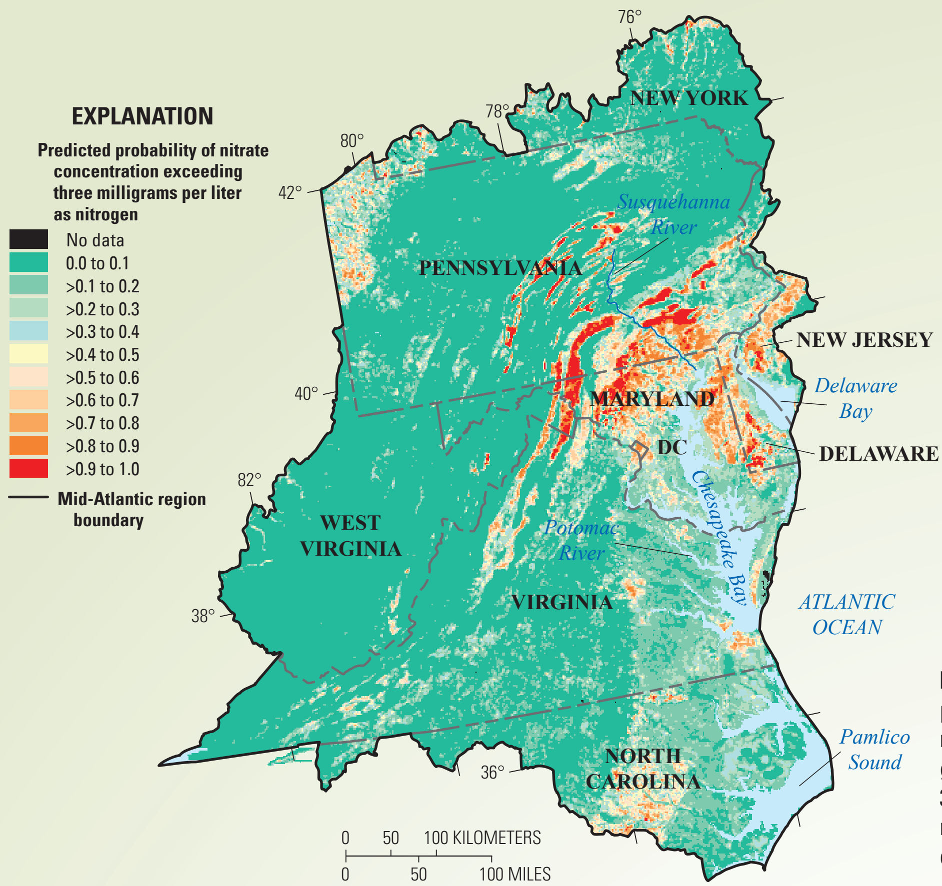


Figure 2.S4. The predicted probability of nitrate concentrations in groundwater exceeding 3 milligrams per liter as nitrogen (Greene and others, 2005).

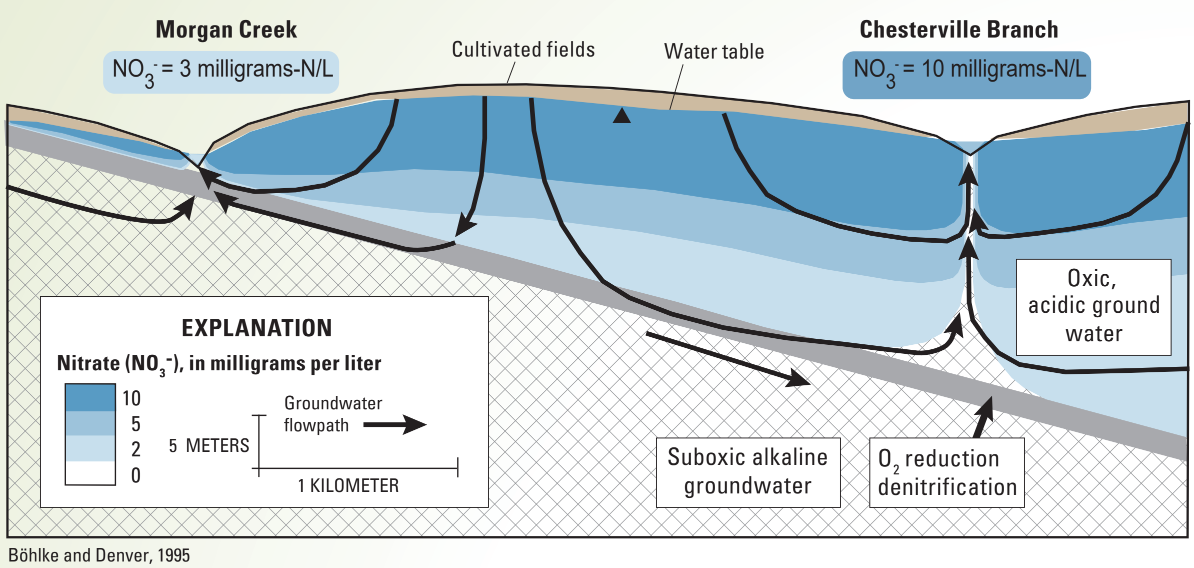


Figure 2.S5. Groundwater nitrate concentrations in two adjacent watersheds having similar agricultural practices on the Eastern Shore of the Chesapeake Bay can be very different owing to the underlying geology (Böhlke and Denver, 1995). N, nitrogen; L, liter.

The long-term monitoring at 123 nontidal streams and rivers allows for comparison of the trends in loads among the streams (Moyer and Blomquist, 2020a). For example, changes in nitrogen yields between 2009 and 2018 for the Susquehanna Basin are shown in figure 2.3*A*. Sites located along larger rivers, such as those on the main stem of the Susquehanna, show the cumulative effects of multiple tributary watersheds upstream of the sampling point. The nitrogen concentrations in small tributary watersheds are more indicative of the local inputs and environmental controls. Nitrogen from various sources in each small watershed is processed differently based on management actions and watershed characteristics such as soils, topography, land use, and aquifer type (Moyer and others, 2017). It is hypothesized that decreases in the yields of nitrogen (improving trends) in the Chesapeake Bay watershed over this time period may be attributed to the implementation of management actions on nitrogen as part of the TMDL (fig. 2.3*B*). However, there are still areas where nitrogen loads have been increasing—such as the Eastern Shore of the Chesapeake Bay. In many places on the Eastern Shore, the groundwater has high concentrations of legacy nitrate in groundwater from crop and animal agriculture in areas with highly permeable, sandy soils and shallow groundwater. This nitrogen in groundwater will continue to slowly discharge to streams and contribute to elevated stream loads for many years owing to the long residence time of the groundwater in this area (see sidebar–Legacy of Nitrogen in Groundwater). Southeastern Pennsylvania has shown decreasing (improving) trends over this time period, but overall, the magnitude of nitrogen loads and yields from this area are high owing to continued crop and animal agriculture on a landscape that is underlain by carbonate aquifers vulnerable to contamination. Carbonate aquifers have subsurface channels that quickly transport water and nitrogen through the aquifer.

Trend computations of total nitrogen loads from these major tributaries to the Chesapeake Bay constitute an important environmental indicator that is linked to the overall health of the Bay. However, it must be noted that a more localized view of inputs and exports for Bay conditions is also important. The Chesapeake Bay has many subestuaries that are strongly influenced by the inputs (rivers, point sources, and groundwater) that directly drain to them and which may locally overwhelm the influence of the waters in the larger main stem of the Bay. Examples of these include the Potomac, the James, and the Patuxent Rivers, which are strongly influenced by wastewater and storm water from metropolitan areas such as Washington, D.C., Richmond, Virginia, and the suburban region between Baltimore and Washington, D.C., respectively. Additionally, subestuaries dominated by agricultural land use such as the Choptank or Nanticoke Rivers on the Delmarva Peninsula can contribute large amounts of nitrogen to these waters, which are important sites of commercial and recreational shellfish harvesting.

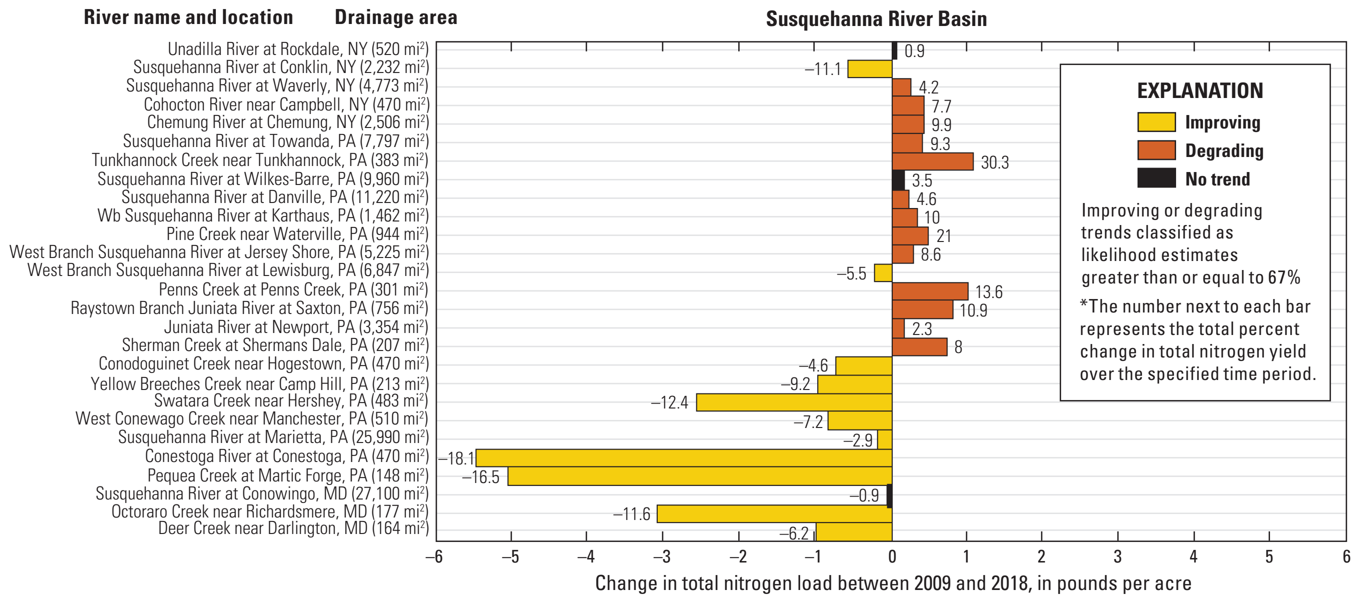
Watershed Stories

The degree of progress in improving water quality is different across watersheds in the Chesapeake Bay owing to land use, population change, nitrogen sources, and management efforts to meet the Bay's TMDL. Five watersheds will be highlighted here to illustrate the wide variety of nitrogen stories over the past three decades (fig. 2.1).

Patuxent River

The watershed of the Patuxent River is a rapidly developing area on the western shore of the Chesapeake Bay (fig. 2.1). The river drains directly into the Bay. One of the major sources of nitrogen to the Patuxent River (Maryland) is wastewater treatment discharges. The State of Maryland has made large investments to implement enhanced nitrogen removal at many of its wastewater treatment plants, including the Patuxent and Little Patuxent facilities (Maryland Department of the Environment, 2020). The result has been a reduction of more than 80 percent in the annual nitrogen loads discharged from the wastewater treatment plants to the Patuxent watershed between 1985 and 2015. This has resulted in a more than 64 percent decrease in the nitrogen load in the Patuxent River over the same time period (fig. 2.4).

A



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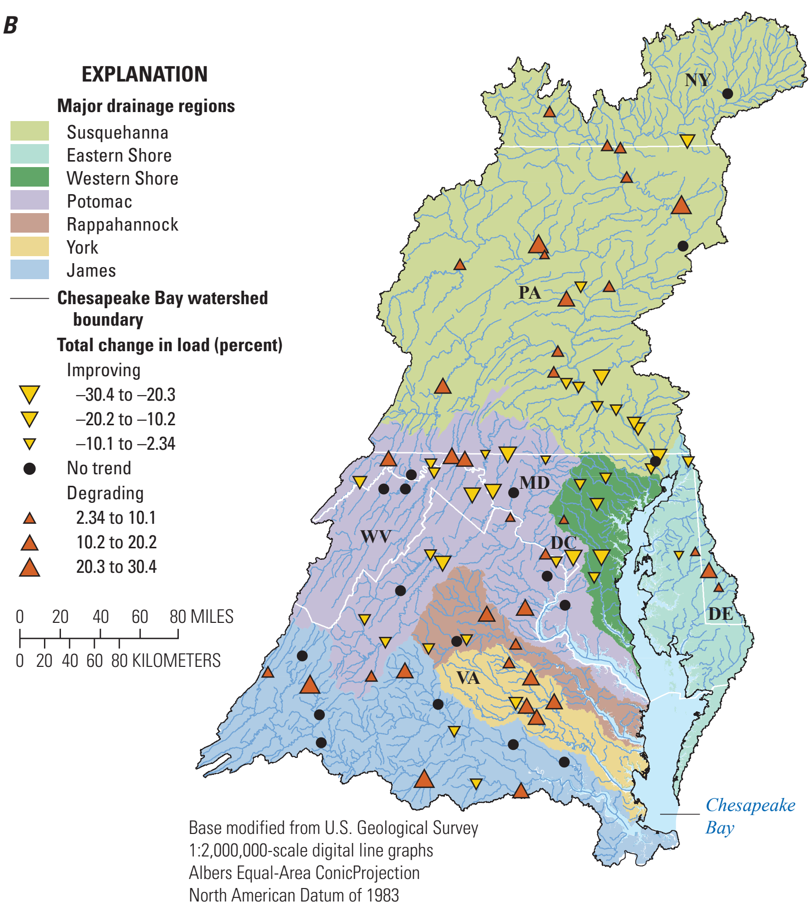

Figure 2.3. Total change in flow-normalized loads from 2009 to 2018 in the ***A***, Susquehanna Basin, expressed in pounds per acre (showing the change in percent units (Moyer and Langland, 2020); and ***B***, percent change in nitrogen loads at all the nontidal monitoring stations in the Chesapeake Bay watershed over the same time period.

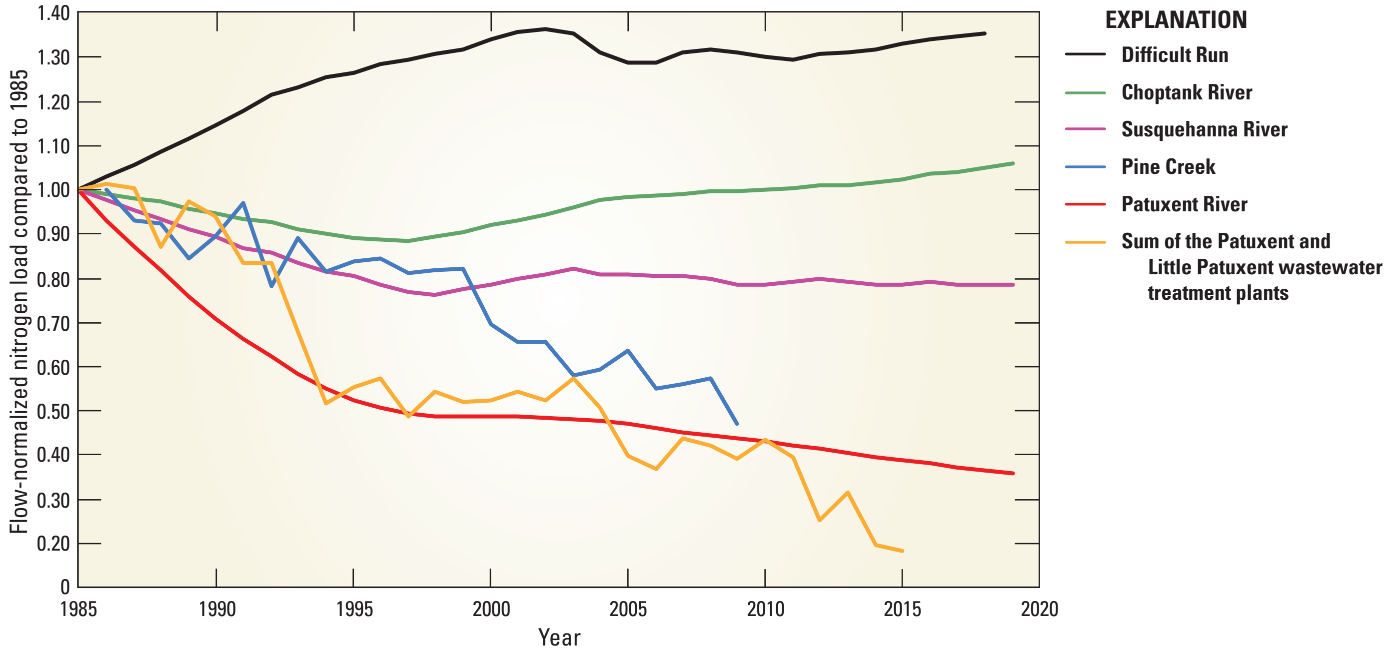

Figure 2.4. Time series of annual flow-normalized nitrogen loads for the Choptank River, Susquehanna River, Patuxent River, and Difficult Run, as well as estimated loads for Pine Creek and for the Patuxent and Little Patuxent wastewater treatment plants (WWTPs). The nitrogen loads shown are relative to 1985 (the load for each year for each stream was divided by the load for 1985 for that stream) to allow an easy comparison of the trends within and across streams. Values >1 reflect an increase in the annual load compared to 1985. Values <1 reflect a decrease in the annual load compared to 1985. Loads for Pine Creek are relative to 1986, the first year of data available. Data sources include Eshleman and others (2013) for Pine Creek; Moyer and Blomquist (2020) for the Choptank River, Susquehanna River, Patuxent River; Moyer and Langland (2020) for Difficult Run; and Hopple and others (2021) for the Patuxent and Little Patuxent WWTPs.

Pine Creek

Pine Creek (Pennsylvania) is an approximately 98-percent forested watershed in the northwest corner of the Chesapeake Bay watershed (fig. 2.1). The predominant source of nitrogen to the watershed is atmospheric deposition, which has been decreasing in the Chesapeake Bay watershed since a peak in the early 1980s (Eshleman and others, 2013; fig. NS.5) as a result of the effectiveness of the Clean Air Act (see chap. 5). Annual nitrogen loads in Pine Creek decreased by more than 50 percent from the beginning of nitrogen monitoring in the watershed in 1986 to 2009 (fig. 2.4). Similar decreases have been observed for many other pristine watersheds in the Chesapeake Bay watershed. Eshleman and others (2013) attribute the reduction in the Pine Creek nitrogen load to the reduction in the atmospheric deposition of nitrogen to the watershed. They also suggest that further reductions in stream loads should be expected if the atmospheric deposition of nitrogen continues to decline.

Difficult Run

Difficult Run (Virginia) flows through an area west of Washington, D.C. that has undergone tremendous urbanization over the past few decades (fig. 2.1). The land use in the watershed has changed from forested to suburban, a transition that opened the potential for greater soil and nitrogen loss to the stream. Development brought an increase in on-site septic systems to the area, as well as the use of lawn fertilizers and an increase in pet waste. These sources, combined with other urban sources of nitrogen, have resulted in a steady increase in the annual nitrogen load in the Difficult Run for more than two decades, 1985–2019 (fig. 2.4). In recent years, with fewer land-use changes and other changing factors (climate, point sources, nonpoint sources, management actions), the annual nitrogen load in Difficult Run has remained relatively constant, with a much smaller rate of change (Moyer and Langland, 2020).

Choptank River

The Choptank River (Maryland) flows through a rich agricultural area on the Eastern Shore of the Chesapeake Bay (fig. 2.1). Corn and soybeans are intensively grown on sandy soils that overlay shallow sandy aquifers. The implementation of agricultural management practices to control the loss of nitrogen has been a focus of water-quality improvement in the watershed (Natural Resources Conservation Service, 2020a,b). In spite of these efforts, the annual loads of nitrogen in the Choptank River have been increasing over the past two decades (fig. 2.3*B*), although figure 2.4 indicates that nitrogen may have decreased in the 1980s and 1990s prior to the increase. The presence of nitrogen in the Choptank River is largely a result of the accumulated reservoir of nitrogen in shallow groundwater from fertilizer and manure use over the past decades (see sidebar–Legacy of Nitrogen in Groundwater). The implementation of more extensive management practices (see chap. 7) in the Choptank River watershed may decrease the loss of nitrogen from the fields, although ongoing intensification of agriculture and legacy nitrogen from groundwater provides a constant source and contributes to the high concentrations in the river (Hirsch and others, 2010). It might be years or decades into the future before the benefits of current management practices are reflected in the loads observed in the Choptank River.

Susquehanna River

The Susquehanna River drains 27,500 square miles (71,000 square kilometers) that includes large parts of New York, Pennsylvania, and Maryland, and the Conowingo Reservoir which sits near the downstream end (fig. 2.1). The watershed has diverse land uses (urban, industrial, agricultural, and undeveloped areas). Over the past few decades, the nitrogen load from various sources to the river over this large area has been successfully reduced through improvements in wastewater treatment, implementation of urban and agricultural management practices, reduction in the area of farmland, and decreases in atmospheric deposition, as can be observed in the reduction of the annual loads of nitrogen from 1985 to the late 1990s (fig. 2.4). Since then, annual loads have not decreased further, largely owing to the effects of the infill of the Conowingo Reservoir (Zhang and others, 2016).

Susquehanna River in Broome County, New York. Courtesy of the Chesapeake Bay Program, used with permission.

Changes in Climate

Chapter 3

By Douglas A. Burns[1], Gopal Bhatt[2], Jeff P. Raffensperger[1], and John W. Clune[1]

Annapolis City Marina in Annapolis, Maryland. Courtesy of the Chesapeake Bay Program, used with permission.

Climate is one of many factors that affect the sources and transport of nitrogen and related hypoxia in the Chesapeake Bay watershed (Tango and Batiuk, 2016). The climate in the watershed has shown strong variations since the Bay as we know it today first formed about 9,000 to 7,000 years ago in the aftermath of glacial melting and sea level rise (Colman and Mixon, 1988). Past ecological changes in the Bay and its watershed have been driven by natural climate variation, and more recently, by an acceleration of changes in water quality associated with human activities that have introduced excess nutrients (Willard and others, 2003). In recent decades, the mid-Atlantic and global climate have been changing at unprecedented rates, largely the result of increasing greenhouse gas concentrations in the atmosphere (Pages 2K Consortium, 2013; Rice and Jastram, 2015). These changes include warmer air and water temperatures, increases in annual precipitation and in the magnitude and frequency of large storms, and climate-driven sea level rise (fig. 3.1; Sallenger and others, 2012; Sagarika and others, 2014; Huang and others, 2017; Vose and others, 2017). Changes observed in the climate of the Bay watershed in recent decades have not been uniform, however, and reflect regional variation driven in part by varying oceanic influence (Meehl and others, 2015).

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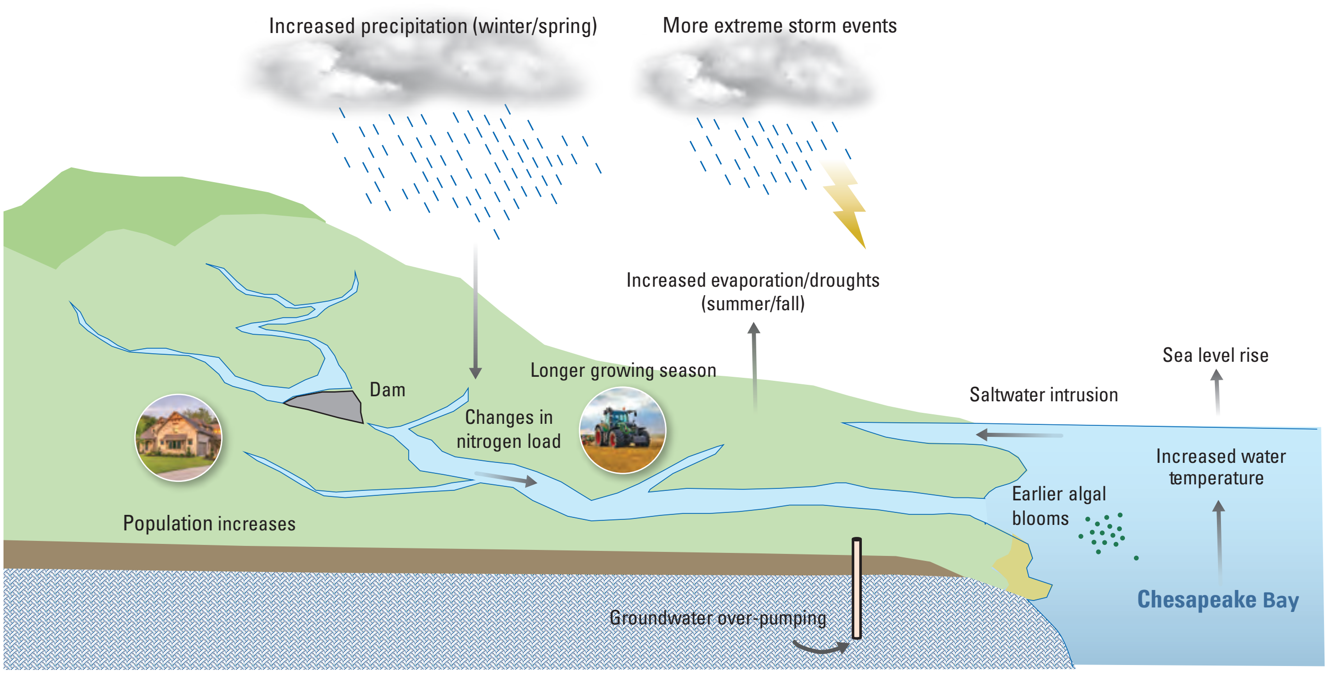


Figure 3.1. Changes in climate and the hydrologic cycle are expected to create warmer and wetter conditions that will impact nitrogen and its effects on the Chesapeake Bay.

The purpose of this chapter is to describe past changes in the climate of the Chesapeake Bay watershed from 1950 to present (2018), and likely future climatic changes. The emphasis of the chapter is on how climatic change affects delivery of nitrogen to the Bay. Some key climatic drivers discussed include temperature-driven effects on nitrogen cycling processes and the roles of precipitation and evapotranspiration on streamflow and delivery of nitrogen to the Bay. The focus is largely on the Bay watershed and not the Bay itself. A goal of this chapter is to frame the complex manner by which climate affects nitrogen transport in the Chesapeake Bay watershed by providing a hierarchical basis for conceptualizing the most relevant climate effects. This conceptualization may assist with development of management priorities.

A paleoclimatic perspective is presented to frame the climate of the past (1950–2018) and immediate future (2019–2050). Finally, a brief overview of how climate changes may affect delivery of nitrogen to the Bay is presented by discussing relevant aspects of nitrogen biogeochemistry, hydrology, and land use.

The Climate Setting

The climate in the Chesapeake Bay watershed is diverse, spanning a gradient from humid continental in the north to humid subtropical in the south (Peel and others, 2007). North-to-south climatic differences can be demonstrated by comparing temperature and precipitation patterns in Binghamton, New York, near the northernmost part of the watershed with those in Richmond, Virginia, near the southernmost part of the watershed (figs. 3.2*A* and ES.3). The mean annual temperature in Richmond is 14.9 °C, which is 7 °C warmer than the mean annual value of 7.9 °C in Binghamton (fig. 3.2*A*). Binghamton reflects not only a higher northern latitude, but also the generally greater elevations in the northernmost part of the watershed on the Appalachian Plateau, compared with the lower elevations in the Piedmont and Coastal Plain provinces in the southeastern part of the watershed where Richmond is located (Thornbury, 1965). Despite north-to-south temperature differences, there are only slight differences in annual precipitation between these cities, with Richmond receiving 111 centimeters (cm) (43.7 inches [in.]) annually compared to 100 cm (39.4 in.) in Binghamton

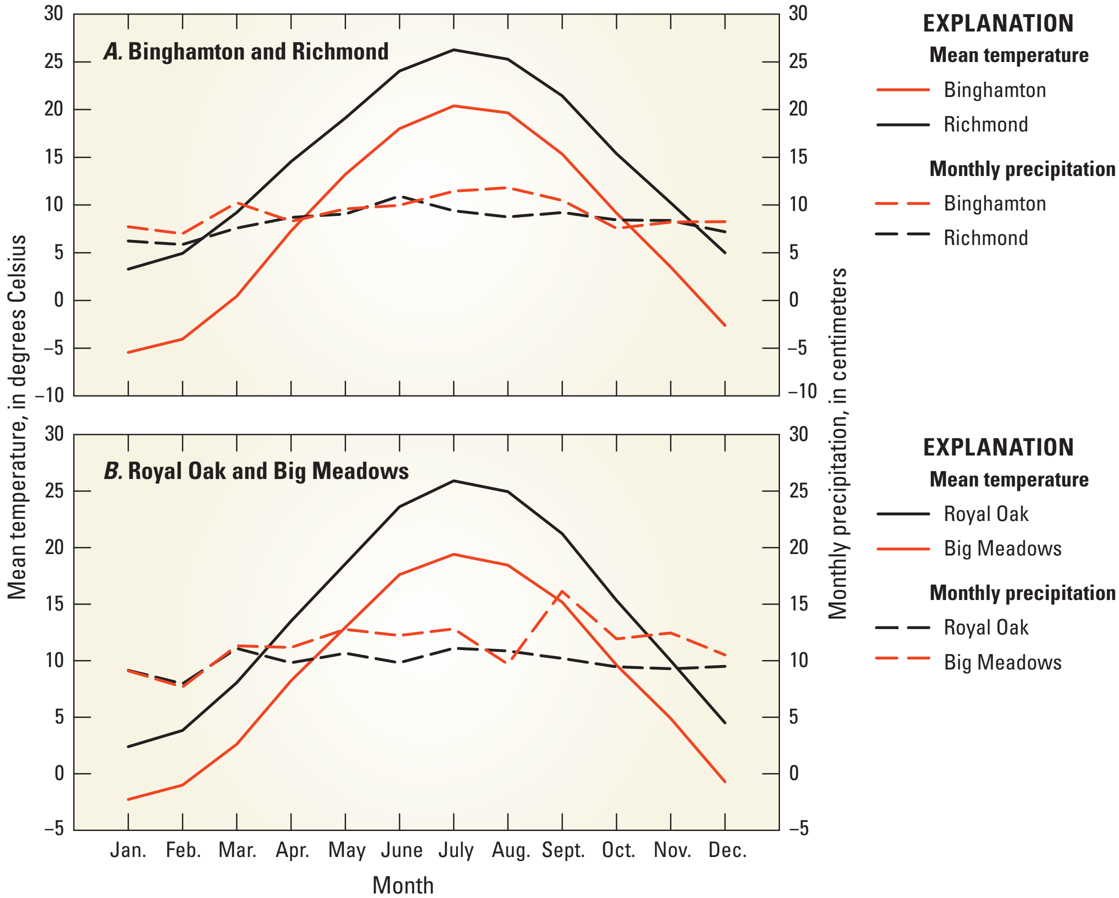


Figure 3.2. Mean monthly temperature and monthly precipitation for the period 1981–2010 at ***A***, Binghamton, New York airport (42.1° N., 75.92° W.) and Richmond, Virginia airport (37.51° N., 77.32° W.), and ***B***, Royal Oak, Maryland (38.74° N., 76.07° W.) and Big Meadows, Virginia (38.52° N., 78.44° W.).

(fig. 3.2 and ES.3). However, annual snowfall in Binghamton is 212 cm (83.5 in.), much greater than the annual value of 26 cm (10.2 in.) in Richmond, reflecting colder winter temperatures in the northern part of the watershed.

There are also strong climatic differences across the watershed owing to differences in elevation only, especially in the southwestern part of the watershed in the Blue Ridge and Valley and Ridge provinces. For example, the mean annual temperature in Royal Oak, Maryland, located on the Eastern Shore at an elevation of 3 meters (m) (9.8 feet [ft]) is 5.6 °C warmer than Big Meadows, Virginia, at a similar latitude in the Blue Ridge, but at an elevation of 1,079 m (3,540 ft) (fig. 3.2*B*). The annual precipitation of 138 cm (54 in.) at Big Meadows is greater than the annual value of 119 cm (46.9 in.) at Royal Oak, reflecting the influence from uplift of moist air masses that enhance precipitation amounts in mountainous parts of the watershed (Konrad, 1995). There are also climatic differences between these two sites that reflect the proximity to the coast of Royal Oak and the inland location of Big Meadows.

Evapotranspiration (ET), a key component of the water balance, is another metric that reflects the climate of a region. There is a broad pattern of increasing annual ET from north to south across the watershed, ranging from about 60 to 90 cm (23.6 to 35.4 in.). This is not primarily a result of the slight north-to-south difference in precipitation, but rather of large increases in air temperature over the same north-to-south gradient that drive increases in annual ET (Wan and others, 2015; Reitz and others, 2017). These ET patterns are important, along with precipitation timing and amount, in regulating the delivery of nitrogen to rivers that discharge to the Bay. The ratio of ET to precipitation increases broadly from north to south across the Chesapeake Bay watershed, indicating that there is a diminished relation between streamflow and precipitation in the southernmost part of the watershed (Reitz and others, 2017); this explains why annual stream runoff is less in the south (McCabe and Wolock, 2011). Beyond the broad north-to-south patterns, considerable spatial heterogeneity is observed owing to factors such as elevation, slope, annual snowfall, land use, and water withdrawal (Liu and others, 2010).

The Past: 1950–2018

Air and Water Temperature

Increasing trends in annual mean air temperature described at local, regional, national, and global scales suggest the climate has become warmer in the Chesapeake Bay and its watershed from the 1950s to the first two decades of the 21st century (Estrada and others, 2013; Kaushal and others, 2010; Rice and Jastram, 2015; Vose and others, 2017). Air temperature has shown distinct patterns of variation in the regions encompassing the Bay watershed and across the United States through this period, with decreases from 1950 to the early 1970s after a peak in the 1940s, and a rapid increase from the early 1970s through recent years (Vose and others, 2017). The decadal rate of increase in air temperature over the past three decades was +0.23 °C in the Bay watershed, similar to the mean rate across the United States (Rice and Jastram, 2015; Vose and others, 2017). Air temperature increases have generally been uniform throughout the Bay watershed (Rice and Jastram, 2015), with winter showing the most widespread and persistent warming, especially in the north (Vose and others, 2017). Other features of air temperature patterns across the United States and along the East Coast that are relevant to the Bay watershed include a greater increase in the minimum than in the maximum daily air temperature, a decrease in extreme annual cold temperatures, and no clear increase in extreme annual warm temperatures (Vose and others, 2017).

Water temperatures in tributaries as well as in the Bay itself have broadly increased in parallel with air temperatures, but water temperature patterns are more diverse and not always clearly synchronous with those of air (Rice and Jastram, 2015). For example, increasing water temperature trends are greater in the southern part of the watershed than in the north, a pattern that is likely a result of increasing discharge in northern streams, which dampen the water temperature trends (Rice and Jastram, 2015). The median decadal rate of increase from 1960 to 2010 was +0.28 °C in freshwater tributaries, whereas Kaushal and others (2010) reported a range of +0.22 to +0.59 °C in three Bay tributaries over varying time periods through 2006–07. Reported increases in water temperature have generally been greater in the Chesapeake Bay than its watershed, with values ranging from +0.4 to +0.8 °C per decade at eight locations from 1984 to 2000–07 (Ding and Elmore, 2015). These increases were evident across 92 percent of the Bay from 1984 to 2011 (Ding and Elmore, 2015). Many factors including dams, irrigation, impermeable surfaces, and shading by forest canopy can alter water temperatures, so some divergence in air-water temperature patterns is expected, especially where human alteration of rivers and the watershed landscape have occurred (Kaushal and others, 2010; Rice and Jastram, 2015). In summary, broad warming of surface air and water in the Chesapeake Bay and its watershed have occurred as the climate has warmed since the 1970s.

Precipitation and Streamflow

Increases in the magnitude, frequency, and intensity of precipitation across the United States have been greatest in the Northeast and upper Midwest, with the largest increases occurring in the fall. These increases have accompanied recent climate change in many regions of the United States over the past century (Easterling and others, 2017). Increasing precipitation amounts are consistent with an observed general acceleration of the hydrological cycle as the climate becomes warmer (Benestad, 2013). Within the Chesapeake Bay watershed, significant increases in both precipitation and stream discharge were widespread from 1927 to 2014, particularly north of the Pennsylvania-Maryland border (Rice and others, 2017). This north-to-south difference in precipitation and streamflow trends across the Bay watershed may be the result of the diverging effects of alternating patterns of sea surface temperature and pressure—such as the El Niño-Southern Oscillation (ENSO) that affects precipitation—and ET at regional, continental, and global scales (Feng and others, 2016). Because of the strong role of these various climate oscillations (ENSO is one among many), attributing trends in precipitation and streamflow to long-term climate change is tenuous in many regions, including the mid-Atlantic (Schulte and others, 2016). This is especially true for streamflow because of the influence of ET as well as human activities on runoff patterns. For example, Rice and others (2017) found little synchronicity between precipitation and streamflow trends across the Chesapeake Bay watershed. Rates of ET have generally increased with climate warming over the past few decades, countering the effects of increasing annual precipitation on streamflow in the Bay watershed (Seong and Sridhar, 2017). These findings highlight the high level of uncertainty that exists in determining how streamflow patterns are changing as a result of long-term climate change and in forecasting likely future patterns.

Sea Level

The elevation of the surface of Chesapeake Bay has been rising in recent decades owing to a combination of natural and anthropogenic factors that include glacial isostatic adjustment, sediment compaction, thermal expansion, modification of ocean currents, and melting of glaciers and ice sheets. Over the past 20,000 years, sea level has been rising globally and along the east coast of the United States, largely as a result of melting of ice following the most recent period of extensive continental glaciation (Miller and others, 2013). However, the rate of sea level rise has likely accelerated during the past few decades owing to both thermal expansion and the melting of ice associated with a warming global climate (Church and White, 2006). In the Chesapeake Bay, the rate of sea level rise during 1950–2009 was about 2 millimeters per year, three- to four-fold greater than the global rate of sea level rise (Sallenger and others, 2012). Future projections indicate that continued global sea level rise is likely and that potential increases range from 0.28 m to as much as 1.31 m over the 21st century (2001–2100) (Mengel and others, 2016). Risks associated with ongoing and future sea level rise in the Chesapeake Bay include coastal erosion and inundation, drowning of salt marshes, and increases in salinity, all of which have ecological effects. For example, increases in salinity, which have been documented and simulated in much of the Bay (Hilton and others, 2008; Rice and others, 2012), affect available habitat for many species. Increased salinity may cause upstream migration of species seeking fresh water, which will likely increase their proximity to sources of nitrogen and other pollutants originating from human activities (Hilton and others, 2008). Considering the complex chemical and physical interactions associated with the rising level in marine influenced parts of the Bay, it is currently unclear whether this rise will heighten or diminish the ecological effects of excess nitrogen in the Bay watershed.

Human Activities

Several human activities that affect nitrogen loads to the Chesapeake Bay such as agricultural practices, land use, and energy consumption are affected by climate change in a complex manner. For example, warmer temperatures and changing precipitation patterns alter nutrient loads and speciation from agricultural landscapes in the Bay watershed (Wagena and others, 2018). But other climate-related factors such as the timing of fertilizer application, choice of crops, effectiveness of management practices aimed at enhancing nitrogen removal, drought frequency, and the need for irrigation by groundwater are among many factors that are driven, in part, by climate change (Bowles and others, 2018). Although it is challenging to understand the complex feedbacks between climate and agriculture as well as those of other human activities, it remains clear that climate change imposes direct responses that affect nitrogen runoff as well as important indirect responses that have long-term implications for nitrogen export. Failure to consider these responses may result in an incomplete understanding of the climate-related drivers of temporal nitrogen export patterns (Abler and others, 2002).

Implications of Climate Change for Nitrogen Loads to the Chesapeake Bay

Among the many aspects of the climate that have changed in recent decades, those with the clearest implications for nitrogen export to the Bay are increasing air and water temperatures and increases in the frequency and magnitude of extreme precipitation events. All the biogeochemical processes that convert nitrogen to its different forms are temperature sensitive, with rates that generally increase with increasing temperature until an optimal temperature is reached (Greaver and others, 2016). These include key microbially mediated processes such as (1) nitrogen mineralization, in which organic nitrogen is converted to ammonium, (2) nitrification, in which ammonium is converted to nitrite and nitrate, and (3) denitrification, in which nitrate is converted to nitrogen gas. Moisture availability regulates the temperature response of microorganisms, however, so the response to climate change is also dependent on changes in precipitation and ET patterns (Borken and Matzner, 2009). The observed increase in the intensity and frequency of large precipitation events as well as the intensification of droughts associated with climate change have already affected patterns of nitrogen export to the Bay and the resulting ecological effects (see chap. 1; Kaushal and others, 2008; Mulholland and others, 2009). An intensifying cyclical pattern of drought followed by an extended wet period has been observed to result in high levels of nitrogen export in the upper Midwest, dubbed "weather whiplash," and Loecke and others (2017) suggest that this pattern may be expected to occur elsewhere with increasing frequency.

Climate change may affect the delivery of nitrogen to the Chesapeake Bay through numerous mechanisms that include altered rates of biogeochemical processes, changes in streamflow owing to changes in precipitation and temperature, and land-use and land-management adaptations (Abler and others, 2002; Greaver and others, 2016; Sinha and others, 2017; Irby and others, 2018). Because of the complex set of drivers involved, quantitative models that consider feedbacks and amplification are helpful to better understand potential climate change outcomes. Land use remains the dominant determinant of riverine nitrogen loads in the watershed, indicating that climate change cannot be adequately studied without jointly considering likely changes in land use and nutrient management (Ciavola and others, 2014). Most importantly, a shift in the focus of climate change studies is evident in recent years from changes in long-term mean climate variables to how climate change affects extreme events such as droughts, floods, coastal storms, and heat waves (Wetz and Yoskowitz, 2013). Changes in the intensity and frequency of extreme climatic events have strong implications for nitrogen transport in the Bay watershed.

The Future: 2019–2050

Increasing concentrations of greenhouse gases like carbon dioxide that trap heat close to the Earth's surface have increased mean annual temperature globally by about 1.0 °C as of 2015, and are estimated to raise global mean annual temperatures an additional 1.5 °C by 2052 (Intergovernmental Panel on Climate Change, 2018). Local climate effects will vary; however thus far, the Northeast is the fastest warming region in the continental United States (Griffiths and Bradley, 2007; Dupigny-Giroux and others, 2018). The Chesapeake Bay watershed is expected to experience a continued increase in temperature and precipitation under an intermediate emission scenario or Representative Concentration Pathway 4.5 (RCP 4.5; fig. 3.3; see sidebar–Selected Pathways for the Future Climate; Seong and Sridhar, 2017**).** Nitrogen and its effects in the Chesapeake Bay watershed are expected to be altered by changes in temperature and the hydrologic cycle creating warmer and wetter conditions (fig. 3.1).

The mean annual temperature across the entire Chesapeake Bay watershed is expected to increase by 2.0 °C by 2050 compared to 1995, with larger differences likely in the northernmost part of the watershed (fig. 3.3*A*; Seong and Sridhar, 2017). Additionally, periodic (or short-term) temperature extremes are also expected along much of the mid-Atlantic (Karl and others, 2009; Wetz and Yoskowitz, 2013). The ecological response to warmer air temperature will likely bring changes in the species composition of native vegetation and introduce better adapted, nonnative species, potentially changing nitrogen exports (Ashton and others, 2005; Gallardo and others, 2016). These exports may change through increases in denitrification and plant nutrient uptake (Schaefer and Alber, 2007). Warmer weather will provide longer growing seasons, fewer spring/fall freezes, faster growing rates, and a greater frequency of high temperatures that have agronomic implications for farmers (Prasad and others, 2010). Air temperature will likely continue to dictate water temperatures (Cronin and others, 2000) and these warming trends are expected to bring shifts in plant and algae phenology, such as earlier spring algae blooms (Edwards and Richardson, 2004), along with changes in the stratification and mixing of saltwater with freshwater within the Bay.

Selected Pathways for the Future Climate

Climate scientists use four major scenarios of how increasing amounts of greenhouse gases could trap heat (change in incoming and outgoing solar radiation) at levels of 2.6, 4.5, 6, or 8.5 watts per square meter by 2100 depending on population, economic growth, energy consumption, and sources and land use (fig. 3.S1; van Vuuren and others, 2011). These scenarios are referred to as representative concentration pathways (RCPs), and are based on assumptions of plausible future changes in a suite of socioeconomic indicators, technology, land use, and ancillary air pollutants. They are often applied in climate models to illustrate the likely range of future climate under future low (RCP 2.6), two intermediate (RCP 4.5 and 6), or high (RCP 8.5) greenhouse gas concentrations (carbon dioxide-equivalent).

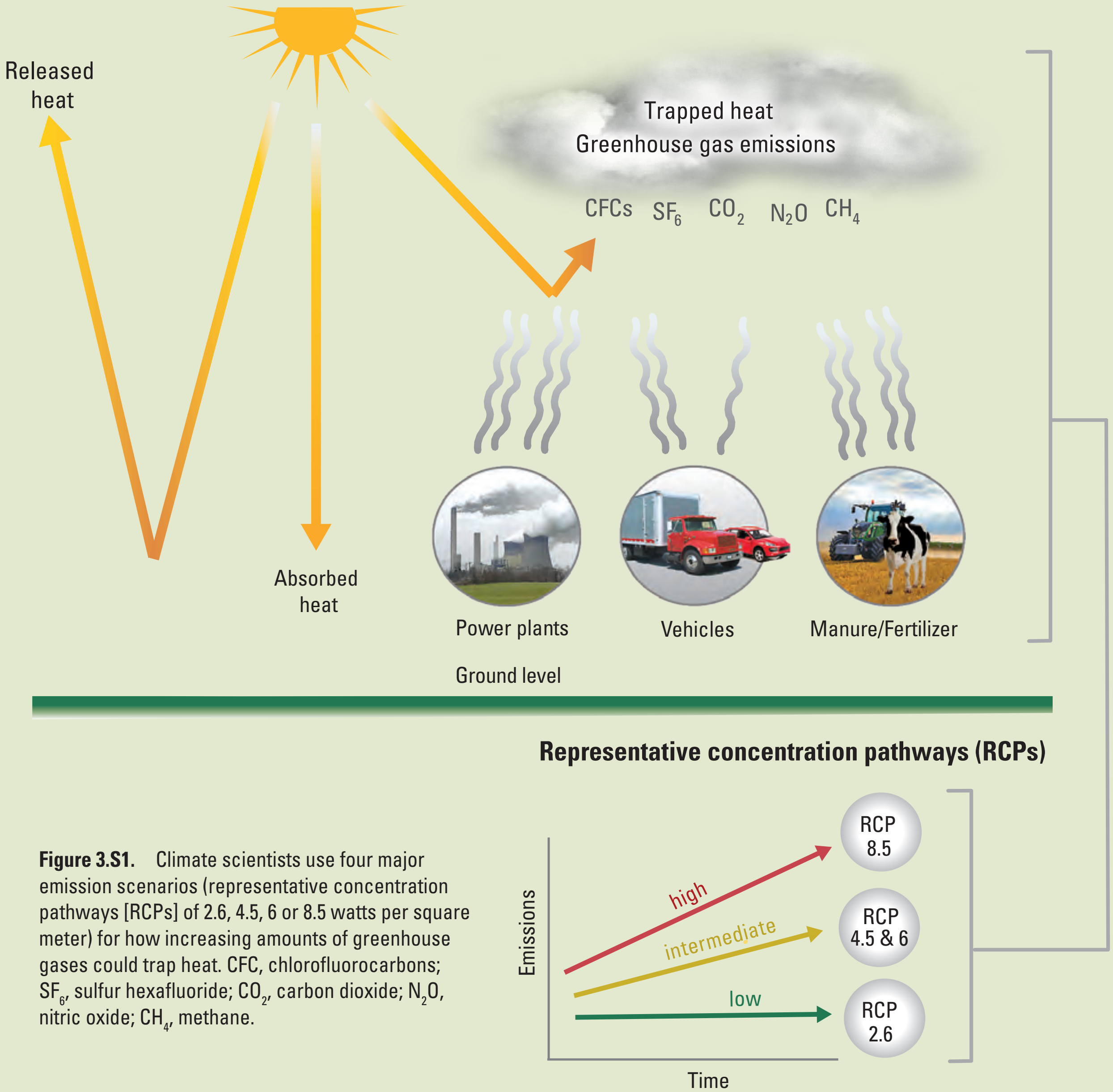


Figure 3.S1. Climate scientists use four major emission scenarios (representative concentration pathways [RCPs] of 2.6, 4.5, 6 or 8.5 watts per square meter) for how increasing amounts of greenhouse gases could trap heat. CFC, chlorofluorocarbons; SF_6, sulfur hexafluoride; CO_2, carbon dioxide; N_2O, nitric oxide; CH_4, methane.

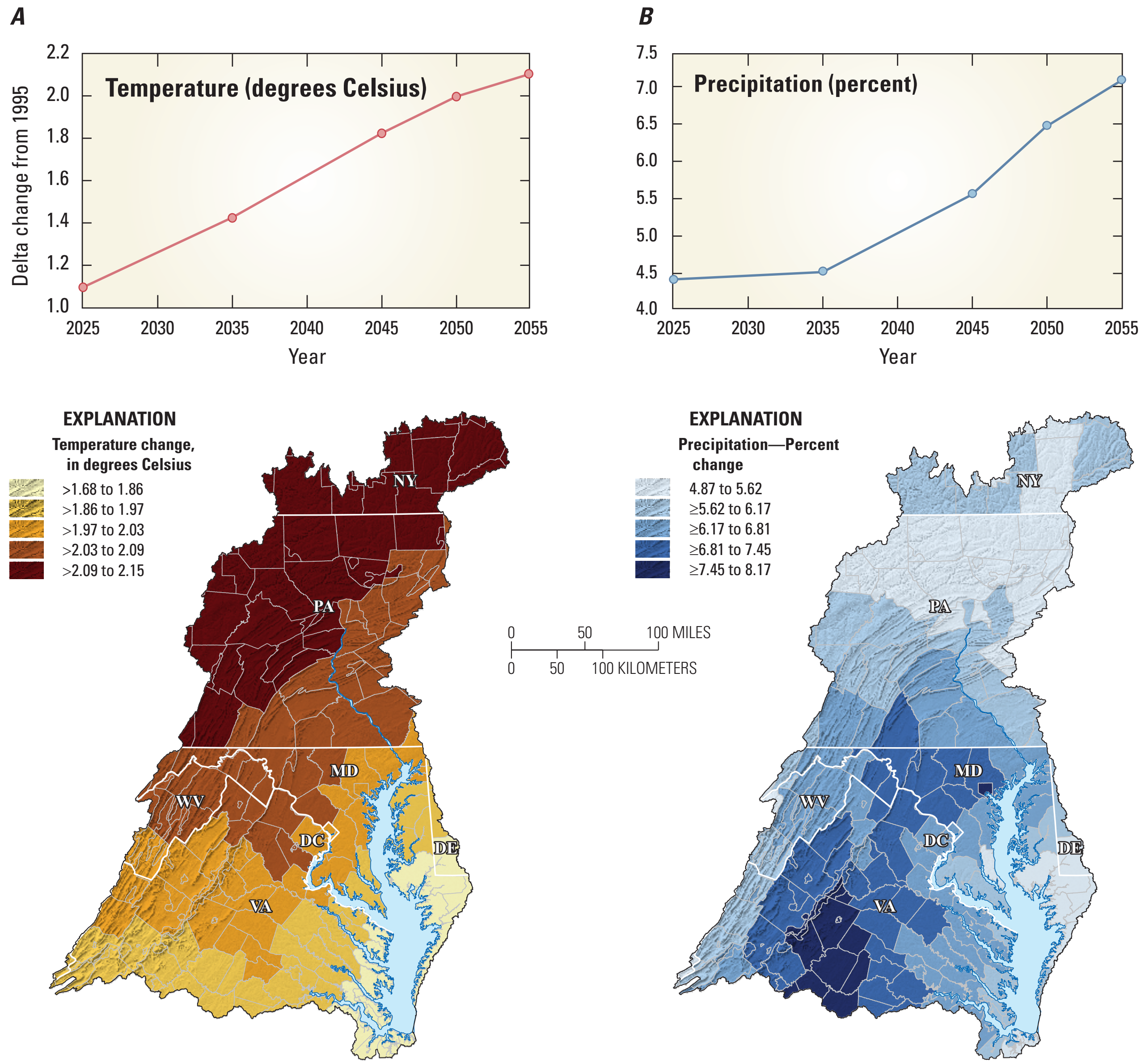


Figure 3.3. The predicted change and spatial variability from 1995 to 2050 for ***A***, mean annual temperature and ***B***, precipitation under the intermediate emission or Representative Concentration Pathway 4.5 (RCP 4.5) scenario for the Chesapeake Bay watershed (Bureau of Reclamation, 2013; Chesapeake Bay Program, 2017). DC, Washington, D.C.; DE, Delaware; MD, Maryland; NY, New York; PA, Pennsylvania; VA, Virginia; WV, West Virginia.

Historical and projected precipitation changes from 1995 to 2050 for the Chesapeake Bay watershed show a 6.3 percent increase in average annual precipitation, with the largest increases occurring in the southern states of Maryland and Virginia (figs. 3.3*B*; Bureau of Reclamation, 2013; Chesapeake Bay Program, 2017). Precipitation is often correlated with streamflow to the Bay and this is important because streamflow regulates the timing of phytoplankton blooms (Najjar and others, 2010). Increased evapotranspiration will bring drier conditions during the summer/fall, with greater variability of intermediate-size intense storms and peak streamflow potentially changing the dynamics of stratification and algal communities in the Bay (Sellner and Kachur, 1987). Almost all climate models concur that wetter conditions than present will occur in the winter and spring, which may increase streamflow and nutrient loads and further feed algal blooms (Hayhoe and others, 2007; Najjar and others, 2010). Additionally, increases in the intensity of winter cyclones are projected because tropical storms are correlated with increases in ocean temperature (Lambert and Fyfe, 2006). These cumulative increases in freshwater flows from basins like the Susquehanna are not only expected to decrease salinity in the upper Bay, but are anticipated to produce changes in nitrogen loads owing to wetter conditions and less denitrification as a result of shorter water residence times before arriving in the Bay (Najjar and others, 2010; Seong and Sridhar, 2017). Furthermore, summer stratification in the Bay is strongly correlated with winter/spring flows from the Susquehanna River and the predicted increases in flows are likely to give way to stratification increases (Najjar and others, 2010).

Predicting future changes in nitrogen yields in response to changes in climate is challenging. For example, the simulated variability in the annual yield of nitrogen exported from corn fields on the Delmarva Peninsula under seven different climate scenarios is shown in figure 3.4. The model period was 28 years and the graph shows the mean ±1 standard deviation of predicted nitrogen yield. In all cases, the annual variability of predicted nitrogen yield within each of the climate change scenarios was greater than the variability among the scenarios. Therefore, it is expected that the effect of different climate scenarios on nitrogen yield would be less than changes observed over time or space at stream monitoring locations.

Adaptation strategies for managing nitrogen impacts to the Chesapeake Bay watershed may be necessary to achieve water quality goals. Urban and regional plans may require updates because warmer environments, increasing population, and changes in land use may bring changes in nutrient loads. Agronomic and conservation practices may have to be adjusted by applying strategies such as shifting planting and cover crop dates (Prasad and others, 2018). Many point sources of nitrogen such as wastewater discharges, combined sewer overflows, and other flood control structures are designed for past precipitation and population regimes and upgrades may be necessary. Some impacts, such as species loss and shifting biodiversity, which affect the nitrogen cycle, may be irreversible (Intergovernmental Panel on Climate Change, 2018). Ultimately, the effects of climate change will depend on human energy consumption, how societies prepare and adapt, and the role that technology plays in mediating the effects of future nitrogen impacts on the Chesapeake Bay.

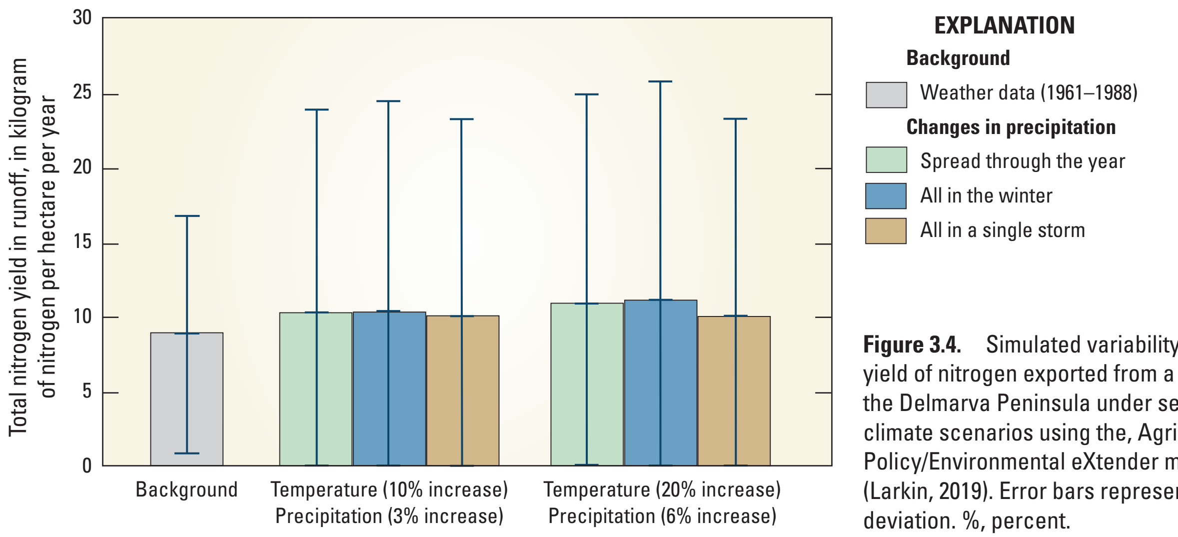


Figure 3.4. Simulated variability in the annual yield of nitrogen exported from a corn field on the Delmarva Peninsula under seven different climate scenarios using the, Agricultural Policy/Environmental eXtender model (APEX) (Larkin, 2019). Error bars represent ±1 standard deviation. %, percent.

A Paleoclimate Perspective

Studying how climate patterns in the distant past have affected nutrient delivery and hypoxia in the Chesapeake Bay can illuminate our understanding of the effects of recent climate change and frame expectations of how future climate change may affect nitrogen loads. Paleoclimatology and paleoecology are fields of investigation that employ analyses of time series recorded in media such as sediment and tree rings to understand past climate extending back hundreds to thousands of years and predating modern instrumented environmental monitoring data. Much has been learned about the history of ecological and climatological change in the Chesapeake Bay watershed through chemistry analysis and plant and animal fossils in age-dated sediment cores from the Bay (Canuel and others, 2017).

Periods of decreased oxygen in the Bay have been identified prior to widespread European colonization in the early 17th century based on measurements of indicators present in sediment (Bratton and others, 2003; Willard and others, 2003). These sediment records represent observations over the past 2,000–3,000 years. Measured changes in sediment isotopic content and in inferred aquatic and terrestrial ecological patterns indicate alternating dry and wet periods that reflect natural climate variability as influenced by multiyear and multidecadal drivers of precipitation and temperature such as the North Atlantic Oscillation (Canuel and others, 2017). Overlying these natural climate signals are increases in the rate of sedimentation in the Bay and related declines in dissolved oxygen and salinity associated with early land clearing driven by European colonization beginning in about the early 17th century (Willard and others, 2003). Continued increases in sedimentation rates are observed that correlate with peak forest harvesting for lumber in the late 1800s and early 1900s. The sharpest and most widespread changes in sediment records are observed beginning in about the 1930s to 1960s that further accelerated in the 1970s, indicating large changes in water quality and more intense hypoxia driven by increasing urbanization and increased application of nitrogen in fertilizer associated with agricultural land use (Cronin and Vann, 2003). The conclusions of the paleoecological and paleoclimatological investigations in the Chesapeake Bay watershed are that natural climate variation can affect precipitation and temperature patterns as well as nutrient delivery to the Bay over decadal time scales. This work shows the potential for anthropogenic climate change as manifested by a greater frequency of large storms and more intense droughts and wet-dry cycles to amplify nitrogen loads to the Bay, which may partly counter management actions to reduce these loads (Cronin and Walker, 2006).

Summary

Because changes in factors such as land use, wastewater management, air quality management, and agricultural best management practices have such a strong impact on nitrogen loads in the Chesapeake Bay watershed, it is challenging at present to quantify the extent to which recent climate change has affected nitrogen export in the Chesapeake Bay watershed. High nitrogen export rates have been associated with large storms (Inamdar and others, 2015), and these storms may be occurring with increasing frequency in the Bay watershed. However, a broad correlation between precipitation amount and stream discharge is not evident across the watershed, despite increasing annual precipitation in the northern part (Rice and others, 2017). Furthermore, many factors associated with climate change favor greater retention or loss of nitrogen, such as enhanced rates of denitrification and nitrogen uptake in aquatic and terrestrial ecosystems (Schaefer and Alber, 2007; Groffman and others, 2018). Focusing exclusively on climate change as reflected by changes in meteorological variables may also overlook other important elements of change, such as sea level rise and the direct ecosystem effects (such as redox chemistry and priming in soils) of increases in atmospheric carbon dioxide concentrations, which have implications for a wide variety of processes that affect ecosystem nitrogen export (Barnard and others, 2005; Song and others, 2018). Nutrient reductions that facilitate submerged aquatic vegetation recovery and reduce hypoxia have also been shown to buffer the effects of coastal marine acidification (Su and others, 2020). Despite the effects of sea level rise or moderate oxygen increases associated with nutrient reductions, hypoxia in the Bay may be overwhelmingly driven by continued warming trends (Ni and others, 2020). Ultimately, the interplay between climate and changing land use will likely govern effects on nitrogen export, and investigations to date show that these two factors are interdependent, necessitating thorough assessments of both to understand patterns of change.

Changes in Hydrology

Chapter 4

By Jeff P. Raffensperger[1], David M. Wolock[1], and John W. Clune[1]

Hydrologic processes, flow paths, and transport of nitrogen are strongly affected by landscape features, including surface and subsurface characteristics such as geology, soils, land cover, and climate (Hamilton and others, 1993; Blomquist and others, 1996; Miller and others, 1997; Shedlock and others, 1999; Bachman and Krantz, 2000; Preston, 2000; Ator and others, 2001). The natural physical, chemical, and biological processes that affect the hydrologic cycle have changed over billions of years. More recent changes in the hydrology of the Chesapeake Bay watershed are the result of centuries of human modification, including changes in land use and land cover, diversions and impoundments including construction and removal of dams, efforts to drain wetlands, irrigation in support of crop production, creation of impervious surfaces, and stormwater management. In addition, withdrawal and use of water in the watershed, for both consumptive and nonconsumptive use, has increased significantly with human population growth and varies spatially (fig. ES.6; Soeder and others, 2007; Dieter and others, 2018). Finally, climate change, including changes in temperature (affecting evapotranspiration and other processes) and the amount, spatial pattern, temporal variability, and intensity of precipitation will impact future hydrology and nitrogen transport in the watershed.

[1]U.S. Geological Survey.

Indian River in the Virginia portion of the Chesapeake Bay watershed. Courtesy of the Chesapeake Bay Program, used with permission.

This chapter describes hydrologic processes (including water storage) and their impact on nitrogen transport and delivery to the Chesapeake Bay. Past and anticipated future changes in those processes are also discussed. The temporal changes and spatial patterns of major water fluxes (precipitation, evapotranspiration, and runoff) are described for three time periods: 1950– 2012 and projections to 2030 and 2060. Future scenarios are based on projected changes in climate. Total runoff to the Bay will be presented for the past and projected for the future. Finally, a discussion of groundwater storage and the groundwater component of streamflow (base flow) highlights the importance of groundwater residence time and its impact on lag times in nitrogen transport within the watershed.

The hydrologic cycle is an endless process linking water in the atmosphere, on the continents, and in the oceans (fig. 4.1). Solar energy drives the hydrologic cycle; gravity and other forces also play important roles. Hydrology is the principal driver of movement or transport of nitrogen in the Chesapeake Bay watershed and needs to be considered at a range of scales (fig. 4.2). The nitrogen load to a body of water is simply the product of the flow of water and the nitrogen concentration; hence, knowledge of both streamflow and nitrogen concentration are the fundamental requirements for quantifying and understanding loads of nitrogen to the Chesapeake Bay (see chap. 2, fig. 2.S1). Hydrologic compartments (groundwater, soil water, streams, lakes, and the atmosphere) store water and nitrogen and may provide opportunities for biological or chemical transformations among the various forms of nitrogen (see Nitrogen Setting of the Chesapeake Bay Watershed).

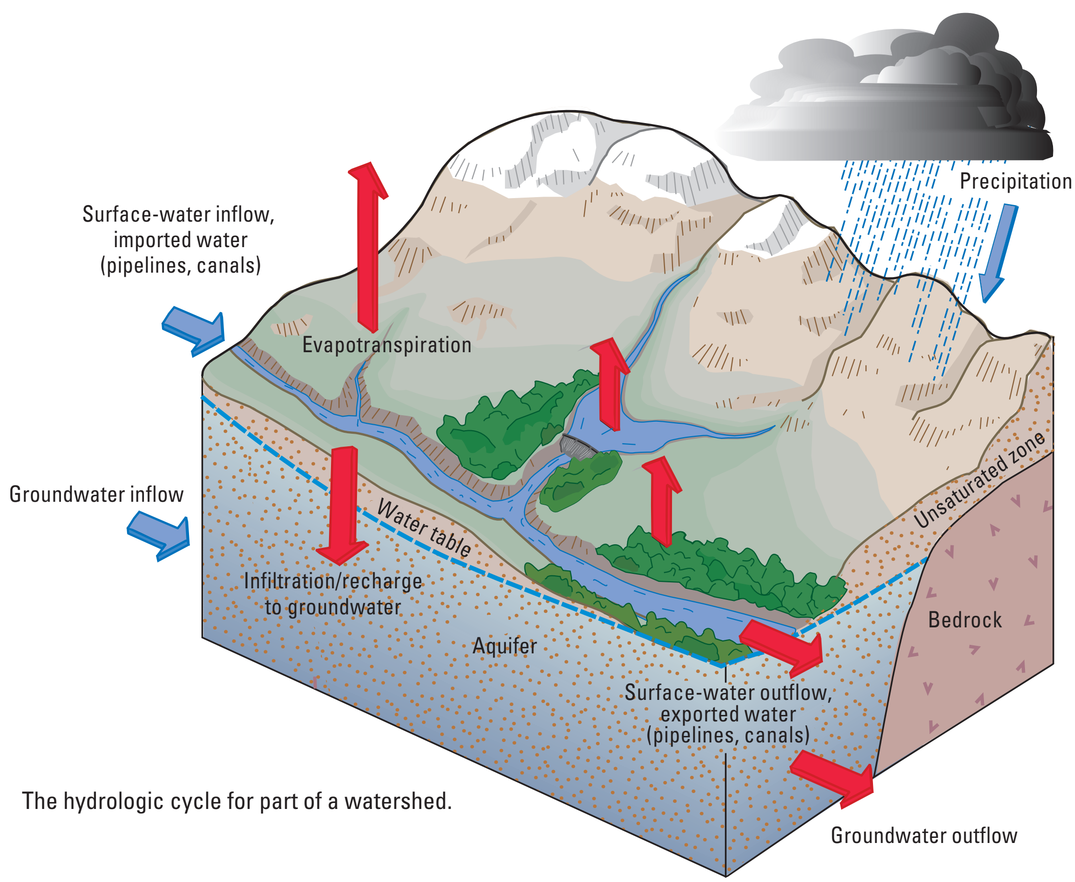


Figure 4.1. The hydrologic cycle for part of a watershed.

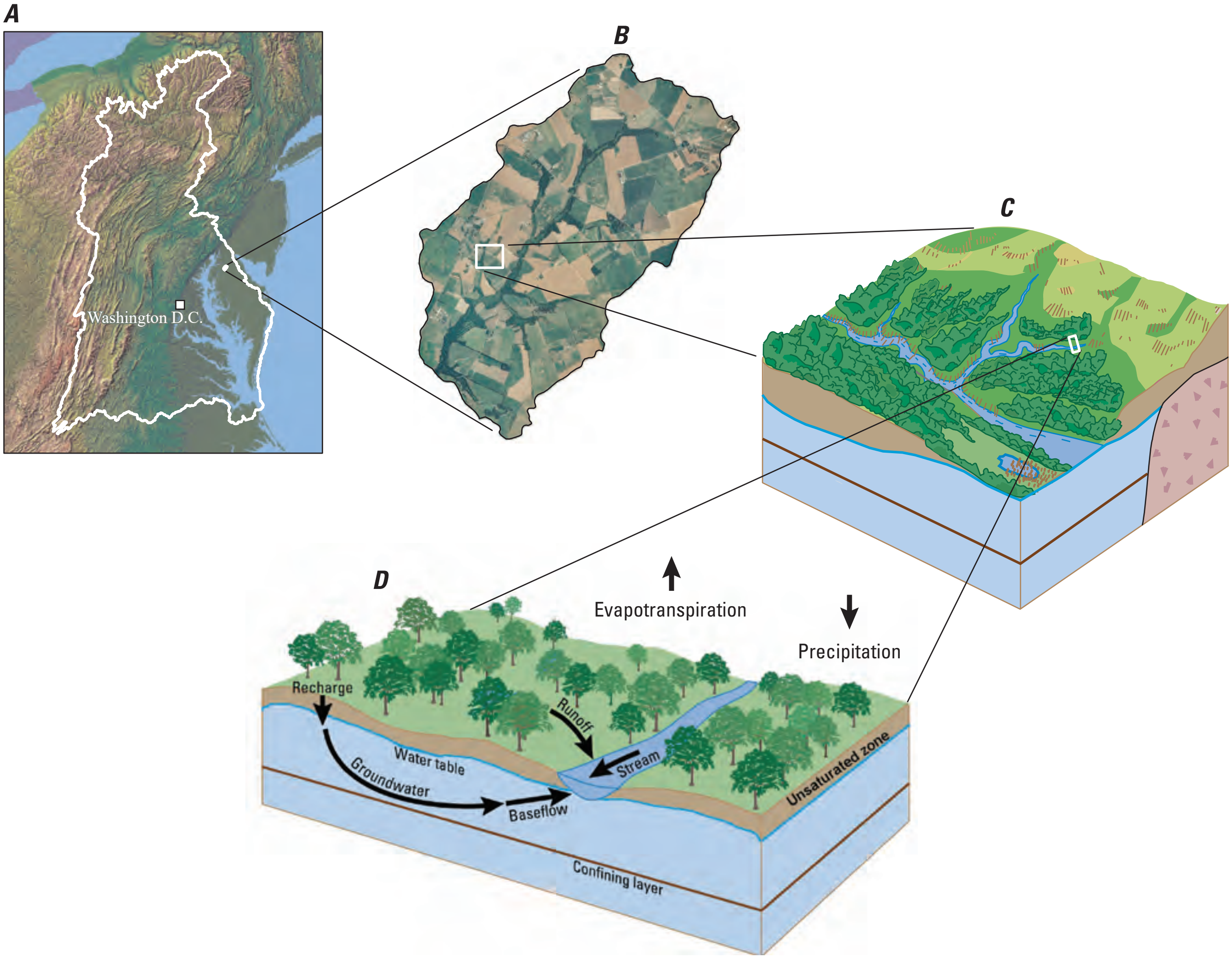


Figure 4.2. Water flow paths and movement are presented at the ***A***, regional scale, represented by the Chesapeake Bay watershed; ***B***, watershed scale, represented by the Morgan Creek watershed, Maryland, ***C***, catchment scale, illustrating the connection between the surface and subsurface; and ***D***, field scale, showing the soil (brown) and groundwater (blue) zones. The arrows represent the water flow paths for the components in the soil-water budget: precipitation, evapotranspiration, runoff, and recharge to groundwater and to stream baseflow (Capel and others, 2018b).

Precipitation onto the land surface is partitioned into evapotranspiration, runoff and recharge (fig. 4.1). There are several flow paths by which precipitation can contribute to runoff, and these different flow paths are important for understanding nitrogen transport. These include direct precipitation onto an open-water surface (stream, lake, estuary), overland flow, shallow subsurface stormflow, and groundwater flow. Overland flow describes water that flows across the land surface and discharges into a stream channel. If overland flow is to occur, water must accumulate at the surface rather than infiltrate into the soil. Overland flow can occur if surfaces are impervious (bedrock, roads, and so forth), the rate of precipitation exceeds that of infiltration, and (or) the ground is saturated. Overland flow in watersheds is one of the most rapid paths that rainfall can follow to the stream channel (Hornberger and others, 2014). Additionally, water that has infiltrated the ground may create localized areas of saturation and, if sufficient permeability and hydraulic gradients exist, induce lateral water movement, a process referred to as shallow subsurface stormflow (Dunne and Black, 1970; Beven, 1981, 2001). This shallow flow path may be moderately fast relative to normal groundwater flow rates, depending on localized heterogeneity in soil permeability, hydraulic gradient, and the presence or absence of macropores (Freeze, 1974; Beven, 1981). Furthermore, the deeper infiltrated water that eventually reaches the water table and the saturated zone beneath becomes groundwater and follows highly variable but relatively slow flow paths before discharging as base flow to streams (fig. 4.3).

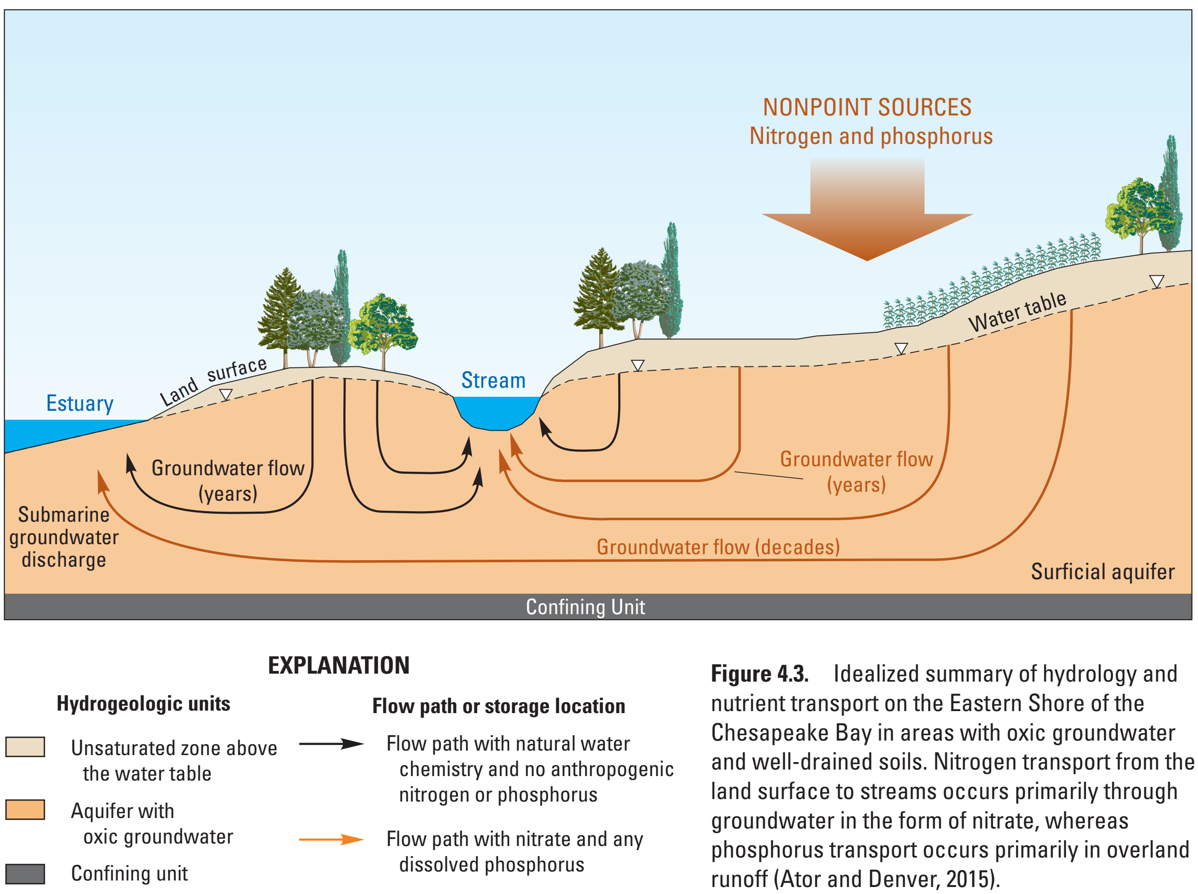


Figure 4.3. Idealized summary of hydrology and nutrient transport on the Eastern Shore of the Chesapeake Bay in areas with oxic groundwater and well-drained soils. Nitrogen transport from the land surface to streams occurs primarily through groundwater in the form of nitrate, whereas phosphorus transport occurs primarily in overland runoff (Ator and Denver, 2015).

The residence time is a measure of how long, on average, a molecule of water spends in a particular hydrologic compartment. Typically, stream and river residence times are shorter (hours to weeks) than groundwater residence times (months to decades). The residence time of groundwater in the Chesapeake Bay watershed varies owing to differences in combinations of rock type and physiographic province (Focazio and others, 1998; Phillips and others, 1999). This has an important bearing on discussions of nitrogen transport within the watershed, and the timing of the environmental response to changes in nutrient management. The resultant groundwater residence times mean source reductions do not always lead immediately to improvements in water quality.

At the time of early European settlement in the 1600s, the Chesapeake Bay watershed was almost entirely covered by forests on a wide variety of soils and the streams, wetlands, groundwater, and nitrogen processing was in balance compared to today (See Historical Setting and chap. 1). The loss of forested habitat had a significant impact on watershed hydrology, enhancing runoff, decreasing recharge, and subsequently affecting nitrogen, sediment, and phosphorus transport. Altered hydrology and transport of nitrogen have continued in the 21st century owing largely to the land-use changes from forested to agricultural and developed land compared to precolonial times. The Bay watershed population grew steadily in the first half of the 20th century, reaching 8.4 million people by 1950. Urbanization and eventually suburbanization altered the hydrologic cycle, creating impervious area that diverts water to streams and rivers at the expense of recharge.

Precipitation nearly equals evaporation in the Bay estuary, and therefore freshwater inputs nearly match the export of brackish water and saltwater (Yang and others, 2015; Marjorie A.M. Friedrichs, Virginia Institute of Marine Science, written comm., 2018). Using a long-term (1980–2012) average for the entire Chesapeake Bay watershed, approximately 1,039 millimeters per year (mm/yr) of precipitation produce 393.1 mm/yr of runoff; the remainder (646.2 mm/yr) is actual evapotranspiration, estimated using the National Hydrologic Model Precipitation-Runoff Modeling System (table 4.1; Markstrom and others, 2015; Hay, 2019a).

Table 4.1. Water budget for the Chesapeake Bay Watershed from 1980–2012.

[Values for October 1, 1980, to September 30, 2012, represent an average for the time period and are based on the Monthly Water Balance Model (Hay, 2019a). Values are given in millimeters per year (mm/yr) and inches per year (in/yr)]

Water-budget component	Annual rate, in mm/yr (in/yr)
Precipitation	1,039 (40.9)
Actual evapotranspiration	646.2 (25.4)
Runoff	393.1 (15.5)

The Past: 1950–2012

From 1950 to 1985, the human population of the watershed increased by 61 percent (5.1 million), from 8.4 to 13.5 million and by 2017 the population was 18.2 million (fig. OV.1; Hopple and others, 2021). Runoff has varied spatially across the watershed, reflecting variations in climate (precipitation, temperature, and evapotranspiration), physiography, land use, and other factors (fig. 4.4). Less runoff occurs in the flatter, more permeable Coastal Plain, especially in areas of southern Virginia and on the Delmarva Peninsula. Population growth has resulted in an increase in impervious surfaces in many locations, causing changes in the hydrology of the watershed (more runoff, less infiltration) and forcing the development of engineering solutions to help manage stormwater in developed areas. Runoff from stormwater causes a number of environmental problems, such as streambank erosion and the transport of excess nutrients from fertilizers, pet waste, and leaf debris into rivers and streams. Stormwater runoff is the fastest growing source of pollution to the Chesapeake Bay (U.S. Environmental Protection Agency, 2021a).

Variations in annual streamflow since 1950 are indicative of changes in landscape characteristics, climate, and anthropogenic modification (including land-use changes, as well as changes in drainage, diversions, and the construction and removal of dams). For example, significant droughts occurred in the 1960s and again in 1999–2002. In general, streamflow to the Bay suggests more wet years and greater variability for the period following the 1960s drought, based on the largest tributary (Susquehanna River at Marietta, PA) (fig. 4.5).

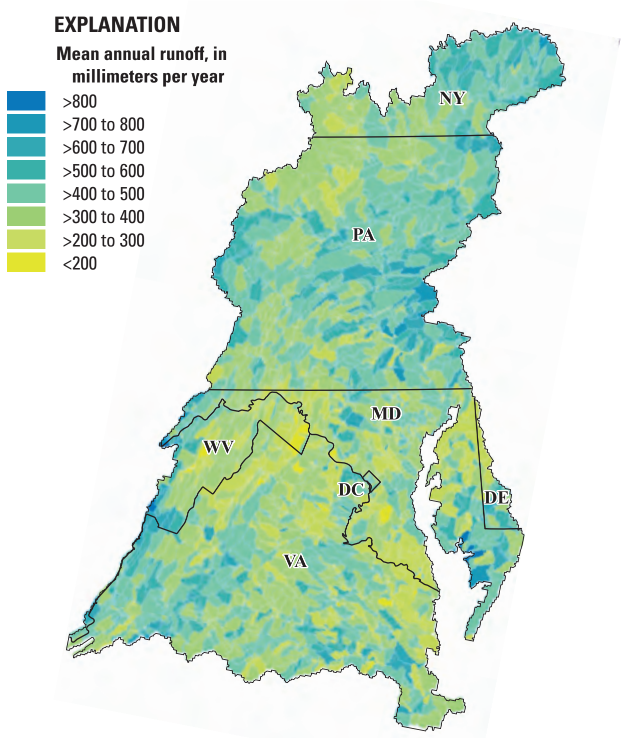


Figure 4.4. Map showing estimated mean annual runoff, 1980–2012, by hydrologic response unit (Flügel and others, 1995), using the National Hydrologic Model Precipitation-Runoff Modeling System and downscaled Global Circulation Model results.

Streamflow varies greatly at short time scales (hours to days) in response to precipitation, including major events such as tropical storms that impact the Chesapeake Bay. Extreme floods on the Susquehanna River over the past two millennia (Toomey and others, 2019), including Hurricane Agnes (June 14-July 6, 1972), had dramatic impacts on the geomorphology, sediment history, and water quality of the Chesapeake Bay. In the aftermath of tropical storm Lee (September 7–15, 2011), concentrations of nitrogen, phosphorus, and suspended sediment were among the highest ever measured at Conowingo Dam at the downstream end of the Susquehanna River Basin in Maryland, where the river flows into the Chesapeake Bay (fig. 2.S2; Hirsch, 2012). The Susquehanna River contributes approximately 51 percent of the total streamflow to the Bay (fig. ES.2*B*; Chesapeake Bay Program, 2017), and therefore exerts a strong and seasonal influence on Bay health (see sidebar–Conowingo Dam, Spring Streamflow, and Predictions of the Chesapeake Bay Hypoxia).

The amount, frequency, and intensity of precipitation increased from 1910 to 1996 in the eastern United States and observed increases were greater in the north compared to the south (Karl and Knight, 1998). Similarly, Rice and others (2017) found that both precipitation and streamflow trends from 1927 to 2014 increased in a south-to-north pattern at representative stations in the Chesapeake Bay watershed. Furthermore, all stations (except one) were projected to have higher annual mean streamflow in 2025 compared to 2014 (Rice and others, 2017).

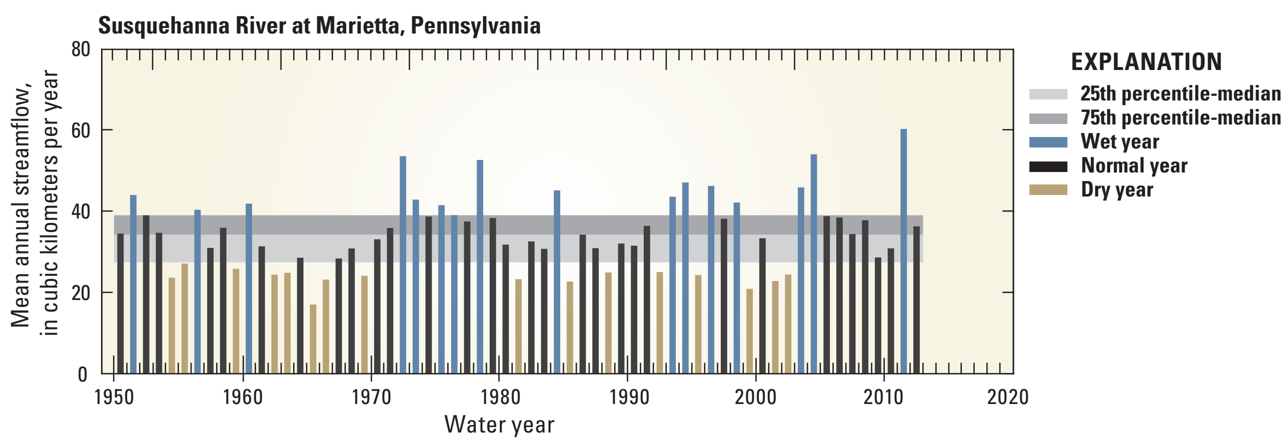


Figure 4.5. Mean annual streamflow at the U.S. Geological Survey stream gage 01576000, Susquehanna River at Marietta, PA, demonstrates interannual variability in streamflow. This station is above the Safe Harbor, Holtwood, and Conowingo Dams on the Susquehanna River. The data show droughts during the 1960s and in 1999–2002, as well as very wet years (2011). Data are calculated for a water year, which runs from October 1 of the previous year through September 30.

Conowingo Dam, Spring Streamflow, and Predictions of Chesapeake Bay Hypoxia

In evaluating hydrologic controls on nitrogen inputs in the Chesapeake Bay watershed, an important factor to consider is the Lower Susquehanna River Reservoir System, which is located on the Bay's largest tributary. The water, as well as the nitrogen, phosphorus, and suspended sediment that are derived from the Susquehanna River Basin and flow past Conowingo Dam (fig. 4.S1) are critically important to the ecologic condition of the Chesapeake Bay. Reductions in nutrients are needed to limit algae blooms that die and sink to the bottom of the Chesapeake Bay and consume oxygen, resulting in hypoxic zones where fish and shellfish cannot survive.

Figure 4.S1. View of the Conowingo Dam on the Susquehanna River in the aftermath of tropical storm Lee. Photograph taken at 4:30 p.m. on September 12, 2011. Streamflow at the time was 220,000 cubic feet per second. Peak streamflow for the flood was 778,000 cubic feet per second at 4:00 a.m. on September 9, 2011. Photograph by Wendy McPherson, U.S. Geological Survey.

Estimates indicate that, on average, the Susquehanna River contributed nearly 41 percent of the nitrogen load to the Chesapeake Bay during 1991–2000 (Hirsch, 2012). As of 1996, the Conowingo Reservoir was annually trapping about 2 percent of the total nitrogen load, as compared with 45 and 70 percent of the total phosphorus and total suspended-sediment loads, respectively (Langland and Hainly, 1997). Seasonal nitrogen loading becomes especially important in the critical spring period, with snowmelt and occasional freshets (a large increase in streamflow caused by heavy rains or melted snow), which may deliver large quantities of nitrogen that can contribute to altered oxygen levels in the Bay. The level of oxygen in the waters of the Chesapeake Bay is a critical factor in determining the health of the Bay's ecosystem. January to May Susquehanna River nitrogen loads have been used to forecast the extent of the summer hypoxic zone, also known as the dead zone (fig. 1.2; Hagy and others, 2004; Murphy and others, 2011), although recent analyses suggest that the summer hypoxic extent is better correlated to watershed-wide nitrogen loads rather than to Susquehanna loads alone (Isabella Bertani, U.S. Environmental Protection Agency, written commun., 2020).

Efforts to control nonpoint sources of nitrogen are hampered because of the lag time between implementation of best management practices (BMPs) and the response in the base flow component of streams (Lindsey and others, 2003). Although agriculture acreage has decreased since 1950, while developed and forested acreage have both increased (fig. OV.1), more intensive agriculture has increased the application of chemical fertilizers to the land (fig. NS.5*A*) and has contributed large amounts of nitrogen to the groundwater system (see chap. 7). A regional groundwater flow model for the shallow aquifer system of the Delmarva Peninsula (Sanford and others, 2012) estimated that residence times are typically less than 10 years near local streams and greater than 100 years near stream watershed divides (fig. 4.6; see sidebar: Groundwater Residence Times).

Groundwater Residence Times

Groundwater discharge as stream baseflow contributes more than half (54 percent) of the total annual flow to streams in the Chesapeake Bay watershed (Bachman and others, 1998). Of the 50 billion gallons (0.19 cubic kilometers) of water that reaches the Chesapeake Bay each day on average, nearly 27 billion gallons (0.10 cubic kilometers) are derived from baseflow (Phillips and others, 1999). Some of the nitrogen applied to the land surface infiltrates into the underlying groundwater system, usually as nitrate. This nitrate is then transported through shallow aquifers and discharges to springs and streams, thereby increasing the nitrogen load to streams. If nitrate is assumed to move at the same rate as the groundwater, the residence time of water can be used to estimate the rates of nitrogen transport. The residence time also provides an estimate of the lag time between implementation of management actions to reduce nutrient loads and a distinguishable improvement in surface-water quality (Phillips and others, 1999; Lindsey and others, 2003).

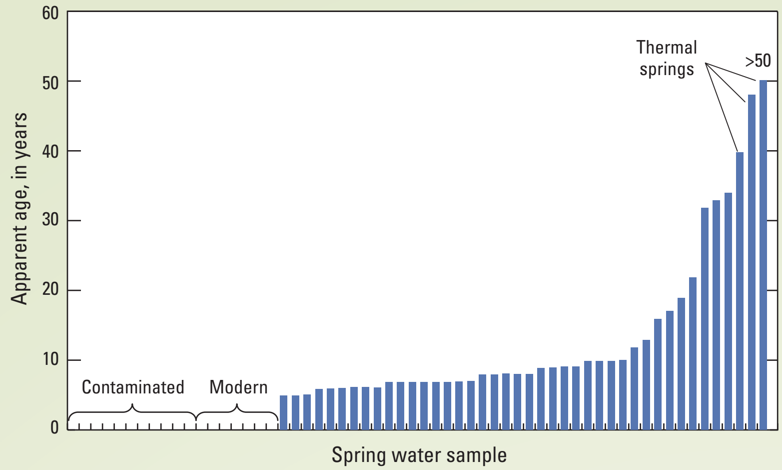


Figure 4.S2. Apparent ages (residence times) of water collected from springs in the Chesapeake Bay watershed in September and November 1996 (Phillips and others, 1999). The apparent ages of water collected from springs sampled ranged from modern (0–4 years) to more than 50 years. The apparent age of water from 75 percent of the springs was less than 10 years, with another 10 percent between 10 and 20 years. The remaining ages were greater than 20 years, and included samples from two geothermal springs, which would indicate the presence of water from deeper groundwater-flow systems. About 20 percent of the samples were contaminated by local sources of chlorofluorocarbons and could not be dated.

Springs are focused areas of groundwater discharge and are useful for understanding residence times. Estimates of residence time based on age-dating of natural spring water samples collected from the Piedmont, Blue Ridge, and other parts of the Bay watershed outside of the Coastal Plain indicate apparent ages from present day to 50 years old or more (figs. ES.4, ES.5, 4.S2; Focazio and others, 1998). More recently, modeling of groundwater flow and transport (Sanford and others, 2012) has further refined this picture, especially for the Coastal Plain (fig. 4.S3). In the Coastal Plain, relatively flat topography and thicker sequences of permeable materials lead to large groundwater storage volumes and older apparent ages.

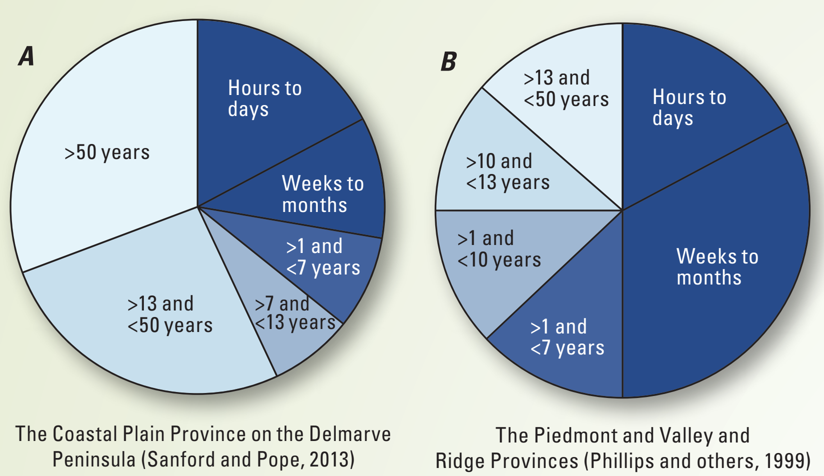


Figure 4.S3. Distributions of estimated baseflow age for the Chesapeake Bay watershed on the ***A***, Maryland and Delaware sections of the Delmarva Peninsula, based on a groundwater-flow model of the region, compared to ***B***, earlier estimates for the Chesapeake Bay watershed using data from the Piedmont and Valley and Ridge Provinces west and north of the Bay. Modified from Sanford and Pope (2013).

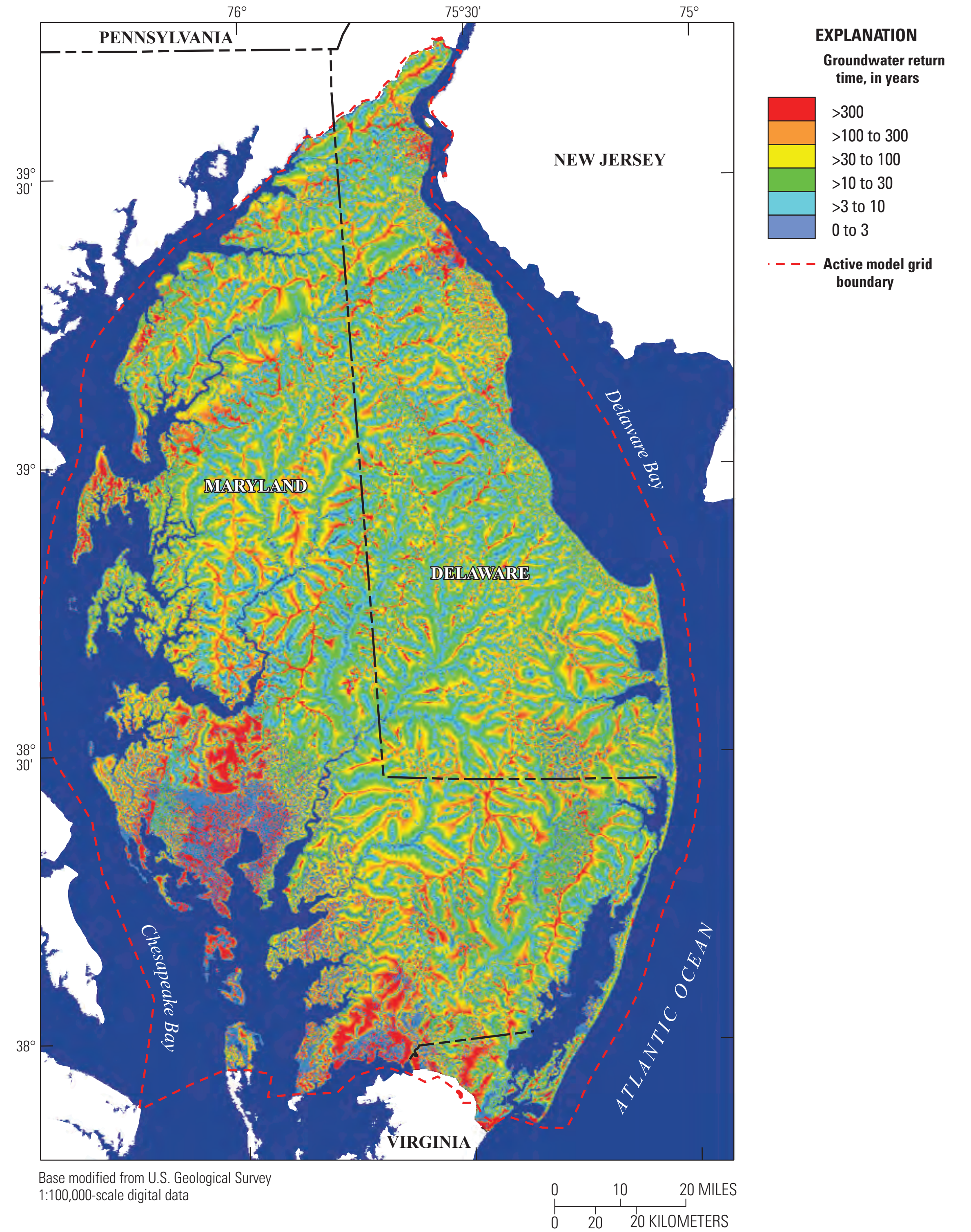
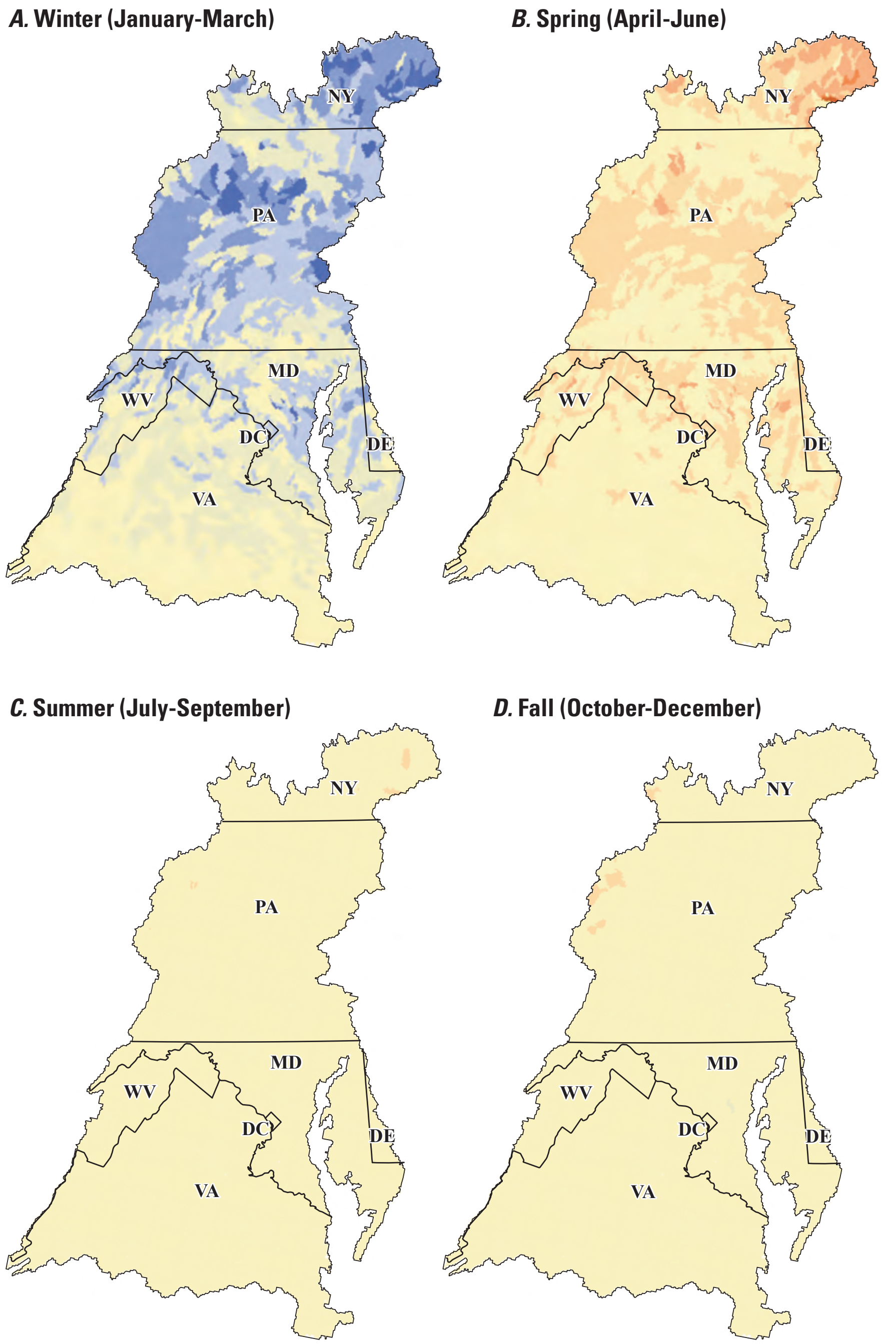


Figure 4.6. Simulated return time or travel time of groundwater traveling from the water table to its discharge location, Delmarva Peninsula (Sanford and others, 2012).

The Future: 2012–2060

Future projections of seasonal precipitation, actual evapotranspiration, and runoff from 214 climate simulations for 19-year periods centered around 2060 (Hay, 2019b) were used to drive additional simulations using the Monthly Water Balance Model (Bock and others, 2017). The summaries of the climate model projections are presented as the 50th percentile (median) of seasonal changes in runoff (fig. 4.7). Runoff is projected to increase towards the northern regions of the Chesapeake Bay watershed during the winter and early spring season (January, February, and March). Similarly, Rice and others (2017) predicted increases in streamflow, considering the median or 50th percentile results, in northern areas. Much of the change in runoff can be explained by changes in precipitation (see chap. 3). Continued warming temperature trends are projected to produce less snowpack in northern regions of the Chesapeake Bay watershed, resulting in earlier runoff in the winter versus the spring (Glas and others, 2019; Ford and others, 2021). This seasonal shift in runoff may affect nitrogen export to streams (Casson and others, 2011; Crossman and others, 2016).



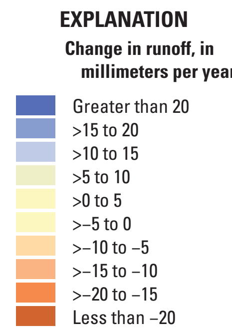


Figure 4.7. Changes in median seasonal average annual runoff in millimeters (mm) for the 50th percentile projected for the future period centered on 2060 (2051-2069) compared to an historical baseline (1980-1999). Based on 214 climate simulations from the downscaled Coupled Model Intercomparison Project-Phase 5 (Hay, 2019b). ***A***, Winter (January-March), ***B***, Spring (April-June), ***C***, Summer (July-September), ***D***, Fall (October-December).

Summary

The components of the hydrologic cycle (precipitation, evapotranspiration, runoff, recharge, groundwater discharge, and streamflow) store, process, and ultimately transport water and nitrogen to the Chesapeake Bay. Changes in the hydrology and transport of nitrogen in the Chesapeake Bay watershed are dependent on environmental characteristics (geology, soils, land cover, and climate), and the result of centuries of human modification that has included changes in land use and land cover, diversions and impoundments, efforts to drain wetlands, irrigation in support of crop production, creation of impervious surfaces, and stormwater management. Nitrogen transport to the Chesapeake Bay is affected by the retention and release of water and nitrogen in surface-water bodies such as reservoirs, lakes, and streams, and by groundwater storage and discharge of water and nitrogen. Each dynamic hydrologic compartment has different characteristics such as volume and residence time, and each has been and will continue to be impacted by development and changes in climate.

Changes in the landscape prior to 1950 (deforestation, dam creation, and early urbanization) impacted the hydrologic cycle and loads of nitrogen. Those changes have accelerated with population growth since 1950. Predictions of future changes in runoff indicate a general increase throughout the watershed, but with notable differences between the northern and southern parts of the watershed and across seasons. Variations in annual streamflow since 1950 are indicative of changes in landscape characteristics, climate, and anthropogenic modification (including land-use changes, changes in drainage, diversions, and construction and removal of dams). For example, significant droughts occurred in the mid-1960s and again in 1999–2002. In general, streamflow to the Bay (as measured at the gage at Marietta on the Susquehanna River, the Bay's largest tributary) suggests more wet years and greater variability for the period since the 1960s drought.

Future projections of seasonal temperature, precipitation, actual evapotranspiration, and runoff from 214 climate simulations were used to drive additional simulations using the Monthly Water Balance Model. Runoff is predicted to increase (based on the future period centered on 2060 compared to an historical baseline) throughout the watershed, with the greatest increases occurring in the northern part of the Chesapeake Bay watershed in the winter/early spring season similar to predictions by Rice and others (2017).

Fields of corn on farms in Loganton, Pennsylvania. Courtesy of the Chesapeake Bay Program, used with permission.

Starrucca Creek in Susquehanna County, Pennsylvania.
Courtesy of the Chesapeake Bay Program, used with permission.

Changes in Atmospheric Deposition of Nitrogen

Chapter 5

By Douglas A Burns[1], Lewis C. Linker[2], Gopal Bhatt[3], and Gary W. Shenk[4]

Hyner View State Park in Clinton County, Pennsylvania. Courtesy of the Chesapeake Bay Program, used with permission.

The atmosphere is an important source of nitrogen to the landscape and waters of the Chesapeake Bay watershed. Atmospheric deposition of nitrogen includes several different chemical forms that are transported to the Earth's surface in wet or dry forms (fig. 5.1). The principal nitrogen bearing constituents in wet deposition (rain and snow) are nitrate (NO_3^-), ammonium (NH_4^+), and organic nitrogen (see Nitrogen Setting of the Chesapeake Bay Watershed). Dry deposition (gases and particles) includes nitrogen oxides (principally nitrogen monoxide [NO] and nitrogen dioxide [NO_2], often termed NO_x), particulate NO_3^-, gaseous nitric acid (HNO_3), ammonia (NH_3), ammonium (NH_4^+), and organic nitrogen (Holland and others, 2005). Organic nitrogen is a complex mix of molecules that originate from both natural and anthropogenic sources such as amino acids and urea (Jickells and others, 2013). Chemically oxidized forms of atmospheric nitrogen deposition such as NO_3^- and NO_x originate primarily as combustion emissions from power plants, industrial sources, and vehicles (fig. 5.1).

[1]U.S. Geological Survey.

[2]U.S. Environmental Protection Agency - Chesapeake Bay Program Office.

[3]Pennsylvania State University - Chesapeake Bay Program Office.

[4]U.S. Geological Survey - Chesapeake Bay Program Office.

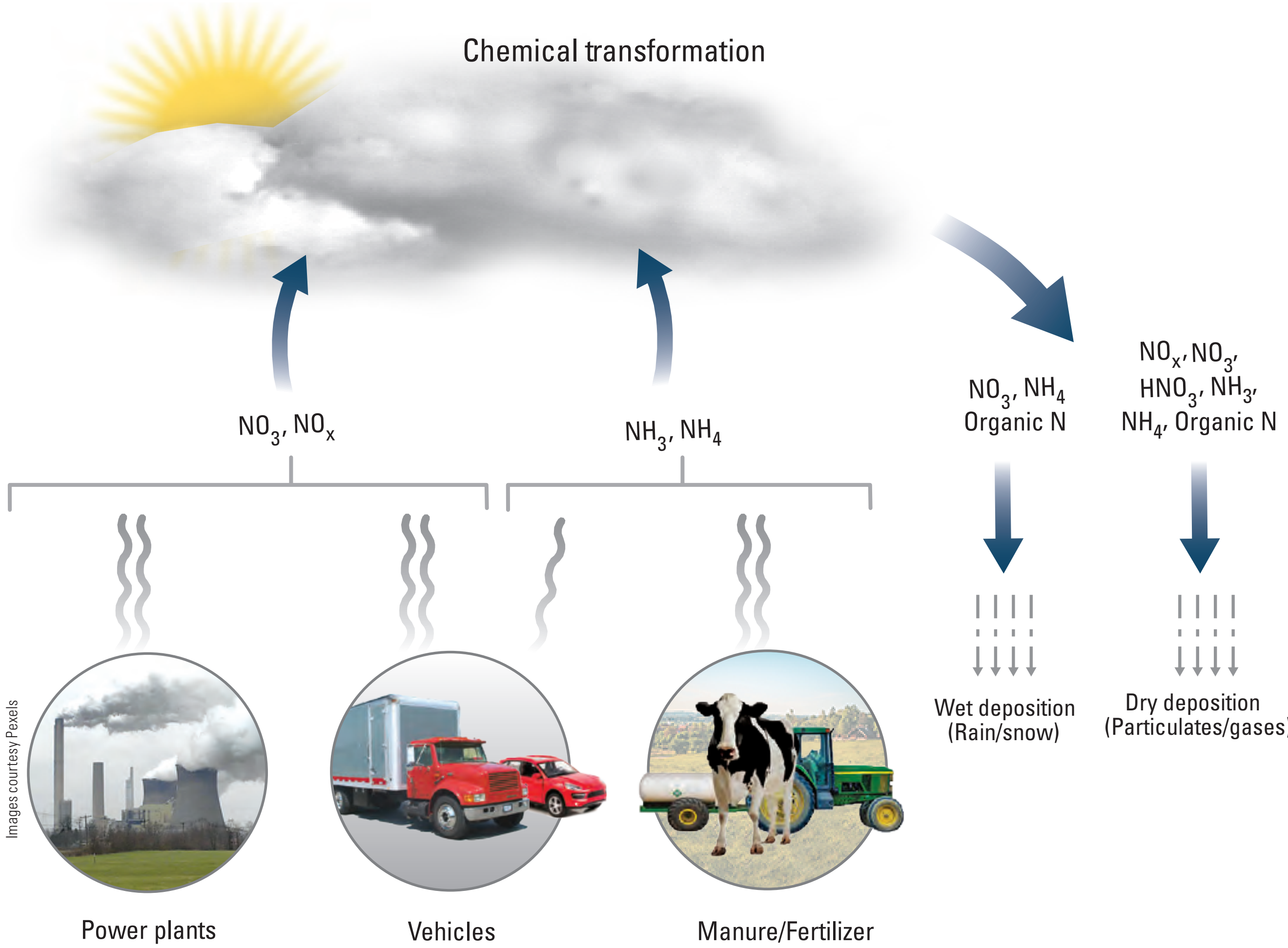


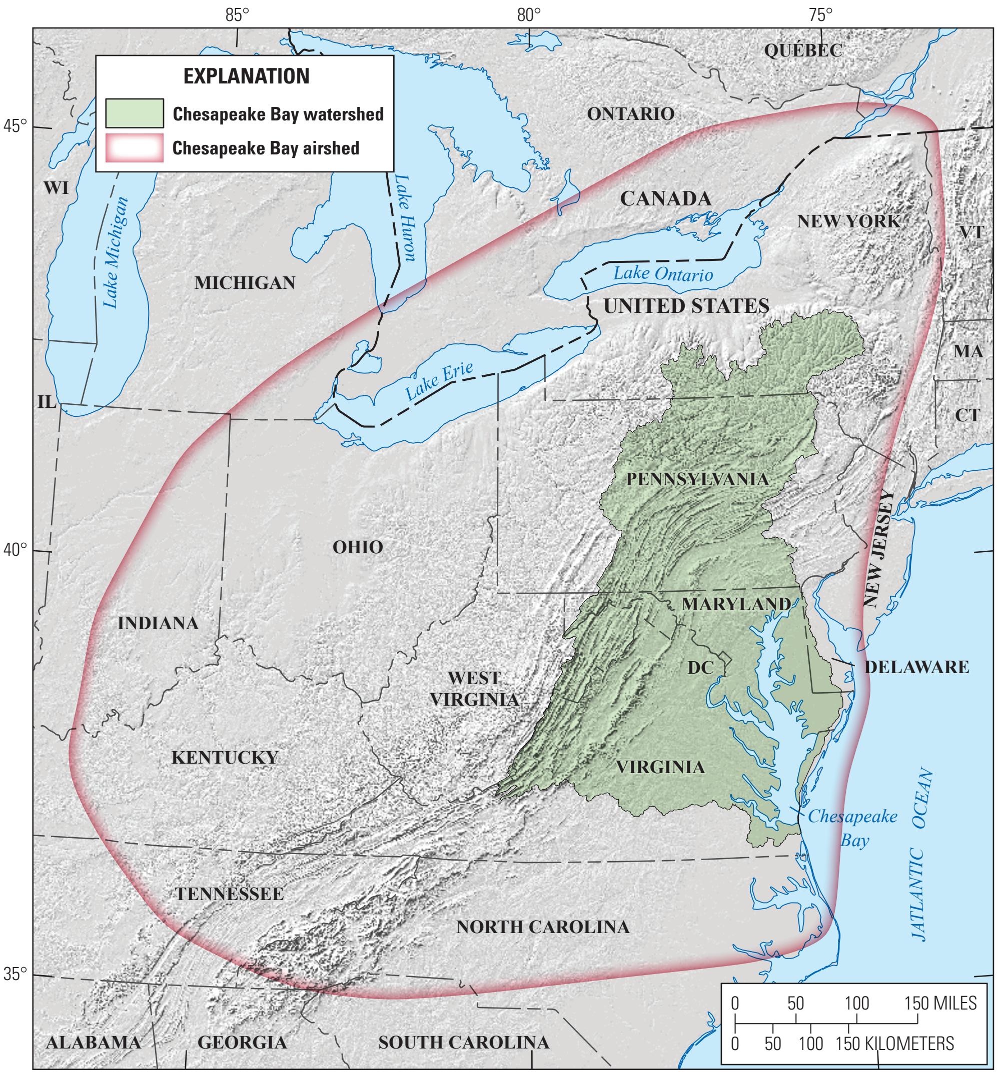
Figure 5.1. Sources and species of atmospheric nitrogen deposition. The illustration shows different sources of nitrogen emissions, which can lead to atmospheric wet and dry deposition downwind from the sources after being transformed and transported in the atmosphere. The most common forms of nitrogen in atmospheric deposition are shown. NO_3^-, nitrate; NH_4^+, ammonium; NH_3, ammonia; HNO_3, nitric acid; organic N, any organic compound that includes nitrogen; NO_x, nitrogen oxides of which nitric oxide and nitrogen dioxide are most abundant.

In the Chesapeake Bay watershed, all forms of atmospheric oxidized nitrogen deposition have decreased since the 1990s, largely as a result of air quality management actions required by the Clean Air Act that have decreased NO_x emissions (fig. NS.5; Linker and others, 2013b; Sickles and Shadwick, 2015; Lloret and Valiela, 2016). Chemically reduced forms of atmospheric nitrogen include NH_3 and NH_4+ in wet and dry deposition, and originate largely from agricultural sources, primarily animal manure and fertilizer, with a minor contribution from vehicles and other emissions. Chemically reduced nitrogen deposition, and across the Bay watershed has shown either a slight decline (Sickles and Shadwick, 2015) or no trend (Linker and others, 2013b) since the 1990s. Organic nitrogen in wet and dry deposition can originate from a variety of sources that include biomass burning, marine aerosols, and other human activities (Jickells and others, 2013).

Atmospheric nitrogen deposition composes about 15 to 32 percent of all nitrogen sources (fig. NS.3; Ator and others, 2011; Linker and others, 2013b; Sekellick and others, 2021) to the Chesapeake Bay, but this fraction has varied over time because atmospheric nitrogen deposition loads have decreased since the 1980s, and nitrogen loads from other sources in the watershed have varied but at different rates over different parts of the watershed (fig. NS.5). Interannual variability of atmospheric nitrogen loads, particularly that of wet deposition, is generally greater than variability of other watershed nitrogen sources and is driven largely by variation in the amount of precipitation (Fisher and Oppenheimer, 1991). Atmospheric deposition presents unique challenges relative to managing other nonpoint nitrogen sources, such as fertilizer, and point-source discharges because air quality policies under the Clean Air Act are implemented at national, regional, and statewide scales based mainly on human health concerns, without direct regard for management of nitrogen in the watershed. Additionally, about half of the oxidized nitrogen emissions that form atmospheric nitrogen deposition originate from outside of the Chesapeake Bay watershed (fig. 5.2; Linker and others, 2013b).

This chapter describes the forms and amounts of atmospheric nitrogen deposition that occur in the Chesapeake Bay watershed, discusses the role of atmospheric nitrogen deposition relative to other nitrogen sources to the Bay, and reports on temporal and spatial patterns of atmospheric nitrogen deposition including the policies that have driven these patterns since 1950. The chapter ends with projections of future atmospheric nitrogen deposition to the year 2050.

The importance of nitrogen in precipitation was first recognized in the 1950s, and the role of nitrogen as an important source to ecosystems came later, in the 1970s (Junge, 1958; Swank and Henderson, 1976). Early estimates focused on wet deposition directly to the Bay and underestimated nitrogen from dry sources and contributions from the watershed (Correll and Ford, 1982; Tyler, 1988; Fisher and Oppenheimer, 1991; Castro and Driscoll, 2002; Howarth, 2007). Accounting for atmospheric nitrogen that is transported from the land is important because of



Albers Conical Equal Area projection.
North American Datum of 1983

Figure 5.2. Estimates of the geographic area, termed "airshed," that contributes to atmospheric deposition nitrogen loads to the Chesapeake Bay watershed from emissions of oxidized nitrogen (NO_x) and reduced nitrogen (NH_3 and NH_4^+). The outer airshed for NO_x is depicted in the figure.

the large size of the watershed compared to that of the Bay, and retention of nitrogen by terrestrial and aquatic biogeochemical processes prior to entering the Bay reduces this load significantly (Fisher and Oppenheimer, 1991). Although met with skepticism, Fisher and Oppenheimer (1991) were among the first to provide comprehensive quantitative estimates to show that atmospheric nitrogen deposition in the form of oxidized nitrogen (25 percent) and reduced nitrogen (14 percent) was a major contributor to the total nitrogen load to the Chesapeake Bay in 1984. Additional studies produced similar estimates and confirmed that atmospheric nitrogen deposition is an important consideration in developing nutrient management strategies (Hinga and others, 1991; Winchester and others, 1995; Howarth and others, 1996; Castro and Driscoll, 2002). Estimates of how much of the total nitrogen load to the Bay consists of atmospheric nitrogen deposition range widely from 15 to 70 percent as atmospheric nitrogen forms, watershed loss rates, availability of measurements, and years considered in the analysis have varied among studies (Paerl, 1995; Jaworski and others, 1997; Meyers and others, 2001; Boyer and others, 2002; Ator and others, 2011; Birch and others, 2011; Linker and others, 2013b). Despite the findings of these studies, atmospheric nitrogen was not widely recognized in the 1987 Bay Agreement (see chap. 2) until persistent decreases from the watershed were observed in the 1990s resulting from enactment of the 1990 Title IV Amendments to the Clean Air Act. Increasing recognition of the important role of atmospheric deposition as a nitrogen source to the Bay became clear in subsequent investigations (Fisher and Oppenheimer, 1991; Howarth and others, 1996; Castro and Driscoll, 2002). As a result, atmospheric nitrogen was incorporated into the Chesapeake Bay Models in 1992 and was first incorporated in load allocations for the 2010 total maximum daily load (TMDL; see chap. 1 sidebar–Largest Total Maximum Daily Load in the Nation; Fisher and Oppenheimer, 1991; Shenk and Linker, 2013).

A challenge for a watershed-based nitrogen management strategy that includes atmospheric deposition is that the airshed, the geographic area whose emissions contribute more than 75 percent of atmospheric nitrogen deposition to the Bay watershed, is about nine times larger than the watershed area for oxidized nitrogen (fig. 5.2). A smaller airshed area exists for reduced nitrogen and extends considerably to the west, north, and south. About half of the atmospheric nitrogen deposition to the Bay watershed originates from inside the watershed (fig. NS.3). Therefore, managing this nitrogen source requires a broader set of stakeholders than just those among the jurisdictions within the watershed. Furthermore, rules and regulations under the Clean Air Act are promulgated for several air pollutants for a variety of reasons but primarily for human health and welfare. Additionally, complex interactions among emission sources and atmospheric chemistry mean that rules designed to address one constituent such as carbon dioxide often have implications for other constituents such as oxidized nitrogen (Driscoll and others, 2015). Following from this broad view, Birch and others (2011) argue that consideration of the nitrogen cycle and its set of cascading steps that deliver nitrogen to the Bay, along with the full array of human health and other environmental effects and costs associated with atmospheric deposition, indicates that controlling atmospheric nitrogen is more cost-effective than controlling other sources such as manure and fertilizer.

Like many other nitrogen sources to the Chesapeake Bay, atmospheric deposition shows considerable spatial variation of about three- to four-fold across the watershed (fig. 5.3). The highest values for total nitrogen from atmospheric deposition of 19.7 to 28.7 kilograms per hectare per year (kg/ha/yr) occur in developed areas such as Washington, D.C., and Baltimore, Maryland, and are shown in dark green in figure 5.3. Such high urban nitrogen deposition is driven largely by high levels of dry, oxidized deposition originating from oxidized emissions from mobile sources. Impermeable surfaces in these developed areas facilitate direct transfer of oxidized nitrogen to nearby surface waters and then to the Bay (Kaushal and others, 2011). Areas of intensive agriculture produce high levels of dry and wet deposition of reduced nitrogen as reflected by values of greater than 20 kg/ha/yr in southeastern Pennsylvania just north of the Bay (fig. 5.3). Broad areas with high levels of 15 to 20 kg/ha/yr of atmospheric nitrogen deposition also occur in the Delmarva Peninsula and a few other areas and are generally associated with high intensity agricultural land use.

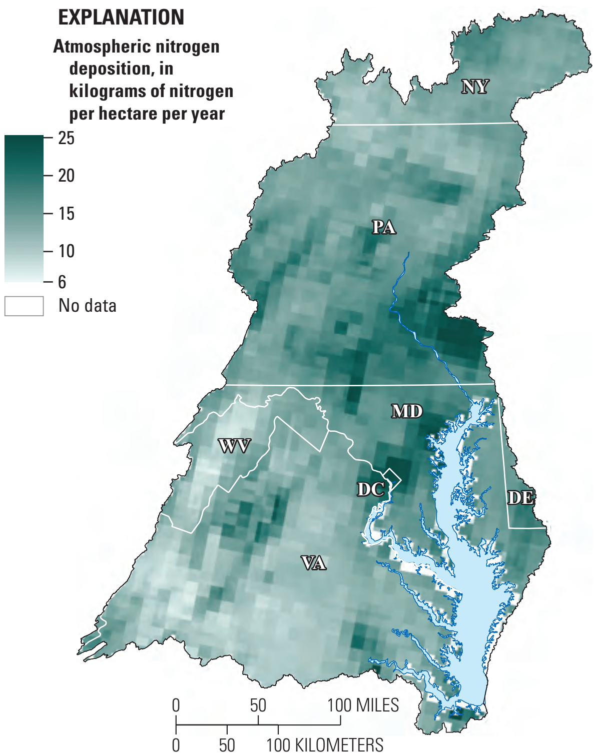


Figure 5.3. Map of annual mean atmospheric inorganic nitrogen deposition to the Chesapeake Bay watershed for the years 1985 to 2005 as simulated in Phase 5.3 of the Chesapeake Bay Model (Shenk and Linker, 2013). DC, Washington, D.C.; DE, Delaware; MD, Maryland; NY, New York; PA, Pennsylvania; VA, Virginia; WV, West Virginia.

The Past: 1950–2017

The earliest measurements of nitrogen species in precipitation within or near the Chesapeake Bay watershed were made in the mid-1950s and generally showed NH_4^+ concentrations of less than 0.1 mg/L and NO_3^- concentrations of 0.3 to 1.0 mg/L (Junge, 1958). Later work in the 1960s included measurements of NO_3^- and NH_4^+ concentrations during several storm events in a forest near Washington, D.C., and showed a general pattern of dilution with precipitation amount (Gambell and Fisher, 1964). Little was known at that time about the principal sources and atmospheric chemistry that governed the spatial and seasonal patterns of atmospheric nitrogen, but later work in the 1970s demonstrated that regional precipitation was acidic owing to the presence of strong acids including nitric and sulfuric acid, with a small contribution from hydrochloric acid (Cogbill and Likens, 1974). Later analysis of the 1950s data (Junge, 1958; Junge and Werby, 1958) confirmed the presence of these same strong acids dating back to that era (Cogbill and Likens, 1974). By the latter half of the 1970s, the first precipitation chemistry monitoring stations were established in the Chesapeake watershed, and in 1978, nationwide monitoring by the National Atmospheric Deposition Program (NADP) commenced (see sidebar–How is Atmospheric Nitrogen Deposition Estimated in the Chesapeake Bay Watershed?; Galloway and Cowling, 1978; National Atmospheric Deposition Program, 2019). The monitoring of dry deposition of nitrogen has a shorter history that only extends back to the late 1980s with the establishment of Clean Air Status and Trends Network sites (U.S. Environmental Protection Agency, 2019b).

How is Atmospheric Nitrogen Deposition Estimated in the Chesapeake Bay Watershed?

Accurate estimates of the atmospheric deposition of nitrogen should consider (1) wet and dry deposition, (2) all known chemical forms, and (3) loads to the watershed and the Bay itself. Approaches have evolved as scientific understanding, the availability of environmental measurements, and models have advanced since the 1980s and 1990s when the first estimates were presented (Tyler, 1988; Fisher and Oppenheimer, 1991). There is no universally accepted approach for estimating atmospheric nitrogen deposition, and methods have varied among investigators (Ator and others, 2011; Linker and others, 2013b). Here, we describe the approach applied in Phase 6 of the Chesapeake Bay Watershed Model in support of the Chesapeake Watershed Agreement.

Wet and dry deposition of nitrogen are estimated separately in the Airshed Model that is part of the Chesapeake Bay Phase 6 Watershed Models. Wet inorganic nitrogen deposition is estimated hourly by a regression model (Grimm and Lynch, 2005) whenever precipitation occurs in the watershed. The regression model was developed with data from 85 stations of the National Atmospheric Deposition Program (National Atmospheric Deposition Program, 2019) and Pennsylvania Atmospheric Deposition Monitoring Network (Pennsylvania Department of Environmental Protection, 2021) that have measured weekly NO_3^- and NH_4^+ loads in precipitation (fig. 5.S1). Predictive variables map wet deposition to the broader landscape and include aspects of land cover, seasonality, emissions, and meteorological data to make hourly estimates. Wet organic nitrogen deposition estimated for the water surfaces only (excludes the watershed) is based on an annual average concentration of 0.05 milligrams per liter (mg/L) and a seasonal range of 0.04 to 0.08 mg/L (Linker and others, 2013b).

Figure 5.S1. A network of atmospheric deposition sampling combined with precipitation measurements provides estimates of nitrogen loads to the Chesapeake Bay watershed. Photograph by Environmental Engineering and Measurement Services, Inc., used with permission.

The Community Multiscale Air Quality Model (CMAQ), a one-atmosphere air simulation model of the North American continent, estimates monthly dry nitrogen deposition of gases and aerosols including NO_x (NO and NO_2), NH_3, NH_4^+, HNO_3, and organic nitrogen to the Bay and its watershed (Byun and Schere, 2006; Pleim and Ran, 2011; Linker and others, 2013b). Monthly values are expressed as daily loads in the watershed model. The CMAQ grid cells in the Phase 6 Chesapeake Bay Watershed model are generally 36 x 36 kilometers (km) across the United States with a nested finer grid of 12 km across the eastern United States covering the Chesapeake airshed and watershed. Because organic nitrogen is assumed to originate largely from wind-driven processes, net dry deposition is assumed only onto water body surfaces which are strong sinks but provide little source, whereas no net deposition is assumed onto terrestrial surfaces for which sources and sinks are in approximate balance (Linker and others, 2013b). Because few measurements of organic nitrogen in atmospheric deposition are available and there is great uncertainty in the full array of sources and sinks, this component has the greatest uncertainty among nitrogen deposition constituents (Jickells and others, 2013).

Extended periods of atmospheric nitrogen deposition measurements are not available in the watershed from 1950 to 1978, prior to establishment of the first NADP stations. However, Gronberg and others (2014) made nationwide spatially gridded estimates of wet deposition of NO_3^- and NH_4^+ for several years during 1955 to 1984 based in part on published values described in the previous paragraph. Furthermore, emissions of oxidized and (or) reduced nitrogen have been estimated for the entire United States and globally at a gridded scale extending from the mid-19th or early-20th century to the late 20th century (Nizich and others, 1996; van Aardenne and others, 2001; Lamarque and others, 2010; U.S. Environmental Protection Agency, 2021b). These estimates generally show gradual increases in oxidized nitrogen emissions beginning in the late 19th or early 20th century that continued until peaking in the 1970s and 1980s. Investigations of the effects of atmospheric deposition on surface water chemistry using simulation models have commonly applied emissions estimates to develop deposition hindcasts for the period prior to the 1970s (Cosby and others, 1985; Tominaga and others, 2010; Zhou and others, 2015). Based on this modeling, atmospheric nitrogen deposition has been generally described as elevated above natural background levels throughout the eastern United States by the 1950s, consistent with the conclusion of Cogbill and Likens (1974) that nitric acid from anthropogenic sources was evident in precipitation by 1950.

Annual loads of NO_3^- and NH_4^+ from wet and dry deposition at four long-term precipitation monitoring sites in the Chesapeake Bay watershed are shown in figure 5.4 along with annual emissions estimates for NO_x and NH_3 for the 12 states that contain the airshed of the Bay to help describe temporal patterns from the 1970s to 2017. These data indicate that NO_x emissions were level or declining slightly from 1970 to 1990, showed a small step decline in 1991, and remained constant through 1997 (fig. 5.4*A*). Beginning in 1998, however, NO_x emissions sharply and persistently declined by 68 percent through 2017. Wet deposition of NO_3^- shows high interannual variation but no persistent trend from 1979 through 1996. Beginning in 1997, despite considerable interannual variation, wet deposition declined sharply by 56 percent through 2017. Emissions of NO_x and wet NO_3^- deposition were synchronous during 1979 to 2017. These patterns are consistent with other analyses of trends in oxidized nitrogen deposition to the Chesapeake Bay watershed and surrounding regions, all of which considered dry deposition (Linker and others, 2013b; Sickles and Shadwick, 2015; Lloret and Valiela, 2016).

Emissions and dry deposition of NO_x in the airshed of the Chesapeake Bay watershed States have corresponding temporal patterns similar to those of emissions and wet deposition of NO_3^- (fig. 5.4*A*). Dry deposition of NO_x decreased by 72 percent from 1989 to 2017, a greater rate of decline than observed for wet deposition of NO_3^-. Airshed emissions of NH_3 are not shown in figure 5.4*B* because although emissions estimates are available, changes in methodologies over time make these data questionable for use in temporal trend analysis (Paulot and others, 2014). The lack of an apparent trend in wet NH_4^+ deposition (fig. 5.4*B*) suggests that NH_3 emissions have changed little over time in the Bay watershed, consistent with the assumptions of other investigators (Linker and others, 2013b; Butler and others, 2016). In contrast, dry NH_4^+ deposition shows persistent declines that appear broadly consistent with those of oxidized N emissions and deposition (fig. 5.4*B*). Other investigators have found similar divergence between temporal patterns of wet and dry NH_4^+ deposition in the eastern United States, which may reflect that particulate NH_4^+ in dry deposition better reflects long-range atmospheric transport than does that of NH_4^+ in wet deposition (Sickles and Shadwick, 2015; Rattigan and others, 2017). Particulate NH_4^+ is linked to sulfate and NO_3^- to balance particle charge, and these two anions act, in part, to limit NH_4^+ particulate concentrations in the atmosphere. So, the apparent decrease in particulate NH_4^+ deposition shown in figure 5.4*B* may better reflect declining sulfur dioxide and oxidized nitrogen emissions from 1989 to 2017 than that of patterns in NH_3 emissions. In contrast, wet NH_4^+ deposition is affected by local scavenging of atmospheric NH_3 and is not well reflected by broad airshed patterns in NH_3 emissions. The 77 percent decrease in NH_4^+ dry deposition is therefore similar to the decrease in wet NO_3^- deposition and dry NO_x deposition from 1989 to 2017.

As atmospheric deposition of oxidized N has declined since the mid-1990s, evidence indicates parallel declines in nitrogen loads delivered to the Bay by streams and rivers. Studies in the headwaters of the Chesapeake Bay watershed where atmospheric nitrogen deposition is the dominant source to streams indicate parallel decreases in NO_3^- loads in wet deposition and stream waters (see chap. 2- Pine Creek section) (Eshleman and others, 2013). A dominant role for declining wet atmospheric NO_3^- deposition in declining stream NO_3^- concentrations and loads can even be discerned at larger basin scales with mixed developed and agricultural land use, suggesting a broad and important role for atmospheric deposition in recovery of the Bay waters (Eshleman and Sabo, 2016).

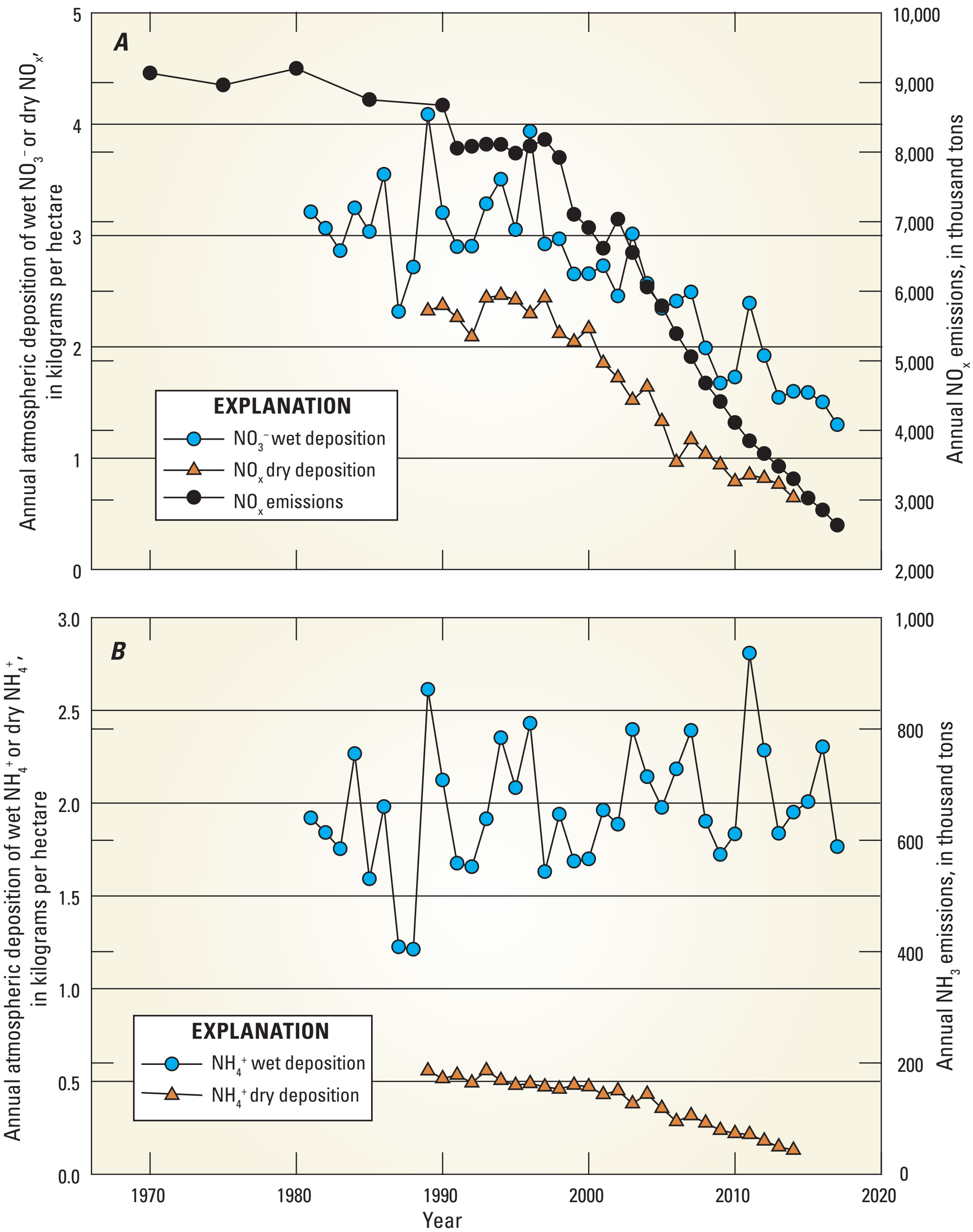


Figure 5.4. Temporal patterns of atmospheric inorganic nitrogen deposition and nitrogen emissions in the Chesapeake Bay watershed. ***A***, Wet NO_3^--N deposition (1979–2017), dry NO_x-N deposition (1989–2017), and NO_x emissions (1970–2017), and ***B***, wet NH_4+-N deposition (1979–2017) and dry NH_4^+-N deposition (1989–2017). Wet deposition is represented by the mean among four sites (MD13, VA13, VA28, WV18; http://nadp.slh.wisc.edu/). Wet deposition is shown only for years when at least three of four sites reported annual values. Dry deposition is represented by the mean of seven sites (ARE128, BEL116, BWR139, CTH110, LRL117, PED108, SHN418, https://www.epa.gov/castnet). Dry deposition values are shown only for years when at least six of the seven sites reported annual values. Emissions data are from the U.S. Environmental Protection Agency (U.S. Environmental Agency, 2021b) and represent the sum of emissions from 12 states that dominate the Chesapeake Bay airshed (Delaware, Indiana, Kentucky, Maryland, North Carolina, New Jersey, New York, Ohio, Pennsylvania, Tennessee, Virginia, West Virginia). For years when state-level emissions data were unavailable but national data were available, a ratio of the 12 states to the entire United Sates was applied. NOx emissions were estimated for 1970, 1975, 1980, and 1985 using the ratio for 1990, and for 1991–95 using the mean 1996–2000 ratio. NO_x are nitrogen oxides of which nitric oxide and nitrogen dioxide are most abundant.

The Future: 2017–2050

Projections of future atmospheric nitrogen deposition require consideration of how rules and regulations currently being implemented as part of the Clean Air Act are likely to affect future oxidized and reduced nitrogen emissions. Future emissions projected under existing or planned Clean Air Act-authorized rules (fig. 5.5) can change with time as legal challenges are made to proposed rules and as new administrations offer new rules. For example, the effects of the Clean Power Plan (Campbell and others, 2019) on emissions were considered in the estimates provided in figure 5.5, but this rule was proposed for repeal by the U.S. Environmental Protection Agency in October 2017 and replaced with the proposed Affordable Clean Energy Rule, which would likely result in slight differences in future trajectories of oxidized and reduced nitrogen emissions (U.S. Environmental Protection Agency, 2018). Furthermore, reduced nitrogen emissions are uncontrolled by Clean Air Act provisions and are dependent on patterns of agricultural land use and management. Further, as solar cells, wind turbines, and other clean energy options continue to become more economically viable relative to conventional thermoelectric generating units, further decreases in oxidized nitrogen emissions are likely beyond those set by air quality standards owing to economic and climate risk motivations.

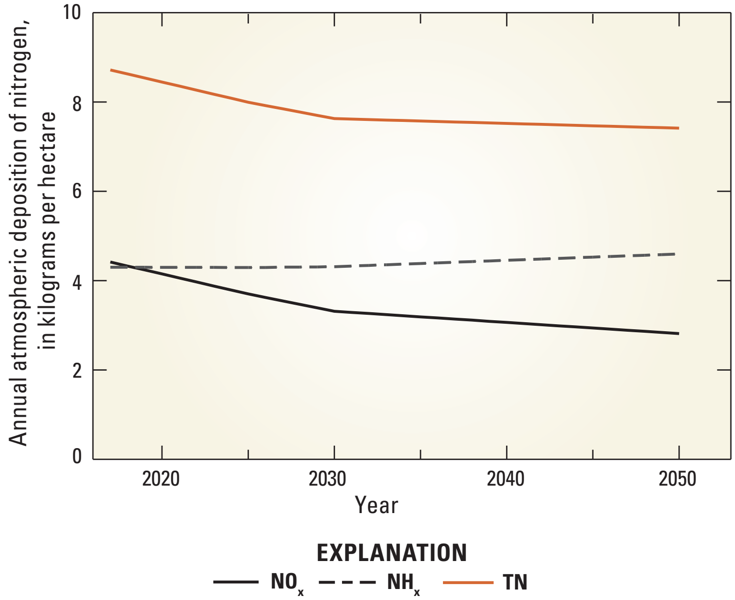
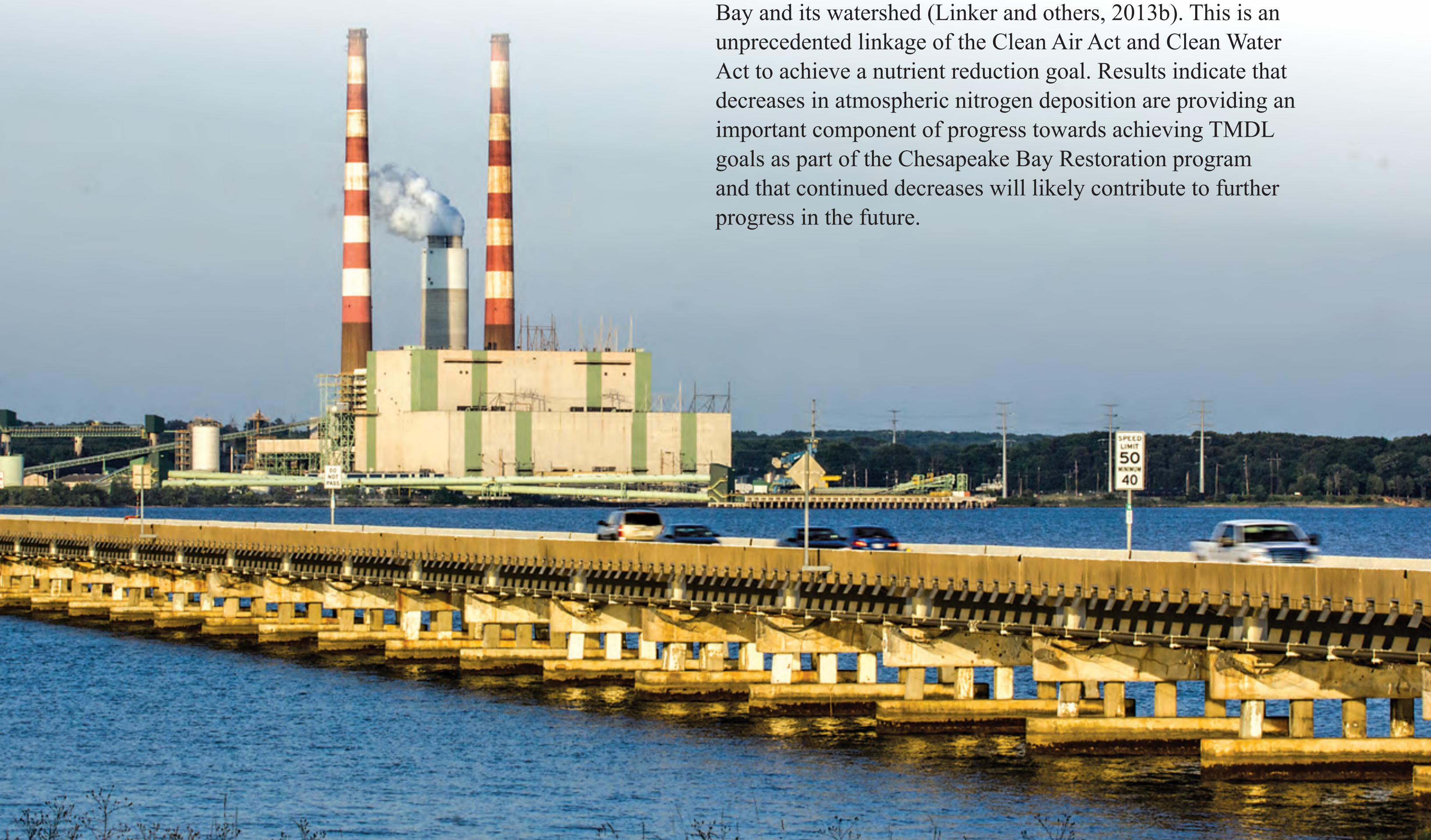


Figure 5.5. Projections of future atmospheric nitrogen deposition to the Chesapeake Bay watershed for 2017–2050. These values are based on Community Multiscale Air Quality (CMAQ) Modeling System model runs for the years 2017, 2025, 2030, and 2050. Linear interpolation was applied to provide estimates for the intervening years. Values are the mean of deposition estimates among all land segments in Phase 6 of the Chesapeake Bay Watershed Model. Climate variation was removed in these projections by applying meteorological conditions for 2011 to the 2017, 2025, and 2030 projections and long-term mean conditions to the 2050 projections. NO_x, nitrogen oxides of which nitric oxide and nitrogen dioxide are most common; NH_x of which ammonia and ammonium are most common; TN, total nitrogen.

Despite uncertainties in future emissions and atmospheric deposition of nitrogen, the model results shown in figure 5.5 indicate a projected decrease of 15 percent in inorganic nitrogen deposition from 2017 to 2050. This change is expected to be driven largely by a 36 percent decrease in oxidized nitrogen deposition, whereas a slight increase of 7 percent is expected in reduced nitrogen deposition. This analysis is based on a constant climate assumption that is not likely given projections of future climate change in the Chesapeake Bay watershed (see chap. 3; Najjar and others, 2010). In a recent investigation, Campbell and others (2019) explored the impact of projected future changes in annual precipitation on wet nitrogen deposition in the watershed. Despite increased precipitation forecasted for 2050 compared to recent years (Hurrell and others, 2013; Powers and others, 2017), Campbell and others (2019) indicate only modest increases in nitrogen wet deposition. Although future climate change may affect many aspects of the global nitrogen cycle beyond just changes in precipitation amount, these results demonstrate that provisions of the Clean Air Act that affect future oxidized and reduced nitrogen emissions are likely to be the strongest drivers of future atmospheric nitrogen deposition to the Bay watershed (Campbell and others, 2019).

Morgantown Generating Station in Charles County, Maryland. Courtesy of the Chesapeake Bay Program, used with permission.



Summary

At the time of the first measurements of NO_3^- and NH_4^+ in precipitation within or near the Chesapeake Bay watershed in the 1950s, atmospheric deposition was already elevated above background levels largely as a result of emissions from coal-fired power plants and mobile sources. However, atmospheric deposition was not generally recognized as an important nitrogen source to the Chesapeake Bay watershed until the early 1990s, when a study estimated that more than 30 percent of the load may originate from this source. Investigations that followed largely confirmed these original estimates, though values varied depending on a variety of assumptions and the period of investigation. Since the mid- to late-1990s atmospheric nitrogen deposition has declined by more than 70 percent in the watershed, driven by provisions of the Clean Air Act that resulted in decreasing oxidized nitrogen emissions that continue to the present (2019). Further decreases are expected through 2050. Atmospheric nitrogen deposition presents unique challenges relative to controlling other nitrogen sources because the contributing airshed extends outside of the Chesapeake Bay watershed, and therefore involves a broader array of stakeholders. Nonetheless, the 2010 TMDL that provides nitrogen source allocation requirements for stakeholders, included a goal of 7.1 million kilograms of annual atmospheric nitrogen deposition to the Bay and its watershed (Linker and others, 2013b). This is an unprecedented linkage of the Clean Air Act and Clean Water Act to achieve a nutrient reduction goal. Results indicate that decreases in atmospheric nitrogen deposition are providing an important component of progress towards achieving TMDL goals as part of the Chesapeake Bay Restoration program and that continued decreases will likely contribute to further progress in the future.

Changes in Land Use

Chapter 6

By Peter R. Claggett[1]

A partly completed residential development borders forested land in Waldorf, Maryland. Courtesy of the Chesapeake Bay Program, used with permission.

Understanding the characteristics of the landscape and how humans use and manage the land is essential for understanding and managing the movement of nitrogen in the Chesapeake Bay watershed. Landscape characteristics include land cover (for example, impervious surfaces, herbaceous grasslands, and tree canopy) and land use (for example, forests, urban areas, golf courses, and cropland), all of which have very different spatial distributions and associated amounts of nitrogen sources (figs. NS.4 and NS.5). Centuries of settlement by Europeans and other immigrants have transformed most of the Coastal Plain and Piedmont physiographic provinces from forest to cropland, pasture, and residential and commercial development (see Historical Setting). Owing to their isolation from ports, poor soils, and rugged terrain, the Appalachian Plateau and the slopes and ridges of the Valley and Ridge physiographic province remain mostly forested (figs. ES.4*A* and ES.5).

In the United States, most land use decisions are local, made by landowners, private individuals, corporations, or local public agencies (Theobald and others, 2000). These land use decisions and their effects on water quality are influenced by the socioeconomic, cultural, regulatory, and policy conditions of their time and are relevant today because they affect the erodibility and nutrient content of soils even in areas that currently appear forested. Therefore, understanding the impacts of land use on the export of nitrogen from the watershed to the Bay requires understanding the historical context for changes in land use.

[1]U.S. Geological Survey - Chesapeake Bay Program Office.

The Past: 1950–2017

No maps or quality aerial images exist to characterize land use conditions in the Chesapeake Bay watershed during the 1950s. Sohl and others (2016) used historical data from the Census of Agriculture and the Decennial Census of Population and Housing to estimate historical land use conditions. Data from that study indicate that there were 3.3 million additional acres (13,355 square kilometers) of agricultural land in 1950 compared to 1985 (Sohl and others, 2016). Urbanization over this period accounted for only 20 percent of the decline in agricultural land, whereas the majority either reverted or was actively managed as natural land (grassland, shrubland, and forest).

From 1950 to 1985, the human population of the watershed increased by 61 percent (5.1 million), from a total of 8.4 to 13.5 million persons (fig. 6.1). During this time, population growth generated land use activities that produced excessive nutrient and sediment inputs to rivers and streams, furthering the declining health of the Bay (see chap. 1). Following World War II, public investments in highway infrastructure coupled with rising incomes, increasing automobile ownership, and negative perceptions of inner cities, led to the creation and proliferation of suburbs (fig. 6.2). New schools and commercial strip malls accompanied residential growth. Residential and commercial developments built from the 1950s through the 1970s had comparatively minimal controls on stormwater and wastewater. National legislation mandating secondary treatment for wastewater treatment plants was not enacted until the 1972 Clean Water Act and national legislation on managing stormwater runoff was not enacted until the 1987 amendments to the Clean Water Act (National Research Council, 2002, 2009).

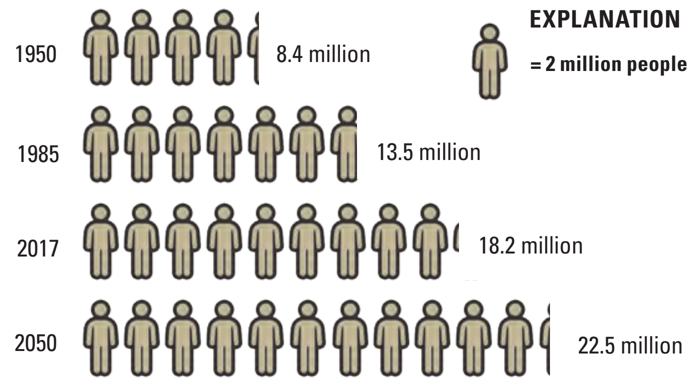


Figure 6.1. Past and projected human population increases in the Chesapeake Bay watershed from 1950 to 2050 (Hopple and others, 2021).

The 1950s were also the beginning of a revolution in agricultural yields through the availability of affordable chemical fertilizers, insecticides, and herbicides, as well as improvements in crop genetics and mechanization. Nationally, corn production levels increased from about 40 to 100 bushels/acre (0.27 to more than 0.67 kilograms per square meter) from the mid-1950s to the early 1980s and continued to increase to more than 150 bushels/acre (1.01 kilograms per square meter) through 2012 (U.S. Department of Agriculture, 2019b). Anthropogenic nitrogen inputs from manure and inorganic fertilizer increased by 92 percent from 1950 to 1982 and declined by 6 percent (2,323 million kilograms) from 1982 to 2012 (Keisman and others, 2018).

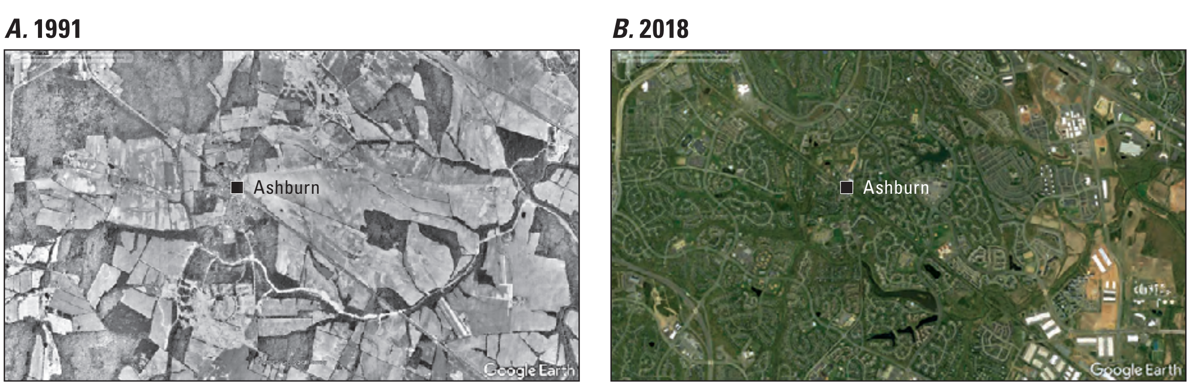


Figure 6.2. Residential and commercial development around Ashburn, Virginia from ***A***, April 1991 to ***B***, April 2018. Base map data from Google, 2015.

The commercial chicken broiler (young chickens raised for meat) industry on the Delmarva Peninsula grew substantially in the 1950s, fueled by the demand for meat, the abundant local supply of menhaden (forage fish in the herring family) for feed, and the integration of poultry production chains (vertical integration) (see chap. 7 sidebar–The Rise of Poultry on the Delmarva Peninsula). On an annual basis, hundreds of millions of broilers are grown in the Bay watershed (Chesapeake Bay Program, 2017). Poultry manure is rich in nitrogen and is applied as fertilizer for corn in fields throughout the Delmarva Peninsula. Land in the Shenandoah Valley and southeastern Pennsylvania (for example, crops and hay fields) also receive extensive manure application from their local animal agriculture (fig. NS.4*C*). The dramatic increase in poultry numbers has increased the amount of manure produced (Keisman and others, 2018). The increasing production of manure and decreasing area of row-crop agriculture since 1950 may have resulted in higher applications of manure, possibly applied in excess of crop nutrient requirements, and could have led to higher transport of manure-derived nitrogen into ground and surface waters (see chap. 2 sidebar–Legacy of Nitrogen in Groundwater; Chesapeake Bay Program, 2017; LaMotte, 2015).

Over the period from 1985 to 2017, human population in the watershed increased by 35 percent or 4.7 million persons (fig. 6.1), while anthropogenic nitrogen inputs were reduced by 32 percent (–117 million pounds or –53 million kilograms) (Chesapeake Bay Program, 2017). The majority of nitrogen reductions resulted from regulatory controls on point sources (for example, wastewater treatment plants and electric power plants) and mobile sources, like automobiles and other vehicles (Shenk and Linker, 2013; Chanat and Yang, 2018; Ator and others, 2019). Reductions in nitrogen yields (kilograms per hectare [kg/ha]) for urban nonpoint sources (for example, lawns, impervious surfaces, and septic systems) were also estimated over this period, despite increases in urban area (Ator and others, 2019). Although reduced nitrogen yields may be associated with best management practices and other human activities (Ator and others, 2019), increases in denitrification rates associated with increases in temperature and precipitation may have contributed significantly to the reductions (Chanat and Yang, 2018). Currently, most of the excess nitrogen exported to the Bay originates from nonpoint sources such as nitrogen leaching from cropland, lawns, septic systems (see sidebar–Household Nitrogen Footprint and Septic Systems), and runoff from roads and rooftops (Chesapeake

Household Nitrogen Footprint and Septic Systems

The food produced for, and consumed by, people is responsible for about two-thirds of a household's nitrogen footprint (Leach and others, 2012). Once consumed, nitrogen in food becomes a waste product that is either transported through sewer pipes to a wastewater treatment plant or treated locally through an onsite or community septic system (fig. 6.S1). If a household is served by sewer and a wastewater treatment facility, 80 to 95 percent of the total nitrogen in wastewater is removed before it is discharged to streams or directly to the Bay (Hertzier and others, 2010). If a household relies on an onsite septic system, roughly 30 to 75 percent of total nitrogen is removed through soil attenuation depending on soil characteristics and proximity to waterways (Swann, 2001; Costa and others, 2002; D'Amato, 2016). Although denitrification coupled with regular pumping can increase septic system nitrogen-removal rates, few onsite systems currently meet these qualifications (D'Amato, 2016).

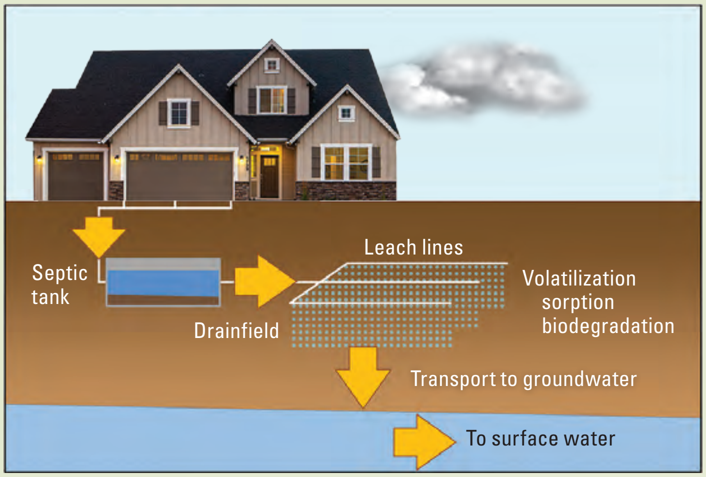


Figure 6.S1. Nitrogen in food becomes a waste product that is treated locally through an onsite or community septic system in many rural areas of the Chesapeake Bay watershed. There are more than 1.9 million septic systems in the Chesapeake Bay watershed that collectively contribute 3.55 million kilograms of nitrogen to the Bay per year on average (Chesapeake Bay Program, 2017). Image courtesy of Laural A. Schaider, used with permission.

Bay Program, 2017). Therefore, greater focus on management of nonpoint sources of nitrogen has been necessary as the limits of cost-effective technology have begun to be reached for point sources.

The first comprehensive long-term estimate of land use conditions, both monitored and modeled, for the Chesapeake Bay watershed spans the period from the mid-1980s to 2025 (Claggett and others, 2013). These data show that land use in the watershed has been and remains mostly natural, composed primarily of deciduous and mixed terrestrial forests (fig. ES.5*A*). Forests, wetlands, grasslands, and other natural land uses contribute the least amount of nitrogen to the Bay on a per-area basis. Agriculture is the second largest land use, composing 23 percent of the watershed in 1985, and contributes the most nitrogen to the Bay on a per-area basis (fig. NS.6; Chesapeake Bay Program, 2017). The majority of cropland in the watershed is planted in corn and soybean, often in rotation with wheat. Of all the agricultural land uses, animal feeding operations contribute the most nitrogen (about 1,700 kilograms per hectare per year [kg/ha/yr]), whereas hay, pasture, and alfalfa contribute the least (about 7–9 kg/ha/yr) (Shenk and Linker, 2013).

From the mid-1980s to 2017, the watershed has become more developed and less agricultural and forested (fig. 6.3). Developed lands in the watershed consist of roads, buildings, lawns and landscaped areas, recreational areas, golf courses, and lands undergoing construction. Impervious surfaces compose about 27 percent of developed lands, whereas the remainder are pervious and consist mostly of turf grass (Claggett and others, 2013). Of all developed land uses, lands undergoing construction contribute the most nitrogen (42 kg/ha/yr), followed by impervious surfaces (18 kg/ha/yr), then lawns and other pervious surfaces in developed areas (14 kg/ha/yr) (Shenk and Linker, 2013).

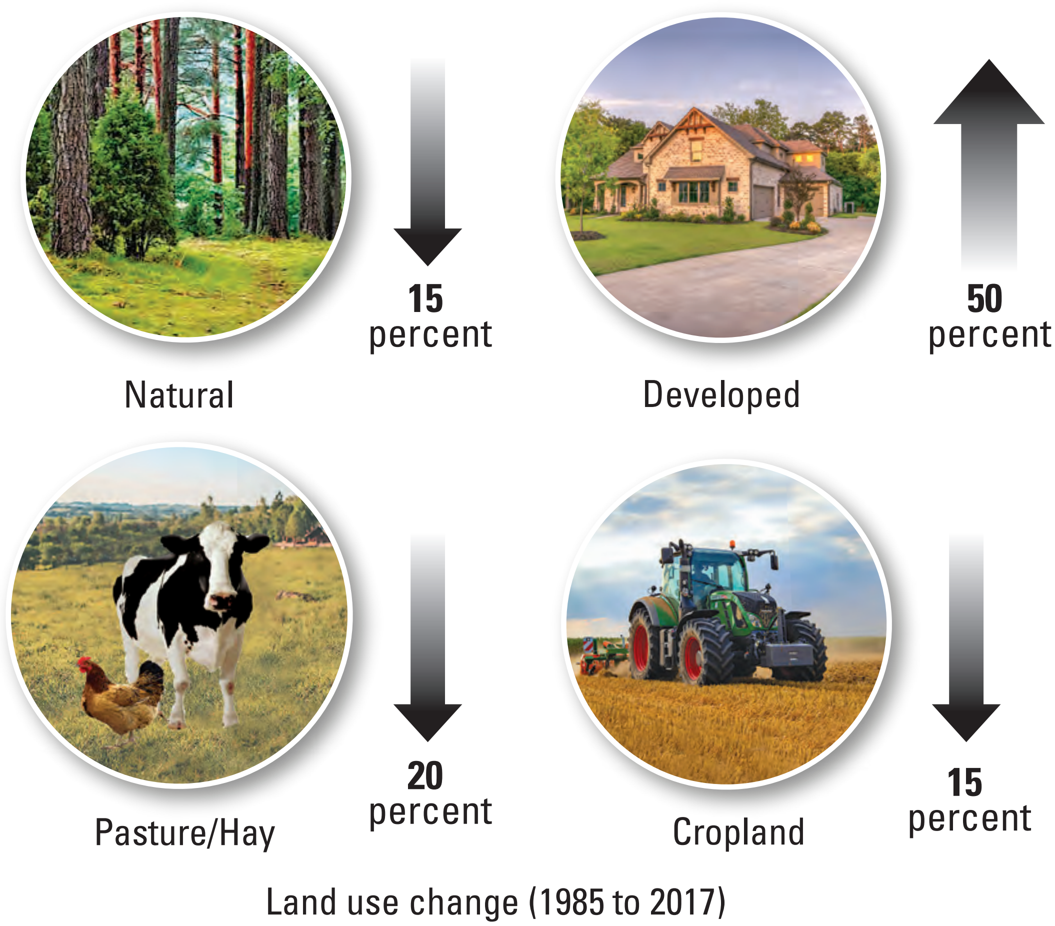


Figure 6.3. Percent land use change in the Chesapeake Bay watershed, 1985–2017 (Chesapeake Bay Program, 2017). Photographs copyright Pexels, used with permission.

The relation between developed lands and nitrogen, however, cannot be simply explained by looking at the extent of the various developed land uses. Development is associated with industrial and municipal point sources of pollution, stationary and mobile sources of atmospheric nitrogen, septic systems, and stream bank and bed erosion exacerbated by storm runoff from impervious surfaces (Gellis and other, 2015; Cashman and others, 2018; Chesapeake Bay Program, 2020). Although these sources are not exclusive to residential and commercial development (a farm utilizes energy produced by power plants; operates trucks, cars, and tractors; and is typically served by an onsite septic system), most mobile sources, impervious surfaces, and septic systems are directly related to residential, commercial, and industrial development. Since 1985, development has occurred mostly at the expense of cropland or pasture (fig. 6.3), which may explain why increases in development throughout the watershed are associated with overall reductions in nitrogen load from urban nonpoint sources (Ator and others, 2020). It should be noted, however, that nitrogen loads from septic systems have increased by 1.8 million kilograms since 1985 along with the number of septic systems (Chesapeake Bay Program, 2017) (see sidebar–Household Nitrogen Footprint and Septic Systems).

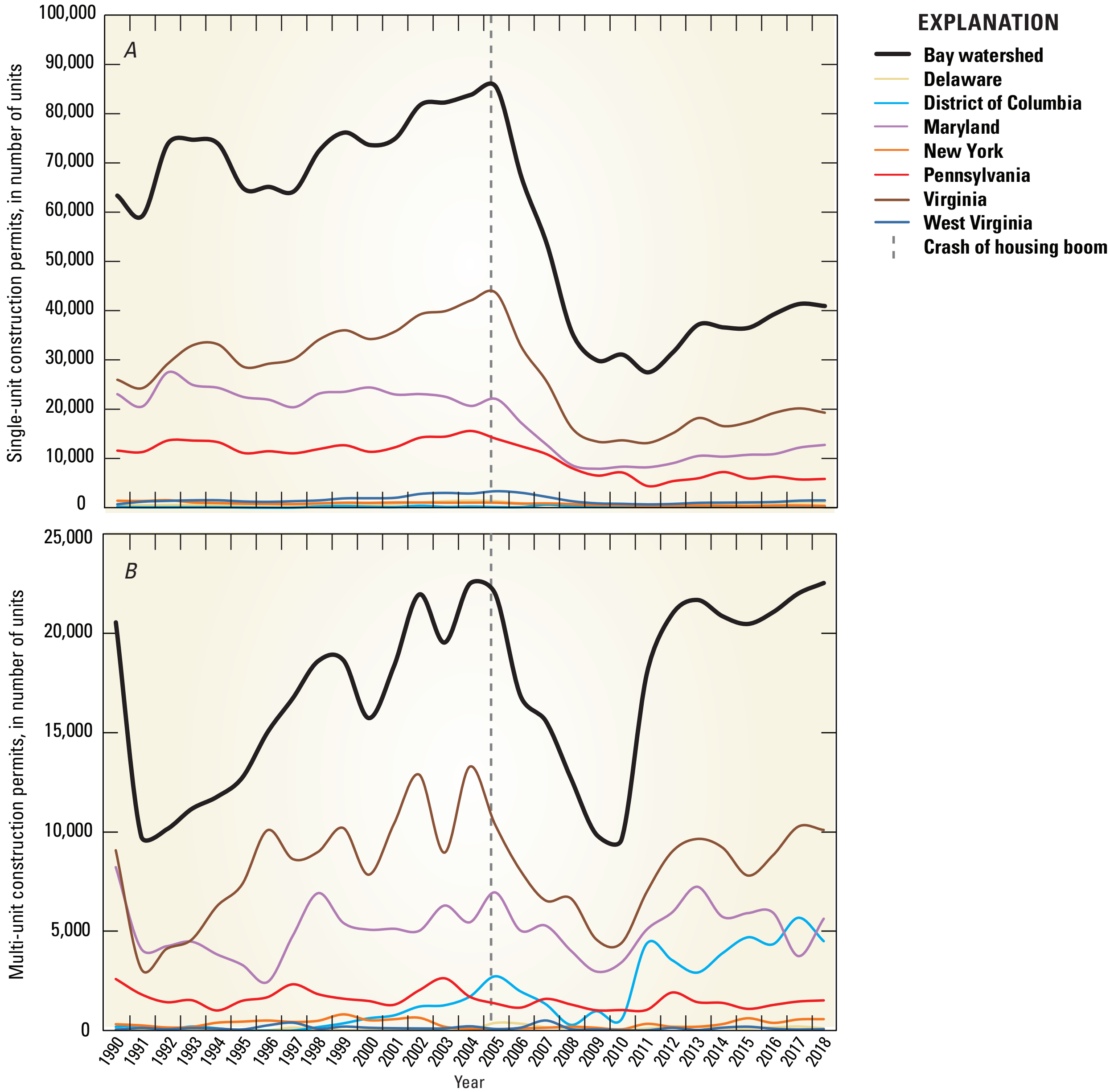
The Future: 2017–2050

When envisioning the future, it is helpful to recognize that changes in the socioeconomic or regulatory conditions that influence land use decisions are complex and can take years or decades to manifest. For example, the national environmental legislation of the 1970s and the formation of the Chesapeake Bay Program in 1983 both followed more than a decade of studies and public concern over pollution and declining environmental conditions (see chap. 1). In contrast, the housing boom from the late 1990s to the burst in 2006, resulted in a steep decline in new construction (fig. 6.4). Even by 2017, new single-family home construction had yet to recover to levels experienced in the 1990s and early 2000s. Although phenomena such as this housing bubble or the recent COVID-19 world pandemic are difficult to predict, long-term trends in demographics, mobility, technology, and sociocultural preferences provide clues about plausible future land use trajectories.

Demographic trends indicate that the population living in the Bay watershed will likely continue to grow by more than 1 million persons per decade through 2050, increasing from 18.2 million (2017 estimate) to 22.5 million persons (figs. 6.1 and OV.1). If the trends of the 1980s through 2010s continue, then future growth will be focused in suburban and adjacent counties, with an emphasis on developing large-lot, single-family homes (fig. 6.5; Blumenberg and others, 2019). Large-lot residential development consumes the most land per-capita compared to other types of development, resulting in large losses of both forests and farms, and will increase nitrogen loads in formerly forested areas and either maintain or decrease nitrogen loads in formerly farmed areas (Ator and others, 2020).

Despite these trends, land use patterns over the next three decades may diverge from those of the past. The Millennial generation, born between 1981 and 1996, represents the baby boomers of the 21st century based on their overall numbers and size relative to other age groups (Frey, 2018). They are distinctly more ethnically and racially diverse and are more likely to prefer living in cities compared to older generations (Frey, 2018). Cities provide cultural and culinary entertainment, transportation amenities, a variety of housing options and employment opportunities, and tend to have more diverse populations compared to suburban and rural areas (Pew Research Center, 2018). Although rural communities that lack opportunities or amenities and older established suburbs are experiencing population declines which may continue into the future, some cities are witnessing a renaissance and others will too if affordable housing, high-quality schools, and low-crime neighborhoods are available (Blumenberg and others, 2019).

Owing to the scarcity of undeveloped land, growth within cities often includes infill development on vacant lots and the redevelopment or revitalization of existing structures and developed lands. These types of development have the lowest per-capita impact on nitrogen loads compared to all other forms because they are composed of mixed-use, multilevel, and multiunit structures and are typically served by public sewer (Kramer and Sobel, 2014). Infill and redevelopment mostly occur in cities with increasing land values and transportation costs (McConnell and Wiley, 2010).



Figures 6.4. Trends in ***A***, single-unit, and ***B***, multi-unit residential construction permits, 1990–2017 (U.S. Census Bureau, 2016).

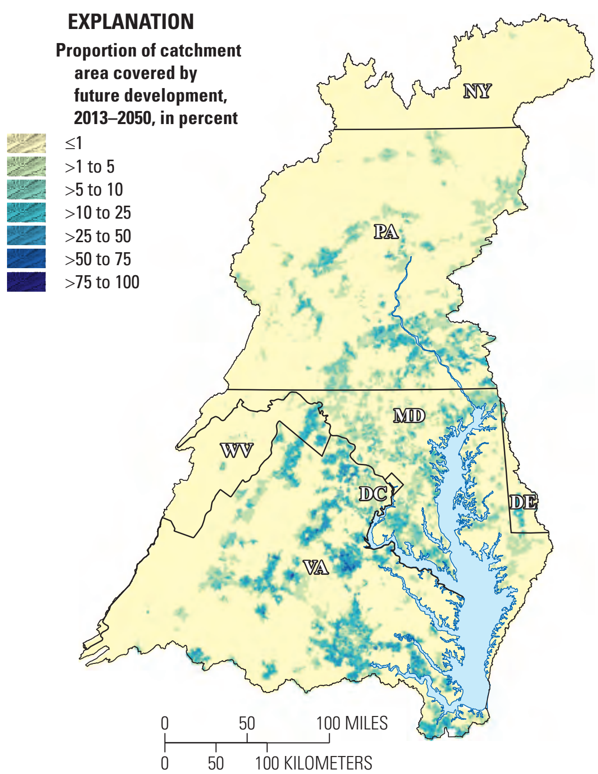


Figure 6.5. Future development in the Chesapeake Bay watershed, 2013–2050. Shading represents the proportion of catchment area covered by future development (Chesapeake Bay Program, 2020). DC, Washington, D.C.; DE, Delaware; MD, Maryland; NY, New York; PA, Pennsylvania; VA, Virginia; WV, West Virginia.

Infill and redevelopment rates for counties and cities in the Bay watershed, estimated using the Chesapeake Bay Land Change Model, average approximately 35 percent and range from below 10 percent in remote rural counties to more than 95 percent in cities such as Arlington, Virginia; Baltimore, Maryland; and Washington, D.C. (Claggett and others, 2014).

In agriculture, new technologies such as precision and vertical farming, hydroponics, aeroponics, aquaponics, manure-to-energy facilities, and changing consumer preferences for sustainably managed local and organic products hold promise that agriculture will evolve towards more efficient management strategies with sustained yields coupled with reduced losses of nitrogen. Growth in the sustainable energy sector (for example, solar, wind, and geothermal) may also facilitate this transition, helping farmers to reduce costs, retire marginal lands, and diversify revenue while reducing pollution (Beiter and others, 2017; Schultz and others, 2021).

Summary

The past does not have to be a prologue to the future. Changing consumer preferences, emerging technologies, land use planning and conservation, and aggressive implementation of controls on nonpoint source pollution from agriculture could present a positive outlook for Bay health. They show that it may be possible to accommodate an additional 4.3 million persons in the watershed over the next few decades while continuing to reduce nitrogen pollution. A future scenario where forests are protected and both people and agriculture are more concentrated, could be a future where pollution is more effectively managed and controlled through technology and regulation, and a future that preserves natural assets for future generations.

Warrenton, Virginia. Courtesy of the Chesapeake Bay Program, used with permission.

Changes in Agricultural Water-Quality Management

Chapter 7

By Ana María García[1], Jennifer L. Keisman[1], Andrew J. Sekellick[1], John W. Clune[1], James S. Webber[1], and Alex M. Soroka[1]

Agriculture in Talbot County, Maryland. Courtesy of the Chesapeake Bay Program, used with permission.

Agriculture has been and continues to be an important part of the economy and heritage of the Chesapeake Bay region, but legacy and current nutrient issues are a major challenge for resource managers (fig. 7.1). Agriculture is the leading contributor of excess nitrogen to the Chesapeake Bay and to many other coastal areas across the Nation whose watersheds include substantial crop and animal production (see Excess Nitrogen Impacts on Coastal Areas across the Nation and the World). As agricultural production has intensified, excess nitrogen exported to the downstream waterbodies and the Bay has increased.

Nitrogen is a critical element in crop nutrition that is most often supplied by animal manure and (or) commercial fertilizer. In areas of confined animal production, such as the Delmarva Peninsula and southeastern Pennsylvania, nitrogen is imported in the form of animal feed and applied as animal manure to local fields for crop growth. Across the watershed, increasing and widespread use of commercial fertilizer throughout crop production has been important in supporting the rising demand for food both locally and nationwide. Unfortunately, excess nitrogen from manure and fertilizer, beyond plant needs, has led to an increased export of nitrogen to groundwater and streams. Consequently, there has been a rise in conservation measures to mitigate water-quality impacts (see chap. 1).

[1]U.S. Geological Survey.

Figure 7.1. Agriculture has been and continues to be an important part of the economy and heritage of the Chesapeake Bay region, but legacy and current nutrient issues are a major challenge for resource managers. Photograph of north central Pennsylvania farm. Courtesy of Wikimedia, used with permission.

The purpose of this chapter is to describe changes in agricultural production and management activities over time as they pertain to nitrogen and water quality. The trends in agricultural land use and in the implementation and effectiveness of conservation practices are highlighted.

The current farming industry in the Chesapeake Bay watershed includes 83,000 farms and an annual agricultural production of $10 billion (see Environmental Setting chapter; Natural Resources Conservation Service, 2018) despite a 45.7 percent decrease in farmland from 1950 to 2012 (fig. 7.2*A*; LaMotte, 2015; Keisman and others, 2018). Half of current farms are solely for crop production, whereas 42 percent are primarily livestock and poultry operations (U.S. Department of Agriculture, 2014). Agriculture in the Chesapeake Bay watershed produces a variety of important commodities such as corn, hay, soybeans, oats, wheat, and barley (fig. 7.2*B*; U.S. Department of Agriculture, 2014). From dairy farming to the poultry industry, the diversity in livestock production is also a distinguishing feature of the region (fig. 7.2*C*).

Agriculture tends to be concentrated on the Coastal Plain and areas with fertile soils, such as the Delmarva Peninsula and southeastern Pennsylvania (fig. ES.5). Unfortunately, many of these regions also tend to have geologic settings (sand/gravel, carbonate aquifers) that are more vulnerable to nutrient contamination (fig. ES.4*A*). Nitrogen and phosphorus inputs from agriculture are the largest source of nutrient inputs to the landscape in the Chesapeake Bay watershed (Boesch and others, 2001; U.S. Environmental Protection Agency, 2010a) and eventually delivered to the Chesapeake Bay (fig. NS.6; Ator and others, 2011; Sekellick and others, 2021).

Current crop agriculture often requires added nitrogen from manure or chemical fertilizer to achieve high crop yields to meet societal demands for food. Plants take up these mineral forms of nitrogen through their root systems to create proteins and other essential biomolecules. Animal production can also contribute significantly to the amount of excess nitrogen. Livestock take up nitrogen in the form of feed and return ammonia and organic nitrogen in the form of manure. In 2010, manure accounted for 39 percent of the nitrogen inputs to the Bay watershed (fig. NS.3). Another environmental impact from animal agriculture is ammonia volatilization, which occurs when manure is applied to the surface of the soil. As a result, atmospheric deposition of nitrogen to the Bay watershed can originate from livestock and soil emissions from agriculture.

The Past: 1950–2017

During the first half of the 20th century, more than half of the land in the Chesapeake Bay watershed was managed as small, highly diversified local farms that produced livestock, dairy products, crops, and vegetables (fig. OV.1; Keisman and others, 2018). For example, according to the 1920 Census of Agriculture, Maryland was the sixth largest vegetable producing state in the U.S. supplying nearby markets (U.S. Census Bureau, 1922). Through the second half of the 20th century, significant changes in agriculture were developed through mechanization, electrification, chemistry, and genetics (Conklin, 2008; Capel and others, 2018a) that resulted in new supply chains and food marketing. Horses were replaced over time by self-propelled, air conditioned, satellite-positioned tractors that significantly increased crop yields. The availability of electric power in rural areas allowed for the mechanization of many old and new farm operations like irrigation. Herbicides, hybrids, and genetically modified crops allowed for the intensification of crop agriculture (Borlaug, 1972; National Research Council, 2000). Advances and efficiencies in animal breeding during this time period brought a four-fold increase in milk production and enlarged the body size of chicken broilers by 400 percent (see sidebar–The Rise of Poultry on the Delmarva Peninsula; Capper and others, 2009; Zuidhof and others, 2014). Antimicrobials allowed for the expansion of large concentrated animal operations and less dispersal of livestock across the landscape (Khachatourians, 1998; U.S. Environmental Protection Agency, 2012).

The most prodigious expansion of agriculture during this time period came from the widespread use of synthetic nitrogen fertilizer, as previous nutrient requirements for crops were limited by crop rotation (for example, fallow years to replenish soils) and local availability of manure (see chap. 1). Nitrogen inputs from fertilizer grew substantially from 1950 to 1971, and then plateaued and slightly declined through 2017 with interannual fluctuations of as much as 117 million pounds (53 million kilograms) (Keisman and others, 2018). Although the plentiful supply of nitrogen dramatically improved crop yields, over time declines in water quality made it evident that almost one-quarter of nitrogen from fertilizer applications and fixation by crops did not contribute to crop growth, but rather was lost to the Chesapeake Bay tributaries (figs. NS.3 and NS.4*B*; Ator and others, 2011).

Moreover, during this time period, agriculture became less diverse as farmland area continued to decline (fig. 7.2*A*). A new model for supplying food emerged with market food chains dominating the

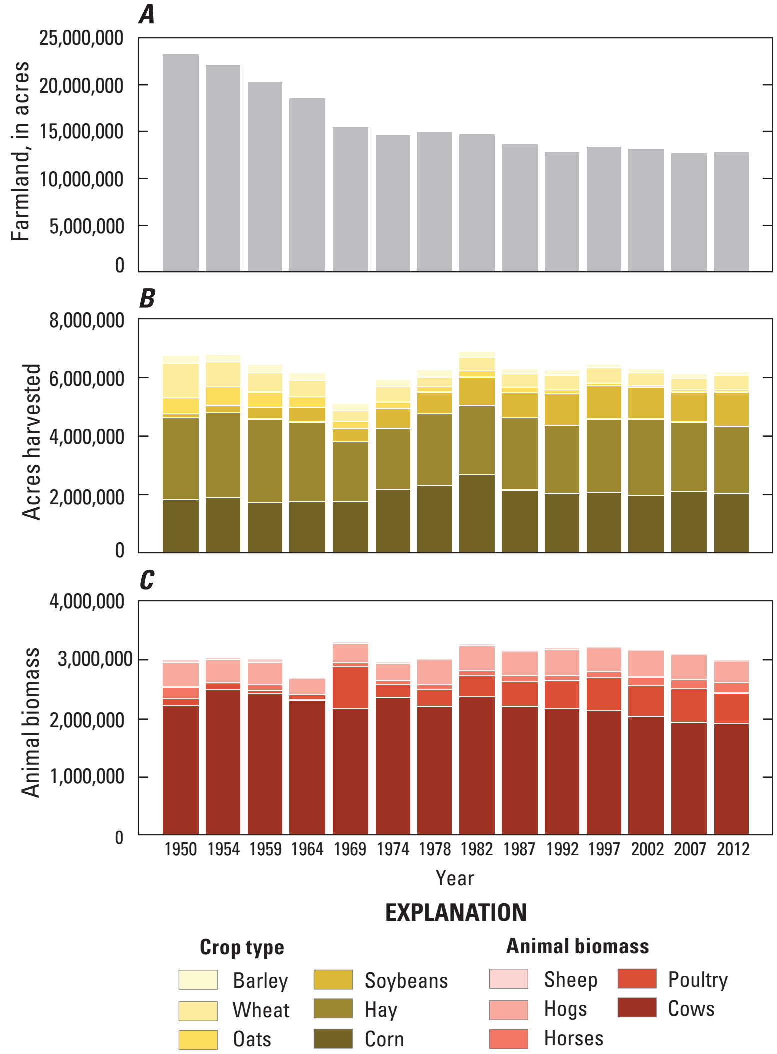


Figure 7.2. Changes in ***A***, farmland acreage, ***B***, acres harvested for six dominant crop types, and ***C***, animal biomass for five major animal types in the Chesapeake Bay watershed, 1950–2012 (LaMotte, 2015; Keisman and others, 2018).

The Rise of Poultry on the Delmarva Peninsula

The Eastern Shore of Maryland that drains to the Chesapeake Bay is located on the Delmarva Peninsula (fig. ES.2) and is the birthplace of commercial chicken production for meat. The raising of backyard flocks of chickens was commonplace before the early 1900s, but the focus had mostly been on egg production. A change in regional agriculture was triggered in 1923, when small farmer Cecile Steele's order of 50 chickens turned out to be 500 (Williams, 1998). She decided to raise the chickens for meat and after a noticeable profit, she built a broiler house for 10,000 birds. This was unknowingly the start of the modern poultry industry in the region that currently produces around 600 million chickens annually (Williams, 1998). The rise in poultry production continues on Maryland's Eastern Shore on the Delmarva Peninsula, but also in other states of the Chesapeake Bay watershed (fig. 7.S1*A*).

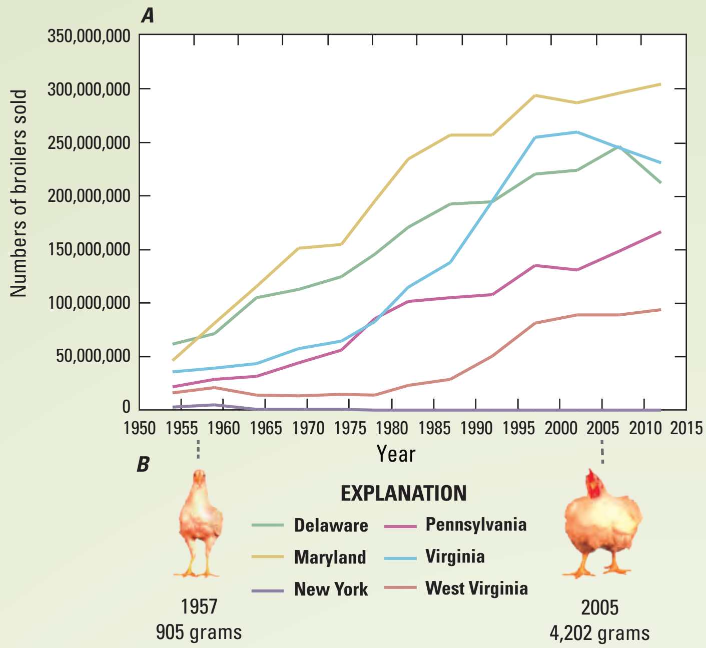


Figure 7.S1. Modern poultry production has ***A***, substantially increased the number of broilers (young chicken raised for meat) sold in the Chesapeake Bay watershed from 1954 to 2012 (LaMotte, 2015), and ***B***, incorporated selective breeding and nutrition improvements that have more than doubled the weight of broilers (Zuidhof and others, 2014).

The U.S. population preference for chicken has surpassed other meats like beef and pork (U.S. Department of Agriculture, 2020). Since the early 20th century, selective breeding and nutrition improvements have more than doubled the weight of broilers (fig. 7.S1*B*), while significantly reducing the days to market, feed consumption, and mortality rate (Zuidhof and others, 2014; National Chicken Council, 2019). Previous small-scale poultry farming was transformed into a spatially expansive (fig. 7.S2) vertical integrated system that has consolidated control over the poultry production chain (for example, breeding, egg hatcheries, broiler farms, feed mills, manure composting, and so forth) and resulted in improved efficiencies and reduction in consumer costs for an industry fueled by the nearby market demands of Philadelphia, Pennsylvania; New York, New York; and Washington, D.C.

Figure 7.S2. Previous small-scale poultry farming on the Delmarva Peninsula in 1992 (***A***) has been transformed into a spatially expansive vertical integrated system by 2018 (***B***). Base map data from Google, 2015.

The transformation and expansion of the poultry industry on the Delmarva Peninsula has led to challenges for the handling of manure and use of commercial fertilizer applied to co-located crops. The Eastern Shore of Maryland (7 percent of the Bay watershed) receives almost twice as much nitrogen and more than twice as many phosphorus applications (per area) as the remainder of the watershed (Ator and Denver, 2015). These larger than average nutrient yields significantly contribute to the exceedance of health standards for drinking water quality for nitrate and concentrations that adversely affect aquatic ecosystems (fig. 7.S3; see chap. 2; Ator and Denver, 2015). The disproportionate delivery of inputs of nitrogen from the Eastern Shore to the Bay is further facilitated by the oxic groundwater environment of permeable sandy soils and surficial aquifers that more easily allow nitrate to remain stable during transport to streams. Reducing fertilizer and manure application rates and (or) further implementation of conservation practices could decrease nitrate concentrations in shallow groundwater in the short term, although with the slow lag time of groundwater traveling through deeper aquifers, may require decades to see similar declines in nitrogen loads in receiving streams and the Bay (see chap. 2 sidebar–Legacy of Nitrogen in Groundwater and chap. 4 sidebar–Groundwater Residence Times; Ator and Denver, 2015).

Figure 7.S3. Nitrogen concentrations in streams of the Eastern Shore of the Chesapeake Bay commonly exceed levels that may adversely affect aquatic ecosystems (Ator and Denver, 2015). Surface-water chemistry estimated from Ator and others (2011). The aggregate quantile reference (AQR) is the water-quality criterion recommended to protect aquatic organisms (U.S. Environmental Protection Agency, 2000).

consumer market and delivering commodities at wholesale discounts; as a result, farms in the Chesapeake Bay became more specialized, vegetable production moved to California, and the number of commodities produced per farm decreased (Bowen and others, 2016). These changes in agriculture, increasing development (see chap. 8), and land retirement in the region led to an overall decline in farmland. Between 1982 and 2012, the overall loss of agricultural lands across the entire Chesapeake Bay watershed was less than 2 percent (fig. 7.2*A*), but local loss of agricultural land was quite variable, with some counties experiencing losses greater than 30 percent (Jantz and others, 2005; Keisman and others, 2018). One consequence of increased specialization was the intensification of animal farming operations and excess manure. When more manure nutrients are produced than can be assimilated on the farm or within the region, overapplication to the land may result, leading to increased nutrient export. Ribaudo and others (2014) found livestock agriculture has become more geographically concentrated among several county clusters within the Chesapeake Bay watershed creating so-called manure hot spots in places like the Delmarva Peninsula and southeastern Pennsylvania (fig. NS.4*C*; Kellogg and others, 2000).

The agricultural industry has been an important partner with the numerous regulatory, policy, management, and voluntary efforts to reduce the effects of excess nitrogen to the Bay (fig. 1.4). Since the first nutrient pollution reduction goals in the 1987 Chesapeake Bay Agreement, significant investments have been made to both restore and preserve water quality in the Bay, making the region a national leader in the adoption of conservation practices to reduce nutrient pollution. Soon after the Chesapeake Bay Agreement, implementation of conservation practices was expanded and by 2014, there was a diverse collection of more than 150 types of practices including nutrient management, cover crops, conservation tillage, and stream buffers (fig. 7.3; Sekellick and others, 2019). Recorded management actions within the Chesapeake Bay watershed vary among the six states and Washington, D.C., depending on regional land use, priorities, programs, and reporting practices.

Understanding the effectiveness of conservation practices is vital towards meeting water-quality goals such as the Bay total maximum daily load (TMDL). Expected eventual reductions in nitrogen from 1985 to 2014 as a result of conservation practices vary spatially across the watershed and are estimated to be as high as 42 percent in areas of the Eastern Shore of the Chesapeake Bay (Sekellick and others, 2019). Across the entire Chesapeake Bay watershed, nitrogen loads were estimated to be reduced by 11 percent or 49 million pounds (22 million kilograms) during the same time period owing to a combination of land retirement and the implementation of best management practices (BMPs) such as animal waste management systems, conservation tillage, and bioretention by ponds and wetlands (fig. 7.4; Sekellick and others, 2019). Natural and restored wetlands, in particular present an opportunity for denitrification and dilution in receiving waters (Denver and others, 2014). Despite these expected eventual reductions in nitrogen, recent research using empirical models suggests that nitrogen loads and yields from most agricultural areas at the large scale did not change substantially in the Bay watershed between the early 1990s and the early 2010s (Chanat and Yang, 2018; Ator and others, 2019) and that management practices may have counteracting effects on nitrogen losses to streams (Ator, 2019). The lag time of water moving through streams and groundwater under past conservation practices means that it may take decades to see the full effects of nutrient reduction efforts (see chap. 2 sidebar–Legacy of Nitrogen in Groundwater and chap. 4 sidebar– Groundwater Residence Times). Monitoring at smaller scales has been used to better measure the water quality response of BMP implementation and help inform conservation planning (see sidebar - Applying Water-Quality Monitoring and Analysis to Help Conservation Efforts).

The Future: 2017–2050

According to estimates, agriculture will need to feed more than 22.5 million people in the Chesapeake Bay watershed by 2050 (fig. 6.1) and the accompanying increases in nitrogen inputs from fertilizer and manure could further degrade downstream waterbodies like the Bay. The future impact of nitrogen from agriculture will depend on management practices, technological advances, and changes in land use and climate.

Conservation practices are expected to continue to increase (fig. 7.3) as states within the Chesapeake Bay watershed prioritize resources to meet the Bay TMDL reduction in nitrogen loads (see chap. 1 sidebar–Largest Total Maximum Daily Load in the Nation). Although conservation practices are largely voluntary across the Bay watershed, jurisdictions like Pennsylvania have begun to ensure compliance with nutrient management and planning through agricultural inspection programs (Pennsylvania

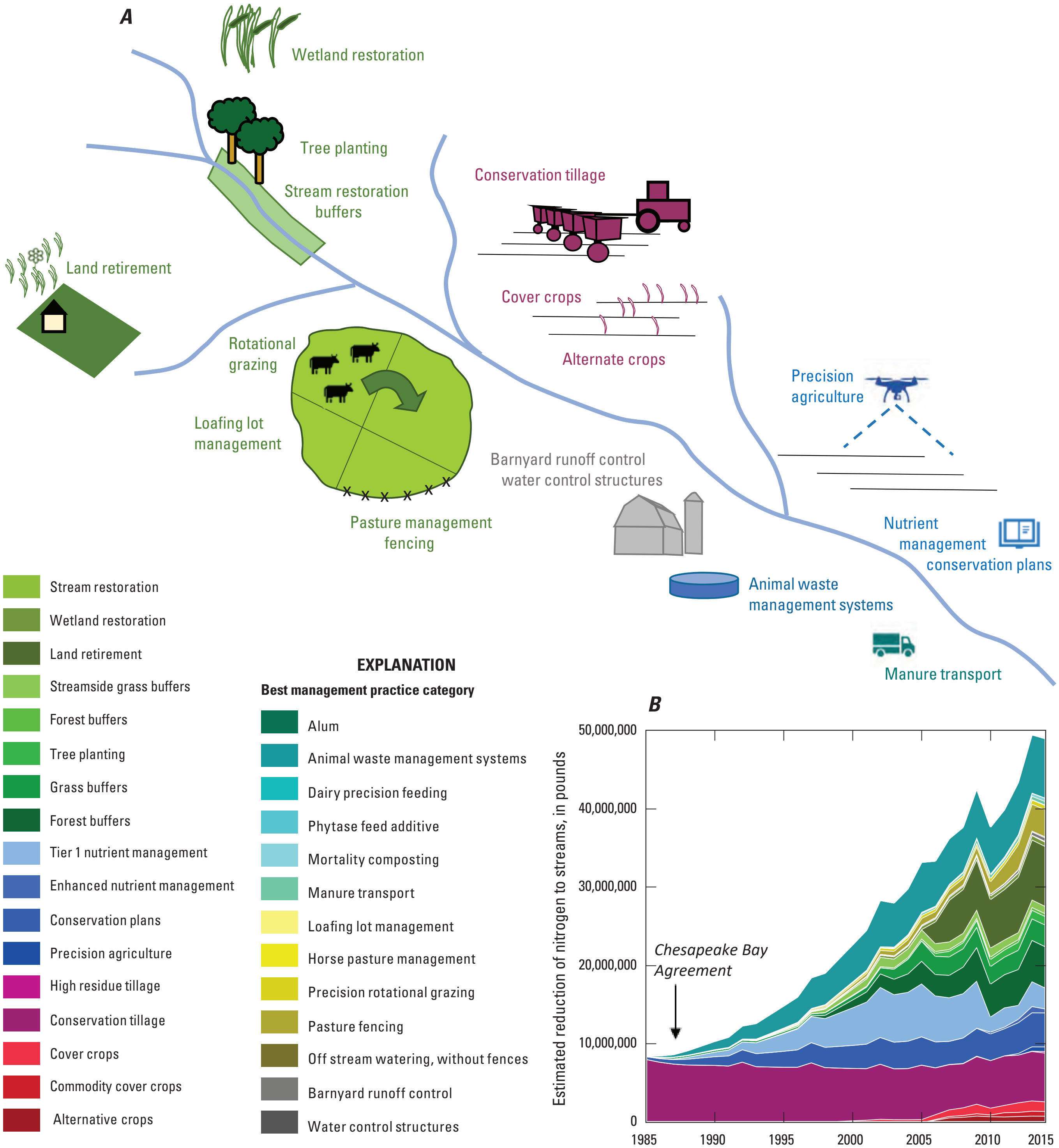


Figure 7.3. The expansion in variety and quantity of conservation practices (***A***) has been estimated to contribute to increased reductions of total nitrogen from 1985 through 2014 (***B***) in the Chesapeake Bay watershed (Sekellick and others, 2019).

Applying Water-Quality Monitoring and Analysis to Help Conservation Efforts

Conservation practices have been implemented throughout the Chesapeake Bay watershed to achieve reductions of nitrogen, phosphorus, and sediment required by the Bay TMDL (Tango and Batiuk, 2016). The suitability and timing of conservation practices to meet nutrient water-quality goals require resource managers to know when nitrogen loads are highest in streams, where the nitrogen is coming from on the landscape, the sources (for example, manure, fertilizer, and

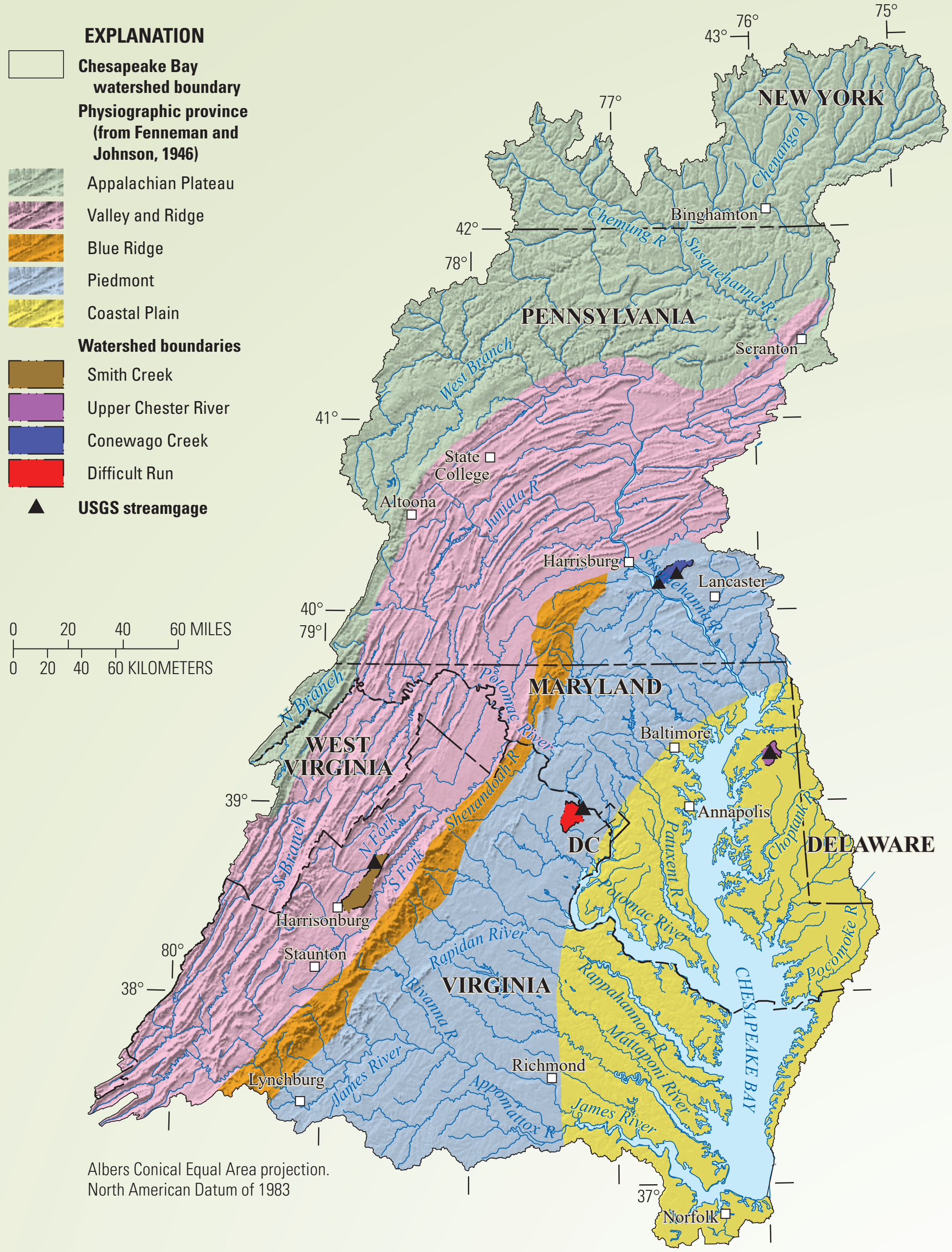


Figure 7.S4. Physiographic provinces and the location of three agricultural study watersheds (Smith Creek, Upper Chester, and Conewago Creek) and one urban watershed (Difficult Run) within the Chesapeake Bay watershed (Hyer and others, 2016).

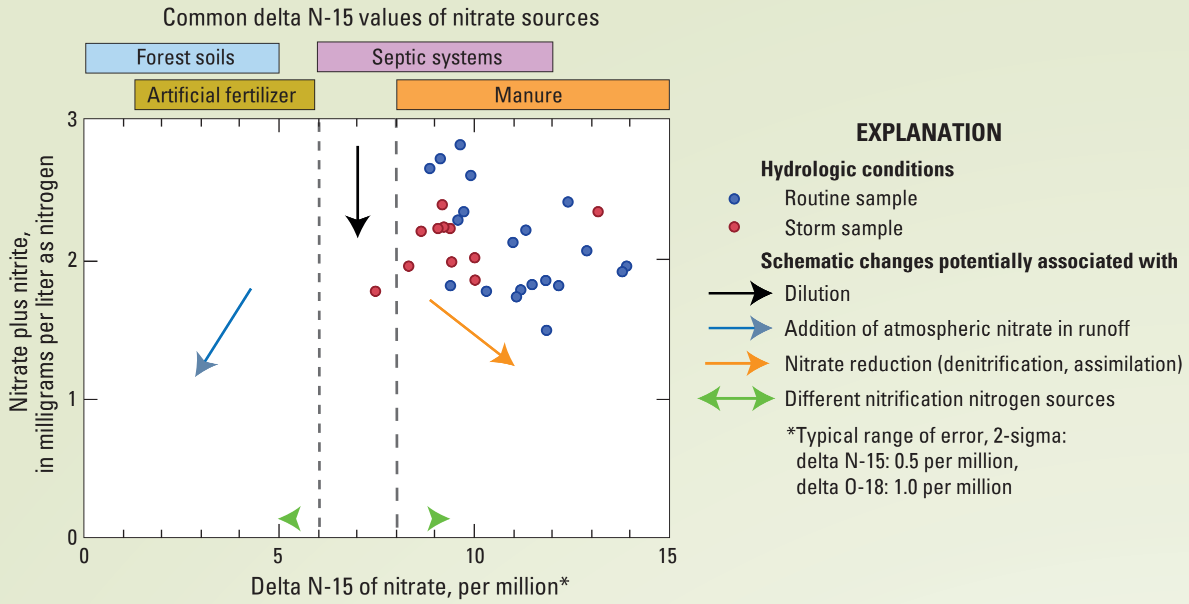


Figure 7.S5. Variability in nitrate isotopes in discrete samples from Smith Creek, Virginia (U.S. Geological Survey site 01632900), and potential sources and causes (shown by arrows) for variability, 2011–2013 (Hyer and others, 2016). N, nitrogen, O, oxygen.

so forth), and the effectiveness of the conservation practices. To help answer these questions, in 2010 the U.S. Geological Survey partnered with the U.S. Department of Agriculture and the U.S. Environmental Protection Agency to increase the implementation of conservation practices in the three small agricultural watersheds of Smith Creek, Va.; Upper Chester, Md.; and Conewago Creek, Pa. (fig. 7.S4). Many of the same conservation practices were commonly used among the watersheds, including nutrient management plans, cover crops, and conservation crop rotation, but implementation of specific conservation practices was more dependent on the local setting. For example, conservation practices in the Upper Chester watershed focused on irrigation management, whereas conservation practices in Smith and Conewago Creek watersheds included stream fencing for animal exclusion and bank restoration.

The Smith Creek watershed is dominated by cattle and poultry production in areas underlain by carbonate and siliciclastic rocks in the Valley and Ridge physiographic province in the Shenandoah Valley of Virginia (fig. 7.S4). Seventy-two percent of the streamflow originates from groundwater discharge (base flow) rather than stormwater runoff and much of the discharge contributes to a single dominant spring. Nitrate isotope data from the limestone springs were generally consistent with manure-derived nitrogen sources (fig. 7.S5).

The Upper Chester watershed contains predominantly row crop agriculture in a sand and gravel aquifer setting in the Coastal Plain physiographic province on the Eastern Shore of Maryland (fig. 7.S4). Nitrate is dominant in groundwater and diluted during storm events (fig. 7.S6). Nitrate isotope data indicate inorganic fertilizer is a dominant source of nitrogen to waterways in the Upper Chester watershed.

The Conewago Creek watershed is characterized by mixed agricultural activities in areas underlain by carbonate and siliciclastic rocks in the Piedmont physiographic province in southeastern Pennsylvania (fig. 7.S4). Modeling data indicate that agricultural sources of manure and fertilizer dominate the input of nitrogen to the watershed (Hyer and other, 2016).

Documenting changes in water-quality over time provides a critical evaluation of conservation practice responses. Although implementation of these conservation practices is encouraging, results indicate only small decreases in nitrogen loads. It will likely be years before the cumulative effects of these practices are detected and future water-quality in these small, agricultural watersheds will depend largely on whether nutrient applications are reduced over time.

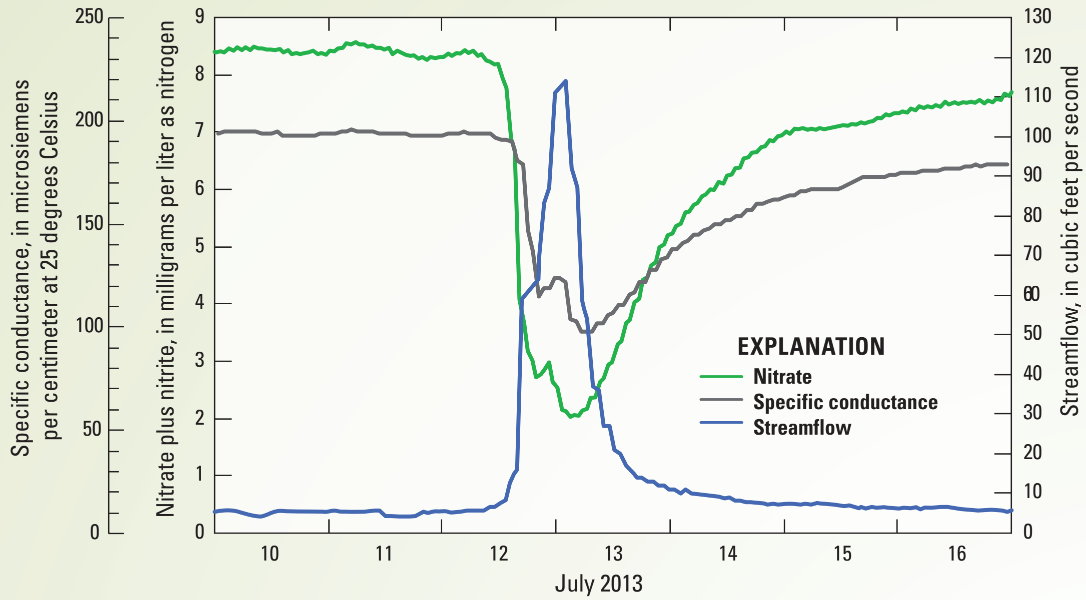


Figure 7.S6. Streamflow hydrograph, continuous nitrate concentrations, and continuous specific conductance data for Chesterville Branch River, Maryland (U.S. Geological Survey site 01493112), July 10-16, 2013.

Department of Environmental Protection, 2018). As the legacy nitrate contamination in the groundwater system continues to be transported to streams and the Bay, the expectation from resource managers is that aquifers will be replenished with water of improved quality from more recent management actions (see chap. 2 sidebar–Legacy of Nitrogen in Groundwater). Observed declines in average nitrogen load from cropland to local streams in carbonate settings between 1992 and 2012 suggest cropland conservation practices may be effective and similar reductions may be achieved for other geologic settings with similar management and land use in the future (Ator and others, 2019).

Autonomous farming has the potential to revolutionize not only productivity for crop yields, but also the efficient use of nitrogen. Through individual agricultural operations or data-sharing collectives, farmers will be able to more easily access data to better manage their farms through drones, satellite imagery, and monitoring sensors deployed on soil, crops, and animals. This type of precision agriculture has the potential to better match nitrogen inputs with crop requirements and to reduce excess nitrogen to waterways (Rütting and others, 2018). Additionally, genetically altered crop varieties have the potential to significantly improve nitrogen use efficiency (Hirel and others, 2011).

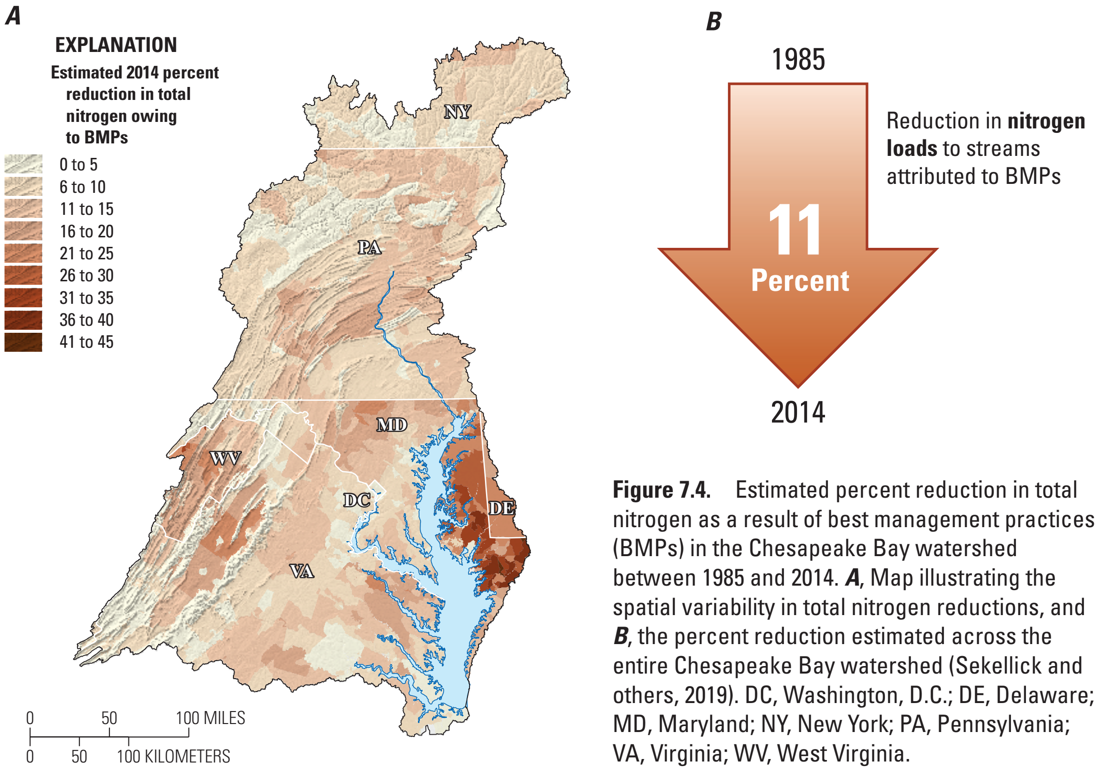


Figure 7.4. Estimated percent reduction in total nitrogen as a result of best management practices (BMPs) in the Chesapeake Bay watershed between 1985 and 2014. ***A***, Map illustrating the spatial variability in total nitrogen reductions, and ***B***, the percent reduction estimated across the entire Chesapeake Bay watershed (Sekellick and others, 2019). DC, Washington, D.C.; DE, Delaware; MD, Maryland; NY, New York; PA, Pennsylvania; VA, Virginia; WV, West Virginia.

Land use changes are expected to alter agricultural activities in the future. Farmland is expected to continue to be converted to developed land in the Chesapeake Bay watershed through 2050. This projection leads to various possible scenarios with respect to water-quality concerns (see chap. 10). Primarily, given the larger amount of nitrogen exported from agricultural lands in comparison to developed lands, it is possible that a reduction in nitrogen will be the result of urbanization. However, simultaneous future environmental trends make it challenging to forecast the magnitude of such a reduction.

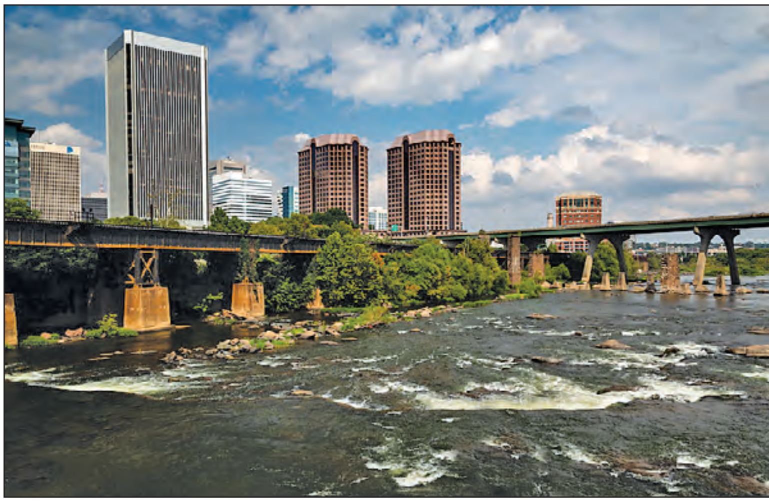
Given the certainty of a warmer future for the Chesapeake Bay watershed, forecasts of agricultural activities embed the anticipated impacts of increased heat stress and ecosystem change. In general, the mid-Atlantic climate is anticipated to become more tropical (see chap. 3). In recent decades, the growing season in agricultural areas in both the eastern and western United States has become longer (Kunkel and others, 2004; U.S. Environmental Protection Agency, 2016). In the mid-Atlantic, the number of days below 32 °F have decreased and days averaging above 75 °F have increased. As many as 20 additional growing days are anticipated by 2040 (Delaware Department of Natural Resources and Environmental Control, 2014; Environment and Natural Resources Institute, Pennsylvania State University, 2015). This could mean greater yields from row crops and increased suitability of double cropping systems, which in turn could lead to greater nutrient and thus fertilizer inputs (Borchers and others, 2014). On the other hand, heat stress is expected to negatively impact livestock operations, increasing costs for producers and reducing overall production, especially for dairy operations, because of unfavorable warming conditions (Wolfe and others, 2008). Additionally, potential increases in spring precipitation may delay crop planting or severe summer drought may reduce growing seasons (Wolfe and others, 2008). Overall, it is uncertain what impact these changes will have on the quality of water resources, but some inferences can be made. Current conservation action plans implemented through the Bay TMDL are expected to have greater reductions in nitrogen, but new nutrient management strategies likely will need to adapt to possibly warmer and wetter conditions in the future.

Crop irrigation on a farm in Queen Anne's County, Maryland.
Courtesy of the Chesapeake Bay Program, used with permission.

Changes in Water-Quality Management in Developed Areas

Chapter 8

By Rosemary M. Fanelli[1], Kristina G. Hopkins[1], Qian Zhang[2], and Paul D. Capel[1]



View of the James River in Richmond, Virginia. Courtesy of the Chesapeake Bay Program, used with permission.

Population growth and expansion of developed land has been, and is expected to be, a major driver of changing water quality in the Chesapeake Bay and its tributaries. Urbanized or highly developed watersheds have a greater nitrogen yield than forested or suburban watersheds (Groffman and others, 2004), making the downstream ecosystems especially vulnerable (Ator and others, 2011). Nitrogen management in developed areas has historically focused on reducing point sources through improved technologies used in wastewater treatment plants (WWTPs; Testa and others, 2008). More recently, urban management practices have been specifically designed and implemented to reduce nitrogen nonpoint (diffuse) loads (for example, bioretention, street sweeping, or stream restoration), as the importance of these nonpoint sources of nitrogen was recognized. As the expansion of developed areas continues, municipalities will increasingly rely on these nonpoint management practices, as well as point-source reductions, to address nitrogen loads. This chapter (1) reviews the sources of nitrogen in developed areas of the Chesapeake Bay watershed and examines how urban expansion has altered the nitrogen cycle, (2) discusses past interventions to mitigate nitrogen loads from developed areas, and (3) considers the future of nitrogen management in developed areas. The future nitrogen management discussion includes the effects of climate change and land use legacies that may hinder management of nitrogen, as well as nascent technologies and approaches that may address some of these issues.

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Figure 8.1. Photographs showing different levels of urban development. Courtesy of the Chesapeake Bay Program, used with permission.

The landscapes of developed areas are different from forested or agricultural areas in their sources, cycling, transport, and management of nitrogen. Although developed regions compose a smaller proportion of the watershed than either agriculture or forest land uses (fig. OV.1), developed landscapes drastically alter the transport, processing, and delivery of nitrogen from the landscape to both local waterways and to the Chesapeake Bay. In 2000, 4.8 percent of the watershed was categorized as developed land, with many areas covered by impervious surfaces such as sidewalks, driveways, roads, and building rooftops. Some developed landscapes are characterized by high-density development, such as downtown Washington, D.C. or Baltimore, Maryland, where most of the land cover is impervious (fig. 8.1). However, most developed land in the Chesapeake Bay watershed is considered medium- to low-density suburban, which is composed of a mix of impervious cover and pervious surfaces, such as parks, yards, and vegetated cover. Farther away from city centers are exurban or rural regions, where there is still development, but with a lower density. In these areas, either undeveloped or agricultural landscapes still dominate.

The largest sources of nitrogen associated with developed areas in the Chesapeake Bay watershed are point sources—discrete points of discharge containing nitrogen (and other constituents) released directly into streams and rivers. Point sources include effluent pipes from WWTPs, industrial facilities, and city sewer systems. Wastewater and industrial effluent usually contain dissolved forms of nitrogen, including nitrate, ammonia, and organic nitrogen. Discharges from city sewer systems (in other words, storm sewer outflows), can also contain nitrogen associated with particles. The largest WWTPs in the Chesapeake Bay watershed by flow are the (1) Blue Plains WWTP, which services the Washington, D.C. area; (2) Back River and Patapsco facilities, which service the Baltimore, Maryland area; and (3) Richmond WWTP in Virginia (Hopple and others, 2021). These plants combined currently treat approximately 610 million gallons (2.3 billion liters) per day of effluent on average (Hopple and others, 2021). As of 2015, there were about 782 WWTPs servicing cities and towns across the Chesapeake Bay watershed (Hopple and others, 2021). In 2010, WWTPs discharged about 23.5 million kilograms of nitrogen to the Chesapeake Bay watershed; about 57 percent of this total was discharged directly into tidal waters (Hopple and others, 2021). Point sources can be a major source of nitrogen loading in some local streams and rivers (fig. NS.3). For example, about 40 percent of the nitrogen load in the Patuxent River, an urban watershed with 53 percent developed land, originates from point sources (see chap. 2; Hopple and others, 2021).

There are also many nonpoint sources of nitrogen in developed areas, including atmospheric deposition, residential inputs of fertilizer, leaky wastewater and storm sewer infrastructure, septic systems, pet waste, and nitrogen derived from sediments stored in rivers, floodplains, and retention basins (Hobbie and others, 2017). These sources are all considered nonpoint sources of nitrogen because they are spread across the landscape and do not originate from a discrete point, like an effluent pipe. As with agriculture landscapes (see chap. 7), nonpoint sources of nitrogen in developed areas can collectively contribute large amounts of nitrogen to streams. Approximately 12 percent of the total nitrogen load to the Chesapeake Bay is estimated to originate from nonpoint sources from developed areas (Ator and others, 2011). Atmospheric deposition from vehicle emissions, industrial sources, and nearby agricultural areas deposit nitrogen on impervious surfaces, which can later be carried by stormwater runoff into local streams during rainfall events. Lawn fertilizer and pet waste are also a source of nitrogen from developed regions in residential areas and have accounted for as much as 59 and 28 percent of urban nitrogen inputs in some developed watersheds, respectively (Hobbie and others, 2017). Leaky sewer systems and septic systems also contribute to nonpoint sources of nitrogen (Divers and others, 2013).

Urban development also profoundly alters the ways in which water moves through the landscape (Paul and Meyer, 2001). The conversion from forest to developed land uses often compacts soil, removes woody vegetation, and covers the land surface with impervious surfaces such as roofs, roads, and sidewalks. Rainfall events generate more surface runoff on these impervious surfaces compared to soils with natural vegetation (Gregory and others, 2006). The reduction in vegetative cover also reduces evapotranspiration rates, which may result in less overall water storage in the watershed (Poff and others, 1997). Increased runoff rates and decreased water storage in developed landscapes create flood risks, so most developed areas include storm sewer networks to route surface runoff away from roads and buildings and into nearby streams (Leopold, 1968). The quick delivery of surface runoff through storm sewer networks can drastically alter streamflow patterns in urban streams (Poff and others, 2006). Changes in streamflow patterns may also cause channel incision, which hydrologically disconnects riparian zones and floodplains from urban stream channels (Hardison and others, 2009).

Hydrologic changes to the landscape alter the transport, processing, and export of nitrogen in developed areas. The storm sewer network often bypasses areas in the uplands and riparian zones where denitrification may naturally occur, thereby reducing the potential for nitrogen removal (Kaushal and Belt, 2012). Low groundwater levels in riparian zones owing to channel incision also reduce denitrification potential (Groffman and others, 2002). Moreover, large increases in streamflow during rainfall events may cause increased streambank erosion in local stream channels, transporting soil particles with their associated nitrogen and contributing to nitrogen exports (Booth and Jackson, 1997). Increased streamflow in the stream channel also scours channel streambed geomorphic features, such as debris dams, which are often hotspots for nitrogen retention (Groffman and others, 2005). Stream burial, which occurs when natural stream channels are replaced with underground piping and then covered with pavement, typically occurs in high-density developed areas, and can also significantly alter nitrogen cycling. For example, Pennino and others (2014) found stream burial to decrease in-stream nitrogen uptake by 39 percent.

Water-quality management of nitrogen in developed areas is critical, given its multiple sources and the numerous ways that it can be transported through the landscape and into streams. Point sources, because of their specific locations, are easier to manage than nonpoint sources. Most point sources, including all WWTPs, are regulated through the Clean Water Act with the National Pollutant Discharge Elimination System (NPDES; U.S. Environmental Protection Agency, 2020), which places limits on either the load or concentration of the effluent being discharged. For example, the NPDES permit for the Blue Plains WWTP limits the annual discharge of nitrogen to 2,127,000 kilograms of nitrogen (see sidebar–Blue Plains Advanced Wastewater Treatment Plant). Advanced techniques in wastewater treatment have been adopted by some WWTPs to meet the growing demand and required limits set forth by NPDES regulations, which are often tied to local and regional total maximum daily loads (TMDLs; see chap. 1 sidebar–Largest Total Maximum Daily Load in the Nation). Although considered to be point sources, combined sewer overflows are more difficult to manage than WWTPs or industrial discharges because of the episodic nature of the timing, duration, and size of their discharges influenced by the amount of stormwater runoff entering the system. Because of this, the actual discharge of water and nitrogen varies from year to year. In Baltimore, the annual nitrogen loads in urban streams are positively associated with the amount of time combined sewer overflows discharge into streams (Reisinger and others, 2019).

Nonpoint sources of nitrogen in developed landscapes are often more difficult to manage than point sources because they are spread across the landscape. Some nitrogen source reduction programs, which are designed to change individual human behavior for managing pet waste and applying lawn and garden fertilizers, have been effective at reducing nitrogen delivered to waterways. For example, educating homeowners on the proper timing and magnitude of lawn fertilizer application rates and watering patterns reduced nitrogen runoff from residential lots (Bachman and others, 2016; Toor and others, 2017). Nonpoint sources of nitrogen can also be managed by controlling stormwater runoff with stormwater management practices such as detention ponds, infiltration basins, and stormwater wetlands (fig. 8.2). Stormwater management practices can store particulate nitrogen or remove nitrogen through denitrification. The implementation of stormwater management practices can also reduce the occurrence of discharge from combined sewer overflows for those municipalities that still have combined sewer systems. Finally, stream restoration practices may be able to increase denitrification in streams by increasing groundwater-surface water interactions or by decreasing particulate nitrogen loads through streambank stabilization and floodplain sediment trapping. Models are used to evaluate the efficiency of stormwater management practices and forest conservation activities for removing nitrogen (Sekellick and others, 2019).

The Past: 1950–2015

Throughout the last century, the population in the Chesapeake Bay watershed has risen steadily, from 8.4 million people in 1950 to 17.3 million people in 2010 (fig. OV.1). Urban centers expanded to accommodate this growing population, with developed land use increasing from 2.3 percent in 1950 to 4.8 percent in 2000 across the watershed (fig. OV.1). Much of the land converted to developed areas in the Chesapeake Bay watershed was originally forest and wetlands. Stormwater management in urban developments in the 1950s and 1960s often

Blue Plains Advanced Wastewater Treatment Plant

The Blue Plains Advanced Wastewater Treatment Plant, located in Washington, D.C., and operated by DC Water, is the largest of its kind in the world (fig. 8.S1*A* and *B*). With a current operating capacity of 1,454 million liters per day (MLD; 384 million gallons per day [MGD]), which represents the 53 MLD (14 MGD) capacity upgrade from 1,400 MLD (370 MGD) in 2010, the facility treats wastewater inflows from the District of Columbia and surrounding counties in Maryland and Virginia. Since its opening in 1937 (see Historical Setting of the Chesapeake Bay Watershed chapter sidebar–Changes in the Potomac River in the 20th Century), the facility has experienced several major upgrades to incorporate new treatment technologies. Consequently, the facility now has primary, secondary, and advanced (tertiary) treatment processes. Beginning in 2000, the facility installed a full-scale nitrogen removal process system, which has resulted in a rapid and substantial decline in nitrogen concentrations in the effluent (Pennino and others, 2016). For example, the annual nitrogen discharge by the facility was reduced by more than 70 percent from 14.1 million pounds per year in 1985. Some of the reasons for this declining trend include but are not limited to implementation of DC Water's Nitrogen Removal Program and the Long-Term Control Plan. The treatment technology enhancements at the Blue Plains WWTP have resulted in appreciable improvements in water-quality and habitat conditions in the downstream Chesapeake Bay estuary (including increased diversity of species and native species), presenting an example of how environmental policies that reduce nutrient inputs can result in improved habitat (Ruhl and Rybicki, 2010). For example, nitrogen concentrations near the surface showed consistent declines over the past 30 years at several tidal monitoring stations that are downstream of the Blue Plains WWTP and located in the tidal fresh region of the Potomac River (fig. 8.S1*C*). In contrast, the nitrogen trend in the Potomac River water-quality station upriver of Blue Plains WWTP has only decreased modestly over that same period.

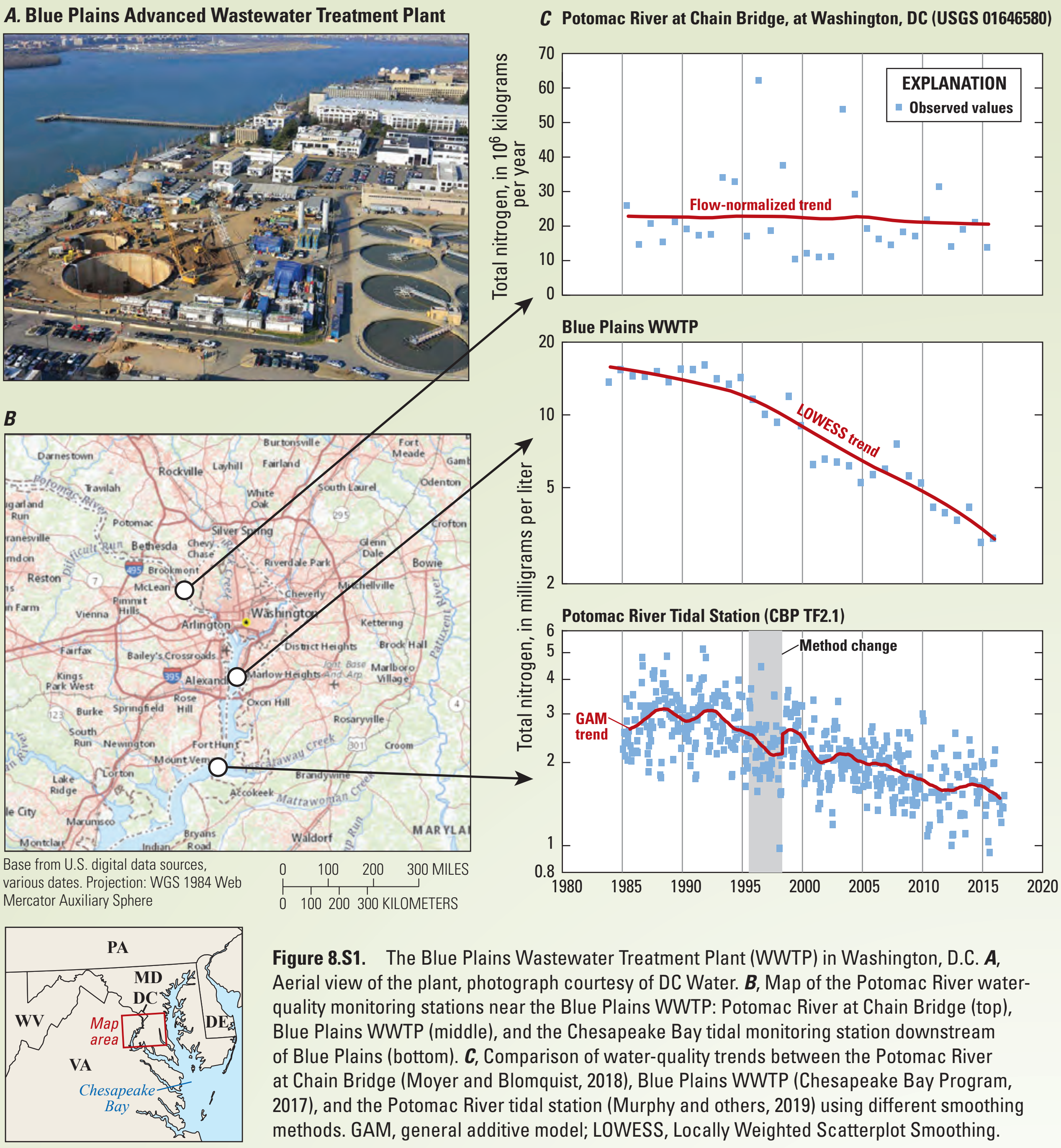

Figure 8.S1. The Blue Plains Wastewater Treatment Plant (WWTP) in Washington, D.C. ***A***, Aerial view of the plant, photograph courtesy of DC Water. ***B***, Map of the Potomac River water-quality monitoring stations near the Blue Plains WWTP: Potomac River at Chain Bridge (top), Blue Plains WWTP (middle), and the Chesapeake Bay tidal monitoring station downstream of Blue Plains (bottom). ***C***, Comparison of water-quality trends between the Potomac River at Chain Bridge (Moyer and Blomquist, 2018), Blue Plains WWTP (Chesapeake Bay Program, 2017), and the Potomac River tidal station (Murphy and others, 2019) using different smoothing methods. GAM, general additive model; LOWESS, Locally Weighted Scatterplot Smoothing.

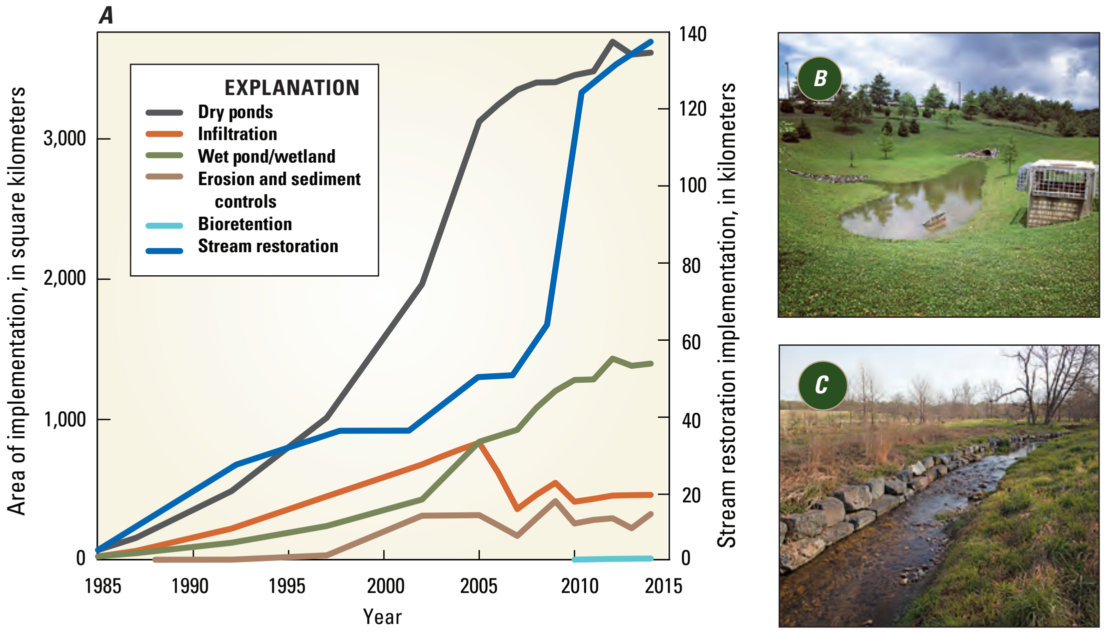

Figure 8.2. A variety of management controls have been implemented to control nitrogen in developed areas and the ***A***, area of implementation of select urban best management practices has increased from 1985 through 2014 in the Chesapeake Bay watershed. Photographs of three management practices used to manage stormwater and associated nonpoint urban sources of nitrogen (Sekellick and others, 2019), including ***B***, a retention pond (Krissy Hopkins, U.S. Geological Survey), ***C***, stream restoration (Will Parson, Chesapeake Bay Program), and *D*, bioretention (Alicia Pimental, Chesapeake Bay Program). Images used with permission.

employed curbs and gutters to convey runoff into streams to avoid local flooding. Urban sources of nitrogen, as well as stormwater, were not regulated or managed much at this time. Municipal and industrial effluents and combined sewer overflows were often discharged directly into local waterways and largely unregulated with respect to nitrogen (National Research Council, 2009); the water quality near cities was quite poor as a result (see Historical Setting of the Chesapeake Bay Watershed chapter sidebar–Changes in the Potomac River in the 20th Century). Poor water quality, a ubiquitous issue across the Nation, prompted the passage of the Clean Water Act in 1972, which established the NPDES program to regulate point sources (fig. 1.4). As the effects of stormwater runoff on streams and rivers were just beginning to be acknowledged, the U.S. Department of Agriculture Soil Conservation Service (now the Natural Resources Conservation Service) developed methods to estimate stormwater runoff volumes and peak discharges from developed areas to aid in the design of stormwater management practices in 1975 (U.S. Department of Agriculture, 2009). In 1987, the Clean Water Act was amended to begin regulating stormwater in two phases: Phase 1 (initiated in 1990) included only larger municipalities, and Phase 2 (initiated in 1999) included smaller municipalities and stormwater quality (National Research Council, 2009). For example, shifts in stormwater management strategies were observed in Baltimore, Maryland, whereby implementation in the 1980s and 1990s was predominantly retention and detention basins, compared to implementation in the late 2000s to 2010s that shifted toward decentralized practices (McPhillips and Matsler, 2018).

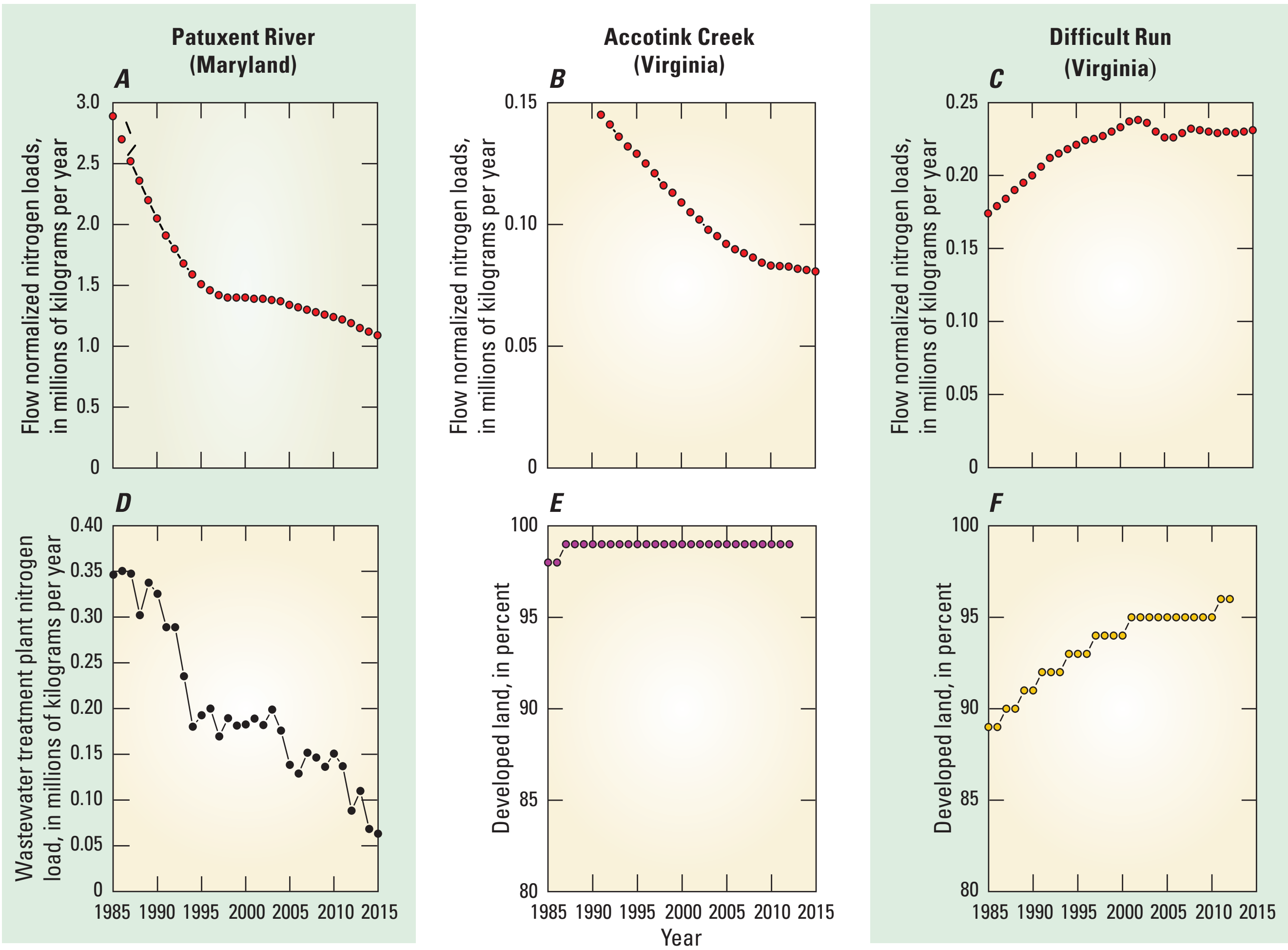

Figure 8.3. Water-quality monitoring in urban watersheds illustrates the effects of urbanization and changes in nitrogen management as shown in ***A*–*C*,** flow-normalized nitrogen loads versus time graphs derived from monitoring data for the Patuxent River (Maryland), Accotink Creek (Virginia), and Difficult Run (Virginia), respectively (Moyer and others, 2017); ***D*,** wastewater loads in the Patuxent River over time (Hopple and others, 2021); ***E*** and ***F*,** percent developed land use over time for the Accotink Creek and Difficult Run watersheds, respectively (Falcone, 2015).

In 1983, the first Chesapeake Bay Agreement was signed, and in 1987, new quantitative goals were established to reduce nitrogen loads entering the Bay (figs. 1.4 and 8.3). The Chesapeake Bay restoration effort prompted many changes in nitrogen management of point and nonpoint sources in the watershed. Upgraded technology in WWTPs (for example, enhanced biological removal) decreased the load of nitrogen discharged from many facilities. In Maryland, 26 WWTPs implemented enhanced nitrogen removal technologies between 2006 and 2012, which reduced nitrogen concentrations in effluent below 3 milligrams per liter (mg/L; Maryland Department of the Environment, 2020). Water-quality improvements in the estuary have been observed downstream of the Blue Plains WWTP as a result of these upgrades (see sidebar–Blue Plains Advanced Wastewater Treatment Plant). The Chesapeake Bay Agreement also encouraged the implementation of nonpoint urban nitrogen management practices. The first urban practices implemented were largely infiltration and detention practices that reduce stormwater runoff volumes (fig. 8.2). Over time, other

What Happens to Stream Water Quality When Agricultural Land is Developed?

Urbanization between 1950 and 1985 accounted for 20 percent of the decline in agricultural land within the Chesapeake Bay watershed (see chap. 6). Green infrastructure and stormwater management practices are being installed in some recently converted areas to minimize the impacts of urbanization on streams and rivers (Hogan and others, 2014). In Clarksburg, Maryland, a suburb of Washington, D.C., long-term monitoring has tracked changes in the concentrations of nitrate in three watersheds to compare (1) an agricultural watershed converted to low-impact development with a high density of infiltration management practices (referred to as the pre- and post-development treatment watershed), (2) an established urban control (for example, developed in the 1990s), and (3) a forested control (fig. 8.S2). Results showed elevated nitrate concentrations (>2 milligrams per liter) in the stream and groundwater during base flow conditions in the predevelopment period in the treatment watershed because of the previous agricultural land use (Hopkins and others, 2017). Nitrate concentrations were the highest in groundwater before the watershed was converted from agriculture to suburban development. In contrast, nitrate concentrations were lowest in the forested area, followed by the older urban control watershed. Although concentrations in the surface water and groundwater declined in the 5 years after development, they were still higher than the forested and older developed areas, suggesting nitrogen was still being flushed from soil and groundwater storage, possibly from the stormwater management practices implemented to enhance infiltration. Such results demonstrate that the impacts of developing land and stormwater management on water quality can lead to surprising and unexpected results (Hopkins and others, 2017). Although nitrate concentrations have generally decreased in areas of new development, nitrogen export has remained relatively constant because the volume of stream base flow has increased (Bhaskar and others, 2015).

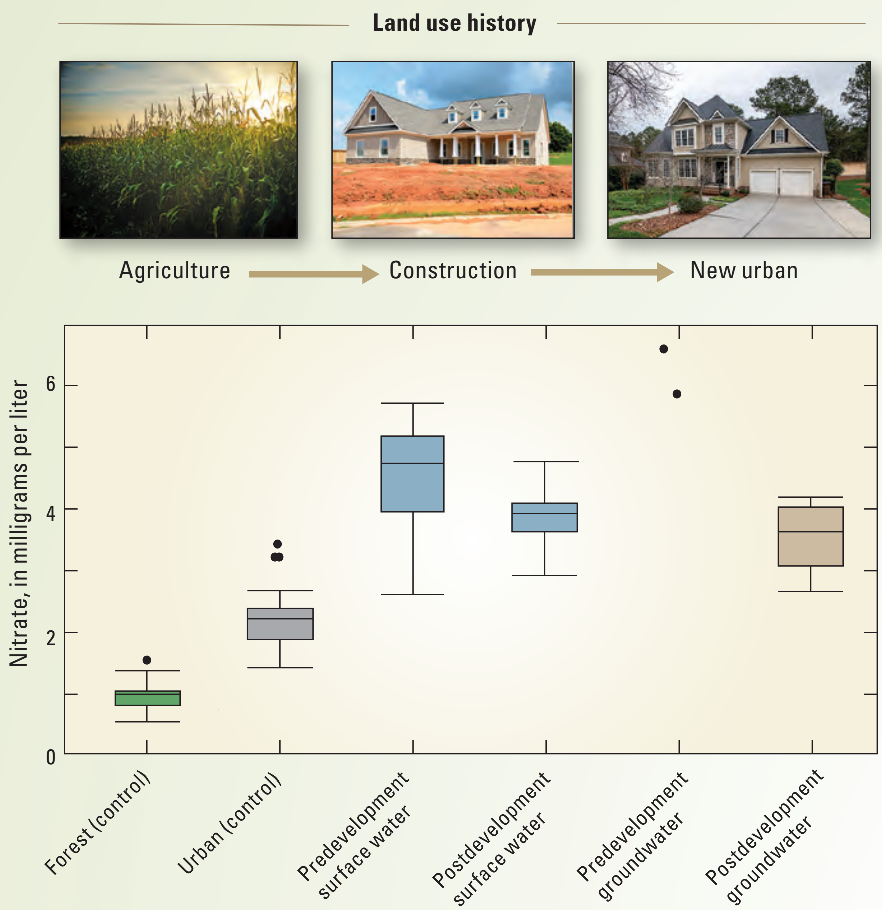

Figure 8.S2. Baseflow water-quality monitoring results from three watersheds in Clarksburg, Maryland: a nearby forested area, an established urban area, and the studied watershed that underwent conversion from agriculture to suburban development in the early 2000s. Predevelopment and post development water-quality results for a stream and a nearby well in the watershed are shown (Hopkins and others, 2017). Images courtesy of Pexels, used with permission

practices began to be adopted, including erosion and sediment control practices, bioretention basins, and stream restoration. In 2010, the Chesapeake Bay TMDL spurred the implementation of even more urban nitrogen management practices, especially stream restoration and bioretention (fig. 8.2).

Long-term water-quality monitoring in urban watersheds illuminates the effects of urban expansion and changes in nitrogen management on water quality. The impact of reduced point source contributions is clearly observed in monitoring data from the Patuxent River, located in central Maryland, where the annual flow-normalized nitrogen load has declined 42 percent from 1990 to 2012, approximating declining WWTP loads over the same time period (fig. 8.3*A,D*). Overall, point source contributions to nitrogen loads across the entire Chesapeake Bay watershed declined by approximately 50 percent from 1992 to 2012 (Ator and others, 2019). Ator and others (2019) suggested that nonpoint nitrogen loading from developed areas to streams also declined by 28 percent from 1992 to 2012, primarily near older urban centers, such as Baltimore, Maryland; Washington, D.C.; and Richmond, Virginia. For example, nitrogen loads are decreasing in the established urban Accotink watershed, located in northern Virginia (fig. 8.3*B,E*). However, in areas that are still undergoing urban development, such as Difficult Run (Reston, Virginia), nitrogen loads continue to increase (figs. 8.3*C,F*; Hyer and others, 2016). Another complicating factor that influences patterns in nitrogen loads from developed areas is the legacy of land use history prior to conversion to developed land (Hopkins and others, 2018). Although land use conversion from agriculture to developed land, in the long-term, typically reduces overall inputs of nitrogen to the landscape (see chap. 9), the change in the hydrologic system as a result of urbanization may initiate short-term flushing of nitrogen stored in soils and groundwater (see sidebar–What Happens to Stream Water Quality When Agricultural Land is Developed?). As developed landscapes expand and mature, water-quality trends in local streams may reflect the overall improvements owing to increased nitrogen management.

The Future: 2015–2050

Future impacts of developed landscapes on the export of nitrogen to the Chesapeake Bay will depend on many factors, including land use conversions to accommodate future urban growth, changes in point sources from a growing population, advances in wastewater treatment, climate change, changes in urban infrastructure, and changes in nonpoint nitrogen management. Human populations are anticipated to continue to increase in the Chesapeake Bay watershed over the next several decades, with most of this growth concentrated in suburban and developed regions (see chap. 6). Although new development practices impact nitrogen pollution, predevelopment land uses are also important (Ator and others, 2020). If future development occurs in previously forested areas, nitrogen loads will increase, whereas the conversion of agriculture to development could result in lower overall nitrogen loads from developing watersheds. However, it’s important to note that streams in developed regions are affected by a myriad of factors besides nitrogen, including other contaminants like phosphorus, sediment, chloride, organic chemicals, and temperature (Paul and Meyer, 2001). Although future growth in developed land in the Chesapeake Bay watershed may result in reduced nitrogen loads entering the Bay, development may also contribute to more widespread stream ecosystem degradation in urban streams owing to the increased prevalence of these other ecological stressors (Walsh and others, 2005).

Management of point sources from developed land for nitrogen will continue to change in the future. Over the past two decades, a considerable effort has gone into the successful reduction of nitrogen discharged from WWTPs into the Bay’s streams (see Historical Setting of the Chesapeake Bay Watershed chapter sidebar–Changes in the Potomac River in the 20th Century). The future of continued nitrogen reductions from wastewater will be dictated by a balance among several factors. It is well established that population will continue to increase in the future, largely in existing and expanding developed areas. More people will generate more nitrogen-containing wastewater. As developed areas grow, nearby areas where houses currently use onsite septic systems will become part of denser developed areas. These areas will likely become sewered with wastewater redirected to treatment plants. In the future, the increased nitrogen loads delivered to WWTPs, due to increased population, could produce an increase in the nitrogen load in treated effluent leaving WWTPs. This increase could be counterbalanced by planned and potential improvements in WWTPs. Enhanced nitrogen removal technologies have been implemented in 75 treatment plants in Maryland since 2006 and 14 more are planned for completion by 2022 (Maryland Department of the Environment, 2020). These enhanced treatment technologies reduce the average annual concentration of nitrogen in effluent (Maryland Department of the Environment, 2020). There are new technologies being developed that could further reduce the nitrogen in wastewater discharge. Generally, enhanced nitrogen removal technologies are used in the larger WWTPs (Hopple and others, 2021). For smaller communities, the treatment of wastewater is performed by less advanced technologies due to cost and the limited volume of waste that needs to be treated. For these smaller communities, the nitrogen loads in wastewater discharges will likely increase as population increases. However, smaller communities only contributed a small percent of the nitrogen from all the Chesapeake Bay watershed wastewater treatment in 2017 (Hopple and others, 2021).

Advancements in nonpoint nitrogen management is anticipated to continue as land continues to be developed (Lucke and others, 2019). For example, areas treated with urban stormwater management in the Baltimore, Maryland region increased by almost 20 square kilometers from 1999 to 2014 (Reisinger and others, 2019). However, as existing stormwater infrastructure ages and design capacities are surpassed, increased export of nitrogen may occur if infrastructure is not properly maintained. Cities will continue to invest in both gray infrastructure (pipes and storage facilities for storing wastewater and stormwater) and green infrastructure (distributed stormwater practices that mimic natural hydrological processes to reduce stormwater runoff). Indeed, many cities in the region are separating their combined sewer systems and/or adding green infrastructure to increase stormwater retention, which may help decrease nitrogen exports (Hopkins and others, 2018). For example, Washington, D.C., is investing in green infrastructure to reduce the inputs of stormwater into the combined sewer system in the Rock Creek watershed (Lim and Yiming, 2018). Although investments in nitrogen management may increase, climate change may offset these advancements. For example, increased rainfall intensity may reduce the effectiveness of stormwater management practices to retain nitrogen (Koch and others, 2014). High intensity rainfall can also increase the discharge from combined sewers (Reisinger and others, 2019), which leads to more untreated effluent discharged into streams. Whether or not future increased investments in nonpoint nitrogen management can keep pace with anticipated future increases in nonpoint nitrogen sources remains unclear.

Summary

Developed areas, although a minor fraction of overall land use in the Chesapeake Bay watershed, are a unique and complex component of the landscape. Historically, developed areas contributed a large fraction of nitrogen to the Chesapeake Bay through both point sources (WWTPs, industrial effluent) and nonpoint sources (atmospheric deposition, residential lawn fertilizer, and so forth). National, regional, and state regulations have changed the management of nitrogen in developed land, including regulations on point sources and on stormwater management. Consequently, the contributions of nitrogen in developed areas from point sources and nonpoint sources to the Chesapeake Bay have declined. Improvements in wastewater treatment, in particular, have been identified as a major driver of declining nitrogen export in major tributaries to the Bay. Point sources, controlled through regulations, will likely continue to play a major role in controlling nitrogen loads in urban streams and rivers. Management of nonpoint sources of nitrogen has been a more recent focus and will likely continue as urban centers expand across the watershed in coming decades. Although management practices to mitigate the effects of nonpoint sources of nitrogen will continue to be implemented with new development, the effectiveness of those management practices in the future is uncertain.

Modeling the Effect of Nitrogen Loads from Multiple Changes in the Watershed

Chapter 9

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Nutrient enrichment of streams and receiving waters has resulted in environmental consequences worldwide (Diaz, 2001), including in the Chesapeake Bay, where nitrogen loading has contributed to seasonal hypoxia (see chap. 1; Scavia and others, 2006; Turner and others, 2006; Testa and others, 2014). As described in preceding chapters, there are multiple sources of nitrogen and processes that influence the delivery of nitrogen to streams and receiving waters in the Chesapeake Bay watershed (fig. 9.1). When reactive (bioavailable) nitrogen is introduced to the land surface, some of it is temporarily stored in the terrestrial environment or lost permanently from the system through natural processes during transport such as denitrification (see Nitrogen Setting of the Chesapeake Bay Watershed chapter sidebar–Reactive [Bioavailable] Nitrogen). As a result, only a fraction of the nitrogen applied from each source is eventually discharged to streams and the Bay. Mathematical tools to integrate information about multiple sources of nitrogen and multiple processes influencing its transport can be used to estimate how much nitrogen is delivered from the terrestrial environment to streams and eventually the Chesapeake Bay. Watershed models are commonly developed and applied for this purpose and can estimate nitrogen loads from each source and from all sources combined. Such models can help provide an understanding of how nitrogen sources and transport processes vary spatially across the watershed and how natural and anthropogenic changes taking place in the watershed are expected to influence the amount of nitrogen delivered to local streams and the Bay. Watershed model results can be used by scientists and resource managers to compare the relative importance of various changes in the watershed to reduce nitrogen loads and subsequently identify management strategies for improving ecosystem health (Miller and others, 2019). This chapter describes how watershed models, specifically, the Spatially Referenced Regression on Watershed Attributes (SPARROW) model, are used as a management tool to identify sources of nitrogen to streams and receiving waters, and to assess the impacts of land use change and management practices on in-stream nitrogen conditions (see sidebar–Use of SPARROW to Identify Nitrogen Sources to Streams and the Bay).

[1]U.S. Geological Survey.

Mapleton Riverside Park in Huntingdon County, Pennsylvania.
Courtesy of the Chesapeake Bay Program, used with permission.

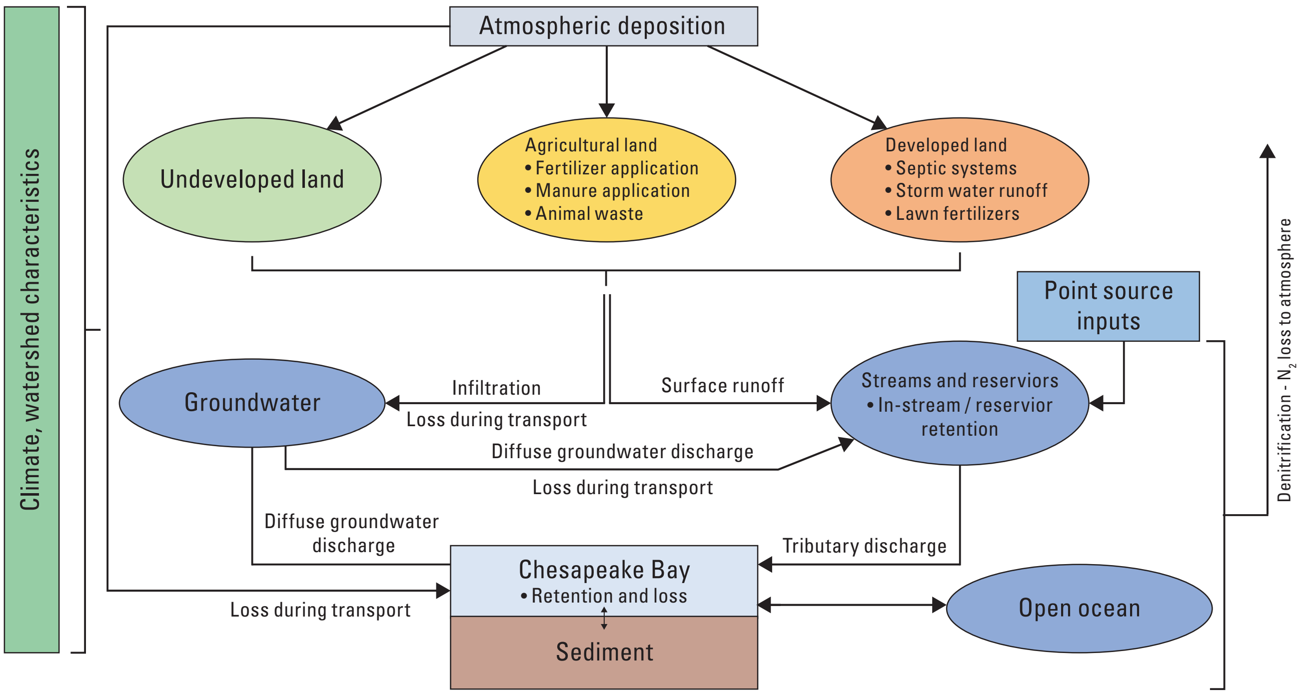

Figure 9.1. Conceptual model illustrating the sources of, and processes by which, nitrogen loads are generated and transported to streams and receiving waters in the Chesapeake Bay watershed. N, nitrogen.

Use of SPARROW to Identify Nitrogen Sources to Streams and the Bay

Spatially Referenced Regression on Watershed Attributes (SPARROW) models estimate the amount of a constituent, such as nitrogen, transported from sources to local streams and delivered to larger water bodies, such as the Chesapeake Bay (Smith and others, 1997). SPARROW models are developed by relating water-quality observations collected throughout a watershed to information on sources and watershed characteristics (fig. 9.S1). SPARROW models estimate long-term average nitrogen loads, yields, and concentrations for each stream reach in the stream network, and to the downstream Bay, including estimates of the contributions from each major source of nitrogen (generally, wastewater discharge, agricultural fertilizer, manure, atmospheric deposition, and developed areas). These models have improved the understanding of the spatial distribution and magnitude of the sources and watershed characteristics that influence the fate and transport of nitrogen at large (regional to national) scales (Smith and others, 1997; Preston and Brakebill, 1999; Ator and others, 2011; Moore and others, 2011).

One of the many benefits that SPARROW-estimated nitrogen contributions from individual sources provides is a framework for prioritizing sources and areas for conservation practices. Model-derived coefficients for sources such as agricultural land, developed land, or atmospheric deposition provide information on the mass of nitrogen from a given source that will be delivered to the local stream. Other model-derived coefficients estimate the losses of nitrogen within streams and reservoirs to provide estimates of the loads exported to the Bay. Additionally, SPARROW models provide information on the most important watershed characteristics affecting nitrogen delivery to streams (fig. 9.S2). These types of model-derived insights provide an opportunity to evaluate how the effects of changes in land use and implementation of land management strategies would be expected to change the loads of nitrogen delivered to local streams and exported to the Bay.

A SPARROW model for nitrogen in the Chesapeake Bay watershed estimated not only cumulative nitrogen loads to the Bay, but also the individual loads for more than 80,000

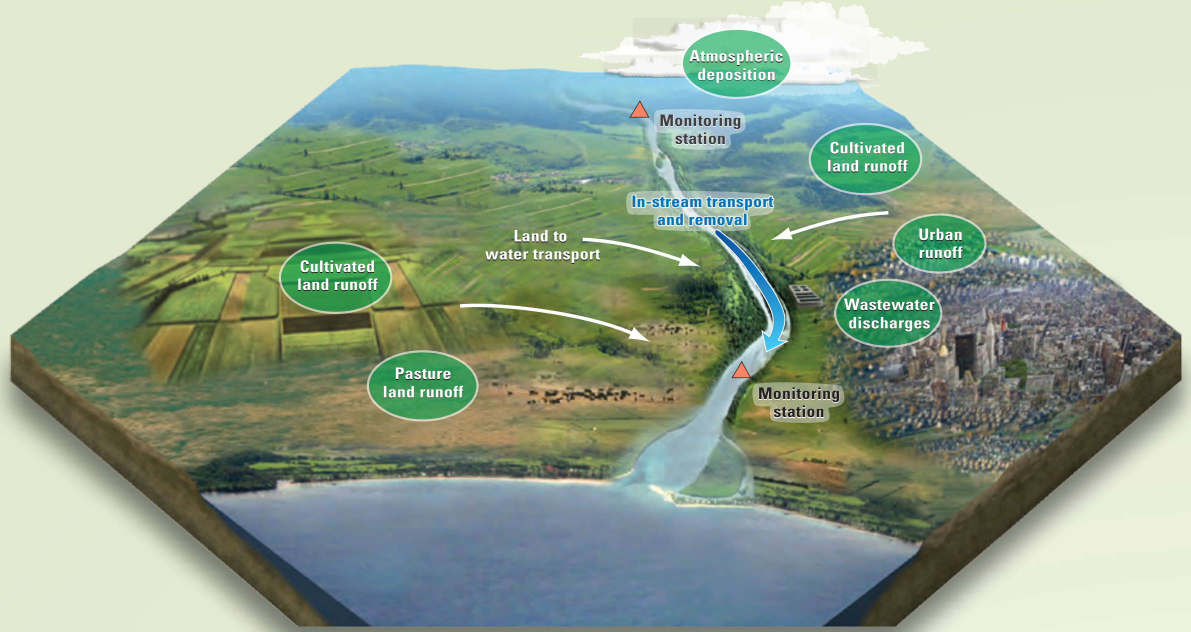

Figure 9.S1. Schematic of SPARROW model components. Adapted from Heinz Center (2008), graphic by Grabhorn Studios.

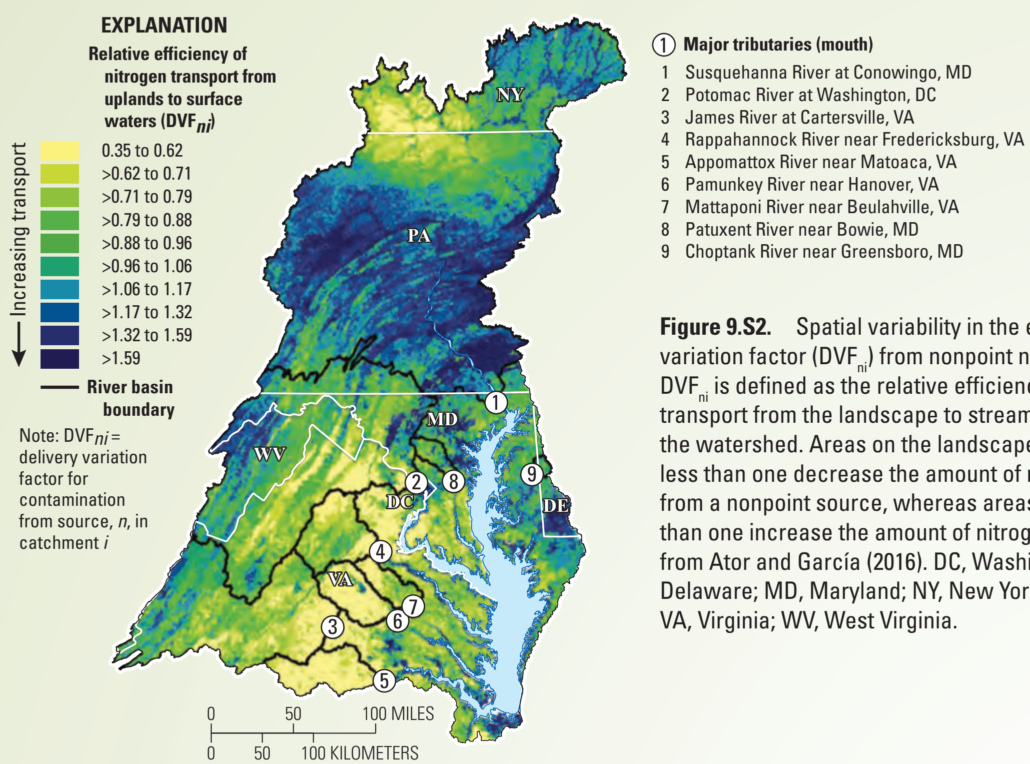

Figure 9.S2. Spatial variability in the estimated delivery variation factor (DVF$_{ni}$) from nonpoint nitrogen sources. DVF$_{ni}$ is defined as the relative efficiency of nitrogen transport from the landscape to streams for each reach in the watershed. Areas on the landscape with DVFni values less than one decrease the amount of nitrogen delivered from a nonpoint source, whereas areas with values greater than one increase the amount of nitrogen delivered. Figure from Ator and García (2016). DC, Washington, D.C.; DE, Delaware; MD, Maryland; NY, New York; PA, Pennsylvania; VA, Virginia; WV, West Virginia.

incremental stream catchments for a period centered on 2002 (Ator and others, 2011; Sekellick and others, 2021). The most statistically significant sources of nitrogen include wastewater discharge, crop fertilizer and fixation, manure, atmospheric deposition, and developed land use. At the Chesapeake Bay watershed scale, crop fertilizer application and fixation were estimated to contribute 45 percent of the nitrogen load to the Bay, with the remaining sources contributing between 9 percent (manure) and 17 percent (atmospheric deposition) of the load (fig. 9.S3). The dominance of fertilizer application and fixation as sources of nitrogen is consistent with studies in other large watersheds, such as the Mississippi River Basin (Alexander and others, 2008; David and others, 2010) and the Laurentian Great Lakes (Robertson and Saad, 2011). In addition to the Ator and others (2011) model for 2002, other SPARROW models for nitrogen in the Chesapeake Bay watershed have been developed for selected time periods in the late 1980s and early and late 1990s (Preston and Brakebill, 1999; Brakebill and others, 2001; Brakebill and Preston, 2004).

Additional advancements in incorporating time-variable source inputs, watershed characteristics, climate forcings, and water-quality constituent predictions into the SPARROW conceptual framework are underway; examples include Spatiotemporal Watershed Accumulation of Net Effects (SWAN; Chanat and Yang, 2018), decadal SPARROW (Ator and others, 2019) and dynamic SPARROW models (R.A. Smith, U.S. Geological Survey, written comm., 2020).

Identification of nitrogen sources and watershed characteristics influencing its fate and transport by SPARROW models provides insight into locations and processes in watersheds that can be targeted for conservation efforts. For example, the finding that human activities contribute a large fraction of nitrogen to the Chesapeake Bay—fertilizer application and fixation for example, are estimated to contribute 45 percent of the nitrogen reaching the Bay—suggests that management and restoration activities aimed at reducing nitrogen inputs from fertilizer sources may be most effective. Further, SPARROW models provide an opportunity to quantify expected nitrogen responses to changes in land use, control of wastewater discharges and atmospheric deposition, and implementation of agricultural and urban management practices. Indeed, testing different modeled scenarios allows for the assessment of stream response to changing conditions and practices in watersheds, which is a major benefit of SPARROW (García and others, 2016) and other water-quality models (see chap. 1 sidebar–The Chesapeake Bay Program Watershed Model; Santhi and others, 2001; Shenk and others, 2012; Shenk and Linker, 2013).

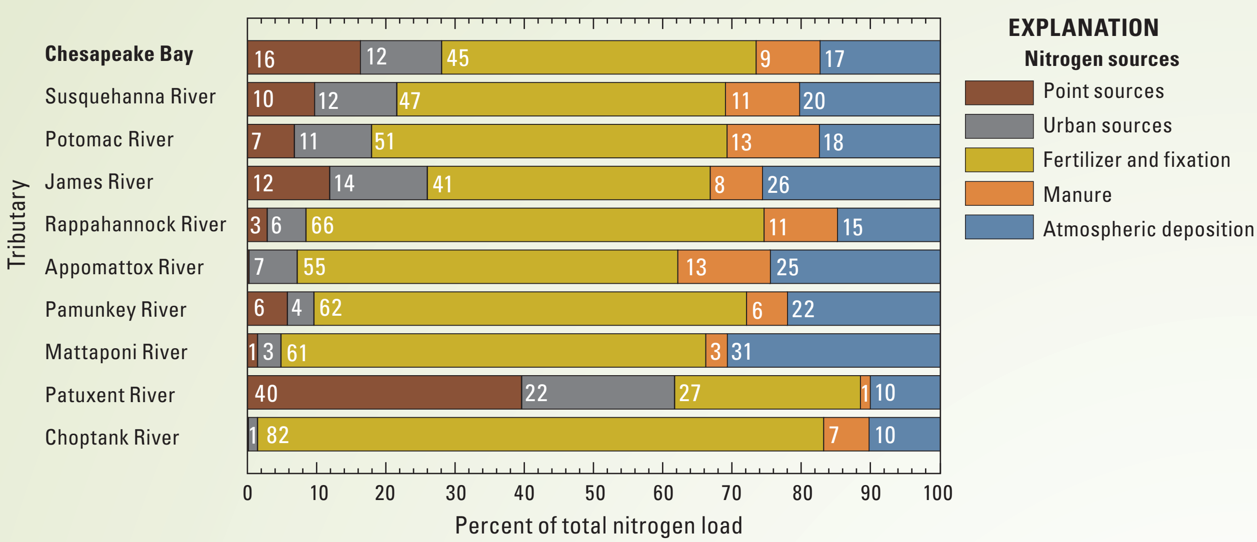

Figure 9.S3. Spatially Referenced Regression on Watershed Attributes (SPARROW) model-derived estimated shares of nitrogen derived from sources to the Chesapeake Bay and major tributaries. Figure from Ator and others (2011).

Only a Fraction of The Nitrogen Input into the Watershed is Delivered to Streams and the Bay

There are numerous nonpoint sources of nitrogen to the Chesapeake Bay watershed, including atmospheric deposition, agriculture, and developed areas. Some of the nitrogen input to the land surface is either temporarily retained or permanently lost from the system during transport. Processes that temporarily retain nitrogen include storage in groundwater that later discharges to streams, storage in soils prior to plant uptake or denitrification, and storage in vegetation that later decays and releases the stored nitrogen. Permanent losses include processes such as denitrification, which result in loss of nitrogen to the atmosphere, and uptake by agricultural crops, which are then harvested and transported out of the watershed. The net result of these retention and loss processes is that only a fraction of the nitrogen input into the watershed is delivered to the Bay (fig. 9.2).

Water-quality models can estimate the amount of nitrogen input into the watershed that is eventually delivered to streams. The amount of nitrogen delivered to streams is a function of the sources of nitrogen and local watershed conditions including climate, geology, soil types, and management choices. The amount of nitrogen delivered to streams from a given source has important implications for expectations regarding how a change in nitrogen loading to the watershed from a given source corresponds to a change in nitrogen loading to the local stream and eventually the Bay. For example, a recent SPARROW model (Ator and others, 2011; Sekellick and others, 2021) estimated that on

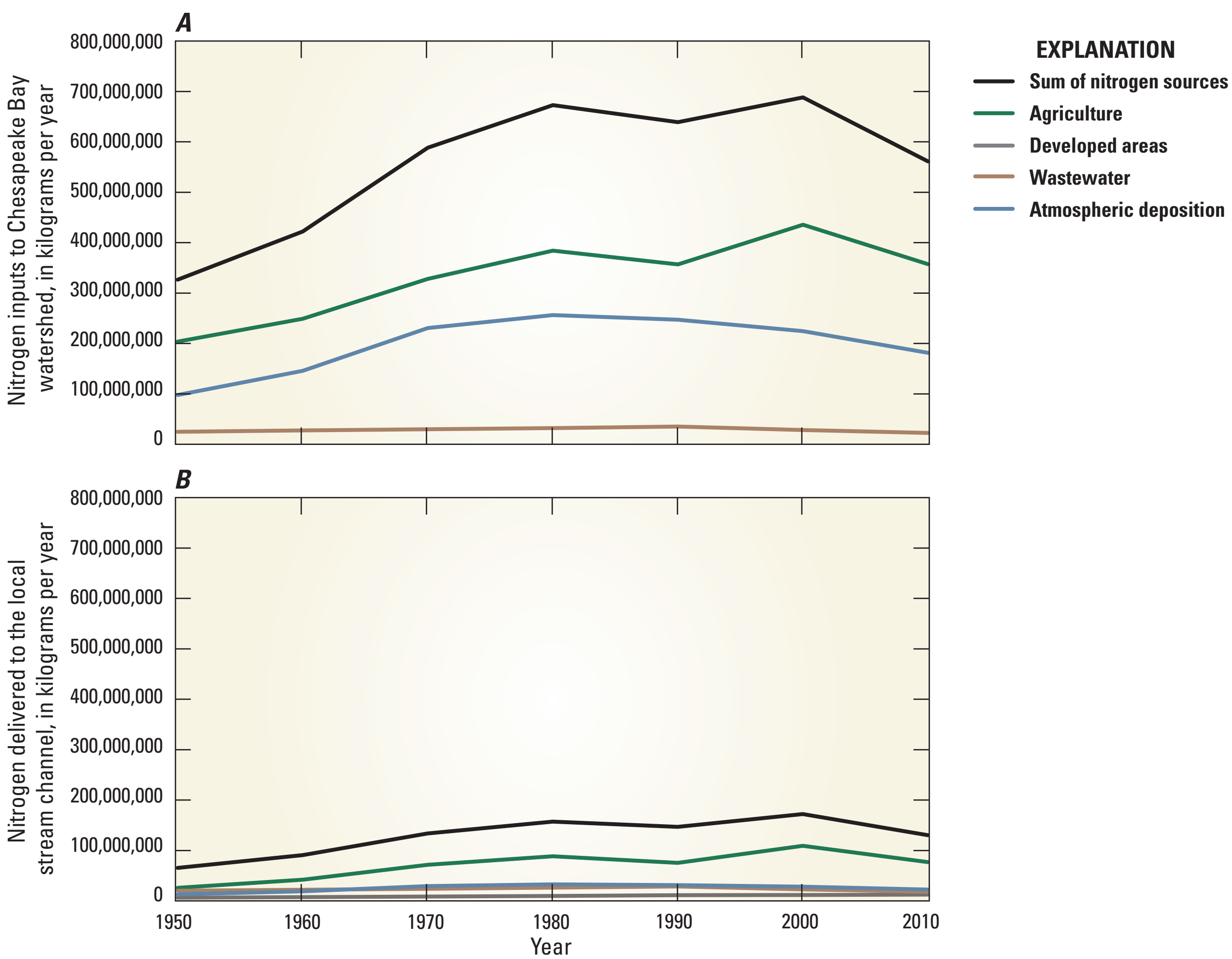

Figure 9.2. Only a fraction of the total nitrogen input into the watershed from a given source (***A***) is delivered to the local stream channel (***B***). Total nitrogen delivered to local streams was estimated by applying source coefficients from Ator and others (2011) to source inputs in ***A***. Estimated delivery to streams is a function of the source of nitrogen to the watershed and local watershed conditions such as climate, geology, and soil type. Land area from developed areas used to estimate nitrogen delivery is shown in figure NS.5 (Hopple and others, 2020, 2021).

average across the Chesapeake Bay watershed, 24 percent of nitrogen applied to agricultural fields as fertilizer and fixed by crops is delivered to streams, whereas 6 percent of nitrogen generated in the watershed as manure is delivered to streams. A large fraction of nitrogen in manure is lost to volatilization prior to application, however, these results still suggest that for an equal mass of nitrogen applied to agricultural fields as fertilizer or as manure, more of the nitrogen applied as fertilizer will be delivered to streams compared with the nitrogen applied as manure.

Water-Quality Models as Tools to Assess the Sensitivity of Nitrogen Loads to Changes in Land Use and Management

SPARROW models estimate long-term average nitrogen loads, yields, and flow-weighted concentrations (see chap. 2 sidebar–Quantifying Nitrogen: Concentrations, Loads, and Yields) in each reach in the stream network and receiving waters such as the Chesapeake Bay. SPARROW models also estimate coefficients (multiplication factors that describe the relative influence of each source or process) for the major sources of nitrogen and landscape characteristics that influence the movement of nitrogen from the landscape to the stream, as well as in-stream and in-reservoir processes that remove nitrogen from the system (see sidebar–Use of SPARROW to Identify Nitrogen Sources to Streams and the Bay). Changes in the amount of nitrogen delivered to local streams in response to changes in land use and source reductions can be estimated by applying source coefficients and information on watershed characteristics that influence the transport of nitrogen from the landscape to local streams. Nitrogen source reductions can be achieved through implementation of conservation practices that proactively control total nitrogen at its source (for example, reduced atmospheric total nitrogen emissions from vehicles or decreases in the mass of total nitrogen applied to fields as fertilizer and manure).

The response of nitrogen yield (nitrogen mass per year divided by watershed area) in local streams to 10 different land use change and source reduction scenarios was estimated using the Ator and others (2011) total nitrogen SPARROW model (fig. 9.3), as was the percent change in local yield (table 9.1). Application of the hypothetical

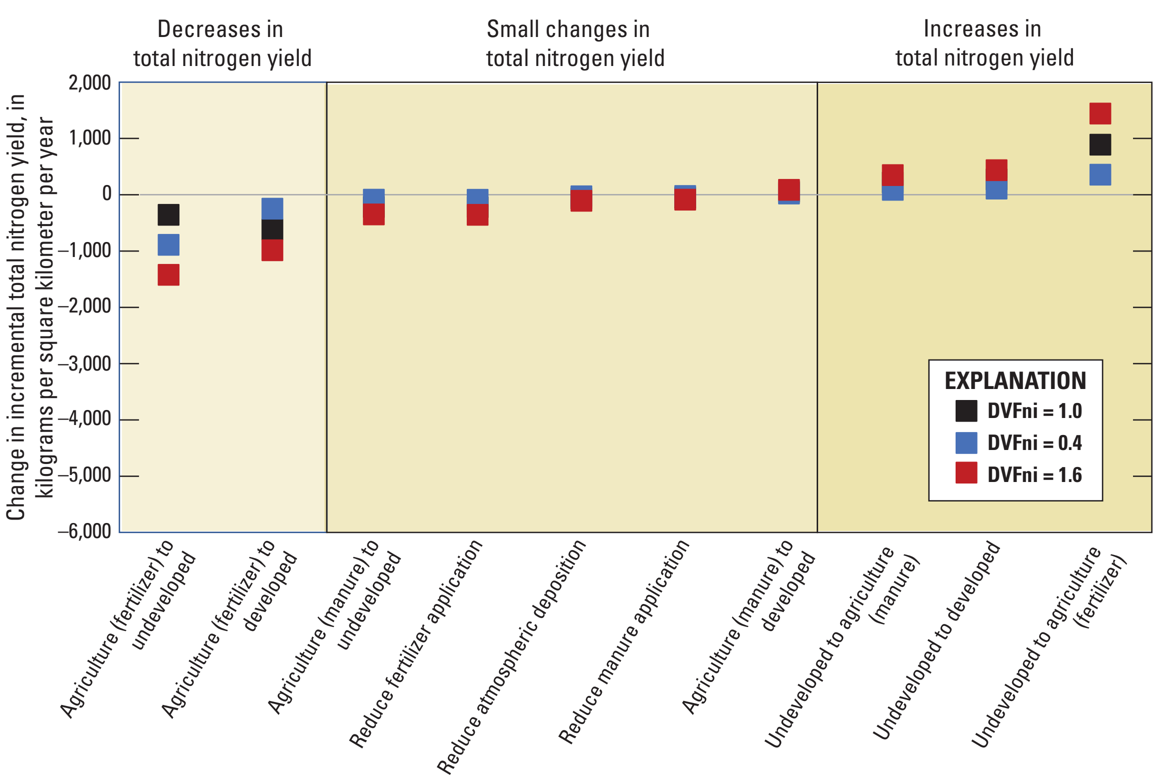

Figure 9.3. Estimated change in total nitrogen (TN) yield to the local stream for 10 different land use change and source reduction scenarios. For source reduction scenarios, the mass of the source in question was decreased by 25 percent in all catchments receiving this source, and the Chesapeake Bay Spatially Referenced Regression on Watershed Attributes (SPARROW) model (Ator and others, 2011; Sekellick and others, 2021) was applied with assumed source magnitudes to quantify TN response in local streams. For land use change scenarios, a 25-percent decrease in the original land use area was coupled with an equivalent increase in the new land use area. Delivery variation factor (DVF_{ni}) values represent the relative efficiency of TN transport from the landscape to streams.

Table 9.1 Predicted percent change in local total nitrogen yield in response to a 25 percent change in non-point source inputs estimated using the Ator and others (2011) SPARROW model. For source reduction scenarios, the mass of the source in question was decreased by 25 percent in all catchments receiving this source. For land use change scenarios, a 25 percent decrease in the area of the original land use was coupled with an equivalent increase in the area of the new land use.

Change in non-point source inputs	Percent change in local yield
Agriculture (fertilizer) to Undeveloped	-23
Agriculture (manure) to Undeveloped	-19
Agriculture (fertilizer) to Developed	-16
Reduced Fertilizer Application	-6
Reduced Manure Application	-5
Agriculture (manure) to Developed	5
Undeveloped to Developed	51
Undeveloped to Agriculture (manure)	82
Undeveloped to Agriculture (fertilizer)	333

modeling scenarios described here requires that assumptions about the magnitudes of total nitrogen assigned to each source. To reflect realistic conditions in the Chesapeake Bay watershed, assumed source magnitudes were informed by published values from the literature. For both fertilizer and manure, the magnitude of total nitrogen inputs was assumed to be 15,000 kilograms per square kilometer per year (kg/km^2/yr). This value is an approximate median value of recommended nitrogen application rates to agricultural fields in Pennsylvania (Penn State Extension, 2005), Maryland (University of Maryland Cooperative Extension, 2009) and Virginia (Virginia Cooperative Extension, 2019). The magnitude of atmospheric deposition was assumed to be 1,000 kg/km^2/yr, which is an approximate median value of total nitrogen deposition in the Chesapeake Bay watershed in 2013 (National Atmospheric Deposition Program, 2018). For land use change scenarios involving agricultural land, increases or decreases in agricultural land area receiving fertilizer or manure were represented by increases or decreases in the mass of nitrogen applied as fertilizer or manure, respectively. Estimated changes in local nitrogen yields in response to land use changes and source reductions were grouped into three categories: (1) decreases in nitrogen yield, (2) small changes in nitrogen yield, and (3) increases in nitrogen yield.

Conversion of agricultural land receiving fertilizer application to either undeveloped or developed land showed the highest potential to reduce nitrogen delivery to local streams, whereas increases in local nitrogen yield were greatest for land use conversions from undeveloped to agricultural or developed land (fig. 9.3). Reductions in source inputs by 25 percent had minimal impacts on estimated local nitrogen yields (source reduction scenarios such as the implementation of BMPs) (table 9.1).

In addition to changes in land use or implementation of conservation practices (BMPs), as represented by source reduction scenarios, conditions in individual watersheds within the Chesapeake Bay watershed also influence the change in nitrogen yields to local streams (fig. 9.3). It is notable that changes in percent yield are independent of watershed characteristics. The relative efficiency of nitrogen transport from the landscape to streams for each reach in the watershed is estimated by the model and is defined as the delivery variation factor. Areas on the landscape with delivery variation factor values less than one decrease the amount of nitrogen delivered from a nonpoint source, whereas areas with values greater than one increase the amount of nitrogen delivered. The Ator and others (2011) Chesapeake Bay nitrogen SPARROW model identified nitrogen transport efficiency as being greatest in areas with (1) more groundwater recharge, (2) a large fraction of the watershed area underlain by Piedmont carbonate rock, (3) limited available water capacity, and (4) a low value for the Enhanced Vegetation Index, which is a measure of the relative density and health of vegetation (Ator and others, 2011). These findings highlight the need to consider watershed characteristics in addition to controlling source inputs when estimating water-quality changes in response to land use changes or implementation of conservation practices.

Although the change in nitrogen yield analysis described above indicates that a 25 percent land use change has a greater potential to decrease or increase local nitrogen yields than does implementation of 25 percent source reductions, this does not indicate that management practices or changes in crop uptake of nitrogen, which have increased in recent years (Byrnes and others, 2020), are ineffective at reducing nitrogen loading to streams. Nitrogen reduction efficiencies of agricultural management practices in the Chesapeake Bay watershed have been estimated to range from 10–15 percent for continuous no-till and 19–65 percent for the implementation of forest buffers (U.S. Environmental Protection Agency, 2010a). To assess the sensitivity of

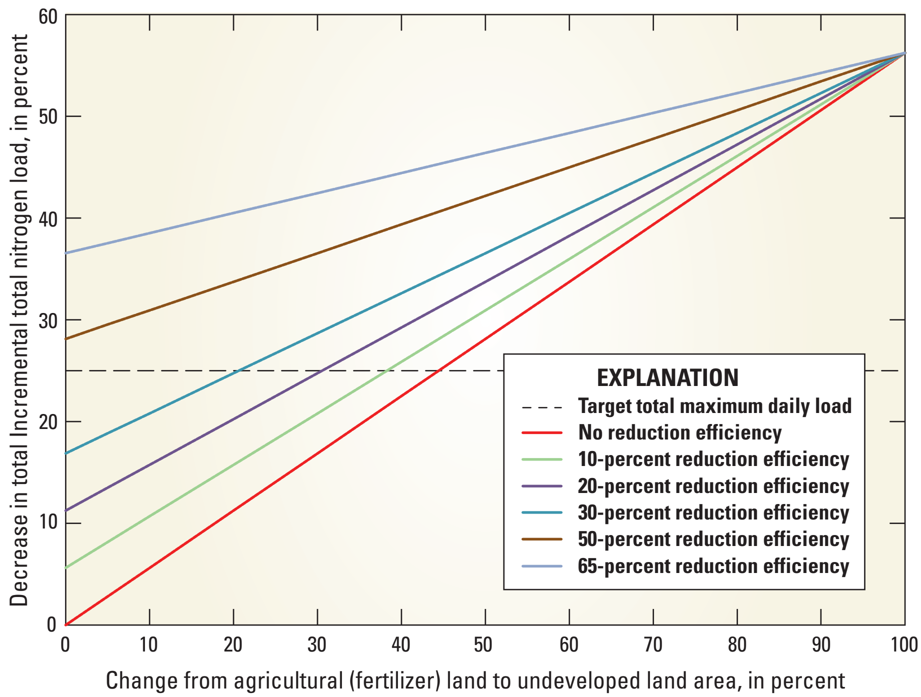

Figure 9.4. Estimated decrease in total nitrogen load to local streams as a function of changing land use from agricultural land receiving fertilizer application to undeveloped land. The Chesapeake Bay SPARROW model for nitrogen was applied with assumed source magnitudes to estimate changes in total nitrogen (Ator and others, 2011; Sekellick and others, 2021). Different lines represent source reductions of varying efficiencies for nitrogen reduction strategies applied to the agricultural land remaining after land use conversion. The horizontal dashed line represents the target 25-percent load reduction.

local nitrogen response to the range of expected nitrogen reduction efficiencies associated with conservation practices in the Chesapeake Bay watershed, the Ator and others (2011) SPARROW model was used with the aforementioned magnitudes of total nitrogen assigned to each source (Miller and others, 2019). This model provided the ability to predict local nitrogen loading to streams in response to changes in land use from agricultural land receiving fertilizer application to undeveloped land, coupled with implementation of a range of source reduction efficiencies (fig. 9.4). An estimated 44-percent change in land use area from agricultural land to undeveloped land would be required to meet a target local nitrogen reduction of 25 percent with no reduction efficiencies in place. The required land use change was 38 percent when coupled with a 10-percent reduction efficiency applied to the remaining agricultural land, 31 percent when coupled with a 20-percent reduction efficiency, and 21 percent with a 30-percent reduction efficiency. No land use change was required to reduce local nitrogen loads by 25 percent for scenarios when the nitrogen reduction efficiency was greater than about 45 percent. Given potential challenges associated with converting large areas of agricultural land to undeveloped or developed land, these results suggest that efforts targeting improvements in nitrogen reduction efficiencies or applying multiple management practices to the same landscape may be an effective approach for mitigating nitrogen loading to local streams in the Chesapeake Bay watershed.

Summary

The amount of nitrogen present in streams in the Chesapeake Bay watershed, and in the Bay itself, comes from numerous sources distributed throughout the watershed and is influenced by both terrestrial and in-stream processes as the nitrogen moves through the system. Mitigating unwanted impacts of stream nitrogen requires disentangling the multiple sources and processes influencing nitrogen transport and delivery. Watershed models have been shown to be effective tools for this purpose. The SPARROW model has identified crop fertilizer and fixation as the dominant source of nitrogen to the Chesapeake Bay, contributing an estimated 45 percent of the total nitrogen load (Ator and others, 2011). The ability of water-quality models to partition nitrogen among sources provides an opportunity to assess the anticipated changes in nitrogen loads to changes in land use, management practices, and climate. The model used in this chapter suggests that changes from agricultural land receiving fertilizer application to undeveloped (fallow) or developed land are likely to result in the largest decreases in nitrogen loads to local streams in the Chesapeake Bay watershed. However, the large-scale implementation of agricultural and urban management practices to control nitrogen are also an effective approach for reducing nitrogen loads to local streams.

Watershed Scale Changes in Nitrogen Export: Past and Future

Chapter 10

By Paul D. Capel[1], Andrew J. Sekellick[1], John W. Clune[1], Richard A. Smith[1], and Matthew P. Miller[1]

Monongahela National Forest in Pendleton County, West Virginia. Courtesy of the Chesapeake Bay Program, used with permission.

Annual inputs and exports of nitrogen to and from the Chesapeake Bay watershed have changed considerably over time (figs. NS.5 and NS.6). The excess nitrogen that reaches the Bay, compared to natural conditions, has had a great impact on the ecological and economic health of the Chesapeake Bay and surrounding watershed (see chap. 1). Natural and human activities will continue to affect the amount of nitrogen exported to the Bay in the future. Water-quality resource managers have varying degrees of control over the movement of nitrogen to the Bay through strategies such as long-term planning, implementation of technologies, economic incentives, influence on societal decisions and voluntary actions, and regulation (fig. OV.2, Linker and others, 2013a).

The export of nitrogen from the Chesapeake Bay watershed to rivers and streams has been well documented through monitoring studies (see chap. 2). The extent of excess nitrogen loads to streams is well documented since the mid-1980s (see chaps. 2 and 5), and the amount of nitrogen discharged from wastewater treatment plants has been documented since 1984. Relatively less is known about the extent of nitrogen contamination of the hydrologic compartments in the watershed before these monitoring efforts began. However, monitoring studies in the Chesapeake Bay watershed (see chap. 2) and in other watersheds across the Nation (Dubrovsky and others, 2010), combined with mechanistic studies of natural and human-impacted processes, have led to a deep understanding of nitrogen behavior and movement through water, air, and soil in natural, agricultural, and built environments.

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Models are used to simulate the complex interactions of nitrogen in the environment, quantify the relative importance of different sources, extrapolate to unmonitored areas, and provide predictions for the future (see chap. 1 sidebar–The Chesapeake Bay Program Watershed Model; chap. 9). Models are also used as decision-support tools to evaluate the effects of alternative management practices and decisions. For this report, the SPARROW model was used to make hindcasts and forecasts of nitrogen export from the watershed to the Bay for the century from 1950 to 2050 (see chap. 9 sidebar–Use of SPARROW to Identify Nitrogen Sources to Streams and the Bay; Ator and others, 2011). The nitrogen exports were modeled until 2010 based on historical data for nitrogen sources through 2012 and then future scenarios were projected in decadal time steps to 2050 (Hopple and others, 2020, 2021).

The estimated inputs of nitrogen into the Chesapeake Bay watershed are better known for the years after 1983 compared to the period before because better records are available for land use, crop and animal agriculture, wastewater treatment discharges, and atmospheric deposition of nitrogen in recent years. Sekellick (2017) reconstructed past nitrogen inputs to the watershed from fertilizer and manure. In a comparable manner, Hopple and others (2021) reconstructed past nitrogen inputs to the watershed from atmospheric deposition (Hopple and others, 2020; Burns and others, 2021), wastewater treatment discharges for the time period 1950–2012, and changes in developed land areas. The land use changes (fig. 10.1) and sources of nitrogen (fig. 10.2) provide the basis for hindcasting the past exports of nitrogen to the Bay (Hopple and others, 2020, 2021).

Forecasts of nitrogen exports from the watershed to the Bay are based on scenarios of future changes in land use and nitrogen inputs. For this report, six scenarios were created to illustrate a range of possible futures for the export of nitrogen to the Bay. The conceptual description for each of the six scenarios are presented in figure OV.S1. The six scenarios are focused on management decisions in wastewater treatment and agriculture (both crop and animal production) that can affect the future exports of nitrogen to the Bay. All six scenarios had common assumptions for population and land use (figure OV.S1, fig. 10.2*C*), climate (no change as compared to 2012), and atmospheric deposition (see wet deposition shown in fig. 10.2*A*).

The changing land use and inputs of nitrogen from the past (1950–2012) are combined with the six future scenarios (2013–2050) as the basis for continuous hindcasts and forecasts of nitrogen export to the Bay over the period 1950–2050.

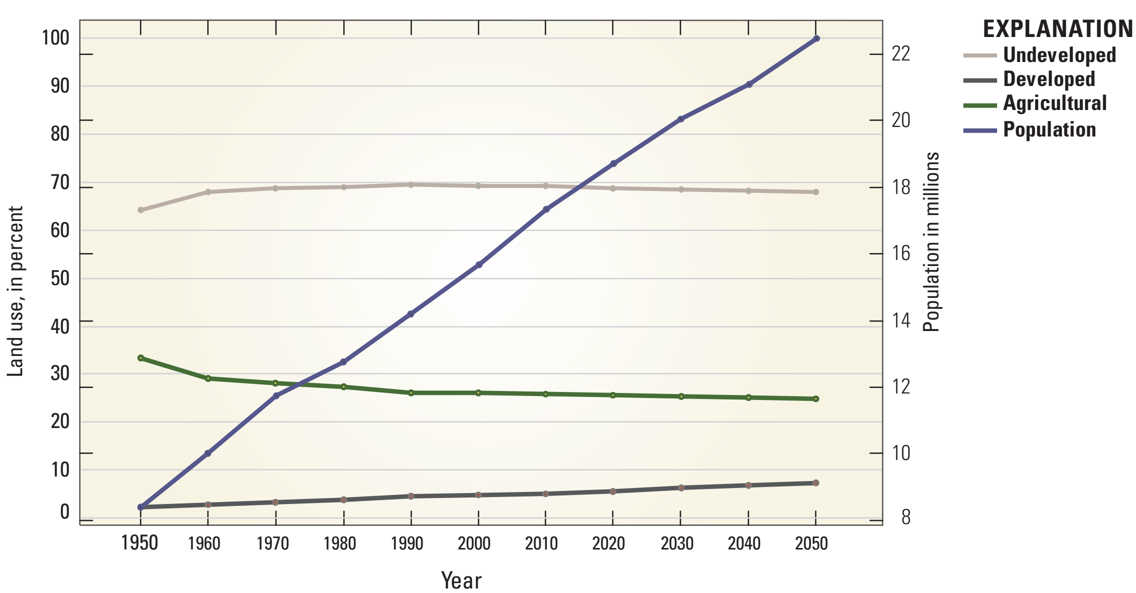


Figure 10.1. Changes in land use and population in the Chesapeake Bay watershed, 1950–2050 (Hopple and others, 2021).

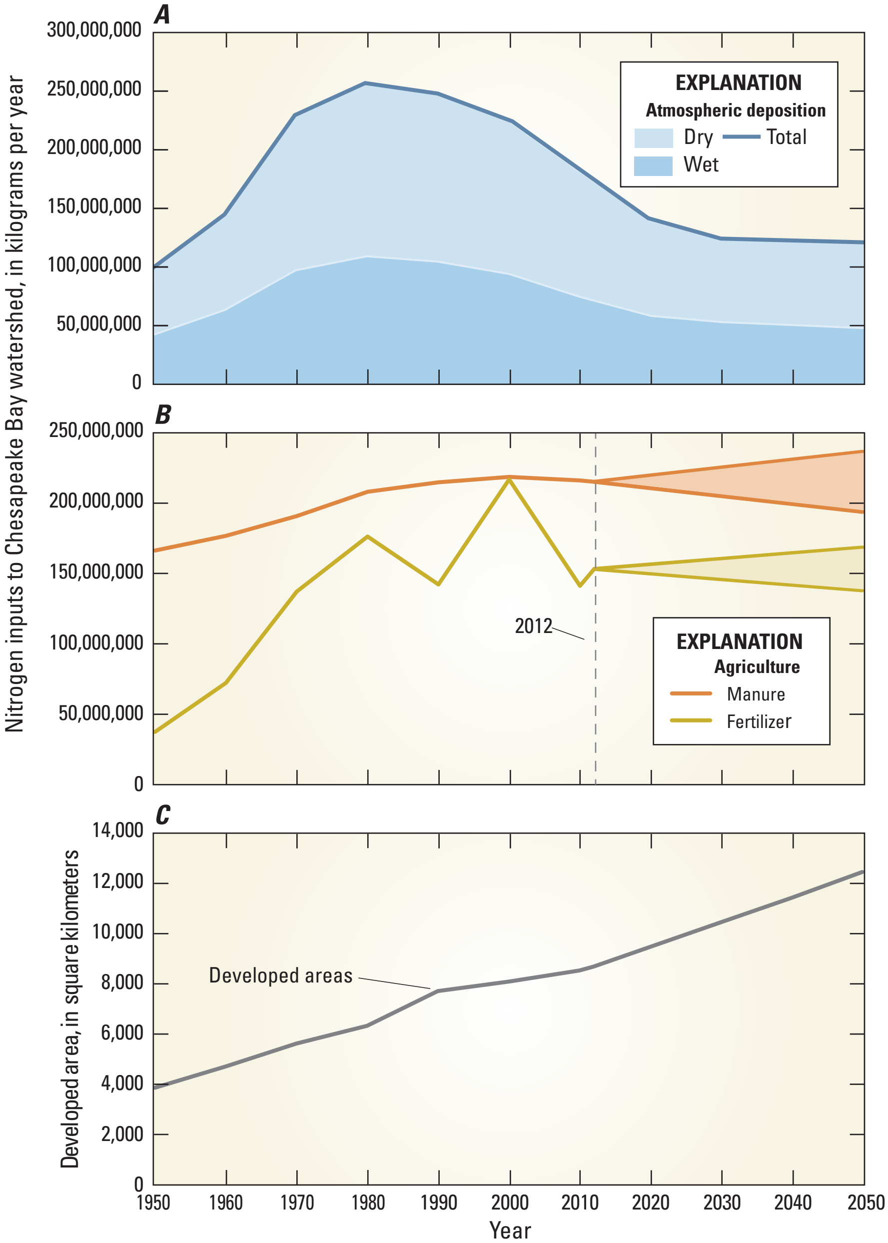


Figure 10.2. Annual nitrogen inputs to the Chesapeake Bay watershed, 1950–2050, from ***A***, atmospheric deposition, ***B***, fertilizer used in agriculture and manure from animal agriculture, and ***C***, developed areas. For fertilizer, manure, and point sources, two future scenarios are shown to bracket the boundaries of the realistic changes. The vertical black dashed line denotes 2012, which is the last year of measured data upon which the future projections were generated (Hopple and others, 2020, 2021).

Hindcasting the Past (1950–2012) and Forecasting the Future (2013–2050)

Nitrogen Inputs to the Chesapeake Bay Watershed

The inputs of nitrogen to the Chesapeake Bay watershed explicitly considered in this report include chemical fertilizer (including direct crop fixation), manure from agriculture, discharges from wastewater treatment plants, nonpoint sources of nitrogen from developed areas, and atmospheric deposition. The nitrogen imported to the watershed as human and animal food is captured in the wastewater and manure values, respectively. The nonpoint source nitrogen generated within the developed areas of the watershed (pet waste, vegetation waste, and so forth), is included as a nonspecific developed source. Septic systems are represented in the modeling of this report as part of the nonpoint sources from developed areas that are not otherwise specified in the model (see chap, 6 sidebar–Household Nitrogen Footprint and Septic Systems; Ator and others, 2011). Only the inputs of wet deposition were included owing to the limitations of the model (fig. 10.2*A*; Ator and others, 2011).

The relative importance of the nitrogen sources has varied over time (fig. 10.2); each source has had a unique historical trajectory. Nitrogen from atmospheric deposition increased substantially from 1950 to the early 1980s as a result of increased emissions from the transportation and industrial sectors (figs. 10.2*A* and 10.3*A*; see chap. 5). Since the 1980s, atmospheric nitrogen has decreased owing to the effects of the enactment of the Clean Air Act—a decrease that is projected to continue for the next few decades. Nitrogen from manure continually increased from 1950 to 2012 owing to the intensification of animal agriculture in some areas of the watershed (figs. 10.2*B* and 10.3*A*; see chap. 7). Nitrogen from chemical fertilizer had an overall increase over this same period, but there was considerable variability during the past few years owing to changes in crops (both type and yield), weather, and economics (demand for the crops grown in the watershed). The nonpoint inputs from developed areas cannot be directly measured but are linked to the size of the developed area, which has increased over time (fig. 10.2*C*; Ator and others, 2011). Nitrogen from wastewater treatment plants increased from 1950 to the early 1990s owing to increases in population. Since about 1990, the nitrogen sources from wastewater decreased as a result of the implementation of enhanced nitrogen removal technologies in many developed areas (fig. 10.3*A*; see chap. 8).

Nitrogen Exports to the Chesapeake Bay Watershed

The nitrogen delivered (exports) from the various sources to the Bay watershed was modeled for each decade over the century, 1950–2050 (fig. 10.3*B*). These modeling results are based on the combination of historical data (1950–2012) and six possible future scenarios for the Bay watershed (fig. OV.S1). All six future scenarios have common population growth and land use change assumptions, including the increase in the developed areas. Two of the scenarios explore the future export of nitrogen owing to population growth, with and without new implementation of enhanced wastewater nitrogen removal technologies. Four additional scenarios explore the future export of nitrogen owing to changes in agriculture (both crop and animal). One of the scenarios explores changes in agriculture, such as intensification in crop or animal agriculture, changes in crops owing to changes in climate, and

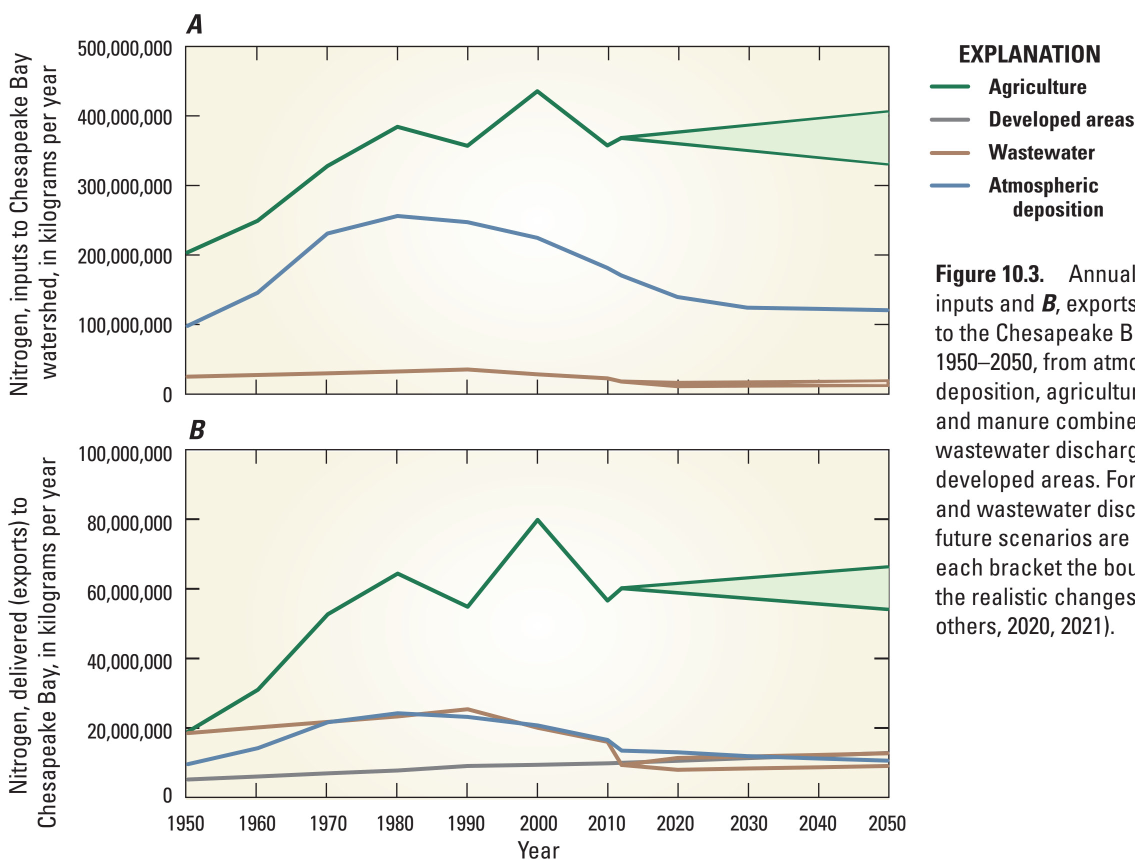


Figure 10.3. Annual nitrogen ***A***, inputs and ***B***, exports (delivered) to the Chesapeake Bay by source, 1950–2050, from atmospheric deposition, agriculture (fertilizer and manure combined), wastewater discharge, and developed areas. For agriculture and wastewater discharge, two future scenarios are shown for each bracket the boundaries of the realistic changes (Hopple and others, 2020, 2021).

increased fertilizer use owing to increased yields that uses a 10-percent increase in nitrogen inputs over the time period 2013–2050. Other scenarios assume a 10-percent decrease in nitrogen inputs from agriculture owing to a combination of changes including (1) increased numbers and efficiencies of management practices implemented, (2) increased areas of cropland in conservation programs or land retired from production, (3) decreased application rates of fertilizer, and (4) decreased manure production (and release to fields) owing to a combination of decreased numbers of animals, or the development of new technologies that decrease the amount of nitrogen available to the Bay. Scenarios 3 and 6 are meant to bracket the boundaries of the realistic changes that might occur in agriculture in the Chesapeake Bay watershed over the next few decades.

These SPARROW modeling results suggest that in 1950, one of the greatest sources of nitrogen delivered (exported) to the Chesapeake Bay watershed was wastewater discharge (fig. 10.3), a finding consistent with wastewater treatment practices during this period (see Historical Setting of the Chesapeake Bay Watershed chapter sidebar: Changes in the Potomac River in the 20th Century). Agriculture (fertilizer and manure) and atmospheric deposition were also dominant contributors of nitrogen and it is during this time period (starting in the late 1940s) that nitrogen chemical fertilizers were first widely available. Nitrogen exports delivered to the Chesapeake Bay watershed were lowest from developed areas during this time.

Over the next few decades, until the 1980s, the nitrogen loads from all five sources, particularly agriculture and to a lesser extent atmospheric deposition, continued to increase as a result of population growth and agricultural intensification. The water quality of the Bay declined over this time period in response to excess nitrogen, phosphorous, sediment, and other stresses on the system (see chap. 1). After the Clean Air Act was enacted in 1972, the atmospheric deposition source of nitrogen began to substantially decrease owing to reduced nitrogen emissions. Since the 1980s, this source has continued to decrease and is predicted to continue to decrease through 2050. Wastewater and atmospheric deposition loads have decreased and their relative importance to the export of nitrogen to the Bay has also decreased.

Although nitrogen from developed and agricultural areas contributed less to the Bay watershed during the past, loads have increased from both sources and are predicted to continue to increase as a result of the expansion of developed areas (increased population) and intensification of animal agriculture (increased animal counts). The export of nitrogen from wastewater began to decline in the 1990s and continued to decline in the following decades after implementation of improved nitrogen removal technologies. Two future (after 2012) scenarios for nitrogen from wastewater are shown in figure 10.3*B*. The scenario that has greater nitrogen export in 2050 assumes continued population growth with no new implementation of enhanced nitrogen removal technologies. The scenario with the lower nitrogen export in 2050 is based on the same population growth assumption, but also assumes new implementation of enhanced nitrogen removal technologies at the treatment plants for the rest of the major developed areas in the watershed.

Agriculture is projected to continue to be the largest source of nitrogen to the Bay. The annual export has continually increased over time, proportional to the increased use of fertilizer and manure (figs. 10.2*B* and 10.3*A*). In the 1950s, agriculture exports were similar to wastewater delivered to the Bay. In the 1980s to 2010s, agriculture contributed two or three times as much nitrogen as either wastewater or atmospheric deposition. Looking to the future, agriculture is predicted to contribute four to five times more nitrogen to the Bay watershed than any of the other sources.

The data in figure 10.3 are presented again in figure 10.4, but in a stacked format (cumulative area). The top of the colored area represents the total load of nitrogen entering the Bay each year from all five sources. The peak estimated nitrogen load to the Bay in 2000 of 129 million kilograms per year (kg/yr) was similar to other studies that estimated 132 million kg/yr (Ator and others, 2011) and 144 million kg/yr (Roberts and others, 2009) for the same year. Fertilizer and manure have been combined as a single agricultural source. From 1970 to 2050, agriculture was estimated to be the dominant contributor (approximately 50 percent or more) of nitrogen to the Bay, which is consistent with estimates of nitrogen loading to estuaries from other studies in the Chesapeake Bay watershed (Roberts and others, 2009; Ator and others, 2011) and nationwide (Castro and others, 2003).

The relative importance of agriculture as a source can be seen throughout the modeled time period. The sum of all sources of nitrogen to the Bay increased in an exponential fashion from 1950 until the 1980s. Many of the current water-quality problems in the Bay started or expanded during this time period. Between the 1980s and 1990s, the export of nitrogen fluctuated around 120 million kg/yr, depending on the amount of fertilizer used. Starting in 2010, the model suggests there has been a decrease in nitrogen export to the Bay owing to the combined effects of nitrogen reduction strategies initiated by the total maximum daily load (TMDL) and (or) the general fluctuation or a decrease in use of fertilizer. Both future scenarios shown in figure 10.4, which represent the extremes of realistic changes in agriculture in the watershed, suggest an annual export of nitrogen that presents a challenge to meeting the goals of the current TMDL. This suggests that an increased effort to reduce nitrogen from agriculture, even more than the 10-percent reduction of nitrogen inputs used in this model, would be necessary to meet water-quality and ecological goals (see chap. 1 sidebar–Largest Total Maximum Load in the Nation). The sources of nitrogen from wastewater and developed areas will likely continue to increase simply as a result of the increase in population in the watershed.

Spatial Patterns in Nitrogen Export Yields from Streams to the Chesapeake Bay

The past and future SPARROW model results can be viewed spatially; figure 10.5 depicts the export yields of nitrogen across the watershed over select years. Nitrogen yield allows an equal comparison for all areas (see chap. 2 sidebar–Quantifying Nitrogen: Concentrations, Loads, and Yields).

The increase in nitrogen yields through time from 1950 to 2010 is shown in figure 10.5*A–D*. In 1950, the nitrogen yields (mass of nitrogen per square kilometer) were greatest in the developed areas owing to wastewater discharge. Over the years, the areas that contributed nitrogen to the Bay increased as development spread and agriculture intensified in the watershed. The highest nitrogen yields were typically found in areas with prime soils and generally oxic groundwater, such as the Coastal Plain and areas underlain by carbonate aquifers, where nitrate is transported without major losses. There are many undeveloped (forested) areas of the watershed that will likely continue to contribute only minimal yields of nitrogen in the future. The export nitrogen yields from the watershed for the year 2050 for four of the future scenarios are shown in figure 10.5*E–H*. At this scale, the model results

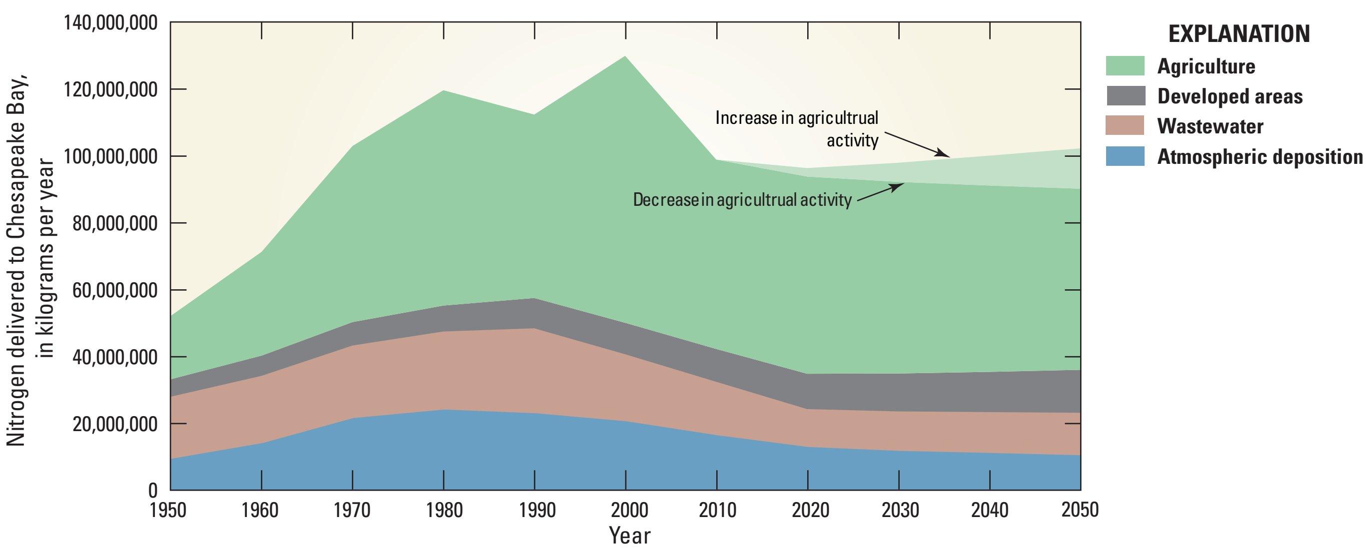


Figure 10.4. Stacked annual nitrogen loads exported to the Chesapeake Bay by source, 1950–2050. Fertilizer and manure are combined into a single agricultural source for the modeled time period after 2010 with two future scenarios: (1) increased intensity of both crop and animal agriculture, and (2) decreased intensity of both crop and animal agriculture (fig. OV.S1). The agricultural scenarios are within this envelope. Only the future scenario for constant wastewater treatment technology is shown (Hopple and others, 2020, 2021). Results are presented for the decadal increments used in modeling simulations.

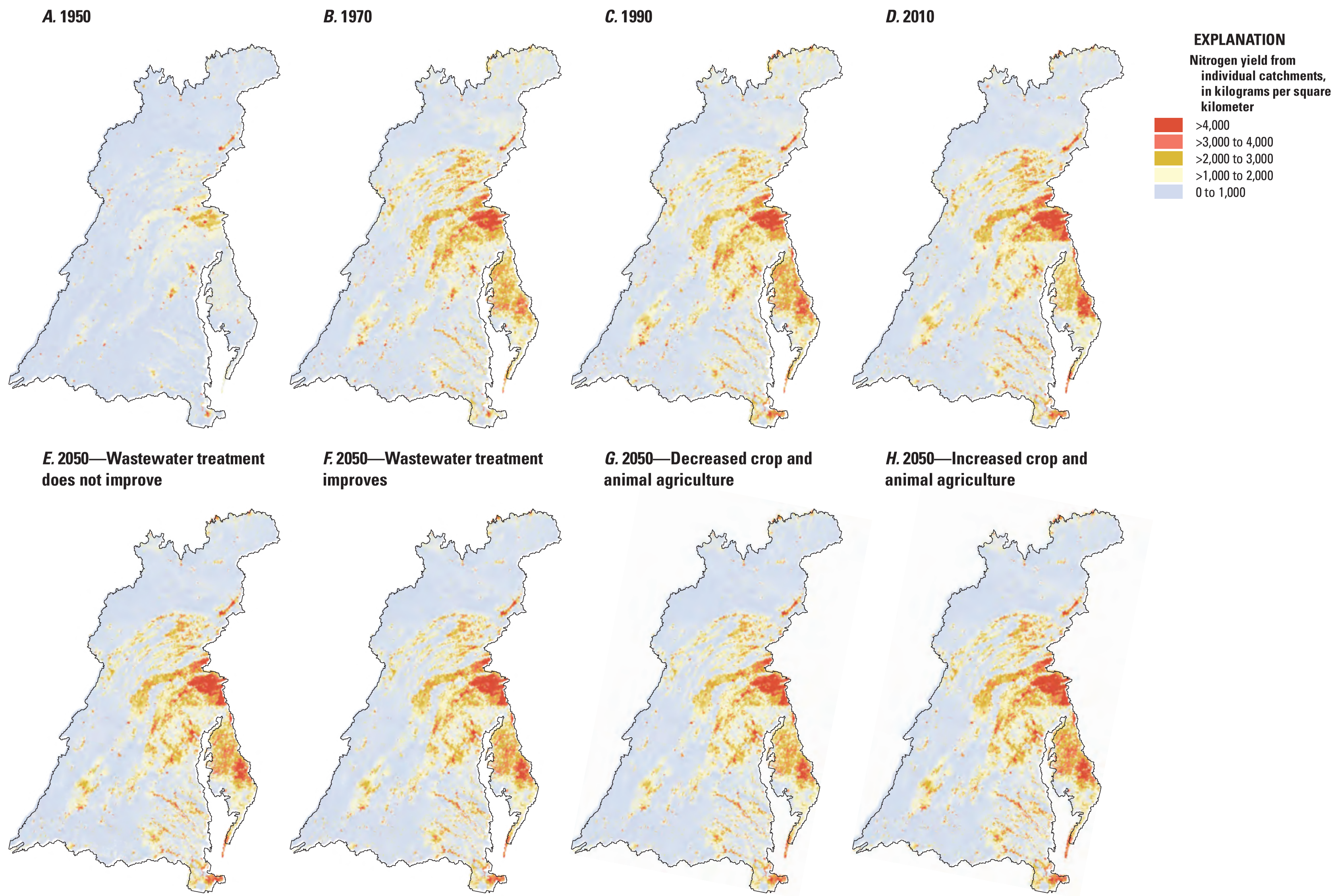


Figure 10.5. The spatial distribution of nitrogen yields from individual catchments in the Chesapeake Bay watershed for ***A***, 1950; ***B***, 1970; ***C***, 1990; ***D***, 2010; ***E***, 2050 with constant wastewater discharge; ***F***, 2050 with enhanced wastewater treatment technology; ***G***, 2050 with decreased crop and animal agriculture; and ***H***, 2050 with increased crop and animal agriculture (Hopple and others, 2020, 2021).

have subtle differences, but divergences can be seen for the nitrogen yields from areas of the Delmarva Peninsula when comparing the two scenarios for agriculture that are most different (fig. 10.5*G*,*H*). The results of the limited number of future scenarios presented here show subtle local differences, but no major regional differences in 2050 compared to 2010, presenting a challenge for watershed implementation plans and meeting the goals of the current TMDL and beyond (see chap. 1 sidebar–Largest Total Maximum Load in the Nation).

Lessons Learned from Modeling Nitrogen Change for a Century

The ability to analyze inputs of nitrogen to the watershed and exports to the Chesapeake Bay over time using SPARROW modeling provides a powerful tool for resource managers. The water-quality problems from excess nitrogen that were observed in the Bay in the last few decades of the 20th century led to the current TMDL. The human activities that occurred during the previous century laid the foundation for the water-quality problems that accelerated in the 1950s and beyond. Human activities related to urbanization, agriculture, and industry provided excess nitrogen to streams, groundwater, and the atmosphere. The continually increasing population led to increased nitrogen from wastewater treatment plants and septic tanks.

Over the past few decades, Federal, state, and local decision makers have set in motion different types of controls to reduce excess nitrogen in the environment (fig. 1.4). Starting in the 1970s, the Clean Water Act and Clean Air Act went into effect at the national scale. The jurisdictions composing the Chesapeake Bay watershed initiated many of their own policies and laws, particularly in response to the Chesapeake Bay Program (Clean Water Act, 33 U.S.C. § 1251 et seq.; 40 C.F.R. §§ 104.1). Wastewater treatment plants were upgraded and more focus was put towards urban planning and development. More effective and more extensive agricultural and urban management practices were encouraged and implemented, and with the TMDL in 2010 these various efforts were combined and quantified to reach living resource-based water quality standards for the tidal Bay by 2025 through nutrient reductions. The coordinated efforts included every level of government from Federal to state and local; each became responsible for their own nutrient reduction targets. Many of the final, practical, everyday decisions are made by resource managers, agricultural producers, and residents of rural and urban environments. The decisions made for their lawns, fields, livestock, pets, and maintenance on septic systems are important for reaching the goals of healthy and sustainable ecosystems in the Bay.

SPARROW nitrogen modeling provides a long-term perspective on water-quality changes over the century 1950–2050 by presenting past water quality issues and the beginnings of improvements in water quality. Modeled results suggest that the water quality of the Bay could continue to be problematic, as quantified by projections that indicate reducing nitrogen will take significant effort. Excess nitrogen loads owing to atmospheric deposition and wastewater discharges to the Bay have decreased substantially (Ator and others, 2020). There are also some early indications that perhaps increasing management practices or climate change may result from declines in nitrogen yields from developed nonpoint sources and in some agricultural settings (Chanat and Yang, 2018; Ator and others, 2019). However, if population and developed areas continue to increase, so might nitrogen loads. Agriculture is the largest contributor of excess nitrogen to the Bay and even with the most environmentally favorable future scenario in the analysis—a 10-percent decrease in the nitrogen inputs released from agriculture—achieving and maintaining the nutrient reductions called for in the Chesapeake Bay TMDL will be a challenge.

The question then becomes, at the time horizon of decades, what are the changes that can be implemented to reach the water quality goals? The future scenario assumption of a 10-percent reduction in nitrogen inputs from agriculture fertilizer and (or) manure purposely lacks detail, as there are numerous possible pathways to these reductions, some of which are specified in the target nutrient reductions designed to achieve the living resource and habitat-based Chesapeake water quality standards by 2025 that are described in the Phase III watershed implementation plans (WIPs), which were completed and approved in 2019 (U.S. Environmental Protection Agency, 2019a). In this analysis, the generalized 10-percent reduction in agricultural nitrogen inputs could, for example, be achieved with a combination of practices including (1) decreased application rates of fertilizer, (2) increased numbers and efficiencies of agricultural management practices, (3) increased areas of cropland moved to conservation programs or land retired from agricultural production, (4) development of new technologies that decrease the export of nitrogen from fields, and (5) decreased numbers of animals. The specifics of the decision-making process that will reduce excess nitrogen coming from urban and agricultural areas over future decades will be difficult, just as the decisions have been over the past few decades toward meeting the TMDL goals. Nevertheless, resource managers will be challenged with making decisions and implementing initiatives for the benefit of the ecological and economic health of the Bay and for the residents of the watershed.

Excess Nitrogen Impacts on Coastal Areas Across the Nation and the World

By Ana Maria Garcia[1] and John W. Clune[1]

The acceleration of the nitrogen cycle in the Chesapeake Bay documented in this report is an epitome of the larger continental-scale biogeochemical patterns brought on by the current period (Anthropocene) where humans now are the dominant influence on the environment. A century ago, the idea that human activity could alter global cycles like that of nitrogen seemed inconceivable (fig. F1). However, land use changes such as the intensification of agriculture have dramatically altered the flow of nutrients, resulting in unprecedented global-scale transfers of nitrogen, phosphorus, and other nutrients (Galloway and others, 2008). The past century has been an inflection point, as new technologies accelerated nitrogen export through human development and new agricultural practices were adopted to satisfy a rapidly expanding population.

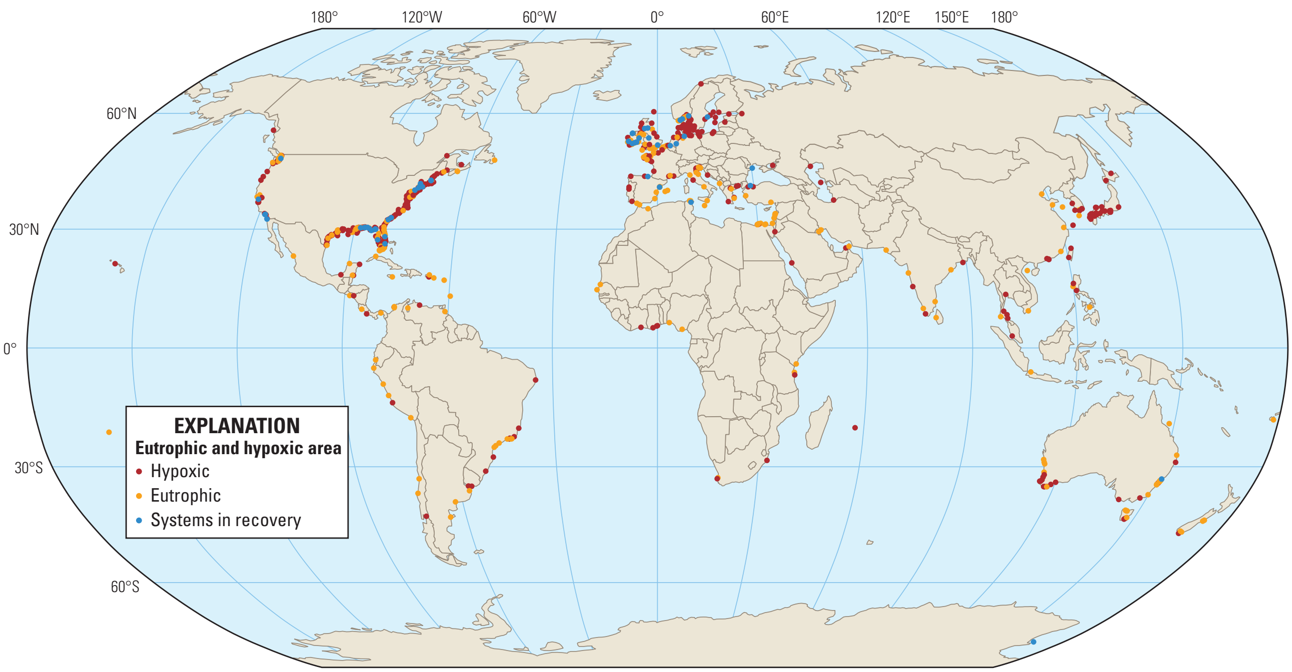
As a result of this disruption, coastal zones highly influenced by human-related activities around the world have seen increases in eutrophic conditions (Selman and others, 2008), which have led to an increase in harmful algal blooms, hypoxia, and declines in aquatic life. Across the United States, this impact is notable in many coastal environments like the Chesapeake Bay (Bricker and others, 2008), for example, from harmful algal blooms in the mid-Atlantic and Lake Erie to red tide outbreaks off the coast of Florida. In 2007, the National Oceanic and Atmospheric Administration performed an assessment of U.S. estuaries and found that 65 percent of assessed systems show moderate to high levels of impairment. Conditions were predicted to worsen for 65 percent and improve in only 19 percent of the assessed estuaries in the future (Bricker and others, 2008).

Sections of the Gulf of Mexico routinely make headlines as having the Nation’s largest dead zone, where recurring hypoxic events have meant massive losses to fisheries and natural ecosystems. The size of the hypoxic zone varies depending on climatic patterns; it reached a maximum in 2017 of 22,730 square kilometers (8,776 square miles) (Lu and others, 2020)—an area the size of New Jersey—and was nearly as large in 2019 owing to extensive flooding in the Midwest, which led to the transport of large volumes of nutrient rich runoff from agricultural lands. Consequently, if the hypoxic zone continues or worsens, fishermen and coastal state economies will be greatly impacted.

By 2050, the world is expected to have 9.8 billion people and the impacts on estuaries from excess pollutants like nitrogen will be felt by nearly half of the global population residing near coastal areas (Elliott and others, 2019). Ten of 18 million people residing in the Chesapeake Bay watershed live along or near the Bay shoreline (Chesapeake Bay Program, 2019b). Many of the same drivers and challenges outlined in this report are common among bays and watersheds throughout the world. Fortunately, effective solutions based on science can be shared worldwide to address legacy pollution and better mitigate future impacts on estuaries so they may be sustainable for ecosystem and societal services (Kennish, 2002; Elliott and others, 2019).

Photograph of coastline in St. Mary’s County, Maryland. Courtesy of the Chesapeake Bay Program, used with permission.

[1]U.S. Geological Survey.



Base map from Natural Earth. Data source: Diaz, R., M. Selman. and C. Chique. 2011. Global Eutrophic and Hypoxic Coastal Systems. World Resources Institute (2013). Eutrophication and Hypoxia: Nutrient Pollution in Coastal Waters. docs.wri.org/wri_eutrophic_hypoxic_dataset_2011-03.xls Robinson projection, WGS 1984

Figure F1. Global map of eutrophic and hypoxic areas (World Resource Institute, 2013).

Final Thoughts

By John W. Clune[1]

The Chesapeake Bay is a national treasure for the people living in its watershed and beyond. It is the country's largest estuary where freshwater from six states and our Nation's capital meets the saltwater of the Atlantic Ocean. The Bay's watershed provides habitat for diverse species of aquatic (like the famous blue crab) and terrestrial life while also providing drinking water, food, and recreation for more than 18.2 million people (22.5 million people by 2050). Within the Bay's watershed, there are thriving centers of agriculture, industry, business, government, tourism, culture, and heritage—and vast areas of remarkable beauty. These attributes and abundant resources have drawn people to the Bay and its watershed for centuries, but ever-increasing pressures from society have taken a toll on the land and water. Changes in climate, hydrology, land use, air quality, farming intensity, population, resource extraction, and developing land have caused enormous quantities of excess nitrogen, as well as phosphorus, sediment, and other contaminants, to reach the Bay and contributed to the degradation of water quality, resulting in fundamental changes in the Bay's ecosystem. The Chesapeake Bay is not the only estuary combating nutrient enrichment. There are numerous coastal areas around the Nation and the world that have been impacted by excess nitrogen and other contaminants. The watersheds of these coastal areas have been undergoing and will continue to undergo substantial population growth, urban expansion, crop and animal agriculture intensification, and land use change.

The Chesapeake Bay partnership took on a commitment to actively control the excess nutrients and sediment exported from the watershed to achieve and maintain the Bay's water quality and ecosystem health. Meeting this challenge has already resulted in positive change in areas, but the work to reach sustainability will continue through 2050 and beyond. Can the Chesapeake Bay, and similar estuary communities around the world, find ways to flourish and live sustainably, while at the same time managing nutrients that maintain healthy terrestrial and aquatic ecosystems? This will be a challenge for the future. Meeting the simultaneous goals of sustainability for the Bay's human community and terrestrial and aquatic ecosystems will require the continued cooperative effort among citizens, producers, consumers, scientists, governments, and policy makers to develop and implement sound strategies that wisely use the resources of the Bay. The science, management, and regulation efforts to save the Chesapeake Bay are unprecedented and if the Nation's largest estuary can rebound, they will serve as a model for the world.

[1]U.S. Geological Survey.

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Glossary

Anoxia Waters that contain less than 0.2 milligrams per liter (mg/l) of dissolved oxygen

Base flow Component of streamflow that can be attributed to groundwater discharge to streams.

Best management practice (BMP) Mitigation methods that aim to prevent or reduce nutrients and other pollutants from entering surface water or groundwater.

Biological fixation The natural process by which nitrogen gas (N_2) from the atmosphere is converted by bacteria into reactive (biological) forms of nitrogen that can then be used by aquatic (for example, algae) and terrestrial plants.

Combined sewer outflows Sewers designed to collect both runoff and wastewaters, but during high flows (storm events) volumes can exceed the capacity of the sewer system or the treatment plant and flow directly to a stream or other receiving water body without treatment.

Complete treatment Wastewater influent flow that has undergone screening, grit removal, primary treatment, secondary treatment, nutrient reduction, disinfection, and dechlorination processes.

Concentration The mass of nitrogen in a particular volume of water (such as milligrams per liter of water as nitrogen).

Denitrification Biologically mediated process by which reactive (bioavailable) nitrogen is converted to nitrogen gas (N_2).

Dry deposition of nitrogen The transfer of reactive nitrogen-bearing gases and particles from the atmosphere to Earth's surface. Nitrogen present in cloud and fog droplets, though technically not dry by definition, is often included in measurements of dry deposition.

Eutrophication Increased plant and algae production in a waterbody commonly caused by nutrient enrichment, particularly phosphorus and nitrogen. As the algae die and decompose, oxygen is depleted from the water, causing impairments to aquatic life.

Evapotranspiration The return of moisture to the atmosphere from the soil, water surface, or plants.

Floodplains Land area adjacent to stream inundated on average every 1 or 2 years.

Hypoxia Waters that contain less than 2 parts per million of dissolved oxygen.

Impervious surfaces Surfaces that allow limited or no infiltration (parking lots, roofs, and so forth) of precipitation into the soil and groundwater.

Load The mass of nitrogen moving past a location over a period of time (such as kilograms per year as nitrogen).

Lag time Delayed time interval for which implementation of management practices produce subsequent response in the nutrient concentrations in groundwater, streams, and estuaries.

Organic nitrogen Any organic molecule that includes nitrogen in its structure.

Overland flow Water that flows across the land surface and discharges into a stream channel.

Oxidized nitrogen Inorganic forms of reactive nitrogen that include oxygen. In these forms, nitrogen is in an oxidation state that ranges from +1 to +5. The most common forms of oxidized nitrogen in the atmosphere are NO_x (NO and NO_2), nitrate (NO_3^-), and nitric acid (HNO_3).

Paleoclimate The climate of the distant past prior to the widespread availability of instrumental records, a time scale that extends back hundreds to thousands of years.

Paleoclimatology The study of paleoclimate, often using records such as tree rings and sediment.

Paleoecology The study of ancient ecology, prior to the widespread availability of instrumental records. Paleoecologists use records such as tree rings and sediment, and their work can inform the study of paleoclimatology.

Primary treatment Removal of suspended solids that can be settled out by gravity through sedimentation tanks during wastewater treatment.

Reduced nitrogen Inorganic forms of reactive nitrogen that include hydrogen, but not oxygen. In these forms, nitrogen is in an oxidation state that ranges from 0 to –3, though –3 is by far the most prevalent. The most common forms of reduced nitrogen are ammonia (NH_3) and ammonium (NH_4^+).

Riparian zones Water-dependent lands along streams and lakes where transitions occur between terrestrial and aquatic parts of a watershed.

Residence time Measure of how long, on average, a molecule of water spends in a particular hydrological compartment (in other words, groundwater, stream, and so forth).

Secondary treatment Removes the soluble organic waste that escapes primary treatment and provides further removal of the suspended solids during wastewater treatment.

Shallow subsurface stormflow Water that has infiltrated the ground and creates localized areas of saturation, and with sufficient permeability and hydraulic gradients, induces lateral water movement.

Stormwater management practices Structural controls to slow down and treat stormwater runoff to prevent excess nitrogen, phosphorus, and sediment entering streams.

Streambank erosion Forces exerted by flowing water exceed the resisting forces of bank materials and vegetation and often results in the release of sediment into streams.

Surface runoff Precipitation that runs off the land surface directly to streams or rivers.

Wet deposition of nitrogen The transfer of reactive nitrogen-bearing species from the atmosphere to Earth's surface as precipitation in the forms of rain, snow, and ice.

Wet weather treatment Wastewater influent flow rate greater than specified discharge that receives treatment comprising primary sedimentation, followed by disinfection and dechlorination

Yield The load of nitrogen normalized to an area (such as kilograms per hectare per year as nitrogen).

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