Technical Report for the Delaware Estuary and Basin

2022

Partnership for the DELAWARE ESTUARY

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The Partnership for the Delaware Estuary, host of the Delaware Estuary Program, leads collaborative, science–based efforts to improve the Delaware River and Bay, which covers portions of Delaware, New Jersey, and Pennsylvania.

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Graphics

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Abstract

The Technical Report for the Delaware Estuary and Basin (TREB) analyzes the most accessible, comprehensive and recent data on the status and trends of more than 50 environmental indicators, including a diverse suite of indicators that are relevant for water, habitat, and living resources. Taken together, the condition of these indicators reflects the overall environmental health of the Delaware River and Bay, and the watershed that drains into it. This report is produced every five years by the Partnership for the Delaware Estuary, which coordinates the Delaware Estuary Program, a part of the National Estuary Program. The TREB serves as the technical foundation for "State of the Estuary" reports for the public. There are eight key indicator categories: watersheds and landscapes, climate change, water quantity, water quality, sediments, habitats, living resources, and restoration progress. Scientists and managers examined historic, recent, and predicted future changes in each indicator's status to develop an understanding of trends. Finally, this report describes future actions and needs that can strengthen indicator reporting and potentially improve environmental conditions. The results from this assessment suggest that the current health of the Delaware Estuary and River Basin in 2022 is "fair," reflecting a mix of positive and negative trends. The overall assessment of "fair" health is unchanged from TREB 2017, TREB 2012, and the smaller State of the Estuary Report published in 2008.

The land and waterways that we discuss here are part of the traditional territory of the Lenni-Lenape, called

Lenapehoking

The Lenape People lived in harmony with one another upon this territory for thousands of years. During the colonial era and early federal period, many were removed west and north, but some also remain among the continuing historical tribal communities of the region: the Nanticoke Lenni-Lenape Tribal Nation; the Ramapough Lenape Nation; and the Powhatan Renape Nation, the Nanticoke of Millsboro Delaware, and the Lenape of Cheswold Delaware. We acknowledge the Lenni-Lenape as the original people of this land and their continuing relationship with their territory. In our acknowledgment of the continued presence of Lenape people in their homeland, we affirm the aspiration of the great Lenape Chief Tamanend, that there be harmony between the indigenous people of this land and the descendants of the immigrants to this land,

"as long as the rivers and creeks flow, and the sun, moon, and stars shine."

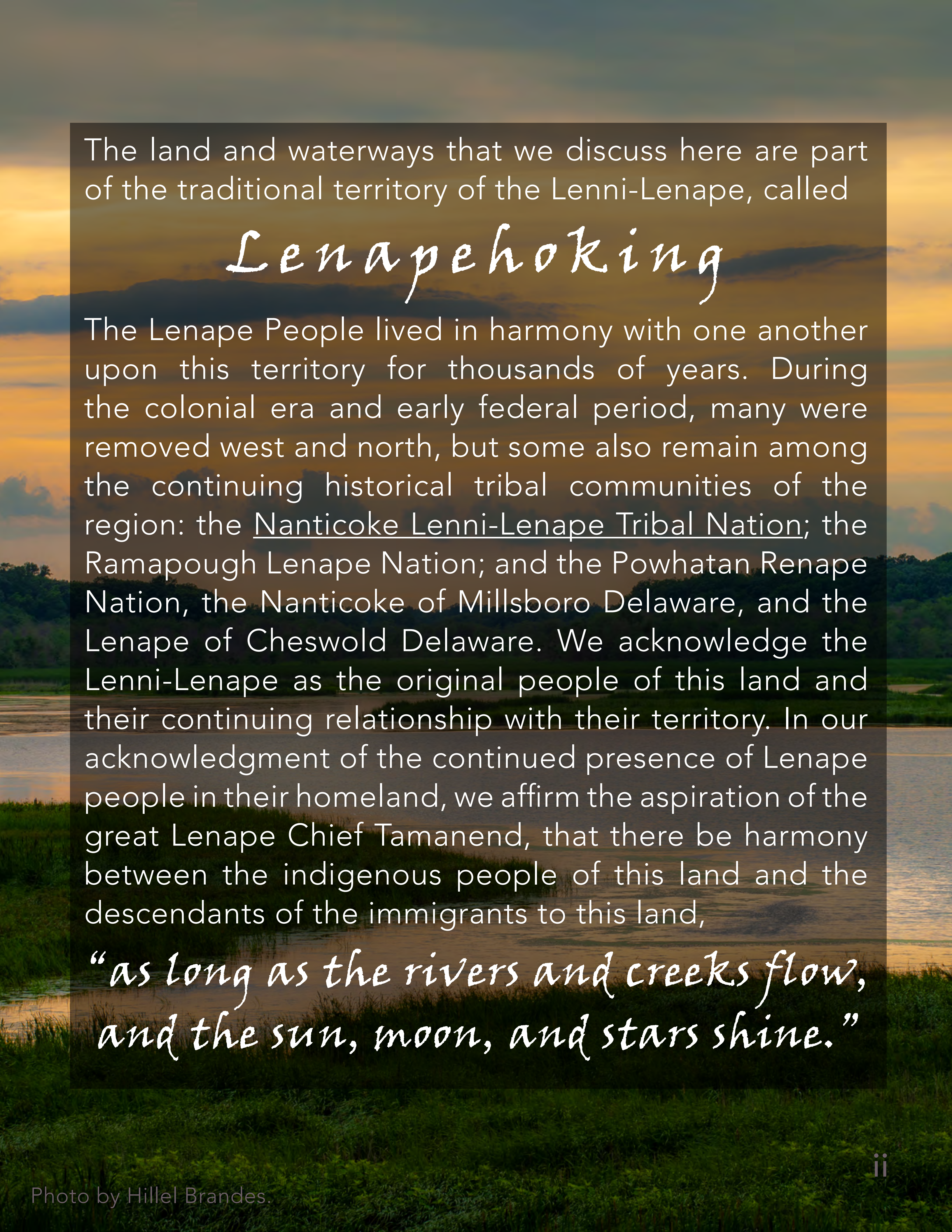

Photo by Hillel Brandes.

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Executive Summary

The purpose of the 2022 Technical Report for the Delaware Estuary and Basin (TREB) is to assess the overall environmental condition of the watershed by examining the status and trends of key indicators that reflect the health of its natural systems. Meeting this goal is challenging because the Delaware River Basin is a large and complex watershed, encompassing more than 35,000 square kilometers (>13,500 square miles) and extending from headwater streams and mountains in New York State to the coastal plain and ocean near Cape May, NJ, and Cape Henlopen, DE. The Delaware Estuary forms the lower 52% of the overall basin. The watershed spans four ecoregions, is home to 8.6 million people, and supplies drinking water to another 5 million in New York City and northern New Jersey living outside the basin. Hundreds of plant and animal species live in balance with people in diverse habitats, including many ecological treasures. The region also has a storied history, starting with rich Native American peoples and extending through the birth of the United States and the Industrial Revolution, up to the present day where it continues to function as a nationally important economic center and strategic port.

Environmental indicators are aspects of the environment which can be quantified and are representative of prevailing local conditions. The approach used in this report was to gather, analyze and interpret the most extensive and recent data for a broad suite of more than 50 indicators that represent different facets of the natural ecosystem, such as water quality, living resources, habitats, land cover, and climate. The last section of the report includes indicators that reflect our progress in preserving and restoring natural systems. When considered together, this indicator-based report provides a comprehensive picture of the status and trends in environmental health of the Delaware Estuary and Basin.

The eight chapters of TREB are organized topically into the following sections: watershed and landscapes, climate change, water quantity, water quality, sediments, habitats, living resources, and restoration. Each section includes a number of different indicators, written by a different set of authors with expertise relevant to the topic. For each indicator, authors present and interpret the most recent available data and summarize any actions and needs that could strengthen future indicator reporting or lead to improved environmental conditions.

On balance, the results from this assessment suggest that the current health of the Delaware Estuary and Basin in 2022 is "fair," reflecting a mix of positive and negative trends. The status of many indicators is good, and others are not so good. Trends for some indicators appear to be improving, while others appear to be worsening. The overall assessment of "fair" health is unchanged from TREB 2017, TREB 2012 and the smaller State of the Estuary Report in 2008.

Although the "fair" overall health assessment is unchanged since 2008, it reflects substantial improvement compared to earlier decades for many key indicators. For example, advances in wastewater treatment and implementation of the Clean Water Act led to dramatic improvement in dissolved oxygen in the river's urban corridor over the past 40+ years. These improvements in many facets of water quality have supported healthier living resources, demonstrated by the propagation of signature species such as sturgeon and increasing public interest in river recreation. Unfortunately, the continued loss and degradation of important habitats and emerging threats associated with climate change could or may undermine the recent recovery. Meanwhile, the human population in the watershed continues to increase, resulting in expanding human activities are likely to increasingly tax our natural resources and require management diligence, especially with regard to water withdrawals, forest cutting, wetland loss, and development. These challenges will be exacerbated by a shifting climate, especially increasing temperature, precipitation, sea level, and salinity. Of particular note, future predictions for many of the key climate indicators in the 2022 TREB reflect a much higher level of certainty compared to the 2017 TREB, largely because of more robust datasets and stronger recent trends.

Where possible, the future status and trends of indicators are also discussed in the context of the expected increase in human activities and climate change. As one example, warming water (from climate change) holds less dissolved oxygen, which is vital for aquatic animals such as fish. Oxygen deficits can also be exacerbated by excess nutrients from runoff, which in turn fuel microbial respiration. With increased water temperature and potentially greater nutrient runoff from more people, it is plausible to expect the trajectory of past improvements in dissolved oxygen conditions to reverse course. Therefore, even more effort to manage dissolved oxygen will be needed as compared to the past. This report includes many other similar examples of past successes and emerging threats.

The cumulative impacts to natural resources from both anthropogenic alterations and shifting climate conditions are difficult to predict. Hence, continued careful monitoring of the indicators reported in this report will be critical so that environmental managers can make adaptive decisions to maintain crucial life-sustaining ecosystem services, which are worth billions of dollars per year. Specifically, to address future environmental challenges while preserving prosperity in the region, agencies, scientists, and others must work together to:

- Sustain and strengthen the effectiveness of monitoring, protection and restoration efforts by focusing on a set of shared, strategic priorities
- Set science-based goals that plan for change as part of the natural landscape
- Adopt realistic environmental targets that focus on preserving and enhancing key life-sustaining features
- Apply an ecosystem-based approach to management that considers cumulative impacts and ecological linkages
- Facilitate collaboration among states, federal agencies and other sectors to implement the Delaware Esturary Comprehensive Conservation and Management Plan (CCMP), which was updated in 2019. The CCMP is a guiding document developed by partners involved with the Delaware Estuary Program, which is the congressionally designated National Estuary Program for the Delaware River and Bay.

The information, perspectives, and future needs stated in this report reflect the best current scientific consensus of the authors that drafted individual sections and do not necessarily represent the official views of the Partnership for the Delaware Estuary, other members of the Delaware Estuary Program, or any other participating entity or specific author. This report is a collective, peer reviewed effort which attempted to coordinate a consistent style and content among sections; however, the written presentations and depth of analysis will reflect (or vary in accordance with) the availability of data, methods of presentation, and analytical rigor that are appropriate for different fields and different writing styles of various authors. Examples of key findings in this report are summarized in the table below which shows both improving and declining environmental conditions (Table 0.1). The list is not prioritized, and many similar examples can be found in various report sections. Scattered throughout the TREB are additional features that showcase recent case studies or hot topics.

Table 0.1 Top positive (A) and negative (B) findings from the 2022 Technical Report for the Delaware Estuary and Basin. Impact scores are qualitative and based on relative overall impact to estuary and basin wide health, and immediacy of action need. Impact scores of 1 for positives are very good, whereas a score of 6 for a negative is detrimental. Averaging all impact scores yields a total score of 3.5, or an overall "fair" rating for the reporting period's Estuary and Basin health.

A.

Chapter	Positives		
	Indicator	Condition	Impact
Watersheds	Protected lands	Estuary and Basin has >2,900 sq mi of protected lands, with a increase by 1.3% in the last decade	2
Water Quantity	Water Withdrawals	Peak water withdrawals occurred in 2006-2007 and have subsequently declined	2
Water Quality	Dissolved Oxygen	Concentrations increased dramatically 1960s to present	1
Sediments	Total Suspended Sediment	Declined from 2005-2010 to 2017-2021, especially in the Lower Estuary (but this could also have negative effects to tidal wetlands in the Bayshore)	3
Habitats	Fish Passage	Between 2017-2021, 29 dams have been removed in the Delaware River Basin	1
Living Resources	Population Increases	Osprey, blue crab, American eel, and sturgeon populations have increased	1
Climate	Temperature	Not yet a significant increase in hot temperature extremes, despite average warming trends	3
Restoration	Habitat Type	Increase in restored acres in 2017-2022, compared to 2006-2011 and 2012-2016	2

B.

Chapter	Negatives		
	Indicator	Condition	Impact
Watersheds	Land Cover	Development increased by ~17.5 acres per day from 1996-2016	6
Water Quantity	-	No negatives observed	-
Water Quality	Temperature	Water temperatures are possibly increasing, but more monitoring and analysis will be required	4
	Contaminants	Many fish consumption advisories remain; ecotoxins in pharmaceuticals and personal care products remain a concern	6
Sediments	Contaminants	Sediment contaminant concentrations highest in areas of the Estuary near Environmental Justice communities	6
Habitats	Tidal Wetlands	From 1996-2016, 340 hectares of tidal wetland were lost; percentage losses were >15% for tidal freshwater wetlands	5
Living Resources	Population Decreases	Striped Bass, Weakfish, White Perch, and freshwater mussel populations show signs of decline	5
Climate Change	Sea Level Rise	Sea levels rose between ~4-6 cm per decade from 1992-2021 in the Estuary	5
Restoration	Regulatory Climate	The time and complexity of permits required to do restoration may be increasing	4

Introduction

The 2022 Technical Report for the Delaware Estuary and Basin (TREB) reviews the status and trends in extent or health of more than 50 environmental indicators as a way to take a holistic and scientific look at the current health of the Delaware Estuary and Basin. Environmental indicators are specific, measurable markers that are used to assess the condition of the environment and indicate whether conditions are improving or worsening over time.[1]

Additionally, indicators help raise awareness about important environmental issues, serve as tools for evaluating the effectiveness of management actions, and can function as early warning signals for detecting adverse changes in environmental quality[1]. Indicators were chosen based on data availability and an indicator's ability to tell something important about the status of the natural resources, water quality, and climate conditions of the Delaware Estuary and its watershed. This report provides the best possible current synthesis of status and trends for the important environmental indicators that could be examined.

This assessment report was led by the Delaware Estuary Program's Science and Technical Advisory Committee (STAC) in collaboration with many other contributing scientists and managers. Indicators were selected by the STAC and approved by core members of the Delaware Estuary Program: Delaware River Basin Commission, Delaware Department of Natural Resources and Environmental Control, New Jersey Department of Environmental Protection, Pennsylvania Department of Environmental Protection, Philadelphia Water Department, and Partnership for the Delaware Estuary. Other authors, contributors, and reviewers represented dozens of academic, non-profit, and business organizations.

The purpose of this report is to compile a scientific synthesis of the most recent status and trends data into a technical report, which can serve as the basis for translation products such as State of the Estuary Reports (PDE) and State of the Basin Reports (DRBC) that are written for the public. The 50+ indicators were chosen to reflect a broad suite of physical, chemical, biological, and watershed features of the ecosystem that have sufficient spatial and temporal datasets with consistent methodologies over time. The 2022 TREB includes a new "trial" indicator, but some 2012 or 2017 indicators were omitted due to insufficient recent data or other issues. As before, most indicators reflect monitoring data for "things" such as numbers of fish or concentrations of pollutants. To enrich future indicator reporting, we anticipate that indicators will be developed that reflect ecological processes, such as rates of primary production, carbon sequestration or biofiltration. Although some analyses were not able to be completed for some important resource conditions in this TREB, the balance of indicator data covered in this report reflects the best possible regional perspective on overall environmental conditions and trends in the Delaware Estuary and Basin.

TREB results are also vital for measuring the progress made toward implementing the Comprehensive Conservation and Management Plan (CCMP) for the Delaware Estuary. By tracking indicators and assessing their status and trends every 5 years, periodic revisions and updates to CCMP strategies can be responsive to changing conditions. To assist with such CCMP updates and guide environmental managers and scientists, this report includes future "Actions and Needs" for each indicator. In many cases, these actions and needs call for improved coordination and/or monitoring. Where data are currently incomplete or unavailable, PDE will continue to work with partners to sustain and improve monitoring to address data gaps and facilitate data sharing and management.

1. U.S. EPA. 2015. Indicator Development for Estuaries. Available at: https://www.epa.gov/sites/default/files/2015-09/documents/indicators_manual.pdf

Organization of the TREB

The sample framework for TREB is the entire Delaware River Basin, although the focus for some indicators is particular sub-watershed areas such as the Delaware Estuary study area which forms the lower half of the Basin. Indicators are grouped into eight chapters, beginning with watershed traits and land use in Chapter 1. The watershed regions considered in this report extend from headwater streams in New York to the mouth of Delaware Bay between Cape May, NJ and Cape Henlopen, DE. The nomenclature and coverage of various watershed regions and sub-regions in TREB are described in the series of maps in Figure 2. Note that these maps highlight the land areas of the watershed (~12,900 square miles), but when the open waters of Delaware Bay are included, the total watershed area is 13,539 square miles.

Climate change is the subject of Chapter 2, which was moved forward in the order of TREB 2022 chapters because of the pivotal importance of changing conditions that serve as drivers for everything that follows. Water resource indicators are discussed next in Chapters 3 and 4, followed by sediment indicators in Chapter 5. Habitat and living resource indicators are examined in Chapters 6 and 7, respectively. Indicators reported in Chapters 1-7 focus on status and trends in environmental conditions; in Chapter 8, we focus on measures of progress for improving conditions through protection and restoration efforts.

Using the TREB

No single indicator or chapter is diagnostic for overall environmental conditions in the Delaware Estuary and Basin. Thus, the information in this report should be interpreted carefully. Changes in some indicators may also not necessarily reflect declining or improving conditions per se, but instead reflect natural variability. For example, it is possible that some species or conditions are actually improving at the expense of others, due to complex ecological inter-relationships. Since monitoring programs typically count or measure "things" rather than "processes," the indicators primarily reflect structural elements of the system rather than ecological functions or ecosystem services. In some cases, this reporting effort was hampered because certain components of the ecosystem that could serve as strong indicators were not able to be included due to insufficient data. The development of this report therefore allows us to assess not only the state of the environment, but also the state of our knowledge and understanding. Furthermore, the restoration chapter is a recent attempt to begin using available data to assess our management progress in preserving, enhancing and restoring environmental conditions, in addition to assessing intrinsic environmental conditions (which is the focus of most of the rest of this report).

For information on the status and trends of any specific indicator (e.g., American Eels), refer to the appropriate section. To obtain an overall status summary for the Delaware Estuary and Basin, one can refer to the Executive Summary although we recommend reviewing the entire report for several reasons. Many indicators interact through complex physical, chemical, and biological relationships, and a complete review facilitates a fuller understanding of the status of functional interrelationships (how the system is working) in addition to any single parameter (what is present). For example, the population abundance of some fish species may depend on others through predation or competition relationships (striped bass versus weakfish, both are never abundant at the same time). Suspended sediment in the water can both represent a pollutant (e.g., in non-tidal tributaries) as well as an essential limiting resource (e.g., for tidal wetlands).

The Delaware Estuary and Basin also has many unique features, such as having one of the world's largest tidal freshwater ecosystems. This is why salinity rise (with climate change and other factors) is more of a threat in this watershed compared to many other estuaries. The natural high turbidity in part of the Estuary is thought to help to stem eutrophication problems by light shading of phytoplankton blooms, despite high nutrient loadings. This runs counter to the stereotypical understanding of estuarine eutrophication processes and is an example of why ecosystem models need to be tailor-designed for the Delaware Estuary. By cross-comparing results among chapters and reading the collective authors' narratives, one can obtain a better understanding of such unique features and complex interactions.

It is difficult to assign a single grade ("good", "fair" or "poor") to the overall Delaware Estuary and Basin given the different trends observed. Taken together, however, an integrated analysis of all chapters provides the best possible basis for judging the holistic environmental condition of the Delaware Estuary and Basin.

Regional Divisions of the Delaware Estuary and Basin

To simplify status and trend analyses, the Delaware Estuary and Basin are divided into four different "watersheds" or "regions". Additional geospatial resolution (e.g. sub-watersheds) varies among indicators, depending on the coarseness of datasets and scientific intent. Geospatial resolution of sub-regions therefore varies from coarse (e.g., nontidal versus tidal; see page 6) to moderate (e.g., ten sub-watersheds; see pages 7 and 8) to fine (e.g., twenty-one sub-regions similar to HUC12s; see page 9).

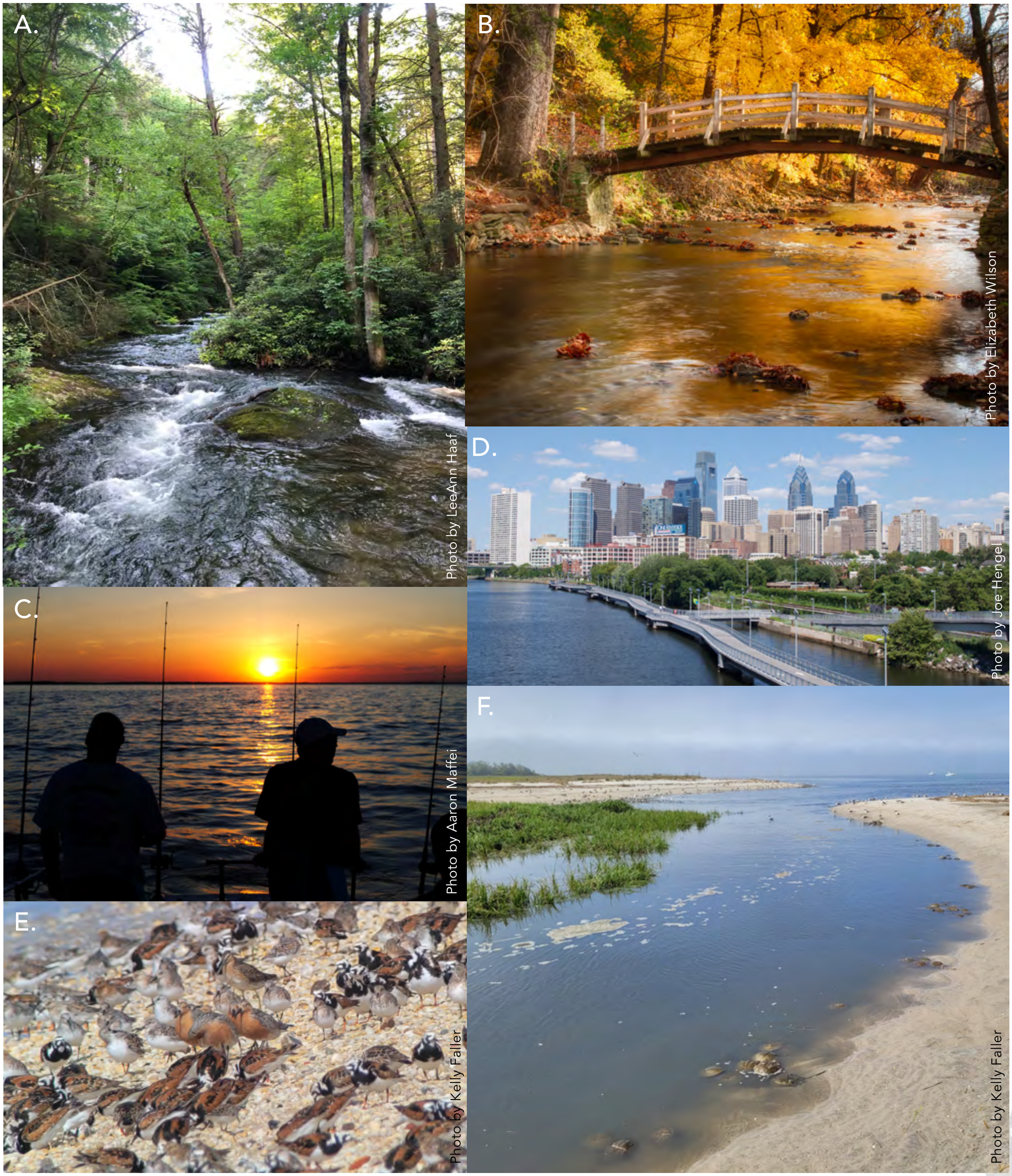


Figure 1 Examples of the sub-regions (see map on page 7) of the Delaware Estuary and Basin: Wild Creek, Albrightsville, PA (A, Central); Valley Creek, Valley Forge, PA (B, Lower); a view of Philadelphia, PA (D, Lower); Fortescue, NJ (C, Bay); shorebirds (E) and horseshoe crabs (F) in NJ (Bay).

Delaware Estuary and Basin

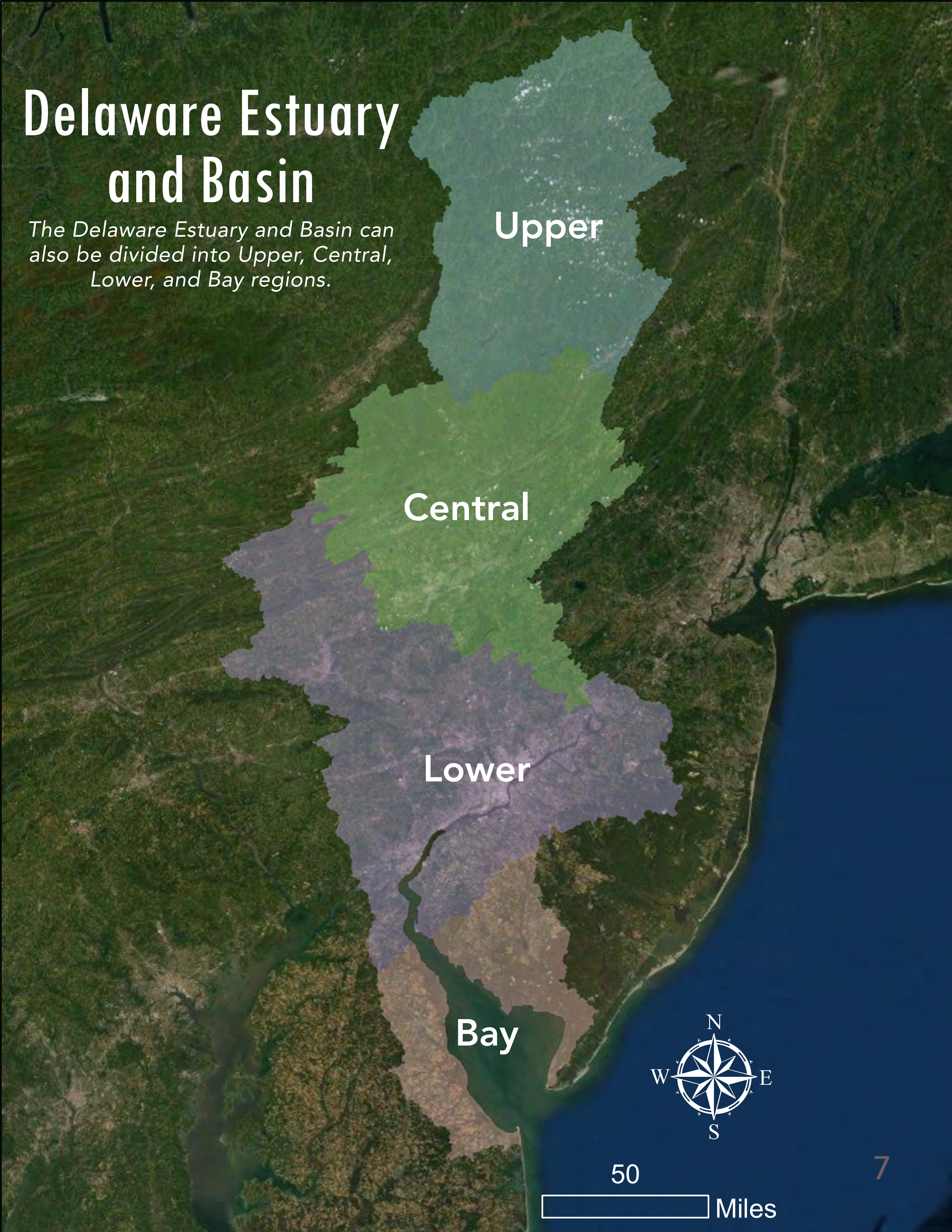
The Delaware Estuary and Basin watershed spans New York, Pennsylvania, New Jersey, and Delaware, with a small portion in Maryland.

Delaware Estuary and Basin

The watershed consists of two macroregions: the upper Basin, which contains headwaters of the Delaware River, and the lower Estuary, which encompasses watersheds of the Schuylkill Valley, as well as the tidal Delaware River and Delaware Bay.

Delaware Estuary and Basin

The Delaware Estuary and Basin can also be divided into Upper, Central, Lower, and Bay regions.

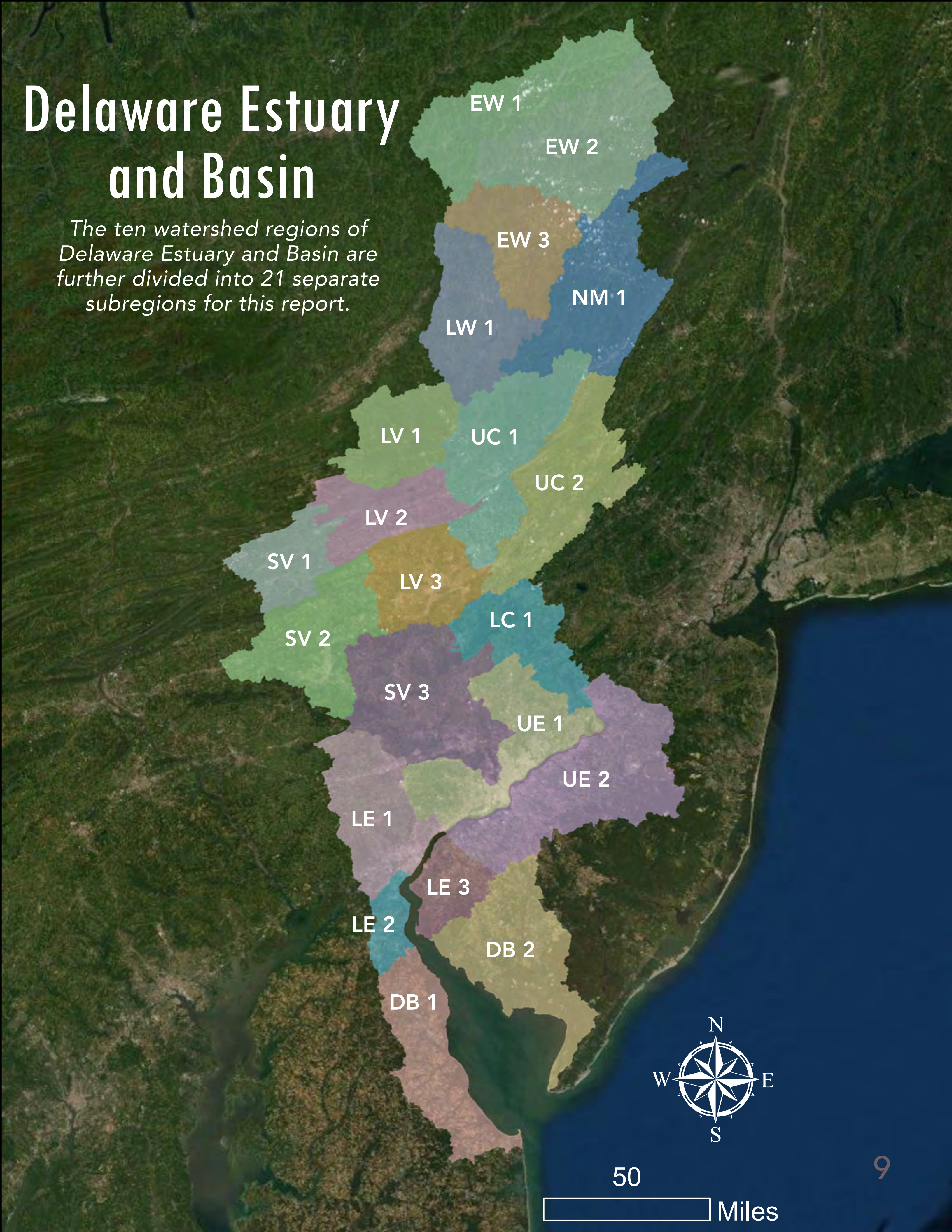


Delaware Estuary and Basin

The Delaware Estuary and Basin watershed is composed of 10 watershed regions.

Delaware Estuary and Basin

The ten watershed regions of Delaware Estuary and Basin are further divided into 21 separate subregions for this report.



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TECHNICAL REPORT FOR THE ESTUARY AND BASIN

Watersheds & Landscapes

December 2022 | Report No.22-05

Partnership for the Delaware Estuary—Host of the Delaware Estuary Program

1 Watersheds & Landscapes

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Cover photograph by Gavin Brown

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1. Watersheds & Landscapes

Abstract

The Delaware Estuary and Basin saw a 4% gain in total population, nearly +865,000, from 2000 to 2020. White populations have decreased, while minority and Hispanic populations have increased. In 2016, land cover in the entire Estuary and Basin was ~60% natural lands, 21% agricultural and 17% developed. From 1996-2016, ~210 mi² was developed— a rate of ~17.5 acres per day. Over that same period, the Estuary and Basin lost ~106 mi² of agricultural and ~99 mi² of natural land, which approximates to 8.9 and 8 acres lost per day, respectively. Imperviousness was greatest in watersheds surrounding Philadelphia; watersheds in the Upper Region had the least imperviousness. In line with increased development, imperviousness has also increased. Despite increased development, the Estuary and Basin has >2,900 square miles of protected land (>22% total land area), which increased from 2010-2020 by 162.9 mi², or 1.3% of the total land area.

1.1 Population

Description of Indicator

Human population can have a direct impact on water quality and habitat within a watershed. In general, the more densely populated an area is, the more stress on the environment. These stresses can impact our natural resources, such as forests, wetlands, and water resources.

The Delaware Estuary and Basin, with a land area of nearly 12,900 square miles, is highly variable in the number of people living within its watersheds. The relatively sparse but increasing population of the tidal portions to the south give way to the densest areas along the I-95 corridor and the Philadelphia environs, up to approximately the limit of tide near Trenton, NJ. Above this point, the population is much less dense, except for some relatively urbanized areas in the Schuylkill and Lehigh Valleys. The upper reaches of the Basin, in upstate New Jersey, Pennsylvania, and New York, are highly forested with quite low population densities.

Data source and processing methodology

In order to derive the population for each decennial census year in the analysis (2000, 2010, and 2020) on a watershed basis, it is necessary to estimate, since population data are not provided by the U.S. Census Bureau on a watershed basis. For each year of the decennial census, tabular data were obtained on a census block and/or block group level (census blocks are the smallest tabulation area produced by the Census Bureau). Census data for 2020 were obtained through an early data release for the purposes of the reapportionment program. It is possible that future data releases will be slightly different as the data are adjusted after a period of quality control.

To enable the arbitrary calculation of population on any spatial unit (e.g., by watershed or by county with each watershed), the population information for each census block was linked to the geographic information system (GIS) data layer for the appropriate year. Since the delineation of census blocks is not static but may change from year to year, a separate GIS data layer for each year's census blocks was obtained and linked to the tabular data.

Information for the population used in this analysis included total population (by census block) for

2000, 2010, and 2020, as well as racial makeup by census block for 2010 and 2020. Two different type breakdowns were made—first the minority population, which consists of any self-reported race that is not "white alone." Individuals who reported themselves as a single race other than white were included, as well as anyone who indicated that they were of more than one race. Secondly, the Hispanic population was determined for 2010 and 2020, based on respondents who self-identified as "Hispanic." Note that minority/non-minority and Hispanic are not exclusive categories. It is possible to self-report as racially white, or any other race (or mixed race), and also Hispanic. For this reason, minority/non-minority and Hispanic populations' status are considered separately.

Population was summarized across two units, states and counties, within the Delaware Estuary and Basin, and population was based on the 21 watersheds within the Basin. GIS data layers for each of these geographic areas were used to summarize population data for each of the three decennial census years. For both layers, the area of water was removed so that all data are based on land area, where habitation is possible.

To help visualize the current population and trends over the past two decades (2000-2020) the tabular data were linked to the GIS layer of 21 watersheds and presented cartographically.

Present Status

Over the past two decades, there has been a fairly large rise in population within the Basin. The highest increases were observed in Delaware (+24%), then Pennsylvania (+11%), and New Jersey (+7%). New York was the only state to see a slight loss of population (-0.4%). Pennsylvania comprises the largest portion of the Basin in area, with over half (50%) of the land area, followed by New Jersey (23%), New York (19%), Delaware (8%), and Maryland (0.1%) (Fig. 1.1.1A, Table 1.1.1).

Table 1.1.1 Current population in the Delaware Estuary and Basin by state.

State	Sq. Miles	% of Basin	2020 Population	% of Population
Delaware	977	7.6%	773,858	9.0%
Maryland	9	0.1%	8,304	0.1%
New Jersey	3,021	23.5%	1,986,397	23.0%
New York	2,393	18.6%	116,980	1.4%
Pennsylvania	6,455	50.2%	5,742,422	66.6%
Total	12,855	100.0%	8,627,962	100.0%

Similarly, based on total population, Pennsylvania contains 67% of the Basin's inhabitants, New Jersey 23%, Delaware 9%, New York 1.4%, and Maryland 0.1% (Fig 1.1.1B). Refer to Table 1.1.3 for a full summary of population from 2000, 2010, and 2020, by state and county, based on U.S. Census Bureau Decennial Census figures.

Between 2000 and 2010, population in the Delaware Estuary and Basin increased by nearly 500,000, representing a 6.3% increase, exceeding 8 million inhabitants for the first time. Between 2010 and 2020 the population expanded by more than an additional 360,000, a 4.5% increase.

As of the 2020 decennial census, there were approximately 8,628,000 people living in the Delaware Estuary and Basin. Nearly a third (2,750,000 or 32%) of the population resided in the Upper Estuary watershed containing the Philadelphia metro area (UE1), with the second most populous watershed being UE2, or the greater Camden, NJ region (1,400,000 or 16%) (Table 1.1.2). Minority and Hispanic populations were summarized by watershed for the census years 2010 and 2020 (Figs 1.1.2-1.1.5). The highest minority

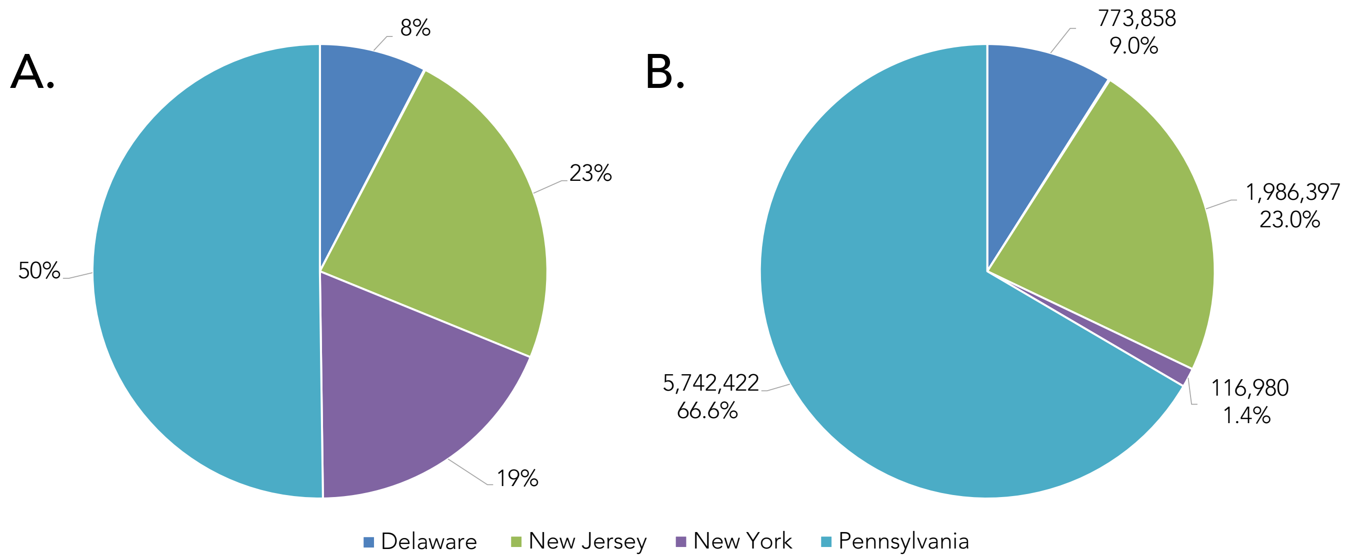


Figure 1.1.1 Percent area within the Basin for each state (A) and the 2020 population sizes by state (B).

populations were also in UE1 and UE 2, with, respectively, 1,278,028 (39.9%) and 549,656 (17.1%) of the Basin's minority population. Similarly, the highest Hispanic population was in those two watersheds, with 303,414 (27.6%) and 210,252 (19.2%), respectively.

Figures 1.1.6 through 1.1.11 shows the distribution and percentages of population within the Delaware Estuary and Basin. The maps on the left show the population density for each demographic category, while the maps on the right show the total populations (and percentages of the Delaware Estuary and Basin total) for each demographic by each 21 watersheds in the Delaware Estuary and Basin. The watersheds in the upper Basin have quite low populations and population densities, with the highest populations in the Lower Central and Upper Estuary watersheds. The Lower Estuary and Delaware Bay watersheds are relatively lower in population but are experiencing growth.

Past Trends

Between 2000 and 2020, the population of the Delaware Estuary and Basin increased by nearly 865,000. Most of the change occurred in the watersheds in the greater Philadelphia region. Based on the decennial census, total population went from 7,763,062 in 2000, to 8,256,005 in 2010 (a 6.3% increase), and 8,627,962 in 2020 (a 4.5% increase).

Figure 1.1.12 shows the trend in population in the Delaware Estuary and Basin, by state, for each decennial census year (2000, 2010, 2020). Tables 1.1.3 and 1.1.4 summarize the change in total basin population between 2000 and 2020 by each county and state. Figure 1.1.13 presents the total change by state between 2000 and 2020. Figure 1.1.14 shows the change as a percentage difference by state.

The map in Figure 1.1.15 shows the change in total population by watershed with total population and percent change indicated. Table 1.1.5 summarizes the changes in total population by watershed in the Delaware Estuary and Basin. Figure 1.1.16 shows the change in minority populations by watershed. Table 1.1.6 summarizes the changes in minority population by watershed. Figure 1.1.17 shows the change in the Hispanic population by watershed. Table 1.1.7 summarizes the changes in minority populations by watershed.

Table 1.1.2 Population of demographic categories by watershed, and the percentage of that category relative to the entire Basin.

Watershed	Total	% Total population	Minority	% Minority population	Hispanic	% Hispanic population
UE 1	2,754,029	31.9%	1,278,028	39.9%	303,414	27.6%
UE 2	1,395,425	16.2%	549,656	17.1%	210,252	19.2%
SV 3	1,075,332	12.5%	327,771	10.2%	62,387	5.7%
LE 1	745,356	8.6%	273,270	8.5%	81,524	7.4%
LV 3	565,233	6.6%	177,400	5.5%	127,685	11.6%
SV 2	372,478	4.3%	115,168	3.6%	98,011	8.9%
UC 2	230,823	2.7%	47,324	1.5%	26,117	2.4%
UC 1	255,010	3.0%	71,443	2.2%	36,504	3.3%
DB 2	248,567	2.9%	99,386	3.1%	60,948	5.6%
DB 1	224,810	2.6%	85,692	2.7%	20,127	1.8%
LC 1	167,844	1.9%	35,808	1.1%	9,766	0.9%
LV 2	97,980	1.1%	18,349	0.6%	14,737	1.3%
SV 1	85,862	1.0%	8,959	0.3%	4,190	0.4%
LE 2	104,047	1.2%	48,769	1.5%	8,305	0.8%
NM 1	84,644	1.0%	24,627	0.8%	14,783	1.3%
LE 3	60,514	0.7%	18,320	0.6%	6,081	0.6%
LW 1	55,847	0.6%	6,262	0.2%	2,874	0.3%
LV 1	48,109	0.6%	13,631	0.4%	7,008	0.6%
EW 1	22,297	0.3%	2,762	0.1%	986	0.1%
EW 3	18,712	0.2%	1,944	0.1%	1,010	0.1%
EW 2	15,241	0.2%	1,870	0.1%	1,083	0.1%
TOTAL	8,628,162	100.0%	3,206,439	100.0%	1,097,793	100.0%

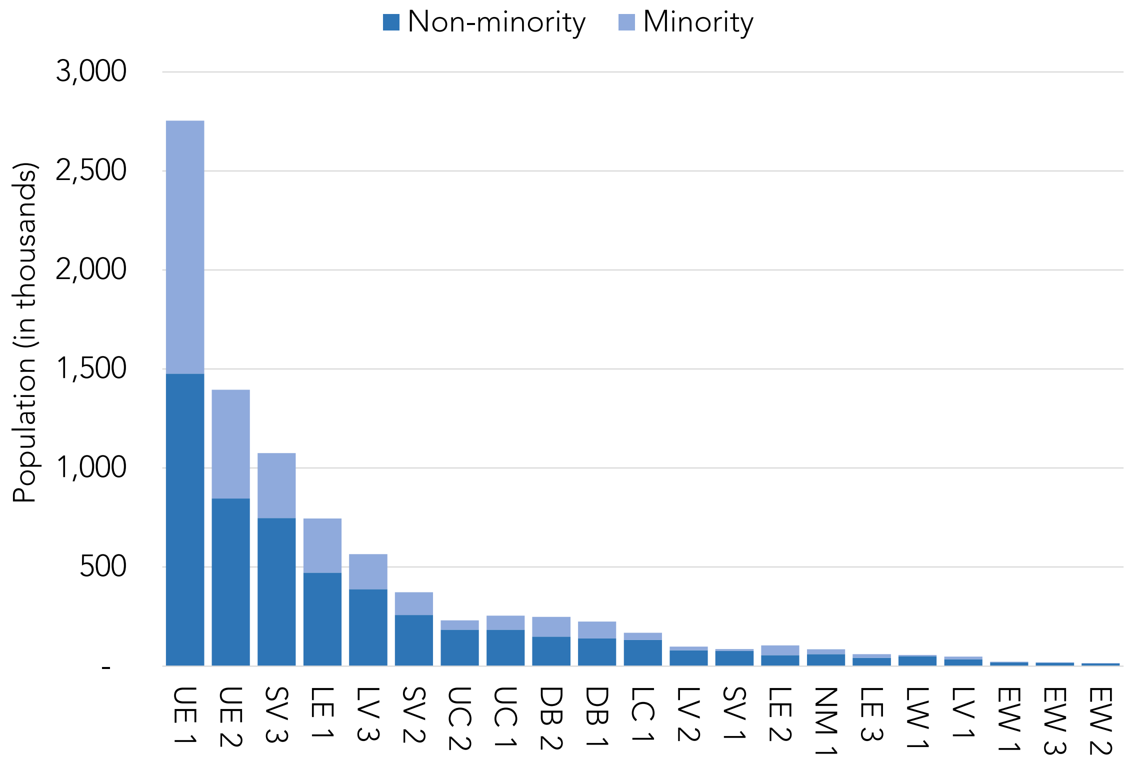


Figure 1.1.2 Minority and non-minority population in the Delaware Estuary and Basin (2020).

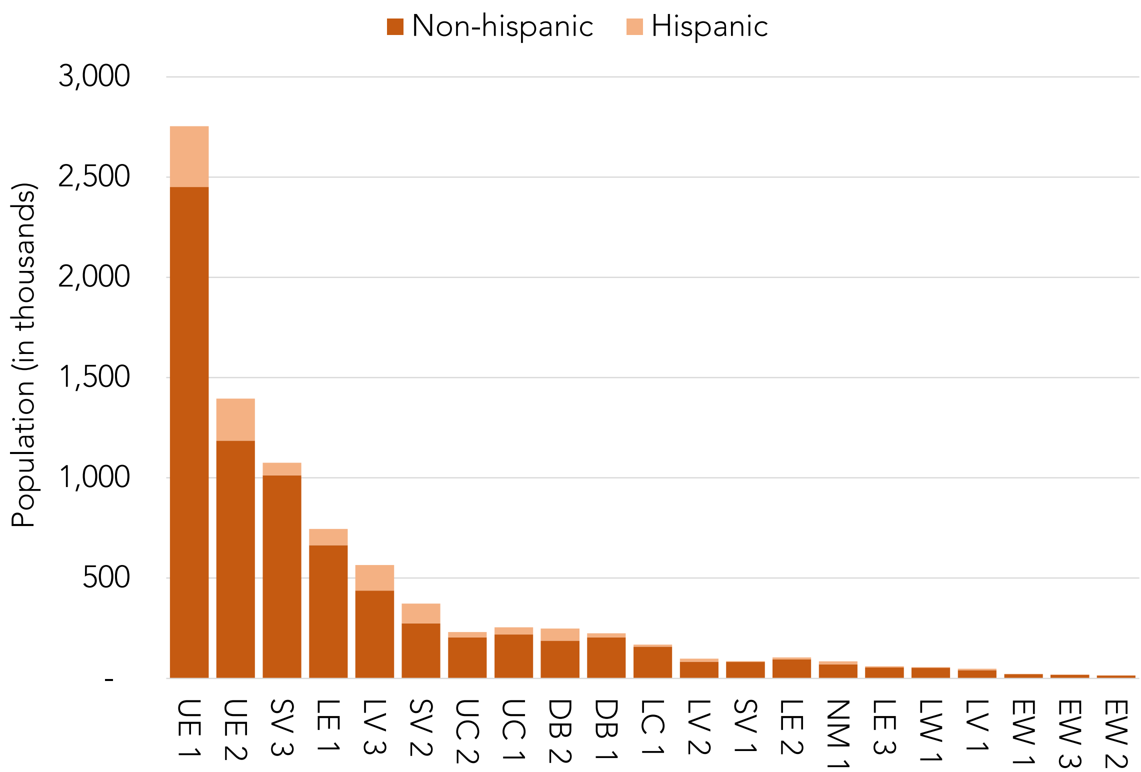


Figure 1.1.3 Hispanic and non-hispanic population in the Delaware Estuary and Basin (2020).

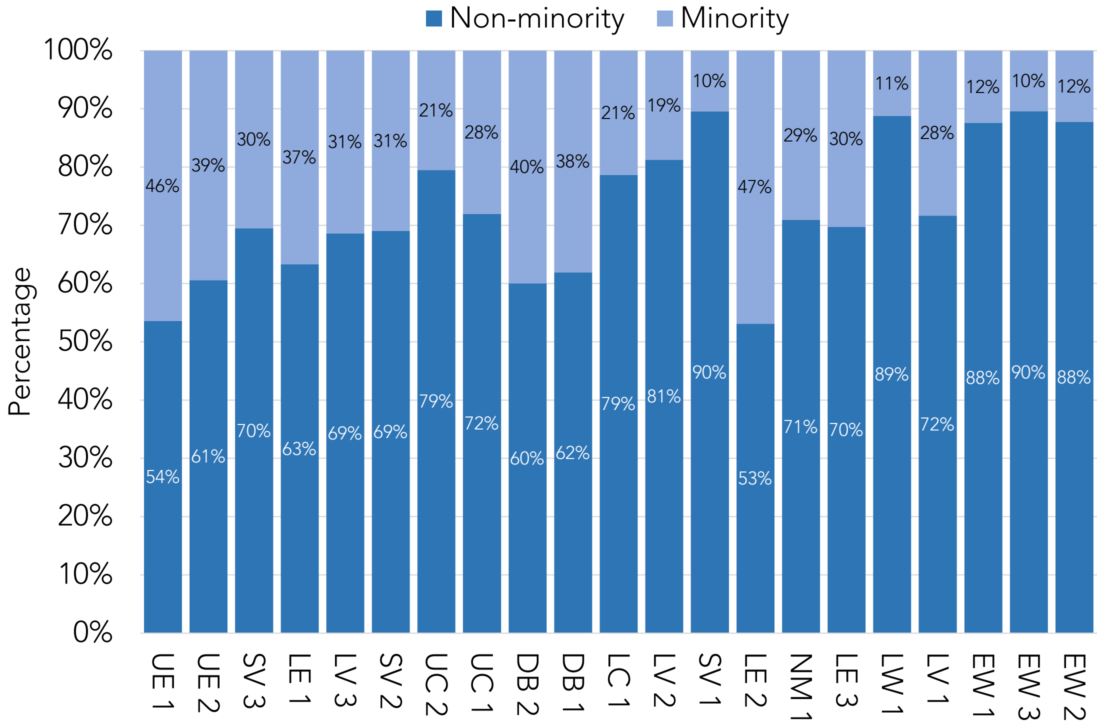


Figure 1.1.4 Minority and non-minority population (%) in the Delaware Estuary and Basin (2020). Watersheds are shown in descending order of total population size, from left to right.

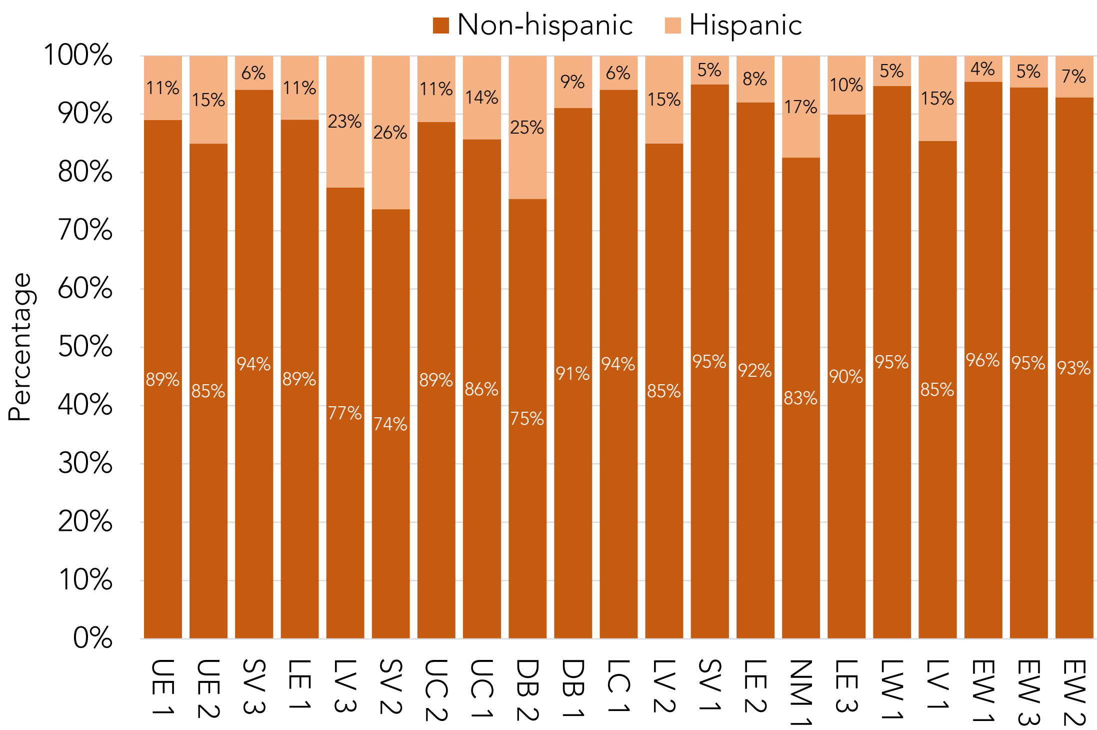


Figure 1.1.5 Hispanic and non-hispanic population (%) in the Delaware River Basin (2020). Watersheds are shown in descending order of total population size, from left to right.

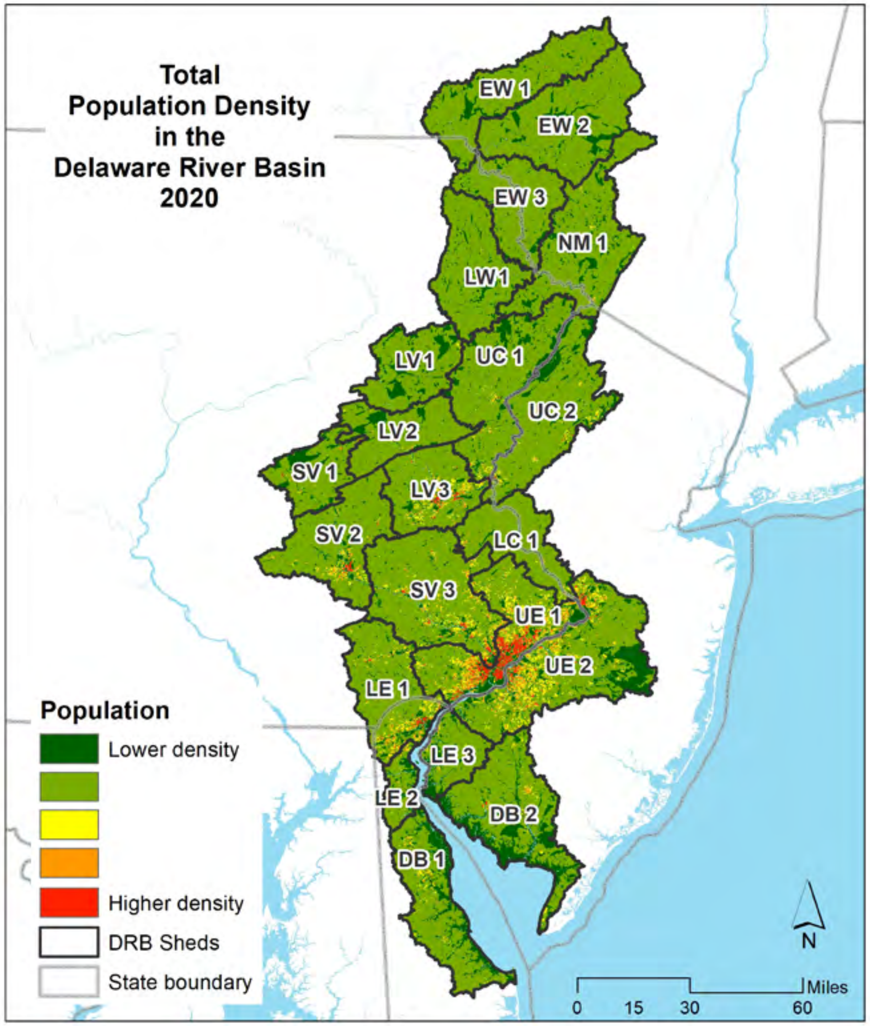


Figure 1.1.6 Population density in the Delaware River Basin (2020).

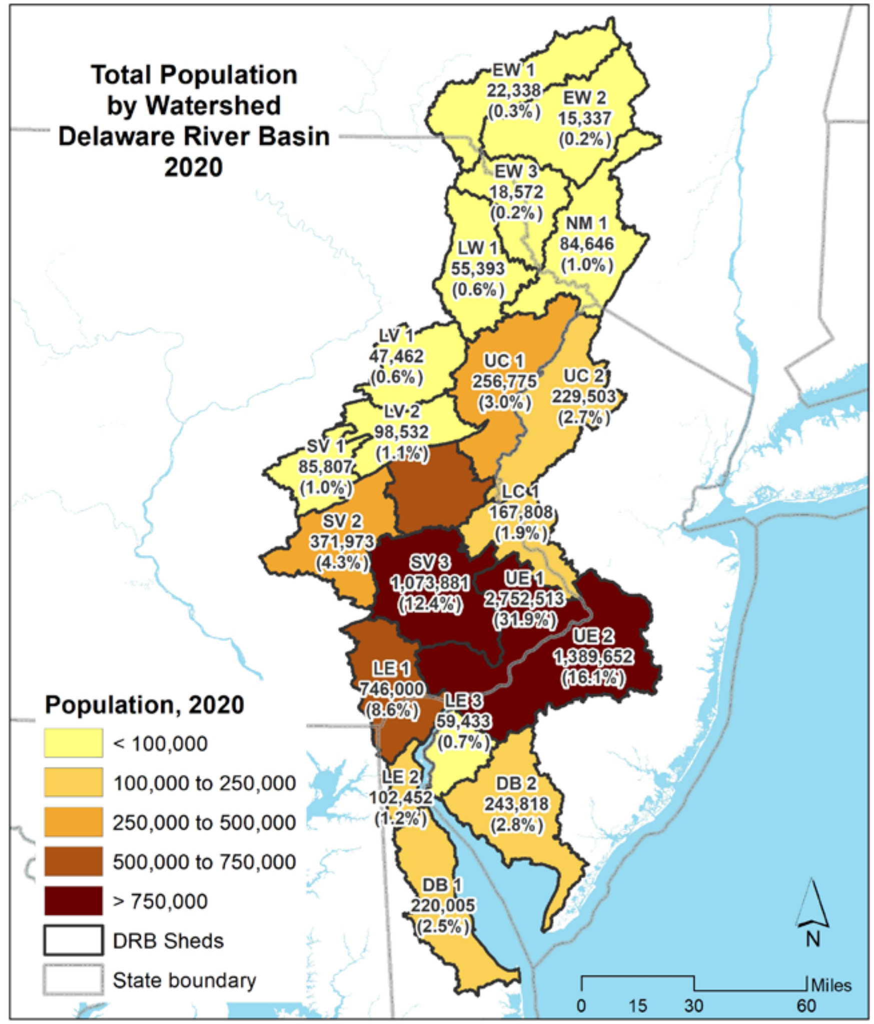


Figure 1.1.7 Total population by watershed in the Delaware River Basin (2020).

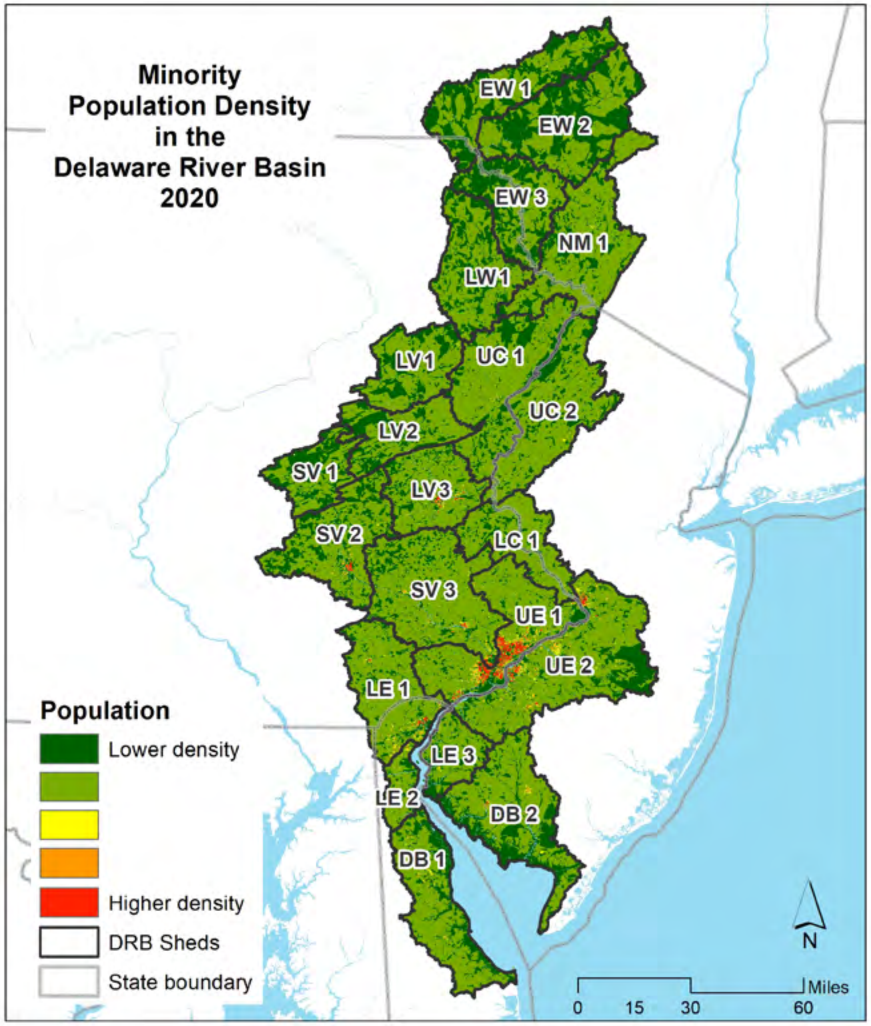


Figure 1.1.8 Minority population density in the Delaware River Basin (2020).

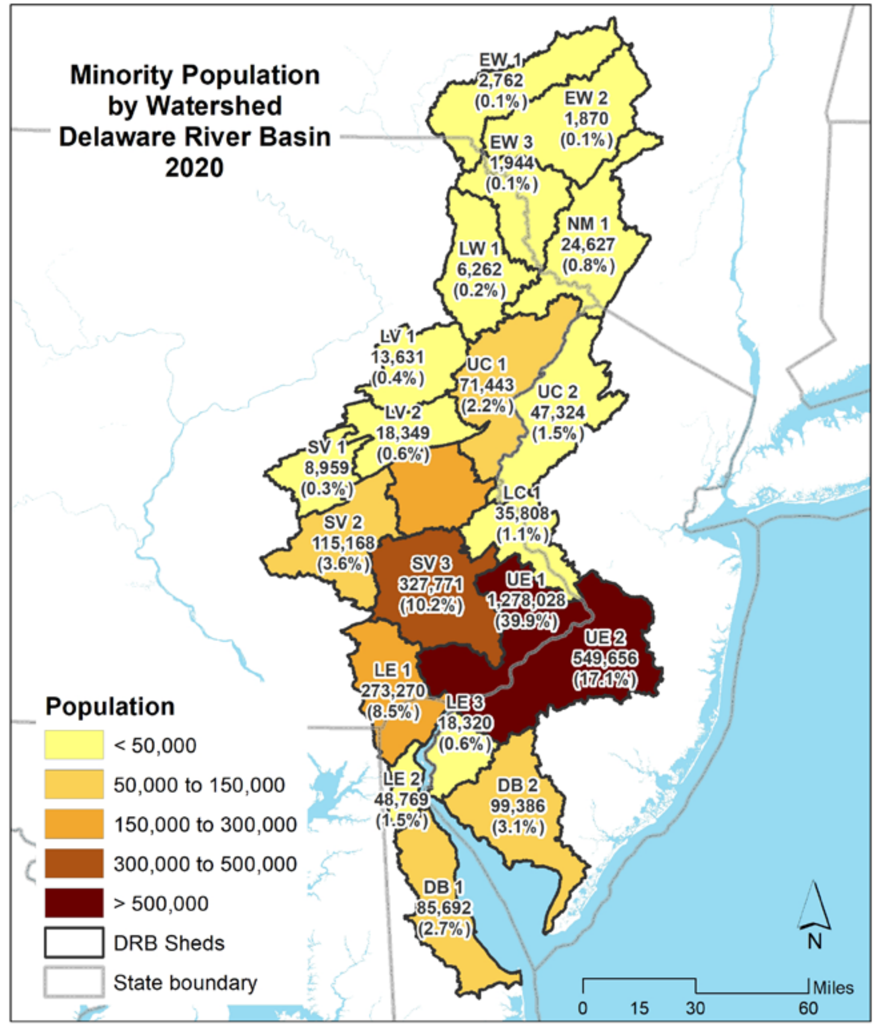


Figure 1.1.9 Minority population by watershed in the Delaware River Basin (2020).

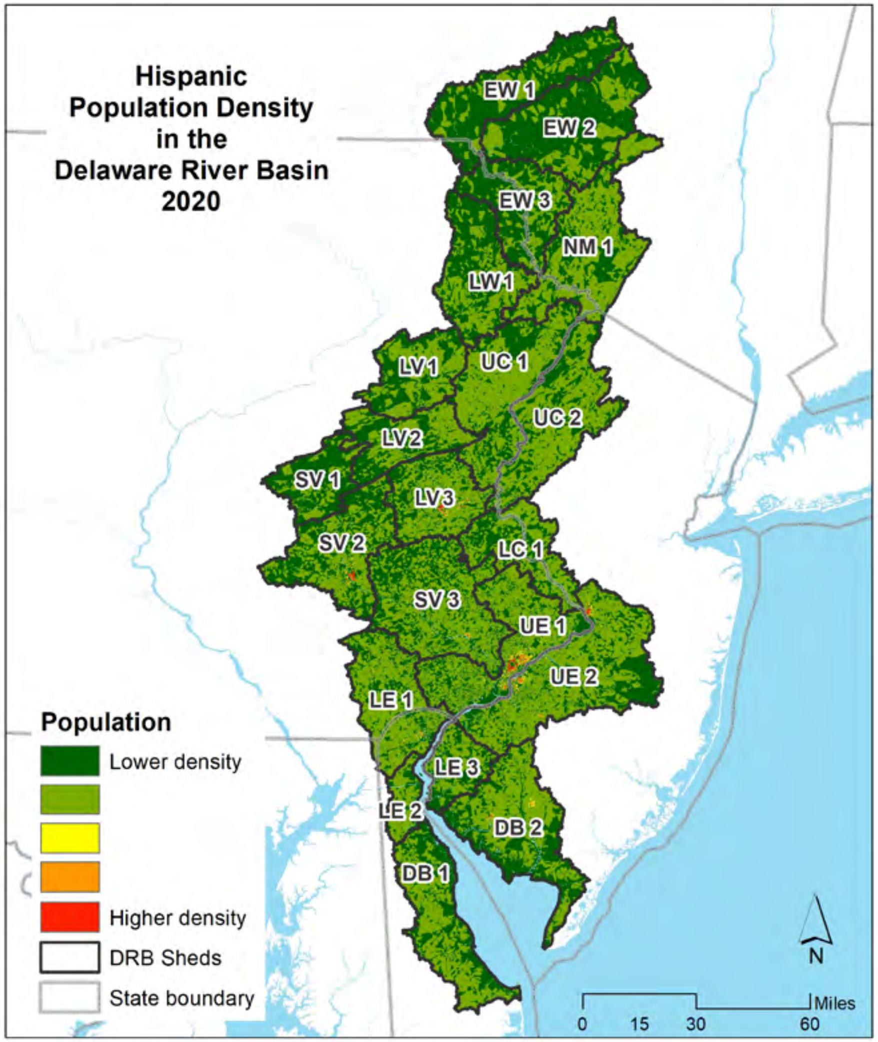


Figure 1.1.10 Hispanic population density in the Delaware River Basin (2020).

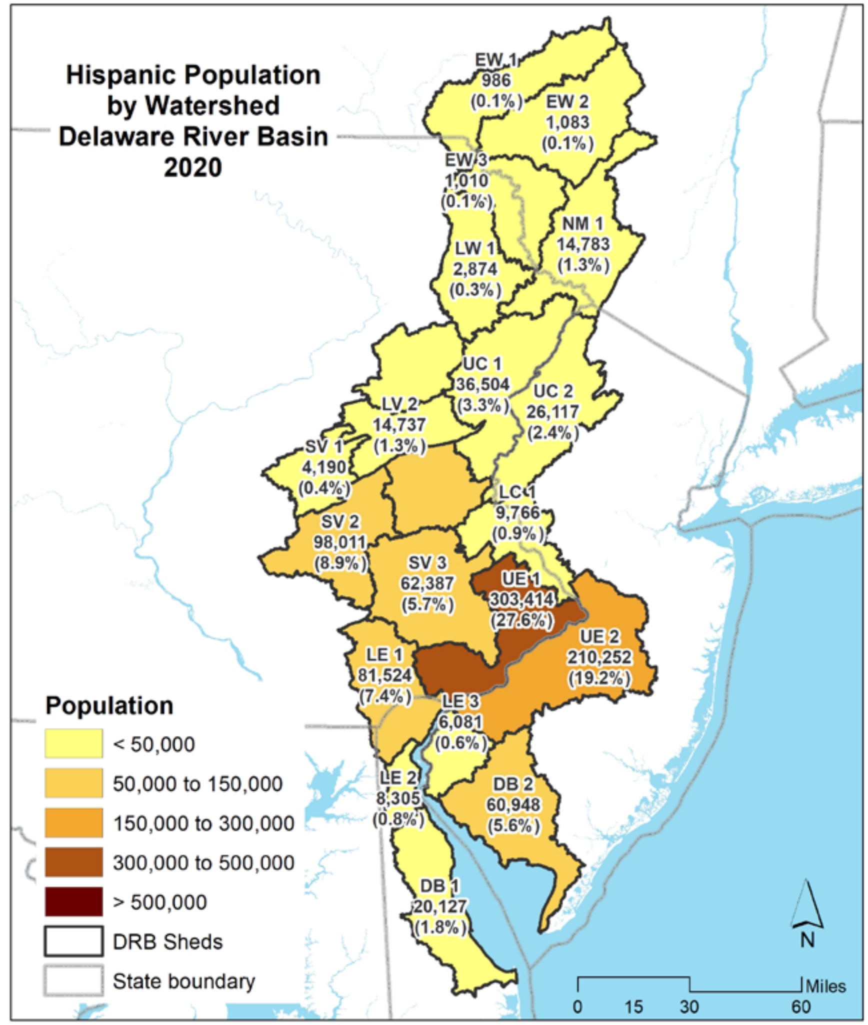


Figure 1.1.11 Hispanic population by watershed in the Delaware River Basin (2020).

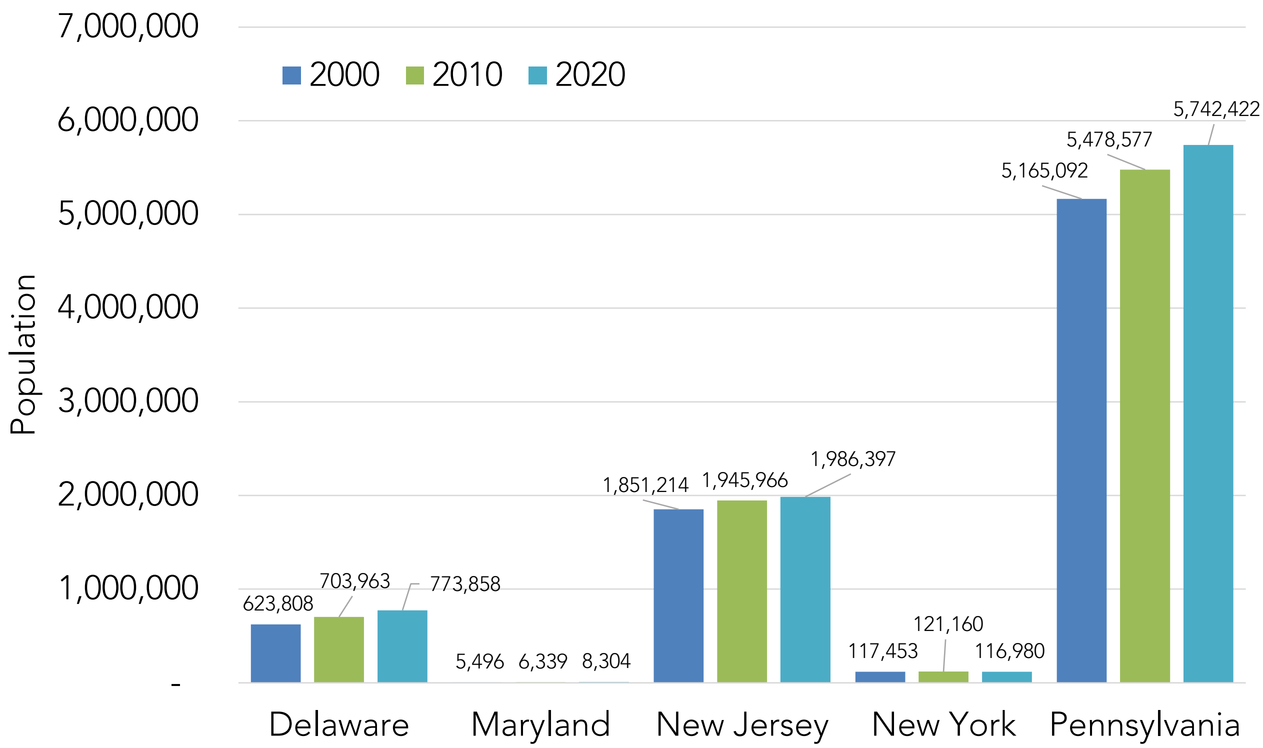


Figure 1.1.12 Population in the Delaware River Basin by year and state.

Table 1.1.3 Populations over time and population change by county for Pennsylvania.

County	Population			Population Change			
	2000	2010	2020	2010-20	% 2010-20	2000-20	% 2000-20
Berks County	361,361	397,634	413,877	16,243	4%	52516	15%
Bucks County	593,922	622,157	642,830	20,673	3%	48908	8%
Carbon County	59,011	65,979	65,183	-796	-1%	6172	10%
Chester County	396,849	453,757	485,275	31,518	8%	88426	22%
Delaware County	544,561	553,166	571,917	18,751	3%	27356	5%
Lackawanna County	5,597	6,426	7,007	581	10%	1409	25%
Lancaster County	737	1,086	959	-127	-17%	222	30%
Lebanon County	14,981	17,021	18,931	1,910	13%	3950	26%
Lehigh County	305,656	343,054	367,486	24,432	8%	61830	20%
Luzerne County	21,373	23,161	24,293	1,132	5%	2921	14%
Monroe County	137,583	169,172	167,532	-1,640	-1%	29949	22%
Montgomery County	751,287	802,342	857,629	55,287	7%	106342	14%
Northampton County	273,549	304,002	319,209	15,207	6%	45660	17%
Philadelphia County	1,518,220	1,525,400	1,607,416	82,016	5%	89196	6%
Pike County	46,493	57,177	58,203	1,026	2%	11710	25%
Schuylkill County	87,298	85,893	84,839	-1,054	-1%	-2460	-3%
Wayne County	46,613	51,151	49,836	-1,315	-3%	3223	7%
Pennsylvania Total	5,165,092	5,478,577	5,742,422	263,845	5%	577,331	11%

Table 1.1.4 Population over time and population change within the Delaware Basin by county for Delaware, Maryland, New Jersey, and New York.

County	State	Population			Population Change			
		2000	2010	2020	2010-20	% 2010-20	2000-20	% 2000-20
Kent County	DE	107,850	141,346	164,526	23,180	21%	56676	53%
New Castle County	DE	486,336	519,130	553,336	34,206	7%	67000	14%
Sussex County	DE	29,622	43,487	55,996	12,509	42%	26374	89%
Delaware Total		623,808	703,963	773,858	69,895	11%	150,051	24%
Cecil County	MD	5,496	6,339	8,304	1,966	36%	2808	51%
Maryland Total		5,496	6,339	8,304	1,966	36%	2,808	51%
Atlantic County	NJ	4,766	5,470	5,323	-147	-3%	557	12%
Burlington County	NJ	413,729	439,697	450,658	10,961	3%	36929	9%
Camden County	NJ	440,664	442,152	450,197	8,045	2%	9533	2%
Cape May County	NJ	31,758	30,845	31,820	975	3%	62	0%
Cumberland County	NJ	146,771	156,901	154,232	-2,669	-2%	7461	5%
Gloucester County	NJ	231,921	258,306	272,431	14,125	6%	40510	17%
Hunterdon County	NJ	32,555	35,139	33,061	-2,078	-6%	506	2%
Mercer County	NJ	259,121	269,344	284,477	15,133	6%	25356	10%
Monmouth County	NJ	9,850	12,360	12,434	74	1%	2584	26%
Morris County	NJ	27,023	30,575	31,782	1,206	4%	4759	18%
Ocean County	NJ	10,228	11,724	10,342	-1,382	-14%	114	1%
Salem County	NJ	64,553	65,976	64,947	-1,029	-2%	394	1%
Sussex County	NJ	76,429	78,917	75,323	-3,594	-5%	-1106	-1%
Warren County	NJ	101,846	108,559	109,370	811	1%	7524	7%
New Jersey Total		1,851,214	1,945,966	,986,397	40,431	2%	135,183	7%
Broome County	NY	2,364	2,292	2,179	-113	-5%	-185	-8%
Chenango County	NY	120	103	39	-65	-54%	-81	-68%
Delaware County	NY	32,448	32,865	29,589	-3,276	-10%	-2859	-9%
Greene County	NY	224	236	236	0	0%	12	5%
Orange County	NY	17,693	18,250	17,262	-988	-6%	-431	-2%
Schoharie County	NY	124	135	75	-60	-49%	-49	-40%
Sullivan County	NY	63,440	66,332	67,021	689	1%	3581	6%
Ulster County	NY	1,040	946	580	-366	-35%	-460	-44%
New York Total		117,453	121,160	116,980	-4,180	-4%	(473)	0%
Delaware Estuary and Basin Total		7,763,062	8,256,005	8,627,962	371,957	5%	864,899	11%

Figure 1.1.18 shows the population change between 2010 and 2020 for each of the five demographic categories—total population, minority population, Hispanic population, non-minority population, and non-Hispanic population—within the Delaware Estuary and Basin. Figure 1.1.19 shows the percentage change in each demographic category 2010-2020.

Figure 1.1.20 shows the change in the minority population between 2010 and 2020, by watershed. Figure 1.1.21 shows the change in minority population between 2010 and 2020 by watershed as a percentage. Figure 1.1.22 shows the change in Hispanic population between 2010 and 2020 by watershed. Figure 1.1.23 shows the change in Hispanic population between 2010 and 2020 by watershed as a percentage.

The Delaware Estuary and Basin saw a 4% gain in total population, despite an area-wide decrease of over 481,000 (8.1%) of the total population of white residents in the basin over the ten-year period. This discrepancy was due to an increase in the minority population, which showed a nearly 850,000 (36%) increase, and the Hispanic population, which increased by over 307,000 (39%) over the ten-year period.

Future Predictions

Based on the trajectory of current demographic trends reported here, the proportion of non-white (i.e., people of a single race other than white, or of multiple races) will likely increase over the next decade. Similarly, the number and proportion of people who identify as Hispanic is also likely to increase. However, the overall population increase is likely to decline, which is concurrent with current national population trends (Vespa et al. 2020, Davis et al. 2022). It is also likely that areas in some of the watersheds, such as in the Schuylkill and Lehigh Valleys and the coastal Delaware Bayshore will continue to experience a majority of the increase in population pressure. In contrast, other areas may face challenges of a static or possibly declining population.

Actions and Needs

With an increasingly diverse and urbanized population within the Delaware Estuary and Basin, water quality and watershed health will become increasingly important for these regions and their inhabitants. Environmental impacts may increase in areas such as the Lehigh and Schuylkill Valleys and the Delaware Bayshore. Impacts on all water resources, including degradation in quality, increased flooding, groundwater pressures, and wetlands degradation will likely become an increasing concern in those areas. Given that the watersheds where population pressures are greatest are also often environmentally sensitive, it will be important to plan for protections for these resources and to secure sufficient funding, resources, and political will to anticipate and counter potential negative impacts.

Summary

Given the trajectory of demographic trends reported here, the pressures on the resources of the Delaware Estuary and Basin will likely continue to increase. While the overall population rate of increase may be starting to flatten, the relative increasingly urbanized population will present challenges in the coming decades. These challenges may be seen in terms of impervious cover, potential pollution sources, and competing interests for water resources. As populations become more diverse, we also must ensure there is equity in access to resources, especially in environmental justice communities. Planners and policymakers need to anticipate differential changes and plan accordingly in each of the Delaware Estuary and Basin's watersheds.

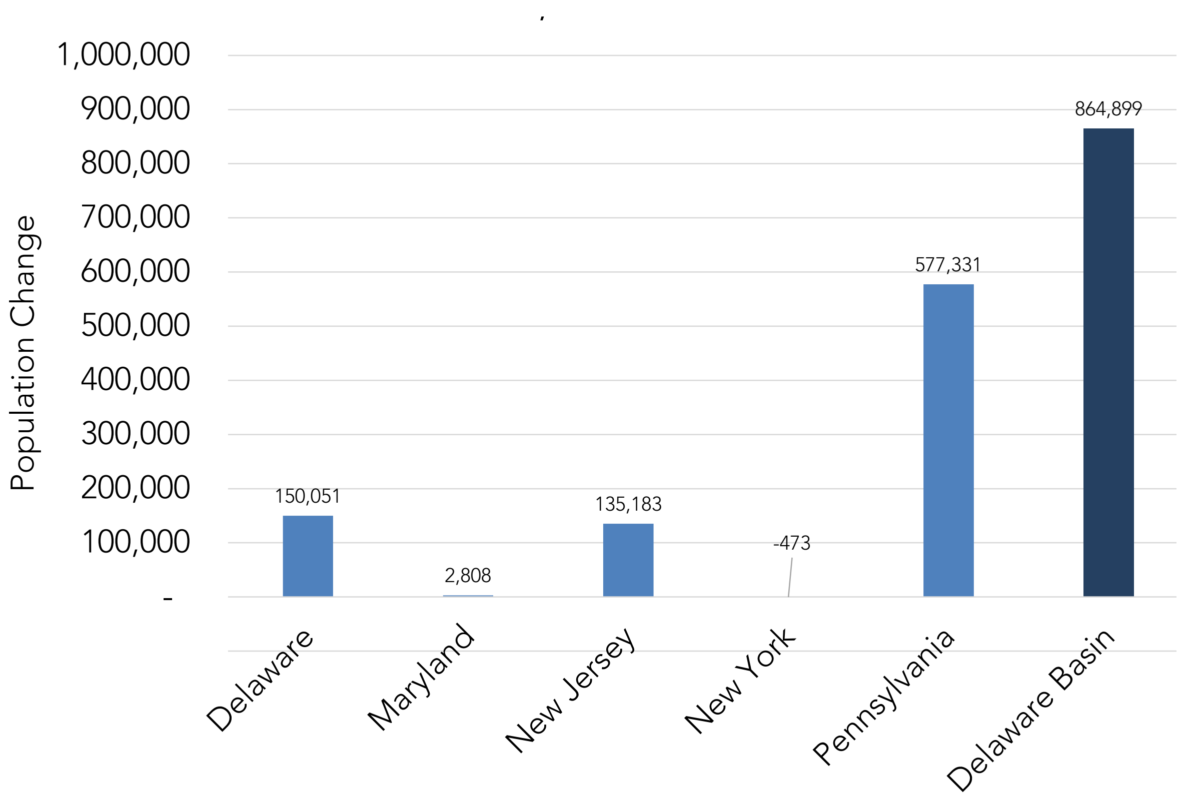


Figure 1.1.13 Population change in the Delaware River Basin by state (2000-2020).

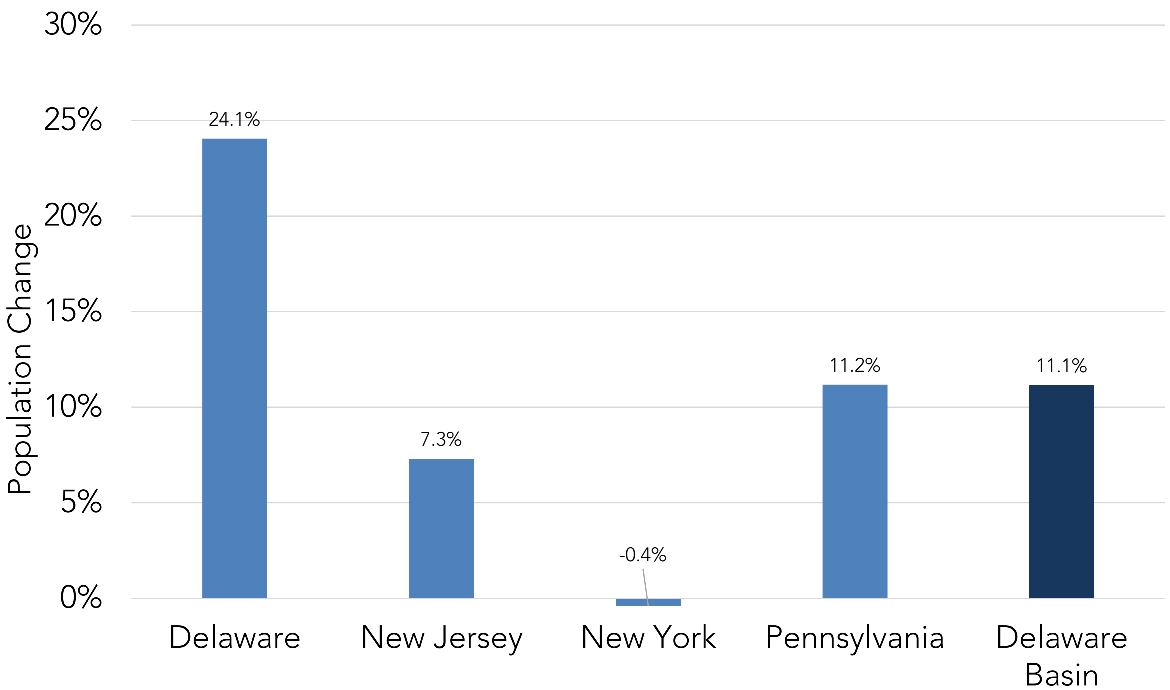


Figure 1.1.14 Percent population change in the Delaware River Basin by state (2000-2020). Maryland is not depicted here as the total population is low compared to other states.

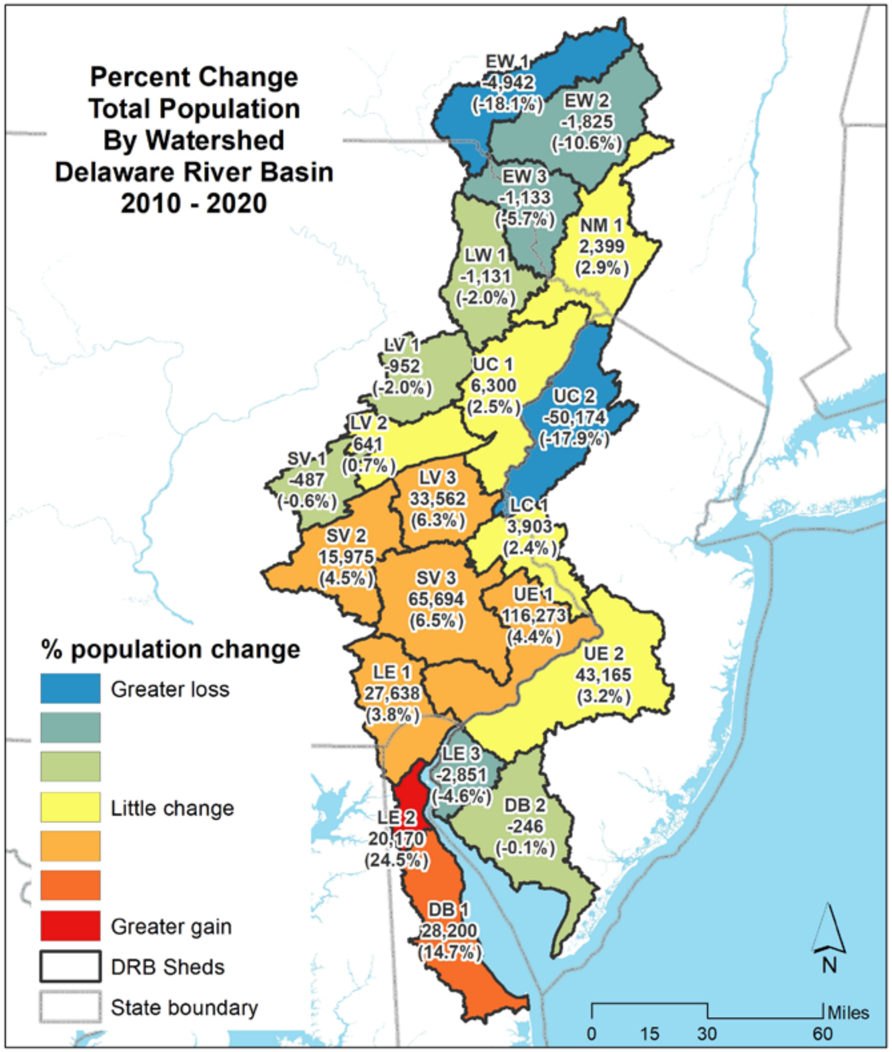


Figure 1.1.15 Percent change in total population by watershed in the Delaware River Basin (2010-2020).

Table 1.1.5 Percent change in total population by watershed in the Delaware River Basin (2010-2020).

Watershed	Population		Population Change	
	2010	2020	2010-20	% 2010-20
UE 1	2,635,302	2,754,029	118728	5%
UE 2	1,349,250	1,395,425	46175	3%
SV 3	1,008,438	1,075,332	66895	7%
LE 1	718,651	745,356	26705	4%
LV 3	528,693	565,233	36540	7%
SV 2	355,870	372,478	16608	5%
UC 2	231,057	230,823	-234	0%
UC 1	249,463	255,010	5547	2%
DB 2	251,514	248,567	-2947	-1%
DB 1	194,917	224,810	29894	15%
LC 1	163,951	167,844	3892	2%
LV 2	98,037	97,980	-57	0%
SV 1	86,992	85,862	-1131	-1%
LE 2	84,217	104,047	19830	24%
NM 1	82,813	84,644	1832	2%
LE 3	59,703	60,514	811	1%
LW 1	57,068	55,847	-1221	-2%
LV 1	48,434	48,109	-324	-1%
EW 1	24,020	22,297	-1723	-7%
EW 3	19,891	18,712	-1179	-6%
EW 2	17,124	15,241	-1882	-11%
TOTAL	8,265,405	8,628,162	362,758	4%

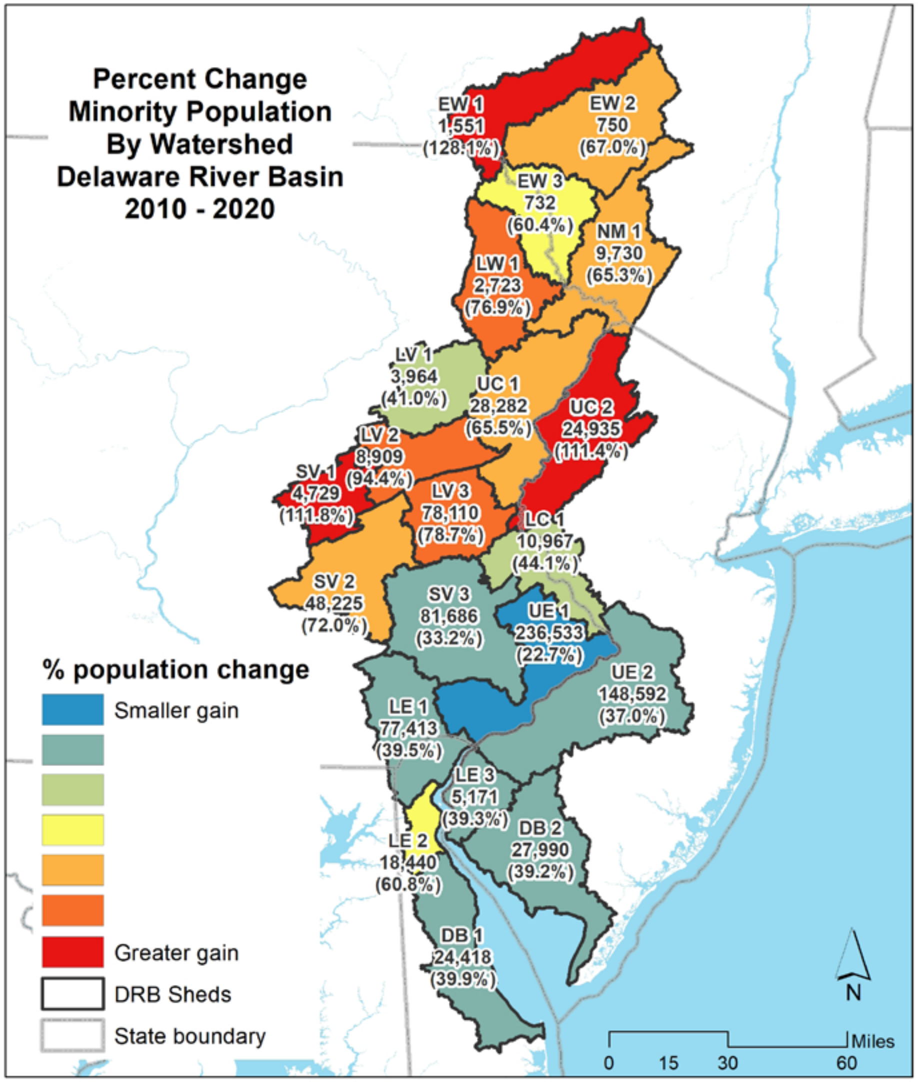


Figure 1.1.16 Percent change in minority population by watershed in the Delaware River Basin (2010-2020).

Table 1.1.6 Percent change in minority population by watershed in the Delaware River Basin (2010-2020).

Watershed	Population		Population Change	
	2010	2020	2010-20	% 2010-20
UE 1	1,041,495	1,278,028	236,533	23%
UE 2	401,064	549,656	148,593	37%
SV 3	246,085	327,771	81,686	33%
LE 1	195,857	273,270	77,413	40%
LV 3	99,290	177,400	78,110	79%
SV 2	66,943	115,168	48,225	72%
UC 2	22,389	47,324	24,935	111%
UC 1	43,161	71,443	28,282	66%
DB 2	71,396	99,386	27,990	39%
DB 1	61,274	85,692	24,418	40%
LC 1	24,841	35,808	10,967	44%
LV 2	9,440	18,349	8,910	94%
SV 1	4,230	8,959	4,729	112%
LE 2	30,329	48,769	18,440	61%
NM 1	14,897	24,627	9,730	65%
LE 3	13,149	18,320	5,171	39%
LW 1	3,539	6,262	2,723	77%
LV 1	9,667	13,631	3,964	41%
EW 1	1,211	2,762	1,550	128%
EW 3	1,212	1,944	731	60%
EW 2	1,120	1,870	750	67%
TOTAL	2,362,589	3,206,439	843,849	36%

Table 1.1.7 Percent change in Hispanic population by watershed in the Delaware River Basin (2010-2020).

Watershed	Population		Population Change	
	2010	2020	2010-20	% 2010-20
UE 1	230,021	303,414	73,393	32%
UE 2	149,650	210,252	60,603	40%
SV 3	39,374	62,387	23,013	58%
LE 1	63,676	81,524	17,847	28%
LV 3	88,415	127,685	39,270	44%
SV 2	66,477	98,011	31,534	47%
UC 2	16,970	26,117	9,147	54%
UC 1	25,943	36,504	10,561	41%
DB 2	48,351	60,948	12,597	26%
DB 1	13,866	20,127	6,261	45%
LC 1	6,286	9,766	3,480	55%
LV 2	8,660	14,737	6,077	70%
SV 1	1,890	4,190	2,301	122%
LE 2	5,234	8,305	3,071	59%
NM 1	11,032	14,783	3,751	34%
LE 3	4,199	6,081	1,882	45%
LW 1	2,169	2,874	705	33%
LV 1	5,736	7,008	1,272	22%
EW 1	758	986	228	30%
EW 3	848	1,010	162	19%
EW 2	988	1,083	95	10%
TOTAL	790,544	1,097,793	307,249	39%

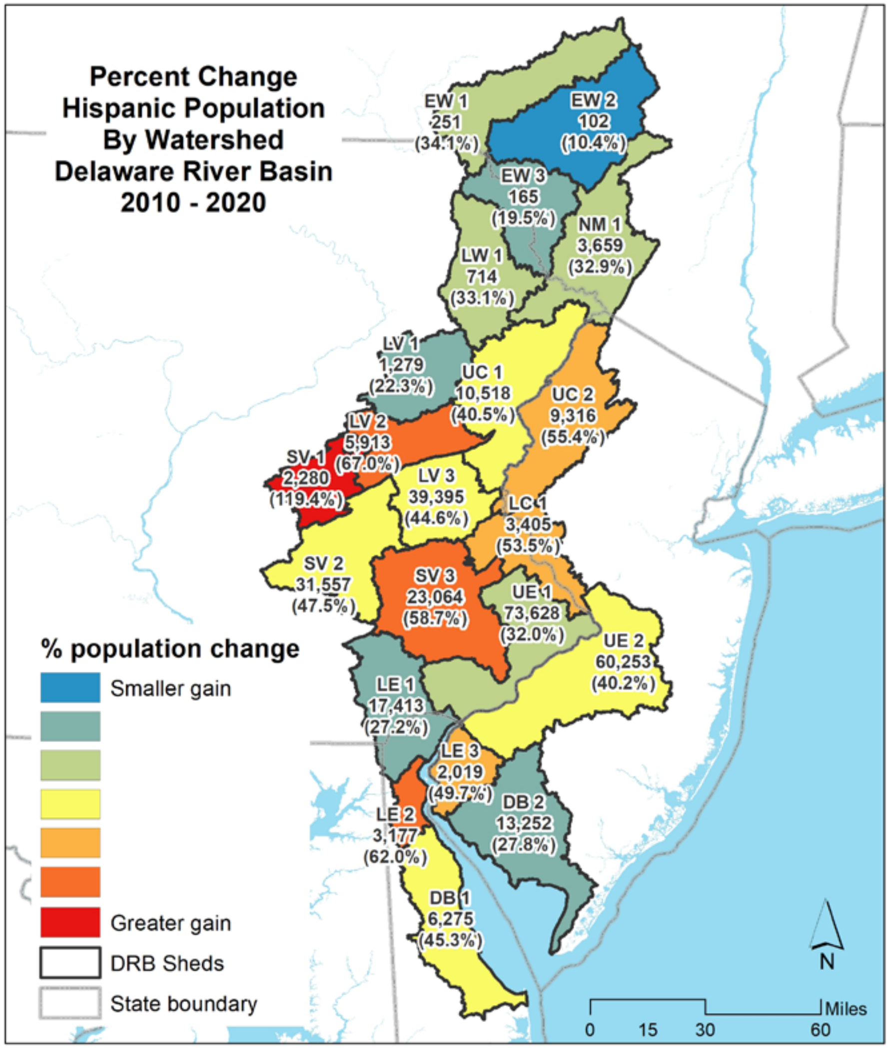


Figure 1.1.17 Percent change in Hispanic population by watershed in the Delaware River Basin (2010-2020).

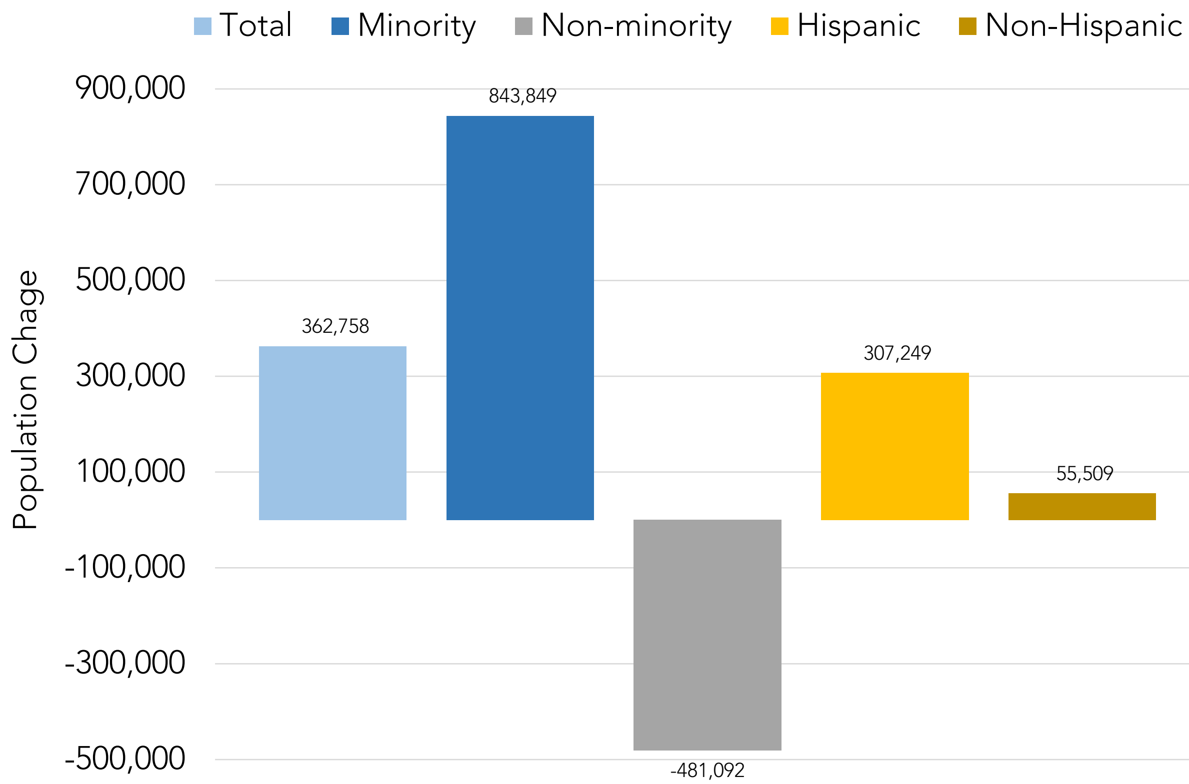


Figure 1.1.18 Population change by type (2010-2020).

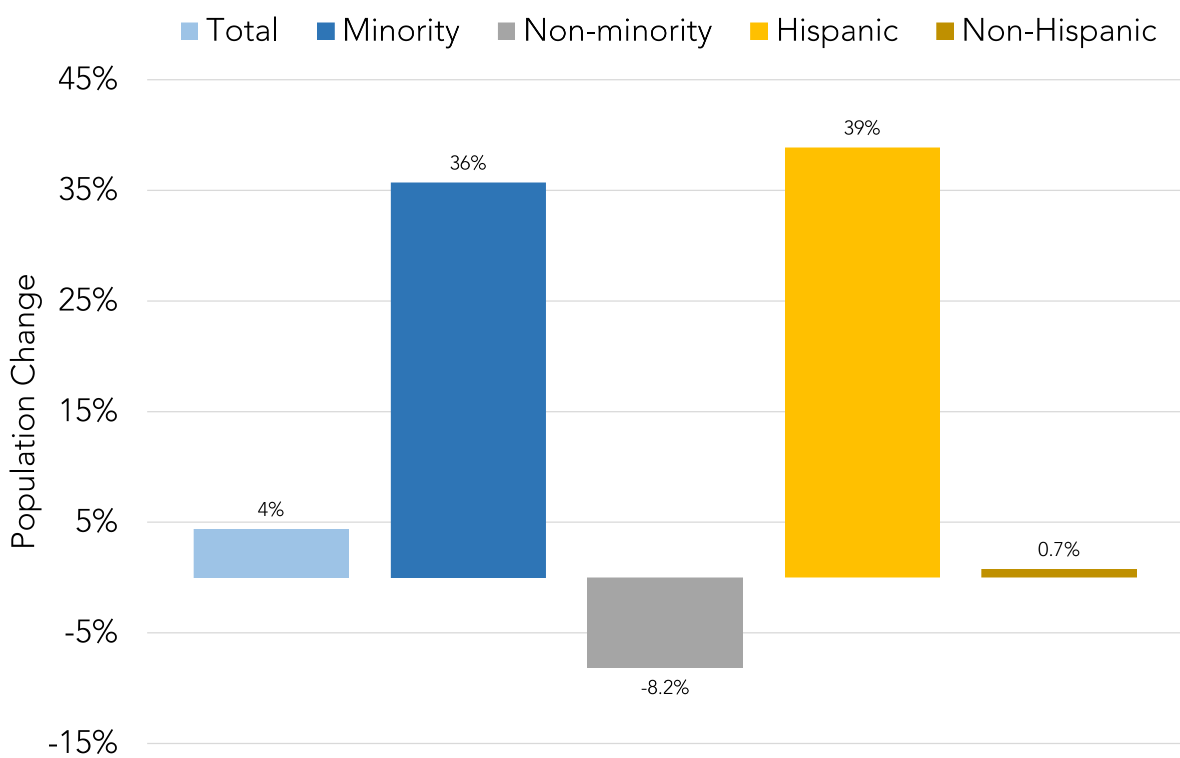


Figure 1.1.19 Percent population change by type (2010-2020).

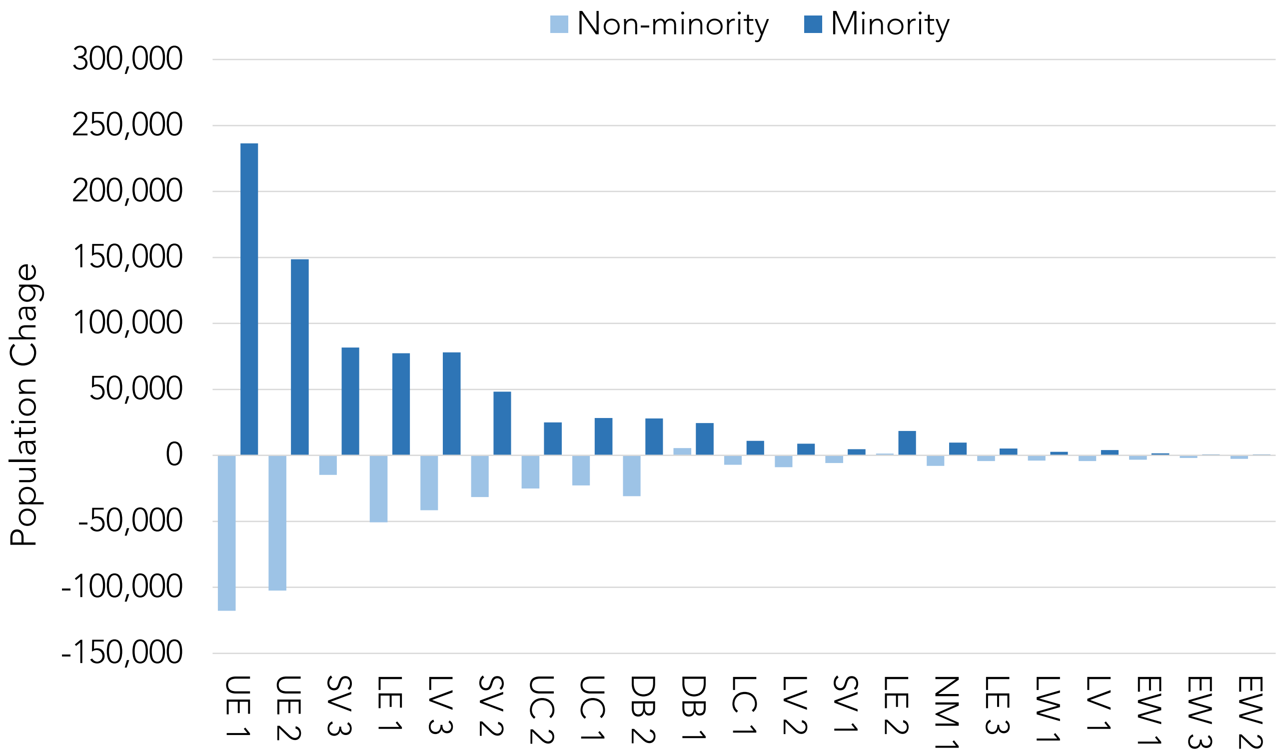


Figure 1.1.20 Minority and non-minority population change (2010-2020).

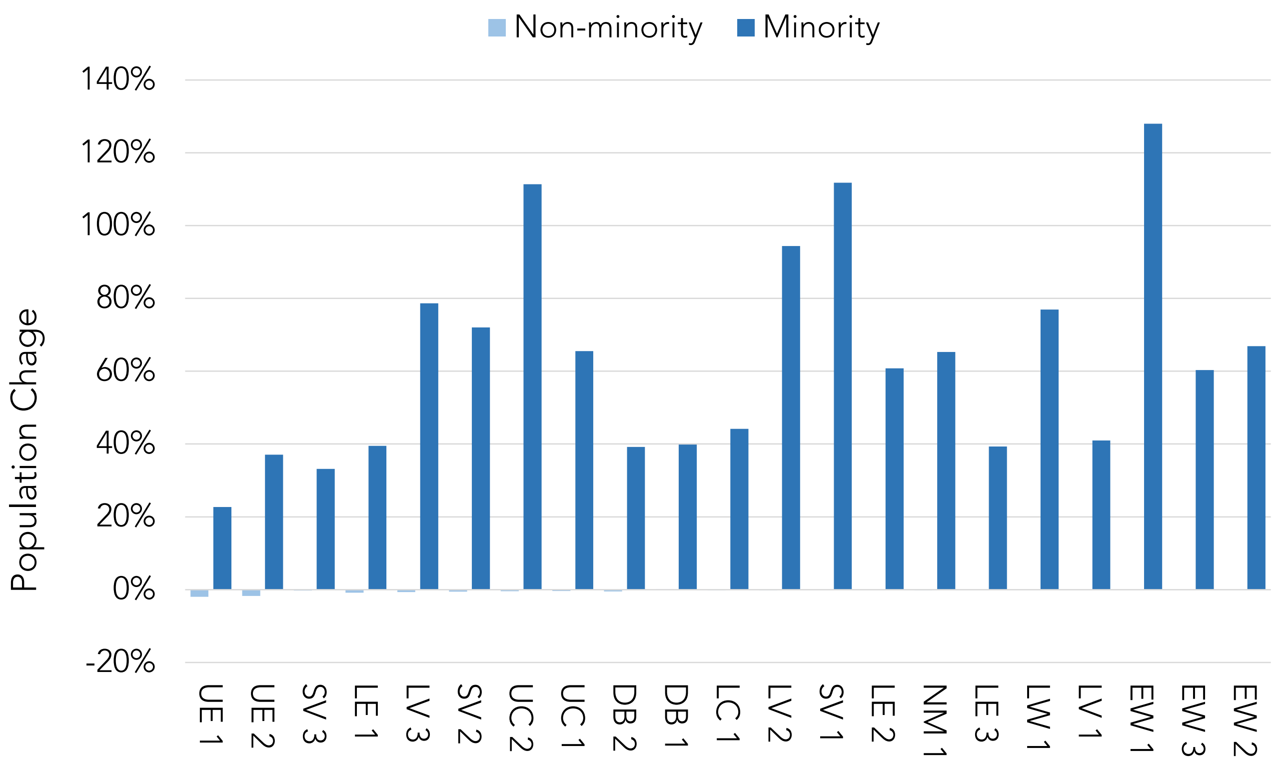


Figure 1.1.21 Minority and non-minority percent population change (2010-2020).

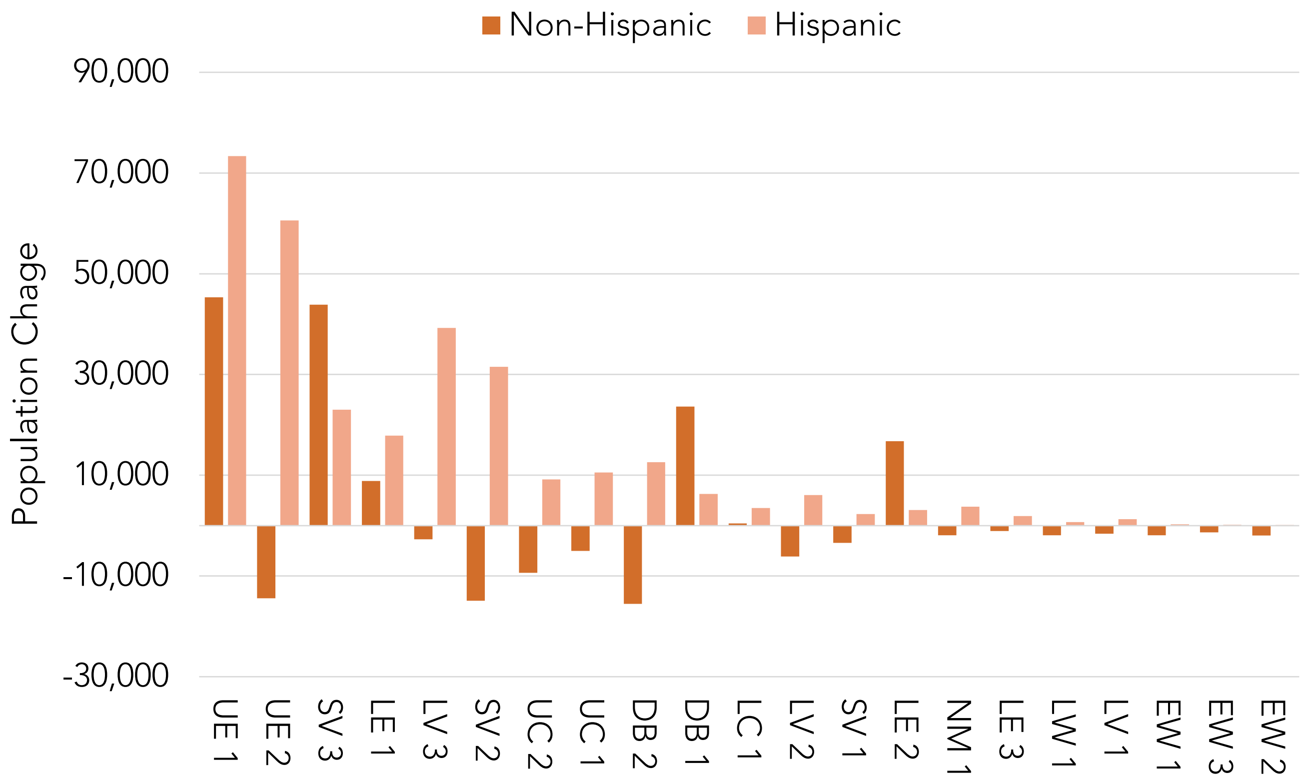


Figure 1.1.22 Hispanic and non-Hispanic population change (2010-2020).

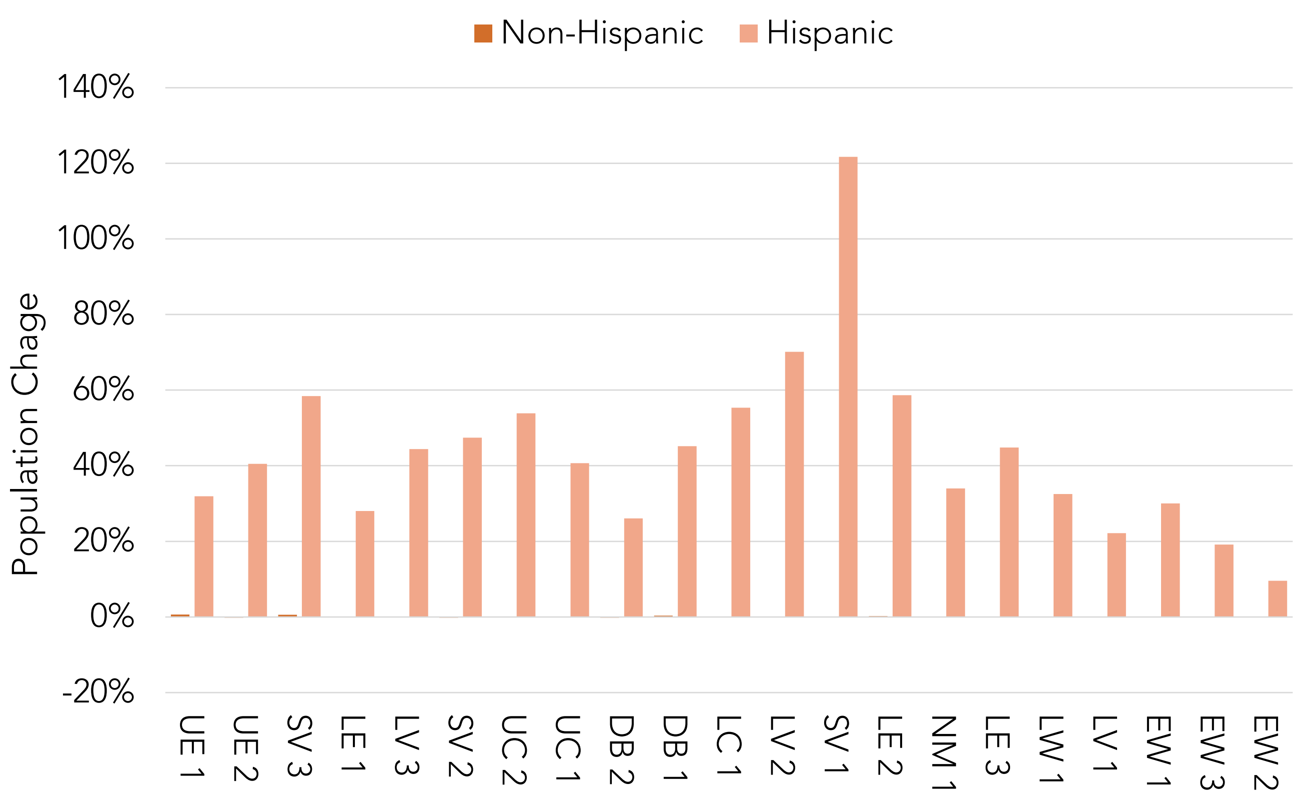


Figure 1.1.23 Hispanic and non-Hispanic percent population change (2010-2020).

1.2 Land Cover

Description of Indicator

Land cover describes what is physically present on a particular area of ground. Land cover is pertinent for determining the effects of a land cover type on the overall health of a watershed, including aspect of water quality (Kauffman et al. 2011). For instance, a cover type of high-intensity developed land would have characteristic impacts related to its intensity, largely independent of what particular land use practice is there. When assessing overall watershed health at a broad scale, land cover is useful because it is comparable across a variety of landscapes and is relatively consistent across time. What is physically present on the landscape has a profound and lasting impact on the overall environmental health of those landscapes.

Land cover is constantly changing. In some cases the changes are natural, resulting from vegetative succession, uplift, volcanism, erosion, wildlife migration, or natural climatic shifts. Other changes occur due to human interaction and agency on the land. Processes such as the development of towns and cities, transportation and communication networks, population dispersal or migration, and agricultural activity have occurred throughout human history and led to significant and sometimes dramatic changes in the landscape. Other processes, such as human-induced climate change, air and water pollution, famine and disease, and widespread dispersal of invasive species, among other factors, are the indirect result of human activity, and can also lead to major impacts on the landscape.

Some changes are quite gradual, taking decades or longer, while others can occur quite quickly, over the span of a few years or even more rapidly. These changes often have a profound impact on the health of watersheds and water resources, and the societies and organisms which depend on them. Understanding how the land cover of an area is changing over time is an important metric to determine if conditions are degrading, improving, or holding steady. Using a high-quality, high-resolution dataset over a decadal time frame is a useful way of tracking those changes and understanding potential future trends. By comparing changes in the landscape with other metrics such as water quality, it is possible to quantify the effects of those changes over time. Land cover is not spatially consistent across the Delaware Estuary and Basin, and so the change in land cover over time can be highly variable as well. Land cover is often a reflection of economic, demographic, societal, political, and regulatory factors. By studying the nature of land cover and particularly how fast it is changing, watershed professionals can understand patterns in the landscape and can anticipate future changes, and seek to lessen their negative impacts.

Land cover types

In this chapter, four broad land cover types are assessed: developed land, agricultural land, natural lands, and open water. Developed land is generally associated with lower-quality habitat and lower water quality values than more natural types of land cover. More highly developed areas have higher degrees of imperviousness, as well as a greater number of human activities that can potentially affect water quality, exacerbate flooding, and reduce base flows in streams. Agricultural land cover can have significant impacts on stream and watershed health, depending on the type of practices employed, and the character of the crop or animal operation. Increased nutrients, sediment, and bacteria are often a result of upstream farming practices. The types of practices on the ground can also have a great impact on overall water quality and watershed health; proper management of waste and runoff are crucial to protecting the water resources of the Basin, particularly since agriculture makes up a large proportion of the Delaware Estuary and Basin.

Natural lands consist of forests and wetlands. Forest cover is associated with pre-development conditions of water quality and hydrology. Forests cycle nutrients and carbon dioxide, capture rainfall and inhibit erosion, and play an important role in water quality, quantity, and habitat provision for aquatic and terrestrial wildlife. The presence of forests is a hallmark for defining high-quality watersheds. Wetlands, both freshwater and tidal, are a natural land cover type and therefore associated with clean and healthy watersheds. Wetlands provide a wide variety of ecosystem services, including water purification, habitat provision, flood protection, pollution reduction, recreation, sea level rise, and storm-surge amelioration.

Open water includes lakes, ponds, and streams in the Delaware Estuary and Basin. The largest body of water in the watershed are its namesakes, the Delaware River and Bay, but that area is not included in land cover analysis by watershed. The "other" category consists mostly of barren land, transitional land, and beaches/dunes. This category makes up a very small portion of land cover in the Estuary and Basin.

Data sources and processing methodology

Land cover status is important to a holistic assessment of the current condition and past or future trends within a watershed. In order to determine the land cover status for the nearly 13,000 square miles, Delaware Estuary and Basin, it was important to use a consistent source of data to assure that comparisons across the watersheds, as well as across years, are valid. The National Oceanic and Atmospheric Administration (NOAA) administers the Coastal Change Analysis Program (C-CAP), which characterizes near coastal land cover (within an ~200-mi boundary of the US coastline). Unlike the National Land Cover Dataset, C-CAP classifictions include more refined coastal habitat classifications, such as estuarine wetlands. Land cover data are available for the past several decades at approximately five-year intervals. This source is useful since it is consistent over time and across the study area. This source has been used in previous editions of the Technical Report for the Estuary and Basin (TREB) and provides another useful "snapshot" of watershed conditions in the Delaware Estuary and Basin. The latest vintage of C-CAP land cover data was used for the land cover analysis. Previous years were also analyzed for land cover trends. Table 1.2.1 presents the classification of land cover types in C-CAP data and the corresponding simplified category that is used for all subsequent analyses in this TREB. Note that while the C-CAP category "shrub/scrub" is not specifically agricultural, it was included in this category as its spectral signature is similar to agriculture.

Table 1.2.1 C-CAP classification scheme.

Developed	Agriculture	Natural Lands		Open Water	Other
		Forest	*Wetlands*		
Low intensity	Cultivated Land	Deciduous	Palustrine Emergent	Water	Unconsolidated shore
Medium intensity	Pasture/Hay	Evergreen	Estuarine Emergent	Aquatic beds	Transitional land
High intensity	Grassland	Mixed	Forested		Other
Open space	Shrub-scrub		Shrub-scrub		

NOAA C-CAP data are based on satellite imagery at a 30-meter ground resolution, which is classified into over 20 land cover categories. A simplified scheme of six categories was used (see the following section). The six categories of land cover used to characterize the Delaware Estuary and Basin are: developed (or developing), agriculture, forest, wetland, open water, and other. This simplified classification scheme is helpful in quantifying the character (and changes) within the landscapes of the Delaware Estuary and Basin at a watershed scale and affords a generalized picture of their condition.

Data were obtained through NOAA's C-CAP program through their online portal for 1996, 2001, 2006, 2011, and 2016 (the latest available data for this program). Each dataset covered the entire Delaware Estuary and Basin and is based on Landsat satellite imagery at 30-meter ground resolution. Note that previous TREB reports have used the same data source (C-CAP) but the entire five-year series was re-acquired, due to some minor classification changes noted in some of the previous datasets since the last TREB report in 2017. The data from 2016 were used for this section (land cover), and all years were used in consideration of the changes over time (see Section 1.3). All land cover data were obtained in the Universal Transverse Mercator (UTM) projection, Zone 18 North, cast on the North American Datum of 1983 (NAD83), with coordinates in meters.

To assess the condition of the land cover across the Delaware Estuary and Basin, the 21 watersheds within the Estuary and Basin were used as a cataloging unit on which to summarize the data. The total land cover area was calculated for each watershed. To determine the land cover characteristics of each sub-category within the overall Delaware Estuary and Basin, and all levels of watershed hierarchy including regions, sub-regions, each of the 10 watershed divisions, and the data for the 21 watersheds were aggregated to produce summary information in this hierarchy. See Figure 1.2.1 for a schematic representation of the assessment units and reporting hierarchy within the Delaware Estuary and Basin. For each of the 21 watersheds (the smallest level of hierarchy, represented by the numbered divisions in the lowest row), land use profiles were calculated. The proportion by land area of the Delaware Estuary and Basin represented by each region is presented in Figure 1.2.2.

Also using C-CAP data, we performed a broad four category classification of land cover changes in the Delaware Estuary and Basin between 1996 and 2016. Changes are more difficult to verify at a lower resolution, and so we chose to represent land cover changes using these broader land cover categories.

Figure 1.2.1 Basin assessment units and reporting hierarchy.

Basin									
Nontidal						Estuary			
Upper Region			Central Region			Lower Region			Bayshore Region
East-West 1, 2, 3	Lacka-waxen	Neversink-Mongaup	Upper Central 1,2	Lehigh Valley 1,2,3	Lower Central	Schuylkill Valley 1,2,3	Upper Estuary 1,2	Lower Estuary 1,2,3	Bayshore 1,2

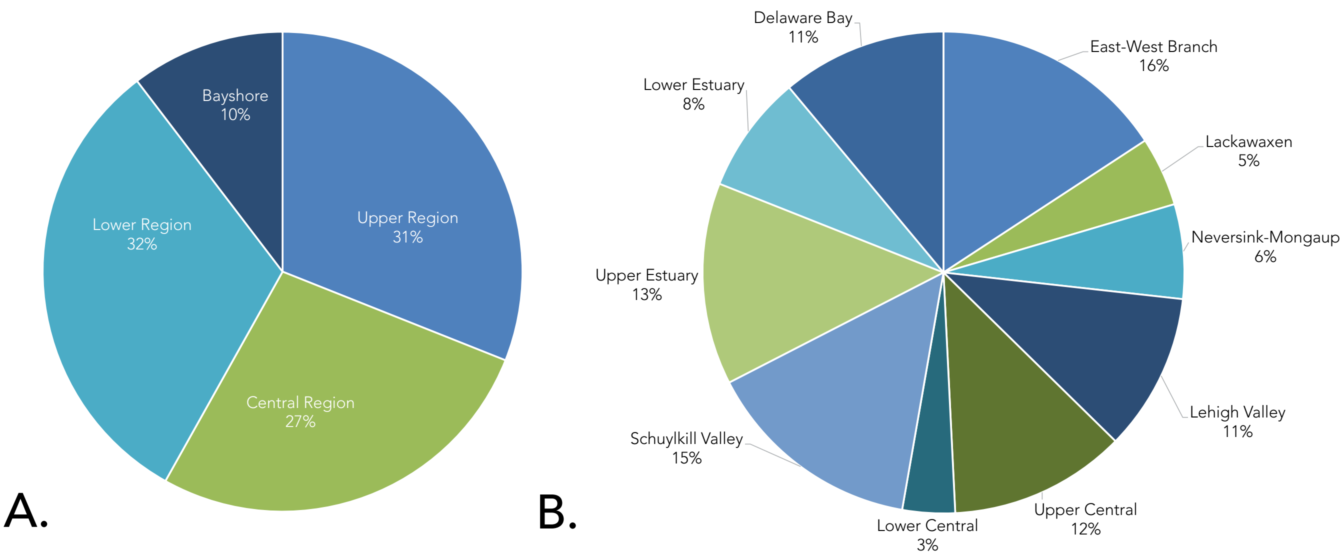


Figure 1.2.2 Proportion of land area by region (A) and subregion (B)

The four categories were Developed, Agricultural, Natural Lands, and Other (see Table 1.2.1). Importantly, "Forest[1]" includes forest C-CAP classes for forest (see Table 1.2.1) as well as salt and freshwater forested/scrub-shrub wetlands; and "Other" includes emergent wetland, barren, open water, etc. Additionally, "Any (non-Developed)" were categories classified as Forest or Agriculture in 1996, but classed one of the "Other" classes in 2016.

Present Status

The Delaware Estuary and Basin comprises approximately 12,862 mi^2 (33,312 km^2) within Delaware, New Jersey, New York, Pennsylvania, and Maryland (Tables 1.2.2, 1.2.3). Over half (53%) falls within the non-tidal watersheds of the Upper Basin. The remainder forms the Delaware Estuary (i.e., the watersheds of the tidal portion of the Basin). The Delaware Bay itself is in the lower portion of the Estuary, and covers 752 mi^2 (1,948 km^2), resulting in a total area of 13,614 mi^2 (35,268 km^2) for the Basin (including the Delaware Bay). With the Bay included, more than half (50.2%) of the Basin is the Delaware Estuary (and under the scope of the National Estuary Program). All land cover analysis has excluded the Bay. Figure 1.2.3 shows the land cover of the Delaware Estuary and Basin.

Maps in the following figures (Figs 1.2.4, 1.2.5) show the proportion of human-manipulated land cover categories as a percentage of total land cover by watershed in the Delaware Estuary and Basin. The most

Table 1.2.2 Area of each portion of the Upper Basin.

Basin Division	Non-tidal/Upper Basin						
Region	East-West	Lackawaxen	Neversink-Mongaup	Vehigh Valley	Upper Central	Lower Central	Schuylkill Valley
Watershed division or subregion	EW1, EW2, EW3	LW	NM	LV1, LV2, LV3	UC1, UC2	LC1	SV1, SV2, SV3
Square miles	2,029	597	816	1,361	1,527	454	1,891
Square kilometers	5,256	1,547	2,113	3,524	3,956	1,175	4,898
% of Region	59%	17%	24%	41%	46%	14%	41%
% of Region	30%	9%	12%	20%	23%	7%	31%
% of Basin	16%	5%	6%	11%	12%	4%	15%

Table 1.2.3 Area of each portion of the Estuary.

Basin Division	Estuary			
Region	Upper Estuary	Lower Estuary	Bayshore	
			DE	NJ
Watershed division or subregion	UE1, UE2	LE1, LE2, LE3	DB1	DB2
Square Miles	1,743	1,020	634	789
Square Kilometers	4,515	2,642	1,642	2,044
% of Region	37%	22%	45%	55%
% of Region	29%	17%	10%	13%
% of Basin	14%	8%	5%	6%

1. Forested wetlands could also be considered with wetland categories, but for this broad scale, we consider these land covers to be more similar to other forested systems rather than grouped with emergent wetlands.

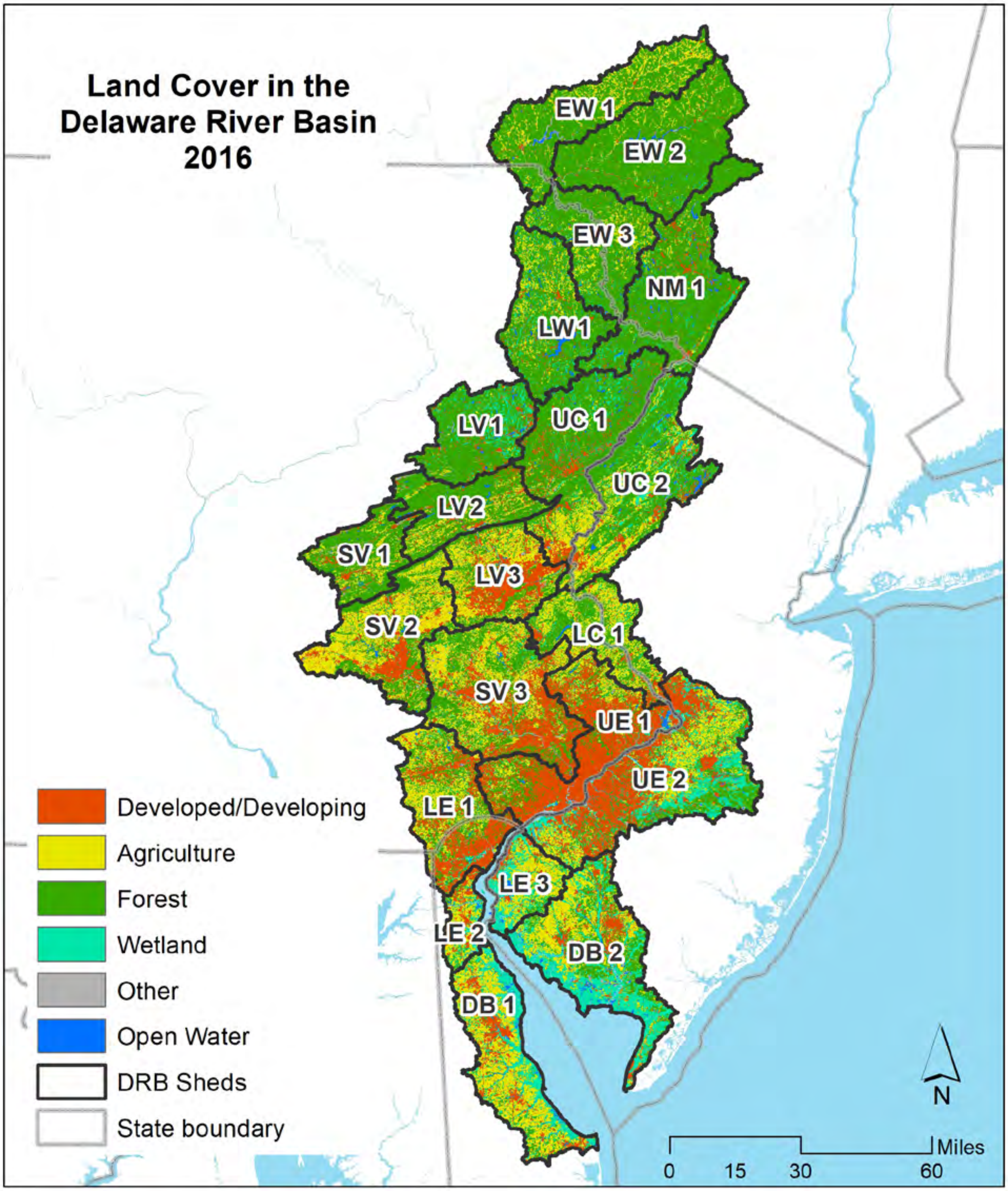


Figure 1.2.3 Land cover in the Delaware River Basin (2016).

developed areas are the watersheds surround the Philadelphia metropolitan area. Watershed UE2 is 61% developed, with less development moving out from there. Northern watersheds are the least developed, with 5% developed area or less. This reflects the historic character of the watershed. The distribution of natural cover is nearly the reverse of areas of higher development; the highest percentage of the natural cover occurring in the northern watersheds, including East-West Branch (EW 1, 2), Lackawaxan (LW 1), and Neversink-Mongaup (NM 1).

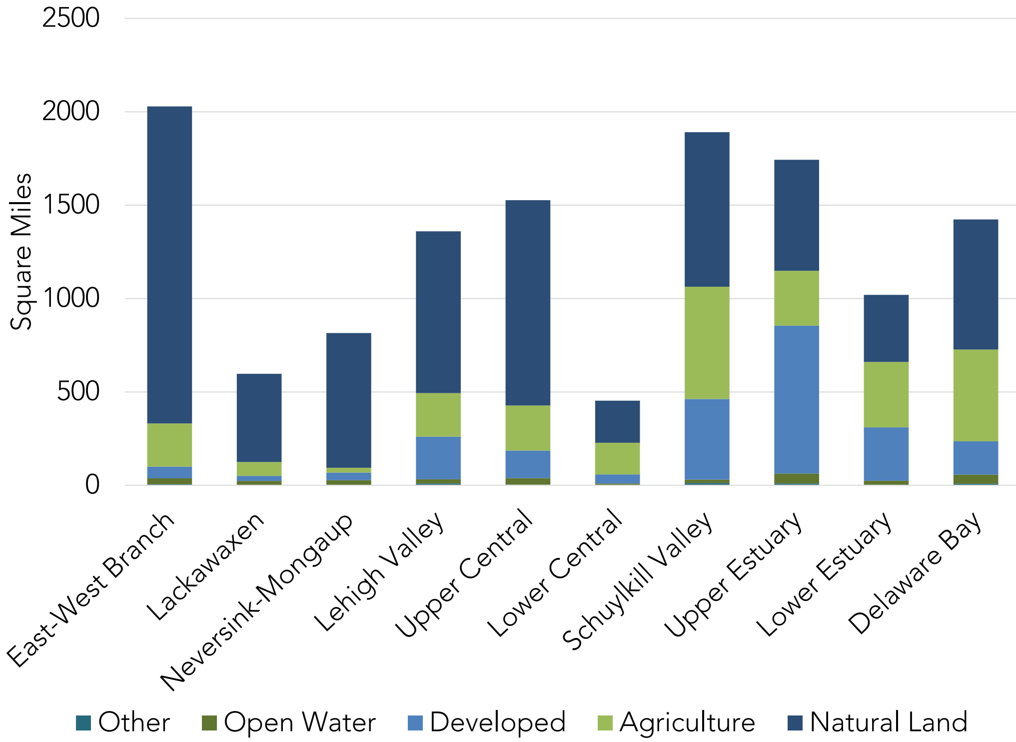


Figure 1.2.4 Land cover in the Delaware River Basin (2016).

Agriculture is a relatively small portion of the most urbanized watersheds. Areas of southeast Pennsylvania and central New Jersey have the highest proportions of agriculture, while the most northern and most highly forested watersheds have the lowest percentages of agricultural land cover.

Overall, the basin is nearly 60% natural lands, with 21% in agriculture and 17% developed. Figure 1.2.6 shows the amount of each land cover type by watershed in the Delaware Estuary and Basin.

The Estuary portion of the Delaware Estuary and Basin comprises the lower four watersheds (with 10 of the 21 watersheds), including all watersheds whose rivers flow into the main stem of the Delaware River below the tidal limit. The Delaware Estuary includes the most populous and developed portions of the Delaware Estuary and Basin, as well as the highest amount of agricultural land. The Delaware Bay watershed is also characterized by a high percentage of tidal wetland along the Bayshore.

Figure 1.2.7 shows the percent land cover types for the upper Basin (non-tidal watersheds) (A), and the Estuary (B). Figure 1.2.8 shows the land cover for both the Estuary, the upper Basin (non-tidal watersheds), and the Delaware Estuary and Basin as a whole. Table 1.2.4 presents a summary of all ten watersheds in the four regions of the Delaware Estuary and Basin, including total area in square miles and as a percentage, as well as for the upper Basin, Estuary, and whole watershed.

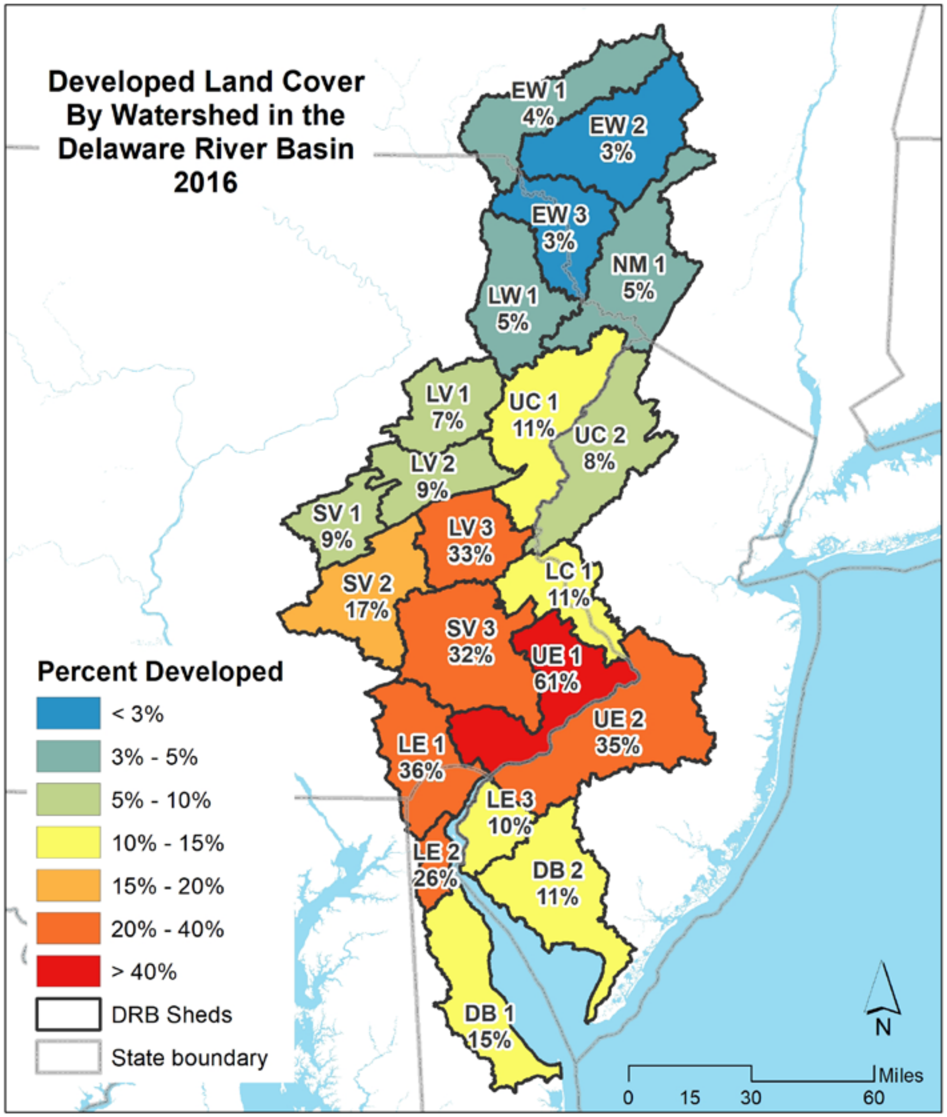


Figure 1.2.5 Developed land cover in the Delaware River Basin (2016).

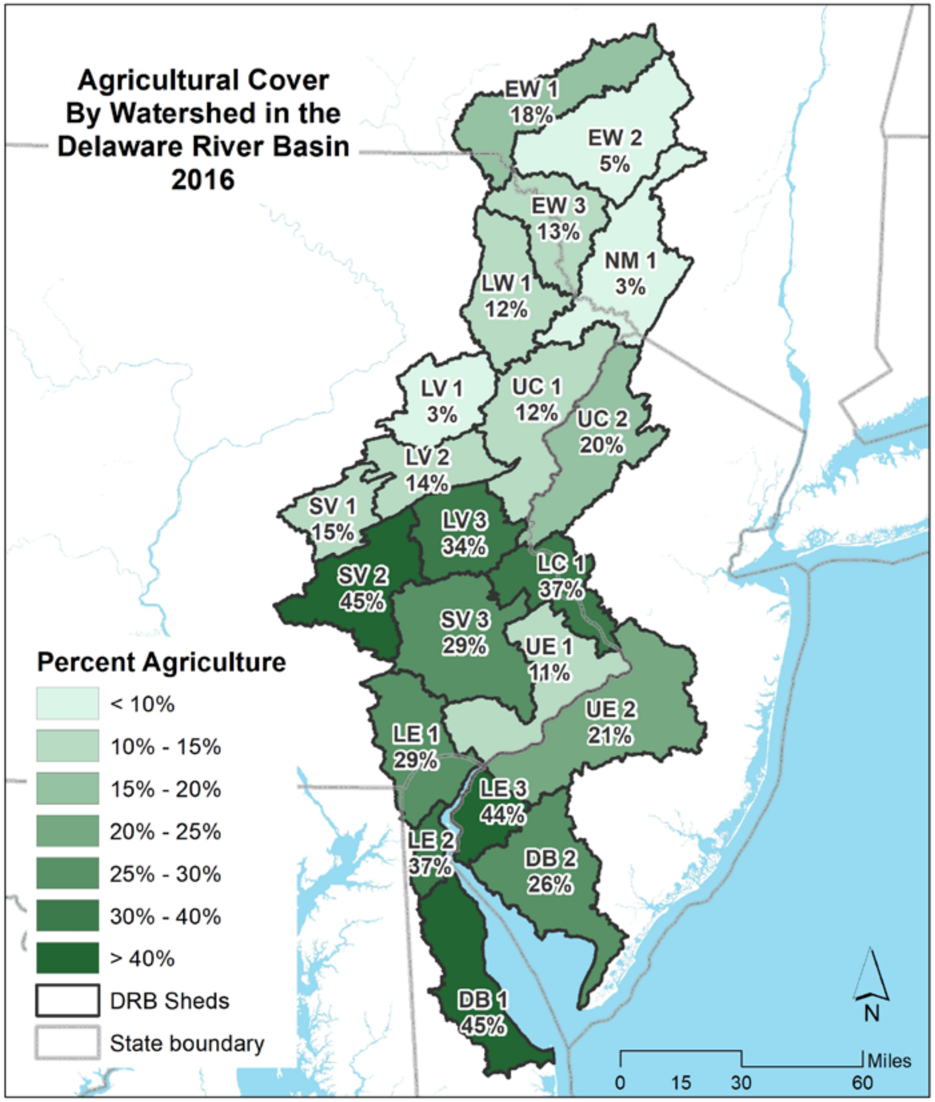
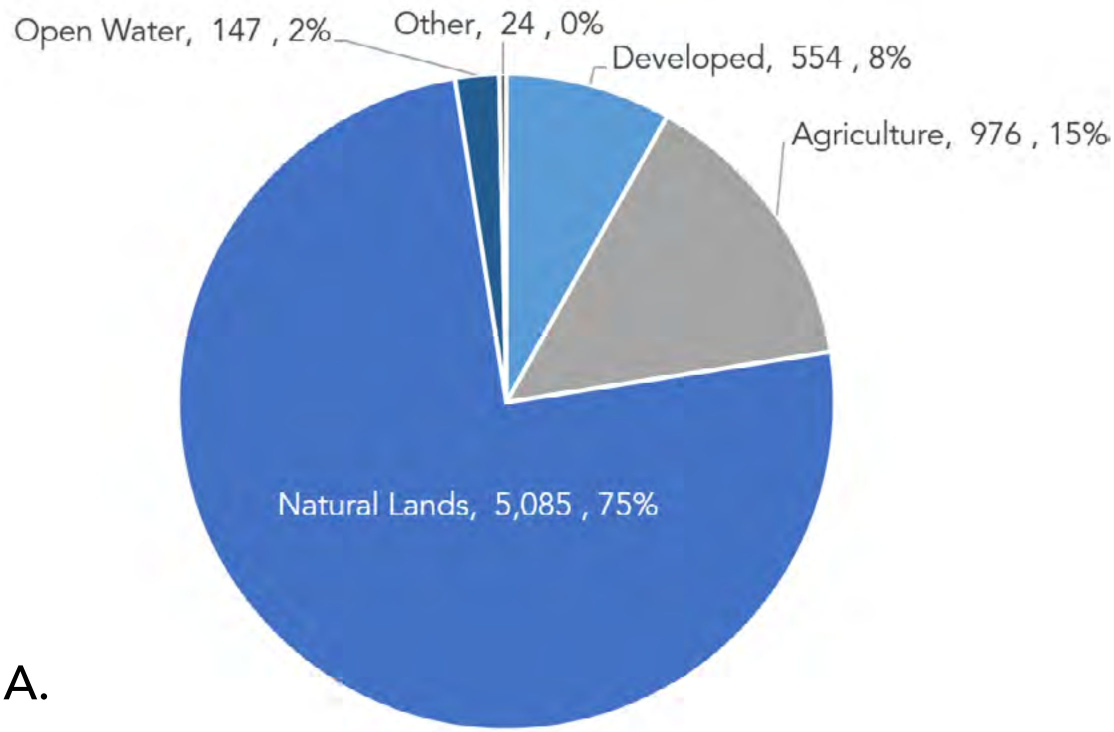
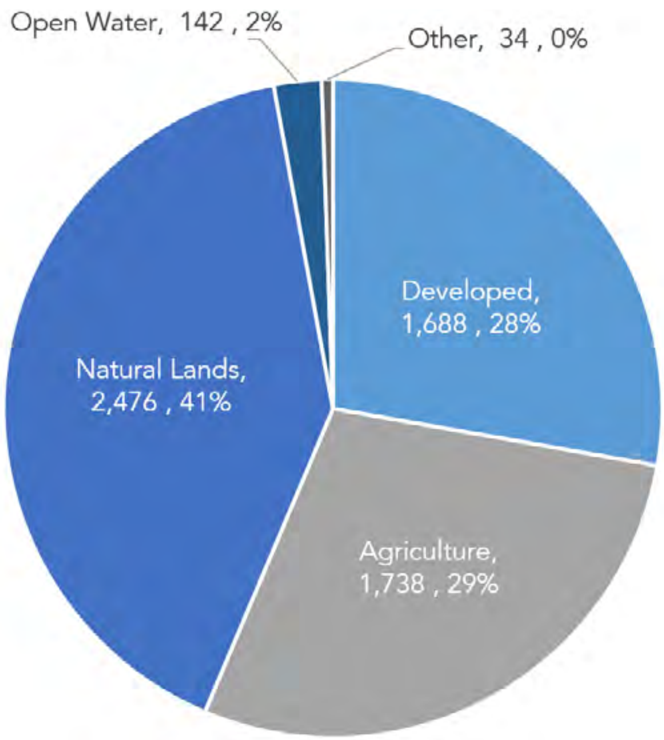


Figure 1.2.6 Agricultural land cover in the Delaware River Basin (2016).



A.



B.

Figure 1.2.7 Land cover proportions of the Upper Basin (A) and Estuary (B) in 2016. Values next to cover class represent land cover in square miles, followed by the percent of each cover class relative to the total land area for each region.

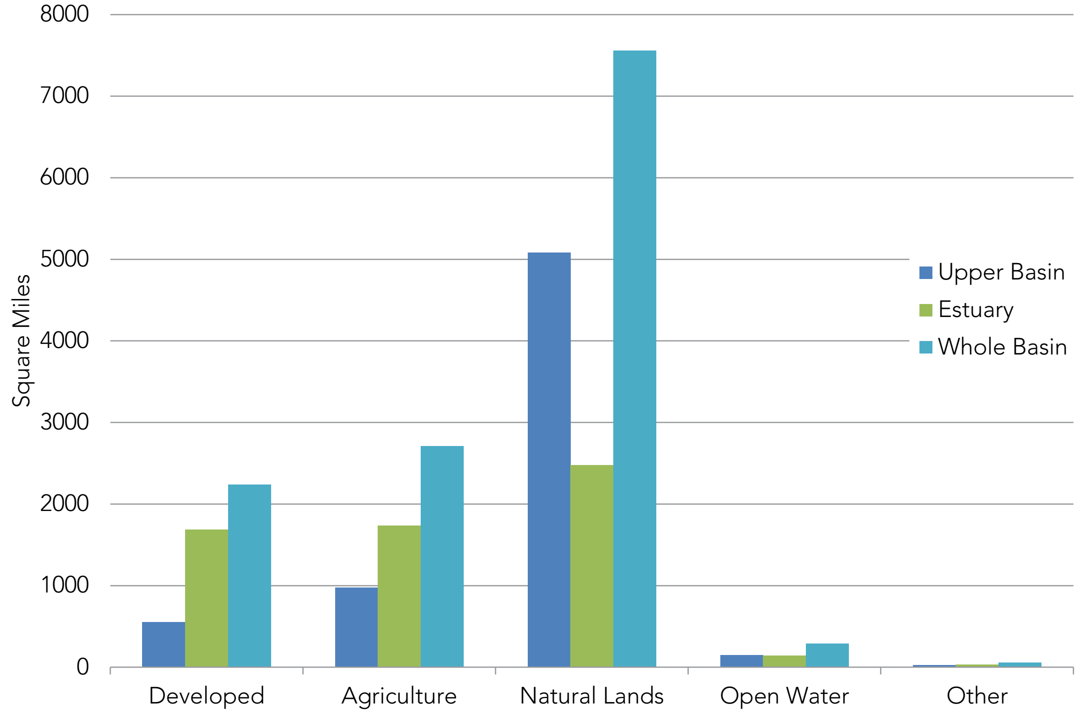


Figure 1.2.8 Land cover in the Delaware River Basin.

Table 1.2.4 Land cover by region and watershed in the Delaware River Basin.

Region	Watershed	Developed		Agriculture		Natural Lands		Open water		Other		Total
		mi²	%	mi²	%	mi²	%	mi²	%	mi²	%	mi²
Upper	East-West Branch	62	3.1%	231	11.4%	1,698	83.7%	32	1.6%	6	0.3%	2,029
	Lackawaxen	27	4.5%	74	12.4%	473	79.2%	23	3.8%	1	0.1%	597
	Neversink-Mongaup	41	5.0%	26	3.1%	721	88.4%	26	3.2%	2	0.3%	816
Central	Lehigh Valley	228	16.7%	234	17.2%	866	63.7%	23	1.7%	10	0.7%	1,361
	Upper Central	148	9.7%	241	15.8%	1,099	72.0%	35	2.3%	4	0.3%	1,527
	Lower Central	48	10.6%	170	37.4%	226	49.8%	9	2.0%	1	0.2%	454
Lower	Schuylkill Valley	432	22.8%	601	31.8%	827	43.7%	18	1.0%	12	0.7%	1,891
	Upper Estuary	792	45.5%	294	16.9%	594	34.1%	53	3.0%	10	0.6%	1,743
	Lower Estuary	286	28.0%	350	34.4%	360	35.3%	22	2.1%	3	0.3%	1,020
Bayshore	Delaware Bay	178	12.5%	492	34.5%	696	48.9%	49	3.5%	8	0.6%	1,423
Upper Basin		554	8.2%	976	14.4%	5,085	74.9%	147	2.2%	24	0.3%	6,784
Delaware Estuary		1,688	27.8%	1,738	28.6%	2,476	40.7%	142	2.3%	34	0.6%	6,077
Basin Total		2,241	17.4%	2,713	21.1%	7,561	58.8%	289	2.2%	57	0.4%	12,862

Past Trends

Changes in the landscape have generally been a progression of natural to more "human-influenced" land uses. Certainly, much of the trajectory of land cover in the Estuary and Basin, particularly in the southern portion, has been from natural cover to urban/suburban and agricultural uses. Land cover, including natural lands and agriculture, thus tends toward development and lower intensity development toward higher intensity development. Another trend seen throughout many portions of the Estuary and Basin is the displacement of agriculture farther from urban and population centers, as existing agricultural lands are progressively developed. Previously forested land is then claimed for agriculture, as the focus of food production shifts away from the higher land rents of the urban cores.

Prior to urbanization and following the period of early European colonization in the early 17th century (Kauffman 2010), much of the land of the Estuary and Basin had been heavily forested. While the Lenni Lenape, the indigenous people of the Estuary and Basin, cleared some land using fire or other means, the preponderance of watersheds was largely natural. By the early 20th century, natural cover was at its lowest ebb due to urban and agricultural expansion and colonial resource exploitation. Some of this trend has reversed due to economics, regulation, and land-use policies, but the pace of development has increased, especially in the previously less-populated areas of the upper watersheds in the Lehigh and Schuylkill Valleys and the Delaware Bayshore.

The Upper Basin has been and continues to be predominated by forest cover, despite more recent trends in recreational uses and extraction such as hydraulic fracturing ("fracking") for natural gas (although high volume hydraulic fracturing was recently banned within the Delaware River Basin, see the Final Rule, February 2021). The driving forces of land use change in our watersheds are based on competition for resources and space. With a limited amount of ground and an increasing population, this trend will likely continue for some time.

Analysis of the C-CAP land cover data for the past twenty years Basin-wide shows that since 1996:

- Approximately 210 square miles were converted to developed land, a 10.3% increase.
- One hundred and six square miles of agricultural land was converted to another use (mostly developed land, but some to forest), a net loss of 3.8%.
- Nearly 94 square miles of forested land was lost over the period, of 1.5% of the total.
- The Basin lost 5.7 square miles of freshwater and tidal wetlands (a net loss of 0.5%).

Figure 1.2.9 shows the overall changes, in square miles, in the Basin as well as in the Non-tidal (upper Basin) and Estuary portions. Figure 1.2.10 summarizes the changes within each region of the Delaware Estuary and Basin between 1996 and 2016. Developed land increased considerably over the period, focused largely on the Estuary portion of the Delaware Estuary and Basin over the period 1996 to 2016. Much of this expansion came as agriculture was converted to developed land, which occurred almost exclusively in the Estuary. Natural land cover loss over the period was more evenly distributed throughout the extent of the Delaware Estuary and Basin. Figure 1.2.11 presents the percentage changes in land cover types by region in the Delaware River Basin. Figures 1.2.12 and 1.2.13 show maps of the percent change in developed and agricultural land by region, respectively.

See Table 1.2.5A for a summary of net changes (mi^2) and Table 1.2.5B for percentage changes within the regions of the Basin and the Basin overall.

Figure 1.2.9 Land cover change in the Delaware River Basin (1996-2016).

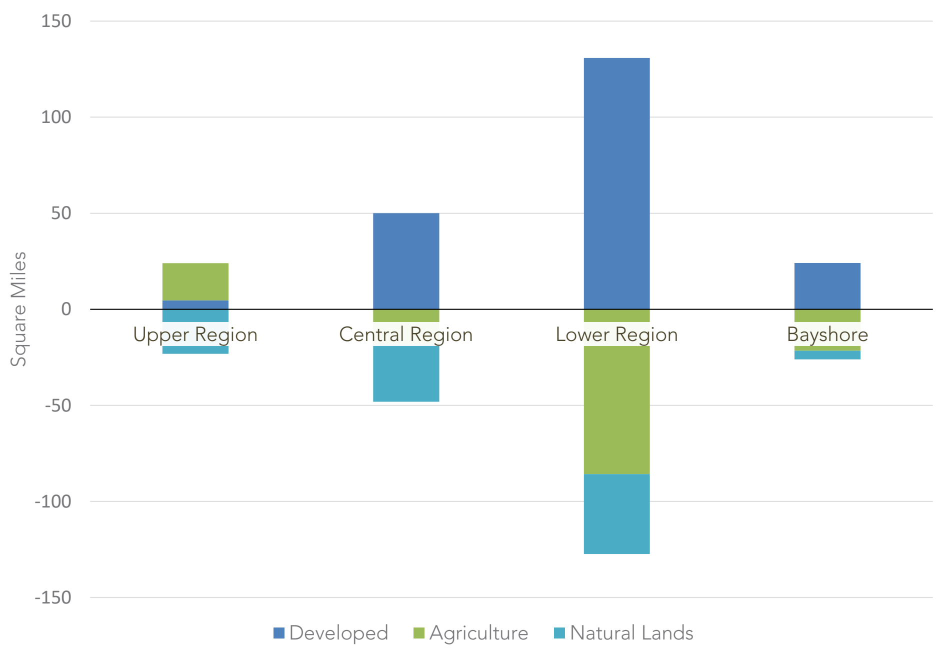


Figure 1.2.10 Land cover change (mi^2) by Region in the Delaware River Basin (1996-2016).

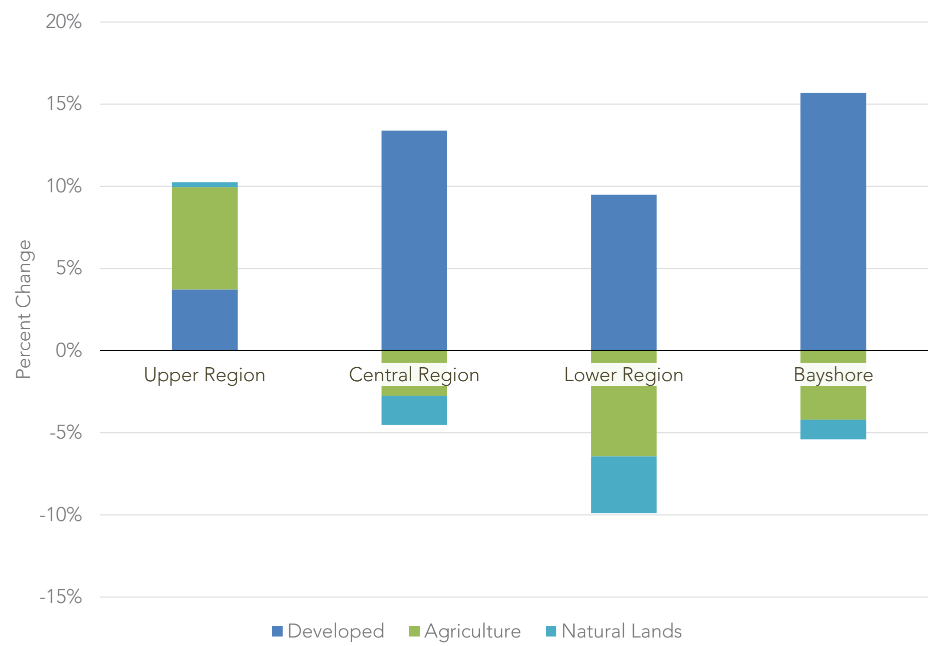


Figure 1.2.11 Land cover change (%) by Region in the Delaware River Basin (1996-2016).

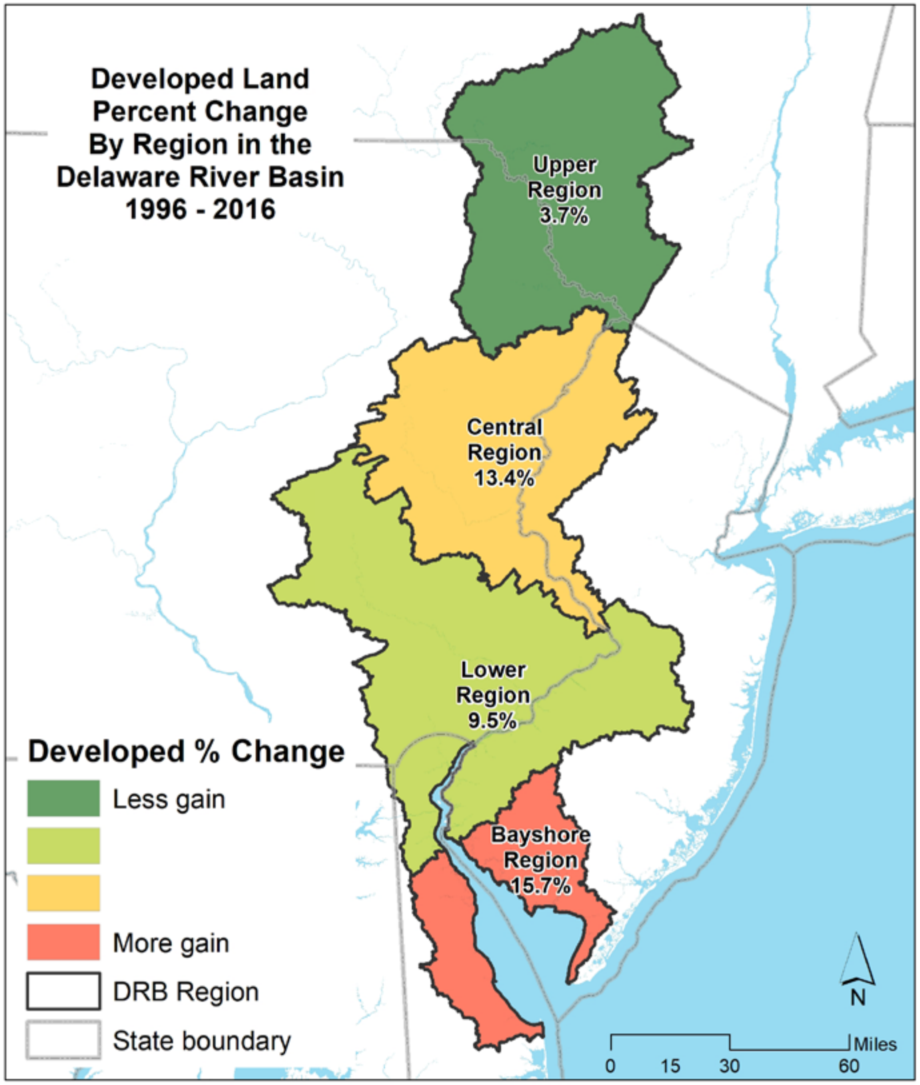


Figure 1.2.12 Developed land cover change (%) in Regions of the Delaware River Basin.

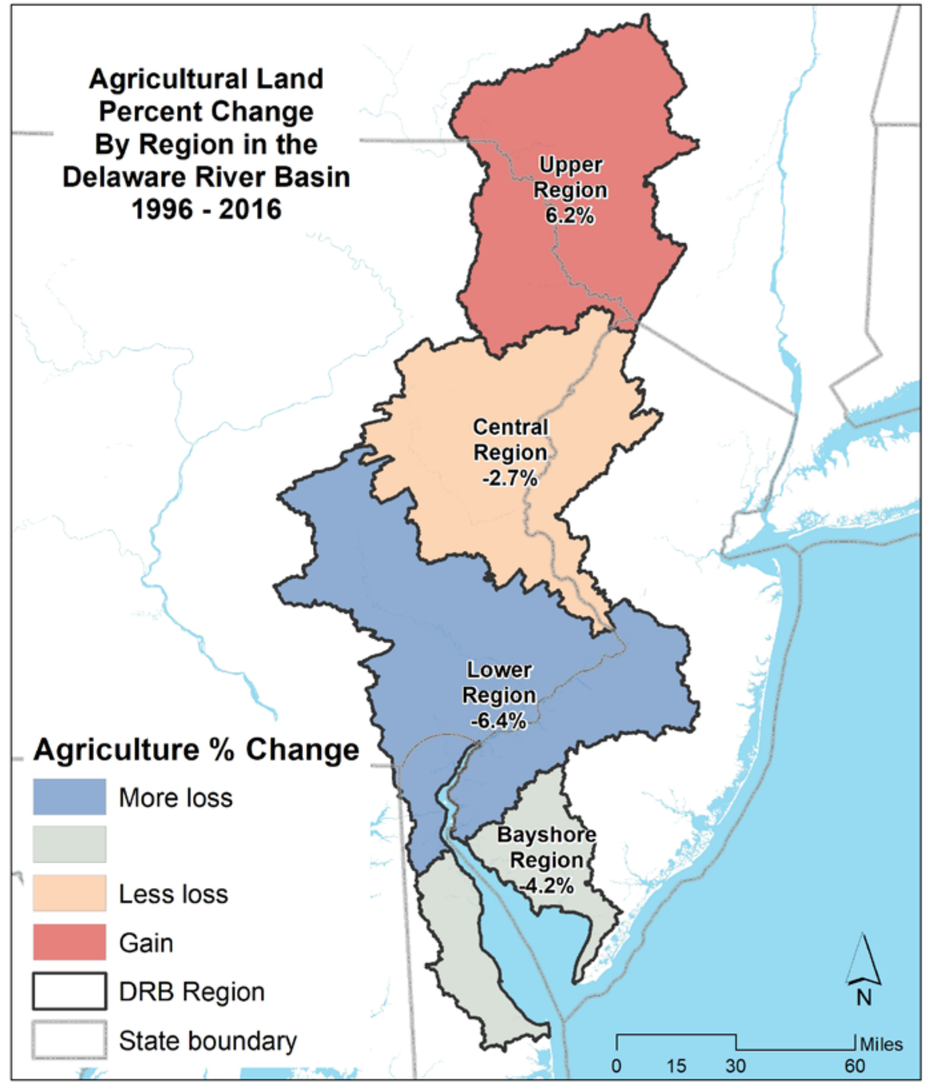


Figure 1.2.13 Agricultural land cover change (%) in Regions of the Delaware River Basin.

Table 1.2.5 Summary of net changes (mi^2, A, and %, B) within the Regions of the Delaware River Basin and the Basin overall from 1996-2016.

		Developed	Agriculture	Natural lands	Open Water	Other
A.	Upper Region	4.7	19.4	-23.2	-1.1	0.2
	Central Region	50.1	-18.2	-29.9	-0.0	-1.9
	Lower Region	130.8	-85.7	-41.6	0.5	-4.0
	Bayshore	24.1	-21.6	-4.5	1.8	0.2
	Basin Total	209.7	-106.1	-99.2	1.1	-5.5
B.	Upper Region	3.7%	6.2%	0.3%	-1.3%	2.3%
	Central Region	13.4%	-2.7%	-1.8%	-0.1%	-11.3%
	Lower Region	9.5%	-6.4%	-3.5%	0.6%	-13.8%
	Bayshore	15.7%	-4.2%	-1.2%	3.7%	1.8%
	Basin Total	10.3%	-3.8%	-1.9%	0.4%	-8.8%

Land Cover Changes by Watershed

The overall increase in developed area in the period between 1996 and 2016 is evident, particularly in the watersheds of the upper portions of the Estuary, in the greater Philadelphia region (Schuylkill Valley, Upper and Lower Estuaries). Significant development is also occurring in more outlying suburbanizing watersheds including the Lehigh Valley, Upper Central, and Lower Estuary. The Delaware Bayshore is also seeing development pressure. Agricultural loss in the period 1996 to 2016 is most significant in the watersheds of the Estuary, which are also seeing the highest amount of newly developed land.

Figure 1.2.14 illustrates the net change in land cover type across the basin by the ten watershed groups arranged north to south (reading left to right), between 1996 and 2016. Figure 1.2.15 shows the same information as a percentage. The maps in Figure 1.2.16 and 1.2.17 show the changes, as a percentage, across the period 1996 to 2016, by the 21 watershed of the Delaware Estuary and Basin for developed and agricultural lands, respectively. Table 1.2.6A summarizes the land cover changes (in square miles) in the Delaware Estuary and Basin between 1996 and 2016 (the shaded portion of the table indicates watersheds that are part of the Delaware Estuary).Table 1.2.6B shows the percentage change in the watersheds of the Basin, as well as the Upper (non-tidal) and Estuary portions of the Basin.

Rate of Land Cover Change

Rates of change in land cover types, including developed land, agriculture, and natural lands, also varied across the time period from 1996 to 2016. Rate of change for developed land peaked in the period between 2001 and 2006, as did the rate of agricultural loss (4.0% and -2.1%, respectively). Each subsequent period saw a smaller change, as a percentage, of those two land cover types. Figure 1.2.18 shows the rates of changes as a percentage of each of the three land cover types for each 5-year period between 1996 and 2016.

The largest change (total across the watershed) is Agriculture to Forest (34%) (Fig 1.2.19 and 1.2.20). This was likely caused by succession, as scrub or fallow farmland reverts to trees, and the same to some extent for pasture/grasslands (these may or may not have actually been cultivated in 1996, but still lumped into the "Agricultural" class). The next largest change was Agriculture to Developed (31%).

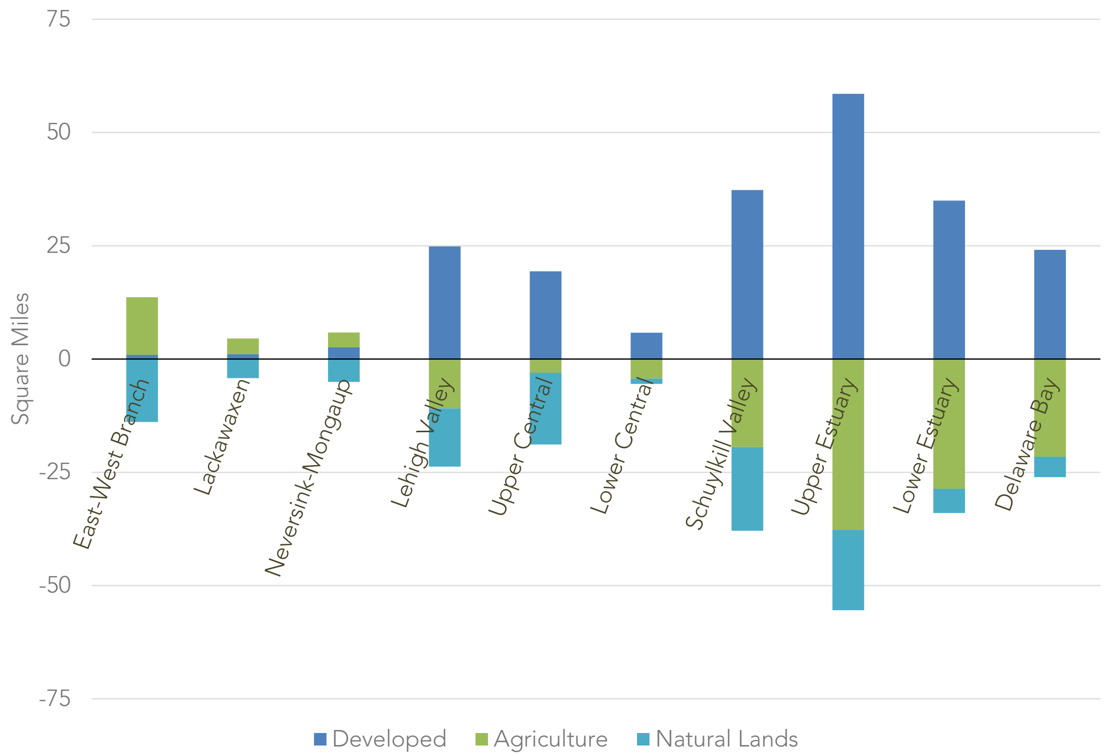


Figure 1.2.14 Land cover change (mi²) by subregion in the Delaware River Basin (1996-2016).

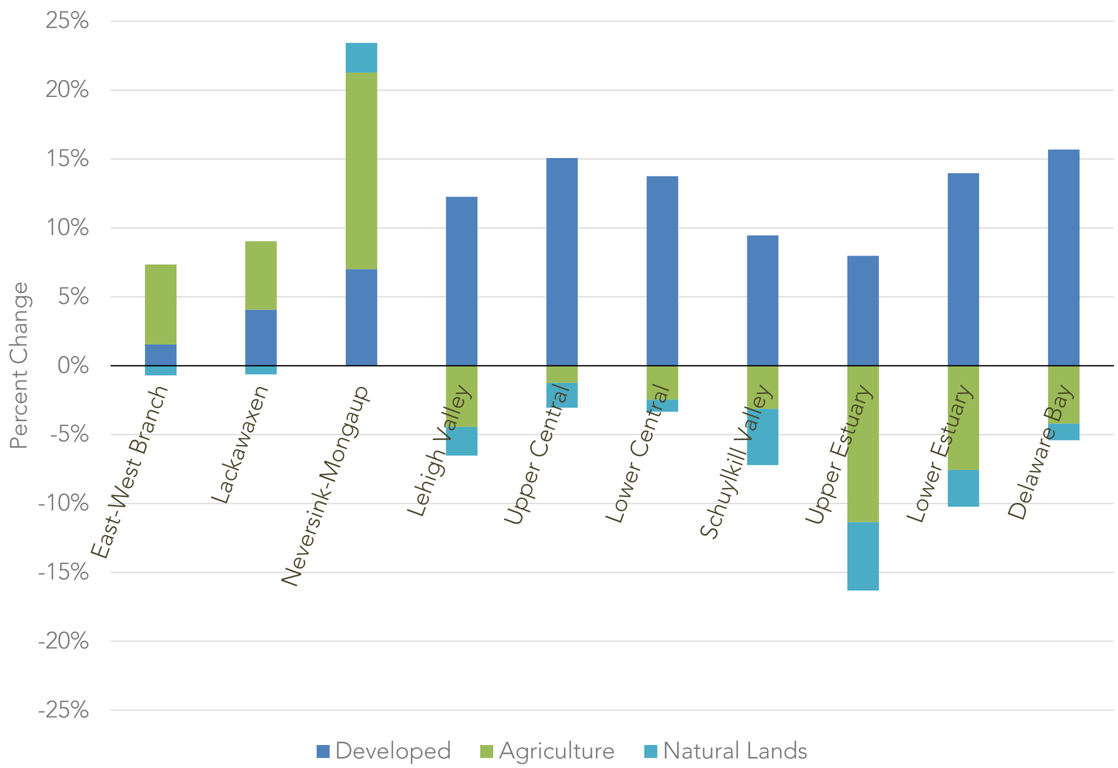


Figure 1.2.15 Land cover change (%) by subregion in the Delaware River Basin (1996-2016).

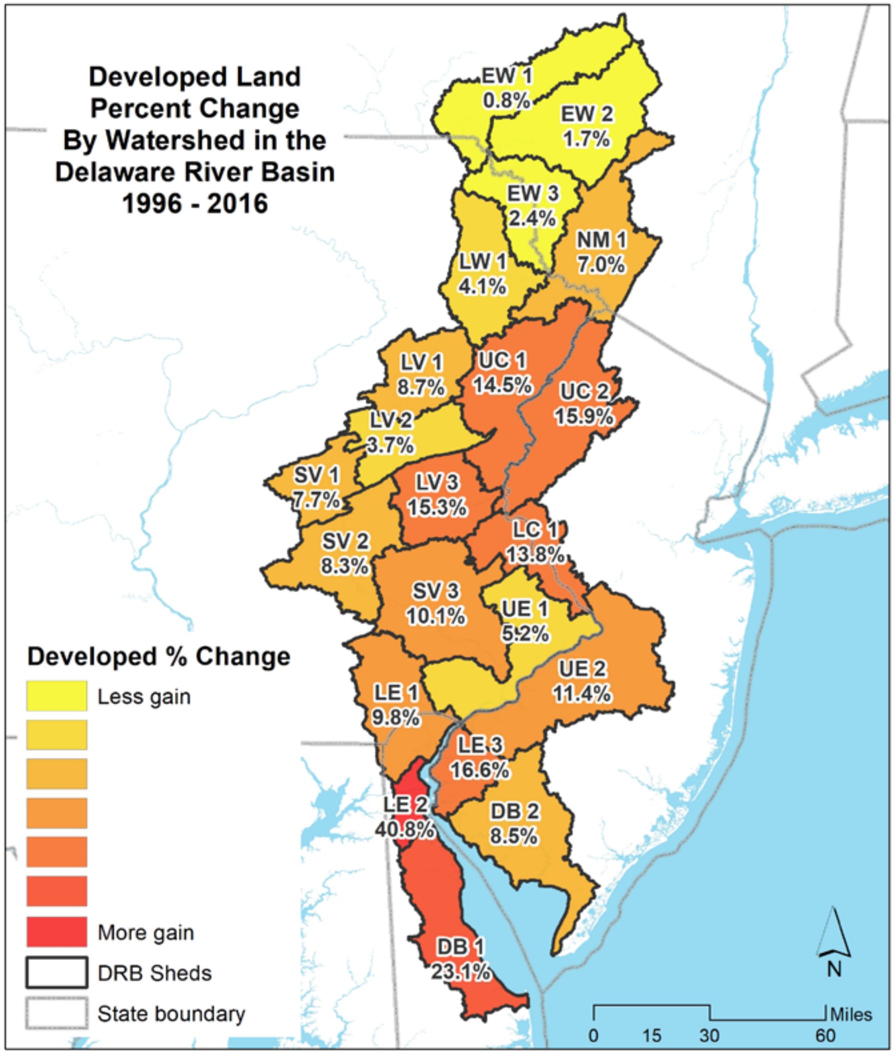


Figure 1.2.16 Developed land cover change (%) by watershed in the Delaware River Basin (1996-2016).

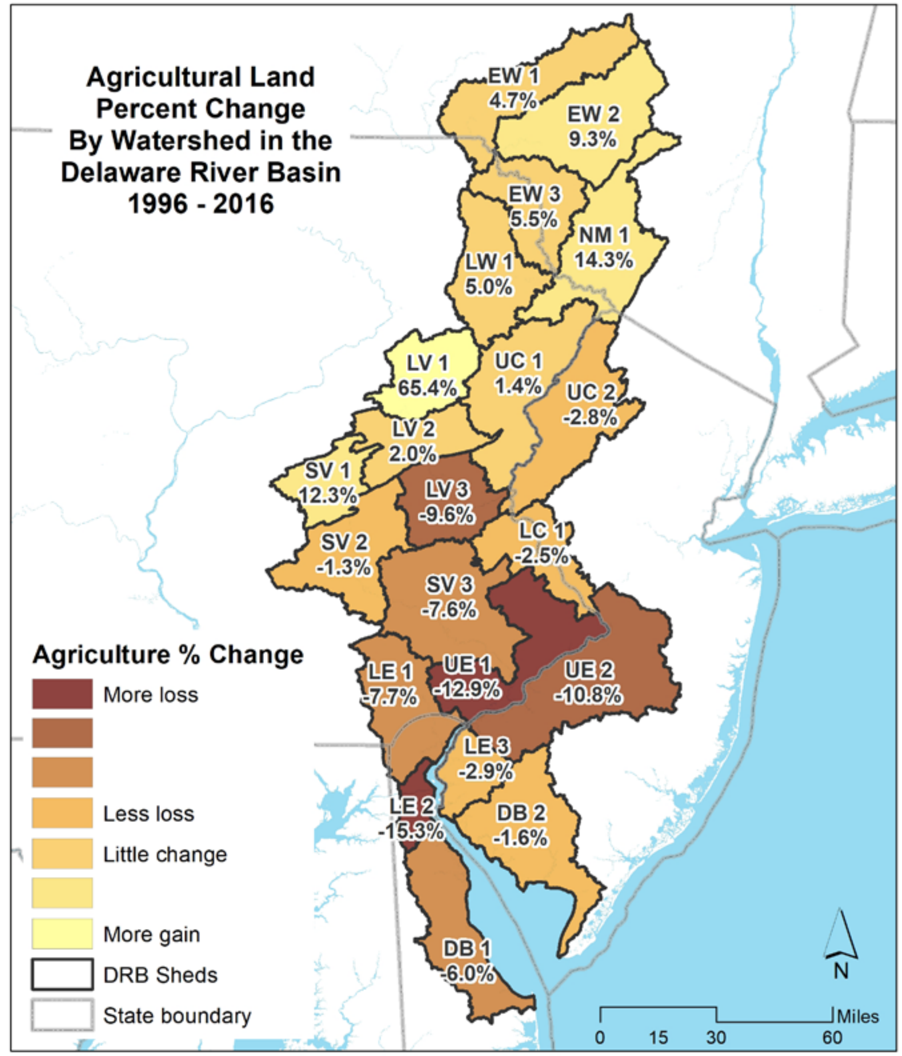


Figure 1.2.17 Agricultural land cover change (%) by watershed in the Delaware River Basin (1996-2016).

Table 1.2.6 Land cover change (mi², A, and %, B) by region and watershed in the Delaware River Basin (1996-2016).

	Region	Name	Developed	Agriculture	Natural Lands	Open Water	Other
A.		East-West Branch	0.9	12.7	-13.9	0.6	-0.4
	Upper Region	Lackawaxen	1.1	3.5	-4.2	-0.5	0.1
		Neversink-Mongaup	2.7	3.2	-5.1	-1.3	0.4
		Lehigh Valley	24.9	-10.9	-12.9	0.8	-1.9
	Central Region	Upper Central	19.4	-3.0	-15.9	-0.8	0.3
		Lower Central	5.8	-4.3	-1.2	-0.0	-0.3
		Schuylkill Valley	37.3	-19.4	-18.5	0.9	-0.3
	Lower Region	Upper Estuary	58.5	-37.7	-17.8	-0.3	-2.8
		Lower Estuary	35.0	-28.6	-5.4	-0.1	-0.9
	Bayshore Region	Delaware Bay	24.1	-21.6	-4.5	1.8	0.2
	Upper Basin		54.7	1.2	-53.1	-1.1	-1.7
	Delaware Estuary		155.0	-107.3	-46.1	2.3	-3.8
	Whole Basin		209.7	-106.1	-99.2	1.1	-5.5
B.		East-West Branch	1.5%	5.8%	-0.7%	2.0%	-6.1%
	Upper Region	Lackawaxen	4.1%	5.0%	-0.6%	-2.1%	25.9%
		Neversink-Mongaup	7.0%	14.3%	2.2%	-4.6%	26.6%
		Lehigh Valley	12.3%	-4.4%	-2.1%	3.6%	-16.0%
	Central Region	Upper Central	15.1%	-1.2%	-1.8%	-2.3%	8.0%
		Lower Central	13.8%	-2.5%	-0.9%	-0.2%	-23.0%
		Schuylkill Valley	9.5%	-3.1%	-4.1%	5.2%	-2.3%
	Lower Region	Upper Estuary	8.0%	-11.4%	-5.0%	-0.6%	-21.9%
		Lower Estuary	14.0%	-7.6%	-2.7%	-0.3%	-24.7%
	Bayshore Region	Delaware Bay	15.7%	-4.2%	-1.2%	3.7%	1.8%
	Upper Basin		11.0%	0.1%	-0.9%	-0.8%	-6.7%
	Delaware Estuary		10.1%	-5.8%	-3.1%	1.6%	-10.3%
	Whole Basin		10.3%	-3.8%	-1.9%	0.4%	-8.8%

Future Predictions

Currently, the watershed is characterized by relatively undeveloped, forested land to the north in the headwaters of the Delaware River, with increasing development moving down the main stem and relatively less development along the less-populous Bayshore. Agriculture is prevalent in the more coastal watersheds and in the regions surrounding the urban Philadelphia core. In the Bayshore (DB 1, 2), Lower Estuary (LE 1, 2, 3), Schuylkill Valley (SV 1, 2, 3), and Lower Central (LC 1) watersheds agriculture is approximately one-third of the total land cover. As land development often correlates or sometimes outpaces population growth (Gao and O'Neill 2021), it is likely that areas of existing development will densify, with agricultural and forested land converting to more developed land cover as populations in the Delaware Estuary and Basin increase (see Section 1.1).

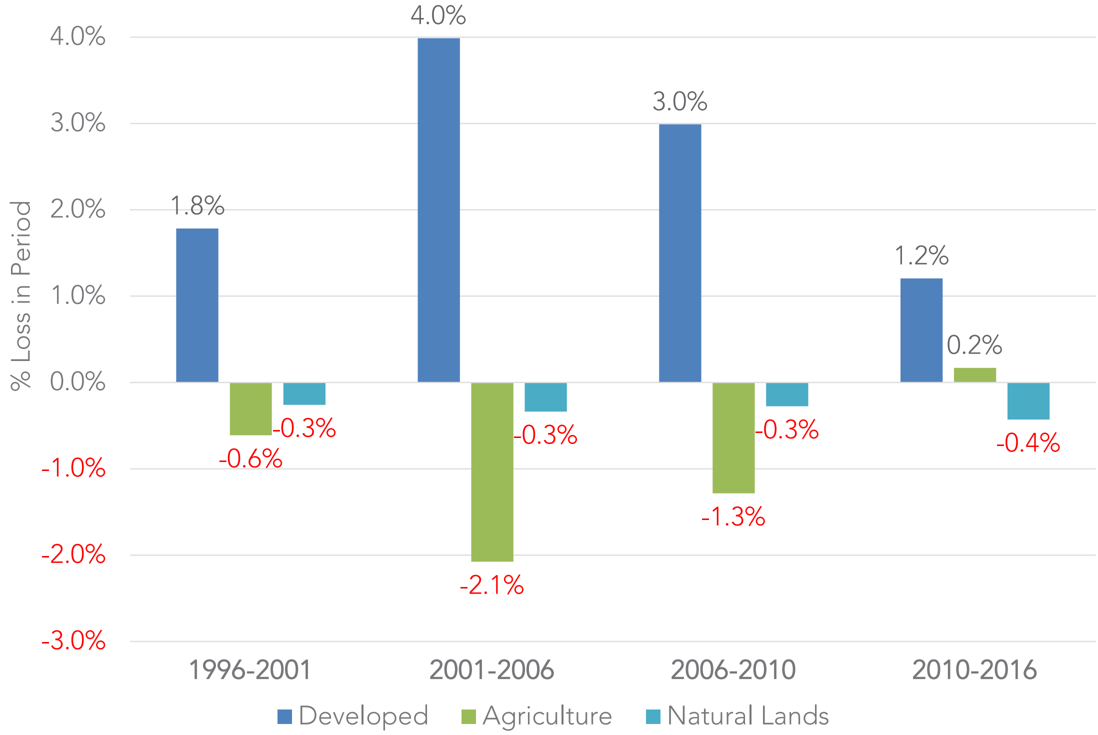


Figure 1.2.18 Rate of land cover change (% per 5-year increment) in the Delaware Estuary and Basin (1996-2016).

Net land cover change in the landscape occurs in conjunction with other trends, including demographic, economic, environmental, and regulatory changes. For example, periods of economic expansion can lead to a surge in development pressure and population expansion, which adds to residential and commercial development. Increased residential and commercial development often includes the conversion of agricultural land to houses, businesses, roads, and other uses. Climate-related effects can also affect the character of land cover. As wetlands are lost, urban areas respond to increased flooding risks and successional changes occur with changing temperature regimes. The regulatory framework at the local to the national level, including zoning ordinances, and environmental constraints (for instance as enforced through the Clean Water Act) can have a significant effect over time on the overall character of land cover. Strong planning and management strategies implemented at the local and regional scale can also help ameliorate some of the negative impacts of uncontrolled changes in the Basin's watersheds.

It is likely that many of the trends being felt in the Delaware Estuary and Basin will continue, with one of the most significant drivers being the overall economic environment. The previous decade saw a significant slow-down in economic activity with the 2008-2009 recession, but also saw a fairly robust recovery in the past several years. The change in the economic outlook and the strong pent-up demand has led to a building boom in many areas. The latest data from 2016 has perhaps not fully captured this trend, but in the next several years it is possible development pressures will increase.

Demographically, the rate of population increase has leveled off, and in some areas, begun to decline. In line with trends nationally, it is likely that population pressures will become less intense so that the impacts that come with population increase will moderate. In conjunction with effective and environmentally sound policies and regulations at the local, state, and federal levels, overall watershed health in the Estuary and Basin is possibly going to stabilize and possibly improve in coming years.

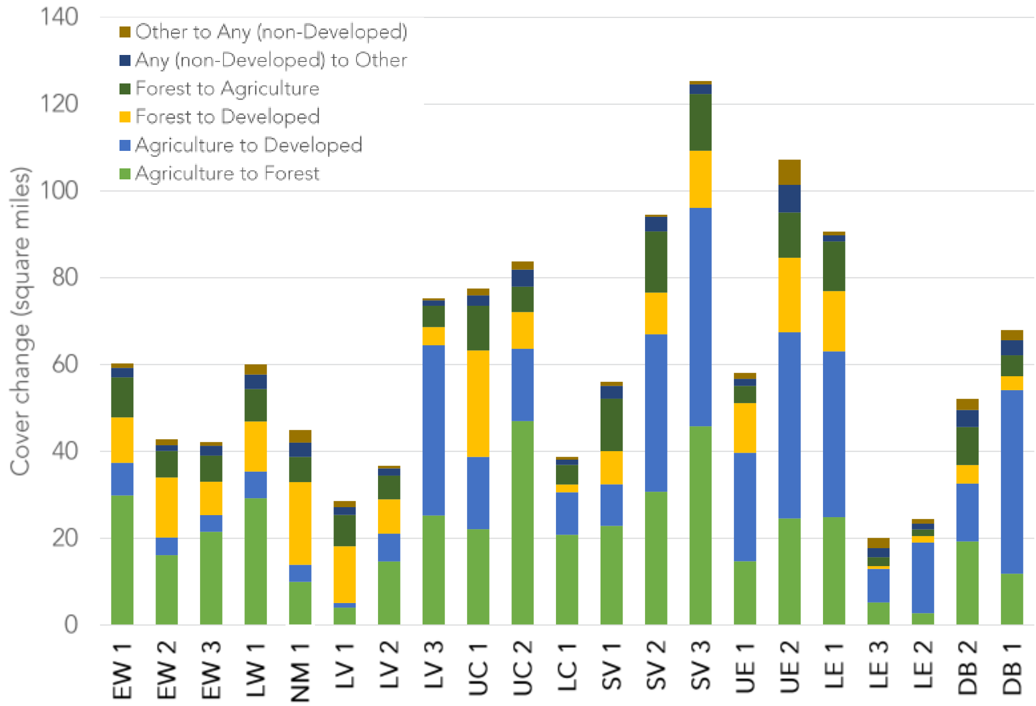


Figure 1.2.19 Cover class changes by broad category in the Delaware Estuary and Basin watersheds from 1996 to 2016.

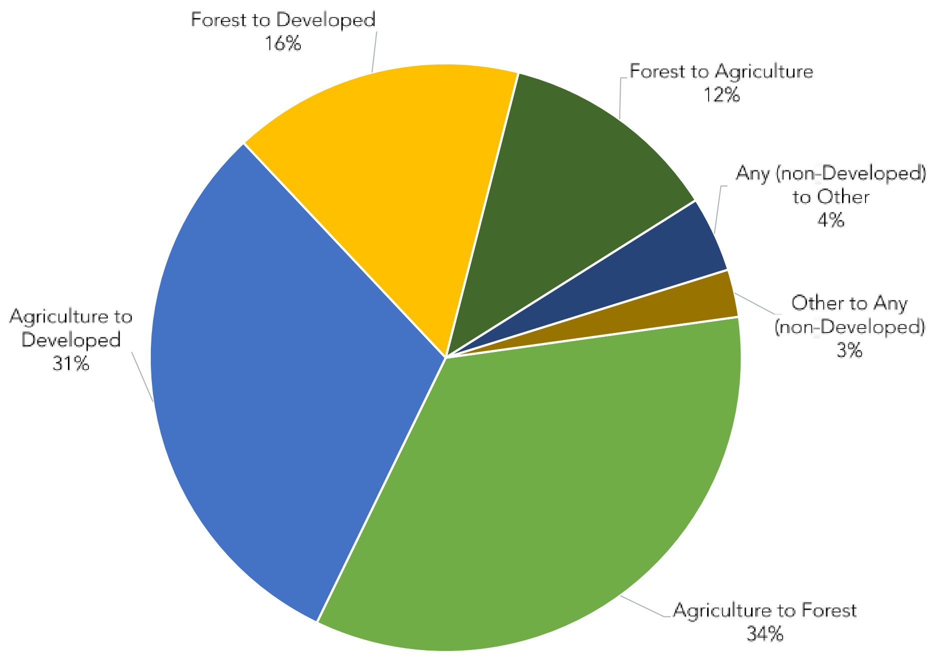


Figure 1.2.20 Total cover class changes by broad category in the Delaware Estuary and Basin from 1996 to 2016.

Actions and Needs

Land cover is one of the most significant factors in the overall health and water quality of the Delaware Estuary and Basin's watersheds. It is therefore important to protect the natural portions of the watersheds, particularly those areas experiencing the highest expected growth in population and developed land uses. Regional planning is a key component of protecting the highly variable landscapes of the watershed from degradation. Given the many diverse governmental and regulatory structures within the Estuary and Basin, centralized coordination is key. Organizations such as the Partnership for the Delaware Estuary and the Delaware River Basin Commission (DRBC), along with private and non-profit partners (especially watershed organizations), are crucial for protecting the resources that impact water quality, watershed health, and human and environmental well-being. Coordinating among these, through efforts such as the Delaware River Watershed Initiative, a multi-year, multi-state collaborative addressing water quality in the Delaware Estuary and Basin, can be an important conduit for the sharing of ideas and coordination of activity.

Implementation in urbanized areas, where watersheds are highly impacted by development, of practices designed to remove pollutants and ameliorate flooding and runoff are components of an overall watershed protection strategy. Agricultural practices designed to remove potential pollutants before they enter the waterways are also crucial to fostering water quality and watershed health. Finally, protecting the natural areas of the watershed, especially forest buffers and wetlands, helps those areas perform functions that serve to protect streams and rivers, and the overall health of the watershed.

- Coordinate with federal and state agencies to maintain and improve geospatial and other data collection, maintenance, and dissemination.
- Focus on highly critical land cover types, such as wetlands and forest, in coordinating the collection methods, sensors, attributes, and timing of data. In particular, focus on C-CAP and the National Wetlands Inventory to ensure spatial and temporal compatibility.
- Identification and inventory of forested areas critical to water resources and habitat.
- Identification and prioritization of the threats and opportunities for watershed protection across the Estuary and Basin, recognizing that these will vary widely across the region.
- Inventory and understanding of land use regulatory frameworks across the Delaware Estuary and Basin.
- Support regional approaches to watershed management and strategic planning efforts to protect natural land cover, in particular in riparian corridors.
- Bolster coordination among local, state, regional, and federal regulatory agencies, local watershed groups and other non-profits, and the academic community. Increased coordination could include outreach to local partners by regional watershed organizations (such as the PDE), in particular to help act as liaison to local, state, and federal regulators, such as the USEPA and state departments of environmental protection, as well as to provide direct technical support.
- Inventory funding opportunities for land protection in the service of clean water and watershed protection.
- Support for robust and comprehensive monitoring of progress and trends to inform decision makers and enable program assessments.

Summary

Land cover in the Delaware Estuary and Basin is highly variable among Regions. Since what occurs on the land upstream will have a direct impact on all downstream lands, it is fortunate that many of the headwaters in the watersheds of the Basin, including the main stem of the Delaware River, are highly natural and protected. Towards the south, development and human density increase, peaking in the watersheds of the Philadelphia conurbation. Upstream in the tributaries of the main stem of the Delaware River many watersheds are characterized by agriculture and smaller urban centers, particularly in the Schuylkill Valley, Lehigh Valley, and the Lower Estuary in southeast Pennsylvania. The Bayshore is also agricultural, with large areas of tidal wetlands, but there are also development and population pressures, which are likely to increase in coming years.

Nearly 210 square miles within the Delaware Estuary and Basin was developed over the period of 1996-2016, at a rate of 17.5 acres per day. While the Basin lost over 106 square miles (8.9 acres per day) of agricultural land, over 99 square miles (~8 acres per day) of natural land. Table 1.2.7 presents the changes in each major land cover class across the 20-year period between 1996 and 2016, in square miles (total and annually) and acres on a daily basis.

Table 1.2.7 Daily change in each land cover class across the 20-year period between 1996 and 2016.

Land Cover Type	Net change (mi^2)	Annual change (mi^2/yr)	Daily rate	
			mi^2/day	acre/day
Developed	209.68	9.98	0.0274	17.5
Agriculture	-106.09	-5.05	-0.0138	-8.9
Natural Land	-99.20	-4.72	-0.0129	-8.3
Open Water	1.15	0.05	0.0001	0.1
Barren/Bare	-5.54	-0.26	-0.0007	-0.5

Developed land increased in every watershed between 1996 and 2016; while natural lands declined. Agricultural land also declined in total area, with increases only in the northern watersheds of the Basin. The Estuary portion of the watershed experienced the highest increase in development (nearly 74% of all increase), about half of the natural land loss, and nearly all the agricultural losses experienced across the Basin. The Upper (non-tidal) portion of the Delaware Estuary and Basin experienced more net natural land loss than the Estuary over the period, but a smaller amount in percentage terms.

The watersheds of the Lower Region experienced the greatest increases in developed land as well as the most loss of agricultural and natural lands. As a percentage increase, however, both the Central and Bayshore Regions saw more intensive development pressure. The conversion of natural land to either agriculture or developed land is the most significant trend in terms of water quality and watershed health. Natural land changes are variable throughout the Delaware Estuary and Basin, but losses in the upper (non-tidal) portions of the Basin have increased in the latest period of available data (2010 -2016) and represent a larger value in square miles (albeit not percent) than the loss in the Estuary.

Land cover, as considered here, can be a helpful indicator of overall health of the Delaware Estuary and Basin at the landscape scale. When considering land cover impacts at higher levels of detail it can be helpful to consider other factors and data sources such as higher-resolution land us, topography, transportation networks, and crop or forestry information, as well as ground-verified data.

1.3 Impervious Cover

Description of Indicator

Impervious cover is defined as features on the ground that prevent water from infiltrating, causing it to run off. Imperviousness as a watershed metric is the measure of the degree to which an area of the ground is covered by such features. Examples of impervious cover include roads, parking lots, rooftops, and any other hard or impermeable surfaces. Impervious cover disrupts the normal hydrologic cycle, in which a portion of atmospheric precipitation is able to percolate into the ground and help recharge the groundwater table. Water that runs off rather than infiltrating the ground can cause problems with pollution entering streams and other waterbodies and more potential for flooding. High percentage of impervious cover leads to more polluted waters, streams that are more flood-prone, and lower streamflow in dry periods. Measuring imperviousness gives an indication of the overall health of a watershed.

As the percentage of impervious cover increases in a watershed, the overall stream quality tends to decline (Fig 1.3.1). Studies have shown that impervious cover in the landscape can negatively affect stream quality and watershed health at levels between 5% and 10% of the total area. Above the 10% threshold streams are considered "impacted", and when imperviousness exceeds 25% of the watershed area, streams are "non-supporting" in terms of adequate water quality, habitat, hydrology and channel stability (Schueler et al. 2009). A survey of 225 research projects assessing the correlation of degree of impervious cover stream and aquatic life condition compiled by the Center for Watershed Protection links the presence of impervious cover to a list of impacts, including: reduced macroinvertebrate and fish diversity; decline in biological function; increase in stream temperature; decline in channel stability and fish habitat; and compromised wetlands water quality and water level fluctuation (Schueler et al. 2009).

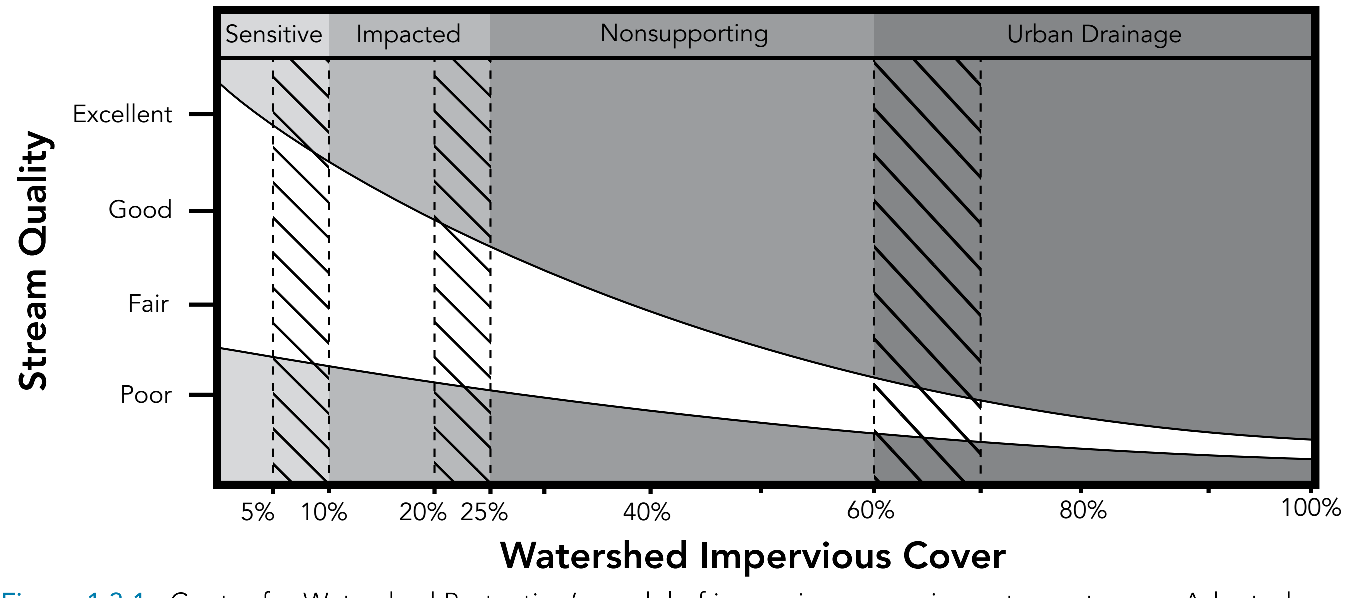


Figure 1.3.1 Center for Watershed Protection's model of impervious cover impacts on streams. Adapted from Schueler et al. (2009).

Data Source and Processing Methodology

Measures of impervious cover can be derived in several ways. Using satellite imagery and image processing techniques it is possible to determine the characteristic imperviousness on a given area of the ground. The USGS National Land Classification Database (NLCD) consists of a series of image products including land cover, forest cover, and imperviousness, based on a 30 by 30 meter ground resolution (pixel size). These data are available for the U.S. starting in 2001, every few years, through 2019. For this study the NLCD data for impervious cover for the years 2001, 2006, 2011, 2016, and 2019 were used to assess both the current level of imperviousness on a watershed basis, as well as the change in imperviousness by watershed over that time period.

Other methods include direct measurement of impervious cover through photo interpretation of aerial photography, or use of representative imperviousness values based on land cover classifications. This latter technique has been used in previous States of the Estuary and Basin reports. Specifically, the NOAA Coastal Change Analysis Program (C-CAP) data was used to apply a representative value of imperviousness based on land cover type. While this method provides a flexible, consistent, and easily replicable method for determining imperviousness on a watershed basis, the data is not available for dates later than 2016. Additionally, the NLCD data has the advantage of being specifically produced to quantify imperviousness in the continental U.S. A drawback is that the data are reported on a percentage basis, with each 30 by 30-meter pixel assigned a percentage score, rather than a fixed amount of impervious cover. However, this limitation was addressed by calculating for each pixel the predicted amount (area) of impervious cover within a given pixel.

Another potential limitation is that the data are developed only for urbanized areas; areas that do not comprise a non-natural land cover type are not considered in calculation of the impervious percentage. Therefore, though the data are well-suited for analysis at a watershed scale, they should not be used at scales requiring a high level of detail (e.g., the neighborhood scale). The consistency and temporal resolution (i.e., every few years), as well as the availability of recent data, made this the preferred data source for impervious analysis for this report.

Determination of the current status of imperviousness on a watershed basis was performed for the Delaware Estuary and Basin and Estuary based on the 2019 NLCD impervious cover dataset. To analyze change over time, data for 2001, 2011, and 2019 were also considered. The impervious layers for each of these years were used to calculate the total amount of imperviousness for each pixel of the datasets. The total amount of imperviousness by watershed was calculated by summing the amount of impervious within each watershed. Statistics were compiled at three scales: the USGS Hydrologic Unit Code 12 digit watershed (HUC12; the smallest watershed division available on a national basis), as well as the Delaware River Basin Commission designations consisting of ten watersheds in the Delaware Estuary and Basin, plus a further division into 21 smaller watersheds, see Table 1.3.1.

Present Status

Impervious cover in the Delaware Estuary and Basin varies dramatically across its 21 watersheds. Higher levels of development correlates to a higher degree of imperviousness. Based on 2019 data, watersheds in the greater Philadelphia area have the highest degree of imperviousness, with the watershed containing Philadelphia at more than 25% impervious, while those on the Delaware Bayshore and in the Central Region have somewhat lower values. The Upper Region is the least affected by impervious cover, with most values well under 2% (Figures 1.3.2, 1.3.3).

Impervious at the HUC 12 level shows the distribution of imperviousness at a higher resolution, with the area around Philadelphia showing the highest degree of imperviousness, with additionally high values in the I-95 corridor in Delaware, and in the upper reaches of the Schuylkill and the Lehigh River Valleys (Figures 1.3.2, 1.3.3).

Figure 1.3.4 presents the current (2019) imperviousness, by watershed in the Delaware Estuary and Basin, as a percent of land cover. Figure 1.3.5 shows the distribution of HUC 12s based on their percentage imperviousness in 2019. The red and orange lines in the graph indicate the level of imperviousness above which streams are considered "Nonsupporting" and "Impacted", respectively, according to the Center for Watershed Protection's Impervious Cover Model (ICM). Out of a total of 428 HUC 12s, 65 watersheds (15.2%) are above the 10% Impacted threshold, while 29 (6.8%) are Nonsupporting. In 2019, of the 21 watersheds, one (UE1) is above the threshold for being Nonsupporting, and an additional four (UE2, LE1, LV3, and SV3) are above the 10% Impacted impervious threshold.

Table 1.3.1 Watersheds, watersheds, and regions of the Delaware River Basin.

Part of the Basin	Region	Watershed	Watershed	Sq mi
Upper Basin	Upper	East-West Branch	EW 1	655.9
			EW 2	840.2
			EW 3	523.2
		Lackawaxen	LW 1	597.4
		Neversink-Mongaup	NM 1	815.8
	Central	Lehigh Valley	LV 1	451.2
			LV 2	460.1
			LV 3	479.3
		Upper Central	UC 1	778.3
			UC 2	744.8
		Lower Central	LC 1	453.7
Estuary	Lower	Schuylkill Valley	SV 1	342.0
			SV 2	655.7
			SV 3	893.3
		Upper Estuary	UE 1	701.0
			UE 2	1,042
		Lower Estuary	LE 1	603.0
			LE 2	154.9
			LE 3	262.3
	Bay	Delaware Bay	DB 1	634.0
			DB 2	789.3

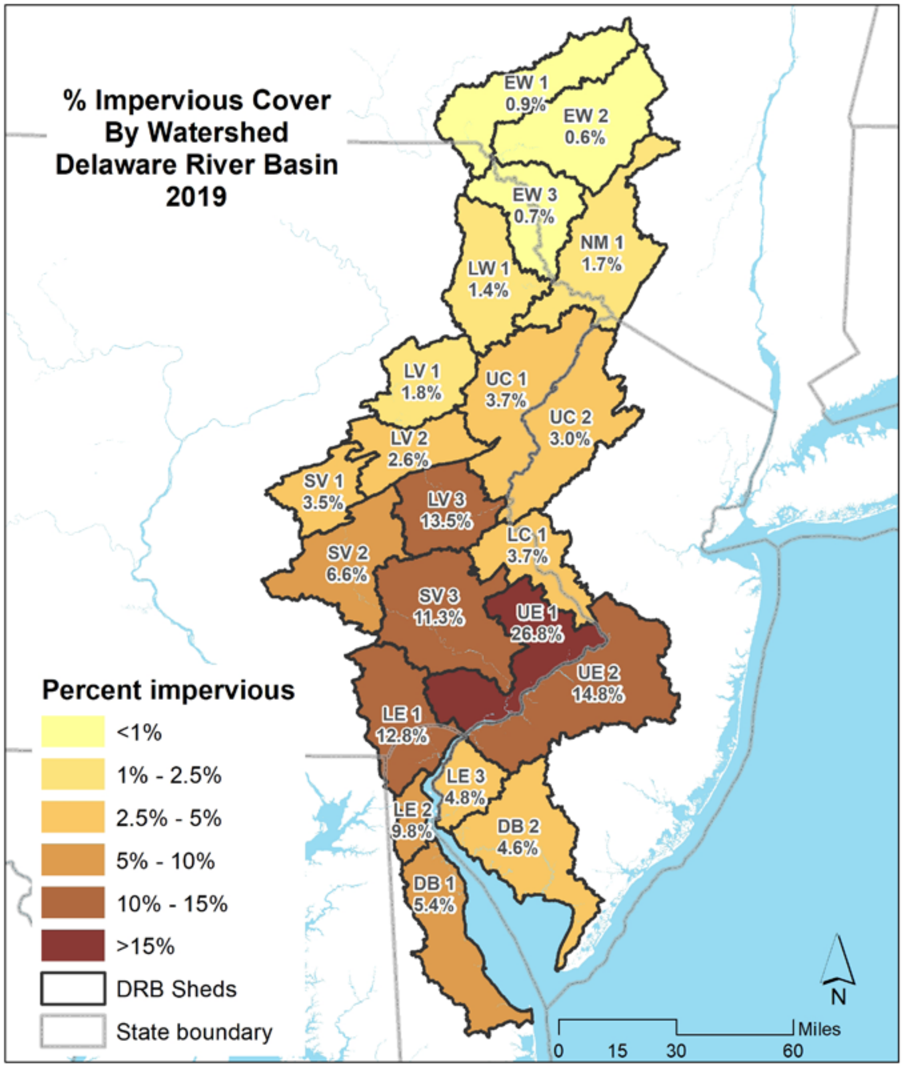


Figure 1.3.2 Percent impervious cover by the 21 watersheds in the Delaware River Basin (2019).

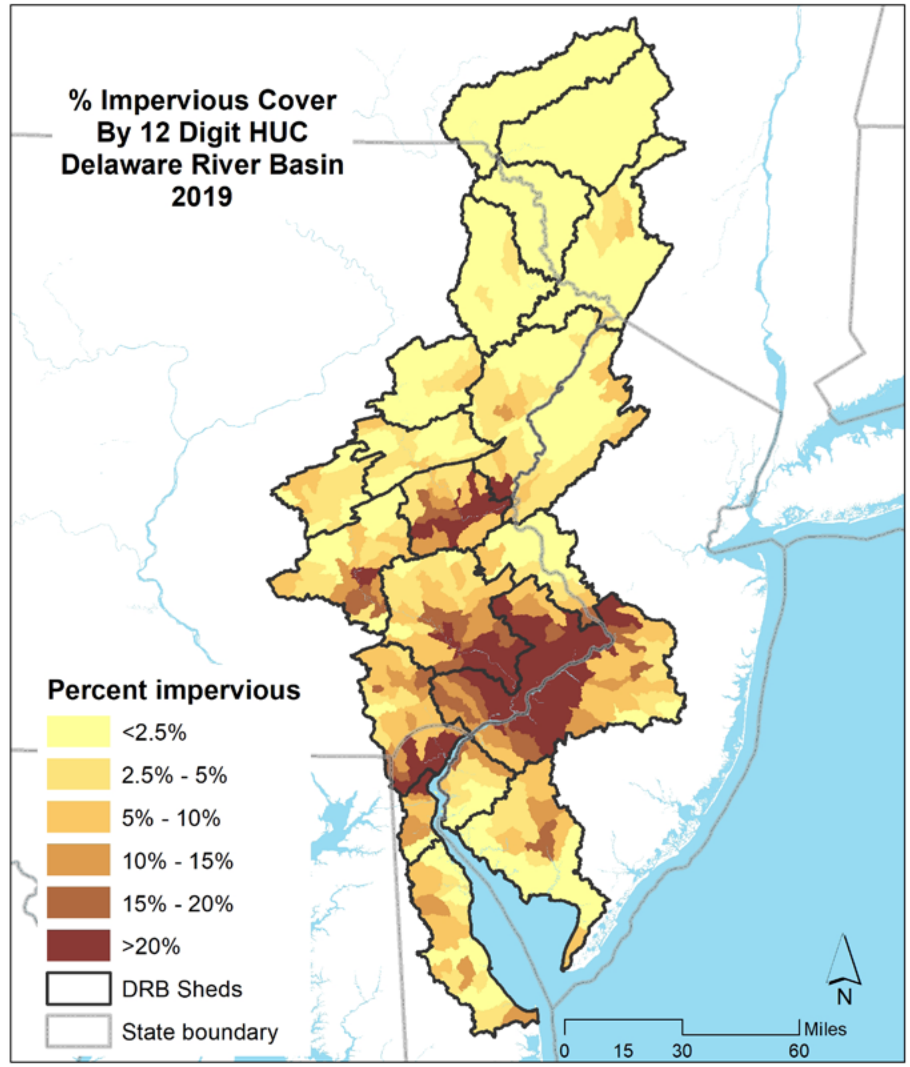


Figure 1.3.3 Percent impervious cover by 12-digit HUC watershed in the Delaware River Basin (2019).

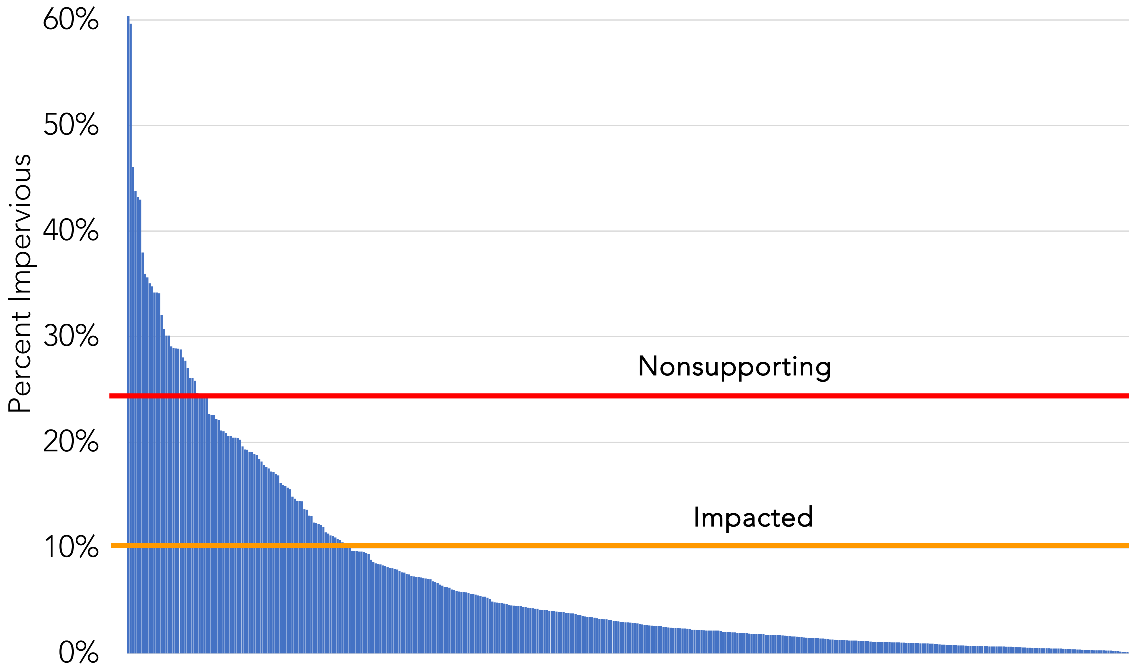


Figure 1.3.4 Histogram of percent impervious cover by 12-digit HUC watershed in the Delaware River Basin (2019).

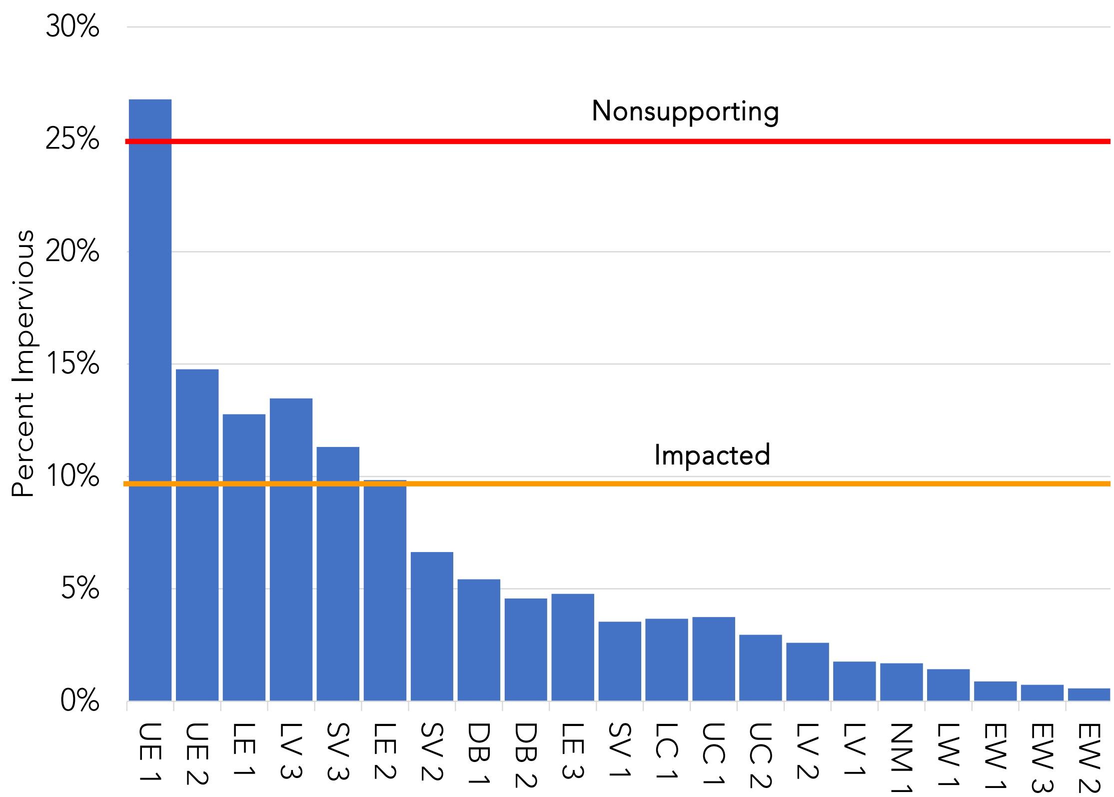


Figure 1.3.5 Histogram of percent impervious cover by watersheds in the Delaware River Basin (2019).

Trends

Over time, as watersheds become increasingly developed, the amount of imperviousness also increases. Watersheds with low levels of imperviousness that see a significant increase in development will experience a large percent increase in impervious cover. The most significant increases were seen in the UE2 in New Jersey, LV3 in Pennsylvania, and LE2 in Delaware (Table 1.3.2). These are not the most highly developed watersheds (i.e. those comprising the Philadelphia area and the I-95 corridor), but in the areas adjacent to those watersheds, indicating an increase in development, and thus stress on the watershed health. Maps in Figures 1.3.6 and 1.3.7 presents the percentage change in imperviousness for each HUC12 and 21 watersheds, respectively, in the Delaware Estuary and Basin based on the NLCD between 2001 and 2019. The highest changes were seen in the greater Philadelphia area (i.e., LV3, SV3, UE2, and LE2). The watersheds of the Upper Region of the Delaware Estuary and Basin remained low in imperviousness and showed little change over the period.

Figure 1.3.8 presents trends in imperviousness by watershed, for the years 2001, 2011, and 2019. Also shown are the thresholds for Impacted (orange line) and Nonsupporting (red line) watersheds. The sharp rise in imperviousness in the watersheds surrounding the most highly urbanized areas is notable, as is the nominal increase in the less highly impervious watersheds of the upper Delaware Estuary and Basin. Figure 1.3.9 presents the increases in imperviousness for Delaware Estuary and Basin HUC 12s 2001–2019.

Table 1.3.2 Percentage change for each watershed, by region within the Delaware Estuary and Basin.

Part of the Basin	Region	Watershed	Watershed	Impervious cover			Impervious cover change
				2001	2011	2019	2001-19
Upper Basin	Upper	East-West Branch	EW 1	0.8%	0.8%	0.9%	0.1%
			EW 2	0.5%	0.5%	0.6%	0.1%
			EW 3	0.6%	0.7%	0.7%	0.1%
		Lackawaxen	LW 1	1.2%	1.3%	1.4%	0.2%
		Neversink-Mongaup	NM 1	1.4%	1.6%	1.7%	0.3%
	Central	Lehigh Valley	LV 1	1.4%	1.7%	1.8%	0.3%
			LV 2	2.4%	2.5%	2.6%	0.2%
			LV 3	11.6%	12.8%	13.5%	1.9%
		Upper Central	UC 1	3.0%	3.5%	3.7%	0.7%
			UC 2	2.5%	2.8%	3.0%	0.5%
		Lower Central	LC 1	3.2%	3.5%	3.7%	0.5%
Estuary	Lower	Schuylkill Valley	SV 1	3.2%	3.4%	3.5%	0.3%
			SV 2	5.9%	6.4%	6.6%	0.7%
			SV 3	9.9%	10.8%	11.3%	1.4%
		Upper Estuary	UE 1	25.3%	26.2%	26.8%	1.5%
			UE 2	13.1%	14.2%	14.8%	1.6%
		Lower Estuary	LE 1	11.6%	12.4%	12.8%	1.2%
			LE 2	6.8%	8.6%	9.8%	3.0%
			LE 3	3.9%	4.4%	4.8%	0.9%
	Bay	Delaware Bay	DB 1	4.0%	4.9%	5.4%	1.4%
			DB 2	4.0%	4.4%	4.6%	0.6%

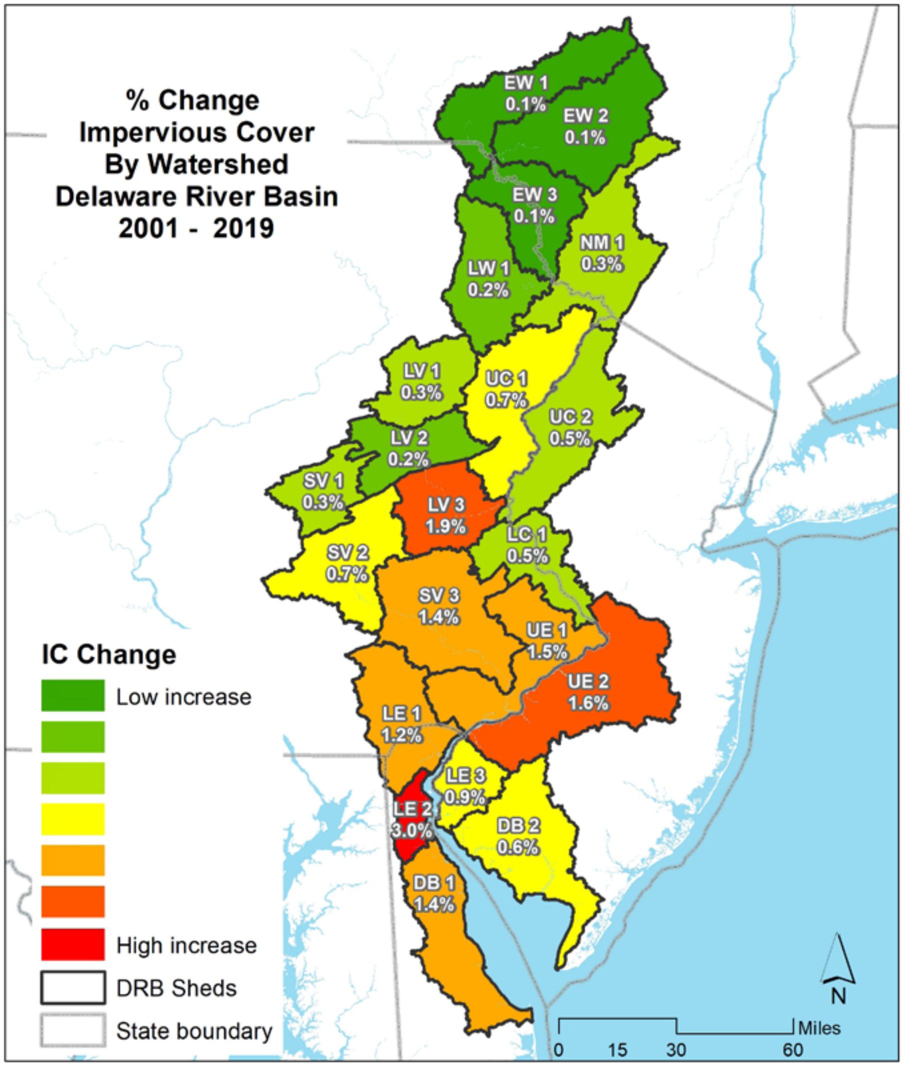


Figure 1.3.6 Percent change in impervious cover by the 21 watersheds in the Delaware River Basin (2019).

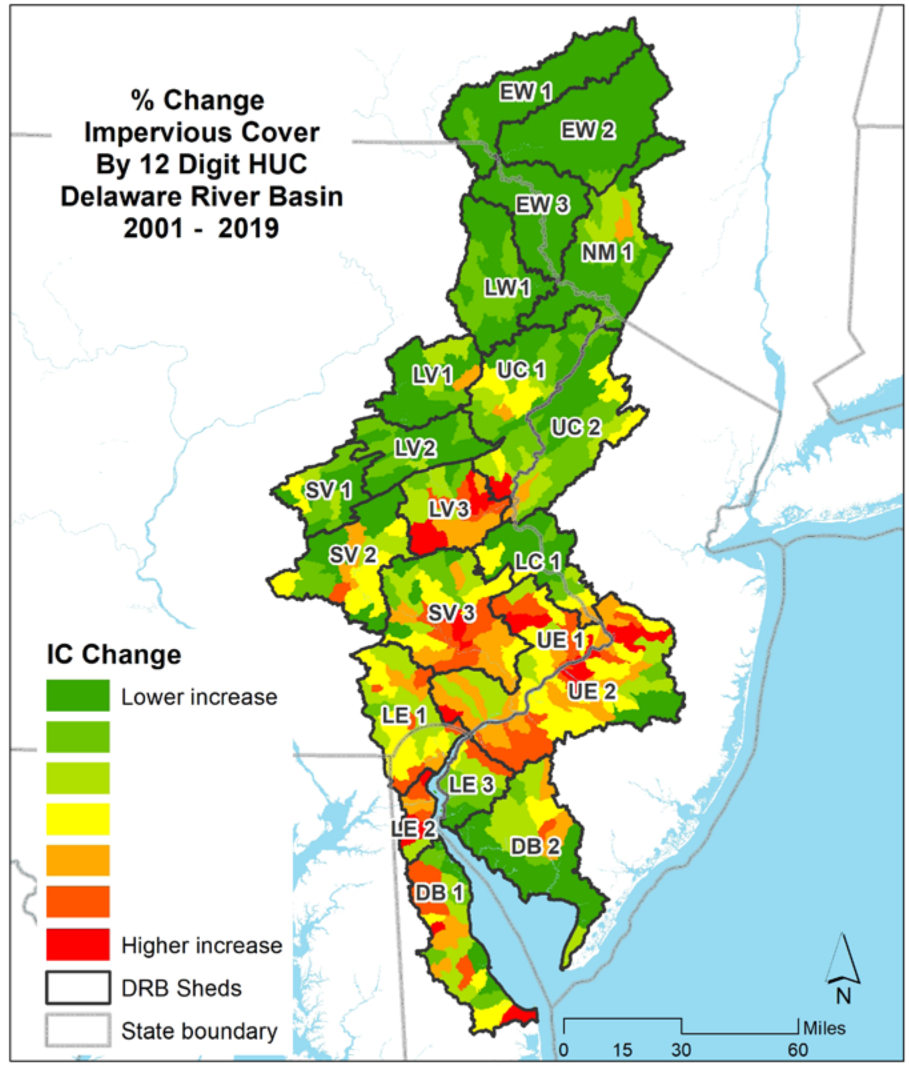


Figure 1.3.7 Percent change in impervious cover by 12-digit HUC watershed in the Delaware River Basin (2019).

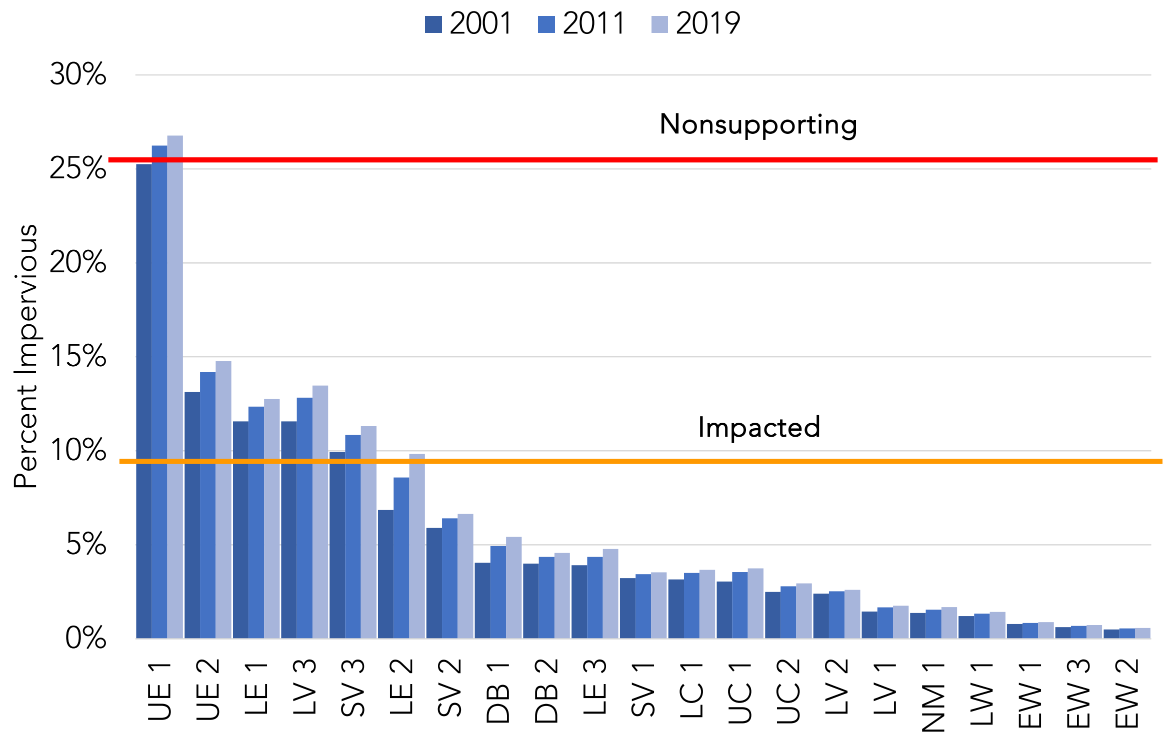


Figure 1.3.8 Percent changes in impervious cover by watershed from 2001-2019.

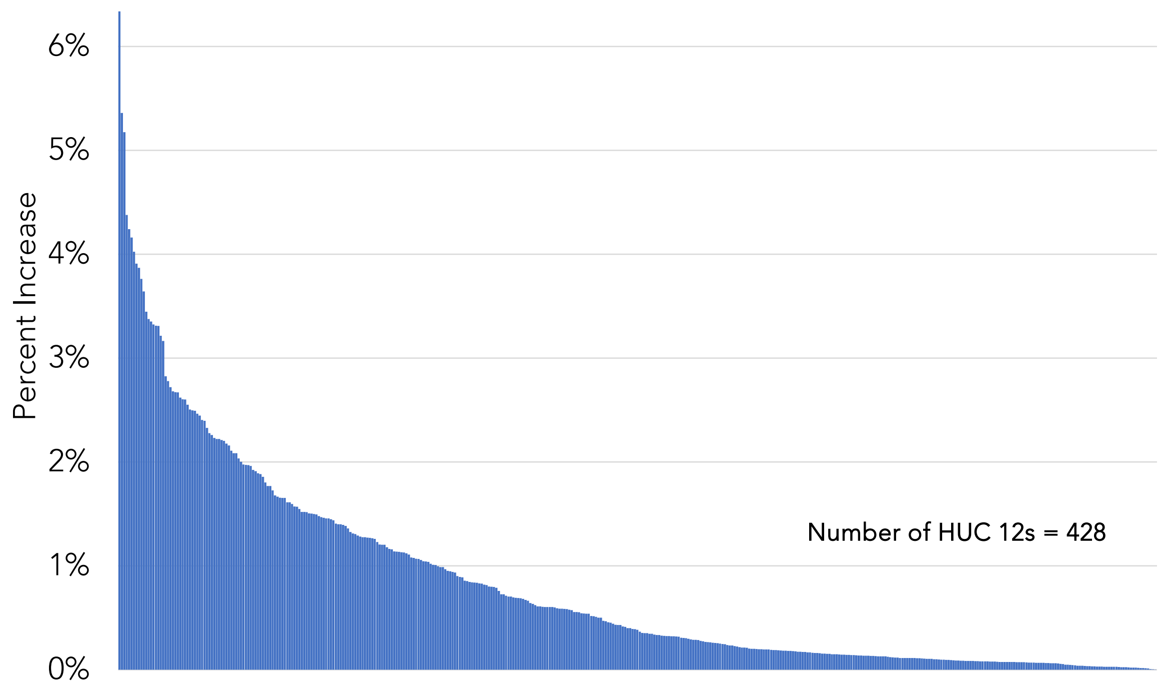


Figure 1.3.9 Histograms of increases in the percent impervious cover from 2001-2019 by HUC12.

Table 1.3.3 shows the total number of HUC 12s in the Basin that are either Impacted (above 10% impervious) or Nonsupporting (over 25% impervious). Of the 428 HUC 12s, 55 (12.9%) were Impacted in 2001, and by 2019 that number had increased to 65 (15.2%). In 2001, 26 (6.1%) of HUC 12s were Nonsupporting, and by 2019 that had increased by three to 29 (6.8%). In all, the number of Impacted or Nonsupporting HUC 12 watersheds increased from 81 (18.9%) in 2001 to 94 (22%)in 2019.

Table 1.3.3 Number of HUC12s that are impacted or nonsupporting.

IC Impact	2001	2011	2019
Total HUC 12s	428	428	428
Impacted (I)	55 (12.9%)	61 (14.3%)	65 (15.2%)
Nonsupporting (N)	26 (6.1%)	29 (6.8%)	29 (6.8%)
All (I & N)	81 (18.9%)	90 (21%)	94 (22%)

Future Predictions

While there is a clear trend upward in imperviousness in the watersheds of the Delaware Estuary and Basin, future directions will depend on land cover, demographic, and economic trends. It is possible to mitigate the negative impacts of imperviousness through proper watershed management. Local and regional planning efforts can improve issues of stormwater, flooding, water quality, and other problems that stem from an excess amount of runoff due to impervious cover. As population and development pressures increase, along with the effects of climate change, impervious cover will remain a primary concern for water resources managers and governmental entities.

Actions and Needs

Two efforts are important to limit the negative effect of imperviousness on the watersheds of the Delaware Estuary and Basin. Large scale planning through bodies such as the federal and state governments, the Delaware River Basin Commission, National Estuary Programs, and many local partners will be required to address water quality and quantity issues caused by excessive runoff from impervious cover.

Funding for transformational efforts at addressing climate change and the impacts of human development should be provided at higher levels. While the Delaware Estuary and Basin has been less-well funded than other large regional basins, recent trends in infrastructure investment, along with public/private partnerships such as the Delaware River Watershed Initiative (DRWI) will help address these problems, and will serve as a catalyst for further investments across the basin.

Summary

Impervious cover is a key metric for tracking and predicting watershed health in terms of water volume, water quality, stream and riparian habitat, and drinking water supply. The trend in increased imperviousness will threaten watersheds at all scales. Smaller urbanizing catchments can be particularly vulnerable, given the increasing transformation of land to developed land cover types, along with relatively weaker local regulations and public will to control imperviousness and the resultant runoff.

1.4 Protected Lands

Description of Indicator

Protected lands are those defined as permanently non-developable (i.e., to urban or suburban uses) because it is either owned outright (fee simple ownership) by a government or other entity which explicitly protects it, or it has an easement that limits or precludes development by the owner due to a legal agreement with a third party. Fee simple protected land is owned by federal, state, or local governments, or by private or non-profit entities, and the owner restricts development of the land. Such land might include state or national parks, preserves, wildlife management areas, wildlife refuges, historic parks, recreational areas, or homeowner association (HOA) owned open space within developments, among others. Eased land can be any open land to which the development rights have been permanently sold or transferred by the owner to a third party. The third party can be a land trust, local or state government, or other private entity. The easement can be designated for a specific purpose, such as agriculture, wildlife conservation, forestry, historic or cultural resources, or other use.

The degree of protection in a watershed afforded by such permanently protected land is important to the current and future health of that watershed. Areas with a high percentage of protected land will undergo less potential development pressures in the future, and are therefore more likely to sustain a higher level of ecological, recreational, historic, and water-quality related integrity. Areas without such protections remain vulnerable to degradation due to more intensive development.

Land which is protected and accessible to the public also provides benefit to the human population in terms of outdoor recreation and ability to enjoy associated benefits to health and well-being. The amount of publicly accessible protected land is also an important indicator of how the presence of protected land can benefit the population of the Delaware Estuary and Basin.

Data sources and processing methodology

The USGS Gap Analysis Program (GAP)[1] has tracked the location and extent of public open space through their Protected Areas Database of the United States (PAD-US). The data are derived from a wide variety of sources, including local, state, national datasets and data provided by private and non-profit entities. The program periodically compiles and inventories protected land nationally, including those owned by federal, state, regional, local government entities, as well as land owned by private and non-profit organizations for the specific purpose of protecting their resources. Additionally, land that is eased by various public, private, and non-profit entities is included in the database. The program also tracks which lands are open to the public and to what degree (e.g., open access versus restricted due to fees or other constraints).

PAD-US data have undergone several iterations over time, starting with Version 1.0 in 2009. The program has released periodic updates to the dataset (available in various GIS formats) since then, with the latest

1. The mission of the USGS Gap Analysis Program (GAP) is providing state, regional and national assessments of the conservation status of native vertebrate species and natural land cover types and facilitating the application of this information to land management activities. The PAD-US geodatabase is required to organize and assess the management status (i.e., apply GAP Status Codes) of elements of biodiversity protection. GAP seeks to increase the efficiency and accuracy of PAD-US updates by leveraging resources in protected areas data aggregation and maintenance as described in "A Map of the Future," published following the PAD-US Design Project (July, 2009) available at: http://gapanalysis.usgs.gov/padus/vision/ with updates coming soon. While PAD-US was originally developed to support the GAP Mission stated above, the dataset is robust and has been expanded to support the conservation, recreation and public health communities as well. Additional applications become apparent over time. See the GAP Website http://gapanalysis.usgs.gov/padus/resources/ or the companion site http://protectedlands.net/uses for more information.

version, and the version used in the current analysis (Version 2.1), having been released in September, 2020. While each iteration of the data theoretically can be used as a "snapshot" in time to determine the overall level of protection (and public access) for each particular year, the data are not easily comparable across time-scales. This is due to the difficulty in compiling data from disparate sources, and in manually cross-walking datasets to make sure they are accurate and not duplicative. If data from one era are missing a source, or if data from two sources overlap partially or entirely, for instance, comparison between releases is problematic. The PAD-US program, in fact, on their website explicitly discourages such comparisons.

While the data in Version 2.1 has been greatly enhanced to be more complete and accurate than that in previous versions, there were still issues with the data which required some additional manual editing and additions. Since the date of establishment is not available for many properties (both fee owned and eased), the decision was made to only derive two periods for trend analysis: pre- and post-2010. To determine the dates of particular tracts of land that were not included in the data already, other data sources, such as previous versions of the PAD-US data as well as Internet searches of the relevant organization/owner were used. Where dates could not be determined, the parcel or property was assumed to have been protected prior to 2010 for analysis purposes. In some cases lands may have been protected in phases, with many years (or decades) separating the protection of different parcels. In this event, the date used was for the primary, largest, or first major parcel protected. Also, in some cases ownership may have changed from one entity (from state to federal, for instance), which complicated the analysis.

While the data in Version 2.1 of PAD-US was structured and cleaned more thoroughly than in previous versions, some editing, such as in the case of overlapping polygons was still necessary in a few instances. There were also incidences of missing or incomplete information (including dates of protection), and this was addressed using ancillary data, such as The Nature Conservancy's 2018 Secured Lands database. New Jersey's Department of Agriculture was also used to fortify the dataset of eased lands in the New Jersey portion of the Delaware Estuary and Basin.

Once the data were enhanced using the steps outlined above, analysis was undertaken in GIS to determine the extent of the characteristics of interest, by each of the 21 watersheds of the Delaware Estuary and Basin. Firstly, for each fee owned preserved property, the area was calculated based on ownership type for each period, pre- and post- 2010. The database included military installations and school district lands, so these were excluded before the areas were summarized. Table 1.4.1 shows the ownership types used for the analysis. Note that "Local" owner types include county and municipal government, while "Unknown/Other" includes unknown owners and lands with joint ownership. Similarly, the area of permanently eased lands was calculated (for both pre- and post-2010 time periods), based on the type of easement. Table 1.4.1 presents the easement types used in the analysis.

To determine the amount of protected land that is also accessible to the public, information from the PAD-US database was used. Land was considered open if there was open access or access restricted by fee or membership. In the database these have the code "OA" and "RA," respectively. Areas that were closed to access, or of unknown status were not included as providing public access ("XA" and "UA", respectively, in the database).

For each component of the PAD-US data considered, the total summary of land by category was tabulated and mapped. For the area of fee owned and eased lands the pre- and post-2010 trend was also considered. For public access, only the latest (2020) conditions were analyzed and presented.

Table 1.4.1 Ownership and easement types in the PAD-US.

Type	Description	Database code
Owner	Federal	FED
	State	STAT
	Local	LOC
	Non-profit/NGO	NGO
	Private	PVT
	Unknown/other	UNK / JNT
Easement	Agricultural	AGRE
	Conservation	CONE
	Forestry	FORE
	Historical	HCAE
	Recreational	RECE
	Unknown/other	OTHE / UNK

Present Status

The following maps (Figures 1.4.1 to 1.4.4) show the distribution and types of protected land in the Delaware Estuary and Basin in 2020, including fee simple ownership (i.e., land that is owned completely) and easements (i.e., lands with legal development restrictions). Fee lands are symbolized according to the owner type, while easements are presented by type of easement. Within the 12,857 square mile Delaware Estuary and Basin in 2020, there were nearly 2,050 square miles of land protected through fee simple ownership, or 15.9% of the total land area in the basin. Of the protected lands, the most was protected by states—nearly 1,350 square miles, or 10.5%—with the next highest share protected at the local level (counties and municipalities), with 327 square miles, or over 2.5% of the land area. Federal lands and NGO lands were the next most significant holdings, with 178.5 and 141.4 square miles (1.4% and 1.1%), respectively. In 2020 land protected through easements made up over 862 square miles of the Delaware Estuary and Basin, which translates to 6.7% of the total land area. Of these, 567 were agricultural easements (4.4%) with other/unknown easements making up the next largest proportion, at nearly 148 square miles (1.15%). Conservation easements were also a significant proportion of protected land, representing over 126 square miles, or nearly 1% of the total land area in the Delaware Estuary and Basin. Figure 1.4.5 shows the proportions of fee simple owned land by ownership and eased land by type. Table 1.4.2 summarizes the total areas and percentages for land protected through easement in the Delaware RIver Basin.

The highest proportion of fee simple owned protected land are found in the upper reaches of the basin, in the headwaters of the Lehigh Valley (LV 1) with nearly 42% of the land protected, and along the main stem of the Delaware River above the head of tide (Upper Central and Neversink-Mongaup watersheds). The lands along the Delaware Bayshore also have a relatively high proportion of protected land; Delaware Bay watersheds have nearly 30% protection in New Jersey, and nearly 20% on the Delaware side. The more urbanized watersheds in the central portions of the Delaware Estuary and Basin have somewhat lower degrees of protected land.

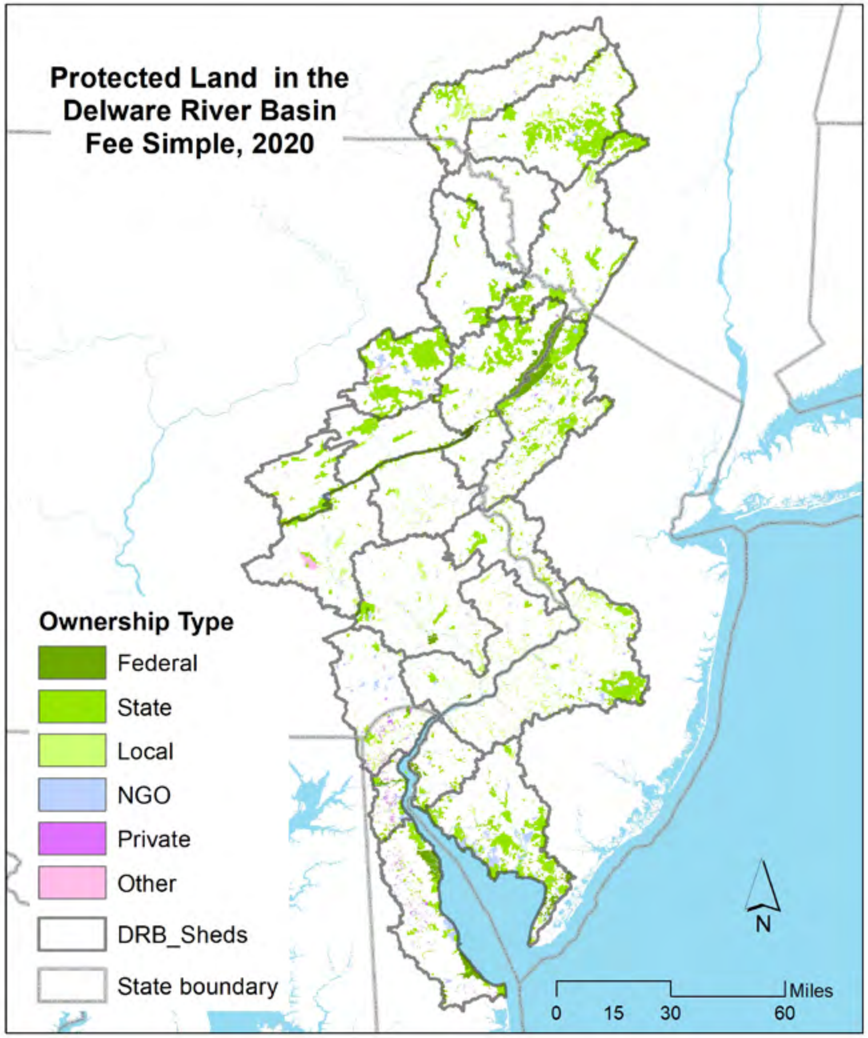


Figure 1.4.1 Fee simple protected land in the Delaware River Basin (2020).

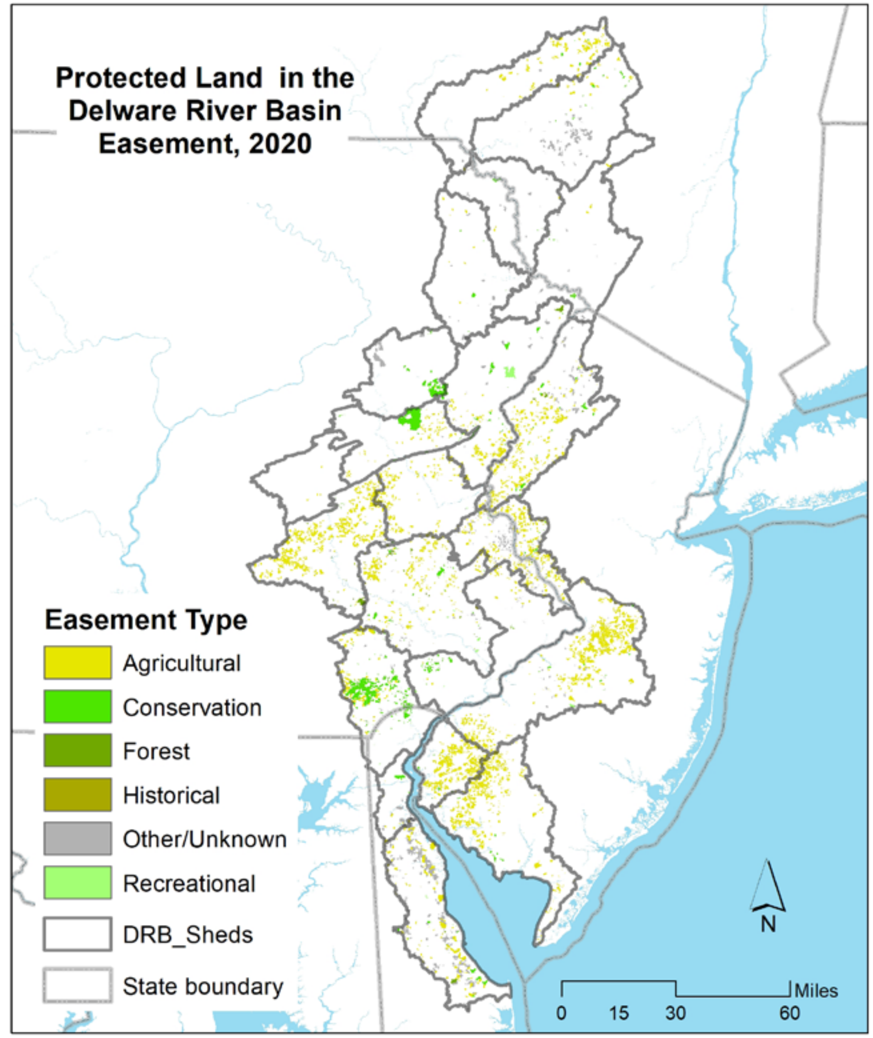


Figure 1.4.2 Easement protected land in the Delaware River Basin (2020).

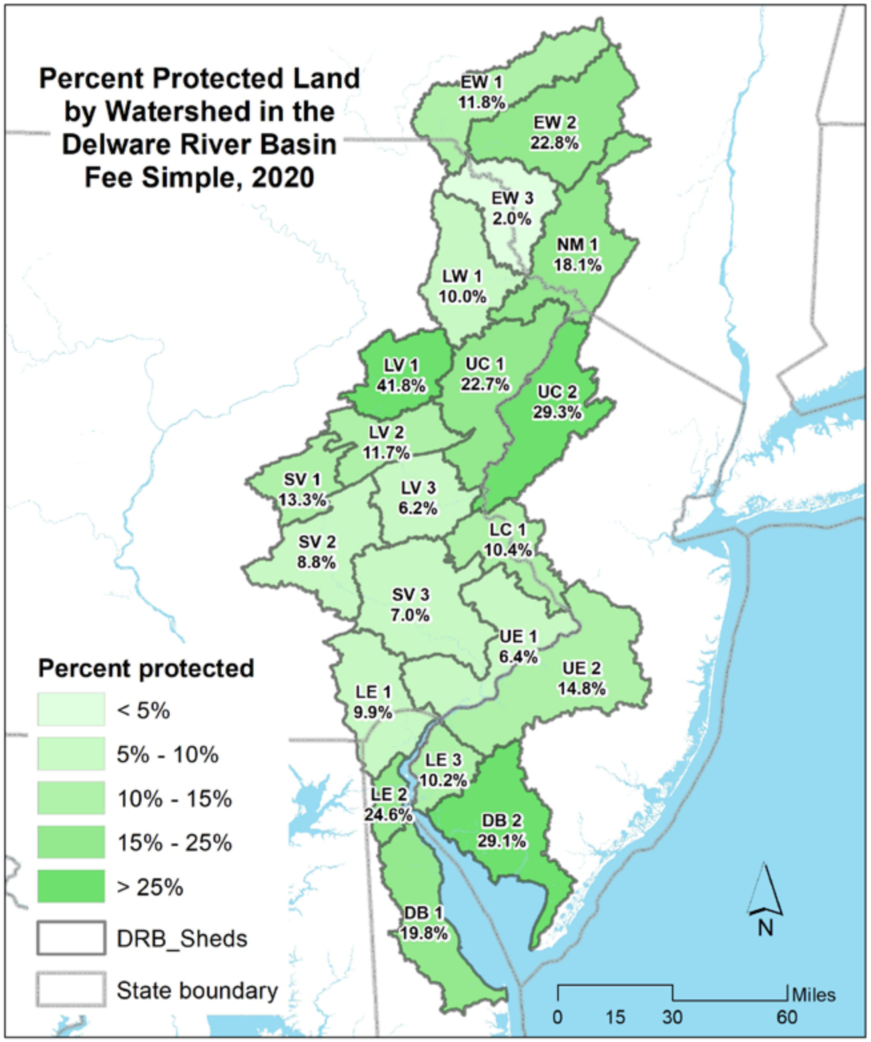


Figure 1.4.3 Percent fee simple protected land in the Delaware River Basin (2020).

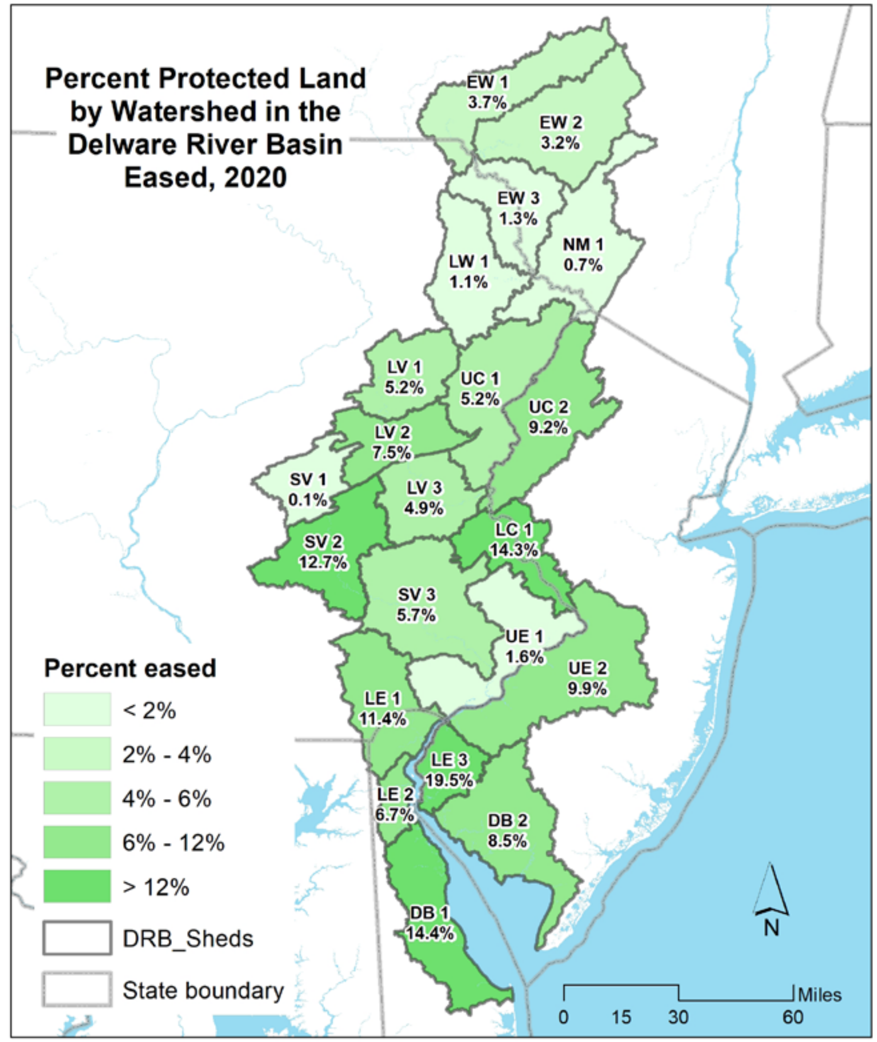
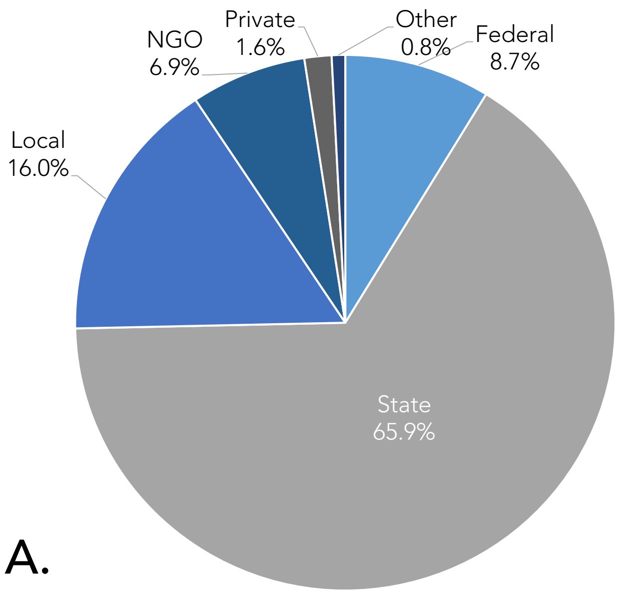


Figure 1.4.4 Percent easement protected land in the Delaware River Basin (2020).



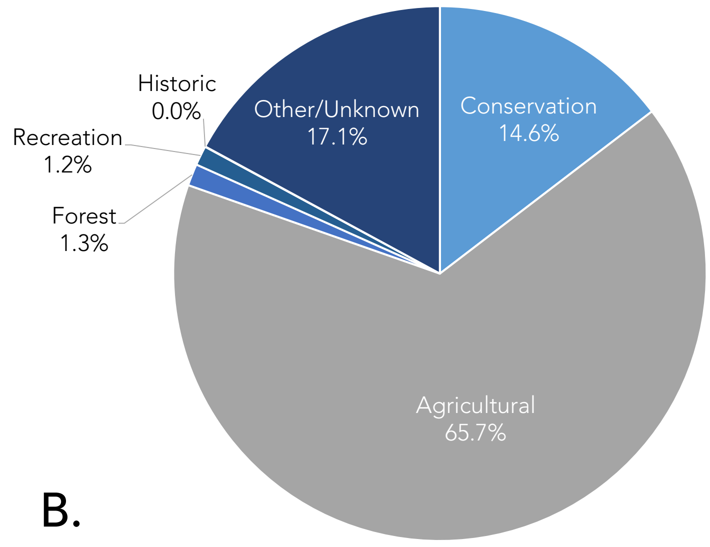


Figure 1.4.5 Proportion of protected land by fee simple ownership (A) and easement type (B).

Conversely, easements tend to be more prevalent in the central portion of the basin, with Schuylkill Valley (SV 2) at 12.7% protected, and Lower Central (LC 1) at 14.3% protected. The upper reaches of the basin have lower proportions of eased land, but are highly forested and protected through ownership. Table 1.4.2 presents the total fee owned and eased protected lands in the Delaware Estuary and Basin in 2020 by major watershed. Overall in the basin, there are over 2,900 square miles of protected land, or 22.6% of the total land area.

Access to protected lands by the public is an important factor in promoting watersheds as a direct public benefit. Allowing access to open spaces is an important factor in peoples' quality of life, and can have the effect of promoting and encouraging further protection measures. Table 1.4.3 summarizes the square mileage and percentage of publicly accessible protected land in the Delaware Estuary and Basin, by watershed. The map in Figure 1.4.7 shows the properties in the Delaware Estuary and Basin that are open to the public, either with no restrictions, or through fee- or membership-based access. Central portions of the basin have the most land available, as a percentage of total land area, for public access, with an average of approximately 25% in the Upper Central watershed, and 40% in the headwaters of the Lehigh Valley watershed (LV 1)(Figures 1.4.6, 1.4.7). Other areas, including the upper portion of the basin, watershed (EW 2) and the Delaware Bayshore (DB 2) also have a high proportion (19% and 25%, respectively) of protected land accessible to the public. The Estuary portion of the Delaware Estuary and Basin has 728 square miles of protected land accessible to the public, or 12% of the total land area. The Upper Basin (non-tidal portion) has 1,095 square miles publicly accessible, over 16% of the total land area.

Trends

The total area of protected land tends to change rather slowly over time, as the process of purchasing land or going through the easement process can be lengthy. This analysis used a 10-year time frame (between 2010 and 2020) due to the difficulty of compiling establishment date information compiled prior to widespread availability of GIS data (for instance through programs such as the PAD-US). The maps in Figures 1.4.8 and 1.4.9 show the change, by each of the 21 watersheds of the Delaware Estuary and Basin, in fee owned protected land and easement, between 2010 and 2020.

Table 1.4.2 Fee owned and easement protected lands in the Delaware River Basin.

Region	Watershed		Fee Owned		Easements		Total Protected	
	Name	mi²	mi²	%	mi²	%	mi²	%
Upper	East-West Branch	2,029	281	13.8%	59	2.9%	339	16.7%
	Lackawaxen	597	60	10.0%	6	1.1%	66	11.0%
	Neversink-Mongaup	816	148	18.1%	6	0.7%	153	18.8%
Central	Lehigh Valley	1,361	269	19.7%	79	5.8%	348	25.6%
	Upper Central	1,523	395	25.9%	109	7.2%	504	33.1%
	Lower Central	454	47	10.4%	65	14.3%	112	24.7%
Lower	Schuylkill Valley	1,891	165	8.7%	135	7.2%	301	15.9%
	Upper Estuary	1,743	200	11.4%	114	6.5%	313	18.0%
	Lower Estuary	1,020	124	12.2%	131	12.8%	255	25.0%
Bayshore	Delaware Bay	1,423	355	25.0%	159	11.1%	514	36.1%
Upper Basin		6,780	1,199	17.7%	324	4.8%	1,523	22.5%
Delaware Estuary		6,077	844	13.9%	539	8.9%	1,383	22.8%
Basin Total		12,857	2,043	15.9%	863	6.7%	2,906	22.6%

Table 1.4.3 Protected lands with public access in the Delaware River Basin.

Region	Watershed		Protected land with public access	
	Name	mi²	mi²	%
Upper	East-West Branch	2,029	209	10.3%
	Lackawaxen	597	59	9.9%
	Neversink-Mongaup	816	138	17.0%
Central	Lehigh Valley	1,361	257	18.9%
	Upper Central	1,523	388	25.5%
	Lower Central	454	45	9.8%
Lower	Schuylkill Valley	1,891	163	8.6%
	Upper Estuary	1,743	181	10.4%
	Lower Estuary	1,020	86	8.4%
Bayshore	Delaware Bay	1,423	299	21.0%
Upper Basin			6,780	1,095
Delaware Estuary			6,077	728
Basin Total			12,857	1,824

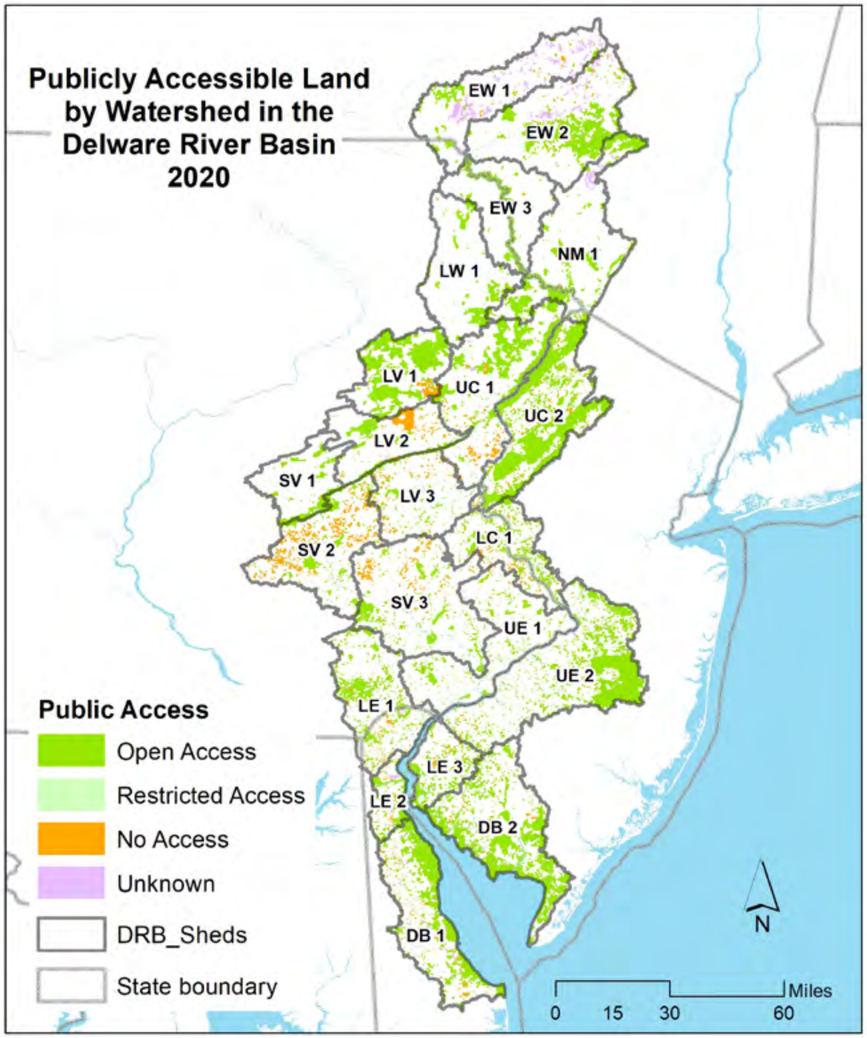


Figure 1.4.6 Public access to protected lands in the Delaware River Basin (2020).

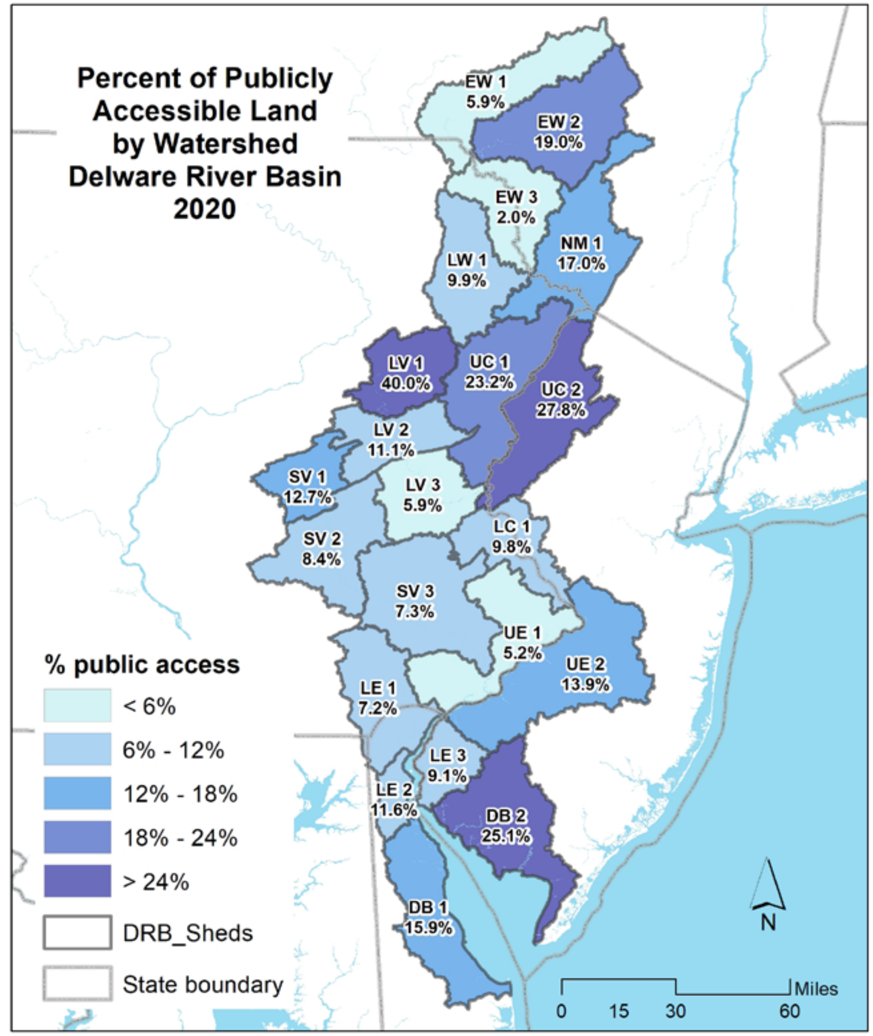


Figure 1.4.7 Proportion of public access to protected lands in the Delaware River Basin (2020).

The watersheds with the highest increase in fee owned protected land are in the Lower Estuary (LE 1), with 2.6 square miles added between 2010 and 2020, UC 1, with 3 square miles added, and EW 1 and EW 2, with 3.2 and 2.3 square miles added, respectively. The increase in the East-West watersheds (EW 1 and EW 2) are due primarily to purchases made the New York City to retain open lands for reservoir protection. In the Lower Estuary (LE 1), the increase was due primarily to acquisitions and transactions relating to the First State National Historical Park, while in Upper Central (UC 1) most changes were due to purchases by a variety of localities and non-governmental organizations." Overall, the Delaware Estuary and Basin added 13.9 square miles, or 0.1% of the total land area, of fee owned protected land, and 149 square miles (1.2% of the total land area). See Tables 1.4.4 and 1.4.5 for the basin-wide totals and change for fee owned and eased protected land by owner type and easement type, respectively. Figure 1.4.8 presents the change in fee simple protected land, by ownership type and as a total for the Delaware Estuary and Basin. Figure 1.4.9 presents the change in land protected through easement, by easement type and as a total for the Delaware Estuary and Basin.

Table 1.4.4 Ownership and easement types in the PAD-US dataset.

Protected Land, Easements	2010		2020		Change	
	mi²	%	mi²	%	mi²	%
Conservation	81.53	0.63%	126.18	0.98%	44.65	0.35%
Agricultural	466.48	3.63%	567.03	4.41%	100.55	0.78%
Forest	11.05	0.09%	11.37	0.09%	0.32	0.00%
Recreation	9.94	0.08%	10.19	0.08%	0.25	0.00%
Historic	0.16	0.00%	0.16	0.00%	-	0.00%
Other/Unknown	144.35	1.12%	147.65	1.15%	3.30	0.03%
Total	713.51	5.55%	862.58	6.71%	149.06	1.16%

Table 1.4.5 Ownership and easement types in the PAD-US dataset.

Protected Land, Fee Owned	2010		2020		Change	
	mi²	%	mi²	%	mi²	%
Federal	176.71	1.37%	178.43	1.39%	1.72	0.01%
State	1,346.81	10.48%	1,347.01	10.48%	0.19	0.00%
Local	320.05	2.49%	326.61	2.54%	6.56	0.05%
NGO	136.27	1.06%	141.44	1.10%	5.17	0.04%
Private	33.08	0.26%	33.15	0.26%	0.07	0.00%
Other	16.38	0.13%	16.52	0.13%	0.13	0.00%
Total	2,029.31	15.78%	2,043.17	15.89%	13.86	0.11%

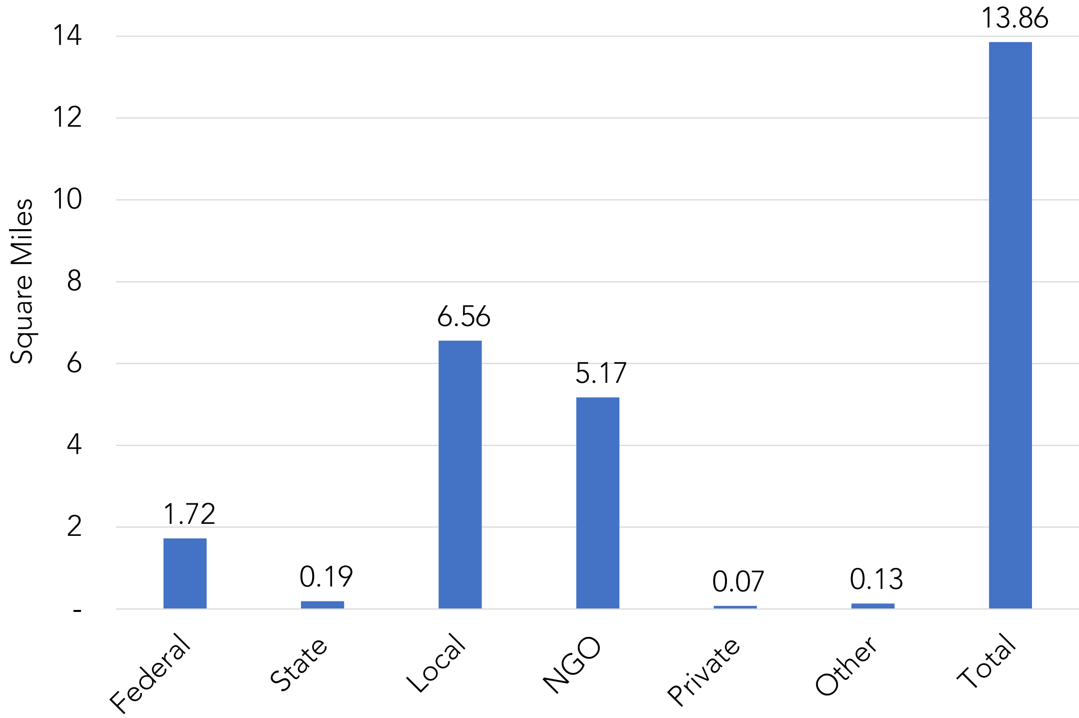


Figure 1.4.8 Change in fee simple protected land by ownership (2010-2020).

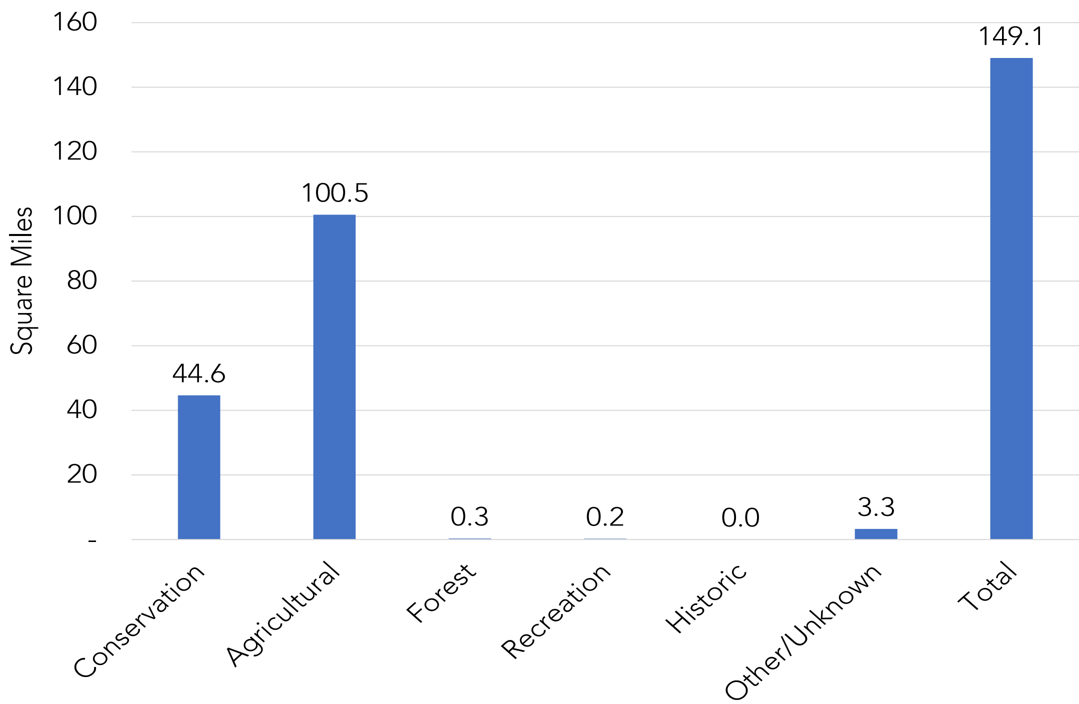


Figure 1.4.9 Change in eased land by type (2010-2020).

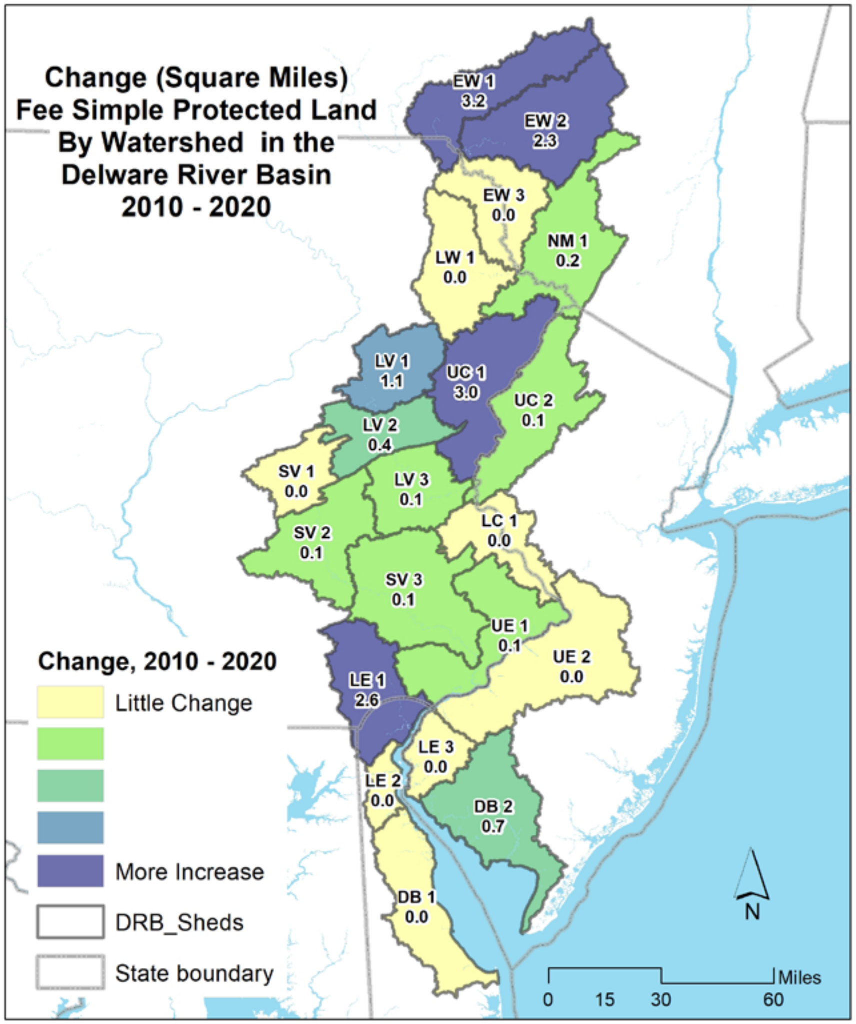


Figure 1.4.10 Change in fee simple protected land in the Delaware River Basin (2010-2020).

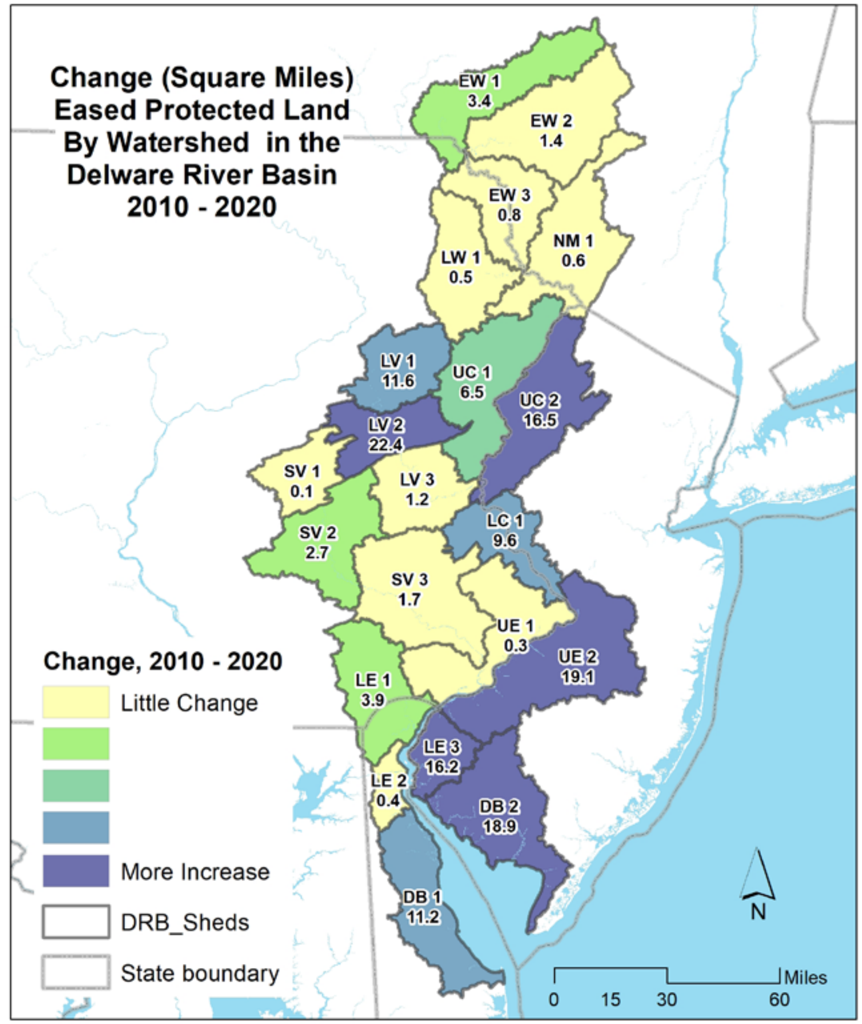


Figure 1.4.11 Change in easement protected land in the Delaware River Basin (2010-2020).

Future Predictions

The pace of land protection is fairly slow, so it is unlikely that very large changes to the overall percentage of protection is going to occur. There is a broad recognition, however of the many benefits of land preservation. Many public entities at the federal, state, county, and local levels have active programs to foster and encourage preservation efforts. New developments often have open space requirements, and the many land conservation organizations are active in various locations throughout the Delaware Estuary and Basin.

Easement programs provide funding and support for preservation of current conditions, such as through agricultural preservation and conservation easements. The amount of land in easement is therefore steadily increasing, though it is generally focused only in certain locations. Recognition of the multiple benefits of preserving open space is widely recognized by the public and by public, private, and non-profit entities.

Actions and Needs

Coordination among federal, regional, state, local, and private/non-profit organizations remains important, as competing interests among watershed stakeholders becomes more intense. Through regional-scale efforts such as the Delaware River Watershed Initiative (DRWI), and organizational frameworks such as the Partnership for the Delaware Estuary (PDE) and the Delaware River Basin Commission (DRBC), this coordination has been improving. Data collection and coordination efforts, such as the PAD-US are necessary to help compile and disseminate information. Accurate and complete layers of data are critical as organizations prioritize their focus to maximize benefit to the overall health of the watershed. Support of such regional and national coordination should continue. Future iterations of the data will become more refined and comprehensive; therefore, this effort should be supported to periodically provide updates.

Programs that target preservation of land need to be properly funded, and should be given priority by governments and regulators. Public awareness of the importance both of open space and protection of existing land-based resources should also be a priority and coordinated at the watershed scale. Creation of additional areas of protected land that are also open to the public is important for the well-being of watershed residents, and for the promotion of understanding of the importance of protections for watershed health.

Summary

As the pool of available open land which could be protected is always shrinking, planning for and prioritizing and coordinating future protection efforts is crucial. While many areas of the Delaware Estuary and Basin are fairly well-protected by fee ownership and easements on land, overall the basin has over 2,900 square miles of protected land (more than 22% of the total land area). Some watersheds, however have lower levels of protection. Some areas are already developed, but many have ample opportunities to increase the amount of protection afforded either through ownership or easement.

Public accessibility to protected open space is an important factor to the well-being of the basin's inhabitants. Currently, the Estuary portion of the Basin has about 12% of land protected and publicly accessible. The Upper Basin watersheds has over 16% of the total land area accessible to the public, with the Delaware Estuary and Basin overall having 14.2% of its land area accessible. The combination of robust levels of protection and increasing access to protected open spaces for the public will go far in fostering watershed health and multiple benefits to the inhabitants of the Delaware River Basin.

References

Davis, James C., Anil Rupasingha, John Cromartie, and Austin Sanders, "Rural America at a Glance, 2022 Edition," Economic Information Bulletin No. 246, U.S. Department of Agriculture, Economic Research Service, Washington, DC, 2022.

Gao, J. and O'Neill, B., 2021. Different Spatiotemporal Patterns in Global Human Population and Built-Up Land. Earth's Future, 9(8), p.e2020EF001920. https://agupubs.onlinelibrary.wiley.com/doi/pdfdirect/10.1029/2020EF001920

Kauffman Jr, G.J., 2010. The Delaware River revival: Four centuries of historic water quality change from Henry Hudson to Benjamin Franklin to JFK. Pennsylvania History: A Journal of Mid-Atlantic Studies, 77(4), pp.432-465. https://journals.psu.edu/phj/article/download/59953/59771

Kauffman, G.J., Homsey, A.R., Belden, A.C. and Sanchez, J.R., 2011. Water quality trends in the Delaware River Basin (USA) from 1980 to 2005. Environmental monitoring and assessment, 177(1), pp.193-225. https://link.springer.com/content/pdf/10.1007/s10661-010-1628-8.pdf

Schueler, T. R., L. Fraley-McNeal, and K. Cappiella. 2009. "Is Impervious Cover Still Important? Review of Recent Research."Journal of Hydrologic Engineering, 14(4).

Vespa, Jonathan, Lauren Medina, and David M. Armstrong, "Demographic Turning Points for the United States: Population Projections for 2020 to 2060," Current Population Reports, P25-1144, U.S. Census Bureau, Washington, DC, 2020.

2

TECHNICAL REPORT FOR THE ESTUARY AND BASIN

Climate Change

December 2022 | Report No.22-05

Partnership for the Delaware Estuary—Host of the Delaware Estuary Program

2 Climate Change

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Contributors Kristen Kavanagh, LeeAnn Haaf, Kirk Raper
Cover photograph by LeeAnn Haaf

Table of Contents

2. Climate Change

Abstract

Past and likely future changes in the climate of the Delaware Estuary and Basin are presented. From a historical analysis of air temperature, precipitation, streamflow, sea level, and ice and snow indicators, an overall picture of dramatic and accelerating climate change in the Delaware Estuary and Basin emerges, one that is largely consistent with expectations from increases in greenhouse gases. These climate trends are extremely likely to continue into the next few decades regardless of greenhouse gas emissions. Climate change beyond mid-century will depend strongly on the emissions pathway. Hence, a combination of aggressive local adaptation and global emissions reduction is needed to avoid the worst impacts of anthropogenic climate change in the Delaware Estuary and Basin. Adaptation planning should compensate for unjust policies, such as redlining, that have led climate impacts to fall disproportionately on the most vulnerable communities.

What is Climate Change?

Climate is among the most important factors determining the character of a watershed or an estuary. Climate influences the type of natural and managed vegetation growing on a landscape as well as the flora and fauna in tidal and non-tidal aquatic habitats. The abundance of fresh water available for human use is largely dictated by the balance of precipitation and evapotranspiration, which are driven by climate. One of the most serious impacts of climate change is flooding, which is worsening with warming via sea level rise and more extreme precipitation (Douville et al. 2022; Fox-Kemper et al. 2022). Though water quality is generally considered to be driven by human activity on the landscape, it is increasingly recognized that climate can act as a stress multiplier by, for example, exacerbating harmful algal blooms during unusually warm periods and reducing water clarity during extreme precipitation events (Paerl and Huisman 2008; Coffey et al. 2018). Because of the strong influence of climate, it must be considered in restoration planning. In summary, all the topics covered by this report—watersheds and landscapes, water quantity, water quality, sediments, aquatic habitat, living resources, and restoration—are all profoundly influenced by climate.

Climate change is defined by the Intergovernmental Panel on Climate Change (IPCC) as changes in the mean and/or variability of climatic properties (e.g., air temperature and precipitation) that persist for an extended period, typically decades or longer, due to natural internal processes or external forcings. One of the main external forcings of climate change is the addition of greenhouse gases (GHGs) to the atmosphere via anthropogenic emissions (Myhre et al. 2013). Globally, there is an urgency to minimize potential future impacts of climate change through the reduction of anthropogenic GHG emissions, particularly carbon dioxide (CO_2), because of their ability to increase radiative forcing (Myhre et al. 2013). Since the beginning of the Industrial Age in the late 1700s to December 2021, the atmospheric CO_2 concentration has increased from about 280 ppm to 417 ppm (Joos and Spahni 2008; NOAA Global Monitoring Laboratory). Over the past decade, the atmospheric CO_2 concentration has increased at a rate of 2.5 ppm per year (Friedlingstein et al. 2022).

This chapter describes how the climate of the Delaware Estuary and Basin has changed in the past and discusses how it may change in the future. The focus is on air temperature and precipitation throughout the watershed with additional analysis of changes in streamflow, ice jams, and sea level. This chapter follows a similar structure as the corresponding chapter from the 2017 report, with a few differences. Sea level has been added as an indicator, given its importance to tidal wetlands, salt water intrusion, and flooding of coastal communities. Since the impacts of changes in climate and other environmental conditions are unequal across socioeconomic and racial groups, we have also included discussions that highlight two particular instances, redlining and nuisance flooding (Climate Features 1 and 2). Wind speed was removed as a climate change indicator as it has been found to be more dependent on land use change and management than climate change (Vautard et al. 2010). Snow cover is now included with precipitation in Section 2.2.

Methods

Although different datasets and procedures were applied to analyze the different indicators, there were several common methods used in the analysis of most indicators. All trends were calculated using the nonparametric Theil-Sen slope estimator (Theil 1950; Sen 1968). The statistical significance of each trend was tested using the nonparametric Mann-Kendall test (Mann 1945; Kendall 1955) at a significance level of α=0.05. For time series that exhibited statistically significant autocorrelation, which increases the likelihood of falsely identifying a spurious trend as significant, the trend and significance were corrected using the trend-free prewhitening method (Coen et al. 2020; Yue et al. 2002). These statistical methods were provided by the Python package "pyMannKendall" (Hussain et al. 2019). Trends were calculated for both the full extent of each time series and for the most recent 30 years (1992–2021). To merge data from multiple stations into a single time series, anomalies were calculated by subtracting each station's mean value over the standard 30-year climate normal time period (1991-2020) prior to averaging the station data.

Some of the indicator trends presented here are broken down by season and/or by sub-watershed, such as upper Basin or Estuary (i.e., Fig 2.1). The seasons were defined as December to February (DJF; winter), March to May (MAM; spring), June to August (JJA; summer), and September to November (SON; fall). Finally, for daily data (temperature and precipitation extremes and streamflow), if a year or season at a given station had more than 5 days of missing or flagged data in any month, the data from the entire year or season were excluded from the analysis to reduce the potential biases caused by incomplete data.

2.1 Air Temperature

Description of Indicator

Air temperature is one of the most important climate change indicators as it directly influences snowpack, evapotranspiration, growing season, water temperature, and other fundamental characteristics of watersheds and estuaries.

Monthly mean near-surface air temperature was obtained from version 2.5 of the U.S. Historical Climatology Network (USHCN) database. A complete description of the dataset and data processing is provided in Menne et al. (2009), Menne et al. (2015a), and Menne et al. (2015b); an abbreviated description is presented here. Most data in the USHCN are a subset of the data from the National Oceanographic and Atmospheric Administration's (NOAA's) Cooperative Observer Program (COOP). The COOP data stations included in the USHCN dataset are relatively long, stable, and amenable to adjustments for non-climatic changes (such as station relocations).

The COOP data are at daily resolution. During processing for inclusion in the USHCN dataset, the data are extensively screened for erroneous daily values. For example, data that show strong spatial or temporal inconsistency are flagged. The monthly USHCN dataset was derived from the daily dataset in several steps. First, means for a given month were computed if no more than nine daily values were flagged or missing for that month. Second, the monthly dataset was subjected to further consistency checks that are qualitatively similar to the checks for the daily data. Third, the data were adjusted for time of observation, which has undergone significant change in the U.S. Fourth, a "change-point" detection algorithm was used to adjust the temperature for other inhomogeneities, such as change in station location, change in instrumentation, and change in nearby land use (e.g., urbanization). As in the 2017 report, these adjustments are needed in order to correct for the substantial cooling effect of changes in equipment and observation times.

The 15 USHCN stations located in or near the Delaware Estuary and Basin were selected for analysis (Fig 2.1, Table 2.1). The analysis distinguished between the upper and lower portions of the Delaware Estuary watershed. The Delaware Estuary, also sometimes referred to here as the lower watershed, is defined by those basins that deliver freshwater directly to the tidal portion of the estuary, which is located below Trenton, NJ. The upper Basin, or upper watershed, drains to the Delaware River above Trenton. There are 8 USHCN stations in the Estuary and 7 in the upper Basin.

The period 1910–2021 was selected for analysis based on the monthly dataset because every station has a value during this time period (some being filled by interpolation).

Table 2.1 USHCN stations in the Delaware Estuary and Basin.

#	Name	State	ID number	Latitude (deg)	Longitude (deg)	Elevation (m)	Start–end
1	Deposit	NY	302060	42.0628	-75.4264	304.8	1963–2011
2	Pleasant Mt. 1 W	PA	367029	41.7394	-75.4464	548.6	1926–2016
3	Port Jervis	NY	306774	41.3800	-74.6847	143.3	1910–2020
4	Stroudsburg	PA	368596	41.0125	-75.1906	140.2	1912–2021
5	Belvidere BRG	NJ	280734	40.8292	-75.0836	80.2	1983–2018
6	Palmerton	PA	366689	40.8000	-75.6167	125.0	1918–1997
7	Allentown AP	PA	360106	40.6508	-75.4492	118.9	1948–2021
8	Reading 4 NNW	PA	367322	40.4269	-75.9319	109.7	1974–2007
9	West Chester 2 NW	PA	369464	39.9708	-75.6350	114.3	1910–2016
10	Moorestown	NJ	285728	39.9511	-74.9697	13.7	1910–2008
11	Indian Mills 2 W	NJ	284229	39.8144	-74.7883	30.5	1910–2019
12	Wilmington Porter Res.	DE	079605	39.7739	-75.5414	82.3	1942–2020
13	Newark Univ. Farm	DE	076410	39.6694	-75.7514	27.4	1942–2020
14	Dover	DE	072730	39.2583	-75.5167	9.1	1910–2018
15	Milford 2 SE	DE	075915	38.8983	-75.4250	10.7	1916–2001

Past Trends

Annual-mean temperature has increased significantly at the 95% confidence interval over the last 112 years in both the upper and lower watersheds (Fig 2.2, Table 2.2). Based on these trends, temperature has increased by roughly 1.7 °C (3.0 °F) over the last 112 years. This rate is consistent with the predicted effect of GHGs (Najjar et al. 2009). The estimated trend in annual mean temperature during the past 30 years of about 0.06 °C (0.10 °F) per year is around four times greater than during the last 112 years, indicating an acceleration in warming.

Since 1910, significant warming trends were also present in both portions of the watersheds for all seasons (Fig 2.3, Table 2.2). In the recent 30-year period, both watersheds show significant temperature increases for all seasons except the winter.

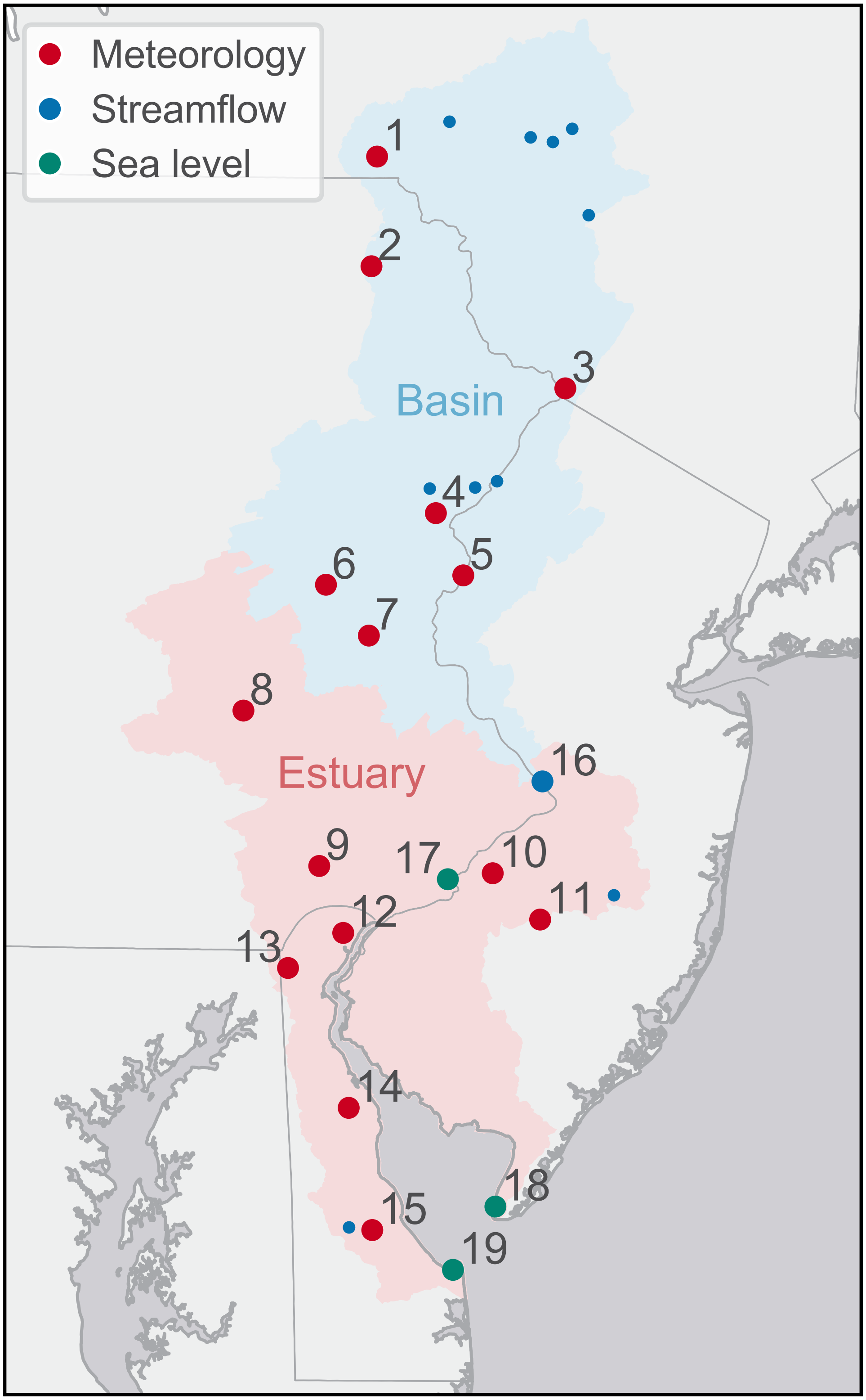
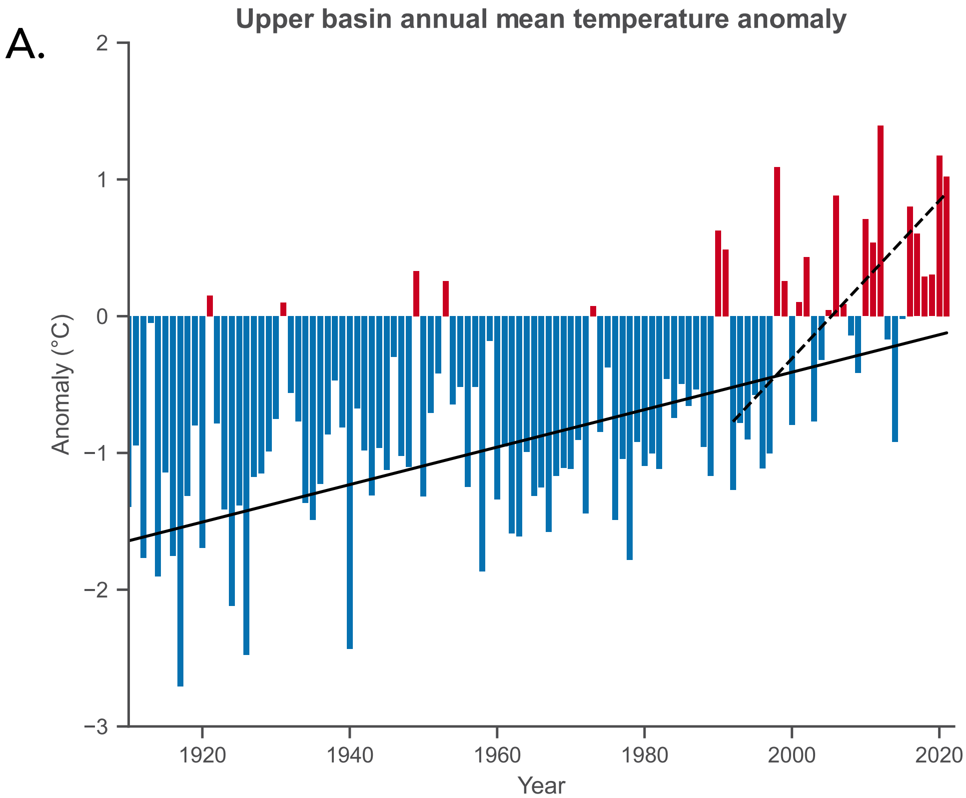


Figure 2.1 Stations in the Delaware Estuary (red shaded region) and Basin (blue shaded region) for: air temperature and precipitation (red circles - see Table 2.1), streamflow (blue circles - #16 is the Trenton gauge), and sea level (green circles - #17 is Philadelphia, #18 is Cape May, and #19 is Lewes tidal stations).



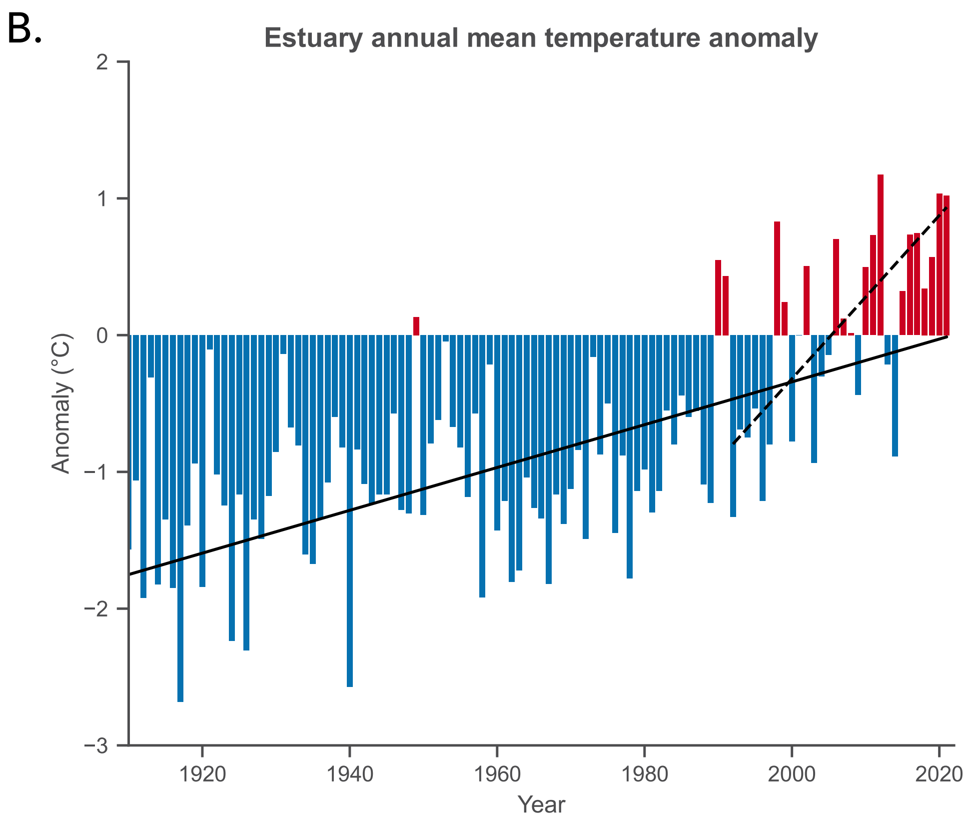


Figure 2.2 Annual mean temperature anomalies with respect to the 1991-2020 average in the Upper Basin (A) and Estuary (B). The solid and dashed lines are linear trends for the 1910–2021 and 1992–2021 periods, respectively.

Table 2.2 Temperature trends in the Delaware Estuary and Basin. Significant trends are bold (95% confidence).

Seasonal Subset	Temperature Trend (°C/decade)			
	1910-2021	p-value	1992-2021	p-value
		Upper Basin		
Annual	**0.14**	**$2.4e^{-10}$**	**0.58**	**0.002**
DJF	**0.19**	**0.00028**	0.35	0.39
MAM	**0.13**	**0.00013**	**0.54**	**0.019**
JJA	**0.13**	**$3.0e^{-07}$**	**0.49**	**0.011**
SON	**0.13**	**$2.5e^{-05}$**	**0.66**	**0.0029**
		Estuary		
Annual	**0.16**	**$1.6e^{-12}$**	**0.60**	**0.0012**
DJF	**0.19**	**0.00045**	0.36	0.43
MAM	**0.16**	**$5.7e^{-06}$**	**0.59**	**0.014**
JJA	**0.17**	**$5.5e^{-12}$**	**0.49**	**0.0027**
SON	**0.13**	**$1.2e^{-06}$**	**0.63**	**0.0013**

The addition of only six years of data since the 2017 report has changed the temperature trends substantially. Every p-value decreased and every trend increased. The decrease in p-values is notable for the 30-year trends, which were predominantly larger than 0.05 in the 2017 report (1986–2015) and are predominantly less than 0.05 here (1992–2021). The acceleration in warming is higher as well, a result of the 30-year trends in this analysis being nearly twice as large as the 30-year trends in the 2017 report.

The Delaware Estuary and Basin is part of the Northeastern United States coastal region, which has warmed more rapidly than regions just inland as well as the North American average and the Northern Hemisphere average, according to a study by Karmalkar and Horton (2021). This study also provided evidence indicating that the high rate of warming is associated with the warming of Northeastern United States coastal waters, which, in turn, appears to be related to the slowdown of the North Atlantic Ocean's overturning circulation and the northward movement of the Gulf Stream.

Future Predictions

Future temperature changes in the Delaware Estuary and Basin are strongly dependent on the amount of future GHG emissions (Hayhoe et al. 2018). If GHG emissions continue to increase throughout the 21 st century (represented by the IPCC's RCP8.5 emissions scenario), global climate models (GCMs) project that some parts of the DEB will be 7 °C (12.6 °F) warmer at the end of the 21st century compared to 1986-2015. On the other hand, if emissions peak by mid-century and then decline (the RCP4.5 scenario), the DEB is projected to be 4 °C (7.2 °F) warmer. The IPCC issued its latest and sixth assessment report (AR6) beginning in late 2021 with GHG scenarios now broken into Shared Socio-economic Pathways, cross-referenced by the approximate level of radiative forcing, and the use of updated GCMs (IPCC 2021). To our knowledge, the AR6 GCMs have not yet been applied to studies of the Delaware Estuary and Basin region, but an analysis comparing projections of the AR6 GCMs to those of the previous generation of GCMs for North America reveals a smaller spread among models across the continent and slightly cooler and wetter projections for the Northeast United States (Martel et al., 2022).

Studies concur that air temperatures will increase in all seasons, but there is less agreement in the seasonality of warming. One study using the CMIP5 GCMs and RCP8.5 scenario, the largest increase

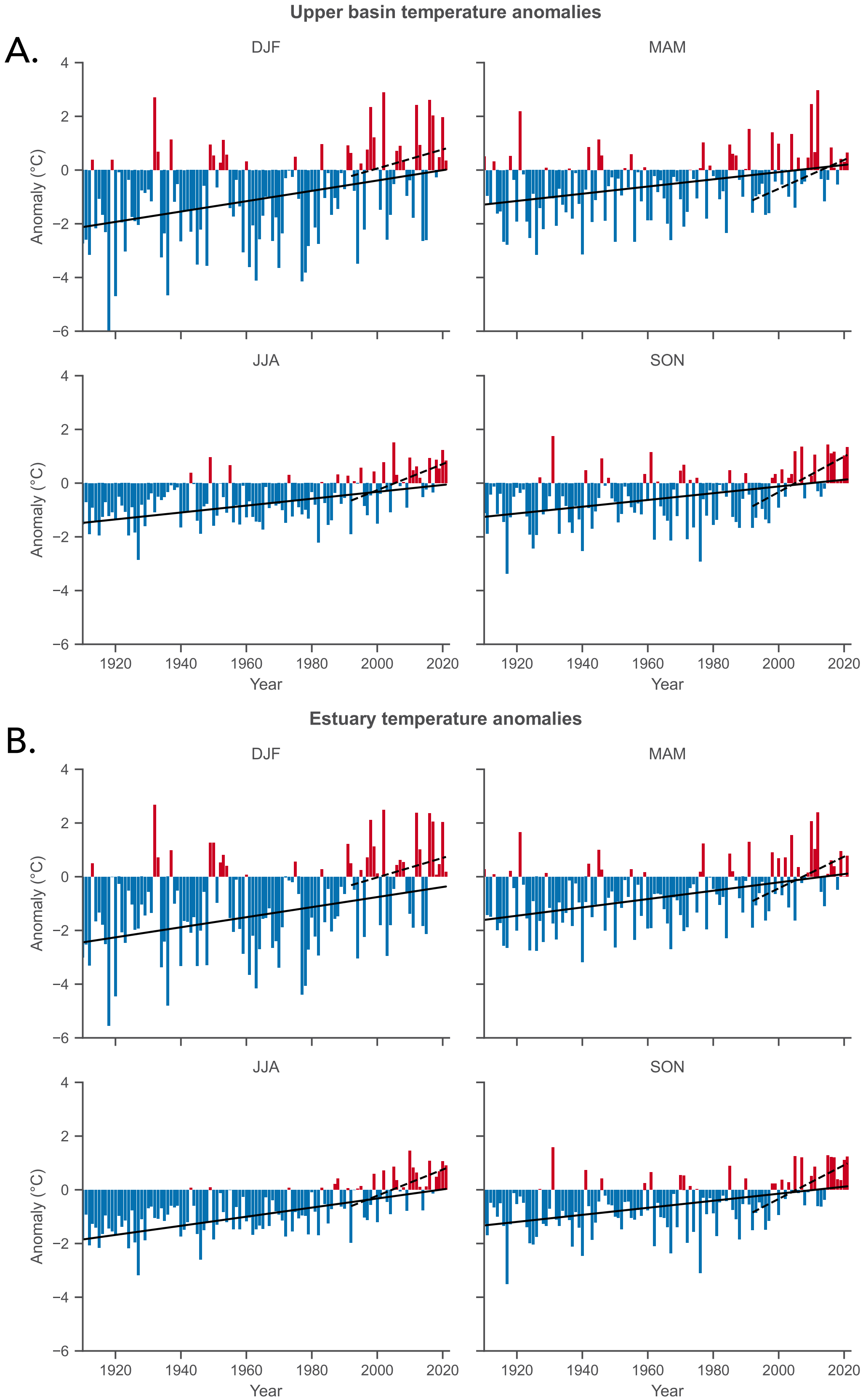


Figure 2.3 Seasonal mean temperature anomalies with respect to the 1991-2020 average in the Upper Basin (a) and Estuary (b). The solid and dashed lines are linear trends for the 1910–2021 and 1992–2021 periods, respectively.

in monthly mean air temperatures for 2041–2100, compared to means from 1971–2000, will occur in winter with a secondary peak in summer, but these changes will be more acute in the northern end of the Delaware Estuary and Basin (Lynch et al. 2016). A second study that employed a suite of regional climate models and the relatively high A2 emissions scenario produced a similar seasonal pattern that, on average, predicted greater winter warming in the northern region of the upper Basin, but higher summer warming in the Estuary region by the mid-21 st century (Rawlins et al. 2012). Finally, a third study that focused on a small watershed dominated by agriculture, located in the lower Estuary subregion of the Delaware Estuary and Basin, also predicted the greatest air temperature change during the winter months under both RCP4.5 and 8.5 climate scenarios for years 2021-2040, but with the least change occurring in the summer (Giri et al. 2020). The difference in future seasonal temperature predictions between these three studies could be due to several factors, such as the use of different GCMs, spatial resolution, climate scenarios, time periods, and land cover. Regardless, any increases in summer heating in the Estuary will have a greater impact on the more urbanized areas of the region, such as Philadelphia, which are already experiencing urban heat island (UHI) effects, especially in redlined communities (See Climate Feature 1).

Actions and Needs

The rapid rate of anthropogenic warming in the Delaware Estuary and Basin means that actions are needed now to adapt to climate change, which will only become more severe as GHG levels continue to increase. The analysis here has been focused on air temperature, but stream water temperatures are also increasing in the watershed by about 0.3 °C per decade (median of eight streams, Kaushal et al. 2010), similar to the air warming rates. It seems very likely that Delaware Bay is warming as well, like its estuarine neighbor just to the south, Chesapeake Bay (Hinson et al. 2022), but the underlying analysis has yet to be conducted.

Summary

The Delaware Estuary and Basin has warmed substantially over the past century, and the rate of warming has increased dramatically. This accelerated warming is probably the clearest and most dramatic indicator that the climate of the Delaware Estuary and Basin is changing rapidly. Identifying this rapid increase in temperatures is important because we are still striving to understand ecological responses to climate change, such as how increasing temperatures have affected vegetation and wildlife geographic range and species composition of communities, which will ultimately impact future ecosystem functions and longevity (Blois et al. 2013).

2.2 Precipitation

Description of Indicator

Precipitation, either as rain or snowfall, is important for maintaining terrestrial and aquatic habitats (see Chapter 6), as well as replenishing groundwater and drinking water reserves (see Chapter 3). However, too much precipitation causes flooding, which can be devastating to crops, infrastructure, and housing due to flooding.

As with temperature, monthly precipitation totals were acquired from the USHCN version 2.5 dataset and underwent the USHCN screening procedure similar to the procedure for temperature except there is no time-of-observation correction.

Past Trends

Annual precipitation has increased by 1.2 cm per decade with 95% confidence in the upper Basin, but not in the Estuary (Fig 2.4, Table 2.3). The estimated trend in annual precipitation over the most recent 30 years in the upper Basin is about three times larger than over the last 112 years, but is not statistically significant. In contrast to the upper Basin, the estimated recent 30-year trend in annual precipitation of 0.56 cm per decade in the Estuary is roughly 30% less than the trend over the past 112 years. Precipitation totals have increased significantly in the fall in both the upper Basin and Estuary over the last 112 years, but have only increased significantly in the summer in the upper Basin over the last 30 years (Fig 2.5, Table 2.3). The precipitation trends reported here are qualitatively similar to those in the 2017 report. The trends appear to be driven, in part, by increases in atmospheric blocking over Greenland and the negative phase of the North Atlantic Oscillation, which lead to enhanced moisture transport from the south (Simonson et al. 2022).

Table 2.3 Precipitation trends in the Delaware Estuary and Basin. Significant trends are bold (95% confidence).

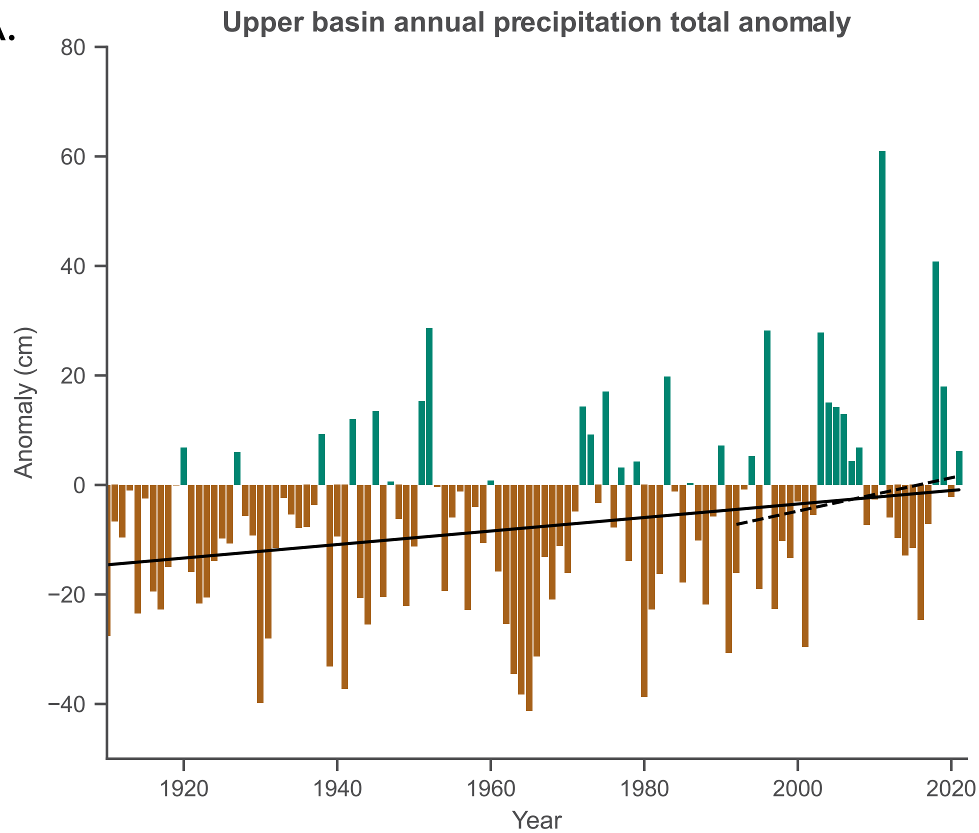
Seasonal Subset	Precipitation Trend (cm/decade) 1910-2021	p-value	1992-2021	p-value
		Upper Basin		
Annual	**1.2**	**0.013**	3.0	0.48
DJF	0.29	0.1	0.76	0.45
MAM	0.14	0.39	-0.77	0.69
JJA	0.27	0.36	**4.5**	**0.042**
SON	**0.69**	**0.012**	-0.59	0.75
		Estuary		
Annual	0.95	0.058	0.56	0.89
DJF	0.098	0.56	0.58	0.67
MAM	0.27	0.21	-1.3	0.35
JJA	0.12	0.71	3.8	0.087
SON	**0.69**	**0.0051**	-1.3	0.52

Warming has dramatically impacted snowfall and snow cover. The ratio of snow to total precipitation decreased throughout the northeast US, including the Delaware Estuary and Basin, from 1949 to 2005 (Feng and Hu 2007). A more recent study showed that the snow/rain event ratio declined by as much as 4% per decade between 1978 and 2019 (Shi and Liu 2021). Finally, days per year with snow cover declined by several days per decade from 1960 to 2019 (Ford et al. 2021).

Future Predictions

There is a strong model consensus towards future increased winter precipitation in the northern half of North America, including all of the Delaware Estuary and Basin (Hayhoe et al. 2018; Lynch et al. 2016; Rawlins et al. 2012; Thibeault et al. 2014). A performance-weighted average of regional climate model simulations under the A2 emissions scenario yields a 10-14% increase in winter precipitation throughout the Delaware Estuary and Basin by 2041–2070 (Rawlins et al. 2012). Under the RCP8.5 scenario, precipitation is projected to increase by 15-20% in winter and 10-15% in spring for most of the northeastern US, including the Delaware Estuary and Basin, by the late 21st century compared to 1986-2015 (Hayhoe et al. 2018). However, winter precipitation as snowfall is projected to decrease by 42-84% by 2099 across the

A.



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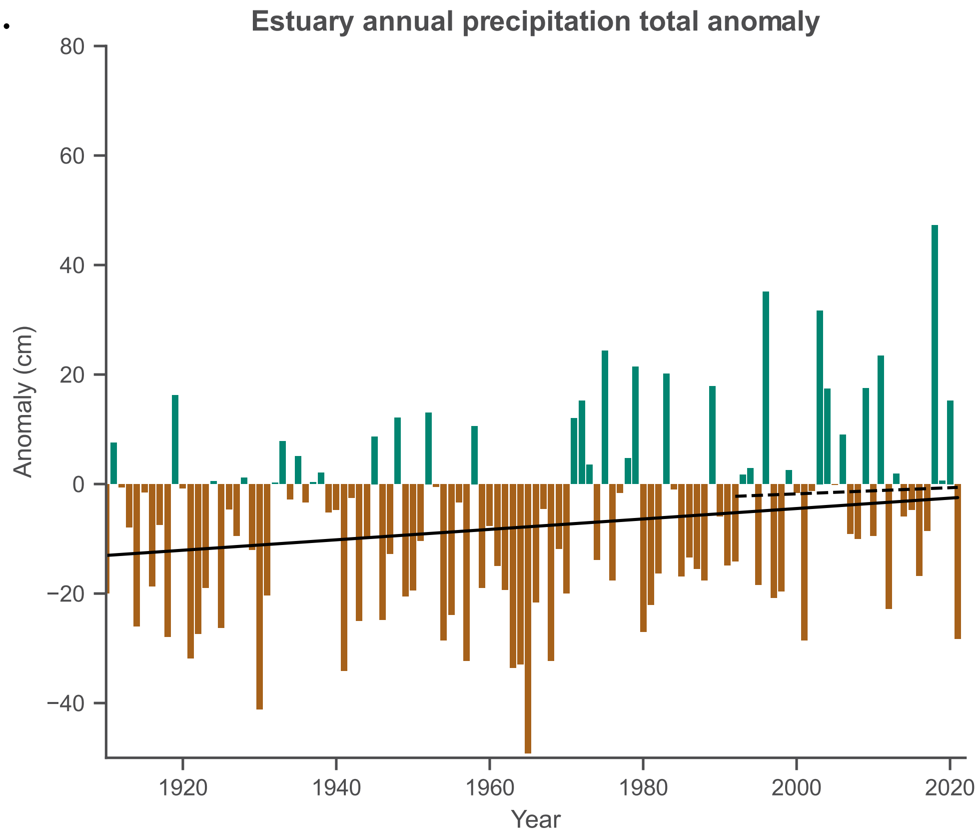

Figure 2.4 Annual mean precipitation anomalies with respect to the 1991-2021 average in the Upper Basin (a) and Estuary (b). The solid and dashed lines are linear trends for the 1910–2021 and 1992–2021 periods, respectively.

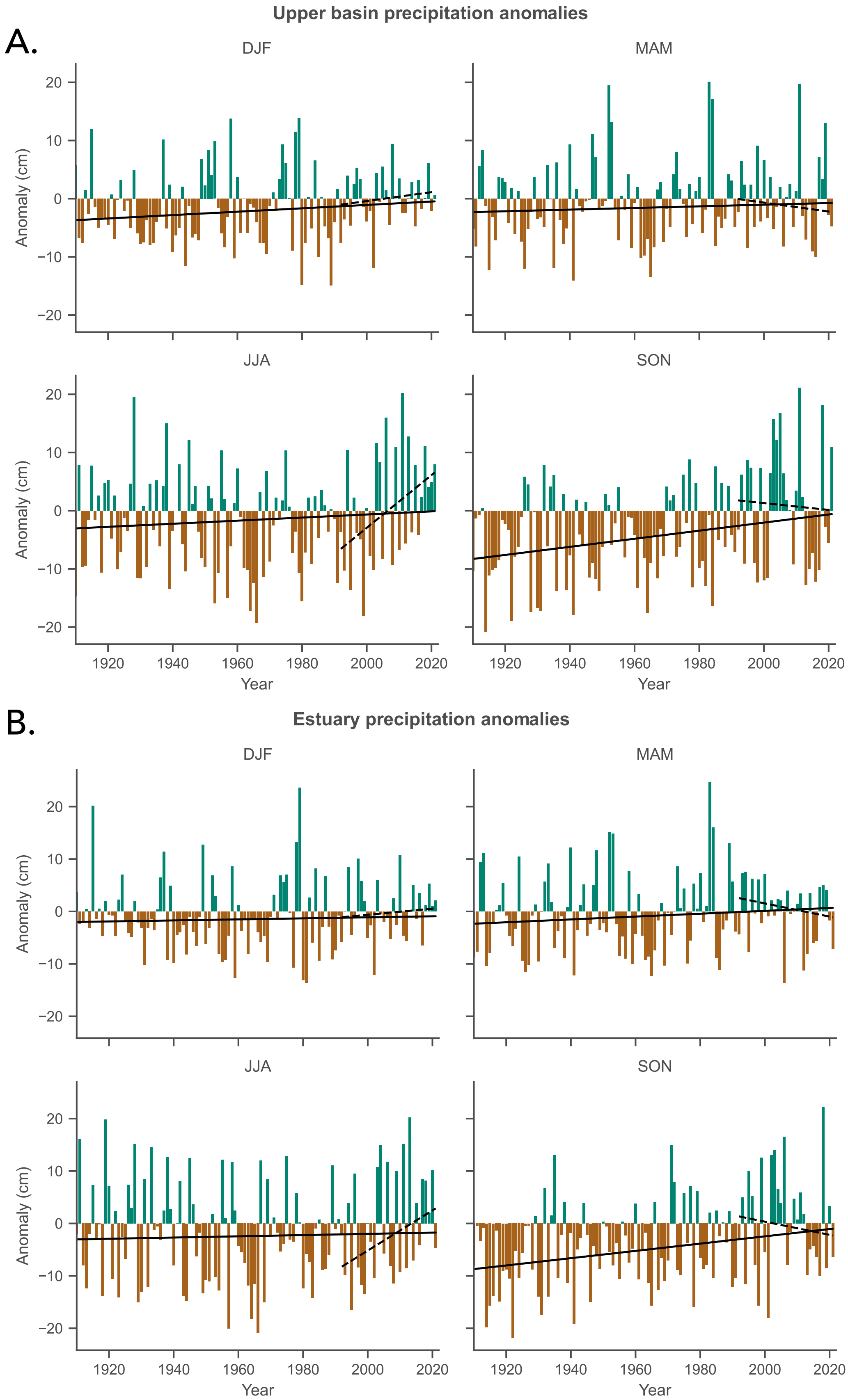

Figure 2.5 Seasonal mean precipitation anomalies with respect to the 1992-2021 average in the Upper Basin (a) and Estuary (b).

Delaware Estuary and Basin due to increasing temperatures causing a shift in the snow-rain transition zone (Ning and Bradley 2015; Hawkins and Woltemade 2021). Most studies also show that increased spring precipitation is likely in the Delaware Estuary and Basin, while there is less agreement in changes in summer and fall precipitation (Hayhoe et al. 2018; Hawkins and Woltemade 2021; Lynch et al. 2016; Rawlins et al. 2012).

Actions and Needs

The understanding of long-term changes in precipitation is not as clear as it is for temperature, where the effect of greenhouse gas emissions has been shown to be substantial. Although the increased annual precipitation observed in the Delaware Estuary and Basin and broader Northeast U.S. is consistent with the modeled effect of climate change on precipitation, the observed increase is greater than predicted by climate models (Knutson and Zeng 2018). Furthermore, climate models project that the largest increases in precipitation will occur in winter and spring, whereas the observed trends during the last century are largest in autumn. Understanding the causes of these differences is important for improving projections of future precipitation change in the Delaware Estuary and Basin.

The past decreases in snowfall and snow cover reported in the literature and continued declines that are projected indicate that dramatic changes in winter hydrology are underway. These changes not only represent a shift in the character of winter in the Delaware Estuary and Basin but should also be of concern to water resource managers and local economies that rely on winter recreation.

Summary

There is some evidence that annual and fall precipitation have increased in the Delaware Estuary and Basin (Table 2.3). Precipitation is projected to increase in the future, mainly during winter and spring. Several snow indicators have shown declines, which will likely continue into the future. Additional research is needed to reconcile the observed increase in autumn precipitation with climate model projections for increases in winter and spring.

2.3 Extremes: Air Temperature & Precipitation

Description of Indicator

Extreme air temperature and precipitation have direct impacts on human health and wellbeing, terrestrial and aquatic ecosystems, and infrastructure (see Chapters 1 and 3).

Trends in five extreme event indices were calculated: (1) Frost days, the number of days per year with low temperatures below 32 °F (0 °C); (2) Hot days, the number of days per year with high temperatures above 90 °F (32.2 °C); (3) consecutive dry days (CDD), the maximum number of successive days without precipitation per year; (4) R45, the number of days per year with heavy (> 4.5 cm or 1.78 in) precipitation; and (5) RX5day, the annual maximum five-day precipitation total (Figs 2.6 and 2.7, Table 2.4).

Daily data from the Berkeley Earth Surface Temperature (BEST; berkeleyearth.org) dataset were used to calculate the frost days and hot days indicators. Like the USHCN monthly temperature, and unlike most other daily temperature datasets, the BEST dataset contains daily station data that have been adjusted for relocations, changing observation times, and other inconsistencies. However, unlike other data used in this report, the BEST daily temperature data have been averaged onto a 1° x 1° latitude-longitude grid. The frost days and hot days indicators were calculated for each grid cell between latitude 39° to 42° N and longitude 76° to 74° W, which covers most of the Delaware Estuary and Basin, and then averaged over the grid cells. Both indicators used data from 1910 to 2021, consistent with the monthly temperature indicators.

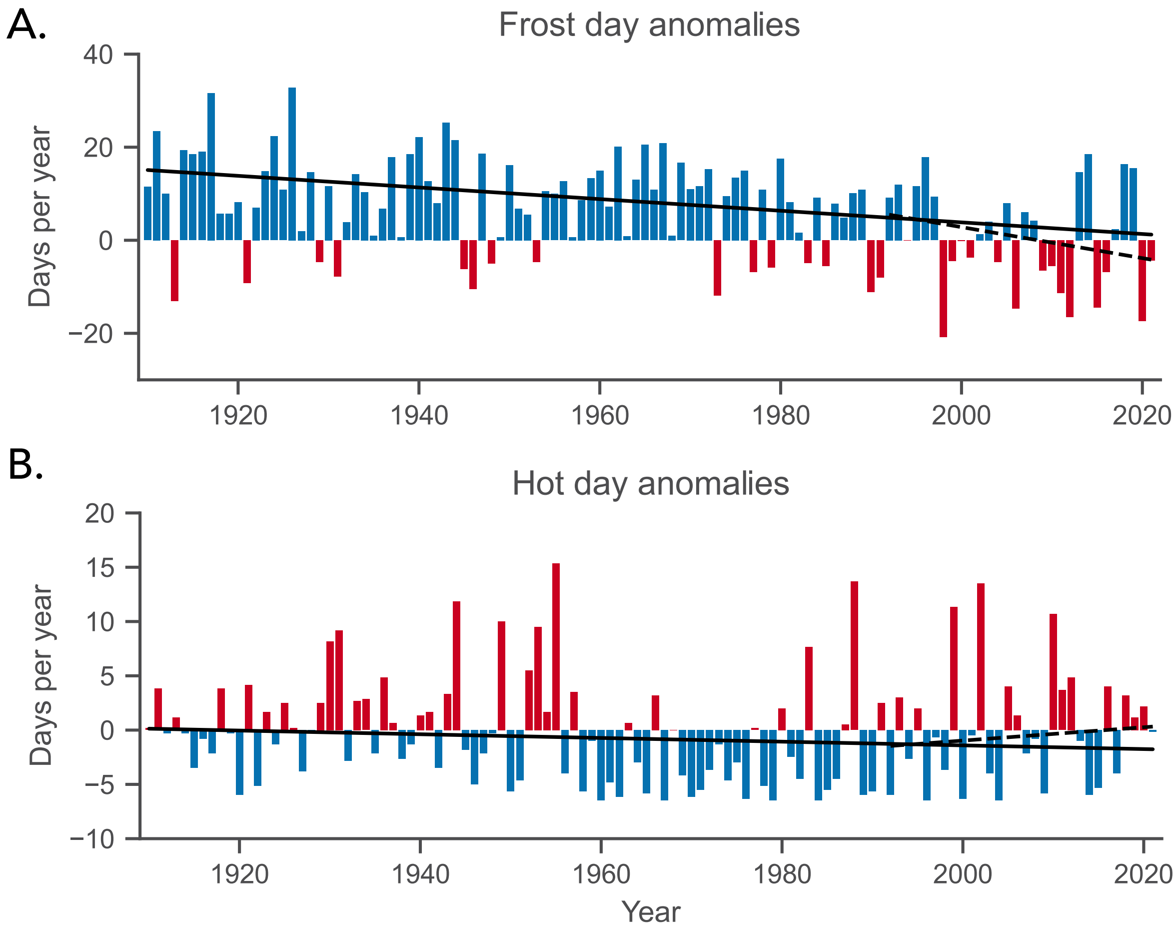

Figure 2.6 Temperature extremes: frost day (a) and hot day (b) anomalies.

Table 2.4 Trends in extremes in the Delaware Estuary and Basin. P-values are in parentheses, and significant trends are bold (95% confidence).

Metric	1991-2020 Average	Trend (per decade) 1910-2021	Trend (per decade) 1992-2021
Entire Watershed			
# of days below 32 °F	119	**-1.2 (0.00029)**	-3.3 (0.25)
# of days above 90 °F	6.49	-0.17 (0.19)	0.61 (0.45)
Upper Basin			
Annual max # consecutive dry days	15	-0.095 (0.26)	-0.3 (0.57)
# days/year with precip. >4.5 cm	3	**0.16 (0.00011)**	0.25 (0.41)
Annual max 5-day precip. total	11	0.057 (0.42)	-0.19 (0.8)
Estuary			
Annual max # consecutive dry days	18	0.13 (0.19)	-0.86 (0.53)
# days/year with precip. >4.5 cm	3.8	**0.20 (1.3e-06)**	0.36 (0.38)
Annual max 5-day precip. total	12	0.15 (0.055)	0.22 (0.71)

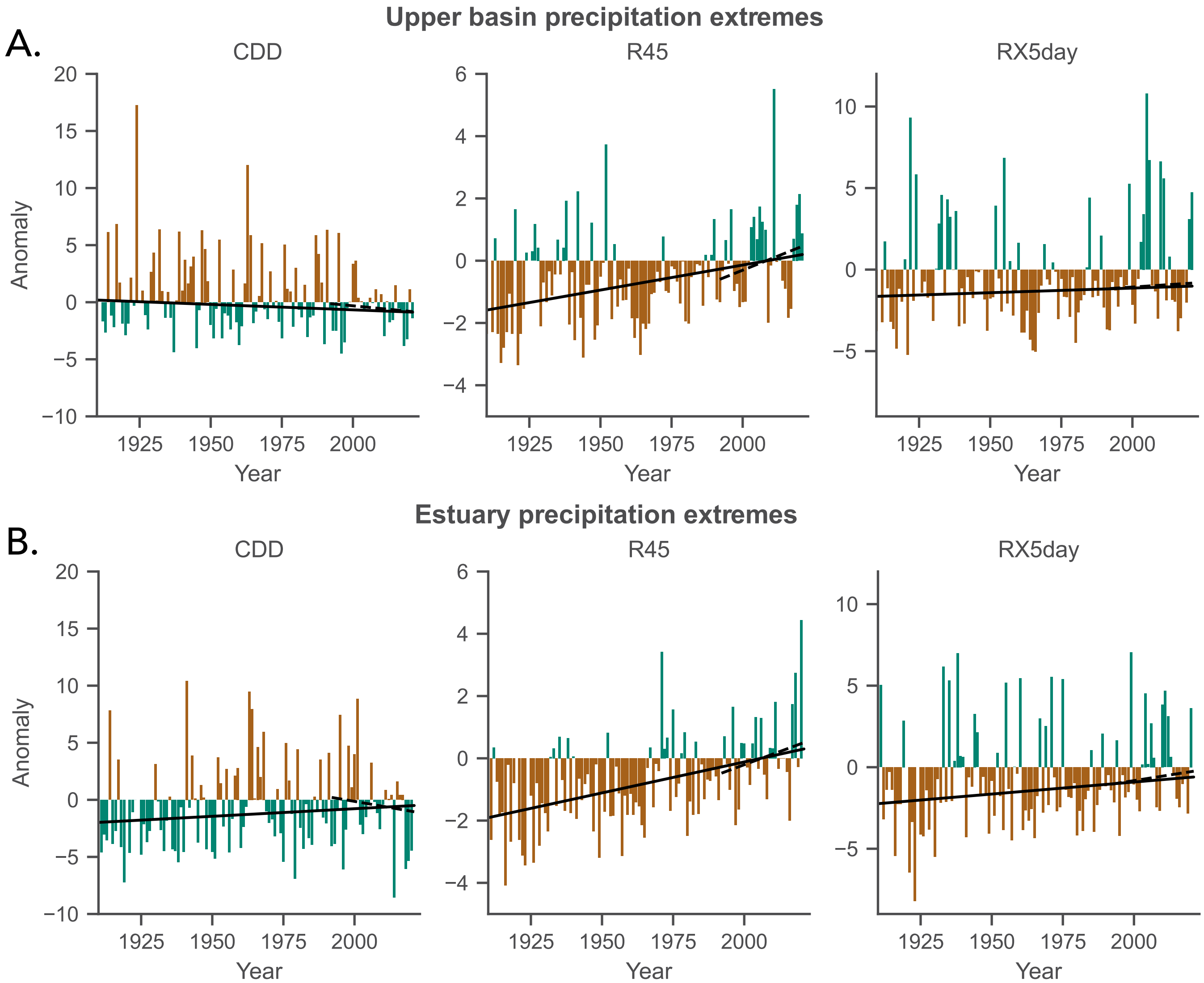

Figure 2.7 Precipitation extremes for the Upper Basin (a) and Estuary (b). Anomalies are with respect to the 1992-2021 average. CDD = annual maximum consecutive dry days, R45 = number of days per year with precipitation >4.5 cm, and RX45day = annual maximum 5-day precipitation total.

The Global Historical Climatology Network (GHCN) daily dataset collected at the USHCN stations (Table 2.1) was used for the precipitation analysis. Unlike the monthly data used for air temperature and precipitation analyses (see Sections 2.1 and 2.2), the daily data are not adjusted for changes in station location, instrumentation, or time of observation, which may result in significant biases and artificial trends. Precipitation data that were given any quality control failure flags in the dataset were removed. A day was deemed dry if the reported precipitation total was less than 1 mm. Missing days were assumed to be wet for the CDD metric and dry for the RX5day metric. Precipitation extreme indices were averaged over the upper Basin and the Estuary.

Past Trends

Table 2.4 presents the trends for the two extreme temperature indices, which are generally consistent with a warming climate. For the entire Delaware Estuary and Basin watershed there is a significant decrease in the number of frost days over the last 112 years by 1.2 days per decade, but the rate of decline in the number of frost days during the more recent 30-year period is almost three times higher (Fig 2.6A). Consistent with these results is the trend towards fewer freezing days found throughout the Northeast U.S. (Brown et al. 2010; Thibault and Seth 2014) and a declining trend in cold winters (Ford et al. 2021). Thibeault and Seth (2014) further showed that the decline in frost days throughout the Northeast US is less than predicted by GCMs from 1951 to 2010. Trends in the number of days above 90 °F in the Delaware Estuary and Basin are negative over the past 112 years, but are positive over the recent 30-year period; however, both trends are not significant at the 95% confidence level. These equivocal results are consistent with those of Thibault and Seth (2014), who examined several high temperature indices (though none exactly like ours) and found both increasing and decreasing trends from 1951 to 2010.

The two heavy precipitation indices, R45 and RX5day, are increasing except for the maximum 5-day precipitation in the upper Basin over the last 30 years (Table 2.4). However, the only significant trend is in R45, which shows an upward trend over the past 112 years of 0.16 and 0.20 days per decade for the upper Basin and Estuary, respectively (Fig 2.7). This may appear to be a small change but is, in fact, substantial, because there are so few days of heavy precipitation. The average number of R45 days per year for the 1991–2020 reference period is about 3 to 4, which is an increase of about 50% compared to the average over the past 112 years (Table 2.4). Consistent with our results are the findings of increases in extreme precipitation in the Delaware Estuary and Basin by Howarth et al. (2019) and in the Northeast US by Thibault and Seth (2014), who also showed that the observed trends are of the same sign but about half the magnitude of those simulated by GCMs from 1951 to 2010.

Tropical systems appear to be responsible for the increases in extreme precipitation in the Delaware Estuary and Basin (Howarth et al. 2019), as they are for total precipitation (Simonson et al. 2022). Indeed, globally, as well as in the North Atlantic Ocean, the fraction of tropical cyclones that are major (sustained winds greater than 50 meters per second), have increased significantly since 1980 (Kossin et al. 2020). The same is true for the number of landfalling major tropical cyclones, though the trend is more significant at the global scale than for the North Atlantic (Wang and Toumi 2022).

There are no significant trends in the maximum number of consecutive dry days per year in the Delaware Estuary and Basin (Table 2.4), a result that is consistent with observations and GCM simulations of the Northeast US during 1951–2010m (Thibeault and Seth 2014).

In general, the trends in extremes reported here are similar to those in the 2017 report.

Future Predictions

Both extreme wet and extreme dry events are expected to become more common by the end of the 21st century (Kreeger et al. 2010; Hayhoe et al. 2018; Wuebbles et al. 2014; Janssen et al. 2014; Thibault and Seth 2014), with larger changes in scenarios of higher GHG emissions. The frequency and intensity of heavy precipitation events are projected to increase over 40% compared to events observed between 1986–2015 based on the high RCP8.5 scenario (Hayhoe et al. 2018). In addition, the frequency, intensity, and size of North Atlantic tropical cyclones that could impact the Delaware Estuary and Basin with heavy precipitation are predicted to increase by the year 2095 compared to tropical cyclones observed between the years 1980-2005 (Marsooli et al. 2019). Other sources of tropical moisture, including atmospheric rivers, are also projected to increase (Hsu and Chen 2020). By the middle of the 21st century, climate models also project a large increase in the number of days per year above 90 °F in the Northeast US, and a decrease in the number of days below freezing in the Delaware Estuary and Basin, even under moderately low emissions scenarios (Horton et al. 2014; Williamson et al. 2016; Vose et al. 2017).

Actions and Needs

The stronger warming signal in the minimum temperature metric (frost days) compared to the maximum temperature metric (hot days) is broadly consistent with other studies (DeGaetano and Allen 2002; Brown et al. 2010). However, other studies have also found that trends in extreme temperature metrics are highly sensitive to the choices of time period and data homogenization method, and more detailed analysis is needed to examine the historical drivers of extreme temperatures and whether the trends are consistent with climate model projections. In addition, due to the size and variable topography and land use of the Delaware Estuary and Basin, changes in temperature and precipitation extremes may have high spatial variability. Future climate trend analyses of temperature and precipitation extremes are needed at higher spatial resolution to identify immediate needs of communities at high risk of climate change impacts, such as the Eastwick community in southwestern Philadelphia, PA (see Climate Features 1 and 2). Despite any uncertainties, the threat of increasing extreme temperature and precipitation is great enough to warrant actions to limit negative impacts through actions including green infrastructure development in urban areas, which will decrease heat stress and flooding.

Summary

The intensity and frequency of extreme temperature and precipitation events are difficult to examine directly and even harder to predict. Despite increased overall temperatures in the Delaware Estuary and Basin over the past century (see Section 2.1), no significant increase in high temperature extreme events was detected in this analysis. On the other hand, heavy precipitation events increased in frequency in both the upper and lower basin. Most climate scientists predict increasing extreme events in the future, such as tropical cyclones in the North Atlantic, but there is still a lot of uncertainty in predicting changes at small scales, such as those of the Delaware Estuary and Basin.

2.4 Streamflow

Description of Indicator

Streamflow, the volume of water flowing in a stream, river, or channel, is influenced by climate change through changes in precipitation, the timing and volume of snowmelt, and evapotranspiration.

Daily streamflow data measured in the Delaware River at Trenton, New Jersey, from 1913 to 2020 were obtained from the United States Geological Survey. Since the flow at Trenton is significantly influenced by upstream reservoir releases to meet an established flow objective set by the Flexible Flow Management Program, data from ten smaller, unregulated (natural flow) tributaries were also included (Delaware River Basin Commission 2021; United States Geological Survey 2017; Table 2.5). The tributaries were selected from those that are noted in the Hydro-Climatic Data Network dataset (Slack et al. 1993) as measuring natural, unregulated streamflow and having a complete record of quality data. We further limited the selection to stations having complete daily data during 1981 to 2020. The gauges are concentrated in the upper Basin; only two are south of Trenton (see Fig 2.1). Data from the tributaries were analyzed for trends for the years 1958 to 2020 when data were available at every gauge. Like other daily data, flow data from the Delaware River and the tributaries were filtered to remove years and seasons with more than 5 days of data missing in any month.

To homogenize the tributary river data, standardized anomalies were calculated based on the years 1991-2020 for each gauge. The standardized anomaly Q' was calculated as:

$$Q' = \frac{Q - \bar{Q}}{Q_\sigma}$$

where Q is the time series of annual or seasonal mean streamflow, $\bar{Q}$ is the 1991–2020 mean of the time series, and Q_σ is the 1991–2019 standard deviation of the time series.

Table 2.5 Unregulated tributaries included in streamflow analyses.

Station ID	Name
01413500	East Br Delaware R at Margaretville, NY
01414500	Mill Brook near Dunraven, NY
01415000	Tremper Kill near Andes, NY
01423000	West Branch Delaware River at Walton, NY
01435000	Neversink River near Claryville, NY
01439500	Bush Kill at Shoemakers, PA
01440000	Flat Brook near Flatbrookville, NJ
01440400	Brodhead Creek near Analomink, PA
01466500	McDonalds Branch in Lebanon State Forest, NJ
01484100	Beaverdam Branch at Houston, DE

Past Trends

Streamflow at Trenton, NJ, has varied substantially over the past 109 years, with some years departing from the 1991-2020 mean of 365 m^3 per second, shown as anomalies, by more than 50% (Figs 2.8 and 2.9, Table 2.6). It is notable that the four largest positive anomalies in annual streamflow have occurred since 1998. Aside from a large increase in winter streamflow over the full time period (1913-2020), no trend in streamflow was statistically significant at the 95% confidence level. Despite lack of significance, the trends over the last century are generally consistent with trends over the last 30 years, with positive trends observed in winter and summer and negative trends in spring.

Since the streamflow at Trenton is highly regulated, trends at the natural flow tributary sites give insight to climate change effects on streamflow across the Delaware Estuary and Basin. Overall, the seasonal trends observed at Trenton are similar to the trends at the smaller tributaries (Fig 2.10). Most unregulated sites show increasing trends in streamflow in the winter, summer and fall, but negative in the spring. Although only a few trends are statistically significant, positive trends are present at every one of the 10 unregulated gauges except one in winter, two in summer, and one in autumn. Trends at all 10 gauges are negative in spring, although the spring trends tend to be closer to zero, meaning little change over time, than the trends in the other seasons.

The streamflow trends at Trenton and the unregulated site are qualitatively consistent with those in the 2017 report.

Low flows in streams can have negative ecological impacts and hence trends in associated indices, such as the 7-day period in a year with the lowest mean flow (annual 7-day low flow), are important for monitoring and management. According to Hammond and Fleming (2021), throughout most of the Delaware Estuary and Basin, annual 7-day low flows have increased as a result of increasing precipitation. The study showed, however, annual 7-day low flows decreased in the Coastal Plain province, where water use and impervious surface area have increased.

A notable feature of streamflow in the Delaware Estuary and Basin is the prominent drought of the 1960s (Fig 2.8), which can also be seen in precipitation (Fig 2.7). Indeed, a water budget analysis revealed that low precipitation, not high temperature, was the dominant driver of the very low streamflow of the 1960s (McCabe and Wolock 2020). The drought was dramatic in its impact, leading to saltwater intrusion that threatened drinking water supplies (Hull and Titus 1986). Nevertheless, a 1,500-year record of drought from tree rings revealed that the 1960s were not historically unusual (McCabe and Wolock 2020). Remarkably, the cause of the 1960s drought is still poorly understood. Using GCMs, Seager et al. (2012) argued that the drought was the result of natural variability of the atmosphere. Schulte et al. (2016), on the other hand, conducted a wavelet analysis that showed that the Southern Oscillation, a natural feature of the coupled ocean–atmosphere system, could account for 40% of the streamflow anomaly associated with the drought.

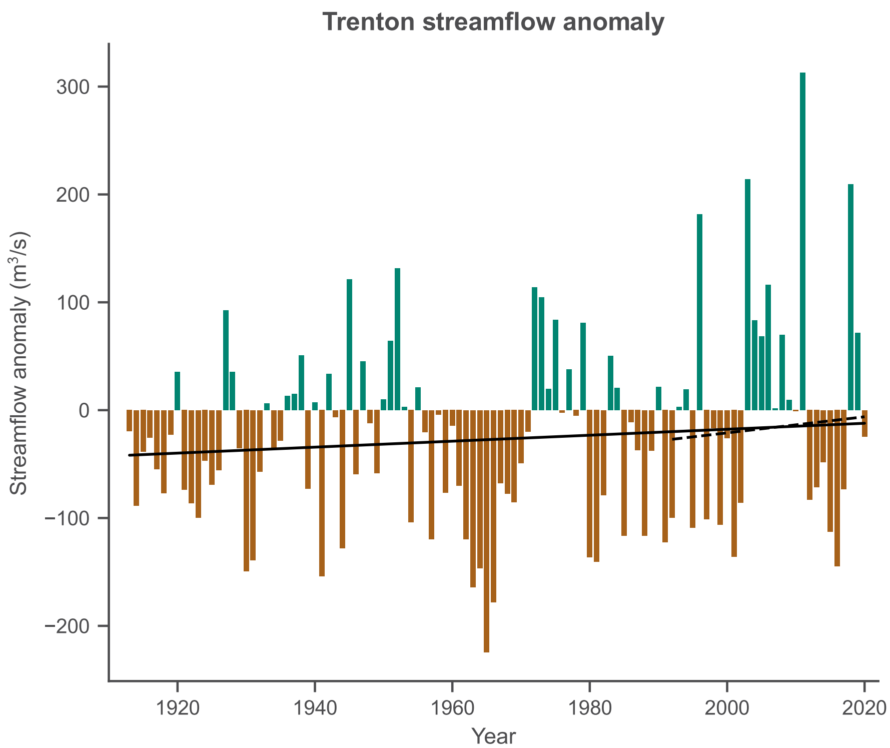

Figure 2.8 Time series of annual average streamflow anomalies (with respect to the 1991-2020 average of 365 m³ per second) at Trenton, NJ. The solid and dashed lines are linear trends for the 1913-2020 and 1991-2020 periods, respectively.

Future Predictions

Hydrological model simulations forced by GCMs project decreasing runoff from April through November in some areas of the Delaware Estuary and Basin (Williamson et al. 2016). Annual mean runoff, however, is predicted to increase, primarily as a result of increased winter precipitation (Williamson et al. 2016, Hawkins and Woltemade 2021). The positive trend observed in winter streamflow, which is consistent with the previous 2017 report but also now statistically significant, is in line with climate projections for increased winter precipitation and decreased snow storage (see Section 2.2). Model simulations of the nearby Chesapeake Bay watershed also show increasing winter runoff, although a decrease in annual mean runoff becomes more likely with higher emissions scenarios and later time periods (Hawkins 2015). The application of GCM projections to hydrologic and hydraulic models in Pennsylvania revealed that flood hazards and exposure are projected to increase, especially in the central Delaware River Basin (Sharma et al. 2021).

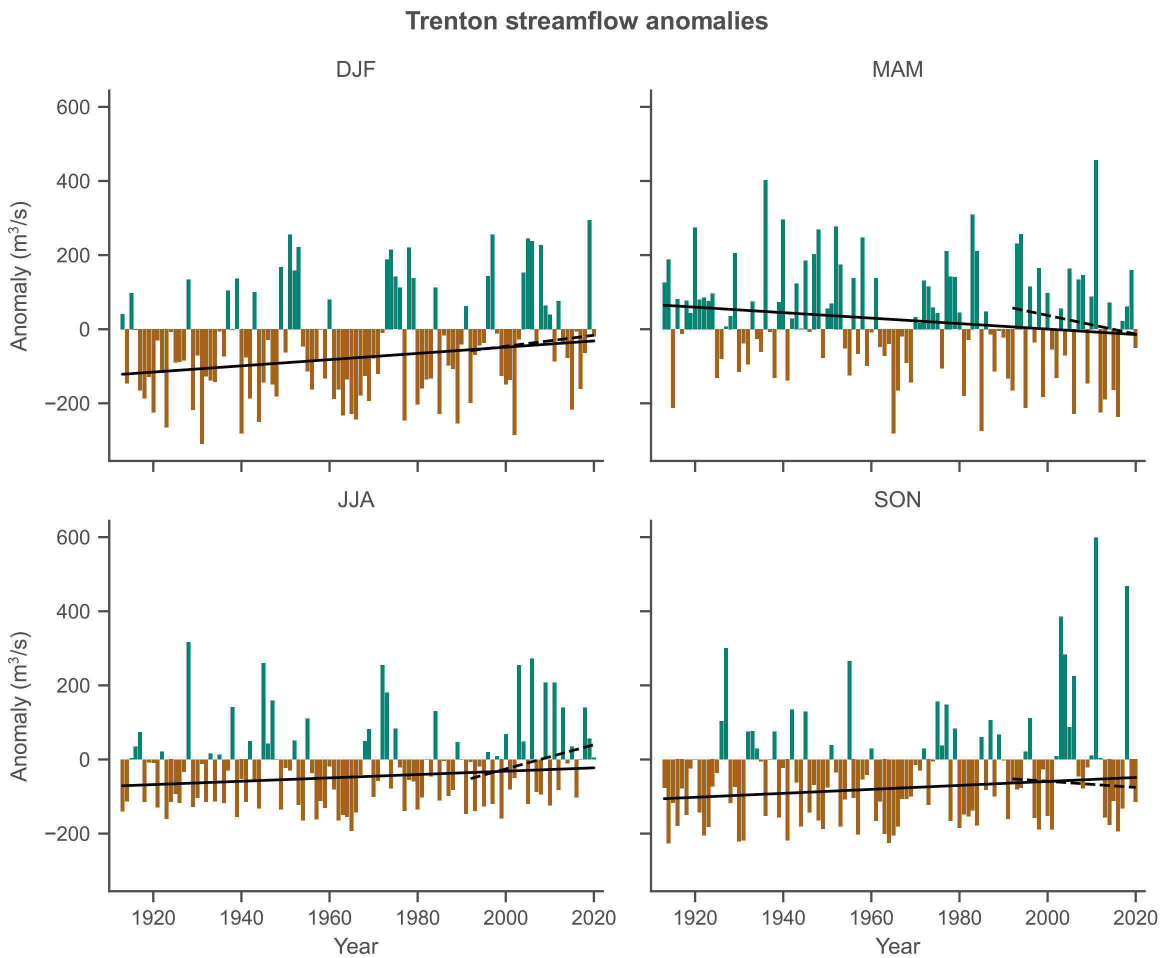

Figure 2.9 Time series of seasonal mean streamflow anomalies (with respect to the 1991-2020 average) at Trenton, NJ. The solid and dashed lines are linear trends for the 1913-2020 and 1991-2020 periods, respectively.

Table 2.6 Streamflow trends. Significant trends are bold (95% confidence).

Seasonal Subset	Streamflow (m^3 per second per decade)			
	1910-2020	p-value	1991-2020	p-value
Annual	2.8	0.29	7.5	0.81
DJF	**8.4**	**0.028**	15	0.69
MAM	-7.4	0.12	-25	0.44
JJA	4.5	0.1	33	0.061
SON	5.4	0.15	-8.2	0.72

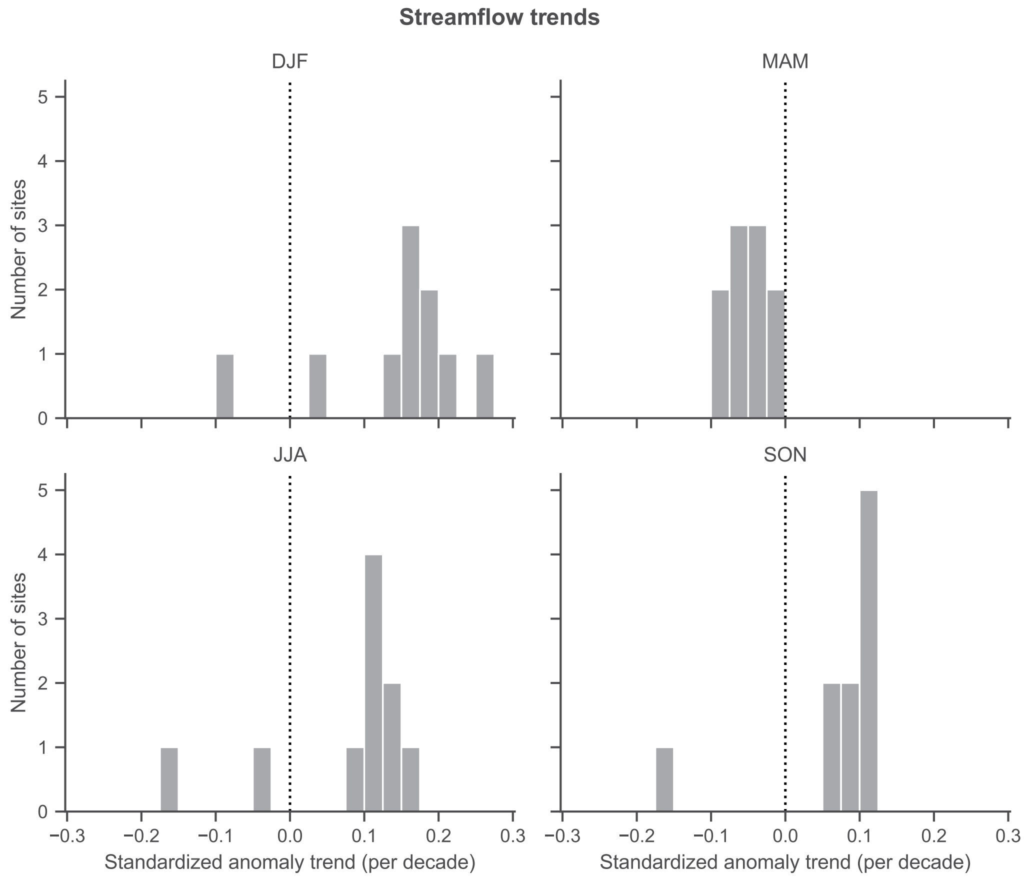

Figure 2.10 Histograms of trends in standardized seasonal streamflow anomaly at the 10 tributary sites during 1958-2020.

Actions and Needs

Continued monitoring of stream and river flows is critically important to track changes in the water budget of the Delaware Estuary and Basin, especially within the Estuary where there are only a few monitoring stations, which affects estuarine salinity and freshwater availability for people and the environment (see Chapter 3). Future changes in flow, particularly in winter and summer, are likely to be driven by a combination of the amount and type of precipitation (see Sections 2.2 and 2.3) and evapotranspiration changes, which may not be simulated well by global climate models, so understanding and accounting for uncertainty in future flow is necessary.

Summary

Most streamflow trends in the Delaware River and its tributaries are not statistically significant. In the future, increased streamflow is expected in winter and early spring, primarily as a result of increased precipitation in the form of rainfall, and reduced streamflow in the summer is possible.

2.5 Ice Jams

Description of Indicator

Ice jams occur primarily when chunks of ice flowing down river clump together, eventually blocking the flow of the river and causing sudden flooding of river banks and the surrounding areas. They are considered a secondary indicator of climate change as ice jams are influenced by air temperature and precipitation.

Occurrences of ice jams were obtained from the Ice Jam Database of the U.S. Army Cold Regions Research and Engineering Laboratory (White 1996). The database contains reports of ice jams in numerous rivers of the northern United States. Analyzed here are annual counts (by water year) of ice jams occurring anywhere on the Delaware River from Trenton upstream, including the West and East Branches.

Past Trends

The number of ice jams that have been reported over the past century in the Delaware River has been declining (Fig 2.11). This is possibly a result of underreporting of ice jams in the more recent past (White 1996). However, winter warming of the watershed has occurred (see Section 2.1, Table 2.2), which is expected to lead to fewer ice jams. Indeed, as Fig 2.12 shows, there is a moderate negative correlation between the number of ice jams and the upper Basin winter mean temperature. Using the same non-parametric Mann-Kendall method that was used to determine the trend in other indicators, the correlation between upper watershed winter mean temperature and the number of reported ice jams is -0.33 ($p < 1\text{x}10^{-4}$).

Future Predictions

It is reasonable to expect fewer ice jams in the future due to predicted higher winter temperatures (see Section 2.1) and decreases in the number of frost days (see Section 2.3).

Actions and Needs

Declining ice jams may be one of the clearer impacts of a warming winter climate, but reporting issues make ice jams a problematic indicator. Other indicators of changing winter climate should be considered in future TREBs, including the snow/rain event ratio (Shi and Liu 2021) and days per year with snow cover (Ford et al. 2021).

Summary

The frequency of ice jams along the Delaware River has decreased, and the decline is directly correlated with the increasing mean winter temperature across the watershed. Since winter temperatures are predicted to increase markedly in the future, ice jams are likely to become less frequent. Inconsistent ice jam reporting suggests that other indicators of winter climate, such as snowfall and snow cover, may be more robust.

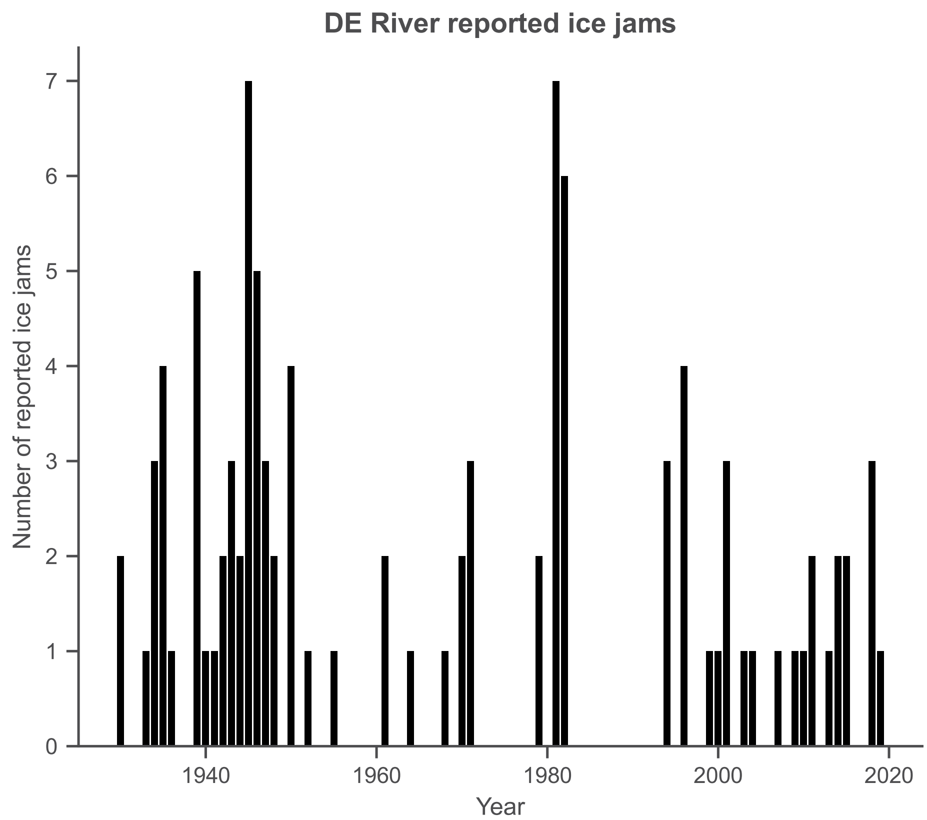

Figure 2.11 Reported ice jams on the Delaware River.

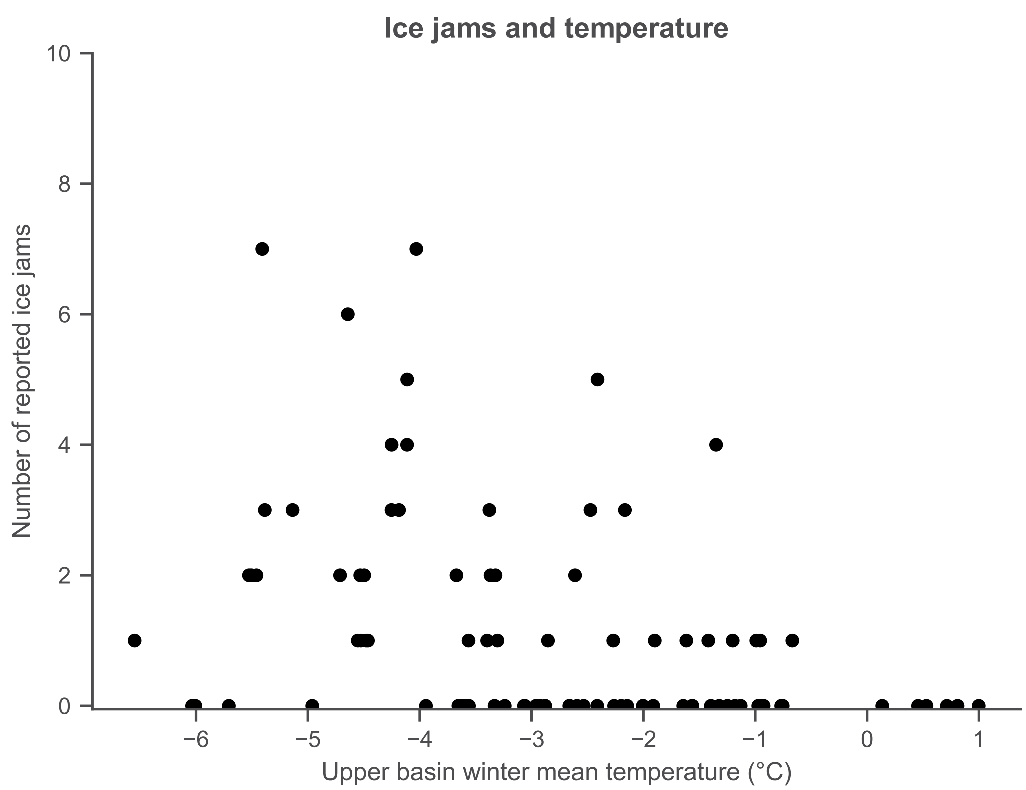

Figure 2.12 Comparison of the number of ice jams and winter temperatures.

2.6 Sea Level

Description of Indicator

Sea level is an indicator of climate change as it is impacted by increasing global air temperatures in a variety of ways, such as the addition of water to the oceans by ice-mass loss and by thermal expansion. Relative sea level is also affected by other factors, such as isostatic rebound and subsidence due to groundwater and oil withdrawals. Rising sea levels are a major threat to global coastal communities and estuarine habitats because of increased high tide flooding, infiltration of saline water into groundwater systems, and upstream progression of the salt front which can impact drinking water intakes (see Chapter 3). The long-term rate of global sea level rise from 1900 to 2018 is about 1.6 cm per decade, but more recent rates (1993–2018) indicate global mean sea level is increasing at 3.4 cm per decade (Frederikse et al. 2020).

Mean sea levels within the Estuary were obtained from the Permanent Service for Mean Sea Level database for three locations: (1) Philadelphia, Pennsylvania (Station ID: 135), (2) Cape May, New Jersey (Station ID: 1153), and (3) Lewes, Delaware (Station ID: 224) (Fig 2.1). Sea level data are annual means calculated from monthly means with gaps of up to 3 months filled using linear interpolation. Recent publications were also reviewed to identify historical trends in tidal range, connections to mean sea level, and projected changes.

Past Trends

Within the Delaware Estuary, both the long-term and recent sea level trends are positive and significant at the 95% confidence level (Table 2.7). Over the past 120 years, sea level at Philadelphia has been rising at a rate of 3.07 cm per decade (Fig 2.13). From 1992 to 2021, sea level at Philadelphia has increased at a rate of 4.7 cm per decade, indicating that the rate of sea-level rise is increasing in the Delaware Estuary, as it is globally. At the mouth of Delaware Bay, sea level has risen at an average rate of 5.7 cm per decade over the past 30 years, for a total of about 17 cm (7 inches). These high rates of sea-level rise are exacerbated by geologic contributions of around 2 cm per decade (Kopp 2013). Sea-level rise has contributed to increasing salinity throughout much of the estuary (Ross et al. 2015).

Table 2.7 Sea level trends in the Delaware Estuary. P-values are in parentheses and significant trends are bold (95% confidence).

Station	Sea level trends (cm per decade)	
	1901-2021	1992-2021
Philadelphia, PA	**3.07 (< 1e^{-4})**	**4.7 (0.00025)**
Cape May, NJ	-	**5.8 (3.9e^{-08})**
Lewes, DE	-	**5.6 (1.1e^{-07})**

Tidal amplitudes and overall tidal ranges in the Estuary have also varied historically, in part due to sea-level rise (Ross et al. 2017). Anomalies in the highest astronomical tide at tide gauges throughout the Estuary have been positively correlated with anomalies in mean sea level (Devlin et al. 2019). Channel deepening has also influenced tidal range, causing it to more than double over the first half of the 20th century in the upper estuary, between Philadelphia and Trenton (DiLorenzo et al. 1993; Pareja-Roman et al. 2020).

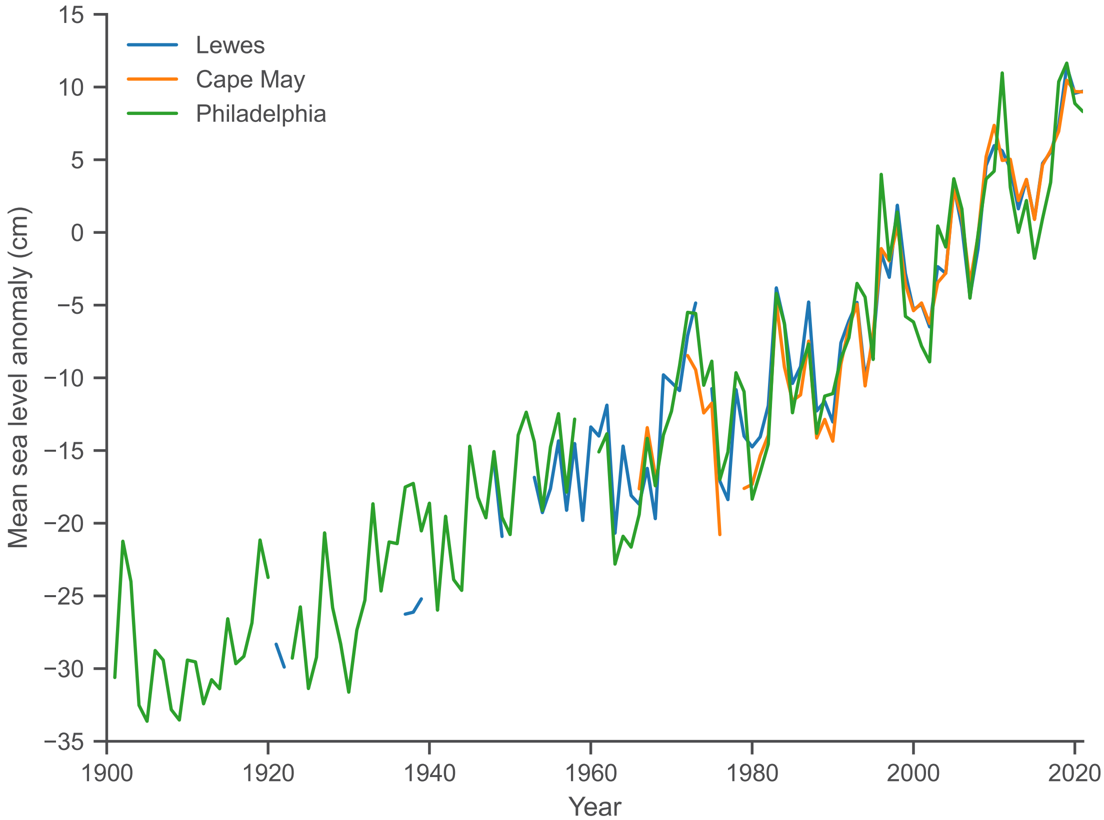

Figure 2.13 Annual mean sea level anomalies with respect to the 1991-2020 means.

Increases in tidal range have increased the frequency of "nuisance flooding" during high tides at Philadelphia (Li et al. 2021). The water level height when nuisance flooding occurs is based on the mean high high water level at each station, which is about 1.9 ft (0.58 meters) above sea level for the locations discussed here (Sweet et al. 2022). At Philadelphia, nuisance flooding events have been increasing over the past 20 years with a peak of 15 instances in 2011 alone (National Ocean Service). The monitoring stations at Lewes, Delaware, and Cape May, New Jersey, have also experienced increases in high tide flooding events since the 1980s (National Ocean Service).

Future Predictions

Mean sea level along the contiguous United States is predicted to increase by 30 cm by the year 2050 (Sweet et al. 2022). Based on the RCP8.5 low and high probability scenarios, sea level in Delaware Bay is projected to increase between 52 cm and 153 cm by the year 2100 when compared to mean sea level observed in the year 2000 (Callahan et al. 2017). In the near term, rising mean sea level and an increase in tidal range associated with a regular 18.6-year tidal cycle are projected to result in a substantial increase in nuisance flooding in the 2030s and moderate flooding in the 2050s (Thompson et al. 2021).

The positive historical correlation between tidal range and mean sea level suggests that future sea level rise is likely to continue to cause increased tidal range beyond the expected cyclical increase in the 2030s. Model simulations agree with this hypothesis; Lee et al. (2017) projected that 1 m of sea-level rise would cause mean tidal range in the upper Estuary to increase by 25 cm, with smaller increases in the Bay. However, Lee et al. (2017) found that the simulations were sensitive to the treatment of shorelines in the model. Sea-level rise increased tidal range when the coast was modeled like a hardened shoreline; if instead low-lying coastal land was inundated as the sea level rose, the model tidal range decreased.

Saltwater intrusion is likely to increase in the Delaware Estuary as a result of sea-level rise (Ross et al. 2015). Such salinity increases are projected to negatively impact industry, including electricity generation (Shirazi et al. 2019), and reduce the large tidal-fresh portion of the estuary, which harbors important and unique species.

Actions and Needs

Coastal communities across the United States are already feeling the impacts of rising sea levels. Adaptations currently used to prevent or reduce flooding due to rising sea levels range from sandbags to living shorelines to higher seawalls, even including elevating structures. However, future predicted sea levels compounded with increasing storm intensities will likely be devastating to coastal communities, regardless of adaptations (see Climate Feature 2). For example, the heights of peak storm tides of cool-season storms, like Nor'easters, are predicted to increase in counties along the Delaware River by the late 21st century because of more intense storms tracking further inland (Pringle et al. 2021). With a projected additional increase in sea level of over 1 meter along the Delaware River, tropical and cool-season storms will become more damaging and far reaching. The Delaware Estuary and Basin is already starting to experience unprecedented storms and their impacts. After landfall in Louisiana as a category 4 storm, Hurricane Ida stalled over the northeastern US in September 2021, causing flash flood events and a storm surge of up to a meter above normal tides (Beven II et al. 2022). Increases in tidal range due to sea-level rise and periods of higher range in the 2030s and 2050s will exacerbate due to flooding storms, particularly in the upper Estuary; detailed projections of these changes and adaptations to cope with regular flooding are urgently needed.

Severe flooding may occur when multiple flood types, such as pluvial flooding (directly due to heavy local precipitation), fluvial flooding (due overflowing river banks), and coastal flooding (due to high tides and storm surge), happen simultaneously. Such compound flooding is likely to increase in the future in many parts of the world, including the Delaware Estuary, as a result of heavy precipitation, higher sea-levels, and stronger storms. Most of the United States is expected to see increases in flooded area from 2020 to 2050 (Bates et al. 2021) that will disproportionately impact Black communities (Wing et al. 2022). Coastal areas, including the Delaware Estuary, are expected to be hardest hit, and the effects by the end of the century under high emissions are projected to be dramatic. Specifically, a severe compound pluvial and storm surge flood, which is when 100-year extreme rainfall occurs simultaneously with 100-year extreme sea level, is projected to change from an occurrence every 270 years to every 7 years on average throughout the Delaware Estuary, a 36-fold increase in likelihood (Gori et al. 2022).

Summary

Changes on decadal to century time scales in sea level and tidal range are indicators of climate change and can have drastic impacts on coastal communities along the Delaware Bay and within the tidal reach of the lower Delaware River. At Philadelphia, the rate of sea level rise has increased and is correlated with increases in tidal range. These and future projected increases will impact other climate change indicators, such as flooding events, particularly within communities already impacted by nuisance flooding (see Climate Feature 2), changes in coastal habitats (see Chapter 6), and movement of the salt front (see Chapter 3). Without an appreciable reduction in GHG emissions, in addition to adaptations to modern increases in sea level, communities and habitats along the Delaware Bay and River will likely experience severe negative impacts.

Take-Home Messages

This chapter describes how the climate of the Delaware Estuary and Basin has changed in the past using historical data collected across the watershed and bay and discusses how climate may change in the future based on model projections from scientific literature. By conducting our own data analysis for past changes and using the peer-reviewed literature for past and future changes, an overall picture of dramatic climate change in the Delaware Estuary and Basin emerges, one that is largely consistent with expectations from increases in greenhouse gases (Table 2.8). The influence of global warming on primary indicators (temperature, precipitation, and sea level) and secondary indicators (ice, snow, and coastal flooding) is clear.

The three primary indicators that most closely track expectations from global warming are mean air temperature, frost days, and sea level, which are not only increasing but also accelerating, trends that are extremely likely to continue. Historic trends in hot days are more equivocal, but future increases are all but certain. Although our analysis of historic extreme precipitation trends depends on the particular metric considered, the totality of evidence, including published studies, indicates that very strong rain events are increasing. Future increases in such extremes are very likely. Winter streamflow is increasing and is predicted to continue to increase, but changes in streamflow throughout the year and by how much will depend on changes in precipitation, air temperature, water use, and impervious land cover.

Coastal flooding and ice and snow indices are considered secondary indicators of climate change because they are directly influenced by primary indicators, such as air temperature and precipitation. Nuisance flooding events along the coast have increased in frequency and are expected to further increase with rising sea levels, and storm surges and heavy precipitation caused by increasingly stronger storms. Ice and snow indicators, such as ice jams and snowfall, have decreased in extent and frequency due to warming and will be further reduced in the future.

Climate across the Delaware Estuary and Basin is rapidly changing. Although climate change will affect every corner of the Delaware Estuary and Basin, vulnerable communities, such as redlined neighborhoods and residents living within high-risk flood zones, will experience the biggest impacts. To decrease the severity of future climate change, emissions of greenhouse gases into the atmosphere must be immediately reduced. Now is also the time to enact adaptation strategies to prepare for the changes to come. Communities, natural habitats, and infrastructure must be made more resilient to these future changes through, for example, the creation of green spaces, increased protections of ecosystems, reduction of impervious surfaces, increased access to public health services, and prioritizing affordable housing in low risk areas.

Table 2.8 Summary of historical and future climate impacts on the Delaware Estuary and Basin.

Climate Change Indicator	Trends: Historical	Trends: Future	Main Messages
Air Temperature	↑ (High)	↑ (High)	Warming is occurring in all seasons.
Days over 90 °F	↑ (Low)	↑ (High)	
Days under 32 °F	↓ (High)	↓ (High)	
Precipitation	↑ (Medium)	↑ (Medium)	Precipitation is increasing and becoming more intense.
Extreme Precipitation	↑ (Medium)	↑ (High)	
Streamflow	↑ (Low)	↑ (Low)	Winter streamflow is increasing.
Sea Level	↑ (High)	↑ (High)	Sea level is increasing rapidly.
Coastal Flooding	↑ (High)	↑ (High)	Occurrences of flooding are increasing.
Ice and Snow	↓ (Medium)	↓ (High)	Ice and snow indices are decreasing due to warming.

Confidence in Trend

- ↑ Low
- ↑ Medium
- ↑ High

References

Bates, P. D., N. Quinn, C. Sampson, A. Smith, O. Wing, J. Sosa, J. Savage, G. Olcese, J. Neal, G. Schumann, L. Giustarini, G. Coxon, J. R. Porter, M. F. Amodeo, Z. Chu, S. Lewis-Gruss, N. B. Freeman, T. Houser, M. Delgado, A. Hamidi, I. Bolliger, K. E. McCusker, K. Emanuel, C. M. Ferreira, A. Khalid, I. D. Haigh, A. Couasnon, R. E. Kopp, S. Hsiang, and W. F. Krajewski. 2021. Combined modeling of US fluvial, pluvial, and coastal flood hazard under current and future climates. Water Resources Research, 57(2), e2020WR028673

Beven II, J. L., A. Hagen, and R. Berg. 2022. Hurricane Ida (AL902021). National Hurricane Center Tropical Cyclone Report. National Oceanic and Atmospheric Administration and National Weather Service

Blois, J. L., P. L. Zarnetske, M. C. Fitzpatrick, and S. Finnegan. 2013. Climate change and the past, present, and future of biotic interactions. Science, 341(6145), 499-504

Brown, P. J., R. S. Bradley, and F. T. Keimig. 2010. Changes in extreme climate indices for the Northeastern United States, 18702005. Journal of Climate, 23, 6555–6572

Callahan, J. A., B. P. Horton, D. L. Nikitina, C. K. Sommerfield, T. E. McKenna, and D. Swallow. 2017. Recommendation of sea-level rise planning scenarios for Delaware: Technical Report, prepared for Delaware Department of Natural Resources and Environmental Control (DNREC) Delaware Coastal Programs. 116pp

Coen, M. C., E. Andrews, A. Bigi, G. Martucci, G. Romanens, F. P. A. Vogt, and L. Vuilleumier. 2020. Effects of the prewhitening method, the time granularity, and the time segmentation on the Mann–Kendall trend detection and the associated Sen's slope. Atmospheric Measurement Techniques, 13, 6945-6964

Coffey, R., M.J. Paul, J. Stamp, A. Hamilton, and T. Johnson. 2018. A review of water quality responses to air temperature and precipitation changes 2: Nutrients, algal blooms, sediment, and pathogens. Journal of American Water Resources Association, 5(4), 844-868.

DeGaetano, A. T., and R. J. Allen. 2002. Trends in twentieth-century temperature extremes across the United States. Journal of Climate, 15, 3188–3205

Delaware River Basin Commision. 2021. Flow Management. <https://www.nj.gov/drbc/programs/flow/flow-mgmt.html> (accessed 06-08-2022)

Devlin, A. T., J. Pan, and H. Lin. 2019. Extended spectral analysis of tidal variability in the North Atlantic Ocean. Journal of Geophysical Research: Oceans, 124, 506-526

DiLorenzo, J. L., P. Huang, M. L. Thatcher, and T. O. Najarian. 1993. Dredging impacts on Delaware Estuary tides. In Estuarine and Coastal Modeling III: Proceedings of the 3rd International Conference, pp. 86-104

Douville, H., K. Raghavan, J. Renwick, R.P. Allan, P.A. Arias, M. Barlow, R. Cerezo-Mota, A. Cherchi, T.Y. Gan, J. Gergis, D. Jiang, A. Khan, W. Pokam Mba, D. Rosenfeld, J. Tierney, and O. Zolina. 2021. Water Cycle Changes. In Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu, and B. Zhou (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1055–1210.

Feng, S. and Q. Hu. 2007. Changes in winter snowfall/precipitation ratio in the contiguous United States. Journal of Geophysical Research: Atmospheres 112

Ford, C. M., A. D. Kendall, and D.W. Hyndman. 2021. Snowpacks decrease and streamflows shift across the eastern US as winters warm. Science of The Total Environment 793, 148483

Fox-Kemper, B., H.T. Hewitt, C. Xiao, G. Aðalgeirsdóttir, S.S. Drijfhout, T.L. Edwards, N.R. Golledge, M. Hemer, R.E. Kopp, G. Krinner, A. Mix, D. Notz, S. Nowicki, I.S. Nurhati, L. Ruiz, J.-B. Sallée, A.B.A. Slangen, and Y. Yu. 2021. Ocean, Cryosphere and Sea Level Change. In Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu, and B. Zhou (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1211–1362

Frederikse, T., F. Landerer, L. Caron, S. Adhikari, D. Parkes, V. W. Humphrey, S. Dangendorf, P. Hogarth, L. Zanna, L. Cheng, and Y. Wu. 2020. The causes of sea-level rise since 1900. Nature, 584, 393-397

Friedlingstein, P., M. O'Sullivan, M.W. Jones, R.M. Andrew, L. Gregor, J. Hauck, C. Le Quéré, I.T. Luijkx, A. Olsen, G.P. Peters, W. Peters, et al. 2022. Global carbon budget 2022. Earth System Science Data, 14(11), 4811-4900.

Giri, S., R. G. Lathrop, and C. C. Obropta. 2020. Climate change vulnerability assessment and adaptation strategies through best management practices. Journal of Hydrology, 580, 124311

Gori, A., N. Lin, D. Xi, and K. Emanuel. 2022. Tropical cyclone climatology change greatly exacerbates US extreme rainfall–surge hazard. Nature Climate Change, 12(2), 171-178

Hammond, J. C. and B. J. Fleming. 2021. Evaluating low flow patterns, drivers and trends in the Delaware River Basin. Journal of Hydrology, 598, 126246

Hawkins, T. W. 2015. Simulating the impacts of projected climate change on streamflow hydrology for the Chesapeake Bay watershed. Annals of the Association of American Geographers, 105, 627–648

Hawkins, T. W. and C. J. Woltemade. 2021. Impact of projected 21st century climate change on basin hydrology and runoff in the Delaware River Basin, USA. Journal of Water and Climate Change, 12.1, 60-81

Hayhoe, K., D. J. Wuebbles, D. R. Easterling, D. W. Fahey, S. Doherty, J. Kossin, W. Sweet, R. Vose, and M. Wehner, 2018. Our Changing Climate. In Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II. [Reidmiller, D. R., C. W. Avery, D. R. Easterling, K. E. Kunkel, K. L. M. Lewis, T. K. Maycock, and B. C. Stewart (eds.)]. U.S. Global Change Research Program, Washington, DC, USA, pp. 72–144

Hinson, K. E., M. A. Friedrichs, P. St-Laurent, F. Da, and R. G. Najjar. 2021. Extent and causes of Chesapeake Bay warming. JAWRA Journal of the American Water Resources Association, doi: 10.1111/1752-1688.12916

Howarth, M. E., C. D. Thorncroft, and L. F. Bosart. 2019. Changes in extreme precipitation in the northeast United States: 1979–2014. Journal of Hydrometeorology, 20(4): 673-689

Horton, R., W. Easterling, R. Kates, M. Ruth, E. Sussman, A. Whelchel, D. Wolfe, and F. Lipschultz. 2014. Ch. 16: Northeast. Climate change impacts in the United States: The Third National Climate Assessment, J. M. Melillo, Terese (T. C.) Richmond, and G. W. Yohe, Eds., U.S. Global Change Research Program, 317–395

Hsu, H.-H. and Y.-T. Chen. 2020. Simulation and projection of circulations associated with atmospheric rivers along the North American northeast coast. Journal of Climate, 33(13): 5673-5695

Hull, C. and J. G. Titus. 1986. Greenhouse effect, sea level rise, and salinity in the Delaware Estuary. United States Environmental Protection Agency. 36pp

Hussain, M. M. and I. Mahmud. 2019. pyMannKendall: a python package for non parametric Mann Kendall family of trend tests. The Journal of Open Source Software, 4, 1556

IPCC. 2021. Summary for Policymakers. In: Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu, and B. Zhou (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 3−32.

Janssen, E., D. J. Wuebbles, K. E. Kunkel, S. C. Olsen, and A. Goodman. 2014. Observational- and model-based trends and projections of extreme precipitation over the contiguous United States. Earth's Future, 2, 99–113

Joos, F. and R. Spahni. 2008. Rates of change in natural and anthropogenic radiative forcing over the past 20,000 years. Proceedings of the National Academy of Sciences, 105(5), 1425-1430

Karmalkar, A. V., and R. M. Horton. 2021. Drivers of exceptional coastal warming in the northeastern United States. Nature Climate Change, 11(10), 854-860

Kaushal, S. S., G. E. Likens, N. A. Jaworski, M. L. Pace, A. M. Sides, D. Seekell, K. T. Belt, D. H. Secor, and R. Wingate. 2010. Rising stream and river temperatures in the United States. Frontiers in Ecology and the Environment, 8(9), 461-466

Kendall, M. G. 1955. Rank Correlation Methods. 2nd ed. Hafner, New York

Kopp, R. E. 2013. Does the mid-Atlantic United States sea level acceleration hotspot reflect ocean dynamic variability? Geophysical Research Letters, 40(15), 3981-3985

Kossin, J. P., K. R. Knapp, T. L. Olander, and C. S. Velden. 2020. Global increase in major tropical cyclone exceedance probability over the past four decades. Proceedings of the National Academy of Sciences, 117(22), 11975-11980

Knutson, T. R. and F. Zeng. 2018. Model Assessment of observed precipitation trends over land regions: Detectable human influences and possible low bias in model trends. Journal of Climate, 31(12), 4617-4637

Kreeger, D., J. Adkins, P. Cole, R. Najjar, D. Velinsky, P. Conolly, and J. Kraeuter. 2010. Climate change and the Delaware Estuary: Three case studies in vulnerability assessment and adaptation planning. Partnership for the Delaware Estuary, PDE Report No. 10-01, 1-117 pp

Lee, S. B., M. Li, and F. Zhang. 2017. Impact of sea level rise on tidal range in Chesapeake and Delaware Bays. Journal of Geophysical Research: Oceans, 122, 3917-3938

Li, S., T. Wahl, S. A. Talke, D. A. Jay, P. M. Orton, X. Liang, G. Wang, and L. Liu. 2021. Evolving tides aggravate nuisance flooding along the U.S. coastline. Science Advances, 7, eabe2412

Lynch, C., A. Seth, and J. Thibeault. 2016. Recent and projected annual cycles of temperature and precipitation in the Northeast United States from CMIP5. Journal of Climate, 29, 347–365

Mann, H. B. 1945. Nonparametric tests against trend. Econometrica, 13, 245–259

Marsooli, R., N. Lin, K. Emanuel, and K. Feng. 2019. Climate change exacerbates hurricane flood hazards along US Atlantic and Gulf Coasts in spatially varying patterns. Nature Communications, 10:3785

McCabe, G. J. and D. M. Wolock. 2020. Hydro-climatic drought in the Delaware River Basin. Journal of the American Water Resources Association, 56(6), 981-994

Menne, M. J., C. N. Williams, and R. Vose. 2015a. U.S. Historical Climatology Network (USHCN). <https://www.ncei.noaa.gov/products/land-based-station/us-historical-climatology-network>

——, ——, and ——. 2015b. United States Historical Climatology Network Daily Temperature, Precipitation, and Snow Data. <http://cdiac.ornl.gov/epubs/ndp/ushcn/daily_doc.html>

Menne, M. J., C. N. Williams Jr., and R. S. Vose. 2009. The U.S. Historical Climatology Network Monthly Temperature Data, Version 2. Bulletin of the American Meteorological Society, 90, 993–1007

Myhre, G., D. Shindell, F.-M. Bréon, W. Collins, J. Fuglestvedt, J. Huang, D. Koch, J.-F. Lamarque, D. Lee, B. Mendoza, T. Nakajima, A. Robock, G. Stephens, T. Takemura and H. Zhang. 2013. Anthropogenic and Natural Radiative Forcing. In: Climate Change 2013: The Physical Science Basis. Contribution of Working group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T. F., D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P. M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA

Najjar, R., L. Patterson, and S. Graham. 2009. Climate simulations of major estuarine watersheds in the Mid-Atlantic region of the US. Climatic Change, 95, 139–168

National Ocean Service. Center for Operational Oceanographic Products and Services. NOAA Tides and Currents, Coastal Inundation Dashboard. <https://tidesandcurrents.noaa.gov/inundationdb/> (accessed 07/12/2022)

Ning, L. and R. S. Bradley. 2015. Snow occurrence changes over the central and eastern United States under future warming scenarios. Scientific Reports, 5(1), 1-8

NOAA Global Monitoring Laboratory. Trends in Atmospheric Carbon Dioxide, Global Monthly Mean CO2 (Full Record). <gml.noaa.gov/ccgg/trends/global.html> (accessed 03/08/2022).

Paerl, H.W. and J. Huisman. 2008. Blooms like it hot. Science, 320(5872), 57-58.

Pareja-Roman, L. F., R. J. Chant, and C. K. Sommerfield. 2020. Impact of historical channel deepening on tidal hydraulics in the Delaware Estuary. Journal of Geophysical Research: Oceans, 125, e2020JC016256

Pringle, W. J., J. Wang, K. J. Roberts, and V. R. Kotamarthi. 2021. Projected changes to cool-season storm tides in the 21st century along the Northeastern United States coast. Earth's Future, 9, e2020EF001940

Rawlins, M. A., R. S. Bradley, and H. F. Diaz. 2012. Assessment of regional climate model simulation estimates over the northeast United States. Journal of Geophysical Research Atmospheres, 117, D23112, doi:10.1029/2012JD018137

Ross, A. C., R. G. Najjar, M. Li, M. E. Mann, S. E. Ford, and B. Katz. 2015. Sea-level rise and other influences on decadal-scale variations of salinity in a coastal plain estuary. Estuarine Coastal and Shelf Science, 157, 79-92

Ross, A. C., R. G. Najjar, M. Li, S. B. Lee, F. Zhang, and W. Liu. 2017. Fingerprints of sea-level rise on changing tides in the Chesapeake and Delaware Bays. Journal of Geophysical Research: Oceans, 122, 8102–8125

Schulte, J., R. G. Najjar, and M. Li. 2016. The influence of climate modes on streamflow in the Mid-Atlantic Region of the United States. Journal of Hydrology: Regional Studies, 5, 80-99

Seager, R., N. Pederson, Y. Kushnir, J. Nakamura, and S. Jurburg. 2012. The 1960s drought and the subsequent shift to a wetter climate in the Catskill Mountains Region of the New York City Watershed. Journal of Climate, 25(19), 6721-6742

Sen, P. K. 1968. Estimates of the regression coefficient based on Kendall's tau. Journal of the American Statistical Association, 63, 1379–1389

Sharma, S., M. Gomez, K. Keller, R. E. Nicholas, and A. Mejia. 2021. Regional flood risk projections under climate change. Journal of Hydrometeorology, 22(9), 2259-2274

Shi, S., and G. Liu. 2021. The latitudinal dependence in the trend of snow event to precipitation event ratio. Scientific Reports 11, 18112

Shirazi, Y. A., E. W. Carr, G. R. Parsons, P. Hoagland, D. K. Ralston, and J. Chen. 2019. Increased operational costs of electricity generation in the Delaware River and Estuary from salinity increases due to sea-level rise and a deepened channel. Journal of Environmental Management, 244, 228-234

Simonson, J. M., S. D. Birkel, K. A. Maasch, P. A. Mayewski, B. Lyon, and A. M. Carleton. 2022. Association between recent U.S. northeast precipitation trends and Greenland blocking. International Journal of Climatology

Slack, J. R., A. M. Lumb, and J. M. Landwehr. 1993. Hydro-Climatic Data Network (HCDN) Streamflow Data Set, 1874-1988. U.S. Geological Survey, Reston, VA

Sweet, W. V., B. D. Hamlington, R. E. Kopp, C. P. Weaver, P. L. Barnard, D. Bekaert, W. Brooks, M. Craghan, G. Dusek, T. Frederikse, G. Garner, A. S. Genz, J. P. Krasting, E. Larour, D. Marcy, J. J. Marra, J. Obeysekera, M. Osler, M. Pendleton, D. Roman, L. Schmied, W. Veatch, K.D. White, and C. Zuzak. 2022. Global and Regional Sea Level Rise Scenarios for the United States: Updated Mean Projections and Extreme Water Level Probabilities Along U.S. Coastlines. NOAA Technical Report NOS 01. National Oceanic and Atmospheric Administration, National Ocean Service, Silver Spring, MD

Theil, H. 1950. A rank-invariant method of linear and polynomial regression analysis, i, ii, and iii. Proceedings of the Koninklijke Nederlandse Akademie Wetenschappen, Series A Mathematical Sciences, 53, 386-392, 521-525, 1397-1412

Thibeault, J. M. and A. Seth. 2014. Changing climate extremes in the Northeast United States: observations and projections from CMIP5. Climatic Change, 127, 273-287

Thompson, P. R, M. J. Widlansky, B. D. Hamlington, M. A. Merrifield, J. J. Marra, G. T. Mitchum, and W. Sweet. 2021. Rapid increases and extreme months in projections of United States high-tide flooding. Nature Climate Change, 11, 584-590

United States Geological Survey. 2017. Flexible Flow Management Program. Office of the Delaware River Master. <https://webapps.usgs.gov/odrm/ffmp/flexible-flow-management-program> (accessed 06-08-2022)

Vautard, R., J. Cattiaux, P. Yiou, J.-N. Thepaut, and P. Ciais. 2010. Northern Hemisphere atmospheric stilling partly attributed to an increase in surface roughness. Nature Geoscience, 3, 756–761

Vose, R. S., D. R. Easterling, K. E. Kunkle, A. N. LeGrande, and M. F. Wehner. 2017. Temperature changes in the United States. Climate Science Special Report: Fourth National Climate Assessment, Volume I. [Wuebbles, D. J., D. W. Fahey, K. A. Hibbard, D. J. Dokken, B. C. Stewart, and T. K. Maycock (eds.)], U.S. Global Change Research Program, Washington, DC, USA, 185-206

Wang, S., and R. Toumi. 2022. More tropical cyclones are striking coasts with major intensities at landfall. Scientific Reports, 12(1), 5236

White, K. D. 1996. A new ice jam database. Water Resources Bulletin, 32, 341–348

Williamson, T. N., E. A. Nystrom, and P. C. D. Milly. 2016. Sensitivity of the projected hydroclimatic environment of the Delaware River basin to formulation of potential evapotranspiration. Climatic Change, 139(2), 1–14

Wing, O. E. J., W. Lehman, P. D. Bates, C. C. Sampson, N. Quinn, A. M. Smith, J. C. Neal, J. R. Porter, and C. Kousky. 2022. Inequitable patterns of US flood risk in the Anthropocene. Nature Climate Change, 12(2), 156-162

Wuebbles, D., G. Meehl, K. Hayhoe, T. R. Karl, K. Kunkel, B. Santer, M. Wehner, B. Colle, E. M. Fischer, R. Fu, A. Goodman, E. Janssen, V. Kharin, H. Lee, W. Li, L.N. Long, S. C. Olsen, Z. Pan, A. Seth, J. Sheffield, and L. Sun. 2014. CMIP5 climate model analyses: Climate extremes in the United States. Bulletin of the American Meteorological Society, 95, 571–583

Yue, S. and C. Y Wang. 2002. Applicability of prewhitening to eliminate the influence of serial correlation on the Mann-Kendall test. Water Resources Research, 38, doi: 10.1029/2001WR000861

Climate Feature 1

Redlining and Urban Heat Islands

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The Pennsylvania State University

Racist housing policies, such as redlining, have left poor neighborhoods in many cities more susceptible than wealthier neighborhoods to the urban heat island effect and air pollution (Hoffmann et al., 2020; Locke et al., 2021; Lane et al., 2022). The term "redlining" derives from discriminatory practices of the federal government's Home Owners' Loan Corporation (HOLC), which developed maps in the 1930s for use by mortgage lenders to assign grades to different neighborhoods; those deemed unfavorable were outlined in red. In 1968 the Fair Housing Act brought an end to redlining, in addition to other discriminatory home-lending policies, as part of the Civil Rights Act (United States Federal Reserve, 2017).

Hoffman et al. (2020) studied the effect of redlining on temperature and land use in 108 US cities, finding that neighborhoods with low HOLC grades are much hotter that those with high grades, a result that reflects the greater density of parks, trees, and other green infrastructure in wealthier neighborhoods, which experienced a greater infusion of funds for development. Specifically, redlined urban neighborhoods in the United States (grade D: hazardous) have land surface temperatures that are, on average, 2.6 °C higher than non-redlined areas (grade A: desirable). In Philadelphia, Camden, and Trenton, the same study found these differences to be much larger: 5.2, 4.3, and 4.6 °C, respectively (Fig 2.14).

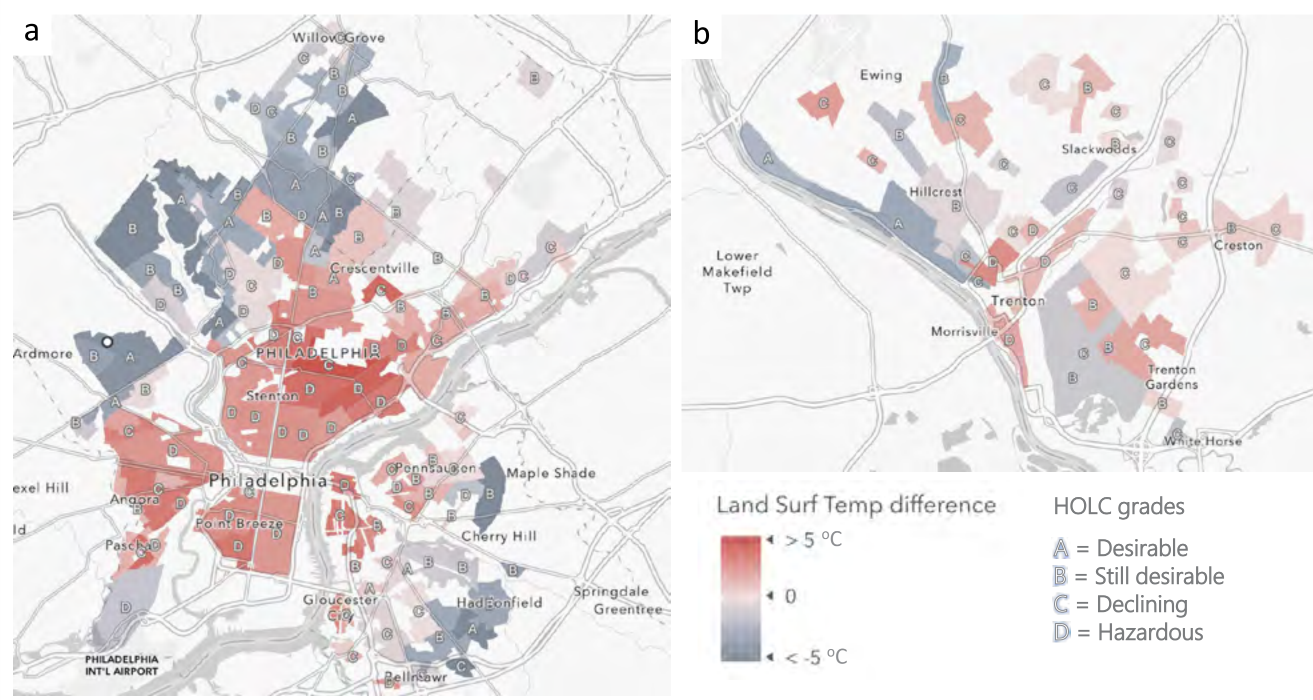

Figure 2.14 Maps of a) Philadelphia, Camden, and b) Trenton indicating the deviation of land surface temperature from the city-wide average (colors) and grades by the Home Owners' Loan Corporation (HOLC; letters). Source: Science Museum of Virginia and Esri. Data from Hoffman et al. (2020). https://www.arcgis.com/apps/dashboards/73e329457b6644e7aeff13ecce43c8d8

References

Hoffman, J. S., Shandas, V., & Pendleton, N. (2020). The effects of historical housing policies on resident exposure to intra-urban heat: A study of 108 US urban areas. Climate, 8(1), 12.

Lane, H. M., Morello-Frosch, R., Marshall, J. D., & Apte, J. S. (2022). Historical Redlining Is Associated with Present-Day Air Pollution Disparities in US Cities. Environmental Science & Technology Letters. https://doi.org/10.1021/acs.estlett.1c01012

Locke, D. H., Hall, B., Grove, J. M., Pickett, S. T. A., Ogden, L. A., Aoki, C., Boone, C. G., & O'Neil-Dunne, J. P. M. (2021). Residential housing segregation and urban tree canopy in 37 US Cities. npj Urban Sustainability, 1(1), 15. https://doi.org/10.1038/s42949-021-00022-0

United States Federal Reserve (2017). Consumer Compliance Handbook. V. Federal Fair Lending Regulations and Statutes. Board of Governors of the Federal Reserve System. https://www.federalreserve.gov/boarddocs/supmanual/cch/fair_lend_fhact.pdf

Climate Feature 2

Flooding in a Marginalized Community

Sofi Courtney[1] and Jill Arriola[2]

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An example of the interaction between systemic racism and climate change is the Eastwick neighborhood in Southwest Philadelphia, Pennsylvania (Fig 2.15). Eastwick resides at low elevations, mostly below 10 feet (~3 meters) above sea level, borders the John Heinz Wildlife Refuge and the Philadelphia International Airport, and is nestled between the Delaware River, Schuylkill River, Cobbs Creek, and Darby Creek. A majority of what is now Eastwick was once a freshwater tidal marsh that was drained and diked for farming and horticulture in the 18th and 19th centuries, then later filled with dredged material to create land for housing and the airport in the 20th century (Krulikowski 2014). In this era, the community was subjected to redlining (also see Climate Feature 1), a racist housing policy that devalued property in neighborhoods of color, by Federal mortgage agencies who cited both African-American residents and flood risk as reasons to disinvest in the neighborhood (Nelson et al.). Today, the majority of the Eastwick public land parcel is within a Special Flood Hazard Area that consists of a 1% annual chance flood zone, otherwise known as the 100-year floodplain, and a 0.2% annual chance flood zone, or the 500-year floodplain (Fig 2.15)(FEMA 2021).

Although Eastwick has predominantly been a low-density neighborhood due to the difficulty of building in the neighborhood's wetland conditions, early 20th century Eastwick still managed to be one of the few racially integrated neighborhoods in Philadelphia (Cahn 2014). Before the urban renewal era in the 1950's, Eastwick was valued by the local residents for its pastoral qualities and was affectionately termed "The Meadows." However, the neighborhood struggled with environmental justice issues, including blight, a fire in a local landfill, and nuisance and major flooding events (McKee 2001). In 1958 the city of Philadelphia implemented large-scale redevelopment urban renewal projects in Eastwick centered around seizing private property using eminent domain. The city then proceeded to contract out creation of new housing tracts and relocate African American residents from other developing areas of Philadelphia into Eastwick (Cahn 2014). Most of this development plan failed, resulting in demolition of 2,000 acres of neighborhoods and the displacement of 8,000 residents (McKee 2001). Much of the housing that was constructed during redevelopment did not account for the low-lying nature of the landscape and resulted in buildings that are regularly flooded or are sinking into the marsh. The attempted redevelopment of Eastwick also destroyed acres of tidal wetlands, which provide a natural ecosystem service of mitigating flooding and storm surges (Cahn 2014).

In addition to discriminatory policy and planning, climate change is seriously threatening the continued existence of the neighborhood as increasing heavy rain and rising sea levels (see sections 2.3 and 2.5) have been dramatically intensifying flooding in residential neighborhoods (Phillips 2021). Today, the interaction of climate adaptation and environmental justice can still be observed in Eastwick. For example, there are ongoing discussions on rehousing current residents to new city-owned housing as Philadelphia's first climate migrants (Phillips 2021). This move is controversial, not only because of the history of development in the neighborhood (McKee 2001), but because the new housing still resides in a 500-year flood plain (Phillips 2021) and there are conflicting stakeholder interests between future economic development and flood mitigation of the vacated land. Despite activism by a neighborhood coalition and extensive community participation in city planning processes, the priorities of Eastwick residents have not been addressed in city land management decisions. Eastwick community groups continue to express concern that any development outside of conservation as open land will result in increased flooding in the surrounding neighborhood, especially under the threat of climate change.

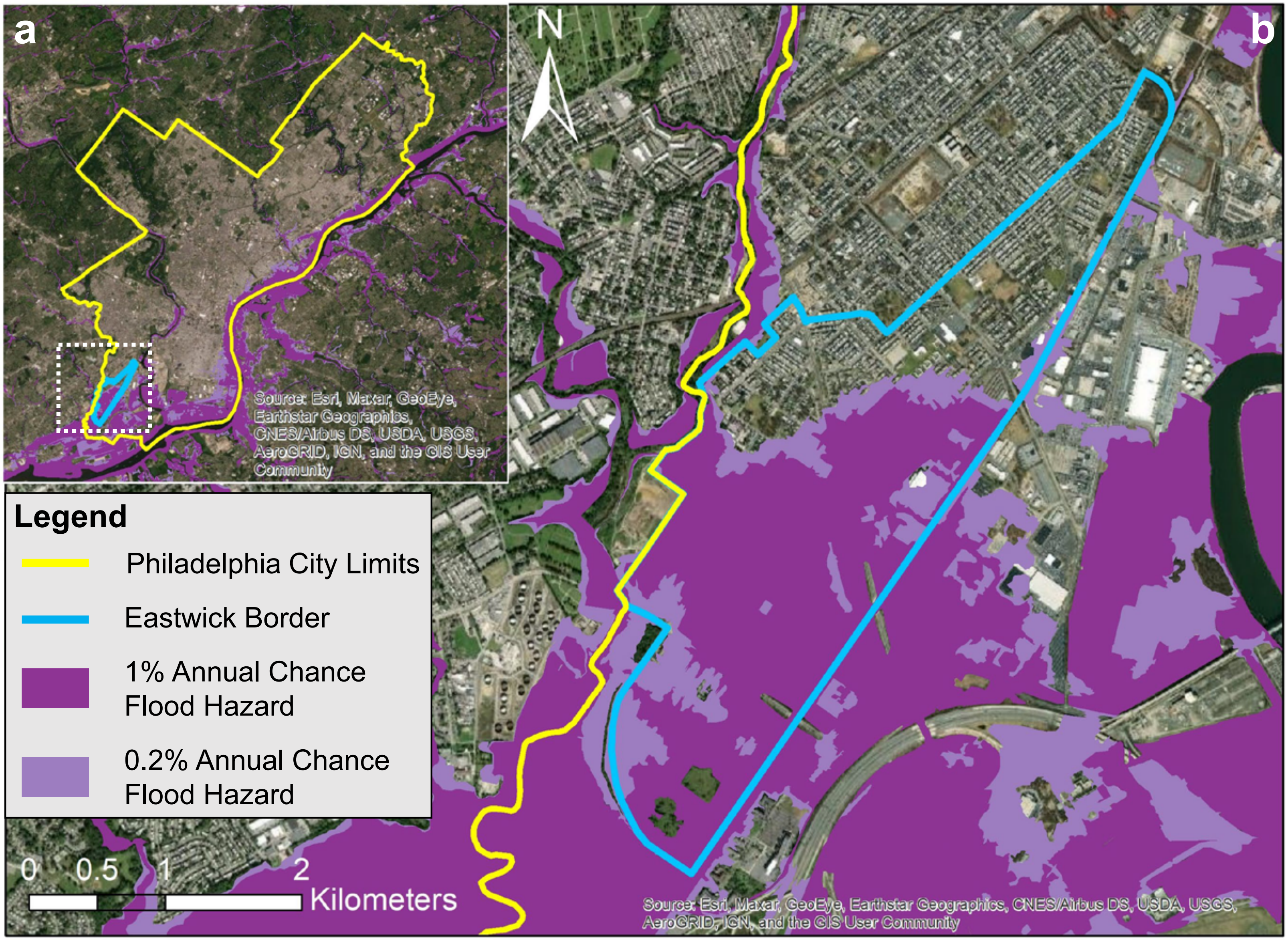


Figure 2.15 a) The Eastwick neighborhood (blue outline) resides in the southwest corner of Philadelphia (yellow outline). The 1 and 0.2 % annual chance flood hazard zones (dark and light purple, respectively) fall mainly along the Delaware River on the south and east sides of the city. b) A closer view of Eastwick and its vulnerability to major flooding events. (Sources: ESRI, FEMA, and pasda.psu.edu)

References

Cahn, A.L., 2014: On retiring blight as policy and making Eastwick whole. Harvard Civil Liberties Law Review, 49, 449-492.

Federal Emergency Management Agency (FEMA), 2021: Pennsylvania Releases New Mapping Tool to Show Flood Risk. https://www.fema.gov/press-release/20210607/pennsylvania-releases-new-mapping-tool-show-flood-risk (Accessed 03-22-2022).

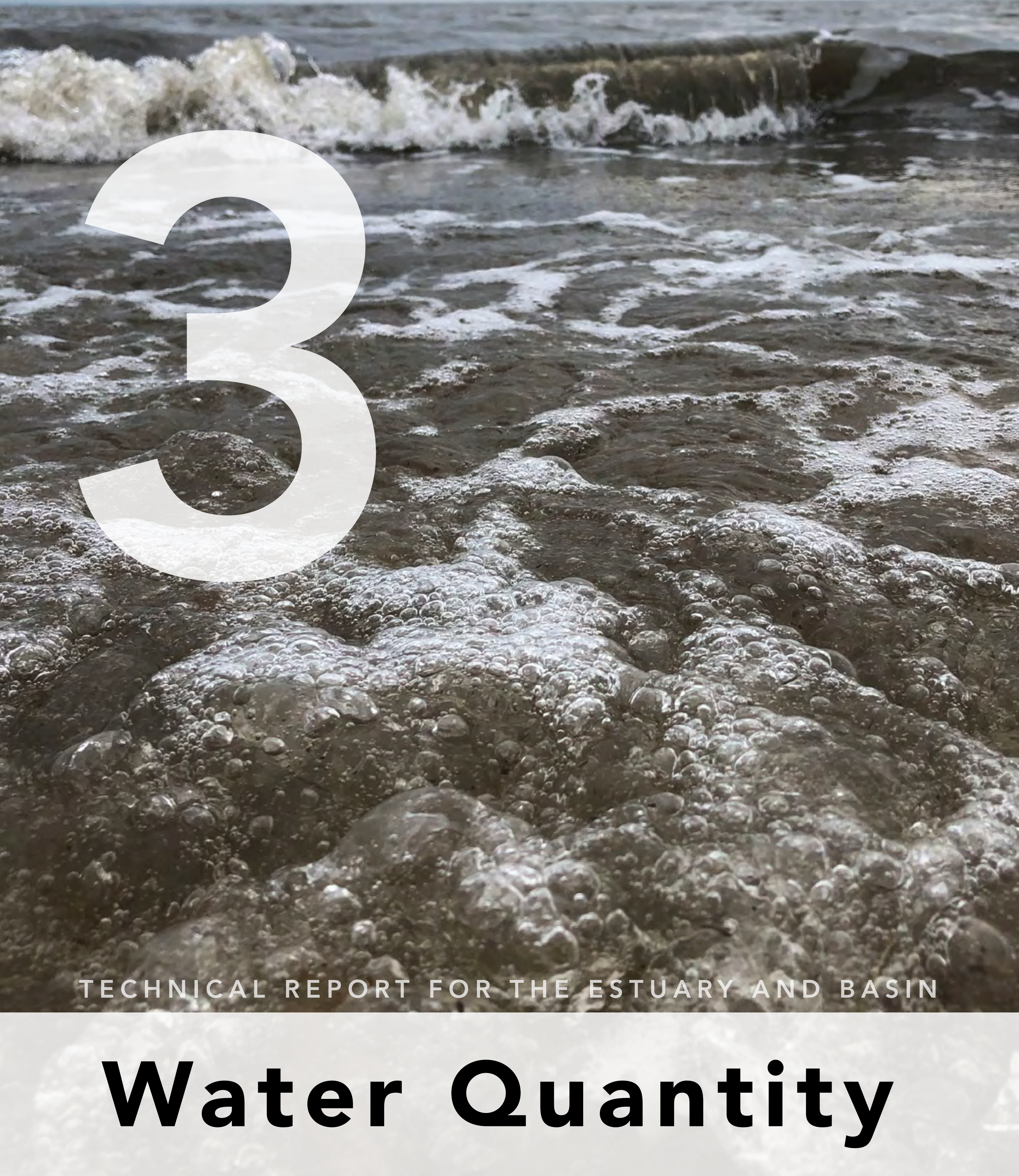
Krulikowski, A.E. 2014. Southwest Philadelphia. The Encyclopedia of Greater Philadelphia. https://philadelphiaencyclopedia.org/locations/southwest-philadelphia/ (Accessed 03-22-22).

Martel, J.-L., F. Brissette, M. Troin, M., R. Arsenault, J. Chen, T. Su and P. Lucas-Picher. 2022. CMIP5 and CMIP6 model projection comparison for hydrological impacts over North America. Geophysical Research Letters, 49(15): e2022GL098364

McKee, G. A., 2001: Liberal ends through illiberal means: Race, urban renewal, and community in the Eastwick section of Philadelphia, 1949-1990. Journal of Urban History, 27(5), 547-583.

Nelson, R.K., Winling, L., Marciano, R., Connolly, N. et al., "Mapping Inequality," American Panorama, ed. Robert K. Nelson and Edward L. Ayers. Accessed March 28, 2022, https://dsl.richmond.edu/panorama/redlining/#loc=12/39.932/-75.267&city=philadelphia-pa&area=D23&adview=full&adimage=2/67.676/-165.938.

Phillips, S., 2021: Eastwick residents could become Philly's first climate migrants. A new plan sees them moving to city-owned higher ground. https://whyy.org/articles/eastwick-residents-could-become-phillys-first-climate-migrants-a-new-plan-sees-them-moving-to-city-owned-higher-ground/ (Accessed 03-22-22).



3

TECHNICAL REPORT FOR THE ESTUARY AND BASIN

Water Quantity

Partnership for the DELAWARE ESTUARY

December 2022 | Report No.22-05

Partnership for the Delaware Estuary—Host of the Delaware Estuary Program

3 Water Quantity

Michael Thompson, Sara Sayed, Chad Pindar & Amy Shallcross

Delaware River Basin Commission

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Contributors Emily Pirl, LeeAnn Haaf, Steven Domber
Cover photograph by LeeAnn Haaf

Table of Contents

3. Water Quantity

Abstract

Although the Delaware Estuary and Basin drain only four-tenths of one percent of the total continental United States land area, its water resources provide drinking water for over 13.3 million people in four states—approximately 4 percent of the total population of the United States. Based on 2020 data, ground and surface water withdrawals from the Delaware Estuary and Basin are estimated to total 6,390 million gallons per day (MGD), out-of-Basin diversions total 594 MGD, and in-basin consumptive use is 263 MGD. With such a demand for water, it is essential that humans continue to monitor and plan for these uses–not only to ensure that the demands can be met, but to help maintain a sustainable relationship with the environmental needs of the Basin and the creatures which inhabit it. To this point, several indicators assessing water quantity and hydrology are outlined in this chapter. Aptly captured by a Native American proverb: "The frog does not drink up the pond in which he lives."

Data Sources

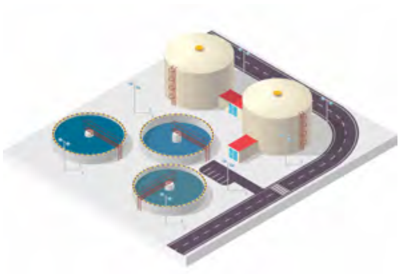
Several of the indicators described in this chapter are based on water withdrawal datasets. These data are typically reported annually by water users (i.e., industrial and public water systems) to the state environmental agencies. As each state has differing requirements, nomenclature, and structure in how data are retained, Delaware River Basin Commission (DRBC) manages its own water use database. This process allows DRBC to compile and assess data on water withdrawals and consumptive use throughout the Delaware Estuary and Basin (also referred to as the Delaware River Basin herein) with some degree of consistency across state boundaries. In some cases, DRBC even supplements data from state agencies with that collected through DRBC programs. This integrated database for the Delaware River Estuary and Basin is typically updated annually and requires a substantial degree of quality assurance/quality control to verify that data are not duplicated or missing.

3.1 Water Withdrawals: Tracking Water Supply & Demand

Description of Indicator

Accurate and comprehensive water use information enables the proper assessment, planning, and management of water resources. As reporting improves, so does our accounting and understanding of the need for water among various water-using sectors. Almost all data are based on withdrawals reported to state agencies. The two exceptions are the self-supplied domestic withdrawal sector (i.e., a private residence utilizing a well that serves a single home, where data is estimated based on population analyses), and the hydroelectric power category within the power generation sector (which has some data estimated based on net electricity generation).

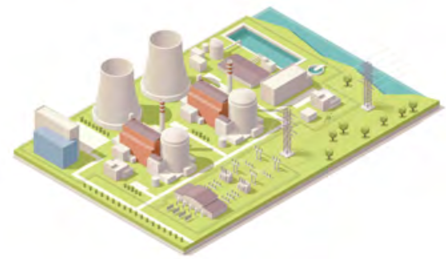
The indicator presented in this section of the report is focused on water withdrawals and discusses the data in terms of withdrawal sectors (which may contain multiple withdrawal categories) (Fig 3.1). As is defined in this section, "water use" may refer to either the withdrawal or end-use of water. This is important when considering public water supply, and how the data are categorized. For example, consider a hypothetical public water supplier operating a withdrawal(s), and then distributing the water to numerous customers (e.g., 60% domestic, 30% industrial, 10% commercial). In this scenario, the withdrawal would be described



(PWS) Public Water Supply
Water withdrawn by a facility meeting the definition of a public water supply system under the Safe Drinking Water Act (Pub. L. No. 93-523, 88 Stat. 1660), or subsequent regulations set forth by signatory parties.

(DIV) Out-of-Basin Diversions
Withdrawals of water for public water supply exported from the Delaware River Basin by the Decree Parties in accordance with a 1954 U.S. Supreme Court Decree (U.S. Supreme Court, 1954).

(SSD) Self-Supplied Domestic
Water withdrawal for domestic use for residents who are not served by a public water supply system; it is assumed in this study that all self-supplied groundwater withdrawals are groundwater.



(PWR) Power Generation
Water withdrawn/diverted by facilities associated with the process of generating electricity. Within the Delaware River Basin, this refers to water withdrawn/diverted by both thermoelectric and hydroelectric facilities.

(IND) Industrial
Water withdrawals by facilities associated with fabrication, processing, washing, and cooling. This includes industries such as chemical production, food, paper and allied products, petroleum refining (i.e., refineries), and steel. Due to the generally close relationship, water withdrawn for groundwater remediation purposes is also included in this sector.

(IRR) Irrigation
Water withdrawals which are applied by an irrigation system to assist crop and pasture growth, or to maintain vegetation on recreational lands such as parks and golf courses. This does not include withdrawals/diversions associated with aquaculture.

(MIN) Mining
Water withdrawals by facilities involved with the extraction of naturally occurring minerals. This includes operations such as mine dewatering, quarrying, milling of mined materials, material washing and processing, material slurry operations (e.g. sand), dust suppression and any other use at such facilities.

(OTH) Other
Facilities not categorized by previous sectors, including but not limited to aquaculture, bottled water, commercial (e.g. hotels, restaurants, office buildings, retail stores), fire suppression, hospital/health, military, parks/recreation, prisons, schools, and ski/snowmaking.

Figure 3.1 Summary of common withdrawal sectors. All images on this page are copyrighted and used by Dreamstime.com, used in accordance with licenses.

by one withdrawal category (public water supply), whereas the end-use would be described by multiple water use categories (domestic, industrial, and commercial).

Present Status

Understanding water withdrawals, consumption, and supply is integral to the management of water resources, which helps provide for a sustainable balance between conservation and utilization. In recent years, our understanding of how water is withdrawn from the system and subsequently used has greatly improved. This increase in knowledge also includes a better understanding of the underlying data management systems, meaning more timely and comprehensive assessments can be made. Figure 3.2 shows the Basin-wide picture of water withdrawals, out-of-Basin diversions, and consumptive use, by sector, based on the 2020 calendar year water use data; the data shown represents daily average withdrawals on an annual basis (DRBC 2022a).

Key Delaware River Basin Water Withdrawal Facts:

- Based on 2016 data, an estimated 13.3 million people rely on water from the Delaware River Basin for their daily water needs (Byun et al. 2019). Approximately 8.3 million people live in the Delaware River Basin, and the volume of diversions to New York City and northeastern New Jersey is sufficient to supply water to an additional 5 million people.
- Based on 2020 data, ground and surface water withdrawals from the Delaware River Basin are estimated to total 6,390 million gallons per day (MGD), out-of-Basin diversions total 594 MGD, and in-basin consumptive use is 263 MGD.
- Approximately 95% of all water used in the Delaware River Basin is obtained from surface waters.
- Three dominant use sectors account for over 68% of total water withdrawals; these sectors are thermoelectric power generation (49%), public water supply (12%), and industrial (7%).

Past Trends

DRBC recently published a comprehensive report assessing historical water withdrawals and consumptive use in the Delaware River Basin from 1990-2017, with projections provided through the year 2060 (Thompson & Pindar 2021). For the most part, the study evaluated individual withdrawal systems (i.e., a facility which operates withdrawal sources together) which are associated with DRBC regulatory approvals (excluding the self-supplied domestic and irrigation sectors). The more than 600 systems which were individually reviewed included about 3,700 sources and account for the overwhelming majority of reported water withdrawals by volume (greater than 99%). The remaining withdrawal systems not associated with DRBC regulatory approvals (likely operating below review thresholds) included about 2,450 sources and subsequently account for under 1% of the total average withdrawal. Based on the volumetric proportion of data, subject to the project's review process, it is assumed that the historical time series represents actual (or observed) conditions. The historical water withdrawals from the Delaware River Basin are presented in Figure 3.3, color-coded by withdrawal sector and updated to include data from 2018-2020, which was beyond the scope of Thompson & Pindar (2021).

Future Predictions

In addition to compiling a historical time series of withdrawal data for the Delaware River Basin, Thompson & Pindar (2021) also project future withdrawals through the year 2060. The detailed methodology behind the projections is contained within that report; however, it can be summarized by a few key notes:

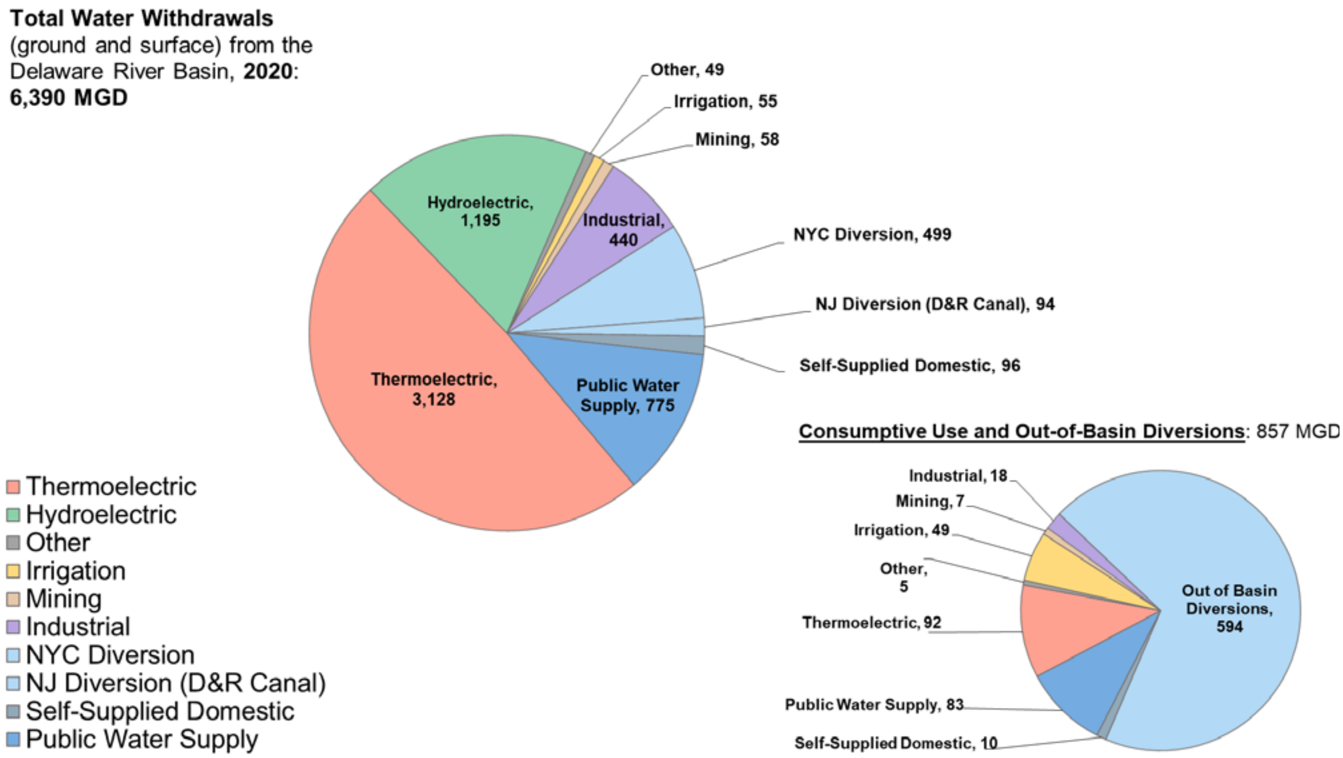


Figure 3.2 Total water withdrawals from the Delaware River Basin, 2020.

- Primary method: The majority of projected sectors (excluding out-of-Basin diversions, self-supplied domestic, and irrigation) relied on the extrapolation of historical withdrawal data. Projections were performed on a system-by-system basis, often at finer scales, allowing results to meet planning objectives. Pertinent metadata was often a consideration in determining the most appropriate regression(s) for each system.
- Out-of-Basin diversions: Due to the complex nature of the operation of these sources, trend extrapolation was not considered an appropriate means for providing a projection of withdrawal. Instead, the last five years of data in this study (2013-2017) were averaged and used as the projected value for each source.
- Self-supplied domestic: Withdrawal estimates for self-supplied domestic water users were based on population distribution in relation to public water supply service areas (considering per-capita water rates). Projections were performed utilizing county-level population projections for 2020-2100 for one of five Shared Socioeconomic Pathways (SSPs), which represent different ways in which the United States may be expected to grow in this century (M. E. Hauer 2019; M. Hauer & CIESIN 2021; O'Neill et al. 2014).
- Irrigation: Due in part to the large number of sources and approvals, irrigation withdrawals were aggregated to and projected at the smallest applicable planning scales, differentiating groundwater from surface water. Projections were performed considering climatic variables from a downscaled regional climate model.

The projections from each of these methods were performed in a way, such that the results may be aggregated to various planning scales (e.g. hydrologic unit codes, state boundaries, Basin-wide, and various combinations of sectors). The Basin-wide projection is presented in Figure 3.3 as the solid blue line, with 80% and 95% predictive intervals in varying shades of gray. Some high-level conclusions from Thompson & Pindar (2021) regarding the projection results are:

- Peak water withdrawal from the Delaware River Basin has likely already occurred (in 2005 and 2006 it was estimated to be approximately 9.917 billion gallons per day).
- The Basin-wide water withdrawals from the Delaware River Basin are projected to continue decreasing, from a projected value of 6,921 MGD in 2020 to about 6,289 MGD in 2060, however, the estimated value based on reported data for 2020 is 6,390 MGD. The largest projected decreases were for thermoelectric power (-322 MGD) and hydroelectric power (-292 MGD), followed by public water supply (-30 MGD) and self-supplied domestic (-5 MGD). The remaining sectors returned mild increases.
- Historical decreases in water withdrawals by thermoelectric facilities are shown to be strongly correlated with decreases in energy generation from coal-fired steam-turbine facilities using once-through cooling. These findings are consistent with other studies at the national level which highlight the closure of many such facilities.
- The population residing within the Delaware River Basin has historically increased and is projected to continue increasing. Despite a growing Basin-wide population, public water supply withdrawals have historically decreased and are projected to continue decreasing. The pattern of increasing population and decreasing withdrawals (Fig 3.4) is assumed to be related to advances in leak detection and water conservation by purveyors, regulatory efforts such as plumbing standards,

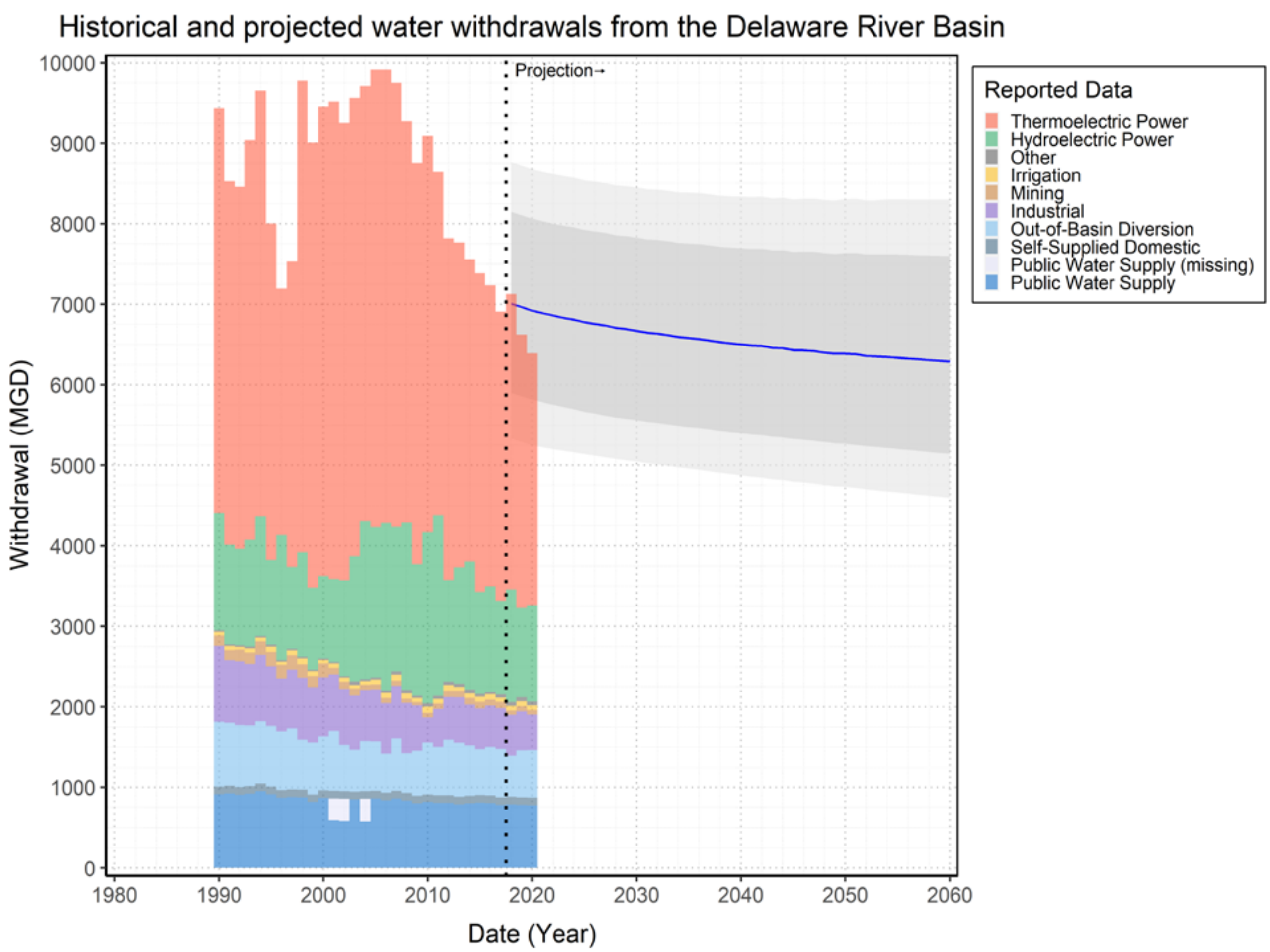


Figure 3.3 Historical and projected water withdrawals from the Delaware River Basin initially published in Thompson & Pindar (2021). The predictive interval shown represents the aggregated predictive intervals for all sectors. The figure has been amended with complete years of data through 2020.

and general public awareness of water conservation. While decreasing in total, withdrawals have increased in several systems where there are population growth regions (i.e., where water conservation practices cannot offset the more rapid increase in population).

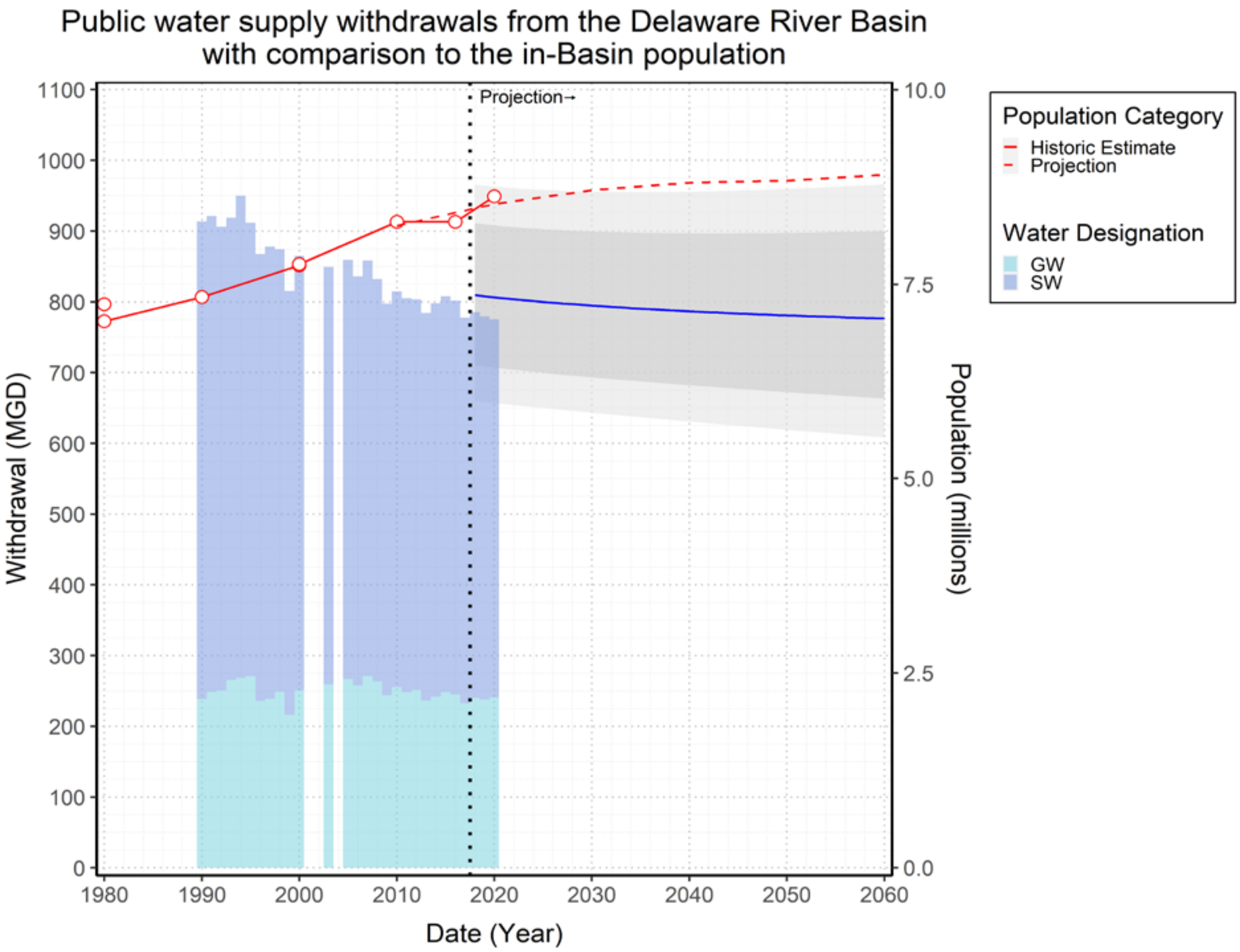


Figure 3.4 Average annual withdrawal rates from the Delaware River Basin (MGD) by public water supply systems, categorized by source water type. Additionally, previous population estimates for in-Basin population as performed by DRBC are graphed as red circles. Data through 2017 and data on projections for both public water supply withdrawals and in-Basin population are adopted from Thompson & Pindar (2021). This graphic has been updated with withdrawal data through 2020, as well as the U.S. Census Bureau data for 2020 as part of the FY2023-2025 Water Resources Program (DRBC 2022a). Years of significant data gaps in withdrawal data have been omitted.

Actions and Needs

The quality and completeness of water withdrawal data are continuously advancing due to the progression of state reporting programs, electronic web-based reporting, and advancements in data sharing, such as online portals and reports. It is likely that advancements in data sharing and availability will continue into the future. Known data gaps exist in certain sectors (mining) where electronic reporting and data sharing are assumed to be not on par with other sectors, which is a possible area for future improvement. There is continuous improvement in reporting compliance and accuracy within the irrigation sector. Improvements in obtaining and sharing the most recent data related to water withdrawals and use within the Delaware River Basin will continue to support advances in quantifying the instream needs of aquatic ecosystems, which are necessary for achieving a balance between instream and off-stream (withdrawal) water needs.

Summary

Advances in the collection and reporting of water withdrawals, primarily by state agencies, have improved our understanding of water use in the Delaware River Basin and its watersheds. A recent comprehensive study performed by DRBC assessed historical water withdrawals and consumptive use in the Delaware River Basin from 1990-2017, with projections provided through the year 2060 (Thompson & Pindar 2021); data on historical withdrawals and projections have been made publicly available on the DRBC website. This dataset has helped DRBC support many important findings, such as that peak water withdrawal from the Delaware River Basin has likely occurred (around 2006 it was estimated to be approximately 9.917 billion gallons per day). Withdrawals by public water supply systems have decreased over time and are projected to continue decreasing despite growing populations in the Basin. Withdrawal by thermoelectric facilities decreased by almost 2,500 MGD since 2007 due largely to reduced operation and/or closure of coal-fired facilities using once-through cooling technology; consumptive use by this sector has been and is projected to remain relatively constant as more evaporative cooling is utilized. Overall, Basin-wide water withdrawals are projected to continue decreasing, from a projected value of 6,921 MGD in 2020 to about 5,670 MGD in 2060; the estimated value based on reported data for 2020 is 6,390 MGD.

3.2 Consumptive Use

Description of Indicator

The previous section largely focused on total withdrawals of water from the Delaware River Basin; however, another important planning metric considers how much of that withdrawn water is actually returned to the immediate environment (or consequently how much of the withdrawal is consumed). For the DRBC, consumptive use of water is defined in 18 CFR Part 420 as:

> *"...the water lost due to transpiration from vegetation in the building of plant tissue, incorporated into products during their manufacture, lost to the atmosphere from cooling devices, evaporated from water surfaces, exported from the Delaware River Basin, or any other water use for which the water withdrawn is not returned to the surface waters of the basin undiminished in quantity."*

Generally, the consumptive use of water is calculated by applying a percentage to the total withdrawal of water (that percentage can vary significantly between withdrawal sectors). For example, Thompson & Pindar (2021) provide a table of the default consumptive use ratios for each sector and the category used in that report (along with cited literature as applicable). Irrigation is highly consumptive (an estimate of 90% or greater is often used) as the water is absorbed by plants, soil, or lost to evaporation. Public water supplies are typically considered to have a consumptive use of 10%, as only a small portion of the water used in homes and cities is evaporated, and the majority is returned to the hydrologic system via sewerage systems. Thompson & Pindar (2021) exclusively applied default consumptive use ratios in all but two sectors for the study (and consequently this analysis); thermoelectric power generation and self-supplied industrial. For these two sectors, under certain circumstances, consumptive use data are reported annually, and wherever possible calculated historical average consumptive use rates were applied.

Present Status

Understanding the consumptive use of water is an integral component of adequate water resource planning. Figure 3.2 shows the Basin-wide picture of water withdrawals, out-of-Basin diversions, and in-

Basin consumptive use, based on the 2020 calendar year water use data; the data shown represents daily average rates on the annual basis (DRBC 2022a).

Key Delaware River Basin Water Consumptive Use Facts:

- The largest component of consumptive use for the Basin are out-of-Basin diversions, which were established as part of the 1954 Supreme Court Decree and are managed separately from other withdrawals and discharges in the Basin.
- Of the remaining 263 MGD of in-Basin consumptive use, the four major components are thermoelectric facilities (92 MGD, 35%), public water supply (83 MGD, 32%), irrigation (49 MGD, 19%), and industrial (17 MGD, 6.7%).

Past Trends

DRBC recently published a comprehensive report assessing historical water withdrawals and consumptive use in the Delaware River Basin from 1990-2017, with projections provided through the year 2060 (Thompson & Pindar 2021). The historical water withdrawals from the Delaware River Basin were presented in Figure 3.3. From these data, it is possible to calculate the portion of withdrawals, which is consumptively used. The corresponding historical consumptive water use in the Delaware River Basin is presented in Figure 3.5. As the out-of-Basin diversions are such a significant portion of Figure 3.5, it is sometimes beneficial to assess each of the other four sectors individually, as shown in Figure 3.6. From these two figures, and considering the previous conclusions on total withdrawals, additional findings can be summarized:

- Public water supply: The trends shown in consumptive use by the public water supply sector mirror the withdrawal trends because this sector's consumptive use is calculated with a default consumptive use rate of 10%. Therefore, findings of declining consumptive use contrary to increasing populations are expected.
- Irrigation: The irrigation sector is comprised of multiple uses, of which the primary is agricultural irrigation (other examples: golf courses, tree nurseries, sports fields). These withdrawals were determined to be strongly correlated with climatic variables and were projected using Regional Climate Model data (Thompson & Pindar 2021). Withdrawal reporting compliance and accuracy appear to be increasing. Consumptive use for this sector is also calculated using a default rate of 90%.
- Industrial: The industrial sector has most of the consumptive use data calculated using reported facility-specific consumptive use rates, or adopted from regulatory approvals; Thompson & Pindar (2021) provides a graphic showing the proportions of data by a calculation method. Overall, the industrial sector has shown significant declines in both withdrawals and consumptive use, often attributed to declines in production at specific facilities (e.g., the U.S. Steel Fairless Plant stopped iron and steel production in 1991, Bethlehem Steel ceased production in 1995, there was a temporary shutdown of Delaware City Refinery in 2010, and Philadelphia Energy Solutions closed in 2019). Despite these historical declines, the projection for industrial withdrawals and consumptive use suggests a lower plateau and continuation at a relatively constant rate.
- Thermoelectric: Water withdrawn for thermoelectric power generation is most commonly used for cooling (e.g., non-contact cooling water). The specific technology used for cooling at a given facility drastically affects the amount of water used. For example, recirculating cooling systems withdraw less water than once-through systems but generally have much higher rates of evaporation and therefore consumptive use. Historical decreases in water withdrawals by thermoelectric facilities were shown to be strongly correlated with decreases in energy net generation from coal-fired

steam-turbine facilities using once-through cooling (Thompson & Pindar 2021). The decrease in withdrawals by thermoelectric facilities between 2007-2017 was around 1,920 MGD, of which about 1,850 MGD was attributable to facilities using once-through cooling. Therefore, it is not surprising that the decreases in total withdrawals have not translated into reduced consumptive use, as shown in Figure 3.6A. To further investigate this concept, consumptive use data can be categorized by each facility's cooling system type, as shown in Figure 3.7. This graphic quantifies that an increasing proportion of overall consumptive use is attributed to facilities with recirculating cooling towers, which comparatively withdraw less water but operate at higher rates of consumptive use than facilities with once-through cooling. The Commission manages the replacement of consumptive use from electric generating and cogenerating facilities during critical hydrologic conditions in accordance with Resolution 2018-5, which accounts for a facility's location in the basin and its impact on the salt front.

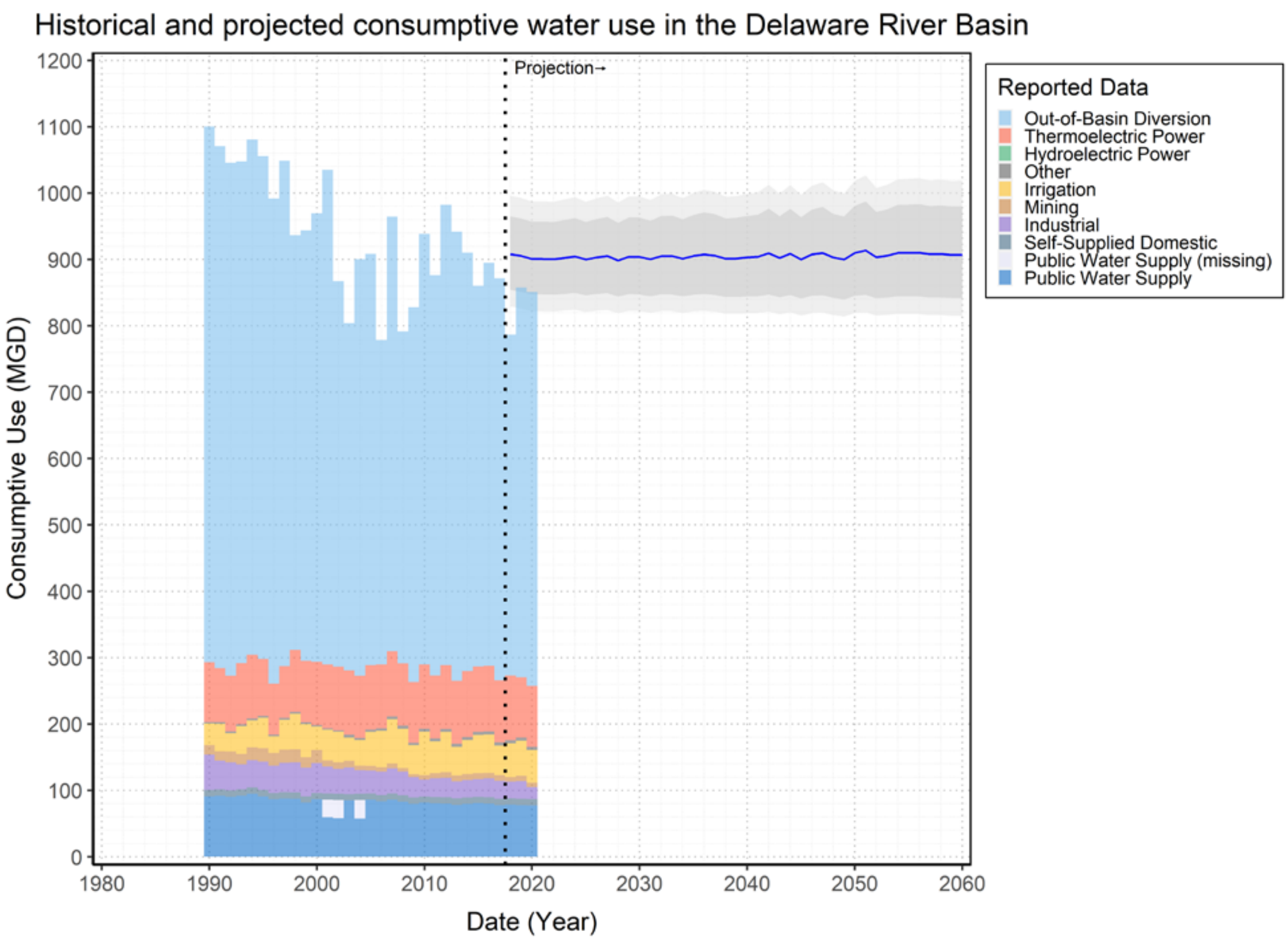


Figure 3.5 Historical and projected consumptive water use in the Delaware River Basin initially published in Thompson & Pindar (2021). The predictive interval shown represents the aggregated predictive intervals for all sectors, excluding the out-of-Basin diversions which did not have a calculated predictive interval. The figure has been amended with complete years of data through 2020.

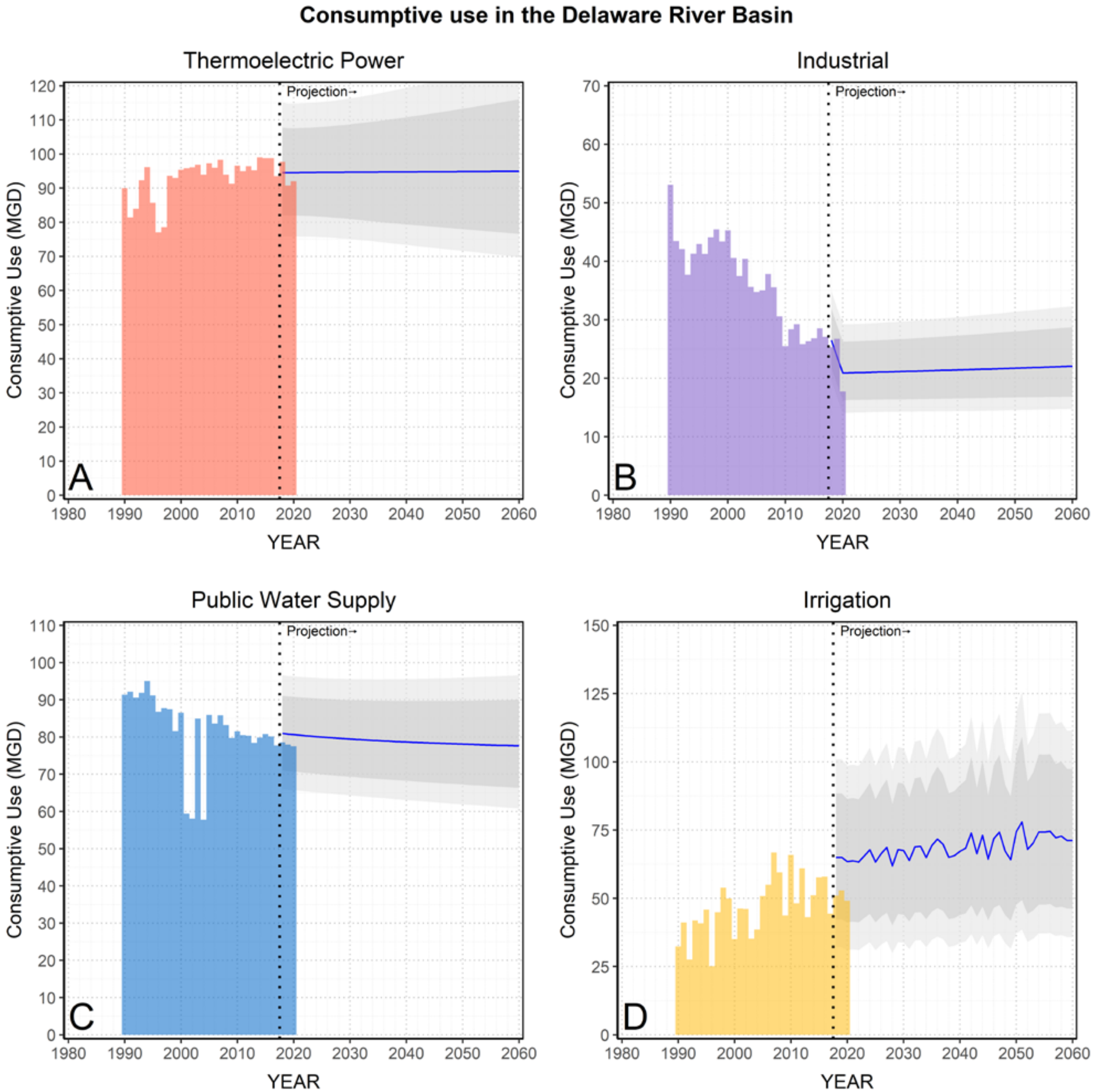


Figure 3.6 Historical and projected consumptive water use in the Delaware River Basin for major sectors.

Future Predictions

Consumptive use trends over the past three decades were shown in Figure 3.5, as well as a Basin-wide projection as determined in Thompson & Pindar (2021); individual sectors trends and projections were shown in Figure 3.6. At the Basin-scale, consumptive use was projected to remain relatively stable. Considering the four highlighted sectors:

- Public Water Supply (PWS): Projections show continued decreases in consumptive use, consistent with the projections of overall withdrawal, based on the same drivers previously discussed. This is due to the method of calculating consumptive use for public water supply.

- Irrigation: Consumptive use was projected using a multivariate model developed based on climate data from a regional climate model for the Delaware River Basin area. Irrigation volumes were shown to share a direct relationship with temperature; therefore, increasing temperatures in the regional climate model helped drive a projection of increasing irrigation withdrawals and consumptive use.
- Industrial: Projections accounted for one major facility shutdown, as the event occurred during model development. However, barring additional circumstances such as the major facility shifts discussed previously, the projection shows a relatively stable and even slightly increasing projection of consumptive use.

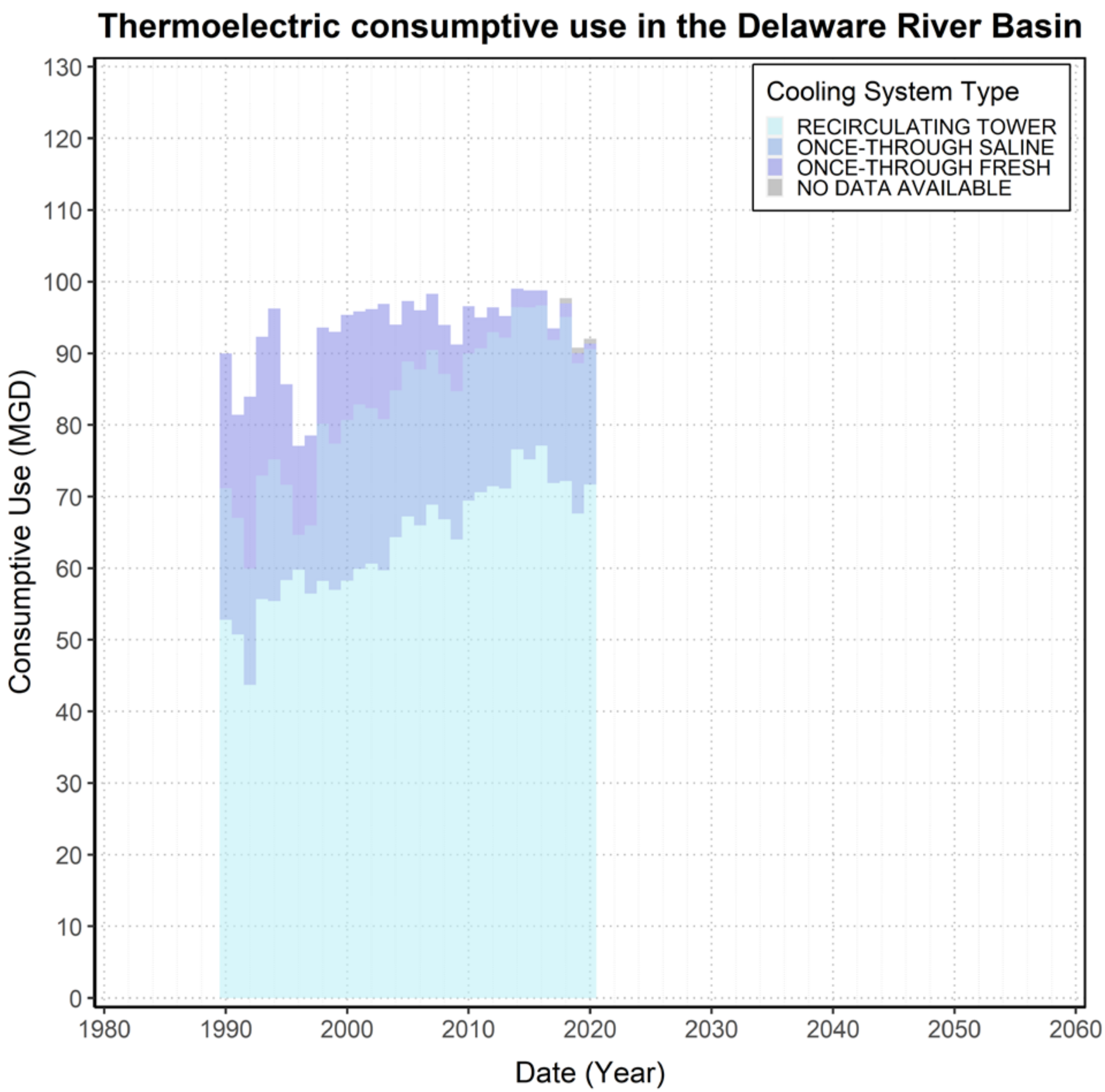


Figure 3.7 Historical consumptive water use in the Delaware River Basin by thermoelectric facilities (same data as Figure 3.6A) color-coded by cooling system type, initially published in Thompson & Pindar (2021). This graphic has been updated with withdrawal data through 2020.

- Thermoelectric: Projections of consumptive water use at thermoelectric facilities were almost constant; it is important to note that the projection methodology did not consider the potential for developing new power generation facilities and is only projecting existing water use patterns given a continuation of the current trends, outlined in the assumptions of Thompson & Pindar (2021). If new facilities were to be constructed, it is likely that they would rely on cooling towers, and potentially increase the overall consumptive use without greatly affecting tends in total withdrawals.

Actions and Needs

An accurate consumptive use characterization for a watershed requires a detailed analysis of each water use sector to determine consumptive use factors representing site-specific conditions. For example, at a small watershed scale, the simple assumption of 10% consumptive use for a PWS system that withdraws from the watershed but discharges wastewater outside the watershed would be inaccurate. This would need to be modeled as 100% consumptive or as an export from the sending watershed and an import of wastewater (minus the 10% consumptive use) to the receiving watershed. More detailed tracking models that link withdrawal volumes more explicitly to discharge volumes are being applied in the Delaware River Basin, such as by New Jersey Geological Survey's Water Transfer Data System.

Summary

An understanding of consumptive water use provides additional insight into water use patterns and is an important indicator in the management of water resources. Within the Delaware River Basin, the largest consumptive uses are the out-of-Basin diversions. Beyond these diversions, the four major components are thermoelectric facilities, public water supply, irrigation, and industrial facilities (constituting approximately 92% of the in-Basin consumptive use). Slightly downward consumptive use trends are expected to continue in the public water supply sector, while neutral trends may continue in the thermoelectric and industrial sectors. Irrigation withdrawals were shown to be correlated with climatic variables, and projections suggest the possibility for mild increases in consumptive use, due primarily to the projected increase in ambient air temperature.

3.3 Groundwater Availability

Description of Indicator

Stress on a groundwater resource system can occur when withdrawals exceed natural recharge. Withdrawal of groundwater through wells is stress superimposed on a previously balanced groundwater system. The response of an aquifer to pumping stress may result in modifications to recharge of the aquifer, a decrease in the natural discharge to streams, a loss of storage within the aquifer, or a combination of these effects. Additionally, impacts may extend beyond the limits of the aquifer being monitored. Therefore, it is important to monitor and assess groundwater withdrawals on various planning scales to better understand where areas of potential stress might exist within the Delaware River Basin. One such tool is a groundwater availability screening method developed in 2006 by the USGS in cooperation with the DRBC (Sloto & Buxton 2006). The methodology assesses the Delaware River Basin as 147 separate sub-basins (Fig 3.8), each of which has stream baseflow from groundwater characterized based on the underlying geology. These baseflows are presented as recurrence intervals (for example, a 25-year recurrence interval is the groundwater baseflow to a particular stream that is expected to occur once in 25 years, abbreviated as RI-25). The method then looks at the net groundwater withdrawals as a percentage of the groundwater baseflow for each sub-basin to assess levels of stress. This method has limited applicability in the Coastal Plain portion of the Basin, which has a complex confined aquifer network; the method is only suited to evaluate withdrawals from unconfined aquifers.

Two major areas, primarily within the watersheds of the Upper Estuary and Schuylkill Valley, have shown signs of potential stress and are recognized as critical or protected areas. These groundwater management areas are shown in Figure 3.8, and are referred to as:

- The Southeastern Pennsylvania Groundwater Protected Area (SEPA-GWPA), which is largely underlain by fractured bedrock, and
- Critical Area 2 in south-central New Jersey, which is underlain by unconsolidated sediments.

DRBC assesses groundwater availability in the SEPA-GWPA using a similar methodology as Sloto & Buxton (2006). The SEPA-GWPA is divided into 76 different sub-basins and groundwater baseflow values were calculated in previous USGS studies in cooperation with DRBC (Schreffler 1996; USGS 1998). The New Jersey Department of Environmental Protection (NJDEP) is the agency responsible for the management and monitoring of Critical Area 2, which is focused on the confined aquifer network in the Atlantic Coastal Plain. New and/or expanded withdrawals in both critical areas are limited and managed, subject to specific regulations that allocate the resource on the basis of a sustainable long-term yield.

Present Status

DRBC has calculated groundwater availability in the Delaware River Basin and for the Southeastern Pennsylvania Groundwater Protected Area (SEPA-GWPA) based on net groundwater withdrawal data from 2020 (Figures 3.9 and 3.10). The present status of groundwater availability in the Basin and in the SEPA-GWPA are summarized below.

Basin-wide: Withdrawal data was adjusted to represent "net" groundwater withdrawals, which is assumed to be the difference between the total groundwater withdrawal from a sub-basin and water recharge occurring within the same sub-basin. Figure 3.9 indicates that one sub-basin is currently between 50.1% and 75% of its 25-year annual baseflow, and two sub-basins are between 25.1% - 50% of its 25-year annual baseflow. The 26 grayed-out basins in Figure 3.9 represent areas where DRBC determined that the methods developed by Sloto & Buxton (2006) are not applicable because groundwater withdrawals in those areas are predominately from confined aquifers in the Coastal Plain (DRBC 2022b). A primary assumption made in Sloto & Buxton (2006) assumes that groundwater withdrawals from the Coastal Plain are from unconsolidated sediments only, as confined aquifer networks may have regional influences which extend beyond sub-basin boundaries.

SEPA-GWPA: The Southeastern Pennsylvania Groundwater Protected Area is an area of 1,200 square miles that includes 127 municipalities, primarily in Bucks, Chester, and Montgomery counties. As municipal boundaries are not defined based on hydrologic parameters, 76 hydrologic sub-basins were delineated to cover all municipalities within SEPA-GWPA (Fig 3.10). DRBC monitors these sub-basins regarding groundwater withdrawals, well interferences, and municipal water supply planning; furthermore, withdrawal limits have been established for each based on previous work by USGS in cooperation with DRBC (Schreffler, 1996; USGS, 1998). The following summary of conditions is based on an analysis using groundwater withdrawal data reported to the Pennsylvania Department of Environmental Protection (PADEP) adjusted to represent net groundwater withdrawals. DRBC will continue to update Delaware River Basin usage with current PADEP water withdrawal data.

- In Figure 3.10, the 2020 net groundwater withdrawal in three of the sub-basins is currently between 50.1% and 75% of the respective annual sub-basin withdrawal limit, and one sub-basin is above the respective withdrawal limit. Sub-basin SP-29 (Schuylkill-Crow Creek) has historically

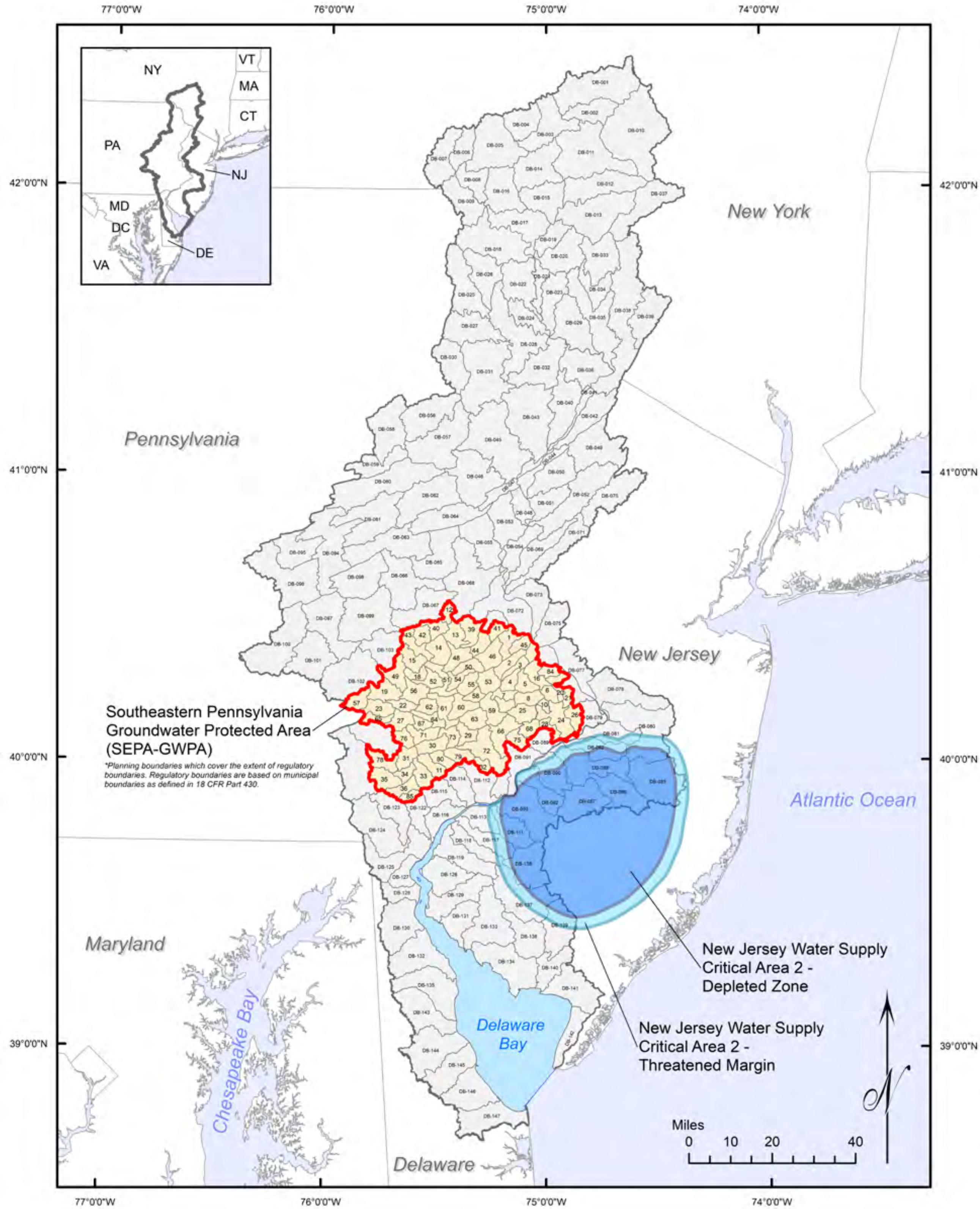


Figure 3.8 Groundwater management areas in the Delaware River Basin.

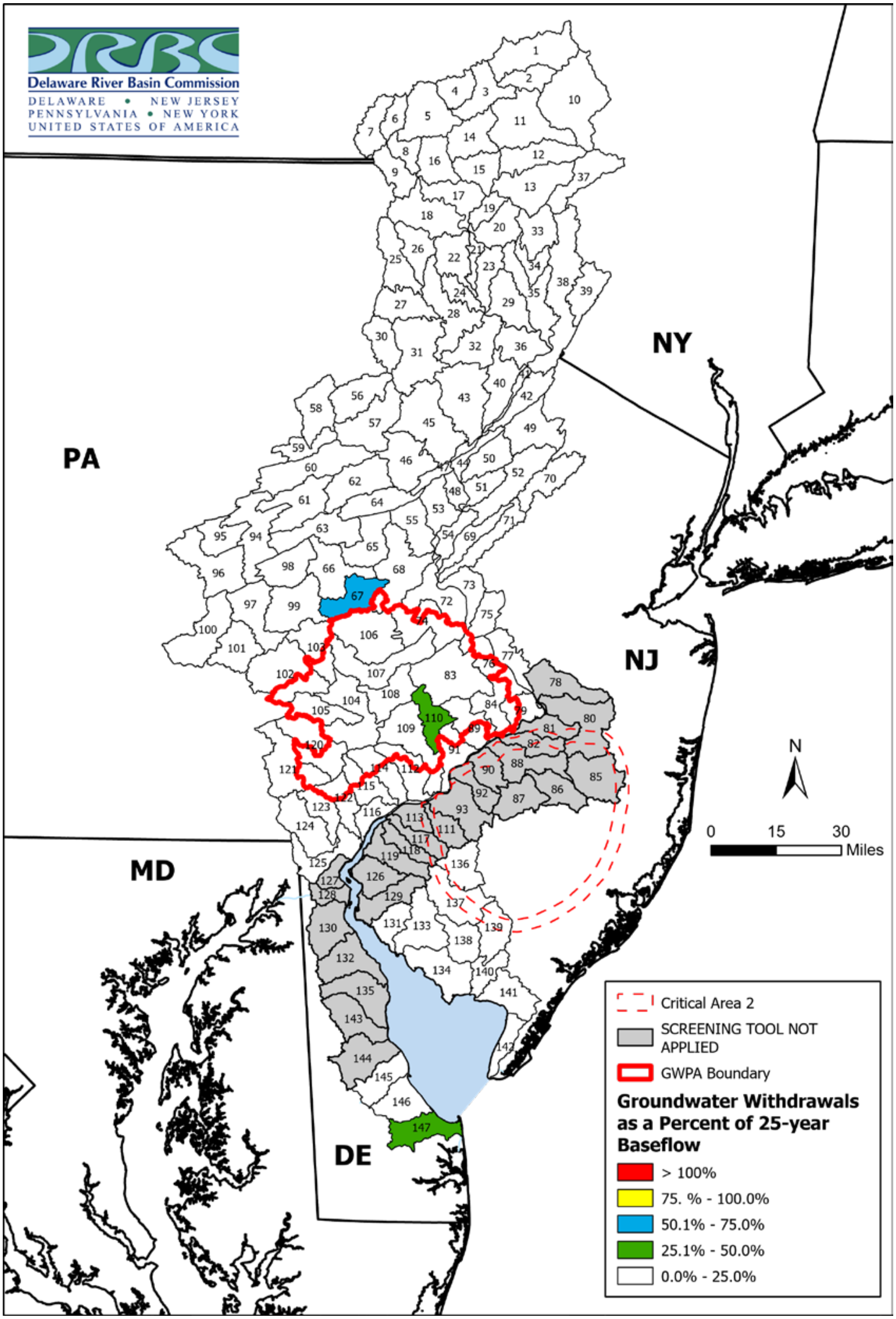


Figure 3.9 Net groundwater withdrawals from the Delaware River Basin for CY2020. Greyed-out sub-basins denote areas where the screening tool was not applied.

been above its withdrawal limit because a large withdrawal from a quarry reservoir is counted as a groundwater withdrawal.

- Reductions in total annual net groundwater withdrawals have been observed over the past two decades. "Conjunctive use" strategies (i.e., adaptive use of both ground and surface water) and regional alternatives to the local supplies are strategies being employed within the SEPA-GWPA.

Critical Area 2: The New Jersey Water Supply Critical Area 2 was designated on July 20, 1993, by administrative order (NJAC 7:19-8.5). This area includes the Potomac-Raritan-Magothy (PRM) aquifer system, with extents based on findings from Eckel & Walker, 1986. The New Jersey Department of Environmental Protection (NJDEP) and the United States Geological Survey (USGS) regularly monitor groundwater levels in the affected aquifers of Critical Area 2 in southern New Jersey, and assessments indicate that withdrawals have significantly decreased beginning with the program's inception in 1996. Data provided by NJDEP for 2019 show that there were about 17,434 million gallons withdrawn from the "Depleted Zone," and about 5,377 million gallons withdrawn from the "Threatened Margin" of the PRM aquifer.

Past Trends

DRBC recently published a comprehensive report assessing historical water withdrawals and consumptive use in the Delaware River Basin from 1990-2017, with projections provided through the year 2060 (Thompson & Pindar 2021). Using data provided as part of that study, a time series of historical net withdrawals for the Delaware River Basin and for the SEPA-GWPA were compiled and updated through 2020 (Figures 3.11 and 3.12). The 30 years of data show that net groundwater withdrawals have changed at both the Basin scale and within the SEPA-GWPA, and this knowledge can help make better planning decisions.

Basin-wide: Groundwater withdrawals in the Basin have varied between about 340 MGD to 400 MGD for the past 30 years (Fig 3.11). More recently annual net groundwater withdrawals hover around 330 MGD with a slight decrease in the past few years. Net withdrawals by sector have shifted with a decrease in industrial withdrawals in the past thirty years.

SEPA-GWPA: As shown in Figure 3.12, the reduction in net groundwater withdrawals in the SEPA-GWPA is largely perceived to be due to the adoption of sub-basin withdrawal limits by DRBC in 1999 and increased reliance on surface water derived from the Delaware River. Over the period from 2000 to 2020, cumulative net groundwater withdrawals from the SEPA GWPA have steadily decreased (Fig 3.12). Groundwater pumping stress in several sub-basins has been alleviated by the Point Pleasant diversion in Pennsylvania, which transfers surface water from the Delaware River to serve populations in several SEPA-GWPA municipalities. This diversion has provided a conjunctive use solution that has reduced the reliance on groundwater in several sub-basins. Other aspects of the management program administered by the DRBC in this area include a water conservation program and a lower withdrawal threshold triggering regulatory review (10,000 gallons/month, as compared to 100,000 gallons/month elsewhere in the Delaware River Basin).

Critical Area 2: The New Jersey Water Supply Critical Area 2 was established by the State of New Jersey in 1993 and has resulted in reduced withdrawals from the Potomac-Raritan-Magothy (PRM) aquifer system. Many of the municipalities are now served by surface water diverted from the Delaware River near Delran, NJ. Strategies such as this have led to decreasing withdrawal volumes from both the Depleted Zone and the Threatened Margin of Critical Area 2 (Fig 3.13A). Consequently, aquifer levels have risen and an example is shown in the graph from USGS Elm Tree 3 Observation well (Fig 3.13B), which is located more than 700 ft below the land surface in the Middle PRM aquifer in Camden, NJ.

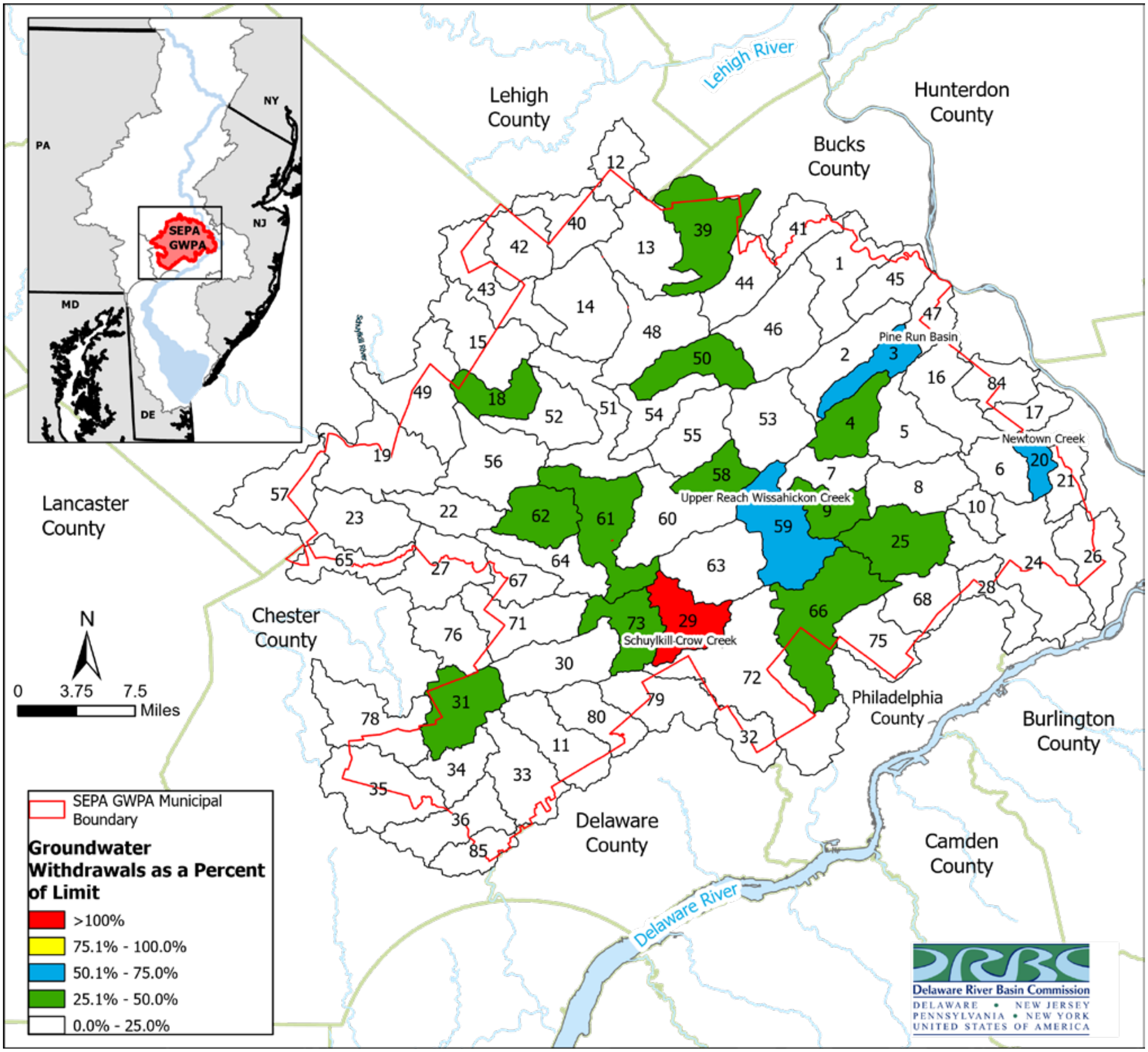


Figure 3.10 Net groundwater withdrawals from the Southeastern Pennsylvania Groundwater Protected Area for CY2020.

Future Predictions

In addition to compiling a historical time series of withdrawal data for the Delaware River Basin, Thompson & Pindar (2021) also provide projections of groundwater withdrawals through the year 2060. These data were published with the report and have been converted to net groundwater withdrawals for the purposes of this study. The detailed methodology behind the projections is contained within the report, but it can be summarized both Basin-wide and for the specially protected areas in the Basin:

Basin-wide: It is projected that net groundwater withdrawals will remain relatively stable with a slight decrease over time (Fig 3.11). At the time of this report, no sub-basins are expected to exceed the RI-25 baseflow determined by Sloto & Buxton (2006).

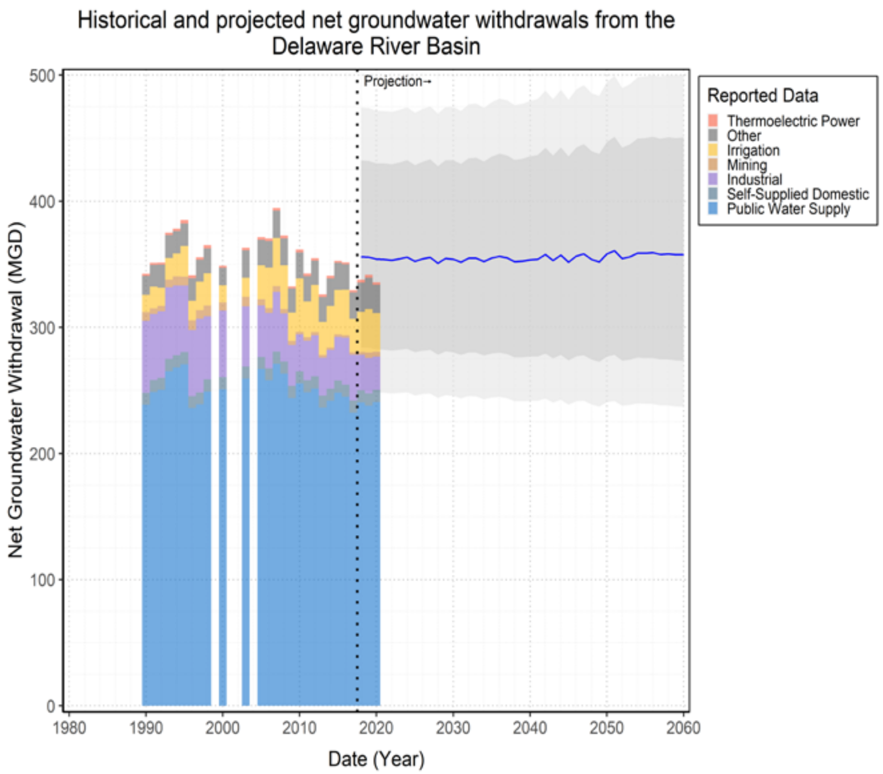


Figure 3.11 Net groundwater withdrawals in the Delaware River Basin 1990-2020. There are known data gaps present for 2001, 2002, and 2004.

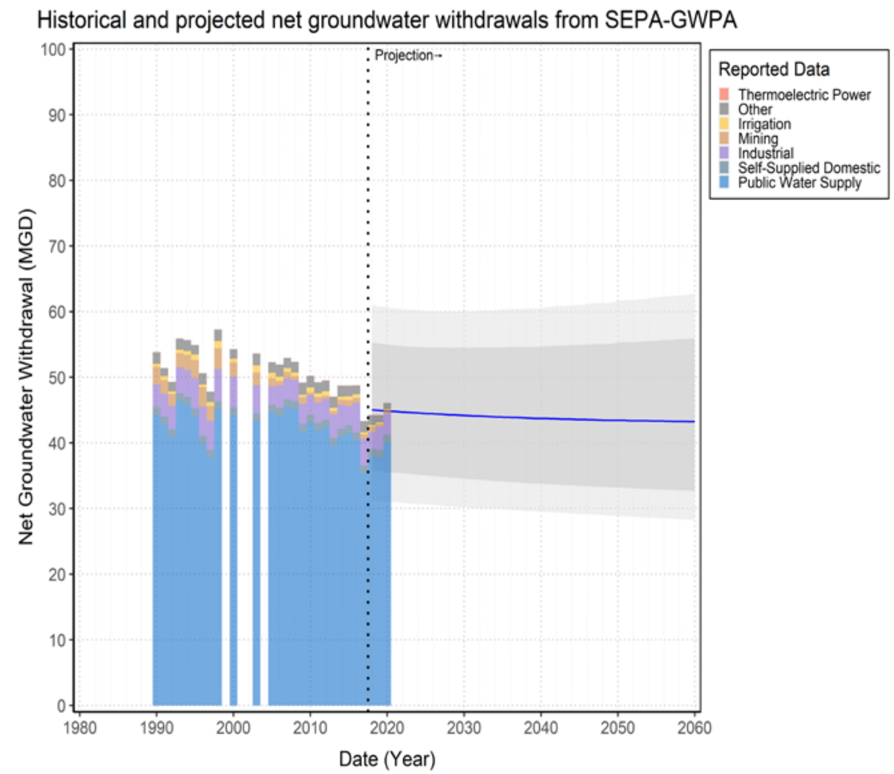


Figure 3.12 Net groundwater withdrawals in the Southeastern Pennsylvania Groundwater Protected Area 1990-2020. There are known data gaps present for 2001, 2002, and 2004.

SEPA- GWPA & Critical Area 2: Groundwater conditions in the SEPA-GWPA and NJ Critical Area 2 are expected to continue to improve over time due to management strategies of the DRBC, Pennsylvania, and New Jersey (Figures 3.12 and 3.13).

Actions and Needs

In recent years, progress has been made to improve water use reporting and these improvements should be continued in order to provide necessary data to monitor and assess conditions in sensitive areas such as the SEPA-GWPA and the New Jersey Water Supply Critical Area 2. In addition, improvements can be made in our understanding of the dynamics distinguishing the confined and unconfined aquifers in the coastal plain. The current Sloto & Buxton (2006) method does not apply to the coastal plain portion of the Basin and an improved understanding would help DRBC and other resource agencies better assess and manage groundwater availability in the region.

Finally, assessing the seasonality of both natural groundwater baseflow and net groundwater withdrawals would allow for a better understanding of seasonal trends and potential stresses in groundwater availability. Certain water use sectors tend to withdraw groundwater more during the summer months, likely resulting in the current annual depiction of availability as an underestimate for certain months of the year. A better understanding of these trends could help with DRBC's and other agencies' availability assessments.

Summary

Overall, the increase in data availability and advancement in data quality have made it possible to better understand historical trends and make future projections for net groundwater withdrawals. Paired with assessments of groundwater baseflows to surface water streams, this net groundwater withdrawal data is used to determine groundwater availability. Historical trends in groundwater withdrawals showed potential stresses in certain sub-basins in Southeastern Pennsylvania and in the Coastal Plain of New Jersey. This led to the development of two groundwater management areas: SEPA-GWPA and Critical Area 2.

Groundwater availability screening tools are used to assess groundwater availability on the Basin wide scale (Sloto & Buxton 2006) and for SEPA-GWPA (Schreffler 1996; USGS 1998). In an analysis of calendar year 2020 Basin-wide groundwater availability indicates that it is generally stable. Analyses of both groundwater management areas show a continued decrease in groundwater withdrawals.

Basin-wide net withdrawals are projected to remain stable with a slight decrease in the next forty years. In the groundwater management areas, groundwater withdrawals are expected to continue to decrease as management strategies continue to be implemented. The two management areas described in this section are examples of successful, proactive management strategies that could be applied to other areas undergoing stress due to the withdrawal of groundwater.

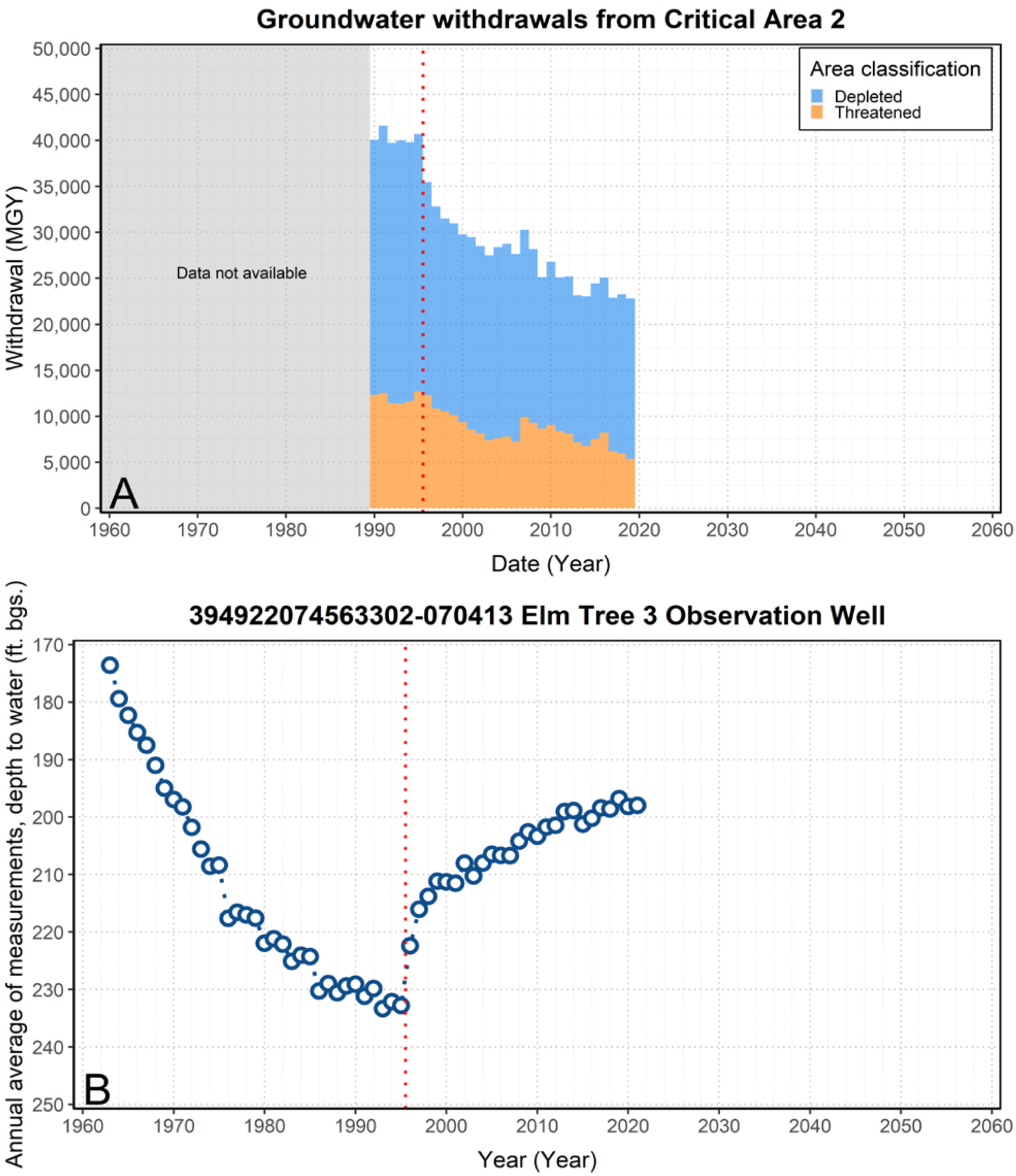


Figure 3.13 (A) Withdrawals from the PRM from 1990-2020 show significant reductions since the inception of Critical Area 2 in the early 1990s. Source: I. Snook, NJDEP, February 2022. (B) Example of rebounding groundwater levels in the upper PRM of NJ Critical Area 2, measured in feet below ground surface (ft. bgs.). Elm Tree 3 observation well, Camden Co., NJ. Period of records shown (02/22/1963 – 12/27/2021), data presented as an annual average of measurements. Source: USGS, February 2022.

3.4 Salt Front Location & Movement

Description of Indicator

The salt front is an estimation of where the seven-day average chloride concentration equals 250 ppm (parts per million) along the tidal Delaware River. The location of the salt front plays an important role in the Delaware River Basin water quality and drought management programs because upstream migration of brackish water from the Delaware Bay during low-flow and drought conditions could increase sodium concentrations in public water supplies, presenting a health concern. Critical intakes on the Delaware River that could be adversely affected by salinity moving upstream are the Philadelphia Water Department's Baxter intake and the New Jersey American Water Company's Delran intake (Fig 3.14). Both intakes are located at approximately river mile 110 (river km 176). In addition, upstream migration of the salt front may have adverse impacts on the PRM aquifer, as high rates of pumping in the PRM draw tidal river water into the aquifer. If the salt front moves too far upstream for an extended period, the presence of sodium could reduce the quality of water in the aquifer.

Present Status

Good: Drinking water intakes in the tidal river are protected by the reservoir releases to meet flow objectives. The water quality in the PRM aquifer remains good.

Past Trends

The salt front moves upstream and downstream with each tidal cycle and seasonal variations in freshwater flow. For most of the year, the location of the salt front is between the Commodore Barry Bridge (RM 82/ KM 131) and Artificial Island (RM 54/KM86). During droughts and periods of low inflow to the Estuary, water is released from reservoirs, in accordance with normal and drought operating plans, to augment flows to meet a daily flow objective of 3,000 CFS (84.9 CMS) in the Delaware River at the Trenton, NJ gage. The program has been effective and, since the 1970s, the salt front has remained more than 18 miles below the drinking water intakes, protecting drinking water supplies in the most urbanized area of the Estuary (Figures 3.14 and 3.15).

Future Predictions

Sea level rise and potential changes to the variability and seasonality of flow from anticipated increases in temperature and precipitation may create additional challenges for the management of the salt front in the future. Additional challenges may be the availability of water for releases from storage due to the need to rehabilitate aging infrastructure and increased demands.

Actions and Needs

An evaluation is needed to determine the ability of the existing drought management plan to impede the upstream movement of the salt front, considering factors such as sea level rise and changes in hydrology resulting from an anticipated increase in temperature and precipitation. An investigation into the contribution of additional sources of chlorides, such as road salts, on the estuary salinity and associated impacts, is appropriate.

Summary

Flow management strategies have been successful in impeding the upstream movement of the salt front and have effectively protected drinking water intakes in the most densely populated area of the Basin.

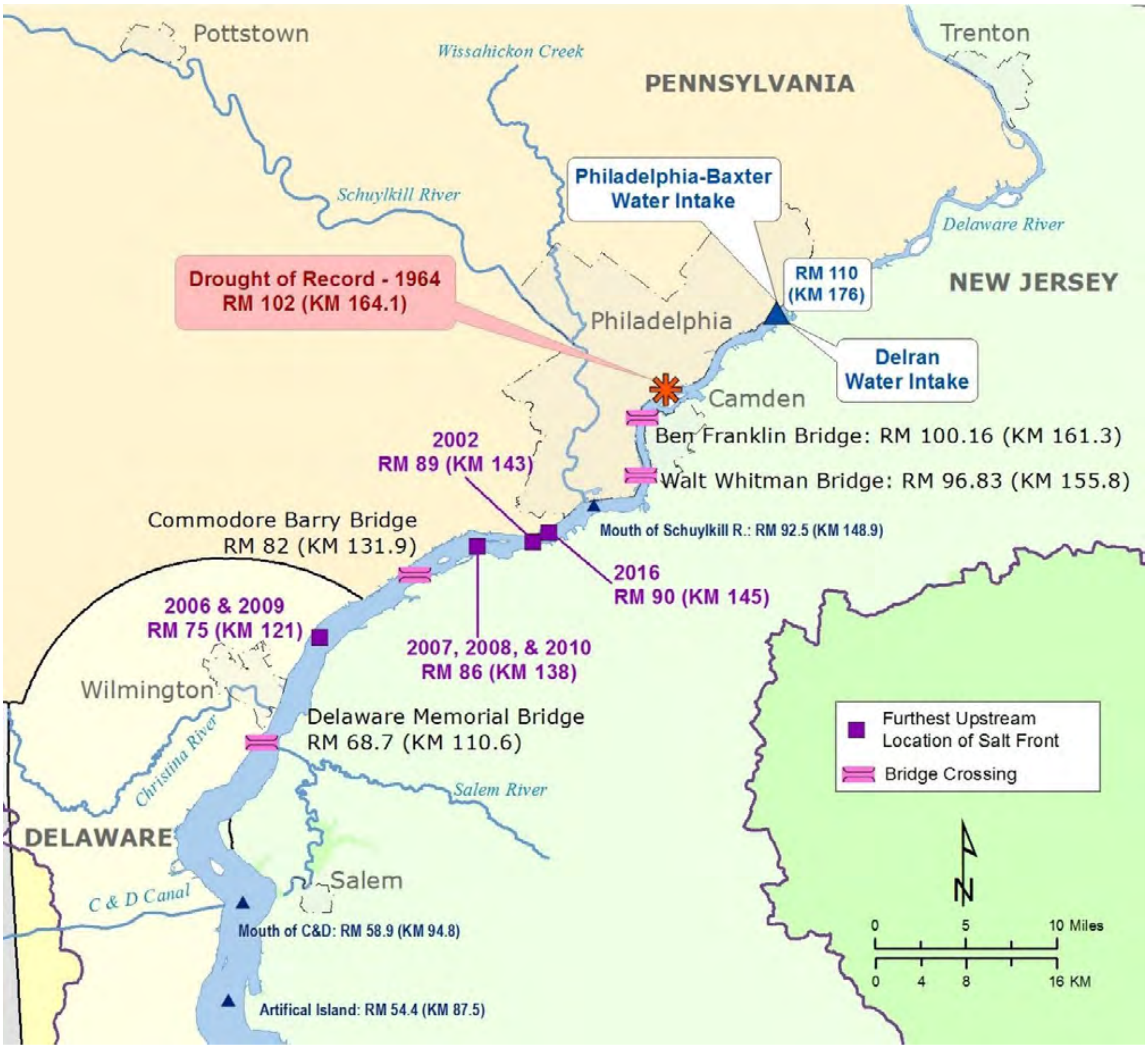


Figure 3.14 Map of historic salt front locations.

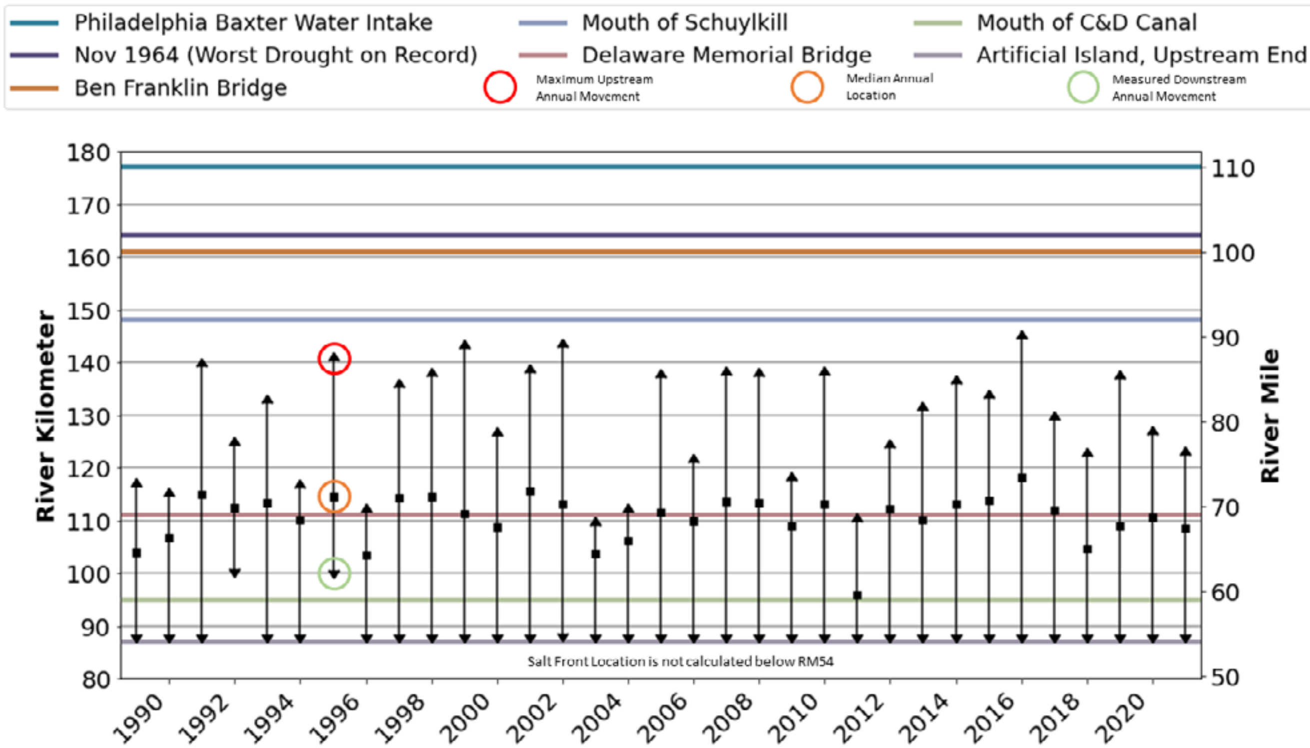


Figure 3.15 Range of Annual Salt Front Locations From 1989-2021. The salt front river mile location is estimated by DRBC using data from the USGS water quality measurements.

References

18 CFR Part 420. (2019). Basin Regulations - Water Supply Charges. With amendments through July 1, 2019. Delaware River Basin Commission. https://www.nj.gov/drbc/library/documents/water_chargesCFR.pdf

Byun, S. A., Kwityn, E., Pindar, C., & Thompson, M. (2019). State of the Basin 2019. Ewing, New Jersey. Delaware River Basin Commission. https://www.nj.gov/drbc/about/public/SOTB2019.html

DRBC. (2022a). Delaware River Basin Commission Water Resources Program FY 2023-2025. West Trenton, New Jersey.

DRBC. (2022b). Estimated Groundwater Availability in the Delaware River Basin, 1990-2060. Authored by S. Sayed, M. Thompson, C.E. Pindar, S. Beganskas. Report in process and anticipated for publication in 2022.

Eckel, J. A., & Walker, R. L. (1986). Water Levels in Major Artesian Aquifers of the new Jersey Coastal Plain, 1983. Water-Resources Investigations Report 86-4028. Trenton, New Jersey. U.S. Geological Survey. https://pubs.usgs.gov/wri/1986/4028/report.pdf

Hauer, M., & CIESIN. (2021). Georeferenced U.S. County-Level Population Projections, Total and by Sex, Race and Age, Based on the SSPs, 2020-2100. Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). https://doi.org/10.7927/DV72-S254

Hauer, M. E. (2019). Population projections for U.S. Counties by age, sex, and race controlled to shared socioeconomic pathway. Scientific Data, 6, 190005. https://doi.org/10.1038/sdata.2019.5

NJAC 7:19-8.5. N.J. Admin. Code § 7:19-8.5 - Water Supply Critical Area II. State of New Jersey. https://www.nj.gov/dep/rules/rules/njac7_19.pdf

O'Neill, B. C., Kriegler, E., Riahi, K., Ebi, K. L., Hallegatte, S., Carter, T. R., Mathur, R., & van Vuuren, D. P. (2014). A new scenario framework for climate change research: The concept of shared socioeconomic pathways. Climatic Change, 122(3), 387–400. https://doi.org/10.1007/s10584-013-0905-2

Schreffler, C. L. (1996). Water-use analysis program for the Neshaminy Creek basin, Bucks and Montgomery counties, Pennsylvania: Water-Resources Investigations Report 96-4127, Prepared in cooperation with the Delaware River Basin Commission. Lemoyne, Pennsylvania. U.S. Geological Survey. https://doi.org/10.3133/wri964127

Sloto, R. A., & Buxton, D. E. (2006). Estimated Ground-Water Availability in the Delaware River Basin, 1997-2000. Scientific Investigations Report 2006-5125--Version 1.1. Reston, Virginia. U.S. Geological Survey. https://pubs.usgs.gov/sir/2006/5125/pdf/sir2006-5125_ver1.1.pdf

Thompson, M. Y., & Pindar, C. E. (2021). Water Withdrawal and Consumptive Use Estimates for the Delaware River Basin (1990-2017) With Projections Through 2060:. DRBC Report No: 2021-4. West Trenton, New Jersey. Delaware River Basin Commission. https://www.nj.gov/drbc/library/documents/water-use/DRBC_2021-4_Water2060_Final_101421.pdf

USGS. (1998). Southeastern Pennsylvania ground water protected area: Open-File Report 98-571 (Open-File Report). U.S. Geological Survey. https://doi.org/10.3133/ofr98571

4

TECHNICAL REPORT FOR THE ESTUARY AND BASIN

Water Quality

Partnership for the DELAWARE ESTUARY

December 2022 | Report No.22-05

Partnership for the Delaware Estuary—Host of the Delaware Estuary Program

4 Water Quality

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4. Water Quality

Abstract

Understanding trends in water quality is an important factor for the protection of both aquatic life and human health in the Delaware River and Estuary. In this chapter, we evaluated trends in dissolved oxygen, temperature, specific conductivity, pH, chloride, nitrate, ammonia, phosphate, and toxic contaminants in both tidal and non-tidal waters using publicly available data from the National Water Quality Portal. Trends varied by parameter and location with some parameters showing improvement, others showing degradation, and some remaining relatively constant. Factors such as climate change, sea-level rise, and development can have major impacts on water quality in the Delaware River and Estuary and work should continue to closely monitor how these drivers may be impacting water quality and its linkages to both aquatic life use and human health. Additionally, work should continue to monitor contaminants of emerging concern as they continue to arise in the system.

4.1 Chemistry and Nutrients

Data Sources and Processing

Data used in this report come primarily from United States Geological Survey (USGS) continuous monitors, USGS discrete monitoring, and Delaware River Basin Commission (DRBC) monitoring programs. Aggregated available data sets were also queried via the National Water Quality Data Portal. Where multiple data sets exist, the authors relied on data for which we had the best firsthand knowledge of quality assurance and quality control. There are unlimited options available for sub-setting data and presenting it graphically. The author chose data periods and graphical representations in each instance that conveyed the best understanding of the data.

4.1.1 Tidal – Chemistry and Nutrients

4.1.1.1 Dissolved Oxygen

Description of Indicator

Dissolved oxygen (DO) refers to the concentration of oxygen gas incorporated in water. Oxygen enters water both by direct absorption from the atmosphere, which is enhanced by turbulence and weather, and as a by-product of photosynthesis from algae and aquatic plants. Sufficient DO is essential to growth and reproduction of aerobic aquatic life such as fish and invertebrates. Oxygen levels in water bodies can be depressed by the discharge of oxygen-depleting materials (measured in aggregate as biochemical oxygen demand (BOD) from wastewater treatment facilities and stormwater runoff), from the decomposition of organic matter including algae generated during nutrient-induced blooms, and from the oxidation of ammonia and other nitrogen-based compounds. Conditions where oxygen levels are depressed are called hypoxic, when there is no oxygen this is called anoxic. The Delaware Estuary has historically been plagued by anoxic and hypoxic conditions that resulted from the discharge of raw and poorly treated wastewater. Although the Estuary has seen a remarkable recovery since the 1960s, evidenced by fish such as striped bass and sturgeon now exhibiting some level of propagation in the Estuary, dissolved oxygen remains a critical issue for the Estuary because of continued depression of oxygen levels below 100% saturation.

Present Status

Dissolved oxygen is measured monthly at multiple locations as part of DRBC's Delaware Estuary Water Quality Monitoring Program (formerly called the Boat Run) and continuously by USGS year-round at Reedy Island (01482800), and April through November at Chester (01477050), and the Ben Franklin Bridge (01467200). DRBC's water quality standard for dissolved oxygen in the Estuary is a 24-hour average concentration not less than 5.0 mg/L in Zone 2, 3.5 mg/L in Zones 3, 4, and the upper portion of Zone 5, 4.5 mg/L in the middle portion of Zone 5, and 6 mg/L in the lower portion of Zone 5. In the most recent Delaware River and Bay Water Quality Assessment, greater than 99% of observations met criteria in Zones 2 through 4, while greater than 95% and 90% of observations met criteria in Zone 5 and 6, respectively. DRBC has developed a daily near real-time assessment of DO comparing the 24-hour mean concentrations at USGS monitors to the DRBC surface water quality standard available online.

USGS continuous monitor data (Fig 4.1.1) shows that dissolved oxygen concentrations are highest at Reedy Island (River Mile 54.1, Zone 5), lower at Chester (River Mile 83.1, Zone 4) and lowest at the Ben Franklin Bridge (River Mile 100.05, Zone 3).

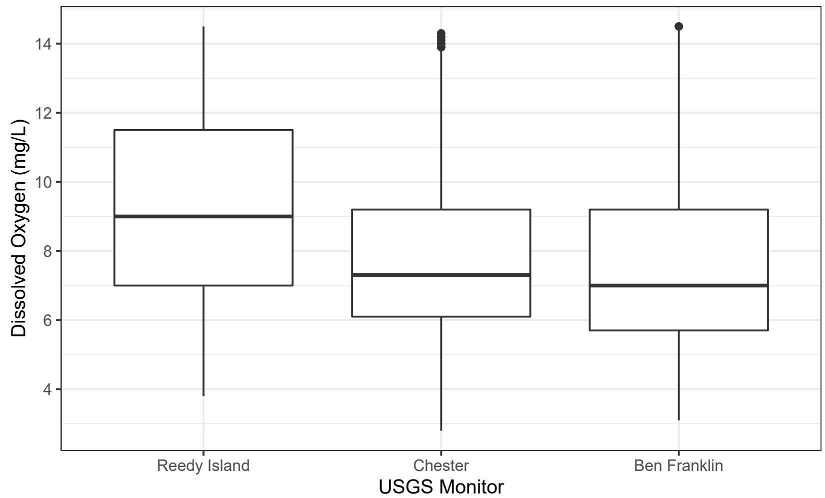

Figure 4.1.1 Delaware Estuary Dissolved oxygen measured by USGS continuous monitors, 2011 through 2020. Boxplots display the median (horizontal line), interquartile range (boxes), values up to 1.5 times the interquartile range (whiskers) and outliers greater than 1.5 times the interquartile range (dots).

Past Trends

USGS's continuous dissolved oxygen measurements began in 1964. Historically, DO concentrations are lowest in mid-summer. As shown in Figure 4.1.2, the July dissolved oxygen concentrations were historically below the current Zone 3 standard of 3.5 mg/L in the 1960s and 1970s. Improvements in DO became apparent through the 1980s as municipal wastewater treatment facilities added secondary treatment for sewage. From the mid 1990s onward, criteria were mostly met, although DO concentrations exhibit a high level of variability from year to year. DO at the Ben Franklin Bridge was mostly above 5 mg/L from 2017 to 2019, but 2020 represented a year with lower DO. Even though DO conditions were lower in 2020, the 3.5 mg/L criterion was still met. Figure 4.1.3 shows box plots for daily minimum DO at the same location, over the same time period.

Figure 4.1.4 is a box and whisker plot of all July daily mean % of dissolved saturation values by year for the Delaware River at the Ben Franklin Bridge. Since % of saturation was not historically reported at this location, values were computed using the daily mean water temperature and atmospheric pressure, and assuming specific conductance of 229 uS/cm (the median for this location for this period of record).

Future Predictions

Documentation of fish spawning in the Delaware Estuary (Silldorff 2015) and the designation of the Delaware Estuary as Critical Habitat for Atlantic Sturgeon (50 CFR Part 226.225) have highlighted a gap between the protectiveness of the current dissolved oxygen standard (24-hour mean concentration not less than 3.5 mg/L) in Zones 3, 4, and the upper portion of Zone 5 and the current ecological function of

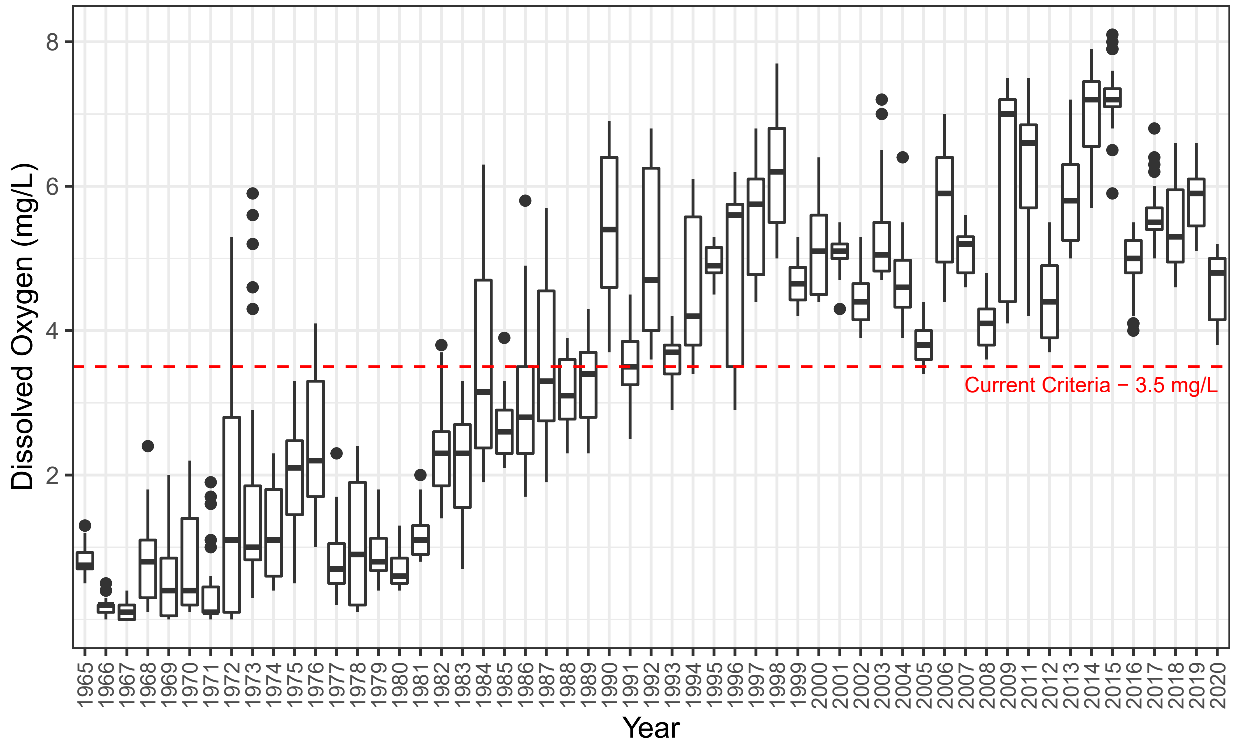

Figure 4.1.2 Delaware Estuary July daily mean dissolved oxygen concentrations by year at USGS at Ben Franklin Bridge, 1965 through 2020.

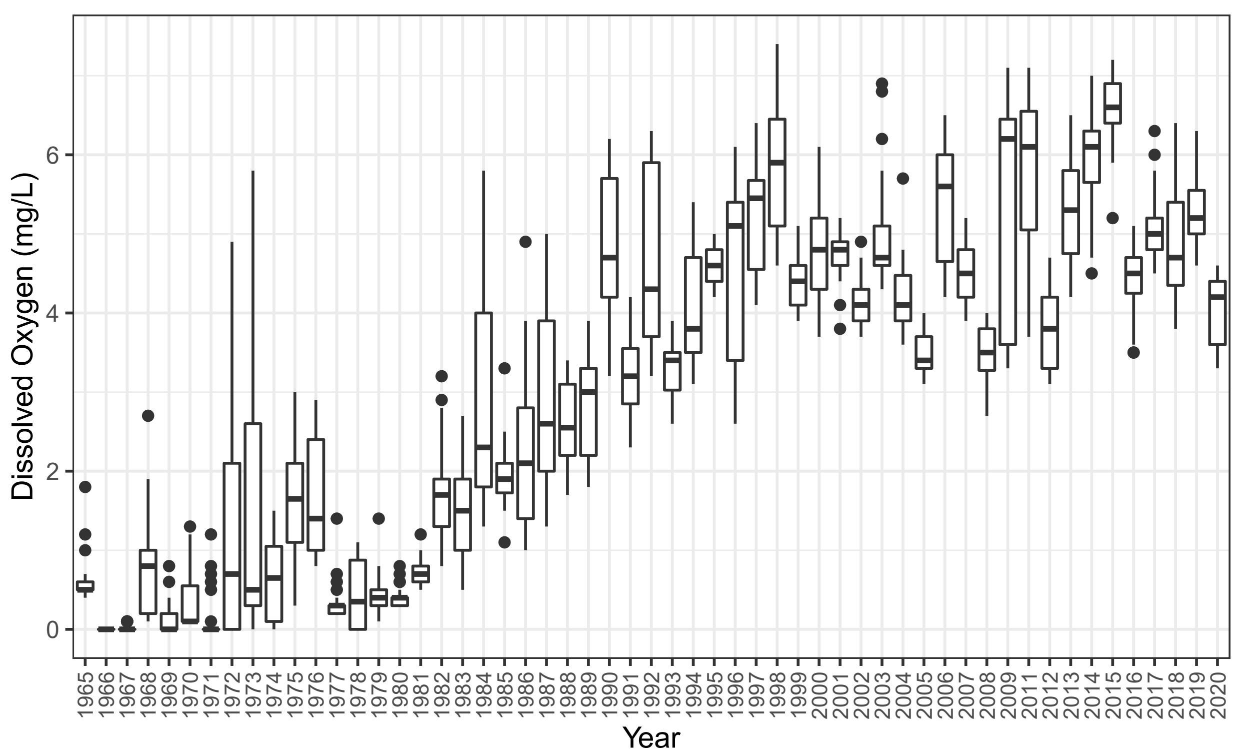

Figure 4.1.3 Delaware Estuary July daily minimum dissolved oxygen concentrations by year at USGS at Ben Franklin Bridge, 1965 through 2020.

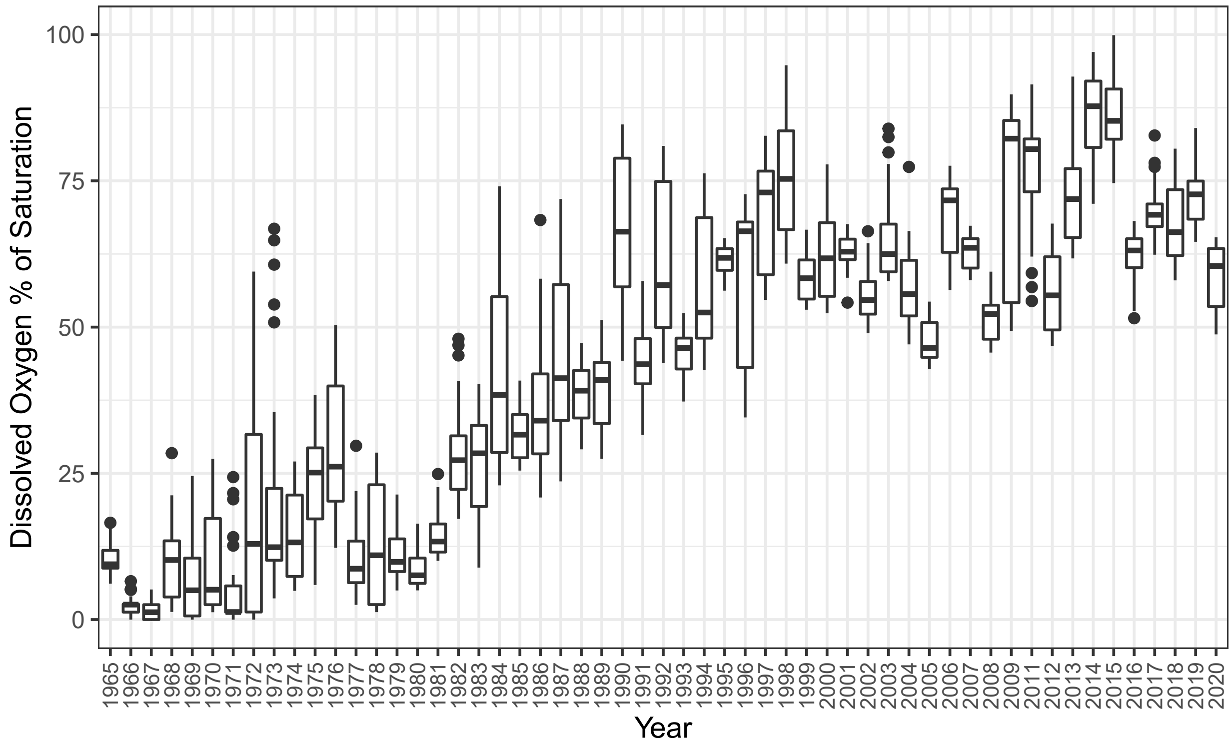

Figure 4.1.4 Delaware Estuary July mean daily dissolved oxygen percent of saturation by year at USGS at Ben Franklin Bridge, 1965 through 2020.

the Estuary. Achievement of higher DO concentrations will likely require tighter controls of the discharge of nutrients, especially ammonia. DRBC is currently in the process of developing a eutrophication model for the Delaware Estuary that will allow us to determine nutrient allocations needed to achieve higher dissolved oxygen concentrations.

Actions and Needs

Large amounts of dissolved oxygen data are collected in the Delaware Estuary each year. These practices will need to continue to provide data for the evaluation of any potential changes to aquatic life use designations in the Delaware Estuary. Additionally, high-resolution DO data in areas of importance to the endangered Atlantic sturgeon would be useful to aid conservation efforts for this species.

Summary

The long-term trend of DO in the Delaware Estuary shows remarkable improvement from near anoxic conditions in the 1960s and 1970s to nearly always above criteria today. In order to capture and retain the recoveries in fish spawning that have followed the recovery in DO, DRBC is seeking to determine the appropriate designated aquatic life uses of the Delaware River Estuary and the water quality criteria necessary to protect these uses.

4.1.1.2 Nutrients

Description of Indicator

The general category of "nutrients" is comprised of many different chemical compounds, including several species of nitrogen and phosphorus containing compounds. For this indicator, we considered specific chemical substances including nitrate, ammonia, and phosphate as being representative of nutrients. Nitrate and phosphate both have the advantage of being relatively quantifiable in the Estuary and having a long measurement record (decades).

The Delaware Estuary has both high loadings and high concentrations of nutrients relative to other estuaries in the United States (National Estuary Program Coastal Condition Report, 2006). The effects from these high nutrients are not well-understood but monitoring in the Estuary shows signs of suboptimal ecological health, including a persistent summer dissolved oxygen sag in the urban corridor of the Estuary. Nutrient loading to the Estuary is suspected to be a potential cause of either suboptimal ecological conditions or the dissolved oxygen sag. Therefore, high nutrient loading is one of the main candidates for understanding the Estuary's ecological health. Although nutrients are high, the worst eutrophication symptoms (such as anoxia, fish kills, and harmful algal blooms) are not currently seen in the Delaware Estuary.

Present Status

Phosphate measured as part of the DRBC Delaware Estuary Water Quality Monitoring Program shows highest concentrations near the most urbanized portion of the Estuary with lower concentrations near the head of tide and the mouth of the Bay as shown in Figure 4.1.5.

Ammonia and nitrate concentrations in the Estuary currently are typically less than 1 mg/L for ammonia and typically less than 3 mg/L for nitrate. Naturally-occuring levels of ammonia or nitrate in water are typically less than 1 mg/L. Highest concentrations are observed in the urbanized mid area of the Estuary, with somewhat lower concentrations near the head of tide (reflecting lower concentrations in the non-tidal river) and substantially lower concentrations at the mouth of the Bay, as shown in Figures 4.1.6 and 4.1.7 below. This pattern suggests loadings originating in the Estuary, especially in the urbanized area. As stated previously, although nutrient concentrations in the Delaware Estuary are high, hypoxia and harmful algal blooms are not observed. Monitoring for ammonia has been performed by the University of Delaware, and since 2009 by the DRBC Delaware Estuary Water Quality Monitoring Program (formerly the Boat Run), with funding from the USGS. Nitrate concentrations in particular, as in Figure 4.1.7 below, show structure suggesting higher loads in the urbanized portion of the Estuary with dilution and possible uptake in the Bay.

Past Trends

To assess trends, data from the DRBC Delaware Estuary Water Quality Monitoring Program (formerly called the Boat Run) were queried, from the late the mid 2000's through 2020.

Nitrate is quantifiable throughout the data record and is expected to be the most prevalent form of nitrogen in the Delaware Estuary, thus providing a good approximation of nitrogen trends over time. Since nitrate in the Estuary has a defined spatial structure (Fig 4.1.7), we selected measurements between river mile 65 and 195 as representative of the highest, uniform concentrations in the Estuary. Figure 4.1.8 below, depicting annual boxplots, demonstrates relatively consistent concentrations since the mid-2000's.

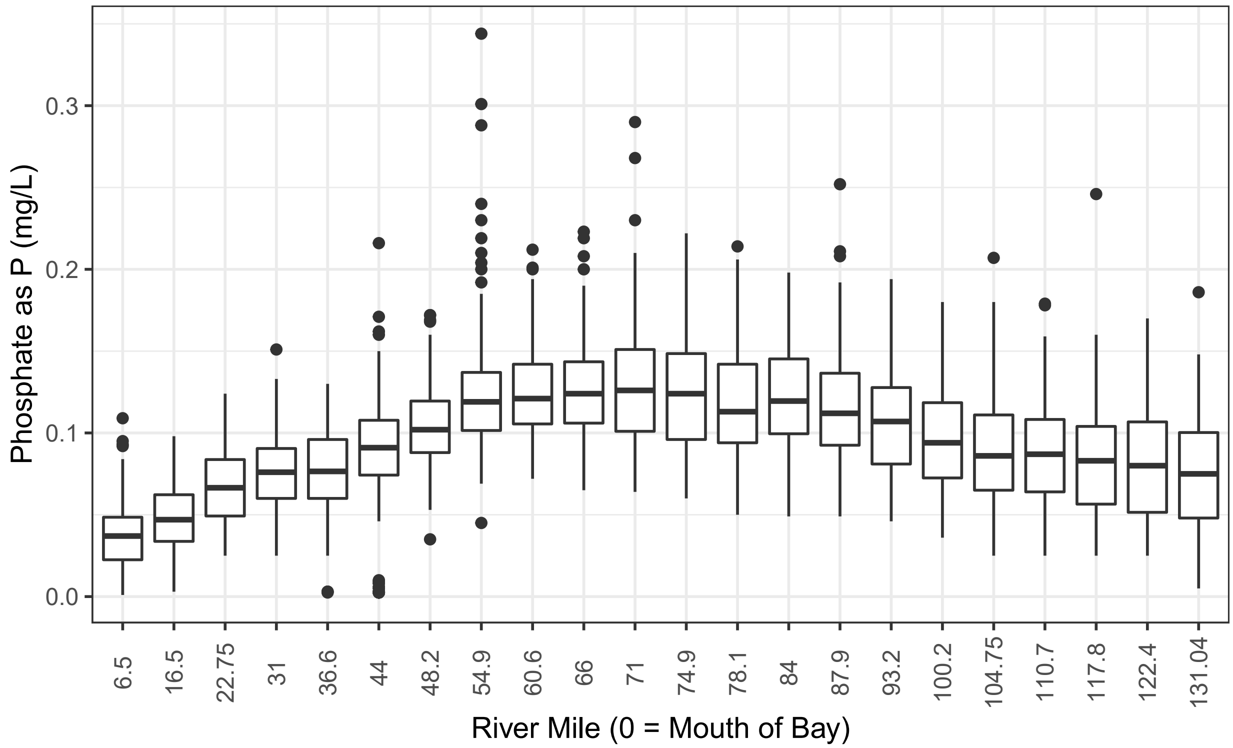

Figure 4.1.5 Phosphate by river mile in the Delaware Estuary, 2008 through 2020.

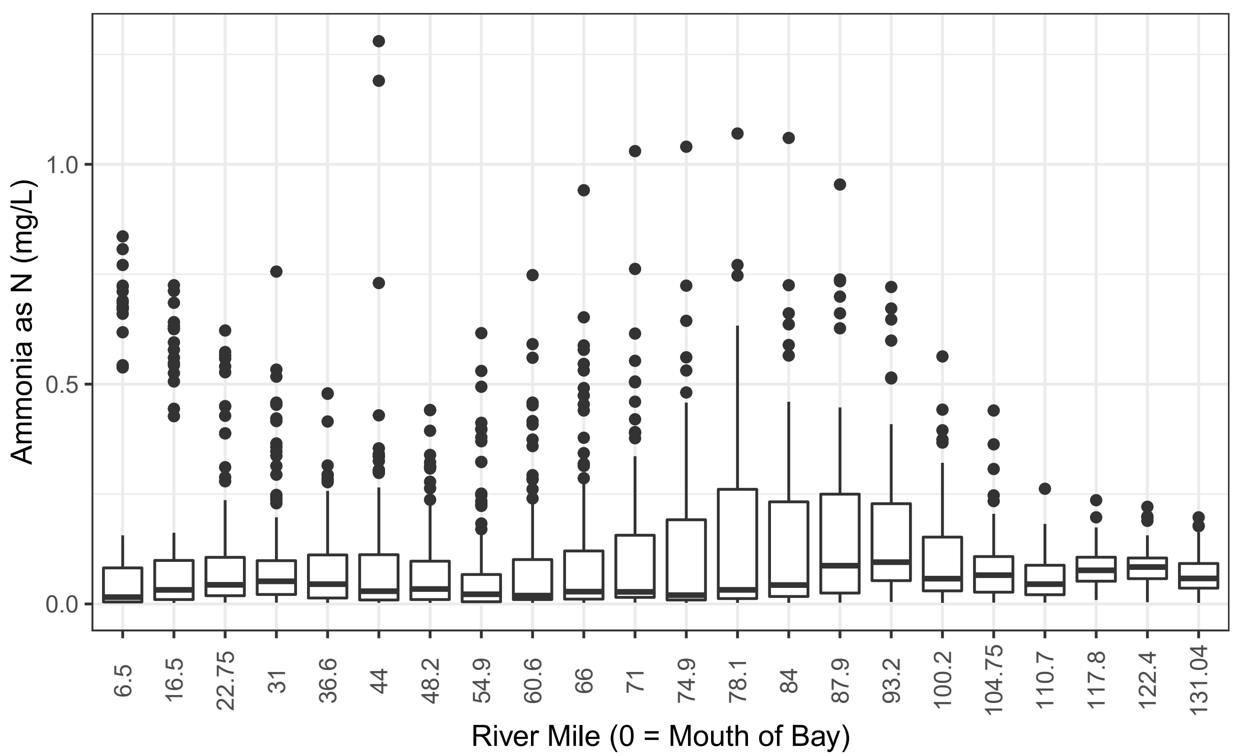

Figure 4.1.6 Ammonia by river mile in the Delaware Estuary, 2009 through 2020.

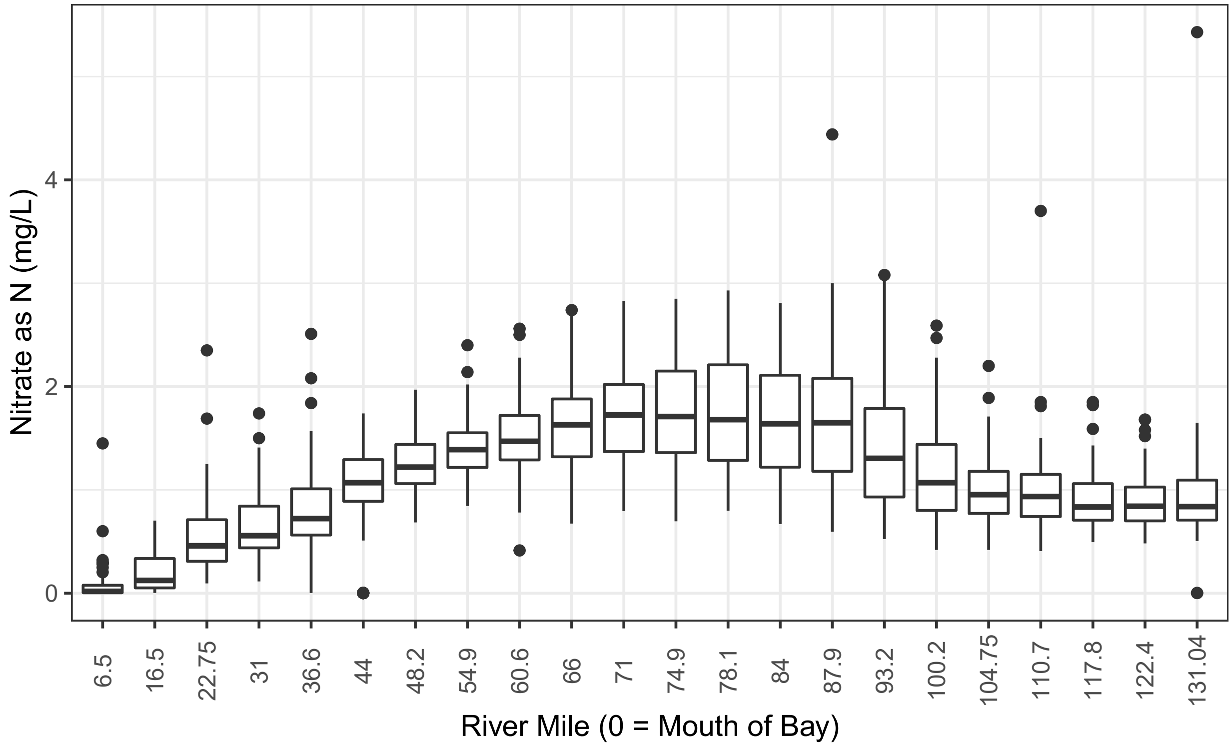

Figure 4.1.7 Nitrate by river mile in the Delaware Estuary, 2008 through 2020.

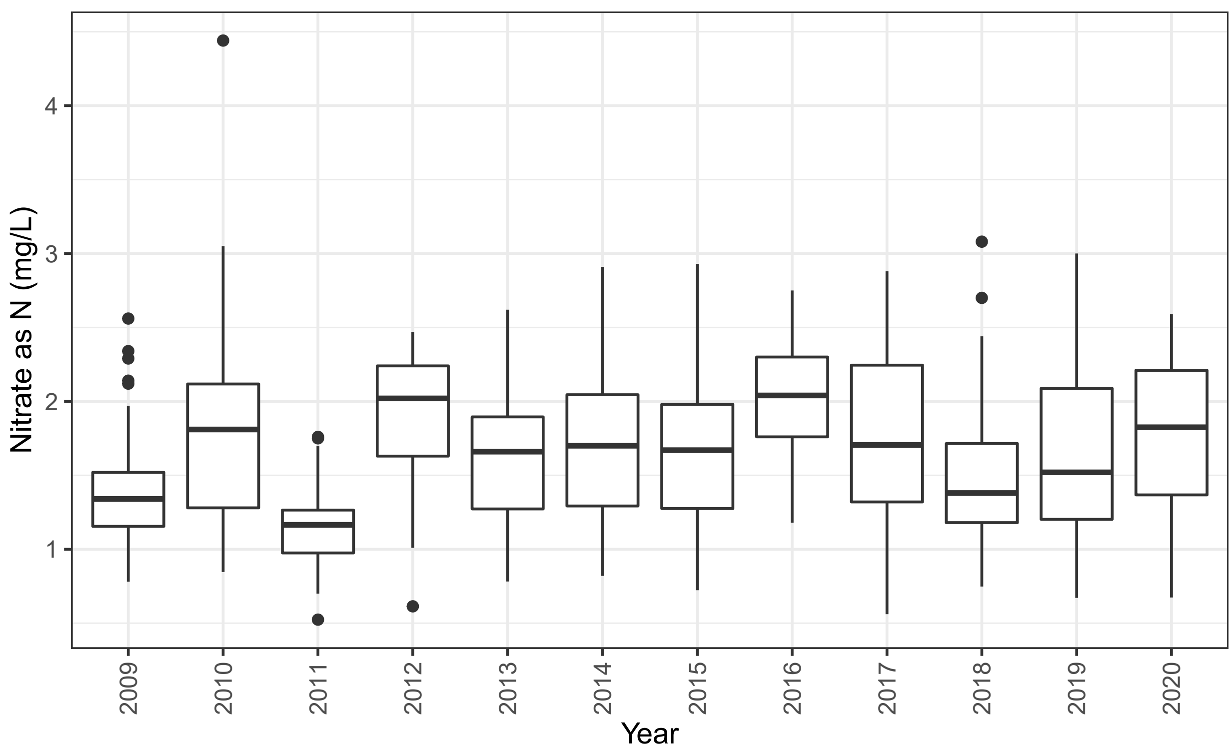

Figure 4.1.8 Annual nitrate between river mile 65 and 195, 2009 through 2020.

Since phosphate data are sparse and shows less spatial structure, we selected all Estuary phosphate measurements to generate the trend shown in Figure 4.1.9 below. Similar to nitrate, Figure 4.1.9 below demonstrates relatively consistent concentrations since the mid-2000's.

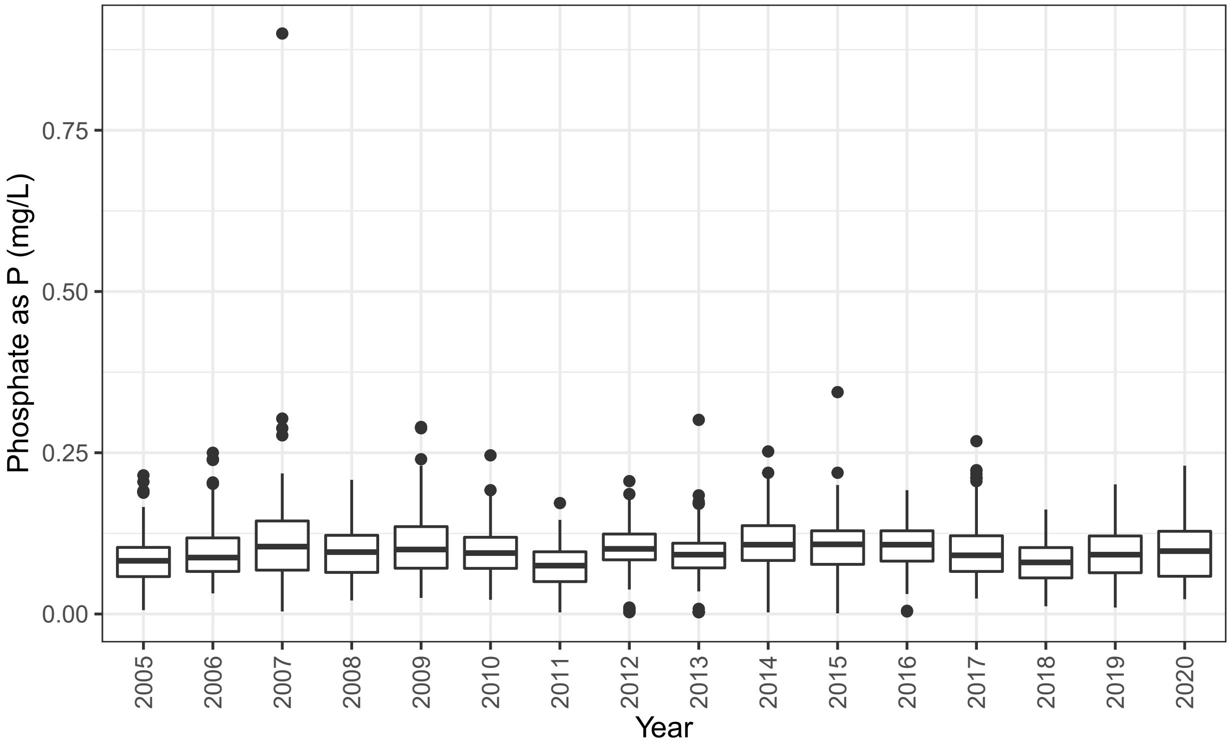

Figure 4.1.9 Historic phosphate in the Delaware Estuary from 1967 to 2016.

Future Predictions

As mentioned previously, documentation of fish propagation in the Estuary and proposals to designate the Estuary as essential fish habitat for Atlantic Sturgeon compel the identification and adoption of more protective dissolved oxygen criteria. Conceptually, achievement of more protective dissolved oxygen standards will likely need to be achieved through tighter effluent limits on nutrients, especially ammonia. DRBC is in the process of developing a eutrophication model for the Delaware Estuary. This model will allow DRBC to determine what level of dissolved oxygen is achievable and what reductions on nutrient discharges will be needed to achieve these limits.

Actions and Needs

DRBC is nearing completion of its Estuary Eutrophication Model. Coordination with partner agencies and stakeholders will be critical throughout the final stages of this project.

4.1.1.3 Salinity

The Delaware Estuary is believed to contain one of the largest freshwater tidal prisms in the world and provides drinking water for over one million people. However, salinity could greatly impact the Delaware's suitability as a source for drinking water, if salt water from the ocean encroaches on the region's drinking water intakes.

Description of Indicator

Salinity is usually estimated via direct measurement of other parameters, such as chloride or specific conductivity, with salinity operationally defined in terms of conductivity in standard references such as Standard Methods for the Examination of Water & Wastewater (APHA, AWWA, WEF 2005).

One important metric for understanding the importance of salinity concentrations in the Delaware Estuary is the location of the 250 mg/L chloride concentration based on drinking water quality standards originally established by the U.S. Public Health Service, also known as the "salt front."

The salt front location fluctuates along the tidal Delaware River as streamflow increases or decreases in response to precipitation, diluting or concentrating chlorides in the River. The seven-day average location of the salt front is used by the DRBC as an indicator of salinity intrusion in the Delaware Estuary. Generally, the salt front is located between river miles 70-75. DRBC's Drought Operating Plan focuses on controlling the upstream migration of salty water from the Delaware Bay during low-flow conditions in Basin rivers and streams. As higher salinity water moves upstream, it may increase corrosion of surface water users' infrastructure, particularly industry, and increase sodium concentration in treated drinking water, which is a health concern for sensitive customers. Zone 2 (as defined in the DRBC Water Code) in the Delaware Estuary contains large Pennsylvania and New Jersey drinking water intakes at river mile 110. As a result and water quality objectives include a maximum 15-day average concentration of 50 mg/L chloride. Salinity repulsion policies governing upstream reservoir releases work to repel the salt front and maintain chloride concentrations below the water quality objective in Zone 2.

Water releases from five reservoirs are used to help dilute the higher salinity water during low streamflow conditions. Three reservoirs —Pepacton, Neversink and Cannonsville—are owned by New York City and are located in the Delaware River's headwaters in the Catskill Mountains in New York State. When full, these three reservoirs hold 271 billion gallons of water in total. Two additional reservoirs—Blue Marsh and Beltzville—are located in Pennsylvania along the Schuylkill River in Berks County and the Lehigh River in Carbon County, respectively. These two lower basin reservoirs hold nearly 20 billion gallons of water in total, when full. For more information on salt front management see Section 5 of the Water Quality Chapter.

Present Status

Figure 4.1.10 shows the chloride concentrations from the DRBC Delaware Estuary Water Quality Monitoring Program (formerly called the Boat Run). A sharp transition between river miles 75 and 78 (near Marcus Hook) is evident.

Past Trends

To determine whether the recent trends are evident in the DRBC data, we plotted boxplots by year from river mile 75 (at the change in spatial structure) and river mile 105 (nearest to major drinking water intakes). The 2000 to 2020 data at river mile 75 (Fig 4.1.11) shows high variability from year to year, but no obvious trend. The same plot for river mile 105 (Fig 4.1.12) suggests some slight elevation from 2015 through 2019, however it is difficult to discern if this is a trend.

The best means of assessing long term historical salinity trends in the estuary is by looking at the long-term continuous specific conductivity results collected by the USGS at the Ben Franklin Bridge, Chester, and Reedy Island. At each of those locations, data are available beginning in 1964. Figures 4.1.13, 4.1.14, and 4.1.15 below suggest that the drought of record in the 1960s strongly influences the oldest data bin. All plots indicate lower conductivity values than the drought of record and year-to-year variability

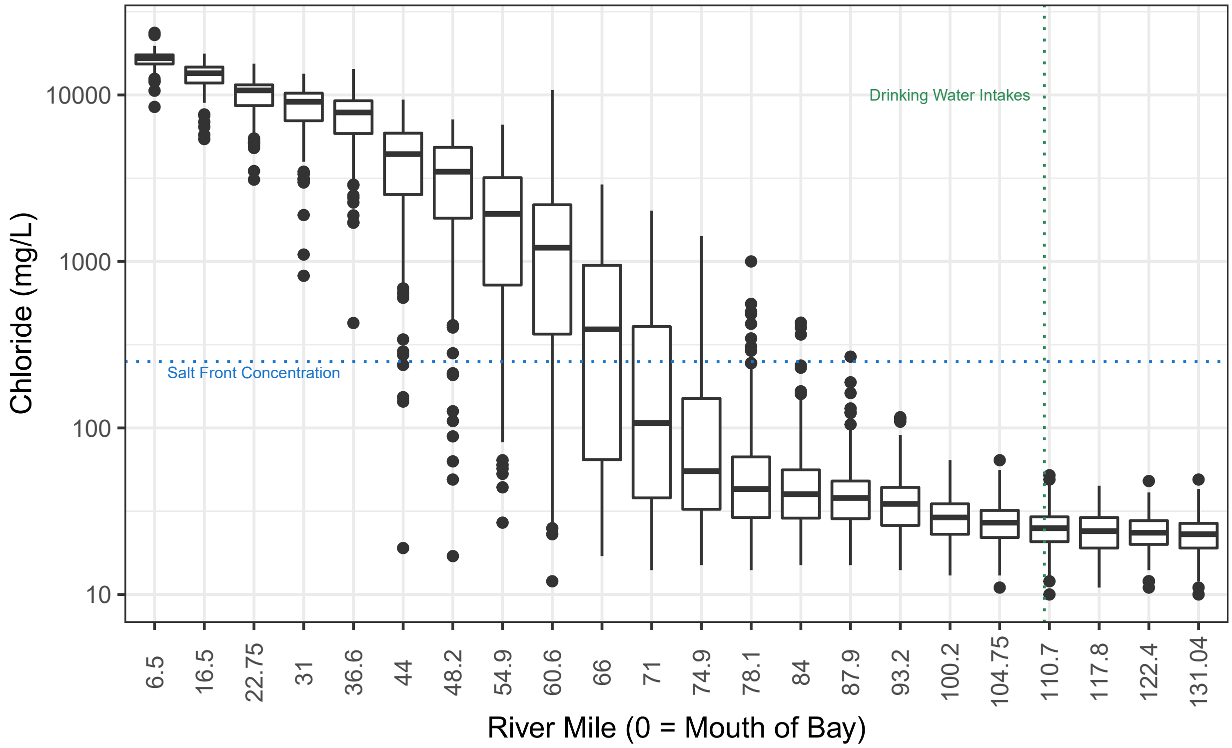

Figure 4.1.10 Chloride concentration ranges by river mile in the Delaware Estuary, 2005 through 2020.

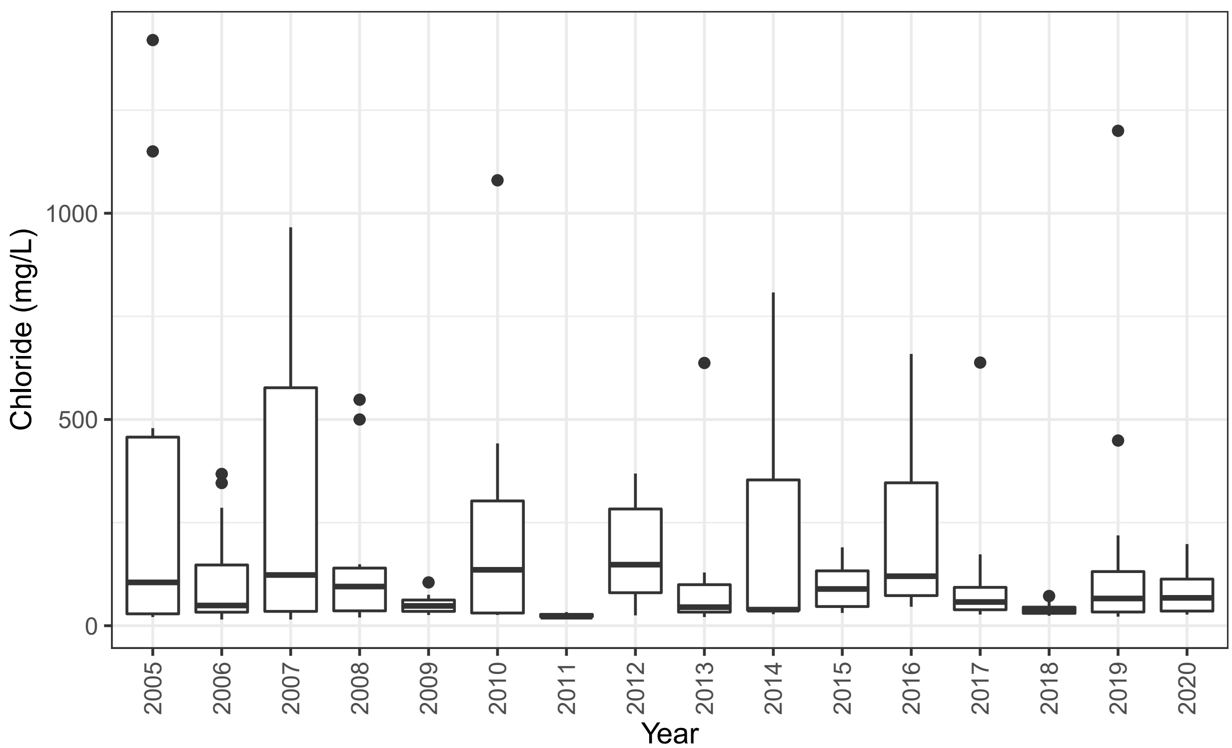

Figure 4.1.11 Recent chloride concentrations by year at river mile 75, 2005 through 2020.

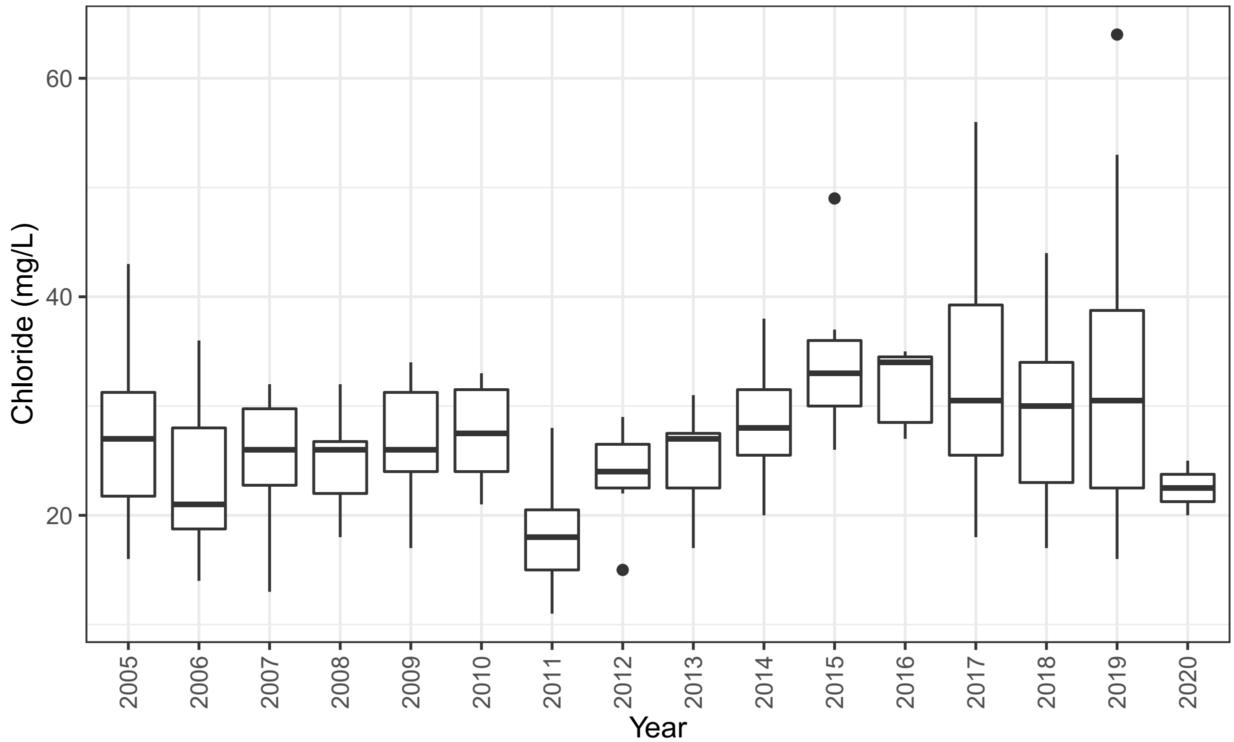

Figure 4.1.12 Recent chloride trends by year at river mile 105, 2005 through 2020.

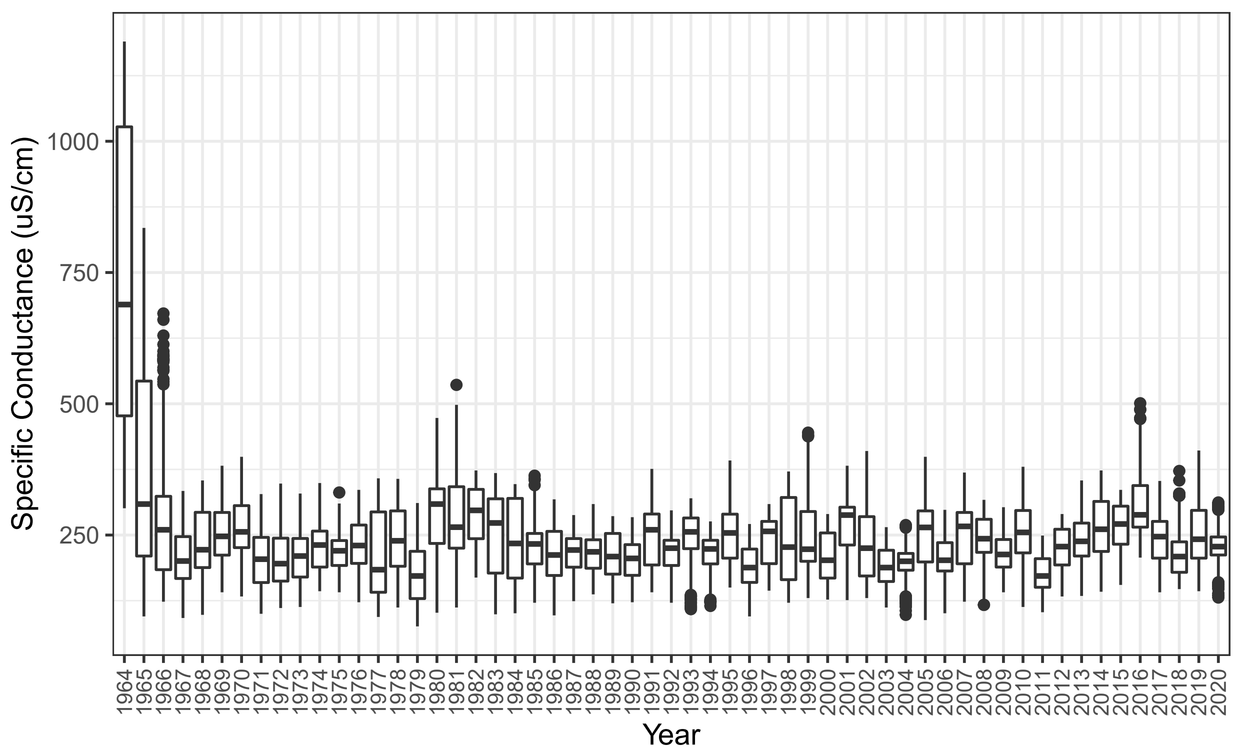

Figure 4.1.13 Long-term specific conductivity box and whisker plots at USGS 01467200, Ben Franklin Bridge.

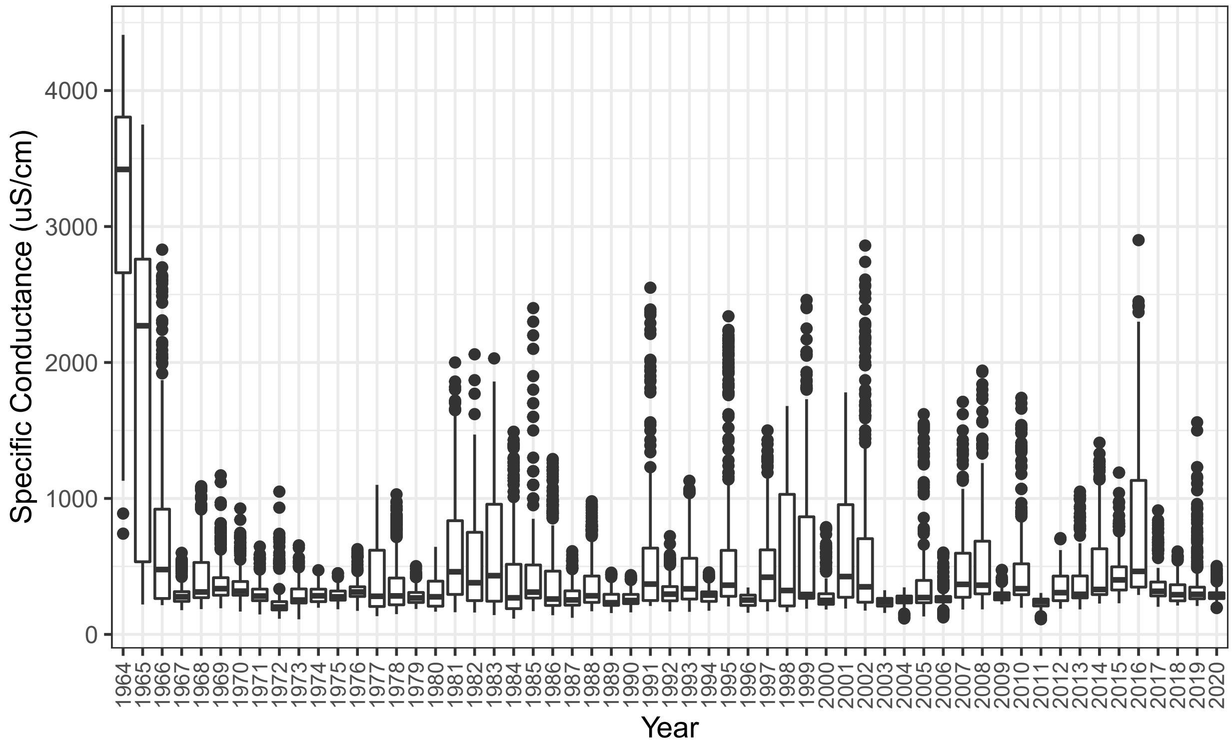

Figure 4.1.14 Long-term specific conductivity box and whisker plots at USGS 01477050, Chester.

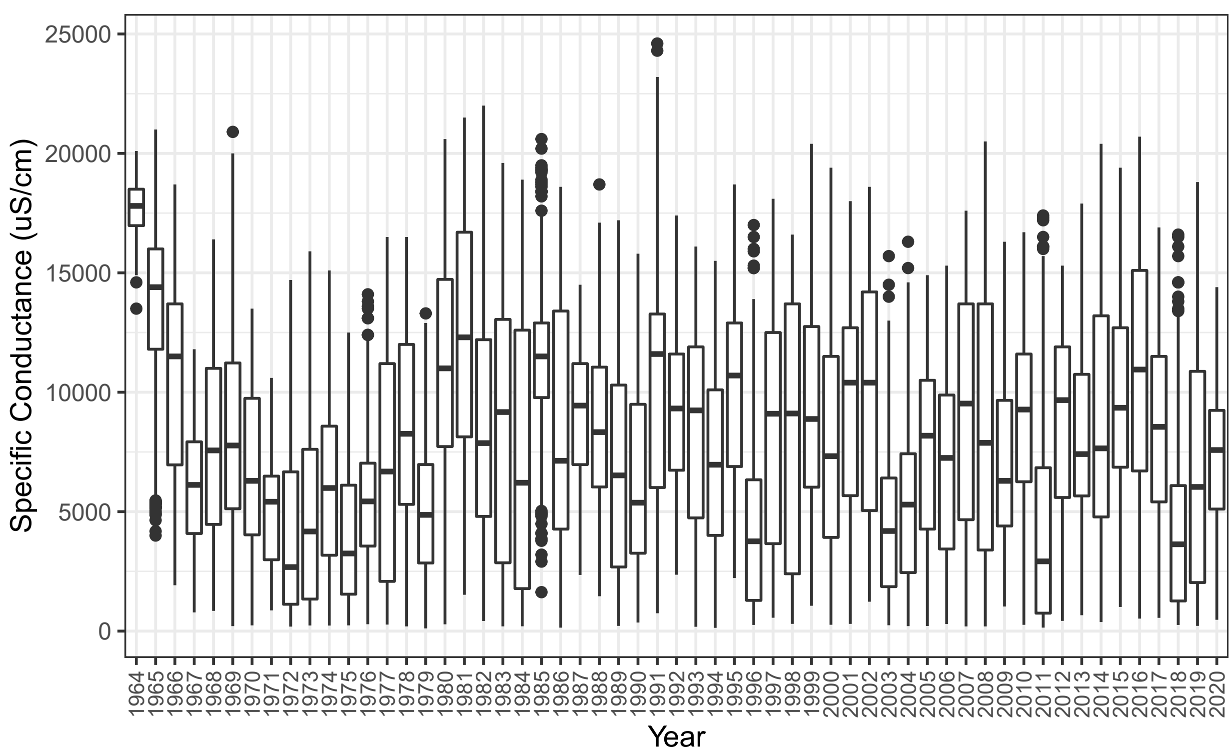

Figure 4.1.15 Long-term specific conductivity box and whisker plots at USGS 1482800, Reedy Island.

(especially at Reedy Island). Ben Franklin and Chester both demonstrate distributions over the last year or two that are within the variability displayed over the last decade.

Future Predictions

Sea level rise associated with global climate change is expected to change the salinity regime of the Delaware Estuary. A model report prepared by the U.S. Army Engineer Research and Development Center (Kim and Johnson, 2007) shows predicted mean increases in salinity between 1996 and 2040 of 14% at Delaware Memorial Bridge, 16% at Chester, PA, and 10% at the Ben Franklin Bridge from sea level rise alone. When combined with other likely drivers, such as channel deepening and changes in consumptive water use over that same period, the forecasted increases in salinity are approximately 22%, 29%, and 18% at the Delaware Memorial Bridge, Chester, and the Ben Franklin Bridge, respectively.

Actions and Needs

Predictive modeling to establish the linkage between sea level and resultant salinity is needed to assess the expected future salinity spatial regimes. Some level of modeling has been completed and used for this purpose, but longer term forecasts under a wider range of conditions are needed to identify critical conditions and begin to evaluate solutions.

Summary

Estuary salinity patterns impact the availability of drinking water and the spatial domains of aquatic living resources. Definitive trends in historic data are not evident from relatively simple assessment tools. Given the importance of the salt front, more refined predictive tools allowing longer term forecasts are needed.

4.1.1.4 pH

Potential of hydrogen (pH) is the mathematical notation for the negative log of the hydrogen ion concentration (-log[H+]) and indicates an acid, neutral, or base condition.

Description of Indicator

The pH of surface waters can be an important indicator of ecological function and productivity, and pH impacts the bioavailability and toxicity of pollutants such as metals and ammonia. Currently, DRBC's pH criteria for the Estuary is between 6.5 and 8.5.

Present Status

Figure 4.1.16 shows the box and whisker plots of discrete pH values measured at each of the Estuary USGS continuous monitoring stations, compared to the minimum and maximum pH criteria in DRBC's water quality standards. Although the distributions differ by location, all values are within the DRBC criteria.

Past Trends

To assess temporal changes in pH, we developed box and whisker plots of pH by year. Results continue to demonstrate an increase in pH over the period of record at Ben Franklin (Fig 4.1.17) and an even more pronounced increase at Chester (Fig 4.1.18).

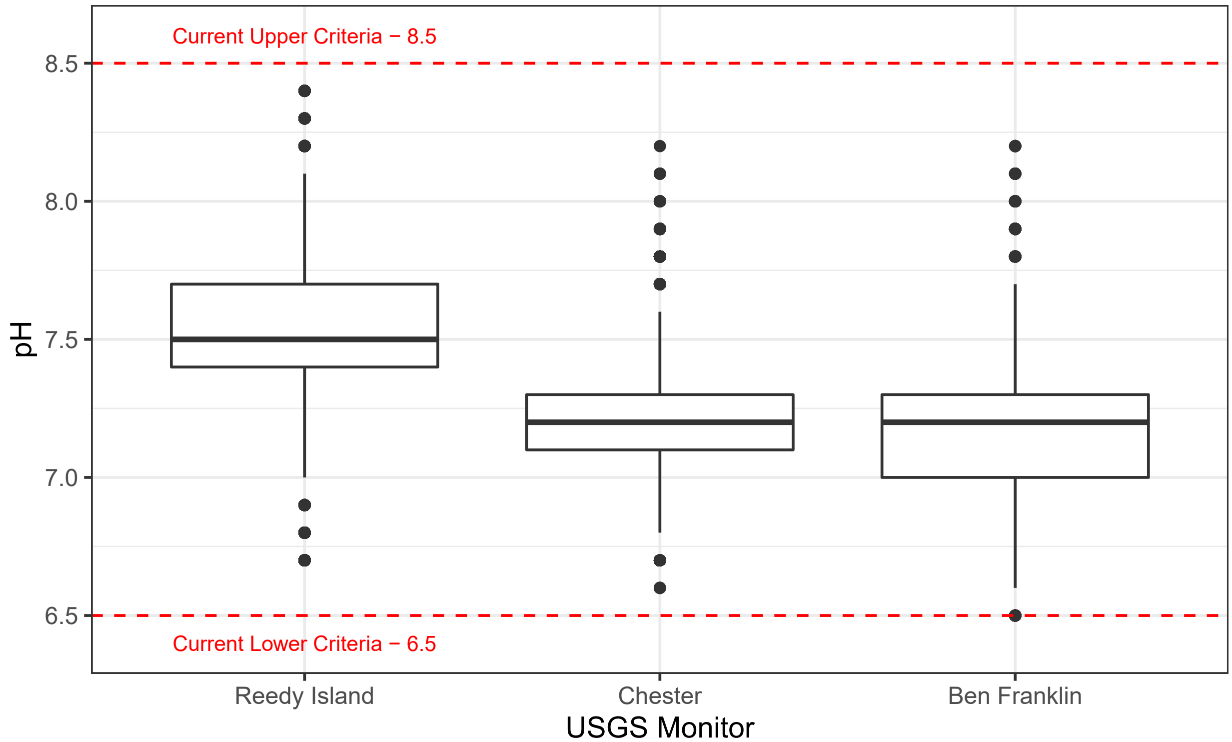

Figure 4.1.16 pH measurements at 3 USGS Delaware Estuary monitors, 2011 through 2020.

This phenomenon was noted in the previous TREB report (2017) and is likely linked to the gross pollution historically found in the urban corridor of the Delaware Estuary and the remarkable progress at eliminating some of this pollution over the past 40 years. Because human and industrial wastes received little or no treatment through the 1960s and 1970s, the carbonaceous and nitrogenous compounds in these wastes were used as food sources for microbes in the Estuary, which in turn used up the available dissolved oxygen and created an oxygen block around Philadelphia. In addition to using the oxygen, the waste products from this microbial respiration included carbon dioxide and additional hydrogen ions (acids) which historically caused depression of pH that closely mirrored the sag in dissolved oxygen (Culberson 1988). The improved treatment of both municipal and industrial waste over the past 40 years has therefore been linked to both improvements in dissolved oxygen and pH for the Delaware Estuary, with stronger trends at both the Ben Franklin Bridge and Chester. In addition, this same period has seen the cessation of highly acidic industrial waste inputs to the Delaware Estuary, which may have also contributed to these temporal observations.

Future Predictions

NOAA and others have documented the occurrence of ocean acidification. In the absence of other reactions, we might expect the pH to decrease at the ocean boundary, with a corresponding decrease in pH propagated from the ocean into the Estuary. The more complex dynamic of the Estuary, however, suggests that pH levels may be increasing. Further improvements to waste treatment in the urban corridor could lead to further improvements in pH for those freshwater zones of the Estuary. Thus with the processes driving pH in both directions, it is impossible to predict if pH values will continue to rise, level off, or if ocean acidification will pass a tipping point causing pH trends to reverse toward a more acidic estuary.

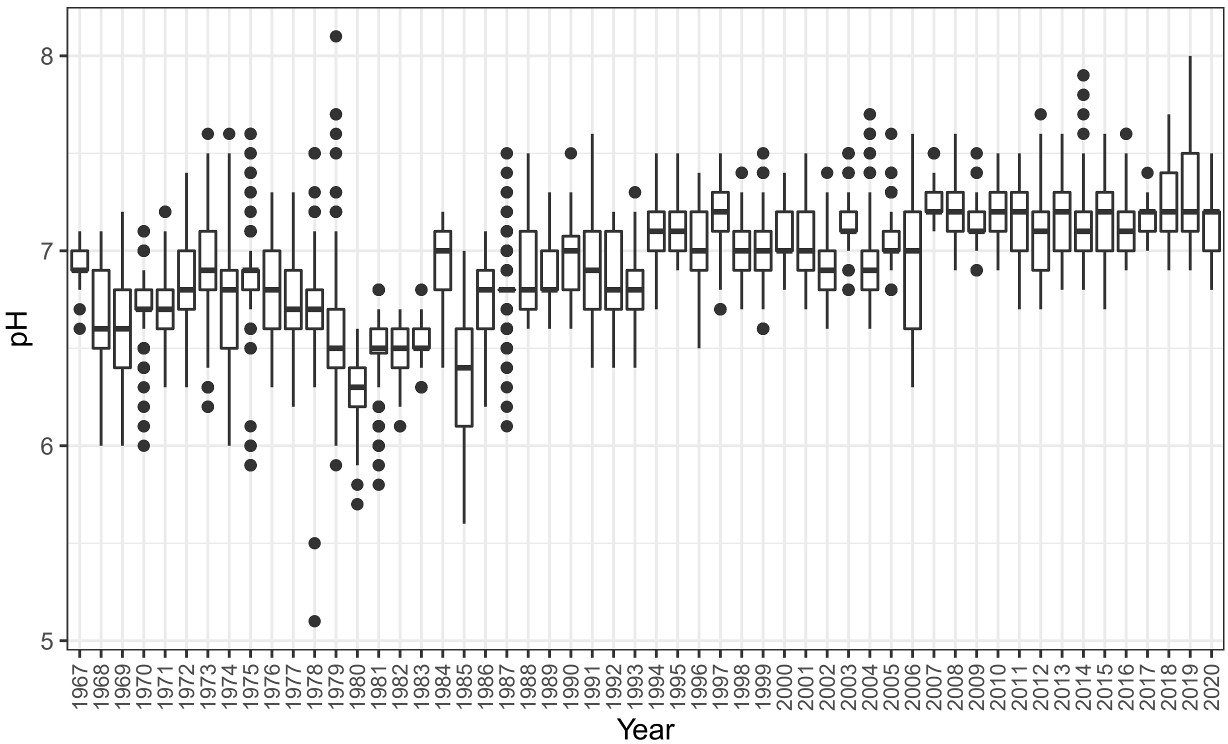

Figure 4.1.17 pH box and whisker plot by year at USGS 01467200, Ben Franklin Bridge, 1967 through 2020.

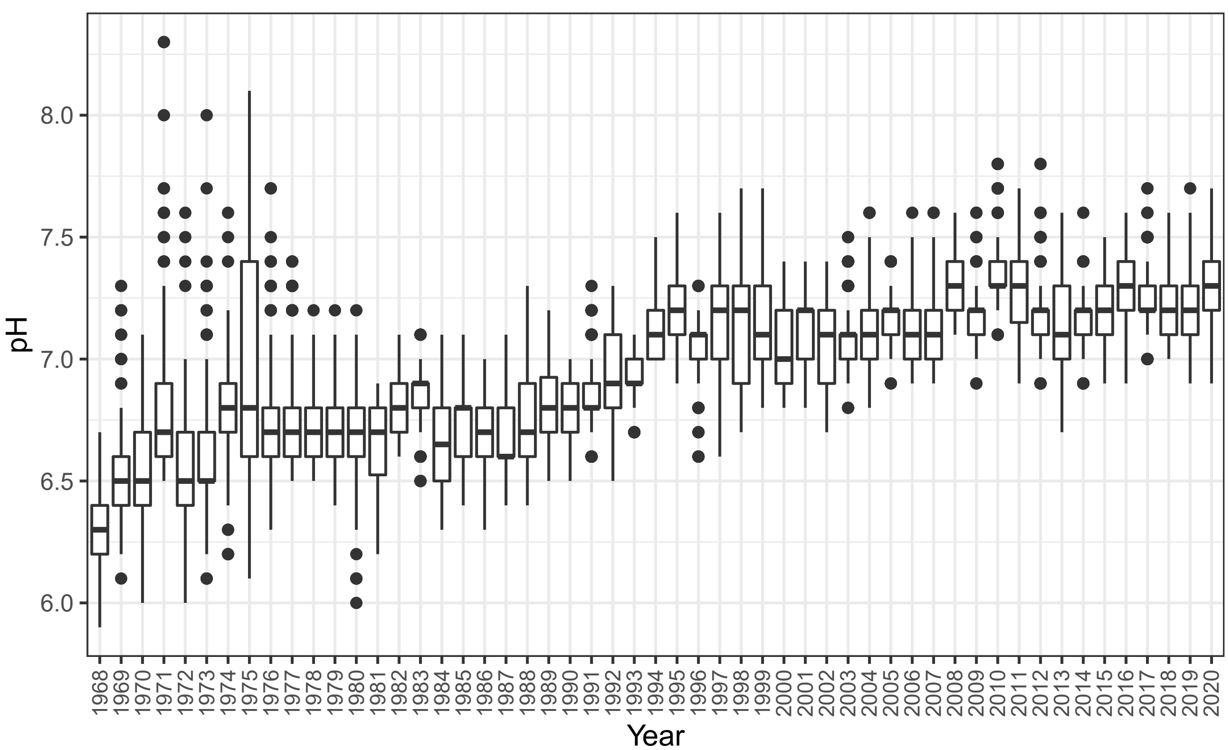

Figure 4.1.18 pH box and whisker plot by Year at USGS 01477050, Chester, 1968 through 2020.

Actions and Needs

A better understanding of the Estuary carbon cycle and its impact on pH is needed. Models that can integrate the countervailing processes of ocean acidification and decreased microbial respiration could help elucidate the short and long-term likelihoods of continued changes in pH and carbon availability.

Summary

Further improvements to waste treatment in the urban corridor could lead to further improvements in pH for those freshwater zones of the Estuary. Thus with the processes driving pH in both directions, it is impossible to predict if pH values will continue to rise, level off, or if ocean acidification will pass a tipping point causing pH trends to reverse toward a more acidic Estuary.

4.1.1.5 Temperature

Description of Indicator

Water temperature is an important factor for the health and survival of native fish and aquatic communities. Temperature can affect embryonic development; juvenile growth; adult migration; competition with nonnative species; and the relative risk and severity of disease. Estuary Temperature Criteria are expressed in DRBC regulations by day of year.

Near real-time assessment of temperature criteria in the Delaware Estuary is provided on DRBC's water quality dashboard, comparing measurements from USGS and NOAA ports monitors to day-of-year temperature criteria.

Present Status

Maximum daily water temperatures recorded at USGS continuous monitors at Ben Franklin Bridge and Chester from 2011 to 2020 were compared to DRBC's zone specific day-of-year temperature criteria (Fig 4.1.19). Although most observations were below (meeting) criteria, some exceedances were evident.

Determination of the importance of these criteria exceedances is confounded by the strong role played by atmospheric conditions. Work performed for the 2008 Integrated Assessment suggested that estuary water temperatures were strongly influenced by air temperatures and cloud cover. Brief periods of water temperatures elevated above criteria can have stressful impacts upon aquatic life species, delaying or interrupting spawning, feeding, and development of young. Extremely high temperatures or extended periods above criteria can result in death or detrimental avoidance behavior.

Past Trends

In the context of global climate change, we want to determine whether water temperatures have shifted perceptibly during the period of observational record. We assessed daily mean water temperatures from the USGS monitors at the Ben Franklin Bridge (since 1964), Chester (since 1965), and Reedy Island (since 1970). Minimum and maximum daily temperature records extend back slightly further.

For the entire period of record through 2016 for each of the 3 monitors, the median of the mean daily temperature for each day of the year was determined. For example, the daily mean temperature was examined for each May 15th, for every year from the 1960s or 1970, and determined the median of that set. DRBC then compared each May 15th temperature to the median of all May 15th temperatures at that location, to see if the differences changed over time. Figure 4.1.20 shows the mean daily temperature

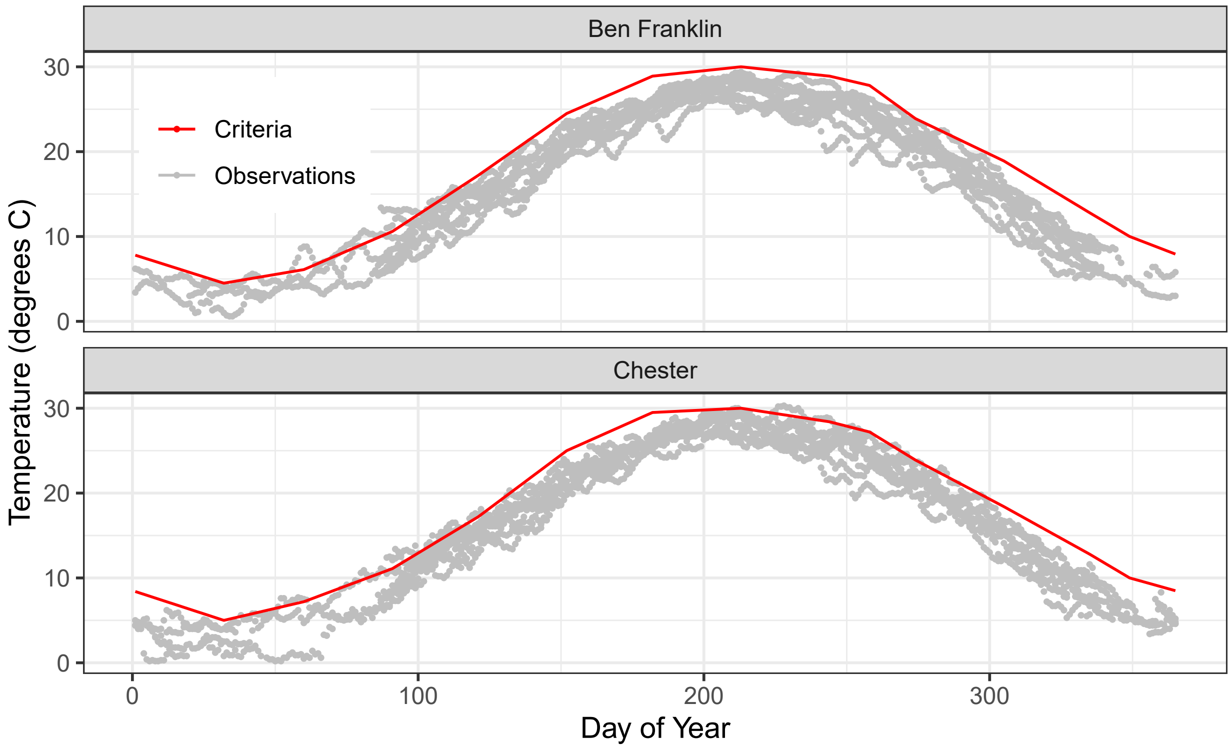

Figure 4.1.19 Temperature observations compared to DRBC day of year criteria, at Ben Franklin and Chester, 2011 through 2020.

measurements by day of year, and the median for each day of year for the USGS continuous monitor at Chester.

As in the previous TREB, portions of the yearly cycle were examined where broad day-to-day shifts were minimized (summer and winter). Figures 4.1.21 and 4.1.22 show the residuals (mean daily water temperature – median water temperature for that day of year) for the Ben Franklin Bridge during the summer and Reedy Island during the summer (where the strongest indication for any trend was evident). Consistent with the prior TREB, this analysis suggested a slightly decreasing summer temperature trend at the Ben Franklin Bridge station, but an increasing summer temperature trend at the Reedy Island station. As suggested in the previous TREB, these apparently opposite trends could be reflecting different sets of drivers. It seems reasonable to conclude that the Reedy Island increasing temperature trend is reflective of documented climate change, while the Ben Franklin Bridge station could be reflecting reductions in industrial thermal loads in the urbanized portion of the Estuary over that same time period.

Future Predictions

In its 2008 report, the Union of Concerned Scientists used output from global circulation models to predict that the climate in Pennsylvania would shift toward a climate more similar to Georgia over the next 60 years. Intuitively, this seems to suggest that water temperatures will increase in that same time period. Some temperature drivers, such as sea level rise and shifts in industry and landscape, may impose counter-acting forces which cannot be easily estimated.

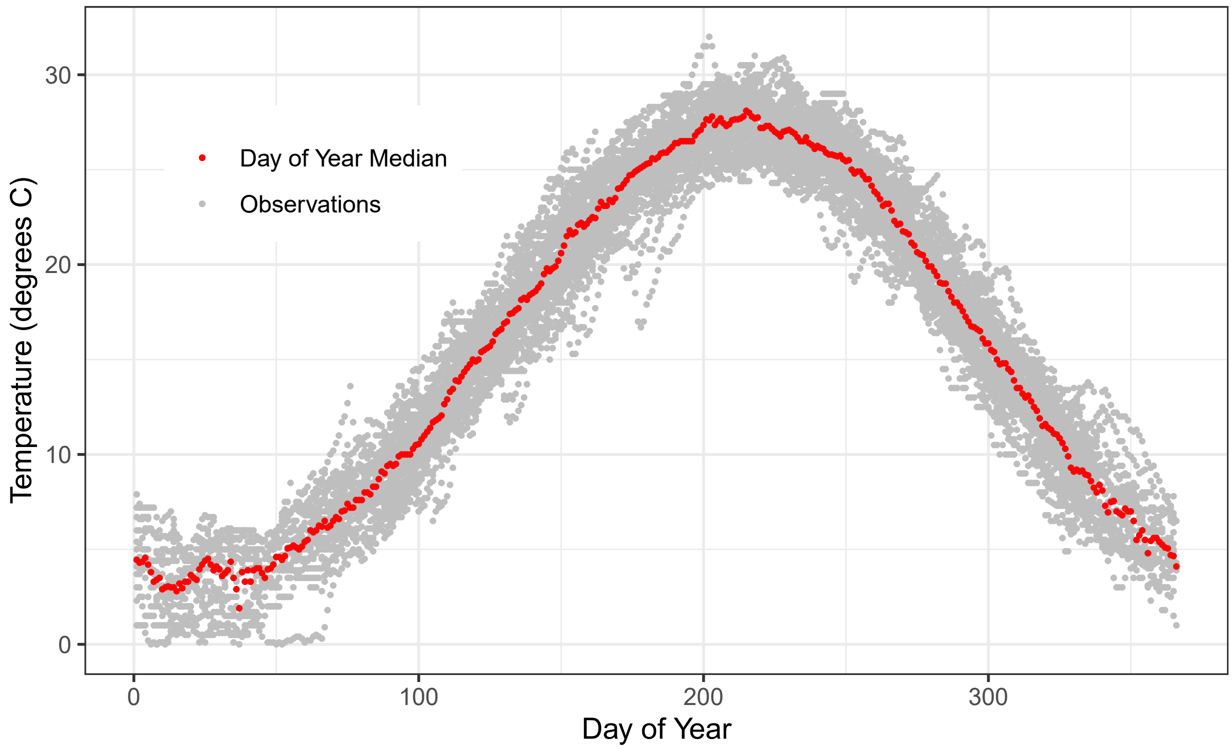

Figure 4.1.20 Period of record temperature observations including median by day of year at Chester, 1964 through 2020.

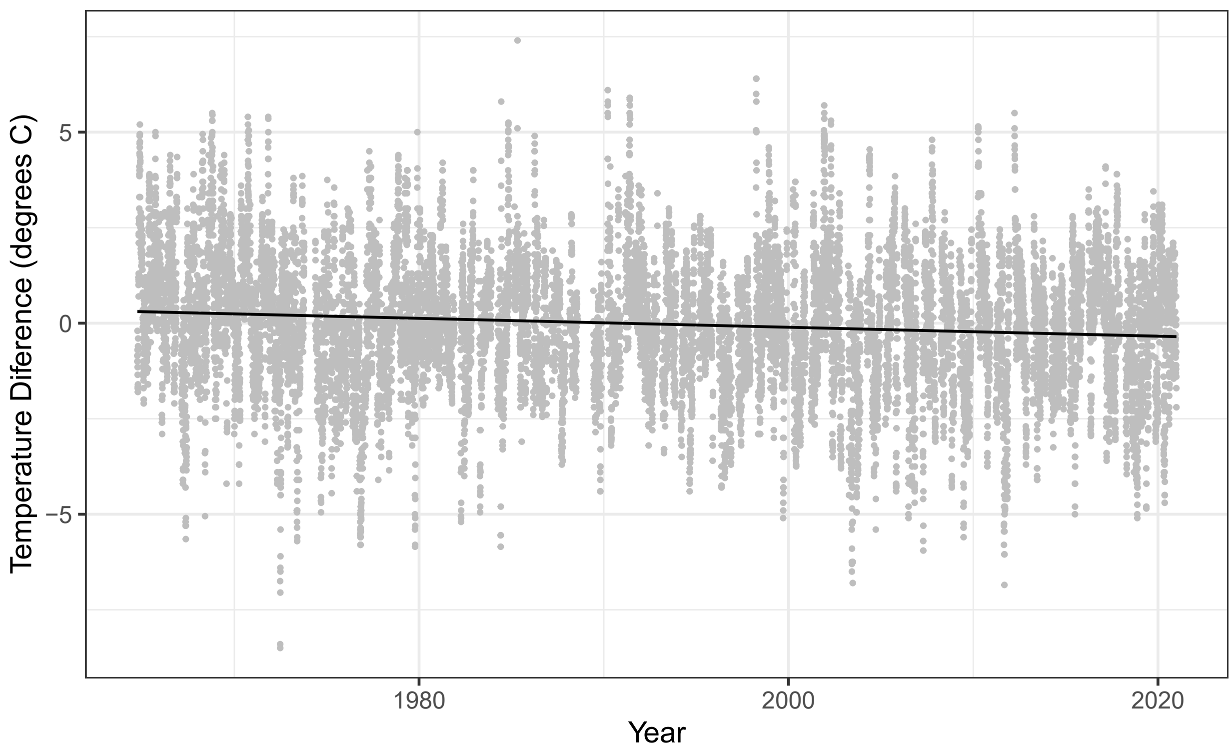

Figure 4.1.21 Delaware River summer residuals at USGS 01467200, Ben Franklin Bridge. A linear trend line was added for illustrative purposes.

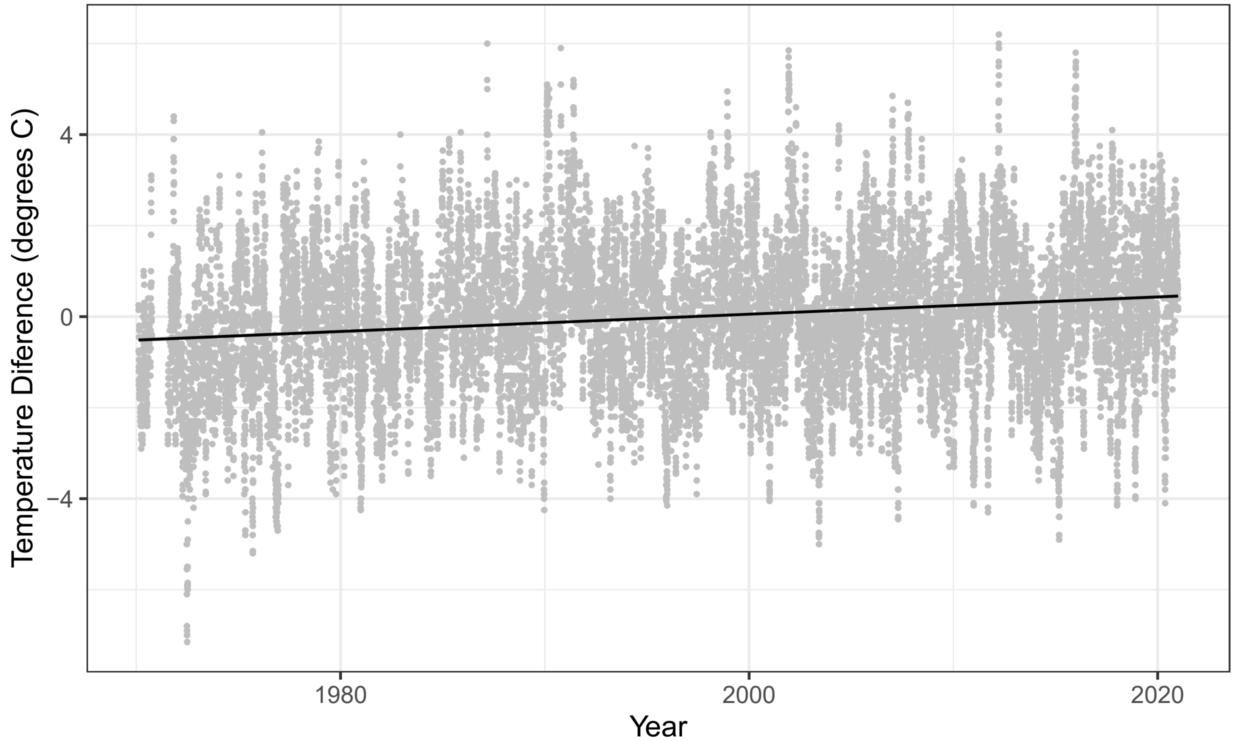

Figure 4.1.22 Delaware River summer residuals at USGS 01482800, Reedy Island Jetty. A linear trend line was added for illustrative purposes.

Actions and Needs

In order to gain a firmer understanding of how different temperature drivers are influencing the Delaware Estuary, and ultimately to understand how global climate change may be manifested, a more rigorous evaluation is needed. This evaluation may need to include a temperature model that integrates the various drivers.

Summary

Delaware Estuary water temperatures are influenced by multiple drivers including meteorological forces, terrestrial and ocean water inputs, and municipal and industrial thermal loads. A review of the current status shows that 90% or more of daily observations are meeting temperature criteria. An analysis of historic trends suggests that the overlapping temperature drivers make it difficult to understand how water temperatures have changed over the last five decades. A more rigorous assessment, which explicitly accounts for overlapping temperature drivers, is desirable.

4.1.2 Non-Tidal – Chemistry and Nutrients

4.1.2.1 Dissolved Oxygen

Dissolved oxygen (DO) refers to the concentration of oxygen gas incorporated in water. Oxygen enters water both by direct absorption from the atmosphere, which is enhanced by turbulence, and as a by-product of photosynthesis from algae and aquatic plants. Sufficient DO is essential to growth and reproduction of aerobic aquatic life. Oxygen levels in water bodies can be depressed by the discharge of oxygen-depleting materials (measured in aggregate as biochemical oxygen demand, BOD, from wastewater treatment facilities), from the decomposition of organic matter including algae generated during nutrient-induced blooms, and from the oxidation of ammonia and other nitrogen-based compounds.

Description of Indicator

For our review of oxygen values in the Basin, we looked at two different expressions of DO: concentration (i.e., as mg/L), and percent of saturation. DO concentration provides a direct comparison to water quality criteria and to aquatic life affects levels. Percent of saturation gives an indication of the oxygen content relative to saturation due to temperature and salinity.

Present Status

We queried the National Water Quality Data Portal for all summer measurements of DO in the Delaware River Basin (tidal and non-tidal) from 2011 through 2020 and plotted their location and concentration (Fig 4.1.23). This mapping shows the availability of spot measurements and the concentration. Because DO concentrations are typically characterized by a daily peak in late afternoon and a pre-dawn daily low due to photosynthetic processes, continuous monitors are preferable to daytime spot measurements, which miss the daily low concentrations. In addition, continuous monitors provide a depth and continuity of data that could not be replicated with spot measurements. USGS continuous monitors provide a more complete DO distribution, but at fewer locations. We compared box and whisker plots of summer DO from USGS monitors at the Brandywine Creek at Chadds Ford, PA, the Christina River at Newport, DE, the Delaware River at Trenton, NJ, the Lehigh River at Glendon, PA, and the Schuylkill River at Norristown, PA (Fig 4.1.24). Although the distributions are different at the different locations, the majority of values are above 5 mg/L (the threshold between fair and poor health identified in the previous TREB).

Past Trends

Extended time series data sets are less plentiful in the non-tidal Basin than they are in the Estuary. However, the Delaware River at Trenton has been monitored with a continuous water quality monitor by USGS since 1962. We visualized past trends at this location with annual box plots of daily average DO saturation (Fig 4.1.25). This analysis suggests minor inter-year variation with no discernible trends. DO saturation at this location is often > 100%. Unlike estuary stations, DO at the Delaware at Trenton is strongly influenced by re-aeration due to its wide, shallow, high gradient reaches and photosynthesis from attached algae.

Future Predictions

Non-tidal DO appears to be relatively stable. Regulatory programs, such as the DRBC's Special Protection Waters regulations are designed to preserve water quality. If potential DO problems may exist (such as in Frankford Creek), long term efforts to minimize combined sewer overflows (CSO) are likely to reduce the frequency and magnitude of exceedances over time.

Actions and Needs

Continued monitoring and enhancement of monitoring networks, especially in the realm of continuous real time monitors, will help ensure preservation of water quality and identify reaches where DO is less than optimal.

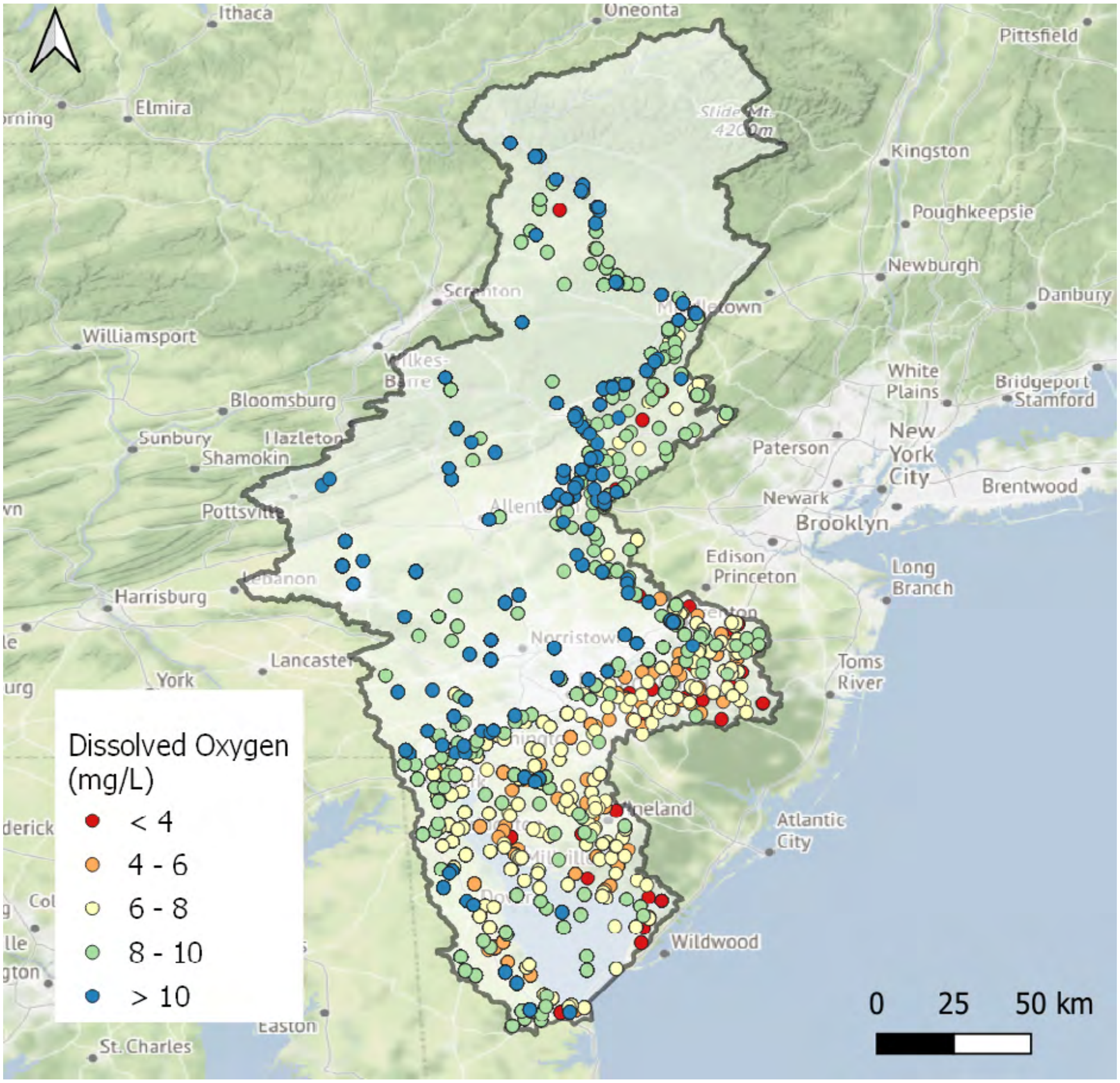

Figure 4.1.23 Summer surface water dissolved oxygen (mg/L) observations in the Delaware River Basin, 2011-2020.

Summary

Available data suggests that DO levels are reasonably good in many locations, with a few areas of localized low DO. The trend at Trenton suggests that DO is stable at relatively high saturation. We expect healthy levels of dissolved oxygen levels to persist under current regulations, with improvements at impacted sites over the long term. Expansion of continuous real-time monitoring capability in the Basin is recommended.

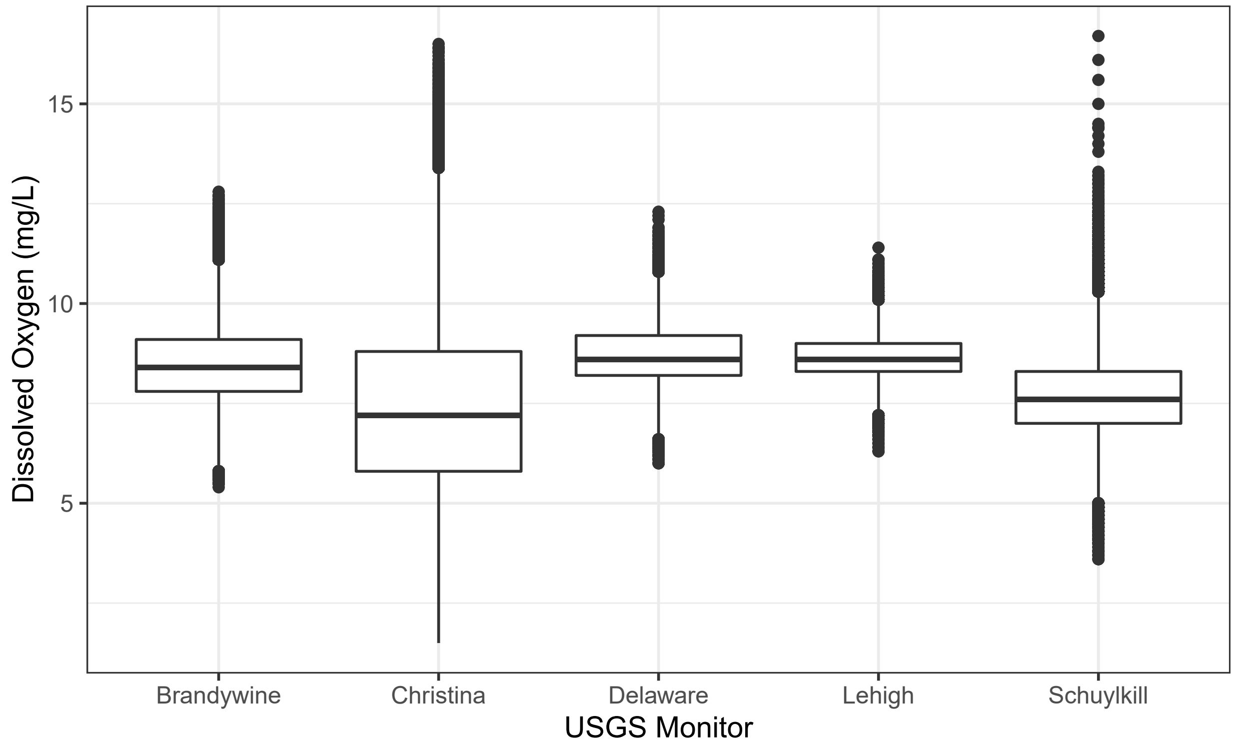

Figure 4.1.24 Box and whisker plot of summer dissolved oxygen (mg/L) from USGS continuous monitoring stations in the Delaware Basin, 2011 through 2020.

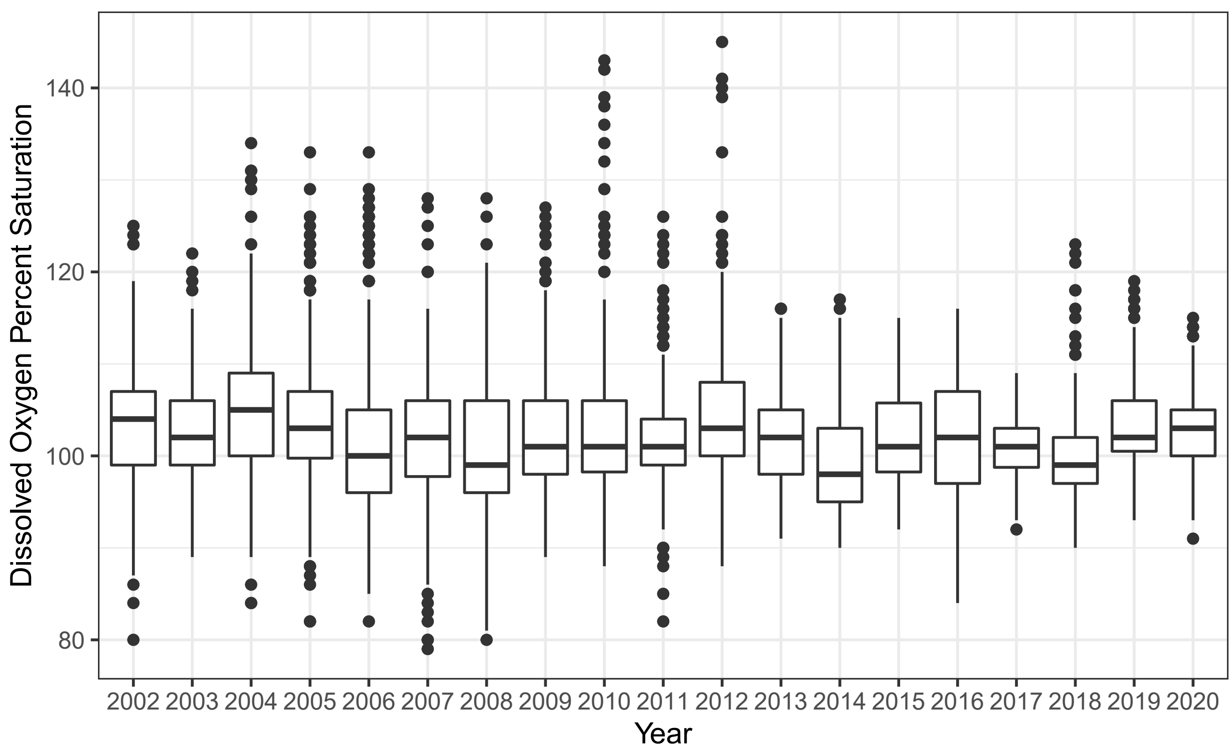

Figure 4.1.25 Mean daily dissolved oxygen percent of saturation by year at Delaware River at Trenton, 2002 through 2020.

4.1.2.2 Nutrients

A nutrient is any substance assimilated by living things that promotes growth. The term is generally applied to nitrogen and phosphorus, although it can also be applied to trace nutrients like silica and iron. According to US EPA, "High levels of nitrogen and phosphorus in our lakes, rivers, streams, and drinking water sources cause the degradation of these water bodies and harm fish, wildlife, and human health." This problem is widespread—more than half of the water bodies in the United States are negatively affected in some way by nitrogen and phosphorus pollution.

Description of Indicator

As part of its Special Protection Waters (SPW) regulations, DRBC has defined Existing Water Quality (EWQ) concentrations of several nutrients including total nitrogen, ammonia, nitrate, total Kjeldahl nitrogen, total phosphorus, and orthophosphate at multiple mainstem Delaware River Boundary Control Points (BCPs) and tributary Interstate Control Points (ICPs). DRBC adopted SPW regulations for Upper and Middle Delaware in 1992, using existing data available at that time to define EWQ, and permanently designated the Lower Delaware as SPW waters in July 2008, using data collected during 2000 through 2004 to define EWQ.

Present Status

We queried nitrate and total phosphate measurements in surface water in the Delaware River Basin from the National Water Quality Data Portal for the period 2011 through 2020. Locations and results are plotted in Figures 4.1.26 and 4.1.27 below. Figure 4.1.26 suggests relatively lower nitrate concentrations in the upper portion of the Basin, with higher values seen lower in the basin and within the Schuylkill sub-watershed. Figure 4.1.27 suggests lower phosphate concentrations in many locations with slightly higher levels seen near the urbanized and Estuary portion of the Basin. For the nitrate basin map, values were limited to those within the range from the first quantile to the 99th quantile, to minimize the scale impact of outliers.

Past Trends

In 2016, DRBC completed a project demonstrating that its Special Protection Waters (SPW) program is effective at protecting clean water and has even allowed improvements in nutrient concentrations. DRBC compared baseline water quality data initially collected from 2000-2004 to the assessment period of 2009-2011 at 24 sites located on the Delaware River and tributaries. For most nutrient parameters, at most locations, there were no measurable changes to existing water quality, and nutrient parameters showed improvement at most sites. All monitoring locations other than Pohatcong Creek, NJ demonstrated maintenance or improvement of nutrients for Existing Water Quality. The Pohatcong Creek site indicated measurable water quality change toward more degraded status in total nitrate and nitrite as well as total nitrogen (TN). DRBC's SPW program is designed to prevent degradation where existing water quality is better than the established water quality standards through management and control of wastewater discharges and reporting requirements. The report and details about the SPW program are available online.

In 2017 USGS completed an assessment of long term trends in water quality in New Jersey, including stations on the Delaware River and within the Basin. This assessment corroborates nutrient improvements in the nontidal Delaware River. That report is available online.

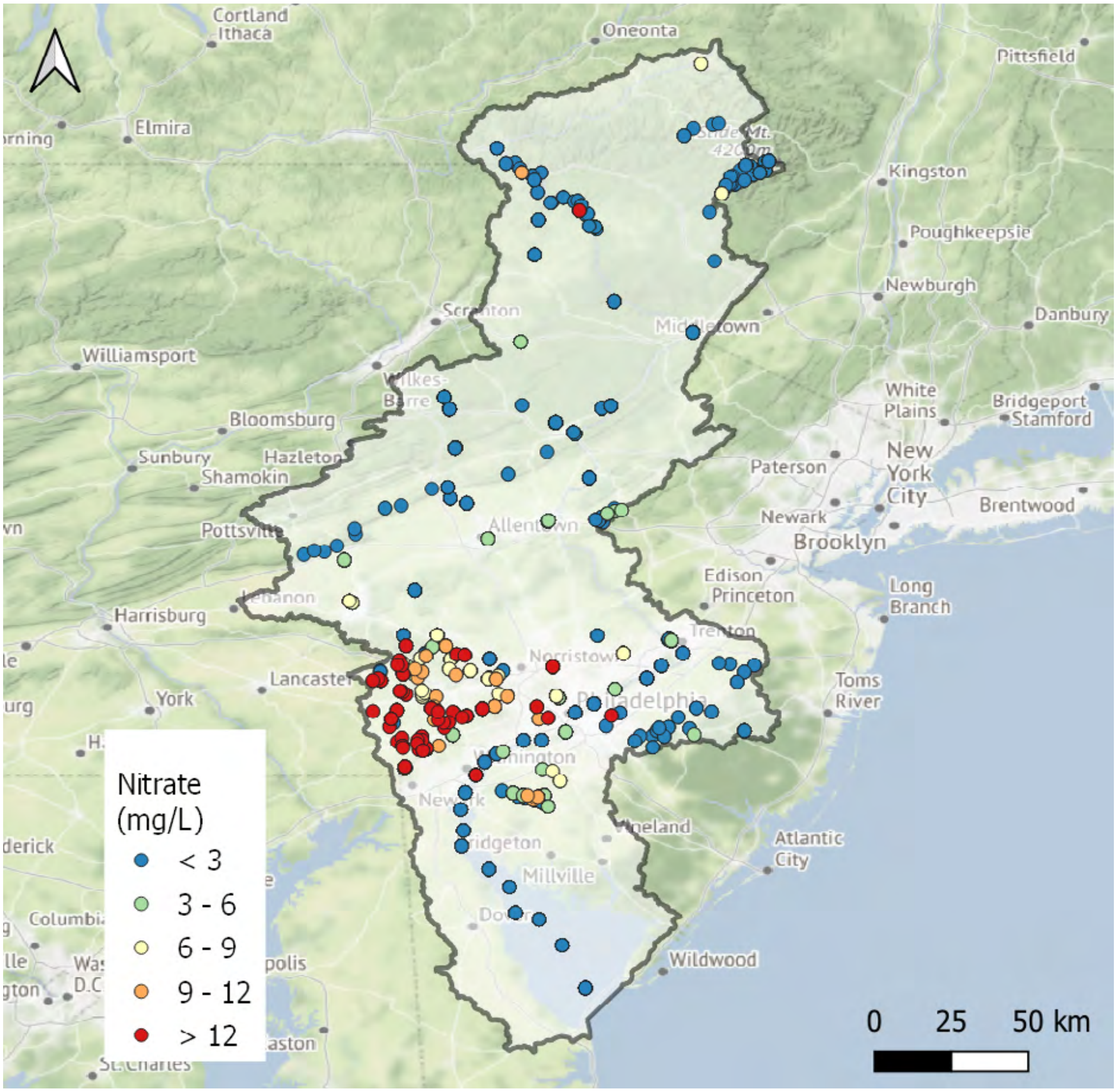

Figure 4.1.26 Surface water nitrate (mg/L) observations in the Delaware River Basin, 2011 through 2020.

Future Predictions

US EPA has prioritized nutrient criteria development in the United States for over 15 years, with states, interstates, and tribes serving as the lead agencies for understanding how nutrients function in their aquatic systems and what nutrient loadings and/or concentrations are needed to sustain healthy biological conditions long-term. As this effort to develop criteria comes to fruition, it is reasonable to presume that some subset of tributaries will be above criteria, and actions will be taken to remedy the exceedances. Thus it is reasonable to expect some continued modest decrease in nutrient concentrations.

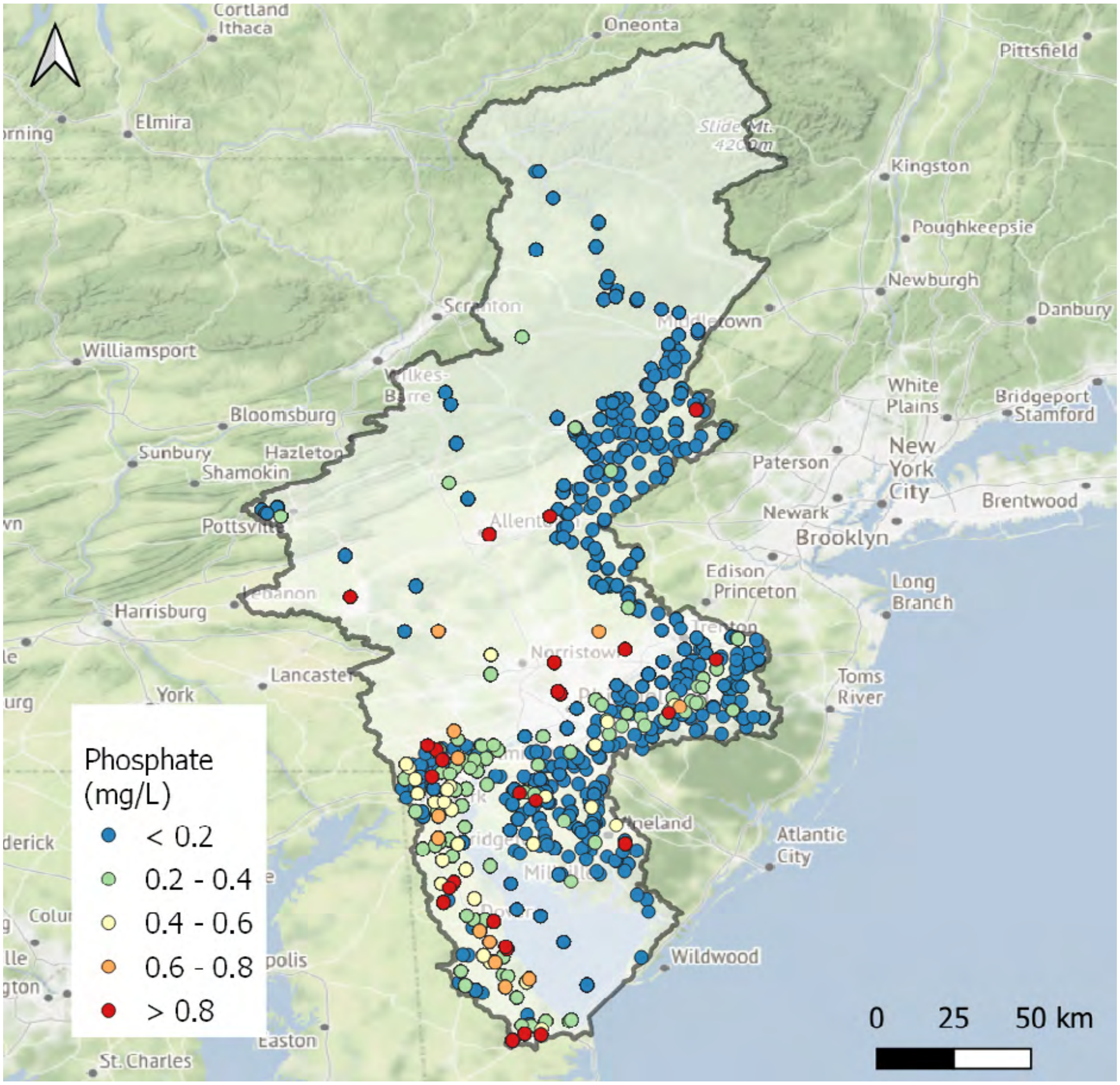

Figure 4.1.27 Surface water phosphate (mg/L) observations in the Delaware River Basin, 2011 through 2020.

Actions and Needs

The most important actions needed are the completion of the assessment to determine if existing water quality has been maintained at boundary control points and interstate control points. In addition, the continued development of numerical nutrient criteria is needed to ensure ecological health of basin waters.

Summary

The Assessment of Existing Water Quality performed by DRBC in 2016 suggests that at most of the locations evaluated for most nutrient parameters, conditions are being maintained or improving. The USGS assessment completed in 2017 corroborates these findings for the non-tidal Delaware River in New Jersey.

4.1.2.3 pH

pH is the mathematical notation for the negative log of the hydrogen ion concentration (-log[H+]) and indicates an acid, neutral, or base condition.

Description of Indicator

The pH of surface waters can be an important indicator of ecological function and productivity, and pH impacts the bioavailability and toxicity of pollutants such as metals and ammonia. Currently, DRBC's criteria for the Delaware River requires pH to be between 6.5 and 8.5.

Present Status

Boxplots of summer pH from USGS monitors at the Brandywine at Chadds Ford, PA, the Christina River at Newport, DE, the Delaware at Trenton, NJ, the Lehigh at Glendon, PA, and the Schuylkill at Norristown, PA, from 2011 through 2020 show different distributions in pH by location (Fig 4.1.28). Since pH can react to productivity, summer was selected to capture this influence. DRBC's criteria for the Delaware River requires pH to be between 6.5 and 8.5. Figure 4.1.29 shows the boxplot of pH instantaneous measurements by year at the Delaware River at Trenton from 2011 through 2020. The applicable criteria are plotted as red lines, and the plot shows that during many years, as many as 25% of the measured values exceed the upper limit criteria of pH = 8.5. Exceedances of the criteria are permissible when due to natural conditions, but more work is needed to evaluate what proportion of these exceedances are attributable to natural conditions. Some criteria violations are attributable to high pH conditions during periods of high primary production, although nutrient concentrations may contribute to the frequency and magnitude of pH exceedances through stimulation of algae and aquatic plants.

Past Trends

We developed a box plot of the daily median pH values at the Delaware River at Trenton by year for the period 2000 through 2020, shown in Figure 4.1.30 below. No clear trend is indicated.

Future Predictions

Observations of pH appear to be relatively stable in the non-tidal portion of the Basin. Continued stable pH, within the already observed ranges, seems likely.

Actions and Needs

More effort is needed to understand and evaluate routine exceedances above a pH value of 8.5 at Trenton. Although this could be a violation of the surface water quality standard, it would be permissible if due to natural conditions. While nutrients may play a role, we have also observed pH excursions above 8.5 in the upper portion of the River, where nutrient concentrations are substantially lower and considered to be oligotrophic. Efforts should be taken to better understand the drivers of these high pH events.

Summary

The pH of surface waters has long been recognized as both a natural and human-induced constraint to the aquatic life of fresh and salt water bodies, both through direct effects of pH and through indirect effects on the solubility, concentration, and ionic state of other important chemicals. Observations of pH at some locations, such as Trenton, show ranges frequently outside of criteria. A portion of this diel swing, however, is attributable to natural primary production.

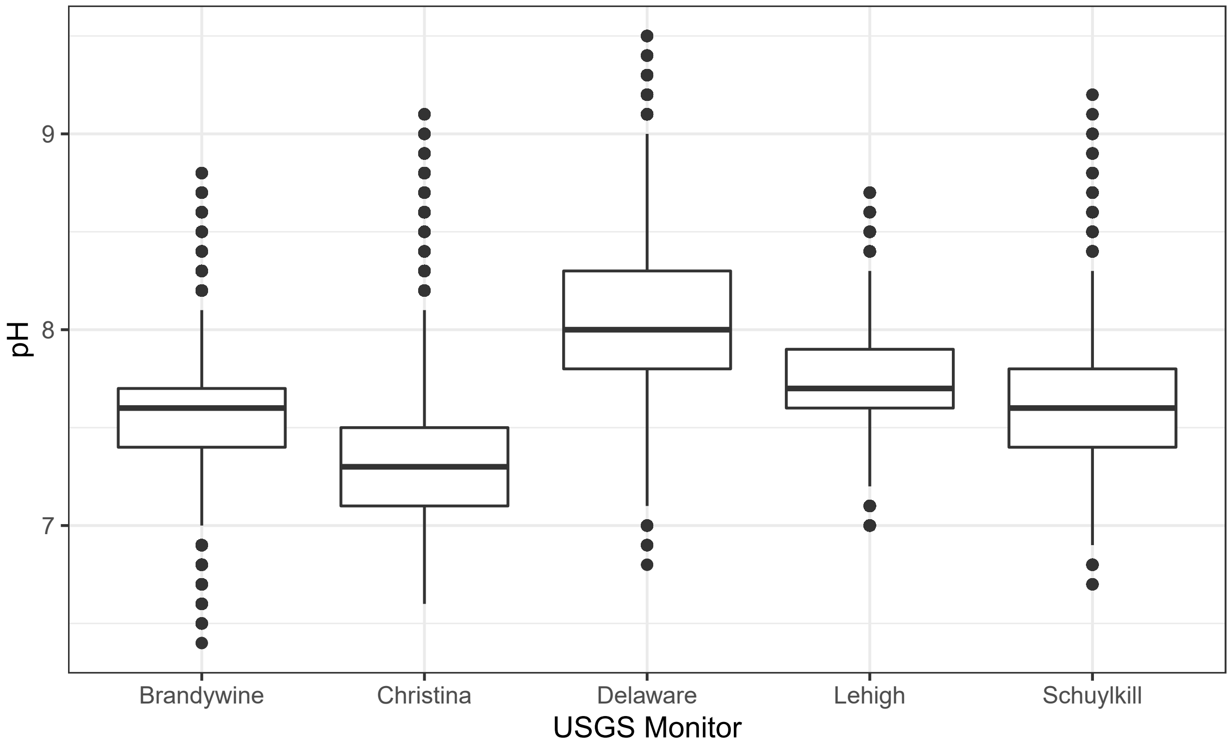

Figure 4.1.28 Summer pH observations at 5 USGS continuous Delaware Basin water quality meters 2011 through 2020.

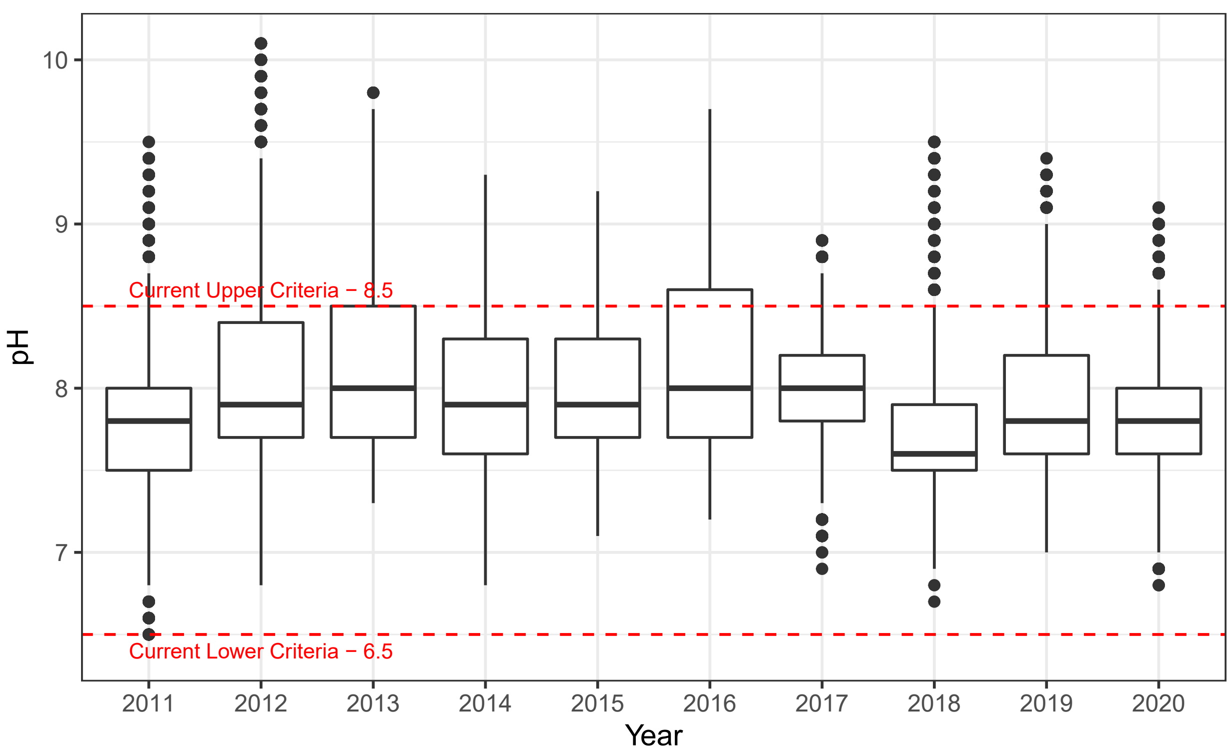

Figure 4.1.29 Instantaneous pH measurements by year, Delaware River at USGS 01463500, Trenton, 2011 through 2020.

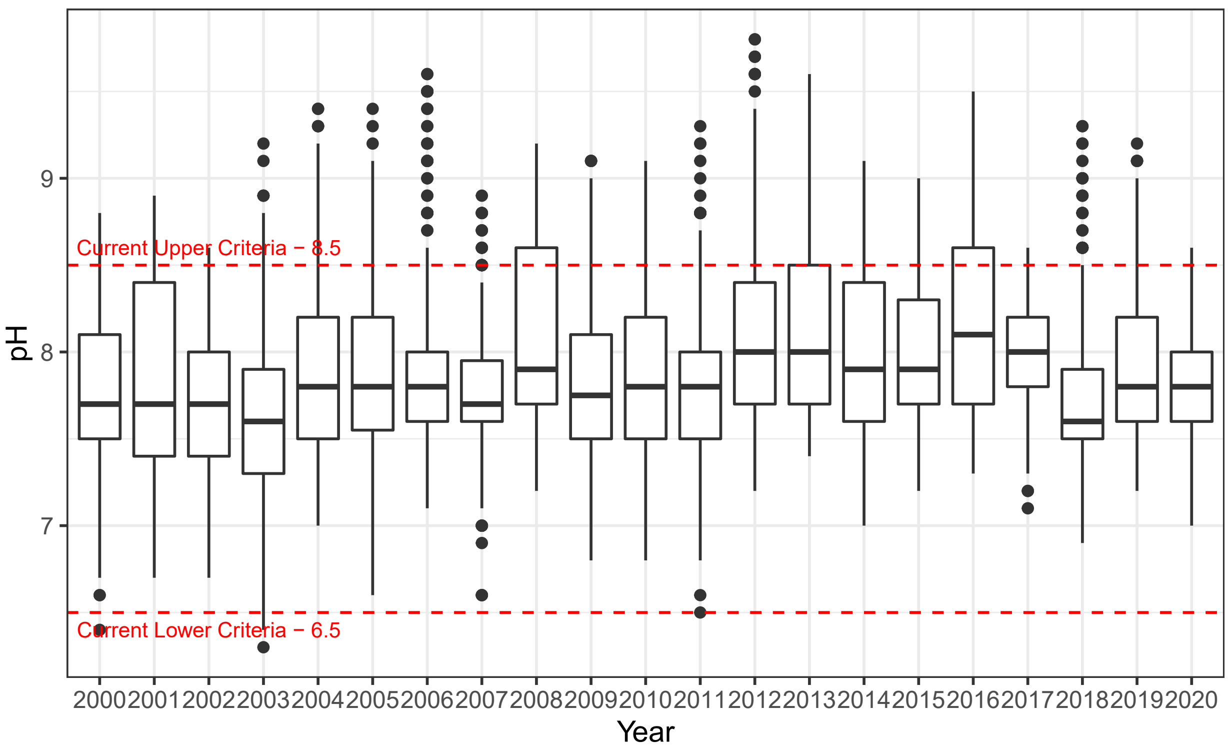

Figure 4.1.30 Daily median pH box and whisker plot at USGS 01463500, Trenton, 2000 through 2020.

4.1.2.4 Temperature

Water temperature is an important factor for the health and survival of native fish and aquatic communities. Temperature can affect embryonic development; juvenile growth; adult migration; competition with nonnative species; and the relative risk and severity of disease. Temperature assessment in the non-tidal Delaware River is confounded by artificially lowered temperatures from reservoir releases in the upper portion of the River and the lack of protective ambient criteria.

Description of Indicator

Currently, DRBC's criteria for temperature in the non-tidal River is oriented toward point discharge thermal mixing zones. As such, there are no specific temperature thresholds protective of the aquatic communities in the River and its tributaries. Pennsylvania, however, has adopted seasonally specific temperature criteria for warm water fisheries, which will be used for comparison in the upcoming section. Continuous temperature monitors are deployed at twelve stations in the non-tidal basin, including the East and West Branches of the Delaware, and the Delaware River at Callicoon, NY, Barryville, NY, and Trenton. Temperature regimes in the non-tidal Delaware are influenced by reservoir operations. Bottom discharges from the Cannonsville and Pepacton Reservoirs release colder water than would naturally occur.

Present Status

Figure 4.1.31 shows the summer temperature distributions at four USGS monitors in the mainstem Delaware River at Lordville, NY, Callicoon, NY, above Lackawaxen near Barryville, NY, and Trenton, from 2011 through 2020. This plot demonstrates the shift in temperature from the reservoir-influenced cold water upstream to warmer temperatures downstream. To assess whether the temperature regimes observed in the river were protective of aquatic communities, we compared the continuous measurements at Trenton to the Pennsylvania criteria for warm water fisheries. As shown in Figure 4.1.32 below, although the majority of observations are below (meeting) criteria, there are numerous violations, most frequently in the spring.

Past Trends

Long-term temperature records at Trenton (1954 through 2020) were evaluated to determine if the number of 'violations' would have increased over time (had those criteria been in place). As shown in Figure 4.1.33, no discernible trend in the number of violations per year is evident from the data.

Future Predictions

Temperature at Trenton appears to be stable over the continuous monitor period of record. Therefore, temperature at Trenton is expected to remain stable for the foreseeable future. Individual subwatersheds may see increases associated with development, increased impervious cover, and loss of tree canopy. In addition, global climate change is expected to exert upward pressure on water temperatures.

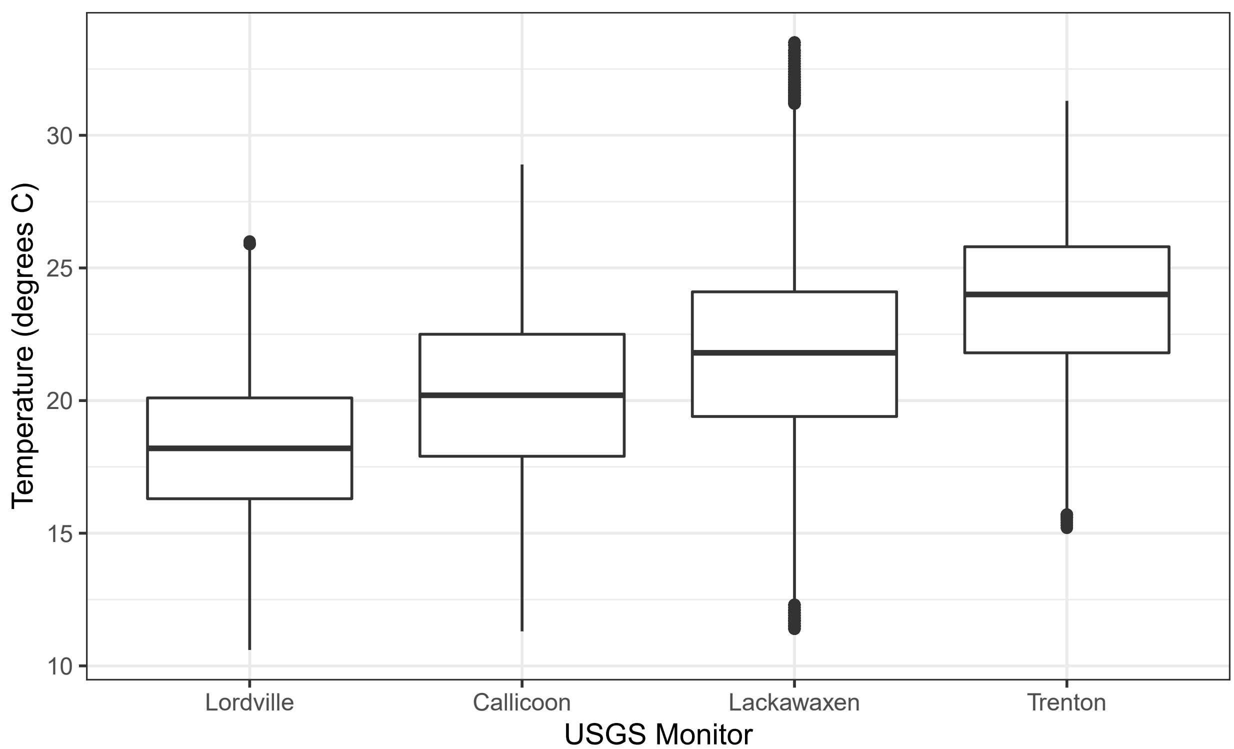

Figure 4.1.31 Summer water temperature box and whisker plot along the main stem of Delaware River, 2011 through 2020.

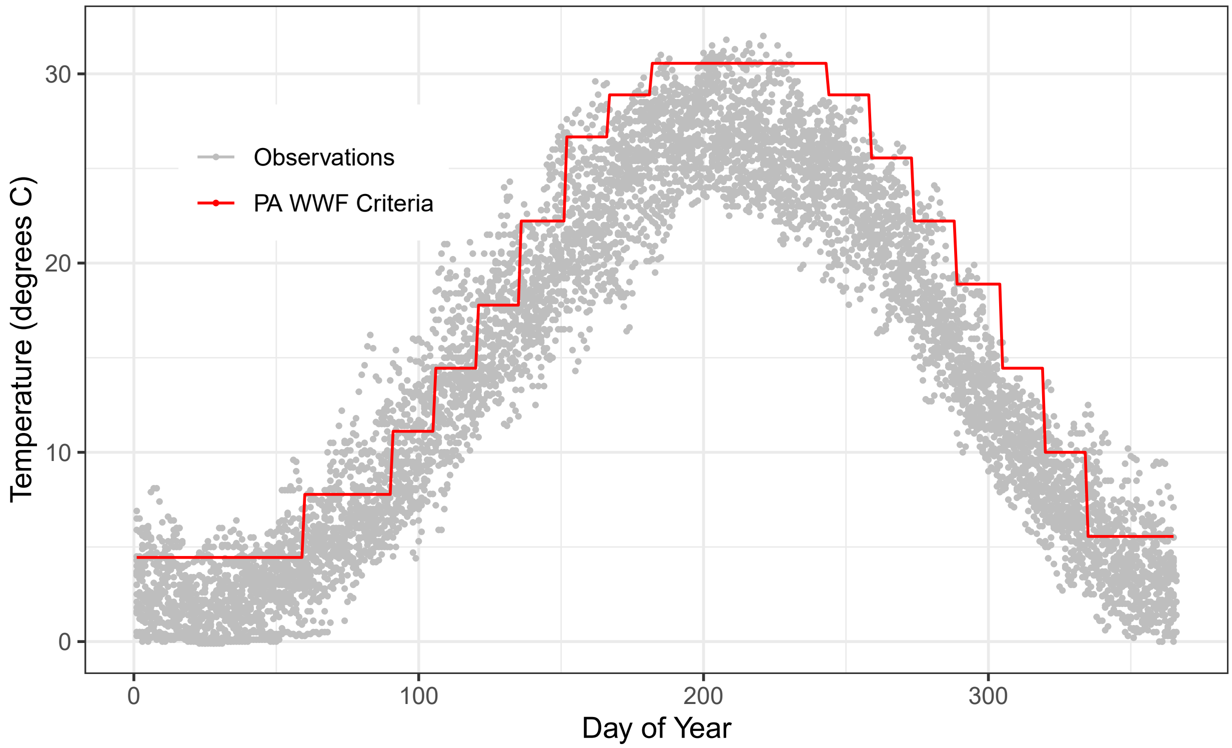

Figure 4.1.32 Comparison of maximum daily water temperature by day of year at USGS 01463500, Trenton to PA Warm Water Fishery Temperature Criteria, 2011 through 2020.

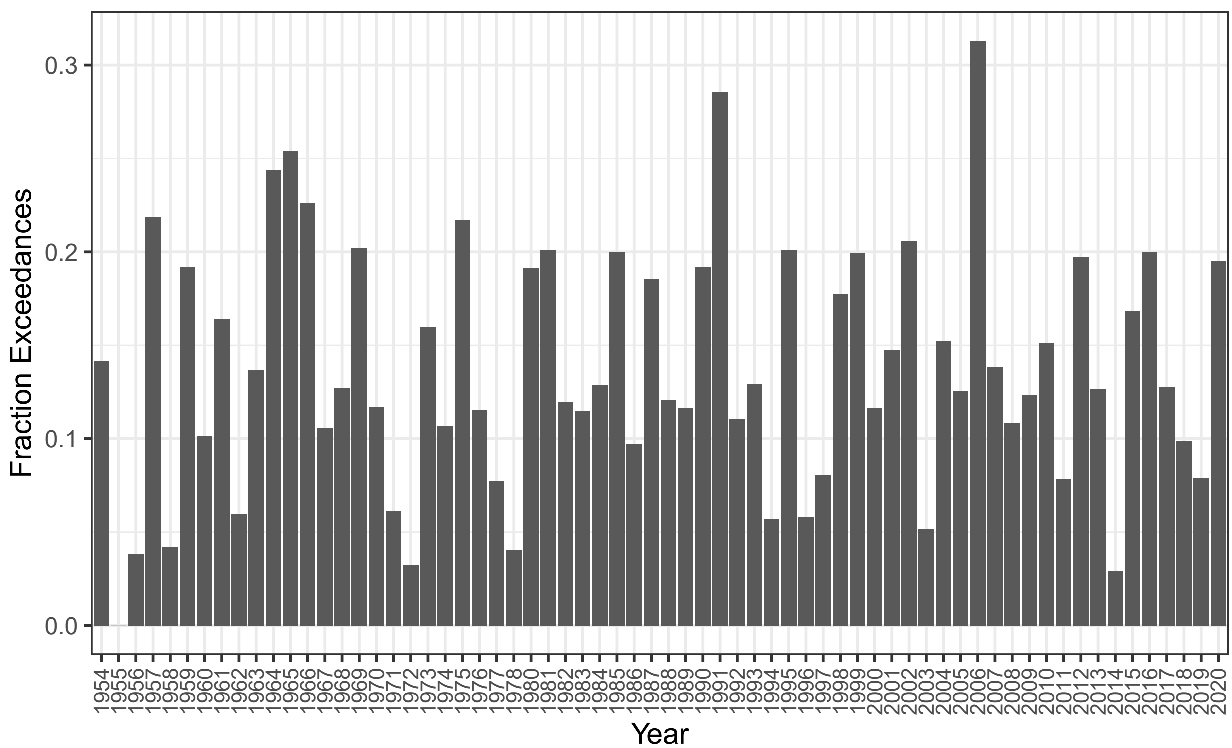

Figure 4.1.33 Fraction of water temperature readings exceeding PA WWF criteria by year along Delaware River at USGS 01463500, Trenton.

Actions and Needs

The development of temperature criteria in the non-tidal portion of the Delaware River should be prioritized to help protect aquatic communities and allow meaningful interpretation of presently collected data. In addition, stronger linkages between meteorological drivers and resultant water temperatures are needed, so that assessors can distinguish between natural conditions and anthropogenic thermal loads.

Summary

Temperature assessment in the non-tidal Delaware River is confounded by artificially lowered temperatures from reservoir releases in the upper portion of the river and the lack of protective ambient criteria. A comparison to Pennsylvania's warm water criteria shows exceedances at Trenton. The majority of exceedances occur in the spring.

References

American Public Health Association (APHA), American Water Works Association (AWWA), Water Environment Federation (WEF). 2005. Standard Methods for the Examination of Water and Wastewater. 21st edition

Culberson, C.H. 1988. Dissolved oxygen, Inorganic carbon, and the acid-base system in the Delaware Estuary. pgs. 58 to 76 –in—S.K. Majumdar, E.W. Miller, and L.E. Sage (eds) "Ecology and Restoration of the Delaware River Basin." Penn. Acad. of Sci. press. 431 pp.

Kim, K.W. and Johnson, B.H. (2007). "Salinity Re-Validation of the Delaware Bay and River 3D Hydrodynamic Model with Applications to Assess the Impact of Channel Deepening, Consumptive Water Use, and Sea Level Change," U.S. Army Engineer Research and Development Center, Vicksburg, MS.

National Estuary Program Coastal Condition Report. 2006. Reference EPA-842/B-06/001. 2006. Office of Water/Office of research and development, Washington, DC. 20460.

Silldorff, E. 2015. Existing Use Evaluation for Zones 3, 4, & 5 of the Delaware Estuary Based on Spawning and Rearing of Resident and Anadromous Fishes. Delaware River Basin Commission.

4.2 Contaminants

4.2.1 Tidal – Contaminants

4.2.1.1 Contaminants

The "Contaminants" indicator is a general category for specific elements and compounds with varying degrees of toxicity to aquatic life and human health. They include metals, pesticides, and other pollutants that have entered the environment from industrial and commercial facilities; oil and chemical spills; non-point sources such as roads, parking lots, and storm drains; and wastewater treatment plants and sewage systems. Some pollutants resist breakdown and accumulate in the food chain. They can contaminate water at levels that cause harm to the environment and public health.

Description of Indicator

Water quality monitoring data from multiple organizations including the Delaware River Basin Commission (DRBC), Delaware Department of Natural Resources and Environmental Control (DNREC), New York State Department of Environmental Conservation (NYSDEC), New Jersey Department of Environmental Protection (NJDEP), Pennsylvania Department of Environmental Protection (PADEP), and the United States Geological Survey (USGS) are compared to stream quality objectives and narrative standards (equivalent to state and EPA's surface water quality criteria) to evaluate water quality. DRBC has stream quality objectives for human health and aquatic life used in assessment of the tidal portion of Delaware Basin from the head of tide at Trenton to the mouth of the Delaware Bay (Zones 2 through 6) that reflect current scientific information and harmonize DRBC criteria with basin states' criteria. In addition, a narrative standard applicable to waters of the Basin requires that: "the waters shall be substantially free from substances in concentrations or combinations which are toxic or harmful to human, animal, plant, or aquatic life."

Present Status

For a recent report on the extent to which waters of the Delaware Estuary and Bay are attaining designated uses, see the Delaware River and Bay Water Quality Assessment (2020). Some contaminants identified in the report for additional monitoring and assessment efforts to assure water quality in the Estuary and Basin are the metals copper and aluminum, the pesticide dieldrin produced by the oxidation of aldrin pesticides and polycyclic aromatic hydrocarbons (PAHs).

Past Trends

Since the passage of the Clean Water Act in 1972 and other legislation, contaminant loadings have been reduced. Sediment cores may yield some insight into estuary pollution histories. Sediment cores collected in 2001 in marshes bordering the water column in Zone 4 and upper Zone 5 were analyzed for the metals silver, cadmium, chromium, cobalt, copper tin and lead. By analyzing slices along the core, time history of toxic pollutants can be measured. The results indicated for most metals a 2-to-5-fold increase between the early 1950s until the late 1960s or early 1970s, with gradual decreases thereafter. Lead and tin displayed a 10-fold increase after 1950 followed by decreasing levels after the early 1970s (Church et al., 2006). The organic chlorine contaminants PCBs in sediment cores from a different study exhibited peak concentrations in the 1960s and 1970s. Concentrations were lowest in the southern portion of the Bay increasing upstream (Vile et al., 2010). It is reasonable to expect that sediment time histories reflect the broader trends that occurred in water column estuary concentrations, with generally increasing concentrations until concentrations peaked in the early 1970s, followed by decreasing concentrations thereafter.

Future Predictions

Based on historical trends, levels of heavy metals and other contaminants are likely to continue decreasing. Many legacy contaminants have been banned or limited, so trends are expected to be stable or decreasing. However, with increasingly sensitive analytical methods in use to measure contaminants (e.g., inductively coupled plasma mass spectrometry) and more complex models to evaluate toxicity (e.g., multiple linear regression; USEPA, 2017), increased coordination of water quality criteria and assessment methodologies is needed for a better understanding of contaminants for additional monitoring and assessment (e.g., copper, aluminum, pesticides and PAHs) and to prioritize environmental management efforts.

Actions and Needs

Coordination among basin states and agencies should continue to ensure the use of appropriate analytical techniques and assessment methodologies to evaluate the effects of contaminants on water quality.

Summary

Trends for specific contaminants may result from regulatory restrictions on use, changes in loading rates or degradation of the contaminant in the environment, but effective management is needed to maintain water quality and efficiently address elevated contaminant levels.

4.2.1.2 Fish Contaminant Levels

Certain chemicals tend to concentrate ("bioaccumulate") in fish to levels thousands of times greater than the levels in the water itself. The resulting concentrations in fish and the attendant health risks to those individuals who consume the fish, such as recreational and subsistence anglers, are of concern to government agencies and the public.

Description of Indicator

Bioaccumulative contaminants have been monitored over an extended period in fish tissue collected from the Delaware River. Bioaccumulation of contaminants in fish tissue such as polychlorinated biphenyls (PCBs), mercury, dioxins/furans, some per- and polyfluoroalkyl substances (PFAS), dichlorodiphenyltrichloroethane (DDT), and other chlorinated pesticides are influenced by physical-chemical properties of the contaminant, fish species, age, migration, and food habits as well as other environmental factors such as season of fish collection.

Present Status

While programs are in place to reduce the concentrations of toxic pollutants that bioaccumulate, Delaware River basin states issue "advisories" containing meal advice for consumers of recreationally caught fish and shellfish to minimize the risk to human health. These advisories list the water bodies, fish species, and number of meals recommended to minimize the risk. In some cases, no consumption of any fish species from a water body or more stringent consumption guidelines for pregnant women and children is advised. These advisories are updated based upon recent fish tissue concentration data. A summary of recent fish consumption advisories in the Delaware River is available online.

The bioaccumulative contaminants currently causing fish consumption advisories in the Delaware River Estuary are polychlorinated biphenyls, mercury, and perfluorooctane sulfonate.

The following links provide additional information on state-issued fish consumption advisories:

- New Jersey | NJDEP Fish Consumption Advisories
- Delaware | DNREC Fish Consumption Advisories
- New York | NYDEC Fish Health Advisories
- Pennsylvania | PADEP Fish Consumption Advisories

Past Trends

A number of bioaccumulative compounds are monitored in fish collected from the tidal Delaware River. Trends will differ depending on the contaminant of interest. For example, declining levels of polychlorinated biphenyls (PCBs) in tidal fish reflect the efforts to reduce PCB loadings through the implementation of Total Maximum Daily Loads (Stage 1 PCB TMDLs). PCBs are a class of man-made compounds that were manufactured and used extensively in electrical equipment such as transformers and capacitors, paints, printing inks, pesticides, hydraulic fluids, and lubricants. PCBs are considered legacy pollutants since their manufacture and use have generally been banned by federal regulation since 1978. However, mobilization and recycling of legacy PCBs results in ongoing sources of PCBs to the Delaware River Estuary and Bay including: industrial and municipal wastewater treatment plants, combined sewer overflows (CSOs) and municipal separate storm sewer systems (MS4s); contaminated sites; tributaries and boundaries such as the ocean and the C&D Canal; nonpoint source runoff directly to the estuary; atmospheric deposition and exchange of PCBs between estuary waters and the atmosphere; and sediments contaminated by PCBs. In addition, PCBs may also be incidentally created as a byproduct from certain manufacturing processes, such as dye and pigment production. DRBC, in close coordination with the co-regulating states in the estuary (DE, PA, and NJ) as well as USEPA Regions II and III, has developed draft Stage 2 PCB TMDLs that incorporate enhanced loading data and implementation requirements as well as revised criteria.

Per- and polyfluoroalkyl substances (PFAS) are another contaminant of interest in fish. Elevated levels of perfluorononanoate (PFNA) and perfluoroundecanoate (PFUnA) were observed in tidal fish fillet collected from the Delaware Estuary by the DRBC in the 2000s. PFNA and PFUnA are two of the many PFAS found in a variety of industrial and household products such as stain-repellant textiles, firefighting foams, and paper coatings. Sources of PFAS in the environment include direct emissions from the manufacturing, use, and disposal of surfactants and other materials containing PFAS as well as indirect emissions by transformation of precursor substances such as fluorotelemer alcohols. PFAS discharges to rivers can arise from waste treatment facilities, especially wastewater treatment facilities receiving industrial waste or landfill leachates. Manufacturing and use of some PFAS have been reduced or eliminated in the USA through a voluntary stewardship agreement between the USEPA and major manufacturers. There have been numerous other federal and state initiatives in recent years to manage PFAS exposure. Significant decreases in PFNA and PFUnA concentrations have been observed in fish fillet from the tidal river. However, fish fillet in some species and locations within the Delaware River Basin continue to be contaminated with a different PFAS perfluorooctanesulfonate (PFOS) at levels exceeding recommended regional risk advisory limits on fish consumption.

Future Predictions

Given the lasting nature of the fish tissue contaminants and the many sources of the contaminants, it is reasonable to presume that concentrations will remain relatively constant. Even the effects of regulatory water quality management efforts may take decades to be reflected in tissue concentrations.

Actions and Needs

Pollution minimization efforts are necessary to bring about the needed reductions in tissue concentrations. Cooperative efforts among state and federal agencies and other partners to reduce bioaccumulative contaminants in the Delaware River should continue and be expanded.

Summary

Trends for specific contaminants may result from regulatory restrictions on use, changes in loading rates or degradation of the contaminant in the environment. Trajectories for contaminant reduction in fish may be long depending on the contaminant of concern, but effective management is needed to facilitate these trajectories.

4.2.1.3 Emerging Contaminants

Emerging contaminants (ECs) or contaminants of emerging concern are substances that have entered the environment through human activities, may have new or changing information that increases the level of concern, and pose a real or perceived threat to the environment. Current regulatory approaches are inadequate to address these contaminants and the increasing public concern over their environmental and human health implications.

Description of Indicator

Thousands of new chemicals are created and used in commerce each year. Keeping pace with the knowledge of the risk in water to human and ecological health of these substances is challenging. A growing body of information on the adverse effects of ECs, as well as improved analytical detection methods, have caused increased interest and concern about how these substances impact our water resources. Perfluoroalkyl and polyfluoroalkyl substances (PFAS) and pharmaceuticals and personal care products (PPCP) are two very different classes of ECs that can illustrate the challenges in understanding their presence, sources, pathways, and persistence in the environment. PFAS are found in a variety of industrial and household products. Human and wildlife exposure to PFAS is widespread. Increasing evidence suggests adverse effects of PFAS to human health and the environment. Human health risks from exposure through drinking water and fish consumption are areas of concern. Therefore, understanding occurrence and exposure risk is important to protect water resources. A wide range of pharmaceutical and personal care products (PPCP) including prescription medicines, over the counter medicines (OTC), antibiotics, and antibacterials including active pharmaceutical ingredients considered essential for maintaining or improving health are also entering the environment. Until recently, the fate and transport of many common PPCP were not of great concern. However, some of these synthetic compounds may ultimately pose harm to the environment. PPCP are of concern because of their biological effects, resistant to degradation, widespread and increasing use, and because conventional wastewater treatment plants are not designed to remove them. Concentrations of PPCP have been shown to be generally higher in urbanized and industrialized areas similar to those present in the tidal Delaware River.

Past Trends

Multi-year surveys of ECs have been conducted in the Delaware River. PFAS have been investigated in fish fillet over an 18-y period (2004–2021). The sample period coincides with actions to reduce or eliminate the release of certain PFAS to the environment. Elevated levels of perfluorononanoate (PFNA) and perfluoroundecanoate (PFUnA) were initially observed in tidal fish fillet. While significant decreases in PFNA and PFUnA concentrations were observed in fish fillet from the tidal river during the timeframe of the study, changes in concentrations of other PFAS in tidal and nontidal fish were less substantial (MacGillivray, 2021). A multi-year survey of PPCP (2007, 2008, 2009 and 2017) in the main stem tidal Delaware River surface water conducted by the DRBC did not observe significant trends. Based on environmental occurrence, aquatic ecotoxicity, potential human health effects and analytical feasibility, fifteen PPCP were identified as candidates for additional monitoring and assessment in surface waters of the tidal main stem Delaware River (MacGillivray, 2013). In another study of the occurrence of pharmaceuticals, hormones, and organic wastewater compounds in Pennsylvania waters, estrogenic compounds were present at higher concentrations in the Delaware River Basin (Reif et al., 2012). DRBC and Temple University's Water Environment and Technology (WET) Center also sampled and analyzed surface water from Delaware River tributaries to increase the understanding of loading, distribution, and potential risk of PPCP in highly urbanized areas significantly impacted by wastewater treatment plant effluents. The measured environmental concentrations of the target compounds presented a detailed picture of urban and industrial impacts on subwatershed receiving waters (Vilimanovic et al, 2020). In surface water samples collected on September 25, 2017 from the mainstem Delaware River, PPCP prioritized for further study continue to be detected (Fig 4.2.1).

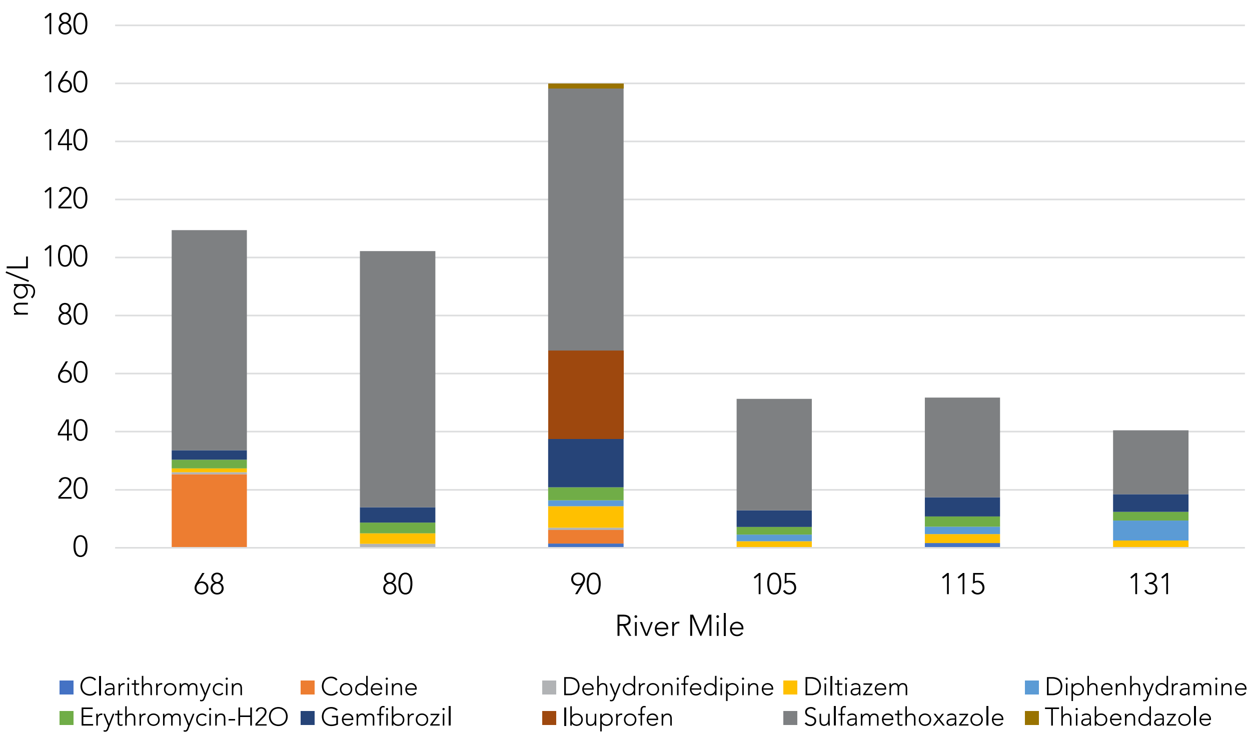

Figure 4.2.1 PPCP in surface water collected on September 25, 2017, from the Delaware River.

Present Status

The DRBC, basin states, U.S. EPA and others are working to increase knowledge of ECs within the Delaware River Basin. Increasing knowledge of ECs can inform source water protection and water quality assessment. Some links to those efforts are listed here.

- Per- and Polyfluoroalkyl Substances (PFAS) | US EPA
- Contaminants of Emerging Concern including PPCP | US EPA
- Contaminants of Emerging Concern | Delaware River Basin Commission (nj.gov)
- PFAS Information | NJDEP - Division of Science and Research
- PFAS in Delaware | DNREC Alpha
- Contaminants of Emerging Concern | PADEP (pa.gov)

Fish fillet continued to be contaminated with perfluorooctanesulfonate (PFOS) at levels exceeding recommended regional risk advisory limits on fish consumption (MacGillivray, 2021). DRBC monitoring and other surveys continue to detect PFAS in surface water but appear to be below regional and national guidelines in areas designated as drinking water sources (i.e., river miles ≥ 95) (Fig 4.2.2). PPCP have been detected in the tidal Delaware River at concentrations comparable to compounds and concentrations measured in other studies of surface water in urban areas (Vilimanovic, 2020). Based on environmental occurrence, aquatic ecotoxicity, and potential human health effects to sensitive populations, PPCP are identified for focused study in the tidal Delaware River.

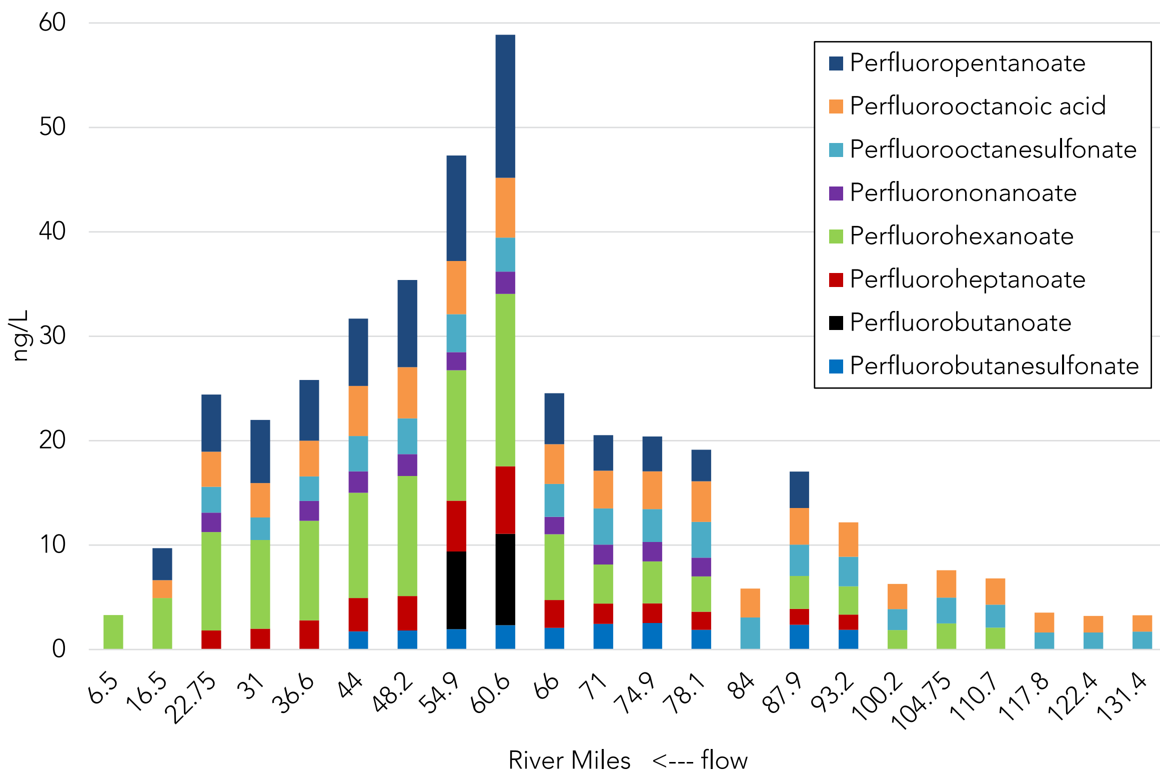

Figure 4.2.2 PFAS in surface water collected on September 20, 2021, from the Delaware River.

Future Predictions

Multiple factors are expected to contribute to the future rise of ECs in the environment including population and climate changes. With a larger and aging basin population, an increase use of prescription and over the counter (OTC) pharmaceuticals is expected to increase pharmaceuticals in the environment. According to this TREB (see Chapter 2), temperatures and precipitation in the Delaware River Basin appear to be increasing. The reports further indicate anticipation of extreme weather events, such as tropical storms, flooding, short-duration dry periods and a sea level rise. Drought events will result in low flow that can lead to a greater concentration of ECs in surface waters. This is prevalent for tributaries where the flow is dominated by the WWTP discharge. The large flooding scenarios have the potential where WWTPs bypass the plant raw influent directly into the basin due to operational safety of the plant, thus increasing the amount of untreated ECs. Such events are not uncommon in the Greater Philadelphia area and the nearby city of Camden, NJ, where plants can be overwhelmed by the flow coming from the combined sewer outfall (CSO). These predictions increase future concerns for ECs in surface waters (Vilimanovic et al, 2020). Identification of new contaminants in the environment will continue to be a challenge. When an EC is identified, replacement or alternative substances in the same class are sometimes not a significant improvement causing a regrettable substitution. New approaches to assessing ECs such as by groupings of compounds (e.g., commercial use, chemical properties, or similar toxicological modes of action) can be made (Wilkinson, 2017). Incorporation of bioanalytical screening methods that integrate exposure to complex mixtures should be considered (Maruya, 2014). Approaches for evaluating ECs in water in a broader context, to supplement the current chemical-by-chemical approach for development of MCLs and water quality standards have been identified (NJDEP SAB, 2020). In addition, sustainable practices like incorporating more closed loop processing with recapture technology would prevent or reduce off-site contamination by ECs (NJDEP SAB, 2020)

Actions and Needs

The effects of ECs in estuarine and coastal waters are not well studied. Future work should evaluate the sources as well as the fate and effects of ECs in the Delaware River water column, sediment, and biota. Available information on the environmental effects of ECs is rapidly increasing and any assessment should be updated periodically using current information. Challenges with addressing ECs will persist into the future in the absence of focused efforts. Prevention, identification, assessment, and management intervention actions need to improve for water resource managers. More effective coordination and utilization of resources are needed to streamline EC understanding and decision-making to improve prevention and response (ASDWA, 2019). See the Association of State and Territorial Solid Waste Management Officials webinar titled "How to Develop a State-led Contaminants of Emerging Concern Program" for more guidance information.

Some general recommendations to establish a more focused EC program include:

- Establish a EC workgroup with a EC nexus;
- Develop a EC Problem Statement;
- Engage partners (agencies, private and public) for collaboration;
- Develop a formalized process to identify the next contaminant of concern;
- Evaluate available occurrence monitoring data/trends;
- Identify sources, fate, and effects;

- Develop a watch list of candidate EC;
- Apply screening criteria to prioritize and assess EC action;
- Identify strategies for monitoring, managing, and addressing specific EC.

Actions identified to minimize EC impacts in the Delaware Estuary and Bay include:

- Public education - source reduction (e.g., drug take back locations and information on program in each basin state, NY State Pollution Prevention Institute, and Endocrine Disrupting Compound (EDC) Footprint Calculator)
- Support decision-making about significant sources of EC exposure by identifying safer alternatives to CEC. (e.g., Interstate Chemicals Clearinghouse)
- Identification and prioritization of substances for which development and adoption of water quality standards is appropriate.
- Chemical Action Plans - comprehensive plans to identify, characterize, and evaluate all uses and releases of a chemical and to recommend actions to protect human health and the environment (e.g., Interim Chemical Action Plan for Per- and Polyfluorinated Alkyl Substances; PFAS Action Team Initial Report).
- Evaluate pre-treatment options and treatment technologies in coordination with basin states and USEPA.

Summary

Potential risks from ECs should be continually updated with recent occurrence data and current science. Risks to designated uses in Delaware Estuary and Bay include the potential of aquatic ecotoxicity from ECs to impair maintenance and propagation of fish and aquatic life and the potential of human health effects from ECs to impair source water and/or fish consumption. Monitoring and assessment tends to emphasize well known and regulated chemicals. ECs in aquatic environments require very different monitoring tools than pollutants that are acutely toxic to wildlife and humans. Chemical-specific approaches cannot efficiently monitor all ECs. Low concentrations and mixtures of ECs present challenges to water quality assessment. The integration of effects-based tools, passive sampling, non-targeted chemical analysis, and traditional targeted chemical analysis is a proposed strategy for monitoring and prioritization of EC. Cooperation among national and regional partners is needed along with education to address the increased interest and concern about how these substances impact our water resources.

4.2.1.4 Whole Effluent Toxicity

Description of Indicator

The tidal Delaware River contains numerous industrial and municipal facilities with National Pollutant Discharge Elimination System (NPDES) effluent discharges. Whole effluent toxicity (WET) testing is a useful approach in the protection of aquatic life by using toxicity tests to measure toxicity of effluents along with the chemical-specific control approach. The two primary advantages to using WET testing over individual chemical-specific controls are: (1) WET tests evaluate the integrated effects of all chemicals in an aqueous sample and (2) WET tests can measure toxicity caused by other compounds for which a chemical-specific numeric criterion has not been established or do not have an approved parameter specific analytical test method. Chronic toxicity tests can detect effects at a much lower dose than acute toxicity tests providing a better estimate of the safe concentration of effluents in receiving waters. Therefore, chronic toxicity tests have a greater potential to produce more ecologically relevant data.

Past Trends

Of the twelve largest individual dischargers in the estuary, six dischargers exhibited a decreasing trend for at least one test species and six dischargers exhibited no trend in 2002 to 2014. Surveys measuring survival, growth, and reproduction in aquatic organisms indicated that the samples from sites tested in the main stem of the Delaware River and from the majority of its tributaries do not produce chronic toxicity. The surveys identified tributaries that warrant further assessment for toxicity (MacGillivray et al., 2011). For more information on DRBC's toxicity monitoring see https://www.state.nj.us/drbc/programs/quality/ambient-tox.html#2.

Present Status

Most effluent discharges to the Delaware Estuary are currently monitored for WET. Based on existing water quality regulations for the estuary, no adverse effects should be observed in toxicity tests with undiluted ambient water (USEPA 1991, 2008b). Monitoring toxicity is an essential component of programs designed to protect this valued resource and to assess compliance with regulatory standards.

Actions and Needs

Recommendations for future WET monitoring in the Delaware Estuary include continued coordination among the basin states, DRBC and USEPA to generate consistent WET testing throughout the estuary, and full compliance with WET monitoring by estuary dischargers. Since the use of a numerical model to predict ambient toxicity from effluent data is complicated by possible additive effects of chronic toxicity, it is recommended that continued efforts be made to monitor not only effluent from discharges but also the ambient environment to ensure that the Delaware River Estuary supports aquatic life from toxicity.

Summary

Limiting chronic toxicity in effluents decreases the impact of point source discharges on water quality in the Delaware Estuary. Monitoring for WET for point source discharges in the Delaware Estuary keeps a focus on controlling toxicity in effluents.

4.2.2 Non-Tidal – Contaminants

4.2.2.1 Contaminants

The "Contaminants" indicator is a general category for specific elements and compounds varying degrees of toxicity to aquatic life and human health. They include metals, pesticides, and other pollutants that have entered the environment from industrial and commercial facilities; oil and chemical spills; non-point sources such as roads, parking lots, and storm drains; and wastewater treatment plants and sewage systems. Some pollutants resist breakdown and accumulate in the food chain. They can contaminate water at levels causing harm to the environment and public health.

Description of Indicator

Water quality monitoring data from multiple organizations (DRBC, DNREC, NYSDEC, NJDEP, PADEP and USGS, previously defined in Section 4.2.1) are included in water quality assessments of the Delaware River including data from DRBC enhanced studies of non-tidal (Zone 1) metals. Toxic pollutants data are collected using USEPA approved or equivalent methods with the level of monitoring varying by Zone and toxic pollutant.

Present Status

To ensure attainment and maintenance of downstream water quality standards and to facilitate consistent and efficient implementation and coordination of water quality-related management actions in shared interstate waters protected for public water supply, the most stringent ambient water quality criteria for human health for New York or Pennsylvania are compared to surface water data in non-tidal DRBC Water Quality Management Zones 1A and 1B. The most stringent ambient water quality criteria for human health for Pennsylvania or New Jersey is compared to surface water data in non-tidal DRBC Water Quality Management Zones 1C, 1D, and 1E. For waters protected for use by fish and other aquatic life, the most stringent ambient water quality criteria apply in non-tidal shared interstate waters. The report "2020 Delaware River and Bay Water Quality Assessment (2020)" describes concerns for the support of human health due to PCB and mercury concentrations and the need for further evaluation of aluminum, cadmium and copper in non-tidal segments of the river.

Past Trends

Data and detection insufficiencies make determination of past trends difficult.

Future Predictions

As monitoring and assessment procedures are refined, and criteria updated to reflect current research, appropriate end points can be defined along with the non-tidal zone contaminant concentrations relative to those endpoints. In the face of improving management, it is reasonable to expect improvements in water quality and declines in concentrations of priority pollutants; however, it is more likely that levels will remain relatively the same at their current levels. Although some upward pressure is likely to be exerted by population growth, these influences may be more than countered by economic shifts and effective water quality management.

Actions and Needs

Continuity in monitoring programs, continued assessments, and continued updates in criteria are all needed to maintain water quality and effectively decrease levels where levels are elevated. Additional monitoring and assessment of toxic contaminants in the non – tidal portion (Zone 1) of the Delaware River is recommended.

Summary

Trends for specific contaminants may result from regulatory restrictions on use, changes in loading rates or degradation of the contaminant in the environment, but effective management is needed to maintain water quality and efficiently decrease levels where contaminant levels are elevated.

4.2.2.2 Fish Contaminant Levels

Certain chemicals tend to concentrate (i.e., bioaccumulate) in fish to levels thousands of times greater than the levels in the water itself. The resulting concentrations in fish and the attendant health risks to those individuals who consume the fish, such as recreational and subsistence anglers, are of concern to government agencies and the public.

Description of Indicator

Bioaccumulative contaminants have been monitored over an extended period in fish fillet collected from the Delaware River. Bioaccumulation of contaminants in fish tissue is influenced by physical-chemical properties of the contaminant, fish species, age, migration, and food habits as well as other environmental factors such as season of fish sampling.

Present Status

While programs are in place to reduce the concentrations of toxic pollutants that bioaccumulate, Delaware River basin states issue "advisories" containing meal advice for consumers of recreationally caught fish to minimize the risk to human health. These advisories list the water bodies, fish species, and number of meals recommended to minimize the risk. In some cases, no consumption of any fish species from a water body or more stringent consumption guidelines for pregnant women and children is advised. These advisories are revised based upon recent fish tissue concentration data. A summary of fish consumption advisories in the Delaware River is available online.

The following websites provide additional information on state-issued fish consumption advisories: New Jersey, Delaware, New York, and Pennsylvania.

Past Trends

A number of bioaccumulative compounds are monitored in fish collected from the Delaware River. Trends will differ depending on the contaminant of interest. In the non-tidal river, mercury contamination has been a past and present cause of fish consumption advisories. Mercury in atmospheric deposition from burning of coal and other fossil fuels contaminate fish. Products containing mercury that are improperly disposed in the garbage or washed down drains end up in landfills, incinerators, or sewage treatment facilities and can contaminate fish. Mercury is converted to methylmercury by bacteria in the environment. In surface water, the bacteria are eaten by plankton and other small creatures, which in turn are eaten by small fish, then larger fish. Mercury doesn't easily leave the body of an organism, so the amount of mercury bioaccumulates in species up trophic levels. Past trends in the United States indicate that concentrations of mercury in fish decreased during the 1970s-1980s but showed no widespread trends during the 1990s -2000s. (Wentz et al., 2014) While nontidal species generally have lower concentration of PFAS when compared to tidal fish, the highest concentration of PFAS in the four species tested by the DRBC are found in nontidal small mouth bass. PFUnA in smallmouth bass collected in 2004 was elevated with a trend of significant decrease in concentration over time (MacGillivray 2021). While PFOS concentrations in fish fillet also appear to be slowly decreasing (MacGillivray 2, some species and locations within the Delaware River Basin continue to be contaminated with PFOS at levels exceeding recommended regional risk advisory limits on fish consumption. A review by Land et al., 2018 of time trends in biota found insignificant changes in PFOS concentrations in North America.

Future Predictions

Given the lasting nature of many fish tissue contaminants, it is reasonable to presume that concentrations will remain relatively constant. For many compounds, even the effects of regulatory water quality management efforts will likely take decades to be reflected in tissue concentrations.

Actions and Needs

Pollution minimization efforts are necessary to bring about the needed reductions in tissue concentrations. Cooperative efforts among state and federal agencies and other partners to reduce emissions of bioaccumulative contaminants to the Delaware River should continue and be expanded.

Summary

Trends for specific contaminants may result from regulatory restrictions on use, changes in loading rates or degradation of the contaminant in the environment. Trajectories for contaminant reduction in fish may be long depending on the contaminant of concern, but effective management is needed to facilitate these trajectories.

4.2.2.3 Emerging Contaminants

Emerging contaminants (ECs) or contaminants of emerging concern are substances that have entered the environment through human activities, may have new or changing information that increases the level of concern, and pose a real or perceived threat to the environmental. Current regulatory approaches are inadequate to address these contaminants and the increasing public concern over their environmental and human health implications.

Description of Indicator

Thousands of new chemicals are created and used in commerce each year. Keeping pace with the knowledge of the risk in water to human and ecological health of these substances is challenging. A growing body of information on the adverse effects of ECs, as well as improved analytical detection methods, have caused increased interest and concern about how these substances impact our water resources. Perfluoroalkyl and polyfluoroalkyl substances (PFAS) and pharmaceuticals and personal care products (PPCP) are two very different classes of ECs that can illustrate the challenges to understand their presence, sources, pathways, persistence and how they degrade in the environment. PFAS are found in a variety of industrial and household products. Human and wildlife exposure to PFAS is widespread. Increasing evidence suggests adverse effects of PFAS to human health and the environment. Human health risks from exposure through drinking water and fish consumption are areas of concern. Therefore, understanding occurrence and exposure risk is important to protect water resources. A wide range of pharmaceutical and personal care products (PPCP) including prescription medicines, over the counter medicines (OTC), antibiotics, and anti-bacterials including active pharmaceutical ingredients considered essential for maintaining or improving health are entering the environment. Until recently, the fate and transport of many common PPCP were not of great concern. However, some of these synthetic compounds may ultimately pose harm to the environment. PPCP are of concern because of biological effects, resistant to degradation, widespread and increasing use, and wastewater treatment plants not designed to remove them.

Present Status

The DRBC, basin states, USEPA and others are working to increase knowledge of ECs within the Delaware River Basin. The work is generally focused on the urbanized tidal areas in the Delaware River Basin. Some links to those efforts are listed here.

- Per- and Polyfluoroalkyl Substances (PFAS) | US EPA
- Contaminants of Emerging Concern including Pharmaceuticals and Personal Care Products | US EPA
- Contaminants of Emerging Concern | Delaware River Basin Commission (nj.gov)
- PFAS Information | NJDEP - Division of Science and Research
- Contaminants of Emerging Concern | PADEP (pa.gov)

Actions and Needs

Because of concerns about potential effects of ECs on human health and aquatic life, future work should evaluate the sources as well as the fate and effects of ECs in the Delaware River water column, sediments and biota.

Summary

Potential risks from ECs should be continually updated with recent occurrence data and current science. Risks to designated uses in Delaware River include the potential of aquatic ecotoxicity from ECs to impair maintenance and propagation of fish and aquatic life and the potential of human health effects from ECs to impair source water and/or fish consumption. Monitoring and assessment tends to emphasize well-known and regulated chemicals. ECs in aquatic environments require very different monitoring tools than pollutants that are acutely toxic to wildlife and humans. Chemical-specific approaches cannot efficiently monitor all ECs. Low concentrations and mixtures of ECs present challenges to water quality assessment. The integration of effects-based tools, passive sampling, non-targeted chemical analysis and traditional targeted chemical analysis is a proposed strategy for monitoring and prioritization of EC. Cooperation among national and regional partners is needed along with education to address the increased interest and concern about how these substances impact our water resources.

References

ASDWA, 2019. Recommendations Report. Contaminants of Emerging Concern Workgroup. https://www.asdwa.org/wp-content/uploads/2019/05/ASDWA-ACWA-Report-on-Contaminants-of-Emerging-Concern-2019.pdf

Church, T.M., Sommerfield, C.K., Velinsky, D.J., Point, D., Benoit, C., Amaroux, D., Plaa, D., and Donard, O.F.X. 2006. Marsh Sediments as Records of Sedimentation, Eutrophication and Metal Pollution in the Urban Delaware Estuary. Marine Chemistry 102, 72-95.

Land M, De Wit CA, Bignert A, Cousins IT, Herzke D, Johansson JH, Martin JW. 2018. What is the effect of phasing out long-chain per- and polyfluoroalkyl substances on the concentrations of perfluoroalkyl acids and their precursors in the environment? A systematic review. Environ Evidence 7:4.

MacGillivray, A.R. 2013. Contaminants of Emerging Concern in the Tidal Delaware River Pilot Monitoring Survey 2007-2009. Delaware River Basin Commission. July 2012, revised August 2013. https://www.nj.gov/drbc/library/documents/contaminants-of-emerging-concernAug2013rev.pdf

MacGillivray, A.R. 2021 Temporal trends of PFAS in Delaware River Fish, USA. Integrated Environmental Assessment and Management. 17(2) 411-421.

MacGillivray, AR, DE Russell, SS Brown, TJ Fikslin, R Greene, RA Hoke , C Nally and L O'Donnell. 2011. Monitoring the Tidal Delaware River for Ambient Toxicity. Integr. Environ. Assess. Manag.: 7 (3) 466-477.

Mayura et al., 2014. An Adaptive, Comprehensive Monitoring Strategy for Chemicals of Emerging Concern (CECs) in California's Aquatic Ecosystems. Integrated Environmental Assessment and Management. 10(1): 69-77.

NJDEP SAB, 2020. Approaches for Addressing Drinking Water and Wastewater Contaminants of Emerging Concern (CECs) in a Broader Context: Identification, Ranking and Treatment Removal. https://www.nj.gov/dep/sab/sab_cec.pdf

Reif, A.G., Crawford, J.K., Loper, C.A., Proctor, Arianne, Manning, Rhonda, and Titler, Robert, 2012, Occurrence of pharmaceuticals, hormones, and organic wastewater compounds in Pennsylvania waters, 2006–09: U.S. Geological Survey Scientific Investigations Report 2012–5106, 99 pU.S Environmental Protection Agency. 1991. Technical support document for water quality-based toxics control. EPA/505/2-90-001.

U.S. Environmental Protection Agency. 2008b. National coastal condition report III—NCCR III. EPA-842-R-08-002. [cited 2010 November 20]. Available from: http://www.epa.gov/owow/oceans/nccr/

U.S. Environmental Protection Agency. 2017. Draft Aquatic Life Ambient Water Quality Criteria for Aluminum. EPA-822-P-17-001

Vilimanovic D, Andaluri G, Hannah R, Suri R, MacGillivray A R. 2020. Occurrence and aquatic toxicity of contaminants of emerging concern (CECs) in tributaries of an urbanized section of the Delaware River Watershed. AIMS Environ Sci 7:302–319

Vile, M. A., S. Neubauer, D. J. Velinsky, N. Weston. 2010. Linking Impacts of Climate Change to Carbon and Phosphorus Dynamics Along a Salinity Gradient in Tidal Marshes. USEPA Star Report https://ansp.org/~/media/Files/ans/research/pcer-de-bay/EPA_Climate-Change-Delaware-Salt-Marsh.ashx?la=en

Wentz, D.A., Brigham, M.E., Chasar, L.C., Lutz, M.A., and Krabbenhoft, D.P., 2014, Mercury in the Nation's streams—Levels, trends, and implications: U.S. Geological Survey Circular 1395, 90 p., http://dx.doi.org/10.3133/cir1395.)

Wilkinson, 2017. Occurrence, fate and transformation of emerging contaminants in water: An overarching review of the field. Environmental Pollution 231(1): 954-970

5

Sediments

Partnership for the DELAWARE ESTUARY

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Partnership for the Delaware Estuary—Host of the Delaware Estuary Program

5 Sediments

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Cover photograph by LeeAnn Haaf

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5. Sediments

5.1 Sediment Loading and Availability

Introduction

Estuaries in the Mid-Atlantic region are recognized as traps for watershed-derived sediments and it has been estimated that less than 5% of these sediments are transported to the continental shelf or deep sea (Meade 1982). The mean supply of sediment delivered to the Delaware Estuary from its watershed has been estimated to be 1–2 × 109 kg year $^{-1}$ (Mansue and Commings 1974) and is largely attributable to its three largest rivers: the Delaware River, the Schuylkill River and the Brandywine-Christina River which have mean annual discharges of 330, 77 and 19 $m^3 s^{-1}$, respectively, and together supply ~80% of the total freshwater inflow to the Delaware Estuary (Sommerfield and Wong 2011). Seasonal variations in sediment discharge to the estuary covary with freshwater inflow which peaks in March and experiences a minimum in September (Ross et al., 2015). An important feature of many estuaries, including the Delaware Estuary, is the estuarine turbidity maximum (ETM), which in the Delaware Estuary is generally located 70-115 km up-estuary (Fig 5.1.1). The ETM is a permanent feature of the Delaware Estuary and results from seaward advection of fluvial sediment combined with the landward flux of suspended sediment driven by gravitational circulation (Delaware Estuary Regional Sediment Management Plan 2013). The ETM acts as both a trap and reservoir for sediments and the total mass of sediment suspended in the water column can approach the mean annual input from the watershed (Sommerfield and Wong 2011).

Sediment plays an important role in the function of the Delaware Estuary. First, light limitation restricts phytoplankton growth despite extremely high anthropogenic nutrient inputs (Penncock 1985; McSweeney et al., 2017). Thus, reductions in turbidity could allow for an increased phytoplankton growth and enhanced vulnerability of the Estuary to anthropogenic nutrient inputs. Sediments also play an important role in the maintenance of Bay beaches and wetlands. Tidal wetlands are ecologically and economically valuable ecosystems that are threatened by climate change and their ability to adapt to rising seas is intimately tied to sediment availability (Weston 2014). Similarly, beaches are also threatened by the confluence of disruptions to sediment transport pathways and climate change. A recent study suggested that half of the world's beaches could disappear by the end of the century (Vousdoukas et al., 2020). As such, reductions in sediment inputs may compromise survival of bay beaches and wetlands. However, in the context of aquatic habitats— such as for fish and submerged aquatic vegetation— increased sediment is an ecosystem stressor.

Description of Indicator

Here, we use total suspended solids (TSS) as an indicator of water column suspended sediment concentration in the Delaware Estuary (see Boat Run data; accessible from the Delaware River Basin Commission website, also available on the Delaware Water Quality Portal). TSS refers to the dry weight of suspended particles in a sample of water that can be trapped by a filtering apparatus. TSS is comprised of both inorganic and organic material. While clear water and low TSS are generally indicative of good ecosystem health (Teodosiu et al., 2015), the decline of water column TSS may threaten the future sustainability of Delaware Bay tidal wetlands and beaches, and causes increasing phytoplankton blooms in Delaware Bay.

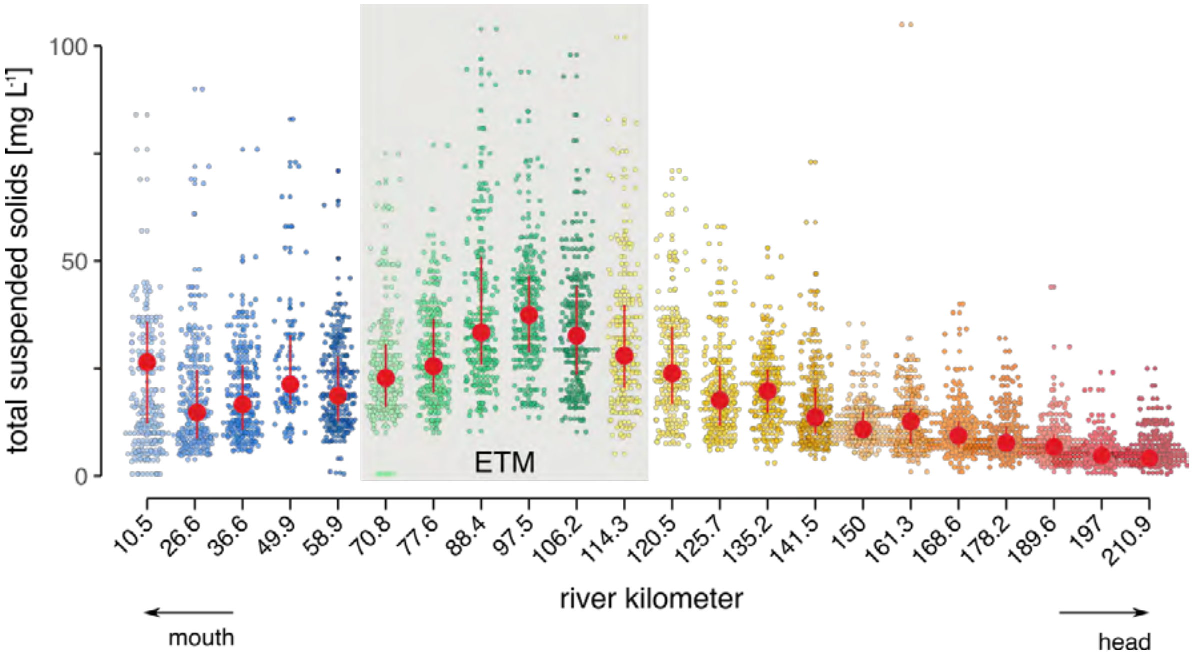

Figure 5.1.1 Spatial patterns in total suspended solids measured down the axis of Delaware Bay from 2005-2021. The estuarine turbidity maximum (ETM) is indicated at river km 70-115. Red circles denote mean at that river km, inter-quartile range is indicated by red lines. Data from DRBC 2022.

Present Status

TSS (measured from 2017-2021) varies spatially across the Delaware Estuary, from 6.33 mg L^{-1} at the most landward station monitored in the upper estuary, on the tidal Delaware River, to 8.83 mg $^{-1}$ in the most seaward station monitored where the Delaware Bay meets the ocean, with peak values at 43.0 mg L^{-1} at the estuarine turbidity maximum. The mean value measured from 2017-2021 was 19.3 mg L^{-1} (Fig 5.1.2).

Past Trends

Data suggests that suspended sediment measured as water column total suspended solids have been declining in Delaware Bay. Examination of TSS trends in stations measured from spring to fall as part of the "Boat Run" along the axis of Delaware Bay from 2005-2021 (Fig 5.1.1) suggest that suspended sediment concentration in most regions of the estuary is declining (Fig 5.1.2). On average, TSS values in 2017-2021 were found to be 16.7% lower than found in 2005-2010. Larger differences were observed at the mouth of the estuary, smaller differences were observed at the head of the estuary and minimal differences were observed at the ETM (Fig 5.1.2). This data agrees with Weston's analysis of water column suspended sediment inputs to Delaware Bay, which suggests that sediment inputs have declined from the 1950s to the present, and further that Mid-Atlantic streams are undergoing declines in suspended sediment fluxes of 2-3% year (Weston 2014). This decline in TSS may be expected based on the "cycle of urbanization" proposed by Wolman as urban development tends to result in increasing and then decreasing watershed sediment inputs (Wolman and Schick 1967). Additionally, regulations that require erosion control may be contributing to decreased sediment inputs. An additional explanation is increased accommodation space - the space available for sediment deposition - is increasing due to dredging and sea level rise. This increased accommodation space may be creating sediment sinks that lead to declines in sediment mixing over tidal cycles (Van Maren et al., 2016).

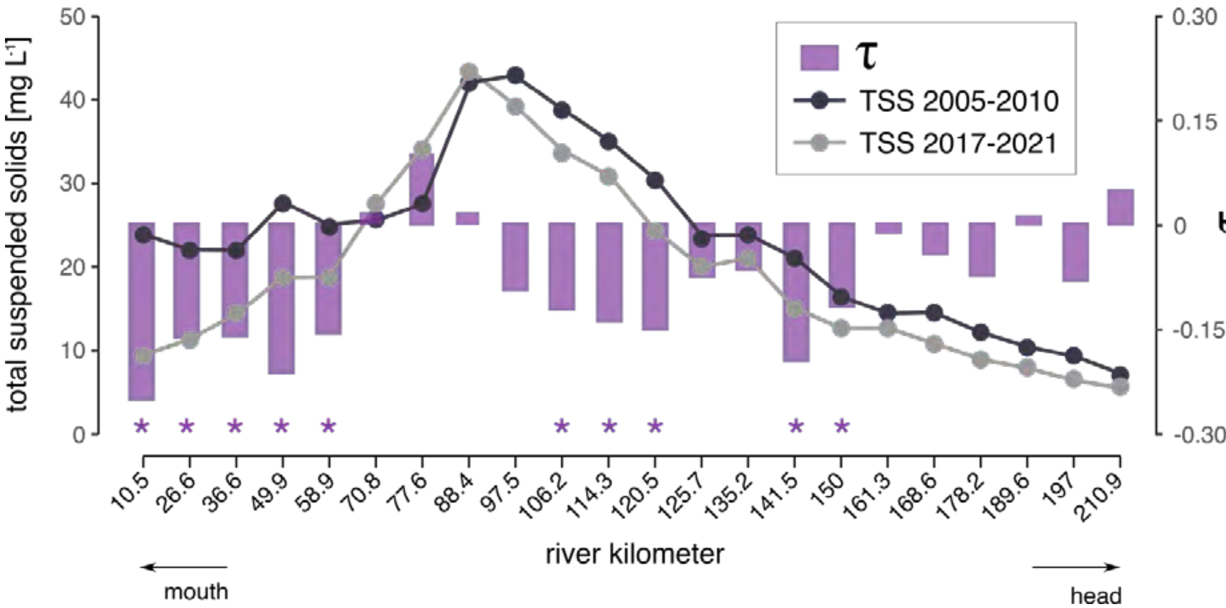

Figure 5.1.2 Differences in total suspended solids measured between 2005-2010 and 2017-2021. Generally, declines were observed between 2005 and 2021. A Mann-Kendall test was used for trend detection at each river kilometer station (R Core Team 2019; McLead 2022), where negative values for τ reflect negative trends. Statically significant trends are denoted by a star: at 17 of 22 sampling locations, TSS was lower in 2017-2021 than during 2005-2010. Negative trends were statistically significant at 10 of 17 sampling locations. No sites had statistically significant positive trends.

Future Predictions

Confounding factors may contribute to alterations in TSS trends. Increasing frequency and intensity of storms may exacerbate watershed sediment export (Chen et al. 2020). Alternatively, TSS declines may continue if watershed sediment inputs have declined due to better sediment and watershed management.

Actions and Needs

Analysis of sediment rating curves could reveal whether trends in estuarine turbidity are a function of reduced watershed inputs, altered circulation, or increased accommodation space. While Weston (2014) and Kauffman et al. (2011) have reported on trends in TSS and sediment discharge, neither have completed an analysis of sediment rating curves to detect changes in slope. An analysis of rating curves, might tell us if similar sized flow events are now carrying less sediment than in the past. Additionally, looking at trends in TSS in areas that are developing rapidly may help determine the effects of land use change on estuarine turbidity. Finally, analysis of historic satellite imagery, which dates to the 1970s, might also reveal longer term trends (or lack thereof) that are not revealed by analyzing about 15 years of sediment monitoring data.

Perspectives on Diversity, Equity, Inclusion, and Environmental Justice

Sediment inputs indirectly affect people because sustaining beaches and wetlands depend on sediment inputs and because high sediment inputs degrade quality in streams. Interactions between environmental equity and sediment are described in sections 5.3 and 4.4.

Summary

Here we assess estuarine turbidity through examination of trends in TSS from data collected from 2005-2021. These data suggest that suspended sediment concentrations in the estuary have declined 1.4% per year from 2005 to 2021. Trends in the future will depend on the interplay between increasing storm intensity which may increase suspended sediment discharge vs. improved sediment management practices which may decrease TSS. While declining turbidity may be beneficial for fish and submerged aquatic vegetation, the rapidity by which loads are declining may threaten the survival of beaches and tidal wetlands, which depend on suspended sediment to build elevation with sea level rise.

References

Chen, C. N., Tfwala, S. S., & Tsai, C. H. (2020). Climate change impacts on soil erosion and sediment yield in a watershed. Water, 12(8), 2247.

Delaware Estuary Regional Sediment Management Plan. (2013). Sediment Quality and Dynamics.

http://www.state.nj.us/drbc/library/documents/RSMPaug2013final-report.pdf

Delaware River Basin Commission. (2022). Delaware Water Quality Portal. http://demac.udel.edu/waterquality/ Accessed online 24 May 2022

Kauffman, G. J., Homsey, A. R., Belden, A. C., & Sanchez, J. R. (2011). Water quality trends in the Delaware River Basin (USA) from 1980 to 2005. Environmental monitoring and assessment, 177(1), 193-225.

Mansue, L.J. and A.B. Commings, (1974). Sediment Transport by Streams Draining into the Delaware Estuary. U.S. Geological Survey Water-Supply Paper 1532-H.

McLeod, A.J. (2002). Kendall: Kendall Rank Correlation and Mann-Kendall Trend Test. R package version 2.2.1. https://CRAN.R-project.org/package=Kendall

McSweeney, J. M., Chant, R. J., Wilkin, J. L., & Sommerfield, C. K. (2017). Suspended-sediment impacts on light-limited productivity in the Delaware estuary. Estuaries and Coasts, 40(4), 977-993.

Meade, R. H. (1982). Sources, sinks, and storage of river sediment in the Atlantic drainage of the United States. The Journal of Geology, 90(3), 235-252.

Pennock, J. R. (1985). Chlorophyll distributions in the Delaware estuary: regulation by light-limitation. Estuarine, Coastal and Shelf Science, 21(5), 711-725.

R Core Team (2021). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Ross, A. C., Najjar, R. G., Li, M., Mann, M. E., Ford, S. E., & Katz, B. (2015). Sea level rise and other influences on decadal-scale salinity variability in a coastal plain estuary. Estuarine, Coastal and Shelf Science, 157, 79-92.

Sommerfield, C. K., & Wong, K. C. (2011). Mechanisms of sediment flux and turbidity maintenance in the Delaware Estuary. Journal of Geophysical Research: Oceans, 116(C1). https://doi.org/10.1029/2010JC006462

Teodosiu, C., Robu, B., Cojocariu, C., & Barjoveanu, G. (2015). Environmental impact and risk quantification based on selected water quality indicators. Natural Hazards, 75(1), 89-105.

Van Maren, D. S., Oost, A. P., Wang, Z. B., & Vos, P. C. (2016). The effect of land reclamations and sediment extraction on the suspended sediment concentration in the Ems Estuary. Marine Geology, 376, 147-157.

Vousdoukas, M. I., Ranasinghe, R., Mentaschi, L., Plomaritis, T. A., Athanasiou, P., Luijendijk, A., & Feyen, L. (2020). Sandy coastlines under threat of erosion. Nature climate change, 10(3), 260-263.

Weston, N. B. (2014). Declining sediments and rising seas: an unfortunate convergence for tidal wetlands. Estuaries and Coasts, 37(1), 1-23.

Wolman, M. G., & Schick, A. P. (1967). Effects of construction on fluvial sediment, urban and suburban areas of Maryland. Water Resources Research, 3(2), 451-464.

5.2 Tidal Wetland Sediment Accumulation

Introduction

At more than 65,000 ha, tidal wetlands are a characteristic habitat of the Delaware Estuary (Haaf et al., this volume), cover about one-third of its area, and thus play an important role in coastal sediment budgets (Stammermann and Piasecki 2012). In Delaware Bay, terrestrial sediments enter the basin and accumulate in tidal wetland or open water sedimentary environments. Coastal sediment accumulation is a critical factor in determining tidal wetland survival relative to climate change. If marsh sediment accumulation rates are exceeded by that of tidal flooding increases associated with sea level, the marsh falls in elevation relative to the tides and may convert to open water (Orson et al. 1985). Conversely, if marsh sediment accumulation matches or exceeds rates of tidal flooding increases, inundation will not increase, and the marsh can survive within its existing footprint. Thus, sediment accretion is an important factor in predicting coastal habitat transitions in response to climate change (Kirwan et al. 2010).

However, the role of sediment accretion in tidal wetland survival under current and projected climate change is nuanced. First, the rate of sediment accumulation and the rate of marsh elevation change can be partially decoupled (Nolte et al. 2013). A marsh may accumulate large volumes of sediment, but still lose elevation relative to the tides due to subsurface consolidation and organic matter decomposition. In some portions of the Estuary, organic matter plays an important role in marsh accumulation (Haaf et al., 2022), so decreased vegetation health could be the driver of elevation loss or accumulation deficit. Secondly, the positive feedback between inundation and sediment accumulation means that high rates of accumulation can also be treated as a sign of a rapidly submerging marsh, rather than one that is resilient to climate change (Ganju et al. 2015). Lastly, a recent analysis of marsh elevation and habitat loss in the region suggests that higher elevation marshes are being replaced by open water even more rapidly than low elevation marshes (Elsey-Quirk et al. 2022). This suggests that high rates of sediment deposition, and high elevation, do not prevent marsh loss due to open water conversion.

A further dimension of marsh sediment accumulation focuses on the ability of tidal wetlands to sequester significant volumes of carbon and thus contribute to climate change mitigation (Drake et al. 2015). Tidal wetlands are particularly effective at doing this due to high rates of primary production, sediment deposition, and anoxic soils that limit decomposition rates, and high soil salinities that inhibit methanogenesis (Bridgham et al. 2006; Poffebarger et al. 2011). Tidal wetlands sequester more carbon than most other ecosystems including terrestrial forests (McLeod et al. 2011).

Description of Indicator

Here, we report on marsh sediment accretion and carbon accumulation as indicators of the important functions of sediment dynamics in the Delaware Estuary. Sediment accumulation is measured using the feldspar marker horizon method. A layer of feldspar, which is a dense, white-colored group of minerals that are used as flux agents in glass and ceramics industries, is laid out in a tidal marsh, and over time, soil cores are taken to assess the thickness of material that has deposited above the marker bed (Cahoon and Turner 1989). Carbon sequestration is reported as carbon accumulation measured in radiometrically dated sediment cores (^{210}Pb, ^{137}Cs) interpolated across the Delaware Estuary (Champlin et al. 2020). These rates of sediment accumulation integrate over different time intervals (yearly for marker bed; decadal for radiometric dating). While some research suggests that yearly vs. decadal measures are not directly comparable, little difference was observed for the Delaware Estuary in previous study (Champlin et al. 2020).

Sediment wetland carbon accumulation can be used to estimate the greenhouse gas mitigation function of wetlands. Here we report on sediment carbon density in tidal wetland soils, as well as estimates of the economic value of this resource using the social cost of carbon (Carr et al. 2018).

Present Status and Past Trends

Sediment accumulation rates varied throughout the Delaware Estuary, with broadly higher rates of sediment accumulation in the upper estuary, and lower rates near the mouth of the Bay (Fig 5.2.1). At some sites, sediment accretion rates were greater than rates of sea level rise, while in other locations sediment accretion rates were less than rates of sea level rise. Overall, analysis of sediment accretion data based on radiometric dating found a mean tidal wetland sediment accretion rate of 2.57 ± 2.03 kg m^{-2} yr^{1} in the Delaware Estuary (Boyd et al. 2017). Given that the Delaware Bay has 67,084 ha of tidal wetlands, this translates to a 1.7 ± 1.4 x 109 km yr^{1} of sediment deposited in tidal wetlands, which is similar to the amount of sediment delivered to the estuary from watershed sources at 1-2 x 109 kg yr^{1} (Mansue and Commings, 1974). Although it can be expected that there are other sediment sinks in the estuary, the high uncertainty in inputs and deposition rates makes it difficult to balance a sediment budget.

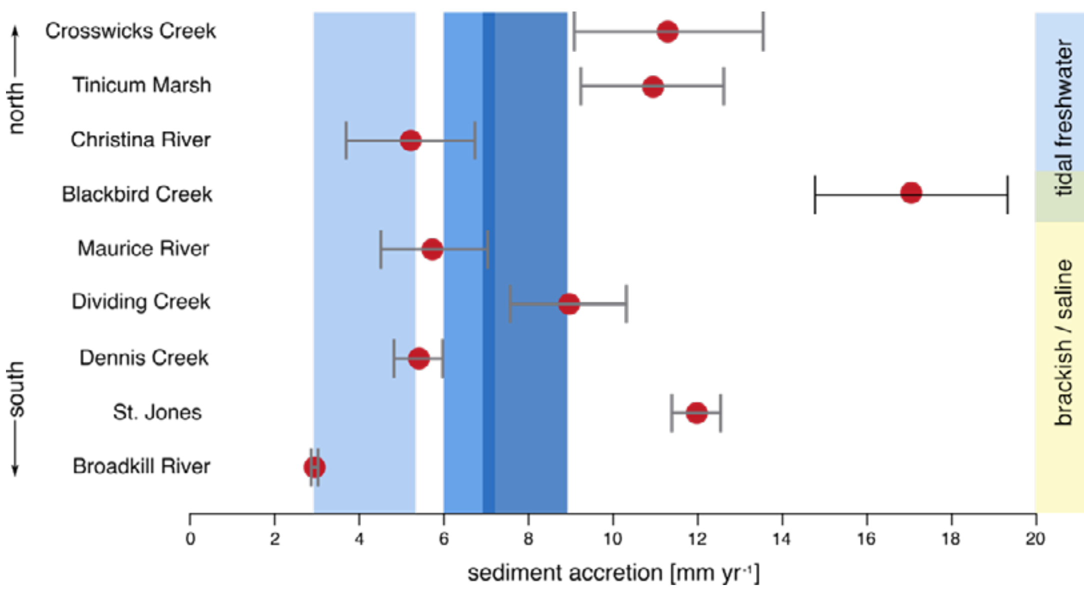

Figure 5.2.1 Sediment accumulation rates, based on marker horizons, measured in the Delaware Estuary relative to rates of long-term sea level rise based on trends from tide gauges (light blue), 19-yr sea level rise (medium blue), and 19-yr rise in mean high water (MHW) (darkest blue). Ranges in sea level rise depict variability between the different gauges. Stations are organized from north to south, and integrate measurements from several plots. Uncertainties for sediment accumulation measures are standard error. Original data found in Haaf et al. 2022; St Laurent et al., 2020. Tide gauge data is from Lewes, DE (8557380), Cape May, NJ (8536110), Reedy Point, DE (8551910), and Philadelphia, PA (8545240) (NOAA 2022a; b; c; d; e).

Interpolation mapping extrapolated to the extent of Delaware Bay wetlands suggests that tidal wetland carbon accumulation averages 127,000 ±103,200 tons of organic C yr^{-1} (based on marker bed measures) or 153,500 ± 58,600 tons of organic C yr^{-1} (based on radiometric dating) (Champlin et al., 2020) (also see Wetlands Feature 3 "A Closer Look at Blue Carbon in the Delaware Bay"). Carr and others, in a study of wetland carbon sequestration in the Delaware Estuary, estimated the annual carbon sequestration value to be $42,000 for a square kilometer of tidal wetlands, given a social cost of carbon of $37.15 per ton of CO_2, a 3% discount rate, and a 2.2% annual increase in the social cost of carbon (Carr et al., 2018). Applying this estimate over the present area extent of tidal wetlands in the estuary (estimated by Carr at 704 km^2), gives a sequestration value of $3.66 billion.

Future Predictions

A variety of factors suggest that watershed-derived sediment supply will not be a sufficient to support tidal wetlands in coming decades. Although uncertainties are high, the current sediment delivery from the Delaware watershed roughly equals the amount of sediment deposited in Delaware Estuary tidal wetlands, suggesting that the sediment supply is limited. Sea level rise rates are increasing, which causes the formation of additional accommodation space, and past epochs of rapid sea level rise have been associated with erosion of tidal wetlands in Delaware Bay (Fletcher et al., 1992). In addition, there are suggestions of declining water column turbidity (see Fig 5.1.1) and sediment supply (Weston 2014). Thus, it appears declining sediment availability may threaten the survival of tidal wetlands in Delaware Estuary. However, declining turbidity may be beneficial to other aquatic organisms, such as submerged aquatic vegetation and fish.

It is generally accepted that carbon accumulation will increase with sea level rise, as increased flooding leads to increased sediment deposition. Increasing the rate of sediment accumulation will result in increased carbon burial (Wang et al., 2019). However, increased carbon burial may be somewhat offset by marsh erosion, if the carbon stored in these wetlands is remobilized and decomposes to yield carbon dioxide or methane.

The survival of tidal wetlands with sea level rise will also depend to a degree on their capacity to migrate upslope. Coastal development may also limit the capacity of tidal wetlands to transgress (Mitchell et al., 2020).

Actions and Needs

As mentioned previously, a better understanding of sediment inputs to Delaware Bay will help ascertain whether sediment demands are outstripping supply. It is important to acknowledge and regulate sediment as both as an ecosystem stressor, and a critical component of resilience to climate change, as robust coastal sediment supplies are important for maintenance of wetlands and beaches.

To better utilize the carbon sequestration capacity of tidal wetlands, additional studies of methane emissions across Delaware will help ascertain where restoration and conservation may best promote carbon sequestration benefits. Carbon sequestration rates are highest in the fresher part of the Estuary, where methane emissions may be expected to be greater as well.

Perspectives on Diversity, Equity, Inclusion, and Environmental Justice

Wetland sediment and carbon accretion affect people because sustaining wetlands depend on sediment inputs, and carbon sequestration within wetlands mitigates climate change. Accelerating sea level rise will also reconfigure coastal areas, and attention to the human dimensions of these landscapes is necessary to ensure that the vulnerability of communities is not unfairly amplified in service to ecological resilience goals (Jurjonas and Seekamp 2020, Bhattachan et al. 2018; Van Dolah et al. 2020). Although it is increasingly recognized that disinvestment has led to disproportionate harms to some overburdened communities, which are often composed of populations that are minoritized, low-income, indigenous, rural, or otherwise lacking in opportunity for public participation. In some states, however, efforts have been made to address these wrongs. For instance, during 2020 New Jersey passed the country's strongest environmental justice bill, which seeks to reduce environmental harms in overburdened communities. It is important to consider such issues proactively, as sediment management and climate change adaptation may also lead to disproportionate impacts on overburdened communities.

Summary

This assessment of sediment and carbon accumulation in Delaware Bay tidal wetlands reveals several important findings. Sediment accumulation varies throughout the Bay with trends towards more rapid sediment accumulation in the freshwater portion of the estuary. Overall, wetland sediment accumulation roughly matches sediment supply delivered via coastal watersheds, meaning that as sea level rise accelerates in the future, sediment supply may become limiting. The carbon sequestered in tidal wetlands is a valuable resource and its further preservation can aid in carbon dioxide emissions mitigation.

References

Bhattachan A, Jurjonas MD, Moody AC, Morris PR, Sanchez GM, Smart LS, Tailliea PJ, Emanuela ER, Seekamp EL (2018) Sea level rise impacts on rural coastal social-ecological systems and the implications for decision making. Environ Sci Policy 9:122–134. https://doi.org/10.1016/j.envsci.2018.10.006

Bridgham SD, Megonigal JP, Keller JK, Bliss NB, Trettin C (2006) The carbon balance of north American wetlands. Wetlands 26(4):889–916

Boyd, B. M., Sommerfield, C. K., & Elsey-Quirk, T. (2017). Hydrogeomorphic influences on salt marsh sediment accumulation and accretion in two estuaries of the US Mid-Atlantic coast. Marine Geology, 383, 132-145.

Cahoon, D. R., & Turner, R. E. (1989). Accretion and canal impacts in a rapidly subsiding wetland II. Feldspar marker horizon technique. Estuaries, 12(4), 260-268.

Carr, E. W., Shirazi, Y., Parsons, G. R., Hoagland, P., & Sommerfield, C. K. (2018). Modeling the economic value of blue carbon in Delaware estuary wetlands: historic estimates and future projections. Journal of environmental management, 206, 40-50.

Champlin, L., Velinsky, D., Tucker, K., Sommerfield, C., Laurent, K. S., & Watson, E. (2020). Carbon sequestration rate estimates in Delaware Bay and Barnegat Bay tidal wetlands using interpolation mapping. Data, 5(1), 11.

Drake, K., Halifax, H., Adamowicz, S. C., & Craft, C. (2015). Carbon sequestration in tidal salt marshes of the Northeast United States. Environmental Management, 56(4), 998-1008.

Elsey-Quirk, T., Watson, E. B., Raper, K., Kreeger, D., Paudel, B., Haaf, L., ... & Velinsky, D. J. (2022). Relationships between ecosystem properties and sea level rise vulnerability of tidal wetlands of the US Mid-Atlantic. Environmental Monitoring and Assessment, 194(4), 1-25.

Fletcher III, C. H., Knebel, H. J., & Kraft, J. C. (1992). Holocene depocenter migration and sediment accumulation in Delaware Bay: a submerging marginal marine sedimentary basin. Marine Geology, 103(1-3), 165-183.

Ganju, N. K., Kirwan, M. L., Dickhudt, P. J., Guntenspergen, G. R., Cahoon, D. R., & Kroeger, K. D. (2015). Sediment transport-based metrics of wetland stability. Geophysical Research Letters, 42(19), 7992-8000.

Haaf, L., Watson, E. B., Elsey-Quirk, T., Raper, K., Padeletti, A., Maxwell-Doyle, M., Kreeger, D., & Velinsky, D. J. (2022). Sediment accumulation, elevation change, and the vulnerability of tidal marshes in the Delaware Estuary and Barnegat Bay to accelerated sea level rise. Estuaries and Coasts, 45(2), 413-427.

Haaf, L., D. Kreeger, and A. Homsey. 2017. Intertidal Habitats. Technical Report for the Delaware River and Basin. PDE Report No. 17-07. pages 177-195.

Jurjonas M, Seekamp E (2020) 'A commons before the sea:' Climate justice considerations for coastal zone management. Clim Dev 12(3):199-203. https://doi.org/10.1080/17565529.2019.1611533

Kirwan, M. L., Guntenspergen, G. R., D'Alpaos, A., Morris, J. T., Mudd, S. M., & Temmerman, S. (2010). Limits on the adaptability of coastal marshes to rising sea level. Geophysical research letters, 37(23).

Mansue, L.J. and A.B. Commings, 1974. Sediment Transport by Streams Draining into the Delaware Estuary. U.S. Geological Survey Water-Supply Paper 1532-H.

Mitchell, M., Herman, J., & Hershner, C. (2020). Evolution of tidal marsh distribution under accelerating sea level rise. Wetlands, 40(6), 1789-1800.

Mcleod, E., Chmura, G. L., Bouillon, S., Salm, R., Björk, M., Duarte, C. M., ... & Silliman, B. R. (2011). A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. Frontiers in Ecology and the Environment, 9(10), 552-560.

Nolte, S., Koppenaal, E. C., Esselink, P., Dijkema, K. S., Schuerch, M., De Groot, A. V., ... & Temmerman, S. (2013). Measuring sedimentation in tidal marshes: a review on methods and their applicability in biogeomorphological studies. Journal of Coastal Conservation, 17(3), 301-325.

NOAA 2022a. Sea Level trends. Access online 14 May 2022. https://tidesandcurrents.noaa.gov/sltrends/

NOAA 2022b. Lewes, DE - Station ID: 8557380 Access online 14 May 2022. https://tidesandcurrents.noaa.gov/stationhome.html?id=8557380

NOAA 2022c. Cape May, NJ - Station ID: 8536110. Access online 14 May 2022. https://tidesandcurrents.noaa.gov/stationhome.html?id=8536110

NOAA 2022d. Reedy Point, DE - Station ID: 8551910. Access online 14 May 2022. https://tidesandcurrents.noaa.gov/stationhome.html?id=8551910

NOAA 2022e. Philadelphia, PA - Station ID: 8545240. Access online 14 May 2022. https://tidesandcurrents.noaa.gov/stationhome.html?id=8545240

Orson, R., Panageotou, W., & Leatherman, S. P. (1985). Response of tidal salt marshes of the US Atlantic and Gulf coasts to rising sea levels. Journal of Coastal Research, 29-37.

Poffenbarger HJ, Needelman BA, Megonigal JP (2011) Salinity influence on methane emissions from tidal marshes. Wetlands 31:831–842

Stammermann, R., & Piasecki, M. (2012). Influence of sediment availability, vegetation, and sea level rise on the development of tidal marshes in the Delaware Bay: A review. Journal of Coastal Research, 28(6), 1536-1549.

St Laurent, K. A., Hribar, D. J., Carlson, A. J., Crawford, C. M., & Siok, D. (2020). Assessing coastal carbon variability in two Delaware tidal marshes. Journal of Coastal Conservation, 24(6), 1-16.

Van Dolah, E. R., Miller Hesed, C. D., & Paolisso, M. J. (2020). Marsh migration, climate change, and coastal resilience: Human dimensions considerations for a fair path forward. Wetlands, 40(6), 1751-1764.

Wang, F., Lu, X., Sanders, C. J., & Tang, J. (2019). Tidal wetland resilience to sea level rise increases their carbon sequestration capacity in United States. Nature communications, 10(1), 1-11.

Weston, N. B. (2014). Declining sediments and rising seas: an unfortunate convergence for tidal wetlands. Estuaries and Coasts, 37(1), 1-23.

5.3 Sediment Quality

Introduction

Estuaries are among the most productive ecosystems found globally (Underwood and Krompkamp 1999), and can also play a crucial role in the life history and development of many aquatic species. Estuaries receive anthropogenic inputs from upstream point and non-point sources and from metropolitan areas and industries located near estuaries (Chapman and Wang 2001). This pollution can not only negatively affect aquatic organisms, but also humans that harvest fish and seafood for consumption. Thus, it is critical that sediment contamination and its significance be assessed.

This assessment focuses on trace metals and metalloids as pollution indicators in the Delaware Estuary measured in the 2008-2010 benthic inventory of Delaware Bay (Kreeger et al. 2010). These metals and metalloids tend to result in decreased diversity, decreased abundance, increased mortality, and behavioral changes among benthic species (CCME 2022a;b;c;d;e;f;g). Fish consumption advisories in Delaware Bay are based on a number of contaminants, including mercury (DRBC 2022).

Description of Indicator

Here we examine sediment concentrations of arsenic, cadmium, chromium, copper, lead, mercury, nickel, and zinc as indicators of sediment pollution. We leverage a dataset collected in 2009, as part of a projected entitled the Delaware Estuary Benthic Inventory (or DEBI), that inventoried marine invertebrates, and sediment characteristics, including metal concentrations at more than 200 stations across the Delaware Estuary (Kreeger et al. 2010). Metal concentrations were measured using EPA methods 200.7 (ICP-OES), 200.8, (ICP-MS) and 245.5 (atomic adsorption). Arsenic and cadmium were measured using ICP-OES, chromium, copper, lead, nickel, and zinc were measured using ICP-MS, and mercury was measured using atomic absorption spectroscopy. Values were compared against sediment quality guidelines and Probable Effect Levels for protection of aquatic life (Table 5.3.1).

Table 5.3.1 Sediment quality guidelines (SQG) for metals and metalloids, and Probable Effect Levels (PEL) (CCME 2022a;b;c;d;e;f;g; Ingersoll et al. 2000). The lowest of freshwater and marine SQG and PELs were utilized.

Metal or metalloid	Mean value (mg kg^{-1}) Delaware Bay	Percentage of stations exceeding PEL	SQG (mg kg^{-1}) non polluted	PEL (mg kg^{-1}) polluted
arsenic	7.35	4.4%	<5.9	17
cadmium	0.45	0.89%	<0.6	3.5
chromium	24	19%	<37.3	90
copper	14	19%	<18.7	108
lead	23	16%	<30.2	91.3
mercury	0.10	15%	<0.13	0.486
nickel	13	0.89%	<20	36
zinc	96	20%	<123	271

Present Status and Past Trends

As of 2010, a significant portion of sediment samples exceeded PEL levels, including chromium (19%), copper (19%), lead (16%), mercury (15%), and zinc (20%). Values were lower for arsenic (4.4%), cadmium (0.89%), and nickel (0.89%). Generally, the sediments had the highest concentrations of pollutants in the tidal Delaware River, near Wilmington, and the lowest concentrations in Delaware Bay (Fig 5.3.1-4). Sediment cores from Oldmans Creek and Woodbury Creek, in the more urban and industrialized tidal freshwater portion of the Delaware River, were analyzed for metals from cores taken in summer of 2001. Sediment cores from accretionary environments record temporal changes in contamination. Near-surface sediments were 2-4 times as enriched in arsenic, cadmium, chromium, copper, and zinc, and 10x as enriched in lead (values for mercury and nickel were not reported) compared to depths dated prior to the 1950s (Church et al, 2006). Near-surface sediments often had lower values than peaks from deeper depths (e.g., 1970s), suggesting improvements in sediment contamination over recent decades (Church et al, 2006).

Future Predictions

Given that sediment contamination appears to have declined recently relative to the 1950s (Church et al. 2006), we may expect sediment contamination to decrease if deeper sediments are not remobilized by storms. However, emerging contaminants (PFAS, microplastics) may increase even as previously regulated pollutants decline. In addition, some metals can be mobilized by changes as pH in well (Xeng et al., 2015), meaning that ocean acidification may increase metal pollution.

Actions and Needs

Spatial and temporal analysis of additional pollutants is needed to help constrain where sediment contamination issues occur. More recent sediment data, or dated from sediment cores collected in depositional environments, can help reveal where sediment contamination is declining. Additional focus might include other compounds linked with fish consumption advisories. To assist in recovery from contamination, and to recover the ability to consumer fish from Delaware Bay, pollution sources would need to be reduced and/or contaminated sediments would need to be remediated.

Perspectives on Diversity, Equity, Inclusion, and Environmental Justice

As an indicator of environmental justice (EJ) concerns, we analyze the metal and metalloid concentrations to determine whether they are disproportionately higher in EJ communities. To determine the location of environmental justice communities, we utilized the methodology used by the PADEP, which defines an environmental justice community as a census track inhabited by a 30% or more Hispanic or non-white population and/or where 20% or more of the population lives below the poverty line, using the data from the 2015 community survey and 2010 census (PA DEP 2022; Figure 4.3.5). To look at sites across the Delaware Bay, we used a 5-km buffer around the EJ communities. We found that there was no significant difference in sediment contamination in areas bordering EJ communities versus areas not bordering EJ communities.

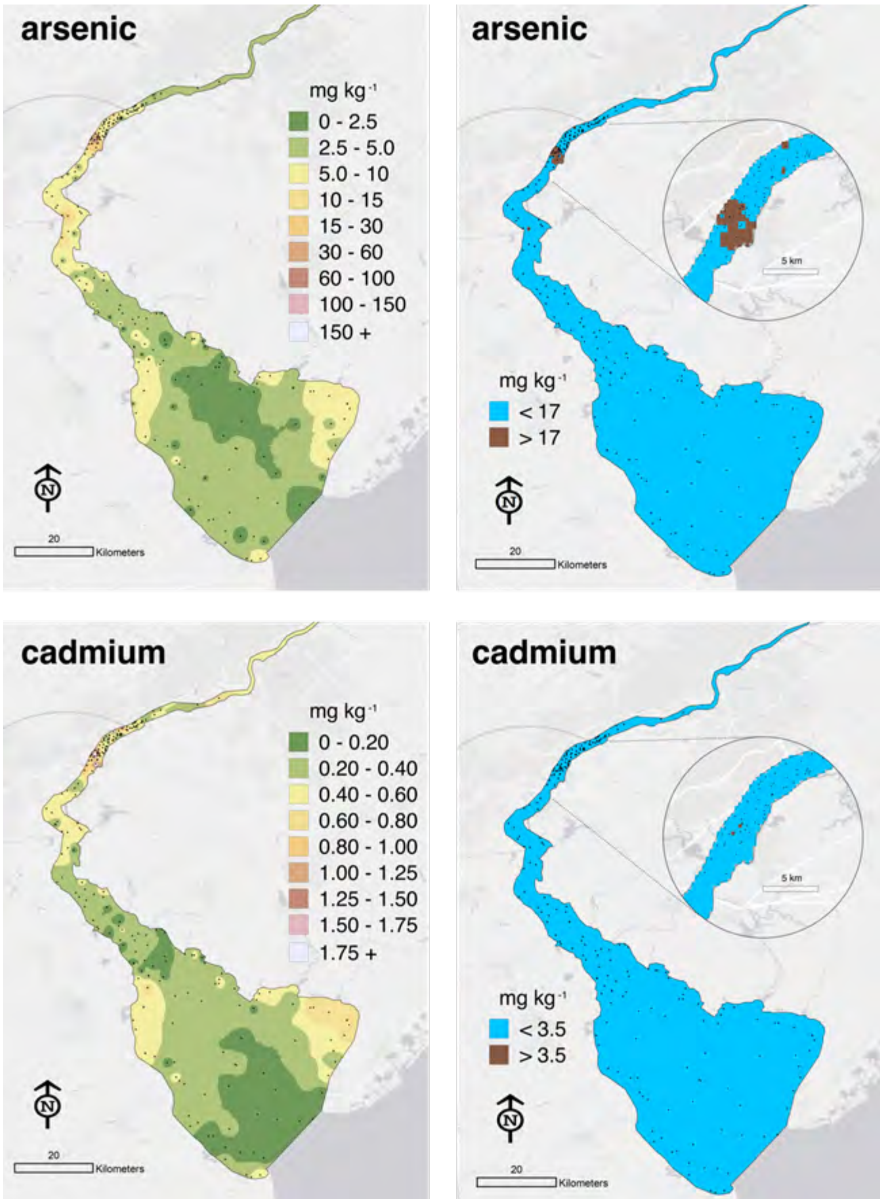

Figure 5.3.1 Interpolation maps of sediment arsenic and cadmium concentrations across the Delaware Estuary. Maps at left show metal concentrations. Maps at right show areas where sediment metal concentrations exceed Probably Effect Levels.

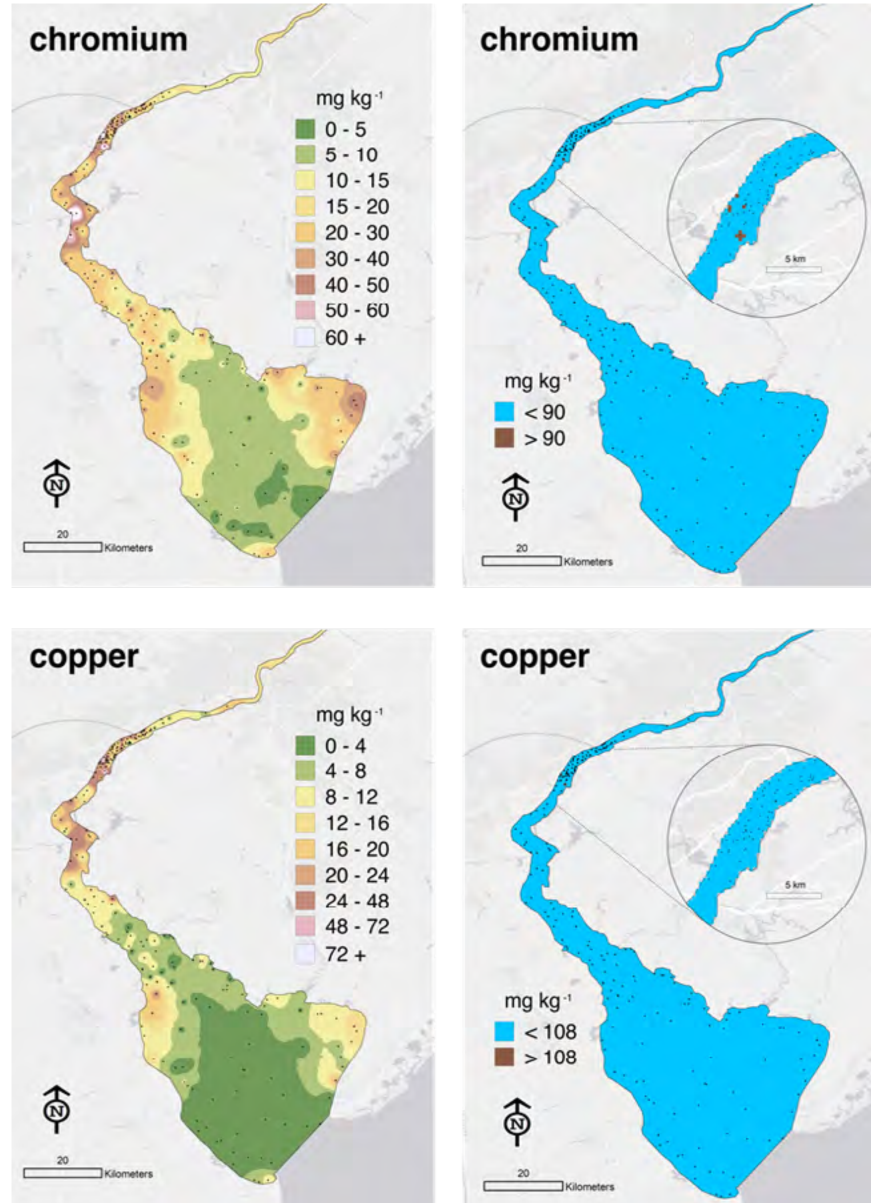

Figure 5.3.2 Interpolation maps of sediment chromium and copper concentrations across the Delaware Estuary. Maps at left show metal concentrations. Maps at right show areas where sediment metal concentrations exceed Probably Effect Levels (which are minor for chromium and for copper are non-existent).

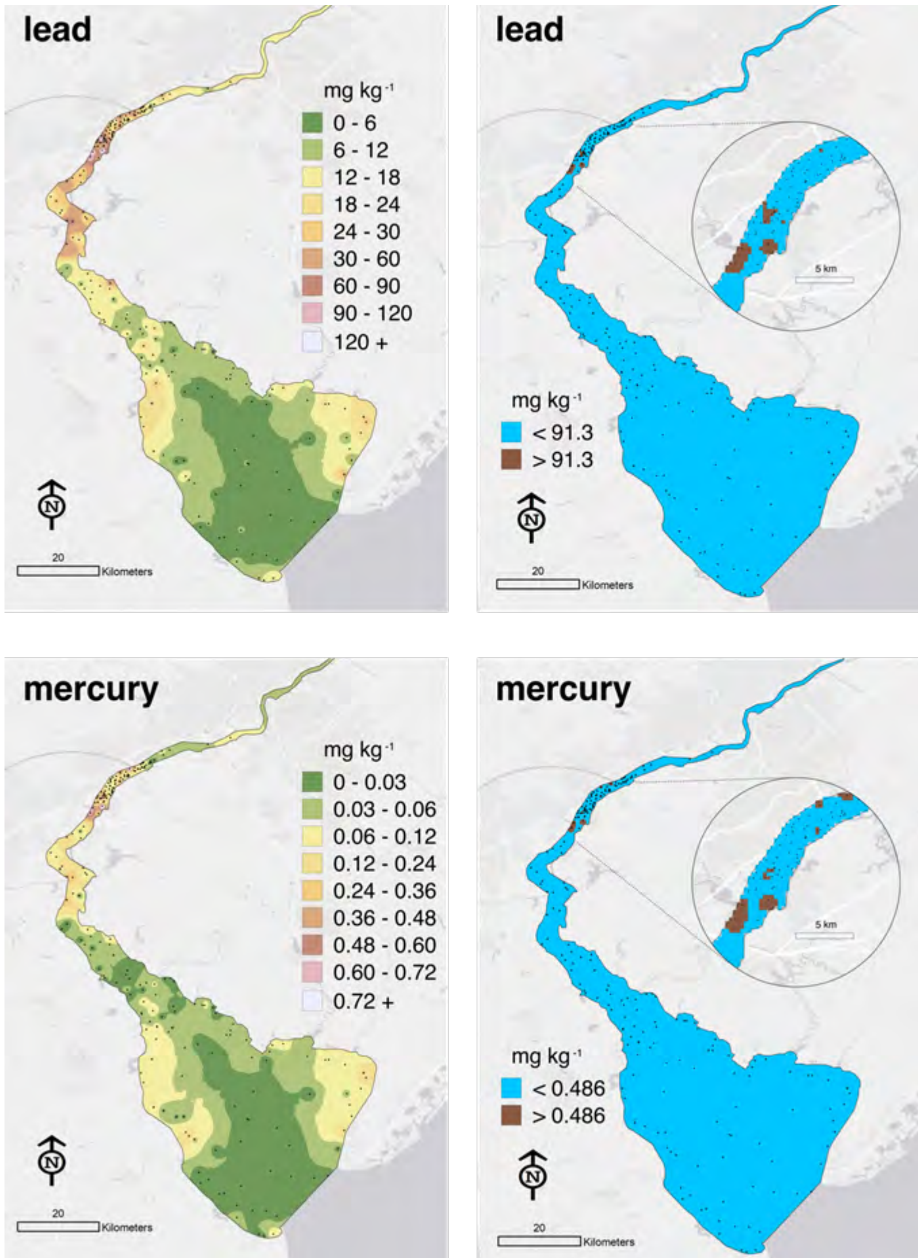

Figure 5.3.4 Interpolation maps of sediment lead and mercury concentrations across the Delaware Estuary. Maps at left show metal concentrations. Maps at right show areas where sediment metal concentrations exceed Probably Effect Levels.

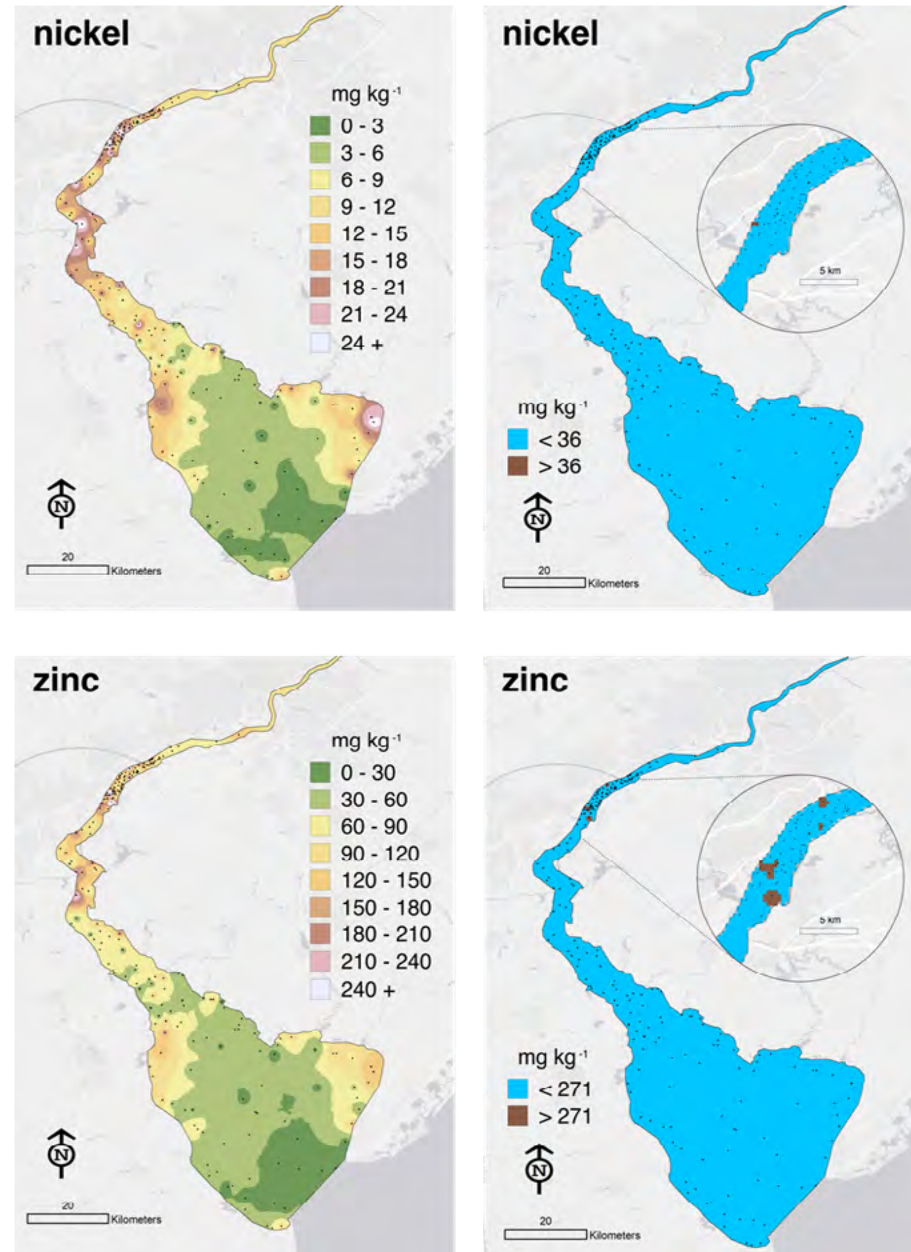

Figure 5.3.3 Interpolation maps of sediment nickel and zinc concentrations across the Delaware Estuary. Maps at left show metal concentrations. Maps at right show areas where sediment metal concentrations exceed Probably Effect Levels.

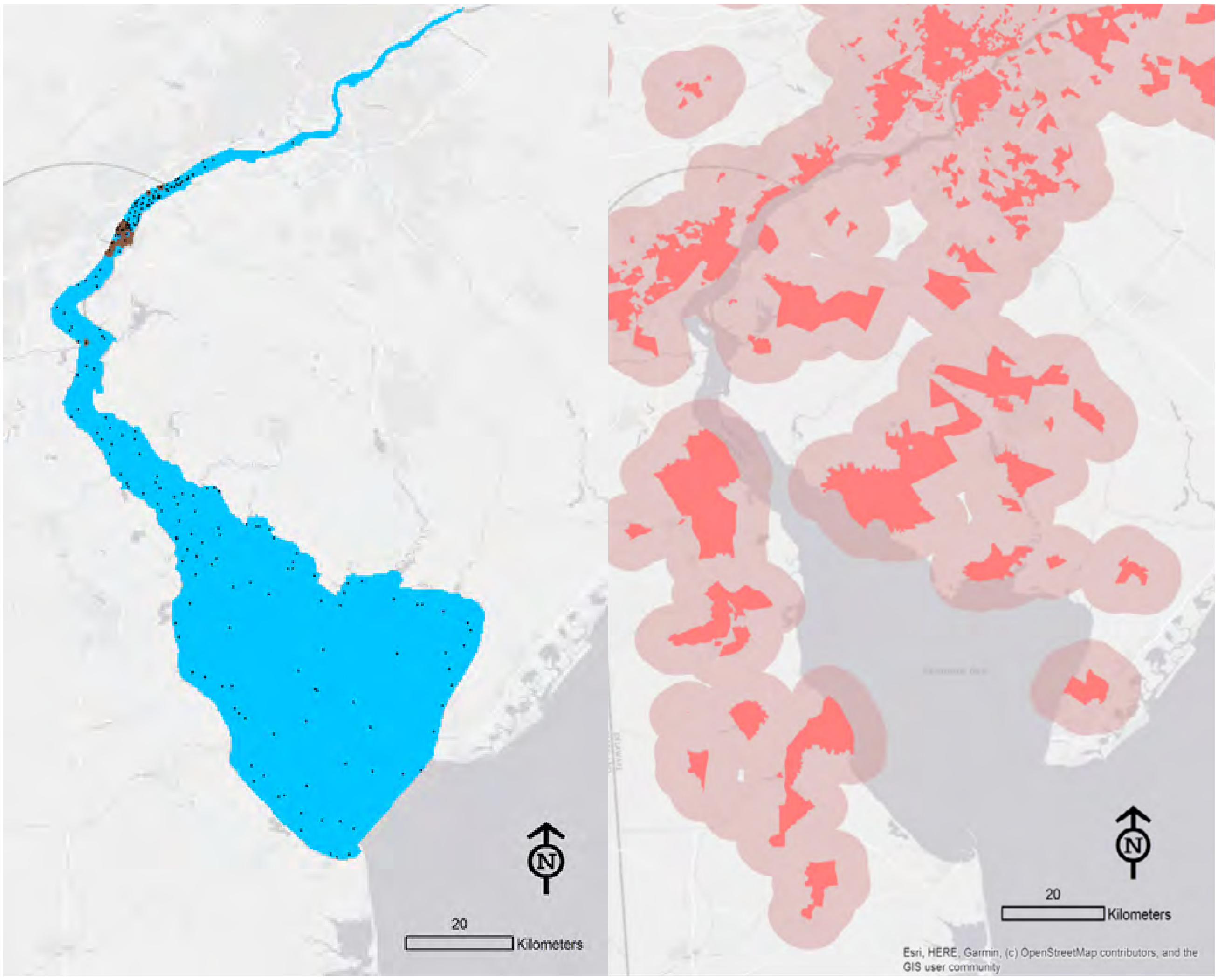

Figure 5.3.5 Location of where sediment concentrations exceed Probable Effect levels and location of environmental justice communities in the vicinity of the Delaware Estuary. Map at left shows location of PEL exceedances; the map at right shows EJ community locations (in red) and 5-km buffers (in rose).

Summary

Sediment metal concentrations were used as indicators of polluted sediments. Greater sediment pollution was found in the upper portion of the Delaware Estuary, but no significant differences was found in sediment pollution comparing waters adjacent to EJ communities vs. waters not adjacent to EJ communities. Based on analysis of dated sediment cores, sediments are more contaminated today than during the 1700s and 1800s, although sediment pollution has decreased over past decades.

References

Chapman, P. M., & Wang, F. (2001). Assessing sediment contamination in estuaries. Environmental Toxicology and Chemistry: An International Journal, 20(1), 3-22.

Canadian Council of Ministers of the Environment. (2022a). Canadian Sediment Quality Guidelines for the Protection of Aquatic Life. Arsenic. Canadian Environmental Quality Guidelines. Accessed online 11 June 2022. https://ccme.ca/en/res/arsenic-canadian-sediment-quality-guidelines-for-the-protection-of-aquatic-life-en.pdf

Canadian Council of Ministers of the Environment. (2022b). Canadian Sediment Quality Guidelines for the Protection of Aquatic Life. Chromium. Canadian Environmental Quality Guidelines. Accessed online 11 June 2022. https://www.ccme.ca/en/res/chromium-canadian-sediment-quality-guidelines-for-the-protection-of-aquatic-life-en.pdf

Canadian Council of Ministers of the Environment. (2022c). Canadian Sediment Quality Guidelines for the Protection of Aquatic Life. Copper. Canadian Environmental Quality Guidelines. Accessed online 11 June 2022. https://www.ccme.ca/en/res/copper-canadian-sediment-quality-guidelines-for-the-protection-of-aquatic-life-en.pdf

Canadian Council of Ministers of the Environment. (2022e). Canadian Sediment Quality Guidelines for the Protection of Aquatic Life. Lead. Canadian Environmental Quality Guidelines. Accessed online 11 June 2022. https://www.ccme.ca/en/res/lead-canadian-sediment-quality-guidelines-for-the-protection-of-aquatic-life-en.pdf

Canadian Council of Ministers of the Environment. (2022f). Canadian Sediment Quality Guidelines for the Protection of Aquatic Life. Mercury. Canadian Environmental Quality Guidelines. Accessed online 11 June 2022. https://www.ccme.ca/en/res/mercury-canadian-sediment-quality-guidelines-for-the-protection-of-aquatic-life-en.pdf

Canadian Council of Ministers of the Environment. (2022g). Canadian Sediment Quality Guidelines for the Protection of Aquatic Life. Zinc. Canadian Environmental Quality Guidelines. Accessed online 11 June 2022. https://www.ccme.ca/en/res/zinc-canadian-sediment-quality-guidelines-for-the-protection-of-aquatic-life-en.pdf

Church, T. M., Sommerfield, C. K., Velinsky, D. J., Point, D., Benoit, C., Amouroux, D., ... & Donard, O. F. X. (2006). Marsh sediments as records of sedimentation, eutrophication and metal pollution in the urban Delaware Estuary. Marine Chemistry, 102(1-2), 72-95.

Delaware River Basin Commission (DRBC). (2022). Fish consumption advisories. https://www.nj.gov/drbc/programs/quality/fish-consumption.html. Accessed online 13 June 2022

Ingersoll, C. G., MacDonald, D. D., Wang, N., Crane, J. L., Field, L. J., Haverland, P. S., ... & Smorong, D. E. (2001). Predictions of sediment toxicity using consensus-based freshwater sediment quality guidelines. Archives of Environmental Contamination and Toxicology, 41(1), 8-21.

Kreeger, D., A.T. Padeletti, and D.C. Miller. (2010). Delaware Estuary Benthic Inventory (DEBI) An exploration of what lies beneath the Delaware Bay and River. Partnership for the Delaware Estuary, PDE Report No. 11-06. 1 –X pp.

Pennsylvania Department of Environmental Protection [PA DEP]. (2022). PA Environmental Justice Areas. How does DEP identify Environmental Justice (EJ) areas?. Accessed online 1 June 2022. https://www.dep.pa.gov/PublicParticipation/OfficeofEnvironmentalJustice/Pages/PA-Environmental-Justice-Areas.aspx

Underwood GJC, Kromkamp J., (1999). Primary production by phytoplankton and microphytobenthos in estuaries. Adv Ecol Res 29: 3–153.

Zeng, X., Chen, X., & Zhuang, J. (2015). The positive relationship between ocean acidification and pollution. Marine Pollution Bulletin, 91(1), 14-21.

5.4 Sediment Management

Introduction

The earliest navigation improvements within the Delaware Estuary that involved dredging began in 1890 to meet the growing needs of waterborne commerce in the region. The U.S. Army Corps of Engineers has been the principal agency responsible for the construction and subsequent maintenance dredging of Federal navigation projects authorized by Congress. The first project was the construction of a 7.9 meter (26 ft) deep channel from Philadelphia to naturally deep water in the Bay in 1898. Between 1890 and 1942, the Delaware River Philadelphia to the Sea channel was incrementally deepened to 9.1 meters (30 ft), 11.0 meters (36 ft), and to a channel depth of 12.2 meters (40 ft). Congress additionally authorized the deepening of this channel to 13.7 meters (45 ft) in 1992, and this work was completed in 2020. Regular maintenance dredging will be required to maintain the channel at this depth.

Dredged material is typically placed in upland Confined Disposal Facilities (CDFs) or an open water disposal area (Buoy 10) (Fig 5.4.1),which have limited capacity and are challenging to establish. Beneficial use of sediment for beach or wetland enhancement can help reduce reliance on the current CDF facilities and can help meet needs for coastal protection and tidal wetland restoration given accelerating rates of sea level rise and coastal erosion. However, sediment contamination limits the beneficial use of sediment in some parts of the Delaware Estuary, and there is broad concern among the public about the human health and ecological consequences of dredging and dredged material placement (Dwinell et al. 2003). Elsewhere in Delaware Bay, factors such as cost, authority, and supply and demand gaps have limited its use.

Figure 5.4.1 Location of selected Confined Disposal Facilities in Delaware Bay.

Beneficial use of sediment includes adding sediment to beaches, dunes, and in some recent cases even including tidal wetlands (Ganju 2019). The goals of beach nourishment projects include the replenishment of narrowing beaches, and to prevent flooding and storm damage to adjacent infrastructure. Dredge sediment has also been used beneficially to remediate oxygen-depleted subaqueous borrow pits and to introduce sediment into the nearshore littoral system. Beach nourishment has been conducted along the Delaware Bay shore since the 1950s, although analysis of beach nourishment tracked by the American Shore and Beach Protection Association suggests that beach nourishment on the New Jersey side of the Bayshore has been extremely limited (Elko et al. 2021). In 2012, the Army Corps of Engineers released a report developing target sites for beneficial use, analyzing sediment toxicity, and coupling maintenance dredging with beach nourishment needs in Delaware, with the intention of wider implementation. Beneficial reuse focuses on matching areas with sediment generated through dredging. To match available sediment to beneficial use sites in the state of Delaware, three main data sources have been utilized: a population density and infrastructure index, social vulnerability index, and an environmental and cultural resources index (USACOE 2017; 2018).

Description of Indicator

Although sediment management occurs in various forms in the Delaware Bay, we report on beach nourishment as an indicator of demands for sediment to maintain beaches (Elko et al. 2021). Funds spent on beach nourishment were inflation adjusted to 2020 dollars. However, many projects were missing funding allocations. To estimate the amount spent on projects with missing data, we used an empirical relationship between cubic yards of sediment added to beaches and the cost for the 30 projects that had both sources of data available ($y=35.054x$; $r^2=0.979$; where y is cost and x is the cubic yards of sediment used to nourish beaches). The database we depended on for this information reported that, although New Jersey ranks first in the country in the amount of sediment added to Atlantic coast beaches (3 cubic yards for each foot of beach per year), New Jersey does not support regular nourishment of Delaware Bay beaches.

Present Status and Past Trends

Between 2010 and 2020, there were an estimated 3.6 million cubic yards of sediment placed on Delaware Bay beaches at an estimated cost of $124 million, which includes some upland sources (Fig 5.4.2). This exceeds the amount of beach nourishment performed over the time period from 1980 to 2010. Overall, we see beach nourishment increasing from the 1950s to 1980s, declines in the 1990s and 2000s, with increases over the past decade. This increase in beach nourishment between 2010-2020 is driven by two large projects, one at Broadkill Beach, and one at Prime Hook Wildlife Refuge that account for 78% of the total.

Future Predictions

Future changes to sediment management in the estuary are predicted to occur based on port expansion and climate change. Dredging to maintain the newly deepened navigation channel will require maintenance dredging. Port expansion projects and development will drive future dredging needs.

Globally, shorelines have receded over the past century globally in response to sea level rise, and this is true even where human interference is not a factor, as recession is occurring even on sparsely occupied and little developed coasts (Leatherman 1990). In the Delaware Bay, rates of shore erosion over the past century have averaged 2-3 m yr^{-1}, which is significantly greater than the U.S. Atlantic average (Maurmeyer 1979; French 1990). Marshy shorelines have supported the most rapid erosion rates, and pre-Holocene sediment deposits had slower erosion rates.

In the future, erosion rates may be expected to increase, and it appears likely that sediment demands for beach nourishment will increase concomitantly. If the aim is to retain beaches in their current position, significant beach nourishment will be required. An additional complication in the Delaware Bay is that barrier beaches help shelter tidal wetlands— which ring the bay— from erosion. Where barrier beaches have been lost over the past century, shoreline erosion can accelerate rapidly (Fig 5.4.3). If beaches along the Delaware Bayshore are not maintained, a major reconfiguration of the coast will likely result.

With some exceptions, sandy beaches faced with sea level rise transgress inland, although they may be maintained in place if sediment supply is adequate or may drown if sediment supply is constricted or sea level rise is rapid (Lorenzo-Trueba Ashton 2014). The rollover of beaches and sandy barriers subject to sea level rise is expected to occur as the barrier narrows, and is overwashed by storms. This moves sediment from the beach face to the rear of the beach or barrier, and which allows the barrier to enter a rollover phase (Leatherman 1979). This natural process of rollover can be prevented by coastal development which fixes the beach or barrier island in place.

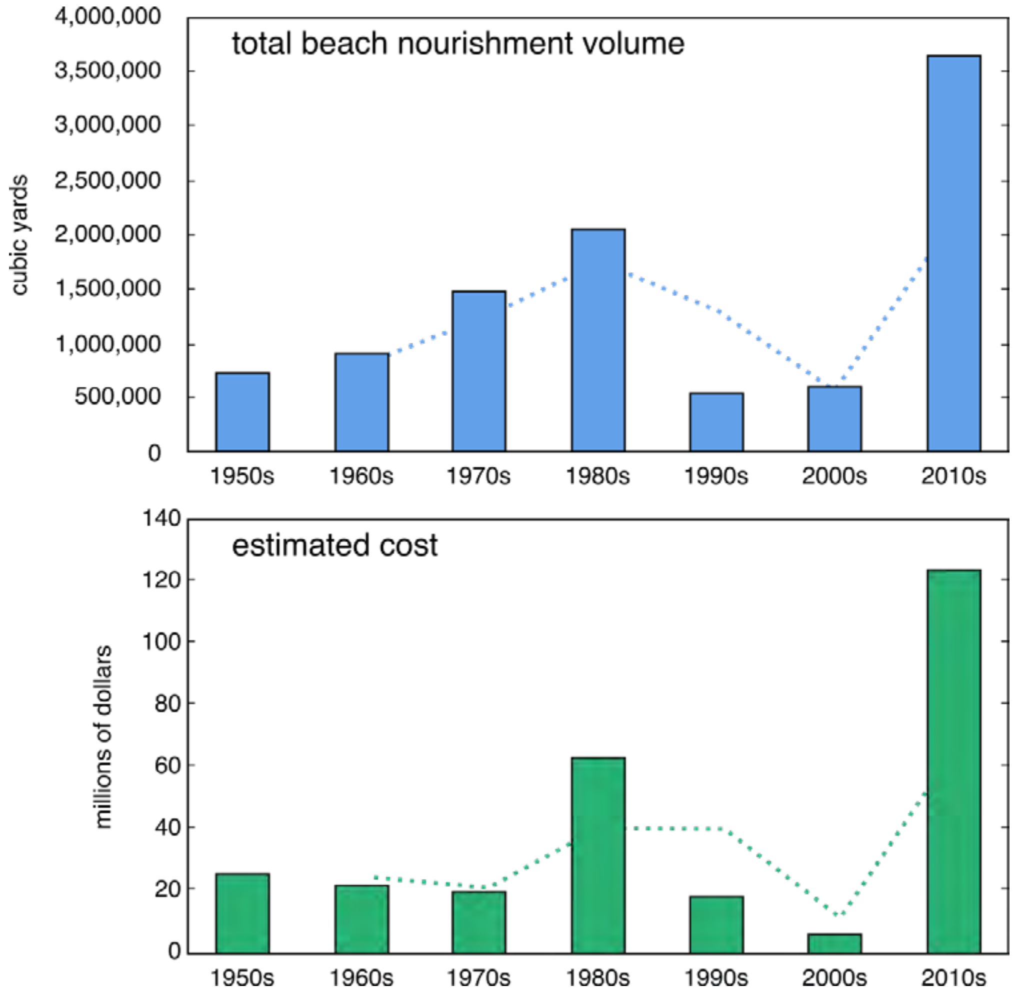

Figure 5.4.2 Estimates of total volume of beach nourishment that occurred in Delaware Bay from the 1950s-present and the cost spent on beach nourishment (Elko et al. 2021). Dashed lines show a 2-period moving average.

Actions and Needs

If beaches along the New Jersey side of the Delaware Bayshore are to be maintained the into future, they will require additional sand placement. The utilization of sediment to build marsh elevation requires further study and consideration. Specifically, it is important to consider the timeline over which plants are expected to recolonize beneficial use sites, and function of the marsh to be restored.

Perspectives on Diversity, Equity, Inclusion, and Environmental Justice

As an indicator of environmental justice concerns, we analyze the location of CDFs to determine whether they are disproportionately cited in EJ communities. Secondly, we examined the location of beach nourishment relative to these same communities. To determine the location of environmental justice communities, we utilized the methodology described above (PA DEP 2022; see Fig 5.3.5). Overall, we found that all CDFs apart from the Reedy Point site (where the C&D canal enters the Delaware Bay) were located in or adjacent to EJ communities (Fig 5.4.1). However, the entire upper stretch of the Delaware River is in or adjacent to EJ communities. For beach nourishment, we found that $186 million has been spent on beach nourishment in non-EJ communities vs. $38 million spent in EJ communities. South of the D&C canal (where beaches begin), about 46% of the shoreline is adjacent to EJ communities versus 54% adjacent to non-EJ communities so we might expect similar amount of nourishment in these two areas of shoreline, although there are specific constrains on where the State of Delaware nourishes beaches.

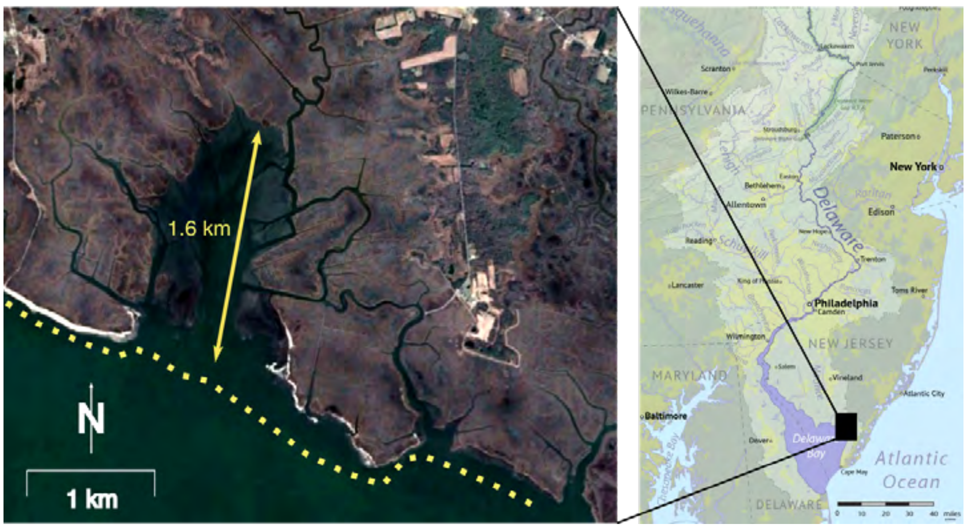

Figure 5.4.3 Shore edge erosion in an area of Delaware Bay where the beach has eroded and disappeared. This, combined with marshlands that had subsided due to levee construction, led to a rapid shoreline transgression (Smith et al. 2017). On areas to the northwest and southeast, less shoreline erosion has occurred in areas where the beach (which reflects white) has persisted, and more shore erosion has occurred in places where the beaches have disappeared. Image from 2017. Shoreline edge in 1985 is shown in dashed yellow. Imagery sources: (Wikimedia 2018; Google Earth 2022)

Summary

Overall beach nourishment and sediment management have increased over the past decade with the dredging of the Delaware Bay shipping channel to 45 feet. Beach nourishment has increased to support the maintenance of eroding Delaware beaches, although recent increases are tied to large projects at Broadkill Beach and Prime Hook Wildlife Refuge. Given climate change, additional beach nourishment may be needed if Delaware Bay beaches are to survive into the future. Navigation dredging projects of all scales may be able to provide sediment to remediate shoreline erosion through beach nourishment. In some cases, sandy beaches have been compromised or on the Delaware Bay shore, exposing marshes to wind-wave forces, which appears to enhance the rate of erosion. Several projects focused on the beneficial use of sediment to build resilience of coastal wetlands to climate change have been implemented in Delaware Bay and the US Northeast, however, questions remain about the costs and benefits of these projects, if the beneficial use of dredge sediment can sustain drowning coastal marshes on decadal and longer timescales, and how they fit into a program of coastal climate change adaptation.

References

Ganju, N. K. (2019). Marshes are the new beaches: Integrating sediment transport into restoration planning. Estuaries and Coasts, 42(4), 917-926.

Wikimedia. (2018). A map of the Delaware River drainage basin, showing tributaries and major cities/ towns. Made using USGS National Map data.

Carr, E. W., Shirazi, Y., Parsons, G. R., Hoagland, P., & Sommerfield, C. K. (2018). Modeling the economic value of blue carbon in Delaware estuary wetlands: historic estimates and future projections. Journal of environmental management, 206, 40-50.

Dwinell, D., Delorey, J., & Fade, L. (2003). The History, Development and Implementation of the Long Term Management Strategy for the Placement of Dredged Material in the San Francisco Bay Region. In Dredging'02: Key Technologies for Global Prosperity (pp. 1-15).

French, G.T. (1990). Historical shoreline changes in response to environmental conditions in West Delaware Bay. MS thesis, University of Maryland.

Google earth V 7.3.4.8642. (12 May 2022). Delaware Bay, USA. 39°11'2.30"N, 74°55'53.43"W, DigitalGlobe 2022. http://www.earth.google.com [10 June 2022].

Leatherman, S. P. (1979), Migration of Assateague Island, Maryland, by inlet and overwash processes, Geology, 7, 104–107.

Leatherman, S. P. (1990). Modelling shore response to sea level rise on sedimentary coasts. Progress in physical geography, 14(4), 447-464.

Lorenzo-Trueba, J., & Ashton, A. D. (2014). Rollover, drowning, and discontinuous retreat: Distinct modes of barrier response to sea-level rise arising from a simple morphodynamic model. Journal of Geophysical Research: Earth Surface, 119(4), 779-801.

Maurmeyer, E.M. (1978). Geomorphology and evolution of transgressive estuarine washover barriers along the western shore of Delaware Bay. PhD dissertation, University of Delaware.

PA DEP. (2022). PA Environmental Justice Areas. https://www.dep.pa.gov/PublicParticipation/OfficeofEnvironmentalJustice/Pages/PA-Environmental-Justice-Areas.aspx

Elko, N., Briggs, T.R., Benedet, L., Robertson, W., Thomson, G., Webb, B.M., Garvey, K. (2021). A Century of U.S. Beach Nourishment. Ocean & Coastal Management, 199(2021) 105406, ISSN 0964-5691, https://doi.org/10.1016/j.ocecoaman.2020.105406

Sommerfield. Mechanisms of sediment flux and turbidity maintenance in the Delaware Estuary Rating curve = Mansue, L.J. and A.B. Commings. (1974). Sediment Transport by Streams Draining into the Delaware Estuary. U.S. Geological Survey Water-Supply Paper 1532-H.

Smith, J. A., Hafner, S. F., & Niles, L. J. (2017). The impact of past management practices on tidal marsh resilience to sea level rise in the Delaware Estuary. Ocean & coastal management, 149, 33-41.

The Nature Conservancy and New Jersey Department of Environmental Protection (2021). Beneficial use of dredged material to enhance salt marsh habitat in New Jersey: Project summary and lessons learned. https://www.nj.gov/dep/dsr/wetlands/beneficial-use-dredged-material-project-summary-lessons-learned.pdf

United States. Army. Corps of Engineers. Philadelphia District. Delaware. Department of Natural Resources and Environmental Control. 2018. Delaware Beneficial Use of Dredged Material for the Delaware River Feasibility Study. https://www.nap.usace.army.mil/Portals/39/docs/Civil/Reports/DMU/DE_DMU_Main%20Report_Volume%201.pdf?ver=2018-09-19-114020-967. United States. Army. Corps of Engineers. Philadelphia District, 234 pp.

United States. Army. Corps of Engineers. Philadelphia District. New Jersey Department of Environmental Protection. 2017. Beneficial Use of Dredged Material for the Delaware River Feasibility Study. https://www.nap.usace.army.mil/Portals/39/docs/Civil/Reports/DMU/NJ-DMU-Feasibility-Report-FEB-2020.pdf?ver=2020-01-31-104457-903, 169 pp.

Sediment Feature

Beneficial Use of Sediment to Build Tidal Wetland Elevation

Metthea Yepsen

New Jersey Department of Environmental Protection

Tidal marshes are an integral part of the Delaware Estuary. They form a charismatic green band that cleans water; provides critical habitat and food sources for fish, shellfish, and birds; and buffers coastal communities from storms and erosion. However, the continued existence of many tidal wetlands is threatened by sea level rise and anthropogenic alterations. To maintain healthy salt marsh vegetation, marshes must accrete sediment and plant matter to gain elevation at a rate that keeps pace with sea level rise and subsidence. Some salt marshes are stressed and literally "drowning" because they cannot gain surface elevation at a rate that keeps pace with accelerating sea level rise.

The beneficial use of dredge sediment to build marsh elevation has been recently tested at two sites that border Delaware Bay: Fortescue Fish and Wildlife Management Area, NJ, and Prime Hook National Wildlife Refuge, DE. An increase in marsh elevation reduces inundation, promoting the growth of vegetation. The vegetation in turn stabilizes the marsh soil and promotes further accretion and increased elevation via sediment trapping and root production. It forms a positive feedback loop that may increase marsh resilience to climate change.

While both projects used dredge sediment to restore dunes, beach, and marsh to sustain wildlife, there were important differences between the two projects. In NJ, several pilot projects were conducted, including but not limited to Fortescue, to build coastal ecosystem resilience to climate change as well as build capacity and advance new sediment management concepts. In Delaware, the Prime Hook restoration was a costly and complex project that incorporated beneficial use of sediment to revegetate freshwater impoundments to sustain wildlife and protect local communities. The Fortescue test site cost $4.8 million for 6.6 acres of thin-layer placement on the marsh, 1.5 acres of beach nourishments, and 2.5 acres of dune restoration (NJDEP & TNC 2021). Prime Hook's project cost $38 million for 4,000 acres of tidal marsh. Both of these projects presented several logistical challenges, and have a mixture of successes and lessons learned. Together, however, these projects reflect a paradigm shift in sediment management, and a reversal in the prohibition of placing sediment fill material in wetlands, which was not allowed for several decades after the passage of the Clean Water Act.

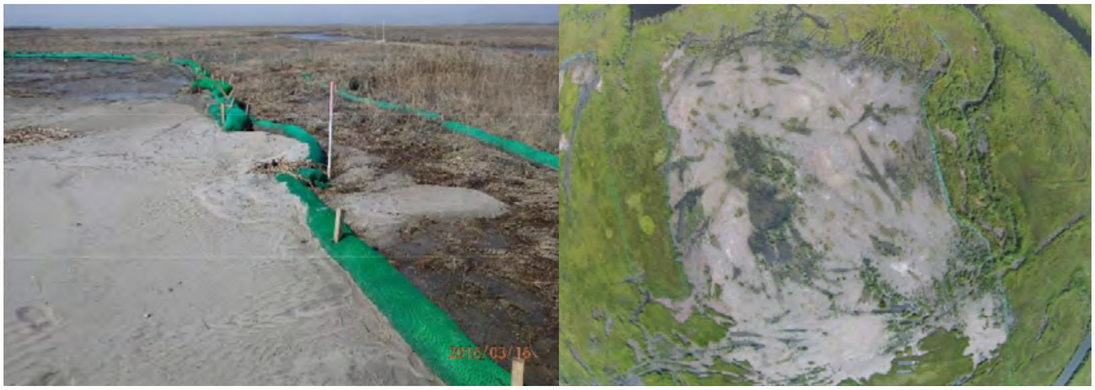

Figure 5.4.4 Close up of dredge material deposited on marshlands at Fortescue, NJ on the Delaware Bayshore (left) and landscape view (right) after placement was complete in 2016. For more information see TNC and NJ DEP 2021.

6

TECHNICAL REPORT FOR THE ESTUARY AND BASIN

Habitats

Partnership for the DELAWARE ESTUARY

December 2022 | Report No. 22-05

Partnership for the Delaware Estuary— Host of the Delaware Estuary Program

TECHNICAL REPORT FOR THE ESTUARY AND BASIN

6 Habitats

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Cover photograph by LeeAnn Haaf

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LeeAnn Haaf & Danielle Kreeger
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LeeAnn Haaf[2]
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6.4 Fish Passage 292

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6. Habitats

6.1 Forests

Abstract

Forests currently account for 49% of the land cover in the Delaware Estuary and Basin (1.6 million hectares). Forests in the Delaware Estuary and Basin are 80% deciduous, 14% mixed, and 6% evergreen. Over the period from 1996 to 2016, the total percentage of forest loss in the Estuary and Basin was over 24,000 hectares (1.5%). Future predictions indicate that forest conversion rates will continue as development continued and climate change adds extra pressure to forested ecosystems. Monitoring forested land cover is crucial to establishing restoration goals and developing actions to support good health of the Delaware Estuary and Basin.

Description of Indicator

Forests offer a myriad of ecosystem services including clean air and water protection, carbon sequestration, and climate change mitigation. Forest are critical habitats for an array of different plants and animals. Forests are also economically valuable to the timber and energy sectors. Access to forested spaces improves human quality of life through recreation, aesthetic value, a sense of place, in addition to providing other physical and mental health benefits. Sustaining these values for future generations requires collaboration among stakeholders, including state and federal agencies, landowners, industry professionals, conservation organizations, communities, and policymakers. Moreover, the collaboration between states is necessary when focusing on watershed-scale forest-related issues. In this chapter, we review the current status of forests using select metrics (from United States Forestry Service Forest Inventory and Analysis datasets; USFS FIA) and assess forest cover change using NOAA's Coastal Change Analysis Program (C-CAP) data across the Delaware Estuary and Basin from 1996-2016. We used state-specific forestry action management plans to synthesize ongoing efforts, compare state priorities, and identify possible needs for the Estuary and Basin.

Data Sources

Forest cover data were obtained through NOAA's C-CAP program through their online portal for the years 1996, 2001, 2006, 2011, and 2016 (for more information on the C-CAP dataset used here, see Chapter 1). The data from 2016 were used to determine the present status indicator for this section, with all years used to assess trends over time. C-CAP land cover data for forests were divided into broad forest type categories of evergreen forests, deciduous forests, and mixed forests (Table 6.1.1, Fig 6.1.1). For present forest conditions, modeled data layers from the USFS FIA were also used to show the distribution of more specific forest types, stand densities, stand size classes, as well as forest productivity. Additionally, for the present status, we review current state-level forested land ownership from state forestry action plans, as forest ownership has ramifications for how management actions are implemented. See Chapter 1 for a schematic representation of the Estuary and Basin assessment units and reporting hierarchy within the Delaware Estuary and Basin.

Table 6.1.1 Forest categories and type definitions

Category	Definition	Forest types
Deciduous	Dominated by tree species that drop leaves in autumn; broadleaf and/or hardwoods	Oak / hickory Maple / beech / birch Elm / ash / cottonwood Aspen / birch
Evergreen	Dominated by tree species that retain leaves through winter; conifers/softwoods and non-deciduous hardwoods	White / red / jack pine Spruce / fir Longleaf / slash pine Loblolly / shortleaf pine Exotic softwoods
Mixed	Equally composed of deciduous and evergreen tree species	Oak / pine Oak / gum / cypress

Present Status

Forest Cover

In 2016, there were 6,343 mi^2 (1.6 million hectares) of forested land within the Delaware Estuary and Basin. Forested land makes up approximately 49% of all land cover within the Estuary and Basin. Forest cover is highest (>70%) in the Upper Region watersheds, but declines to 40-70% in the Central and Lower Region watersheds, with the Bay Region watersheds typically having <30% forest cover (Figure 6.1.2).

Forests across the entire Delaware Estuary and Basin are 80% deciduous, 14% mixed, and 6% evergreen. In the Upper, Lower, and Central Regions, deciduous forests account for >75% of forest cover with <10% evergreen forests, and the remainder being mixed forests. In the Bay Region, <60% of forests are deciduous, with 30% being mixed and 13% being evergreen. Generally, forest cover is low around the Philadelphia-Camden and Wilmington corridor of the Estuary (i.e., UE1, western UE2, and southeastern LE1; see Figure 6.1.2) due to the distribution of development and urbanization.

As the Estuary and Basin span a broad latitudinal gradient, with varying physiographic characteristics (e.g., geology, soil type, altitude), forest types vary among subregions (Figs 6.1.3-6.1.4). In the mountainous, northern watersheds of the Upper Region (i.e., Catskill mountains), forest types are dominated by maple/beech/birch (*Acer, Fagus, Betula* spp) communities. Moving south towards the Pocono Mountains in the Central Region, forest communities shift to a dominance of oak/hickory (*Quercus, Carya* spp), a pattern which continues to the Lower and Bay Regions of the watershed. In UE2 and DB2 of the Bay Region, forest types also principally include loblolly/shortleaf (*Pinus* spp; a category that includes pitch pine)--these areas are associated with the New Jersey pine barrens.

Condition metrics

A suite of condition metrics are available from the USFS, but for this review, we consider productivity, stand density, diameter classes, stand ages, and ownership. Productivity indices identify potential tree growth modeled from the mean annual increment of fully stocked natural stands. Stand density indices reflect the modeled number of trees greater than 10" in diameter per acre, which can be used to surmise tree occupancy— it should be noted, however, that dense stands are not always in better ecological condition. Size and age classes reflect forest maturity, which could have an impact on forest biodiversity and resilience. Lastly, forest ownership is reviewed as a potential condition metric for forests because of the challenges associated with implementing large-scale management actions on privately-owned land.

Figure 6.1.1 Diverse forests in the Delaware Estuary and Basin include hardwood forests (A, D, G), mixed forests (B), Atlantic white cedar swamps (E), and evergreen forests dominated by pines (C, with F showing forest structure after wildfire).

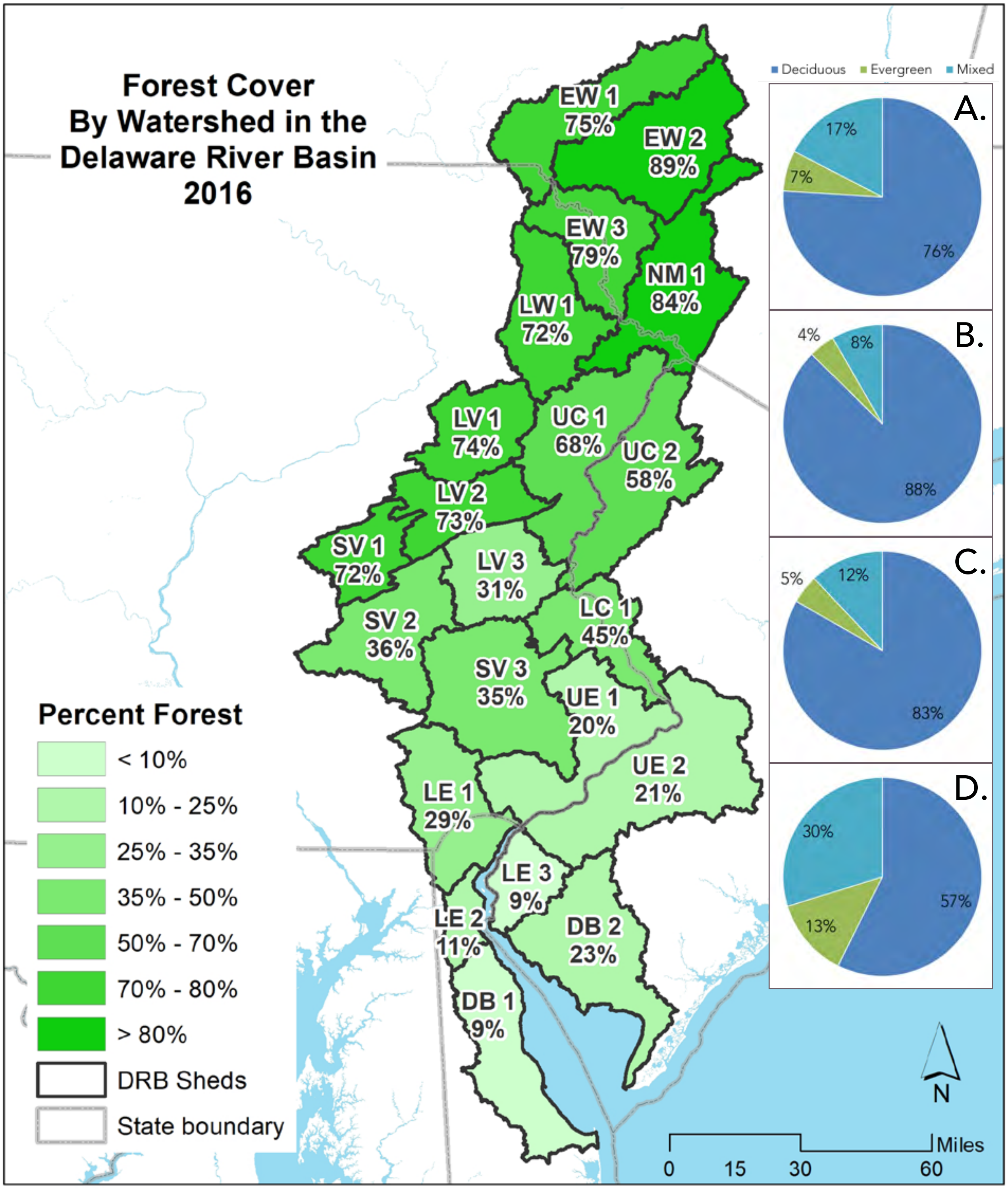

Figure 6.1.2 Forest cover in sub-watersheds of the Delaware Estuary and Basin (2016). Insets show the relative percentage of forest types in the (A) Upper (EW 1, EW 2, EW 3, NM1, LW 1), (B) Central (LV 1, LV 2, LV 3, UC 1, UC 2, LC 1), (C) Lower (SV 1, SV 2, SV 3, UE 1, UE 2, LE 1, LE 2, LE 3) and (D) Bay regions (DB 1, DB 2).

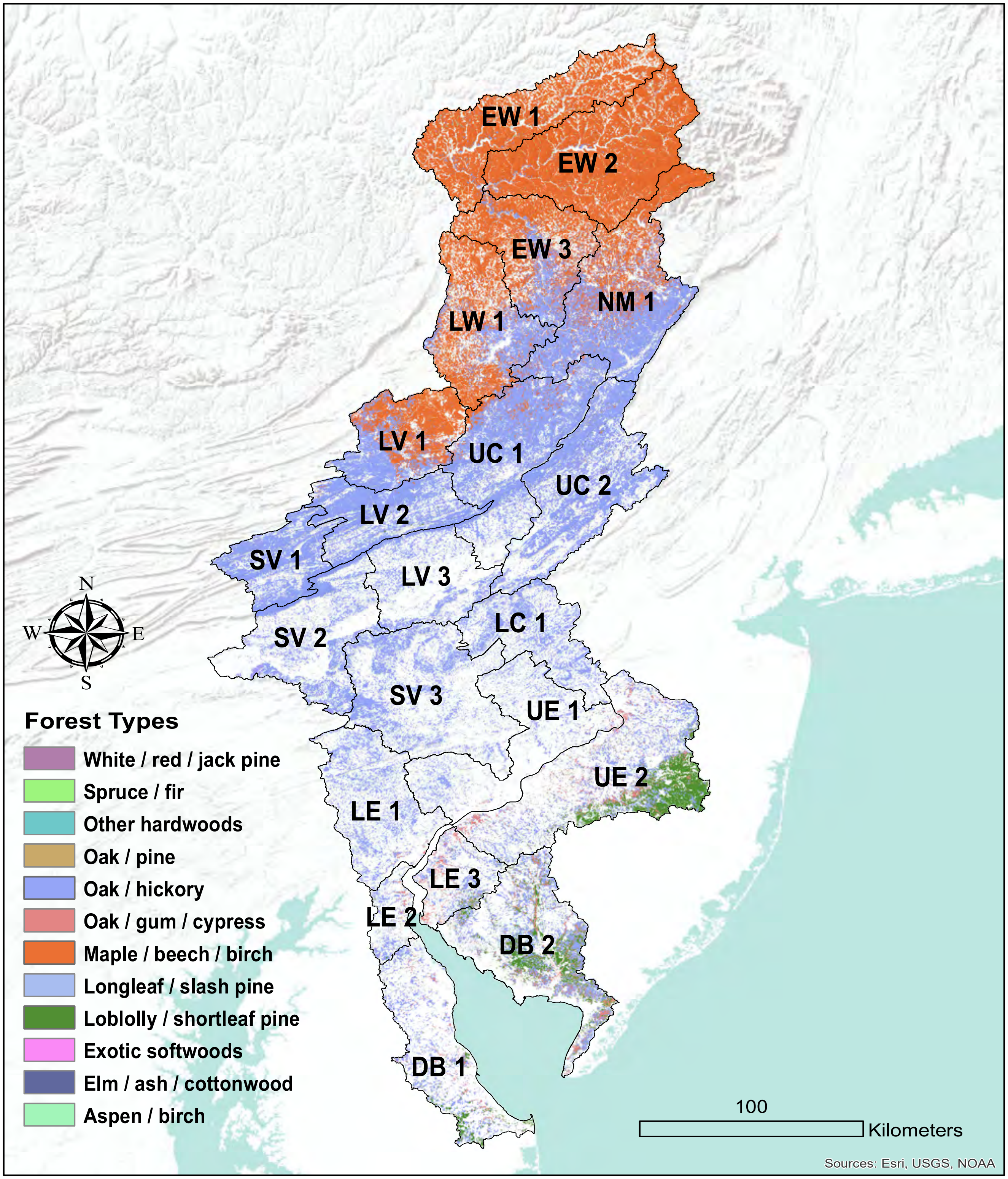

Figure 6.1.3 Distribution of forest types across the Delaware Estuary and Basin from USFS FIA datasets (2018). Forest types are predominantly maple/beech/birch in the most northern watersheds, with a majority of the remaining watersheds of the Delaware Estuary and Basin dominated by oak/hickory. Forest types also include notable coverage by oak/gum/cypress along the Delaware River (southeastern UE 1, western UE2, eastern LE 2, and western LE 3). Further, central DB 1 and eastern UE 2 also have significant cover of loblolly/shortleaf pine.

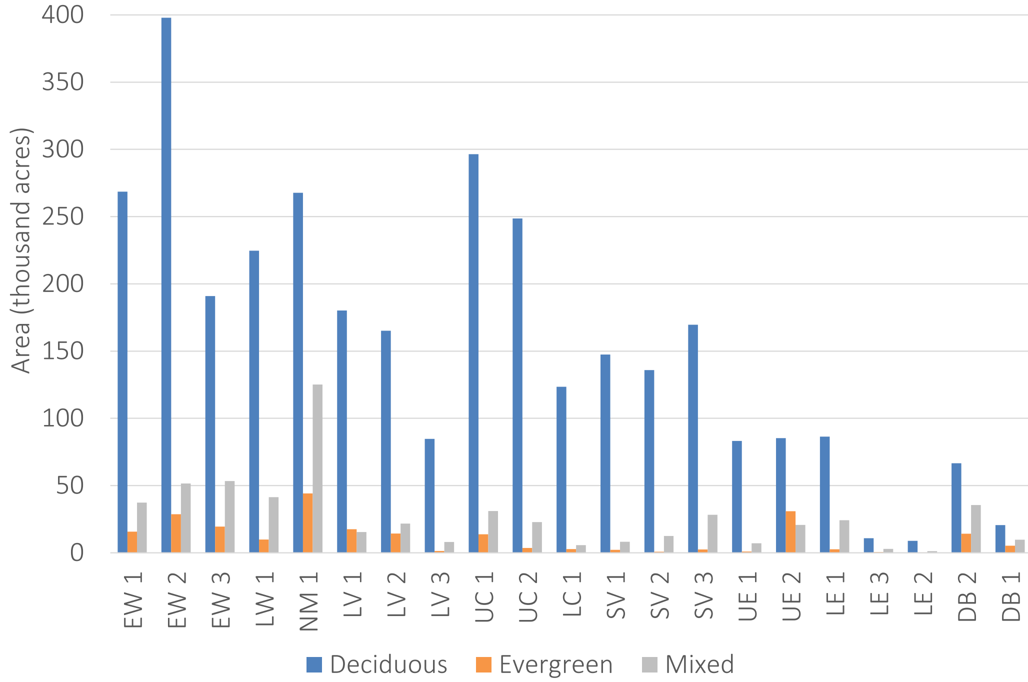

Figure 6.1.4 Forest cover by type in sub-watersheds of the Delaware Estuary and Basin (2016).

Indices for forest productivity and stand density match patterns associated with total forest cover across the Estuary and Basin (Figs 6.1.5-6.1.6). For instance, high productivity and stand densities are distributed in the Upper and Central regions where development and possible fragmentation is low. In the Philadelphia-Camden and Wilmington corridors, productivity and density indices are low, reflecting the intense urbanization there. In SV2, LV3, and northwestern parts of SV3, agriculture also dominates the landscape (see Chapter 1), and likely as a result, indices for forest productivity and density are low. In the areas of UE2 and DB2 that correspond with forests that benefit from protections of New Jersey Pinelands regulations, forest productivity and density indices are regionally high.

Across the states in the Estuary and Basin, forested lands are predominantly of large diameter classes (Fig 6.1.7). As forests grow older, they also grow larger. The majority of trees in the estuary region fall into a large diameter class, also considered “sawtimber”. The lack of diversity in size class across the region is directly related to the age structure of forests. Without the recruitment of saplings creating a heterogeneous forest structure, wildlife habitat and ecosystem services are reduced. New Jersey and Delaware, which have a less robust timber industry, are particularly vulnerable to this issue.

Across states, forests also lack diversity in age classes (Fig 6.1.8). Homogeneous forest age structures threaten the economic productivity and ecological health of forests. Forest age structure is related to time since past disturbance (Pan et al 2011). Overall, forests are maturing while regeneration rates have declined, resulting in a shortage of early successional forests as well as old-growth forests. Roughly 80 years ago, agriculture in the Northeast was largely abandoned and croplands were left to succeed into the mature forests we see today (Irland 1999, Thompson et al. 2013).

Forest ownership varied across the states within the Estuary and Basin. In New York and Pennsylvania, privately owned forests represent 74% and 70%, respectively, of each state’s forested land. In New Jersey and Delaware, privately-owned forests accounted for 48% and 47% of forested lands in each state. State-initiated management plans can face challenges when attempting to implement action items in privately-owned forested systems. For instance, states with higher private ownership of forests face greater risks

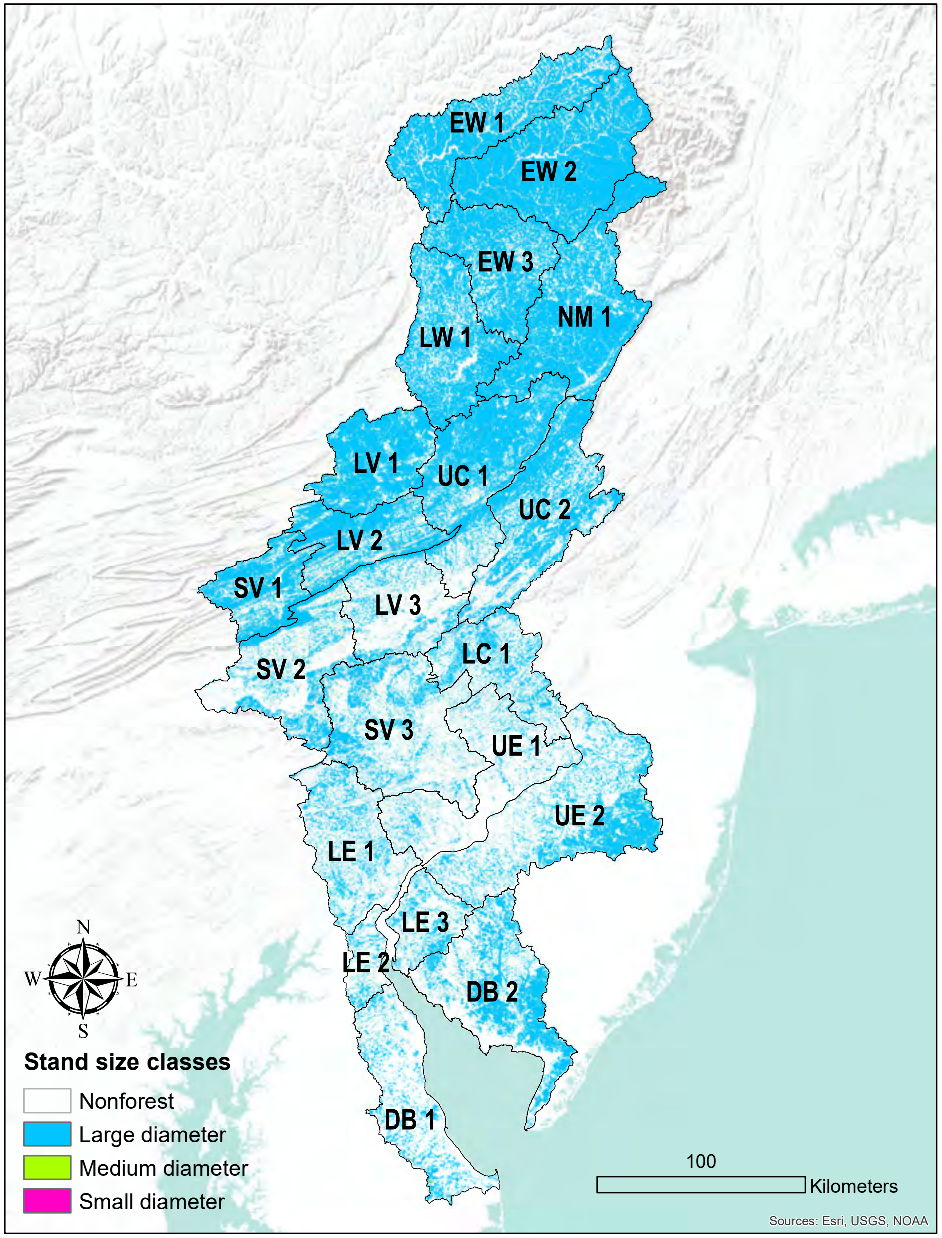

Figure 6.1.5 Mean forest stand size from USFS FIA datasets (2018). Forest stands are predominantly large diameter (blue) across the Delaware Estuary and Basin.

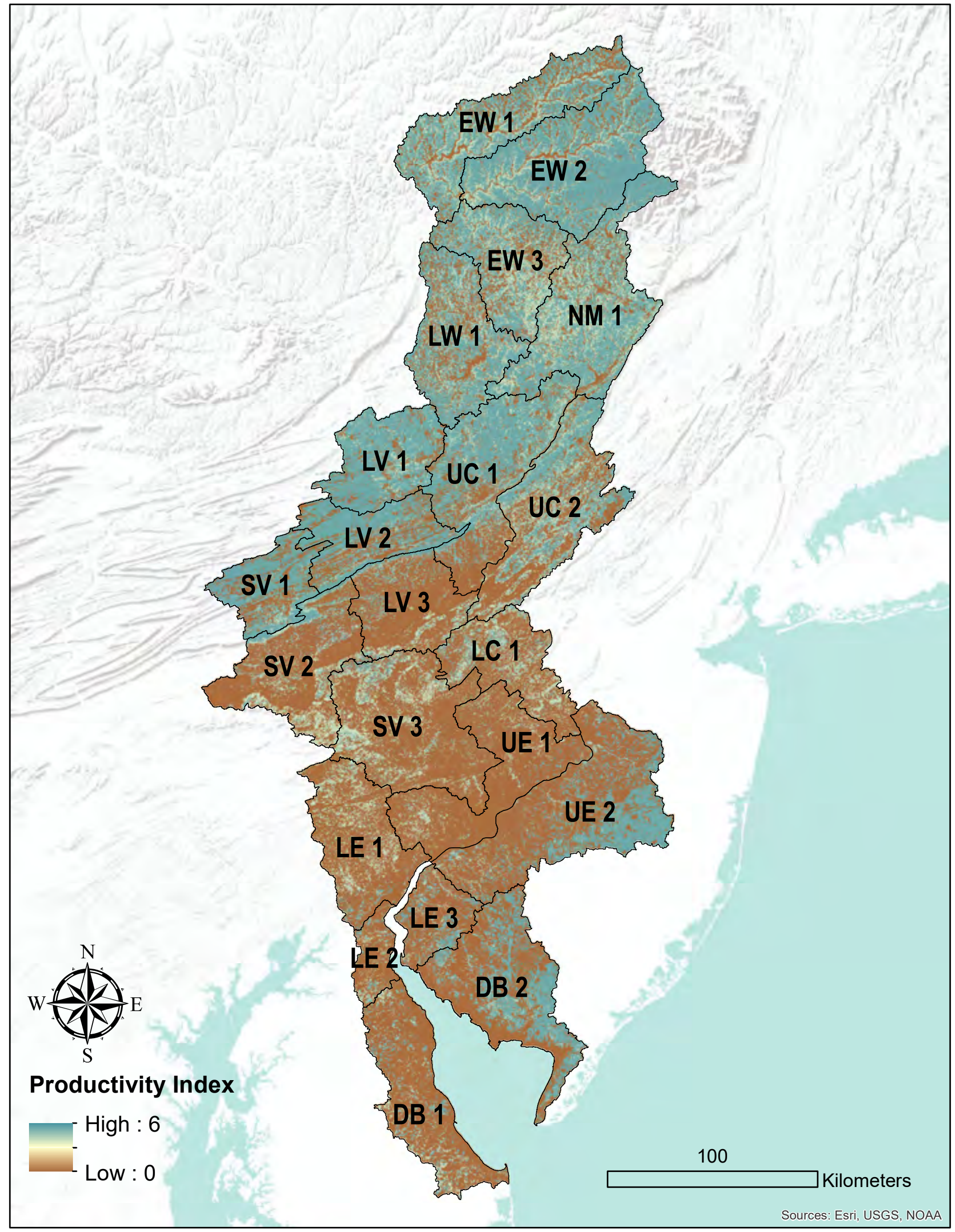

Figure 6.1.6 Modeled mean forest productivity from USFS FIA datasets (2018).

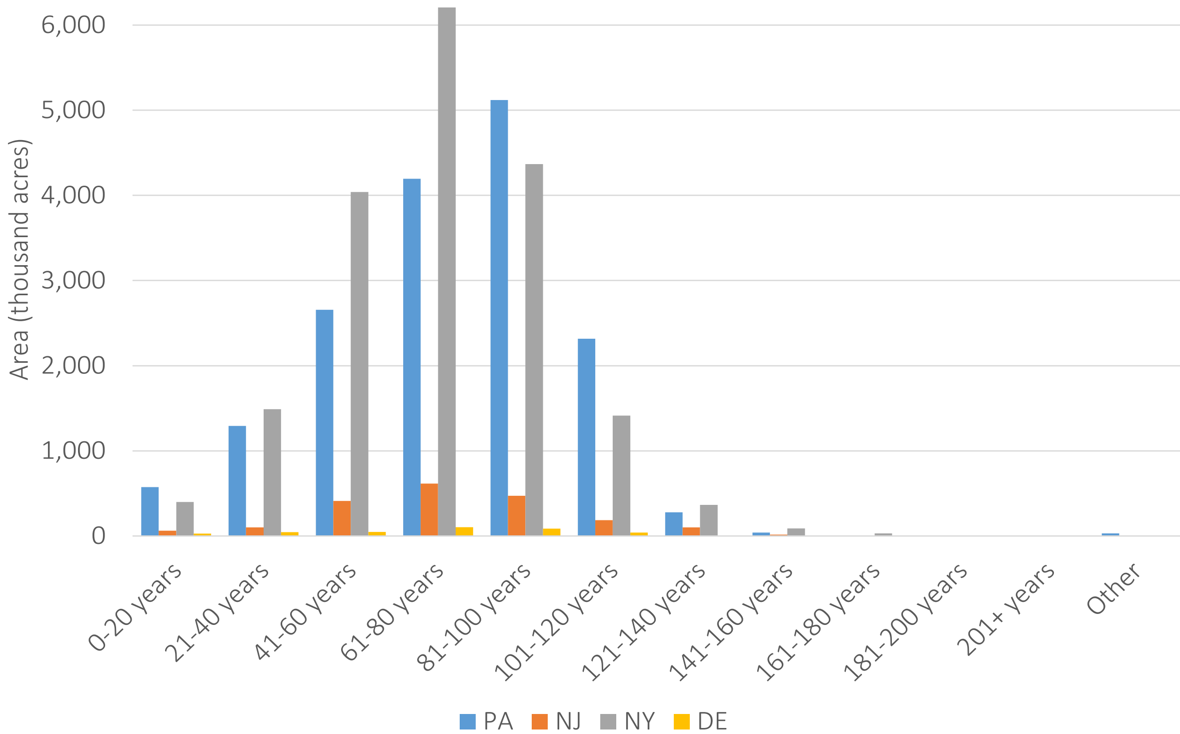

Figure 6.1.7 State-wide forest stand-age class distributions for the four major states that comprise parts of the Delaware Estuary and Basin (2019).

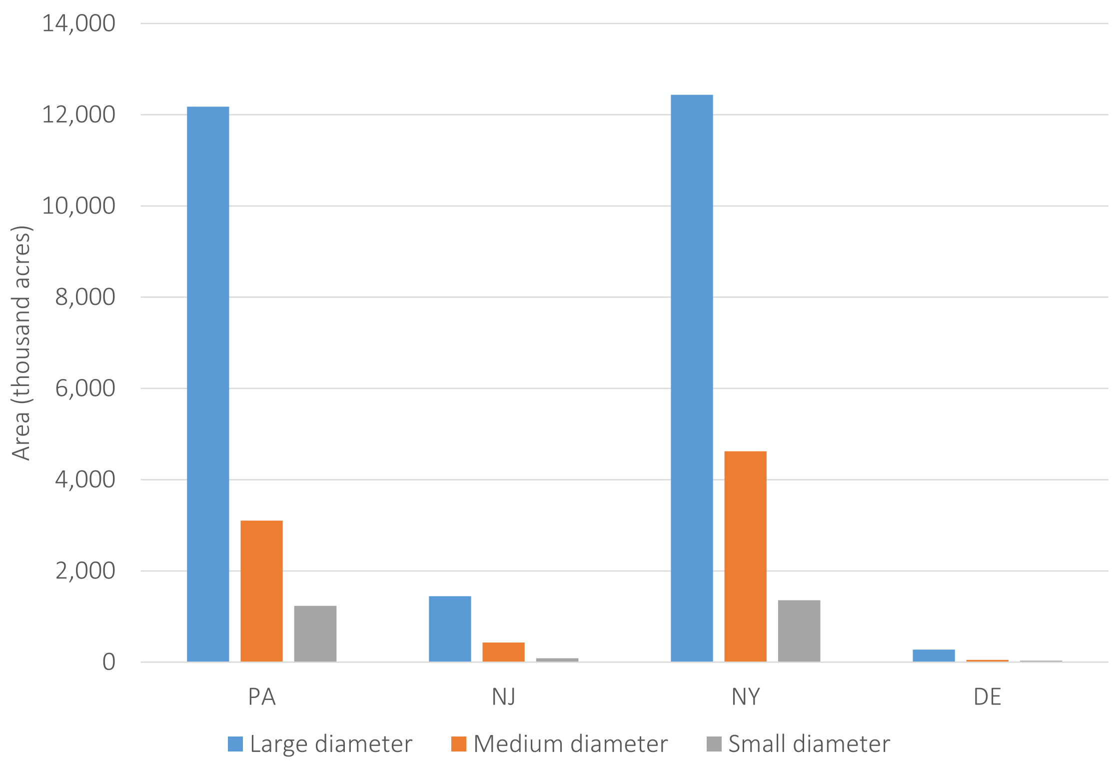

Figure 6.1.8 State-wide forest stand-size class distributions for the four major states that comprise parts of the Delaware Estuary and Basin (2019).

for lack of forest regeneration due to the high-cost burden of implementing action. In Pennsylvania, where a majority of forests are privately owned, few landowners have the resources to implement actions to support regeneration.

Trends

Current forest cover in the Delaware Estuary and Basin is likely only a fraction of what it was when only indigenous tribes managed the land. During European colonization, forest land was converted to agricultural land, a process that continued for centuries with clearing practices accelerating during the industrial age (Irland 1999, Thompson et al. 2013). As industrial technologies changed and development patterns changed in the early to mid-nineteenth century, forest clearing declined and reforestation likely increased while agricultural land in the eastern U.S. was abandoned (Irland 1999, Thompson et al. 2013). Since then, forest losses have likely remained small yet consistent over time as urban and suburban landscapes expand. Lastly, changing land-use history or anthropogenic manipulations have also contributed to landscape-wide forest community changes, such as the loss of American chestnuts through an introduced fungal pathogen and the proliferation of early-mid successional, mesic species such as red maple (*Acer rubrum*)(Irland 1999, Thompson et al. 2013).

Forest loss is a significant component in overall land cover changes in the Delaware Estuary and Basin, totally 93.5 mi^2 or 24,200 hectares from 1996 to 2016. Forest cover change is a crucial indicator of overall watershed health, as the more forested a watershed, the better condition it is in terms of habitat, water quality, and ecosystem service provision. Over the period from 1996 to 2016, the total percentage of forest loss was 1.5%. Between each year of the analysis, there were variable rates of loss, with the most significant loss coming in the most recent period (between 2010 and 2016), with the loss of 0.5% of total forest cover. The second most significant drop occurred between 2001 and 2006, at a 0.4% loss.

Figure 6.1.9 shows the forest loss in square miles between each time span (i.e., between each year of available data) of the analysis, and across the entire period (1996 to 2016). The data are distinguished between Upper Basin (non-tidal) watersheds, Estuary watersheds, and the Basin as a whole. Figure 6.1.10 presents the changes in forest cover by time span as a percentage. Changes in forest cover also varied by forest type. The C-CAP data provide details on the forest cover type, including deciduous, evergreen, and mixed forests. Figures 6.1.9 and 6.1.10 also show the change for each of these three forest types across each 5-year period between 1996 and 2016. While the trend in forest cover is consistently down, with minor exceptions, there is variability in the percentage losses in the three categories of forest. The greatest loss in evergreen forest (slightly more than 0.6%) occurred between 1996 and 2001, while the greatest loss in deciduous forest cover (nearly 0.6%) occurred between 2010 and 2016. Only the period between 2001 and 2006 saw an overall increase in forest cover, for both evergreen and mixed forest types. All four states' Forest Action Plans additionally suggest that the composition of forest communities is changing. These species shifts raise concerns about the future forest compositions, notably, their ability to provide critical food resources to wildlife. Potential reductions in mast species such as beech and oak towards species such as red maple will undoubtedly change forest trophic interactions.

Future Predictions

Forest cover has continuously declined across the Delaware Estuary and Basin, likely due to anthropogenic pressure. Anthropogenic pressure is likely to remain consistent or increase over time (see Chapter 1), and so, development or parcelization will continue to further net losses of forests, as well as fragmentation and lower habitat connectivity. Additionally, climate change is likely to drive large-scale community changes in forest cover and type (Forest Feature 1).

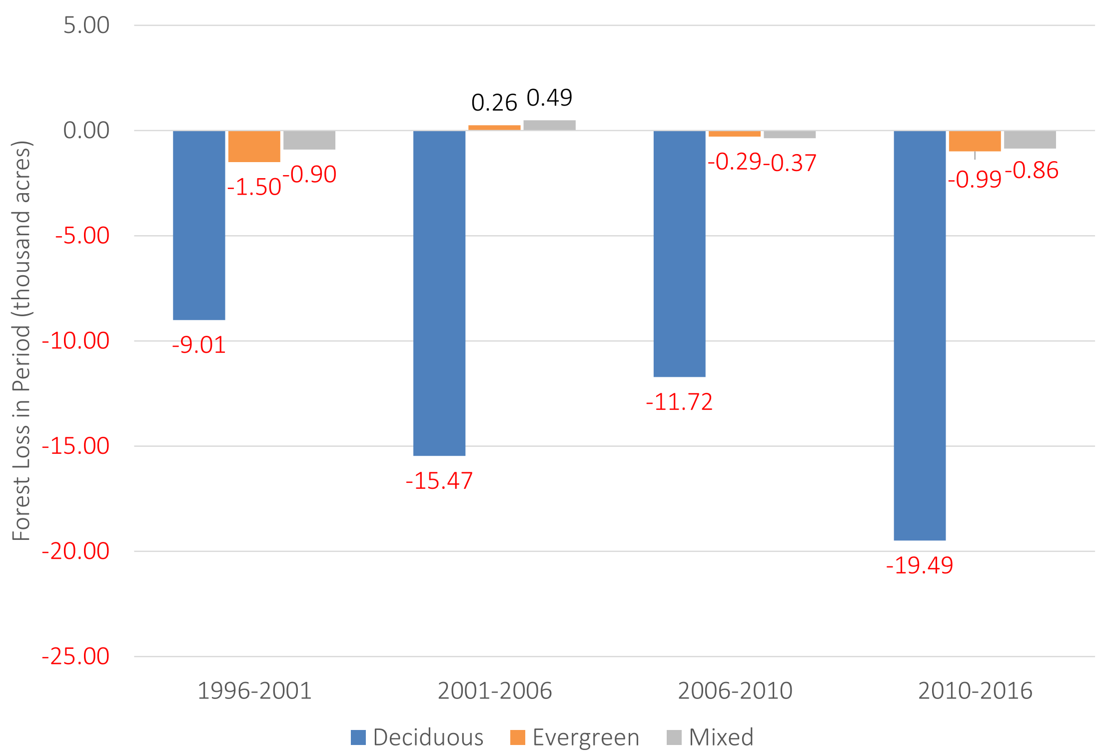

Figure 6.1.9 Total forest losses by type in the Delaware Estuary and Basin between 1996 and 2016.

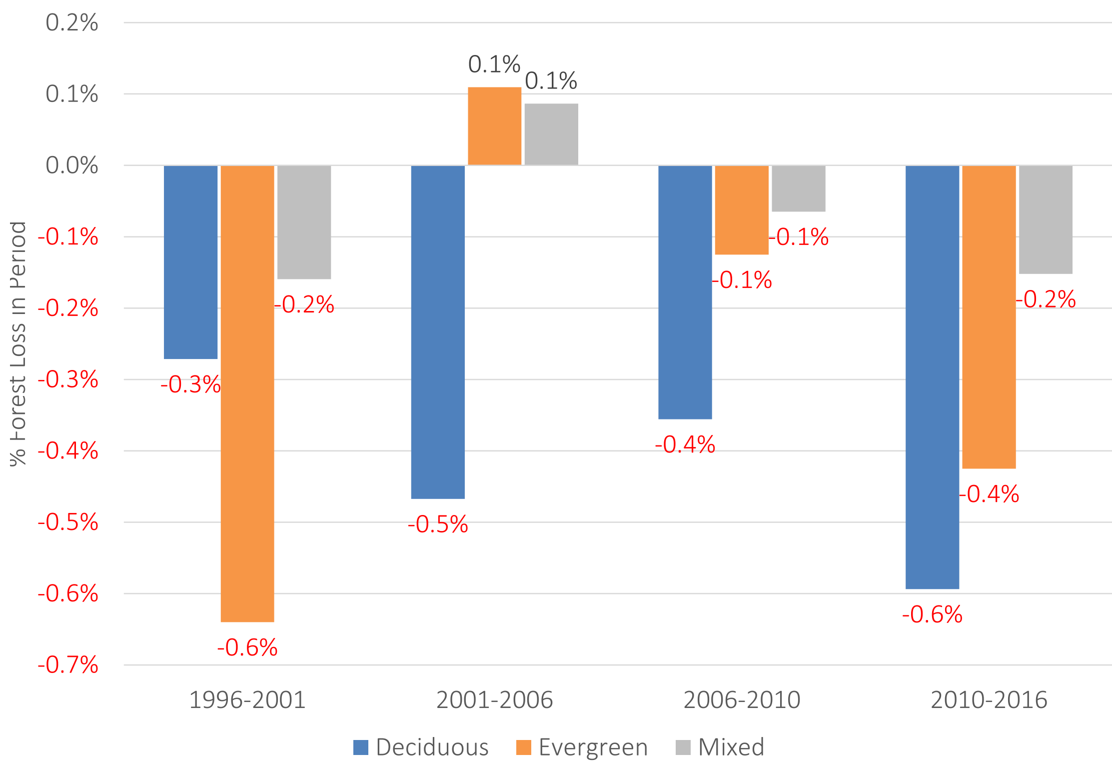

Figure 6.1.10 Rate of forest losses by type in the Delaware Estuary and Basin from 1996 to 2016.

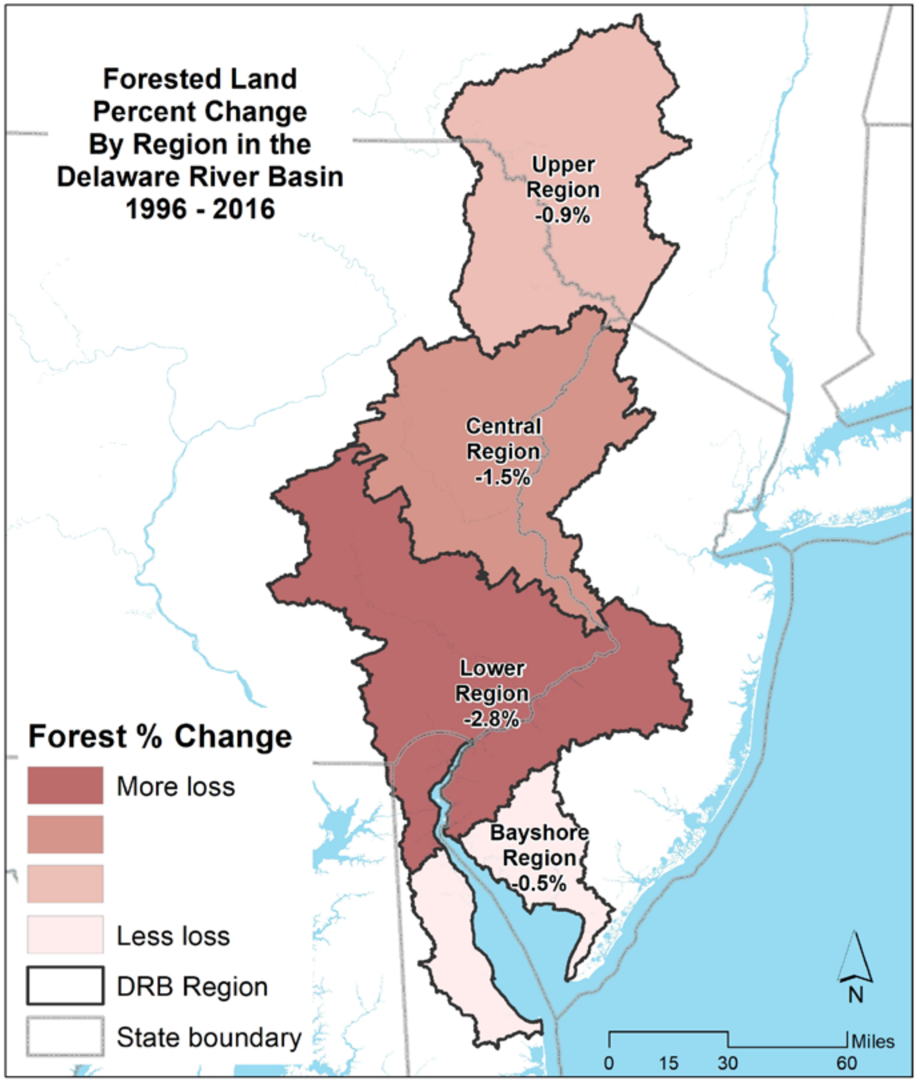

Figure 6.1.11 Forest cover percent change by Region of the Delaware Estuary and Basin (1996-2016).

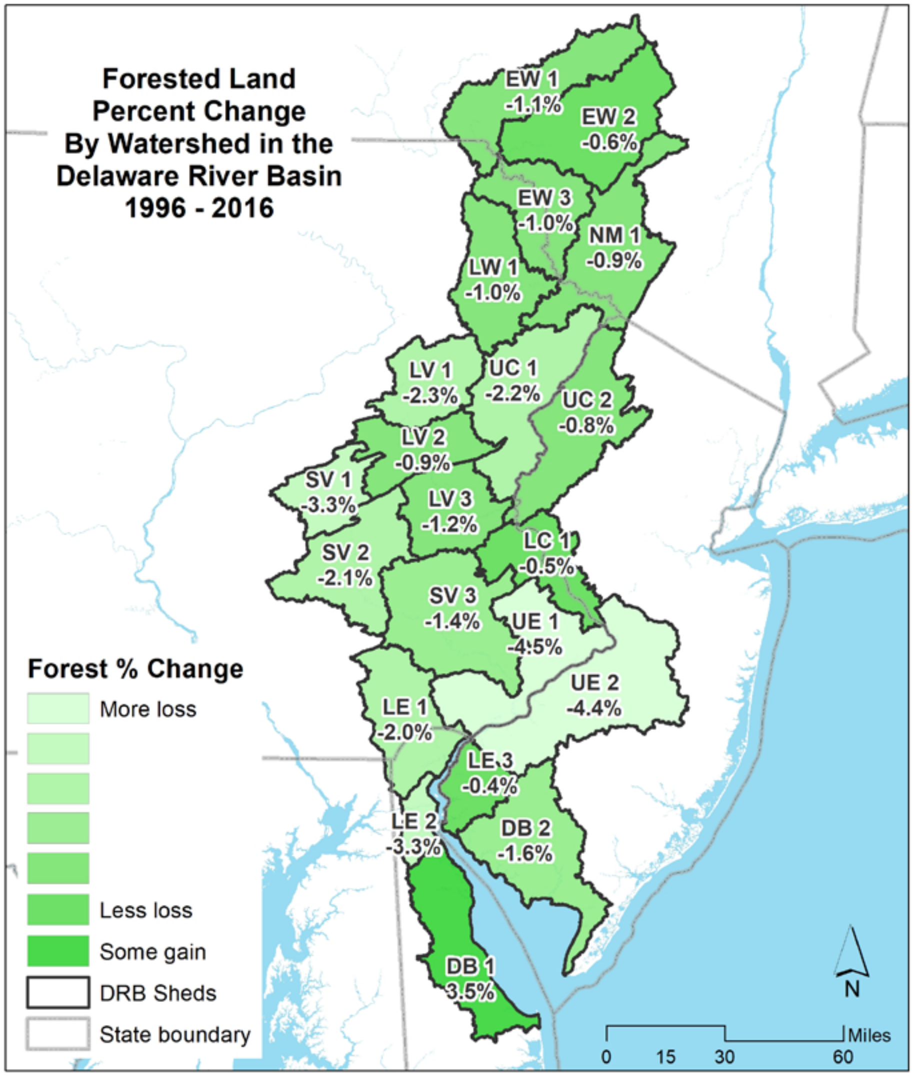

Figure 6.1.12 Forest cover percent change by watershed of the Delaware Estuary and Basin (1996-2016).

Damage-causing agents are likely to increase in influence across the Delaware Estuary as the climate changes, forests fragment, and populations increase. Invasive species are a focal point across all state-specific Forest Action Plans within the Estuary and Basin, with a heavy emphasis on rapid detection and monitoring as tools for management. The Estuary is especially susceptible to invasive pests as it is a hub for domestic and international transportation via roadways, railways, airways, and waterways. These pests are exacerbated by white-tailed deer population overabundance. Deer selectively browse native vegetation, giving invasive species a competitive advantage. Fire suppression aids the transport of native and non-native pests by creating dense, even-aged stands and can reduce species diversity, leaving the forest more susceptible to catastrophic damage.

Other notable drivers of future forest change include, but are not limited to, the inadequate regeneration of forests, biodiversity loss, forest maturation, and forested wetland acreage losses. All of these drivers reduce critical habitats for animal and plant species, especially those of conservation concern. Forest cover percent change by region and by watershed are depicted in Figures 6.1.11 and 6.1.12, respectively.

Actions & Needs

Every ten years, the federal Farm Bill requires states to compile a Forest Action Plan. These plans were first compiled in 2010. The recent 2020 Forest Action Plans represent the second iteration of this national forest planning effort which focuses on understanding the issues facing our nation's forests and the actions necessary to mitigate those issues. Some important issues facing the states in the Delaware Estuary and Basin include:

- Forest conversion to development and agricultural land
- Fragmentation and parcelization into smaller forest tracts
- Regulations for private forest ownership make buying, holding, and maintaining forestland expensive, putting pressure on owners to sell or follow unsustainable practices
- Socioeconomic inequalities reflected in unequal rates of urban tree cover
- Forest fire suppression exacerbating wildland fire risk

As per the requirements of the Farm Bill, each state addressed the following three top national priorities in their action plans:

- Conserve and Manage Working Forest Landscapes for Multiple Values and Uses
- Protect Forests from Threats
- Enhance Public Benefit from Trees and Forests

Each state broke down the priorities into sub-issues and provided actionable forest protection measures. In Table 6.1.2 we present a summary of priority items proposed in state-specific forestry action plans for New York, New Jersey, Pennsylvania, and Delaware.

Summary

Forests provide important ecosystem services such as reducing stormwater runoff and cleaning our air and water. Across the watershed, forests account for roughly half of the total land cover, but this is likely only a fraction of what it was 400+ years ago. Although forest types vary by region, states in the Estuary have similar issues with connectivity, cover, and condition. Development around the estuary has been

Table 6.1.2 Brief summary of state-specific forestry management priorities and possible risks.

Topic	New Jersey	New York	Delaware	Pennsylvania	USFS
Priorities	• Tree density • Forest age structure • Species of Concern • Biodiversity • Fragmentation, invasive species, land use change/disturbance, and climate change • Climate and Carbon • Fragmentation and Habitat • Damage-Causing Agents	• Keep New York's forests as forests ("Forests as Forests") • Keep New York's forests healthy ("Healthy Forests") • Ensure forests benefit humans and all living creatures ("Forests for People") • Support, protect, and appreciate New York's forests ("People for Forests") • Defending forests from invasive plants and insects • Excessive forest clearing and fragmentation • Worsening forest regeneration • Diversity, equity, and inclusion • Indigenous knowledge/ values and commitment to an increased level of engagement	• Forest Health and Functionality • Forest Markets • Sustainable Forest Management • Public Awareness and Appreciation of Forests • Reducing threats of development, fragmentation/ parcelization • Soil and water quality protection and enhancement	• Land Use Change • Forest Health • Sustainable Forest Management • Climate Change • Communicating Natural Resource Values • Energy Management & Development • Wildland Fire and Public Safety • Plant and Animal Habitat • Forest-related Economy and Jobs • Forest Recreation • Water and Soil	• Conserve and manage working forest landscapes for multiple values and uses. • Protect forests from threats. • Enhance public benefits from trees and forests.
Possible risks	White-tailed deer over abundance, disease, fragmentation/parcelization/development, climate change, invasive species				

increasing and will continue to increase. Forest losses have slowed since mid-century, but losses have continued overall, with slightly higher recent losses (0.5% per year from 2010-2016). Adaptively managing remaining forests, and preserving forests where possible, will be critical to their future functioning and the health of the Estuary and Basin.

Forests are facing several issues, such as but not limited to, age/size class distribution, reduced regeneration rates, conversion, and damage-causing agents. The state-specific forestry action plans provide a framework for understanding key issues to forests in each state, possible action items, and gaps in management practices. This review identifies similarities, strengths, and possible gaps for management across the watershed. Reductions in fragmentation, more forest protected lands and/or state management areas (purchases, easements, etc), increased prescribed burning (reduced fire suppression), ecological-based management tactic implementation (increase age class diversity and biodiversity, etc), and support for urban forestry are key themes to address moving forward.

References

Delaware Department of Agriculture Forest Service, 2020, Delaware Forest Resource Assessment, https://delawaretrees.com/2020_del_forest_resource_assessment.pdf.

Forest buffers. Chesapeake Bay Program. (n.d.). Retrieved June 24, 2022, from https://www.chesapeakebay.net/issues/forest_buffers

Irland LC. 1999. The Northeast's changing forests. Cambridge, MA: Harvard University Press.

New York State Department of Environmental Conservation, 2020, New York State Forest Action Plan, https://www.dec.ny.gov/docs/lands_forests_pdf/nysfap.pdf.

National Oceanic and Atmospheric Administration, Office for Coastal Management. "C-CAP Regional Land Cover." Coastal Change Analysis Program (C-CAP) Regional Land Cover. Charleston, SC: NOAA Office for Coastal Management. Accessed June 2022 at www.coast.noaa.gov/htdata/raster1/landcover/bulkdownload/30m_lc/.

Pan, Y., Chen, J.M., Birdsey, R., McCullough, K., He, L. and Deng, F., 2011. Age structure and disturbance legacy of North American forests. Biogeosciences, 8(3), pp.715-732. https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0072540

Pennsylvania Department of Conservation and Natural Resources Bureau of Forestry, 2020, Pennsylvania Forest Action Plan, http://elibrary.dcnr.pa.gov/GetDocument?docId=3861659&DocName=PA%20Forest%20Action%20Plan%202020.pdf.

State of New Jersey Department of Environmental Protection NJ Forest Service, 2020, New Jersey State Forest Action Plan, https://nj.gov/dep/parksandforests/forest/njsfap/docs/njsfap-final-12312020.pdf.

Thompson, J.R., Carpenter, D.N., Cogbill, C.V. and Foster, D.R., 2013. Four centuries of change in northeastern United States forests. PloS one, 8(9), p.e72540.

Forest Feature 1

Loblolly Pine and Eastern Hemlock Distributional Shifts with Climate Change

Kelly Faller & LeeAnn Haaf
Partnership for the Delaware Estuary

Climate change will have sweeping effects on forest health and composition in the future. One of the many impacts of human-induced climate change is increasing temperature. Temperatures in the Mid-Atlantic are projected to increase 0.5 to 4.2°C (2.2 to 7.6°F) by the end of the century. With these changes, we expect that the current distribution of tree species will shift northward in response to warming temperatures. The Delaware Estuary, sitting at the confluence of many tree species' northern or southern range extent, will likely see forest community compositional shifts as species distributions expand poleward to match their temperature thresholds.

This feature focuses on the climate change-induced species distribution changes of loblolly pine (*Pinus taeda*; Fig 6.1.13A) and eastern hemlock (*Tsuga canadensis*; Fig 6.1.13B). These species are valuable examples of range shifts as the Delaware Estuary and Basin is the southern-most extent, and/or elevation limit, of eastern hemlock (Fig 6.1.13C) and the northern-most extent of loblolly pine (Fig 6.1.13D). Hemlock and loblolly pine are both commercially valuable woods and their shifting ranges will impact the timber industries throughout the states that constitute the Delaware Estuary and Basin.

Here we present USFS Climate Change Atlas (v 4.0) models of loblolly pines and eastern hemlock distributional changes based on current distributional, physiographic, and climatic information (Butler-Leopold et al. 2018). We chose to showcase predicted distributional changes of loblolly pine and eastern hemlock under the highest emission climate scenario predicted by the Intergovernmental Panel on Climate Change (RCP 8.5). This climate prediction is frequently referred to as "business as usual", as it predicts climate change if there was no change in policy or action with respect to carbon dioxide emissions.

Under continued carbon dioxide emissions, loblolly pine will likely become more common in the lower estuary as its distribution shifts northward, further into New Jersey and Pennsylvania (Fig 6.1.13E). Contemporaneously, as temperatures warm, eastern hemlock is likely to experience northward shifts outside of the watershed— until this species becomes rare within the Basin (Fig 6.1.13F).

Citations

Butler-Leopold, P.R., Iverson, L.R., Thompson, F.R., Brandt, L.A., Handler, S.D., Janowiak, M.K., Shannon, P.D., Swanston, C.W., Bearer, S., Bryan, A.M. and Clark, K.L., 2018. Mid-Atlantic forest ecosystem vulnerability assessment and synthesis: a report from the Mid-Atlantic Climate Change Response Framework project. Gen. Tech. Rep. NRS-181. Newtown Square, PA: US Department of Agriculture, Forest Service, Northern Research Station. 294 p., 181, pp.1-294.

USFS Climate Change Atlas, Trees, v 4.0. 2022. <https://www.fs.fed.us/nrs/atlas/tree/> Accessed July 5, 2022.

Intergovernmental Panel on Climate Change (IPCC). 2022. Climate Change 2022: Impacts, Adaptation and Vulnerability. <https://www.ipcc.ch/report/ar6/wg2/> Accessed July 5, 2022.

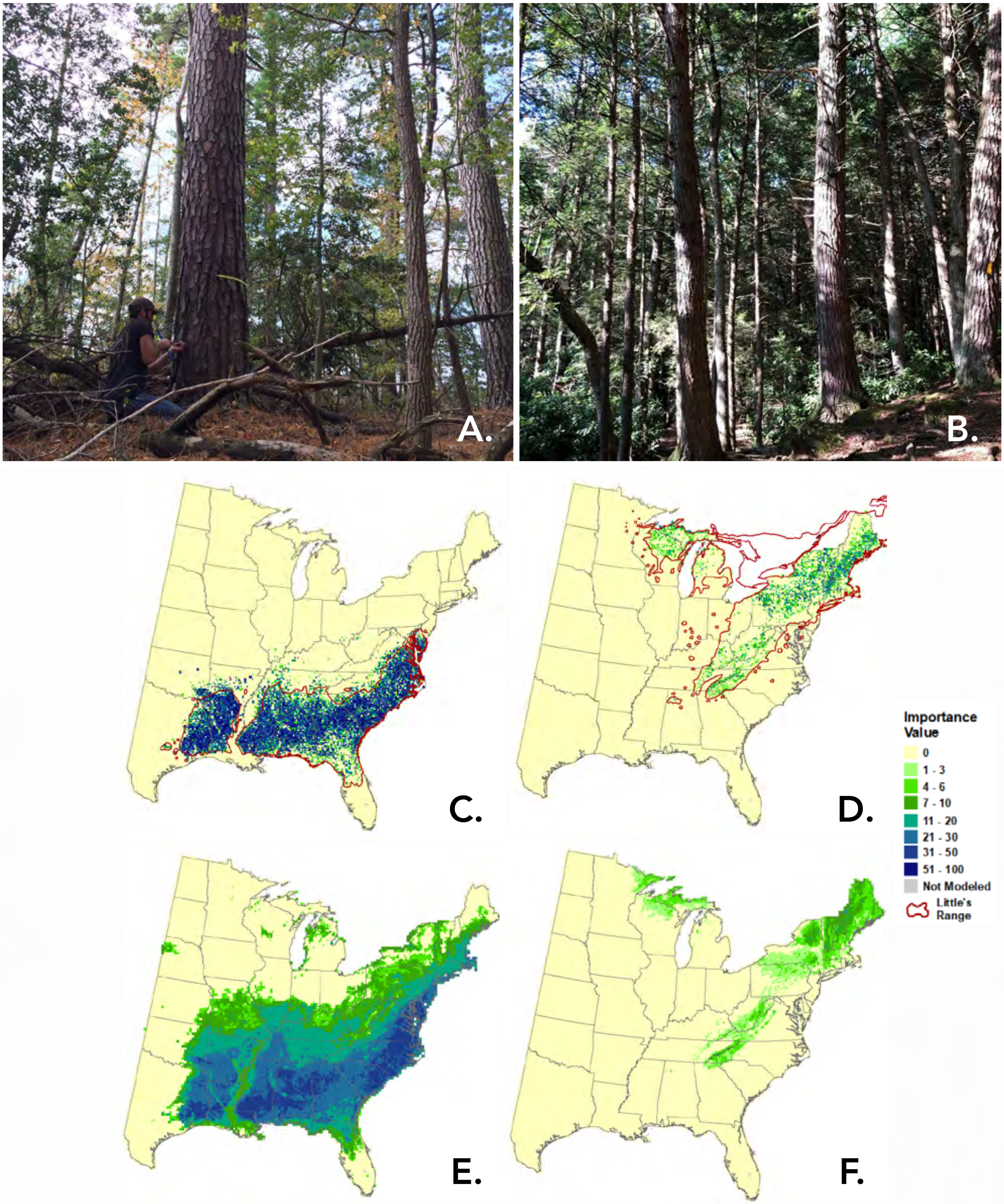

Figure 6.1.13 Distributions of loblolly pine (*Pinus taeda*, A) and Eastern hemlock (*Tsuga canadensis*, B) in the eastern United States from current (C, D, respectively) to future distribution under RCP 8.5 (E, F) modelled by USFS FIA.

6.2 Submerged Aquatic Vegetation

Description of Trial Indicator

Submerged aquatic vegetation (SAV) is an essential habitat for fish and other wildlife (Fig 6.2.1). It provides spawning, nursery, and protective habitat to juvenile fish and bivalves such as freshwater mussels (Lubbers et al. 1990; Heck et al. 2003). SAV services include shoreline stabilization, carbon sequestration, nutrient filtration, sediment entrapment and wave energy dissipation (Kemp et al., 1984; Orth et al., 2010; Ward et al., 1984; Waycott et al., 2009; Jaskinski et al., 2021). Despite these ecologically important attributes, historical SAV studies in the tidal Delaware Estuary are limited (Schuyler, 1988; Schuyler et al., 1993).

For this report, submerged aquatic vegetation is being piloted as a trial indicator. To create a foundation for protecting and ultimately restoring SAV habitat in the tidal Delaware Estuary, a baseline understanding of the distribution, density, and composition of SAV was needed. Since 2017, acoustic data has been collected and analyzed to begin to characterize these metrics. Furthermore, preliminary observational data has been recorded to start to understand habitat suitability, interannual trends and factors impacting long-term stability. Currently, there is not enough data over time to analyze trends and methods are still evolving. Therefore, a full indicator report is not currently feasible. This trial indicator reports some preliminary data and observations but is not able to discuss potential indicators of watershed health.

Methods

Typically, quantifying SAV is assessed through annual aerial images that are used to map SAV beds. Aerial flights and assessment of imagery are dependent on favorable water clarity. The Delaware Estuary is a naturally turbid watershed making aerial imagery an unreliable method for quantifying SAV extent. Because of this, hydroacoustic surveys were determined to be the most suitable and effective method for detecting and mapping SAV.

To quantify SAV in the Delaware Estuary, a single beam echosounder (BioSonics© MX Habitat Echosounder) was used to detect submerged vegetation. The transducer sends pings of sound straight down through the water column (Fig 6.2.2). When the sound encounters an object, such as SAV or bottom sediment, it echoes off the surface. This return echo is detected by the transducer, which interprets how long it took and at what amplitude (dB) the echo returned. The return echoes are displayed in an echogram (Fig 6.2.3). Using BioSonics © propriety software package, Visual Aquatic and Visual Acquisition, SAV beds were able to be delineated and mapped in a web-based mapping application that shows the extent and height of SAV across a limited time series (2017-present).

Figure 6.2.1 SAV captured on a telescoping rake in the Tidal Delaware River.

Field Sampling

Using an echosounder mounted to a vessel, transects parallel to the shoreline were surveyed. The distance off the edge of the shoreline varies depending on the bathymetry and bottom type. Parallel transects were used to identify 'presence/absence' of SAV. When SAV was confirmed along the

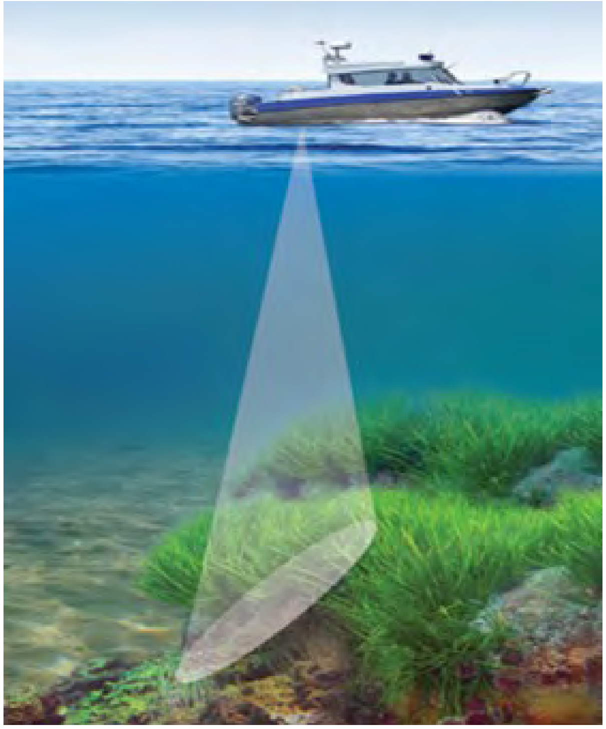

Figure 6.2.2 Example of a single beam echosounder sending pings down the water column (image from BioSonics).

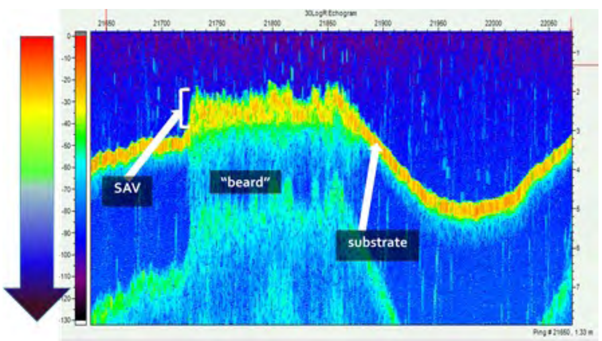

Figure 6.2.3 Example of a single beam echosounder.

Figure 6.2.4 Example of boat tracks for collection. The tight zigzag patterns mimics 'mowing the lawn'.

parallel transect, transects perpendicular to shore were surveyed to determine the spatial extent of the beds. The pattern to delineate the bed is similar to 'mowing the lawn' where the boat moves in lines perpendicular to shore to map the entire length and width of the bed. Figure 6.2.4 shows this collection pattern. Using Visual Acquisition, Biosonics Field Data Collection Software program, while pinging the echosounder, an echogram would display the bottom substrate and where SAV might be present. During perpendicular transects, field verification was used by dropping a telescoping rake at random points throughout the bed. Where possible, species identification was noted. Species composition has not been a regular variable measured in the annual acoustic survey, however where possible coordinate pins were dropped on the software and field notes captured species. Species that were identified include *Vallisneria americana*. In future years, there is plans to have a more concerted effort to inventory species identification.

Ground-Truthing

EPA's Scientific Dive Unit completed numerous dives to field verify the echosounder data as well as to collect other scientific field data. Divers used quadrats to determine percent coverage as well as species composition and noted the presence of bivalves, if observed. Divers had difficulty collecting data due to low visibility, strong currents, and the heavy accumulation of fine sediments within the SAV beds (Fig 6.2.5). Select long-term monitoring stations were piloted with the intent of having annual scientific dives.

Post-Processing

After the data is collected, it is post processed using BioSonics Visual Aquatics (formerly Visual Habitat). The software interpolates the data in two phases. First, the program auto detects and delineates the bottom surface drawing the hard bottom providing the bathymetry for the area. In areas of high density SAV beds, some manual editing is required using the 'draw tool' to identify and delineate the bottom substrate. After the bottom is defined, the program then auto-detects and delineates the vegetation canopy. Some editing may be required to further delineate the canopy using the software's pen tool. Processed data were exported to comma separated value (CSV) format to facilitate loading into a geographic information system (GIS) platform and a database management system.

Inverse distance weighting (IDW) technique was then used to interpolate the data (Fig 6.2.6). IDW is a nearest neighbor approach that relies less on geo statistics and underlying assumptions. IDW assumes that points with close proximity geographically will tend to have similar values. This high-density survey allowed the interpolation of raster datasets depicting plant coverage. The interpolated raster data was then clipped 1 meter from the closest sample point to ensure only areas covered by the survey were represented. Coverage is reported in the percent of pings that recorded vegetative cover (i.e., 0, 20, 40, 60, 80, 100).

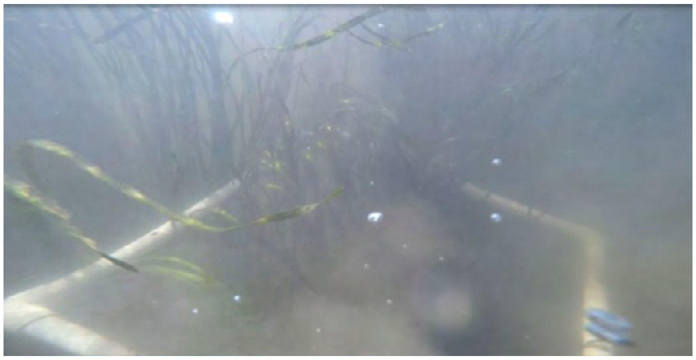

Figure 6.2.5 A quadrat used by scientific divers quantifying the percent coverage of SAV.

Data from 2017-2020 are available online for viewing and download online. In 2017 and 2018, data was collected parallel to shore to determine the presence or absence of SAV. In 2019, the survey refined its methods and limited scope allowing more detailed bed delineation. The 2020 and 2021 field season was more limited due to travel restrictions from COVID-19.

Present Status

The project, which focused on inventorying the distribution and density of SAV, has completed five years of monitoring (2017-2021). The first two years, 2017-2018, set out to completely map the Delaware Estuary which includes the tidal Delaware River and the Delaware Bay. However, no SAV was identified in the meso- or polyhaline zones of the river from approximately the Delaware Memorial Bridge south to the mouth of the Delaware Bay. Therefore, starting in 2019, the survey scope was re-focused to the tidal Delaware River which includes the head of tide near Trenton, NJ down to the maximum turbidity zone near the Delaware Memorial Bridge.

There are a few possibilities why no SAV (seagrasses) were found in meso- and polyhaline areas. One is that species like *Zostera marina* never thrived in the system historically (Schuyler 1988; Schuyler et al., 1993). Another possibility is that the survey did not look in the right areas and/or at the right time. Since time and resources were a factor, the decision was made to focus on the freshwater tidal where SAV was found and thriving.

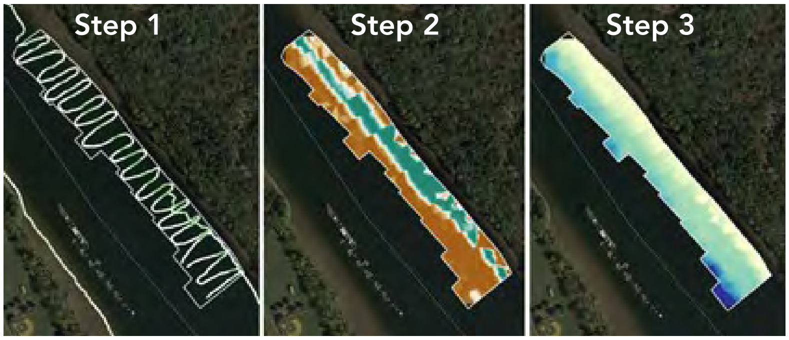

Figure 6.2.6 A time series showing the process of interperolation using IDW.

Basic analysis of distribution and density of SAV was analyzed for 2019 and was summarized among the Delaware River water quality zones (Table 6.2.1). Coverage is reported in the percent of pings that recorded vegetative cover (i.e., 0, 20, 40, 60, 80, 100). The map of Little Tinicum Island (Fig 6.2.7) and the surrounding areas demonstrate the percent coverage of SAV. The darker the shade of green, the denser the bed is. As the survey continues, time-series trends of changes in bed density and size will be able to be collected and shared to determine the stability of SAV health in the system.

Table 6.2.1 Summary of SAV distribution and density in Delaware River water quality zones.

Water Quality Zones	SAV Density (Acres)
Zone 2	1213
Zone 3	628
Zone 4	1352
Zone 5	109

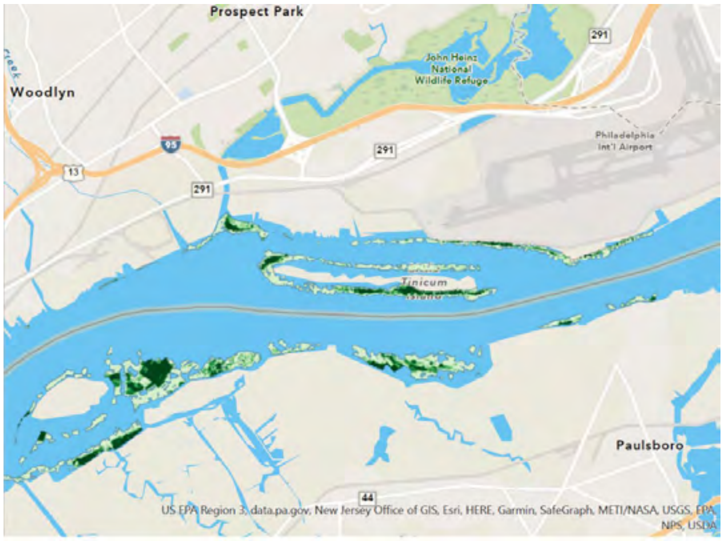

Figure 6.2.7 A map of Little Tinicum Island in the Delaware River displaying the density of SAV. Dark green indicates high plant density.

Zone 1 – No surveys completed as this area is not tidal.

Zone 2 – Head of tide to Pennypack Creek

The head of tide starts in Zone 2 near Trenton, NJ. SAV beds from Roebling, NJ north to the survey limit are narrow and close to the banks (Fig 6.2.8). Bathymetry within this area consisted of a narrow shallow shelf close to the banks which is likely a limiting factor for SAV. Overhanging terrestrial vegetation also shades portions of the banks. South of Roebling to the Burlington Bristol Bridge, SAV beds get broader as the bathymetry of the river provides greater shelf areas and favorable depths that allow for light penetration at mean high water and remain submerged at mean low water. Areas south of the Burlington Bristol Bridge to Pennypack Creek have broader beds and greater shelf area with favorable water depths. Some beds, although not continuous, extend almost 150 m (500 ft) from the shoreline toward the center of the river, with multiple bands of beds.

Zone 2 also contains Newbold and Burlington Islands. The Newbold Island back channel was not surveyed due to water depths being too shallow for boat passage. SAV beds were identified along the main channel side of the island, along a stone revetment on the southern portion of the island, as well as the northern forested portion of the island. The southern tip of Burlington Island has a significant bed, which was thoroughly surveyed in 2019-2021. The channel side of the island, as well as portions of the back channel of the island, had SAV beds in 2019. The areas lacking SAV beds were deeply scoured, dredged for marina space, and/or too exposed at low tide.

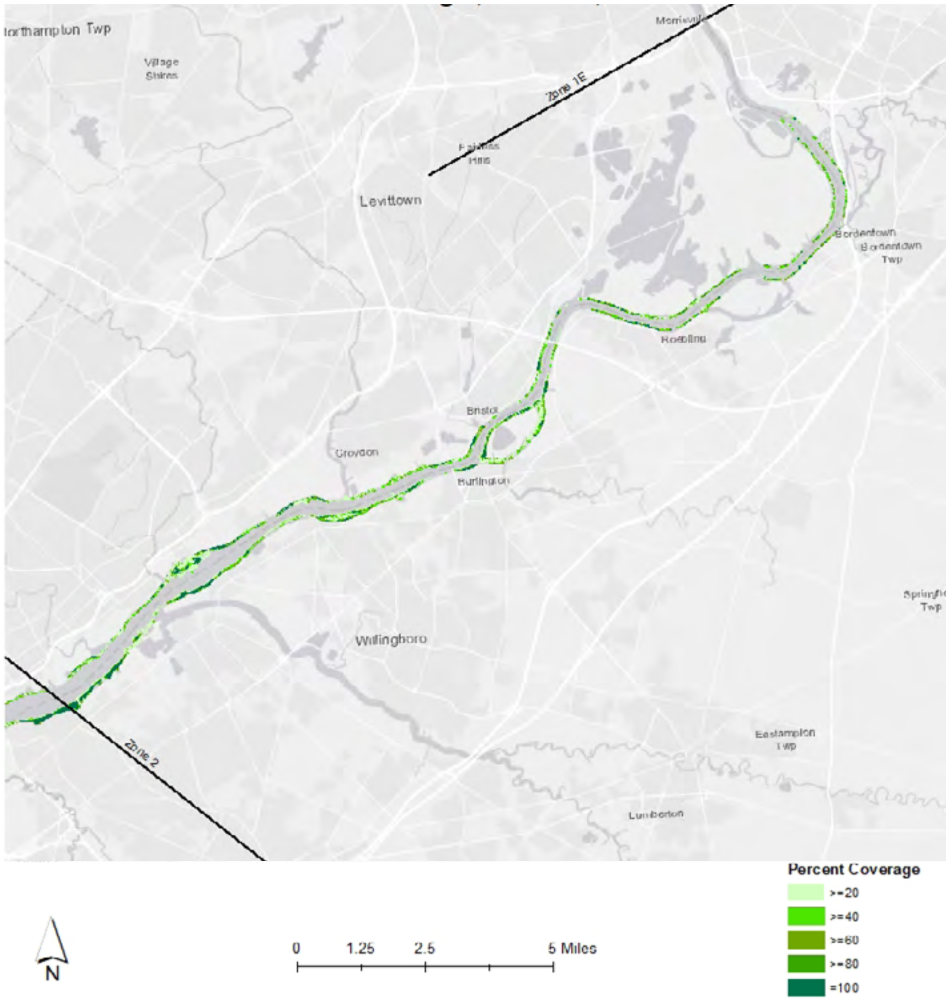

Figure 6.2.8 SAV coverage, Zone 2, 2019.

Zone 3 – Pennypack Creek to Big Timber Creek

SAV in Zone 3 was most abundant north of Petty Island (Fig 6.2.9). A broad bed thrives just south of the Pennsauken Creek. A bed on the northern tip of Petty Island was also intensely surveyed in 2019-2021. The cove associated with the northern tip of Petty Island has significant bed(s) consisting of a variety of different species. However, the channel ward side of the island as well as most of the back channel of the island SAV has not been observed. In 2021, a small area was observed just south of the bridge to access the island. From Petty Island to the south of the Walt Whitman Bridge, SAV is limited to areas that have not been dredged. The survey did not investigate within the old piers or behind structures due to navigation hazards, but SAV may be present within these areas. Downstream of the Philadelphia/Camden port facilities and along the bulkhead of the Navy Yard, the beds get broader as the bathymetry of the river provides greater shelf area and at favorable depths for SAV. The beds are close to the riverbanks. The Navy Yard site is another location where thorough data has been collected.

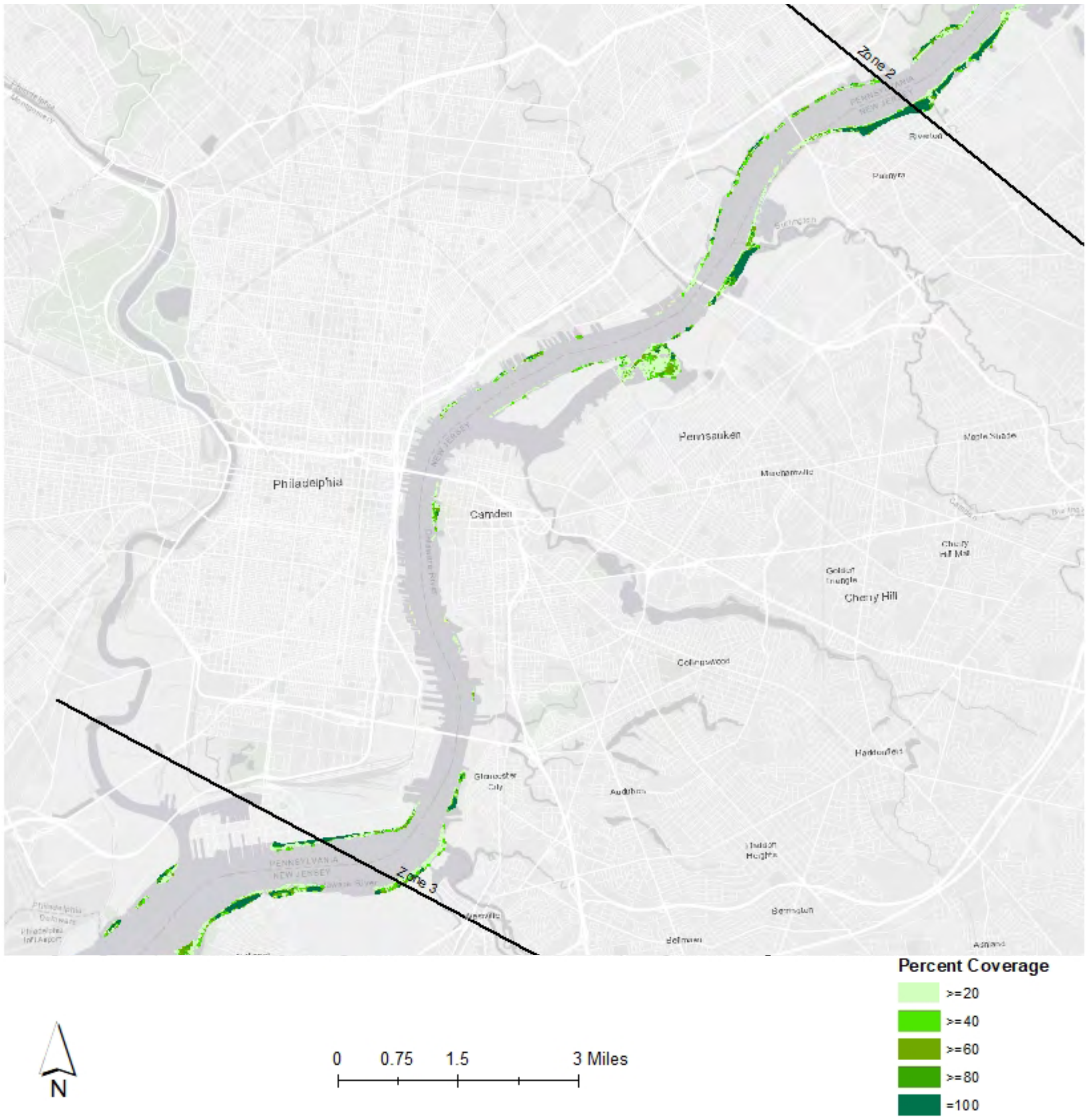

Figure 6.2.9 SAV coverage, Zone 3, 2019.

Zone 4 - Big Timber Creek to Marcus Hook/Raccoon Creek

Zone 4 contains abundant SAV beds (Fig 6.2.10). Many of the areas lacking SAV along the main channel, however, have been dredged. The Red Bank Battlefield area has significant bed(s) consisting of a variety of different species. A submerged wave screen located north of Chester Island supports a large bed of SAV that sits over 600 m (2,000 ft) channel ward of the shoreline. Another large bed is established at the mouth of Darby Creek. From Darby Creek south along the Pennsylvania shoreline, SAV is sparse, and the area has been dredged.

This Zone contains Tinicum and Chester Islands. The channel ward side of Tinicum Island consists of narrow beds of SAV close to the shoreline. The southern tip of the island contains a broad bed that stretches into the back channel. At the back side of the island, SAV is sparse but emergent spatterdock (*Nuphar advena*) is prevalent. The portion of back-channel along Hog Island Road also supports a broad bed of SAV that narrows and transitions to spatterdock as the road turns north. Chester Island has a large SAV bed on the south side of the submerged wave screen discussed above. There is also a bed along a revetment on the back channel of the island and a small dense bed on the southern tip of the island. Moving south, the amount of SAV decreases significantly.

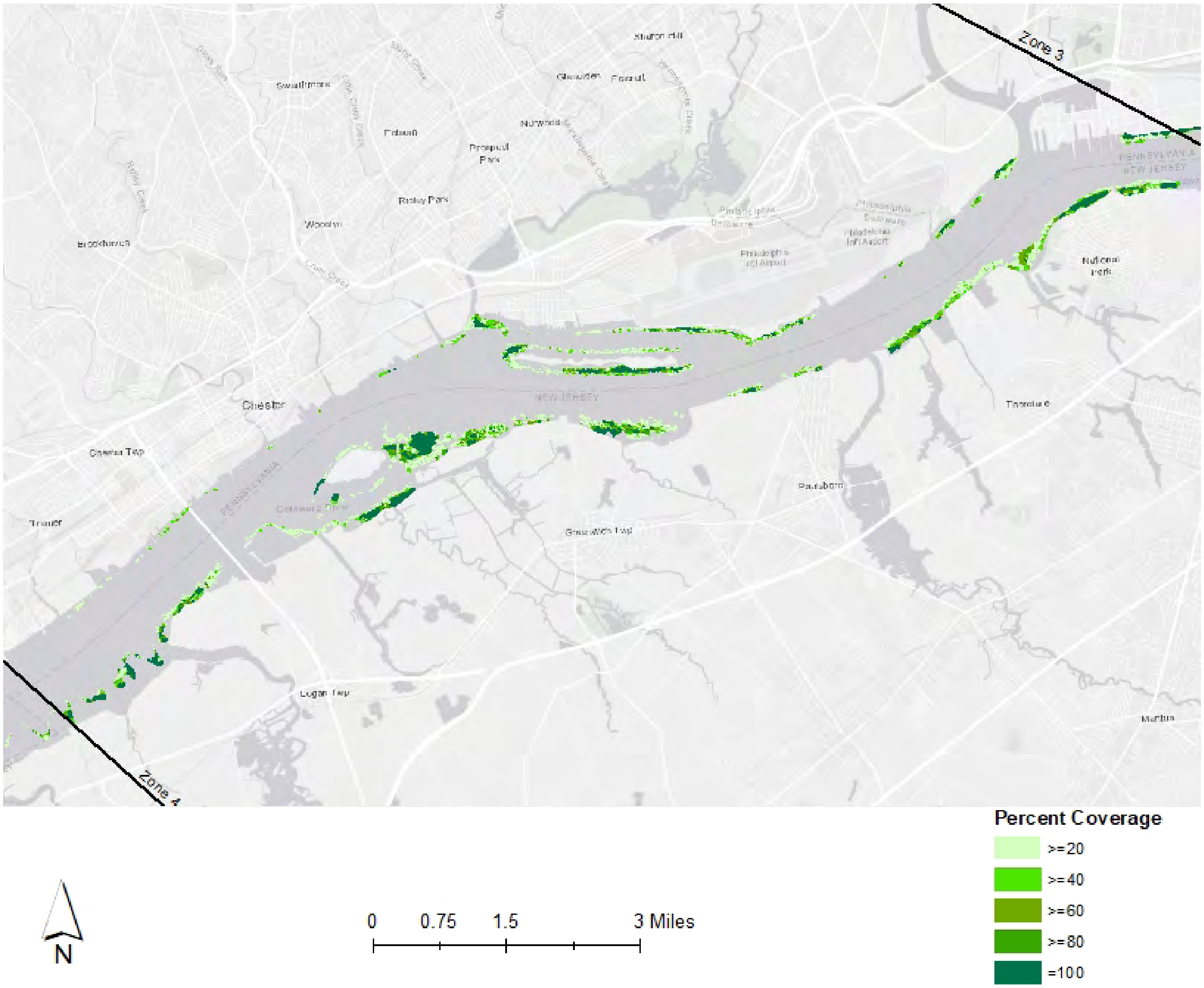

Figure 6.2.10 SAV coverage, Zone 4, 2019.

Zone 5 - Marcus Hook – Raccoon Creek Delaware State line south

SAV in Zone 5 is minimal and sparse (Fig 6.2.11). While the bathymetry and water depths would support SAV, very little SAV was observed in the 2019 survey. The SAV that was observed was mostly on the New Jersey side, close to the Commodore Barry Bridge. A small amount of SAV was observed along the New Jersey coastline. This area is the maximum turbidity zone in the Estuary and high concentrations of suspended sediments may interfere with light reaching the bottom substrate. Additional studies are required to determine the limiting factors of SAV distribution in this Zone.

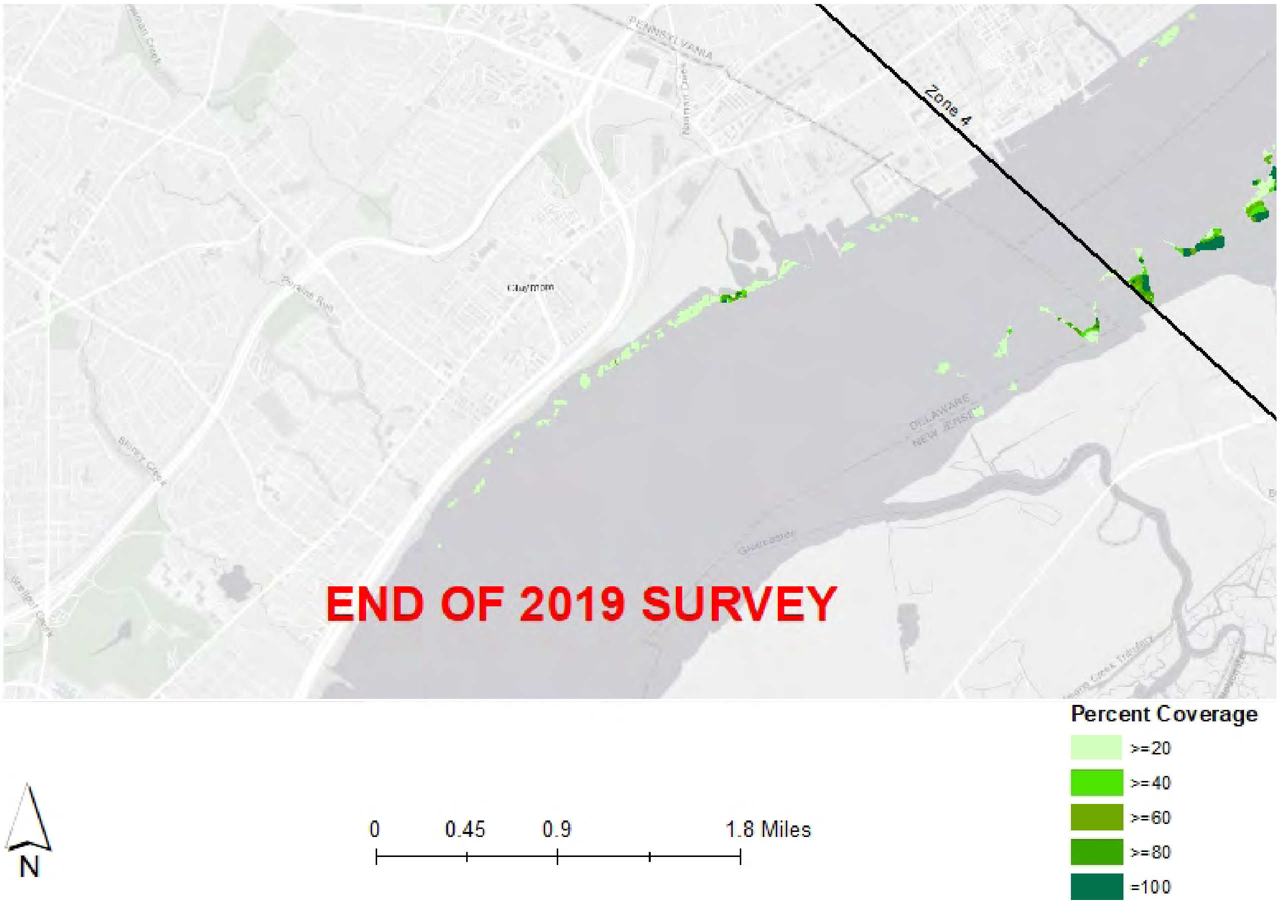

Figure 6.2.11 SAV coverage, Zone 5, 2019.

Regulatory Authorization

Impacts on SAV must be considered during the decision-making process to issue a permit for dredge and/or fill material. Data from this study is being used by applicants and agencies to review proposed project sites for potential impacts on SAV beds. If SAV is present within the project area, the applicant may need to consider alternative locations to avoid and minimize the negative impacts on SAV. If SAV cannot be avoided and a permit is issued, mitigation may be required.

Past Trends

There is a lack of information on the historic distribution of SAV species in the Delaware Estuary (Schuyler, 1988; Schuyler & Kolega, 1993). It was once believed in the scientific community that SAV did not thrive in this system due to the limited light availability, turbidity, excess nutrients, and/or dredging activity occurring in the system. Some of these limiting factors, like turbidity, are natural features of the Delaware Estuary. Up until 2011-2012, however, researchers lacked adequate equipment and technical expertise to properly inventory the size, extent, and species of the beds. Also, the Philadelphia Water Department (PWD) performed a study in 2011 in which they observed emergent and submerged species, but the area of study did not extend beyond the city limits.

In 1988, the Academy of Natural Sciences published the most cumulative historical and present account of SAV plant diversity in the tidal Delaware River and its tributaries. The report indicated that no previous work had been done on submergent or planmergent (plants with floating leaves) in the watershed. It was reported that numerous plants of *Vallisneria americana, Myriophyllum spicatum, Elodea nuttallii,* and *Najas flexilis* were present in the portion of the Estuary between Trenton and Philadelphia. That is consistent with the preliminary plant identification found in the current research. In this report, it was acknowledged that the larger tidal range might play a factor in plant diversity in the main Delaware River channel. It was concluded that some plants that appeared prior to the 1930s were then found to be absent in the river. Other plants remained or retreated to tributaries where tide ranges were smaller or did not span as far down the river as once before. This study showed that SAV diversity declined. Although this study looked at species distribution, it did not synthesize plant coverage or bed densities throughout the watershed. This study provided valuable information on species occurrences but did not indicate historical SAV bed density or areal cover, making it difficult to measure what the historical acreage of SAV in the tidal Delaware River was (Schuyler, 1988; Schyler et al. 1993).

Understanding the role of SAV in the Delaware Estuary and its impact on water quality and habitats has been identified as an important data gap. A 2006 white paper on the Status and Needs of Science in the Delaware Estuary authored by members of the Delaware Estuary Scientific and Technical Advisory Committee identified Submerged Aquatic Vegetation as data gap particularly given their importance to benthic communities (Kreeger et al. 2006). The 2021 Delaware Estuary Monitoring Inventory and Needs Assessment also identified SAV as a top monitoring priority (Partnership for the Delaware Estuary, 2021).

Future Predictions

SAV habitat in the tidal Delaware River is subject to many potential threats. However, without long-term data, it is difficult to predict how these changes could impact SAV survival. SAV in the tidal Delaware River has been subjected to historical dredging and fill events over time and has survived, despite the altered location of the riverbanks, changing bathymetry, and increase in sedimentation. Continuing to monitor the effects and understanding the impacts of physical changes is important to SAV protection.

Climate change is another threat to SAV health. As sea levels rise, increasing water heights with few shallow areas left to retreat due to hardened shorelines could cause significant losses to SAV in the Delaware. Shifting salt lines, another result of sea level rise, has the potential to impact freshwater SAV habitat. As storms increase in the watershed, large flushing events may lead to significant increases in nutrient and sediment loads that impact SAV growth by limiting light. Water quality improvements in the Delaware River could lead to significant SAV expansion. Improved water quality might also provide improved ways to monitor (i.e., aerial imagery or drones) and allow researchers to access tributaries and other shallower bodies within the estuary.

Actions and Needs

Several actions are needed to understand the value that SAV may play in the Delaware Estuary.

Species Inventorying and their distribution

This project has largely been focused on capturing the broad distribution and density of SAV beds in the Tidal Delaware River. Researchers capture sample grabs at select sites throughout the survey and note species collected, but no systematic approach to fully capturing in-depth underwater analysis of species diversity and geographic distribution of those species have been undertaken. Understanding the species diversity and their distribution could aid in future restoration projects and capture SAV ecosystem services.

Understanding Habitat Suitability

SAV has specific habitat and growth requirements, usually determined by light attenuation. The tidal Delaware River has an approximate 1.8 meter (~6 feet) tidal range that is also subjected to activities like dredging. Understanding the factors that determine SAV suitability is necessary to understand potential restoration and preservation. It's important to also understand what species can thrive under varying conditions such as large tidal ranges, changes in water quality variables, exposure to high energy, increases in sedimentation, and loss of light. Additionally, it is critical to understand the necessary sediment type and shoreline needed to create and maintain SAV habitats. To best preserve and restore SAV habitat, studies are needed to understand the physical, chemical, and biological factors required for SAV beds to thrive.

Piloting Restoration and Planting

SAV restoration is an important consideration for the Delaware River watershed. SAV provides many ecosystem services and increasing SAV distribution would improve water quality and provide essential habitat to key species. Little information is available to guide restoration managers on how best to restore SAV beds. Developing restoration guides that specify when to collect seeds, plugs or transplant and where and when to plant them is needed for successful restoration.

Summary

SAV is a vital habitat that provides many ecosystem services and protecting and restoring it is essential to supporting clean and healthy watersheds. SAV in the tidal Delaware Estuary is expansive and diverse, contrary to the previously accepted notion that this naturally turbid system could not support abundant SAV. Continuing to monitor SAV and establishing it as an indicator of water quality and benthic habitat is an important next step to understanding the overall health of the Delaware Estuary.

References

Heck, K. L., Hays, G. C., & Orth, R. J. (2003). Critical evaluation of the nursery role hypothesis for seagrass meadowns. Marine Ecology Progress Series 253, 123-136.

Jaskinski, D., Gurbisz, C., & Landry, B. (2021). Small-scale SAV Restoration in Chesapeake Bay: A Guide to the Restoration of Submerged Aquatic Vegetation in Chesapeake Bay and its Tidal Tributaries. cheapeakebay.net.

Kemp, W. M., Boynton, W. R., Twilley, R. R., Stevenson, J. C., & Ward, L. G. (1984). Influences of submersed Vascular Plants on Ecological Processes in Upper Chesapeake Bay. In The Estuary as a Filter (pp. 367-369). Academic Press.

Kreeger, D., Tudor, R., Sharp, J., Kilham, S., Soeder, D., Mawell-Doyle, M., Kraeuter, J., Frizzera, D., Hameedi, J., and Collier, C. (2006). White Paper on the status and needs of science in the Delaware Estuary. Wilmington: Partnership for the Delaware Estuary.

Lubbers, L., Boynton, W. R., & Kemp, W. M. (1990). Variations in structure of estuarine fish communities in relation to abundance of submersed vascular plants. Marine Ecology Progress Series 64, 1-14.

Orth, R. J., Williams, M. R., & Marion, S. R. (2010). Long Term Trends in Submersed Aquatic Vegetation in the Chesapeake Bay, USA, Related to Water Quality. Estuaries and Coast, 33, 1144-1163.

Partnership for the Delaware Estuary. (2021). Monitoring Inventory and Needs Assessment for the Delaware Estuary: Addendum to the 2019 Comprehensive Conservation and Management Plan for the Delaware Estuary. Wilmington, DE.

Schuyler, A. E. (1988). Submergent and Planmergent Flora of the Freshwater Portion of The Delaware Estuary. In Ecology and Restoration of the Delaware River Basin (pp. 157-168). Philadelphia: Pennsylvania Academy of Natural Sciences.

Schyler, A. E., Anderson, S. B., & Kolaga, V. J. (1993). Plant zonation changes in the tidal portion of the Delaware River. Academy of Natural Sciences of Philadelphia, (pp. 263-266). Philadelphia.

Ward, L. G., Kemp, W. M., & Boynton, W. R. (1984). The influence of waves and seagrass communities on suspended particulates in an estuarine embayment. Marine Geology, 59(1-4), 85-103.

Waycott, M., Duarte, C. M., Carruthers, T. J., Orth, R. J., Dennison, W. C., Olyarnik, S., . . . Williams, S. L. (2009, July 28). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. The Proceedings of the National Academy of Sciences, 106(30).

6.3 Wetlands

Wetlands are areas where water inundates soils, or saturates at or near the surface of the soil all year, seasonally, or periodically. The gradient of saturated conditions in wetlands create characteristic hydric soils, which in turn supports specially adapted plants (hydrophytes). Wetlands also have key ecosystem functions, or services, that support clean water and provide habitat for a variety of plants and animals. The status and trends in wetland cover are therefore important indicators of the health of the Delaware Estuary and Basin.

In Chapter 1, land cover classes were assessed broadly, with both forests and wetlands constituting "natural land cover." Here, in Chapter 6.3, we specifically summarize wetland cover patterns. In the following sections of this chapter, we further address the two main classes of wetlands in the Estuary and Basin: 1) tidal wetlands, which are characteristic of the Estuary portion of the watershed; and 2) non-tidal wetlands, which occur outside of tidal fluctuations throughout the Estuary and Basin. Additional descriptions of these wetland types, as well as their respective status and trends, are in Chapters 6.3.1 Tidal Wetlands and 6.3.2 Non-tidal Wetlands.

Present Status & Trends

The proportion of wetlands across the Delaware Estuary and Basin varies greatly by region and watershed (Fig 6.1). The Bay Region has the greatest proportion of wetlands (~22-36%) due to the expansive tracts of salt and brackish tidal wetlands in the Estuary. In the Lower region, where development is greatest within the Estuary and Basin, however, wetland cover is substantially less. Wetland cover is often <5% of the watersheds in southeastern PA, surrounding Philadelphia. Wetland cover is closer to 20% in southwestern NJ, where there is more suburbia as well as protections provided within the Pinelands area along the eastern edge of the UE2 watershed. Wetland cover ranges from ~6-13% in the Central Region and ~1-5% in the Upper Region, where wetland cover on the landscape is restricted due to elevation gradients within mountainous terrain.

On average from 1996-2016, the Lower Region experienced the greatest loss of wetland cover (-6.4%), followed by the Bay (-4.2%) and Central Regions (-2.7%) (Fig 6.2). The Upper Region experienced wetland cover gains (+6.2%). At the watershed scale, proportional wetland losses were greatest in the most developed watersheds, such as UE1, SV 3, and LE1, and especially in SV 2, which has intensifying development and agriculture (Chapter 1) (Fig 6.3). Wetland losses in the Bay Region were higher in Delaware (-1% for LE 2 and DB 1) than losses in New Jersey (~-0.4% for LE 3 and DB 2), perhaps due to greater increase in development in those areas of Delaware (>20%) compared to New Jersey (<20%).

NM 1 and EW 2 had notable wetland gains, although the mechanisms for these gains are unclear. It is possible that, with increasing development in these areas, stormwater pond creation could have an observable net effect on wetland cover, especially as wetland cover in those areas is naturally low—but this is highly speculative. In other areas with increasing development, like UC 1 and UC 2, where natural wetland cover is higher than NM 1 and EW 2, stormwater ponds gains may offset only a fraction of losses. Understanding the spatiotemporal patterns of stormwater pond creation is a critical need within the Delaware Estuary and Basin, and a need that is further described in the non-tidal wetland section of this chapter.

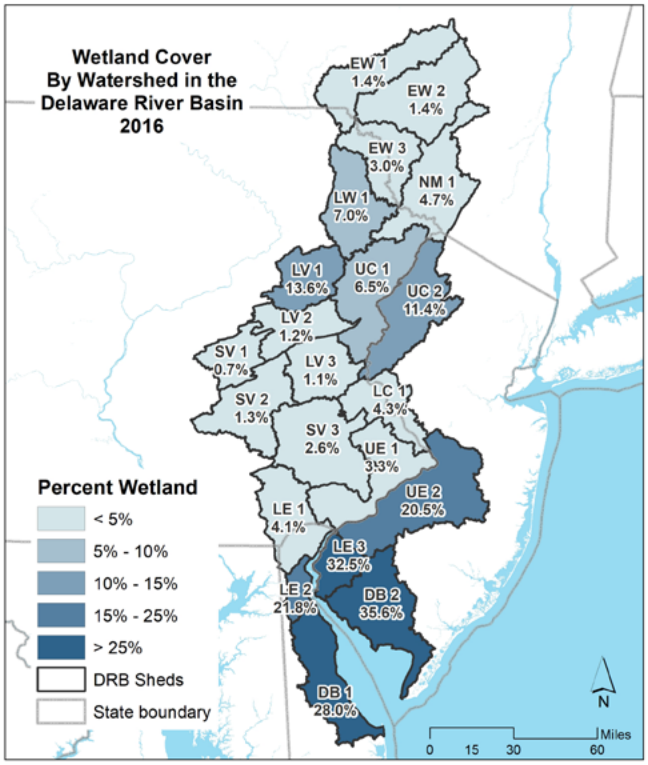

Figure 6.1 Wetland cover by watershed in the Delaware Estuary and Basin, 2016.

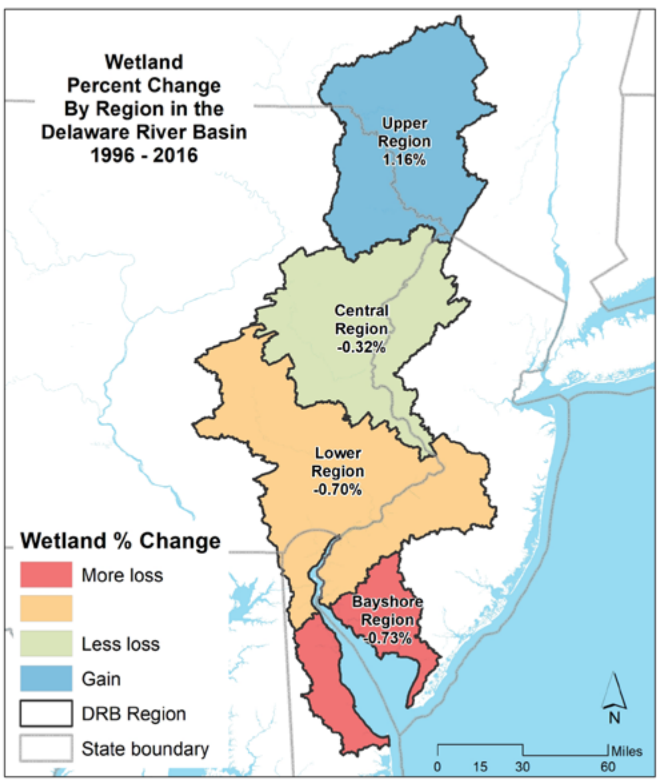

Figure 6.2 Wetland cover change by region in the Delaware Estuary and Basin, 1996-2016.

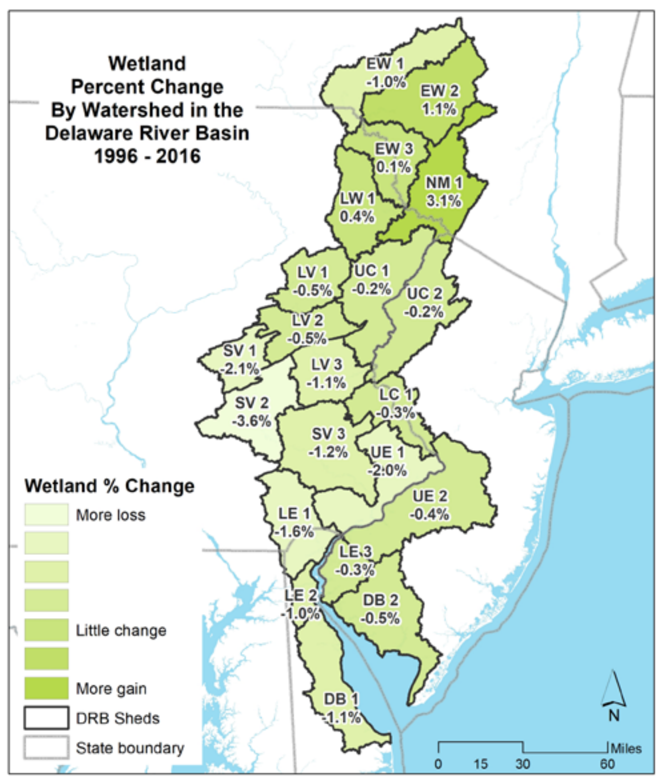

Figure 6.3 Wetland cover change by watershed in the Delaware Estuary and Basin, 1996-2016.

6.3.1 Tidal Wetland Cover

Abstract

Tidal wetlands are likely some of the most ecologically and economically important natural habitats. Data from the National Wetlands Inventory suggests that there are 125,447 hectares (309,986 acres) of tidal wetlands in the Delaware Estuary, which is ~8% of the total land area of the Estuary. More than half of the tidal wetlands in the Delaware Estuary are salt/brackish marshes (59%), with freshwater tidal marshes covering <5%. Between 1996–2016, however, tidal wetlands of the Delaware Estuary experienced a net decline of 340 hectares (840 acres), which is an average loss of 17 hectares (42 acres or 0.56%) per year, as NOAA C-CAP data suggested. Future projections imply that these losses will likely increase due to existing degradation and accelerating sea level rise, especially if robust intervention measures are not undertaken. Research, monitoring, proactive management, and on-the-ground actions are urgently needed to minimize ongoing losses.

Introduction

Tidal wetlands are vegetated aquatic habitats which occur in the intertidal zone between open water and upland areas not regularly exposed to tidal flooding. Tidal wetlands are among the most productive habitats in the world and are critical components of the interaction between land and water in the Estuary. They perform a wide variety of vital ecosystem services, such as protecting inland areas from tidal and storm damage; storing water, carbon, and other nutrients; providing important habitat to a wide variety of wildlife, including imperiled species of birds; filtering and storing contaminants to sustain water quality; providing spawning and nursery habitat for commercial fisheries; and supporting active and passive recreation and aesthetic value. Tidal wetlands are therefore regarded as the most critical habitat type in the Delaware Estuary for supporting broad ecological health and good water quality. Assuring that tidal wetlands remain intact and continue to provide these critical functions is therefore fundamental to the overall good quality of the Delaware Estuary and Basin as a whole.

Tidal wetlands occur within the tidal extent of the Delaware Estuary, spanning a broad salinity gradient from the head-of-tide near Trenton, New Jersey, down to the mouth of Delaware Bay at Cape May, New Jersey, and Cape Henlopen, Delaware. This area is contained within the Bay (DB1, DB2) as well as the Upper (UE1, UE2) and Lower Estuary (LE1, LE2, LE3) subregions of the Estuary region of the Delaware Estuary and Basin. In the Delaware Estuary, the largest portion of tidal wetlands are the salt marshes that fringe the Delaware Bay. These tidal wetlands are mostly salt marshes, dominated by smooth cordgrass (*Spartina alterniflora*), with some areas mixed with salt hay (*Spartina patens*) and salt grass (*Distichlis spicata*) (Fig 6.3.1A). In the upper stretches of many tidal creeks, nationally rare communities of freshwater tidal vegetation can be dominant wherever salt concentrations are below 3 ppt (Fig 6.3.1B). Upstream of the salt line (a location where water salinities average 0.25 ppt) and within tidal reaches, the Delaware River and its tributaries support fringing and expansive freshwater tidal wetlands in the Lower and Upper Estuary regions. These freshwater tidal wetlands mainly consist of marshes dominated by perennial grasses, sedges and rushes, but there are some scrub/shrub and forested tidal wetlands as well. Typically, freshwater tidal wetlands contain a greater number of species than salt marshes; a few diagnostic species are annual wild rice (*Zizania aquatica*), cattails (*Typha* sp.), dotted smartweed (*Polygonum punctatum*) and forbs such as spatterdock (*Nuphar advena*), wapato (*Sagittaria latifolia*) and tuckahoe (*Peltandra virginica*).

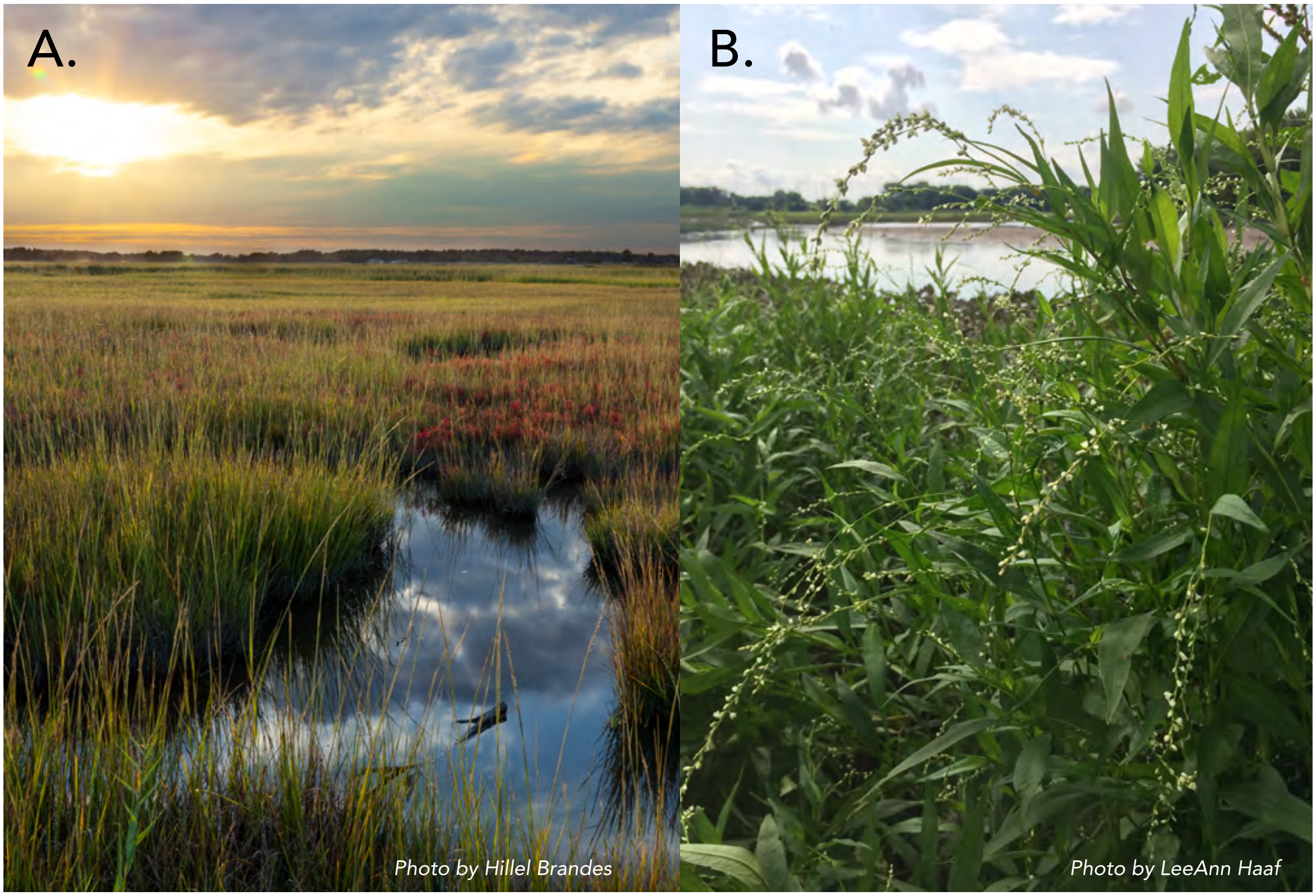

Figure 6.3.1 Tidal wetlands in the Delaware Estuary span a salinity gradient, which includes salt marsh (A) and freshwater tidal marsh (B).

Description of Indicator

In considering tidal wetland habitats one of the leading environmental indicators for the Estuary and Basin as a whole, the science and management community of the Delaware Estuary and Basin elevated tidal wetland extent as a top priority for monitoring and management (Kreeger et al. 2006). In this chapter, landscape-level data are synthesized to assess our best current understanding of tidal wetland composition in the Estuary, as well as understanding how extent varied over space and time. This landscape-level analysis also represents Tier 1 of the Mid Atlantic Coastal Wetland Assessment (MACWA), which is a multi-tiered, multi-partner program that coordinates research and assessment of coastal wetlands in the region (see Wetlands Feature 1- Mid-Atlantic Coastal Wetland Asssessment).

Methods

Status

National Wetlands Inventory: Data on wetland distribution were gathered for each state from the U.S. Fish and Wildlife Service (USFWS) National Wetlands Inventory (NWI). The NWI is a nationwide program that inventories wetlands of the United States through aerial imagery interpretation and ground-truthing. The NWI provides detailed, consistent, high resolution data that enables differentiation of wetland types; however, it is of limited value in trend analyses for the whole system because of the different times that data are collected in different states and areas. For instance, the latest NWI data in New Jersey are from approximately 2007, 2017 in Delaware, and 2015 in Pennsylvania. Although other mapping efforts have been carried out for the general region (e.g., Carr et al. 2018; Correll et al. 2019), NWI remains the most routinely assessed and updated wetland dataset for the entire Delaware Estuary.

To determine the current extent of the various types of tidal wetlands in the Estuary, the latest of each of three state-wide NWI wetlands were used. Wetland types were categorized using the classification scheme developed by Cowardin (Cowardin et al. 1979). A simplified classification was developed to allow for a synoptic assessment of status of broad categories of wetlands within the Estuary, with special attention to the differentiation of freshwater tidal and saltwater wetlands (Fig 6.3.2). Generally, tidal wetlands were classified as forest, shrub-scrub, emergent (both estuarine marsh and freshwater tidal marsh), riverine, lake, unconsolidated, and aquatic bed. Freshwater tidal wetlands were determined by isolating palustrine wetlands with tidal flood classifications (freshwater tidal flood classifications of S, R, V, and T), which served as the freshwater tidal footprint of the Estuary.

Trends

Coastal Change Analysis Program Determining landscape level changes in different wetland types of the Delaware Estuary requires consistent data in both space and time. Since NWI lacks temporal consistency, wetlands data were derived from the National Oceanic and Atmospheric Administration's (NOAA) Office of Coastal Management Coastal Change Analysis Program (C-CAP) datasets. These data are derived from Landsat imagery at a 30m ground resolution and are routinely assessed in 4–6 (typically 5) year intervals. Years for the C-CAP land cover data for the Delaware Estuary and Basin are 1996, 2001, 2006, 2010, and 2016.

C-CAP data are most useful for trend analyses as they are not as resolved as NWI (C-CAP is assessed at the 1:100,000 scale, whereas NWI are assessed at the 1:24,000), and may have larger classification errors. Previous assessment of the comparability of the wetland categories of the C-CAP land cover data with NWI indicates that the data are comparable to a relatively small percentage difference, particularly for salt/brackish wetlands (i.e., estuarine emergent) wetlands. As with NWI, accuracy/precision issues among various mapping methods have been noted (Weis et al. 2021), yet C-CAP remains the most methodologically consistent dataset for temporal trend analysis of wetlands for the entire Delaware Estuary. Therefore, C-CAP data were used to assess the Trends in tidal wetlands for this report, whereas NWI data were used to determine Status.

Categories of wetlands distinguished by the C-CAP are: Palustrine Forested, Palustrine Scrub/Shrub, Palustrine Emergent, Estuarine Forested, Estuarine Scrub/Shrub, Estuarine Emergent, Unconsolidated Shore, and Palustrine Aquatic Bed. Here, classifications beginning with Estuarine are considered salt or brackish tidal wetlands. At approximately the salt front, classifications consist of Palustrine categories, even though freshwater tidal marshes exist within this corridor. Therefore, the freshwater tidal footprint derived from NWI data was used to further isolate freshwater tidal wetlands for C-CAP trend analysis.

Results

Status

Tidal wetlands (freshwater tidal, brackish, and saline) cover 125,447 hectares (309,986 acres), which is about 8% of the total land area of the Estuary (~1.6 million hectares) (Fig 6.3.4). This is comparatively higher than the national figure of 5.5% area of wetlands in the contiguous U.S. (Dahl 2011). Emergent salt and brackish marshes account for 59% (73,390 hectares or 181,351 acres) and freshwater tidal marshes account for 4.5% (5,630 hectares or 13,912 acres) of all tidal wetlands in the Estuary. Intertidal and subtidal unconsolidated habitats (beaches, shorelines, mudflats) are 20% of tidal wetlands in the Estuary (24,730 hectares or 61,109 acres), while the remaining 16.5% of tidal habitats consist of tidal forested and shrub-

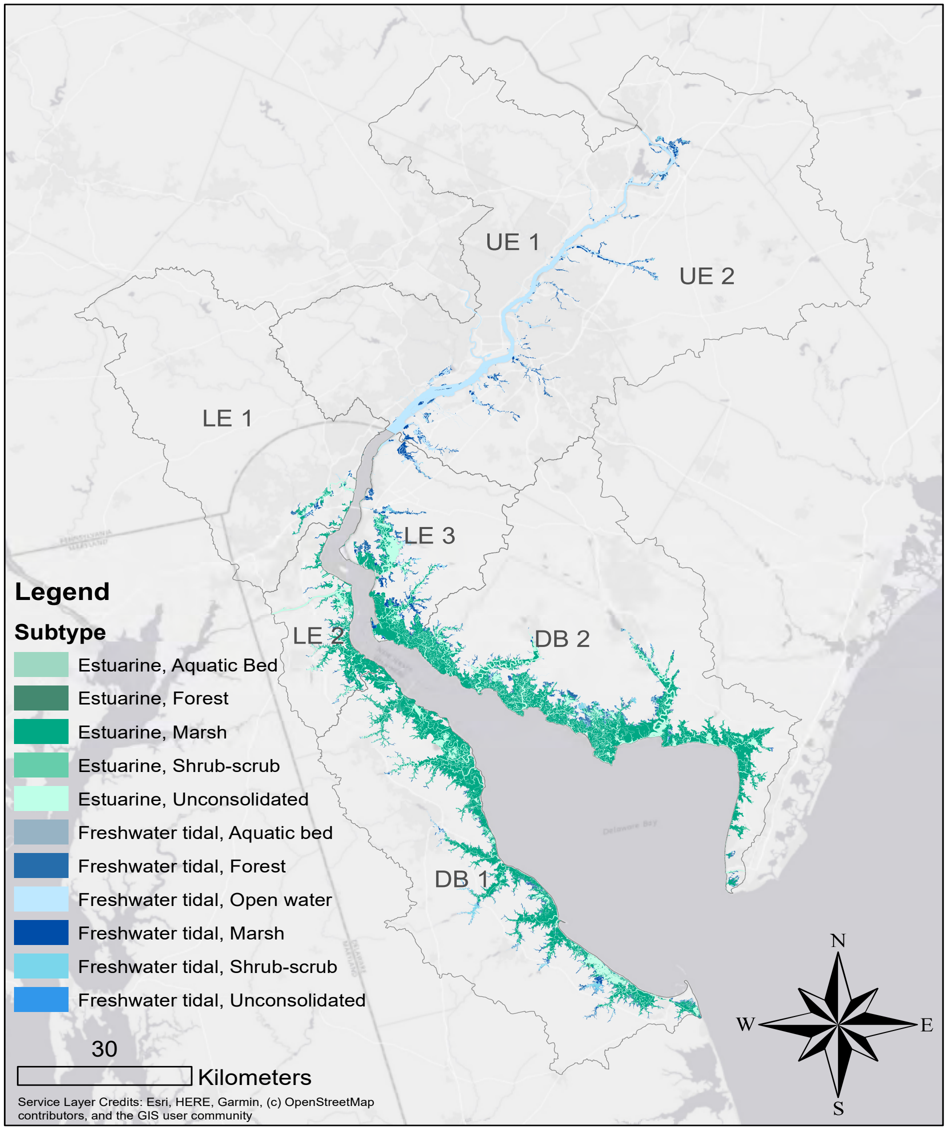

Figure 6.3.2 Tidal wetland cover in the Delaware Estuary based on the most recent NWI data.

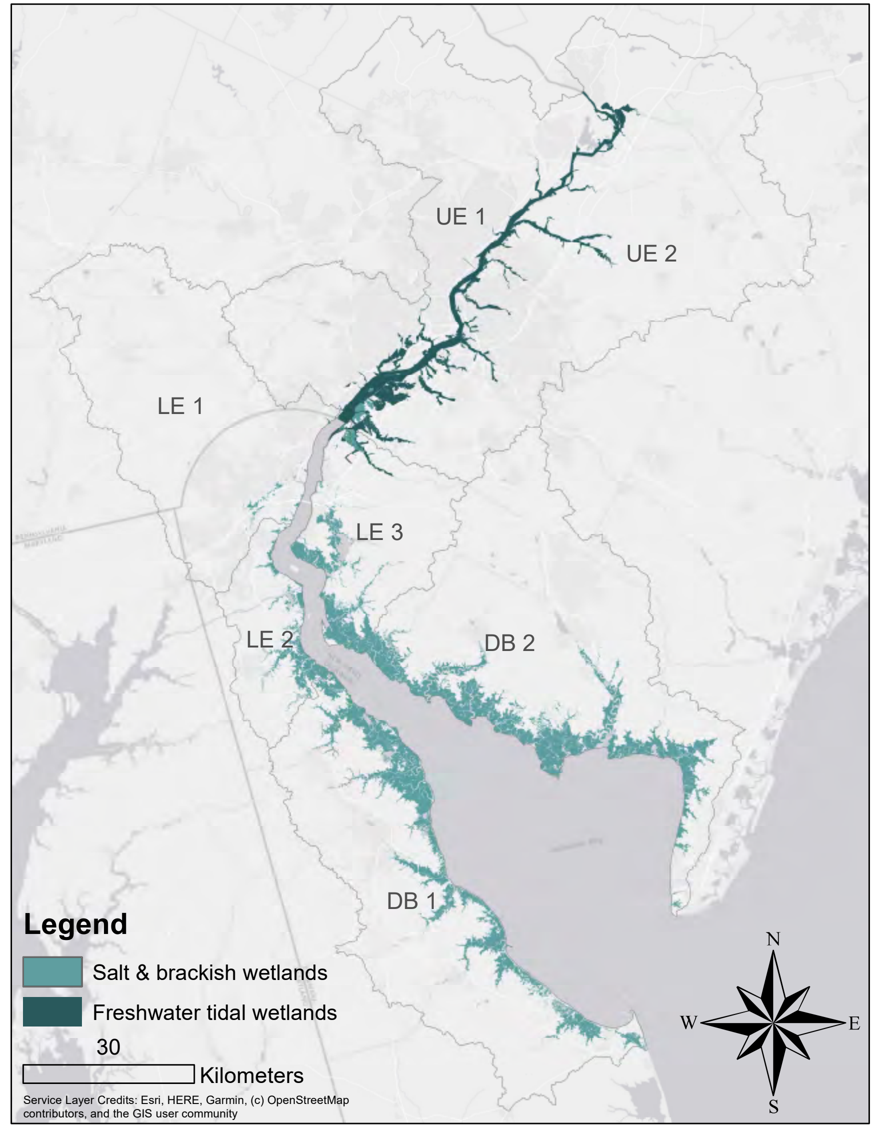

Figure 6.3.3 Tidal wetland cover in the Delaware Estuary based on 2016 NOAA C-CAP data.

scrub habitats. In the following figures, summary information on tidal wetland acreage based on the latest NWI data was divided by subregion (Fig 6.3.4, Tables 6.3.1-6.3.2) and state (PA, NJ, and DE; Fig 6.3.5, Table 6.3.3).

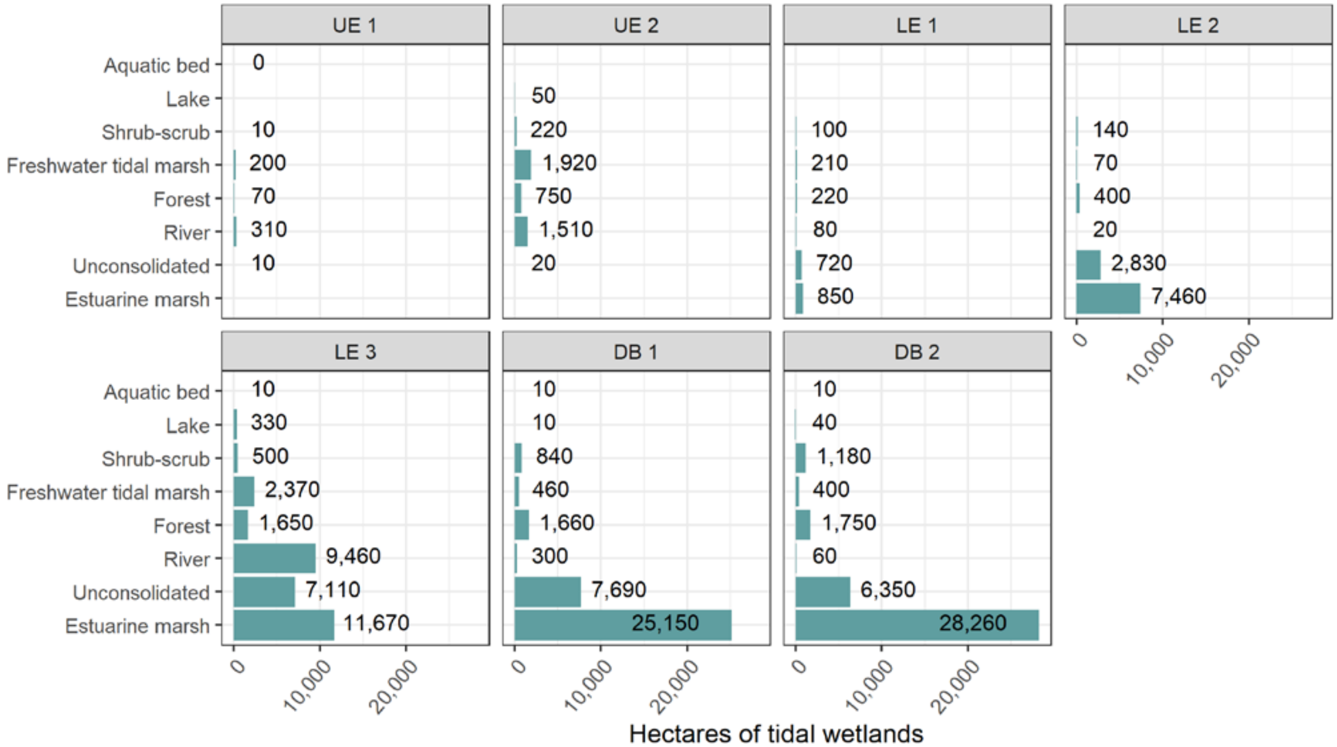

Figure 6.3.4 Tidal wetland cover (in hectares) of tidal wetland types by region based on NWI classifications.

Trends

Between 1996–2016, tidal wetlands of the Delaware Estuary experienced a net decline of 340 hectares (840 acres), which is an average decline of 17 hectares (42 acres or 0.56%) per year (Figs 6.3.6-6.3.8; Table 6.3.4). The largest areas of loss were in the New Jersey Bayshore (DB2), which saw a decrease of 354 hectares (501 acres), and Delaware Bayshore (DB1), which saw a decrease of 52 hectares (128 acres). Freshwater tidal wetlands in the Upper Estuary, as well as portions of the Lower Estuary, also experienced declines ranging from 11 to 37 hectares (27–91 acres). Net change from 1996–2016 showed a very small increase (0.18 hectare or 0.44 acre) in tidal wetland acreage in subregion LE1, in northern Delaware. Although losses in the upper estuary add up to smaller acreage, losses are proportionately larger; for instance, a loss of 17 hectares from 1996–2016 in UE1 translates to a 2.4% average annual decline, whereas 203 hectares lost is a <1% average annual decline for DB2.

Freshwater tidal wetland losses totaled 1,023 hectares (2,528 acres) between 1996–2016 in the Estuary, which were mostly driven by conversion to development, open space/agriculture/grasslands, and forest/shrub-scrub habitats (Table 6.3.5). These losses were countered by 506 hectares (875 acres) of freshwater tidal wetland gains, which were driven mostly by conversion from open water, bare or unconsolidate land, and forest/shrub-scrub habitats. Brackish/salt wetland losses in the Estuary totaled 568 hectares (1,404 acres), which were driven by conversion to open water and unconsolidated/bare land (Tables 6.3.3, 6.3.4). These losses were countered by 341 hectares (843 acres) of gains, which were driven by conversion from open water and non-tidal wetlands. Taken together, there was a net loss of 509 hectares (1,258 acres) of freshwater tidal wetlands and 226 hectares (558 acres) of brackish/salt wetland in the Delaware Estuary during the twenty year study period.

Table 6.3.1 Cover of tidal wetlands in the Delaware Estuary in the Lower Region based on NWI data.

Subregion	Type	Hectares	% of tidal wetlands	% of subregion area
UE 1	Aquatic bed	1	0.239%	0.001%
	Forest	72	11.7%	0.039%
	Freshwater tidal marsh	203	33.2%	0.112%
	Riverine	311	50.7%	0.171%
	Shrub-scrub	14	2.29%	0.008%
	Unconsolidated	12	1.93%	0.007%
UE 2	Forest	748	16.8%	0.277%
	Freshwater tidal marsh	1,920	42.9%	0.710%
	Lake	52	1.17%	0.019%
	River	1,510	33.8%	0.558%
	Shrub-scrub	222	4.97%	0.082%
	Unconsolidated	17	0.388%	0.006%
LE 1	Estuarine marsh	851	39.1%	0.544%
	Forest	220	10.1%	0.141%
	Freshwater tidal marsh	207	9.49%	0.132%
	Riverine	83	3.80%	0.053%
	Shrub-scrub	96	4.43%	0.062%
	Unconsolidated	719	33.0%	0.460%
LE 2	Estuarine marsh	7,460	68.4%	18.6%
	Forest	404	3.70%	1.007%
	Freshwater tidal marsh	66	0.600%	0.163%
	Riverine	17	0.158%	0.043%
	Shrub-scrub	140	1.28%	0.348%
	Unconsolidated	2,830	25.9%	7.05%
LE 3	Aquatic bed	9	0.028%	0.014%
	Estuarine marsh	11,700	35.3%	17.1%
	Forest	1,650	4.98%	2.41%
	Freshwater tidal marsh	2,370	7.16%	3.47%
	Lake	334	1.01%	0.490%
	Riverine	9,460	28.6%	13.9%
	Shrub-scrub	495	1.50%	0.726%
	Unconsolidated	7,110	21.5%	10.4%

Table 6.3.2 Cover of tidal wetlands in the Delaware Estuary in the Bay Region based on NWI data.

Subregion	Type	Hectares	% of tidal wetlands	% of subregion area
DB 1	Aquatic bed	7	0.020%	0.004%
	Estuarine marsh	25,200	69.6%	15.3%
	Forest	1,660	4.60%	1.01%
	Freshwater tidal marsh	462	1.28%	0.281%
	Lake	11	0.0290%	0.006%
	Riverine	301	0.83%	0.183%
	Shrub-scrub	836	2.31%	0.508%
	Unconsolidated	7,690	21.3%	4.68%
DB 2	Aquatic bed	5	0.0140%	0.003%
	Estuarine marsh	28,200	74.2%	13.8%
	Forest	1,750	4.60%	0.856%
	Freshwater tidal marsh	404	1.06%	0.198%
	Lake	43	0.113%	0.021%
	Riverine	61	0.161%	0.030%
	Shrub-scrub	1,180	3.09%	0.576%
	Unconsolidated	6,350	16.7%	3.10%

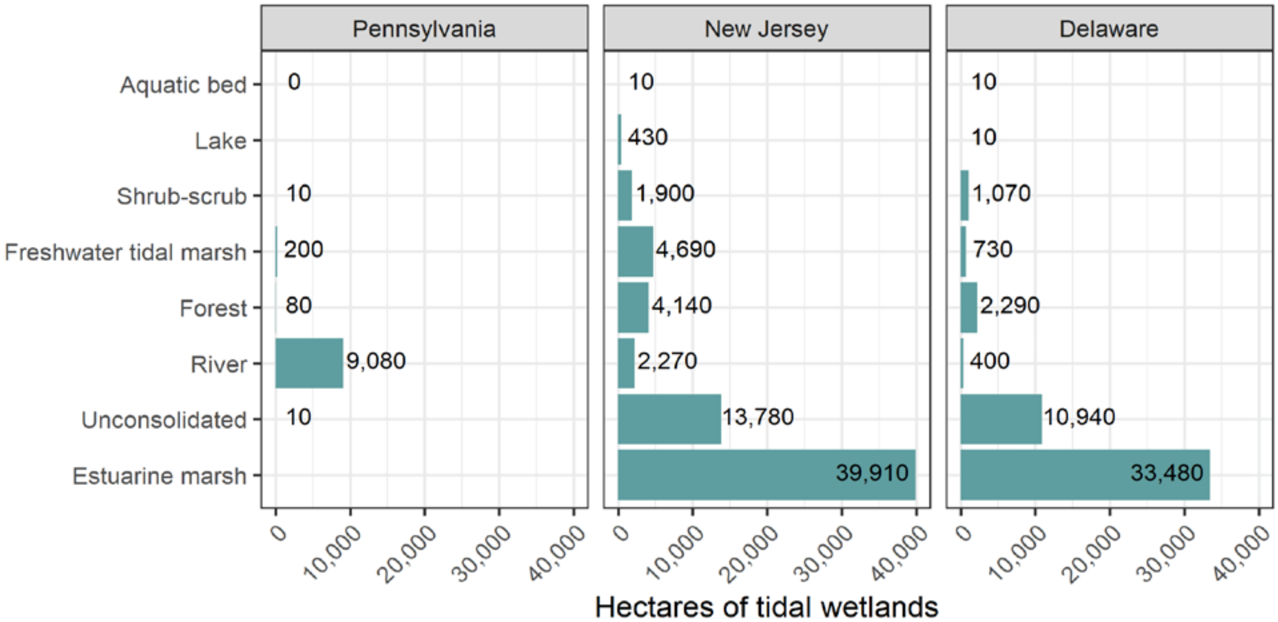

Figure 6.3.5 Tidal wetland cover (in hectares) of tidal wetland types by state based on NWI classifications.

Table 6.3.3 Cover of tidal wetlands in the Delaware Estuary by state based on NWI data.

State	Type	Hectares	% of wetlands
Pennsylvania	Aquatic bed	1	0.0156%
	Forest	78	0.832%
	Freshwater tidal marsh	203	2.17%
	Riverine	9,080	96.7%
	Shrub-scrub	14	0.144%
	Unconsolidated	12	0.126%
New Jersey	Aquatic bed	15	0.0217%
	Estuarine marsh	39,900	59.5%
	Forest	4,140	6.17%
	Freshwater tidal marsh	4,690	6.99%
	Lake	430	0.640%
	Riverine	2,270	3.38%
	Shrub-scrub	1,900	2.83%
	Unconsolidated	13,800	20.5%
Delaware	Aquatic bed	7	0.02%
	Estuarine marsh	33,500	68.4%
	Forest	2,290	4.67%
	Freshwater tidal marsh	734	1.50%
	Lake	11	0.0218%
	Riverine	401	0.820%
	Shrub-scrub	1,070	2.19%
	Unconsolidated	10,940	22.4%
Total	Aquatic bed	23	0.0186%
	Estuarine marsh	73,400	58.5%
	Forest	6,505	5.19%
	Freshwater tidal marsh	5,630	4.49%
	Lake	440	0.351%
	Riverine	11,700	9.36%
	Shrub-scrub	2,980	2.38%
	Unconsolidated	24,700	19.7%
	All	125,500	-

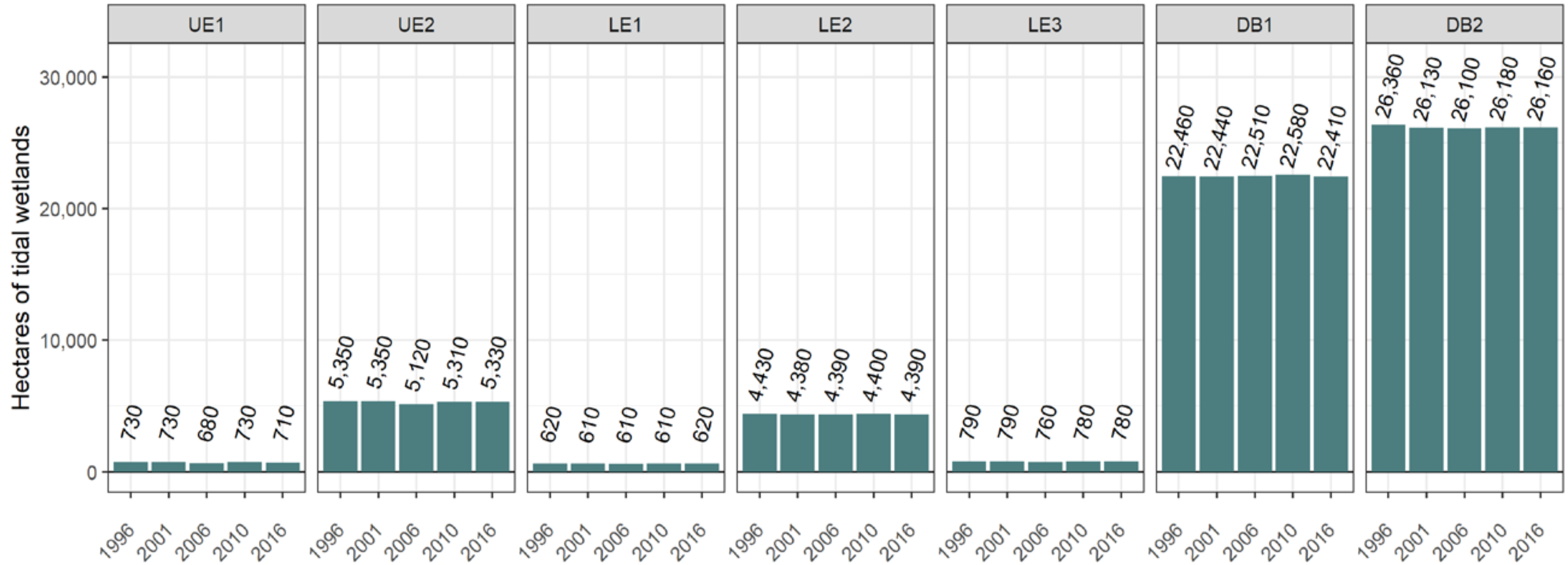

Figure 6.3.6 Total tidal wetland cover in the Estuary over time by subregion

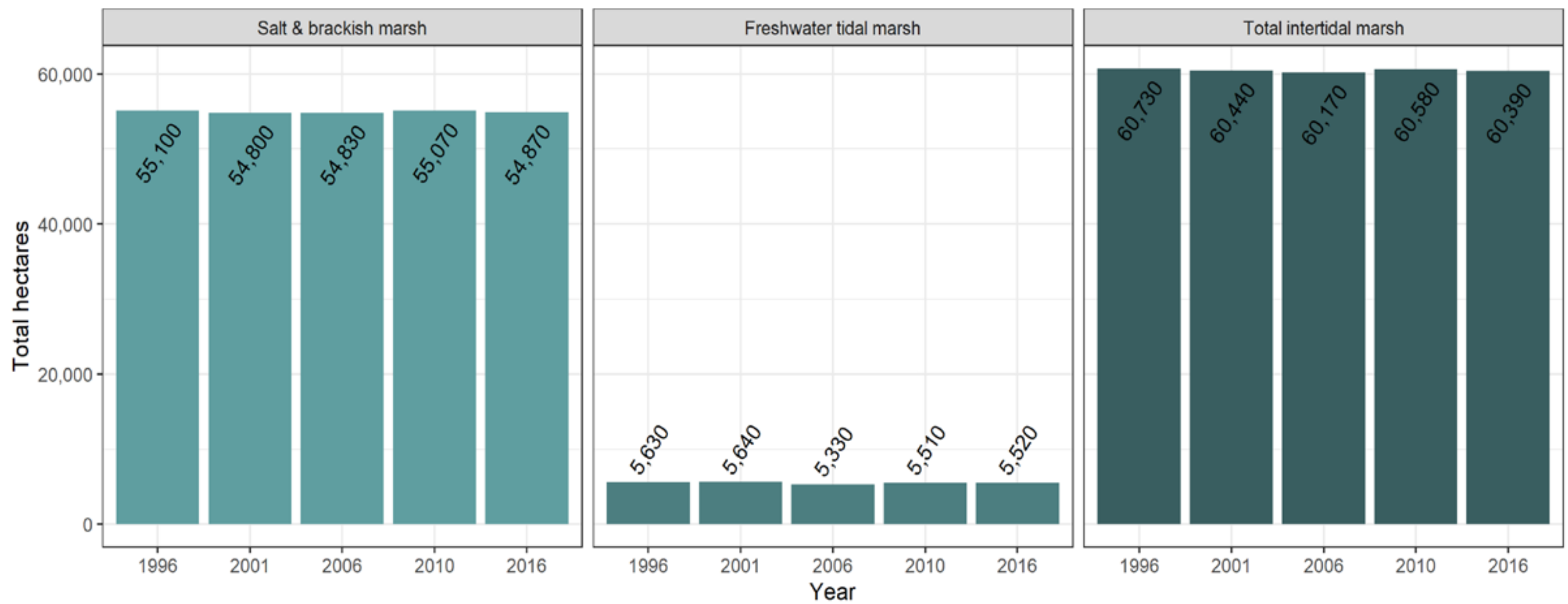

Figure 6.3.7 Total tidal wetland cover in the Estuary over time by type.

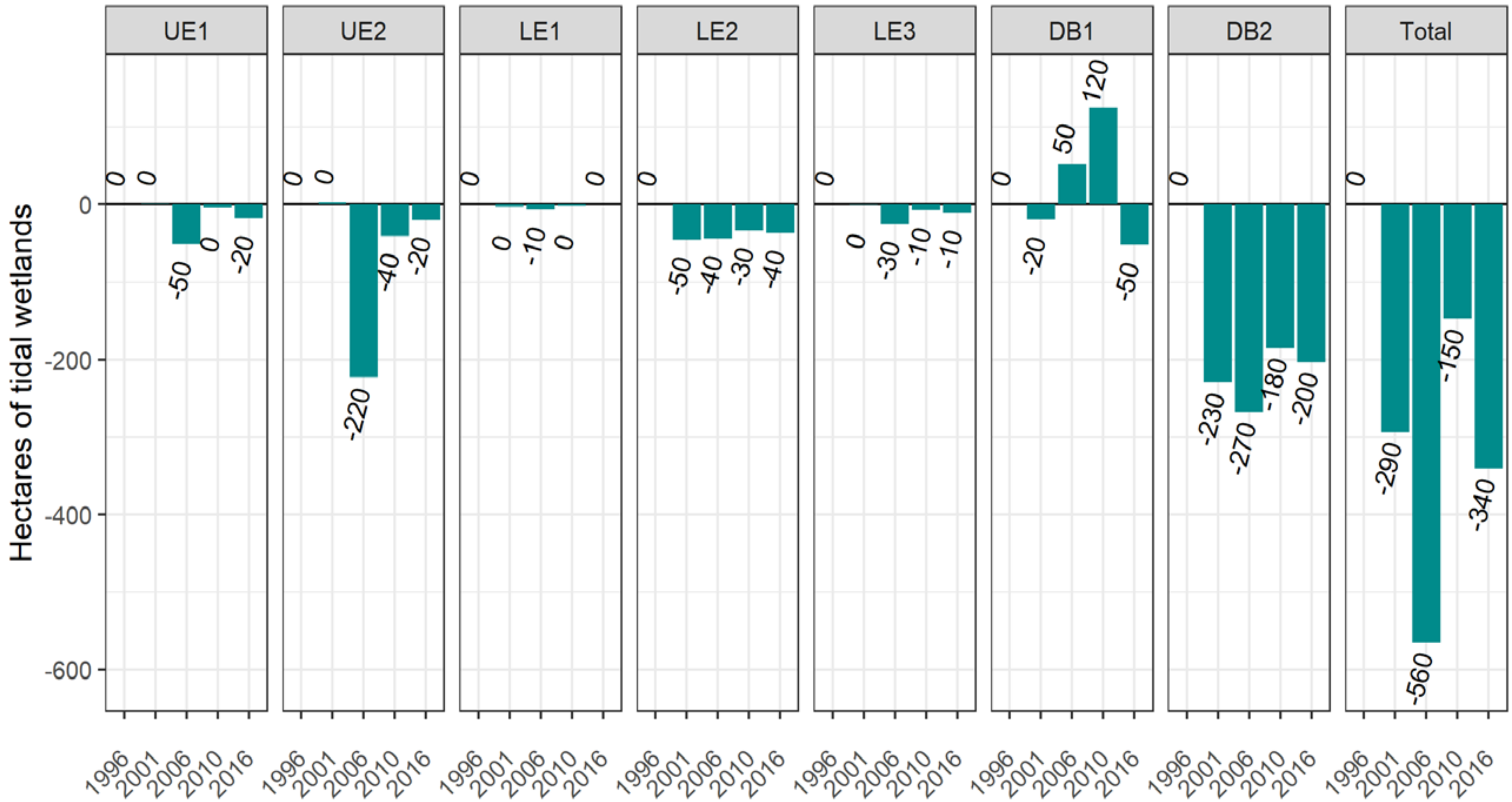

Figure 6.3.8 Tidal wetland cover changes relative to the 1996 baseline.

Table 6.3.4 Tidal wetland cover change from 1996-2016.

Subregion	Hectares		Net change	
	1996	2016	ha	%
UE1	731	713	-17	-2.4%
UE2	5,346	5,326	-20	-0.37%
LE1	615	615	0.18	0.029%
LE2	4,429	4,392	-37	-0.83%
LE3	786	775	-11	-1.4%
DB1	22,458	22,406	-52	-0.23%
DB2	26,363	26,160	-203	-0.77%
Total	60,730	60,390	-340	-0.56%

Table 6.3.5 Conversions of tidal wetlands from 1996 to 2016.

Wetland Type	Land use type	Loss		Gain		Net	
		ha	%	ha	%	ha	%
Freshwater tidal	Development	236	17.4%	-	-	-236	-17.4%
	Open space, agricultural, & grassland	328	24.1%	61	4.50%	-267	-19.6%
	Forest & scrub-shrub	209	15.4%	110	8.10%	-99	-7.30%
	Unconsolidated & bare	95	7.00%	44	3.20%	-51	-3.80%
	Open water	93	6.90%	139	10.20%	46	3.30%
	Aquatic bed	62	4.60%	-	-	-62	-4.60%
Salt & brackish marsh	Development	39	0.06%	-	-	-39	-0.06%
	Open space, agricultural, & grassland	74	0.12%	0.4	0.001%	-74	-0.12%
	Forest & scrub-shrub	1	0.002%	-	-	-1	0.002%
	Unconsolidated & bare	116	0.18%	2	<0.001%	-114	-0.18%
	Open water	337	0.53%	236	0.37%	-101	-0.16%
	Aquatic bed	0.5	0.001%	-	-	-0.5	0.001%
	Non-tidal wetland	-	-	103	0.16%	103	0.16%

Discussion

Tidal wetlands were historically lost in the Delaware Estuary primarily through their reclamation for agriculture and other purposes. Carr et al. (2018) estimated that the Delaware Estuary might have lost 22,400 hectares (55,352 acres) of tidal wetlands from 1776 to 2011. The most extensive losses occurred mostly before 1950. Direct losses of wetlands have slowed since ~1975, likely due to protections afforded by provisions in the 1972 Clean Water Act. Since 2000, total tidal wetland acreage has oscillated between approximately 60,000 and 70,000 hectares (148,000-173,000 acres) (Carr et al. 2018; this study).

As no other habitat types rival tidal wetlands in productivity, the net loss of ecosystem services are disproportionately large compared to acreage losses. Although interannual variability exists and some gains can be noted, all tidal wetland acreage losses are nevertheless impactful considering their benefits to people, fish and wildlife, and water quality. Historical and current human-mediated disturbances on estuarine systems are considerable, and development pressures are likely to continue to increase (see Chapter 1). These stressors are additionally exacerbated by climate change and sea level rise.

Mechanisms of Loss

Rising sea level is a significant contributor of tidal wetland decline in the Delaware Estuary (Carr et al. 2018). Tidal wetlands naturally have the ability to keep pace with rising sea levels through feedbacks that result in the accumulation of mineral and/or organic materials. These feedbacks, however, can be outpaced when sea level rise surpasses biological and geomorphological thresholds. When outpaced, tidal wetlands drown and convert to mudflats or open water. Nationally, 96.4% of tidal wetland losses were due to conversion to open water, with about 3.5% attributable to human effects in upland areas (Stedman and Dahl 2008). This likely holds true for the Delaware Bay, as Kearney et al. (2002) and Kearney and Riter (2011) discerned through satellite imagery that marshes in the Delaware Bay had decreasing vegetative cover and increasing proportions of open water over time. Coastal managers in Delaware and New Jersey also report rapidly expanding interior open water in the Delaware Estuary.

Anthropogenic stressors have contributed directly to degradation of tidal wetlands and, therefore, have likely contributed to their losses in the Delaware Estuary. Degradated tidal wetlands are less resilient to perturbations or stressors, and more likely to erode, drown, and deteriorate on short time scales. Coastal wetland stressors include a mix of practices such as mosquito control ditching, continued incremental filling, lack of regulatory oversight, regulatory loopholes for developers, shoreline hardening, hydrological alterations such as dredging, and pollution. Hydrological alterations, fill, diking, ditching, among numerous other stressors, have been directly observed from on-the-ground inventories of tidal wetland conditions in the Delaware Estuary (see Mid Atlantic Tidal Rapid Assessment reports). Historical diking for salt hay farming has led to low relative elevations of many tidal wetlands in New Jersey, which decreases their resilience to sea level rise and increases the probability that they convert to open water. As a result, ~3,600 hectares of New Jersey salt marsh (in LE3, DB2) converted to open water as a direct result of historical diking from 1931 to 2017 (Smith et al. 2017). Kearney et al. (2002) also found that more than two-thirds of the salt marshes studied in both the Chesapeake and Delaware Bays were in degraded condition. Additionally, nutrient loading might reduce the need for ample below ground production, potentially impairing a marsh's ability to keep pace with sea level rise (Deegan et al. 2012).

Tidal marshes need ample sediment supplies to keep pace with sea level, so regional sediment management is also a concern for tidal marsh sustainability in the Delaware Estuary. The Delaware Estuary is a naturally muddy system, but more sediments are removed each year through maintenance dredging than enter the system through surface runoff. The overall budget (inputs and outputs of sediments at the Estuary scale) appears to be in balance despite regular sediment removal during channel dredging, and so, it is likely that the budget is subsidized by inputs of sediments from eroding or disintegrating tidal wetlands (Delaware Estuary Regional Sediment Management Plan 2013). Sediment management in the

Estuary should continue to consider how to retain sediment within the system, such as through thin layer placement or other types of beneficial reuse of dredged materials, in order to provide tidal wetlands the provisions they need to keep pace with rising sea levels and compensate for, or mitigate, erosional losses.

Prognosis

The rate of relative sea level rise (SLR) is critically important for determining the fate of tidal wetlands in the Delaware Estuary. SLR is currently estimated between 4-5 mm/yr in the Delaware Estuary (see Chapter 2). The upper threshold of SLR at which tidal marshes can build vertically, however, is about 10 mm/yr (D'Alpaos et al. 2011). SLR is likely to rise ~1.5 meters from 2000 to 2100 under high carbon dioxide emissions, which averages approximately 15 mm/yr (Callahan et al. 2017; Kopp et al. 2019). A new NOAA report further suggests that, under a high emissions scenario, the Northeastern U.S. may see 0.54 m of rise between 2000 and 2050 (~10.8 mm/yr), but rise might exceed 2 m from 2000 to 2100 (20 mm/yr) (Sweet et al. 2022). These sea level rise projections suggest that the limit at which tidal marshes can keep pace with SLR will likely be breached before 2100 unless significant actions are taken to aid the vertical accretion and migration of tidal wetlands.

As sea levels rise and thresholds are reached, declines in tidal wetland acreage are expected. Based on Sea Level Affecting Marsh Model (SLAMM, V.6) predictions for 2100 with 1 m of SLR, a net loss of 18,000 hectares of tidal wetlands were predicted (Kassakian et al. 2017). As the Estuary has approximately 125,000 hectares of tidal wetlands based on NWI data, this would be about a 14% loss. Other recent SLAMM modeling of focal Delaware Bay salt marshes suggest that total marsh gains might first increase, subsidized by the expansion of regularly flooded areas, until thresholds in SLR are reached (between 0.8-1.0 m of SLR). Thereafter, total declines are likely to become evident (Stamp et al. 2019).

As these SLAMM runs were augmented using on-the-ground data from MACWA efforts, it is apparent that site-specific tidal wetland conditions play critical roles in the sustainability of tidal wetlands with continued, and likely accelerating, SLR (Stamp et al. 2019). In a linear model analysis of tidal wetland acreage trends relative to sea level, Carr et al. (2018) found that for every 1 cm rise in sea level, wetland area historically declined by 169 hectares (418 acres). Given current rates of SLR (~5 mm/yr), 169 hectares would be lost every 20 years. However, C-CAP data here suggests that 340 hectares of tidal wetlands have been lost in the 20 years from 1996 to 2016. Sea level rise acceleration in combination with degraded conditions, driven human disturbances in 19th and/or 20th centuries, have likely contributed to these additional losses.

Climatic changes, such as warming temperatures and altered precipitation patterns, will also likely affect tidal wetland extent in the Delaware Estuary (see Chapter 2). On one hand, a longer growing season and warmer temperatures likely will increase total tidal marsh productivity (Kirwan et al. 2009). Warming trends are also expected to increase the incidence and intensity of coastal storms, including nor'easters and hurricanes. Storm-related damages to tidal wetlands, such as excessive erosion, submergence, and salt intrusion could exacerbate other threats and stressors mentioned above. Storms may provide pulses of sediment to tidal wetlands that help them build vertically, but damages may outweigh benefits, especially in already degraded or fragmented wetlands.

Moreover, tidal wetlands face barriers to landward migration within the Delaware Estuary. The potential for tidal wetlands to migrate landward is affected by slope, soils, degree of development, and other ecological considerations (such as upland habitat plant community composition). Areas that do not allow tidal wetlands to migrate landward must accrete in place and stave off excessive erosion to preserve acreage or drown. Despite that forest conversion to marsh is a leading hypothesis of future salt marsh gains, C-CAP data in the Delaware Estuary suggested that no salt/brackish marsh net gains were caused directly by upland forest conversion. However, ongoing research in the Delaware Estuary has found that forested areas in the region are indeed converting to salt marshes as sea levels rise (Smith 2013; Sacatelli et al. 2020; Feature 2 - Coastal Forest Die Back). In this report, gains appeared to be most notably driven by the conversion of open water and non-tidal wetlands, which also includes forested wetlands (this study; Osland et al. 2022). Patterns of non-tidal wetland conversion, emergent and forested, will therefore be an important area of additional consideration and research. More research on processes that drive marsh migration is needed to help support planning for coastal land use adaptations as rising sea levels and tidal wetland losses cause land cover transitions.

Actions and Needs

Sea level rise, salinity intrusion, anthropogenic disturbance, outdated management paradigms, and pollutants will likely continue to contribute to degradation and loss of tidal wetlands in the Delaware Estuary unless very swift actions are taken to abate these impacts. Managers should carefully consider how a projected loss of tidal wetlands in the Delaware Estuary might affect coastal communities (lives and property) and regional economies (fisheries and shellfisheries, property values, nutrient criteria for industry). The following are actions and needs to aid efforts to reduce tidal wetland losses:

Proactive Adaptive Management

Despite the dynamic nature of the coastline, many regulatory policies continue to treat the landscape as fixed in place. Restoration paradigms set goals based on historic conditions rather than future sustainability. It is generally still easier to obtain a permit for a bulkhead or other hard structures, which can contribute to degradation of tidal wetlands, than it is for a living shoreline. The state of Delaware had taken a lead in making living shoreline permitting and construction easier, and is now being followed by New Jersey and Pennsylvania. Ditching, diking, excavating, and filling of tidal wetlands still occur, often without a good understanding or monitoring of the consequences. New active policies and tactics are needed to both facilitate the horizontal migration and vertical accretion of tidal marshes. Marsh migration management plans are needed and will require conflict resolution and education. To adapt to both climate change and continued watershed development, tidal wetland managers and landowners will need to adjust targets, expectations, and tactics to sustain the most tidal wetland habitat in the future.

In order to address threats to tidal wetlands in Delaware Estuary, an approach combining policy and regulatory remedies and actions on the ground is still needed. The 1972 Clean Water Act, the 1972 Coastal Zone Management Act, and the 1982 Coastal Barriers Resources Act, are evidence of the importance of tidal wetlands in the policy and legal arena. Many states and counties have followed the lead of federal agencies and implemented their own regulations covering such wetland protection measures as buffer requirements, impervious cover limitations, and implementation of federal National Pollutant Discharge Elimination System (NPDES) and total maximum daily load (TMDL) guidelines.

Continued promulgation and refinement of regulations and policies is a critical need, as demonstrated by the various emergency measures that are already underway or being called for in some Delaware and New Jersey areas (e.g. Prime Hook, Delaware; Sea Breeze, New Jersey; Maurice Township, New Jersey) where tidal wetland losses are contributing to the decline of coastal communities. Given accelerating development and population pressures, as well as increases in relative sea level rise and climate change, these measures will need to be augmented to maintain the current integrity of the tidal wetlands. In particular, local differences in the extent of regulatory protection for wetlands poses a challenge to maintaining consistently high levels of wetland quality and function in the Estuary.

Continued Monitoring and Scientific Study

Another need for protecting tidal wetland extent is continued and complete data on their current conditions, as well as scientific information on factors that drive losses and best management practices to avert such losses. Although monitoring efforts have been underway through MACWA efforts (Feature 1 - Mid-Atlantic Coastal Wetland Assessment), the synthesis and continued support of these programs is critical in understanding the complexities that affect tidal wetland condition, extent, and sustainability into the future. These data are also useful for prioritizing and planning intervention strategies across the Estuary.

Investment in tidal marsh monitoring and science is difficult to fund at the multi-state scale of the Delaware Estuary. However, the benefits of tidal wetlands are beginning to be captured and capitalized (e.g. flood protection, nutrient and carbon capture, fish production; see Feature 3 - Blue Carbon). As markets for ecosystem services develop in the future, there could be increasing demand for essential information on trends in tidal wetland extent and condition, as well as tactics to protect and enhance tidal wetlands. However, until markets evolve that can generate needed resources to sustain monitoring and assessment there will continue to be a need to collaborate and leverage funds to fill vital information gaps.

On-the-ground Action

Given the rapid pace of change in tidal wetland extent and health, swift action to physically protect or enhance tidal wetlands is warranted to stem losses even if monitoring and scientific information are still developing. Seaward protections and marsh enhancements can be difficult to implement due to permitting, logistical and funding challenges. However, there are efforts to explore beneficial use of sediments for enhancement (Delaware Estuary Regional Sediment Management 2013), develop new types of hybrid living shorelines and craft estuary-wide strategies for intervention implementation (e.g., Delaware Estuary Living Shoreline Initiative).

New prioritization and decision-making tools are becoming available to facilitate the design and implementation of invention projects using on-the-ground data, which will improve success rates. In the Delaware Estuary, these tools include New Jersey Coastal Resilience Collaborative Prioritization Tool, the New Jersey Coastal Ecological Restoration and Adaptation Plan, as well as the Living Shoreline Feasibility Model, the Wetland Assessment Tool for Condition and Health, and Marsh Futures.

References

Callahan, John A., Benjamin P. Horton, Daria L. Nikitina, Christopher K. Sommerfield, Thomas E. McKenna, and Danielle Swallow. 2017. Recommendation of Sea-Level Rise Planning Scenarios for Delaware: Technical Report, prepared for Delaware Department of Natural Resources and Environmental Control (DNREC) Delaware Coastal Programs.

Carr, Edward W., Yosef Shirazi, George R. Parsons, Porter Hoagland, and Christopher K. Sommerfield. 2018. Modeling the Economic Value of Blue Carbon in Delaware Estuary Wetlands: Historic Estimates and Future Projections. Journal of Environmental Management 206: 40–50. https://doi.org/10.1016/j.jenvman.2017.10.018.

Correll, Maureen D., Wouter Hantson, Thomas P. Hodgman, Brittany B. Cline, Chris S. Elphick, W. Gregory Shriver, Elizabeth L. Tymkiw, and Brian J. Olsen. 2019. Fine-Scale Mapping of Coastal Plant Communities in the Northeastern USA. Wetlands 39. Wetlands: 17–28. https://doi.org/10.1007/s13157-018-1028-3.

Cowardin, Lewis M., Virginia Carter, Francis C. Golet, and Edward T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. FGDC-STD-004-2013. Second Edition. https://doi.org/FWS/OBS-79/31.

D'Alpaos, Andrea, S. M. Mudd, and Luca Carniello. 2011. Dynamic response of marshes to perturbations in suspended sediment concentrations and rates of relative sea level rise. Journal of Geophysical Research: Earth Surface 116: 1–13. https://doi.org/10.1029/2011JF002093.

Dahl, T.E. 2011. Status and Trends of Wetlands in the Conterminous United States 2004 to 2009. U.S. Department of the Interior; Fish and Wildlife Service, Washington, D.C.: 108.

Deegan, Linda a., David Samuel Johnson, R. Scott Warren, Bruce J. Peterson, John W. Fleeger, Sergio Fagherazzi, and Wilfred M. Wollheim. 2012. Coastal eutrophication as a driver of salt marsh loss. Nature 490: 388–392. https://doi.org/10.1038/nature11533.

Delaware Estuary Regional Sediment Management Plan (DERSMP). 2013. Prepared by the Delaware Estuary Regional Sediment Management Plan Workgroup. http://www.nj.gov/drbc/library/documents/RSMPaug-2013final-report.pdf

Kassakian, Jennifer, Ann Jones, Jeremy Martinich, and Daniel Hudgens. 2017. Managing for No Net Loss of Ecological Services: An Approach for Quantifying Loss of Coastal Wetlands due to Sea Level Rise. Environmental Management. Springer US: 0–1. https://doi.org/10.1007/s00267-016-0813-0.

Kearney, Michael S., and J. C. Alexis Riter. 2011. Inter-annual variability in Delaware Bay brackish marsh vegetation, USA. Wetlands Ecology and Management 19: 373–388. https://doi.org/10.1007/s11273-011-9222-6.

Kearney, Michael S., Andrew S. Rogers, John R. G. Townshend, Eric Rizzo, David Stutzer, J. Court Stevenson, and Karen Sundborg. 2002. Landsat imagery shows decline of coastal marshes in Chesapeake and Delaware Bays. Eos, Transactions American Geophysical Union 83: 173. https://doi.org/10.1029/2002EO000112.

Kirwan, Matthew L., Glenn R. Guntenspergen, and James T. Morris. 2009. Latitudinal trends in Spartina alterniflora productivity and the response of coastal marshes to global change. Global Change Biology 15: 1982–1989. https://doi.org/10.1111/j.1365-2486.2008.01834.x.

Kopp, R. E., C Andrews, A Broccoli, A Garner, D Kreeger, R Leichenko, N Lin, et al. 2019. New Jersey's Rising Seas and Changing Coastal Storms: Report of the 2019 Science and Technical Advisory Panel November 2019.

Kreeger, D., R. Tudor, J. Sharp, S. Kilham, D. Soeder, M. Maxwell-Doyle, J. Kraeuter, D. Frizzera, J. Hameedi and C. Collier. 2006. White paper on the status and needs of science in the Delaware Estuary. Partnership for the Delaware Estuary, Report #06-01. 72 pp.

Osland, M.J., Chivoiu, B., Enwright, N.M., Thorne, K.M., Guntenspergen, G.R., Grace, J.B., Dale, L.L., Brooks, W., Herold, N., Day, J.W. and Sklar, F.H., 2022. Migration and transformation of coastal wetlands in response to rising seas. Science Advances, 8(26), p.eabo5174.

Sacatelli, R, R. G. Lathrop, and M Kaplan. 2020. Impacts of climate change on coastal forests in the northeast US. Rutgers University. https://doi.org/10.1007/978-3-030-32878-8_7.

Smith, Joseph A.M., Steven F. Hafner, and Lawrence J. Niles. 2017. The impact of past management practices on tidal marsh resilience to sea level rise in the Delaware Estuary. Ocean and Coastal Management 149. Elsevier Ltd: 33–41. https://doi.org/10.1016/j.ocecoaman.2017.09.010.

Smith, Joseph A M. 2013. The Role of Phragmites australis in Mediating Inland Salt Marsh Migration in a Mid-Atlantic Estuary. PLoS ONE 8. https://doi.org/10.1371/journal.pone.0065091.

Stamp, Jenifer, Anna Hamilton, Marissa Liang, Jonathan Clough, Marco Propato, LeeAnn Haaf, and Jordan West. 2019. Application of the sea-level affecting marshes model (SLAMM) to the lower Delaware Bay, with a focus on salt marsh habitat. Washington, DC.

Stedman, Susan Marie, and Thomas E Dahl. 2008. Status and Trends of Wetlands In the Coastal Watersheds of the Eastern United States 1998 to 2004: 36.

Sweet, W.V., B.D. Hamlington, R.E. Kopp, C.P. Weaver, P.L. Barnard, D. Bekaert, W. Brooks, et al. 2022. Global and Regional Sea Level Rise Scenarios for the United States. Silver Spring, MD.

Weis, Judith S., Elizabeth Burke Watson, Beth Ravit, Charles Harman, and Metthea Yepsen. 2021. The status and future of tidal marshes in new jersey faced with sea level rise. Anthropocene Coasts 4: 168–192. https://doi.org/10.1139/anc-2020-0020.

Wetlands Feature 1

Mid Atlantic Coastal Wetland Assessment

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1. Partnership for the Delaware Estuary; 2. Barnegat Bay Partnership

The Partnership for the Delaware Estuary (PDE) and the Barnegat Bay Partnership (BBP) established the Mid Atlantic Coastal Wetlands Assessment (MACWA) in 2008 to address critical knowledge gaps regarding the current conditions of coastal wetlands and future risks to these wetlands posed by climate change and sea level rise. Shortly thereafter, the Academy of Natural Sciences at Drexel University, New Jersey Department of Environmental Protection (NJDEP), and Delaware Department of Natural Resources and Environmental Control (DNREC) joined the effort. Before MACWA, no single entity had been able to consistently assess or track the extent and condition of coastal wetlands across the Delaware Estuary and Barnegat Bay. Consequently, only patchy or inconsistent data existed on coastal wetland status and trends, despite the importance of these data to decision makers. From the onset, MACWA objectives sought to develop a comprehensive coastal wetlands assessment program to establish a baseline of current conditions, perform ongoing monitoring, and conduct special studies to answer questions about the drivers of coastal wetland health and function. MACWA has continued to evolve with the help of its numerous regional partners.

For the first thirteen years of its implementation, MACWA was composed of 4 tiers, which followed the three basic tiers recommended by EPA, plus an additional tier aimed to facilitate research to answer specific questions about coastal wetland condition or function. In 2020, MACWA partners highlighted the persistent yet ever-expanding need to address education, diversity, equity, inclusion, and justice in the realm of coastal resilience. Components of MACWA already sought to engage different audiences, such as using social media to help the public appreciate the beauty of coastal wetlands, or presenting data at regional or national scientific conferences, but a need to be more intentional about reaching even broader audiences and helping rectify environmental inequities and injustices was identified. To codify this, a fifth tier of MACWA, titled Social Science and Science Communication, was proposed. The five MACWA tiers as established are:

- Tier 1 – Landscape level assessments
- Tier 2 – Rapid Assessments through the Mid Atlantic Tidal Rapid Assessment (MidTRAM)
- Tier 3 – Special research studies
- Tier 4 – Intensive monitoring through Site Specific Intensive Monitoring (SSIM)
- Tier 5 – Social Science and Science Communication

MACWA initiatives, or the sub-programs implemented to support the framework, are selected based on how methodological objectives aligned with each tier. Tier 1 assessments or those focused on determine coastal wetland extent. We have completed these studies relative to National Estuary Programs requirements to produce State of Estuary reports, like the TREB. MidTRAM, which DNREC originally developed (~2007) and has since been adopted into the MACWA framework, represents Tier 2. Tier 3 includes special studies that seek to understand specific components of coastal wetland condition and function. These studies have covered topics such as restoration project monitoring, blue carbon assessments, living shoreline monitoring, as well as sediment or dendrochronological studies. SSIM, which we first piloted in 2010, represents Tier 4. We routinely collect SSIM data at designated stations and fulfill the need for understanding long-term dynamics in coastal wetland systems.

Major findings

Tier 1 Chapter 6.2.1 in this TREB report

Tier 2 PDE, BBP, and DNREC have implemented Tier 2 rapid assessments in numerous watersheds throughout the Mid-Atlantic region (Figure 6.3.9). These assessments have provided a better understanding of wetland conditions and stress distribution on a watershed-wide scale. Land managers and restoration practitioners have used MidTRAM data to better understand stressor-response gradients, determine natural differences among different areas or marsh types, and recently to help prioritize areas for restoration potential. MidTRAM data from several watersheds in coastal New Jersey, Pennsylvania and Delaware indicate that most studied tidal wetlands had some degree of stressed condition. Most of the study areas also exhibited areas that were both severely and moderately stressed (Figure 4). The Christina River watershed had the lowest average score due to numerous instances of altered hydrology in the downstream section of the watershed. On the other hand, the Mullica Estuary is one of the least disturbed study areas and received higher final MidTRAM scores with no sites having a severely stressed final score. Data collected through MidTRAM is being included in the Penn State Riparia Database to disseminate wetland condition information to practitioners, scientists and land managers.

Tier 3 Special studies conducted through tier 3 projects have sought to address research questions related to wetland condition, function, ecosystem services, and/or restoration (Table 1). Protocols and data furnished from Tiers 1, 2 and 4 help to inform experimental design and shape the findings and conclusions of these studies. PDE and BBP are currently integrating these data into various tools to help

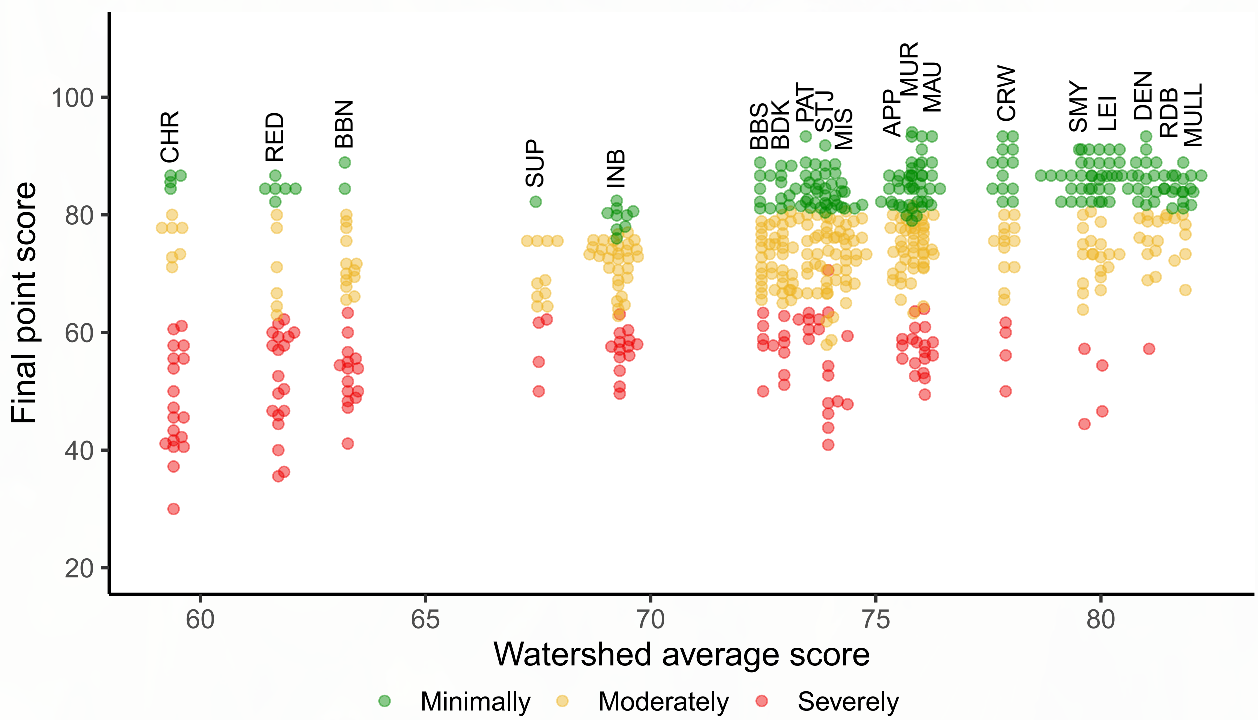

Figure 6.3.9 Final MidTRAM scores throughout New Jersey, Pennsylvania, and Delaware indicate stressed wetlands (yellow or red points) existed at a majority of studied watersheds. Studied watersheds are: Christina River (CHR), Red Lion Creek (RED), Barnegat Bay North (BBN), Supawna NWR (SUP), Inland Bays), Barnegat Bay South (BBS), Broadkill River (BDL), Pennsylvania Tidal (PAT), St Jones River (STJ), Mispillion River (MISP), Appoquinimink Creek (APP), Murderkill (MUR), Maurice River (MAU), Crosswicks Creek (CRW), SMyrna River (SMY), Leipsic River (LEI), Dennis Creek (DEN), Reeds Beach (DB), and the Mullica River (MULL).

prioritize areas for potential restoration. Tier 3 studies can also be conducted at the local level such as current groundwater studies in targeted priority coastal forests that are investigating the effects of saltwater intrusion and groundwater dynamics in coastal forest transition zones. For more, see [PDE's MACWA website.](#)

Tier 4 Long term monitoring in Barnegat Bay and the Delaware Estuary has produced extensive long-term monitoring datasets, including those on water quality, soil quality, elevation, and vegetation community metrics. These datasets provide essential baseline information about how coastal wetland conditions can vary through time and in response to changing environmental conditions. Whether tidal marshes are keeping pace with rising sea level is one of the chief study questions for Tier 4 monitoring. From Tier 4 SSIM monitoring, we found that the rate of elevation change at most of these study sites does not keep pace with recent rates of sea level rise (Figure 6.3.10). This is especially true for the study sites in Delaware Bay where none of the salt marsh monitoring sites are keeping pace with the observed increase in sea level of 5.5 cm from 2011-2021. In Barnegat Bay, the West Creek/Dinner Point (WC) site saw a large increase in elevation in 2013 due to sediment side casting from Open Water Marsh Management (OMWM), a mosquito management technique.

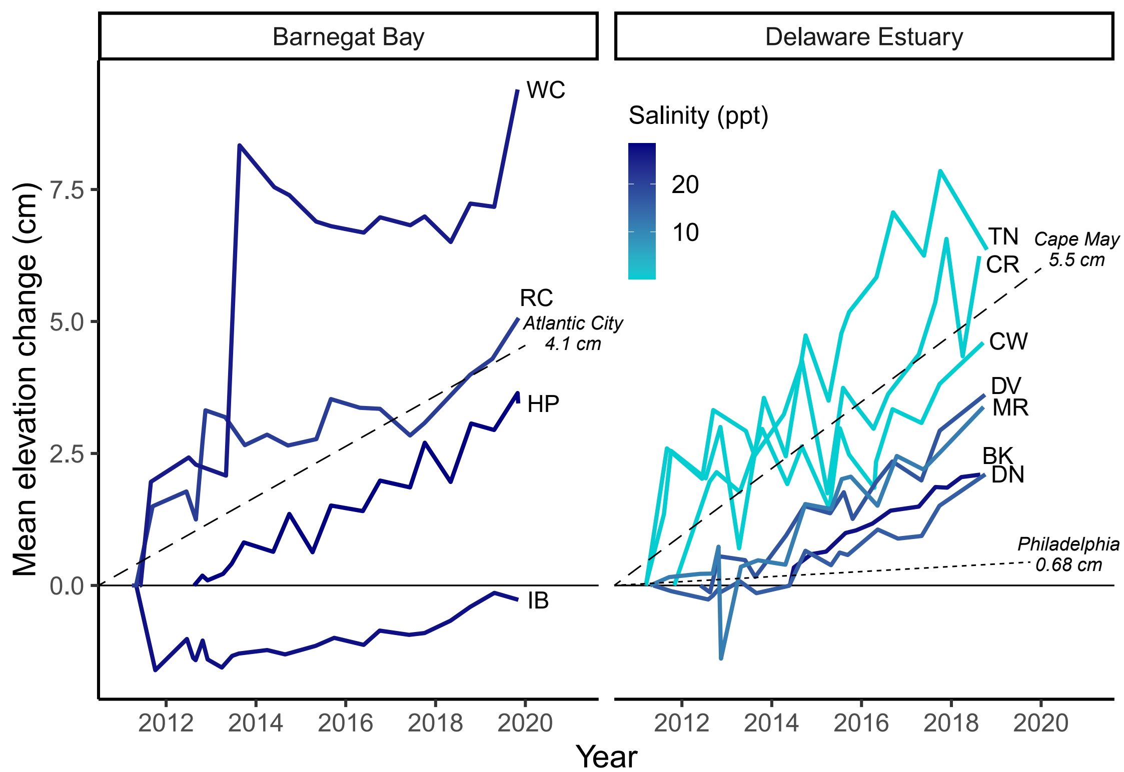

Figure 6.3.10 Mean elevation change at long term monitoring sites in Barnegat Bay and the Delaware Estuary from 2011-2019. The dashed lines represent mean sea level (MSL) where the slope of the line is the change in sea level from the average monthly sea level in 2011 (starting point of 0) to the average monthly MSL in 2019 (give as the difference in cm). In the Delaware Estuary, freshwater tidal sites (turquoise) should be compared to the change in MSL at Philadelphia (a change of 0.68 cm), whereas the saline sites (dark blue) should be compared to MSL at Cape May (a change of 5.5 cm). Sites are West Creek (WC), Reedy Creek (RC), Horse Point (HP), Island Beach (IB) in Barnegat Bay, and Tinicum (TN), Christina River (CH), Crosswicks Creek (CW), Dividing Creek (DV), Maurice River (MR), Broadkill River (BK) and Dennis Creek (DN) in the Delaware Estuary.

Wetlands Feature 2

Coastal Forest Dieback in the Delaware Bay

Rachael Sacatelli, Richard G. Lathrop, Marjorie Kaplan
Rutgers, The State University of New Jersey

The coastal forests around the Delaware Bay are commonly a mix of deciduous hardwoods and evergreen conifers with the species composition dependent upon the site level soil moisture gradient (Anderson et al., 2013a; Butler-Leopold et al., 2018; Janowiak et al., 2018). In addition to providing protective buffering of inland areas against coastal storms (Williams et al., 2003; Barbier et al., 2011; Duarte et al., 2013), these forests provide a range of ecosystem services including carbon storage (McGarvey et al., 2015, Fahey et al., 2010), valuable timber resources, and habitat for a diversity of rare plants and animals including a number of species of concern (Global Rank G1-G4) (Anderson et al., 2013b).

Various studies have documented that these coastal forests are showing signs of stress evidenced by trees at the forest-tidal salt marsh edge dying back and the forests transitioning into tidal salt marsh ecosystems (Smith, 2013; Kirwan and Gedan, 2019; Sacatelli, 2020). These areas have been referred to as "ghost forests" denoting the presence of standing dead trees within or fringing the edge of salt marsh ecosystems (Able et al., 2018; Kirwan and Gedan, 2019; Ury et al., 2021). While this phenomenon of coastal forest dieback and replacement with salt marshes as sea level rises has been ongoing for millennia (Clark, 1986; Able 2021), there is widespread concern that accelerating local sea-level rise (Kopp et al. 2019; Gornitz et al., 2019; Boesch et al. 2018; Sweet et al., 2017b; Oppenheimer et al., 2019) and intensifying coastal storms and associated surges (Gornitz et al., 2019) may be hastening this process (Schieder and Kirwan, 2019).

The dieback of the forest ecosystem at the salt water tidal marsh edge (or marsh/forest ecotone) and transition to tidal salt marsh ecosystems is often referred to in the literature as the migration (or transgression) of the salt water tidal marsh into the coastal forest (Wasson et al. 2013, Kirwan and Gedan, 2019, Hussein, 2009; Smith, 2013; Schieder et al., 2018; Sacatelli, 2020). This coastal dieback and subsequent marsh migration for selected areas along the Delaware Bay has recently been mapped using historical aerial photography between the years 1940 and 2015 (Figure 6.3.11) (Sacatelli, 2020). The study found an average rate of coastal forest dieback was 3.2 m/yr, ranging from 0.4m/yr to 8.3m/yr depending on the location (Sacatelli, 2020). When examined in temporal increments, these rates also show evidence of accelerating in recent years (Sacatelli, 2020).

Subsequently, a Mid-Atlantic region-wide assessment has been undertaken to map those coastal forest areas vulnerable to future sea level rise. The consensus SLR estimates determined for New Jersey (https://www.nj.gov/dep/climatechange/pdf/2019-stap-report-summary.pdf) suggest that 1 foot of SLR could be reached by 2050. This 1' rise in sea level is estimated to put over 250,000 acres of coastal forest at risk of dieback 1' (Lathrop et al., in prep). However, when compared with historical change analysis (described above), the results suggest that low-lying forested areas predicted to be at risk under future sea level rise are already experiencing stress and dieback at current SLR levels (Lathrop et al., in prep). We can expect that this phenomenon of ghost forests along Delaware Bay to be an ever present feature into the foreseeable future.

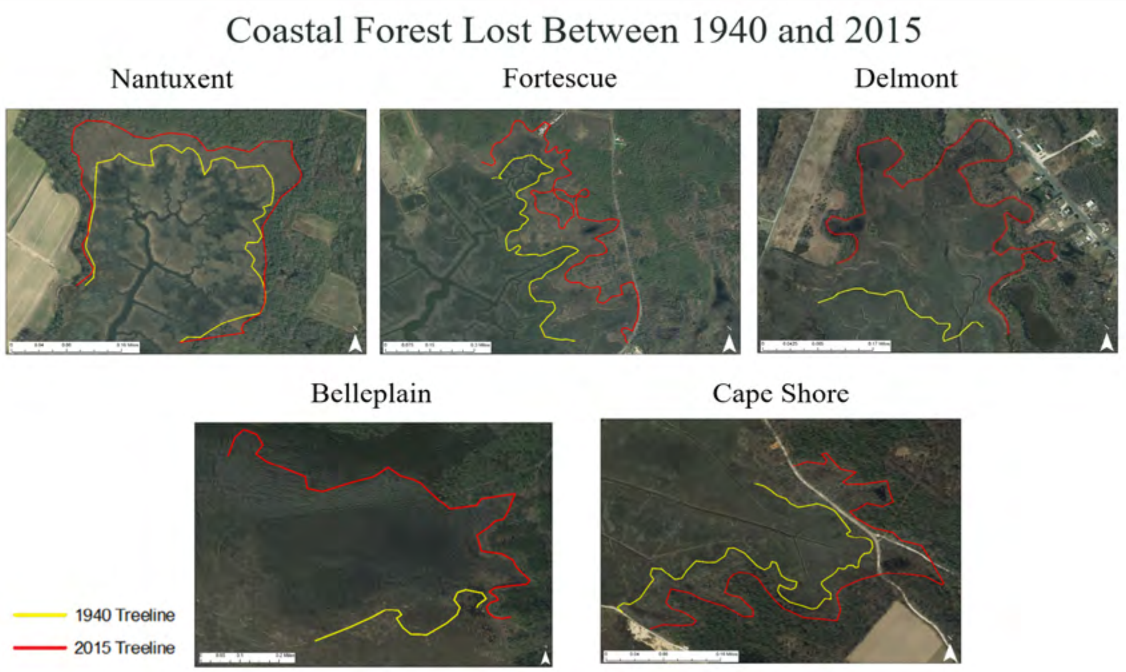

Figure 6.3.11 Locations of the treeline as of 1940 and 2015 at 5 sites along the New Jersey coast of the Delaware Bay. The area between the two lines is the area of forest lost between 1940 and 2015.

References

Able, K.W., J. Walker and B.P. Horton. (2018). Ghost forests in the Mullica Valley: Indicators of sea-level rise. Sojourn 2(2):87-96.

Able, K. W. (2021). From Cedar Cemeteries to Marsh Lakes: a Case Study of Sea-Level Rise and Habitat Change in a Northeastern US Salt Marsh. Estuaries and Coasts. 44. 10.1007/s12237-021-00946-x.

Anderson, C.J.; Lockaby, B.G.; Click, N. (2013a). Changes in wetland forest structure, basal growth, and composition across a tidal gradient. The American Midland Naturalist. 170(1): 1-13. https://doi.org/10.1674/0003-0031-170.1.1.

Anderson, M.G. M. Clark, C.E. Ferree, A. Jospe, A. Olivero Sheldon and K.J. Weaver. (2013b). Northeast Habitat Guides: A companion to the terrestrial and aquatic habitat maps. The Nature Conservancy, Eastern Conservation Science, Eastern Regional Office. Boston, MA. http://nature.ly/HabitatGuide.

Barbier, E. B., Hacker, S. D., Kennedy, C., Koch, E. W., Stier, A. C., & Silliman, B. R. (2011). The value of estuarine and coastal ecosystem services. Ecological monographs, 81(2), 169-193.

Boesch, D. F., Boicourt, W. C., Cullather, R. I., Ezer, T., Galloway Jr, G. E., Johnson, Z. P., ... & Li, M. (2018). Sea-level rise: projections for Maryland 2018. 27 pp. University of Maryland Center for Environmental Science, Cambridge, MD.

Butler-Leopold, P. R., Iverson, L. R., Thompson, F. R., Brandt, L. A., Handler, S. D., Janowiak, M. K., ... & Clark, K. L. (2018). Mid-Atlantic forest ecosystem vulnerability assessment and synthesis: a report from the Mid-Atlantic Climate Change Response Framework project. Gen. Tech. Rep. NRS-181. Newtown Square, PA: US Department of Agriculture, Forest Service, Northern Research Station. 294 p., 181, 1-294.

Clark, J. S. (1986). Coastal forest tree populations in a changing environment, southeastern Long Island, New York. Ecol. Monogr. 56, 259–277. doi: 10.2307/2937077

Duarte, C. M., Losada, I. J., Hendriks, I. E., Mazarrasa, I., & Marbà, N. (2013). The role of coastal plant communities for climate change mitigation and adaptation. Nature Climate Change, 3(11), 961-968.

Fahey, T. J., Woodbury, P. B., Battles, J. J., Goodale, C. L., Hamburg, S. P., Ollinger, S. V., & Woodall, C. W. (2010). Forest carbon storage: ecology, management, and policy. Frontiers in Ecology and the Environment, 8(5), 245-252.

Gornitz, V. M., Oppenheimer, M., Kopp, R., Orton, P., Buchanan, M., Lin, N., Horton, R. & Bader, D. A. (2019). New York City Panel on Climate Change 2019 Report Chapter 3: Sea Level Rise. Annals of the New York Academy of Sciences. https://doi.org/10.1111/nyas.14006.

Hussein, A. H. (2009). Modeling of sea-level rise and deforestation in submerging coastal ultisols of Chesapeake Bay. Soil Science Society of America Journal, 73(1), 185-196.

Janowiak, M. K., D'Amato, A. W., Swanston, C. W., Iverson, L., Thompson, F. R., Dijak, W. D., ... & Brandt, L. A. (2018). New England and northern New York forest ecosystem vulnerability assessment and synthesis: a report from the New England Climate Change Response Framework project. Gen. Tech. Rep. NRS-173. Newtown Square, PA: US Department of Agriculture, Forest Service, Northern Research Station. 234 p., 173, 1-234.

Kirwan, M. L., & Gedan, K. B. (2019). Sea-level driven land conversion and the formation of ghost forests. Nature Climate Change, 9(6), 450.

Kopp, R.E., C. Andrews, A. Broccoli, A. Garner, D. Kreeger, R. Leichenko, N. Lin, C. Little, J.A. Miller, J.K. Miller, K.G. Miller, R. Moss, P. Orton, A. Parris, D. Robinson, W. Sweet, J. Walker, C.P. Weaver, K. White, M. Campo, M. Kaplan, J. Herb,and L. Auermuller. (2019). New Jersey's Rising Seas and Changing Coastal Storms: Report of the 2019 Science and Technical Advisory Panel.Rutgers, The State University of New Jersey. Prepared for the New Jersey Department of Environmental Protection. Trenton, New Jersey.

Lathrop, R., Sacatelli, R., Kaplan, M., Allen, B., & Carleton, G. (In Prep). Coastal Forest Dieback in the Northeast US.

McGarvey, J. C., Thompson, J. R., Epstein, H. E., & Shugart Jr, H. H. (2015). Carbon storage in old-growth forests of the Mid-Atlantic: toward better understanding the eastern forest carbon sink. Ecology, 96(2), 311-317.

Oppenheimer, M., B.C. Glavovic, J. Hinkel, R. van de Wal, A.K. Magnan, A. Abd-Elgawad, R. Cai, M. Cifuentes-Jara, R.M. DeConto, T. Ghosh, J. Hay, F. Isla, B. Marzeion, B. Meyssignac, and Z. Sebesvari. (2019). Sea Level Rise and Implications for Low-Lying Islands, Coasts and Communities. In: IPCC Special Report on the Ocean and Cryosphere in a Changing Climate [H.-O. Pörtner, D.C. Roberts, V. Masson-Delmotte, P. Zhai, M. Tignor, E. Poloczanska, K. Mintenbeck, A. Alegría, M. Nicolai, A. Okem, J. Petzold, B. Rama, N.M. Weyer (eds.)]. In press.https://www.ipcc.ch/srocc/chapter/chapter-4-sea-level-rise-and-implications-for-low-lying-islands-coasts-and-communities/

Sacatelli, R. M. (2020). A look at upland salt marsh edge migration in New Jersey (Masters Thesis, Rutgers University-School of Graduate Studies). https://rucore.libraries.rutgers.edu/rutgers-lib/62908/

Schieder, N. W., Walters, D. C., & Kirwan, M. L. (2018). Massive upland to wetland conversion compensated for historical marsh loss in Chesapeake Bay, USA. Estuaries and coasts, 41(4), 940-951.

Schieder, N. W., and M. L. Kirwan. (2019). Sea-level driven acceleration in coastal forest retreat. Geology 47: 1151– 1155.

Smith, J.A. (2013) The role of Phragmites australis in mediating inland salt marsh migration in a mid-Atlantic estuary. PloS one 8 (5), e65091

Sweet, W. V., Kopp, R. E., Weaver, C. P., Obeysekera, J., Horton, R. M., Thieler, E. R., & Zervas, C. (2017). Global and regional sea level rise scenarios for the United States. NOAA Technical Report NOS Co-OPS 083.

Ury, E.A., Yang, X., Wright, J.P. & Bernhardt, E.S. (2021). Rapid deforestation of a coastal landscape driven by sea-level rise and extreme events, Ecological Applications, 10.1002/eap.2339, 31(5).

Wasson, K., Woolfolk, A. & Fresquez, C. (2013). Ecotones as indicators of changing environmental conditions: rapid migration of salt marsh-upland boundaries. Estuaries and Coasts 36: 654– 664.

Williams, K., MacDonald, M., & Sternberg, L. D. S. L. (2003). Interactions of storm, drought, and sea-level rise on coastal forest: a case study. Journal of Coastal Research, 1116-1121.

Wetlands Feature 3

A Closer Look at Blue Carbon in the Delaware Bay

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What is Blue Carbon?

Vegetated coastal ecosystems, including tidal marshes, seagrass meadows, and mangroves, are being increasingly recognized for their potential to store and sequester "blue carbon" in both sediments and biomass. When sediments accrete in a saturated tidal environment, the anaerobic conditions allow decomposing organic material to build up over the course of hundreds or thousands of years, thus supporting long-term carbon storage. This process can be over ten times more efficient than even the most productive forested ecosystems; despite their comparatively small acreage, salt marshes have an estimated carbon burial rate of approximately 4.8 million metric tons of carbon per year globally (McLeod et al. 2011).

Carbon accumulation in tidal marshes can be influenced by a variety of factors, including vegetation type, primary productivity, sediment and nutrient fluxes, tidal range, accretion rates, and salinity. These complex geomorphological and biogeochemical interactions tend to be highly variable and site-specific, and as a result they are not well understood (Holmquist et al. 2018; Sheng et al. 2015; Reid et al. 2013; Poffenbarger et al. 2011; Kirwan et al. 2016). Human impacts and climate change factors can also impact a tidal marsh's ability to store and sequester carbon; increases in development pressure, sea level rise, and severe coastal storms can impact marsh health, alter accretion regimes, and exacerbate edge erosion, thus influencing tidal marsh carbon fluxes (Wigand et al. 2014; Alldred et al. 2017; Wedge and Anderson 2017; Logan 2018; Matzke and Elsey-Quirk 2018; Martin et al. 2018; Morris et al. 2002, Kirwan et al. 2016, Rogers et al. 2019; IPCC 2019; Lane et al. 2016). More research is needed in order to further the use of blue carbon sequestration and storage for climate mitigation efforts.

Status and trends of blue carbon storage and sequestration in the Delaware Estuary

Based on organic matter density data collected at various depths (0-134 cm) and locations, tidal marshes in the Delaware Estuary currently store an average of 0.039 g C cm^{-3} ± 0.013 (CCRCN 2022; organic carbon estimation methodology from Craft et al. 1991). This average is slightly higher than the national average found by Holmquist et al. 2018 in an inventory of tidal marsh carbon across the country, but it has similar variability (0.027 g C cm^{-3} ± 0.013). Annual carbon accumulation rates in the Delaware Bay may also be higher than those in nearby tidal marsh environments such as Barnegat Bay, especially in the upper bay (Figure 6.3.12; Champlin et al. 2020). This difference may be due to factors such as higher primary productivity, greater suspended sediment availability, and larger tidal range. Despite greater carbon sequestration rates, the lower salinity of the Delaware Estuary means greater methane emissions as well. While polyhaline marshes (salinity >18 ppt) emit minimal methane (1 ± 2 g m^{-2} yr^{1}), marsh types with lower salinities (0-18 ppt) can be highly variable, emitting anywhere between 16 ± 11 g m^{-2} yr^{1} and 150 ± 221 g m^{-2} yr^{1}; these marshes have the potential to be net sources rather than sinks (Poffenbarger et al. 2011; Settelmyer 2018; Mitsch et al. 2013).

Delaware Estuary marshes have higher annual accretion rates than Barnegat Bay marshes, potentially due to increased sediment inputs facilitated by higher tidal ranges and lower elevation (Haaf et al. 2022).

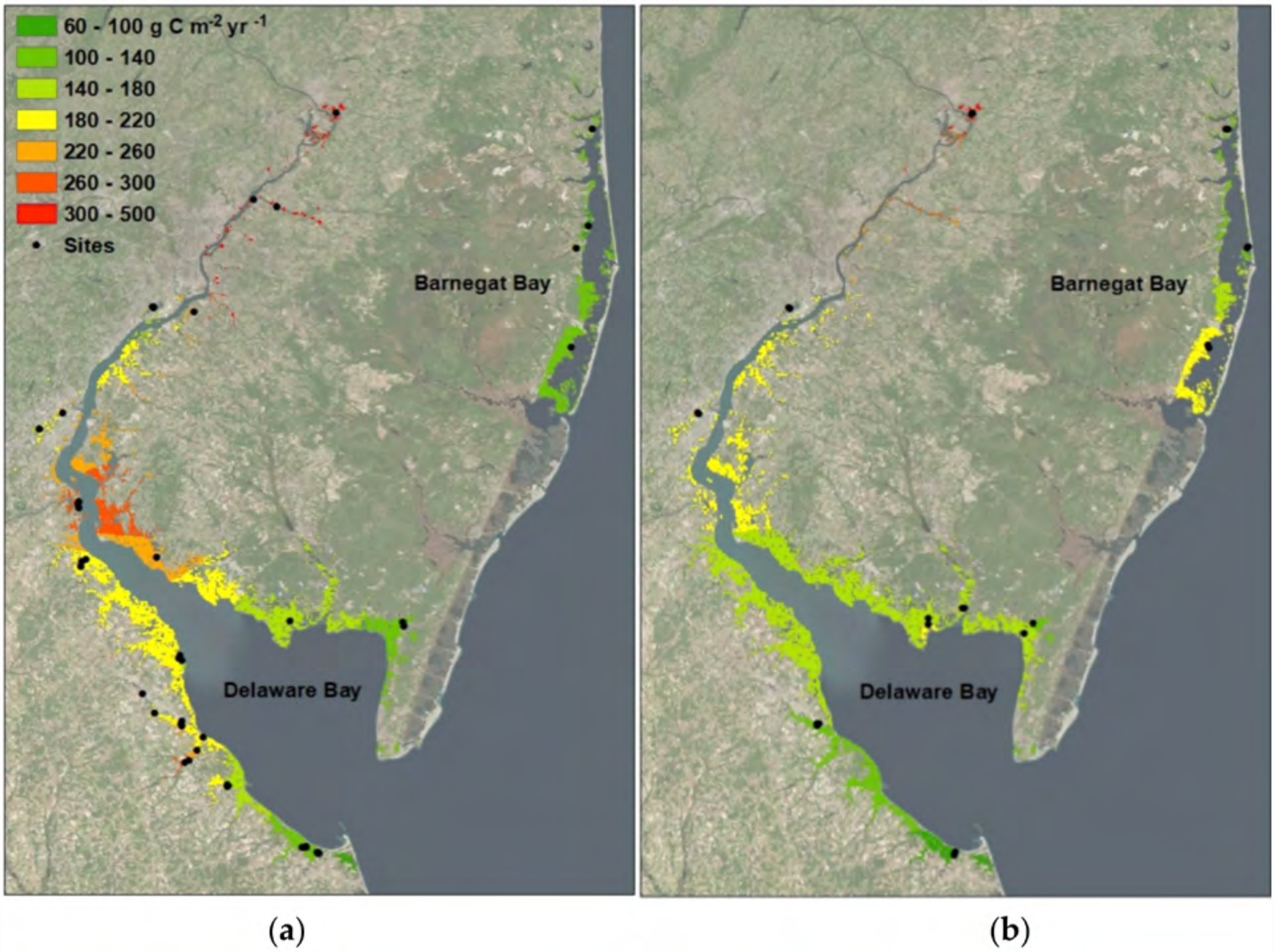


Figure 6.3.12 Interpolation maps of yearly carbon accumulation in Delaware Bay and Barnegat Bay, based on accumulation rates estimated by using (a) 137Cs dating and (b) from marker bed measures. Reprinted from "Carbon Sequestration rate estimates in Delaware Bay and Barnegat Bay tidal wetlands using interpolation mapping" (Champlin et al., 2020).

This difference in accretion may continue over the next 30 years, thus allowing Delaware Bay marshes to keep pace with rising sea levels after other nearby marshes have drowned (Warnell and Olander 2020); however, sea level rise has been accelerating in recent decades, and shallow subsidence has been lowering Delaware Estuary marsh elevation (Haaf et al. 2022). Projections under an intermediate to high emissions scenario suggest that unless accretion rates increase, current Delaware Bay marshes are likely to be submerged by 2100 (Haaf et al. 2022). Assuming an intermediate global mean sea level rise scenario (1 m by 2100, Sweet et al. 2017), these marshes are also likely to cumulatively emit more carbon than is sequestered throughout the entire 80-year period, leaving only the newly migrated marshes as net carbon sinks (Figure 6.3.13; Warnell and Olander 2020).

A variety of protection, restoration, and enhancement techniques can provide greater opportunity for blue carbon accumulation in the Delaware Estuary. While additional research is necessary to ascertain the most effective courses of action, reducing edge erosion with living shorelines, raising marsh elevation with dredge sediment placement, increasing salinity and accretion rates through hydrologic restoration, and protecting land for marsh migration are all possible approaches that could help us take full advantage of our blue carbon resources in the fight against climate change.

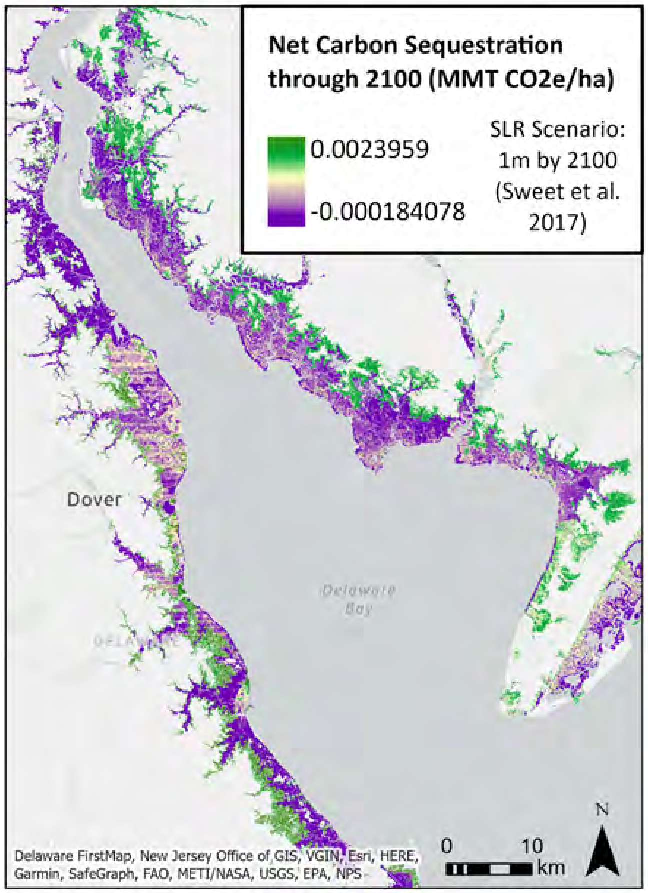


Figure 6.3.13 Figure 2. Net carbon sequestration through 2100, assuming the Sweet et al. 2017 intermediate sea level rise scenario (1m by 2100). Data from "Coastal protection and blue carbon mapping for six Mid-Atlantic states"(Warnell & Olander, 2020).

Sources Cited

Alldred, M., A. Liberti, and S.B. Baines. 2017. Impact of salinity and nutrients on salt marsh stability. Ecosphere 8(11): e02010. 10.1002/ecs2.2010

Boyd, B., C.K. Sommerfield, T. Quirk, V. Unger. 2019. Dataset: Accretion and sediment accumulation in impounded and unimpounded marshes in the Delaware Estuary and Barnegat Bay. Smithsonian Environmental Research Center. Dataset. https://doi.org/10.25573/data.9747065.v1

Boyd, B. and Sommerfield C.K. 2016. Marsh accretion and sediment accumulation in a managed tidal wetland complex of Delaware Bay. Ecological Engineering 92:37-46. https://doi.org/10.1016/j.ecoleng.2016.03.045

Boyd, B. 2012. Comparison of sediment accumulation and accretion in impounded and unimpounded marshes of the Delaware Estuary. Master's Thesis, University of Delaware, Department of Marine Studies. http://udspace.udel.edu/handle/19716/12831

CEC. 2021. North American Updates in Blue Carbon Science, Conservation and Collaboration. Montreal, Canada: Commission for Environmental Cooperation. 18 pp.

Champlin, L., D. Velinsky, K. Tucker, C. Sommerfield, K. St. Laurent, and E. Watson. 2020. Carbon Sequestration rate estimates in Delaware Bay and Barnegat Bay tidal wetlands using interpolation mapping. Data 5(11). doi:10.3390/data5010011

Coastal Carbon Research Coordination Network. 2022. Coastal Carbon Atlas. Accessed 4/29/2022. https://ccrcn.shinyapps.io/CoastalCarbonAtlas/

Craft, C.B., E.D. Seneca, and S.W. Broome. 1991. Loss on ignition and Kjeldahl digestion for estimating organic carbon and total nitrogen in estuarine marsh soils: Calibration with dry combustion. Estuaries 14(2):175-179

Haaf, L., E.B. Watson, T. Elsey-Quirk, K. Raper, A. Padeletti, M. Maxwell-Doyle, D. Kreeger, and D. Velinsky. 2022. Sediment Accumulation, Elevation Change, and the Vulnerability of Tidal Marshes in the Delaware Estuary and Barnegat Bay to

Accelerated Sea Level Rise. Estuaries and Coasts 45:413-427. doi:10.1007/s12237-021-00972-9

Holmquist, J.R. et al. 2018. Accuracy and Precision of Tidal Wetland Soil Carbon Mapping in the Conterminous United States. Scientific Reports 8:9478 DOI:10.1038/s41598-018-26948-7

Holmquist, James R., Windham-Myers, Lisamarie, Bliss, Norman, Crooks, Stephen, Morris, James T., Megonigal, J. Patrick, Troxler, Tiffany, Weller, Donald E., Callaway, John, Drexler, Judith, Ferner, Matthew C., Gonneea, Meagan E., Kroeger, Kevin D., Schile-Beers, Lisa, Woo, Isa, Buffington, Kevin, Boyd, Brandon M., Breithaupt, Joshua, Brown, Lauren N., Dix, Nicole, Hice, Lyndie, Horton, Benjamin P., MacDonald, Glen M., Moyer, Ryan P., Reay, William et al. 2018. Dataset: Accuracy and Precision of Tidal Wetland Soil Carbon Mapping in the Conterminous United States: Public Soil Carbon Data Release. Distributed by Smithsonian Research Online.

IPCC. 2019. Summary for policymakers. In: Pörtner H-O, Roberts DC, Masson-Delmotte VP, et al., eds. IPCC Special Report on the Ocean and Cryosphere in a Changing Climate. Geneva, Switzerland: IPCC (in press).

Kirwan, M.L., D.C. Walters, W.G. Reay, and J.A. Carr. 2016. Sea level driven marsh expansion in a coupled model of marsh erosion and migration. Geophysical Research Letters 43:4366-4373 doi:10.1002/ 2016GL068507

Lane, R.R., S.K. Mack, J.W. Day, R.D. DeLaune, M.J. Madison, and P.R. Precht. 2016. Fate of Soil Organic Carbon During Wetland Loss. Wetlands 36(6):1167–1181.

Laurent, Kari A. St., D.J. Hribar, A.J. Carlson, C.M. Crawford, and D. Siok (2020): Dataset: Assessing coastal carbon variability in two Delaware tidal marshes. Smithsonian Environmental Research Center. Dataset. https://doi.org/10.25573/serc.13315472.v1

Logan, J.M. 2018. Salt marsh aboveground production in New England estuaries in relation to nitrogen loading and environmental factors. Wetlands 38(6):1327-1340

Martin, R.M., C. Wigand, E. Elmstrom, J. Lloret, and I. Valiela. 2018. Long-term nutrient addition increases respiration and nitrous oxide emissions in a New England salt marsh. Ecology and Evolution 8(10):4958-4966

Matzke, S. and Elsey-Quirk, T. 2018. Spartina patens productivity and soil organic matter response to sedimentation and nutrient enrichment. Wetlands 38(6):1233-1244

Mcleod, E., G.L. Chmura, S. Bouillon, R. Salm, M. Bjork, C.M. Duarte, C.E. Lovelock, W.H. Schlesinger, and B.R. Silliman. 2011. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. Frontiers in the Ecology of the Environment 9(10):552-560

Mitsch, W.J., B. Bernal, A.M. Nahlik, U. Mander, L. Zhang, C.J. Anderson, S.E. Jorgensen, and H. Brix. Wetlands, carbon, and climate change. Landscape Ecology 28:583-597. https://doi.org/10.1007/s10980-012-9758-8

Morris, J.T., P.V. Sundareshwar, C.T. Nietch, B. Kjerfve, and D.R. Cahoon. 2002. Responses of coastal wetlands to rising sea level. Ecology 83(10):2869-2877

Nelson, J.L. and Zavaleta, E.S. 2012. Salt Marsh as a Coastal Filter for the Oceans: Changes in Function with Experimental Increases in Nitrogen Loading and Sea-Level Rise. Plos One DOI: 10.1371/journal.pone.0038558

Orson, R.A. and Sim, R.L. 1990. Rates of sediment accumulation in a tidal freshwater marsh. SEPM Journal of Sedimentary Research 60. http://dx.doi.org/10.1306/D4267631-2B26-11D7-8648000102C1865D

Poffenbarger, H.J., B.A. Needelman, and J.P. Megonigal. 2011. Salinity Influence on Methane Emissions from Tidal Marshes. Wetlands 31:831-842

Reid, M.C., R. Tripathee, V.R. Schafer, and P.R. Jaffe. 2013. Tidal marsh methane dynamics: Difference in seasonal lags in emissions driven by storage in vegetated versus unvegetated sediments. Journal of Geophysical Research: Biogeosciences 118:1802-1813

Rogers, K., J.J. Kelleway, N. Saintilan, J.P. Megonigal, J.B. Adams, J.R. Holmquist, M. Lu, L. Schile-Beers, A. Zawadzki, D. Mazumder, and C.D. Woodroffe. 2019. Wetland carbon storage controlled by millennial-scale variation in relative sea-level rise. Nature 567:91-95

Sheng, Q., L. Wang, and J. Wu. 2015. Vegetation alters the effects of salinity on greenhouse gas emissions and carbon sequestration in a newly created wetland. Ecological Engineering 84:542-550

Settelmyer, S., 2018. Delaware Bay Blue Carbon Feasibility Study: Phase 2 Report. Version 2.0. Prepared by TerraCarbon for The Nature Conservancy.

Sweet, W.V., R.E. Kopp, C.P. Weaver, J. Obeysekera, R.M. Horton, E.R. Thieler, and C. Zervas. 2017. Global and regional sea level rise scenarios for the United States. NOAA Technical Report NOS CO-OPS 083. https://tidesandcurrents.noaa.gov/publications/techrpt83_Global_and_Regional_SLR_Scenarios_for_the_US_final.pdf

Unger, V., T. Elsey-Quirk, C. Sommerfield, D. Velinsky. 2016. Stability of organic carbon accumulating in Spartina alterniflora-dominated salt marshes of the Mid-Atlantic US. Estuarine, Coastal and Shelf Science 182A: 179-189. https://doi.org/10.1016/j.ecss.2016.10.001.

Warnell, K., and Olander, L. 2020. Data from: Coastal protection and blue carbon mapping for six mid-Atlantic states. Duke Research Data Repository. https://doi.org/10.7924/r4pg1qw8p

Wedge, M., and Anderson, C.J. 2017. Urban Land use Affects Resident Fish Communities and Associated Salt Marsh Habitat in Alabama and West Florida, USA. Wetlands 37:715-727

Wigand, C., C.T. Roman, E. Davey, M. Stolt, R. Johnson, A. Hanson, E.B. Watson, S.B. Moran, D.R. Cahoon, J.C. Lynch, and P. Rafferty. 2014. Below the disappearing marshes of an urban estuary: Historic nitrogen trends and soil structure. Ecological Applications 24(4):633-649

6.3.2 Non-tidal Wetland Cover

Abstract

Non-tidal wetlands are ecologically important natural habitats. Data from the National Wetlands Inventory suggests that there are over 240,00 hectares (595,000 acres) of non-tidal wetlands in the Delaware Estuary and Basin. About half of the non-tidal wetlands in the Delaware Estuary and Basin are forested (52%), with lacustrine wetlands constituting 15%, shrub-scrub wetlands constituting 10%, and emergent wetlands constituting ~8%. Between 1996–2016, however, non-tidal wetlands of the Delaware Estuary and Basin experienced a net decline of over 1,000 hectares (2,600 acres), which is an average loss of 13 hectares (32 acres or 0.022%) per year, as NOAA C-CAP data suggested. Future projections imply that these losses will likely increase due to degradation and perhaps climate change. Research, monitoring, proactive management, and on-the-ground actions could help minimize ongoing losses.

Introduction

Non-tidal wetlands are vegetated aquatic habitats that occur along rivers, lakes, and streams, as well as within topographic depressions or on top of clay lenses. Non-tidal wetlands can exist along a gradient of flooding, from permanently to periodically or seasonally inundated (Figure 6.3.14). Non-tidal wetlands are highly productive habitats and are critical for water flow, storage, and health in the Estuary and Basin. They perform a wide variety of vital ecosystem services (de Groot et al. 2018). These services include storing water, carbon, and other nutrients; providing habitat to a wide variety of wildlife, including imperiled species of birds, mammals, insects, and plants; filtering and storing contaminants to sustain water quality; as well as supporting recreation and aesthetic value. Non-tidal wetlands are habitat to 37% of rare plant taxa in Delaware (William McAvoy, pers. comm. 2022) and 57% in New Jersey (Kathleen Walz, pers. comm. 2022). Non-tidal wetlands are therefore one of the most critical habitat types in the Delaware Estuary for supporting broad ecological health and good water quality. Assuring that non-tidal wetlands remain intact and continue to provide these critical functions is therefore fundamental to the overall good quality of the Delaware Estuary and Basin as a whole.

Figure 6.3.14 Non-tidal wetlands in the Delaware Estuary and Basin are diverse. They span a broad inundation gradient, which includes seasonally wet swamps (A) and emergent wetlands (B).

Non-tidal wetlands occur throughout the Delaware Estuary and Basin. Non-tidal wetlands occur beyond tidal reaches in the Estuary (Lower and Bayshore regions), and encompass all wetland types in the Basin (Upper and Central regions). There is a great diversity of non-tidal wetland types in the Delaware Estuary and Basin, but in this Chapter, non-tidal wetlands are categorized into a few broad categories: *aquatic beds, riverine wetlands, lacustrine wetlands, emergent wetlands, shrub-scrub wetlands, forested wetlands, and unconsolidated.* Aquatic beds are permanently flooded wetlands, riverine wetlands occur along rivers or large streams. Lacustrine wetlands occur within lakes or other enclosed bodies of water. Herbaceous plants dominate emergent wetlands. Woody plants dominate shrub-scrub wetlands less than 6 m (20 ft.) tall whereas woody plants greater than 6 m (20 ft.) tall dominate forested wetlands (also called swamps). Lastly, unconsolidated non-tidal wetlands are those areas with sparse vegetation such as mudflats, shorelines, or water body bottoms.

Description of Indicator

Non-tidal wetland habitats are one of the leading environmental indicators for the Estuary and Basin as a whole. In this chapter, we synthesized landscape-level data to assess our best current understanding of non-tidal wetland composition in the Estuary and Basin, as well as discern how acreage varies over space and time.

Data Sources

Present status

National Wetlands Inventory We gathered data on wetland distribution for each state from the U.S. Fish and Wildlife Service (USFWS) National Wetlands Inventory (NWI). The NWI is a nationwide program that inventories the nation's wetlands through aerial imagery interpretation and ground-truthing. The NWI provides detailed, consistent, high-resolution data that enables differentiation of wetland types; however, it is of limited value in trend analyses for the whole system because of the various data collection times in different states and areas. For instance, the latest NWI data in New Jersey are from approximately 2007; in Delaware in 2017; and in 2015 in Pennsylvania.

To determine the current extent of the various types of non-tidal wetlands in the Estuary, the latest of each of three state-wide NWI wetlands were used. We categorized wetland types using the classification scheme developed by Cowardin (1979), but then used a simplified classification scheme to allow for a synoptic assessment of the status of broad categories of wetlands within the Estuary and Basin. Non-tidal wetlands were classified as *aquatic bed, emergent wetland, forest, lacustrine, lacustrine aquatic bed, lacustrine emergent, riverine aquatic bed, riverine emergent, riverine streambed, shrub-scrub, and unconsolidated.* We excluded freshwater tidal wetlands from this analysis by isolating palustrine wetlands with tidal flood classifications (freshwater tidal flood classifications of *S, R, V,* and *T*).

Trends

Coastal Change Analysis Program Determining landscape-level changes in different wetland types of the Delaware Estuary requires consistent data in both space and time. Since NWI lacks temporal consistency, wetlands data are derived from the National Oceanic and Atmospheric Administration's (NOAA) Office of Coastal Management *Coastal Change Analysis Program* (C-CAP) datasets. These data were from Landsat imagery that has a 30 m ground resolution; C-CAP routinely assesses this imagery in 4–6 (typically 5) year intervals. Years for the C-CAP land cover data for the Delaware Estuary and Basin are 1996, 2001, 2006, 2010, and 2016[1].

1. States or regions were not delineated using satellites from the same epoch, as interpretation requires high-quality, cloud-free imagery, so the choice of dates to consider was limited by data availability.

C-CAP data are most useful for trend analyses, as they are not as resolved as NWI (C-CAP has a 1:100,000 scale, whereas NWI has a scale of 1:24,000), and may have larger classification errors. Previous assessment of the comparability of the wetland categories of the C-CAP land cover data with NWI indicates that the data are comparable to a relatively small percentage difference. Although accuracy/precision limitations among various mapping methods have been noted (Weis et al. 2020), C-CAP remains the most methodologically consistent dataset for temporal trend analysis of wetlands in the Delaware Estuary and Basin. Therefore, we used C-CAP data to assess the *Trends* in non-tidal wetlands for this report, whereas NWI data are used to determine *Status*.

Categories of non-tidal, freshwater wetlands (i.e., palustrine) distinguished by the C-CAP are forested, scrub-shrub, emergent, unconsolidated shore, and aquatic bed. Freshwater tidal wetlands (Chapter 6.3.1) were excluded from these analyses.

Results

Status

Non-tidal wetlands cover 240,744 hectares (594,891 acres), which is about 7% of the total land area of the Estuary and Basin (~3.3 million hectares) (Figures 6.3.15-6.3.20). About half of the non-tidal wetlands in the Delaware Estuary and Basin are forested (52%), with lacustrine wetlands covering 15%, shrub-scrub wetlands covering 10.8%, and emergent wetlands covering about 8%. Unconsolidated habitats (shorelines, mudflats) are 10.8% of non-tidal wetlands in the Estuary and Basin (25,881 hectares or 63,953 acres). In the following figures, summary information on non-tidal wetland acreage based on the latest NWI data was divided by subregion (Fig 6.3.16-6.3.20, Tables 6.3.6-6.3.9). and state (PA, NJ, and DE; Fig 6.3.21, Table 6.3.10).

Trends

Between 1996–2016, non-tidal wetlands of the Delaware Estuary and Basin experienced a net decline of 340 hectares (840 acres), which is an average decline of 34 hectares (84 acres or 0.043%) per year (Figs 6.3.23-6.3.26; Table 6.3.11). The largest areas of non-tidal wetland loss were in the Lower Estuary and Bayshore, which saw a net decrease of 646 and 622 hectares (1,596 and 1,537 acres), respectively (Figs 6.3.28-6.3.32). The majority of those losses were emergent wetlands. Non-tidal wetlands in the Central region also experienced a net decline of 183 hectares (452 acres), which were mostly forested and emergent wetlands. In the Upper region, there was a net increase in non-tidal wetlands, where forested wetlands dominated losses.

Freshwater non-tidal wetlands losses totaled 1,764 hectares (4,359 acres) between 1996–2016 in the Estuary and Basin. Conversion to development and open space/agriculture/grasslands mostly drove these losses (Fig 6.3.33; Tables 11,12). About 1,510 hectares (3,731 acres) of freshwater non-tidal wetland gains countered those losses, which occurred by conversion from open water.

Discussion

Losses and prognosis

Non-tidal wetlands were historically lost in the Delaware Estuary and Basin primarily through conversion for agriculture and development. Human-mediated disturbances in these systems are considerable, and development pressures from increasing human populations are likely to continue to increase (Chapter 1). Losses due to developmental pressure are especially possible where state-specific legislation does not offer protection against losses or degradation (e.g., ample buffer distances).

Climatic changes, such as warming temperatures and altered precipitation patterns (Chapter 2), will likely

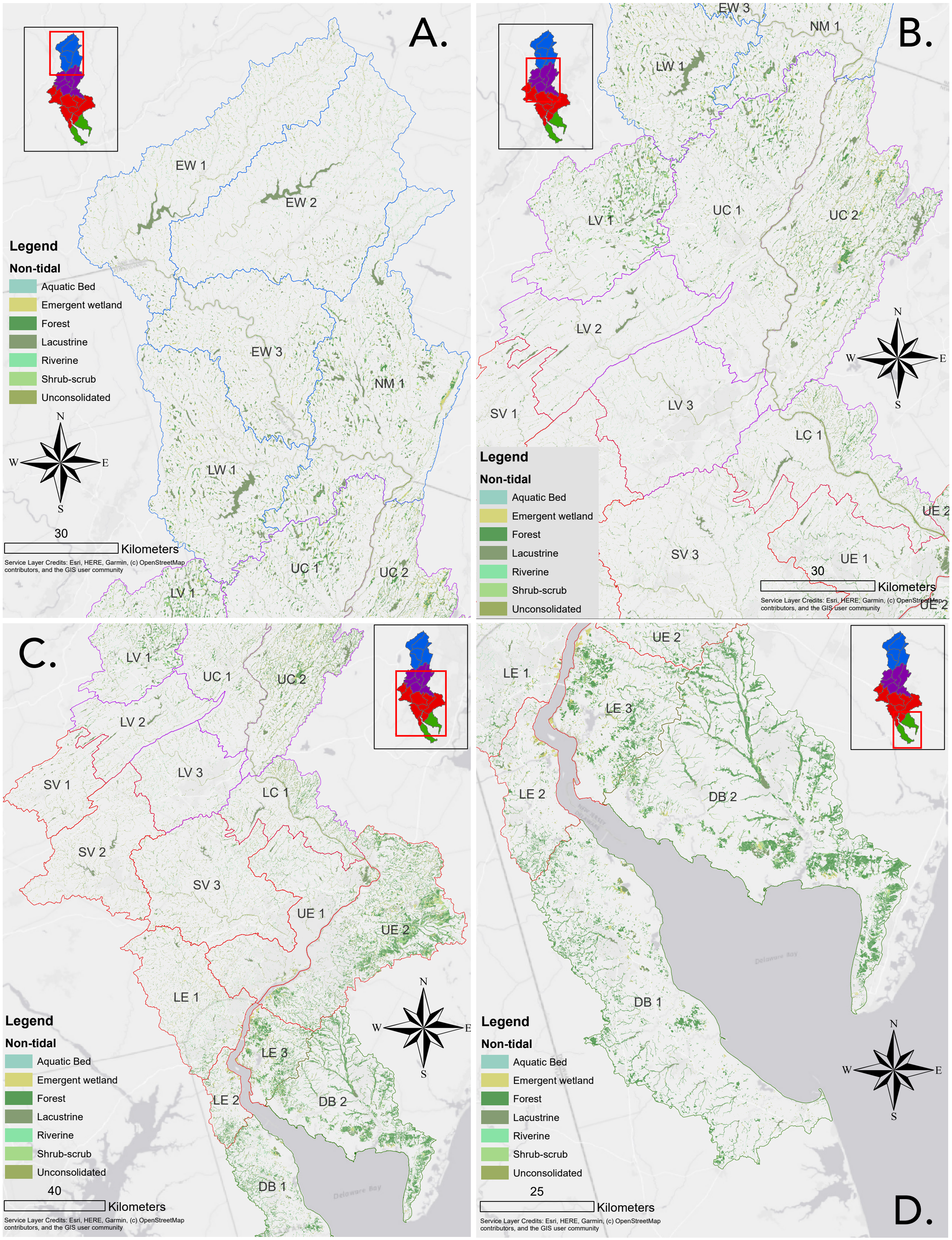


Figure 6.3.15 Cover of generalized NWI classifications of non-tidal wetlands in the Upper (A), Central (B), Lower (C), and Bay (D) regions of the Delaware Estuary and Basin.

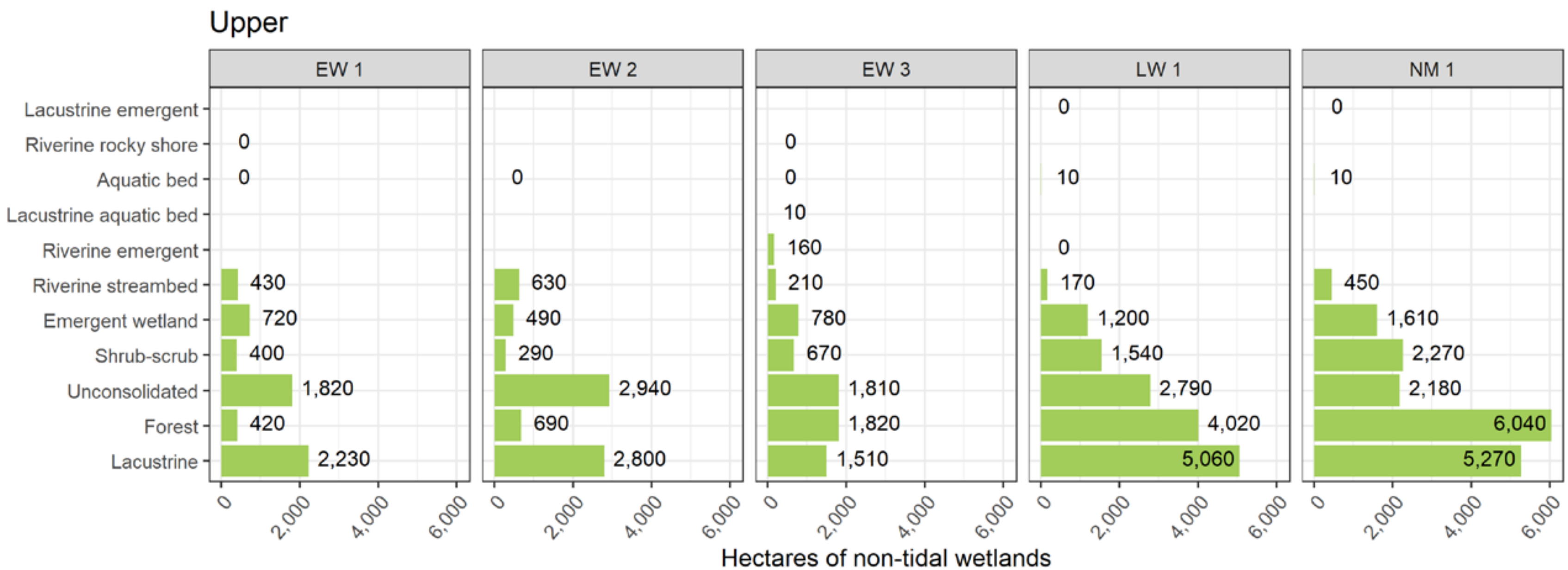


Figure 6.3.16 Cover (in hectares) of non-tidal wetland types in the Upper region based on NWI classifications.

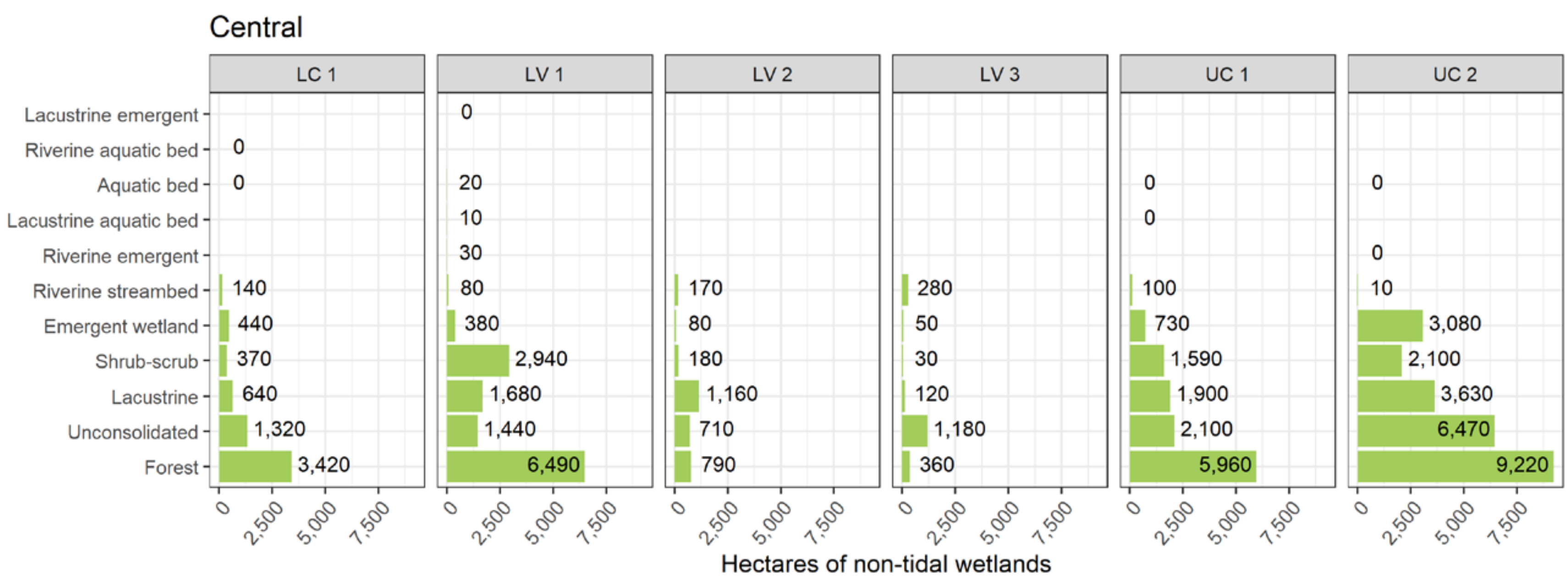


Figure 6.3.17 Cover (in hectares) of non-tidal wetland types in the Central region based on NWI classifications.

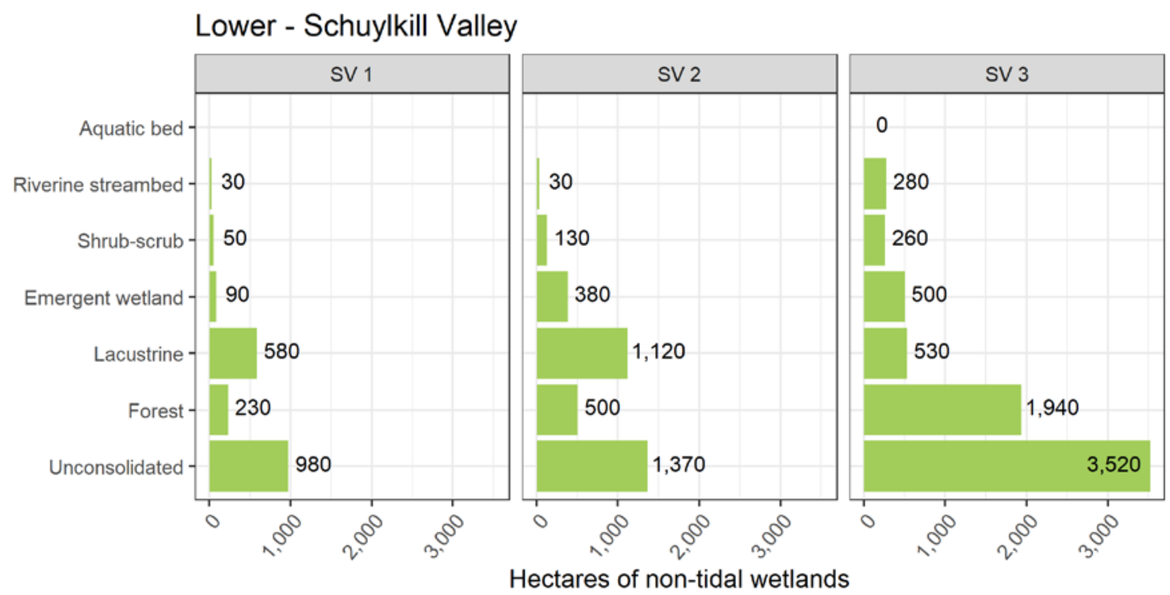


Figure 6.3.18 Cover (in hectares) of non-tidal wetland types in the Schuylkill Valley of the Lower region based on NWI classifications.

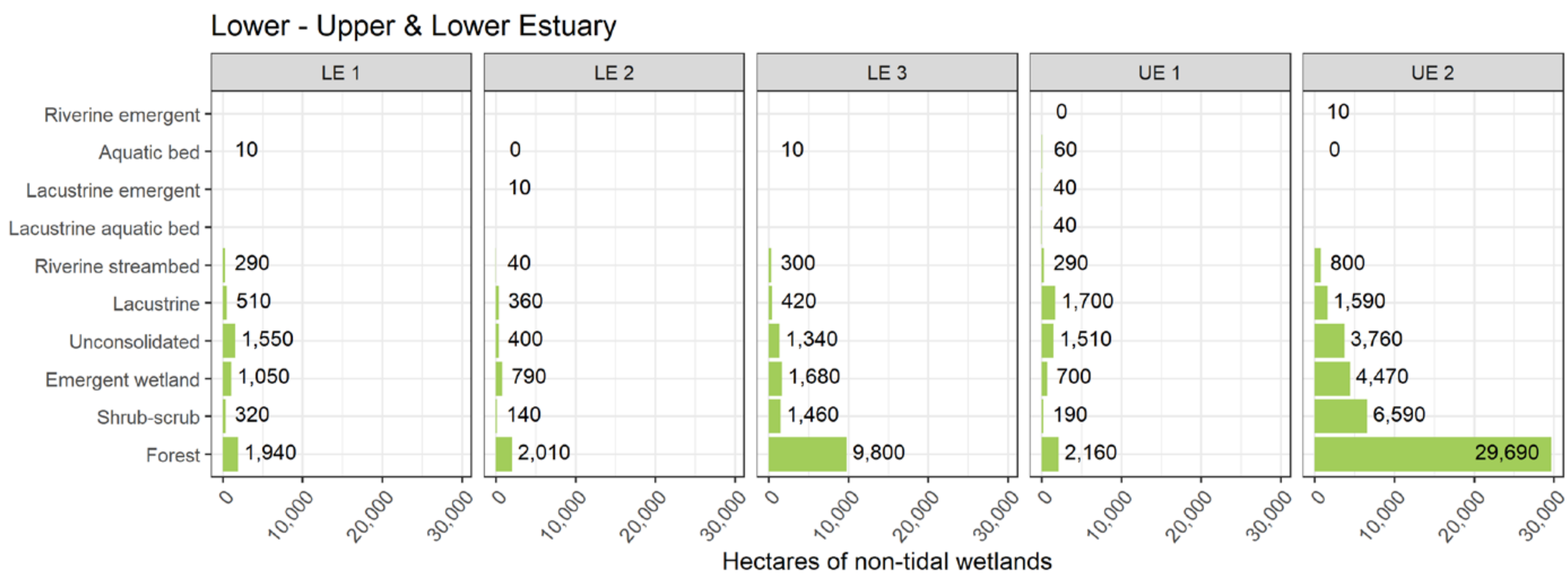


Figure 6.3.19 Cover (in hectares) of non-tidal wetland types in the Upper and Lower Estuary of the Lower region based on NWI classifications.

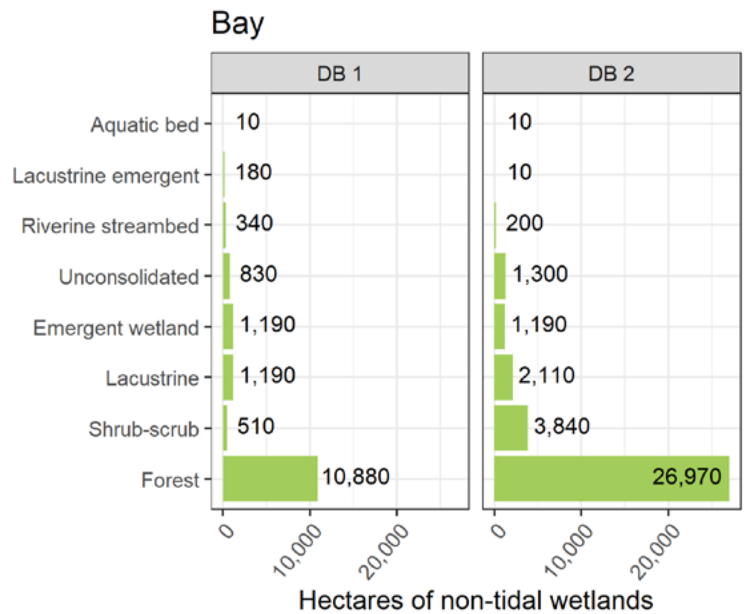


Figure 6.3.20 Cover (in hectares) of non-tidal wetland types in the Bay region based on NWI classifications.

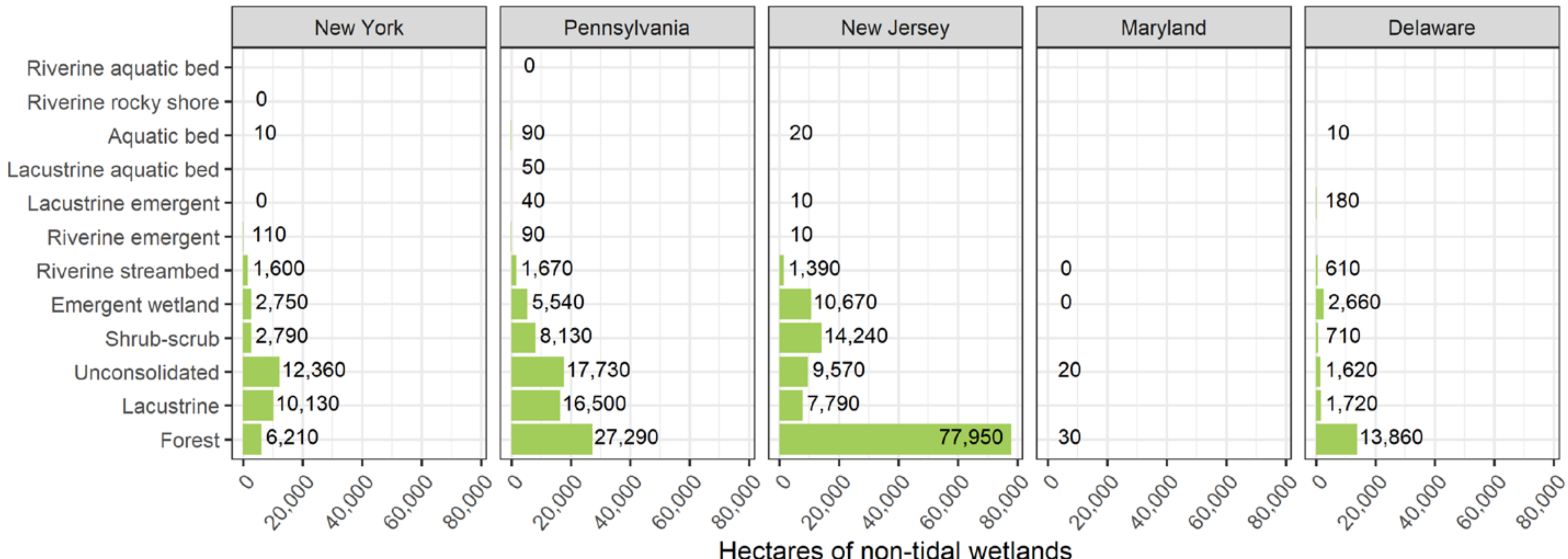


Figure 6.3.21 Cover (in hectares) of non-tidal wetland types in the Bay region based on NWI classifications.

Table 6.3.6 Cover of non-tidal wetlands in the Upper region of the Delaware Estuary and Basin.

Subregion	Type	Hectares	% of wetlands	% of region
EW 1	Aquatic Bed	0	0.00%	0.000%
	Emergent wetland	724	12.01%	0.420%
	Forest	420	6.97%	0.243%
	Lacustrine	2234	37.04%	1.294%
	Riverine rocky shore	2	0.03%	0.001%
	Riverine streambed	433	7.18%	0.251%
	Shrub-scrub	403	6.69%	0.234%
	Unconsolidated	1815	30.09%	1.052%
EW 2	Aquatic Bed	1	0.01%	0.000%
	Emergent wetland	491	6.26%	0.225%
	Forest	686	8.75%	0.315%
	Lacustrine	2799	35.68%	1.285%
	Riverine streambed	633	8.07%	0.291%
	Shrub-scrub	295	3.76%	0.135%
	Unconsolidated	2939	37.47%	1.350%
EW 3	Aquatic Bed	0	0.00%	0.000%
	Emergent wetland	782	11.23%	0.577%
	Forest	1816	26.05%	1.339%
	Lacustrine	1509	21.65%	1.113%
	Lacustrine aquatic bed	6	0.08%	0.004%
	Riverine emergent	164	2.35%	0.121%
	Riverine rocky shore	1	0.02%	0.001%
	Riverine streambed	209	3.00%	0.154%
	Shrub-scrub	671	9.62%	0.494%
	Unconsolidated	1812	26.00%	1.336%
NM 1	Aquatic Bed	8	0.05%	0.004%
	Emergent wetland	1608	9.02%	0.760%
	Forest	6045	33.92%	2.859%
	Lacustrine	5267	29.55%	2.491%
	Lacustrine emergent	0	0.00%	0.000%
	Riverine streambed	450	2.53%	0.213%
	Shrub-scrub	2267	12.72%	1.072%
	Unconsolidated	2176	12.21%	1.029%
LW 1	Aquatic Bed	9	0.06%	0.006%
	Emergent wetland	1195	8.08%	0.772%
	Forest	4021	27.19%	2.596%
	Lacustrine	5064	34.25%	3.270%
	Lacustrine emergent	2	0.01%	0.001%
	Riverine emergent	1	0.01%	0.001%
	Riverine streambed	166	1.12%	0.107%
	Shrub-scrub	1544	10.44%	0.997%
	Unconsolidated	2786	18.84%	1.799%

Table 6.3.7 Cover of non-tidal wetlands in the Central region of the Delaware Estuary and Basin.

Subregion	Type	Hectares	% of wetlands	% of region
UC 1	Aquatic Bed	2	0.02%	0.001%
	Emergent wetland	732	5.90%	0.361%
	Forest	5956	48.06%	2.936%
	Lacustrine	1905	15.37%	0.939%
	Lacustrine aquatic bed	1	0.01%	0.001%
	Riverine streambed	104	0.84%	0.051%
	Shrub-scrub	1592	12.85%	0.785%
	Unconsolidated	2101	16.96%	1.036%
UC 2	Aquatic Bed	0	0.00%	0.000%
	Emergent wetland	3083	12.57%	1.597%
	Forest	9222	37.61%	4.777%
	Lacustrine	3633	14.82%	1.882%
	Riverine emergent	3	0.01%	0.001%
	Riverine streambed	14	0.06%	0.007%
	Shrub-scrub	2101	8.57%	1.088%
	Unconsolidated	6465	26.37%	3.349%
LV 1	Aquatic Bed	18	0.13%	0.015%
	Emergent wetland	383	2.93%	0.327%
	Forest	6487	49.64%	5.546%
	Lacustrine	1683	12.88%	1.439%
	Lacustrine aquatic bed	12	0.09%	0.010%
	Lacustrine emergent	0	0.00%	0.000%
	Riverine emergent	25	0.20%	0.022%
	Riverine streambed	77	0.59%	0.066%
	Shrub-scrub	2938	22.48%	2.512%
	Unconsolidated	1443	11.05%	1.234%
LV 2	Emergent wetland	84	2.70%	0.075%
	Forest	794	25.54%	0.712%
	Lacustrine	1163	37.43%	1.044%
	Riverine streambed	171	5.50%	0.153%
	Shrub-scrub	183	5.90%	0.165%
	Unconsolidated	712	22.92%	0.639%
LV 3	Emergent wetland	52	2.56%	0.042%
	Forest	364	17.97%	0.293%
	Lacustrine	123	6.07%	0.099%
	Riverine streambed	278	13.74%	0.224%
	Shrub-scrub	29	1.41%	0.023%
	Unconsolidated	1179	58.25%	0.949%
LC 1	Aquatic Bed	0	0.00%	0.000%
	Emergent wetland	440	6.94%	0.374%
	Forest	3415	53.95%	2.904%
	Lacustrine	644	10.18%	0.548%

Table 6.3.8 Cover of non-tidal wetlands in the Central and Lower region of the Delaware Estuary and Basin.

Region	Subregion	Type	Hectares	% of wetlands	% of region
Central	LC 1 (con't)	Riverine aquatic bed	0	0.01%	0.000%
		Riverine streambed	144	2.27%	0.122%
		Shrub-scrub	368	5.81%	0.313%
		Unconsolidated	1320	20.84%	1.122%
Lower	SV 1	Emergent wetland	87	4.46%	0.098%
		Forest	229	11.72%	0.258%
		Lacustrine	581	29.81%	0.656%
		Riverine streambed	27	1.37%	0.030%
		Shrub-scrub	51	2.63%	0.058%
		Unconsolidated	975	50.01%	1.100%
	SV 2	Emergent wetland	382	10.82%	0.225%
		Forest	501	14.17%	0.295%
		Lacustrine	1,121	31.73%	0.660%
		Riverine streambed	32	0.90%	0.019%
		Shrub-scrub	125	3.55%	0.074%
		Unconsolidated	1,372	38.83%	0.807%
	SV 3	Aquatic Bed	2	0.02%	0.001%
		Emergent wetland	499	7.10%	0.216%
		Forest	1,941	27.60%	0.838%
		Lacustrine	529	7.53%	0.229%
		Riverine streambed	276	3.92%	0.119%
		Shrub-scrub	263	3.74%	0.114%
		Unconsolidated	3,523	50.10%	1.522%
	UE 1	Aquatic Bed	56	0.84%	0.031%
		Emergent wetland	701	10.51%	0.386%
		Forest	2,161	32.36%	1.189%
		Lacustrine	1,701	25.47%	0.936%
		Lacustrine aquatic bed	35	0.53%	0.019%
		Lacustrine emergent	36	0.53%	0.020%
		Riverine emergent	2	0.03%	0.001%
		Riverine streambed	289	4.33%	0.159%
		Shrub-scrub	189	2.84%	0.104%
		Unconsolidated	1,506	22.56%	0.829%
	UE 2	Aquatic Bed	3	0.01%	0.001%
		Emergent wetland	4,466	9.52%	1.653%
		Forest	29,687	63.29%	10.990%
		Lacustrine	1,591	3.39%	0.589%
		Riverine emergent	8	0.02%	0.003%
		Riverine streambed	802	1.71%	0.297%
		Shrub-scrub	6,587	14.04%	2.438%
		Unconsolidated	3,762	8.02%	1.393%

Table 6.3.9 Cover of non-tidal wetlands in the Lower and Bay region of the Delaware Estuary and Basin.

Region	Subregion	Type	Hectares	% of wetlands	% of region
Lower	LE 1	Aquatic Bed	8	0.14%	0.005%
		Emergent wetland	1,054	18.58%	0.674%
		Forest	1,943	34.25%	1.243%
		Lacustrine	513	9.04%	0.328%
		Riverine streambed	291	5.13%	0.186%
		Shrub-scrub	315	5.56%	0.202%
		Unconsolidated	1,549	27.30%	0.990%
	LE 2	Aquatic Bed	0	0.01%	0.001%
		Emergent wetland	786	21.01%	1.957%
		Forest	2,006	53.65%	4.998%
		Lacustrine	362	9.67%	0.901%
		Lacustrine emergent	6	0.16%	0.015%
		Riverine streambed	37	0.98%	0.091%
		Shrub-scrub	143	3.83%	0.357%
Bay	DB 1	Aquatic Bed	5	0.03%	0.003%
		Emergent wetland	1,191	7.87%	0.724%
		Forest	10,881	71.94%	6.617%
		Lacustrine	1,191	7.87%	0.724%
		Lacustrine emergent	178	1.18%	0.108%
		Riverine streambed	339	2.24%	0.206%
		Shrub-scrub	513	3.39%	0.312%
		Unconsolidated	828	5.47%	0.504%
	DB 2	Aquatic Bed	7	0.02%	0.003%
		Emergent wetland	1,189	3.34%	0.581%
		Forest	26,970	75.74%	13.183%
		Lacustrine	2,107	5.92%	1.030%
		Lacustrine emergent	5	0.01%	0.003%
		Riverine streambed	200	0.56%	0.098%
		Shrub-scrub	3,836	10.77%	1.875%
		Unconsolidated	1,295	3.64%	0.633%

Table 6.3.10 Cover of non-tidal wetlands in the Delaware Estuary and Basin by State.

State	Type	Hectares	% of wetlands
New York	Aquatic Bed	9	0.026%
	Emergent wetland	2,749	7.644%
	Forest	6,209	17.265%
	Lacustrine	10,129	28.164%
	Lacustrine emergent	0	0.001%
	Riverine emergent	110	0.305%
	Riverine rocky shore	3	0.008%
	Riverine streambed	1,597	4.442%
	Shrub-scrub	2,793	7.766%
	Unconsolidated	12,364	34.380%
Pennsylvania	Aquatic Bed	95	0.123%
	Emergent wetland	5,538	7.180%
	Forest	27,293	35.384%
	Lacustrine	16,498	21.390%
	Lacustrine aquatic bed	54	0.070%
	Lacustrine emergent	37	0.048%
	Riverine aquatic bed	0	0.001%
	Riverine emergent	85	0.110%
	Riverine streambed	1,673	2.170%
	Shrub-scrub	8,133	10.545%
	Unconsolidated	17,725	22.980%

State	Type	Hectares	% of wetlands
New Jersey	Aquatic Bed	21	0.017%
	Emergent wetland	10,666	8.769%
	Forest	77,947	64.080%
	Lacustrine	7,790	6.404%
	Lacustrine emergent	5	0.004%
	Riverine emergent	8	0.006%
	Riverine streambed	1,389	1.142%
	Shrub-scrub	14,239	11.706%
	Unconsolidated	9,574	7.870%
Maryland	Emergent wetland	1	2.093%
	Forest	34	63.054%
	Riverine streambed	2	4.288%
	Unconsolidated	16	30.566%
Delaware	Aquatic Bed	6	0.027%
	Emergent wetland	2,659	12.440%
	Forest	13,862	64.861%
	Lacustrine	1,724	8.067%
	Lacustrine emergent	184	0.860%
	Riverine streambed	607	2.838%
	Shrub-scrub	713	3.338%
	Unconsolidated	1,617	7.568%

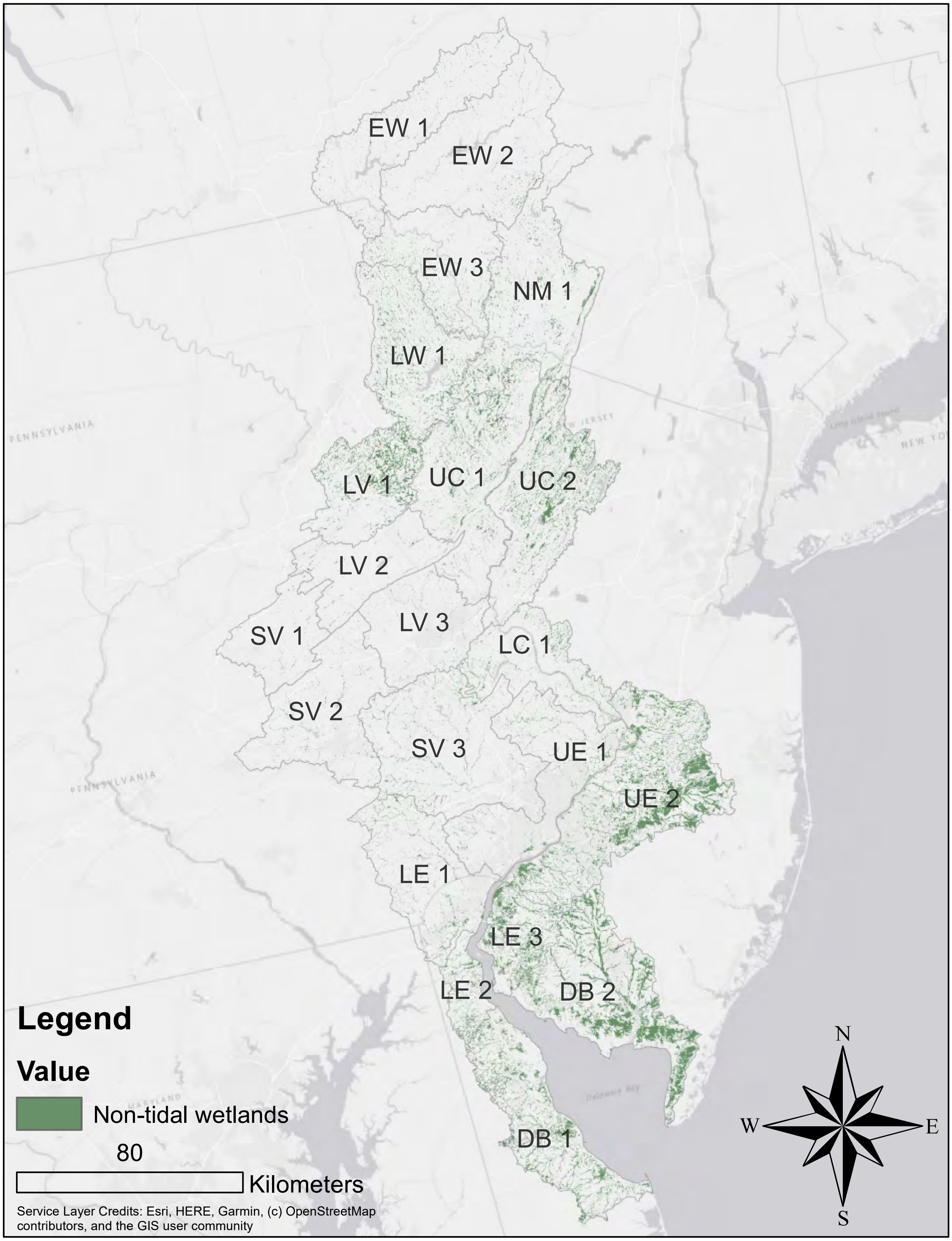


Figure 6.3.22 Cover of C-CAP non-tidal wetlands in the Delaware Estuary and Basin.

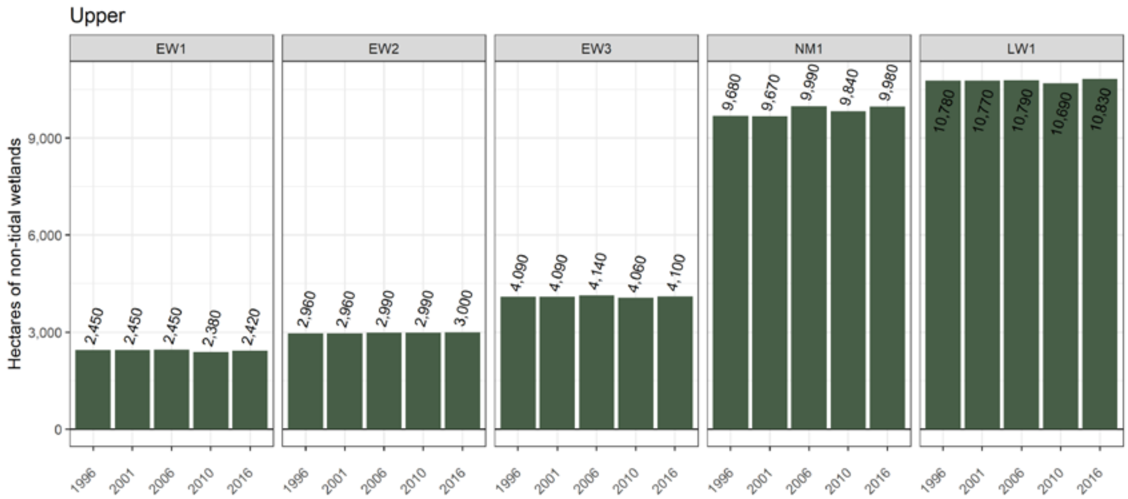


Figure 6.3.23 Total non-tidal wetland cover in the Upper region over time by subregion.

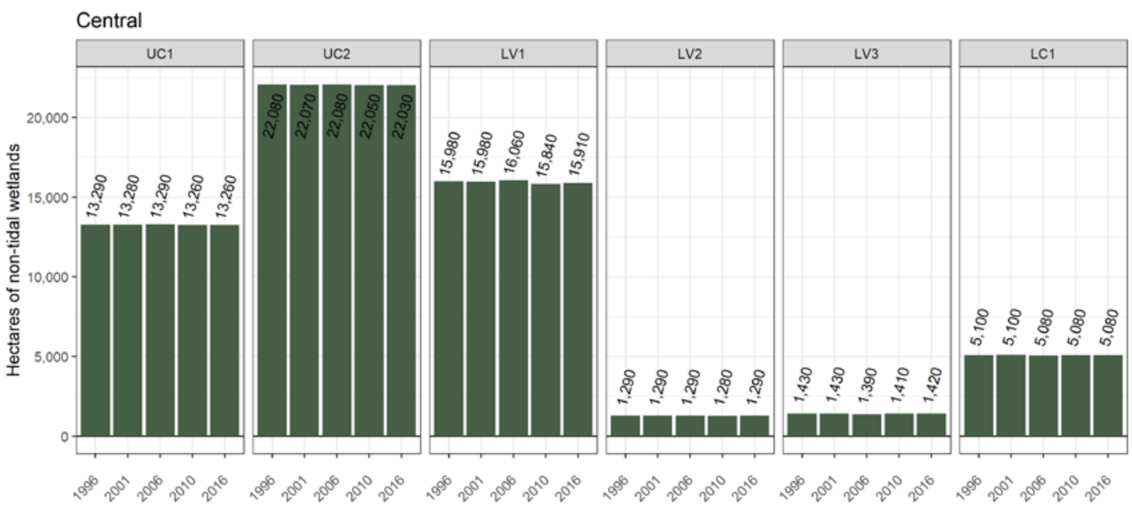


Figure 6.3.24 Total non-tidal wetland cover in the Central region over time by subregion.

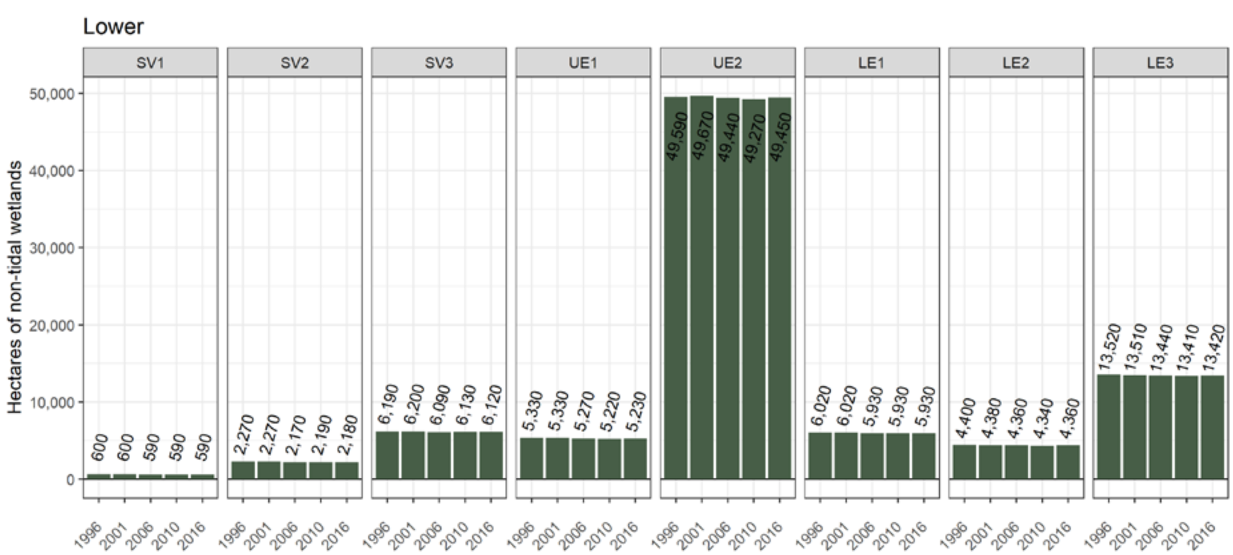


Figure 6.3.25 Total non-tidal wetland cover in the Lower region over time by subregion.

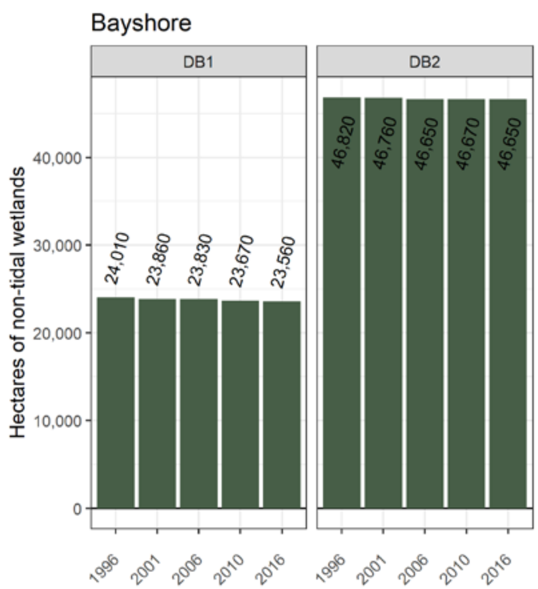


Figure 6.3.26 Total non-tidal wetland cover in the Estuary over time by subregion.

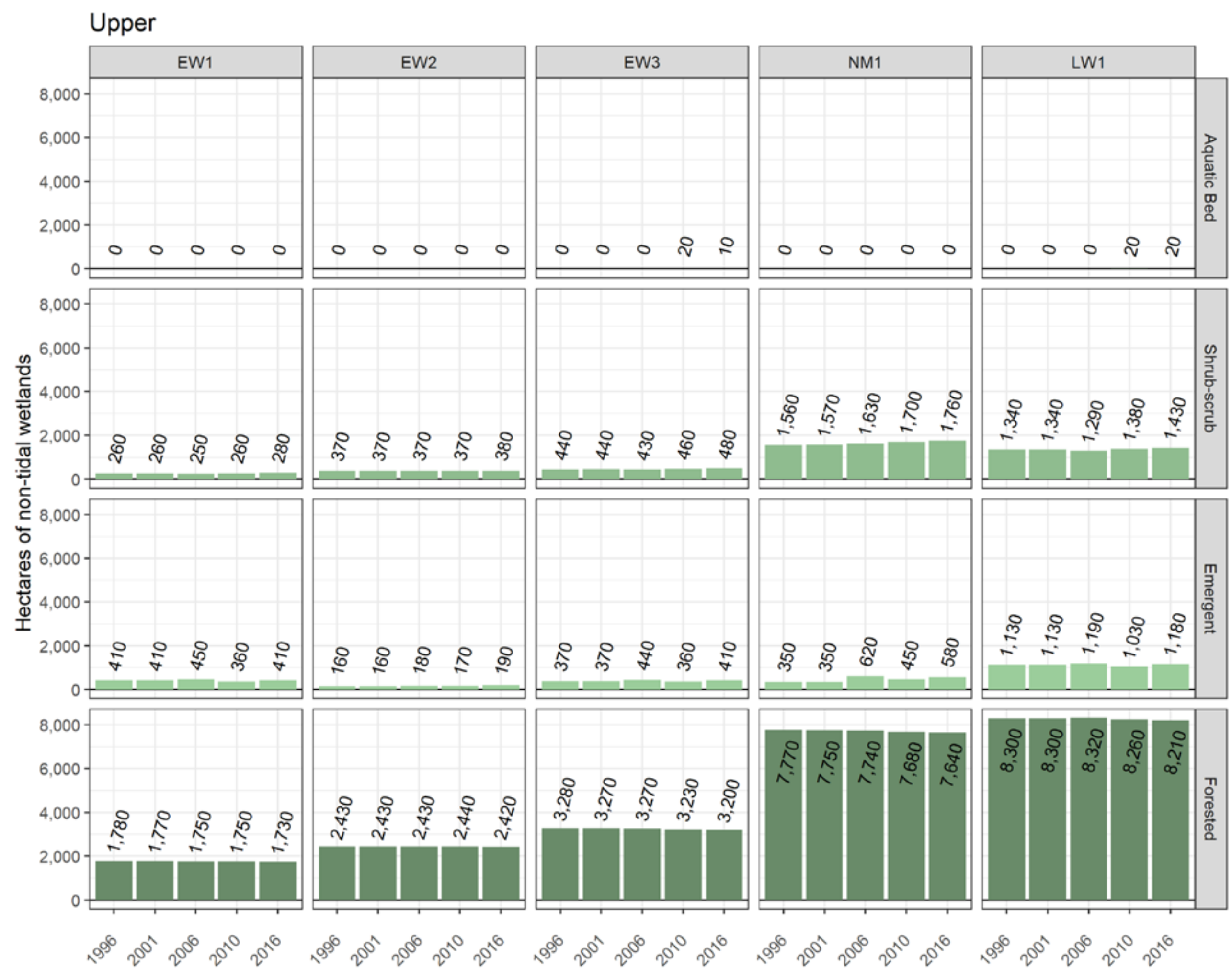


Figure 6.3.27 Total non-tidal wetland cover in the Upper over time by wetland type.

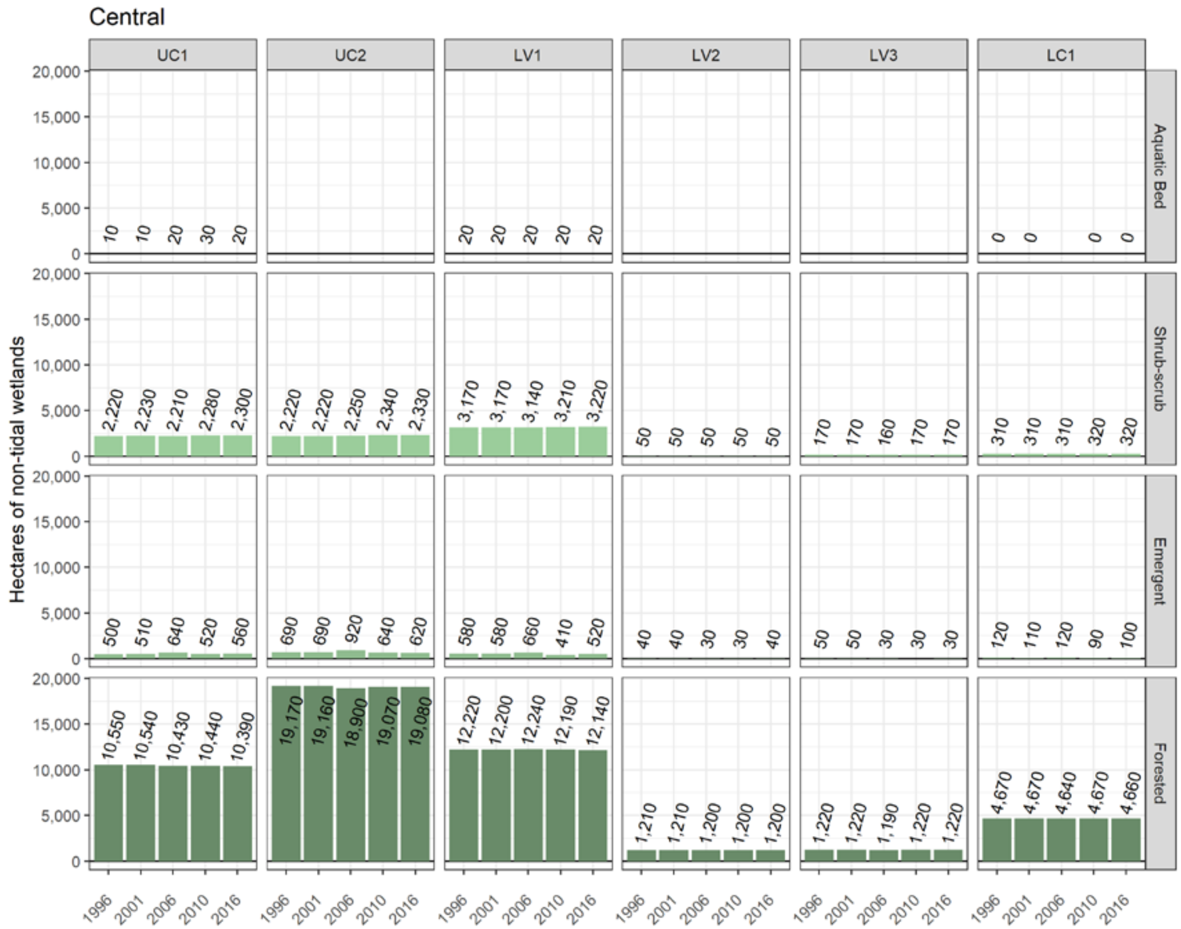


Figure 6.3.28 Total non-tidal wetland cover in the Central over time by wetland type.

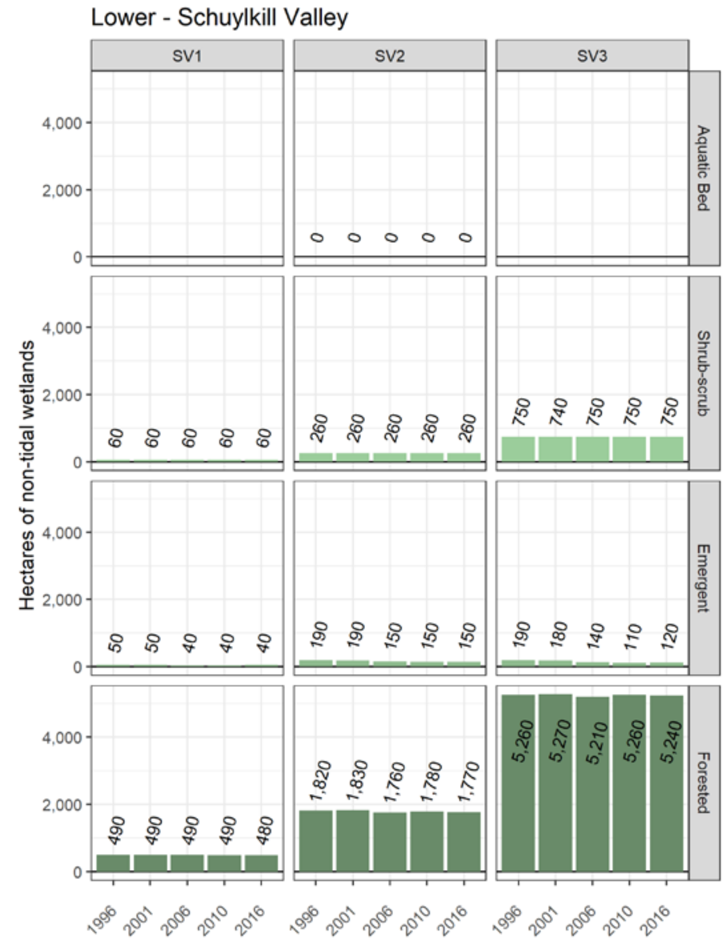


Figure 6.3.29 Schuylkill Valley total non-tidal wetland cover in the Lower region over time by wetland type.

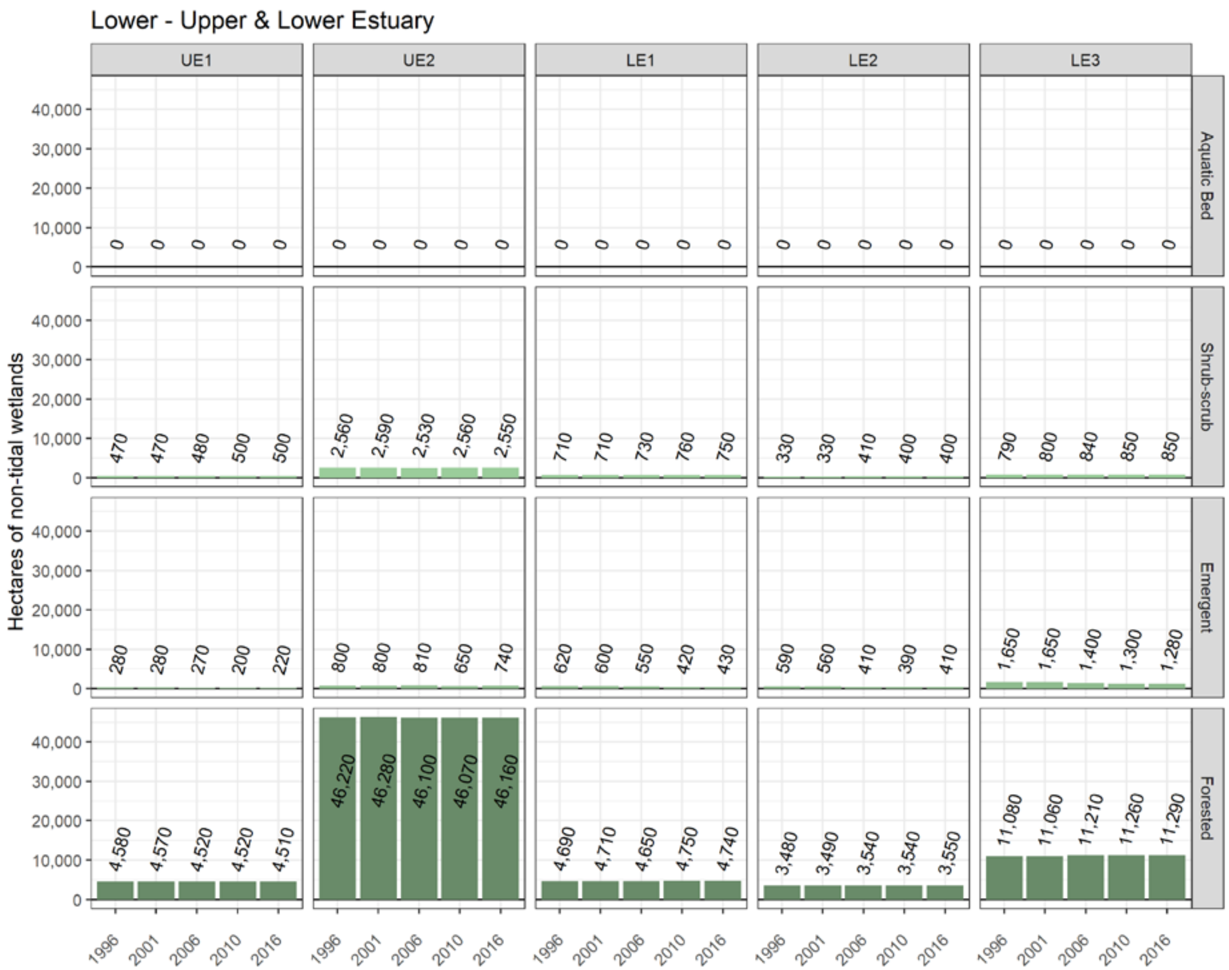


Figure 6.3.30 Upper and Lower Estuary total non-tidal wetland cover in the Lower region over time by wetland type.

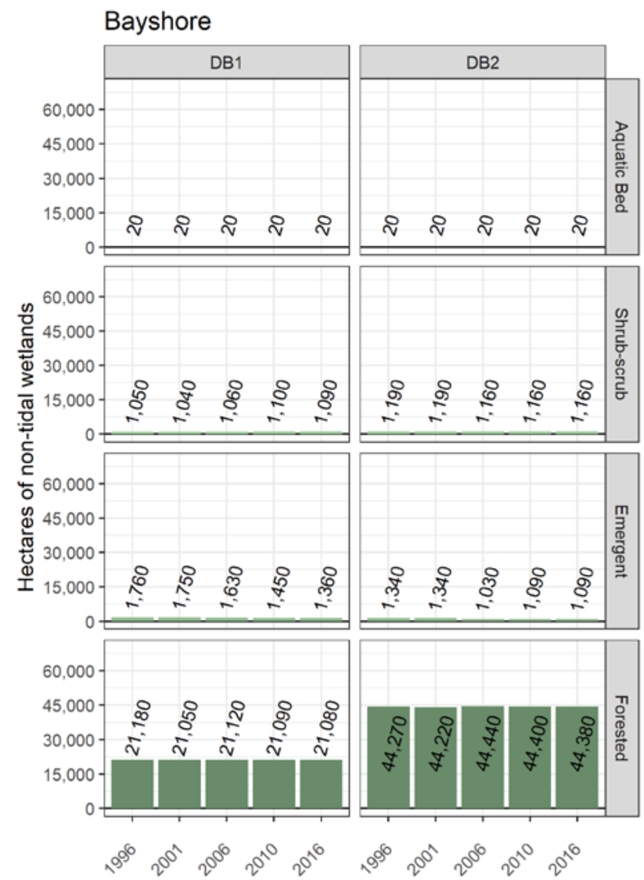


Figure 6.3.31 Total non-tidal wetland cover in the Bayshore region over time by wetland type.

Figure 6.3.32 Non-tidal wetland cover by region and type over time.

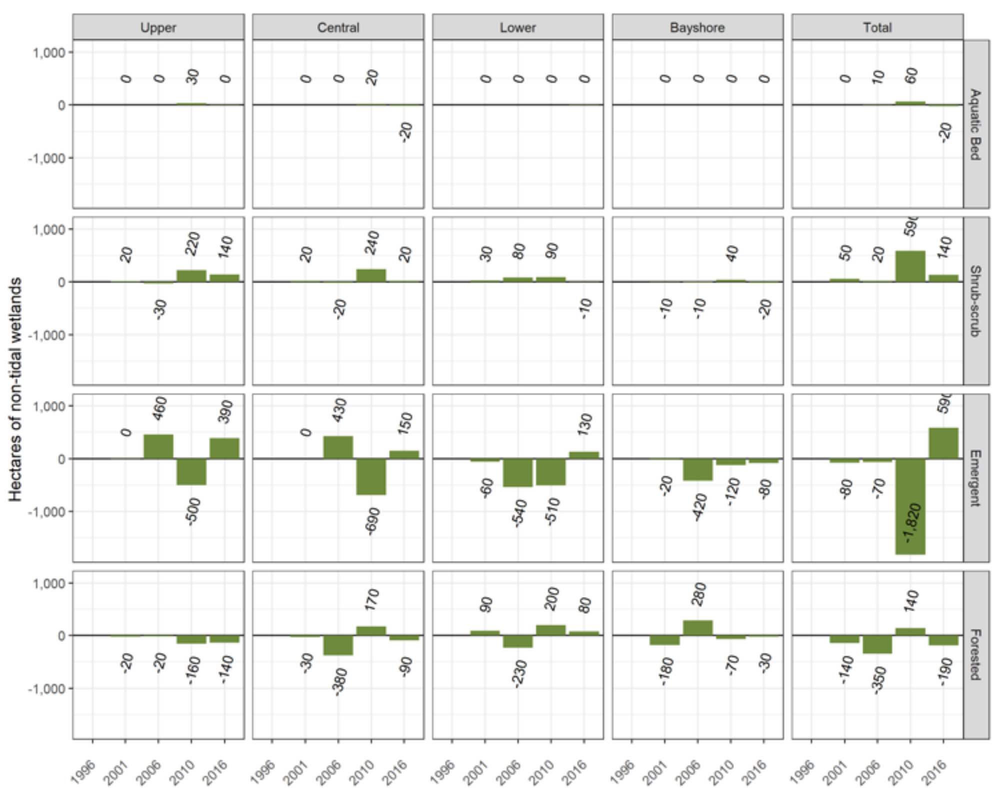


Figure 6.3.33 Non-tidal wetland cover changes relative to the 1996 baseline.

Table 6.3.11 Non-tidal wetland cover change from 1996-2016.

Region	Type	Hectares		Net change (ha)	Change/yr	
		1996	2016		ha	%
Upper	Aquatic Bed	8	40	32	1.6	20.0%
	Shrub-scrub	3,973	4,318	345	17	0.43%
	Emergent	2,418	2,764	346	17	0.72%
	Forested	23,551	23,209	-342	-17	-0.07%
Central	Aquatic Bed	31	41	10	0.5	1.61%
	Shrub-scrub	8,129	8,386	257	13	0.16%
	Emergent	1,983	1,868	-115	-5.8	-0.29%
	Forested	49,025	48,690	-335	-17	-0.03%
Lower	Aquatic Bed	6	7	1	0.05	0.83%
	Shrub-scrub	5,935	6,126	191	9.6	0.16%
	Emergent	4,369	3,399	-970	-49	-1.11%
	Forested	77,616	77,748	132	6.6	0.01%
Bayshore	Aquatic Bed	36	44	8	0.4	1.11%
	Shrub-scrub	2,238	2,242	3	0.2	0.01%
	Emergent	3,100	2,457	-643	-32	-1.04%
	Forested	65,454	65,464	10	0.5	0.001%
Total	Aquatic Bed	81	132	51	2.6	3.15%
	Shrub-scrub	20,275	21,071	796	40	0.20%
	Emergent	11,871	10,488	-1,383	-69	-0.58%
	Forested	215,646	215,112	-534	-27	-0.01%

Table 6.3.12 Conversions of non-tidal wetlands from 1996 to 2016.

Land use type	Loss		Gain		Net	
	ha	%	ha	%	ha	%
Development	522	0.21%	0	-	-522	-0.207%
Open space, agricultural, & grassland	601	0.24%	247	0.10%	-354	-0.140%
Forest & scrub-shrub	388	0.15%	68	0.03%	-320	-0.127%
Unconsolidated & bare	125	0.05%	17	0.01%	-108	-0.043%
Open water	25	0.01%	1172	0.47%	1147	0.455%
Aquatic bed	0	-	5.9	0.002%	5.9	0.002%
Salt & brackish	103	0.04%	0	-	-103	-0.041%

affect non-tidal wetland extent in the Delaware Estuary and Basin. On one hand, a longer growing season and warmer temperatures likely will increase total non-tidal marsh productivity. Enriched atmospheric carbon dioxide will likely also enhance primary productivity within C_3-photosynthesis-dominated non-tidal wetlands (i.e., *Schoenoplectus* sp., *Bolboschoenus* sp., *Typha* sp.) (Ecker et al. 2020). Increased precipitation may sustain wetlands and could cause increased runoff into streams. Yet, heavier rains coupled with more intense periods of drought may damage wetland integrity through plant stress (Garsson et al. 2014) and drive soil geochemical changes (Stirling et al. 2020). For non-tidal wetlands in areas proximal to tidal wetlands, conversion to salt and brackish tidal marshes are additional pressures of climate change brought on by rising sea levels (Herbert et al. 2015). Here, we found that over 100 hectares (247 acres) of non-tidal wetlands were lost from 1996-2016 due to their conversion to salt and brackish wetlands. Saltwater intrusion and increased tidal will drive freshwater non-tidal wetland conversion to salt/brackish wetlands as sea levels continue to rise.

Actions and Needs

Non-tidal wetlands are especially vulnerable to degradation and fragmentation across the landscape. Active management of non-tidal wetlands includes tactics aimed at reducing fragmentation and relieving key pressures such as deer/grazer overabundance, invasive dominance, excessive runoff, and narrow buffers. Additionally, as wetlands are especially vulnerable to fragmentation, efforts to protect and preserve non-tidal wetlands should include tactics that aim to connect fragmented wetlands through easements, land purchases, and increase regulatory protection measures where appropriate. These strategies are especially key in protecting high-value wetlands, like those that are relatively undisturbed, have high functionality and biodiversity, and those that serve as habitat for protected, rare, or other species of concern.

Continued Monitoring and Scientific Study

Better mapping of the variety of non-tidal wetlands in the Delaware Estuary and Basin is a fundamental need to enact more meaningful protections and research on these systems. Mapping efforts might include more coupled high-resolution satellite imagery analysis with on-the-ground verification (Mahdianpari et al. 2020). It will be important to continue to improve products, like NWI, for difficult to map non-tidal wetlands, such as ephemeral wetlands (like vernal pools) or those which have been historically manipulated, degraded, or converted. Improved mapping might also help differentiation between human-created wetland-like areas (e.g., stormwater catchments) and naturally occurring non-tidal wetlands. The number of stormwater ponds has drastically increased with increasing development. For instance, in Delaware from 1992-2007, nearly 65% of "new" wetlands were stormwater ponds (Tiner et al. 2011). Although wetland vegetation may colonize these ponds giving them similar imagery signatures to natural non-tidal wetlands, stormwater ponds do not function similarly to natural systems. With imagery analysis, the increase in stormwater ponds might artificially buffer losses of natural non-tidal wetlands. Research on how to improve the function of stormwater ponds might also benefit wildlife and provide more ecosystem services if trends in stormwater pond creation continue.

Further research on non-tidal wetland functions, as well as their responses to environmental changes, such as development and climate change, is also a continued need in the Delaware Estuary and Basin. Systematic or holistic watershed-wide assessments of vulnerability based on hydrogeomorphic classes, adaptive capacities, and exposures may provide critical direction or prioritization schemes for at-risk non-tidal wetlands in the region (Warthrop et al. 2019). Research on or quantification of the various ecosystem services provided by non-tidal wetlands will also be important for prioritizing active management. For instance, more information on carbon storage potential, geochemical cycling, biodiversity, flood protection, as well as the cultural or social value of the various types of non-tidal wetlands will be important for determining the return on investment of restoration or management activities. Ecosystem service information could support the active protection of these habitats when the preservation or restoration of these systems directly competes with development interests. Climate change may also alter ecosystem functioning, so research on how increasing temperature and atmospheric CO2, as well as changing precipitation patterns, affect non-tidal wetland functions is another critical research need (Moomaw et al. 2018).

Non-tidal wetlands across the landscape represent the intimate connection between the ecology of surface landscape and its connections with hydrology (see Feature 4 for future directions on hydrological condition assessments). Wetlands act as an essential buffer for high quality water resources (surface and ground), and it will be important to research how to maintain, preserve, ad enhance wetland acreage to protect good water quality as populations in the region increase (Chapter 1). Groundwater, in addition to surface water, is a key source of drinkable water for many people who reside in the Delaware Estuary and Basin. And so, understanding the connections among non-tidal wetland type, condition, extent, and aquifer recharge/resilience, especially in critical drinking water areas (Chapter 3), is also important to ensure the longevity or sustainability of aquifers as a source of drinking water.

References

Cowardin, L.M., 1979. *Classification of wetlands and deepwater habitats of the United States*. Fish and Wildlife Service, US Department of the Interior.

Ecker, M., Kelley, D. and Sato, H., 2020. Modelling the effects of CO2 on C3 and C4 grass competition during the mid-Pleistocene transition in South Africa. *Scientific reports*, *10*(1), pp.1-8.

Garssen, A.G., Verhoeven, J.T. and Soons, M.B., 2014. Effects of climate-induced increases in summer drought on riparian plant species: A meta-analysis. *Freshwater Biology*, *59*(5), pp.1052-1063.

Herbert, E.R., Boon, P., Burgin, A.J., Neubauer, S.C., Franklin, R.B., Ardón, M., Hopfensperger, K.N., Lamers, L.P. and Gell, P., 2015. A global perspective on wetland salinization: ecological consequences of a growing threat to freshwater wetlands. *Ecosphere*, *6*(10), pp.1-43.

Mahdianpari, M., Granger, J.E., Mohammadimanesh, F., Salehi, B., Brisco, B., Homayouni, S., Gill, E., Huberty, B. and Lang, M., 2020. Meta-analysis of wetland classification using remote sensing: A systematic review of a 40-year trend in North America. Remote Sensing, 12(11), p.1882.

Moomaw, W.R., Chmura, G.L., Davies, G.T., Finlayson, C.M., Middleton, B.A., Natali, S.M., Perry, J.E., Roulet, N. and Sutton-Grier, A.E., 2018. Wetlands in a changing climate: science, policy and management. *Wetlands*, *38*(2), pp.183-205.

Stirling, E., Fitzpatrick, R.W. and Mosley, L.M., 2020. Drought effects on wet soils in inland wetlands and peatlands. *Earth-Science Reviews*, *210*, p.103387.

Tiner, R.W., M.A. Biddle, A.D. Jacobs, A.B. Rogerson and K.G. McGuckin. 2011. Delaware Wetlands: Status and Changes from 1992 to 2007. Cooperative National Wetlands Inventory Publication. U.S. Fish and Wildlife Service, Northeast Region, Hadley, MA and the Delaware Department of Natural Resources and Environmental Control, Dover, DE. 35 pp https://documents.dnrec.delaware.gov/Admin/DelawareWetlands/Documents/Delaware%20Wetlands%20Status%20and%20Changes%20from%201992%20to%202007%20FINAL2012.pdf

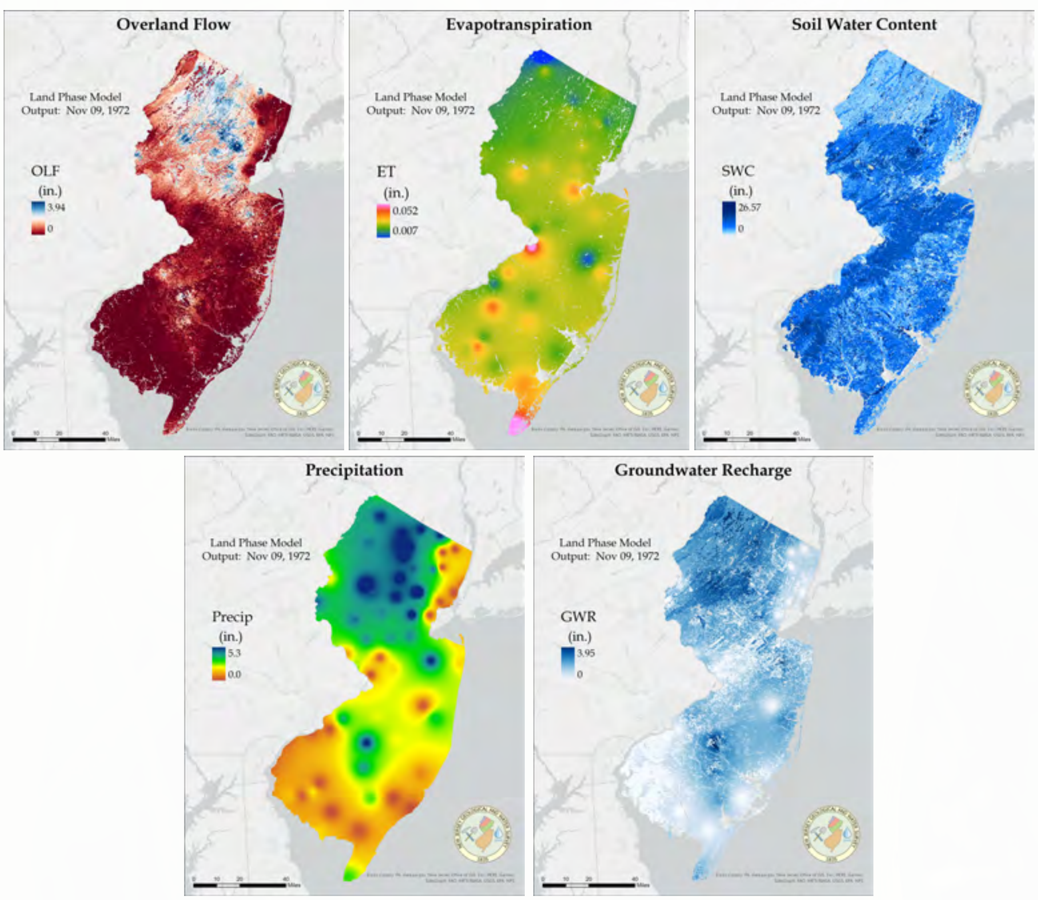
Wardrop, D.H., Hamilton, A.T., Nassry, M.Q., West, J.M. and Britson, A.J., 2019. Assessing the relative vulnerabilities of Mid-Atlantic freshwater wetlands to projected hydrologic changes. Ecosphere, 10(2), p.e02561.

Weis, J.S., Harman, C., Ravit, B Watson, E.B. 2020. The status and future of tidal marshes in New Jersey faced with sea level rise. New Jersey Department of Environmental Protection Science Advisory Board Report. https://dep.nj.gov/wp-content/uploads/sab/sab-salt-marsh.pdf

Land Phase Hydrologic Model

Richard Grabowski
New Jersey Geologic and Water Survey

The New Jersey Geologic and Water Survey (NJGWS) is in the initial phases of developing a land phase hydrologic model to analyze the status and trends of hydrologic processes across New Jersey, including the State's various land uses, land covers, and soils, from 1921 through near current times. The model simulates daily hydrologic processes including precipitation, overland flow, evapotranspiration (based on surface air temperature and daylight hours), ground water recharge, and soil water content and builds off of previous NJGWS research on groundwater recharge documented in Geological Survey Report 32. Initial results targeted for 2022 are expected to be aggregated to watershed management areas and counties. Trends associated with climate change will be assessed using moving 30-year average values of the hydrologic processes. Model results combined with analyses of the status and trends of water use data are expected to improve understanding of the factors associated with changing stream flow trends that affect water availability, water quality and aquatic habitats. As time and resources allow, NJGWS plans to work toward downscaling results to smaller areas such as riparian corridors. Since significant work is still required by NJGWS to refine the model, the specific outcomes of the model and modeling process will evolve and be better defined over time. The Partnership for the Delaware Estuary (PDE) and NJGWS are discussing the feasibility of applying the model's methodology to the Delaware Estuary and the Delaware Basin. Example outputs for a single day are shown below.



6.4 Fish Passage

Introduction

The Delaware River is unique along the Atlantic Coast in that it is free flowing along its entire length. Migratory fish have access to the entire mainstem river and far up into its headwaters where in other similar East Coast aquatic systems they have long been extirpated. Diadromous fish like American shad, alewife, blueback herring, striped bass, sea lamprey, and American eel can travel over 300 miles (483 km) from the mouth of the river up to its origin (and back out to the ocean) without being blocked by a barrier. For this reason the Delaware holds enormous potential for the recovery of migratory fish, but there are still over 1,400 dams on tributaries to the Delaware River. Reduced aquatic connectivity, the fragmentation of river habitats by dams, road-stream crossings (e.g., culverts) and other aquatic barriers, pose primary threats to aquatic species in the Delaware River Basin. These barriers limit the ability of sea-run fish to reach freshwater spawning and rearing habitats in important tributaries and prevent resident fish populations from moving between other critical habitats. Some resident species, such as the tessellated darter, also serve as host fish for certain freshwater mussels. Consequently, the ability of fish like this one to move within a stream system is also critical for freshwater mussels, which rely on host fish to disperse their young and colonize new habitats.

Description of the Indicator

The number of fish barriers (dams only) were calculated using the results of the Northeast Aquatic Connectivity project completed in 2011 and updated 2017. These results were furthered updated within the Delaware River Basin in 2021 as part of the development of a Restoration Roadmap for American Shad, Alewife and Blueback Herring in the Delaware Basin (Figures 6.4.1-6.4.2; DeSalvo et al. 2022). The number of dams removed in the Delaware River was calculated using American River's dam removal database. To be included in the database, a significant portion of the dam must have been removed for the full height of the dam, such that ecological function, natural river flow and fish passage can be restored at the site (American Rivers 2017).

Present Status

The Delaware River is distinguished by being the longest free-flowing river in the eastern United States. Many tributaries lack dams in their downstream portions and thus allow migratory fish like river herring to access spawning habitat downstream of any barrier. For example, the Rancocas, Flatbrook, Beaverkill, Paulins Kill, and Neversink River systems all have significant habitat available for migratory fish. Progress has been made opening up tributaries via dam removal over the last 15 year, however, that success is tempered by the lack of success in effective fish passage that was hoped for with the installation of fish ladders on the dams in the Schuylkill and Lehigh River in particular.

Past Trends

Between 2012-2016, 43 dams were removed in the Delaware River Basin, and between 2017-2021, 29 dams were removed in the Delaware River Basin (Figure 6.4.3). Although the number of dam removals has declined over the last five years, it should be noted the COVID-19 pandemic likely had a significant impact on the ability to move dam removals forward in 2020 and 2021. The most successful models for dam removal success seem to be a sustained focus on a watershed by a conservation group or a group of partners working toward a shared goal. Wildlands Conservancy has been active in the Lehigh Watershed over the last decade with a significant number of dam removals. Efforts by the New Jersey

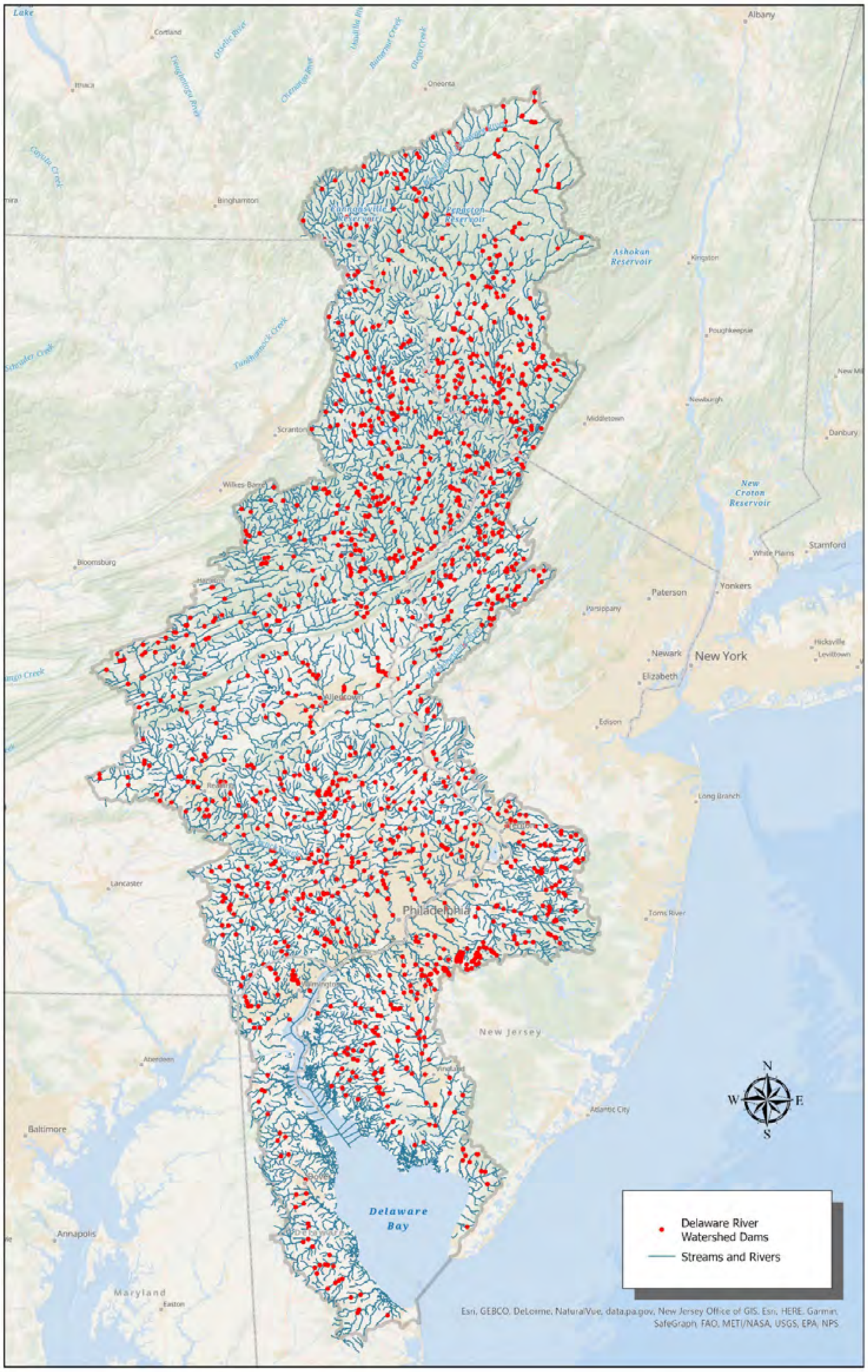


Figure 6.4.1 Fish barriers (dams only) in the Delaware Estuary and Basin from the results of the Northeast Aquatic Connectivity project completed in 2011 and updated 2017.

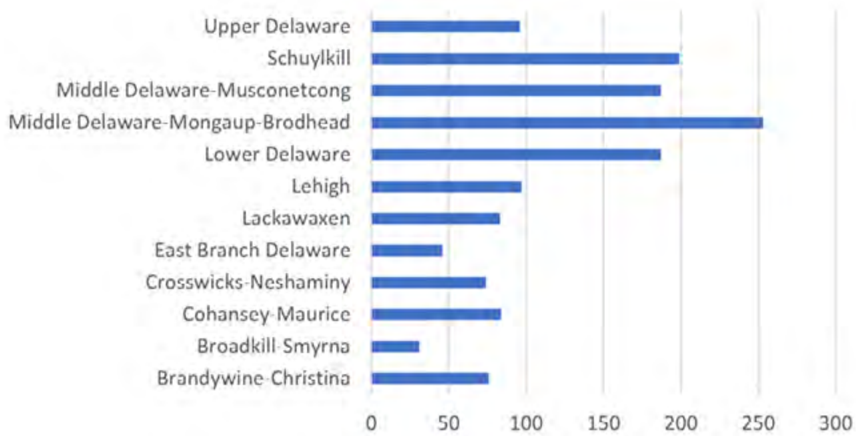


Figure 6.4.2 Total dams in the Delaware Estuary and Basin per HUC8 watershed.

Chapter of the Nature Conservancy have opened up miles of the Paulins Kill and returned American Shad to the Paulinskill for the first time in 100 years. Over the last decade the Musconetcong River Watershed Association, USFWS, NJDEP and other partners have worked to remove a total of 5 dams on the Musconetcong River. Studies are underway for the sixth dam on the river to be removed. American Shad have returned to the Musconetcong River after a 250-year absence. The City of Wilmington, DE removed the first dam on Brandywine Creek. The Brandywine River Restoration Trust has formed and is leading the effort to remove or mitigate the next 10 barriers on this river. While many fishways were added over the last 15 years or so, in most places they have not resulted in significant improvements in passage.

Historically, fish ladders have been the tool of first choice when dealing with connectivity issues; however, fish ladders should be a tool of last resort as dam removal is the best fish passage technology (ASMFC 2010). Technical fishways have been determined to be generally ineffective at passing American Shad (Haro & Santos 2012). While river herring may be successful using ladders in smaller streams, past efforts to restore populations of American Shad to the Lehigh and Schuylkill rivers via fish ladders have been largely unsuccessful. Across the basin many fish ladders are not maintained or monitored—even during the spawning run season—and those that did have passage, often passed nominal numbers of fish. The States of New Jersey and Delaware are both utilizing eDNA technology to determine fish ladder effectiveness on their dams.

Future Predictions

The enormous ecological impact of dams on the successful spawning and rearing habitat of migratory fish is increasingly recognized and highlighted by organizations and coalitions. In New Jersey there is now a Statewide Dam Removal Partnership (SDRP), which is a collaboration of nonprofits and government agencies that seeks to advance the removal of antiquated, dangerous, or ecologically detrimental dams. These types of partnerships can leverage expertise and accelerate the pace of dam removals. Climate change is increasing the frequency and severity of precipitation events (Blenkinsop et al. 2021) which can lead to dam failures or increased flooding due to a dam. Flooding concerns are already driving the effort to remove two dams in the Pequest River in New Jersey which will also significantly improve fish passage.

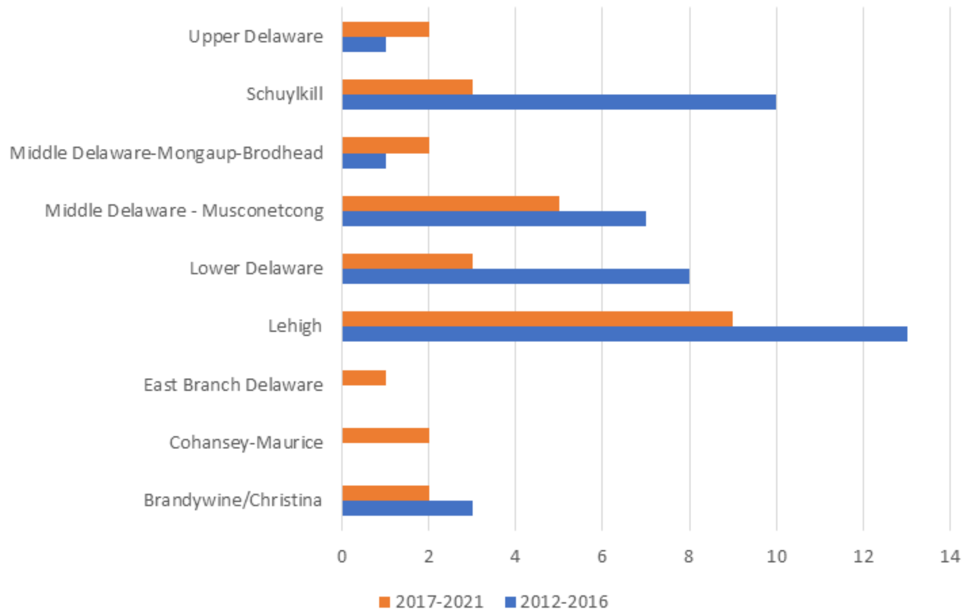


Figure 6.4.3 Total dams removed in the Delaware Estuary and Basin per HUC8 watershed between 2017-2021 (orange) and 2012-2016 (blue).

Actions and Needs

- While the restoration roadmap prioritizes dams, a basin wide assessment of road crossings and culverts to identify and prioritize fish passage impediments beyond dams is needed.
- Removing dams can be a long and complicated process; in the Delaware River basin there is a need for more experienced dam removal project managers within the conservation community. Increase opportunities for dam removal project manager training for conservation practices.
- Historic canals have proven a significant barrier to advancing dam removal on priority river systems. Although the original use for these canal systems is long gone, many of the canal towpaths have been turned into recreation trails along water-filled canals. Dams still provide the water to these canals and solutions to removing the dam while maintaining water in the canal have proven to be difficult to overcome. Innovative solutions to balance the needs of migratory fish passage and historic and recreational interests are needed.

Summary

Climate change is already impacting fish populations. Now and into the future it will be critical for our migratory fish to utilize the full extent of the mainstem as well as numerous tributaries of different size classes. The ability of these fish to utilize a variety of streams of different sizes may have greater reproductive potential to protect against negative impacts from environmental disturbances (Hillborn et al. 2003, Schindler et al. 2010). The Delaware River Restoration Roadmap identifies 45 priority barriers on 13 tributaries across the basin to focus shad and river herring restoration efforts and highlights key actions to significantly improve fish passage and restore habitat. Sustained federal funding through the Delaware Watershed Conservation Fund and new aquatic connectivity funding opportunities through the 2021 Infrastructure and Investment Act should provide the broader restoration community with much needed funding to restore spawning and rearing habitats for migratory fish.

References

ASMFC (Atlantic States Marine Fisheries Commission). 2010. Fish Passage Working Group Upstream Fish Passage Technologies for Managed Species. Retrieved from: https://www.asmfc.org/uploads/file/FishPassTechnologyForASMFCspecies_Oct2010.pdf

ASMFC (Atlantic States Marine Fisheries Commission). 2020 American Shad Benchmark Stock Assessment and Peer Review Report. Prepared by the ASMFC American Shad Stock Assessment Review Panel. Retrieved from: http://www.asmfc.org/uploads/file/5f43ca4eAmShadBenchmarkStockAssessment_PeerReviewReport_2020_web.pdf

Blenkinsop, S., Muniz Alves, L., and Smith, A.J.P., 2021: ScienceBrief Review: Climate change increases extreme rainfall and the chance of floods. In: Critical Issues in Climate Change Science, edited by: C. Le Quéré, P. Liss & P. Forster. https://doi.org/10.5281/zenodo.4779119

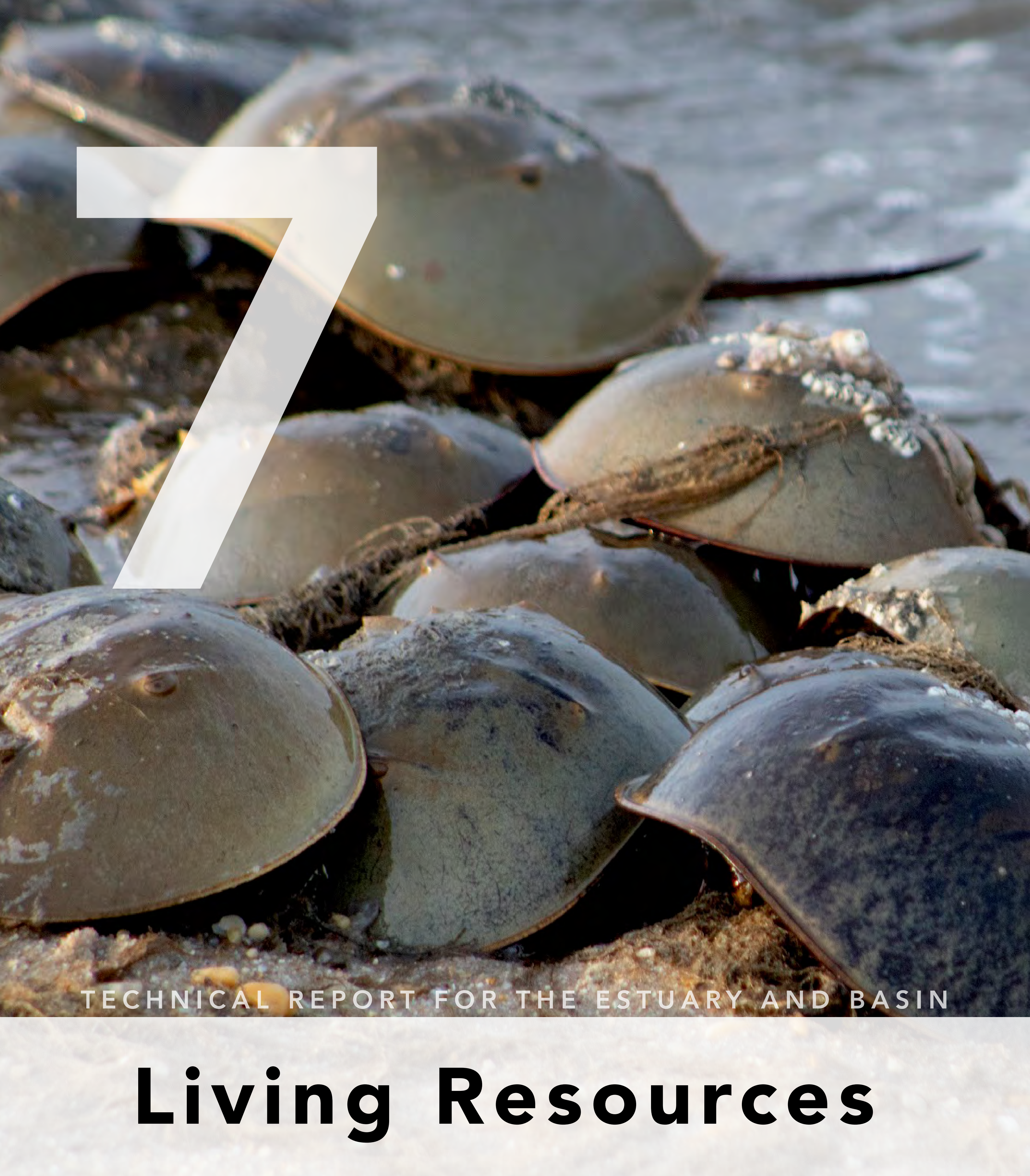
DeSalvo, L. L., M. DeLucia, E. H. Martin, and D. H. Keller. 2022. Delaware River Basin Restoration Roadmap for American Shad, Alewife, and Blueback Herring. The Nature Conservancy's final report to National Fish and Wildlife Foundation (Grant No.: 65162). The Nature Conservancy, Harrisburg, Pennsylvania

Haro A, Castro-Santos T. 2012. Passage of American Shad: paradigms and realities. Marine and Coastal Fisheries. 4:252–261

Hilborn, R., Quinn, T. P., Schindler, D. E., & Rogers, D. E. (2003). Biocomplexity and fisheries sustainability. Proceedings of the National Academy of Sciences of the United States of America, 100(11): 6564–6568.

Rivers, American (2017): American Rivers Dam Removal Database. Figshare. https://doi.org/10.6084/m9.figshare.5234068.v2 Retrieved: 03/03/2022

Schindler, D.E, R. Hilborn, B. Chasco, C.P. Boatwright, T.P. Quinn, L.A. Rogers and M.S. Webster. 2010. Population diversity and the portfolio effect in an exploited species. Nature 465: 609–612.



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TECHNICAL REPORT FOR THE ESTUARY AND BASIN

Living Resources

Partnership for the DELAWARE ESTUARY

December 2022 | Report No.22-05

Partnership for the Delaware Estuary— Host of the Delaware Estuary Program

TECHNICAL REPORT FOR THE ESTUARY AND BASIN

7 Living Resources

Contributors Jenny Shinn (ed), Sheila Eyler, Ronald Heun, Gregory Lech, Ella Rothermel, and Dan Millemann

Cover photograph by Kate Layton

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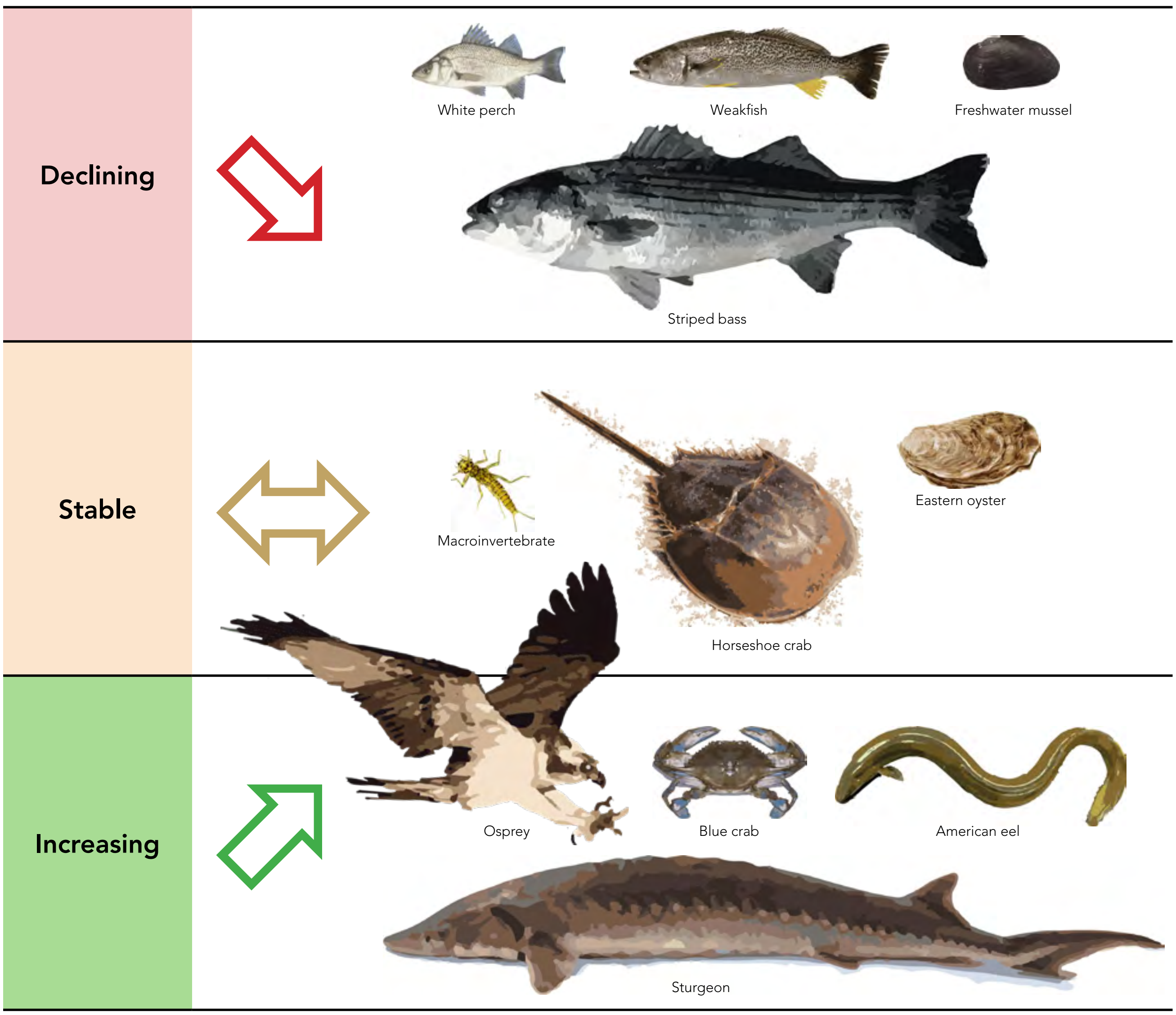
Abstract

The Living Resources Chapter examines eleven indicators across the Delaware River Estuary and Basin, spanning multiple taxa from osprey to macroinvertebrates. Experts that manage or study each living resource provide an introduction, description of the indicator, status, trends, future predictions, and conclude with actions and needs. Of the indicators discussed, 4 are showing signs of population decline (white perch, weakfish, striped bass, freshwater mussel), 3 are stable (macroinvertebrates, horseshoe crabs, oysters) and 4 are on the rise (osprey, blue crab, American eel, strugeon)(see graphic summary below). Feature stories within this chapter highlight the burgeoning shellfish aquaculture sector including a view on food justice, and new research characterizing long-term fish community trends in the Estuary. Importantly, this chapter only focuses on a few animal populations within the Delaware Estuary and Basin; more information on other monitored fisheries (i.e., stock assessments) can be found in reports by the Mid-Atlantic Fishery Management Council, Atlantic States Marine Fisheries Commission, as well as the Delaware River Basin Fish and Wildlife Management Cooperative (e.g., the American Shad Habitat Plan for the Delaware River).



7. Living Resources

7.1 Invertebrates

7.1.1 Blue Crab

The blue crab (*Callinectes sapidus)* is a member of the swimming crab family Portunidae and inhabits estuarine habitats throughout the western Atlantic, from Nova Scotia (although rare north of Cape Cod), along the Gulf of Mexico and Caribbean, to northern Argentina, and along western South America as far south as Ecuador (Williams 1979*)*.

Blue crab spawning occurs primarily in the summer months in mid to lower Delaware Bay with peak larval abundance occurring in August (Dittel and Epifanio 1982). Larvae are exported from the estuary into the coastal ocean where they undergo a 3-6 week, seven stage, zoeal development period in surface waters (Epifanio 1995; Nantunewicz et al. 2001). Models describe an initial southward transport of zoeae along the inner continental shelf within the buoyant estuarine plume after exiting the estuary (Epifanio 1995, Garvine et al. 1997). Northward transport back toward the estuary is provided by a wind-driven band of water flowing northward along the mid-shelf. Across-shelf transport into settlement sites in Delaware Bay is accomplished by coastal Ekman transport tied to discrete southward wind events (nor'easters) in the fall. These discrete wind events may have a large effect on larval recruitment and settlement success in the bay and strongly influence year class strength.

Females mate immediately after their pubertal molt into sexual maturity, usually late in their first year of life (late spring, summer). Sperm is stored over their remaining lifetime from this single mating event. Mated females can begin producing eggs in that summer and early fall over multiple clutches, continuing through to a second spawning season (Churchill 1921; Van Engle 1958; Darnell et al. 2009). Darnell et al. (2009) observed up to seven clutches for females in North Carolina. Prager et al. (1990) estimated fecundity per batch as over $3x10^6$ eggs.

Blue crabs hold an important ecological role as opportunistic benthic omnivores, with major food items including bivalves, fish, crustaceans, gastropods, annelids, nemertean worms, plant material, and detritus (Guillory et al. 2001). Post-settled blue crabs have been shown to have a key effect on infaunal community structure, particularly through major predation on bivalves such as the eastern oyster (*Crassostrea virginica*) (Eggleston 1990), hard clam (*Mercenaria mercenaria)* (Sponaugle and Lawton 1990), wedge clam (*Rangia cuneata)* (Darnell 1958), soft-shelled clam (*Mya arenaria)* (Blundon and Kennedy 1982; Smith and Hines 1991; Eggleston et al. 1992), and other bivalve species (Blundon and Kennedy 1982), and through indirect mortality on infaunal species from mechanical disturbance of sedimentary habitats caused by foraging (Virnstein 1977). Fish are primary predators on blue crabs, with more than 60 known fish predator species (Guillory et al. 2001).

Another very important source of predation on blue crabs occurs from cannibalism, as blue crabs make up as much as 13% of their diet (Darnell 1958). Cannibalism appears to increase with increasing crab predator size and is heaviest during the period of juvenile recruitment (Mansour 1992). Adult predation may be a key factor in density-dependent regulation of juveniles (Peery 1989).

Overfishing and stock sustainability in Delaware Bay became a serious concern in the mid 1990s, after a prolonged period of rising fishing effort and three-fold increase in landings from 1985-1995 (Fig 7.1.1.1). These fears peaked after bay-wide landings reached a record 12.7 million pounds in 1995 and then subsequently dropped by more than 46% in 1996.

Concern for the stock in 1998 prompted the 138th General Assembly of the State of Delaware to direct its Division of Fish and Wildlife to prepare a fishery management plan and quantitative assessment of the stock. Subsequent stock assessments revealed high fishing mortality rates in Delaware Bay in close proximity to the management threshold (fishing mortality F=1.3) suggesting that the stock was fully exploited (Helser and Kahn 1999; Wong 2010).

Description of Indicator

Perhaps the most-studied fishery species in Delaware, the blue crab has been very closely monitored since 1978 with monthly trawl surveys conducted by the Delaware Division of Fish and Wildlife (DDFW). Using biological information collected from these surveys, together with year-round collections of landings reports, the DDFW assesses the size and status of the Delaware Bay blue crab stock on an annual basis. This annual stock assessment is funded by the National Oceanic and Atmospheric Administration.

Present Status

Fisheries

The Delaware Bay blue crab stock supports a multi-million dollar, bi-state (NJ & DE) fishery. Delaware Bay landings and ex-vessel value in 2020 remained at historically high levels, reaching 9.3 million pounds and 14.1 million dollars combined in DE and NJ (Fig 7.1.1.1). Delaware Bay landings are generally split equally between the two States (51%:49%, DE:NJ). The current, five-year, average, baywide landings are the fourth highest on record since 1973, while the running, five-year, average ex-vessel value ($15.1M) is at the historical record high.

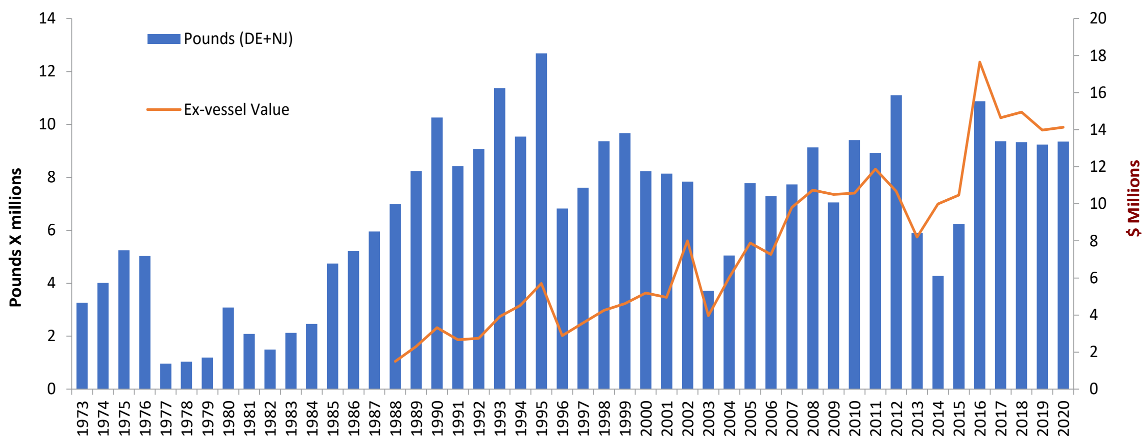


Figure 7.1.1.1 Total commercial and recreational harvest (lb) and commercial ex-vessel value in the States of Delaware and New Jersey.

In the State of Delaware, the blue crab (*Callinectes sapidus*) is by far the most valuable commercial fishery species. The ex-vessel value of blue crab landings in Delaware is worth more than three times the value of all other commercial fisheries combined.

The blue crab is also the most-numerous recreational fishery species harvested in Delaware Bay, exceeding two million crabs annually (Wong, unpublished). Recreational harvest accounts for about 3% and 15% of the total annual Delaware Bay landings in Delaware and New Jersey, respectively.

The pot fishery, by far, harvests the lion's share (86%) of Delaware's crab landings and value (Fig 7.1.1.2). Male crabs make up about 2/3 of the pot landings, in stark contrast to the female-dominated winter dredge fishery landings (Fig 7.1.1.3).

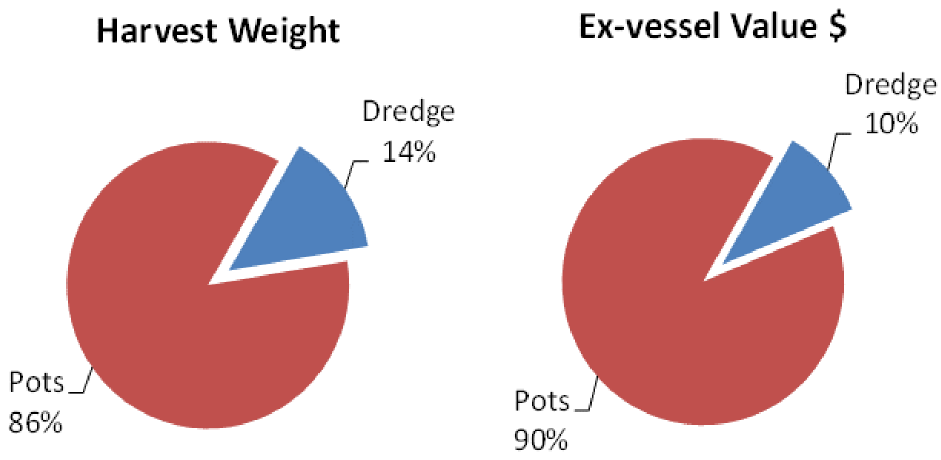


Figure 7.1.1.2 Total harvested weight and ex-vessel dollar value by gear type over the most-recent five years of Delaware landings data.

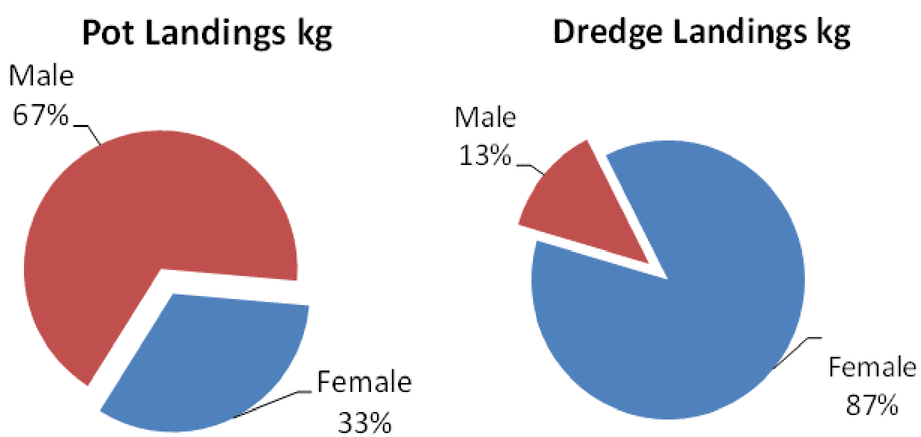


Figure 7.1.1.3 Sex-composition of pot and dredge fishery landings over the most-recent five years of Delaware landings data.

Stock Size and Status

The DDFW conducts an annual stock assessment of the Delaware Bay blue crab stock. Modeling work shows the stock to be at high levels of abundance and at relatively low levels of fishing mortality, and to be in excellent condition relative to historical metrics dating back to 1978 (Wong 2021). Juvenile recruitment has increased considerably (after bottoming in 2012-2013), reaching record highs in 2019, and persisting at robust, above-average levels for the past six years. Recent, sustained, high levels of recruitment (beginning in 2015), concomitant with a short period of low harvest (2013-2015), have now led to the highest levels of adult abundance since the population peak of the 1980s and 90s (Figs 7.1.1.5; 7.1.1.6). Consequently, fishing mortality rates have declined appreciably over recent years, at levels well below management thresholds. Terminal year model estimates of juvenile and adult abundances are at healthy levels. Total stock size stands at 203 million crabs, above the 43 year mean and median of 153 and 132 million (Fig 7.1.1.4) (Wong 2021). The near-term future outlook is promising, given the observed, elevated, stock productivity in recent years.

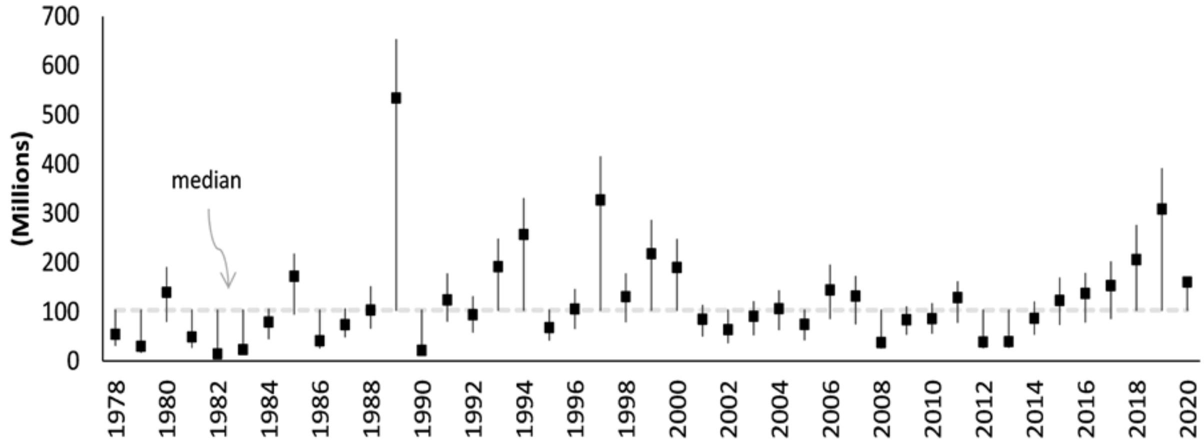


Figure 7.1.1.4 Absolute abundance (on Sep 1) of juvenile crabs with 95 percent confidence intervals estimated from the population assessment model (Wong 2021).

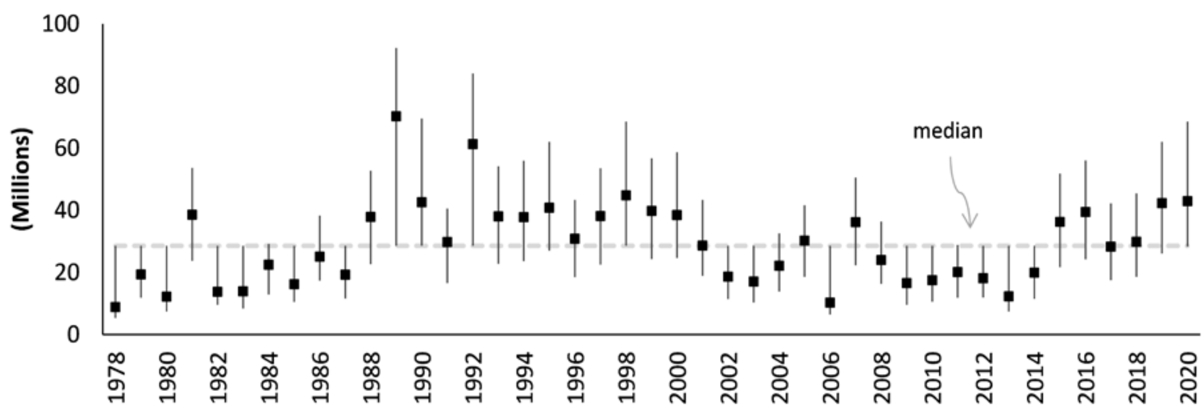


Figure 7.1.1.5 Absolute abundance (on Sep 1) of adult crabs fully recruited to the fishery with 95 percent confidence intervals estimated from the population assessment model (Wong 2021).

Past Trends

A period of high stock productivity occurred for about 15 years from 1985 to 1999 (Fig 7.1.1.6). During this period, DDFW crab indices were at or above median levels for 13 of 17 years. Weak year classes became commonplace for the next 15 year period from 2000 to 2014. Currently, the DDFW has observed consistently robust juvenile recruitment for six consecutive years (2015 to current).

Future Predictions

The near-term outlook for the fishery is promising given robust juvenile recruitment. Young-of-the-year (YOY) recruitment is typically a good predictor of future Delaware Bay landings (Fig 7.1.1.7). Furthermore, high stock productivity could continue to benefit from extended spawning seasons and increased egg clutches as temperatures remain elevated in Delaware Bay.

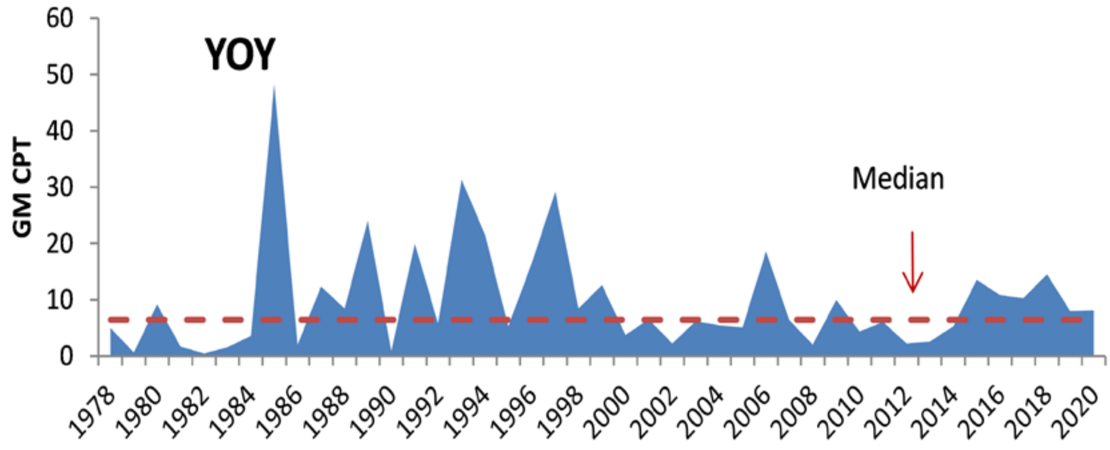


Figure 7.1.1.6 Young-of-the-year blue crab relative abundance from the DFW Delaware Bay trawl survey.

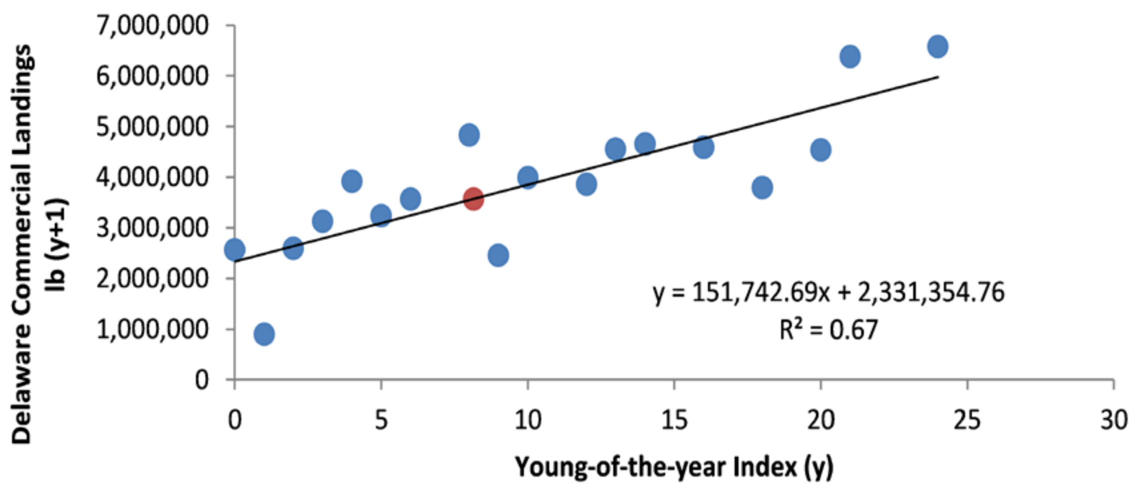


Figure 7.1.1.7 YOY abundance as a predictor of ensuing Delaware commercial landings (1978-2020). Red dot indicates predicted recruitment in 2021.

Actions and Needs

Continued close monitoring of stock abundance through monthly trawl surveys and accurate reporting of fishery landings are needed to protect, assess, and manage this important fishery stock.

Summary

The Delaware Bay blue crab stock is presently at historically high levels of adult abundance, and at low levels of fishing mortality (Wong 2021). Juvenile recruitment has risen sharply (after a prolonged, two-decade period of weak year classes) and has remained at above-average levels for six consecutive years, which bodes well for the stock and fishery in the near future.

References

Blundon, J.A. and V.S. Kennedy. 1982. Mechanical and behavioral aspects of blue crab, *Callinectes sapidus*, predation on Chesapeake Bay bivalves. J. Exper. Mar. Biol. and Ecol. 65:47-65.

Booth, K.J. and L.G. Martin. 1993. Striped bass feeding behavior and the potential effect on the blue crab population in the Chesapeake Bay. Maryland Department of Natural Resources, Tidewater Administration Fisheries Technical Memo No. 2 (Series), 7 pages.

Churchill, E.P. 1921. Life history of the blue crab. Bull. U.S Bureau of Fisheries for 1917-1918. 36:91-128.

Darnell, R.M. 1958. Food habits of fishes and larger invertebrates of Lake Pontchartrain, Louisiana, an estuarine community. Publication of the Institute of Marine Science, U. Texas 5:353-416.

Eggleston, D.B., R.N. Lipcius and A.H. Hines. 1992. Density-dependent predation by blue crabs upon infaunal clam species with contrasting distribution and abundance patterns. Mar. Ecol. Prog. Ser. 85:55-68.

Epifanio, C.B. 1995. Transport of blue crab (*Callinectes sapidus*) larvae in the waters off Mid-Atlantic states. Bull. Mar. Sci. 57:713-725.

Garvine R.W., Epifanio C.C., Epifanio C.E. and K.C. Wong.1997. Transport and recruitment of blue crab larvae: a model with advection and mortality. Estuar Coast Shelf Sci 45:99–111.

Guillory, V., (11 other authors). 2001. The blue crab fishery of the Gulf of Mexico, United States: A regional management plan. Gulf States Marine Fisheries Commission, P.O. Box 726, Ocean Springs, MS 39566.

Helser, T.E., and D.M. Kahn. 1999. Stock assessment of Delaware Bay blue crab (*Callinectes sapidus*) for 1999. Department of Natural Resources and Environmental Control, Delaware Division of Fish and Wildlife, 89 Kings Highway, Dover, DE 19901.

Mansour, R.A. 1992. Foraging ecology of the blue crab, *Callinectes sapidus*, in lower Chesapeake Bay, Ph.D. Dissertation. College of William and Mary, Williamsburg, VA.

Natunewicz, C., Epifanio, C.E., and R.W. Garvine. 2001. Transport of crab larvae patches in the open ocean. Mar. Ecol. Prog. Ser. 222:143-154.

Peery, C.A. 1989. Cannibalism experiments with the blue crab (*Callinectes sapidus* Rathbun): potential effects of size and abundance. Thesis (M.A.)--College of William and Mary. (Series) viii, 29 leaves, [2] leaves of plates. Virginia Institute of Marine Science.

Prager, M.H., J.R. McConaugha, C.M. Jones, and P.J. Geer. 1990. Fecundity of blue crab, *Callinectes sapidus*, in Chesapeake Bay: biological, statistical, and management considerations. Bull. Mar. Sci. 46:170-179.

Smith, L.D. and A.H. Hines. 1991. The effect of cheliped loss on blue crab *Callinectes sapidus* foraging rate on soft-shell clams *Mya arenareia*. J. Exp. Mar. Biol. and Ecol. 15(2):245-256.

Speir, Harley J. 2001. A review of predation on blue crabs in Chesapeake Bay. Maryland DNR. Fisheries Tech. Memo Series no.24.

Van Engle, W.A. 1958. The blue crab and its fishery in Chesapeake Bay. Part 1-Reproduction, early development, growth, and migration. Commercial Fisheries Review 20:6-17.

Virnstein, R.W. 1977. The importance of predation by crabs and fishes on benthic infauna in Chesapeake Bay. Ecology 58:1199-1217.

Wong, Richard. 2010. Stock assessment of the Delaware Bay blue crab (*Callinectes sapidus*). Department of Natural Resources and Environmental Control, Delaware Division of Fish and Wildlife, 89 Kings Highway, Dover, DE 19901.

Wong, Richard. 2021. Stock assessment summary of the Delaware Bay blue crab (*Callinectes sapidus*). Department of Natural Resources and Environmental Control, Delaware Division of Fish and Wildlife, 89 Kings Highway, Dover, DE 19901.

7.1.2 Horseshoe Crab

Horseshoe crabs (*Limulus polyphemus*) are benthic (or bottom-dwelling) marine arthropods that use both estuarine and continental shelf habitats. Although it is called a "crab," it is grouped in its own class (Merostomata), which is more closely related to the arachnids and scorpions than blue crabs and other crustaceans. Horseshoe crabs range from the Yucatan peninsula in Mexico to northern Maine in the US, with the largest population of spawning horseshoe crabs in the world found in the Delaware Bay.

Each spring, adult horseshoe crabs migrate from deep waters in bays and the ocean to spawn on intertidal estuarine sandy beaches (Shuster and Botton 1985). Beaches within estuaries, such as the Delaware Bay, are believed to be preferred because they are low energy environments protected from wind and waves, thus reducing the risks of stranding during spawning events. Spawning generally is initiated by increasing water temperatures and occurs from May through June in Delaware Bay, with the peak spawning activity occurring on the evening new and full moon high tides in mid to late May and early June (Smith et al. 2017).

Horseshoe crabs are characterized by high fecundity, high egg and larval mortality, and low adult mortality. Horseshoe crabs spawn multiple times per season associated with high tides (Chabot and Watson 2010). Adult females spawn typically on multiple days per season digging multiple nests during a spawning bout to deposit clusters of approximately 3,650 to 4,000 eggs in each nest (Fig 7.1.2.1), which is 10-20 cm below the beach surface (Brockmann and Penn 1992, Brousseau et al. 2004, Weber and Carter 2009, Smith et al. 2010, Beekley and Mattei 2015). An average size female in Delaware Bay, measuring 265 mm in prosomal width (i.e., the widest part of the animal), lays approximately 88,000 eggs annually (Shuster and Botton 1985). Adult males approach the spawning beaches more frequently than females to fertilize eggs either while attached in a mating position, known as amplexus, or gathered around a spawning female (Brockmann and Penn 1992). Egg development is dependent on temperature, moisture, and oxygen content of the nest environment (Vasquez et al. 2015) (Fig 7.1.2.2A). Eggs hatch between 14 and 30 days after fertilization (Fig 7.1.2.2B) (Botton et al. 1992).

Figure 7.1.2.1 (A) Horseshoe crabs (Limulus polyphemus) returning to the Delaware Bay at sunrise after spawning. (B) A horseshoe crab egg clutch in nest. Photo credit: Elizabeth Bouchard, Rutgers University

Juvenile horseshoe crabs generally spend their first and second summer on the intertidal flats, usually near the breeding beach where they hatched (Botton and Loveland 2003). As they mature, juvenile horseshoe crabs move into deeper water, eventually mixing with multiple cohorts and ages in areas offshore. Horseshoe crabs molt 16 to 17 times over 9 to 11 years to reach sexual maturity (Shuster and Sekiguchi 2003). Based on multiple lines of evidence from tagging studies, growth of epifaunal slipper shells (*Crepidula fornicata*), and demographic modeling, horseshoe crabs can live more than 20 years with tagged adults observed to remain at large up to 17 years indicating a maximum age of at least 27 years (ASMFC 2019).

Larvae and juveniles feed on benthic algae and animals, such as small polychaetes and nematodes (Carmichael et al. 2009). As horseshoe crabs mature, their diet shifts to a variety of benthic macrofauna, mainly blue mussel (Mytilus edulis) and surf clam (*Spisula solidissima*), but also razor clam (*Ensis* spp.), macoma clam (*Macoma* spp.), wedge clam (*Tellina* spp.), and fragile razor clam (*Siliqua costata*) (Botton and Haskins 1984, Botton and Ropes 1989).

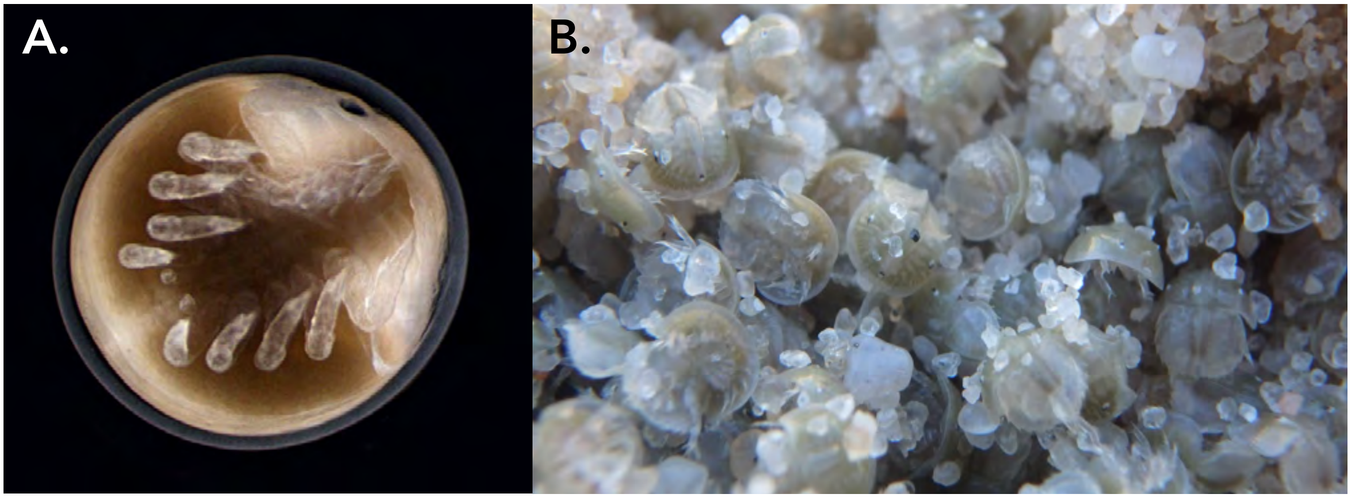


Figure 7.1.2.2 (A) A horseshoe crab embryo. Photo credit: Julia Van Etten, Rutgers University. (B) Newly hatched juvenile horseshoe crabs on the Delaware Bayshore. Photo credit: Elizabeth Bouchard, Rutgers University

Shorebirds feed on horseshoe crab eggs in areas of high spawning densities most notably within the Delaware Bay where horseshoe crab eggs are considered essential food for several shorebird species (Karpanty et al. 2006 and 2011, Botton et al. 2022). Delaware Bay is an important migratory staging area for shorebirds in North America. Horseshoe crabs place egg clusters at depths greater than 10 centimeters, which is deeper than most shorebirds can probe (Weber and Carter 2009, Karpanty et al. 2011). Shorebirds feed on the eggs that are brought to the surface by bioturbation due to nesting activity and wave action (Nordstrom et al. 2006).

Adult and juvenile horseshoe crabs make up a portion of the diet of the loggerhead sea turtle (Caretta caretta) in Delaware (Seney 2007). Horseshoe crab eggs and larvae and adults are also a seasonally preferred food item of a variety of invertebrates and finfish.

Historically, overharvest of horseshoe crabs has caused multiple cycles of population reduction and rebuilding (Kreamer and Michels 2009). Between the 1850s and the 1920s, it is estimated that over one million horseshoe crabs were harvested annually for fertilizer and livestock feed (Kreamer and Michels 2009). More recently horseshoe crabs have been taken in substantial numbers (e.g., over 5 million pounds in 1996 probably exceeding previous harvest periods) to provide bait primarily for the American eel (*Anguilla rostrata*) and whelk (*Busycon carica* and *Busycotypus canaliculatus*) fisheries (Botton et al. 2022). Since the early 2000s, harvest of horseshoe crabs for bait has been regulated and restricted by the Atlantic States Marine Fisheries Commission (ASMFC). Since 2013, sex-specific harvest in the Delaware Bay region has been managed under an adaptive management framework (ASMFC 2021 and 2022).

Horseshoe crabs are also collected by the biomedical industry to produce Limulus Amebocyte Lysate (LAL). This industry bleeds individuals and releases the animals live after the bleeding procedure. LAL is used world-wide to test medical products such as influenza serum, pacemakers, artificial joints, and other items to help ensure public safety from bacterial contamination (Levin et al. 2003). Mortality of horseshoe crabs associated with LAL production is estimated to be on average 15 percent and ranges from 4 to 30 percent depending on blood volume withdrawn and handling stress (ASMFC 2019). Synthetic alternatives to LAL have been developed based on recombinant technology (Botton et al. 2022). But the alternatives are waiting for product development and validation before adoption by commercial industry.

Description of Indicator

The peer-reviewed stock assessment conducted decennially since 1999 by the ASMFC represents the state of science indication of status for the horseshoe crab population in Delaware Bay. The ASMFC stock assessment integrates information from multiple surveys from the Delaware Bay region. The most recent stock assessment conducted in 2019 established the use of a catch multiple survey analysis (CMSA) to estimate abundance.

The CMSA integrates indices from the Delaware Adult Trawl, New Jersey Ocean Trawl, and Virginia Tech Trawl surveys to estimate abundance of primiparous (1st year of spawning) and multiparous (>1 year of spawning) males and females (Figs 7.1.2.3 and 7.1.2.4). Additional scientifically valid surveys, such as egg survey data (Smith et al. 2022), could be incorporated into the CMSA in future assessments. The use of CMSA was endorsed by two peer-review panels and adopted by the ASMFC management board for use in conservation decisions. Detailed descriptions of the surveys and CMSA methods, along with peer review comments, are available in ASMFC reports (ASMFC 2019 and 2021). The first peer-review panel was convened for the 2019 stock assessment. The second panel reviewed the revised adaptive resource management framework (ARM) in 2021.

The ARM has been used to inform ASMFC management of horseshoe crabs since 2013 (ASMFC 2021, McGowan et al. 2015; Fig 7.1.2.5). The objective of the ARM is to manage sustainable harvest of Delaware Bay horseshoe crabs while maintaining ecosystem integrity and supporting red knot (*Calidris canutus*) recovery by providing adequate stopover habitat for migrating shorebirds. Towards that end, the CMSA model was used to predict future horseshoe crab abundance resulting from harvest levels. And the CMSA model was combined with an integrated population model for predicting red knot abundance in response to horseshoe crab abundance and other factors (ASMFC 2021, Tucker et al. 2022).

Present Status

The peer-reviewed 2019 Horseshoe Crab Benchmark Stock Assessment evaluated the stock status by region, finding the Delaware Bay population to be stable (ASMFC 2019). The assessment concluded that overfishing and overfished status were unlikely for Delaware Bay horseshoe crabs because of low fishing mortality and relatively high recent abundances.

Past Trends

The horseshoe crab population in Delaware Bay declined in the early 1900s due to overharvest for fertilizer (Kreamer and Michels 2009). But the population increased from the mid-20th century through the 1980s after use for fertilizer stopped. Overharvest for bait caused the population to decline in the 1990s. Coordinated inter-state harvest regulation to address the overharvest started in 1999 (ASMFC 2019). The peer-reviewed 2019 Horseshoe Crab Benchmark Stock Assessment evaluated the stock status by region, finding the Delaware Bay population to be stable (ASMFC 2019).

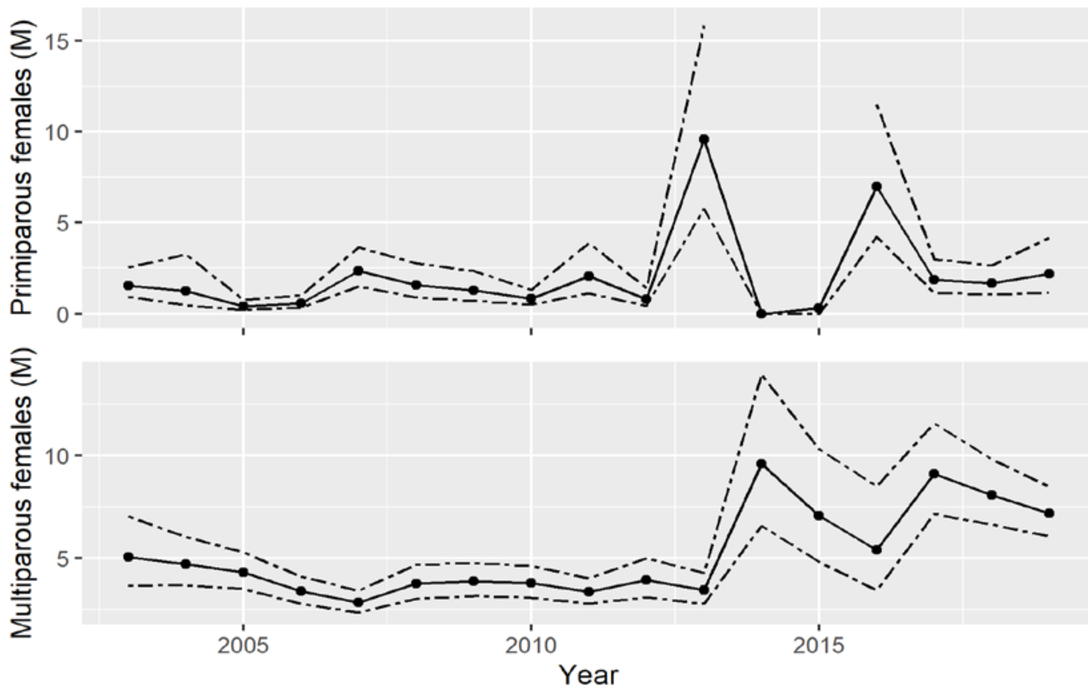


Figure 7.1.2.3 Catch multiple survey analysis (CMSA) estimated abundance (solid lines) of primiparous (first-spawning) and multiparous (previously spawned) females (millions, M) in the Delaware Bay population of horseshoe crabs (*Limulus polyphemus*) with 95% confidence limits (dashed lines) for 2003 to 2019 (ASMFC 2021 and 2022). The upper bound for primiparous females exceeded the scale in 2013-2015 and is not shown.

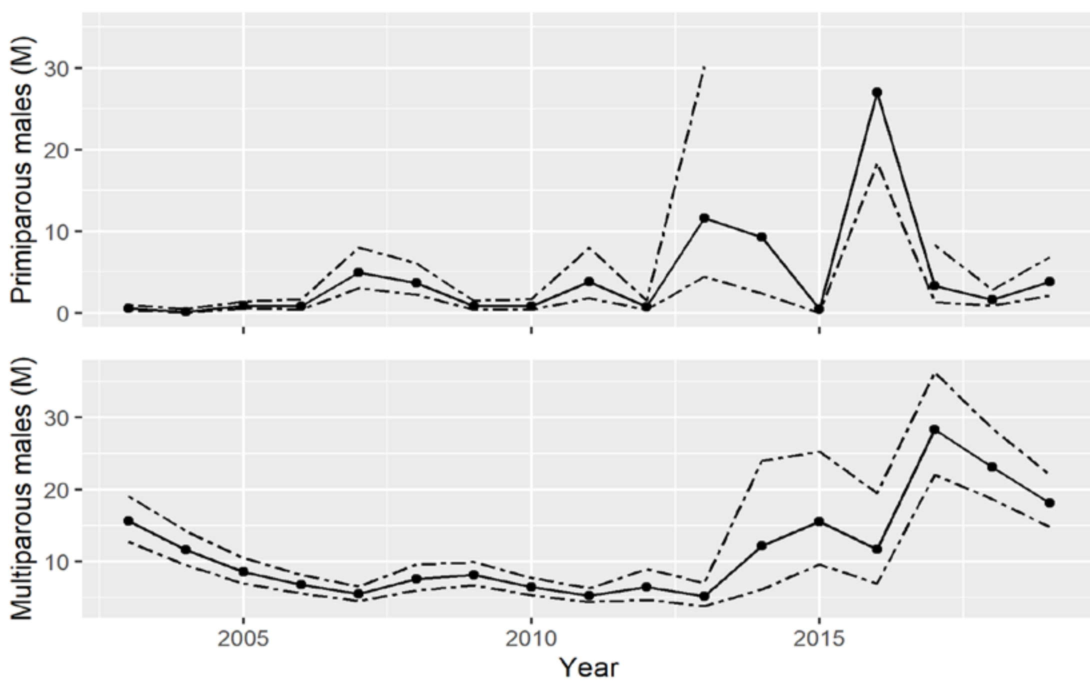


Figure 7.1.2.4 Catch multiple survey analysis (CMSA) estimated abundance (solid lines) of primiparous (first-spawning) and multiparous (previously spawned) males (millions, M) in the Delaware Bay population of horseshoe crabs (*Limulus polyphemus*) with 95% confidence limits (dashed lines) for 2003 to 2019 (ASMFC 2021 and 2022). The upper bound for primiparous females exceeded the scale in 2013-2015 and is not shown.

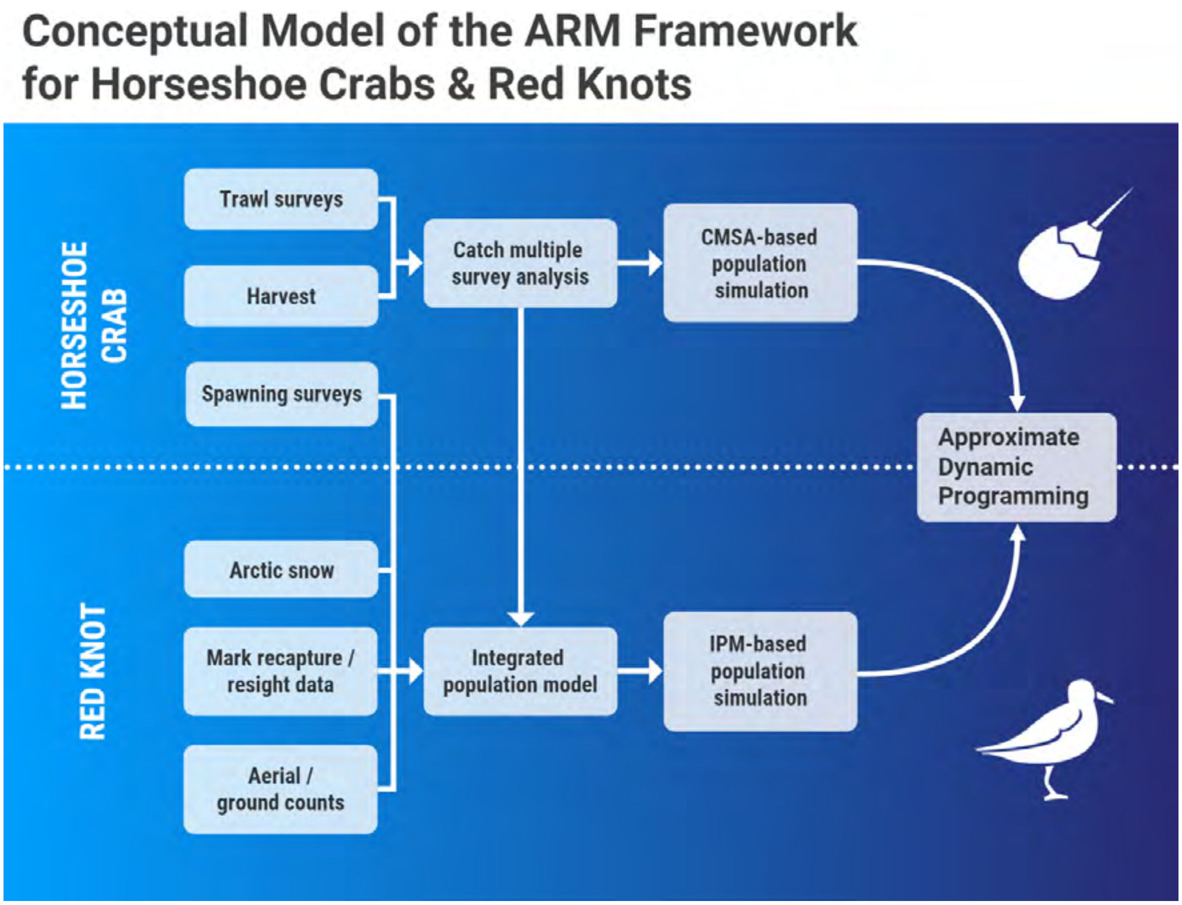


Figure 7.1.2.5 The adaptive resource management (ARM) framework used by the Atlantic States Marine Fisheries Commission to inform management of horseshoe crabs (*Limulus polyphemus*) in the Delaware Bay region (Sarah Murray, ASMFC).

Future Predictions

The ASMFC has implemented monitoring programs and restricted harvest of horseshoe crab with stated goals of maintaining a sustainable population for current and future generations of the fishing and non-fishing public, migrating shorebirds, and other dependent wildlife, including federally listed loggerhead sea turtles. The National Marine Fisheries Service has established a horseshoe crab sanctuary off the mouth of Delaware Bay, the Carl N. Shuster Sanctuary (Smith et al. 2016). Watermen have voluntarily implemented the use of bait bags that reduce their need for bait by preventing bait from being consumed by non-target species (Walls et al. 2002). The biomedical industry has voluntarily implemented management practices to reduce stress to animals being held for bleeding (Botton et al. 2022). These measures can be expected to allow the spawning population to increase over time by reducing harvest and indirect mortality (ASMFC 2019).

Because horseshoe crabs are long-lived and females do not reproduce typically until at least 10 years old, population rebuilding can take decades (Sweka et al. 2007). Whether the recent horseshoe crab population levels are sufficient to support migratory shorebirds has been debated in the scientific literature (Karpanty et al. 2016, Smith et al. 2022). However, the data indicate that the management actions to limit harvests, combined with voluntary reductions in bait use by commercial fishers, have allowed the population to increase in recent years compared to the early 2000s (ASMFC 2019 and 2021, Smith et al. 2022).

Actions and Needs

Under the ARM Framework, empirical models that describe the dependence and interaction of red knots and horseshoe crabs can be evaluated over time by monitoring the populations and updated with new data (McGowan et al. 2015; Fig 7.1.2.5). The monitoring programs that provide data for the CMSA model and the Shorebird Monitoring Program that result in mark-recapture estimates of red knot abundance are essential to implement this Framework (ASMFC 2021). Also, the Delaware Bay spawning survey is used to estimate the proportion of spawning that occurs during the red knot migration and is another important input to the ARM modeling (ASMFC 2021). Ensuring consistent funding to support these important monitoring programs will be critical to increase our understanding and reduce our uncertainty regarding how these two populations interact.

Summary

Management of horseshoe crab harvest coupled with voluntary measures by the bait and biomedical industries has allowed spawning populations of horseshoe crabs in Delaware Bay to increase over time compared to the early 2000s. Managing for horseshoe crab abundance near carrying capacity will be important to support surface eggs needed by current and future shorebirds during their stopover in Delaware Bay. However, shifts in carrying capacity related to climate change and coastal development could alter spawning densities relative to what were believed to be historical levels.

Since a portion of the red knot population that passes through Delaware Bay winters at the tip of South America and breeds in the high Arctic, other factors outside of Delaware Bay can, and probably are, affecting these populations. Work to better understand the dependence of red knots on Delaware Bay is being carried out, in part, through a cooperative Adaptive Management Framework.

References

Atlantic States Marine Fisheries Commission (ASMFC). 2019. 2019 Horseshoe Crab Benchmark Stock Assessment. Arlington, VA. 271 pp. (http://www.asmfc.org/uploads/file/5cd5d6f1HSCAssessment_PeerReviewReport_May2019.pdf)

Atlantic States Marine Fisheries Commission (ASMFC). 2021. Revision to the Framework for Adaptive Management of Horseshoe Crab Harvest in the Delaware Bay Inclusive of Red Knot Conservation and Peer Review Report. Arlington, VA. 302 pp. (To be posted at http://www.asmfc.org/species/horseshoe-crab under Stock Assessment Reports).

Atlantic States Marine Fisheries Commission (ASMFC). 2022. Overview of Horseshoe Crab ARM Revision and Peer Review. Arlington, VA. 237 pp. (http://www.asmfc.org/uploads/file/61f2f18aHSC_ARM_RevisionOverview_Jan2022.pdf)

Beekey MA, Mattei JH (2015) The mismanagement of Limulus polyphemus in Long Island Sound, U.S.A.: what are the characteristics of a population in decline. In: Carmichael RH, Botton ML, Shin PKS, Cheung SG (eds) Changing global perspectives on horseshoe crab biology, conservation and management. Springer, New York, pp 433–461

Botton ML, Haskins HH (1984) Distribution and feeding of the horseshoe crab, Limulus polyphemus, on the continental shelf off New Jersey. Fishery Bulletin 82:383–389

Botton ML, Loveland RE (2003) Abundance and dispersal potential of horseshoe crab Limulus polyphemus larvae in the Delaware estuary. Estuaries 26:1472–1479

Botton ML, Loveland RE, Jacobsen TR (1992) Overwintering by trilobite larvae of the horseshoe crab Limulus polyphemus on a sandy beach of Delaware Bay (New Jersey, USA). Marine Ecology Progress Series 88:289–292

Botton ML, Loveland RE, Munroe D, Bushek, Cooper JF (2022) Identifying the major threats to American horseshoe crab populations, with emphasis on Delaware Bay. In: Tanacredi JT, Botton ML, Shin PKS, Iwasaki Y, Cheung SG, Kwan KY, Mattei JH (eds) International horseshoe crab conservation and research efforts. Springer (doi.org/10.1007/978-3-030-82315-3)

Botton ML, Ropes JW (1989) Feeding ecology of horseshoe crabs on the continental shelf, New Jersey to North Carolina. Bulletin of Marine Science 45:637–647

Brockmann HJ, Penn D (1992) Male mating tactics in the horseshoe crab Limulus polyphemus. Animal Behaviour 44:653–665

Brousseau LJ, Sclafani M, Smith DR (2004) Acoustic-tracking and radio tracking of horseshoe crabs to assess spawning behavior and subtidal habitat use in Delaware Bay. North American Journal of Fisheries Management 24:1376–1384

Carmichael RH, Gaines E, Sheller Z, Tong A, Clapp A, Valiela I (2009) Diet composition of juvenile horseshoe crabs: implications for growth and survival of natural and cultured stocks. In: Tanacredi JT, Botton ML, Smith DR (eds) Biology and conservation of horseshoe crabs. Springer, New York, pp 521–534

Chabot CC, Watson WH III (2010) Circatidal rhythms of locomotion in the American horseshoe crab Limulus polyphemus: underlying mechanisms and cues that influence them. Current Zoology 56:499–517

Karpanty SM, Fraser JD, Berkson J, Niles LJ, Dey A, Smith EP (2006) Horseshoe crab eggs determine red knot distribution in Delaware Bay. Journal of Wildlife Management 70: 1704-1710.

Karpanty SM, Cohen J, Fraser JD, Berkson J (2011) Sufficiency of horseshoe crab eggs for red knots during spring migration stopover in Delaware Bay USA. Journal of Wildlife Management 75: 984-994.

Kreamer G, Michels S (2009) History of horseshoe crab harvest on Delaware Bay. In: Tanacredi JT, Botton ML, Smith DR (eds) Biology and conservation of horseshoe crabs. Springer, New York, pp 299–313

Levin J, Hochstein HD, Novitsky TJ (2003) Clotting cells and Limulus amebocyte lysate: an amazing analytical tool. In: Shuster CN Jr, Barlow RB, Brockmann HJ (eds) The American horseshoe crab. Harvard University Press, Cambridge, pp 310–340

McGowan CP, Smith DR, Nichols JD, Lyons JE, Sweka J, Kalasz K, Niles LJ, Wong R, Brust J, Davis M, Spear B (2015) Implementation of a framework for multi-species, multi-objective adaptive management in Delaware Bay. Biological Conservation 191: 759–769

Nordstrom KF, Jackson NL, Smith DR, Weber RG (2006) Transport of horseshoe crab eggs by waves and swash on estuarine beach: implications for foraging shorebirds. Estuarine Coastal and Shelf Science 70:438-448

Seney EE (2007) Musick JA (2007) Historical diet analysis of loggerhead sea turtles (Caretta caretta) in Virginia. Copeia 2:478–489

Shuster CN Jr, Botton ML, 1985. A contribution to the population biology of horseshoe crabs Limulus polyphemus (L.) in Delaware Bay. Estuaries 8: 363-372.

Shuster CN Jr, Sekiguchi K (2003) Growing up takes about ten years and eighteen stages. In: Shuster CN Jr, Barlow RB, Brockmann HJ (eds) The American horseshoe crab. Harvard University Press, Cambridge, pp 103–132

Smith, D. R., Brockmann, H. J., Beekey, M., King, T. L., Millard, M. J., and Zaldívar Rae, J. (2016). Conservation status of the American horseshoe crab (Limulus polyphemus): a regional assessment. Reviews in Fish Biology and Fisheries 27, 135–175. doi: 10.1007/s11160-016-9461-y

Smith DR, Brousseau LJ, Mandt MT, Millard MJ (2010) Age and sex specific migration timing and frequency of horseshoe crab spawning in Delaware Bay: insights from a largescale radio telemetry array. Current Zoology 56:563–574

Smith JAM, Dey A, Williams K, Diehl T, Feigin S, Niles LJ (2022) Horseshoe crab egg availability for shorebirds in Delaware Bay: dramatic reduction after unregulated horseshoe crab harvest and limited recovery after 20 years of management. Aquatic Conservation: Marine and Freshwater Ecosystems (doi.org/10.1002/aqc.3887)

Sweka JA, Smith DR, Millard MJ (2007) An age-structured population model for horseshoe crabs in the Delaware Bay area to assess harvest and egg availability for shorebirds. Estuaries Coasts 30:277–286

Tucker AM, McGowan CP, Lyons JE, DeRose-Wilson A, Clark NA (2022) Species-specific demographic and behavioral responses to food availability during migratory stopover. Population Ecology 64(1): 19-34

Vasquez MC, Murillo A, Brockmann HJ, Julian D (2015) Multiple-stressor interactions influence embryo development rate in the American horseshoe crab, Limulus polyphemus. J Exp Biol 218:2355–2364

Walls EA, Berkson J, Smith SA (2002) The horseshoe crab, Limulus polyphemus: 200 million years of existence, 100 years of study. Reviews in Fisheries Science 10(1): 39-73

Weber RG, Carter DB (2009) Distribution and development of Limulus egg clusters on intertidal beaches in Delaware Bay. In: Tanacredi JT, Botton ML, Smith DR (eds) Biology and conservation of horseshoe crabs. Springer, New York, pp 249–266

7.1.3 Eastern Oyster

Oysters are a dominant structural and functional member of the Delaware Bay benthos. The species native to Delaware Bay is *Crassostrea virginica* (Gmelin, 1791), commonly called the eastern or American oyster. Eastern oysters are reef builders that provide hard substrate and create structural complexity in an environment otherwise dominated by sand and mud. This species occurs from Nova Scotia to Florida, throughout much of the Gulf of Mexico and south to Brazil. In some areas like South Carolina and Georgia, it can form extensive intertidal reefs but in Delaware Bay it is predominantly subtidal where it is protected from freezing and ice scour. In addition to providing habitat for many other species, oysters filter large quantities of water that enhance nutrient cycling within the system. Oysters have been harvested from Delaware Bay since pre-colonial times, and current harvests are carefully managed to support a sustainable fishery. Oysters have also been cultivated in Delaware Bay for more than a century in both intertidal and subtidal habitats of the lower Delaware Bay.

Oysters occur throughout Delaware Bay from Artificial Island to the mouth of the bay and extend up into tributaries until salinity falls below tolerable average levels of about 5 ppt. Some oysters live intertidally, often on or within ribbed mussels along creek banks or attached to other hard substrates, natural or otherwise, within the lower intertidal zone. Nevertheless, the vast majority of the oyster population exists subtidally on reefs or beds that occur in the upper portion of the Bay above Egg Island Point on the New Jersey side and Port Mahon on the Delaware side upbay to Artificial Island. About 90% of the oysters in this region occur on the New Jersey side of the Bay.

Oysters may begin spawning in Delaware Bay as early as May or as late as September, but most spawns take place in July and August. Females can release all their eggs at once or partially spawn multiple times, but an average mature female may produce 2 to 60 million eggs during a single spawn. Typical spawns in a hatchery yield 1 to 15 million eggs. The fertilized eggs produce free swimming larvae within 24 hours that remain in the water column for two to three weeks before attaching to whatever hard substrate they can find, preferably clean oyster shell. During this process known as "setting" or "settlement", the settling larvae glues its left valve to the hard substrate then undergoes a metamorphosis, losing its ability to swim and taking on the morphology of a juvenile. Subsequent growth rate depends on the temperature, salinity and food availability of the site where the oyster attaches and varies both seasonally and annually. By fall the Young-of-Year (YOY) oysters can range in size from a few millimeters to 40 or 50 mm with an average of around 25 mm. Little or no growth takes place during the winter, and young oysters are heavily preyed upon by oyster drills, flatworms, small crabs and other predators. By the next fall most surviving oysters reach 30 to 65 mm depending on the location within the salinity gradient. Lower salinity areas have slower growth, but there are fewer predators so survival is better. Average growth to market size (3 inches = 76 mm) typically takes from 3 to 6 years in Delaware Bay, again depending on the location along the salinity gradient.

The oyster and the oyster reef assemblage are important to the general ecology of the bay. The assemblage of organisms that develop on an oyster reef was recognized in the late 1800s as a community and described as a biocoenose by Möbius. This concept was the forerunner of what we now know as community ecology. In addition to the structure that oysters provide, they are also a major functional part of the ecosystem because oysters filter water for food. This filtration process removes particulate material from the water column and deposits it on the sediment surface where some of it becomes food for other organisms or is broken down by bacteria. This filtration and deposition is an important pathway for nutrient cycling in estuaries. In some estuaries, oyster filtration can clarify water enough to increase light penetration and facilitate growth of sea grasses but Delaware Bay is so turbid that this facilitation does not occur.

Two oyster diseases are present in Delaware Bay. MSX is caused by *Haplosporidium nelsoni*, and dermo or Perkinsosis is caused by *Perkinsus marinus*. Both pathogens are protozoans and neither affects humans, but they are eventually lethal to oysters. There is clear evidence that the native oyster population has developed a relatively high level of resistance to MSX (Ford and Bushek 2012), but resistance to dermo has not developed to any major extent (Bushek et al. 2012). Since 1989 dermo has been a major factor controlling oyster population levels on the higher salinity oyster beds in Delaware Bay from Ship John Light south.

Description of Indicator

Given the role oysters play as a keystone species, the size and condition of the oyster population is an excellent indicator of overall ecosystem health in the Delaware estuary. Luckily, the commercially harvestable oyster beds of the New Jersey portion of Delaware Bay have been surveyed in the fall and winter since 1953 (Fegley et al. 2003). In the earlier years, the survey took place from September through the winter, but since 1989 the survey time frame has been reduced to about four, non-consecutive days between October and November. To create a survey domain, each of the beds or reefs were divided into 0.2-min latitude x 0.2-minute longitude grids (~25 acres or 10,171 m^2) and the grids were assigned to one of three strata, high, medium, and low quality (Fig 7.1.3.1). Strata were delineated by relative density. For each surveyed bed, high quality strata represent high density areas containing ~50% of the population, medium density areas contain ~48% of the population, and low density areas contain just 2% of the population. A random selection of these grids on the high and medium quality strata are sampled each year using a commercial oyster dredge. The number of grids sampled in each strata is determined by optimally allocating survey effort to minimize overall survey error. Low quality grids are not sampled and the abundance of oysters on those grids, about 2% of the population, are never used in setting the quota for annual harvest. This annual survey is also supplemented by regular monitoring of disease, mortality and harvesting at weekly to monthly intervals, providing a comprehensive picture of the status of the population and the fishery it supports. Each year the Haskin Shellfish Research Laboratory (HSRL) convenes a Stock Assessment Workshop (SAW) where population status and trends are presented. At the SAW, a Stock Assessment Review Committee (SARC) is asked to make recommendations for sustainable levels of harvest and for ways to improve assessment or management of the population. Details presented at these workshops are published in annual stock assessment reports available at http://hsrl.rutgers.edu/SAWreports/index.htm.

Past Trends

There were substantial oyster harvests from Delaware Bay in the middle 1800's, and by the latter part of that century extensive importation of seed onto leased bottom in the lower Delaware Bay enhanced the numbers of market oysters over what the Bay alone could produce. Active survey of the seed bed resource did not take place until 1953, and annual records are available since that date (Fig 7.1.3.2). The survey was initiated during a period of low abundance and just a few years before the oyster disease MSX substantially reduced the total numbers of oysters in the bay. The following decade was a period of low abundance, but it was followed, from the late 1960's until the mid 1980's, by a period of high abundance. This was terminated by another MSX epizootic in 1985, and the emergence of dermo in 1989 which has dominated the population dynamics across the oyster beds ever since. In the late 1950's the natural oyster bed oyster population averaged about 2.8 billion adult individuals. In the peak years of the 1970's to the mid 1980's the average oyster population was tenfold higher at 17 billion individuals during a period when disease pressure was virtually non-existent.

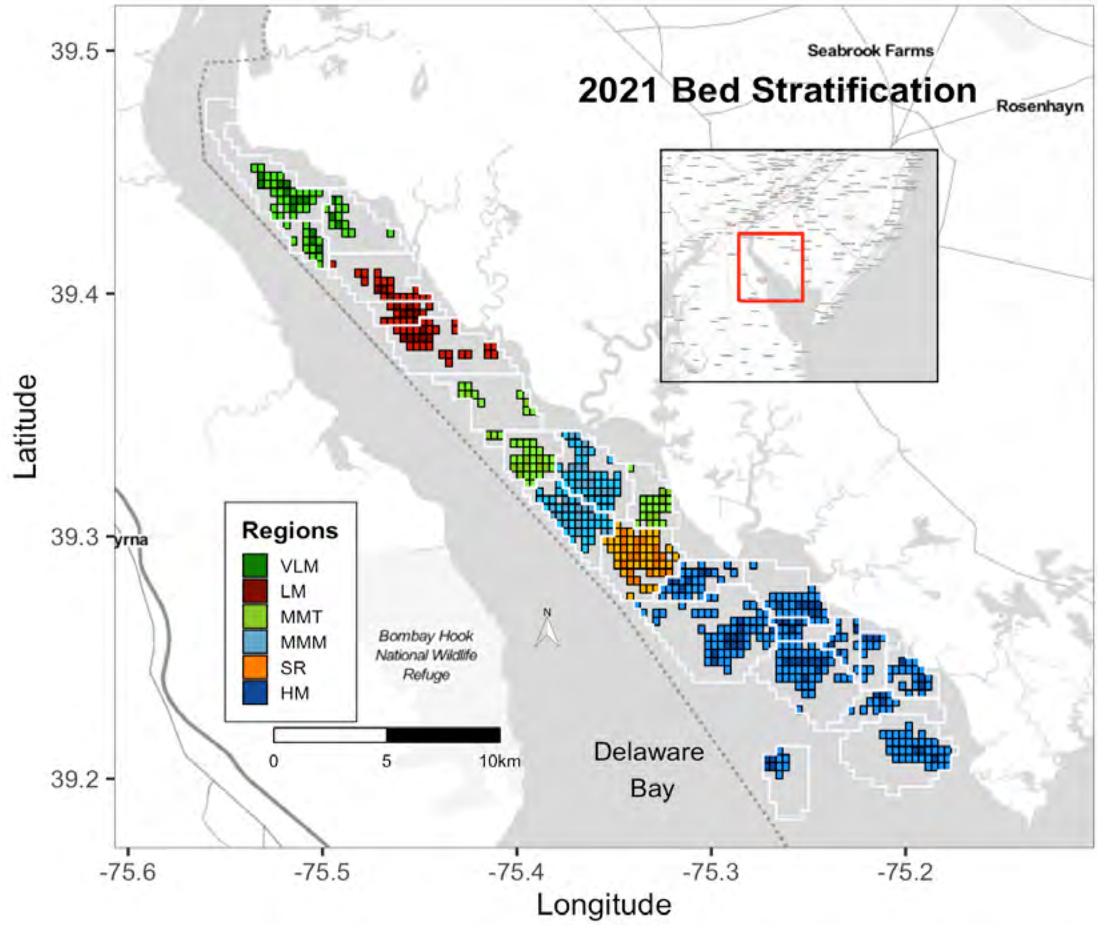


Figure 7.1.3.1 The assessed oyster beds of Delaware Bay, NJ colored by region (see Legend) with the 2021 strata designations. White outlines indicate the complete boundary of each bed with the high and medium quality strata grids in dark and light colors, respectively; black outlines indicate beds that were resurveyed in 2021. Strata designations are calculated within-bed not within-region. Gray areas in each bed indicate low quality strata. Annual assessments include samples from each bed's high and medium quality strata only. Each grid is 0.2" latitude x 0.2" longitude, approximately 25 acres (10.1 hectares).

Present Status

In 2006, the SARC established target and threshold abundance reference points based on the 1990-2005 time series. It was concluded that this time period represented the scope of oyster population dynamics in the present climate and disease regime in the Delaware Estuary. A target was calculated as the median of total and market-size (>63.5mm) oyster abundance and a threshold was calculated as ½ the target. These values have since provided a reference level against which the current population size could be compared. In addition, the 2006 SARC developed fishing exploitation reference points that constrained the annual harvest to be within a range of fixed exploitation. Since these two management tools were put into place, the exploitation rates of all oysters and market-size oysters have rarely gone above 2% and 4% respectively (Fig 7.1.3.3), the abundance of market-size oyster has never been below the target (Fig 7.1.3.4B), and the quota has typically fallen between ~80,000 and ~120,000 bushels (Fig 7.1.3.5). In effect, these management measures stabilized both the population and the fishery.

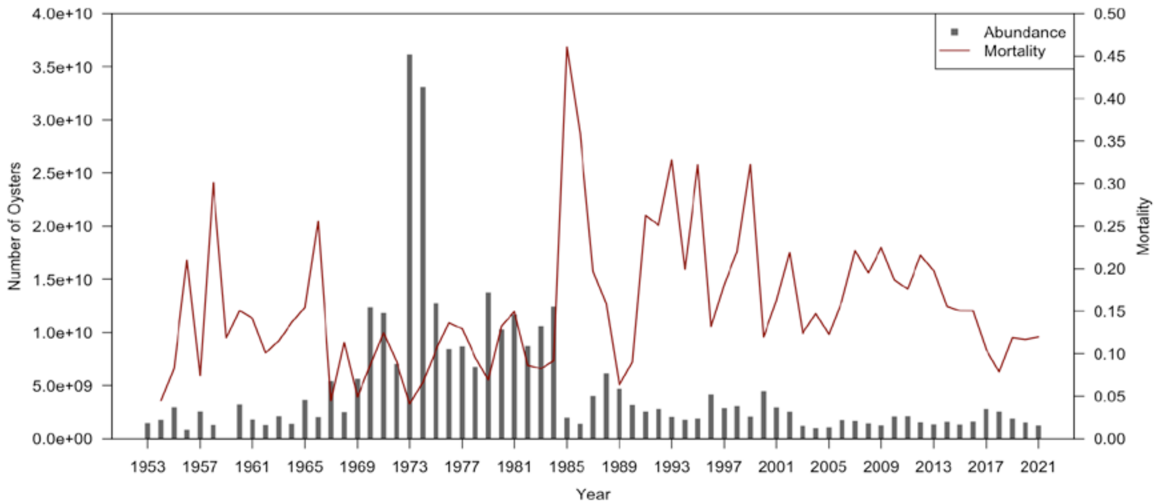


Figure 7.1.3.2 Time series of total oyster abundance (left axis) and natural mortality rate.

In 2021, a total of 238 grids scattered across the high and medium quality strata of all twenty-six beds were sampled to estimate the status of the stock. Figure 7.1.3.4 details the recent (1990-present) times series for several key population metrics. A large recruitment event in 2016 and 2017 (Fig 7.1.3.4C) led to a temporary spike in sub-market abundance (Fig 7.1.3.4B) and total abundance (Fig 7.1.3.4A). The total abundance has since returned to being between 1 and 2 billion individuals the last few years where it has typically fallen during the last ~20 years. Despite large fluctuations in sub-market and total abundance, market-size abundance has remained relatively stable and is again well above the target in 2021 (Fig 7.1.3.4B). In addition, natural mortality continues to be low in recent years relative to the last several decades (Fig 7.1.3.4D).

Future Predictions

Since the intensity of oyster disease and recruitment cannot be predicted, management decisions can only influence how heavily the population is exploited and how much habitat is added to promote recruitment. There is no evidence that harvest has had substantial effects on the population dynamics of oysters in Delaware Bay since at least the late 1960's. Therefore, there is no good reason to reduce exploitation any further. While current recruitment levels indicate the stock is not recruitment limited, substrate, or available habitat, may be limited. This suggests that until the amount of habitat increases, likely via persistent, large-scale shell planting, the population will remain stable at the current level. Shell planting is the addition of clean, recycled shell to the bay bottom to provide attachment surfaces for oysters. Presently, the oyster industry taxes itself at a rate that ensures it replaces what shell it harvests. Profit margins are such that increased taxes for shell planting are not likely to be a viable mechanism for increasing shell planting efforts. Ideally, shell planting would be on the order of half a million to a million bushels of shell each year. Current efforts are between 100,000 and 200,000 bushels. If external funding sources could be secured to plant additional shell and increase available habitat for spat to settle and grow on, perhaps the population could grow beyond what the current environment and habitat appear to support.

As long as oyster population dynamics in higher salinity areas are controlled by dermo and MSX, changes in the oyster population will be linked to salinity. The funnel shaped geomorphology of Delaware Bay makes the area available for development of oyster reefs decrease from the mouth of the bay toward the fall line. This unique geomorphology in combination with ongoing sea-level rise suggests that the area available for prime oyster habitat will decline in the future. Other factors such as channel deepening, extraction of groundwater, and consumptive use of Delaware River freshwater supplies all imply that salinity will rise even if climate change causes increased rainfall. In 2011, however, excessive rainfall from Tropical Storms Lee and Irene depressed salinity throughout the bay for several weeks causing up to 75% mortality on the uppermost beds (Munroe et al. 2013). Those beds recovered rapidly with higher than anticipated recruitment. However, in 2018 and 2019 extended periods of heavy rainfall led to depressed salinity values again and mortality was observed near 50%. The likelihood of continued freshets in the upper bay region with similar impacts is expected to increase with climate change.

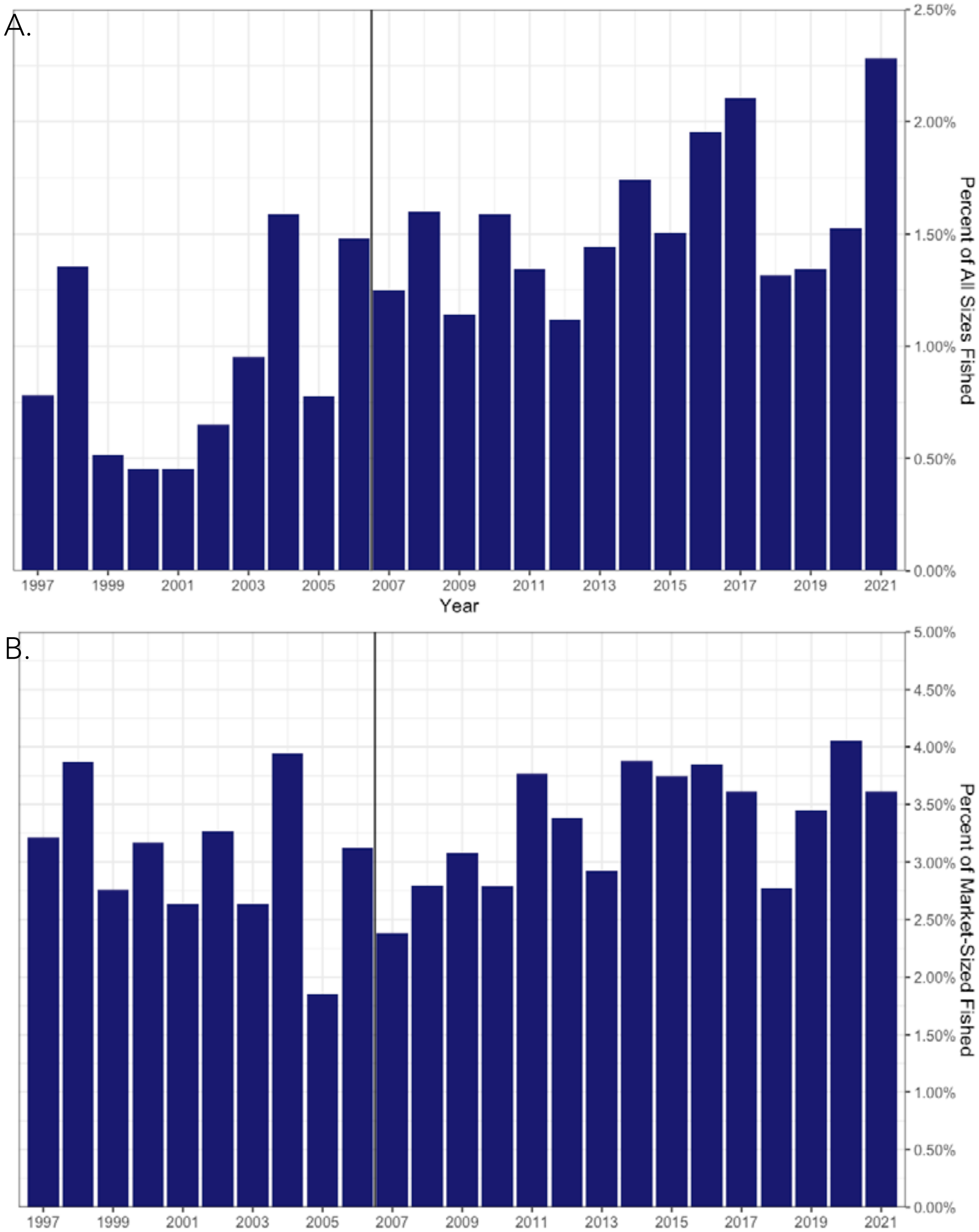


Figure 7.1.3.3 Fishing mortality as a percentage of (A) total oyster abundance and (B) the market-sized oyster abundance (≥2.5”). Reference points began in 2007 (vertical line).

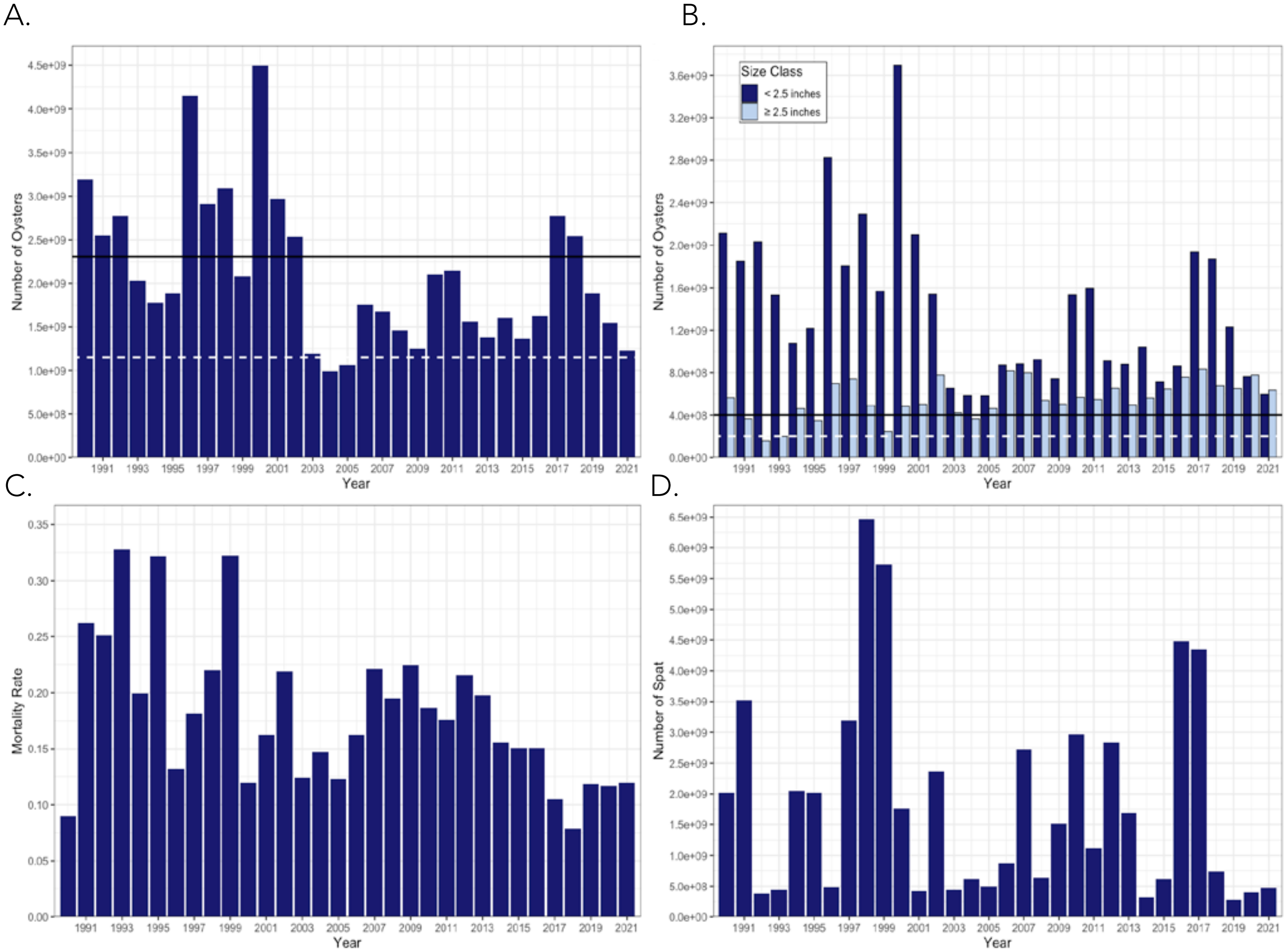


Figure 7.1.3.4 Long-term time series summary for the population. Top panels (A, B): total abundance (≥ 20 mm) and size class abundances (≥ 20 mm). Bottom panels (C, D): mortality rate and spat abundance (< 20 mm). Dashed horizontal lines represent the threshold and solid horizontal lines represent the target for abundance in panel A and for market abundance in B.

Actions and Needs

Maintaining the annual oyster population and oyster disease surveys is essential to sustainable management of this resource. In addition to continuing these survey efforts that support the annual stock assessment, actions could be taken to manage the population more actively. First, some attention should be devoted to evaluating the dynamics of the Hope Creek, Fishing Creek, and Liston Range beds to create a management plan for that region of Delaware Bay that accounts for the unique environmental conditions and population demographics in that region of Delaware Bay. Second, plans should be developed to manage the likely continued rise in salinity in Delaware Bay given its importance to the long-term viability of key oyster beds. For instance, exploration of the application of a salinity mitigation bank might be warranted. Third, development of a bay-wide environmental monitoring system for temperature and salinity should be implemented. Such a system would assist with interpreting changes in population demographics as the environment changes. Last, and likely most importantly, plans should be developed for enhancing habitat and recruitment through shell planting beyond the current efforts that are funded by the industry bushel tax.

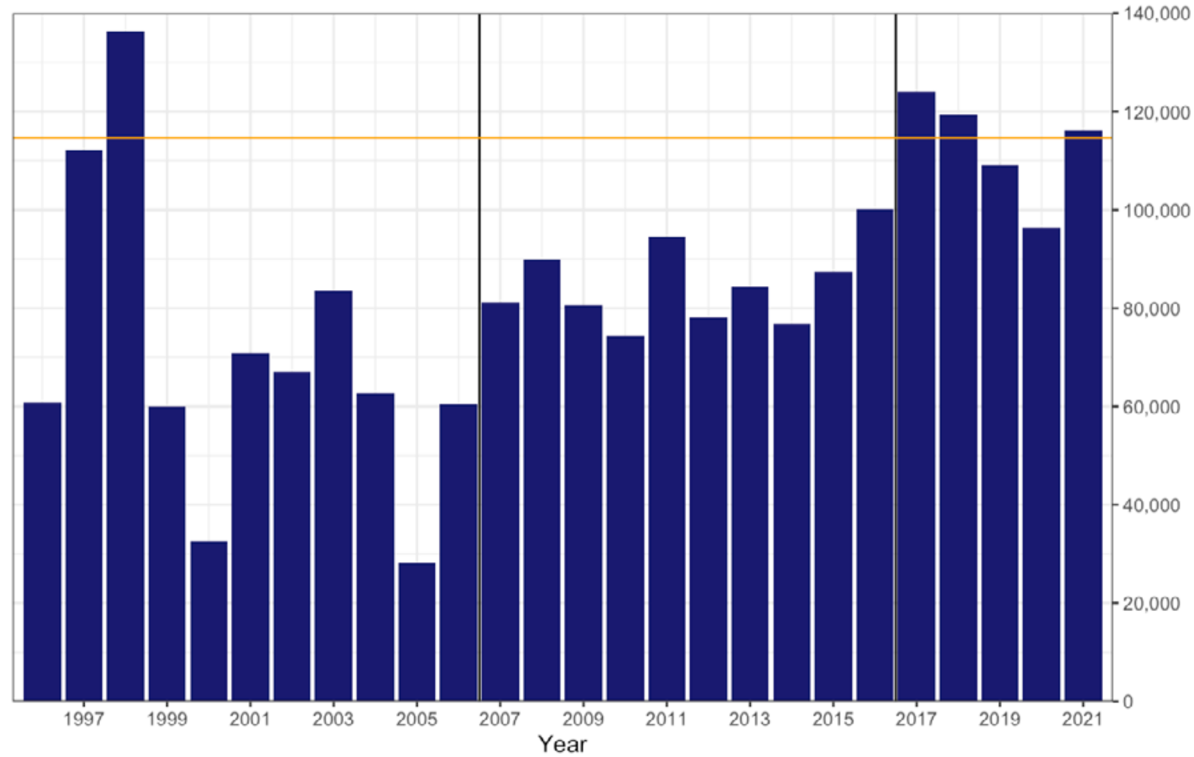


Figure 7.1.3.5 Number of bushels harvested from the natural oyster beds of Delaware Bay since the inception of the direct-market program in 1996. The 25-year average harvest is 84,130 bushels. The vertical line shows when reference points were instituted in 2007. The achieved quota for 2021 was 116,194 bushels after transplant (orange line).

Oyster Aquaculture

Oyster aquaculture continues to expand in the Delaware Bay with new developments in breeding for disease resistance and growth as well as technological advances in cultivation systems (see Living Resources Feature 1 - Oyster Aquaculture in the Delaware Bay). Policies and regulations are being developed to guide this growth in a sustainable manner. Little growth in aquaculture is presently occurring in the intertidal areas of the southern portion of the Bay due to concerns about possible conflicts with the federally listed threatened species, *Calidris canutus rufa*, commonly known as red knots. An adaptive management system has been employed to help minimize potential negative impacts of oyster farming on the species. Meanwhile, advances in gear technology have led to the growth of oyster aquaculture in deeper waters in the mainstem of the Bay away from intertidal areas.

Summary

The oyster is a keystone species that helps sustain a diverse community assemblage through habitat provisioning, contributes to key geochemical processes like nutrient cycling in the estuary, and provides socioeconomic benefits to the communities that surround the Delaware estuary by providing food and supporting jobs and fishing infrastructure. The dynamics of the oyster population in Delaware Bay are controlled by a balance between recruitment and growth, and disease and fishing related mortality. Recruitment, growth, and disease respond to environmental changes such as the annual temperature cycling and salinity (freshwater input) and thus cannot be predicted or controlled. However, fishing mortality, and to some extent recruitment, can be controlled by limiting exploitation rate and planting clean shell to promote recruitment respectively. Fishing exploitation has been fixed at a low, sustainable level for some time and this has led to a stable population of adult oysters. If the population is to grow, the capacity to support additional oysters will likely need to come from larger scale shell planting efforts from funding sources outside of the fishing industry.

References

Bushek, D., S.E. Ford and I. Burt. 2012. Long-term patterns of an estuarine pathogen along a salinity gradient. J. Marine Research, 70(2-3):225-251.

Fegley, S.R., S.E. Ford, J.N. Kraeuter and H.H. Haskin. 2003. The persistence of New Jersey oyster seedbeds in the presence of MSX disease and harvest: The role of management. Journal of Shellfish Research, 22:451-464

Ford, S.E. and D. Bushek. 2012. Development of resistance to an introduced marine pathogen by a native host. J. Marine Research, 70(2-3):205-223.

Morson, J. M., D. Bushek, and J. Gius. 2022. Stock Assessment Workshop – New Jersey Delaware Bay Oyster Beds (24th SAW) Final Report. Haskin Shellfish Research Lab., Port Norris, NJ. 87pp.

Munroe, D., A. Tabatabai, I. Burt, D. Bushek, E.N. Powell, and J. Wilkin. 2013. Oyster Mortality and Disease in Delaware Bay: Impact and Recovery Following Hurricane Irene and Tropical Storm Lee. Estuarine, Coastal and Shelf Science, 135:209-219.

Oyster Aquaculture in the Delaware Bay

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Shellfish aquaculture has become an important and growing sector in New Jersey's coastal economy. Extensive oyster farming in Delaware Bay has historical significance. Traditional oyster farming in the Delaware involved both moving submarket oysters from oyster bars to down bay shellfish leases where more favorable oyster growth occurred and "planting" shell on leased bottom to capture recruitment of wild oysters. At the Delaware Bay oyster fishery's peak thousands of acres of shellfish leases were managed in this way, essentially operating as non-structural oyster farms. Oystermen worked their leases independently. With the emergence of MSX disease in the late 1950s, the lease planting strategies were nearly abandoned as the practice was no longer profitable in an environment with significant loss to disease. Today, traditional bottom planting still occurs, albeit at an effort lower than historical levels.

Contemporary structural oyster aquaculture began in the southern portion of the Delaware Bay shoreline on the Cape May Peninsula in the 1990s following advances in genetic breeding that yielded disease resistant oyster stocks that could survive in the presence of the endemic oyster disease, MSX, that decimated wild oyster populations. This shoreline is distinct in that it features narrow beaches, small creeks, and extensive intertidal sand flats that are exposed at low tide. The intertidal flats extend 1000-1700 feet offshore and are characterized by a series of sand bars (ridges) and sloughs (runnels) that run parallel to the beach. Such conditions are ideal for rack and bag cultivation systems in which oysters are grown in stiff mesh bags that are secured to rebar racks that maintain the oysters about 10-15 inches off bottom (Fig 1). The farms are accessed by land and tended during low tide when the oysters are exposed for a period typically 2-6 hours long.

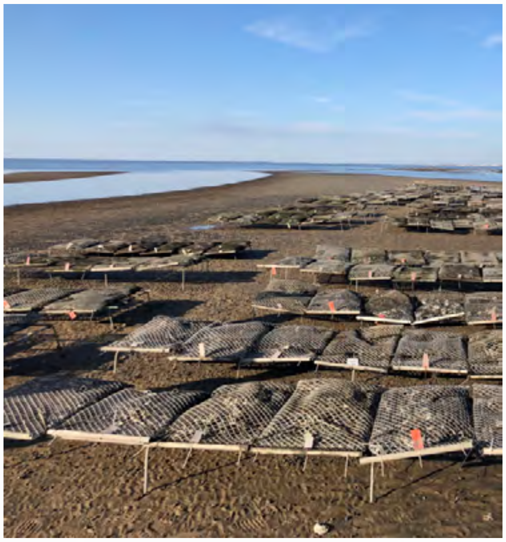

Figure 1. Rack and bag oyster cultivation system in the lower Delaware Bay. Photo credit: Michael Whiteside.

A variety of oyster seed is available from hatcheries located along the east coast. Farmers typically plant seed from 2 to 15 mm in size. Production from hatchery to market on average takes 1.5 to 2 years (Fig 2) and requires extensive husbandry involving washing to remove biofouling, adjusting the volume of oysters in cages to optimize growth, sorting, grading and culling oysters. Market size and harvest season is determined by the farmer and is independent of wild harvest restrictions. Some farms harvest year-round. Growers typically use high volume trash pumps to remove biofouling organisms from the oysters using water pressure. A major oyster farm pest is the spionid mud worm *Polydora cornuta*, which aggregates as thick colonies surrounded in mud that can smother oysters if not removed. Labor costs associated with biofouling removal is significant. A second mud worm *Polydora websteri* causes unsightly internal shell blisters that negatively affect oyster health and market value. Oyster farming is a year-round endeavor; however, most activity occurs from March through December. Winter loss can be significant and some farmers protect stocks by moving them to overwintering areas with deeper water or cold storage.


Figure 2. Farm raised oysters (*Crassostrea virginica*) from Delaware Bay that are ready for market. Photo credit: Lisa Calvo.

Oyster farms may be permitted in the Bay via three mechanisms: Riparian Grants, Shellfish Leases or Aquaculture Development Zone (ADZ) Leases. Riparian Grants are rare, and convey rights to private individuals holding the riparian as owners of the bay bottom. Oyster farming on riparian grants requires additional approval to grow shellfish and requires permits from New Jersey Division of Land Use Regulation (DLUR), New Jersey Bureau of Water Monitoring (BMWM), New Jersey Department of Agriculture (NJDA) and the U.S. Army Corps of Engineers (Corps). More often, tidally flooded lands are held in the public trust. In the 1850s, a system was developed for the private lease of bay bottom for the purpose of planting and growing oysters. Oyster farmers can obtain annually renewable shellfish leases from the Delaware Bay Shellfish Council. Oyster farmers operating on these private shellfish leases require authorizations from DLUR, the New Jersey Tidelands Resource Council (TRC), BMWM, NJDA, and the Corps.

Lastly, ADZ leases were established in 2011 to cluster structural aquaculture to facilitate management, streamline permitting, and minimize user conflicts. ADZ leases are pre-permitted. ADZs have been established in three areas in Middle Township in Cape May County, New Jersey. Two areas are approximately 5 miles offshore (ADZ-2 and ADZ-3) and one area (ADZ-4) is located in the nearshore intertidal about 1 mile south of Pierces Point. Twenty-four 1.5 acre parcels were established at ADZ-4 and a combined 1100 acres are available as 10 acre parcels at ADZ-2 and ADZ-3. Growers leasing ADZs still require permits from BMWM and NJDA.

NJ statewide production in 2016 was two million oysters with a farm gate value of $1.37 million. Wholesale pricing averaged $0.62 per piece and direct sale price averaged $0.91 (Calvo 2018). In 2021, there were eleven active oyster farms enterprises in Delaware Bay and NJ statewide oyster aquaculture production was reported to be 3.8 million oysters (pers. communication Craig Tomlin, NJDEP). That production number, however, is likely a stark underestimate of actual production due to reporting limitations. Farm raised oysters are valued for their high quality and are mainly sold to raw bars and restaurants for the half-shell market. However, as a consequence of the COVID-19 pandemic, oyster farms established or expanded direct-to consumer markets. There is currently no oyster aquaculture occurring in the Delaware State portion of the Delaware Bay.

In addition to their economic value, Delaware Bay oyster farms provide many ecological benefits akin to wild oyster populations. An increasing body of literature demonstrates the ecological services provided by the oysters on oyster farms including water filtration, nutrient cycling and sequestration, and habitat provisioning (Alleway et al. 2019, Coen et al 2007). Both regulators and coastal communities are interested in environmental benefits that may be provided by shellfish farms. It is well established that aquaculture gear alters habitat (e.g. increases structural complexity) and diversifies the benthic community of the habitat in which the gear is placed (e.g. O'Beirn et al. 2004, DeAlteris et al. 2004). Marenghi et al. (2010) observed that 2 types of oyster aquaculture gear (rack and bag, and floating cages) supported similar assemblages of finfish and invertebrates as restored oyster reefs in the Delaware Inland Bays. Most recently, the use of underwater cameras has been employed to quantify habitat provisioning by oyster farms in coastal Connecticut (Mercaldo-Allen et al. 2021), Washington

(Muething et al. 2020 and Ferriss et al. 2021) and New Jersey (Shinn et al. 2021). In New Jersey, videos were collected from both an off-bottom oyster farm using two different gear types (cages and floating bags) (Fig 3) and a marsh edge in Barnegat Bay. The overall community associated with the cage habitat differed from the other two habitats and furthermore, provides evidence of habitat provisioning for both finfish and invertebrates by intertidal oyster farms that could operate similarly to a naturally structured habitat (Shinn et al. 2021).

Figure 3. A sheepshead (*Archosargus probatocephalus*) feeds on sessile organisms attached to structured, oyster aquaculture gear in New Jersey. Photo Credit: Jenny Shinn.

A recent study examined the water quality benefits oysters in the Delaware Estuary provide by conducting field experiments at an oyster farm in Rehoboth Bay and Delaware Bay to quantify farm-specific year-round oyster filtration services (Barr, 2022). Field experiments were conducted seasonally in 2020 and 2021 using a flow-through filtration chamber with ambient water from the farm sites to calculate the biomass of particles oysters removed from the water column. The results broadly showed oyster filtration physiology differed between locations and through the year. Moreover, oysters at one intertidal farm in Delaware Bay were estimated to remove 43.8 tons of particulate matter from the water column acre^{-1} year^{-1}.

The expansion of oyster farming in the lower Bay is limited by potential conflict with important migratory shore birds which aggregate in the area in Spring. Aquaculture operations occur within portions of the same tidal flats used by the migratory shorebirds, including the threatened rufa subspecies of the red knot (*Calidris canutus rufa*), ruddy turnstone (*Arenaria interpres*), sanderling (*C. alba*), and semipalmated sandpiper (*C. pusilla*). These intertidal oyster farms also occupy a small portion of available horseshoe crab (*Limulus polyphemus*) spawning beaches. The co-location in both time and space of horseshoe crabs and migratory shorebirds and the oyster farm activities presents a unique socio-economic-ecological interaction that has been studied recently. Maslo et al. (2020) assessed the impact of oyster aquaculture as practiced in the lower Delaware Bay on the distribution and foraging behavior of shorebirds. While red knot numbers were slightly reduced during oyster tending activities, red knot abundance was predominantly determined by the presence of other foraging shorebirds (Maslo et al. 2020). This pattern was consistent for all focal species, strongly suggesting that factors other than oyster aquaculture are the primary drivers of shorebird distribution during the stopover period (Gillings et al. 2007). Separately, Munroe et al. (2020) characterized the interactions between oyster farms and mature horseshoe crabs accessing spawning habitat. Observations of horseshoe crabs showed conclusively that neither crab access to spawning habitat, nor their use of mudflats is altered by the presence of rack-and-bag oyster aquaculture gear (Munroe et al., 2020).

Since the ESA listing of the red knot as threatened in 2015, industry growth has been directed away from this nearshore area of conflict and led to the development of enhanced technology for cage culture of oysters in subtidal upper Bay areas. Oyster aquaculture in this region requires very robust bottom cages and large vessels capable of handling such cages (Fig 4). Significant investment has been made by two farms to expand bottom culture capacity supporting a significant proportion of NJ's oyster aquaculture production.

In accordance with Endangered Species Act (ESA), the USFWS has developed a Programmatic

Figure 4. Large cage used in subtidal oyster aquaculture in the Delaware Bay. Photo credit: Jenny Shinn

Biological Opinion (PBO) for structural aquaculture operations in portions of the Delaware Bay in Cape May County, New Jersey for the Corps (USFWS 2016). The PBO considers the potential impacts on the threatened red knot by the Corps' issuance of aquaculture permits in the action area. It includes an Incidental Take Statement for existing oyster farms and establishes certain actions that the aquaculture farmers must take, known as Conservation Measures (CMs), to reduce the potential harm of oyster farming on the red knot population. "The aim of the Biological Opinion is "to benefit or promote the recovery of" the red knot. The CMs include restrictions on gear placement, farm work hours, and access to farms. The changes associated with the PBO have resulted in the closure of two farm sites and limits future expansion of aquaculture in certain areas." The Biological Opinion applies for a 10-year period (2016-2026) and establishes a mechanism for adaptive management.

It is important to understand the ecological role that oyster farms play in various nearshore coastal habitats near the Delaware Estuary and around the world and to appreciate the role that changes in ecosystem indicators may make to the vitality of the industry.

References

Alleway, H. K., Gillies, C. L., Bishop, M. J., Gentry, R. R., Theuerkauf, S. J., and Jones, R. (2019). The ecosystem services of marine aquaculture: valuing benefits to people and nature. Bioscience 69, 59–68. doi: 10.1093/biosci/biy137

Barr, Janine. 2022. Wild and farm Eastern oyster (*Crassostrea virginica*) contributions to improved water quality in the mid-Atlantic: contemporary and future climate estimates. MS Dissertation. Department of Marine and Coastal Sciences, School of Environmental and Biological Sciences, Rutgers, the State University of New Jersey. New Brunswick, New Jersey. 134 pages. In Preparation.

Calvo L. 2018. New Jersey Shellfish Aquaculture Situation and Outlook Report 2016 Production Year. New Jersey Sea Grant NJSG-18-

Coen, L. D., Brumbaugh, R. D., Bushek, D., Grizzle, R., Luckenbach, M. W., Posey, M. H., et al. (2007). Ecosystem services related to oyster restoration. Mar. Ecol. Prog. Ser. 341, 303–307. doi: 10.3354/meps341303

DeAlteris J, Kilpatrick BD, Rheault RB (2004) A comparative evaluation of the habitat value of shellfish aquaculture gear, submerged aquatic vegetation and a non-vegetated seabed. J Shell Res 23(3):867-874

Ferriss B, Veggerby K, Bogeberg M, Conway-Cranos L and others (2021) Characterizing the habitat function of bivalve aquaculture using underwater video. Aquacult Environ Interact 13:439-454. https://doi.org/10.3354/aei00418

Gillings, S., Atkinson, P. W., Bardsley, S. L., Clark, N. A., Love, S. E., Robinson, R. A., ... & Weber, R. G. 2007. Shorebird predation of horseshoe crab eggs in Delaware Bay: species contrasts and availability constraints. Journal of Animal Ecology, 76(3), 503-514.

Marenghi F, Ozbay G, Erbland PJ, Rossi-Snook K (2010). A comparison of the habitat value of sub-tidal and floating oyster (*Crassostrea virginica*) aquaculture gear with a created reef in Delaware's Inland Bays, USA. Aquacul Int 18(1):69-81

Maslo, B., Burkhalter, J. C., Bushek, D., Yuhas, T., Schumm, B., Burger, J., & Lockwood, J. L. (2020). Assessing conservation conflict: Does intertidal oyster aquaculture inhibit foraging behavior of migratory shorebirds?. Ecosphere, 11(5), e03097.

Mercaldo-Allen R, Clark P, Liu Y, Phillips G and others (2021) Exploring video and eDNA metabarcoding methods to assess oyster aquaculture cages as fish habitat. Aquacult Environ Interact 13:277-294. https://doi.org/10.3354/aei00408

Muething KA, Tomas F, Waldbusser W, and Dumbauld BR (2020) On the edge: assessing fish habitat use across the boundary between Pacific oyster aquaculture and eelgrass in Willapa Bay, Washington, USA. Aquacult Env Interac 12: 541-557

Munroe, D. M., Grothues, T. M., Cleary, N. E., Daw, J., & Estrada, S. (2020). Oyster aquaculture does not impede spawning beach access for Atlantic horseshoe crabs *Limulus polyphemus*. Aquaculture Environment Interactions, 12, 81-90.

O'Beirn FX, Ross PG, Luckenbach MW (2004) Organisms associated with oysters cultured in floating systems in Virginia, USA. J Shell Res 23(3):825-829

Shinn, J.P., D.M. Munroe, J.R. Rose, 2021. A Fish's-Eye-View: Accessible Tools to Document Shellfish Farms as Marine Habitat In New Jersey. Aquacult Env Interact 13: 295-300 DOI: https://doi.org/10.3354/aei00407

USFWS. Biological Opinion on the Effects of Existing and Expanded Structural aquaculture of Native Bivalves in Delaware Bay, Middle Lower Township Cape May County, New Jersey on the Federally Listed Red Knot (*Calidris canutus rufa*). 2016 175 p.

Living Resources Feature 2

Adopting Principles of Food Justice for Equitable Oyster Aquaculture Industry Development

Cristina Sandolo, PSM, Ocean Food Systems

Introduction

While many Black watermen were critical to our region's oyster industry historically, the aquaculture industry lacks racial diversity, particularly at the business ownership level. Recent work by Sandolo (2021) analyzed barriers to entry and the history of Black oystermen in the region, and Food Justice, to determine if Food Justice themes could be adopted to promote socioeconomic diversity among business owners. 13 Oyster Producers and 12 Industry Experts from MD, DE, and NJ and 3 Food Justice Experts were interviewed about industry history, management and statistics, and food justice. Over 85 peer-reviewed and non-peer-reviewed resources were also analyzed. Results indicate that exploitation of Black watermen and historic barriers to wealth accumulation have disproportionately fueled barriers to entry today. The research identified 8 major barriers to entry in aquaculture and 5 core Food Justice themes that were then developed into a framework of recommendations (Sandolo, 2021). The framework presents an opportunity for industry managers to facilitate engagement among a more diverse population of both entrepreneurs and consumers, promoting economic development, nutrition security, and environmental restoration in the Delaware Estuary and beyond.

What is Food Justice?

Historically, communities of color have been exploited in the food system and disproportionately excluded from key positions and from land and business ownership (Horst et al, 2017). Black farmers have experienced racial discrimination from lenders and subsidy programs. Farmland loss in the 1900s impacted Black farmers 3 to 4 times more than white farmers (Alkon, 2007). Efforts to build community determination of the food system date back at least the Black Panther Party's 1960's community food programs, such as free breakfast programs led by and serving the Black community, transforming roles from consumer to decision-maker and doer (Alkon, 2007). In 1996, the Community Food Security Coalition convened to create a vision for Food Justice (Bradely & Herrera, 2016) and the movement has grown dramatically since then, as a response to racial inequities and a corporate-controlled food system (Clendenning et al., 2016). The Food Justice movement seeks to create equity in all aspects of the food system, from seed to table, including land and business ownership, labor rights, and consumer access, through systemic and structural change (Horst et al, 2017). Sandolo (2021) defined Food Justice as the right of all people to grow, process, sell, distribute and consume their food of choice, and to have the means to do so, such as land ownership, business ownership and access to water and other resources and identified 5 core themes:

- Creating systemic change throughout the food system
- Having an Antiracist and Anti-exploitation Approach
- Focusing on Community Determination of a Just Food System: Acknowledge, Invest in Talent and Leadership of Communities of Color

- Ensuring The Right to the Food System: The Right to Business Ownership and Livable Wages; the Right to Land and Resources; The Right to Food

- Valuing People, Culture, Community, and Environment

Food Justice Relevance to Oyster Industry

As with land farming, seafood resources were stolen from indigenous communities, privatized, and exploited through slavery. Opportunities for advancement, business development, and resource ownership in the 1800s' oyster boom were mostly limited to white watermen. Oystering did provide a unique sense of freedom and some monetary benefits for Black oystermen. And, some Black watermen did own and captain ships. Captain James McBride (Fig 1) is described as New Jersey's only Black oyster fleet owner; his wife, Lucy M. McBride took on the enterprise after he tragically died while at sea (Delaware Bay Watermen's Memorial, 2013). Sam Turner joined the Civilian Conservation Corps and sent money to his father so they could buy land in 1939. They opened a shucking business, employing 40 shuckers (Anderson, 1998). These stories should be memorialized and amplified, celebrating the integral role of Black watermen in the region. Still, most Black watermen remained in lower-level positions through the mid 20th Century, even with a deep knowledge of the marine environment and skills in navigation and oyster harvesting and processing (Chiarappa, 2018) (Fig 2).

Figure 1. Captain James Elwood McBride was the first African American boat owner and captain in the Delaware Bay region. He is pictured with his wife, Lucy McBride. Photo provided courtesy of the Delaware Bay Watermen's Memorial.

Laws and codes prevented most Black watermen from owning or operating boats and travel required permission from a white person (Clarke, 2021). William Wallace remembers two failed attempts by his dad, a Black oysterman in Chesapeake Bay, to secure a loan for a boat purchase. Fearing a loss of labor, the owner of the boat his dad labored on discouraged the banker from giving the loan (Anderson, 1998). Robert Morgan, an oysterman from Delaware, described: "...there was a lot of Black mates who were just as qualified as any white man but just couldn't make it. If they were somebody's son they'd have been captain. . . you know how things work. We're below the Mason-Dixon Line here. Things change slow"(Chiarappa, 2018).

The impacts of marginalization in the industry were only exacerbated by systemic oppression preventing Black people from equal participation in the economy. Systems of slavery and structural racism in labor, the housing market, and credit (such as disproportionate rates of denials for loans for businesses and homes and limited business opportunities) have caused racial discrepancies in generational wealth accumulation still evident today (Herring & Henderson, 2016; Miller, 2011). The demographics of today's aquaculture industry illustrate that many Black watermen were excluded from the financial benefits and legacy that the booming industry could have provided.

Figure 2. Shucking house in Commercial Township, NJ. Photo provided courtesy of Bayshore Center at Bivalve. Photographer: Harvey W. Porch.

Barriers to Entry: Oyster Aquaculture in DE and NJ

Although the wild oyster fishery is somewhat limited to general entry today due reduced oyster stocks and harvest limitations in most estuaries, the aquaculture sector is poised to grow. However, a lack of both historical awareness and gender and racial diversity exist in aquaculture today (Sandolo 2021). 11 of the 12 oyster producers interviewed in the study identified as white men. Furthermore, just 7 acknowledged the historic role of Black watermen; 1 made an "assumption" that Black watermen participated while another had "no idea." Barriers to entry were identified by interviewees as:

- Racial and Gender Bias
- Exclusiveness. Those without relationships (e.g., from previous generations' success) have difficulty gaining entry.
- Access to Capital. Startup costs are high.
- Permitting Processes. Lengthy permitting processes and public protests prevent those without disposable income, time, and social capital from participating.
- Waterfront Access. A lack of public waterfront access for small businesses limits the pool of participants to those who can afford coastal land or dock space.
- Exposure to the Industry.
- Experience on the Water.
- Lack of Support and Technical Assistance.

Recommendations for the Industry

Industry managers can utilize a Food Justice framework to eliminate barriers preventing equal participation among people of color and to expand consumption of oysters. Recommendations include (Table 1):

1. **Develop an ant-racist approach**. Hire a consultant to train staff in adopting an antiracism, equity, or Food Justice Framework.
2. **Acknowledge Leaders of Color**. Memorialize Black talent, leadership, and labor. Integrate the history of watermen of color into management plans and permitting processes. Develop multilingual material.
3. **Acknowledge and Eliminate Exploitation and Harm**. Equip staff with knowledge: integrate the history, including the harm, into training and meetings.
4. **Eliminate Racial and Gender Bias**. Require antiracism and anti-bias training at all staff levels.
5. **Facilitate Access to Capital**. Establish and expand equity-based, no- or low-interest loans and grants. Integrate equity lens into application review to prioritize applicants of color. Provide Technical Assistance in applying for funds and in business planning. Promote livable wages to facilitate advancement.
6. **Create More Waterfront Access**. Provide dock space and waterfront space to descendants of watermen of color and interested entrepreneurs of color. Promote development of systems that require less onshore activity.
7. **Boost the Buying Market**. Integrate goals in equitable food access into industry plans. Invest in marketing and promotion: Engage historically marginalized communities; tell the story and build pride. Initiate farm-to-institution programs for producers to sell to state institutions. Engage growers in planning. Incentivize oyster purchases made with Federal Benefits. Facilitate shellfish handling certification for growers.

Table 1. Recommendations table that lists action steps aligned with Food Justice themes, that could reduce barriers to entry and increase Food Justice within the oyster aquaculture regions studied. Acronyms used include: People of Color (POC); Food Justice (FJ); Technical Assistance (T.A.).

RECOMMENDATIONS: REDUCING BARRIERS TO ENTRY IN MD, DE AND NJ OYSTER AQUACULTURE				
FOOD JUSTICE THEMES AND SUBTHEMES		BARRIERS TO ENTRY		
		Racial and Gender Bias Exclusiveness	Permitting Access to Capital Technical Assistance	Waterfront Access Exposure to Industry Experience on the Water
Creating systemic change throughout the food system.		Hire a consultant to lead all staff through training in: 1) the history of the industry, 2) antiracism and 3) adopting an Antiracism, Equity or Food Justice framework.		
Antiracist and Anti-exploitation Approach	Acknowledging Harm, History, and Trauma	Require antiracism training for all industry employees Memorialize Black talent, leadership and labor; acknowledge exploitation Integrate history into staff training and management practices	Integrate antiracism / equity lens into permit process Create / expand no- and low-interest and forgivable loans and grants Invest in technical assistance in permitting / business planning / grant applications technical assistance for entrepreneurs of color.	Utilize antiracism framework in outreach and engagement programs; tell the story of the industry Provide dock / working waterfront space to descendants of watermen of color and entrepreneurs of color Build pride / connection to the industry in the region through marketing
	Reparations			
Community Determination of a Just Food System	Acknowledge, Invest in Talent and Leadership of Communities of Color	Invest in people of color as the future of aquaculture.	Position interested descendants of watermen and people of color as compensated decision-makers	Invest in and compensate people of color as leaders Invest in organizations led by people of color, internships, and community engagement to amplify industry exposure
	Community Power and Community-Determination	Compensate descendants of watermen of color to tell industry history		Provide opportunities for people to explore aspects of aquaculture outside of farming.
The Right to the Food System	Right to Food	Pay for shellfish handling certification to facilitate direct-to-consumer sales	Integrate equitable food access goals into management plans Invest in regional marketing / build pride	Establish incentive programs for oyster purchases made with Federal Benefits.
	Right to Business Ownershp and Livable Wages	Integrate industry history into creative place-making in urban areas.	Hold informational / T.A. sessions in non-coastal areas and in communities of color Develop oyster farm-to-institution programs with state universities, hospitals, and other institutions Simplify permitting and provide multiple languages and various avenues for information sharing.	Invest in paid apprenticeships / externships as opportunities for entrepreneurs to build skills and successful businesses Invest in educational programming (K-College) and in organizations led by people of color to conduct youth education in fisheries and seafood
	Right to Land and Resources	Engage the public through events and online.	Develop low-cost dock access for oyster entrepreneurs who do not have convenient private dock access. Develop "right to farm" policies that prohibit or minimize complaints	Invest in working waterfront space and on-deck equipment Ensure leases are not dependent on landownership Create Internship / Apprenticeship programs at state agencies, small aquaculture businesses, universities, and organizations to boost their capacity while increasing engagement
Valuing People, Culture, Community, and Environment	The Right to Human Dignity	Establish multilingual outreach and communications across all platforms and initiatives.	Promote livable wages via grant / loan requirements Highlight industry's role in historic regional development through product marketing Eliminate / minimize public protest process and expand on pre-approved Aquaculture Zones	Invest in memorializing the history, celebrating the individuals, and acknowledging historic injustices. Compensate families of black oystermen for decision-making roles in planning memorials.
	Community / Collaboration	Sponsor events that showcase the history of the oyster industry. (e.g. at Bivalve Center, in inland cities)	Support businesses through grants, marketing, and institutional purchasing Conduct community education to build social capital	Invest in organizations led by people of color for community outreach and education (e.g. urban communities; coastal communities; incarcerated communities) to increase interest in the industry and increase the buying market
	Environmental Connection and Stewardship	Invest in people of color to lead environmental education programming	Expand oyster purchasing programs for restoration Prioritize environmentally beneficial coastal activities in state, county, municipal plans.	Invest in environmental organizations to conduct community engagement about the industry and the environmental benefits of aquaculture

8. **Boost Exposure to the Industry and its History**. Memorialize the history and acknowledge injustices. Position families of Black oystermen as decision makers in planning memorials. Invest in organizations led by people of color to amplify exposure. Establish compensated apprenticeships and internships within state agencies, aquaculture businesses, and partners (state universities, HBCUs, community colleges, city high schools and creative place-making initiatives in urban areas.

9. **Modify Permitting Processes**. Do not require land ownership for leases. Protect growers from personal attacks: eliminate public protests, encourage bottom culture; conduct community education; establish the "right to farm" in community-scale aquaculture; position activities that are beneficial to coastal ecosystems as priorities within development plans. Expand pre-approved aquaculture areas. Expedite and incentivize descendants of watermen of color for loans and grants. Improve timeliness of application review: engage apprentices in application review. Provide permit application technical assistance.

Conclusion

Now is the time for the oyster industry to recognize the historic barriers to advancement faced by oysterers of color that have led to modern day barriers to entry in aquaculture. Ignoring historical wrongdoings and opportunities for repair is no longer acceptable. Increasingly, consumers consider their food choices' impacts, evidenced by the terrestrial Food Justice movement, from which the oyster industry has an opportunity to learn. Using the framework developed by Sandolo (2021) would promote socioeconomic diversity among business owners and position aquaculture as a reflection of Food Justice in our region, boosting demand and popularity and creating triple societal benefits: small business development, healthy ecosystems, and nutritional food.

References

Alkon, A. H. (2007). Growing Resistance: Food, Culture and the Mo' Better Foods Farmers' Market. Gastronomica, 7(3), 93–99. https://doi.org/10.1525/gfc.2007.7.3.93.

Anderson, H. (1998). Black Men, Blue Waters African Americans on the Chesapeake. Marine Notes, April.

Bradley, K., & Herrera, H. (2016). Decolonizing Food Justice: Naming, Resisting, and Researching Colonizing Forces in the Movement. Antipode, 48(1), 97–114. https://doi.org/10.1111/anti.12165

Chiarappa, M. J. (2018). Working the Delaware estuary: African American cultural landscapes and the contours of environmental experience. Building and Landscapes, 25(1), 64–91. https://doi.org/10.5749/buildland.25.1.0064

Clarke, W. M. (2021, February). Water Bound. Cheseapeake Quarterly. https://storymaps.arcgis.com/stories/1c6f8f445a7c43e5982e6e4682df537e

Clendenning, J., Dressler, W. H., & Richards, C. (2016). Food justice or food sovereignty? Understanding the rise of urban food movements in the USA. Agriculture and Human Values, 33(1), 165–177. https://doi.org/10.1007/s10460-015-9625-8

Delaware Bay Watermen's Memorial. (2013). Contemplation - Reflection - Remember the Past - Envision the Future. http://watermensmemorial.org/about.htm

Herring, C., & Henderson, L. (2016). Wealth Inequality in Black and White: Cultural and Structural Sources of the Racial Wealth Gap. Race and Social Problems, 8(1), 4–17. https://doi.org/10.1007/s12552-016-9159-8

Horst, M., Mcclintock, N., & Hoey, L. (2017). The Intersection of planning, urban agriculture, and food justice: A review of the literature. Journal of the American Planning Association, 83(3), 277–295. https://doi.org/10.1080/01944363.2017.1322914

Miller, M. C. (2011). Land and Racial Wealth Inequality. American Economic Review, 101(3), 371–376. https://www.jstor.org/stable/29783772.

Sandolo, C. (2021). Adopting Principles of Food Justice for Equitable Oyster Aquaculture Industry Development. Graduate Program in Ocean Foods Systems, University of New England, Maine, USA.

7.1.4 Freshwater Mussels

Description of Indicator

Freshwater mussels are filter feeding bivalve mollusks that live in lakes, rivers, and streams (Fig 7.1.4.1). Similar to oysters, freshwater mussels provide many important functional and structural benefits for natural ecosystems. Supporting and regulating ecosystem services include increasing water clarity, filtering particulate pollutants, and enhancing nutrient storage and cycling (Vaughn et al. 2008, Newton et al. 2011, Hoellein et al. 2017, Kreeger et al. 2018). Freshwater mussels can serve as habitat engineers that increase complexity, buffer hydraulic stresses, reduce erosion, and enrich habitats for benthic plants, invertebrates and fish (Gutiérrez et al. 2003, Lopez and Vaughn 2021, Sansom et al. 2018, 2020, Vaughn et al. 2008). Their cultural benefits are diverse, ranging from their historical importance for Native Americans and the shell button, pearl and jewelry industries to their current use for myriad bioassessment and educational programs. For summaries of the ecosystem services provided by mussel assemblages, see Strayer 2017, Vaughn 2018, Atkinson et al. In Press).

Figure 7.1.4.1 Eastern Pondmussel (*Sagittunio nasutus*) juveniles exhibiting new growth along ventral margin. Photo credit: Kurt Cheng, Partnership for the Delaware Estuary.

The potential beneficial effects of mussel beds on water quality are generating increasing research and restoration interest (Kreeger et al. 2018, Strayer et al. 2019, Wood et al. 2021). Although vastly depleted in numbers and species richness compared to historical conditions, enough freshwater mussels appear to remain in the Delaware River Basin to materially contribute to water quality by their filtration (Kreeger and Kraeuter 2010, Anderson and Kreeger 2010). For example, Kreeger (2008) measured the abundance of the Eastern Elliptio (*Elliptio complanata*) in the Brandywine River and used survey data from Dr. W. Lellis (2001) (U.S.Geological Survey) to estimate that there are at least 4 billion adult mussels of this species across the Basin. Pairing these survey data with measured physiological processing rates, this species was estimated to filter about 10 billion liters of water per hour across the Basin, which is roughly 250 times the volume of freshwater entering the tidal estuary (Kreeger and Kraeuter 2010). A similar approach was used to estimate that representative beds of freshwater mussels in the tidal Delaware River upstream from Philadelphia filter more than a million gallons of water and 8 tons of suspended particles per day per hectare (Fig 7.1.4.2) (Kreeger et al. 2013).

Freshwater mussels grow more slowly than their marine counterparts. They also live longer (50 years or more) and have complicated reproduction strategies dependent on fish hosts (Haag 2012). As long-lived, relatively sedentary creatures that process large amounts of water over their soft tissues, freshwater mussels are particularly sensitive to water quality and contaminants (e.g., Wang et al. 2007). The health, population abundance, and species diversity of freshwater mussels therefore represent excellent bioindicators of freshwater systems, particularly over long periods of time. Unfortunately, freshwater mussels are typically not sampled effectively as part of traditional macroinvertebrate assessments (see section 7.1.5), and so data on the status and trends of freshwater mussel populations are scarce.

Figure 7.1.4.2 Freshwater mussels are filter-feeding bivalves that efficiently remove microparticulate matter, resulting in improved water clarity, greater light penetration, and beneficial transformation of many filtered pollutants. In this outreach demonstration, both tanks received the same water, but the addition of live mussels to the tank on the right had dramatically enhanced water quality within 4 hours. Photo credit: Danielle Kreeger, Partnership for the Delaware Estuary.

Present Status

Freshwater mussels are among the most imperiled of all animals and plants in North America (Nobles and Zhang 2011), which has the world's greatest diversity of this taxonomic group with nearly 300 species (Williams et al. 2017). More than 70% have special conservation status (Williams et al. 1993, FMCS 2016). At least 12 species are native to the Delaware River Basin (Ortmann 1919, Campbell and White 2010, Kreeger and Kraeuter 2010); however, all but one species is currently reported to be uncommon (Kreeger and Kraeuter 2010).

To assess present status, survey data were gathered from disparate sources and analyzed for the past 25 years from the portions of Delaware, New Jersey, and Pennsylvania that comprise the Delaware River Basin. Most survey data were qualitative reports of species presence or absence rather than quantitative assessments of size class distribution and abundance. Due to limited data availability and the sensitive nature of heritage datasets, the current status of mussel resources was analyzed as species richness at the sub-watershed scale (i.e., by tributary or hydrologic unit code (HUC) unit). This analysis suggests that the overall condition of freshwater mussel populations is poor in streams where water quality, habitat degradation, dams, and other factors have progressively eliminated or reduced mussel populations over the past 100 or more years (Thomas et al. 2011).

Joint surveys in southeastern Pennsylvania by the Partnership for the Delaware Estuary (PDE) and the Academy of Natural Sciences of Drexel University between 2000 and 2010 found that only 4 of >70 Piedmont stream reaches contained any freshwater mussels (Thomas et al. 2011). Even the most "common" native species are presently patchy in distribution and limited in abundance. Furthermore, most mussel populations that have been found appear to lack juveniles and be comprised mainly of older individuals, suggesting that most populations in Piedmont streams are not successfully reproducing. In contrast, surveys for freshwater mussels in Coastal Plain streams of southern Delaware and New Jersey suggest that mussel populations are not quite as degraded with regard to the number of streams that still retain extant populations of at least one native species (Cheng and Kreeger 2015). Similarly, extensive surveys of the undammed and tidal reaches of the mainstem Delaware River have revealed sometimes large beds of mussels (5-100 per square meter) (Lellis 2001, 2002, Kreeger et al. 2011). Several species found in the tidal Delaware River in 2010-1011 were previously believed to have been extirpated from the basin because they had not been reported in the published literature since Ortmann's surveys 100 years earlier (Ortmann 1919). Importantly, quantitative surveys of the Delaware River between Philadelphia, Pennsylvania, and Trenton, New Jersey, revealed a few locations that still contained abundant juvenile mussels, demonstrating that natural reproduction is still occurring within that zone of the river where up to 6 mussel species can still be found (Kreeger et al. 2013, 2015).

Taken together, these richness and occurrence data indicate that the condition of mussel assemblages on Coastal Plain streams and the tidal Delaware River is healthier compared to assemblages in non-tidal tributaries of Piedmont streams, including the Schuylkill River. Anecdotal observations of mussel fitness and sizes support this observation, as evidenced by lower shell erosion, richer tissue biochemistry, and a more diverse size class range in Coastal Plain locations compared to Piedmont streams. Caging and tagging relocation studies in Delaware and Pennsylvania indicate that the food and water quality of most streams can currently support mussel fitness, growth and survival (Kreeger and Padeletti 2011, Gray and Kreeger 2014, Cheng and Kreeger 2015). We therefore infer that the main current culprits for the poor status of mussel populations is impaired habitat or limited fish hosts that are needed for reproduction. Since freshwater mussels rely on fish, usually species-specific relationships, for successful reproduction dams that block fish passage can disrupt reproduction and gene flow (McMahon 1991, Neves 1993).

Freshwater mussels require stable streambeds that they can burrow into, and stormwater scouring causes unstable conditions that wash mussels out or leads to very course sediments or bedrock. Shoreline alterations can also disturb or degrade mussel habitat, and dredging can result in a direct mussel take. Even in streams that have been the subject of deliberate remediation, restoration practices rarely consider the habitat needs of freshwater mussels (Wood et al. 2021).

Past Trends

The most comprehensive historical regional mussel survey was conducted in Pennsylvania between 1909 and 1919 (Ortmann 1919). Ortmann's surveys described the bivalve species that were present or absent in many representative watersheds. However, even by that time, dams and water quality degradation may have already affected mussel communities. Nevertheless, the study provided an excellent benchmark for gauging long-term trends in the mussel assemblage for the past 100 years, especially related to species diversity. Ortmann (1919) reported 12 species of native mussels in the Delaware River Basin, most of which were present at that time in southeastern Pennsylvania (Fig 7.1.4.3A). Although species richness was highest in the mainstem Delaware River even then, at least five species were present in several tributary watersheds, including the Schuylkill and Brandywine.

In contrast, Figure 7.1.4.3B depicts the current species richness of native mussels for those sub-watersheds where surveys have been completed since 1996 (Thomas et al. 2011; updated in 2022 by the PDE mussel database). Although the richness appears to have been preserved in the mainstem Delaware River and a few tidal tributaries in New Jersey, only one or no species has been detected in the past 25 years in most surveyed tributary streams of Delaware and Pennsylvania (Fig 7.1.4.3B). Taken together, these data suggest a long-term decline in mussel diversity in most areas of the lower Delaware River Basin where survey data exist.

The comparison between Figures 7.1.4.3A and 7.1.4.3B also suggests that the range of native mussel occurrence has shrunk significantly during the last 100 years in streams where historic and recent survey data exist. This decline appears to be continuing. For example, two species were found there as recently as 1998-2001, but no mussels have been found since 2002 in the upper White Clay Creek, Pennsylvania, despite annual surveys by PDE (Fig. 7.1.4.3).

The leading causes of historic mussel declines in the Delaware River Basin have been habitat and water quality degradation, like elsewhere in the United States (Haag 2012, FMCS 2016). As discussed above, the water quality and food within the system appears to now be sufficient to support mussel recolonization in many areas. However, as noted above, habitat degradation, fish passage blockages, and the inattention to mussel needs in stream restoration efforts are factors that continue to impede mussel recolonization in historically occupied habitats.

Future Predictions

Since the decline of native mussel biodiversity has been attributed to habitat and water quality degradation, the future prospects for freshwater mussels are likely to hinge on careful watershed management, inclusion of mussel requirements in stream restoration practices, and deliberate efforts to conserve and restore mussel populations. Human population is expected to grow by 80% this century in the basin, which threatens to exacerbate the stressors that have been affecting mussels for probably hundreds of years.

Climate change also threatens freshwater mussels because of increased thermal stress, periodic drought, and increased flooding and stormwater scouring (Kreeger et al. 2011, Najjar 2015). Freshwater mussels are especially sensitive to bed instability and inputs of fine sediments to the system, and so stormwater and flood scouring represent threats that are expected to increase with climate change. Salinity rise may also threaten mussels living in freshwater tidal areas, especially during droughts and storm events which can lead to localized spikes in conductivity (Cheng et al. 2021). Periodic dips in dissolved oxygen concentrations could constrain mussels in some areas such as deeper areas of the tidal Delaware River.

Since freshwater mussels depend on fish hosts for larval dispersal, it is unlikely that southern mussel species will be able to expand northward to fill niches that open if northern species are extirpated. The Eastern Pearlshell, *Margaritifera margaritifera*, is an example of a cold-adapted species that uses brook trout as a host – its present distribution in southeast Pennsylvania is constrained to a few cold headwater streams and below reservoirs in the upper Schuylkill Basin which release colder water from the bottom. Assisted migration of warm-adapted southern species represents a potential climate adaptation tactic, but the willful introduction of species that are not native to this region might carry unforeseen risks and is at odds with current management paradigms.

Enhanced conservation and restoration efforts have the potential to offset projected continued declines in freshwater mussels (Kreeger and Padeletti 2011). Given the severely weakened status of freshwater mussel richness, range, and abundance, it is vital that any extant populations be protected. Although some streams may no longer be as suitable for mussels as they were historically, results from pilot reintroduction trials during 2007-2017 at more than a dozen locations in Delaware and Pennsylvania (Gray and Kreeger 2014, Kreeger et al 2014, 2015, Cheng and Kreeger 2017) suggest the majority of historic streams and ponds are still capable of sustaining mussels, but natural recolonization is prevented because of either inhibited movements of suitable fish hosts or unsuitable habitat conditions. Mussel restoration in these areas can be accomplished by improving habitat conditions and promoting fish passage. Where suitable habitat exists (or is created) but mussels are still depleted or absent, stocking of hatchery-propagated mussel seed having appropriate genetic composition has the potential to expedite recovery (Kreeger et al. 2018, 2022). For example, new restoration approaches such as building mussel beds within urban living shoreline projects (Morgan et al. 2022) have the potential to also boost mussel carrying capacity via habitat enhancement, and mussel bed establishment in remediated areas may require stocking. Recent findings also suggest that freshwater mussels could be introduced into man-made aquatic systems such as stormwater management ponds to either promote water quality or diversify rearing locations for hatchery-produced mussels (Gentry et al. 2022). Growing interest in mussel-mediated ecosystem services, such as water quality benefits, could energize mussel restoration in the Delaware River Basin and beyond (Kreeger et al. 2018, Wood et al. 2021).

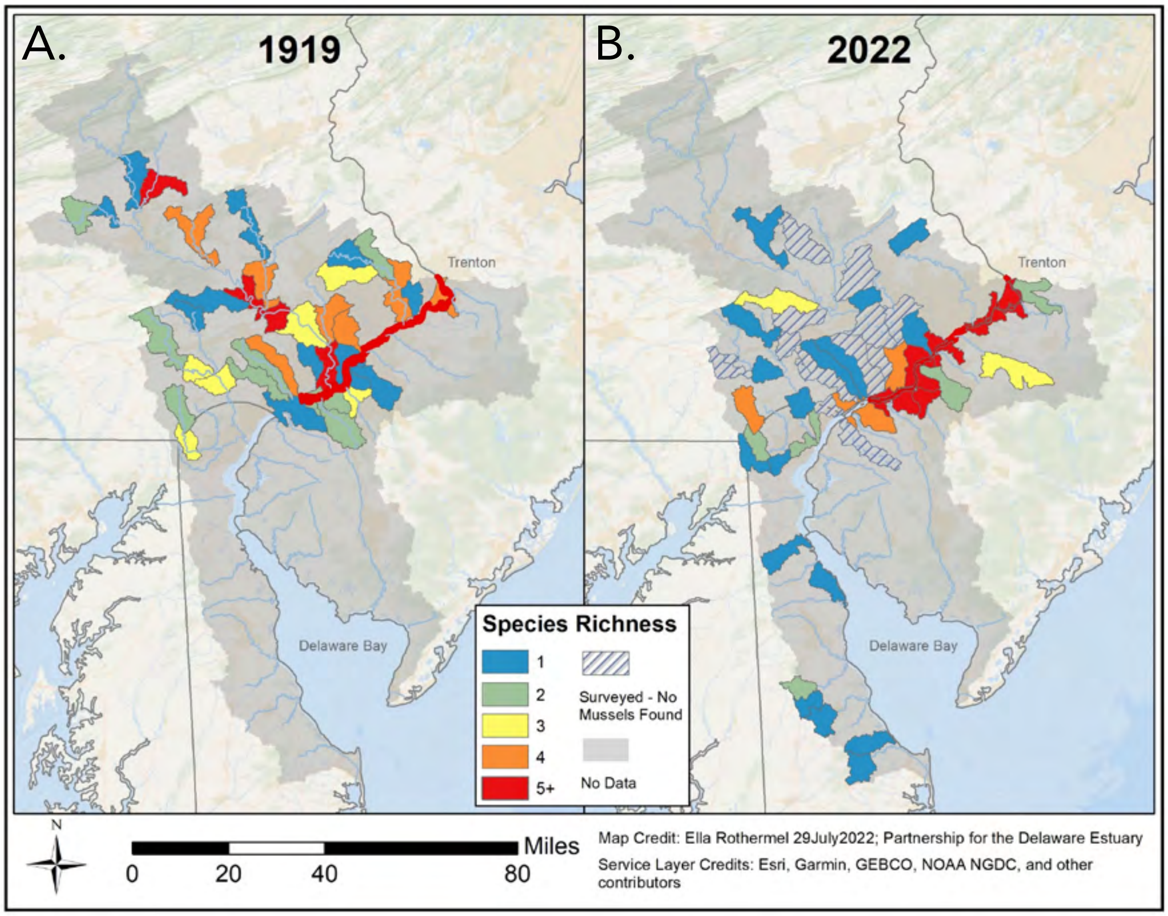

Figure 7.1.4.3 A) Species richness of native freshwater mussels reported in surveys conducted between 1919-1996, based on available data obtained by PDE. Surveys were primarily conducted by W. Ortmann prior to 1920 and A. Bogan during the 1980's. B) Species richness of native freshwater mussels reported in surveys conducted between 1996-2016. Surveys were primarily conducted by PDE with assistance in some areas by the Academy of Natural Sciences of Drexel University, Environmental Protection Agency Region 3 dive unit, Philadelphia Water Department, and the Western Pennsylvania Conservancy.

Actions and Needs

More proactive freshwater mussel monitoring for species presence and population health is needed across the Delaware Estuary and River Basin. Freshwater mussels are not targeted in routine macroinvertebrate assessments, and so mussel surveys are rarely performed despite their value for assessing long-term status and trends of aquatic health. Hence, survey data are not available for most sub-watersheds of the Basin for at least 25 years, if ever. Improved coordination and data sharing would also facilitate development of better indicators and a coordinated watershed restoration strategy. New survey technologies for mapping mussel beds and suitable habitats are being developed and could be used to fill vital data gaps, identify mussel conservation areas, and help prioritize restoration areas. Critical habitat for mussel beds should be mapped and protected. The confirmation of natural freshwater mussel propagation and rare species in the tidal freshwater zone of the Delaware River is important because these represent potential source populations and broodstock to support the restoration of genetically appropriate mussels in other areas of the Basin. Quantitative surveys of mussel beds in this span of the Delaware River suggest that the presence of rarer species may depend on high densities of more common mussel species; hence, conservation and restoration efforts should target the entire mussel assemblage in their natural species abundance ratios. More research is needed to clarify these interspecific relationships. Research is also needed to understand the life history, habitat requirements, and ecological importance of native species of freshwater mussels, which are still understudied compared to marine bivalves (e.g. oysters) or freshwater macroinvertebrates (e.g. insects). Due to the lack of mussels for experiments, there are few case studies that have empirically determined outcomes from large-scale mussel restoration efforts.

Thanks to recent advances in mussel propagation and rearing technology, we now have the ability to produce juvenile mussels and rear them quickly in ponds for use in restoration and enhancement projects (Patterson et al. 2018). Given the current status of freshwater mussels nationwide, it is widely accepted that this technology will be needed to help address mussel conservation and restoration goals (FMCS 2016). This is because natural populations are so depleted and mussel dispersal mechanisms so restricted that it is unlikely that they can recover on their own, and even in the places where natural recovery may be viable it would take a very long time due to their slow growth rates. As part of the Freshwater Mussel Recovery Program that PDE and partners have been coordinating since 2007, more than 50,000 juvenile mussels representing four species native to the Delaware River Basin were propagated between 2017 and 2022. These mussels were produced from Delaware River broodstock at either the Fairmount Water Works demonstration hatchery in Philadelphia, owned and operated by the Philadelphia Water Department, or the USFWS Harrison Lake National Fish Hatchery in Virginia. These mussels were reared to sizes that can withstand most predators and then released into various types of aquatic habitats where they continue to be monitored.

Building on these successes, increased hatchery capacity is needed to supply mussels for regional restoration, enhancement, bioassessment, research and engagement programming. PDE estimates that more than 1 billion mussel seed would be needed to address these needs within the Delaware River Basin. To help address this constraint, a "Mussels for Clean Water Initiative" was launched in 2019 to increase mussel production capacity and test new mussel restoration and enhancement concepts, particularly at sites that are impaired for microparticulate pollutants such as total suspended solids, nutrients, and human pathogens. This initiative aims to produce up to 500,000 juvenile mussels per year. Similar to fish and oyster propagation and stocking, there are various genetic, biosecurity and ecological considerations that exist with mussel hatcheries (e.g., see Strayer et al. 2019). Efforts to reverse mussel declines or enhance mussel-mediated ecosystem services using propagated animals should be mindful of these concerns and proactively address them with best management practices, research, and monitoring. Monitoring of restoration outcomes is aided by electronic tagging methods, biochemical and physiological fitness measures, new genetic markers, and a variety of other emerging technologies (e.g., Kreeger and Padeletti 2011, Gray and Kreeger 2014, Cheng and Kreeger 2014).

These various advances in restoration and monitoring tools offer hope that mussel declines can start to be reversed in the Delaware River Basin and elsewhere. However, in many areas mussel recovery will not be possible until all of the root issues causing their declines have been addressed. For example, best management practices to manage stormwater runoff are needed to promote more stable mussel habitats. Stream restoration practices should consider mussel habitat suitability. Water quality managers should consider mussel sensitivity to specific conductivity, dissolved oxygen, ammonia, metals and other parameters that could cause chronic or acute stress to natural mussel assemblages. Finally, permitting agencies should consider impacts to both common and rare mussel species when weighing damages from natural resource injuries, or dredging or development projects.

Summary

A robust community of freshwater mussels should be evident throughout most freshwater aquatic habitats of the Delaware River Basin, including diverse species that fill different ecological niches in streams, rivers, ponds and lakes. Unfortunately, the present status of the dozen native species of mussels is poor in most areas of the Basin, especially in flashy Piedmont streams and areas with impediments to fish passage. Poor status was judged by the reduced biodiversity, abundance, and range for this taxonomic group. Continued watershed development and climate change represent increasing threats. As long-lived organisms that are sedentary, mussels are a useful indicator of site-specific conditions over longer time scales, compared to short-lived or mobile fauna. The depleted status of natural mussel populations is concerning because of emerging data showing that mussel beds provide a diverse array of ecosystem services. For example, growing research is strengthening our understanding of the water quality benefits of healthy mussel assemblages and the economic basis for an increased restoration investment particularly at sites with impaired water quality. Careful watershed management, inclusion of mussels in traditional restoration, and more vigorous mussel conservation and restoration would help to reverse historic mussel declines and offset future threats. The few areas that still harbor healthy, diverse, and reproductive mussel beds, such as a few areas of the mainstem Delaware River, merit careful protection. Many areas that have lost mussel beds can now be restored or bolstered using new technologies. Mussel beds might also be included in man-made systems to provide ecosystem services, similar to treatment wetlands. The greatest improvements for water and habitat quality will likely be achieved by a basin-wide shellfish strategy that conserves and restores native bivalves living in different niches throughout the river-to-sea continuum.

References

Anderson, R.M. and D. A. Kreeger. 2010. Potential for impairment of freshwater mussel populations in DRBC Special Protection Waters as a consequence of natural gas exploratory well development. Written testimony to the Delaware River Basin Commission. November 23, 2010. 14 p. <http://www.delawareestuary.org/ science_reports_ other_groups.asp> (accessed 9/9/11).

Atkinson, Hopper, Kreeger, Lopez, Maine, Sansom, Schwalb and Vaughn. In Review. Gains and gaps in knowledge surrounding freshwater mollusk ecosystem services. Freshwater Mollusk Biology and Conservation.

Campbell, C.A. and B. St. John White. 2010. Development of an aquatic gap analysis for the Delaware River Basin. Gap Analysis Bulletin 17: 26-29.

Cheng, K. M. & D. A. Kreeger, 2015. Current Status and Restoration of Freshwater Mussels in Northern Delaware. Partnership for the Delaware Estuary, Wilmington, DE. PDE Report No. 15-04. <https://s3.amazonaws.com/delawareestuary/pdf/PDE-Report-15-04_ Current+Status+and+Restoration+of+Freshwater+Mussels+in+Northern+Delaware.pdf> (accessed 5/11/17)

Cheng, K. M. & D. A. Kreeger, 2017. Determination of Filtration Capacity and Pollutant Removal of Freshwater Mussels in Delaware Streams. Partnership for the Delaware Estuary, Wilmington, DE. PDE Report No. 17-04. (accessed 5/11/17)

Cheng, K. M., Haaf, L. R., Kreeger, D. A., O'Hara, B. and Gentry M. J., 2021. Enhancing Water Quality by Restoring Freshwater Mussels in an Urban River: Assessing and Testing Prospects in Relation to Salinity. Partnership for the Delaware Estuary, Wilmington, DE. PDE Report No. 21-02.

Freshwater Mollusk Conservation Society. 2016. A national strategy for the conservation of native freshwater molluscs. Freshw. Mollusk Biol. Conserv. 19: 1-21.

Gentry, M. J. K. M. Cheng, L. Morgan and D. A. Kreeger. 2022. Municipal stormwater ponds as low-maintenance, high-impact grow-out sites for juvenile freshwater mussels in the Delaware River Basin, USA. Abstract. J. Shellfish Res. 41(2)

Gray, M. and D. Kreeger. 2014. Monitoring fitness of caged mussels (Elliptio complanata) to assess and prioritize streams. Aquatic Conservation: Marine and Freshwater Ecosystems. vol. 24. pp. 218-203.

Gutiérrez, J. L., C. G. Jones, D. L. Strayer, and O. O. Iribarne. 2003. Mollusks as ecosystem engineers: the role of shell production in aquatic habitats. Oikos 101:79-90.

Haag, W. R. 2012. North American Freshwater Mussels: Ecology, Natural History, and Conservation. New York, NY: Cambridge University Press.

Hoellein, T. J., C. B. Zarnoch, D. A. Bruesewitz, and J. DeMartini. 2017. Contributions of freshwater mussels (Unionidae) to nutrient cycling in an urban river: filtration, recycling, storage, and removal. Biogeochemistry 135:307-324.

Kreeger, D.A. 2005. A holistic concept for the conservation and propagation of populations of freshwater, brackish and estuarine bivalves for ecosystem services. J. Shellfish Res. 24(2): 662

Kreeger, D., J. Adkins, P. Cole, R. Najjar, D. Velinsky, P. Conolly, and J. Kraeuter. June 2010. Climate Change and the Delaware Estuary: Three Case Studies in Vulnerability Assessment and

Adaptation Planning. Partnership for the Delaware Estuary, PDE Report No. 10-01. 1-117 pp. <https://s3.amazonaws.com/ delawareestuary/pdf/Climate/Climate%20Change%20and%20the%20Delaware%20Estuary%20&%20Appendicies_PDE-10-01.pdf> (accessed 5/11/17)

Kreeger, D.A., K. Cheng, P. Cole and A. Padeletti. 2014. Partnership for the Delaware Estuary. 2014. Reintroduction of Freshwater Mussels into the Red and White Clay Creeks, DE. PDE Report No.14-02. http://delawareestuary.s3.amazonaws.com/PDE-Report-14-02_Re-Introductionof-FreshwaterMusselsintoRedClayandWhiteClayCreeksDE.pdf> (accessed 5/11/17)

Kreeger, D.A., K. M. Cheng, M. J. Gentry, L. M. Morgan and J. A. Moody. 2022. Beyond restoration aquaculture: testing the propagation, rearing and deployment of freshwater mussels for water quality enhancement of impaired habitats. Abstract. J. Shellfish Res. 41(2).

Kreeger, D., P. Cole, K. Cheng, L. Butler, D. Ross, A. Padeletti, R. Thomas, J. D'Agostino, L. Wool. 2015. Freshwater Mussel Scientific Research and Community Involvement for Improved Water Quality in Urban Pennsylvania Coastal Zone Project. Partnership for the Delaware Estuary final report to the Pennsylvania Coastal Management Program. PDE Report No. 15-02. 41p. <http://delawareestuary.s3.amazonaws. com/PDE-Report-15-02_Freshwater%20Mussel%20Scientific%20Research%20and%20Community%20Involvement%20for%20Improved%20Water%20Quality%20in%20the%20Urban%20PA%20Coastal%20Zone. pdf> (accessed 5/11/17).

Kreeger, D., P. Cole, M. Mills, L. Butler, A. Padeletti, R. Thomas, J. D'Agostino. 2013. Connecting people to aquatic biodiversity: freshwater mussel surveys in Pennsylvania's coastal zone. Partnership for the Delaware Estuary final report to the Pennsylvania Coastal Management Program. PDE Report No. 13-02. 65 p. <https://s3.amazonaws.com/delawareestuary/pdf/ScienceReportsbyPDEandDELEP/PDE-Report-13-02_ Freshwater+Mussel+Surveys+in+Pennsylvania%27s+Coastal+Zone.pdf> (accessed 5/11/17).

Kreeger, D. A., C. M. Gatenby, and P. W. Bergstrom. 2018. Restoration potential of several native species of bivalve mollusks for water quality improvement in mid-Atlantic watersheds. Journal of Shellfish Research 37: 1121-1157.

Kreeger, D., R. Thomas, S. Klein, A. Padeletti, and W. Lellis. 2011. Recent discoveries of rare freshwater mussels (Unionidae) in the urban corridor of the Delaware Estuary. Proceedings for the Fourth Delaware Estuary Science & Environmental Summit. P. Cole and D. Kreeger (eds.), Partnership for the Delaware Estuary. PDE Report No. 11-01. 154 p. <http://www.delawareestuary.org/science_reports_partnership.asp> (accessed 9/12/11)

Kreeger, D. and J. Kraeuter. 2010. Ecologically significant bivalve molluscs of the Delaware Estuary, Appendix N, In: Climate Change and the Delaware Estuary: Three Case Studies in Vulnerability Assessment and Adaptation Planning. Partnership for the Delaware Estuary, Report No. 10-01. 23 p. <http://www. delawareestuary.org/science_reports_ partnership.asp> (accessed 9/9/11).

Kreeger, D. and A. Padeletti. 2011. Freshwater mussels: "old growth" survivors for "new age" restoration. Estuary News 21(4): 5-6.

Lellis, W.A. 2001. Freshwater Mussel Survey of the Delaware Scenic and Recreational River, Qualitative Survey 2000. Report to the National Park Service, February 20, 2001. 6 p. plus appendices.

Lellis, W.A. 2002. Freshwater Mussel Survey of the Delaware Water Gap National Recreation Area, Qualitative Survey 2001. Report to the National Park Service, March 8, 2002. 12 p. plus appendices.

McMahon, R. F. 1991. Mollusca: Bivalvia. In: J. H. Thorp and A. P. Covich (eds.), Ecology and Classification of Freshwater Invertebrates. Academic Press Inc., New York. pp. 315 397.

Morgan, L., M. Gentry, K. Cheng, J. Moody and D. A. Kreeger. 2022. Coupling freshwater mussel and shoreline restoration practices in the Delaware Estuary, USA. Abstract. J. Shellfish Res. 41(2)

Najjar, R. G. 2015, Potential impact of climate change on bivalves in the tidal freshwater region of the Delaware River. Final report to Pennsylvania Sea Grant. 23 p.

Neves, R.J. 1993. A state-of-the-unionids addres. P. 1-1- in K.S. Cummings, A.C. Buchanan, and L. M. Koch (eds.). Conservation and management of freshwater mussels. Proc. Upper Mississippi Riv. Conserv. Comm. Symp., Rock Island, IL.

Newton, T. J., S. J. Zigler, J. T. Rogala, B. R. Gray, and M. Davis. 2011. Population assessment and potential functional roles of native mussels in the Upper Mississippi River. Aquatic Conservation: Marine and Freshwater Ecosystems 21: 122-131.

Nobles, T., and Zhang, Y. 2011. Biodiversity loss in freshwater mussels: importance, threats, and solutions. Biodiversity Loss in a Changing Planet. Intech, Rijeka, 137-162.

Ortmann, A. E. 1919. A monograph of the naiades of Pennsylvania. Part III. Systematic account of genera and species. Memoirs of the Carnegie Museum. 8(1):1-384. Partnership for the Delaware Estuary. 2008. State of the Estuary Report. PDE Report #08-0. 36 p.

Pennsylvania DEP Report 2020. Qualitative Survey of the Spatial Extent of Freshwater Mussel Beds in the Pennsylvania Portion of the Tidal Delaware River using Side Scan Sonar Imaging and Underwater Video. http://www.depgreenport.state.pa.us/elibrary/GetDocument?docId=3420602&DocName=DELAWARE%20ESTUARY%20SIDE%20SCAN%20MUSSEL%20SURVEY.PDF%20%20%3cspan%20style%3D%22color:green%3b%22%3e%3c/span%3e%20%3cspan%20style%3D%22color:blue%3b%22%3e%28NEW%29%3c/span%3e%2010/1/2022# (Accessed on 8/22/2022)

Patterson, M.A., R.A. Mair, N.L. Eckert, C.M. Gatenby, T. Brady, J.W. Jones, B.R. Simmons and J.L. Devers 2018. Freshwater Mussel Propagation for Restoration, 1st Edition, Cambridge University Press, https://www.cambridge.org/core/books/freshwater-mussel-propagation-for-restoration/2373C012E661FF1533D3FB5082FC1ABA

Sansom, B. J., S. J. Bennett, J. F. Atkinson, and C. C. Vaughn. 2020. Emergent hydrodynamics and skimming flow over mussel covered beds in rivers. Water Resources Research 56: 1–17.

Sansom, B. J., J. F. Atkinson, and S. J. Bennett. 2018. Modulation of near-bed hydrodynamics by freshwater mussels in an experimental channel. Hydrobiologia 810: 449–463.

Strayer, D. L., J. Geist, W. R. Haag, J. K. Jackson, and J. D. Newbold. 2019. Essay: Making the most of recent advances in freshwater mussel propagation and restoration. Conservation Science and Practice 1. 10.1111/csp2.53

Thomas, R., D. Kreeger, S. Klein, A. Padeletti, Z. Ruge, M. Gray and C. Gatenby. 2011. Occurrence of freshwater mussels (Unionidae) in surveyed streams of southeastern Pennsylvania, 2000-

2010. Proceedings for the Fourth Delaware Estuary Science & Environmental Summit. P. Cole and D. Kreeger (eds.), Partnership for the Delaware Estuary. PDE Report No. 11-01. 154 p. <http://www.delawareestuary.org/science_reports_ partnership.asp> (accessed 9/12/11)

Vaughn, C. C. 2018. Ecosystem services provided by freshwater mussels. Hydrobiologia 810: 15-27.

Vaughn, C. C., S. J. Nichols, and D. E. Spooner. 2008. Community and foodweb ecology of freshwater mussels. Journal of the North American Benthological Society 27: 409-423.

Wang, N., C. G. Ingersoll, D. K. Hardesty, C. D. Ivey, J. L. Kunz, T. W. May, F. J. Dwyer, A. D. Roberts, T. Augspurger, C. M. Kane, R. J. Neves, and M. C. Barnhart. 2007. Acute toxicity of copper, ammonia, and chlorine to glochidia and juveniles of freshwater mussels (Unionidae). Environmental Toxicology and Chemistry 26:2036-2047.

Williams, J. D., M. L. Warren, K. S. Cummings, J. L. Harris, and R. J. Neves. 1993. Conservation status of freshwater mussels of the United States and Canada. Fisheries 18: 6-22.

Williams, J. D., Bogan, A. E., Butler, R. S., Cummings, K. S., Garner, J. T., Harris, J. L., Johnson, N. A. and Watters, G. T. 2017. A revised list of the freshwater mussels (Mollusca: Bivalvia: Unionida) of the United States and Canada. Freshwater Mollusk Biology and Conservation, 20(2): 33-58.

Wood, Bukaveckas, Galbraith, Gattis, Gray, Ihde, Kreeger, Mair, McLaughlin, Hahn, and Harvey. 2021. Incorporating Freshwater Mussels into the Chesapeake Bay Restoration Efforts. Chesapeake Bay Workshop Publication. Annapolis, MD.

7.1.5 Macroinvertebrates

Introduction

Freshwater benthic macroinvertebrate communities, home to a diverse group of organisms such as those shown in Figures 7.1.5.1A and 7.1.5.1B, are useful indicators of the ecological integrity of the Delaware River watershed for several reasons. A variety of macroinvertebrates live in every aquatic environment, and they are functionally important in several ecological roles. They are widely acknowledged to be good indicators of water quality because they are directly impacted by and highly sensitive to changes in water quality. Furthermore, they have been studied extensively in all parts of the Delaware River Basin.

Figure 7.1.5.1 (Top) Newly emerged adult dragonfly (photo credit PA DEP); (bottom); *Drunella* spp (photo credit NJ DEP).

It is difficult to aggregate and summarize data about this indicator for a multi-state area like the Delaware Estuary and Basin because the various organizations that produce data (including state environmental agencies) all use different methods of sampling and analysis. The differences in methods limit the comparability of data from different sources. Therefore, data are compared using the common approach for all states, grades of condition (e.g. good, fair, poor). Assuming a rough comparability between these grades of condition, data from various sources can be brought together and presented side-by-side to approximate a basin-wide assessment.

An explanation of how this complex situation came about may help explain what this indicator tells us about the ecology of the Delaware River Basin broadly. The discussion may also help readers to appreciate something about benthic macroinvertebrates and their importance, and to understand more about the way environmental agencies perform water quality management in the United States.

Description of Indicator

The word "benthic" indicates animals that live on, or in, the substrate at the bottom of a waterbody. The word "macroinvertebrates" designates invertebrate animals that are large enough to be seen without the aid of magnification. In aquatic habitats, benthic macroinvertebrates are a broad group of organisms representing several phyla. The group includes roundworms, flatworms, mollusks, and several kinds of arthropods. Insects are a particularly important class of animals in the group, because of their abundance and diversity in the freshwater biota. Some aquatic invertebrates that are commonly encountered are dragonflies, mayflies, mosquitoes, and water striders.

To be more precise, the indicator being discussed here is freshwater benthic macroinvertebrates that live in streams. Thus, those macroinvertebrates that live in lakes, ponds, wetlands, and tidal waters are excluded. These distinctions are primarily made because the nature of the information most easily available, is mostly for "wadable" streams. Wadable streams are relatively easy to survey, and these smaller waterbodies are where most states have focused their sampling efforts (Fig 7.1.5.2). Most states have been sampling and compiling data about benthic macroinvertebrates since the 1970s or 1980s. The reason lies in what these animals say about the water quality of the environments in which they live. Using a procedure called "bioassessment," the biological condition of macroinvertebrate communities is analyzed to provide information about pollution and other water quality problems. A bioassessment protocol is a set of standard practices describing how streams should be surveyed to produce data about ecological condition.In most states, bioassessment is used for multiple purposes, but the most widespread application of bioassessment is for the purpose of assessing a state's streams for the attainment of water quality standards. This program of assessment arises from the states' obligations under the Federal Clean Water Act.

Figure 7.1.5.2 Scientists sampling for macroinvertebrates in New Jersey (left) and Pennsylvania (right). Photo credits NJ DEP and PA DEP respectively.

The Federal Clean Water Act (and its amendments through 1987) requires states to develop water quality monitoring programs. States report to the U.S. Environmental Protection Agency (USEPA) on the quality of their waters using the biennial "305 (b) report" and the "303 (d) list." In most states, these biennial reports are now usually merged into a single document called the "Integrated Assessment" or the "Integrated Report." The states are charged with assessing their waterways' conditions for various water uses, including, for example, public water supply, recreation, or aquatic life. The condition of macroinvertebrate communities is usually connected specifically to aquatic life uses. Results of bioassessments are used to determine if a waterway is "attaining" or "not attaining" the State's water quality standard, a threshold condition determined by the state.

Over the past 20 to 30 years, bioassessment has become increasingly important to environmental agencies, as advances have been made in the scientific understanding of watershed degradation and its effects. It is now widely acknowledged that biological indicators represent an essential means of determining the condition of natural waters. Some of the reasons for this are:

- Bioassessments provide information that is directly relevant to the goals of water pollution law (that is, that waters should be able to support aquatic life).

- Bioassessments provide information about long-term, chronic, or episodic stressors that are otherwise difficult to monitor.

- Bioassessment methods can be used to assess fish or periphyton (algae) in addition to macroinvertebrates. However, macroinvertebrates may be the most broadly useful of these biological groups, for reasons that include the following:
    - Macroinvertebrates are relatively easy to sample and analyze,
    - Macroinvertebrates are less mobile than fish, and thus they provide a better representation of the condition of a particular location, and
    - Macroinvertebrates are abundant and utilize diverse niches, which allows for a detailed determination of their condition over a wide gradient.

Methods of collection and analysis must be standardized and consistently applied if data are to be comparable. However, there is no single macroinvertebrate protocol that is universally applicable in all circumstances. Natural variation sometimes dictates that protocols should differ, for the assessment of streams from substantially different environments. In addition, the needs and resources of the organization doing the sampling sometimes determines what protocol will be applied, since there are some protocols that demand more time and resources, while others can be done more rapidly. While there are broad similarities between many of the protocols, they usually differ from one another in their various details. A brief discussion of some of the variables will illustrate the reasons for this complexity. Every macroinvertebrate bioassessment protocol must include a description of each of the steps listed below. Within each of these four steps, there can be variations in methodology, as indicated by the following discussion:

1. Sampling: According to most protocols for wadable streams benthic macroinvertebrates should be sampled using hand-held nets. The bioassessment protocol specifies details such as the exact shape of the net, the size of the mesh, and how the net should be handled in a stream. The protocol describes how to select sampling sites in the field and how to combine the material from grab samples to make a composite. The protocol further specifies how many organisms are needed to make a representative sample (typically between 100 and 300 individuals), and provides techniques for ensuring that those organisms are picked from the sample using an unbiased randomization method.

2. Identifying organisms: The bioassessment protocol specifies whether a collection of organisms will be identified in the field and returned to the stream alive or preserved and identified in a laboratory. Field methods usually involve family-level identification, while laboratory methods often provide for identification to genus or to species. Laboratory analysis requires more time and effort but provides more information. Whether the identification is done in the field or the lab, the product of this step is a list of the macroinvertebrate taxa found at a site, along with the number of individuals of each taxon.

3. Applying bioassessment metrics: The list of organisms produced in the previous step is analyzed by applying bioassessment metrics. This involves various methods of grouping and counting the organisms by types (by taxa). A variety of bioassessment metrics have been presented in scientific literature. Some metrics involve counting the number of different taxa found in a sample (assessing sample diversity); while other metrics involve counting the number of individuals of certain taxa or in certain groups of taxa (assessing community structure). Applying metrics often requires grouping taxa together by what is known about their ecological roles or characteristics. For example, there are several commonly-used metrics that take into account the relative "pollution tolerance" of the various taxa. Applying any metric to the list of taxa for a sample produces a numerical score. It is generally agreed that no single metric

provides enough information to stand alone as a means of assessing water quality. Therefore, most states apply a suite of several metrics.

4. Applying an index: An Index of Biological Integrity (IBI) is a method of combining and integrating information from several bioassessment metrics. It involves applying a series of mathematical transformations to each sample's metric scores and then combining them to give a single numerical index score. Typically, an index score for the so-called "reference condition" is developed using data from sites that are known to be undisturbed and that are judged to be appropriate reference sites based on regional and ecological considerations. Sample data are compared to reference conditions using the numerical scores calculated using the index. Increasing degrees of disturbance (or pollution) are indicated by scores that range further and further from the reference score. For state agencies, one of the main purposes of their bioassessment work is to identify those streams that are divergent enough from the reference condition that they are determined to be "not attaining" the state's water quality standards for aquatic life use. Typically, the threshold that is used to determine attainment is linked to a particular numerical score using the appropriate index.

The "Present Status" and "Past Trends" sections of this chapter are based on data from six different sources, namely the four Delaware River Basin states, Delaware River Basin Commission (DRBC), and the Academy of Natural Sciences of Drexel University. These six organizations all use different macroinvertebrate protocols in their programs for stream assessment. In addition to this interstate variability, there is also intrastate variability, because some states actually use more than one protocol to account for natural variation. A brief description is provided of how each of the organizations that contributed data has designed their respective programs for producing macroinvertebrate data.

Delaware Delaware is a small state with relatively little natural variability, but it does straddle a significant eco-regional divide. Delaware's land area is divided between the Middle Atlantic Coastal Plain eco-region and the Northern Piedmont eco-region. In the Coastal Plain, where streams have a low-gradient character, the state's bioassessment program specifies the use of the protocol developed by a USEPA-sponsored, multi-state workgroup called the Mid-Atlantic Coastal Streams Workgroup (USEPA 1997). In the Piedmont, the state specifies the use of methods documented in USEPA's 1999 Rapid Bioassessment Protocols report (Barbour et al. 1999). The structural and ecological differences between coastal plain streams and piedmont streams dictate several differences between the two protocols. For both stream categories, Delaware specifies that macroinvertebrate samples are to be preserved and identified in a laboratory, with most taxa identified to genus. Both protocols also utilize a multi-metric index. Of the assessment stations that make up the data set for Delaware's Delaware Estuary Basin, 46% are from the Piedmont and 54% are from the Coastal Plain.

Pennsylvania In 2006, after 10 years of effort, Pennsylvania completed their first statewide bioassessment survey, which was done using a modified version of the USEPA Rapid Bioassessment II Protocol from the document referenced above (Barbour et al. 1999). This method used field identification of organisms and family-level taxonomy. At about the same time, the state decided to refine their biomonitoring program and implement major changes to the bioassessment protocols. Pennsylvania's new program has the State's streams divided into four major ecological categories, each of which is assessed by a different data collection protocol and assessment method. Each collection protocol specifies particular sampling requirements, and each assessment method specifies how metrics and index calculations should be applied.

The largest group of streams in Pennsylvania is categorized as wadable riffle-run streams, which are assessed using the "Wadable Freestone Riffle-Run Stream Macroinvertebrate Assessment Method" (Shull 2017). The method specifies making a certain number of collections from shallow gravel-bottom or cobble-bottom riffle habitat, and then compositing and randomly sub-sampling to give a 200-organism sub-sample. The sub-sample is preserved and identified in a laboratory to genus, and a multi-metric IBI is applied to the taxa list. The preferred seasons for sampling are between November and May, so as to avoid sampling during the summer emergence period of many important insects. However, a method for summer samples is also available when agency workload requires that stream assessments continue through the summer months. The summer samples method provides a modified analysis to account for the effects of seasonal emergence on the invertebrate community. During the summer months, many insects emerge as winged adults, and their aquatic forms are notably absent from stream-collected samples. In light of this, practitioners of bioassessment have two choices. They may avoid sampling during the time of year when the benthic community is likely to be altered by emergence, or they may develop protocols that are specifically tailored to each particular seasonal condition. Freestone Streams account for 95% of the assessments performed in Pennsylvania's Delaware River Basin.

Pennsylvania's second stream category consists of low-gradient streams that are lacking in riffle habitat. Pennsylvania uses the phrase "Multi-Habitat" to refer to this stream category and method. For Multi-Habitat sites, the sampling protocol is designed to provide a means of capturing representative organisms from several specific kinds of habitats (including, for example, coarse submerged debris, submerged aquatic vegetation, and deposits of coarse particulate organic matter). A specific multi-metric analysis and IBI are applied for assessments (Pulket 2017). This category is somewhat similar to the Mid-Atlantic Coastal Plain Streams "Coastal Plain" streams discussed above in the "Delaware" section, as well as to the "Coastal Plain (Non-Pinelands)" category discussed below in the "New Jersey" section. However, the analogy is not exact, because many of Pennsylvania's Multi-Habitat sites are not in the coastal plain but in low-gradient topography in plateau regions, such as the Pocono region of northeast Pennsylvania. Multi-Habitat assessments account for 4% of the assessments performed in Pennsylvania's Delaware River Basin.

The third category of streams, limestone streams, is assessed using the method for "True" Limestone Streams (Williams 2017). This method is specifically for spring-fed streams with high alkalinity and constant year-round temperature. These streams are considered ecologically unique and are important as cold-water fish habitat. The protocol specifies the collection of two samples from riffle habitat, composited and sub-sampled to make a 300-organism sample, followed by laboratory-identification of organisms to genus. A specific multi-metric analysis and IBI are applied. Limestone Streams account for 1% of the assessments performed in Pennsylvania's Delaware River Basin.

Pennsylvania's fourth stream category includes the large semi-wadable rivers. Semi-wadable rivers are defined as predominantly free-flowing systems with drainage areas >1,000 mi^2 and have physical characteristics that allow for riffle and run sections to occur with relative frequency. These river systems tend to lack a well-defined and navigable U-shaped channel for any significant distance and frequently present difficulties for both wadable and non-wadable macroinvertebrate data collection protocols. Well over half of the large rivers within Pennsylvania are considered semi-wadable. Pennsylvania uses the "Semi-Wadable Large River Macroinvertebrate Assessment Method", which applies a multi-metric IBI to make assessments, much like the previous methods discussed (Shull 2018). Samples from large semi-wadable rivers are not reported here because they constituted less than 1% of the total assessments performed in Pennsylvania's Delaware River Basin.

New Jersey In the early 1990's, New Jersey began its Ambient Macroinvertebrate Network (AMNET). Each station within this network of freshwater, non-tidal, wadable rivers and streams is sampled once every five years using a rotating basin approach (NJDEP 2007). Assessments are made utilizing one of three multi-metric indices developed for use in New Jersey. The three indices are: the High Gradient Macroinvertebrate Index (HGMI), which applies to the streams of northern New Jersey in the Highlands, Ridge and Valley, and Piedmont ecoregions; the Pinelands Macroinvertebrate Index

(PMI), which applies to rivers and streams within the Pinelands National Reserve and 5-km buffer surrounding this boundary; and the Coastal Plain Macroinvertebrate Index (CPMI), which applies to the Coastal Plain of southern New Jersey excluding the Pinelands National Reserve and buffer. For New Jersey's most recent sampling in the Delaware River Basin, 37% of stations were assessed by the HGMI, 42% by the CPMI, and 21% by the PMI.

New York New York's biological monitoring program began in 1972, with the first surveys done on the state's large rivers, using artificial substrate samplers. Since 1984, New York has used a "Rapid Assessment" method in the state's wadable streams, for both special studies and as part of the statewide ambient water quality monitoring program. In 1987, the statewide program was re-designed to use a rotating cycle of monitoring and assessments called Rotating Integrated Basin Studies (RIBS). Under the current RIBS schedule, chemical and biological monitoring is conducted in all of the state's 17 major drainage areas over a five-year period (Bode et al. 2002). Riffle habitat is targeted for biological sampling of wadable streams. Non-wadable waters are monitored using artificial substrate samplers. The index period for wadable stream sampling is from July through September. Individual metrics characterizing the benthic macroinvertebrate community are combined to form a multi-metric index called the Biological Assessment Profile. There is no differentiation of streams by eco-region; however, modification of the sampling methods and assessment metrics are used for low-gradient, sandy- bottom streams. Samples are preserved and identified in the laboratory to genus or species.

DRBC As an interstate agency, DRBC takes responsibility for assessing the mainstem Delaware River where it forms a border between states. Since 2001 DRBC has collected benthic macroinvertebrate samples annually at about 25 fixed sites on the Delaware River. These sites range from Hancock, NY (river mile 331/533 km) to just above the head-of-tide at Trenton, NJ (river mile 137/220 km). All samples are collected from gravel- or cobble-dominated riffle habitats. Sampling generally occurs in the late summer, with the central sampling window being August and September. The samples are preserved for laboratory identification, and the organisms are generally identified to genus. The analytical methodology is based on a multi-metric IBI with a 100-point range.

The Academy of Natural Sciences of Drexel University (ANSDU) collects data for the **Delaware River Watershed Initiative (DRWI**; 4states1source.org), among other projects. ANSDU has coordinated and performed monitoring for the DRWI around on-the-ground restoration and protection projects since 2013. Projects take place in sub-watershed "clusters" in 8 geographies across the basin: Brandywine-Christina (BWC), Middle Schuylkill (MS), Schuylkill Highlands (SH), Upper Lehigh (UL), Poconos-Kittatinny (PK), New Jersey Highlands (NJH), Kirkwood-Cohansey [Aquifer], and Upstream Suburban Philadelphia (PHL). The sampling sites and methods are designed to capture the potential effects of stream recovery from implementation of agricultural Best Management Practices (BMPs) and land preservation. The quantitative sampling methods were applied with the goal of tracking small changes over time. For example, when sufficient aggregation of BMPs leads to nutrient and sediment reductions, the data are a fine enough resolution to show stream ecosystem response (Kroll & Abell, 2015; Kroll et al., 2019).

ANS and Stroud Water Research Center (SWRC) field teams collected baseline data at three types of sites. Integrative sites capture larger areas of the cluster drainage and were sampled to characterize the overall condition of sub-watershed clusters, sampled in 2013 and 2015. Baseline sampling occurred in 2014 and 2016 at project sites – upstream and downstream of on-the-ground projects, where streams are smaller and improvement of degraded waters or maintenance of good conditions might be expected as a result of DRWI activities. The third site type represents Focus Areas in 2017-2020—smaller regions within sub-watershed clusters where on-the-ground work is focused. The Focus Area sites are typically located at the watershed outlets of the Focus Area, or the "pour points." Sampling follows the same protocols at all sites, producing comparable and complementary data that over time can give a comprehensive picture of ecosystems within the basin at different spatial scales. For the DRWI, fish, macroinvertebrate and diatom communities are assessed, along with habitat assessment and water chemistry (Kroll et al., 2021). Macroinvertebrate data are collected by Stroud Water Research Center (SWRC) from April to June. Surber samplers are used to collect macroinvertebrates at 16 riffle locations in each site and composited into 4 samples. A portion of each of the 4 samples is taken to the laboratory and identified to genus or species. The data are presented using the PA DEP IBI here to align with other data sets.

Present Status

For this Technical Report, the status of macroinvertebrates in nontidal waterways of the Delaware Estuary and Basin is determined using the data produced by the States for their biennial water quality reporting. All four basin states and DRBC report results of water quality monitoring to USEPA for the biennial 303(d) list, sometimes called the Integrated List of Waters, or the Integrated Assessment. For this Technical Report, the states have provided the most recent bioassessment data they were able to share. Some state-by-state details are given in the sections below, and in the accompanying figures.

Delaware

Present status is given by data from 87 individual assessments, performed between 2006 and 2009. Four grades of condition were reported: excellent condition, good condition, moderately degraded, and severely degraded. These grades were translated to excellent, good, fair, and poor for consistency with other states. The aggregated data are presented in Figure 7.1.5.3.

Pennsylvania

Present status is given by data from 1,993 assessments, spanning more than 20 years of time. Four grades of condition are reported: excellent, good, fair, and poor. The aggregated data are presented in Figure 7.1.5.4.

New Jersey

The Ambient Macroinvertebrate Network (AMNET) has produced several rounds of survey results for each of the state's major basins. Four condition categories are used: excellent, good, fair, and poor. Present status is given by data collected since the previous report (Burke and Bright 2012). This includes two sampling events at stations in the Delaware River Basin: 2011-2013 (264 stations) and 2016-2018 (270 stations). The aggregated data are presented in Figures 7.1.5.5 and 7.1.5.6, respectively.

New York

Present status is given by data from 107 stations, collected from 2014 -2019. Four grades of condition are reported: non-impacted, slightly impacted, moderately impacted, and severely impacted. These grades were translated to excellent, good, fair, and poor for consistency with other states, however in this report, no sites in the basin scored "poor". The aggregated data are presented in Figure 7.1.5.7.

Delaware River Watershed Initiative (DRWI) data show a variety of water quality levels throughout the sub-watershed clusters. The assessments are based on 564 samples from 185 sites. The aggregated data are shown in Figure 7.1.5.8.

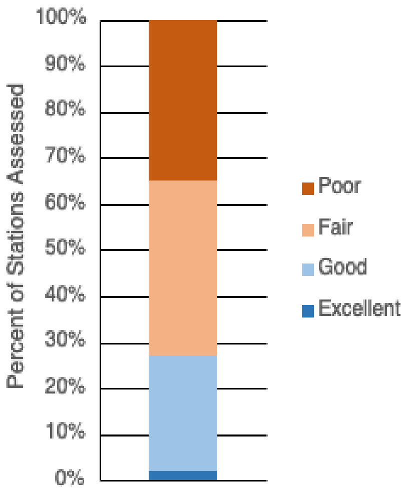

Figure 7.1.5.3 Bioassessment station data for Delaware's Delaware River Basin from 87 individual assessments, performed between 2006 and 2009.

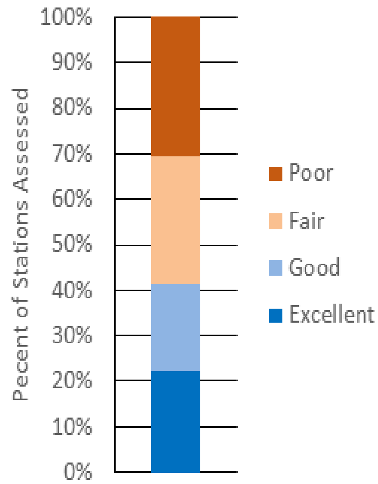

Figure 7.1.5.4 Bioassessment station data for Pennsylvania's Delaware River Basin for 1,993 assessments.

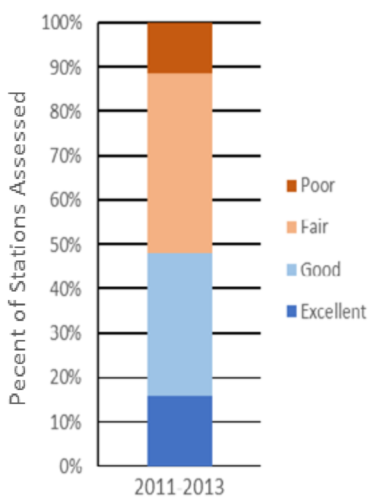

Figure 7.1.5.5 Bioassessment station data for New Jersey's Delaware River Basin, AMNET program, 2011-2013.

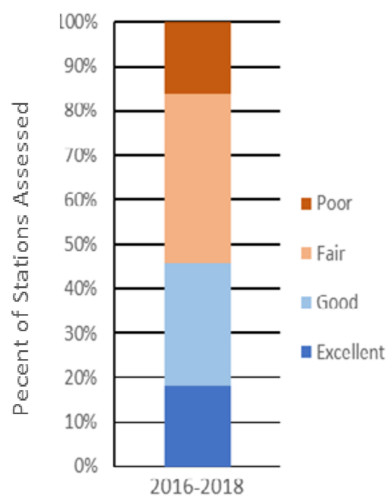

Figure 7.1.5.6 Bioassessment station data for New Jersey's Delaware River Basin, AMNET program, 2016-2018.

All data from the four basin states and the DRWI were compiled to evaluate biological condition at a basin-wide scale (Fig 7.1.5.9). Discrepancies in field and analytical methods between the basin states complicate direct comparison of IBI scores, however a qualitative comparison of the condition of macroinvertebrate communities is possible. Each state categorizes its biological assessments into four qualitative categories that describe the biological condition of the stream. As seen above, slight differences in terminology exist between the states so conditions were mapped to the following categories: Excellent, Good, Fair, and Poor. A biological condition was then calculated at the HUC12 sub-watershed scale by assigning values to the qualitative scores (Excellent=4, Good=3, etc.) and averaging the scores. Each HUC 12 sub-watershed was color coded based on condition (Excellent = 3.5-4, Good = 2.5-3.5, Fair = 1.5-2.5, Poor = 1-1.5).

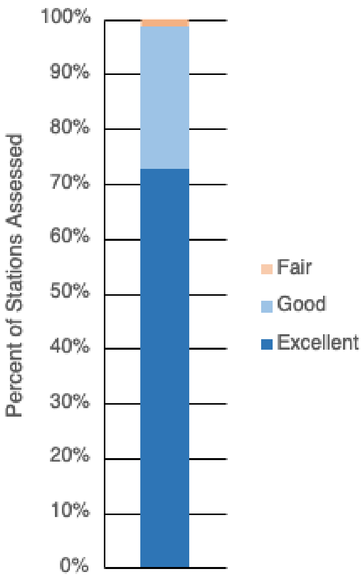

Figure 7.1.5.7 Bioassessment station data for New York's Delaware River Basin sites, 107 stations, collected from 2014-2019.

An important assumption of this analysis is that each state's qualitative interpretation of condition is the same (i.e., Excellent in PA describes a similar condition as Excellent in NJ). In reality, there may be differences in how each state categorizes its results which may lead to some bias in visual interpretation of the map. Additionally, the amount of available data can bias the results. For instance, in NY, each sub-watershed often contains only a single sampling point, which ultimately defines the condition for the entire unit.

Considering the Delaware River Basin as a whole, there may be some broad regional conclusions that can be drawn from the bioassessment data (Fig 7.1.5.9). New York is the state with the lowest percentage of low-scoring stations, and apparently the best overall condition. Delaware is the state with the highest percentage of low-scoring stations; and New Jersey and Pennsylvania are in

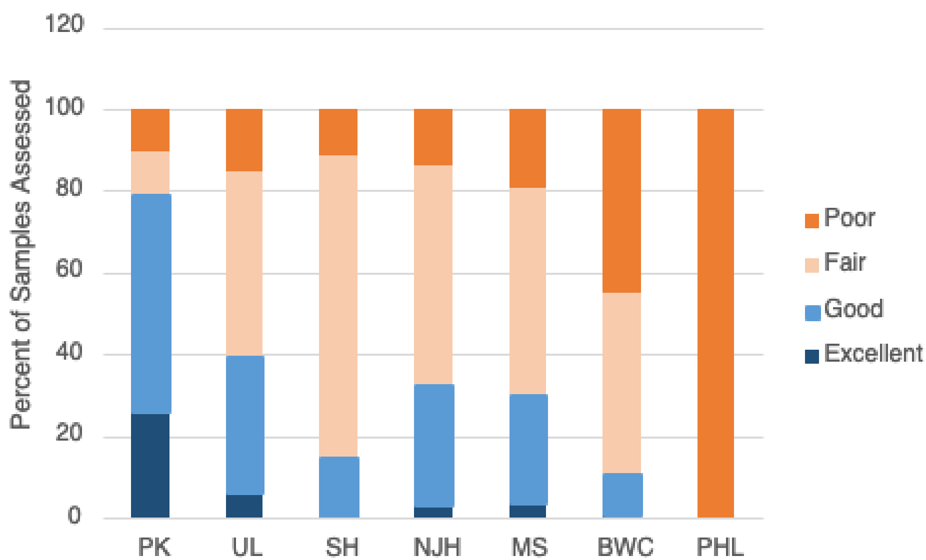

Figure 7.1.5.8 Bioassessment station data for the DRWI's Delaware River Basin samples based on 564 samples from 185 sites.

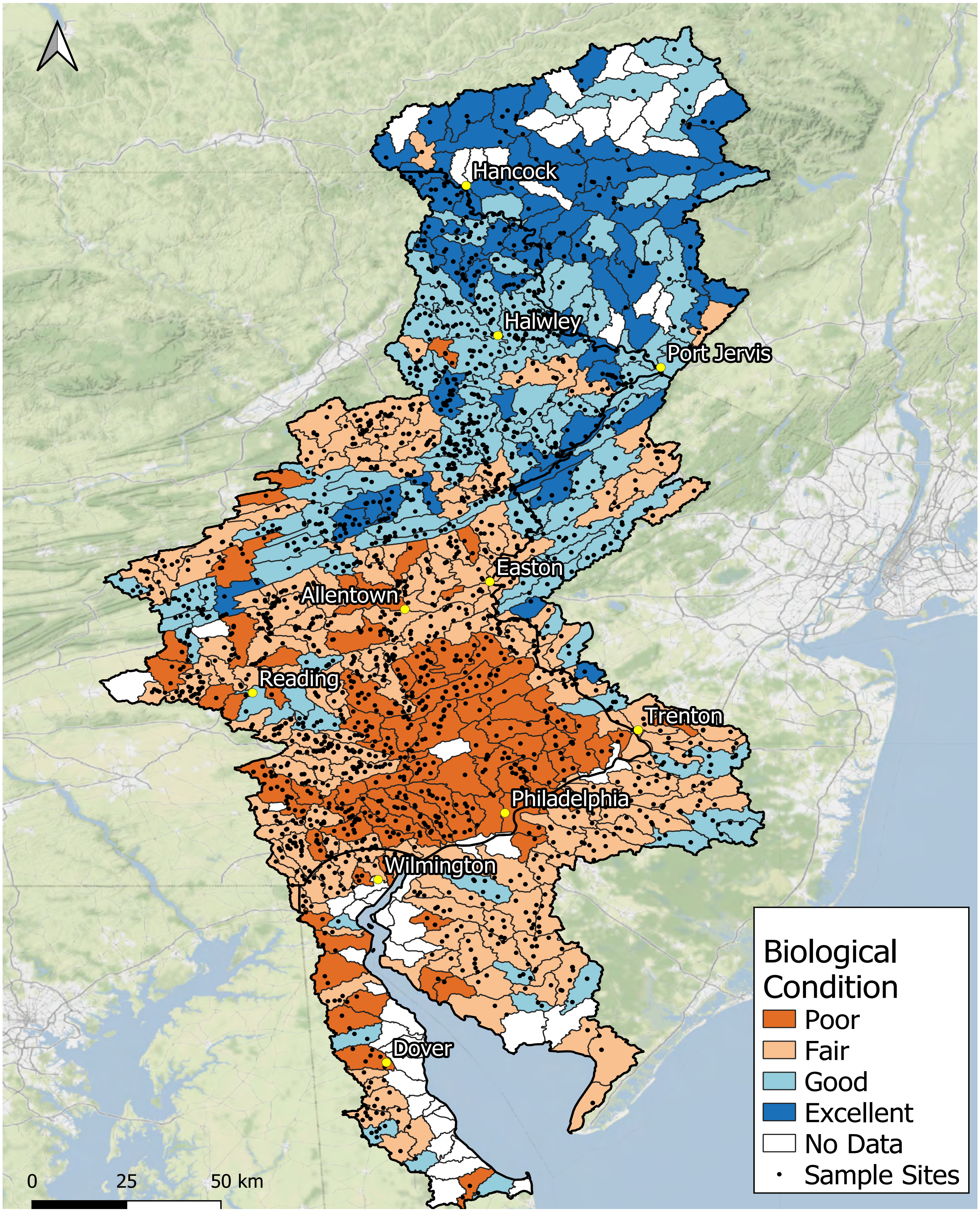

Figure 7.1.5.9 Biological condition complied using data from NJ, DE, NY, PA and the DRWI across the Delaware Estuary Basin.

between. These observations suggest that the condition of benthic macroinvertebrates is generally better in the upper portions of the Delaware River Basin, farther from the coast, and closer to "headwaters." This corresponds to what may be expected based on a general understanding of water quality problems in this Basin. Good water quality is generally expected (hence macroinvertebrate quality) to correlate negatively with urban land cover, which is mostly in the Lower Basin, and positively with forested land cover, which is mostly in the Upper Basin.

The data suggest the above conclusions as though there were a basin-wide survey, however this is not exactly the case. The data presented in this report, particularly for the states of Delaware and Pennsylvania and for the DRWI, may not represent a random selection of sites, as would have been ideal if this had truly been a basin-wide survey of ambient conditions. In Pennsylvania this is due to the fact that the state has not yet completed a full survey of the Basin using their revised bioassessment protocol. In Delaware, the available data is skewed towards lower-quality waterways, which were prioritized for monitoring. For the DRWI, sites are chosen relative to on-the-ground projects, and are sampled to show if improvements can be observed with aggregation of these projects.

Benthic macroinvertebrate community condition is affected primarily by water quality and habitat disturbance. There are many reasons why conditions at a particular site may appear to be degraded. Furthermore, the Basin being discussed is large and diverse. For these reasons, it would probably be inappropriate to draw further conclusions from the data presented. When biomonitoring results cause a state agency to list a stream as "impaired," the agency is supposed to attribute the impairment to a "source" and a "cause." The Integrated List for each state contains information about these "source" and "cause" determinations for each listing, but the terminology that is used is complex. Because of this complexity, an attempt was not made to gather or analyze "source" and "cause" information for the present report. Readers who are interested in examining the sources and causes of impairments listed by the states are referred to the Integrated List documentation for Pennsylvania, New Jersey, and New York.

Past Trends

Monitoring of trends is one of the stated goals of the biomonitoring program in most of the states. However, trend detection is more easily said than done. Reporting trends is difficult at the present time because of the nature of the available data; in Delaware and Pennsylvania, sufficient data was not obtained to present any kind of trend. Several more years of work will be necessary before meaningful time series will be generated for Pennsylvania and Delaware. We can discuss trends for New Jersey, New York, and for the mainstem Delaware River (DRBC data), based on the collected data.

New Jersey New Jersey's AMNET program has a dataset spanning over 20 years in the Delaware River Basin, with sampling conducted roughly once every 5 years. Data from three successive rounds of the AMNET program spanning 1995-2018 are presented. Data presented here shows 5 rounds completed to date. While there are fluctuations in the percentage of sites in each of the 4 condition categories round-to-round, no discernible trend is observed (Fig 7.1.5.10). In each round, the highest percentage of sites is in the fair category, with the majority being in the good-fair range. Also, most often, the lowest percentage of sites is in the poor range.

New York Over the years, New York has collected multiple rounds of data for a certain number of stations in the Delaware River Basin. In 2004, the state published a report entitled "30-Year Trends in Water Quality of Rivers and Streams in New York State Based on Macroinvertebrate Data, 1972-2002." (The report is available online at https://nysl.ptfs.com/data/Library1/98990.pdf). That report compared the results of surveys conducted between 1992 and 2002 to an earlier set of data collected before 1992.

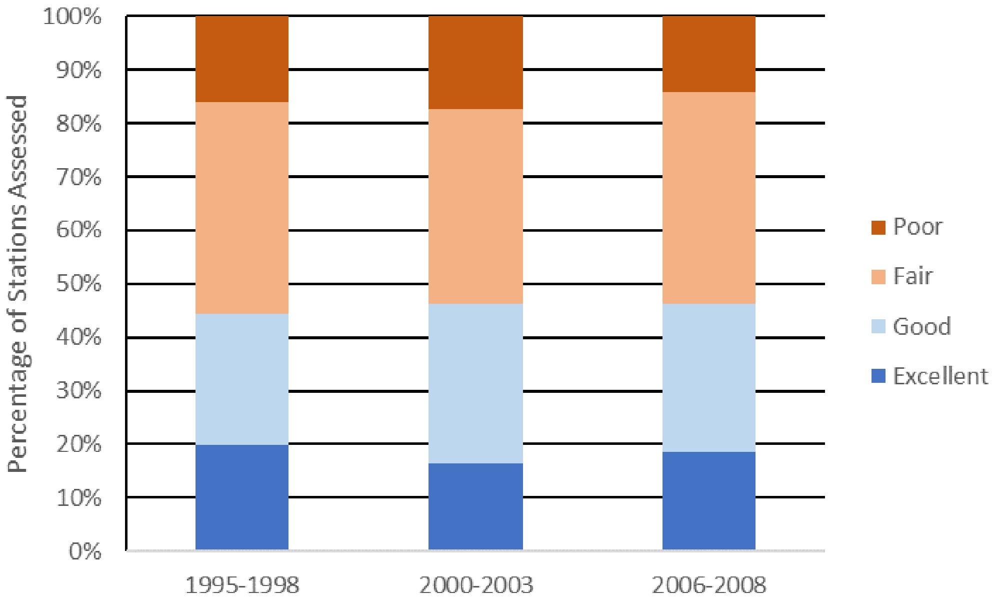

Figure 7.1.5.10 Bioassessment data for three successive rounds of New Jersey's AMNET program in the Delaware River Basin, 1995-2008.

For the present report, the recent data (2003 – 2010) was compared to the data from the 1990s that appears in the state's "30-Year Trends" report. The comparison reveals that the changes that occurred from the 1990s to the 2000s were very small. The total number of stations with assessment data in both decades was 37. Of those, 28 scored the same both times, while 9 scored differently. Five stations changed from "non-impacted" to "slightly impacted," and four others changed from "slightly impacted" to "non-impacted." Thus, the overall difference in the Basin appears to be very small. Figure 7.1.5.11 presents this comparison as a chart.

DRBC Because DRBC's sampling team has returned to the same stations for several years on a regular basis, their data set appears to offer an opportunity to look at bioassessment data in a time series. Some of these data are presented as a chart in Figure 7.1.5.12. Data is presented from 25 stations, collected between 2008 and 2017. Stream condition is shown as the percentage of sites meeting DRBC's IBI threshold for attainment of aquatic life use. Sites are grouped by DRBC Water Quality Zone and year to visualize trends spatially and temporally. Spatially, macroinvertebrate communities in the upper river (Zone 1a) almost always indicate high water quality. Moving downstream, the proportion of sites meeting DRBC's IBI threshold for attainment of aquatic life use decreases. Temporally, there is year-to-year variability, but it appears that there are no clear trends.

DRBC's technical staff believe that some of the variability observed here can be attributed to particular events or conditions. It is thought that a severe summer drought or a major flood can affect aquatic life enough to produce anomalous scores using the bioassessment metrics and index.

Future Predictions

The future condition of the benthic macroinvertebrates in the Delaware River Basin can be expected to follow the various causes of waterway impairment. Any attempt to project future conditions in the Basin would be speculative, particularly in light of the challenges of determining past trends from macroinvertebrate data.

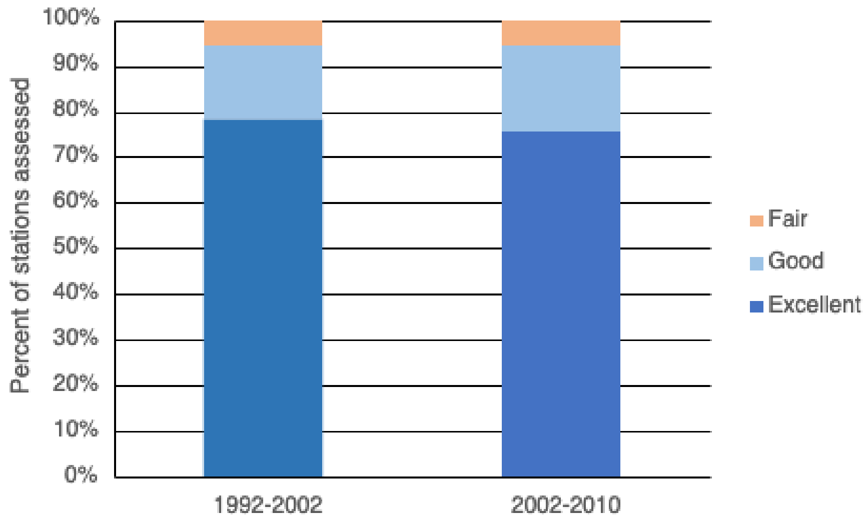

Figure 7.1.5.11 Bioassessment station data for New York's Delaware River Basin sites. The same 37 sites were sampled in each time period, 1992-2002 and 2002-2010.

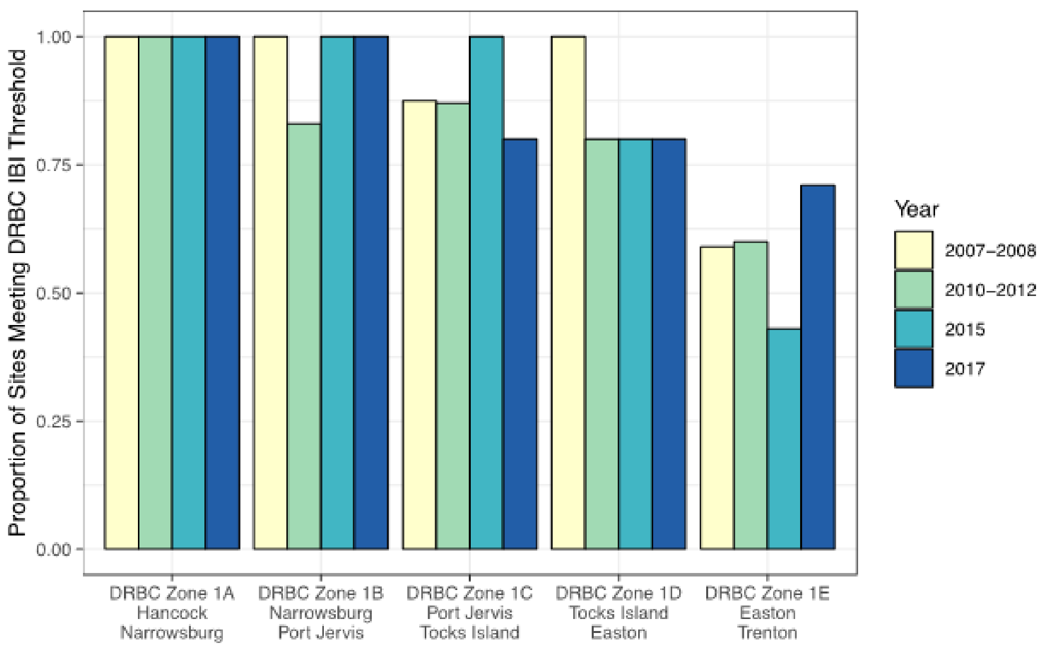

Figure 7.1.5.12 DRBC bioassessment data for mainstem Delaware River by Water Quality Zone from 2007 - 2017.

Actions and Needs

The Delaware River Basin has received increased investments to maintain and improve water quality and ecosystem health over the past decade through private and public funding sources. Most notably, the Delaware River Basin Act allocates funding for restoration, protection, monitoring and research through congressional approval each year. Nonetheless, the data and associated information indicate that urban areas are still impacted by Combined Sewage Overflows and a lack of tertiary treatment of wastewater to remove nutrients. Non-point source pollution, especially runoff of stormwater and nutrients, continues to affect streams of all sizes in the watershed, but several federal, state and regional programs are working to reduce these inputs. Changing development patterns have led to increased populations and associated urban lands in forested areas throughout the Basin. Therefore, attention must continue to be paid to changing water quality and ecosystem health, and measures to protect and restore the Basin's waterways are needed to ensure these natural resources remain intact for future generations.

Bioassessment of macroinvertebrates is a well-established practice in state environmental agencies, and it may be expected to continue for the foreseeable future. Bioassessment has become a core element of the regulatory system for protecting water quality in the United States. Over time, it may be expected that the uses of bioassessment data will be refined as the datasets grow and as organizations gain experience with the interpretation of information produced. While small and often unknown to the general public, macroinvertebrates provide an important contribution to streams as a food source to fish and other predators, among other functions. Macroinvertebrate communities indicate stream health on local and watershed levels, making them the indicator most used to designate stream quality in state assessment programs. Macroinvertebrate data are used in conjunction with water chemistry, habitat assessments, diatom and fish community data to identify stream characteristics, ecosystem conditions, and potential sources of disturbance.

Summary

Benthic macroinvertebrates are a diverse and important natural resource. They are well known to people who are concerned with water quality and watershed health, but ignored or taken for granted by most people in the general public. Macroinvertebrates are not normally considered for specific management actions of any kind. The management actions that affect benthic macroinvertebrates are essentially the same management actions that affect water quality and aquatic habitats. It is expected that macroinvertebrates can be allowed to thrive by preventing water pollution and by protecting or restoring natural habitat conditions in waterways.

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References

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish, Second Edition. USEPA 841- B-99-002. U.S. Environmental Protection Agency, Office of Water, Washington, D.C.

Burke, D., G. Bright. 2012. "Chapter 6.12 - Macroinvertebrates" in the Technical Report for the Delaware Estuary and Basin. Partnership for the Delaware Estuary. PDE Report No. 17-07, pp. 283-298.

Kroll, S.A., R. Abell, 2015. Points of Departure: Baseline Conditions in the Subwatershed Clusters of the Delaware River Watershed Initiative, Academy of Natural Sciences of Drexel University, 65 pp. http://www.ansp.org/~/media/Files/ans/research/pcer-other/Points%20of%20Departure.ashx-?la=en.

Kroll, S.A., R.J. Horwitz, D.H. Keller, B.W. Sweeney, J.K. Jackson, L.B. Perez, 2019. Large-scale protection and restoration programs aimed at protecting stream ecosystem integrity: The role of science-based goal-setting, monitoring, and data management. Journal of Freshwater Science 38(1): 23-39.

Kroll, S.A., H.C. Oakland, A.M. Frohn, 2021. Macroinvertebrate and diatom indicators of ecosystem integrity for setting targets in agricultural BMP restoration. Ecosphere, doi:10.1002/ecs2.3798.

Pulket, M. (editor), 2017. Wadeable multihabitat macroinvertebrate assessment method. Chapter 2.3, pages 36–45 in D. R. Shull, and R. Whiteash (editors). Assessment methodology for streams and rivers. Pennsylvania Department of Environmental Protection, Harrisburg, Pennsylvania.

Shull, D. R. (editor), 2017. Wadeable freestone riffle-run stream macroinvertebrate assessment method. Chapter 2.1, pages 2–24 in D. R. Shull, and R. Whiteash (editors). Assessment methodology for streams and rivers. Pennsylvania Department of Environmental Protection, Harrisburg, Pennsylvania.

Shull, D. R., 2018. Semi-wadeable large river macroinvertebrate assessment method. Chapter 2.4, pages 46–74 in D. R. Shull, and R. Whiteash (editors). Assessment methodology for streams and rivers. Pennsylvania Department of Environmental Protection, Harrisburg, Pennsylvania.

U.S. Environmental Protection Agency, 1997. Field and Laboratory Methods for Macroinvertebrate and Habitat Assessment of Low Gradient, Nontidal Streams. Mid-Atlantic Coastal Streams (MACS) Workgroup, Environmental Services Division, Region 3, Wheeling, WV, 23 pp.

Williams, A. (editor), 2017. Wadeable freestone riffle-run stream macroinvertebrate assessment method. Chapter 2.2, pages 25–35 in D. R. Shull, and R. Whiteash (editors). Assessment methodology for streams and rivers. Pennsylvania Department of Environmental Protection, Harrisburg, Pennsylvania.

Relevant References for Methods, Data and Integrated Reports

Delaware Department of Natural Resources and Environmental Control, 2008. State of Delaware 2008 Combined Watershed Assessment Report (305(b)) and Determination for the Clean Water Act Section 303(d) List of Waters Needing TMDLs. Delaware Department of Natural Resources and Environmental Control. 204 pp.

Delaware River Basin Commission, 2010. 2010 Delaware River and Bay Integrated List Water Quality Assessment. Delaware River Basin Commission. 52 pp.

New Jersey Department of Environmental Protection, 2010. 2010 Integrated Water Quality Monitoring and Assessment Methods. (Appendix F of the 2010 New Jersey Water Quality Monitoring and Assessment Report). New Jersey Department of Environmental Protection. 79 pp.

New Jersey Department of Environmental Protection, 2007. Standard Operating Procedures, Ambient Biological Monitoring Using Benthic Macroinvertebrates. New Jersey Department of Environmental Protection. 44 pp.

New York Department of Environmental Conservation, 2009. The New York State Consolidated Assessment and Listing Methodology. New York Department of Environmental Conservation. 23 pp.

Lookenbill, M. J., and R. Whiteash (editors), 2021. Water quality monitoring protocols for streams and rivers. Pennsylvania Department of Environmental Protection, Harrisburg, Pennsylvania.

Shull, D. R., and R. Whiteash (editors), 2021. Assessment methodology for streams and rivers. Pennsylvania Department of Environmental Protection, Harrisburg, Pennsylvania.

Pennsylvania Department of Environmental Protection, 2022. 2022 Pennsylvania Integrated Water Quality Monitoring and Assessment Report, Clean Water Act, Section 305(b) Report and 303(d) List. Pennsylvania Department of Environmental Protection, Harrisburg, Pennsylvania.

7.2 Vertebrates

7.2.1 Osprey

Introduction

Osprey (*Pandion haliaetus*; Fig 7.2.1.1) are found on all continents except Antarctica. This large bird of prey has regained a prominent position in the Delaware Estuary after having experienced dramatic population declines in the mid-20th century particularly associated with environmental contaminants.

Osprey typically arrive in the Delaware Estuary in mid to late March and begin nesting in April (Hess et al. 2000, Wurst and Clark 2021). Nest sites include live or dead trees but they have particularly adapted to nesting on artificial structures such as nesting platforms, utility poles, channel markers, and duck blinds (Poole et al. 2002). Nests are comprised of sticks and other materials opportunistically found and individual nests can be used for many years as they gradually increase in size.

Nestlings fledge in the summer and join adults migrating south in early September. Wintering likely occurs mostly in the Caribbean and South America for osprey nesting in the Delaware Estuary (Martell et al. 2001).

Osprey occur near water where they feed almost exclusively on fish that make up 99% of their diet (Poole et al. 2002). Osprey are highly adapted for opportunistically capturing a variety of species of fish common in the Delaware River watershed such as menhaden and summer flounder. Some of their adaptations include oily feathers to reduce water absorption, nostrils that can shut when in water, and spines on their toes along with a reversible outer toe to help them keep a secure grip on fish. At times osprey may plunge nearly completely underwater in pursuit of their prey but are generally restricted to foraging on fish near the surface. Therefore, they are most concentrated in areas where there are abundant fish in shallow water or schooling fish near the surface and are able to forage more effectively in calm sunny weather.

Bald eagles (*Haliaeetus leucocephalus*) and great horned owls (*Bubo virginianus*) are known to take fledgling osprey and raccoons (*Procyon lotor*) occasionally take eggs (Poole et al. 2002). Raptors and other birds will take over osprey nests as their own nesting sites. Bald eagles also often rob osprey of the fish they have caught.

Figure 7.2.1.1 (Left) Adult osprey on its nesting platform. Photo credit: Charlie Lister; (Right) An osprey family in their nest. Photo credit: Deborah Freeman.

Description of Indicator

New Jersey, Delaware, and Pennsylvania have osprey monitoring and conservation programs. Nest checks by ground observers are conducted to determine active nests and productivity between the end of April and mid-July. Every five years more comprehensive surveys may be done including aerial observation of nests. Each state works independently on their monitoring programs with their own protocol and therefore the reported data upon which this indicator is based vary in type and quality. Delaware Division of Fish and Wildlife has relied mostly on volunteer monitoring since 2014 and does not count all nests in the State but does examine productivity of selected nests. New Jersey Division of Fish and Wildlife staff in partnership with the Conserve Wildlife Foundation of New Jersey perform ground surveys and also rely on citizen scientist Osprey Watchers to monitor some nests from a distance and have the most data on osprey in the watershed. Pennsylvania has been developing a database of known osprey nests. The Pennsylvania Game Commission relies on a combination of staff observations and citizen scientist Osprey Watchers.

The data sets gathered by the states, although limited and not inclusive of all nesting activity, do suggest a generally positive trend for the number of active nests and productivity. Figure 7.2.1.2 shows data for New Jersey, Figure 7.2.1.3 for Pennsylvania and Tables 7.2.1.1 and 7.2.1.2 show data for Delaware.

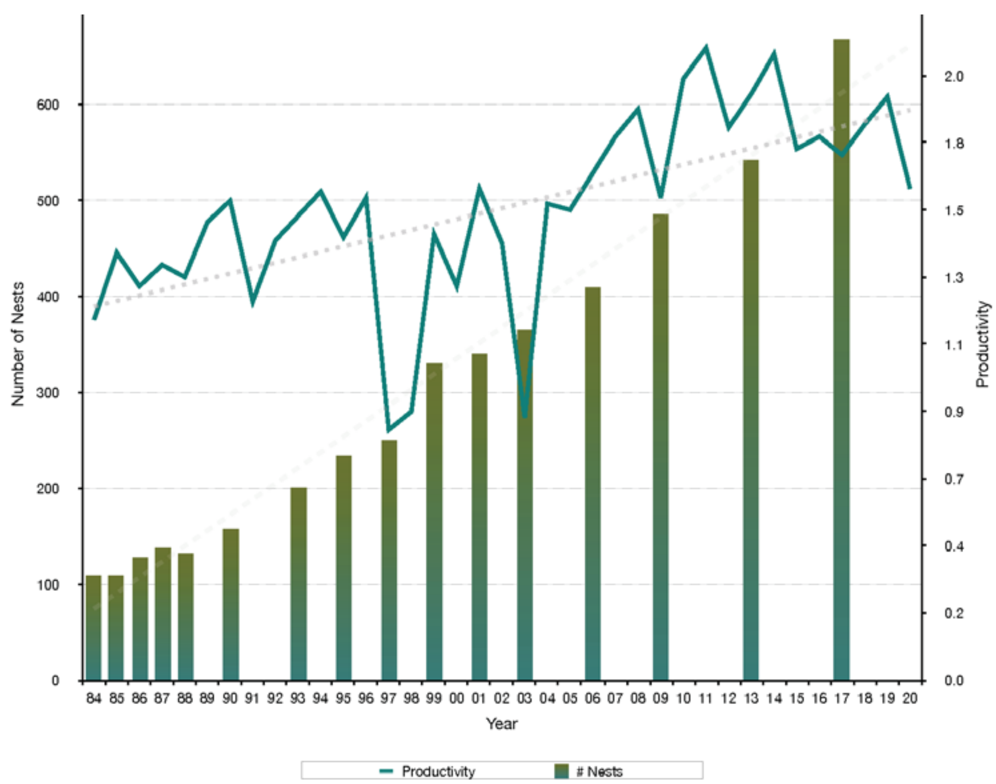


Figure 7.2.1.2 New Jersey State-wide osprey nesting population (bars) and productivity (line) 1984-2021 (Wurst and Clark 2022).

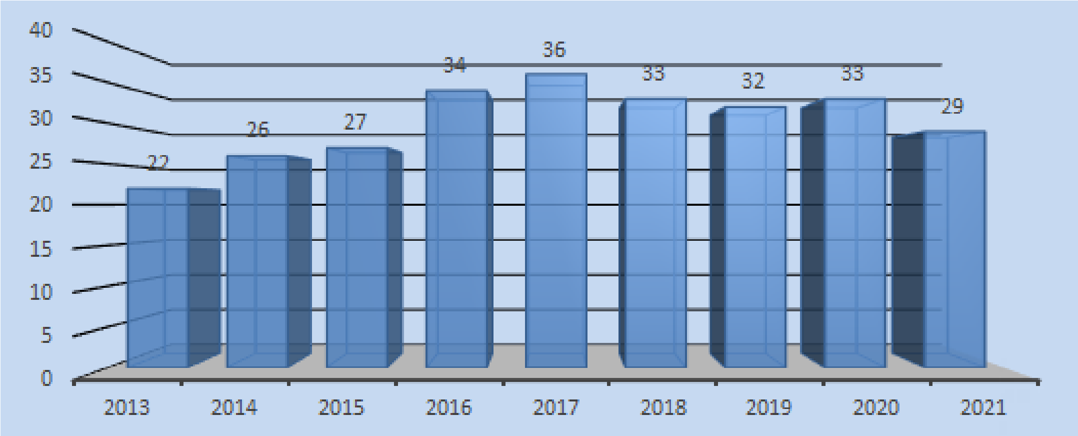


Figure 7.2.1.3 Pennsylvania active osprey nests 2013-2021 in the Delaware River watershed from Northampton County to Delaware County (Morgan and Barber 2021: unpublished report).

Table 7.2.1.1 Delaware state-wide osprey nest surveys 2003, 2007, and 2014 (DE DFW 2014: unpublished report). Active nests are those where eggs or chicks were seen in a nest during at least one survey, Successful nests are those in which at least one chick reached banding age, and Chicks refers to the number observed in the nest during any survey.

Year	Active nests	Successful nests	Chicks	Chicks/active nest
2003	119	77	135	1.1
2007	173	136	293	1.7
2014	197	103	424	2.2

Table 7.2.1.2 Delaware volunteer osprey nest monitoring project 2014–2020 (Brown and Robinson, 2020: unpublished report). Active nests are those with eggs or chicks confirmed (includes incubating posture) and Chicks refers to the number of successfully fledged chicks.

Year	Active nests	Successful nests	Chicks	Chicks/active nest
2014	91	64	141	2.2
2015	69	54	82	1.8
2016	115	70	98	1.4
2017	72	69	93	1.3
2018	90	74	115	1.6
2019	83	66	120	1.8
2020	97	83	138	1.7

Present Status

Osprey appear to be doing well in the Delaware Estuary and are a noteworthy conservation success story for the area. New Jersey still lists osprey as threatened, but Pennsylvania and Delaware no longer do. Productivity, as measured by fledglings observed, consistently appears to be higher than needed for a stable population. For example, New Jersey that has the largest dataset between the three states, has seen productivity rates over 0.80 (the minimum needed for a stable population) every year data was collected since 1997 (Wurst and Clark 2022).

Population levels may be at least back to what is believed to have been the level prior to the widespread use of dichlorodiphenyltrichloroethane (DDT)(Henny et al. 2010). Studies of osprey eggs and diet and water quality show that the birds continue to be exposed to environmental contaminants. These include legacy contaminants, such as PCBs and DDT, and more novel contaminants, such as pharmaceutical ingredients, but the exposures are below levels believed to warrant concern (Bean et al. 2018, Rattner et al. 2018).

Past Trends

Historically abundant, osprey populations declined precipitously in the Northeast from the 1950s through the 1970s due exposure from DDT, from widespread use to control mosquitoes, and other contaminants that enter the osprey food chain and consequently impact breeding success of ospreys (Henny et al. 2010, Bierregaard et al. 2014). The location of osprey at the top of the aquatic food chain means they can be exposed to high concentrations of contaminants that have been biomagnified through their food chain from relatively low concentrations in the sediment and water column. Delaware Estuary populations remained somewhat depressed into the 1990s partly due to high organochloride levels still in the system, such as polychlorinated biphenyls (PCBs) and legacy concentrations of the metabolites of DDT, but since then levels of organochlorides have gradually lowered allowing productivity to improve. Additionally, in the 19th and into the mid early 20th century, osprey were shot, their nests were destroyed, or their eggs were collected far more often than now (Bierregaard et al. 2020).

Future Predictions

The outlook for osprey continues to be good in the Delaware Estuary and they are expected to continue to rebound and thrive. Osprey adapt well to some anthropogenic disturbance, such as moderate coastal development and boat traffic. As long as nesting sites remain, fish populations do well, and novel contaminant threats are not created, osprey should sustain population levels currently observed.

Actions and Needs

Osprey have shown flexibility for nesting structures. However, the most common nesting structure for osprey in the estuary are nesting platforms put up specifically for them. These structures are susceptible to collapse especially as nests increase in weight over years (Wurst and Clark 2021) or with the possibility of increased storm intensity associated with climate change. Continued installation and especially maintenance of nesting platforms will ensure a continued robust population. For example, New Jersey generally is not installing new nesting platforms to increase the population further but is working to maintain the existing platforms to maintain the current robust population (Wurst and Clark 2022). New platforms are now mostly installed only when a current nest must be relocated from an unsuitable structure in New Jersey.

Contaminants likely to biomagnify into osprey have declined in the estuary and in the osprey and their eggs. However, due to their reliance on fish, osprey will continue to potentially be exposed to some concentrations of legacy contaminants, such as PCBs and DDT metabolites, as well as novel contaminants in the water and sediment through their prey. We do not yet fully understand the impact of some of these contaminants and therefore osprey warrant continued observation (Bierregaard et al. 2014).

Volunteers are needed for monitoring nests and productivity. Due to limited resources, the states do not know of all nests or track productivity of all known nests. More support for these programs will enhance the current data sets to further indicate osprey population health in the watershed. Those interested in volunteering, establishing nesting structures, reading more detailed reports or that have questions about osprey should contact the State agencies responsible for osprey conservation:

NJ: http://www.conservewildlifenj.org/protecting/projects/osprey/

DE: https://dnrec.alpha.delaware.gov/fish-wildlife/conservation/osprey-monitoring/

PA: https://www.pgc.pa.gov/InformationResources/GetInvolved/Pages/OspreyNestSurvey.aspx

Summary

Osprey populations in the Delaware Estuary are a success story. They demonstrate the value of reducing contaminants in our environment and taking conservation actions. The success of osprey conservation and the continued monitoring of the populations shows how volunteers can work with the states to make a difference in the estuary.

References

Bean T.G., B.A. Rattner, R.S. Lazarus, D.D. Day, S.R. Burket, B.W. Brooks, S.P. Haddad, and W.W. Bowerman. 2018. Pharmaceuticals in water, fish and osprey nestlings in Delaware River and Bay. Environmental Pollution 232:533-545. https://doi.org/10.1016/j.envpol.2017.09.083

Bierregaard, R.O., A.F. Poole, and B.E. Washburn. 2014. Ospreys (*Pandion haliaetus*) in the 21st Century: Populations, migration, management, and research priorities. Journal of Raptor Research 48(4):301-308. 2014. http://dx.doi.org/10.3356/0892-1016-48.4.301

Bierregaard, R.O., A.F. Poole, M.S. Martell, P. Pyle, and M.A. Patten. 2020. Osprey (*Pandion haliaetus*), Version 1.0. In P.G. Rodewald (Ed.), Birds of the World. Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bow.osprey.01

DE DFW. 2014. Final Report: Delaware Osprey Monitoring Program Report. Unpublished report. Delaware Division of Fish and Wildlife, Dover.

Henny, C.J., R.A. Grove, J.L. Kaiser, and B.L. Johnson. 2010. North American osprey populations and contaminants: Historic and contemporary perspectives. Journal of Toxicology and Environmental Health, Park B 13:579-603. https://doi.org/10.1080/10937404.2010.538658

Hess, G.K., R.L. West, M.V. Barnhill, and L.M. Fleming. 2000. Birds of Delaware. University of Pittsburgh Press, Pittsburgh.

Martell, M.S., C.J. Henny, P.E. Nye, and M.J. Solensky. 2001. Fall migration routes, timing, and wintering sites of North American ospreys as determined by satellite telemetry. The Condor 103: 715-724. https://doi.org/10.1093/condor/103.4.715

Morgan, J. and P. Barber. 2021. Draft summary of 2021 osprey nesting in Pennsylvania's Southeast Region. Unpublished report. Pennsylvania Game Commission Bureau of Wildlife Management. August 2021.

Poole, A.F., R.O. Bierregaard, and M.S. Martell. 2002. Osprey. Number 683. In A. Poole and F. Gill (Eds.). The Birds of North America. The Birds of North America, Inc., Philadelphia, Pennsylvania.

Rattner, B.A., R.S. Lazarus, T.G. Bean, P.C. McGowan, C.R. Callahan, R.A. Erickson, and R.C. Hale. 2018. Examination of contaminant exposure and reproduction of ospreys (*Pandion haliaetus*) nesting in Delaware Bay and River in 2015. Science of the Total Environment 639:596-607. https://doi.org/10.1016/j.scitotenv.2018.05.068

Brown, J. and S. Robinson. 2020. Delaware Citizen Osprey Monitoring Program Data Summary. Unpublished report. Delaware Division of Fish and Wildlife, Dover.

Wurst, B. and K. Clark. 2021. 2020 New Jersey osprey project report. Kathleen Clark, NJ Division of Fish and Wildlife - Endangered and Nongame Species Program and Benjamin Wurst, Conserve Wildlife Foundation of New Jersey. February 5, 2021. https://www.nj.gov/dep/fgw/ensp/pdf/osprey20.pdf

Wurst, B. and K. Clark. 2022. 2021 New Jersey osprey project report. Kathleen Clark, NJ Division of Fish and Wildlife - Endangered and Nongame Species Program and Benjamin Wurst, Conserve Wildlife Foundation of New Jersey. April 2022.

7.2.2 Atlantic Sturgeon

In 2012, the National Marine Fisheries Service declared the New York Bight Distinct Population Segment of Atlantic sturgeon to be endangered. The Delaware River spawning stock is part of this "population segment" and is officially considered Endangered under the Endangered Species Act. The Delaware River population of Atlantic sturgeon has been determined to be genetically similar to those of the Hudson River, but through range-wide genetic analysis of nuclear DNA, at least 6 sub-populations were suggested including one for the Delaware River distinguishable from the Hudson River stock (King et al. 2001).

During the late nineteenth century, the fishery for Atlantic sturgeon peaked. The Delaware River landings were an order of magnitude higher than landings from all other estuaries and rivers combined. One estimate based on these landings was that the Delaware River spawning stock included 180,000 mature females (Secor 2002). Secor estimated the next highest stock as 20,000 females for the entire Chesapeake Bay. This past abundance suggests that the Delaware may still have the potential to produce a large population of sturgeon.

Mature Atlantic sturgeon migrate from the sea to fresh water in advance of spawning. In the Delaware River, first-maturing females are likely to be at least 15 years old. Males can be somewhat younger at first spawning. Spawning occurs in flowing fresh waters with a hard bottom. Shed eggs are 2-3 mm in diameter and become sticky when fertilized. They frequently become attached to hard substrates or submerged detritus until hatching in several days. After hatching occurs, juveniles (Fig 7.2.2.1) remain in fresh water for several years but have been documented to out-migrate to coastal areas in their 3rd year. Sweka et al. (2006), found that juvenile sturgeon preferred soft bottom habitats at depths greater than 6.3 meters in the Hudson River. Once juveniles out-migrate from their natal river they are known to frequent distant estuary systems (Secor et al. 2000); tagged age-0 fingerlings stocked in the Hudson River in 1994 were found in the Chesapeake and Delaware Bays in 1997 (Bain 1998).

Mature individuals also frequent estuaries distant from their natal river. Studies performed in the Hudson River using pop-up satellite archival tags showed that the majority of adult Atlantic sturgeon captured and tagged in the Hudson during spawning season eventually out-migrated to the mid-Atlantic Bight; one individual, however, traveled north to the Bay of Fundy and another went south to coastal Georgia (Erickson et al. 2011). Mature Atlantic sturgeon are of great potential commercial value for both flesh and roe, the latter being known as caviar. Although there is an occasional report of Atlantic sturgeon caught with rod and reel, the species is not known for recreational fishing importance.

Figure 7.2.2.1 Juvenile Atlantic sturgeon collected by the Delaware Division of Fish and Wildlife in the Delaware River. Photo credit: DDFW.

Descriptions of Indicators

The primary indicator, which monitors the annual production of young sturgeon in the tidal Delaware River, is the mean catch per unit of effort (CPUE) of young sturgeon from the small-mesh gill net survey conducted by the Delaware Division of Fish and Wildlife (Park 2020).; specifically it is the mean catch per hour per square meter of gill net (Fig 7.2.2.2). The survey employs four gill nets 91.5 meters long and 2.4 meters deep; two of these consisted of 5.1 cm stretch mesh and two consisted of 7.6 cm stretch mesh. The nets were constructed from 0.33 mm diameter clear monofilament, with a 29.5 kg/182.9 m lead line and a 1.3 cm foam core rope with floats every 4.57 meters. Sampling was conducted at least twice a week from October through December and was weather dependent. Anchor nets were set diagonal to the current approximately 45 minutes prior to slack tide and pulled at the onset of the next tide. Sampling was limited to slack tide due to strong tidal currents.

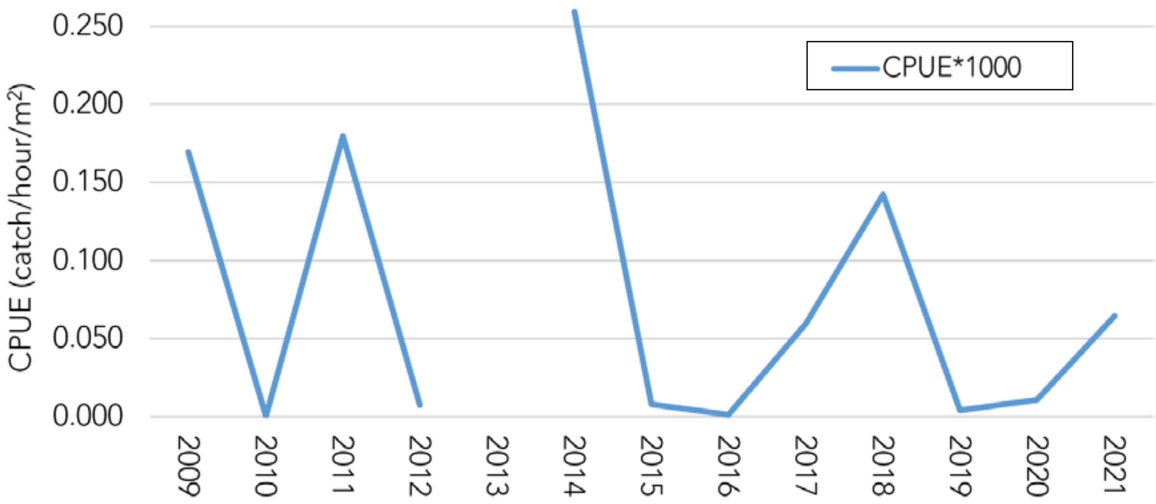


Figure 7.2.2.2 Primary indicator. Relative abundance trend of age 0 Atlantic Sturgeon in the Delaware River. The indices are the catch per unit effort in small mesh gill-nets set in the tidal Delaware River.

The secondary indicator is the relative abundance index of larger juvenile and some adult sturgeon from by the Delaware Division of Fish and Wildlife's Adult Groundfish research trawl survey, which is conducted monthly from March through December at nine fixed sites in the Bay (Fig 7.2.2.3). The Division conducts the survey on the sixty-two-foot-long (19 meters) research vessel, the "First State". An otter trawl with a thirty-foot, six-inch (9.3 m) long headrope is towed along the bottom. Nine sites are sampled each month from March through December. The index of abundance is the mean catch-per-tow (cpt) of Atlantic sturgeon over the course of each year.

Present Status

Recent results from the primary indicator, the small-mesh gill-net survey conducted with similar effort annually since 2014 (Fig 7.2.2.2), showed high abundance in 2014 and 2018, with lower levels in other years. The 2016 index was almost certainly reduced by the overlapping of the gill net survey that year with a sturgeon relocation project conducted for the U. S. Army Corps of Engineers Sturgeon Relocation Project (280 sturgeon relocated fall of 2016 – all numbers refer to Atlantic sturgeon relocated; data provided by Environmental Research and Consulting, Inc). That project was part of the larger project to deepen the shipping channel in the River right in the reach that is a favored location for young sturgeon, near Marcus Hook and the Pennsylvania-Delaware border. Bedrock in the navigation channel had to be fractured using a drilling and blasting technique. In order to reduce possible mortality of sturgeon caused by blasting used to deepen the channel, the Corps contracted with a commercial fishing vessel to trawl the channel to capture sturgeon and then relocate them up river. With its large net (almost three times larger than the net used for the Delaware Division of Fish and Wildlife's Delaware Bay research groundfish trawling), the trawler was effective at capturing sturgeon.

Some of the relocated sturgeon had been tagged, and some of them returned and were then caught a

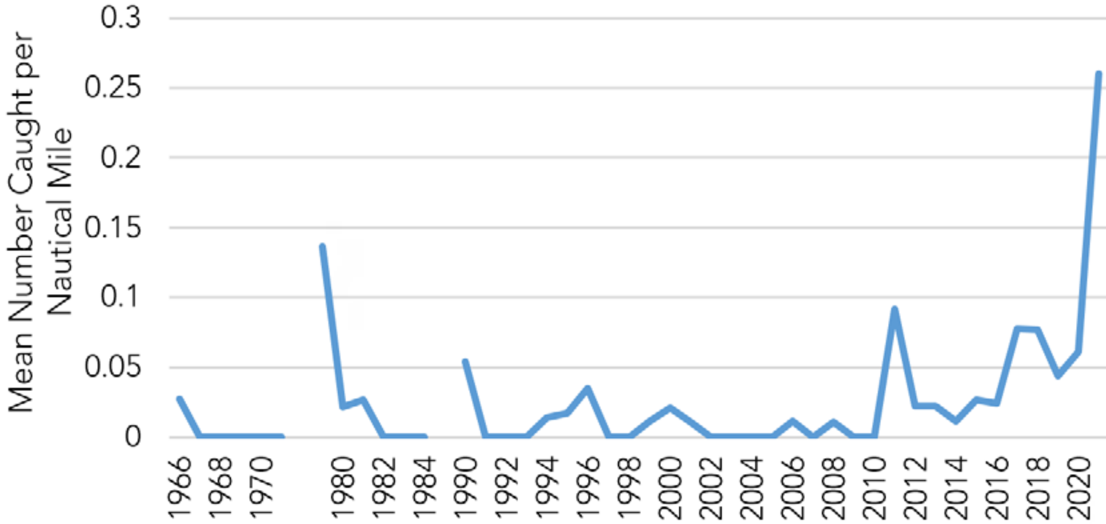


Figure 7.2.2.3 Secondary indicator. Relative abundance trend of older juvenile and adult Atlantic Sturgeon based on the mean number caught per nautical mile by the Delaware Division of Fish and Wildlife's Adult Fish Trawl Survey in Delaware Bay (M. Greco, DDFW, personal communication). The survey has been conducted from 1966 through 1971, 1979 through 1984 and 1990 through the present. The survey design varied between periods and within periods prior to 1990. Survey sampling gear was consistently an otter bottom trawl with a thirty-foot headrope.

second time at the Marcus Hook area, substantially downriver from where they were released, upstream of Philadelphia. Consequently, it was difficult to estimate a reduction in the gill net survey in the fall catch from a relocation the previous winter.

The relocation project operated from 2016 through 2020 during fall and winter. In 2017, the index of abundance was based on the gill-net survey operation prior to the trawling operation in November (fall 2017- winter 2018: 3,211 sturgeon relocated). No trawling was conducted during the fall of 2018, but there was extensive trawling during the winter of 2019 (1,355 sturgeon relocated). A minimal amount of trawling was conducted in the fall of 2019 and the winter of 2020 (170 sturgeon relocated)

A potential cause of the relatively low young-of-year index in 2015, 2019, and 2020 could be mortality of young sturgeon due to hypoxia. Kahn and Fisher (2012) combined earlier relative abundance estimates from this survey with data collected by the United States Geological Survey on temperature and oxygen content of the river. They developed a hypoxia hypothesis that years with low levels of dissolved oxygen (DO) in the nursery area of the river have caused high mortality of young-of-year sturgeon, which is reflected in relatively low relative abundance indices in the gill net survey in those years.

The current Delaware River Basin Commission's (DRBC) criteria for dissolved oxygen, enacted in 1967, is only 3.5 mg/liter as an average over a 24 hour period, meaning the DO level can drop below that level for several hours and still meet the 24 hour average criteria. When river temperatures increase in July and August to as much as 30 ˚C., 3.5 mg/L is only 40% oxygen saturation, which is considered hypoxia. Consequently, the current DRBC oxygen criteria allows for hypoxic conditions, which laboratory tests have shown reduces survival, especially of young-of-year sturgeon at high temperatures (Niklitschek and Secor 2009). At this time of year, young-of-year sturgeon are only one to two months old.

Recent data seems consistent with the hypoxia hypothesis. In 2015, DO saturation registered below

50%, and in 2019, 2020 and 2021 DO saturation levels reached as low as 40% in an area of the river from at least Chester, PA to Philadelphia, PA (Fig 7.2.2.4). Relative abundance was very low in 2019 and 2020, and was low in 2021 (Fig 7.2.2.2). Juvenile sturgeon collected in 2019 were also smaller than those collected in other years, suggesting that low oxygen conditions or the increase in salinity observed that year may have had a negative effect on growth rate, as well as survival, as reported by Niklitschek and Secor (2009).

An alternative explanation of fluctuations in the relative abundance of young-of-year sturgeon is that spawning and hatching success vary markedly among years. This could be due to fluctuations in the number of spawning adults, or due to combined effects of various environmental factors such as timing of fluctuations of flow, weather events and temperature. In either case, if such random effects of spawning density or environmental variables are the cause of variation in relative abundance of young-of-year sturgeon, no obvious correlation between relative abundance and dissolved oxygen levels would be expected. As the data collection proceeds, a test of the hypoxia hypothesis will be feasible.

A recent estimate of total abundance of juvenile sturgeon of ages 0-1 years in the Delaware River was presented by Hale et al. 2016. Utilizing tag-recapture methods, they estimated abundance in 2014 as 3,656 (95% CI = 1,935–33,041), and abundance in 2018 was estimated to be 5,846 (95 CI = 2,394-14,446) (Park 2020). We are seeing high variability, with estimates of abundance between the low thousands and fifteen thousand annually. While trawling was not conducted in 2014, these numbers are roughly comparable to the numbers collected during the relocation project.

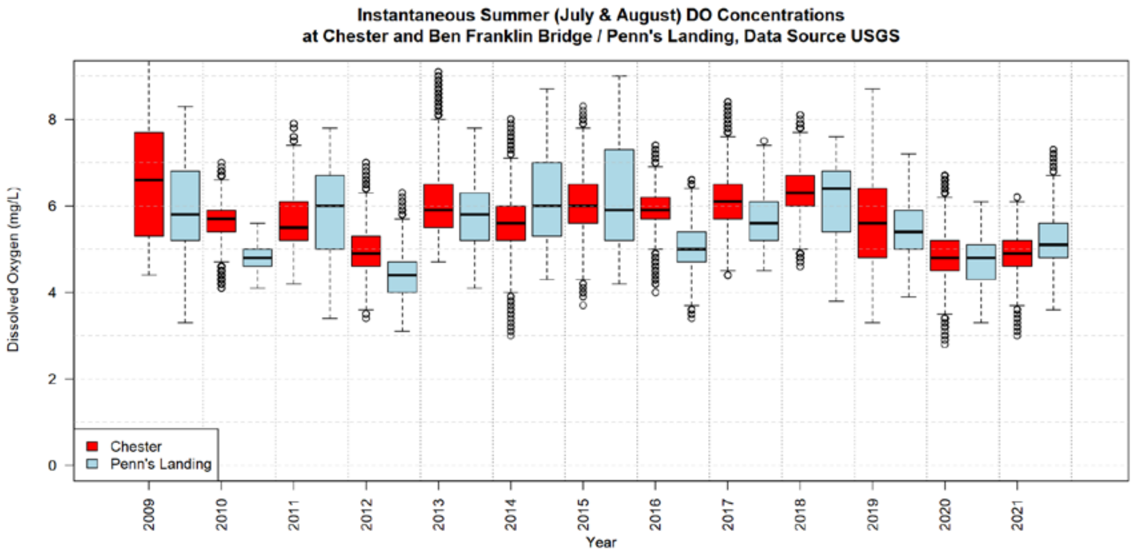
Hale et al. also tracked locations of ages 0-1 sturgeon using acoustic tags during 2014. They located these young fish in concentrations centered on the Delaware-Pennsylvania border, specifically the Marcus Hook anchorage, the Cherry Island Flats adjacent to Wilmington, and the Chester Range of the river and nearby areas.

Acoustic tracking continued in 2015-2019, and similar results were obtained (Park 2020). Marcus Hook is the prime nursery location. The deep Marcus Hook reach is the location of the Marcus Hook Anchorage, which can often be identified visually because tankers frequently are anchored in the river there.

The secondary indicator has shown a consistently increasing trend in abundance of sub-adults in the Bay (Fig 7.2.2.3, M. Greco, Delaware Division of Fish and Wildlife, personal communication), building upon a similar but lower trend, reported in the 2017 Technical Report on the Delaware Estuary (Kahn 2017). Sturgeon caught in this trawl survey may be from other spawning stocks, since sturgeon are known to wander the Atlantic coast, entering various estuaries. However the sharp increase in catch in 2021 coincides with the large year class documented in 2018, since the lengths of many of the sturgeon collected in the trawl survey were that of three-year olds.

During the last decade, there has been about a doubling of reports to the Delaware Division of Fish and Wildlife of sturgeon carcasses in the Delaware estuary compared to the previous decade. Brown and Murphy (2010) initially documented the appearance of carcasses in the Bay and River. Most carcasses bear evidence of propeller strikes from ships. The recent increase may, in part, have been caused by an increased reporting rate, since the Division has conducted a campaign directed at anglers by requesting reports of observed sturgeon mortalities. Another likely cause of this increase in reported carcasses is the apparent increase in abundance of larger sturgeon indicated by the Division's research trawl survey (Fig 7.2.2.3).

Researchers at Delaware State University have worked with the Division to investigate these reports. Sixty-one percent of the carcasses that were measured through 2019 were of adult length (over 1500 mm; approximately 5 feet). In 2020, for example, nine carcasses were reported to the Division, of which



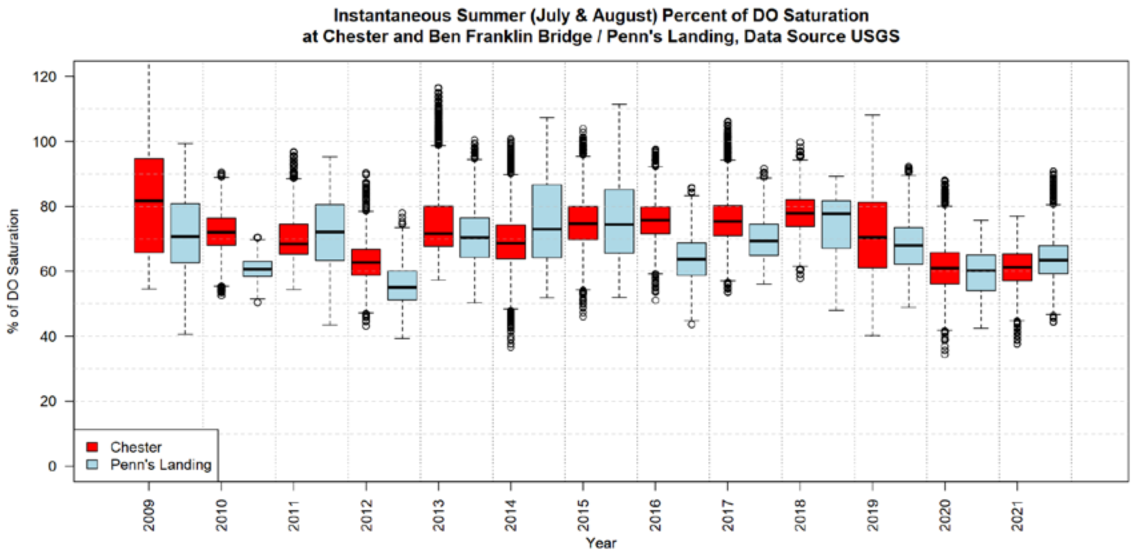


Figure 7.2.2.4 Dissolved oxygen readings conducted by the United States Geological Survey in the tidal Delaware River from 2009 through 2021 during July and August, based on two separate dissolved oxygen meters. One meter is located at Chester, Pennsylvania and the other was at the Benjamin Franklin Bridge in Philadelphia until it was moved the Penns's Landing in central Philadelphia. Measurements were conducted every 30 minute or every 15 minutes. The solid line in each box is the median value. The upper and lower boundary of each box represent the upper and lower quartiles of the measurements, respectively. The length of the box is known as the interquartile range. The dashed lines are drawn from the top of each box to the largest observation within 1.5 interquartile ranges of the top and conversely a dashed line is drawn from the bottom of each box to the smallest observation within 1.5 interquartile ranges of the bottom. The chart was provided by John Yagecic, Delaware River Basin Commission.

six appeared to have been killed by ship strikes; eight of these were measured and half appeared to be of adult size (Park 2021). There has been concern expressed that this mortality from ship strikes can repress the ability of the stock to rebuild.

Past Trends

Heavy fishing in the late nineteenth century in the lower tidal river probably caused a decline in the population, although no data or analysis documents that hypothesis. The production of eggs from mature females was high enough that a railroad line was constructed to Delaware City, Delaware to haul the eggs, which were held in barrels. They were apparently shipped to Russia for processing into caviar.

The failure of the stock to recover during the twentieth century is likely to be due to sturgeon's sensitivity to oxygen levels and the depletion of oxygen in the Delaware. Sturgeon have been described as more sensitive to oxygen levels than rainbow trout (Klyashtorin 1982). Webster et al. (1914) reported an oxygen reading in Philadelphia of only 1 mg of oxygen per quart of water. In 1946, Ellis et al. (1947) found dissolved oxygen (DO) levels less than 2.0 mg/l for thirty-five to forty miles of the River in spring and fall from Pennsville, NJ, 6 miles below Wilmington to several miles above Philadelphia. Sharp (2010) in a comprehensive review of hypoxia in the river, reports data and estimates of oxygen levels from 1880 to 2005.

The method of using small-mesh gill nets to monitor young-of-year sturgeon (the primary indicator) was only introduced in 2008 by Fisher (2009). There is no indication from this indicator of an increasing trend in abundance of young sturgeon in the river; in fact, the survey shows reduced abundance from 2015 on. The interference of the sturgeon relocation project clouds this picture for some years; the capture of hundreds to thousands of young sturgeon by the relocation project, however, verifies that substantial levels of reproduction have been occurring consistently.

The secondary indicator (Fig 7.2.2.3) had a couple of years early in the research trawl survey with relatively high indices of subadults and adults. The previous highest index, though, which was for 1981, has now been surpassed by the 2021 value. The last nine years have had higher indices, on average, than the earlier years back to the 1960s. Consequently, this indicator does indicate an increase in abundance in Delaware Bay of adults and subadults over the last decade.

Actions and Needs

The discussion above outlines the problem with the current oxygen criterion, which is dangerous to young sturgeon. While the regulations imposed in 1967 have produced increased oxygen levels in the river compared to earlier decades, when levels were below 2 ppm for forty kilometers of river in summer (Ellis et al. 1947), the continuing presence of high levels of nitrogen-based biological oxygen demand in the river is well-documented and the Delaware River Basin Commission is currently considering requiring improvement in DO by reducing input of ammonia and other sources of nitrogen into the river while raising the criteria for oxygen.

Summary

The data presented above show that Atlantic sturgeon are reproducing successfully in most years in current conditions in the river, with hundreds to thousands of sturgeon ages 0 – 1 present in the river. Delaware Bay is hosting more consistent levels of subadults and adults then during the previous decade. While ship strikes are causing some mortality on sub-adults and adults, we don't have an estimate of the mortality rate caused by this factor. The question is to what extent this mortality will affect the population growth rate. Ship-strike mortality will have some negative effect on the population, but the amount of reduction in population growth rate is unknown.

References

Bain, M. 1998. Hatchery Atlantic sturgeon released in the Hudson River showing up in the Delaware River and Chesapeake Bay. Cornell Sturgeon Notes 5:14. Newsletter of the New York Cooperative Fish and Wildlife Research Unit- Department of Natural Resources. Cornell University, Ithaca, NY.

Brown, J.J. and G.W. Murphy. 2010. Atlantic Sturgeon Vessel-Strike Mortalities in the Delaware Estuary. Fisheries 35:72-83.

Erickson D. L., A. Kahnle, M. J. Millard, E. A. Mora, M. Bryja, A. Higgs, J. Mohler, M. DuFour, G. Kenney, J.Sweka and E. K. Pikitch. 2011. Use of pop-up satellite archival tags to identify oceanic-migratory patterns for adult Atlantic Sturgeon, Acipenser oxyrinchus oxyrinchus Mitchell, 1815. Journal of Applied Ichthyology 27:356–365.

Fisher, M. 2009. Atlantic sturgeon progress report. State Wildlife Grant Project T-4-1. Delaware Division of Fish and Wildlife, Dover.

Hale, E.A., I., Park, M.T. Fisher, R.A. Wong, M.J. Stangl and J.H. Clark. 2016. Abundance Estimate for and Habitat Use by Early Juvenile Atlantic Sturgeon within the Delaware River Estuary, Transactions of the American Fisheries Society. 145:6, 1193-1201.

Kahn, D. M. 2017. Atlantic Sturgeon. Technical Report for the Delaware Estuary and Basin. Partnership for the Delaware Estuary. Wilmington.

Kahn, D. M. and M. Fisher. 2012. Endangered Atlantic Sturgeon in the Delaware River require higher standards for dissolved Oxygen. Delaware Division of Fish and Wildlife, Dover. Available at Reports by Others - Partnership for the Delaware Estuary

King, T.L., B.A. Lubinski, and A.P.Spidle. 2001. Microsatellite DNA variation in Atlantic sturgeon (Acipenser oxyrinchus oxyrinchus) and cross species amplification in the Acipenseridae. Conservation Genetics 2:103-119.

Klyashtorin, L. B. 1982. The ability of sturgeons (Acipenseridae) to regulate gas exchange. J. Ichthyology 21.

Niklitschek, E. J. and D. H. Secor. 2009. Dissolved oxygen, temperature and salinity effects on the ecophysiology and survival of juvenile Atlantic sturgeon in estuarine waters: I. Laboratory results. Journal of Experimental Marine Biology and Ecology 381: 150-160.

Park I.A. 2021. State of Delaware Annual Compliance Report for Atlantic Sturgeon. Delaware Division of Fish and Wildlife

Park, I.A. 2020. Conservation and Recovery of Juvenile Sturgeons in the Delaware River. Delaware Division of Fish and Wildlife. Final Report to National Marine Fisheries Service. Award NA16NMF4720072

Secor, D. H. 2002. Atlantic Sturgeon fisheries and stock abundances during the late nineteenth century. American Fisheries Society Symposium 28:89-98

Secor, D.H., and 9 co-authors. 2000. Dispersal and growth of yearling Atlantic sturgeon Acipenser oxyrinchus, released into the Chesapeake Bay. U.S. National Marine Fisheries Service Fishery Bulletin 98:800-810.

Sharp, J. H. 2010. Estuarine oxygen dynamics: What can we learn about hypoxia from long-term records in the Delaware estuary? Limnology and Oceanography. 55:535-548.

Sweka, J.A., J.Mohler, M.J. Millard. March 2006. Relative Abundance Sampling of Juvenile Atlantic Sturgeon in the Hudson River. Technical Report to the NY State Department of Environmental Conservation. Hudson River Fisheries Unit. New Paltz, NY. 46 pp.

Webster, G. S., G. E. Dateman and W. L. Stevenson. 1914. Report on the Collection and Treatment of the Sewage of the City of Philadelphia. Philadelphia Dept. of Pub. Wks., Philadelphia. 187 pp

7.2.3 White Perch

White perch (*Morone americana*; Fig 7.2.3.1) are one of the most abundant fish in the Delaware Estuary and probably the most widespread, found in nearly all the waters of the Delaware Estuary, from the lower bay to uppermost reaches of the estuary's many tidal tributaries. White perch support important recreational and commercial fisheries throughout the estuary.

Figure 7.2.3.1 White perch being measured for total length. Photo by Jenny Shinn, Rutgers University.

White perch are closely related to striped bass, but the white perch is a much smaller fish. Although the Delaware state record white perch was 2 pounds 9 ounces (1.2 kg), any white perch over one pound is considered large. Delaware Estuary white perch display anadromous tendencies in that large aggregations of white perch move into tidal tributaries in spring to spawn and then out into the deeper waters of the Estuary to overwinter, but, unlike striped bass, white perch rarely leave the Estuary. White perch numbers in the Delaware Bay and River typically increase during fall and remain high through winter, then decrease during spring and summer (Miller 1963, PSEG 1984), while white perch numbers in the tidal tributaries show the opposite trend (Smith 1971). However, white perch were caught year-round in both the Delaware Estuary (de Sylva et al 1962) and the tidal tributaries (Smith 1971), so the evidence was inconclusive about the extent of white perch movements. On a smaller spatial scale, Jones et. al (2014) used acoustic telemetry in Alloway Creek, NJ, to determine white perch exhibit high site fidelity and variable movement patterns but prefer *Spartina* spp. and mixed marshes to those invaded by *Phragmites australis*. In addition, landlocked white perch populations have thrived for years in most of the freshwater ponds in the headwaters of Delaware Estuary tidal tributaries (Martin 1976).

White perch spawn in the Delaware River (Miller 1963, PSEG 1984) and most of the Delaware Estuary tidal tributaries (Miller 1963, Smith 1971, Clark 2001). Spawning occurs from early April through early June, but May is usually the peak spawning month (Miller 1963, Smith 1971, PSEG 1984). Young-of-the-year white perch, are found in both the Delaware Estuary (PSEG 1984) and the lower salinity reaches of tidal tributaries (Smith 1971, Clark 2001). White perch feed almost exclusively on small invertebrates from their larval through juvenile stages, and then add fish to their diet as they reach maturity (PSEG 1984). Most male and female white perch mature at two and three years, respectively (Wallace 1971). Delaware Estuary white perch have been aged to ten years old and some may live longer than that, but white perch older than six years old are rare (Clark 2001).

White perch tolerate a wide range of environmental conditions, as would be expected of such a ubiquitous fish. White perch have been collected from water with temperatures ranging from 2.2° C (Rohde and Schuler 1971) to 35.5° C (Clark 1995) and at salinities ranging from 0 (Shirey 1991) to 35 (Clark 1995). White perch catch per unit effort is greatest in fresh and oligohaline waters of Delaware tidal tributaries (Clark 2001). White perch in the Delaware Bay, particularly juveniles, are caught more frequently at lower salinities; not many are caught at higher, ocean salinity levels. Both juveniles and adults are caught at lower temperatures (Oleynik 2020)(Fig 7.2.3.2). Smith (1971) collected white perch at a dissolved oxygen level of 2.2 ppm in Blackbird Creek, DE and Clark (1995) caught white perch at a dissolved oxygen level of 2.0 ppm in a high-level tidal impoundment near the Little River, DE, but neither report indicated whether the fish showed signs of stress at these low dissolved oxygen levels.

White perch were among the top five finfish species harvested commercially in Delaware from the time

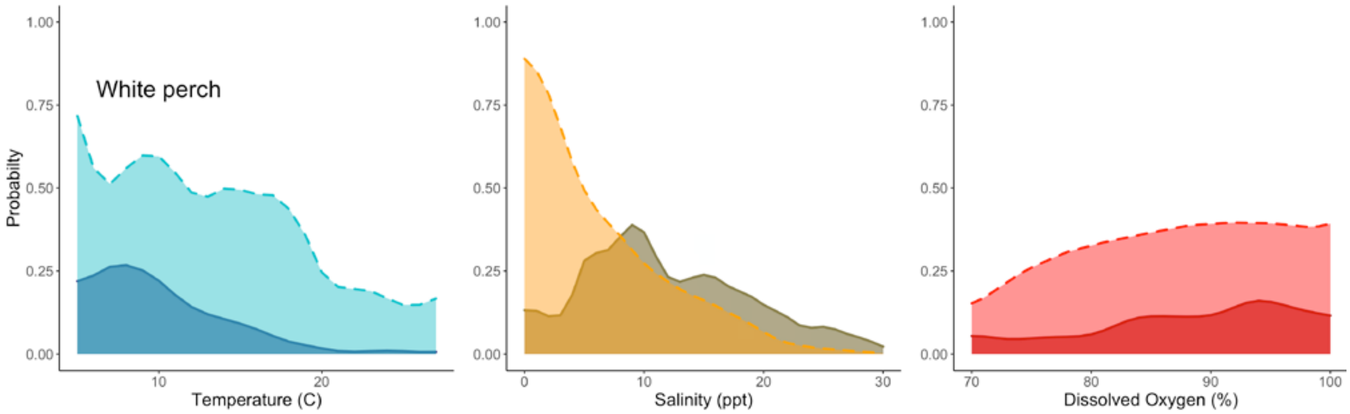


Figure 7.2.3.2 Probability of catching white perch in the Delaware Bay across a range of temperature (°C), salinity (ppt), and dissolved oxygen (% saturation). Solid line represents the 30-foot trawl used in DDFW's Adult Groundfish Research Trawl Survey (1966-2019) and dashed line represents the 16-foot trawl used in DDFW's Juvenile Finfish Research Trawl Survey (1978-2019) (Oleynik, 2020).

Delaware began requiring commercial catch reporting in the 1980s until the mid-2010s, but landings have since decreased substantially. Landings averaged 77,868 lbs during 2010 through 2015, with the highest landings, 157,947 lbs, reported in 2011, but only averaged 10,152 lbs during 2016 through 2020 (Fig 7.2.3.3). The decline in landings likely reflects a decline in the Delaware Estuary white perch population, but other factors also contributed to the landings decline. Although gourmets consider the white perch to be one of the finest tasting fish available, market demand for white perch has declined over the years, which has contributed to the decline in landings. Most fishing effort for white perch was expended during late fall through winter and into early spring. Delaware Bay was the source for most commercially-caught white perch, but substantial landings also came from Delaware River and several tidal tributaries. Commercial white perch landings in Delaware Estuary counties in New Jersey (Salem and Cumberland) averaged 24,333 lbs per year during 1995 through 2000, but, as with Delaware, landings have declined and averaged 7,652 lbs per year during 2015 through 2020.

White perch were among the top ten fish species harvested recreationally in Delaware annually since 2000. The annual, mean estimated recreational harvest during 2000 through 2020 was 97,618 pounds, with harvests greater than 150,000 pounds reported in 2010, 2011, 2014, and 2019 (personal communication from the National Marine Fisheries Service, Fisheries Statistics Division. 10/22/21). White perch are sought after for their mild, flaky meat; however, there are several consumption advisories for white perch in the Delaware Estuary (NJDEP 2021, DNREC 2018).

Description of Indicator

This indicator uses the white perch young-of-the-year (YOY) index derived from the Delaware Division of Fish and Wildlife's (DDFW) Juvenile Finfish Trawl Survey. The juvenile finfish trawl survey uses a 16′ trawl to sample 39 inshore Delaware Bay and River stations monthly during April through October. The YOY index is calculated as the geometric mean number of YOY white perch caught by the juvenile finfish trawl survey during June through October in Delaware Bay and River (Greco 2021). The white perch YOY index is an indicator of year-class strength and may indirectly be an indicator of future spawning stock abundance. The geometric mean white perch YOY index was only above the 1990 through 2016 time series geometric mean of 0.80 YOY white perch per tow twice during the past ten years of 2011 through 2020 (Fig 7.2.3.4).

Present Status

Delaware white perch commercial landings exceeded 100,000 lbs. in both 2009, 2010 and 2011; the only time landings have exceeded 100,000 lbs. for three consecutive years in the 1951 through 2020 time series, but landings have since declined and have been below the time series mean since 2015 (Fig 7.2.3.3). The YOY Index and commercial landings suggest the Delaware Estuary white perch population has declined since the last update of this report in 2017.

The white perch YOY index was below the time series mean YOY index value 80% of the time from 2011-2021, which suggests the Delaware Estuary white perch spawning population has had poor spawning success during this period. Although the white perch YOY index has not been used as a predictor of future spawning stock abundance or future commercial catches, the low YOY index values may be a factor in the decrease in commercial landings reported during 2013 through 2020.

Past Trends

Delaware white perch commercial landings were the longest-term time series available to assess past trends in white perch abundance (Fig 7.2.3.5), but white perch landings were affected by several factors other than the white perch population, such as fishing effort, conditions during the fishing season, gears used, market demand etc. Delaware white perch landings were high for several years during the 1950s, were low during most of the 1960s and 1970s, rose during the 1980s, were near or above the time series mean during the 1990s through 2015, and have since declined to levels not seen since in decades. While Delaware's precipitous decline in commercial landings since their historic peak in 2011 may be the result of poor fishing or market conditions during the following years, it may also be a result of poor recruitment to the fishery during this time as suggested by the low YOY index during most years between 2011 through 2020. Both the YOY index and the commercial landings suggest that the Delaware Estuary white perch population undergoes cyclical expansions and declines.

Future Predictions

The white perch's ability to inhabit almost all waters of the Delaware Estuary may buffer it from some of the extreme population fluctuations seen in other species, but habitat protection, particularly for areas of the Estuary in which white perch spawn, is important for the continued viability of this fish. Although the 8-inches is likely conservative, an establishment of a 6+ inch minimum size limit for white perch by all states in the Delaware Estuary will help ensure that most perch may spawn before they recruit to the fisheries.

Actions and Needs

The 8-inch (21 cm) minimum size limit for white perch, established by Delaware in 1995, has been effective in allowing almost all white perch to spawn at least once before recruiting to the fisheries. All states in the Delaware Estuary should establish an 8-inch minimum size for white perch to ensure that most white perch may spawn before they recruit to the fisheries.

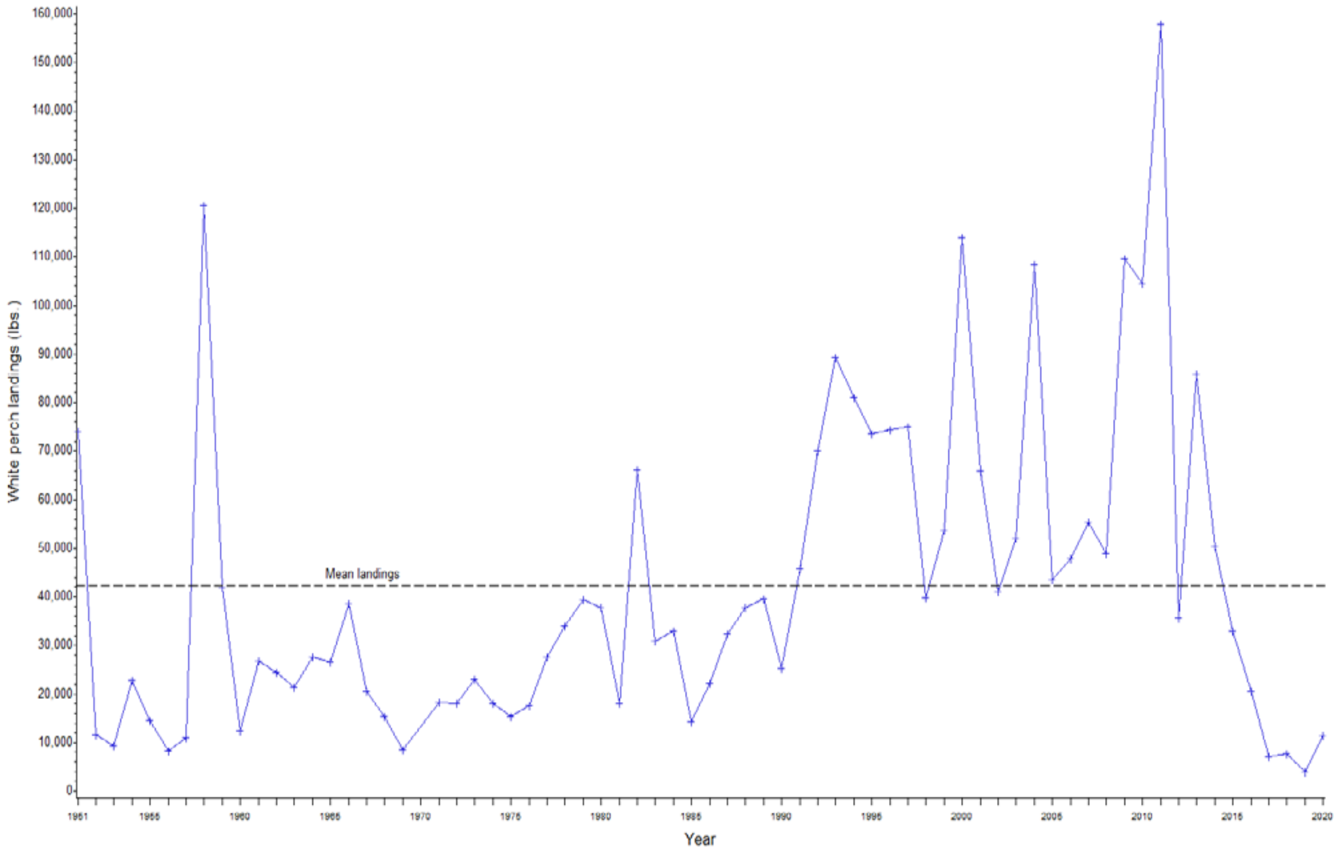


Figure 7.2.3.3 Delaware commercial white perch landings (lbs) during 1951 through 2020.

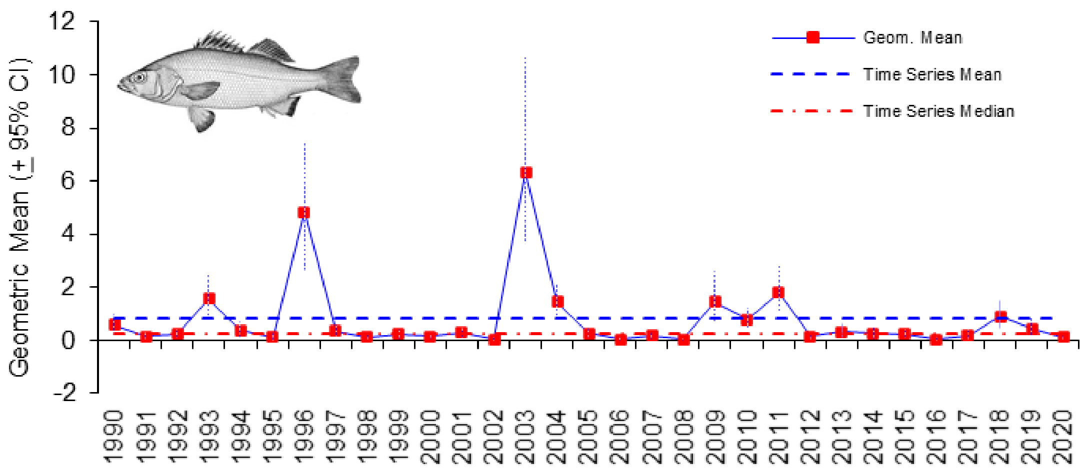


Figure 7.2.3.4 White perch YOY index (number of YOY white perch caught per trawl tow) from the DDFW Juvenile Trawl Survey from 1990 through 2020.

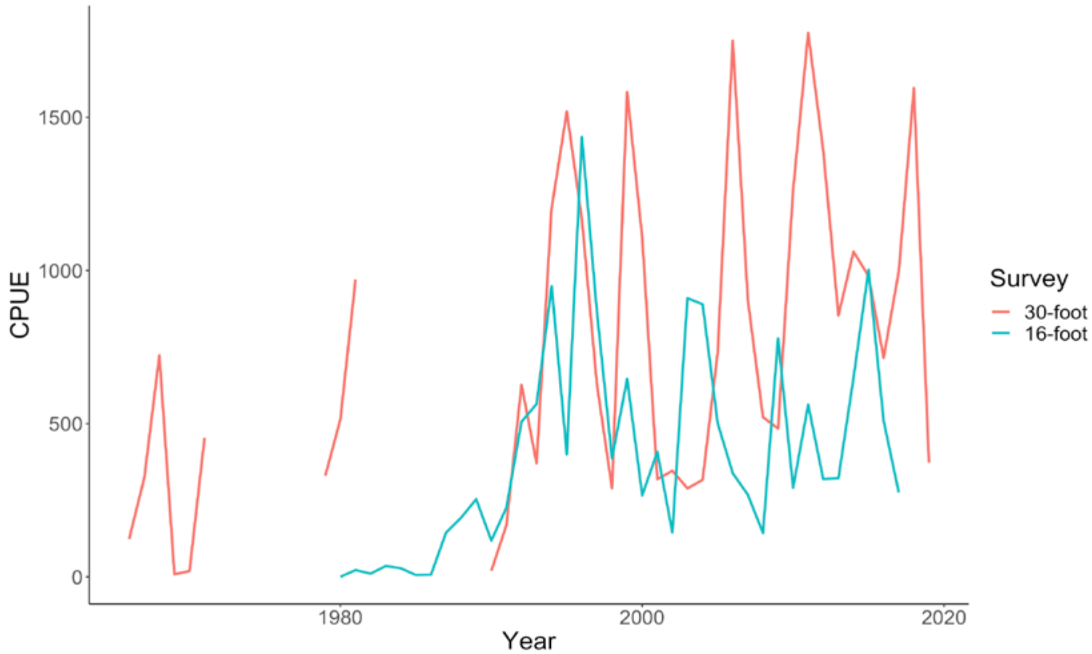


Figure 7.2.3.5 White perch CPUE (catch per unit effort) from DDFW's Adult Groundfish Research 30-foot Trawl Survey (1966-2019) and DDFW's Juvenile Finfish Research 16-foot Trawl Survey (1978-2019) (Oleynik, 2020).

White perch often spawn in areas of the Delaware River and in the upper reaches of Delaware Estuary tidal tributaries that have been subject to intense development pressure in the past 60 years. These are spawning habitats for many fish species in addition to white perch and these habitats should be protected.

Summary

White perch are one of the most abundant and widespread fish in the Delaware Estuary. White perch support important commercial and recreational fisheries. Although the white perch population in the Delaware Estuary seems to be maintaining itself, its abundance has declined over the past decade, so some basic management measures must be taken to ensure the population can rebound.

References

Clark, J. 1995. Impoundment estuarine interactions. Federal Aid in Fisheries Restoration Project F-44-R-5. Final report. Delaware Division of Fish and Wildlife, Dover, DE. 186 pp.

Clark, J. 2001. Anadromous species investigations. Federal Aid in Fisheries Restoration Project F-47-R-10. Job Number 1 - VI: White Perch Management Plan for the Delaware Basin. Delaware Division of Fish and Wildlife, Dover, DE. 54 pp.

de Sylva, D. P., F. A. Kalber, Jr., and C. N. Shuster, Jr. 1962. Fishes and ecological conditions in the shore zone of the Delaware River estuary, with notes on other species collected in deeper waters. Information Series, Publication 5. University of Delaware Marine Laboratories, Lewes, DE. 164 pp.

DNREC. 2018. Delaware Fish Consumption Advisories. Delaware Department of Natural Resources and Environmental Control & Delaware Division of Public Health.

Greco, M. 2021. Coastal finfish assessment survey. Federal Aid in Fisheries Restoration Project F-42-R-30. Annual report. Delaware Division of Fish and Wildlife, Dover, DE. 406 pp.

Jones, K.M.M., P.E. McGrath, P.E., K.W. Able. 2014. White perch Morone americana (Gmelin, 1789) habitat choice and movements: Comparisons between *Phragmites*-invaded and *Spartina* reference marsh creeks based on acoustic telemetry. Journal of Experimental Marine Biology and Ecology 455: 14-21.

Martin, C. C. 1976. Delaware lentic water management program. Federal Aid in Fisheries Restoration Project F-28-R-1. Annual report. Delaware Division of Fish and Wildlife, Dover, DE. 42 pp.

Miller, L. W. 1963. Growth, reproduction, and food habits of the white perch, Roccus americanus (Gmelin), in the Delaware River estuary. Master's thesis. University of Delaware, Newark, DE. 62 pp.

NJDEP. 2021 Fish Smart. Eat Smart. A Guide to Health Advisories for Eating Fish and Crabs Caught in New Jersey Waters. NJ Department of Environmental Protection & NJ Department of Health.

Oleynik, H. A. 2020. Changes in a Mid Atlantic estuary: trends and drivers of the fish and macroinvertebrate community in Delaware Bay. (Publication No. 1237124312) [Master's thesis, University of Delaware]. https://udspace.udel.edu/handle/19716/28649

Public Service Electric and Gas Company (PSEG). 1984. White perch (*Morone americana*): A synthesis of information on natural history, with reference to occurrence in the Delaware River and estuary and involvement with the Salem Generating Station. Public Service Electric and Gas Company, Newark, NJ. Appendix X. 629 pp.

Rhode, F. C. and V. J. Schuler. 1971. Abundance and distribution of fishes in the Delaware River in the vicinity of Artificial Island. Ichthyological Associates for Public Service Electric and Gas, Newark, NJ. 232 pp.

Shirey, C. 1991. Stream and inland bays fish survey. Federal Aid in Fisheries Restoration Project F-37-R-5. Final report. Delaware Division of Fish and Wildlife, Dover, DE. 231 pp.

Smith, B. 1971. An ecological study of the Delaware River in the vicinity of Artificial Island. Delaware Progress Report 3 (Part V). Ichthyological Associates for Public Service Electric and Gas, Newark, NJ. 291 pp.

Wallace, D. C. 1971. Age, growth, year class strength, and survival rates of the white perch, *Morone americana* (Gmelin), in the Delaware River in the vicinity of Artificial Island. Chesapeake Science 12: 205-218.

7.2.4 Striped Bass

Striped bass are large, predatory fish of the family Moronidae with dark horizontal stripes extending from the opercula to the caudal peduncle (Fig 7.2.4.1). This species has been found to inhabit tidal creeks and rivers, jetties, beaches and relatively open water in the Bay, River and ocean depending upon age and time of year. Striped bass are frequently referred to as rockfish because of a historic association with oyster reefs which were known as oyster rocks in the Mid-Atlantic region. Some younger, smaller individuals inhabit portions of the Delaware River Estuary year-round, unlike other potentially large predators such as weakfish, bluefish, large sharks and sea turtles which occur within the estuary seasonally. The Delaware Division of Fish and Wildlife, hereafter the Division, has conducted a survey to measure spawning stock biomass since 1996. Additionally, the Division has started to explore the use of acoustic telemetry to better identify migratory corridors and trends in habitat utilization. Preliminary results coupled with older tagging studies indicate that a large portion of the Delaware River spawning stock, primarily females, engage in a spring coastal migration to Southern New England and eastern Long Island; mature females spawn in the River prior to migrating up the coast annually. However, most tagged male bass remain in the estuary or nearby ocean waters year-round. Further, the Division has found evidence of exchange between the Chesapeake and Delaware Bays via the Chesapeake and Delaware Canal, indicating these fish use the canal as a migratory corridor between estuaries.

Figure 7.2.4.1 Scott Newlin of DDFW holds the largest striped bass collected in the DE spawning stock survey to date. This gravid female was caught in the Delaware River, in Delaware waters and measured 49 inches (125 cm). Photo credit: Ian Park.

Once considered extirpated by some biologists prior to the improvement of dissolved oxygen levels in the 1980s, the Delaware River population is now one of the major spawning stocks on the Atlantic coast, along with the Hudson River and Chesapeake Bay stocks. Management action for striped bass can be traced as far back as pre-Colonial times, when use of striped bass for fertilizer was banned. The Delaware River spawning stock declined greatly by the mid-twentieth century, in response to frequent, prolonged periods of hypoxia and anoxia in the late spring through early fall in the spawning grounds from Philadelphia through Wilmington reaches (ASMFC 1981; Kahn et al., in press), with some areas having persistent DO concentrations at zero during the summer months in the 1950s and 1960s (Sharp 2010). The Delaware River oxygen content increased during the 1970s and 1980s due to the Clean Water Act, which produced pollution reduction and upgrades to the sewage treatment plants along the River. During the 1980s, production of striped bass young of year increased gradually with a large surge in 1989 (Figure 7.2.4.2).

Striped bass feed on a variety of fishes and invertebrates throughout their life cycle with a general increase in prey size concomitant with individual growth. Younger bass feed primarily on smaller invertebrates including zooplankton, insects, worms, and amphipods. However, juveniles will also feed on fish larvae and small pelagic fish species as growth and ontogeny progress. Larger bass have been found to predominately prey on small pelagic fish species such as anchovies, river herring, Atlantic silverside and Atlantic menhaden (Griffin and Margraf, 2003) with secondary prey items including larger invertebrate species (e.g. blue crab, Atlantic rock crab and American lobster; Pruell et al. 2003; ASMFC 2013).

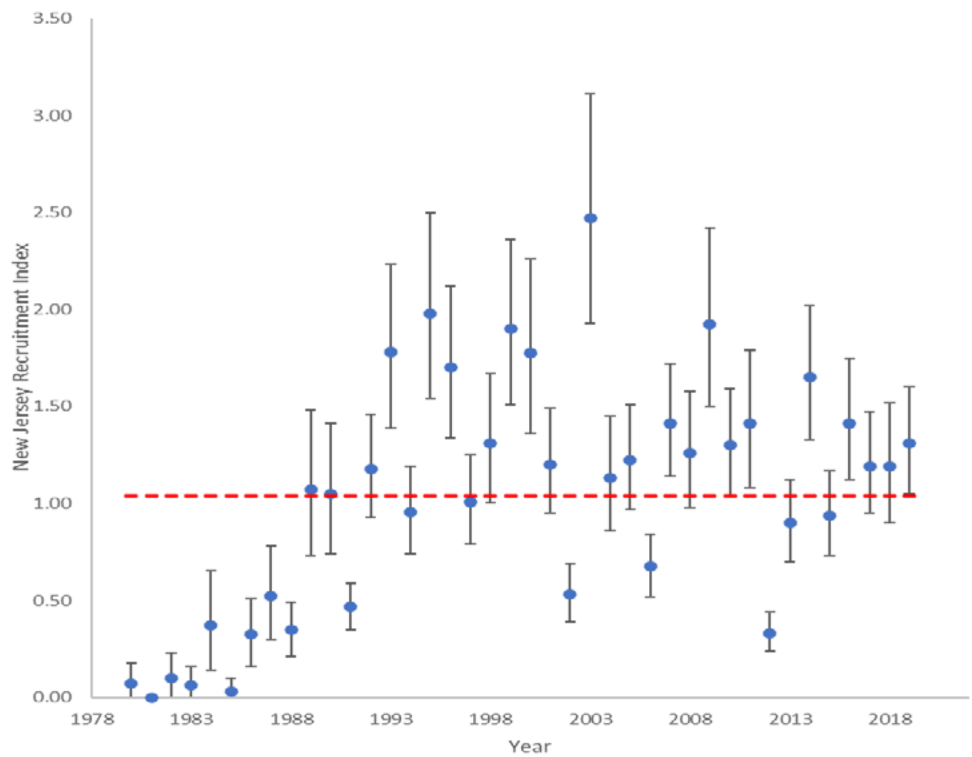


Figure 7.2.4.2 The annual Delaware River Recruitment Index, the geometric mean number of young-of-year bass caught per seine haul, with the time series mean shown by the red dashed line. Source: New Jersey Division of Fish, Game and Wildlife.

Striped bass spawning grounds exist in tidal fresh water in the Delaware River generally above detectable concentrations of salinity. However, the Division has observed spawning activity in nearby tidal waters with salinities ranging from 0.5 – 5.0 ppt. Similarly, a previous study demonstrated that striped bass successfully spawn within a narrow range of very low salinities (0.70-1.5 ppt) in the Chesapeake and Delaware Canal (Johnson and Koo 1975; Greene et al. 2009). The Delaware spawning survey usually finds more fish in April in Delaware waters from the Delaware Memorial Bridge up to the Delaware-Pennsylvania line. However, the New Jersey shore is typically where the majority of spawners congregate, along with the Cherry Island Flats, which are shoals in the River opposite of Wilmington, DE. As the season progresses into May, the temperature and salinity tend to increase, and spawning striped bass are more commonly collected in Pennsylvania waters up to the Philadelphia Navy Yard. Spawning usually terminates by the beginning of June. By September, young-of-year striped bass are several inches long, not typically exceeding four inches in total length before November. A recent study (Oleynik, 2020) examined the Division's annual trawl surveys data and determined striped bass are more frequently caught at lower salinity levels and lower temperatures across life history stage. Juveniles, in particular, are more likely to be caught in low salinity areas close to the mouth of the Delaware river and up into the river while adults are more likely to be caught in a greater range of salinities (Fig 7.2.4.3).

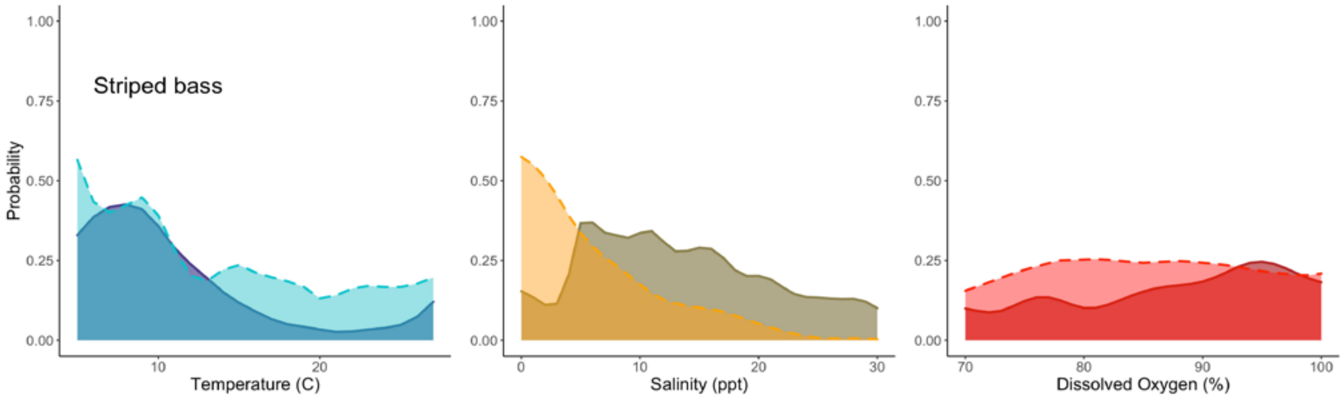


Figure 7.2.4.3 Probability of catching striped bass in the Delaware Bay across a range of temperature (°C), salinity (ppt), and dissolved oxygen (% saturation). Solid line represents the 30-foot trawl used in DDFW's Adult Groundfish Research Trawl Survey (1966-2019) and dashed line represents the 16-foot trawl used in DDFW's Juvenile Finfish Research Trawl Survey (1978-2019) (Oleynik, 2020).

In addition to being integral to the ecology of the estuary, striped bass are of economic benefit to both the States of Delaware and New Jersey. Delaware has a commercial fishery targeting the species. Currently, this fishery has the highest economic value of any of Delaware's commercial fin fisheries and is second only to the commercial blue crab fishery in terms of total ex-vessel value in the state. In 2019, Delaware commercial fishers generated more than $450,000 in ex-vessel value from striped bass landings (Fig 7.2.4.4). However, the State of New Jersey has banned the commercial harvest of striped bass for decades. Despite the difference between the commercial activities of the two states, both Delaware and New Jersey have a large recreational fishery (Fig 7.2.4.6), which ranks as one of the most popular in both states. The species is one of a few inshore species that can achieve big game size, with occasional fish exceeding 50 pounds (23 kg). However, there are several consumption advisories for striped bass in the Delaware Estuary (NJDEP 2021, DNREC 2018).

Description of Indicator

Two indicators from the Delaware River Estuary serve to measure the relative health of the striped bass population: the Delaware Spawning Stock Survey and the New Jersey Recruitment Survey. Both surveys use a geometric mean to provide a quantitative annual index of two biological parameters so we can compare performance through time. The first index, a geometric mean of the number caught per unit of electrofishing effort on the spawning grounds in April and May, is a measure of the reproductively capable abundance of the stock (Fig 7.2.4.5). The second index, the geometric mean of the number of young-of-year bass caught per seine haul, is a measure of the annual reproductive output of the stock (Fig 7.2.4.2).

Present Status

Survival to age one varies annually in response to a multitude of factors, including but not limited to, adult spawning intensity, hydrodynamic properties affecting transport and retention, growth, quantity and quality of larval prey and corresponding larval condition. A large year class at the young-of-year stage often results in a greater number of recruits into the fishery several years later. Regardless of the observed fluctuations between years, the overall status of the Delaware River spawning stock is trending downward and management strategies are being evaluated to reverse that trend. The most recent assessment has shown that the female spawning stock biomass has been below the management threshold since 2013, thus triggering new management actions to reduce catch.

Past Trends

The 2018 stock assessment found that striped bass are presently being overfished and are experiencing overfishing relative to the biological reference points (NEFSC 2018). However, the most recent assessment still shows that female spawning stock biomass and recruitment are above the levels seen in the 1980s which resulted in a harvest moratorium being imposed. Improvements to water quality and a successful management regime are cited as the principal reasons for the dramatic improvement in the population. Within the Delaware River Estuary, the annual Spawning Stock Survey index has varied from 0.48 to a high of 4.10, with a mean of 2.05 from 1996-2019 (Fig 7.2.4.4). The index was generally higher from 1996-2005 compared to the period from 2005-2019. However, a great deal of inter-annual variability is present in the index.

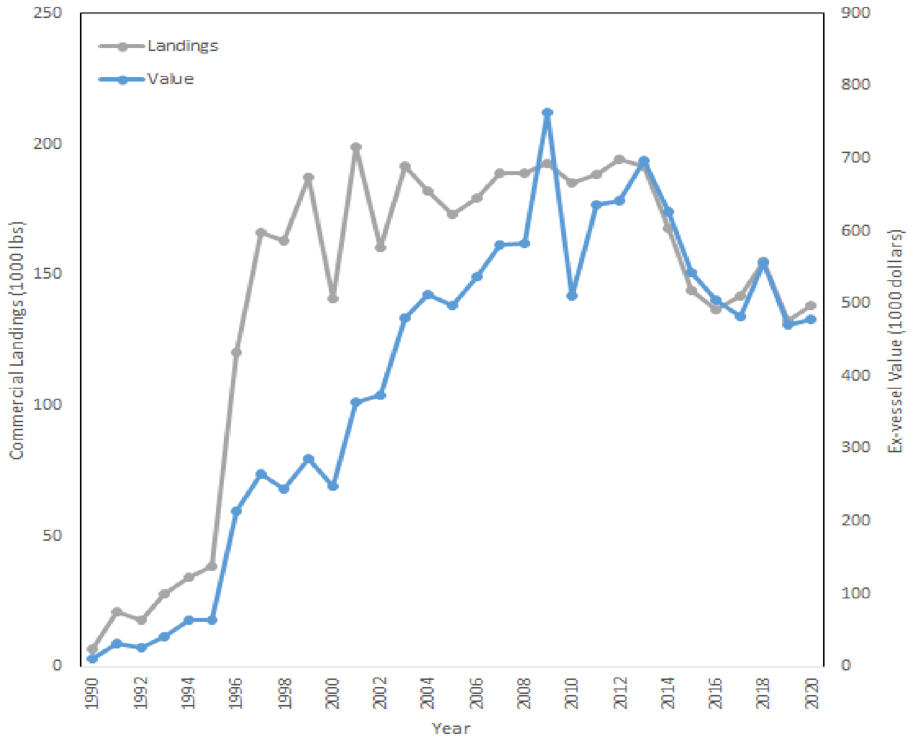


Figure 7.2.4.4 The total annual landings and ex-vessel value of commercially caught Atlantic striped bass in the State of Delaware.

The annual New Jersey Recruitment Survey index has ranged from 0.03 to 2.47, with a time series mean of 1.04 from 1980-2019 (Fig 7.2.4.5). Similar to the Spawning Stock Survey index, the recruitment index observes substantial inter-annual variability, but 2016-2019 the index remained above the time series mean.

Future Predictions

The striped bass fishery is managed under relatively conservative regulations to maintain high levels of spawning stock biomass. The current reference points were enacted to protect a coast-wide spawning stock biomass target of 125% of the 1995 levels (the year the species was declared recovered by the ASMFC). Aggressive management actions to rebuild the spawning stock biomass predicts that the female spawning stock biomass will be recovered by 2023.

When examining the number of striped bass caught per recreational trip in Delaware, a similar pattern of high inter-annual variability compared to the Delaware Spawning Stock Survey becomes apparent (Fig 7.2.4.6), demonstrating the inherent irregularity in annual harvest. Despite the second lowest value since the turn of the century being observed in 2016, the past several years have shown an increase in striped bass being caught per recreational trip. The recreational catch per trip was generally higher in the last twenty years than the time series average suggesting that the species has been managed to maintain relatively high levels of productivity.

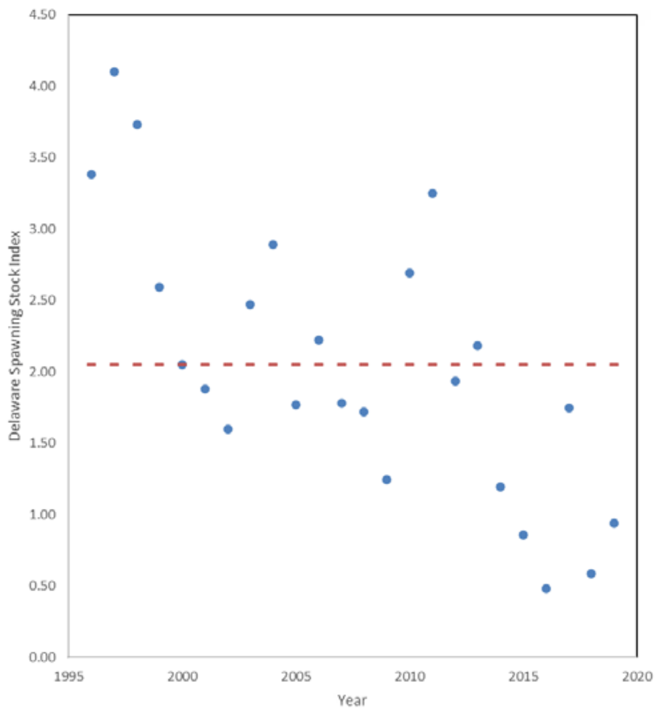


Figure 7.2.4.5 The annual Delaware Spawning Stock Survey index with the time series mean shown by the red dashed line.

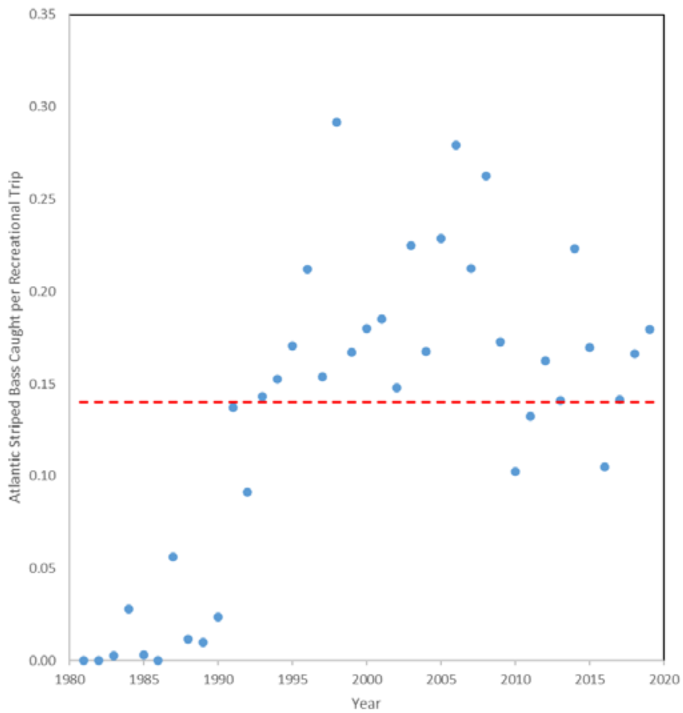


Figure 7.2.4.6 Annual index of recreationally caught Atlantic striped bass caught per trip with the time series mean shown by the red dashed line.

Actions and Needs

In order to ensure sustainable levels of future harvest, we need to continue monitoring long term trends in biomass and recruitment, responding when necessary with management action. The most recent stock assessment reported that overfishing was occurring. One of the main culprits of the overfishing finding was a new estimate of recreational discard mortality being calculated. Several management actions have been implemented to decrease discard mortality, including a coastwide mandate on the use of circle hooks, and limiting recreational fishing during times of the year when water temperatures are at their peak.

Summary

Striped bass are large, predatory fish that are important to the ecology of the Delaware River Estuary and the economy of the surrounding states. In response to conservative historical management measures and improved habitat availability and thanks to enhanced water quality conditions, the species has rebounded from historic lows to new highs in abundance. It has come to represent a significant management success and continues to provide a sustainable fishery resource. In order to continue to sustainably harvest striped bass, we will need to continue long term monitoring programs and advance our mathematical modeling to better approximate the dynamics of an ever-changing environment.

References

ASMFC. 1981. Interstate Fisheries Management Plan for the Striped Bass of the Atlantic Coast from Maine to North Carolina. A report prepared by the State of Maryland Department of Natural Resources, Tidal Fisheries Division as part of the Interstate Fisheries management Program. 329 p.

ASMFC. 2013. Striped Bass Benchmark Stock Assessment for 2013. A report prepared by the Striped Bass Technical Committee for the Atlantic Striped Bass Management Board. 967 p.

DNREC. 2018. Delaware Fish Consumption Advisories. Delaware Department of Natural Resources and Environmental Control & Delaware Division of Public Health.

Greene, K. E., J. L. Zimmerman, R. W. Laney, and J. C. Thomas-Blate. 2009. Atlantic coast diadromous fish habitat: A review of utilization, threats, recommendations for conservation, and research needs. Atlantic States Marine Fisheries Commission Habitat Management Series No. 9, Washington, D.C.

Griffin, J. C. and F. J. Margraf. 2003. The diet of Chesapeake Bay striped bass in the late 1950s. Fisheries Management and Ecology 10: 323–328.

Johnson, R. K., and T. S. Y. Koo. 1975. Production and distribution of striped bass (Morone saxatilis) eggs in the Chesapeake and Delaware Canal. Chesapeake Science 16: 39-55.

Kahn, D. M., P. Himchak, Cl. A. Shirey, M. Fisher, T. Baum, R. Allen, G. Murphy, M. Kaufman and R. W. Miller. In press. Restoration of the Delaware River Stock of Striped Bass.

Kahn, D. M., R. W. Miller, C. A. Shirey, and S. Grabowski. 1998. Restoration of the Delaware River stock of striped bass. Delaware Division of Fish and Wildlife, Dover.

NJDEP. 2021 Fish Smart. Eat Smart. A Guide to Health Advisories for Eating Fish and Crabs Caught in New Jersey Waters. NJ Department of Environmental Protection & NJ Department of Health.

Northeast Fisheries Science Center (NEFSC). 2018. 66th Northeast Regional Stock Assessment Workshop (66th SAW) Assessment Report: Part B. Striped Bass Stock Assessment. Northeast Fisheries Science Center Reference Document 19-08 (https://www.nefsc.noaa.gov/publications/crd/crd1908/)

Oleynik, H. A. (2020). Changes in a Mid Atlantic estuary: trends and drivers of the fish and macroinvertebrate community in Delaware Bay. (Publication No. 1237124312) [Master's thesis, University of Delaware]. https://udspace.udel.edu/handle/19716/28649

Pruell, R. J., B. K. Taplin and K. Cicchelli. 2003. Stable isotope ratios in archived striped bass scales suggest changes in trophic structure. Fisheries Management and Ecology 10: 329–336.

Rugolo L.J., Crecco V.A., and M.R. Gibson. 1994. Modeling stock status and the effectiveness of alternative management strategies for Atlantic coast striped bass. Washington (DC): ASMFC. A Report to the ASMFC Striped Bass Management Board. 30 p.

Sharp, J. 2010. Estuarine oxygen dynamics: What can we learn about hypoxia from long time records in the Delaware Estuary? Limnology and Oceanography 55, 535-548.

7.2.5 Weakfish

Weakfish (*Cynoscion regalis*) is a marine fish that is member of the drum family Sciaenidae. Locally, weakfish often go by other common names such as gray trout or sea trout; although they are of no relation to the "true" trout family Salmonidae. Weakfish occur along the Atlantic coast from Nova Scotia, Canada to southeastern Florida, but are most common from New York to North Carolina. Weakfish once dominated Delaware's recreational and commercial landings in the 1970s and 1980s and the species was named the Delaware State Fish in 1981. With the onset of spring and the warming of coastal waters, adult weakfish begin a northerly inshore migration from offshore waters off the Carolina coast to nearshore coastal waters and estuaries to spawn. Spawning in the Delaware Estuary occurs in the shallows and on shoals in the middle and lower Bay and generally begins in May with sporadic, secondary spawning taking place throughout the summer. Larger weakfish, over several pounds, which were extremely common in the 1970s and 1980s and less so in the later 1990s, spawn in the spring and then leave the Bay. These larger fish may then migrate to southern New England. Younger, smaller adult weakfish tend to stay in the Bay all summer and do spawn more than once given their propensity to reach sexual maturity by age 1 (Nye et al. 2008). From late spring through early fall, young-of-the year (YOY) weakfish are found throughout the estuary from the lower Bay up into the Delaware River (Fig 7.2.5.1). In recent years, age 0 weakfish have started to appear in surveys in mid to late June. Young weakfish are fast growing, often reaching a length of six to eight inches (15-20 cm) before leaving the Bay in the fall to migrate south as water temperatures decline. Due to the above life history characteristics, weakfish in Delaware Bay have a strong relationship to temperature and are much more likely to be caught at higher temperatures, in warmer months across life history stage. While adult weakfish are caught across a range of salinity, juvenile weakfish are more likely to be caught at higher salinity levels (Oleynik 2020) (Fig 7.2.5.2).

Figure 7.2.5.1 Young-of-the-year weakfish collected in the Delaware Bay. Photo credit: Jenny Shinn, Rutgers University.

Weakfish feed on a variety of prey ranging from invertebrates like crustaceans, and mollusks to various fish species. Younger fish feed on mysid shrimp, also known as opossum shrimp, and sand shrimp, which can be very abundant in mats of grass detritus washed out of marshes. Larger weakfish are more piscivorous, feeding mainly on other fish, primarily members of the Clupeidae family like Atlantic menhaden. Larger weakfish are also cannibalistic, feeding on YOY weakfish (Merriner 1975; Thomas 1971). Due to these and other life history traits, there are several human consumption advisories for weakfish in the Delaware Estuary (NJDEP 2021, DNREC 2018).

Weakfish abundance and catches have been declining coast-wide since the late 1990s. The 2019 Weakfish Stock Assessment Update indicates that Weakfish continue to be depleted and have been since 2003 (ASMFC 2019). A coast-wide stock assessment completed in 2006 found natural mortality had increased beginning in 1996, eventually causing the stock to decline (ASMFC 2006). That assessment developed a hypothesis that predation and possibly competition from striped bass and spiny dogfish caused the large increase in natural mortality that led to the weakfish decline. Although coast-wide YOY indices remained relatively steady with low levels of adult harvest, the population did not show signs of recovery. A stock assessment conducted in 2009 examined other potential factors that could affect natural mortality in addition to predation, including seasonal variables such as water temperature, and large-scale, environmental phenomena including the Atlantic Multidecadal Oscillation (NEFSC 2009).

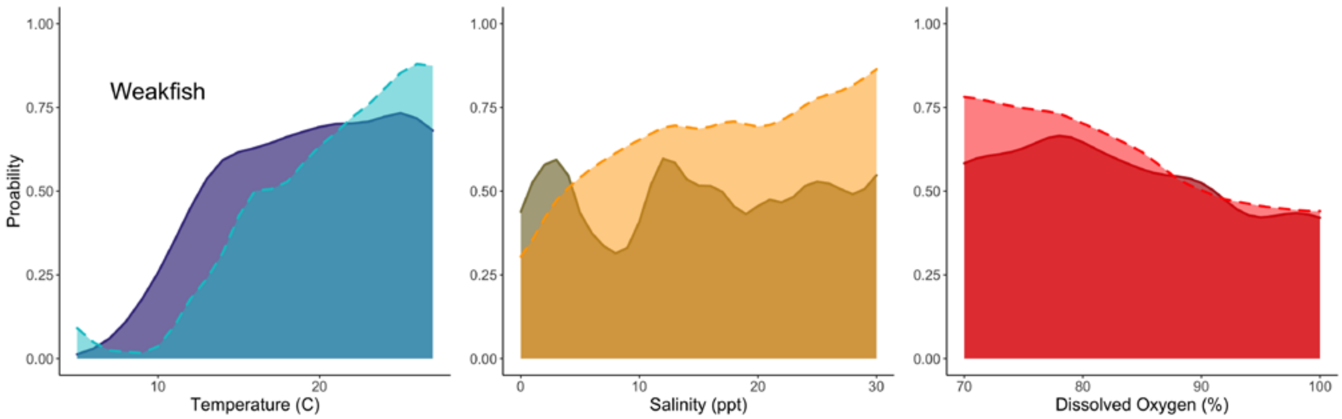


Figure 7.2.5.2 Probability of catching weakfish in the Delaware Bay across a range of temperature (°C), salinity (ppt), and dissolved oxygen (% saturation). Solid line represents the 30-foot trawl used in DDFW's Adult Groundfish Research Trawl Survey and dashed line represents the 16-foot trawl used in DDFW's Juvenile Finfish Research Trawl Survey from years 1978 to 2019 (Oleynik 2020).

The most recent peer reviewed assessment conducted in 2016 utilized several methods to estimate time-varying mortality including the relationship between catch and the Atlantic Multidecadal Oscillation (ASMFC 2016). As with the 2016 assessment, the 2019 assessment update supported the hypothesis that natural mortality has increased since 1996 but was unable to determine the underlying cause or causes. Tagging work also estimated an increasing trend in natural mortality and suggested that increased predation from bottlenose dolphin was driving that trend (Krause 2019).

Description of Indicator

The primary indicator of weakfish productivity in the Delaware River Estuary is the mean catch per nautical mile of weakfish in the adult groundfish research trawl survey, conducted using a 30-foot otter trawl net in Delaware Bay by the Delaware Division of Fish and Wildlife. This survey has been conducted since 1966 (1966-71, 1979-84 and 1990-present) and is conducted monthly from March through December at nine fixed stations in Delaware Bay.

Weakfish relative abundance in the 30-foot trawl survey has generally followed a declining trend since 1996 (Fig 7.2.5.3) and total mortality estimates have correspondingly increased. Despite annually ranking among the top 1 or 2 (by number) fish species encountered in the trawl survey, weakfish abundance remains below the historical average for the survey (Greco 2021). The age structure of weakfish remains truncated similar to the age structure found in the early 1990s with 82% of survey catch being less than age two.

A secondary indicator of weakfish productivity in the Delaware River Estuary is the index of relative abundance of YOY weakfish as measured by the Delaware Division of Fish and Wildlife's Juvenile Finfish Research Trawl Survey. This survey has been conducted annually since 1980 and samples monthly from April through October at 33 fixed stations in the Delaware Bay and River utilizing a 16-foot semi-balloon otter trawl. Abundance of YOY weakfish declined in 2020 relative to 2019 and remains below the time series mean for the fifth consecutive year (Fig 7.2.5.4) (Greco 2021). Weakfish annually rank among the top species collected in the juvenile trawl. However, as with the relative abundance in the 30-foot trawl survey, the YOY index for weakfish has also followed a declining trend since 1996 (Fig 7.2.5.4).

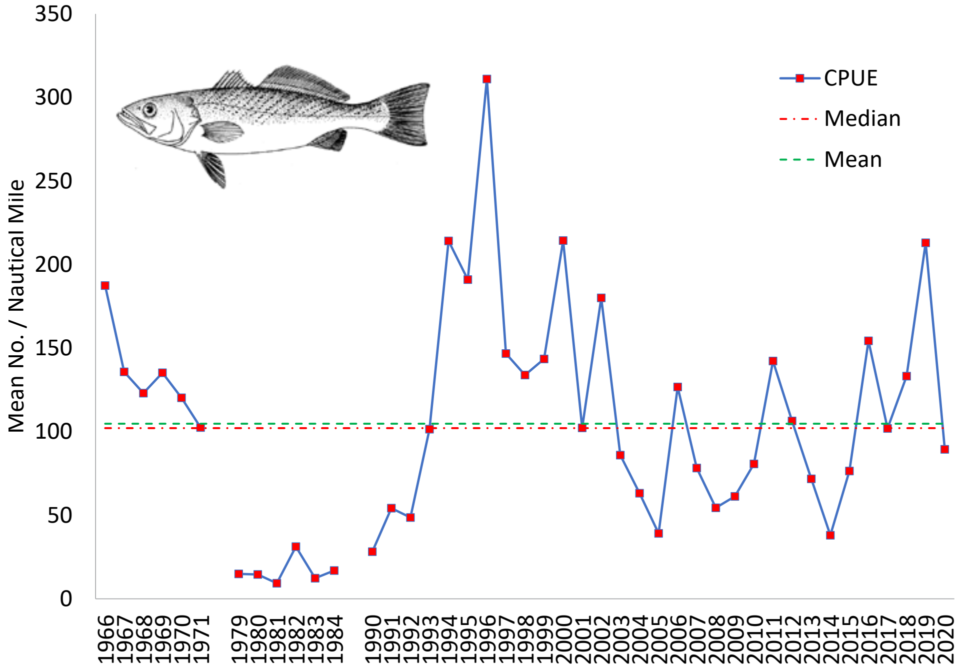


Figure 7.2.5.3 Weakfish relative abundance (mean number per nautical mile), time series (1966 – 2020) mean and median as measured in 30-foot trawl sampling in Delaware Bay.

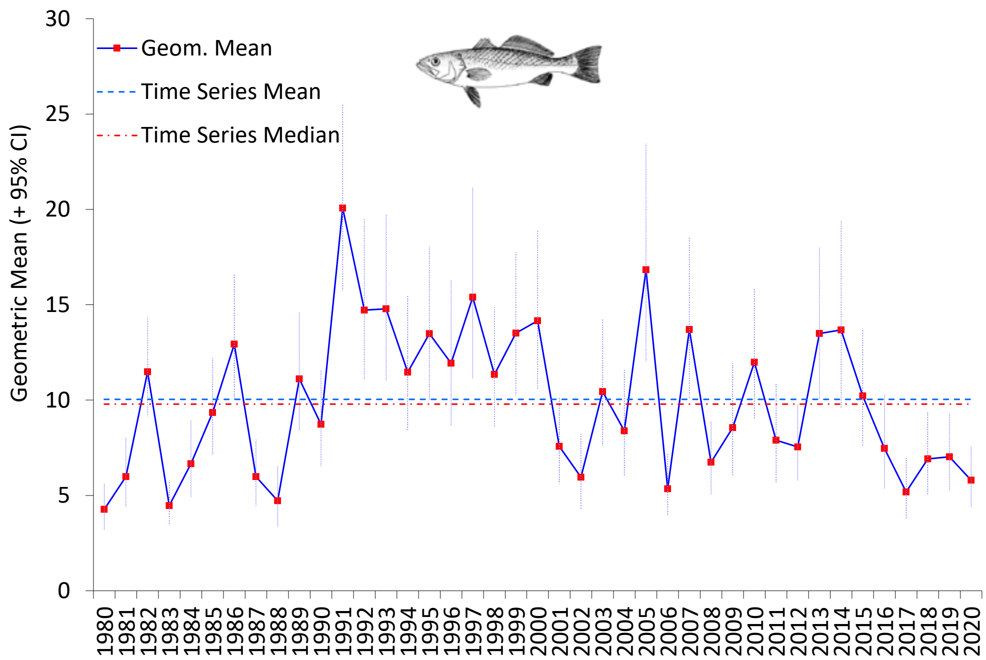


Figure 7.2.5.4 Relative abundance of young of the year (YOY) weakfish from 1980 through 2020, time series (1980-2019) mean and median as measured by 16-foot trawl sampling in the Delaware Estuary.

Present Status

Despite a top-5 adult abundance level in Delaware's adult trawl survey in 2019; YOY recruitment in the estuary continues to be below the historical average, the coast-wide weakfish stock is considered depleted and has been since 2003 as detailed in the latest peer reviewed stock assessment (ASMFC 2016). Under the new reference points proposed in the latest assessment, the stock is considered depleted when the coast-wide estimated spawning stock biomass is below 30% of the estimated average biomass over the period 1982-2014. The 2019 assessment update estimated the spawning stock biomass to be 4.24 million pounds in the terminal year of the assessment update (2017). Despite slight increases in total abundance and spawning stock biomass in recent years, the stock is well below the spawning stock biomass threshold and has been since 2003. Results of the latest assessment update indicated that total mortality ($Z = 1.45$) was above the target ($Z = 1.03$) and threshold ($Z = 1.43$). However, the assessment indicated that natural mortality has been increasing since the mid-1990s. As such, the weakfish population has been experiencing high levels of total mortality, which has prevented the stock from recovering (ASMFC 2016).

Past Trends

Weakfish were at moderate abundance prior to the 1970s, when they began an explosive rise in abundance and size. By the late 1970s, Delaware Bay had become famous throughout the Mid-Atlantic region as a destination for catching trophy-sized weakfish in the spring spawning run. By the late 1980s, this fishery declined somewhat; however, the Delaware commercial fishery landed over 200,000 pounds of weakfish as late as 2001.The Atlantic States Marine Fisheries Commission imposed significant fishery restrictions coast-wide in the mid-1990s, and, in response, abundance and catches initially began to increase through the late 1990s, before declining during the 2000s. So, although the fishery has not regained the high catches and trophy sizes seen in the 1970-1980 period, it did produce higher catches of legal-size weakfish for many in the mid- to late 1990s, before its ultimate decline. By 2007, Delaware commercial landings declined to 27,000 pounds. By 2010, no directed fishery was allowed on the Atlantic coast; only a small amount of bycatch was legal.

Future Predictions

The 2016 stock assessment indicated that in recent years, slight increases coast-wide in total abundance, spawning stock biomass and recruitment of age 1 fish have occurred. However, the stock remains well below the recommended threshold.

Actions and Needs

None.

Summary

Currently, weakfish reproduction continues at moderate levels. Survivorship to catchable size, however, has declined greatly, to the point that catches of legal-size weakfish are uncommon in Delaware Bay. The cause of the decline has been linked to factors such as predation by striped bass and spiny dogfish, competition with striped bass for Atlantic menhaden and changes in environmental conditions (ASMFC 2006, NEFSC 2009). However, the most recent stock assessment (ASMFC 2016) claimed that explicit factors leading to the decline of weakfish require more investigation.

References

ASMFC. 2006. Weakfish Stock Assessment Report, 315 pp. Available at http://www.asmfc.org/uploads/file/2006WeakfishStockAssessment.pdf

ASMFC. 2016. Weakfish Stock Assessment and Peer Review Report. 435p.

DNREC. 2018. Delaware Fish Consumption Advisories. Delaware Department of Natural Resources and Environmental Control & Delaware Division of Public Health.

Greco, M. J. 2019. Coastal finfish assessment survey. Federal Aid in Fisheries Restoration Project F-42-R-30-2. Annual Report. Delaware Division of Fish and Wildlife, Dover.

Krause, J.R. 2019. Sources, Magnitude, and Timing of Weakfish Mortality across the U.S. East Coast [dissertation]. Raleigh, NC. 22 p.

Merriner, J. V. 1975. Food habits of the Weakfish, Cynoscion regalis, in North Carolina waters. Chesapeake Science. 16: 74-76.

NEFSC. 2009. 48th Northeast Regional Stock Assessment Workshop (48th SAW) Assessment Summary Report. US Dept Commerce, Northeast Fisheries Science Center Ref Doc. 09-10; 50 p. Available from: National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-1026, or online at http://www.nefsc.noaa.gov/nefsc/publications/

NJDEP. 2021 Fish Smart. Eat Smart. A Guide to Health Advisories for Eating Fish and Crabs Caught in New Jersey Waters. NJ Department of Environmental Protection & NJ Department of Health.

Nye, J.A., Targett, T.E., and Helser, T.E. 2008. Reproductive characteristics of weakfish in Delaware Bay: implications for management. N Am J Fish Manage. 28: 1-11.

Oleynik, H. A. (2020). Changes in a Mid Atlantic estuary: trends and drivers of the fish and macroinvertebrate community in Delaware Bay. (Publication No. 1237124312) [Master's thesis, University of Delaware]. https://udspace.udel.edu/handle/19716/28649

Thomas, D. L. 1971. The early life history and ecology of six species of drum (Sciaenidae) in the lower Delaware River, a brackish tidal estuary. Ichthyological Associates. Delaware Progress Report 3 (Part III). 247 pp.

7.2.6 American Eel

American eels (*Anguilla rostrata*) (Fig 7.2.6.1) are very unique among fishes of the Delaware River Estuary. Being catadromous, eels spend most of their lives in fresh and estuarine water, only returning to the open ocean to spawn. It is believed that all American eels spawn in the Sargasso Sea off the southern coast of the United States (Miller et. al. 2014). American eels are also semelparous, meaning they spawn once and die. Larval stage eels (leptocephali) hatched from buoyant eggs are leaf-like in shape and drift on ocean currents westward to the eastern Gulf of Mexico and Atlantic coast of the U.S. All American Eels are currently believed to spawn in one aggregation, and therefore offspring, with few exceptions, are genetically indistinguishable (Cote et. al. 2012). Larval eels are not believed to return to the particular waters from which their parents came, but rather migrate up the coast with the Gulf Stream and move inshore in a randomized fashion. Recent findings suggest that ingressing juvenile eels are capable of conspecific cuing, using olfaction to select waters that are already occupied by other eels (Schmucker et. al. 2016).

Figure 7.2.6.1 American eel (yellow phase) collected in the Delaware Estuary. Photo credit: Jennifer Pyle, NJDEP.

Figure 7.2.6.2 American eels in the elver and glass phases collected in the Delaware Estuary. Photo credit: Jennifer Pyle, NJDEP.

As leptocephali near freshwater environs in the late winter to early spring, they metamorphose into clear, very small eels known as glass eels. Some eels will move far up into non-tidal portions of Delaware River tributaries, often very small streams. Others remain in brackish water in tidal tributaries of the Bay and River. Once glass eels reach freshwater, they undergo pigmentation, eventually reaching the "yellow" phase of their life history, named as such for their yellow-green coloration. American eels spend most of their life in the "yellow" stage, residing in tributaries and the Delaware River for up to 30 years (Able and Fahay 1998) until they reach sexual maturity and the last stage of their life cycle, the "silver" phase (Fig 7.2.6.2). A number of physiological changes occur during the silvering process: the skin thickens, the body fattens, the shape and color of the pectoral fins change, the digestive tract degenerates, and the eyes become enlarged. These changes are thought to be beneficial for migration through the open ocean back to the Sargasso Sea (Facey and Van den Avyle 1987).

Delaware and New Jersey have, historically, had significant commercial fisheries for yellow eels, prosecuted in the Bay and it's tidal tributaries. Delaware landings ranged above 100,000 pounds until 2008 when shortages in bait supply, namely female horseshoe crab, suppressed more recent annual landings (Fig 7.2.6.3). Eels are used by recreational fishers for bait to catch striped bass and large pelagic fishes such as tunas and billfish. A fairly robust bait market exists in the southeastern United States as well for cobia, catfish, and land-locked striped bass. Size of bait eels varies depending upon the quarry targeted but all must meet the legal minimum size of nine inches (23 cm). The second market for eels is a food market both in this country and in Europe, where they are regarded as a delicacy. Eels are shipped live or frozen to Europe. There are several consumption advisories for eating American eel meat harvested in the Delaware Estuary (NJDEP 2021, DNREC 2018).

Delaware's eel fishery is reliant on a source of good bait; fishers say that much of the year, the only bait that will catch significant numbers of eels is female horseshoe crabs. With the restrictions on horseshoe crab harvest along the Atlantic coast, availability has dwindled and the price of bait has increased considerably. The price of bait has negatively impacted Delaware's eel landings in two ways. First, the catchability of other baittypes including fish wracks and blue crabs is not as great as it is for horseshoe crabs. Secondly, many eel fishermen accustomed to catch rates of pots employing horseshoe crab baits have left the fishery presumably due to a decline in profitability. As a result, a sharp decline in commercial landings have been observed since regulations were enacted (2007) banning the harvest of female horseshoe crabs in the Delaware Bay region (Fig 7.2.6.3). In more recent years, aquaculture facilities in Europe and Asia have supplanted the need for wild-caught eel product.

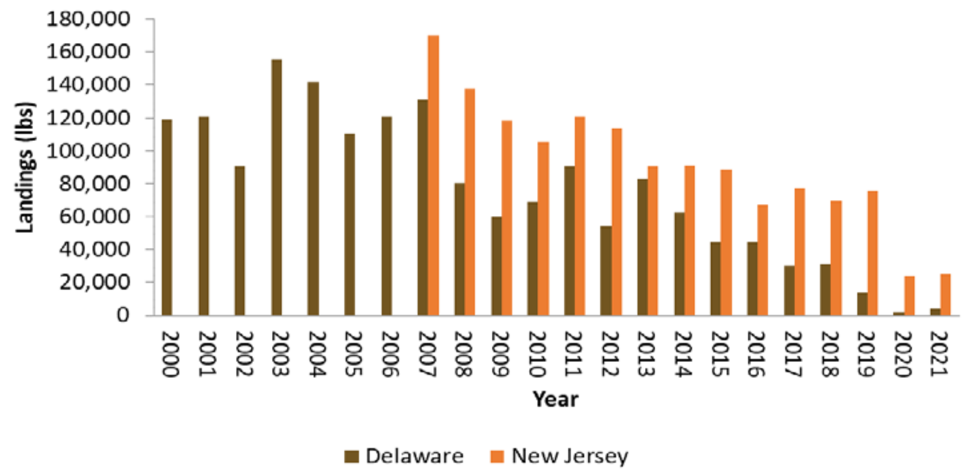


Figure 7.2.6.3 American eel commercial landings for the years: 2000 – 2021 in Delaware and 2007-2021 (data from 2021 are preliminary) in New Jersey.

The American eel population is managed under regulations developed by the Atlantic States Marine Fisheries Commission. Coast-wide populations have declined in recent years, due to several potential factors including the relatively slow rate of maturation, high levels of stage specific mortality, fishing mortality on a wide range of year classes prior to spawning, continued habitat loss in the form of dams and other impediments to upstream migration, and changes in oceanic conditions. Additionally, the introduced Asian parasite, *Anguillicola crassus*, is now wide-spread in the American eel population, as it has been documented in every State on the Atlantic coast. Relatively little is known about the overall effects this parasite has on the population, but the fact that it weakens, and in some cases, totally destroys the eel's swim bladder intuitively equates to a negative impact on infected eels. The United States Fish & Wildlife Service conducted a review of the species status in order to determine whether it should be listed under the Endangered Species Act (ESA). The Service had previously concluded in 2007 that there was no basis for listing eels as threatened or endangered. After reviewing the data again in 2015, the USFWS decided that listing the American eel under the ESA was again not warranted (USFWS 2015).

Description of Indicator

The index of eel relative abundance is developed from 13 trawl survey stations in the lower Delaware River by the Delaware Division of Fish and Wildlife Juvenile Finfish Trawl Survey. The net is a 16-ft (4.8-m) semi-balloon trawl with a 0.5-in (1.3-cm) cod end liner towed by 62-ft (19-m) R/V First State. The geometric mean catch-per-tow, using catch data collected from April through June from 1982-2021, is used to estimate an index of abundance (Fig 7.2.6.4). Catch typically consists of eels from ages 0 to 7, with 3 years of age representing the most frequent age observed in the catch (DDFW unpublished data). All eels captured in this survey are yellow-phase.

A linear regression line was found to best represent the index as a function of year, which explains a statistically significant portion of the annual variability (P = 0.01, R^2 = 14.6%; Fig 7.2.6.4). Such patterns raise the possibility of decadal-scale oscillations in climate affecting recruitment into the stock. Changes in cyclical climatic events have been found to affect patterns of abundance through cumulative effects on ecosystem processes including, but not limited to spawning success, primary productivity and larval transport (Nye et al. 2014).

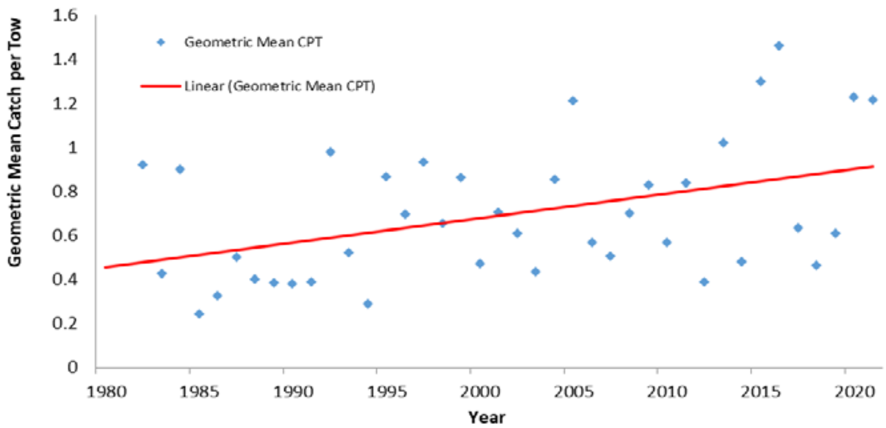


Figure 7.2.6.4 Index of relative abundance of American eels in the tidal Delaware River, based on catch per tow at 13 stations from April –June from 1982-2021. The index is the geometric mean catch per tow. The predicted line was fitted as a linear regression, P = 0.01, R^2 = 14.6%.

Present Status

Eel abundance in the estuary as represented by the index, has generally increased over the time series with the last four years exhibiting the highest abundance estimates of the last decade (Fig 7.2.6.4). All indications from anecdotal accounts from fishermen and biologists are that eel abundance is currently very high. Glass eel abundance surveys in Delaware and New Jersey have documented above average recruitment over the past decade. Although these surveys do not occur within the Delaware River watershed, they generally speak to recruitment trends in the region.

Past Trends

Abundance declined somewhat during the 1980s, but increased to higher levels in the mid-2000s.

Sykes and Lehman (1957) reported that eel weirs were so numerous on the non-tidal Delaware River that they trapped and killed many, if not most, young-of-year shad migrating downriver in early fall. These weirs targeted the so-called silver eel stage, which are adults migrating down river and out to spawn in the Sargasso. Smiley (1884) described "hundreds of traps" in the River between Lackawaxen, PA and Hancock, NY. The relatively high number of fishing weirs would suggest much heavier fishing mortality occurred on silver eels many decades ago. In recent years, nine weirs have been operating in the Delaware River, in New York. Due to the panmictic nature of the American eel population, high fishing mortality in the upper Delaware River may not affect the number of new recruits arriving to the Delaware River Basin from the Sargasso Sea annually.

Future Predictions

There are no apparent basis for future predictions, but the coast wide nature of the spawning aggregation suggests that even if the Delaware estuary spawning numbers would decline, the estuary could still receive relatively high levels of annual recruits.

Actions and Needs

Although the main stem of the Delaware River is un-dammed, hundreds of dams still block passage along its tributaries; many are low head dams under private ownership and in poor operating condition. In addition, there are thousands of culverts for roads that cross the tributaries. And in many areas the riparian forested buffer along the streams has been removed, leaving the stream exposed to sun and dramatically increased non-point source sediment and pollution run off. Fish passage and riparian restoration would help improve habitat for eel by increasing connectivity and improving in-stream habitat by providing shade and structure in these tributaries.

Summary

Eel populations in the estuary declined in the late 1980s and increased in the mid-2000s. This increasing trend has continued through to 2020. Annual recruitment in Delaware has been stable at a high level for the past four years. Harvest controls put in place through interstate management of the resource should bode well for sustainability of the fishery. Habitat initiatives such as dam removal, when practical, open up quality habitat in the upper portions of Delaware River tributaries.

Acknowledgements

New Jersey commercial landings data were provided courtesy of Jennifer Pyle at NJ Department of Environmental Protection, Division of Fish and Wildlife.

References

Able, K. W., & Fahay, M. P. 1998. First year in the life of estuarine fishes in the middle Atlantic Bight. Rutgers University Press.

Cote, C. L., P. A. Gagnaire, V. Bourret, G. Verreault, M. Castonguay, and L. Bernatchez. 2013. Population genetics of the American Eel (Anguilla rostrata): F_{st} = 0 and North Atlantic Oscillation effects on demographic fluctuations of a panmictic species. Molecular Biology 22, 1763-1776.

DNREC. 2018. Delaware Fish Consumption Advisories. Delaware Department of Natural Resources and Environmental Control & Delaware Division of Public Health.

Facey, D. E., and M. J. Van Den Avyle. 1987. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (North Atlantic): American eel.

Miller, M. J., S. Bonhommeau, P. Munk, M. Castonguay, R. Hanel, and J. D. McCleave. 2014. A century of research on the larval distributions of the Atlantic eels: a re-examination of the data. Biological Reviews 000-000.

NJDEP. 2021 Fish Smart. Eat Smart. A Guide to Health Advisories for Eating Fish and Crabs Caught in New Jersey Waters. NJ Department of Environmental Protection & NJ Department of Health.

Nye, J. A., M. R. Baker, R. Bell, A. Kenney, K. H. Kilbourne, K. D. Friedland, E. Martino, M. M. Stachura, K. S. Van Houtan, and R. Wood. 2014. Ecosystem effects of the Atlantic Multidecadal Oscillation. Journal of Marine Systems 133:103–116.

Schmucker, A. K., N. S. Johnson, H. S. Galbraith, and W. Li. 2016. Glass-eel-stage American Eels respond to conspecific odor as a function of concentration. Transactions of the American Fisheries Society 145:712–722.

Smiley, W. (1884). US Fisheries Commission Bulletin. 4: 469-470.

Sykes, J. E., & Lehman, B. A. 1957. Past and present Delaware River shad fishery and considerations for its future. US Fish and Wildlife Service. No. 46.

USFWS (U. S. Fish and Wildlife Service). 2015. Endangered and threatened wildlife and plants; 12-month finding on a petition to list the American eel as threatened or endangered. Federal Register 80:95 (10 October 2015): 60834–60850.

Fish Population Trends in Delaware Bay: Climate and Community

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University of Delaware

Trawl survey data from Delaware Bay revealed a shift in the fish community over the last three decades (Oleynik 2020). In this feature box, we'll explore long-term fish community trends and explain possible mechanisms for these changes.

Delaware Fish and Wildlife conducts two fish and macroinvertebrate trawl surveys which take place once a month between April and November at fixed sampling stations in Delaware Bay (Fig 1). The smaller, 16-foot survey, samples along the coastline of the Bay and up into the Delaware River, and typically catches smaller, juvenile animals, while the larger, 30-foot survey, samples in the deepest parts of the Bay along the shipping channel typically catch larger, adult animals. The two surveys date back to the 1960s, making them some of the longest-running state trawl surveys in the U.S.

Figure 1. A diversity of fish and macroinvertebrates caught on the Delaware Fish and Wildlife trawl surveys (left). Crew on the Delaware Fish and Wildlife R/V First State, pulling up the trawl net during the monthly trawl survey (right). Photo credits Delaware Division of Fish & Wildlife.

Nearly 200 species of finfish and shellfish were caught in the trawl surveys since 1960. Many of these species are transient marine species, which move in and out of the Bay, and up and down the coast, throughout the year, making Delaware Bay an ecologically and economically important habitat for the entire East Coast. Our analyses of these trawl survey data revealed changes in the overall fish and macroinvertebrate community composition through time, with a significant increase in species richness, or the number of species caught in the surveys, since 1990. We also found a significant increase in temperature and a significant decrease in dissolved oxygen in Delaware Bay, indicating a change in the physical environment (Fig. 2).

To explore fish community changes further, we assessed species accumulation, which represents the relationship between species richness and the area or effort sampled. In this case, a species accumulation curve represents the accumulation of species when sampled from individual trawls within the entire Delaware Bay. The species accumulation curves for each year were calculated and then linearized to determine their slope. The slope of species accumulation curves has been correlated to community metrics including abundance, species richness, and species diversity, and it may be a more efficient measure of overall community 'health' than a combination of these community metrics (Novaglio et al. 2016). We found that slopes increased since 1990, echoing the increasing trend in species richness (Fig. 2). Studies have shown that disturbed ecosystems that experienced greater anthropogenic pressure typically have lower species accumulation slopes than undisturbed ecosystems (McClanahan 1994; Flather 1996; Cannon et al. 1998; Tittensor et al. 2007). Thus, the increase in species richness and species accumulation may indicate an increase in overall community health through time in Delaware Bay. This trend in richness and accumulation may be, at least in part, due to habitat restoration and improvements in water quality that have taken place in Delaware Bay since the 1980s.

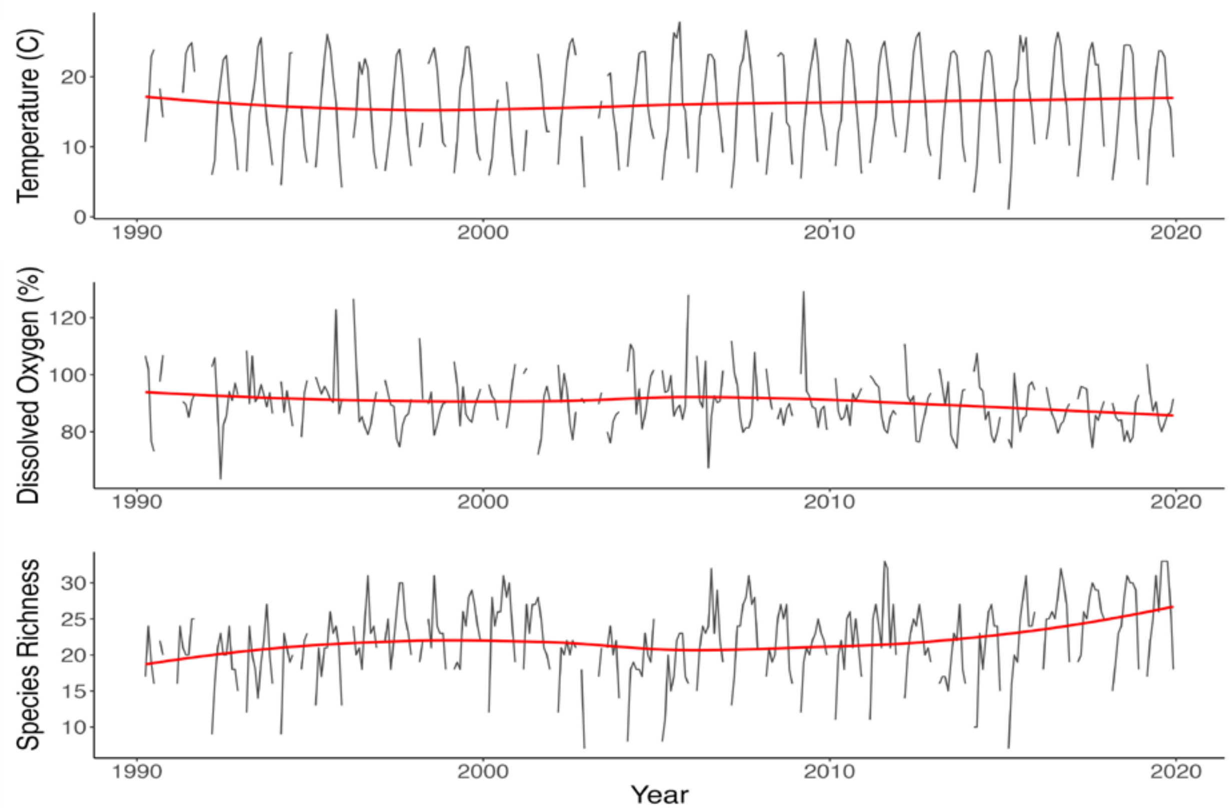

Figure 2. Monthly time series of temperature (°C), dissolved oxygen (percent saturation), and species richness (# of species) in Delaware Bay from 1990 to 2019. The red line represents a LOESS smoother to show the general trend.

While restoration and water quality may have helped to improve community health, further analysis also revealed community-wide reorganizations in Delaware Bay through time. Delaware state scientists noticed that some species, such as winter flounder, were caught frequently in the trawl surveys in the 1960s, 70s, and 80s, but began to disappear, and are almost never caught in the present day. Upon examination of single species patterns of occurrence through time, this appeared to be true for a number of other species as well, which typically have more northern distributions. These included species like winter skate (*Leucoraja ocellata*), silver hake (*Merluccius bilinearis*), and Atlantic herring (*Clupea harengus*) for which Delaware is at the southern extent of their distribution. Similarly, species with more southern ranges that were not caught in the early years of the survey began to show up in later years. These included species like sandbar shark (*Carcharhinus plumbeus*), spot (*Leiostomus xanthurus*), and bluntnose stingray (*Dasyatis say*), for which Delaware Bay is at the Northern extent of their distribution (Fig. 3, Table 1).

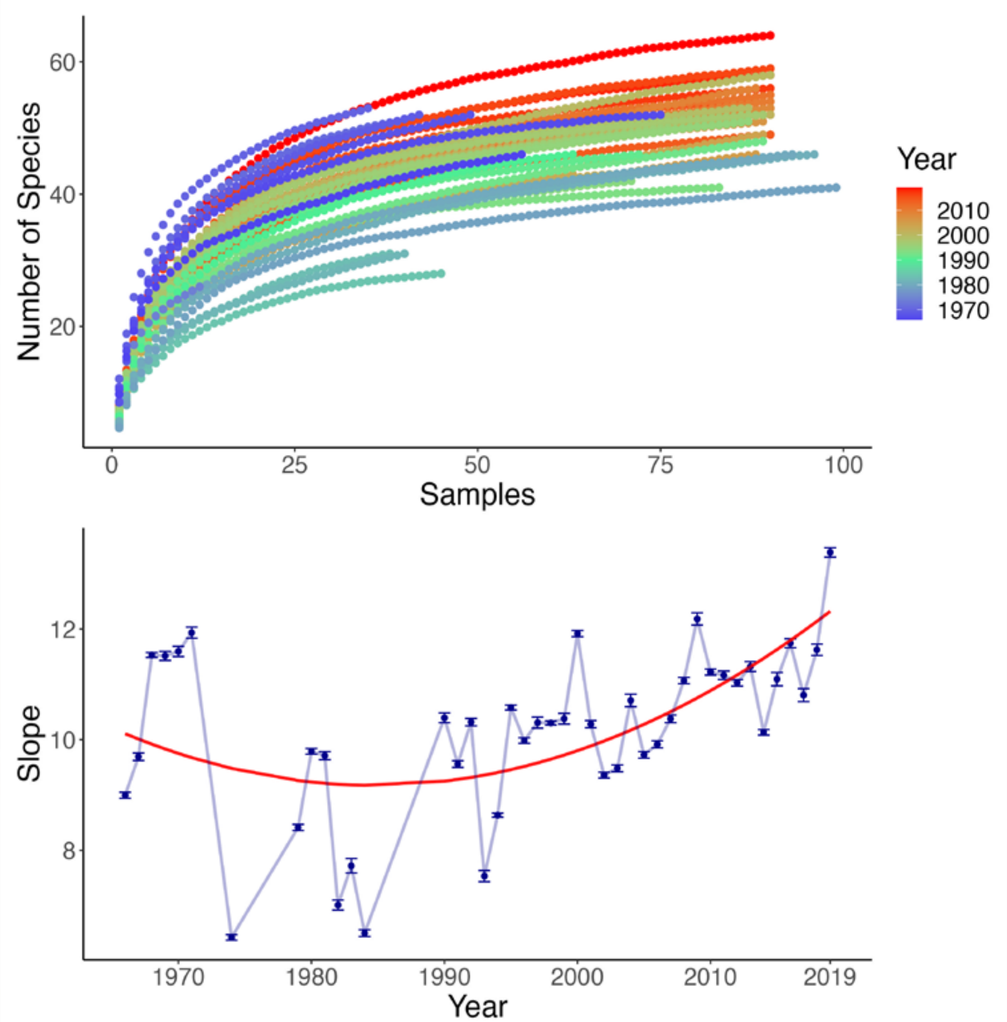

Figure 3. Species accumulation curves, color represents year (top); Slope and standard error of species accumulation curve by year (1990-2019) with a weighted quadratic fit in red (bottom).

To explore possible climate-related mechanisms of community change, mean mid-latitude of the marine community in Delaware Bay was assessed through time. Mid-latitude was characterized for each species using the information in FishBase, and the mean mid-latitude was taken for the whole community in each year to see the trend through time. These results show that the mean mid-latitude of the community has decreased through time, meaning that more southern range species are now dominating the community composition (Fig. 4). Mean preferred temperature was also explored in a similar way; the mean preferred temperature for each species was taken from a temperature range in FishBase, and then mean preferred temperature for the entire community was characterized through time. This trend showed an increase in mean preferred temperature through time, indicating that species that inhabit warmer waters are being caught more frequently in Delaware Bay (Fig. 4). These trends indicate a clear shift in the community through time, and the increase in species richness and species accumulation suggest that Southern range species are possibly showing up faster than Northern range species are disappearing. One possible ecological explanation for the difference in rates, indicated in prior research, is that species ranges generally expand faster poleward than they contract at the lower latitudes of their distribution due to climate change (Parmesan and Yohe 2003). While increasing temperature has not yet been directly linked to these changes in community in the Delaware Bay, temperature was found to be the greatest driver of temporal and spatial variability in the estuarine fish community (Oleynik et al. *in review*). As climate change continues to drive changes in physical conditions, particularly in more vulnerable marine environments like estuaries, the community in Delaware Bay will likely continue to change into the future.

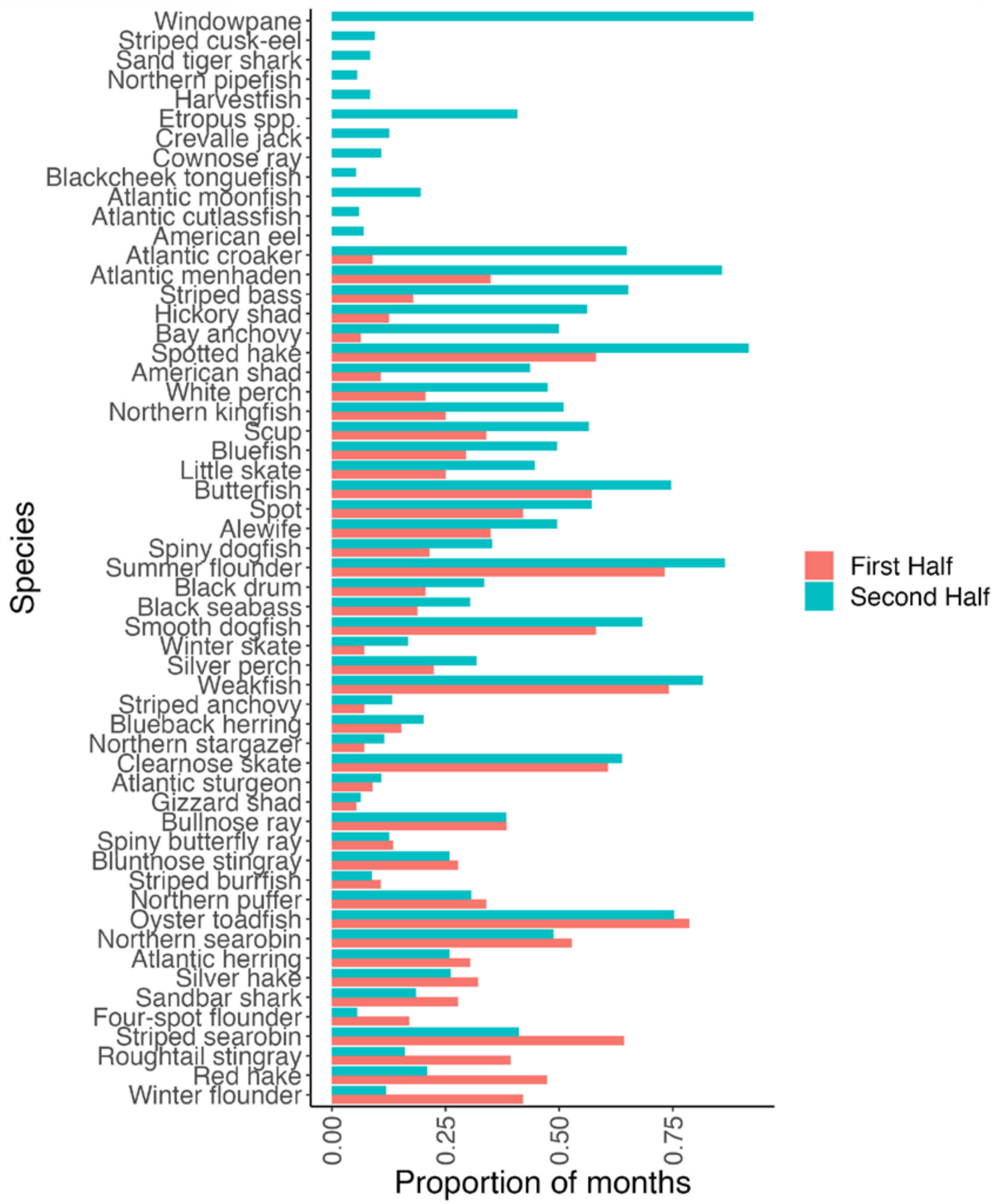

Figure 4. Proportion of months each species was caught in the 30-foot survey for the first half of the time series (1966-1984) in red and the second half, (1990-2019) in blue. Species occurring in less than 5% or more than 95% of months were excluded.

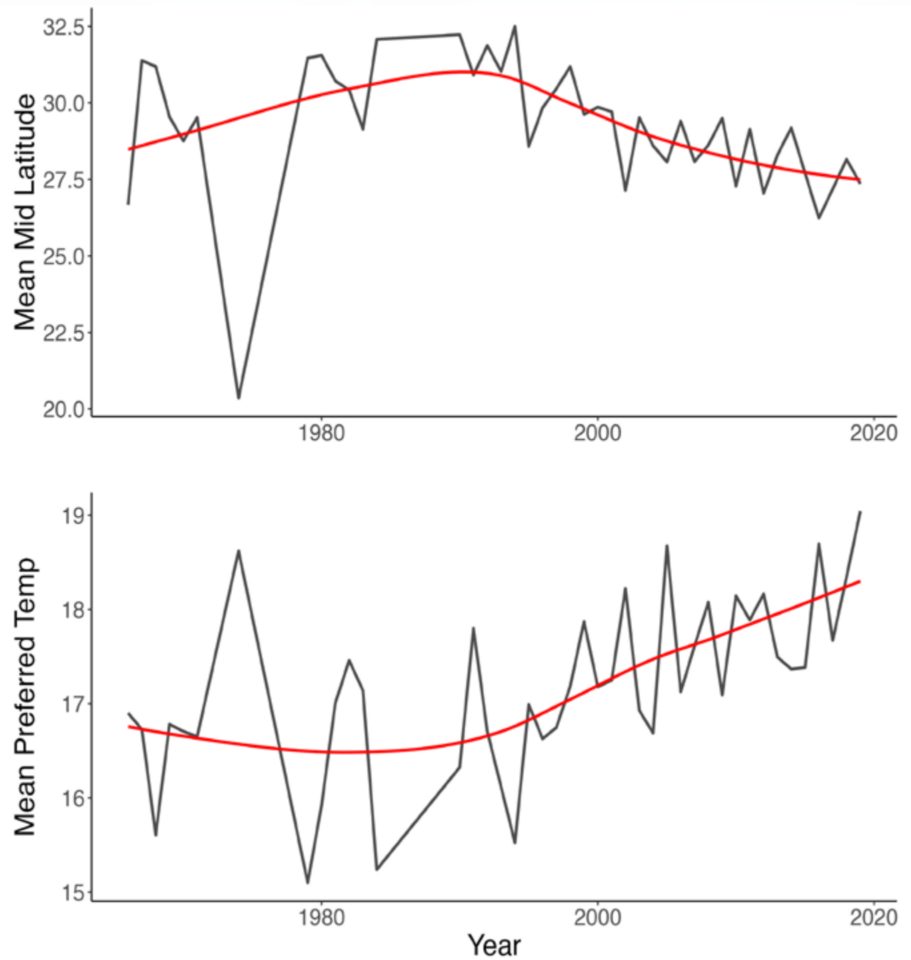

Figure 5. Mean preferred temperature (°C) and mean mid latitude from fishbase.org by year for fish species from 1966 to 2019 from the 30-foot survey. The red line represents a LOESS smoother to show the general trend.

Table 1. Occurrence of each species caught in the 30-foot survey by decade. Green "x" represents caught, white "0" represents not caught.

Species	1960s	1970s	1980s	1990s	2000s	2010s
The following fish were caught across all decades, from 1960s to 2010s:	Atlantic Menhaden, Atlantic Sturgeon, Black Drum, Black Seabass, Blueback Herring, Bluefish, Bluntnose Stingray, Butterfish, Clearnose Skate, Conger Eel, Four-spot Flounder, Gizzard Shad, Hickory Shad, Hogchoker, Little Skate, Northern Kingfish, Northern Puffer, Northern Searobin, Oyster Toadfish, Red Hake, Roughtail Stingray, Sandbar Shark, Scup, Silver Hake, Smooth Dogfish, Spiny Butterfly Ray, Spiny Dogfish, Spot, Spotted Hake, Striped Bass, Striped Burrfish, Striped Searobin, Summer Flounder, Weakfish, White Perch, Windowpane, Winter Flounder					
Inquiline Snailfish	x	0	0	0	0	0
Sea Raven	x	0	0	0	0	0
Snowy Grouper	x	0	0	0	0	0
Three-spine Stickleback	x	0	0	0	0	0
Longhorn Sculpin	x	x	0	0	0	0
Orange Filefish	x	x	0	0	0	0

Species	1960s	1970s	1980s	1990s	2000s	2010s
Striped Killifish	x	0	x	0	0	0
Fringed Flounder	x	x	x	0	0	0
Grubby	x	x	x	0	0	0
Ocean Pout	x	x	x	0	0	0
Feather Blenny	0	0	0	x	0	0
Mummichog	0	0	0	x	0	0
Pinfish	0	0	0	x	0	0
Goosefish	x	x	0	x	0	0
Smallmouth Flounder	0	0	x	x	0	0
Atlantic Silverside	0	0	0	0	x	0
Northern Sennet	0	0	0	0	x	0
Rock Gunnel	0	0	0	0	x	0
Sea Lamprey	0	0	0	0	x	0
Planehead Filefish	x	x	0	0	x	0
Shortnose Sturgeon	0	0	0	x	x	0
Tautog	x	x	x	x	x	0
Atlantic Hagfish	0	0	0	0	0	x
Atlantic Sharpnose Shark	0	0	0	0	0	x
Atlantic Stingray	0	0	0	0	0	x
Atlantic Thread Herring	0	0	0	0	0	x
Banded Rudderfish	0	0	0	0	0	x
Rough Scad	0	0	0	0	0	x
Sheepshead	0	0	0	0	0	x
Threadfin Shad	0	0	0	0	0	x
Atantic Angel Shark	x	x	0	0	0	x
Blue Runner	x	x	0	0	0	x
Atlantic Spadefish	0	0	x	0	0	x
Inshore Lizardfish	0	0	x	0	0	x
Cobia	0	0	0	x	0	x
Gray Triggerfish	0	0	0	x	0	x
Lined Seahorse	0	0	0	x	0	x
Spanish Mackerel	0	0	0	x	0	x
Smooth Butterfly Ray	x	x	0	x	0	x
Striped Mullet	x	x	0	x	0	x
Atlantic Cutlassfish	0	0	0	0	x	x
Banded Drum	0	0	0	0	x	x
Dusky Shark	0	0	0	0	x	x
Naked Goby	0	0	0	0	x	x
Thresher Shark	0	0	0	0	x	x
White Catfish	0	0	0	0	x	x

Species	1960s	1970s	1980s	1990s	2000s	2010s
Florida Pompano	0	0	x	0	x	x
Atlantic Mackerel	0	0	0	x	x	x
Blackcheek Tonguefish	0	0	0	x	x	x
Channel Catfish	0	0	0	x	x	x
Etropus spp.	0	0	0	x	x	x
Northern Pipefish	0	0	0	x	x	x
Cownose Ray	x	0	0	x	x	x
Atlantic Moonfish	0	x	0	x	x	x
Bay Anchovy	x	x	0	x	x	x
Lookdown	x	x	0	x	x	x
Pigfish	x	x	0	x	x	x
Silver Perch	x	x	0	x	x	x
Striped Anchovy	x	x	0	x	x	x
Winter Skate	x	x	0	x	x	x
American Eel	0	0	x	x	x	x
Crevalle Jack	0	0	x	x	x	x
Harvestfish	0	0	x	x	x	x
Northern Stargazer	x	0	x	x	x	x
Striped Cusk-eel	x	0	x	x	x	x
American Shad	0	x	x	x	x	x
Atlantic Croaker	0	x	x	x	x	x

References

Cannon, C. H., Peart, D. R., & Leighton, M. (1998). Tree species diversity in commercially logged Bornean rainforest. *Science*. https://doi.org/10.1126/science.281.5381.1366

Flather, C. H. (1996). Fitting species-accumulation functions and assessing regional land use impacts on avian diversity. *Journal of Biogeography*. https://doi.org/10.1046/j.1365-2699.1996.00980.x

Froese, R. and D. Pauly. Editors. 2021. FishBase. World Wide Web electronic publication. www.fishbase.org.

McClanahan, T. R. (1994). Kenyan coral reef lagoon fish: effects of fishing, substrate complexity, and sea urchins. *Coral Reefs*. https://doi.org/10.1007/BF00303637

Novaglio, C., Smith, A. D. M., Frusher, S., & Ferretti, F. (2020). Identifying historical baseline at the onset of exploitation to improve understanding of fishing impacts. *Aquatic Conservation: Marine and Freshwater Ecosystems, 30*(3), 475–485. https://doi.org/10.1002/aqc.3264

Oleynik, H. A. (2020). Changes in a Mid Atlantic estuary: trends and drivers of the fish and macroinvertebrate community in Delaware Bay. (Publication No. 1237124312) [Master's thesis, University of Delaware]. https://udspace.udel.edu/handle/19716/28649

Oleynik, H.A., J. J. Bizzarro, E. A. Hale, & A. B. Carlisle. *In review*. Environmental drivers of biogeography and community structure in a Mid-Atlantic Estuary. *Oecologia*

Parmesan, C., & Yohe, G. (2003). A globally coherent fingerprint of climate change impacts across natural systems. *Nature*. https://doi.org/10.1038/nature01286

Tittensor, D. P., Micheli, F., Nyström, M., & Worm, B. (2007). Human impacts on the species-area relationship in reef fish assemblages. *Ecology Letters*. https://doi.org/10.1111/j.1461-0248.2007.01076.x

8

TECHNICAL REPORT FOR THE ESTUARY AND BASIN

Restoration

Partnership for the DELAWARE ESTUARY

December 2022 | Report No.22-05

Partnership for the Delaware Estuary—Host of the Delaware Estuary Program

TECHNICAL REPORT FOR THE ESTUARY AND BASIN

8 Restoration

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Contributors Brittany Wilburn and Metthea Yepsen
Cover photograph by Kate Layton

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8. Restoration

Abstract

The goal of this chapter is to present an assessment of the current state and perceived trajectory of ecological restoration and its related factors since 2017. Data from EPA's National Estuary Program's Online Reporting Tool indicated an increase in restoration since the 2012-2016 time period, while protection/ maintenance remained largely constant. A poll of restoration practitioners indicated a generally positive trajectory regarding innovation, investment, implementation, monitoring, and outreach, but still had concerns regarding the complexity and level of effort required by the regulatory process. Recommendations include the further development of cross-sector relationships to move towards greater regulatory ease especially to facilitate innovative tactic implementation, as well as increased partnerships between larger, more established, and community-based restoration entities to promote a more equitable distribution of restoration funds. Finally, larger projects that target ecosystem-based restoration will be needed to promote greater resiliency in the face of a changing climate and sea level rise.

Introduction

The goal of this chapter is to present gathered information on various important aspects associated with ecological restoration efforts within the Delaware Estuary. Further, this chapter will present an assessment of the current state of ecological restoration and where certain aspects of ecological restoration have progressed, regressed, or remained static since the 2017 reporting.

Although no entity has quantified the cumulative management and restoration progress across the entire Delaware River Basin, an initial summary was provided for the lower basin in 2012 from the National Estuary Program Online Reporting Tool (NEPORT). Later, the restoration indicator was updated in the 2017 Technical Report for the Delaware Estuary and River Basin. This was the most recent occasion in which the restoration indicator was updated.

The indicators presented in this chapter serve to provide information on three areas related to restoration efforts in the Delaware Estuary:

1. Trends in measurable size (acres) of previous and ongoing restoration efforts;
2. Trends regarding the types (diversity) of prior and ongoing restoration efforts; and
3. Information on restoration-adjoining influences (i.e., innovation, level of investment, regulatory climate, implementation, monitoring, and outreach), provided by polling qualified professionals who work in the restoration sciences.

This chapter includes a synthesis of the three aforementioned items to provide context for the current state of restoration practices and to help guide the development of an estuary-wide vision toward improving restoration technology, understanding, cooperation, and strategy to optimize restoration efforts in the Delaware Estuary.

When evaluating restoration, it is important to recognize how "restoration" fits into the umbrella framework of "resource management." Resource management consists of three fundamentally distinct areas: preservation, restoration, and maintenance. Preservation may include physical and/or regulatory resource protection, acquisition, and buffering. Maintenance can include maintaining the status quo by activities associated with fieldwork and surveys/monitoring. Restoration is the practice of engaging in an action that increases a resource in terms of habitat size, quality, diversity, etc., which leads to a net gain in ecological functions and values. As a general rule, preservation and maintenance tend to be more often practiced due to lower levels of uncertainties and implementation risk, while restoration carries with it higher uncertainties and higher implementation risk. As such, restoration is harder to fund, gain approval for, and is proportionately the least practiced of the three resource management areas. Alignment between planning and funding is crucial. Often, design projects are not "ready" when construction funds become available, or conversely designed projects (from conceptual to 100% engineered) are not implemented because funds are not available. The ebb and flow of funding create a moving target to align with restoration needs. It is widely viewed that more intensive restoration will be required to combat these changing base-level conditions.

In this way, preservation/maintenance and restoration work together to sustain natural resources (at their current condition or in line with their natural evolution, if still practical) and the services they provide. For areas experiencing severe stress from a variety of sources like anthropogenic pressures or climate change, restoration can play an important role. If one area is not implemented with the others, the resource management underachieves or even breaks down. The severity of the impaired system and the magnitude of the goals (both habitat- and species-specific) will affect the balance in which preservation, management, and restoration are applied (i.e., restoration may have the largest role in severely impaired systems which are epicenters for impaired species of concern). The absence of natural or man-induced restoration makes it nearly impossible for long-term natural resource sustainability. Nature has some level of resilience, a sustainable capability combined with a healing component. But depending on what the drivers and goals are for a particular resource, natural resilience may not suffice in the face of rapid base-level changes and a system decoupled due to human perturbation or natural disasters and fluxes.

Similar to resource management, restoration can be further classified into three subgroups: creation, enhancement, and governance, with governance being an increasingly significant factor in modern-day restoration. Creation involves establishing functions and values for a resource where one was not currently present. Enhancement generally is defined as improving something that currently exists. Governance is the political, regulatory, and education component of restoration (Fig 8.1). Socio-based drivers and obstructions have been demonstrated to periodically alter the governance aspect of restoration. For the purposes of this section, governance will mean non-physical influences. The response to Superstorm Sandy brought about a rapid integration of new partners in restoration, advancements in technologies, a willingness to push the boundaries with respect to scale and regulatory framework, and a willingness to invest in restoration initiatives, but to some extent, those advancements have stalled or been overshadowed by certain elements of the community who are less willing to take risks, have a lack of understanding, and/or have a reluctance towards change.

Additionally, one other significant factor considered in this chapter is the influence of restoration technology. The extent of development, testing, and implementation of restoration technologies, in the face of today's climate-based and socioeconomic challenges, may greatly influence public and political opinion and could be considered the catalyst in determining the scale of landscape-scale restoration implementation success or restoration stagnation and failure. Caution in the use of experimental technologies or larger scales is required, but our current technological state and resistance to change in methodologies can become a hindrance or wall. Restoration technology is concurrently limited with room for continued development driven by creativity, expertise, and scientific commitment. There is a need for technically-based development, testing, and evaluation and a willingness to take calculated risks in efforts to improve the technology of new techniques. Unfortunately, there are limited individuals and programs who are willing and able to fill that role. Our environment and climate are ever-changing, meaning technology does not have the luxury to "rest on one's laurels."

Restoration Indicators

In common usage, the term "restoration" implies some form of remediation or improvement that returns a resource or certain resource functions to some former condition or location. In some cases, however, targeting historic conditions is inappropriate because the viable location for a resource or habitat may have shifted in response to changing environmental conditions (e.g., salinity, tidal inundation, temperature). In other cases, the structure and function of restored systems may never match that of undisturbed systems, and various tools are used to set appropriate criteria that define a project's success. In acknowledging the difficulty of fully repairing disturbed systems, restoration practitioners have adopted various definitions of restoration and restoration-type activities. For example, in its 1992 report titled Restoration of Aquatic Ecosystems, the National Research Council defined restoration as the "return of an ecosystem to a close approximation of its condition prior to disturbance." The Society for Ecological Restoration defines ecological restoration as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed." Additionally, New Jersey, Delaware, and Pennsylvania all have regulatory definitions for each term, but this document will focus on the broader definitions which generally apply in each state.

The concept of restoration is further clarified by defining different types of restoration-related activities. There are many management actions that can be considered restoration activities, such as land and habitat protection, flow management, and pollutant regulation. However, for the purposes here, "restoration" refers to on-the-ground actions that either create, enhance, or restore natural resources. With more precise and expansive data provided in the future, management progress could be broadened to include any actions or decisions that lead to improvements in environmental conditions as assessed by the indicators in Chapters 1-7. This includes the elimination or reduction of stressors that degrade natural conditions. In addition to the traditional restoration of past natural conditions, the following terms describe activities that are considered part of the restoration for the purposes of this chapter.

This chapter reports on two restoration indicator categories: physical and non-physical (Fig 8.1). ***Physical indicators*** refer to discrete, on-the-ground restoration applications. This TREB reports on the following physical indicators: ***number of acres restored and protected***, their respective ***habitat*** types, and associated ***dollars spent***. These indicators have been reported in previous TREB reports and attempt to track the progress of landscape-level restoration activities. Restoration success would be an ideal indicator to include but is harder to track due to a lack of data regarding the proportion of individual projects that have been able to meet defined goals through the tracking of relevant metrics. The application of goal-based monitoring frameworks, such as the ones developed in NJ and DE, coupled with a project progress registry would facilitate this type of tracking. Although these physical indicators can provide a summary of restoration progress, they are influenced by a variety of non-physical activities and circumstances that provide context to the quantifiable restoration activities. For example, if there is a drop in the number of annual acres restored, this may not be due to a lack of effort by restoration practitioners but may be a result of the influence of changes in available funding or regulatory requirements. Additionally, a decline in the restoration activities of a specific habitat may result in a divergence between the innovative expectations of a funding entity (e.g., hesitant to fund unproven tactics), the region's practitioners (e.g., submitting innovative project designs), or a shift in interest between habitats within the restoration community. In this TREB effort, six ***non-physical indicators*** have been selected for evaluation to provide context for the physical indicators and to drive discussion regarding the future direction of restoration in the Delaware Estuary: ***innovation, level of investment, regulatory climate, implementation, monitoring,*** and ***outreach***.

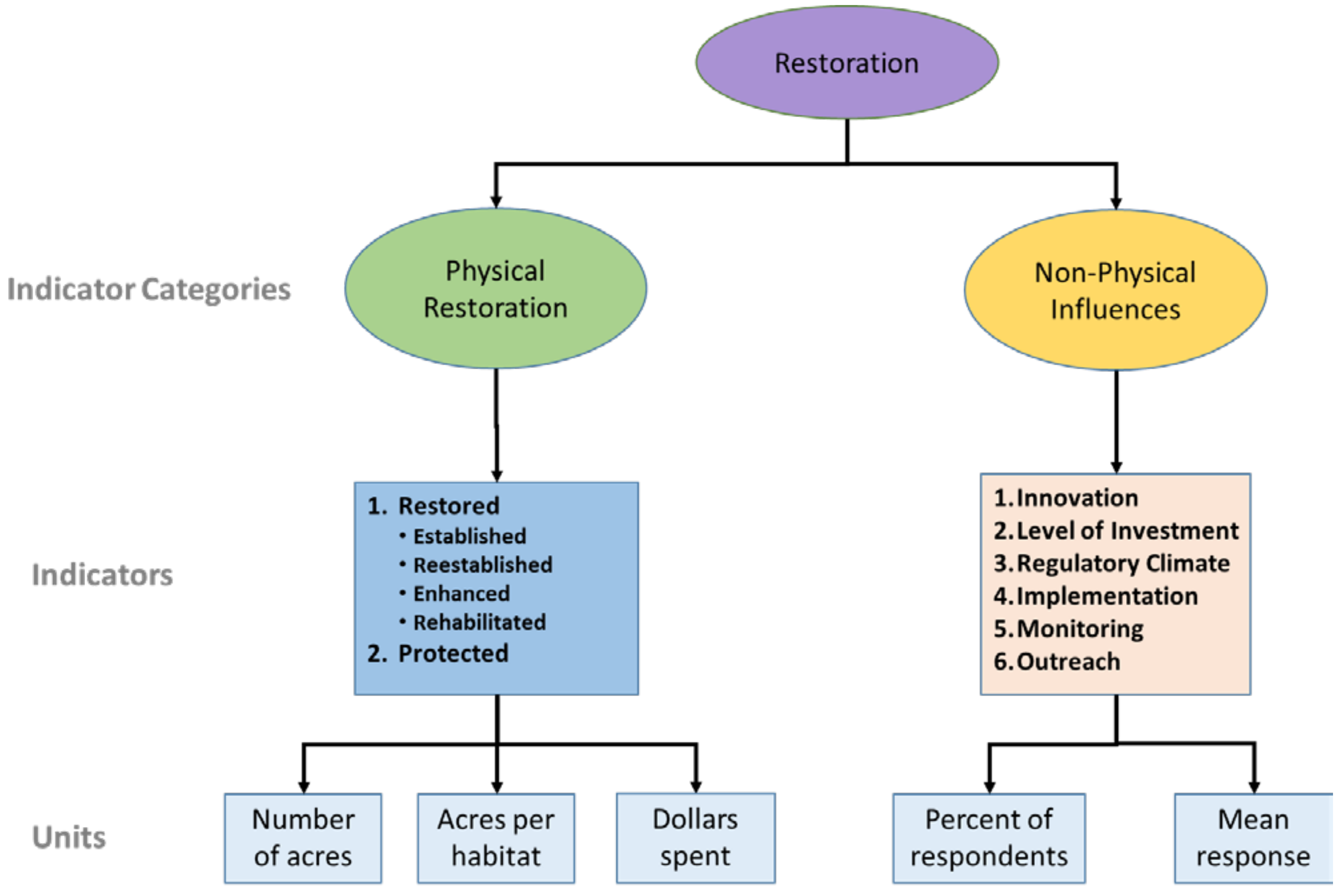


Figure 8.1 Diagram of restoration indicators nested in either physical restoration or non-physical influences categories with their associated reported units. Data for physical restoration indicators were compiled from the EPA NEPORT reporting system, and data for non-physical indicators were collected through a survey of restoration practitioners and other individuals in adjacent professional fields (e.g., regulatory agents, academic researchers).

8.1 Physical Restoration

Description and Methods

Restoration data from multiple states and programs are challenging to collect and analyze, and many important resources are found in the Delaware Estuary and Basin. Considering the tremendous habitat diversity, numerous geopolitical boundaries, and large size of the watershed, efforts to track restoration progress are hampered by limited data availability and lack of standardized reporting among the many different agencies and programs that are responsible for the restoration. One of the most straightforward ways to track habitat restoration is to determine acres restored annually, focusing on voluntary actions (and not reparative, regulatory-based actions such as mitigation projects). Ideally, restoration activities should also be assessed for specific habitat types. In the future, it would be beneficial to also assess the functionality to restored habitats, since a particular site could be "restored" significantly without any net increase in acreage. Since no database exists to track watershed-wide restoration, as a starting point for this effort, we discuss acreage data that have been reported as restored (and/or protected) by each state (New Jersey, Pennsylvania, and Delaware) annually using EPA's National Estuary Program Online Reporting Tool (NEPORT).

NEPORT is a web-based database that EPA has developed for National Estuary Programs (NEPs) to track annual acreage of habitat improvement efforts as part of the goals of the 1996 Comprehensive Conservation Management Plan (PDE 2019) for the Delaware Estuary. Unfortunately, there is no coordinated tracking system at this time to determine how many net acres have been restored or gained/lost in the watershed, and NEPORT is not comprehensive as it reflects only those data that have been voluntarily provided by partners of the Partnership for the Delaware Estuary. Since NEPORT is not comprehensive and generally focuses on the lower half of the Delaware River Basin, data for this indicator is largely conservative at the watershed scale. However, since this NEPORT reporting approach has been active for more than a decade it is possible to examine broad restoration trends using NEPORT data as an indicator, but it should be noted that EPA occasionally modifies the NEPORT data collection and reporting process. Another advantage of NEPORT data is that the tracking program excludes actions associated with mitigation (e.g. Natural Resource Damage Assessment, Supplemental Environmental Project), which are designed simply to correct for discrete injuries. Although protection efforts are not the focus of this chapter (see above), NEPORT data for protected acreage are also shown here for comparison purposes. NEPORT tracks restoration data in four categories which are used in this chapter to explore and evaluate restoration progress in the Delaware Estuary:

- **Establishment** (also referred to as "creation") is the manipulation of physical, chemical, or biological conditions to facilitate the development of a target habitat that is representative of natural conditions but that did not previously exist at the project location. Establishment results in acres gained for the target habitat. For example, establishment occurs when a wetland is placed on the landscape by some human activity on a non-wetland site (Lewis, 1989). Typically, established wetlands are created by the excavation (or addition) of upland soils to achieve elevations that will support the growth of wetland species through the establishment of appropriate hydrology.

- **Reestablishment** is the manipulation of physical, chemical, or biological characteristics of a site with the goal of returning natural/historic habitat types and functions to the site. Reestablishment results in the rebuilding of a former habitat and a gain in acres for that target habitat.

- **Enhancement** is the manipulation of physical, chemical or biological characteristics of a site to strengthen ecological conditions and functions, such as for the purpose of improving water quality, flood water retention, or wildlife habitat. Enhancement typically results in improvement of the structure and/or function without an increase in acreage.

- **Rehabilitation** is similar to enhancement and is defined by the US EPA as the manipulation of the physical, chemical, or biological characteristics of a site with the goal of repairing the natural/historic functions of a degraded habitat. Rehabilitation results in a gain of habitat function but does not result in a gain of acres for that habitat.

- **Protection** is defined as the removal of a threat to, or preventing the decline of, natural healthy environmental conditions. This includes management actions such as land acquisition for public parks, conservation easements, deed restrictions, etc., or other designations to prevent alteration of natural site conditions. This term also includes activities commonly associated with the term "preservation." Although protection efforts are critically important for sustaining ecological function, they do not result in net habitat gains. Although not a direct restoration activity, protection might be considered restoration progress towards a net increase in ecosystem function, relative to no action or loss, and as NEPORT tracks protection efforts, they are included here.

In addition to assessing the amount of area restored in the aforementioned categories, NEPORT also tracks the types of habitat per category to ensure that restoration progress reflects the balance of habitats that have suffered the most degradation. Healthy estuaries depend on a complex mix of habitats, with each estuary possessing a unique character and habitat assemblage. Although the Delaware Estuary and Basin is home to dozens of different habitats and ecological communities, it is most distinct because of its abundant, protective forests in the headwaters, broad freshwater tidal area that supports rare biotic assemblages, and a wealth of coastal wetlands that fringe the tidal estuary. These systems purify our water, provide clean air to breathe, and furnish other critical goods and services enabling the survival of both people and natural communities. For example, coastal wetlands are a hallmark feature of the Delaware Estuary and are critical for supplying diverse benefits to people and the environment, and we have lost more than half of our coastal wetlands mainly due to direct filling and development (see Chapter 6.3). Forests are similarly vital for sustaining source water quality and other services, and forest losses continue to be swift due to development (see Chapter 6.1). To get the greatest benefits, voluntary (non-mitigation) attempts to rebuild these habitats should reflect the natural balance of types that characterizes the watershed. Data from NEPORT were examined to discern the types of habitats that have generated the greatest restoration attention since 2006.

8.1.1 Acres Protected and Restored

Present Status

Recent restoration progress was examined qualitatively by contrasting the types of efforts made in the Delaware Estuary from 2006-2021, as reported in NEPORT. Since 2006, a generally equal amount of land has been protected and restored (Fig 8.2a). Enhancement and rehabilitation accounted for 92% of the restoration activities and were similar in percent of total areas restored. Since 2017 when the last TREB was published, restoration activities accounted for 58% of the total activities, with rehabilitation and enhancement accounting for 98% of the area restored (Fig 8.2b). As protection does not necessarily improve ecological conditions, NEPORT does not give a clear representation of actual net ecological improvement.

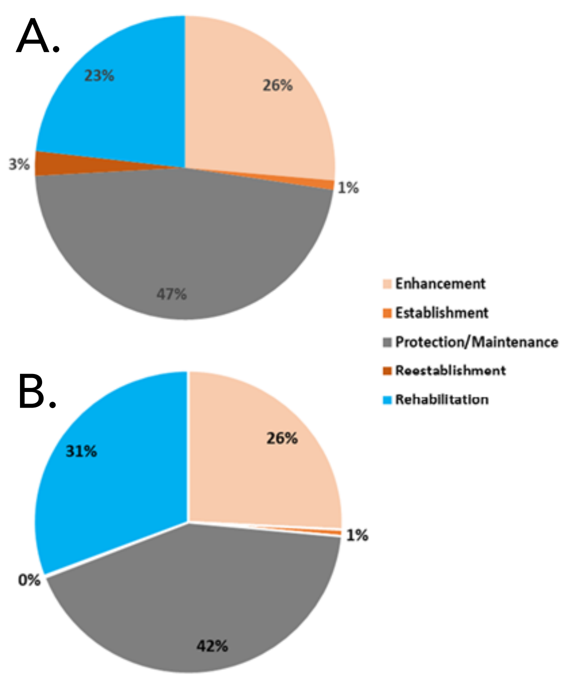


Figure 8.2 Proportions of the land area (acres) protected vs restored (i.e., enhancement, establishment, reestablishment, and rehabilitation) 2006-2021 (a) and 2017-2021 (b, period since last TREB) as reported in NEPORT.

Past Trends

As a National Estuary Program, PDE is responsible for setting restoration goals every year. Since the advent of NEPORT tracking in 2000, the total number of acres reported to NEPORT each year represents a modest 0.017% of the total area of the Delaware River Basin. As noted above, tracking restoration is challenging because PDE must rely on voluntary reporting by partners. Annual variation in restoration investment also takes place since projects are typically grant-funded and are subject to fluctuating availability of funds. Despite these caveats, restoration progress since 2006 has been considerable, typically exceeding the annual goal set by PDE and US EPA for the combination of protected and restored acres (Fig 8.3). Prior to 2011, this annual goal was 2,250 acres. Due to declining acreage that was protected or restored between 2007 and 2010, this annual goal was changed in 2011 to1,500 acres. In multiple single years since 2006 protection efforts surpassed restoration efforts, largely due to data reporting from programs such as New Jersey Green Acres which provides funding for land acquisition projects (Fig 8.3). However, in each of the last three TREB reporting timeframes, cumulative restoration activities (i.e., enhancement, establishment, reestablishment, and rehabilitation) have accounted for more area per period than protection/maintenance efforts (Fig 8.4). Interestingly, restoration activities appear to spike in 6-year cycles (Fig 8.4 - 2007, 2013, & 2019), potentially related to funding cycles (Fig 8.3).

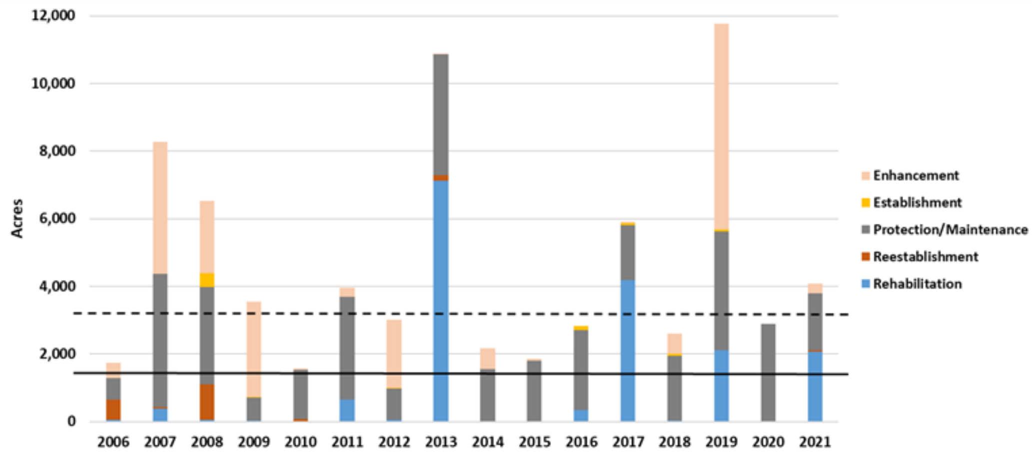


Figure 8.3 Acres restored and protected annually between 2006 and 2021, with the four types of restoration reported separately. For comparison, the annual NEPORT goals are shown for 2006–2010 and 2011–2021 (dashed and solid black lines, respectively).

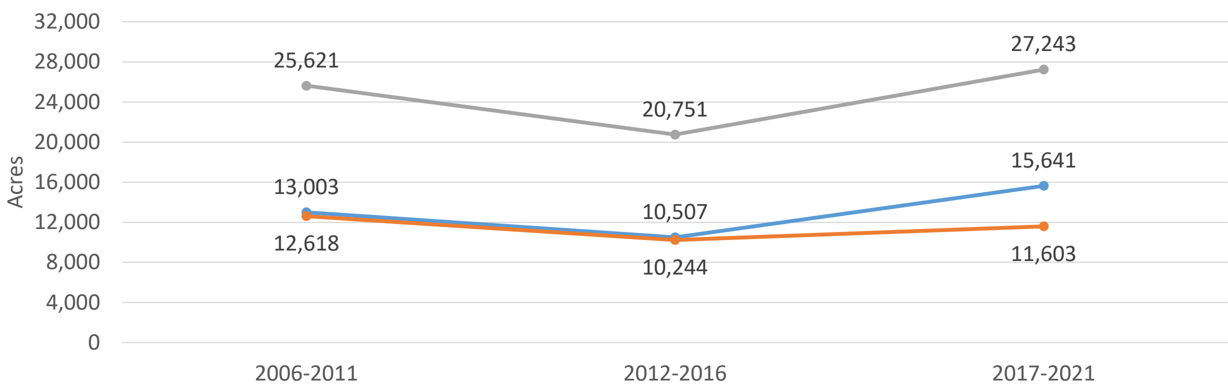


Figure 8.4 Total areas restored (blue), protected (orange), and cumulative (gray) for approximately each of the last three TREB reporting periods.

8.1.2 Habitat Types Protected and Restored

Present Status

Cumulatively since 2006, forests/woodland (32%), tidal wetlands (24%), and forested wetlands have had the greatest acreage restored or protected (Fig 8.5a). This ranking is similar to the 2017-2021 data with the exception of beaches (10%) having the third-largest area restored/protected, and forest/woodland accounting for a greater proportion of the total acreage (47%, Fig 8.5b).

Past Trends

The amount of area protected and restored varies widely among years and among habitat types (Fig 8.6). This variability is due mainly to fluctuations in funding from year to year, as well as shifts in reporting from various state and local partners who report data to NEPORT. Although it is difficult to draw any conclusions from these limited data, the potential of an overall downward trend hypothesized in the 2017 TREB is not as apparent in 2021, largely due to stable activity between 2016 and 2021, as well as the large increase in 2019 (Fig 8.6). Grouping activities by restored (enhancement, establishment, reestablishment, and rehabilitation) and protected, the decline observed in both activities in the 2017 TREB (2012-2016) did not continue, and increases in the restoration and protection were present in the 2017-2021 data (Fig 8.7). Of note are substantial increases in restoration of tidal wetlands, beach, and agriculture, while field and meadow and riparian activities were not reported.

Trajectories and Future Predictions

The amount of area restored per year in the Delaware Estuary (per NEPORT) through non-mitigation, voluntary actions is dependent on funding, especially from state and federal agencies. The restoration need is high, as judged by the continuing losses of critical habitats. However, we are optimistic that in the long term, the pace of restoration will hasten as our understanding of the ecological and economic consequences of inaction increases, and as predicted increases in funding become available. For example, water resources in the Delaware Estuary sustain a $10 billion per year economy, and the loss and degradation of natural systems is certain to have serious economic consequences (Kauffman 2011). In the short term, we anticipate that restoration progress could be undermined if federal investment in environmental programs is reduced, as has been proposed. Fortunately, non-profit organizations such as the National Fish and Wildlife Foundation, in partnership with USFWS and William Penn Foundation, have recognized the importance and scale of the restoration need, contributing substantial resources through the Delaware River Program's

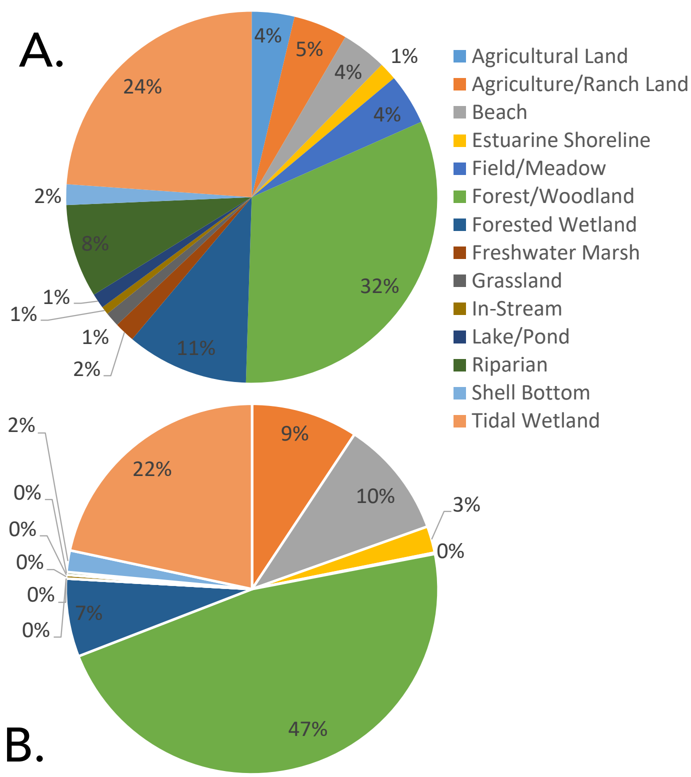


Figure 8.5 Proportions of the total habitat area (acres) either protected or restored (i.e., enhancement, establishment, reestablishment, and rehabilitation) 2006-2021 (a) and 2017-2021 (b) period since the last TREB as reported in NEPORT.

Delaware Watershed Conservation and Restoration Funds to support habitat restoration. With sustained or increased investment by other state and local entities, and potentially new public-private partnerships, we anticipate that the Delaware Estuary Program will continue to meet the annual 1500-acre goal. Trajectories and predictions are further discussed through the context of the non-physical indicators in *8.2 Non-Physical Influences and Efforts.*

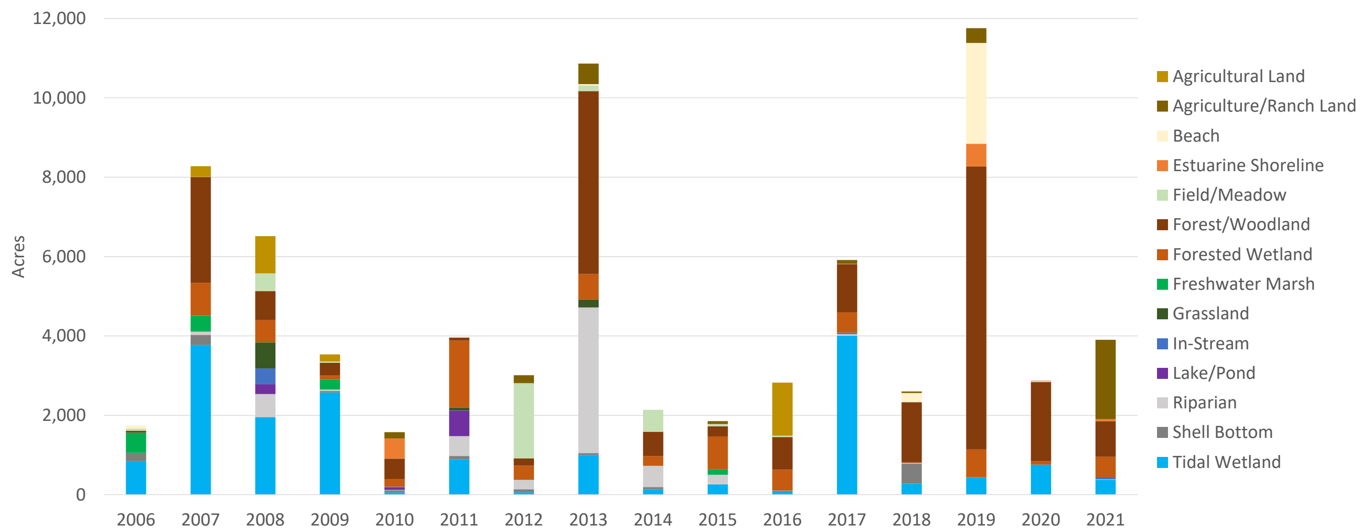


Figure 8.6 Acres restored and protected annually per target habitat between 2006 and 2021.

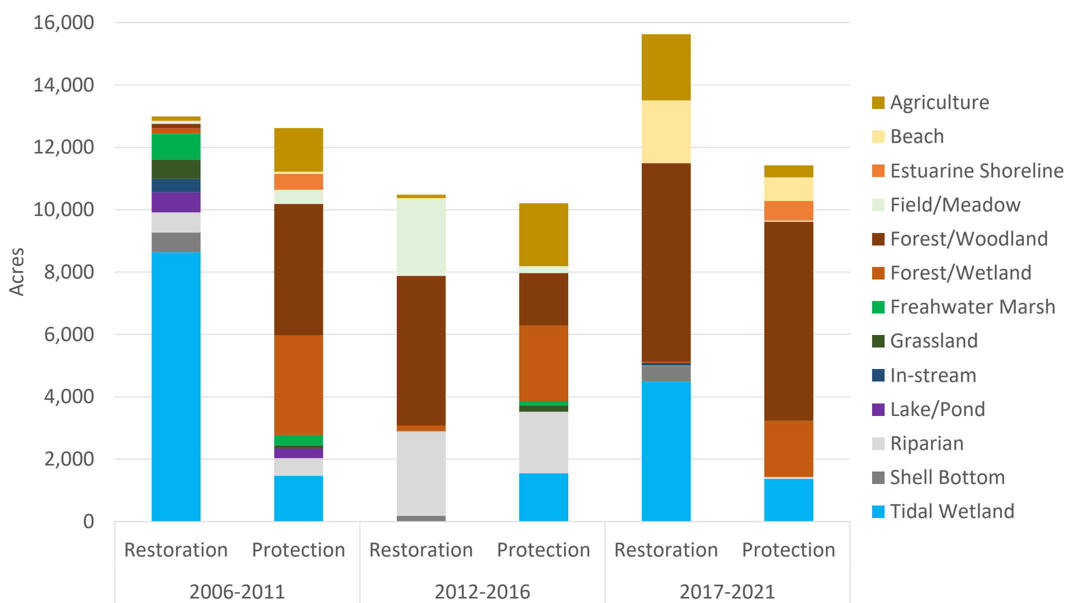


Figure 8.7 Acres restored and protected per habitat between 2006-2011, 2012-2016, and 2017-2021.

As noted in the habitats chapter of this document, more than half of tidal wetlands have been lost in the Delaware Estuary compared to pre-settlement conditions, and between 1996 and 2010, nearly 2% of tidal wetland acreage was lost. Future projections suggest that 119,000 acres (48,000 hectares) will be lost by 2100, assuming that the sea level rises by one meter (Kassakian et al. 2017; Kreeger et al 2010). Forests continue to be lost at an even faster rate, and the cumulative impacts from the development and other contemporary challenges threaten to hasten loss rates in the future. Continued focus on tidal wetlands and forests is therefore warranted. Other habitats that have been prioritized such as shellfish beds are arguably even more vital, but they are also smaller in size and harder to capture in terms of acres.

Unfortunately, hundreds of thousands of acres of natural habitats have been destroyed or significantly altered in the Delaware Estuary watershed during the past 15 years despite many governmental protections (see Chapter 6). One of the top goals in the Comprehensive Conservation Management Plan (CCMP) for the Delaware Estuary is the restoration, protection and enhancement of natural habitats (strategies H1.2, H1.3, H1.4, H2.3, H3.1, H3.2, and H3.4, PDE 2019). Future monitoring and assessment reports would also be strengthened by development of enhanced tracking tools for restoration data, enabling better comparisons with land use data on habitat losses such as those associated with development. The balance of habitat types restored and protected in the past 11 years can be analyzed with data from the NEPORT. Since 2006, 66% of money spent was for protection relative to restoration efforts cumulatively, and was generally greater annually (Fig 8.8). The large increase in rehabilitation in 2017 can be attributed to a single $38 million berm and dike modification project focusing on tidal wetland habitat (Figs 8.8 and 8.9). The contribution of this single project accounts for 72% of the 2017 total dollars spent and 41% of the total restoration dollars spent between 2006-2021. Quantitative measures of land area restored annually in the Delaware Estuary can be an effective way to track management progress, and analysis of limited data suggests that some progress has been made since 2006 .

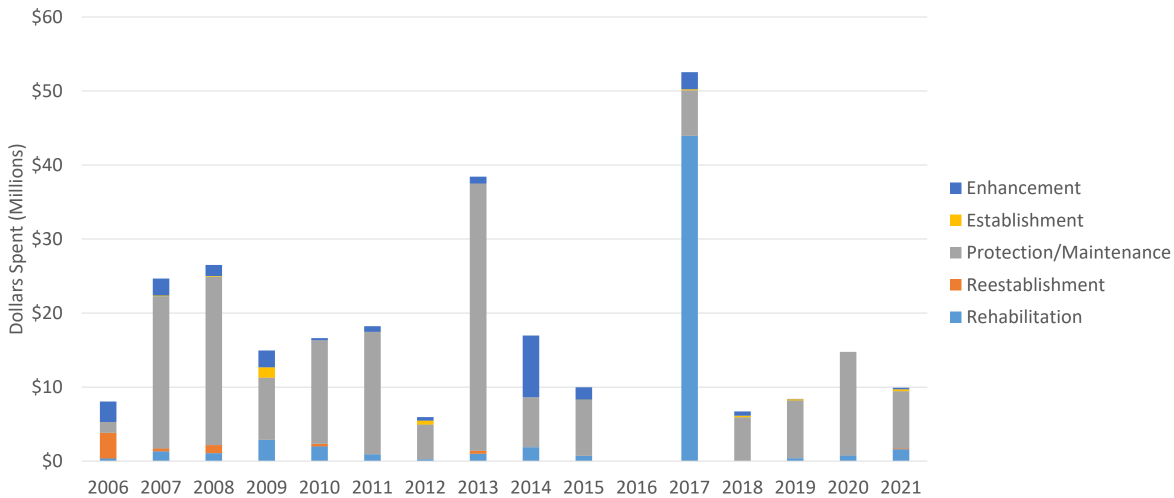


Figure 8.8 Dollars spent for restoration (broken out by four types) and protection between 2006 and 2021.

Unless funding and capacity for natural areas restoration is significantly increased within the Delaware River Basin, it will be vital that the limited investments be spent wisely through the prioritization of key areas and habitat types that are most critical to the character and functionality of the Delaware Estuary, using the best available scientific data. A complementary approach utilizing the wealth of monitoring data generated by DELEP partners and newly developed restoration prioritization tools can be used to guide strategic investments. Ultimately, for effective restoration at the regional scale, a balance needs to be maintained between innovation, investment, and outreach. To protect habitats to the best of our ability, proven techniques need to be employed, but as stressors change we need to be prepared to envision and test new methodologies. Implementation will be driven by funding availability for the tried and true as well as innovative projects, coupled with regulatory mechanisms to facilitate their application and evolve with lessons learned through monitoring. Finally, dissemination of knowledge through outreach will help to engage stakeholders and temper expectations regarding effects and timelines. These non-physical influences on restoration practices are evaluated and explored in the following section.

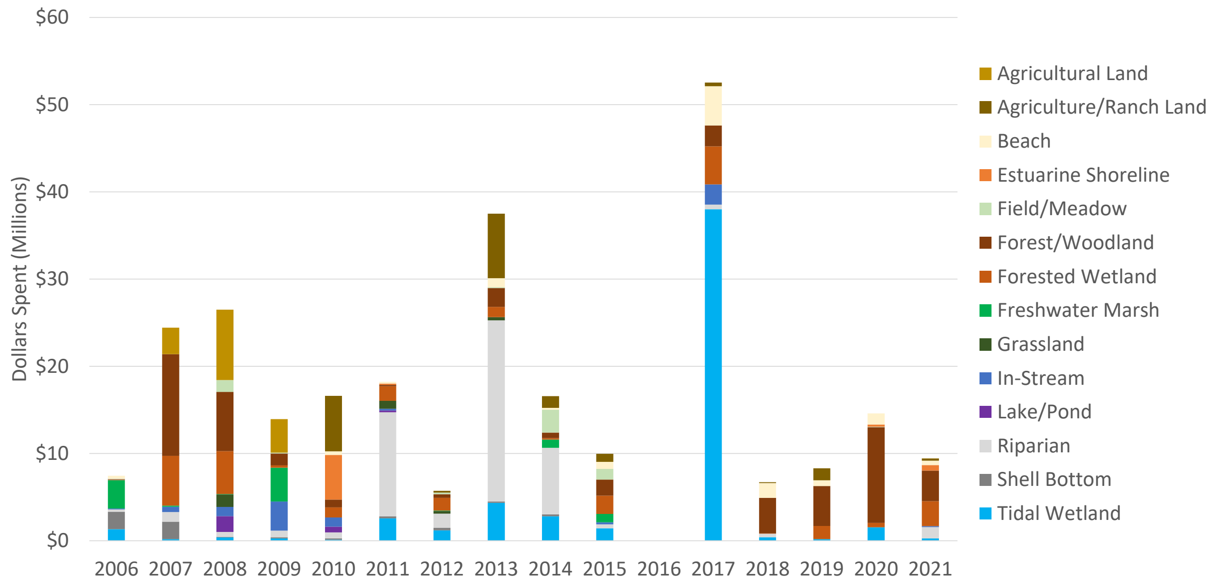


Figure 8.9 Dollars spent on restoration and protection annually per target habitat between 2006 and 2021.

8.2 Non-Physical Influences and Efforts

Description and Methods

Data for non-physical indicators were collected through an online questionnaire. The questionnaire consisted of 45 questions (Appendix 8.A) asking the respondent to select the best answer based on their experience across the following categories (q=number of questions per category):

- **Respondent Information** (q=7): Introductory questions for the evaluation of the sample pool to help provide context to the questionnaire results. The questions accounted for the respondent's area of expertise, time involved in restoration, and current sector

- **Innovation** (q=5): Innovation questions aimed to quantify the perception in the development of new ideas and approaches or new applications of existing ideas and approaches. Innovation can be applied to restoration techniques, technology, and/or tactic design or implementation.

- **Level of Investment** (q=5): These questions asked the user to consider changes in factors related to funding opportunities for restoration efforts.

- **Regulatory Climate** (q=6): These questions asked the user to reflect on changes in regulatory considerations, permitting, and time frames as they relate to restoration project requirements and implementation.

- **Implementation** (q=5): Implementation refers to the actual execution of restoration activities at a specific location, such as construction, deployment of fill material, excavation, or any activity that alters or adds to the landscape. These questions asked the user to relay their experience related to on-the-ground activities.

- **Monitoring** (q=6): Monitoring refers to any activity conducted with the explicit intent to collect information related to any facet of the project that will be used to better understand one or more processes potentially affected by the restoration activity. This can include project monitoring conducted on consistent, sporadic, or opportunistic time frames, and for required (i.e., permit requirement) or optional (i.e., principal investigator/partner initiated) activity. These questions asked the user to relay their experience related to these activities.

- **Outreach** (q=6): Outreach refers to activities intended to disseminate information regarding restoration needs, activities, and/or results to any stakeholder (e.g., different sectors, the general public, educators, other practitioners). These questions asked the user to relay their experience related to these activities.
- **Additional questions** (q=5): These questions required answers that could not be specifically fit into a standardized set of options (see below), and were asked individually. All were related to categories 2-7 above and were used to provide additional context to average categorical responses (see below).

Invited respondents (n=155) were selected for their direct knowledge and/or expertise regarding restoration activities in the Delaware Estuary. Respondents were sourced from the following groups:

- PDE committee members: Ecological Implementation Committee (EIC), Science & Technical Advisory Committee (STAC).
- Regional ecological committees and work groups: Delaware Living Shorelines Committee, NJ Coastal Resiliency Collaborative, Tidal Wetlands Monitoring Network, and the DE & NJ Salt Marsh Work Group.
- Regional NGOs: The Nature Conservancy, American Littoral Society, DE Center for Inland Bays, and Barnegat Bay Partnership.
- State entities: NJ and PA Departments of Environmental protection and DE Department of Natural Resources and Environmental Control - additionally, potential respondents were selected from multiple internal sectors, including the regulatory divisions.
- Federal entities: Environmental Protection Agency Regions 2 and 3, US Fish and Wildlife Service, and the Army Corps of Engineers.
- Restoration practitioners who are directly involved in one or more of the interview categories.

Although not all respondents' work singularly focuses on activities in the Delaware Estuary, their time and experience led to accumulated knowledge of these activities in the Delaware Estuary within the associated time frame of 2017–2021.

All questions were multiple-choice, with the exceptions of the demographic and the additional questions (categories 1 & 8, respectively), and asked the user to answer each question by selecting from the following list of standardized responses with associated point values in parentheses to inform analysis:

- Substantially increased (5)
- Moderately increased (4)
- Not changed (3)

- Moderately decreased (2)
- Substantially decreased (1)

Standardized responses, although more general in terms of the detail of information provided per respondent, allowed for the calculation of some summary statistics that reflect the general thoughts of the group as a whole. The evaluation included the relative proportion of responses and mean response per question, and by calculating a grand mean response per category (e.g., innovation, regulatory climate). Mean responses were assigned factor levels as follows:

- Substantially increased: 4.01-5.00
- Moderately increased: 3.01-4.00
- Not changed: 3.00
- Moderately decreased: 2.99-2.00
- Substantially decreased: 1.99-1.00

Results

Respondent Demographics

A table of responses is provided for each question below. Responses with the highest percentages are highlighted in green.

Sectors of questionnaire respondents were represented variably, with the highest percentage of respondents coming from state and federal agencies (Table 8.1). The majority of respondents have been involved with restoration in the Mid-Atlantic region for 10–20 years, while the lowest percentage of respondents has been involved less than three years (Table 8.2).

The habitats that questionnaire respondents worked in more than any others were tidal freshwater and upland habitats, with many respondents working across categorical options (Table 8.3). The top three areas of involvement indicated by respondents were site evaluations, monitoring, and project design, followed closely by installation and outreach (Table 8.4).

The majority of respondents indicated that the organization they represent is not a partner of DELEP or a member of any DELEP committees such as the STAC or the EIC (Table 8.5). While a majority of respondents do not represent DELEP-affiliated organizations or sit on any DELEP committees, about two-thirds of questionnaire respondents indicated that their organization has partnered on projects with PDE (Table 8.6). The majority of respondents indicated that the organization they represent is not a partner of DELEP or a member of any DELEP committees such as the STAC or the EIC. While a majority of respondents do not represent DELEP-affiliated organizations or sit on any DELEP committees, about two thirds of questionnaire respondents indicated that their organization has partnered on projects with PDE.

Table 8.1 Percentage of respondents who selected each response regarding their affiliated sectors.

#	Demographics Questions	Federal	State	Private	Academic	NGO	Other
2	Sector:	23%	23%	18%	9%	20%	7%

Table 8.2 Percentage of respondents associated with respective years involved in the restoration field in the region. Years 3-5 are those since 2017.

#	Demographics Questions	0-3	3-5	5-10	10-20	20-30	30+
3	How many years have you been involved with restoration in the Mid-Atlantic?	6%	10%	28%	30%	14%	12%

Table 8.3 Percentage of respondents associated with each environment associated with restoration practice in the Delaware Estuary.

#	Demographics Questions	Upland	Tidal Freshwater	Tidal Brackish/ Saltwater	Non-tidal Freshwater	All of the above	Other
4	Typical working habitat (choose up to 3)	23%	23%	18%	9%	20%	7%

Table 8.4 Typical restoration practice of respondents. Note that participants were able to select as many options as they wanted and that the number for each option in the table indicates the number of times that option was chosen, not a percentage of participants that chose it.

Demographics Question #5	
What area of restoration practice are you typically involved in (choose all that apply)?	#
Site evaluations	55
Distributing funds	25
Project design	50
Permitting	38
Installation	48
Monitoring	51
Training	20
Outreach	45
Regulating	10
Other	13

Table 8.5 Percentage of respondents affiliated or not affiliated with DELEP organizations or DELEP-partnering organizations.

#	Demographics Questions	Yes	No
6	Is your organization a Delaware Estuary Program (DELEP) partner or a member of any DELEP committees? DELEP partners are a group of regional organizations that, with PDE, work together to coordinate and implement long-term plans for the Delaware Estuary. DELEP committees include the Steering Committee, Estuary Implementation Committee (EIC), and the Science and Technical Advisory Committee (STAC).	46.99%	53.01%

Table 8.6 Percentage of respondents whose organizations have or have not partnered on projects with PDE.

#	Demographics Questions	Yes	No
7	Has your organization partnered on projects with PDE?	67.07%	32.93%

Categorical Questions

A table of responses is provided for each category of questions below (innovation, level of investment, etc…), including the percentage of respondents that selected each answer (significant increase, moderate increase, etc…), the mean response for each question, and a translation of the mean into a categorical response. Coloration for each question reflects the level of categorical; response as follows:

- Substantially increased — dark green
- Moderately increased — light green
- Not changed — white
- Moderately decreased — light red
- Substantially decreased — dark red

Innovation (Q8-12)

Responses indicate a positive trend in innovation (Table 8.7). More than 74% of respondents believe there has been a positive trend in on-the-ground innovation and the use of feedback to inform lessons-learned and project design (Q8–11). Most agencies have recognized the desire and need for restoration innovation, created restoration-specific general permits with the intent to facilitate restoration projects, partnered with other entities for restoration training and demonstration projects in efforts to advance restoration science, and even performed their own restoration studies and pilot projects with at least some of the goals being associated with innovation. Generally, respondents indicated that the regulatory process since 2017 has moderately increased the application of innovation (Q12).

Although innovation tends to be trending in a positive direction over the past 5 years, there are still many challenges to overcome. Over the last 5 years, there has been a high level of turnover in the restoration community across sectors (e.g., agencies, NGOs, and the private sector) representing a loss of institutional knowledge and experience. Regarding the neutral or negative effects associated with regulatory processes (Q12), agencies may be in a difficult position regarding how to support innovation opportunities that may not align with the currently accepted categories. Most general permits, by their nature, are developed to facilitate routine, "low risk", applications, and as such these permits typically do not accommodate innovation. One aspect of innovation not addressed in this questionnaire concerns project size. Upscaling is a form of innovation due to its unique challenges relative to small projects. Elements related to managing large volumes of material, handling larger material, project coordination, project duration, permitting, monitoring, outreach, etc, can all have unexpected challenges, needs, and costs. How these elements are dealt with can be a form of innovation. Finally, although innovation can lead to technical restoration advancements, practitioners and agencies may be hesitant to embrace innovation because of concern regarding the under-performance of novel techniques. Efforts need to be made to develop innovative, multi-sector projects with the single goal of advancing the state of science and knowledge, with a science-based tolerance towards the negative implications of less than ideal outcomes. The idea of "breaking a few eggs to make a cake" applies to innovation, provided there are reasonable checks and balances. For example, appropriate monitoring and assessment can support innovation through the development of a better understanding of the appropriate application of goal-based tactics. Standardized monitoring can provide quantitative information to inform innovation needs (e.g., the need for new tactics to meet specific goals under site-specific conditions), and how to transform lessons learned from short-term failures into long-term solutions.

Table 8.7 Summary statistics for each question asked in the innovation section of the survey. Percentage of respondents that selected each categorical response are noted in columns 3-7: SI=substantially increased, MI=moderately increased, NC=not changed, MD=moderately decreased, and SD=substantially decreased. Means (column 8) were calculated based on the method described in Section 8.2, and reflect the average score among respondents per question, which is translated back to a categorical response in the last column (9). Coloration reflects increases (dark and light green), no change (white), and decreases (light and dark red).

#	Innovation Questions: Since 2017,	SI (%)	MI (%)	NC (%)	MD (%)	SD (%)	Mean	Categorical Translation
8	The use of feedback, lessons-learned, and/or results to inform project designs has ____.	22.5	58.75	16.25	2.50	0.00	4.01	SI
9	The general level of innovation applied by regional organizations has____.	11.54	69.23	17.95	1.28	0.00	3.91	MI
10	The level of innovation applied by the organization they represent has____.	21.79	52.56	25.64	0.00	0.00	3.96	MI
11	The proportion of new and proposed projects that incorporated (not just discussed) sea level rise into their design has____.	20.78	53.25	23.38	2.60	0.00	3.92	MI
12	The effects or actions associated with the regulatory process has ____ the opportunity for the application of innovation.	2.60	44.16	40.26	7.79	5.19	3.31	MI
						Overall Categorical Result	3.82	MI

Level of Investment (Q13-17)

Of note, question 16 was phrased so that an increase indicated a negative effect. To account for this in the mean calculation and overall interpretation, the categorical scores (i.e., SI =5, MI = 4, …., SD = 1) was reversed (e.g., SI = 1,…, SD = 5) for these questions (*).

Overall, level of investment is trending in a positive direction (Table 8.8). Nearly 65% of respondents believe there has been an increasing trend in restoration funding opportunities (Q13), and more than 70% believe there has been an increasing proportion directed towards implementation (Q14). Despite this perceived trend, more than 55% of respondents do not feel there has been any change regarding monitoring funding (Q15), match requirements (Q16), or recipient diversity (Q17). As such, it may be construed that the trend of increased funding may be slightly outpacing the trends associated with the administration of funding as experienced by the respondents. It is worth noting that 30.26% of respondents felt that match requirements were resulting in increased difficulties related to application submission (Q16). Although this is not the majority, it is a notable percentage of respondents.

Table 8.8 Summary statistics for each question asked in the level of investment section of the survey. Percentage of respondents that selected each categorical response are noted in columns 3-7: SI=substantially increased, MI=moderately increased, NC=not changed, MD=moderately decreased, and SD=substantially decreased. Means (column 8) were calculated based on the method described in Section 8.2, and reflect the average score among respondents per question, which is translated back to a categorical response in the last column (9). Coloration reflects increases (dark and light green), no change (white), and decreases (light and dark red).

#	Investment Questions: Since 2017,	SI (%)	MI (%)	NC (%)	MD (%)	SD (%)	Mean	Categorical Translation
13	Funding for restoration opportunities has ____.	19.48	45.45	23.38	11.69	0.00	3.73	MI
14	The proportion of available funding that is focused on implementation rather than project planning has ____.	20.78	50.65	24.68	3.90	0.00	3.88	MI
15	The proportion of available funding for project monitoring has ____.	2.60	29.87	58.44	9.09	0.00	3.26	MI
16	Match requirements have ____ the level of difficulty in applying for funds.	0.00	5.26	57.89	30.26	6.58	2.62	MI*
17	The distribution of funding across diverse recipients has ____.	6.67	30.67	57.33	4.00	1.33	3.37	MI
						Overall Categorical Result	3.37	MI

Funding sources are looking to find recipients that offer the best chance of success. Established and more networked entities likely still receive a large portion of funding, while smaller entities may be viewed as a higher funding risk. Often, these smaller entities have to rely on smaller local grants, raise their own capital, or compromise their projects to obtain funding. But smaller entities may have more personal investment in the communities in which they are located, resulting in high energy and deep commitment. Partnerships are perhaps the most effective means to get funding to the most diverse range of users while addressing many of the risk-related issues that concern funders such as financial stability, experience, and a proven track record. Larger entities are beginning to recognize the unique role and perspective smaller, community-based entities can provide including the local expertise, local support, landholdings, and commitment. Although not a perfect solution for every situation, partnerships are becoming more common and popular.

Regulatory Climate (Q18-23)

Of note, questions 18, 19, & 21 were phrased so that an increase indicated a negative effect. To account for this in the mean calculation and overall interpretation, the categorical scores (i.e., SI =5, MI = 4, …., SD = 1) was reversed (e.g., SI = 1,…, SD = 5) for these questions (*).

Overall, respondents felt there was a moderate decrease in restoration project facilitation from the regulatory sector (Table 8.9). Although respondents felt there was a moderate increase in permit options (Q20) and in-agency willingness to work with applicants (Q22), the level of effort and time involved in the permitting process(Q18, Q19), as well as the complexity and perceived technical competency of agencies (Q21, Q23) negatively affected the overall perception of the regulatory climate. It should be noted that nearly half of the respondents felt that there was no change in the level of effort (Q19), regulatory options (Q20), complexity (Q21), or technical competence (Q23), so these positive and negative perceptions are slight. It is also important to note that nearly half of the respondents were from agencies. It is unclear if or how responses to these questions differ among subsectors of respondents, but future questionnaire evaluations should evaluate results by sector or subgroups of sectors.

Table 8.9 Summary statistics for each question asked in the regulatory climate section of the survey. Percentage of respondents that selected each categorical response are noted in columns 3-7: SI=substantially increased, MI=moderately increased, NC=not changed, MD=moderately decreased, and SD=substantially decreased. Means (column 8) were calculated based on the method described in Section 8.2, and reflect the average score among respondents per question, which is translated back to a categorical response in the last column (9). Coloration reflects increases (dark and light green), no change (white), and decreases (light and dark red).

#	Regulatory Climate Questions: Since 2017,	SI (%)	MI (%)	NC (%)	MD (%)	SD (%)	Mean	Categorical Translation
18	The amount of time to complete the permit process has ____.	13.04	30.43	40.58	25.94	0.00	2.59	MI*
19	The change in level of effort and associated costs of acquiring permits has ____.	18.18	25.76	48.48	6.06	1.52	2.47	MI*
20	The regulatory options for permitting a project have ____.	5.88	39.71	47.06	5.88	1.47	3.43	MI
21	The complexity of the permitting process has ____.	9.09	31.82	54.55	3.03	1.25	2.56	MI*
22	The general willingness of agencies to work with the applicant has ____.	4.35	47.83	34.78	11.59	1.45	3.42	MI
23	The perception of technical competence within permitting agencies has ____.	1.45	21.74	47.83	20.29	8.70	2.87	MD
	Overall Categorical Result						2.89	MD

Interestingly, the majority of respondents (52.18%, Q22) indicated an increase in the general willingness of agencies to work with applicants. This is an intriguing point to consider in discussions of the permitting process as a whole and how it can be improved. With regard to funding, the permitting sector of the restoration process has not been allocated funds in the same regard as other sectors, like installation, over the last 5 years. So while costs have increased to obtain permits (i.e., increased time requirements), fewer funds have been allocated to managers for this process. If additional funds were allocated towards the regulatory process, agencies would theoretically be able to invest in improving the efficiencies of their offices, such as: additional training, additional hiring to avoid long approval times, higher pay for tenured personnel to better incentivize these employees to remain in important positions where staff is needed, and better employment benefits to attract new hires. Because respondents indicated on average that there has been a moderate increase in cost, effort, and complexity, it has likely become frustrating that the process has not gotten easier or smoother, only more expensive. However, because a majority of respondents indicated an increase in agencies' willingness to work with applicants, it is possible that some stakeholders are experiencing improved relationships with regulatory personnel, which could be helpful in the future in navigating rising direct and indirect costs.

Implementation (Q24-28)

Of note, question 28 was phrased so that an increase indicated a negative effect. To account for this in the mean calculation and overall interpretation, the categorical scores (i.e., SI =5, MI = 4, …., SD = 1) was reversed (e.g., SI = 1,…, SD = 5) for these questions(*).

Respondents conveyed an overall moderate increase in activities and qualities associated with restoration project implementation (Table 8.10). More than half the respondents perceived increases in the rate of implementation (50.72%, Q24), quality of projects (57.97%, Q25), the quantity that considers future changes in their design (76.47%, Q26), and the quantity aimed at testing new techniques (75.74%, Q27). Additionally, more than 65% of respondents perceived no change and ~24% of respondents indicated a moderate decrease associated with project failure or under-performance (Q28). This suggests that existing projects have generally been successful and that more restoration work has been accomplished in the Delaware Estuary over the last 5 years than prior to 2017.

The perceived increase in project quality since 2017 could be attributed to a few factors. Stakeholders in the Delaware Estuary indicated that in the last 5 years, there has been an increase in the trial of new techniques and applications in restoration projects. While there are always lessons learned when attempting new methods, the use of new technology or different procedures could be heightening project quality due to better efficiency or could be more cost-effective if these technologies save time, resources, or labor. Given this, managers attempting new techniques in their restoration projects should provide updates on the success of a new method and how it affected their project(s) at meetings with partners or by presenting results at conferences. Given that 76% of respondents indicated a perception of increased consideration of future conditions like sea level rise and habitat transgression in project development, these components should be at the forefront of long-term restoration work. A heightened consideration of future conditions implies that restoration work could provide managers with the potential opportunity to save money and time in future restoration considerations.

Monitoring (Q29-34)

Respondent responses indicated metrics associated with monitoring are trending in a positive direction (Table 8.11). More than 70% of respondents felt that the availability of project monitoring guidance (Q31) and the understanding of the effects of restoration projects as a result of monitoring (Q34) had increased. Although there were indications of moderate increases on average, between 50%-60% of respondents felt there had been no change in optional (Q29) or required (Q30) monitoring activities, as well as standardization (Q32) or funding (Q33) of monitoring activities.

The overall moderate increase with approximately half of respondents indicating no change in standardization of monitoring is interesting. There have been standardized monitoring guidance documents available in NJ and DE since 2017, which many of the respondents of this questionnaire contributed to or reviewed. It's possible that only those who participated in the creation of the documents actually use them, and expanded promotion may facilitate further standardization. But this has apparently not reduced the perception of understanding that monitoring provides, and that increases in funding paired with regulatory requirements may further support greater standardization, but a prescription to specific metrics and methods needs to be considered carefully.

Table 8.10 Summary statistics for each question asked in the implementation section of the survey. Percentage of respondents that selected each categorical response are noted in columns 3-7: SI=substantially increased, MI=moderately increased, NC=not changed, MD=moderately decreased, and SD=substantially decreased. Means (column 8) were calculated based on the method described in Section 8.4.1, and reflect the average score among respondents per question, which is translated back to a categorical response in the last column (9). Coloration reflects increases (dark and light green), no change (white), and decreases (light and dark red).

#	Implementation Questions: Since 2017,	SI (%)	MI (%)	NC (%)	MD (%)	SD (%)	Mean	Categorical Translation
24	The rate of individual project implementation has ____.	10.14	40.58	34.78	11.59	2.90	3.43	MI
25	The average quality of implemented projects has ____.	7.25	50.72	37.68	4.35	0.00	3.61	MI
26	The quantity of implemented restoration projects (e.g., sea-level rise, habitat transgression) that consider a change in future conditions rather than immediate conditions alone has ____.	16.18	60.29	20.59	2.94	0.00	3.90	MI
27	The quantity of implemented projects aimed at testing new techniques or applications has ____.	11.45	64.29	21.43	2.86	0.00	3.84	MI
28	The percentage of failures or underperformance associated with implemented projects has ____.	1.49	8.96	65.67	23.88	0.00	3.12	MD*
				Overall Categorical Result			3.58	MI

Outreach (Q35-40)

More than 80% of respondents felt that information regarding restoration needs (Q35), activities (Q36), and positive outcomes (Q37) has increased since 2017. More than half of the respondents felt that messaging clarity has increased (59%, Q40), and ~50% perceived no change in outreach regarding negative lessons learned (Q38) and messaging consistency (Q39, Table 8.12). Overall, these results indicate that outreach has been moderately increasing since 2017 in multiple areas. These positive results regarding messaging are encouraging but there may be an issue regarding communicating the negative (i.e., project "failure") restoration lessons learned/results. This is fundamentally one of the most important aspects to advance the practices of restoration but also represents one of the hardest aspects for practitioners to be open about and discuss openly. Part of this may be due to a need to show positive results, as negative bias can skew the perception of effectiveness. For example, multiple positive outcomes may be overshadowed by a single negative result; therefore, outreach focuses on the positives rather than the negative. Further conversation in appropriate language to discuss "failure" is needed, as well as the proper way for these to be communicated.

Table 8.11 Summary statistics for each question asked in the implementation section of the survey. Percentage of respondents that selected each categorical response are noted in columns 3-7: SI=substantially increased, MI=moderately increased, NC=not changed, MD=moderately decreased, and SD=substantially decreased. Means (column 8) were calculated based on the method described in Section 8.2, and reflect the average score among respondents per question, which is translated back to a categorical response in the last column (9). Coloration reflects increases (dark and light green), no change (white), and decreases (light and dark red).

#	Monitoring Questions: Since 2017,	SI (%)	MI (%)	NC (%)	MD (%)	SD (%)	Mean	Categorical Translation
29	Optional project monitoring has ____.	5.71	32.86	51.43	8.57	1.43	3.32	MI
30	Regulatory monitoring requirements have ____.	2.94	38.24	55.88	2.94	0.00	3.41	MI
31	The availability of project monitoring guidance has ____.	23.19	49.28	23.19	4.35	0.00	3.91	MI
32	Standardization of project monitoring across projects has ____.	8.70	37.68	53.62	0.00	0.00	3.55	MI
33	Funding for monitoring has ____.	0.00	27.54	59.42	11.59	1.45	3.13	MI
34	Understanding of effects of restoration projects as the result of monitoring has ____.	31.43	40.00	28.57	0.00	0.00	4.03	SI
					Overall Categorical Result		3.56	MI

Additional Questions

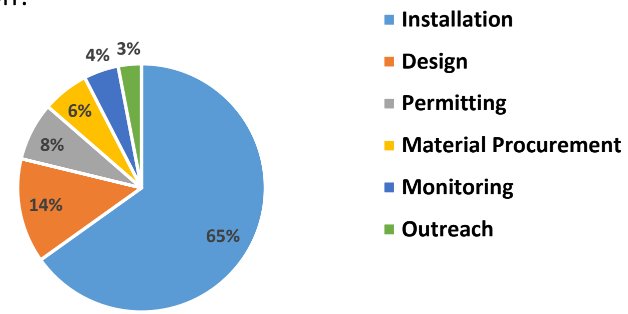
Respondents felt that installation receives the greatest amount of funding allocation (65%), while the next highest response was project design which received 13.64% (Q41) (Fig 8.10). The majority of respondents (74%) indicated that they receive most of their information about restoration topics via work group and committee discussions and/or presentations, such as in meetings (Fig 8.10). Only 8% of respondents said they receive information through peer-reviewed journals, which represents the "established" or "accepted" means of knowledge transfer, and no one said they received this information from tabling/ outreach events (Fig 8.10). This is likely due to the sampling pool not being the target audience of these events and being on the "inside" regarding information sharing (Q42). When asked what the best method for absorbing information about restoration activities were, the majority of respondents selected the site visit/field trip option (34%), followed by work groups (30%) and conferences (17%, Q43) (Fig 8.10). Interestingly, although work groups and conferences were identified as the primary sources of receiving information regarding restoration activities, that site visits ranked higher for information absorption show the essential, valuable nature of these activities are still seen by the stakeholders as a very important means of absorbing information (Q43) (Fig 8.10).

The majority of respondents (71%) selected that only some of the requirements involved in the regulatory process seem appropriate while others only seemed appropriate to be of interest only to the permitting agency. Traditionally, agencies focused on performance-based monitoring for the purpose of determining if the proposed project successfully resulted in the establishment or enhancement of a specific resource type and provided the associated expected functions. The responses to this question suggest that regulatory requirements are going beyond this fundamental purpose for monitoring and heading more toward special interests or perhaps more research-related interests. Of interest is why this trend is occurring. Is it a result of lost institutional knowledge, new innovations, or one or more other reasons? Also, how are the additional monitoring requirements burdening the permitted project? However, 20% of respondents indicated that their perception of requirements is generally appropriate for their projects, so more discussion regarding regulatory monitoring requirements is warranted.

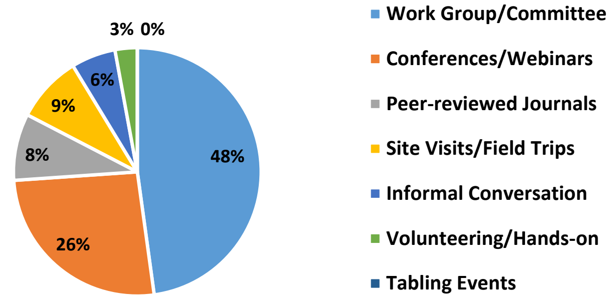
Table 8.12 Summary statistics for each question asked in the outreach section of the survey. Percentage of respondents that selected each categorical response are noted in columns 3-7: SI=substantially increased, MI=moderately increased, NC=not changed, MD=moderately decreased, and SD=substantially decreased. Means (column 8) were calculated based on the method described in Section 8.2, and reflect the average score among respondents per question, which is translated back to a categorical response in the last column (9). Coloration reflects increases (dark and light green), no change (white), and decreases (light and dark red).

#	Outreach Questions: Since 2017,	SI (%)	MI (%)	NC (%)	MD (%)	SD (%)	Mean	Categorical Translation
35	Information being relayed to stakeholders regarding restoration needs has ____.	15.94	68.12	14.49	1.45	0.00	3.99	MI
36	Information being relayed to stakeholders regarding restoration activities has ____.	17.65	63.24	17.65	1.47	0.00	3.97	MI
37	Information being relayed to stakeholders regarding positive restoration lessons learned/results has ____.	15.94	68.12	13.04	2.90	0.00	3.97	MI
38	Information being relayed to stakeholders regarding negative (i.e., project "failure") restoration lessons learned/results has ____.	4.35	34.78	53.62	5.80	1.45	3.34	MI
39	The consistency in outreach messaging (e.g. what has been working, costs, feasibility of techniques) regarding restoration needs, activities, and/or results has ____.	5.88	36.76	54.41	2.94	0.00	3.46	MI
40	The clarity in outreach messaging has ____.	10.29	48.53	35.29	5.88	0.00	3.63	MI
				Overall Categorical Result			3.73	MI

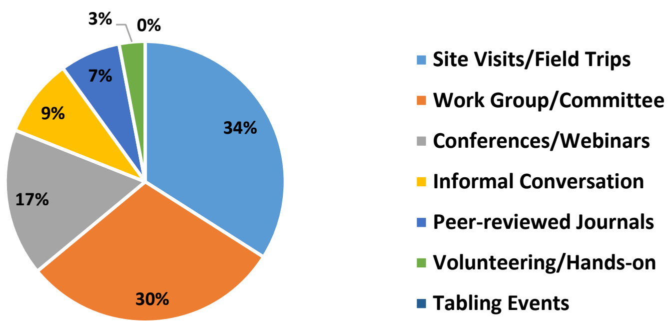
Q41: Which area of restoration has experienced the greatest increse in awardee resource allocation?



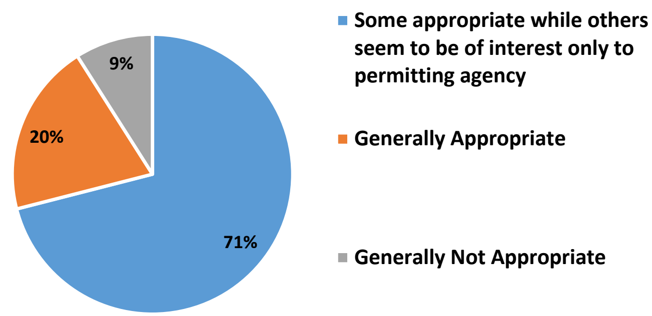
Q42: How do you receive most information regarding restoration needs, activities, and outcomes?



Q43: How do you best absorb informtion regarding restoration needs, activities, and outcomes?



Q44: For their associated projects, regulatory monitoring requirements are....



Q45: My Top three restoration needs looking forward to 2027 are....

Figure 8.10 Summary of responses from additional non-physical questions.

The top three restoration needs looking forward to 2027 as selected by questionnaire respondents were salt marsh edge (16%), salt marsh platform (14%), and the protection of transgression zones (14%, Q45). These topics are common in many regional restoration work groups but may reflect the imbalance of the questionnaire respondents. However, Q4 indicated that the most common area of restoration that respondents worked in were upland habitats and freshwater tidal habitats. Thus, these results suggest somewhat of a consensus among restoration practitioners of varietal backgrounds and expertise that these three priorities are currently the most important across the field. Additionally, each agency and NGO has a unique mission or programmatic goal for one or multiple natural resources. Laws, regulations, policies, programmatic management plans, and precedence control how an entity applies its mission and goals toward innovative technology. Because of this dynamic, it is not uncommon for entities to have nonparallel views on certain applications of innovations. It often takes time, coordination, and teamwork between entities to sort out such inconsistencies. Given this, it is promising that respondents of different entities and backgrounds seem to agree on what needs to be prioritized in the restoration field.

Discussion

Moving Forward with Restoration Efforts in the Delaware Estuary

Overall, restoration activities have been increasing in the Delaware Estuary 2017-2021 relative to the 2006-2011 and 2012-2016 periods (Fig 8.4), largely focused on tidal wetlands, forests/woodland, agriculture, and beaches (Fig 8.7). Moving forward, the top identified needs for restoration in the estuary are the salt marsh edge, salt marsh platform, and transgression zones (Fig 8.10, Q45), despite the fact that the largest self-identified restoration focus areas by respondents were upland and tidal freshwater systems (Table 8.3). Additionally, the questionnaire suggests that the level of investment for restoration is trending in a positive direction, but uncertainties lie with the acquisition and distribution of funding, as well as the matching funds' requirements (Fig 8.11, Table 8.8). Future efforts should include simplifying the grant application process and finding additional means to get funding to both the larger and smaller restoration entities, while either reducing the match requirements or broadening the range of qualifying funds.

There has been a hard push towards innovation by a small nucleus of practitioners. There appears to be a call for further increases in innovation by stakeholders; however, many current issues make innovation difficult, including the costs and complexity of acquiring permits as well as acquiring funding for those innovations (Tables 8.8, 8.9). Yet, stakeholders felt that there was a willingness of regulatory agents to work with application and support innovation (Tables 8.7, 8.9). Innovation can be complicated and often site-specific. A successful innovative project is often copied. For example, if the new user of the technology does not fully understand the use of the innovation, negative consequences or under-achievement of outcomes may occur. As such, information transfer between entities is essential to maximize the continued success of innovation. Despite these variations in conception in practitioners, innovation still scored well indicating a general feeling that the opportunity and momentum are in place to move forward if challenges in funding and permitting can be overcome (Table 8.7)

Perception of regulatory climate is the only clearly under-performing metric (Figure 8.11, Table 8.9). There has been limited change in restoration project regulations over the past five years, other than the development of new general permits; otherwise, costs have increased, and obtaining permits has become more arduous (Table 8.9). This may have caused project managers to feel inclined to adjust the scopes of future projects to make them more narrow (or aligned with more commonly accepted practices or scales) to counteract high permitting pricing or to avoid unnecessary requirements of permitting agencies. If this were to occur, other aspects of restoration project management would be impacted, and may lead to more conservative restoration work in the Delaware Estuary rather than projects developing that are expansive and growing. Moreover, if project managers are not capable of or are unable to oversee cost increases for the same procedures, it could contribute to fewer projects being implemented and a generally less efficient regulatory process. Relatedly, a smaller subset (20%) of questionnaire respondents reported that they felt regulatory monitoring requirements were generally appropriate for their associated projects (Fig 8.10, Q44). Conversely, 71% of respondents felt that while some requirements are sometimes appropriate, other times these requirements only seem to be of interest to the permitting agency. This trend could be viewed in several different ways, depending on the role of the agency. It may represent a loss of institutional knowledge within regulatory programs as a result of personnel turnover, with hefty responsibility placed on new staff. It could also represent a disconnect between regulatory requirements, and the rapidly advancing suite of tools and methodologies employed by practitioners.

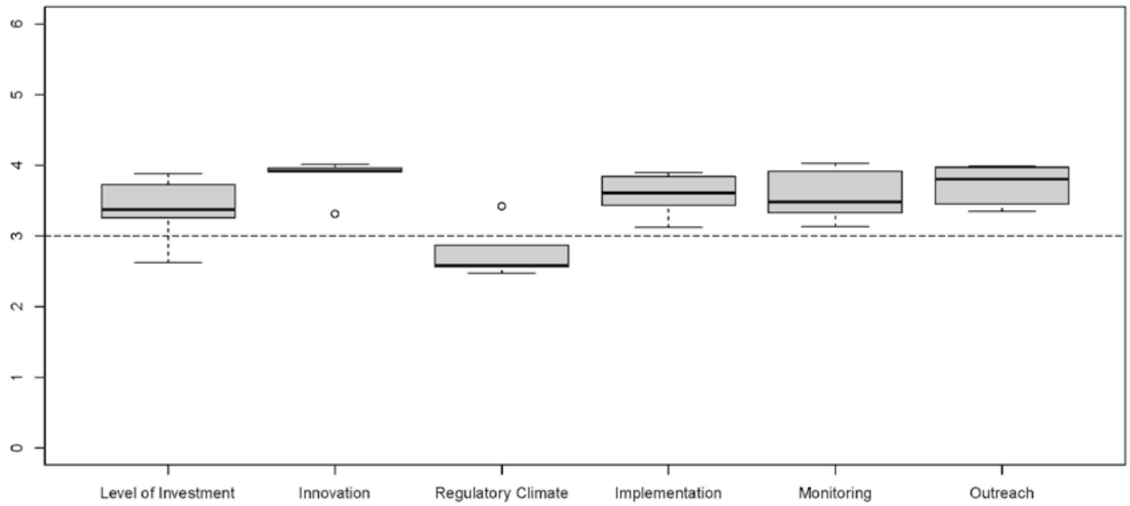


Figure 8.11 Boxplot of average responses to questions regarding each metric in the restoration survey. All metrics other than Regulatory Climate had an average response of Moderately Increased or greater for each survey question.

While costs have risen and general permits are developing, the main change in the regulatory climate over the past 5 years has been the turnover of personnel. This has resulted in the net loss of experience, institutional knowledge, further innovation, and other technological advancement abilities. Thus, there is a need for outreach education and experiences for the regulatory community, including middle management. As such, regulatory climate is linked to outreach in that outreach needs to create a focus on agency-targeted workshops and partnerships. To date, outreach has been consistently focused on external stakeholders (e.g., the general public), but outreach for internal personnel would also be beneficial. A key way to implement change in the restoration field will be to refocus some outreach towards practitioners and new agency individuals, as this will be integral to improving the overall regulatory climate, better allocation of funds, more easily identifying gaps, and improving project efficiency.

Additionally, the current status of outreach in the restoration field presents a huge bias in considering positive results and shying away from negative results. Negative outcomes as a result of learning are important to consider when implementing new policies and management techniques; however, positive outcomes protect methodology, so when results are positive, general stakeholders don't hear about any drawbacks concluded from results and therefore skew willingness to continue to learn, and potentially inhibit the ability or opportunity to secure funding in the future to continue learning new lessons relative to the restoration field.

Monitoring is a key component in advancing restoration practice and assessing the effectiveness of cutting-edge innovation. Unfortunately, maximizing monitoring efforts is becoming increasingly difficult due to the increased scope of investigation, regulatory requirements, and pressures of optimizing funding to get the most acres/miles of restoration completed. As many agency partners are unable to provide funding for monitoring, and since most budgets have been finalized long before regulatory requirements are known, the restoration community needs to become more adept in building a baseline and post-construction monitoring costs into initial project scopes, increasing efficiencies and regulatory assurance. Moreover, investment results evaluated through monitoring can support arguments for increased funding for monitoring in years to come.

The collection and assimilation of monitoring data lead to the refinement of methodologies and practices, which provide ecological and institutional benefits. Lessons learned enable the evolution of practices and, in some cases, transferability. One consistent issue is the comparative disparity in monitoring between more established and well-funded programs/projects and smaller grass roots or community-based efforts. Smaller project partners usually have a tighter budget and may not be able to implement a full suite of monitoring protocols that regulatory agencies normally require or expect with better-funded projects. The lack of scaling regarding requirements and methodologies creates a burden of these efforts that may diminish project size below the thresholds necessary for effectiveness. Successful landscape-level restoration efforts will be the result of a synergy between community-driven, grassroots efforts, and larger-partnered planning efforts. Growth of practices, innovation, expanding knowledge of the importance of ecosystems in communities, and why conservation and restoration practices are important all need to work in conjunction at the time of project implementation. A lack of experience and scientific knowledge across the regulatory community, grant reviewers, and partners has led to the juxtaposition of equity in restoration and understanding of how scientific knowledge is gained and utilized. We need advancements in scientific knowledge, especially with emerging issues (e.g. geochemistry in elevation enhancement projects, wave attenuation, climate change feedback, over-engineering, “soft engineering”, etc.), but not at the expense of community and civic engagement-led restoration. Scaling of requirements and methodologies, paired with a better understanding of the value of qualitative monitoring can help us achieve more equitable outcomes. In some cases, a photo time series can provide as much understanding of a project’s outcomes and its impact on a community as more precise metrics, depending on the goals of the effort.

As restoration practitioners and the supporting community work to advance practices, it needs to be recognized that as a whole we are lagging behind in our responses and scaling in restoration with response to climate change and relative sea level rise. The observed losses of vegetated tidal habitat due to sea level rise and anthropogenic-induced disequilibrium have resulted in extreme variations in wetland conditions across the watershed. “Hotspots’ of wetland loss and/or decreased wetland function illustrate the need to increase the scale (spatial size) and the rate at which wetland restoration projects are conducted. The annual rate of restoration (i.e., acres and the number of projects) should outpace the average annual losses, but this is currently not the case. The losses of coastal habitat have been recognized and monitored for decades, but the effects of anthropogenic pressures have brought to light the true magnitude of the decoupled nature of the evolution of these aquatic and coastal systems with the rate of sea level rise and changes in the distribution of sediments within the system. Loss of habitat function and spatial distribution mark an increase in the number of obligate species that are imperiled as climate change, sea level rise, and anthropogenic impingement shrink the ecosystem landscape. Restoration is not an immediate process or cure for the ails of the systems as there is a lag in the restored function which can vary significantly between habitat types and biotic communities. The lack of large-scale (i.e., larger footprint and total area) restoration, especially in tidal marshes, indicates that the restored areas that will anchor the habitats for impaired species 3 to 10 years from now should already have been restored or been in the process of being implemented.

Demonstration projects are necessary to exemplify the effectiveness of practices to resource managers and regulatory agencies, but there is a lack of urgency in moving these demonstration projects to the appropriate scale for landscape-level effects. A mosaic of small projects can be effective in some habitat types and locations (especially in urbanized landscapes) but it is understood that generally larger patches and connected habitats (corridors) create a better mosaic that is ultimately needed in the estuary. A collection of smaller patches' habitat function is lower than that of a larger continuous block, as there is not a linear relationship between discrete blocks to continuous patches in ecological function. Large blocks or patches are more desirable as they inherently have higher resilience and ability to ecologically and spatially adapt to changes in their base-level and abiotic drivers. Climate change is affecting the entire region, and our restoration focus needs to have that same corresponding attention. Larger connected projects on the wider landscape are needed, but there are concerns regarding transferability that may slow larger intervention efforts.

Significant differences in how restoration practices and scales should be implemented on the landscape exist between states, counties, and federal agencies. There will be hesitancy in implementing innovative or larger-scale projects, as the fear of failure can dampen risk-averse entities' enthusiasm for supporting these types of projects. Questionnaire respondents indicated that the percentage of implemented projects that have failed or underperformed has moderately decreased in the last 5 years, but this may be to more conservative goal alignment rather than full intervention success so this concern is still valid among practitioners (Table 8.10). The transferability of project design themes, specifications, and methods from one state or organization to another, or outside the region, is recognized as a major issue that is impeding the expansion of implementation and increasing project size. Successful projects and practices outside the region need to be effectively integrated into the restoration community toolbox if future goals and objectives to combat climate change are expected to be achieved. Leveraging partners' lessons learned and experiences is a fundamental necessity that needs to be addressed for the restoration community and has to be a focal point for future investment in the communities' resources.

The future of restoration needs to be aligned around a focus on ecosystem-scale function, and not solely focus on habitat- or species-specific goals. We have learned that restoration projects that take an ecosystem approach tend to be more resilient and establish a better baseline for future natural evolution as they provide more opportunities for spatial ecological response. Anthropogenic influences have decoupled their natural evolution and we need to ameliorate these mistakes, and potentially erroneous management actions, by restoring the foundational physical aspects of the coastal systems (i.e., hydrology, elevation, etc.). An ecosystem approach on a larger area (on a basin or drainage area landscape) incorporating abiotic and biotic feedback, supports greater resilience for species and people through a wider variety of associated benefits (e.g., flood attenuation, nutrient removal, wildlife-dependent activities). Despite the rapid evolution of the landscape and its habitats due to climate change and human use, the natural resources community often becomes entrenched in static views and ideas of what the landscape setting and evolution will soon look like. For example, not all erosion is bad, beaches do not belong everywhere, and some locational appropriateness can change over time with regard to restoration in the Estuary. Becoming comfortable with natural processes rather than imposing an ideological will may be the result of a cultivated idea or generally, the right decision made for a certain site or area. Not working on something that may be impacted by natural processes is not necessarily a lack of action being taken.

Recommendations

- Salt marsh edge stabilization, salt marsh platform elevation enhancement, and protection of transgression zones were identified as the top three necessities for restoration in the next five years (Fig 8.10) and should be prioritized moving forward. A need for some kind of increase in consideration of future conditions in project development was also identified, potentially including any or all of these top 3 considerations, despite questionnaire respondents representing different entities and aspects of the restoration field.

- Innovation is generally increasing (Table 8.7), but specific regulatory pathways for innovative tactics, possibly with more rigorous monitoring requirements to broaden restoration implementation knowledge, may facilitate further tactic development to meet changing spatial and ecological restoration needs.
- The funding process should have a simplified grant application process and find additional means to get funding to a diversity of restoration entities spanning a range of entity types and sizes. A scaled process that encourages entities that do not have larger capacities and experiences would greatly expand the work to the smaller community level which would be better implemented in communities that have not been engaged and create better "buy-in" by implementing self-identified community projects.
- Although there is a general increase in the costs, effort, and complexity of the regulatory process, there is also a perceived increase in the willingness of regulatory agencies to work with applicants (Table 8.9), indicating the potential for the development of cross-disciplinary relationships. It is important to keep relationship development moving in a positive direction, especially with newer agents, as these relationships may be a source of future modifications to ease the regulatory process for innovation and shortened timelines.
- Information on new technologies should continue to be shared in the restoration community to allow for continued development and refinement of new methods and techniques and avoid misunderstanding the use of new tools. The means by which this knowledge sharing should be implemented is unclear, but will always be rooted in formal and informal communication, and anchored in field-based knowledge transfers with constant and consistent communication between the restoration and the regulatory communities.
- The restoration community generally feels that project monitoring (required and optional) has been increasing (Table 8.11), but almost 60% of those polled indicated that there has not been any change in the standardization of monitoring methodologies, despite the development of a restoration monitoring framework in 2016 (Yepsen et al. 2016) developed by a multi-sector panel. Greater effort should be made to promote the existing monitoring documents, with periodic revisions to integrate new metrics and methods.
- Connecting related projects together on a wider spatial scale will assist in reaching restoration targets. Project teams including partners of various sizes with varying levels of project experience to tackle larger restoration objectives could facilitate more effective implementation with more thorough community representation. Restoration entities should maintain communication to better understand the results of implementation at multiple scales to design more comprehensive and effective projects.
- The restoration community needs to continue conveying negative outcomes of restoration efforts (Table 8.12), framed not as failures, but as key steps in a scientific inquiry process aimed at moving effective restoration practices forward. These appropriately framed results can help showcase how innovation, even if unsuccessful in meeting project-specific goals, can be successful in narrowing the focus of tactic development towards what works rather than burring these outcomes in fear of negative perception resulting in potential reapplication.

References

Delaware Living Shoreline Committee. (2018). Developing monitoring plans for living shoreline projects in Delaware: A goal-based framework. A report prepared by the Delaware Living Shorelines Committee Standards of Practice Subcommittee. https://www.delawarelivingshorelines.org/

Kassakian, J., Jones, A., Martinich, J. and Hudgens, D. (2017). Managing for no net loss of ecological services: an approach for quantifying loss of coastal wetlands due to sea level rise. Environmental management, 59(5), pp.736-751.

Kauffman, G. (2011). Economic Value of the Delaware Estuary Watershed. Institute for Public Administration, University of Delaware. 78 p. http://www.ipa.udel.edu/publications/DelEstuaryValueReport.pdf

Kreeger, D., J. Adkins, P. Cole, R. Najjar, D. Velinsky, P. Conolly, and J. Kraeuter. (2010). Climate Change and the Delaware Estuary: Three Case Studies in Vulnerability Assessment and Adaptation Planning. Partnership for the Delaware Estuary, PDE Report No. 10-01. 117 p. https://s3.amazonaws.com/ delawareestuary/pdf/Climate/Climate%20Change%20and%20the%20Delaware%20Estuary%20&%20 Appendicies_PDE-10-01.pdf

Lewis III, R.R. (1989). Creation and restoration of coastal wetlands in Puerto Rico and the US Virgin Islands. Wetland Creation and Restoration: The Status of the Science. Island Press, Washington, DC, pp.103-125.

Partnership for the Delaware Estuary. (2019). A Comprehensive Conservation & Management Plan for the Delaware estuary, 2019 Revision. https://s3.amazonaws.com/delawareestuary/2019+DelEst+Revised+CCMP.pdf

Yepsen, M., Moody, J., Schuster, E., eds (2016). A Framework for developing monitoring plans for coastal wetland restoration and living shoreline projects in New Jersey. A report prepared by the New Jersey Measures and Monitoring Workgroup of the NJ Resilient Coastlines Initiative, with support from the NOAA National Oceanic and Atmospheric Administration (NOAA) Coastal Resilience (CRest) Grant program (NA14NOS4830006).

Appendix 8.A: Stakeholder Questionnaire

Introduction

Thank you for taking the time to take the Partnership for the Delaware Estuary's 2021 questionnaire regarding non-physical influences on restoration practices in the Delaware Estuary. Your answers to the following questions will be used to develop the Restoration chapter of the Delaware Estuary Program 2022 Technical Report for the Estuary and Basin (TREB). The purpose of the TREB is to assess the overall environmental condition of the watershed by examining the status and trends of key indicators that reflect the health of its natural systems. The TREB is produced every five years, with the last publication in 2017. The Restoration chapter was first introduced in 2017 with the objective to provide information on restoration efforts and progress. As restoration practices are influenced by many factors other than the implementation efforts themselves, this questionnaire is designed to gather information on some of those factors that are not frequently evaluated. You have been selected as a prospective participant due to your expertise and experience in local and/or regional restoration activities. This questionnaire should not take more than 30 minutes, and all input and communication will be kept confidential unless otherwise permitted. If you have any question, comments, or concerns, please reach out to Joshua Moody, PDE's Restoration Programs Manager at jmoody@delawareestuary.org

Section 1: Respondent Description

Please let us know a little about yourself. Name is not required, but please fill out the other sections to the best of your ability. Answers to these questions will be used to partition data and contextualize summary statistics to identify overarching factors that contribute to particular perspectives and experiences.

1. Name (not required)

2. Sector

- Federal
- State
- Private
- Academic
- NGO
- Other

3. How many years have you been involved in restoration in the Mid-Atlantic?

- 0 to 3 Years
- 3 to 5 Years (i.e., since 2017)
- 5 to 10 years
- 10 to 20 years
- 20 to 30 years
- 30+ years

4. Typical Working Habitat (choose up to three)

- Upland
- Tidal Freshwater

- Tidal Brackish/Saltwater
- Non-Tidal Freshwater
- All of the above
- Other

5. What area of restoration practice are you typically involved in (choose all that apply)?

- Distributing Funds
- Site Evaluations
- Project design
- Permitting
- Installation
- Monitoring
- Training
- Outreach
- Regulating
- Other

6. Is your organization a Delaware Estuary Program (DELEP) partner or a member of any DELEP committees? DELEP partners are a group of regional organizations that, with PDE, work together to coordinate and implement long-term plans for the Delaware Estuary. DELEP committees include the Steering Committee, Estuary Implementation Committee (EIC), and the Science and Technical Advisory Committee (STAC).

- Yes
- No

7. Has your organization partnered on restoration projects with PDE?

- Yes
- No

Section 2: Categorical Questions

Please select a response from the provided options to the statements in the following six categories from the list of options provided for each that is most reflective of your experience and understanding. Generally, the answers will follow a pattern of declining agreement, including a neutral option as follows:

- Substantially increased
- Moderately increased
- Not changed
- Moderately decreased
- Substantially decreased

Please answer all questions by considering the time frame 2017-2021 in the Delaware Estuary.

Innovation: Innovation is the development of new ideas and approaches or new applications of existing ideas and approaches. Innovation can be applied to restoration techniques, technology, and/or tactic design or implementation. For each of the following questions, please select the answer most reflective of your experience.

1. Since 2017, the use of feedback, lessons-learned, and/or results to inform project design has____.
2. Since 2017, the general level of innovation applied by regional organizations has ________.
3. Since 2017, the level of innovation applied by the organization I represent has ________.
4. Since 2017, the proportion of new and proposed projects that incorporated (not just discussed) sea level rise into their design has______.
5. Since 2017, the effects or actions associated with the regulatory process have _______ the opportunity for the application of innovation.

Level of Investment: These questions ask the user to consider changes in factors related to funding opportunities for restoration efforts. For each of the following questions, please select the answer most reflective of your experience.

1. Since 2017, funding for restoration opportunities has______.
2. Since 2017, the proportion of available funding that is focused on implementation rather than project planning has______.
3. Since 2017, the proportion of available funding for project monitoring has______.
4. Since 2017, match requirements have ______ the level of difficulty in applying for funds.
5. SInce 2017, the distribution of funding across diverse recipients has ______.

Regulatory Climate: These questions ask the user to consider changes in regulatory considerations, requirements, and timeframes as they relate to restoration project implementation and requirements. For each of the following questions, please select the answer most reflective of your experience.

1. Since 2017, the time required to complete the permit process has______.
2. Since 2017, the level of effort and associated costs of acquiring permits ______.
3. Since 2017, the regulatory options for permitting a project have______.
4. Since 2017, the complexity of the permitting process has _______.
5. Since 2017, the general willingness of the agencies to work with the applicant has ________.
6. Since 2017, the general perception of technical competence within permitting agencies has _______.

Implementation: Implementation refers to the actual execution of restoration activities at a specific-location, such as construction, deployment of fill material, excavation, or any activity that alters or adds to the landscape. For each of the following questions, please select the answer most reflective of your experience.

1. Since 2017, the rate of individual project implementation has ______.
2. Since 2017, the average quality of implemented projects has ______.
3. Since 2017, the number of restoration projects implemented that consider a change in future conditions (e.g., sea level rise, habitat transgression) rather than just the immediate condition has ______.
4. Since 2017, the number of implemented projects aimed at testing new techniques or applications has ______.
5. Since 2017, the percent of failures or under performance associated with implemented projects has _______.

Monitoring: Monitoring refers to any activity conducted with the explicit intent to collect information related to any facet of the project that will be used to better understand one or more processes potentially affected by the restoration activity. This can include project monitoring conducted on consistent, sporadic, or opportunistic time frames, and for required (i.e., permit requirement) or optional (PI/partner initiated) activity. For each of the following questions, please select the answer most reflective of your experience.

1. Since 2017, optional project monitoring has ______.
2. Since 2017, regulatory monitoring requirements have ______.
3. Since 2017, the availability of project monitoring guidance materials has ______.
4. Since 2017, standardization of monitoring across projects has ______.
5. Since 2017, funding for monitoring has ______.
6. Since 2017, monitoring has ________ my understanding of restoration.

Outreach: Outreach refers to activities intended to disseminate information regarding restoration needs, activities, and/or results to any stakeholder (e.g., different sectors, general public, educators, other practitioners). For each of the following questions, please select the answer most reflective of your experience.

1. Since 2017, information being relayed to stakeholders regarding restoration needs has ______.
2. Since 2017, information being relayed to stakeholders regarding restoration activities has ______.
3. Since 2017, information being relayed to stakeholders regarding positive restoration lessons learned/results has ______.
4. Since 2017, information being relayed to stakeholders regarding negative (i.e., project "failure") restoration lessons learned/results has ______.
5. Since 2017, the consistency in outreach messaging (e.g. what has been working, costs, feasibility of techniques) regarding restoration needs, activities, and/or results has ______.
6. Since 2017, the clarity in outreach messaging has ______.

Section 3: Additional Questions

Please select one response from the provided options to the questions below that is most reflective of your experience and understanding.

1. Since 2017, which area of restoration has experienced the greatest increase in awardee resource allocation?

- Design
- Permitting
- Material Procurement
- Installation (material installation costs and associated labor)
- Monitoring
- Outreach

2. How do you receive most of your information regarding restoration needs, activities, and outcomes/ lessons learned?

- Peer-reviewed Journal Articles
- Presentations at Conferences or on Webinars
- Work Group/Committee Discussion and/or Presentations
- Site Visits/Field Trips
- Tabling/Outreach/Education Events

- Informal Conversation
- Volunteering/Hands-on Experience

3. How do you best absorb information regarding restoration needs, activities, and outcomes/lessons learned?

- Peer-reviewed Journal Articles
- Presentations at Conferences or on Webinars
- Work Group/Committee Discussion and/or Presentations
- Site Visits/Field Trips
- Tabling/Outreach/Education Events
- Informal Conversation
- Volunteering/Hands-on Experience

4. Which statement best reflects your thoughts on regulatory monitoring requirements:

- They are generally appropriate for their associated projects.
- They are generally not appropriate for their associated projects
- Some are appropriate while others seem to be of interest only to the permitting agency.

5. Looking forward to 2027, where do you see the biggest restoration need (select your top three)?

- Salt marsh edge stabilization
- Freshwater tidal marsh edge stabilization
- Salt marsh platform elevation enhancement
- Freshwater tidal marsh platform elevation enhancement
- Dune construction and stabilization
- Beach replenishment
- Protection of transgression zones (e.g., for forests, wetlands)
- Shellfish restoration (across salinity gradient)
- Stream stabilization
- Island Creation/Protection
- SAV Restoration
- Soft Bottom Habitat
- Reforestation
- Freshwater Wet Meadow
- Freshwater Forested Wetlands
- Freshwater Instream Habitat
- Riparian Habitats and BMPs
- Urban Greenspace Reclamation
- Brownfield Reclamation
- Pedestrian Access

Appendix A

Justice, Equity, Diversity, and Inclusion in the Delaware Estuary and Basin

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Justice, Equity, Diversity, and Inclusion (JEDI) are central to the long-term health of the people that reside within the Delaware Estuary and Basin. Twenty years ago, PDE released the first State of the Estuary Report. Since then, we have become increasingly aware of the need to address systemic issues, such as environmental justice and racism, which currently plague the Estuary and Basin. Within this TREB, and those that will follow, PDE and its partners will continue to find ways to open discussions about possible inequities with respect to environmental health within the Delaware Estuary and Basin. The first step in a JEDI framework was to perform a JEDI audit of the 2017 TREB. Here, we review the findings of our audit to steer and inspire future work.

The JEDI audit was a multi-step process. We began this audit by researching environmental issues as they intersect with race and socioeconomic backgrounds of the Delaware Estuary and Basin. This enabled us to review the 2017 TREB and identify places where environmental justice issues were relevant. We were particularly mindful of racial diversity, socioeconomic diversity, land usage and ownership, and opportunities for education initiatives. The next step of our process was propose environmental justice metrics to use moving forward. We determined a need to identify equity gaps for things such as environmental jobs, home ownership, funding trends, and engagement opportunities. Although we could not address every question, metric, or suggestion in the 2022 TREB, this framework provides necessary direction for future TREB reports.

Specific environmental issues with equity or justice focus included flooding, air pollution, soil contamination, blood lead levels, tree cover, access to clean water, access to boat launches, and location of environmental justice communities. Metrics for engagement include adaptive capacity of communities, incorporation of local knowledge in management and municipality connectedness. Providing the necessary support that the Lenni Lenape— the indigenous people of the Delaware River Watershed— say they need to participate in all discussions about the management of this land as they desire is also an essential step in dismantling some of the harm done by colonization.

General notes

Be mindful of:

- Racial diversity
- Socioeconomic diversity
- Land usage and ownership
- Education initiatives
- There should be an accessible online database for the data used, where possible

Moving forward:

- Moving climate change to the front so environmental indicators act as a response to the greater issues
- Metric: using EJ communities in relation to spatial mapping
- Highlight EJ NJ law passed last year
- Listing each EJ definition for each state in the watersheds
- Utilizing census data and EJ Screen for EJ analyses
- Is there information on consumption advisories? Where would that best go?

Watersheds and Landscapes

Overarching Suggestions

- Be mindful to not ignore: racial diversity, socioeconomic diversity, land usage and ownership, and educational attainment.
- Populations
- Including information on racial diversity, socioeconomics, land ownership / segregation, and educational attainment can help set the stage for later discussion of economic and racial justice metrics.
- May include EPA Environmental Justice (EJ) screening tool.
- One idea that could be propagated throughout this chapter is to identify environmental justice communities based on the methodology of the PA DEP (basically 30%+ non-white and/or 20%+ below federal poverty line). This is more of a binary metric, while the EPA EJ screening tool is a continuous variable. If you locate the EJ communities, you can then look for differences in different metrics (current land cover, land cover change, impervious cover, open space, etc) based on values for EJ communities in comparison to non-EJ communities. This would translate into a simple metric of inequality.
- Consideration of how the demographics are changing.
- Is gentrification happening? Where?

Land use/ land cover

- A metric or map that tracked differences in land cover by race / income (or in EJ community vs. not) could be useful.
- Maps that track changes in relation relative to EJ communities (or racial and socioeconomic divisions) could be useful.
- Who lives in the spaces where land cover is changing?

Impervious cover

- A metric to show the difference in impervious surface coverage in EJ vs non-EJ communities.
- Land cover and impervious surface coverage is related to heat stress, which is an important climate change vulnerability for EJ communities.

Public open space

- A metric to show the difference in public open space access for EJ vs non-EJ communities (such as using a cost-distance function)
- Who is utilizing the open space?

Public access points

- Show whether there is inequality or not in EJ vs. non-EJ communities with respect to access

Natural capital value

- Are environmental jobs / workers mainly white?

- Is there demographic information for jobs that could be used to show if there is equality or inequality across race?
- Actions and needs section can focus on need for equity in access to green jobs

Water Quantity

Overarching Suggestions

- How do you think about water usage in an environmental justice framework?
- Are upstream users claiming water at the expense of water quantity / quality downstream?
- Do all have equal access to the resources? Is there patterns related to affluence?
- Per capita water use: which areas use more resources? Which use less? Why?
- Need more on tributaries
- Effects on underrepresented communities along tributaries should be addressed
- How does prioritizing NYC affect communities around the Delaware River and Delaware Bay?
- Is management of access to the resources done equitably?

Water Withdrawals: Tracking Supply & Demand

- Break down by race and socioeconomic standing for basin water use
- Who has used water from the basin in the past?
- Are there any concerns about water use? Is it different for different communities? If not, why not?
- Do certain populations use more water than others? Why?
- If certain populations consume more water, what specific solutions can we offer them

Per Capita Water Use

- Who lives in the eight sub-basins
- If certain populations consume more water, what specific solutions can we offer?
- Who has or where is leaky/poor infrastructure?

Groundwater Availability

- Who lives in areas most stressed?

Salt Front Location & Movement

- What are the downstream ecological and societal effects of water diversion upstream?

Water Quality

Overarching Suggestions

- How does water quality affect the ecosystem and human populations?
- How has water quality historically affected the ecosystem and human populations?
- Give concrete ideas for solutions and improvements
- Mention water quality issues such as CSO overflows, plastics, emerging contaminants (PFAS), trash in waterways.
- Report focuses on mainstem, which we agree integrates the watershed. However, this obscures where in the watershed are input/contamination hotspots. Can some information from tributaries (e.g., from USGS data) be integrated?
- Water security as it relates to water quality is an important issue. Specific examples that could be touched on: drought, freshwater availability (relative to salt line drift or salt intrusion)
- Access to potable water in at risk communities need to be more tightly monitored

Tidal

Dissolved oxygen

- Should we talk more about the locations where the DO is being tested? For example Philadelphia has a lot of marginalized populations and is close to the airport. Other sites should be built out the same way.
- How has this historically affected the ecosystem and human populations?

Contaminants

- Metals, pesticides and polycyclic aromatic hydrocarbons (PAHs): how are these affecting the ecosystem and human population?

Fish Contaminant Levels

- What other contaminants have been measured on a time series?
- How do contaminants affect people?
- Is there anything that can be done to treat the effects people experience if eating these fish?

Emerging Contaminants

- Is there anything that can be done to treat the effects people experience of eating these fish and shellfish?
- Give concrete ideas for solutions and improvements

Whole effluent toxicity

- Does toxicity vary based on whether the community is an EJ community?

Non-tidal

- Emerging Contaminants: Expand on effects of pharmaceuticals
- Temperature Progress on PA's seasonally specific temperature criteria for warm water fisheries?

Sediments

Overarching Suggestions

- Are some communities exposed to toxic sediment, due to being located near sources of contamination, polluting industries and waste management?
- Who lives by the filled dredged materials lots? How are they affected?
- How are different communities affected by sediment loss or erosion?
- Are dredging activities carried out equitably?

Dredging Activity

- How does dredging affect surrounding communities? Does the dredged sediment contain hazardous material?
- What types of land settlements tend to need dredging? Are routine dredging activities prioritized equitably?
- Is there equal funding/opportunity for dredging needs? Commerce vs. recreation vs. both?
- Where are the upland dredging disposal sites?

Habitats

Overarching Suggestions

- How can we be proactive in promoting environmental equity?
- Where are there equity gaps in habitat quality and how can they be remedied?
- Greenspace – How does it fit in?
- Environmental equity analysis could be done on most spatially explicit data – comparing EJ to non-EJ communities.
- For riparian corridors specifically: Overall, cities and urbanized areas play an important role in environmental quality. Cities that have high density have a lower carbon footprint per person, and people are able to live more sustainably – not driving as much, consuming less resources, and altering the land cover over a much smaller footprint than folks that live in rural areas, suburbs, and exurban areas. In this context, we need to be able to determine what biodiverse, healthy, and revitalized cities look like so that we can create them across a development gradient. Here, it is important to just adjust the tone as to not put as strong a value judgment on urbanization.
- Action items should incorporate equity concerns, especially focusing on the importance of restoration, high quality habitat, and public access to greenspace in cities generally and overburdened neighborhoods specifically.

Subtidal habitats

- How have human activities contributed to the status of soft or hard bottom habitat communities?
- The 2012 TREB reports high benthic diversity in the marine part of the estuary and low diversity in the freshwater portion, indicating polluted condition in the freshwater portion of the Estuary. This suggests poor habitat quality in the freshwater part of the Estuary, where human populations and environmental justice communities are. This could be added to the discussion.

Nontidal habitats

Freshwater Wetlands

- Do these overlap with residential areas? Commercial? Where are they? Who lives here?
- Add action items related to environmental equity, such as: determine whether wetland policy has created inequities as to who receives benefits of wetlands.

Fish Passage

- How does dam removal affect communities?
- Dam removal prioritization should include equity as part of its return.

Hydrological Impairment

- Talk about the location of this, where it is, who they effect and how
- Stormwater runoff fees – Need for equitable education so the public votes/ understands needs for them, particularly in the context of climate change.

Living Resources

Overarching Suggestions

- Include consumption advisories

Climate Change

Overarching Suggestions

- Include discussion of the importance of equity in resilience planning and recovery from climatic event emergencies.
- What types of climatic events are likely to affect EJ communities? Some examples include extreme flooding and heat stress.
- Integrate information about how temporal trends in climate change are affecting particular communities compared to others
- How has temperature changed by socioeconomic status?
- How do these indicators affect people? How are people affected differently?

Air Temperature

- What are the effects of air temperature on animals, humans, ecosystems, infrastructure etc.?
- How does it disproportionately affect people by race or socioeconomic standing?
- How will warming affect day-to-day life of different socioeconomic groups?

Precipitation

- How does increased precipitation affect people, ecosystems, animals, infrastructure?
- Are people affected by indigenous status, race, or socioeconomic factors?

Extremes: Air Temperature and Precipitation

- How is this affecting air temperature on animals, humans, ecosystems, infrastructure etc.
- How does it disproportionately affect people by race, socioeconomic standing?

Streamflow

- What about development trends & destruction of habitats?
- How does it disproportionately affect people by race, socioeconomic standing?

Restoration

Acres Restored Annually

- How do restored areas compare with racial diversity or socioeconomic diversity?
- Which parts are identified as needing help, but then not getting it? Who lives there?
- Which parts of the watershed are increasing in protection? Who lives there?
- Which parts of the basin have improved the most since data has been collected? What are the data trends? Who lives there?
- Are there different trends in funding based on position within the watershed or socioeconomic status?
- How will different people in different parts of the basin be affected?

Restores Habitat Types

- Who lives in or near the greatest habitat losses? Can we balance investing in those systems?
- Add maps showing different restored habitats, racial diversity, and socioeconomic diversity
- Are there different trends in funding based on position within the watershed or socioeconomic status?
- Who lives where the most critical habitats are?

Restoration Need

- Who lives in the other watersheds that are more funded?

Table A1. List of all acronyms referenced throughout this document referring to entities, committes, agencies, etc., relative to the indicators updated in the 2022 TREB.

Acronym	Definition
ADZ	Aquaculture Development Zone
ANSDU	Academy of Natural Sciences at Drexel University
AMNET	Ambient Macroinvertebrate Network
ARM	Adaptic Resource Management
ASMFC	Atlantic States Marine Fisheries Commission
BEST	Berkeley Earth Surface Temperature
BBP	Barnegat Bay Partnership
BMP	Best Management Practice
BOD	Biochemical Oxygen Demand
C-CAP	Coastal Change Analysis Program
CCMP	Comprehensive Conservation Management Plan
CDF	Confined Disposal Facility
CM	Conservation Measure
CMSA	Catch Multiple Survey Analysis
COOP	Cooperative Observer Program
CPMI	Coastal Plain Macroinvertebrate Index
CPUE	Catch Per Unit Effort
CSO	Combined Sewer Outflow
DDFW	Delaware Division of Fish and Wildlife
DDT	Dichlorodiphenyltrichloroethane
DEB	Delaware Estuary and Basin
DEBI	Delaware Estuary Benthic Inventory
DELEP	Delaware Estuary Program
DELSI	Delaware Living Shoreline Initiative
DLUR	Division of Land Use Regulation
DNREC	Department of Natural Resources and Environmental Control
DO	Dissolved Oxygen
DRBC	Delaware River Basin Commission
DRWI	Delaware River Watershed Initiative
EC	Emerging Contaminant
EDC	Endocrine Disrupting Compound
EIC	Estuary Implementation Committee
EJ	Environmental Justice
EPA	Environmental Protection Agency
ESA	Endangered Species Act
ETM	Estuarine Turbidity Maximum
EWQ	Existing Water Quality

FIA	Forest Inventory and Analysis
FMRP	Freshwater Mussel Recovery Program
GAP	Gap Analysis Program
GCM	Global Climate Model
GHCN	Global Historical Climatology Network
GHG	Greenhouse Gases
GIS	Geographic Information Systems
HGMI	High Gradient Macroinvertebrate Index
HOA	Homeowner Association
HOLC	Home Owners' Loan Corporation
HUC	Hydrologic Unit Code
ICP	Interstate Control Points
IDW	Inverse Distance Weighting
IPCC	Intergovernmental Panel on Climate Change
JEDI	Justice, Equity, Diversity, and Inclusion
LAL	Limulus Amebocyte Lysate
MACC	Monitoring Advisory and Coordination Committee
MACWA	Mid Atlantic Coastal Wetland Assessment
MGD	Million Gallons per Day
MidTRAM	Mid Atlantic Tidal Rapid Assessment Method
MS4	Municipal Separate Storm Sewer System
MuCWI	Mussels for Clean Water Initiative
NAIP	National Agriculture Imagery Program
NEP	National Estuary Program
NEPORT	National Estuary Program Online Reporting Tool
NGO	Non-governmental Organization
NJBWM	New Jersey Bureau of Water Monitoring
NJDA	New Jersey Department of Agriculture
NJDEP	New Jersey Department of Environmental Protection
NJGWS	New Jersey Geologic and Water Survey
NLCD	National Land Classification Database
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
NWI	National Wetlands Inventory
NYSDEC	New York State Department of Environmental Conservation
OTC	Over the Counter
PADEP	Pennsylvania Department of Environmental Protection
PAD-US	Protected Area Database of the United States
PAH	Polycyclic Aromatic Hydrocarbons
PBO	Programmatic Biological Opinion

PCB	Polychlorinated Biphenyls
PDE	Partnership for the Delaware Estuary
PEL	Probable Effect Levels
PFAS	Polyfluoroalkyl Substances
PFNA	Perfluorononanoate
PFOS	Perfluorooctanesufonate
PFuNA	Perfluoroundecanoate
PMI	Pinelands Macroinvertebrate Index
PPCP	Pharmaceuticals and Personal Care Products
PWD	Philadelphia Water Department
PWS	Public Water Supply
SARC	Stock Assessment Review Committee
SAV	Submerged Aquatic Vegetation
SAW	Stock Assessment Workshop
SC	Steering Committee
SEPA-GWPA	Southeastern Pennsylvania Groundwater Protected Area
SLAMM	Sea Level Affecting Marsh Model
SLR	Sea Level Rise
SOE	State of the Estuary
SPW	Special Protection Waters
SQG	Sediment Quality Guidelines
SRWC	Stroud Water Research Center
SSIM	Site Specific Intensive Monitoring
SSP	Shared Socioeconomic Pathway
STAC	Science and Technical Advisory Committee
TAC	Toxics Advisory Committee
TMDL	Total Maximum Daily Load
TREB	Technical Report for the Estuary and Basin
TSS	Total Suspended Solids
USACOE	United States Army Corps of Engineers
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
UTM	Universal Transverse Mercator
WET	Whole Effluent Toxicity
YOY	Young-of-the-year